Assessment of Water Quality Responses to Sediment Removal in Lake Hancock

Prepared for

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and

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Executive Summary

Water quality in Lake Hancock has been characterized as “poor” based on the State of Florida’s Trophic State Index (TSI) since at least 1970. More recently, Lake Hancock’s water quality was verified as impaired for nutrients by the Florida Department of Environmental Protection (FDEP) due to levels of total nitrogen (TN), total phosphorus (TP) and biological oxygen demand (BOD) that far exceed the State of Florida’s threshold screening values.

The extensive and organic-rich sediments in Lake Hancock are either a net sink for nutrients from the water column or a net source for nutrients into the water column. Results from this and other studies indicate that these sediments store phosphorus from both current and historical external delivery, and that phosphorus is “pulsed” into the water column during the resuspension of sediment material on windy days, as well as remineralized from bottom sediments.

Polk County and FDEP contracted with PBS&J to examine the potential benefits to water quality in Lake Hancock due to sediment removal activities. Results from these efforts, as related to potential improvements in lake water quality, were then placed into the context of potential Total Maximum Daily Load (TMDL) determinations for Lake Hancock and its watershed tributaries

The project design included two concurrent field experiments. The first (Task 1) was a manipulative experiment to determine if variation in the levels of nutrient availability could bring about decreases in blue-green algal vigor and/or abundance. The second experiment (Task 2) was designed to evaluate whether partial removal of the lake’s sediments would likely result in significant improvements in water quality.

Results from Task 1 show clearly that light intensity affects primary production in Lake Hancock such that photosynthetic rates decrease at higher light intensities. These results suggest that phytoplankton communities in Lake Hancock are currently adapted to low light levels that accompany the very low water clarity. Other results from Task 1 show evidence of an immediate increase in nitrogen fixation rates upon phosphorus enrichment. These results suggest that nitrogen fixation is sensitive to relatively low concentrations of phosphorus enrichment, and that rates of nitrogen fixation are “saturated” at TP concentrations of 0.16 mg/liter.

Results from Task 2 show that levels of TN were reduced between 13 and 53 percent with sediment removal, while levels of TP were reduced between 21 and 77 percent. Levels of total suspended solids were reduced between 35 and 78 percent, while turbidity levels decreased by 14 to 75 percent. Chlorophyll *a* concentrations were reduced by 5 to 31 percent with sediment removal. Ratios of TN to TP increased between 26 and 111 percent with sediment removal, resulting in TN:TP ratios that are considered less favorable for blue-green algae,. However, TSI values exhibited a lesser degree of change with sediment removal, with reductions of 1 to 12 percent found.

These findings are consistent with the hypothesis that Lake Hancock is a net sink for phosphorus loads that accumulate in the sediments. At the same time, the lake is a net source of nitrogen

(created via nitrogen fixation) which is exported from the lake and eventually into the Peace River and Charlotte Harbor.

Only by reducing TP loads from both external and internal loading sources could TP levels be reduced to biologically relevant levels, i.e., levels that may affect blue-green algae production. Load reduction strategies that do not appropriately incorporate sediment nutrient loads might not improve water quality to the desired level. Consequently, the stormwater-only TP load reduction of 75.2 percent called for in the draft TMDL for Lake Hancock appears to be problematic.

An appropriate target for TP levels in Lake Hancock would likely be between 0.08 and 0.16 mg / liter. In contrast, TP levels in the lake averaged 0.52 mg/liter between 1992 and 2004. Results from Task 2 show that TP levels within dredged cylinders in December 2006 averaged between 0.13 and 0.21 mg /liter and represent TP reductions of 37 and 77 percent. In May 2007, phosphorus levels in dredged cylinders averaged between 0.41 and 0.51 mg /liter. Although these levels were higher than levels in December 2006, they represent reductions of 21 and 69 percent.

The results from Task 2 suggest that sediment removal could substantially reduce TP levels in Lake Hancock and if results can be scaled up to whole-lake expectations, TP levels could potentially be reduced to biologically relevant concentrations for at least portions of the year.

Further, results from Task 2 suggest that levels of TP expected with sediment removal would result in TN:TP ratios less favorable for blue-green algae. However, these TP levels would still be higher than the TP concentration found to saturate nitrogen-fixation (0.16 mg TP/liter).

An additional assessment was conducted that examined the probability of sediment resuspension in Lake Hancock. This assessment relied on a series of scenarios, including one that incorporated the removal of 3 feet of lake sediments combined with a 1.8 feet increase in water level (the number used in the Southwest Florida Water Management District's proposed lake level modification project). Results from this assessment suggest that an increase in the lake's effective water depth of 4.8 feet would reduce the frequency of whole-lake sediment resuspension events from approximately once a week to just over once every two weeks. **Since productivity of blue-green algae increases in response to phosphorus pulses from sediment resuspension, actions that decrease sediment resuspension by increasing the effective water depth (e.g. increasing water levels or removing sediments), are likely to improve water quality in Lake Hancock.**

Table of Contents

List of Tables

List of Figures

List of Photos

Appendix A

Appendix B

1.0 Introduction

Lake Hancock, with a surface area of approximately 4,550 acres, is the third largest lake in Polk County (ERD, 1999). The contributing watershed is approximately 131 square miles in size, for a watershed to open water ratio of 18:1. The major tributaries to Lake Hancock are the Banana Creek sub-basin (13,578 acres), the Lake Lena Run sub-basin (11,754 acres) and the North Saddle Creek sub-basin (49,034 acres). The only distributary from Lake Hancock is South Saddle Creek which ultimately discharges into the Peace River.

Lake Hancock has been characterized as having "poor" water quality, using the State of Florida's Trophic State Index (TSI), since at least 1970 (Polk County, 2005), and concerns over poor water quality in the lake have existed as far back as the 1950s (ERD, 1999). More recently, Lake Hancock's water quality was verified as impaired for nutrients based on data collected between January 1997 and June 2004 (EPA, 2005). Levels of total nitrogen, total phosphorus and biological oxygen demand exceeded the State of Florida's threshold screening values, all by considerable amounts (EPA, 2005). The poor water quality in Lake Hancock has prompted in a number of reports focusing on strategies to improve its condition.

1.1. Previous Lake Restoration Reports

In 1986, Zellars-Williams developed a nutrient budget and restoration strategy for Lake Hancock that pointed to the potential benefits of sediment removal on lake water quality.

In 1999, ERD completed a nutrient budget and water quality modeling effort that concluded that even if nutrient loads from base flow and stormwater runoff were reduced by 75 percent (an implausible scenario) water quality in the lake would remain "poor" based on TSI values. The report indicated that an estimated that a 50 percent reduction in nutrient loads from baseflow and stormwater, in combination with a whole-lake dredging effort, could potentially improve the mean TSI value to 70, the threshold between "fair" and "poor" water quality (ERD, 1999). This report also highlighted the significant reductions in pollutant loads to the Peace River that would be expected to occur by various outfall treatment alternatives. As an example, the least effective outfall treatment alternative investigated by ERD (1999) would still reduce nitrogen loads leaving the lake by 92,380 kg / year, an amount greater than the 20 year horizon for population-growth related nitrogen increases expected for the entire Peace River watershed (81,500 kg / yr; Coastal Environmental, 1995).

Also in 1999, a draft report entitled "Lake Hancock Habitat Enhancement Plan" was prepared by the Florida Fish and Wildlife Conservation Commission. This restoration plan included a number of activities, including reestablishing historical aquifer levels to restore seepage into North Saddle Creek, and reestablishing native vegetation along the lake's shoreline and in the lower reaches of Banana Creek. As in the ERD (1999) report, this plan also highlighted the need to construct a downstream treatment wetland for lake discharges, particularly during the latter phases of lake restoration efforts, when the lake's sediments would be removed and transported to either "in-lake disposal sites" or to "isolated nearby disposal sites".

In a report conducted for Polk County and the Florida Department of Environmental Protection (FDEP) by CDM (2002), a variety of lake restoration techniques were assessed. A number of options were developed, and the highest-rated options all included the following components: 1) wetland treatment of discharges from the lake, 2) chemical treatment/settling pond treatment of inflows from North Saddle Creek, 3) wetland treatment of inflows from Banana Creek, and 4) passive drawdown and mechanical excavation of bottom sediments in nearshore areas. Regardless of the techniques employed to restore water quality within the lake, CDM (2002) concluded that the wetland treatment of lake discharges component should be incorporated into any potential restoration strategies, as it would both further improve the quality of water leaving the lake, and also provide an “environmental safeguard” for other activities, such as hydraulic dredging of the lake’s sediments.

After reviewing these and other reports, the Southwest Florida Water Management District (District) concluded that “The results of the study (from ERD, 1999) showed little benefit for in-lake water quality treatment in regard to the high cost of treatment. Based on the findings of the study the option to treat waters leaving the lake was chosen.” Thus, the District has not concluded that projects intended to improve the quality of the lake’s water were inappropriate, but that the benefits might not be worth the costs involved.

Currently, two of the four restoration techniques proposed by CDM are scheduled for implementation. Polk County has already constructed a wetland treatment system that treats approximately 50 percent of the Banana Creek inflows to Lake Hancock. Preliminary results indicate a significant improvement in the water quality, thereby further reducing the external nutrient load to the lake. Additionally, the District is scheduled to construct a wetland treatment system to improve approximately 50 percent of the water leaving Lake Hancock. Consequently, dredging to improve water quality in the lake would not be in conflict with the District’s chosen course of action to construct. Also, the District’s decision to pursue the lake outfall treatment option is consistent with the recommendation from the CDM report (2002) that such a system would be needed to act as an “environmental safeguard” for some of the more aggressive potential lake restoration activities.

Using methods outlined in the Impaired Waters Rule (IWR- Florida Administrative Code 62.303) the Florida Department of Environmental Protection (FDEP) concluded that both Lake Hancock and Lower Saddle Creek were verified as impaired for nutrients and dissolved oxygen (DO). Lake Hancock was verified as impaired in 2003 for nutrients based on an annual average Trophic State Index (TSI) value of 83.5, which exceeded the IWR threshold value of 60 during the verification period (the Verified Period for the Group 3 basins was from January 1, 1997 to June 30, 2004). The median values for total nitrogen (TN), total phosphorus (TP), and biochemical oxygen demand (BOD) for the lake for the verified period were 4.46, 0.49, and 9.9 mg/liter, respectively (FDEP 2005). In comparison, 90 percent of Florida lakes have a median TN or TP concentration below 2.5 and 0.29 mg/liter, respectively (FDEP 2005). Ninety percent of Florida streams have a median BOD below 5.1 mg/liter (FDEP 2005).

In addition to being listed as impaired for nutrients, FDEP listed Lake Hancock as impaired for DO because more than 19 percent (27/140) of DO values collected during the verified period were below the freshwater DO standard of 5 mg/liter (FDEP 2005).

South Saddle Creek was verified as impaired for nutrients based on chlorophyll *a* values that exceeded the IWR listing threshold of 20 µg/liter for freshwater streams (FDEP 2005). During the verified period, annual average chlorophyll a levels ranged from 89 to 164 µg/liter (FDEP 2005). Reflecting the extremely high levels of organic matter discharging from Lake Hancock, the median value for biological oxygen demand (BOD) in South Saddle Creek during the verified period was 6.5 mg/liter, a very high level (FDEP 2005).

Water quality data for nutrients, chlorophyll *a* and DO during the period of 1992 to 2004 is summarized in Table 1, as reported in FDEP (2005).

Table 1. Water quality from Lake Hancock and South Saddle Creek (data from FDEP 2005).

Waterbody	Parameter	Number of Samples	Minimum	Mean	Median	Maximum
Lake Hancock	TN (mg/L)	134	1.19	4.57	4.40	12.03
Lake Hancock	TP (mg/L)	135	0.04	0.52	0.49	1.01
Lake Hancock	Chl-a (µg/L)	77	16.0	175.2	148.0	558.0
Lake Hancock	DO (mg/L)	140	0.38	8.26	8.43	15.70
South Saddle Creek	TN (mg/L)	75	0.39	2.95	2.50	11.02
South Saddle Creek	TP (mg/ L)	134	0.01	0.63	0.45	10.50
South Saddle Creek	Chl-a (µg/L)	111	1.0	101.2	79.3	612.0
South Saddle Creek	DO (mg/L)	130	0.31	6.08	6.26	15.56

Numerous studies conducted within the Lake Hancock watershed all come to a similar conclusion – both Lake Hancock and South Saddle Creek exhibit symptoms of extreme hypereutrophication, with exceptionally high levels of nitrogen, phosphorus, chlorophyll *a* and BOD, and highly variable levels of DO.

In response to repeated findings of exceedingly poor water quality, pollutant loading models and pollutant load reduction strategies have been developed for Lake Hancock, to guide in lake restoration activities.

1.2. Nutrient Loading Models

In a detailed nutrient budget developed for Lake Hancock, ERD (1999) measured TN and TP loads into the lake from stormwater runoff, direct rainfall, baseflow, and in-lake seepage. They also measured nutrient loads leaving the lake via South Saddle Creek. Loads from these four sources totaled an estimated 175,879 kg TN entering the lake annually, along with an estimated 35,086 kg TP per year. Measured loads leaving the lake via South Saddle Creek totaled 271,700 kg / yr for TN and 23,180 kg / yr for TP.

Based on these calculations, approximately 34 percent of the TP loads into the lake remain in the lake. In contrast, there is a surplus of 95,821 kg TN leaving the lake on an annual basis, compared to the amount calculated to enter the lake. This represents a "surplus" of nitrogen equal to 35 percent of the total load leaving the lake.

A possible explanation for these findings is that the lake is a net sink for phosphorus loads, which accumulate in the sediments, while simultaneously being a net source for nitrogen, which is then exported out of the lake, and eventually into the Peace River via South Saddle Creek.

In their nutrient loading model, FDEP (2005) first attempted to determine whether nitrogen or phosphorus (or both) would likely be the nutrient limiting phytoplankton production in Lake Hancock. The technique used to determine the limiting nutrient(s) is outlined below.

Step 1 – ratios of TN to TP are calculated for each sampling event during the verified period (1997 to 2004) when both TN and TP are available

Step 2 – if median values of TN or TP exceed screening levels (the highest 30 percent of statewide values for the appropriate water body type) that nutrient which exceeds the screening level would be identified as a potential nutrient causing the impairment of TSI thresholds

Step 3 – the individual ratios of TN to TP for data during the verified period are compared, and if <u>all</u> TN to TP ratios calculated are less than 10 (by weight) nitrogen is determined to be the limiting nutrient. If <u>all</u> ratios are greater than 30, then phosphorus is determined to be the limiting nutrient. If neither of the above criteria are met, then the lake is determined to be limited by both nitrogen and phosphorus.

During the verified period, the median TN:TP ratios for Lake Hancock and South Saddle Creek (reported by FDEP [2005]) were 9.49 and 7.22, respectively. These median values would suggest nitrogen as the limiting nutrient. But as not all TN:TP ratios calculated were less than 10, nutrient load reduction goals for Lake Hancock were developed by FDEP (2005) for both nitrogen and phosphorus.

1.3. Influence of Nitrogen vs. Phosphorus on Lake Hancock Phytoplankton

The poor water quality of Lake Hancock is due either directly or indirectly to the very high levels of phytoplankton in the lake. In the lake, values of chlorophyll *a* (an indicator of phytoplankton abundance) can reach levels as high as 800 µg/liter (Parsons, 2005). In 2004, the annual average chlorophyll *a* level in Lake Hancock was 152 µg/liter (Polk County, 2005) more than seven times higher than the 20 µg/liter level that FDEP uses to classify lakes as "impaired" for nutrients.

Based on criteria developed by the State of Florida, ratios of TN to TP are used to infer the nutrient that is most likely to limit algal biomass. Ratios of 10 or less suggest limitation by nitrogen, ratios greater than 30 suggest limitation by phosphorus, and ratios between 10 and 30 suggest co-limitation. Recent data (Polk County, 2005) show an annual average TN:TP ratio of 9.6. At three sites visited monthly, Parsons (2005) reported TN:TP ratios that ranged between 8.6 and 8.7. And in a report that focused on the isotopic signature of Lake Hancock's sediments, Brenner *et al*. (2002) concluded that these sediments appeared to be associated with algal detritus from nitrogen-fixing blue green algae from the overlying water column.

These results suggest that nitrogen is the nutrient limiting algal production in Lake Hancock and that the "surplus" nitrogen exported from the lake is due to nitrogen-fixation by the lake's high levels of blue green algae (i.e., Brenner 2002). Not all the surplus nitrogen is exported from the lake, as Brenner (2002) found that sediment accumulation in the lake was a significant destination for an unknown portion of the lake's nitrogen budget. The conclusion that high rates of nitrogen fixation are at least partly responsible for the very high levels of nitrogen in Lake Hancock appears to be supported by recent water quality modeling work (FDEP 2005) wherein "internal loading" of TN was required to account for the lake's measured phytoplankton biomass. Therefore, it is important to determine what factors are most likely responsible for the dominance of blue-green algae in Lake Hancock's waters.

Early work by Smith (1983) suggested that low TN:TP ratios (<29), which could be caused by excessive phosphorus loading, favor dominance by blue-green algae. However, this conclusion may be overly simplistic. A broader and Florida-based assessment by Canfield *et al.* (1989) found little evidence for a dominant role of TN:TP ratios in determining blue-green algae abundance. Instead, Canfield *et al.* (1989) concluded that blue-green algae become proportionally more important in Florida lakes simply as overall phytoplankton biomass increases. Other researchers (i.e., Downing *et al.* 2001) also concluded that overall nutrient supply, rather than ratios of nitrogen to phosphorus, was the best predictor of the dominance of phytoplankton communities in lakes by blue-green algae.

A potential mechanism that would allow nutrient supply alone, rather than TN:TP ratios, to lead to lakes dominated by blue-green algae was originally outlined by Shapiro (1973). In this conceptual framework, it was first noted that blue-green algae have competitive advantages over other algal forms, in that they have preferential uptake of both free CO_2 and inorganic phosphorus, compared to non blue-green species. Additionally, blue-green algae can "fix" nitrogen gas (N_2), by converting biologically-inert (to most species) N_2 into ammonium and then assimilating ammonium into the amino acids, etc. needed for cellular growth. **The net result is that blue-green algae can more rapidly take up inorganic phosphorus and the inorganic carbon needed for photosynthesis, and they can access a pool of nitrogen that is biologically unavailable to other types of algae.**

Work by Shapiro (1973), combined with an understanding of the bicarbonate buffer system, can be used to explain the dominance of blue-green algae in Lake Hancock. The chemical equations used to describe the bicarbonate buffer system are summarized below:

$$CO_2 + H_20 \leftrightarrow H^+ + HCO_3^- \leftrightarrow 2H^+ + CO_3^{2-}$$

dissolved carbon dioxide		hydrogen ion	biocarbonate ion	hydrogen ion	carbonate ion

Shapiro (1973) then proposed the following scenario, to describe how increases in nutrient loads in general – regardless of the TN:TP ratio, can lead to dominance by blue-green algae.

1. Starting with a "healthy" lake with a mixed assemblage of algal species, an increased supply of nutrients (either TN and TP) would increase algal biomass.

2. As biomass and productivity increase, free CO_2 levels would decrease as this form of carbon is used for carbon fixation via photosynthesis.

3. Reduced levels of CO_2 would bring about an increase in pH, as the bicarbonate buffer system compensates (shifts to the left in the equation shown above) in response to reduced levels of free CO_2.

4. Continued reductions in the levels of CO_2 would bring about further increases in pH.

Levels of pH are important to lake water quality, as the form of inorganic carbon available for photosynthetic uptake varies with pH, as illustrated in Figure 1.

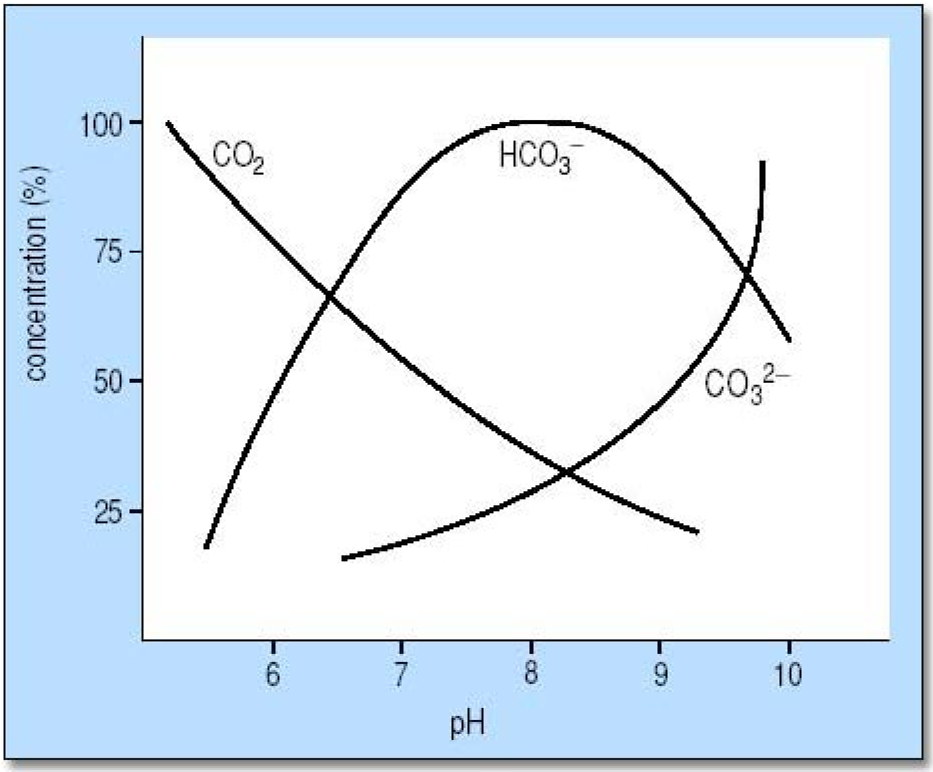


Figure 1. Influence of pH on forms of carbon dioxide in freshwater lakes.

As lake pH levels increase to values higher than 7 standard units (su), the form of carbon available for algal uptake becomes increasingly bicarbonate (HCO_3^-) rather than free CO_2. During the period of 1985 to 1999, the mean value for pH in Lake Hancock was 9.18, with a minimum value of 7.38 and a maximum value of 10.13 (ERD 1999). Under those conditions, the predominant form of carbon available would be bicarbonate and/or carbonate (CO_3^{2-}).

Loftus *et al.* (1979) and Tortell (2000) concluded that those species of phytoplankton such as blue-green algae that can directly utilize bicarbonate as a carbon source for photosynthesis, without having to first convert bicarbonate to CO_2 via the enzyme carbonic anhydrase, would have a competitive advantage in high pH lakes. In this sense, it would appear that the blue-green algae that dominate Lake Hancock's phytoplankton community might enjoy competitive advantages over other phytoplankton groups that allow them to become the dominant algal form in Lake Hancock.

1.4. Role of Lake Sediments on Water Quality

In Polk County, ground water in the surficial and the intermediate aquifers typically flow from the region of the Lake Wales Ridge to areas of lower elevation (FDEP 2005). However, declines in the Floridan aquifer due to groundwater withdrawals has created a downward movement of ground water into lower aquifer systems throughout the Upper Peace River (FDEP 2005). As such, evidence for groundwater seepage (i.e., ERD 1999) might be compromised due to the lack of a sufficient gradient in surficial aquifer hydrologic head pressure to cause water to “flow upwards” from sediments below Lake Hancock into the open waters of the lake. A layer of organic muck ranging between 1 to 4 feet in depth covers the bottom of Lake Hancock (FDEP 2005).

At its simplest, the extensive sediments in Lake Hancock are either a net sink for nutrients from the water column, or they are a net source for nutrients into the water column. Unfortunately, this question has not yet been unequivocally answered. It appears that Lake Hancock sediments play a more complicated role in nutrient delivery and storage. The sediments store phosphorus from historic external delivery; however, phosphorus is “pulsed” into the water column from the resuspension of sediment material during windy days.

2.0 Scope of Work

In the context of the topics outlined above, Polk County and FDEP contracted with PBS&J to conduct a study to determine the potential benefits to water quality in Lake Hancock from sediment removal activities. In addition, Polk County, the Southwest Florida Water Management District (District) and the FDEP are also interested in potential improvements in lake water quality within the context of the pending TMDL (Total Maximum Daily Loads) determinations for Lake Hancock and its watershed tributaries.

The overall project design involves the implementation of two concurrent field studies. The first study (Task 1) was a manipulative experiment to determine if variation in the levels of nutrient availability could bring about decreases in concentrations of blue-green algal vigor and/or abundance. The second study (Task 2) was another manipulative experiment designed to determine if the partial removal of the lake's sediments would likely result in significant improvements in water quality.

2.1. Task 1 - Manipulative Phytoplankton Response Study

On two occasions (December 2006 and July 2007) a series of experiments were carried out to measure rates of N_2- and CO_2-fixation in Lake Hancock. In December 2006, three sites were visited in Lake Hancock. However, in July 2007 the lake's water level was exceptionally low and the sampling and incubation equipment could not be realistically taken out to the center of the lake using the available boat. Instead, water samples were taken from a jetty on the west side of the lake where the incubation experiments were completed. This process used a high-sensitivity stable isotope tracer method to quantify rates in bottles suspended from the jetty (Montoya *et al.* 1996). The December 2006 experiment determined rates of N-fixation from the three locations throughout the lake (described in Task 2). The second experiment (May 2007) involved estimations of N-fixation rates within the lake, but also two additional experiments – a assessment of the role (if any) of light levels on rates of N-fixation, and an assessment of the role (if any) of levels of phosphorus availability on N-fixation rates.

Photo 1. Preparation of N_2 fixation incubation samples from Lake Hancock.

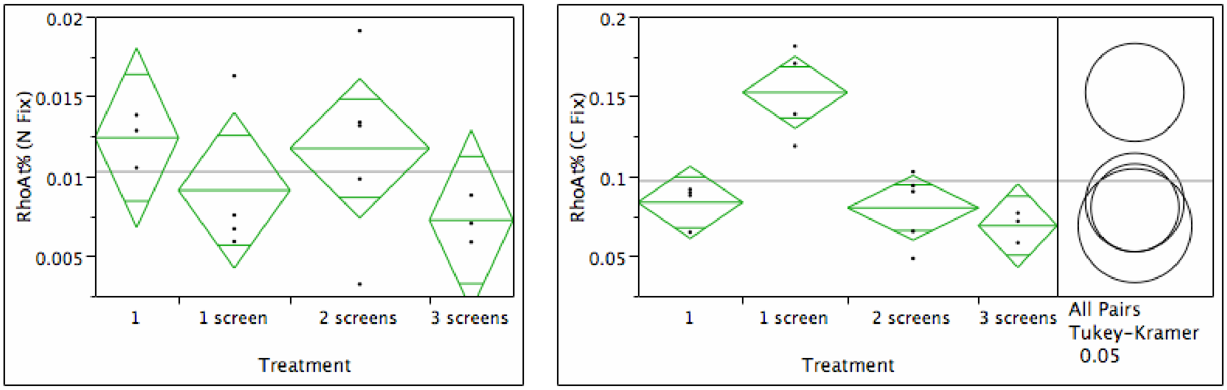
2.1.1. Baseline Rates from the December and July Experiments

Biomass in the lake was very high at the time of the July 2007 manipulative experiments. Baseline N_2-fixation rates of 16.2 ± 4.5 nmol N L^{-1} h^{-1} (mean ± SE, N=9) were comparable to rates measured in December 2006 but were lower than measured rates from March 2005 (133 ± 22 nmol N L^{-1} h^{-1}, N=7). The baseline CO_2-fixation rates were 130 ± 27 nmol C L^{-1} h^{-1} (mean ± SE, N=9). These rates are small relative to the very high biomass we observed (particulate nitrogen [PN] levels of 705 ± 34 µmol N L^{-1} and particulate carbon [PC] of 5100 ± 260 µmol C L^{-1} (mean ± SE, N=9).

The ratio of specific CO_2- and N_2-fixation rates was relatively low (V_C:V_N = 12.4 ± 3.5), implying that N_2-fixers made up a modest fraction (ca. 8 percent) of the photosynthetically active biomass at this time of year. Both instantaneous rates were very low relative to the biomass of the lake, perhaps reflecting the high temperatures and low lake levels in May 2007.

2.1.2. Effect of Light Intensity (July 2007)

An experiment was carried out during the July 2007 experiments to test the effect of light intensity on both N and C-fixation rates. The experiment used neutral density screening to vary the light intensity in bottles suspended at the surface of the lake. These treatments (0, 1, 2, and 3 screens) correspond roughly to 100 percent, 50 percent, 25 percent, and 12.5 percent of ambient irradiance. Results (Figure 2) indicate no significant effect of light intensity on N-fixation rates. In contrast, light intensity had a significant effect on CO_2-fixation rates (also Figure 2), with higher rates in the bottles exposed to 50 percent of ambient irradiance. The other three light treatments did not differ significantly.



Differences not significant

Level			Mean
1 screens	A		0.15242173
0 screens		B	0.08355443
2 screens		B	0.08017355
3 screens		B	0.06896480

Figure 2. Effect of light intensity on N- and CO_2-fixation rates in July 2007. The 1-screen (ca. 50 percent ambient light) treatment showed a significantly higher CO_2-fixation rate than the others). Results of a Tukey's HSD multiple comparisons test are shown below the CO_2-fixation results. The differences among the N_2-fixation rates were not significant

These results show clear evidence for photoinhibition (reduced photosynthetic rates with increased light levels) of primary production at surface irradiances. The CO_2-fixation rate in the ~50 percent irradiance treatment is roughly double the rate under full surface light. In contrast, results showed no evidence for light limitation of N-fixation.

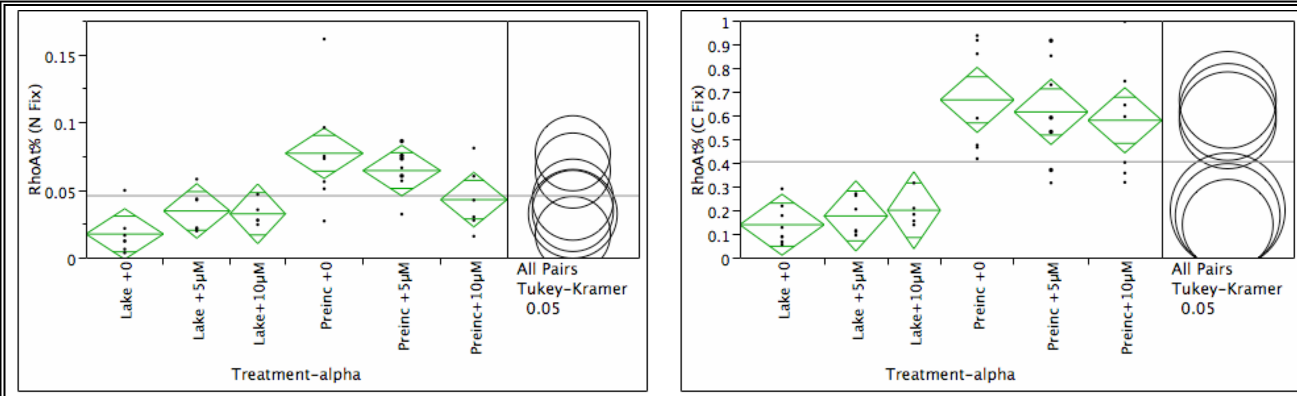
2.1.3. Effect of Phosphorus (July 2007)

An additional study was carried out during the July 2007 experiments to test the effect of P availability on metabolic rates on short (hour) and longer (day) time scales:

1. On 11 and 12 July, N- and CO2-fixation rates were measured in lake water after amending samples with 0, 5, and 10 µM PO43-.

2. On 10 July, large (4.5L) bottles were filled with lake water and amended with 0, 5, and 10 µM PO43-. These bottles were incubated at the surface of the lake through 12 July. N- and CO2-fixation rates in water from these bottles were measured on 11 and 12 July (i.e., after roughly 24 and 48 h of incubation).

The general expectation was that P amendments would increase the rate of N_2-fixation, and possibly CO_2-fixation. The data show a more complex response pattern that suggests experimental artifacts could affect the ability to discern P-enrichment effects (Fig. 3).



Level				Mean
Preinc +0	A			0.07700947
Preinc +5µM	A	B		0.06422441
Preinc+10µM	A	B	C	0.04275980
Lake +5µM		B	C	0.03442438
Lake+10µM		B	C	0.03230808
Lake +0			C	0.01737547

Level			Mean
Preinc +0	A		0.66535144
Preinc +5µM	A		0.61438396
Preinc+10µM	A		0.57914861
Lake+10µM		B	0.19948642
Lake +5µM		B	0.17551063
Lake +0		B	0.13862776

Figure 3. Effect of P amendment on N-fixation (left) and CO_2-fixation (right) rates in July 2007. Results of a Tukey's HSD multiple comparison are shown below the plots.

Major implications of the results of this experiment include the following:

1. CO_2-fixation rates showed no P effect but did show a significant difference between the experiments carried out with fresh lake water and the preincubated water. Although the differences were not significant, there was a decrease in CO_2-fixation rates with increasing PO_4^{3-} amendments in the preincubated samples. This suggests that the preincubations had an impact on biomass and perhaps on phytoplankton community composition. Measurements of PN and PC concentrations showed no significant differences between the fresh lake water and the preincubated water, however.

2. N_2-fixation rates were higher in the preincubated samples than in fresh lake water, but the multiple comparisons test showed overlap in the rates between the lake and preincubated samples. For lake samples, the two PO_4^{3-} amendments showed significantly higher rates of N-fixation than the unamended control, while the preincubated samples showed a **decrease** in N-fixation rates with increasing PO_4^{3-} amendments. This last result was surprising and suggests that significant changes occurred in the diazotroph (N-fixing) community in the preincubated samples.

2.1.4. Results from the N_2- and CO_2 Fixation Studies

Results from the manipulative studies show clear evidence that light intensity affects primary production but not N-fixation rates in Lake Hancock (at least in July 2007). There was significant photoinhibition (i.e., decreased rates with higher light intensities), suggesting that phytoplankton communities in Lake Hancock are currently adapted to low light levels that accompany the very low water clarity.

The results of the PO_4^{3-} amendments show evidence for an immediate increase in N-fixation upon exposure to PO_4^{3-}. Since responses to the two PO_4^{3-} levels were not significantly different, these data suggest that N-fixation rates are sensitive to relatively lower concentrations of PO_4^{3-}, and that rates of N-fixation are "saturated" at PO_4^{3-} concentration of 5 µM (equivalent to a TP concentration of 0.155 mg/liter). Longer incubations produced evidence of an inhibition of N-fixation by exposure to PO_4^{3-} for 1 to 2 d, which may reflect changes in community composition, perhaps including competition between diazotrophs (N-fixers) and other phytoplankton.

Additionally, the results of the N-fixation rates will be compared to output from nutrient loading models produced by ERD and FDEP below, to examine the ability of in-situ N-fixation to account for "missing" levels of nitrogen within Lake Hancock.

2.2. Task 2 - Manipulative Sediment-Water Quality Response Study

For the second study, an experimental design similar to that used to assess the value of sediment removal strategies for Lake Maggiore (in St. Petersburg) was conducted, as described below.

2.2.1. Cylinder Preparation and Deployment

Three locations were selected within Lake Hancock (North- LH1, Central-LH2 and South-LH3) to allow for heterogeneity in water and sediment depth. At each location, the water and sediment depth were measured using a clear 2" diameter acrylic tube to determine the dimensions of the mesocosms. Large 6' diameter aluminum mesocosms were assembled from two sections. The bottom portion ranged from 3' to 3.5' tall, depending on sediment depth. This section of the mesocosm was designed to prevent exchange with the surrounding sediment matrix. The cylinder had an open top and bottom with a solid wall. The cylinder was pushed through the organic sediment layer into the sandy bottom of the lake to ensure complete isolation. The top portion ranged from 5' to 5.5' tall, depending on water depth. This section isolated the water column and was comprised of an aluminum frame covered with a double layer of reinforced nylon 10 millimeter clear polyvinyl material. The construction of these cylinders resulted in the isolation of the water column within the cylinders while still allowing for natural wave action and light penetration. The maximum height of each mesocosm was 8.5' (Photo 2). The experiment was completed twice, during December 2006 and May 2007. Two cylinders were deployed at each location on November 30, 2006 and May 2, 2007 (Photo 3). The cylinders were removed between the December and May sampling periods to reduce the influence of the cylinder affect on water quality. One cylinder at each location was randomly selected for dredging and subsequently dredged on December 1, 2006 or May 3, 2007, using a Mud Hog pump (Photo 4). Reference sites were established in close proximity to each of the three pairs of dredged and undredged mesocosms (Figure 4). Prior to cylinder removal, three replicate cores were taken at each site, the sediment and water depths were recorded.

Photo 2. Constructed aluminum mesocosm about to be deployed in Lake Hancock.

Photo 3. Example of deployed mesocosm.

Photo 4. At each site in Lake Hancock, one mesocosm was randomly selected to be dredged using a MudHog pump.

Figure 4. Locations of the three paired locations for cylinder mesocosms.

2.2.2. Water Quality Sampling

For both the December and May experiments, water samples and water quality parameters were collected at each of the nine sites on four sampling dates (Table 2). A suite of physical, chemical and biological parameters were measured (Table 3). The water quality laboratory of the Polk County-Natural Resources Division analyzed all parameters but biological oxygen demand (BOD) which was analyzed by PhosLab Co. in Lakeland, Florida. All samples were collected 20cm below the water surface within each cylinder and at each reference site. Additionally, a profile of the water column was completed using a HydroLab. Temperature, conductivity, pH and Dissolved Oxygen were recorded at 20 cm intervals. On the first and third sampling dates, water samples were collected without manipulation of the water column. On the second and fourth sampling dates, a paddle was used to mix the isolated water column to simulate windy day conditions that regularly occur on the lake. Initially, the undredged cylinder at each location was "stirred" to determine the amount of effort necessary to reach high turbidity conditions. A preliminary and post-stirring turbidity reading were taken to establish the increase in turbidity due to mixing, using a Hach 2100P Turbidimeter. The same amount of effort (i.e., seconds spent stirring the water column) was then used at the dredged and reference sites at each location. Water quality samples and water quality profile readings were taken after the mixing was completed.

Table 2. Sampling Days for the mesocosm experiments.

Condition	Sampling Dates	
Not-Mixed	December 6th, 2006	May 7th, 2007
Mixed	December 8th, 2006	May 9th, 2007
Not-Mixed	December 11th, 2006	May 14th, 2007
Mixed	December 13th, 2006	May 16th, 2007

Table 3. Water Quality Parameters measured for each sample.

Parameter	Unit	Method
Physical		
pH		HydroLab
Temperature	°C	HydroLab
Dissolved Oxygen (DO)	mg/liter	HydroLab
Specific Conductivity	uhoms	HydroLab
Secchi Depth	cm	Secchi Disk/ measuring tape
Turbidity (Field)	NTU	Hach 2100P Turbidimeter
Turbidity (Lab)	NTU	2130 B
Color	Pt-Co	2120 B
Total Suspended Solids (TSS)	mg/liter	2540 D
Chemical		
Total Nitrogen (TN)	mg/liter	351.2 & 4500-NO3F
Total Kjeldahl (TKN)	mg/liter	351.2
Total Phosphorus (TP)	mg/liter	EPA365.4
Nitrate+Nitrite (NOx)	mg/liter	4500-NO3F
Ortho-Phosphate (SRP)	mg/liter	4500 P-F
Biological		
Chlorophyll *a*-corrected (Chl a)	µg/liter	102000 H
Biological Oxygen Demand (BOD)	mg/liter	SM 5210B

2.2.3. Data Analysis

Data Comparison

PBS&J completed a comparison between the period of record data (n=209) available from the Polk County Water Atlas website (http://www.polk.wateratlas.usf.edu/) and the mesocosm experiment (n=72) completed by PBS&J. The mesocosm experiment data included the duplicate samples taken to satisfy QA/QC requirements for the laboratories and field collection protocol. An additional sample was collected on each of the sampling dates (except on December 6th, 2006). The earliest data point included within our analysis for the period of record was in 1967, the most recent in 2007. Not all parameters are available over the same time period. The objective of the data comparison was to determine if the values measured during the mesocosm experiment were comparable to ambient samples collected within the Lake.

Turbidity Data Analysis

PBS&J analyzed nutrient data based on simultaneous measurements of turbidity for both the period of record and the mesocosm experiment time period. The Florida Department of Environment Protection defines the threshold for an increase in turbidity based on water quality standard as "no more than a 29 NTU increase from background". The average turbidity (background) of the historical data was calculated (33 NTU). In Lake Hancock, the threshold was calculated to be 62 NTU (i.e., 33+29=62). Therefore, any turbidity measurement recorded greater than 62 could be classified as "impaired". The turbidity measurements for the period of record were sorted into values < 62 and values ≥ 62 (n=205 and n=22, respectively). A linear regression was determined for each nutrient parameter based on turbidity for each data set (impaired, not impaired and experiment) to determine if a statistically significant relationship existed ($p<0.05$).

Mesocosm Data Analysis

All data collected during the mesocosm experiment were analyzed to determine if any statistically significant differences could be detected between treatments ($p<0.05$). Using StatGraphics™ Centurion Ver. XV, the data were analyzed to compare "Not-Mixed" to "Mixed" conditions for each treatment type (Reference, Dredged and Undredged).

Dissolved Oxygen 5-day Experiment

A dissolved oxygen (DO) experiment was designed to compare oxygen depletion of water discharged out of Lake Hancock into South Saddle Creek. The initial DO concentration was measured at the Saddle Creek location from the structure on July 16, 2007 using a YSI water quality meter. Water was collected in four amber 500ml glass jars and immediately covered in aluminum foil to eliminate light contamination. Additionally, water was collected and agitated until super saturation was reached based on measurements of DO. A second set of four amber 500ml glass jars was filled with the supersaturated oxygenated water and covered in aluminum foil. All eight DO experiment bottles were transported to the Tampa PBS&J office. DO was measured twice a day at 8am and 4 pm until Thursday, July 19, 2007 using a YSI. Additional measurements were made Friday afternoon (July 20th) and the morning of Monday, July 23rd.

TP Model

Resuspension of sediments is a common internal source of additional TP to lake systems. Multiple factors were identified for Lake Hancock which could facilitate the resuspension of sediments, such as wind speed, wind direction, rainfall and lake stage. Daily wind speed, wind direction and rainfall data were downloaded for Lake Alfred from the UF/ IFAS Florida Automated Weather Network website (http://fawn.ifas.ufl.edu). Lake Alfred is located northwest of Lake Hancock in Polk County. The daily stage from USGS gauge FLO-782 was obtained from the Polk County Wateratlas website (www.polk.wateratlas.usf.edu). Stage values ranged from 95.5 to 101.5 ft. Data from January 1, 1998 to July 24, 2007 were available for all parameters. Data for these multiple external factors were loaded into a multiple regression model to determine if a statistically significant relationship was evident between TP concentrations as the dependent variable, and wind speed, wind direction, rainfall and lake stage as potentially significant independent variables ($p<0.05$). A one, two and three day lag was applied to both wind speed and wind direction and during a separate analysis, rainfall data, to determine if a delayed effect was apparent. Additionally, cumulative rainfalls for two, three and four day periods were explored to identify a potentially more robust model fit. The resulting model indicated that daily wind speed, daily rainfall and daily stage explained 29.3 percent of the variation of TP in Lake Hancock. The equation is below:

$$TP = 5.93133 + (0.633035*rainfall) + (0.062019*\ windspeed) - (0.0581623*\ stage)$$

Where TP is in mg/l, rainfall is in inches, windspeed is in mph and stage is in decimal feet.

*Note: each independent variable was measured in a different unit and has a different range of values.

Using the above equation, lake stage alone was modified based upon several lake level manipulation scenarios, including a proposed action by the District to increase the seasonal high water level of Lake Hancock by 1.8 feet, to allow for increased wet weather storage for recovering minimum flows in the Upper Peace River (Table 4). Unaltered wind speed and rainfall data were used in combination with the altered stage data to calculate a modeled TP concentration for Lake Hancock based upon the following scenarios:

Table 4. Changes in effective water depths (feet) with different sediment removal and lake level modification scenarios.

Scenario	Condition	Change in Effective Water Depth (ft)
1	Current	NA
2	Sediment Removal (1 ft)	1.0
3	Proposed Lake Level Modification (1.8 ft)	1.8
4	Sediment Removal (2 ft)	2.0
5	Proposed Lake Level Modification (1.8 ft) and Sediment Removal (1 ft)	2.8
6	Sediment Removal (3 ft)	3.0
7	Proposed Lake Level Modification (1.8 ft) and Sediment Removal (2 ft)	3.8
8	Proposed Lake Level Modification (1.8 ft) and Sediment Removal (3 ft)	4.8

2.2.4. Results of Sediment Removal Study

Mesocosm Profile

Three cores were completed at each location to measure the water and sediment profile during both December and May sampling. The average core profile for each site is displayed in **Figures 5 and 6**. During December, sediment in the dredged cylinders was significantly less than the reference and undredged cylinders. There was a 35-52 cm change in sediment depth due to dredging compared to the undredged cylinders. Additionally, a 46 to 86 percent increase in water volume in the dredged cylinders compared to the undredged cylinders and reference stations was calculated. The total volume of water within the cylinders ranges from a minimum of 2,100 gallons (undredged) to a maximum of 5,100 gallons (dredged). In comparison, the volume of sediment ranged from a minimum of 300 gallons (dredged) to a maximum of 2,000 gallons (undredged). In the May, sediment in the dredged cylinders was significantly less than the reference and undredged cylinders. There was a 31-50 cm change in sediment depth due to dredging compared to the undredged cylinders. Additionally, a 79 to 151 percent increase in water volume in the dredged cylinders compared to the undredged cylinders and reference stations was calculated. The total volume of water within the cylinders ranges from a minimum of 940 gallons (undredged) to a maximum of 3,400 gallons (dredged). In comparison, the volume of sediment ranged from a minimum of 610 gallons (dredged) to a maximum 2,300 gallons (undredged).

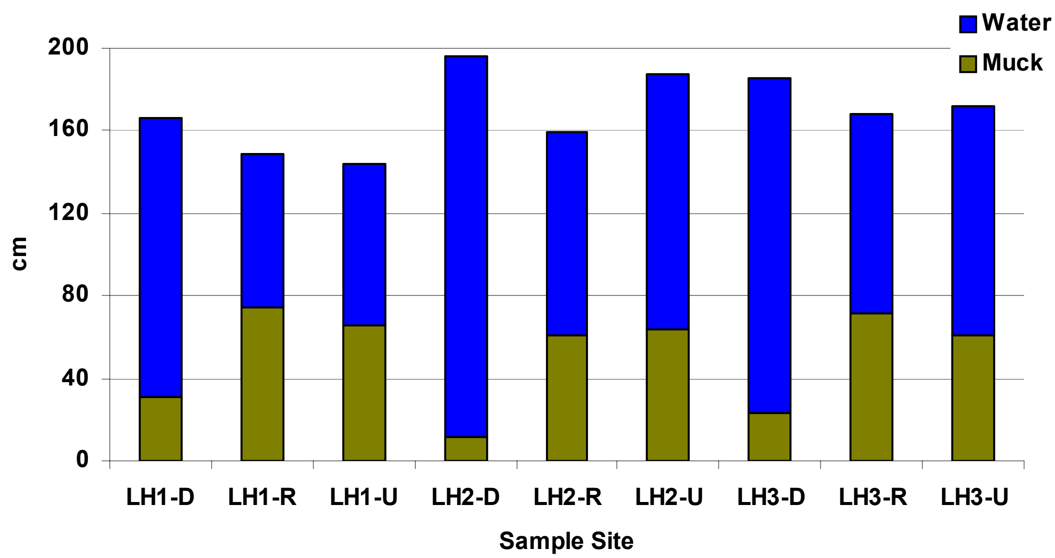


Figure 5. December 2006 average muck and water depth at each site.

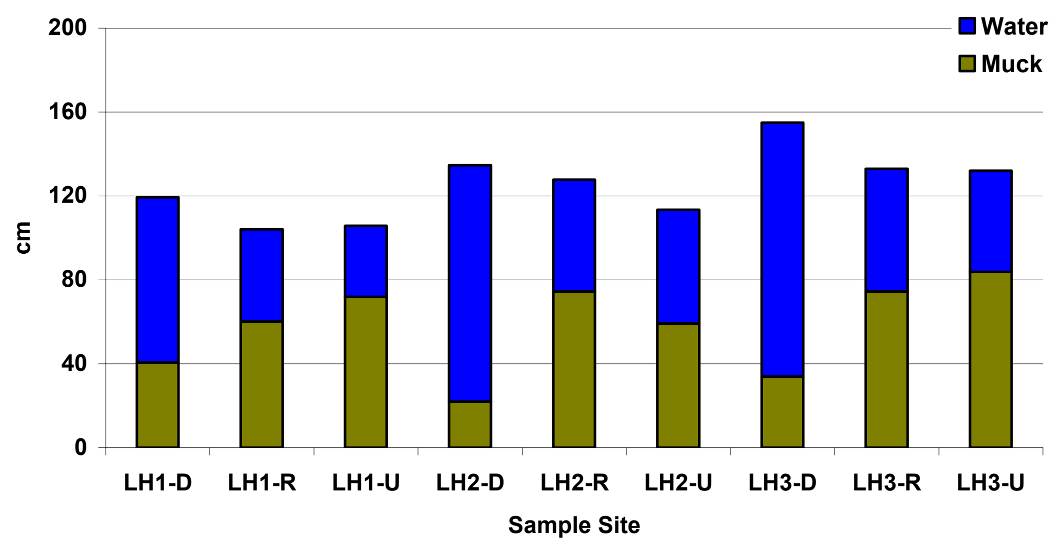


Figure 6. May 2007 average muck and water depth at each site.

Historical Data Comparison

Several parameters collected by Polk County Natural Resources Division at Lake Hancock were compared to data collected during the sediment removal mesocosm experiment. All figures for the data comparison are located in **Appendix A**. For all comparisons, the data collected during the mesocosm were within range of the historical dataset. The historical comparison confirms the installation and use of the aluminum cylinders for the experiment did not create an environment so different from natural conditions that data analysis would be of no value. In other terms, if the experimental data had resulted in a dramatic increase or decrease for a particular parameter it would be difficult to distinguish between a potential treatment effect (i.e., dredged vs. non-dredged comparisons) versus the effect of cylinder placement itself.

Turbidity Data Analysis

The turbidity values measured by the Polk County Natural Resources Lab were used for all analysis. A strong linear relationship (r^2=.4053) was observed in Lake Hancock between Total Suspended Solids (TSS) and Turbidity (**Figure 7**).

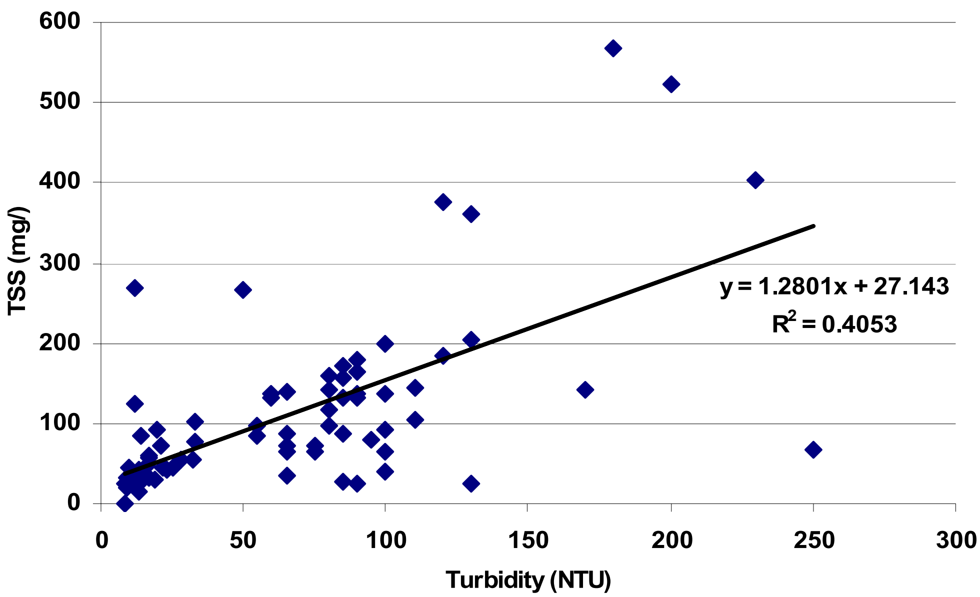


Figure 7. Linear Relationship between Lake Hancock TSS and Turbidity.

The average turbidity values for each treatment, for both December and May, are shown in **Figures 8 and 9,** respectively. In December, turbidity values in the unmixed cylinders were reduced by 41 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, turbidity values in December were 75 percent lower in dredged vs. undredged cylinders. In May, turbidity values in the unmixed cylinders were reduced by 14 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, turbidity values in May were 60 percent lower in dredged vs. undredged cylinders.

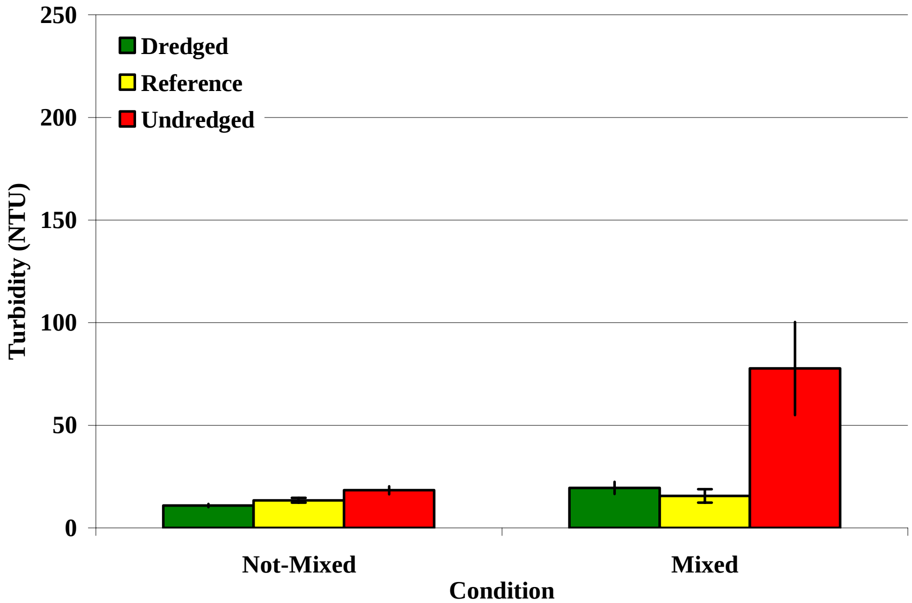


Figure 8. December 2006 average turbidity results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

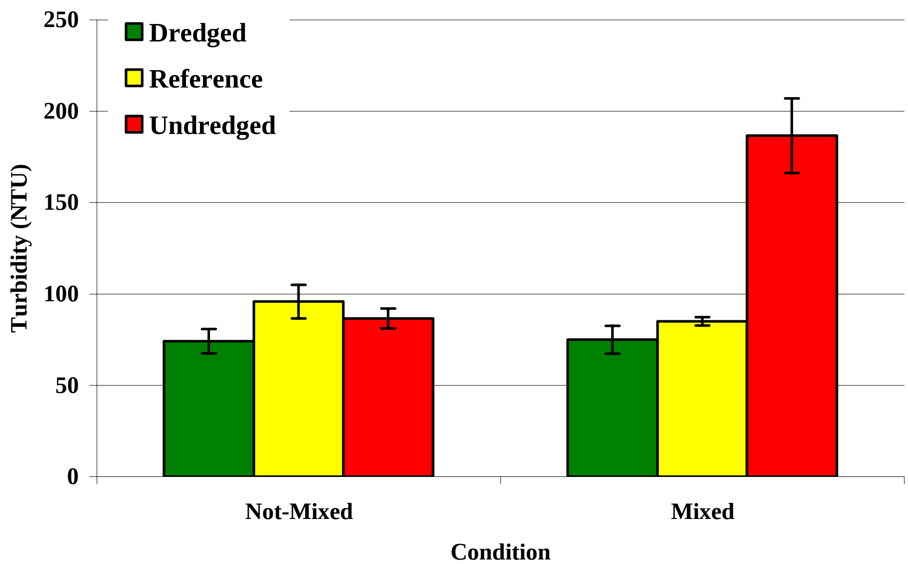


Figure 9. May 2007 average turbidity results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

In Lake Hancock, there is a strong statistically significant relationship between levels of TP and turbidity **(Figure 10)**. However, correlative analyses do not allow for the determination of a cause and effect relationship. That is, based on this data set alone, it would not be possible to determine whether high levels of turbidity were caused by high concentrations of TP, or whether high concentrations of TP were caused by high turbidity. Therefore, the value of the "mixing" experiment was that it would shed further light on the relationship between these two variables. As shown in **Figures 11 and 12**, it appears that the act of mixing the water column so as to increase the turbidity in the water column is sufficient to increase the concentration of TP in the water. These results would indicate that a significant pool of TP in Lake Hancock is in the sediments, which can be resuspended into the water column upon mixing, thus increasing the levels of TP in the water as well (see below).

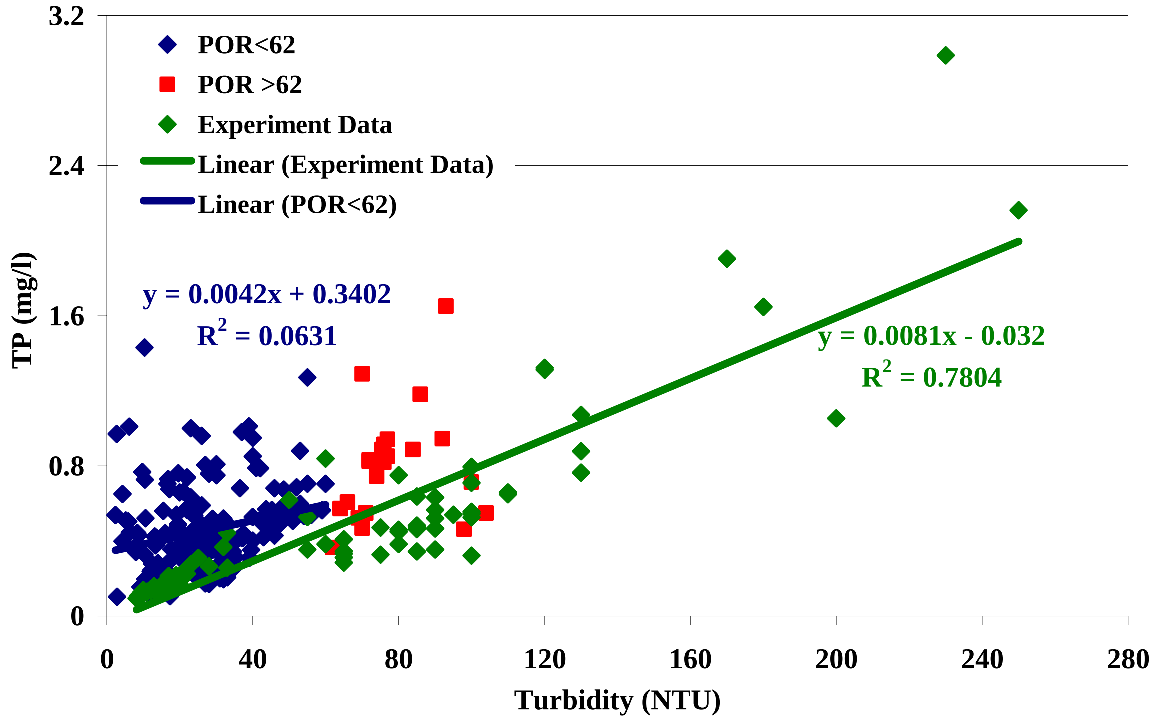


Figure 10. Linear trend relationships between Total phosphorus data under below and above FDEP maximum turbidity values (turbidity=62) and experimental conditions.

The average TP concentrations for each treatment, for both December and May, are shown in **Figures 11 and 12,** respectively. In December, TP concentrations in the unmixed cylinders were reduced by 37 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, TP concentrations in December were 77 percent lower in dredged vs. undredged cylinders. In May, TP concentrations in the unmixed cylinders were reduced by 21 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, TP concentrations in May were 69 percent lower in dredged vs. undredged cylinders.

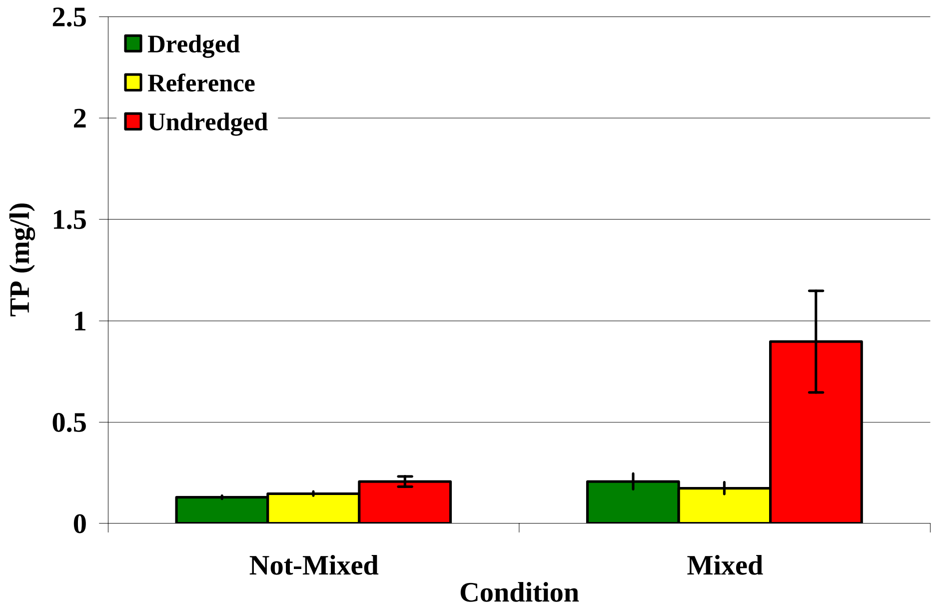


Figure 11. December 2006 average total phosphorus results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

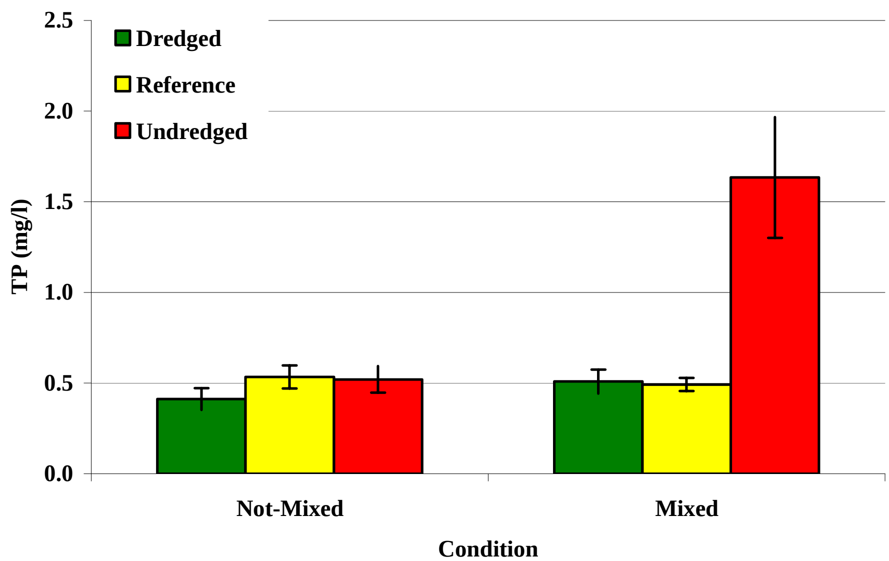


Figure 12. May 2007 average total phosphorus results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

As was the case with TP, there is a strong statistically significant relationship between levels of TN and turbidity in Lake Hancock **(Figure 13)**. However, correlative analyses do not allow for the determination of a cause and effect relationship. As shown in **Figures 14 and 15**, it appears that the act of mixing the water column so as to increase the turbidity in the water column is sufficient to increase the concentration of TN as well.

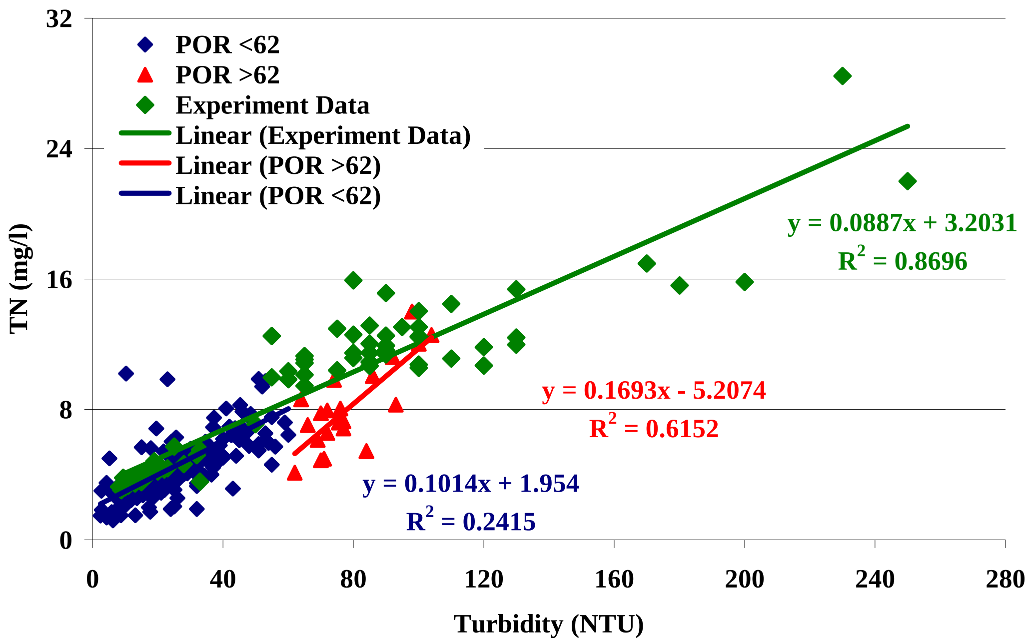


Figure 13. Linear trend relationships between Total nitrogen data under below and above FDEP maximum turbidity values (turbidity=62) and experimental conditions.

The average TN concentrations for each treatment, for both December and May, are shown in **Figures 14 and 15,** respectively. In December, TN concentrations in the unmixed cylinders were reduced by 20 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, TN concentrations in December were 53 percent lower in dredged vs. undredged cylinders. In May, TN concentrations in the unmixed cylinders were reduced by 13 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, TN concentrations in May were 33 percent lower in dredged vs. undredged cylinders.

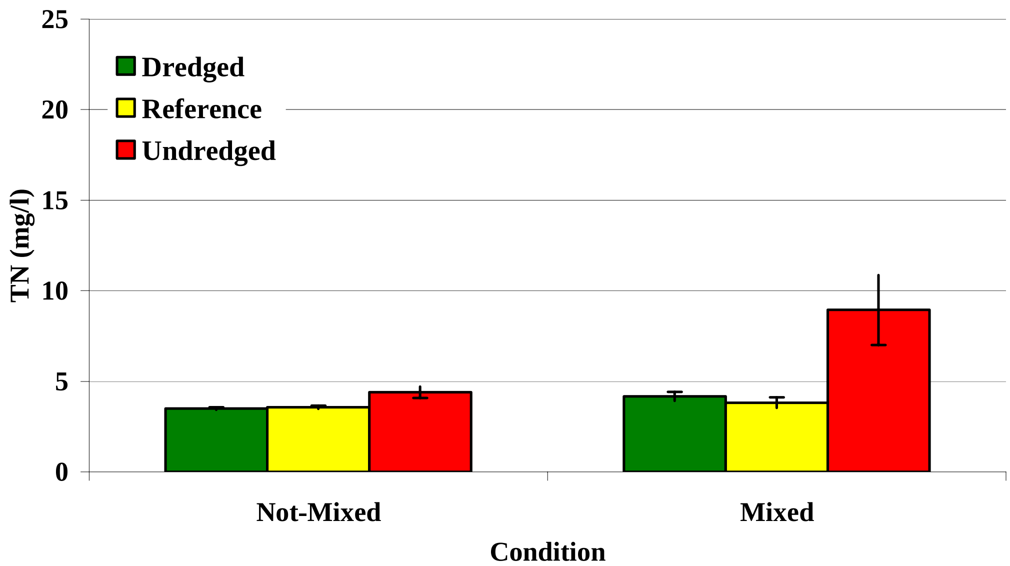


Figure 14. December 2006 average total nitrogen results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

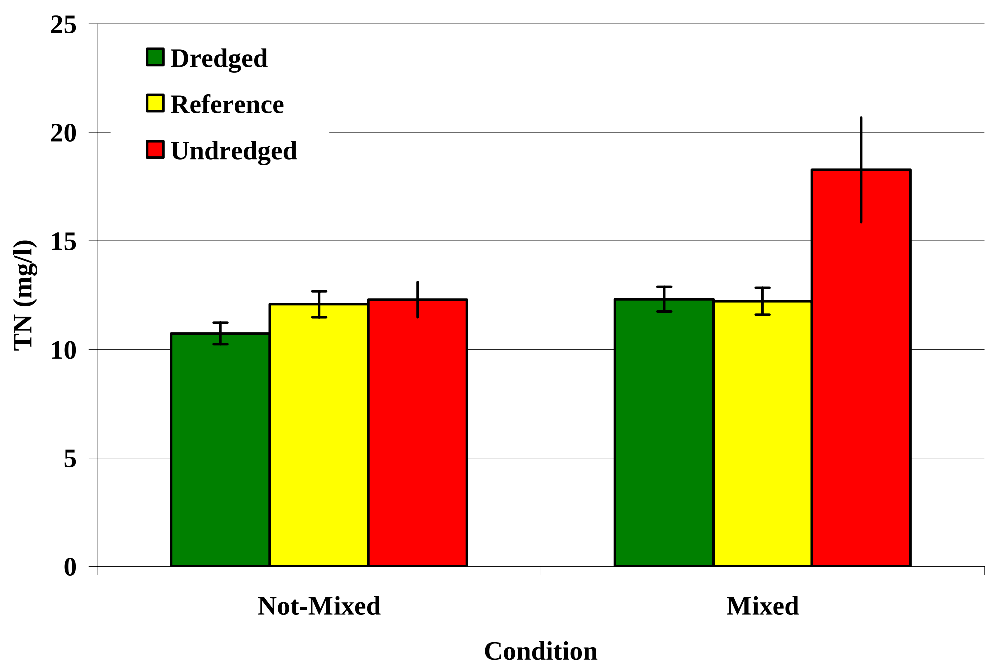


Figure 15. May 2007 average total nitrogen results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

The average TN:TP ratios for each treatment, for both December and May, are shown in **Figures 16 and 17,** respectively. In December, TN:TP ratios in the unmixed cylinders were increased by 26 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, TN:TP ratios in December were 101 percent higher in dredged vs. undredged cylinders. In May, TN:TP ratios in the unmixed cylinders were increased by 11 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, TN:TP ratios in May were 111 percent higher in dredged vs. undredged cylinders. During both December and May sampling events, TN:TP ratios in undredged and mixed cylinders dropped below the "Redfield Ratio" (by weight) of 16, a value thought to represent a threshold below which conditions would be favorable for the development of phytoplankton blooms for species able to access inorganic forms of nitrogen (i.e., conditions favorable for blue-green algae; edfield 1958).

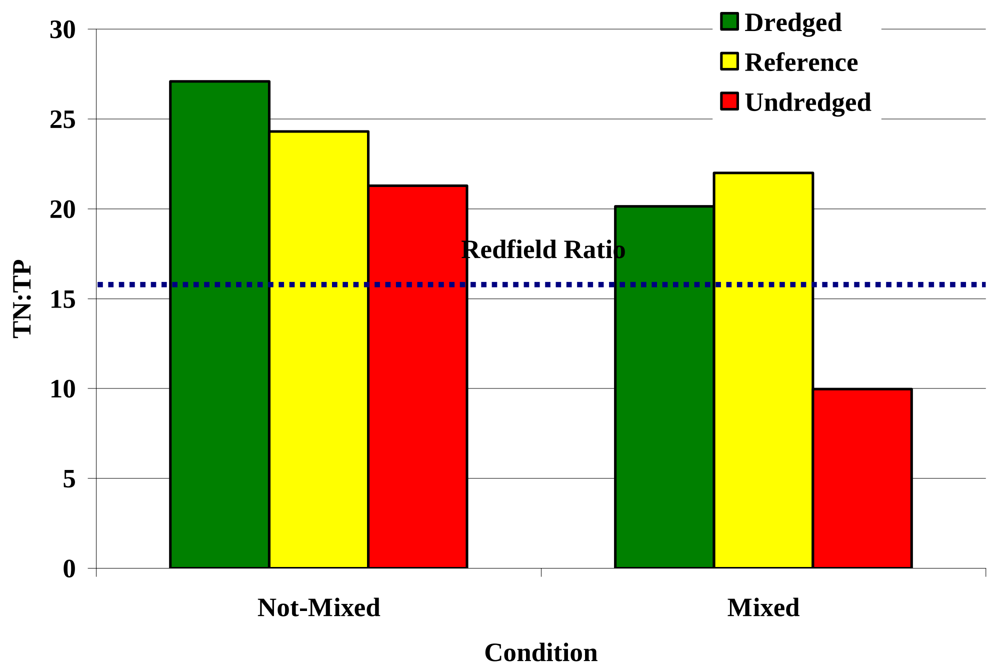


Figure 16. December 2006 average total nitrogen:total phosphorus results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

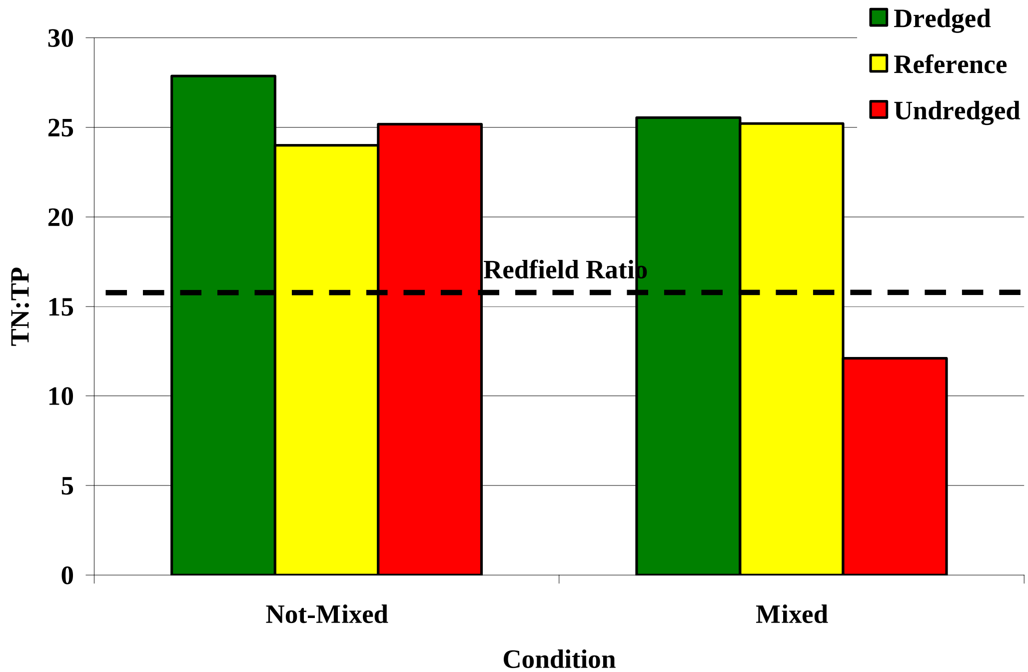


Figure 17. May 2007 average total nitrogen:total phosphorus results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

In Lake Hancock, there is a strong statistically significant relationship between chlorophyll *a* concentrations and turbidity **(Figure 18)**. As shown in **Figures 19 and 20**, it appears that the act of mixing the water column so as to increase the turbidity in the water column is sufficient to increase chlorophyll *a* concentrations in the water, although the magnitude of the increase with mixing is much less than what was seen with responses of TN and TP.

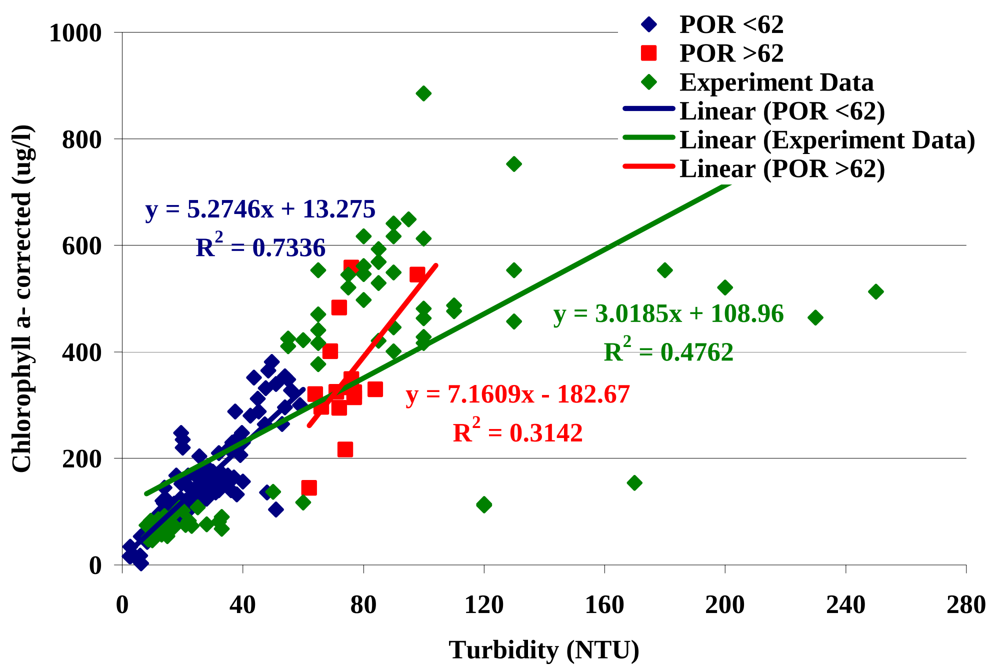


Figure 18. Linear trend relationships between chlorophyll *a* (corrected) data below and above FDEP impairment level turbidity values (turbidity=62) and experimental conditions.

The average chlorophyll *a* concentration for each treatment, for both December and May, are shown in **Figures 19 and 20,** respectively. In December, chlorophyll *a* concentrations in the unmixed cylinders were reduced by 21 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, chlorophyll *a* concentrations in December were 31 percent lower in dredged vs. undredged cylinders. In May, chlorophyll *a* concentrations in the unmixed cylinders were reduced by 5 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, chlorophyll *a* concentrations in May were 13 percent lower in dredged vs. undredged cylinders.

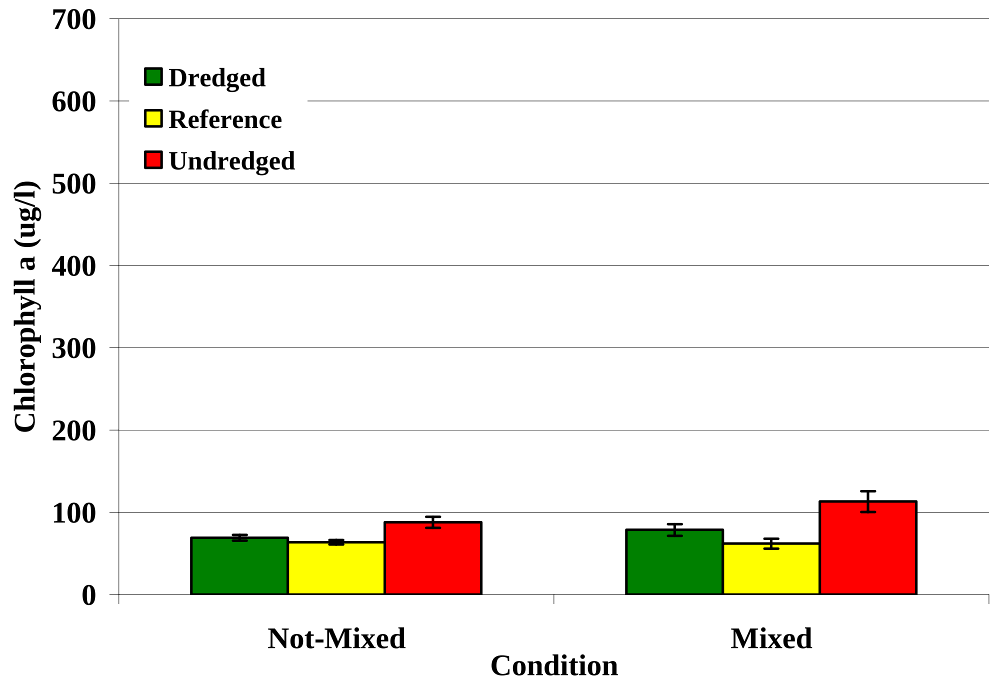


Figure 19. December 2006 average chlorophyll *a* results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

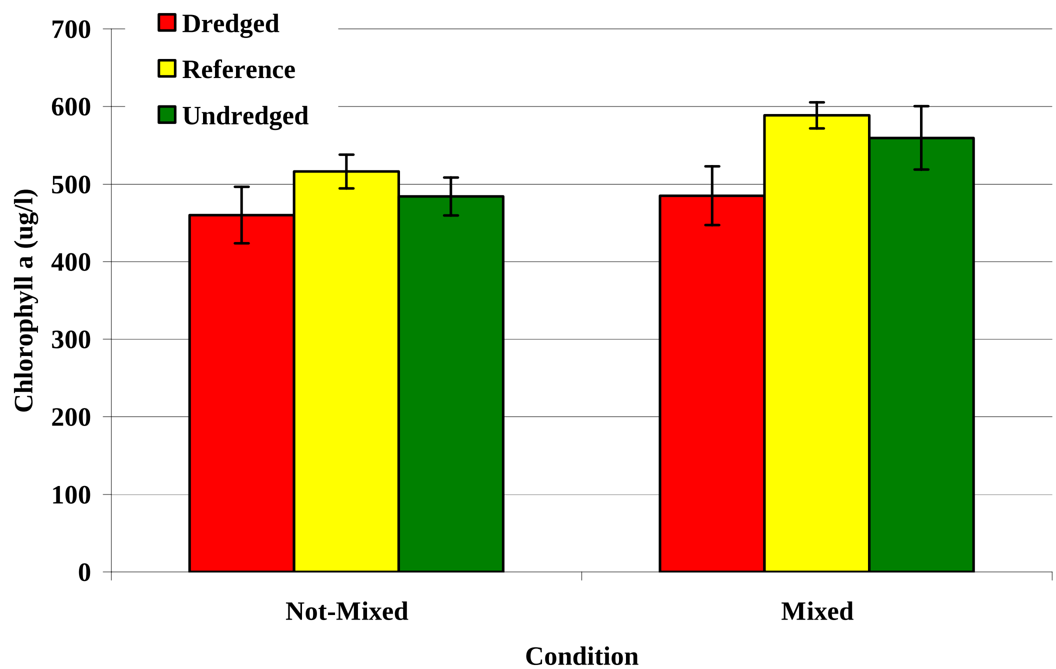


Figure 20. May 2007 average chlorophyll *a* results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

The average TSI value for each treatment, for both December and May, are shown in **Figures 21 and 22,** respectively. In December, TSI values in the unmixed cylinders were reduced by 4 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, TSI values in December were 12 percent lower in dredged vs. undredged cylinders. In May, TSI values in the unmixed cylinders were reduced by 1 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, TSI values in May were 7 percent lower in dredged vs. undredged cylinders.

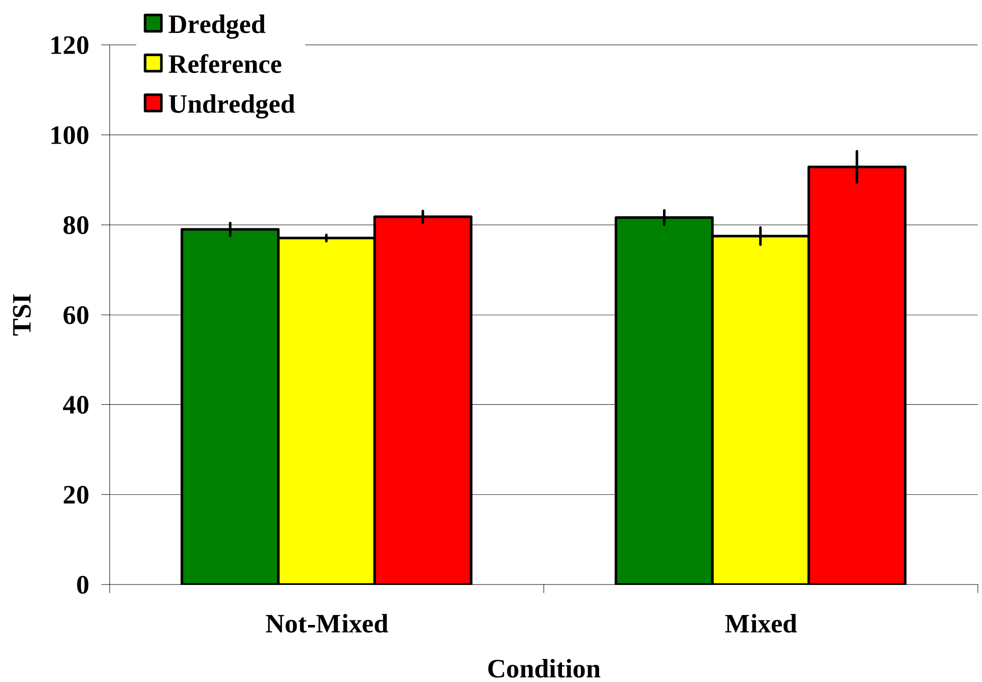


Figure 21. December 2006 average TSI results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

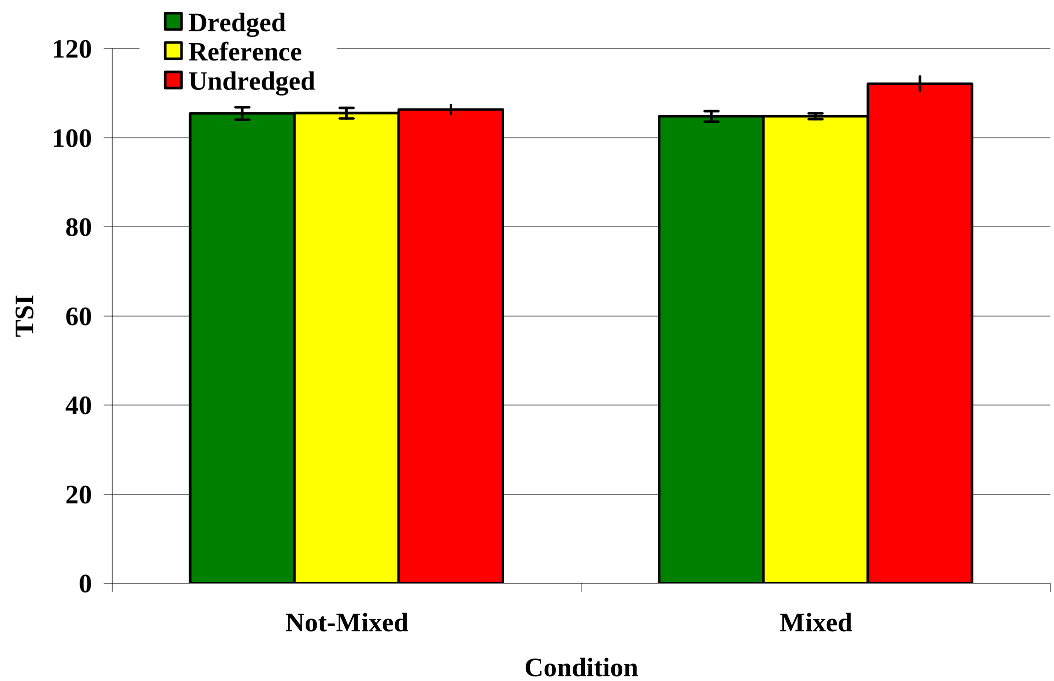


Figure 22. May 2007 average TSI results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

By comparing changes in the concentrations of TN, TP, and TSS in unmixed cylinders, it was possible to estimate a "flux rate" for each parameter not associated with physical resuspension of sediments. These flux rate estimates might represent rates of re-mineralization of TN and TP, and potential recycling and/or other process for TSS.

Flux rate estimates for TN, TP and TSS are shown in **Figure 23**. These results suggest that during the December sampling event, significant amounts of TP might be remineralized back into the water column. As such, a mechanism in addition to resuspension of bottom sediments could be associated with increasing TP availability into the water column. Flux rates for TN and TSS were much lower for the December event. During the May event, no evidence was found for fluxes of TN, TP or TSS into the water column other than the previously described mechanisms associated with resuspension of bottom sediments.

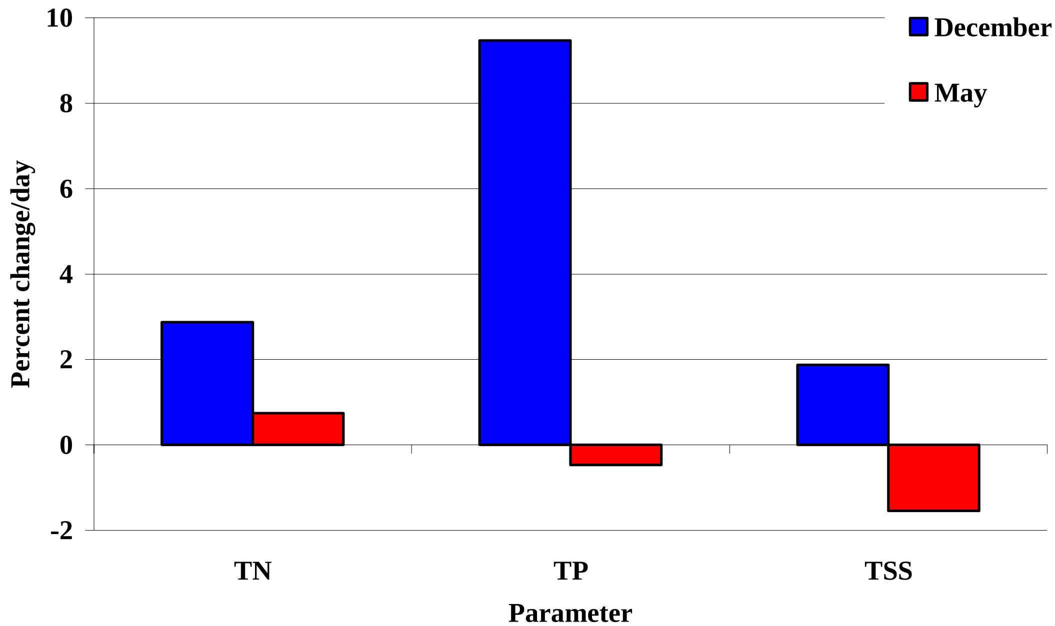


Figure 23. Differential flux rates of TN, TP, and TSS in the undredged, unmixed cylinders from the sediment to the water column.

Dissolved Oxygen (DO)

All Hydrolab Profiles (DO, pH, temperature and conductivity are in Appendix B), with the following text and graphics were used to summarize the relevance of experimental results, as regards levels of DO in both Lake Hancock and the Upper Peace River. As noted in Tomasko *et al.* (2006), the Upper Peace River is characterized by regularly occurring incidents of low levels of DO. For the Peace River at Bartow (see **Figure 24**) levels of DO not only fall below the State of Florida's minimum DO standard of 4 mg/liter, they often fall well below 2 and even 1 mg/liter of DO. For the rest of the Peace River, only after the passage of an event such as Hurricane Charley do DO levels fall to those regularly seen at the Bartow gage site (Tomasko *et al.* 2006).

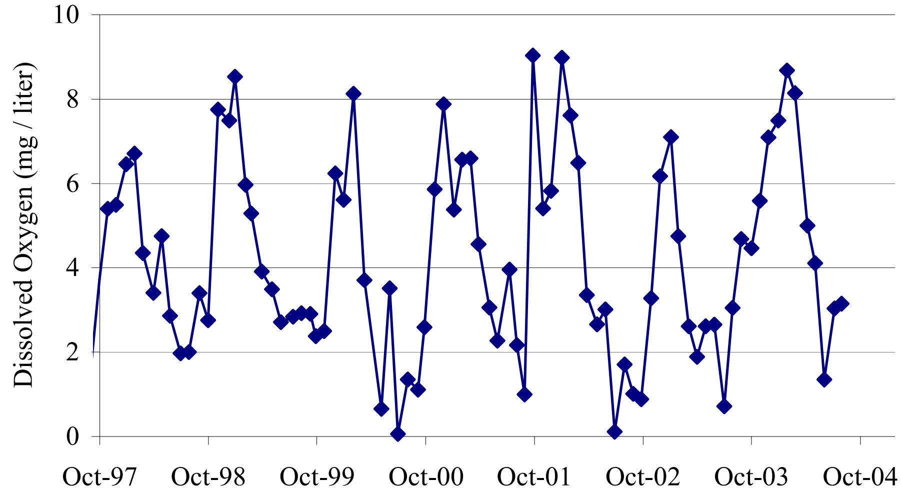


Figure 24. Dissolved Oxygen at the Peace River at Bartow sampling site.

When comparing DO levels at Bartow versus flows discharged out of Lake Hancock, the overall relationship is one wherein increased flows out of Lake Hancock are correlated with reductions in DO in the Peace River (**Figure 25**). These results suggest that a statistically significant impact on DO levels in the Peace River is the poor quality of water discharged from Lake Hancock through the P-11 structure on South Saddle Creek. Depressed levels of DO in downstream waters could be due to two phenomena – low DO levels in discharged water, or high levels of biological oxygen demand (BOD) of discharged waters.

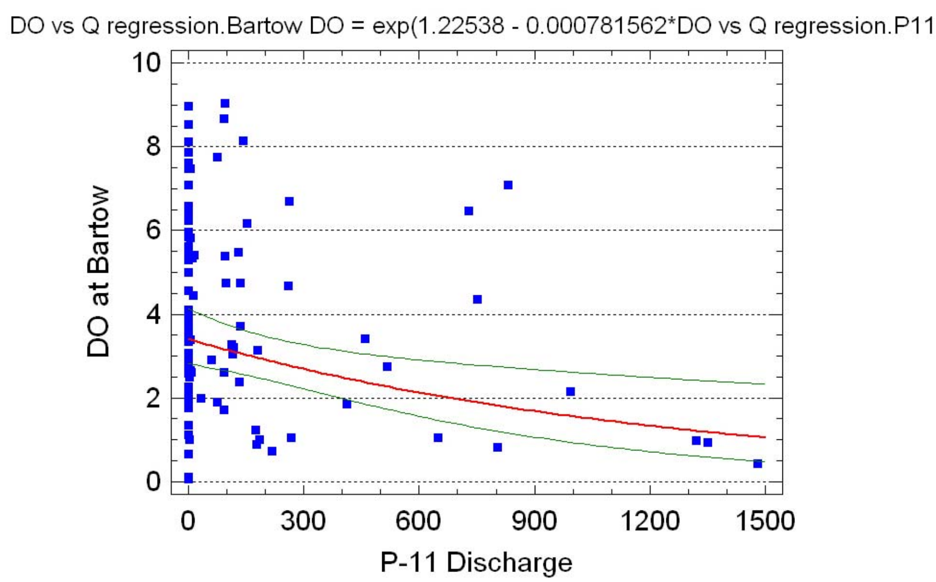


Figure 25. A significant regression of the Discharge from the P-11 structure and Dissolved oxygen on the Peace River at Bartow exists.

The average BOD value for each treatment, for both December and May, are shown in **Figures 26 and 27,** respectively. In December, BOD values in the unmixed cylinders were reduced by 13 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, BOD values in December were 13 percent lower in dredged vs. undredged cylinders. In May, BOD values in the unmixed cylinders were higher by 3 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, BOD values in May were 6 percent lower in dredged vs. undredged cylinders.

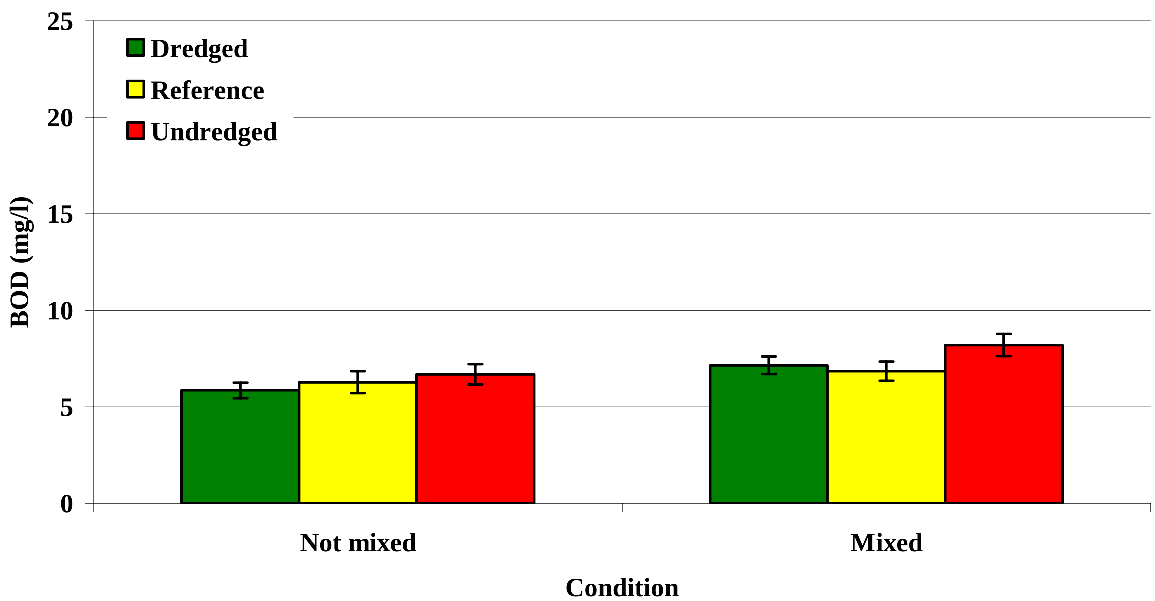


Figure 26. December 2006 average BOD results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

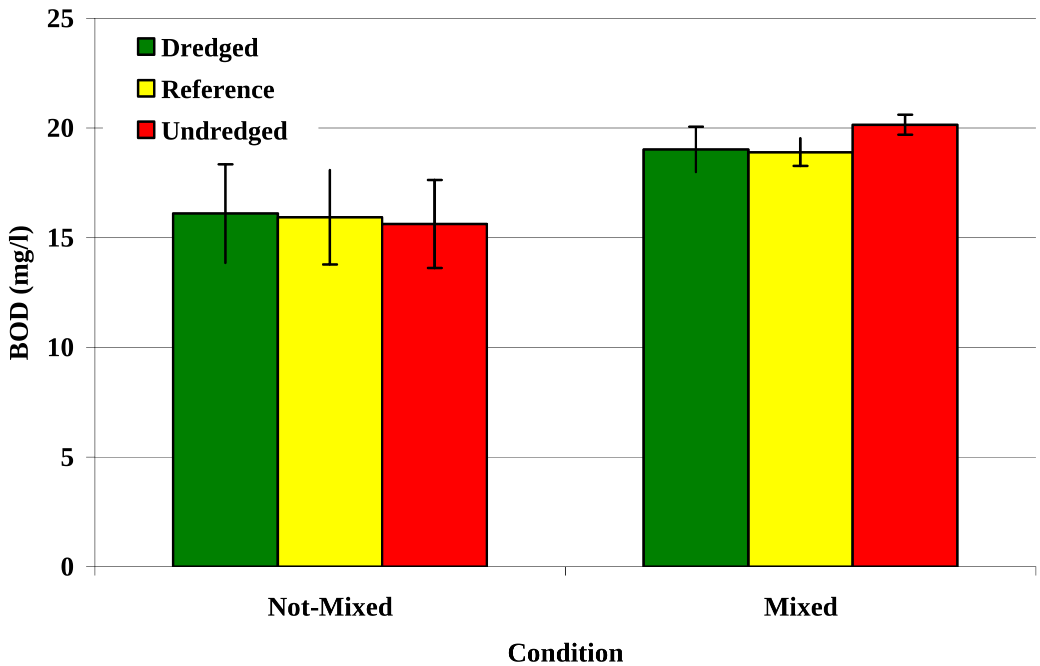


Figure 27. May 2007 average BOD results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

Table 5 summarizes the effects of a comparison of dredged versus undredged cylinders, for both mixed and non-mixed treatments, for both December and May. Levels of TN were reduced between 13 and 53 percent with sediment removal, while levels of TP were reduced between 21 and 77 percent. TSS was reduced between 35 and 78 percent, with turbidity down by 14 to 75 percent. Chlorophyll *a* concentrations were reduced, comparing dredged versus undredged cylinders, by 5 to 31 percent. BOD was reduced by 13 percent for December samples, but showed either a slight increase (not-mixed) or a small decrease (6 percent) in May. Ratios of TN to TP increased between 26 and 111 percent, indicating conditions would tend toward those less favorable for blue-green algae, which may benefit from low TN:TP ratios. However, TSI values exhibited a lesser degree of change, compared to both nutrient and chlorophyll *a* concentrations, with reductions of 1 to 12 percent found, when comparing dredged to undredged cylinders.

Table 5. Percent Change of Dredged Versus Undredged Cylinders.

	December 2006		May 2007	
	Not-Mixed	Mixed	Not-Mixed	Mixed
Chl a (corrected)	-21	-31	-5	-13
TN	-20	-53	-13	-33
TP	-37	-77	-21	-69
TSS	-35	-72	-42	-78
Turbidity	-41	-75	-14	-60
TN:TP	26	101	11	111
BOD	-13	-13	3	-6
TSI	-4	-12	-1	-7

When considering the inverse relationship between DO levels for the Peace River at Bartow and flows out of Lake Hancock (see Figure 25), this relationship could be derived from either low levels of DO in discharged water, or high BOD levels of discharged water. Levels of DO within Lake Hancock itself have been shown to vary between less than 1 mg/liter and higher than 10 mg/liter within the course of a single day, at a single location (Parson 2005). Such extreme variation in DO levels is to be expected in a hypereutrophic lake, where rapid rises in DO would occur when irradiance levels allow for photosynthesis to occur, and rapid reductions in DO would occur when irradiance levels drop down below the light requirements of the ambient phytoplankton communities. Therefore, water samples taken during daylight hours would be expected to have elevated levels of DO, while samples taken at night or early in the day would be expected to have low levels of DO.

In contrast, the very high levels of BOD in Lake Hancock (see figures 26 and 27) would exert a significant negative impact on DO levels in the Peace River, regardless of the time of day.

Results shown in **Figure 28** illustrate that whether samples of Lake Hancock were agitated or not, DO levels in sample bottles fell below the State of Florida's DO minimum of 4 mg/liter within 5 days. Interestingly, agitation of collected water did not produce a lasting benefit - DO levels in agitated samples were not different from unagitated samples within 24 hours. Instead, agitation seemed to be associated with a more rapid reduction in DO levels, such that agitated samples had DO levels less than 2 mg/liter at the end of 5 days, values even lower than the paired but unagitated samples.

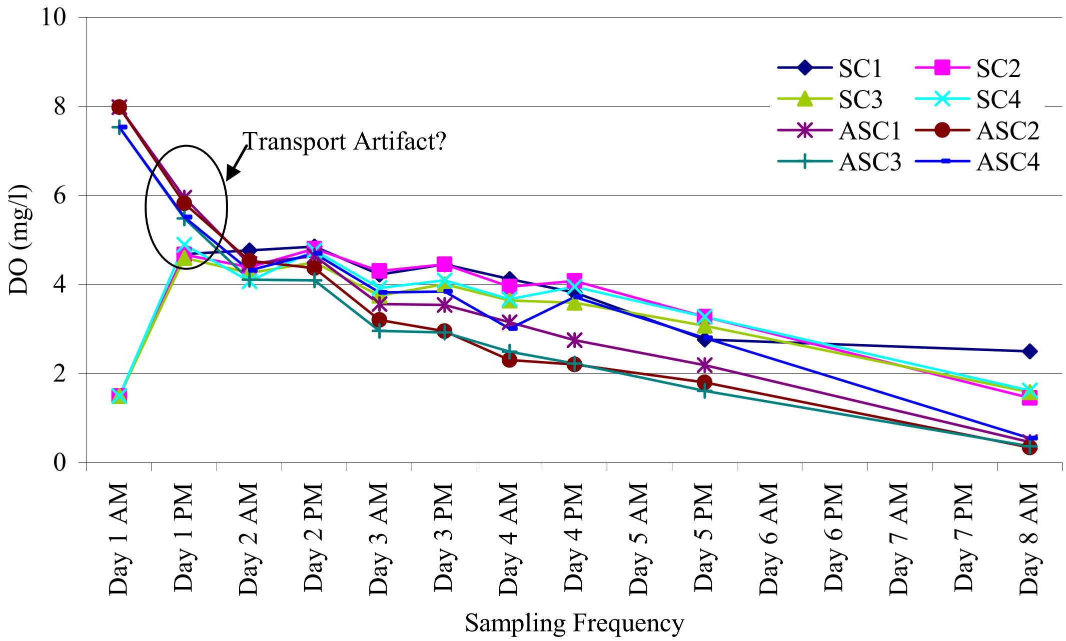


Figure 28. Results of experiment measuring dissolved oxygen over a week for ambient (SC1 to SC4) and agitated (ASC1 to ASC4) water collected at the P-11 structure.

3.0 Discussion

Data derived from the two experimental field studies and a review of historical data sets were analyzed to address the questions listed below.

1. Is there a phosphorus and/or nitrogen level below which blue-green algae in Lake Hancock no longer have a competitive advantage over other algal types?

2. Can removal of the organic sediments in Lake Hancock reduce phosphorus and/or nitrogen concentrations below threshold levels?

3. Will the removal of organic sediments from Lake Hancock likely result in a measurable improvement in water quality (e.g., TN, TP, chl A, BOD, TSI), and a reduction in the abundance of nuisance blue-green algae?

In addition, the results from these studies were interpreted in the context of the following questions, as well:

Based on the potential effects of atmospheric deposition, nitrogen fixation, and sediment re-suspension and re-mineralization as they relate to the lake's nutrient budget and the potential for improvements in water quality following sediment removal activities; **will sediment removal achieve the TSI proposed for Lake Hancock by FDEP (2005)?**

These questions are addressed in the following sections.

3.1. Is there a "threshold" value for TN and/or TP for blue-green algae?

Smith (1983) suggested that low TN:TP ratios (<29), brought about by excessive phosphorus loading, produce conditions that favor dominance by blue-green algae. However, Canfield *et al.* (1989) found little evidence for an effect of TN:TP ratios on blue-green algae abundance. Instead, Canfield *et al.* (1989) concluded that blue-green algae become proportionally more important in Florida lakes simply as overall phytoplankton biomass increases. This view has been supported by Downing *et al.* (2001) who concluded that overall nutrient supply, rather than ratios of nitrogen to phosphorus, was the best predictor of blue-green algal abundance.

In terms of TN:TP ratios, it is important to remember that levels of nitrogen in Lake Hancock are not simply a function of external loading. Significant levels of nitrogen fixation occur in both Lake Hancock (Figures 2 and 3) and Lake Jesup (Seminole County; PBS&J 2007), and nitrogen fixation is believed to be an issue in Lake Seminole as well (Pinellas County 2007). Clearly, nitrogen fixation via photosynthetic blue-green algae is likely to be a major source of nitrogen in Lake Hancock, as water column values of TN are much higher than levels of TN in stormwater runoff entering the lake. For example, levels of TN in Lake Hancock between 1992 and 2004 averaged 4.57 mg/liter (FDEP 2005), while TN levels of runoff from North Saddle Creek, which supplies 87 percent of the tributary inflow volume into Lake Hancock, were measured at 1.24 mg/liter (Table 5-4 in ERD 1999).

In contrast, the disparity between in-lake levels of TP and stormwater runoff is much less than that the disparity between in-lake levels of TN and TN levels in the stormwater entering the lake. Between 1992 and 2004, TP values in the lake averaged 0.52 mg/liter (FDEP 2005), while TP levels of runoff from North Saddle Creek, which supplies 87 percent of the tributary inflow volume into Lake Hancock, were measured at 0.44 mg/liter (Table 5-4 in ERD 1999). Perhaps associated with the removal of point source discharges from the cities of Auburndale and Lakeland (ERD 1999), levels of TP in Lake Hancock declined from an average of 0.96 mg liter in the 1980s to an average of 0.48 mg/liter in the 1990s (Table 2-5 in ERD 1999), a 51 percent decline. Levels of TN decreased by only 10 percent during that same time period. Chlorophyll *a* concentrations appear to have declined by only 5 percent, when comparing values in the 1980s to values in the 1990s (Table 2-5 in ERD 1999).

Recent reductions in nutrient loads, comparing Lake Hancock water in the 1980s to water in the 1990s, has been mostly associated with reductions in levels of TP, rather than TN. This has resulted in an increase in TN:TP ratios from a decadal average of 6.7 in the 1980s to 12.2 in the 1990s (ERD 1999), a shift that <u>could</u> foretell a shift in TN:TP ratios and subsequently result in less favorable conditions for blue-green algae (i.e., Smith 1983). Even if Smith (1983) is mistaken and nutrient ratios are irrelevant, there have been overall reductions in both TN and TP over the past two decades.

For reasons presented above, levels of TN within Lake Hancock are higher than what can be explained by loading models such as those constructed by ERD (1999) and FDEP (2005). Consequently, stormwater retrofit activities designed to reduce nitrogen loads to the lake might have little impact, as much of the nitrogen in the lake appears to be actually “created” within the lake itself from N_2 gas diffusing into the lake from the atmosphere (which is approximately 78 percent nitrogen). The draft TMDL for Lake Hancock (FDEP 2005), which calls for a 75.2 percent reduction in stormwater loads of TN, cannot be supported, as it does not properly account for the role of in-situ nitrogen fixation in the lake.

Similarly, the draft TMDL (FDEP 2005) may not accurately reflect the response of Lake Hancock water quality to changes in phosphorus loads. The draft TMDL does not appear to reflect the apparent disconnect between significant reductions in levels of TP in Lake Hancock (51 percent decline from 1980s to 1990s) compared to the minor or absent concurrent decline in chlorophyll *a* concentrations (5 percent decline from 1980s to 1990s).

In its Draft TMDL report on Lake Hancock, FDEP used the Watershed Assessment Model (WAM) and the Bathtub Model to predict expected changes in water quality in Lake Hancock in response to modeled reductions in nutrient loads. The time period for model calibration was 1994 to 2003. In response to TN and TP load reductions in excess of 75 percent, water quality in Lake Hancock was predicted to improve such that a long-term TSI value of 74.4 would be achieved. However, a trend analysis of water quality from Lake Hancock suggests that substantial reductions in the levels of TN and TP in the lake (**Figure 29**) have not been accompanied by concurrent reductions in levels of chlorophyll *a* (**Figure 30**). As noted above, TN levels in the lake are strongly affected by in-lake nitrogen fixation, and reductions in external loads may have little to no direct effect on TN values in the lake.

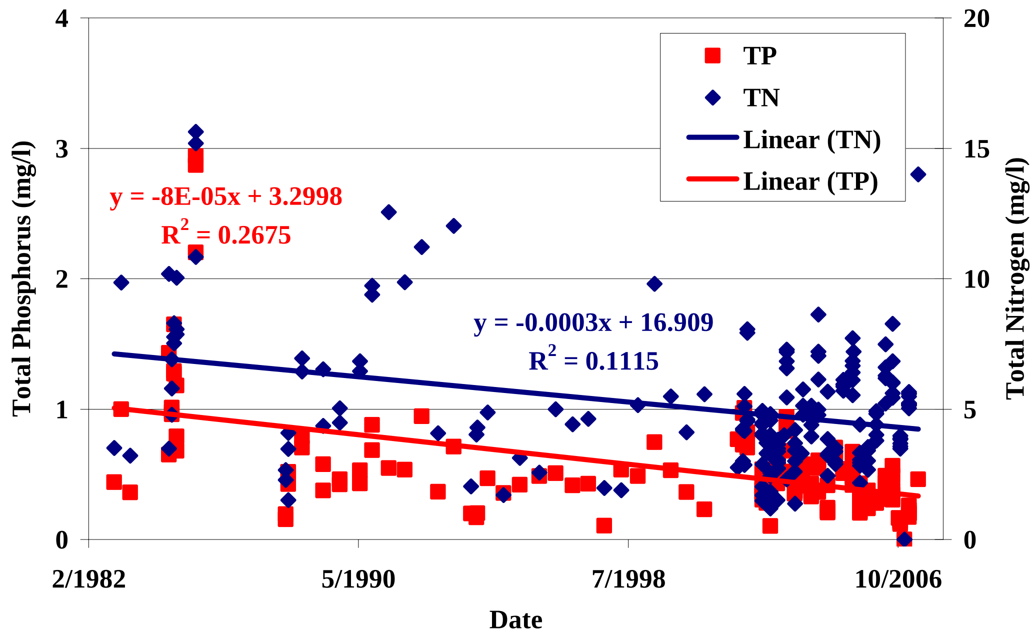


Figure 29. A significant decline ($p<0.05$) in Total Nitrogen and Total Phosphorus concentrations is observed in Lake Hancock from the 1980's to present.

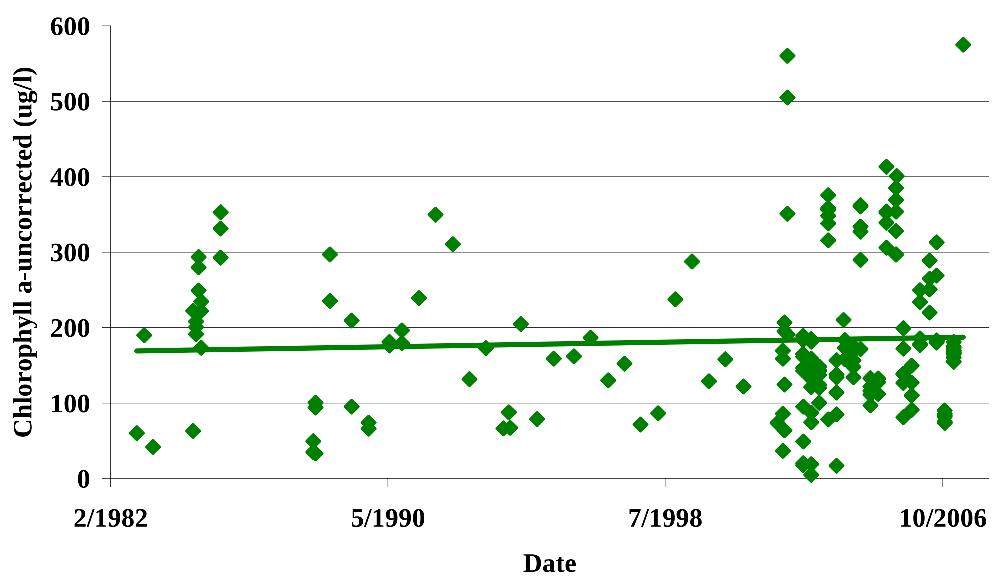


Figure 30. No significant change in uncorrected chlorophyll *a* concentrations are observed in Lake Hancock from the 1980's to present.

A highly relevant question, related to the appropriateness of the draft TMDL goal of a 75 percent TP reduction, is the following:

If chlorophyll a concentrations in Lake Hancock did not show a significant response to recent (1990s) and substantial reductions in TP levels in the lake, why would further reductions in external loads (as called for in the draft TMDL) be expected to bring about reductions in chlorophyll a?

While this report does not contradict the value of reducing external nutrient loads to Lake Hancock, it does suggest that internal loads from remineralization and resuspension of bottom sediments are a significant and continuing source of TN and TP. Further, our results suggest that sediment removal activities could reduce in-lake TP values to levels close to potential "thresholds" that allow blue-green algae to dominate.

In their nutrient budget for Lake Hancock, ERD (1999) measured TN and TP loads into the lake from stormwater runoff, direct rainfall, base flow, and in-lake seepage. Loads were also measured as they left the lake via South Saddle Creek. When loads into the lake from these four loading sources were totaled, 175,879 kg TN was calculated to enter the lake annually. Measured loads leaving the lake via South Saddle Creek totaled 271,700 kg/ yr for TN. Measured TP loads into the lake were estimated at 35,086 kg TP per year, with TP loads leaving the lake via South Saddle Creek estimated at 23,180 kg / yr.

Based on these calculations, approximately 34 percent of the TP loads into the lake remain in the lake. In contrast, there is a surplus of 95,821 kg TN leaving the lake on an annual basis, compared to the amount calculated to enter the lake. This represents a "surplus" of nitrogen equal to 35 percent of the estimated load leaving the lake.

These findings are consistent with the hypothesis that Lake Hancock is a net sink for phosphorus loads, which accumulate in the sediments, while simultaneously being a net source for nitrogen (created via nitrogen fixation), which is then exported out of the lake, and eventually into the Peace River and Charlotte Harbor via South Saddle Creek.

It would seem that only by reducing TP loads from both external and internal loading sources could TP levels be further reduced (in addition to prior reductions), and load reduction strategies that do not appropriately incorporate sediment nutrient availability might not be able to create desired water quality. As such, it would appear that the lack of incorporation of a sediment-based load reduction strategy for Lake Hancock makes the stormwater-only TP load reduction of 75.2 percent perhaps insufficient. That is, if internal loads are not taken into account, then this "reservoir" of older, now reduced external TP loads will continue to contribute to present day water quality issues.

3.2. Can removal of the organic sediments in Lake Hancock reduce phosphorus and/or nitrogen concentrations below threshold levels?

Activities that would reduce phosphorus loads, both internal and external, to the lake might in turn increase TN:TP ratios to levels that would no longer favor the dominance of blue-green algae, if Smith (1983) is correct.

Based on that premise, the period of record for water quality was examined to determine if "target" levels of TP could be developed, based on the influence of TP on TN:TP ratios. As with any ratio, the TN:TP ratio can be impacted by changes in both the numerator (TN) and the denominator (TP). Since TN levels vary as a function of nitrogen-fixation, they are not wholly controlled by external loads of TN. As such, TN "targets" may not prove useful. In contrast, TP levels are not directly influenced by "fixation" and their levels more accurately reflect the influence of both internal and external loads.

The relationship between levels of TP and TN:TP ratios is graphed in **Figure 31**. Based on the best fit equation for this relationship, the TP level that would correlate with a TN:TP ratio of 29:1 was determined. A TN:TP ratio of 29:1 (by weight) was suggested by Smith (1983) as being a "threshold" value below which conditions would favor dominance by blue-green algae. To keep TN:TP ratios above 29:1, phosphorus levels would have to remain lower than 0.08 mg TP/liter.

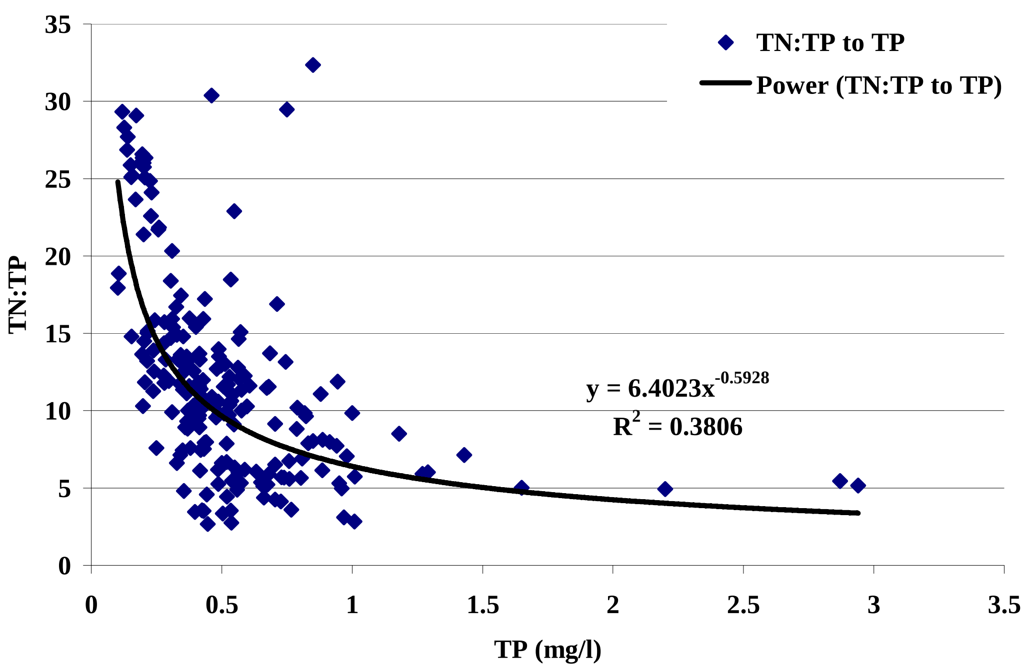


Figure 31. Regression model relating TP concentration to the Total Nitrogen: Total Phosphorus ratio in Lake Hancock for the period of record.

Based on the response of N-fixation rates to phosphorus additions, rates of N-fixation appear to be "saturated" at TP levels somewhat less than 0.16 mg TP / liter (see Figure 3). The actual saturation threshold could not be calculated, but rates of N-fixation were not further increased by TP levels higher than 0.16 mg / liter.

It would seem reasonable to conclude that an appropriate target for TP levels in Lake Hancock would be somewhere between 0.08 and 0.16 mg / liter. TP levels within the lake averaged 0.52 mg/liter between 1992 and 2004 (FDEP 2005).

In comparison, phosphorus levels within dredged cylinders in December 2006 averaged 0.13 and 0.21 mg TP/liter for non-mixed and mixed treatments, respectively. These results represent TP reductions of 37 and 77 percent, compared to non-dredged cylinders. In May 2007, phosphorus levels within dredged cylinders averaged 0.41 and 0.51 mg TP/liter for non-mixed and mixed treatments, respectively. Although these levels were higher than those found in December 2006, they still represented substantial reductions of between 21 and 69 percent, compared to undredged cylinders (see Table 5).

The results from the mesocosm experiments suggest that sediment removal could result in substantial reductions in TP levels within Lake Hancock. If these results can be scaled up to whole-lake expectations, TP levels could be reduced to potentially biologically relevant concentrations (levels that may affect blue-green algal abundance), at least portions of the year.

As an additional assessment of the ability of sediment removal to reduce TP levels, wind speed, wind direction, rainfall and lake stage were compared to TP concentrations within Lake Hancock. Data for these multiple external factors were loaded into a multiple regression model to determine if a statistically significant relationship was evident between TP concentrations as the dependent variable, and wind speed, wind direction, rainfall and lake stage as potentially significant independent variables. The resulting model containing daily wind speed, daily rainfall and daily stage explained 29.3 percent of the variation of TP in Lake Hancock. The equation is below:

$$TP = 5.93133 + (0.633035^{*}\text{rainfall}) + (0.062019^{*}\text{ windspeed}) - (0.0581623^{*}\text{ stage})$$

Where TP is in mg/l, rainfall is in inches, windspeed is in mph and stage is in decimal feet.

*Note: each independent variable was measured in a different unit and has a different range of values.

Using the above equation, the variable of lake stage alone was modified based upon several lake level manipulation scenarios, including a proposed action by the SWFWMD to increase the seasonal high water level of Lake Hancock by 1.8 feet, to allow for increased storage for recovering minimum flows in the Upper Peace River (Table 6 [same as Table 4]). Unaltered wind speed and rainfall data were used in combination with the altered stage data to calculate modeled TP concentrations for Lake Hancock based upon the following scenarios:

Table 6. Changes in effective water depths (feet) with different sediment removal and lake level modification scenarios.

Scenario	Condition	Change in Effective Water Depth (ft)
1	Current	NA
2	Sediment Removal (1 ft)	1.0
3	Proposed Lake Level Modification (1.8 ft)	1.8
4	Sediment Removal (2 ft)	2.0
5	Proposed Lake Level Modification (1.8 ft) and Sediment Removal (1 ft)	2.8
6	Sediment Removal (3 ft)	3.0
7	Proposed Lake Level Modification (1.8 ft) and Sediment Removal (2 ft)	3.8
8	Proposed Lake Level Modification (1.8 ft) and Sediment Removal (3 ft)	4.8

Using the equation that models TP concentrations vs. stage (among other external factors) the expected TP concentration associated with these eight scenarios was calculated, as shown in **Figure 32**.

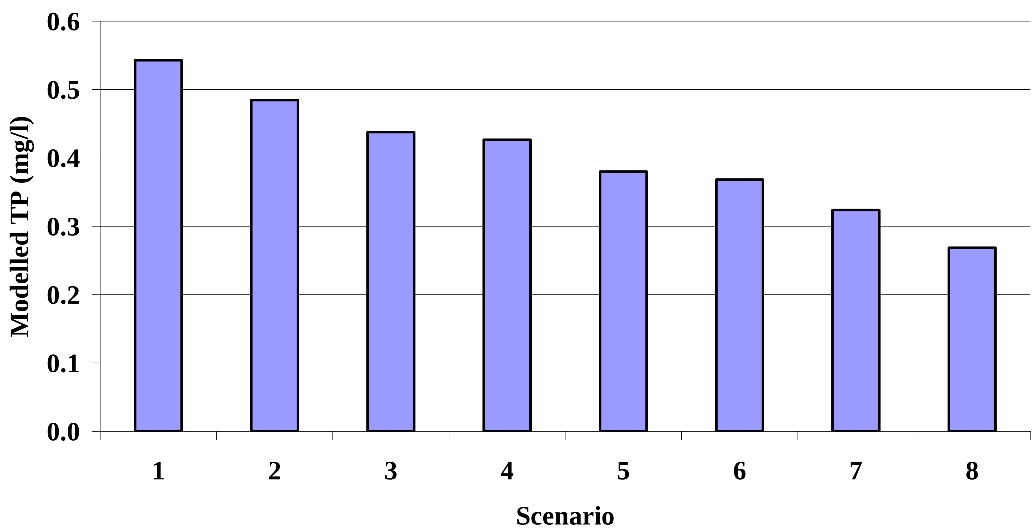


Figure 32. Modelled TP concentration using real rainfall and windspeed for multiple scenarios where lake stage was modified (see Table 6).

Under scenario 8, where the SWFWMD's lake level modification project has increased water levels by 1.8 feet, and 3 feet of muck has been removed from the lake bottom, the effective distance between the lake's surface and the bottom (at least in the wet season) would be increased by 4.8 feet. Under such a condition, TP concentrations within Lake Hancock would be expected to average 0.27 mg/liter, a 52 percent reduction from the current (1992 to 2004) average of 0.52 mg TP / liter (see Table 1). The predicted TP level of 0.27 mg / liter is 14 percent lower than the average (n = 4) of the post-dredging TP levels for the four scenarios investigated (December and May, Mixed and non-Mixed). Also, the 52 percent reduction in TP levels (0.52 going to 0.27) is nearly identical to the average (n = 4) percent reduction in TP levels calculated when comparing dredged to undredged cylinders for the four scenarios investigated (December and May, Mixed and non-Mixed).

Based on the modeled reductions in TP levels associated with each of the eight scenarios described above, the TN:TP ratio expected, based on TP concentrations, was estimated using the equation generated from **Figure 31**. These calculated TN:TP ratios are shown in **Figure 33**.

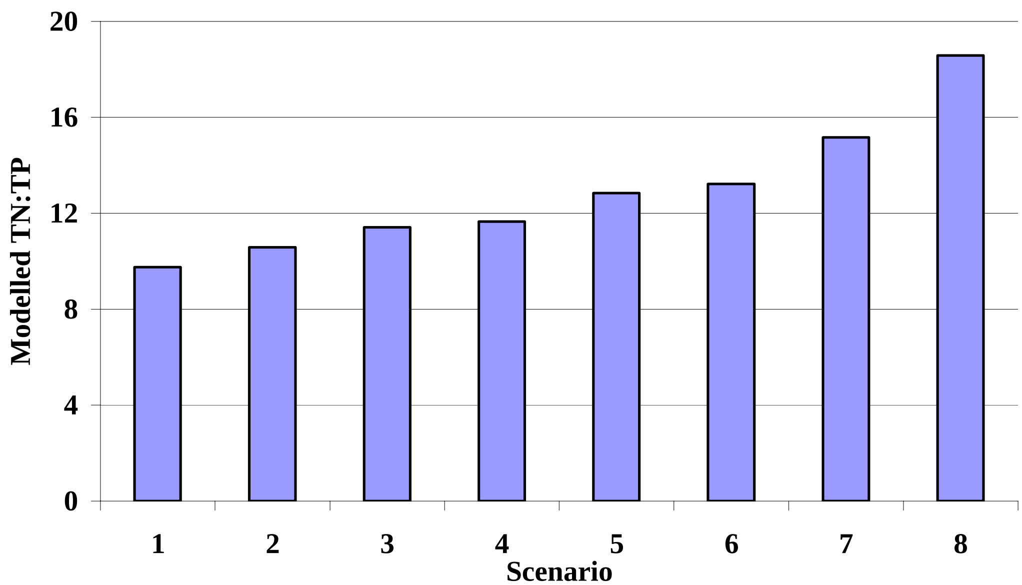


Figure 33. Modelled TN:TP for multiple scenarios changing the effective water depth of Lake Hancock (see Table 6).

Under scenario 8, where the effective distance between the lake's surface and the bottom (at least in the wet season) would be increased by 4.8 feet, TP concentrations within Lake Hancock would be expected to average 0.27 mg/liter, and the TN:TP ratio would be expected to be 18.6. While this value is lower than the potential "threshold" TN:TP ratio of 29 discussed by Smith (1983) it is higher than the often-quoted TN:TP value of 16 commonly called the "Redfield Ratio". This ratio, while not appropriate under all conditions, is nonetheless a fairly robust indicator of conditions favorable (lower than 16) or unfavorable (higher than 16) for the development of nuisance algal blooms (i.e., Redfield 1958).

As such, both the manipulative sediment removal study and the TP modeling efforts described above indicate that substantial reductions in TP concentrations could be achieved with the removal of the highly organic bottom sediments of Lake Hancock. Further, these results suggest that levels of TP that might occur with sediment removal (0.27 to 0.31 mg TP/liter) would be expected to result in TN:TP ratios less favorable for domination by blue-green algae. However, these potential TP levels are higher than the TP concentration that was high enough to saturate nitrogen-fixation (0.16 mg TP/liter; see Figure 3). The results of these studies cannot answer the question as to whether or not sediment removal activities would drop TP levels low enough to result in a lake no longer dominated by blue-green algae.

3.3. Will the removal of organic sediments from Lake Hancock likely result in a measurable improvement in water quality for other water quality parameters?

As a final test of the potential benefits to water quality associated with sediment removal, the Dynamic Ratio concept outlined by Bachmann *et al.* (2000) was examined. In a comprehensive survey of 36 Florida Lakes, Bachmann *et al.* (2000) found that water quality could be predicted, with fairly good results, based on a term called the "Dynamic Ratio". The Dynamic Ratio is calculated as the square root of the size of the open water of the lake (in square kilometers) divided by the average depth of the lake (in meters). Using information within ERD (1999), and converting to metric units, the Lake Hancock Dynamic Ratio is thus:

$$\text{Dynamic Ratio} = \text{Square Root of } 18.4 \text{ km}^2 / 1.5 \text{ meters} = 2.86$$

The Dynamic Ratio was then calculated for Lake Hancock under Scenario 8 (Table 6) wherein 3 feet of muck would be removed and the lake's seasonal high water level would be raised by 1.8 feet (equating to an average water depth of 3.0 meters). The Dynamic Ratio for Lake Hancock under Scenario 8 is thus:

$$\text{Dynamic Ratio} = \text{Square Root of } 18.4 \text{ km}^2 / 3.0 \text{ meters} = 1.42$$

Within Bachmann *et al.* (2000), equations are derived that allow for the estimation of levels of TN, TP, and chlorophyll *a*, as related to Dynamic Ratios. However, Lake Hancock's water quality is sufficiently poor that it is an "outlier" to their data set, with higher levels of TN and TP, and much higher levels of chlorophyll *a*, than most of the lakes they examined. However, the equations in Bachmann *et al.* (2000) can be used to calculate estimates of the amount of time that the entirety of the lake bottom would be expected to be disturbed by wind-driven waves. This equation is:

$$y = -2.45 + 6.47\,x;$$

Where, y = percent of time that 100 percent of the lake bottom would be disturbed, and

x = Dynamic Ratio

Using this equation, the percent of time that 100 percent of the bottom of Lake Hancock would be disturbed calculates out to 16.1. Under Scenario 8, with an increase in the effective water depth of 4.8 feet, the percent of time that 100 percent of the bottom of Lake Hancock would be disturbed calculates out to 6.7. Taking the inverse of these two numbers, present day conditions in Lake Hancock are such that 100 percent of the bottom of the lake would be expected to be disturbed every 6.2 days (100 / 16.1 = 6.2). Under Scenario 8, this time interval would increase to once every 14.9 days (100 / 6.7 = 14.9).

If the Dynamic Ratio developed by Bachmann *et al.* (2000) accurately reflects the probability of sediment resuspension in Lake Hancock, the removal of 3 feet of organic muck, combined with an increased water level of 1.8 feet, could reduce the frequency of whole-lake sediment resuspension events from approximately once a week to just over once every two weeks. As nitrogen fixation and carbon fixation rates respond quite rapidly to pulses of phosphorus enrichment (see Figure 3) and as resuspension of sediments brings about an increased level of TP within the water column, it would seem reasonable to conclude that increases in the effective water depth of Lake Hancock would likely bring about an increase in water quality, due in part to conditions being less favorable for resupension of bottom sediments.

Reductions in the internal loads of TP and increased water depths would be expected to decrease levels of chlorophyll *a* in Lake Hancock. These actions would also be expected to decrease levels of TSS and turbidity within the lake, perhaps allowing for increased water clarity. Increased water clarity could potentially allow submerged vegetation to become established, at least in the shallowest fringes of the lake. Additionally, reductions in BOD might be expected to occur with sediment removal, especially on windy days. A reduction in BOD levels would not only benefit the lake itself, but it would likely benefit the downstream water bodies of South Saddle Creek, and even the Upper Peace River (at least at Bartow).

3.4. Will sediment removal achieve the TSI goal proposed for Lake Hancock by FDEP (2005)?

An overriding conclusion of this report, based on an analysis of data from Lake Hancock, is that extreme caution should be used when making predictions on expected changes in nutrient loads and water quality in such a lake. Brenner *et al*. (2002) concluded that Lake Hancock's water quality was so poor that typical water quality models should not be used for management purposes, and that only the broadest of water quality characterization predictions (i.e., classification as hypereutrophic or mesotrophic) would be appropriate. As such, the use of traditional (or even "improved") water quality models would not be appropriate tools for setting water quality goals for TMDL purposes. Even the calculation of nutrient loads, a vital first step in water quality modeling techniques, has proven extremely problematic in Lake Hancock. In their detailed nutrient budget for Lake Hancock, ERD (1999) calculated a surplus of 95,821 kg TN leaving the lake on an annual basis, compared to the amount calculated to enter the lake from all sources. This excess nitrogen can be accounted for via nitrogen fixation by blue-green algae, although rates appear to vary in space and time. In their TMDL modeling efforts, FDEP (2005) had to invoke "in-lake processes" to balance the nitrogen budget for Lake Hancock. Without these unmeasured processes, in-lake TN values were significantly higher than modeled values for the lake.

In consideration of these phenomena, it might be best to use a "common-sense" approach to water quality restoration, with models used only sparingly, if at all, until more complete data sets of in-lake processes are developed.

This, however, does not mean that lake restoration techniques should not be continued and or investigated for Lake Hancock. This report finds no reason why the original course of action outlined by CDM (2002) would not be appropriate today.

In their report to Polk County and FDEP, CDM (2002) assessed the value of a variety of lake restoration techniques. The recommended course of action for Lake Hancock included the following components: 1) wetland treatment of discharges from the lake, 2) chemical treatment/settling pond treatment of inflows from North Saddle Creek, 3) wetland treatment of inflows from Banana Creek, and 4) passive drawdown and mechanical excavation of bottom sediments in nearshore areas. Currently, two of the four restoration techniques proposed by CDM are scheduled for implementation or have been recently completed. Polk County has already constructed a wetland treatment system that treats approximately fifty percent of the Banana Creek inflows to Lake Hancock. Preliminary results indicate a significant improvement in discharged water quality, thereby further reducing the external nutrient load to the lake. Additionally, the District is scheduled to construct a wetland treatment system to cleanse approximately fifty percent of the water leaving Lake Hancock.

Results from this study do not conflict with these four recommended actions. However, they can be interpreted such that task 4, the excavation of bottom sediments, could be of such importance that this activity should not be restricted to nearshore areas alone. Techniques that would allow for the removal of bottom sediments throughout the lake would be of greater benefit than techniques that were restricted to the nearshore areas alone.

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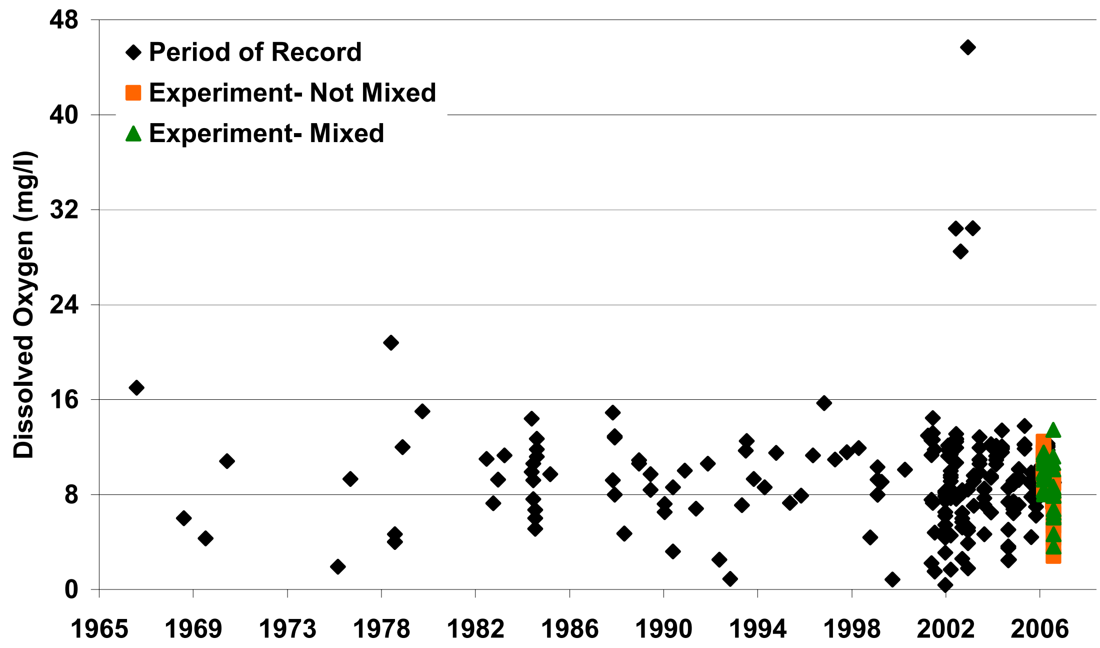


Figure A1. Polk County wateratlas period of record dissolved oxygen comparison to PBS&J mesocosm experiment in Lake Hancock.

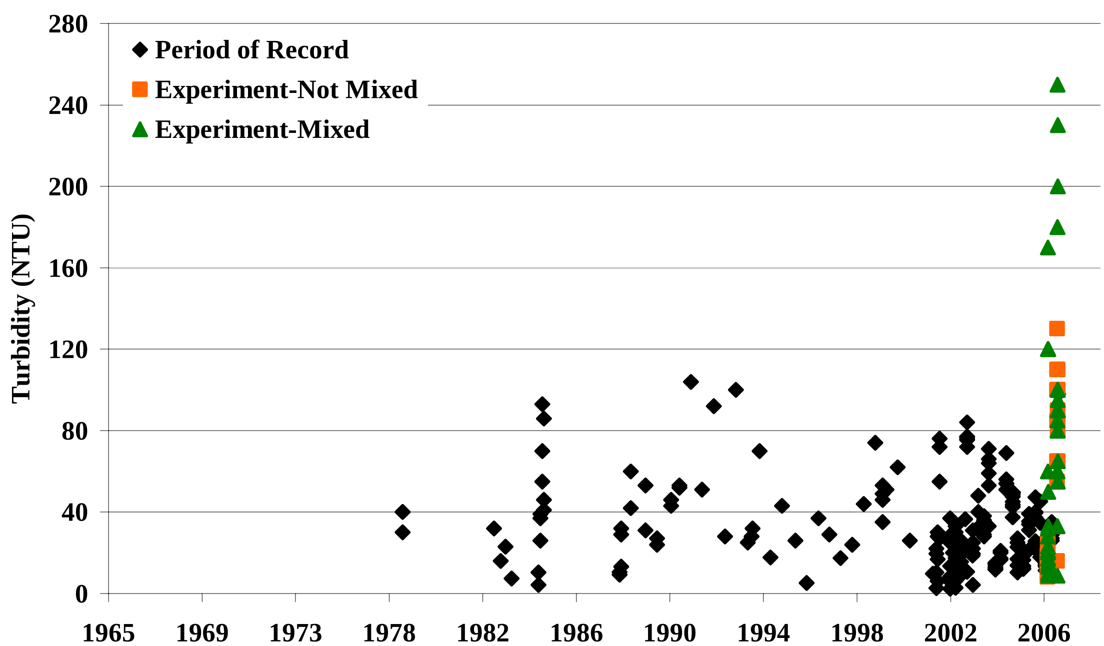


Figure A2. Polk County wateratlas period of record turbidity comparison to PBS&J mesocosm experiment in Lake Hancock.

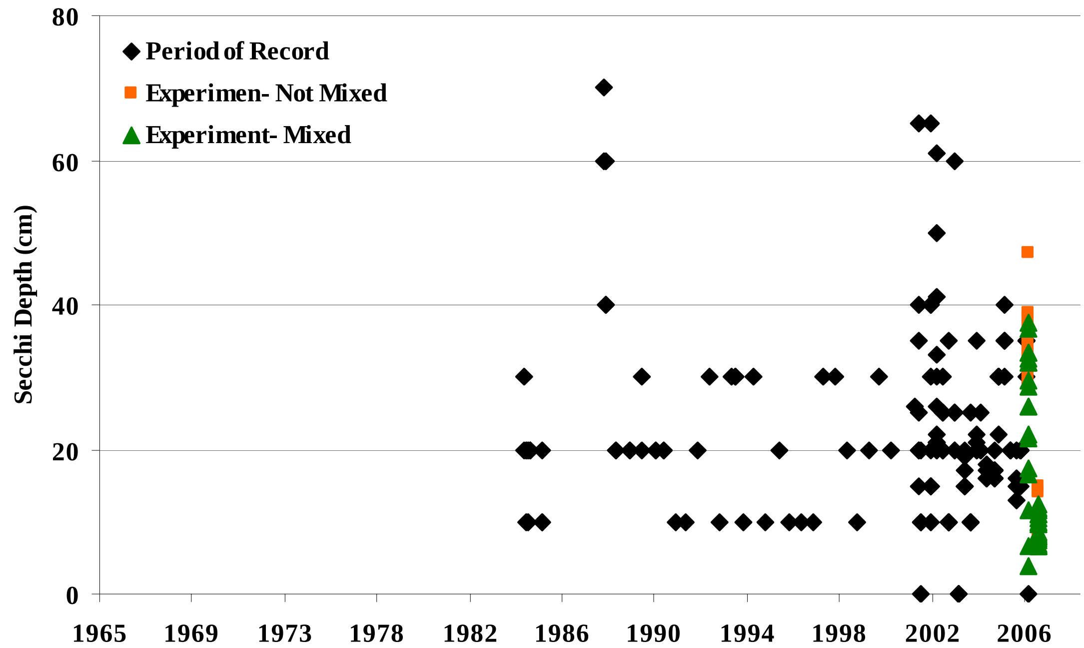


Figure A3. Polk County wateratlas period of record secchi depth comparison to PBS&J mesocosm experiment in Lake Hancock.

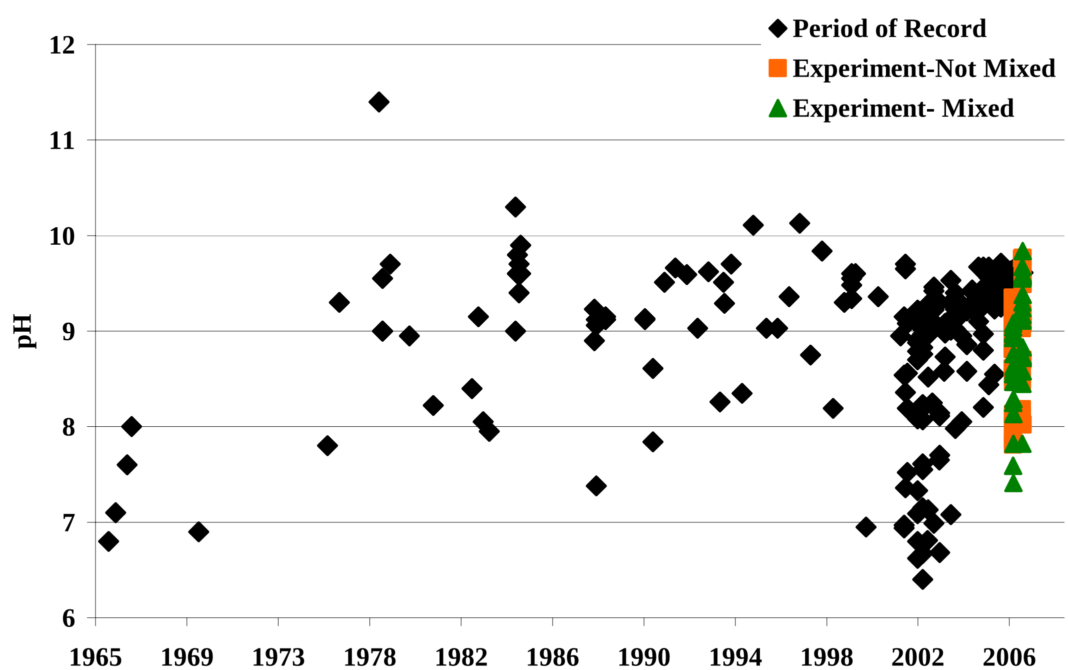


Figure A4. Polk County wateratlas period of record pH comparison to PBS&J mesocosm experiment in Lake Hancock.

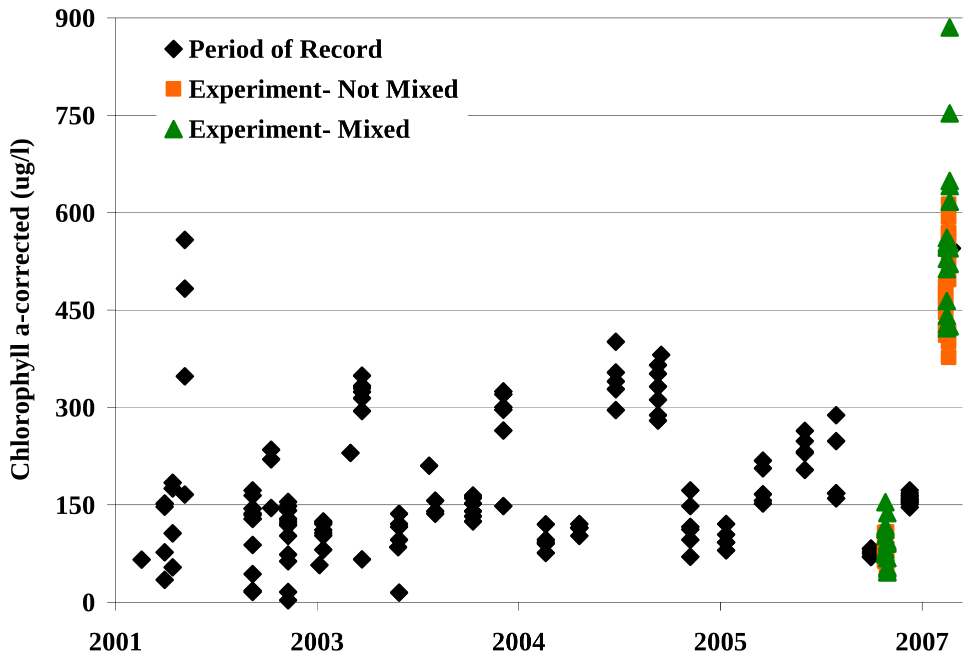


Figure A5. Polk County wateratlas period of record chlorophyll *a* comparison to PBS&J mesocosm experiment in Lake Hancock.

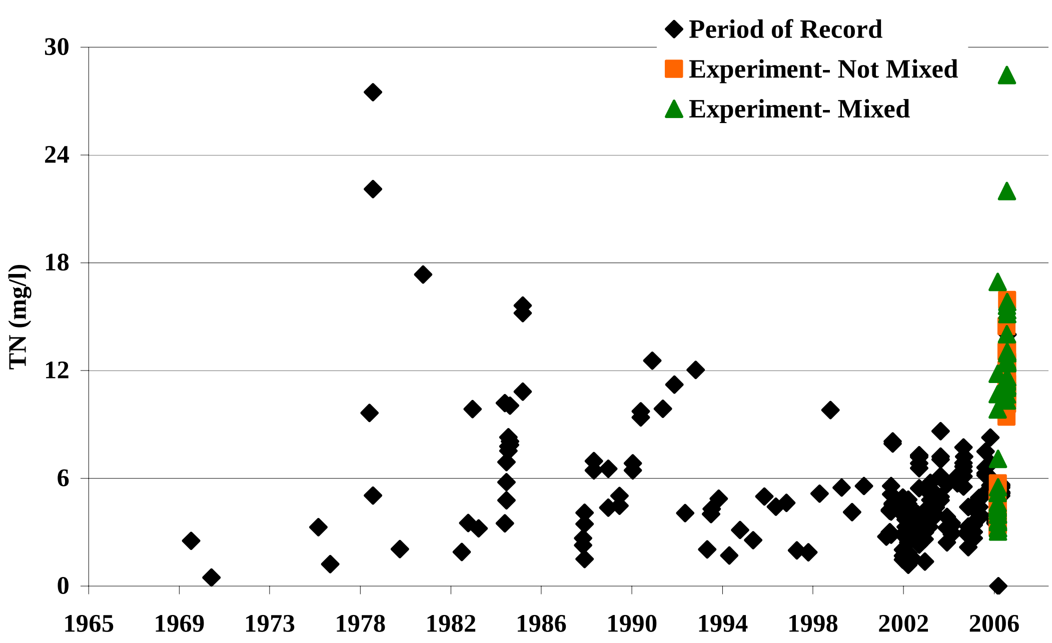


Figure A6. Polk County wateratlas period of record Total Nitrogen comparison to PBS&J mesocosm experiment in Lake Hancock.

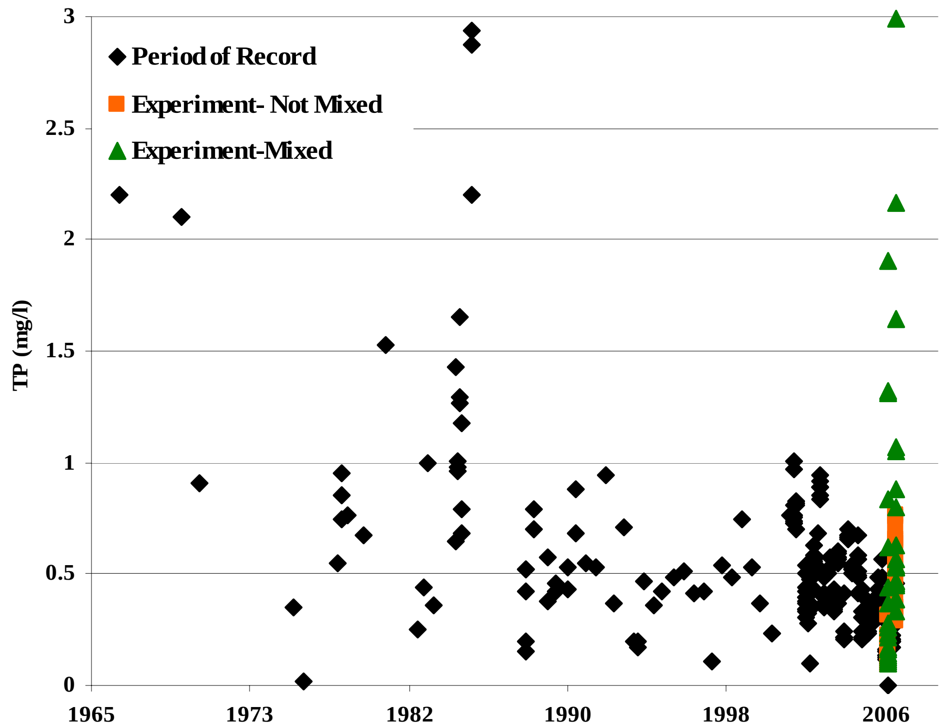


Figure A7. Polk County wateratlas period of record Total Phosphorus comparison to PBS&J mesocosm experiment in Lake Hancock.

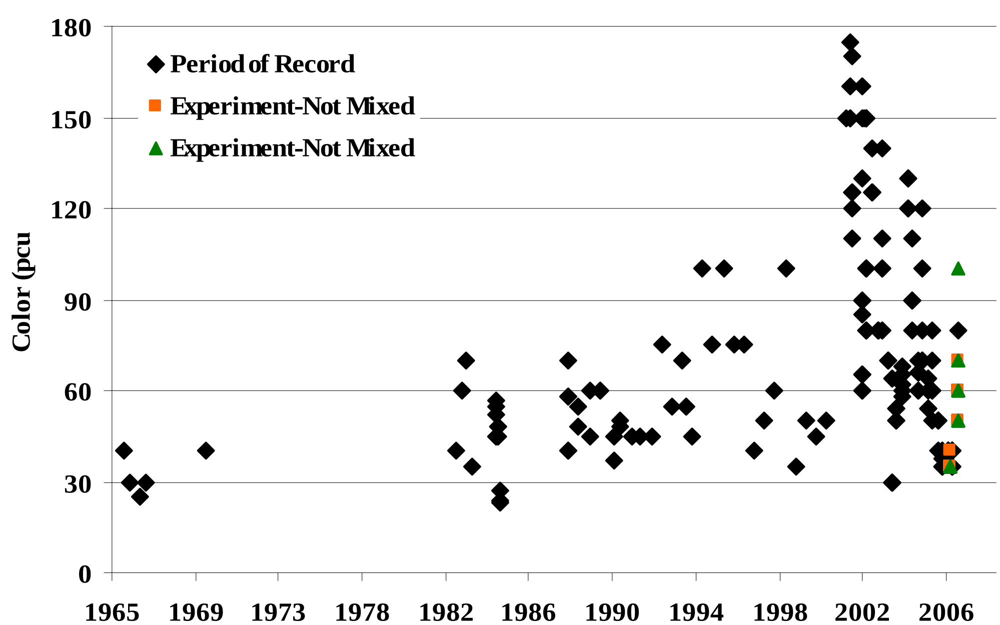
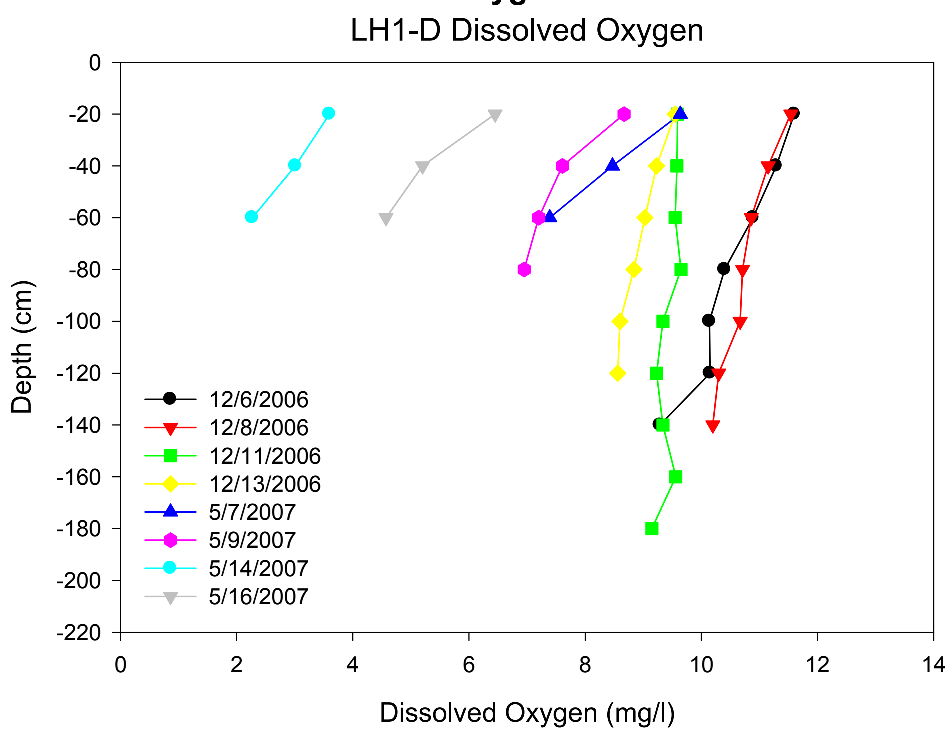
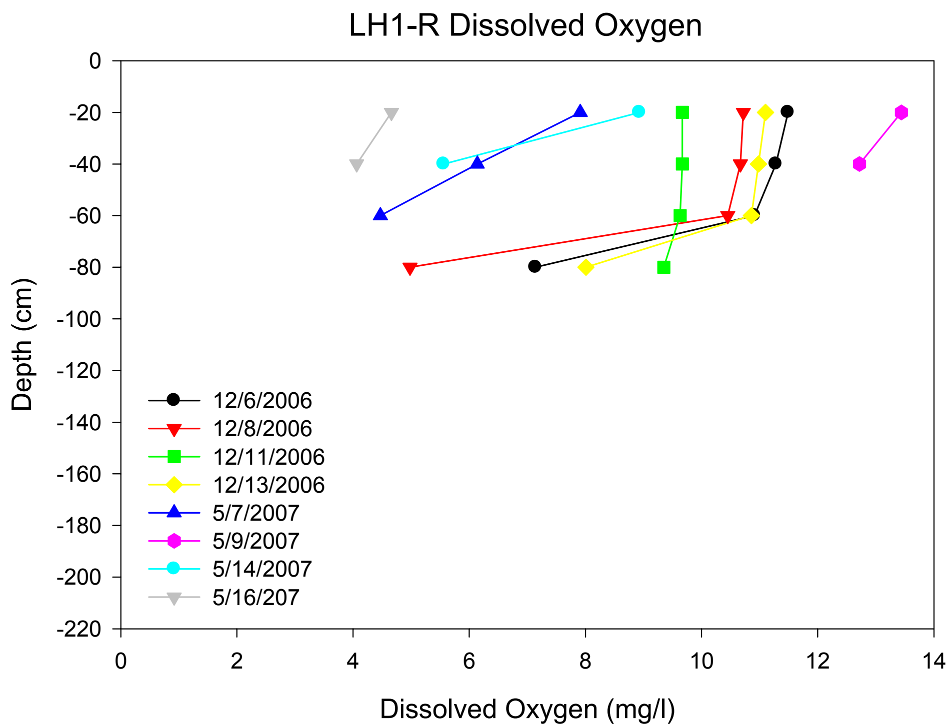
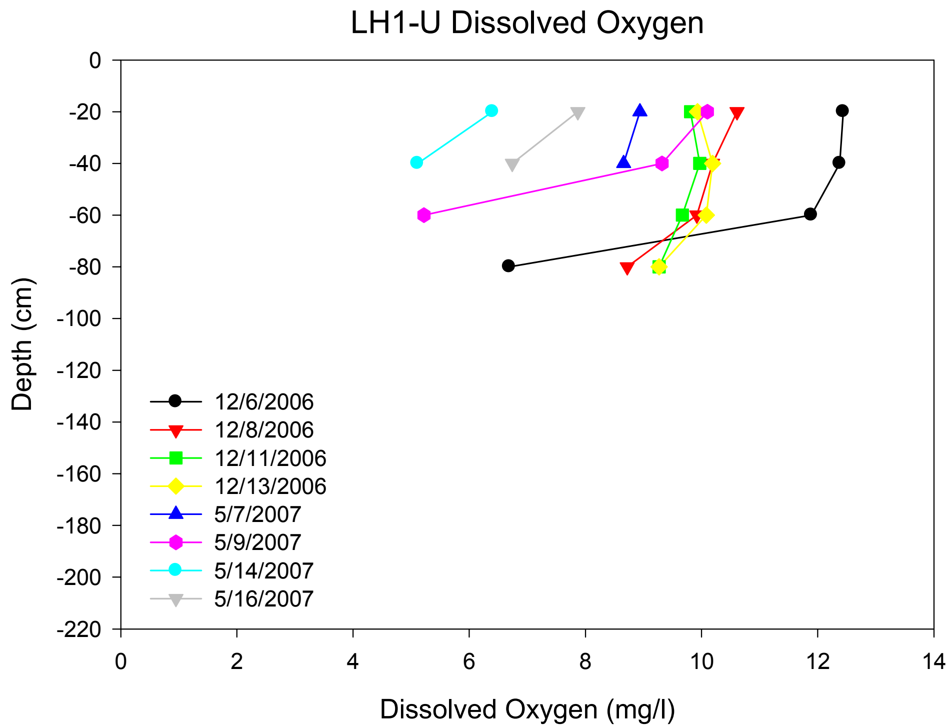


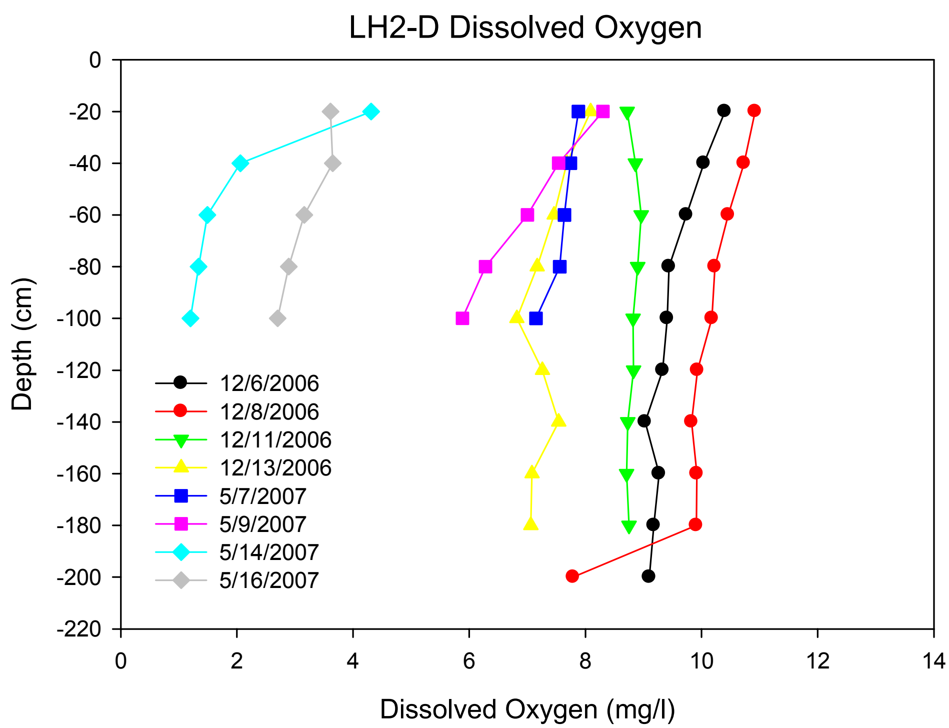
Figure A8. Polk County wateratlas period of record color comparison to PBS&J mesocosm experiment in Lake Hancock.

Dissolved Oxygen Profiles






LH1-U Dissolved Oxygen
Depth (cm)
Dissolved Oxygen (mg/l)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
0
2
4
6
8
10
12
14
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007


LH2-D Dissolved Oxygen
Depth (cm)
Dissolved Oxygen (mg/l)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
0
2
4
6
8
10
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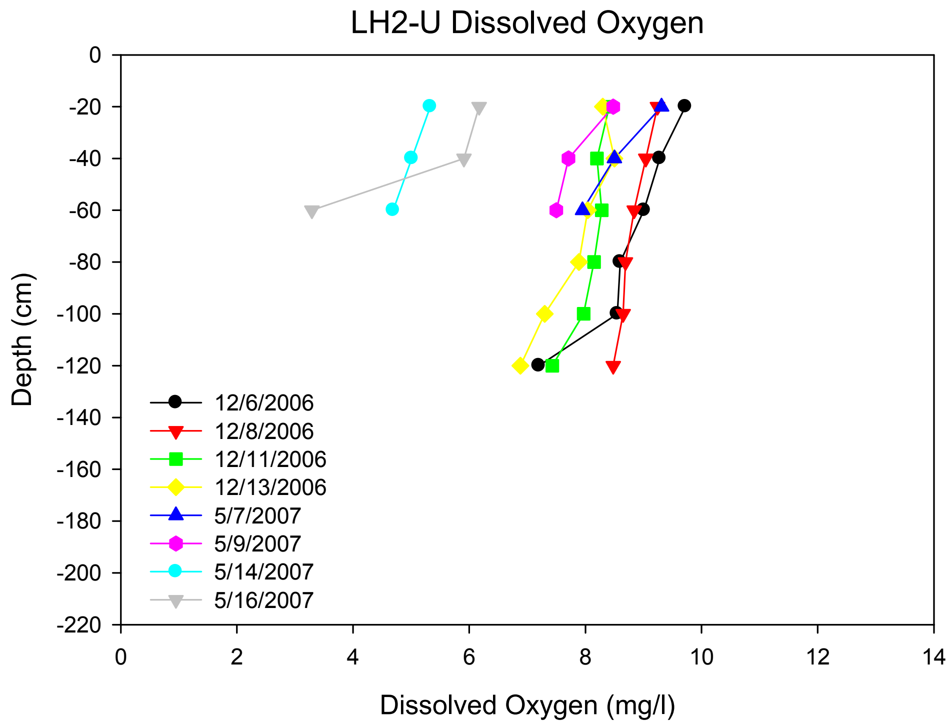
5/14/2007
5/16/2007


LH2-U Dissolved Oxygen
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
0
2
4
6
8
10
12
14
Dissolved Oxygen (mg/l)
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
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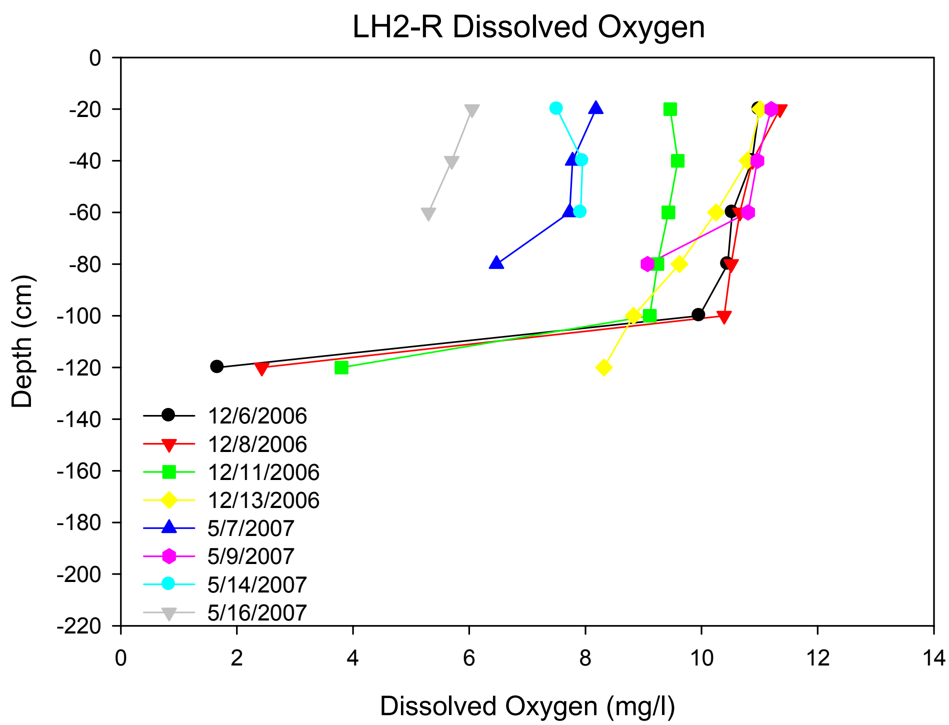
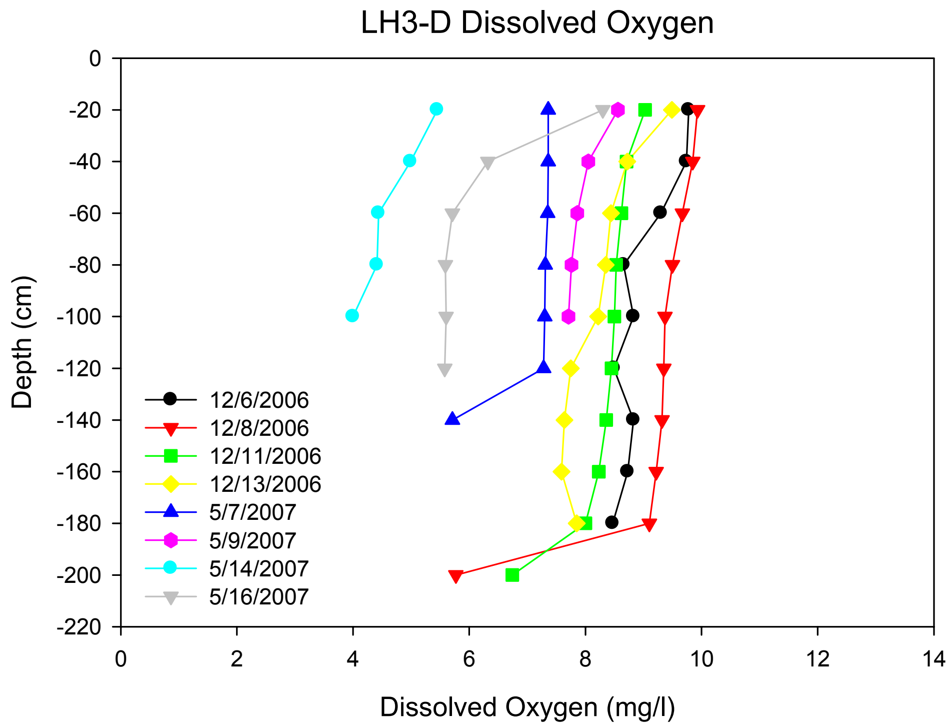
5/14/2007
5/16/2007


LH2-R Dissolved Oxygen
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
0
2
4
6
8
10
12
14
Dissolved Oxygen (mg/l)
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007


LH3-D Dissolved Oxygen
Depth (cm)
Dissolved Oxygen (mg/l)
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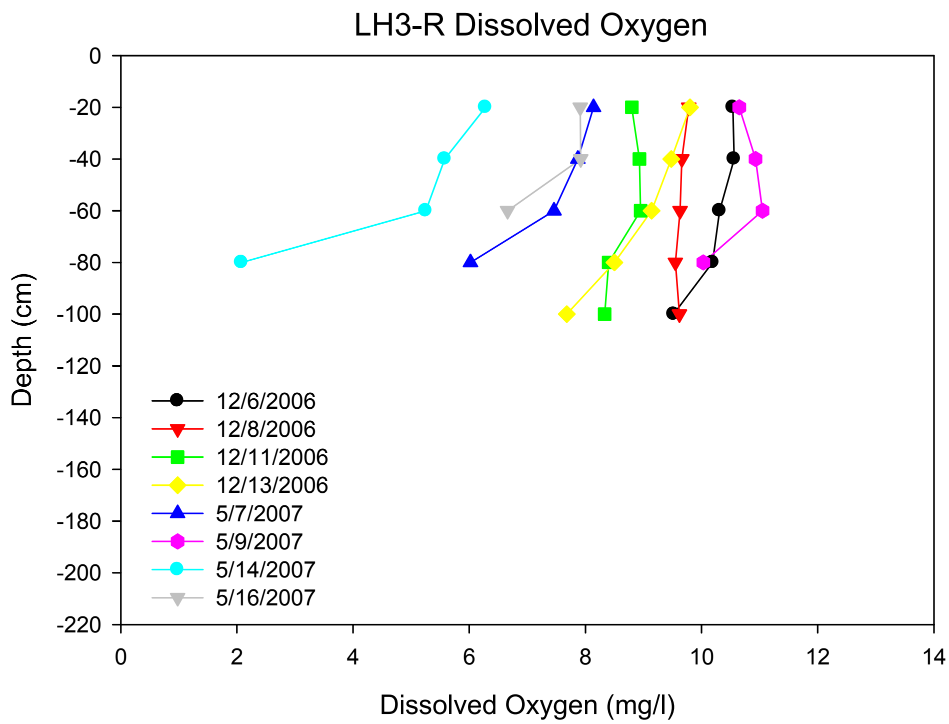
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
0
2
4
6
8
10
12
14
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007


LH3-R Dissolved Oxygen
Depth (cm)
Dissolved Oxygen (mg/l)
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-20
-40
-60
-80
-100
-120
-140
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-180
-200
-220
0
2
4
6
8
10
12
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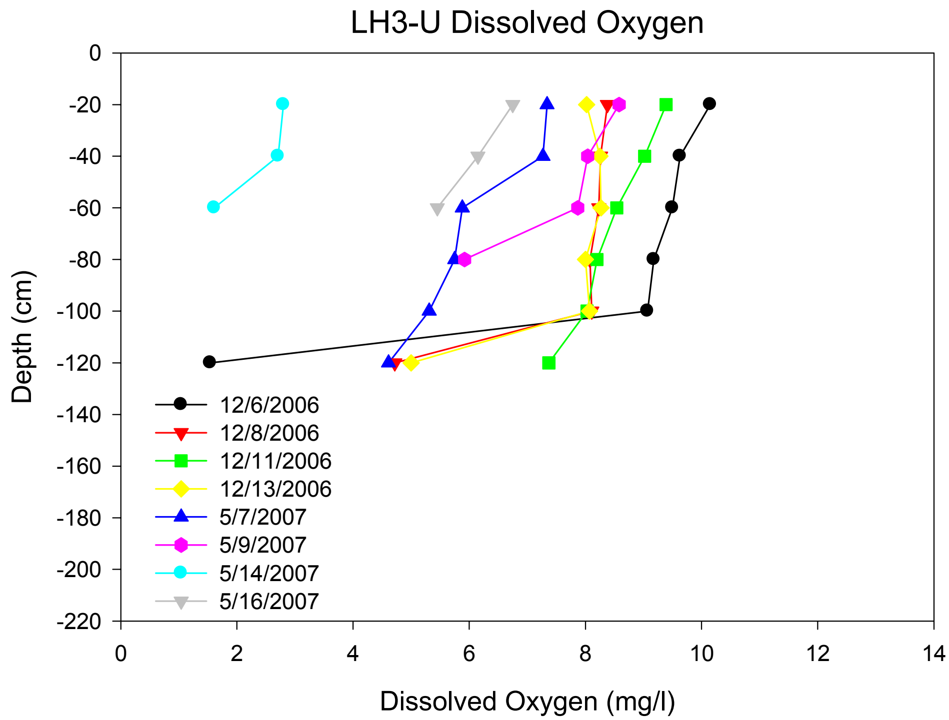
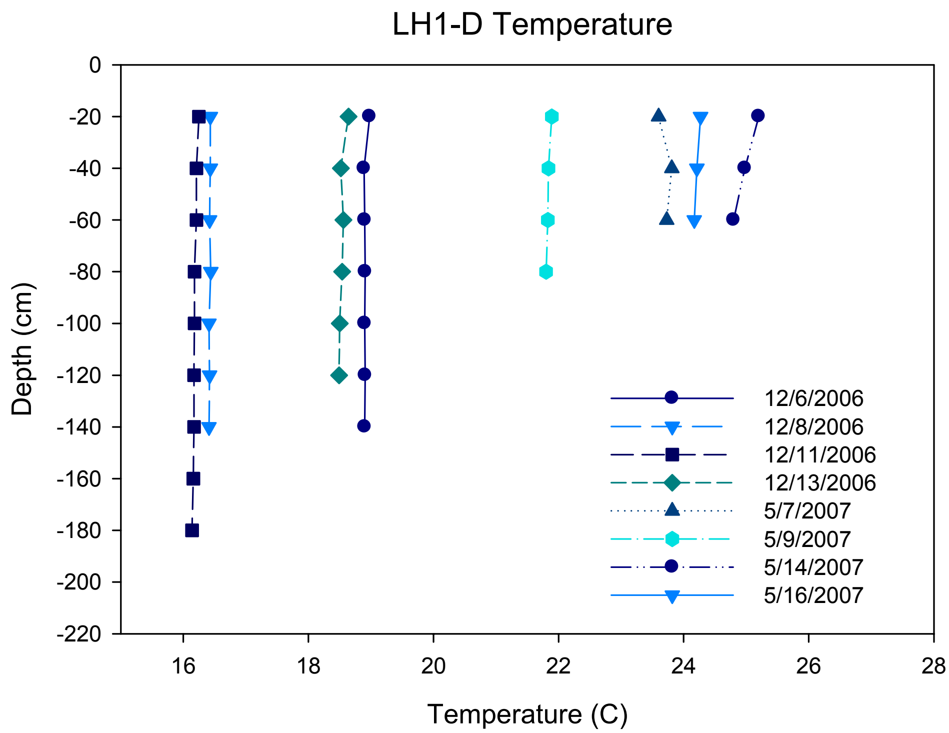
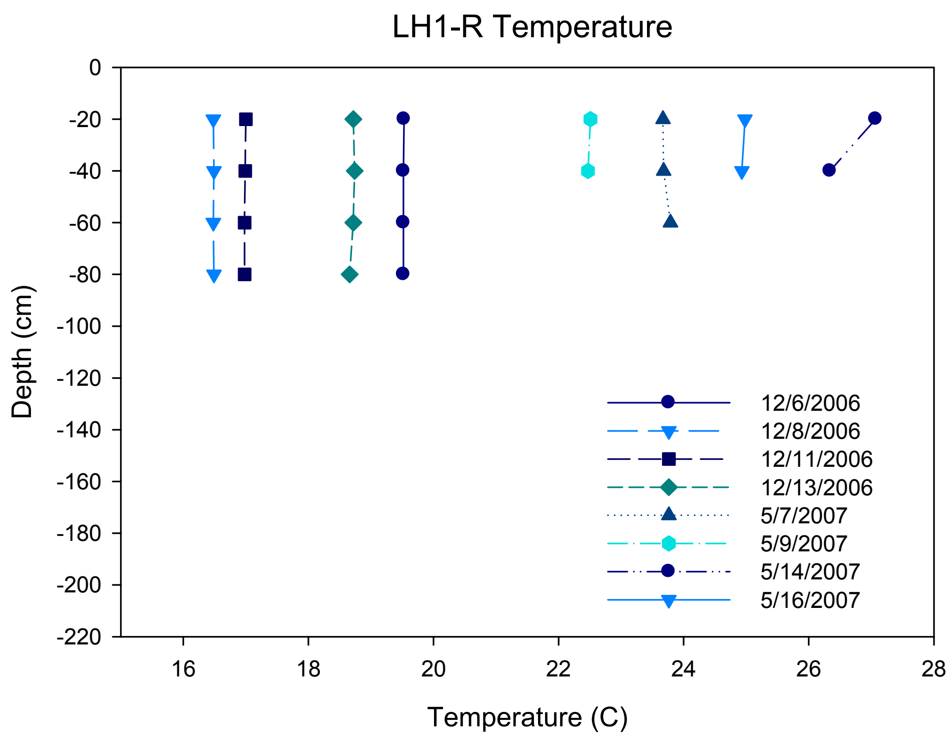
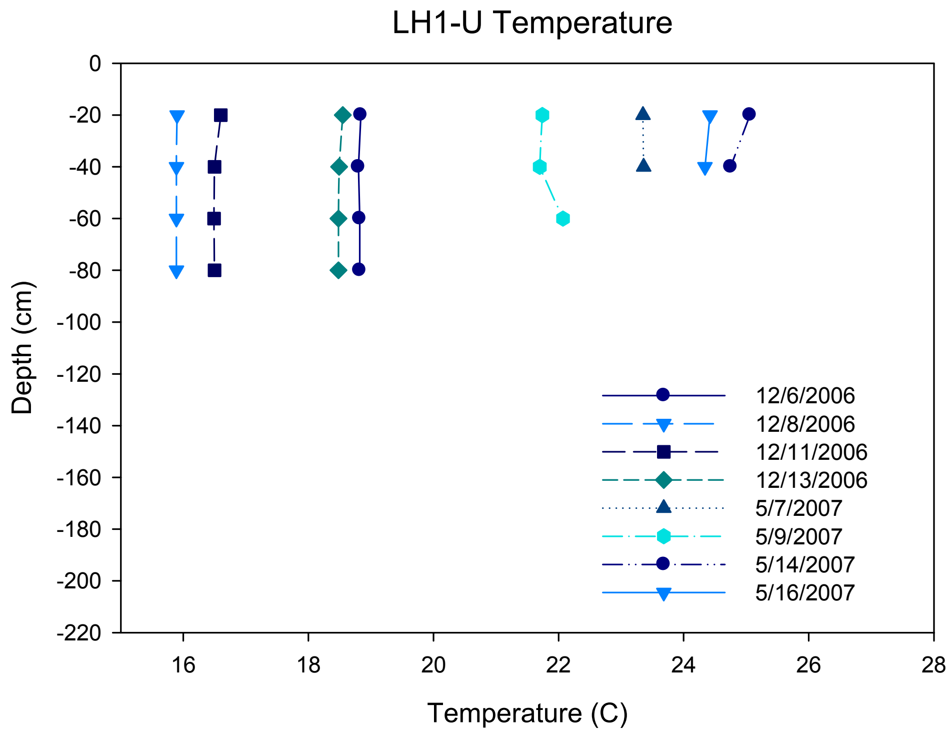
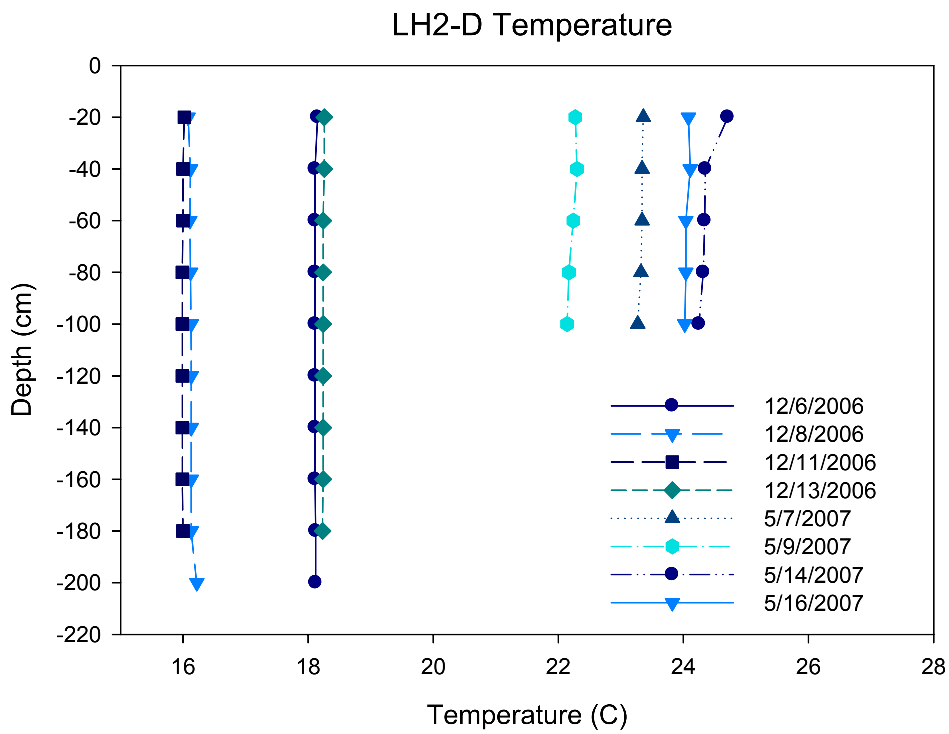

LH3-U Dissolved Oxygen
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
0
2
4
6
8
10
12
14
Dissolved Oxygen (mg/l)
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007

Temperature Profiles






LH1-U Temperature
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
16
18
20
22
24
26
28
Temperature (C)
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007


LH2-D Temperature
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
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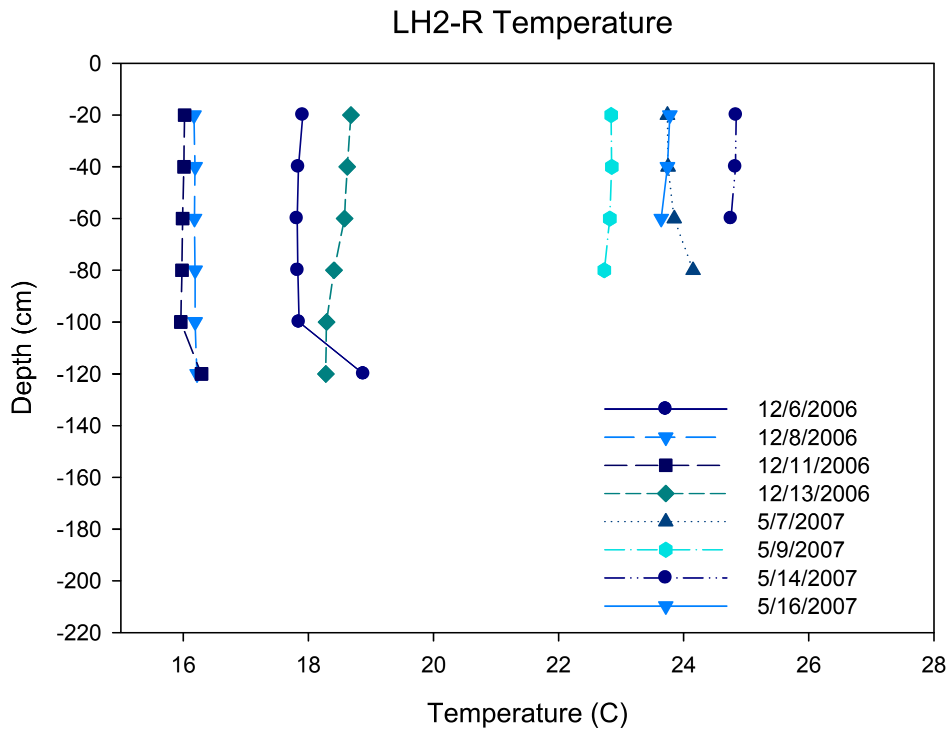
16
18
20
22
24
26
28
Temperature (C)
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007


LH2-R Temperature
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
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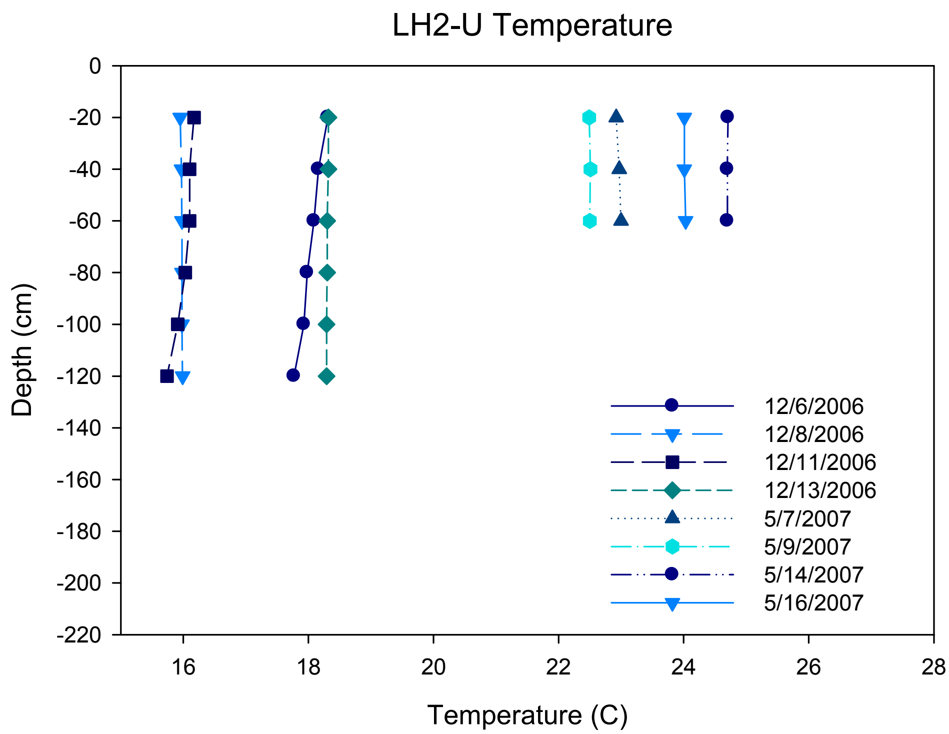
16
18
20
22
24
26
28
Temperature (C)
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007


LH2-U Temperature
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
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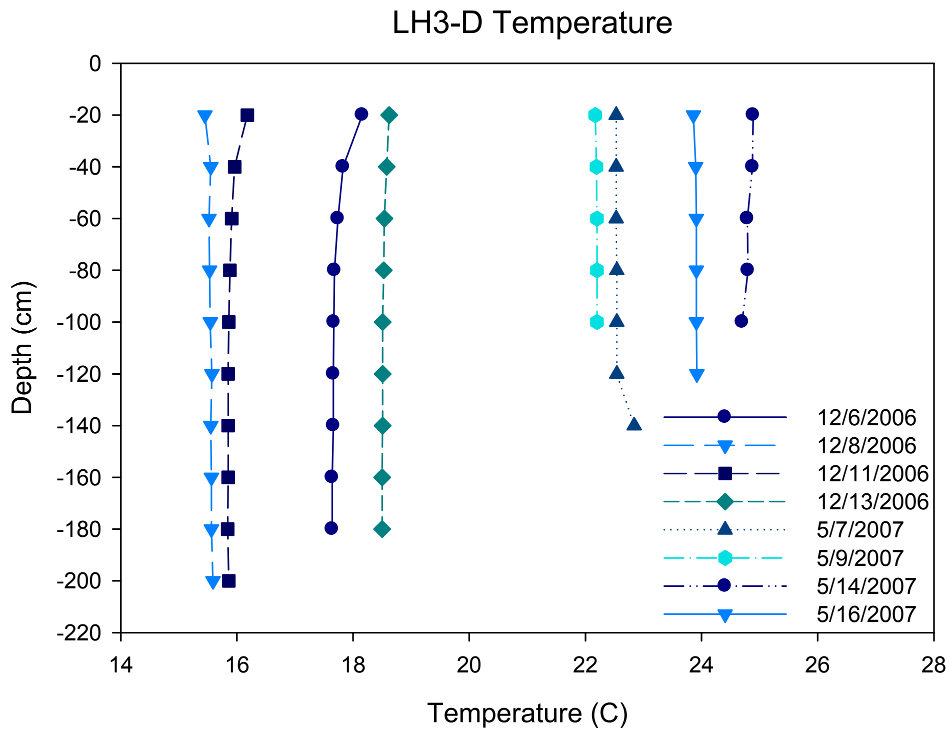
16
18
20
22
24
26
28
Temperature (C)
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007


LH3-D Temperature
Depth (cm)
Temperature (C)
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-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
14
16
18
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22
24
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28
12/6/2006
12/8/2006
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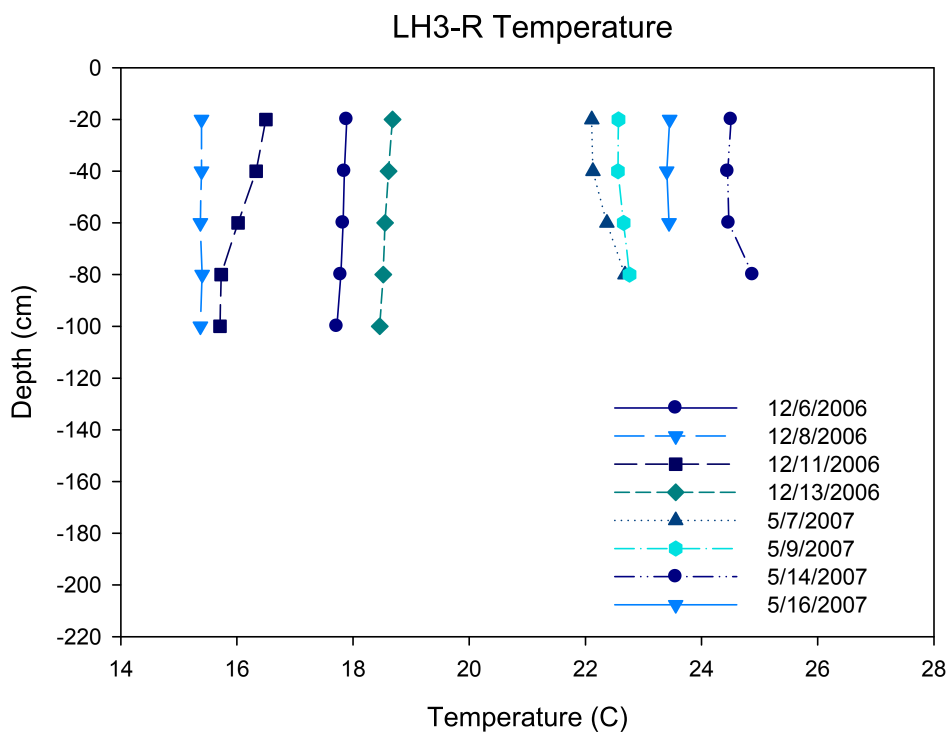
5/14/2007
5/16/2007


LH3-R Temperature
Depth (cm)
Temperature (C)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
14
16
18
20
22
24
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28
12/6/2006
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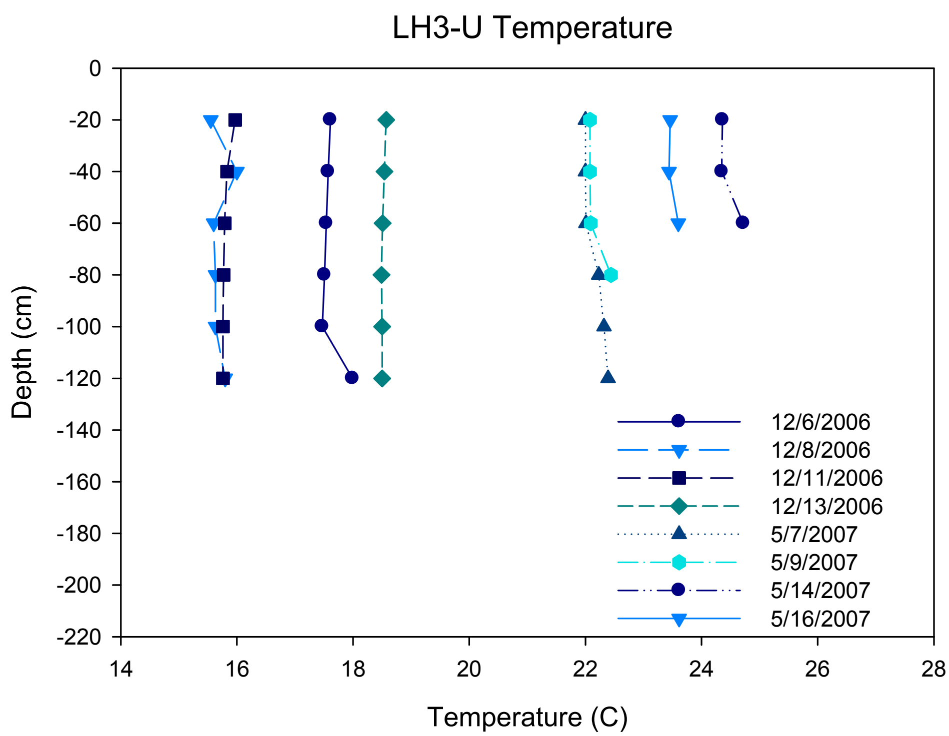
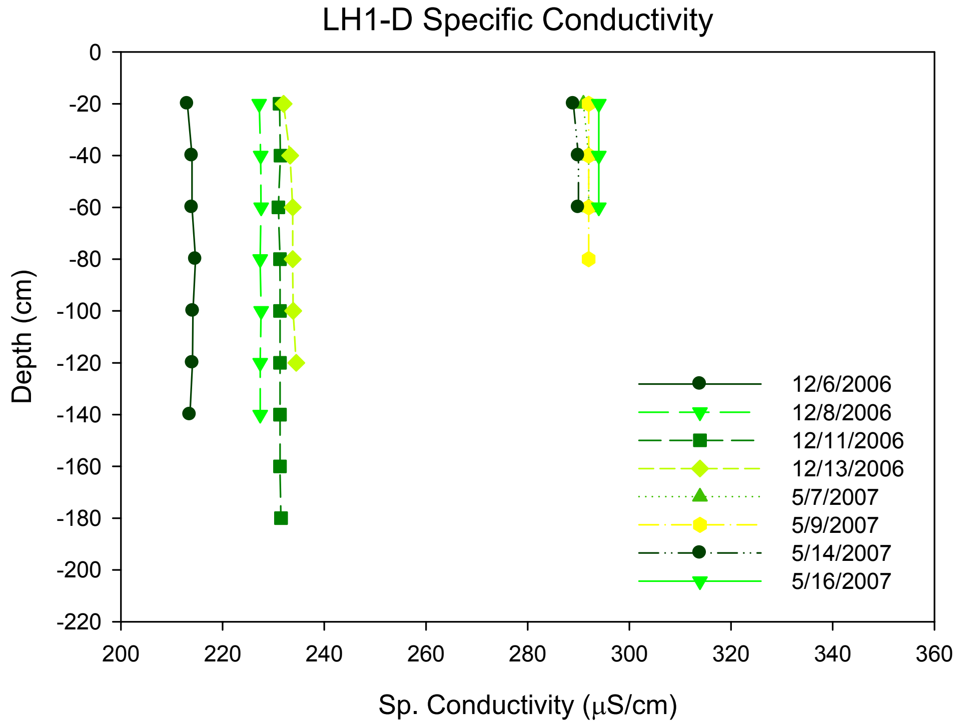
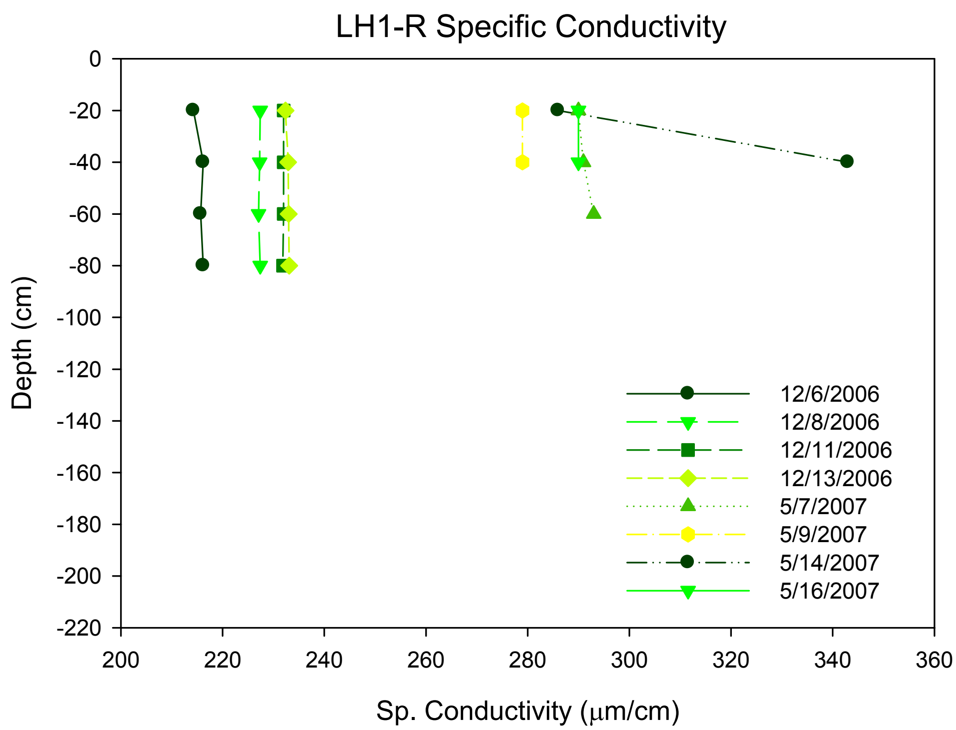
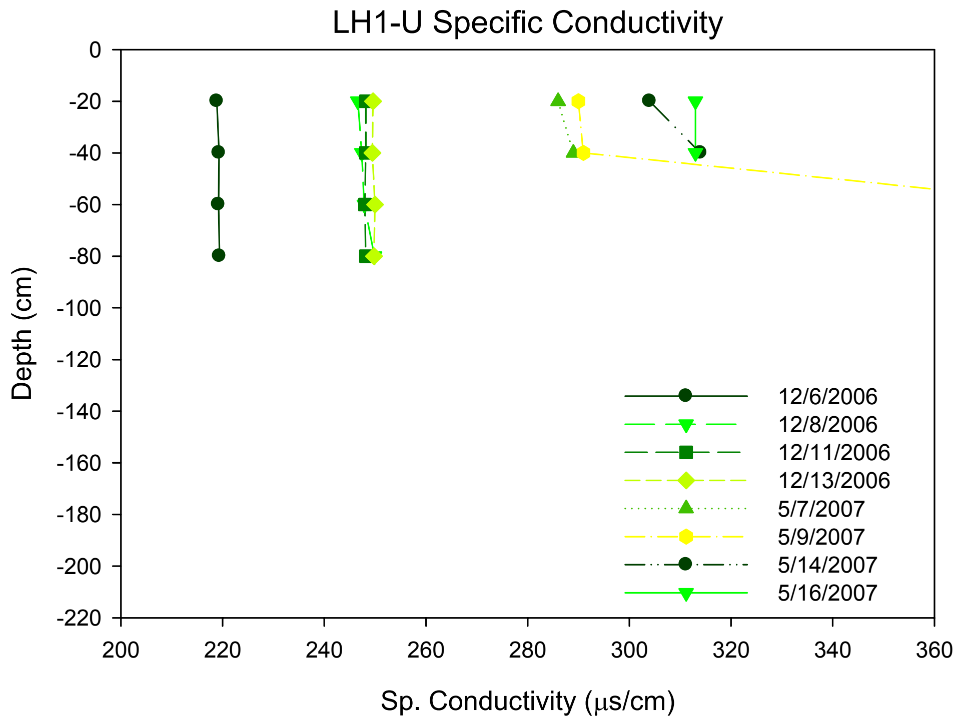
5/9/2007
5/14/2007
5/16/2007

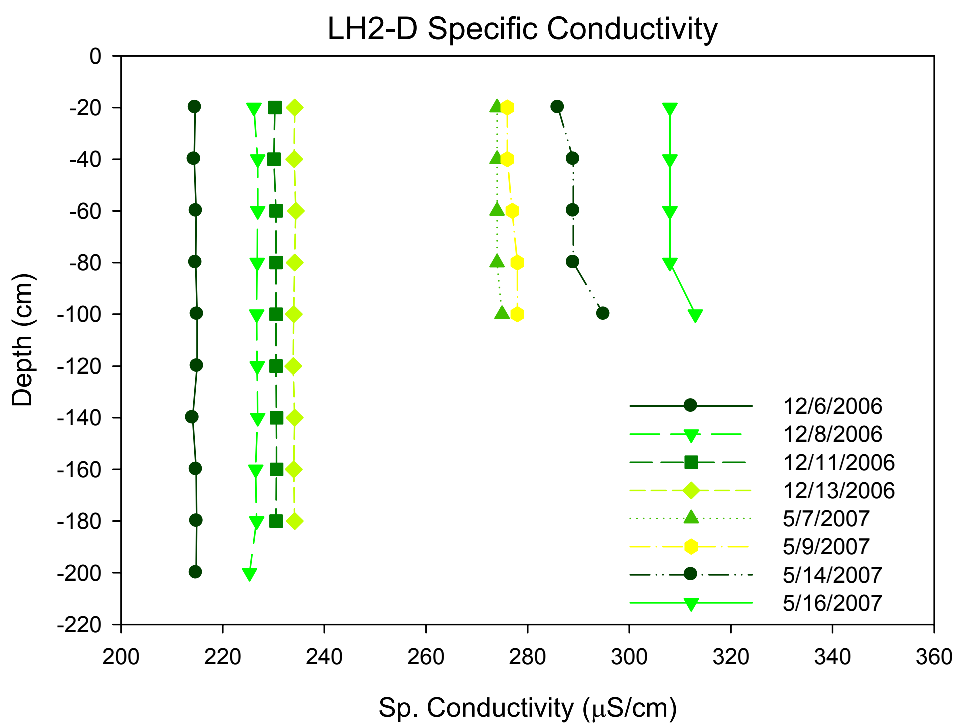

LH3-U Temperature
Depth (cm)
Temperature (C)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
14
16
18
20
22
24
26
28
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007

Specific Conductivity Profiles






LH1-U Specific Conductivity
Depth (cm)
Sp. Conductivity (µs/cm)
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007


LH2-D Specific Conductivity
Depth (cm)
Sp. Conductivity (µS/cm)
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
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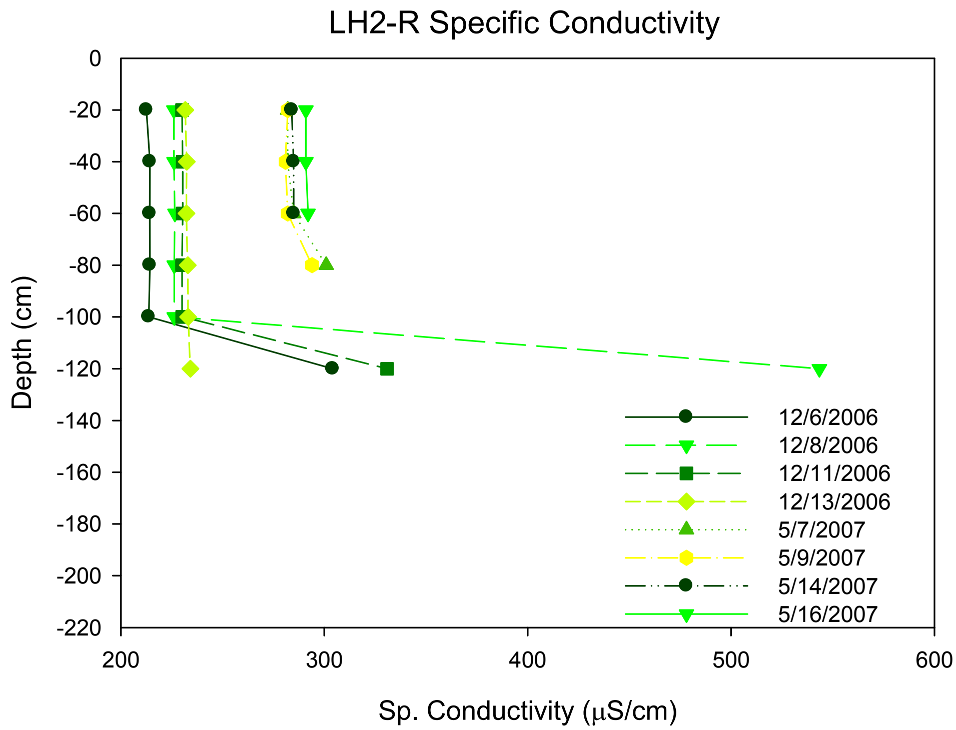
5/14/2007
5/16/2007



LH2-R Specific Conductivity
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
200
300
400
500
600
Sp. Conductivity (µS/cm)
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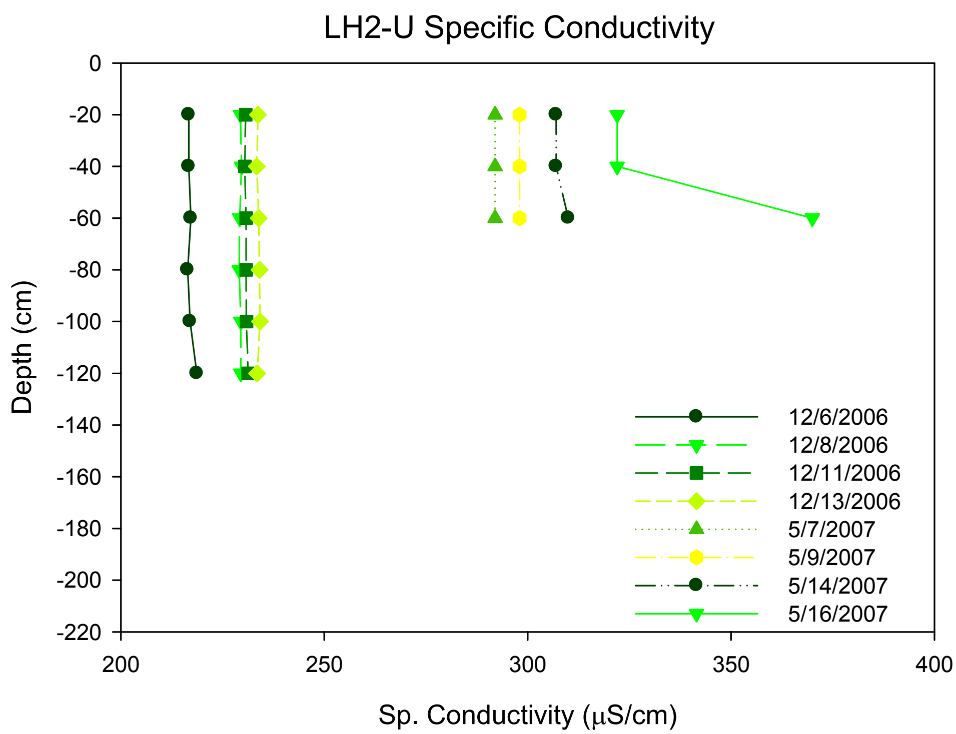
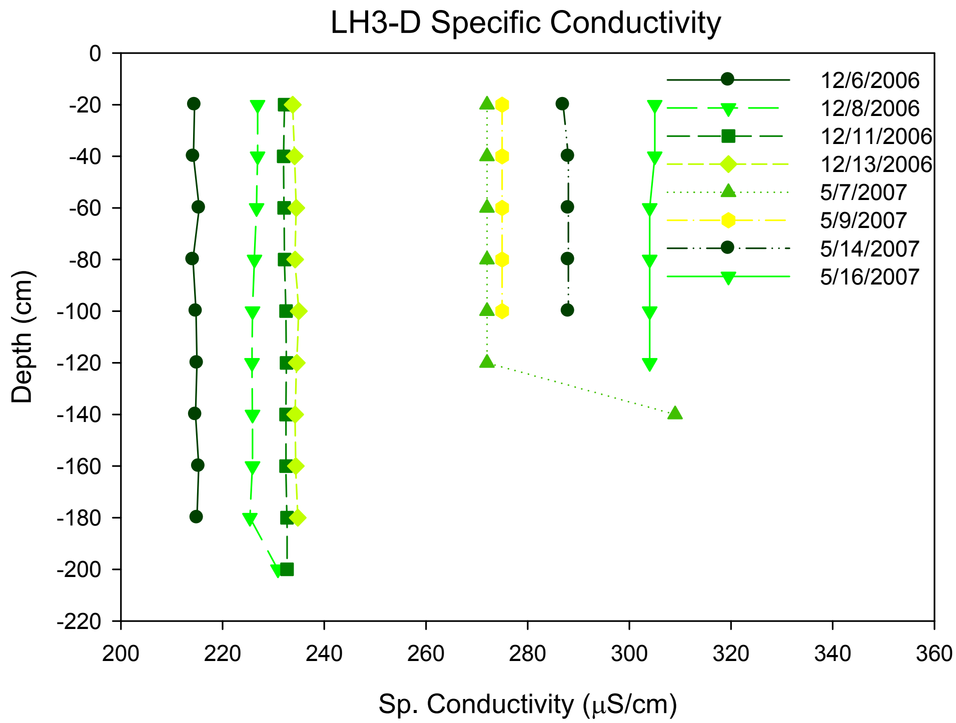
5/14/2007
5/16/2007



LH2-U Specific Conductivity
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
200
250
300
350
400
Sp. Conductivity (µS/cm)
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007


LH3-D Specific Conductivity
Depth (cm)
Sp. Conductivity (µS/cm)
0
-20
-40
-60
-80
-100
-120
-140
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-180
-200
-220
200
220
240
260
280
300
320
340
360
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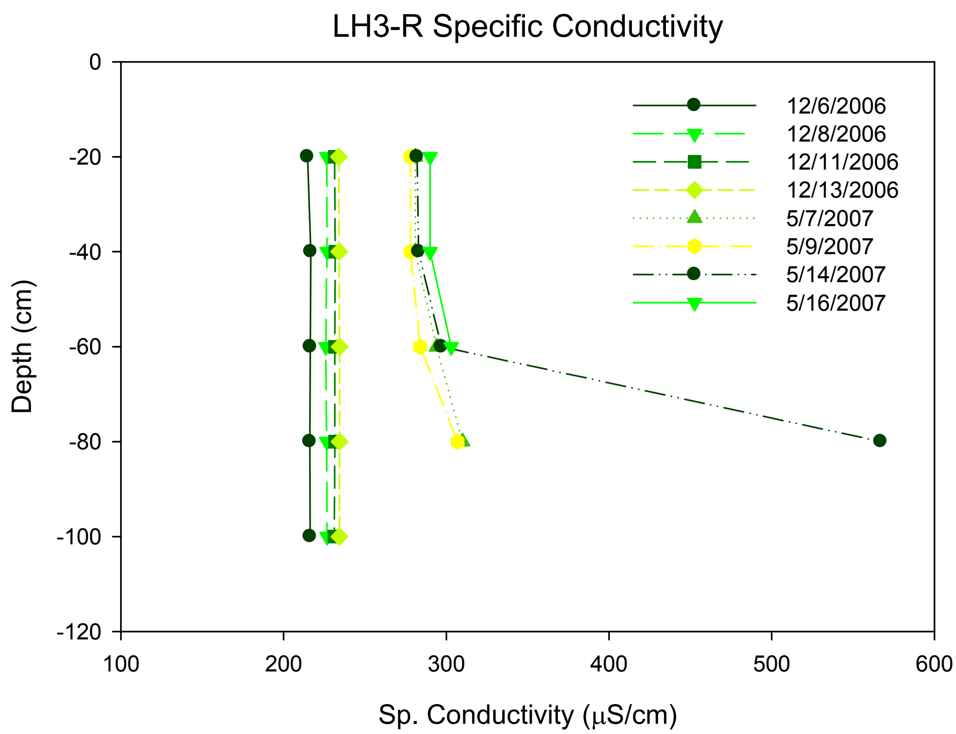
5/14/2007
5/16/2007


LH3-R Specific Conductivity
Depth (cm)
Sp. Conductivity (µS/cm)
0
-20
-40
-60
-80
-100
-120
100
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400
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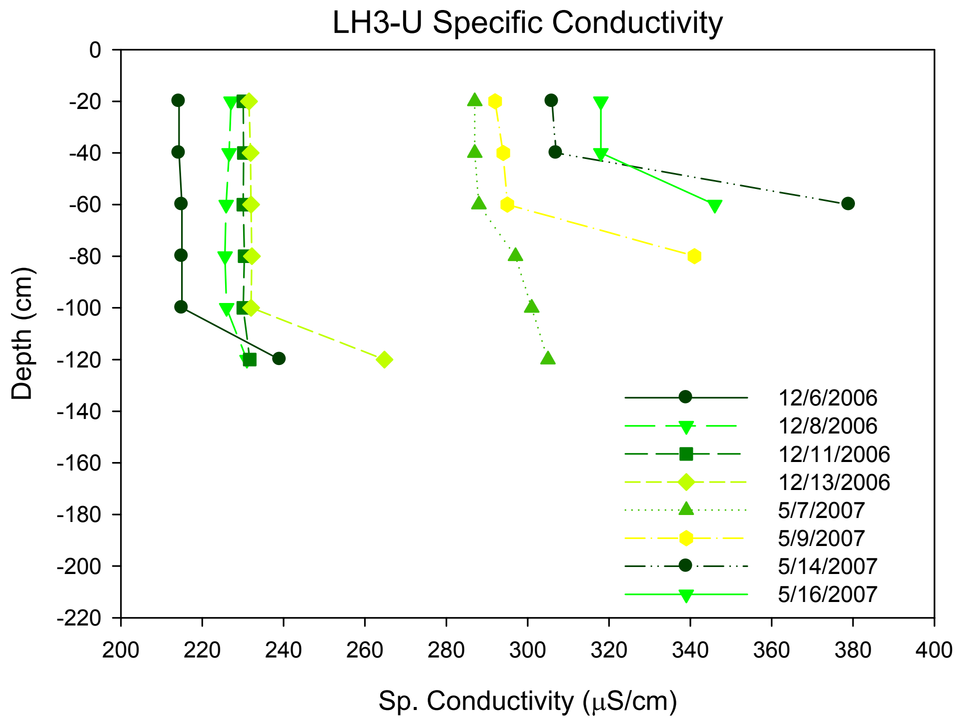
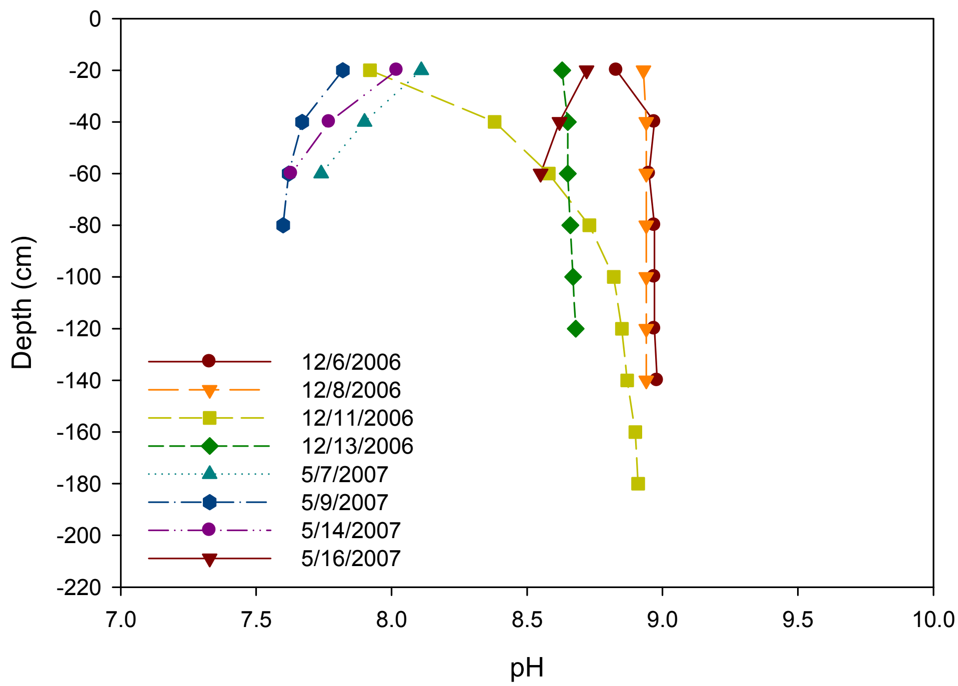
5/14/2007
5/16/2007



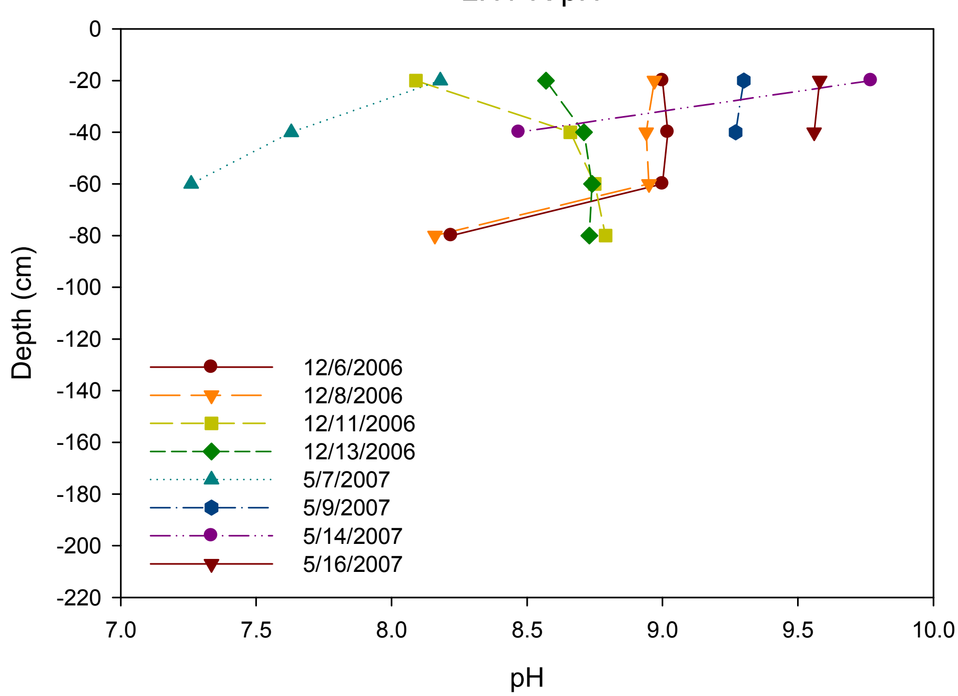
LH3-U Specific Conductivity
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
200
220
240
260
280
300
320
340
360
380
400
Sp. Conductivity (μS/cm)
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007

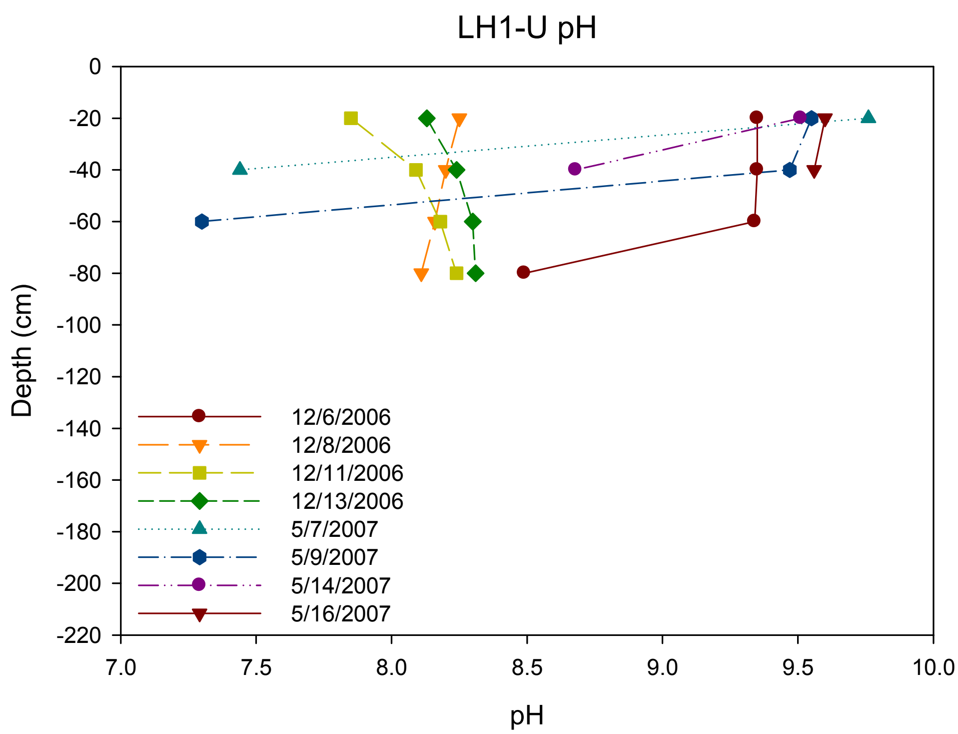
pH Profiles

LH1-D pH



LH1-R pH




LH1-U pH
Depth (cm)
pH
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
7.0
7.5
8.0
8.5
9.0
9.5
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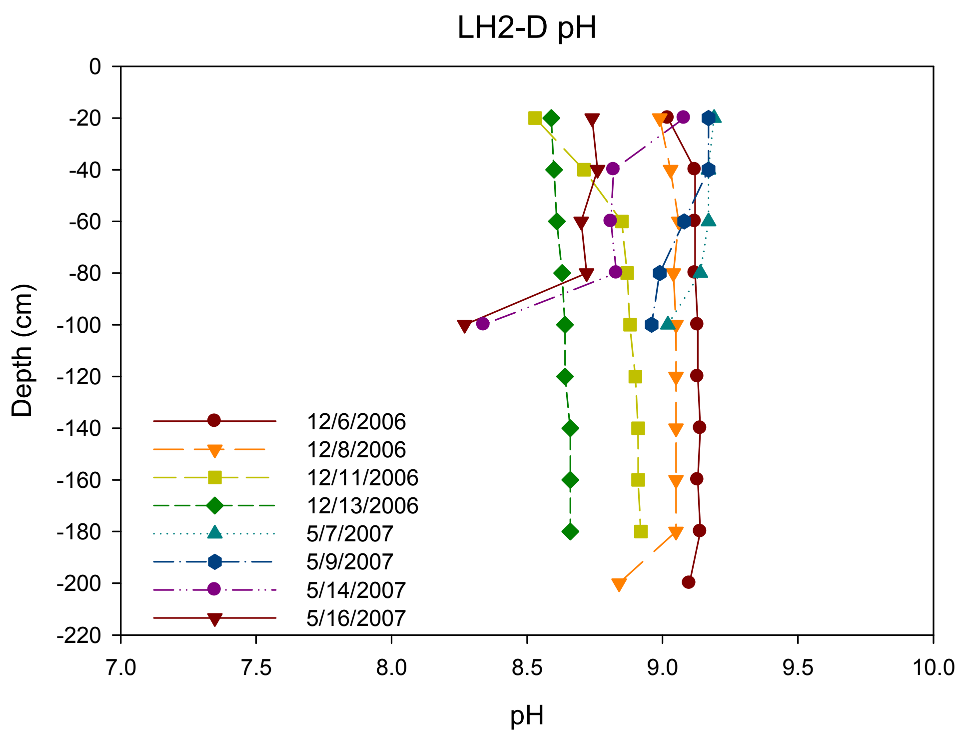
5/14/2007
5/16/2007


LH2-D pH
Depth (cm)
pH
0
-20
-40
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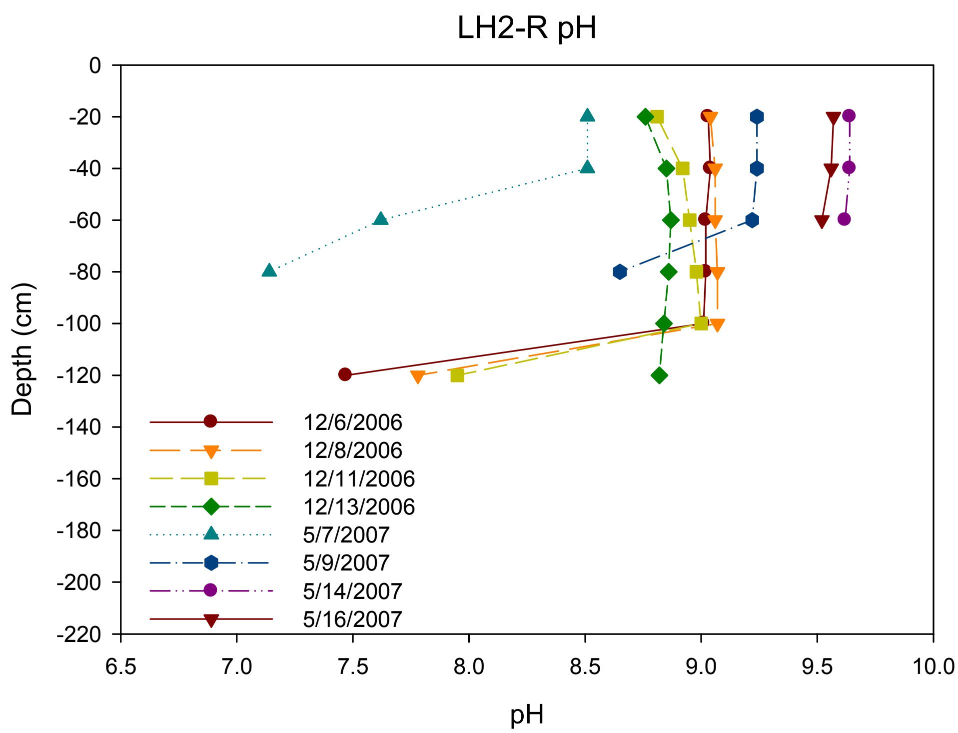
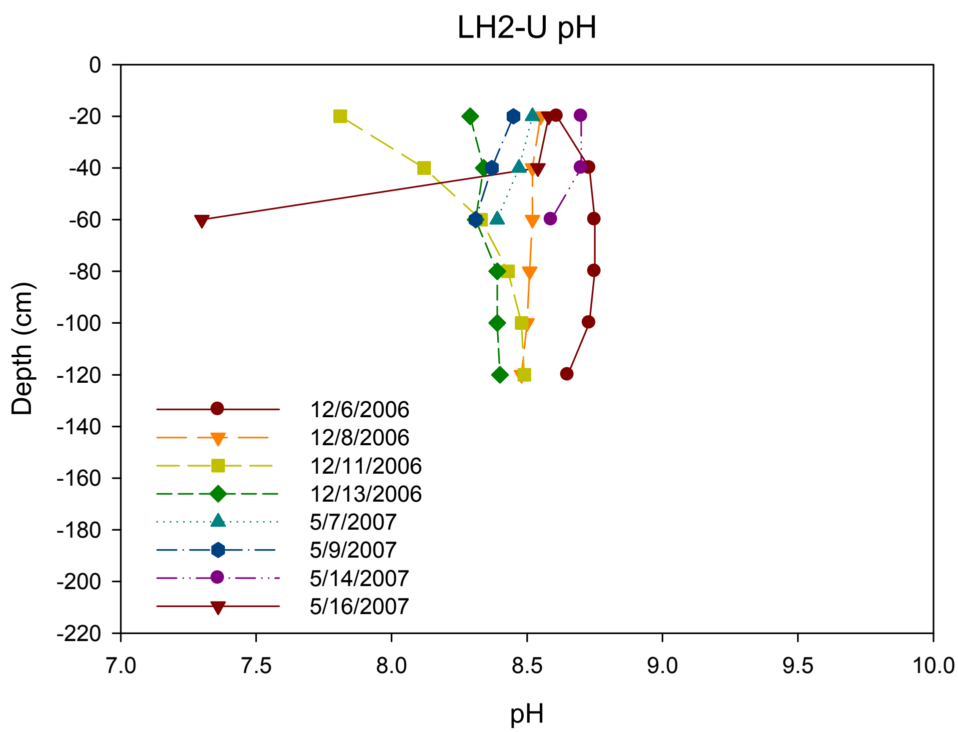
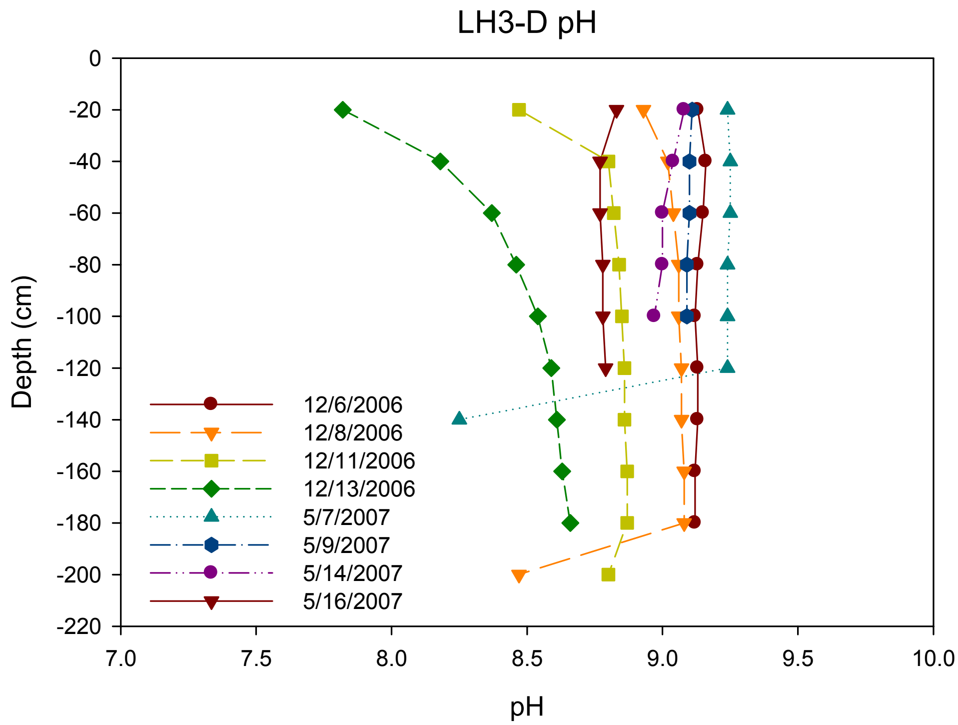
-80
-100
-120
-140
-160
-180
-200
-220
7.0
7.5
8.0
8.5
9.0
9.5
10.0
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007

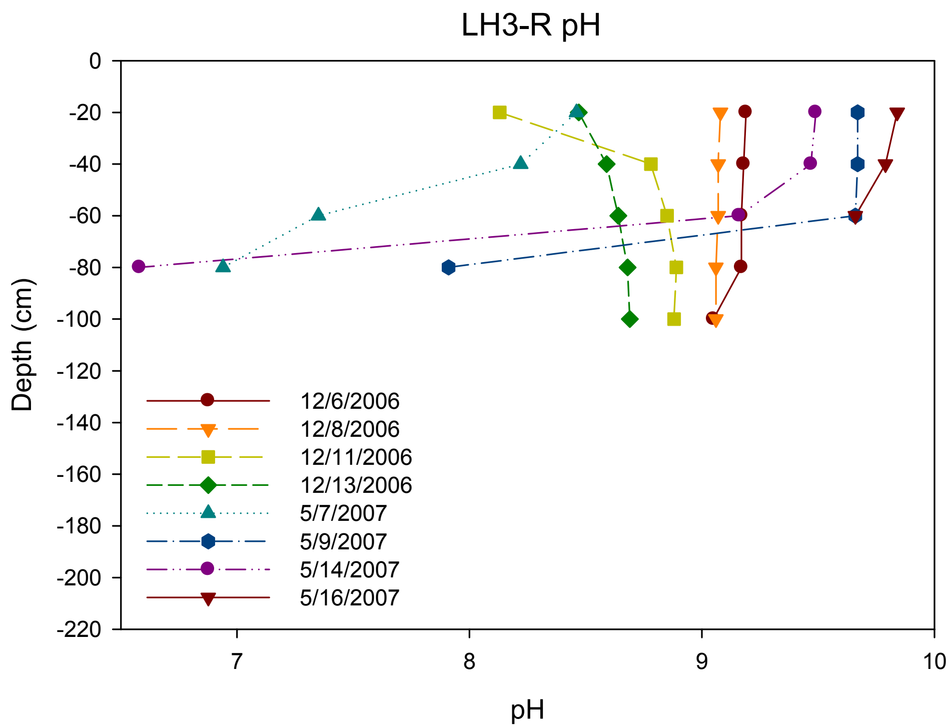


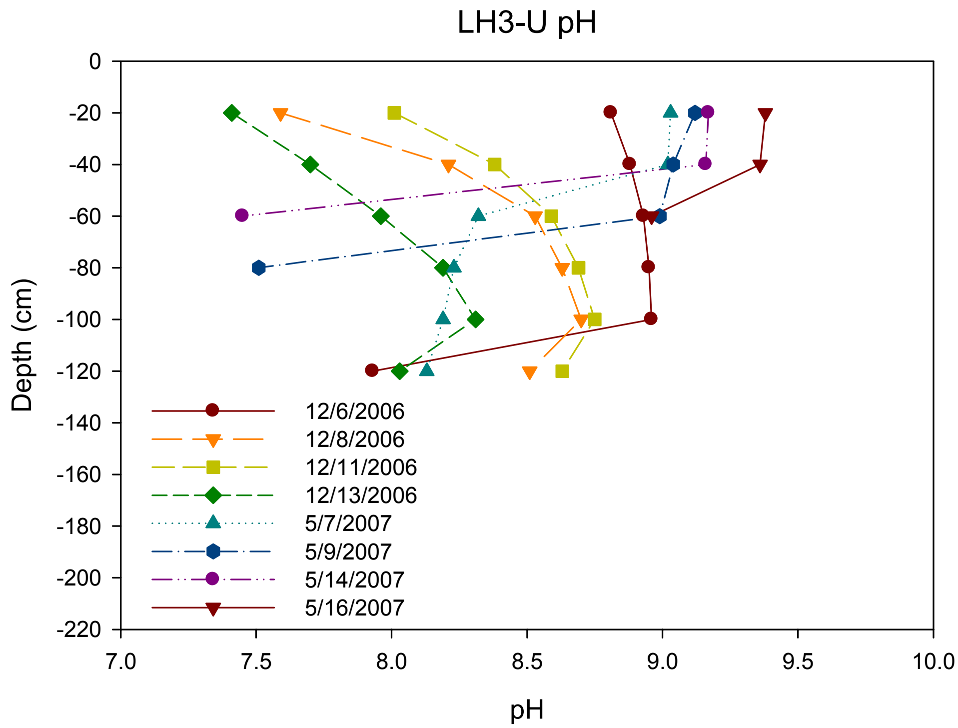
LH2-R pH
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
6.5
7.0
7.5
8.0
8.5
9.0
9.5
10.0
pH
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007



LH2-U pH
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
7.0
7.5
8.0
8.5
9.0
9.5
10.0
pH
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007


LH3-D pH
Depth (cm)
pH
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
7.0
7.5
8.0
8.5
9.0
9.5
10.0
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007


LH3-R pH
Depth (cm)
pH
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
7
8
9
10
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007


LH3-U pH
Depth (cm)
0
-20
-40
-60
-80
-100
-120
-140
-160
-180
-200
-220
7.0
7.5
8.0
8.5
9.0
9.5
10.0
pH
12/6/2006
12/8/2006
12/11/2006
12/13/2006
5/7/2007
5/9/2007
5/14/2007
5/16/2007