A Multiple Lake Method to Analyze for Temporal Variations

by

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The Soap and Detergent Association initiated a lake monitoring survey from 1978 through 1983 to ascertain the effectiveness of an interim detergent phosphorus ban in the state of Wisconsin. A paired-lake approach was employed in this study. Lakes expected to be impacted by wastewater effluent nutrient loadings were monitored along with lakes not impacted by effluent. Physical, chemical and biological parameters were recorded. The ensueing statistical analysis has based comparisons between geographically proximate lake pairs, one representing an effluent impacted lake and one not. Initial work has focused upon the use of a linear model to determine whether a time-dependent change in either total phosphorus or chlorophyll-a concentration or Secchi depth could be determined between the lake pairs.

Biological parameters included algal cell concentrations erumerated by algal division. This report applies our linear modeling technique to these parameters and also investigates the data requirements which would be necessary in order to increase the accuracy of determining a change associated with imposition of the detergent phosphorus ban.

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I. INTRODUCTION

Legislation banning the use of phosphorus in laundry detergents has been considered in several areas of the United States as a way to mitigate the eutrophication of surface waters, particularly lakes [Maki, et. al., 1984]. The state legislature of Wisconsin enacted a three-year ban on the use of phosphorus in detergents which became effective 1 July 1979. The purpose of an interim ban was to allow an assessment of any immediate regional water quality response which could be associated with its imposition. The Soap and Detergent Association initiated a Wisconsin lake study program in 1978, and continued it through 1983, in order to provide information which would contribute to an evaluation of the effectiveness of the ban.

Ten lakes were selected and monitored during the study period [Figure 1]. It may be noted that the concentration of selected lakes is in the north, however, this is consistent with the natural aggregation of lakes within the state [Lillie and Mason, 1983]. The study lakes were divided into three groups each consisting of a "test" lake[s] and a "reference" lake. Test lakes were those for which historical evidence, available from either the National Eutrophication Survey [NES] or the Wisconsin Department of Natural Resources [WDNR], indicated that a percentage of their nutrient loading resulted from the discharge of sewage effluent. Effluent loadings, when linked to the presence of wastewater treatment plants, occured as either direct discharge to the surface water or land discharge within the drainage basin, or resulted from the discharge of individual septic tank/leach field systems. Reference lakes were those determined to not be impacted by sewage effluent from those sources.

As shown in Figure 1, test and reference lakes were grouped. The basis for the groupings lies both in geographic proximity and in regional geology. In comparing data collected from different lakes those factors which would contribute to normal fluctuations in water quality data, namely climatic conditions [i.e. temperature, rainfall amounts and frequency], were sought to be minimized by considering only geographically proximate test-reference lake pairs. In addition, each lake group lies in a unique geologic region of the state [Prescott, 1962]. The Teal lake group, in the northwest, is located within a region of granite topsoil underlain by crystalline rock. The Little Bearskin lake group, in the northeast, is located within a region of sandy soils, also underlain by crystalline rock. The Fish-Swan lake pair, in the southeast, is located in a region of glaciated limestone soil underlain by a limestone bedrock.

Morphological characteristics of the lakes and their drainage basins have been reported [Clifford, et. al., 1986]. Similarities between test-reference lake pairs occurred, overall, for most of the cases. The study lakes ranged in size from 36 to 425 hectares. Hydraulic residence times for the lakes varied from less than five days to over four years. All of the study lake basins fostered some residential development. Drainage basins for the lakes in the Little Bearskin group were generally forested (for the group an average of 70% of the basin area) while agriculture can be cited as the most prominent land usage around the Swan-Fish lake pair (an average of 68%). The basins surrounding Teal and Balsam lakes are chiefly forested and marshy. Butternut Lake's basin is a mix of forested and agricultural land (40% and 34% respectively) while Elk Lake's basin is, for the most part, residential (approximately 45%). Moss and Fish lakes are seepage lakes; the remaining study lakes are drainage lakes.

II. MONITORING METHODOLOGY

Samples and measurements were taken from 5 to 9 times each year of the study, during ice-free conditions, and from at least two sites on each lake. Two meter integrated water samples were collected. For each site, the following parameters were recorded: temperature, dissolved oxygen, conductivity, Secchi depth, chlorophyll-*a*, total phosphorus, total filterable phosphorus, total oxidized nitrogen, total kjeldahl nitrogen, and ammonia. An aliquot of the deep-site integrated sample (collected at the location of maximum depth, as determined by bathymetric charts) was preserved in an amber glass bottle with Lugol's solution (APHA, 1976) for identification and enumeration of phytoplankton species. Analysis of these samples for algal species was conducted under the auspices of the Environmental Research Group of St. Paul, MN and followed the taxonomic key of Prescott (1962). The algal genera identified are listed in

Table 1. Among all of the study lakes over 114 different algal genera were identified.

Enumerative data were grouped according to the algal divisions Chlorophyta [green algae], Cyanophyta [blue-green algae], Cryptophyta, Chrysophyta, Euglenophyta, Pyrrhophyta, and Bacillariophyceae [diatoms], the sub-class of the division Chrysophyta. These data were recorded in computer files.

A more complete account of the sampling and analytical methodology employed in the determination of physical and chemical parameters, as well as the statistical analysis conducted upon the parameters Secchi depth, chlorophyll-a concentration and total phosphorus concentration have been reported [Clifford, et.al., 1986]. Subsequent to the analysis of these parameters similar statistical techniques were applied to the algal enumerative data. The results of this analysis follows.

III. ANALYTICAL METHODOLOGY

A. Data

Empirical Probability distributions for the enumerative data, recorded by algal division, as well as tranforms of these data, were calculated using a mainframe statistical package [MIDAS, Statistical Research Laboratory, University of Michigan, 1976]. This analysis confirmed base-10 log normal distributions for modelling of the divisions Chlorophyta, Cyanophyta, Cryptophyta, Chrysophyta and the sub-class Bacciliophyceae.

Because of the infrequent occurence of data values for the divisions Euglenophyta and Pyrrhophyta, a distributional form could not be identified for these divisions. Since the covariance analysis to be employed in the statistical analysis is a parametric test, the divisions Euglenophyta and Pyrrhophyta were omitted from subsequent analysis. In addition, members of the sub-class Bacillariophyceae [diatoms] were found to make up a significant portion of the Chrysophyte community within these study lakes [approximately 51%], hence, only the diatoms were considered in statistical analysis.

Arithmetic yearly means [Table 2] characterize the range of algal cell concentrations found throughout the study. Summer means [June, July, August] followed trends similar to yearly means. It may be noted that data were not collected for the test lakes in 1979, the year the ban was imposed. In addition monitoring of Fish Lake was discontinued following 1981 when excessive macrophyte growths inhibited sampling. Monitoring of Butternut, Swan and Teal lakes was continued through 1983.

Striking increases in yearly mean algal cell counts may be noted throughout the study period, in particular for the Cyanophytes on Butternut and Swan lakes. These increases typically extend over several orders of magnitude. Following a logarithmic transform of the data, these increases appear monotonic and linear. A temporal plot of the data for the test-reference lake pair Elk and Teal [Figure 2] provides evidence of the variability of individual measurements throughout the year. A mid-season maximum is apparent, during some years, for the Cyanophytes [blue-greens] and the Chlorophytes [greens]. No other seasonal trends can be easily discerned.

Simple statistical correlations between biological and physical/chemical parameters [Table 3] provide insight into the general behavior of the algal enumeration data. Blue-green algae's [Cyanophytes] strong correlation with chlorophyll-a concentration reflects the dominance of this division in the study lakes. Cyanophytes averaged 41% of all algal genera found in all the lake samples. Algal cell abundance and a decrease in Secchi depth are noted for all but the Cryptophytes. All groups demonstrate their seasonal variablity with relation to total phosphorus concentration and temperature. Diatoms and Cryptophytes appear to dominate during periods of low temperature and high phosphorus concentration [i.e. spring and fall overturns] while the summer maxima of Cyanophytes and Chlorophytes is evidenced in these group's correlation with temperature [$r = 0.26, 0.44$] and lower correlation with phosphorus [$r = 0.26, 0.18$].

Average cell concentrations ranged from 7267 cells/mL for Butternut Lake to 671 cells/mL for Fish Lake. Concentrations were lowest for the two seepage lakes, Moss and Fish. For most of the lakes Cyanophytes were the dominant division, however, the diatoms made up the largest percentage [41%] of the algal community of Elk Lake most probably due to the low hydraulic residence time [<5 days] and shallow depth [mean depth = 1.5 meters] which would help maintain suspension of the heavy cell frustules.

B. Covariance Analysis for Each Test Lake

For each test-reference lake pair, a covariance analysis was performed upon the algal cell counts, as enumerated by division, using the following model:

$$\log y_{ti} = \beta_0 + \beta_1 \log y_{ri} + \beta_2 B_i + E_i$$

The term y_{ti} represents the i-th observation on the test lake, y_{ri} represents the corresponding i-th observation on the reference lake, and B_i is a dummy variable with value 0 for pre-ban observations and value 1 for post ban observations.

The result of this analysis is displayed in Table 4. In all, 28 cases (four algal groups, seven lake pairs) were evaluated. For 23 of the cases a statistically significant (at the 5% level) linear relationship was determined for divisional cell counts between proximate test and reference lakes, based on the significance of β_1. The coefficent β_2 represents the post-ban change between test and reference lake and is significant, at the 5% level, for eight of the 28 cases. However, for three of these eight cases a linear relationship between test and reference lake cannot be determined. Of the five cases where a post-ban change is detected, and where a linear relationship can be determined, three cases occur for a single lake pair, the Elk-Teal lake pair, and that for Chlorophytes, Cyanophytes and diatoms. One case occurs for the diatoms between the Butternut-Teal lake pair and one case occurs for the Cryptophytes between Enterprise and Little Bearskin lake.

Linear relationships exist for most algal groups between test-reference lake pairs with the exception of the Swan-Fish lake pair. Contributing to this is most probably the lack of 1982 data for Fish Lake as well as differences in volume [16×10^6 versus 6.3×10^6 m.^{3} for Swan and Fish lakes, respectively] and hydraulic residence time [160 versus 1410 days, respectively] which would be expected to slow the reaction of Swan Lake, as compared to Fish Lake, to seasonal fluctuations.

The value of r-squared [r^2], the proportion of variability for the linear model explained by factors affecting both test and reference lakes, ranges from 0.41 to 0.75 when considering the algal concentration data. This is a greater range than that found [0.13 to 0.54] when the same model was applied to data for Secchi depth, total phosphorus and chlorophyll-a between these same lake pairs. For those three physical-chemical parameters, a linear relationship between test-reference lake pairs, significant at the 5% level, was observed in 17 of 21 cases [versus 23 of 28 cases for algal concentration data].

C. Combined Covariance Analysis for All Test Lakes

For a simultaneous examination of all test lakes, a covariance analysis was performed using the following model:

$$\log y_{ti} = \beta_{0_g} + \beta_1 \log y_{ri} + \sum_{j=1}^{g} \alpha_j D_{ji} + \sum_{j=1}^{g} \delta_j D_{ji} \log y_{ri} + \delta B_i + \sum_{j=1}^{g} f_j D_i B_i + E_i$$

Each D_j is a dummy or indicator 0, 1 variable depending upon whether or not the observation is from the j-th test lake or not, and B is a 0,1 variable for pre- or post-ban [g denotes the number of test lakes minus one]. This model first adjusts the total test lake response variablity for a potential relationship with the corresponding reference lake response and for individual test lake differences and then examines for the effect of imposition of the ban. The stepwise model considers, in order, the log test lake response as the sum of a general intercept, a linear component relationship with the log reference lake response, an adjustment in the intercept for the specific test lake, an adjustment in the slope for the specific test lake, an adjustment in the general intercept for pre/post-ban and an adjustment in the intercept for a specific test lake for pre/post-ban. In this analysis additional degrees of freedom become available for statistical testing purposes, giving a greater capability of detecting differences.

The results of this analysis [Table 5] show a strong test-reference lake correlation for the four algal divisions considered [r^2 = 0.39 to 0.60]. Little intercept adjustment is needed when considering individual test lakes and almost no adjustment is made in the model's slope. There is only a marginal adjustment in the model intercept for diatoms when the ban response is considered [Δr^2 = 0.11], and for Cyanophytes [Δr^2 = .06]. However, an examination of the yearly means indicates that this ban response is in the positive direction. In other words, the monotonic increase in diatom concentrations seen for the lakes throughout the study period is somewhat greater for test lakes than for reference lakes, a response which would not be attributable to the phosphate ban.

The reference lake covariate comprises a significant proportion of the model's variability (Table 6). Consideration of the test lake differences did not affect the model significantly. This is in direct contrast to the covariate analysis for Secchi depth, total phosphorus and chlorophyll-a where the greatest proportion of variability in the model occurred in the intercept adjustment for individual test lakes (values of 0.34, 0.49 and 0.22, respectively; for comparison, observe the values for algal concentration, by division, of 0.06, 0.10, 0.14 and 0.04). This would point to the stability of biological indicators, i.e. algal concentration, in environmental impact analysis utilizing the test-reference lake approach. Those parameters typically considered, Secchi depth or chlorophyll-a concentration, which are functions of algal concentration, suffer as indicators most probably due to the robustness of their determination.

A shift associated with the ban was detectable in only six of the 28 cases (Table 7). Four of these shifts were associated with diatoms. In all six cases the indicated change was positive, a response which would not be expected due to the ban. The number of slope estimates that are statisitcally significant (p-value < 0.05) is an indication that the relationship of the reference lakes to the test lakes is accounting for a statistically significant proportion of the variability. This relationship was usefull in making the analysis more sensitive.

D. Multivariate Analysis

A general multivariate analysis, taking into account the covariance structure of the data, was carried out. The measurements for a given lake and year were considered to be a single multivariate variable, or vector, for the purposes of analysis. For each post-ban year, the vector analyzed actually consisted of the differences from the corresponding sampling times for the single pre-ban year. The model can be written mathematically as follows:

$$X_{ijk} = \mu_k + Y_{ik} + L_{jk} + e_{ijk}$$

$(i = 1,2,3,4;\ j = 1,2,3,4,5,6,7,8,9;\ k = 1,2,3,4,5,6,7,8)$

Here X_{ijk} is the k^{th} observation on lake j in year i, μ_k is the k_{th} component of the grand mean, Y_{ik} is the k^{th} component of the effect of year i, and L_{jk} is the k^{th} component of the effect of lake j. For a further explanation of the parameter estimations see Clifford, et. al. [1986] and Morrison [1976].

The analysis was performed using data constructed from the differences of the log test lake responses and the corresponding log reference lake responses, an analysis of the test lake data adjusted for a potential relationship with its corresponding reference lake. A graph [Figure 3] of the simultaneous confidence intervals on the differences between post- and pre-ban years for each point in time that was sampled show no observable effect of the detergent phosphorus ban.

Trautmann, et. al. [1982] discussed the relationship of sample size and the chance of detecting a minimum amount of change [represented in terms of standard deviation units] in limnologic parameters following a detergent phosphorus ban. If corresponding measurements have been made on a reference lake and then employing, as a model, a linear relationship between test and reference lake measurements, the original test lake variance can be reduced by a factor of $1 - r^2$ [where r is the correlation between test and reference lake data]. For example, if $r^2 = 0.50$ then the test lake variance is halved and the standard deviation σ reduced to 0.71σ. Hence, a sample size big enough to detect a change of 3 standard deviation units with test lake data only would detect a change of 2.1 test lake standard deviation units if one incorporated the reference lake relationship with $r^2 = 0.50$. For the variable discussed in this paper [divisional algal cell concentrations] the value of r^2 ranged from 0.39 to 0.60. Hence, the incorporation of the test-reference lake relationship improves detectability of a shift for a given sample size of test lake data. In a sense, this is an improvement of precision of test lake inferences using concomitant data on a reference lake. If one reference lake can be used for several test lakes, an efficient sampling program can be realized.

IV. CONCLUSION

This study has shown that a link between a ban on detergent phosphorus and a change in divisional algae concentrations does not exist for the seven lakes studied in Wisconsin. This result concurs with an earlier finding that the interim detergent phosphorus ban did not effect the concentrations of total phosphorus or chlorophyll-*a* or the Secchi depth for the same seven lakes. Following the study a re-evaluation of the phosphorus budgets for the lakes determined that only four of the seven test lakes could ascribe detergent phosphorus as a potential nutrient source. Therefore, for three of the lakes, in addition to the reference lakes, a ban on detergent phosphorus could have no effect upon water quality. The statistical analyses employed have borne this out and shown that any shifts in algal concentrations between test and reference lakes move in a direction which would not be attributable to the detergent phosphorus ban.

The three statistical models used in this analysis have determined a significant relationship between geographically proximate lakes in terms of divisional algal concentrations. The strength of this relationship is greater than that when common limnological trophic indicators, such as total phosphorus or chlorophyll-*a* concentration or Secchi depth, are considered. A test-reference lake monitoring approach, coupled with consideration of biological parameters, would figure to be a effective strategy in environmental impact analysis.

V. ACKNOWLEDGEMENTS

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FIGURE 1. LOCATIONS OF THE WISCONSIN STUDY LAKES

FIGURE 2. Temporal Plot of Algal Enumeration Data, Elk Lake (T) vs. Teal Lake (R).

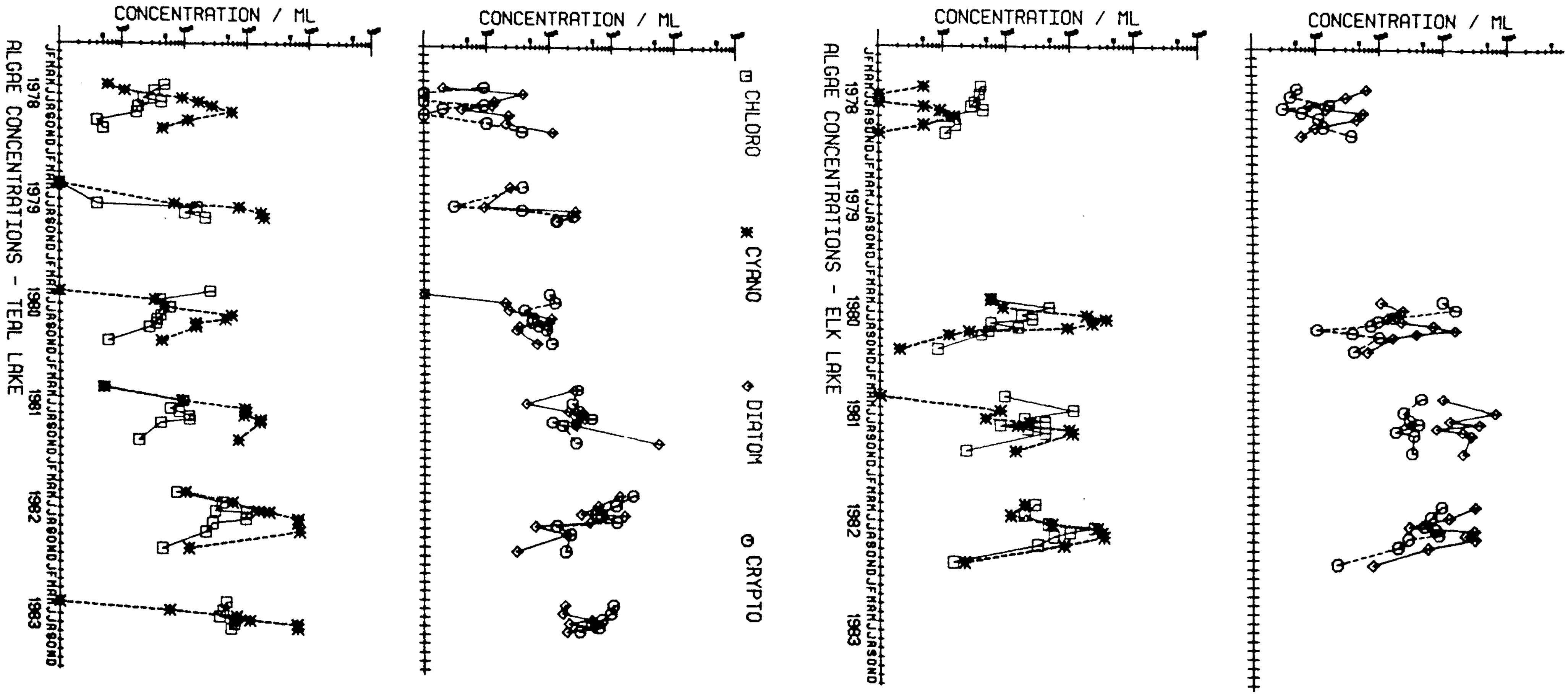

FIGURE 3. Estimated differences and 95% confidence bounds for post-ban effects. Analysis of differences of logarithmic transform of data between test lakes and reference lakes.

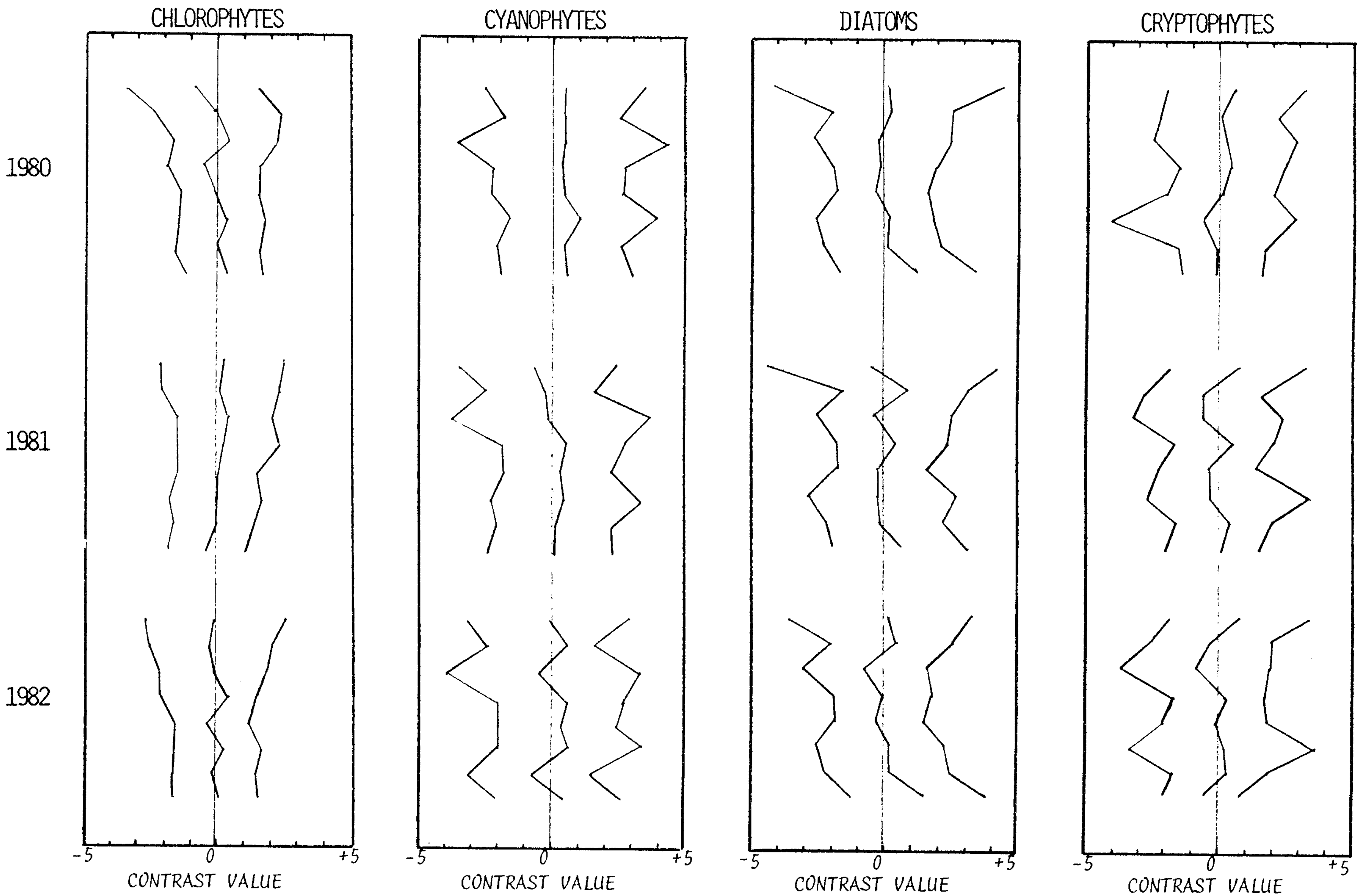

Table 1. Algal Genera Identified During the Wisconsin Lakes Study.
(Genera are listed under name of phytoplankton family).

CHLOROPHYTA		CYANOPHYTA (Cyanobacteria)	CHRYSOPHYTA	DIATOMS	CRYPTOPHYTA	EUGLENOPHYTA	PYRRHOPHYTA
Acathosphaera	Kirchneriella	Anabaena	Attheya	Achnanthes	Chlorochromonas	Euglena	Ceratium
Actinastrum	Lagerheimia	Anacystis	Dinobryon	Amphiprora	Cryptomonas	Lepocinelis	Glenodinium
Ankistrodemus	Micraetinium	Aphanizomenon	Mallomonas	Amphora	Rhodomonas	Phacus	Hemidinium
Arthrodesmus	Mougeotia	Chroococcus	Ophiocytium	Asterionella		Trachelomonas	Peridinium
Binuclearia	Oocystis	Coelosphaerium	Stipitococcus	Caloneis			
Carteria	Pachycladon	Cloeocaspa	Vaucheria	Cymatopleura			
Cerasterias	Pandorina	Cloeothece	[unicellular]	Cymbella			
Characium	Pediastrum	Gomphosphaeria		Diatoma			
Chlamydomanas	Planktospharia	Lyngbya		Epithemia			
Chlorococcum	Pleadorina	Merismopedia		Eunotia			
Chodatella	Quadrigula	Microcystis		Fragillaria			
Closteriopsis	Radiococcus	Nostoc		Gomphonema			
Closterium	Scenedesmus	Oscillatoria		Gyrosigma			
Coelastrum	Schroederia	Phormidium		Melosira			
Cosmarium	Selenastrum	Polycystis		Meridion			
Crucigenia	Sphaerocystis	Spirulina		Navicula			
Desmidium	Spondylosium			Nitzschia			
Dictyosphaerium	Staurastrum			Pinnularia			
Elakatothrix	Stylosphaeridium			Rhizosolenia			
Euastrum	Tetradesmus			Stauroneis			
Eudorina	Tetraedon			Stephanodiscus			
Franceia	Tetraspora			Surirella			
Geminella	Treubaria			Synedra			
Gloeocystis	Ulothrix			Synura			
Golenkinia	Volvox			Tabellaria			
Gonatozygon	Westella			Vaucheria			
Gonium	Xanthidium			[small pennate]			
Haematococcus							
[palmelloid green]							
[unicellular coccoid]							
[unicellular flagellate]							

Table 2: Yearly Arithmetic Means for Algal Concentrations (Conc./mL)

	1978	1979	1980	1981	1982	1983
Butternut						
Chloro	27		49	113	400	356
Cyano	405		1886	3226	14182	13543
Diatom	13		62	264	664	526
Crypto	32		230	397	859	731
Balsam						
Chloro	35		27	124	532	
Cyano	243		588	1180	1907	
Diatom	32		204	1211	1500	
Crypto	13		75	805	1214	
Elk						
Chloro	27		137	304	629	
Cyano	5		939	322	1306	
Diatom	29		381	2393	1658	
Crypto	12		324	299	503	
Townline						
Chloro	18		111	145	304	
Cyano	164		444	2007	3280	
Diatom	25		88	590	663	
Crypto	12		177	342	1007	
Enterprise						
Chloro	24		56	215	483	
Cyano	52		504	1097	1206	
Diatom	35		81	470	719	
Crypto	21		177	404	1136	
Moss						
Chloro	15		40	92	278	
Cyano	26		164	675	1252	
Diatom	13		25	154	273	
Crypto	10		94	333	507	
Swan						
Chloro	7		63	127	634	557
Cyano	200		916	3093	6760	10316
Diatom	23		59	1023	1498	405
Crypto	3		145	272	1381	540
Fish						
Chloro	18	35	44	107		
Cyano	37	884	216	810		
Diatom	9	44	12	143		
Crypto	10	44	106	238		
Teal						
Chloro	24	95	62	65	397	500
Cyano	159	864	177	825	3012	2407
Diatom	28	133	45	875	557	278
Crypto	9	88	78	262	783	676
L. Bearskin						
Chloro	15	50	86	147	473	
Cyano	26	667	192	398	2139	
Diatom	24	141	81	422	490	
Crypto	29	190	161	533	795	

Table 3: Pearson Product Moment Correlation Coefficients (r) for Log Values of Parameters

	Chlorophyll-*a*	Secchi Depth	Total Phosphorus	Temperature
Chlorophytes	.16	-.18	.26	.26
Cyanophytes	.45	-.24	.18	.44
Diatoms	.20	-.21	.37	-.01
Cryptophytes	.10	.03	.23	-.03

Table 4: Covariance Analysis for Individual Test Lakes

Reference Lake	Fish				Teal											
Test Lake	Swan				Balsam				Butternut				Elk			
Parameter	CHLO	CYAN	DIAT	CRYP	CHLO	CYAN	DIAT	CRYP	CHLO	CYAN	DIAT	CRYP	CHLO	CYAN	DIAT	CRYP
Intercept β_0	.29	1.28	.65	.21	.54	.85	.97	.51	.05	.25	.43	.73	.55	-.66	.92	.61
Post-Ban Change β_2	.65	.85	.24	1.17*	.14	.31	.99*	.18	.18	.62	.78*	.51	.39*	1.30*	1.25*	.47
Standard Error (S.E.)	.26	.55	.26	.52	.22	.16	.28	.34	.24	.33	.21	.48	.19	.29	.24	.34
Slope β_1	.45	.26	.84*	.26	.68*	.65*	.30	.72*	.86*	.85*	.46*	.48*	.68*	.64*	.34*	.55*
S.E. of Slope	.23	.40	.17	.28	.16	.08	.15	.16	.17	.15	.11	.24	.14	.14	.13	.17
R-squared	.38	.20	.64	.52	.48	.72	.31	.72	.55	.54	.53	.50	.56	.45	.41	.69
Δ R-squared	.14	.08	.01	.09	.01	.03	.20	.00	.01	.05	.15	.02	.06	.23	.29	.02

Reference Lake	Little Bearskin											
Test Lake	Enterprise				Moss				Townline			
Parameter	CHLO	CYAN	DIAT	CRYP	CHLO	CYAN	DIAT	CRYP	CHLO	CYAN	DIAT	CRYP
Intercept β_0	.65	.23	1.08	.24	.35	-.06	-.21	.33	.53	1.00	.54	-.15
Post-Ban Change β_2	.06	-.27	.58*	.70*	.31	-.11	.37	.61	.45	-.21	.63	.50
Standard Error (S.E.)	.29	.26	.26	.24	.36	.28	.28	.36	.31	.36	.34	.37
Slope β_1	.64*	1.03*	.30	.63*	.55*	1.00*	.75*	.50*	.49*	.80*	.50*	.80*
S.E. of Slope	.21	.13	.15	.15	.27	.14	.17	.22	.24	.18	.20	.23
R-squared	.50	.75	.38	.75	.44	.74	.63	.52	.50	.48	.44	.65
Δ R-squared	.00	.01	.09	.05	.01	.00	.02	.04	.03	.01	.06	.02

TABLE 5.

Combined Covariate Analysis.

Model Steps	Parameter	Sum of Squares	D.F.	Mean Square	Test Stat.	R^2	ΔR^2
Reference Lake Response	Chloro	45.71	1	45.71	210.6*	.47	.47
	Cyano	114.82	1	114.82	275.3*	.49	.49
	Diatom	60.82	1	60.82	220.4*	.39	.39
	Crypto	98.18	1	98.18	362.3*	.60	.60
Intercept Adjustment for Test Lakes	Chloro	5.09	6	.85	3.92*	.52	.05
	Cyano	21.43	6	3.57	8.56*	.58	.09
	Diatom	21.20	6	3.53	12.78*	.53	.14
	Crypto	5.05	6	.84	3.10*	.63	.03
Slope Adjustment for Test Lakes	Chloro	1.16	6	.19	.88	.53	.01
	Cyano	1.96	6	.33	.79	.59	.01
	Diatom	2.18	6	.35	1.30	.54	.01
	Crypto	1.82	6	.30	1.11	.64	.01
Intercept Adjustment for Post/Pre-Ban	Chloro	2.36	7	.34	1.57	.56	.03
	Cyano	14.34	7	2.05	4.92*	.65	.06
	Diatom	17.22	7	2.46	8.91*	.65	.11
	Crypto	5.43	7	.78	2.88*	.67	.03
Residual	Chloro	42.91	198	.217			
	Cyano	82.52	198	.417			
	Diatom	54.67	198	.276			
	Crypto	53.58	198	.271			

* p value less than 0.01

TABLE 6.

Proportion of Variability Explained by Various Sources.

Source	Measurement	Proportion of Variability Explained by Model	Proportion of Total Variability
Reference Lake Covariate	Chloro	0.84	0.47
	Cyano	0.75	0.49
	Diatom	0.60	0.37
	Crypto	0.89	0.60
Test Lake Difference	Chloro	0.12	0.06
	Cyano	0.15	0.10
	Diatom	0.23	0.14
	Crypto	0.06	0.04

TABLE 7. Estimates of the Post-/Pre-ban shift and Slope-Coefficent for Corresponding Reference Lakes.

	Chlorophytes		Cyanophytes		Diatoms		Cryptophytes	
Lake	Post/Pre chng. in intercept	Slope	Post/Pre chng. in intercept	Slope	Post/Pre chng. in intercept	Slope	Post/Pre chng. in intercept	Slope
Townline	.23 [.33]	.48 [.25]	-.21 [.34]	.80* [.17]	.63* [.29]	.50* [.17]	.50 [.35]	.80* [.22]
Swan	.43 [.41]	.32 [.33]	.85 [.51]	.26 [.33]	.24 [.39]	.84* [.25]	1.17* [.55]	.26 [.32]
Balsam	-.08 [.39]	.51 [.29]	.31 [.43]	.65* [.21]	1.00* [.38]	.29 [.21]	.18 [.53]	.72* [.29]
Butternut	-.04 [.39]	.71* [.29]	.62 [.43]	.76* [.21]	.78* [.38]	.50* [.21]	.51 [.53]	.48 [.29]
Elk	.17 [.39]	.67* [.29]	1.30* [.43]	.64* [.21]	1.24* [.38]	.33 [.21]	.47 [.53]	.54 [.29]
Enterprise	-.16 [.47]	.64 [.35]	-.27 [.47]	1.03* [.24]	.58 [.41]	.28 [.24]	.70 [.49]	.63* [.30]
Moss	.09 [.47]	.55 [.35]	-.11 [.47]	1.00* [.24]	.37 [.41]	.64* [.24]	.61 [.49]	.50 [.30]

* p-value less than 0.05

[ ] Standard Deviation