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Ecological Research Series

MATHEMATICAL MODELING OF PHYTOPLANKTON IN LAKE ONTARIO

Part 2

Simulations Using Lake 1 Model

Environmental Research Laboratory
Office of Research and Development
U.S. Environmental Protection Agency
Duluth, Minnesota 55804

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Part 2

Simulations Using Lake 1 Model

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ABSTRACT

The results of a series of simulations of the response of the open lake region of Lake Ontario to various levels of nutrient input are described in this report. The simulations use a simplified dynamic model of phytoplankton - nutrient interactions in a vertically segmented structure. The lake is assumed to be well-mixed in the horizontal direction.

The problem of long term simulations (10-20 years) that draw on short term observation and verification periods (5 years) is discussed and it is indicated that the overall loss rates of nutrient are of particular importance. Under a hypothesized, but reasonable, set of model parameters, the simulations indicate that the present observed open lake phytoplankton biomass of Lake Ontario does not appear to be in equilibrium with the present input nutrient load. Therefore, if the present load is continued,it is estimated that spring peak phytoplankton chlorophyll in the epilimnion will continue to increase to a new level about 45% higher than present levels. The interaction of nitrogen and phosphorus is also described by the simulations and the results indicate a tendency for nitrogen limitation to be an increasing dominant factor in controlling the spring bloom.

A " pastoral" simulation using load estimates, indicative of conditions prior to man's intensive activity provides an approximation of an early state of the lake. This "hindcast" indicates that spring phytoplankton levels were some 40% less than present levels and average annual epilimnion biomass under equilibrium with present loads is about twice that under pastoral conditions.

A series of analyses is also conducted comparing simulations from the dynamic model to estimates made from simplified plots of loading versus lake geometry. The results from the dynamic model indicate that a reduction in external nutrient load does not result in an accompanying decrease in

phytoplankton biomass, due to the hypothesized non-equilibrium condition of Lake Ontario. The dynamic model results are therefore in contrast to the results one would obtain from using "admissable" loading concept which indicates an improvement in lake trophic status.

Analysis of lake response to the U.S.-Canada Water Quality Agreement (WQA) loads using the hypothesized parameters indicates about a 6% reduction in peak phytoplankton at equilibrium.

The implications of the results appear to be of some importance since the analyses indicate that it may be difficult to achieve measurable reductions below present levels of phytoplankton biomass in the open lake. From a decision and policy making viewpoint then, the simulations tend to indicate that maximum point source nutrient control for Lake Ontario will, at best, be a "holding"action rather than a significant improvement in the status of the open lake.

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SECTION I

CONCLUSIONS

For large lakes such as Lake Ontario, it is difficult to estimate the effects of various nutrient input levels due to the short observation period of the lake relative to its size. Verification analyses would of necessity have to be conducted over a number of years up to at least a decade in order to estimate overall loss rates of nutrients from the lake system. The long term simulations are particularly sensitive to these decay rates. Nevertheless, it is concluded that a reasonable set of kinetic parameters can be hypothesized and simulations can be carried out under different input nutrient conditions.

Estimates of present nutrient loading to Lake Ontario can vary widely (± 15%) partly due to difficulty in accurately estimating Niagara River input. For phosphorus, the variation in the Niagara River loading above is equivalent to a population of about 1.6 million or about 25% of the present population. It will be difficult then to estimate future changes in input load because of this inherent variability.

Simulations using the simplified Lake 1 Model were carried out over a range of nutrient conditions. It should be stressed that the conclusions to follow represent open lake responses under hypothesized (albeit reasonable) system kinetics. Near shore responses and resulting conclusions may be considerably different than indicated below.

Four principal load conditions were used as mile posts: 1) Continuation of present load, phosphorus=75,000 lbs P/day 2) pastoral loads, phosphorus=20,500 lbs. P/day, 3)Vollenweider reduction, phosphorus=46,900 lbs P/day, and 4) U.S.-Canada Water Quality Agreement load, phosphorus=54,800 lbs. P/day. Nitrogen loading was used at 883,000 lbs N/day for all simulations except the pastoral conditions where the loading used was 406,000 lbs N/day.

Under a continuation of present loads and a "reasonable" set of system parameters, it is estimated that spring peak phytoplankton chlorophyll in the epilimnion will continue to increase to a new level about 45% higher than present peaks. About 8-10 years would elapse to reach this new equilibrium. Average annual biomass in the epilimnion is computed to increase by about 20%. The simulation also indicates an increasing tendency for the spring bloom to be controlled more by nitrogen than phosphorus.

Simulations made under "pastoral" conditions representing estimated background loading prior to man's 20th Century activities indicates spring peak chlorophyll values about 7 μg/l or some 40% less than present levels. This simulation therefore provides a first approximation to the state of Lake Ontario prior to the turn of the century. The results indicate average annual phytoplankton chlorophyll in the epilimnion to be about 2.6 μg/l as compared to 5.8 μg/l annual average in equilibrium with the present load. It is estimated therefore that the average annual phytoplankton in the epilimnion at equilibrium with present loads is about twice the level that existed under some previously unstressed environment. The load however has increased by about 3.7 times over the same period.

A simulation conducted using the "admissable" loading from empirical plots of Vollenweider (a 40% phosphorus reduction) indicate an increase in spring peak biomass to about 15 μg/l chlorophyll. The results indicate an exception to the general axiom that a reduction in external loads will result in an improvement in water quality. This is in contrast to the implication inherent in the use of loading plots which indicate that a reduction in external load will of necessity improve lake quality. If the hypothesis that Lake Ontario is not yet in equilibrium with the present loads is correct, then the "admissable" loading does not result in an improvement in water quality. One concludes therefore from this one comparison between two models,that empirical loading plots may not be appropriate for large lakes such as the Great Lakes.

The Water Quality Agreement (WQA) load simulation (a 27% phosphorus reduction) indicated that the open lake phytoplankton of Lake Ontario may continue to increase for a period of about 15 years or until the late 1980's. The WQA phosphorus load is calculated to result in only about a 6% reduction in peak phytoplankton at equilibrium. Given the variability in load estimates and observed fluctuations in open lake biomass such a change would be difficult to detect. This is not meant to imply that the WQA program is not a good one, but simply that under the stated hypotheses the computations indicate that the hopes for an expected response of Lake Ontario may not be as high as anticipated under the Agreement.

It is estimated that to maintain present open-lake phytoplankton peak biomass, a total phosphorus loading of about 35,000 lbs P/day would be necessary. This loading represents conditions of approximately the 1940's and also represents about a 73% reduction in present loading above the pastoral background load. Since only about 60% of the total load discharged is from point sources, the results indicate that it may be difficult to achieve measurable reductions below present levels of phytoplankton biomass.

Finally, it should be noted strongly again that the conclusions from these simulations are indicative of open lake conditions and do not reflect near-shore responses which may be quite different and further that the simulations are based on an hypothesized, but apparently reasonable set of kinetics. Also, research into predicting the future dynamic behavior of phytoplankton in large lakes is still very much in its infancy which would indicate that additional research may lead to varying conclusions. Nevertheless, policies and decisions will still have to be made even though future research may suggest adjustments and corrections.

SECTION II

RECOMMENDATIONS

Because of the importance of near shore versus open lake problems, it is recommended that a detailed verification and analysis of the Lake 3 model be conducted. Some simulations should be carried out with the Lake 3 model to delineate the time for the near shore to reach a new dynamic equilibrium.

Since the simulations using the Lake 1 model were particularly sensitive to the overall loss rates of the nutrients, it is recommended that: 1) a modeling framework of phosphorus chemistry be developed and verified to attempt to define these critical parameters 2) investigations begin into the development and verification of a sediment model to utilize the only available historical trace of the state of Lake Ontario.

Finally, and most importantly, it is strongly recommended that additional analyses be conducted of Lake 1 model responses. Continuing up-dating and verification together with simulations under a variety of conditions, should be carried out preferably by one of the operating agencies in the Great Lakes.

SECTION III
INTRODUCTION

PURPOSE OF RESEARCH

Attention is directed in this report to the utilization of a simplified model of phytoplankton dynamics of Lake Ontario (the Lake 1 model) to estimate the lake-wide response to various levels of nutrient loading. The Lake 1 model is therefore viewed as an initial framework for estimating whole lake phytoplankton response and to provide some input into the ongoing decision making process on Lake Ontario.

A range of external nitrogen and phosphorus loading is examined and the sensitivity of the results to various model parameters is examined. All of the work is aimed at providing estimates of phytoplankton biomass to a first approximation. It should be stressed that the results presented herein are to serve only as general indicators of the direction to be expected under remedial nutrient control programs.

SCOPE OF RESEARCH

This report follows Part 1, Model Development and Verification[1] and builds on that work. As such, the details of the model are not presented and only the general structure and a summary of the results of the verification analysis are given. The geographical scope is lake-wide and the simulations are,therefore, for the whole lake only. Near shore problems are not considered herein. The measure of eutrophication is taken as the phytoplankton chlorophyll a. The emphasis is on the response of the open lake total biomass to a range of nutrient loading and the sensitivity of the response to varying estimates of model parameters and coefficients.

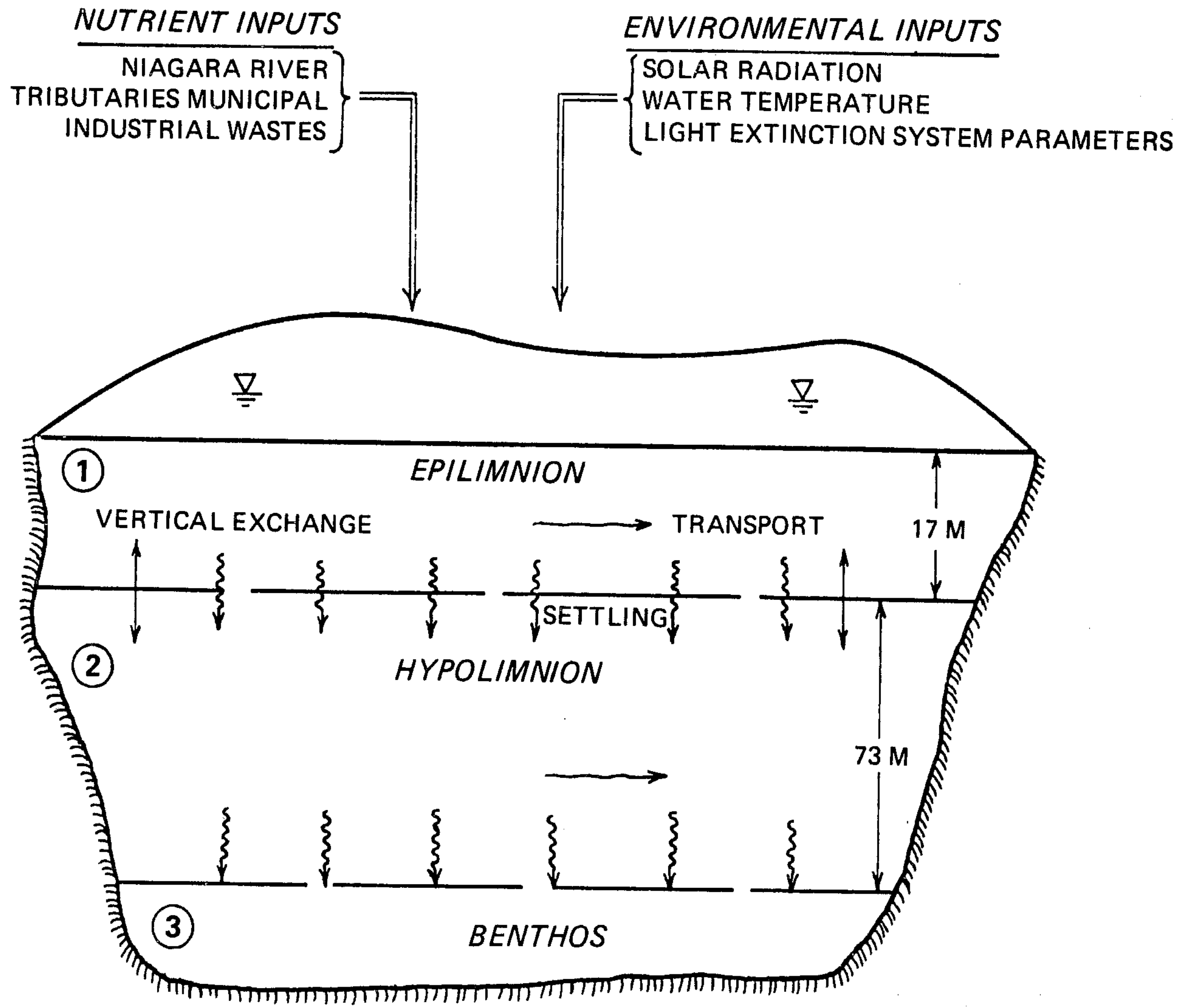

Figure 1 Major physical features included in Lake 1 model

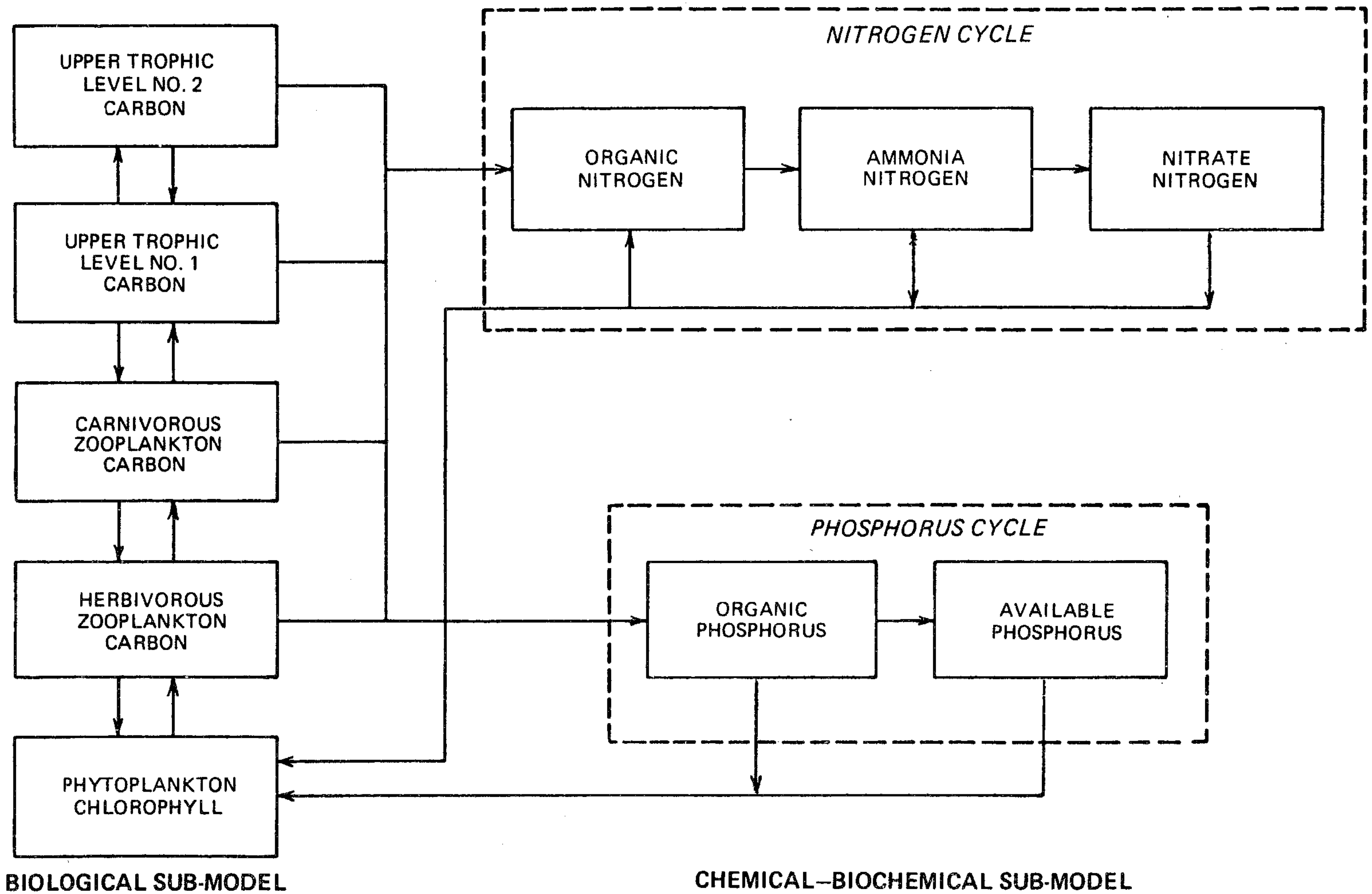

Figure 2 System diagram - Lake 1 model

LAKE 1 MODEL REVIEW

Fig. 1 shows the geometry of the Lake 1 model. The principal features included in the model are:

a) a two layer system with a sediment layer, the mixing and stratification being accomplished by vertical exchange
b) phytoplankton settling
c) external environmental inputs of nutrients
d) external environmental inputs of solar radiation, water temperature and other system parameters.

The system diagram showing the interaction of the key variables is given in Fig. 2. Ten dependent variables are included and incorporate the major features of the interactions of phytoplankton, zooplankton and nutrients. Table 1 gives the basic physical data used and complete details are given in (1).

Extensive analyses and summary data from 1967-1970 formed the basis for verification of the model. The results of the verification indicated that the model provides a reasonable comparison to observed lake-wide average values of chlorophyll, zooplankton carbon, and various forms of nitrogen and phosphorus. The analyses indicate that the spring growth phase and peak phytoplankton biomass are primarily controlled by increasing light and temperature and phosphorus limitation. The mid-summer minimum in phytoplankton is estimated to be due primarily to zooplankton grazing and nitrogen limitation. The broad fall peak in phytoplankton is a complex interaction of nutrient regeneration (up to five times the external nutrient inputs), subsequent nutrient limitation and then the fall overturn. Both nitrogen and phosphorus are important nutrients in this dynamic succession.

TABLE 1

BASIC PHYSICAL DATA OF THE LAKE 1 MODEL

Segment No.	Segment Interface	Volume ($m^3 x 10^6$)	%	Depth (Meters)	Surface Area (meters^2)	cfs	Flow m^3/sec	%
1		297,000	19	17		43,500	1232	19
	1-2				$1.64.10^{10}$			
2		1,373,000	81	73.3		188,500	5323	81
	2-3				$0.89.10^{10}$			
3 (sediment)		-		0.15*				

Note: Vertical dispersion coefficient between segments No. 1 and 2 varied from 0→ 6.7 cm^2/sec (0→0.78 m^2/day)

* Segment #3 depth is arbitrary

Although the model parameters used in the verifications are all considered reasonable and within reported literaturn ranges, no claim is made as to the uniqueness of the particular parameter set that was finally derived. Nevertheless, the conclusion of the model development and verification stage of the work indicated that a sufficient base had been established to use the model for preliminary simulations of various levels of nutrient reduction. The simulations using the Lake 1 model are therefore the primary topic of this report.

THE SIMULATION PROBLEM IN LARGE LAKES

The estimation of future levels of water quality on large lake systems is complicated by the long detention time of the lake and the usual relatively short period of observed data on the state of the lake. Changes in lake water quality are therefore difficult to perceive on a year to year basis. For example, for Lake Ontario, with an eight year hydraulic detention time, the relevant time scale of interest is on the order of tens of years; i.e. it may take 10-20 years for the lake to respond to changes in external inputs. Fig. 3 shows this effect. The observation time for Lake Ontario of about 5 years is short compared to the response time. A difficult problem in prediction is therefore presented: namely, the estimation of long term system response based on a short observational period. The problem is somewhat analogous to attempting to estimate the frequency of occurrence of a one in ten year drought flow based on one or two years of record.

In the verification analyses reported on in Part I of this work, the time variable responses computed for a period of one year were responsive primarily to the specified initial conditions rather than the

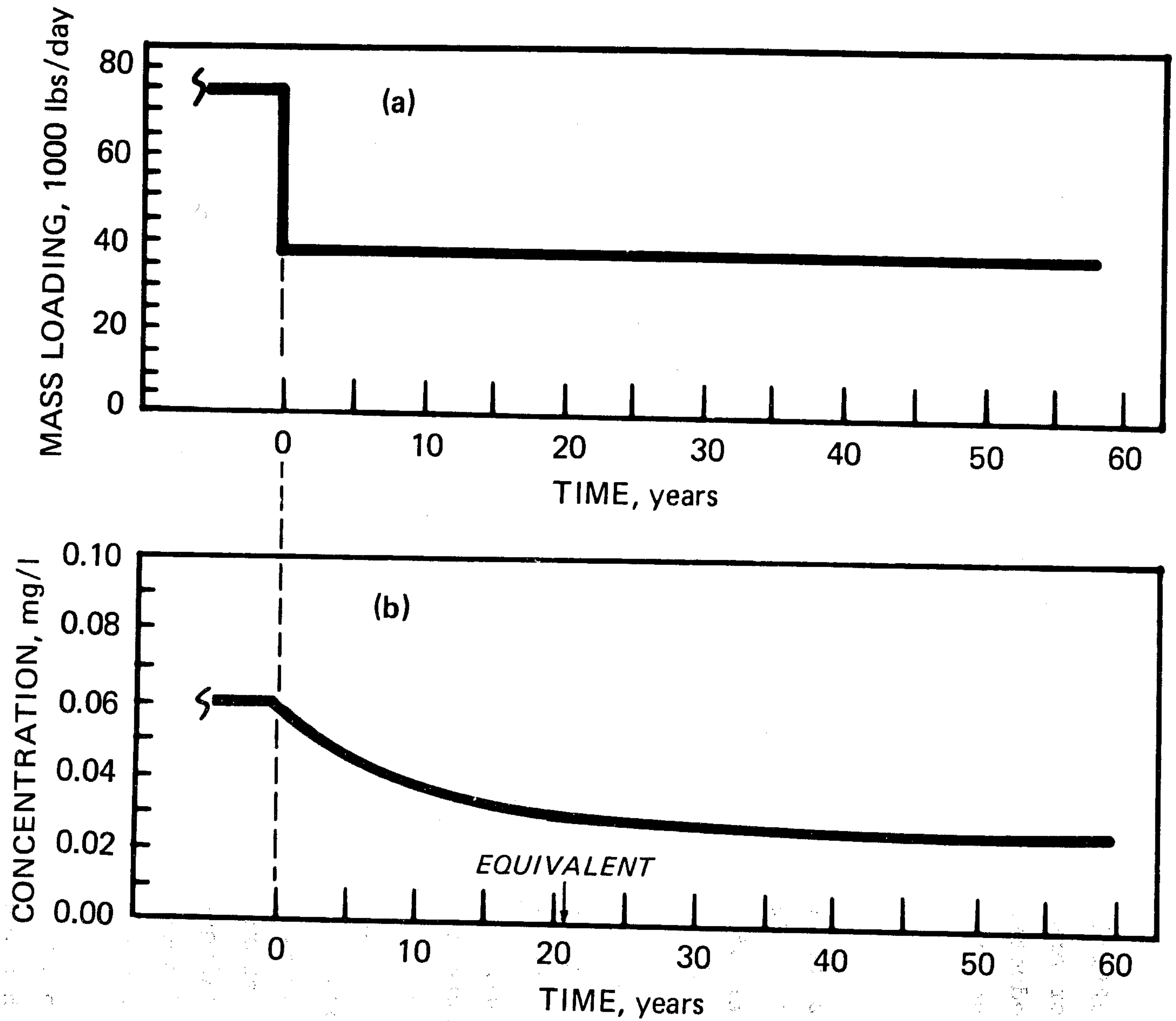

Figure 3. Time to equilibrium, conservative variable a) reduction in load, b) response in concentration

external inputs. This, of course, can be seen from a nutrient balance equation for a well-mixed lake. Thus,

$$c = \frac{W}{Q + KV} (1 - e^{-(1/t_o + K)t}) + c_o e^{-(1/t_o + K)t} \qquad (1)$$

where c=the whole lake nutrient concentration (mg/l), W is the external source of nutrients (kg/day), V is the lake volume, Q is the flow (m^3/sec) through the lake, t_o is the hydraulic detention time (=V/Q), K(l/day) is the overall decay of the nutrient due to settling or chemical reactions, and t is time.

If one is now approaching a year of data or even several years of data, two estimates must be made. First, the external load, W must be estimated and then the overall loss rate of the nutrient given by K must be determined. The external nutrient load is estimated from a variety of data sources (see next section) but usually for a given year or group of years close to the sampling for the nutrients. The overall decay rate is either estimated as part of the settling of the phytoplankton as in the dynamic phytoplankton model or is estimated from the nutrient data itself. But, the latter course of action assumes the lake to be in equilibrium with the present load, an assumption that cannot be checked until the lake has actually been observed for a period of at least one-two detention times (8-16 years for Lake Ontario). The dilemma is made clear by an example from Lake Ontario.

"Present" total phosphorus load to the Lake is about 34,000 kgP/day (75,000lbs P/day). If total phosphorus were completely conserved (K=0), the total within-lake phosphorus concentration, p_t, at equilibrium is simply from Eq.(1), (for a flow of 6570 m^3/sec (232,000 ft^3/sec)),

$$p_t = \frac{W}{Q} \qquad (2)$$

or p_t = .06 mg/l. Now, average observed total phosphorus concentrations in Lake Ontario during the period 1967-70 and 1972-73 is about 0.02 mg P/l. For this initial condition, the concentration at the end of the first year would be about 0.025 mg/l; close to the observed value. Fig. 4 indicates these results and shows the dominant effect of the initial condition during the first year. Three choices are open now to the analyst: a) assume the present load is not in equilibrium with the present concentrations and that the system is conservative, b) assume the lake is not in equilibrium with the present concentrations but that there is some loss in the system, or c) assume that the lake is in equilibrium with the present load and estimate the decay rate from the data.

For a lake with a large volume such as Lake Ontario, ($1.67 \cdot 10^{12} m^3$), a value of K = .001/day, gives a value of p_t at equilibrium of about 0.015 mg/l, again close to the present observed concentration. The question then is: "Is a value of K=.001/day reasonable?" Unfortunately, the verification analysis does not necessarily provide the needed information. Fig. 5 shows that for Lake Ontario the difference at the end of one year of analysis with or without a value of K is too slight to determine a reasonable estimate.

Only two courses of action appear open for Lake Ontario. If long term data on some aspect of the lake biomass behavior were available, a long term simulation would provide information on the decay rate. Only the sediments seem to contain some hope for constructing a long term record of the state of the lake. The difficulty at the present time of course, is that it is not yet possible to deal in a reasonable way with the available sediment data on pore water chemistry or on the chemistry of the solid phase. The second course of action therefore appears to be the most fruitful. Reasonable hypotheses on the phosphorus and nitrogen components (e.g. phytoplankton and detrital settling) can be formulated and tested on the dynamic model. Simulations based on those results can then be prepared but with full recognition that the long term behavior has only been grossly approximated.

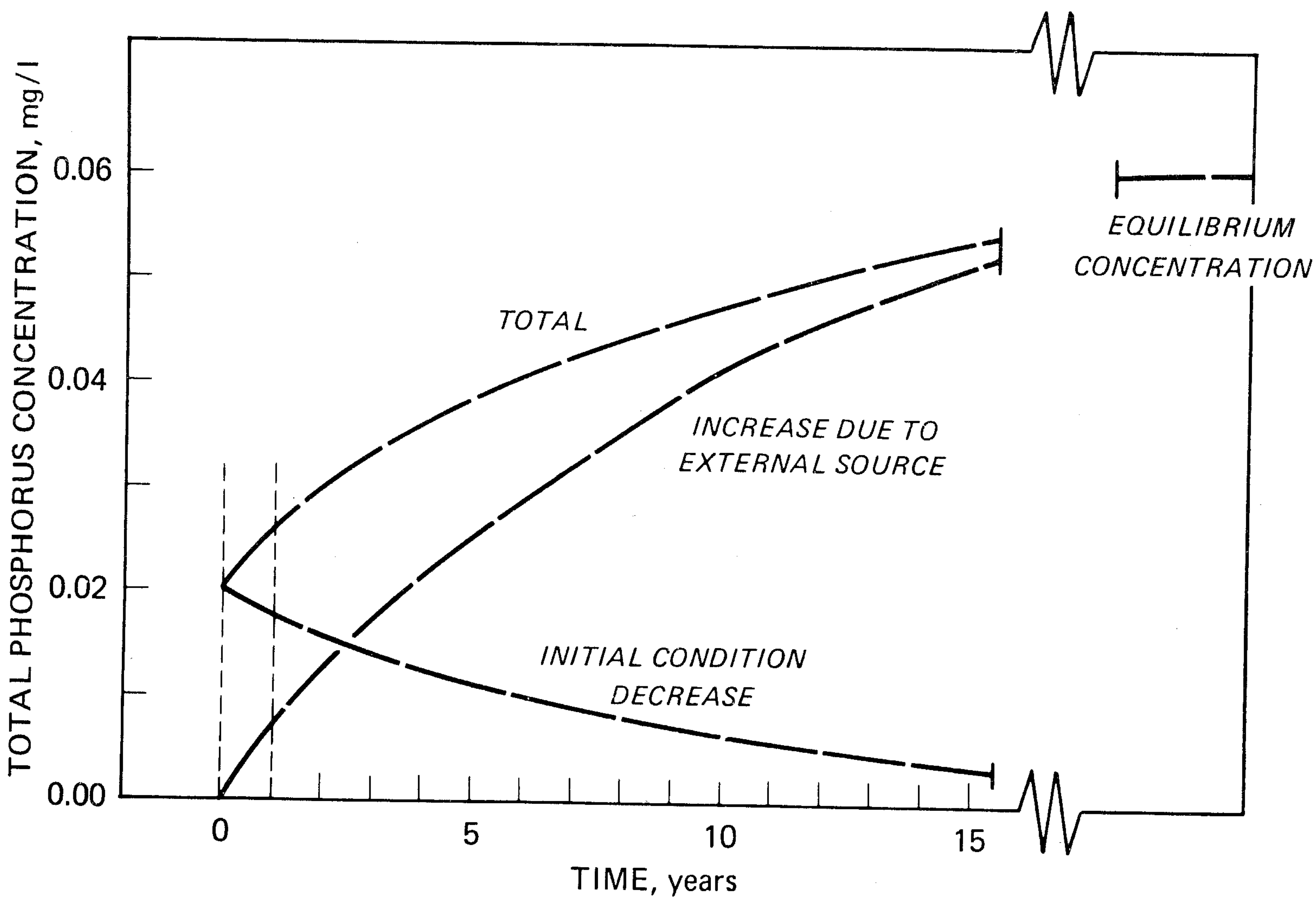

Figure 4. Illustration of dominant effect of initial condition during a one-year analysis

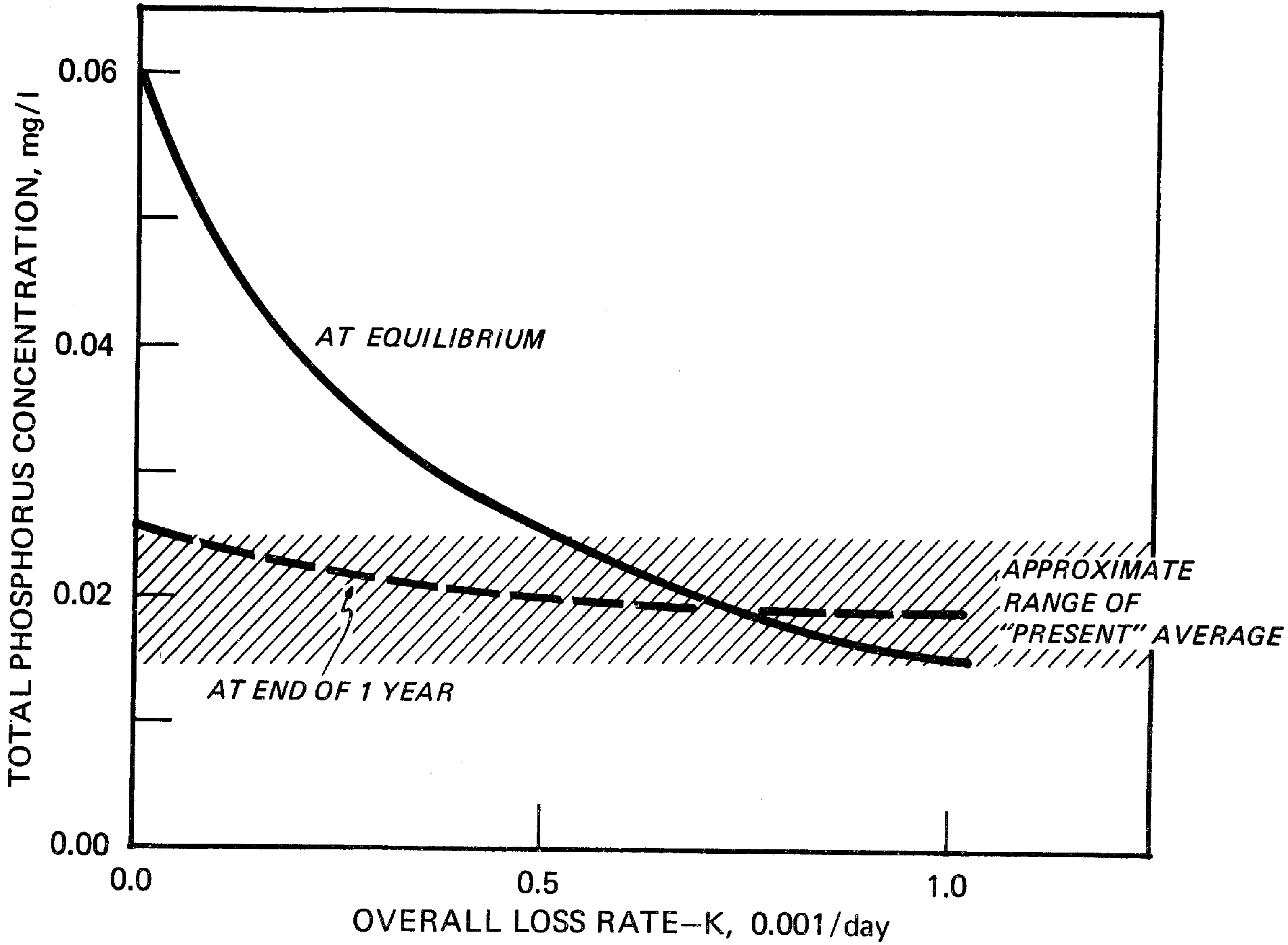

Figure 5 The insensitivity of a one-year calculation to the overall loss rate

SECTION IV

EXTERNAL NUTRIENT INPUTS FOR SIMULATIONS

"PRESENT" INPUTS

The principal sources of nutrients to the whole lake are: 1) the Niagara River, including input from Lake Erie and waste discharges on the Niagara River itself, 2) other tributary inputs in the Lake Ontario basin, 3) direct discharges of municipal and industrial wastes, 4) local drainage to the Lake and, 5) atmospheric inputs. The major contributions are from the first three categories although atmospheric inputs may prove to also be significant.

The estimated 1966-67 loading to the Lake is shown in Table 2 as estimated by the IJC.

TABLE 2
ESTIMATED 1966-67
NUTRIENT LOADINGS

SOURCE	NITROGEN			PHOSPHORUS		
	lbs/Day	Metric tons /Day	%	lbs/Day	Metric tons /Day	%
Niagara	522,200	236.9	59	42,200	19.1	56
Tributaries	191,000	86.6	22	15,600	7.1	21
Municipal	72,600	32.9	8	16,200	7.3	22
Industrial	97,300	44.1	11	1,000	.5	1
Total	883,100	400,5		75,000	34.0	

As shown, the major input is the Niagara River which includes discharges on the Niagara River itself as well as the output from Lake Erie. Direct municipal and industrial discharges to the Lake account for about 20%, the remainder of the load enters from tributaries to the Lake.

Additional sampling was conducted during IFYGL by both the U.S. and Canada. Casey and Salbach[3] have summarized the IFYGL inputs and Table 3 compares the two estimates.

TABLE 3

ESTIMATED NUTRIENT INPUTS TO LAKE ONTARIO[(2,3)]
(1966-67 and 1972)
1,000 lbs/day[(a)]

SOURCE	Total Phosphorus 1966-67	1972	Total Nitrogen 1966-67	1972
Niagara	42.2	45.9	522.2	482.4
Tributaries	15.6	20.3	191.0	224.9
Municipal	16.2	20.9	72.6	93.3
Industrial	1.0	0.4	97.3	20.3
Sub-Total	75.0	87.5	883.1	820.9
Atmospheric[(b)]	9.9	9.9	137.1	137.1
Groundwater[(b)]	0.2	0.2	2.0	2.0
Total	85.1	97.6	1022.2	960.0

a) 1,000 lbs/day = 0.454 metric tons/day

b) Atmospheric and groundwater inputs for 1966-67 assumed equal to 1972 input.

For total phosphorus, the load estimates (excluding precipitation and groundwater) are reasonably close and the difference in Niagara River and tributary loads can be partially accounted for by the higher flows during IFYGL. The total nitrogen load estimates vary more widely and not in the direction of increasing flow. Further, there is a substantial change in the estimate of the industrial nitrogen contribution between 1966-67 and 1972. The relative magnitude of the atmospheric input can also be noted.

One of the difficulties in the simulation analyses was the specification of expected future loading or the loads that were discharged to the Lake in prior years. The difficulty results primarily from varying load estimates at different times such as indicated in Table 3 and from different sampling procedures. The importance of the Niagara River input indicates a need to estimate the range in load to be expected from the Niagara River even if point source load reduction were accomplished.

VARIABILITY OF THE INPUTS

As shown in Tables 2 and 3, one of the primary inputs of nutrients to the Lake is from the Niagara River which has an average annual flow of about 202,000 cfs with a mean annual standard deviation of about 19,000 cfs. Therefore, about 70% of the time, the average annual flow is between about 180,000 to 220,000 cfs, or a range of 40,000 cfs. Fig. 6 shows the frequency distribution of annual flows for the Niagara River for the period 1860-1972. Flow during the first 9 months of IFYGL averaged about 228,000 cfs for the months of April-December, 1972. Niagara River flow during the 1966-67 nutrient loads estimated by the International Joint Commission (IJC)[2] was about 190,000 cfs. The two periods for which nutrient load estimates were made differed in Niagara flow by about 40,000 cfs.

The nutrient concentrations of the Niagara River vary over a fairly wide range. For example, during the

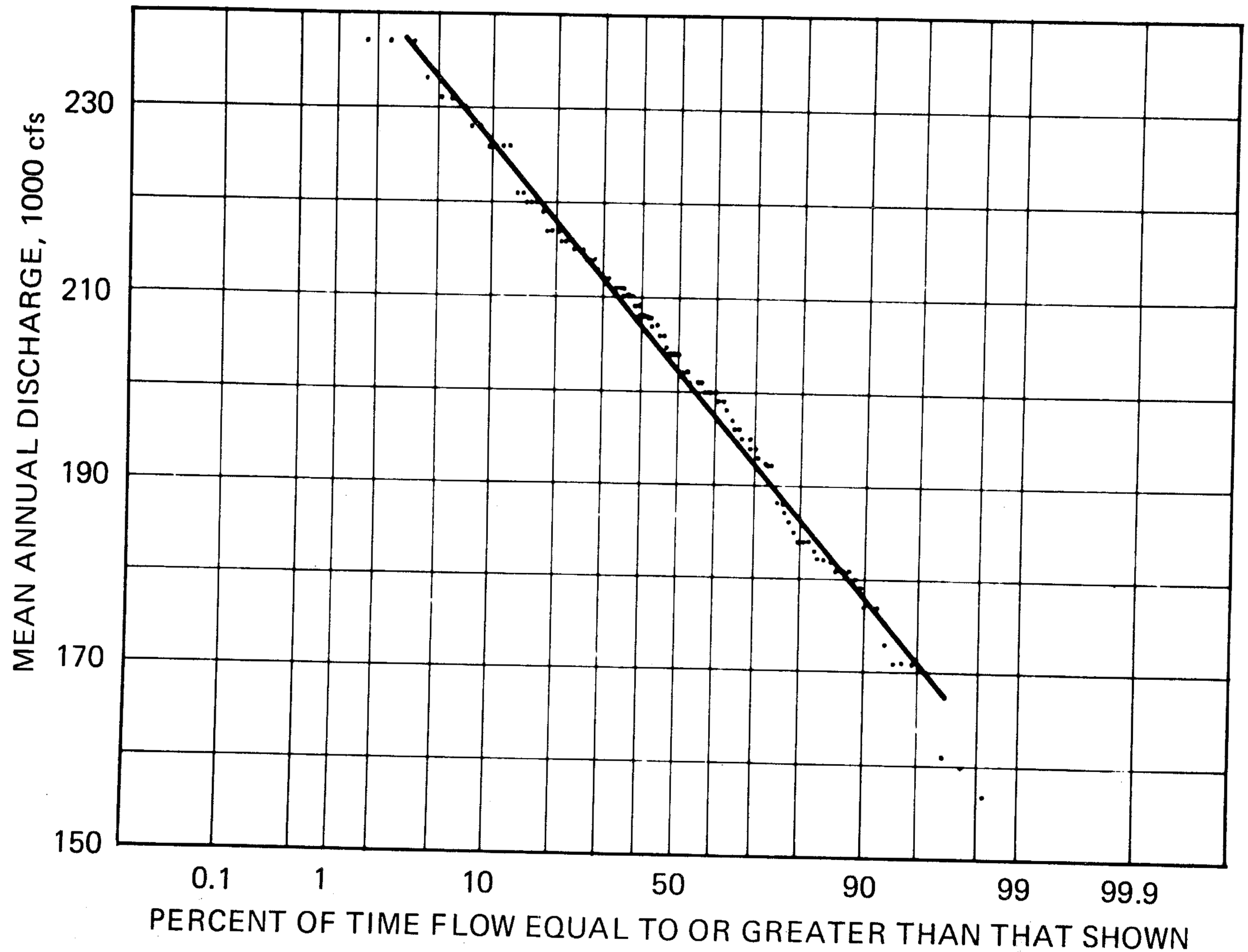

Figure 6. Frequency distribution mean annual flows of Niagara River (1860-1972)

1966-67 estimates, the equivalent total phosphorus concentration estimated for the Niagara River was about .04 mg P/l which was the approximate mean concentration measured during IFYGL. The standard deviation of the concentration is about .006 mg P/l or about 15% of the mean. One can then calculate the mean and standard deviation of the nutrient mass input as follows (assuming that flow and concentration are uncorrelated):

$$\bar{W} = \bar{Q}\bar{c} \tag{3}$$

and

$$s_W = \sqrt{(\bar{Q})^2 (s_c)^2 + (\bar{c})^2 (s_Q)^2 + s_Q^2 \, s_c^2} \tag{4}$$

where $\bar{Q}$ is the mean flow, $\bar{c}$ is mean concentration, $\bar{W}$ is mean mass input, s_Q, s_c and s_w are the standard deviations of flow, concentration and mass input respectively. Applying Eqs. (3) and (4) to nitrogen and phosphorus for the 1966-67 and the 1972 IFYGL estimates gives the results shown in Table 4. Particular attention is drawn to the last column, i.e. the range of the expected fluctuation in the nutrient loads. This range is taken as two standard deviations and as shown is about 30% of the average load or the load from the Niagara River may vary ±15% around the average.

It can be assumed as a lower bound that tributary inputs together with the municipal, industrial and other inputs vary by approximately the same percentage. Casey and Salbach[4] estimate a range of 14-21% for total phosphorus for the Genesse, Oswego and Block River. Overall then, one may expect the load estimates to vary by about ±15%. This will, of course, assume some considerable importance in any simulations.

TABLE 4

NUTRIENT LOADINGS FROM NIAGARA RIVER

		1966-67[2]	1972 IFYGL[3]	Average $\bar{x}$	Estimated Std. Dev. s	Range of Expected Fluctuation =2s
Flow	(cfs)	190,000	228,000	202,000	19,000	38,000
	(m^3/sec)	5,300	6,400	5,700	540	1,100
Total Phosphorus (as P)	(lbs/day)	42,200	45,900*	42,400	7,700	14,400
	(metric tons/day)	19.1	20.8	19.3	3.5	7.0
Total Nitrogen (as N)	(lbs/day)	522,200	482,400	470,400	62,300	124,500
	(metric tons/day)	236.9	218.8	214.	28.0	56.

*Canadian estimate = 51,100 lbs/day (23,200 kg/day)

SIMULATION INPUTS

A range of conditions on the external nutrient inputs has been examined. However, in order to place the "present" loads into an historical perspective, a preliminary analysis was made of the nutrient inputs that might have existed at some distant time in the past. These loads were termed the Pastoral Loads. In addition, a review was made of nutrient reductions as suggested by Vollenweider[5] and the Great Lakes Water Quality Agreement[6]. The simulations were therefore prepared using a wide range of input nutrient loads with the Pastoral Loads as a baseline condition.

Pastoral Loads

It is difficult to evaluate whether presently observed conditions in the Lake represent a "serious" condition or a condition close to "desirable". If water quality objectives are set without considering what the state of the water body would be if man had not happened on the stage, unrealistically high expectations might occur. It must always be remembered that even before the coming of man, there was phytoplankton biomass in Lake Ontario. The question is, "What is a reasonable level of biomass for Lake Ontario at some time in the past?" An estimate must therefore be made of the loads that existed before any substantial effect by man's activities.

The settlement of the Great Lakes Basin caused the nutrient loads being placed on the lakes to increase greatly. Lake Ontario's nutrient loads are heavily influenced by man, both within its basin and by man's influence on the upper lakes. Keeping in mind that point sources are the most readily controlled sources of pollution, the problem posed was to estimate what Lake Ontario water quality would be if only background, non-point sources of nutrients were allowed to enter the lake. This setting was viewed as a "pastoral" condition, similar to the time when the basins were principally rural areas, with no major population centers, and hence no significant human waste water affecting the lake.

The conditions which determined the pastoral simulation were as follows. The entire basin was considered to be a rural area with no significant human waste water entering the lake directly. The nutrient load from the Niagara River was assumed to reflect a "pastoral" condition in the upper lakes. It was assumed that the present conditions on Lake Huron could be used as an approximation to what the pastoral conditions would be in Lake Erie.

Loehr[7] summarizes characteristics of rural runoff and reports 0.20 to 0.28 lbs of total phosphorus per acre per year and total nitrogen loading of 1.3 to 2.9 pounds per acre per year in runoff from rural areas containing no significant human wastewater contributions to the streams. Loading rates of 2.0 lbs nitrogen/acre/year and 0.25 lbs phosphorus/acre/year were therefore used for the basin contribution to the nutrient loads. The Niagara River load was calculated using southern Lake Huron mean concentration data (Great Lakes Water Quality Board[8].) The organic nitrogen load was assumed to be ten times the organic phosphorus load. Table 5 shows the estimates of the pastoral loads used in the simulations.

Historical Loads and the U.S.-Canada Agreement

The estimates of the pastoral loads provide a lower bound on the external nutrient inputs to the Lake.

TABLE 5

ESTIMATED "PASTORAL" NUTRIENT LOADINGS

	Lake Ontario Basin Component		Niagara River Component		Total
	lbs/acre/yr	lbs/day	mg/l	lbs/day	lbs/day
Total Nitrogen	2.0	122,000	0.270	283,800	405,800
Organic - N	1.4	86,400	0.0450	47,300	133,700
Total Inorganic-N	0.6	36,600	0.225	236,500	273,100
Ammonia - N	0.1	6,100	0.034	35,700	41,800
Nitrate - N	0.5	30,500	0.191	200,800	231,300
Total Phosphorus	0.25	153,000	.005	5,200	20,500
Organic P	0.238	14,500	.0045	4,700	19,200
Inorganic P	0.012	800	.0005	500	1,300

Lake Ontario Basin: 2.227×10^7 acres (90,132 km^2)

Niagara Flow: 195,000 cfs

The estimates of the present loads (1967-72) provide a measure that indicates the increase in load that has occured over the past 50-100 years. Two additional input patterns are important: a) the historical load pattern between the present and say, the turn of the century and, b) the expected future load pattern.

Any attempt at estimating the past nutrient inputs over the preceding five decades is paved with many difficulties. The effects of input from Lake Erie, population growth, the introduction of phosphorus detergents and varying land use practices are several examples of important phenomena that contribute to the nutrient input. In addition, as noted previously, normal flow and concentration variations in the Niagara River input alone can vary by as much as $\pm$ 15%. A detailed study would be necessary to delineate each of these components. Such a study is clearly outside the scope of this work. However, in order to provide at least some basis for placing the loads used in the simulation in an historical context, the simplest and most crude analysis was performed to estimate the historical load pattern since 1900. Because of the importance of phosphorus and the U.S.-Canada Agreement on phosphorus (see below), the historical analysis was restricted to that nutrient. The procedure to estimate the phosphorus loads was as follows:

a) The population tributary to Lake Ontario was determined using estimates by O'Connor and Mueller[9]

b) Given the 1970 population and load (exclusive of the pastoral input) a per capita loading of phosphorus was determined.

c) A per capita loading of domestic waste water was estimated.

d) A time history of the per capita loading was graphically estimated considering the introduction of detergent phosphorus beginning in 1950.

e) The resulting intermediate loads were then calculated, to which was added the pastoral input.

f) Atmospheric inputs were not included in order to provide a basis for comparison to the U.S.-Canada Agreement loads. The results of this analysis are shown in Table 6.

TABLE 6

APPROXIMATE HISTORICAL PHOSPHORUS INPUTS

(Atmospheric Inputs Not Included)

Year	Estimated Trib.Popul. (Millions)	Estimated lbs/day/ Capita	Load 1000 lbs/day	Load & Pastoral Input 1000 lbs/day
1900	2.4	.003	7.2	27.2
1910	2.7	.003	8.2	28.2
1920	3.1	.003	9.3	29.3
1930	3.5	.004	14.2	34.2
1940	4.0	.004	16.2	36.2
1950	4.6	.007	32.2	52.2
1960	5.3	.009	47.7	67.7
1970	6.0	.011	66.0	86.0

Fig. 7 is a plot of the approximate total phosphorus load and for the latter two decades the range due to variations in Niagara River flow and concentration is also plotted. This range represents $\pm$ 7700lbsP/day as shown in Table 4 and is reasonable since the range in Niagara River annual flows from 1950-1972 was from 161,000 cfs to 231,000 cfs. This flow range covers the expected range in Niagara River mean annual flow.

As shown in Table 6 and Fig. 7, total phosphorus loads to the whole of Lake Ontario have increased by about 15,000 lbs P/day for each 10 years since about 1950. The total load is now about three times the load at the turn of the century and about 2.4 times the 1940 load, prior to the introduction of detergents. As shown in Fig. 7,

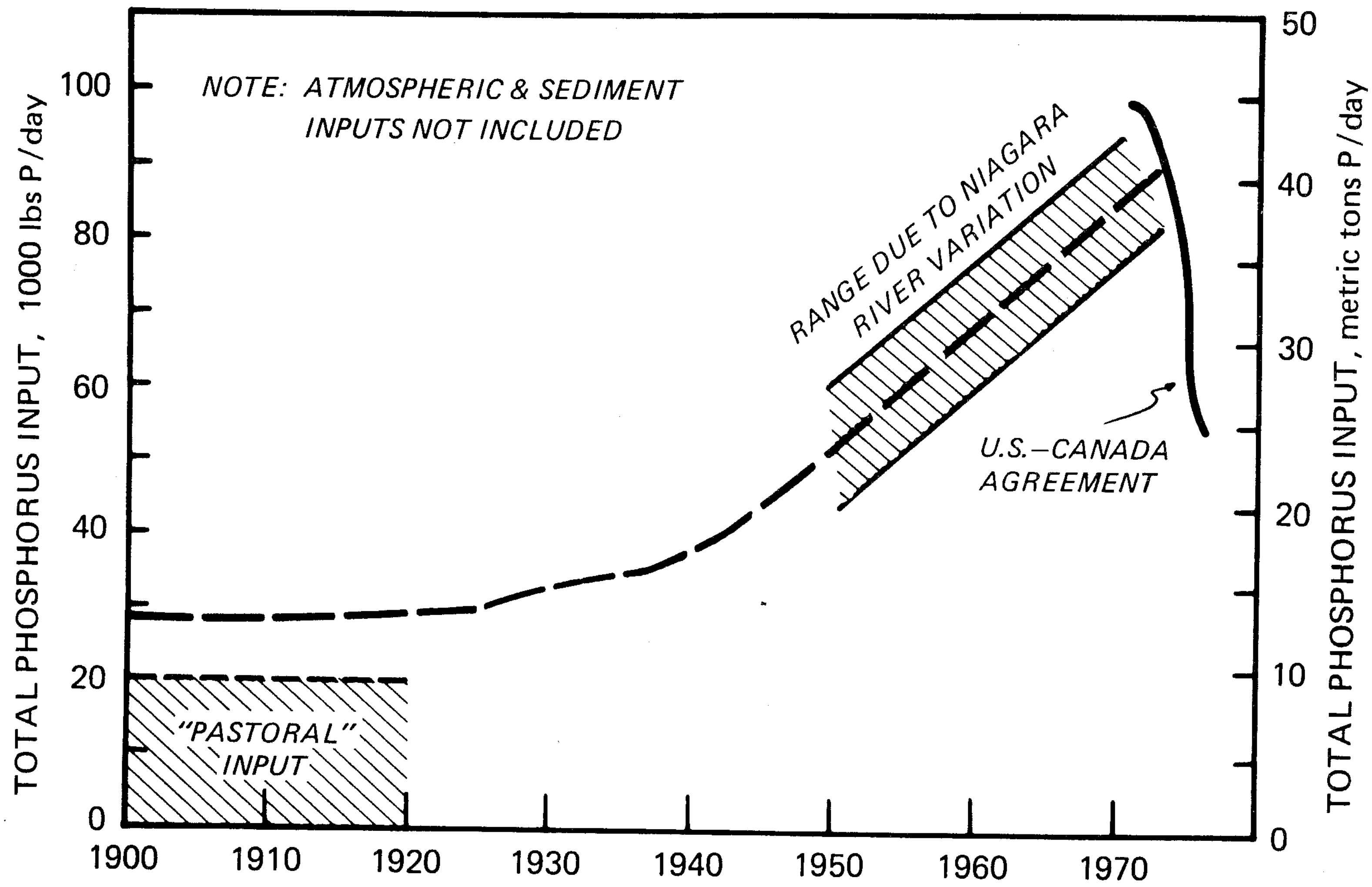

Figure 7. Approximate historical phosphorus inputs to Lake Ontario and the U.S.-Canada Agreement inputs

the range due to changes in Niagara River input (see Table 4) is significant and probably will mask any further attempts at detailed refinement of the estimates such as land use breakdowns. Indeed, for a range of 14,400 lbs P/day (equal to ± one standard deviation), and an average per capita loading of 0.009 lbs P/day, the variation in Niagara River input is equivalent to a basin population of 1.6 million or about 25% of the present population. It should be stressed again that the loads shown are illustrative only and represent only the crudest of estimates.

The simulation question discussed earlier assumes particular importance in the light of these estimates. The phosphorus input to the Lake has been increasing significantly over the past two decades and there have been significant year to year changes due to changes in Niagara River flow. As such, the state of the Lake in 1967-70 and again in 1972 the years of intensive sampling, represents some integrated average response of the past loads.

Fig. 7 also shows the loads promulgated under the U.S.-Canada Agreement on phosphorus control[6] Several points can be noted. The load estimated in 1971 of 98,600 lbs P/day (44.8 metric tons/day) appears to be an over-estimate of the input so that future estimates of input may show an apparent reduction. The range of input load from the Niagara River will increase the difficulty of estimating the changes in load as part of the Agreement. For all practical purposes, in terms of simulation, the Agreement loads represent an approximate step function decrease (or instantaneous decrease) in load to conditions of the early 1950's.

In the light of the range of loads shown in Fig.7, extending from a pastoral level of some 20,000 lbs P/day to maximum levels of about 100,000 lbs P/day, simulations were prepared assuming input nutrient loads covering this range.

It should also be noted that this range of external inputs includes any uncertainty in the possible release of phosphorus from the sediments to the hypolimnion. Bannerman et al.[10] estimated the annual contribution of inorganic phosphorus from the sediments to be about 1.4×10^6 kg P/year (8,500 lbs P/day) or 10% of the total phosphorus input (see Table 3). As indicated previously, the range of the external load input is about $\pm$ 18% so that the contribution from the sediments would tend to be masked and in any event is covered by the external loading range.

Nevertheless, the importance of the sediments as a phosphorus source cannot be completely ignored especially as external sources are reduced. For example, at the U.S.-Canada Agreement load of 54,800 lbs P/day, the sediment input rises to 16%. Under future load reduction therefore, further attention should be directed towards the role of the sediments.

SECTION V
RESULTS OF SIMULATIONS

A variety of simulations have been carried out using the Lake 1 model and the range of loads indicated in Section IV. Except for one illustrative case, the procedure followed in each simulation was similar. The Lake 1 model kinetic structure was used including the initial conditions of the 1967-70 period. A new external load was then imposed, representing a step function decrease or increase in the load. The model was then run until a new dynamic equilibrium was obtained. No attempt was made to estimate an actual future load time history; rather, a range of external conditions was imposed to illustrate the nature of the Lake response.

As discussed in Section III, the verification analysis is most responsive to the initial conditions as opposed to the external inputs. Yet, one of the key parameters, the overall loss rate of a nutrient is critical to the response of the lake over the long term. This is shown in Fig. 5. The verification analysis provides only an estimate of the loss of phytoplankton nitrogen and phosphorus to the sediments. It is not possible to estimate from the short observation period available the decay of detrital and other forms of organic nutrients or the decay of dissolved inorganic forms. The following range of conditions was therefore used in the simulations:

1) Non-living organic nitrogen and phosphorus assumed at two levels (a) conservative or (b) A loss from the system at a rate of .001/day (equivalent to an approximate settling rate of .1 m/day)

2) Phytoplankton phosphorus and nitrogen settling rate of .1 m/day.

3) Inorganic forms of nitrogen and phosphorus are assumed to be conservative.

This range of conditions on the decay or loss of nutrient from the system is believed to be reasonable. However, it should be stressed that the only real check to date is on the second condition. The results of the verification analyses indicated that the dynamic behavior of the phytoplankton can be verified with a settling velocity of about 0.1 m/day for the Lake 1 model. It appears plausible to assign a similar settling velocity to the other organic forms although some fraction of that form is undoubtedly dissolved. Consequently, this form of nutrient was assumed at the two levels of K=0 and K=0.001/day to illustrate the sensitivity of the solutions to varying loss rates of organic nutrients. The question of the decay or loss of inorganic forms is considerably more difficult. Chemical mechanisms of co-precipitation and minerialization may be the cause of a loss of inorganic forms, however the degree to which this loss may occur in Lake Ontario is not known. It appears however, that such a loss is probably not significant and as such, it is assumed that the inorganic nutrient forms are conserved. The importance of this assumption is discussed below.

"Reasonable" kinetics for the simulations presented in this chapter therefore include loss rates of .001/day for the organic forms, a zero loss for the inorganic forms and a phytoplankton settling rate of 0.1 m/day.

CONTINUATION OF "PRESENT" INPUTS

The first series of runs examined the model response due to a continuation of present inputs where "present" was used as the nitrogen and phosphorus input distribution shown in Table 7. This distribution is identical to that used in the verification analysis[1]. Atmospheric and sediment inputs are not included explicitly but are incorporated in the range of results discussed below.

TABLE 7

ASSUMED "PRESENT" NUTRIENT LOAD DISTRIBUTION

System	Nutrient Load Metric Tons/Day	1000 lbs/Day
Nitrogen		
Non-living Organic N	250.0	551.4
Ammonia N	8.5	18.8
Nitrate N	141.8	312.9
Total	400.3	883.1
Phosphorus		
Non-living Organic P	25.9	57.1
Inorganic P	8.1	17.9
Total	34.0	75.0

Figure 8 shows the dynamic behavior of the phytoplankton biomass in the epilimnion for a continuation of the loads indicated in Table 7 and for a decay of organic nutrients of 0.001/day. This represents a "reasonable" condition on the system decay coefficients. As can be seen, the spring peak of phytoplankton reaches a new dynamic equilibrium after about 8-10 years or about equal to the detention time of the Lake.

The spring peak under this "reasonable" condition reaches a maximum value of over 16μg chlor./l or about 45% higher than the present peak. The fall peak also increases to just under 10 μg/l from present values of about 8μg/l or about 25% increase. The dynamic behavior of the phytoplankton under these conditions can be understood further from Fig. 9 which also shows the behavior of the inorganic nitrogen and available phosphorus. All variables are for the epilimnion. As shown, the lake as described by a "reasonable"

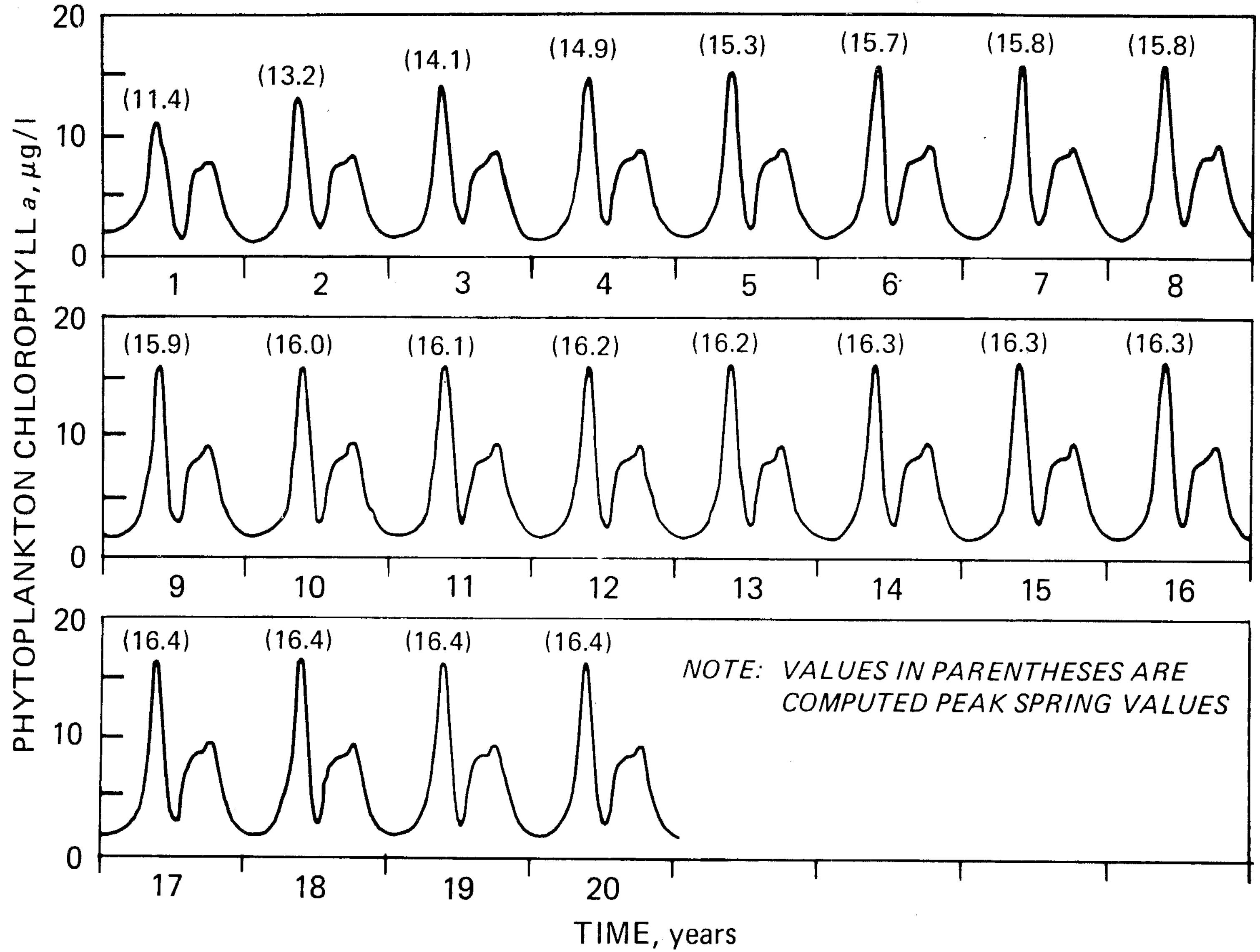

Figure 8. Dynamic behavior of phytoplankton biomass in epilimnion-continuation of "present" inputs

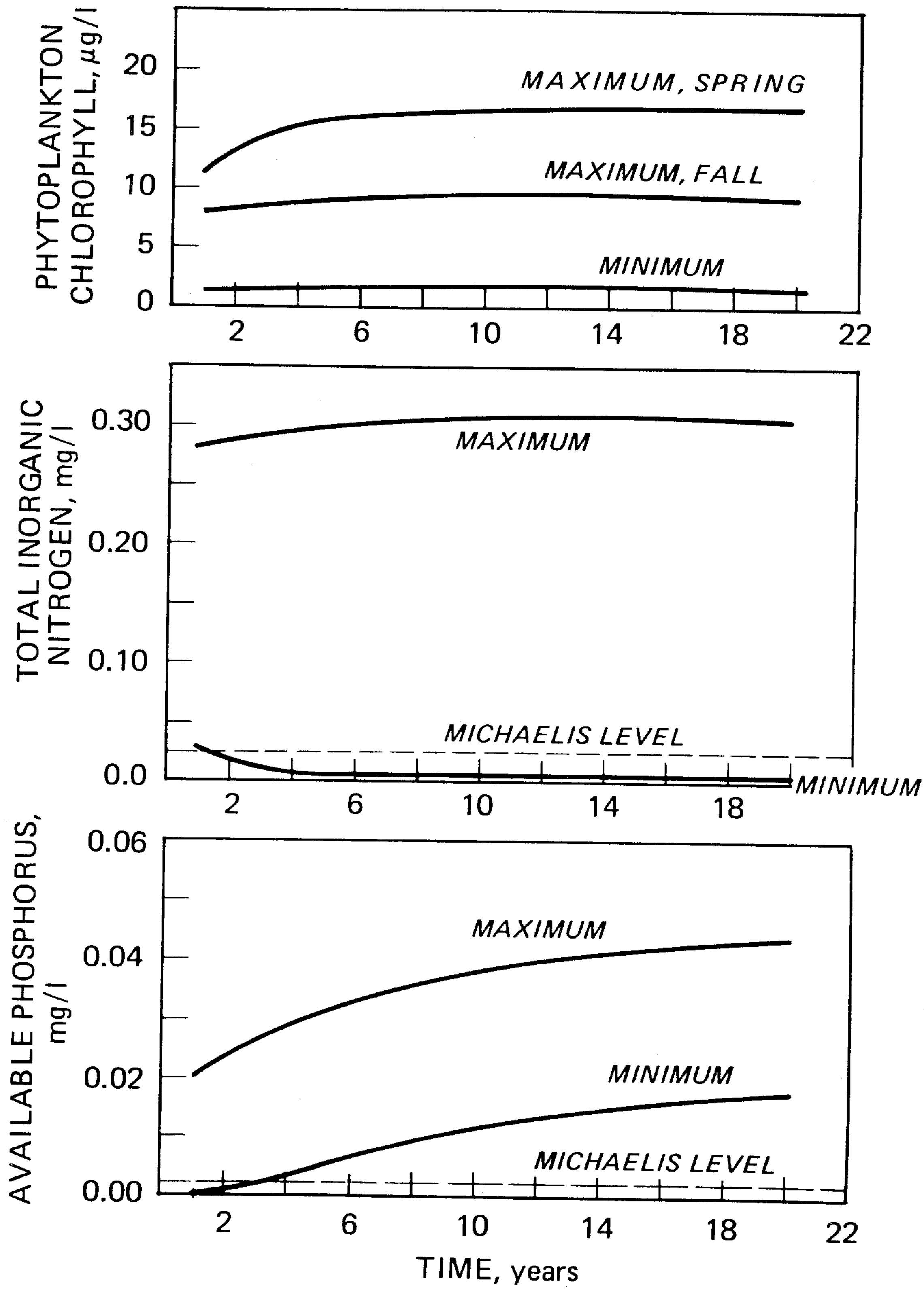

Figure 9. Dynamic behavior of phytoplankton, inorganic nitrogen and available phosphorus in epilimnion-continuation of present inputs

set of system parameters is not in equilibrium with respect to the present loads. Peak biomass continues to increase with the continuation of present loads. In contrast to the change in the peak spring biomass, the average annual biomass in the epilimnion is computed to change by about 0.9 μg/l, an increase of 20% or less than half the increase in the peak spring concentration. Changes then in spring peak concentrations are not paralleled by equal changes in the average annual concentrations. This is discussed more fully below. Also, after some initial time, the increase in biomass is controlled by the nitrogen and not the phosphorus as indicated by the increase in the minimum level of phosphorus about the Michaelis level. "Michaelis level" refers to the concentration of nutrient at which the growth rate of the phytoplankton is half of the maximum growth rate. This can also be seen in Fig. 10 in which N/KM + N represents the nutrient limitation for either nitrogen or phosphorus and KM is the appropriate Michaelis level. The nutrient balance in the spring is surprisingly sensitive to the distribution of the nitrogen and phosphorus concentrations. This can also be seen from the present data which indicate that the spring peak is controlled primarily by phosphorus but that nitrogen levels are approaching levels in the spring that could control growth. The simulation using the mix of nitrogen and phosphorus input loads as shown in Table 7 indicates that nitrogen may be the nutrient that will affect growth more than the phosphorus. This, of course, depends on the nitrogen-phosphorus input load distribution. A full summary across all load distributions is given below. The results as shown in Fig. 10 are however extremely interesting since they show that the lake may be in a delicate balance and that what appears to be a limiting nutrient during one period of years may not continue to affect growth in the same manner in later years. In both the spring and fall, nitrogen assumes a relatively greater role in nutrient limitation than does phosphorus. The effect is particularly

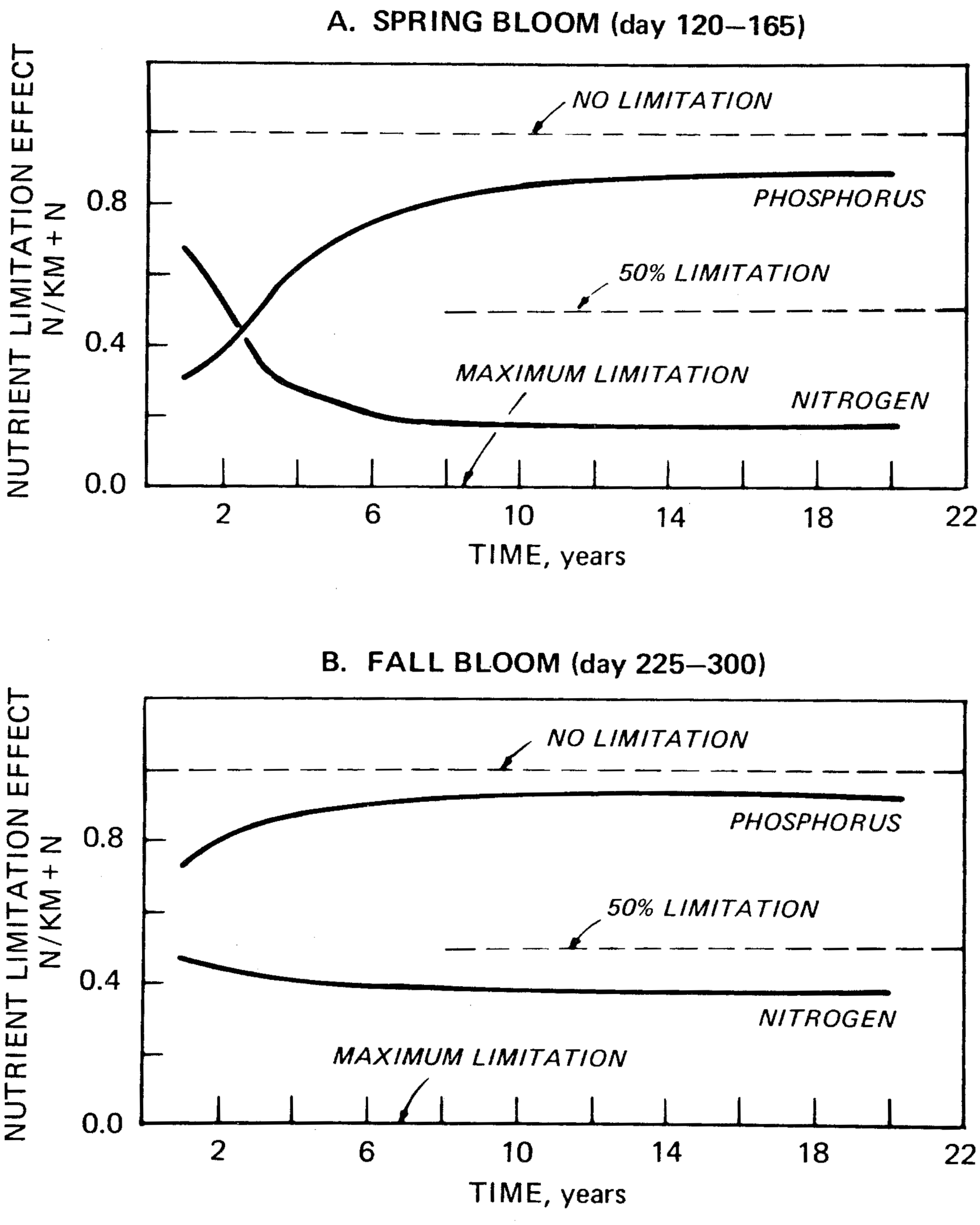

Figure 10. Nutrient limitation effect under continuation of present loads in epilimnion

noticeable in the spring bloom where after about 3-4 years, nitrogen becomes more limiting. It whould be stressed however, that the model does not include any nitrogen fixing algae which would alter the nutrient limitation effect especially in the fall when blue green algae would be dominant.

Also, the computed shift to nitrogen limitation reflects, to some degree, the particular model structure that is used in the simulation. The evolution of a nitrogen-limited system is a much more complex phenomena of species adaptation and readjustment of the upper trophic levels than is indicated by the Lake I model. Nevertheless, the results are interesting and do indicate a general direction and sensitivity of the Lake to the two primary nutrients.

Figs. 11 and 12 show the behavior of the annual average values for segment #1 (0-17 meters), segment #2 (17-90 meters) and the lake average concentration, weighted volumetrically. Referring to the chlorophyll concentrations in Fig. 11, the substantial difference between annual average epilimnion level and the lake average can be noted. Also, the relatively small change of 13% in the lake average concentration can be contrasted to the change in the peak concentration of 45% shown in Figs. 8 and 9. The total inorganic nitrogen (TIN) plot shown in Fig. 11 indicates that the lake is in equilibrium on an annual average basis with that nutrient. The effect of the nitrogen limitation is quite clear from a comparison of the TIN and the available phosphorus. The latter nutrient is continuing to increase although the biomass in the epilimnion has reached an equilibrium level governed essentially by the conversion of the TIN in the upper layer. Note however, that the stoichiometric conversion does not apply to the annual averages but to the peak values shown in Figs. 8 and 9. That is, the approximately 165 μg TIN/l in segment #1 determines the

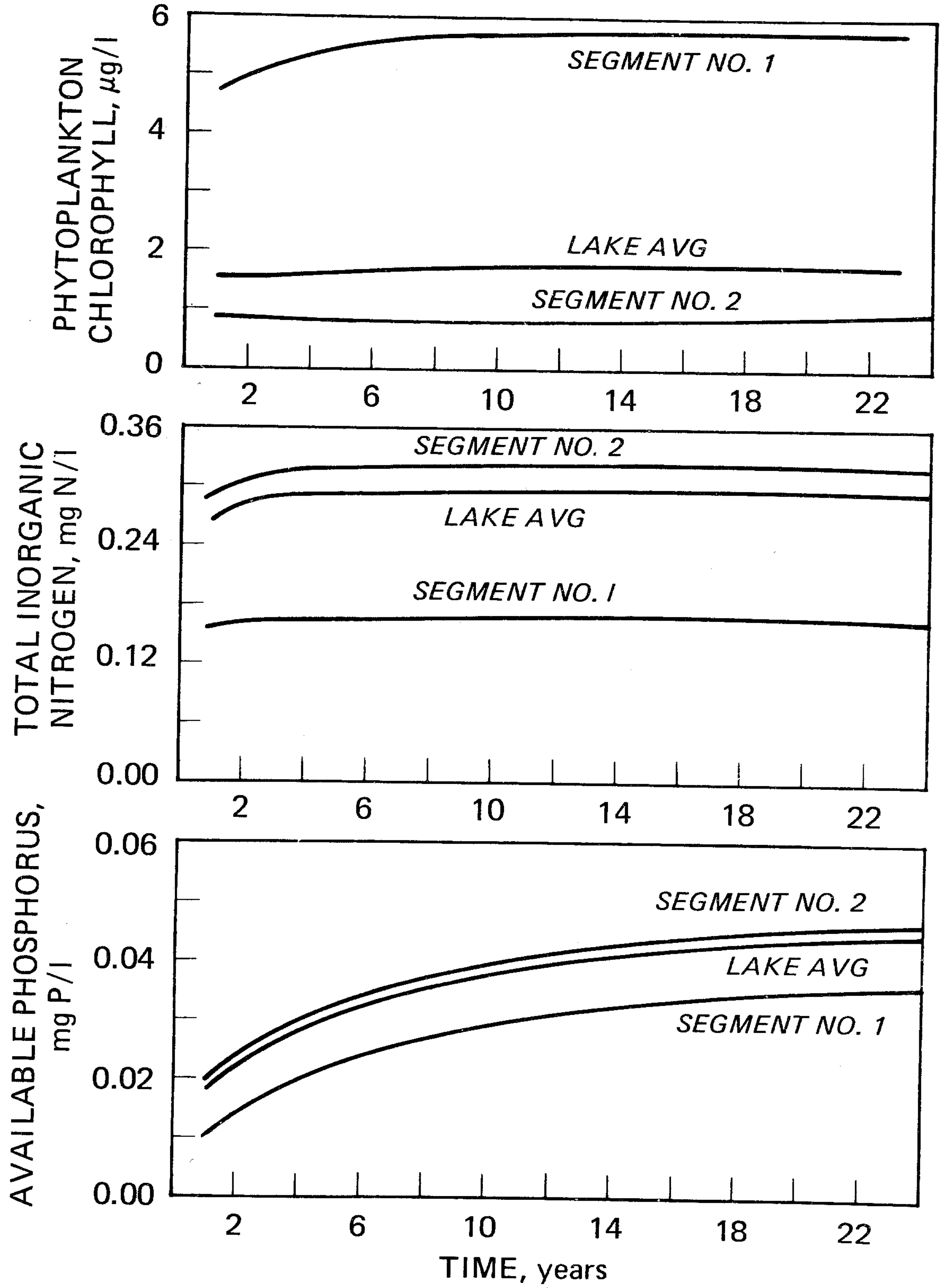

Figure 11. Yearly average changes - continuation of present inputs

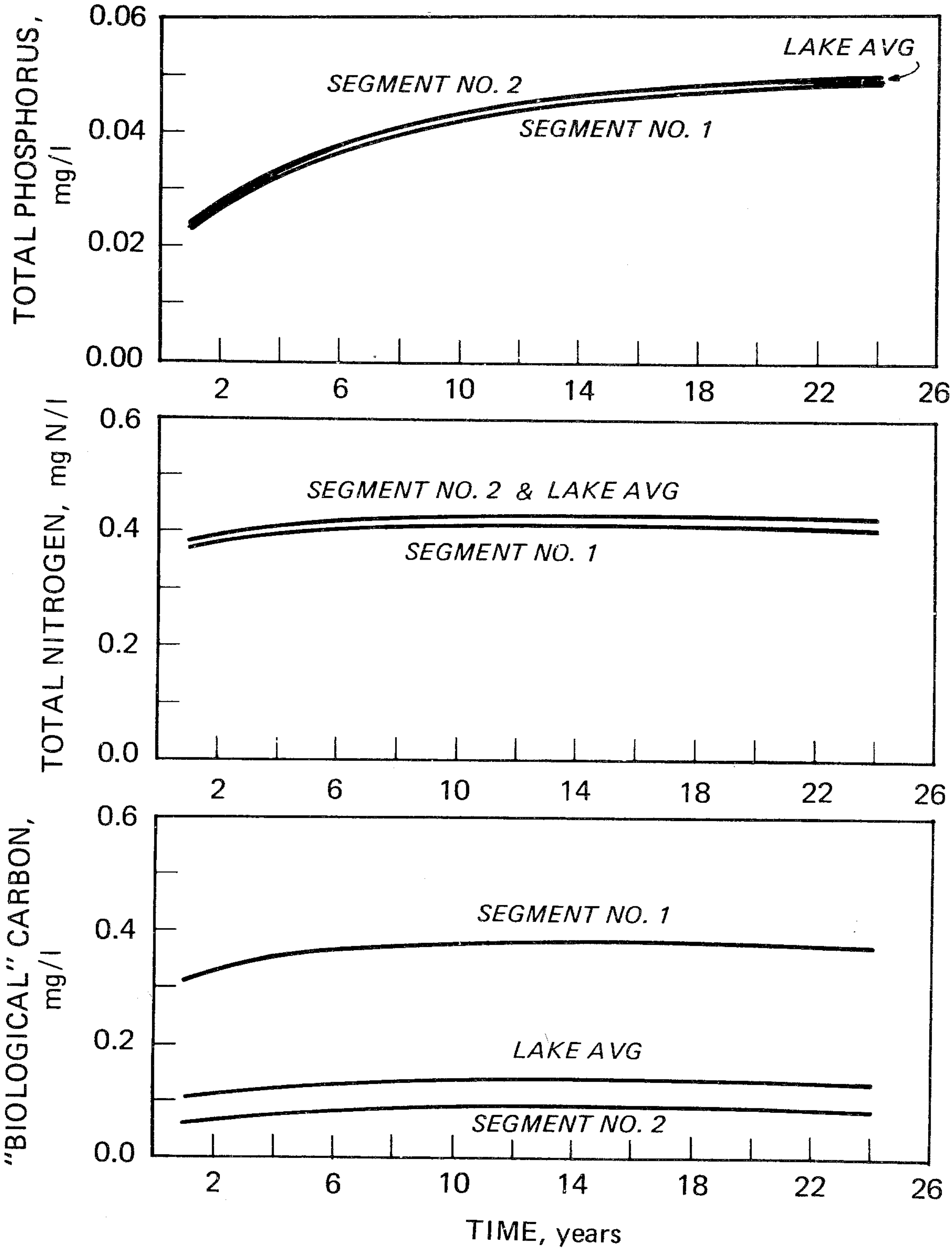

Figure 12. Yearly average change (cont.) - continuation of present inputs

peak level of phytoplankton of 16.4 μg/l. Because of the nature of the spring and fall blooms, a similar simple relationship does not exist between the TIN and the annual average chlorophyll.

Fig. 12 shows the total phosphorus, nitrogen and "biological" carbon (phytoplankton and zooplankton). Again, the total phosphorus and nitrogen plots show that the lake is not in equilibrium with the present phosphorus load. It is clear from Fig. 12 and the preceding figures that it would be difficult to make meaningful statements about phytoplankton biomass just from models that projected annual whole lake averages of total phosphorus or nitrogen. For example, an annual lake average of 50 μgP/l is computed which stoichiometrically would yield a phytoplankton biomass of 50 μg chlorophyll/l which is significantly higher than that calculated by the dynamic kinetic model. A similar argument applies to the total nitrogen.

One concludes therefore that a continuation of present loads will result in a continual increase in peak biomass for about a decade and that nitrogen would become increasingly important as a limiting nutrient. Peak biomass is estimated to increase by 45% over present levels under a continuation of present nutrient inputs. Further, models and analyses that deal only with annual lake averages and total nutrients may be severely in error in projections, at least when compared to the results of the dynamic model.

RESPONSE UNDER ZERO INPUT

In order to further study the behavior of the Lake under different load conditions, a run was constructed which utilized the present initial conditions of all variables with zero external nutrient inputs. The simulation therefore is intended to show the behavior of the model dynamics as the lake "winds down." The run is not intended in any way to be a realistic representation of the behavior of the lake in the past but rather is simply another insight into the behavior of the model with no external forcing functions.

Fig. 13 shows the phytoplankton chlorophyll in the epilimnion over a 16 year computation with zero nutrient input and present initial conditions. (The first year computation differs slightly from that in Fig. 8 due to some differences in recycle kinetics.) The results show that the fall bloom is the first to disappear and is essentially gone by about 5-6 years and the system is characterized for the remainder of the computation by a single peak. The peak occurs later and later in the year so that by the 10th year, the peak occurs close to the end of June.

Figs. 14 and 15 show the change in the maximum and minimum values of key variables for the zero nutrient input case. The rapid decrease in the zooplankton is interesting and accounts in part for the disappearance of the fall phytoplankton bloom. The decrease of the zooplankton reduced the fall recycling of nutrients leading to the decline of the fall bloom. Fig. 15 shows that both nitrogen and phosphorus interact to control phytoplankton growth as indicated by the minimum values of both nutrients being at or below Michaelis levels. The maximum values of the chemical variables and the phytoplankton appear to exhibit a type of first order decay over time although the zooplankton decline and the available phosphorus (for the first three years) are exceptions. The equivalent first order decay for the maximum phytoplankton chlorophyll is about .00063/day.

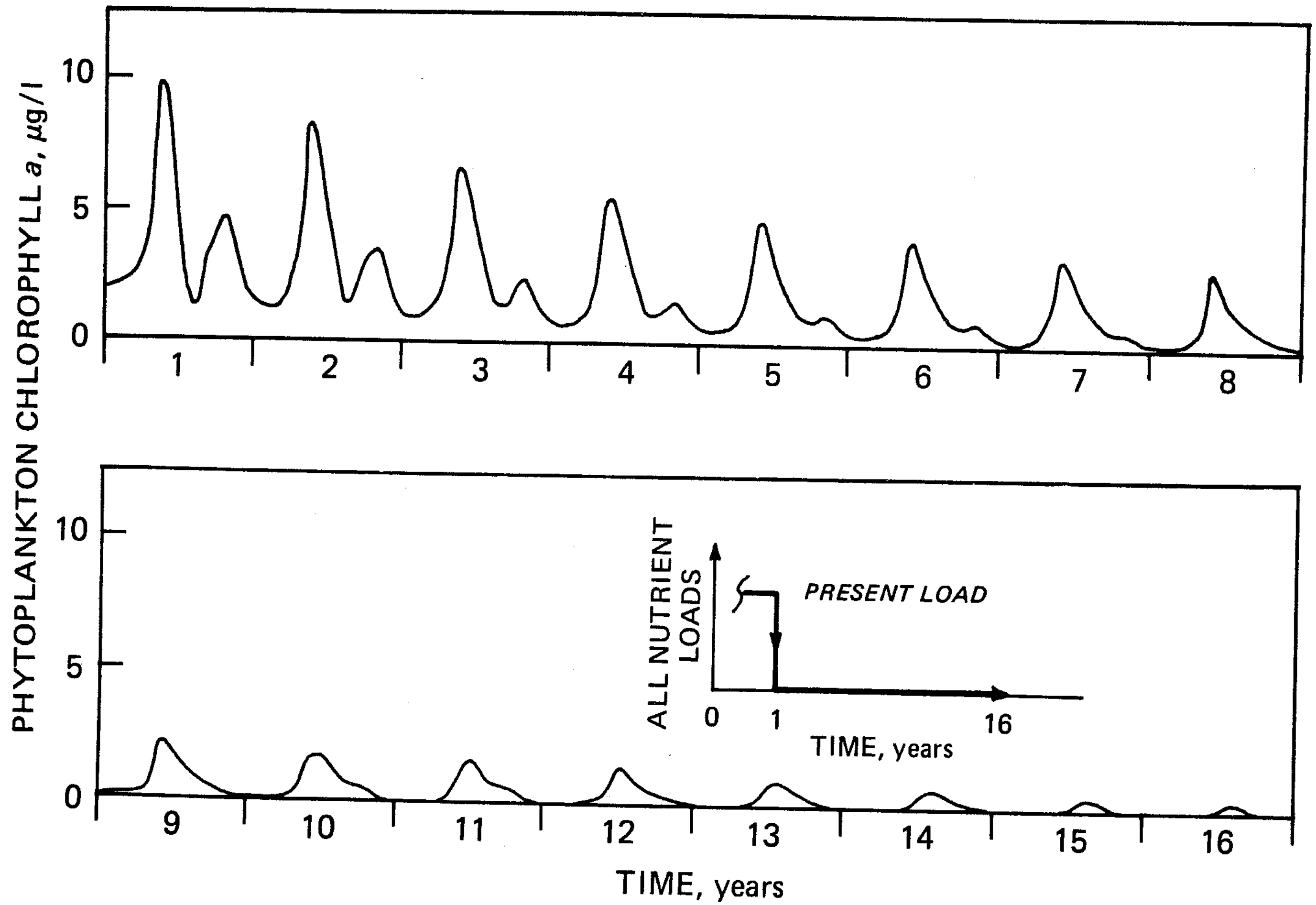

Figure 13. Phytoplankton behavior, epilimnion, present initial conditions, zero nutrient input

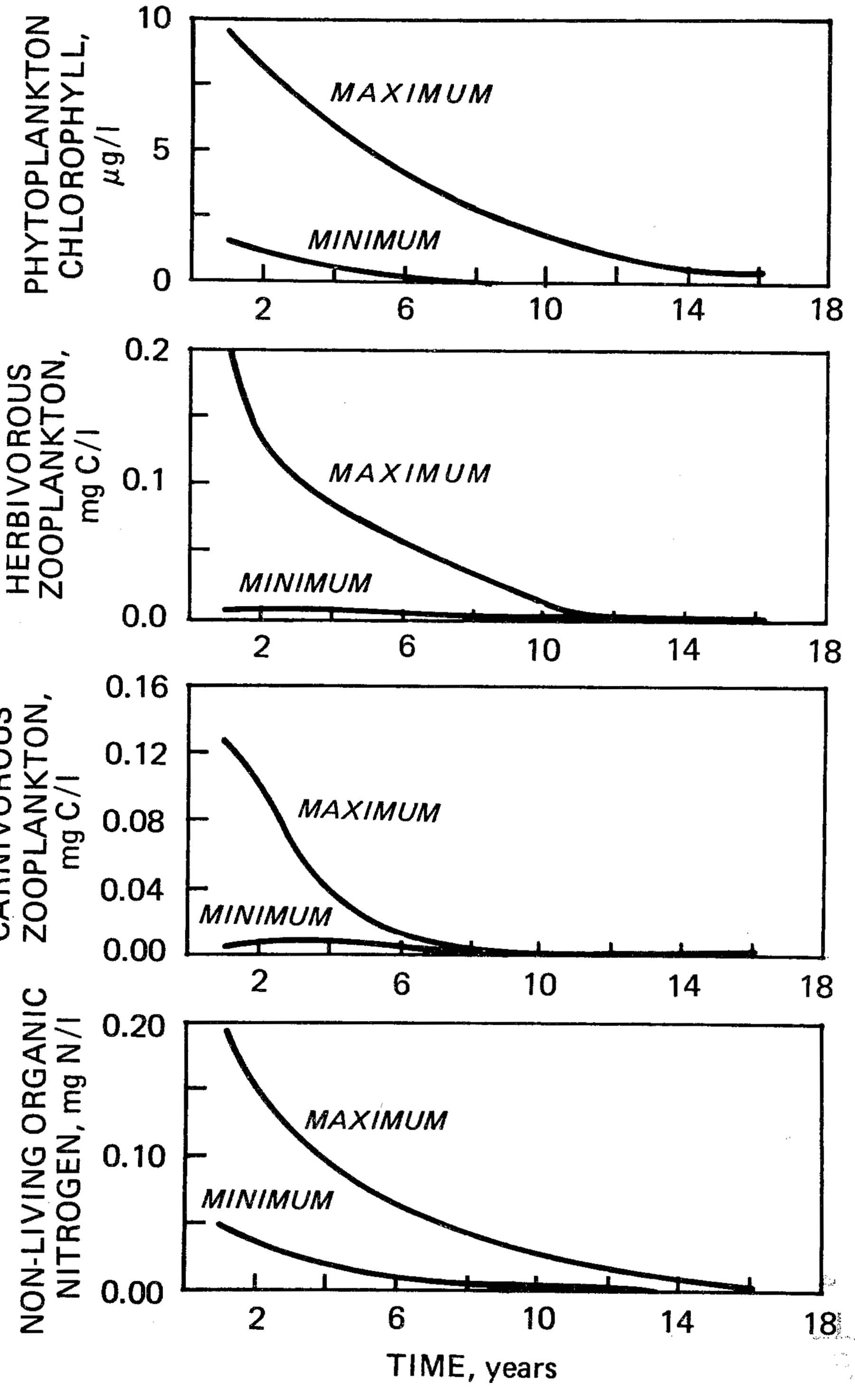

Figure 14. Maximum and minimum values, epilimnion, present initial conditions, zero nutrient input

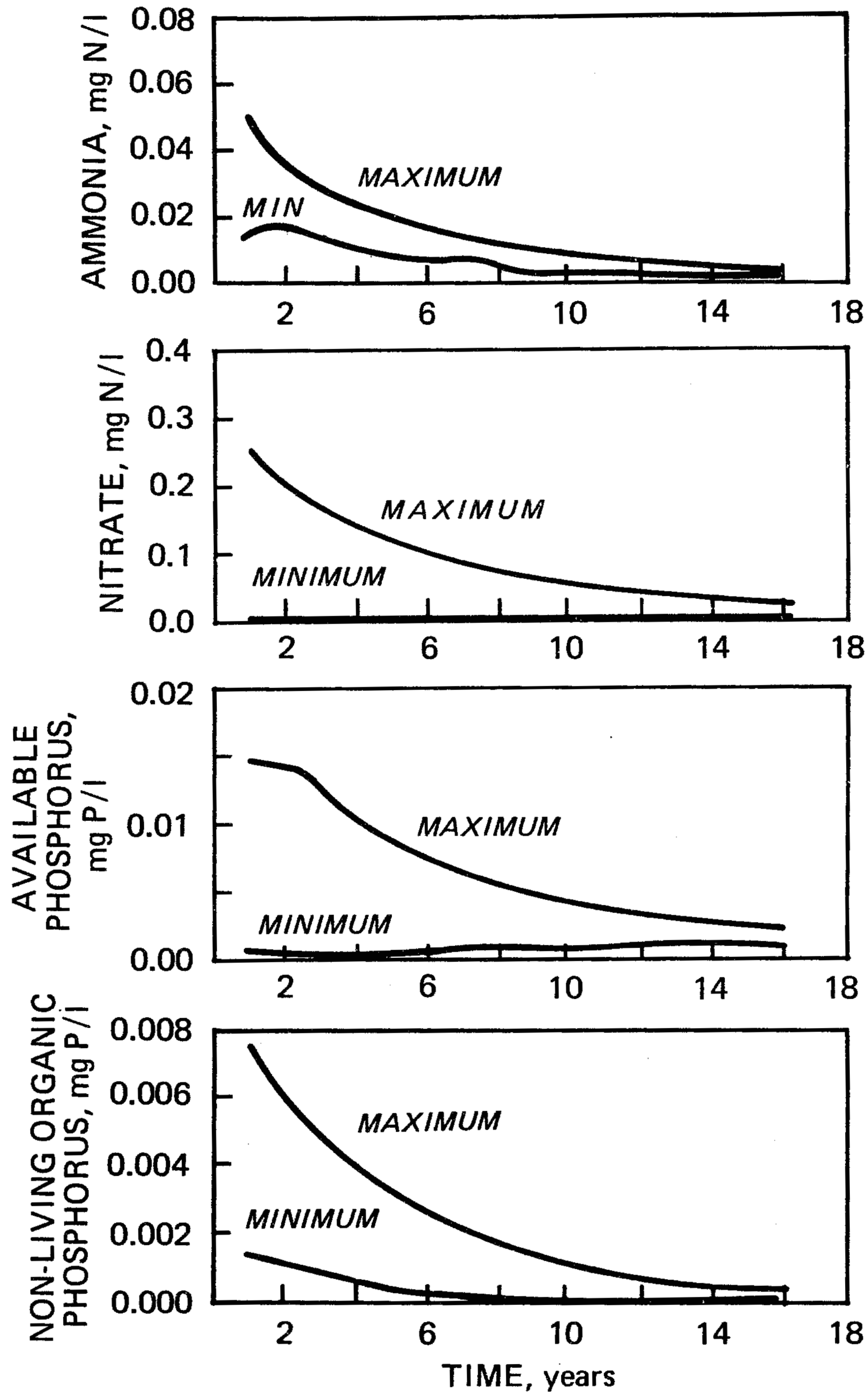

Figure 15. Maximum and minimum values-(cont.) epilimnion, present initial conditions, zero nutrient input

"PASTORAL" RESPONSES

As discussed in Section IV, an estimate was made of the nutrient loads that prevailed at some earlier time, a so-called pastoral condition. The loads are shown in Table 5. The results of such a simulation provide an approximate basis for measuring the degree to which present biomass exceeds some earlier level. Using "reasonable" long term kinetics, i.e. sinking of phytoplankton and decay of the non-living organic nutrient fraction, (.001/ day), the results are summarized in Figs. 16-20.

As shown in Figs. 16 & 17, the spring peak value is estimated to decrease to about 7 μg/l, a decrease of about 40% from present peak values. The broad fall peak is estimated to decrease to less than 4 μg/l, a decrease of almost 50% from present fall values. Fig. 17 can be contrasted to Fig. 9, the continuation of present loads. As the comparison indicates, the pastoral load system is primarily phosphorus limited in the spring. This is further indicated by Fig. 18(a) which shows phosphorus limitation in the spring at a constant level of about 0.28 although nitrogen approaches the 50% limitation in later years. Fig. 18 also shows the nutrient limitations for the fall bloom and surprisingly indicates that the growth during that time is progressively more nitrogen limiting. This again may be an effect of the particular nitrogen-phosphorus load distributions used for the pastoral case. A different mix of nutrients would result in a different nutrient limitation effect.

The yearly average changes computed for the pastoral loads are shown in Figs. 19 and 20. Phytoplankton chlorophyll on the annual average in the epilimnion decreases by about 40% to an equilibrium value of less than 3 μg/l. The time to equilibrium for each of the variables is about 10-15 years. The results of the computation on the state of the Lake under these pastoral conditions are summarized in Table 8.

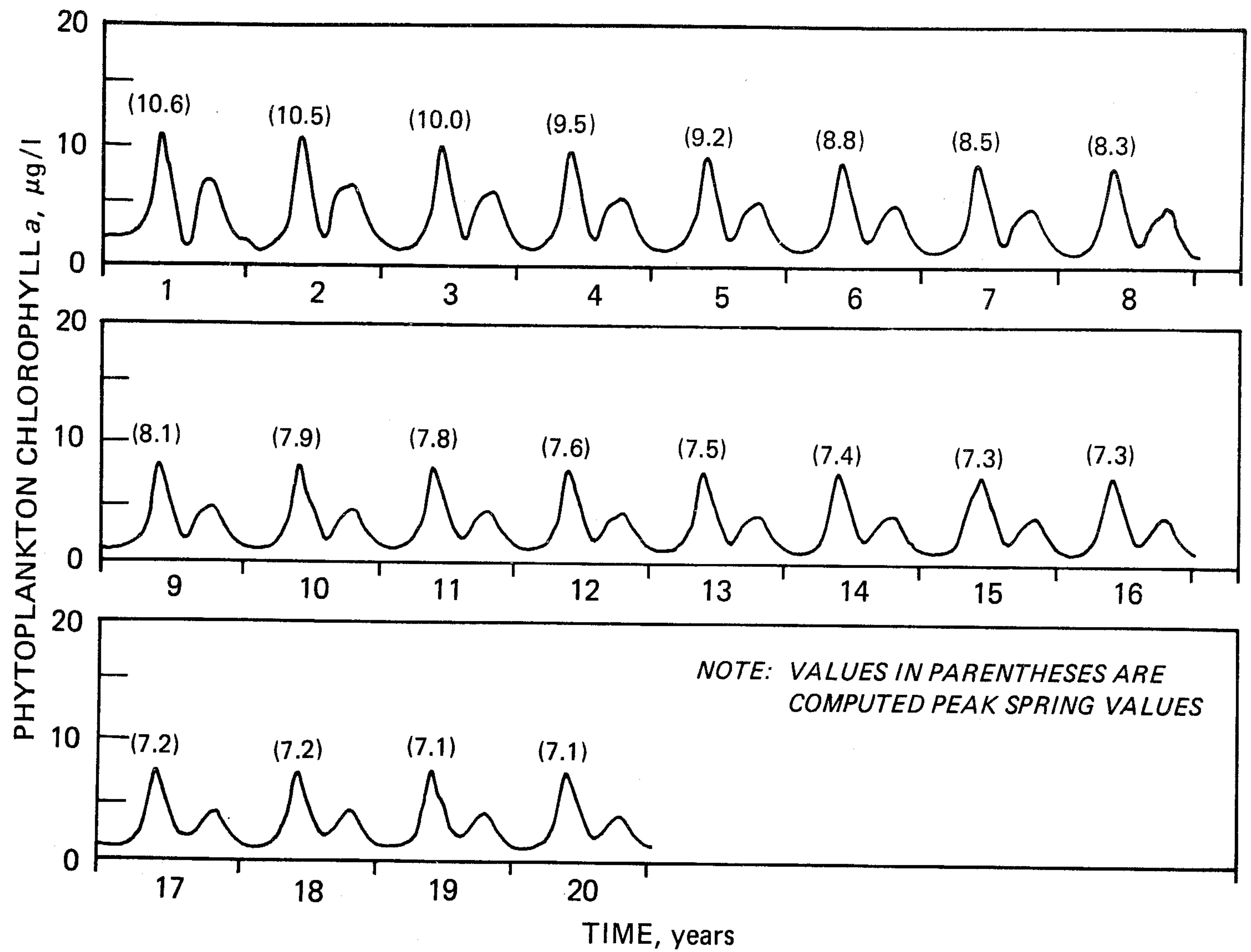

Figure 16. Dynamic behavior of phytoplankton biomass in epilimnion, "pastoral" inputs

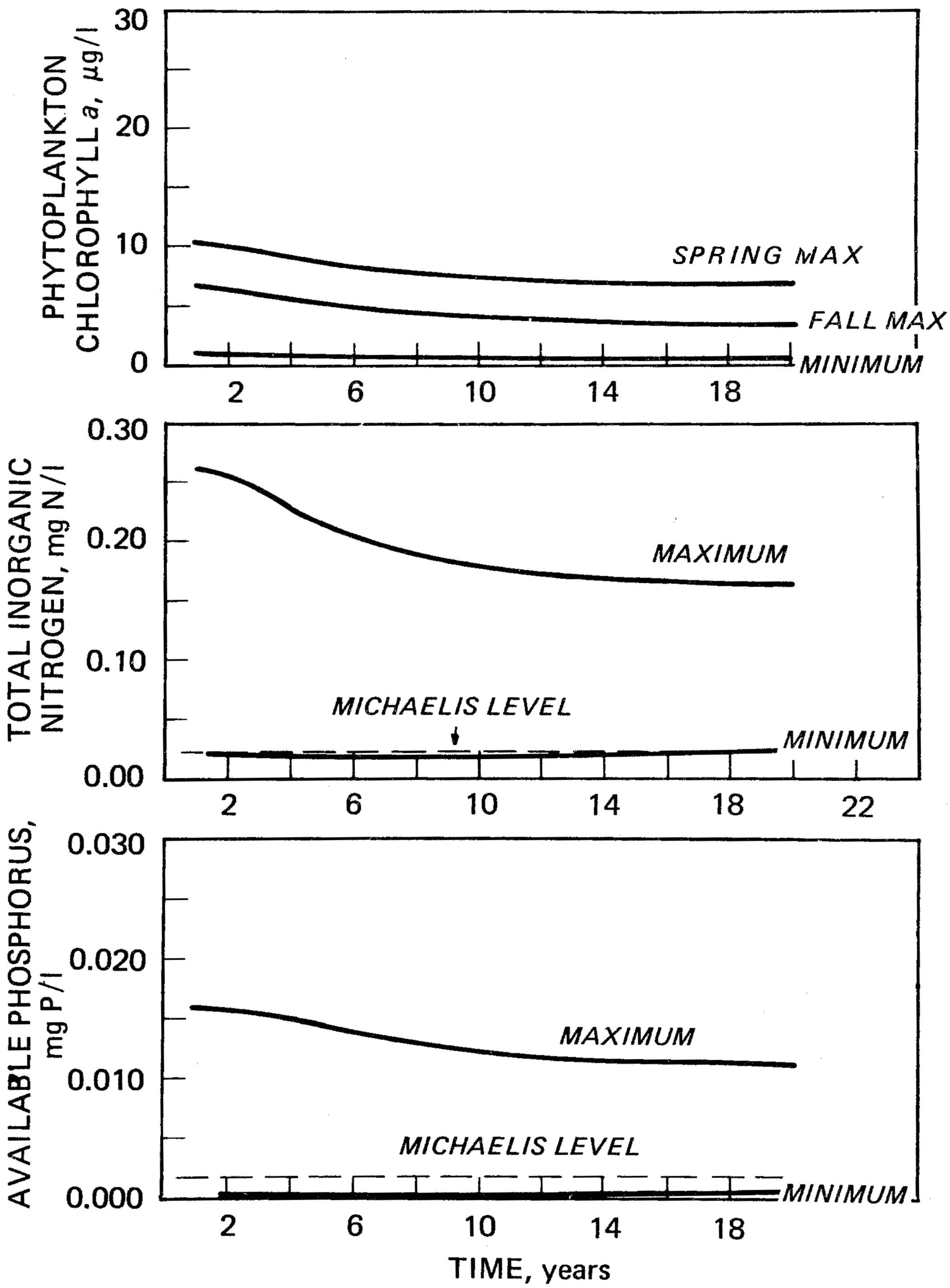

Figure 17. Dynamic behavior of phytoplankton, inorganic nitrogen and available phosphorus in epilimnion "pastoral" inputs

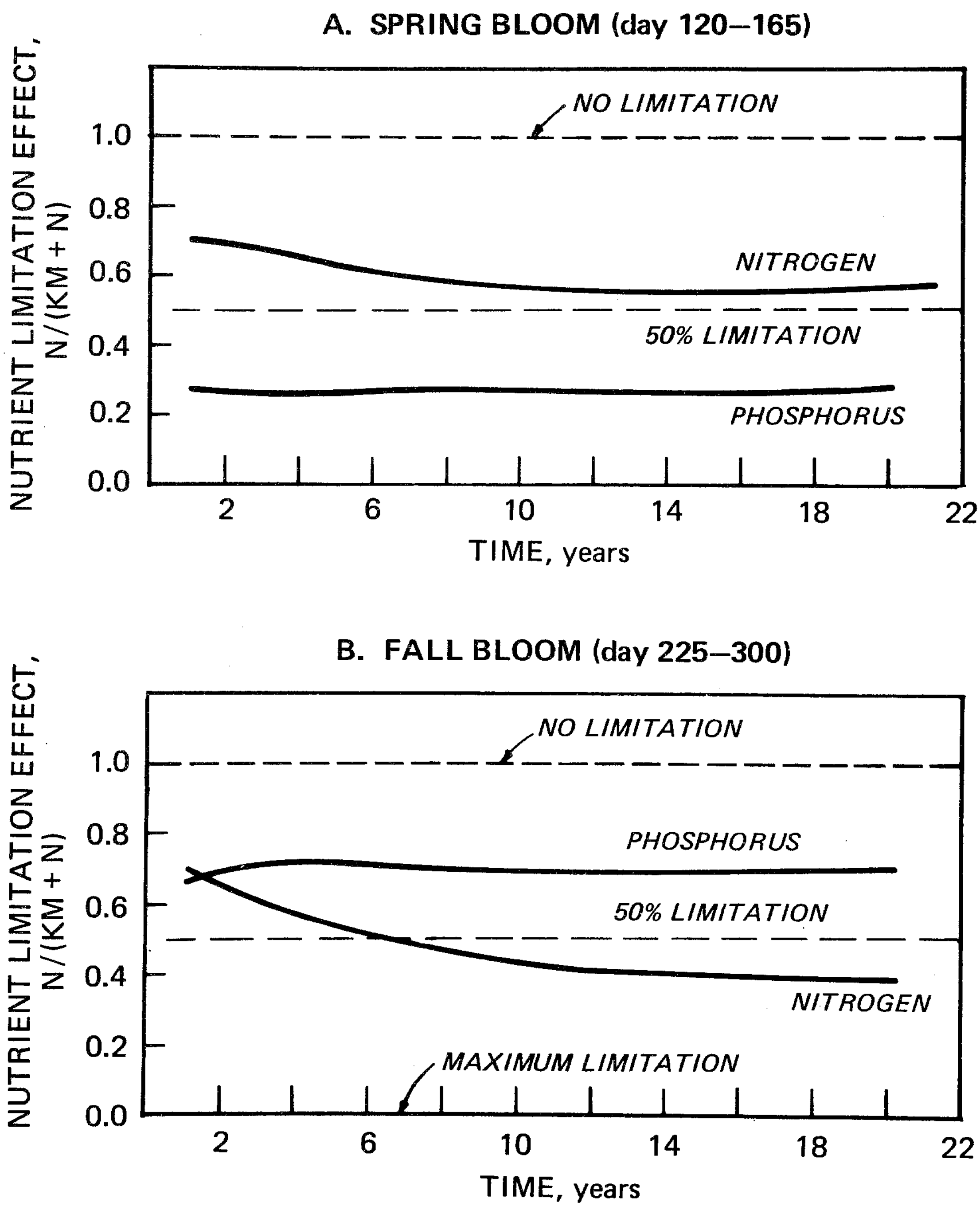

Figure 18. Nutrient limitation effect under pastoral inputs-epilimnion

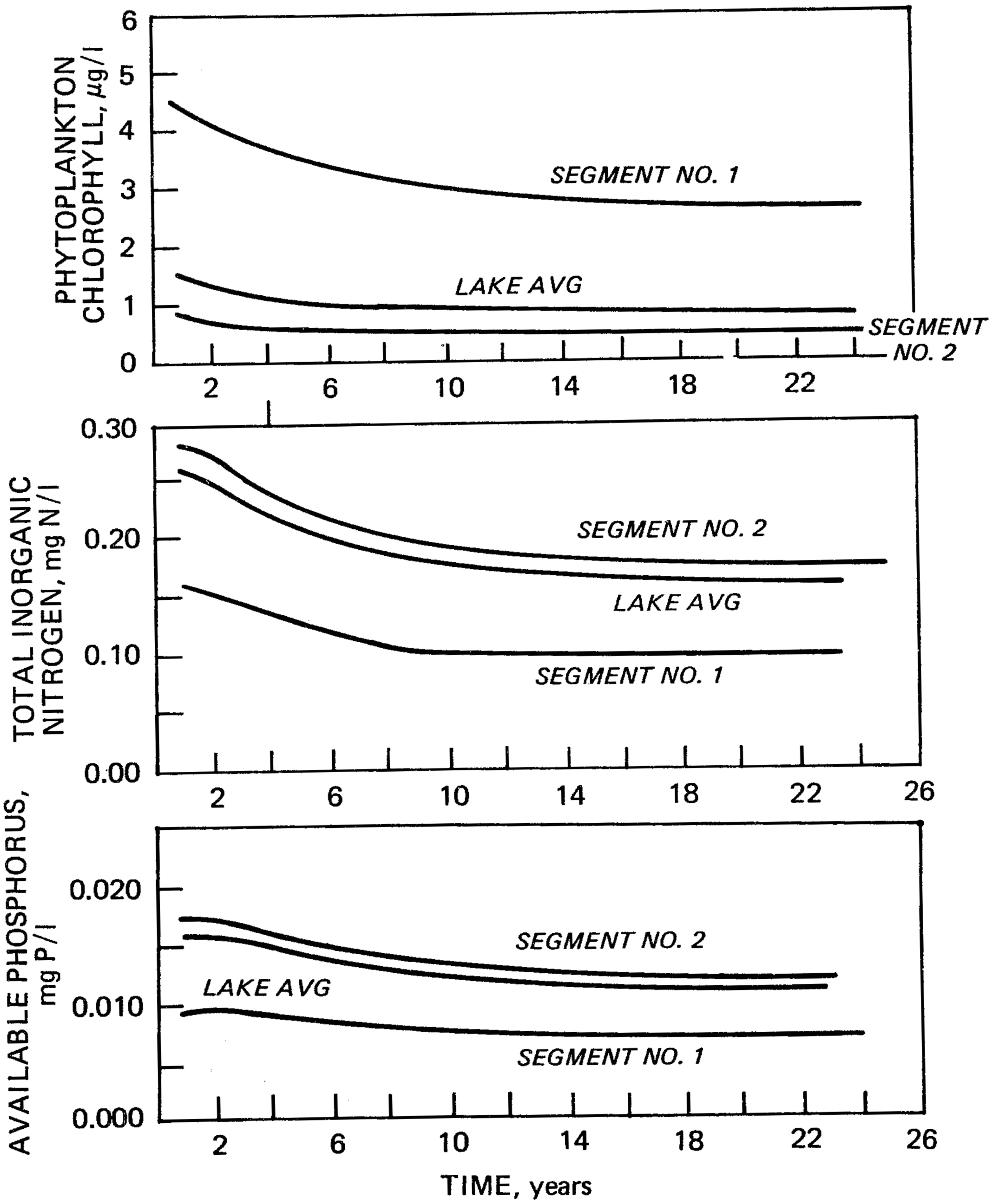

Figure 19. Yearly average changes - pastoral inputs

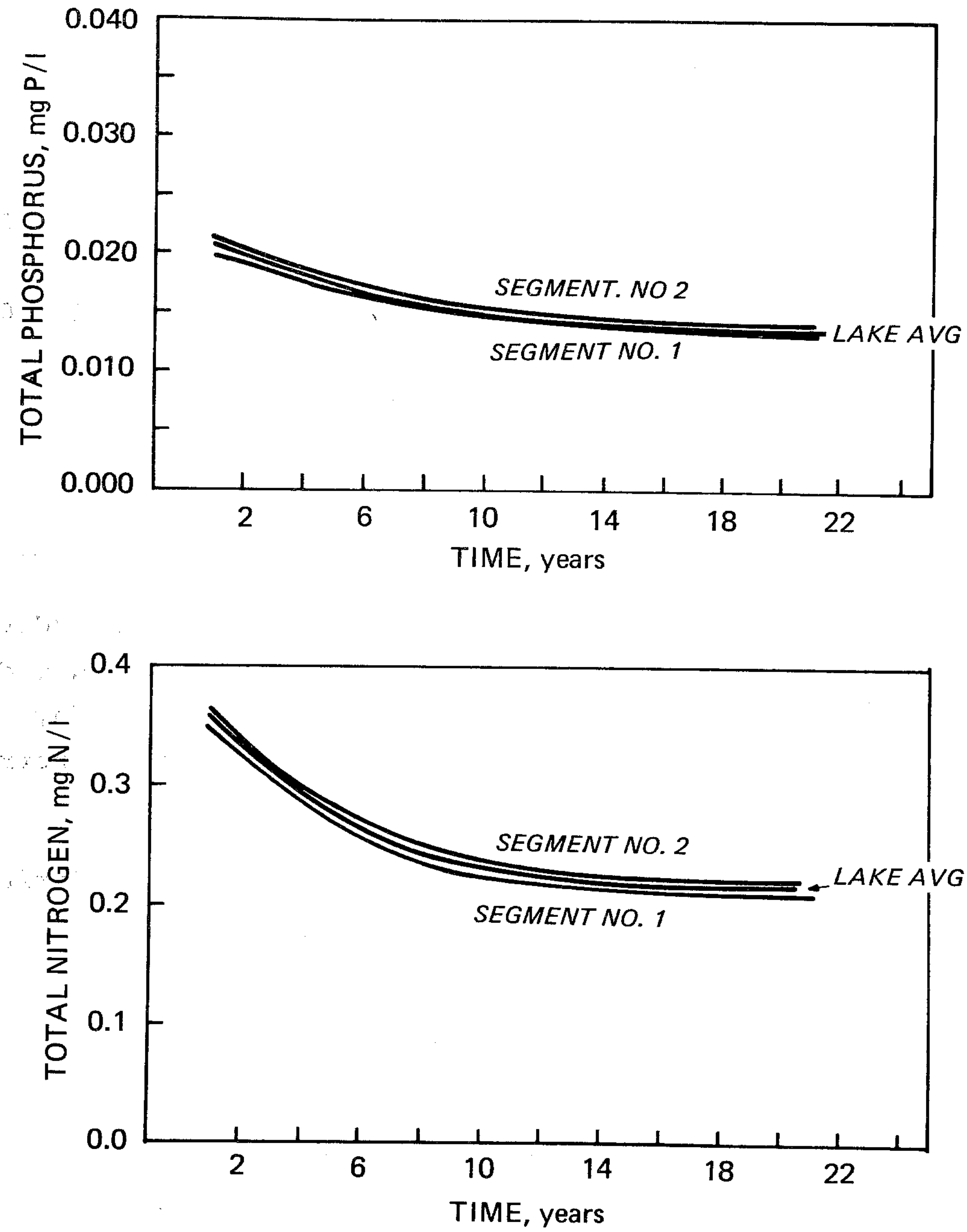

Figure 20. Yearly average change(cont.)-pastoral inputs

TABLE 8

COMPUTED STATE OF LAKE ONTARIO
UNDER "PASTORAL" CONDITIONS[1]

Variable	Max. During Year[2]	Min. During Year[2]	Annual Average 0-17m	Annual Average 17-50m	Annual Average Whole Lake
Chlorophyll "a"-μg/l	7.0	0.9	2.6	0.4	0.8
Total Inorganic Nit.-mg N/l	0.16	.025	0.10	0.17	0.16
Available Phosphorus mg P/l	0.011	0.001	.007	.012	0.011
Total Phosphorus mg P/l	0.014	0.012	.013	.014	.014
Total Nitrogen mg N/l	0.22	0.19	.21	.22	.22

[1]Pastoral Conditions - 20,500 lbs P/day (9,300 kg P/day)
-406,000 lbs N/day (184,200 kg N/day)
- Algal sinking rate - 0.1 m/day
- Decay of non-living organic nutrients-.001/day

[2]Maximum and minimum occur at different times for different variables.

Chlorophyll for the whole lake average decreases only slightly from a level of about 1.6 μg/l at present to about 1.1 μg/l under the pastoral conditions. This is a decrease of 30%, somewhat lower than the epilimnion annual average.

Overall, the simulation indicates a type of lower bound that one can use as a measure of the increase in biomass that has occurred in Lake Ontario due to increased nutrient inputs. The results indicate that the average annual phytoplankton chlorophyll in the epilimnion under the pastoral loading was about 2.6 μg/l. This level compares to an estimate of 5.7 μg/l as the annual average that is in equilibrium with the present load. It is estimated therefore that the average annual phytoplankton in the epilimnion is about twice the level that existed under some previously unstressed environment.

REDUCTION OF NUTRIENT INPUTS

A variety of simulations were carried out under different combinations of nitrogen and phosphorus inputs and under different levels of key system parameters. A complete summary is given in Section VI. Some details of two particular levels of nutrient reductions are given here because of their importance. The first level is that indicated by Vollenweider [5], referred to as the "Vollenweider" reduction and the second level is that required by the Great Lakes Water Quality Agreement[6] referred to as the "Water Quality Agreement" reduction.

Vollenweider Reduction

In his pioneering paper [5] on the relationship between external nutrient input and resulting eutrophic state of lakes, Vollenweider provided a basis for determining the allowable nutrient load to bring a lake to a more desirable eutrophic state. For Lake Ontario, an "admissable" loading from Vollenweider is about 0.4 gms/m^2-year of total phosphorus or about 47,000 lbs P/day.

This would represent a reduction of about 40% from a present input of about 75,000 lbs/day. Empirical plots similar to the original plots of Vollenweider appear to be in wide use. It is therefore of interest to determine whether the results of the dynamic simulation would agree with a projected decline in eutrophic state as estimated by Vollenweider.

Accordingly, a long term dynamic simulation was run using 47,000 lbs P/day and present nitrogen loads of 883,000 lbs N/day with the reasonable kinetics of each of the earlier runs. The results are summarized in Fig. 21 and as shown, the 40% reduction in phosphorus does not result in a concomitant reduction in phytoplankton biomass. The simulation actually indicates an increase in peak biomass to about 15 μg chlorophyll/l. The system remains phosphorus limited in the spring as shown by the minimum values of phosphorus approaching the Michaelis level, although nitrogen also has an important effect on growth. The calculations indicate therefore that a 40% reduction in phosphorus input actually results in an increase in biomass over present conditions and, from these calculations, does not result in an improvement over present conditions. This, of course, can be anticipated given the kinetics used in the simulations which indicated the Lake Ontario is not in equilibrium with the present inputs. These results indicate that the present observed condition is approximately in equilibrium with a load that is less than the load Vollenweider projected as necessary for improvement in the lake. This is a reflection of one of the hazards of the empirical plots which assume that the present observed condition of a lake is in equilibrium with the present observed input nutrient load. This does not appear to be the case for Lake Ontario.

The simulation shown in Fig. 21 was for an "immediate" reduction of 40% of total phosphorus, i.e. a step function drop in load from present loads to 47,000 lbs P/day. One would normally expect that load reductions are actually

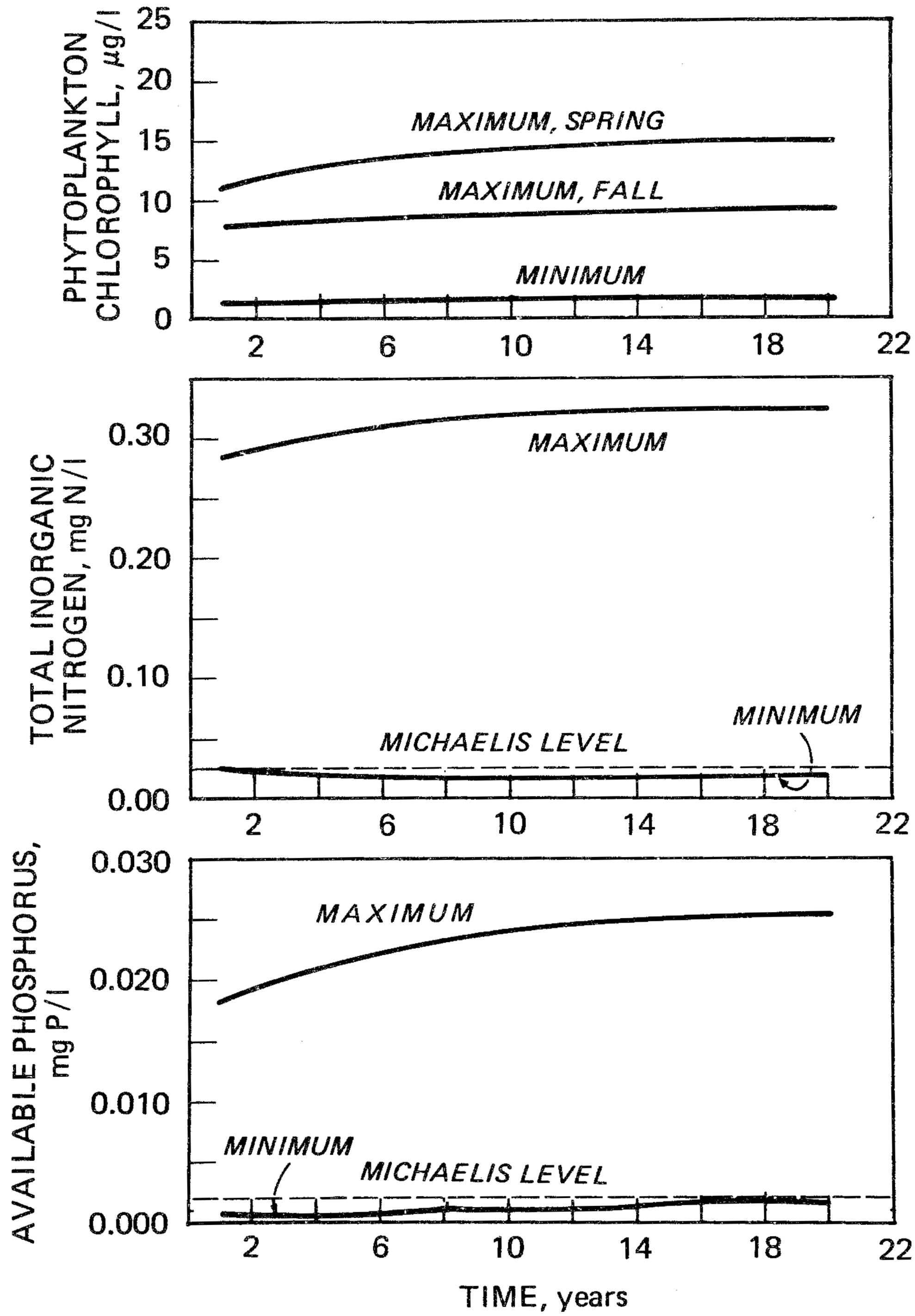

Figure 21. Dynamic behavior of phytoplankton, inorganic nitrogen and available phosphorus in epilimnion-Vollenweider reduction

accomplished over a period of time. A run was therefore prepared assuming that the reduction is accomplished linearly over a 10 year period. The results comparing the two patterns of load reduction are shown in Fig.22. Under a 10 year time interval to accomplish the 40% reduction, peak biomass would continue to increase to 16μg/l or about 45% higher than present levels and then gradually decrease to the new equilibrium value of about 15μg/l. As one would expect the time to reach a new equilibrium increases to about 25 years.

One concludes therefore from an examination of the Vollenweider reduction case, that Lake Ontario phytoplankton biomass, on a lake wide average would continue to increase and under a 10 year load reduction period would reach peak values of 15 μg chlorophyll/l. A new equilibrium level of 15 μg/l would be reached in about 25 years. The results indicate a surprising exception to the general axiom that a reduction in external loads will result in an improvement in water quality. If the hypothesis that Lake Ontario is not yet in equilibrium with the present loads is correct, then a reduction in load will not necessarily result in an improvement in water quality. The results shown in Figs. 21 and 22 illustrate this exception and indicate the importance of considering the dynamic behavior of large lakes in decisions regarding nutrient reductions.

Water Quality Agreement Loads

Because of the international importance of the phosphorus loads agreed to by the United States and Canada [6], a simulation was prepared using the agreed upon phosphorus input. Fig. 7 shows the approximate historical phosphorus inputs to Lake Ontario and the United States - Canada Agreement loads. Because the Water Quality Agreement (WQA) loads are to be accomplished over a 5 year period, the load pattern can be approximated by a step function decrease. The simulation therefore

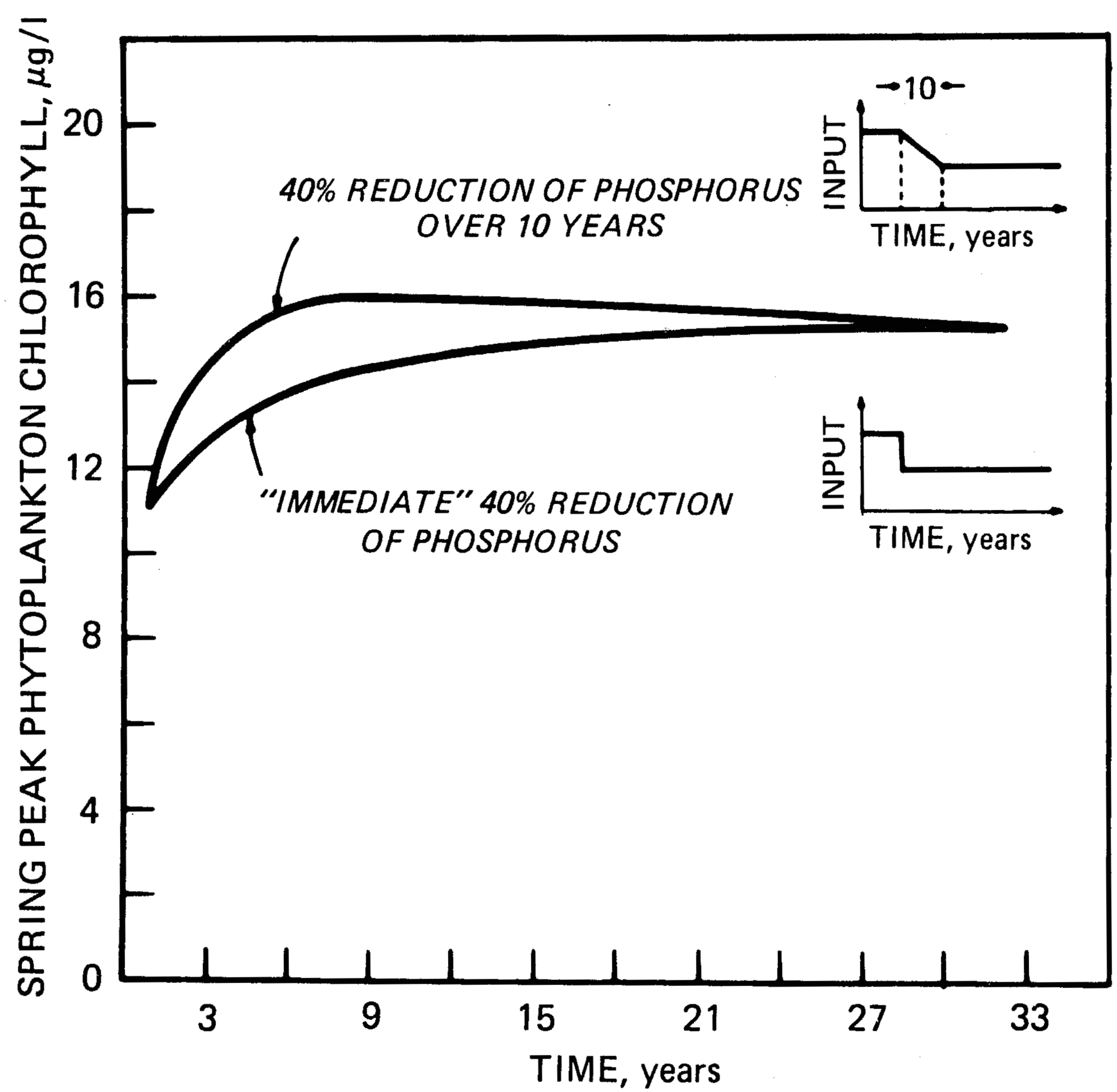

Figure 22. Comparison of "immediate" Vollenweider phosphorus reduction with a 10 year reduction period

considers an immediate drop in phosphorus load to a level of 54,800 lbs P/day or an overall reduction of less than 30%. Nitrogen loads were retained at present levels. Kinetics are as before (phytoplankton sinking velocity of 0.1 m/day and first order loss of non-living organic nutrients of .001/day).

The results are summarized in Figs. 23-25 and as one might expect from the preceding discussion, phytoplankton biomass is computed to continue to increase under the WQA loads. Fig. 23 indicates that peak values are estimated to increase to over 16 µg/l or about 50% higher than present peak levels. The system tends to be phosphorus limited. The yearly average changes shown in Figs. 24 and 25 indicate the relative insensitivity of whole lake annual averages as a measure of response. The epilimnion average annual responses indicates a time to equilibrium of about 10 years, yet peak biomass reaches equilibrium in about 15 years.

These calculations indicate that under the WQA loads, the phytoplankton biomass of the open lake may continue to increase for about a period of 15 years or until the late 1980's. The 27% reduction in phosphorus would result in only about a 6% reduction in peak phytoplankton at equilibrium. This is not meant to imply that the WQA program is not a good one. These computations indicate that the hopes for an expected response of Lake Ontario may not be as high as anticipated under the Agreement.

Comparison to Empirical Loading Plots

As indicated previously, one of the classical works in lake eutrophication and the effects of nutrient loadings is that of Vollenweider [5]. His work represented one of the first synthesis of water quality data related to accelerated eutrophication of lakes, with external sources of nutrients, due to such inputs as "natural" runoff, agricultural runoff and point sources of municipal and industrial wastes. An appeal of Vollenweider's

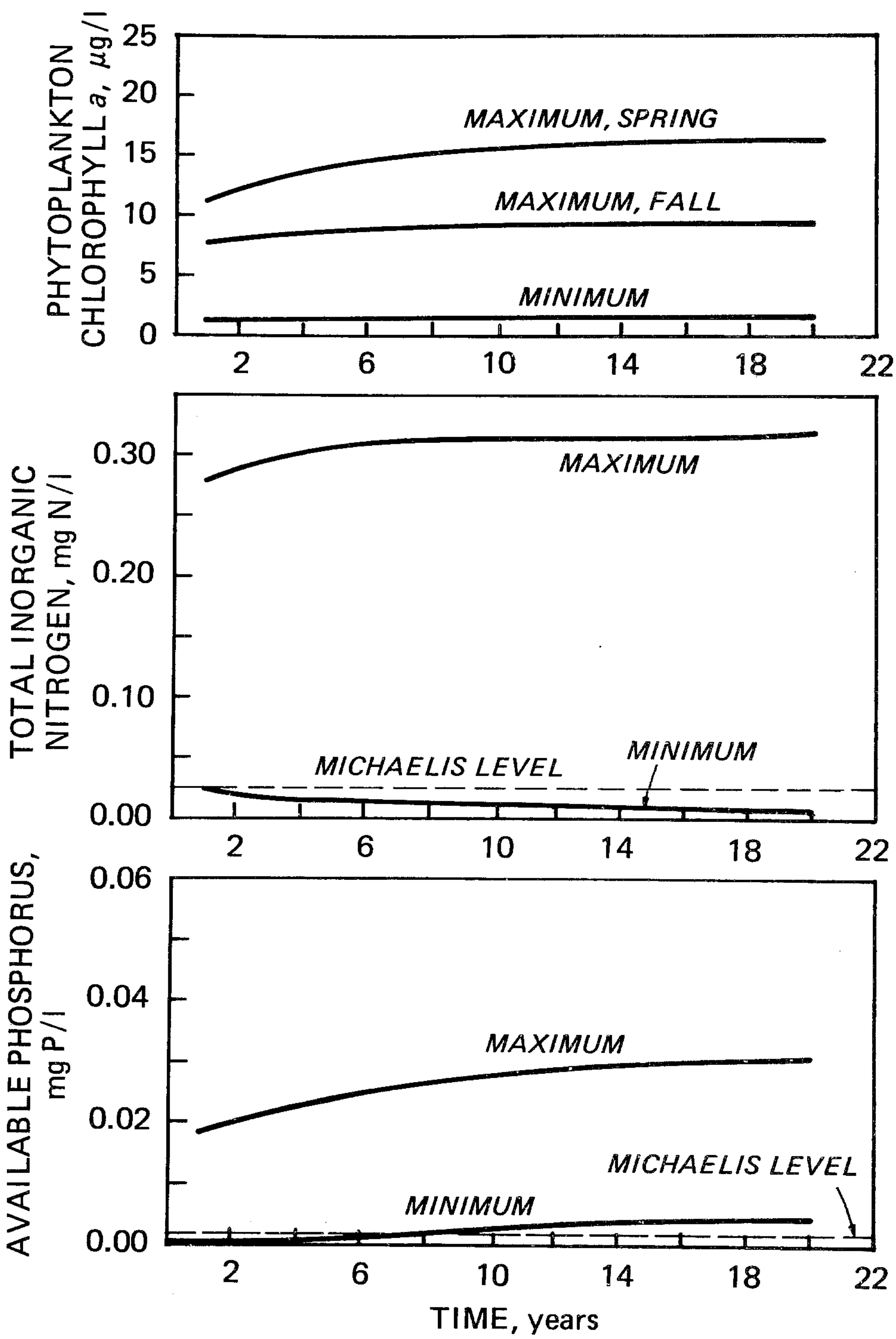

Figure 23. Dynamic behavior of phytoplankton, inorganic nitrogen and available phosphorus in epilimnion-Water Quality Agreement reduction

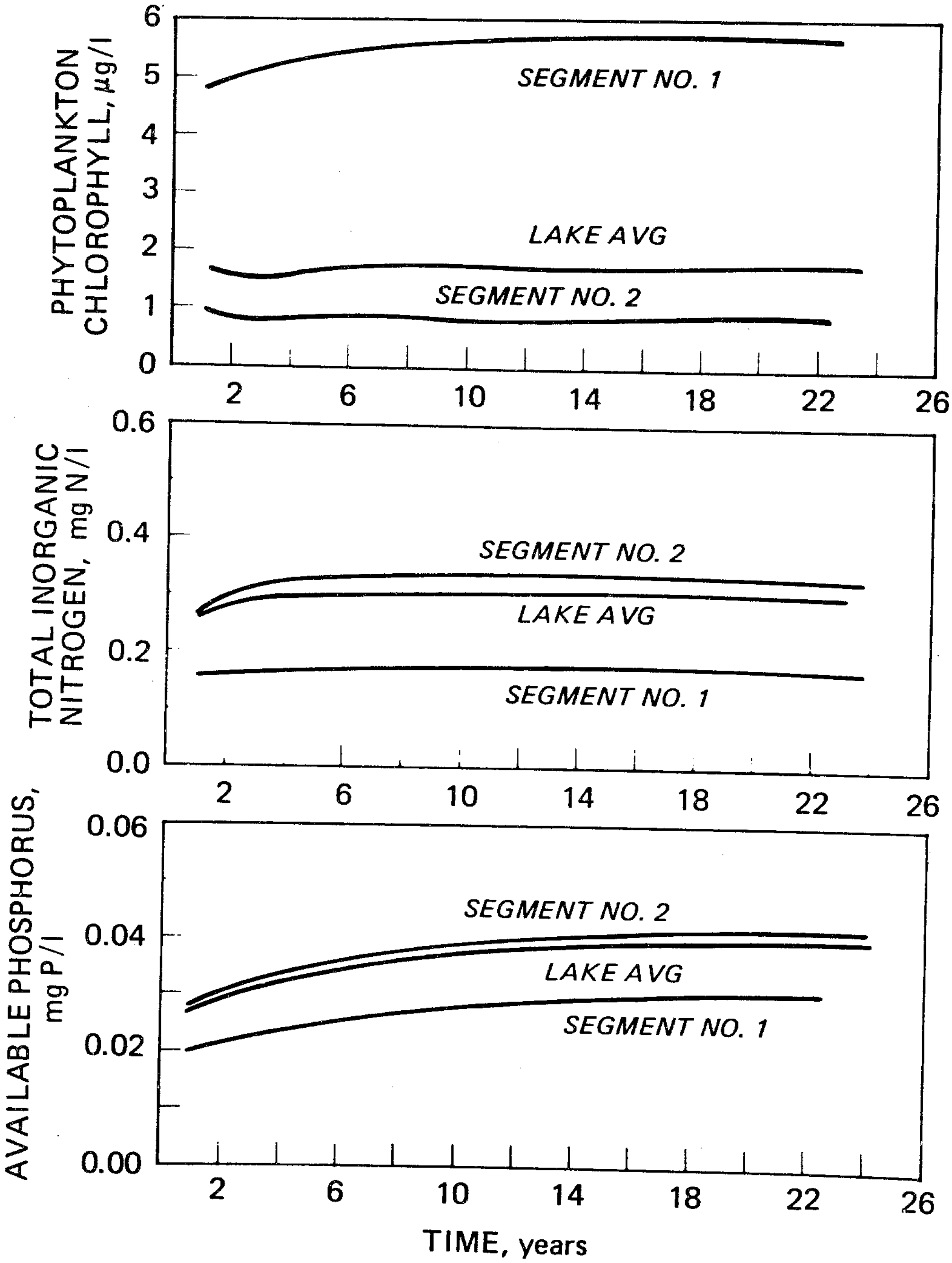

Figure 24 Yearly average changes - Water Quality Agreement loads

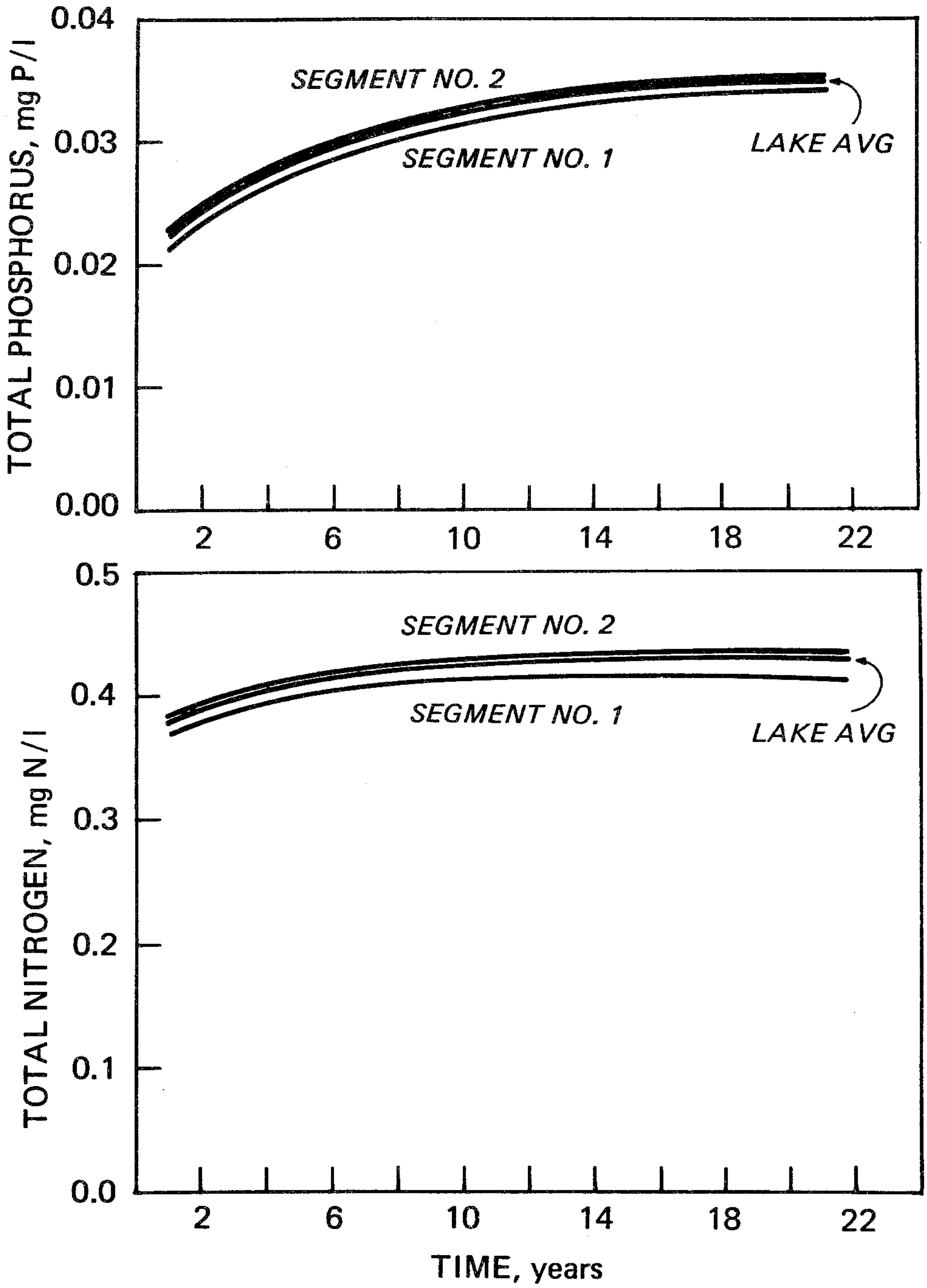

Figure 25. Yearly average changes (cont.)-
Water Quality Agreement loads

analysis is its simplicity. There was very little available to environmental managers of lake water quality that linked external loadings (which are controllable in various degrees) to a measure of eutrophication. The graphical plot of nutrient loading to the lake such as phosphorus in grams/m^2 - year as a function of mean depth of the lake with a general division into eutrophic or oligotrophic lakes provided a basis for decision making. For a given depth of the lake, the "admissable" loading can be read directly from the plot. This in turn can be translated into treatment requirements. Indeed, for the Great Lakes system, the analysis of Vollenweider presumably formed an important input into the Great Lakes Water Quality Agreement [6] and also for nutrient controls in other lakes of the world.

Others such as Dillon and Rigler [11] and Bachman and Jones [12] have also attempted to relate various measures of eutrophication to loading rate or nutrient concentrations. Dillon [13] has summarized and critiqued nutrient budget models extending from 1963 to 1974. All of the models to date, including a detailed analysis by Vollenweider [14] deal with a nutrient such as total phosphorus as the starting point. Inferences are then drawn from the models as to relative degrees of eutrophication. In addition to these models, various empirical plots, as noted previously, have been developed to relate loading to response or nutrients in the lake to phytoplankton biomass. There are at least three major shortcomings to these attempts at exploring observed behavior: a) the empirical plots are non-dynamic and assume a one to one correlation between observed nutrient and/or biomass concentrations and input loading, b) as a corollary, the models and plots do not directly relate nutrient loading to the resulting plant biomass and, c) the models and plots do not explore the interactions between two or more nutrients. Further, the models generally assume a completely mixed lake or attempt in a minimal way to include stratification.

The dynamic model of phytoplankton behavior [1] which forms the basis for the simulations reported herein provides a means for displaying the relationship between loading and nutrient plots and a dynamic mathematical modeling framework. The hope is that each of the approaches can be demonstrated to result from assumptions made on the more generalized dynamic phytoplankton model. In this way, the underlying unity and direction will be more apparent. While the assumption of a well mixed lake is quite restrictive and not necessary, it is used here in order to provide some comparability between the analyses.

For a completely mixed lake, the following equations may be used to represent the dynamics of the phytoplankton: [1]

$$V \frac{dP}{dt} = V\ (G\ \ -\ D\ )\ P - V\ K_s\ P - QP \qquad (5)$$

$$V \frac{dZ}{dt} = V\ (G_Z - D_Z)\ Z - QZ \qquad (6)$$

$$V \frac{dp_1}{dt} = VS_1 - Qp_1 - K_{12}\ p_1 + W_1 \qquad (7)$$

$$V \frac{dp_2}{dt} = K_{12}p_1 - Qp_2 - a_{pp}G_pP + W_2 \qquad (8)$$

where P = phytoplankton chlorophyll (μg/l), G_p and D_p = phytoplankton growth and death rate respectively, (l/day) K_s = phytoplankton sinking rate (l/day) which incorporates the sinking velocity, vertical dispersion effects and depth of water, Z = zooplankton carbon (mg/l), G_Z and D_Z represent growth and death rate of zooplankton (l/day), p_1 = dissolved and detrital organic phosphorus (mg/l), S_1 = overall source of organic phosphorus due to plankton respiration and excretion (mg/l-day), p_2 = inorganic available phosphorus (mg/l), K_{12} = decomposition of organic phosphorus (l/day), a_{PP} is the phosphorus chlorophyll ratio, W_1 and W_2 represent external loadings of organic and inorganic phosphorus respectively (kg/day) and V, (km^3), Q (m^3/sec) are the lake volume, and outflow respectively. For simplicity only one nutrient has been considered here, the lake is not assumed to stratify and the only loss of nutrient is through phytoplankton settling, assumptions which can be quite important. Further, it should be recognized that all growth, death and predation terms are complicated non-linear functions of such factors as nutrients, light, temperature and grazing rates. Details of the full lake model are given in the earlier report on the Lake Ontario model [1].

If total phosphorus is used as the relevant nutrient variable, then it can be shown that summing Equations (5) and (8) and using phosphorus equivalents of all plankton gives

$$V \frac{dp_t}{dt} = - Qp_t - VK_s P + W_t \qquad (9)$$

where p_t is the total phosphorus concentration and W_t is the total phosphorus external loading.

Note that one of the forcing functions for the total phosphorus is the phosphorus equivalent of the sinking phytoplankton biomass. As such, strictly speaking, Eq. (9) is not solvable since P is a variable. However, keeping P constant temporarily, and combining with W_t as a sink of phosphorus, the solution to Eq. (9) is simply,

$$p_t = \frac{W_t - K_s VP}{Q} \left(1 - \exp\left(-\frac{t}{t_o}\right)\right) + (p_t)_o \exp\left(-\frac{t}{t_o}\right) \qquad (10)$$

where $(p_t)_o$ is the initial concentration of total phosphorus and t_o is the lake detention time $(=\frac{V}{Q})$. Eq. (10) shows that the time to a new steady state for a constant input is reached in about 2-3 times the detention time of the lake. For Lake Ontario, this would be about 16-24 years. However, the effect of the sinking biomass and losses of organic nutrient fraction serves to reduce that time to perhaps 10-15 years based on the previously discussed long term simulation runs.

This effect of the sinking biomass can also be seen by letting the phytoplankton biomass be some fraction of the total phosphorus, i.e.,

$$P = \alpha \, p_t \tag{11}$$

and then the solution is

$$p_t = \frac{W_t}{Q} \left(1 - \exp - (\alpha K_s + 1/t_o)\, t\right) + (p_t)_o \exp(-t/t_o) \tag{12}$$

The exponent is therefore increased by αK_s and therefore the time to reach a new steady state is decreased. For Lake Ontario, and a continuation of present loads (Figs. 11 and 12), P is about 6 μg/l, p_t is about 50 μg/l (at equilibrium) and K_s is estimated at 0.1 m/day ÷ 90 m and αK_s is about 0.00013/day. For an eight year detention time, $1/t_o$ is .00034/day and the value of the exponent is therefore about .00047/day or an equivalent response time of 5.8 years. The loss of phosphorus through the sinking biomass therefore reduces the response time by about 2.3 years. The difficulty, of course, in any practical problem is that the ratio P/p_t is exactly the variable that must be projected under a different loading regime and hence is not explicity known.

Therefore, under constant loadings in time, presently observed total phosphorus would be correlated with waste loadings some years earlier. But, loadings to Lake Ontario have, of course, not been constant in time

(Fig. 7) so that the presently observed total phosphorus is correlated with some overall average loading over approximately the past few decades.

This simple dynamic analysis on a total nutrient such as phosphorus serves to illustrate the uncertainty underlying plots of loading rate and nutrient for the same year. This was also shown in the simulation using the Vollenweider reduction which indicated that biomass would continue to increase even after the reductions were accomplished.

Another difficulty is that the relationship between biomass and the relevant total nutrient is confounded in the empirical plots. That is, the direct computation of phytoplankton biomass under different external nutrient loading is not possible from equations that deal only with the total nutrient concentration in the lake. This can be seen by the results summarized in the preceding discussion (see especially Figs. 11 and 12 and 24 and 25.) Nevertheless, accepting the notion that one is always seeking a simple representation of complex phenomena, the steady state assumption in Eq. (9) can be made, giving

$$0 = Qp_t - VK_sP + W_t$$

or $$W_t = Qp_t + VK_sP \tag{13}$$

Dividing through by the surface area of the lake and recognizing that Q/A is H/t_o, where H is the average depth of the lake, one obtains

$$\frac{W_t}{A} = \frac{H}{t_o} p_t + K_sHP \tag{14}$$

The input loading is now an areal rate (gms/m^2-year).

Taking logarithms of both sides of (14) gives

$$\log \left(\frac{W_t}{A}\right) = \log (p_t/t_o + K_s P) + \log H \tag{15}$$

A comparison of Eq. (15) to the plot of Vollenweider [5] indicates that the slope of the latter should be unity (for the model used here), and that the intercept is a function of the total phosphorus concentration, the detention time and the phytoplankton biomass and sinking rate.

The intercept, therefore, of a plot of log H versus log W_t/A is a variable depending on a complex interaction of biomass and total phosphorus concentration. Vollenweider [5] of course, recognized the fact that biomass and the total phosphorus are related and provides an excellent qualitative discussion of these interactions (5. p. 78ff.). Eq. (15) shows quantitatively that the more eutrophic lakes would generally have larger values of the intercept, thus providing some basis for the division originally made by Vollenweider. The intercept also indicates that it is not possible, in general, to predict the biomass level for a change in W_t/A due to the confounding of the biomass level with the total in-lake phosphorus concentration.

Only if it is assumed that $P = \alpha\, p_t$ as in Eq. (11) then:

$$\log \frac{W_t}{A} = \log \left(P\left(\frac{1}{\alpha t_o} + K_s\right)\right) + \log H \tag{16}$$

which permits a direct relationship between biomass and external loading for constant α. The difficulty with Eq. (16) is the necessity to specify the fraction, α which may vary under different eutrophic states. For example, Table 11 (page 76) shows that α did indeed vary significantly for the simulations carried out in this report.

Recently, Vollenweider and Dillon [15] have reexpressed the loading versus depth plot to include the detention time and have suggested a plot of loading against depth divided by the detention time. It can be noted from Eq. (14) that this plotting is not possible without again confounding the detention time in the intercept, i.e.,

$$\frac{W_t}{A} = p_t + K_s t_o P \frac{H}{t_o} \qquad (17)$$

$$\text{or} \quad \log W_t/A = \log (p_t + K_s t_o P) + \log \frac{H}{t_o} \qquad (18)$$

As shown, the detention time also appears in the intercept as well as the independent variable which explains the "bending" of the lines using Eq. (18). However, whether Eq. (18) or Eq. (15) (or other variations thereof) are used, the phytoplankton biomass is still incorporated in the intercept and therefore not directly predictable.

Other formulations have therefore attempted to incorporate the phytoplankton biomass as a plotting variable. For example, Dillon and Rigler [11] have explored log of the summer phytoplankton biomass versus the logarithm of the spring total phosphorus concentration. Because of the specified time periods such a plot is a representation of Eqs. (5)-(8) in a complex way and is strictly an empirical relationship not readily derivable from Eqs. (5)-(8). However, some insight into the predictive capability of such plots can be obtained by rearranging Eq. (14). Therefore,

$$P = \frac{1}{K_s} \left(\frac{W_t}{V} - p_t/t_o\right) \qquad (19)$$

or taking logarithms,

$$\log P = - \log K_s + \log \left(\frac{W_t}{V} - p_t/t_o\right) \qquad (20)$$

which indicates that the abscissa includes both the external loading and within-lake total phosphorus concentration. Bachman and Jones [12] have used Eq. (20) but plotted log P versus log W_t/V. The dilemma of trying to predict biomass response is made clear by Eq. (20).

Due to the interaction of biomass, nutrient concentration in the lake and external mass nutrient loading, it is not possible to directly estimate the response since a reduction in loading requires an estimate of the new lake total phosphorus which in turn requires an estimate of the new phytoplankton biomass. Only by simultaneous solution of a set of equations such as Eqs. (5)-(8) can such predictions be directly made.

Of course, the assumption of Eq. (11) can again be made to give

$$\log P = - \log H \left(\frac{1}{\alpha t_o} + K_s\right) + \log W_t/A \qquad (21)$$

which as before requires an a priori specification of α. Eq. (21) however would appear to be a useful equation to use for plotting purposes but may be risky to use for prediction purposes.

One concludes, therefore, that simplified plots of loading rate versus lake geometry and flushing rates or plots of a presumed limiting nutrient and biomass while interesting in describing the general trends in lakes, are too crude a level of analysis to premit meaningful statements to be made about the effects of reduction in waste load on phytoplankton biomass. Indeed, the one comparison between a dynamic model and the loading plot of Vollenweider [5] as indicated above produces results in conflict with the projection from the loading plot. Such plots will always show an implied improvement in lake status as load is decreased, yet the dynamic results show that if the lake is not in equilibrium (such as is hypothesized for Lake Ontario) then the projections from the empirical plots may be significantly in error. Such plots may be of some use for short detention time lakes which reach equilibrium quickly. However, one would conclude from the results presented in this Section that empirical loading plots are not appropriate for large lakes such as the Great Lakes.

SECTION VI

SUMMARY OF LAKE RESPONSES TO NUTRIENT INPUTS

The preceding results indicated a variety of responses of Lake Ontario to external nutrient inputs. A summary and comparision of the responses is given in this Section together with the results of simulations over a complete range of nutrient loads.

SENSITIVITY OF PRINCIPAL SIMULATIONS

Table 9 shows a summary of the principal load simulations which highlight key points of departure in various proposed load reductions. The results of the preceding Section indicated the change in peak spring phytoplankton chlorophyll under a long term kinetic structure that included a decay of the non-living organic nutrient forms of 0.001/day. The most pessimistic assumption with respect to phytoplankton changes would be to assume that the organic fraction (as well as the inorganic fraction) of the nutrients is conserved. Under that kinetic assumption then, the only loss of nutrient from the system is from sinking phytoplankton which, in this model, are eliminated from the system when the biomass reaches the sediment.

The most optimistic assumption would be to assume that the Lake is presently in equilibrium with respect to the present nutrient loads, especially the phosphorus load. As indicated previously, this implies some loss of inorganic phosphorus through a mechanism of chemical precipitation. While there is considerable uncertainty over the potential effect of this mechanism, the possibility of a sink of inorganic phosphorus does exist. Therefore, a series of runs were prepared for the load conditions of Table 9 and incorporating the most optimistic kinetics, most pessimistic kinetics and the "reasonable" set of kinetics used in the last chapter. The results are shown in Fig. 26.

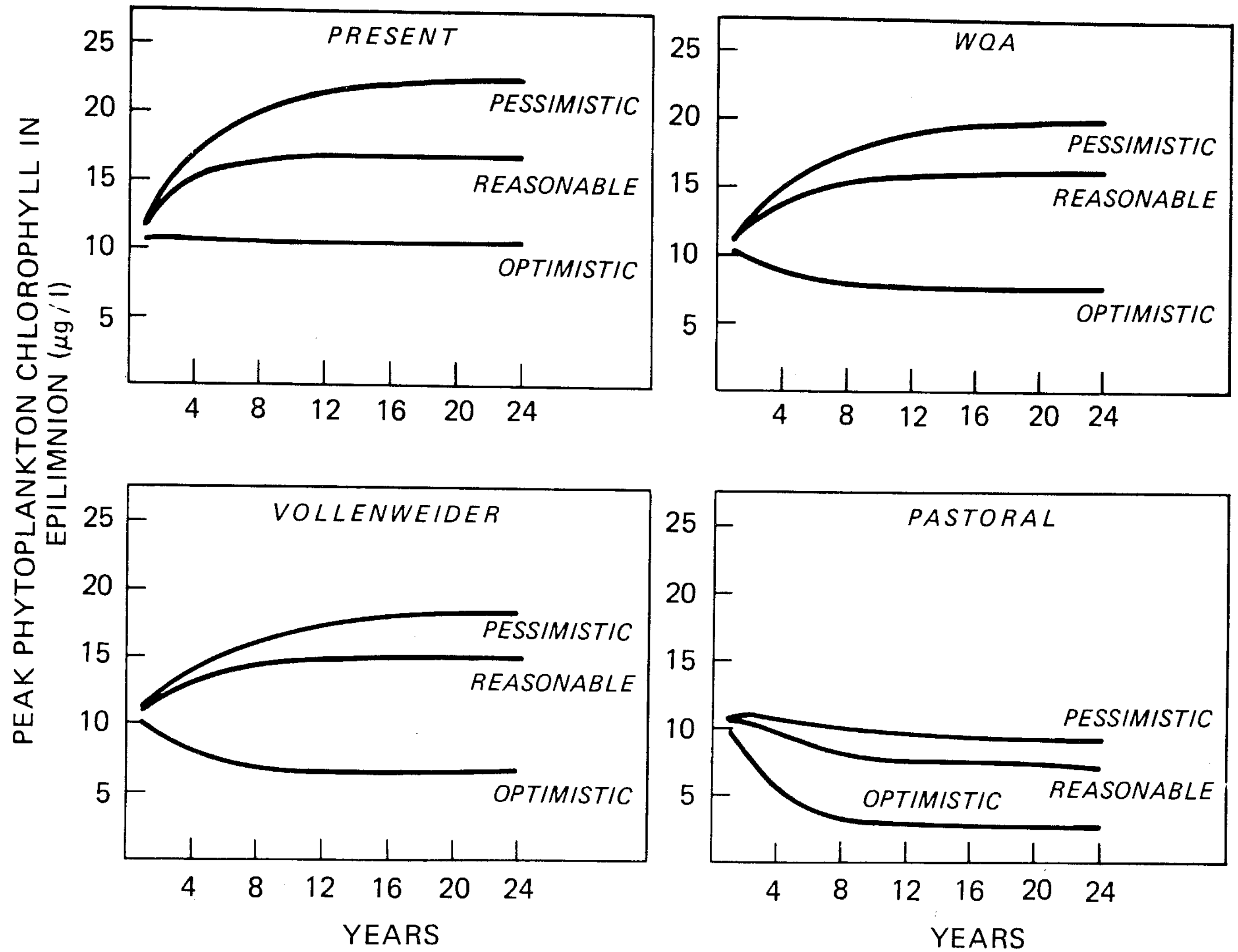

Figure 26 Sensitivity of phytoplankton response to kinetic assumptions

As one would expect, the responses vary widely over the range of assumed conditions. Under the most optimistic setting, the response of the Lake is immediate (a direct consequence of the equilibrium assumption) and positive as contrasted to the reasonable set of kinetics. However, while the optimistic setting is possible, the degree to which the inorganic phosphorus is removed from the Lake is quite uncertain, at the present time. Therefore, it appears more reasonable that such removal by chemical precipitation is not occurring in any substantial amount (to say, within ±10-20% of the incoming load) and that the primary removal mechanism is by settling of particulate phosphorus forms. The remainder of this chapter therefore, summarizes the results of the simulations under the "reasonable" set of kinetics used earlier.

SUMMARY OF SIMULATIONS USING REASONABLE KINETICS

Fig. 27 is a summary plot of the principal simulation conditions, each under the reasonable kinetic assumptions. The most obvious result in Fig. 27 is the relatively small change between the continuation of present loads and the WQA loads which indicates that it may be difficult to detect any substantive change under the WQA. It should be stressed again however, that all the results shown and discussed previously are for the Lake 1 model which assumes a horizontally well-mixed lake. The results in Fig 27 are therefore for lake wide conditions in the epilimnion. Near shore responses may be quite different. Further, as previously indicated in Fig. 26, peak values may reach as high as 22 μg chlorophyll/l under the pessimistic assumption and a continuation of present loads. The implementation of the WQA loads would reduce this "worst" case to about 20 μg/l. Under the reasonable kinetics of Fig. 27 that include organic decay, the change occasioned by the WQA would be about 1 μg/l.

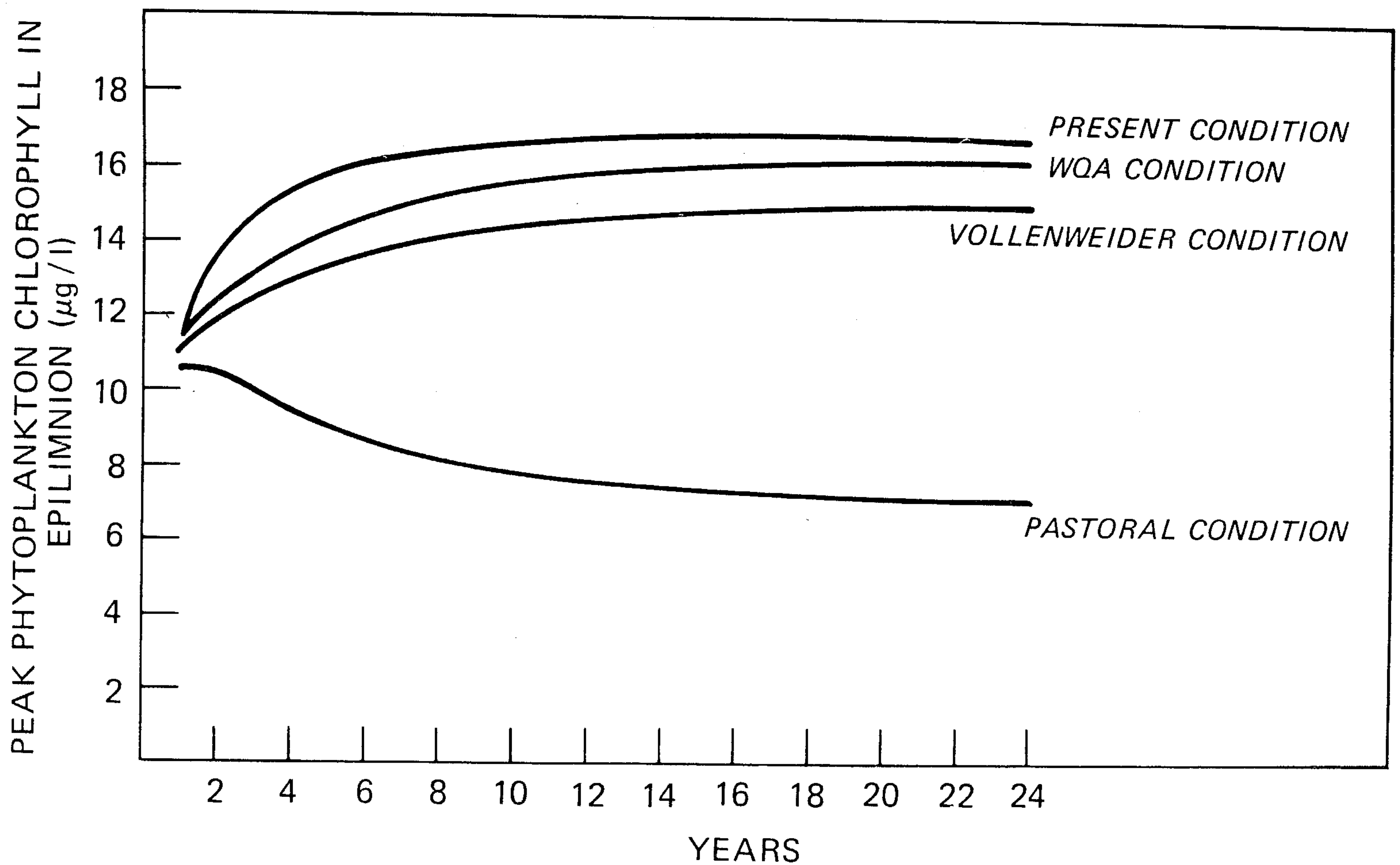

Figure 27 Summary of peak phytoplankton response. Reasonable kinetics: phytoplankton settling = 0.1 m/day, K organic = 0.001/day, K inorganic = 0.0

TABLE 9

SUMMARY OF PRINCIPAL

LOAD CONDITIONS

Simulation Condition	Total Phosphorus lbs/day	Metric tons/day	% of present	Total Nitrogen lbs/day	Metric tons/day	% of present
Continuation of Present Load	75,000	34.0	100	883,000	400.4	100
Water Quality Agreement Load	54,800	24.8	73	883,000	400.4	100
Vollenweider Reduction	46,900	21.3	63	883,000	400.4	100
Pastoral Loads	20,500	9.3	27	405,900	184.1	46

Table 10 is a further elaboration of the system responses for the reasonable kinetics. The relatively small impact of the WQA loads are well as the Vollenweider reduction case can be seen. In fact, the model computations indicate that at the very best, one could expect about a 40% reduction from present phytoplankton levels.

A further breakdown of the nutrient components at equilibrium in the epilimnion is given in Table 11 and plotted in Fig. 28. Reasonable kinetics again apply. The most interesting feature of the response is the relative insensitivity of the average annual response in phytoplankton over a range of phosphorus reduction. Furthermore, one of the important ratios in the empirical annual average plots as discussed in Section V and given in Eq. (16) is the ratio of phytoplankton biomass to total nutrient. Table 11 shows this ratio to increase from .12 to .20 and Fig. 28 shows the continued increase in total phosphorus but without an accompanying increase in phytoplankton chlorophyll. A constant assumption on the ratio, α, as required by Eq. (16) for use in the empirical plots does not appear to be a viable assumption for Lake Ontario, at least when compared to the computed results of Lake 1. The reason the ratio changes, is of course due to the fact that the Lake is estimated to be progressively more nitrogen limited as one increases the phosphorus load. This is further illustrated in Fig. 29 which shows the phytoplankton biomass under difficult levels of phosphorus loading. As shown, for a range of phosphorus load from present loads of about 75,000 lbs P/day to about 45,000 lbs P/day, the chlorophyll level is relatively insensitive. At further reductions, the slope is approximately linear. These results indicate the hazards, and uncertainty of simple models of lake nutrients and phytoplankton biomass. In the case of Lake Ontario, two factors seem to preclude the use of simplified models, a) the relatively long detention time of the lake and b) the interaction of both nitrogen and phosphorus as important growth limiting nutrients.

TABLE 10

ESTIMATED CHANGE IN PHYTOPLANKTON BIOMASS UNDER DIFFERENT SIMULATION CONDITIONS

Simulation Condition (1)	Ratio: Simulation / Present			
	Spring Peak (2)	Annual Average (3)		
		0-17m	17-90m	Lake Average
Continuation of Present Loads	1.55	1.25	1.05	1.15
Water Quality Agreement Loads	1.50	1.20	1.00	1.10
Vollenweider Reduction	1.35	1.20	0.95	1.05
Pastoral Loads	.65	.55	.5	.5

(1) See Table 9 for loads for each condition. K organic = .001/day.

(2) Present peak assumed at 11 µg/l.

(3) Present annual average assumed at 4.7 µg/l (0-17m); 0.9 µg/l (17-90m) and 1.6 µg/l (lake average)

TABLE 11

COMPUTED DISTRIBUTION OF PHOSPHORUS AND NITROGEN COMPONENTS AT EQUILIBRIUM

Annual Average - Segment #1 (0-17 m)

Simulation Condition (1)		Total Nutrient conc. μg/l	Equivalent Phytoplankton		Equivalent Zooplankton		Non-Living Organic Forms		Inorganic Nutrient	
			μg/l	% (2)	μg/l	%	μg/l	%	μg/l	%
Continuation of Present Loads	P	50	5.8	12	1.7	3	5.9	12	36.6	73
	N	408	58.	14	17.	4	168.	41	165	41
Water Quality Agreement Loads	P	35	5.7	17	1.7	5	5.4	15	22	63
	N	415	57.	14	17	4	167.	40	174	42
Vollenweider Reduction	P	29	5.6	19	1.7	6	5.2	18	16.5	57
	N	421	56.	13	17	4	165.	39.0	183	44
Pastoral Loads	P	13	2.6	20	1.0	8	2.4	18.0	6.9	54
	N	205	26.	13	10.	5	70.	34.	99	48

(1) See Table 9 for loads for each condition

(2) Percent of total nutrient concentration

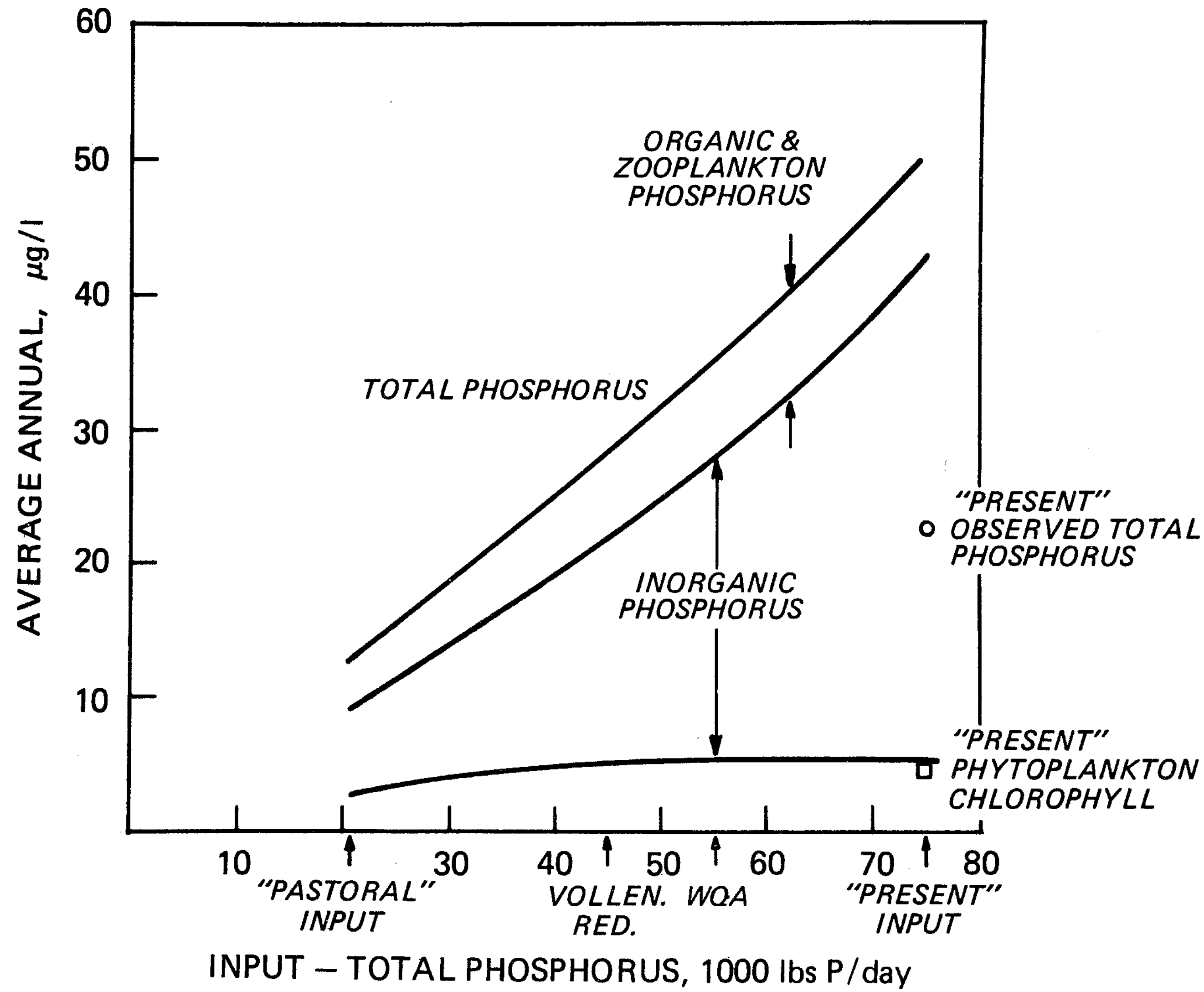

Figure 28 Summary of phosphorus components computed at equilibrium, 0-17m, nitrogen loads as in Table 9, reasonable kinetics

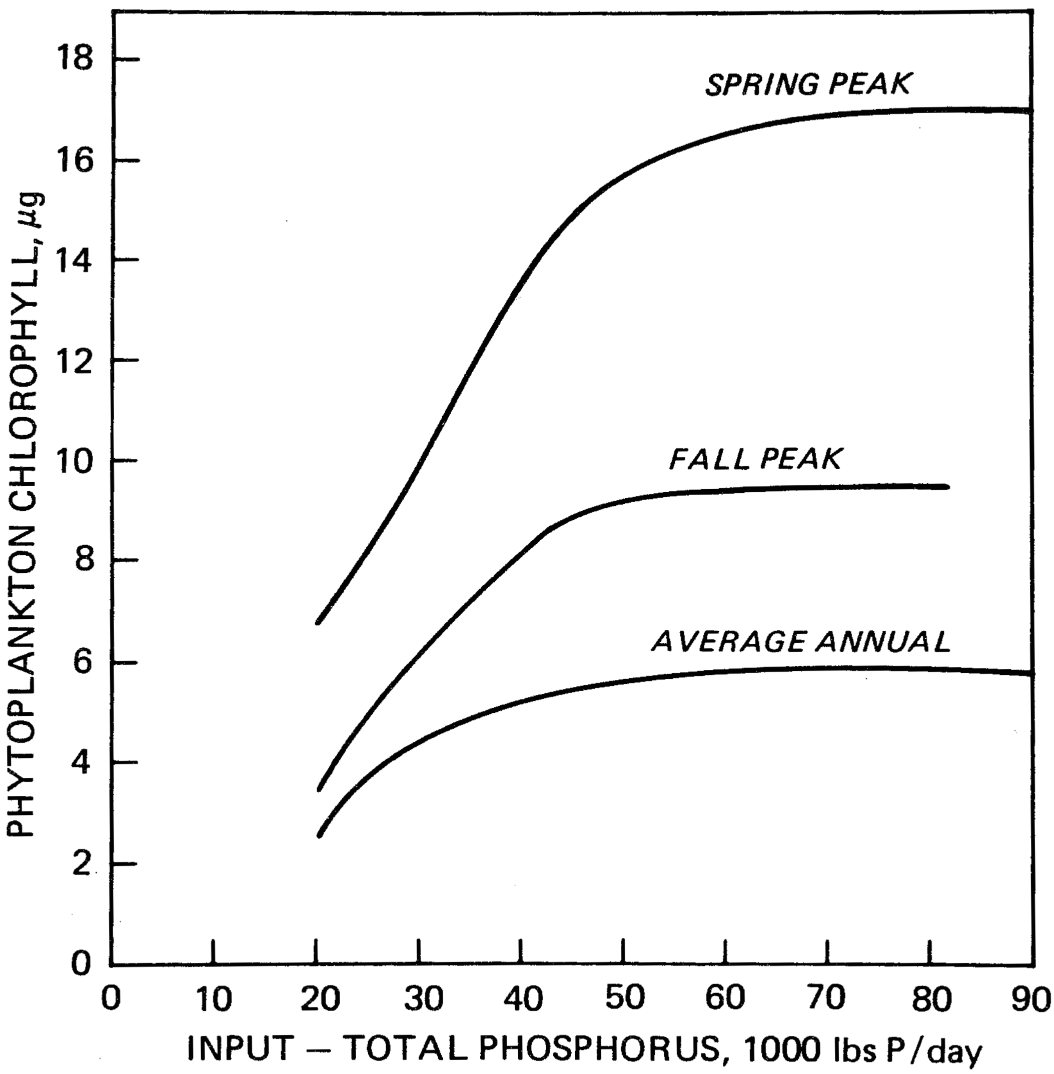

Figure 29 Spring and Fall peaks and annual average chlorophyll at equilibrium in epilimnion, reasonable kinetics

SUMMARY FOR RANGE OF NUTRIENT LOADS

In order to explore the full behavior of the Lake 1 model response, a complete series of simulations was conducted over a range of nutrient inputs under two kinetic assumptions on the organic decay coefficient, representing the pessimistic and reasonable cases. Therefore, inorganic forms of the nutrient are assumed conserved under the two assumptions. The results are shown in Figs. 30-32 and permit a first estimate to be made of the open Lake response for 0-17 meters under different combinations of nitrogen and phosphorus loading. The shape of the functional relationship in Figs. 30 - 32 is particularly interesting and shows the regions where either phosphorus or nitrogen is limiting. For example, if nitrogen loadings are kept at present levels, the model will become more and more phosphorus limited as the phosphorus load is reduced. On the other hand, if nitrogen levels are at present levels and phosphorus loads continue to increase, the simulations indicate that nitrogen will become progressively more limiting and there will be no increase in peak biomass even though phosphorus loads are increasing.

Plots such as Figs. 30 - 32 are useful for assessing effects of uncertainty in load estimates as discussed in Section IV. Effects of sediment releases of nutrient can be estimated by simply adding the net flux of nutrient to the external load. Figs. 30 - 32 show, in this regard that if the sediment input is approximately 10% of the external load[10], the phytoplankton response is relatively insensitive to this source except in the region where either nutrient may be limiting.

IMPLICATIONS FOR DECISION MAKING

One can also determine from Fig. 31 that in order to maintain present peak phytoplankton chlorophyll at about 11 μg/l, the total phosphorus loading must be reduced to about 35,000 lbs P/day or about a 53% reduction of present inputs. But, since it is estimated that pastoral phosphorus loads were about 20,500 lbs P/day, the total load of 35,000 lbs P/day represents actually about a 73% reduction of the total nut-

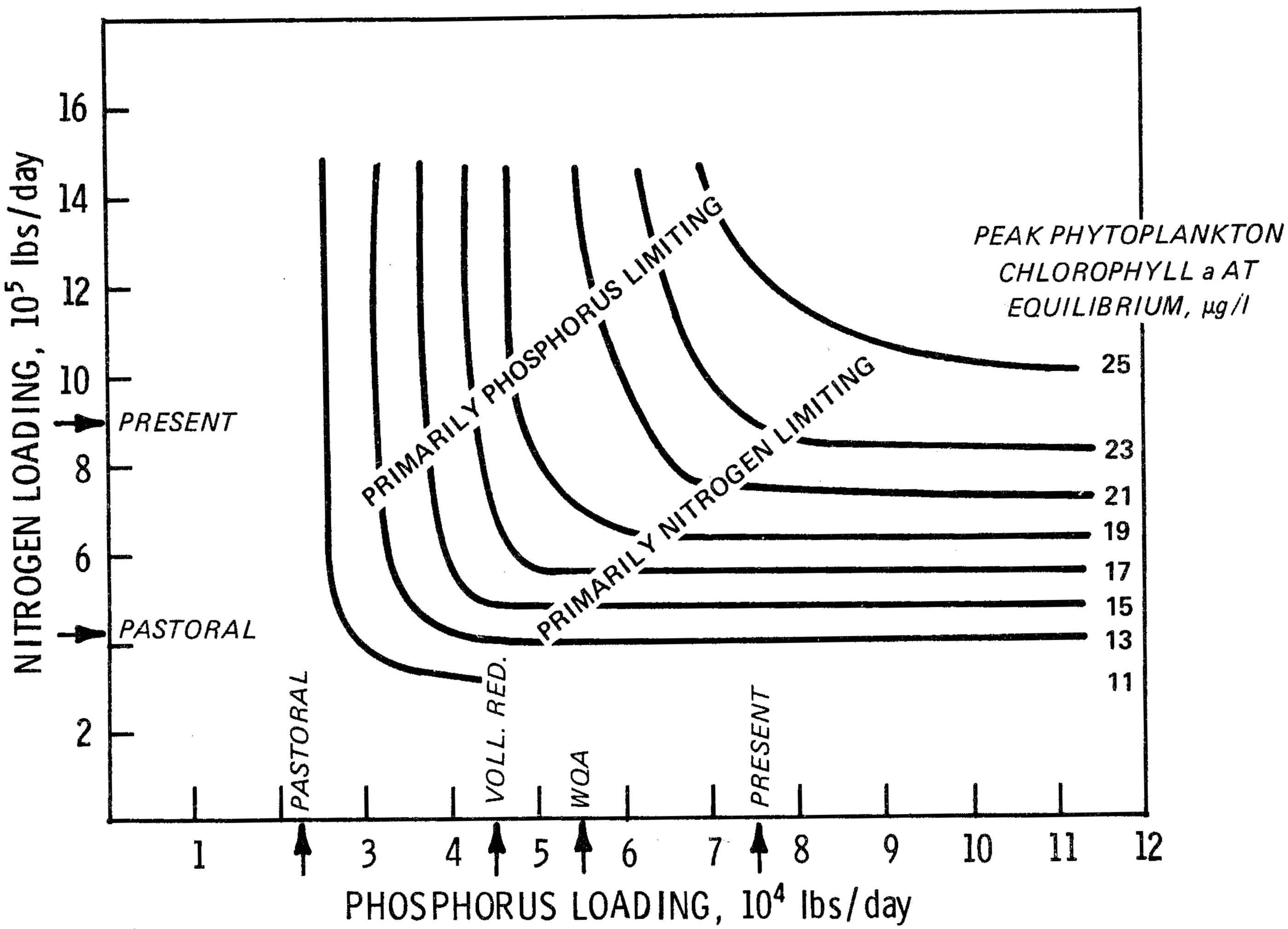

Figure 30 Peak phytoplankton chlorophyll, 0-17m, as a function of nitrogen and phosphorus input - K organic = 0.0, K inorganic = 0.0

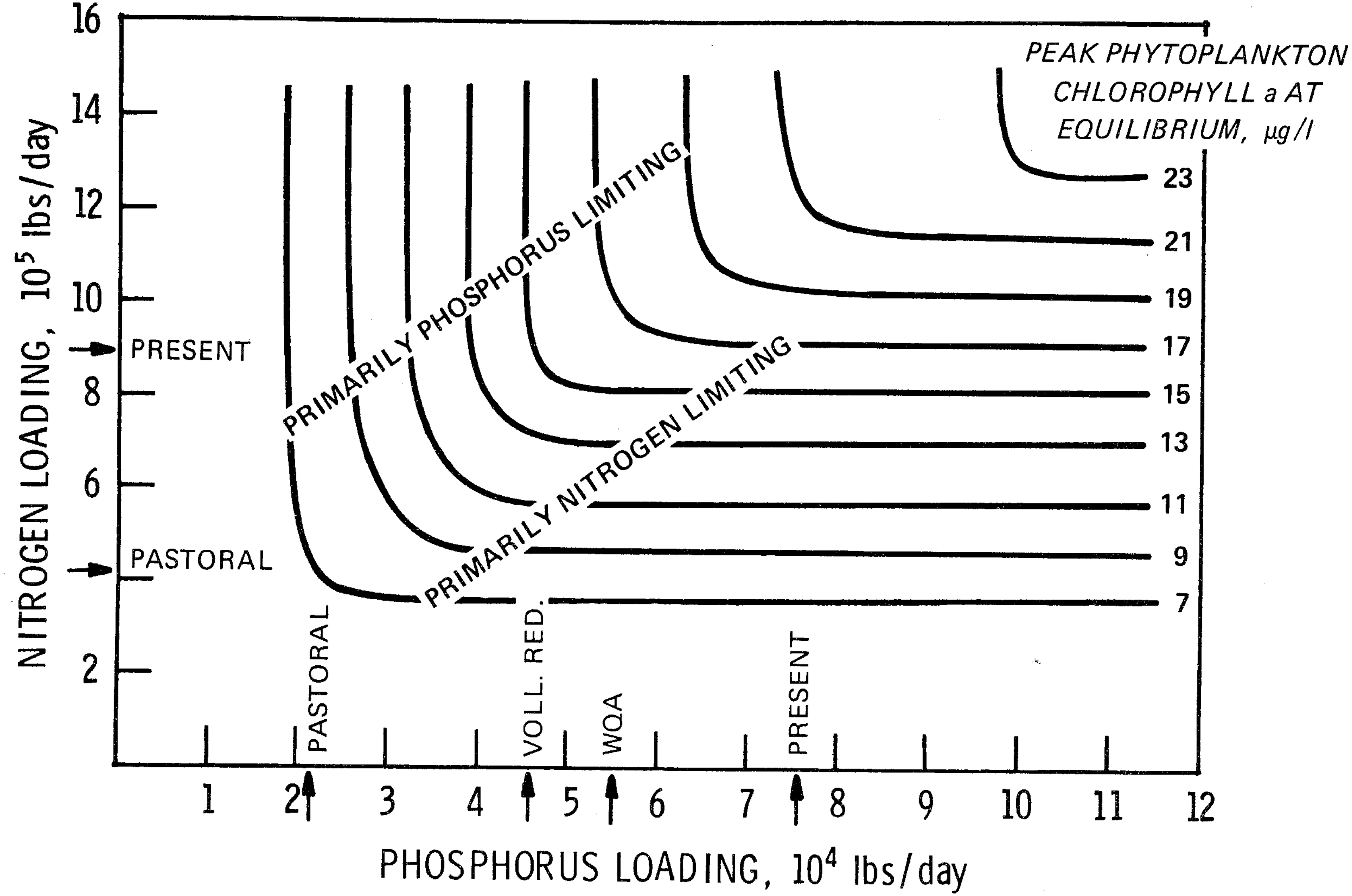

Figure 31 Peak phytoplankton chlorophyll, 0-17m, as a function of nitrogen and phosphorus input - K organic = 0.001/day, K inorganic = 0.0

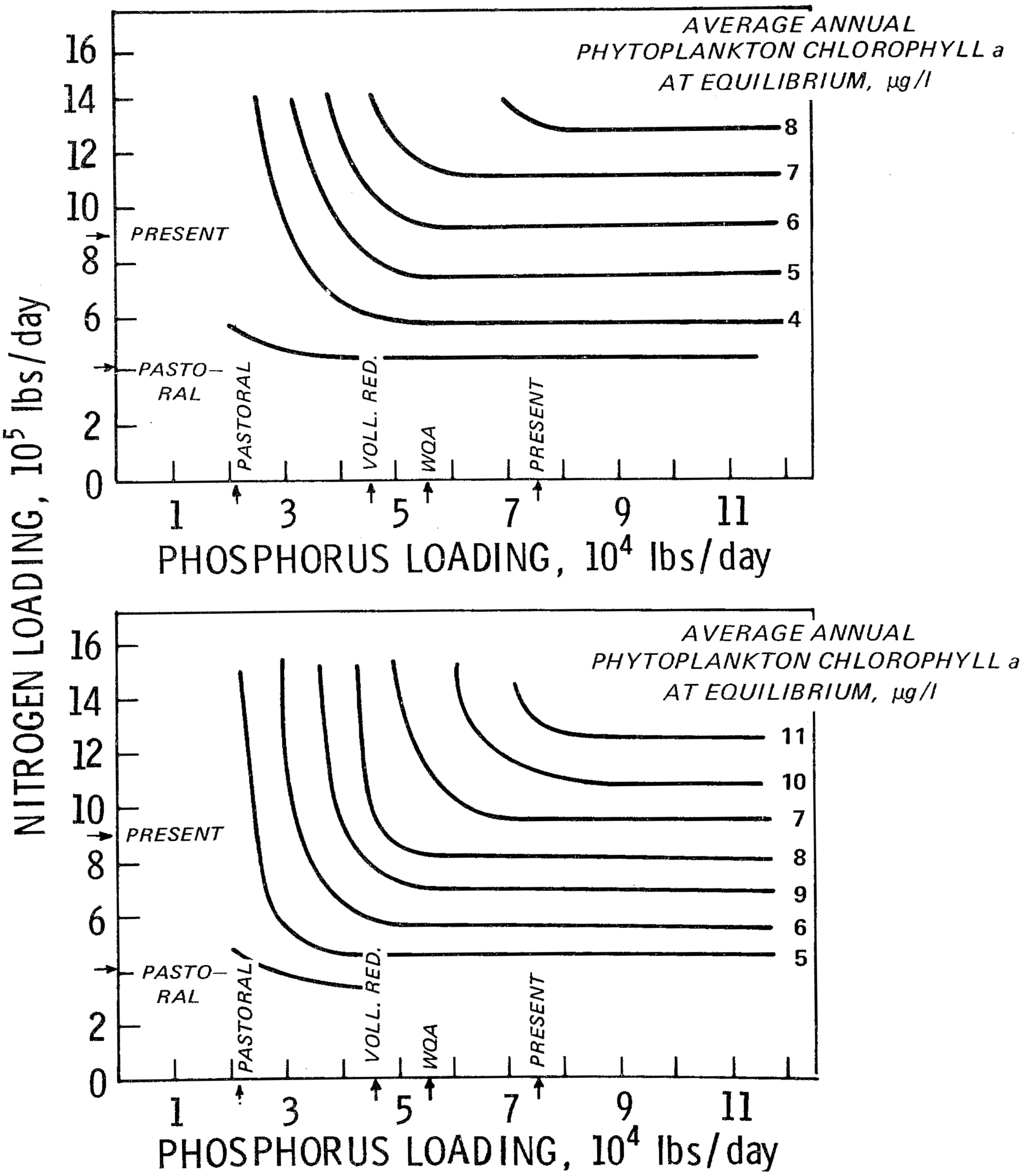

Figure 32 Average annual phytoplankton chlorophyll, 0-17m as a function of nitrogen and phosphorus input-Top fig.: reasonable kinetics, bottom fig.: pessimistic kinetics

rient input. This total includes both point municipal and industrial inputs which are reasonably controllable as well as agricultural non-point sources for which control procedures are not yet available. Finally, only about 60% of the total load discharged to the lake is from point sources. The results therefore tend to indicate that if present conditions are to be maintained, nutrient inputs from Lake Erie together with some form of non-point control as well as extensive point source control will be required. It may therefore be difficult to achieve reductions below present values of biomass.

The results of these simulations indicate that the language of policy documents such as the Water Quality Agreement may be somewhat overly optimistic; e.g. "The Government of the United States of America and the Government of Canada, determined to restore and enhance water quality in the Great Lakes System;..." (Ref.(6), page 2.) This is not to say that the present policy is not a good one. Quite the contrary, the phosphorus removal policy presently being implemented is certainly in the right direction and was formulated based on the the best available information at the time.

This research has simply indicated that hoped for reductions in eutrophication of Lake Ontario may not be realized until additional phosphorus reductions significantly beyond the WQA are achieved. The primary reason for this effect is the hypothesis that Lake Ontario eutrophication is continuing to increase due to the present input so that the nutrient loads and lake wide quality are not in equilibrium.

Finally, it should be noted strongly again that these simulations are indicative of open lake conditions and do not reflect "near-shore" responses which may be quite different. Also, research into the dynamic behavior of phytoplankton in large lakes (or for lakes in general) is still very much in its infancy. Nevertheless, policies and decisions will continue to have to be made even as ongoing research may indicate conflicting results. It is hoped that this work has provided some additional insight into the behavior of Lake Ontario to aid in the development of further policies on nutrient control for the lake.

SECTION VII

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16. ABSTRACT

The results of a series of simulations of the response of the open lake region of Lake Ontario to various levels of nutrient input are described. The simulations use a simplified dynamic model of phytoplankton - nutrient interactions in a vertically segmented structure. The analysis of long term simulations (10-20 years) indicates that the overall loss rates of nutrient are of particular importance. Under reasonable set of model parameters, the simulations indicate that the present observed open lake phytoplankton biomass of Lake Ontario does not appear to be in equilibrium with the present input nutrient load. Therefore, if the present load is continued, it is estimated that spring peak phytoplankton chlorophyll in the epilimnion will continue to increase to a new level about 45% higher than present levels. The interaction of nitrogen and phosphorus is also described by the simulations and the results indicate a tendency for nitrogen limitation to be an increasing dominant factor in controlling the spring bloom. A estimated "pastoral" load simulation, indicative of conditions prior to man's intensive activity, indicates that spring phytoplankton levels were some 40% less than present levels and average annual epilimnion biomass under equilibrium with present loads is about twice that under pastoral conditions.

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