Upper Klamath Lake monitoring program: preliminary analysis of status and trends for 1990-2009

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Introduction

The Klamath Tribes, in a cooperative agreement with the Bureau of Reclamation, have been monitoring water quality variables in Upper Klamath (UKL) and Agency lakes (AL) since 1990 at locations throughout the two lakes (Table A.2, Figure 2.1). Currently, limnological data are collected biweekly from April through October at 10 standardized stations (Kann 2008). This report summarizes water quality data collected over the last 20 years, including the more important patterns and trends as well as related hydrological and climate variables. Measurements of chlorophyll-*a* (Chl) receive particular emphasis because of the central role of phytoplankton in determining water quality in these hypereutrophic lakes. The phytoplankton is dominated by the bloom-forming cyanobacterium *Aphanizomenon flos-aquae* (AFA). Temperature, wind, nutrients and lake level are also of special interest, as they are thought to underly variability in magnitude and timing of the blooms as well as water quality (Kann and Welch 2005). Dissolved oxygen (DO), pH and un-ionized ammonium also require special attention because—along with temperature—they affect habitat quality for two important fish populations in the lake: Lost River (*Deltistes luxatus*) and shortnose (*Chasmistes brevirostris*) suckers. The two populations have been listed as endangered since 1988.

Although the relatively small effort allocated for this study precludes an in-depth treatment of the underlying causal mechanisms for phytoplankton variability and its effect on water quality, it does include a preliminary examination of selected mechanisms underlying chlorophyll variability. By focusing primarily on differences among water quality variables by location, season and year of observation this study is intended to serve as a foundation for these larger goals, because any such differences can ultimately be used to suggest and test hypotheses regarding the underlying mechanisms. The primary goals of this report were to utilize the 20-year limnological data base to 1) determine spatial and temporal patterns among water quality variables, 2) perform preliminary time series trend analyses, and 3) provide an exploratory multivariate analysis of selected mechanisms underlying chlorophyll variability.

2

Analysis methods

2.1 Data sources

This report is based mainly on data collected by the Klamath Tribes monitoring program. The observed and derived variables obtained as part of the monitoring program are summarized in Table A.1. The methods are described in a report by the Klamath Tribes (2006), as well as earlier reports.

Although at least 15 sites have been sampled at times (Table A.2), of particular interest are nine core sites that have been sampled throughout the period 1990-01-18 to 2009-10-21: PM, NB, WB, ML, ER, SB, MN, AS and AN (the latter since 1991; Figure 2.1). The frequencies of individual measurements for each observed water quality variable at each core site are shown in Table 2.1.

Data averaged over the water column and month provide the basis for many of the analyses herein. Table 2.2 summarizes data availability by showing the number of core sites that have chlorophyll data for each month. As can be seen from this table, the month range May–September is of particular interest because it encompasses the onset and decay of the annual AFA bloom and is almost complete in terms of data availability. None of these months are missing for 1993, 1995 and 1998–2009, a total of 14 years. Comparisons among sites and years can be carried out without bias introduced by ignoring or interpolating missing months as long as we confine ourselves to this month range and these years. Some analyses are insensitive to the amount of missing data occurring here, in which case all years can be used. We can also increase the number of years for analysis when attention can be confined to a narrower month range such as June–July, the most likely period for the annual bloom maximum. In still other cases, the year range can be extended by using a subset of the core sites.

Climate data, obtained from the National Climatic Data Center on-line archive

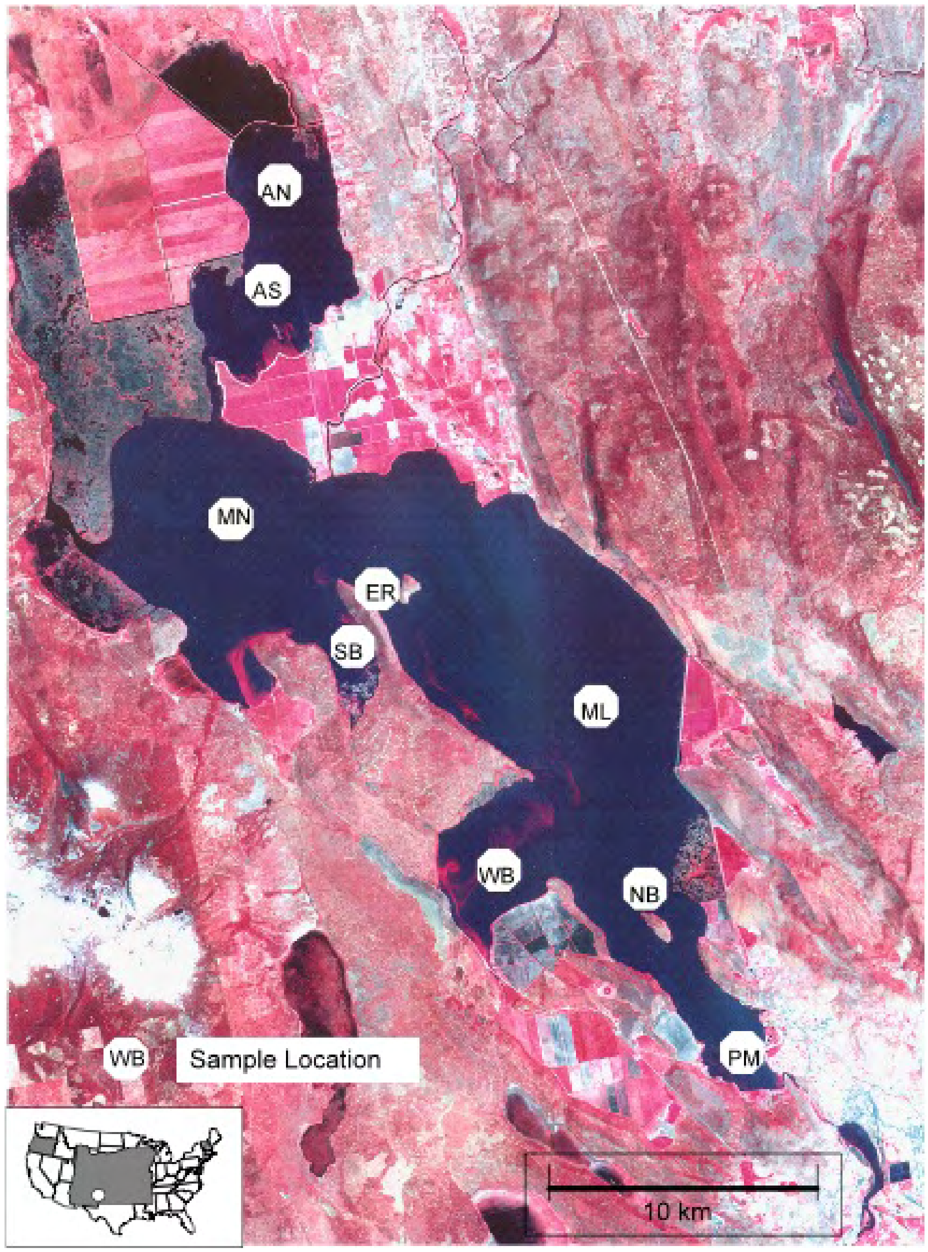

Figure 2.1: Upper Klamath Lake and Agency Lake monitoring sites.

Table 2.1: Upper Klamath Lake monitoring program: frequency of observations per variable and site, 1990-01-18 to 2009-10-21

	PM	NB	WB	ML	ER	SB	MN	AS	AN
SECCHI	1019	989	902	1233	2333	1193	1376	919	910
LIGHT	949	922	838	1106	1971	1038	1188	805	852
T	796	756	686	1012	2148	983	1178	752	717
DO	792	754	681	1009	2140	978	1174	751	717
PSAT	792	754	681	1009	2140	978	1174	751	717
PH	782	745	674	1000	2116	964	1160	739	713
COND	769	730	659	972	2059	937	1127	717	692
ORP	117	109	103	140	291	134	148	96	115
TP	251	247	246	265	295	278	284	273	218
SRP	248	246	245	262	290	275	284	273	219
TN	251	247	246	265	298	279	287	275	218
NO	237	234	234	251	274	266	274	262	212
NH	245	242	241	258	282	275	280	269	216
MCHL	244	250	246	268	291	275	284	275	221
PHAE	246	250	245	267	290	276	283	271	221

(`http://www.ncdc.noaa.gov/oa/ncdc.html`) are for the for the Klamath Falls Airport station (WBAN ID 94236). Lake and outflow hydrology were obtained from the Bureau of Reclamation hydrology database for the Klamath Basin Area (`http://www.usbr.gov/mp/kbao/operations/water/index.html`).

2.2 Data analysis

Trends in monthly data were determined by the nonparametric Seasonal Kendall test (Hirsch and Slack 1984). The overall trend slope is computed as the median of all slopes between data pairs within the same season (no cross-season slopes contribute to the overall slope estimate). This is sometimes referred to as the *Theil* or *Sen* slope. Two criteria were used to insure that data records for different variables and locations represented the same period so that trend results were comparable: Tests were conducted for a particular water quality variable and station only if at least 50% of the total possible number of monthly values in the beginning and ending fifths of the record were present in the record; in addition, more than 50% of the maximum possible number of comparisons had to be present for at least 75% of the months of interest.

We used Kendall's tau test to determine the significance of a monotonic time series trend for annualized data at individual sites (Mann-Kendall test; Mann 1945). We also used the Regional Kendall test, which is similar in concept to the Seasonal Kendall

Table 2.2: Number of sites with chlorophyll data available for each month (columns) and year (rows).

	1	2	3	4	5	6	7	8	9	10	11	12
1990	8	0	8	0	8	8	8	4	8	8	4	0
1991	0	8	0	8	0	9	9	9	9	8	9	0
1992	0	9	9	9	9	9	9	9	8	0	0	0
1993	0	0	0	9	9	9	9	9	9	9	9	0
1994	0	0	9	3	9	9	9	9	5	5	1	0
1995	0	1	1	9	9	9	9	9	9	9	9	1
1996	1	1	6	7	6	6	6	6	6	6	0	0
1997	1	1	9	8	9	9	9	9	8	8	0	0
1998	0	0	0	9	9	9	9	9	9	9	0	0
1999	0	0	0	0	9	9	9	9	9	0	0	0
2000	0	0	0	9	9	9	9	9	9	9	0	0
2001	0	0	0	0	9	9	9	9	9	9	0	0
2002	0	0	0	0	9	9	9	9	9	9	0	0
2003	0	0	0	9	9	9	9	9	9	7	0	0
2004	0	0	0	9	9	9	9	9	9	0	9	0
2005	1	1	1	1	9	9	9	9	9	9	1	1
2006	0	0	0	9	9	9	9	9	9	9	0	0
2007	0	0	0	9	9	9	9	9	9	9	0	0
2008	0	0	0	9	9	9	9	9	9	0	0	0
2009	0	0	0	9	9	9	9	9	9	9	0	0

test, to increase the power of detection by combining results from individual sites (Helsel and Frans 2006). None of these Kendall tests were corrected for correlation in space or time, which means that the significance levels may be biased low. This has no effect when the tests do not suggest a trend, i.e., the null hypothesis is not rejected, but it implies that a smaller than usual p-value may be advisable for accepting a trend on the basis of the tests. Although the tests can be adjusted for serial correlation, in this preliminary investigation no adjustments were made.

Shifts in time series were amplified visually by first scaling the series—substracting the long-term mean and dividing by the standard deviation—and then transforming each observation of the scaled series by calculating the cumulative sum of all values up to and including that observation. This *CUSUM* transformation is commonly used for monitoring industrial processes and has also been employed in many ecological studies (Manly 2008).

We used multidimensional scaling—also known as *principal coordinates analysis*—to portray the difference among sites with regard to water quality characteristics (Cox and Cox 1994). Sites are located on a 2-dimensional plot, with similar sites clustering together. Multidimensional scaling takes a set of dissimilarities among the measurements at different sites and returns a set of points such that the distances between the points are approximately equal to the dissimilarities. We used Euclidean distances as a measure of the dissimilarity among time series observations of water quality variables for different sites.

We also used *mixed-effects models*, which describe the relationship between a response variable and one or more predictor variables grouped according to some classification (Pinheiro and Bates 2000). The models consist of *fixed effects*, which are parameters associated with the entire population, and *random effects*, which are associated with the individual classifications. In this report, random effects represent the different sampling sites. Models with multiple variables were subjected to a backward elimination of factors, and nested models were compared using a likelihood ratio statistic with associated $p-$value. A value of $p < 0.05$ was used to indicate a significant deterioration of model performance when a factor was dropped.

Box plots shown in this report are classic box-and-whiskers plots: Horizontal line is the median; box is the interquartile range (IQR); whiskers extend to all points within 1.5 times the interquartile range; and solid circles are outlying points. A notch is drawn in each side of the boxes in some plots. If the notches of two different plots do not overlap this is "strong evidence" that the two medians differ (Chambers et al. 1983).

All calculations and tests, unless otherwise specified, were carried out with the R language and environment for data analysis (R Development Core Team 2009).

3

Water quality characteristics of the core sites

3.1 Lake-wide summaries

In this section we summarize water quality data on a lake-wide basis and for Upper Klamath and Agency lakes separately. The distinction between the two lakes is made both on a geographical and a water quality basis. Although Upper Klamath and Agency lakes tend to differ in a number of attributes, there can also be greater differences among stations within a given lake than between lakes, depending on the water quality variable and time.

Table 3.1 summarizes the individual measurements by their means and quartiles for each variable (both lakes combined). The lake is shallow, with a median depth of 2.6 m at the sampling sites, and turbid, with a median Secchi depth of less than 1 m. It is hypereutrophic, total phosphorus (TP) and total nitrogen (TN) reaching up to 2.0 and 11.8 mg L^{-1}, respectively. Over the period of record, conditions stressful for fish populations were evident, including hypoxia and anoxia, pH values over 10, temperatures of 30 °C or more and total ammonium more than 7 mg L^{-1}. Chlorophyll-a levels can reach over 1 mg L^{-1}, which is at the upper level of what can be observed in natural waters. This table clearly portrays the lake as one of extreme water quality, at least at times.

We compared observations in Upper Klamath and Agency lakes for key water quality variables. Water column means were averaged over all sites and dates for each year in each lake. Boxplot distributions of these annual May–September means are plotted in Figure 3.1. Note that un-ionized ammonia (NH_3), Chl and Phae have all been log_{10}-transformed because they were highly skewed and difficult to compare visually otherwise. By first averaging over sites and seasons, we reduced the correlations among observations within each distribution. This implies in turn that non-overlapping notches in Upper Klamath versus Agency Lake boxplots provide more

Table 3.1: Upper Klamath Lake monitoring program: summary of data by water quality variable, May–September, 1990-01-18 to 2009-10-21. See Table A.1 for measurement units.

	Min.	1st Qu.	Median	Mean	3rd Qu.	Max.
ZMAX	0.6	2.1	2.6	3.3	3.5	8.6
SECCHI	0.0	0.6	0.9	1.0	1.2	3.5
LIGHT	0.0	15.7	268.8	886.7	1917.0	3797.0
T	9.3	15.8	18.6	18.3	20.9	30.5
DO	0.0	6.9	8.2	8.1	9.4	21.3
PSAT	0.0	84.5	97.2	99.6	114.0	299.5
PH	6.9	8.3	9.1	8.9	9.5	10.8
COND	40.0	102.4	108.0	107.9	112.2	156.7
ORP	33.0	303.0	328.0	331.8	364.8	437.0
TP	7.5	90.0	161.0	183.0	234.0	1990.0
SRP	1.5	14.0	38.0	59.8	80.0	564.0
TN	171.0	933.2	1845.0	1969.0	2554.0	11850.0
NO	0.0	5.0	10.0	22.0	20.5	470.0
NH	0.0	19.0	59.0	237.0	276.0	7645.0
MCHL	0.0	16.0	68.7	103.8	145.5	1436.0
PHAE	0.0	1.9	3.7	10.3	7.7	1232.0

compelling evidence that their medians actually differ. The plots show that Upper Klamath Lake appears to have less transparency, lower temperatures, lower DO, lower TP and SRP, higher TN, and higher Chl during the May–September period. Despite showing an overall lower distribution during May–September, Chl peaks tend to be higher in Agency Lake during the early part of the season due to differences in bloom timing between the two lakes: AL generally undergoes earlier bloom initiation, peak and decline in Chl than does UKL (Kann and Smith 1999, Kann 2010). At least some of these water quality differences are probably an artifact of sampling time (Section 3.2), although the seasonal differences in bloom dynamics between the lakes indicates factors other than sampling time drive the observed differences in Chl. Table 3.2 summarizes individual observations of water column means for the two lakes, i.e., the data underlying Figure 3.1 before averaging over sites and seasons within each lake.

Table 3.2: Summary of individual sampling day observations of water column means during May–September, 1990-01-18 to 2009-10-21 in each lake. See Table A.1 for measurement units.

	Min.	1st Qu.	Median	Mean	3rd Qu.	Max.
AL SECCHI	0.0	0.7	1.1	1.2	1.6	3.5
UK SECCHI	0.0	0.6	0.8	0.9	1.1	2.7
AL T	10.5	17.1	19.8	19.4	21.9	26.3
UK T	9.9	15.9	18.6	18.3	20.7	26.4
AL DO	1.7	7.3	8.8	8.7	10.3	17.4
UK DO	0.2	7.2	8.4	8.2	9.4	14.0
AL PH	7.1	8.2	9.0	8.9	9.6	10.7
UK PH	7.1	8.5	9.2	9.0	9.5	10.4
AL COND	82.7	102.3	107.5	108.3	113.8	136.0
UK COND	80.3	103.4	108.4	108.3	112.4	150.6
AL TP	7.5	100.0	184.5	230.4	315.8	1130.0
UK TP	29.0	83.8	154.0	169.2	221.0	1990.0
AL SRP	4.0	45.0	94.5	127.4	185.1	564.0
UK SRP	1.5	11.0	25.7	40.9	63.0	221.0
AL TN	184.0	633.0	1205.0	1437.0	1940.0	6975.0
UK TN	171.0	1060.0	1970.0	2112.0	2680.0	11850.0
AL NO	0.0	5.0	10.0	23.8	22.5	470.0
UK NO	0.0	5.0	9.5	21.3	20.0	376.0
AL NH3	0.0	1.5	9.6	54.6	24.9	1869.0
UK NH3	0.0	3.3	13.9	94.3	86.7	3112.0
AL MCHL	0.0	8.8	38.3	79.0	113.3	986.0
UK MCHL	0.1	20.8	77.0	110.7	151.9	1436.0
AL PHAE	0.0	1.0	2.2	6.7	5.0	236.2
UK PHAE	0.0	2.1	4.0	11.1	8.5	1232.0

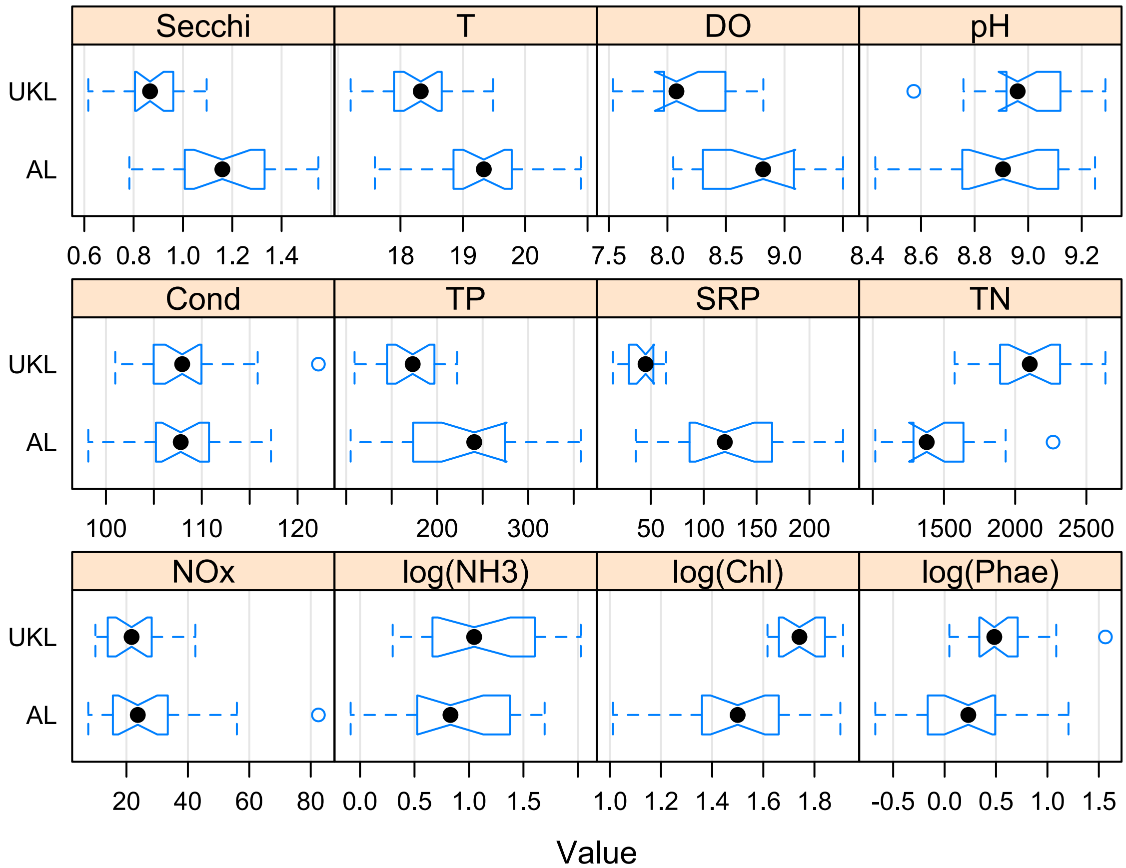

Figure 3.1: Boxplots of annual means for May–September, 1990–2009 in Upper Klamath and Agency lakes. See Table A.1 for measurement units.

3.2 Site-specific summaries

3.2.1 Water column depths and stability

Mean depths at each site were determined from time series of measured depths z_{max} and corresponding lake surface elevations $elev$. Data were fit to a mixed-effects model: $z_{max} = b + \beta_1 elev$, where b is a random effect depending on station and β_1 is constant. The fitted slope β_1 is 1.00 ± 0.01, exactly the expected value. The model yields the following values (m) for elevation = 1262 m (4141 ft), which is both the long-term mean and median:

PM	NB	WB	ML	ER	SB	MN	AS	AN
2.21	1.97	1.76	2.82	6.46	2.48	3.03	1.64	2.18

The differences in depths, although mostly small in an absolute sense, are quite large in a relative one, with ER being 3.7 times deeper than WB. Recall that ER, the deepest site, is also the most anomalous with respect to several water quality variables. In addition, MN is 1.8 times deeper than AS and may also account for some of the observed differences between the two lakes.

Before examining site-specific differences in water temperature and the related factor water column stability, it is important that we consider time of day of sampling. Sampling order is systematic, starting at PM and ending at AN. This implies that sampling time of day differs systematically among sites, as is made clear in Figure 3.2. It also implies that certain variables will differ systematically by site as an artifact of sampling time. Indeed, this appears to be the case with surface temperature, which shows an almost systematic increase from south to north consistent with time of day (Figure 3.2). The slight deviations of WB and SB from the overall pattern may be due to their semi-isolated embayment nature. In contrast, bottom and mean temperatures (data not shown) show a subdued effect, which means that the vertical temperature difference increases from south to north at sampling times.

The *Relative Thermal Resistance to Mixing* (RTRM) is a dimensionless index of water column stability based on vertical differences in water density. RTRM is used here to refer to the density difference between surface and bottom waters divided by the density difference of water between 4 and 5 °C. Low RTRM values indicate little resistance to wind-induced mixing, and vice-versa for high RTRMs. Calm periods generally produce the latter. The distribution of RTRMs by station during the last two decades is plotted in Figure 3.2. As would be expected from the spatial trend of vertical temperature differences, it increases from south to north. The apparent sampling time bias is a large one with respect to RTRM. For example, a linear regression model for $rtrm = \alpha_0 + \alpha_1 tod$, where tod is sampling time of day and the α_i are constants, accounts for 21% of the total variability, based on the (adjusted) R^2 value.

3.2.2 Water quality

A comparison of RTRM among stations is therefore biased by the sampling time of day. The bias should also affect comparisons of temperature and probably other water quality variables to the extent that they respond to short-term diel weather cycles, including heating/cooling and wind mixing. In principle, phytoplankton biomass, nutrients, DO, pH and ammonium could all be affected. However, some of the bias can be avoided by examining water column averages, because the sampling time effects probably manifest at least partially as a vertical redistribution of materials in the water column. In addition, parameters including chlorophyll and nutrients are water column integrated samples and are thus less likely to be affected by time of sample collection. Conseqently, we focus on water-column averages in this section and in most of what follows. But bias clearly remains. For example, diel changes in vertical stratification and wind mixing could change not only the vertical distribution

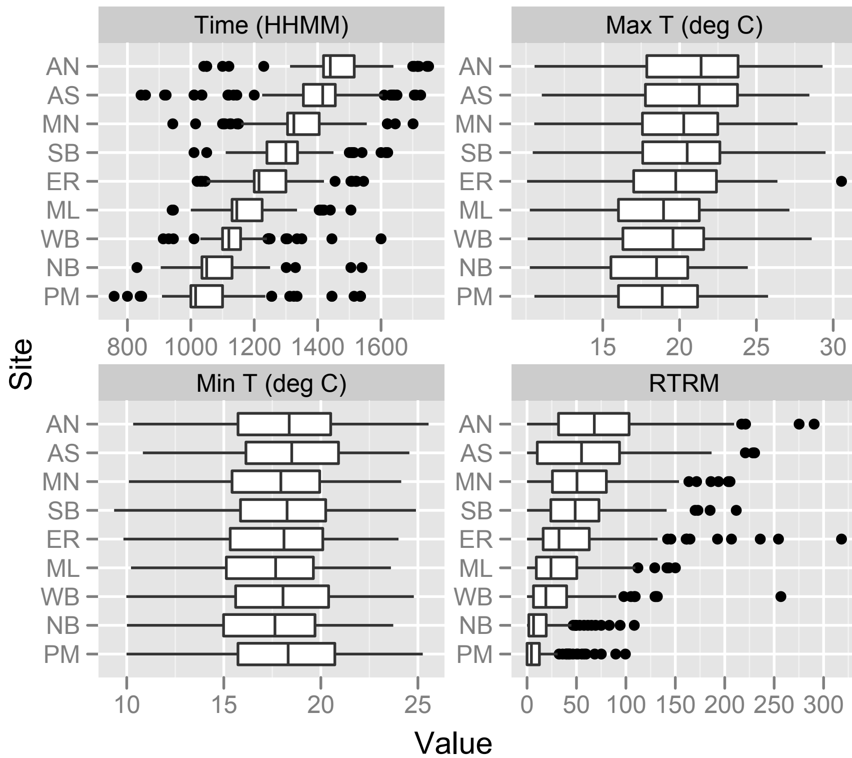

Figure 3.2: Sampling time of day, measured temperatures, and calculated RTRM by site.

of TP but also the extent of resuspension of surficial sediments containing additional phosphorus. It is possible that some correction could be made to the data based on time of day given the large number of observations to work with, but this is a challenging undertaking that is not feasible within the limited scope of this project. Moreover, comparisons among years and seasons, rather than among sites, should not be affected by the sampling time issue.

The distributions of individual temperature, DO, pH and Secchi depth observations by site during May–Sep, 1990–2009 are illustrated in Figure 3.3. Notched box plots are not used in this case, as the individual sampling-day means are highly correlated in space and time, and notches would not be a reliable guide to significant differences. Sites are arranged by latitude, from south to north. We cannot discount at least a partial effect of sampling time on any differences among sites that could be part of a south-north trend. However, the deeper ER station stands out with respect to DO in a way that cannot be explained by sampling time, with its median DO the lowest and the entire bottom quartile < 5 mg L^{-1} DO. One explanation is that DO is highly influenced by photosynthesis in UKL, and light extinction is sufficiently high

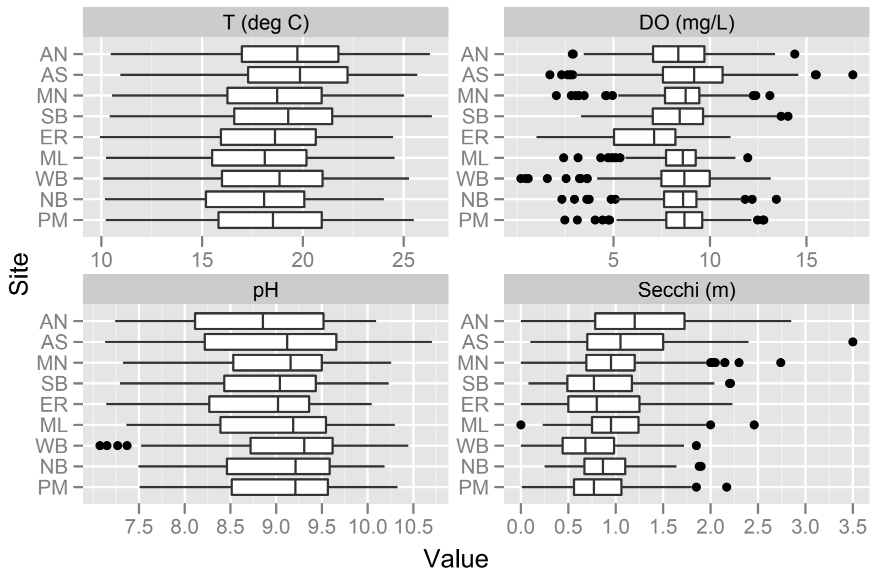

Figure 3.3: Distributions of water-column means of temperature, DO, pH and Secchi depth by site during May–Sep, 1990–2009.

that the lower depths of ER are more frequently below the photic zone than at other stations (Kann 2010).

The distributions of chlorophyll-a (Chl), its degradation product phaeophytin (Phae), and major N and P fractions are illustrated in Figure 3.4. (NO_3 is not shown because the many measurements at the limit of detection—5 μg L^{-1}—imply that its distribution is distorted.) Since these parameters are water column integrated they are less likely to be influenced by time of sample collection. WB tends to have the highest Chl and Phae. The highest values for un-ionized NH_3 are found at ER, followed by the two embayments WB and SB. The embayments are also highest in TN (and TP of the Upper Klamath Lake sites). Note that Phae appears to have an unusual number of outlying values.

Conditions deleterious for Lost River and shortnose suckers in the lake can be described in terms of temperature, DO, pH and un-ionized NH_3 (Perkins et al. 2000, Terwilliger et al. 2003). The relevant concentrations vary according to the specific critera used to assess toxicity. Table 3.3 shows the frequency of deleterious conditions for one particular choice. The values for temperature, DO and pH are the high stress thresholds for Upper Klamath Lake suckers determined by M.E. Loftus in a 2001 study (quoted by Lindenberg et al. 2008). The value for NH_3 is the lowest mortality threshold determined by Meyer and Hansen (2002). The results are tabulated as the proportion of times that the most extreme value in the water column was greater than

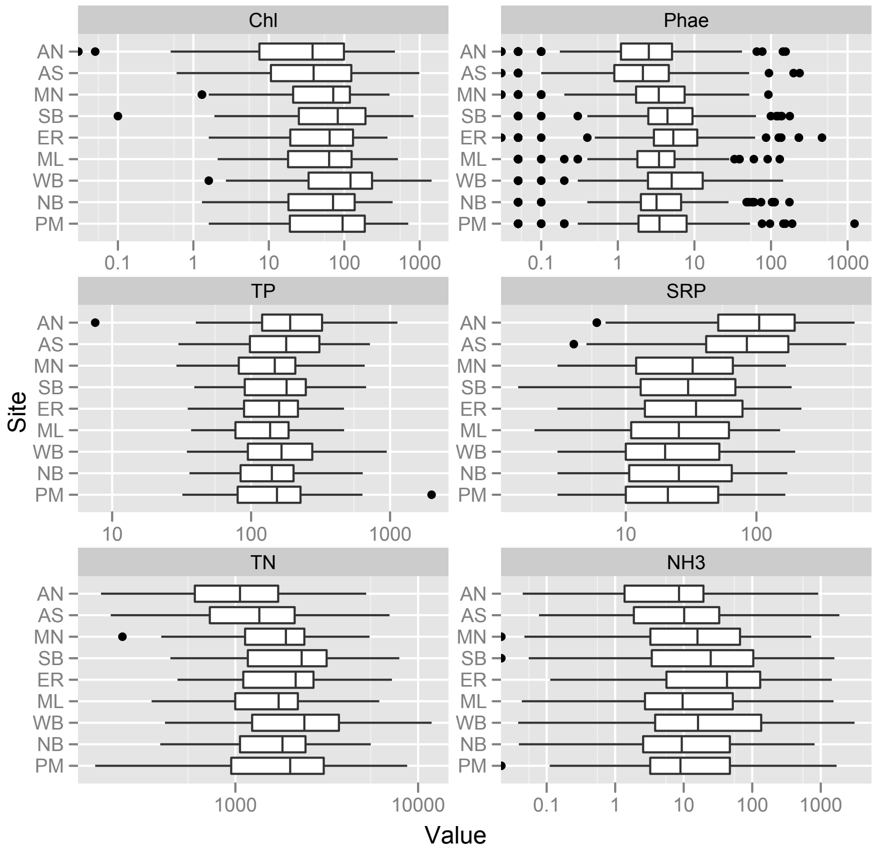

Figure 3.4: Distributions of water-column means of phytoplankton-derived pigments and various N and P fractions by site during May–Sep, 1990–2009. All measurement units are μg L^{-1}.

(or less than, for DO) the indicated concentration. Although excessive temperatures are not very frequent, the high proportion of exceedances for both DO and pH stands out at many sites. Low DO at ER occurs 25% of the time, with SB not far behind at 22%. WB and AN both have low DO more than 10% of the time. High pH is also pervasive throughout the two lakes, ranging from a low of 18% at ER to 30% and above in Agency Lake. Stressful un-ionized NH_3 conditions occur more frequently than temperature but less so than DO and pH. It does exceed 10% at WB and PM, however.

Table 3.3: Proportion of water column observations in which temperature (T), DO, pH and un-ionized ammonia N exceed the conditions shown.

Site	T > 28 °C	DO < 4 mg/L	pH > 9.7	NH_3-N > 370 μg L^{-1}
PM	0.00	0.01	0.20	0.10
NB	0.00	0.04	0.24	0.06
WB	0.01	0.10	0.29	0.11
ML	0.00	0.02	0.20	0.05
ER	0.01	0.25	0.18	0.06
SB	0.01	0.22	0.23	0.07
MN	0.00	0.06	0.22	0.03
AS	0.01	0.09	0.31	0.04
AN	0.02	0.12	0.30	0.04

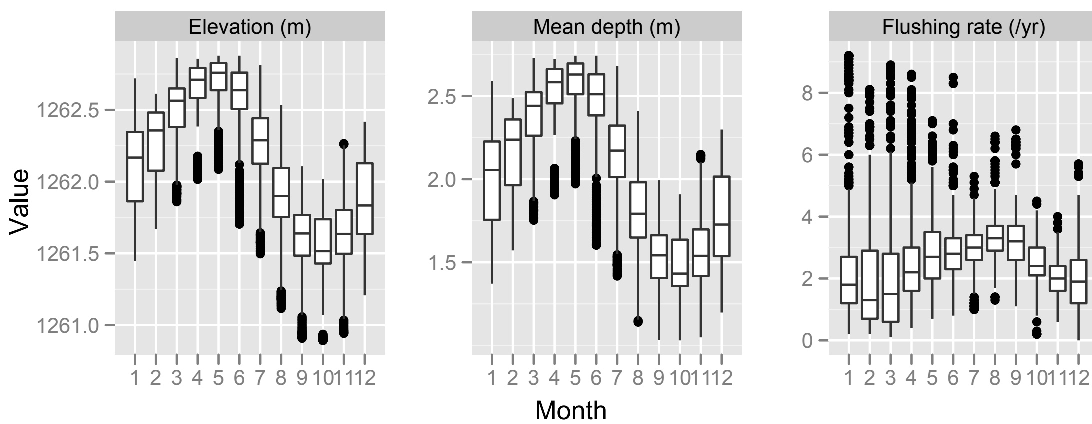

Figure 3.5: Seasonal pattern of hydrological variables affecting mass balances in the two lakes.

3.3 Seasonal patterns

3.3.1 Stability and related factors

On a seasonal basis, median monthly surface elevation of the lake peaks in May at 1262.8 m (4143.0 ft) and declines to a minimum in October of 1261.5 m (4138.8 ft), a drop of 1.3 m (Figure 3.5). The corresponding values for mean lake depth are 2.65 and 1.44 m, a slightly smaller difference of 1.2 m because the lake surface area decreases slower than the volume as surface elevation decreases. The flushing rate is at a minimum of 1.3 yr^{-1} in February and more than doubles to a maximum of 3.3 yr^{-1} in August.

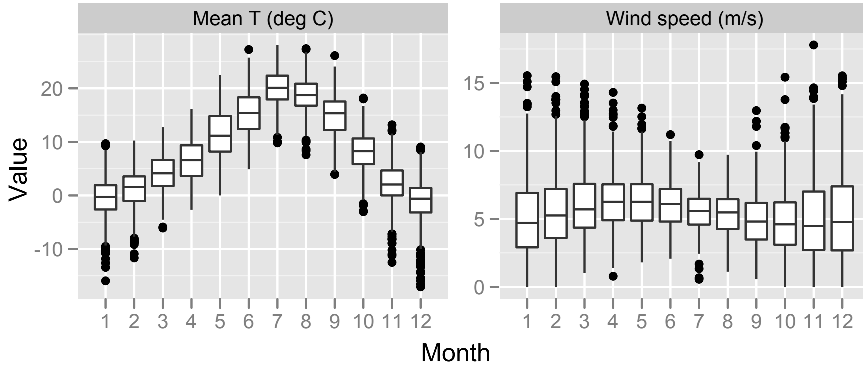

Figure 3.6: Seasonal pattern of weather variables at Upper Klamath Lake.

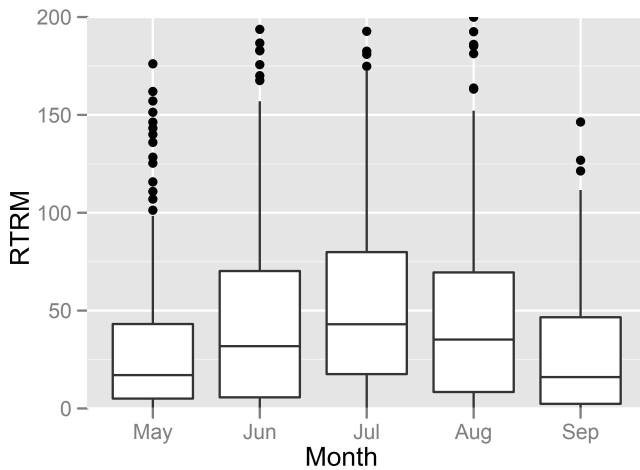

Figure 3.7: Seasonal pattern of RTRM at Upper Klamath Lake.

Median monthly temperature rises from a minimum of -0.8 °C in December to a maximum of 20.1 in July (Figure 3.6). Median wind speed, measured as the daily maximum of the 4-hr running mean, peaks in April at 6.2 m s^{-1} and declines to 4.4 in November. Although summer is not the time of lowest median wind speed, higher values ($>$ 9 m s^{-1}) are less likely. Partially in response to higher air temperature and reduced wind stress, median water column stability reaches its maximum of 43 in July, more than 2.5 $\times$ its median value in either May or September (Figure 3.7).

3.3.2 Water quality

Here we summarize the seasonal patterns of key water quality variables. First, the distributions of temperature, DO, pH and Secchi depth by month during May–Sep, 1990–2009 are illustrated in Figure 3.8. Temperature increases rapidly from a median of 14.3 °C in May to over 21 in July and then falls more slowly over the next two months. Median DO first increases with photosynthesis from 8.5 mg L^{-1} in May to 9.4 in June but then falls in July to 8.0 as hypoxic conditions become common. Unlike DO, median pH continues to increase from 8.1 in May to 9.5 in July, an interesting disparity that bears further examination as they are both influenced strongly by photosynthetic and respiratory processes. Secchi depth decreases from 1.3 m in May to 0.73 in July and remains low for the duration of the season.

The distributions of Chl, TP, TN and NH_3-N by month are illustrated in Figure 3.9. Chl is essentially a mirror of Secchi depth, as the phytoplankton is the major contributor to transparency variability. Chl increases from a median of 7.1 μg L^{-1} in May to 124 in July (and a maximum of 1400 μg L^{-1} in July for water-column means). Similarly, TP and TN mirror the rise of Chl. Median TP increases from a low of 60 μg L^{-1} in May to 220 in July, median TN from 680 μg L^{-1} in May to 2270 in July. These are very large increases in each nutrient, over three-fold. NH_3-N rises rapidly from 1.6 μg L^{-1} in May to 39 in July, responding to an increase in temperature, pH and organic matter substrate, as well as decreasing DO.

The distributions of total inorganic N (TIN = NO_3 + NO_2 + NH_4 + NH_3) and soluble reactive P (SRP) by month are summarized in Figure 3.10. Median TIN varies from 48 to 162 μg L^{-1} and median SRP from 19 to 74 μg L^{-1}. TIN showed no obvious pattern (although upper quartile values begin to increase in July). However, SRP decreased from May to June as Chl increased, indicating algal uptake, and then clearly increased in July. The July increase in SRP appears to be due to reduced uptake as the annual AFA bloom achieves a maximum and then declines. Both TIN and SRP end up higher at the end of the 5-month period than the beginning. The remaining two plots are of the *weight* ratios of TP:TN and TIN:SRP. The ratio of N to P in phytoplankton cells typically averages about 7.2 on a weight basis (16 on a molar basis). Ratios of N:P < 4 indicate probable N limitation and N:P > 14 indicate probable P limitation. The median monthly TN:TP ratio here varies from 11 to 14 and is therefore in an ambiguous range, although there are many cases where it lies above 14 or below 4. The median TIN:SRP ratio, on the other hand, varies from only 1.7 to 4.6 and indicates that N-limitation is much more likely than P-limitation.

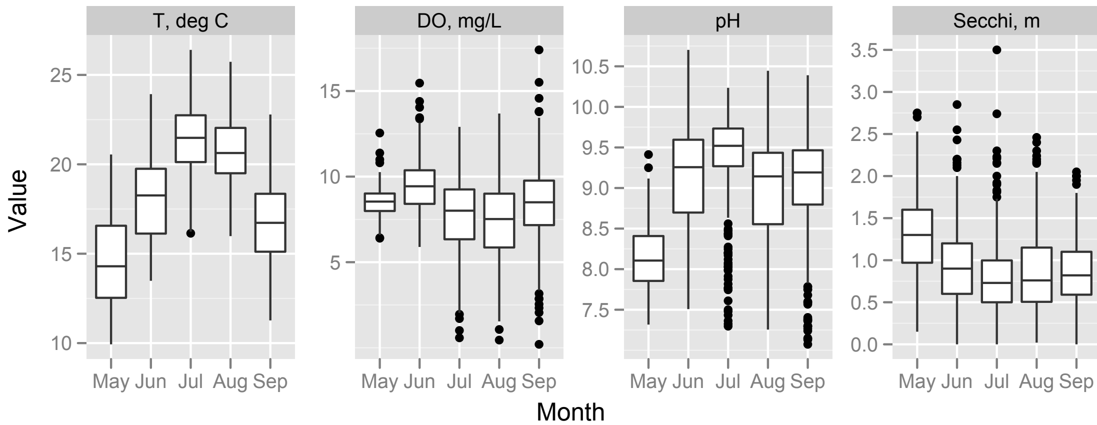

Figure 3.8: Distributions of temperature, DO, pH and Secchi depth by month, 1990–2009.

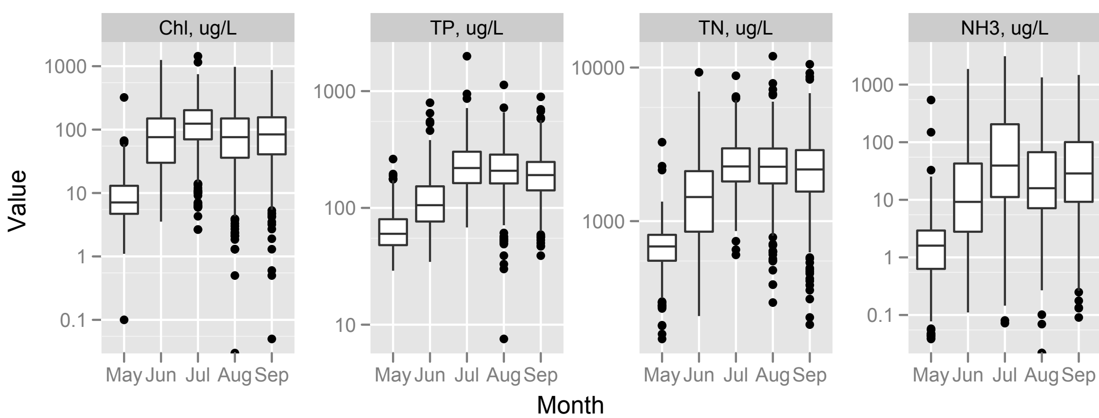

Figure 3.9: Distributions of Chl, TP, TN and NH3 by month, 1990–2009.

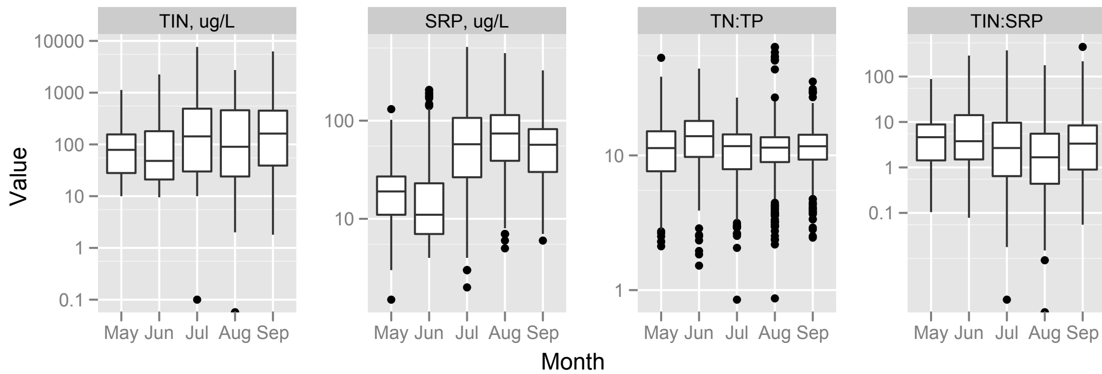

Figure 3.10: Distributions of TIN, SRP, TN:TP weight ratio and TIN:SRP weight ratio by month, 1990–2009.

3.3.3 Site differences

Underlying the boxplots describing seasonality is a large variability from year to year, and from site to site. We scaled each chlorophyll series to have the same mean and standard deviation for each year and then overlaid them in order to illustrate this variability (Figure 3.11). Although the details are difficult to discern, this graph makes clear that there is no dependable seasonal pattern. Moreover, not only is there variability among sites in terms of seasonal pattern, but groupings of sites with similar patterns change from year to year. In some years such as 1994 and 2009, the sites behaved more or less coherently, but they showed little coherence in most other years. Because the transit time among stations is of the same order as the sampling time (Wood et al. 2008), each series represents phenomena originating at the site mixed with those of surrounding sites. The relative "closeness" in behavior of two stations can be visualized in two dimensions with multidimensional scaling (Figure 3.12). WB and SB frequently stand out from the others, reflecting their relative isolation from the main circulation. PM is also often anomalous. This variability raises an important question about how to examine causal factors underlying bloom magnitude and timing. Even when the relationship is linear between some bloom attribute and a causal factor, the relationship can be obscured if data are first composited across sites.

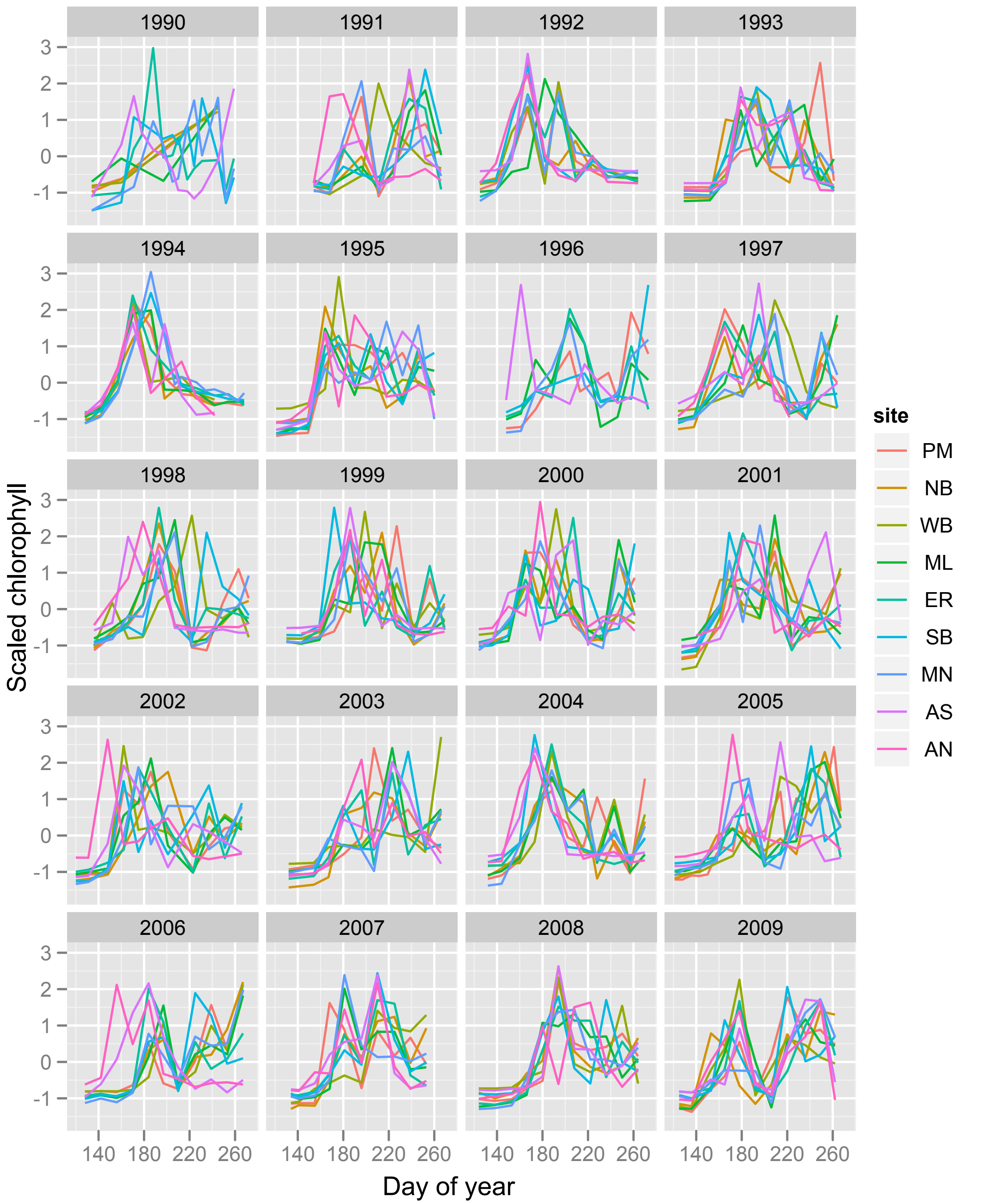

Figure 3.11: Scaled chlorophyll versus day of the year for different years and sites.

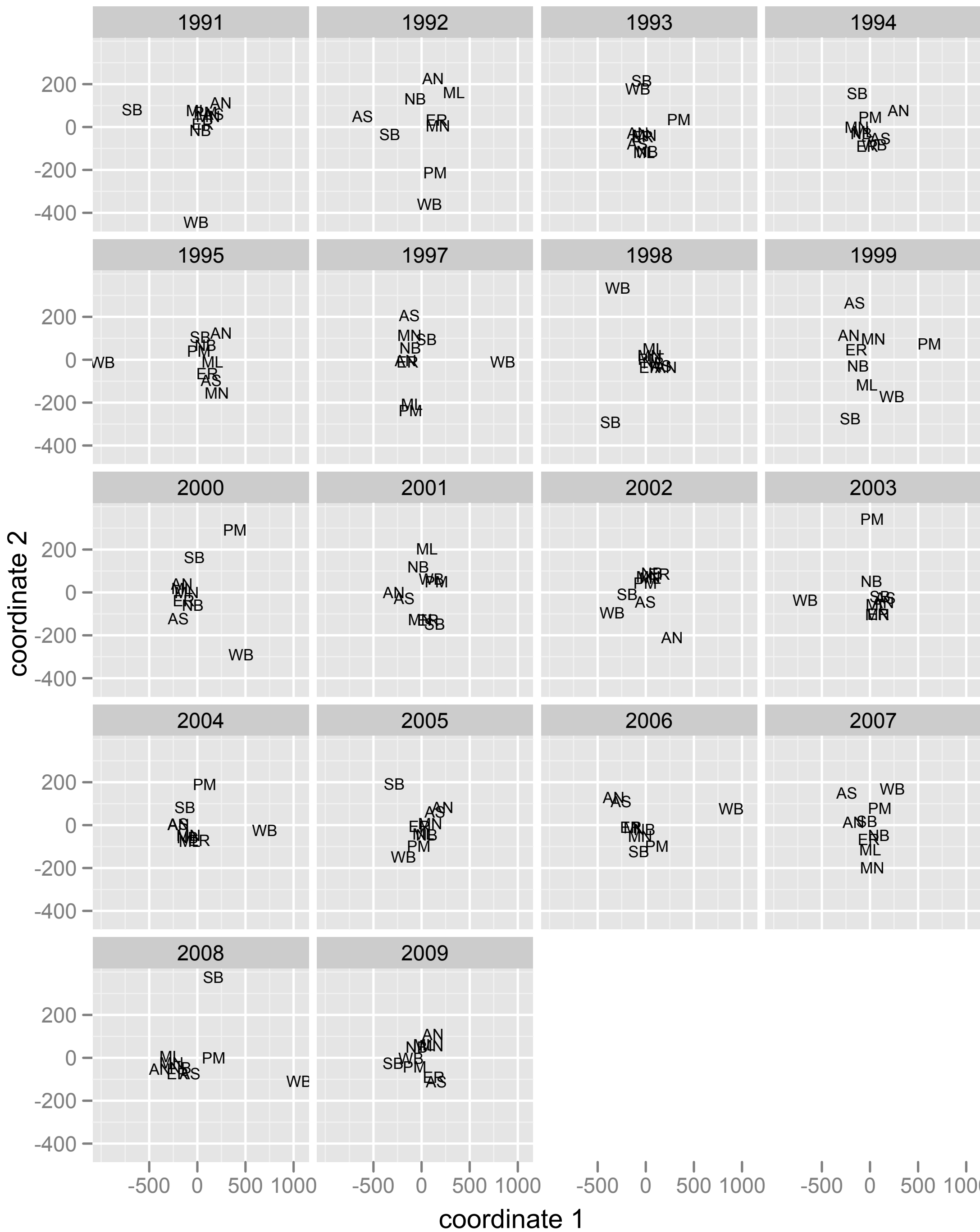

Figure 3.12: Stations arranged in two dimensions according to the similarity of their seasonal patterns in each year.

4

Trends over time

4.1 Water quality

4.1.1 Trends

How has the frequency of extreme values for these variables changed throughout the years 1990–2009? We tested the trend in the annual proportion of deleterious conditions by site for each of the four variables in Table 3.3, using the Mann-Kendall test. None of the $4 \times 9 = 36$ time series exhibited a significant time trend ($p >= .12$). We also combined sites using the Regional Kendall test in order to conduct a more powerful test of trend, but again none of the four frequencies showed a trend ($p >= .15$). The year-to-year variability in these frequency variables apparently overwhelms the presence of any secular change.

Trends in the actual variables, as opposed to the frequency of extremes in these variables, were detected by the Seasonal Kendall test (Table 4.1). Of the variables tested, the following exhibited a trend ($p < 0.05$) at one or more sites: Secchi depth, DO, TN, NO_x and, especially, phaeophytin.

We again combined sites using the Regional Kendall test, without regard to the sign of the trends at individual sites (Table 4.2). Secchi depth, temperature, DO, SRP and phaeophytin all showed evidence of a regional-scale trend. If the test is repeated for chlorophyll using just sites with the same sign, then the six northern stations show a significant subregional trend of -1.2 μg L^{-1} yr^{-1} or -1.3% yr^{-1} ($p = 0.01$). In contrast, if we also test any of the remaining variables using just sites with the same signs, we still obtain no significant trends ($p \geq 0.05$). The regional trend results thus point to an increase in Secchi depth and SRP over the record, a decrease in temperature, DO and phaeophytin, and a decrease in chlorophyll except for the southernmost stations. Note that the trend slopes themselves are nonparametric estimates and can differ substantially from the slope of a linear trend.

Table 4.1: Seasonal Kendall test of trend.

	PM	NB	WB	ML	ER	SB	MN	AS	AN
Secchi depth, m/yr:									
sen.slope	-0.003	0.002	0.005	0.011	0.006	0.017	0.013	0.005	0.004
p.value	0.723	0.718	0.238	0.082	0.282	0.004	0.026	0.496	0.786
temperature, °C/yr			:						
sen.slope	0.005	0.001	-0.011	0.004	0.008	-0.012	-0.016	-0.006	-0.03
p.value	0.844	0.948	0.646	0.777	0.814	0.659	0.393	0.754	0.356
DO, mg/L/yr:									
sen.slope	-0.008	-0.024	-0.046	-0.023	-0.035	-0.04	-0.04	-0.013	-0.073
p.value	0.716	0.159	0.108	0.156	0.049	0.049	0.032	0.483	0.014
pH, yr:$^{-1}$									
sen.slope	0.012	0.006	0.008	0.014	0.007	0.003	0.009	0.003	-0.009
p.value	0.122	0.326	0.258	0.068	0.317	0.638	0.176	0.643	0.338
conductivity, μS/cm/yr:									
sen.slope	0.058	-0.017	0.158	0.039	0.077	0.162	0.028	-0.045	-0.163
p.value	0.611	0.904	0.235	0.684	0.38	0.261	0.817	0.743	0.141
TP, μg/L/yr:									
sen.slope	0.5	-0.214	0.5	0.806	0.75	0.182	0.15	-0.362	-0.785
p.value	0.411	0.793	0.731	0.143	0.401	0.791	0.86	0.777	0.57
SRP, μg/L/yr:									
sen.slope	-0.15	0.045	0.062	0.083	0.417	0.5	0.359	1.121	0.8
p.value	0.437	0.831	0.694	0.57	0.324	0.105	0.149	0.152	0.607
TN, μg/L/yr:									
sen.slope	16.187	4.5	9.361	11.294	9.444	8.45	2.833	1.406	0.537
p.value	0.042	0.682	0.232	0.056	0.102	0.508	0.768	0.823	0.901
NO_3+ NO_2, μg/L/yr:									
sen.slope	0	0.133	0	0.231	0	0	0	0	0.364
p.value	0.56	0.221	0.428	0.091	0.437	0.499	0.363	0.806	0.013
NH_3, μg/L/yr:									
sen.slope	-0.038	0.053	0.214	0.094	0.412	0.291	0.355	-0.05	-0.039
p.value	0.888	0.723	0.352	0.442	0.06	0.232	0.095	0.661	0.804
chlorophyll-*a*, μg/L/yr:									
sen.slope	0.403	0.525	0.35	-0.025	-0.25	-0.486	-0.4	-0.108	-0.567
p.value	0.303	0.145	0.376	0.881	0.48	0.185	0.171	0.709	0.356
phaeophytin, μg/L/yr:									
sen.slope	-0.228	-0.182	-0.383	-0.195	-0.321	-0.3	-0.347	-0.14	-0.119
p.value	0.003	0.007	0.002	0.001	0	0	0	0.004	0.038

Table 4.2: Regional Kendall test for trend.

	trend	percent	p-value
Secchi	0.009	0.932	0.000
temperature	-0.015	-0.080	0.017
DO	-0.023	-0.275	0.000
ph	0.004	0.048	0.090
conductivity	0.071	0.066	0.372
TP	0.323	0.177	0.562
SRP	0.552	0.930	0.047
TN	3.905	0.200	0.623
NO_x	-0.150	-0.643	0.489
NH_3	0.103	0.126	0.672
chlorophyll	-0.620	-0.602	0.135
phaeophytin	-0.370	-3.648	0.000

4.1.2 Chlorophyll trends in more detail

The composite chlorophyll time series for the six northern and three southern core sites are shown in Figure 4.1. The data are first converted to log-anomalies: $x \to \log_{10} x/\mu$, so a log-anomaly of 1.5 corresponds to a chlorophyll value of $10^{1.5} = 31.6 \times \mu$, where μ is the mean. The positive log-anomalies are then divided into two groups by their mean, and similarly for the negative ones, giving a total of four groups distinguished by color. Although these "tile" plots involve a transformation and condensation of the underlying observations, they provide a good picture of the way a monthly series evolves in terms of both seasonal pattern and long-term level. At the northern sites, which exhibit a regional trend, the highest two categories became less frequent after the 1990s, although that may now be changing again. There is also a suggestion of a shift in timing of the bloom maximum to later in the season, especially since 2002. The southern sites do not have a significant long-term trend. If anything, the trend appears to be in the opposite direction, i.e., increasing, although it is not statistically significant. Similar to the northern sites, there is the appearance of a shift in bloom timing to slightly later in the year. The trend at the northern sites is by no means a simple long-term decline: bloom magnitudes have increased during the last few years of record, and any systematic changes identified here require ongoing monitoring to determine whether the long-term trends will continue.

A closer look at the shifts at individual sites confirms the notion of a decrease in recent years but does not suggest any overwhelming synchrony in the timing of this change. We averaged the chlorophyll for May–September at each site, interpolated the few missing values in the resulting annualized time series, and then calculated the CUSUM-transformed chlorophyll (Figure 4.2). The CUSUM-transformed series

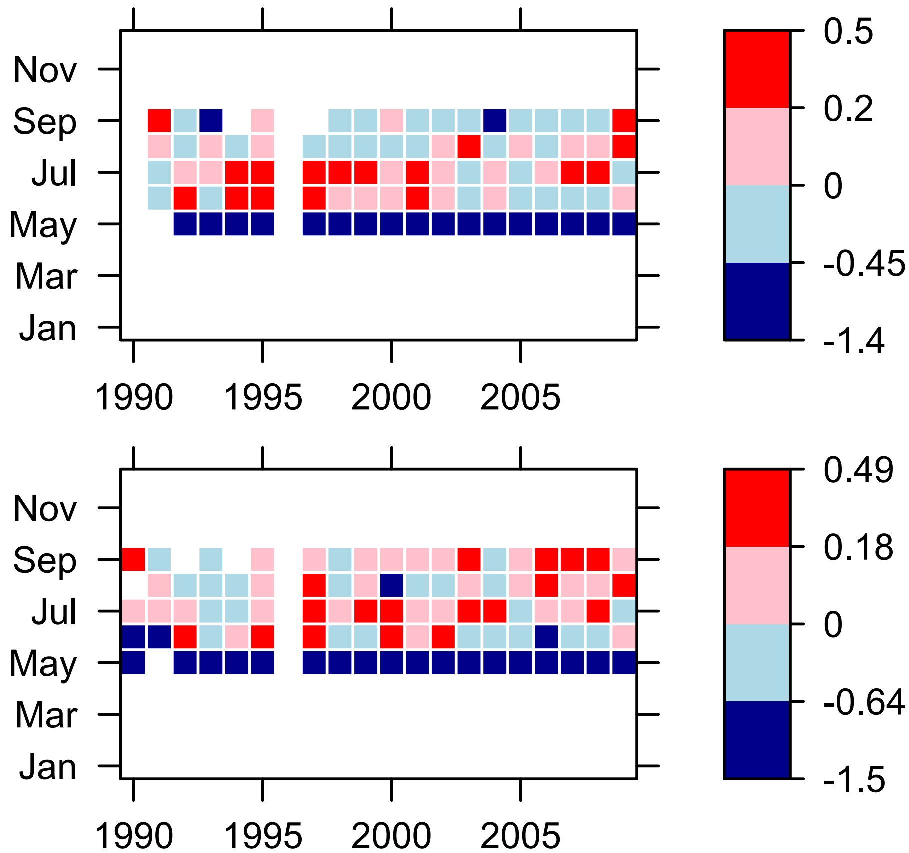

Figure 4.1: Log-anomaly plots of composite monthly time series for chlorophyll-a (μg L^{-1}). (Top panel) Sites ML, SB, ER, MN, AS, AN. (Bottom panel) Sites PM, NB, WB.

decline when the current observation is less than the long-term average. The plots imply that most recent years have been below the long-term average, but they do not identify a unique turning point. For ML, ER, SB and MN, 2001–2002 may have some significance as a turning point. But for Agency Lake sites, 1996–1997 appears to be more important. The three southern stations have weaker, less consistent recent declines. Note that the shift at AN in 1997 coincides with the first several years of the Bureau of Land Management (BLM) program, including removal of cattle grazing, to restore 12 km^2 of wetland near the mouth of the Wood River. It is also interesting to note that, although not statistically significant, the trend for AN (-0.57 μg L^{-1} yr^{-1}) was the most negative of all station trends (Table 4.1).

How strong is the evidence for a change in the maximum or the timing? Figure 4.3 highlights the differences among years in the annual maximum. We first examined the CUSUM-transformed *maximum annual* water column means of chlorophyll to exam-

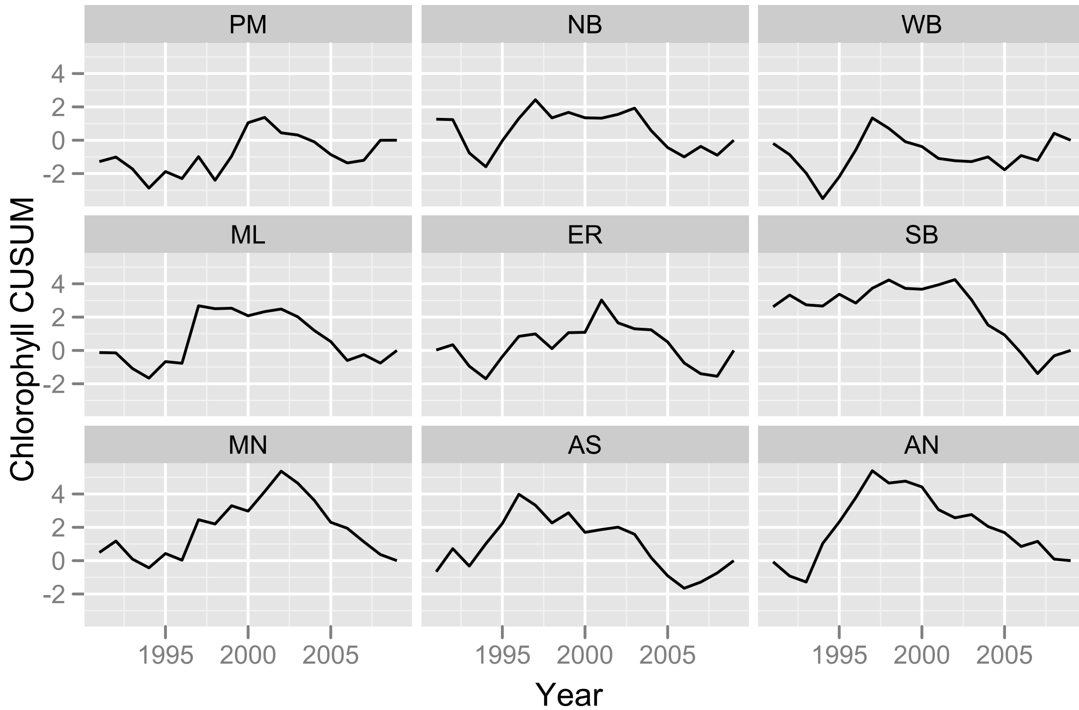

Figure 4.2: CUSUM-transformed mean May-Sep chlorophyll at the core sites.

ine possible changes in the maximum (Figure 4.4). As in the case of mean chlorophyll, the annual maxima showed a tendency to lower values after around 2000, more or less, depending on the site. A comparison with Figure 4.3 is a useful reminder that these represent *tendencies* superimposed on much interannual variability. Although the overall Mann-Kendall test of trend in the maximum was barely significant for MN ($p = 0.041$) and not significant for the others ($p > 0.10$), the Regional Kendall test suggested a modest lake-wide regional decline in the maximum with Sen slope of 2.6 μg L^{-1} yr^{-1} since 1990 ($p = 0.026$).

There is also some evidence for a change in the seasonal timing. Table 4.3 shows the number of times the annual maximum occurred in a particular month, by site. July is most common for the Upper Klamath Lake sites and June for the Agency Lake sites. But a later maximum in September is also not unusual, especially at the southern sites. The day of the year in which the maximum occurred did not show any clear trend according to the Mann-Kendall test ($p > 0.12$). But the "fulcrum" or center of gravity of the chlorophyll distribution (i.e., the time which divides the area under the chlorophyll curve for each year into two) does show some increasing trends, especially at NB where the Mann-Kendall test indicates significant trends at

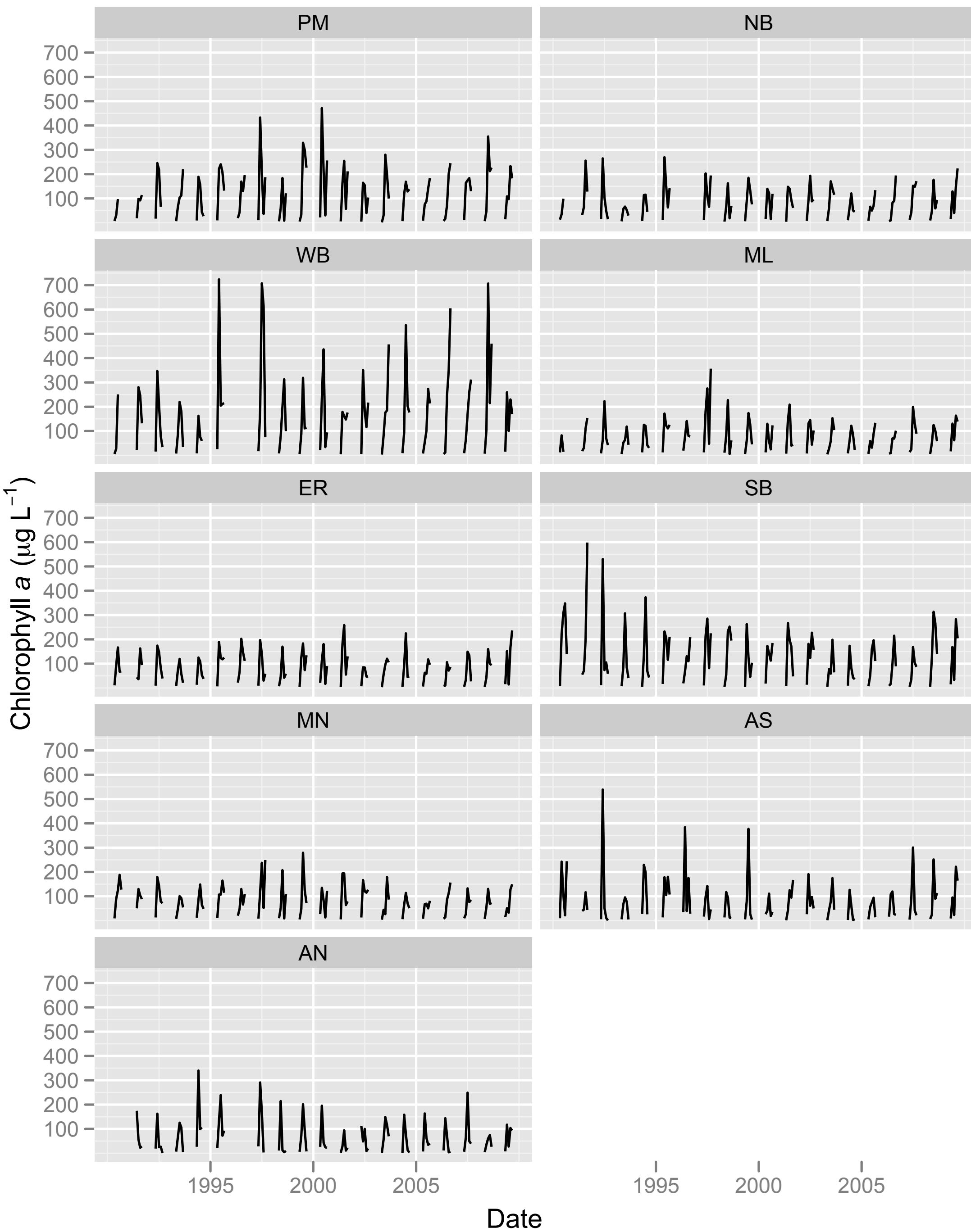

Figure 4.3: Monthly chlorophyll time series at the core sites.

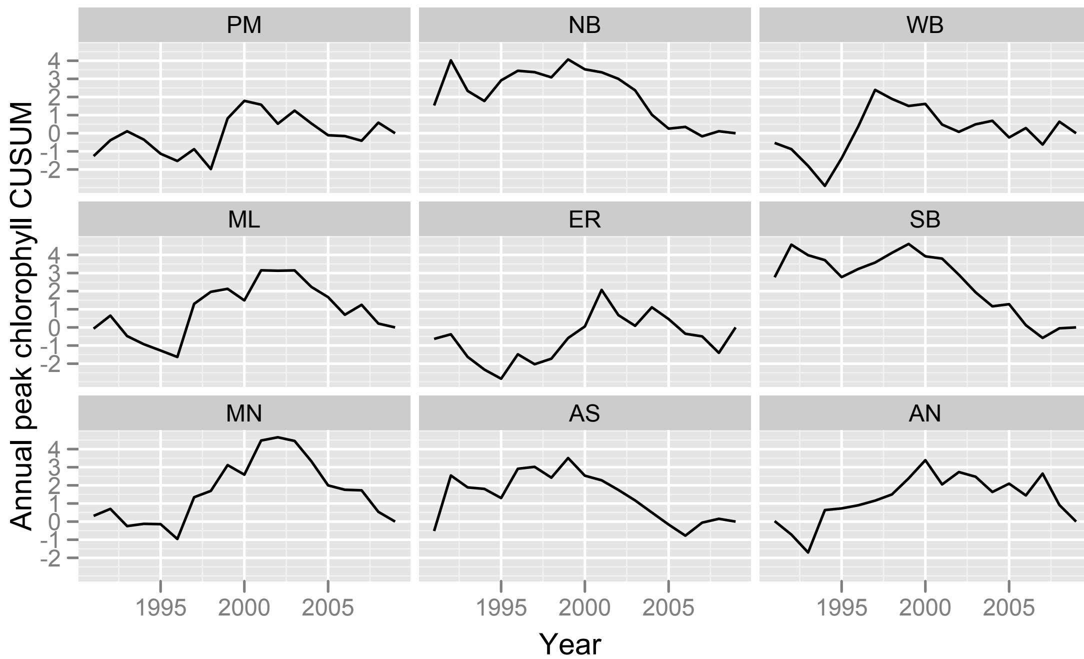

Figure 4.4: CUSUM-transformed annual peak chlorophyll at each of the core sites.

Table 4.3: Frequency of occurrence of maximum annual chlorophyll.

	May	Jun	Jul	Aug	Sep
PM	0	5	7	2	6
NB	0	3	8	3	5
WB	0	5	9	2	3
ML	0	2	10	2	6
ER	0	7	10	3	0
SB	0	5	6	6	3
MN	0	2	13	2	3
AS	0	7	6	5	2
AN	1	10	5	1	1

the $p < 0.01$ level (Figure 4.5). The Sen slope for these trends ranges from 0.64 d yr^{-1} at MN to 2.2 at NB, or a total of several weeks over the observation record. The overall Sen slope given by the Regional Kendall test was 1.0 d yr^{-1} ($p = 0.000018$).

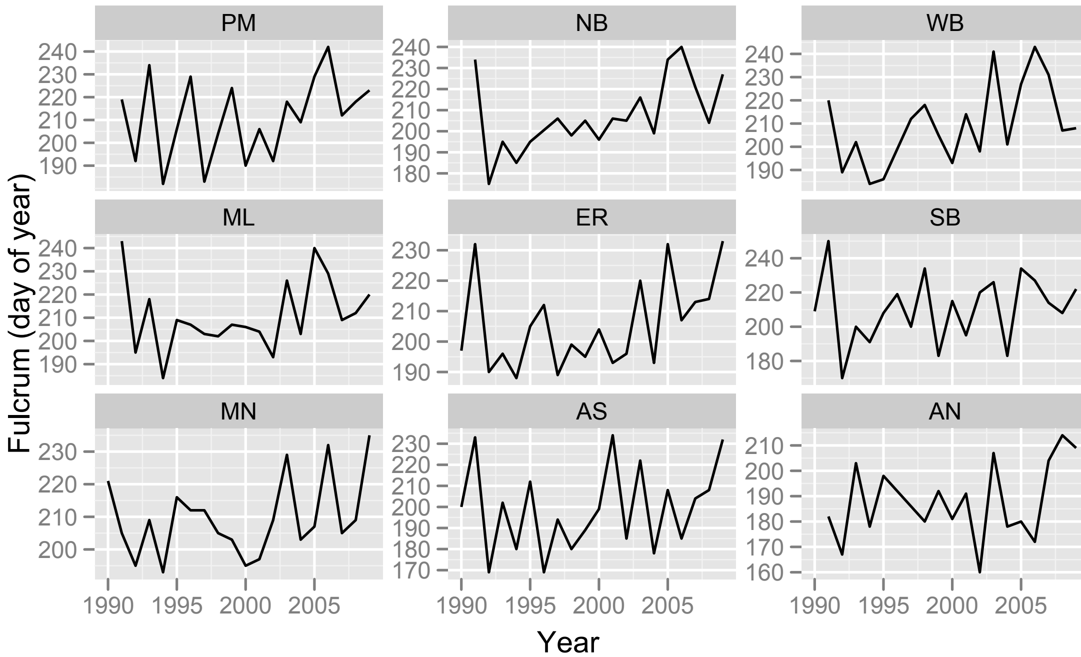

Figure 4.5: Annnual fulcrum or center of gravity of the chlorophyll pattern for each year and site.

4.1.3 Secchi depth and chlorophyll trends

Of the significant water quality trends other than chlorophyll, all of them could, in principle, be an effect of the chlorophyll trend. For example, lower Chl, indicating lower algal biomass, could cause in increase in SRP due to lower uptake. Only the trend in Secchi depth could plausibly have *generated* the chlorophyll trend. To examine this possibility, we estimated the linear model relating secchi depth to chlorophyll in which the intercept was allowed to vary with year. More formally, we estimated the linear mixed-effects model $1/secchi = b + \beta_1 Chl$, where b is a random effect of the year and β_1 is constant (Pinheiro and Bates 2000). The rationale for using inverse Secchi depth is given by, for example, Swift et al. (2006). Secchi depths < 0.1 m were removed, as they are highly uncertain and their inverses can be very large. The analysis was also confined to the six northern core sites accounting for the chlorophyll trend. The fixed effects were $b = 0.80 \pm 0.05$ and $\beta_1 = 0.0060 \pm 0.0001$. So the overall average intercept, presumably representing non-phytoplankton matter, has an effect comparable to $0.80/0.0060 = 133$ μg L^{-1} Chl. Consequently, a sizable increase in non-phytoplankton matter, equivalent to an increase in the intercept, could in principle cause a decrease in water clarity. In fact, though, we found no evidence for any increasing trend in the intercept (Figure 4.6; $p = 0.31$ using the Mann-Kendall test

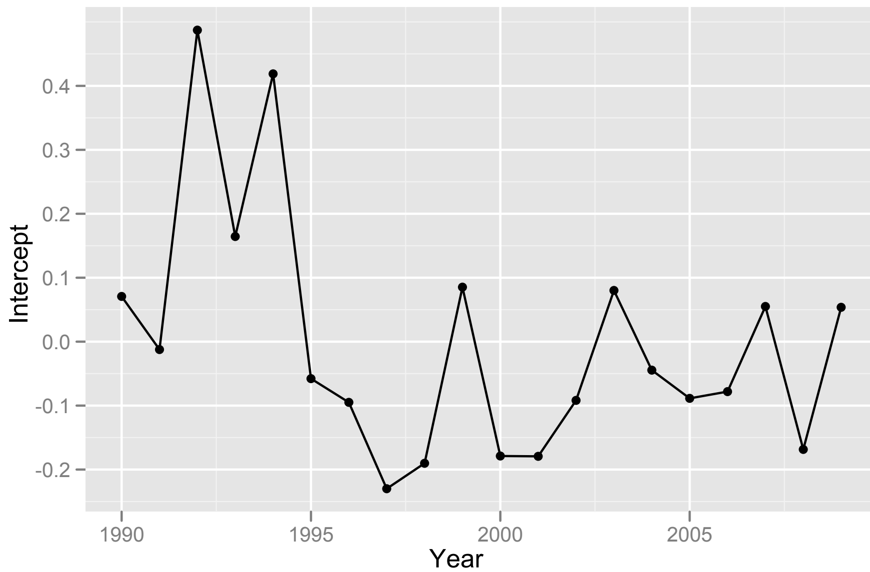

Figure 4.6: Annual intercept values (random effects) determined from the linear mixed-effects model for inverse Secchi depth versus chlorophyll for the six northern core sites.

for trend) and therefore no evidence that the chlorophyll decrease was *caused* by a decrease in water clarity.

4.1.4 Site differences

These composite data disguise much variability among sites with regard to long-term trends. As an example, we scaled the chlorophyll data by month and site and plotted the resulting series over the years (Figure 4.7). The different sites were relatively coherent in May, but the coherence deteriorated over the next two months and remained low. To illustrate this more quantitatively, consider the median absolute deviations (MAD, defined as the median of the absolute deviations from the data's median) of the data for each month and year. The median values of MAD for May through September were 8.0, 87, 123, 91 and 100, respectively.

We used multidimensional scaling to visualize the differences among sites in two dimensions. The results for each month suggest that the two bay sites, WB and SB, tend to be the most anomalous (Figure 4.8). Even AS and AN can be far apart (June). Only NB, ML, ER and MN can be said to cluster consistently.

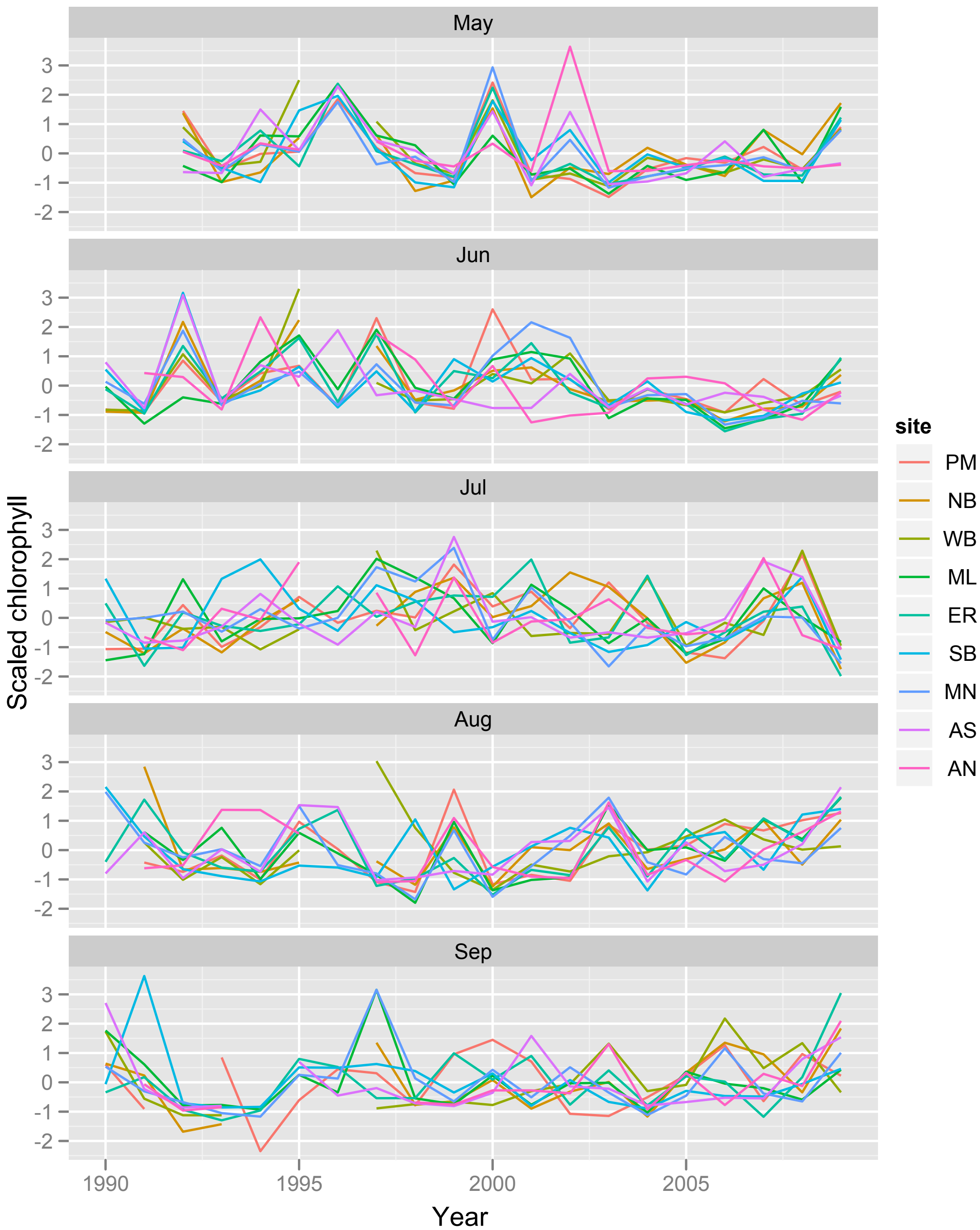

Figure 4.7: Scaled chlorophyll versus year for different months and sites.

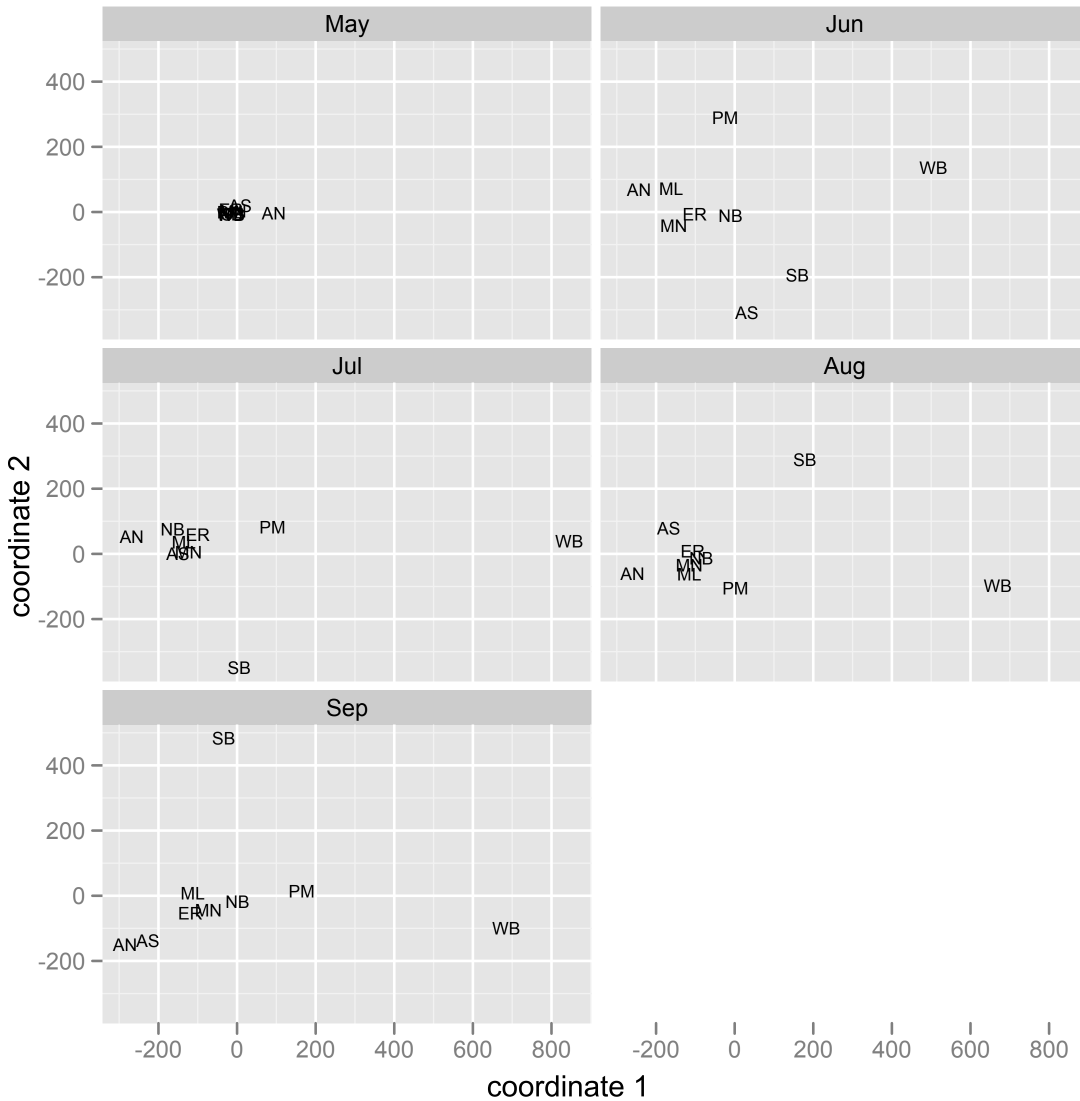

Figure 4.8: Results of multidimensional scaling of chlorophyll series in Figure 4.7.

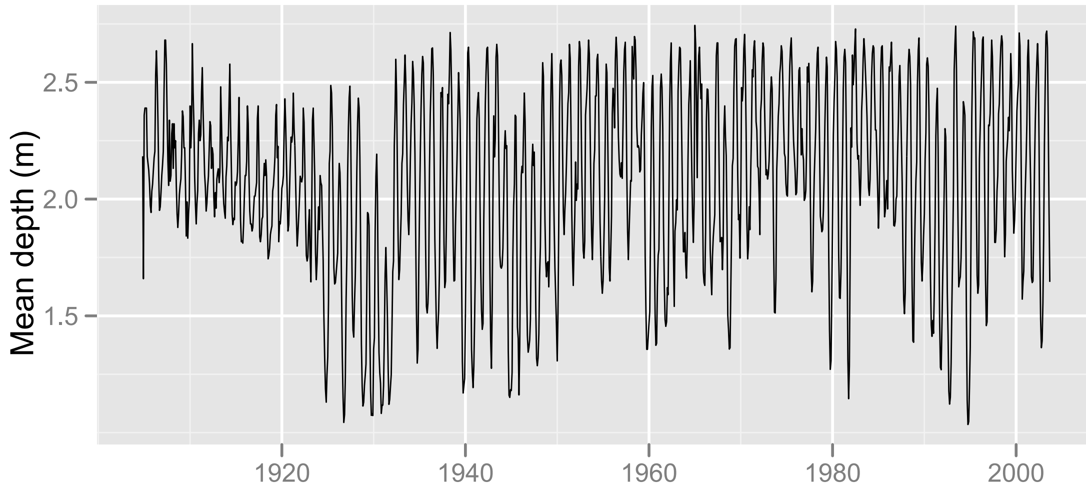

Figure 4.9: Monthly average of mean depth for both lakes combined, 1904–2003.

4.2 Hydrology

The depth of Upper Klamath Lake fluctuates widely from month to month, year to year, and even among decades (Figure 4.9). Although the annual average at the end of the record is similar to that of the early 20th century, the seasonal variability has increased dramatically. For example, the average coefficient of variation of the monthly means was 0.094 in the 1910s compared with 0.226 in the 1990s. More recent values of mean depth for both lakes separately are plotted in Figure 4.10, and the elevation time series for all months are summarized in Figure 4.11. The corresponding flushing rate for both lakes combined is in Figure 4.12. Although there are no long-term trends in these variables (according to the Seasonal Kendall test: $p > 0.27$), shifts in their patterns appear to persist for at least several years. These shifts are large enough in magnitude to influence certain water quality variables, at least in principle. For example, the annual minimum in monthly mean flushing rate varied over 10-fold from 0.19 in 1991 to 2.3 in 1996. Also, trends in surface elevation were consistently positive during Dec–May and negative during Jun–Nov, even though trends at individual months were not significant (Figure 4.13).

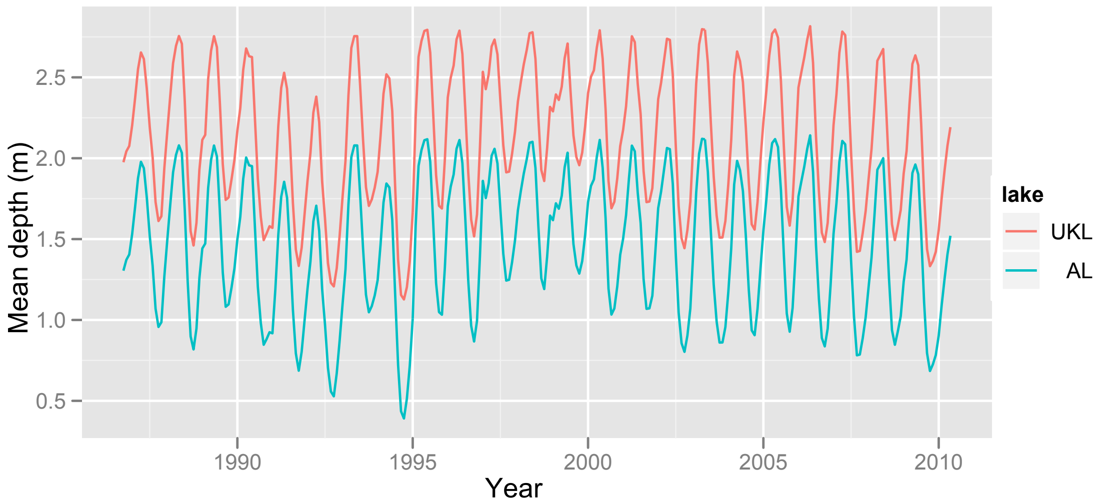

Figure 4.10: Monthly average of mean lake depth, 1987–2009.

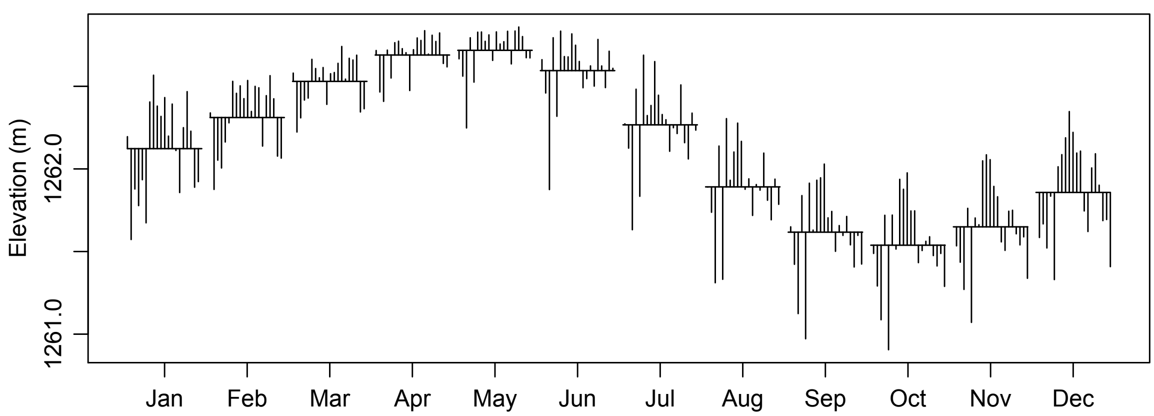

Figure 4.11: Elevation time series for each month separately. *Horizontal lines,* long-term mean elevation for each month; *vertical lines,* deviation from the long-term monthly mean for each year.

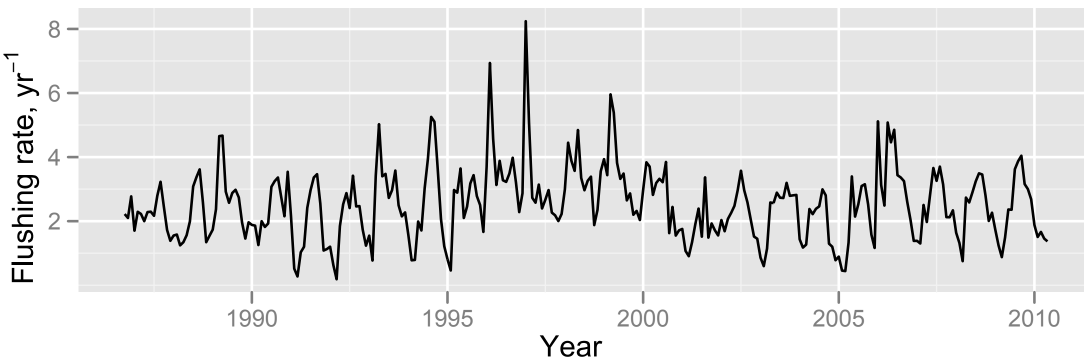

Figure 4.12: Monthly averages of flushing rate (outflow/volume) for both lakes combined.

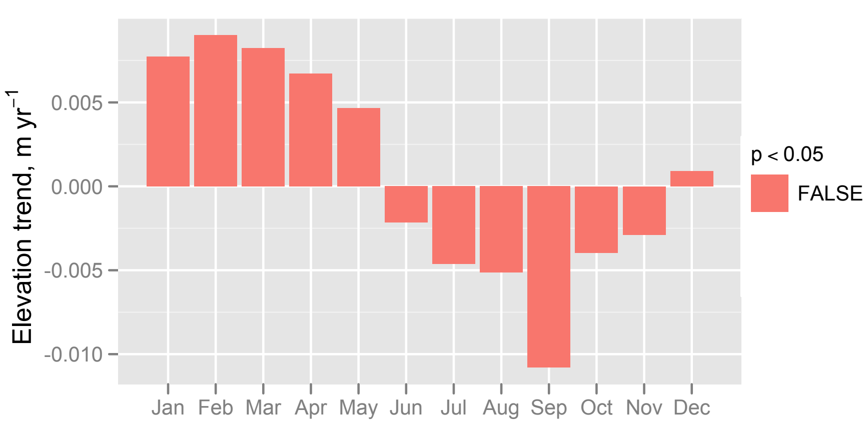

Figure 4.13: Trends in lake surface elevation by month, 1990–2009. *Red,* indicates lack of significance according to the Mann-Kendall test.

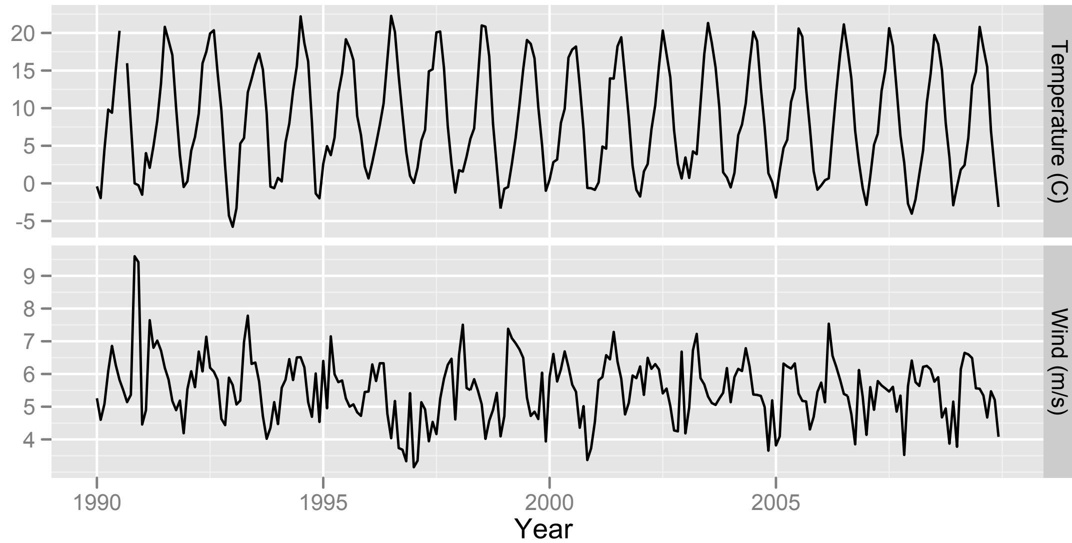

Figure 4.14: Monthly time series of mean daily air temperature (top panel) and maximum daily 4-hr running mean of wind speed (bottom panel).

4.3 Climate

We examined time series of mean daily air temperature and maximum daily 4-hr running mean of wind speed for any systematic changes during the period of water quality measurement (Figure 4.14).

The Seasonal Kendall test suggested a downward trend in mean daily air temperature with a small slope of -0.058 °C yr^{-1} ($p = 0.0026$). A closer examination of the individual months showed that trends were significant in September and October (Figure 4.15), equivalent to ca. 2 °C cooler over 20 years. In contrast, we could find no long-term trends in wind speed, either for the time series as a whole or for individual months.

The *Relative Thermal Resistance to Mixing* (RTRM) is an important variable under at least partial control by the weather (Kann and Welch 2005). There is a great deal of interannual and seasonal variability, as well as differences among sites (Figure 4.16). Based on the Seasonal Kendall test, we found no long-term trends at any of the stations, nor even a consistent sign in the trend:

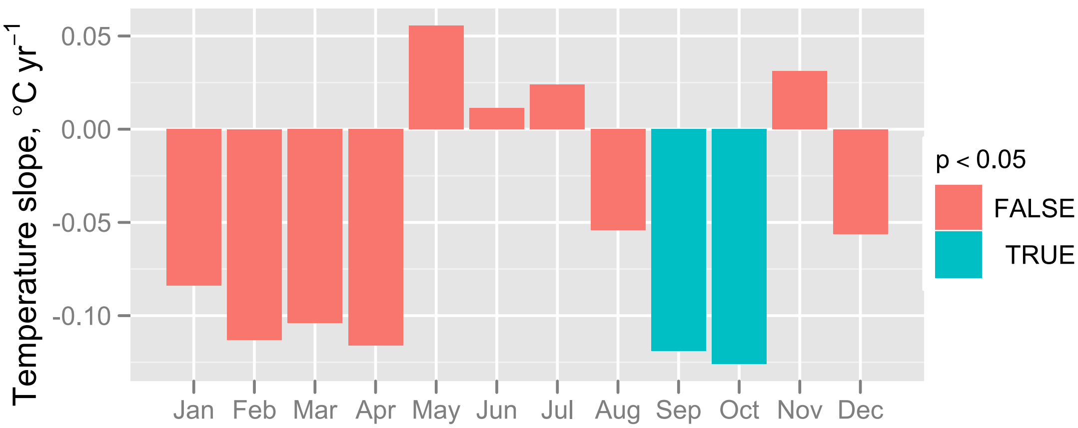

Figure 4.15: Air temperature trends during 1990–2008 and their statistical significance according to the Mann-Kendall test.

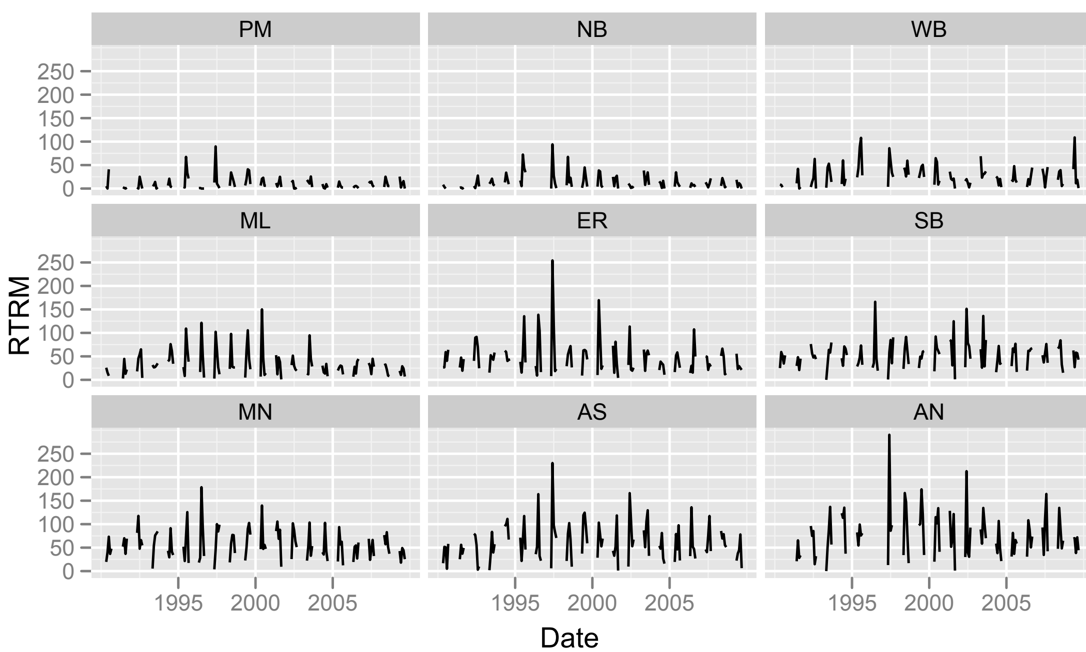

Figure 4.16: RTRM monthly time series.

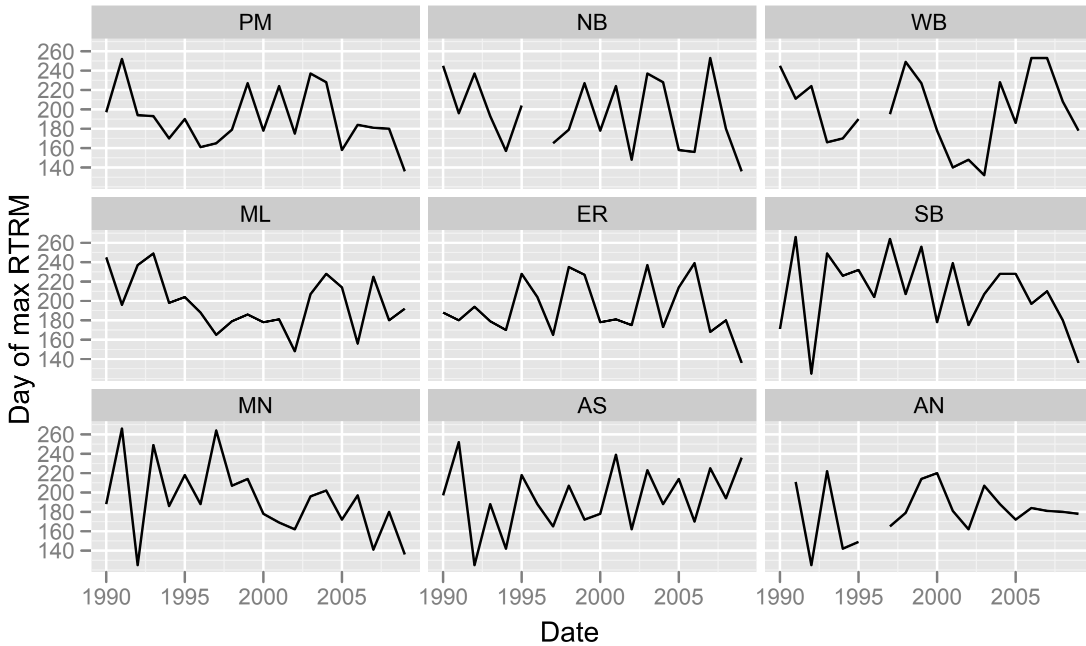

Figure 4.17: Day of the year that RTRM reached a maximum at each site.

	PM	NB	WB	ML	ER	SB	MN	AS	AN
sen.slope	0.05	-0.04	0.00	-0.73	-0.87	0.49	-0.42	0.38	0.48
sen.slope.pct	0.49	-0.30	0.00	-2.15	-1.83	0.92	-0.74	0.62	0.63
p.value	0.52	0.73	1.00	0.10	0.06	0.26	0.30	0.56	0.48

We also found no trends (Mann-Kendall) in the annual maximum RTRM at any site, nor any trend (Regional Kendall) for the lake as a whole. We did, however, find a possible trend in the timing of the annual maximum (Figure 4.17). Several Upper Klamath Lake sites appeared to decrease and the Agency Lake sites appeared to increase. Although only MN had a significant site-specific trend ($p = 0.038$), the Regional Kendall trend was significant for Upper Klamath Lake as a whole ($p = 0.003$); the Agency Lake trend was not significant ($p = 0.23$). The Upper Klamath Lake Sen slope amounted to -1.55 d yr^{-1}, or about a month over the 20-year period. It remains to be determined, however, if the estimated trend could be an artifact of changes in sampling time of day over the observational record (Section 3.2.1).

5

Determinants of chlorophyll levels

The course of *Aphanizomenon* concentrations during May–September is perhaps the central water quality phenomenon in the lake, responding to nutrient enrichment and hydrological/climate conditions favorable for growth and resulting in hypoxia and other conditions deleterious to fish populations. Several attributes of the bloom are of interest to the extent that they characterize its size and shape quantitatively. These include the mean; the maximum size; the timing of the maximum; and other features such as secondary maxima, bloom duration, rate of decline at bloom end, etc. Another study of the causal factors underlying variability of chlorophyll and other factors was completed recently (Morace 2007); although the scope for our analysis is relatively limited, we briefly examine a different approach on a single attribute, namely, mean water column chlorophyll.

There are several "external" factors that can be expected to exert influence on chlorophyll concentrations. We focus on weather variables—daily mean temperature (°C) and daily maximum of the running 4-h mean of wind speed (m s^{-1})—and a morphometric/hydrological variable—lake surface elevation (m above sea level). Temperature and wind both affect water column stratification and therefore vertical mixing in the lake, which in turn has effects on nutrient supply and light availability. Temperature, in addition, directly affects growth rates. Depth—in addition to an influence on flushing rate—affects the water column light climate and the ability of wind-induced mixing to re-aerate bottom waters and resuspend bottom material. As pointed out above (Sections 4.2, 4.3), there is much interannual variability in these factors that could have an impact on phytoplankton biomass. We excluded flushing rate because its typical magnitude is quite small compared with net growth and loss rates of chlorophyll, although other water quality variables such as nutrient pools may be affected.

Temperature and elevation change relatively slowly and it is not too important whether daily means or simple moving averages are used (Figure 5.1). The wind variable, on the other hand, undergoes relatively rapid fluctuations and the choice of averaging period is more important. The wind spectrum shows a variability peak at

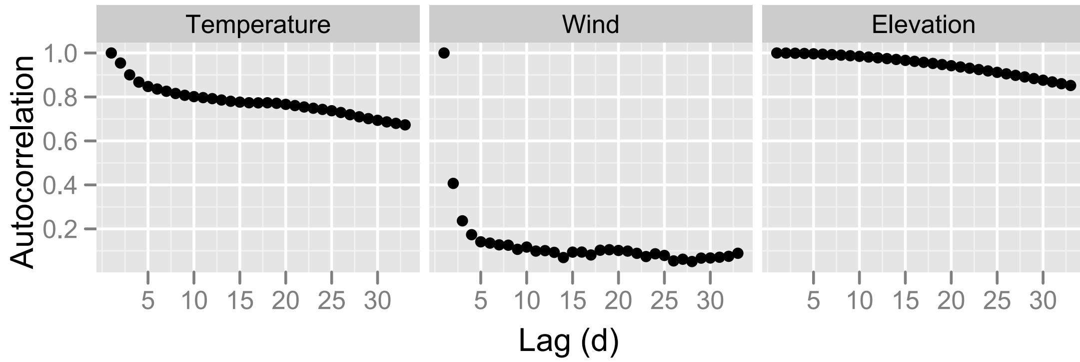

Figure 5.1: Autocorrelation functions for the climate and hydrological factors.

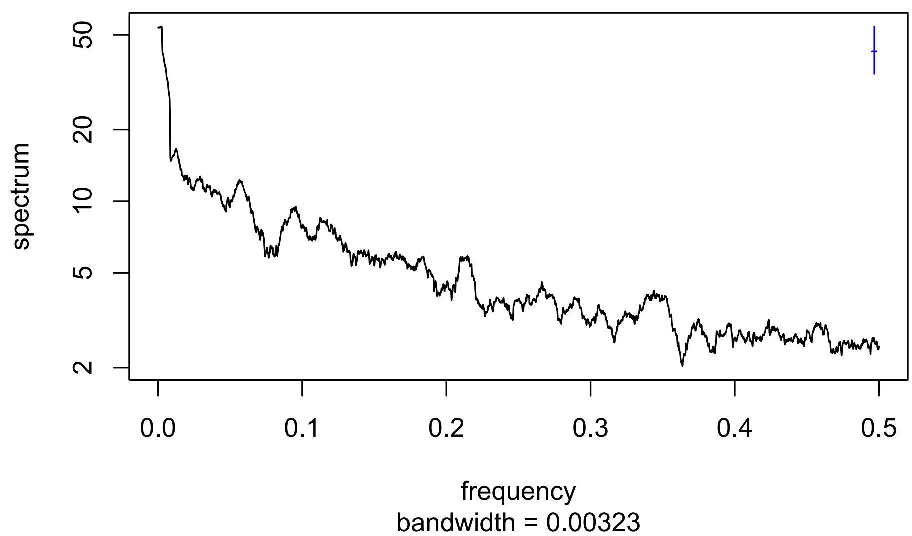

Figure 5.2: Smoothed periodogram estimate of the spectral density function for maximum daily 4-h running mean wind speeds. *Vertical line,* 95% confidence interval for the spectrum.

a minimum wavelength of about 3 d (maximum frequency of about 0.33 d^{-1}), which can be taken as an average length for a weather system at the lake, for the purposes of this analysis (Figure 5.2). Accordingly, we use a 3-d simple moving average for the weather variables.

We binned all data by station and month, using the median, and normalized the chlorophyll values by taking their logarithm (to the base 10). For each month, May–Sep, we then fit the following linear mixed-effects model to water quality observations during that month: $log_{10}(chl) = b + \beta_1 temp + \beta_2 wind + \beta_3 elev$, where the variables are as described above, b is a random effect dependent on station, and the β_i are constant parameters. The fitted models were then subjected to a backward elimination of factors to determine which ones were necessary to account for chlorophyll variability, within the context of the model and data. For this first exploration, we did not try to account for spatial correlation among sites, which means that the significance levels reported are biased to lower levels. Thus, the results presented here should be considered part of an exploratory phase of analysis in which effects of external variables should be further refined.

First, we present some raw plots of the individual data to see what relations they suggest. Figures 5.3-5.5 show plots of chlorophyll versus the "predictor" variables by site and month. The straight lines in these plots are robust regression lines (using an M estimator: Venables and Ripley 2002) that are less sensitive to outliers than ordinary regression, and the shaded areas are 95% CIs. Several features stand out in these graphs. First, different sites can have different slopes as well as different intercepts. Our first models account for intercept differences only. Second, slopes can change during the season (compare June and August temperature effects, for example, in Figure 5.3), which suggests that pooling data over too long a period could disguise causal relationships. In fact, biweekly binning may turn out to be preferable to the monthly bins used here. Third, some slopes are determined by a few influential points, as is the case for surface elevation (Figure 5.5). In some cases, the slopes are not significant (e.g., ML in June, when the 95% CI is large enough to contain lines of positive *or* negative slope), but in others they may be valid despite the few points (e.g., MN in June). Finally, these relationships are univariate and they can break down when multiple variables are tested simultaneously. In any case, the signs of the slopes in June, for example, make biological sense, with higher temperature apparently favoring growth in early summer, and wind and water depth each opposing development of phytoplankton populations at this time.

Fitting models for each month, followed by backward elimination of factors, led to a choice of predictor variables for each month (Table 5.1). No important factors could be identified for September. The site effects account for some of the variability by accounting for different mean chlorophyll levels at each site, so it is useful to compare correlations between observed and predicted values for the final model with the null model that has site effects only. Although at least one predictor variable was identified for models in May–Aug, only in June and July did the predictors contribute substantially to correlations, so we will examine those months more closely. Note that the β_i were negative in the final model for all months, May–Aug.

Figure 5.6 shows the site-specific fits for the June model. Note the large differences among sites in how well the model describes the observations. Table 5.2 summarizes

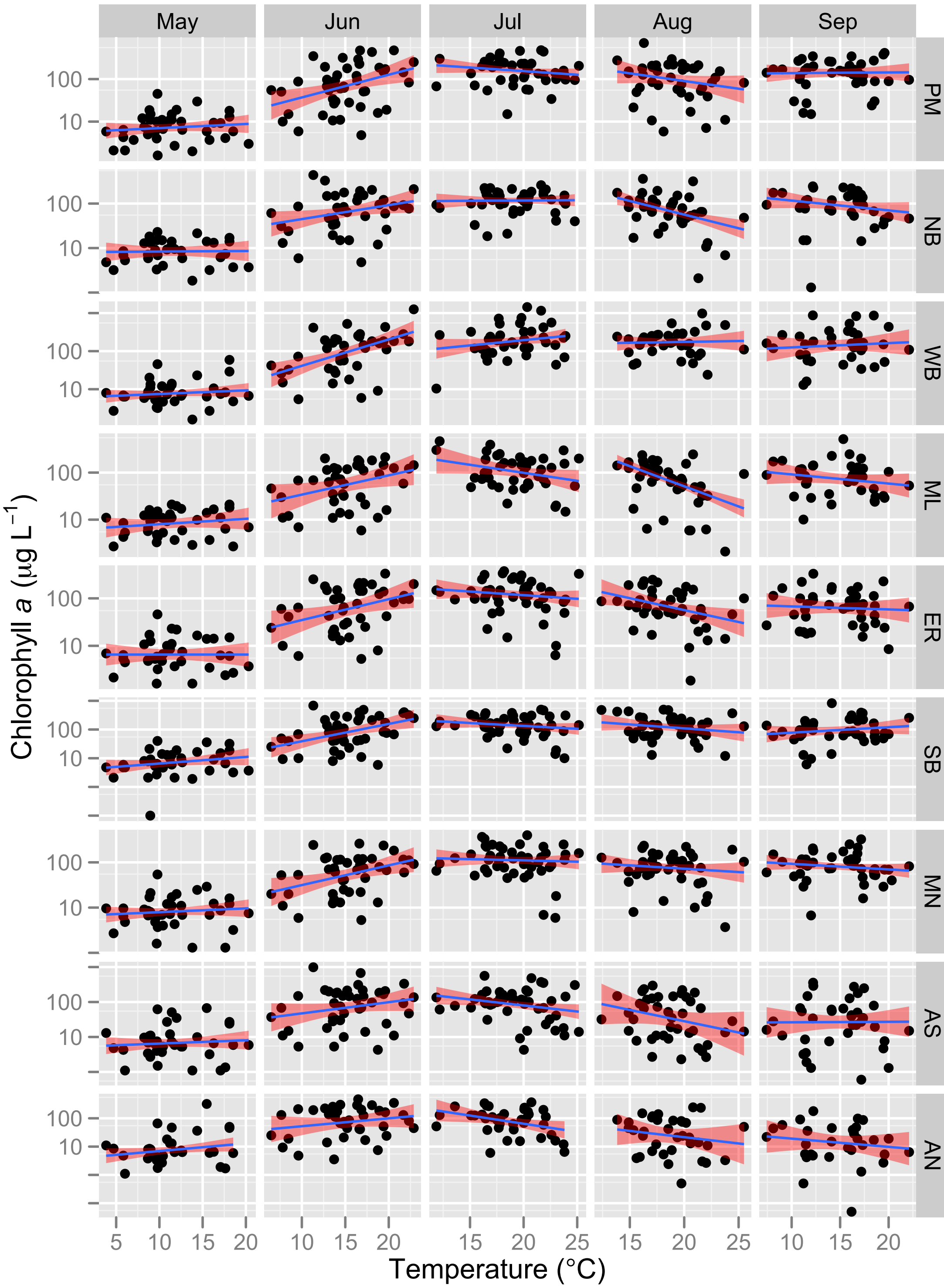

Figure 5.3: Observations of chlorophyll versus (3-d simple moving average of) daily mean air temperature.

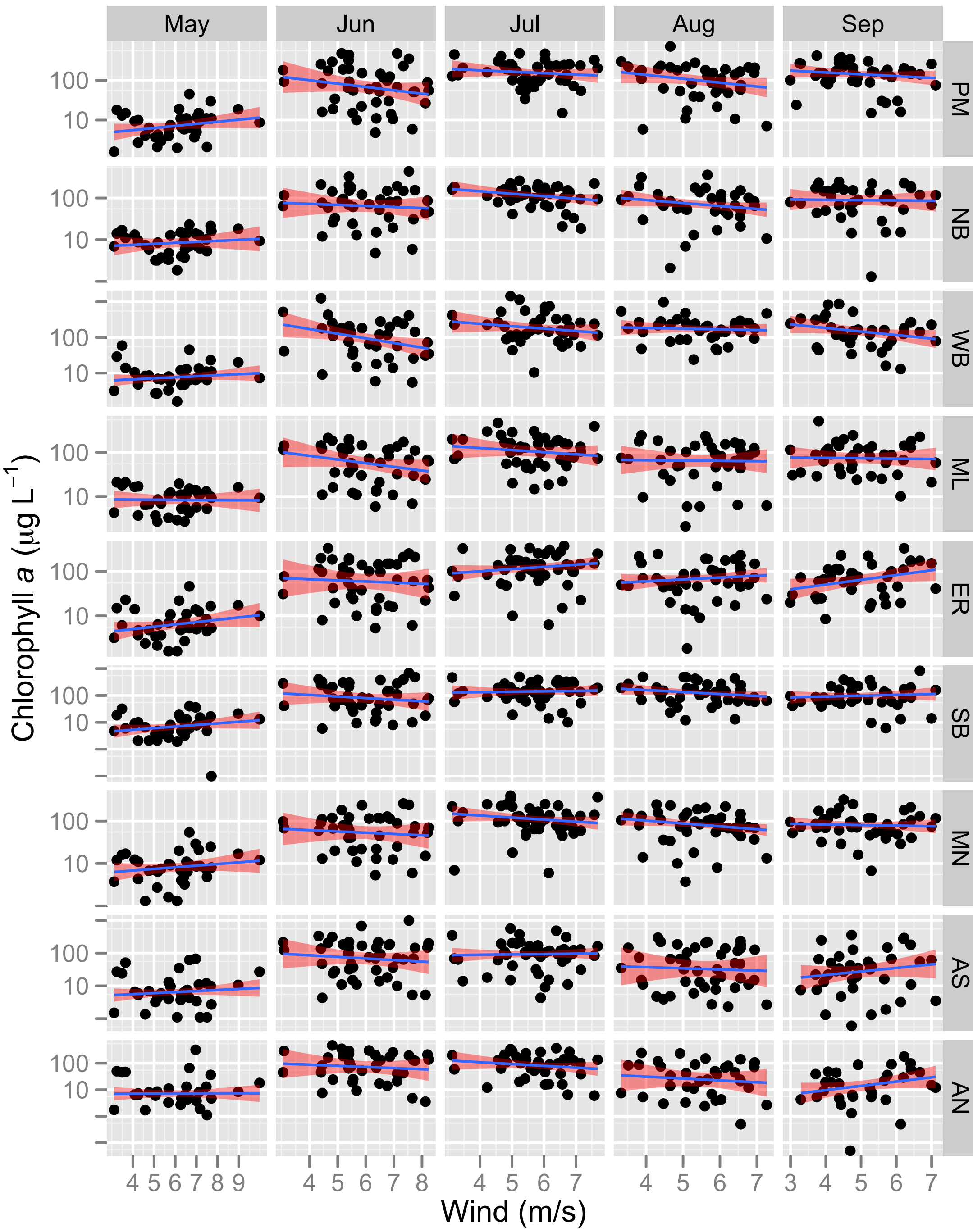

Figure 5.4: Observations of chlorophyll versus (3-d simple moving average of) daily maximum running 4-h wind speed.

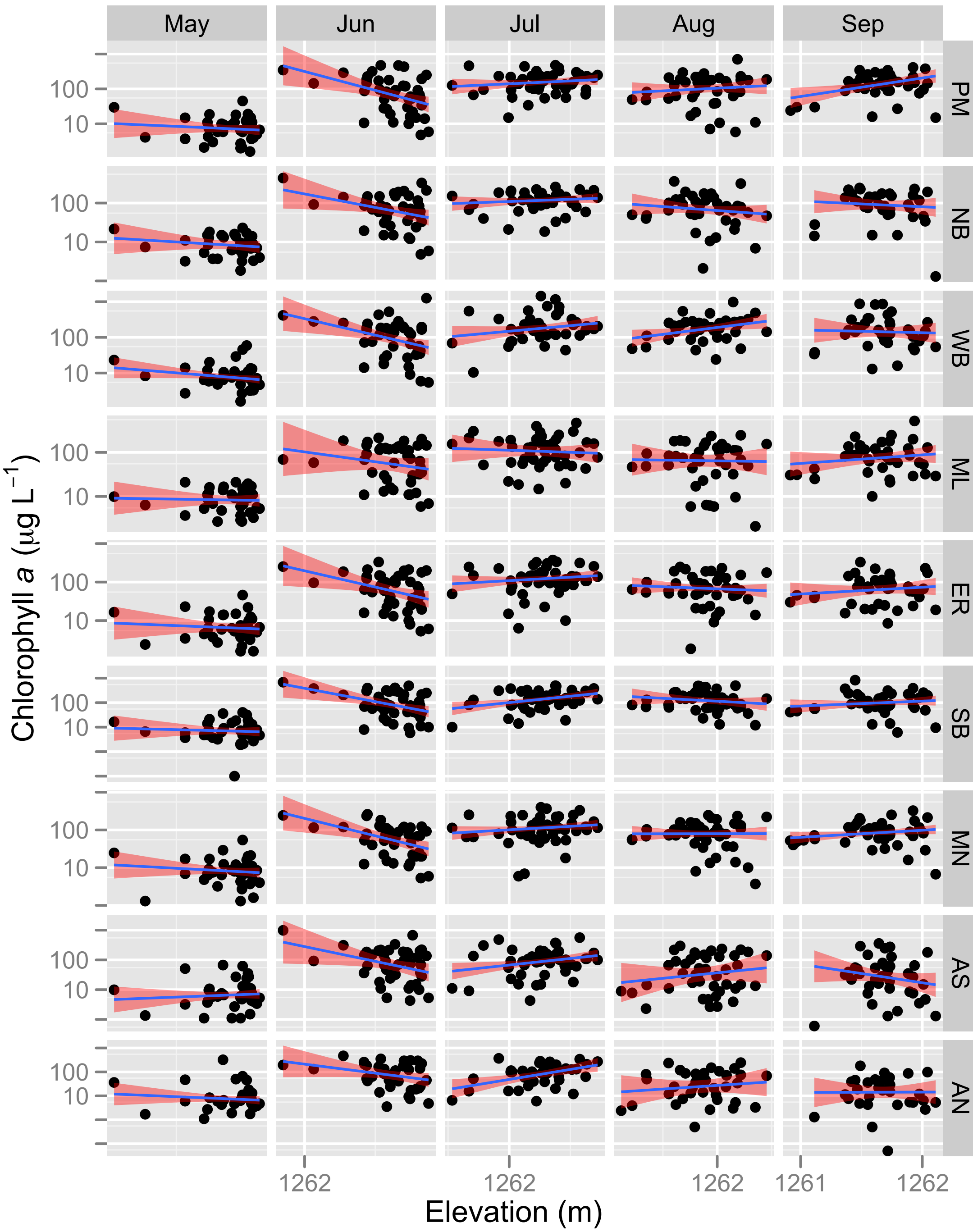

Figure 5.5: Observations of chlorophyll versus lake surface elevation.

Table 5.1: For each monthly model, the factors retained, the correlation between observed and predicted values for the null model, and the same for the final model.

Month	Factors	Null	Final
May	elevation	0.144	0.169
June	elevation, wind	0.193	0.362
July	temperature, wind	0.370	0.435
August	wind	0.526	0.527
September	no factors	0.617	0.617

fixed-effects components of the models for June and July. The coefficients do appear to be statistically significant, although further refinement of the p-values should be undertaken by accounting for spatial correlation. The table also includes a column labeled "effect". These numbers represent the effect that a "typical" change in the variable would have on chlorophyll levels, according to the model. It is calculated as $10^{iqr \times value}$, where iqr is the interquartile range, i.e., the difference between the first and third quartiles for a specific variable and month, and $value$ is the corresponding parameter value. 10 is raised to the power of $iqr \times value$ in order to account for the log_{10}-transform. The resulting number is the fraction of the original chlorophyll that would remain after increasing the variable from the first to third quartile. For example, an increase in elevation in June from 1262.501 to 1262.763 m could lead to a 37% decrease in chlorophyll levels according to the model. The potential effects for wind and temperature are smaller but notable.

Although preliminary, the above approach provides evidence for the effect of wind, temperature and lake elevation on UKL mean Chl values, in varying combinations and to varying degrees during May–August. Another point of this exercise is to suggest the importance of building models that make direct use of site effects, which can enhance the ability to determine relationships that may be obscured when binning data from different sites together. As pointed out above, further model refinement that incorporates spatial correlation is the next logical step. Such an approach can also be utilized to determine relationships among these and other predictor variables and such response variables as DO levels problematic for endangered fish. For example, Morace (2007) noted that both water temperature and lake level appear to be important variables in explaining the variance observed in minimum DO at Midnorth (MN).

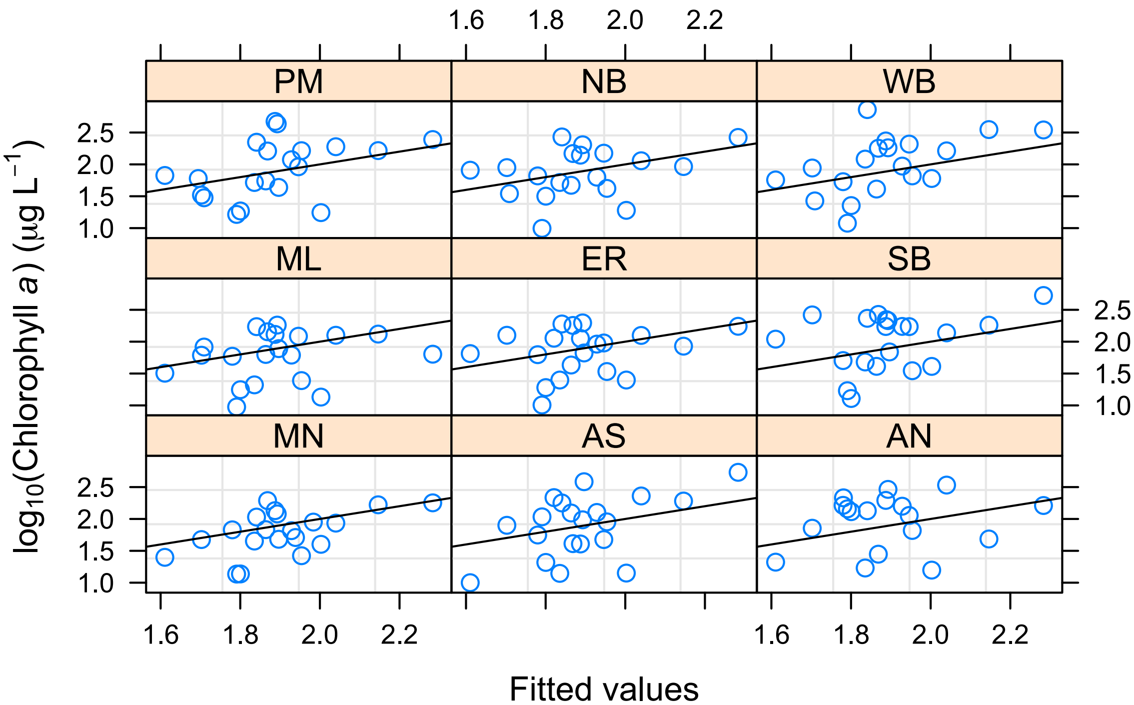

Figure 5.6: Observed versus fitted values for the June model. *Straight line,* line with intercept 0 and slope 1.

Table 5.2: Fixed-effects components of models for June and July.

	Value	Std.Error	DF	t-value	p-value	Effect
June						
intercept	981	199	165	4.92	< 0.001	
wind	-0.116	0.0366	165	-3.19	0.002	0.72
elevation	-0.775	0.1580	165	-4.91	< 0.001	0.63
July						
intercept	3.20	0.334	165	9.59	< 0.001	
temperature	-0.0391	0.0118	165	-3.31	0.001	0.83
wind	-0.0662	0.0276	165	-2.40	0.018	0.87

6

Summary and conclusions

- The Klamath Tribes have been monitoring water quality of Upper Klamath and Agency lakes on a biweekly basis since 1990. Of particular interest are nine core sites throughout the lakes that have been sampled almost consistently from May through September.

- The lake is shallow, turbid, and hypereutrophic, with occasionally extreme Chl concentrations and resulting levels of DO, un-ionized NH_3 and pH stressful to fish. The phytoplankton is dominated by the bloom-forming cyanobacterium *Aphanizomenon flos-aquae*. Two important fish populations in the lake—Lost River (*Deltistes luxatus*) and shortnose (*Chasmistes brevirostris*) suckers—have been listed as endangered since 1988.

- Compared with Agency Lake, Upper Klamath Lake appears to have less transparency, lower temperatures, lower DO, lower TP and SRP, higher TN and higher Chl, based on overall observations during 1990–2009.

- Because sites are sampled in a fixed order at different times during the day, certain measurements—e.g., surface temperature and RTRM—probably reflect a bias due to time of day. Contrasts between Agency and Upper Klamath lakes for these parameters may be affected by this bias. A time-of-day adjustment could conceivably be calculated for the data because of the large number of observations available. In the interim, using water-column averaged measurements and comparing the same set of sites between sampling days, as opposed to different sites on the same day, will help to avoid biased results.

- The most anomalous sites with respect to water quality—in particular, DO, Chl, NH_3, TN and TP—were the embayment sites WB and SB, and the deepest site ER. Based on frequency of deleterious temperature, DO, pH and NH_3 conditions, these same sites as well as the southernmost (PM) and northernmost (AN) sites were the least suitable for fish over the whole record.

- Conditions deleterious for Lost River and shortnose suckers in the lake can be described in terms of temperature, DO, pH and un-ionized NH_3. Although excessive temperatures were not very frequent, a high proportion of exceedances for both DO and pH occurred at certain sites ($\geq$ 25% of the observations). Stressful un-ionized NH_3 conditions occurred more frequently at certain sites ($\geq$ 10% of the observations) than temperature but less so than DO and pH.

- Based on the entire record, median lake surface elevation peaked in May and then declined until October, whereas median flushing rate rose through August. Median temperature and water column stability was highest in July. Median wind speed peaked in May and then declined slowly through the summer growing season.

- Median DO was at a minimum in August, including hypoxic and anoxic conditions for certain sites and years. pH peaked in July, as did Chl, TP, TN and NH_3. The weight ratios of TIN:SRP were suggestive of N-limitation, especially as the growing season progressed.

- Water quality at each site represents phenomena originating at the site mixed with those of surrounding sites. Not only is there variability among sites in terms of seasonal pattern of water quality, but groupings of sites with similar patterns change from year to year. WB and SB frequently stand out from the others, reflecting their relative isolation from the main circulation. PM is also often anomalous.

- The frequency of deleterious conditions for DO, pH, NH_3 and temperature showed no significant long-term trends, either on a specific site or combined regional basis.

- Trend testing suggested a long-term increase in Secchi depth and SRP over the record, a decrease in temperature, DO and phaeophytin, and a decrease in Chl except for the southernmost stations.

- Most recent years have been below the long-term average Chl, but there is no unique turning point and the trend is not monotonic. For northern Upper Klamath Lake, 2001–2002 may have some significance as a turning point, but for Agency Lake sites, 1996–1997 appears to have been more important. The shift at AN in 1997 coincided with the first several years of the Bureau of Land Management (BLM) program to restore 12 km^2 of wetland near the mouth of the Wood River. The annual maximum bloom size also showed a tendency to lower values after around 2000, more or less, depending on the site.

- July was the most common month for the annual Chl maximum at the Upper Klamath Lake sites, and June at the Agency Lake sites. But a later maximum in September was not unusual, especially at the southern sites. Although the day of the annual maximum showed no clear long-term trend, the center of gravity of the May–September Chl series shifted to later in the season, at a regional long-term rate of about 1 d yr^{-1}, or about three weeks over the observational record.

- The increase in Secchi depth was due to the decrease in Chl, as opposed to a decrease in mineral suspensoids.
- The different sites were relatively coherent over the years with respect to Chl levels in May, but the coherence deteriorated for the next two months and remained low for August and September. Once again, WB and SB tended to be the most anomalous.
- There were no detectable long-term trends in lake surface elevation or flushing rate. Trends in surface elevation were consistently positive during Dec–May and negative during Jun–Nov, even though trends at individual months were not significant.
- Mean daily air temperature had a small negative long-term trend, but no trend was found for wind speed or in mean or maximum RTRM. There appears to have been a long-term trend in the timing of the annual RTRM maximum in Upper Klamath Lake to earlier in the year, but further analysis is required to determine if this shift could be an artifact of changes in sampling time of day over the years.
- Plots of Chl by month versus air temperature, wind speed and lake elevation showed that different sites can have different slopes in addition to different intercepts, and that slopes can change sign during the season.
- Linear mixed-effects models that allowed different intercepts for different sites were used to examine the relation between Chl and these three external factors, and backward elimination was used to pare down the models to predictive factors that mattered. In the context of these models and data, surface elevation was important in May–Jun, wind in Jun–Aug and temperature in July. Only in June and July did the models contribute substantially to explaining variability. The potential effect of the external factors in those months was large: when their value changed from the first to third quartiles of their observed distribution, the resulting Chl changes ranged from 13 to 37%.
- Although preliminary, the modeling approach provided evidence for the effect of wind, temperature and lake elevation on UKL mean Chl values, in varying combinations and to varying degrees during May–August. The results suggest the importance of building models that make direct use of site effects, which can enhance the ability to determine relationships that may be obscured when binning data from different sites together.

- Further studies of this nature should consider the following refinements:

  1. Biweekly instead of monthly binning of data;
  2. Time series models that allow lagged relationships between response and predictor variables;
  3. Lagged differences or ratios of Chl, rather than Chl itself, as the response variable, in order to assess effects on net growth and loss rates rather than biomass;
  4. Incorporation of nutrient loading as a factor;
  5. Incorporation of spatial correlation among sampling sites;
  6. Expanding the scope of mixed-effects models, including site-specific effects on slopes and nonlinear relations.

- Additional modeling studies utilizing this and similar approaches could be used to study the effect of other predictor variables such as phosphorus and nitrogen, as well as different response variables such as peak chlorophyll concentration, timing of chlorophyll peak and decline, magnitude of chlorophyll decline, and other water quality variables such as dissolved oxygen and pH.

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Appendix A

Water quality sites and variables

Table A.1: Upper Klamath Lake monitoring program primary and derived variables.

Name	Description	Units
ZMAX	Maximum Depth	m
SECCHI	Secchi Depth	m
T	Temperature	C
DO	Dissolved Oxygen	mg/L
PSAT	Percent Saturation	percent
PH	pH	pH units
COND	Specific Conductance	uSiemens/cm
TP	Total Phosphorus	μg/L
SRP	Soluble Reactive Phosphorus	μg/L
TN	Total Nitrogen	μg/L
NO	Nitrate+Nitrite Nitrogen	μg/L
NH	Ammonia Nitrogen	μg/L
MCHL	Chlorophyll a	μg/L
PHAE	Phaeophytin	μg/L
LIGHT	Photosynthetically Active Radiation (PAR)	μE m^{-2} s^{-1}
LIGHTS	PAR at surface	μE m^{-2} s^{-1}
LIGHT0	PAR at 0m	μE m^{-2} s^{-1}
EXTCOEFA	Light Extinction Coefficient	per m
LIGHT1	PAR at 1m	μE m^{-2} s^{-1}
PER01	Percent of 0m light remaining at 1m	percent
MA1PH	Maximum pH	pH units
MI1T	Minimum T	C
MA1T	Maximum T	C
MI1DO	Minimum DO	mg/L
MA1DO	Maximum DO	mg/L

continued

Table A.1 continued

Name	Description	Units
MI2PSAT	Minimum PSAT	percent
MA2PSAT	Maximum PSAT	percent
DOWGTM	DO water column weighted average	mg/L
TWGTM	T water column weighted average	C
PHWGTM	pH water column weighted average	pH units
PSATWGTM	PSAT water column weighted average	percent
DODIF	Surface to bottom DO difference	mg/L
PHDIF	Surface to bottom pH difference	pH units
TDIF	Surface to bottom T difference	C
PSATDIF	Surface to bottom PSAT difference	percent
LIGHTDIF	Surface to bottom LIGHT difference	μE m^{-2} s^{-1}
TPCR	TP to Chl ratio	
RTRM	Relative Thermal Resistance to Mixing	
TNTP	TN to TP ratio	
TIN	Total Inorganic Nitrogen	μg/L
TINSRPR	TIN to SRP ratio	
SRPTP	SRP to TP ratio	
PCHLR	TP to Chl ratio	
CHLTPR	Chl to TP ratio	
CHLTNR	Chl to TN ratio	
CHLTIN	Chl to TIN ratio	
TINTPR	TIN to TP ratio	
CHLSRP	Chl to SRP ratio	
UNION or NH3	Un-ionized Ammonia Nitrogen	μg/L
ZPHOTIC	Depth of photic zone	m
PERPHOTIC	Percent water column in photic zone	percent

Table A.2: Upper Klamath Lake monitoring program sites

Site no.	Site	Description	Code	Latitude	Longitude
1.0	AS	AGENCY SOUTH	KL0010	42.52358	121.9843
1.5	ASJ	AGENCY JAKES		42.53143	121.9311
2.0	ER	EAGLE RIDGE	KL0006	42.42208	121.9433
3.0	ML	MID LAKE	KL0005	42.36908	121.8487
4.0	MN	MID NORTH	KL0008	42.44136	121.9988
5.0	NB	NORTH BUCK IS.	KL0003	42.30839	121.8562
6.0	PB	PELCIAN BAY		42.45988	122.0764
7.0	PM	PELICAN MARINA	KL0002	42.23803	121.8104
8.0	SB	SHOALWATER BAY	KL0007	42.40725	121.9631
9.0	WB	WOCUS BAY	KL0004	42.32639	121.9200
10.0	WS	WEATHER STATION		42.53143	121.9311
11.0	AN	AGENCY NORTH	KL0011	42.56056	121.9474
12.0	FB	FREMONT BRIDGE	KL0001	42.23873	121.8047
13.0	PBL	PELICAN BAY INTERFACE		42.45349	122.0580
14.0	CP	COON POINT	KL0009	42.43572	122.0282

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