Chlorophyll–Phosphorus Relations in Individual Lakes. Their Importance to Lake Restoration Strategies†

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■ The response of algal biomass to nutrient reduction is critically evaluated in 16 north temperate lakes by using data from the literature. The analysis confirms that reductions in total phosphorus concentration in the lakes are typically accompanied by consistent declines in chlorophyll. The data also suggest that this response can be expected whether a lake is phosphorus or nitrogen limited, although the magnitude of the response may differ. This is in contrast to the conclusion of a recent report which suggests that a threshold response is involved. Furthermore, the responses of some lakes appear unique and may not be accurately predicted by using current global eutrophication models. Modifications of these models to account for additional factors are urged, in order that these prediction errors may be decreased in the future.

In recent years the Dillon–Rigler (*1*) and Vollenweider (*2*) models of phosphorus loading have been used widely to justify phosphorus reduction as a lake restoration measure. At the heart of such models are empirical relationships relating total phosphorus to algal biomass, expressed either as chlorophyll (*3–5*) or as cell volume (*6, 7*). These estimates of algal biomass in turn are used to predict Secchi disk transparency (*1, 5, 6, 8*).

Because of the importance of phosphorus to eutrophication, a recent national study was made to evaluate the effects of eight different phosphorus control options and to identify those options which would have the most significant impact on water quality in U.S. lakes (*9*). Although it was found that several of these options would lead to significant reductions of total P in U.S. lakes, the report concludes that these reductions in phosphorus would have little effect on algal standing crop (chlorophyll *a*). We feel that this conclusion is incorrect and that the apparent insensitivity of chlorophyll to the phosphorus control options results from the chlorophyll/total P model used in the report. This model (Figure 1), which was developed from analysis of 493 lakes taken from the 1972–1975 National Eutrophication Survey (NES), assumes that the best predictive equation is that for the line chlorophyll a = 1.0 total P. The model thus predicts that chlorophyll concentrations in lakes would not decline unless phosphorus were first reduced to a "threshold" concentration defined by the 1:1 line. As a result, the model is extremely insensitive to changes in total P. The behavior of this model contrasts sharply with the Dillon–Rigler (*3*) and Jones–Bachmann (*4*) models, which predict that chlorophyll will respond in a continuous manner to changes in nutrients.

It is important that we know whether a lake's response to changes in total P will be immediate or whether a "threshold" concentration indeed exists, because this response will in part dictate phosphorus management strategies. The purpose of this paper is to evaluate critically the responses of north temperate lakes which have undergone nutrient reductions, in order that we may decide which of these alternatives is correct.

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Much of the data to follow is plotted on linear rather than logarithmic scales even though we have noted by analysis of residuals (*10*) that the relationships may be log/log. We have chosen linear scales because we wish to compare our data directly with the threshold model. No differences in interpretation result regarding the validity of the threshold model.

Analysis of the Model's Assumptions

It is first important to understand why the threshold model differs so significantly in form from previously published chlorophyll/phosphorus relationships (reviewed in ref *6*). The first and probably the greatest source of variance in the NES data set stems from the period of measurement. Although sampling was done from March to October, generally only one datum point was gathered during the summer months. Because of this seasonal bias, it is not surprising that there is so much scatter in the chlorophyll/TP relationship. Megard (*11, 12*), for example, has emphasized that there is a high correlation between chlorophyll and phosphorus concentrations in the various bays of Lake Minnetonka during summer, but *not* during spring and autumn. A plot of all of Megard's (*11*), data without regard to season (Figure 2A), shows a high degree of scatter which closely resembles that found in Figure 1. Remarkably, these data also seem to be bounded on the left by a 1:1 chlorophyll/phosphorus line. However, when May–September means are used (Figure 2B), a strong correlation is evident between chlorophyll and TP in Lake Minnetonka. It is clear that this relationship is obscured in Figure 2A by the inclusion of spring and fall data.

There are a number of factors that help explain the lack of correlation of algal biomass to total phosphorus during spring and fall. During the early spring, algal biomass is most likely controlled by climatic conditions such as light and temperature. In Lake Constance (Germany), for example, phytoplankton production during April–May seems to be influenced strongly by temperature and irradiance (*13, 14*) and less by nutrients. In late spring the algal standing crop was controlled by herbivorous zooplankton. In addition to regulation by temperature, light, and zooplankton, spring crops of algae may also be limited by silicon (*7*) when the algae are predominantly diatoms. Declining temperatures and light availability undoubtedly have a marked controlling influence on algal biomass in the fall as well. Thus, while the threshold model presumes that there should be a correlation between chlorophyll and phosphorus during *all* three seasons, there are clear reasons why this need not be so.

Empirical Test of the Model

The seasonal variability in chlorophyll yield suggests that the data base from which the threshold model was developed was inappropriate. However, this is insufficient evidence by itself that the "threshold" model is incorrect. We therefore tested the predictive ability of this model by using data from lakes outside the NES study.

There are basically two approaches which can be used to test the model. The first is to gather data on algal biomass and phosphorus from a wide spectrum of lakes and then to use the

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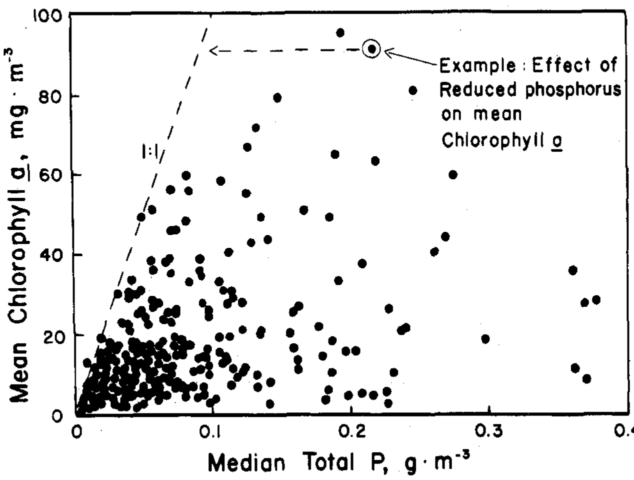

Figure 1. Mean chlorophyll *a* vs. median total phosphorus (493 lakes) (modified from ref *9*). Note example, where the concentration of total P must be reduced to ca. 0.09 g m^{-3} (90 mg m^{-3}) before a response in chlorophyll *a* is expected.

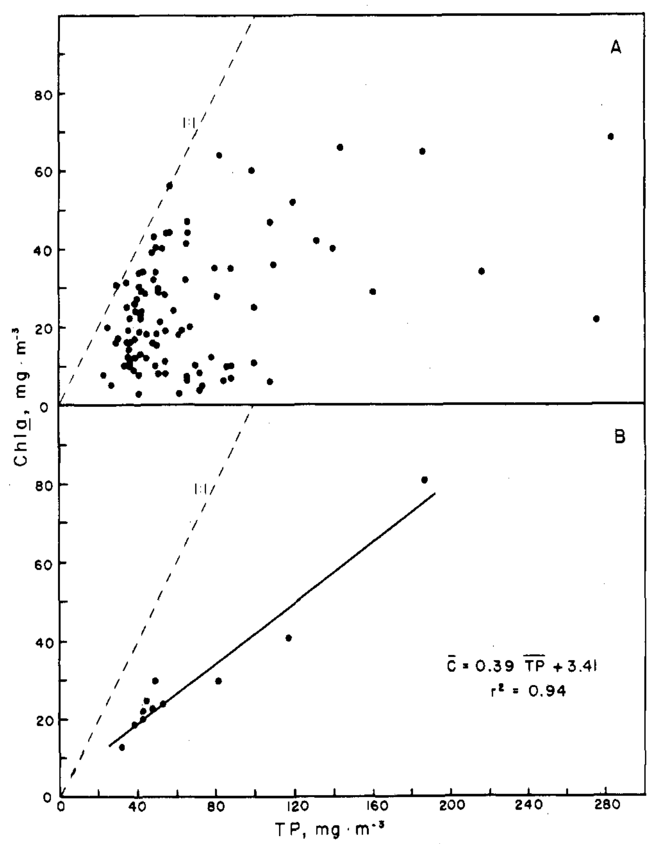

Figure 2. Phosphorus dependence of chlorophyll *a* in 11 bays of Lake Minnetonka, 1968–1969: (A) all dates without regard to season; (B) May–September means (from ref *11*).

line of best fit between these two variables to predict the response of a *single* lake to nutrient reduction. This approach assumes that individual lakes will, on the average, behave in a manner predicted by the composite response of many lakes. This is essentially the line of reasoning originally followed by Dillon and Rigler (*1*). For reasons described later, we have not used this procedure.

The second approach, which we have adopted in this paper, involves an examination of individual case histories of lakes where there have been changes in the euphotic zone phosphorus concentration. This approach does not presume that every lake responds to phosphorus in an identical manner, and is, we feel, a fairer and more appropriate test of the model. Thus, we have directed our analysis toward lakes where nutrient abatement, or natural variation in phosphorus loading, has led to measurable reductions of total P. We have partitioned these lakes into N- and P-limited categories, because Sakamoto (*15*) observed that chlorophyll yield is altered in nitrogen (N)-limited systems. We assume N limitation where the ratio of total N to total P ($\overline{TN} : \overline{TP}$) is <10, or where the total inorganic N/phosphate P ratio ($\overline{TIN} : \overline{SRP}$) is <5 (*16*).

An attempt was made to obtain growing season means of chlorophyll *a* ($\bar{c}$), total phosphorus ($\overline{TP}$), and total nitrogen ($\overline{TN}$) for each lake. Where total-N data were not available, attempts were made to obtain means for total inorganic nitrogen ($\overline{TIN}$) and phosphate ($\overline{SRP}$). The data were analyzed by using standard linear regression and correlation techniques (*17*). The strength of the relationships was evaluated by r^2, the proportion of the total variance associated with the regression. The slopes of all regressions for which a value of r^2 is given in Table I were different from zero at the 95% level of significance. Tests of differences between slopes and intercepts were made by constructing 95% confidence intervals around their mean values (*17*, p 58).

Results

Nutrient/Chlorophyll Relationships in Single Lakes. A survey of the literature provided 16 lakes in which $\overline{TP}$ was found to decrease significantly over a period of several years (Table I). The most extensive data on the response of phytoplankton biomass to nutrient reduction were those from Lake Washington (Seattle). As noted previously by Edmondson (*18*, *30*), the recovery of the lake following sewage diversion in 1963 involved a consistent decrease of both $\overline{TP}$ and $\bar{c}$ (Figure 3).

Nutrient loading to Shagawa Lake (Minnesota) was reduced following tertiary treatment of wastewater entering the lake. Although marked reductions of total P have occurred in the lake, trends in chlorophyll *a* have not been as consistent (*19*, *31*). A significant regression between $\bar{c}$ and $\overline{TP}$ is nonetheless evident (Figure 4A).

A decrease in algal biomass after nutrient reduction was also seen in Brown's Bay of Lake Minnetonka (Minnesota). Wastewater was diverted from the Lower Lake during the summers of 1971–1972, and the concentrations of both phosphorus and chlorophyll *a* in Brown's Bay dropped significantly (*12*, p 104).

In contrast to lakes Washington, Shagawa, and Minnetonka, restoration of Green Lake (Seattle) was accomplished by nutrient dilution. The flushing rate was increased from 0.84 to 2.5 yr^{-1} by pumping in low-nutrient water from the Seattle domestic supply (*20*). A steady and continuous decline in both TP and chlorophyll resulted from these measures (Figure 4B).

Cline's Pond (Oregon) exemplifies another restoration technique—that of nutrient inactivation. Treatment of the pond with sodium aluminate in 1971 led to dramatic reductions in both total P and chlorophyll (Figure 5A). During 1974, Cline's Pond was divided into two approximately equal volumes with a suspended plastic curtain, and the curtain was sealed to the bottom (*21*). Experimental treatment of one-half with zirconium tetrachloride caused an immediate decrease in both total P and chlorophyll (Figure 5A). A similar response was noted in Twin Lake (Ohio) following treatment with aluminum sulfate (*22*, *32*; Figure 5B).

A comparable covariation of phosphorus and chlorophyll is evident from recent lake restoration experiences in Canada. Until 1971, Gravenhurst Bay (Ontario) received effluent from a nearby wastewater treatment plant. Following the initiation of phosphorus removal at the plant, total phosphorus loading

Table I. Data Used To Investigate Chlorophyll–Phosphorus Relations

lake (symbol)	yr	$\overline{TP}$, mg m^{-3}	$\bar{c}$, mg m^{-3}	N:P [a]	period of measurement and method of treatment	regression eq [b]	r^2	ref
Washington (W)	1957	20.7	12.9	16	June–Sept, $\overline{TP}$, $\overline{TN}$;	$\log \bar{c} = 1.20 \log \overline{TP} - 0.55$,	0.94	*30*
	1958	18.4	12.4	27	July–Aug, $\bar{c}$;	$m \pm 0.18$, $b \pm 0.25$,		
	1962	53.5	31.8	10	wastewater diversion	$\bar{c} = 0.62\ \overline{TP} - 1.30$,	0.91	
	1963	60.2	34.8	9		$m = \pm 0.11$, $b = \pm 3.64$		
	1964	51.2	41.0	10				
	1965	48.3	24.8	12				
	1966	37.8	23.9	17				
	1967	26.3	14.9	17				
	1968	21.1	12.7	22				
	1969	13.9	6.8	24				
	1970	16.7	8.8	23				
	1971	13.4	6.1	22				
	1972	13.1	7.2	25				
	1973	14.9	4.7	18				
	1974	10.7	4.3	23				
	1975	10.5	3.9	25				
Minnetonka (Brown's Bay)	1968	53.3	24.3		May–Sept; wastewater diversion			*11, 12*
	1969	44.4	21.2					
	1973	36	13					
	1974	35	19					
Shagawa (S)	1971	52.6	23.8	4*	wastewater P removal	$\bar{c} = 0.48\ \overline{TP} - 0.64$,	0.53	*32*
	1972	49.6	28.8	9*		$m \pm 0.45$, $b \pm 18.6$		
	1973	49.2	22.8	5*				
	1974	34.5	13.1	11*				
	1975	34.8	17.2	24*				
	1976	36.4	21.9	13*				
	1977	36.5	13.0					
	1978	31.7	19.7					
Cline's Pond (C)	1970	290	187	9	April–Sept, sodium aluminate	$\log \bar{c} = 0.96 \log \overline{TP} - 0.04$,	0.92	*21*
	1971	60	54	19		$m \pm 0.85$, b 1.65		
	1971	60	54	19	May–Sept, zirconium chloride	$\bar{c} = 0.60\ \overline{TP} + 15.4$,	0.97	
						$m \pm 0.33$, $b \pm 50.3$		
control:	1974	74.5	75.5	>8*				
exptl:	1974	35.5	21.4	>9*				
Twin (T)								
east (control)	1974	39.3	17.4	32*	June–Sept, aluminum	$\log \bar{c} = 2.23 \log \overline{TP} - 2.41$	0.93	*22, 32*
	1976	34.4	8.8	>5*	sulfate	$m \pm 1.89$, $b \pm 2.95$		
west (exptl)	1974	48.0	20.0	>35*		$\bar{c} = 0.70\ \overline{TP} - 12.9$,	0.90	
	1976	25.4	5.3	>5*		$m \pm 0.70$, $b \pm 26.3$		
Green (Gr)	1959	65	44.5	5	"summer"; lake	$\log \bar{c} = 2.17 \log \overline{TP} - 2.25$,	0.99	*33, 20*
	1965	42	20.5		flushing	$m \pm 0.73$, $b \pm 1.13$		
	1966	26	7.5			$\bar{c} = 0.92\ \overline{TP} - 16.2$,	0.99	
	1967	20	3.3			$m \pm 0.21$. $b \pm 8.9$		
Gravenhurst Bay (G)	1969	42	10.6	11	May–Oct; wastewater	$\log \bar{c} = 1.00 \log \overline{TP} - 0.67$, $m \pm 0.86$, $b \pm 1.32$	0.64	*35*
	1970	39	5.1	13	P removal	$\bar{c} = 0.27\ \overline{TP} - 1.61$,	0.72	
	1971	52	13.8	12		$m \pm 0.19$, $b \pm 6.98$		
	1972	35	8.1	15				
	1973	33	6.9	17				
	1974	25	5.0	22				
	1975	20	5.0	29				
Little Otter	1971	86	36		Aug–Sept, $\overline{TP}$, $\bar{c}$;			*24*
	1972	13.3	2.5	24	May–Oct, $\overline{TN}$; wastewater diversion			

Table I (*Continued*)

lake (symbol)	yr	$\overline{TP}$, mg m^{-3}	$\bar{c}$, mg m^{-3}	N:P [a]	period of measurement and method of treatment	regression eq [b]	r^2	ref
Loch Leven (L)	1968	98	92	16	May–Sept; natural variation	$\log \bar{c} = 0.98 \log \overline{TP} + 0.01$, $m \pm 0.27$, $b \pm 0.52$ $\bar{c} = 0.94\ \overline{TP} + 1.66$ $m \pm 0.26$, $b \pm 24.8$	0.99 0.99	*25,35*
	1969	99	96	16				
	1970	79	77	15				
	1971	67	64	19				
Ekoln (E)	1972	78	25	15	May–Sept; wastewater P removal			*26*
	1973	67	13	22				
	1974	48	12	32				
	1975	28	8.3	72				
Boren (B)	1973	27	7.4	17	May–Sept; wastewater P removal	$\log \bar{c} = 0.38 \log \overline{TP} + 0.32$, $m + 0.14$, $b \pm 0.20$ $\bar{c} = 0.10\ \overline{TP} + 4.43$, $m \pm 0.08$, $b \pm 2.24$	0.98 0.94	*26*
	1974	42	8.6	10				
	1975	25	7.4	16				
	1976	13	5.5	39				
Norrviken (N)	1969	267	130	17	June–Sept; wastewater diversion	$\log \bar{c} = 0.58 \log \overline{TP} + 0.58$, $m \pm 0.44$, $b \pm 1.01$ $\bar{c} = 0.27\ \overline{TP} + 29.7$, $m \pm 0.24$, $b \pm 43.4$	0.51 0.46	*27*
	1970	260	57	8				
	1971	220	86	12				
	1972	207	94	11				
	1973	186	108	13				
	1974	158	67	12				
	1975	98	68	18				
	1976	92	53	15				
	1977	92	46	16				
	1978	99	47	17				
Edssjön	1970	291	85	8	June–Sept; wastewater diversion			*27*
	1971	329	95	8				
	1972	373	129	8				
	1973	327	95	8				
	1974	213	62	9				
	1975	217	138	11				
Oxundasjön	1970	169	55	10	June–Sept; wastewater diversion			*27*
	1971	211	39	6				
	1972	138	50	10				
	1973	108	27	11				
	1974	109	40	14				
	1975	111	42	13				
Ramsjön	1972	596	157	4	June–Sept, May–Sept; wastewater P removal			*36*
	1973	269	95	7				
	1974	387	101	5				
Ryssbysjön	1973	408	40	4	May–Sept; wastewater P removal			*36*
	1974	212	23	5				

[a] Asterisk denotes N:P ratio determined as $\overline{TIN}:\overline{SRP}$; in all other cases N:P was estimated as $\overline{TN}:\overline{TP}$. [b] 95% confidence limits are listed for both slope (m) and intercept (b) for all significant regressions ($P \le 0.05$).

to the bay was reduced by 60% (*23*, *34*), and an immediate decrease in both $\overline{TP}$ and $\bar{c}$ occurred (Figure 6). An even more dramatic improvement in water quality was seen in Little Otter Lake (Ontario), which received industrial wastes containing a polyphosphate descaling agent. Use of the descaler was discontinued in 1971 (*24*), and an immediate reduction of both phosphorus and chlorophyll in the lake was observed (Table I).

Significant decreases in chlorophyll concentration may also occur from year to year in lakes which are not being managed or restored. Although no phosphorus control measures were apparently in effect during the years 1968–1971, a 50% reduction in chlorophyll in Loch Leven (Scotland) resulted from a steady decline in $\overline{TP}$ (Figure 7). These changes in $\overline{TP}$ are attributable in large part to unpredictable year-to-year fluctuations in phosphorus loading, phosphorus retention, and flushing rate (cf. ref *25*).

When data from Scandinavian lakes are examined, similar continuous relationships between chlorophyll and total P are evident. Advanced wastewater treatment for phosphorus was expanded in Sweden in 1968, and a program to evaluate the response of lakes receiving wastewater effluent was begun in

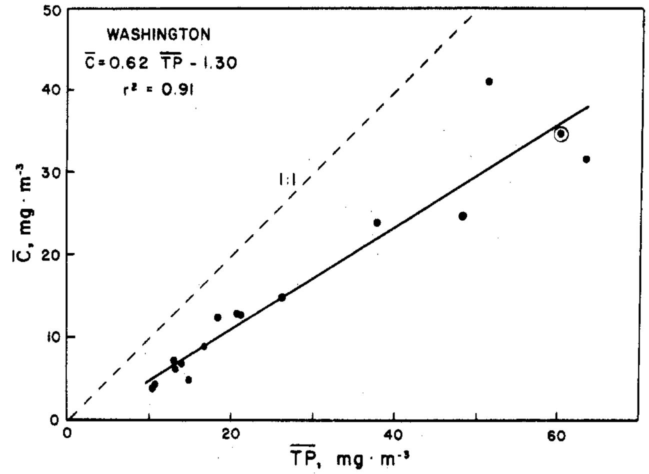

Figure 3. Phosphorus dependence of chlorophyll *a* in Lake Washington, 1957–1975. Circled point denotes year of probable N limitation.

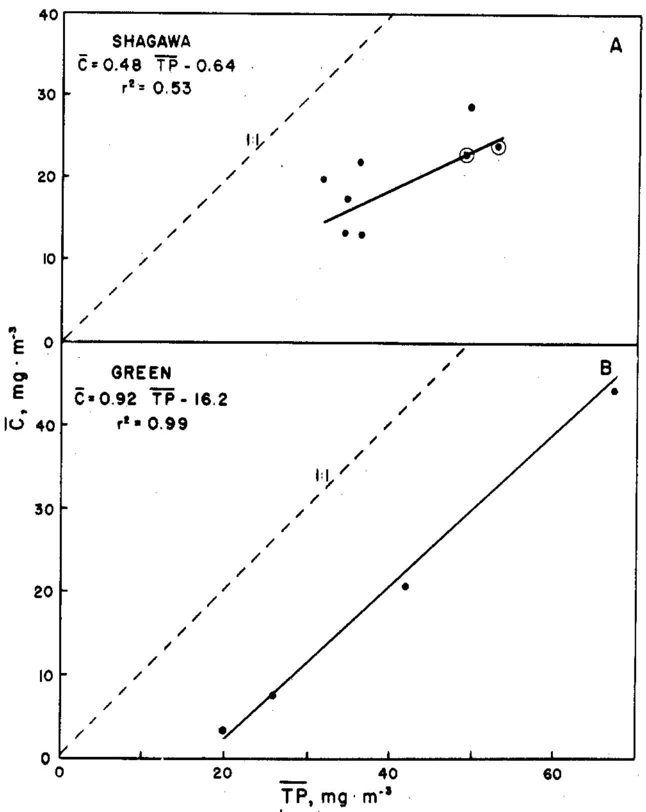

Figure 4. Phosphorus dependence of chlorophyll *a* in (A) Shagawa Lake, 1971–1978, and (B) Green Lake, 1959–1967. Circled points denote years of probable N limitation.

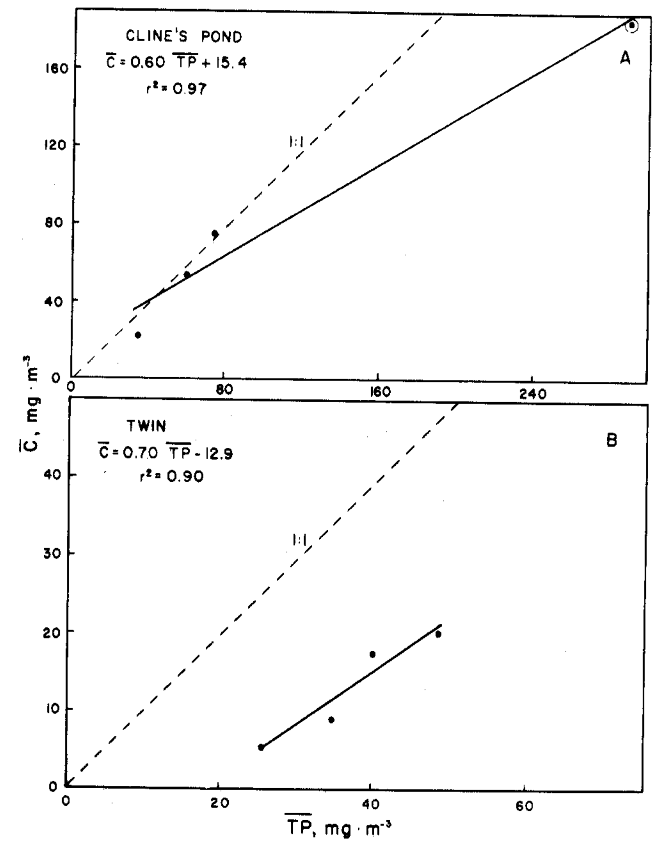

Figure 5. Phosphorus dependence of chlorophyll *a* in (A) Cline's Pond, 1971–1974, and (B) Twin Lake, 1974–1976. Circled point denotes year of probable N limitation.

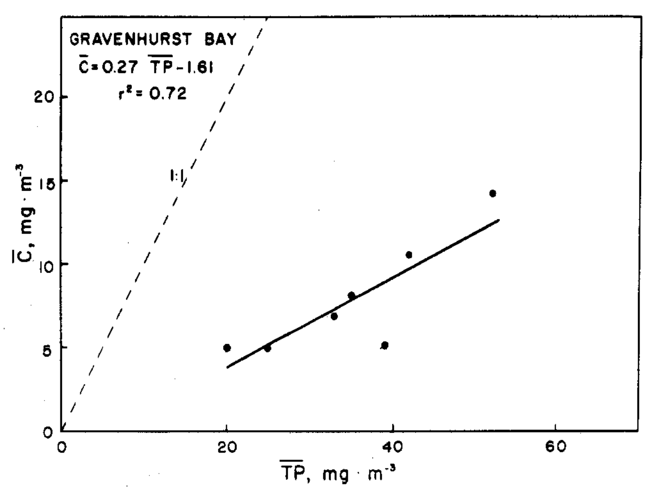

Figure 6. Phosphorus dependence of chlorophyll *a* in Gravenhurst Bay, 1969–1975.

1972 (*26*). The responses of lakes Ekoln and Boren are presented in Figure 8A, and it is clear that an immediate improvement in water quality resulted from phosphorus removal. Similarly, diversion of wastewater from Lake Norrviken (*27*) also led to consistent declines in both $\overline{TP}$ and $\bar{c}$ (Figure 8B). The only exception to this pattern occurred in 1970, when algal biomass seemed to be N limited ($\overline{TN}:\overline{TP}$ = 8) and may have been regulated by intense zooplankton grazing as well (*28*).

The effects of N limitation may also be seen in the case of Oxundasjön, Sweden (Figure 9A), where, despite large reductions in total P, no statistically significant reduction was noted in $\bar{c}$ (*27*), in contrast to the preceding examples. In this case, much of the scatter appears to be caused by changes in the N:P ratio over time (Table I). However, if the data from 1971 ($\overline{TN}:\overline{TP}$ = 6) are excluded, a marked decreasing trend in both $\bar{c}$ and $\overline{TP}$ is evident. Figure 9B shows further that three Swedish lakes that are definitely N limited also responded to P reduction. For example, in the case of Edssjön, if the data for 1975 (the only P-limited year) are omitted, there is a very strong relationship of $\bar{c}$ to $\overline{TP}$ (r = 0.91). Ramsjön and Ryssbysjön (*27*) also appear to show good regressions of $\bar{c}$ on $\overline{TP}$, although the number of data points is too low to test for significance.

Discussion

When data from the 16 lakes are taken individually (Table I and Figures 3–9), it is clear that reductions in the mean

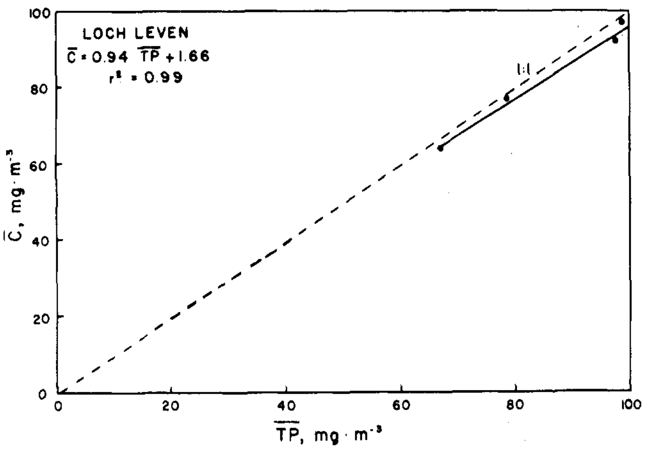

Figure 7. Phosphorous dependence of chlorophyll *a* in Loch Leven, 1968–1971.

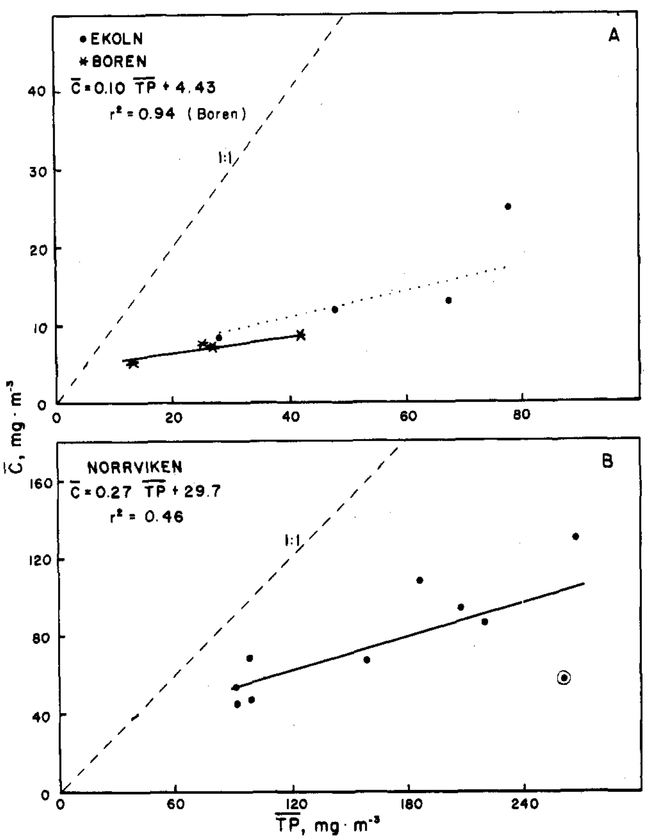

Figure 8. Phosphorus dependence of chlorophyll *a* in (A) Lake Ekoln, 1967–1975, and Lake Boren, 1973–1976, and (B) Lake Norrviken, 1969–1978. Circled points denote years of probable N limitation.

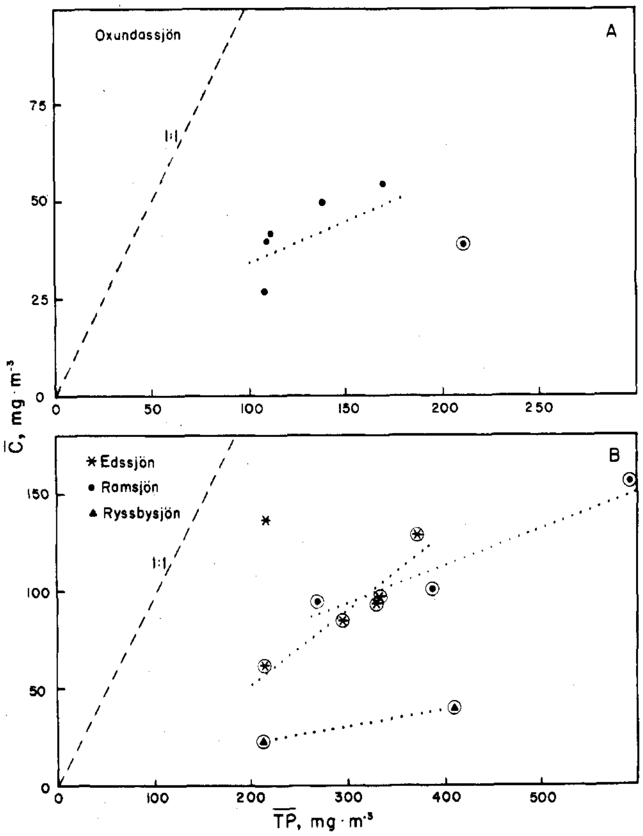

Figure 9. Phosphorus dependence of chlorophyll *a* in (A) Oxundasjön, 1970–1975, and (B) Edssjön, 1970–1975, Ramsjön, 1972–1974, and Ryssbysjön, 1973–1974. Circled points denote years of probable N limitation. Dotted lines denote trend lines (linear regressions not statistically significant).

chlorophyll concentration typically accompanied reductions in mean total phosphorus. However, in only one case (Loch Leven) would the threshold model have adequately predicted the response. Therefore we repeat our belief that this is an unsatisfactory approach to predicting responses of algal biomass in individual lakes to phosphorus management strategies. We have shown that P-limited lakes (presumably representing a large fraction of NES lakes) respond immediately to P reduction, and our data suggest that even N-limited lakes respond to P reduction. Therefore we believe that the most important argument against the threshold model is the use of inappropriate data in its construction; e.g., too large a proportion of the data were from spring and fall. We urge that models intended for predictive purposes be based on data collected during the summer months, when algal biomass is most closely related to nutrient concentration and when public interest in such factors as algal abundance and type, and transparency, is at its peak.

We also wish to stress that global models such as the Sakamoto (*15*), Dillon–Rigler (*3*), Jones–Bachmann (*4*) type, although based on summer data, may not accurately predict what will happen in individual lakes. This point has been made before (e.g., ref *28*) on the basis of the great variability in chlorophyll at any given concentration of phosphorus. Our (P-limited) data, plotted on a log/log basis as in Figure 10A, show a relationship similar to that of Sakamoto, Dillon–Rigler, and Jones–Bachmann and also show similar variability. However, it is clear from Figures 3–9 and from Figure 10B, where the P-limited regressions are shown separately for each lake, that one reason for the variability is that many lakes seem to have responded to P reduction in a unique manner, with significant differences between the slopes and the intercepts of their regressions (Table II). This is not surprising when one considers the multitude of factors which affect chlorophyll production in lakes. For example, chlorophyll yield at a given total phosphorus concentration is influenced strongly by the mixed depth (*37, 38*), by herbivore abundance (*39*), and by turbidity (*40*). The TN : TP ratio (*15*) can also influence algal biomass, and, in fact, a recent reanalysis of the NES data set (*40*) shows clearly that "... the higher N : P ratio lakes generally have a higher yield of chlorophyll per unit phosphorus, and many of the lakes with high phosphorus values and low chlorophyll concentrations also had low N : P ratios."

These are sound limnological reasons why every lake should not respond identically to nutrient reduction. Thus, a key assumption in the practical application of these and other (*2*,

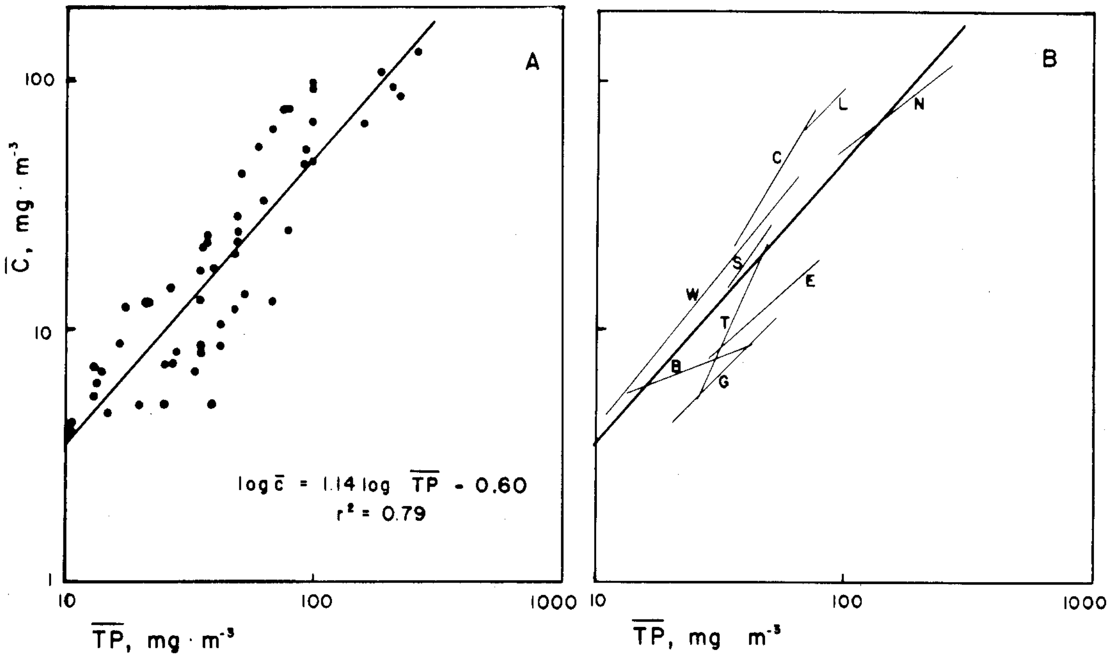

Figure 10. (A) Phosphorus dependence of chlorophyll *a* in nine phosphorus-limited north temperate lakes, as judged by N : P criteria outlined in the text. Each point represents one year's mean data. (B) Phosphorus dependence of chlorophyll *a* in nine phosphorus-limited north temperate lakes. Each line represents the regression for an individual lake (for key to symbols see Table I).

29) global models—that individual lakes will respond in an identical fashion—is unwarranted. This is not to say that global models should be abandoned, however. These models have already proved valuable worldwide in the prediction of the effects of lake restoration, and they will continue to be useful until newer, more complex models have been tested.

Conclusions

A recent (*9*) study of phosphorus control options examined the effects of eight phosphorus management strategies on the mean annual concentrations of total P in the NES lakes. However, despite the fact that *all* control options produced lower values of total P in these lakes, the model relating chlorophyll to the changes in TP could not show measurable benefits to the lakes in terms of apparent water quality. We feel that this failure to show concrete benefits resulted in part from the use of an inappropriate chlorophyll–phosphorus model which could not accurately predict the declines in algal biomass that accompany nutrient reduction. It is hoped that this fact will be recognized before any conclusions are drawn concerning the relative values of different phosphorus control strategies. From our own point of view, phosphorus reduction remains the keystone of lake restoration.

The global models which are currently used to assess the impact of phosphorus removal have been developed from a very heterogeneous population of lakes, and this variability is reflected in part as the noise typically associated with chlorophyll–phosphorus regressions. We have tried to reduce some of this noise by using time-series data from individual lakes. In doing so, we have observed that many of the 16 lakes tended to behave uniquely in their response to nutrient reduction. We urge that practical users of global models take note of this, and we suggest that they not expect all lakes to respond exactly as the models predict. Current chlorophyll–phosphorus models are simply one stage in the evolution of our lake management tools. We expect to see these models modified to consider the effects of other factors—and thus to see that our prediction errors will decrease in the future. For example, a preliminary chlorophyll–phosphorus model taking TN : TP ratios into account has been developed (*41*, *42*), and modifications to include mixed depth, zooplankton grazing, and turbidity will undoubtedly follow as other investigators attack these problems as well. We hope that limnologists will continue to search for other factors which influence the response of lakes to nutrients and that the result will be a logical, stepwise reduction in the residual error in our models.

Table II. Summary Matrix Showing Statistically Significant Differences between Chlorophyll–Total Phosphorus Regressions for Nine Lakes

A. Lower left: logarithmic regressions. Upper right: arithmetic regressions. Letter designates significant ($P \leq 0.05$) difference between two slopes.

Lake	W	C	T	Gr	G	L	B	N	S
W		—	—	—	a	—	a	—	—
C	—		—	—	—	—	a	—	—
T	—	—		—	—	—	—	—	—
Gr	a	—	—		a	—	a	a	—
G	—	—	—	—		a	a	—	—
L	—	—	—	a	—		a	a	—
B	a	—	—	a	—	a		—	—
N	—	—	—	a	—	—	—		—
S	—	—	—	—	—	—	—	—	

B. Same format as above, except letter designates significant differences between two intercepts.

Lake	W	C	T	Gr	G	L	B	N	S
W		—	—	a	—	—	—	—	—
C	—		—	—	—	—	—	—	—
T	—	—		—	—	—	—	—	—
Gr	a	—	—		—	—	a	—	—
G	—	—	—	—		—	—	—	—
L	—	—	a	a	—		—	—	—
B	a	—	a	a	—	—		—	—
N	—	—	a	a	—	—	—		—
S	—	—	—	—	—	—	—	—	

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Particle Collection in Cyclones at High Temperature and High Pressure

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Introduction

In recent years there has been a renewed interest in the performance of cyclone dust collectors at high temperature and high pressure. This interest is related to the need for reliable sampling and particulate control equipment for advanced coal conversion and combustion processes. Applications at temperatures up to 1200 °C and pressures up to 5000 kPa are being considered.

Only very limited experimental data have been reported, and these are insufficient for evaluating the effects of temperature and pressure on the mechanisms responsible for particle deposition in cyclones. An understanding of these fundamental mechanisms is essential to evaluate and develop design models for high-temperature and high-pressure cyclones.