Physical, Chemical, and Biological Characteristics of Upper Klamath Lake, Oregon During Summer 2002

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Introduction

Since the late 1800's, summer fish kills have occurred in Upper Klamath Lake (UKL), one of the largest freshwater lakes in the Western United States. The principal populations of the federally listed endangered Lost River (*Deltistes luxatus*) and shortnose (*Chasmistes brevirostris*) suckers endemic to shallow UKL, Oregon are subject to periodic catastrophic and annual localized mortality events caused by adverse water quality such as critically low dissolved oxygen (DO) levels in late summer and hypereutrophic conditions. In recent years, from 1995 to 1997, such die-offs killed a substantial portion of the adult population of these suckers. These fish were abundant at the turn of the century but their numbers have declined substantially.

Water supplies for irrigated agriculture were severely curtailed in 2001, thereby maintaining high lake levels for protecting endangered suckers in UKL but there is considerable controversy regarding lake levels and water quality of the lake. Higher lake levels may result in reduced blue-green algal blooms and improved water quality conditions due to dilution effects. Shallower water depths may promote mixing, aeration, and possibly higher DO levels. Determining suitable water elevations (higher vs. lower) would be a step forward in providing some relief for poor water quality since at present it is impossible to predict what lake elevation would be appropriate to alleviate any of the problems at UKL. The occurrence of low DO levels is a primary contributor to adverse water quality and catastrophic fish die-offs in UKL. Water quality in the reservoir is thought to be greatly influenced by the dynamics of the blue-green algae *Aphanizomenon flos-aquae* (AFA) (Kann, 1998). If fewer suckers of all life-stages die from adverse water quality, more will contribute to sucker population sizes and reproduction.

The focus of Phase 1 of this study was to collect 1) physical data, nutrients, total suspended solids (TSS), and chlorophyll *a* during algal blooms, and again during algal collapse to determine additional sampling needs for a UKL monitoring program; and 2) water respiration rates from dusk to dawn to be used in a risk assessment model (for Phase 2) (Miranda et al. 2001) for determining potential estimates of lake area that would be affected by critically low DO. The greatest value from predictions derived from risk assessment models is recognition and description of generalized patterns, not specific quantitative predictions.

Study Site

Upper Klamath Lake is located in south-central Oregon on the eastern slopes of the Cascades. UKL is a large, shallow lake that covers 36,360 ha of surface area and a mean depth of 2.4 m. Williamson River provides the major inflow waters entering the lake. Since 1921, discharge from UKL has been controlled by Link River Dam. Historically, UKL was considered eutrophic (Kann, 1998), progressively becoming hypereutrophic through the years (U.S. Army Corps of Engineers, 1978). A recent paleolimnological study by Eilers et al. (2001) found that cyanobacteria have been present in the lake for more than a thousand years. Nutrient concentrations in lake sediments increased during the 20th century, and AFA, a species not present 150 years ago, showed major increases in recent decades. These changes accompanied changes in land use, including deforestation, wetland drainage, and increased agricultural activity.

Methods

UKL was sampled at 12 sampling sites (Fig. 1) from July 22 to 25 and from September 9 to 10, 2002. Stations 1 to 11 and ERS1 (called ERS in this report) (Fig.1) are all well established sampling sites in the northern part of the lake from which historical data exists. Station 8 represents Shoalwater Bay, Station 11 represents Ball Bay, and Station ERS1 was located south of the established ERS site. Depth profiles for water temperature, DO, pH, and specific conductance were collected from surface to bottom at 1 m depth increments with a Hydrolab multiparameter probe (Surveyor 3). Secchi depth transparency was recorded using a 20 cm black and white disk. Duplicate water samples for chlorophyll analysis were taken from below the surface (0.5 m) and just above the bottom with a Van Dorn sample bottle. From 250 to 500 ml of water was filtered through a Whatman GF/C filter (47mm, particle retention 1.0 μm) immediately after collection. Filters were sealed in coin envelopes, stored on dry ice in a cooler, and frozen until processed. Samples were extracted with 90 % acetone, analyzed with a spectrophotometer for chlorophyll *a* (chl *a*) by Reclamation's Environmental Chemistry Laboratory (DECL), Denver Technical Service Center (TSC) (Standard Methods, 1998, Method 10200 H(2)). All of the above samples were collected during early morning and late afternoon/evening.

Water samples for soluble reactive phosphorus (SRP) (0.001 mg/L detection limit (dl)), total phosphorus (TP) (0.003 mg/L dl), nitrate+nitrite nitrogen (NO_3+NO_2-N) (0.003 mg/L dl), ammonia-nitrogen (NH_3-N) (0.003 mg/L dl), total Kjeldahl nitrogen (TKN) (0.05 mg/L dl), and total suspended solids (TSS) (4 mg/L dl) were collected from Stations 5, 6, 7, 8, 11, and ERS (Fig. 1) at the surface (0.5 m) and just above the bottom. Five hundred milliliters were collected for TSS and 125 ml for all nutrients. Water samples for NH_3-N, NO_3+NO_2-N, and SRP were filtered through Gelman cellulose acetate filters (0.45 µm pore size) and stored on ice; samples for TP and TKN were unfiltered and acidified with 0.5 mL of 10 % sulfuric acid. Morning nutrient collections began at dawn. Nutrients were again collected from late afternoon into evening. All nutrient samples were shipped on ice within 48 hours to DECL for analysis. Nutrients were analyzed with a Perstorp segmented flow analyzer. Un-ionized ammonia concentrations were calculated according to Messer et al. (1984), and Holdren and Montano (2002). TSS was determined gravimetrically after drying at 103 to 105 °C.

Triplicate water samples for determining water respiration rates were collected in late afternoon, from the surface to immediately off the bottom, using a weighted swimming pool hose at Stations 1 to 11 and ERS. Water was emptied from the hose into a bucket and initial DO was recorded with a portable DO meter. Dark BOD bottles were filled to the top, capped with dark stoppers, and then submerged in a cooler filled with lake water.

Initial time was recorded to indicate start of incubation. Additional water was added to the cooler as needed to keep BOD bottles at lake water temperature while all bottles were filled at each station. All BOD bottles were then transferred from the cooler to bottle racks and deployed until dawn directly in the lake at the Shoalwater Bay boat ramp until dawn. Final DO, temperature, and time were recorded at dawn for each BOD bottle. Oxygen consumption was calculated from change in DO over time (days) recorded in $g/m^3/day$.

Results and Discussion

Water Temperature

Water temperature ranged from 21 to 28 °C during sampling in July and from 14.5 to 22 °C in September at 0.3 to 0.5 m (Figs. 2a-13a). Perkins et al. (2000) found that mean daily water temperatures were 16 to 22 °C during those periods (1995 to 1997) corresponding with fish kills. Warmest water temperatures were recorded from the shallowest stations (Stations 1 and 2) in the late afternoon of July 24 (a calm period). The previous sampling day had been breezy. Temporary thermal stratification was recorded at most stations for this brief calm period (Figs. 2a-13a). In addition, a sharp thermocline (0.5 to 3.0 m) developed in the afternoon of both July 23 and 24 in the trench at Station ERS (Fig. 13a), following isothermal conditions that existed in the morning hours. During September sampling, the lake was typically unstratified in the morning and the water column became weakly stratified by late afternoon which is common in shallow hypereutrophic lakes. Sheffer (1998) reported that temporary thermal stratification in calm weather had the potential for lowering DO in the water column.

Temporary thermal stratification of the water column coinciding with reduction in DO occurred only periodically during this study.

Dissolved Oxygen

DO concentrations fluctuated dramatically from morning to evening as well as from one day to the next (Figs. 2b-13b). This is typical of hypereutrophic lakes (Barica, 1980) and has been well documented in past summers at UKL (Burleson, 2002; Perkins et al., 2000; Kann, 1998). Critically low DO concentrations below 4 mg/L were observed occasionally. Lowest bottom DO levels at sampling stations were: 10 mg/L at very shallow Stations 1 and 2; 6 mg/L (July) at Station 5; 5 mg/L (July) at Station 7; 4 mg/L (July) at Station 3; 4 mg/L (September) at Station 6; 3.5 mg/L (September) at Station 9 and 11(Ball Bay); 3.0 mg/L at Station 10 (July); 2.8 mg/L (July) at Station 4; 2.5 mg/L (July) at Station 8 (Shoalwater Bay); and 0.09 mg/L (July) at Station ERS (trench). The lowest DO levels below 1.0 m in depth were consistently reported at the trench (Station ERS) in July.

Reduced DO levels were recorded for the calm morning of July 23, 2002, following a night of severe rain storms, increased mixing in the lake, and on-set of temporary stratification (Figs. 2b-13b). DO levels of < 4 mg/L at a depth of 1 m only occurred at Station 4 and in Shoalwater Bay at Station 8. Perkins et al. (2000) describes in detail the lethal and sublethal effects of DO levels on fish (i.e. Lost River and shortnose sucker juveniles). Fish kills coincided with prolonged DO levels typically < 4 mg/L for 10 to 24 hours per day at a depth of 1 m. By evening of July 23, DO levels increased in the water column. Anoxic conditions also occurred from 3 to 11 m at Station ERS (trench) the afternoon of July 24. Perkins et al. (2000) observed that 1997 was the only year (from 1995 to 1997) that DO of < 4 mg/L occurred at Mid North even though fish kills were observed for all three years. DO levels below 4 mg/L were not reported from Station 6 (Mid North) during this study. During this study, DO levels changed rapidly and critically low levels disappeared within a 12 hour period. Low DO levels were mostly restricted to bottom depths below 1.0 m. In fact, chl *a* continued to peak in September and did not appear to be in sharp decline as observed in previous years (Perkins, 2000).

Oxygen percent saturation was often greater than 100 % and at times reached over 200 % of saturation at the upper depths of the water column and in the late afternoon/evening hours coinciding with DO levels that often exceeded 15 mg/L. DO levels increased to a high of 17 mg/L in the lake in September at Station 1. DO supersaturation commonly occurred at the shallowest stations (Stations 1 and 2). At the somewhat deeper stations, oxygen saturation declined rapidly in the water column. During July sampling, oxygen saturation decreased to 35 % at Station 8 and declined to 1 % of saturation at Station ERS.

pH

Wide diurnal fluctuations of surface pH levels ranged from 8.6 to 9.7. Levels approaching or exceeding pH 9.5 were only observed at shallow Stations 1, 2, and the deep trench (Station ERS) (Figs. 2c-13c). Day to day fluctuations were often more dramatic at the shallower stations where mixing throughout the water column rapidly occurred during windy periods. Morning pH readings were typically lower than afternoon/evening pH readings, common in eutrophic lakes that undergo intensive photosynthesis of phytoplankton (Wetzel, 1975). During the evening of July 24, the pH gradient from surface to bottom depths decreased sharply at many of the stations, corresponding with calm weather conditions. The pH declined always below the top one-meter of the water column. Near surface pH levels were similar for both July and September, most often above 9.0 and at times above pH 9.5. High levels of pH may cause phosphorus releases from bottom sediments and promote higher un-ionized ammonia levels (Kann and Smith, 1999; Holdren and Montano, (2001). Perkins et al. (2000) reported that mean pH declined below 9.0 in late summer each year, and corresponded to declines in algal blooms. During the fish kills of the 1990's, the pH was less than 9.0 and highest fish kills occurred when pH levels were 7.5 to 8.5. Levels remained above 8.5 throughout this study both in July and September mainly due to fluctuating algal biomass as indicated by chl *a* concentrations. In fact, greater peaks in chl *a* concentrations occurred in September than in July at Stations 3, 6, and 10. During this study there were no observed signs of a massive algal die-off at any of the sampling stations, and therefore pH remained at high levels.

Secchi Depth

Low secchi depth transparencies, ranging from 0.3 to 1.6 m, were unremarkable and comparable to other hypereutrophic lakes (Holdren and Montano, 2002). Secchi depth readings were greatly influenced by weather conditions, resuspended sediment, and patchiness of AFA. Lowest secchi depths were consistently recorded from Station ERS (0.6 to 0.7 m), coinciding with high chl *a* concentrations.

Table 1. Range of secchi depths from Stations 1 to 11 and ERS based on sampling conducted during morning and afternoon hours in July and September, 2002.

Station	Secchi Depth Range (m)
1	0.3-0.6
2	0.5-0.9
3	0.9-1.2
4	1.1-1.2
5	1.2-1.6
6	0.9-1.2
7	0.7-1.5
8	0.4-1.5
9	1.0-1.3
10	1.0-1.2
11	1.1-1.3
ERS	0.6-0.7

Chlorophyll *a*

Chl *a* concentrations fluctuated widely from morning to evening, from day to day, from July to September, from station to station, and by depth (Figs. 14-19). It was common for chl *a* to change by 100 to 200 µg/L, and by as much as 600 µg/L (Station 8 in Shoalwater Bay) (Fig. 17b) within a 12 to 24 hour period. The distribution of AFA in the water column appeared to be influenced more by calm vs. windy conditions than by time of collection (morning vs. late afternoon).

Chl *a* was comprised mostly of the dominant blue-green alga *Microcystis* sp. was also observed occasionally floating on the surface. At times, bottom concentrations surpassed near surface chl *a* during both calm and breezy periods at Stations 1, 2, 3, 4, 5, 7, 9 and 10. By contrast, Burleson (2002) reported that chl *a* concentrations during summer 2001 were always greater near the surface than near bottom depths. Water column mixing in this study may have provided a mechanism by which algae were propelled into bottom depths with reduced light. However, AFA did not remain in limited light conditions for extended periods of time. Perkins et al. (2000) indicated AFA that existed in conditions with reduced light for long periods of time may actually have contributed to the algal crash, and consequently hypoxia and fish kills from 1995 to 1997.

Greatest surface chl *a* concentrations of 1328 µg/L were reported from Station ERS (trench area) on the calm evening of July 24, 2002. There were also high chl *a* concentrations of 782 µg/L reported from Station 8 in Shoalwater Bay on the cloudy, smoky, and calm morning of July 25; chl *a* the previous evening was only 112 µg/L.

During this study, AFA continued blooming in September and an algal crash did not appear to be eminent indicated by levels of pH , DO, and chl *a*. A large algal decline preceded each fish kill in 1995 to 1997 as reported by Perkins et al. (2000) and a steep decline in chl *a* was reported from their Mid North station (Station 6 in this study). Chl *a* levels at Station 6, peaked to about 170 µg/L on the morning of September 10, 2002 (increasing by 100 µg/L from the previous evening), coinciding with levels of acceptable DO. Further supporting, that algal biomass was still blooming at some stations in the lake.

Nutrients

Nutrients fluctuated greatly at all stations sampled (5, 6, 7, 8, 11, and ERS) (Figs. 20-25). Nutrient levels showed great diurnal variability due to resuspended sediment fluctuations, AFA distribution, DO levels, pH swings, and most importantly weather conditions. For example, ammonia nitrogen levels were extremely unpredictable and were not always higher in the morning than in late afternoon as reported by Burleson (2002).

SRP values were generally above 0.040 mg/L (Figs. 20a–25a). Maximum SRP levels generally occurred at the bottom depths during July and were variable in September. Surface SRP levels ranged from 0.025 to 0.088 mg/L at 0.1 m and 0.004 to 0.107 mg/L at bottom. Greatest SRP levels were recorded at bottom depths from Station ERS on July 24 and 25 during a calm period when the station was stratified and anoxic. At times, minimum bottom DO and pH levels coincided with maximum SRP at any given station.

TP levels were greater than 0.100 mg/L (Figs. 20b-25b) at all stations. TP levels at Station 8 (Shoalwater Bay) increased to 0.575 mg/L on July 25, coinciding with peaks in chl *a*. Twelve hours earlier, TP levels were about 0.200 mg/L at 0.1 m (Fig. 23b). Greater surface TP concentrations of 0.800 mg/L were reported on July 24 at Station ERS, and levels of 1.6 mg/L occurred on September 9, 2002. Spikes in TP levels were also observed at Stations 5 and 6 in September. An important source of phosphorus in UKL is derived from bottom and resuspended sediments under elevated pH levels. Elevated pH increases phosphorus flux to water column by solubilizing iron-bound phosphorus in both bottom and resuspended sediments as high pH causes increased competition between hydroxyl ions and phosphate ions decreasing the sorption of phosphate on iron (Welch, 1992). The pH levels typically ranged from 9.0 to over 9.5 at all stations.

Ammonia concentrations varied with depth, time of day, and weather conditions. Morning ammonia-nitrogen levels were not always greater than levels present in late afternoon (see Figs. 20c-25c, Station 6 morning of Sept. 9, 2002). In addition, surface NH_3-N levels were greater than bottom levels at times. Ammonia-nitrogen (NH_3-N) levels ranged from 0.024 to 0.800 mg/L. Higher levels of NH_3-N occurred at Stations 8, 11, and ERS (Shoalwater Bay, Ball Bay, and trench area, respectively). TKN ranged from 1.3 to 31.0 mg/L (Figs. 20d-25d). Greatest TKN levels were reported during a calm period from 0.1 m at Station ERS. Nitrate-nitrite nitrogen levels ranged from 0.005 to 0.040 mg/L (Figs. 20e-25e). Lowest levels were present at Station ERS, and remained below 0.016 mg/L.

Un-ionized ammonia levels (Table 2) were calculated based on pH and temperatures according to Holdren and Montano (2002). NH_3 (un-ionized ammonia) levels of 0.02 mg/L is the listed U.S. EPA (1976) criterion for freshwater aquatic life whereas the range of 0.20 to 2.0 mg/L is the acute lethal concentration for a variety of fish. In UKL, during July and September, levels were often found to be near or exceeding the 0.02 mg/L level at Stations 5, 6 (mid-north), and 7. Surface and bottom NH_3 concentrations always exceeded the 0.02 mg/L level at Stations 8, 11, and ERS. Un-ionized ammonia exceeded 0.20 mg/L at Station ERS on the evening of July 23 and levels observed to cause toxic effects in freshwater systems. By contrast, during 1997, summer un-ionized ammonia commonly exceeded the acute lethal concentration of 0.20 mg/L, and peaked to 2.0 mg/L in Shoalwater Bay (Perkins et al. 2000) whereas summer levels were lower in 1995 and 1996. During all these years, un-ionized ammonia was < 0.1 mg/L during the peak of the fish kill (Perkins et al., 2000) and coincided with decline in pH and chl *a* (Perkins et al., 2000).

Table 2. Unionized ammonia levels (NH_3) at Stations 5, 6, 7, 8, 11, and ERS. Samples were collected at surface (0.1 m) and immediately above the bottom depth.

Station	Date	Time	NH_3 (mg/L)	NH_3(mg/L)
			surface	bottom
5	7/23/2002	722	0.021	0.019
5	7/23/2002	1720	0.017	0.013
5	7/24/2002	1725	0.012	0.016
5	7/25/2002	845	0.014	0.013
5	9/9/2002	845	0.012	0.008
5	9/9/2002	1820	0.018	0.007
5	9/10/2002	800	0.019	0.023
5	9/10/2002	1811	0.020	0.018
6	7/23/2002	705	0.081	0.026
6	7/23/2002	1740	0.043	0.038
6	7/24/2002	1745	0.029	No data
6	7/25/2002	705	0.008	0.007
6	9/9/2002	820	0.013	0.007
6	9/9/2002	1800	0.02	0.046
6	9/10/2002	745	0.017	0.018
6	9/10/2002	1745	0.033	0.022
7	7/23/2002	625	0.018	0.014
7	7/23/2002	1850	0.094	0.084
7	7/24/2002	1936	0.009	0.028
7	7/25/2002	856	0.008	0.009
7	9/10/2002	733	0.029	0.028
7	9/10/2002	1720	0.038	0.025
8	7/23/2002	1910	0.092	0.077

8	7/24/2002	2010	0.034	0.024
8	7/25/2002	845	0.020	0.050
8	9/9/2002	730	0.018	0.026
8	9/9/2002	1720	0.096	0.042
8	9/10/2002	1905	0.055	0.054
11	9/9/2002	905	0.056	0.019
11	9/9/2002	1658	0.066	0.046
11	9/10/2002	833	0.044	0.036
11	9/10/2002	1640	0.070	0.057
ERS	7/22/2002	1715	0.084	0.084
ERS	7/23/2002	815	0.106	0.076
ERS	7/23/2002	1525	0.277	0.021
ERS	7/24/2002	1515	0.466	0.003
ERS	7/25/2002	800	0.133	0.035
ERS	9/9/2002	655	0.027	0.076
ERS	9/9/2002	1530	0.016	0.063
ERS	9/10/2002	635	0.027	0.074
ERS	9/10/2002	1520	0.070	0.113

The ratio of N:P often is a good indicator of the limiting nutrient in aquatic systems (Reynolds, 1986). The TN (TKN+ NO_3-N +NO_2-N) :TP ratios at Stations 5, 6, 7, 8, 11, and ERS were very similar. TKN comprised the greatest proportion of TN. The ratio ranged from 13:1 (Station 5) to 16:1 (Station ERS), indicating that algal growth will be limited by the supply of phosphorus present with a N:P ratio, > 7 (Reynolds, 1986). Kann and Walker (2001) explained that in UKL, a shallow hypereutrophic lake, algal biomass in general and blue-green algae in particular would possibly show substantial reductions in response to reduction in P loading, even when P concentrations remain in the hypereutrophic range (> 0.100 mg/L TP) after restoration. Ratios of TIN (NH_3-N + NO_3-N + NO_2-N) to SRP often provide a better indication of nutrient limitation than TN:TP ratios due to inorganic nutrient forms that are more directly available to support algal growth. This may be important in UKL, in particular, where significant amounts of

N and P may be tied up in algal cells. Ratios of TIN:SRP ranged from 1.5:1 to 5:1, indicating nitrogen limiting (N:P < 7) (Schindler, 1977), not phosphorus limiting conditions. Even so, the dominant nitrogen-fixing blue-green alga, AFA, has the ability to augment nitrogen uptake in a nitrogen-limiting system (Kann, 1993; 1998). High phosphorus input and relative nitrogen deficiency resulting in a low N to P ratio is characteristic of hypereutrophic lakes (Barica, 1980).

Total Suspended Solids

TSS were typically below 20 mg/L (Fig. 26) with some variation in depth. Mean surface TSS ranged from 6 to 27 mg/L and mean bottom TSS ranged from 8 to 29 mg/L. TSS levels were variable due to weather conditions. During windy periods, TSS levels increased as much as 4 to 5 fold from the previous sampling collection. At times, TSS was higher at bottom depths (Stations 5, 6, and ERS) than at the surface during these peak events. TSS appeared to settle out rapidly in UKL after brief periods of sediment resuspension from wind action. Peaks in TSS coincided with increasing chl *a* concentrations at Stations 5, 6, 7, and 11. Elevated pH contributes to release of phosphorus from bottom and resuspended sediments thereby elevating AFA biomass and pH even further.

Water Respiration Rates in the Water Column

During periods of AFA blooms and crashes, oxygen consumption rates from dusk to dawn indicate depletion of accumulated DO from daytime photosynthesis. The variability between samples was high due to patchiness of AFA at some of the stations (Table 3), resulting in differences in final concentrations of algae in each BOD bottle.

Table 3 . Water respiration rates (gO_2/m^3/day) at Stations 1 to 11, and ERS for July 23 to 24 and September 9 to 10, 2002. Water samples were collected in late afternoon/evening, and BOD bottles were incubated from dusk to dawn at Shoalwater Bay.

Date	Station	Depth (m)	Mean (gO_2/m^3/day)	Std Dev (gO_2/m^3/day)	Date	Station	Depth (m)	Mean (gO_2/m^3/day)	Std Dev (gO_2/m^3/day)
7/23/2002	1	0-1.0	3.71	1.28	9/9/2002	1	grab	3.08	1.94
7/24/2002	1	0-1.0	2.60	2.13	9/10/2002	1	grab	11.71	0.38
7/23/2002	2	0-1.25	4.76	1.16	9/9/2002	2	0-0.5	3.12	0.80
7/24/2002	2	0-1.25	4.39	0.87	9/10/2002	2	0-0.5	5.06	0.26
7/23/2002	3	0-1.5	2.38	1.40	9/9/2002	3	0-1.0	3.99	2.13
7/24/2002	3	0-1.5	3.67	1.73	9/10/2002	3	0-1.0	7.63	0.92
7/23/2002	4	0-2.25	3.71	0.79	9/9/2002	4	0-1.75	3.32	1.29
7/24/2002	4	0-2.25	1.09	1.19	9/10/2002	4	0-1.75	3.05	2.59
7/23/2002	5	0-1.5	2.59	0.79	9/9/2002	5	0-1.0	3.48	0.19
7/24/2002	5	0-1.5	1.47	0.61	9/10/2002	5	0-1.0	4.34	1.11
7/23/2002	6	0-2.25	1.94	0.39	9/9/2002	6	0-1.5	2.80	1.63
7/24/2002	6	0-2.25	2.62	1.21	9/10/2002	6	0-1.5	5.59	0.35
7/23/2002	7	0-2.5	1.33	0.31	9/9/2002	7	0-2.0	4.13	0.90
7/24/2002	7	0-2.5	2.88	0.87	9/10/2002	7	0-2.0	4.66	0.93
7/23/2002	8	0-2.5	1.33	0.81	9/9/2002	8	0-2.0	1.49	0.33
7/24/2002	8	0-2.5	3.90	0.39	9/10/2002	8	0-2.0	1.96	0.60
7/23/2002	9	0-2.0	2.13	0.90	9/9/2002	9	0-1.5	3.71	3.18
7/24/2002	9	0-2.0	5.56	1.19	9/10/2002	9	0-1.5	5.24	1.37
7/23/2002	10	0-3.25	2.67	1.81	9/9/2002	10	0-2.5	5.43	0.60
7/24/2002	10	0-3.25	4.13	0.21	9/10/2002	10	0-2.5	2.32	0.15
7/23/2002	11	0-2.0	1.80	0.53	9/9/2002	11	0-1.5	2.21	0.98
7/24/2002	11	0-2.0	3.60	0.36	9/10/2002	11	0-1.5	1.83	0.09
7/23/2002	ERS	0-11.0	4.62	1.64	9/9/2002	ERS	0-11.0	2.92	0.60
7/24/2002	ERS	0-11.0	4.11	0.79	9/10/2002	ERS	0-11.0	3.72	0.36

Algal material was not uniformly dispersed in the water column (see discussion on chl *a*). Much of the patchiness in the water column was the result of windy conditions during a specific late afternoon/evening collection. Mean respiration rates ranged from 1.33 to 4.62 gO_2/m^3/day on July 23 and from 1.09 to 5.56 on July 24. Mean respiration rates ranged from 1.49 to 5.43 gO_2/m^3/day on September 9 and from 1.83 to 11.71 gO_2/m^3/day on September 10. Highest respiration rates occurred on September 10 at Stations 1, 3, 6, and 9. Some of the lowest rates were reported from Stations 8, 11, and ERS (Shoalwater Bay, Ball Bay, and the trench area). By contrast, Wood (2001) found the sediment oxygen demand (SOD) to be greater than 10.2 g/m^2/day in late summer at Ball Bay (Station 11). A SOD value of this magnitude could potentially deplete the water column of oxygen in a few days (Wood, 2001), whereas peak uptake in the water column could completely deplete O_2 in less than one day even without any SOD.

Respiration rates were generally greater in September than in July indicating signs of decreasing algal biomass, although Stations 6, 9, and 10 had greater maxima chl *a* in September. One of the objectives of sampling during summer 2002 was to collect oxygen consumption rates during the bloom and then collapse of AFA. AFA blooms were captured in July but it was difficult in catching the collapse of the monoculture in September during these collections. The algae were undergoing blooms at some stations and gradually dying off at others. A massive die-off of AFA causes anoxia due to accelerated respiratory oxygen demand (Miranda et al., 2001) and these conditions would have presumably resulted in higher respiration rates than what has been reported in the data (Table 3). Miranda et al. (2001) reported water respiration rates (WR) of 4.56 to 16.8 gO_2/m^3/day and an average of 10.56 in a hypereutrophic oxbow lake in Mississippi. These rates are higher than what was found at UKL, primarily due to greater summer water temperatures. Oxygen consumption rates from summer 2002 will be used further in an oxygen risk assessment model (Miranda et al. 2001) to assess the importance of lake level management in UKL during periods of critically low DO from mid to late summer.

Conclusions

Physical, nutrient, TSS, and chl *a* data during July and September 2002 reflected a highly dynamic hypereutrophic lake. Limnological interactions in UKL are extremely complex, compounded by weather conditions (calm vs. windy days), and variable lake elevations. Drawing conclusions about the interactions of the physical, chemical, and biological processes in the lake are therefore difficult.

DO levels approached critically low concentrations (< 4 mg/L) at 1.0 m depth for only short periods at Stations 4, Station 8 (Shoalwater Bay), and Station ERS (deep trench). Un-ionized ammonia reached toxic levels (> 0.2 mg/L) only at Station ERS (trench). AFA was widely dispersed but patchy and greatest chl *a* levels occurred at Station 8 and ERS. Water respiration rates were higher in September than July, but there was not a total collapse of AFA during this study. In summary, data were highly variable and fluctuated widely.

Water column respiration data from this study will be used as input into a DO risk assessment model for predicting suitable water levels to alleviate critically low DO. Water elevation is one of the most controversial issues involving hypereutrophic conditions of UKL and previous results give no clear indications of relationships between lake levels and DO concentrations (NRC, 2002).

All other data collected from this study, as well as, the wealth of information that already exists will be used to formulate a future UKL plan. It is difficult to come up with solutions to the many problems in the lake when interactions are not fully understood by the many studies that have already been conducted on this system. An additional future monitoring study will only supplement all of the present data collected. Finding the "correct" next step that will lead to alleviating some of the problems of UKL is one of the most difficult tasks at hand.

Our recommendation at present would be to compile all data that have been collected in the past, evaluate what has been done, and determine if it is possible to link weather data with lake elevation and DO. We realize that weather conditions are so variable that this may be an impossible task to complete.

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