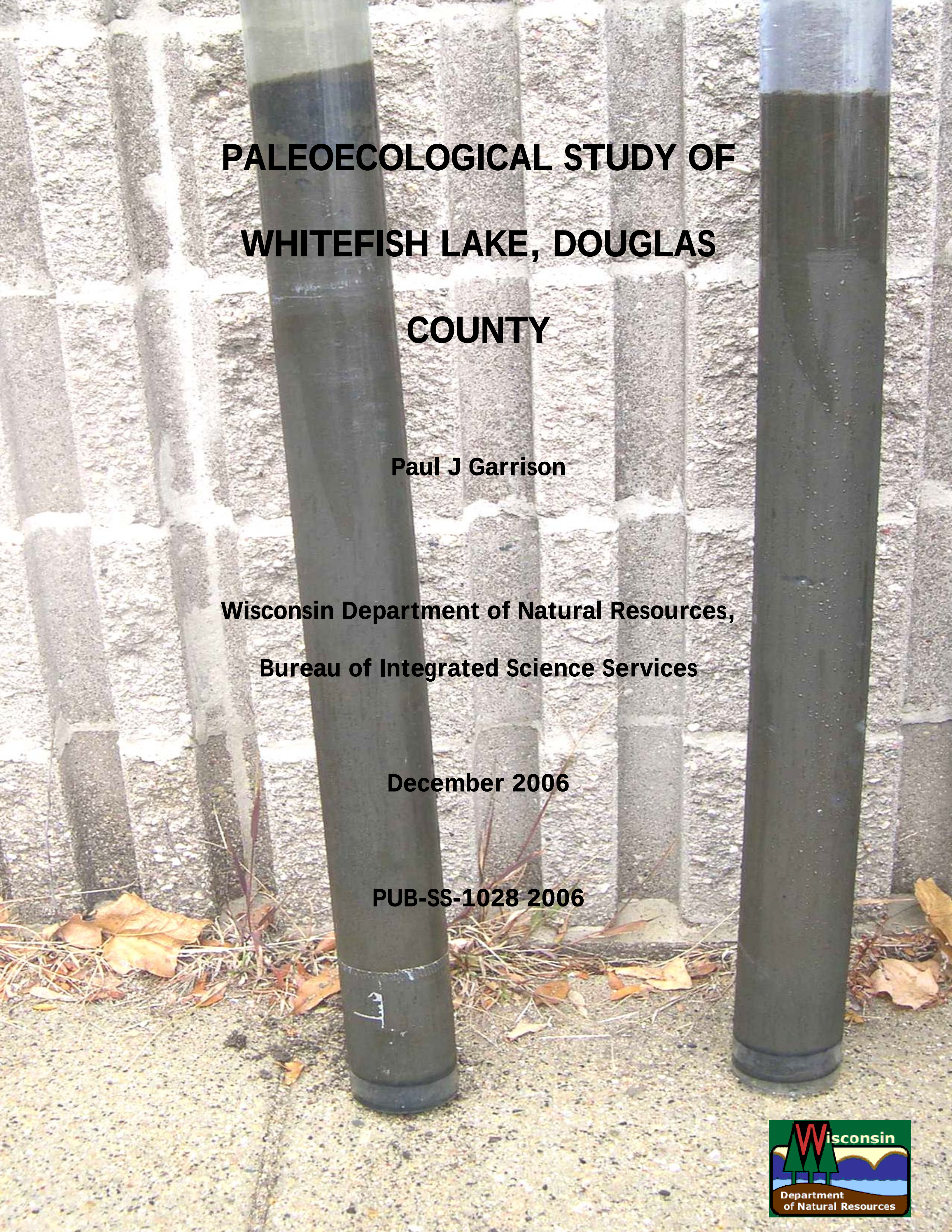

PALEOECOLOGICAL STUDY OF WHITEFISH LAKE, DOUGLAS COUNTY

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Introduction

Questions often arise concerning how a lake's water quality has changed through time as a result of watershed disturbances. In most cases there is little or no reliable long-term data. Questions often asked are if the condition of the lake has changed, when did this occur, what were the causes, and what were the historical condition of the lake? Paleoecology offers a way to address these issues. The paleoecological approach depends upon the fact that lakes act as partial sediment traps for particles that are created within the lake or delivered from the watershed. The sediments of the lake entomb a selection of fossil remains that are more or less resistant to bacterial decay or chemical dissolution. These remains include diatom frustules, cell walls of certain algal species, and subfossils from aquatic plants. The chemical composition of the sediments may indicate the composition of particles entering the lake as well as the past chemical environment of the lake itself. Using the fossil remains found in the sediment, one can reconstruct changes in the lake ecosystem over any period of time since the establishment of the lake.

Whitefish Lake is a 832 acre lake located in Douglas County. The maximum depth is 102 feet with a mean depth of 30 feet. Three sediment cores were each collected from the North and South Basins of the lake on 28 October 2004. The cores were collected with a gravity core with a plastic tube having an inside diameter of 62.8 cm. The cores were collected from the deep areas of each basin (Figure 1). The location of the coring site in the South Basin was 46º 12.227' north, 91º 52.635' west in a water depth of 95 feet (29.0 m). The location of the coring site in the North Basin was 46º 13.223' north, 91º 52.364' west in a water depth of 54 feet (16.5 m). The top 30 cm of the cores were sectioned into 1 cm intervals while the remainder of the cores were sectioned into 2 cm slices. The longest core from the South Basin was 70 cm and the longest core from the North Basin was 66 cm. The core was dated by the ^{210}Pb method. The diatom community was analyzed to assess changes in nutrient levels and changes in the macrophyte community and geochemical elements were examined to determine the causes of changes in the water quality and changes in oxygen conditions in the bottom waters.

Results and Discussion

Dating

In order to determine when the various sediment layers were deposited, the samples were analyzed for lead-210 (^{210}Pb). Lead-210 is a naturally occurring radionuclide. It is the result of natural decay of uranium-238 to radium-226 to radon-222. Since radon-222 is a gas (that is why is sometimes is found in high levels in basements) it moves into the atmosphere where it decays to lead-210. The ^{210}Pb is deposited on the lake during precipitation and with dust particles. After it enters the lake and is deposited 2in the lake sediments, it slowly decays. The half-life of ^{210}Pb is 22.26 years (time it takes to lose one half of the concentration of ^{210}Pb) which means that it can be detected for about

Figure 1. Map of Whitefish Lake showing the two coring sites. The water depth of the coring site in the South Basin was 95 ft and the depth in the North Basin was 52 ft.

130-150 years. This makes ^{210}Pb a good choice to determine the age of the sediment since European settlement began in the mid-1800s. Sediment ages and mass sedimentation rates were determined by two ways: (1) constant rate of supply (CRS) model (Appleby and Oldfield, 1978) and the piecewise CRS model of Appleby (1998, 2001). Accumulation rates of geochemical variables were computed for each sediment depth by multiplying the bulk sediment accumulation rate (g cm^{-2} yr^{-1}) by the corresponding concentration (mg g^{-1}) of each constituent in the bulk sediment.

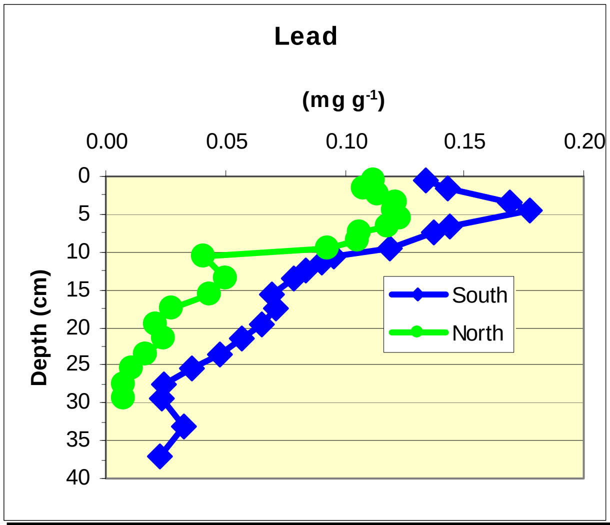
There can be problems with this dating technique. For example, when sediment has moved after it was deposited, large changes in sediment deposition over the last 150 years, and errors associated with lab analysis with sediments that are over 100 years old. For these reasons the accuracy of the ^{210}Pb dates is verified by other methods. These methods usually involve measuring parameters that are known to have been deposited at a certain time and comparing stratigraphic changes in the core in Whitefish Lake with other lakes in the region.

Cesium-137 (^{137}Cs) can be used to identify the period of maximum atmospheric nuclear testing (Krishnaswami and Lal, 1978). The peak testing occurred by the USSR in 1963 and thus the ^{137}Cs peak in the sediment core should represent a date of 1963. Another element that can be used to verify the dating model is the profile of stable lead. Stable lead has an historical pattern of deposition that is very consistent among lakes, with lead concentrations increasing from around 1880 to the mid-1970s, and decreasing to the present. The decline of lead is largely the result of the discontinued use of bonded leaded gasoline in the mid-1970s (Gobeil et al. 1995; Callender and Van Metre 1997).

In both cores, with the CRS model the peaks of cesium-137 and stable lead were placed at depths that were to young. In both cores the peaks were at the same depth (4-5 cm) which the model indicates was a date of about 1992. Since the ^{137}Cs and the stable lead peaks should be at different depths, one or both of these variables is not a good indicator of the correct date. The profile of stable lead for the South Basin has a very distinct peak which is typical of lead profiles from other lakes. In the North Basin the peak is not as sharp (Figure 2) as in the South Basin. In neither basin is the ^{137}Cs peak very sharp (not shown). It is likely that the cesium profile does not accurately reflect cesium deposition. Other studies have found that ^{137}Cs can be mobile and thus its profile does not always accurately reflect its depositional history.

The piecewise CRS model was constructed by calculating the ^{210}Pb flux for independently dated intervals determined by other chronostratigraphic markers (Appleby 1998, 2001). Because the stable lead profile appeared better than the cesium profile, the lead profile was used to construct the piecewise CRS model. The lead peak at 4-5 cm was assumed to represent a date of 1976 and the initial rise in lead at 27-28 cm was assumed to be 1880. Although the peak is broader in the North Basin core (Figure 2) the peak also occurred around a depth of 4-5 cm. It is likely that the sediment at this site has undergone some mixing which resulted in smearing of the peak.

Another indication that the rise in stable lead at 27-28 cm represents the time period around 1880 is the decline in organic matter at this depth (Figure 2). Other studies have shown that this decline is the result of watershed activities which result in an increase in the soil erosion (Engstrom et al.,



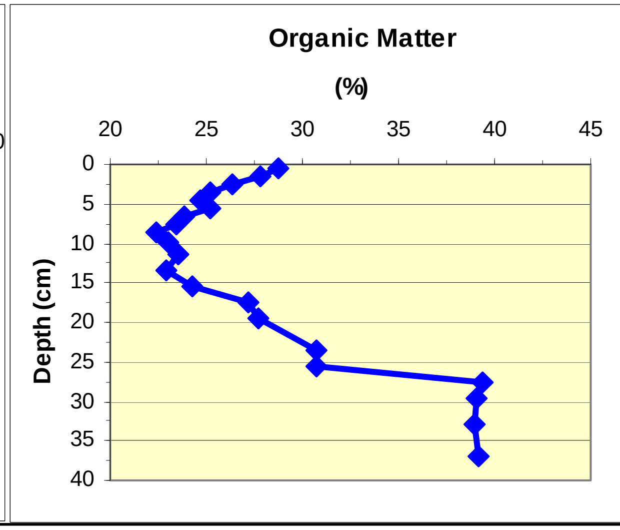


Figure 2. Profiles of stable lead in the cores from both basins. The peak is assumed to represent a date of 1976 and the initial increase is a date of 1880. Profile of organic matter in the South Basin. The beginning of the decline at 27-28 cm indicates an increase in the input of inorganic material

1985; Garrison, 2000a,b; Garrison, 2003; Garrison and Wakeman, 2000). This erosion is largely composed of inorganic material which dilutes the organic matter and thus organic matter concentrations decline.

The CRS model and the piecewise CRS model give different results for dates and the bulk sedimentation rate. Calculated dates and sedimentation rate are given in Appendices 1 and 2. Since the lead peak in the South Basin is very sharp and the initial rise of lead and decline in organic matter occur at similar depths in this core it is assumed that the piecewise CRS model is the most accurate. The sedimentation rate shown in Figure 4 and dates of the subsequent graphs are those derived using the piecewise CRS model.

Sedimentation Rate

The mean mass sedimentation rate for the South Basin the last 160 years was 0.008 cm^{-2} yr^{-1} and for the North Basin the rate was somewhat lower at 0.005 cm^{-2} yr^{-1} (Figure 3). These are the lowest rates I have measured for 42 Wisconsin Lakes. The average linear rate for the South and North Basins was 0.23 cm yr^{-1} which equates to less than 0.1 inch of sediment per year.

To account for sediment compaction and to interpret past patterns of sediment accumulation, dry sediment accumulation rates were calculated. In the North Basin the sedimentation rate was constant from the period 1870 until about 1970. During the last 3 decades the rate has steadily increased and

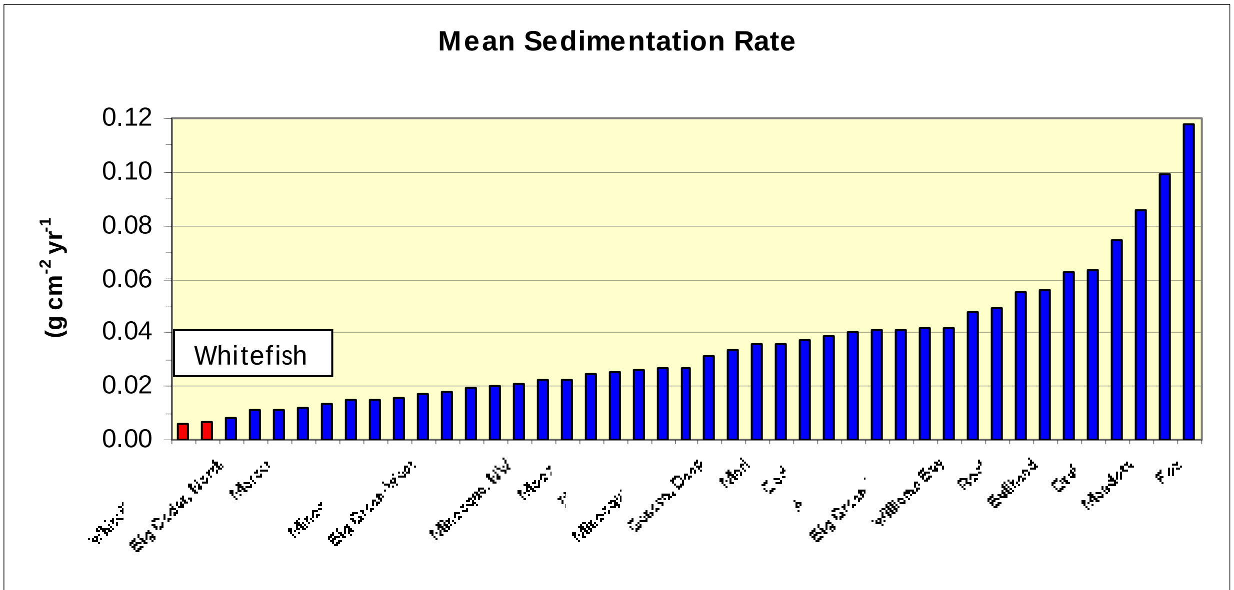


Figure 3. Mean sedimentation rate for the last 180 years for 42 Wisconsin lakes. The North Basin has the lowest mean sedimentation rate of all the lakes while the South Basin had the second lowest sedimentation rate.

at the present time it is near its highest rate at 0.009 cm^{-2} yr^{-1} (Figure 4). The increased rate is likely the result of anthropogenic activities in the watershed, e.g. shoreland development.

In the South Basin the historical sedimentation rate was higher than in the North Basin (0.005 vs 0.008 cm^{-2} yr^{-1}). This is likely because the South Basin is deeper and sediment focusing occurs at a higher rate. This is confirmed by the higher ^{210}Pb inventory in the South Basin. Around 1920, the sedimentation rate began declining and this continued until the 1970s. It is unclear what was the cause of this. As with the rate in the North Basin, the rate has steadily increased during the last 3 decades.

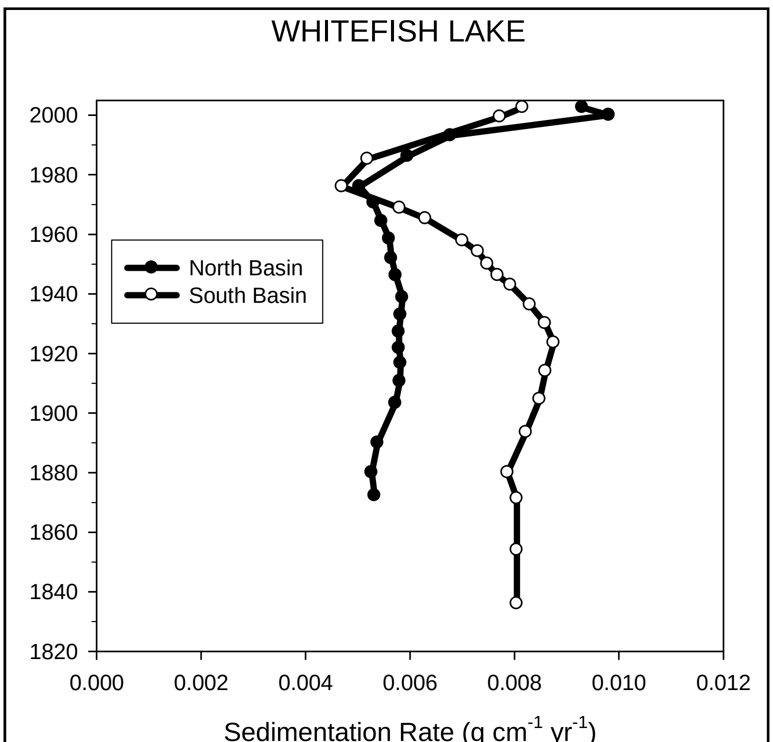


Figure 4. Sediment accumulation rate for the North and South Basins of Whitefish Lake. The higher historical rate in the South Basin is likely because of its deeper depth which facilitates sediment focusing. In recent years the North Basin exhibits a higher rate which is likely the result of anthropogenic activities in the lake's watershed.

Sediment Geochemistry

Geochemical variables are analyzed to estimate which watershed activities are having the greatest impact on the lake (Table 1). The chemical aluminum (Al) is found in soil particles, especially clays. Changes in Al are an indication of changes in soil erosional rates throughout the lake's history. Zinc (Zn) is associated with urban runoff because it is a component of tires and galvanized roofs and down-spouts. Nutrients like phosphorus and nitrogen are important for plant growth, especially algae and aquatic plants. Calcium is an indication of the use of soil amendments for lawns. Manganese is an indication of changes in oxygen levels in the bottom waters.

Table 1. Selected chemical indicators of watershed or in lake processes.

Process	Chemical Variable
Soil amendment	calcium
Soil erosion	aluminum, titanium
Urban	zinc, copper
Anoxia	manganese
Nutrients	phosphorus, nitrogen

In the South Basin the concentration of the geochemical variables of interest were essentially unchanged during the nineteenth century and the twentieth century until about 1920 (Figure 5). From about 1920 until about 1960, aluminum (indicative if soil erosion) steadily increased a small but significant amount. The decline in aluminum concentrations after 1960 indicates are decline in soil erosion.

Concentrations of calcium, which is often used as a soil amendment in sandy soils such as found around Whitefish Lake, has been increasing since 1970 (Figure 5). This likely reflects increased suburbanization from shoreland development around the basin. The nutrient phosphorus has been increasing during the same time period that calcium concentrations have been increasing. This is not a good sign for the lake as phosphorus is usually the nutrient that is in shortest supply and thus increasing concentrations will have a more profound impact on the lake's water quality. Nitrogen does not show the same trend as phosphorus. The decline in nitrogen and organic matter concentrations between the period 1920 and 1970 is the result of dilution from materials brought into the lake from soil erosion. This dilution as a result of increased soil erosion has been found in other lakes (Engstrom et al., 1985; Garrison, 2000a,b; Garrison, 2003; Garrison and Wakeman, 2000). Nitrogen concentrations in-

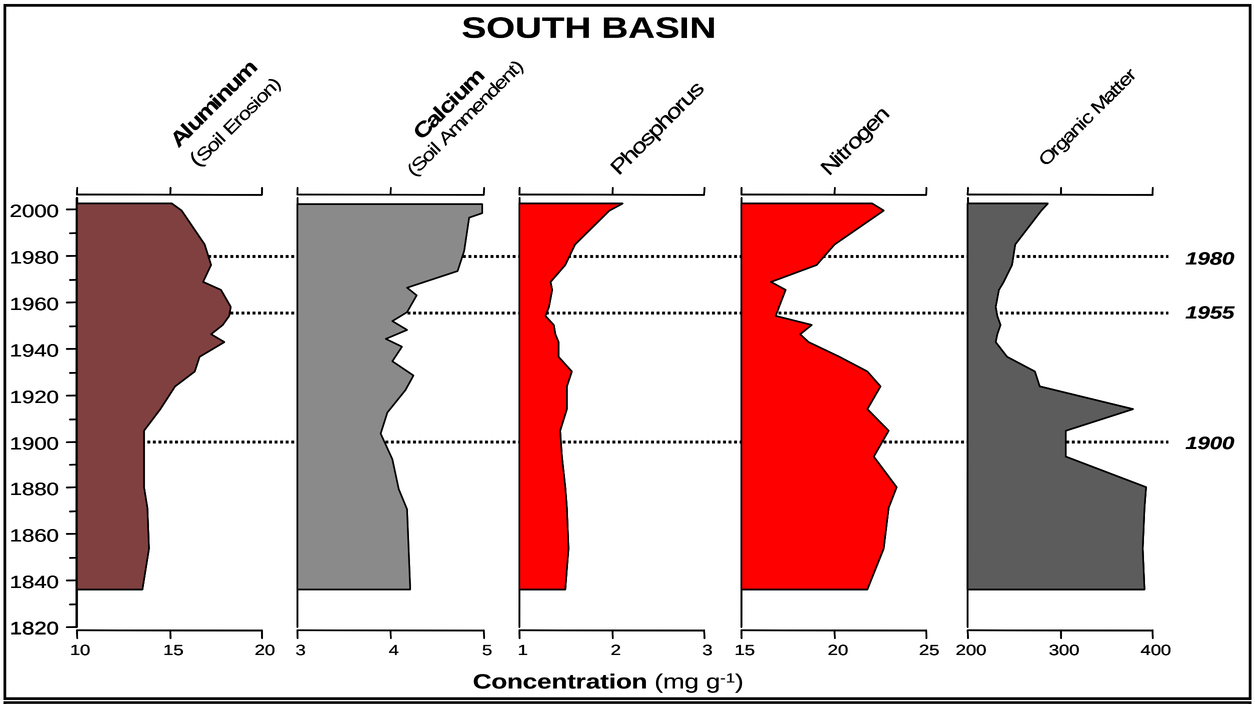


Figure 5. Profiles of the concentration of selected geochemical elements in the South Basin core. Aluminum profiles are indicative of soil erosional rates in the watershed. Calcium is often used as a soil amendment in lawns. Nitrogen and phosphorus profiles reflect changes in nutrient deposition rates while organic matter profiles indicate deposition of biological compounds produced within the lake and those entering the lake from the watershed.

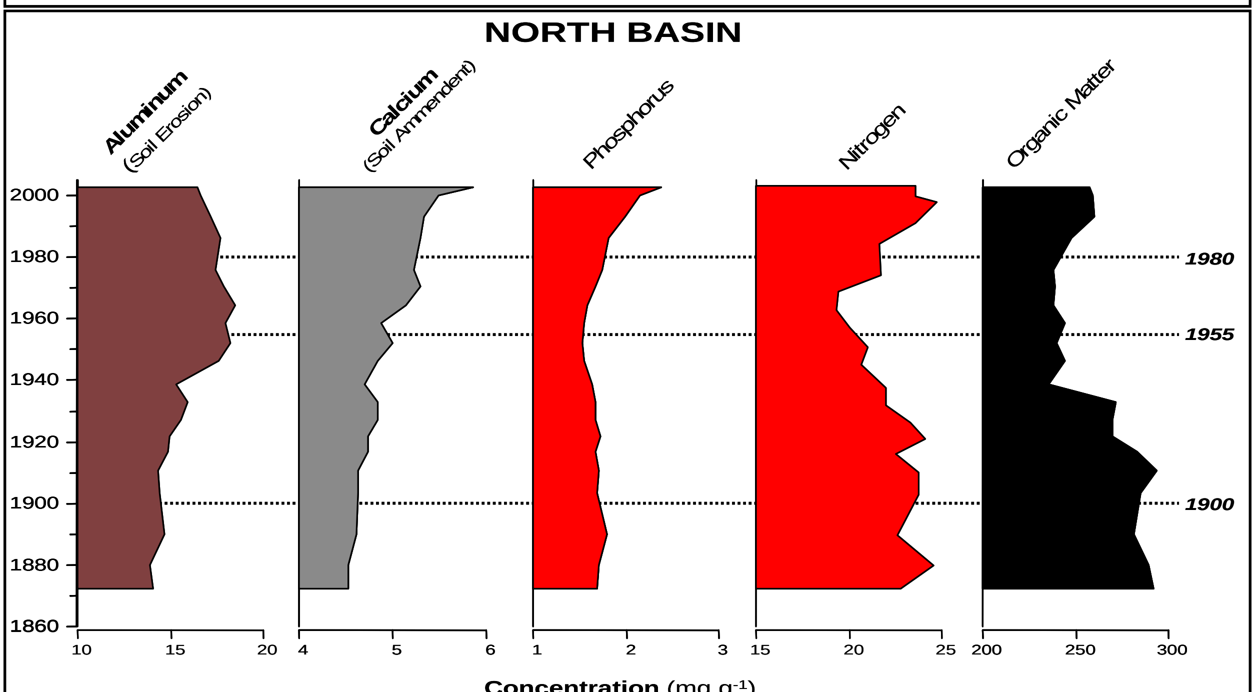


Figure 6. Profiles of the concentration of selected geochemical elements in the North Basin core. The general trends of declining soil erosion since 1960 but increased deposition of calcium and phosphorus are similar to the South Basin.

creased at a faster rate than organic matter since 1980. This likely indicates that like phosphorus, nitrogen is entering the lake from anthropogenic sources such as shoreline development.

In the North Basin the concentration profiles are generally similar to those observed in the South Basin. Concentrations were generally unchanged until the twentieth century (Figure 6). Soil erosion (Al) increased as in the South Basin although it began later, around 1940. Since 1960, the rate has been steadily declining much as was observed in the other basin. Calcium concentrations increase at the top of the core, likely as a result of its application on lawns. The nutrient phosphorus has been increasing since about 1980 (Figure 6), with highest concentration occurred at the top of the core. Nitrogen also showed a similar increase at the top of the core.

As the bottom waters become increasing devoid of oxygen, manganese (Mn) is mobilized from the sediments. This manganese then moves into the deepest waters resulting in enrichment of manganese in the sediments of the deeper waters. While this also occurs with iron, it happens sooner with manganese and manganese tends to stay in solution longer (Jones and Bowser 1978). Therefore as the bottom waters loss oxygen manganese is preferentially moved with respect to iron (Engstrom et al. 1985). The result is that with the loss of oxygen, the ratio of iron to manganese (Fe:Mn) declines (Mn increases). Figure 7 shows the profiles of Fe:Mn in the cores from both basins. In both basins the ratio increases during the middle of the twentieth century as a result of the increase in soil erosion. Iron,

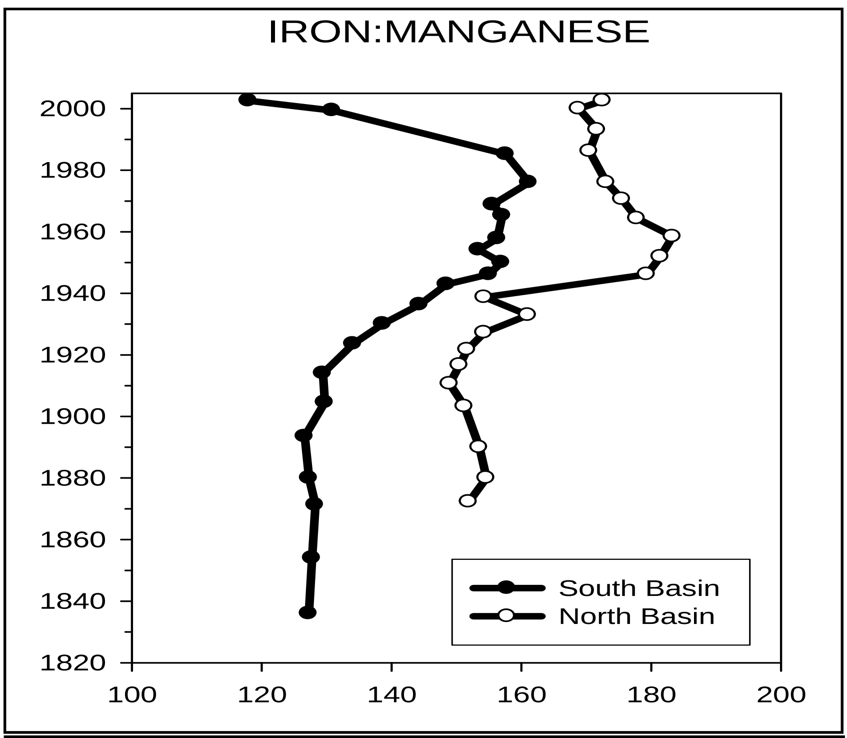


Figure 7. Profiles of Fe:Mn for both basins. The decline in the ratio in the South Basin since 1985 indicates that the bottom waters are

and to a lesser extent manganese, are found in soils so their concentrations increase with higher soil erosion rates. In the deeper South Basin, the ratio has been steadily declining since 1985. This indicates that during the last 20 years, the bottom waters are beginning to lose oxygen. The decline in hypolimnetic oxygen is a classic sign of increased eutrophication of a lake. The North Basin has not experienced a decline in the Fe:Mn which indicates that oxygen levels are not declining in this basin. Although oxygen is present in the bottom waters of both basins throughout the year, the sediment core indicates that in the South Basin there is a loss of oxygen very near the sediment water interface for at least part of the year. This loss has only been occurring during the last couple of decades and is an indication an increase in the lake's productivity.

Diatom Community

Aquatic organisms are good indicators of water chemistry because they are in direct contact with the water and are strongly affected by the chemical composition of their surroundings. Most indicator groups grow rapidly and are short lived so the community composition responds rapidly to changing environmental conditions. One of the most useful organisms for paleolimnological analysis is diatoms. They are a type of alga which possess siliceous cell walls and are usually abundant, diverse, and well preserved in sediments. They are especially useful as they are ecologically diverse and their ecological optima and tolerances can be quantified. Certain taxa are usually found under nutrient poor conditions while others are more common under elevated nutrient levels. They also live under a variety of habitats, which enables us to reconstruct changes in nutrient levels in the open water as well as changes in benthic environments such as aquatic plant communities. Figure 8 shows photographs of three diatom species that were common in the sediment cores.

In both basins, the dominant diatoms are those found in the open water of the lake, although in the North Basin diatoms that grow on the lake bottom or on macrophytes are an important part of the community. This is not surprising since Whitefish Lake is a deep and relatively large lake with a large pelagic zone. The most common diatoms in both cores are *Aulacoseira ambigua* and *Cyclotella glomerata* (Figure 9) which are indicative of low nutrients, especially phosphorus. There is an increase in *C. glomerata* during the period of 1900 to 1930 in both basins. This likely indicates a disturbance in the watershed. This may have been logging followed by early cottage construction.

The diatoms *Fragilaria crotonensis* and *Asterionella formosa* are indicative of moderate nutrient levels. The increase in these diatoms after 1955 indicates that there has been a slight increase in nutrients since that time. This is likely the result of shoreline development. An indication that the increase in phosphorus in small was the fact that the diatom *Cyclotella comensis* was not present in the core.

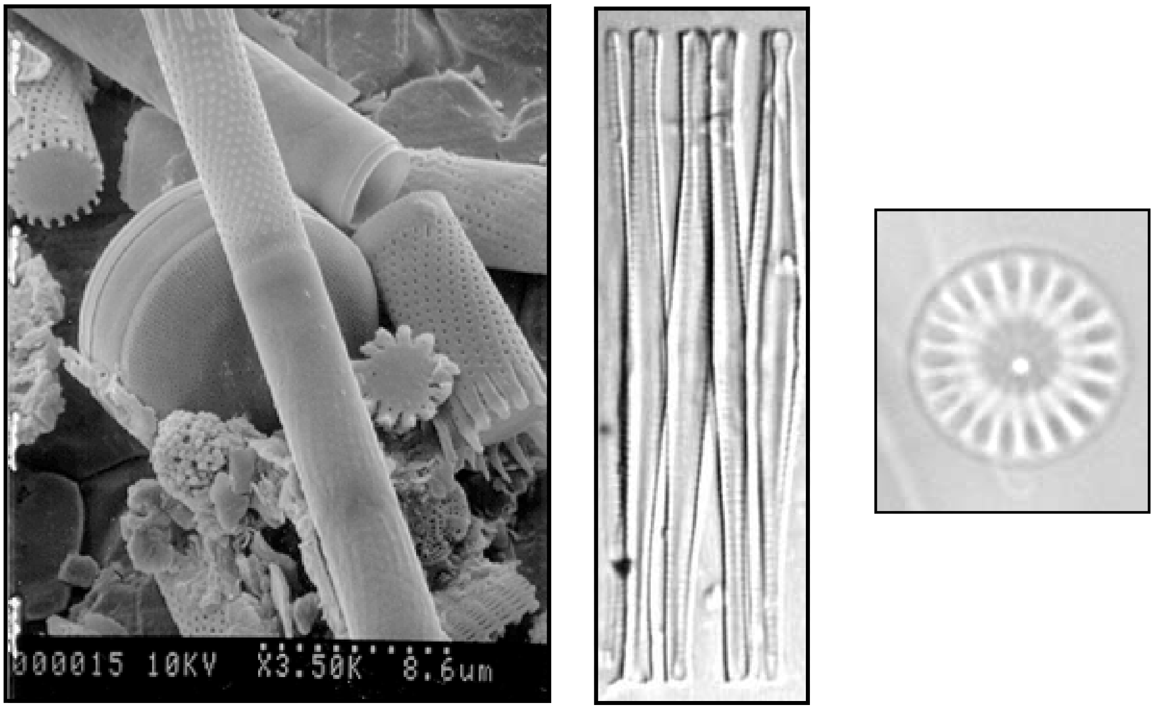


Figure 8. Micrographs of the diatoms *Aulacoseira* (left), *Fragilaria crotonensis* (middle), and *Cyclotella glomerata* which were common in the cores from both basins. All of these diatoms are typically found floating in the open water. The *Aulacoseira* pictured is commonly found in low nutrient waters, *F. crotonensis* is usually found in waters with moderate levels of nutrients.

This diatom is thought to have been introduced into the Great Lakes from northern Europe during the 1950s (Stoermer et al. 1985; 1990; 1993). *C. comensis* seems to indicate slightly elevated phosphorus levels (Schelske et al. 1972; Wolin and Stoermer 2005). It has been found in a number of northern Wisconsin lakes that possess good water quality but are showing signs of eutrophication (Garrison 2005a,b).

Another significant change in the diatom community was the increase in benthic *Fragilaria* (Figures 9 and 10). These diatoms typically grow on aquatic plants. This indicates there has been an increase in the density or type of macrophytes. This is especially evident in the core from the North Basin. It would be expected that this would be most evident in the North Basin since the coring site is closer to shallower water where plants would be expected to grow. Since around 1955, more of the macrophytes have been the type that have larger leaves and grow higher in the water column. This is also evident by the decline in planktonic diatoms starting in the 1950s and continuing until the present time. This increase in plant growth most likely is the result of shoreline development. Cores from other lakes in northern Wisconsin have shown that one of the most common impacts of shoreline development has been an increase in macrophyte growth (Fitzpatrick et al. 2003; Garrison 2005a, b;

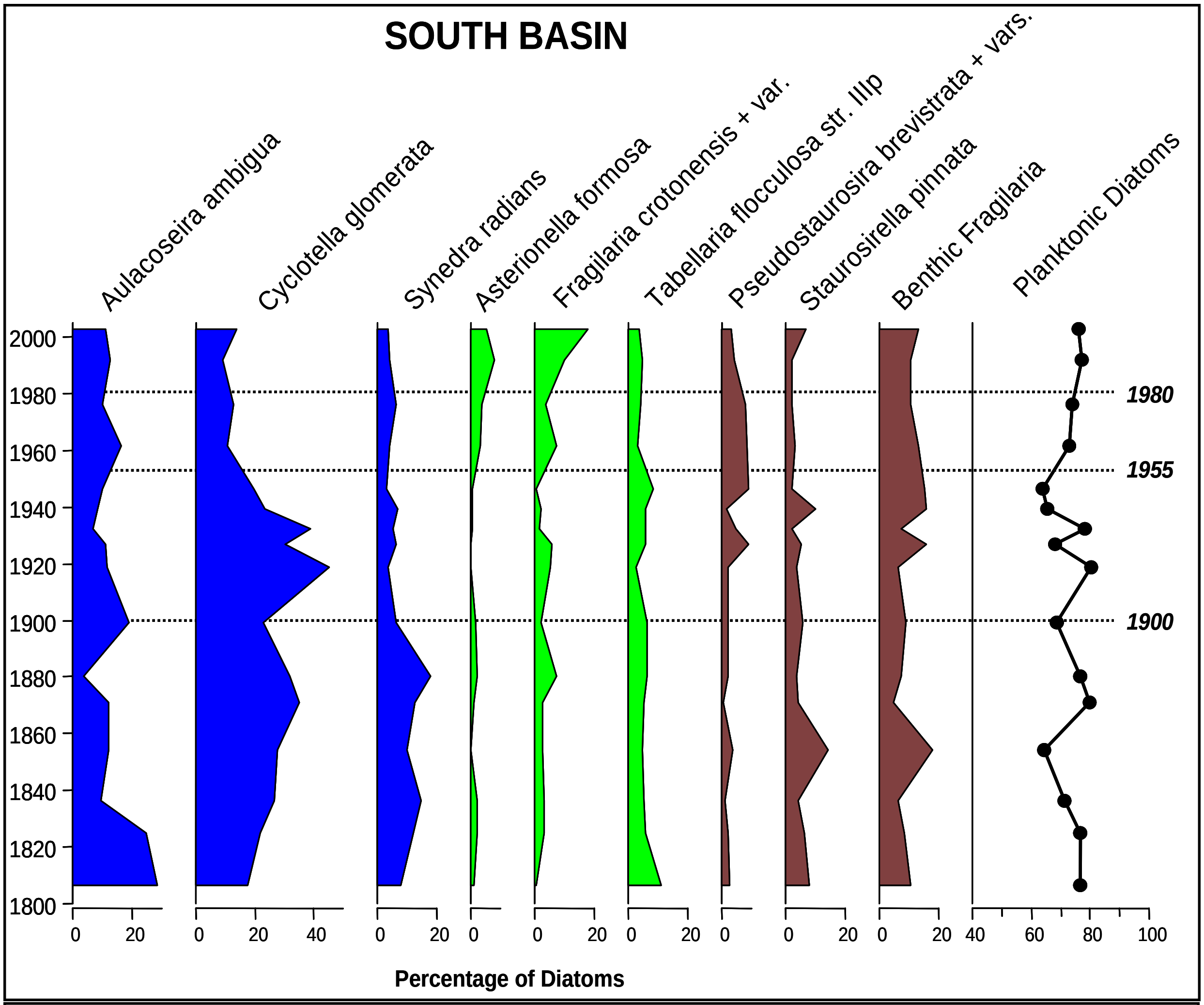


Figure 9. Profiles of the most common diatoms in the South Basin core. The taxa in blue are indicative of low nutrients. The diatoms in green indicate moderate nutrient levels. The diatoms depicted in brown grow attached to aquatic plants and are indicative of the amount of macrophytes.

Garrison 2006).

Diatom assemblages historically have been used as indicators of trophic changes in a qualitative way (Bradbury, 1975; Anderson et al., 1990; Carney, 1982). In recent years, ecologically relevant statistical methods have been developed to infer environmental conditions from diatom assemblages. These methods are based on multivariate ordination and weighted averaging regression and calibration (Birks et al., 1990). Ecological preferences of diatom species are determined by relating modern limnological variables to surface sediment diatom assemblages. The species-environment relationships are then used to infer environmental conditions from fossil diatom assemblages found in the sediment core.

The diatom inferred phosphorus levels in both cores at the surface where somewhat higher than ac-

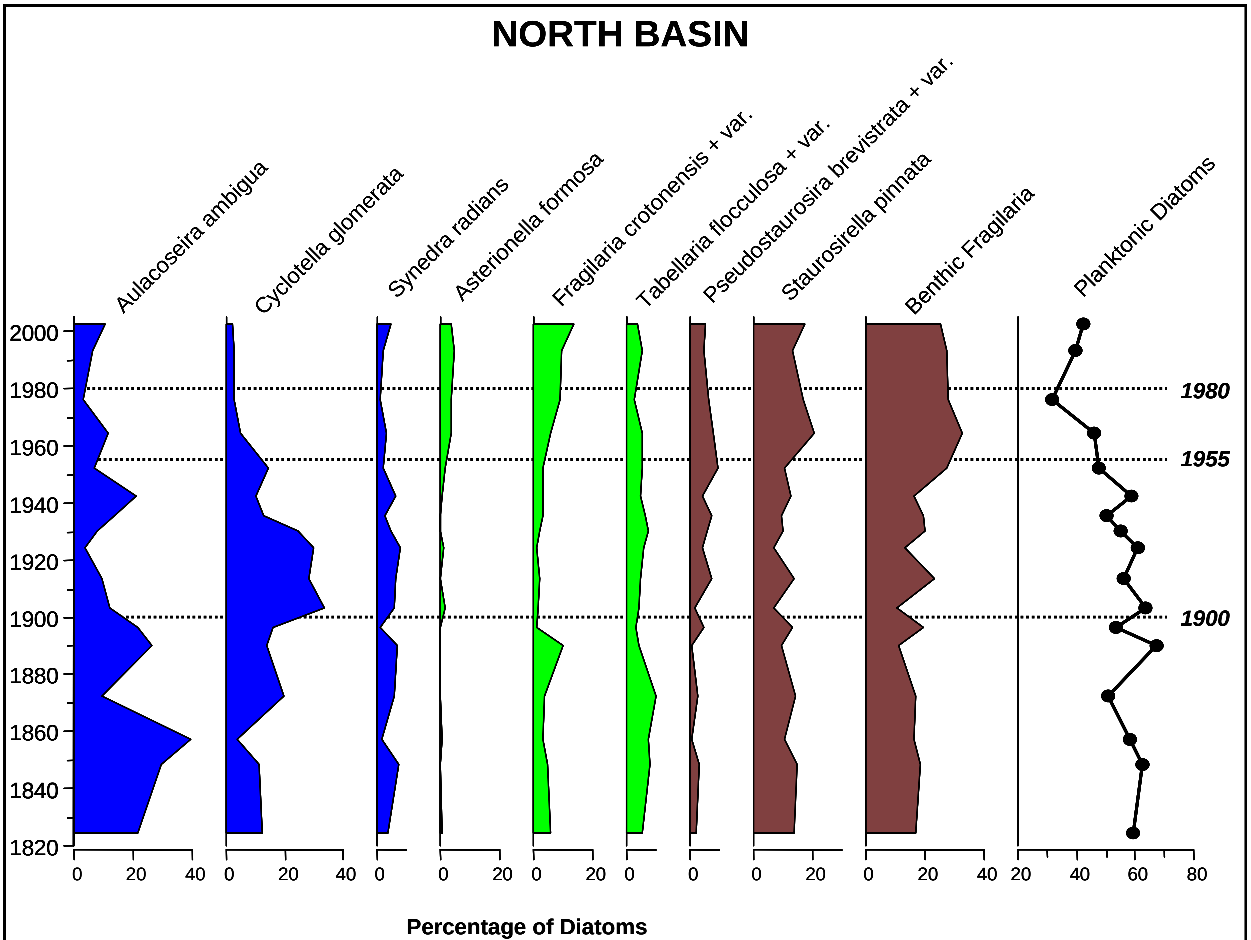
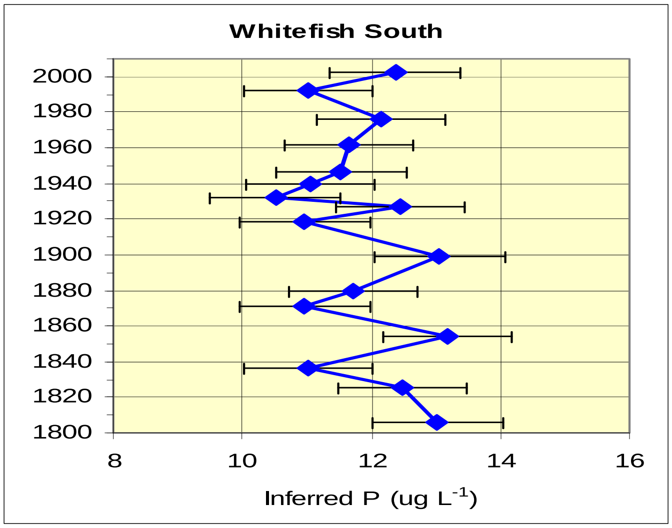


Figure 10. Profiles of the most common diatoms in the North Basin core. The changes in the diatom community at this site is similar to the South Basin although there is a higher percentage of benthic *Fragilaria* because the coring site was closer to the shore.

tual measured values. The model overestimates the phosphorus levels because there are few lakes in the calibration data set that have phosphorus values similar to Whitefish Lake. The measured values are 6-9 µg L^{-1} while the diatom inferred values are 12 µg L^{-1} in the South Basin and 18 µg L^{-1} in the North Basin. Even with the estimated values are higher than the measured values, the predicted direction of the trends should be reasonably accurate. In the South Basin, phosphorus levels changed little throughout the core. Concentrations varied from 11 to 13 µg L^{-1} (Figure 10). In the North Basin the concentrations during the 1800s were about 12 µg L^{-1} (Figure 11). Phosphorus levels began to increase around 1960 and reached their highest values near the top of the core. The predicted surface value is 18 µg L^{-1} which is higher than measured values. The higher phosphorus values are largely driven by the increase in benthic *Fragilaria* which prefer a wide range of phosphorus concentrations and therefore often predict higher concentrations (Wilson et al. 1997; Bennion et al. 2001). It is likely that phosphorus levels have increased in the North Basin in recent years but probably no more than 2-3 µg L^{-1}.



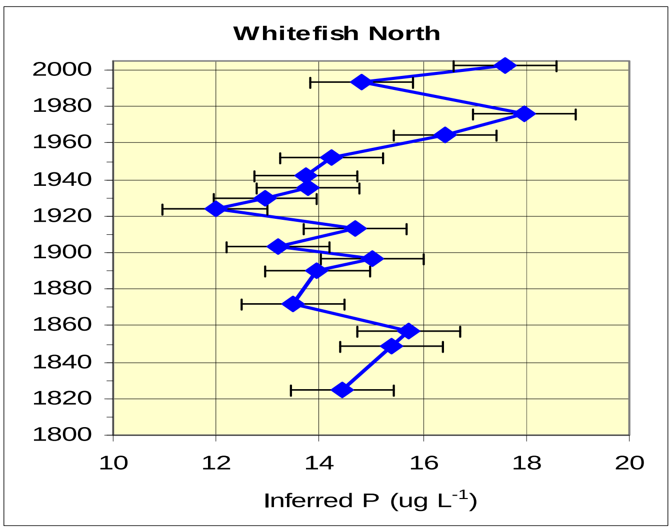


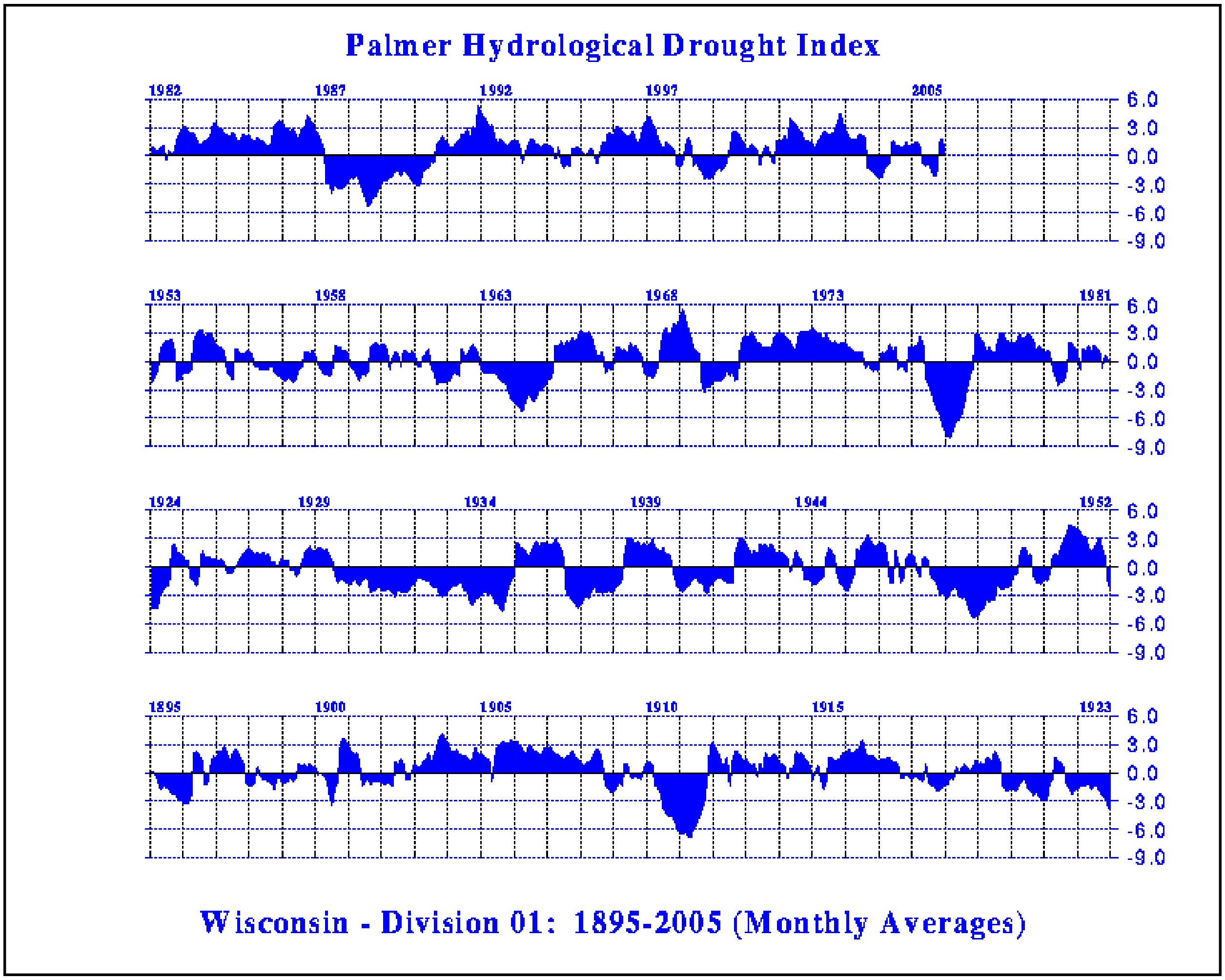
Figure 11. Diatom inferred phosphorus values. The model over predicts the surface phosphorus concentrations but the direction of the trend should be accurate. It is likely that the predicted increase in phosphorus in the North Basin since 1960 is too high. It is likely that the phosphorus level has increased 2-3 µg L^{-1}.

Impact of Water Level Changes

One of the purposes of conducting the present study was to estimate if changing water levels in Whitefish Lake have an adverse affect upon the lake's water quality. Specifically, it was hypothesized that higher water levels would result in the addition of nutrients, e.g. phosphorus, to the lake as vegetation was flooded and consequently decayed.

The U.S. Department of Commerce has compiled the Palmer Hydrological Drought Index (PHDI) for the northwestern part of Wisconsin since 1895. This index estimates the duration and intensity of longterm drought-inducing circulation patterns. The intensity of droughts are reflected in the PHDI and estimate the effect of droughts on groundwater levels and thus the impact on lake levels in groundwater dominated systems like Whitefish Lake.

Figure 12 shows the PHDI for the period 1895-2005 for the region that includes Whitefish Lake. The PHDI indicates that Whitefish Lake should have had higher waterlevels for a number of periods during the last 100 years. It is expected that when lake levels are low in Whitefish Lake there should be a higher percentage of benthic diatoms and consequently a higher percentage of planktonic diatom during high water years. This trend is most likely to be reflected in the cores from the North Basin since this basin has a larger littoral area than the South Basin. There is a slight indication that between the period of 1880 and 1940 that there is a decline in benthic *Fragilaria* (Figure 10) during periods of higher PHDI values. However, since the late 1940s there is no relationship. This indicates that anthropogenic activities during the last 60 years have a much greater impact on the diatom community than do waterlevels.



http://climvis.ncdc.noaa.gov/cgi-bin/ginterface

Figure 12. The Palmer Hydrological Drought Index for northwestern Wisconsin for the period 1895-2005. This index should reflect groundwater levels around Whitefish Lake. When the index is high, it is expected that water levels would be elevated in the lake and when the index is low water levels in Whatfish Lake should be lower.

In summary, the sediment cores from Whitefish Lake indicate that the lake possesses an oligotrophic trophic state but changes have occurred in the lake during the last century. The mean sedimentation rate is very low in both basins. In fact they are the lowest that I have measured in 42 lakes in Wisconsin. However, in the North Basin the sedimentation rate at the present time is higher than that measured during the 1800s. In both basins the soil erosional rates peaked around 1960 and rates have steadily declined since then. While sediment delivery has declined during the time period, there has been an increase in phosphorus deposition. Along with the increase in phosphorus was an increase in calcium. Since this frequently is used as a soil amendment in lawns it is likely that lawn maintenance is at least partially responsible for the increased phosphorus. Another early sign of eutrophication is the decline in oxygen levels in the bottom waters in the South Basin.

The diatom community indicates that the lake has low nutrient levels but there has been a small increase in phosphorus in recent years. The increase in the abundance of *F. crotonensis* and *A. formosa* since 1955 supports the indication of eutrophication from the declining oxygen levels in the deep waters. The diatom community also indicates there has been an increase in the density of the macrophyte community since 1955. This likely means macrophytes now present in the lake possess larger leaves and grow higher in the water column. This has been found in almost all northern Wisconsin lakes which have shoreline development.

While Whitefish Lake possesses very good water quality with excellent clarity and low algal populations, the core study indicates that eutrophication is occurring, especially during the last 25 years. Steps should be taken now to protect the lake's water quality since it is more difficult to reverse eutrophication once the lake has been degraded.

- The mean sedimentation rate for the last 150 years for Whitefish Lake in both basins is the lowest measured in Wisconsin lakes.

- In the South Basin, the sedimentation rate was largely unchanged from the 1830s until 1920. It declined until 1970 and then increased to the top of the core.

- In the North Basin the highest sedimentation rate occurred after 1980. For the last 25 years the rate has increased with the highest rate occurring during the last 5 years.

- Aluminum, which is indicative of soil erosion, increased during the first half of the twentieth century. Since 1955, soil erosion rates have been declining in both basins.

- There has been an increase in phosphorus deposition in both basins during the last 25 years. Since this is associated with calcium, a soil amendment for lawns, the increased nutrients is probably from shoreline development.

- The diatom community indicates that phosphorus levels in the North Basin have increased about 2-3 $\mu g\ L^{-1}$in the last 40 years. Phosphorus levels in the South Basin are unchanged during the last 200 years, While nutrient levels increased slightly or not at all, the most significant change in the lake was an increase in the growth of aquatic plants since 1960.

- Perhaps the most significant indication of the pressures exerted by shoreline development is the decline in oxygen levels in the bottom waters of the lake. This has not occurred in the North Basin but it began around 1980 in the South Basin.

- The cores show that Whitefish Lake has been impacted by shoreline development beginning around the 1920s. Runoff from development led to an increase in macrophytes in the 1960s. This impact has accelerated during the last 15 years resulting in an increase in sediment infilling and loss of oxygen in the bottom waters. If this trend continues, the lake will be further degraded with nutrient levels increasing, resulting in a loss of water clarity.

- Fluctuating lake water levels have had minimal impact on phosphorus levels in the lake. Since 1950, shoreline development has had a much larger impact than fluctuating water levels.

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Appendix 1. South Basin sedimentation rates and dates for the CRS and piecewise CRS models.

Depth of Core (cm)		Sedimentation Rate (mg cm^{-2} yr^{-1})		Year at Interval Bottom	
Interval top	Interval bottom	CRS	Piecewise CRS	CRS	Piecewise CRS
0	1	8.7	8.2	2003	2003
1	2	9.0	7.7	2000	1999
3	4	10.6	5.2	1994	1985
4	5	11.1	4.7	1991	1976
6	7	11.2	5.8	1985	1969
7	8	12.4	6.3	1981	1965
9	10	12.1	7.0	1975	1958
10	11	12.8	7.3	1972	1954
11	12	11.3	7.5	1968	1950
12	13	12.5	7.7	1965	1946
13	14	15.1	7.9	1962	1943
15	16	14.2	8.3	1957	1936
17	18	15.4	8.6	1952	1930
19	20	14.2	8.8	1947	1924
21	22	9.9	8.6	1941	1914
23	24	12.7	8.5	1935	1905
25	26	10.3	8.2	1929	1893
27	28	11.9	7.9	1923	1880
29	30	9.0	8.0	1916	1871
32	34	6.8	8.0	1897	1854
36	38	3.1	8.0	1880	1836

Appendix 2. North Basin sedimentation rates and dates for the CRS and piecewise CRS models.					
Depth of Core (cm)		Sedimentation Rate (mg cm^{-2} yr^{-1})		Year at Interval Bottom	
Interval top	Interval bottom	CRS	Piecewise CRS	CRS	Piecewise CRS
0	1	8.5	9.3	2002	2003
1	2	8.3	9.8	1999	2000
2	3	7.5	6.8	1995	1993
3	4	8.0	6.0	1991	1986
4	5	6.6	5.0	1986	1976
5	6	6.9	5.3	1980	1971
6	7	6.5	5.5	1974	1964
7	8	6.9	5.6	1969	1958
8	9	6.2	5.6	1962	1952
9	10	6.8	5.7	1957	1946
11	12	11.3	5.9	1950	1939
13	14	11.8	5.8	1944	1933
15	16	11.9	5.8	1939	1927
17	18	13.0	5.8	1934	1922
19	20	14.8	5.8	1930	1917
21	22	12.0	5.8	1925	1911
23	24	10.6	5.7	1918	1903
25	26	6.0	5.4	1908	1890
27	28	14.0	5.3	1900	1880
29	30	6.5	5.3	1890	1872

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