Twenty Years of Water Quality Sampling in Devils Lake, With a Specific Focus on the Effect of Water Levels, 1995 -2014

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1.0 Introduction

Devils Lake is a terminal, polymictic lake in northeastern North Dakota. Devils Lake has been subject of extreme fluctuations in lake elevation throughout its history, with the most recent being an almost 30-foot increase from 1993 to 2008 (data from USGS) (Figure 1). Around 1830, Devils Lake reached a lake elevation of about 1,846 feet above sea level (asl), according to Upham's (1895) account of Devils Lake "having attained, about the year 1830, a level of 16 feet higher than its low stage in 1889." Water level in 1889, however, was not the lowest on record, as the lake's elevation decreased to about 1,400 feet asl in 1940 with an average depth of about 0.7 meters (Swenson and Colby, 1955). After 1940, Devils Lake experienced several dramatic increases and decreases in water level, with a generally upward trend. The volume of water in Devils Lake has increased most recently, highlighted by an increase of greater than 200,000 acre-feet in 1993 (Williams-Sether, 1996).

Two primary drainages in the basin are Channel A and Big Coulee. Channel A drains the Dry Lake, Edmore, Starkweather, and Sweetwater areas, while Big Coulee drains Lake Irvine, Lake Alice, Chain of Lakes, and Mauvais Coulee. Spring runoff is the major source of water, with the majority of runoff entering the system through West Bay via Big Coulee (naturally) and Six Mile Bay via Channel A. According to the U.S. Geological Survey (USGS), about 85 percent of the water contributed to Devils Lake enters through these two sources (Vecchia, 2002). Extreme fluctuations in water level throughout the Devils Lake basin was formerly seen as a result of increased cultivation in the region influencing water retention (Pope, 1908), whereas today the fluctuation is thought to be more an effect of drastic changes in the cycling of precipitation and evaporation (Pusc, 1993) with groundwater inputs and outputs having a minimal but constant effect (Wiche and Pusc, 1993). Further, there was a 426% increase in rural wetland area in Nelson County from 1992 to 2001due to a shallower water table (Todhunter and Rundquist, 2004), likely having a positive impact on water quality and biological communities throughout the Devils Lake region.

When water levels are elevated, Devils Lake naturally drains to Stump Lake (at 1,447 feet asl via the Jerusalem Outlet) and the Sheyenne River (above 1,459 ft asl via the Tolna Coulee). In the early 2000s, Devils Lake began contributing to Stump Lake and has continued to rise to 1,451.6 feet asl as of December 31, 2014 (USGS). Mean annual precipitation increased from 18.3 inches from 1950-1979 to 22.4 inches from 1980-2006 (Vecchia, 2008), contributing to the dramatic increase in water levels over the past twenty years. The absence of a natural surface outlet at lower lake elevations and annual evaporation (30.9 inches per year from 1980-2006 [Vecchia, 2008]) frequently exceeding annual precipitation are important causes of the high ionic concentrations and changes in nutrient concentrations.

Historically, ion concentrations in Devils Lake tend to have an inverse relationship with water level (Fritz, 1990; Wiche et al., 2000). For example, recorded concentrations of specific conductance in Main Bay have ranged from 11,300 µS/cm in 1954 to 2,560 µS/cm in 1955, increasing to 9,400 µS/cm by the end of 1959 (Mitten et al., 1968), then decreasing to about 5,400 µS/cm in the early 1990s (Lent, 1994), thus demonstrating the fluctuating concentrations of ions in Devils Lake. Further, nutrient concentrations in Devils Lake (e.g., nitrogen [N] and phosphorus [P]) show high inter- and intra-annual variability, likely due to internal cycling as many nutrients attach to sediments which sporadically are re-suspended into the water column. For example, Lent (1994) found total phosphorus (TP) concentrations in Main Bay varied from 210 µg/L in July to 90 µg/L in October. Similarly, Lent (1994) found that total ammonia (as N) decreased from 200 µg/L to 30 µg/L from July to October. These nutrients play a vital role in plant and algal production in lakes; for example, N:P ratios less than 30:1 are more likely to experience a higher biovolume of cyanobacteria (Jacoby et al. 2000). Such drastic changes in ionic and nutrient concentrations likely have a significant impact on algal communities in Devils Lake.

Historically, there have been few investigations into the phytoplankton community in Devils Lake. Verch and Blinn (1971) found that cyanobacteria species *Aphanizomenon flos-aquae* and *Microcystis aeruginosa* dominated the photic zone from mid- to late-summer, with green algae species dominating more in the spring months. Similarly, Leland and Berkas (1998) found that cyanobacteria species dominated phytoplankton samples during summer months and found inter-annual spring dominance by chrysophytes, chlorophytes, and cryptophytes. Diatom species appear to be relatively scant and variable in Devils Lake (e.g., Verch and Blinn, 1971; Leland and Berkas, 1998), which may be due to relatively high concentrations of specific conductance inhibiting growth of these species (Blinn, 1993).

A popular fishing destination in the State and the Midwest region, Devils Lake is managed for many common game species, including northern pike, walleye, and yellow perch (ND Game and Fish). At times, however, northern pike have completely disappeared from Devils Lake, attributed to loss of spawning grounds due to desiccation and relatively high ion concentrations (Pope, 1908). Historically, Mauvais Coulee (which dried up in 1889) was thought to play a key role in reproduction of northern pike, likely due to “fresher” water from the north entering the lake (Pope, 1908).

The Devils Lake Water Quality Monitoring Project (DLWQMP) was initiated to characterize and analyze chemical, physical, and biological properties of Devils Lake. This project specifically focuses on the effect that aforementioned fluctuations in water volume have on ionic and nutrient concentrations in Devils Lake. Results from this assessment will help to understand the effect of water level dynamics on water quality in Devils Lake, and to anticipate potential effects on chemical, physical, and biological properties of the Lake as lake elevation continues to change.

2.0 Methods

2.1 Sampling sites

The DLWQMP began in March 1995 with six monitoring sites: Six Mile Bay (380221), Creel Bay (380222; ended February 1999), Main Bay (380233), East Bay (380234), East Devils Lake (380235), and West Bay (380236). Since that time, additional sites were added throughout the Devils Lake chain, including: Southeast East Bay (380237; began May 1996, ended February 1999), East Minnewauken Flats (380250; began October 1995, ended August 2000), West Minnewauken Flats (380251; began October 1995, ended February 1999), Southwest West Bay (384160; began August 1997, still an active site), and Pelican Lake (385029; began May 1999, still an active site). Sites that are currently sampled are listed in Table 1.

2.2 Water Quality Sampling

During the period from 1995 through 2014, annual sampling effort typically resulted in four sampling trips: winter (January-March), spring (in May), summer (July-August), and fall (October). From 1996 through 1998, however, three summer samples were taken for each site (one each in July, August, and early-September). Water quality samples were collected at each site for total nutrients (particulate and dissolved), metals, cations/anions, and chlorophyll-α in accordance with procedures in NDDoH Standard Operating Procedures (http://www.ndhealth.gov/). Temperature-dissolved oxygen profiles were taken at 1-meter intervals from surface to bottom using a YSI multi-parameter water quality sonde. A Secchi disk was used to measure light transparency to help determine the effect algae and/or suspended sediments have on the photic zone. For open-water samples (May-October), chlorophyll-α samples were taken with a two-meter water column sampler. Surface water quality samples were collected for each trip, preserved, and analyzed by the North Dakota Department of Health, Chemistry Division laboratory. Analytes measured are provided in NDDoH (2011).

Table 1: List of sites currently sampled as part of the Devils Lake Water Quality Monitoring Project with coordinates.

Station ID	Description	Latitude	Longitude
380221	Six Mile Bay	48.10530	-99.02549
380233	Main Bay	48.03265	-98.95333
380234	East Bay	48.05927	-98.84115
380235	East Devils Lake	47.95486	-98.64633
380236	West Bay	48.01335	-99.11192
384160	Southwest West Bay	48.04898	-99.21235
385029	Pelican Lake	48.14634	-99.16399

Temperature and dissolved oxygen profiles were recorded annually, but only years 2010 through 2014 are presented in this report to show variance and continuity among years. Due to the Devils Lake water level rising nearly 20 feet since the inception of this project, Pearson, two-tailed correlations were used to relate lake elevation to measured analytes. Lake elevation can also be used as a surrogate for increasing time because of their high correlation ($R > 0.85$).

To analyze trophic state of each basin, TSI scores were calculated using total phosphorus (TP), Secchi disk (SD) transparency, and chlorophyll-α (Carlson, 1977). TSI scores were then compared using methods from Carlson (1992) to measure inconsistencies among trophic state scores (i.e., chlorophyll-α).

Phytoplankton community similarities were analyzed using dendrogram plots for relatedness (using Bray-Curtis similarities) of specific sampling events (Oksanen et al., 2013). All statistical tests and figures were performed and created using R software (2013).

Average daily lake elevations were downloaded for USGS gage station 05056500 (from waterdata.usgs.gov/) for all available data going back to 1930.

3.0 Results

From 1995 through 2014, as part of the DLWQMP, water quality parameters were measured within Devils Lake at least four times per year (at least once seasonally). There were, however, some instances where stations could not be sampled (e.g., unsafe water conditions, no access to site), so not all stations were sampled on an annual or seasonal basis. Despite this, for every season and station, there seems to be an adequate sample size for statistical comparison of measured ions and nutrients across a gradient of water levels.

Over the study period, water levels in Devils Lake had a range of about 23 feet, ranging from 1,431 feet (winter 1995) to 1,454 feet (spring 2011) (Figure 1), with annual rises of two or more feet asl occurring five times in the past twenty years. Over this time period, these fluctuating water levels appear to have a strong effect on ion concentrations in the lake.

3.1 Land Cover Change

According to information from the National Land Cover Dataset (NLCD) (1992, 2001, 2006, 2011) the Devils Lake sub-basin (8-digit HUC 09020201) changed most throughout the 1990s. The amount of open water (including Devils Lake itself) increased in the Devils Lake watershed from 3.16% in 1992 to 9.18% by 2001 (Table 2). Wetlands in the watershed (mostly as emergent, herbaceous wetlands) increased from 7.92% of all cover in 1992 to 10.74% in 2001 (Table 2). There was a corresponding decrease over the same time with cultivated crops (64.09% in 1992, 59.65% in 2001) and grassland (14.72% in 1992, 6.88% in 2001) (Table 2). While there were tremendous increases in open water and wetland area from 1992 to 2001, there was relatively little increase in wetland area from 2001 to 2011 (Table 2).

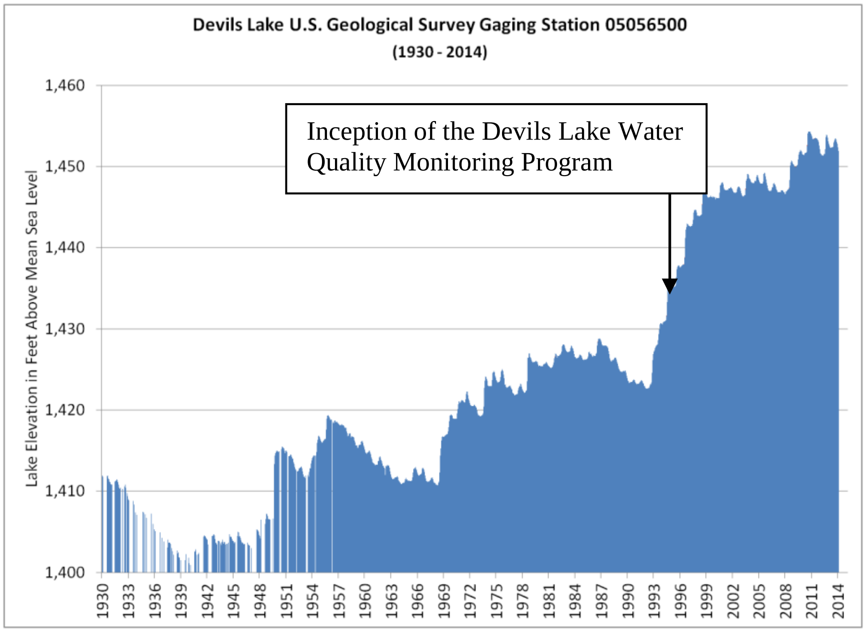

Figure 1: Average daily elevation of Devils Lake at USGS Station 05056500 near Devils Lake from October 1930 to November 2014.

Table 2: Land use classes and percentages in the Devils Lake drainage area (8-digit HUC 09020201) for 1992, 2001, 2006, and 2011.

Land Use Class	1992 NLC Percentage	2001 NLCD Percentage	2006 NLCD Percentage	2011 NLCD Percentage
Open Water[3]	3.16%	9.18%	9.49%	9.65%
Developed, Open Space	n/a[1]	3.71%	3.72%	3.70%
Developed, Low Intensity	0.12%	0.35%	0.35%	0.34%
Developed, Medium Intensity	n/a[1]	0.04%	0.05%	0.07%
Developed, High Intensity	0.04%	< 0.01%	< 0.01%	0.01%
Commercial	0.24%	n/a[2]	n/a[2]	n/a[2]
Barren Land	0.04%	< 0.01%	< 0.01%	< 0.01%
Deciduous Forest	2.61%	0.58%	0.57%	0.57%
Evergreen Forest	< 0.01%	0.05%	0.05%	0.05%
Grassland	14.72%	6.88%	6.75%	6.64%
Pasture/Hay	6.98%	8.41%	8.30%	8.25%
Cultivated Crops	64.09%	59.65%	59.87%	59.83%
Woody Wetlands	0.03%	0.39%	0.38%	0.38%
Emergent Wetlands	7.92%	10.74%	10.53%	10.57%

[1]Not represented in the 1992 NLC data
[2]Not represented after 1992 NLC data
[3]Includes Devils Lake itself

3.1 Six Mile Bay

The sampling station at Six Mile Bay (Station ID 380221) is about seven miles west of the City of Devils Lake (Figure 1.1). The station at Six Mile Bay has been active since 1995. This station is slightly protected from both east and west winds, which, theoretically, should make thermal stratification more common in Six Mile Bay during summer months. Much of the major inflow to Devils Lake enters through Six Mile Bay via Channel A, which, in conjunction with Big Coulee, supplies much of the surface water flow and plays a key role in lake elevation dynamics.

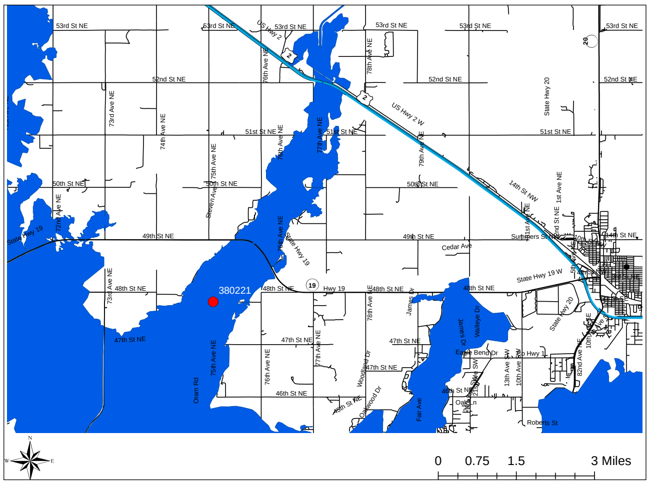

Figure 1.1: Map showing location of sampling site in Six Mile Bay.

Annual winter sampling

In the winter, Six Mile Bay is typically well-oxygenated to the bottom. Only once in the past five years (2014) was the dissolved oxygen (DO) less than the State's standard of 5 mg/L (Figure 1.2). There is some reverse thermal stratification during winter months, with three years since 2010 exhibiting a change of 1ºC m^{-1} (Figure 1.2). Near-sediment temperatures in Six Mile Bay in the winter are the warmest throughout the lake.

Concentrations of many analytes measured in the winter decreased from 1995 through 2014. Cations decreased with increasing lake elevation, driven by decreases in potassium ($R = -0.799$) and sodium ($R = -0.651$; Figure 1.3). Conversely, calcium (major cation) increased significantly with increasing lake elevation ($R = 0.502$). Anions also decreased, driven by chloride ($R = -0.900$) and sulfate ($R = -0.792$) (Figure 1.3). Specific conductance ($R = -0.856$; Figure 1.3) and total dissolved solids (TDS) ($R = -0.803$) decreased with increasing lake elevation.

Nutrient concentrations have not changed significantly with increasing water levels in Six Mile Bay. Concentrations of dissolved phosphorus (DP) ($R = 0.060$) and total phosphorus (TP) ($R = 0.117$; Figure 1.4) had no relationship with water levels in Six Mile Bay, with concentrations relatively high compared to the rest of the lake. Similarly, Nitrate + Nitrite (N+N) (R = -0.293), Kjeldahl Nitrogen (TKN) (R = -0.406), and Total Nitrogen (TN) (R = -0.094; Figure 1.4), were not significantly related to changing lake level.

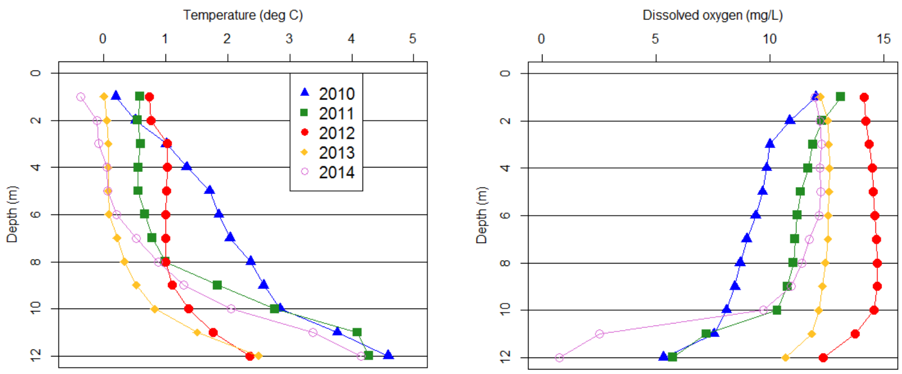

Figure 1.2: Winter temperature (left) and DO (right) profiles for Six Mile Bay from 2010 through 2014.

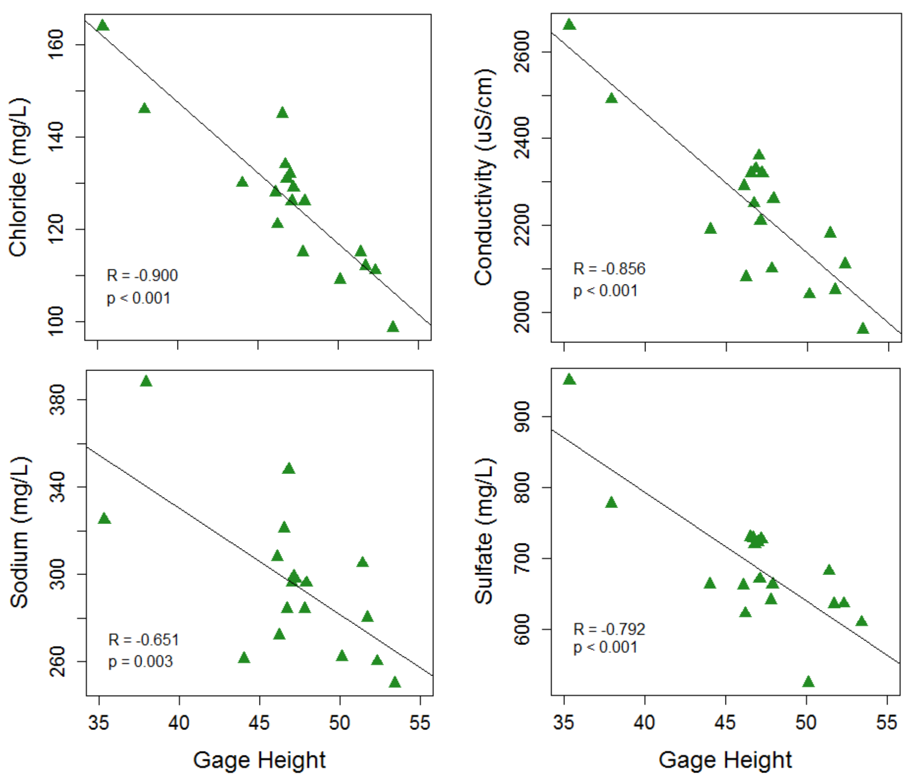

Figure 1.3: Winter concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Six Mile Bay. Displayed value for lake elevation represents values in the 1400s (i.e., 52.5 equals 1452.5).

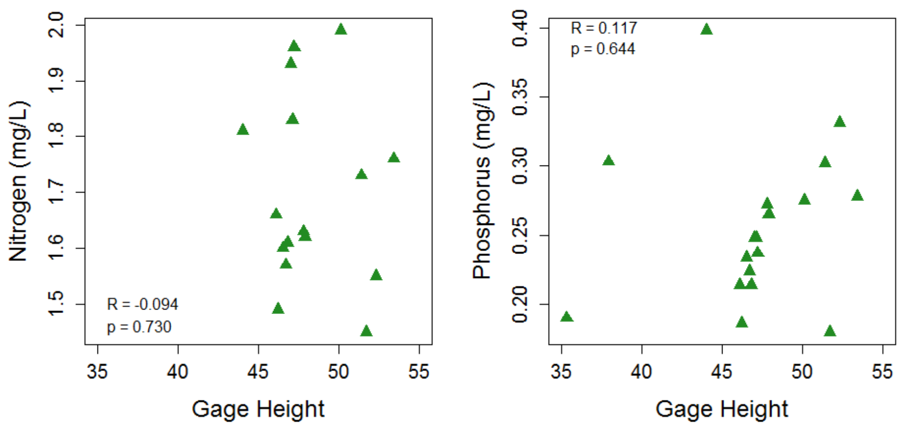

Figure 1.4: Winter concentrations of TN (left) and TP (right) plotted against lake elevation at Six Mile Bay.

Annual spring sampling

Six Mile Bay occasionally exhibits thermal stratification in the spring, with two of the past five years showing stratification (Figure 1.5). These two years (2010 and 2014) had above-average surface temperatures, likely indicating a warmer than average spring. Despite thermal stratification in the spring, Six Mile Bay is rarely below the State's standard for DO, and never during the past five years (Figure 1.5).

Many of the analytes that declined with increasing water levels in the winter increased with increasing water levels in the spring. Anions increased with increasing water levels, driven by an increase in sulfates (R = 0.495; Figure 1.6). Chloride, a major anion which strongly declined in the winter, had no relationship with water levels in the spring (R = 0.015; Figure 1.6). Cations also increased, driven by increases in barium (R = 0.734), calcium (R = 0.793), and magnesium (R = 0.635). There was also a nearly significant increase in sodium in the spring (R = 0.436; Figure 1.6). There was no relationship with specific conductance and water level (R = 0.320; Figure 1.6), while TDS did increase significantly (R = 0.508).

Like winter sampling, spring nutrient concentrations showed almost no relationship to water levels in Six Mile Bay. TN declined significantly with increasing water levels (R = -0.565; Figure 1.7), but neither TKN (R = -0.399) nor N+N (R = -0.193) had any relationship with lake elevation. TP (R = 0.002; Figure 1.7) and DP (R = 0.144) did not have a relationship with increasing water levels.

TSI scores in Six Mile Bay have decreased for both Secchi disk transparency and chlorophyll-α. TSI scores suggest that Six Mile Bay has transitioned from a highly eutrophic (nearly hypereutrophic basin) to a mesotrophic basin in the spring. The Secchi disk score has declined from a peak of 67.36 (0.6 meters [m]) in 1996 to lows of 35.17 (5.6 m) in 2010 and 37.70 (4.7 m) in 2014 (Figure 1.8). Chlorophyll-α TSI scores have followed the same general trend, though the decline has not been as strong and has exhibited much more variance among years. Chlorophyll-α TSI scores have declined from peaks of 61.36 (1997) and 63.97 (2001) to a low of 34.58 (2008) (Figure 1.8).

Comparison among TSI scores suggests that Six Mile Bay, like the rest of Devils Lake, is not phosphorus-limited (P-limited) in the spring (Figure 1.9). Over the past several years, Six Mile Bay appears to be switching from the photic zone being dominated by small particles of non-algal turbidity (evidenced by Secchi-TSI scores being greater than chlorophyll-TSI) to being dominated by larger particles of algae and suggesting strong zooplankton grazing (Figure 1.9).

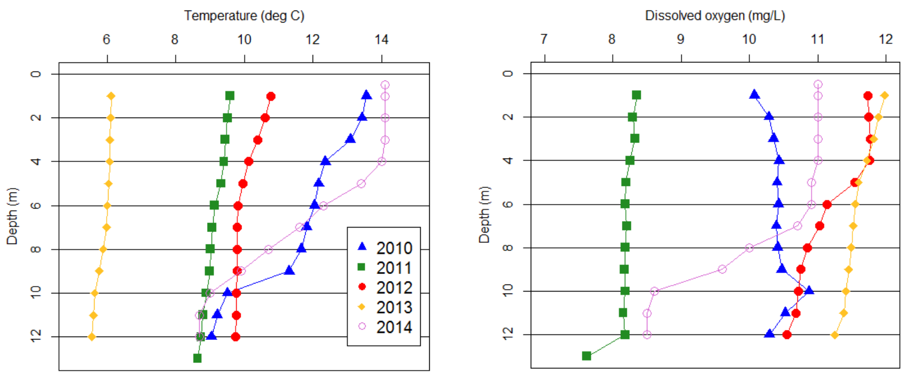

Figure 1.5: Spring temperature (left) and DO (right) profiles for Six Mile Bay from 2010 through 2014.

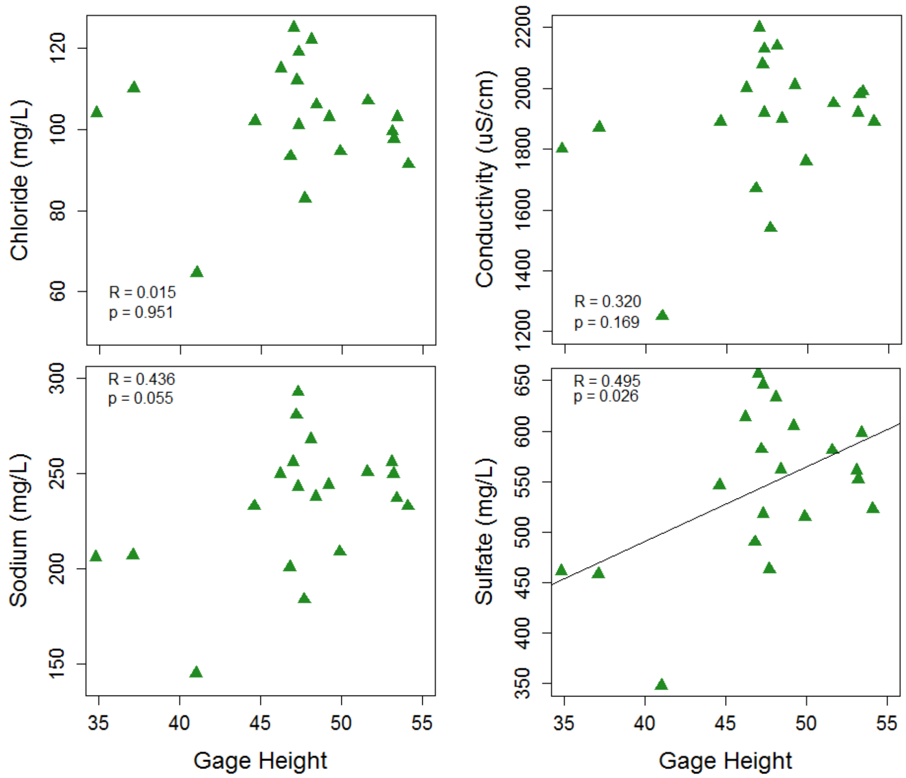

Figure 1.6: Spring concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Six Mile Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

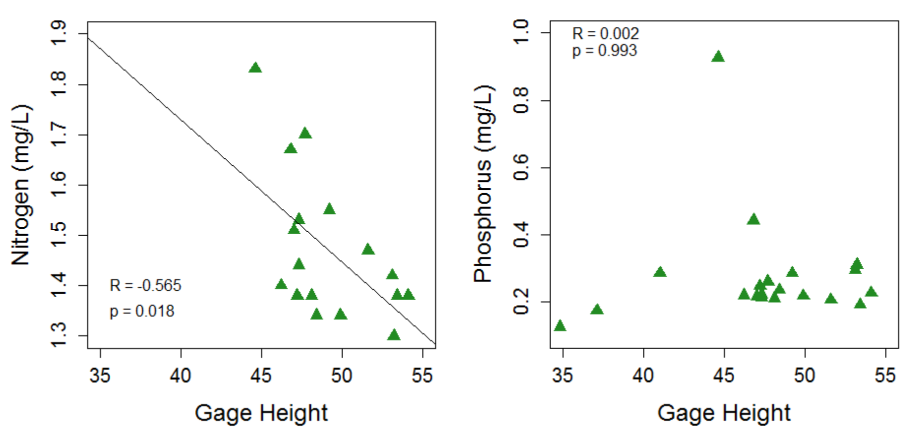

Figure 1.7: Spring concentrations of TN (left) and TP (right) plotted against lake elevation at Six Mile Bay.

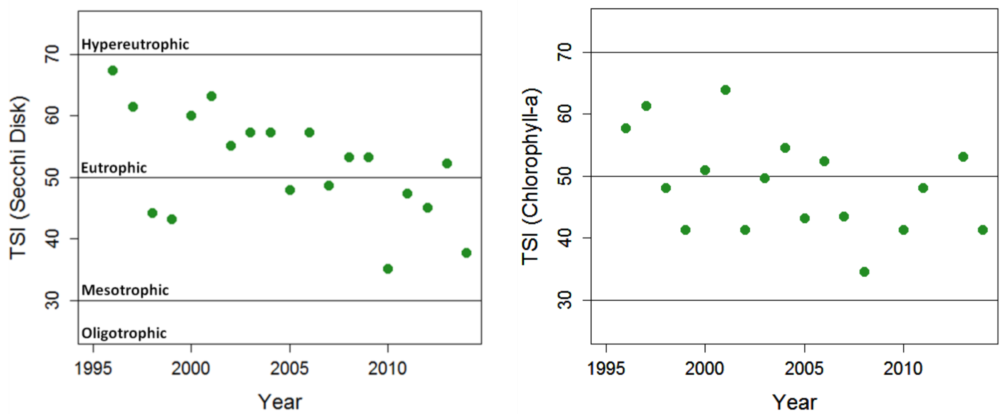

Figure 1.8: Spring TSI scores for Six Mile Bay from 1996 through 2014.

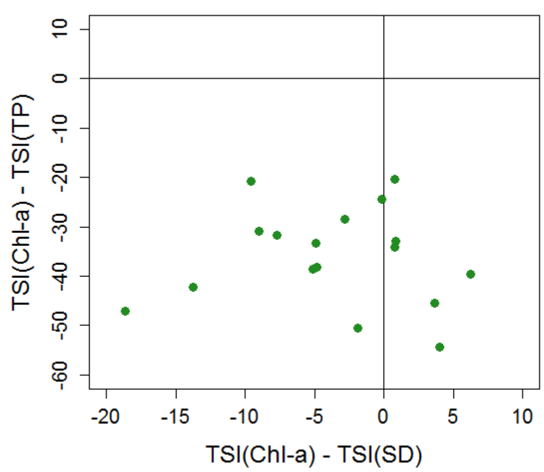

Figure 1.9: Nutrient-limitation graph for spring sampling at Six Mile Bay.

Annual summer sampling

Like other seasons, Six Mile Bay is not always stratified in the summer. In fact, Six Mile Bay only stratified twice since 2010 (2011 and 2014), though it was nearly stratified during all other years except 2012. Lake bottom-DO levels in the summer were below the State's standard three years since 2010 (2010, 2012, and 2014), with all other years decreasing towards the bottom but never below 5 mg/L (Figure 1.10).

Analyte concentrations in the summer have been variable since 1995. Cations have increased with water levels, driven most by increases in calcium (R = 0.910) and magnesium (R = 0.620). There were, however, decreases among cations as well, with copper (R = -0.431) and zinc (R = -0.424) having decreased significantly in Six Mile Bay. Sodium had no relationship with increasing water levels (R = 0.139; Figure 1.11). Anions had little to no significant relationships, with chloride as the only major anion relating to water levels (R = -0.557; Figure 1.11). Sulfate, which had a significant decrease in winter and increase in spring, had no relationship with lake elevation in the summer (R = -0.108). Additionally, neither specific conductance (R = -0.267; Figure 1.11) nor TDS (R = 0.135) related to water levels.

Similar to other seasons, there was no relationship between nutrient concentrations and water levels in Six Mile Bay. TN (R = -0.115; Figure 1.12), N + N (R = 0.285), and TKN (R = -0.188) had no relationship with water levels. Similarly, neither TP (R = -0.053; Figure 1.12) nor DP (R = 0.229) related to changing water levels.

In recent years, TSI scores have been declining during the summer, similar to spring sampling. Secchi disk TSI scores have declined from 58.63 (1.1 m) in 1996 to 46.23 (2.6 m) in 2014, though there was an outlying peak of 73.20 (0.4 m) in 2003 (Figure 1.13). Chlorophyll-α TSI scores increased from 51.00 in 1996 to 66.02 in 2003, but has since declined to 44.84 in 2014 (Figure 1.13). Since 1995, for both Secchi disk transparency and chlorophyll, Six Mile Bay has fluctuated between a mesotrophic and eutrophic basin.

Similar to spring sampling, Six Mile Bay shows no evidence of being P-limited (Figure 1.14), with TSI scores for TP far exceeding those for chlorophyll-α. In recent years, Six Mile Bay has changed from being more dominated by smaller particles to being dominated by larger particles, which is evidence of more efficient zooplankton grazing (Figure 1.14).

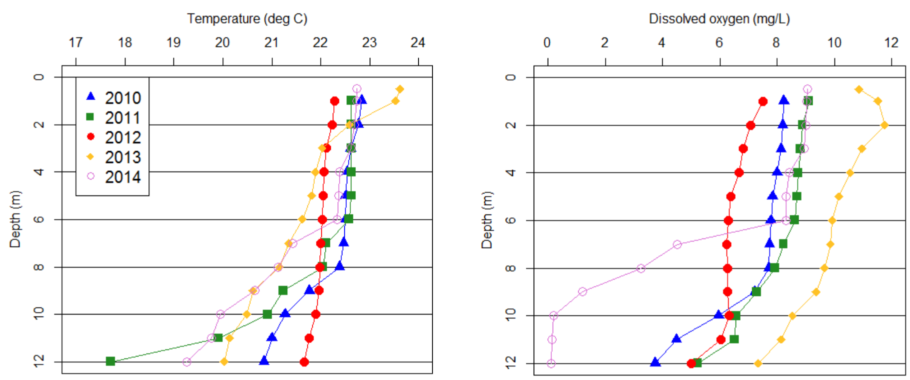

Figure 1.10: Summer temperature (left) and DO (right) profiles for Six Mile Bay from 2010 through 2014.

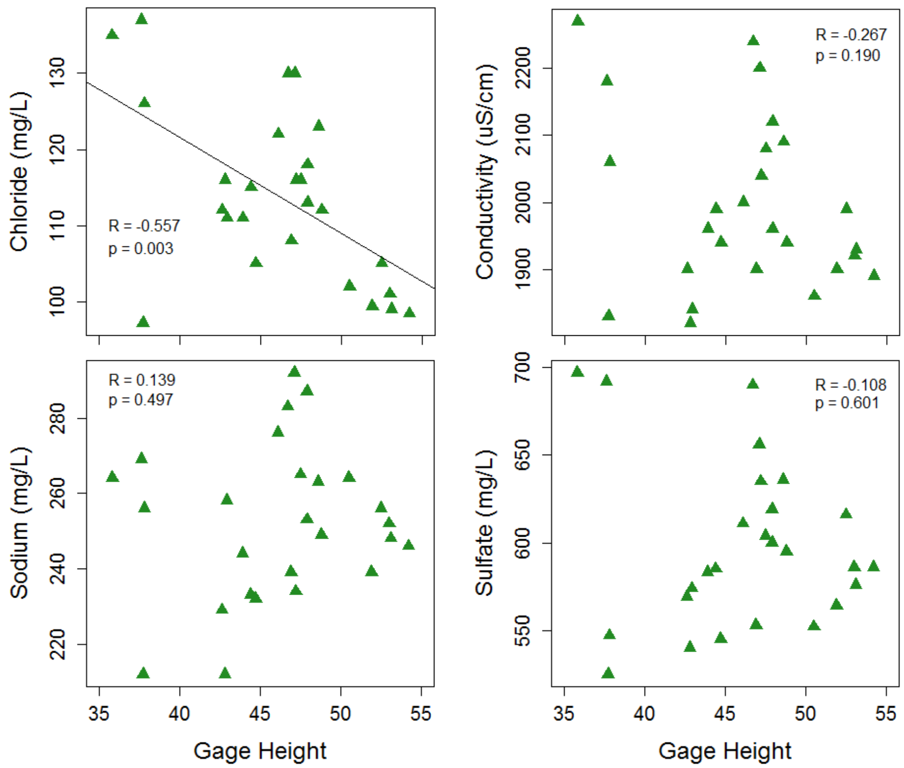

Figure 1.11: Summer concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Six Mile Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

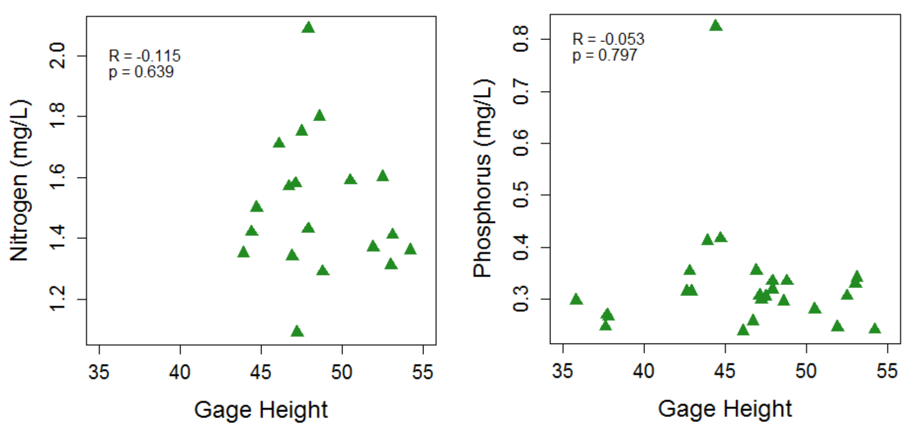

Figure 1.12: Summer concentrations of TN (left) and TP (right) plotted against lake elevation at Six Mile Bay.

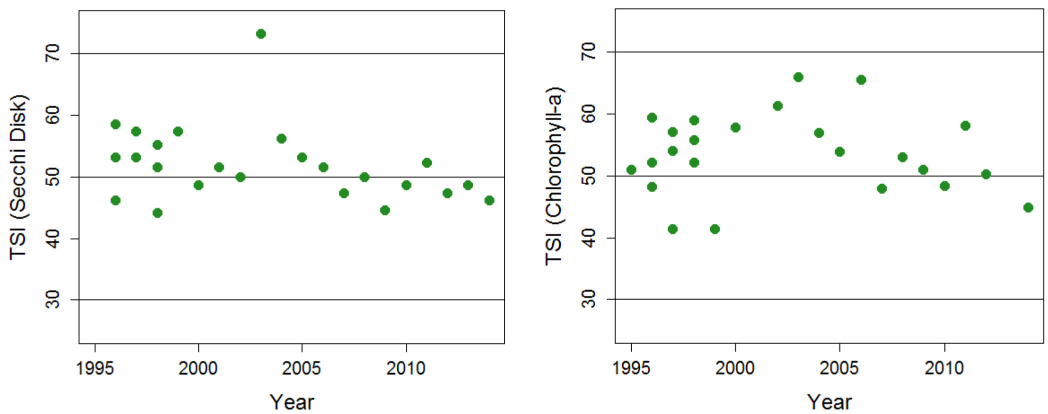

Figure 1.13: Summer TSI scores for Six Mile Bay from 1996 through 2014.

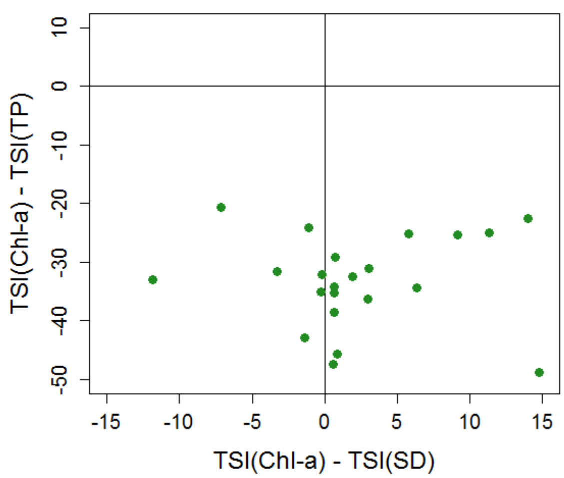

Figure 1.14: Nutrient-limitation graph for summer sampling at Six Mile Bay.

Annual fall sampling

As is common throughout Devils Lake, Six Mile Bay does not thermally stratify in the fall (Figure 1.15). Additionally, this station stays well-oxygenated from top to bottom, not recording a DO reading below the State's standard since 2010 (Figure 1.15).

Over the study period, fall ionic concentrations had variable relationships with increasing lake elevation. Cations had variable relationships with water levels, evidenced by relationships with calcium (R = 0.719), manganese (R = 0.539), potassium (R = -0.551), sodium (R = -0.481; Figure 1.16), and zinc (R = -0.565). Anions decreased over the study period, driven by decreases in chloride (R = -0.857) and sulfate (R = -0.469) (Figure 1.16). Additionally, specific conductance (R = -0.663; Figure 1.16) and TDS (R = -0.476) decreased over the study period.

Nutrient concentrations remained relatively unaffected over the study period. TN (R = -0.193; Figure 1.17) and N+N (R = -0.175) were not related to an increase in water levels, while TKN decreased (R = -0.446). Similarly, TP (R = 0.030; Figure 1.17) and DP (R = 0.059) were not related to the increase in lake elevation.

TSI scores in the fall classify Six Mile Bay as mostly eutrophic, with occasional spikes into mesotrophic and hypereutrophic. TSI scores based on Secchi disk transparency fluctuated since the project's inception in 1995 with a decreasing trend, going from 63.22 (0.8 m) in 1996 and 1997 to 51.53 (1.8 m) in 2007, 2008, and 2013 (Figure 1.18). TSI scores based on chlorophyll-α concentration were much more variable, ranging from 71.24 in 2008 to 34.58 in 2007 (Figure 1.18).

Analysis of TSI scores shows that Six Mile Bay, like the rest of Devils Lake, is not P-limited during fall sampling, with TSI scores based on TP concentration far exceeding those for chlorophyll-α concentration (Figure 1.19). TSI scores based on chlorophyll-α and transparency depths are highly variable, suggesting that Six Mile Bay ranges between dominance by small particles of non-algal turbidity (likely suspended sediment) and large particles of algae in the photic zone (Figure 1.19).

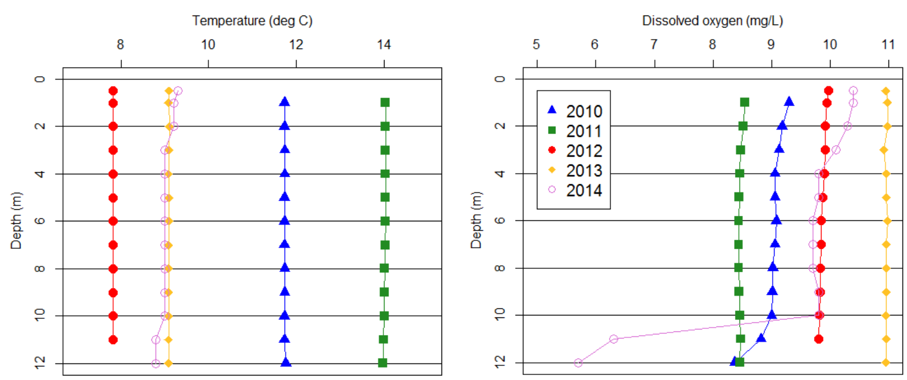

Figure 1.15: Fall temperature (left) and DO (right) profiles for Six Mile Bay from 2010 through 2014.

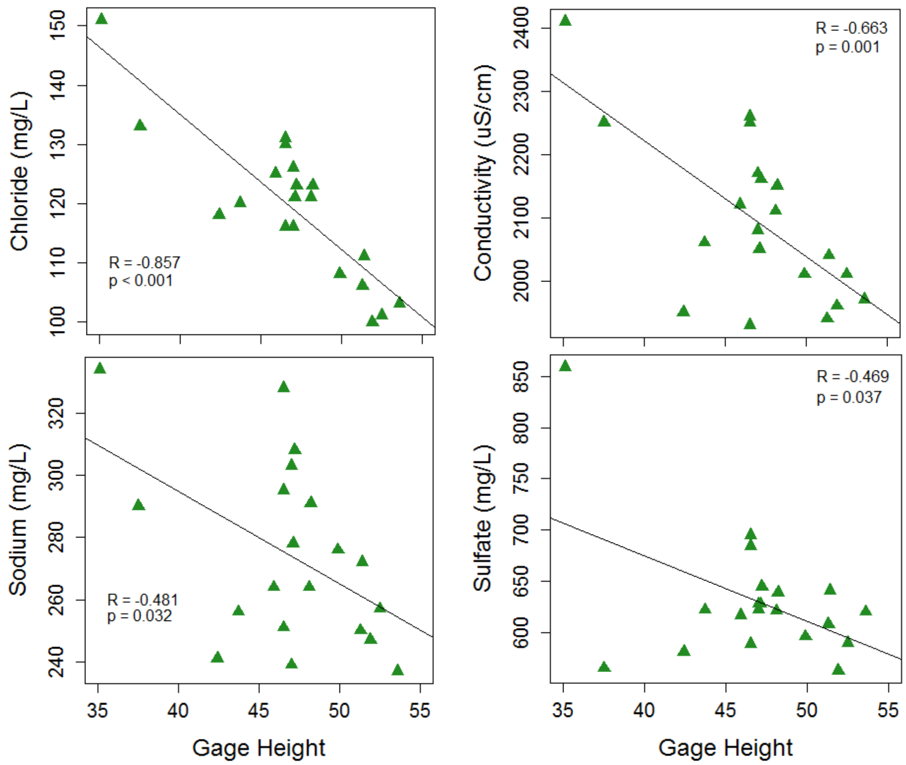

Figure 1.16: Fall concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Six Mile Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

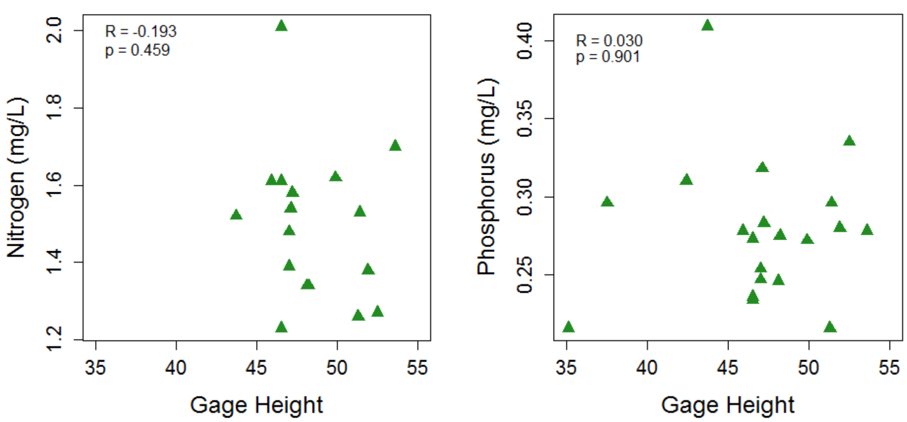

Figure 1.17: Fall concentrations of TN (left) and TP (right) plotted against lake elevation at Six Mile Bay.

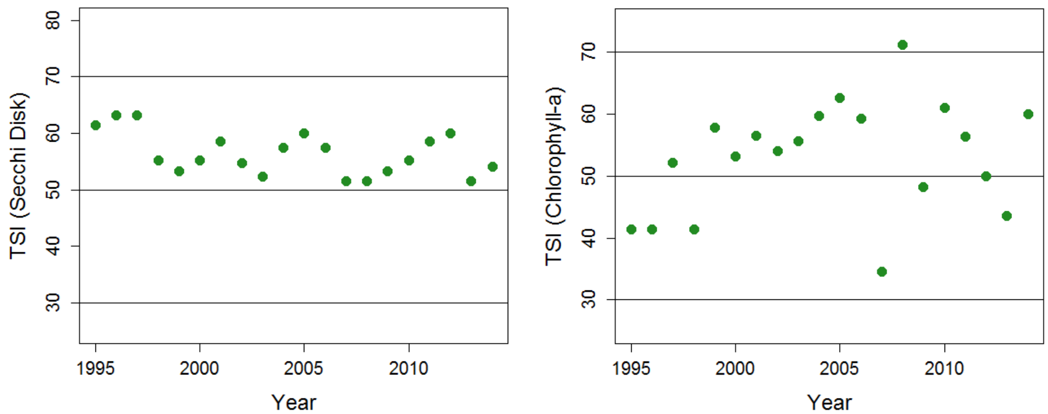

Figure 1.18: Fall TSI scores for Six Mile Bay from 1995 through 2014.

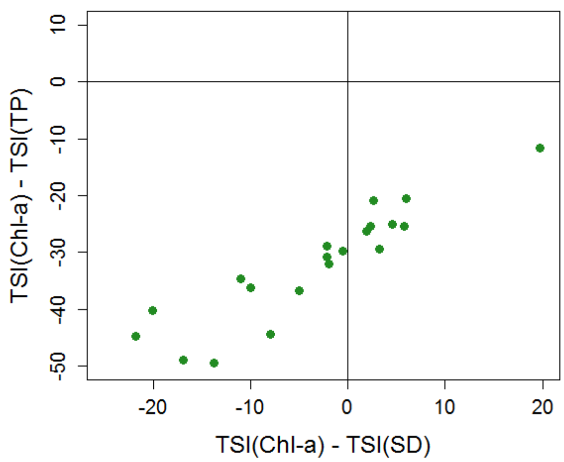

Figure 1.19: Nutrient-limitation graph for fall sampling at Six Mile Bay.

Phytoplankton assemblage sampling

Six Mile Bay is generally dominated by a combination of green and blue-green algae (i.e., cyanobacteria), the latter being most abundant during the summer and fall months. While dominance of certain phyla was common season-by-season in a year, the species composition of these dominating phyla tended to vary from year-to-year (Figure 1.20). *Aphanizomenon flos-aquae*, however, was present in high densities every year (between 10,000-45,000 cells ml^{-1}), with the seasonal exception of most spring samples (only present in May of 1996). Community composition between 1997 and 1999 samples were most similar to each other, regardless of season, while 1996, 1998, and 2000 samples tended to be most closely related (Figure 1.20). At the species-level, however, there were varying proportions of these individual species across seasons (Figure 1.20).

Table 1.1: Presence-absence table showing common phytoplankton species and when they were detected in Six Mile Bay. An "X" signifies that the species was present during that particular season.

	1996			1997			1998			1999			2000		
	Sp	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**
Chlorophyta															
Ankyra judai		X					X	X							X
Chlorella minutissima							X	X	X				X	X	X
Chlorella vulgaris	X	X	X				X								
Choricystis minor	X	X	X				X	X	X				X		X
Monoraphidium contortum	X							X	X			X			
Monoraphidium minutum	X			X	X	X				X	X	X			
Planktosphaeria gelatinosa							X	X	X						
Raphidocelis microscopic									X				X		X
Raphidonema sp.										X	X	X			
Schroederia setigera		X						X	X						
Stylosphaeridium chlorangielloides								X	X						
Chrysophyta															
Chrysococcus cordiformis										X	X	X			
Chrysococcus minutus										X	X	X			
Chrysococcus porifer										X		X			
Chrysococcus punctiformis				X	X	X				X	X	X			
Chrysococcus rufescens				X						X	X	X			
Chrysococcus rufescens var. tri.										X	X	X			
Kephyrion littorale										X	X	X			
Kephyrion sitta										X	X	X			
Ochromonas minuta				X	X	X									
Cryptophyta															
Campylomonas reflexa							X	X	X						
Chroomonas acuta	X	X	X	X	X	X		X	X			X			
Cryptomonas erosa				X	X	X									
Cryptomonas marssonii	X					X						X			
Cryptomonas ovata				X	X	X							X		
Cryptomonas rostrata	X	X										X			
Plagioselmis sp.													X	X	X
Rhodomonas minuta		X		X	X	X				X					
Rhodomonas minuta var. nan.				X	X	X									
Cyanobacteria															
Anabaena curva		X						X	X						
Anabaena flosaquae								X							
Aphanizomenon flosaquae	X	X	X		X	X		X	X		X	X		X	X
Aphanocapsa delicatissima		X											X		
Aphanothece nidulans var. nid.		X	X												
Aphanothece smithii							X	X	X				X	X	X
Gloeocapsa aeruginosa				X	X	X				X	X	X			
Microcystis aeruginosa		X	X		X			X	X			X		X	X
Myxobaktron sp.	X		X						X						
Oscillatoria angustissima					X	X									
Oscillatoria utermoehl								X	X					X	X
Pseudanabaena mucicola		X						X	X					X	
Snowella lacustris	X	X	X												
Woronichinia compacta								X	X						
Woronichinia naegeliana							X	X	X						X
Dinoflagellata															
Ceratium hirundinella		X			X			X						X	

Table 1.1: (cont.)

	1996			1997			1998			1999			2000		
	Sp	Su	Fa	Sp	Su	Fa	Sp	Su	Fa	Sp	Su	Fa	Sp	Su	Fa
Heterokontophyta															
Asterionella formosa	X	X						X					X		
Aulacoseira granulata					X									X	
Aulacoseira granulata var. gra.		X						X	X						
Cyclostephanos sp.				X		X					X	X			
Cyclotella choctawhatcheeana								X	X						X
Cyclotella glomerata				X	X	X				X	X	X			
Cyclotella meneghiniana		X		X		X			X	X	X	X			
Navicula minuscule	X	X	X												
Nitzschia fonticola		X						X							
Nitzschia palea	X							X							
Stephanodiscus hantzschii				X	X					X					
Stephanodiscus minutus	X	X							X						
Stephanodiscus niagarae	X	X	X		X		X	X				X	X		X
Stephanodiscus rotula				X		X			X						

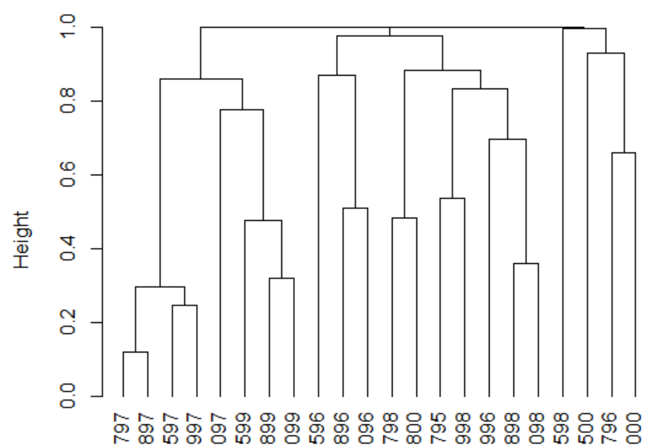

Figure 1.20: Dendrogram clustering sites based on presence and proportion of algal species for each sample. Distances were determined using Bray-Curtis dissimilarity scores. Numbers along the x-axis represent the date of the sample (i.e., 797 is July 1997; 896 is August 1996).

3.2 Main Bay

The sampling station at Main Bay (Station ID 380233) is located east of Graham's Island. The station at Main Bay has been active since 1995 (Figure 2.1). This station has long fetches with westerly and easterly winds, making thermal stratification more difficult except at greater depths. Additionally, Main Bay is one of the deeper areas of Devils Lake, averaging between 15 and 17 meters (49 to 56 feet).

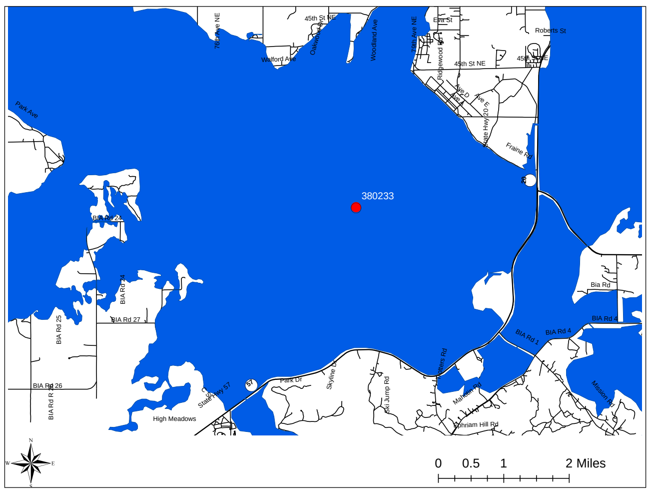

Figure 2.1: Map showing location of sampling site in Main Bay.

Annual winter sampling

Despite having some of the warmest near-bottom temperatures in the basin, Main Bay rarely thermally stratifies in the winter, and has not been observed since 2010 (Figure 2.2). Additionally, the entire water column of Main Bay remains well-oxygenated despite its depth, with DO not being below the State's standard since 2010 (Figure 2.2).

Likely due to dilution-effects from increasing water levels, most analytes in Main Bay declined during winter sampling. Cations decreased with increasing water levels, driven by decreases in copper (R = -0.534), potassium (R = -0.663), and sodium (R = -0.684; Figure 2.3). Barium and calcium, however, did increase with lake elevation (R= 0.590 and 0.751, respectively). Anions decreased with increasing water levels as well, driven by sulfate concentration (R = -0.729; Figure 2.3). Specific conductance (R = -0.913; Figure 2.3) and TDS (R = -0.719) also declined significantly with increasing water levels.

TN had no relationship with increasing water levels (R = 0.065; Figure 2.4), while TKN (R = -0.650) and N+N (R = -0.588) declined significantly with lake elevation. Additionally, TP (R = 0.077; Figure 2.4) and DP (R = 0.026) were not related to changes in water level.

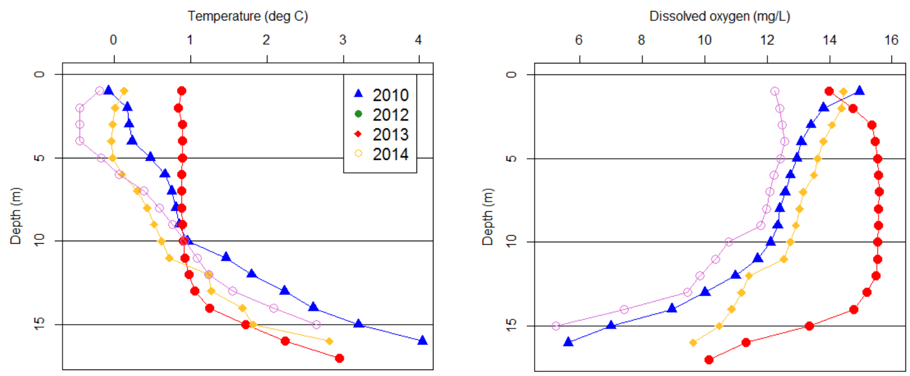

Figure 2.2: Winter temperature (left) and DO (right) profiles for Main Bay from 2010 through 2014.

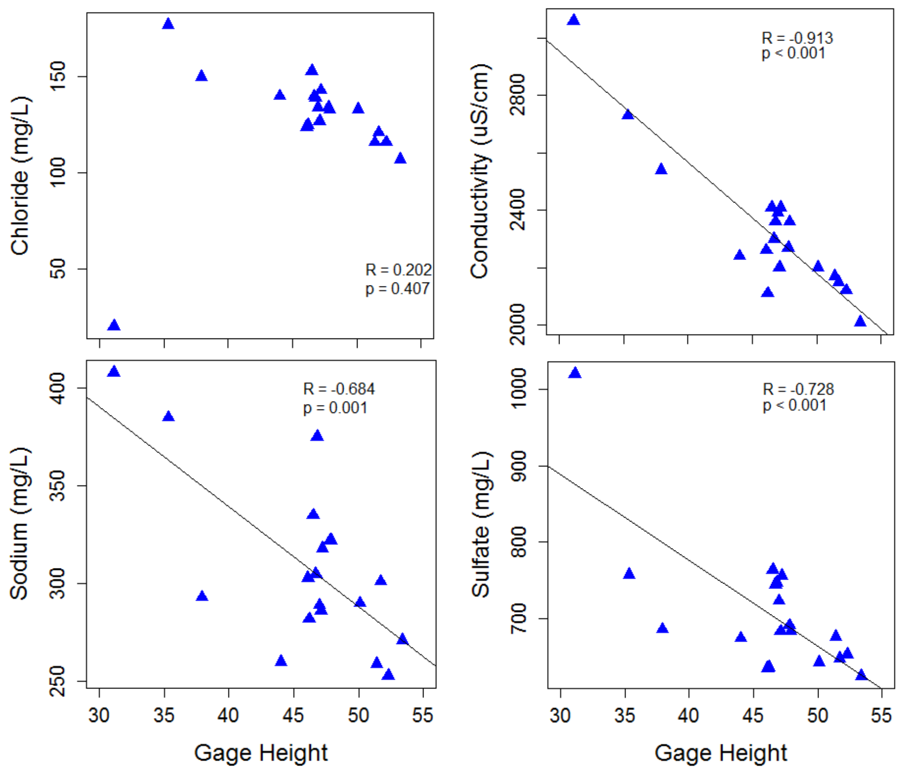

Figure 2.3: Winter concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Main Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

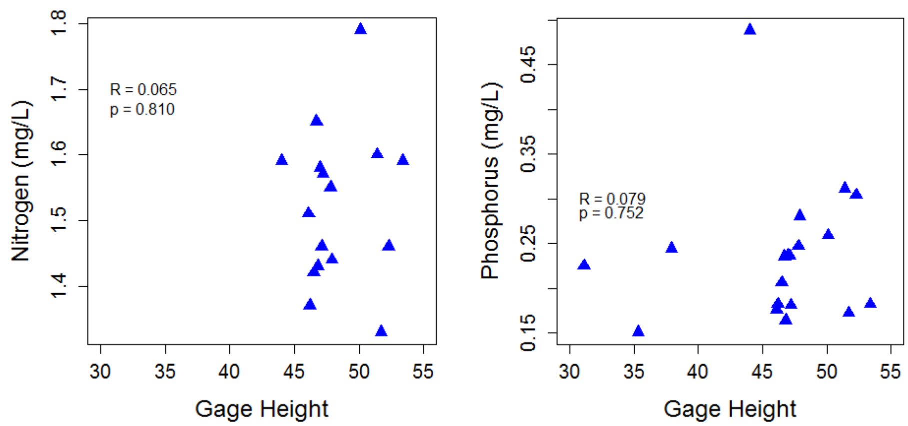

Figure 2.4: Winter concentrations of TN (left) and TP (right) plotted against lake elevation at Main Bay.

Annual spring sampling

While thermal stratification begins to appear in the spring, it is rarely observed due to Main Bay having long fetches with limited protection. The only exception is during 2014 sampling, where the mixing zone was located between three and four meters (Figure 2.5). This constant mixing from strong, spring wind keeps Main Bay well-oxygenated (Figure 2.5).

Similar to winter sampling, analytes commonly decreased with increasing water levels. Cations decreased with increasing water levels, driven by decreases in copper (R = -0.510), potassium (R = -0.818), and sodium (R = -0.870; Figure 2.6). Barium and calcium continued to increase in the spring (R = 0.696 and 0.712, respectively). Anions declined with increasing lake elevation, with chloride (R = -0.917) and sulfate (R = -0.716) decreasing the most (Figure 2.6). Additionally, specific conductance (R = -0.866; Figure 2.6) and TDS (R = -0.781) declined significantly in Main Bay.

Trends among nutrients in the spring in Main Bay were highly variable. For example, TN (R = -0.529; Figure 2.7) and N+N (R = -0.509) decreased significantly with increasing water levels, while TKN had no relationship with lake elevation (R = 0.081). TP (R = -0.069; Figure 2.7) and DP (R = -0.023) also had no relationship with water levels in Main Bay.

TSI scores in Main Bay in the spring have been highly variable since 1995. TSI scores based on Secchi disk have varied from 58.63 (1.1 m) in 1997 to 29.33 (8.4 m) in 2010 with no distinct trend (Figure 2.8). Similar to Secchi disk, chlorophyll-α TSI scores varied greatly since 1995, ranging from 34.58 (2007) to 59.32 (2008).

Similar to the rest of Devils Lake, Main Bay is not P-limited in the spring (Figure 2.9). Since 2008, the basin has been transitioning to being dominated by larger particles of algae, instead of having non-algal turbidity impacting transparency (Figure 2.9).

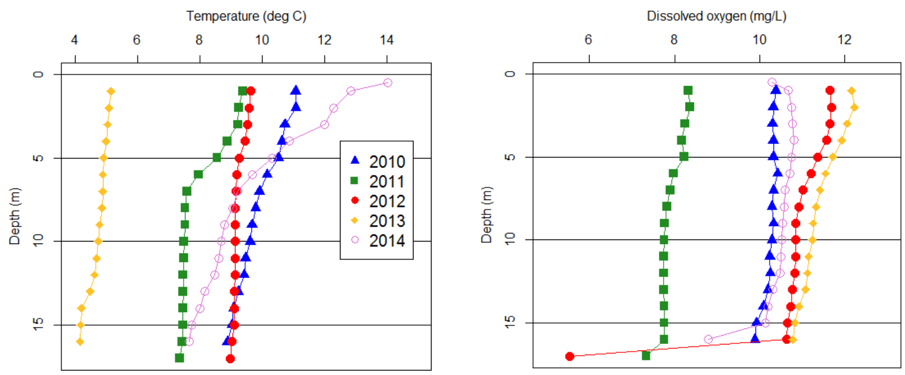

Figure 2.5: Spring temperature (left) and DO (right) profiles for Main Bay from 2010 through 2014.

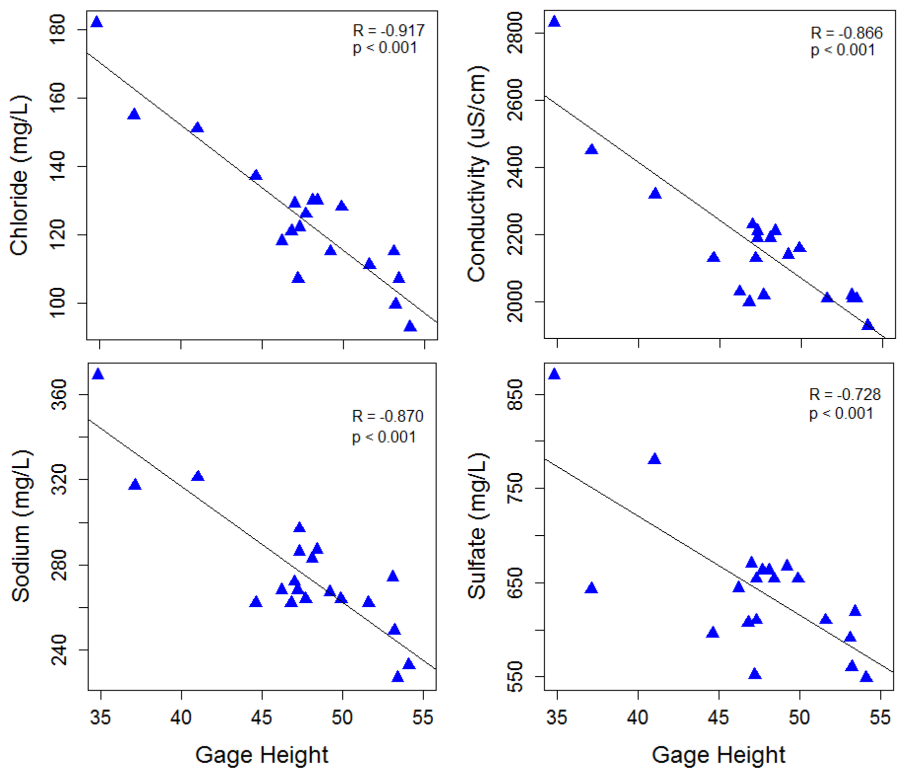

Figure 2.6: Spring concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Main Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

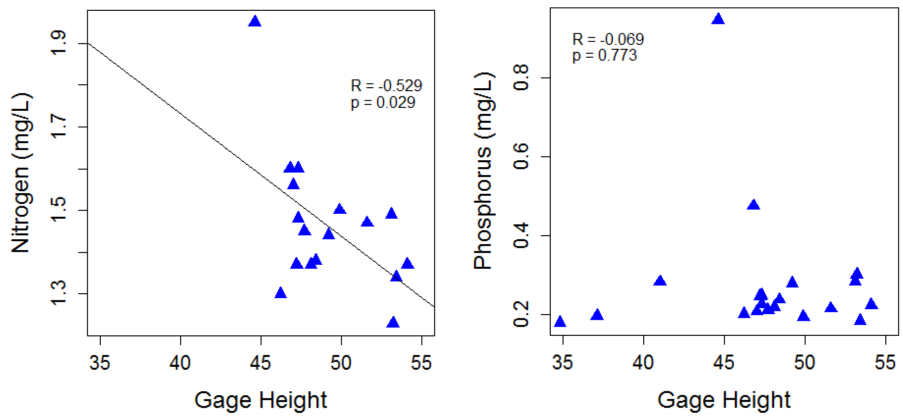

Figure 2.7: Spring concentrations of TN (left) and TP (right) plotted against lake elevation at Main Bay.

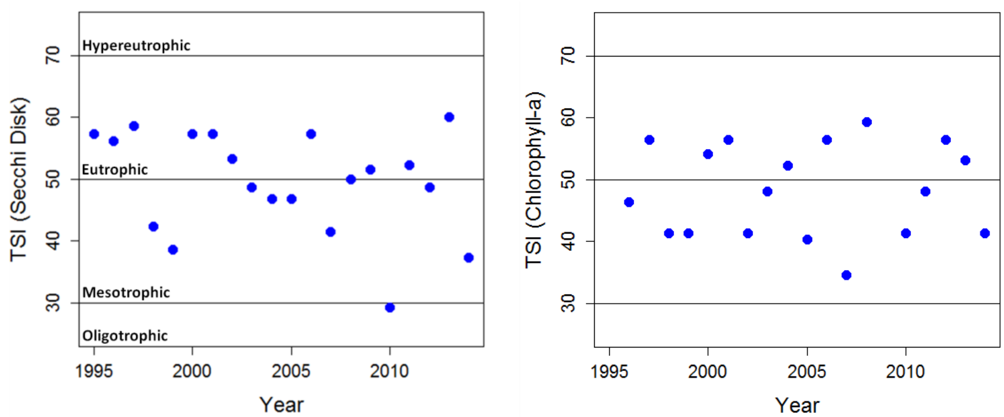

Figure 2.8: Spring TSI scores for Main Bay from 1995 through 2014.

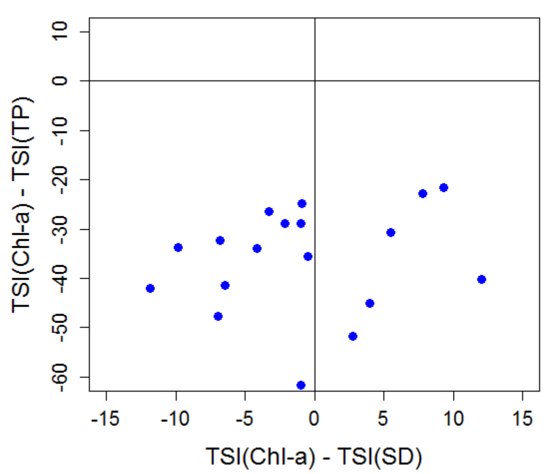

Figure 2.9: Nutrient-limitation graph for spring sampling at Main Bay.

Annual summer sampling

Similar to other seasons, Main Bay temperature and DO remain relatively similar throughout the water column in the summer. There is some thermal stratification, however, typically only one or two meters from the bottom (Figure 2.10). Similarly, DO remains relatively high throughout the water column, and rarely goes below the State's standard (Figure 2.10).

Many analytes continued to decline with increasing water level in the summer. Cations declined which was driven by decreases in copper (R = -0.414), potassium (R = -0.718), and sodium (R = -0.693; Figure 2.11). Conversely, barium (R = 0.438), calcium (R = 0.753), and manganese (R = 0.545) increased with increasing water levels. Anions decreased with increasing water levels, with chloride (R = -0.823) and sulfate (R = -0.639) showing the strongest relationship (Figure 2.11). Specific conductance (R = -0.766; Figure 2.11) and TDS (R = -0.636) also declined with increasing water levels.

Nutrients were variable from 1995 through 2014, and had no relationship with water levels. TN (R = -0.076; Figure 2.12) and TKN (R = -0.281) were not related to increasing water levels, as were TP (R = 0.055; Figure 2.12) and DP (R = 0.272).

TSI scores in the summer showed no trend over time. TSI scores for Secchi disk remained similar since 1995, ranging from 45.16 (2.8 m) in 1996 and 1998 to 60.00 (1.0 m) in 2003 (Figure 2.13). Similarly, chlorophyll-α TSI scores were highly variable. Scores ranged from a peak of 67.50 in 1996 to a low of 34.58 in 2010, with many fluctuations over the twenty years (Figure 2.13). These numbers suggest that Main Bay fluctuates between mesotrophic and eutrophic in the summer (Figure 2.13).

Analyzing TSI scores shows that Main Bay, like the rest of the lake, is not P-limited in the summer months (Figure 2.14). Strongly negative scores for TSI(Chl-α)-TSI(TP) suggests that TP is not a limiting nutrient in the basin (Figure 2.14). Varying scores for TSI(Chl-α)-TSI(SD) suggests that Main Bay fluctuates between being dominated by small and large particles of non-algal and algal turbidity, respectively (Figure 2.14).

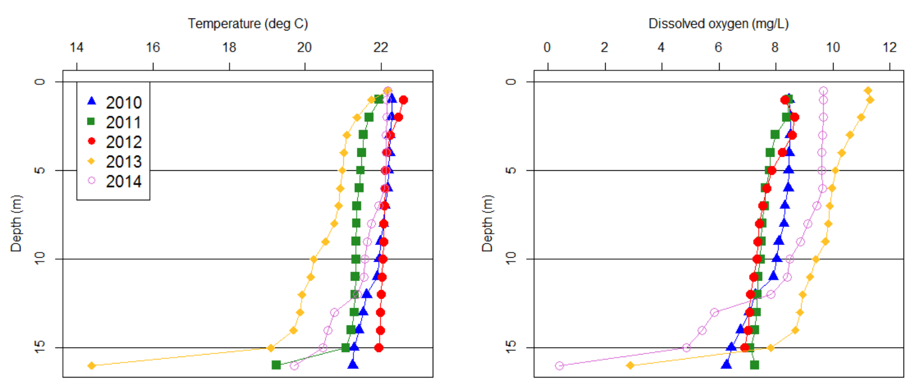

Figure 2.10: Summer temperature (left) and DO (right) profiles for Main Bay from 2010 through 2014.

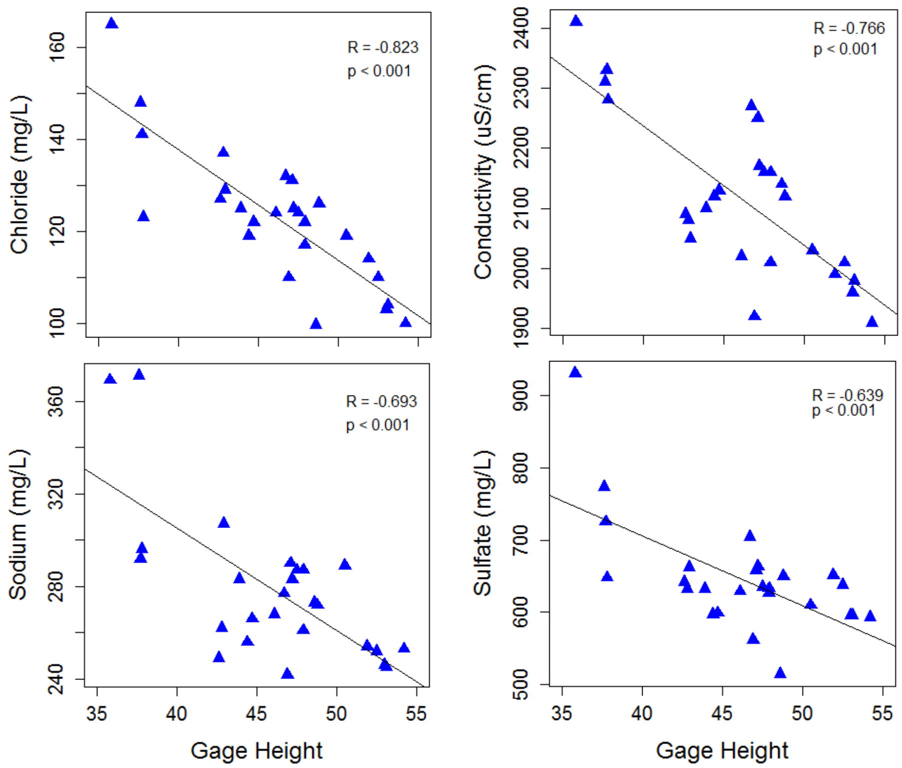

Figure 2.11: Summer concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Main Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

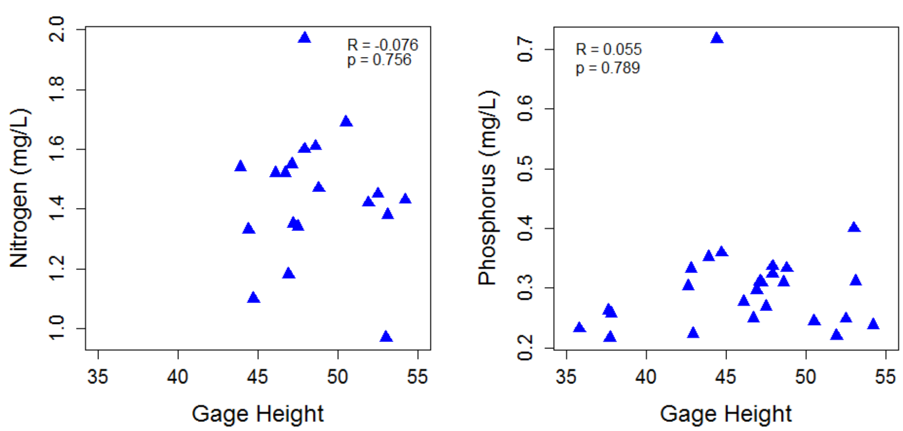

Figure 2.12: Summer concentrations of TN (left) and TP (right) plotted against lake elevation at Main Bay.

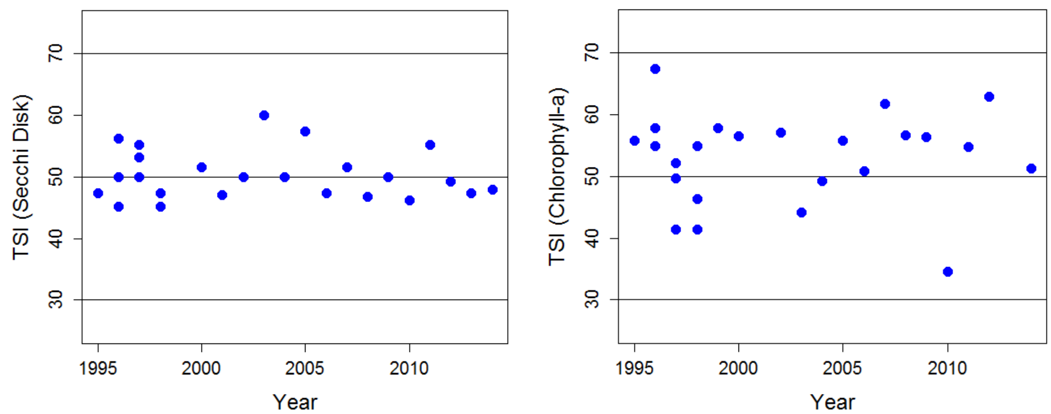

Figure 2.13: Summer TSI scores for Main Bay from 1995 through 2014.

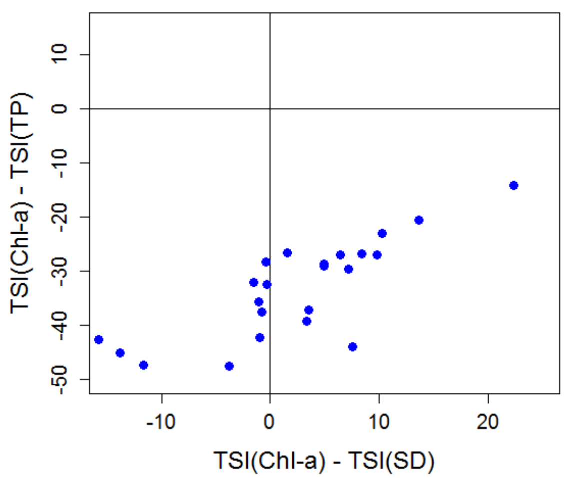

Figure 2.14: Nutrient-limitation graph for summer sampling at Main Bay.

Annual fall sampling

Due to long fetches in Main Bay, this station did not thermally stratify during fall sampling (Figure 2.15). Due to this constant top-to-bottom mixing, DO at Main Bay was not below 8 mg L^{-1} since 2010 (Figure 2.15).

Analytes measured in the fall in Main Bay followed a similar pattern to other seasons. Cations decreased with increasing lake elevation, driven by decreases in potassium (R = -0.654) and sodium (R = -0.720; Figure 2.16). Conversely, calcium (R = 0.811) and manganese (R = 0.452) increased significantly with rising water levels. Anions decreased with increasing lake elevation, driven by decreases in chloride (R = -0.917) and sulfate (R = -0.654) (Figure 2.16). Specific conductance continued to decline over the study period (R = -0.543) (Figure 2.16).

Nutrient concentrations were relatively unaffected by the increase in lake elevation over the study period. TN (R = -0.212; Figure 2.17) and N+N (R = 0.201) did not relate to water level, while TKN decreased significantly over time (R = -0.506). Additionally, neither TP (R = 0.126; Figure 2.17) nor DP (R = 0.260) related to increasing water levels in Main Bay in the fall.

TSI scores based on chlorophyll-α and Secchi disk transparency classify Main Bay as eutrophic in the fall with occasional spikes into mesotrophic and hypereutrophic. TSI scores based on Secchi disk transparency varied from 57.37 (1.2 m) in 1996, 2005, and 2009 to 48.64 (2.2 m) in 1995 and 1996) (Figure 2.18). Chlorophyll-α TSI scores were highly variable over the study period ranged from 63.33 (1998) to 41.38 (1995, 2007) (Figure 2.18).

Analysis of TSI scores confirms that Main Bay, like the rest of Devils Lake, is not P-limited in the fall, as evidenced by TSI scores based on TP concentration being much higher than TSI scores based on chlorophyll-α (Figure 2.19). Additionally, the photic zone is becoming increasingly dominated by larger particles of algae (Figure 2.19), likely due to rising water levels causing reduced mixing and more efficient zooplankton grazing.

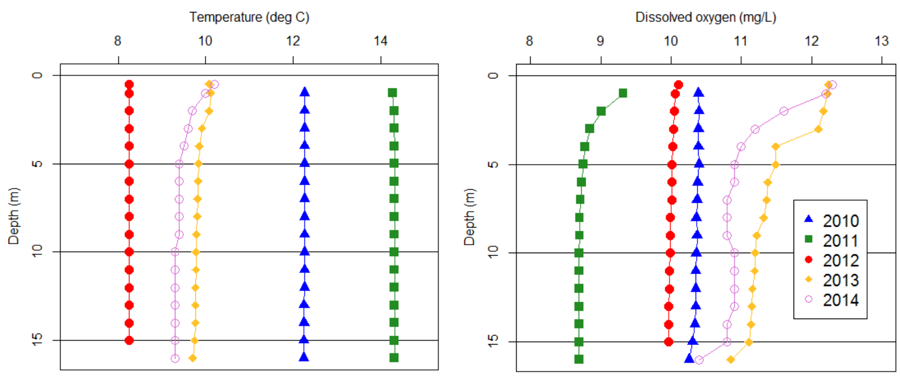

Figure 2.15: Fall temperature (left) and DO (right) profiles for Main Bay from 2010 through 2014.

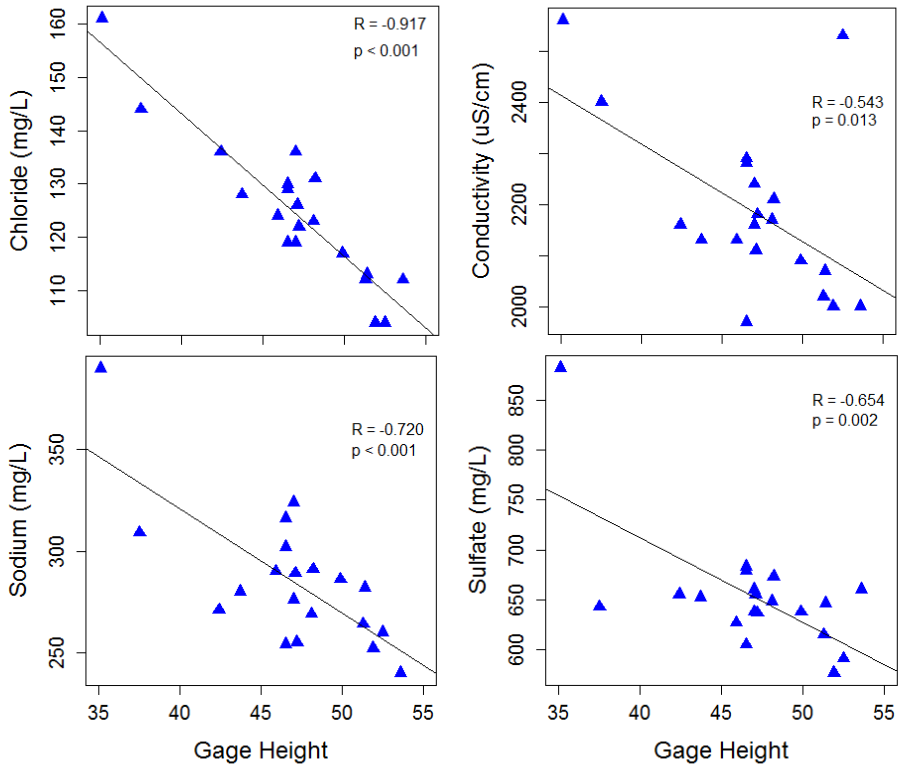

Figure 2.16: Fall concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Main Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

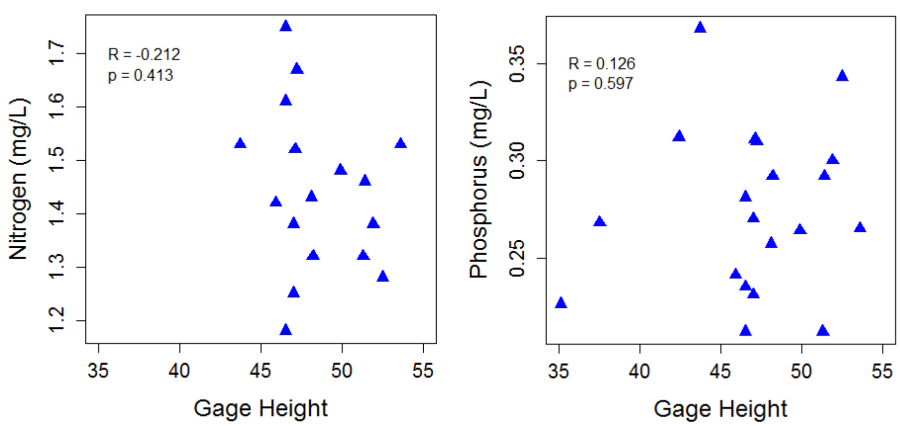

Figure 2.17: Fall concentrations of TN (left) and TP (right) plotted against lake elevation at Main Bay.

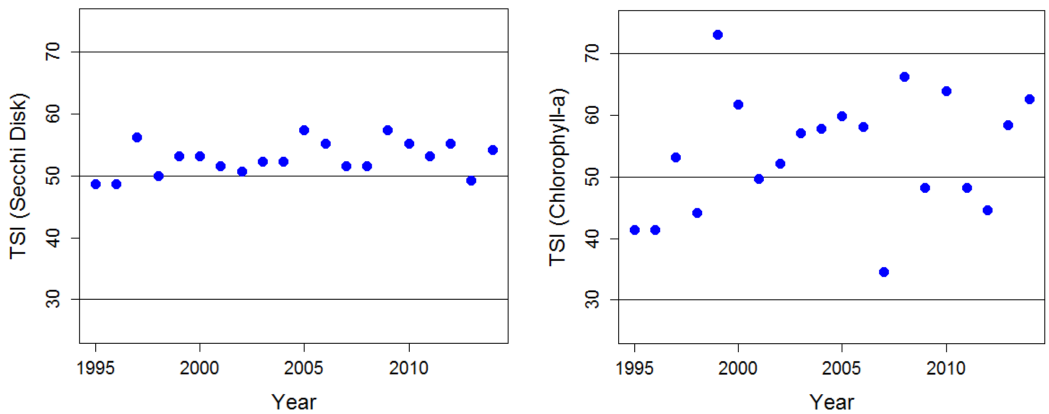

Figure 2.18: Fall TSI scores for Main Bay from 1995 through 2014.

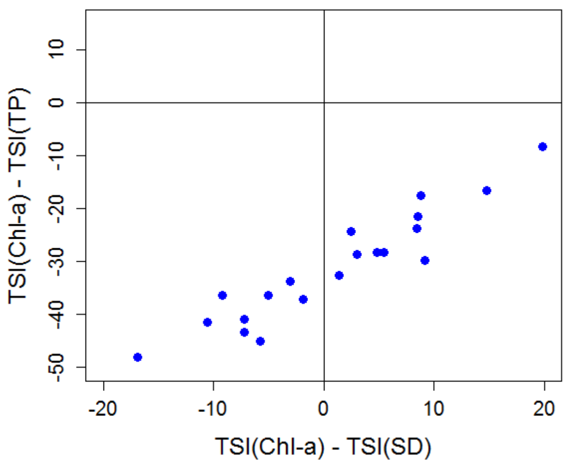

Figure 2.19: Nutrient-limitation graph for fall sampling at Main Bay.

Phytoplankton assemblage sampling

Main Bay is dominated by a combination of green and blue-green algae (i.e., cyanobacteria), the latter generally being most abundant during the summer months. While dominance of certain phyla was common season-by-season, the species composition of these dominating phyla tended to vary from year-to-year (Table 2.1; Figure 2.20). *Aphanizomenon flosaquae*, however, was present in relatively high densities every year (between 2,000-15,000 cells ml^{-1}), with the seasonal exception of most spring samples (only present in May of 1997) and was at low densities all of 1997 (maximum density was 200 cells ml^{-1}). Community composition between 1997 and 1999 samples were most similar to each other, regardless of season, while 1996, 1998, and 2000 samples tended to be most closely related (Figure 2.20).

Table 2.1: Presence-absence table showing common phytoplankton species and when they were detected in Main Bay. An "X" signifies that the species was present during that particular season. Despite multiple samples being taken during summer months from 1996 through 1998, these samples were pared to a single summer sample for this table.

	1996			1997			1998			1999			2000		
	Sp	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**
Chlorophyta															
Ankyra judai		X	X					X							
Chlorella minutissima								X	X				X	X	X
Chlorella vulgaris	X	X						X	X						X
Choricystis minor	X	X	X					X	X				X	X	X
Coelastrum microporum								X	X			X			
Eremosphaera eremosphaeria		X	X					X							
Monoraphidium contortum	X								X						
Monoraphidium minutum					X	X	X				X	X			
Monoraphidium tortile				X		X	X								
Oocystis parva								X	X		X				
Pediastrum boryanum		X						X							X
Planktosphaeria gelatinosa								X	X						
Raphidocelis contorta var. gra.									X				X		X
Raphidonema sp.				X						X	X	X			
Schroederia setigera		X	X					X	X						X
Stylosphaeridium chlorangielloides								X	X						
Chrysophyta															
Chrysococcus cordiformis										X	X	X			
Chrysococcus cordiformis var. ast.										X	X	X			
Chrysococcus minutes										X	X	X			
Chrysococcus punctiformis				X	X	X	X			X	X	X			
Chrysococcus rufescens				X	X					X	X	X			
Chrysococcus rufescens var. tri.										X	X	X			
Kephyrion littorale										X	X	X			
Kephyrion sitta										X	X	X			
Ochromonas minuta				X	X	X	X								
Cryptophyta															
Campylomonas reflexa								X	X						
Chroomonas acuta	X	X	X	X	X	X	X	X	X			X			
Chroomonas nordstedtii					X							X			
Cryptomonas sp.	X	X	X												
Plagioselmis sp.													X	X	X
Rhodomonas minuta	X	X		X	X	X	X			X	X				
Rhodomonas minuta var. nan.				X	X	X	X			X	X	X			
Rhodomonas pusilla					X	X	X			X					

Table 2.1: (cont.)

	1996			1997			1998			1999			2000		
	Sp	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**
Cyanobacteria															
Anabaena curva		X						X							
Anabaena flosaquae		X						X						X	
Anabaena spiroides					X						X	X			
Aphanizomenon flosaquae		X	X	X	X			X	X		X	X		X	X
Aphanocapsa delicatissima		X						X	X					X	
Aphanocapsa incerta								X	X						X
Aphanothece nidulans var. nid.	X	X	X												
Aphanothece smithii								X	X				X	X	X
Gloeocapsa aeruginosa				X	X	X	X			X	X	X			
Lyngbya birgei		X	X					X							
Microcystis aeruginosa		X	X					X	X		X	X		X	X
Oscillatoria utermoehl								X							X
Pseudanabaena mucicola		X						X	X					X	
Woronichinia compacta								X	X						
Woronichinia naegeliana								X	X						
Heterokontophyta															
Aulacoseira granulate var. gra.		X						X							
Cyclostephanos sp.				X	X					X	X	X			
Cyclotella bodanica						X	X				X	X			
Cyclotella choctawhatcheeana									X				X		X
Cyclotella glomerata				X						X	X	X			
Cyclotella meneghiniana				X					X	X					
Nitzschia fonticola								X							
Nitzschia inconspicua					X	X	X			X	X				
Stephanodiscus hantzschii				X	X	X	X				X				
Stephanodiscus minutus	X		X						X						
Stephanodiscus niagarae	X	X	X			X	X	X	X			X	X		X

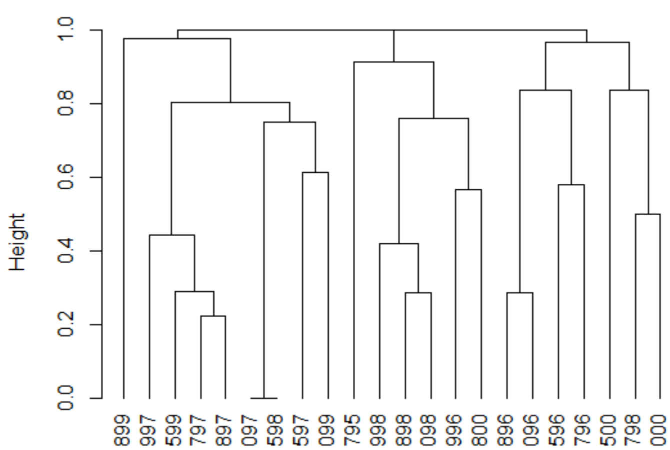

Figure 2.20: Dendrogram clustering sites based on presence and proportion of algal species for each sample in Main Bay. Distances were determined using Bray-Curtis dissimilarity scores. Numbers along the x-axis represent the date of the sample (i.e., 797 is July 1997; 896 is August 1996).

3.3 East Bay

The sampling station at East Bay (Station ID 380234), active since 1995, is south of the City of Devils Lake (Figure 3.1). This station has long fetches with easterly and westerly winds, making thermal stratification more difficult except at greater depths.

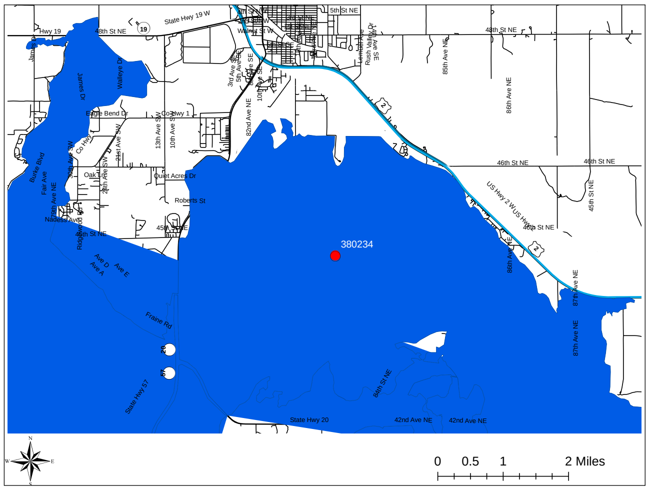

Figure 3.1: Map showing location of sampling site in East Bay.

Annual winter sampling

Although water temperature in East Bay warmed towards the water-sediment interface, there was no thermal stratification from 2010 to 2014 (Figure 3.2). Also, East Bay remained well-oxygenated from top-to-bottom during the winter months despite its depth (Figure 3.2).

Dilution effects appeared to have a strong influence on measured analytes in East Bay since 1995, with many concentrations decreasing significantly. Cations decreased with increasing water levels, driven by decreases in arsenic (R = -0.528), copper (R = -0.670), magnesium (R = -0.588), potassium (R = -0.732), and sodium (R = -0.668; Figure 3.3). Barium and calcium, however, were two cations that increased with rising water levels (R = 0.627 and 0.645, respectively). Anions decreased with increasing lake elevation as well, most evidenced by chloride (R = -0.770) and sulfate (R = -0.818) (Figure 3.3). Also, specific conductance (R = -0.804; Figure 3.3) and TDS (R = -0.784) decreased significantly with increasing water levels.

Nutrient concentrations were also influenced by increasing water levels in East Bay. TN (R = -0.581; Figure 3.4), N+N (R = -0.589), and TKN (R = -0.766) decreased significantly with increasing water level. TP (R = -0.128; Figure 3.4) and DP (R = -0.177), however, were not associated with increasing water levels.

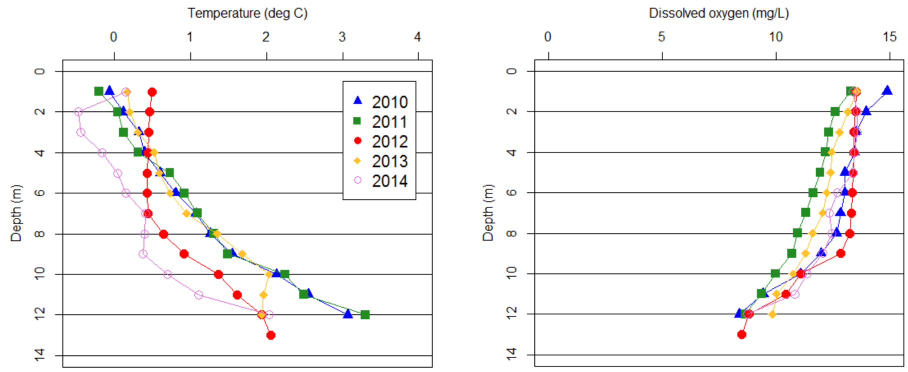

Figure 3.2: Winter temperature (left) and DO (right) profiles for East Bay from 2010 through 2014.

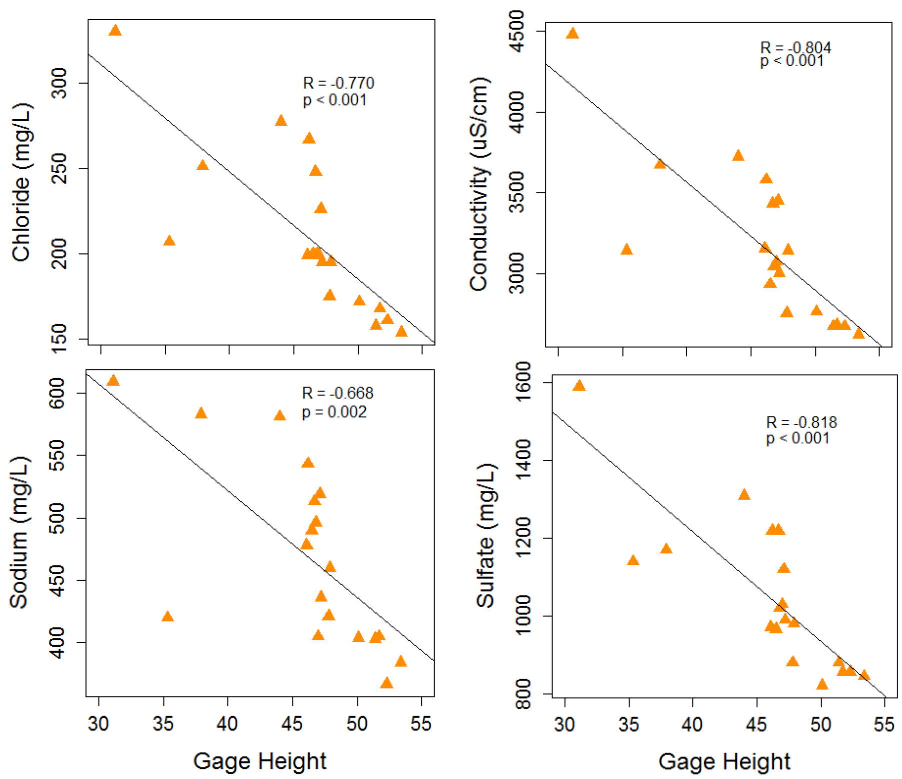

Figure 3.3: Winter concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at East Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

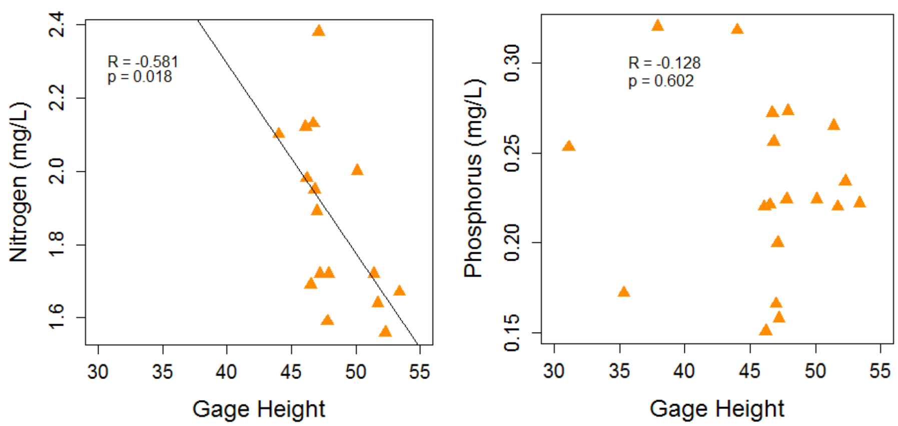

Figure 3.4: Winter concentrations of nitrogen (left) and phosphorus (right) plotted against lake elevation at East Bay.

Annual spring sampling

Like the rest of Devils Lake, thermal stratification is not common in the spring in East Bay, but does occur occasionally. For example, East Bay experienced thermal stratification twice since 2010 (2010 and 2014), with relatively shallow mixing zones (Figure 3.5). DO, however, remains relatively constant from top-to-bottom with only occasional anoxia at the sediment-water interface (Figure 3.5).

Analytes measured in the spring decreased strongly over the study period. Cations decreased with increasing water levels, with arsenic (R = -0.838), magnesium (R = -0.923), potassium (R = -0.950), sodium (R = -0.973; Figure 3.6), and zinc (R = -0.479) being the strongest drivers. Conversely, barium (R = 0.693) and calcium (R = 0.821) increased with increasing water levels. Anions declined with rising water, most influenced by carbonate (R = -0.587), chloride (R = -0.938; Figure 3.6), and sulfate (R = -0.918; Figure 3.4). Additionally, specific conductance (R = -0.973; Figure 3.6) and TDS (R = -0.947) decreased significantly with increasing water levels.

Nutrients followed a similar pattern to other analytes. TN (R = -0.766; Figure 3.7) and TKN (R = -0.773) decreased significantly with increased water level, while N+N was not impacted by lake elevation (R = -0.077). TP (R = 0.071; Figure 3.7) and DP (R = 0.077) were not significantly related to water level.

TSI scores had considerable variation across the sample period, with no trend detected. TSI scores using Secchi disk transparency ranged from 60.00 (1.0 m) in 2001, 2012, and 2013 to 38.65 (4.4 m) in 2010 (Figure 3.8). Chlorophyll-α scores also showed considerable variation, ranging from 55.22 in 2011 to 41.38 in 1998, 1999, 2010, and 2014 (Figure 3.8).

Comparison of TSI scores reveals that East Bay, similar to the rest of Devils Lake, is not P-limited (Figure 3.9). Additionally, the water column is commonly impacted by smaller particles of either algal or non-algal turbidity, hindering Secchi disk transparency (Figure 3.9). In recent years, however, East Bay has been more likely to be dominated by larger algal particles in the photic zone, which is evidence of more efficient zooplankton grazing.

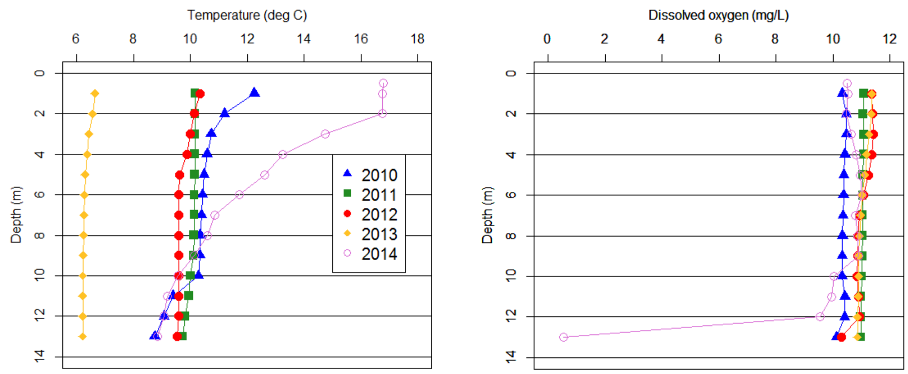

Figure 3.5: Spring temperature (left) and DO (right) profiles for East Bay from 2010 through 2014.

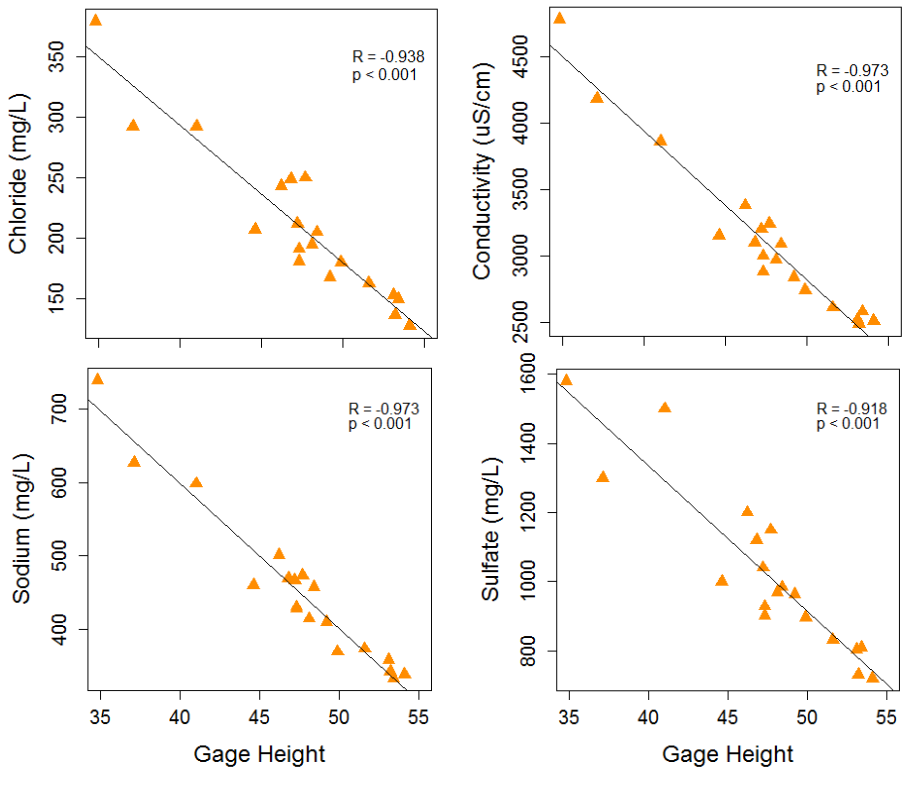

Figure 3.6: Spring concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at East Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

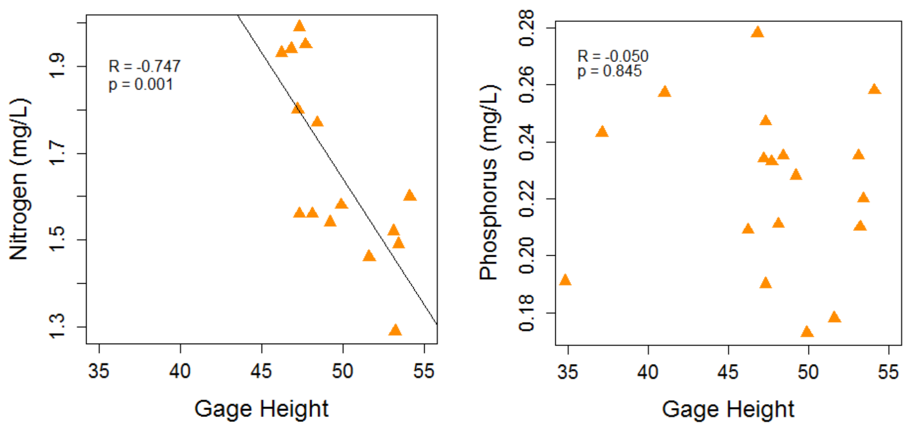

Figure 3.7: Spring concentrations of TN (left) and TP (right) plotted against lake elevation at East Bay.

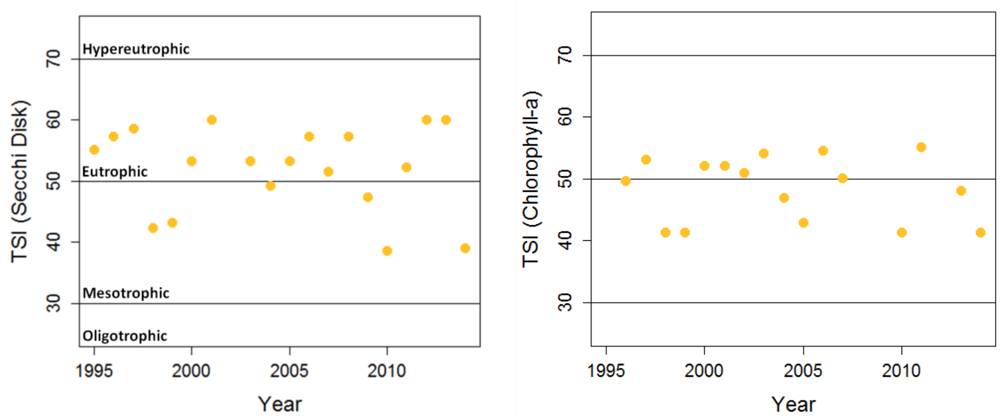

Figure 3.8: Spring TSI scores for East Bay from 1995 through 2014.

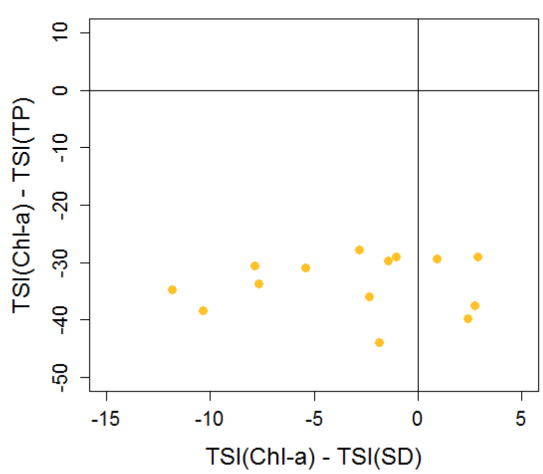

Figure 3.9: Nutrient-limitation graph for spring sampling at East Bay.

Annual summer sampling

East Bay occasionally stratifies during summer sampling, occurring twice since 2010 (2013 and 2014) (Figure 3.10). Decreased DO levels are associated with this stratification, as DO dropped below the State's standard only during years when stratification was observed (Figure 3.10). In 2014, the anoxic zone was relatively shallow compared to other years and basins, reaching 3 mg L^{-1} at 10 meters of depth (Figure 3.10).

Most analytes measured in the summer in East Bay continued to decrease with increasing water levels. Cations decreased with increasing water levels, driven by decreases in arsenic (R = -0.661), magnesium (R = -0.796), potassium (R = -0.873), sodium (R = -0.902; Figure 3.11), and zinc (R = -0.489). Conversely, barium (R = 0.656) and calcium (R = 0.771) increased with increasing water levels. Anions decreased with increasing water levels, with decreases in carbonate (R = -0.502), chloride (R = -0.957), and sulfate (R = -0.928; Figure 3.11). Also, specific conductance (R = -0.958; Figure 3.11), TDS (R = -0.931), and hardness (R = -0.677) decreased strongly with increased lake elevation.

During summer sampling, nutrients were more affected by dilution effects than other seasons. For example, TN (R = -0.457; Figure 3.12) and TKN (R = -0.724) decreased with increasing water levels. TP decreased significantly with rising water levels (R = -0.439; Figure 3.12), while DP was not significantly impacted (R = -0.364).

In the summer, East Bay can typically be classified between mesotrophic and eutrophic. Since 1995, TSI scores based on Secchi depth have steadily decreased, going from 61.52 (0.9 m) in 1997 to 45.16 (2.8 m) in 2013 (Figure 3.13). Chlorophyll-α TSI scores, however, were more variable over time and did not show a trend, ranging from a low of 34.58 in 2010 to a high of 67.05 in 2007 (Figure 3.13).

Comparing TSI scores shows that East Bay is not limited by TP (Figure 3.14), despite a significant reduction since 1995 (Figure 3.12). This analysis also suggests that East Bay is consistently dominated by larger particles of algae, evidenced by higher chlorophyll-α TSI scores and decreasing Secchi disk scores (Figure 3.14).

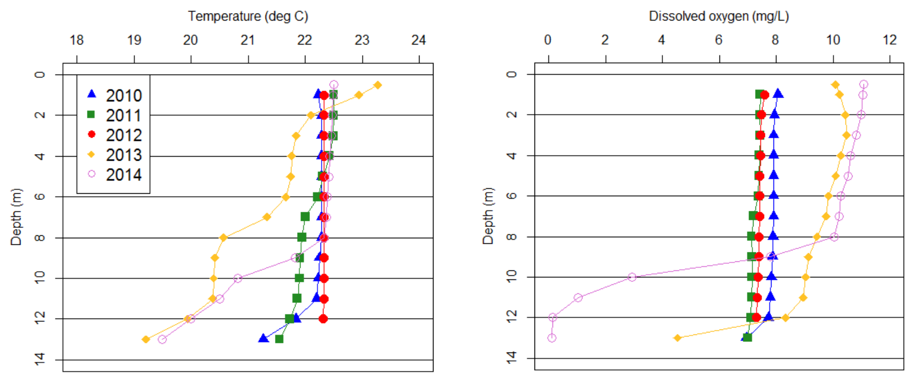

Figure 3.10: Summer temperature (left) and DO (right) profiles for East Bay from 2010 through 2014.

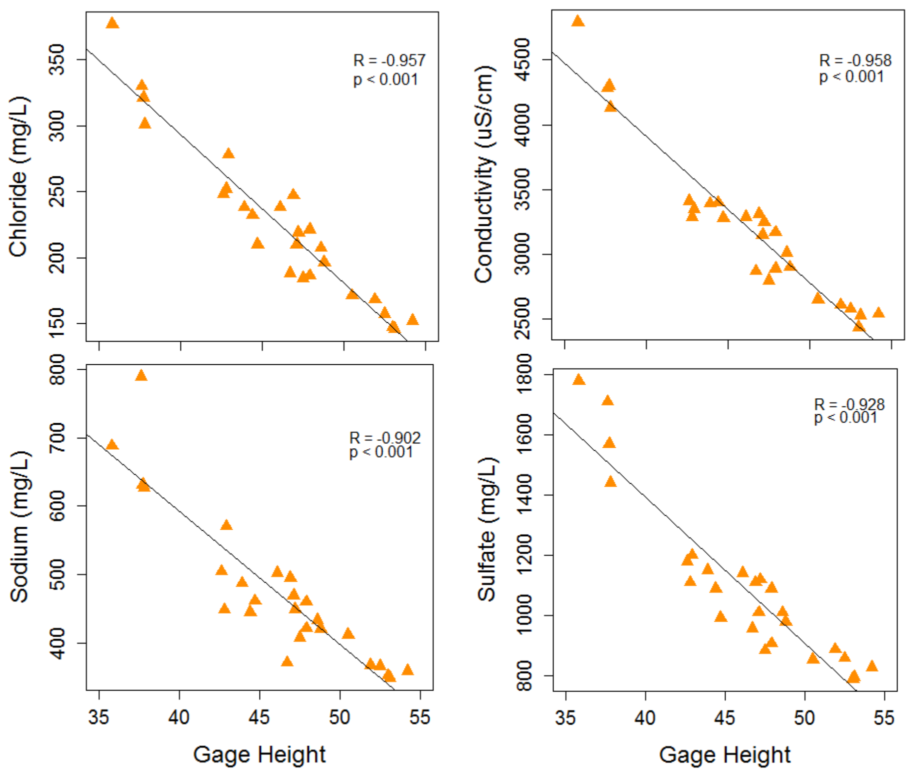

Figure 3.11: Summer concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at East Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

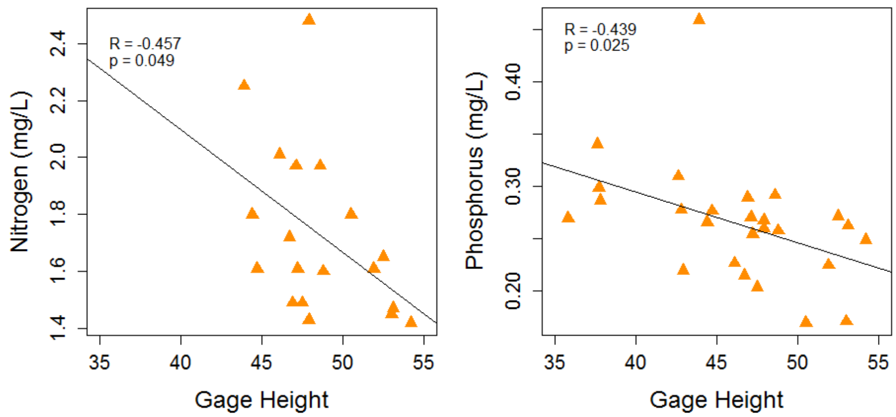

Figure 3.12: Summer concentrations of TN (left) and TP (right) plotted against lake elevation at East Bay.

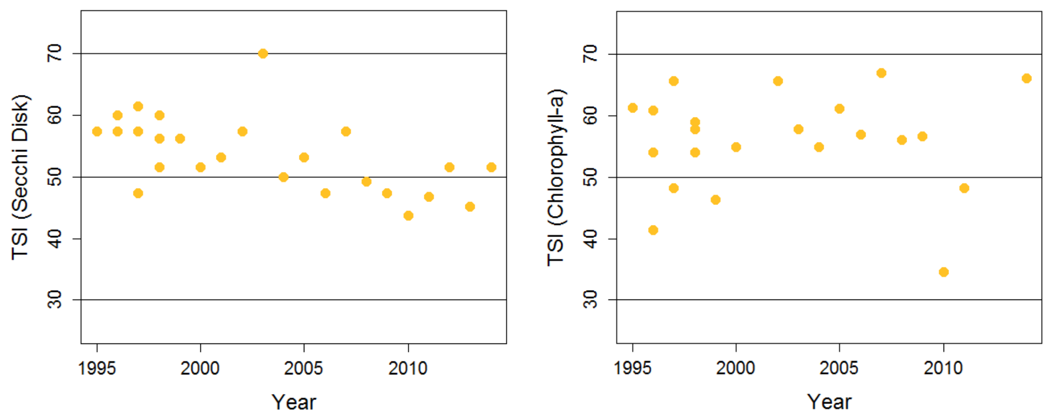

Figure 3.13: Summer TSI scores for East Bay from 1995 through 2014.

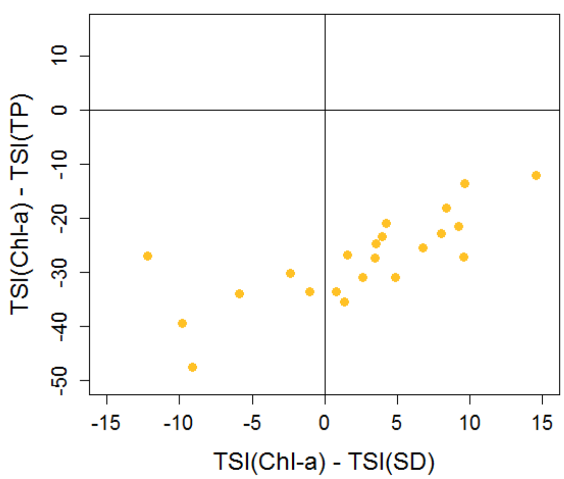

Figure 3.14: Nutrient-limitation graph for summer sampling at East Bay.

Annual fall sampling

Due to long fetches causing almost constant mixing in the fall, this station did not stratify after 2010 (Figure 3.15). Due to this constant mixing, East Bay remains well-oxygenated to the bottom, with DO concentrations not being below 8 mg L^{-1} (Figure 3.15).

Analytes measured in the fall in East Bay continued to decrease over the study period. Cations decreased with increasing water levels, driven by decreases in arsenic (R = -0.837), magnesium (R = -0.800), potassium (R = -0.858), and sodium (R = -0.892; Figure 3.16). Conversely, calcium concentration increased with increasing lake elevation (R = 0.773). Anions decreased with increasing water level, driven by decreases in carbonate (R = -0.487), chloride (R = -0.929; Figure 3.16), and sulfate (R = -0.914; Figure 3.16). Specific conductance decreased significantly over the study period (R = -0.957), from 4,820 µS/cm in 1995 to 2,480 µS/cm in 2014 (Figure 3.16). Further, TDS (R = -0.917) and hardness (R = -0.708) declined significantly with increasing water levels.

Nutrients in East Bay were strongly related to water levels in the fall. TN (R = -0.636; Figure 3.17) and TKN (R = -0.857) declined with increasing water levels, while N+N was not affected by lake elevation (R = 0.167). TP decreased significantly with increasing water level (R = -0.531) (Figure 3.17), while DP was not related to changes in lake elevation (R = -0.140).

TSI scores calculated for Secchi disk transparency and chlorophyll-α concentration classify East Bay as eutrophic in the fall. TSI scores based on transparency ranged from 73.20 (0.4 m) in 1997 to 50.75 (1.9 m) in 2003 (Figure 3.18). TSI scores based on chlorophyll-α concentration ranged from 63.83 in 2008 to 41.38 in 1995 and 2007, with scores being variable over the study period (Figure 3.18).

Analysis of TSI scores suggests that East Bay, similar to the rest of Devils Lake, is not P-limited in the fall, as evidenced by TSI scores based on TP being much greater than those based on chlorophyll-α concentration (Figure 3.19). Additionally, there is much inter-annual variation with the photic zone being dominated by smaller particles of non-algal turbidity during some years and being dominated by larger particles of algae in others (Figure 3.19).

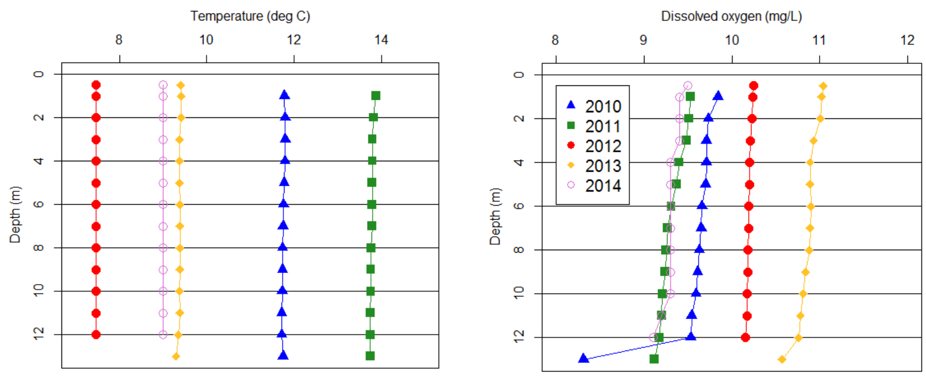

Figure 3.15: Fall temperature (left) and dissolved oxygen (right) profiles for East Bay from 2010 through 2014.

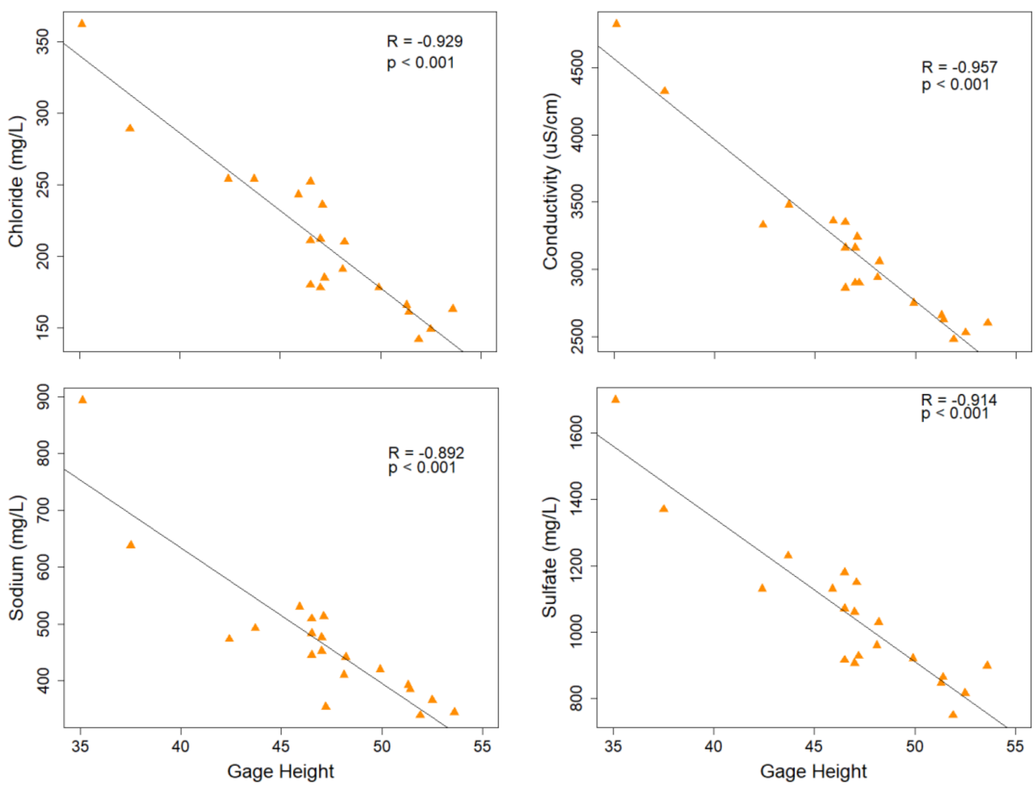

Figure 3.16: Fall concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at East Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

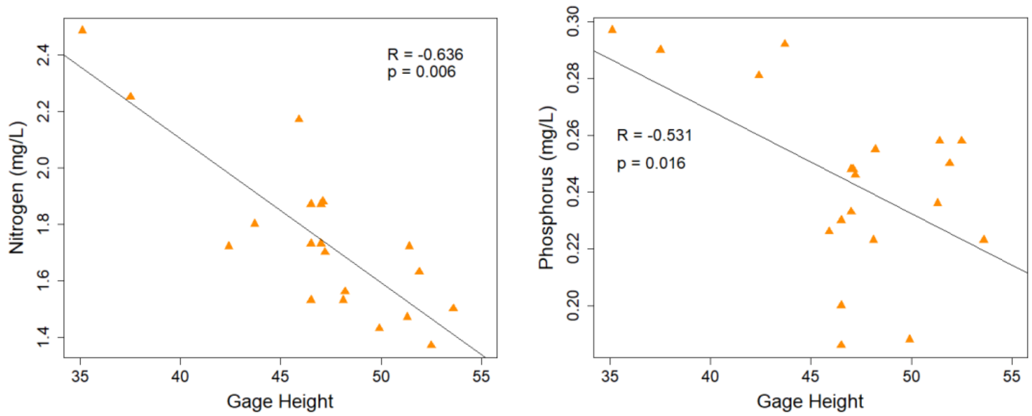

Figure 3.17: Fall concentrations of TN (left) and TP (right) plotted against lake elevation at East Bay.

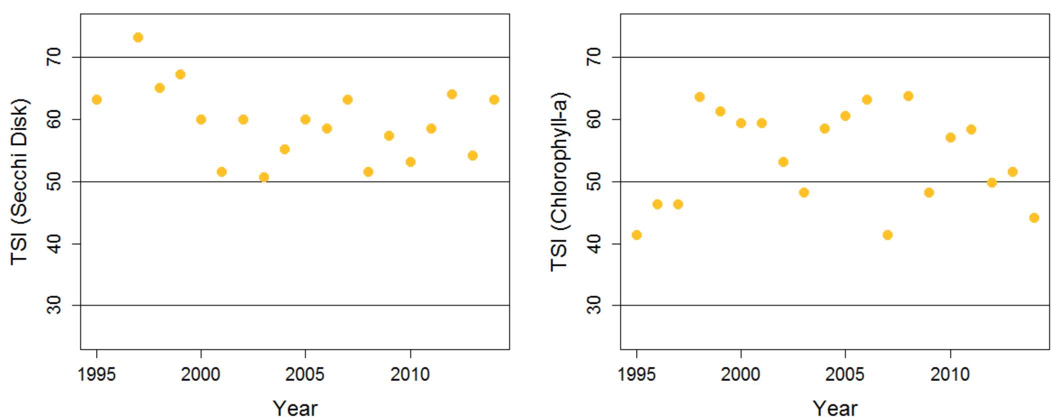

Figure 3.18: Fall TSI scores for East Bay from 1995 through 2014.

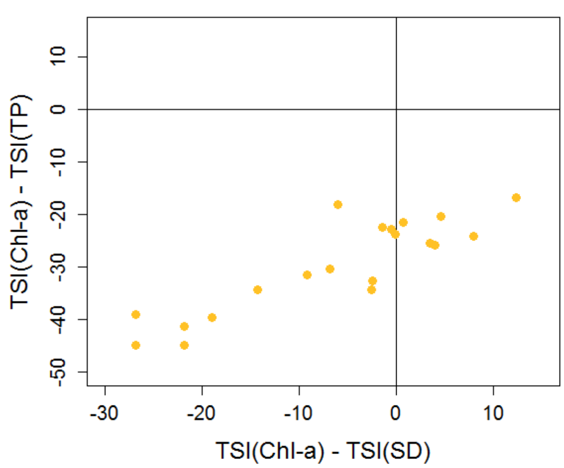

Figure 3.19: Nutrient-limitation graph for fall sampling at East Bay.

Phytoplankton assemblage sampling

The phytoplankton community in East Bay had associations with physical, temporal, and seasonal variables from 1995 to 2000. While certain taxonomic groups had a tendency to dominate communities during specific seasons (e.g., cyanobacteria had high densities in summer), the species associated varied from year-to-year. Algal communities appear to be rotating dominant species from year-to-year, with species from all groups being dominant every other year (Figure XX). For example, *Aphanothece smithii* was the dominant cyanobacteria in 1998 and 2000 sampling, but was absent during 1997 and 1999 (Table XX). Conversely, *Gloeocapsa aeruginosa* dominated the cyanobacteria community in 1997 and 1999, but was not present during 1998 and 2000 sampling (Table XX). Golden algae (*Chrysococcus sp.* and *Kephyrion sp.*) were present in relatively high densities throughout 1999 sampling, but only a few species in 1997.

Table 3.1: Presence-absence table showing common phytoplankton species and when they were detected in East Bay. An "X" signifies that the species was present during that particular season.

	1996			**1997**			**1998**			**1999**			**2000**		
	Sp	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**
Chlorophyta															
Ankyra judai		X	X					X							
Chlamydomonas tetragama	X	X													
Chlorella minutissima							X	X	X				X	X	X
Chlorella vulgaris	X	X					X								
Choricystis minor	X	X	X				X	X	X				X	X	X
Closteriopsis longissima		X						X							
Eutetramorus fottii	X	X	X												
Kirchneriella sp.										X	X	X			
Monoraphidium minutum				X	X	X					X	X			
Pediastrum duplex			X					X	X						
Planktosphaeria gelatinosa							X	X	X						
Raphidocelis microscopica								X	X				X		
Schroederia setigera		X						X	X					X	X
Stylosphaeridium chlorangielloides							X	X	X						
Chrysophyta															
Chrysococcus cordiformis					X					X	X	X			
Chrysococcus minutus										X	X	X			
Chrysococcus porifer										X	X	X			
Chrysococcus punctiformis				X	X	X				X	X	X			
Chrysococcus rufescens					X					X	X	X			
Chrysococcus rufescens var. tri.										X	X	X			
Kephyrion littorale				X	X	X				X	X	X			
Kephyrion sitta				X	X					X	X	X			
Cryptophyta															
Campylomonas reflexa								X	X						
Chroomonas acuta	X	X	X	X	X	X		X	X	X	X	X			
Plagioselmis sp.													X	X	X
Rhodomonas minuta	X				X							X			
Rhodomonas minuta var. nan.				X	X	X				X	X	X			

Table 3.1: (cont.)

	1996			1997			1998			1999			2000		
	Sp	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**
Cyanobacteria															
Anabaena curva		X						X							
Aphanizomenon flosaquae		X	X		X			X	X	X	X	X		X	X
Aphanocapsa delicatissima	X	X	X						X					X	
Aphanothece nidulans var. nid.	X	X	X												
Aphanothece smithii							X	X	X				X	X	
Gloeocapsa aeruginosa				X	X	X				X	X	X			
Gomphosphaeria salina								X	X						
Lyngbya birgei		X						X						X	
Microcystis aeruginosa		X	X					X	X		X	X		X	X
Oscillatoria angustissima					X						X	X			
Oscillatoria utermoehl								X						X	X
Pseudanabaena mucicola		X						X	X						
Snowella lacustris	X	X	X					X							
Woronichinia compacta							X	X	X						
Heterokontophyta															
Cyclostephanos sp.				X						X	X				
Cyclotella bodanica				X				X				X			
Cyclotella choctawhatcheeana								X	X						X
Cyclotella glomerata				X	X					X	X	X			
Cyclotella meneghiniana		X	X	X				X	X	X	X				
Nitzschia fonticola	X	X	X					X	X						
Nitzschia inconspicua					X	X				X		X			
Stephanodiscus hantzschii	X			X						X	X	X			
Stephanodiscus niagarae		X	X				X	X	X			X	X		X

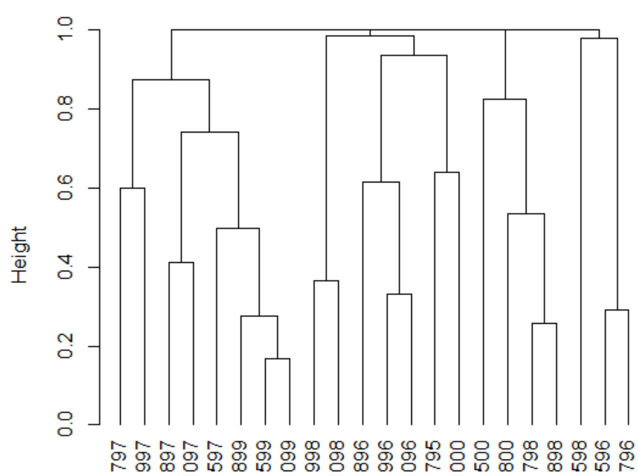

Figure 3.20: Dendrogram clustering sites based on presence and proportion of algal species for each sample at East Bay. Distances were determined using Bray-Curtis dissimilarity scores. Numbers along the x-axis represent the date of the sample (i.e., 797 is July 1997; 896 is August 1996).

3.4 East Devils Lake

The sampling station at East Devils Lake (Station ID 380235), active since 1995, is southeast of the City of Devils Lake (Figure 4.1). Accessibility to East Devils Lake can be difficult as there is no boat ramp. Instead, this site is accessed by going under the bridge at County Road 353 (i.e., Woods Rutten Road). This station is slightly protected from westerly winds, but has long fetches with easterly winds, making stratification difficult depending on prevailing wind direction.

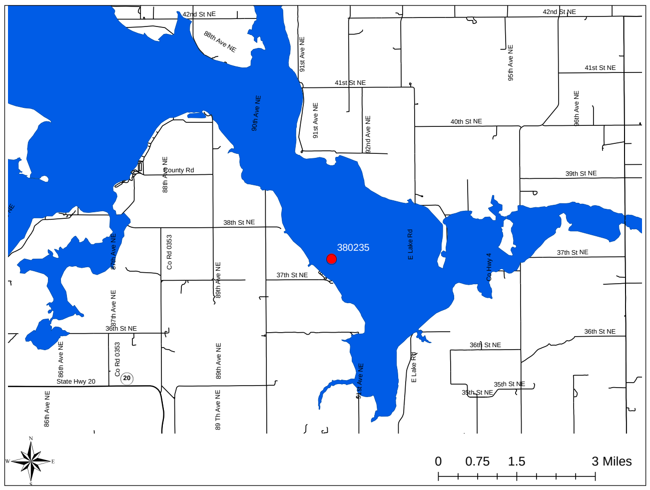

Figure 4.1: Map showing location of sampling site in East Devils Lake.

Annual winter sampling

East Devils Lake has some of the coldest bottom temperatures (approximately 2° C) compared to the rest of the lake. Despite its depth, East Devils Lake did not thermally stratify during winter sampling between 2010 and 2014 (Figure 4.2). Further, East Devils Lake is well-oxygenated, rarely being below 8 mg L^{-1} (Figure 4.2).

East Devils Lake was likely the most impacted basin by the increase in water levels. Cations declined with increasing water levels, most influenced by decreases in copper (R = -0.582), magnesium (R = -0.917), potassium (R = -0.914), and sodium (R = -0.912; Figure 4.3). Conversely, barium increased significantly with increasing water volume (R = 0.606), while calcium was nearly significant (R = 0.451). Anions decreased with increasing water level, driven by decreases in carbonate (R = -0.795), chloride (R = -0.925; Figure 4.3), and sulfate (R = -0.925; Figure 4.3). Also, specific conductance (R = -0.920; Figure 4.3), TDS (R = -0.922), and hardness (R = -0.911) have decreased significantly since 1995. Specific conductance has decreased from 11,000 µS/cm in 1995 to 3,050 µS/cm in 2014 (Figure 4.3), likely due to the influx of fresher water from the western side of the basin.

Like other sites in Devils Lake, nutrient levels in East Devils Lake did not uniformly decline with increasing water levels. Increasing water levels correlated to decreases in ammonia (R =

-0.746), TN (R = -0.612; Figure 4.4), and TKN (R = -0.785), while N+N was not affected by increased water volume (R = -0.065). Additionally, TP (R = 0.043; Figure 4.4) and DP (R = 0.141) concentrations were not related to water level.

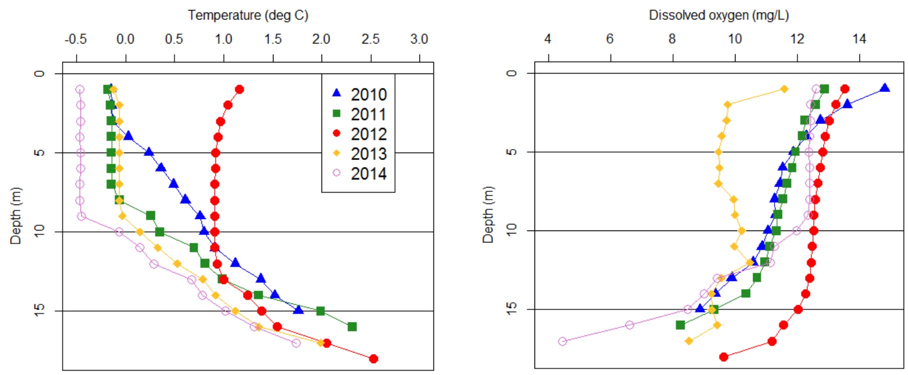

Figure 4.2: Winter temperature (left) and DO (right) profiles for East Devils Lake from 2010 through 2014.

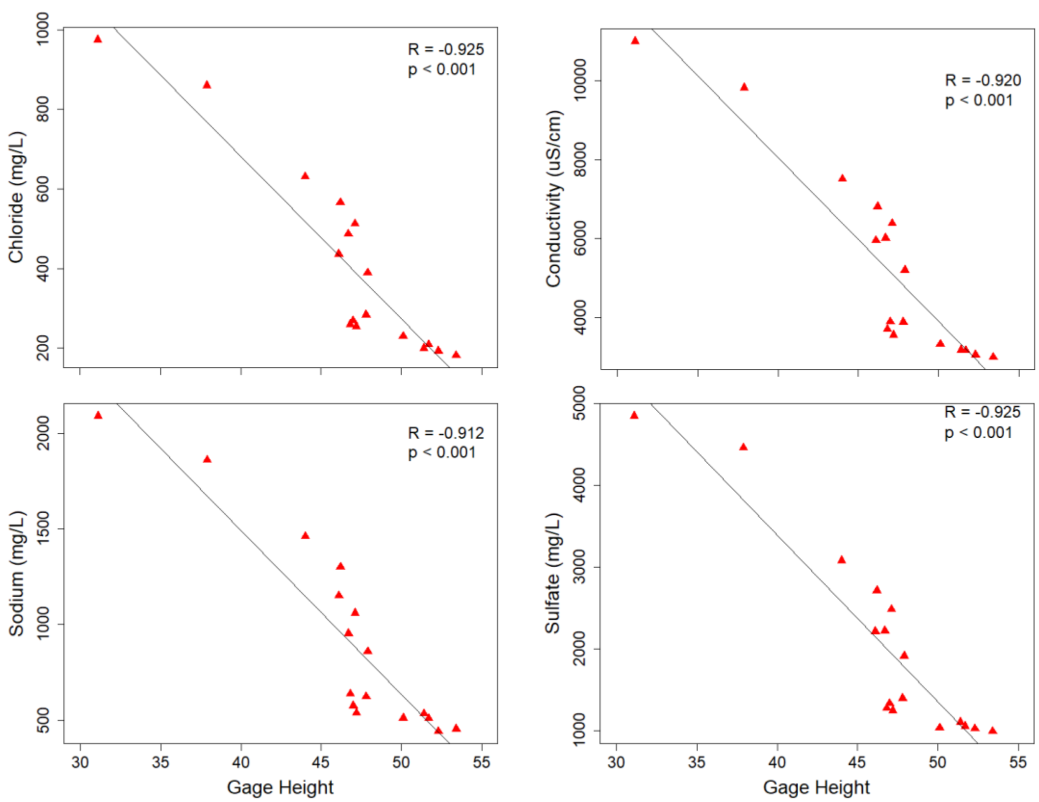

Figure 4.3: Winter concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at East Devils Lake. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

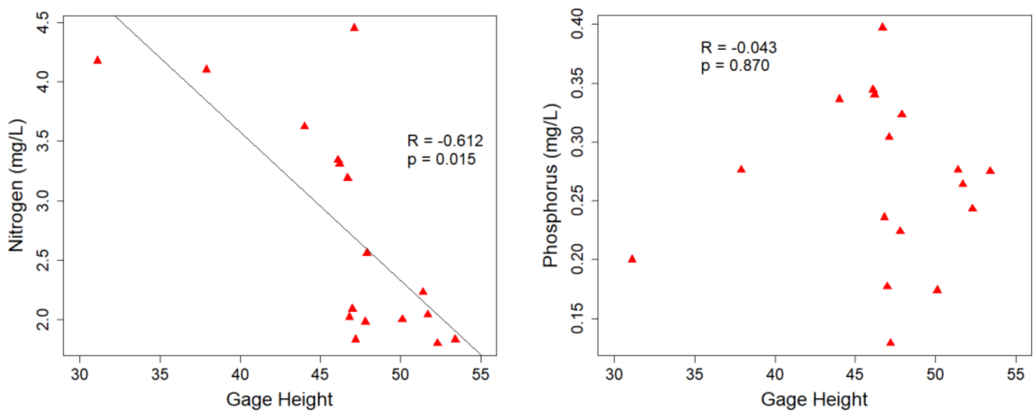

Figure 4.4: Winter concentrations of TN (left) and TP (right) plotted against lake elevation at East Devils Lake.

Annual spring sampling

Like the rest of Devils Lake, East Devils Lake rarely stratifies during spring sampling, with only one occurrence since 2010 (2014) (Figure 4.5). Also, DO levels in East Devils Lake remain relatively high during spring sampling, not measured below 10 mg L^{-1} since 2010 (Figure 4.5).

Analytes in East Devils Lake continued to decline in the spring. Cations decreased with increasing water levels, most influenced by decreases in arsenic (R = -0.790), copper (R = -0.644), magnesium (R = -0.926), potassium (R = -0.924), sodium (R = -0.920; Figure 4.6), and zinc (R = -0.470). Barium, however, increased significantly with increasing water volume (R = 0.659). Anions decreased as well (R = -0.898), driven by decreases in carbonate (R = -0.815), chloride (R = -0.901; Figure 4.6), and sulfate (R = -0.896; Figure 4.6). Also, specific conductance (R = -0.914; Figure 4.6), TDS (R = -0.907), and hardness (R = -0.920) decreased significantly with increasing water volume. Similar to winter sampling, specific conductance decreased drastically, going from 9,900 µS/cm in 1995 to 2,840 µS/cm in 2014 (Figure 4.6).

Levels of nitrogen correlated to water level in East Devils Lake, while phosphorus was not affected. Concentrations of TN (R = -0.766; Figure 4.7) and TKN (R = -0.773) declined strongly with increasing lake elevation, while N+N was not affected (R = -0.077). TP (R = 0.071; Figure 4.7) and DP (R = 0.077) were not impacted by increases in lake elevation.

TSI scores for Secchi disk and chlorophyll-α in the spring classify East Devils Lake as mesotrophic or eutrophic. TSI scores based on Secchi disk ranged from a peak of 60.00 (1.0 m) in 2005 and 2006 to a low of 33.03 (6.5 m) in 2014 (Figure 4.8). TSI scores based on chlorophyll-α were much more variable over the study period, ranging from 64.78 in 2004 to 41.38 in 1997, 1999, 2006, 2010, and 2014 (Figure 4.8).

East Devils Lake follows similar trends to the rest of the lake when it comes to nutrient limitation. East Devils Lake was not P-limited, with TSI(Chl)-TSI(TP) values strongly negative throughout the study period (Figure 4.9). For most of the study period, the basin was mostly dominated by small, non-algal particles, leading to transparency TSI scores consistently being higher than chlorophyll-α TSI scores (Figure 4.9). In recent years, however, the basin has switched more to being dominated by larger particles of algae, leading to higher transparency depths and higher chlorophyll-α concentrations (Figure 4.9).

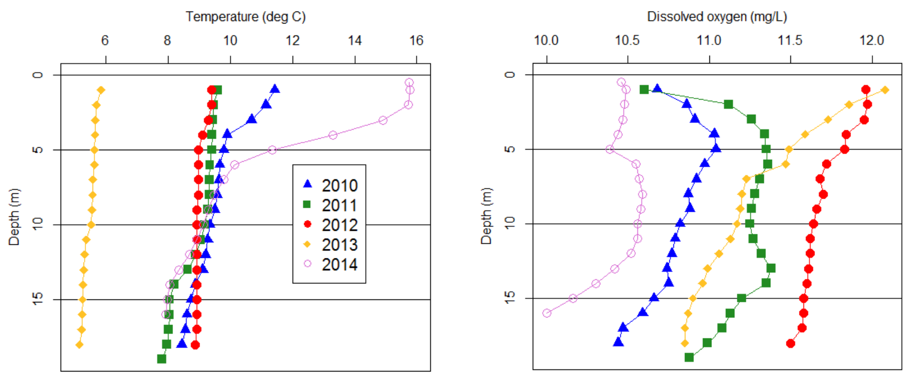

Figure 4.5: Spring temperature (left) and DO (right) profiles for East Devils Lake from 2010 through 2014.

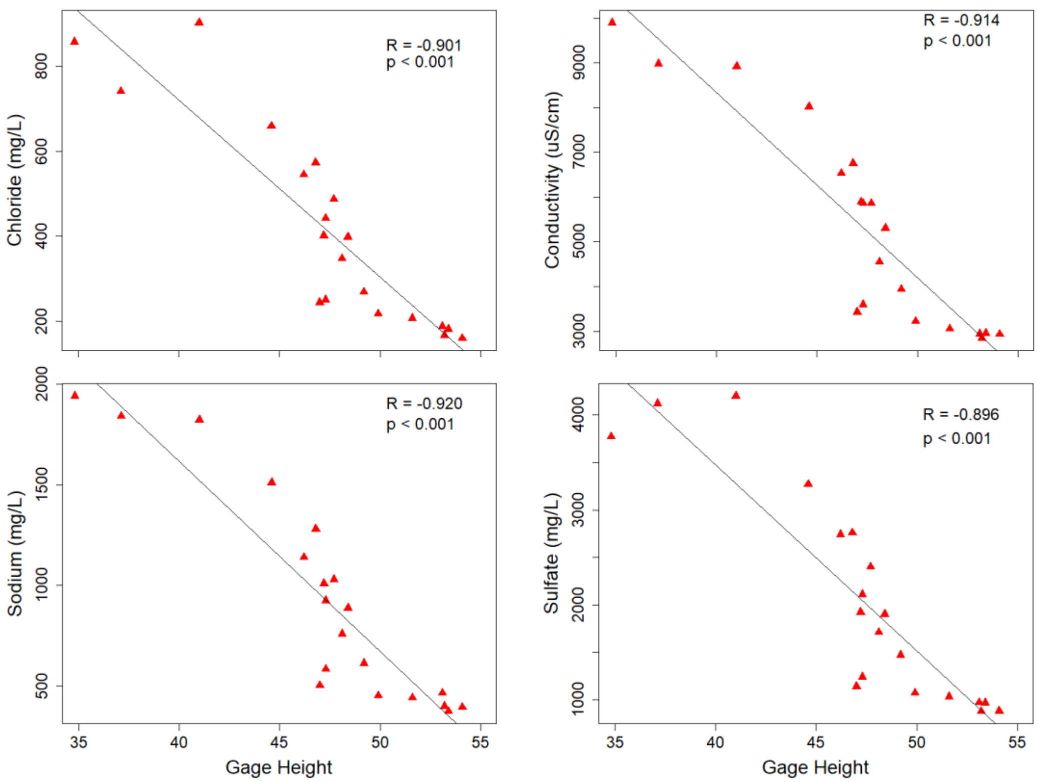

Figure 4.6: Spring concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at East Devils Lake. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

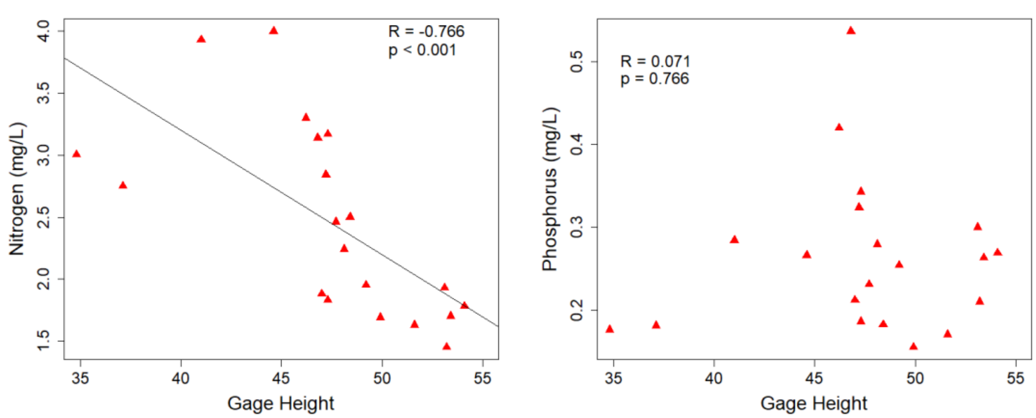

Figure 4.7: Spring concentrations of TN (left) and TP (right) plotted against lake elevation at East Devils Lake.

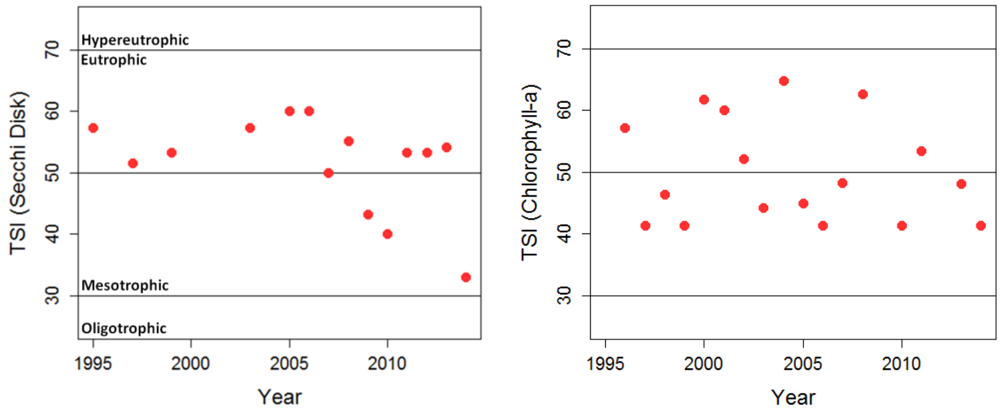

Figure 4.8: Spring TSI scores for East Devils Lake from 1995 through 2014.

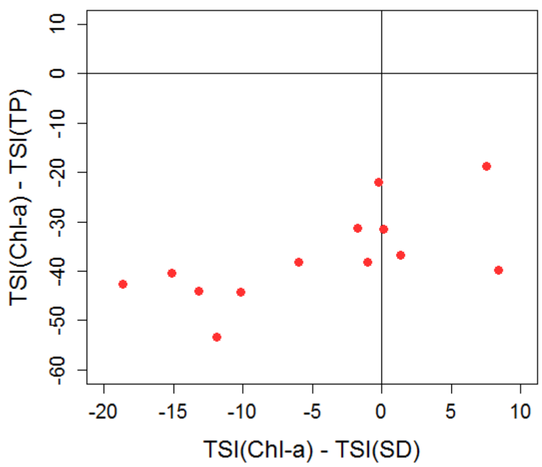

Figure 4.9: Nutrient-limitation graph for spring sampling at East Devils Lake.

Annual summer sampling

Unlike the rest of the lake, East Devils Lake commonly stratifies during the summer months, occurring four times since 2010 (not in 2014) (Figure 4.10). Typically, thermal stratification occurs between 11 to 15 meters of depth (Figure 4.10). Every year since 2010, East Devils Lake has had an area in the water column below the State's standard for DO, starting between 14 and 16 meters of depth (Figure 4.10).

Like the other seasons, summer sampling in East Devils Lake showed strong decreases for measured analytes with increasing water level. Cations decreased with increasing lake elevation, driven by decreases in copper (R = -0.533), magnesium (R = -0.910), potassium (R = -0.901), sodium (R = -0.914; Figure 4.11), and zinc (R = -0.554). Conversely, calcium increased with increasing water levels (R = 0.680). Barium, which had increased during winter and spring sampling, had no relationship with rising water levels. Anions also had a strong decrease with increasing water levels (R = -0.927), most influenced by decreases in carbonate (R = -0.780), chloride (R = -0.921; Figure 4.11), and sulfate (R = -0.929; Figure 4.11). Also, there were strong decreases in specific conductance (R = -0.927; Figure 4.11), TDS (R = -0.925), hardness (R = -0.898), and alkalinity (R = -0.853). Specific conductance decreased from 9,650 µS/cm in 1995 to 2,820 µS/cm in 2014 (Figure 4.11).

Nutrient levels in the summer were affected by changing water levels in East Devils Lake. TN (R = -0.644; Figure 4.12) and TKN (R = -0.801) were strongly correlated with increasing lake elevation. TP was not related to increasing water levels (R = 0.157; Figure 4.12), while DP increased significantly over time (R = 0.417).

TSI scores in East Devils Lake have fluctuated greatly over the sample period, ranging from nearly oligotrophic to hypereutrophic. TSI scores based on Secchi disk transparency have varied from 73.20 (0.4 m) in 1998 to 34.67 (5.8 m) in 1996, with no defined trend since 1995 (Figure 4.13). TSI scores based on chlorophyll-α peaked at 79.08 in 1998, decreasing to a low of 48.18 in 2011, then increasing in 2012 and 2014 (no sample in 2013) to scores around hypereutrophic (Figure 4.13).

Like most of Devils Lake, East Devils Lake is not P-limited in the summer, with phosphorus-TSI scores greatly exceeding TSI scores for chlorophyll-α (Figure 4.14). Additionally, East Devils Lake is commonly dominated by larger particles of algae during the summer months, evidenced by chlorophyll-α TSI scores commonly exceeding Secchi disk TSI scores (Figure 4.14).

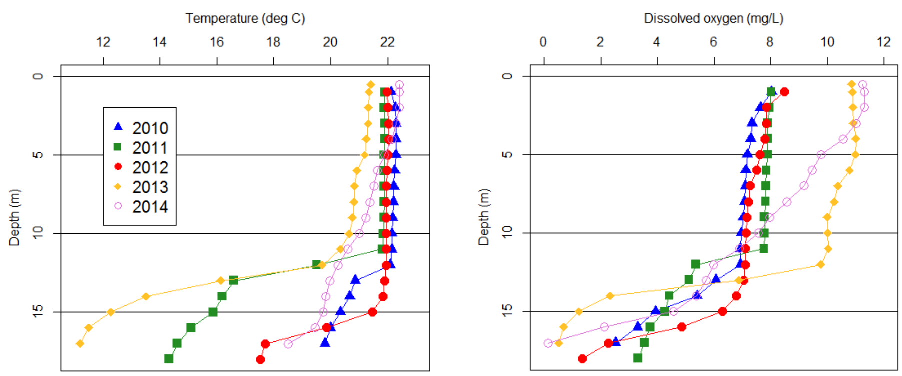

Figure 4.10: Summer temperature (left) and DO (right) profiles for East Devils Lake from 2010 through 2014.

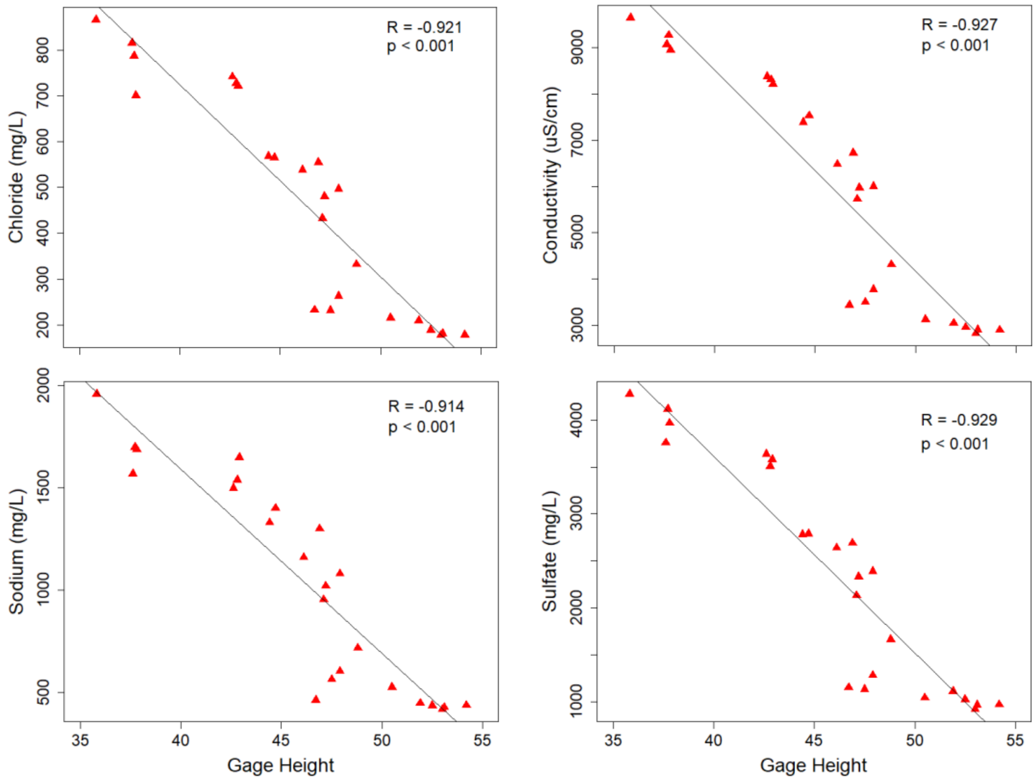

Figure 4.11: Summer concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at East Devils Lake. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

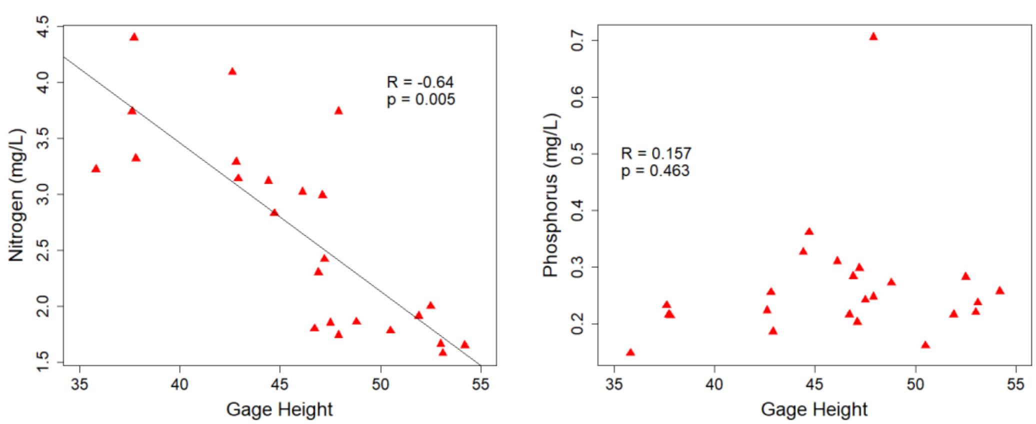

Figure 4.12: Summer concentrations of TN (left) and TP (right) plotted against lake elevation at East Devils Lake.

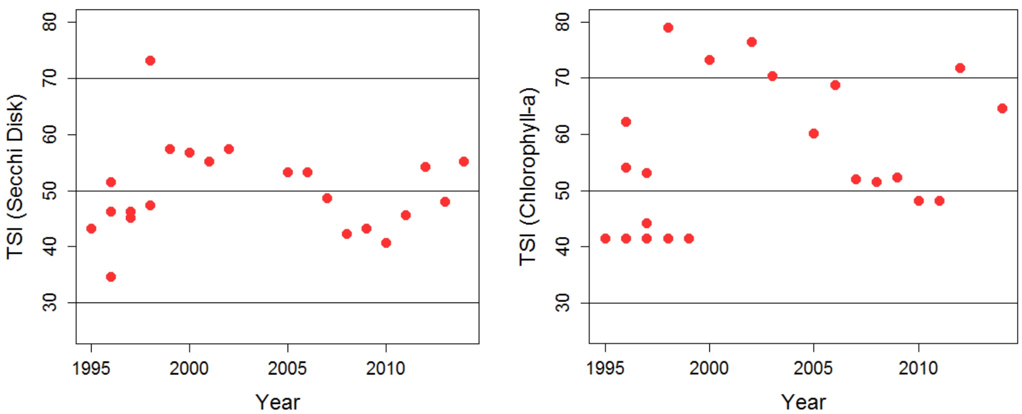

Figure 4.13: Summer TSI scores for East Devils Lake from 1995 through 2014.

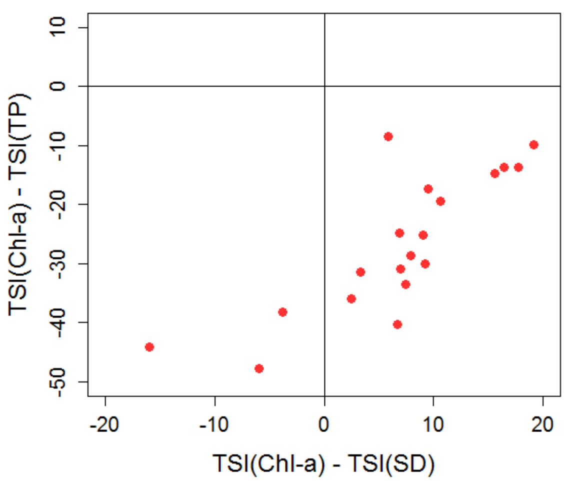

Figure 4.14: Nutrient-limitation graph for summer sampling at East Devils Lake.

Annual fall sampling

Similar to the rest of Devils Lake, the station at East Devils Lake does not thermally stratify during fall sampling (Figure 4.15). Also, East Devils Lake remains well-oxygenated throughout the entire water column, not being below 8 mg L^{-1} since 2010 (Figure 4.15).

Measured ionic concentrations continued to decline in East Devils Lake in the fall due to a high influx of "fresher" water from the western side of the lake. Cations decreased with increasing water level, driven by decreases in arsenic (R = -0.855), copper (R = -0.766), magnesium (R = -0.910), potassium (R = -0.921), sodium (R = -0.918; Figure 4.16), and zinc (R = -0.602). Conversely, barium (R = 0.525) and calcium (R = 0.485) increased with increasing water volume. Anions decreased with increasing water level as well, driven by reductions in carbonate

(R = -0.762), chloride (R = -0.905; Figure 4.16), and sulfate (R = -0.905; Figure 4.16). Specific conductance continued to reduce strongly with dilution (R = -0.920), going from 9,920 µS/cm in 1995 to 2,910 µS/cm in 2014 (Figure 4.16).

There was some effect of water level on nutrient concentrations in East Devils Lake. Concentrations of TN (R = -0.757; Figure 4.17) and TKN (R = -0.860) declined significantly with increasing water levels, while N+N was not affected by the change in lake elevation (R = 0.184). TP (R = -0.183; Figure 4.17) and DP (R = 0.213) were not affected by water level increases.

TSI scores based on chlorophyll-α and Secchi disk transparency classify East Devils Lake as eutrophic, with occasional spikes into mesotrophic and hypereutrophic. TSI scores based on Secchi disk transparency ranged from 77.35 (0.3 m) in 1999 to 46.23 (2.6 m) in 2007 (Figure 4.18). Scores based on chlorophyll-α were still highly variable, but showed a decreasing trend over time, ranging from 70.73 in 2004 to 37.40 in 2007 (Figure 4.18).

East Devils Lake is not P-limited in the fall, as evidenced by TSI scores based on TP being much greater than those based on chlorophyll-α (Figure 4.19). Further, the photic zone in East Devils Lake is typically dominated by larger particles of algae, as evidenced by TSI scores based on Secchi disk transparency typically being larger than those based on chlorophyll (Figure 4.19).

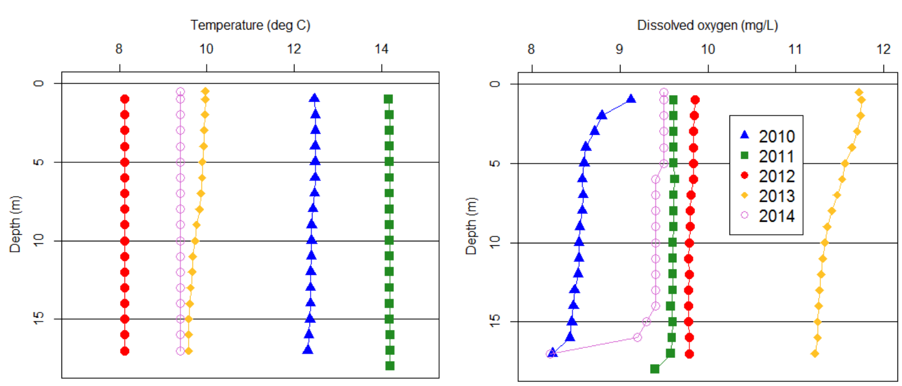

Figure 4.15: Fall temperature (left) and DO (right) profiles for East Devils Lake from 2010 through 2014.

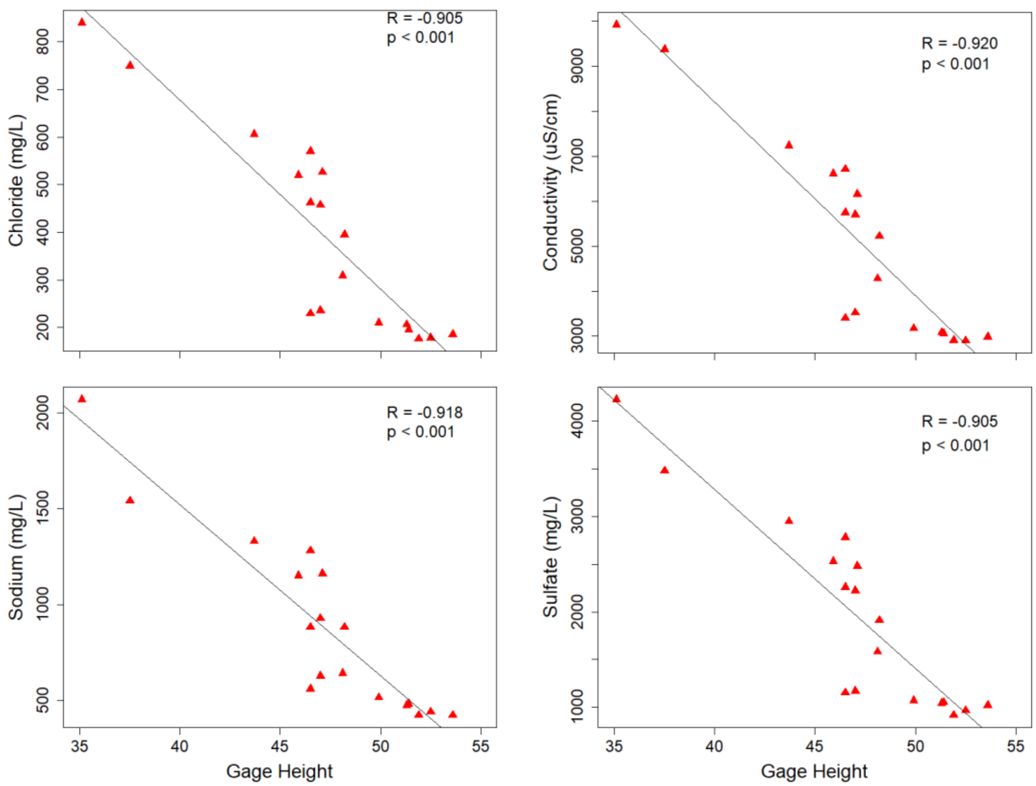

Figure 4.16: Fall concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at East Devils Lake. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

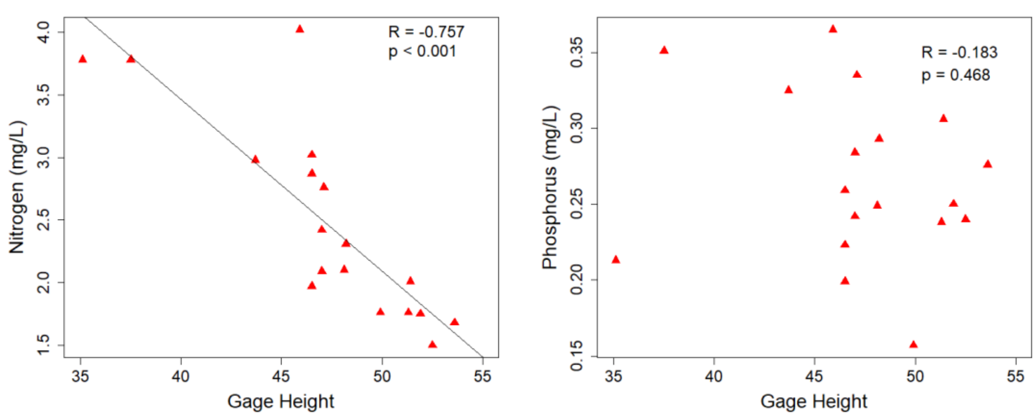

Figure 4.17: Fall concentrations of TN (left) and TP (right) plotted against lake elevation at East Devils Lake.

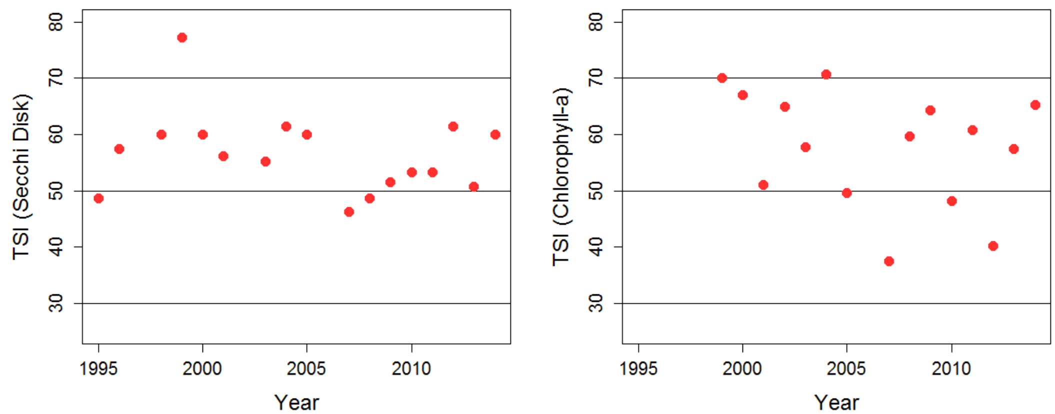

Figure 4.18: Fall TSI scores for East Devils Lake from 1995 through 2014.

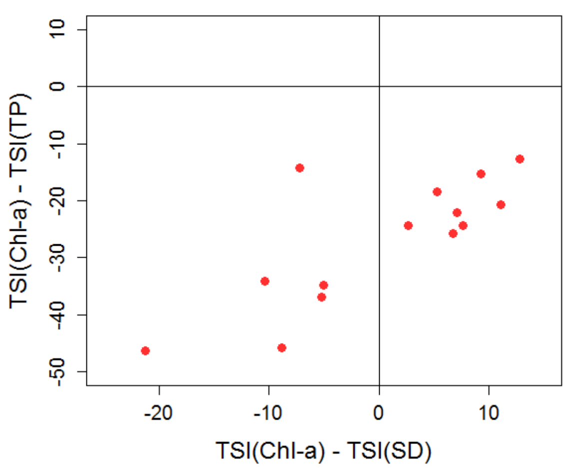

Figure 4.19: Nutrient-limitation graph for fall sampling at East Devils Lake.

Phytoplankton assemblage sampling

Though commonly dominated by cyanobacteria, the algal community in East Devils Lake follows similar patterns to the rest of the basin. Summers of 1996 and 1998 had the highest densities of cyanobacteria of all sampling seasons, with many species represented during those timeframes. Spring samples during those years, however, yielded little or no cyanobacteria in East Devils Lake, and were instead dominated by diatoms and chlorophytes (Table 4.1). Bray-Curtis dissimilarities showed that algal communities in East Devils Lake, like other sites, were most similar every other year, rather than based on seasonality (Figure 4.20). For example, communities from 1997 were most similar to algal communities in 1999 (Figure 4.20).

Table 4.1: Presence-absence table showing common phytoplankton species and when they were detected in East Devils Lake. An "X" signifies that the species was present during that particular season.

	1996			1997			1998			1999			2000		
	Sp	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**
Chlorophyta															
Chlamydomonas tetragama		X				n/a									
Chlorella minutissima						n/a	X	X	X				X	X	X
Chlorella vulgaris	X	X	X			n/a	X		X				X		
Choricystis minor	X	X	X			n/a	X	X	X				X		X
Kirchneriella lunaris						n/a				X	X	X			
Monoraphidium minutum				X	X	n/a				X					
Stylosphaeridium chlorangielloides						n/a	X	X	X						
Chrysophyta															
Chrysococcus cordiformis						n/a				X	X	X			
Chrysococcus minutus						n/a				X	X	X			
Chrysococcus punctiformis				X	X	n/a				X	X	X			
Chrysococcus rufescens				X	X	n/a					X	X			
Chrysococcus rufescens var. com.						n/a				X	X	X			
Chrysococcus rufescens var. tri.						n/a				X	X	X			
Kephyrion cupuliforme						n/a				X	X	X			
Kephyrion littorale				X	X	n/a				X	X	X			
Kephyrion sitta				X	X	n/a				X	X	X			
Cryptophyta															
Chroomonas acuta	X				X	n/a	X	X							
Rhodomonas minuta		X		X	X	n/a				X	X				
Rhodomonas minuta var. nan.				X	X	n/a				X	X	X			
Rhodomonas pusilla					X	n/a				X	X				
Cyanobacteria															
Anabaena curva		X	X			n/a		X							
Aphanizomenon flosaquae		X	X		X	n/a		X	X	X	X	X		X	X
Aphanocapsa delicatissima		X	X			n/a			X						X
Aphanothece nidulans var. nid.	X	X	X			n/a									
Aphanothece smithii						n/a		X	X				X		X
Gloeocapsa aeruginosa				X	X	n/a				X	X	X			
Lyngbya birgei		X				n/a		X					X	X	
Microcystis aeruginosa		X	X			n/a		X	X	X	X			X	X
Microcystis incerta						n/a				X	X	X			
Myxobaktron sp.	X					n/a		X	X						
Pseudanabaena mucicola		X				n/a		X						X	X
Spirulina sp.		X				n/a		X							
Woronichinia compacta						n/a		X	X						
Heterokontophyta															
Chaetoceros muelleri	X	X	X			n/a		X							
Cyclostephanos sp.						n/a				X	X	X			
Cyclotella glomerata						n/a				X	X	X			
Cyclotella meneghiniana			X			n/a				X	X	X			X
Nitzschia fonticola		X	X			n/a		X	X						
Nitzschia inconspicua					X	n/a				X	X	X			
Stephanodiscus niagarae						n/a							X		X

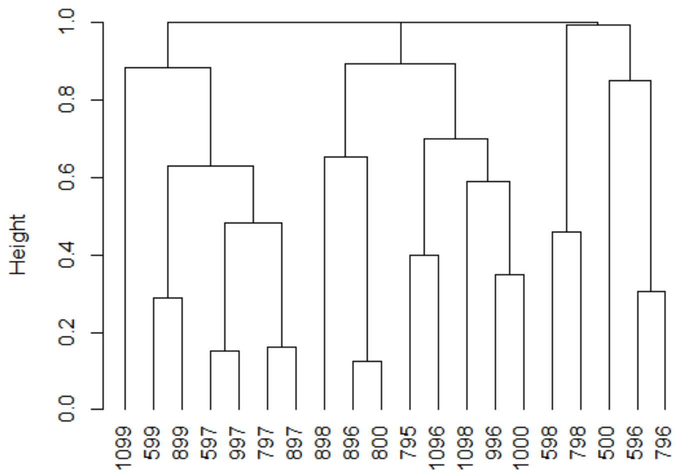

Figure 4.20: Dendrogram clustering sites based on presence and proportion of algal species for each sample at East Devils Lake. Distances were determined using Bray-Curtis dissimilarity scores. Numbers along the x-axis represent the date of the sample (i.e., 797 is July 1997; 896 is August 1996).

3.5 West Bay

The sampling station at West Bay (Station ID 380236) is southwest of Graham's Island (Figure 5.1). This station has long fetches with easterly and westerly winds, making thermal stratification difficult due to constant mixing.

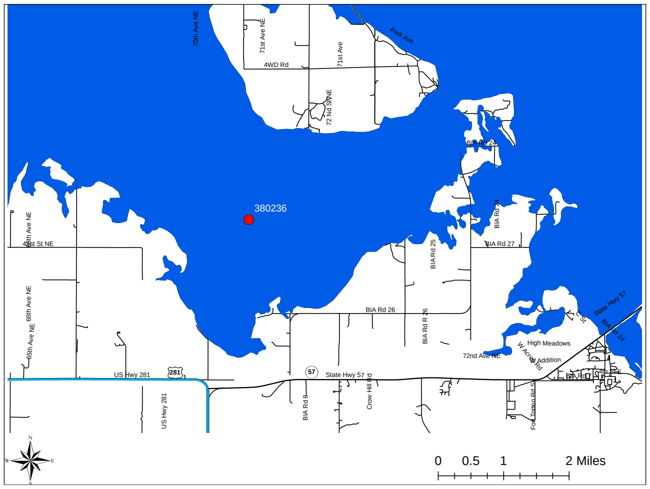

Figure 5.1: Map showing location of sampling site in West Bay.

Annual winter sampling

There is occasional reverse thermal stratification in the winter in West Bay (2011, 2014) with some of the warmest near-bottom water temperatures among sampling sites (Figure 5.2). Other than in 2014, West Bay has stayed well-oxygenated from top to bottom since 2010, with DO rarely below 5 mg/L (Figure 5.2).

Analytes measured in the winter in West Bay were greatly affected by increases in water level. Sodium, a common cation, was not significantly related to increasing water levels over the course of the project (R = -0.235); however, sodium decreased significantly since the opening of the west-end outlet in April of 2005 (R = -0.819; Figure 5.3). Similarly, chloride and sulfate, common anions, were not related to increasing water levels overall (R = -0.032 and 0.361, respectively), but both declined significantly since April 2005 (R = -0.959 and -0.907, respectively; Figure 5.3). Finally, specific conductance declined significantly since the opening of the outlet (-0.951), despite no relationship since 1995 (R = -0.102; Figure 5.3).

The effect of dilution (i.e., lake level increase) on nutrients measured in West Bay in the winter was variable. Based on the 1995 to 2014 sampling period, concentrations of TN (R = -0.817; Figure 5.4), TKN (R = -0.768), N+N (R = -0.573), and ammonia (R = -0.544) declined significantly with increasing water levels. Conversely, neither TP (R = -0.056; Figure 5.4) nor DP (R = -0.072) were related to increases in water level.

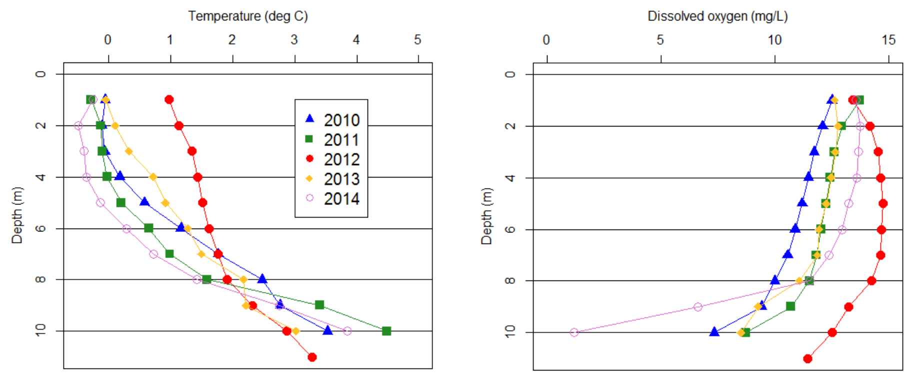

Figure 5.2: Winter temperature (left) and DO (right) profiles for West Bay from 2010 through 2014.

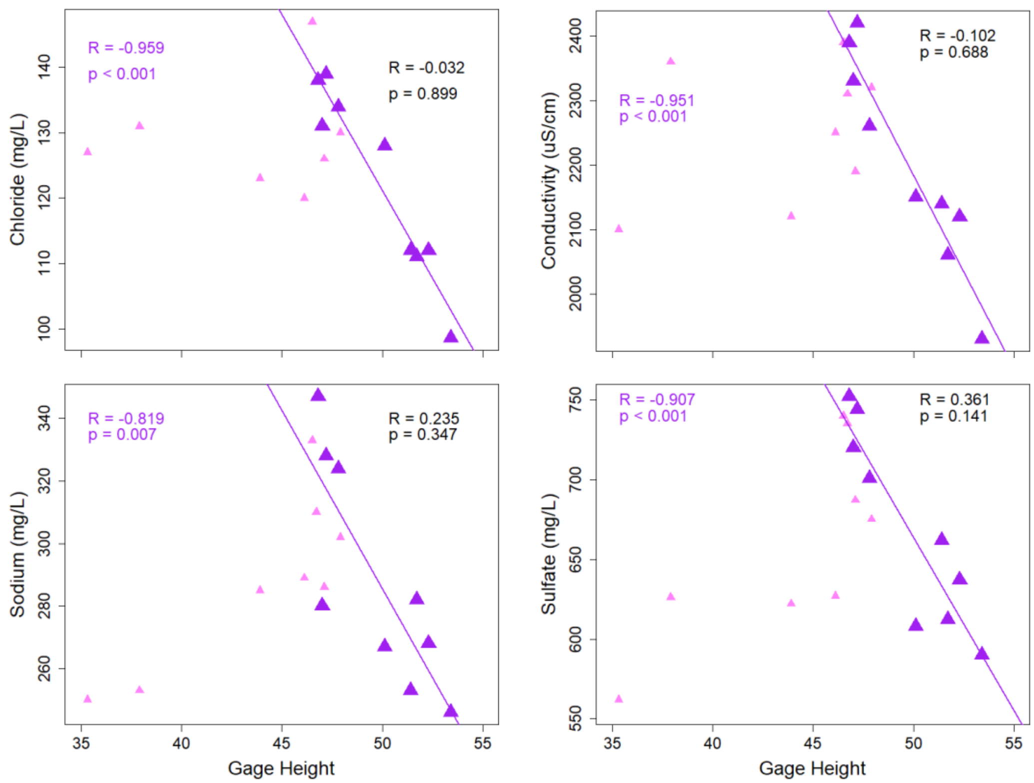

Figure 5.3: Winter concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at West Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5). (▲ is from 2005 through 2014; ▲ is from 1995 through 2004)

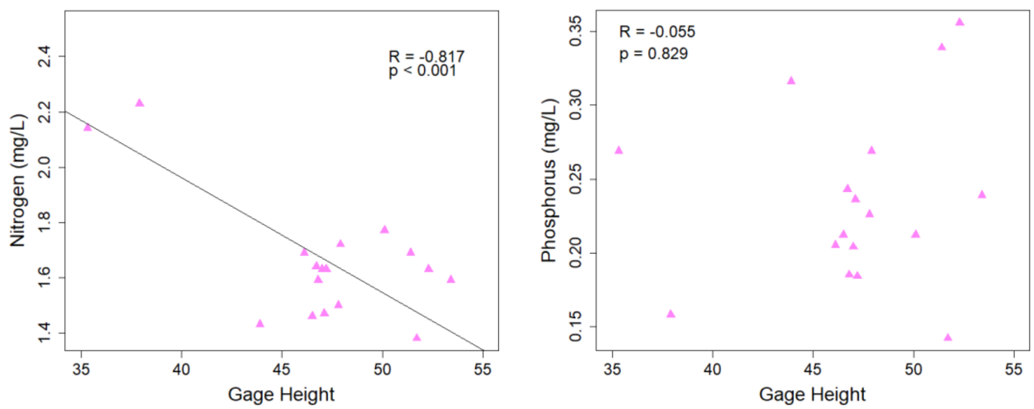

Figure 5.4: Winter concentrations of TN (left) and TP (right) plotted against lake elevation at West Bay.

Annual spring sampling

Due to the aforementioned long fetches at West Bay, thermal stratification is rare for this station, occurring once since 2010 (2014) (Figure 5.5). Due to this constant mixing in the spring, the water column remains well-oxygenated, never below 5 mg/L (Figure 5.5).

Over the study period, there appears to be no effect of lake elevation on measured ionic concentrations; however, there appears to be an initial increase in most measured ions, followed by a subsequent decrease with increased dilution correlating with the opening of the west-end outlet in April 2005. Sodium, a common cation, did not correlate with water level over the project period (R = -0.060), but declined significantly following the opening of the west-end outlet (R = -0.931; Figure 5.6). Chloride and sulfate, common anions, declined following the opening of the west-end outlet (R = -0.931 and -0.899, respectively), despite only the latter significantly increasing over the project period (R = 0.494; Figure 5.6). Specific conductance was not related to increasing water levels over the project period (R = 0.291), but decreased significantly since 2005 (R = -0.971; Figure 5.6).

Trends with nutrient concentrations varied over the study period. Concentrations of TN (R = -0.705; Figure 5.7) and TKN (R = -0.723) decreased with increasing water levels. Concentrations of N+N, however, were not affected by change in lake elevation (R = 0.187). TP increased significantly with rising water levels (R = 0.481; Figure 5.7), while DP was not significantly affected (R = 0.157).

TSI scores varied considerably in West Bay in the spring. Based on TSI scores using transparency and chlorophyll-α, West Bay was classified between mesotrophic and eutrophic in the spring (Figure 5.8). TSI scores based on Secchi disk transparency varied from 63.22 (0.8 m) in 1997 to 33.48 (6.3 m) in 2010 (Figure 5.8). Over the study period, TSI scores based on chlorophyll-α varied greatly, ranging from 59.48 in 1997 to 41.38 in 1996, 1998, 2002, 2004, 2010, and 2014 (Figure 5.8).

West Bay is not P-limited in the spring, with negative scores based on high TP TSI scores and relatively low chlorophyll-α concentrations (Figure 5.9). Throughout most of the study period, West Bay was dominated by smaller particles of algal and non-algal turbidity inhibiting transparency. During the past few years, however, West Bay has begun to be dominated by more, larger algal colonies, indicative of more effective zooplankton grazing.

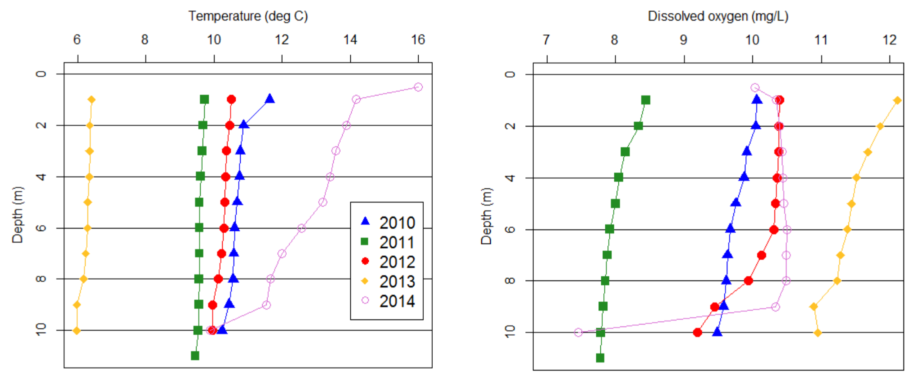

Figure 5.5: Spring temperature (left) and DO (right) profiles for West Bay from 2010 through 2014.

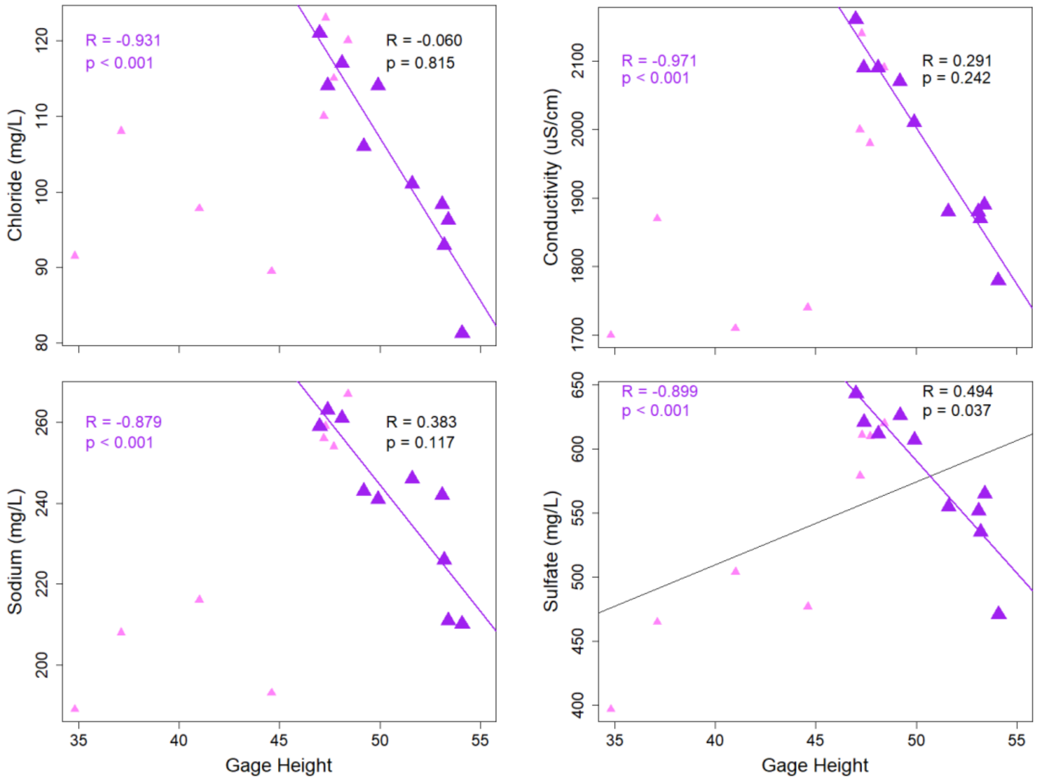

Figure 5.6: Spring concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at West Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5). (▲ is from 2005 through 2014; ▴ is from 1995 through 2004)

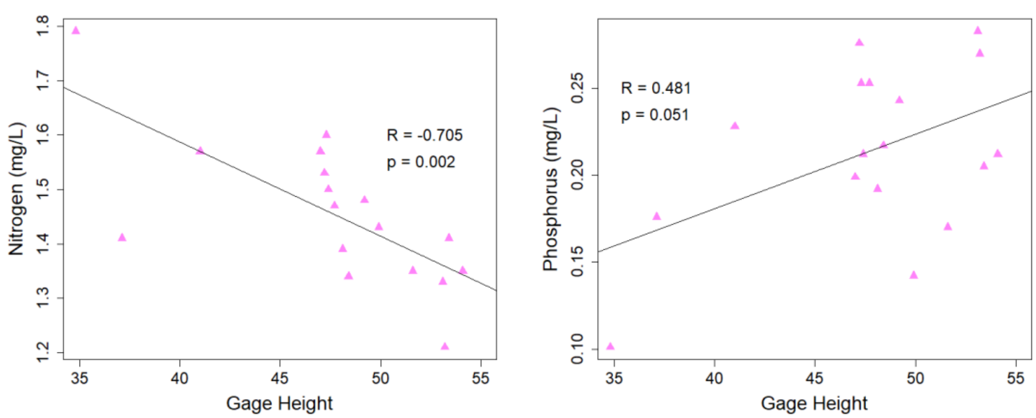

Figure 5.7: Concentrations of TN (left) and TP (right) plotted against lake elevation at West Bay for spring sampling. Values in the upper-right corner are Pearson correlation coefficients (R) and the p-value (significant at $p < 0.05$).

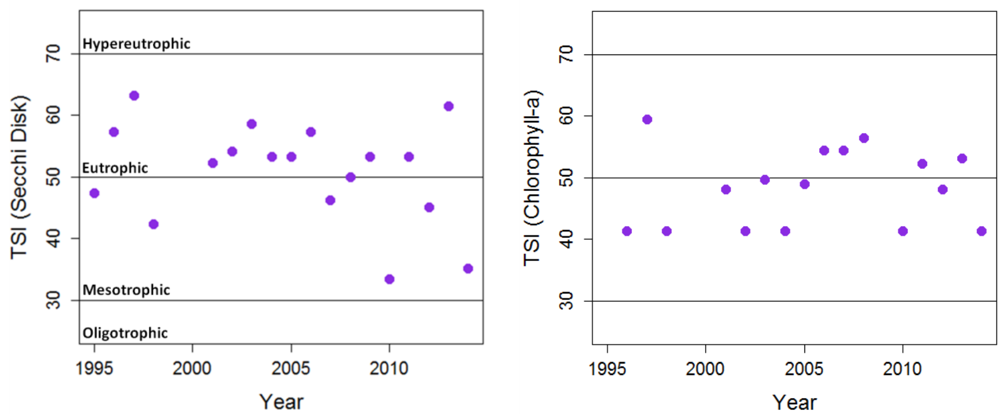

Figure 5.8: Spring TSI scores for West Bay from 1995 through 2014.

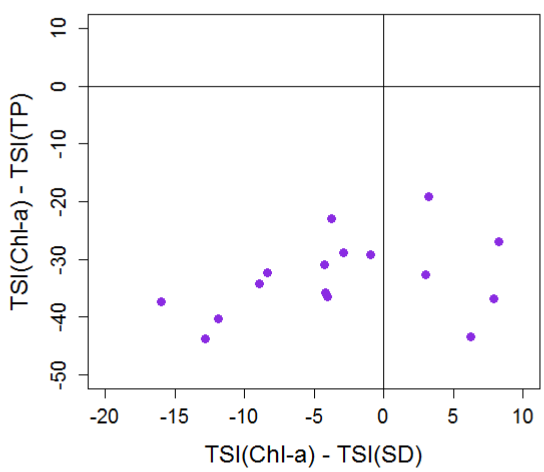

Figure 5.9: Nutrient-limitation graph for spring sampling at West Bay.

Annual summer sampling

The station at West Bay exhibited thermal stratification only once since 2010 (2013) during summer sampling (Figure 5.10). Regular mixing at this site does maintain a well-oxygenated water column, with DO concentrations only going below the State's standard once in 2014 (Figure 5.10).

Trends during summer sampling in West Bay remained similar to other seasons. Sodium, a common cation, increased significantly over the course of the project (R = 0.522), but decreased significantly since the opening of the west-end outlet (R = -0.872; Figure 5.11). Chloride and sulfate, common anions, increased with increasing water levels since 1995 (R = 0.393 [not significant] and 0.672, respectively), but decreased significantly since 2005 (R = -0.990 and

-0.894, respectively; Figure 5.11). Finally, specific conductance decreased strongly since the opening of the west-end outlet (R = -0.970), but has increased since the beginning of the project (R = 0.550; Figure 5.11).

Nutrient concentrations remained relatively unchanged over the study period. TN (R = -0.029; Figure 5.12), TKN (R = -0.048), and N+N (R = 0.304) were not affected by increasing water levels. Similarly, TP (R = -0.022; Figure 5.12) and DP (R = 0.172) were not affected by rising water.

TSI scores remained relatively unchanged over the study period, classifying West Bay between mesotrophic and eutrophic in the summer (Figure 5.13). TSI scores based on Secchi disk transparency ranged from 69.99 (0.5 m) in 2003 to 48.00 (2.3 m) in 2010, with an outlying peak of 93.18 (0.1 m) in 1997 (Figure 5.13). TSI scores based on chlorophyll-α concentration displayed much more variability than those for transparency. Chlorophyll-α TSI scores varied from 68.31 in 2012 to 34.58 in 2007 and 2010 (Figure 5.13).

West Bay is not P-limited in the summer, demonstrated by TSI(TP) scores being much higher than TSI(Chlorophyll-α) (Figure 5.14). There is some inter-annual variability, however, when it comes to the difference between chlorophyll and transparency. West Bay appears to differ annually as to whether the photic zone is dominated by small particles of algal and non-algal turbidity or dominated by large particles of algae, which is indicative of strong zooplankton grazing and/or a lack of strong, whole-column mixing.

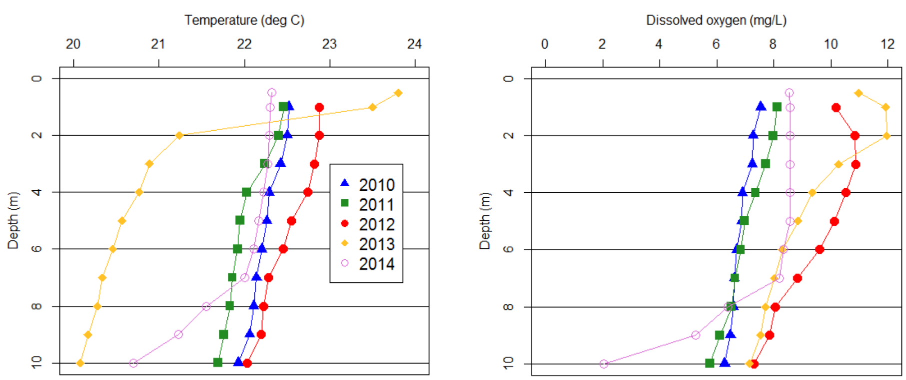

Figure 5.10: Summer temperature (left) and DO (right) profiles for West Bay from 2010 through 2014.

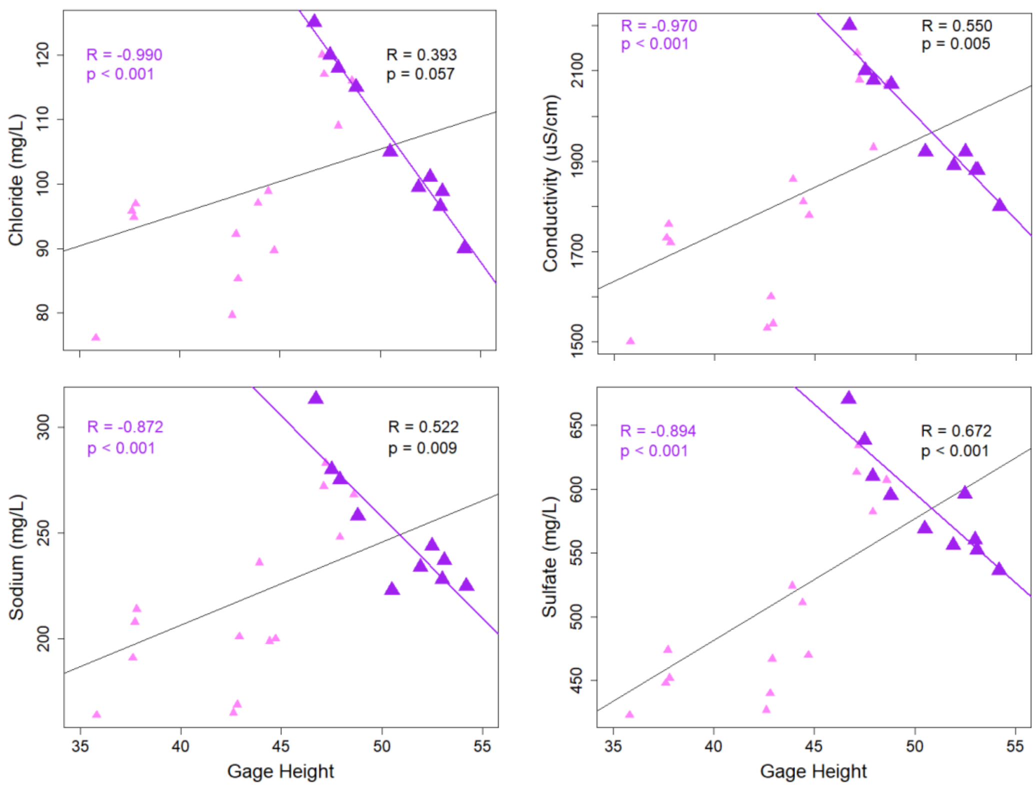

Figure 5.11: Summer concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at West Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5). (▲ is from 2005 through 2014; ▲ is from 1995 through 2004)

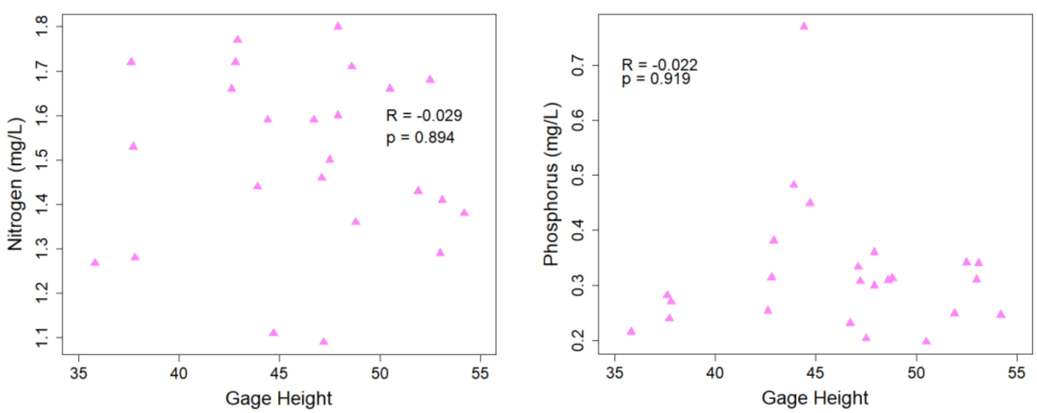

Figure 5.12: Summer concentrations of TN (left) and TP (right) plotted against lake elevation at West Bay.

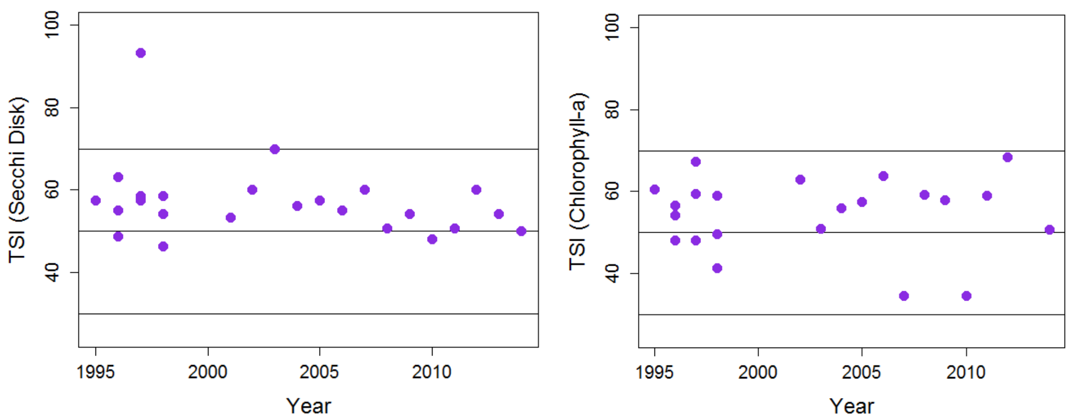

Figure 5.13: Summer TSI scores for West Bay from 1995 through 2014.

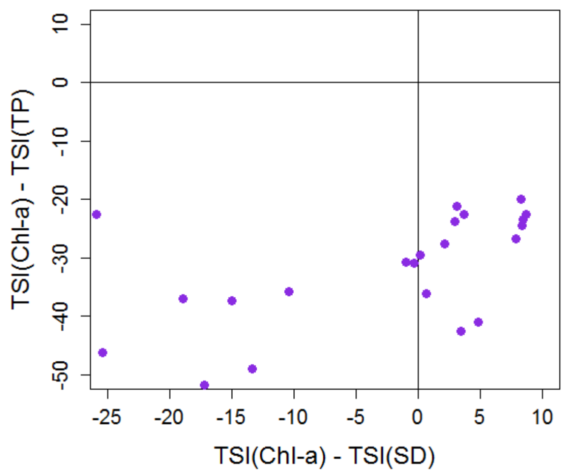

Figure 5.14: Nutrient-limitation graph for summer sampling at West Bay.

Annual fall sampling

West Bay did not thermally stratify in the fall, though it nearly stratified during 2014 sampling (Figure 5.15). Due to whole-column mixing, West Bay remained well-oxygenated, never being below 8 mg L^{-1} (Figure 5.15).

Over the study period, ionic concentrations were occasionally related to water level due to initial increases followed by a subsequent decrease with rising water levels, a pattern observed during other seasonal samples in West Bay. Sodium, a common cation, had no relationship with water level over the course of the project (R = 0.342), but decreased significantly following the opening of the west-end outlet (R = -0.933; Figure 5.16). Chloride and sulfate, common anions, have decreased strongly since the 2005 (R = -0.961 and -0.833, respectively), but was not related to water level since 1995 (R = 0.214 and 0.347, respectively; Figure 5.16). Specific conductance decreased significantly with increasing water levels since the west-end outlet opened (R =

-0.965), but was not related to water level over the course of the project (R = 0.346; Figure 5.16).

The effect of dilution on nutrient concentrations continued to be variable in West Bay in the fall. Concentrations of TN (R = -0.254; Figure 5.17), TKN (R = -0.209), and N+N (R = 0.181) were not related to increasing water levels. Similarly, TP (R = 0.191; Figure 5.17) and DP (R = 0.389) were not significantly related to rising water levels.

TSI scores based on chlorophyll-α and Secchi disk transparency classify West Bay as eutrophic in the fall. TSI scores based on Secchi disk transparency were variable over the study period, but decreased from 65.14 (0.7 m) in 1998 to 50.01 (2.0 m) in 2013 (Figure 5.18). Scores based on chlorophyll-α concentration were more variable, ranging from 66.70 in 2008 to 41.38 in 2007 (Figure 5.18).

West Bay is not P-limited in the fall, as evidenced by TSI scores based on TP being greater than scores based on chlorophyll-α concentration (Figure 5.19). Further, the photic zone in West Bay is typically dominated by small particles of non-algal turbidity (Figure 5.19), likely due to constant mixing causing a re-suspension of sediments.

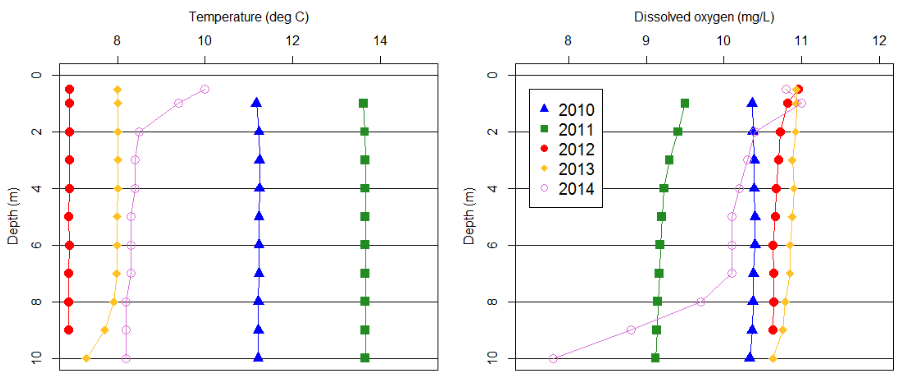

Figure 5.15: Fall temperature (left) and DO (right) profiles for West Bay from 2010 through 2014.

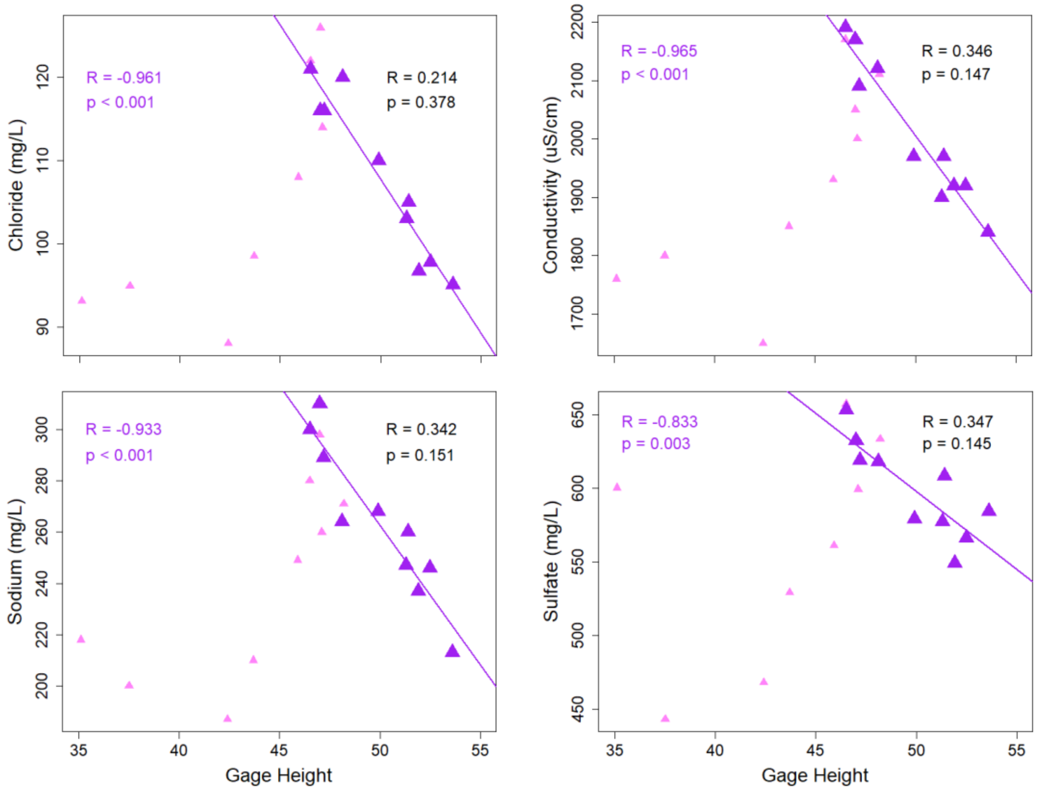

Figure 5.16: Fall concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at West Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5). (▲ is from 2005 through 2014; ▲ is from 1995 through 2004)

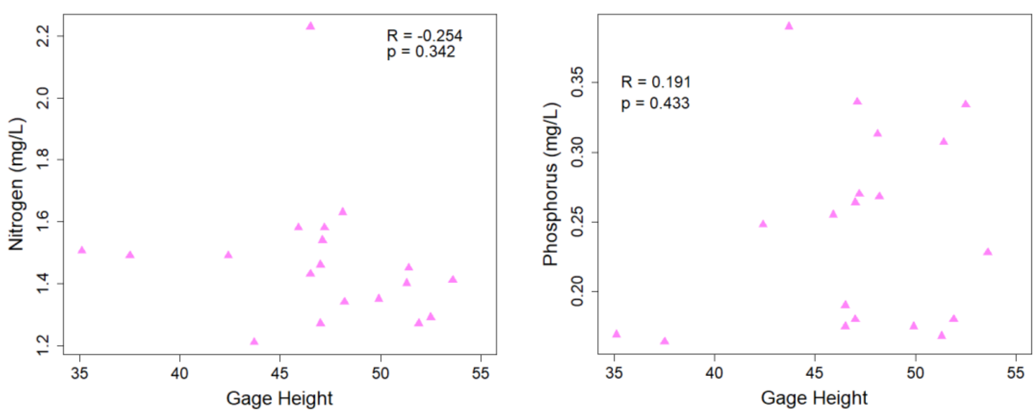

Figure 5.17: Fall concentrations of TN (left) and TP (right) plotted against lake elevation at West Bay. Values in the upper-right corner are Pearson correlation coefficients (R) and the p-value (significant at $p < 0.05$).

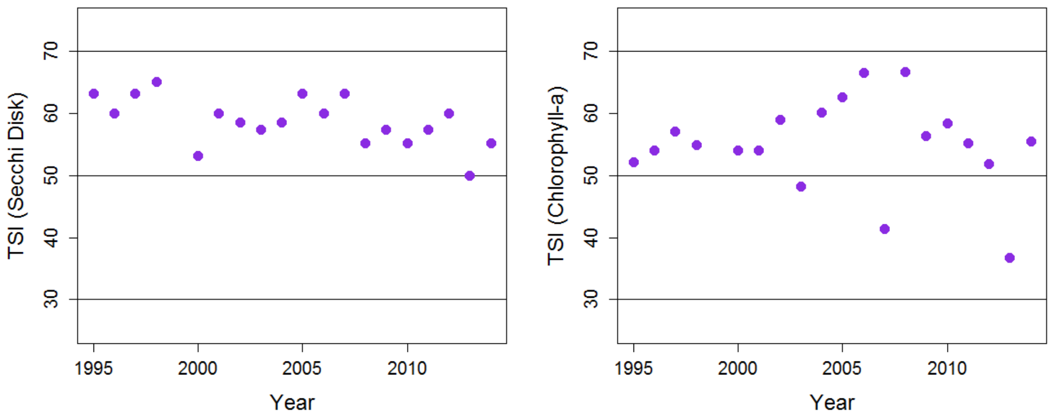

Figure 5.18: Fall TSI scores for West Bay from 1995 through 2014.

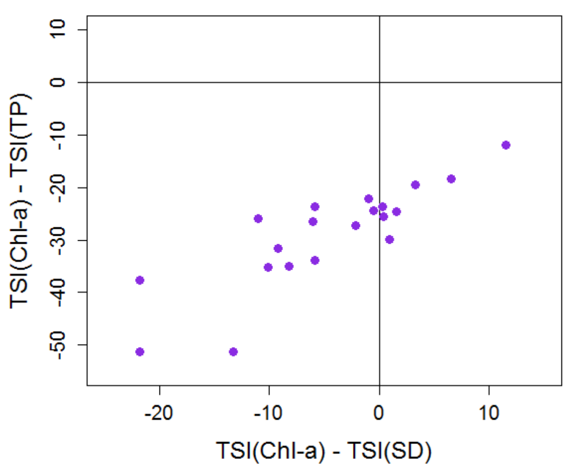

Figure 5.19: Nutrient-limitation graph for fall sampling at West Bay.

Phytoplankton assemblage sampling

Though dominated during summer by cyanobacteria, the West Bay of Devils Lake had a relatively high number of diatom species during most sampling months (dominated by *Nitzchia sp.* and *Stephanodiscus sp.*) (Table 5.1). Cyanobacteria were in highest concentrations during summer and fall sampling in 1998; chlorophytes (*Chlorella sp.* and *Chloricystis* minor) and diatoms, however, were also in relatively high densities during this timeframe. Chrysophytes were identified in low densities during sampling in West Bay (Table 5.1), but this is likely due to a lack of sampling during 1999, during which time Chrysophytes were in relatively high densities throughout the lake. Algal communities in West Bay were almost independent annually, with most years being similar within years, but not between years (Figure 5.20). Unlike other stations in Devils Lake, there did not appear to be a rotating of community composition (i.e., 1996 did not associate to 1998 samples). This lack of similarity could be due to the fact that no samples were collected during 1999 and only fall samples collected in 2000.

Table 5.1: Presence-absence table showing common phytoplankton species and when they were detected in West Bay. An "X" signifies that the species was present during that particular season.

	1996			1997			1998			1999			2000		
	Sp	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**
Chlorophyta															
Ankyra judai		X					X	X		n/a	n/a	n/a	n/a	n/a	
Chlorella minutissima							X	X	X	n/a	n/a	n/a	n/a	n/a	X
Chlorella vulgaris	X	X					X	X	X	n/a	n/a	n/a	n/a	n/a	
Choricystis minor	X	X	X				X	X	X	n/a	n/a	n/a	n/a	n/a	X
Eremosphaera eremosphaeria		X	X							n/a	n/a	n/a	n/a	n/a	
Monoraphidium minutum				X	X	X				n/a	n/a	n/a	n/a	n/a	
Oocystis novae-semliae		X						X		n/a	n/a	n/a	n/a	n/a	X
Oocystis submarina	X	X						X	X	n/a	n/a	n/a	n/a	n/a	X
Planktosphaeria gelatinosa							X	X	X	n/a	n/a	n/a	n/a	n/a	
Raphidonema sp.				X	X					n/a	n/a	n/a	n/a	n/a	
Schroederia setigera		X						X		n/a	n/a	n/a	n/a	n/a	
Chrysophyta															
Ochromonas sp.								X	X	n/a	n/a	n/a	n/a	n/a	X
Cryptophyta															
Campylomonas reflexa							X	X	X	n/a	n/a	n/a	n/a	n/a	X
Chroomonas acuta	X	X	X	X	X	X	X	X	X	n/a	n/a	n/a	n/a	n/a	
Cryptomonas erosa				X	X					n/a	n/a	n/a	n/a	n/a	
Cryptomonas reflexa	X	X	X	X						n/a	n/a	n/a	n/a	n/a	
Cryptomonas rostratiformis	X	X								n/a	n/a	n/a	n/a	n/a	
Cryptomonas sp.		X								n/a	n/a	n/a	n/a	n/a	
Rhodomonas minuta	X	X		X						n/a	n/a	n/a	n/a	n/a	
Rhodomonas minuta var. nan.				X	X	X				n/a	n/a	n/a	n/a	n/a	
Rhodomonas pusilla					X	X				n/a	n/a	n/a	n/a	n/a	
Cyanobacteria															
Anabaena curva		X	X					X	X	n/a	n/a	n/a	n/a	n/a	
Anabaena flosaquae		X						X	X	n/a	n/a	n/a	n/a	n/a	
Aphanizomenon flosaquae		X			X	X		X	X	n/a	n/a	n/a	n/a	n/a	X
Aphanocapsa delicatissima	X		X					X		n/a	n/a	n/a	n/a	n/a	
Aphanocapsa incerta								X		n/a	n/a	n/a	n/a	n/a	X
Aphanothece nidulans var. nid.	X	X	X							n/a	n/a	n/a	n/a	n/a	
Aphanothece smithii								X	X	n/a	n/a	n/a	n/a	n/a	
Gloeocapsa aeruginosa					X	X				n/a	n/a	n/a	n/a	n/a	
Microcystis aeruginosa	X	X	X			X		X	X	n/a	n/a	n/a	n/a	n/a	X
Myxobaktron sp.	X						X	X	X	n/a	n/a	n/a	n/a	n/a	
Oscillatoria angustissima				X	X	X				n/a	n/a	n/a	n/a	n/a	
Oscillatoria utermoehl								X	X	n/a	n/a	n/a	n/a	n/a	X
Pseudanabaena mucicola		X						X		n/a	n/a	n/a	n/a	n/a	
Woronichinia compacta								X	X	n/a	n/a	n/a	n/a	n/a	X
Woronichinia naegeliana								X	X	n/a	n/a	n/a	n/a	n/a	X

Table 5.1: (cont.)

	1996			1997			1998			1999			2000		
	Sp	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**
Heterokontophyta															
Asterionella formosa	X	X	X					X	X	n/a	n/a	n/a	n/a	n/a	
Aulacoseira granulate var. ang.		X	X	X		X			X	n/a	n/a	n/a	n/a	n/a	
Aulacoseira granulate var. gra.		X	X					X	X	n/a	n/a	n/a	n/a	n/a	
Cyclostephanos sp.				X	X	X				n/a	n/a	n/a	n/a	n/a	
Cyclotella bodanica				X	X	X				n/a	n/a	n/a	n/a	n/a	
Cyclotella meneghiniana		X		X				X	X	n/a	n/a	n/a	n/a	n/a	
Nitzschia fonticola		X						X	X	n/a	n/a	n/a	n/a	n/a	
Nitzschia frustulum					X	X				n/a	n/a	n/a	n/a	n/a	
Nitzschia inconspicua					X	X				n/a	n/a	n/a	n/a	n/a	
Nitzschia palea			X					X		n/a	n/a	n/a	n/a	n/a	
Stephanodiscus hantzschii				X	X	X				n/a	n/a	n/a	n/a	n/a	
Stephanodiscus minutus	X	X	X						X	n/a	n/a	n/a	n/a	n/a	
Stephanodiscus niagarae		X	X	X	X			X	X	n/a	n/a	n/a	n/a	n/a	X
Stephanodiscus rotula				X	X					n/a	n/a	n/a	n/a	n/a	

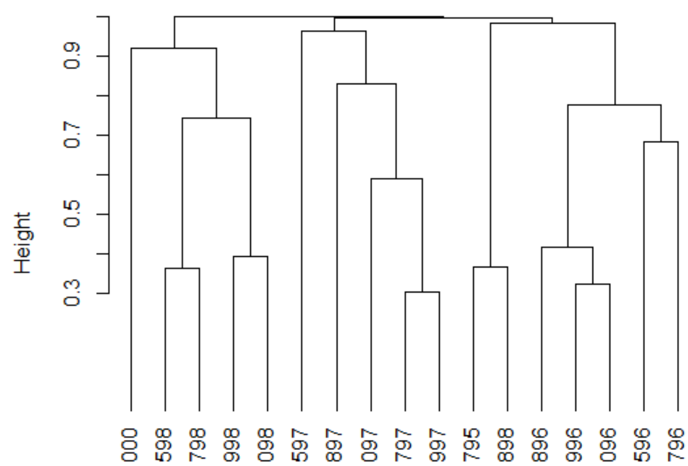

Figure 5.20: Dendrogram clustering sites based on presence and proportion of algal species for each sample at West Bay. Distances were determined using Bray-Curtis dissimilarity scores. Numbers along the x-axis represent the date of the sample (i.e., 797 is July 1997; 896 is August 1996).

3.6 Southwest West Bay

The sampling station at Southwest West Bay (Station ID 384160), active since 1997, is southeast of the City of Minnewaukan (Figure 6.1). This station is slightly protected from westerly winds, but has long fetches with easterly winds and is relatively shallow, which makes thermal stratification difficult. Water quality data from this station represents water going through the west-end outlet, which opened in April 2005.

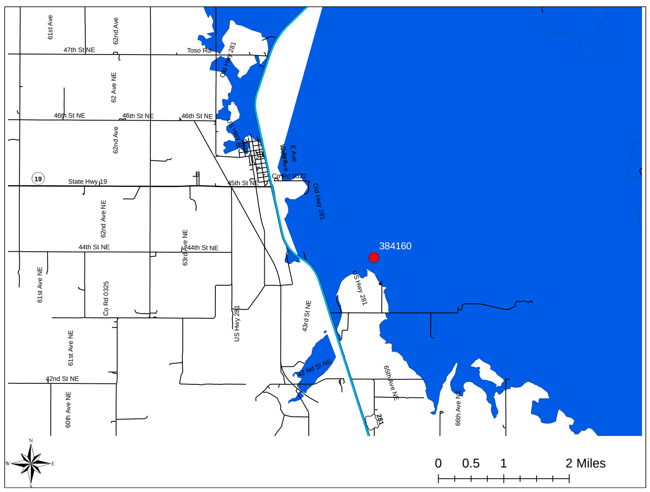

Figure 6.1: Map showing location of sampling site in the Southwest portion of West Bay.

Annual winter sampling

The sampling station at Southwest West Bay thermally-stratified one time during winter sampling (2013). Southwest West Bay has among the coldest average near-sediment temperatures in Devils Lake, likely due to being shallow. DO concentrations remained relatively high under ice-cover, never below 10 mg/L since 2010 (Figure 6.2).

Many analytes followed a similar pattern as the station at West Bay, sharply decreasing after the opening of the west-end outlet. Sodium, a common cation, had no relationship with water level through the project period ($R = -0.148$), but decreased significantly following the opening of the west-end outlet ($R = -0.673$; Figure 6.3). Chloride and sulfate, common anions, decreased strongly after April 2005 ($R = -0.968$ and -0.830, respectively), but only the former was related to rising water levels over the course of the project ($R = -0.539$; Figure 6.3). Specific conductance decreased significantly since the west-end outlet opened ($R = -0.945$), but was not related to water levels over the course of the project ($R = 0.464$; Figure 6.3).

Nutrient concentrations in Southwest West Bay were not significantly related to increasing water levels in the basin. TN ($R = -0.265$; Figure 6.4), TKN ($R = -0.304$), and N+N ($R = 0.036$) were not affected by increasing water volume. Also, TP (R = 0.214; Figure 6.4) and DP (R = 0.029) were not impacted by the increase in lake elevation.

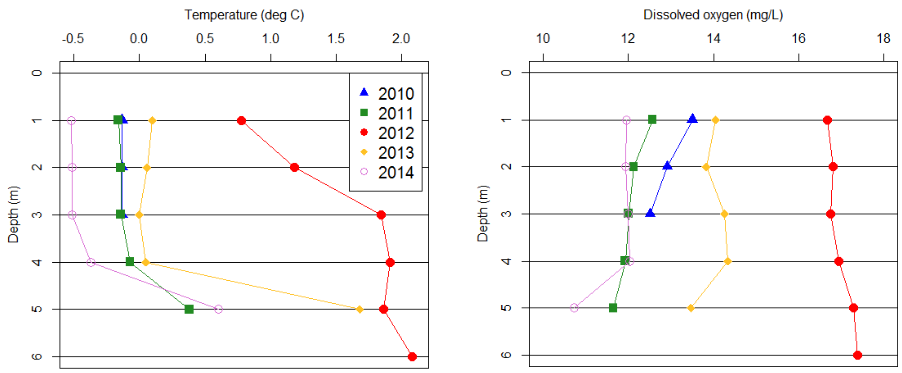

Figure 6.2: Winter temperature (left) and DO (right) profiles for Southwest West Bay from 2010 through 2014.

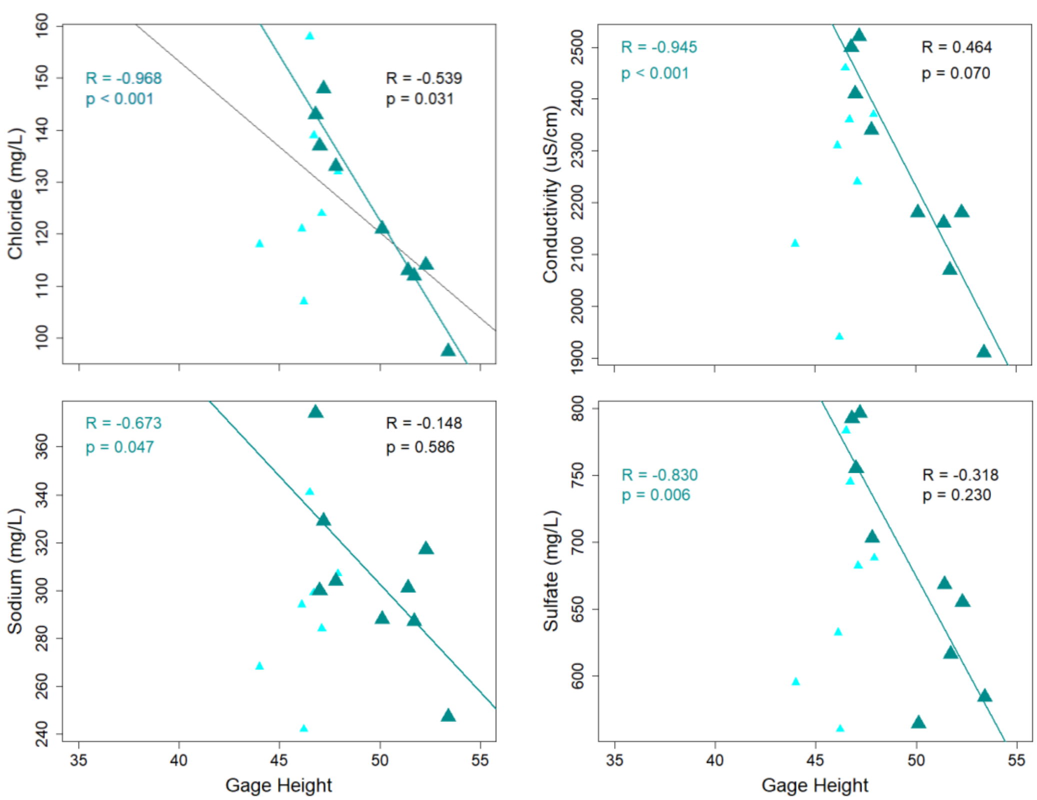

Figure 6.3: Winter concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Southwest West Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5). (▲ is from 2005 through 2014; ▴ is from 1997 through 2004)

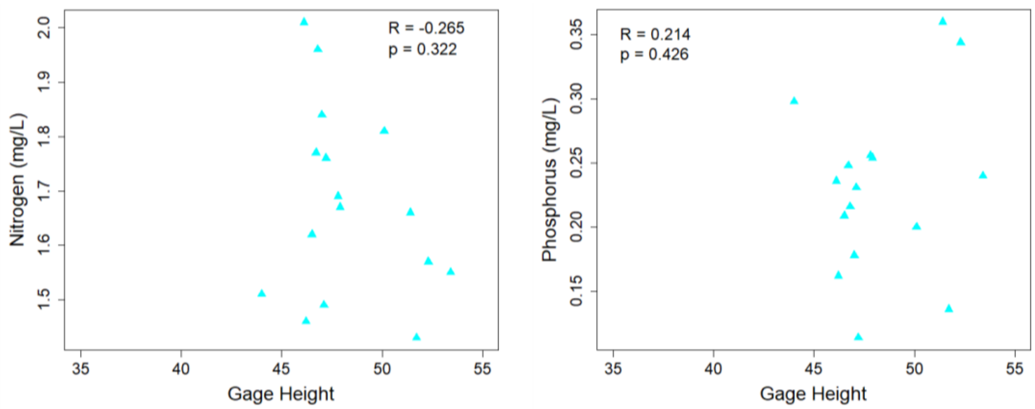

Figure 6.4: Winter concentrations of TN (left) and TP (right) plotted against lake elevation at Southwest West Bay.

Annual spring sampling

Similar to most Devils Lake stations in the spring, Southwest West Bay rarely stratifies, occurring once since 2010 (2014) (Figure 6.5). Due to mixing during the spring months, this station has stayed well-oxygenated, never below 5 mg L^{-1} since 2010 (Figure 6.5).

Most analytes in Southwest West Bay followed a similar pattern to the rest of the seasons at West Bay. Sodium, a common cation, decreased significantly with increasing water volume following the opening of the west-end outlet (R = -0.924), but was not significantly related over the course of the project (R = -0.227; Figure 6.6). Chloride and sulfate, common anions, decreased significantly since April 2005 (R = -0.936 and -0.939, respectively), but were not related to water levels over the entire project (R = -0.346 and -0.154, respectively; Figure 6.6). Specific conductance was not related to water levels over the entire project period (R = -0.132), but decreased significantly following the opening of the outlet (R = -0.965; Figure 6.6).

Concentrations of most nutrients were variable over the study period in Southwest West Bay. There was no relationship between water level and total nitrogen (R = -0.265; Figure 6.7), while TKN significantly decreased with lake elevation (R = -0.559). Additionally, TP (R = -0.214; Figure 6.7) and DP (R = 0.115) were not significantly related to increasing water volume.

Over the study period, TSI scores based on Secchi disk transparency and chlorophyll-α decreased slightly, which classified the lake between mesotrophic and eutrophic. TSI scores based on transparency decreased from 61.52 (0.9 m) in 2000 to 35.17 (5.6 m) in 2014 (Figure 6.8). TSI scores based on chlorophyll-α were a bit more variable over the study period, ranging from 53.19 in 2013 to 41.38 in 1998, 1999, 2008, 2010, and 2014 (Figure 6.8).

Analyzing TSI scores suggests that Southwest West Bay is not P-limited in the spring, based on TSI scores for phosphorus being much greater than chlorophyll-α TSI scores (Figure 6.9). Long fetches open to easterly winds appear to be influencing the photic zone in Southwest West Bay, as it is commonly dominated by smaller particles of algal and non-algal turbidity (Figure 6.9).

Over the past few years, however, this station has increasingly been dominated by larger particles of algae, suggesting that zooplankton grazing is becoming more efficient.

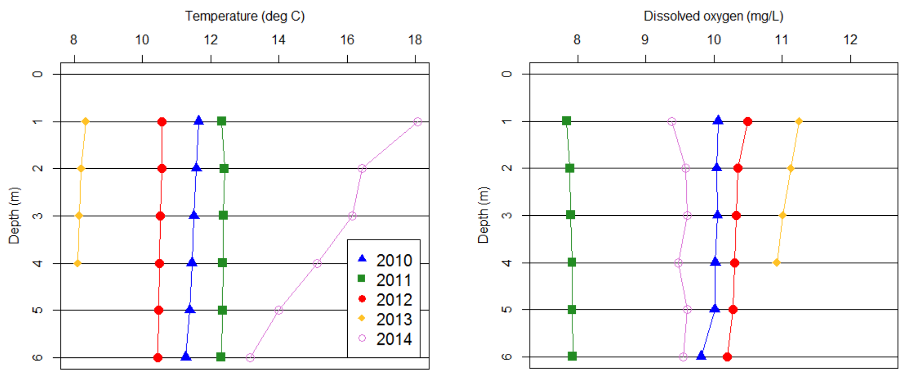

Figure 6.5: Spring temperature (left) and dissolved oxygen (right) profiles for Southwest West Bay from 2010 through 2014.

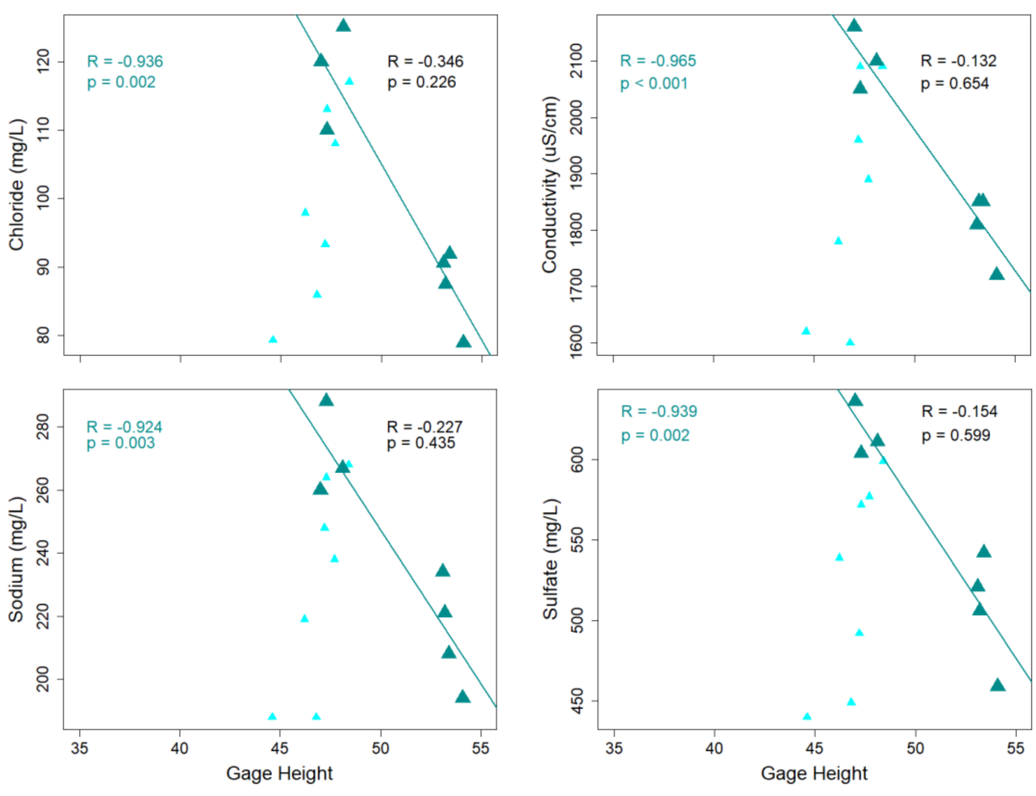

Figure 6.6: Spring concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Southwest West Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5). (▲ is from 2005 through 2014; ▴ is from 1997 through 2004)

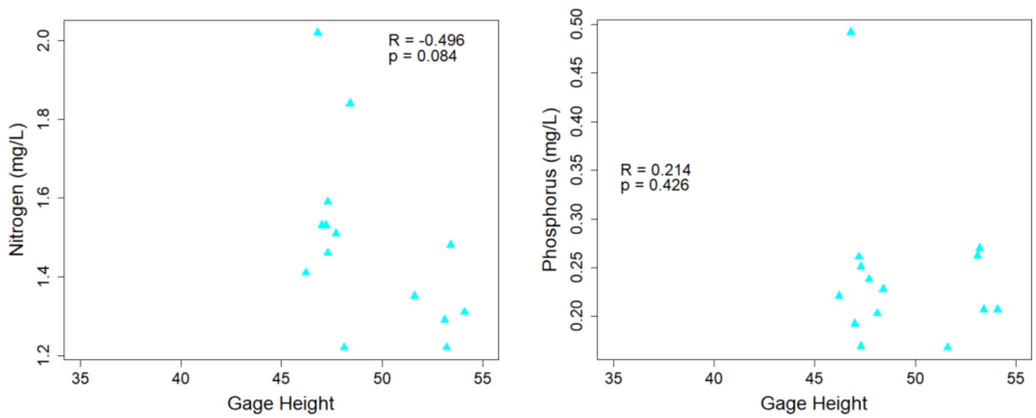

Figure 6.7: Spring concentrations of TN (left) and TP (right) plotted against lake elevation at Southwest West Bay.

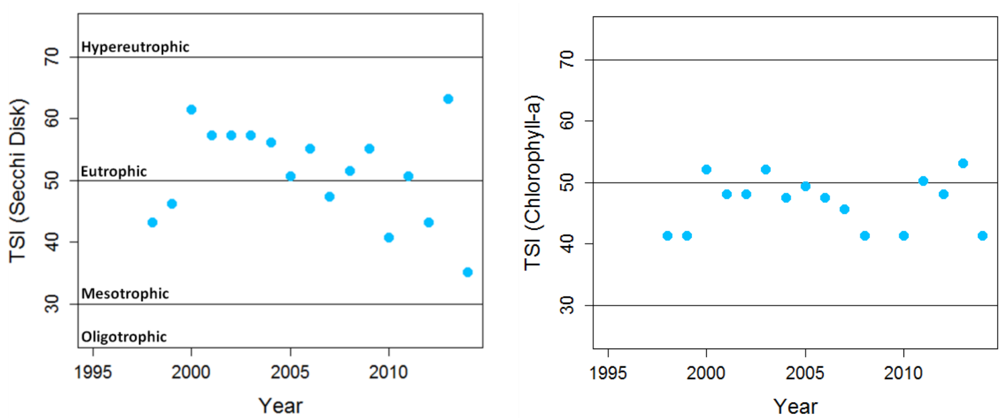

Figure 6.8: Spring TSI scores for Southwest West Bay from 1998 through 2014.

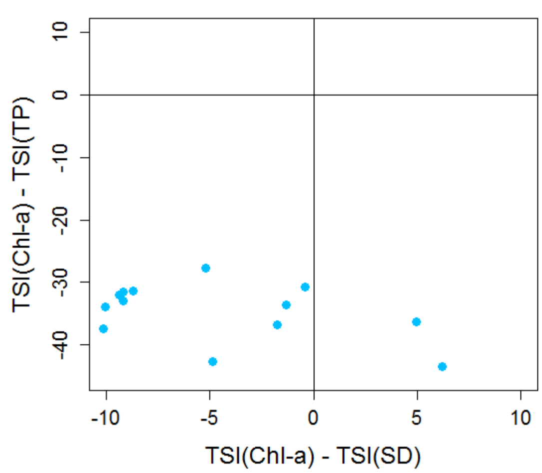

Figure 6.9: Nutrient-limitation graph for spring sampling at Southwest West Bay.

Annual summer sampling

Though it is vulnerable to long fetches from easterly winds, Southwest West Bay does thermally stratify during summer sampling, occurring twice since 2010 (2012, 2013) (Figure 6.10). Despite the thermal stratification, this station remains well-oxygenated to the bottom, only once being below 5 mg L^{-1} (2012) (Figure 6.10).

Analytes measured during summer sampling appeared to follow similar trends to other seasonal monitoring. Sodium, a common cation, showed a nearly significant increase over the course of the project (R = 0.407), but decreased significantly following the opening of the west-end outlet (R = -0.956; Figure 6.11). Chloride and sulfate, common anions, decreased significantly since 2005 (R = -0.896 and -0.918, respectively), but only the latter had a relationship to increased water level over the project period (R = 0.465; Figure 6.11). Specific conductance had no relationship over the project period (R = 0.375), but decreased significantly since 2005 (R = -0.951; Figure 6.11).

Nutrient concentrations measured in Southwest West Bay were not significantly related to increases in water level. TN (R = -0.117; Figure 6.12) and TKN (R = -0.231) were unaffected by increases in water volume. Similarly, DP was not significantly affected by lake elevation (R = 0.124), although the relationship between water level and TP was nearly significant (R = -0.328) (Figure 6.12).

During the summer, Southwest West Bay was strongly eutrophic. Over the study period, there was a slight decrease in TSI score based on Secchi disk transparency, going from 67.36 (0.6 m) in 1997, 1999, and 2000 to 52.35 (1.7 m) in 2011, with a peak of 77.35 (0.3 m) in 2003 (Figure 6.13). TSI scores based on chlorophyll-α concentration were much more variable, ranging from 71.41 in 2007 to 41.38 in 1999 (Figure 6.13).

In the summer, Southwest West Bay was not P-limited, with TSI scores based on TP greatly exceeding those for chlorophyll-α (Figure 6.14). From 1998 through 2005, TSI scores suggest that Southwest West Bay was dominated by smaller particles of algal and non-algal turbidity, demonstrated by TSI scores for Secchi disk transparency being higher than those for chlorophyll-α (Figure 6.14). Since 2005, however, this station switched to being dominated by larger particles of algae (Figure 6.14), likely due to more efficient zooplankton grazing.

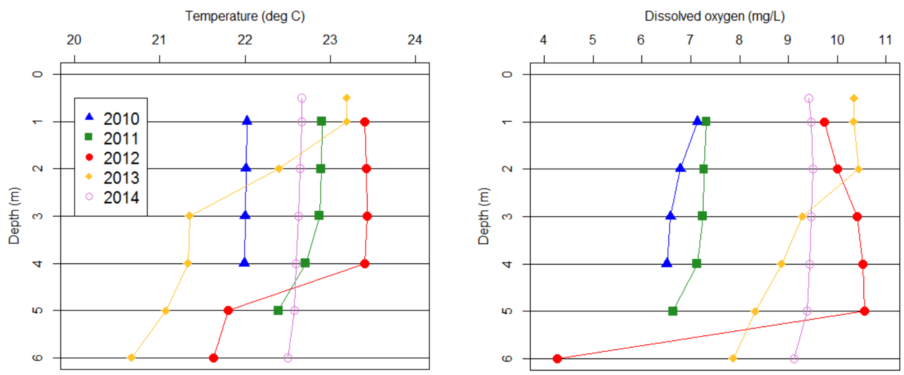

Figure 6.10: Summer temperature (left) and DO (right) profiles for Southwest West Bay from 2010 through 2014.

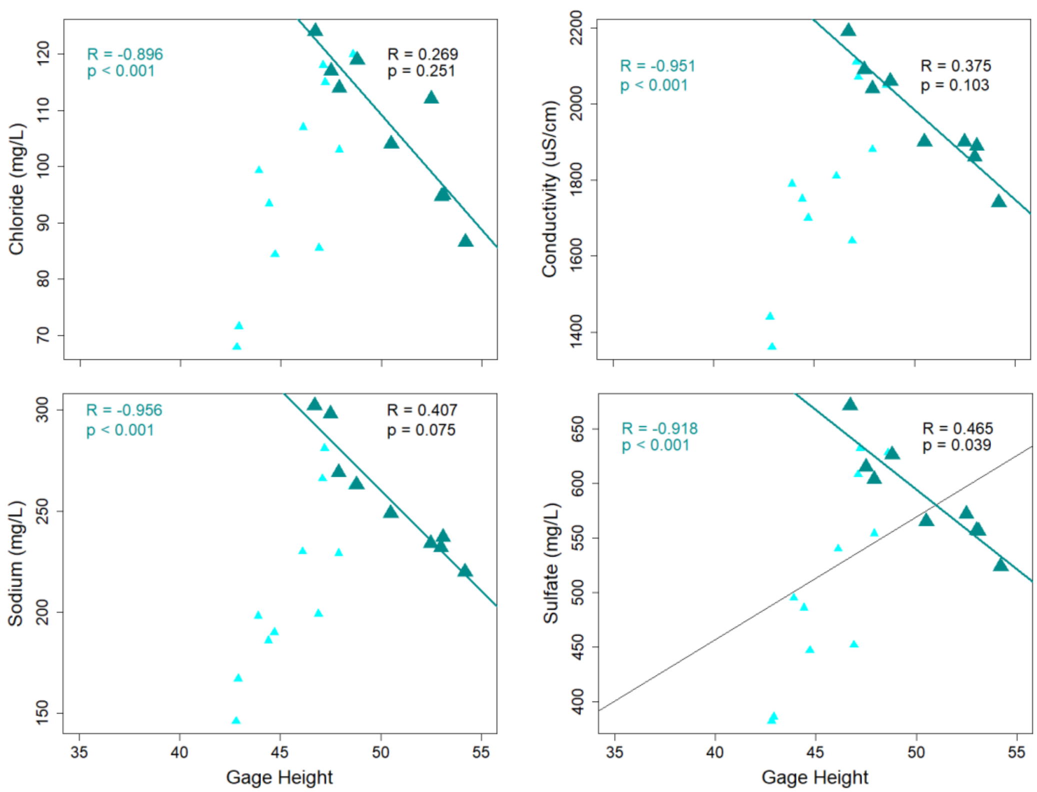

Figure 6.11: Summer concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Southwest West Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5). (▲ is from 2005 through 2014; ▴ is from 1997 through 2004)

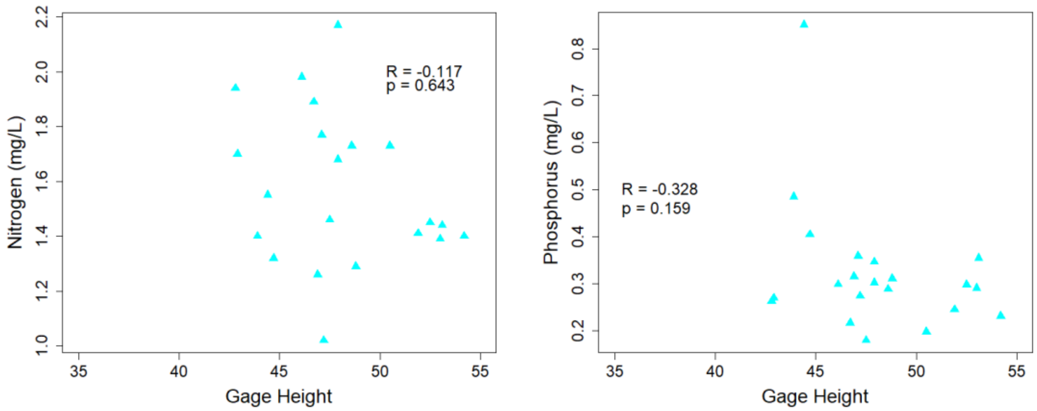

Figure 6.12: Summer concentrations of TN (left) and TP (right) plotted against lake elevation at Southwest West Bay.

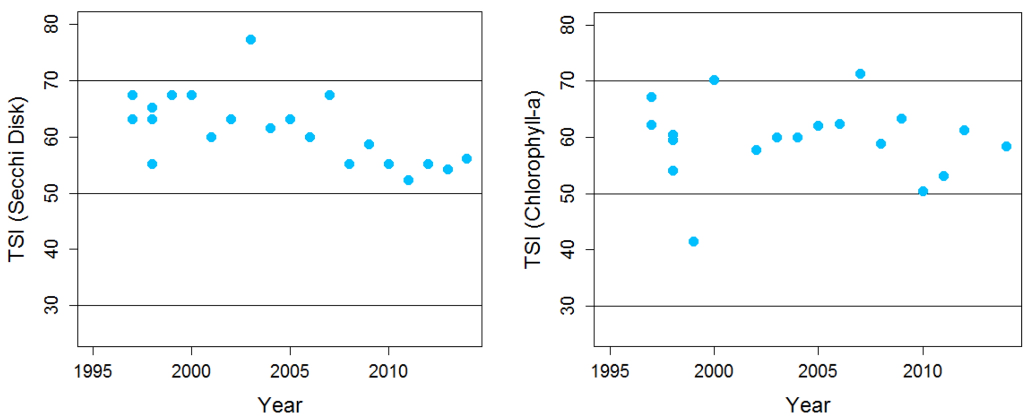

Figure 6.13: Summer TSI scores for Southwest West Bay from 1997 through 2014.

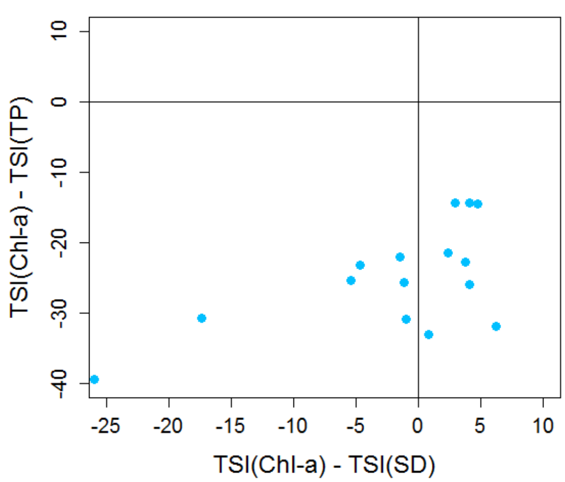

Figure 6.14: Nutrient-limitation graph for summer sampling at Southwest West Bay.

Annual fall sampling

The station at Southwest West Bay did not thermally stratify during fall sampling (Figure 6.15). Due to whole-column mixing at this station, Southwest West Bay remained well-oxygenated to the bottom, never below 9 mg L^{-1} (Figure 6.15).

Over the study period, few measured ions had a statistically significant relationship with increasing water levels in Southwest West Bay, which is likely due to the hump-shaped relationship observed for many analytes. Sodium, a common cation, was not related to increased water levels over the project period (R = 0.201), but decreased significantly since the opening of the west-end outlet in 2005 (R = -0.694; Figure 6.16). Chloride and sulfate, common anions, decreased significantly since 2005 (R = -0.953 and -0.858, respectively), but were not related over the project period (R = -0.138 and 0.220, respectively; Figure 6.16). Specific conductance decreased strongly since the opening of the outlet (R = -0.972), but was not related over the entire project period (R = 0.102; Figure 6.16).

Over the study period, nutrient concentrations were not related to changes in lake elevation. TN (R = -0.137; Figure 6.17), TKN (R = -0.156), and N+N (R = 0.060) did not relate to increases in water level. Also, TP was not significantly related to increases in water level (R = 0.062) (Figure 6.17).

TSI scores based on chlorophyll-α concentration and Secchi disk transparency classify Southwest West Bay as eutrophic in the fall. TSI scores based on Secchi disk transparency declined steadily over the study period, decreasing from 73.20 (0.4 m) in 1998 to 53.23 (1.6 m) in 2013 (Figure 6.18). TSI scores based on chlorophyll-α concentration were much more variable over the study period, ranging from 72.05 in 2008 to 35.21 in 2013 (Figure 6.18).

Analysis of TSI scores suggests that Southwest West Bay is not P-limited in the fall. This is evidenced by TSI scores based on TP being much greater than those based on chlorophyll-α concentration (Figure 6.19). Further, Southwest West Bay appears to be dominated by smaller particles of non-algal turbidity in the fall (Figure 6.19). This is likely due to the station's relatively shallow depth, and therefore sediment is easily re-suspended.

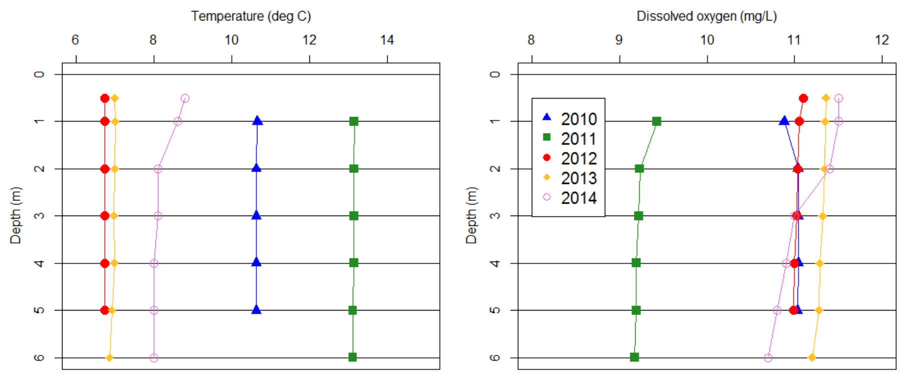

Figure 6.15: Fall temperature (left) and DO (right) profiles for Southwest West Bay from 2010 through 2014.

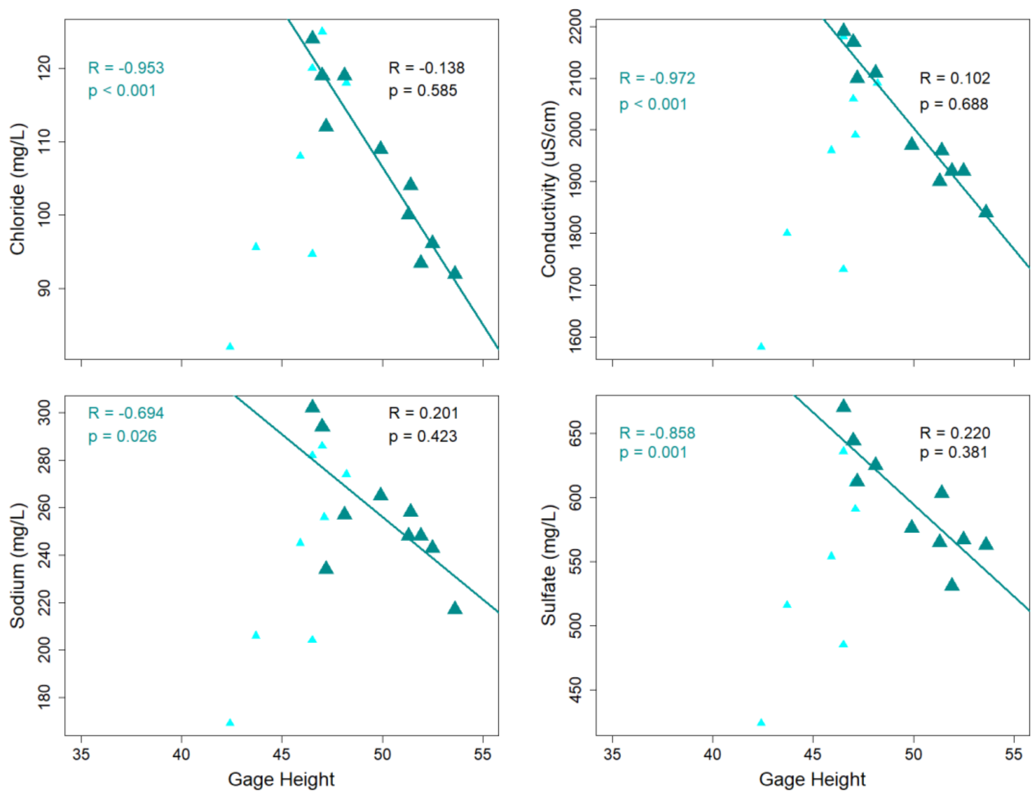

Figure 6.16: Fall concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Southwest West Bay. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5). (▲ is from 2005 through 2014; ▲ is from 1997 through 2004)

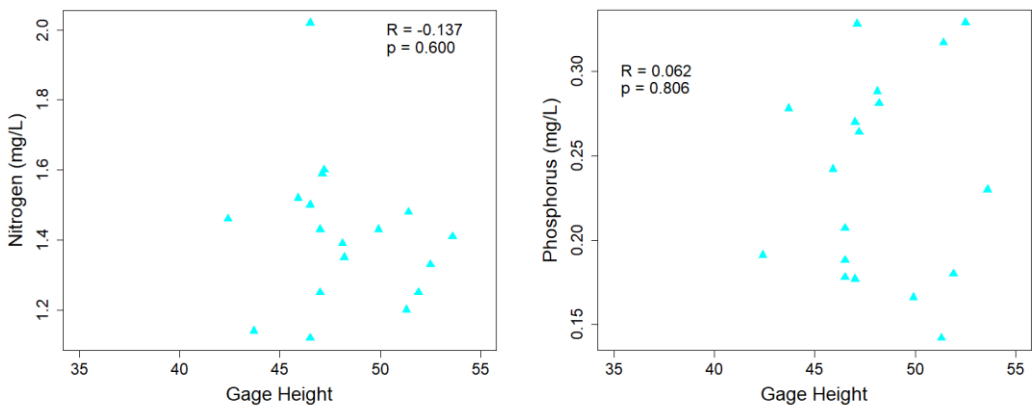

Figure 6.17: Fall concentrations of TN (left) and TP (right) plotted against lake elevation at Southwest West Bay.

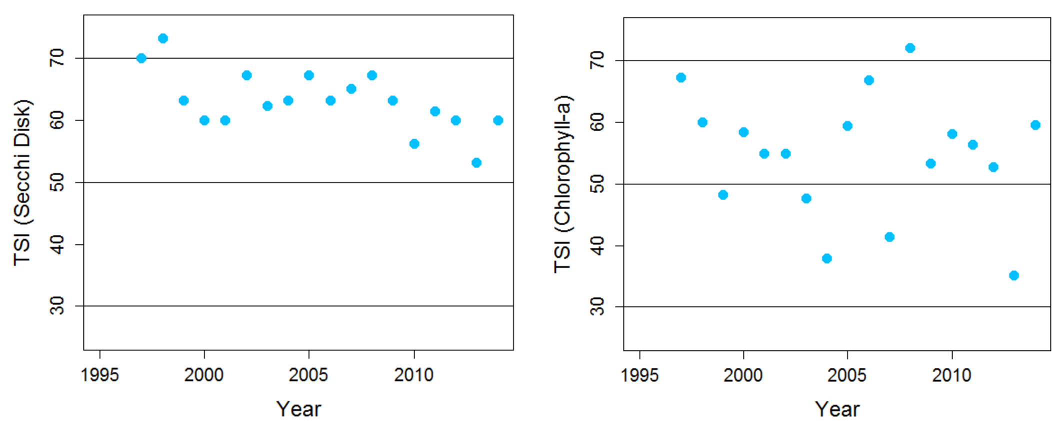

Figure 6.18: Fall TSI scores for Southwest West Bay from 1997 through 2014.

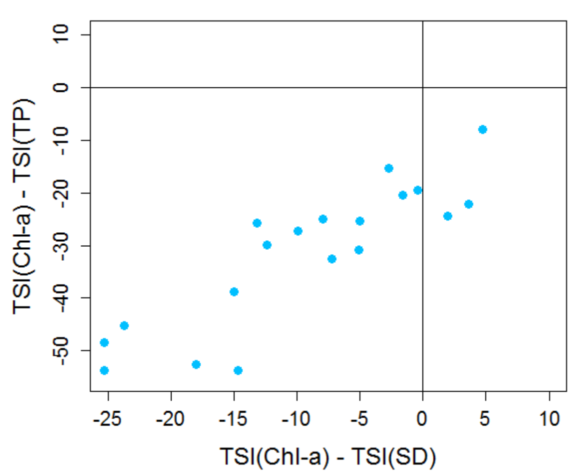

Figure 6.19: Nutrient-limitation graph for spring sampling at Southwest West Bay.

Phytoplankton assemblage sampling

Phytoplankton sampling in Southwest West Bay began during summer 1997 and continued through 2000. Assemblage patterns in Southwest West Bay were very similar to that of West Bay, and could be used to fill-in knowledge gaps in 1999 and 2000 for West Bay. Other than spring samples, the blue-green community was dominated annually by *Aphanizomenon flos-aquae* and *Microcystis aeruginosa*, while all other species of cyanobacteria rotated annually (Table XX). Similarly to the whole basin, golden algae were in highest densities during 1999 samples, but were rare during other sampling periods. Species composition remained in an every-other-year pattern, with 1997 associating closely with 1999, and 1998 associating closely to 2000 (Figure XX).

Table 6.1: Presence-absence table showing common phytoplankton species and when they were detected in Southwest West Bay. An "X" signifies that the species was present during that particular season.

	1997		**1998**			**1999**			**2000**		
	Su	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**
Chlorophyta											
Ankyra judai			X	X							X
Chlorella minutissima			X	X	X				X	X	X
Chlorella vulgaris			X	X	X						
Choricystis minor			X	X	X				X		X
Closterium acutum var. var.				X	X					X	X
Dictyosphaerium ehrenbergianum				X	X						X
Kirchneriella sp.	X	X				X					
Monoraphidium contortum		X					X	X			
Monoraphidium minutum	X	X				X	X	X			
Monoraphidium pusillum	X					X	X	X			
Oocystis parva				X			X	X			
Oocystis submarina				X	X						
Pediastrum boryanum				X					X		X
Planktosphaeria gelatinosa			X	X	X						
Raphidocelis microscopica				X	X						X
Raphidonema sp.						X	X	X			
Schroederia setigera			X	X	X						
Chrysophyta											
Chrysococcus cordiformis						X	X	X			
Chrysococcus cordiformis var. ast.						X	X	X			
Chrysococcus minutus						X	X	X			
Chrysococcus punctiformis	X	X				X	X	X			
Chrysococcus rufescens						X	X	X			
Chrysococcus rufescens var. tri.						X	X	X			
Kephyrion littorale						X	X	X			
Kephyrion rubri-claustri						X	X	X			
Kephyrion sitta						X	X	X			
Cryptophyta											
Campylomonas reflexa			X	X	X						
Chroomonas acuta	X		X	X	X			X			
Cryptomonas erosa	X	X									
Plagioselmis sp.									X	X	X
Rhodomonas minuta var. nan.	X	X				X	X	X			

Table 6.1: (cont.)

	1997		1998			1999			2000		
	Su	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**	**Sp**	**Su**	**Fa**
Cyanobacteria											
Anabaena curva				X	X						
Anabaena flosaquae				X	X					X	
Aphanizomenon flosaquae	X			X	X		X	X	X	X	X
Aphanothece smithii				X	X				X	X	X
Gloeocapsa aeruginosa	X	X				X	X	X			
Gomphosphaeria lacustris var com.	X	X				X	X				
Merismopedia tenuissima				X	X		X	X			
Microcystis aeruginosa		X	X	X	X		X	X		X	X
Myxobaktron sp.				X	X						
Oscillatoria angustissima	X					X	X	X			
Oscillatoria utermoehl			X	X	X					X	X
Pseudanabaena mucicola				X							X
Woronichinia compacta				X	X						
Woronichinia naegeliana			X	X	X					X	X
Heterokontophyta											
Asterionella formosa				X	X				X		
Cyclostephanos sp.	X	X				X	X	X			
Cyclotella chotawhatcheeana				X	X						X
Cyclotella glomerata	X					X	X	X			
Cyclotella meneghiniana				X	X	X		X			X
Navicula sp.				X	X						
Nitzschia fonticola				X	X						
Nitzschia frustulum	X	X					X				
Nitzschia inconspicua	X						X	X			
Nitzschia palea					X						
Stephanodiscus hantzschii	X	X					X	X			
Stephanodiscus niagarae	X	X		X	X		X	X	X		X
Stephanodiscus rotula		X					X	X			

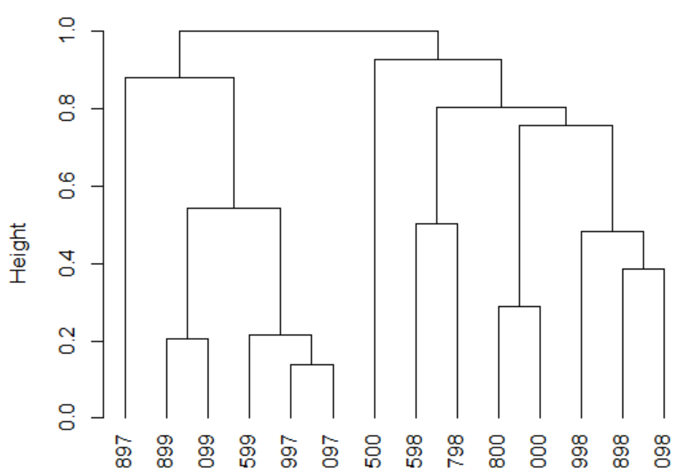

Figure 6.20: Dendrogram clustering sites based on presence and proportion of algal species for each sample at Southwest West Bay. Distances were determined using Bray-Curtis dissimilarity scores. Numbers along the x-axis represent the date of the sample (i.e., 797 is July 1997; 896 is August 1996).

3.7 Pelican Lake

The sampling station at Pelican Lake (Station ID 385029), active since 1999, is 15 miles west of the City of Devils Lake (Figure 7.1). This station has long fetches with easterly and westerly winds, making thermal stratification less likely to occur. Its relatively shallow depth (approximately 7 meters [23 feet]) may also contribute to its difficulty to stratify.

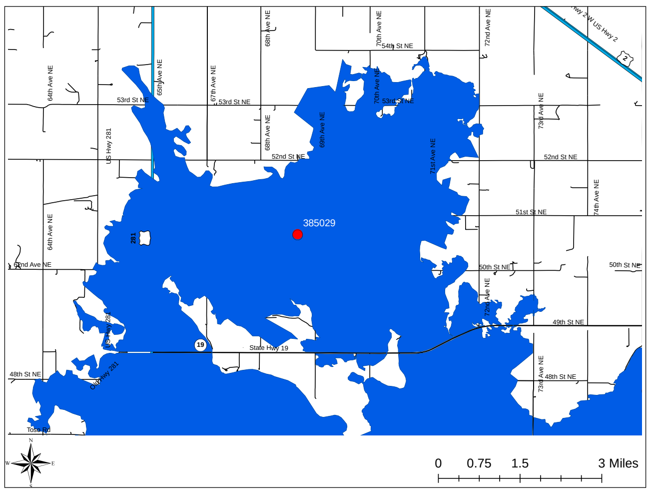

Figure 7.1: Map showing location of sampling site in Pelican Lake.

Annual winter sampling

There is some thermal stratification in Pelican Lake during the winter months, with the basin stratifying twice since 2010 (2010, 2011) (Figure 7.2). Despite this stratification, Pelican Lake remains relatively well-oxygenated from top-to-bottom, never being below the State's standard (Figure 7.2).

Analytes measured in Pelican Lake in the winter had no statistical significance with increases in water level. Decreases in barium (R = -0.464), manganese (R = -0.496), and zinc (R = -0.410) were nearly significant, while increases with pH (R = 0.446) and carbonate (R = 0.422) were also nearly significant. Over the study period, pH increased from a low of 7.29 (2001) to a high of 8.48 (2010, 2014), likely a result of mixing with other relatively high-pH water from the rest of the lake. Additionally, specific conductance in Pelican Lake used to be the lowest in the Chain (1,120 µS/cm, 2001), but has since increased with water-mixing with the rest of Devils Lake (1,890 µS/cm, 2014) (Figure 7.3). Further analysis of Pelican Lake data suggests that trends occurring in the basin are similar to those in West Bay and Southwest West Bay.

Before the influx of water from eastern Devils Lake, winter nutrient concentrations in Pelican Lake were relatively high compared to the rest of the basin. TN (R = -0.725; Figure 7.4) and TKN (R = -0.535) decreased significantly with increasing water levels, while ammonia (R = -

0.075) and N+N (R = -0.389) were unaffected by rising water. TP (R = 0.101; Figure 7.4) and DP (R = 0.195) were not related to increases in lake elevation, and maintained extreme inter-annual variability throughout the study period.

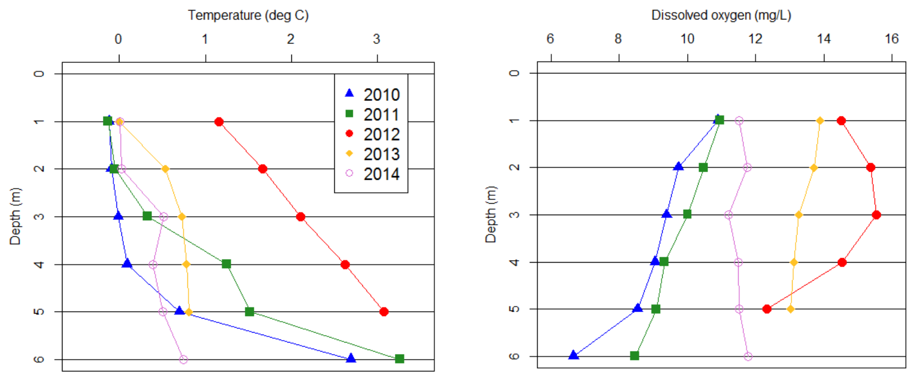

Figure 7.2: Winter temperature (left) and DO (right) profiles for Pelican Lake from 2010 through 2014.

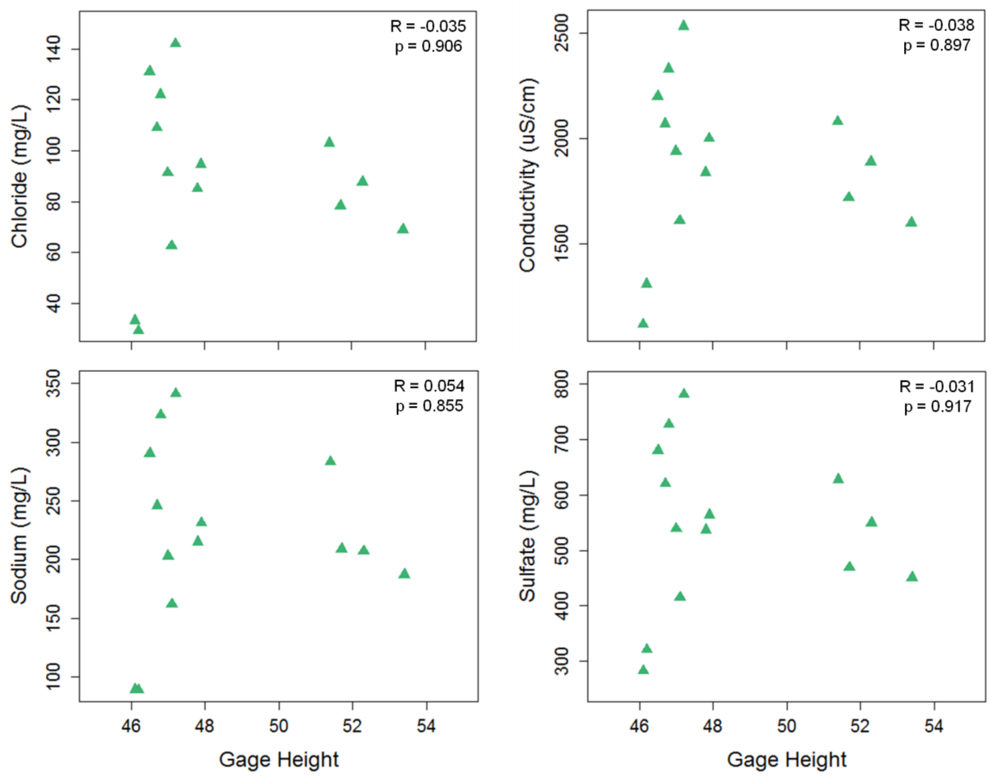

Figure 7.3: Winter concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Pelican Lake. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

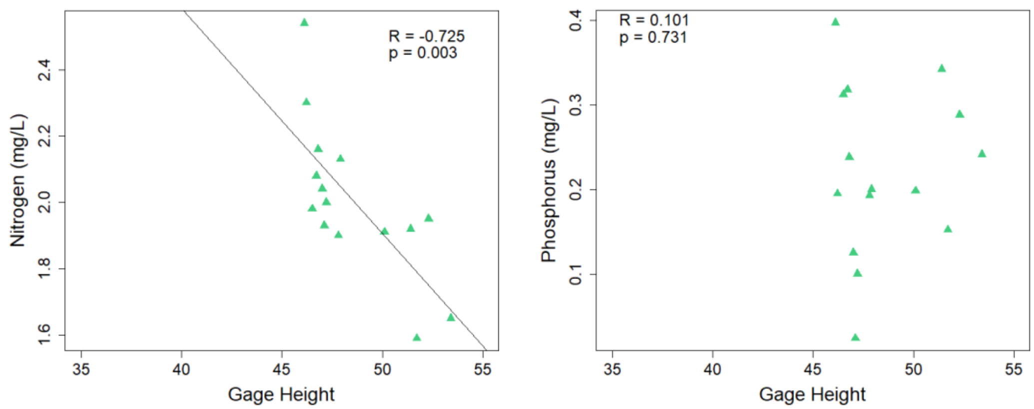

Figure 7.4: Concentrations of TN (left) and TP (right) plotted against lake elevation at Pelican Lake. Values in the upper-right corner are Pearson correlation coefficients (R) and the p-value (significant at $p < 0.05$).

Annual spring sampling

Despite having expanded fetches to easterly and westerly winds, the station at Pelican Lake thermally stratified twice in the spring since 2010 (2010, 2014) (Figure 7.5). Pelican Lake remained well-oxygenated, rarely below 5 mg L^{-1} (2012, 2014) (Figure 7.5).

Similar to measured constituents in the winter, analytes measured in the spring have considerable variability and show little to no relationship with rising water levels (see Figure 7.6). For example, specific conductance showed extreme variability (702 µS/cm to 1,910 µS/cm), but these fluctuations were not related to increases in water volume ($R = -0.063$) (Figure 7.6).

Nutrient concentrations, however, have some relationship with the increases in water level. Concentrations of TN ($R = -0.665$; Figure 7.7), TKN ($R = -0.622$), and ammonia ($R = -0.538$) decreased significantly with increasing water levels. TP ($R = 0.041$; Figure 7.7) and DP ($R = 0.132$), however, were not related to rising lake elevation in Pelican Lake.

TSI scores based on Secchi disk transparency and chlorophyll-α concentration show considerable variability over the study period. TSI scores in the spring classify Pelican Lake between mesotrophic and eutrophic (Figure 7.8). TSI scores based on transparency varied from 58.63 (1.1 m) in 2013 to 36.81 (5.0 m) in 2014 (Figure 7.8). TSI scores based on chlorophyll-α concentration varied from 60.92 in 2002 to 34.58 in 2008 (Figure 7.8).

Analyzing TSI scores suggests that Pelican Lake is not P-limited in the spring (Figure 7.9), based on TSI scores for TP being much greater than those for chlorophyll-α. Additionally, Pelican Lake appears to be dominated by smaller particles of algal and non-algal turbidity, causing TSI scores based on Secchi disk transparency to be higher than those based on chlorophyll (Figure 7.9).

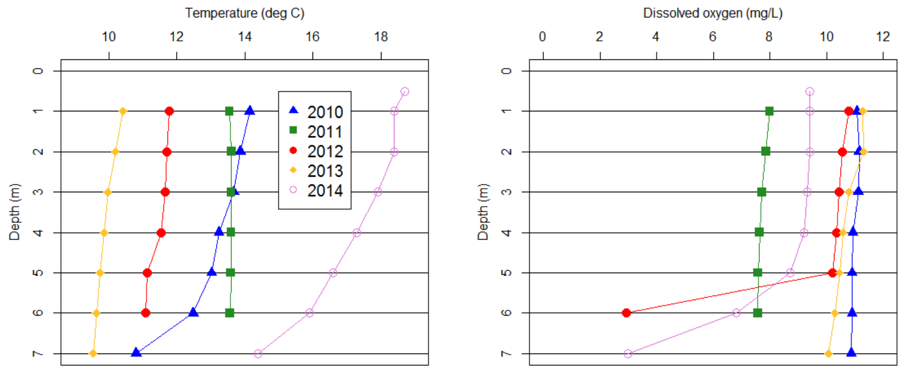

Figure 7.5: Spring temperature (left) and DO (right) profiles for Pelican Lake from 2010 through 2014.

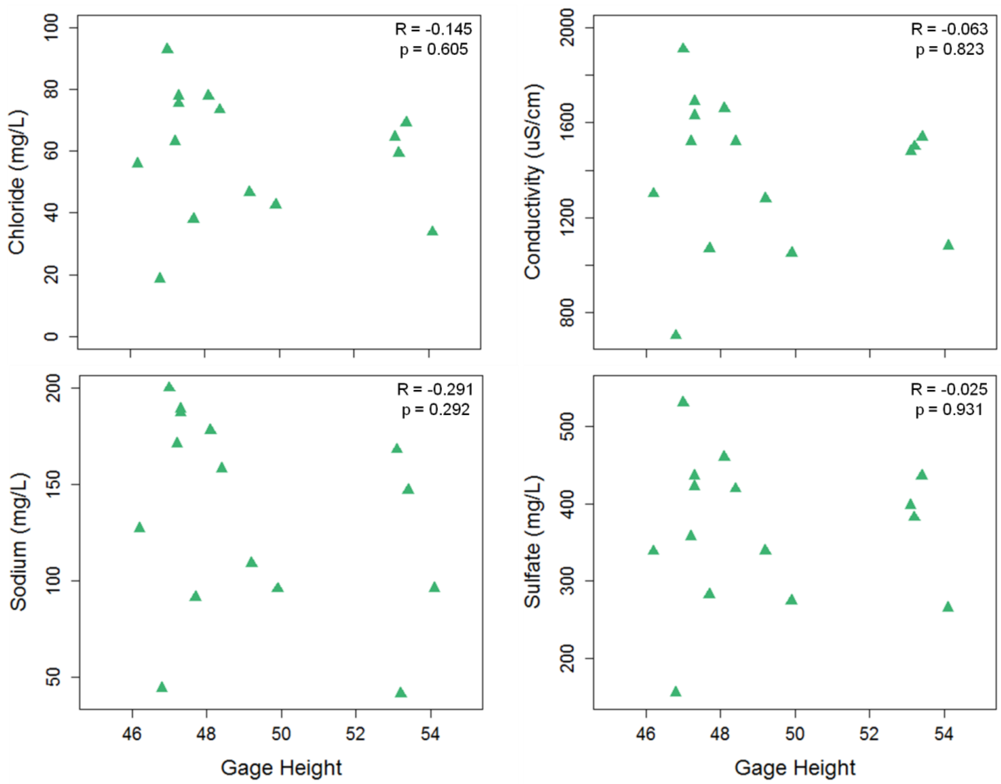

Figure 7.6: Spring concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Pelican Lake. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

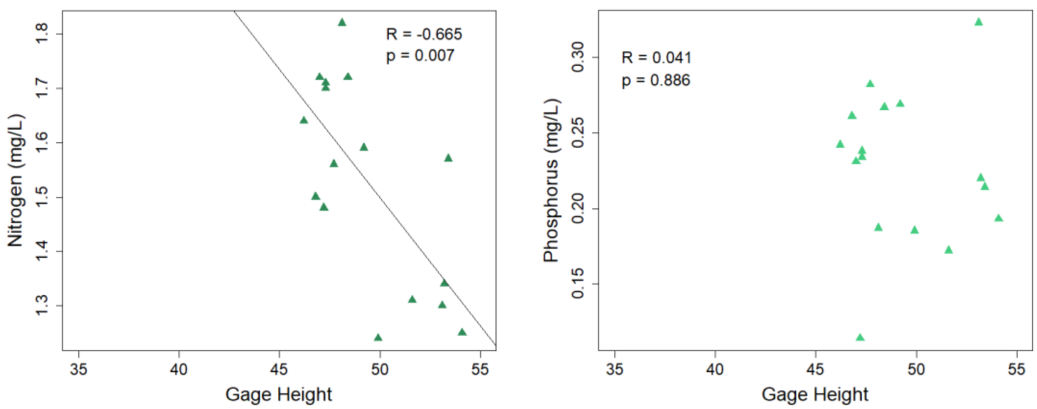

Figure 7.7: Spring concentrations of TN (left) and TP (right) plotted against lake elevation at Pelican Lake.

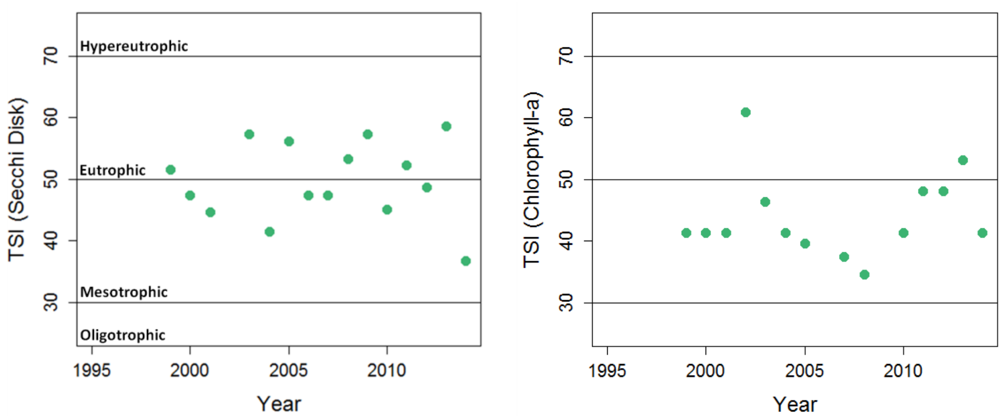

Figure 7.8: Spring TSI scores for Pelican Lake from 1999 through 2014.

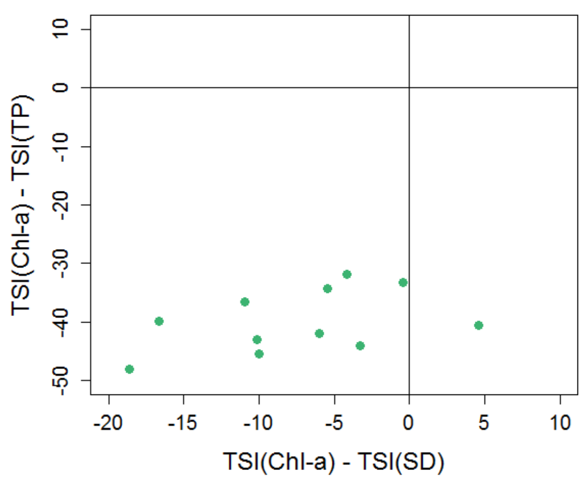

Figure 7.9: Nutrient-limitation graph for spring sampling at Pelican Lake.

Annual summer sampling

Likely due to aforementioned fetches exposed to easterly and westerly winds, Pelican Lake rarely stratifies during summer sampling (only in 2013) (Figure 7.10). Due to this constant mixing of the water column, Pelican Lake remains well-oxygenated in the summer, not being less than 5 mg L^{-1} since 2010 (Figure 7.10).

Pelican Lake continued to display high inter-annual variability among measured analytes, with no constituents relating significantly to increasing lake elevation (see Figure 7.11). Specific conductance, for example, continued to show considerable variability (864 µS/cm to 1,970 µS/cm), but was not related to increasing water levels (R = -0.230) (Figure 7.11).

Nutrient concentrations during summer sampling were not affected by increasing water levels. Concentrations of TN (R = -0.352; Figure 7.12), TKN (R = -0.352), and N+N (R = 0.189) did not significantly relate to increasing lake elevation. TP (R = -0.152; Figure 7.12) and DP (R = 0.104) were also not related to increasing water levels in Pelican Lake.

Over the study period, summer TSI scores in Pelican Lake appeared to decrease slightly, classifying Pelican Lake between eutrophic and hypereutrophic. TSI scores based on Secchi disk transparency ranged from 73.20 (0.4 m) in 2003 to 53.22 (1.6 m) in 2011 (Figure 7.13). Similarly, TSI scores based on chlorophyll-α concentration decreased from 73.34 in 2000 to 60.28 in 2014 (Figure 7.13).

Pelican Lake was not P-limited in the summer; however, there are some years where the basin is nearly P-limited (Figure 7.14). Similar to spring sampling in Pelican Lake, the basin continues to be dominated by smaller particles of algal and non-algal turbidity, evidenced by TSI scores based on Secchi disk transparency being consistently higher than those based on chlorophyll-α (Figure 7.14).

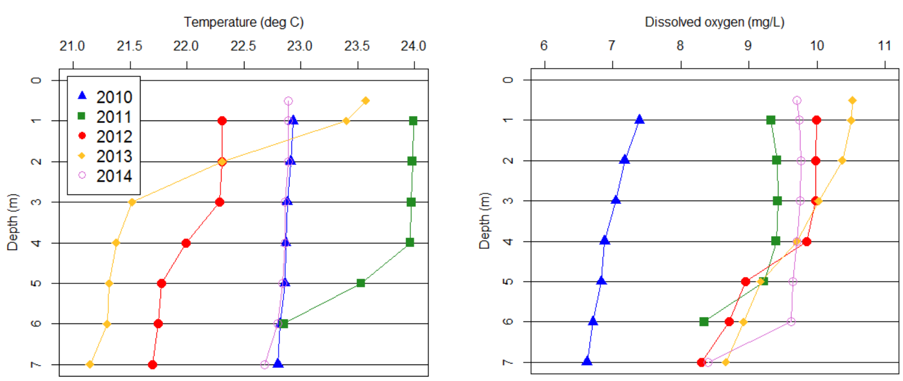

Figure 7.10: Summer temperature (left) and DO (right) profiles for Pelican Lake from 2010 through 2014.

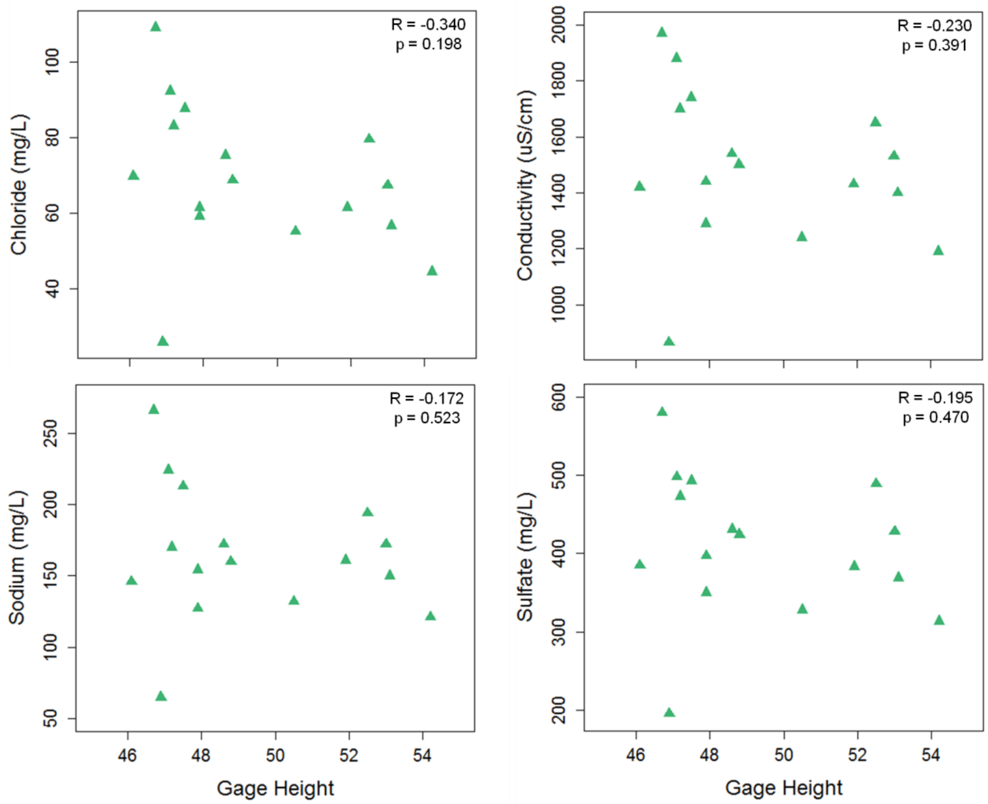

Figure 7.11: Summer concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Pelican Lake. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

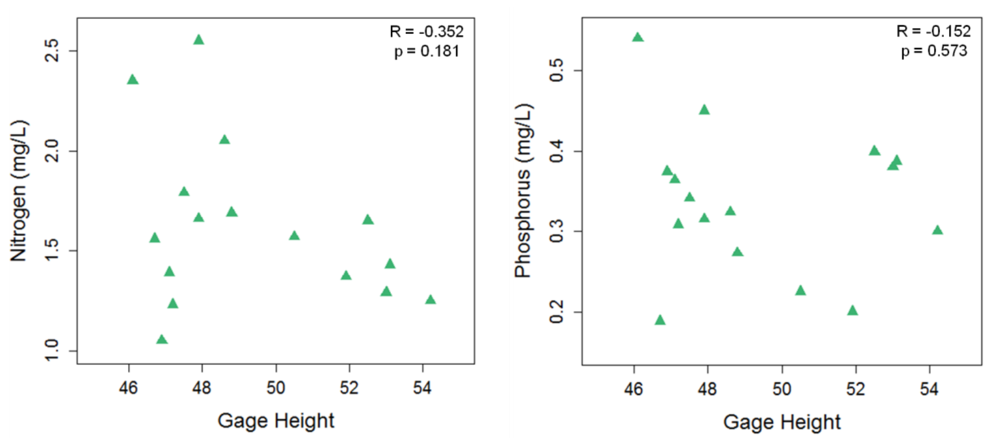

Figure 7.12: Summer concentrations of TN (left) and TP (right) plotted against lake elevation at Pelican Lake.

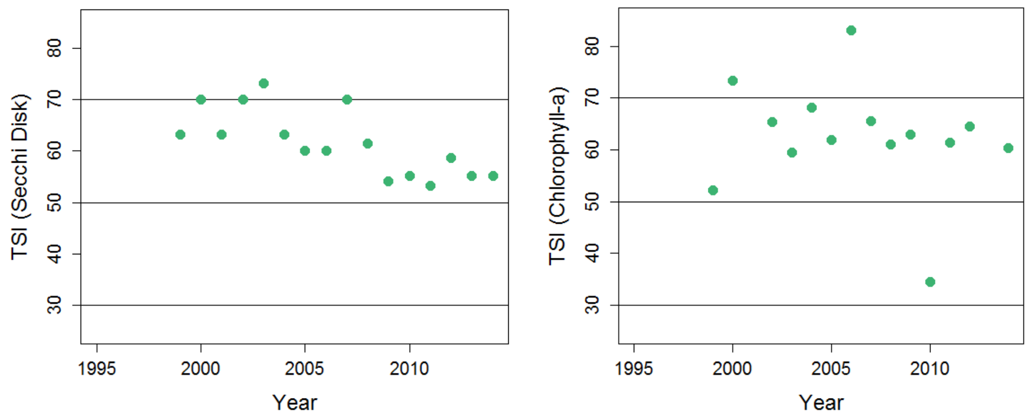

Figure 7.13: Summer TSI scores for Pelican Lake from 1999 through 2014.

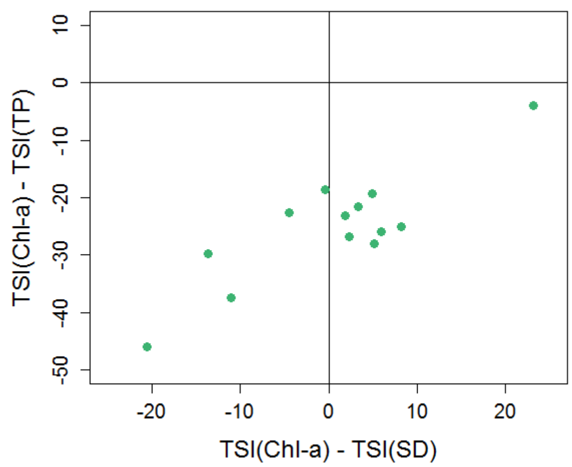

Figure 7.14: Nutrient-limitation graph for summer sampling at Pelican Lake.

Annual fall sampling

Similar to the rest of Devils Lake, Pelican Lake did not thermally stratify during fall sampling (Figure 7.15). Due to this constant mixing, Pelican Lake remained well-oxygenated to the bottom, with DO concentration not less than 8 mg/L (Figure 7.15).

Similar to other sampling seasons, ionic concentrations were highly variable during fall sampling in Pelican Lake (see Figure 7.16). Barium (R = -0.532) and manganese (R = -0.552) declined significantly with increasing water levels. Specific conductance was highly variable over the study period (R = 0.001), ranging from 932 µS/cm to 2,010 µS/cm with no defined trend (Figure 7.16). Further, nutrients were not related to lake elevation (see Figure 7.17).

TSI scores based on chlorophyll-α and Secchi disk transparency classify Pelican Lake as eutrophic in the fall. TSI scores based on transparency depth were variable ranging from 73.20 (0.4 m) in 2007 to 48.64 (2.2 m) in 2013. Chlorophyll-α TSI scores were variable as well, but showed a declining trend, going from 68.80 in 2008 to 55.76 in 2012 (Figure 7.18).

Similar to the rest of the lake, Pelican Lake was not P-limited during fall sampling, as evidenced by TSI scores based on TP being much higher than those based on chlorophyll-α (Figure 7.19). Further, the photic zone has been dominated by small particles of non-algal turbidity, as evidenced by scores based on transparency being higher than those for chlorophyll-α (Figure 7.19).

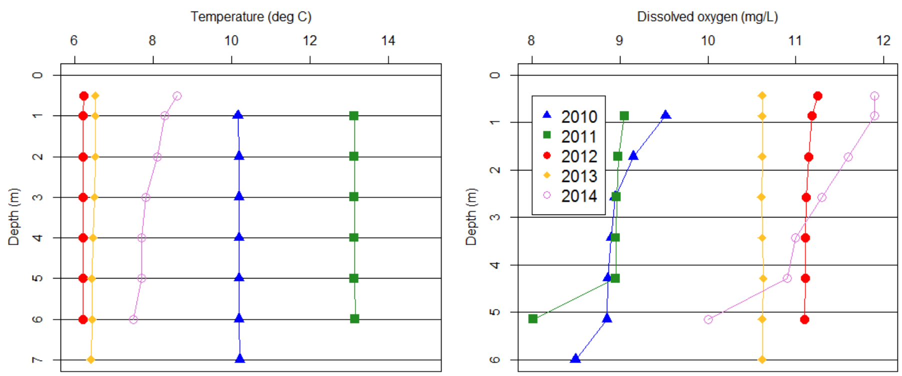

Figure 7.15: Fall temperature (left) and DO (right) profiles for Pelican Lake from 2010 through 2014.

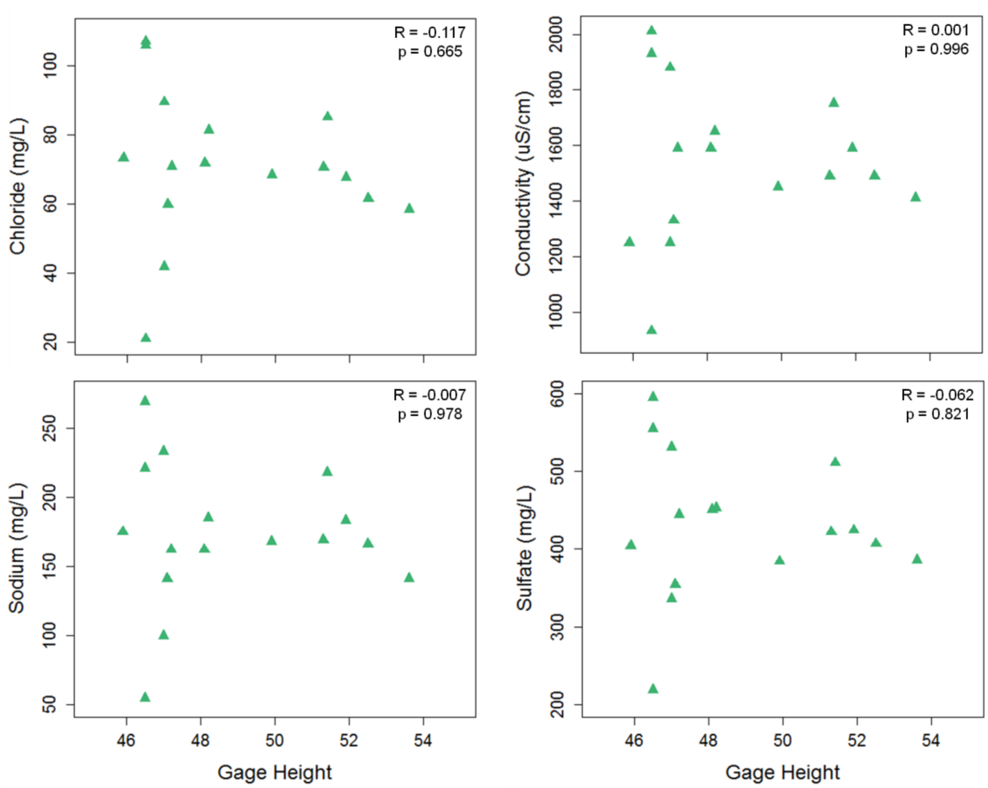

Figure 7.16: Fall concentrations of chloride (top left), conductivity (top right), sodium (bottom left), and sulfate (bottom right) plotted against lake elevation at Pelican Lake. Displayed value for lake elevations represents values in the 1400s (i.e., 52.5 equals 1452.5).

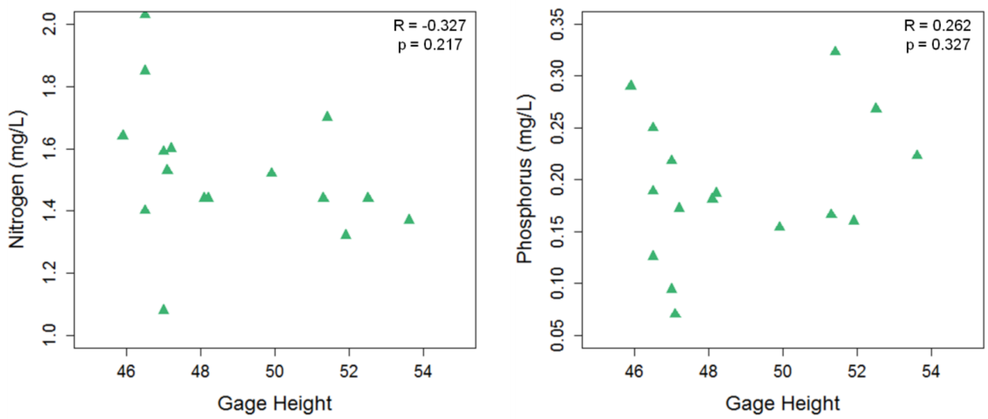

Figure 7.17: Fall concentrations of TN (left) and TP (right) plotted against lake elevation at Pelican Lake.

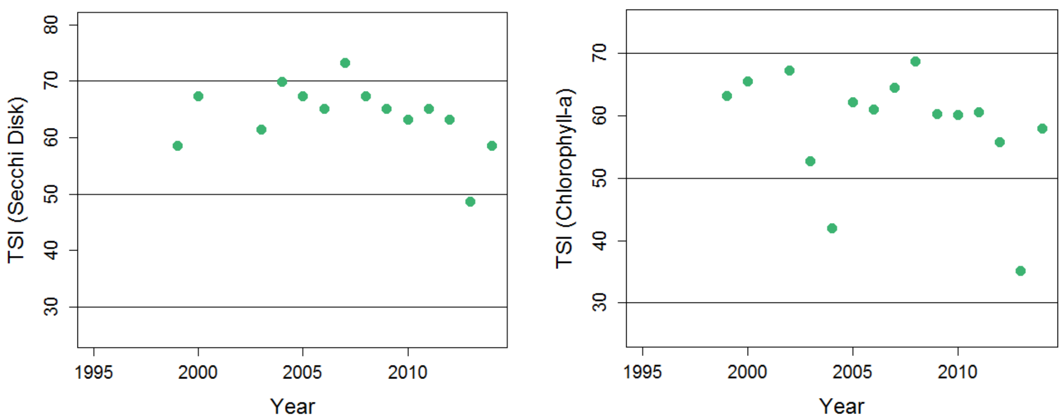

Figure 7.18: Fall TSI scores for Pelican Lake from 2003 through 2014.

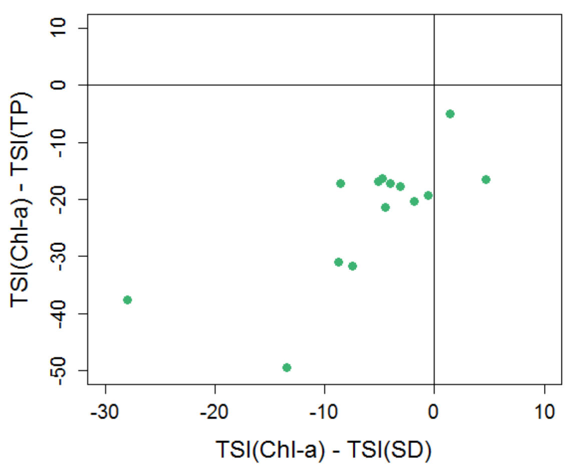

Figure 7.19: Nutrient-limitation graph for fall sampling at Pelican Lake.

Phytoplankton assemblage sampling

Phytoplankton sampling began on Pelican Lake in the spring of 1999. As with all other sites, cyanobacteria dominated during summer and fall months, but remained relatively rare during spring samples (Table 7.1). Chrysophytes and Heterokonts were in relatively high densities in 1999, but were almost unseen during 2000 samples (Table 7.1). Chlorophytes and Cryptophytes were present during both years with species-present being different between 1999 and 2000 (Table 7.1), a pattern observed throughout the whole basin.

Table 7.1: Presence-absence table showing common phytoplankton species and when they were detected in Pelican Lake for 1999 and 2000. An "X" signifies that the species was present during that particular season.

	1999			2000		
	Sp	**Su**	**Fa**	**Sp**	**Su**	**Fa**
Chlorophyta						
Chlorella minutissima				X	X	X
Choricystis minor				X	X	
Closterium acutum var. var.					X	X
Monoraphidium minutum	X	X	X			
Raphidonema sp.	X	X				
Chrysophyta						
Chrysococcus cordiformis	X	X	X			
Chrysococcus cordiformis var. ast.	X	X	X			
Chrysococcus minutus	X	X	X			
Chrysococcus porifer	X	X	X			
Chrysococcus punctiformis	X	X	X			
Chrysococcus rufescens var. com.	X	X				
Chrysococcus rufescens var. tri.	X	X	X			
Kephyrion littorale	X	X	X			
Kephyrion rubri-claustri	X	X	X			
Kephyrion sitta	X	X	X			
Cryptophyta						
Chroomonas acuta	X		X			
Cryptomonas marssonii	X		X			
Plagioselmis sp.				X	X	X
Rhodomonas pusilla	X	X				
Cyanobacteria						
Anabaena spiroides		X	X			
Aphanizomenon flosaquae		X			X	X
Aphanothece smithii				X	X	
Gloeocapsa aeruginosa	X	X	X			
Merismopedia tenuissima		X	X			
Microcystis aeruginosa		X	X		X	X
Oscillatoria angustissima		X	X			
Oscillatoria utermoehl					X	X
Heterokontophyta						
Cyclotella meneghiniana	X		X			
Nitzschia frustulum	X	X	X			
Nitzschia inconspicua	X		X			
Stephanodiscus niagarae		X	X			X
Stephanodiscus rotula		X	X			

4.0 Discussion

Devils Lake is a well-oxygenated, polymictic lake with highly fluctuating lake levels and, subsequently, highly fluctuating ionic and nutrient concentrations. Devils Lake has risen tremendously over the study period, rising from an elevation of 1,431 feet asl in 1995 to a peak elevation of 1,454.3 feet asl in 2011 (USGS), with tremendous intra-annual variability as well. This rise in elevation has brought "fresher" water in to the Devils Lake Basin, and has most likely had positive effects on the biological communities in the basin. Concentrations of most ions decreased sharply with increasing water levels, a relationship that has been observed elsewhere (Fritz, 1990).

Water column dissolved oxygen and temperature readings remained relatively similar from top to bottom during spring and fall sampling, due to constant mixing from long fetches throughout the Devils Lake Chain. Well-oxygenated areas near the water-sediment interface have been observed in other studies (e.g., Sando and Lent, 1995), and can provide good deep-water habitat for numerous fish species. Due to long fetches, thermal stratification remained relatively rare even during summer sampling, a finding consistent with other studies (e.g., Sando and Lent, 1995). Additionally, dissolved oxygen concentrations during summer sampling remained relatively high from top to bottom throughout the chain, agreeing with the findings of Sando and Lent (1995), who reported a mean near-sediment oxygen concentration of 7.7 mg/L. For our study, near-sediment dissolved oxygen levels were extremely variable, but were generally above the State's standard (5 mg/L), though Sando and Lent (1995) found mean near-sediment dissolved oxygen level was lowest during the winter sampling period, averaging approximately 5.3 mg/L.

Over the study period, we found that concentrations of most ions increase progressively from west to east throughout the Devils Lake chain, an observation made during previous studies (e.g., Pope, 1908; Sando and Lent, 1995). Seasonally, ionic concentrations tend to be highest during the winter, lowest in the spring corresponding with increased runoff, and then generally increase towards fall.

Specific conductance ranged from peaks of 2,100 to 11,000 µS/cm (going from west to east) in 1995 compared to 2,120 to 3,050 µS/cm in 2014. These East Devils Lake results compare with concentrations observed as high as 60,000 µS/cm observed in East Devils Lake between 1956 and 1960 in other studies (Mitten et al., 1968). Concentrations remained relatively high through the late 80s, early-90s, with a peak near 20,000 µS/cm (Sando and Lent, 1995) prior to the observed decline through 2014.

The breadth of these values shows the dynamic nature of ionic concentrations in Devils Lake. Over the study period, many measured analytes decreased dramatically in the eastern part of Devils Lake as it became mixed with water from the western side. Conversely, stations in the western part of the lake increased initially (likely with an influx of water from the eastern side), then decreased strongly with increasing water levels. This finding is parallel to that of Mitten et al. (1968) who observed analytes decreasing throughout the basin with increasing water levels, likely due to dilution. Unlike most analytes that tended to decrease with increased water levels, constituents such as barium and calcium increased significantly across the basin. Historically, runoff into Devils Lake has been dominated by calcium, magnesium, and bicarbonate (Mitten et al., 1968) and their increase over the study period may be attributed to flooded soils contributing these ions when inundated. In addition to our findings associating dilution and decreases in Devils Lake analytes, Lent (1994) provides evidence (both empirical and reviewed) for reductions and fluxes occurring within benthic sediments, though these were not measured for our study. Increases due to flux from sediment processes could help explain increases in barium and calcium, despite decreases in the concentrations of several other constituents. Further, Zohary and Ostrovsky (2011) suggest that excessive water level fluctuations, likely similar to those observed in Devils Lake, can contribute significantly to internal cycling of nutrients.

Pope (1908) hypothesized that these high concentrations of ions would have a detrimental effect on reproduction of many fish species within the lake. In fact, Koel and Peterka (1995) found that TDS above 4,250 mg/L was the threshold for hatching success for northern pike and walleye, though success was still low at these concentrations. Further, the TDS threshold for successful hatching in yellow perch and white sucker was 2,400 mg/L (Koel and Peterka, 1995). In this study, TDS in East Devils Lake was 7,570 mg/L in 1995, which likely inhibited any fish production in this section of the lake. Presently, Devils Lake supports a healthy population of game fish, such as walleye and northern pike (ND Game & Fish, personal communication). Historically, during historical low-water conditions, however, Devils Lake mainly supported relatively few hardy fish species (e.g., stickleback, fathead minnow) due to high concentrations of ions, with a documented disappearance of northern pike during this time (Pope, 1908).

Nutrient concentrations within the basin were extremely variable, occasionally showing a relationship with increasing water levels over the study period. Nitrogen- and phosphorus-mass appear to play the biggest role in inter-annual variability of these nutrients, varying with re-suspension of nutrient-rich sediments, breakdown of organic materials, and nutrient-fixation. Despite this, concentrations of TP and DP remain relatively high throughout the basin, with TSI scores commonly exceeding 80 (Hypereutrophic) and concentrations consistently greater than 150 µg/L. While there is a suggested relationship between TP and chlorophyll-α concentrations (from Wetzel et al. 2003), we do not observe this within Devils Lake as both exhibit extreme inter- and intra-annual variability. Sando (1992) reported few external sources being important for nutrient variability, with the exception of increased P due to stormwater and wastewater runoff from the City of Devils Lake. We hypothesize that fluxes in chlorophyll-α concentration over time are a result of these variable fluxes of nutrients, and not due to variation with outside sources of N or P.

Secchi disk transparency depths generally decreased throughout the basin over the study period (1995 through 2014), with non-algal turbidity playing a lesser role in obstructing light penetration. An inverse trend of TSI scores based on Secchi disk transparency and water levels was also observed by Lent (1994). This trend is shown throughout most of the basin when analyzing TSI scores, as TSI scores based on chlorophyll-α concentrations are now exceeding those based on Secchi disk transparency in recent years, suggesting more effective zooplankton grazing in the photic zone. This may also be due to deeper water within Devils Lake not allowing for "complete mixing", with less sediment being suspended in the water column and subsequently improving light transparency. Further, it appears the photic zone in Devils Lake becomes more commonly dominated by larger particles of algae, which may be a result of less mixing due to the increased depth within the basin and not allowing these sediments to re-suspend as often based on phosphorus concentration. For example, transparency depths in Main Bay ranged from 1.0 meter in 2003 to 8.4 meters in 2010. This compares to historical data from Lent (1994) who reported that transparency depths did not exceed 2.5 meters when the lake elevation was much lower in 1965-1968, 1974-1975, and 1989-1991.

While there was predictable intra-annual variability among the phytoplankton community due to seasonality, there was also considerable inter-annual variability of species composition, with many species being present every other year. This association of community similarity every other year is observed at all stations, with many species reaching relatively high densities one year, then completely disappearing the next. It is unclear whether this is a natural phenomenon or if this is driven by other physical or chemical variables. Leland and Berkas (1998) did not observe species being present only every other year, but instead found presence to be less variable, with some species being dominant every year. Additionally, Verch and Blinn (1971) did not observe significant inter-annual variability from 1970-1971 among the phytoplankton community. Additional investigations should be conducted on the phytoplankton community of Devils Lake to determine if these communities are driven more by natural variation in annual composition or if algal diversity is driven by physical and/or chemical factors. For most stations, species of cyanobacteria were at lowest abundance during spring samples, with densities rising significantly during summer and fall, at times comprising greater than 90 percent of the phytoplankton community. *Aphanizomenon flos-aquae* is one of the few species found to be dominant annually throughout much of Devils Lake was also found to be dominant annually from 1989-1994 (Leland and Berkas, 1998), though for both studies it was not dominant during spring sampling. Chrysophytes (i.e., golden algae) had a large increase in species richness and abundance during 1999 sampling throughout Devils Lake. While the cause of this sudden increase is unknown, our analysis suggests it was driven by a decrease in ionic concentrations throughout Devils Lake (e.g., specific conductance declined by approximately 400 µS/cm in Six Mile Bay from 1995-1999). Previous investigations in Stump Lake suggest much lower algal diversity during times of high ionic concentrations (Blinn, 1972). Blinn (1993) found a strong, negative correlation between specific conductance and the presence and abundance of diatom species; thus, the low numbers of these species from our samples may be driven by high concentrations of ions represented by higher specific conductance. With specific conductance declining significantly in recent years, particularly in the eastern section of Devils Lake, a present-day investigation of the phytoplankton community may yield higher richness and abundance of diatom species. The aforementioned constant mixing of the water column due to long fetches in parts of Devils Lake may also lead to diel variability among the phytoplankton community and thus affect community composition in phytoplankton samples collected.
As the Devils Lake basin experiences continual fluctuations in water level, ionic and nutrient concentrations can also be expected to vary. The inter- and intra-annual variability among these analytes can be substantial, likely having an effect on biological processes within Devils Lake.

With the installation of two outlets draining to the Sheyenne River, the elevation of Devils Lake is likely not to increase substantially in the future, with the likelihood of an decrease unknown.

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