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Summary Report

The OECD Cooperative Programme on Eutrophication

Canadian Contribution

Compiled and prepared by

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PREFACE

The report presented here is the Canadian contribution to the OECD Cooperative Programme on Eutrophication. Scope and modus operandi of said programme have been outlined in a Synthesis Report (OECD, Paris, 1980), and have also been summarized in the four already published Regional Reports. Therefore, the reader is invited to consult these reports for detailed information. To make the present report self-contained, a summary paper which provides a conceptual background, and the most significant results of the total programme, is enclosed as Appendix 1.

The present report completes the North American effort for the OECD Programme. However, the philosophy adopted to produce this report differs from that of the initial North American report produced by Rast and Lee (1978), and differs also from the other regional project conception. The Canadian report has been designed to serve as a test case for evaluating the extent of applicability of the OECD results to an independent set of data not included in the Synthesis Report. The data elaborated for the Canadian report have either been provided by individuals, or have been extracted from already published material. People who have provided data are acknowledged separately and pertinent literature is cited in the bibliography. The text has been kept as concise as possible, wherefore citations are minimal. A background data supplement, containing information sheets listing the most pertinent background information for the lakes and basins discussed in this report, is available on request.

ABSTRACT

The OECD International Programme on Eutrophication has been designed for cross sectional comparison of lakes to provide management with simple tools to evaluate nutrient reduction, particularly phosphorus, necessary to alleviate excessive eutrophication. The scope of the Canadian programme was to test the applicability of the overall OECD results on a set of data not included in the original elaboration with three main objectives in mind:

a) clarification of the extent to which lakes of an unspecified nature exhibit statistical properties similar to the OECD lakes

b) clarification and identification of the limits of transferability

c) identification of conditions which need further evaluation.

The Canadian lakes data base represents a collection of information from personal communication with various workers involved in major limnological projects as well as that contained in the literature. It has been subdivided into seven major geographical regions which are treated separately.

In a diagnostic sense, the majority of Canadian lakes tested show statistical behaviour similar to that of the OECD lakes, particularly in regard to the relationships between annual mean chlorophyll and annual mean phosphorus, annual mean phosphorus and flushing corrected inflow phosphorus concentration, and annual mean chlorophyll and flushing corrected

inflow phosphorus concentration. However, application of the results for predictive purposes requires care. A number of limiting conditions were identified under which applicability and transferability of OECD results are either questionable or should be done with utmost care. These include situations where:

a) $z_{eu}/\bar{z}$ (euphotic zone depth/mean depth) is substantially greater than one
b) hydraulic load is high ($q_s > 50$ m/y), flushing rate is more than twice/year ($T_w < 0.5$ yr) and/or lakes with irregular flushing regimes either seasonally or over consecutive years
c) high mineral turbidity or a high degree of humic staining exists
d) N/P ratios are ≤ 5 and/or P exceeds 100 mg/m^3
e) phosphorus is relatively inert (e.g. as apatite) or internal loading is substantial
f) dynamic equilibrium has not been attained as in the case of increasing or decreasing nutrient loads

RÉSUMÉ

Le programme international de l'OCDE sur l'eutrophisation a été mis en oeuvre en vue de comparer des coupes transversales de lacs pour fournir aux chercheurs sur l'aménagement de l'eau des outils simples afin d'évaluer dans quelle mesure il faut diminuer la quantité d'éléments nutritifs, particulièrement le phosphore, pour atténuer l'eutrophisation excessive. Le but du programme canadien était de vérifier l'application de tous les résultats obtenus par l'OCDE à un ensemble de données qui ne paraissent pas dans le rapport initial. Trois principaux objectifs étaient visés:

a) Établir dans quelle mesure des lacs de nature non spécifiée présentent des propriétés statistiques semblables à celles des lacs de l'OCDE;

b) Éclaircir et identifier les limites d'application à d'autres lacs des données de l'OCDE;

c) Identifier les conditions qu'il y aurait lieu d'évaluer de façon plus détaillée.

La base de données sur les lacs canadiens est constituée de données provenant de communications personnelles avec différents chercheurs engagés dans d'importants travaux limnologiques ainsi que des données provenant de la littérature. Elle a été divisée suivant sept grandes régions géographiques qui sont traitées séparément.

Du point de vue diagnostic, la plupart des lacs canadiens étudiés, présentent un comportement statistique semblable à celui des lacs de l'OCDE, particulièrement en ce qui concerne les relations entre la concentration moyenne annuelle de chlorophylle et la concentration moyenne annuelle de phosphore, la concentration moyenne annuelle de phosphore et la concentration

de phosphore dans les eaux d'apport corrigée pour tenir compte du renouvellement et, enfin, la concentration moyenne annuelle de chlorophylle et la concentration du phosphore dans les eaux d'apport corrigée pour tenir compte du renouvellement. Toutefois, il y a lieu de faire preuve de prudence lorsqu'on applique les résultats à des fins de prévision. On a défini des conditions limites dans lesquelles l'applicabilité des résultats obtenus par l'OCDE est douteuse ou devrait être faite avec la plus grande prudence. Parmi ces conditions, on compte:

a) $z_{eu}/\bar{z}$ (profondeur de la zone euphotique/profondeur moyenne) beaucoup plus grand que l'unité;

b) charge hydraulique élevée (q_s > 50 m /année) renouvellement plus fréquent que deux fois par année (T_w < 0,5 année) et/ou lacs présentant des régimes de renouvellement irréguliers, c.-à-d. renouvellement saisonnier, ou renouvellement pendant des années consécutives;

c) turbidité minérale élevée ou degré élevé de coloration humique;

d) rapports N/P $\leq$ 5 et/ou concentration de P supérieure à 100 mg/m^3;

e) phosphore relativement inerte (p. ex. sous forme d'apatite) ou apport interne important;

f) équilibre dynamique encore non atteint, comme dans le cas de charges d'éléments nutritifs croissantes ou décroissantes.

ACKNOWLEDGEMENT

The authors wish to express their thanks to all who have contributed to the presented compilation. The work has not been simple, at times very tedious and frustrating because of the complexity of the subject treated. Such large-scale comparison would never have been possible without the generosity and responsiveness of the many researchers who have been involved. In addition, much of the data collection was on a personal communication basis and only through this cooperation have we been able to remain sensitive to the problems and meanings of the original measurements. This has undoubtedly improved the interpretation and reliability of conclusions. Special thanks go to those who shared their often unpublished results and/or painstakingly reviewed and commented on the initial exposition of the various chapters. In particular, this includes J. J. Kerekes (Atlantic Region), P. Campbell, J. Cornett, J. Kalff, P. Potvin, S. Watson (Quebec Region); P. Dillon (Ontario Shield Region); A. S. Fraser and H. F. H. Dobson (Laurentian Great Lakes); E. J. Fee and D. W. Schindler (Experimental Lakes Area); R. J. Allan (Prairie Lakes); G. J. Brunskill, R. J. Daley, C. J. Gray, R. N. Nordin, T.G. Northcote, L. Patalas and J.G. Stockner (British Columbia Region). Noteworthy contributions also came through the published work of many people and in addition to those mentioned above, this includes M. Dickman (Quebec Region); M. F. P. Michalski, K. Nicholls, F. H. Rigler and W. Scheider

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TABLE OF CONTENTS

LIST OF FIGURES

Page

LIST OF TABLES

LIST OF APPENDICES

INTRODUCTION

THE EUTROPHICATION PROBLEM IN CANADA

- General Background -

Canada, with a total area of 10 million km^2, and a population of nearly 23 million people, has some 756,000 km^2 of fresh water. Streamflow in Canada's rivers has been estimated to be about 100,000 m^3 per second, equivalent to about 50% of Canada's annual precipitation, and represents about 9% of the total flow in all the world's rivers. Much of the usable surface water present in Canada is stored in lakes and reservoirs, or elsewhere in the form of ice and snow.

Canada probably has more lake area than any other country in the world: a complete inventory is not feasible at present. Five hundred and sixty-three large Canadian fresh water lakes exceed a water surface area of 100 km^2; 42 of these exceed 1,000 km^2. Outstanding among these lakes, in terms of size, are the Great lakes, although only parts of these are in Canadian territory. In central Canada, Lake Winnipeg is the largest lake; Great Slave and Great Bear are the largest in the north-western region. Others worthy of mention are Lakes Athabasca, Reindeer, Winnipegosis, Nipigon, Manitoba, Lake of the Woods, Nettilling and Dubawnt. Apart from these large lakes are countless other smaller lakes scattered througout Canada, particularly in the Canadian Shield.

River discharge in the different main river basins varies considerably and consequently, the range of theoretical water residence time is tremendous. Of the smaller lakes, many receive an extremely high hydrological load and have very short water residence times of weeks only, whereas the residence time of some of the large lakes can exceed a hundred years. In addition, variability of the residence time for a particular system is affected by its size and complexity. The Saskatchewan River, with a mean flow of 680 m^3 per second, has a maximum flow of 3,000 m^3 per second, and minimum of 50 m^3 per second. The maximum recorded flow is 4.4 times the mean flow and 60 times the minimum. On the other hand, large systems have the capacity to damp these variations in flushing rate. The St. Lawrence River, flushing the Laurentian Great Lakes system has a mean flow of 6,740 m^3 per second. The maximum is only 1.5 times the mean and about twice the minimum. Residence time and its variability are of considerable importance for the actual susceptibility of Canadian lakes to nutrient load and eutrophication.

- The Present Situation -

In considering the general situation, the eutrophication problem in terms of the total water resource stored in lakes, for Canada, cannot be defined as serious compared to other regions in the world. Most of the Canadian lakes belong to the oligotrophic category. A cross-sectional listing of the 130 lakes, including the lakes considered in this report, showed the following distribution of trophic categories:

a) in terms of number of lakes:
oligotrophic - 73%, mesotrophic - 15%, eutrophic - 12%

b) in terms of surface area:
oligotrophic - 73%, mesotrophic - 13%, eutrophic - 14%

Although these statistics result from availability of data rather than from an unbiased selection, it probably provides a good characterization of the trophic conditions of Canadian lakes, at least for the more inhabited southern belt. The figures would probably change in favour of an even higher percentage of oligotrophic lakes if more northern lakes, in particular some of the large lakes (Athabasca, Great Slave, Great Bear and others) not considered were included in the analysis.

In contrast to this general situation, the eutrophication problem of individual cases can be quite serious; incidences of more or less advanced eutrophication have been reported throughout all the regions of high population density and/or intensified agriculture. Lake size also seems to be of subordinate importance: eutrophied lakes range from small cottage and prairie lakes to Lake Erie exceeding 25,000 km^2.

REPORT ORGANIZATION

- Lakes Considered and Regional Synopses -

For the scope of the present report, the lakes reviewed have been arranged into seven regional groups (cf. Figure 1, Table 1) covering the entire expanse of Canada. Each region is explored separately in the first seven chapters. The groups as defined here do not entirely coincide with geographic entities, nor do they coincide with the natural limnological regions as e.g. proposed by Northcote and Larkin (1966) for

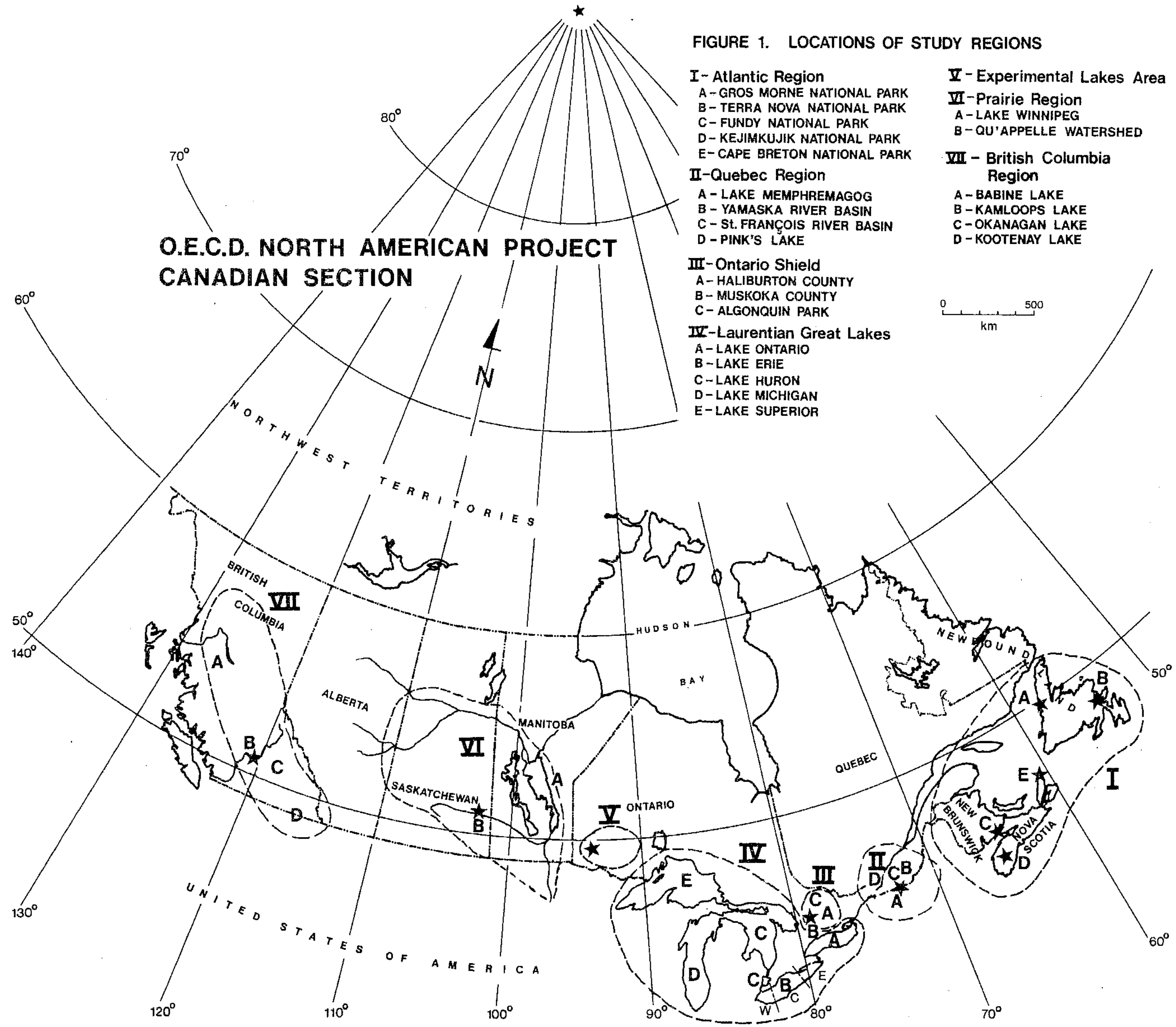

FIGURE 1. LOCATIONS OF STUDY REGIONS

Table 1. List of lakes included in the Canadian OECD analysis (grouped according to regions)

ATLANTIC REGION I

A. Gros Morne National Park, Nfld.	
Western Brook Pond	(WB)*
B. Terra Nova National Park, Nfld.	
Bluehill North Pond	(BN)
Bluehill South Pond	(BS)
Minchin Pond	(M)
Yudle North Pond	(YN)
Pine Hill Pond	(PH)
C. Fundy National Park, N.B.	
Bennet	(Be)
MacLaren's Pond	(ML)
Wolfe	(Wo)
D. Kejimkujik National Park, N.S.	
Grafton	(Gr)
Kejimkujik	(Ke)
Mountain	(Mo)
Pebbleoggitch	(Pe)
Little Kempton	(LK)
E. Cape Breton Highlands National Park, N.S.	
Freshwater	(FW)
MacDougall's	(MD)
Presqu'ile	(Pr)
Warren	(Wa)

QUEBEC REGION II

Lake Memphremagog	
- North Basin	(MN)
- Central Basin	(MC)
- South Basin	(MS)
Pink's Lake	(Pi)
Saint-Francois River Basin	
- Aylmer	(Ay)
- Bowker	(Bo)
- Brompton	(Br)
- Lovering	(Lo)
- Magog	(Ma)
- Massawippi	(Ms)
- Montjoie	(Mo)
- Petit Brompton	(PB)
- Saint-Francois	(SF)
- Stuckely	(ST)
Yamaska River Basin	
- Boivin	(Bv)
- Brome	(Bm)
- Roxton	(Ro)
- Waterloo	(Wa)

ONTARIO SHIELD LAKES REGION III

A. Haliburton County	
Beech	(Be)
Bob	(Bo)
Cameron	(Ca)
Cranberry	(Cr)
Eagle-Moose	(Ea-Mo)
Four Mile	(FM)
Halls	(Ha)
Maple	(Ma)
Oblong-Haliburton	(O-H)
Pine	(Pi)
Raven	(Ra)
Talbot	(Ta)
Twelve-Mile-Boshkung	(TM-B)
B. Muskoka County	
Dudley Bay	(DB)
Jerry	(Je)
Harp	(Hp)
Gravenhurst Bay	(GB)
Joseph	(Jo)
Little Joseph	(LJ)
Little Otter	(Lo)
Muskoka	(Mu)
Rosseau	(Ro)
Simcoe	(Si)
Skeleton Bay	(SB)
C. Algonquin Park	
Brewer	(Br)
Clarke	(Cl)
Costello	(Co)
Found	(Fo)
Kearney	(Ke)
Little McCauley	(LM)
D. Haliburton-Muskoka	
Basshaunt	(Ba)
Bigwind	(Bi)
Blue Chalk	(BC)
Buck	(Bu)
Chub	(Ch)
Crosson	(Cs)
Dickie	(Di)
Glen	(Gl)
Gullfeather	(Gu)
Harp	(Hr)
Jerry	
Little Clear	(LC)
Red Chalk	(RC)
Solitaire	(So)
Walker	(Wa)

* Note: The two letter abbreviations of lakes' names, as given here, are used throughout the figures for identification of points.

LAURENTIAN GREAT LAKES REGION IV

- Lake Superior	(Sp)
- Lake Michigan	(Mc)
- Lake Huron	(Hr)
- Lake Erie	
- Western Basin	(WB)
- Central Basin	(CB)
- Eastern Basin	(EB)
- Lake Ontario	(ON)

EXPERIMENTAL LAKES AREA V

a) Natural Condition
114
120
223
224
228
239
240
303
305
383

b) Artificial Enrichment
226 NE
226 SW
227
261
302 N
302 Total
302 S
304

PRAIRIE LAKES REGION VI

- Lake Winnipeg	
- Qu'Appelle River Lakes	
- Pasqua Lake	(Ps)
- Echo Lake	(Ec)
- Mission (Lebret) Lake	(Ms)
- Katepwa	(Kt)
- Crooked Lake	(Cr)
- Round Lake	(Rd)

BRITISH COLUMBIA REGION VII

Babine Lake	(Bb)
Kamloops Lake	(Km)
Kootenay Lake	(Kt)
Okanagan Lakes	
- Wood Lake	(Wd)
- Kalamalka Lake	(Kl)
- Okanagan Lake	(Ok)
- Skaha Lake	(Sk)
- Osoyoos-N Lake	(Os)

Western Canada. However, the lakes listed in each group have some common features which are either unique or sufficiently unifying to permit comparison also with regions outside Canada. For example, the British Columbia lakes are in many ways similar to the European pre-Alpine lakes; the Ontario, Quebec and Atlantic Region lakes have more similarity with Scandinavian lakes. The Laurentian Great Lakes have some properties of deep pre-Alpine lakes, yet also have unique features which make them hardly comparable to the majority of lakes studied in the OECD Programme. Although there are trends within each region, a wide range of trophic conditions may be represented.

The following is a short synopsis of each of the regions considered.

I Atlantic Region Lakes

The majority of these lakes are small pristine lakes of less than 1 km^2 in size, the largest lake (Kejimkujik) extending to 24 km^2. With the exception of a few lakes studied by Parks Canada (CWS), these lakes are not well known. Those included in this report have prevailing oligotrophic and ultra-oligotrophic characteristics. Besides a few lakes of slightly mesotrophic properties, only McLaren Pond has been indicated as eutrophic.

II Quebec Region Lakes

Several lakes of this region are under extensive study by provincial laboratories, the Institut National de Recherche Scientifique

(INRS-Eau) and other university groups. These lakes range from oligo- to eutrophic. The most important lake included in this grouping is Lake Memphremagog (90 km^2) which is situated at the Canada-US border. This lake is heavily eutrophied from city sewage discharged at its southern end.

III Ontario Shield Lakes

The majority of these lakes bear a similarity to the Atlantic Region lakes, being basically oligotrophic. A common characteristic of many of these lakes is their anastomosis into lake chains or lake complexes which makes simple definition difficult at times (cf. e.g. Muskoka lakes). They share this feature with other Shield lakes as well as lakes in Scandinavia.

Several of these lakes have undergone eutrophication, in part due to cottage development, and in part due to direct city discharges. The most serious case of eutrophication has been Gravenhurst Bay, a bay almost entirely separated from Muskoka Lake.

Also for these lakes, information is in part scant, yet in part excellent studies have been conducted by the provincial laboratories and have been made available for the OECD Programme.

IV Laurentian Great Lakes

The most noted cases of eutrophication are those of the so-called Lower Lakes (Lake Ontario and Lake Erie). The upper Great Lakes (Superior and Huron) instead, are generally in acceptable oligotrophic condition. Exception must be made, however, regarding more localized conditons in all the Great Lakes. Many of these bays show signs of more or less serious degradation due to eutrophication. The following map (cf. Figure 2) which has been taken from the IJC Volume "Environmental Management Strategy for the Great Lakes System, 1978", provides an overview of the prevailing local (inshore) eutrophication conditions of this system. Because of the marked inshore-offshore differentiation, a simple allocation of these lakes to any one of the trophic categories has but limited value. Lake Huron (including Georgian Bay) e.g. for its main water body would have to be classified as oligotrophic, yet at the same time, the large Saginaw Bay exhibits one of the most serious conditions of degradation. Similarly, Lake Michigan, generally speaking, has but localized problems of eutrophication but Green Bay is meso- to eutrophic. Also, in many areas of these lakes the most serious eutrophication problem is excessive development of *Cladophora* along the shores. Quantitative information of this problem is scant, which makes valid comparison between lakes and/or between inshore and offshore conditions impossible for the present.

V ELA Lakes

These lakes are not essentially different from other Shield lakes having prevailing oligotrophic characteristics. The reason for separating them as a specific category is the fact that several of these lakes have been artificially manipulated to study the effects of nutrient loading on pristine oligotrophic lakes. The results of this programme on

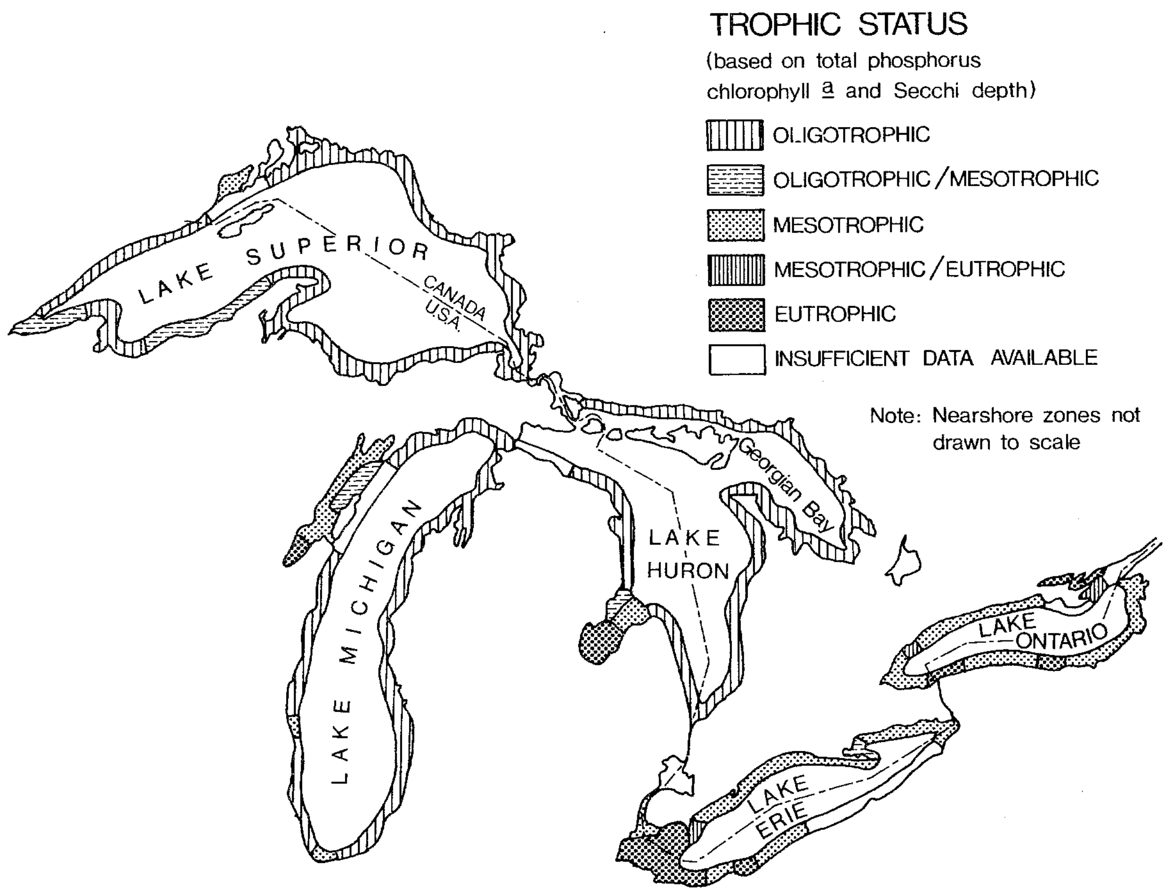

Figure 2. Nearshore trophic condition of the Great Lakes.

whole-lake experimentation under the direction of Dr. D.W. Schindler are well known in limnology for their uniqueness, and the convincing demonstration that eutrophication depends primarily on load of phosphorus and nitrogen.

VI Prairie Lakes

Prairie lakes for the most part are relatively small lakes and ponds, many of which are highly eutrophic or hypertrophic with fish kills during winter or summer situations of anoxia. The special situation presented by lakes of this type has not been considered in this report. Instead, the larger and more economically important lakes have been the subject of this analysis. Data have been most readily available for Lake Winnipeg and lakes of the Qu'Appelle watershed. Lake Winnipeg receives 50% Shield drainage and is eutrophic only in localized areas. The Qu'Appelle chain of lakes has high year-to-year variability in hydraulic and accordingly, nutrient load. In years when hydraulic loading is high, this occurs prevalently during the spring and early summer period with peak values 10 to 20 times above average discharges and 50 to 75 times above minimum average discharges. Biomass development also varies considerably from year to year, ranging from apparent mesotrophy in one year to highly eutrophic conditions in another. Lake Winnipeg, on the other hand, shows less variation, yet exhibits a strong north-south differentiation.

VII British Columbia Interior Valley Lakes

In this group appear a few separate lakes (Babine, Kamloops, Kootenay) and a chain a lakes situated in the Okanagan Valley.

In regard to trophic conditions Babine and Kamloops lake are oligotrophic. Babine lake shows the special feature that some 20% of its phosphorus income comes from salmon carcasses which is difficult to evaluate in its effects on the nutrient budget. Towards the other end of the spectrum, Kootenay lake has experienced strong eutrophication, and is now in a process of recovery. The Okanagan lakes receive in part direct city discharges causing eutrophication, yet these lakes remain strongly differentiated by flushing rates which range from 0.014 to 33 times/year.

- Data and Data Elaboration -

Data. Whereas in the Alpine, Northern and Reservoir Shallow Lakes Project case studies have been initiated and conducted according to an agreed system over a two to three year period, in the Canadian project no specific OECD oriented study was initiated. Available data have been obtained from contributors out of their ongoing studies, or from already published literature. Therefore, the basic data bank is somewhat uneven and in part incomplete, for example where annual means were not available seasonal means were sometimes substituted. More importantly, no analytical intercomparison for method standardization has been made which means that comparability of data is not always warranted.

Apart from this problem, there are intrinsic problems with some of the data, particularly nutrient loading data. Lakes receiving high hydrological loads often show high year to year variability in nutrient loadings which makes correct characterization of such lakes exceedingly difficult. In such cases, loading measurements limited to a one year cycle are almost meaningless. Also, nutrient loading often occurs over a limited time period of the year coinciding with high hydraulic load, in contrast to the more common situation where the load is delivered more gradually.

At the other end of the scale, concerning large lakes, difficulties arise in estimating total loads for the mere size of the catchment areas to be monitored. For example, in regard to the Laurentian Great Lakes, considerable discussion has arisen over the last years as to the significance of reported nutrient loads. Figures taken from various sources may vary considerably.

Also, indirect loading estimates are often plagued with considerable uncertainty of land export coefficients, significance of the more remotely located areas of a catchment system in terms of areal contribution, the potential contribution of septic tank leachates, the importance of animal husbandry, the importance of contributions from detergents, etc.

Accordingly, in judging the validity of correlations and conclusions based on available information, these conditions must be kept in mind. However, every effort has been made, often in direct collaboration with the authors responsible for data, to reduce the level of uncertainty. In several instances the author-provided data have been recalculated for an independent check. What to a reader unfamiliar with the real situation may easily appear as "juggling of the figures", in most cases can be explained with the at times enormous hidden difficulty of separating truth from some initial subjectivity.

Comparability of Canadian data with the OECD data. Transferability of statistical information derived from one set of data to a new set, and hence, comparability between the two sets of data, depend on whether or not the two sets are comparable in number, range, mean values and other statistical parameters. The Synthesis Report contains a full table of all directly and indirectly measured statistical parameters of the programme. In order to examine the Canadian data against this background, a few key parameters have been selected and tabulated in Table 2, together with the corresponding OECD data.

Comparison of the two sets shows that in terms of limnological characterization, such as mean depth and theoretical water residence time (filling time), average inflow and lake phosphorus concentrations and chlorophyll levels, the Canadian data cover about the same range. Also, in regard to the mean values of the basic parameters, mean depth and water residence time, the two sets are comparable.

The essential differences are in the trophic parameters. The geometric means (which are closer to the mode than arithmetic means because of the skewness in data distribution) for average inflow and

Table 2 . Comparison between some mean features of OECD and Canadian lakes.

Variables	OECD Lakes				Canadian Lakes			
	Min.	Geom Av.	Max.	N	Min.	Geom. Av.	Max.	N
$\bar{z}$ (m)	1.7	14.3	313	126	1.1	10.4	148	108
T(w) (y)	.016	1.2	700	112	.017	.91	185	108
$[\bar{P}]_i$ (mg/m³)	4.7	112	1425	95	.35	35	2268	82
$[\bar{P}]_\lambda$ (mg/m³)	3.0	47	750	115	1.9	16.1	1600	82
$[\overline{chl}]$(mg/m³)	.3	8.4	89	96	.55	2.8	33.6	104

	OECD Lakes			Canadian Lakes		
Lake Types	Oligo-	meso-	eutrophic	oligo-	meso-	eutrophic
%	18	17	65	73	15	12
Numbers		(101)			(129)	

average lake phosphorus concentrations as well as for chlorophyll, differ characteristically by a factor of about 3 between the two sets. The lower Canadian means in these parameters reflect the difference in the distribution of trophic categories of water considered. Whereas in the OECD programme the majority of lakes studied are eutrophic, the majority of the Canadian lakes selected belong to the oligotrophic category. It is therefore expected that - whilst in principle the OECD information can be applied to the Canadian data - the analysis may lead to indications for revisions and modifications of the OECD standard relationships. However, it will not be within the scope of this report to pursue such matters beyond this preliminary stage.

Correlations. Instead of attempting to correlate the data on a national project basis (i.e. similar to the US portion of the North American Project) they are treated here with the aim of comparison with the results arising from the other OECD regional projects and the project synthesis. This means that - rather than to establish a fifth set of correlations and regressions - the information is analysed to establish consistency or non-consistency with results from other projects. Indeed, this methodology has not only proved to be more appropriate to the Canadian data than independent analyses, but provides a test study for the applicability of the OECD results, as summarized in the Synthesis Report, to a new set of independent data.

Data have been log-log plotted, and the standard regressions used for comparison are those reported in the Synthesis Report*) The respective coefficients, correlation coefficients and standard errors are given in Table 3.

In addition, these relationships are graphically represented in Appendix Figures A1.1 to A1.17. It should be noted that regression lines denominated as a-lines are those given in the Synthesis Report, and are commonly called the OECD lines. They have been obtained by minimizing y against x (cf. Table 3). The b-lines, instead, represent the regression lines resulting from minimizing x against y, but have been recalculated for x as reference variable. The slope of the third and intermediate line, finally has been calculated from the ratio SE(y)/SE(x).

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*) The rationale for log-log plotting is in part purely practical (i.e. to distinctly plot data spreading over 2 to 3 orders of magnitude on the same figure), in part also justified by statistical consideration. As has been found in elaborating the Synthesis Report, data distribution is highly skewed. With log-log transformation not only the skewness is reduced but also the prediction error is stabilized over the whole range, (i.e. the percent prediction error at the lower and upper end of the scale remains constant). Although visually this may appear for someone who is less used to reading log-log diagrams as data polishing, in effect the error margins at the upper end of the scale are as high as in a linear representation. The added benefit with a log-log representation is that deviations from linearity, and non-conforming situations are clearly recognizable over the whole range of the scale.

Table 3. OECD - Standard Regressions, $Y = A.X^B$ (See graphic representation in Appendix 1).

Fig. A.1.	Y versus X		n	r.	A [2]	B [2]	A [3]	B [3]	$\bar{A}$ [4]	$\bar{B}$ [4]	SE(Y) [5]	SE(X) [6]
1, 2	$[\overline{chl}]$	$[\bar{P}]_\lambda$	77	.88	.287	.962	.110	1.242	.183	1.093	.251	.230
3, 4	$[chl]_{max}$	$[P]_\lambda$	50	.90	.635	1.048	.277	1.294	.429	1.165	.257	.221
5, 6	$[\bar{P}]_\lambda$	$[X]$ [1]	87	.93	1.566	.821	.948	.950	1.229	.883	.193	.219
7, 8	$[\overline{chl}]$	$[X]$ [1]	67	.88	.373	.795	.160	1.027	.251	.904	.256	.283
9, 10	$[chl]_{max}$	$[X]$ [1]	45	.89	.707	.905	.290	1.142	.464	1.017	.284	.279
11, 12	$\bar{S}$	$[\overline{chl}]$	78	- .75	9.32	- .508	22.0	- .903	13.5	- .677	.198	.292
13, 14	$\bar{S}$	$[\bar{P}]_\lambda$	87	- .47	9.89	- .284	505	- 1.284	34.8	- .603	.255	.423
15, 16	$\bar{S}$	$[X]$ [1]	67	- .69	14.8	- .387	85.7	- .812	30.3	- .560	.236	.421
17, -	$[N]_\lambda$	$[X]'$ [1]	42	.92	5.30	.782	2.00	.924	3.33	.850	.157	.182

[1] $[X] = [\bar{P}]_\lambda/(1 + \sqrt{T(w)})$; $[X]' = [\bar{N}]_\lambda/(1 + \sqrt{T(W)})$

[2] a) - Regression

[3] b) - Regression

[4] Average between a) and b)

all reported against X as reference variable

[5] and [6] Standard Errors referring to $^{10}\log$-transformed equation

Table 3 shows that the difference between the standard errors of the two corresponding regression equations for the relationships 1) to 5) is insignificant. Hence, the two regression lines a) and b) are interchangeable for any practical purpose, and the intermediate regression may be the best predictor. In some cases the b-line seems to describe the data better than the a-line. However, in order not to overload the graphical representations, the corresponding confidence ranges (80% and 95%) have been plotted only relative to the a-line. Confidence limits have been calculated from the composite of OECD data for both 80 and 95% probability levels. Test case values which fall within the 80% limits are accepted as showing no difference from the OECD relationship. Values which fall between the 80 and 95% limits are considered suspect as potentially non-conforming, whereas those outside the 95% limits are taken as non-conforming with the OECD relationship.

For the relationships 6 to 8, the large differences in the respective standard errors indicate that only the a-relationship is meaningful (i.e. prediction of Secchi transparency from $[chl]_{\lambda}$, $[P]_{\lambda}$, $[P]_{\lambda}/(1 + \sqrt{T(w)})$, respectively, is valid, but not the inverse).

For other relationships the data base for calculating standard errors and confidence limits was insufficient or not fully elaborated in the Synthesis Report, (e.g. primary production, phytoplankton biomass and oxygen depletion rates). Available Canadian data have been plotted in an attempt to make some preliminary comparison between them, and with the respective provisional OECD regression models. Interpretation of nitrogen data has been made under the assumption that nitrogen is limiting biomass yield only when concentrations are in a proportion to phosphorus of less than that most commonly found in the biomass itself (i.e. about 10:1).

In addition to the cross-sectional examination of regional cases, six examples of culturally imposed non-equilibria are discussed (Chapter 8). The lakes or watersheds treated here have undergone abrupt changes in nutrient loading due to municipal and/or industrial development and subsequent pollution abatement measures. The central point in question is that regarding the time requirement for changes in nutrient loading to produce changes in trophic condition. A summary of Canadian experience to date is presented.

CHAPTER 1. ATLANTIC REGION I

A. Gros Morne National Park, Nfld.

Western Brook Pond	(WB)

B. Terra Nova National Park, Nfld.

Bluehill North Pond	(BN)
Bluehill South Pond	(BS)
Minchin Pond	(M)
Yudle North Pond	(YN)
Pine Hill Pond	(PH)

C. Fundy National Park, N. B.

Bennet	(Be)
MacLaren's Pond	(ML)
Wolfe	(Wo)

D. Kejimkujik National Park, N. S.

Grafton	(Gr)
Kejimkujik	(Ke)
Mountain	(Mo)
Pebbleloggitch	(Pe)
Little Kempton	(LK)

E. Cape Breton Highlands National Park, N. S.

Freshwater	(FW)
MacDougall's	(MD)
Presqu'ile	(Pr)
Warren	(Wa)

1.1 Atlantic Region, Description of Location

The present day lakes of the Atlantic region lie in glacially formed depressions of Devonian and Ordovician sediments (i.e. limestones, sandstones, granites and shales) of Canada's east coast.

The pronounced wetness which characterizes the climate of the maritimes results from convergence of three major airstreams. The precipitation, caused by entrainment and mixing of Arctic, Tropical and Pacific air masses, amounts to an annual mean of 100 to 150 cm (Bryson and Hare, 1974). The persistent cloud cover permits fewer hours of bright sunlight than for any other location in Canada. The period of maximum precipitation is during the winter months and because of the warm Tropical air current from the Gulf Stream, the snow is periodically washed away by unpredictable winter rains. Ice cover may not persist throughout the winter and is more likely to do so in the interior lakes. Maritime springs are cold, prolonged and less dramatic with respect to flooding than the continental interior where entire winter snow accumulations are abruptly delivered to the lakes.

With precipitation in excess of evaporation, podzolization is the overriding process that occurs in what little soil is present. Calcium and iron are leached downwards to form a poorly drained hardpan which leaves the surface prone to bog formation. Lowland boggy areas immediately surrounding lakes are common and the humic materials generated by them eventually wash into the lakes staining them a rusty brown. The degree of colouring is proportional to the size of the drainage basin. Lakes with colour of 20 Hazen units or more in general have drainage basin to lake surface areas greater than 5:1. Clear water lakes also exist in this region and are predominantly in the highlands.

Through influence of the sea, Na and Cl ions are more abundant in these than in most freshwaters. Some lakes are among the lowest in the world in Ca content.

The flora consists of the Acadian forest (spruce, fir and pine with a deciduous hardwood complement of maple, birch and beech). Typical phytoplankton assemblages include Chrysophyceae and diatoms. The inevitable connection of watershed and sea has allowed penetration of the American eel (Anguilla anguilla) to virtually all lakes of the region. The brook trout (Salvelinus fontinalis) is also a common feature of the fauna.

The lakes of this study are located in national parks (Figure I 1) and for the most part, are free from direct human influence. (Exceptions are Freshwater, Kemjimkujik, MacLaren and Wolfe Lakes, which receive nutrients from sewage and lawn fertilizer runoff). The main 'industry' of the region is fishing and forestry. This economy is supplemented by dairy and fruit farming.

Selected limnological features, including total phosphorus, mean and peak chlorophyll a, colour, pH, conductance and selected morphometric features from 18 lakes in five Atlantic Region national parks are presented in Table I 1. With the exception of Western Brook Pond, where only ice-free season data are available, annual mean values are given. Annual phosphorus loadings are available for Kejimkujik and Warren Lake Only.

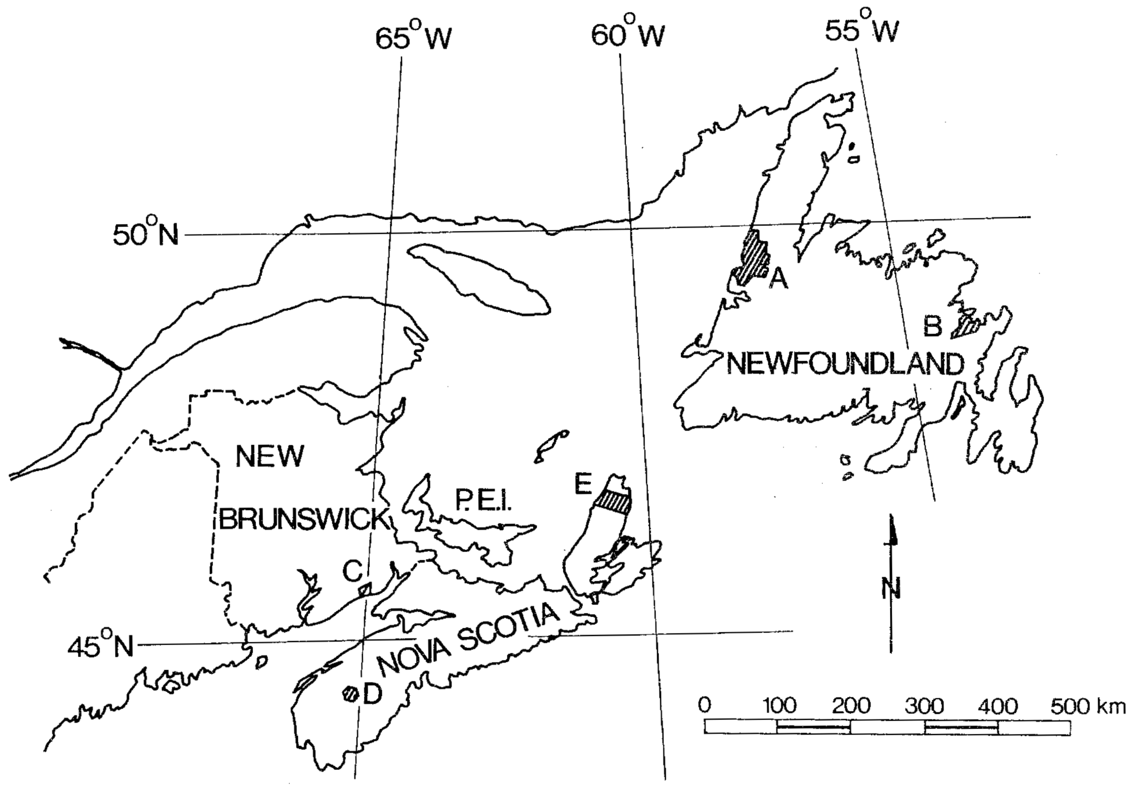

FIGURE I 1. Location of Atlantic Region Lakes.

A Gros Morne National Park
B Terra Nova National Park
C Fundy National Park
D Kejimkujik National Park
E Cape Breton Highlands National Park

Table I 1. Selected limnological features of Atlantic Region lakes (annual mean values).

Lake	Surface Area km^2	Mean Depth m	Flushing Rate times/yr	Chlorophyll a Peak mg/m^3	Chlorophyll a Mean mg/m^3	Total Phos. mgPm^3	Colour Hazen Units	pH	Conductivity μmhos/cm	Trophic Status
A. Gros Morne, Nfld.										
Western Brook Pond	22.8	72.6	0.14	0.9	.55	1.9	6	6.0	35	UO
B. Terra Nova, Nfld.										
Bluehill North Pond	0.16	2.6	13.0	2.1	1.3	6.9	32	6.8	35	O
Bluehill South Pond	1.11	9.2	0.44	1.7	1.1	3.7	21	6.9	35	O
Minchin Pond	0.064	7.0	40.6	3.2	1.7*	7.0	44	6.7	33	O
Yudle North Pond	0.06	2.4	4.7	2.2	0.9	7.7	39	6.7	36	O
Pine Hill Pond	.02	1.1	10.0	4.1	1.4	9.6	27	6.8	83	O
C. Fundy, N.B.										
Bennet	0.31	2.3	28.2	---	1.3	6.0	14	6.4	33	O
MacLaren's Pond	0.007	5.1	8.8	26.3	9.4	20.6	clear	7.1	150	E
Wolfe	0.22	3.8	2.4	7.5	3.6	7.7	4	6.1	25	O
D. Kejimkujik, N.S.										
Grafton	2.7	2.8	7.4	4.3	1.8	10.5	44	6.3	26	O
Kejimkujik	24.33	2.4	6.7	4.0	1.2	9.2	72	5.0	24	O
Mountain	1.36	4.2	1.5	---	1.0	5.1	15	5.3	22	O
Pebbleloggitch	0.33	1.4	3.5	2.9	1.6	10.5	90	4.4	30	O
Little Kempton	0.025	1.4	23.1	15.5	---	27.9	180	5.4	24	M
E. Cape Breton, N.S.										
Freshwater	0.42	6.5	2.0	6.2	2.5	7.5	6	7.1	145	O
MacDougall's	0.05	3.4	9.7	2.8	1.1	5.9	5	6.2	33	O
Presqu'ile	0.044	2.1	14.7	7.0	3.3	13.9	7	7.8	290	M
Warren	0.9	15.9	3.6	2.6	0.7	6.2	50	5.8	30	O

* Ice free average

1.2 Trophic Response-Nutrient Relationships

1.2.1 Chlorophyll-Phosphorus Relationship. There is a recognizable pattern as to the relationship of the three groups of lakes (i.e. lakes with coloured water, cultural nutrient load and clear-water lakes) to the OECD line for both annual mean and maximum concentrations of chlorophyll.

Coloured lakes all fall below the OECD relationships (Figure I 2 and I 3). In these lakes, a large proportion of the total phosphorus represents dissolved organic substances, which are not readily available for phytoplankton growth. Available phosphorus in coloured lakes then is a considerably lower proportion of the total than that of clear-water lakes, and in accordance with this, chlorophyll response appears low in relation to total P. Low pH does not seem to influence this overall trend. The most acidic,coloured lake(Pebbleloggitch, pH - 4.4) and one of the least acidic coloured lakes (Yudle North, pH = 6.7) both occupy positions approximately the same distance below the OECD standard relationship. (Note: Although preliminary data suggests summer nitrogen limitation in some Kejimkujik Park lakes, the role of nitrogen limitation in the coloured lakes has yet to be evaluated; adequate data has not been available.

The four lakes which receive cultural nutrient loads from sewage and lawn fertilizers are above the OECD relationships for both mean and peak chlorophyll. The additional phosphorus load, high in available phosphorus, explains their position above the expected relationship.

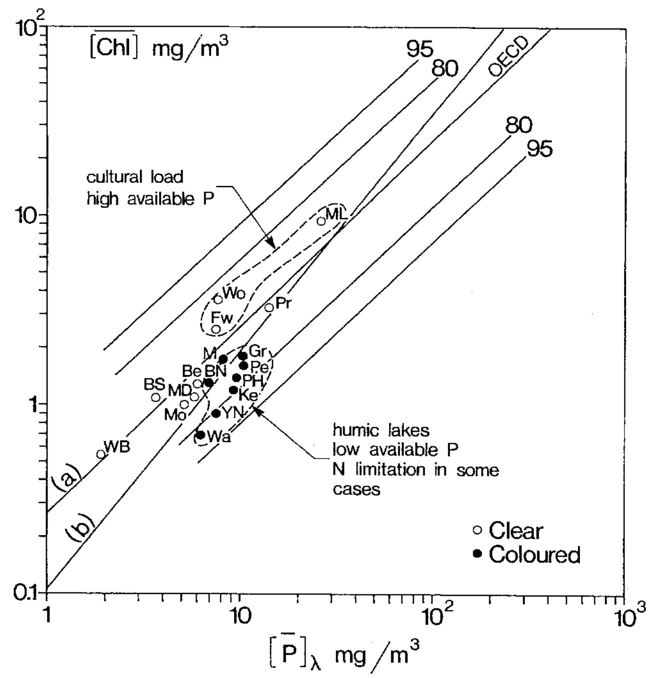

Figure I 2. Annual mean chlorophyll a concentration in relation to annual mean total phosphorus concentration.

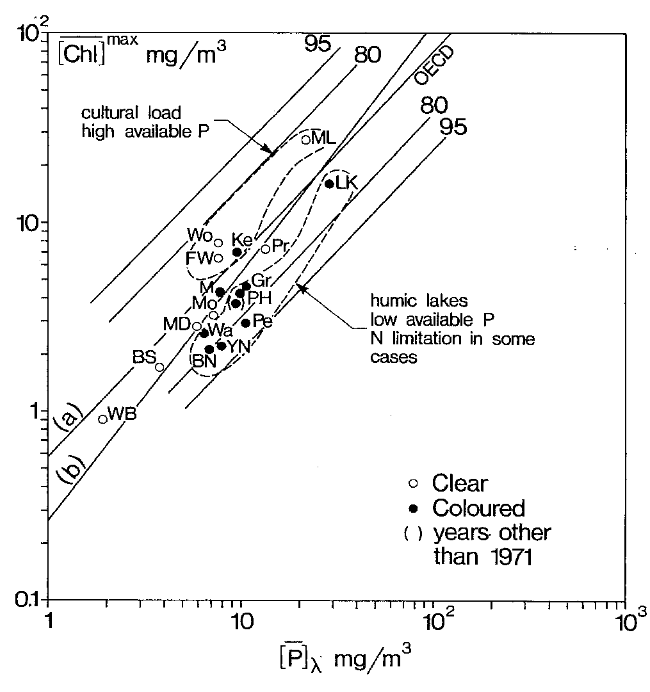

Figure I 3. Peak chlorophyll a concentration in relation to annual mean total phosphorus concentration.

Kejimkujik Lake, which is represented by two points on Figure I 3, can be viewed as a special case. An exceptional peak chlorophyll value of 6.8 mg/m^3 was measured in August, 1971, but in subsequent years the peak values were in the range of 3.5 - 4.1 mg/m^3. Until 1971, a sewage lagoon was drained into the lake in early August. The amount of phosphorus input attributed to this lagoon is estimated as approximately 1% of the annual total phosphorus load. Beginning in 1972 the draining of the lagoon was delayed until after the end of the growing season in late fall when the rate of water renewal is high. Apparently, the high peak chlorophyll observed in 1971 was a response to an increase in available phosphorus caused by the summer sewage discharge high in mineral phosphorus. The peak chlorophyll value fell somewhat above the expected OECD relationship in that year, while in subsequent years the peak chlorophyll values were below the OECD prediction similar to that of other coloured lakes in the Atlantic Region. Apparently, the timing of the discharge of the point source phosphorus load, to take advantage of favourable hydraulic and limnological conditions, was sufficient to eliminate an undesirable trophic response (i.e., high chlorophyll) without reduction of the annual phosphorus load.

It may be argued that lakes with high hydraulic load may have lower average chlorophyll concentrations, due to phytoplankton washout, than lakes with long water retention times. Undoubtedly, this mechanism plays some role in highly flushed Atlantic lakes. However, the distribution of the runoff is highly seasonal in the Atlantic Region and water renewal is at its lowest during the height of the growing season in July and August, when peak chlorophyll values occur. During this period, even in very highly flushed lakes, the rate of water renewal is minimal or non-existent. Thus, the high rate of water renewal may influence the average annual chlorophyll concentration, but would have very little or no effect on peak values. Since both mean and peak chlorophyll values are below the OECD prediction for coloured lakes with high and low water renewal, it can be concluded that the low amount of available phosphorus (and nitrogen limitation where applicable) in humic coloured lakes (rather than the high rate of flushing) keeps chlorophyll values below predictions for the Atlantic Region.

Clear-water lakes show good agreement with the overall OECD relationship.

1.2.2 Prediction of P Concentrations from Loadings. Figure I 4 shows the relationship between predicted phosphorus and measured lake phosphorus concentrations in Kejimkujik and Warren Lake. In both cases the average lake concentrations are higher (13 and 52%, respectively) than obtained from the standard prediction. Although the discrepancy may well be within the statistical uncertainty, particularly for Kejimkujik, a possible explanation other than internal loading (which loses importance in these highly flushed, oligotrophic lakes) may be given for this discrepancy. As mentioned earlier, a considerable portion of the total phosphorus is associated with dissolved organic substances in these humic lakes. This type of phosphorus behaves somewhat like a conservative element and is not eliminated by sedimentation at the same rate as that of available phosphorus which rapidly becomes associated with algal biomass. For this reason, lake concentrations of total phosphorus may lie above the standard prediction. This general trend can also be seen in other coloured lakes in the Atlantic Region.

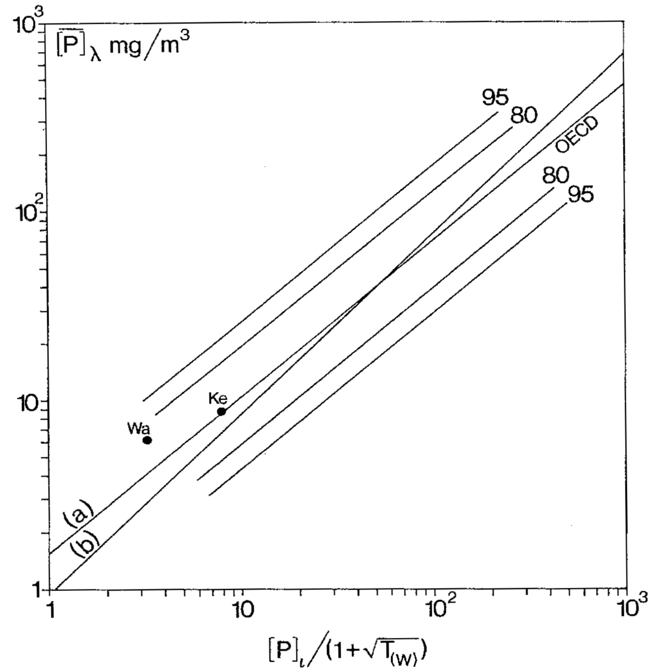

Figure I 4. Annual mean total phosphorus concentration in relation to the flushing corrected annual mean inflow total phosphorus concentration.

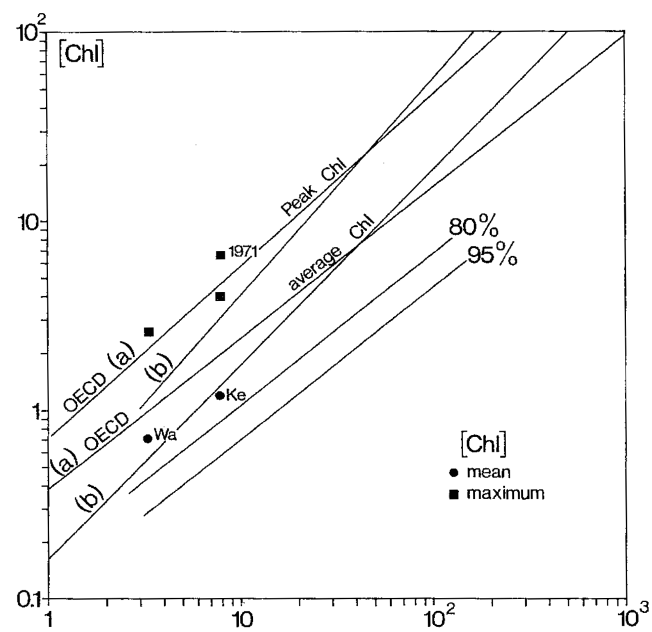

Figure I 5 Annual mean chlorophyll in relation to flushing corrected annual mean inflow total phosphorus concentration.

1.2.3 Prediction of Chlorophyll from Loading. The chlorophyll relationship to loading is the resultant of the previous two relationships and its appearance close to the OECD line is somewhat coincidental (cf. Figure I.5). This is due to the compensating effects of phosphorus concentration high in relation to loading, because of the conservative behaviour of phosphorus associated with dissolved organics, and low chlorophyll response in relation to total phosphorus because of a low percentage in available form. On an annual average basis for chlorophyll the latter effect outweighs the former, but it is interesting to note that in terms of peak values, the discrepancy of low chlorophyll response relative to total phosphorus,diminishes. This may be an indication that the inactivation of phosphorus through association with organics only occurs up to a certain capacity beyond which the percentage of the total which remains available increases. In the case of peak chlorophyll values, the compensating effects are more nearly equivalent and in relation to loading, fall very close to the OECD prediction.

1.3 Region I Conclusions

Non-humic lakes in the Atlantic Region seem to conform with the expected lake behavioural pattern established by the (nearly 150) OECD lakes.

Other lakes in the Atlantic Region exhibit particular properties, e.g. high colour caused by humic substances, which cause them to depart from the standard behavioural pattern derived from the OECD lakes. In these coloured lakes, a substantial part of the total phosphorus present is associated with dissolved organic substances and is not readily available for phytoplankton growth. For this reason, humic, coloured Atlantic Region lakes at a given phosphorus concentration maintain a lower algal biomass as indicated by chlorophyll, than that predicted from the OECD relationship.

Phosphorus associated with dissolved humic substances is not eliminated from a lake as readily as phosphorus associated with phytoplankton. Flushing corrected predictions underestimate lake concentrations of total phosphorus, but this compensates somewhat for the less mobile nature of phosphorus in humic lakes. Therefore, average chlorophyll values predicted from loading fall below but nearer the OECD relationship than those predicted from measured lake phosphorus concentration. Peak chlorophyll is even more closely predicted from loading than average values. This is a consequence of the equalization of the compensating effects. The proportion of total phosphorus which is available for plant growth may increase at times of peak biomass and reflect a finite binding capacity for inactivation of phosphorus.

Experience gained in one lake with a moderately high rate of flushing (6/yr) suggests that by properly timing a moderate amount of point source phosphorus discharge, the undesirably high peak chlorophyll levels can be prevented.

1.4 References (I)

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CHAPTER 2. QUEBEC REGION II

Lake Memphremagog

- North Basin (MN)
- Central Basin (MC)
- South Basin (MS)

Pink Lake (Pi)

Saint-Francois River Basin

- Aylmer (Ay)
- Bowker (Bo)
- Brompton (Br)
- Lovering (Lo)
- Magog (Ma)
- Massawippi (Ms)
- Montjoie (Mo)
- Petit Brompton (PB)
- Saint-Francois (SF)
- Stukely (ST)

Yamaska River Basin

- Boivin (Bv)
- Brome (Bm)
- Roxton (Ro)
- Waterloo (Wa)

2.1 Quebec Region, Description of Location

The Quebec region is one of rolling hills where glaciation has been the main force in shaping the present day landscape. The lakes lie in the southern portion of Quebec where various depressions were created by the scour of sedimentary and metamorphic bedrock and deposition of till upon recession of the last glacier.

The climate is "humid continental interior" (Landsberg, 1973) with cool summers and winters moderated through the influence of the Laurentian Great Lakes to the west. Snow and ice cover persists approximately four months until spring, when the heavy accumulation of the winter's snow is released to the lakes. Precipitation is, on the average, 100 cm per year and with the rather dramatic release of the winter's accumulation, lake levels may rise 1 to 2 metres.

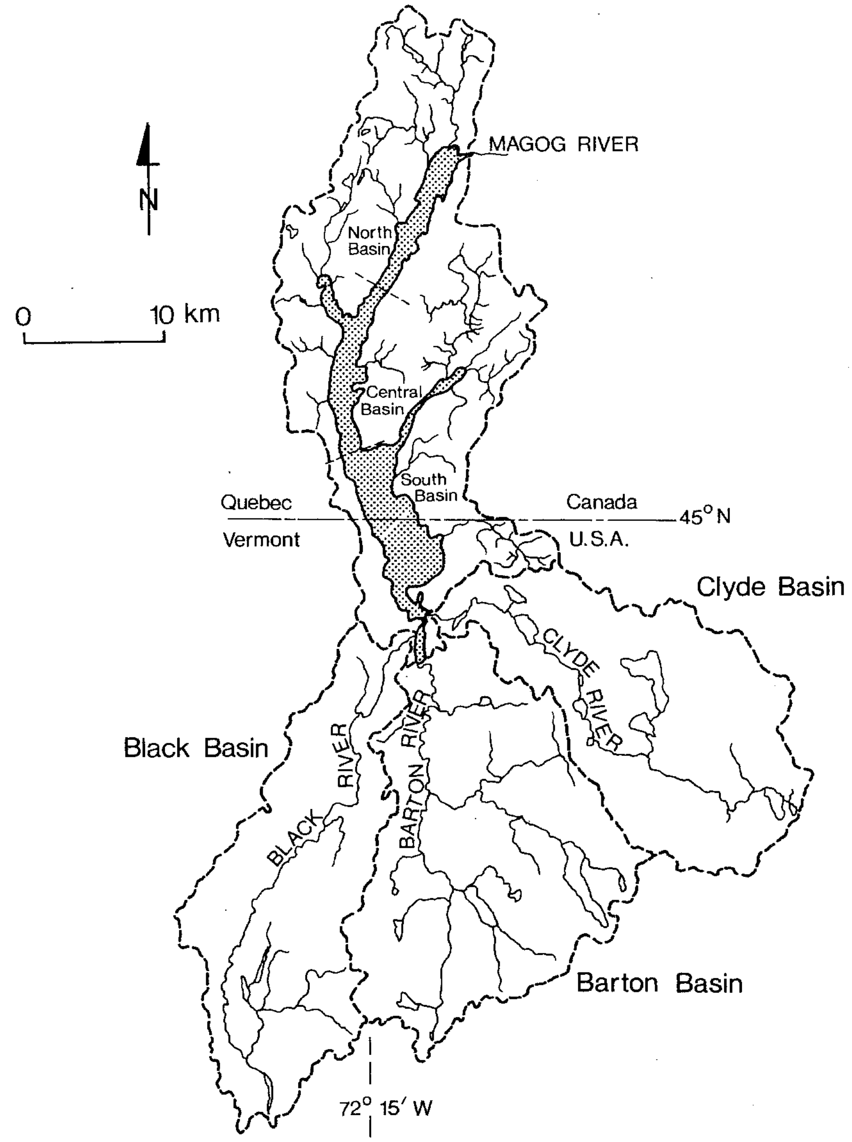
The soil which has developed under this condition of precipitation and the mixed deciduous-coniferous forest is a grey-brown podzol. A mull layer overlies a yellowish brown horizon which retains enough calcium that the soil is not excessively acidic. Although this soil is only of medium quality, the region is important agriculturally because of the plentiful moisture and 4 to 5 month growing season. The area is part of the hay and dairy belt and the abundance of hardwood trees promotes a local furniture industry.

Quebec lakes are mostly used as recreational sites and cottages and farms are the main sources of nutrients to the lakes in excess of natural supplies.

2.1.1 Loading Estimates and Trophic Status. The Lake Memphremagog catchment drains a land area of approximately 1780 km^2. The three major inflows to the lake, the Black, Barton and Clyde rivers, enter the lake at the southern-most end (cf. Figure II 1) and carry with them nutrient laden runoff from an extensive agricultural area, as well as untreated sewage from a number of small towns. Additional nutrients enter the lake nearly directly (at the mouth of the Clyde River) as sewage effluent from the primary treatment plant for the town of Newport (pop. 5,000). Because of the location of nutrient inputs, the southern area of the lake receives a phosphorus load estimated to be 3 times that received by the north or central regions. The remainder of the watershed is largely forested and sparsely populated.

The lake itself is long (≅ 40 km) and narrow (≅ 2.5 km) and morphometry of the basin is such that it is divided into three sub-basins. The southern basin is the most shallow ($\bar{z}$ = 6.9 m) with a surface area (4370 km^2) approximately twice that of each of the other basins. The central basin is the deepest ($\bar{z}$ = 50.9 m) and the northern basin intermediate in depth ($\bar{z}$ = 13.5 m). The ratio of volumes of the 3 basins is roughly 1:3:1 for the south, central and north, respectively.

Morphometric features in combination with nutrient loading have produced a distinct trophic gradient which decreases with distance from the southern, nutrient-rich inflows. Chlorophyll is notably higher (1.6 times) in the south as is macrobenthic biomass (≅ 3 times the northern value). The gradient is further reflected in the distribution of yellow perch in the lake, and biomass of this species was 2.5 times greater in



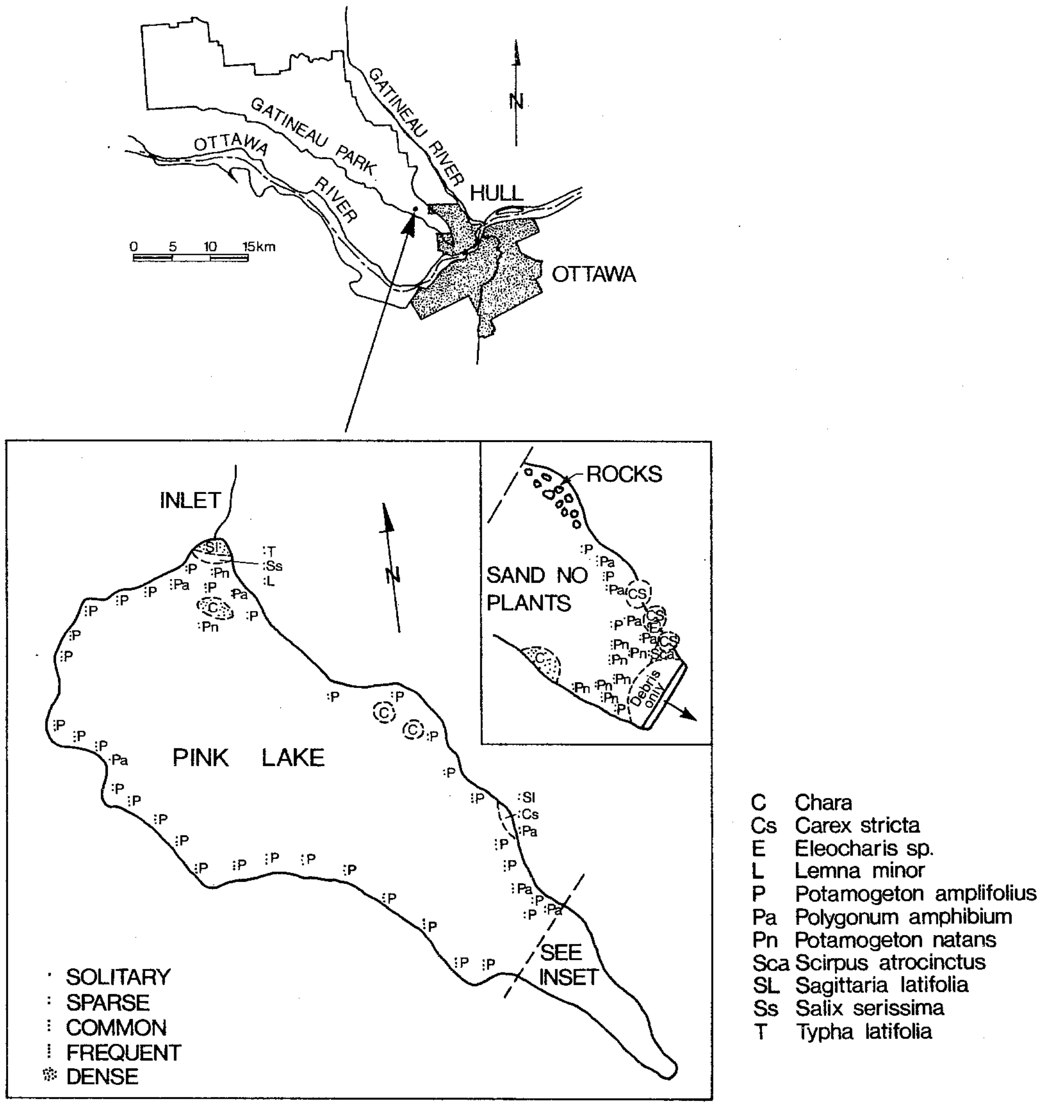
—from Spiller et. al., 1975

Figure II 1. LAKE MEMPHREMAGOG DRAINAGE BASIN the south than in the north. Accordingly, the three basins are treated separately in the section on nutrient loading - trophic response relationships.

Pink Lake is located in Gatineau Park and is surrounded by forest (cf. Figure II 2). It is unusual for lakes of the park in that it has a high calcium carbonate content and is the only alkaline lake there. It is a meromictic lake with moderately high epilimnetic nutrient (N and P) concentrations and even higher hypolimnetic concentrations. Cyanophyceae and Chrysophyceae dominate the summer phytoplankton and in July are often accompanied by a bloom of the green alga Lagerheimia, a species often found in enriched environments. These conditions have resulted in categorization of Pink Lake as one exhibiting natural eutrophy.

The remaining fourteen lakes considered here lie in two Québec river basins of the St. François and Yamaska Rivers. Lakes of the St. François basin receiving relatively high nutrient loads are found in proximity to areas under agricultural use, of high population density, or sometimes both (cf. Table II 1). These same factors influence the lakes of the Yamaska basin, but somewhat more intensively. The percentage of agricultural land surrounding these lakes is higher, hence specific loading estimates tend to be higher than those of the St. François basin. Variations in loadings to the lakes of these two river basins (due to the particular configurations of circumstances) are sufficient to present the entire range of trophic states.

The origin of the St. François-Yamaska River basin data is a thesis - by Potvin (1976) - wherein two methods of calculating loading are compared



—from Aiken and Gillette, 1974

Figure II 2 Location and macrophyte distribution in Pink Lake

Table II 1. Comparison of two phosphorus specific loading estimates and resultant inflow concentrations for Québec lakes.

	% agricultural land	population density † h/km²	L_p mg.m^{-2}.yr^{-1} land use	L_p mg.m^{-2}.yr^{-1} INRS-Eau census*	$[P]_i = \frac{Lp}{q_s}$ mg.m^{-3} land use	$[P]_i = \frac{Lp}{q_s}$ mg.m^{-3} INRS-Eau census*
St.-François River Basin						
Aylmer	12	22	1000	1720	27	47
Bowker	5	88	120	120	26	26
Brompton	5	28	190	320	29	48
Lovering	17	60	295	320	30	33
Magog	22	36	5000	4620	31	28
Massawippi	29	15	1100	1700	41	63
Montjoie	3	23	80	140	17	30
Petit Brompton	0	458	200	125	105	66
Saint-François	15	7	600	1130	35	66
Stukely	2	25	110	170	21	32
Yamaska River Basin						
Boivin	22	44	6200	7230	124	145
Brome	15	31	530	510	64	61
Roxton	21	100	400	650	69	112
Waterloo	20	15	910	1700	70	131

* INRS-Eau census - A modified L_p calculation based on specific details of INRS and Ministry of Natural Resources surveys.

\- Adapted from Potvin, 1976.

† Approximate due to seasonal changes

for these fourteen lakes. The essential difference in the loading estimates lies in the specificity of the initial information concerning nutrient sources. One method, the INRS-Eau 'census' method, is an attempt to improve the relatively coarse resolution of estimates derived from the second, more simplified 'land use' method. The 'census' method is based on more detailed accounts of human and animal populations, fertilizer use, inflow measurements, etc. In most cases 'land use' estimates are considerably lower than those derived from the more specific 'census' method. Results of both are presented in the figures which follow.

A few additional values of 1980 spring phosphorus and chlorophyll concentrations for some lakes of the two river basins were obtained from J. Cornett (pers. comm.) and are presented for comparison with the data from Potvin.

2.2 Trophic Response - Nutrient Relationships

2.2.1 Chlorophyll-Phosphorus Relationship. Chlorophyll means plotted in Figure II 3 tend to be high in relation to the OECD line, yet most lakes fall within the upper 80% confidence limit. If the most recent values from J. Cornett (pers. comm.) are taken, then only Waterloo remains above the 80% limit. It should be noted that the chlorophyll values available for these lakes (with the exception of Memphremagog and Pink) are means for the summer period only, and as such are not directly comparable to the full-year means used to establish the OECD line of Figure II 3. OECD data show that on the average, summer chlorophyll means are 1.5 to 2 times greater than annual means and this is largely why Quebec lakes show a generally high position.

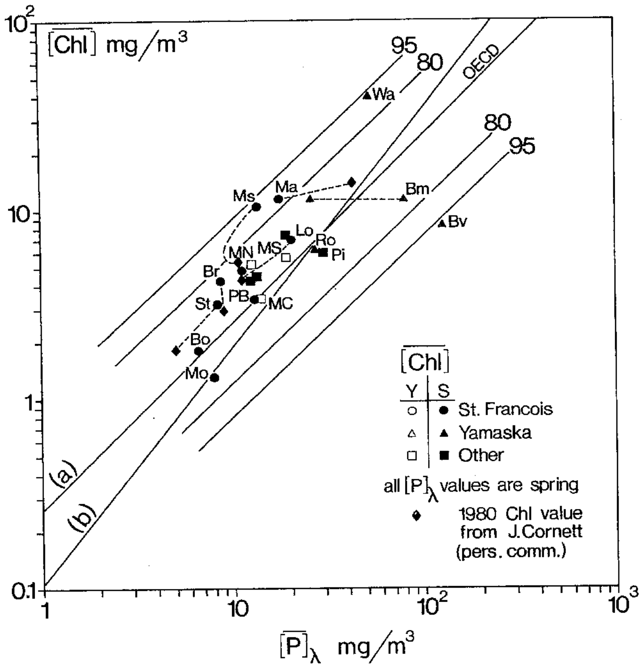


Figure II 3. Annual mean chlorophyll a concentration in relation to spring total phosphorus concentration

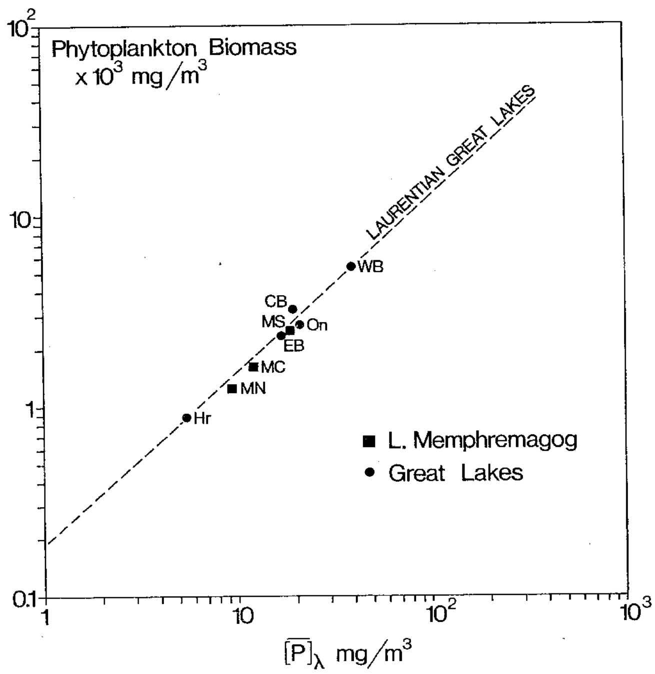


Figure II 4. Annual mean phytoplankton biomass in relation to annual mean total phosphorus concentration

More particularly, Magog, Massawippi and Waterloo obtain exceptionally high summer chlorophyll means, and these lakes receive the heaviest phosphorus loads which originate primarily from sewage discharge or agricultural activities (cf. Table II 1). Factors contributing to the high chlorophyll values observed might be the combination of long daylength and a high percentage of the total P in a form readily assimilated by phytoplankton. In addition, Waterloo is a shallow lake with a mean depth of only 2.9 m and residence of nutrients in the euphotic zone is probably longer than usual.

On the other hand, Boivin, although also heavily enriched, does not attain the high chlorophyll level exhibited by other lakes. This is most likely due to the dense macrophyte growth in this basin (mean depth is only 1.2 m). Efficient competition for nutrients and shading repress the phytoplankton response. An additional factor might be nitrogen limitation. Bioassay results (of N and P additions) and the low N:P ratios found during the spring and summer months indicate that this occurs (P. Campbell, P. Conture and D. Cluis, pers. comm.) and as stated in the thesis by Potvin, may be responsible for the low chlorophyll mean of Boivin. Caution must be used in interpretation of bioassay results, though, since this procedure eliminates the effect of the macrophyte community and may be out of context for Boivin.

2.2.2 Phytoplankton-Phosphorus Relationship. Data for the three basins of Lake Memphremagog lie along the same line as that described by the Laurentian Great Lakes biomass-P relationship. 10 mg TP corresponds to approximately 1500 mg phytoplankton freshweight which implies that

3 to 4 mg of a total of 10 mg P are fixed within phytoplankton cells (given that a 10%/volume carbon content and C/P ratio of 40 applies (cf. Figure II 4). In addition, 1 mg/m³ of chlorophyll corresponds to approximately 600 mg/m³ of phytoplankton which is twice that of the values found in the Alpine investigation. (Further discussion of this topic appears in an addendum to Chapter 5.).

2.2.3 Phosphorus-Loading Relationship. The majority of lakes fall within the 80% confidence interval of the OECD relationship, regardless of which of the two methods is used for estimation of loading (cf. Figure II 5). As previously mentioned, the INRS-Eau 'census' method tends to give values somewhat higher than those obtained from 'land use' estimation and as a result, two lakes (Br and Ms), lie outside the 80% limits and may be overestimates. On the other hand, the one lake (PB) lies outside the 80% limits according to the 'land use' estimate and this is most likely an underestimate. It is probable that the most realisic loading values lie somewhere between the two estimates.

The lake which appears to deviate most in phosphorus concentration relative to its loading is Brome, which lies above the upper 95% boundary. The high 1975 lake concentration (80 mg/m³) cannot be explained by flushing corrected inflow concentration. In 1976, the spring P concentration in Brome was measured at 25 mg/m³ rather than 80 mg/m³, and some doubt about the validity of the 1975 value has been expressed (Potvin, 1976). The 1976 value is very close to prediction from the general relationship. Other lakes with P concentrationsthat tend to be high relative to loadings are Petit Brompton, Lovering,Magog and Montjoie.

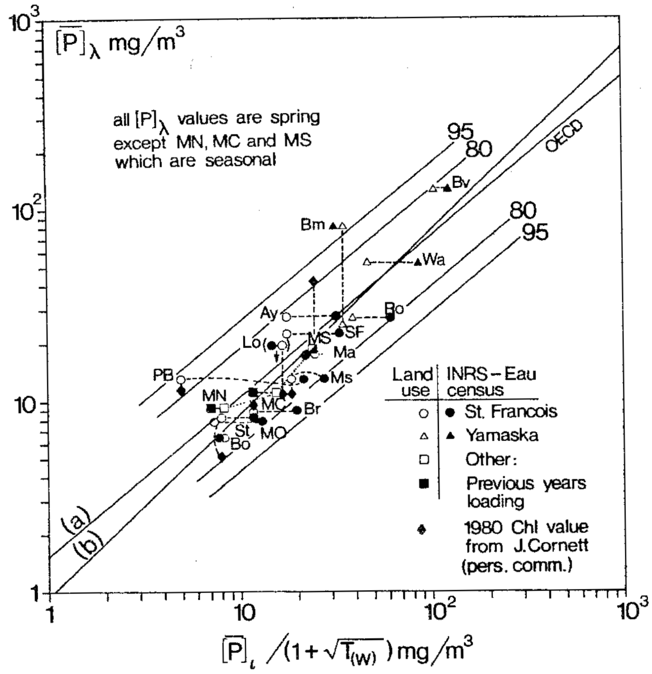


Figure II 5. Spring total phosphorus concentration in relation to flushing corrected inflow total phosphorus concentration

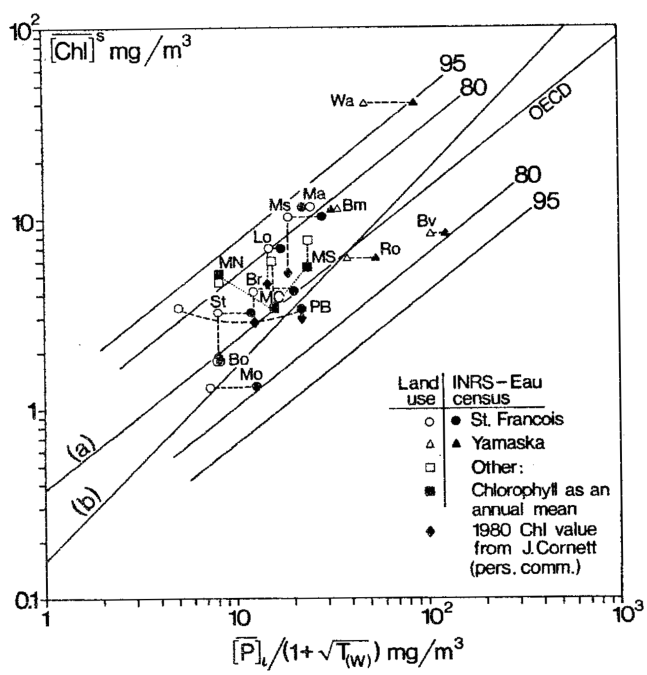


Figure II 6. Mean summer chlorophyll a in relation to flushing corrected inflow total phosphorus concentration

These lakes (including Brome) are known to undergo anoxia at times and phosphorus feedback from the sediments may be responsible for the high position of lake concentrations. Indeed, the high phosphorus concentration (reported by Cornett) for Magog in combination with compliance to the general chlorophyll-P relationship seems to indicate that the total loading for this lake has been underestimated. (Magog receives substantial sewage effluent and remained anoxic summer and winter from 1976 to 1980, P. Potvin, pers. comm.). P feedback from sediments may not be uncommon.

2.2.4 Chlorophyll-Loading Relationship. Chlorophyll values obtain a consistently high position in relation to loading (cf. Figure II 6) and as explained earlier, the fact that these values are summer, rather than annual chlorophyll means, may be at least partial explanation for this general shift of position. (Also, as described in the preceding section, the higher than expected phosphorus levels in some lakes contribute to this). The higher loading values of the 'census' method tend to compensate for the high chlorophyll values, and therefore, lie closer to the OECD line than 'land use' estimates. On the other hand, the ice-free season chlorophyll means from Cornett are very much closer to expectation and may indicate methodological overestimation in the other values.

The high chlorophyll means (both summer and annual) for Lake Memphremagog's north basin may be the effect of wind and flow pattern. There seems to be an export of algal cells produced in the more heavily loaded southern basin to the more lightly loaded northern outflow. The exceptionally high chlorophyll-P relationship in Waterloo (discussed earlier) is also evident here.

2.2.5 Primary Production in Relation to Loading. Primary production data is only available for Lake Memphremagog (cf. Figure II 7). Both the north and south basins fit well with the expectation provided by the OECD relationship. The tendency for production to be high in the north with respect to its loading is supported by the chlorophyll data (also high in relation to phosphorus concentration or loading) and may be the effect of transport of the algal growth response in the southern nutrient-rich basin to the northern outflow which receives lower loading.

2.2.6 Secchi Transparency in Relation to Chlorophyll, Phosphorus and Loading. Lake Memphremagog and Pink Lake lie close to the OECD line, whereas all Secchi readings for lakes of the two river basins are less than would be expected on the basis of chlorophyll, P or flushing corrected inflow concentrations (cf. Figures II 8, II 9, and II 10). Some lakes of the river basins have a silty appearance from clay erosion and some receive dyes from the effluent of the paint manufacturing industry (Carignan, pers. comm.). In general, transparency of the water is controlled by factors other than phytoplankton but lack of quantitative information regarding colour and turbidity prevents further exploration of these factors as explanation.

2.3 Region II Conclusions

In general, the Quebec lakes do not essentially differ from the OECD experience, but a few 'special' cases can be identified by the comparison. Deviations appear to be related to extreme properties in the physical characteristics of the lakes in question which may cause highly

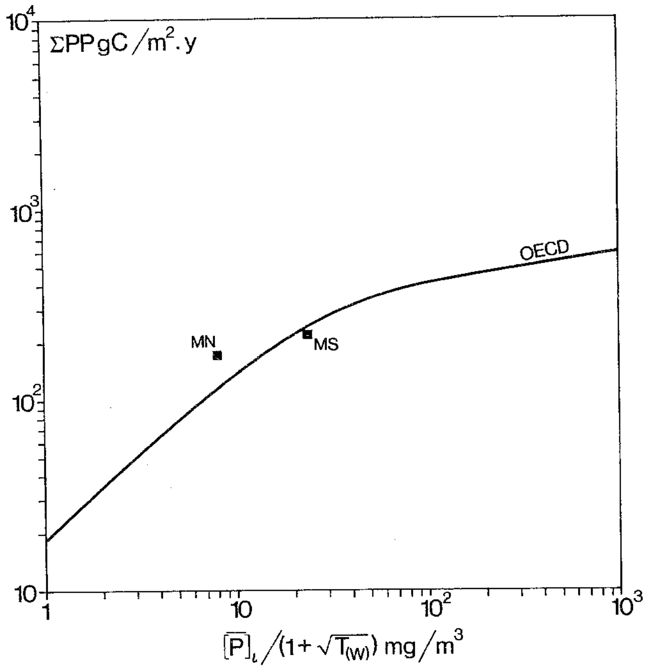


Figure II 7. Annual areal primary production in relation to flushing corrected inflow total phosphorus concentration

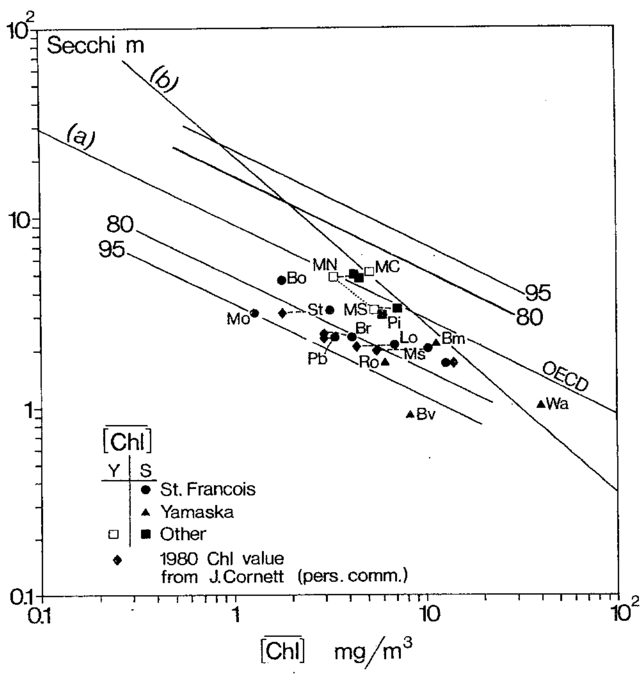


Figure II 8. Secchi transparency in relation to mean chlorophyll a concentration

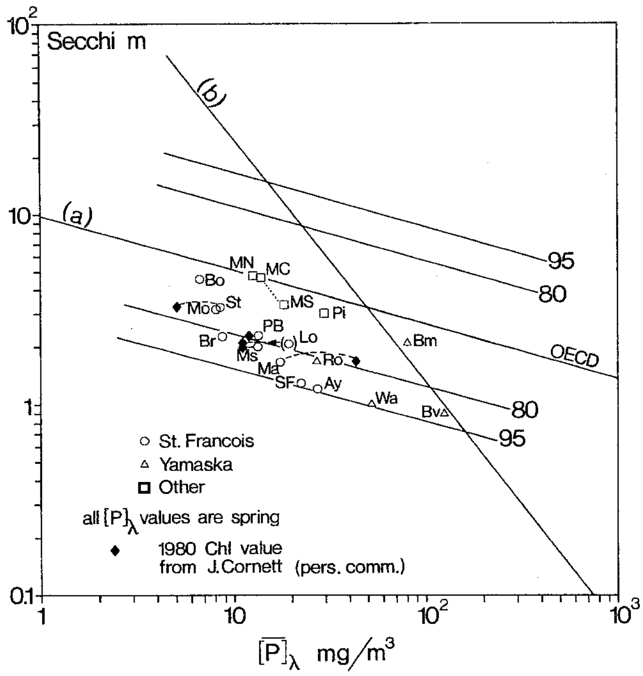


Figure II 9. Secchi transparency in relation to spring total phosphorus concentration

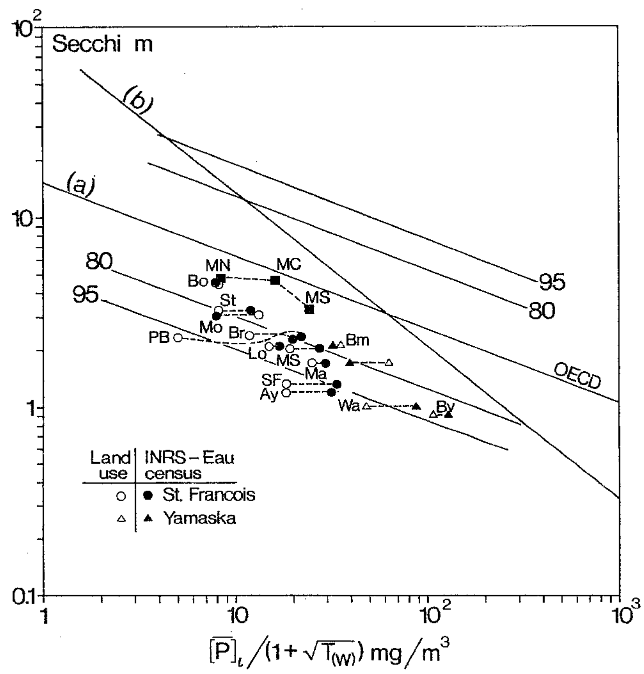


Figure II 10. Secchi transparency in relation to flushing corrected inflow total phosphorus concentration varied effects. A shallow basin may be responsible for a higher than usual chlorophyll response to phosphorus concentration (as in Waterloo) or may cause the opposite effect of low chlorophyll response. Low chlorophyll may be observed if phytoplankton lose importance in the plant community structure (as in Boivin) and macrophytes dominate.

Overall, chlorophyll relationships are not entirely representative because of the fact that the means available are those for the summer period and not the full year. For this reason they tend to be high relative to the OECD prediction for phosphorus concentrations. Despite the tendency for chlorophyll values to be high, nitrogen limitation and/or nutrient competition in combination with shading by macrophytes may be adequate to offset this, resulting in lower than expected chlorophyll means (as in Lake Boivin).

Problems with loading estimate methodology are apparent. Comparison of the more common 'land use' estimates and a more detailed 'census' method may differ by as much as 1:2 (average 1.4 $\pm$ 0.4). In terms of OECD lakes, loading seems to be underestimated by the simplified 'land use' procedure and overestimated by the detailed accounting of sources in the 'census' method. More realistic estimates probably lie somewhere between the two and in fact, the most recent communication with the authors states that export coefficients for forest land may have been overestimated in the original version of the 'census method. Differences in the estimates pose difficulties in relating these to lake response, but the predicted patterns appear to emerge more clearly with advances in estimation procedures and update of annual means.

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On the other hand, Muskoka County lakes are influenced by cottages and municipal and industrial development (cf. Table III 1, following). Effects of these inputs became evident in the mid-1960s when deterioration of water quality and algal blooms were reported in Gravenhurst Bay, Lake Muskoka and Little Otter Lake (cf. Fig. III 1). Gravenhurst Bay received 92% of its total P load through discharge from the Ontario Fire College and Gravenhurst and Ontario Hospital sewage treatment plants. In 1971 80% P removal was effected in these plants, greatly reducing the P load. Most cottages around Lake Muskoka are served by septic tank-tile bed systems, but much of the high-phosphate dishwater and laundry wastewaters and lawn fertilizer leachates bypass these. The Harp and Jerry Lake study was intended as a comparison of cottaged and uncottaged lakes (cf. Fig. III 2) to determine the influence of cottages in phosphorus loading and the extent to which export of this to the lakes should be expected. Since the sandy soils of the area retain only small amounts of phosphorus, they provide little protection from these inputs; hence, in 1971 the Ontario Ministry of Environment recommended a minimum of lawn fertilization and use of low phosphate cleaning compounds to residents. In Little Otter Lake (cf. Fig. III.3), a dense *Anabaena limnetica* bloom occurred in 1971 when Rockwell International (an automobile parts manufacturer) discharged a polyphosphate de-scaling agent into the lake. This was discontinued in 1972 and transparency, phosphorus, chlorophyll and algal levels returned to normal. Fortunately, flushing rates of these lakes are relatively high allowing them to respond more readily than they otherwise would to these abatement measures. As exemplified by the Atlantic Region case, attention to time coordination of maximal inputs

Table III 1. Phosphorus loading to Muskoka lakes.

Lake	L_P $(mg/m^2/y)$	land use (dwellings etc.)	τ_w (years)
Gravenhurst Bay	2370	5900 municipal	1.8
		300 cottages	
Skeleton Bay	1026	45 cottages	0.2
Muskoka	747	13460 municipal	0.9
		5000 cottages	
		4 resorts	
Jerry	316	1 cottage	1.5
Little Otter	238	20 commercial units	0.1
		17 cottages	
Little Joseph	210	81 cottages	2.3
Dudley Bay	195	210 cottages	1.4
Simcoe	190		20.9
Harp	180	75 cottages	3.5
Rosseau	153	230 municipal	4.8
		1690 cottages	
		3 resorts	
Joseph	76	1100 cottages	20.9
		2 resorts	

CHAPTER 3. ONTARIO SHIELD LAKES REGION III

A. Haliburton County

Beech	Be	Four Mile	FM	Raven	Ra
Bob	Bo	Halls	Ha	Talbot	Ta
Cameron	Ca	Maple	Ma	Twelve-Mile-Boshkung	TM-B
Cranberry	Cr	Oblong-Haliburton	O-H		
Eagle-Moose	Ea-Mo	Pine	Pi		

B. Muskoka County

Dudley Bay	DB	Joseph	Jo	Rosseau	Ro
Jerry	Je	Little Joseph	LJ	Simcoe	Si
Harp	Hp	Little Otter	Lo	Skeleton Bay	SB
Gravenhurst Bay	GB	Muskoka	Mu		

C. Algonquin Park

Brewer	Br	Kearney	Ke
Clarke	Cl	Little McCauley	LM
Costello	Co		
Found	Fo		

D. Haliburton-Muskoka

Basshaunt	Ba	Crosson	Cs	Jerry	
Bigwind	Bi	Dickie	Di	Little Clear	LC
Blue Chalk	BC	Glen	Gl	Red Chalk	RC
Buck	Bu	Gullfeather	Gu	Solitaire	So
Chub	Ch	Harp	Hr	Walker	Wa

3.1 Ontario Shield Region, Description of Location

These lakes lie in the 'eastern wet region' of North America (Bryson and Hare, 1974) and annual precipitation is 80 to 100 cm. The soils are poorly drained and sphagnum bogs are common with the result that many of the lakes are 'brown-water' lakes.

This Shield region is characterized by a rugged, rocky landscape. The irregular granitic basins which these lakes occupy have highly convoluted shorelines and numerous islands. They are typically deep relative to surface area and the dilute soft water conditions support the plant or animal communities associated with oligotrophy. The most obvious macrophytes include *Isoetes*, *Nuphar*, *Nymphaea*, *Potamogeton* and *Fontinalis* and the usual catch of the sports fisherman is trout and smallmouth bass.

In the 1970s, year-round occupancy and population growth has led to cultural enrichment and, as is commonly the experience, bloom-forming Cyanophyceae (*Anabaena*) have appeared (Michalski and Conroy, 1973). Formerly, no such component was found amongst the typical Shield assemblage of Chrysophyceae and Diatomae with the more seasonal growth of Cryptomonads and Chlorophyceae. Public concern for the obvious changes in tropic condition of these 'vactionland' lakes has been the impetus for collection of much of the data used in this analysis.

3.1.1 Phosphorus Loadings. Phosphorus loadings to Ontario Shield lakes are generally those occurring naturally. Lakes of Haliburton County and Algonquin Park are surrounded by boreal forest, used solely for recreational purposes, and as such, are devoid of point source loadings.

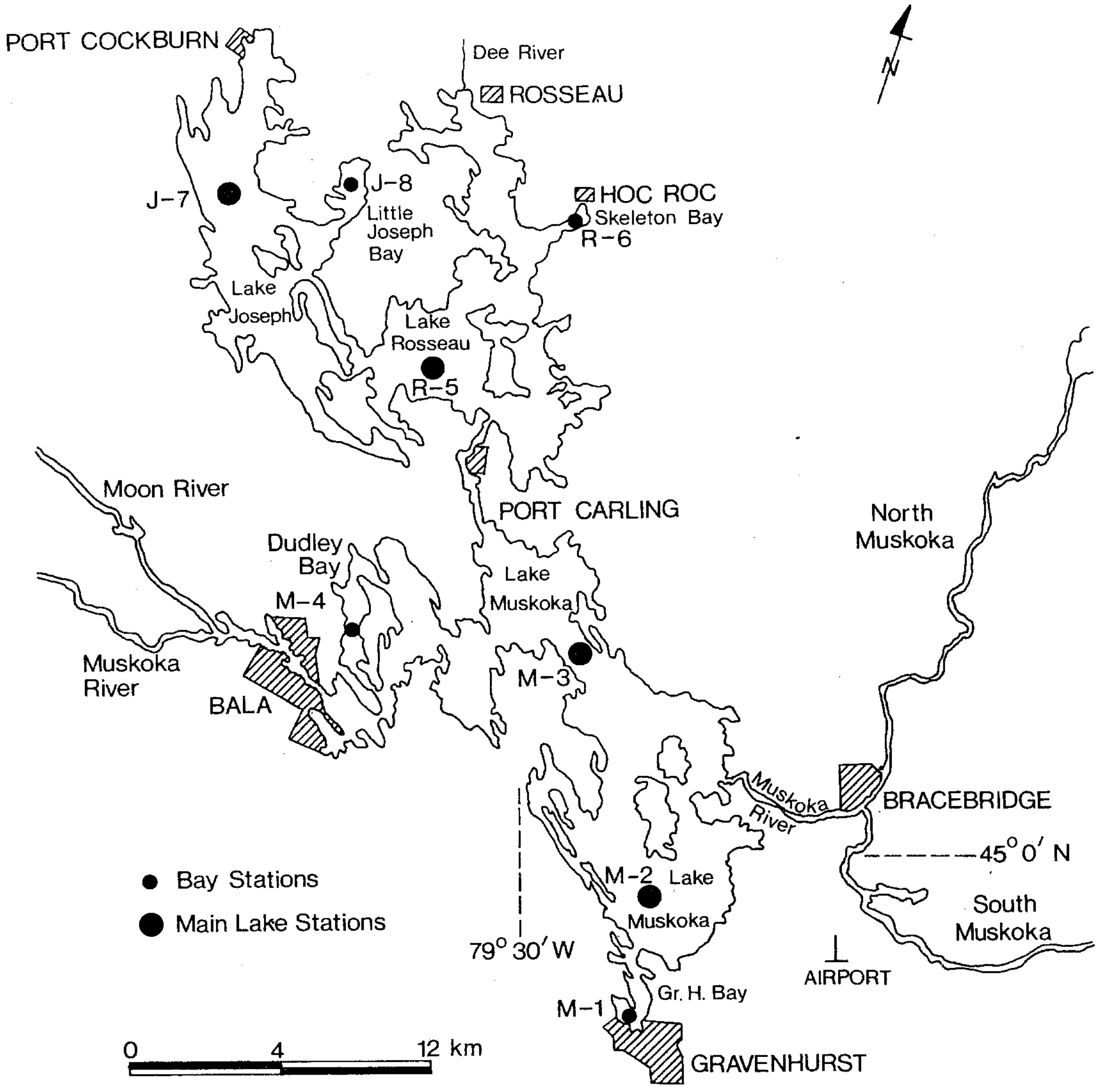

Figure III 1 MUSKOKA LAKES AND THE TOWNS THAT INFLUENCE THEM.

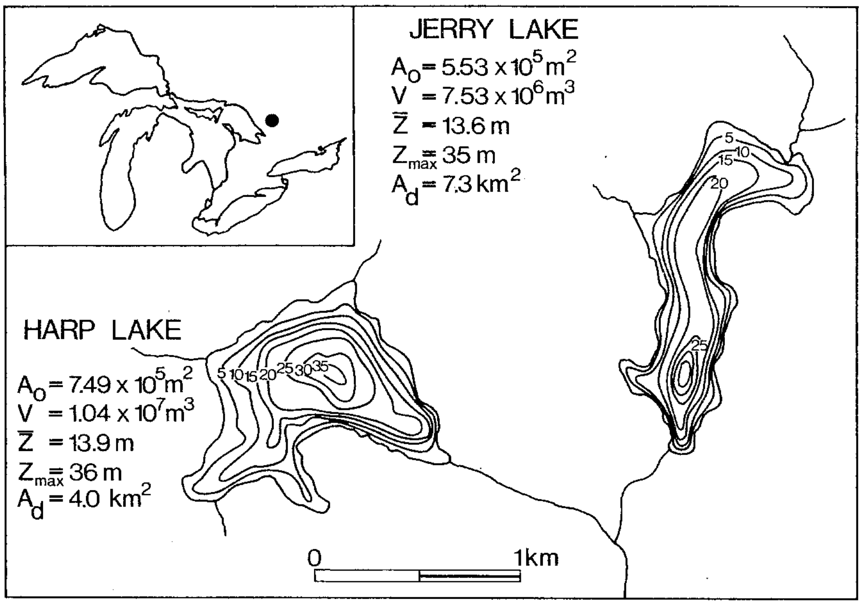

Figure III 2. Harp and Jerry Lakes

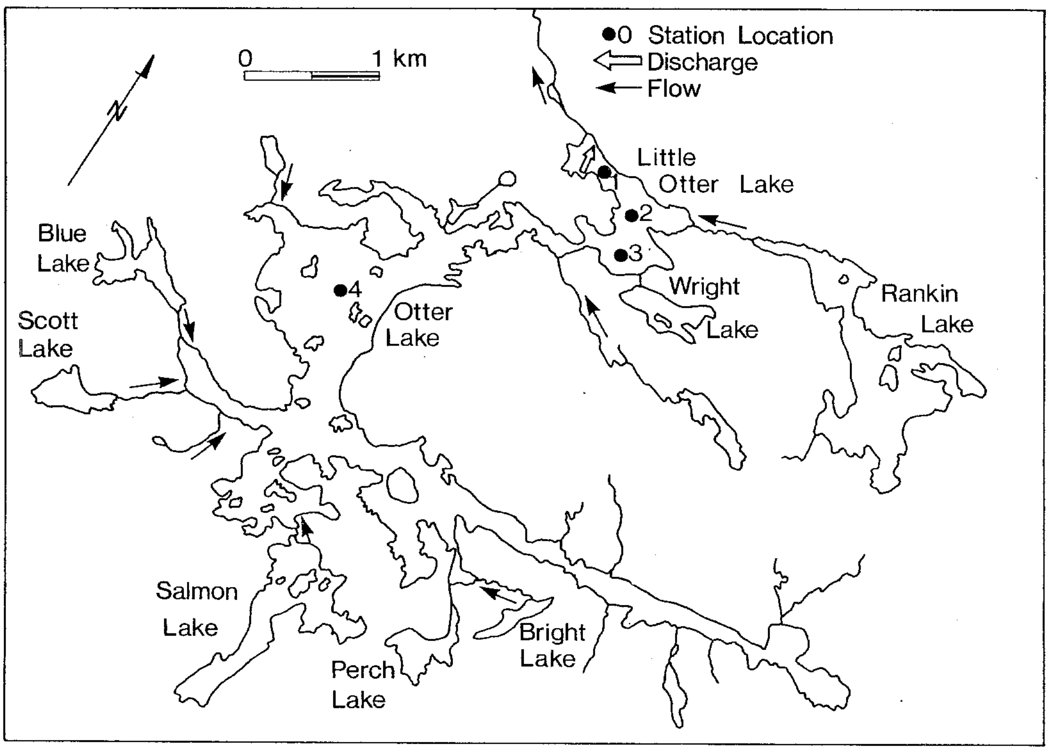

Figure III 3. Otter - Little Otter Lake watershed with periods of high flushing rate may be effective in dampening the trophic response which might otherwise be expected.

In the following discussion, the most recent data of a four-year study is treated separately from the less systematic data extracted from a variety of sources.

3.2 Trophic Response-Nutrient Relationships

3.2.1 Chlorophyll-Phosphorus Relationship. Chlorophyll is plotted against spring phosphorus (at overturn) and annual mean phosphorus in Figs. III 4, III 5, III 6, and III 7. In relation to spring phosphorus, (Fig. III 4) chlorophyll values appear to be low in comparison with the OECD relationship, particularly considering that many of the chlorophyll averages used are those for the growing season rather than representations of the full year. On the other hand, the more recent data of a four-year study of lakes in the same region (Fig. III 5) appears to have high chlorophyll levels in relation to spring phosphorus. Although less scattered when chlorophyll is plotted against the mean phosphorus concentration for the year (Figs. III 6 and III 7), the same relative position of the data sets persists. This may be an indication that chlorophyll extraction procedures have been more complete in the recent study. Yearly values all fall within the 80% margin and the tighter clustering of this data indicates that chlorophyll is better predicted by yearly mean phosphorus concentration than the single value spring phosphorus concentration. An additional factor causing low chlorophyll response

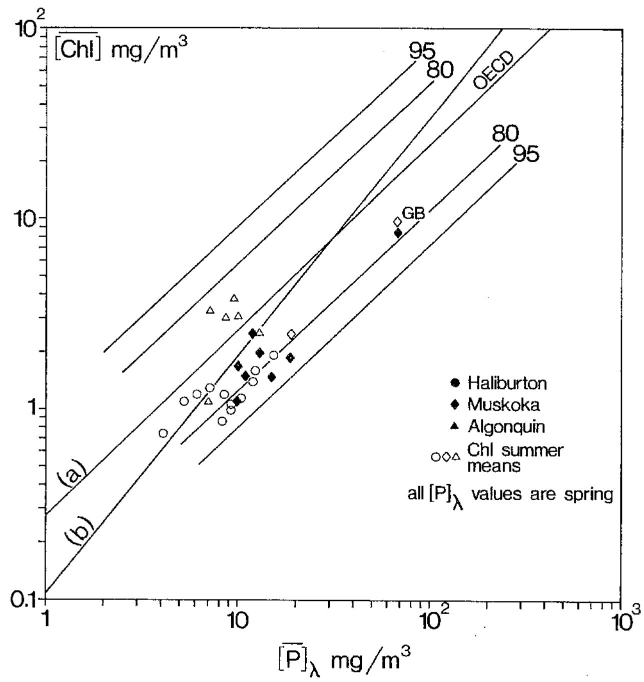

Figure III 4. Annual mean chlorophyll a concentration in relation to spring total phosphorus concentration

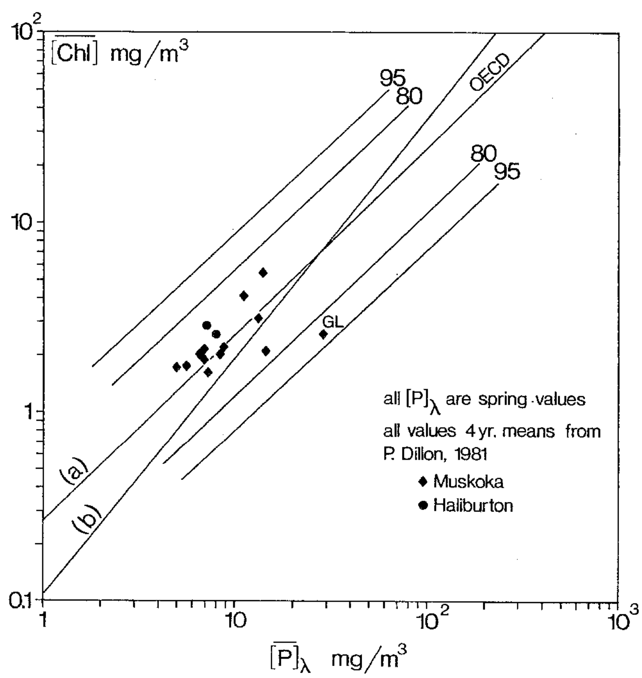

Figure III 5. Annual mean chlorophyll a concentration in relation to spring total phosphorus concentration, 4 year means

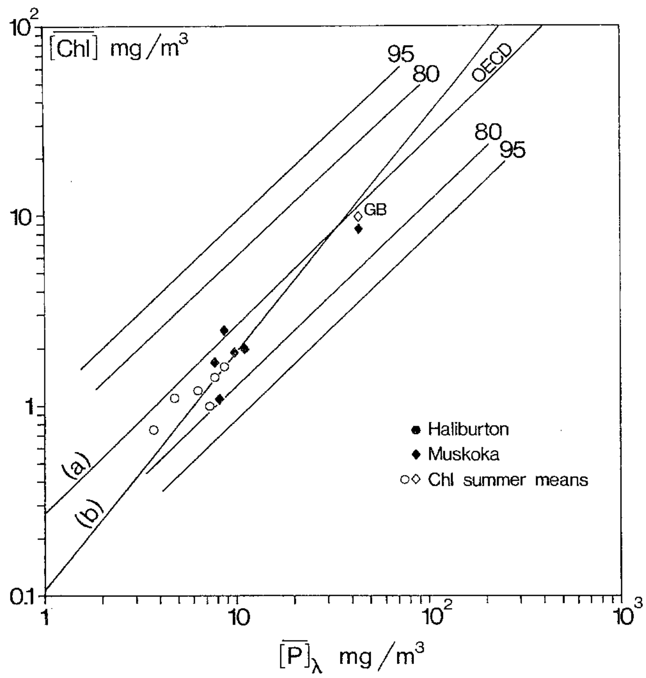

Figure III 6. Annual mean chlorophyll a concentration in relation to mean total phosphorus concentration

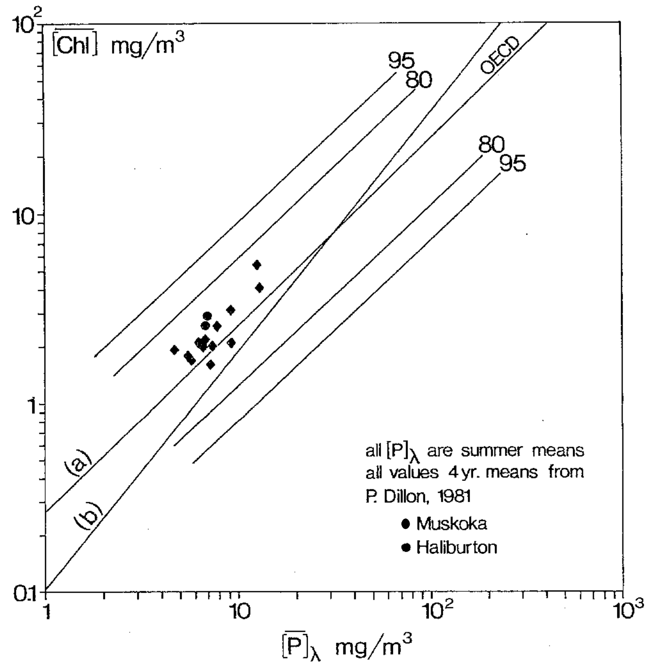

Figure III 7. Annual mean chlorophyll a concentration in relation to mean total phosphorus concentration, 4 year means to phosphorus may be light limitation by dissolved humic substances.

3.2.2 Chlorophyll-Nitrogen Relationship. In the Muskoka lakes (and two Haliburton lakes) there appears to be no relationship of chlorophyll to inorganic nitrogen (Fig. III 8) or 4-year means of total nitrogen found during the growing season (Fig. III 9). Lake N : P ratios are high (on the average approximately 30 : 1) and are are in excess of our assumed biomass requirement of approximately 10 : 1. Gravenhurst Bay with an N : P ratio of 11 seems the only candidate for possible nitrogen limitation, but even this situation has probably changed since 1971 when pre-treatment of wastewaters for phosphorus removal began, N : P ratios have probably resumed a higher value and N limitation would not be expected.

3.2.3 Phosphorus-Loading Relationship. Both spring and yearly mean phosphorus concentrations tend to be low in relation to loadings when compared with OECD findings with, as expected, yearly values consistently below those for the spring (cf. Fig. III 10). To retain perspective, it should be remembered that the OECD reference was established with a majority of lakes having had some previous history of enrichment contemporaneous with land development. In lakes which have been fertilized for several consecutive years, a high percentage of the binding capacity of sediments may be charged . The equilibrium between concentration and input load in these lakes maintains a position higher than could be expected for the unexposed, oligotrophic lakes of the shield where binding capacity of the sediments is high (cf. also Chapter V). Consequently,

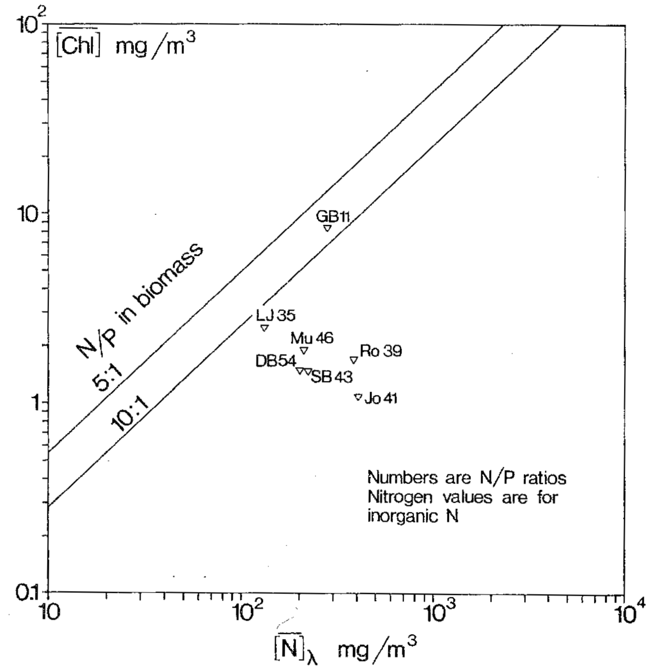

Figure III 8. Annual mean chlorophyll a concentration in relation to mean inorganic nitrogen concentration

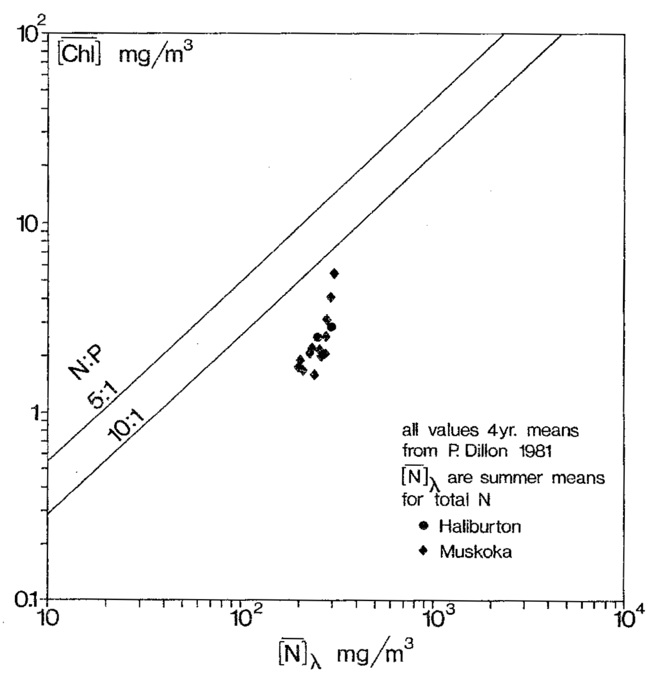

Figure III 9. Annual mean chlorophyll a in relation to total nitrogen concentration of the growing season, 4 year means

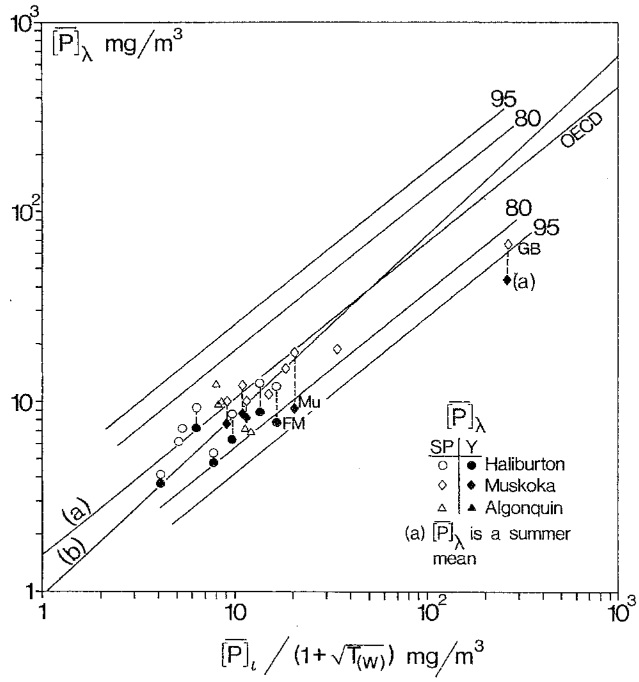

Figure III 10. Annual and spring total phosphorus concentration in relation to flushing corrected inflow total phosphorus concentration

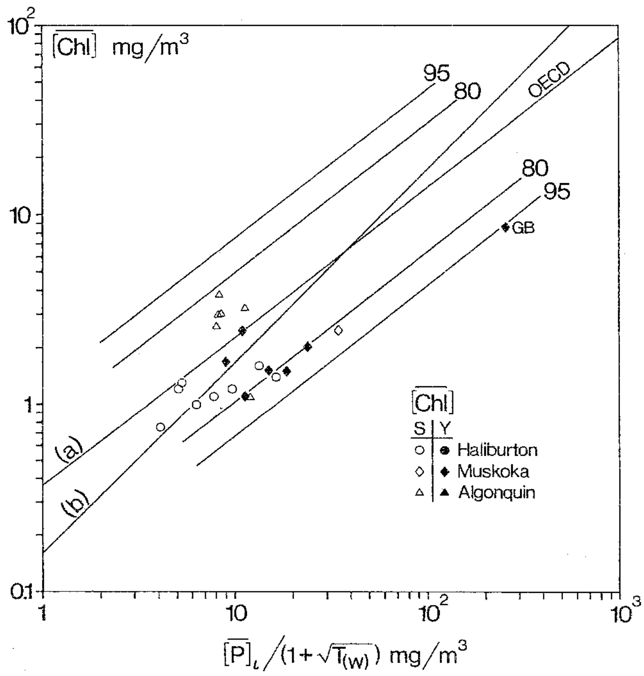

Figure III 11 Annual and summer chlorophyll a concentration in relation to flushing corrected inflow total phosphorus concentration measured phosphorus concentrations are overestimated by flushing corrected inflow concentrations and sedimentation (rather than internal loading) is a dominant process in these lakes.

In the particular case of Gravenhurst Bay, phosphorus concentrations (both spring and summer means) are considerably below what one would expect on the basis of the high loading this bay received prior to abatement measures. Although this result resembles that for the artificially enriched lakes of the ELA (cf. Chapter V), it should be noted that the mode of enrichment is quite different in the two situations. In the ELA lakes, fertilization took place at the surface and low resultant phosphorus concentrations were due to rapid fixation in the sediments. In Gravenhurst Bay, it is possible that nutrient rich wastewaters from treatment plants introduced into the bay's hypolimnion, never mixed into epilimnetic waters. This might explain why phosphorus values of these two situations appear similarly low in relation to loading.

3.2.4 Chlorophyll-Loading Relationship. The general trend is for chlorophyll values to be low in relation to flushing corrected phosphorus inputs from the Haliburton and Muskoka lakes (cf. Fig. III 11). Although most points lie within or near the 80% limits, Gravenhurst Bay lies just beyond the lower 95% limit. The generally low chlorophyll response mirrors the concentration-loading relationship. In contrast, Algonquin lakes lie mostly above the OECD line, but this is the consequence of the relatively high concentrations of chlorophyll observed in relation to phosphorus (cf. III 4). (It is notable that chlorophyll extractions of the Algonquin lake study included a grinding procedure and may be higher for that reason.)

3.2.5 Secchi Transparency in Relation to Chlorophyll, Phosphorus and Loading. Secchi disc readings plotted against mean chlorophyll concentrations (cf. Figs. III 12 and III 13) nearly all fall within the 80% limits of the OECD line. It is notable that both summer and yearly means for chlorophyll are low, and despite use of the summer mean when phytoplankton growth is greatest, transparency is controlled by factors other than chlorophyll. Mean depth of some lakes (notably Raven and Talbot) is less than 1 m and flushing is high (> 5 times per year), but resuspension or influx of silt cannot explain the low position of Secchi transparency either, since the average transparency for lakes of the region flushed in excess of even 10 times per year is approximately 5 m. The tendency for readings to be low is most likely a result of humic colouring.

With regard to the relationship between Secchi depth and spring phosphorus concentration, there is good agreement with the OECD findings (cf. Fig. III 14). Slightly less scatter in the relationship results when Secchi depths are plotted against the more stable summer or yearly means for phosphorus (cf. Fig. III 15 and III 16) and with only two exceptions (cf. Fig. III 14), lakes lie well within the 80% limits. Talbot and Little Otter deviate most from the OECD prediction of Secchi depth from spring [P] and lie near the upper and lower extremes of the limits, respectively (cf. Fig. III 14). In each case, spring phosphorus concentration is uncoupled from the factors most important in decreasing transparency. For Talbot Lake, the high flushing rate (5 times per year),

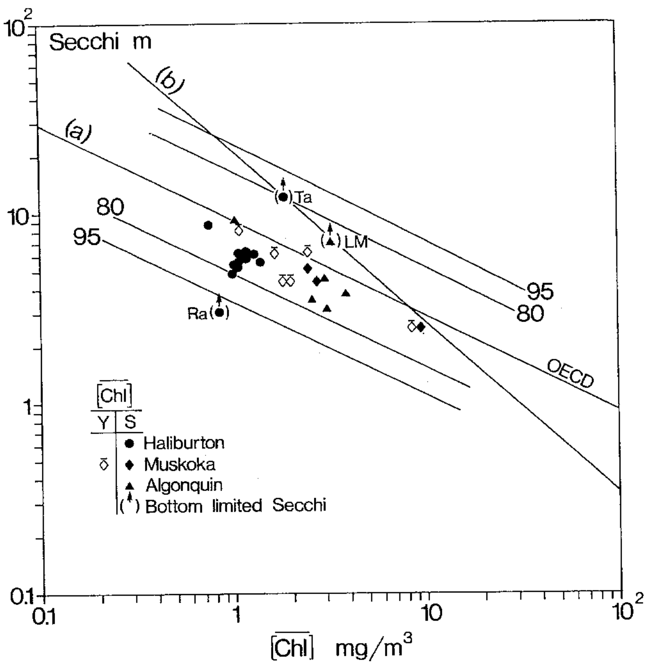

Figure III 12. Secchi transparency in relation to annual and summer chlorophyll a concentration

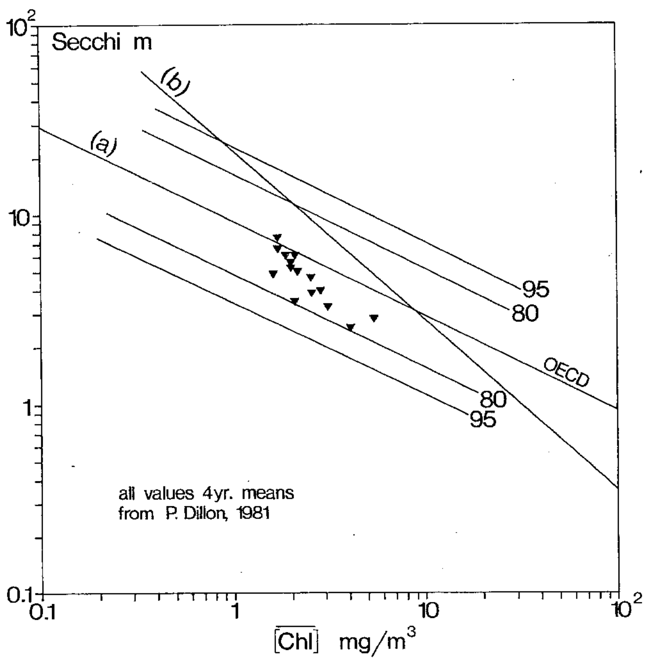

Figure III 13. Secchi transparency in relation to annual chlorophyll a concentration, 4 year means

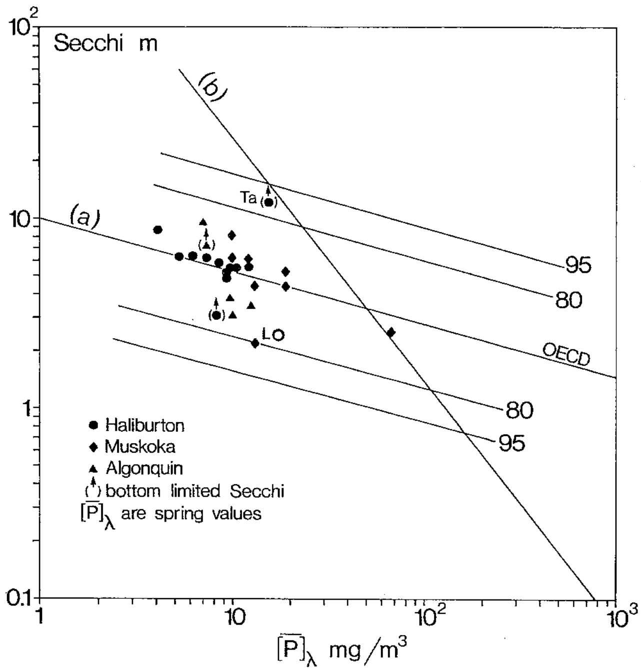

Figure III 14. Secchi transparency in relation to spring total phosphorus concentration

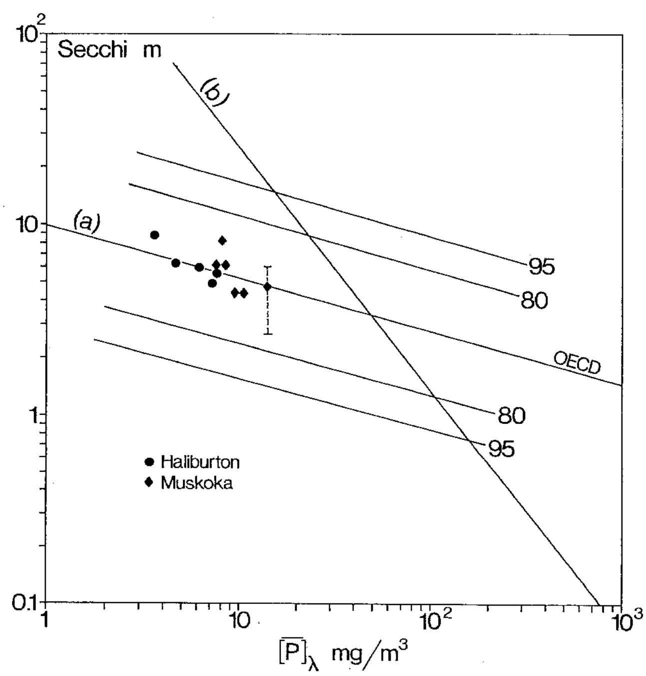

Figure III 15. Secchi transparency in relation to annual mean total phosphorus concentration

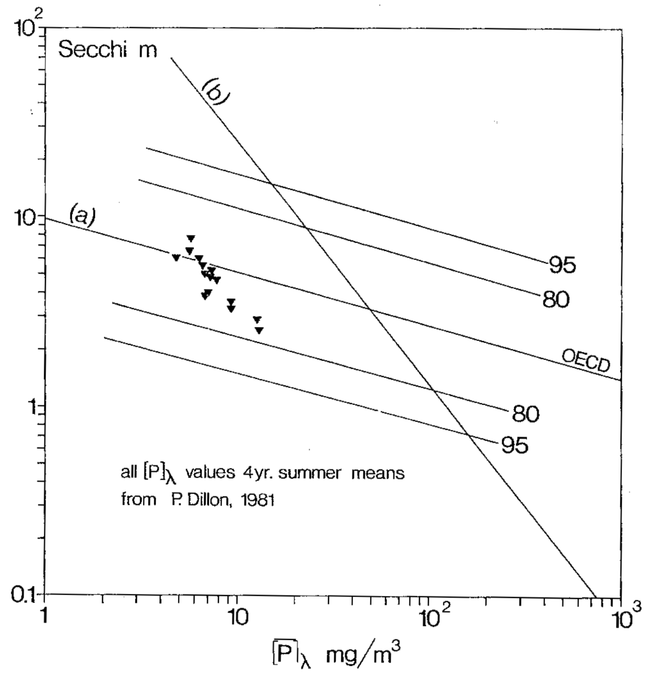

Figure III 16. Secchi transparency in relation to summer total phosphorus concentration, 4 year means

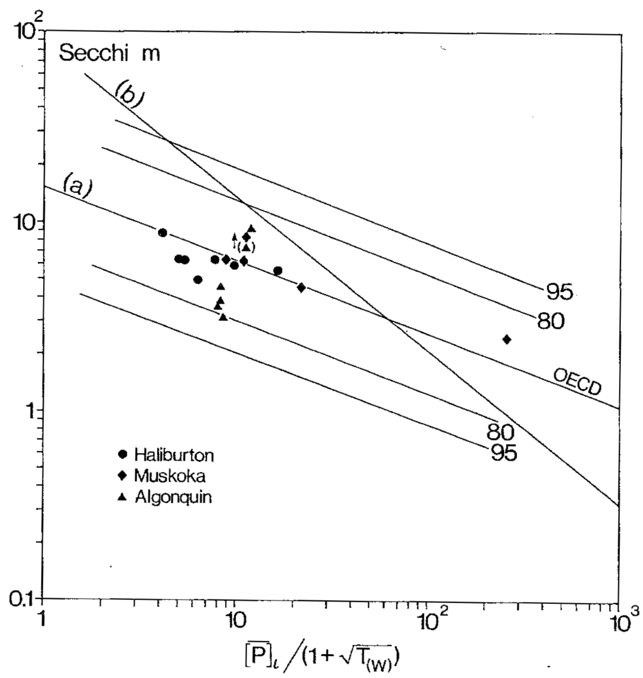

Figure III 17. Secchi transparency in relation to flushing corrected inflow total phosphorus concentration shallow mean depth (0.85 m) and large surface area/volume ratio ($1.07 \ km^2/0.91 \times 10^6 m^3$) are conditions which contribute to high variability and without strict adherence to equivalent sampling periods a lack in coordination of parameters is not surprising. In Little Otter Lake, the spring measurement does not represent the cultural input which follows throughout the growing season, and therefore it is unrelated to the summer chlorophyll build up which results in low mean Secchi transparency.

Secchi transparency plotted against loading in Fig. III 17 is in agreement with the OECD findings and only a few of the Algonquin lakes cluster toward the lower 80% limit. Although in the majority of cases here, low phosphorus in relation to loading is compensated for by high chlorophyll in relation to phosphorus (cf. previous sections), Secchi depths remain low in relation to chlorophyll. Therefore, the Secchi readings also appear low in relation to loading, and factors other than phytoplankton growth are responsible for low Secchi transparency.

3.2.6 Hypolimnetic Oxygen Depletion. The culturally enriched Muskoka bays (Skeleton and Dudley) show oxygen depletion rates far in excess of what would be expected on the basis of the average biomass (as chlorophyll) observed there (cf. Fig. III 18). The eastern and central basins of Lake Erie behave similarly. These locations appear to be analogous to the experimental enrichment of the hypolimnion of Lake 302 N where there were no changes in the algal populations of the epilimnion, but severe oxygen depletion occurred. At least partial explanation of high oxygen depletion rates, in the absence of chlorophyll densities

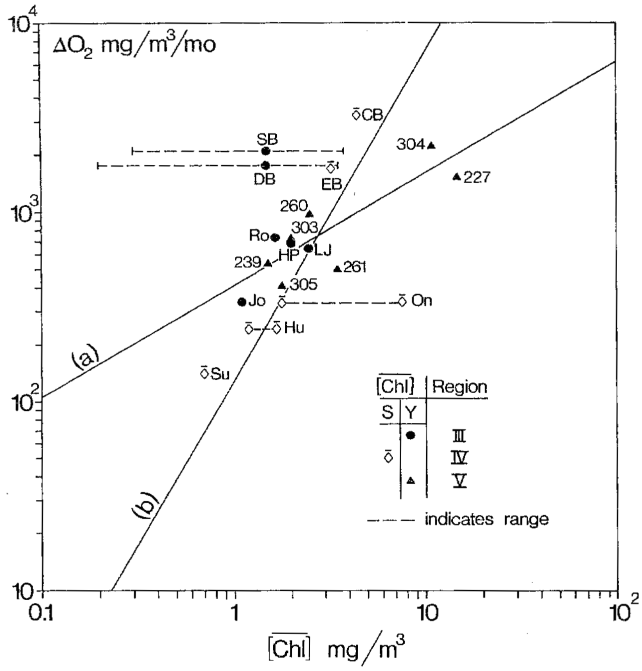

Figure III 18. Monthly hypolimnetic oxygen depletion rate in relation to mean chlorophyll a concentration which would result in this,is the diversion of nutrients or oxydizable materials into hypolimnetic waters.

3.3 Region III Conclusions

It is noticeable that the information base complied from less current and more varied sources relates to the OECD findings with as little scatter as do the results of the most recent (and better coordinated) data of a four-year study of the same location. In addition, the two data sets indicated that the most stable and therefore best predictive relationships, are those between parameters which represent similar time periods, for example mean chlorophyll levels are best predicted by the yearly mean phosphorus concentrations rather than spring measurements.

A dominant process of the region seems to be transfer of phosphorus into the sediments as evidenced by the relatively low positions of both phosphorus lake concentrations and chlorophyll levels in relation to loading estimates. Furthermòre, nitrogen limitation may have only been approached in the single case of culturally enriched Gravenhurst Bay; N : P ratios observed are high (generally 30 : 1) and could not explain the low chlorophyll levels.

Hypolimnetic enrichments (as in the particular situation of bays receiving effluents) may remain isolated in deeper water where they cause oxygen depletion in excess of that predicted by autochthonous biomass production (as measured by chlorophyll) alone. Secchi transparencies of the region are characteristically low in relation to chlorophyll and this is most probably a result of dissolved humic susbstances.

3.4 References (III)

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CHAPTER 4. LAURENTIAN GREAT LAKES REGION IV

- Lake Superior (Sp)
- Lake Michigan (Mc)
- Lake Huron (Hr)
- Lake Erie
  - Western Basin (WB)
  - Central Basin (CB)
  - Eastern Basin (EB)
- Lake Ontario (ON)

4.1 The Laurentian Great Lakes System, Description of Location

The Laurentian Great Lakes System (cf. Figure IV 1) is perhaps the largest single lake system in the world covering a total area of 784,325 km^2 of which 245,126 km^2 represent the surface area of the Great Lakes, and 538,899 km^2 represent land area plus scattered smaller lakes, some of which are still considerable in size compared to other lakes of the world.

The water mass comprised by the Great Lakes amounts to some 22,600 km^3 which represent some 10% of the world's freshwater resource, with more than half contained in Lake Superior alone. On the other hand, the land to lake surface ratio is relatively small (2.2), and the bulk renewal time amounts to some 110 years for the total water mass, however, varying considerably from lake to lake, being shortest for Lake Erie (2.6 y) and longest for Lake Superior (185 y).

Accordingly, these lakes are inland seas rather than lakes in the classical sense, yet, limnologically speaking, they do not differ in their overall characteristics from other freshwater lakes. Physical properties, i.e. current features, thermal regime and inshore-offshore relationships, become more important than in lakes of considerably smaller size.

The humid continental interior climate of the Great Lakes area is basically the same as that for the first three regions (i.e. Atlantic, Quebec and Ontario shield) described, but average annual precipitation is slightly lower at approximately 80 cm/y. The forest which

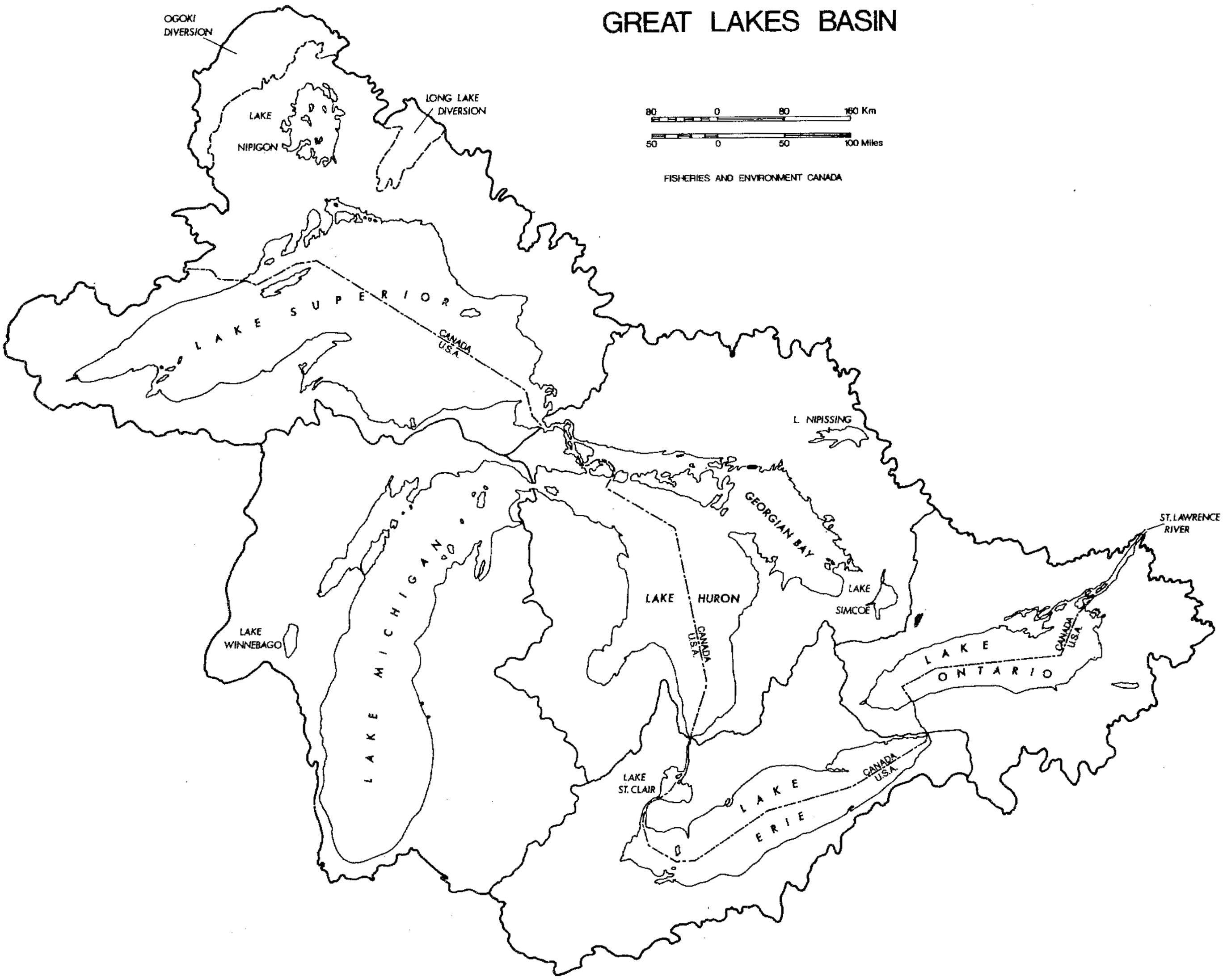

Figure IV 1. Laurentian Great Lakes Basin tolerates this climate is the Great Lakes - St. Lawrence mixture of conifers and hardwoods. The soil developed by the interaction of this particular organic litter and rainfall is a lightly leached grey-brown podzol. Some calcium is still retained by the lower layers such that the soil is not excessively acidic. The underlying bedrock is glaciated sediment of the Ordovician.

Lake Superior, the source lake of the chain, is basically different from the other Great Lakes. The climate is somewhat colder and the surrounding forest is coniferous. The unproductive soil that this litter develops is acidic and highly leached of calcium and other nutrients. The basin lies on clean shield bedrock of granite and granitic gneisses without influence of any subsequent Ordovician sediments, as is the case for the other lakes. In conjunction with the morphological character of great depth, these features have produced a beautifully clear and highly oligotrophic body of water.

Some 50 million people reside in the Great Lakes basin, and water usage is as varied as the basin itself, extending from recreation, commercial and sport fishing, water supply for drinking water, industrial use, cooling water for energy production, to large scale shipping and, of course, as a receptacle of sewage and other dumpings in large amounts.

The pollution aspect, particularly that of the lower lakes (Ontario, Erie) has been the focus of studies conducted which led to a high level international agreement between the United States and Canada in 1972, and large scale cooperation between the two countries through the International Joint Commission (IJC) and its Great Lakes Office located in Windsor, Ontario. Reports issued over the last several years through the IJC are a most important source of information on Great Lakes status.

Prior to the mid-1960s, the characteristics of this system were poorly understood, but since that time, large scale studies have been made by Canada and the United States covering many different aspects, including limnological surveys, sediment distribution, physical properties, land use and connected socio-economic studies, etc. This system is now one of the best understood of any comparable in complexity.

4.2 Land Use and Sources of Phosphorus to the Great Lakes

These are summarized in Tables IV 1 and IV 2. Globally speaking, the Great Lakes basin is covered to about 60% by forest; cropland, pastures and unused land make up some 12% each, and less than 5% is occupied by residential and industrial development. However, within the sub-basins, large differences exist. Most noticeable is the high forest coverage in the Lake Superior basin (> 90%), and the large agriculturally utilized area of the Lake Erie basin (ca. 60%). This latter is also the most densely populated area.

These differences are reflected in the considerable differences in global phosphorus export coefficients, lowest for Lake Superior (19 $kg/km^2.y$) and highest in the Lake Erie basin (ca. 200 $kg/km^2.y$). In this latter, about half of this load results from agriculturally utilized areas; however, some 30% has been attributed to direct discharges from municipal treatment plants, highest in all sub-basins both in terms of % contribution and absolute amounts.

Table IV 1. % Land Use Great Lakes Basin

Basin (km²)	Global*) Export Coeff. kg P/km².y	Urban Land Use: Developed Land (Resid/Commerc./Industr.)	Rural Land Use: Agricultural Land: Cropland	Rural Land Use: Agricultural Land: Pasture	Rural Land Use: Non-Agricultural Land: Forest/Woodland	Rural Land Use: Non-Agricultural Land: Barren/Brush Wetland
Superior (138586)	19.1	.1	.2	1.2	94.5	4.0
Michigan (117408)	39.8	3.5	12.4	11.0	49.8	23.3
Huron (128863)	23.8	1.8	9.3	13.1	65.7	10.0
Erie (78769)	198.3	9.2	39.4	19.7	17.1	14.6
Ontario (75272)	86.3	4.4	10.6	21.0	55.8	8.8
TOTAL 538889		3.2	12.2	11.7	61.1	11.8

*) Including all sources from land; excluding upstream lakes and atmospheric contributions.

From: International Joint Commission Report on Great Lakes Pollution from Land Use Activities (PLUARG). IJC Office, Windsor, Ontario, July, 1978.

Table IV 2. Phosphorus sources to Great Lakes. % contribution 1976

	Superior	Michigan	Huron	Erie	Ontario
Direct Municipal Treatment Plants	2	16	3	32	17
Tributary Municipal Treatment Plants	5	23	8	7	7
Direct Municipal	2	< 1	< 1	2	< 1
Tributary Municipal	< 1	4	2	< 1	< 1
Urban Non-point Direct	< 1	-	< 1	< 1	3
Tributary Diffused	53	30	50	48	28
Atmospheric	37	26	23	4	4
Load from Upstream Lake	-	-	14	5	41

From: IJC Report on Great Lakes Pollution From Land Use ACTIVITIES (PLUARG), International Joint Commission Office, Windsor, Ontario, July, 1978.

The global export coefficient reflects reasonably well the trophic situation of the respective lakes. However, in the case of Lake Ontario, the additional contribution from the upstream lake and the connecting channel is of significance, whereas for the upper lakes (Superior, Michigan, Huron), it is notable that the atmospheric contribution is a significant fraction of the total load, though it is as yet undetermined what biological consequences may be attributed to this source. Specific loading values to the lakes take into account all of these sources.

4.3 Trophic Response - Nutrient Relationships

In regard to the trophic response of the lakes (cf. Table IV 3) it is to be noted that the primary reactors lie inshore which in some cases are more or less separated embayments, as e.g. the highly eutrophic Saginaw Bay of Lake Huron, the open basin of which remains largely oligotrophic. Substantial differences in inshore versus offshore conditions have been found in Lake Ontario.

Lake Huron and Lake Erie cannot be treated as single basin lakes; where - with the exception of Saginaw Bay - this is of minor consequence for Lake Huron in the present context, the differentiation of Lake Erie into three distinct sub-basins cannot be neglected. Therefore, an attempt has been made to estimate loadings to each sub-basin, and connected in-lake parameters are reported separately (cf. Table IV 4).

In regard to data used for the present elaboration, it has been unavoidable to combine information from several sources, and in

IV.8

Table IV 3. Great Lakes Trophic Response

A. Mean values and range of surface phytoplankton biomass in lakes Ontario, Erie, and Huron.

	No. stations	Phytoplankton biomass $x10^3 \times mg/m^3$ Mean	Range	Remarks	Reference
Lake Ontario	27	2.7	0.7–8.5	Jan.–Dec. 1970 13 cruises	
Inshore	7	3.3	0.8–7.4	" "	Munawar and Nauwerck (1971)
Offshore	20	2.6	0.6–9.0	" "	
Lake Erie	24	3.4	1.8–5.2	Apr.–Dec. 1970 10 cruises	
Western Basin[1]	5	5.3	0.8–13.2	" "	Munawar et al. (1973)
Central Basin	12	3.2	0.6–6.0	" "	
Eastern Basin	7	2.4	1–4.2	" "	
Lake Huron					
Mid-lake station	1	0.9	0.3–1.8	Apr.–Dec. 1971 8 cruises	Munawar and Munawar (1973)
Saginaw Bay station	1	8.3	1.6–17.3	" "	

B. Mean range of chlorophyll *a* (mg C/m^3) in the Great Lakes (lakes Michigan and Superior values are not corrected for pheopigments).

	No. stations	Chlorophyll *a* Mean	Range	Remarks	Reference
			Lake Ontario		
Inshore	10	5.2	2.7–12.0	Jan.–Dec. 1970 13 cruises	Glooschenko et al. (1972, 73, 74b)
Offshore	22	3.8	1.8–7.9	" "	
			Lake Erie		
Western basin	5	8.9	3.3–19.3	Apr.–Dec. 1970 10 cruises	" "
Central basin	12	4.4	2.5–9.2	" "	
Eastern basin	8	3.2	1.4–5.4	" "	
			Lake Huron		
Open lake	44	1.7	1.4–2.2	Apr.–Dec. 1971 8 cruises	" "
Saginaw Bay	2	16.9	9.5–27.4	" "	
			Lake Michigan		
	3	2.3	0.3–5.3	May–Nov. 1967 8 cruises	Robertson et al. (1971)
Inshore	2	-	1.1–10.3	May 1970–Feb. 1971 16 cruises	Fee (1971)
Offshore	3	-	0.6–3.7		
			Lake Superior		
	130	-	0.4–9.7	May–Aug. 1973	H. F. Nicholson (unpublished data)

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Table IV 3. (continued)

C. Photosynthesis rates in the Great Lakes.

	Date	mg C/m³/h	mg C/m²/h		mg C/m²/day	g C/m²/yr	Reference
			Lake Ontario				
Inshore stations (10)	Jan.-Dec. 1970	2.9-25.0[a]		27-110[a]	120-1080[a]	190(170)[b]	Glooschenko et al. (1974a)
Offshore stations (22)	Jan.-Dec. 1970	1.7-12.4[a]		17-100[a]			" "
Inshore station (1)	Apr.-Apr. 1972-73	-	1.8-26.7[c]	11-184[d]	119-2003[d]	270[b]	Stadelmann et al. (1974)
Offshore station (1)	Apr.-Apr. 1972-73	-	1.4-33.3[c]	23-178[d]	58-1443[d]	170[b]	" "
			Lake Erie				
Eastern stations (8)	Apr.-Dec. 1970	3.2-13.9[a]		15-120[a]	140-1440[a]	(160)[b]	Glooschenko et al. (1974a)
Central stations (14)	Apr.-Dec. 1970	5.5-21.4[a]		17-141[a]	120-1690[a]	(210)[b]	" "
Western stations (3)	Apr.-Dec. 1970	4.8-146.9[a]		5-397[a]	30-4760[a]	(310)[b]	" "
			Lake Huron				
Stations (13-14)	May-Aug. 1968	4.9-12.0[a]					Parkos et al. (1969)
Stations (40)	Apr.-Dec. 1971	2.2-9.9[a]		21-63[d]	147-698[d]	100(80-90)[b]	Glooschenko et al. (1973)
Saginaw Bay (2)	Apr.-Dec. 1972	4.1-127.2[a]		-	-	-	" "
			Lake Michigan				
Stations (17-24)	July-Oct. 1967	8.1-17.5[a]		-	-	-	Parkos et al. (1969)
Northern stations (11-5)	July-Aug. 1969	2.5-4.1[a]		-	-	-	Schelske and Callender (1970)
Offshore stations (3)	June-Feb. 1970-71	-	0.9-4.2[c]	-	67-1030	121-139	Fee (1971)
Inshore stations (2)	June-Feb. 1970-71	-	1.4-30.1[c]	-	67-1567	187-247	"
			Lake Superior				
Larsmont station (1)	July-Aug. 1961		0.25-0.73[e]	-	183[e]	-	Olson and Odlang (1966)
Stations (23-26)	July-Oct. 1967	2.6-8.8[a]		-	185[f]	-	Parkos et al. (1969)
Stations (20-22)	May-Aug. 1968	4.7-6.4[a]		-		-	" "
Stations (58)	July-Sept. 1973	2.2-2.7[a]		32-35[d]	330-350[d]		C.C.I.W. (unpublished data)

[a]Mean of each cruise.
[b]During the investigation period.
[c]P_{opt}.
[d]In situ values.
[e]Maximum observed values.
[f]Mean of the two seasons.

- From Vollenweider et al., 1974

Table IV 4. Lake Erie Phosphorus Loading.

(Data combined from 1970 to 1975 studies)

	P-Load t/y			Q km^3/y ($\tau_w y$)	$\overline{[P]}_J$ mg/m^3	$[P]_\lambda$ mg/m^3			concentr. predictor $[P]_\lambda^{pred}$ 3)	predicted concentr. $[P]_\lambda$ 4)	R_c meas. %	Average chl. a mg/m^3	
	carry over	external	total ($g/m^2.y$)			spring	max yr	year average				found	predicted 5)
Total Lake from L. Huron	2335 1)	18170 1)	20505 1)	183	112	-	-	-	43	34		-	7.5 - 8.5
to Niagara	3910 2)		(.81)	(2.56)							81		
Western Basin from L. Huron	2335 1)	11060 1)	13395 (3.66)	176 (.16)	76	62	62	46	54	41	54	8.9 - 11.4	9 - 11
Central Basin from W. Basin	6220 2)	5080 1)	11300 (.74)	179 (1.53)	63	18	39	22	28	24	73	4.5 - 5.5	5.2 - 5.9
Eastern Basin from C. Basin	3020 2)	2030 1)	5050	183	28	21	24	16	14	13.5	23	3.3 - 4.3	3.0 - 3.4
to Niagara	3910 2)		(.81)	(.91)									

1) U.S. Corps of Engineers Estimates (1974-75)
2) From: N. Burns 1970/71 Study
3) Predictor = $[P]_J/(1 + \sqrt{\tau_w})$
4) From: Predicted Concentration = OECD Relationship $[P]_\lambda = 1.55 \left\{[P]_\lambda^{pred.}\right\}^{.82}$
5) From: OECD Relationship Chl = $(.37 \div .43) \left\{[P]_\lambda^{pred.}\right\}^{.79}$

part combine values determined for different years. Given the enormous size of these lakes, it is somewhat arbitrary to reduce information to simple averages. Therefore, averages calculated by different people may differ between them; however, accounting for the difficulties connected with data reduction, it is surprising how close such values are in the end.

A weakness of the - for the rest excellent - data base is the insufficiency of nitrogen data, in particular, total nitrogen. Therefore, the respective relationships cannot be explored further in this report.

4.3.1 Chlorophyll-Phosphorus Relationship. Pertinent data are plotted in Figure IV 2 which show an excellent relationship with the overall OECD findings. Lake Superior is slightly low, perhaps due to the relatively low average temperature of this lake.

Slightly at variance from the lake-wide relationship is that resulting from the several subsections of Saginaw Bay. The reason for this is not clear, but it is nevertheless noteworthy that there is consistency in other data points.

4.3.2 Phytoplankton-Phosphorus Relationship. Available data are plotted in Figure IV 3. Although considerable difference in phytoplankton composition between the various lakes exists(cf. e.g. Vollenweider *et al*. 1974), the data points lie practically on one line from which it would result that to 10 mg TP about 1500 mg phytoplankton freshweight (or 1.5 cm^3/m^3 phytoplankton volume) correspond. If a 10% volume carbon content

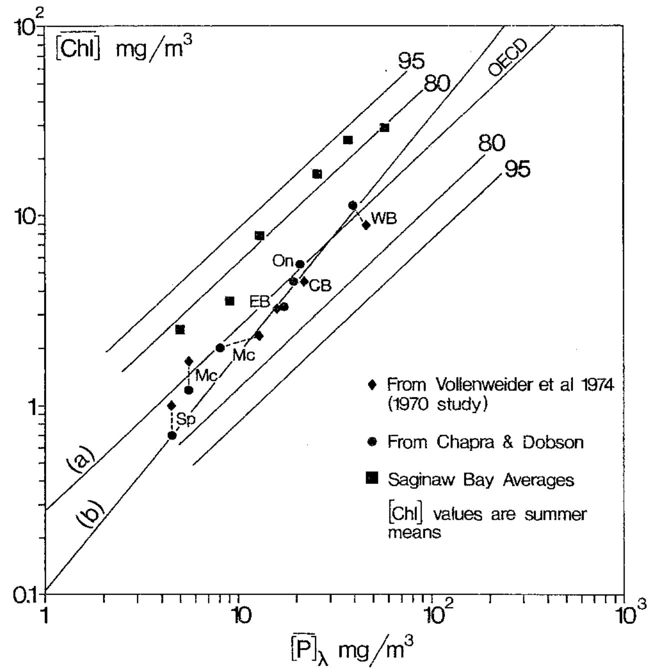

Figure IV 2. Mean summer chlorophyll a concentration in relation to annual mean total phosphorus concentration

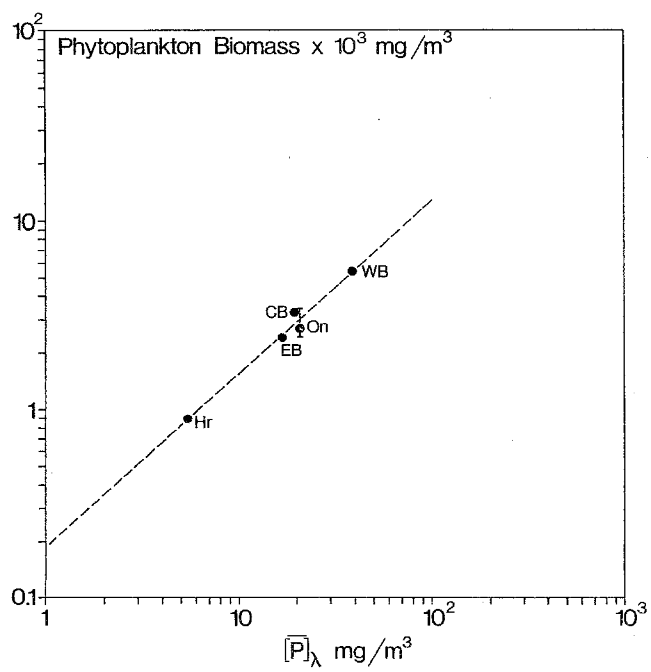

Figure IV 3. Annual mean surface phytoplankton biomass in relation to annual mean total phosphorus concentration and a C/P ratio of 40 apply, this would mean that about 3 to 4 mg P of a total of 10 mg total P are tied up on average in phytoplankton; conversely, it would also mean that 1 mg/m^3 of chlorophyll corresponds to approximately 500 to 600 mg/m^3 (or 0.6 to 0.7 cm^3) of phytoplankton, as found for the Quebec Region. This appears to be roughly double the values resulting from the Alpine Project.

4.3.3 Phosphorus-Loading Relationship. This relationship is plotted in Figure IV 4 and results practically identically with the overall OECD relationship. The slight positive deviation of measured values for the Eastern and Central basins of Lake Erie is probably insignificant, given the difficulty of estimating the transfer between the basins correctly, whereas the OECD relationship would slightly underestimate the actual concentration in Lake Ontario. Whether this is due to a non-equilibrium situation (which would imply from the data that sediment return is important though unlikely) or due to an underestimate of the actual loading, cannot be decided yet. Considering the same displacement observed for Lake Ontario regarding the chlorophyll-loading relationship (cf. Figure IV 5), this latter is a possibility.

4.3.4 Chlorophyll-Loading Relationship. This relationship is plotted in Figure IV 5. The scattering along the OECD relationship appears to be somewhat larger than for the formerly discussed relationships, yet still within conforming limits of confidence. The relatively low position of Lake Superior is in agreement with that already noted previously, whereas for Lake Michigan, uncertainties exist regarding

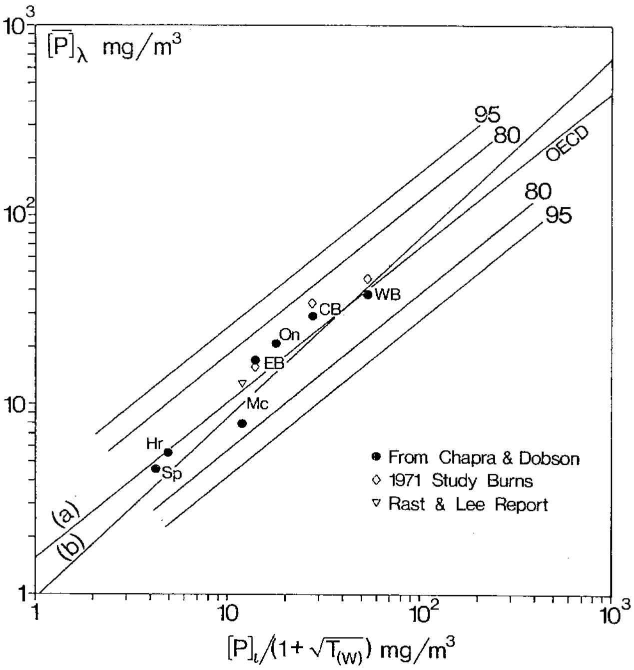

Figure IV 4. Annual mean total phosphorus concentration in relation to the flushing corrected annual mean inflow total phosphorus concentration

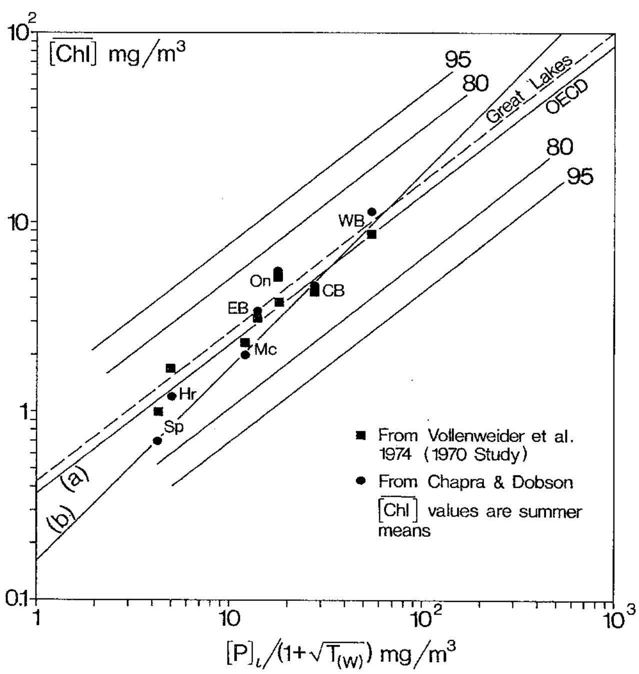

Figure IV 5. Annual mean chlorophyll a concentration in relation to the flushing corrected annual mean inflow total phosphorus concentration the basic data which differ between authors. The somewhat aberrant position of Lake Ontario has already been noted above.

4.3.5 Primary Production-Loading Relationship. Given the enormous difficulties of reliably measuring the primary production of the lakes in question, relatively good estimates exist. These have been plotted against phosphorus loading characteristics in Figure IV 6. In contrast to the other relationships discussed, yearly primary production is not linearly related to loading, and can best be approximated by a hyperbola. Several options are open, yet in the most simple case, yearly primary production in the Great Lakes can be estimated from

$$PP\ (gC/m^2.y) = 500\ \frac{x}{27 + x}\ ,$$

where $x = [P]_l/(1 + \sqrt{\tau_w})$. This is but marginally different from the relationship found in Alpine Lakes. Accordingly, the saturation level for the Great Lakes would be in the order of 500 $gC/m^2.y$; Lake Superior in regard to this relationship, is somewhat low, which corresponds, however, to the other findings.

4.3.6 Hypolimnetic Oxygen Depletion Rates. Estimation of this parameter in the Great Lakes (cf. Figure IV 7) is difficult due to depletion rates being low in the deeper lakes, and in Lake Erie, due to considerable variation from year to year in the hypolimnetic thickness and other parameters. In this latter, the observed depletion rates are rather high, leading in certain years to complete exhaustion of the hypolimnetic oxygen reserve in the Central basin in a layer 3 to 4 m above the sediments.

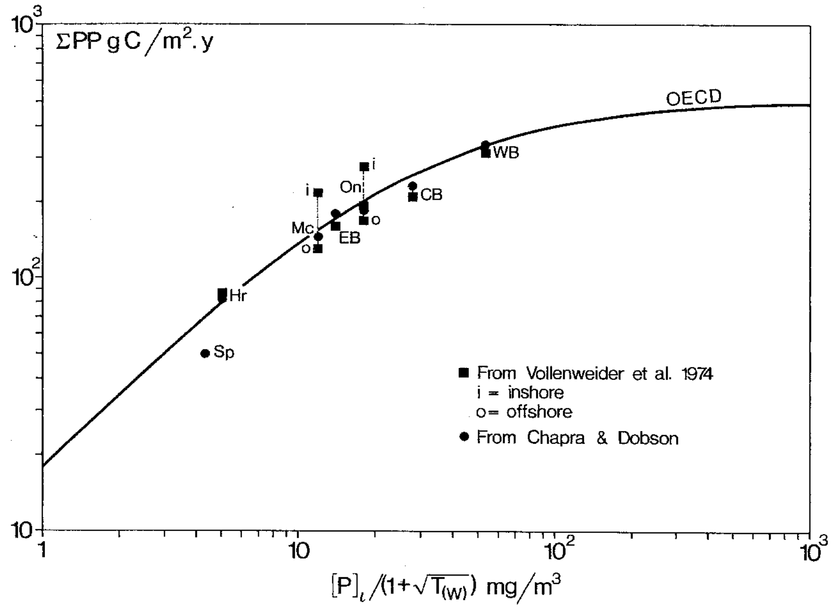

Figure IV 6 Annual areal primary production in relation to flushing corrected annual mean inflow total phosphorus concentration

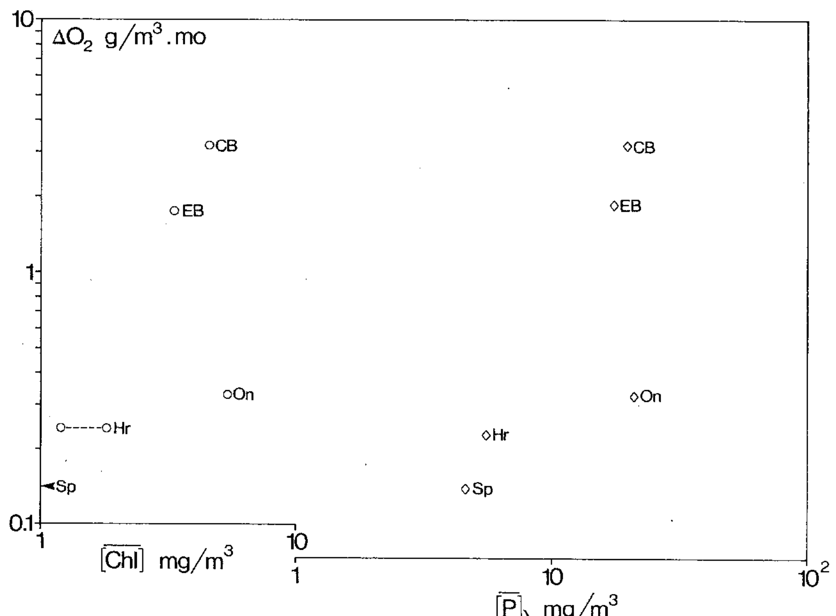

Figure IV 7.

Hypolimnetic oxygen depletion rates in relation to annual mean chlorophyll a concentration and annual mean total phosphorus concentration

Any attempt to calculate areal hypolimnetic depletion rates for comparative purposes has been unsuccessful insofar as the resulting figures are but marginally related to epilimnetic production characteristics. The difficulty arises in part from the uncertainty of accurately defining the hypolimnion, and in part may also be due to physical phenomena. In Lake Erie, e.g., exchange phenomena and vertical transport of oxygen are rather important, reducing the apparent depletion rate which, however, is also counterbalanced by hypolimnetic temperature substantially above 4°C, whereas in the deep lakes exhibiting temperatures below 4°C in accordance with the pressure-density relationship, and in which it is not uncommon to observe slight "oversaturation" even in deep layers, part of the observed apparent depletion rate may be purely physical and not related to oxidative processes.

4.4 Region IV Conclusions

In considering the particular problems connected with Great Lakes research, both logistically and in regard to the spatial and temporal variability, it is surprising that these lakes, better than many other lakes reviewed here, fit the average standard behaviour of lakes resulting from the OECD programme. Except for the oxygen depletion conditions, all parameters examined deviate only marginally from expectation, and hence, application of OECD experience to these lakes will undoubtedly provide a good basis for prediction. This may fail, however, in regard to behaviour of local regions, embayments and waters partly separated from the main body, as well as in inshore-offshore differentiation which in certain lakes is important, and which in this summary report could not be adequately treated.

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CHAPTER 5. EXPERIMENTAL LAKES AREA V

a) Natural Condition

114

120

223

224

228

239

240

303

305

383

b. Artificial Enrichment

226 NE

226 SW

227

261

302 N

302 Total

302 S

304

5.1 Experimental Lakes Area, Description of Location

By the mid-1960s, nutrient pollution in the lower Great Lakes was obvious and a need for systematic investigation of eutrophication became imperative (Johnson and Vallentyne 1971). The Experimental Lakes Area (ELA) of northwestern Ontario (Figure V 1) was established to attempt stepwise unravelling of the complex problem. The preliminary data collected for these lakes forms the basis of the present regional analysis.

Dynamic sculpturing of the Shield by glacial and faulting processes left a haphazard network of drainage systems. The recession of Lake Agassiz some 9,500 B.P. marked the formation of the present lakes. Waters are dilute; Brunskill and Schindler (1971) found the average conductivity of 463 lakes to be 29 μmho/cm, 25C. Secchi depths (usually 1 to 2 metres for the same 463 lakes) are low due to humic stains. Among the lakes chosen for the ELA, transparency is somewhat higher and Secchi depths are usually 4 to 6 metres. Overall, basins of the region are small, deep, of limited transparency and oligotrophic in nature.

Only boreal subclimax forest persists under the taiga conditions of maximal seasonal temperature change with a lengthy (five to six months) snow and ice cover. The frost free period is on the order of 100 days long and total annual precipitation is approximately 70 cm.

Figure V 1. Location of the Experimental Lakes Area
(- from Brunskill and Schindler, 1971).

Excepting the research operations which take place, this rather inacessible landscape is free from human influence and habitation. In addition to the difficulties in transport and travel, there is "the menace of mosquitoes against which the use of smoke, oils or wire screens is imperative" (Mead and Brown 1962). The low density of wildlife surviving these conditions consists of moose, wolves, black bears and loons. The predominant trees tolerating the rocky substrate are black spruce and jack pine. The sparse growth of macrophytes includes *Lobelia dortmana*, *Eriocanton septangulare*, *Nuphar*, *Potamogeton*, *Isoetes* and others. Typical phytoplankton include Chrysophycean genera (*Dinobryon*, *Mallomonas*, *Ochromonas*, *Chromulina*) with diatoms *Asterionella*, *Tabellaria*) and non-bloom-forming Cyanophyceae (*Chroococcus* and *Rhabdoderma*).

In consideration of all, lakes of ELA were chosen as sites for manipulation of factors suspected to stimulate eutrophication for the reasons of abundance, small size and low-level natural loading (Johnson and Vallentyne 1971).

5.1.1 Nutrient Sources and Loadings.

a) Natural Condition. Isolation of the area from cultural effects leaves atmospheric precipitation of major importance as a source and mechanism of nutrient loading. The underlying crystalline bedrock is nearly insoluble with the effect that the nutrient-poor precipitation changes little as it finds its way into the lakes. As such, physical, chemical and biological parameters of the ELA region are found to be directly related to the simple parameter 'catchment area' (Schindler 1971).

Previous documentation of lakes of the ELA describes their response as nutrient sinks; radiotracer experiments indicate that the net flux of phosphorus is into the sediments, hence, sediment feedback is unimportant in terms of phosphorus loading in natural lakes there.

b) Artificial Fertilization

Artificial enrichment experimentation was begun in several lakes in 1969 and histories of enrichment schemes for this and subsequent years are set out in Table V 1. In general, nutrient additions were made to the epilimnions of the lakes by simple surface applications of nutrient solutions but 302 N was enriched hypolimnetically. These enrichment experiments have clearly demonstrated the controlling effect of phosphorus on biomass in this region.

In the discussion that follows, this and other nutrient-trophic state relationships are explored through both variation of the natural condition and that induced through these artificial enrichments. The two sets of data (for natural and fertilized lakes) are in part treated separately and are distinguished by different signs in the figures.

5.2 Trophic Response - Nutrient Relationships

5.2.1 Chlorophyll-Phosphorus Relationship. Data: The chlorophyll data presented in this section represent two time periods, each several years long, and include different seasons and different strata of the lakes. The data set supplied by D. Schindler (pers. commun.) is the earlier of the two and covers the period 1969 to 1974. The chlorophyll means of this period are calculated on the basis of the full year and

Table V 1. Condition of ELA Lakes

A. Natural Lakes. (no enrichments) 114, 120, 223, 224, 226 N (prior to 1973), 239, 240, 261 (prior to 1973), 302 N (prior to 1971 303, 305, 383, 228.

B. Artificial Enrichments. Fertilization histories: All additions were made as 18-21 equal increments during the ice-free season (May - October). Unfertilized lakes of the area receive 0.05 to 0.10 $g/m^2/yr$ of P and 1.2 - 1.5 $g/m^2/yr$ of N. (From Schindler 1975).

Lake	Area	$\bar{z}$	Years	Enrichment	Annual Fertilizer Addition, g/m^2			Chemical Used			Response
	ha	m	Fertilized	of:	P	N	C	P	N	C	
227	5.00	4.4	1969 - 74*	Epilimnion	0.48	6.29	-	H_3PO_4	$NaNO_3$	sucrose	Large increase in standing crop. Domination by greens and blue-greens.
304	3.62	3.2	1971 - 72		0.40	5.2	5.5	H_3PO_4	NH_4Cl	sucrose	Same as 227.
			1973 - 74		none	5.2	5.5	-	NH_4Cl	sucrose	Complete recovery.
226 N	4.86	5.0	1973 - 74		0.59	3.16	6.05	H_3PO_4	$NaNO_3$	sucrose	Large increase in standing crop. Domination by blue-greens.
226 S	4.73	5.7	1973 - 74		none	3.16	6.05	-	$NaNO_3$	sucrose	No response.
261	5.6	2.9	1973 -		0.26	-	-	H_3PO_4	-	-	2 - 3 times increase of normal standing crop.
302 N	12.8	5.7	1972 - 74	Hypolimnion	0.54	2.79	3.73	H_3PO_4	NH_4Cl	sucrose	Slight increase in standing crop during fall overturn.

From Schindler, 1975

* 1975 and later: P enrichment remained constant, N enrichment reduced to 1/3 previous level.

the entire water column. The chlorophyll data reported by Fee (1979), on the other hand, span the period 1973 to 1976, account for the ice-free season (May to October) and include only the euphotic zone (similar to the OECD data). Therefore, this second set of data may exclude chlorophyll maxima sometimes found below the euphotic zone. Data of these two sets are plotted separately in Figures V 2 and V 3 for lakes in the natural condition. The enriched condition is plotted in Figure V 4, which only displays the earlier set and V 5 where yearly means for some recent years were available. (Chlorophyll values of the two time periods are compared in later sections in relation to loading and primary production).

a) Natural Condition. Some difference exists between Schindler's earlier and Fee's (Fee 1979) later chlorophyll values (cf. Figures V 2 and V 3, respectively). Most of the values reported do not deviate significantly from the OECD relationship but 114 and 304 values seem to be high during the years represented in Figure V 2, whereas they remain within the 95% confidence interval in later years represented in Figure V 3. Partial explanation may be existence of chlorophyll maxima below the euphotic zone which would only have been sampled in the earlier data set as in Figure V 2. With regard to the overall pattern, distribution of the points along the OECD lines in Figure V 2 would indicate a tendency of the ELA lakes to produce higher chlorophyll values above, and lower chlorophyll values below the [P] = 10 mg/m³ mark, yet the best-fit regression line does not deviate significantly from the OECD

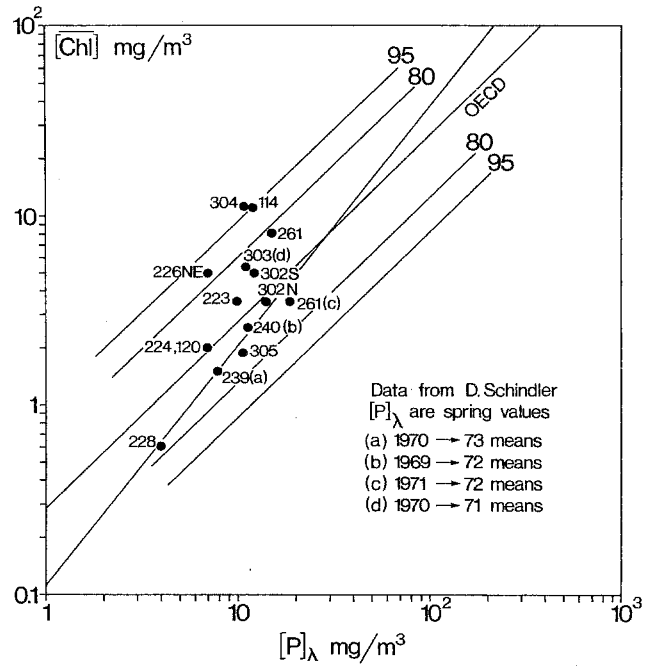

Figure V 2. Natural condition: annual mean chlorophyll a concentration in relation to spring total phosphorus concentration (Schindler)

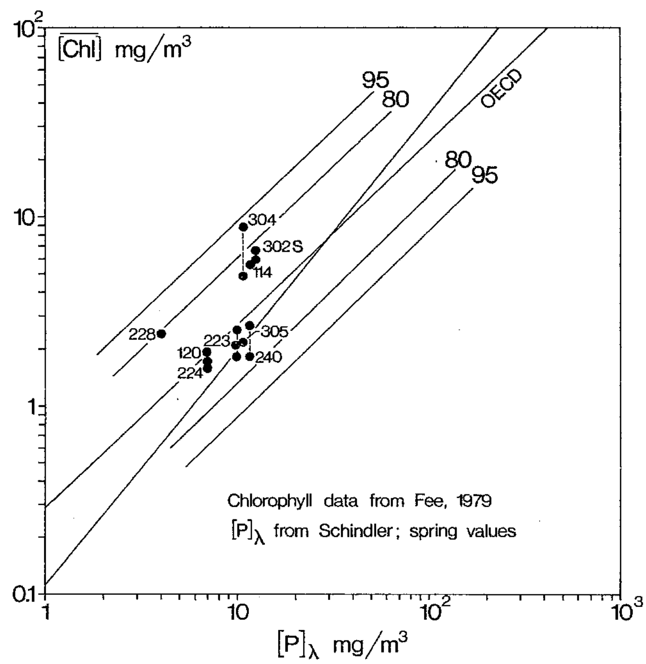

Figure V 3. Natural condition: summer mean chlorophyll a concentration in relation to spring total phosphorus concentration (Fee) regression line. As discussed in the concluding chapter, this tendency becomes clear when the entire range of OECD lakes is considered in combination with the Canadian data.

b) Artificial Fertilization. Chlorophyll values of the fertilized lakes, if plotted against the respective spring phosphorus values, cluster above the OECD relationships (cf. Figure V 4 and V 5). However, it has to be noted that fertilization of these lakes has been done over prolonged periods of time after the spring period; therefore, for these lakes, the OECD plot based on spring phosphorus values cannot be strictly used as reference. This is similar to the situation of natural lakes which obtain their major nutrient loads throughout the growing season, and therefore, in principle, these would also be expected to lie above the OECD line.

Other factors may also contribute to the high position of these chlorophyll values. The type and proportions of nutrients added may have some bearing on this. Nutrients added as artificial enrichment are in a form readily assimilated by phytoplankton and chlorophyll values may reflect this higher than usual percentage of available nutrients. Also, the effect of N:P ratio on chlorophyll production, presently a matter of debate, may also be influential here. (N:P ratios are discussed in the following section). In any case, the N:P ratio may affect species composition which may in turn alter grazing pressure and/or efficiency of quantitative determinations. Grazing pressure in the ELA has been found to be extremely low and R. Peters measured this at 2 to 3%

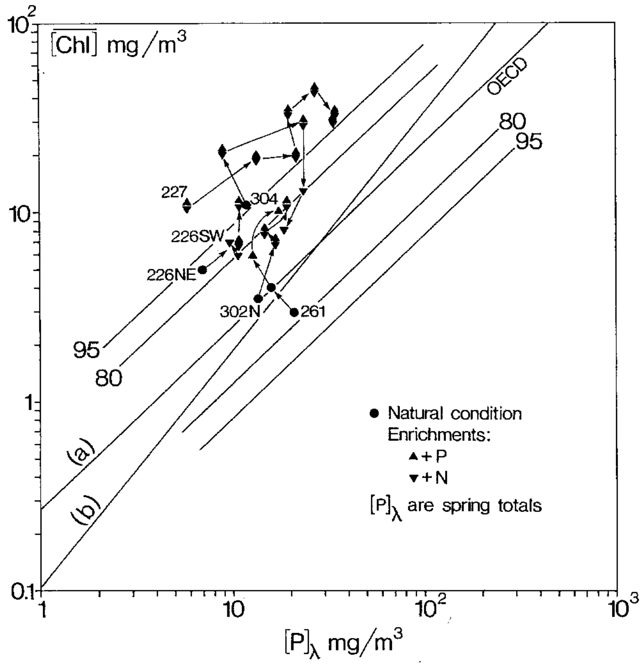

Figure V 4. Enrichment condition: annual mean chlorophyll a concentration in relation to spring total phosphorus concentration

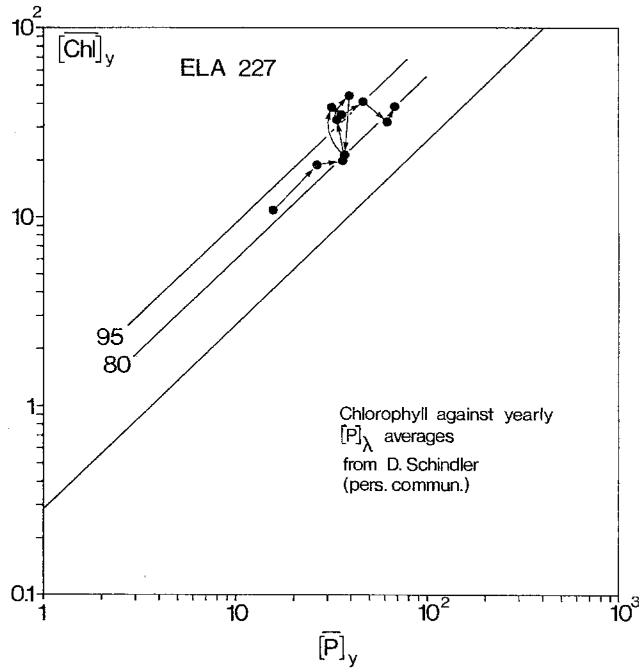

Figure V 5. Enrichment condition: L227, annual mean chlorophyll a concentration in relation to annual mean phosphorus concentration per day in contrast to 25 to 50% measured elsewhere (D. Schindler, pers. comm.). D. Schindler also comments that chlorophyll values may appear high due to high extraction efficiency in the methodology of Stanton et al. and the paucity of large phytoplankton cells in ELA lakes.

Despite high chlorophyll values, fertilized ELA lakes do not respond differently from the natural lakes of the region in terms of loading parameters as discussed later.

5.2.2 Chlorophyll-Nitrogen Relationship

a) Natural Condition. If it is assumed that nitrogen limitation does not occur when N/P ratios are greater than 10, then ELA lakes are not subject to this restriction. The average N/P ratio is 34.2 ranging from 18 to 75 (cf. Table 2) and all lakes lie below the 10:1 limitation line in Figure V 6.

b) Artificial Fertilization. The same limitation in plotting chlorophyll values against spring concentration, as discussed above for phosphorus, applies to nitrogen. However, it is to be noted that chlorophyll values of fertilized lakes relative to spring overturn nitrogen came close, or within the transition zone of nitrogen limitation (cf. Figure V 7). This would indicate that - if the lakes had not been simultaneously fertilized with nitrogen - the natural nitrogen reserve would probably have imposed a temporary ceiling on the response to fertilization with phosphorus alone. Nitrogen limitation is likely to be of only limited duration though, since "nitrogen concentration increases rapidly when phosphorus loading is increased; nitrogen return seems to be enhanced as the lakes eutrophy" (D. Schindler, pers. commun.).

Table V 2. Spring N/P Ratios of ELA Lakes

A. Natural Conditions		Yr.	N	P	N/P
114			448	12	37
120			312	7	45
223			283	10	28
224			196	7	28
226 NE		71	259	7	37
239		70-73	248	8.5	29
240		69-72	267	11	24
261		71-72	340	18.5	18
302 N		71	290	14	21
302 S		72-74	321	12.3	26
303		70-71	312	11	28
304		70-74	1100	17.4	63
305		69,71,74	213	10.7	20
228		74	300	4	75
					$\bar{x}$ = 34.2
B. Artificial Enrichment*			**N**	**P**	**N/P**
227	N+P	69	249	6	42
	N+P	71	856	22	39
	N+P	72	1474	20	74
	N+P	73	989	28	35
	N+P	74	1150	32	36
226 SW	+N	73	353	11	32
	+N	74	448	10	45
261	+P	73	338	13	26
	+P	74	372	18	21
226 NE	N+P	73	338	11	31
	N+P	74	420	11	38
302 N	N+P	72	349	17	21
	N+P	73	413	15	28
	N+P	74	522	19	27
					$\bar{x}$ = 35.4

* See Table V 1 for quantitative description of artificial loadings.

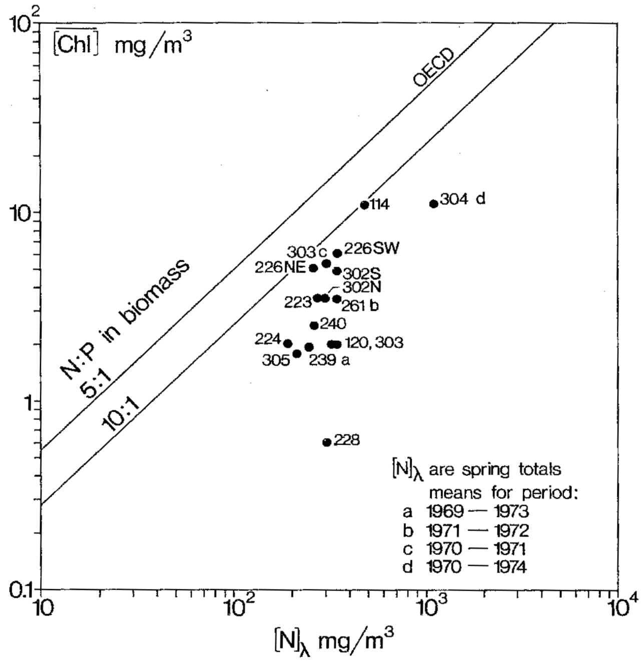

Figure V 6. Natural condition: annual mean chlorophyll a concentration in relation to spring total nitrogen concentration

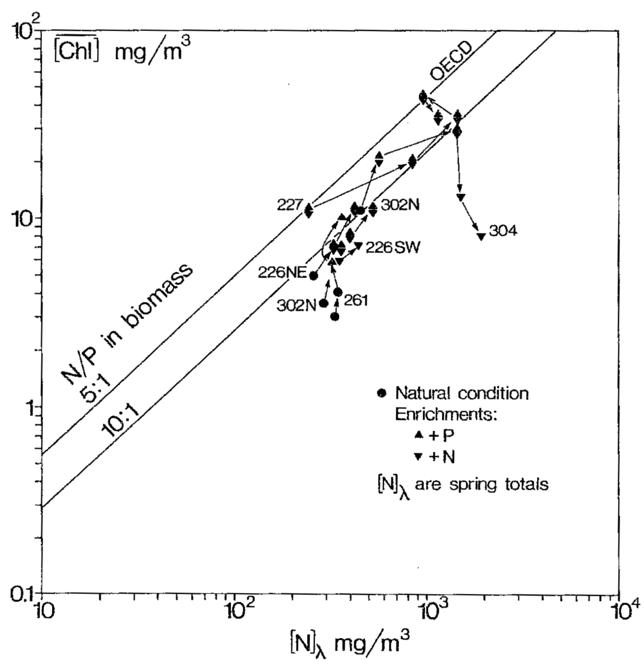

Figure V 7. Enrichment condition: annual mean chlorophyll a concentration in relation to spring total nitrogen concentration

5.2.3 Nutrient Loadings and Concentrations. (cf. Appendices V 1 and V 2).

a) Phosphorus: Natural Condition. The relationship between phosphorus lake concentrations and flushing corrected inflow concentrations is quite diffuse and values are found both above and below the OECD 95% confidence limits (cf. Figure V 8). The values plotted are spring concentrations and these show little dependence on the loading which takes place throughout the remainder of the season. When spring values of the year following loadings are used (cf. Figure V 9), the relationship approaches more nearly that of the OECD lakes and all values fall within the OECD 80% confidence limits. The cause and effect relationship of these parameters is clarified by arrangement of events in time.

b) Phosphorus: Artificial Fertilization. The independence of spring concentrations from loading (i.e. flushing corrected inflow* concentrations) is exaggerated by artificial enhancement of loadings (cf. Figure V 8) and spring concentrations fall far below what might be expected on the basis of this reference. As in the case of the natural condition, there is a shift in the position of data points closer to the OECD lines when spring [P] values lag one year after loadings. In extension of this, it is notable that as the years of enrichment of Lake 227 progress, spring concentrations begin to "catch up" to the loadings received (cf. Figure V 9). Allowance of a three to four year lapse between loadings

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* What has been termed here inflow concentrations $[M]_\iota$ are values calculated similarly to those for natural conditions, but total loading L_M includes nutrients added through artificial fertilization; $[M]_\iota = \frac{L_M}{\bar{z}} T(w)$.

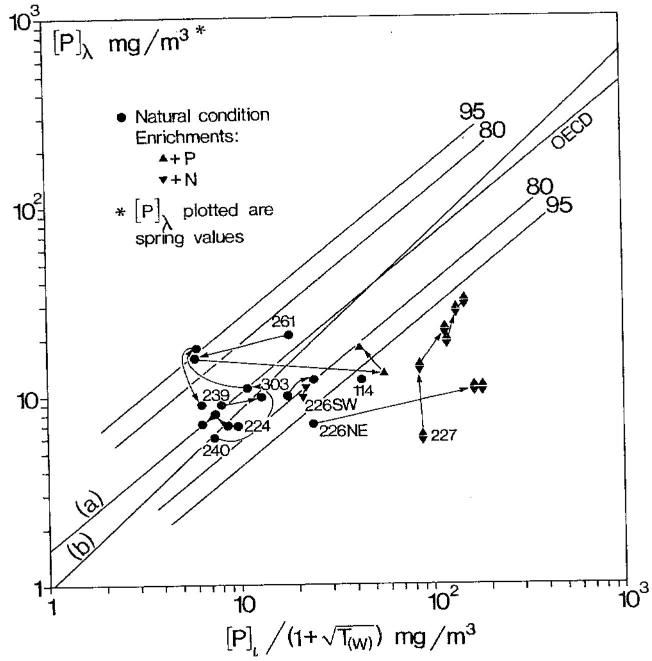

Figure V 8. Total phosphorus concentration in relation to flushing corrected annual mean total phosphorus inflow concentration

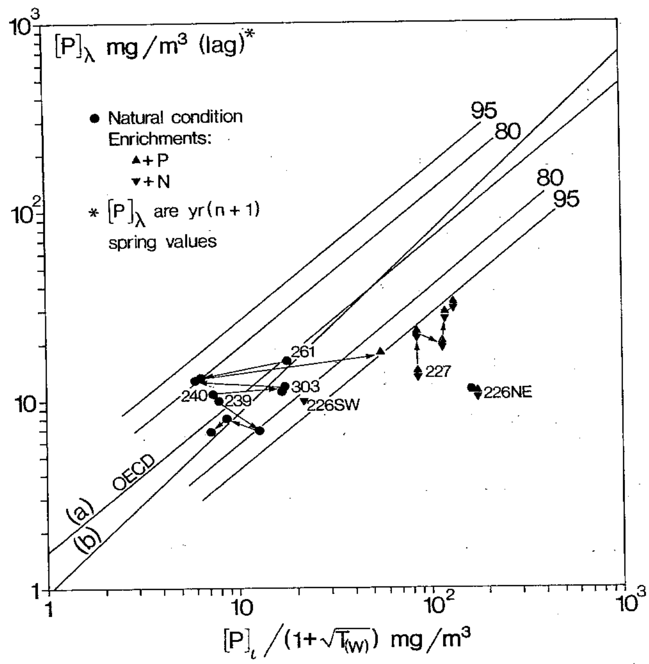

Figure V 9. Total phosphorus concentration (one year after inflow measurements) in relation to flushing corrected annual mean total phosphorus inflow concentration and resultant concentrations for 227 places it within the 95% limits of the OECD line. Thus, phosphorus concentrations of artificially fertilized ELA lakes, in contrast to the natural lakes, do not attain long-term dynamic equilibrium with loadings until several years later. The consequence of this fact is discussed in the conclusion.

c) Nitrogen. Nitrogen concentrations of lakes in the natural condition fall on or within the 95% confidence limits of the OECD line (cf. Figure V 10). Only a slight shift in position relative to the OECD line is made if spring nitrogen concentrations of the year following loadings are used (cf. Figure V 11). In the artificially enriched condition, nitrogen concentrations attain an equilibrium with loadings more rapidly than in the case of phosphorus. This can be seen from the positioning of points relative to the OECD line for the artificially enriched lakes and may explain why these lakes are comparable with the natural condition (cf. Figure V 10 and V 11). An exception to this seems to be lake 226 (NE and SW) which received only half the nitrogen of other artificially enriched lakes. Here, [N] remains low relative to loading even with a lag period of one year (cf. V 11).

5.2.4 Chlorophyll-Loading Relationship. In lakes of the natural condition, chlorophyll means plotted against the respective flushing corrected phosphorus loadings remain within or near the 80% confidence boundaries, and in general terms follow the OECD relationship (cf. Figure V 12). The tendency to lie above the line is a reflection of the high chlorophyll values in relation to [P] as previously discussed.

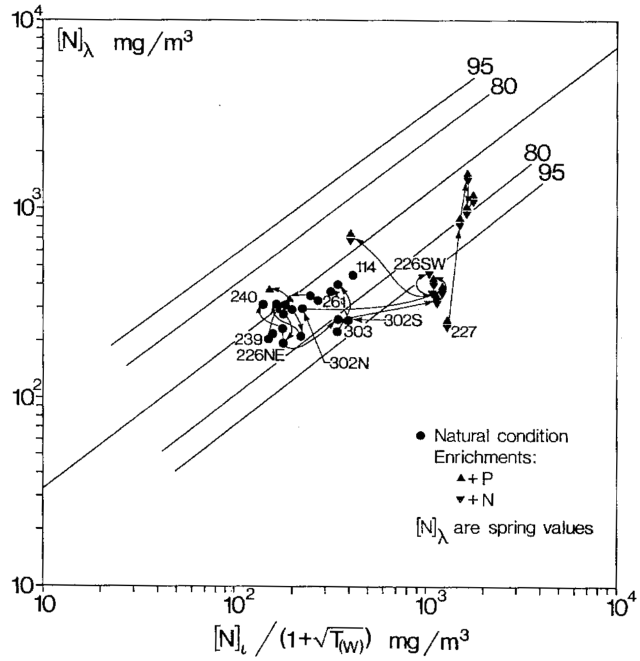

Figure V 10. Total nitrogen concentration in relation to flushing corrected inflow total nitrogen concentrations

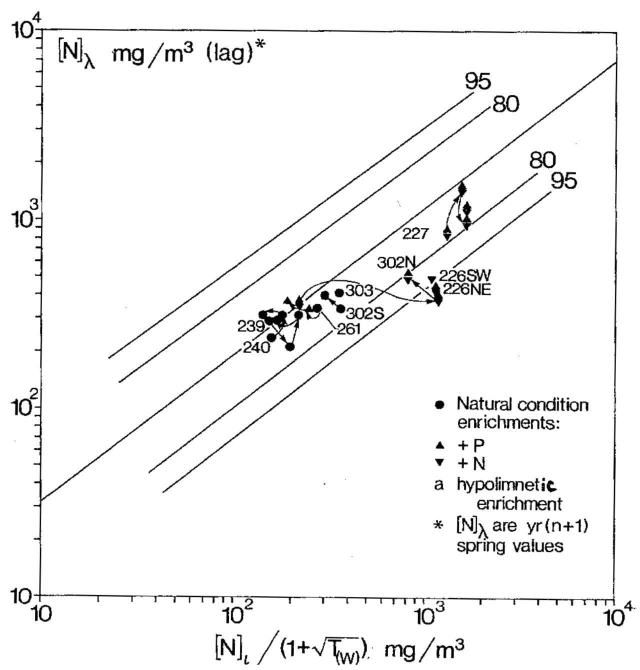

Figure V 11. Total nitrogen concentration (one year after inflow measurements) in relation to flushing corrected inflow total nitrogen concentrations

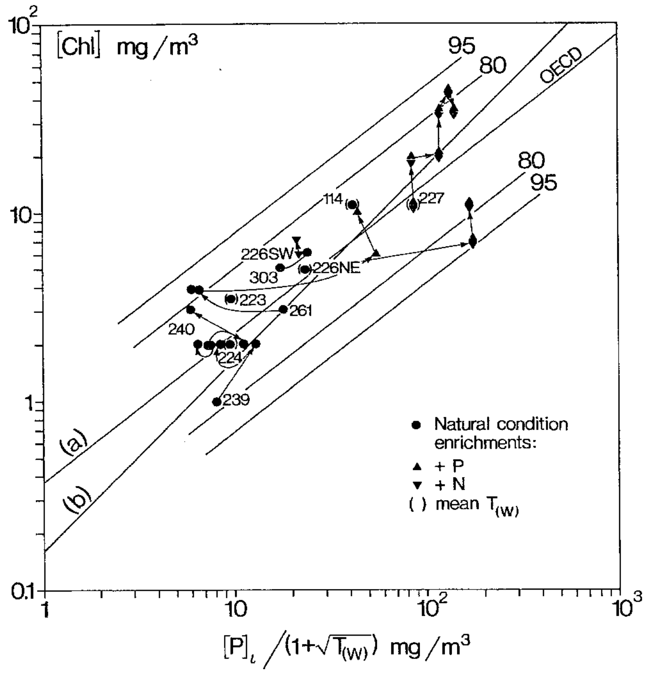

Figure V 12. Annual mean chlorophyll a concentration in relation to flushing corrected inflow total phosphorus concentration

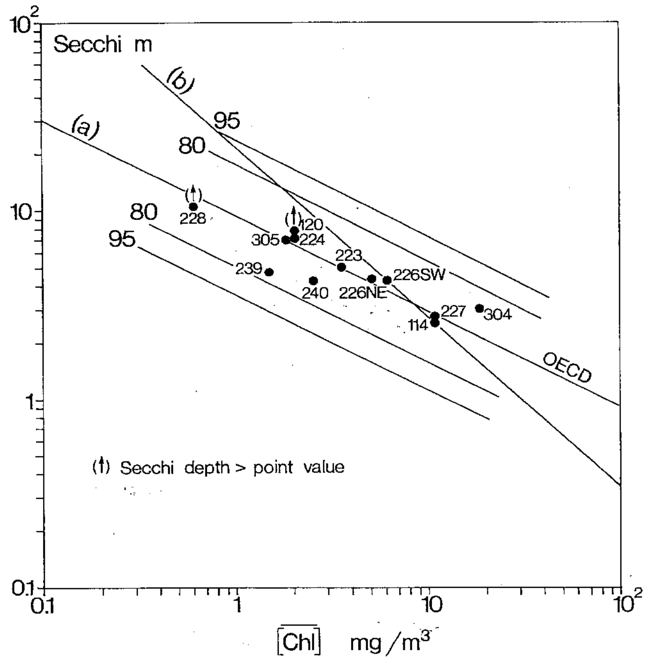

Figure V 13. Secchi transparency in relation to annual mean chlorophyll a concentration

Artificially enriched lakes follow the same pattern and no essential distinction between these and lakes of the natural condition can be made. However, in the case of enrichments, this is largely the effect of two compensating factors. Although [P] is low in relation to loading for enrichments, the tendency for [Chl] to be high in relation to the [P] is exaggerated by the higher than usual percentage of available phosphorus entering via artificial fertilization. This latter effect becomes more apparent as time passes and [P] attains dynamic equilibrium with loadings. In the case of 226 NE only two years of fertilization history are available and during these two years [P] was low in relation to loading, hence, its low position here. Yet the reaction is the same as that of 227; as [P] 'catches up' to what would be expected on the basis of loadings, chlorophyll values increase in relation to loadings.

5.2.5 Secchi Transparency in Relation to Chlorophyll and Phosphorus. All available data on the Secchi disc-chlorophyll relationship fall close to the OECD line, and are definitively within the 80% confidence limits (cf. Figure V 13). The same applies to the Secchi disc-phosphorus relationship (cf. Figure 14). Secchi values for the ELA lakes are a reasonable reference for phosphorus and chlorophyll and confirm these latter measurements. This is important not only in relation to the previous chlorophyll-loading relationship, but also as a reference for judgement of deviations in patterns regarding primary production in the following discussion.

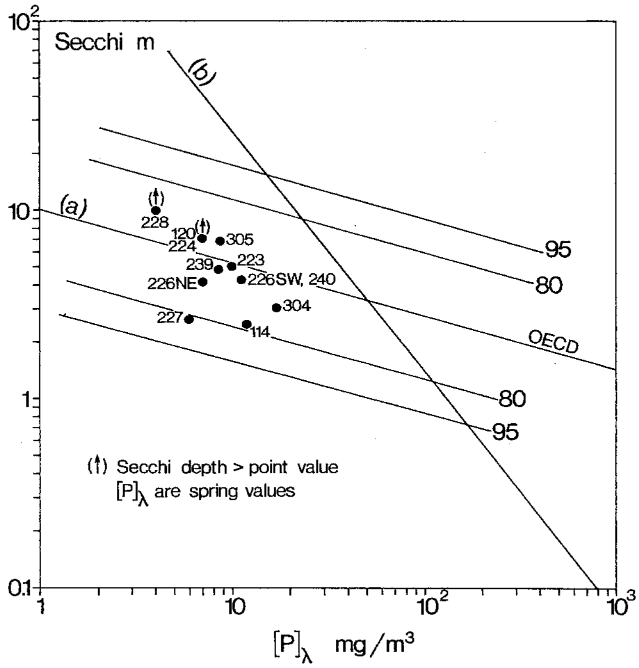

Figure V 14. Secchi transparency in relation to spring total phosphorus concentrations

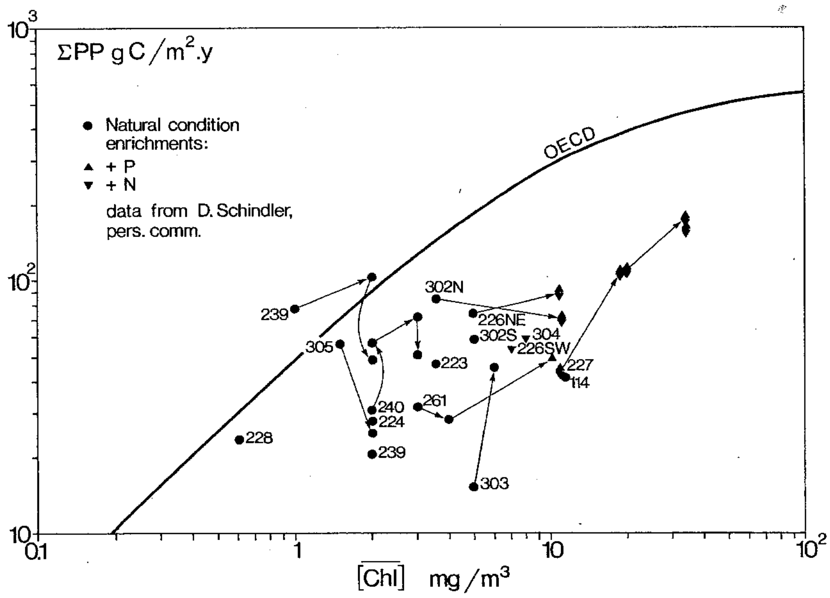

Figure V 15. Annual areal primary production in relation to annual mean chlorophyll a concentration (Schindler)

5.2.6 Primary Production in Relation to Chlorophyll and Loading

The two primary production sets of data available provided by D. Schindler and E. Fee (1979) are a counterpart of the two sets of data described previously in the chlorophyll-phosphorus section. These represent annual means for a period directly following fertilization and growing season means of a period beginning five years after the start of fertilization.

Schindler's early data on primary production, when plotted against chlorophyll and P-loadings respectively (cf. Figures V 15 and V 16) show difficult to explain irregularities. Fee's later data, when plotted against chlorophyll (cf. Figure V 17) on the other hand, show a more consistent pattern. Scattering is reduced considerably, though some irregularities in the behaviour of individual lakes persist. As such, variations in a number of parameters appear to be most erratic immediately following fertilization and tend to dampen out as new equilibria are reached. This may explain why Schindler's earlier values are more irregular than Fee's data describing later years of fertilization.

The position of ELA production values in relation to loading is similar to that of some lakes of Terra Nova and the Muskoka area (cf. V 16) and what is remarkable is that these lakes lie practically one order of magnitude below the OECD line. This also appears in the production-chlorophyll plot. In view of this, chlorophyll values remain at variance in terms of the chlorophyll-phosphorus relationship as discussed previously (and in the biomass-chlorophyll relationship of Addendum V 1).

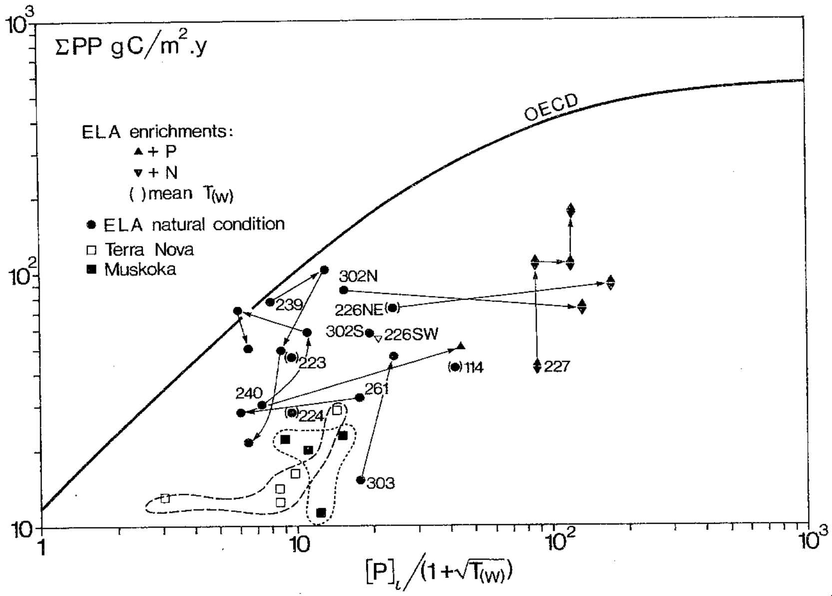

Figure V 16. Annual areal primary production in relation to flushing corrected inflow total phosphorus concentration

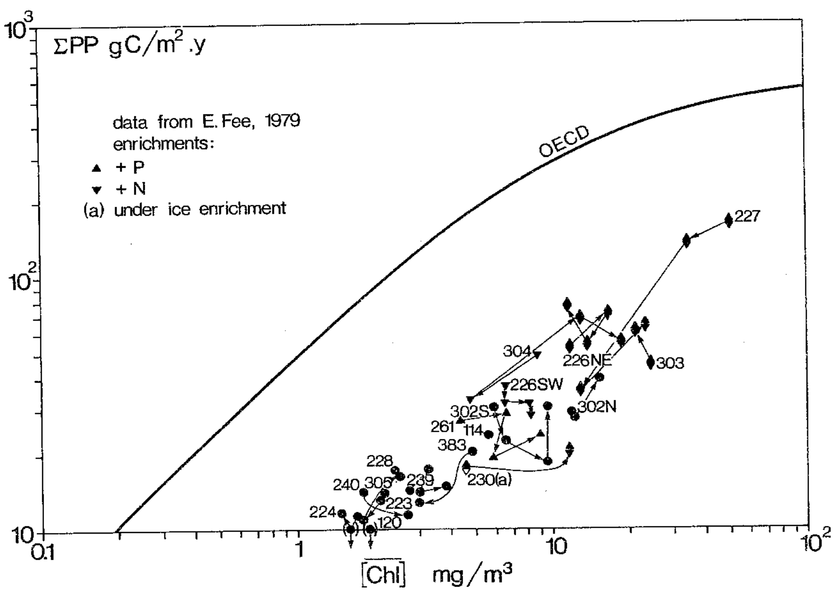

Figure V 17. Annual areal primary production in relation to annual mean chlorophyll a concentration (Fee)

The low positioning can in part be explained by the generally high non-algal extinction coefficient mostly due to the presence of humic substances and other light absorbing sites. With the exception of a few lakes in the area (e.g. 224), this non-algal absorption is at least .6/m, and can be as high as 1.1/m. This is considerably higher than the non-algal absorption in other OECD lakes, which has been estimated in the order of .2 to .3/m on average. This difference in non-algal absorption accounts for some 50% of the lower integral primary production in the ELA lakes.

However, the discrepancy can also be discussed in terms of a lower P utilization of the incoming load. In Figure V 18 the ratio $PP_y/L(P)_y$ has been plotted against $L(P)_y$ which is compared to the same relation calculated for the Alpine Lakes (cf. Fricker 1980). In general, a similar decrease in P utilization as found for the Alpine Lakes, is observed for the ELA lakes, but most of the points pertaining to the ELA lakes lie up to an order of magnitude below the Alpine relationship. Other factors contribute to this: winter production in the ELA is extremely low because of heavy snow accumulations which minimize light penetration, and production values plotted have been corrected for morphometry. Although corrected values are on the average 20% lower than uncorrected values (Fee, 1981), the correction would not be as significant for Alpine Lakes due to the steepness of basin sides and larger size of the latter.

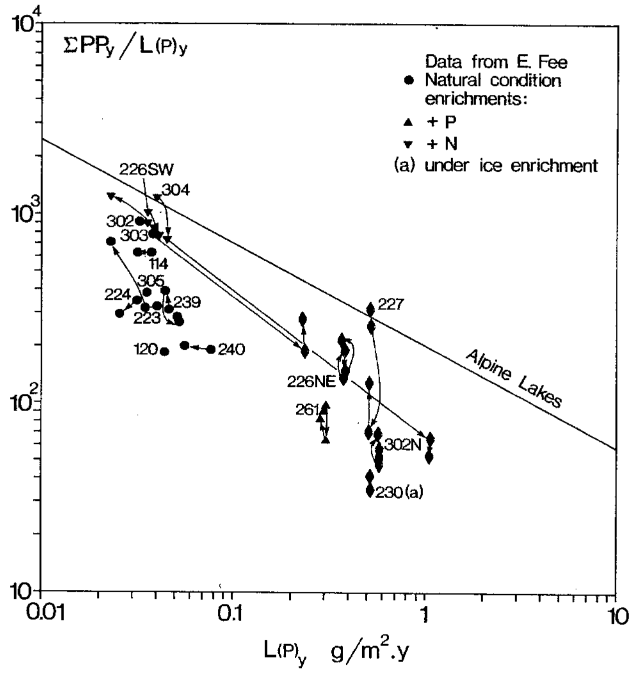

Figure V 18. Annual areal primary production/annual total phosphorus loading ratio in relation to annual total phosphorus loading

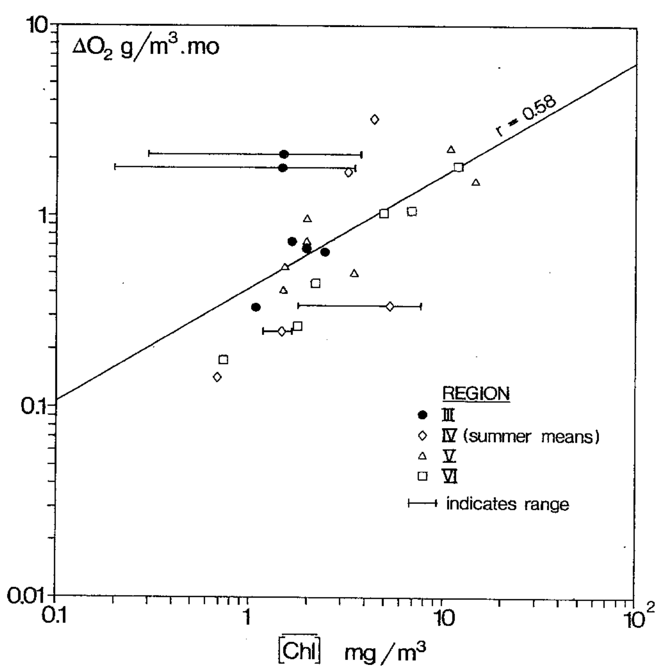

Figure V 19. Mean monthly hypolimnetic oxygen depletion rate in relation to annual mean chlorophyll a concentration

Recapitulating, the following premises have to be collated:

1) The chlorophyll-P relationship for the unfertilized lakes, and the chlorophyll- P -loading relationships for all ELA lakes, do not differ essentially from the corresponding OECD relationships;

2) The biomass-chlorophyll relationship of the ELA lakes corresponds to that of the Alpine lakes (cf. Addendum V 1).

3) The primary production-chlorophyll relationship, and the primary production- P -loading utilization relationship for the ELA lakes are considerably lower in comparison to the Alpine and other OECD lakes.

The inferences to be made from these three points lead to the conclusion that - while in terms of biomass response to loading, ELA lakes conform with the behaviour of other lakes - in dynamic terms, the input required to produce a given level of biomass in the ELA lakes is much lower compared e.g. to the Alpine lakes.

All the implications derived from this conclusion cannot be explored in this report. Yet, it puts into question the suitability of primary production as a comparative trophic index parameter. If the same level of biomass is produced in one case by an input of 1, in another case by an input of 10, then this parameter is of limited value for deriving loading criteria useful for management. Its value for characterizing tropho-dynamic system properties, however, remains unaffected by the above statement. In this sense ELA lakes (and lakes of comparable characteristics) would have to be defined as low dynamic-high output systems in comparison to lakes of e.g. the Alpine type. Morphometric, epi/hypolimnetic relationships, hydrodynamic, biocoenotic structure, and others, appear to be the primary factors needing evaluation in regard to their role in determining tropho-dynamic differences between lakes and groups of lakes.

5.2.7 Hypolimnetic Oxygen Depletion Rates. The regression of hypolimnetic oxygen depletion rate on chlorophyll (given in Figure V 19) shows the relationship between ELA lakes and those of a few other regions. The positioning of lakes seems to be related to the type of enrichment which occurs. In the ELA, this consists of inorganic plant nutrients and in itself does not cause a depletion rate above that from the chlorophyll response and subsequent 'rain' of biomass into the hypolimnion. Among the other lakes, cultural enrichment occurs (cf. eastern and central basins of Lake Erie and Skeleton and Dudley Bay, Figure V 19) and depletion rates are in excess of what could be expected on the basis of the chlorophyll levels, hence, oxidizable materials in excess of phytoplankton production of the epilimnion must enter the hypolimnion.

Hypolimnetic oxygen demand does show some relationship to chlorophyll when extraneous sources of oxidizable materials do not exist, yet simple correlation does not provide a strong enough relationship to be used in a predictive sense ($r = 0.58$). Oxygen depletion rates are dependent on the additional factors of morphometry, ratio between euphotic and tropholytic zones and turbulence (causing entrainment of epilimnetic oxygen or incomplete mixing and buildup of oxygen deficit). An attempt to relate some of these factors is presented in Appendix 3 where a preliminary predictive model is given.

5.3 Region V Conclusions

ELA lakes are comparable with the standard behaviour of OECD lakes, (except in regard to primary productivity). In terms of the response of chlorophyll to phosphorus loading, the OECD standard relationship finds excellent support by the ELA experience where equilibria have been attained. Therefore, one can conclude that this part of the ELA study is directly applicable to eutrophication management. However, in regard to the sediment phosphorus buildup, and indirectly in regard to inferences regarding potential recovery time, the ELA situation shows particularities not directly transferable, and would not be expected to give valid information in regard to recovery time of lakes which have been eutrophied for prolonged periods of time. ELA lakes react very quickly (i.e. more rapidly than one could expect for other situations) to reduced phosphorus load, as shown experimentally, yet the most complete information for L 227 shows that this is likely to be a preliminary stage. Over a ten year period of constant P loading, lake concentrations changed little during the first five years, then increased dramatically over the remainder of the period. Presumably, the rapid rise in phosphorus concentration marked the saturation point for the sediments and as enrichment continued, relationships in this lake approached those of OECD lakes. Sediment history is an extemely important aspect of recovery time.

The particular behaviour of ELA lakes in terms of primary production needs further evaluation and comparison with similar lake situations. It is possible that lakes of the Shield type and similar lakes, exhibit primary production characteristics which deviate essentially from the standard behaviour of the OECD lakes. Uncertain cases in this regard have also been identified among OECD lakes but for lack of sufficient cases for comparison, such deviations could not as yet be satisfactorily analysed.

Addendum V 1

Biomass-Chlorophyll Relationship. Contemporaneous measurements of chlorophyll and biomass during peak periods for a few lakes permit an approximative conversion of chlorophyll to biomass for ELA lakes (cf. Figure V 20), i.e.

$$\text{Biomass (g/m}^3) \simeq .35\ [\text{chl}]^{.86} \qquad r = .90 \qquad ([\text{chl}] \text{ in mg/m}^3).$$

This relationship is close to that found for Alpine lakes but lower than that established for the Great Lakes. How far this difference from the Great Lakes is a reflection of differences in population growth phase, species composition, percentage of nutrients in available form or extraction methodology, is difficult to say. However, it is an indication of the level of uncertainty connected with the use of chlorophyll measurements as an indicator of biomass.

The above relationship and the chlorophyll-P relationship (cf. Figure V 2) permit an approximation of the amount of phosphorus tied up in biomass relative to a given total phosphorus concentration of lake water. Accordingly, 10 mg/m³ chlorophyll correspond to 2500 mg/m³ phytoplankton fresh weight on average for ELA lakes. Assuming a phytoplankton volume/biomass-P ratio of 400 (volume/c $\approx$ 10; C/P $\approx$ 40), then about 6 mg P are associated with 2500 mg/m³ phytoplankton volume. Furthermore, 10 mg chlorophyll are also associated with 40 mg total phosphorus in lake water when the OECD standard relationship is used as reference or about 20 mg/m³ of total phosphorus when the ELA regression is used. Therefore, the amount of phosphorus tied up in biomass would lie between 15 to 30% on average using the OECD reference, or 30 to 60% by the ELA reference.

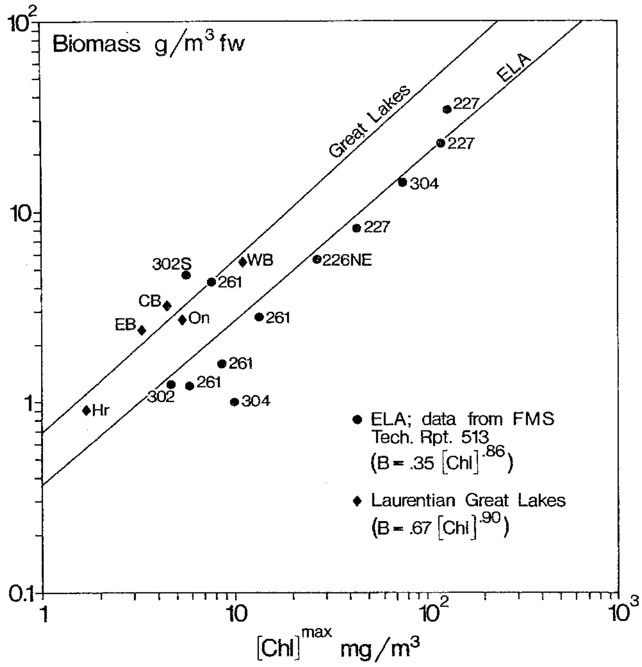

Figure V 20 Phytoplankton biomass in relation to peak chlorophyll a concentration

In consideration of this range of possibilities, it follows that some 15 to 60% of the total P measured in lake water is associated with phytoplankton biomass, and overall, 30% would represent a most likely average estimate.

5.4 References (V)

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CHAPTER 6. PRAIRIE LAKES REGION VI

- Lake Winnipeg
- Qu'Appelle River Lakes
  - Pasqua Lake (Ps)
  - Echo Lake (Ec)
  - Mission (Lebret) Lake (Ms)
  - Katepwa Lake (Kt)
  - Crooked Lake (Cr)
  - Round Lake (Rd)

6.1 Preamble

The seven lakes discussed in the following belong to the Nelson-Saskatchewan River basin draining into Hudson Bay (Fig. VI 1). The term "prairie lakes" is not entirely correct and is used here only in the sense that they belong geographically to the south-central area of Canada. Typical prairie lakes of that region are small in size and depth, often not connected to major river systems, and often are highly eutrophic (hypertrophic). This type of lake has not been included in the OECD study and comparison of their characteristics with the majority of the OECD lakes would be inappropriate.

Further, because of the substantial difference in limnological characteristics, the relatively small lakes of the Qu'Appelle river system have been treated separately from Lake Winnipeg. The latter would be more comparable to other large Canadian lakes, although its particular characteristics and the data available for such lakes do not allow a generalized discussion at this time.

6.2 Lake Winnipeg, Description of Location

Lake Winnipeg, the twelfth largest lake in the world, is a valuable resource to the people of Manitoba and Canada and - water use over the past century has increased greatly. Irrigation has been increasing continuously since the early 1900's. The lake has a sizeable commercial fishery with annual landings of approximately ten million pounds. The beaches of southern Lake Winnipeg are a major recreation attraction and are heavily utilized, especially by residents of Winnipeg.

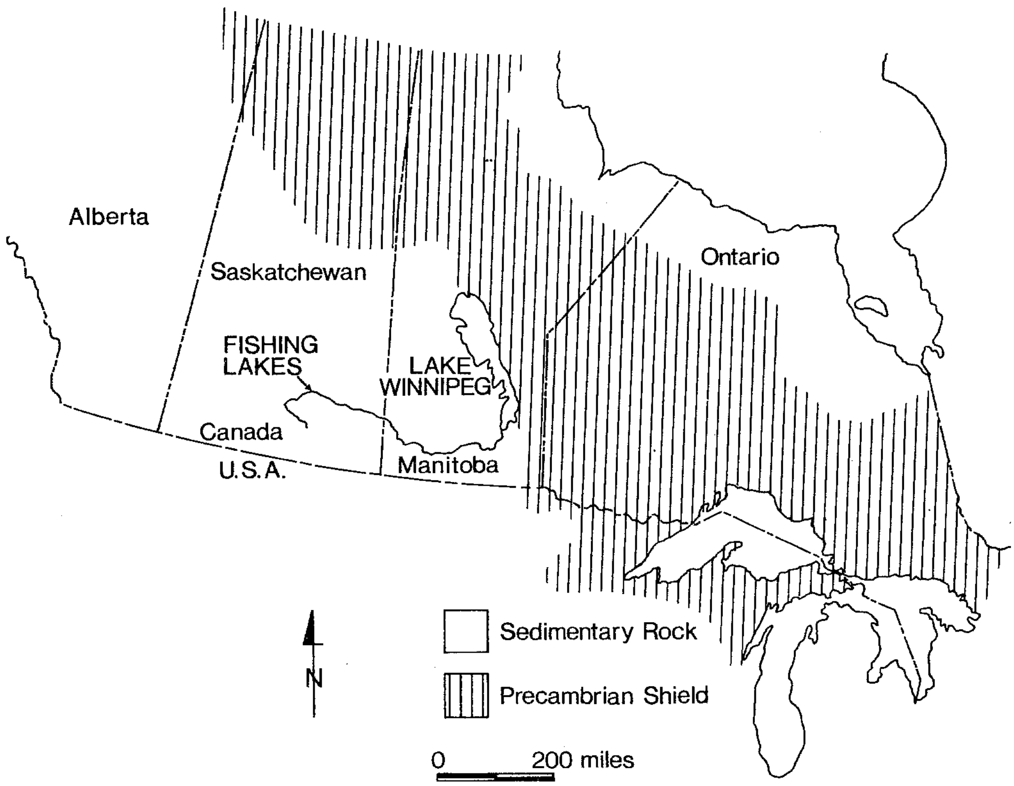

Figure VI 1. DRAINAGE AREA OF LAKE WINNIPEG AND THE FISHING LAKES

Certain portions of the adjacent wetlands are important waterfowl breeding areas.

Urban, agricultural and industrial developments in the Lake Winnipeg basin are not concentrated along the lake shoreline, and consequently, the direct impact of wastes from the more than four million people living in the drainage basin has often been more dramatic in the rivers and streams than in Lake Winnipeg itself. On the other hand, complaints of algal growth and the recent closure of commerical fishing on Lake Winnipeg as a result of mercury concentrations in fish from the lake are clear indications that the sheer size of the lake and its spatial separation from major sources of pollution are not enough to protect water quality.

Climate of the basin may be considered continental with temperatures as high as 38°C in the summer months and lows of -40°C not uncommon during the winter period. Seasonal transition is very rapid and spring and autumn are nearly non-existent.

Precipitation ranges from a mean annual total of more than 43 cm in the eastern and western limits of the basin to less than 22 cm near the southern part of the basin. Over a large portion of the basin, average annual evaporation exceeds average annual precipitation. Hence, calcium and other ions are drawn to the surface by capillary action and the soils are typically pedocals. These soils are varied in texture and depth, and may be as thick as 150 metres at some points.

The huge drainage basin of Lake Winnipeg (nearly 1 million km^2) extends from the Rocky Mountains on the west to the Great Lakes basin on the east and from the Missouri-Mississippi basin on the south. It has three physiographic regions: the mountains and foothills, the plains, and the Canadian Shield. Of the three regions, the plains region is by far the largest, making up approximately 75% of the total area. It is a region of low relief with an average slope from east to west of about 0.5 metres to the kilometer and this leaves drainage patterns poorly developed with many lakes, sloughs and marshes having no defined outlet.

Lake Winnipeg has a surface area of 23,750 sq. km, and maximum depth of 36 m, a mean depth of 12 m, and a theoretical water residence time between 2.9 and 4.3 years. The lake consists of a large north basin and a small south basin, separated by a narrow, irregular shaped channel. The lake is situated on the boundary between the Precambrian Shield on the east and Palaeozoic and Mesozoic sedimentary strata on the west and south (cf. Figure VI 1). Rivers flowing to Lake Winnipeg from the east are generally low in concentrations of nutrients, major ions (specific conductivity <100 μmhos/cm) and suspended material; rivers draining to the lake from the west and south generally have higher concentrations of nutrients, major ions (specific conductivity 300 to 1700 μmhos/cm), and suspended material.

From the studies conducted from 1969 to 1974, the nutrient load to Lake Winnipeg is well known, amounting to 7300 ± 1600 t/y of phosphorus, and 90,000 + 13,000 t/y of nitrogen, of which 78% for phosphorus and 59% for nitrogen enters via the south basin. Accordingly, the average loadings per unit surface calculated for the whole lake amount to 0.31 ± 0.07 $g/m^2.y$ for phosphorus, and 3.79 ± 0.55 $g/m^2.y$ for nitrogen, whereas these loadings into the south basin alone would result in 1.94 ± 0.43 $g/m^2.y$ for phosphorus, and 17.5 ± 2.53 $g/m^2.y$ for nitrogen. (Condensed from Hamilton et al., Brunskill, Kenney).

6.3 Trophic Repsonse - Nutrient Relationships

6.3.1 Chlorophyll - Loading Relationship. In accordance with these loading values, the main problems of eutrophication occur in the southern basin for which chlorophyll values may reach 150 mg/m^3. However, biomass (chlorophyll) values show a strong south to north differentiation, wherefore calculation of average values are difficult. Approximative mean values have been given as 4.2 mg/m^3 for the south basin, and 2.8 mg/m^3 for the whole lake.

The scant figures available for OECD purposes have been plotted in Figure VI 2 and VI 3. The strong negative deviation from the expected line would appear to indicate limitation by some growth factor. However, humic materials, high silt loads and turbulent resuspension of sediments in Lake Winnipeg reduce light penetration. The Secchi disc range reported varies from 0.05 to 3.5 m, which makes it likely that the controlling factor is light availability. On the other hand, bulk water residence time, in contrast to the Qu'Appelle Lakes, is less variable from year to year; however, because of strong water level oscillations and resonance phenomena caused by the particular morphology of Lake Winnipeg, the actual flushing rate of the southern basin could be considerably higher (0.1 to 0.4 years). If corrected for this, the chlorophyll-loading relationship for the south basin would come closer to expectation.

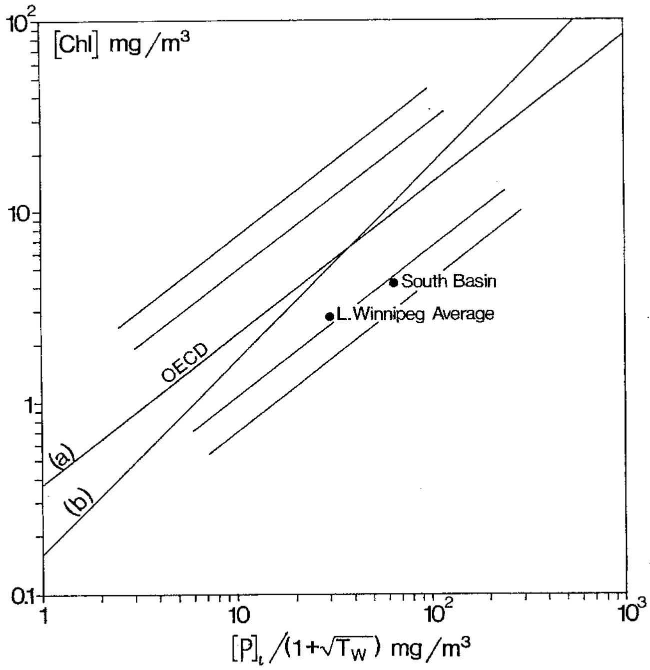

Figure VI 2. Several year mean chlorophyll a concentration in relation to flushing corrected annual mean total phosphorus inflow concentration, Lake Winnipeg

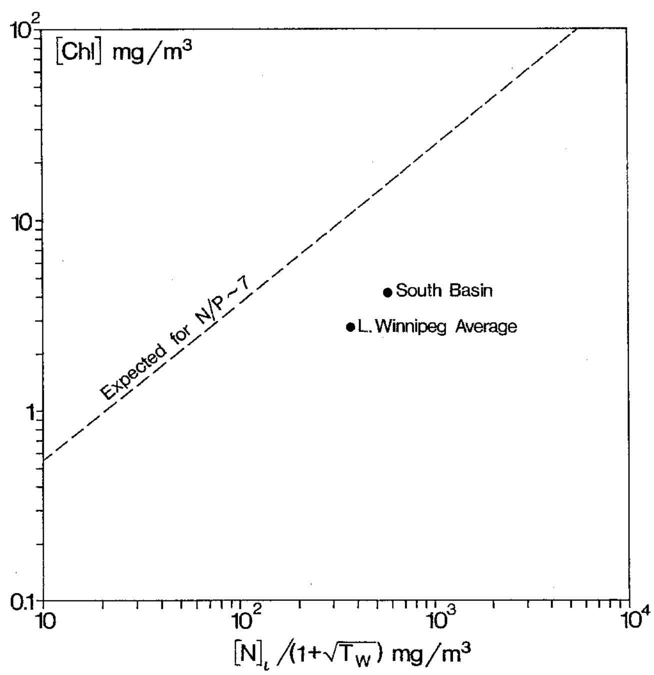

Figure VI 3. Several year mean chlorophyll a concentration in relation to flushing corrected annual mean total nitrogen inflow concentration, Lake Winnipeg

6.4 Assessment for Lake Winnipeg

Although the available data do not entirely satisfy the needs of the OECD study, confrontation of the Lake Winnipeg situation with OECD results shows up in substantial deviations due to physical conditions (light penetration) controlling the production level, rather than due to nutrient limitation. However, a vast amount of material collected by the authors is still under elaboration, and further modifications may be forthcoming.

6.5 Qu'Appelle Valley Lakes, Description of Location

The Qu'Appelle River basin (ca. 52,000 km²) is located between 50° 00' and 51° 30' N Lat. The basin is gently rolling to undulating, with a general elevation above sea level of about 535 m at the Qu'Appelle Dam to about 397 m inside the Manitoba border, where it joins the basin of the Assiniboine River. The Qu'Appelle Valley is a trench some 430 km long and 1 to 3 km wide at the top. It varies from 33 to 106 m below the level of the surrounding terrain and its depth increases eastward. A chain of six lakes formed by expanded sections of the river basin is known as the Fishing Lakes and occupies this valley (cf. Figure VI 4). Upstream of the Fishing Lakes, the Qu'Appelle River system contains several lakes, the largest being Last Mountain Lake (227.5 km²) and Lake Diefenbaker, a large impoundment closed in 1966.

The climate of the general area is dry subhumid bordering on semiarid of the continental type. The mean air temperature varies from -18.3°C in January to 18.3°C in July. The mean annual temperature is

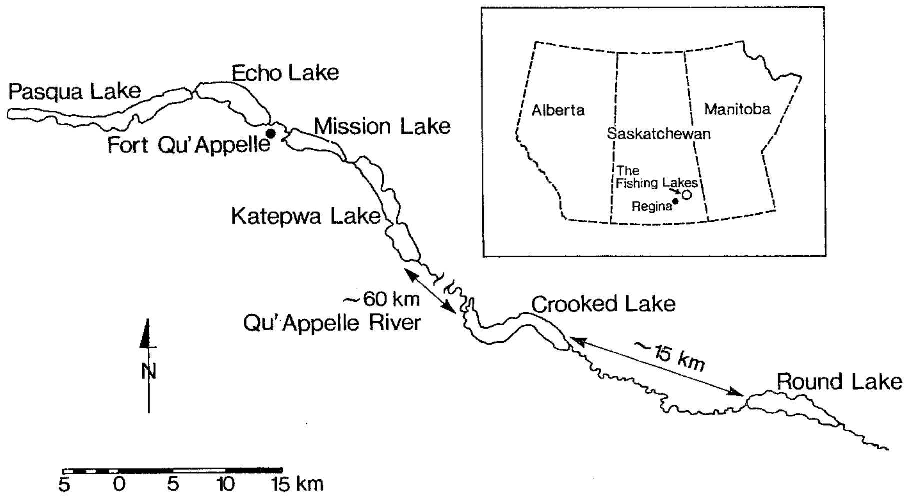

Figure VI 4 The Qu'Appelle Valley Lakes of Saskatchewan

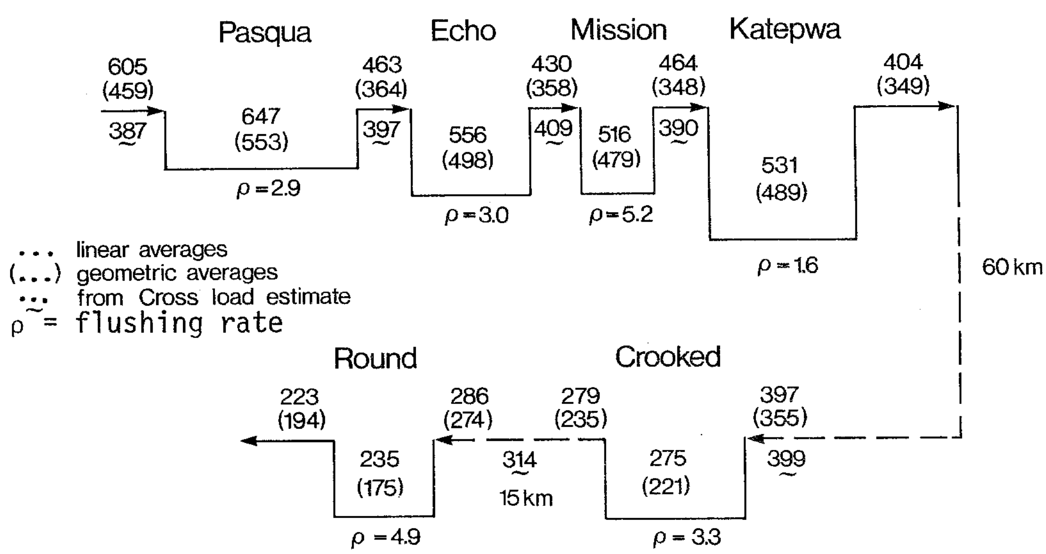

Figure VI 5. Qu'Appelle Phosphorus Budget
(average concentrations, mg/m³)

1.7°C with recorded extremes of 43.3°C and -49°C. The mean annual precipitation is 37.3 cm with about 20% falling as snow, which persists with the ice cover for 5 to 6 months. Over two-thirds of the precipitation falls contemporaneously with the growing season and the resultant runoff carries a high nutrient load with it. The Qu'Appelle Valley, on the other hand, contains elements of the eastern deciduous forest. These include green ash, white birch, black poplar, white elm and box elder. The forest growth also includes the typical tall shrubs found in the black soil zone. In regard to the surrounding zones, the drainage area west of Loon Creek lies in the dark brown soil zone and supports mixed prairie vegetation. The area east of Loon Creek is characterized as the black soil zone, covered by transitional parkland vegetation.

The drainage area is cultivated extensively with wheat, oats and barley as the principal crops. Non-arable uplands, lowlands and valley slopes are grazed by cattle. Irrigation is carried on to a limited extent on the Qu'Appelle Valley floor, with hay, grain and vegetables being the main crops.

The fisheries of the upper Qu'Appelle River system lakes have long been important to people living in the area. Commercial fishing dates back to the late 19th century. In the last decade the principal commercial fish caught were whitefish, ciscoes and buffalo fish. This accounts for 1/3 of Canada's freshwater fish catch.

The lakes of the Qu'Appelle River system are being used on an ever-increasing scale for recreation and summer cottage development. A sport fishery has become well established on these lakes with northern pike, walleye and perch being the principal fish caught.

The lakes as well as the streams are utilized for stock watering, limited irrigation, and drinking water supply. Wascana and Moose Jaw Creeks are used to transmit domestic sewage from Regina and Moose Jaw, respectively, to the Qu'Appelle River and thence, into the Fishing Lakes some 50 km downstream. Before 1975 the sewage received secondary treatment (in effect) in aerated lagoons prior to discharge into the streams, with an excessive nutrient load eventually reaching the Fishing Lakes. In 1975 and 1976 a tertiary treatment plant (90% phosphorus removal) was operating intermittently, and since 1977 for most of the year.

With the series of dams now established on the Qu'Appelle River system and the availability of water from Lake Diefenbaker, the PFRA (Prairie Farm Rehabilitation Association) is in a position to maintain lake levels and also to use the system for flood control. Droughts and floods occur periodically and their effects must be mediated for maximal multiple use benefits.

These lakes are rarely, or only weakly stratified during summer. Stratification is short-lived as a result of almost constant winds and shallow basins (mean depth extending from 5.8 m in Pasqua Lake to 14.1 in Katepwa) which makes them distinct from other lakes of this treatise.

In the lakes treated in the following discussion, which includes the Fishing Lakes (Pasqua, Echo, Mission or Lebret, Katepwa) and some 60 km downstream, Crooked and Round Lakes, salinity is relatively high (dominated by sodium and sulphates). Average flushing time for all lakes is less than one year but the hydraulic load varies so drastically from year to year that flushing rates may vary from less than once per year to more than ten times per year (cf. Table VI 7). This high variation in flushing and the respective differentiation between lakes, is of considerable importance for their limnological characteristics.

Although naturally eutrophic, these lakes have deteriorated in their trophic conditions with the development of agriculture in the prairies, compounded by sewage discharges from the two main centres (Regina and Moose Jaw) lying upstream from the lakes. Overall nutrient load to the lakes appears to be among the highest reported in Canada and elsewhere, resulting approximately in 1/3 from natural runoff, 1/3 from agriculture and 1/3 from municipal components. However, it is noteworthy that the estimated "effective" drainage area from which these lakes receive runoff, is considerably less (ca. 15,000 km^2 at Round Lake) than the total drainage area, yet still high in relation to the surface areas (ranging from 7.6 km^2 for Mission Lake to 19.9 km^2 for Pasqua Lake) of the lakes in question.

These various aspects make these lakes distinct from the majority of lakes studied in the OECD Programme which results in considerable deviation from the standard behaviour of other lakes. (Condensed from Hammer 1971, Allan et al. 1978, and Cross 1979).

6.6 Trophic Response-Nutrient Relationships

6.6.1 Nutritional Conditions in the Lakes. These are summarized for total phosphorus and total nitrogen as averaged over a 6 to 7 year period (1970/1977) in Tables VI 1 to VI 4. Generally speaking, average phosphorus and nitrogen concentrations decrease significantly from Pasqua Lake at the upper end to Round Lake at the lower end of the chain (cf. Table VI 5). However, considerable variations in nutrient content from year to year and throughout the seasons occur in all the lakes independently of their position in the chain. Therefore, averages over a 7 year period are of but limited value to characterize the nutritional conditions of these lakes for any particular year, as evidenced by the high standard deviations. Also, no essential improvement in this regard is possible using log-transformed values and geometric averages instead of linear averages. Linear averages are systematically higher, due to positively skewed distribution of the data sets for each lake. Therefore, geometric averages would seem to be somewhat more representative.

The nutritional conditions of all the lakes, and in particular the Fishing Lakes, are characteristic for hypertrophic lakes. The low average N/P ratio of all these lakes (cf. Table VI 6) points to the possibility that nitrogen limitation, at least temporarily, can occur, but the variability of the N/P ratio over time is considerable (cf. below).

6.6.2 Nutrient Supply and Nutrient Budgets. It is hardly possible to construct indisputable phosphorus and nitrogen budgets for these lakes (cf. Figures VI 5 to VI 7), whether this is attempted from linear or from log-transformed data.

Table VI 1. Qu'Appelle Valley Lakes. Total Phosphorus mg/m^3; Statistical Averages 1970-77

Lake	Linear Average ± St. Dev.	Coeff. Var.	^{10}log Average ± St. Dev.	Coeff. Var.	Geom. Average	No. of Observations
Pasqua	647 ± 372	.57	2.743 ± .242	.09	553	370
Echo	556 ± 206	.37	2.697 ± .211	.08	449	278
Mission	516 ± 196	.38	2.680 ± .185	.07	479	155
Katepwa	531 ± 270	.51	2.689 ± .178	.07	489	280
Crooked	275 ± 172	.63	2.345 ± .321	.14	221	172
Round	235 ± 161	.69	2.242 ± .351	.16	175	101

Table VI 2. Qu'Appelle Valley Lakes. Total Nitrogen mg/m^3; Statistical Averages 1970-77

Lake	Linear Average ± St. Dev.	Coeff. Var.	^{10}log Average ± St. Dev.	Coeff. Var.	Geom. Average	No. of Observations
Pasqua	2994 ± 1976	.66	3.418 ± .215	.06	2620	370
Echo	1989 ± 780	.39	3.283 ± .190	.06	1920	278
Mission	1801 ± 691	.37	3.231 ± .154	.05	1700	155
Katepwa	1880 ± 999	.53	3.230 ± .184	.06	1700	280
Crooked	1827 ± 1031	.56	3.204 ± .220	.07	1600	172
Round	1586 ± 773	.49	3.158 ± .189	.06	1440	101

Table VI 3. Long-term Averages of Phosphorus Inflow-Outflow Concentrations (mg/m³).

Inflow - Outflow	Linear Average ± St.D.	Geometric Average (log ± St. D.)	No. of Observations
Pasqua in	605 ± 472	459 (2.66 ± .34)	95
Pasqua out / Echo in	463 ± 279	364 (2.56 ± .34)	102
Echo out / Mission in	431 ± 213	358 (2.55 ± .30)	95
Mission out / Katepwa in	464 ± 193	348 (2.54 ± .26)	77
Katepwa out	404 ± 185	349 (2.54 ± .26)	98
Crooked in	397 ± 145	355 (2.55 ± .23)	70
Crooked out	279 ± 145	235 (2.37 ± .29)	60
Round in	286 ± 93	274 (2.43 ± .12)	30
Round out	223 ± 102	194 (2.28 ± .26)	37

Table VI4. Long-term Averages of Nitrogen Inflow-Outflow Concentrations. (g/m^3).

Inflow - Outflow	Linear Average ± St.D.	Geometric Average (log ± St.D.)	No. of Observations
Pasqua in	3.75 ± 2.38	3.14 (.496 ± .263)	93
Pasqua out / Echo in	2.57 ± 1.02	2.38 (.377 ± .170)	100
Echo out / Mission in	2.11 ± .73	2.01 (.302 ± .131)	95
Mission out / Katepwa in	2.09 ± 1.16	1.87 (.272 ± .199)	77
Katepwa out	1.86 ± 1.07	1.71 (.234 ± .163)	97
Crooked in	1.76 ± .77	1.63 (.213 ± .167)	68
Crooked out	1.73 ± .88	1.55 (.189 ± .210)	59
Round in	1.52 ± .45	1.46 (.166 ± .117)	28
Round out	1.42 ± .56	1.31 (.118 ± .175)	83

Table VI 5. Confidence level of significance for differences between consecutive lakes (P value).

A. Lake Concentrations	Phosphorus	Nitrogen
Pasqua - Echo	> .99	> .99
Echo - Mission	.05	.98
Mission - Katepwa	not sign.	not. sign.
Pasqua - Katepwa	> .99	> .99
Katepwa - Crooked	> .99	not sign.
Crooked - Round	≈ .05	.05

B. Inflow - Outflow Concentrations	Phosphorus	Nitrogen
Pasqua	.98	> .99
Echo	not sign.	.99
Mission	not sign.	not sign.
Katepwa	.95	not sign.
Pasqua - Katepwa	> .99	> .99
Crooked	> .99	not sign.
Round	> .95	not sign.
Crooked - Round	.99	.99

Table VI 6. Qu'Appelle Valley Lakes. N/P Ratio; Statistical Averages 1970-77

Lake	N/P Ratio $\pm \sigma$	Coeff. Var.	Range	No. of Observations
Pasqua	4.63 $\pm$ 3.96	.96	1 to 33.6	370
Echo	3.58 $\pm$ 1.98	.54	1 to 24.8	278
Mission	3.49 $\pm$ 1.88	.54	1.1 to 20.0	155
Katepwa	3.54 $\pm$ 2.60	.74	1 to 19.5	280
Crooked	6.64 $\pm$ 5.60	.84	1.3 to 46.1	172
Round	6.75 $\pm$ 5.67	.84	1.6 to 43.7	101

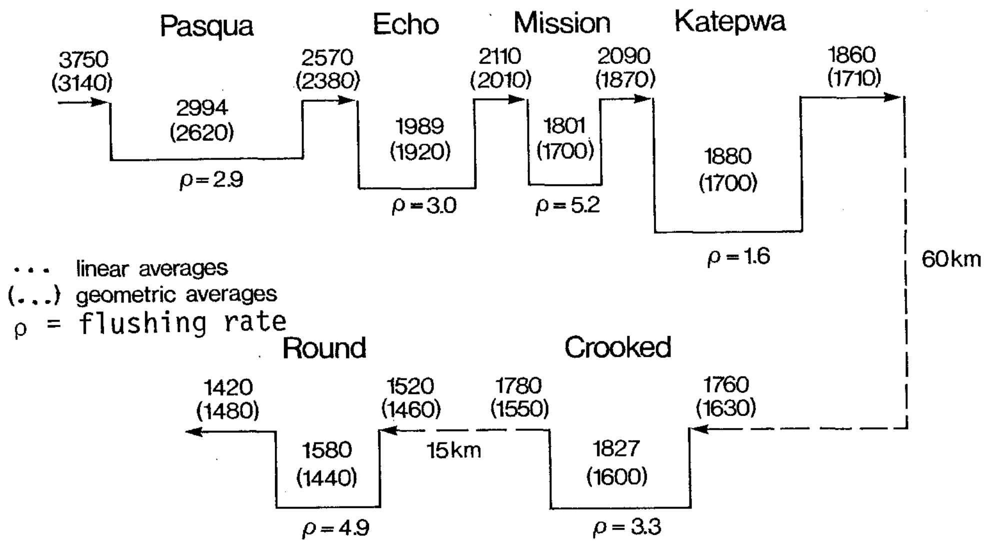

Figure VI 6. Qu'Appelle Nitrogen Budget
(average concentrations, mg/m^3)

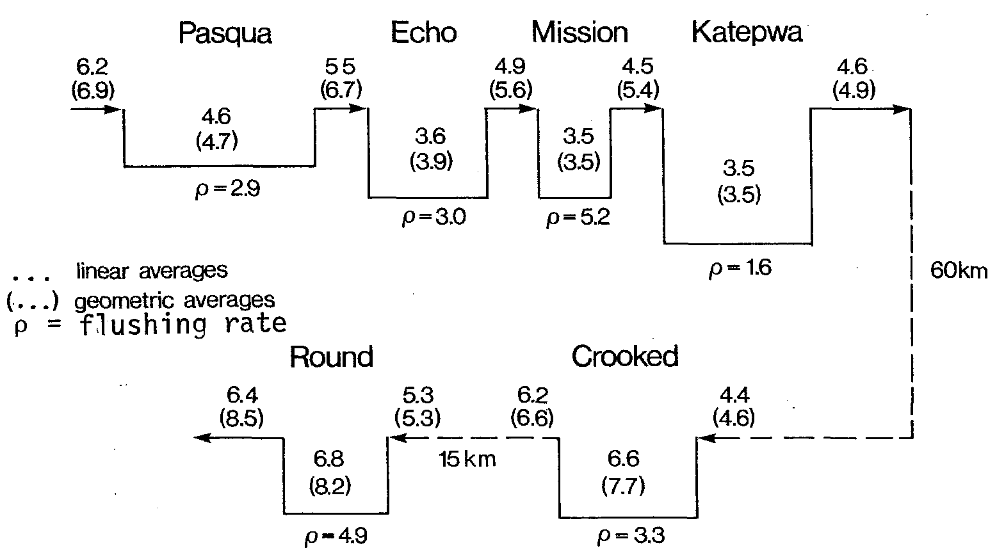

Figure VI 7. Qu'Appelle N/P Ratios

Inflow-outflow concentrations are somewhat more consistent with measured average lake concentrations for nitrogen than those for phosphorus, and as a whole, the lakes act sequentially as sinks for nitrogen. The average nitrogen outflow concentration of Katepwa Lake drops to about 50% of the Pasqua inflow concentration, and the outflow concentration of Round Lake is about 40% of the Pasqua inflow concentration. Considered singly, the differences between inflow-outflow concentrations of Crooked and Round Lakes respectively, are statistically not significant, but the difference between Crooked inflow and Round outflow is significant at the P = .01 level. Therefore, while the upstream Fishing Lakes appear to be a somewhat stronger sink for nitrogen, the two lower lakes also act as nitrogen sinks.

In regard to phosphorus, the picture is confused by the average lake concentrations of the Fishing Lakes being systematically above the estimated average inflow concentrations. Cross (1978), in trying to account for apparently negative retention coefficients, has forwarded the claim that phosphorus supply by way of groundwater infiltration is an important internal source for phosphorus in these lakes. On the otherhand, decrease along the chain and observation that outflow concentrations are systematically below average lake concentrations, are inconsistent with such an assumption. This would mean that - although internal load cannot be ruled out for limited periods of time, over a several year period the Fishing Lakes chain either acts as a modest sink for phosphorus, or as a flow-through system.

The claim for internal supply of phosphorus is further suspect in view of the nitrogen situation; indeed, no corresponding evidence for internal nitrogen supply exists. There is no principal reason to assume that - whilst phosphorus supply by way of groundwater infiltration is possible - the same groundwater flow should not contain substantial amounts of nitrogen to similarly affect the nitrogen budget, as is hypothesized for phosphorus.

It seems that in contrast to the Fishing Lakes, the relative balance of phosphorus and nitrogen in the lower lakes is reversed. Whereas the former are active sinks for nitrogen, but probably not in the same way for phosphorus, the lower lakes retain more phosphorus than nitrogen. Indeed, for the lower lakes (Crooked and Round), the difference between inflow and outflow P concentrations is significant, at least at the $P = .05$ level for either lake.

The impossibility of constructing meaningful nutrient budgets for the Qu'Appelle Lakes depends, in part, on the inadequacy of the data available, and in part, and more importantly, on the enormous year to year variability in the hydrological regime of these lakes, and the connected variability in year to year nutrient loads. This shall be exemplified with the Pasqua Lake situation, which would equally apply to the downstream lakes of the chain. For loading estimates, Cross has used equations derived from correlating daily loads ($Q.[P]_i$) with the hydraulic load (Q). In order to avoid potential autocorrelation, we have instead estimated the relation between inflow concentrations and hydraulic load using the Pasqua Lake inflow data. Our equations are slightly different from the original ones, yet, when used for year by year estimate of phosphorus loads, the end results (though somewhat higher) are not significantly different from Cross's loading estimates (cf. Table VI 7). The variation in yearly nutrient loadings during the 7 year period extends over a scale of between 1 : 3 to 4, and average inflow concentrations vary by a factor of 1 : 2 to 3. Accordingly, if referred to yearly conditions, it becomes practically impossible to deduce from the available data whether inflow, outflow and lake concentrations are statistically distinct or not, i.e. it is not possible to construct nutrient budgets from yearly averages. Given the hydrological characteristics, therefore, reliable yearly nutrient budgets for these lakes would only be obtainable if the sampling programme were spaced more densely in time than that carried out by the investigators. The implications for monitoring design and management are evident, and shall be considered in the summary review.

6.6.3 Chlorophyll. The high variability in nutritional conditions also reflects itself in the high variability of phytoplankton biomass measured as chlorophyll (cf. Table VI 8). Unfortunately, the two sets of information (nutrients, chlorophyll) are not directly comparable because of the less frequent measurement of chlorophyll. Also, the authors report having had difficulties with these measurements. The level of perturbation of the Qu'Appelle Lakes - which is higher for these lakes than for any others considered in this report - is clearly

Table VI 7. Pasqua Lake. Yearly Phosphorus Loading and Average Inflow-Outflow Concentrations.

Year	Load and Average Inflow Concentration from Regression[1]			Average Inflow Conc. from Individual Measurements	Average Outflow Conc. from Individual Measurements	Significance Level of Difference	Mean Lake Concentr.
	10^3 kg/y	$\bar{\rho}$	mg/m³	mg/m³	mg/m³	P	mg/m³
1970	135 (177)	2.66	440 (579)	564 ± 183 (17)[2]	724 ± 220 (16)[2]	.05	
71	107 (133)	1.93	483 (600)	946 ± 691 (9)	678 ± 179 (9)	not	
72	81 (72)	1.02	697 (616)	585 ± 252 (9)	594 ± 263 (7)	not	
73	71 (53)	.62	1002 (743)	523 ± 291 (7)	773 ± 206 (9)	not	
74	240 (155)	5.66	369 (239)	791 ± 901 (10)	234 ± 159 (26)	.01	
75	182 (164)	4.13	384 (347)	410 ± 290 (17)	246 ± 141 (10)	not	
76	196 (147)	4.28	388 (299)	445 ± 413 (15)	354 ± 202 (15)	not	
77	-	-	-	802 ± 390 (11)	453 ± 157 (10)	.05	
Averages	145 ± 63 (129 ± 48)		538 ± 234 (489 ± 191)	605 ± 472 (95) (578) (1970-76)	463 ± 279 (102) (464) (1970-76)	.02	647 ± 372

1) Figures in parentheses = Loadings and mean inflow concentrations from Cross 1978.

2) Figures in parentheses = Number of Observations.

Table VI 8. QU'APPELLE VALLEY LAKES

Chlorophyll mg/m^3 Statistical Averages(1970/77)

Lake	Linear Average $\pm$ St.D.	Coeff. Var.	^{10}log Average $\pm$ St.D.	Coeff. Var.	Geomet. Mean	Peak Observed	No. of Observations
Pasqua	25.5 $\pm$ 33.8	1.33	1.105 $\pm$ .532	.48	12.7	205	162
Echo	33.6 $\pm$ 50.5	1.50	.995 $\pm$ .598	.60	9.9	366	139
Mission	26.0 $\pm$ 37.0	1.42	1.106 $\pm$ .540	.49	12.8	254	64
Ketepwa	21.2 $\pm$ 46.0	2.17	1.044 $\pm$ .449	.43	11.1	420	100
Crooked	15.3 $\pm$ 24.0	1.57	.973 $\pm$ .407	.42	9.4	194	71
Round	9.3 $\pm$ 9.4	1.01	.789 $\pm$ .400	.51	6.2	39	31

evidenced by the standard deviations of log-transformed chlorophyll values. According to experience, the standard deviation of log-transformed chlorophyll data from lakes of normal variability should lie in the order of 0.33 ± 0.07; on the other hand, perturbated systems show up with overdispersion. All the Qu'Appelle Lakes show typical overdispersion of the chlorophyll data, characteristic of lakes of unusual production dynamics.

6.6.4 Nutrient-Chlorophyll Relationship. In Figure VI 8 to VI 12, average and peak chlorophyll levels are plotted against average total phosphorus and average total nitrogen, respectively. The problem of correctly interpreting these plots is similar to that discussed for nutrient budgets. Nonetheless, it can be said that average biomass levels for all the lakes in question are considerably lower than what one would expect from either the standard chlorophyll-phosphorus relationship or a derived chlorophyll-nitrogen relationship. If the biomass of these lakes were controlled univocally by one or the other nutritional factor, then this would result in a closer position of the chlorophyll values to one or the other OECD standard regression line.

Figures VI 8 to VI 10 show that none of the lakes discussed here is meeting the criterion to claim these lakes are phosphorus controlled, whether one considered as a reference concentration the average lake or the average outflow concentration, and also, regardless of whether one considers linear or the corresponding geometric averages. All points remain significantly below the OECD standard line. The same is true when plotting average chlorophyll against average nitrogen

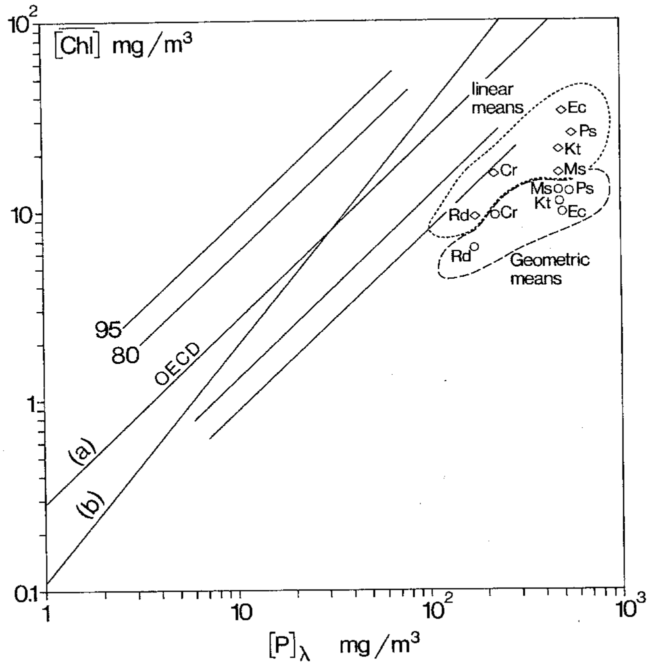

Figure VI 8. Several year linear and geometric mean chlorophyll a concentration in relation to average inlake total phosphorus concentration

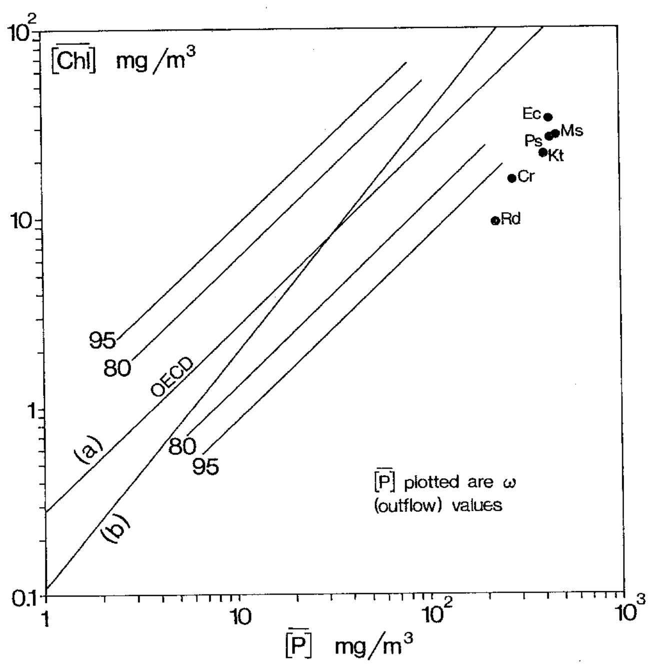

Figure VI 9. Several year mean chlorophyll a concentration in relation to average outflow total phosphorus concentration

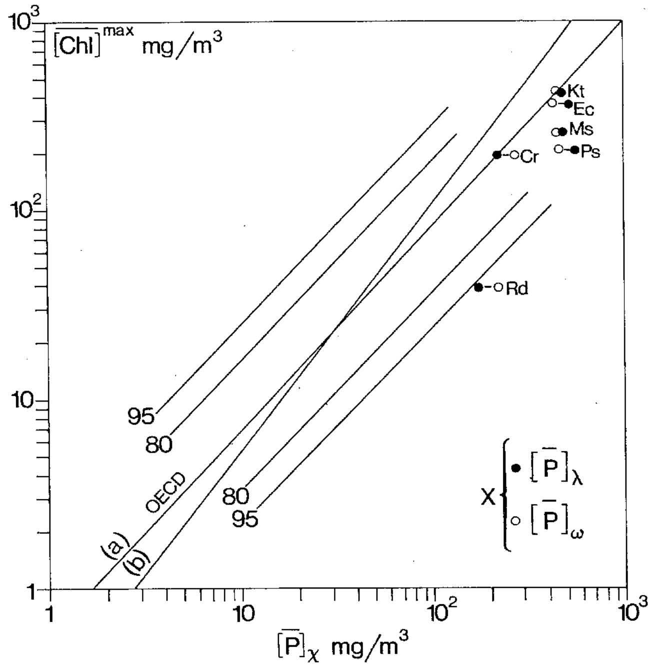

Figure VI 10. Peak chlorophyll a concentration in relation to average inlake and outflow total phosphorus concentration

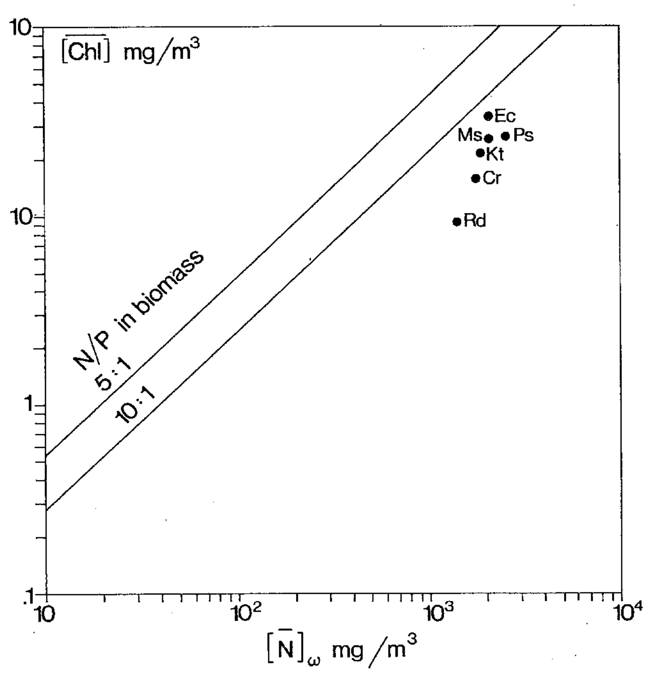

Figure VI 11 Several year mean chlorophyll a concentration in relation to mean outflow total nitrogen concentration

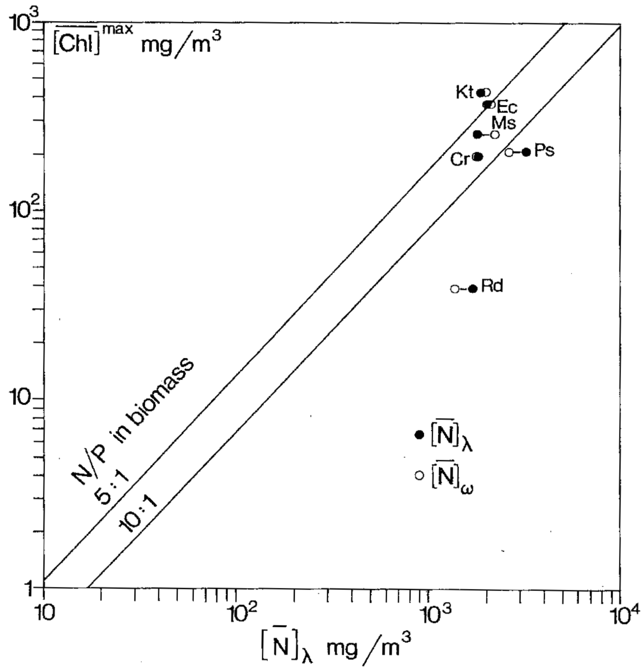

Figure VI 12. Peak chlorophyll a concentration in relation to mean inlake and outflow total nitrogen concentration concentrations (cf. Figure VI 11). All points remain below the transition area which corresponds to a biomass N/P ratio between 5 to 10. However, the positioning of the data points appears to be such as to make nitrogen a more likely candidate to affect production.

A clue to the question of whether nitrogen or phosphorus potentially could act as a factor controlling average biomass could be obtained from considering the N/P ratios. In the Fishing Lakes the N/P ratios are below 5 on average; in the lower lakes, instead, they are close to 7. This may indicate that - whilst the average nutritional level is high in all lakes - the Fishing Lakes might be nitrogen rather than phosphorus controlled, whereas the lower lakes might be phosphorus controlled. This argument, though not supported by the chlorophyll data to the level desired, finds support by that said on nutrient budgets. The Fishing Lakes act as a sink for nitrogen rather than for phosphorus, and *vice versa* the lower lakes act as phosphorus rather than nitrogen sinks (cf. above).

In contrast to average biomass levels, nutritional factors seem to be more important in controlling peak chlorophyll. Indeed, with the exception of Round Lake, peak chlorophyll values are within the uncertainty range of the corresponding OECD relationships (cf. Figures VI 10 and VI 12). Again, nitrogen seems to be of slightly greater importance than phosphorus, though also in this case, its relative role remains rather conjectural.

As already mentioned, the nutritional conditions of the Qu'Appelle Lakes are not the sole factors regulating the production levels of these lakes. Low light transmission (the 1% depth has been reported to be between 1 and 3 m) is one, the effect of which during the summer is compounded by the weak thermal stratification, leading to frequent redistribution of biomass by mixing, to levels below the compensation depth. For lack of information, this aspect cannot be explored.

Besides light, the hydrological regime of the Qu'Appelle system is probably the most important production regulating factor. As Table VI 9 shows, the probability for the yearly flushing rate to exceed 1 per year is at least 60% for all lakes, and for some of these lakes the probability for the flushing rate to exceed five per year is still significant. Such events do not necessarily occur in several distinct pulses, and the lakes may be washed out several times at once.

A satisfactory resolution of the effect of flushing on biomass is not possible at the basis of the data available. During years when flushing rates are high, the biomass found in downstream lakes undoubtedly results in part from carry-over, i.e. it is not produced *in loco*. Also, depending on flow conditions, mixing and water mass displacement, etc., the mineralization of biomass along the transport may be substantially affected. Such influences may not be easily traced by the statistical methodology pursued in the context of the present report. On the other hand, the available data would not permit analysis of these aspects with dynamic techniques.

Table VI 9. QU'APPELLE VALLEY LAKES

Distribution of Flushing Rates over a 7-year Period

	Vol. $10^6 m^3$	Average Yearly Flushing Rate ($1/\tau_w = \rho$)						
Pasqua	114.7	0.62	1.02	1.93	2.66	4.13	4.28	5.66
Echo	114.1	0.6	1.02	1.96	2.70	4.28	4.41	5.78
Mission	66.2	1.04	1.74	3.48	4.83	7.50	7.71	10.05
Katepwa	226.6	0.31	0.50	1.04	1.40	2.26	2.32	2.99
Crooked	123.5	0.61	1.18	2.25	2.92	4.66	5.33	5.94
Round	88.8	.86	1.73	3.32	4.39	7.14	8.15	8.54
Approx. Cumulative Probability % that - ρ >1/ year (line A) ρ >5/ years (line B)		87.5	75	62.5	50	37.5	25	12.5

The influence of flushing on the average production level may to some extent be elucidated by introducing a concentration-time-dose instead of concentration alone, i.e. a concentration corrected for the average water residence time, $[M].\overline{\tau_W}$. Admittedly, this argument may be questionable, yet, it is at least interesting that linear averages for chlorophyll plotted against this concentration-time-dose result in a considerably closer position to expected levels for both phosphorus and nitrogen (cf. Figures VI 13 and VI 14).

Variations on the same theme lead to similar conclusions. The coefficients of variability of chlorophyll, e.g., are positively correlated with the average residence time of the six lakes ($r = .89$, $P = .05$).

Partial and multiple correlation analyses including as independent variables $[P]_\omega$, $[N]_\omega$ and $\overline{\tau_W}$ or $\overline{\rho}$, further show for peak chlorophyll the closest relationship to average residence time or average flushing rates (positive for the former, negative for the latter; significant at the $P = .01$ level). Conversely, average chlorophyll is weakly correlated with residence time or flushing rates with opposite signs relative to the peak chlorophyll relationships. This may be a slight indication for the effect of carry-over as mentioned above.

The corresponding multiple correlation coefficients are significant at the $P = .03$ level. This would indicate that the variability in the three factors selected could explain 70 to 80% of the variability in average or peak chlorophyll. Unfortunately, the resulting multiple regression equations are of but limited applicability because of the negative term in nitrogen. This, as well as unrealistic

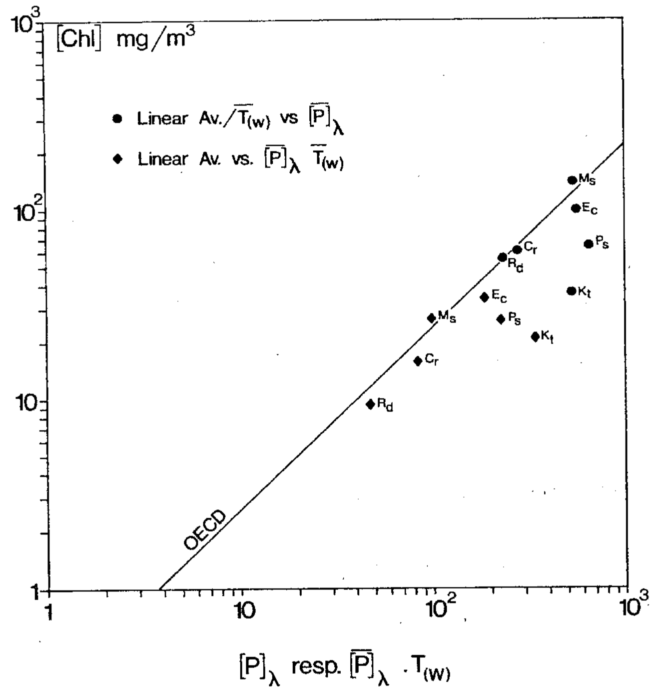

Figure VI 13. Mean annual chlorophyll a concentration in relation to mean phosphorus, respectively, phosphorus x τ_W

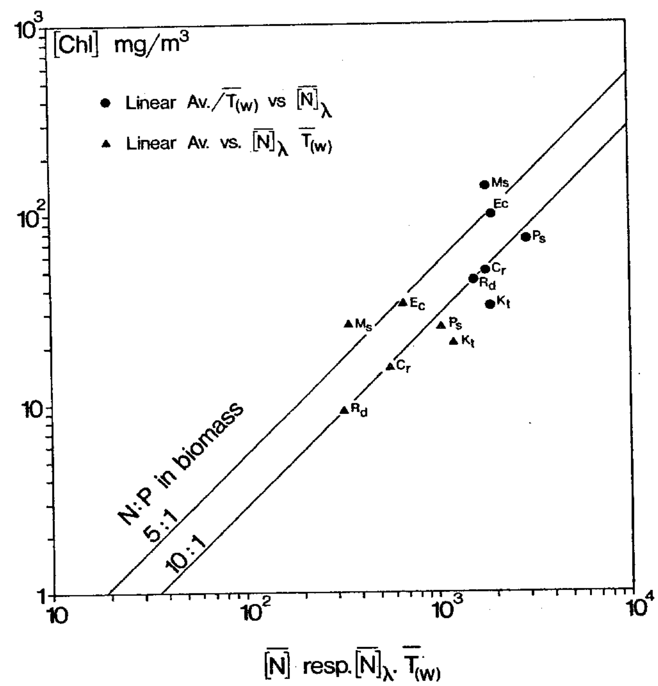

Figure VI 14. Mean annual chlorophyll a concentration in relation to mean annual nitrogen, respectively, nitrogen x τ_W

exponents, makes these equations unsuitable for predictive purposes, but at least serve as indicator for the fact that the interaction between nutritional and flushing conditions are the main factors determining the production levels of the Qu'Appelle system. The details of these interactions, however, are not fully understood as yet.

6.6.5 Prediction of Concentration and Biomass from Loadings. It is evident from the discussion above that both prediction of concentration as well as prediction of biomass from the information base available, remain more than questionable.

As pointed out previously, the uncertainties connected with constructing a nutrient budget are such as to make any reference to the OECD relationships inappropriate, i.e., it is impossible to predict inlake concentrations of the Qu'Appelle Lakes satisfactorily from such relationships and reported loading values. The inlake phosphorus concentrations for the Fishing Lakes are consistently higher than the input-predicted lake concentrations which have been used as argument for internal loadings. The flaw in this latter has been discussed above.

Similarly, the second aspect, i.e. inferences made from loading in regard to expected biomass, leads to unrealistic deductions. Apparently, chlorophyll values are in relatively good agreement with the OECD predictions, if the author-reported loading estimates are taken as reference. However, in considering what has been said above, the apparent agreement must be judged as fortuitous and therefore, also, any inference from it in regard to future behaviour of these systems (e.g. in terms of the effects of reduced loadings) remains questionable.

Therefore, although we have analysed these aspects very carefully, we desist from reporting the details of these analyses; application of the OECD procedures and results to the Qu'Appelle Lake system is possible only in a diagnostic but not predictive sense.

It is also questionable to state which of the four Fishing Lakes is the most productive. Taken at face value, the high average and peak chlorophyll values would point to Echo. However, a preliminary attempt to estimate relative metabolic rates places Pasqua on top, followed by Echo, Katepwa and Mission, in that order, i.e. unrelated to the order which one would derive from the chlorophyll values alone. This is due to the fact that biomass parameters (such as chlorophyll), as they are treated in this report, have a bearing to production dynamics only as long as they truly integrate *in loco* dynamics, whereas the Qu'Appelle system effects of biomass carry-over from the lake at the top of the chain to lakes below, have also to be accounted for.

6.7 Region VI Conclusions

Whatever the final judgement about the conflicting evidence presented here may be, there is no question that the lakes under discussion exhibit particular properties not easily comparable to the standard behavioural pattern derived from other lakes. The relatively low average chlorophyll levels, if juxtaposed to the average nutritional conditions, point to the fact that besides nutritional factors, factors other than these alone are of importance, flushing having been singled out as one. Under less perturbed conditions, biomass would most likely be controlled to a higher degree by nutritional factors (though not exclusively) as e.g. evidenced from the peak chlorophyll-phosphorus relationship being in better agreement with standard expectation. It must be underlined that simplistic application of the OECD results to the Qu'Appelle situation could lead to gross misjudgement in terms of management objectives. What their application to such systems can provide, however, is a diagnostic tool to clearly show that more substantial research is required to understand their behavioural characteristics. Despite the large amount of data collected, the analysis of this data base makes it clear that the coverage was insufficient, particularly in terms of time resolution in critical periods.

In more principal terms, the Qu'Appelle experience demonstrates the need for commensurating a data collection design with the hydrological characteristics of a system. Despite the limitations of the present data set, one can e.g. conclude that - had the collection period been limited to say one to two years out of the seven year study - then even less conclusive results would have emerged. This is a monitor to management that the amount of effort to be put into management-oriented research has to be proportionate to the characteristics of the object under study, i.e. its inherent variability, and that no simple design recipe is available for all cases.

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CHAPTER 7. BRITISH COLUMBIA REGION VII

Okanagan Lakes:	
- Wood Lake	(Wd)
- Kalamalka Lake	(Kl)
- Okanagan Lake	(Ok)
- Skaha Lake	(Sk)
- Osoyoos-N Lake	(Os)
Babine Lake	(Bb)
Kamloops Lake	(Km)
Kootenay Lake	(Kt)

7.1 British Columbia Region, Description of Location

Lakes of this region are located within the interior valleys and plateaux of the British Columbia mountains. These interior valleys and the leeward slopes of the mountains may be extremely dry and significantly warmer (due to rainshadow effect) than the surrounding peaks, where rainfall increases drastically with height. The vegetation of the region reflects these conditions and may range from the xerophytes of a steppe landscape to that of a high rain forest. Geographic conditions and some unusual events important to loading estimates and understanding trophic response are set out in the following descriptions of these lakes.

Okanagan Basin Lakes. While the Okanagan valley lies in a dry belt, there is a gradual change in climatic conditions from south to north. In the extreme southern portion of the valley, average rainfall is 27.5 cm yr^{-1} and in the extreme northern end of the valley it is 44 cm yr^{-1}. There are approximately 162 frost-free days in the south but only 108 in the north.

The valley, which is U-shaped with plateaux rising 1000 to 2500 m on both sides, contains five major lakes interconnected by river flow (cf. Figure VII 1). All the mainstem lakes are dimictic, most having bottom water temperatures above 4°C in midsummer, with mean hypolimnial temperatures consistently different among the lakes. In general, the lakes have an ice-cover in winter which commences about late December and lasts until the middle of March. Okanagan Lake, because of its volume (26 km^3) seldom has a complete ice-cover, although bays and shallow arms often do.

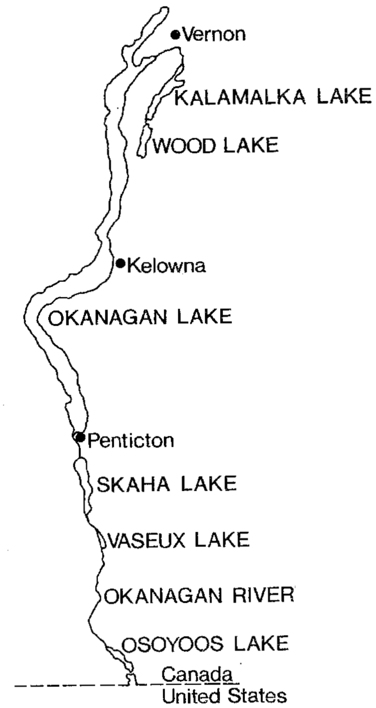

Figure VII 1. MAINSTEM LAKES OF THE OKANAGAN VALLEY, BRITISH COLUMBIA.

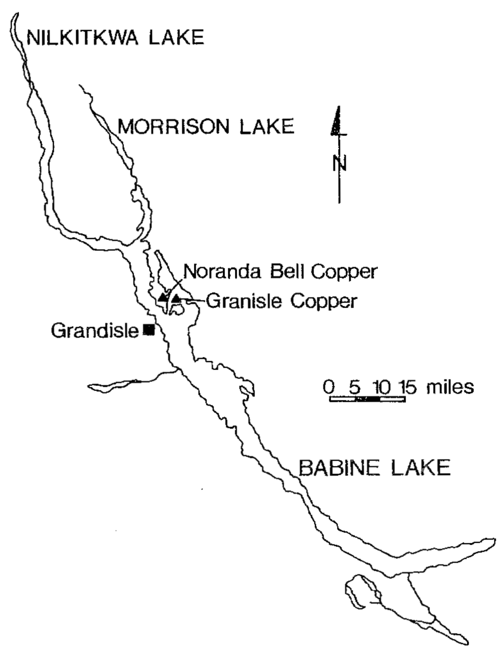

Figure VII 2. BABINE LAKE

There is considerable variation between annual mean water renewal times of the Okanagan Lakes. Those lakes uppermost in the chain have the longest theoretical filling times, while those receiving the outflow of Okanagan Lake (Skaha, Vaseux, Osoyoos) have much shorter filling times (the range for all being 52 to 0.35 years). The major portion of inflow water to the lakes comes during a 3-month period from April to June. Evaporation from Wood, Kalamalka and Okanagan Lakes is a major component of their water budgets (e.g. net evaporation in Okanagan Lakes is equivalent to 33% of the outflow). Except for major tributary streams, most small stream beds are dry from July to November, a result of upstream storage on regulated headwater lakes and land irrigation requirements. About 15% of the mean annual surface runoff to Okanagan Lake is used for irrigation. In addition, annual water replenishment in the lakes is irregular, owing to extreme year-to-year variation in runoff.

The major developments in the valley are associated with agriculture, forest industries and population growth. The soil of the benchlands surrounding the lakes is ideally suited for fruit crops; the higher, open forest lands are grass-covered and serve as valuable range for summer cattle grazing. The three major population centers in the basin, Vernon, Kelowna and Penticton, were organized around 1920 and from 1930 to 1950 regional sewage treatment plants commenced operation, discharging their effluents to the lakes. By 1972 the effects on Skaha Lake were so marked that the city of Penticton installed a tertiary treatment sewage plant and in 1977 the city of Vernon switched from release of effluent to surface waters to use for spray irrigation (Nordin, pers. commun). Land clearance, overgrazing and waste discharge have led to serious degradation of surface water quality and this in combination with the increased water demands by agriculture, industry and a growing population (approximately 200,000 by the 1976 census) have jeopardized an economy heavily dependent on water-based tourism and recreational activities.

Trophic Conditions. Wood, the headwater lake of the chain, stands out as the most eutrophic lake in the system, followed by Osoyoos. Oxygen saturation minima of 6%, 27% and 56% were observed in Wood, Osoyoos, and Skaha, respectively (whereas hypolimnetic oxygen values rarely fall below 80% saturation in Kalamalka or Okanagan). Deterioration of spawning habitats has seriously stressed the salmonoid fish populations in Wood Lake.

Assessment of the trophic condition of Skaha is difficult. Chlorophyll *a* and nutrient levels point to eutrophic conditions, while other criteria (periphyton, Secchi) do not. The short water renewal rate has undoubtedly prevented the rapid deterioration of surface water quality, especially along the more oligotrophic eastern shoreline. Decline of salmonoids over the past 20 years, together with the increased average density of benthos, are signs of change commensurate with sewage discharge, beginning in 1947.

Towards the other end of the scale, Kalamalka and Okanagan have retained basic oligotrophic characteristics despite significant watershed development. Local regions of Okanagan exhibit eutrophic features (Vernon arm and Kelowna foreshore) but these have not appreciably affected the main body of the lake. O_2 values in the hypolimnion have not changed appreciably from the 1935 condition and salmonoids still predominate.

Secchi transparencies vary from 2.6 m in Wood to 9.3 m in Kalamalka (cf. Table VII 1). (Humic substances are not important here).

The following descriptions of phytoplankton refer to the time period when nutrient levels throughout the the basin were highest (i.e. about 1970).

Phytoplankton. Wood Lake has the highest phytoplankton density with an average of 7900 cells·ml^{-1} and average freshweight of 5000 mg.m^{-3}. Previously, blue-green algae dominated at all depths throughout the growing season, but more recently diatoms have dominated the spring flora. *Oscillatoria* spp. and *Aphanizomenon flos-aquae* are common in the summer flora.

Next to Wood, Osoyoos exhibits the highest stock of phytoplankton (yet has the lowest mean annual Secchi reading). Average density is 5470 cells·ml^{-1}. Spring and fall are characterized by pulses of diatoms, and summer by blue-green blooms accompanied by cryptomonads. Common species include *Asterionella formosa, Fragilaria crotonensis, Cyclotella comta, Melosira italica* and *Cryptomonas ovata.*

Phytoplankton density in Skaha Lake averages 3700 cells·ml^{-1}. Diatoms dominate, but there is a blue-green algal peak in late August. The most common diatoms are *Asterionella formosa, Fragilaria crotonensis,* and *Cyclotella comta* and blue-green species include *Aphanizomenon flos-aquae, Aphanothece microscopica,* and *Anabaena circinalis.*

Okanagan Lake phytoplankton densities are low averaging 1500 cells·ml^{-1}. This lake is also dominated by diatoms, but blue-greens are common in midsummer and early fall. Species most often found are *Melosira italica, Cyclotella ocellata, Fragilaria crotonensis, Aphanothece nidulans, Anabaena flos-aquae, Lyngbya limnetica* and *Cryptomonas ovata.*

Table VII 1. Selected chemical, physical and biological parameters: Okanagan Lakes

Lake		Wood	Kalamalka	Okanagan	Skaha	Osoyoos-N
A. Phosphorus (spring overturn)	mg/m^3					
Total-P 1971 [1]		92 85	9.7 8	11.8	20.1	19.5
Ortho-P 1971 [1]		27 65	4.5 1	5.7	5.0	4.4
Total-P 1974-1980 [2]		76 ± 13	6.0 ± 1.2	-	(16)	-
Ortho-P 1974-1980 [2]		41 ± 26	3.3 ± 1.6	-	-	-
Total/Dissolved 1977-1978 *)				7.4/2.8	14.5/3.7	22.3/12.3
B. Nitrogen (spring overturn)	mg/m^3					
Total-N 1971 [1]		526 580	241 250	215	215	248
NO_3-N 1971 [1]		25 195	23 60	10	< 10	13
Total-N 1974-1980 [2]		758 ± 165	251 ± 33	-	-	-
NO_3-N 1974-1980 [2]		173 ± 112	47 ± 10	-	-	-
C. TN/TP		6 (10) 7	25 (42) 31	18	10	13
NO_3-N/ dissolved-P		1 (4.2) 3	5 (14) 60	< 2	< 2	3
D. Chlorophyll	mg/m^3					
a) original values [3]		50	2.5	5	31	23
b) Secchi revised estimates		12	1.8	2.2	5	7
c) 1975-79 summer averages [2]		9.8 ± 6.1	1.3	-	(4.2)	-
d) 1975-79 maximum observed [2]		31	3.5, 6.4†	-	-	-
e) seasonal means 1976-79*)				1.4	3.4	4.9
E. Secchi Disc Averages 1971 [3]	m	2.6	9.3	7.5	4.7	3.1
1976-78 *)				7.7	4.3	3.8
F. O_2-Depletion rates 1971 [1]	mg/m^3.day (g/m^3.month)	62 (1.85)	9 (.27)	15 (.45)	34 (1.02)	35 (1.05)

[1]) Report Williams 1972
[2]) Nordin, pers. comm.
[3]) From Stockner & Northcote 1974
— underlined figures from Gray & Jasper, 1981
*) Truscott and Kelso 1979
† 1979, 1980, respectively (Okanagan Basin Implementation Board, 1981)

Kalamalka Lake phytoplankton are sparse (< 700 cells ml $^{-1}$). As in Skaha and Okanagan, diatoms dominate, but phytoflagellates peak in early summer and fall. The most abundant species here are *Cyclotella ocellata*, *C. kutzingiana*, *Melosira italica*, *Cryptomonas ovata*, *Chromulina spp.* and *Dinobryon sertularia*.

Attached Algae and Rooted Aquatic Vegetation. Over the past decade, a marked increase has occurred in the abundance of rooted aquatic plants, especially Eurasian milfoil and attached microalgae (periphyton) along the shorelines of most of the main valley lakes, where high biological production is caused by the discharge of nutrients. Some areas currently exhibiting extensive weed beds are: Vernon Arm and Kelowna shoreline south of the floating bridge on Okanagan Lake; south end of Wood, east shore of Skaha, Vaseux, and along the west shore and the north and middle basins of Osoyoos Lake. In many of the Okanagan Lakes luxuriant periphyton growths occur in the immediate vicinity of known man-made nutrient discharges, which could jeopardize the reproductive success of beach-spawning kokanee or other sport fish.

Fish. Catches of all fish were lowest in Wood, followed by Kalamalka; those from Okanagan were intermediate with much higher catches being taken from Skaha and by far the most from Vaseux. The contribution of salmonoids to the total catch was lowest in Wood (12%), followed by Vaseux (19%), Skaha (about 35%), and Osoyoos (about 40%) while that in Okanagan and Kalamalka Lakes in nearly all cases was over 40%, sometimes running as high as 70%. Of the 26 different species of fish taken from the mainstem lakes in 1971 (Northcote et al. 1972), only 10 of these occurred in Wood Lake, whereas 14 were recorded from Kalamalka, 15 from Okanagan, Skaha and Vaseux and 20 from Osoyoos.

Babine Lake, (cf. Figure 2). The watershed surrounding Babine Lake (about 10,000 km^2) forms one of the major drainage basins of the Skeena River system, lying some 600 km north of Vancouver.

Precipitation, about 60 cm $year^{-1}$, is distributed fairly evenly throughout the year. Snowfall (2.5 to 3 m) occurs between October and May; the lake is usually ice-covered from December to early May. Discharge (annual mean 47 m^3/sec) reaches peak flows of 110 m^3/sec in June.

In the vicinity of the lake, metamorphosed rocks of the Upper Palaeozoic Group occur. To the north, Upper Triassic to Tertiary formations are associated with volcanicity, and plutonism; scattered economic copper deposits are found. Pleistocene glacial and post-glacial tills and drift cover much of the area.

Development in the Babine watershed has been relatively recent. Clear-cut logging, initiated in the 1950s, resulted by the late 1960s in several openings in excess of 202 hectares of the extensive surrounding forests, primarily composed of white spruce and lodgepole pine. The area remained thinly populated until the 1960s, when the townsite of Granisle was developed, currently accommodating about 2,000 people; about 200 to 300 live on the lake in widely scattered locations. During summer the population increases considerably with an influx of tourists, hunters and seasonal workers. Since 1960 extensive open-pit operations on Copper Island and on Newman Peninsula have been developed. Seepage of mine wastes into Babine Lake could be a potential water quality problem, but no serious effects have as yet been observed on lake flora or fauna.

Trophic Condition. Babine Lake is classed dystrophic on the basis of humic materials, and mixotrophic in terms of annual primary production (33 $g/m^2.y$). Disparity in regional production (higher local production, e.g. in the south basin of the main lake) is thought to be related to lake surface inflow disparity, upwelling and attendant entrainment of hypolimnetic waters common in the south, coupled with a higher surface nutrient inflow.

The phytoplankton of Babine Lake is dominated by diatoms. The spring bloom consists of *Rhizosolenia longiseta*, and *Cyclotella stelligera* followed by *Melosira italica* and *Asterionella formosa* which exhibit peaks of lesser magnitude in June. *Ankistrodemus*, *Cryptomonas and Chromulina* with other flagellated Chrysophytes develop following the decline of the diatom spring bloom. The fall bloom, when occurring, is mainly *Tabellaria fenestrata*, with *Fragilaria spp.* present. *Oscillatoria* and *Chroococcus* are the only blue-greens present in significant numbers, with *Anabaena* attaining lesser prominence in the latter part of the season.

The Babine watershed supports large fisheries of considerable recreational and commercial importance. The major species present include rainbow trout (*Salmo gairdneri*), kokanee (landlocked sockeye, *Onchorhynchus* *nerka*), lake trout (char, *Salvelinus namaycush*) and whitefish (*Coregonus clupeaformis*). The primary commercial species in the Babine system are sockeye salmon with annual catches amounting to some 800,000 to 1,000,000 fish.

Kamloops Lake. Kamloops Lake is a long (25 km), narrow (mean width 2.1 km), deep (mean depth 71 m) lake (cf. Figure VII 3) situated in a dry valley of glacial origin on the Thompson Plateau in south central British Columbia (Ward, 1964).

The physical limnology of Kamloops Lake is dominated by the Thompson River, which has a mean annual inflow rate of 720 m^3 sec^{-1}. Over 60% of the discharge comes in the early summer freshet period, with peak flows near 3400 m^3 sec^{-1} in June and minimum flows of 120 m^3 sec^{-1} in February. As a result of the large and variable discharge, bulk residence time during the year is highly variable (20 - 340 days with a mean of 60 days) (St. John et al., 1976).

Classical stratification does not develop until late summer. Throughout June to October, the inflow water remains cooler than the surface lake water and thus interflows through the epilimnion at depths of 10 - 30 m. Only in late summer, with declining river flows and deep convective mixing of the surface waters, does the lake establish a classical two-layer thermal structure. Direct stratification slowly breaks down through November and December until complete convective overturn results. During this period, inflowing river water either sinks to the bottom, or is confined to the eastern end of the lake.

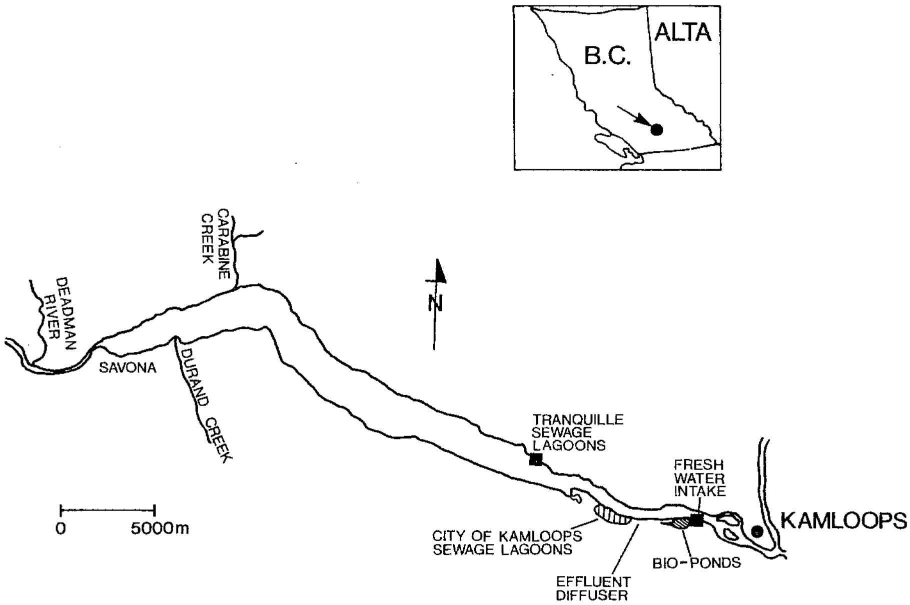

FIGURE VII 3. KAMLOOPS LAKE AND MAJOR GEOGRAPHIC, URBAN, AND INDUSTRIAL FEATURES OF THE AREA (from St. John et al, 1976)

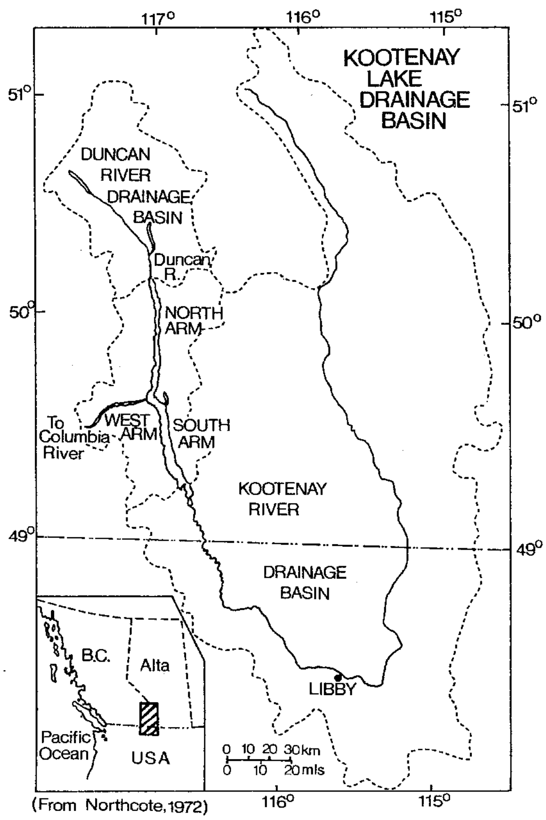

Fig.VII 4. Kootenay Lake Drainage Basin

Cultural eutrophication of Kamloops Lake is expected to be slow. The high flushing rates generally prevent long-term nutrient accumulation in the lake and most of the wastewater nutrients that do accumulate in the winter are flushed from the lake prior to the summer growing period. Point source nutrients are insignificant in the summer epilimnion in comparison to natural loadings and very large increases in pollution inputs will be required to elicit even a small change in nutrient concentrations. In addition, the summer growth rates of phytoplankton are largely subdued by short residence time and high turbidity.

Confirmation that Kamloops Lake has undergone little, if any, eutrophication is found in microbiological and chemical criteria. Its oligotrophic status is apparent in the species composition and low activity of the phytoplankton. Mean annual chlorophyll a and C-14 primary production values are 0.85 mg m^{-3} and 32 g C m^{-2}, respectively. Highest chlorophyll concentrations were found in September but there may also be a near surface peak in June or July. Diatoms (*Fragilaria, Tabellaria*) dominate the phytoplankton (> 50% by carbon) except during the fall maxima when Cryptophytes (*Chroomonas* and *Cryptomonas*) have been found to be the principal genera. Blue-green algae only appear as minor constituents in winter and consist of non-bloom-forming genera. Total bacteria numbers are also low ($3\text{-}7 \times 10^5$ cells$\cdot ml^{-1}$) although on an areal basis, heterotrophic biomass was found to be five times the phytoplankton biomass. Also, oxygen levels are generally near saturation, with lowest values (8.3 mg l^{-1}) at the bottom in October.

Kootenay Lake. Kootenay Lake (532 m) occupies about half the length of a deep, steep-sided valley extending in a northerly direction over 225 km into British Columbia from 49° to at least 51°N latitude (cf. Figure VII 4). It is bounded on the west by the Selkirk Mountains rising 3000 m and on the east by the Purcell Mountains, reaching nearly 3500 m. Its two major influents, Kootenay River entering from the south and Duncan River from the north, also follow this valley. The Kootenay River drainage system covers about 80% of the total drainage basin (45,610 km^2) and contains 90% of the total population (ca. 70,000). The lake discharges near its midpoint through a transverse valley containing the west arm of the lake to its outlet, the lower Kootenay River, which joins the Columbia River. The portions of the main lake north and south of the west arm are commonly referred to as the north and south "arms".

Kootenay Lake is thermally stratified from May to November to about 40 m and has a complex internal circulation system dependent on wind and density differences of its major tributaries. Marked spatial and seasonal changes in transparency are due to high turbidity of the Kootenay River (Daley et al., 1981). The theoretical water renewal time for the main lake is estimated to be some 570 days but that of the west arm about 5.5 days.

The Columbia River system is presently regulated by a number of dams for lake level control and hydroelectric power generation; these include the Duncan Dam and the Libby Dam. These two major impoundments have effected the biology of the lake, yet the extent of it has only undergone initial assessment (cf. Chapter 8 for further discussion of events and consequences for Kootenay Lake). The data used in the present elaboration extend over a period between the Duncan Dam closure (1967) and the Libby

Dam closure (1972). Besides the impoundment alterations, major changes in nutrient load (cf. Section 7.1.1) have taken place which complicates data interpretation.

The first indications of increased algal abundance (planktonic and epibenthic) as well as aquatic weed growth in Kootenay Lake, were noted in the mid- to late 1950s when anglers began reporting slime on their lines, the clarity of the water decreased, the high watermark of the lake became much more distinct, and formerly clear shoals and sand bars of the west arm became covered with rooted aquatic plants. Extensive algal blooms occurred on Kootenay Lake in the summers of 1958 and 1960, imparting offensive odour and taste to the water and to the fish. Major algal blooms occurred in 1965, 1967 and 1973, with minor or localized blooms in other years. No algal blooms occurred in 1976 and algal biomass accumulations have been much lower since the phosphorus load reductions began in 1969.

Primary production seems to have peaked around 350 g C/m^2.y; more consistent measurements over the last years showed a value of 190 g C/m^2.y in 1973 with a further drop to 140 — 170 g C/m^2.y in subsequent years.

Evidence for a two to threefold increase in standing crop of macrozooplankton between 1949 and 1964 in Kootenay Lake has been documented. All species present in 1949 were found in 1964, but *Daphnia galeata* was less numerous whereas *Diaphanosoma leuchtenbergianum* had increased sharply. Since the introduction of *Mysis relicta* in 1949, there has been a population explosion of this omnivorous species with standing stocks to over 1000 individuals per m^2. The effect of this introduction of *Mysis* on the partitioning of carbon between phytoplankton, zooplankton and kokanee has not been ascertained.

Kootenay Lake contains some 20 species of fish, including the six native salmonoid species: kokanee or landlocked sockeye salmon (*Onchorhynchus nerka*), Dolly Varden (*Salvelinus malma*), rainbow trout (*Salmo gairdneri*), mountain whitefish (*Prospoium williamsoni*), Yellowstone Cutthroat trout (*Salmo clarki lewisi*), and pygmy whitefish (*Prosopium coulteri*), but several other species have been introduced. The fishery, largely for the salmonoids, whitefish and lingcod (*Lota lota*) is of economic importance.

7.1.1 Loading Estimates. Because of the inequality of available data for the lakes reviewed here, it is not possible to lump all the information without appropriate comment. Extensive studies of Kamloops Lake and Babine Lake have produced reliable data, excepting that for the phosphorus loading of Babine Lake, which is probably underestimated. Also, Kootenay Lake has been extensively studied, yet the available data refer in part to different years. In addition to this, considerable change in limnological conditions over the study period makes synthetic treatment somewhat difficult. In regard to the Okanagan Lakes, original data required some reinterpretation, therefore, aspects of these lakes are reviewed separately from other lakes with appropriate qualification.

The Okanagan Lakes were extensively studied in early 1970, with a major monitoring programme in 1971. Nevertheless, the data bank with respect to OECD purposes is somewhat weak, particularly in regard to loading estimates and chlorophyll measurements. Indirect phosphorus loading estimates from population density, runoff and interconnecting channels have been made by Patalas; direct loading measurements from flow data and concentrations have been made by B.C. Laboratories (cf. Table VII 2). The original Patalas figures (cf. Patalas 1973) are probably overestimates due to overestimating the phosphorus load originating from population. It is likely that only about 1/2 to 2/3 of the assumed basic load (1700 g per capita per year) from diffuse sources reaches waterways. Recently revised figures provided by Patalas are much closer to those derived from direct measurements.

D. J. Williams (1972) estimated the loadings to Kalamalka and Wood Lakes and pointed out the potentially high relative contribution of groundwater inputs to Wood Lake. In a more recent study (Water Investigations Branch, 1974) however, the groundwater inflows were directly observed to be very low. The evidence that Williams (1972) used for estimating groundwater flows was based on the abnormally low summer heat income of Wood Lake in 1971 (Blanton, 1973). In subsequent years the heat income of the lake has been normal (Water Investigations Branch, 1974) (C. Gray, pers. comm.).

In regard to Babine Lake, the phosphorus load (mostly from diffused sources) has been estimated at 24.2 t/year, about 19% coming from salmon carcasses. This latter is a unique feature among the lakes reviewed in this report (but is known from other coastal British Columbia, Alaskan and Siberian lakes). However, there are strong indications that the load has been underestimated by a factor of 2 to 3. At the basis of the above loading figure, the average basin export coefficient would be only 1.9 $kg/km^2.y$ which is very low. The lake is clearly phosphorus-limited (average Total-P: 6 mg/m^3; dissolved-P < 3; average Total N: 275 mg/m^3; $N(NO_2 + NO_3) \approx 81$ mg/m^3)).

In Kamloops Lake, the total and dissolved phosphorus loads to the lake are very high (17 and 3.5 $g \cdot m^{-2} \cdot yr^{-1}$, respectively). However, the

Table VII 2. Phosphorus Loading Estimates: Okanagan Lakes.

Lake		Wood	Kalamalka	Okanagan	Skaha	Vaseux	Osoyoos-N
A. Patalas[1]	kg/y	1506 (4650)	2350 (8300)	84968 (134300)	21982 (44300)	-	17019 (26700)
B. B.C.[2]	kg/y	1400	2600	87000	26000	(11700)	18000
C. Q x 10^6 m^3/y		10.1 12.7*	21.3	439	475	529	590
Average inflow conc.	mg/m^3						
From A.		149 119	110	194	46		28
From B.		138 110	122	198	55		30
D. Filling Time T(w)	years	20 16	71 42	60	1.2		.35
E. $([P]_i/(1 + \sqrt{T(w)}))$	mg/m^3						
From A.		27 24	12	22	22		18
From B.		25 22	13	23	26		19
F. Lake spring conc. April 1971	mg/m^3	92	9.7	11.8	20.1		19.5
Means TP 1975-79		76	5.5	-	-	-	-

1) Indirect estimates from land runoff and population density (reference year 1969) - revised figures. Original figures in parentheses.

2) From concentration and flow measurements.

— recent figures from C. Gray (pers. commun.); *after Hiram Walker Cooling water discharge (1971).

Table VII 3. Phosphorus Loading Estimates: Wood and Kalamalka Lakes
by D.J. Williams, 1972 kg/y

	Wood		Kalamalka
a) Surface input	260	...	2130
b) Groundwater	870	23[a]	
c) Sewage & septic tanks	1140	...	990 (excl. septic tank conc.)
Total	2270	1423[b]	3120
$[P]_i$ (average inflow conc.)	225	141[b]	146
$[P]_i/1+\sqrt{\tau_{(w)}}$ (theoretical lake conc.)	41	27[b]	15.5

[a] Water Investigations Branch, 1974

[b] Calculated from a

presence of up to 80% biologically inert apatite in the particulate phosphorus pool and the confinement of river water to the lower epilimnion in summer, seriously overestimate the available phosphorus load and the effective bulk residence time, respectively. When corrected for these effects, Kamloops Lake falls well within the oligotrophic category, based on the usual plot of phosphorus load versus the mean depth: flushing time ratio.

Dramatic changes in the phosphorus load to Kootenay Lake have occurred over the past 30 years. An estimate for 1950 of less than 250 t/y (0.6 g/m^2.y) increased tenfold to more than 2500 t/y (6 g/m^2.y) by about 1966. After 1972, phosphorus loadings dropped again to less than 1000 t/y and levelled off close to the earlier loading level of 250 to 300 t/y.

In terms of lake concentration, phosphorus had increased sharply in all three regions of Kootenay Lake from less than 2 mg/m^3 (dissolved-phosphate) in 1949 to over 100 mg/m^3 by 1968. Highest values have consistently occurred in the south arm, with concentrations in some years showing over a hundredfold increase from earlier levels. The tremendous increase was due primarily to discharge operations of the Cominco fertilizer plant (which opened in 1953). Subsequent abatement measures by the plant in 1969 brought on the decline. In comparison, the phosphorus load from domestic sources, other industries and agriculture (less than 8% of the basin has soil suitable for agriculture) has been of minor importance, although their absolute contributions have undoubtedly increased over time.

Nitrogen loading, on the other hand, remained relatively constant with only minor changes during the period when fertilizer plant effluent contained ammonium phosphate. No consistent, long-term changes in nitrogen concentrations in Kootenay Lake have been observed. Even maximum values have not exceeded 300 mg N/m^3 and most overturn values during the time of peak phosphorus concentrations (1972-78) were about 200 mg/m^3.

7.2 Trophic Response Nutrient Relationships

7.2.1 Chlorophyll-Phosphorus Relationship (Okanagan Lakes).

Interpretation of this relationship (cf. Figure VII 5) encounters some difficulty if the original author-provided chlorophyll data are used (cf. Table VII 1). In relation to the OECD findings, chlorophyll estimates appear to be exceptionally high, the following arguments can be brought forward to claim these values are overestimates and not representative for lake conditions in terms of yearly averages which, however, does not exclude that local and/or periodic chlorophyll build-ups may occur.

a) Skaha and Osoyoos lie distinctly above the 95% confidence limit and Wood on the upper boundary of the 80% limit of the OECD relationship (cf. Figure VII 5).

b) Plotted against Secchi disc, 4 out of 5 lakes (Wood, Okanagan, Skaha, Osoyoos) are near or beyond the accepted upper statistical boundaries (cf. Figure VII 6).

c) Conversely, Secchi disc readings match well with what one would expect from average phosphorus concentrations (cf. Figure VII 7).

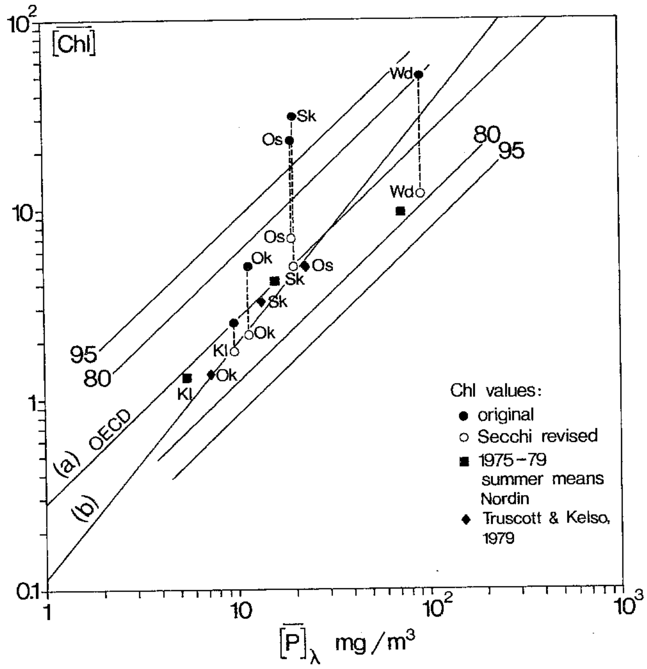

Figure VII 5. Annual mean chlorophyll a in relation to mean total phosphorus concentration: Okanagan Lakes

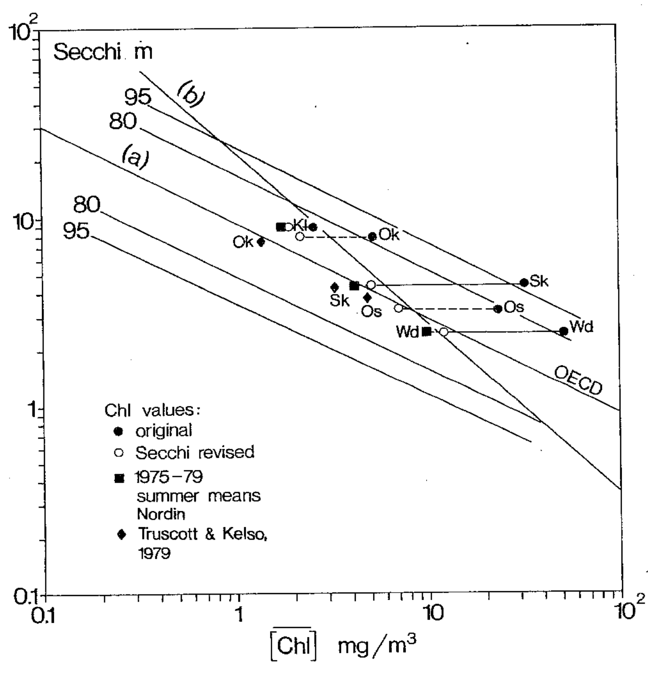

Figure VII 6. Secchi transparency in relation to annual mean chlorophyll a concentration: Okanagan Lakes

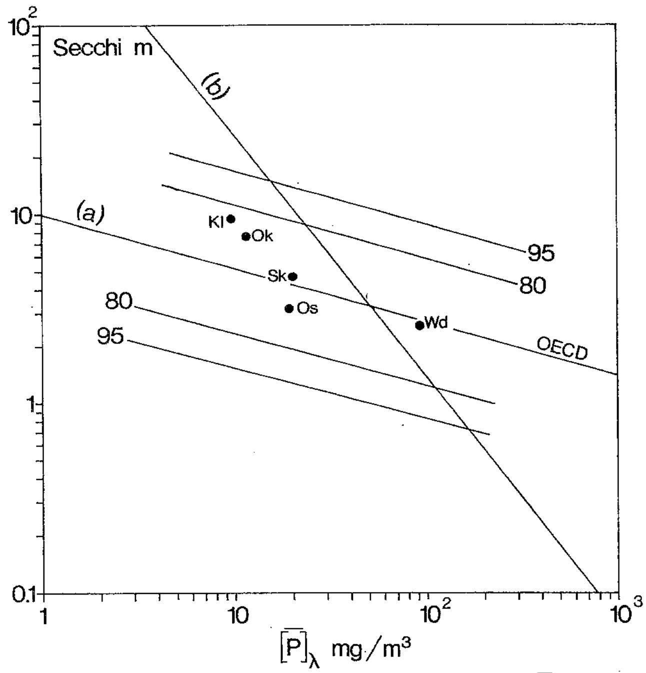

Figure VII 7. Secchi transparency in relation to mean total phosphorus concentration: Okanagan Lakes

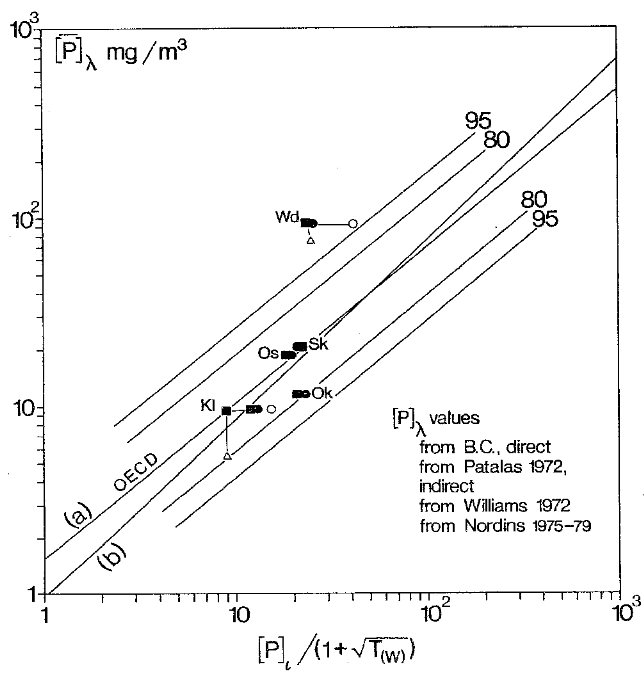

Figure VII 8. Mean total phosphorus concentration in relation to flushing corrected inflow total phosphorus concentration: Okanagan Lakes

d) Rigorous scrutinizing of all data available indicate that both Secchi disc values and average phosphorus concentrations provided for the Okanagan Lakes do not appear to be objectionable.

Combining these several premises, we attempted a correction for chlorophyll using the OECD chlorophyll-Secchi disc relationship as reference. Secchi corrected chlorophyll values have been estimated from the mean between the two regression lines a) and b) to avoid an undue overcorrection, and these 'revised estimates' (cf. Figure VII 6) are reported in Table VII 1.

We recognized that a Secchi disc related correction procedure may be questionable. The phosphorus-Secchi disc relationship is the least reliable of all OECD relationships established and if used in isolation, could lead to substantial errors in judgement. The same also applies, in principle, to the chlorophyll-Secchi disc relationship, with the exception that, in situations of high biomass, i.e. when neither mineral turbidity nor humic substances govern transparency essentially, Secchi disc visibility is closely related to biomass. This is the principle on which many reconstructions of the history of trophic changes in lakes have been based.

Concerning the Okanagan situation, lakes such as Wood, Skaha and Osoyoos judged on the basis of the original chlorophyll data, and the boundaries derived from the overall OECD information, would be close, or in the category of hypertrophic lakes. In this case, neither the phosphorus nor the Secchi disc transparency would coincide with such a classification. What makes our correction justified, however, are a few more recently measured summer values provided by Nordin including the Truscott & Kelso (1979) values which are in close agreement with our revised estimates. (Cf. Table VII 1). In considering all this evidence, we reject the validity of the original chlorophyll values, at least for Wood, Skaha and Osoyoos. However, in trying to remain objective, we report both the author-provided chlorophyll values as well as our revised estimates to allow the reader to make his own judgement.

7.2.2 Loading-Phosphorus Relationship (Okanagan Lakes). In contrast to the discussion on the chlorophyll-phosphorus relationship, the relationship in question here matches satisfactorily with the overall OECD information (cf. Figure VII 8). Wood Lake has to be excepted, insofar as its real load is difficult to estimate. Although earlier estimates of phosphorus loading suggested that up to 60% of the external load entered the lake with groundwater (Williams, 1972), more recent observations of groundwater flow have not shown any significant flow relative to surface runoff (Water Investigations Branch, 1974).

The relative contribution to spring concentrations of phosphorus from sediment return during the summer anaerobiosis has been found to be no higher than the return observed during aerobic periods (Gray and Jasper, 1981) so that high concentrations cannot be explained by anaerobiosis alone (C.B. Gray, pers. comm.).

7.2.3 Chlorophyll-Loading Relationship (Okanagan Lakes). The problem discussed previously regarding chlorophyll is also reflected in the relationship in question here (cf. Figure VII 9). As for the chlorophyll-phosphorus relationship (cf. Figure VII 5), Wood, Skaha, and Osoyoos lie considerably above the 95% confidence limit of the corresponding OECD

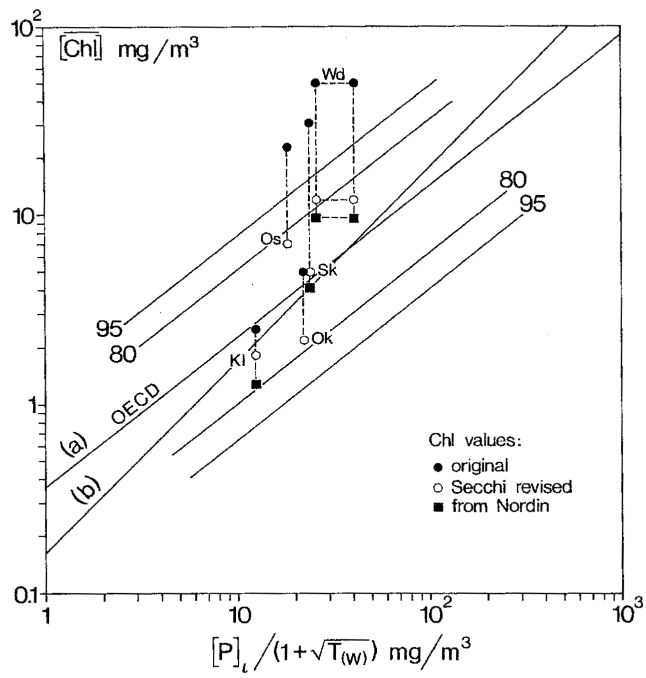

Figure VII 9. Annual mean chlorophyll a concentration in relation to flushing corrected inflow total phosphorus concentration

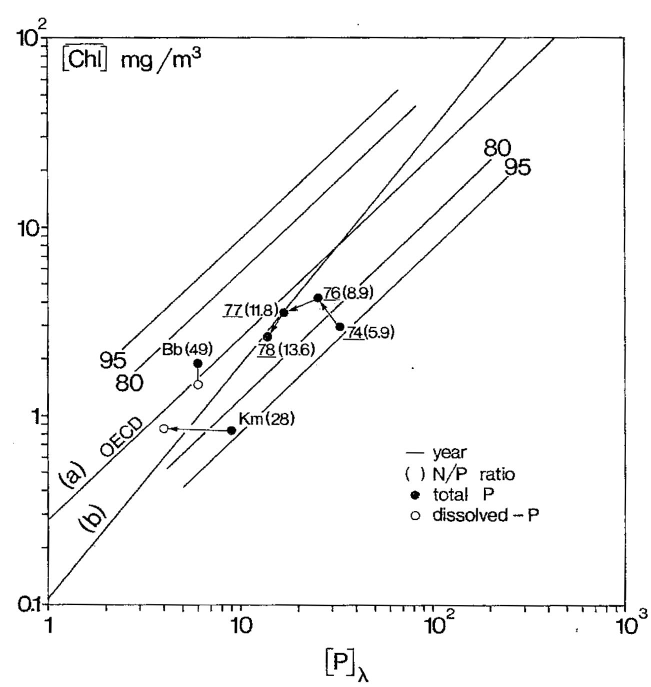

Figure VII 10 Annual mean chlorophyll a concentration in relation to mean phosphorus concentration (dissolved and total): Kootenay, Kamloops and Babine relationship. Also in agreement with what is observed for the former case, considerable improvement is obtained for the three lakes if Secchi disc corrected values are plotted against loading. The corrected value for Wood Lake may appear as overcompensated, yet Nordin's value is even lower. However, in considering the nutritional level and the low N/P ratio, it may well be that Wood Lake is not phosphorus controlled at all, and may possibly be nitrogen controlled. The revised values of Kalamalka and Okanagan remain inconclusive, if not less acceptable.

7.2.4 Chlorophyll-Phosphorus Relationship, Babine Lake, Kamloops Lake, Kootenay Lake. (Cf. Figure VII 10). The three lakes in question fit reasonably well with the overall OECD relationships. For Babine Lake, average chlorophyll values from 1973 and average phosphorus values from 1974 have been used because of the more extended data base for the respective years and parameters. Kamloops Lake, if total phosphorus is used as reference, stays outside the 80% confidence interval. This may be due to the high apatite load of this lake, which, biologically, is of but limited availability. Average chlorophyll plotted against dissolved-P, on the other hand, is close to OECD expectations. However, it may also be possible that the low chlorophyll value is, at least in part, dependant on the relatively low transparency. Kamloops Lake in both plots (Secchi vs Chl; Secchi vs P; cf. Figure VII 15, VII 16) shows up at the lower side.

In regard to Kootenay Lake, the evolutionary pattern of chlorophyll response to phosphorus for the period 1974 to 1978 shows a definite relationship to the N/P ratio (cf. Figure VII 10). In 1972, this ratio was as low as 3.9 but as a consequence of the phosphorus load increasing N/P ratio during the following years, the chlorophyll values moved closer to the OECD line with the interesting phenomenon that averages for 1976/77 stayed higher than the 1974 average. However, it is not possible to prove statistically that this apparent succession is real, i.e. that the 1976 average exceeds the 1974 average. In view of the unsettled question regarding the role of the N/P ratio in co-determining the average chlorophyll values, the phenomenology observed is at least noteworthy. According to Smith and Shapiro (in press) it seems conceivable that in certain cases the average chlorophyll level would temporarily increase prior to decreasing, as a consequence of phosphorus load reduction.

Peak chlorophyll values for the different years from Kootenay Lake, on the other hand, are practically on the OECD line (cf. Figure VII 11), as are the values for Babine Lake and Kamloops Lake, if for the latter one gives more weight to the average dissolved P as reference.

7.2.5 <u>Loading-Phosphorus Relationship (Babine, Kamloops, Kootenay)</u>. Lake phosphorus (spring overturn) concentrations have been plotted against flushing corrected inflow concentrations measured for the same year (cf. Figure VII 12). It is important to keep this in mind in interpreting the particular behaviour of Kootenay Lake. As has been pointed out, the phosphorus load to this lake has undergone drastic reduction, wherefore there is no equilibrium between yearly load and the corresponding spring overturn concentrations. This shows up clearly in the scattering of the data points relative to the OECD standard regression. Even if phosphorus concentration is plotted against the loading measured for the year reduction, increased subsequently to values up to 16 (cf. Table VII 4).
In 1974 chlorophyll values stayed low with an N/P ratio of 5.9, but with an

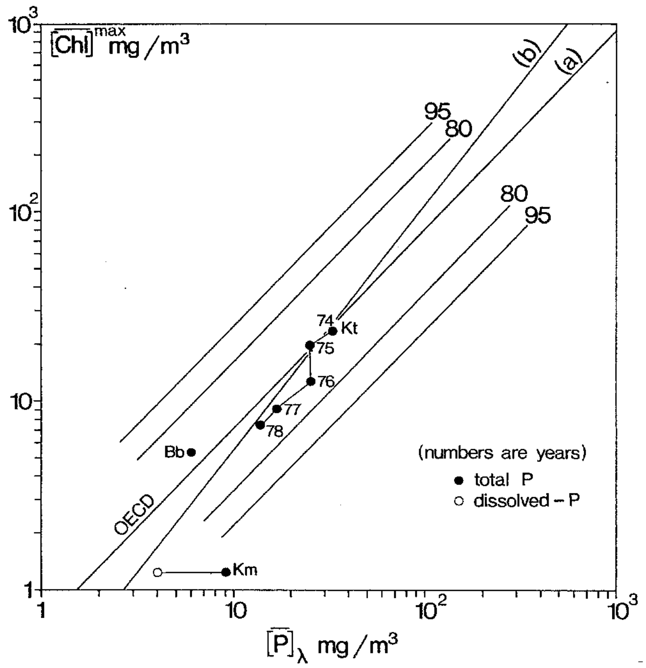

Figure VII 11. Peak chlorophyll a concentration in relation to mean total phosphorus: Kootenay, Kamloops and Babine

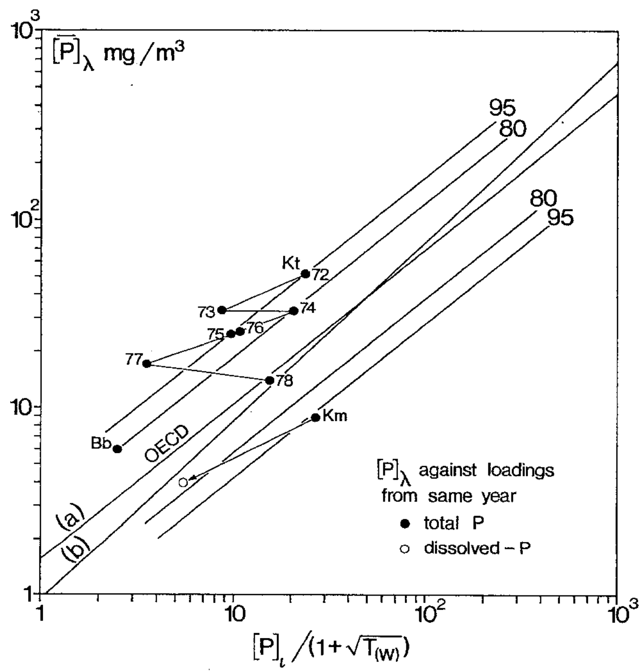

Figure VII 12. Mean total phosphorus concentration (same year) in relation to flushing corrected inflow total phosphorus concentration: Kootenay, Kamloops and Babine

Table VII 4. Kootenay Lake 1972 to 1979 Averages

Year	$[TP]_\lambda$ mg/m³	TN/TP Ratio St. 34 Spring	Chlorophyll (St. 31, 34, 57)*) mg/m³				Peak Value Observed	Primary Production gC/m².y
			0-20 m Composite Samples		1 m Samples			
			Lin. Av.	Geom.Av.	Lin. Av.	Geom. Av.		
1972	52	3.9	-	-	-	-	-	-
1973	33	5.8	-	-	-	-	-	ca. 190
1974	33	5.9	3.00 ± 2.63	2.34	4.02 ± 4.86	2.71	23.6	ca. 140
1975	25	8.5	(3.28 ± 3.40)	-	(4.45 ± 5.57)	-	19.9	-
1976	26	8.9	4.17 ± 2.61	3.58	4.21 ± 3.26	3.35	12.9	-
1977	17	11.8	3.60 ± 2.32	-	3.22 ± 1.92	-	9.0	ca. 170 a)
1978	14	13.6	2.62 ± 1.20	-	2.39 ± 1.47	-	7.3	-
1979	12	15.8	-	-	-	-	-	-
Prior to 1970**)	up to 108	-	-	-	-	-	-	up to 350

*) March to November Averages

**) Data base somewhat scattered and not directly comparable

— From Nordin (pers. comm.)

a) From Daley et al., 1981

n-1, the scatter - though reduced - remains considerable (cf. Figure VII 13). Smoothing the trend by plotting gliding averages of two years against loading averages of the two previous years shows that, in principle, the lake reacts to reduced load approximately proportionally to the achieved reduction (cf. Figure VII 13). However, it is also evident that the time span for the establishment of equilibrium is longer than two years. The equilibrium concentration expected with a load of 250 to 300 t/y should be between 5 and 10 mg P/m^3.

The positioning of Kamloops Lake in terms of total phosphorus is outside the 95% confidence range but reasonably within the 90% confidence range relative to dissolved-P. The discrepancy between actual total phosphorus concentration (9 mg/m^3) and expected concentration (23 mg/m^3) indicates that some 60% of the incoming total load (which is 80% apatite) is rapidly sedimented out (St. John *et al.*, 1976).

In regard to Babine Lake there is reason to assume that its long-term load has been underestimated by a factor of 2. This judgement results not only from the positioning of the lake in the lake concentration-loading diagram, but also from estimation of the areal export coefficient of its catchment system on the basis of the reported load. Accordingly, the export coefficient for phosphorus estimated at 1.9 kg/km^2 is extremely low for a catchment system having the characteristics of those of Babine Lake (cf. above).

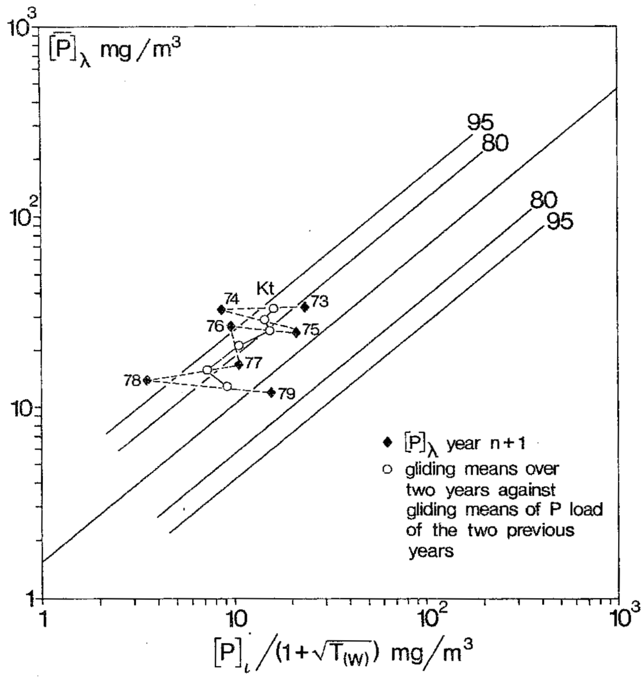

Figure VII 13. Mean total phosphorus concentration (one and two years later) in relation to flushing corrected inflow total phosphorus concentration: Kootenay

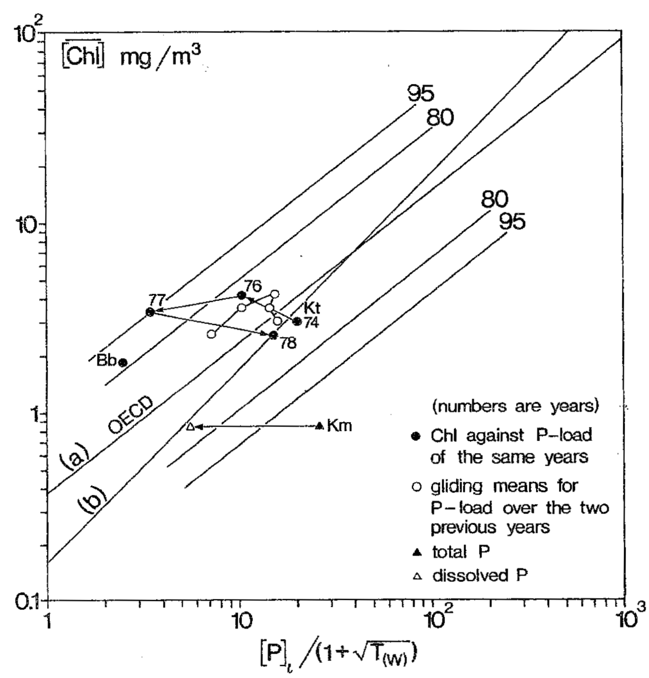

Figure VII 14. Annual mean chlorophyll a concentration in relation to flushing corrected inflow total phosphorus concentration (of same year and mean of two previous years): Kootenay

7.2.6 Chlorophyll-Loading Relationship (Babine, Kamloops, Kootenay). (Cf. Figure VII 14). The scattering observed in this plotting has the same origin as that discussed above for the loading-lake concentration relationship. For Kootenay Lake the chlorophyll values stay within the OECD uncertainty range if the data are smoothed and plotted against the load averaged over the previous two years. Also in this case it can be anticipated that the equilibrium concentration finally attained will be on the order of 2 mg/m³. Babine Lake stays high because of the likely underestimate of loading, whereas the chlorophyll value for Kamloops Lake relates better with the dissolved-P load than with the total-P load.

7.2.7 Secchi Transparency - Chlorophyll and Phosphorus Relationships (Babine, Kamloops, Kootenay). The Babine Lake and the only Kootenay Lake data available are well within the range of the respective OECD relationships (cf. Figures VII 15 and VII 16), and in regard to the Secchi-phosphorus relationship, are comparable with the Okanagan Lakes. This, in turn, corroborates the view that the original author-reported chlorophyll values are questionable for the Okanagan Lakes.

The Kamloops Lake situation, on the other hand, represents a different case. The yearly average Secchi transparency of this lake is only 2.9 m (0.5 to 5 m) and the lake shows consistently low chlorophyll. Therefore, the transparency is not so much dependant upon biomass as either mineral turbidity or colour. Colour does not seem to be important, but river borne turbidity has been reported to be extremely high from May to July (St. John *et al*., 1976).

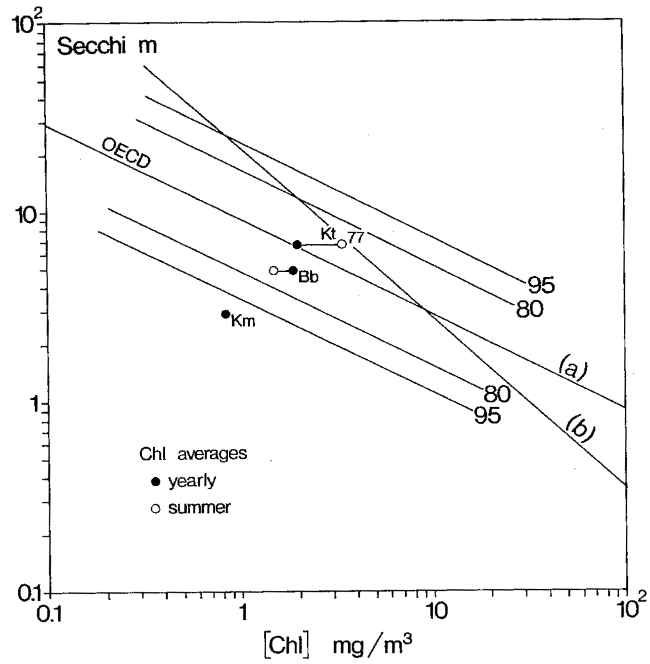

Figure VII 15. Secchi transparency in relation to annual mean chlorophyll a concentration: Kootenay, Kamloops and Babine

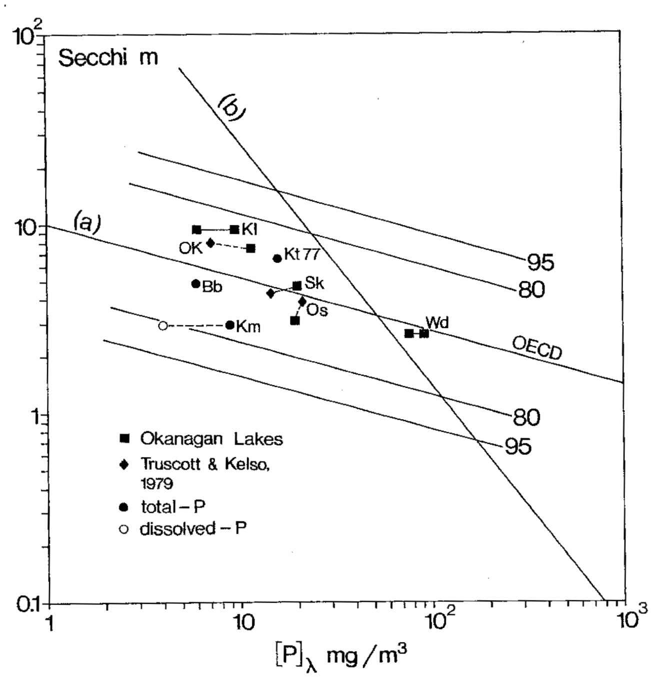

Figure VII 16. Secchi transparency in relation to mean total phosphorus concentration: Kootenay, Kamloops and Babine

7.2.8 Primary Production and Hypolimnetic Oxygen Depletion Rates. Yearly primary production measurements are available for Babine, Kamloops and Kootenay Lakes. These data have been plotted in Figures VII 17 and VII 18 against chlorophyll and phosphorus, respectively. The author-provided value for Babine Lake appears to be low in comparison to the OECD lines. However, recalculation of the yearly primary production from daily measurements shows that 25 to 40 g $C/m^2.y$ is an underestimate, and that the more likely yearly primary production of Babine Lake is 50 to 70 g $C/m^2.y$. If corrected in this sense, then the values of all three lakes stay within the uncertainty limits of the overall OECD data.

Good estimates of hypolimnetic oxygen depletion rates are available for the Okanagan Lakes and Kamloops Lake. In regard to correlating the Okanagan data with chlorophyll, the same problem arises as discussed above for the chlorophyll-P relationship. Also in this case, Secchi disc corrected chlorophyll values appear to be a more reliable reference basis than the original values. The corresponding plottings (cf. Figures VII 19 and VII 20) show that Okanagan oxygen depletion rates, referred to both the Secchi disc corrected chlorophyll and average phosphorus concentration, rank sequentially with the Kamloops Lake values. These latter are unaltered author-reported values.

In a more general context, the oxygen depletion rates for the lakes in question here agree well with corresponding data reported from 23 other lakes (cf. Figures 19 and 20), substantiating the direct dependency of hypolimnetic depletion rates on nutritional and biomass conditions. Some related aspects are reviewed in Appendix 3 (Hypolimnetic Oxygen Depletion Models).

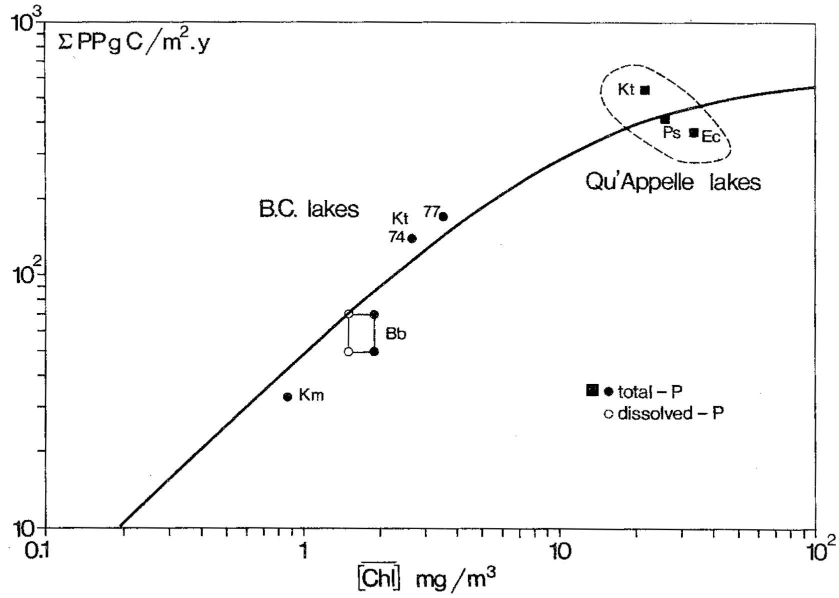

Figure VII 17. Annual areal primary production in relation to mean chlorophyll a concentration: Kootenay, Kamloops and Babine

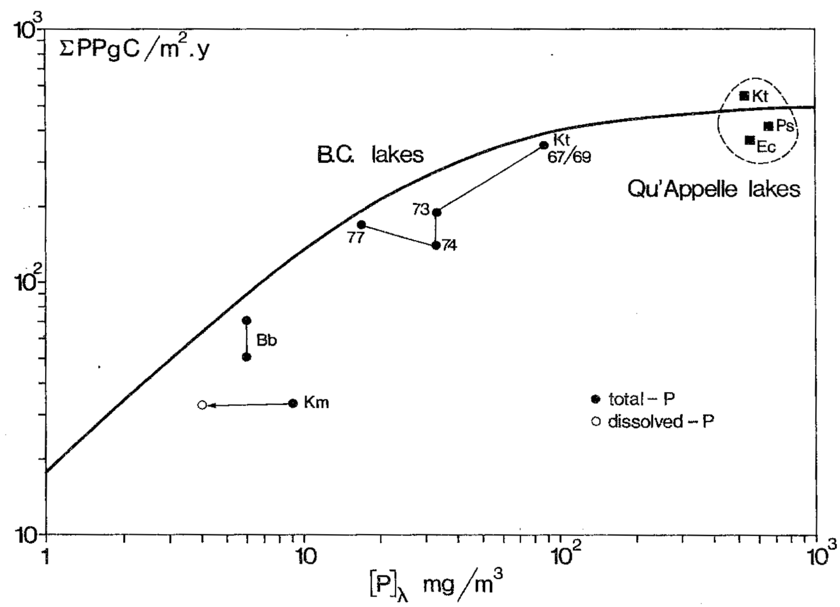

Figure VII 18. Annual areal primary production in relation to mean total phosphorus concentration: Kootenay, Kamloops and Babine

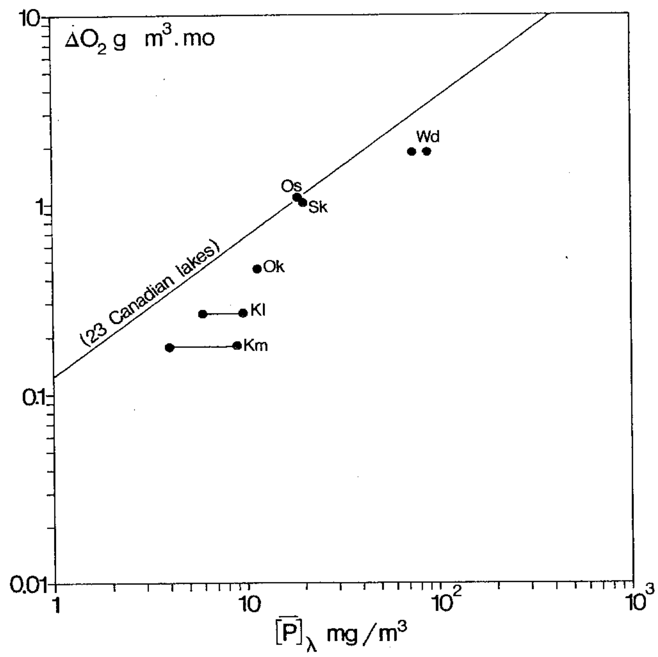

Figure VII 19. Monthly hypolimnetic oxygen demand in relation to mean total phosphorus concentration: B. C. lakes

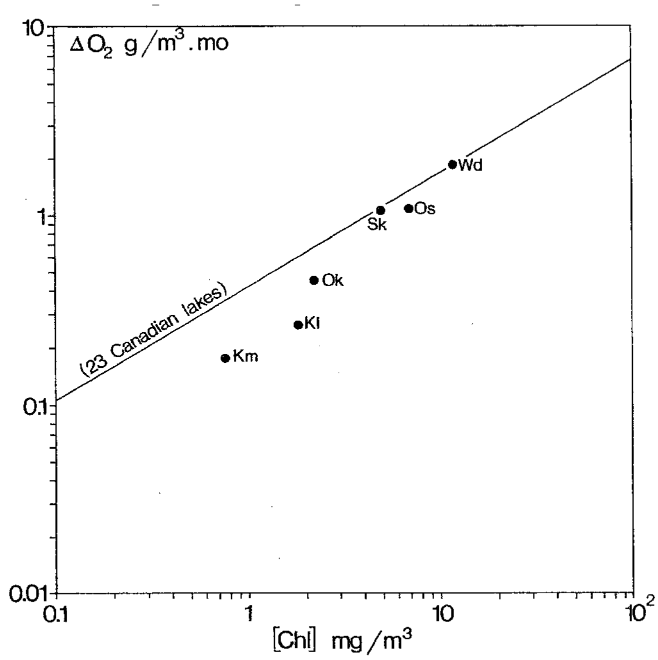

Figure VII 20. Monthly hypolimnetic oxygen demand in relation to mean chlorophyll $\underline{a}$ concentration B. C. lakes

7.3 Region VII Conclusions

Chlorophyll values for the Okanagan Lakes were found to be disproportionately high in relation to both phosphorus concentrations and Secchi transparency, yet Secchi transparency was found to relate to phosphorus concentration in the expected manner. Scrutiny of individual values of Secchi transparency and phosphorus showed that they were reasonable and warranted revision of the accompanying chlorophyll values. Further support of this came from some recent chlorophyll figures which were considerably lower. With revised chlorophyll values all relationships followed OECD expectations.

In Babine Lake, phosphorus concentration and chlorophyll were higher than that which could be expected on the basis of its estimated load. Further consideration showed that an areal export coefficient calculated from the loading value would be questionably low, therefore, loading of this lake has probably been underestimated.

In Kamloops Lake, chlorophyll was found to be low in relation to total phosphorus concentration, but close to expectation if dissolved phosphorus values were used. This is a consequence of the inert nature of apatite-P entering the lake as silt, and possibly the shading effect this has. (Secchi transparency is low in relation to chlorophyll and is probably controlled by mineral turbidity.). In addition, phosphorus concentration is low in relation to the estimated load to Kamloops, which is dependant on the high sedimentation rate of the incoming mineral load (estimated at 60%) and the flow-through hydrology of the epilimnion during summer stratification.

In Kootenay Lake, the OECD standards have been useful in following non-equilibrium conditions in the years following phosphorus reduction. Pairing of parameters from different years to attempt a 'match' for the OECD relationship gives some estimate of the time lag involved in establishment of the new equilibria. Calculation of recovery time for lakes can only be predicted on the basis of individual histories and data for this is limited.

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Chapter 8. Recovery of Lakes

Gravenhurst Bay

Little Otter Lake

Laurentian Great Lakes

- Erie
- Ontario

Qu'Appelle Lakes

Kootenay Lake

8.1 Introduction

In the past 30 years, some major efforts have been made to improve the trophic status of lakes where population growth has magnified both the need for and problems of obtaining clean water. Six such examples follow where nutrient loading is described both historically and in terms of recent attempts to alter it, as well as the consequences of these events for the characteristics we use to evaluate trophic condition. These few historical accounts are most valuable in that they provide an indication of what lakes will react to, and how rapidly and to what extent they will respond. In addition, comparison of responses of lakes with various histories allows,at this stage, at least qualitative evaluation of the importance past loading history has had in determining the present and future trophic conditions of lakes.

8.2 Gravenhurst Bay (Ontario Shield Region III)

8.2.1 Loading History and Trophic Condition. Prior to 1971, Gravenhurst Bay (of Lake Muskoka on the Shield in southern Ontario) received untreated effluent from the 7,000 inhabitants of the town of Gravenhurst. With this loading, a mean depth of 7.6 m and τ_w of 1.8 yr (cf. Table VIII 1), the bay's water clarity became very poor and severe algal blooms occurred, which resulted in an annual depletion of hypolimnetic oxygen. Alleviation of the problem began after 1971 when two sewage treatment plants were constructed and effluents were chemically treated for P removal before discharge into the bay. By 1975 the P

Table VIII 1. Morphometric and hydrologic data for Gravenhurst Bay.

Surface area (A_o)	4.13 km^2
Mean depth (z)	7.6 m
Maximum depth (z_m)	15.8 m
Volume (V)	$3.14 \times 10^7 m^3$
Drainage basin area (A_d)	38.9 km^2
Water replenishment time (T_w)	1.8 yr

Table VIII 2. Phosphorus sources for Gravenhurst Bay. (kg yr^{-1} unless specified)

P-Precipitation: Source	Before 1971	After 1975
Land drainage[1]	180	180
Shoreline cottages[2]	520 - 1290	280 - 690
Gravenhurst discharge	2580	680
Precipitation[3]	150 - 310	150 - 310
Total	3430 - 4360	1280 - 1860
Load (g $m^{-2}yr^{-1}$)	0.81 - 1.06	0.31 - 0.45

[1]Based on a P export coefficient of 4.7 mg $m^{-2}yr^{-1}$ (Dillon & Kirchner 1975 b).
[2]Based on a supply of 1.5 kg P $cap^{-1}yr^{-1}$ prior to 1973, 0.8 kg P $cap^{-1}yr^{-1}$ after 1973 (Dillon & Rigler 1975), and 0% - 60% retention of P in septic tank file field systems.
[3]Based on a phosphorus load in precipitation of 35 - 75 mg $m^{-2}yr^{-1}$ (Dillon & Rigler 1975; Scheider 1978).

Tables 1 & 2: From Dillon _et al._, 1978 concentration was down to 45% (i.e. 20 mg/m^3) of the pretreatment level (cf. Table VIII 2), and the bay showed marked improvement in water quality. Secchi transparency increased by more than 1 m and chlorophyll, biomass and oxygen depletion rates all dropped to about 50% of the pre-treatment levels (cf. Table VIII 3).

Changes in species composition of the phytoplankton accompanied the increased light availability and shift in nutrient proportion; the N:P ratio increased from 11 to 35. *Crytpophyceae, Chlorophyceae* and bloom forming *Cyanophyceae* decreased contemporaneously with increase in *Chrysophycea* and non-bloom forming *Cyanophyceae*. These latter are considered typical cohabitants of hypolimnetic environments and presumably the changes in light climate were responsible for their development.

A comparison of predicted conditions with those actually observed in 1975 shows that improvement was more rapid than expected (cf. Table VIII 4). The flexibility of this response was demonstrated in the following year, 1976, when there was a temporary failure at one of the sewage treatment plants. Efficiency of P-removal by $FeCl_3$ precipitation dropped from 90% to 40% and the bay responded with a dense bloom of *Aphanizomenon*. Once high P-removal efficiency was regained, the previous low biomass level and "typical oligotrophic species composition" were re-established within three months. In this case, rapid recovery of oligotrophic conditions resulted from phosphorus removal.

Table VIII 3. Water quality of Gravenhurst Bay before and after phosphorus precipitation

		[P] ($mg\ m^{-3}$)	TIN ($mg\ m^{-3}$)	TON ($mg\ m^{-3}$)	TN/TP	[chl *a*] ($mg\ m^{-3}$)	Secchi (m)	Algal biomass ($ASU\ ml^{-1}$)	AHOD ($mg\ cm^{-2}d^{-1}$)
Before:	1969	42	65	384	11.1	10.6	2.6	1780	0.053
	1970	39	77	427	13.9	5.1	3.1	1620	0.047
	1971	52	88	527	14.7	13.8	1.9	3620	—
	mean	44	77	446	13.2	9.8	2.5	2340	0.050
After:	1972	35	144	383	17.8	8.1	3.1	1410	—
	1973	33	163	399	18.3	6.9	3.2	1470	0.032
	1974	25	188	356	24.8	5.0	2.7	1710	0.043
	1975	20	257	320	35.2	5.0	3.9	1120	0.025

- from Dillon, *et al.*, 1978

Table VIII 4. Comparison of predicted and measured phosphorus, chlorophyll and Secchi depth values.

	1975	predicted
P_{sp}	26	19 - 27
P_{ss}	20	17 - 23
chl *a*	5.0	5.1 - 8.6
Secchi depth	3.9	2.9 - 2.3

Total phosphorus concentration at spring overturn (P_{sp}) and during thermal stratification (P_{ss}), calculated according to Nichols & Dillon (1977), chlorophyll concentration and Secchi depth during thermal stratification (Dillon & Rigler, 1975).
- from Dillon *et al.*, 1978

8.3 Little Otter Lake (Ontario Shield Region III)

8.3.1 Loading History and Trophic Condition. By the end of the 1960's, Little Otter Lake, a small Shield lake, was directly influenced by approximately 20 cottages, 20 commerical establishments, and Rockwell International - an automobile parts manufacturer. This industry (located at the outflow) used the lake primarily as a supply of cooling water but a small amount of the intake was released as treated effluent. In 1971 a severe bloom of *Anabaena limnetica* limited recreational use of the lake and the Ontario Ministry of the Environment subsequently found that the manufacturer had released 265 kg of phosphorus as a polyphosphate descaling agent in its effluent. It was this, in conjunction with other factors, which provided conditions amenable to bloom formation.

The mean depth of only 4.6 m ensured that this nutrient addition (i.e. 0.436 g/m²/yr) remained exposed to sunlight and temperature conditions favourable for growth. Washout and dilution were minimal in this

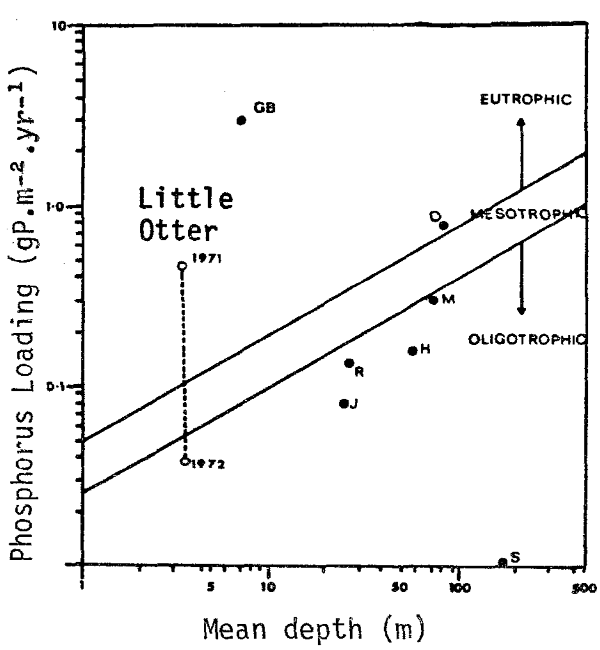


Figure VIII 1.

Return of oligotrophic conditions to Little Otter Lake with 1972 P-loading reduction.

- from Michalski and Conroy, 1973

period since flushing rate dropped from the high average of once per month to nearly zero during the summer. Phosphorus concentrations increased from < 5 to as high as 110 mg/ℓ at the outflow and exceeded 40 mg/ℓ throughout the rest of the lake. No essential change was seen in other nutritional factors.

The extent of this bloom was that of major change in trophic status of the lake (cf. Figure VIII 1). Phytoplankton standing crop increased from 400 to more than 19,000 ASU/ml at the peak of the bloom and chlorophyll a rose from 3 to 46 mg/l. Secchi transparency decreased from more than 5 m to less than 0.1 m.

When use of the descaling agent was discontinued in 1972, the reduction in loading was 92% (i.e. from 0.436 to 0.036 $g/m^2/yr$). Fortunately, the phosphorus which had accumulated in the sediments remained stable and concentrations resumed their former low level in direct response to loading reductions. Recovery was rapid and all former oligotrophic conditions returned that year.

8.4 Lakes Erie and Ontario (Laurentian Great Lakes Region IV)

8.4.1 Loading History. It was concluded in a 1969 report to the IJC that the data of the previous 6 years indicated pollution of the Great Lakes had progressed on both sides of the US-Canada boundary to the point of causing injury to health and property on both sides. Lake Erie was evaluated as in an advanced state of eutrophication and an acceleration of the process was occurring in Lake Ontario. The "injury" took the form of excessive algal growth (cf. Figure VIII 2) which resulted in difficulties in filtration of drinking water, unpleasant beach areas and

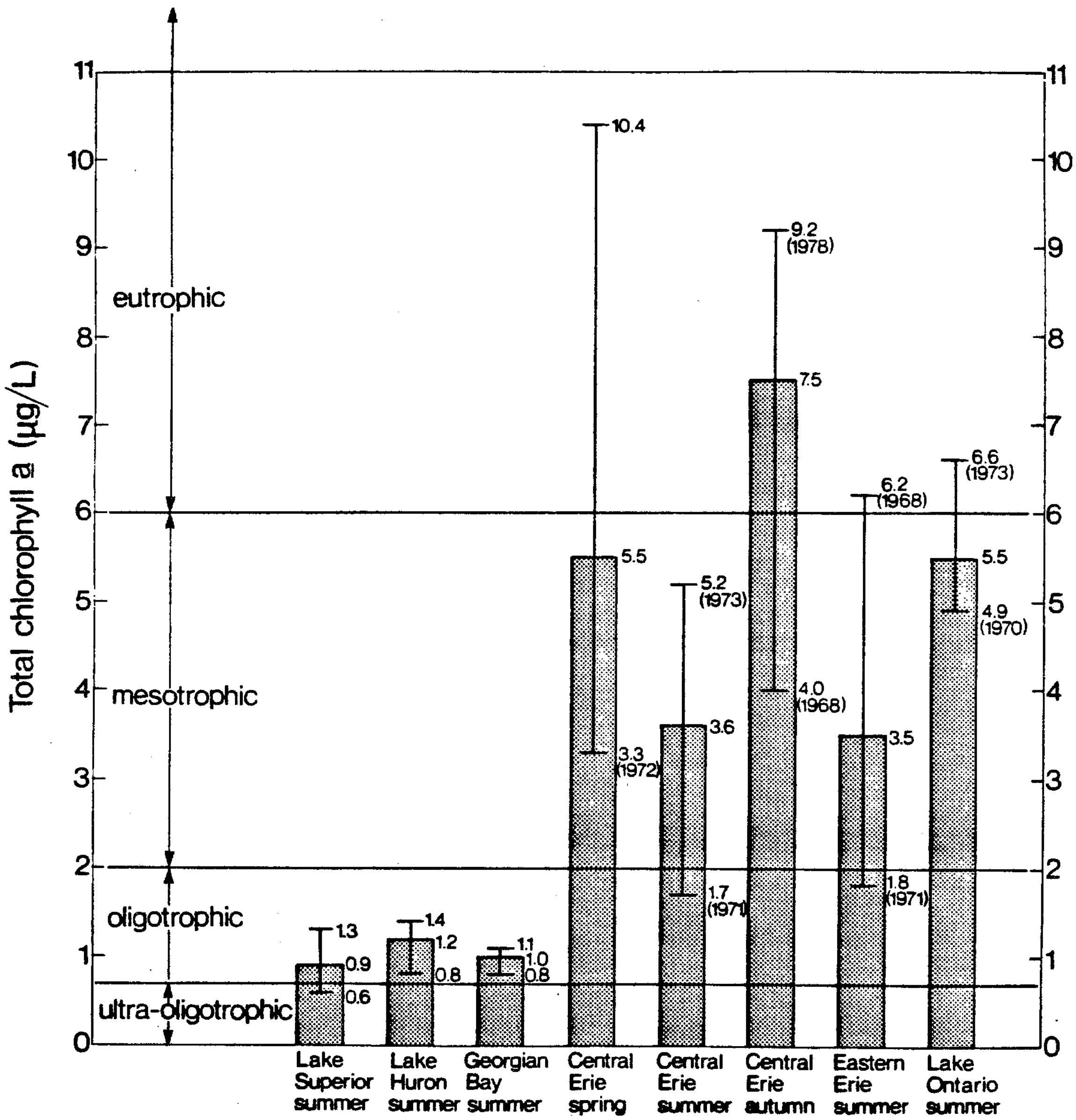


Figure VIII 2. Trophic conditions of the Laurentian Great Lakes in the 1960s and 1970s: Summary of chlorophyll a in offshore surface waters. (Limits are mean values in different years)
- from Dobson, 1981

property devaluation. Decay of this growth was also thought to have caused the secondary problems of oxygen depletion, fish kills and subsequent alteration of the species composition of fish populations to those more tolerant of low oxygen levels but far less desirable as food or for sport. Recognition of these problems resulted in the 1972 "Agreement between the United States of America and Canada on Great Lakes Water Quality" in which it was resolved:

1) that phosphorus loading (the cause of the excessive algal growth) to the lakes would be reduced to levels allowing return of oligotrophic conditions to Lakes Erie and Ontario, with maintenance of those conditions in other lakes;
2) that this could be accomplished through such measures as treatment of municipal and industrial wastes, management of land use and agricultural practices, limitation of the phosphorus content of detergents, etc., and
3) that the target loads to achieve these goals by 1976 were estimated to be 14,600 and 9,100 metric tonnes/yr. for Lakes Erie and Ontario, respectively

The target loads have been revised in the 1979 renegotiation of the Agreement and have been lowered to 11,000 t/y and 7,000 t/y for Lakes Erie and Ontario, respectively.

However, the figures regarding the historical and present loading are a point of contention. Reliable load estimates for such large systems are difficult to attain, and estimates made by different groups vary considerably. In the period between 1970 and 1973, Canadian estimates for Lake Erie (Fraser and Willson, 1981) paralleled those of the US Army Corps of Engineers, (US COE), but later on the estimates diverge substantially (cf. Figure VIII 3). For 1976, the Canadian (Fraser and Willson, 1981) estimate of total phosphorus loading to Lake Erie is 15,800 tonnes and the US COE estimate is 18,400 tonnes. The difference is 2,600 tonnes (16%) higher by US figures. A similar discrepancy exists in the 1976 Lake Ontario estimates, and this is 9,706 tonnes according to the Canadian estimate or 11,803 tonnes according to the US estimate. The difference here is approximately 2,100 tonnes, or 22% higher by US figures (cf. Fig. VIII 4).

Accordingly, it also becomes somewhat tenuous to establish direct cause - effect relationships between variation in load and variations in trophic conditions observed over the same period of time, and earlier. Several attempts have been made, and the best that can be said condenses into the following summary.

8.4.2 Variations and Trends in Phosphorus Concentrations. Dobson (1981) has reported the trends of increasing, then decreasing spring total phosphorus concentrations in Lakes Erie and Ontario over the 1970 to 1980 period (cf. Figure VIII 5). In eastern Lake Erie, concentrations changed from 16 μg/ℓ to a peak of 25 μg/ℓ and back to 16 μg/ℓ. In Lake Ontario, concentrations decreased from approximately 20 to 16 μg/ℓ over this ten year period. In both cases some improvement has been seen from early 1970 levels. Nevertheless, the cause of these trends is difficult to relate to the as yet quantitatively undefined decrease in

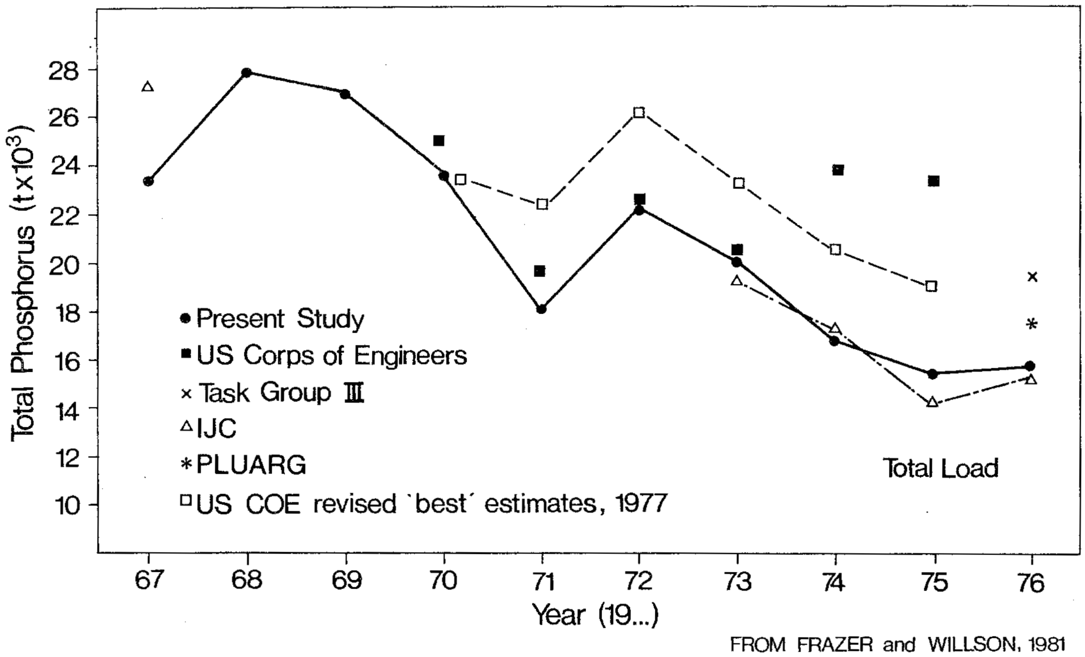


Figure VIII 3. Lake Erie phosphorus loading estimates (1967 to 1976)

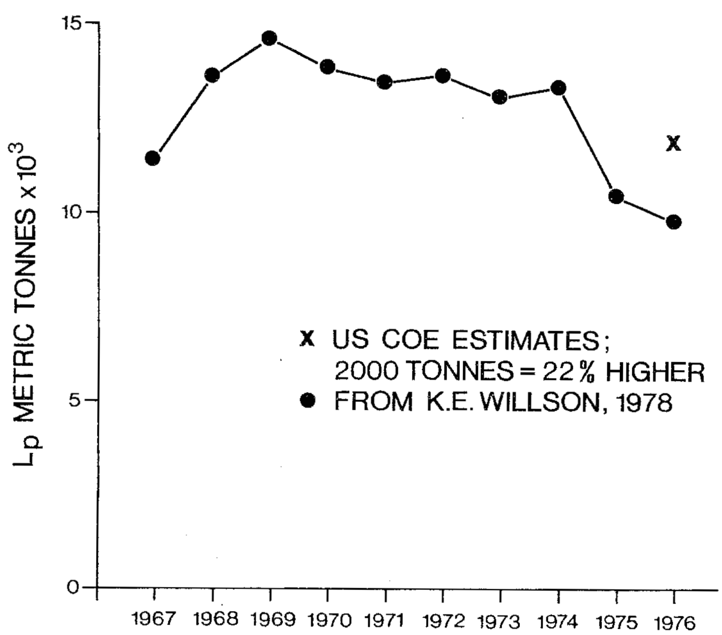


FIGURE VIII 4. LAKE ONTARIO LOADING ESTIMATES (1967 to 1976)

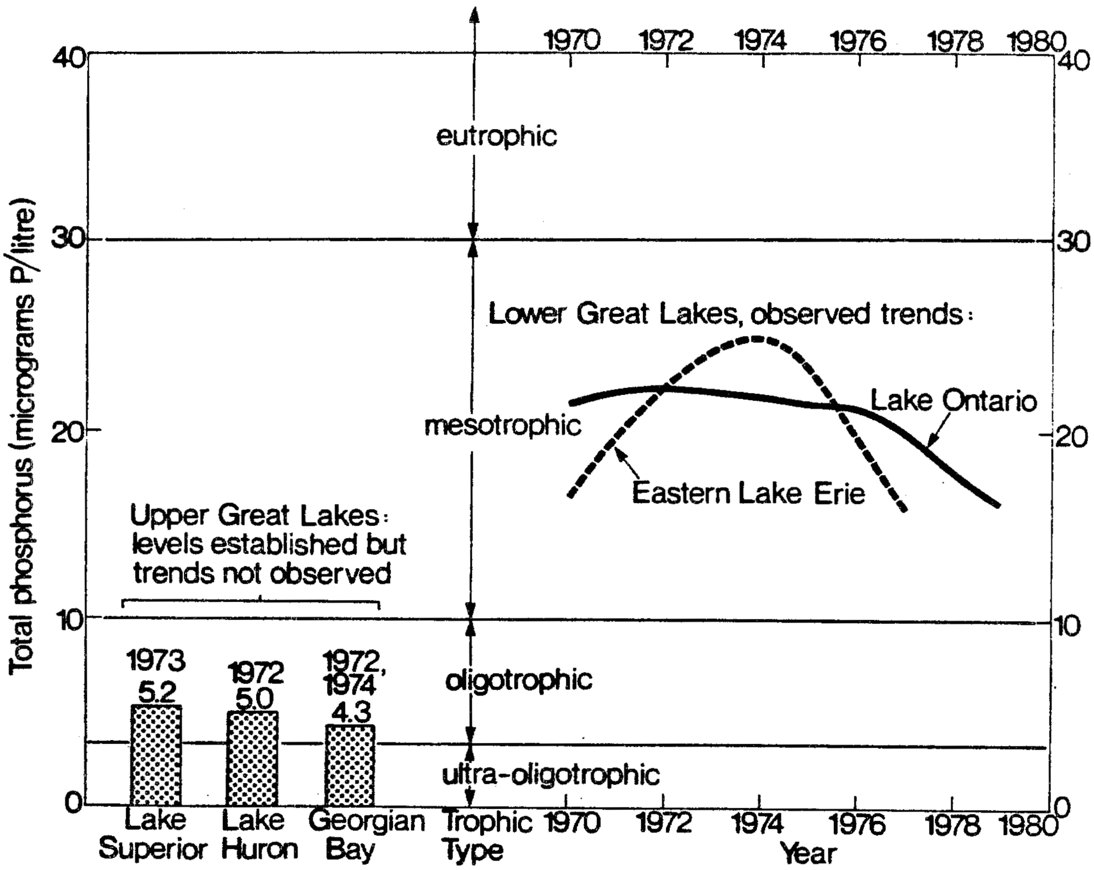


Figure VIII 5. Spring total phosphorus trends in the Great Lakes. - from Dobson, 1981 loading, as it was also observed that lake concentrations in this period varied directly with changes in water level ($r = 0.74$; Dobson, 1981), and therefore, erosional load may at times be the most important determinant of lake concentrations. Early 1970s concentrations were high, parallel to relatively high water levels. In any case, present phosphorus concentrations in both Erie and Ontario are halfway from mid-1970 peaks to the proposed goal for oligotrophic conditions at 10 μg/ℓ.

8.4.3 Variations and Trends in Biomass (Chlorophyll). Watson (1977) has reported a relationship between chlorophyll and phosphorus concentration in the Great Lakes. Although decreasing phosphorus concentrations (as described above) are apparent, an expected decline in chlorophyll levels has not been conclusively demonstrated. Kwiatkowski and El-Shaarawi (1977) found that a high percentage of chlorophyll variability could be explained by physico-chemical parameters, but problems persist in arranging for sensible correlation of chlorophyll and nutrients. Chapra and Dobson (1979) note particularly the high variability due to inshore-offshore differentiation caused by intermittent upwellings and localized influences of river discharges and municipal pollution. In this regard, emphasis on characterization of horizontal spatial and temporal variability in the Great Lakes (with the inevitable sacrifice of studies on vertically related phenomena) has been of primary importance (Vollenweider *et al.*, 1974). Trends in chlorophyll data may not be discernible until some consistent sampling strategy accounting for the variability of these parameters has been developed.

The same problems plague the data of total phytoplankton biomass, However, there is some evidence for a decrease in phytoplankton biomass in Lake Erie. An inshore station monitored for 9 years at weekly intervals showed a 42% decrease of biomass in Lake Erie's western basin between 1970 and 1975 (Nichols, et al. 1977). In another study, offshore station comparison of 1970 with 1979 showed that average phytoplankton biomass had changed from 4.3 to 2.6 (i.e. 40%) in the western basin and from 2.7 to 2.1 (i.e. 22%) in the central basin (Munawar, 1981). Whether or not this decrease is a trend, cannot be evaluated, since intermediate years are not presented. In Lake Ontario the data for 3 offshore stations show that in the years 1970, 1975, 1977 and 1978, biomass was always approximately 1 to 2 g/m³ (Munawar, 1981) and no definite trends appear.

Phytoplankton species composition trends are described in a following section.

8.4.4 Variations and Trends in Secchi Transparency. In the mid-1960s, annual mean Secchi transparency for offshore waters of Lake Ontario was 3.5 m. This decreased to 2.5 m in the early 1970s when phosphorus concentration was maximal and increased again after 1972, when a legal limit was placed on the phosphate content of detergents in Ontario and New York State (cf. Figure VIII 6). A gradual improvement of water clarity was observed and by the late 1970s, Secchi transparency was approximately 3 m.

In Lake Erie no historical trend is available. Secchi depth in the central area of the lake was about 5 m in 1975 to 1978. Such transparencies would hardly suggest an overabundance of phytoplankton during

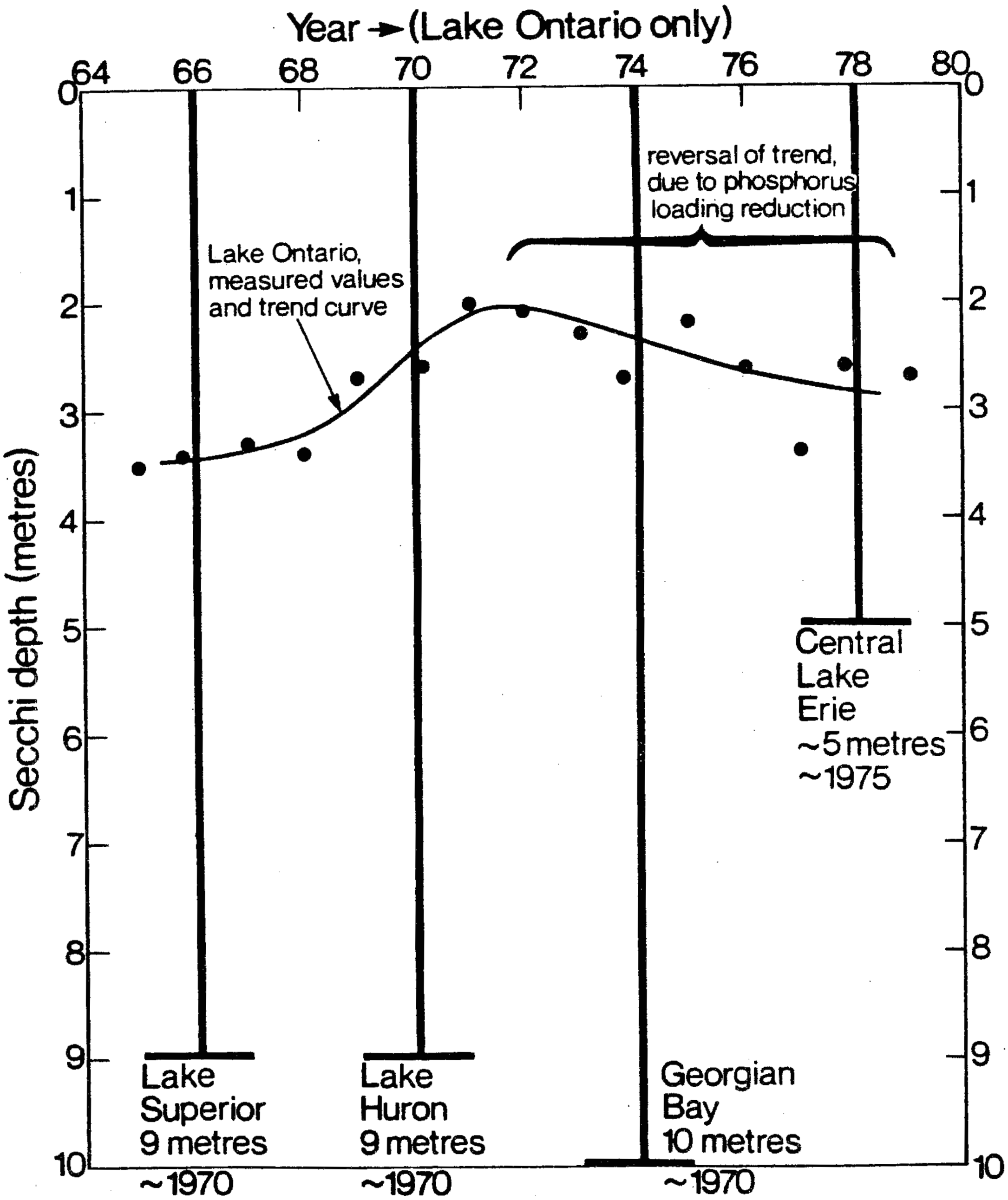


Figure VIII 6. Offshore summer Secchi transparencies of the Great Lakes.

- From Dobson, 1981

these years. It is further to be noted that the reputation Lake Erie has gained as being in an advanced state of eutrophication is largely due to the unpleasant and obvious signs of localized inshore areas (rather than whole-lake deterioration) where, for example, problem growths of *Cladophora* have occurred.

8.4.5 Variations and Trends in Hypolimnetic Oxygen Depletion Rates in Lake Erie. In the late 1960s, anoxic conditions were observed in the hypolimnion of Lake Erie and it was presumed that this was the consequence of the high nutrient load which the lake received at that time. Hence, the IJC set the goal of restoration of year-round aerobic conditions of Lake Erie's central basin to be approached through a prescribed program of loading reductions. Subsequently, reanalysis of the data by Charlton (1980, a) showed that year to year variation was greater than the supposed trend of oxygen depletion, and that oxygen conditions were related to changes in hypolimnetic thickness which was in turn controlled by climatic factors. Biomass (as chlorophyll) was found to have very little influence on the situation and oxygen depletion would persist even if chlorphyll levels were halved (Charlton, 1980, b). This hypothesis, that low oxygen conditions in Lake Erie are a natural consequence of basin characteristics and not solely a recent eutrophication related phenomenon, is also supported by a study of the paleolimnology of Lake Erie's ostracodes. Those species tolerant of low oxygen conditions appear in the sediments before 1850 and possibly as long as 4,000 years ago (Delorme, 1969). Evidently, periodic situations of hypolimnetic anoxia have occurred for a long time. Vollenweider (IJC, 1981) has estimated that to bring the hypolimnetic oxygen concentration, through phosphorus control, to a level established by the Canada-US Agreement, would require a reduction of 75% of the 1976 loading level (i.e. from 20,000 to 5,000 metric tonnes per year). In view of these contentions, the goal of restoration (?) of year-round aerobic conditions of Lake Erie's hypolimnion requires reconsideration.

8.4.6 Historical Trends in Phytplankton Species Composition Changes. a) Lake Erie. The most recent and comprehensive treatment of Lake Erie's species composition history is presented in Harris and Vollenweider's (1981) analysis and interpretation of diatom remains of a core from the central basin of the lake. Prior to the 1850s, the Centrales constituted more than 80% (frequency) of the identifiable diatoms. As such, *Melosira distans* and *Cocconeis disculus*, "species characteristic of clear oligotrophic lakes", were identified in the older sections of the core. The first major change in nutrient loading to the lake took place around 1860 when the Great Black Swamp was drained and major forest clearance took place. This was paralleled in the core by transitions of the *Melosira* species; *M. distans*. and *M. italica* were eventually replaced by *M. islandica* and *M. granulata*. By the 1900s a large number of species commonly found under eutrophic conditions had appeared (Snow, 1903) and major shifts in species composition occurred. *Stephanodiscus niagarae*, common throughout the core, lost relative importance to *S. astraea* and *Coscinodiscus rothii* and *Fragilaria crotonensis* became outnumbered by *F. capucina* (about 1930). These changes were further stimulated in the 1950s when nutrient supply to the lake accelerated with population growth and the introduction of detergents containing phosphates. The Melosira species (*M. islandica* and *M. granulata*) shifted to *M. binderana* around 1960. These long term changes may be summarized as an increase in the percentage of Araphidineae present concurrent with a decrease in the percentage of Mono- and Bi-raphidineae (cf. Figures VIII 7). The core information ends in the late 1960s and this structural trend carries on up to that time.

The recent trends of the 1970s during the period of concerted effort to reduce phosphorus loading to the lake, are contained in two reports, both of which claim substantial decreases in total phytoplankton. Nicholls et al. (1977) found that phytoplankton density (in terms of a.s.u. /ml) at a water treatment plant in the western basin had fallen by 42% and although all classes of algae had decreased, the most obvious decline was in the diatoms, since these comprised up to 80% of the total biomass over the 9 year study (cf. Figure VIII 8). Samples were taken weekly, so the trend at this location seems fairly well substantiated. Another comparison (Munawar, 1981) of two years (with a gap of 8 years between) showed that annual average biomass for the western basin was lower in 1979 than 1971 due to decreases in Cryptophyceae, diatoms and blue-greens, but it was seasonally higher in the fall by 150% because of the increased density of blue-greens and some greens. Similarly, in the central basin, annual average biomass, though lower due to decreases particularly in the diatoms, Dinophyceae and Cryptophyceae, was seasonally higher but in the spring and fall, because of higher densities in these seasons of diatoms, greens and Chrysophyceae.

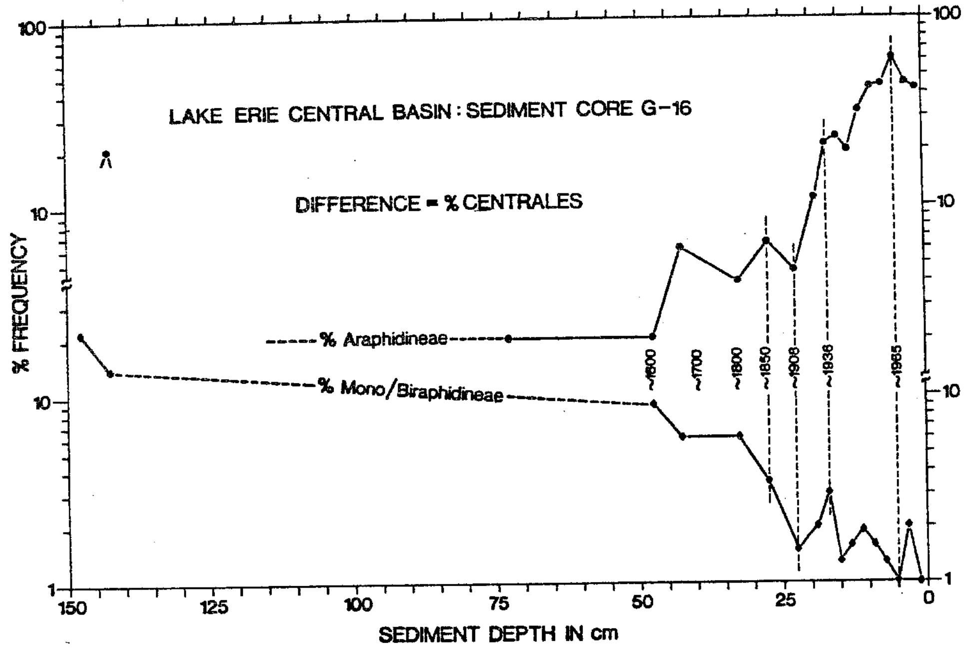
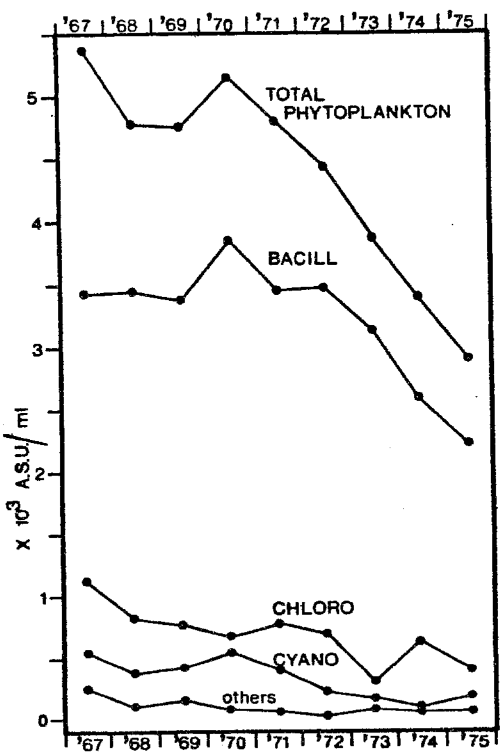


Figure VIII 7. Long term changes of diatoms in the sediments of Lake Erie; from Harris and Vollenweider, 1981.

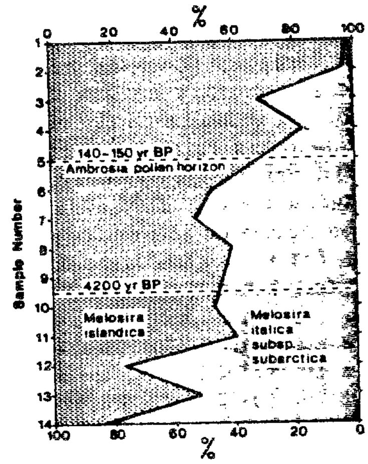


Bacill - Bacillariophyceae
Chloro - Chlorophyceae
Cyano - Cyanophyceae

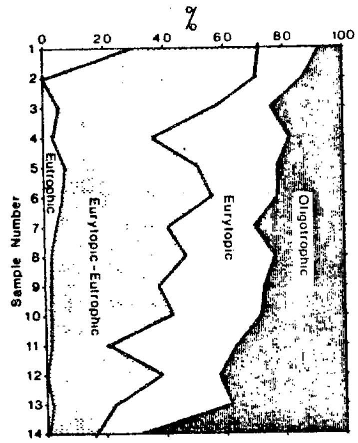
Figure VIII 8. Recent trends in Western Lake Erie phytoplankton - from Nicholls et al., 1977

In summary, the recent community structure of Lake Erie on an annual basis appears to be changing to one with lower relative importance of diatoms and Cryptophyceae because of their absolute declines to one with greens and blue-greens as more obvious components which occur in slightly higher densities than they have previously. How far this generalization for recent years is true could only be known if data for consecutive years were available such that trends could be distinguished from yearly variability.

b) Lake Ontario. The long term species composition history of Lake Ontario, as elucidated from sediment core analysis (Duthie and Sreenivasa, 1971) shows some definite similarity to the trends identified in the previously described Lake Erie core. The most outstanding similarity is the transition in *Melosira* species and, as in Lake Erie, *M. italica* has been replaced by *M. islandica*, particularly coincident with forest clearance about 150 years B.P (cf. Figure VIII 9, a). At that time, all species underwent a large increase, presumably due to increased erosion and heightened nutrient loads. From that point on, all diatoms declined to present, but greatest relative decreases were found in the "oligotrophic group" consisting mostly of *Cyclotella* species, (cf. Figure VIII 9, b). Unlike the Lake Erie core, percentages of Mono- and Biraphidineae and Araphidineae remain nearly constant. It is proposed that the decline of diatom populations over the past 150 years has been part of an overall shift to a more heterogeneous assemblage, most of which would have decayed leaving no trace in the core. This would be the most likely interpretation since an overall decline of the phytoplankton would not be expected on the basis of nutrient loading history.



a) **Proportional representation of two *Melosira* species in 14 sample levels from core section.**



b) **Proportional representation of trophic groups of planktonic diatoms in 14 sample levels from core section.**

Figure VIII 9. Long term a) species and b) compositional changes of diatoms in the sediments of Lake Ontario.
from Duthie & Sreenivasa, 1971

Recent data (Munawar, 1981) for the last decade (1970s) show no definite trends for Lake Ontario as a whole in terms of total biomass. For the four years 1970, 1975, 1977 and 1978, biomass was always between 1 and 2 g/m^3 and the lake on this basis would be regarded as oligo- to oligo-mesotrophic. From the compositional standpoint, Cyanophyceae and Chlorophyceae have declined (in contrast to the slight increase of these groups seen in Lake Erie) and Dinophyceae have increased over the 4 years of data . At present the lake's biomass on an annual basis is for the most part composed of diatoms, Cryptophyceae and Dinoyphyceae, thus Lakes Erie and Ontario have arrived at similar group compositional structures in recent years through rather dissimilar changes.

8.5 Qu'Appelle Lakes (Prairie Region VI)

8.5.1 Loading History and Trophic Conditions. In the prairie region, evaporation exceeds precipitation making water a very scarce and precious commodity. Conservation and recycling of the resource is of prime importance and is largely accomplished through a system of multiple use. The variety of purposes which it must serve include domestic and recreational activities, crop irrigation, livestock watering, industrial use, and support of a large and economically important fishery (i.e. this represents approximately 1/3 of Canada's freshwater catch). In the early 1970s, the population in the Qu'Appelle Lakes watershed had grown, simultaneously intensifying the contradictory aspects of demand for clean water and unintentional nutrient enrichment of the lakes through increased use of fertilizers and the industrial-domestic sewage discharge to inflow tributaries. Dense blooms of blue-green algae were common occurrences in the prairie lakes and were thought to have worsened with the growth of the two main population centers of Regina and Moose Jaw. The chronically eutrophic condition of the 'Fishing Lakes' with the problems of poor water quality eventually gained the attention of federal and provincial governments and became the subject of nutrient budget investigations. In 1973, it was estimated that nearly 50% of the nutrient input originated from the two major municipalities (Cullimore & Johnson, 1971) and the decision was made to spend several million dollars on construction of tertiary treatment plants (which became operative in 1977) for phosphorus removal from the sewage these cities produced. The hope was that this would relieve some of the problems caused by excessive algal growth. A follow-up study of nutrients and algal production in the years (1977-1978) immediately following nutrient reduction efforts showed that chlorophyll _a_ values throughout the Qu'Appelle Lakes remained high (means for various locations varied from 39 to 175 mg/m^3) and phytoplankton biomass accumulations still produced visible signs of eutrophication (Allan and Roy, 1980). Essentially no improvement of conditions was observed. However, judgement of this is difficult because of the extreme natural variability in prairie lakes and lack of long term characterization available, and as Allen and Roy (1980) point out, near drought conditions which persisted throughout the study period may have been the most important factor in determining trophic conditions of the lakes at that time.

In retrospect, it was probably a mistake to draw the conclusion that the eutrophic state of these lakes was related to recent increases in phosphorus loading due to population growth. Historical accounts point out that prior to any settlement of the area "the water was rendered very disagreeable by the great quantity of confervae covering nearly the whole surface .... decaying and rotting under the hot sun" and that thick green slimes were a hindrance to canoe travel. Therefore, the lakes were eutrophic before nutrient loadings increased. In addition, the N:P ratios ($\cong$ 5) are so low that nitrogen is more likely to be the limiting element than is phosphorus (cf. Chapter VI).

Some interesting speculations with regard to the effect of P reductions have been made. One by Allan and Kenney (1978) suggests that slight or insufficient reduction of P may be enough to alter the N:P ratio in such a way that the chlorophyll would actually increase (cf. Figure VIII 10). A proportional decline in water quality would not necessarily follow since the anticipated changes in species composition would be to one less noxious than that of blue-green communities.

Further uncertainty regarding the outcome of P reductions stems from the lag in response expected of environments with long histories of P loading where stores of nutrients have been built up in the sediments. The importance of this for the Qu'Appelle Lakes is unknown since (as discussed in Chapter VI) these lakes act as modest sinks or flow-through systems for P.

8.6 Kootenay Lake (B.C. Region VII)

8.6.1 Loading History and Trophic Condition. Rapid changes in Kootenay Lake began in 1953 (cf Table VIII 5) when the Cominco fertilizer plant began operation and discharge of phosphorus-rich effluent into the southern (Kootenay River) inflow. Within the next 20 years, the two

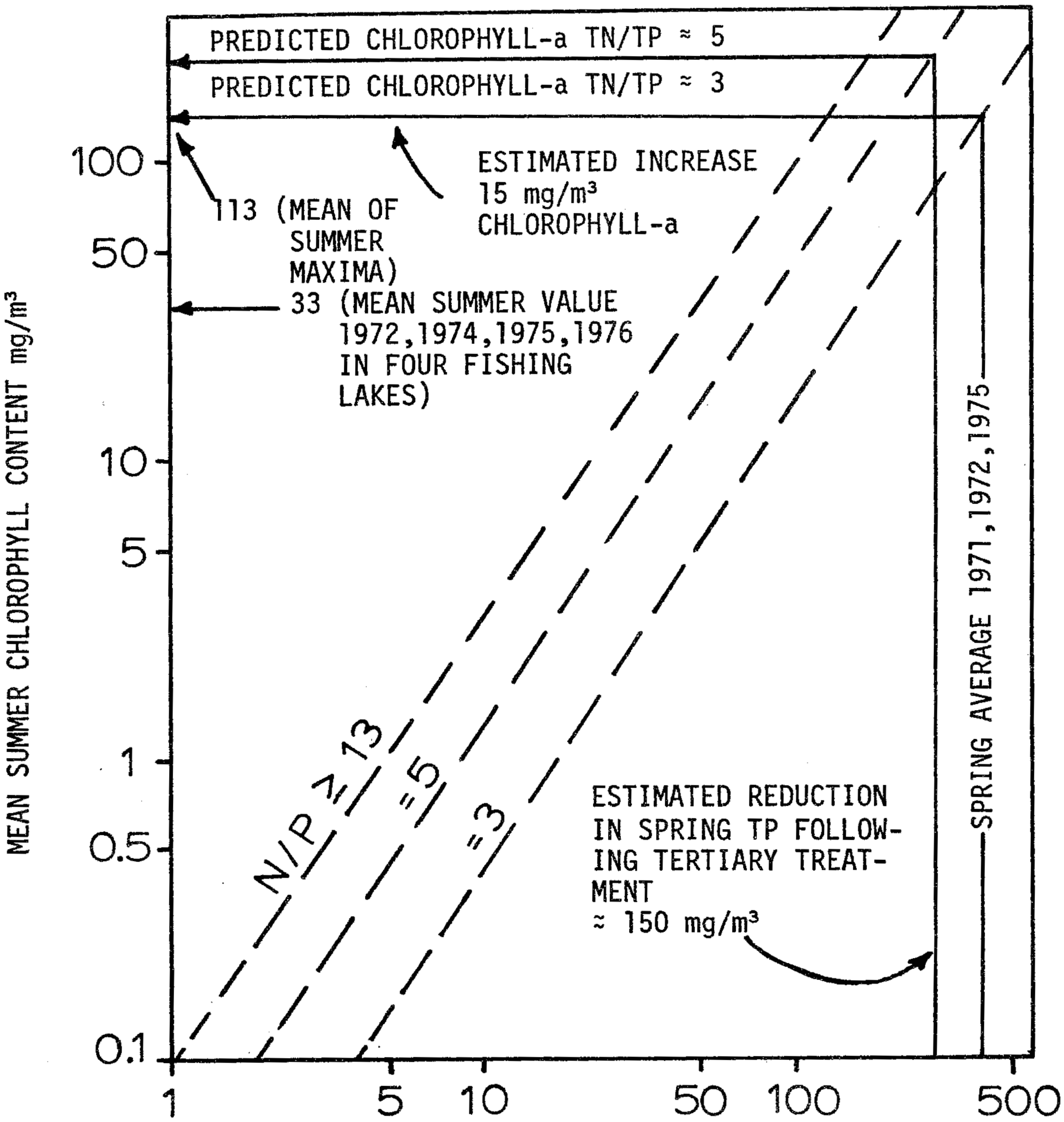


FIGURE VIII 10. Effect of TN/TP ratio on predicted Chlorophyll-a (after Sakamoto, 1966)

\- from Allan & Kenney, 1978

Table VIII 5 . Species composition in response to physical and chemical changes in Kootenay Lake.

Year	Event*	N:P	Phytoplankton [Chl]	Phytoplankton annual biomass maxima mg/m³ So./ No./ Ctr.	Phytoplankton predominant genera of biomass maxima
1950		≃ 15			
1953	1) Cominco fertilizer plant in operation.				
1966-68	2) Duncan Dam closed (1967)	< 1		> 3000[a)]	
1969	3) Pollution abatement legislation: initial P-removal				
1972	4) Libby Dam closed	3.9		- / 792 / 524	{ Oscillatoria Anabaena
1973		5.8		3000/ 695 /1946	Anabaena { Fragillaria Asterionella
1974		5.9	3.0	2663/ - / 606	{ Asterionella Fragillaria
1975	5) P-load from fertilizer plant reduced by 90%	8.5	3.3	230/ - / 251	Fragillaria
1976		8.9	4.2	214/1993 /1401	Fragllaria Stephanodiscus
1977		11.8	3.6	274/ 806 / -	Fragillaria Tabellaria
1978		13.6	2.6		
1979		15.8			
(?) 1984	prediction	25[a)]	< 1-2[a)]		

* numbers correspond to those on Figure VIII 11.
\- from Nordin '79 pers. comm.
a) from Daley et al. ,1980

major inflows were dammed: in the north Duncan Dam began operation in 1967 as a flood control structure and in the south Libby Dam began operation in 1972 as both a flood control structure and a means of generating hydro-electric power. Following completion of the dam, a number of public reports were made of fish kills, large blooms of floating algae (especially in the south) and changes of water level and ice formation in the western outflow. Unfortunately, coincidence in time led many local residents to attribute these changes to dam completion, and it is only the careful 1978 (Daley *et al.*, 1981) analysis of nutrient relationships which reveals satisfactory explanation of the causes of these problems.

8.6.2 Effects of Impoundment. Impoundment may potentially change the depth of the euphotic zone, depth of the epilimnion (mixed zone), residence time, hydraulic load, and consequently, the interactions of all these factors. In the case of Kootenay Lake, impoundments did reduce turbidity such that the depth of the euphotic zone increased, but this had no impact on annual yields of phytoplankton. The turbidity change was, in effect, a difference for only a brief period of the year. In addition, thermal structure, hence depth of the mixed zone during stratification, remained essentially unchanged. Residence time of the epilimnion increased for 1 to 4 months but since both figures are in excess of the generation time of phytoplankton, the decrease in washout rate was inconsequential. With regard to phosphorus loading, the Libby reservoir efficiently retained P such that the downstream lake received a much reduced TP load (≅ 25% less). While phytoplankton populations did decline in response to the decrease in loading, the drop was not proportional. Consideration of the form of phosphorus perhaps provides explanation: much of it was in the form of suspended particulate apatite which is unavailable as a plant nutrient. In addition, the annual cycle of loading was altered by damming. Nutrients which formerly entered the lake during the spring freshet and summer growing period, when stratification aided maintenance of the nutrient pool in the euphotic zone, now were distributed throughout the water column in the isothermal winter period when reservoir releases occurred. This redistribution of nutrient loading had little effect on Kootenay Lake initially because the load which was redistributed was a small percentage of the total supply. Most of the P supply to the phytoplankton at the time of dam closures was made up of the P existing in the epilimnion just after spring overturn and reflected the excessive phosphorus pollution of the late 1960s. As time passes, this overturn supply will be diluted and the growing season river supply will gain importance. It has been estimated that by 1984 the dams will have caused a 50% reduction of the growing season nutrient supplies. Even this reduction, though, in absolute terms, is small in comparison to that which followed pollution abatement (Daley *et al*., 1981).

8.6.3 Effects of Nutrient Loadings. Nutrient loadings and the eventual lake concentration response, were quite substantial in the 1950 to 1980 period, which encompasses both ascending and descending phases. The upsurge of nutrient input began in 1953 when the Cominco fertilizer plant started release of phosphorus rich effluents into the Kootenay River. In 1962 production by the plant had doubled and by 1965 tripled, which increased loading eight times (cf. Figure VIII 11). This rise in loading, from 0.8 to 6 $g/m^2.y$, was followed over the next two years by a

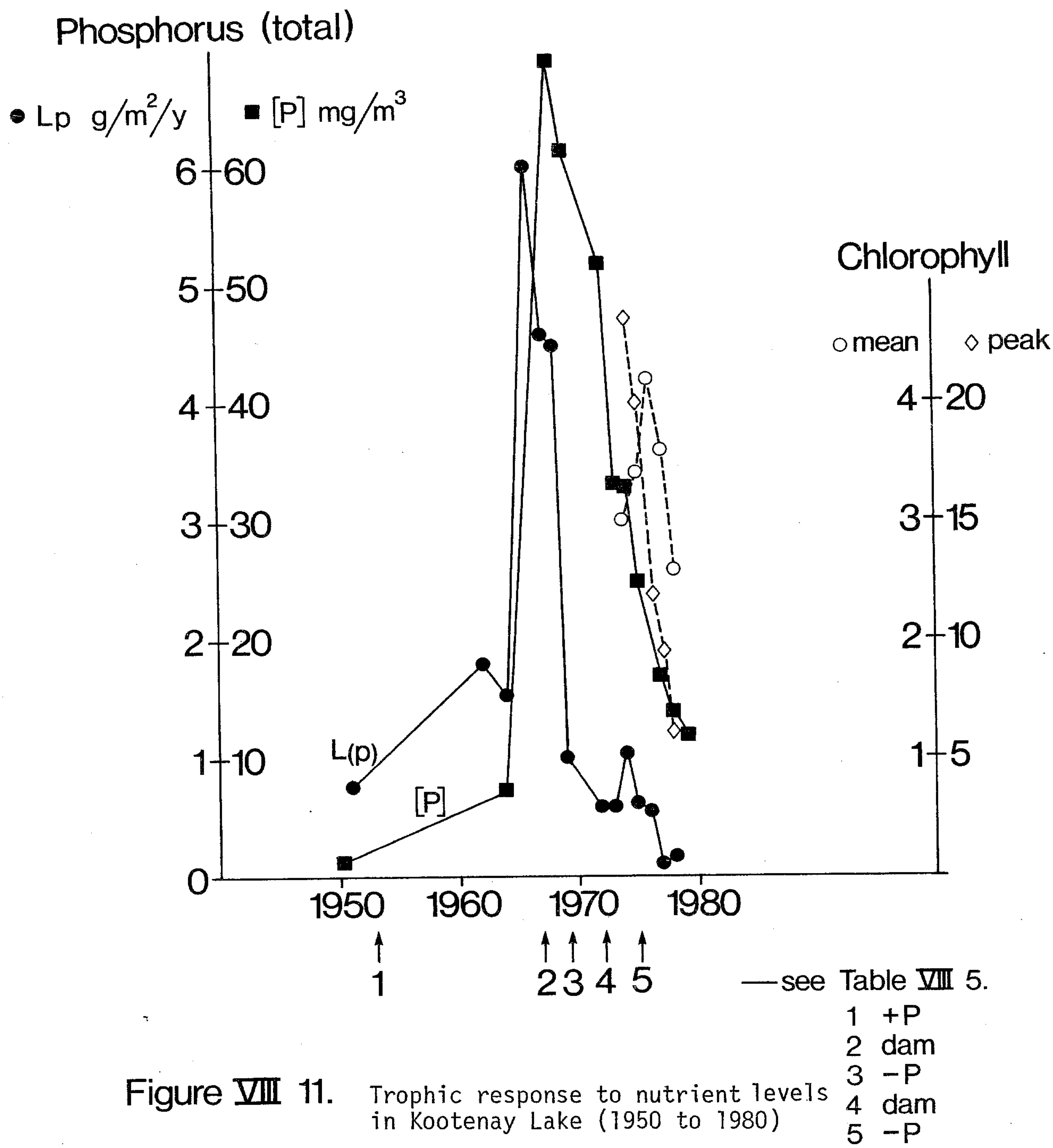


Figure VIII 11. Trophic response to nutrient levels in Kootenay Lake (1950 to 1980) sharp rise in concentrations, from approximately 1 to a maximum of 70 mg/m^3. Pollution control measures at the plant were dramatically improved in 1969 and were improved to a lesser degree in several stages to 1975 when the combined effect of these improvements resulted in a 90% reduction of P releases. The loading reduction occurred essentially in 1969, but the P concentrations have not reached the equilibrium level because the concentrations are still decreasing. Although the residence time is only 1.8 years, which should allow re-equilibration in less than six or seven years, incomplete winter mixing has retarded the washout of the pollution loadings of the 1960s (Wiegand and Carmack, 1981). Equilibrium with the reduced loading is expected in 1984 (Daley et al., 1981).

Changes in phytoplankton abundance and species composition paralleled the nutrient changes. Biomass levels dropped from more than 3000 mg/m^3 (when phosphorus concentrations were highest) to less than 1000 mg/m^3 by 1977 (i.e. 2 years after 90% P reduction had been achieved). Peak and mean chlorophyll levels averaged for 3 stations dropped from highs of 23 to 6 and > 4 to 2.6, respectively, between 1974 and 1978 (Nordin, pers. commun.). Another source (Daley et al., 1981) summarized and documented the trend of decreasing chlorophyll levels as follows:

Period	Average Annual Chlorophyll	
	south arm:	mid-lake
1966-1968	3.9	2.9
1972-1974	2.6	2.3
1976-1978	1.7	2.0

Throughout the period of increasing then decreasing biomass, the ratio of nutrients changed. Nitrogen levels remained at about 200 mg/m^3 while P fluctuations caused N:P ratios to strike a low of < 1 at the time of maximum P concentration. At this point the phytoplankton population was most likely N-limited and the blue-green nitrogen fixer *Anabaena* was particularly conspicuous (cf. Table VIII 5). A heavy bloom of it in August 1973 was alarming enough to qualify as a front page news item of the Vancouver Sun. However, it should be noted that a low N:P ratio is not the only condition controlling the particular species of blue-greens present. In the year preceding the *Anabaena* bloom, the non-nitrogen fixer *Oscillatoria* accompanied *Anabaena* as a major component of the summer and fall phytoplankton, yet the N:P ratio was as low as that during the *Anabaena* bloom. As phosphorus removal went into effect and lake concentrations declined, the blue-green populations declined by 50% and diatoms (especially *Stephanodiscus astraea*) by about 25%.

When Kootenay Lake reaches equilibrium with its reduced P load, by around 1984 (provided present residence time of 1.8 yrs. and nutrient sources remain the same), the expected N:P ratio is 25 and nuisance blooms of blue-green algae are not expected to materialize. At that time, TP concentrations should be < 10 mg/m^3 and the anticipated summer mean chlorophyll level is about 1-2 mg/m^3. The total time for recovery of oligotrophic conditions will have been 10 years from the time when 90% of the Cominco phosphorus wastes were removed from effluents.

8.7 Conclusion for Recovery of Lakes

In all of the six cases presented here, the common variable has been a reduction in phosphorus loading. The lakes subjected to these decreased loadings and concentrations have responded, at least eventually, with proportional decreases in phytoplankton biomass with a general shift away from bloom-forming Cyanophyceae. In lakes which are thoroughly mixed at least once a year, recovery of oligotrophic conditions may be extremely rapid (e.g. Gravenhurst Bay, Little Otter Lake), whereas incompletely mixed lakes may lag behind in their biological response to loading reductions (e.g. Kootenay Lake). Lakes which have apparently responded little to culturally related phosphorus loading reductions are those which by historical reconstructions appear to have had poor trophic conditions before cultural influences existed (e.g. Lake Erie, the Qu'Appelle Lakes).

Insight into the present dynamics of a lake afforded us by the OECD relationships allows more realistic prediction of the consequences future alterations may have, but management of a lake must be tempered by some knowledge of its past and naturally assumed condition.

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Chapter 9. DISCUSSION AND CONCLUSIONS

The OECD Programme has been designed for cross-sectional comparison of lakes to provide management with simple tools to evaluate the nutrient reduction necessary, primarily phosphorus, to alleviate excessive eutrophication in lakes, and to bring lakes back to an acceptably mesotrophic or oligotrophic state.

As explained in the introduction, the scope of the Canadian Report was to test the applicability of the overall OECD results on a set of data which originally was not included in the data elaboration of the overall programme. With this, the following objectives were envisaged:

a) Clarify to what extent an as yet unspecified population of lakes exhibits statistical properties similar to those of the population of OECD lakes;

b) Clarify and assess the limits of transferability of the OECD results to specific lakes, or lake groups of similar nature;

c) Identify particular situations and conditions which need further evaluation.

The following are some of the points emerging from this study.

9.1 Applicability of the OECD Results

Generally speaking, the results of any model or methodology chosen to approach a subject can only be as good as the quality and consistency of the data available for the study. The elaboration of the

Canadian material has shown that not only are data availability and internal consistency not always present, but that the quality of data is sometimes questionable. Many of the study programmes appear to be conducted with inadequate criteria, often depending on resource availability and logistics, rather than on the object of the study and the particular limnological conditions of the lake examined. More often than expected, only partial information is available, often disconnected in time, which makes interpretation exeedingly difficult. In fairness to the Canadian effort, it must be said that similar observations have been made throughout the OECD study. Also, in no sense does it mean that not many excellent specific studies have been carried out in Canada. The point we wish to make is that we are still far from the level of consistency in methodology and approach which is required for comparative studies in eutrophication. Results, conclusions and recommendations of the present report have to be viewed in this light.

In regard to the specific question of how far the OECD results and relationships are transferable to a new set of data, the following can be said, referring first to some general statements, then to some of the limiting conditions, and later on considering some specific areas.

Our study shows that the population of Canadian lakes tested exhibits statistical behaviour which, in principle, is comparable to that of the OECD lake population. This is true for both the bi-variate regression trends between related variables (mean chlorophyll - mean phosphorus concentrations - standard flushing corrected inflow phosphorus concentrations - transparency, etc.), and the magnitude of the respective error pattern.

Some apparently deviating cases could be reduced to data uncertainty, part of which was eliminated through direct discussion with the data originators. However, cases have been identified which definitively must be considered as non-conforming with the overall OECD relationships, and for which transferability and applicability of the OECD results become questionable. Some of the limiting conditions, under which the OECD results may not be valid, or which require special attention, are listed in the following. This includes:

a) Lakes for which the ratio $z_{eu}/\bar{z}$ (euphotic zone depth/mean depth) is substantially greater than one. In general these are lakes having depths of less than 4 m under which conditions it is likely that the littoral production zone (macrophytes, periphytic and benthic algae) dominate the lake metabolism reducing the importance of pelagic algal production. Exceptionally, relatively shallow lakes show behaviour similar to OECD lakes.

b) Lakes having high hydraulic load (q_s > 50 m/y), flushing rates of more than twice/year (T(w) < 0.5 years), and to lakes having very irregular flushing regimes, either seasonally, or over consecutive years.

c) Lakes having high mineral turbidity. Such lakes normally show considerably less chlorophyll than expected from the OECD relationships. Though not to the same extent, humic substances seem to depress chlorophyll levels too, yet annual areal primary production in coloured lakes is generally lower than in less coloured lakes of comparable trophic conditions.

d) Lakes having an N/P ratio ≤ 5. In such lakes nitrogen, rather than phosphorus, will be the limiting factor, and OECD relationships should not be expected to hold. In regard to phosphorus, application of the OECD relationships becomes questionable if annual mean concentrations exceed 100 mg/m^3.

e) Lakes receiving a high fraction of phosphorus load in the form of apatite. Apatite phosphorus should be excluded from calculations of loads.

f) Lakes exhibiting substantial internal load. Such lakes should be excluded from simple application of the OECD relationships. Internal load normally occurs with anoxic hypolimnetic conditions but can also be due to groundwater influxes high in dissolved nutrients. Internal load as a nutrient source is likely to be important if measured lake concentrations are substantially higher than expected ones. Conversely, high negative descrepancy between measured and expected lake concentration is indicative of strong nutrient elimination processes taking place at the water-sediment interface.

g) Lakes not in dynamic equilibrium. This is the case when nutrient loads are either progressively increasing, or decreasing. For situations where internal load does not yet play a major role, the most likely equilibrium conditions can be predicted from OECD results, but behaviour during the transitional phase cannot be predicted.

9.2 Insufficiently Resolved Problems

9.2.1 Chlorophyll and Biomass. It has to be recognized that chlorophyll measurements as substitute for biomass are of but limited value. Statistics on the range over which the chlorophyll/phytoplankton biomass ratio (in terms of phytoplankton volume) can vary, are scarce, but the most frequent range is from 0.3 to 1.2%. However, considerably higher frequency ranges have been reported in the literature (cf. Nichols & Dillon, 1978, Tolstoy, 1979). This potential variability explains part of the high standard error (and therefore part of the prediction uncertainty) of the phosphorus-chlorophyll relationship. Chlorophyll measurements, whenever possible, should be supplemented by direct biomass measurements (species identification, phytoplankton counting and volumetric estimates, etc.).

9.2.2 Chlorophyll-Phosphorus Relationship. Scrutiny of the combined Canadian data indicates further that the standard OECD relationship between average chlorophyll and average lake phosphorus concentration may require revision. A majority of the Canadian data lie below 6 mg chl/m^3 and 20 mg P/m^3, respectively, and fall in a range which is underrepresented in the overall OECD study. Their scattering pattern seems to be better described by the b-line (which has a slope steeper than 1; cf. Figure IX1) which is in better agreement with the findings made by Sakamoto (1966), Dillon & Rigler (1974b), Smith and Shapiro (1981) a.o. Although the number of points, including all data (143), is probably insufficient to decide whether or not an approximate linear regression between chlorophyll and phosphorus provides the best descriptor of this

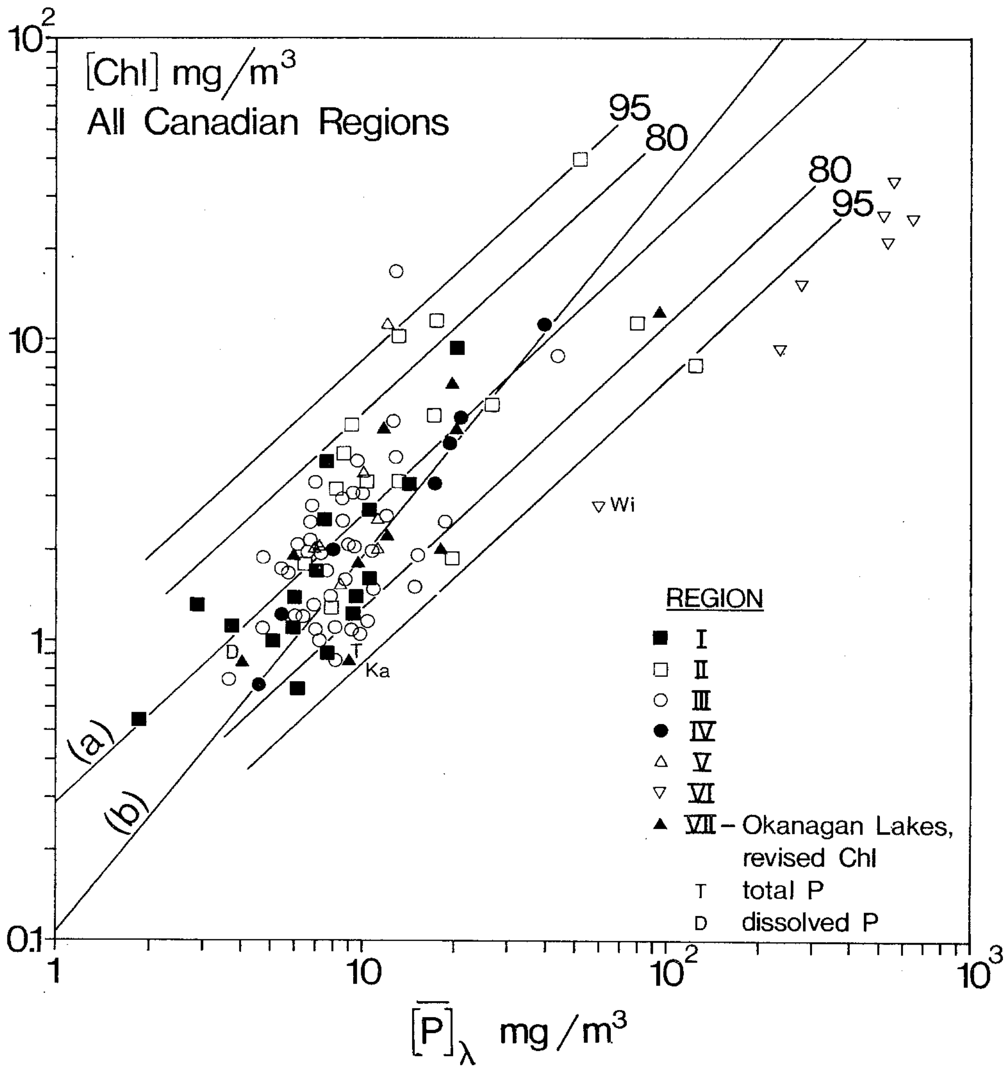

Figure IX 1. Annual mean chlorophyll a concentration in relation to annual mean phosphorus lake concentration relationship, the present stage of information seems to suggest the following generalized pattern:

Range 1) Below 10 mg P/m^3 chlorophyll increases more than linearly proportional to phosphorus concentration.

Range 2) Between 10 and 100 mg P/m^3 the prevailing relationship appears to be linear, but can equally well be described by a slightly more than linear dependency of chlorophyll on phosphorus

Range 3) Above 100 mg P/m^3 the assumption of a deterministic relationship between chlorophyll and phosphorus breaks down.

Also, if the N/P ratio falls below 5 to 7, the chlorophyll/phosphorus ratio is generally low. However, the dependency of the chlorophyll/phosphorus ratio on the N/P ratio may be more complex. This derives from the observation that in regard to the OECD lakes, there is a general tendency (though not a strict rule) for the annual mean chlorophyll values to lie above the a-regression line, if the N/P ratio exceeds 25, and vice versa. This would lead immediately to a multivariate relation pattern between chlorophyll and nutritional conditions, the two primary variables of which are phosphorus <u>and</u> nitrogen.

Other concurrent factors which have a bearing on the chlorophyll/phosphorus relationship are zooplankton grazing, competitive processes with macrophytes and benthic algal production, turbidity, mixing, flushing and other physical processes and factors. From the Canadian study it can be concluded that such interactive relationships are important but their effects on the chlorophyll/phosphorus relationship has not been possible to elucidate more.

9.2.3 Nutrient Loadings. The level of reliability in regard to loading estimates is still a matter of contention. The uncertainty in simple estimates is likely to be in the order of ± 35%, but in worse cases can be ± 50% and more. This makes correlation between loading and inlake conditions often tenuous, a fact which from the management point of view is particularly unsatisfactory.

Indirect estimates of nutrient loadings are only partial substitutes for estimates from direct measurements, but this latter can only be relied upon if the measurement programme has been adequate. The Qu'Appelle study shows clearly that in a system of high year to year variability even pluri-annual efforts do not guarantee the level of resolution which would allow drawing more consistent conclusions. Conversely, the problem connected with indirect estimates is illustrated with the Quebec study.

In regard to potential differences in lake response to concentrated direct discharges vis-à-vis dilute tributary loadings, little can be said from the Canadian data, but in regard to apatite phosphorus, the Kamloops case suggests that this fraction should be excluded in phosphorus loading estimates.

9.2.4 Predictability of inlake nutrient Concentrations from loadings. Considering the uncertainties connected with nutrient load estimates as discussed previously and taking account of other limiting conditions, both phosphorus and nitrogen concentrations can be reasonably predicted from the OECD standard relationships. This is not without some perplexity as the sole modifying parameter used in the OECD relationships is water residence time. A preliminary attempt has been made, using the Canadian material to formulate alternative phosphorus models which (besides average inflow concentration) include also mean depth and hydraulic load. These have been the only factors which could be evaluated in a more consistent fashion.

Stepwise regression analysis shows (cf. Table A 2.1) that inflow phosphorus concentrations can only explain 34% of the variability in inlake concentrations; including water residence time, hydraulic load and mean depth raises predictability to 50 to 60%. The major improvement is due to either water residence time or hydraulic load whereas mean depth seems to play a marginal role. Among the various alternative models, model No. 9 (cf. Table A 2.1) assumes the highest interest, insofar as it relates closest to the OECD standard relationship. Substituting $\bar{z}/q_s = \tau_w$, it can also be written as

$$[P]_\lambda/[P]_i = 1/(1 + 1{,}685\ \tau_w^{.504}/q_s^{.186})$$

which gives an output identical with the OECD standard flushing correction for a hydraulic load of 20 m/y.

Selective testing of these models shows good predictability in many cases, but indicates also that further improvement is necessary. For further discussion cf. Appendix 2.

9.2.5 Primary Production and Hypolimnetic Oxygen Depletion. The strongly deviating pattern of the ELA lakes relative to the OECD relationships points to the weakness of areal primary production as a useful trophic state indicator for comparative purposes. Equal areal production rates can be the result of a variety of different combinations of optical, temperature, mixing and biomass conditions (e.g. a condition sufficient for areal primary production to remain unaltered would already be that the ratio "biomass (chlorophyll)/basic water extinction coefficient" remains constant). Volumetric instead of areal primary production may probably be more meaningful. Nonetheless, within lakes of similar characteristics (e.g. Great Lakes; ELA lakes) annual areal primary production is sufficiently correlated with average chlorophyll, average phosphorus concentration and phosphorus loadings to make lakes within the group comparable, but the relation pattern breaks down between the groups.

In regard to hypolimnetic oxygen dynamics, areal depletion rates are still less meaningful than areal primary production rates, as comparability often breaks down even within lake groups (e.g. Great Lakes). In some lake groups (e.g. B.C. lakes) volumetric depletion rates are relatively consistently correlated with chlorophyll and phosphorus, but the pattern is not transferable to other situations. Improvement is possible, if more complex models are used which include parameters like transparency or extinction coefficients, the epi/hypolimnion volume ratio and hypolimnetic temperature. An attempt has been made to develop such models which is given in Appendix 3, and includes some Canadian lakes.

9.3 Management Implications

As has been discussed in the Synthesis Report, uncritical application of the OECD results can lead to gross evaluation errors (cf. also Appendix 1).

For management application, the OECD results can serve two different purposes.

9.3.1 Diagnostic Application. Information on a given lake can be tested against the standard behaviour of many lakes. With this it is possible to identify outliers, i.e. lakes which show particular properties which cannot be judged solely upon the basis of standard information. Such cases require additional information. Sometimes only specific processes studies may provide the answer.

However, our experience also shows that data outliers may often be due to inadequate basic information (e.g. insufficient number of observations used for calculating mean values), or to inadequate analytical methods used. Sometimes simple calculation errors have been found to be the reason for apparent outliers. Therefore, prior to concluding that non-conforming data points are indicative for non-standard situation, it is important to re-examine the data base.

9.3.2 Predictive Application. In using the OECD results, it should be borne in mind that the loading-trophic response relationships developed for the most part refer to equilibrium conditions, i.e. the most likely response to a certain loading regime. This response pattern is relatively consistent for oligo- to mesotrophic lakes, whereas the response pattern of eutrophic and hypertrophic lakes may be considerably more variable. As pointed out, the response in eutrophic lakes depends on the N/P ratio (limitation by nitrogen), the biological self-shading effect, and other biological, and, at times, physical parameters. In addition, with low hypolimnetic oxygen conditions, internal loading normally increases, and may become as important as external loading.

Within the limits of this applicability, it is possible to estimate from the OECD relationships the levels of nutrient (primarily phosphorus) reduction necessary to revert a lake from an unacceptable eutrophic condition to a more acceptable meso- or oligotrophic situation. Such predictive use of the OECD relationships are based on the assumption of long-term equilibrium conditions, but cannot necessarily give information of how quickly the systems will respond to external load reductions. Further it is possible to estimate from the expected equilibrium conditions roughly the level of expected phytobiomass in terms of chlorophyll. For this, the OECD experience - which includes the Canadian data - shows that in 80% of the cases the annual average $[chl]/[\bar{P}]_{\lambda}$ ratio remains below .375, and in 90% below .500. Only exceptionally, the ratio remains above .80 (cf. Table IX 1). Mean summer chlorophyll concentrations normally range between 1.5 to 2 of the expected annual mean, and peak values are 3 times the annual mean on average. However, exceptional peaks may be 4 to 5 times the annual mean.

The objective for nutrient reduction to be achieved depends on what level of annual mean biomass is considered acceptable relative to the intended water use and related economics. In certain cases, it may not be possible to attain with practical means the nutrient reduction level

Table IX 1. Frequency distribution of [chl]/[P] ratios in Canadian and OECD lakes.

Class midpoint	.0625	.1875	.3125	.4375	.5625	.6875	. ~ .85
Class range	0.000 to .125	.125 to .250	.250 to .375	.375 to .500	.500 to .625	.625 to .750	.750 to 1.0
Frequency							
Canadian lakes (69 = 100%)	15 (21.7)	36 (52.2)	9 (13.0)	5 (7.3)	2 (2.9)	0 (0.0)	2 (2.9)
OECD lakes (78 = 100%)	14 (18.0)	21 (26.9)	21 (26.9)	10 (12.8)	9 (11.5)	3 (3.9)	0 (0.0)
All lakes (147 = 100%)	29 (19.7)	57 (38.8)	30 (20.4)	15 (10.2)	11 (7.5)	3 (2.0)	2 (1.4)

Mean ± St. Dev.

Canadian Lakes: .225 ± .161
OECD Lakes : .293 ± .173
t = 2.42; P = .05

All Lakes : .261 ± .170

required. This then would indicate that additional measures to reduce the undesirable effects of eutrophication should be taken into consideration, either as supplementary, or as substitutive measures. Options of this kind are discussed in the Synthesis Report. However, it is important to evaluate first the nutrient reduction requirements, and proceed with the nutrient reduction option whenever the circumstances are in favour of it.

In cases where the need for supplementary measures may arise, the OECD relationships are insufficient to provide the basis for the most efficient choice, or combination of choices. In such cases, it is warranted to proceed with more complex dynamic models which, however, have to be tuned to the specific case situations.

9.4 Recommendations

9.4.1 Guidelines for Data Elaboration and Testing Against Standard Correlations. Apart from the limiting conditions and uncertainties discussed in the previous chapter, the following positive recommendations are made in regard to proper data elaboration and interpretation.

Nutrient loading - inlake concentration relationship. The basis for any consideration in regard to this relationship should be the proper evaluation of the $[M]_{\lambda}/[M]_{i}$ ratio (mean lake concentration/mean inflow concentration ratio), independent of any model consideration. Theoretically, this ratio is an expression for the residence time of substance M (phosphorus or nitrogen, etc.) relative to the water residence time, and can assume values ≤ 1, but can exceptionally be also > 1. If the ratio is close to 1, then the substance is likely to behave like a conservative element in terms of mass balance properties. If the ratio is $\lessdot 1$, then part of the substance is retained in the lake due to sedimentation, but the ratio may also be $\gtrdot 1$ in cases of increasing load. Conversely, ratios $\geqq 1$ may result when internal loading dominates the system, or the lake is in a washout phase after load has been decreased. It is important in each situation to evaluate the meaning of the $[M]_\lambda/[M]_\iota$ ratio properly prior to proceeding with other interpretation.

In regard to average lake concentrations, it is recommended to estimate these from analyses over a full year cycle whenever possible; in lakes having longer water residence times, average lake concentrations are close to spring overturn concentration. In irregularly flushed lakes, spring concentrations and yearly averages may be substantially different.

Average inflow concentrations are normally estimated from appropriate measurement programmes of all the tributaries and direct inflows. Correct stream gauging has been found to be important; otherwise huge errors may result. It is also advisable to check measured loadings against indirect estimates, and vice versa.

Complex lakes should be broken down appropriately in sub-basins, and inter-basin transfer of nutrients should be taken into account in estimating loads of the sub-basins.

If internal load appears to be important (e.g. as judged from the $[P]_\lambda/[P]_\iota$ ratio), best estimates of their relative magnitude should be attempted. E.g. the magnitude of internal load can be estimated from short-term (bi-weekly, or monthly) consecutive nutrient budget calculations taking into account storage changes in the lake.

Chlorophyll-Nutrient Relationship. Yearly average chlorophyll estimates should be restricted to the euphotic zone and evaluated against yearly average of nutrient concentrations rather than against spring overturn concentrations. This is particularly advisable for lakes with irregular flushing regimes, or lakes which receive their nutrient load mainly during the growing season.

If only growing season chlorophyll averages are available, then yearly averages will be lower by a factor of 1.5 to 2. This has to be taken into account in comparing chlorophyll-nutrient relationships with OECD standard relations. It is also important to evaluate peak chlorophyll values against mineral nitrogen and phosphorus at the time of the peak to evaluate if nitrogen or phosphorus is to be considered as growth limiting. However, over the full growing cycle, the system controlling nutrients may be different from that limiting peak chlorophyll. Finally, in judging chlorophyll nutrient relationships, physical factors, in particular the ratio euphotic zone/mixing depth, and presence or absence of mineral turbidity, should be taken into account, as well as potential grazing pressures by zooplankton.

Use of Standard Correlation. The most important recommendation along this line is that a given lake situation should be evaluated in terms of the total correlation pattern, and not solely on single correlations in isolation. From the total pattern it can be deduced if a single non-conforming situation is due to particular conditions, or rather to data errors. Also, in using the OECD standard relations, or their modifications for predictive purposes, judgement should be based on the total pattern and not solely on single factor prediction. Potential error margins should be taken into account, particularly if there is reason to believe that data points may be uncertain. In regard to nutrient loadings, and prediction of required reduction, it should be borne in mind that most nutrient estimates have a basic uncertainty margin of some ± 35%, and in some cases may even be higher.

9.4.2 Planning and Implementation of New and Follow-up Studies. The many problems encountered in synthesizing the available information also leads to some suggestions of which areas are weak and need improvement, or better attention in future studies. In this context it is important to keep in mind that limnological studies often serve two purposes, the first to investigate a particular situation, to pursue the analysis of a specific subject, etc., the second to provide information which could be used by other researchers for comparative purposes. The OECD study has been relying particularly heavily on the latter. Therefore, the few hints given in the following are direct to this second line of thought.

Inlake Nutrient Conditions and Nutrient Loading Studies. For comparative purposes and correct interpretation, information on inlake nutrient concentration (apart from appropriate breakdown in operational categories: total, dissolved, particulate) should include mean annual concentrations, winter-spring average concentrations, summer epi- and hypolimnetic average concentrations, and epilimnetic minimum of nitrogen and phosphorus components and N/P ratio at peak production and at the end of summer stagnation.

For an improved understanding of the nutrient dynamics, study programmes should be conducted in such a way that allows short term (two weeks to monthly) budgets to be made. Little in this sense is presently available. This should also lead to a better appreciation of internal load as a nutrient source vis-a-vis the external loads.

For highly flushed, or irregularly flushed lakes, one or two year survey programmes are almost meaningless. Study programmes should be desiged in accordance with the expected hydrological regime, and have a temporal coverage and time resolution to include potentially extreme situations (cf. e.g. Chapter VI).

Physical Conditions. Information on physical conditions relative to their effects on nutrient and biodynamics have often been found to be scant. Appropriate studies should include information on optical conditions (basic water absorption, composite absorption), mineral turbidity, humic substance content, information on the variability of the z_{eu}/z_{mix} ratio (euphotic zone/mixing depth), information on mixing cycles, length of summer stratification, ice coverage, etc., and information on epi-hypolimnetic relationships (entrainment), etc.

Hypolimnetic Conditions. Particularly unsatisfactory is the information available in regard to hypolimnetic dynamics, including oxygen depletion, hypolimnetic entrainment, sediment-water exchange phenomena, hypolimnetic redox conditions and build-up of reduced chemical species (Fe^{II}, MN^{II}, H_2S, CH_4, etc.). Such information - which presently is often lacking - should be given in a form that permits quantification of the process as well as providing the basis which would permit estimation of the total hypolimnetic oxygen demand (or corresponding chemical expressions).

Tropho-dynamic Interactions. The bearing of tropho-dynamic interactions on the trophic levels, in particular, their effects on phytoplankton standing crop, is poorly understood. Information on zooplankton standing crop and dynamics often is scarce, and the potential effects on fish is only known from species studies.

Historical Trends. Although historical evaluations and trends can only be reconstructed in part from sediment record, and therefore remains qualitative, provision should be made to allow future trends to be studied over a prolonged period of time, at least for a few well selected cases. The few examples available from the literature show how important long-term studies are for understanding species and population dynamics, and interactive tropho-dynamic aspects. Such understanding becomes relevant to eutrophication cases when external nutrient manipulation is not, or only partially possible, and necessary for improving the prediction capability of eutrophication models in general.

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Appendix 1

BACKGROUND AND SUMMARY RESULTS OF THE OECD COOPERATIVE PROGRAMME ON EUTROPHICATION

prepared by

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FIGURES

TABLES

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R. A. Vollenweider

The Problem of Eutrophication

Early in the 1960 decade, it became obvious that a large number of lakes and reservoirs were rapidly changing their trophic characteristics due to the addition of plant nutrients originating largely from human activities. The main nutrient sources identified were municipal and industrial wastewater and agricultural and urban runoffs.

Eutrophication is the response to this over-enrichment by nutrients (primarily phosphorus and nitrogen) and can occur under natural or man-made conditions. "Man-made" eutrophication, in the absence of control measures, proceeds at an accelerated rate compared to the natural phenomenon. A recent survey (cf. Vollenweider 1979) has shown that eutrophication is one of the main forms of water pollution reported in countries throughout the world. The resultant increase in fertility in in affected lakes, reservoirs, slow-flowing rivers and certain coastal waters causes symptoms such as algal blooms, heavy growth of certain rooted aquatic plants, algal mats, deoxygenation and, in some cases, unpleasant odour, which often affects most of the vital uses of the water, such as water supply, fisheries, recreation or aesthetics. In short, man-made eutrophication of inland bodies of water becomes synonymous with the deterioration of water quality and as such frequently causes considerable extra economic costs.

Man-made accelerated eutrophication can, in principle, be reversed by the elimination or reduction of the nutrient supply from such as municipal and industrial wastewaters, agricultural wastes and fertilizers. In most cases, however, it is not possible to eliminate all sources of nutrient supply. Thus, it is important to understand the qualitative and quantitative relationships which exist between nutrient supply and the degree of eutrophication in order to be able to develop sound lake management strategies to control eutrophication at minimum costs.

History of OECD Activities in Eutrophication

In 1967 a group of experts under the chairmanship of Professor O. Jaag (EAWAG, Zürich) recommended to the OECD that a comprehensive survey be made of the existing literature on eutrophication processes. This led to the publication of a report, "Scientific Fundamentals of the Eutrophication of Lakes and Flowing Waters with Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication" by Vollenweider (1968). This report introduced the concept of nutrient loading and lake response but also stressed the inadequacy of limnological data for broad generalizations and for producing precise guidelines for eutrophication control.

Further, a symposium on "Eutrophication in Large Lakes and Impoundments" was held in Uppsala, Sweden, and the resulting report was published by the OECD in 1970.

In spite of the advances achieved in eutrophication control, many basic questions concerning eutrophication remained unanswered, and it became obvious that a broader limnological data base was required for inter-comparison between bodies of water and assessment of the status of lake eutrophication. The nutrient loading concept and the related concept of loading tolerance had been consolidated and accepted by a large segment of the international scientific community, but controversies whether carbon and other growth factors rather than phosphorus or nitrogen limit algal growth in lakes continued for some time.

In 1971 the OECD established a Steering Group on Eutrophication and in February, 1973, approved and adopted an "Agreed Programme on Evaluation of Eutrophication Control" and charged the Steering Group on Eutrophication Control with the responsibility for developing and co-ordinating the agreed programme, bringing into account its effectiveness, cost and feasibility. Four _ad hoc_ expert groups carried out the programme:

1. Expert Group on Detergents (published 1973);
2. Expert Group on Impact of Fertilizers and Agricultural Waste Products on the Quality of Waters (published 1973);
3. Expert Group on Wastewater Treatment Processes for Phosphorus and Nitrogen Removal (published 1974);
4. Planning Group on Measurements and Monitoring (published 1973).

The three expert groups and the planning group completed their reports in 1972. The planning report "Summary Report of the Agreed Monitoring Projects on Eutrophication of Waters" (published 1973) gave a common system of agreed measurements, guidelines on background data and comments on existing methods of sampling. It also outlined the basis for an international programme of measurements and monitoring of waters being undertaken by interested OECD Member countries. This programme came to a close in 1980 and has resulted in a Synthesis Report and four Regional Reports already being published. A fifth Test Case Report is presently in its final stage.

The Conceptual Background

Scientifically speaking, eutrophication is but a special aspect of water productivity. Seen in this perspective, studies on eutrophication have to respond to the same conceptual references as productivity studies in general. Productivity is the expression of the external physiographic complexes of the system as a whole, as well as of its internal physico-chemical and biological dynamics. Accordingly, the trophic properties of bodies of water, lakes, estuaries, sea coasts or running waters, have to be considered as the resultant of a sequential nexus of geographic, geochemical, climatic, hydrological and other factors.

In applying this concept to eutrophication studies on lakes and reservoirs, the scheme expressed in Figure 1 proposes a quasi deductive procedure to derive the cause effect relationships which determine any observed specific limnological situation from the characteristics of the catchment system by progressing from the general properties of the system to the specific conditions of the water body considered. In the progression from level to level, the degree of freedom for the next level is narrowed down, i.e. the specific properties of the physico-geochemical complex at the top controls the hydrologic and qualitative properties and characteristics of the water deflux level, which in turn determines the limnological level in its connotation "productivity".

In order to bring this concept into perspective, at least one specific transfer compartment and two transfer functions have to be singled out:

A) The vegetation-soil complex acts as an intermediary between the physico-chemical complex and the water property level. Under natural conditions this compartment is practically the only source compartment in terms of nutrients, yet - due to man's intervention - has been substantially altered over the centuries. The historical and modern development in land use, urbanization and industrialization has had effects on both the size of this compartment in terms of nutrients potentially available, and on the transfer function, i.e. the amount of nutrients released per unit of time and unit of surface.

The transfer function from the basin to the receiving waters is expressed in terms of export coefficients (e.g. $kg/km^2.year$) for each source. Point sources*), in general, are expressed in terms of unit load,

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*) It is now customary to distinguish between diffused and point sources. However, such a distinction has primarily operational meaning: point sources, as opposed to diffused sources, in general, offer less difficulty for quantification, and at the same time are more amenable for technological control.

yet in principle, they can also be expressed in terms of export coefficients at the condition that their density distribution can be established. The specific values of these export coefficients vary considerably from situation to situation, depending on the general geographic, climatological, hydrological and other conditions, as well as on the specific land use, urban and industrial development, etc. Export coefficients for phosphorus vary from less than 5 kg/km^2.y to over 500 kg/km^2.y and for nitrogen from less than 50 kg/km^2.y to more than 3000 kg/km^2.y.

In spite of this large variability, it is at times possible for specific geographic regions to apply lump values as has been shown by Vollenweider (1968, 1978) for average European conditions, and by Rast and Lee (1978) for U.S. conditions. However, uncritical transfer of such coefficients to unknown regions can lead to gross error.

B) The nutrient loading concept, as distinct from the transfer function discussed above, refers to the receiving water body and in most general form means the intensity of supply to a given body of water of any chemical factor necessary for plant growth; in our context, however, its meaning has been restricted to nitrogen and phosphorus.

The theoretical limnology for decades has ignored this aspect, or at least neglected it, despite the early announcement made by e.g. Naumann (1932), Åberg and Rodhe (1942), Ohle (1955) a.o. Accelerated eutrophication of bodies of water over the last two or three decades has brought this problem into the open. The nutrient loading concept as defined here implies the connotation of a quantifiable property called "external load" which establishes the functional relationship between the basin and the trophic conditions of the receiving waterbody, and as such, is fundamental to the understanding of the total system.

From the methodological point of view, the quantification of the load-response relationship remains not without certain perplexities. Part of these relate to the question regarding the most appropriate way to express the load. Advantages and disadvantages of various options (e.g. absolute total amounts, specific loading per unit of surface or volume over a selected time-space, average inflow concentrations, etc.) are still a matter of discussion.

More important, however, the loading-trophic reaction relationship cannot be dealt with adequately without due consideration being given also to the fate of the various load components of a given substance within the lake system itself. An improvement over consideration of sole totals could be achieved by distinguishing at least two principal components and corresponding pathways, i.e. one component which enters the internal cycle via an "autotrophic" pathway - and which becomes immediately available to primary producers, and a component which enters into the internal cycle via a "heterotrophic" pathway of a more refractory nature (cf. Figure 2). In part, this aspect relates also to the question of what fraction is, or is not biologically available. In practice, the analytical distinction of these components is only partially possible, yet in order to understand the full array of reactions of different bodies of water to a given (total) load, a clarification of this problem is not without importance. Also, in many cases the internal loading cannot be neglected, though in many lakes this internal component remains far below the importance of the external loading. The exact quantification and dependency on external load is not entirely solved as yet, although essential progress has been made (cf. e.g. Golterman 1980).

However, important in this context is the basic idea that the in-lake bioproduction and recycling machinery (to use a more engineeristic analogy) is fed and driven by the external loading, and maintains itself depending on this external load in a repetitive cyclic steady state as long as no (unidirectional) alteration of the external supply occurs. On the other hand, any (unidirectional) change in the supply function will have as a consequence an alteration of the internal responses of the machinery speeding up or decreasing the velocities of exchange between the compartments, and correspondingly producing a change in size of each compartment.

In pursuing this concept, the question is posed as to how far we can go at present to quantify the postulated relationships. This implies the necessity to establish and to elucidate the function of those parameters which primarily govern the relationship between the external load and the reaction of the body of water. From an applied point of view, such an understanding of the various relationships - always expressed in quantitative terms - would provide the scientific basis to develop criteria to manage the system; in particular, it would provide the basis to estimate the nutrient supply reduction required for lakes which in terms of preset water quality standards, appear to be overfertilized.

The far reaching, practical, i.e. economical, implications of solving these questions have been recognized by OECD and have provided the motivations for the OECD Cooperative Programme on Eutrophication which is the main theme of the following exposé.

Approaches Taken in the OECD Cooperative Programme on Eutrophication

We realize that we have oversimplified the problem considerably, yet this has been done with the intention of bringing the problem into focus. Also, in speaking further on about the OECD Programme, its scope, outcome and results, much oversimplification will be necessary, which does not exclude that the single collaborators, as well as the members of the Steering Committee, are well aware of the many difficulties arising in specific cases in applying a simplified approach.

In order to introduce the rationale for the OECD Cooperative Programme on Eutrophication, it is necessary to recall the situation regarding the level of understanding of the nutrient load-trophic reaction relationship, particularly in regard to nitrogen and phosphorus, some fifteen years ago. At that time, only a few reliable data on nitrogen and phosphorus loadings existed in the whole applied and theoretical limnological literature, and much of the data were no more than crude estimates which hardly permitted any founded generalization. Nonetheless, when the first author proposed in 1968 (cf. Vollenweider, OECD Technical Report) that in principle it was feasible to distinguish between "acceptable" and "excessive" loading, this proposal was welcomed in the scientific community, and immediately had substantial influence on practical decisions as well as in stimulating a plethora of follow-up research.

It rapidly became clear through a number of meetings organized by OECD that only through international cooperation would it become possible to arrive at a sufficiently large amount of comparable data to derive valid quantitative relationships. Therefore, in about 1972 it was decided to launch a major cooperative programme involving a majority of the OECD member countries. Some 18 countries, including more than 50 research centers covering between 100 and 200 lakes, have adhered to this programme. It was conceived to tap into and make use of ongoing research but also to initiate new research. Accordingly, a full uniformity in approach could not expect to be achieved, yet this shortcoming was hoped to be counterbalanced by the large variety of individual lake situations covered by the programme.

How did we develop this programme? It was quite clear from the beginning that the focus would be on nitrogen and phosphorus, but that this aspect would have to be related to the particular geographic and limnological conditions of each lake individually studied. Further, it was necessary to develop a common language, to screen particular techniques and methods as to suitability and reliability, and to select those study items which appeared to be both pertinent to the success of the programme, and logistically feasible, i.e. accessible for most cooperating centres involved. With evolving results, serious thought had to be given to data elaboration and exploration of the most useful way to correlate them.

In order to account for geographic variability, as well as for logistic considerations, we organized the programme into four main projects:

1. An Alpine Project
2. A Northern Project
3. A Reservoir and Shallow Lake Project
4. A lump project for North America

Each project was headed by a regional coordination centre, regional chairmen, plus some consultants forming a Technical Bureau for overall coordination. The first author has had the pleasure of chairing this committee over the last few years, and wishes to acknowledge the the cooperation he enjoyed from his colleagues, particularly Drs. Ambühl, Bernhardt, Forsberg, Golterman, Lee, Löffler, Maloney, Ravera and others, for steering this programme, and the consultants responsible for synthesizing the material into report form (Drs. Kerekes, Clasen, Fricker, Lønholt and Ryding).

Table 1 provides an illustration of the kind of approach taken in developing a common language to identify parameters to be measured, or thought to be necessary to collate the information gathered into a consistent picture. It was understood that not necessarily all parameters would be measured in each individual case; some have been singled out as absolutely essential, whereas others have been left to the choice of the individual centres, in accordance with their capabilities and expertise.

In contrast to an approach of studying but a few examples only in depth, the chosen approach permits covering a wide spectrum of individual cases in an extensive way. Hence, our attention was not primarily focussed on specific mechanisms, but on information that is amenable to statistical analysis. We wish to state this explicitly because at times the philosophy of the programme has been misunderstood, particularly by those who expected a kind of material which could be used for dynamic modelling. From the very beginning, elaboration of the data at the basis of correlation and other comparative techniques thought to be meaningful, was envisaged. From this we expected to determine the cause-effect relationship in the sense of what we may call "statistical behaviour", examples of which shall be given below.

What kind of results have we obtained from this programme? The programme has covered a wide variety of limnological situations, including almost every type of lake and impoundment of the temperate region, a few subtropical lakes and reservoirs, as well as some estuarine situations.

Although the majority of lakes well studied fall into the meso- to eutrophic categories, a sufficient number of lakes representing oligo- and ultra-oligotrophic types have been included.

It is not the place here to discuss the whole array of results, conclusions and implications for practical management. These aspects are covered in the regional reports, in site specific reports and scientific papers, and in the final synthesis report. The integration of the available data has proven to be a worthwhile though difficult task. Such difficulties refer to both conceptual aspects as well as to straightforward problems with data screening, selection and appropriate interpretation. In many cases, it is not immediately available whether a data point represents a particular situation, a general uncertainty, or some unintentional mistake. Frequency of sampling, e.g., is a major factor in determining the reliability of a reported average, but also season to season or year to year variability of the investigated system itself. Superimposed on these problems are problems connected with calculation procedures; the choice of which of the various alternatives to use often remains a matter of taste rather than a matter of objective judgement.

As an example, loading figures represent a key parameter in the whole study, yet little do we know about the inherentuncertainty of any specific value reported. In our judgement, it is unlikely that individual year specific loading figures in most cases are better than ± 25%. The natural year to year variability in loadings, in addition, is found to be in the same order of magnitude (in some cases also considerably higher), so that representative loading estimates have a built-in uncertainty of at least ± 35%. In-lake parameters such as biomass, chlorophyll, nutrient parameters, etc., are affected by similar uncertainties that have to be taken into account in data interpretation and correlation.

In its final output, the OECD Programme has paid attention to the following aspects:

a) the qualitative assessment of the trophic state of bodies of water in terms of a few easily measurable parameters;
b) the dependence of this state on nutritional conditions and nutrient load;
c) translation of these results to the needs of eutrophication control for management.

One of the recurring problems we have run into during the study was the question of how to relate the classical trophic terminology - which is qualitative in nature - with the quantitative information provided in regard to selected parameters. In other words, the question arose of how far it is possible to quantify, in an objective way, the qualitatively defined trophic categories.*)

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*) The pressing need for clarification in this context becomes apparent, if one recalls such examples as Lake Erie, which in the early sixties was "dead", then became "eutrophic", and finally is now considered, at least in regard to the main body of the lake, as mesotrophic.

Though apparently of academic interest, this question is not without meaning, in two ways. Firstly, it relates to what has previously been stated relative to the need of a common language between limnologists themselves. Secondly, it relates to how the limnological terminology applies to practical management. From the practical point of view, there is no unequivocal relationship between the main trophic limnological categories and water usage. The relationship depends on specific use requirements. A categorization of bodies of water for fishery purposes need not necessarily correspond to the one for recreational purposes or to the one for domestic water supply, and none can entirely be matched with the limnological categories. Generally speaking, however, one can say that, proceeding from oligotrophy to eutrophy, multiple use of any water progressively becomes adversely affected with increasing trophy (cf. Table 2). Given this inherent ambiguity, therefore, it is important to attach quantitative meaning to the limnologically defined categories as the basic reference independent of their specific application. The OECD study has led to some interesting and not necessarily anticipated results.

The quantitative information given by the single contributors, together with their subjective judgement, were combined into a 4 x 5 matrix and for each block, mean and standard deviations have been calculated. A log-transformation of the original data was found to be necessary; the results are given in Table 3.

Clearly, most investigators consider a lake to be oligotrophic when the annual mean total phosphorus concentration is < 10 mg P/m^3. It is noteworthy, however, that a few lakes with < 10 mg P/m^3 were classified as either mesotrophic or eutrophic. Careful examination of the data revealed that in these cases the lakes have received an increased nutrient load in recent years, and as a consequence, have undergone some perturbation and change in trophic response. This may be in the form of a noticeable growth of attached filamentous algae along the shore near nutrient inflows, often accompanied by the appearance of a nuisance algae not observed before, however, without producing fundamental repercussions in the overall metabolism of the lake, noticeably its hypolimnetic oxygen conditions. At the other extreme, lakes with an annual total phosphorus concentration > 30 mg P/m^3 and as high as 80 mg P/m^3 were assessed as mesotrophic by some investigators. In these cases, a variety of reasons, e.g. short water residence time or high turbidity, a high rate of grazing by zooplankton in the absence of fish, a.o., prevented the development of a high standing stock of phytoplankton, and hence, the lakes did not exhibit eutrophic characteristics.

In regard to nitrogen, no consistent picture evolved. In particular, it was impossible to separate oligotrophic from mesotrophic lakes, although as a general rule, lakes of more eutrophic characteristics tend to have higher nitrogen concentrations.

A somewhat clearer delineation of trophic categories resulted, however, when allocation was based on chlorophyll a concentrations. In general, lakes were assessed as oligotrophic, mesotrophic or eutrophic when annual mean chlorophyll a con centrations were < 2.5 to 10, or > 10 mg chl a/m^3, respectively. No lake was classified as eutrophic with an annual mean concentration of chlorophyll a < 2 mg/m^3. In regard to "worst case" situations, i.e. peak chlorophyll values, lakes are considered to be oligotrophic, mesotrophic and eutrophic when annual peak chlorophyll a concentrations are around 5, 16 and > 25 mg/m^3, respectively.

What emerged from the assessment of all information available, however, led to the conclusion that there is no possibility of defining strict boundary values between trophic categories. Whilst the progression from oligo- to eutrophy is a gliding one - as has been stressed many times in the past - any one combination of trophic factors, in terms of trophic category allocation, can only be used in a probabilistic sense. The probability distribution for the two single factors, yearly average phosphorus and chlorophyll for the three main categories (oligo, meso, eutrophy) plus the two boundary categories (ultra-oligo and hypertrophic) is exemplified in Figures 3 and 4. E.g. the probability of classification of a body of water having a total phosphorus concentration of 10 mg/m^3, respectively, would be as follows:

	Phosphorus	Chlorophyll
ultra-oligotrophic	10%	6%
oligotrophic	63%	49%
mesotrophic	26%	42%
eutrophic	1%	3%
hypertrophic	0%	0%

In judgement terms, then, such a water body is best classified as oligotrophic with a certain tendency toward mesotrophy. However, exceptionally, such a body of water may have excellent ultra-oligotrophic characteristics, or to the contrary, may show signs of grave deterioration, as e.g. is the case with Lake Mjøsa.

Evidently, this way of looking at trophic categorization has considerable management implication. If, in a given case (e.g. a drinking water reservoir), it is important that certain water quality characteristics are maintained, then the management objectives must be set at some level slightly lower than would be required for maintaining average conditions.

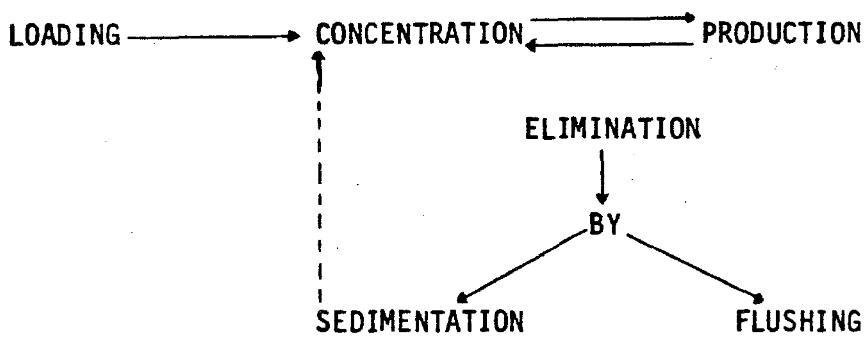
In order to manage a lake with a certain objective in mind, we need knowledge of i) which of the nutritional factors controls the system and ii) what the relationship is between nutrient loading and the trophic reaction of the lake.

In regard to the first aspect, one of the primary results evolving from the data collected is confirmation that in at least 80% of the cases studied phosphorus was found to be the production-controlling factor; some cases remained inconclusive and the rest were identified as nitrogen limited, or controlled by some other factor.

In regard to the second aspect, the relationships between trophic characteristics, such as nutrient concentrations, mean annual chlorophyll, peak chlorophyll and loading, have been shown to be amenable to quantification; in accordance with the programme objectives, these relationships are statistically not deterministically defined. Hence, if these relationships are to be used for prediction, the built-in uncertainty has to be taken appropriately into account.

Restricting the discussion to phosphorus, the results are based on the following methodology:

It has been obvious for some time now that simple relationship between areal or volumetric loading and lake phosphorus levels cannot be established without consideration of sedimentation and flushing (cf. e.g. Vollenweider 1969, 1975, 1976; Dillon 1975, Kerekes 1975). Basically, this relationship has to be thought of as follows:



In the most simple way, this scheme can be expressed mathematically as

$$\frac{d[\overline{P}]_{\lambda}}{dt} = \left(\frac{1}{\tau_w}\right) [\overline{P}]_j - \left(\frac{1}{\tau_p}\right) [\overline{P}]_{\lambda} \qquad (1)$$

where

$\overline{[P]}_\lambda$ = average total lake concentration (which includes both dissolved and particulate phosphorus components)

$\overline{[P]}_J$ = average inflow concentration of total phosphorus

$\overline{\tau_p}$ = average residence time of phosphorus

$\overline{\tau_w}$ = average residence time of water

The righthand terms represent the average rate of supply to and the average rate of loss of total phosphorus from the lake, respectively, and the lefthand terms the corresponding temporal variations of the average lake concentration. Note that in this formulation no specific assumptions are made as to the mechanism of loss.

Several possibilities are open to deal with the above equation, yet in principle, it reduces to evaluating statistically the quotient $\overline{\tau_p}/\overline{\tau_w}$ as a function of parameters controlling the system such as mean depth, epi-hypolimnion ratio, hydraulic load, length of stratification, etc., assuming steady state conditions.*)

From the various attempts made to analyse these relationships, the fact evolved that mean depth and hydraulic load are the most important factors, and that $\overline{\tau_p}/\overline{\tau_w}$ can be approximated by a function of the form

$$\overline{\tau_p}/\overline{\tau_w} = \overline{[P]}_\lambda/\overline{[P]}_J \simeq \frac{1}{1 + a . q_s{}^b \, \bar{z}^c} \qquad (2)$$

*) The term "steady state" is referred to in this context as "repetitive state over time" for which $\Sigma \pm d\overline{[P]}/dt = 0$. Time resolution is 1 year.

Approximative values for *a*, *b* and *c* were found to be 1, -.5 and +.5, so that (2) reduces to

$$\overline{\tau_p}/\overline{\tau_w} = [\overline{P}]_\lambda/[\overline{P}]_j \approx \frac{1}{1 + \sqrt{\bar{z}/q_s}} = \frac{1}{1 + \sqrt{\tau_w}} \tag{2a}$$

(Vollenweider 1976). These findings correspond to results of similar approaches made by Larsen and Mercier (1975), Dillon (1974), Kirchner and Dillon (1975), Chapra (1975), Chapra and Tarapchak (1976), Reckhow (1978), a.o. which all are variations of the same theme.

Accordingly, mean lake phosphorus should be predictable from load, in principle, by

$$[\overline{P}]_\lambda = [\overline{P}]_j/(1 + \sqrt{\tau_w}) \tag{2b}$$

Figure 5 shows that this indeed is the case yet (2b) slightly underestimates concentrations at low levels, and overestimates concentrations at higher levels.

All these formulations, and their variations, contain the underlying assumption that lakes can be treated as mixed reactors in steady state. This is not true for most lakes. It is therefore surprising that simplified relationships of this sort provide a workable basis, which in principle means that a large spectrum of lakes (governed by phosphorus) behave statistically in a similar way. <u>The relationships derived describe the average statistical relation pattern of lakes between phosphorus load and phosphorus concentration</u>.

Used as a diagnostic criterion, these relationships also provide a tool to identify "outliers". The term "outlier", as used here, refers to both statistical and functional variability. Indeed, outliers from the rule may indicate simple data uncertainty as Rast and Lee (1978) have shown to be the case for lakes for which the load has been either under- or over-estimated. However, outlier lakes have also been identified which behave functionally differently; either the assumption of steady state does not apply, their sedimentation quota is above or below normal, or an internal or external disturbance of the system exists. Examples for each possibility could be listed, yet more importantly, in most cases it was possible to identify the reason for deviation.

This experience shows that it would be wrong to discard equation (1) simply because a given data set would not fit it. Strong deviations from this relationship can be used as a diagnostic indication for a particular situation which requires further attention. Conversely, it would also be wrong to blindly apply this relationship for predictive purposes, regardless of special limnological conditions.

The next step in the sequence was to establish the relationship between chlorophyll (yearly and peak values) and nutrient concentrations. Without entering into detail for the present review, it may be said that, on average, the yearly mean chlorophyll concentration was found to be between 25 to 30% of the average total phosphorus (cf. Figure 6A). Peak chlorophyll values (which are of particular importance for practical considerations) on the other hand, resulted as roughly three times average chlorophyll, but exceptionally can be considerably higher (cf. Figure 6B).

Interestingly, chlorophyll apparently also resulted in being correlated to nitrogen in many cases, yet statistical discrimination tests have shown that this is primarily due to coupling of nitrogen with phosphorus. In particular cases, however, the dependence of bioproductivity on nitrogen, as well as on other factors, has been found to be unquestionable. The interaction between phosphorus and nitrogen has been identified as an area which requires further research.

In the light of what has been said thus far, a close relationship between phytoplankton biomass (as measured by chlorophyll) and phosphorus load can be expected. The findings are illustrated with Figures 7A and 7B. In regard to statistical variability, the same applies as said above regarding the relationship between phosphorus loading and concentration. However, it is to be stressed that chlorophyll is but a crude parameter to estimate biomass. Indeed, cases did come to light indicating that the biomass/chlorophyll ratio can vary by a factor of up to 3 and is therefore a major contributor to the scattering observed.

Nevertheless, the biomass (chlorophyll)/loading relationships are perhaps the most important results of the OECD study thus far. Within the range of the identified uncertainties, they permit estimation of the phosphorus reduction necessary to reduce eutrophication to any preset level of biomass. The main conclusion which one can draw from the OECD results is the fact that the production level of any given water body, in principle, is proportioned to its nutrient load, and therefore that load reduction will have effects proportional to the reduction achieved.

In the long run and with consideration that exceptions from this rule exist, it is desirable to base such judgements not solely on standing crop but also on related dynamic parameters. Unfortunately, the OECD study has not permitted convincing establishment of relationships between loading and dynamic parameters, such as primary production and hypolimnetic oxygen depletion rates, etc. This is due, in part at least, to the dearth of usable data points and in part to considerable difficulties in measuring such parameters uniformly. The problem of hypolimnetic oxygen conditions is further compounded by conceptual uncertainties (e.g. oxygen depletion rates versus apparent or potential oxygen deficit).

The following is a short account of the present state of the art.

The relationship between primary production and phosphorus deviates structurally from the chlorophyll - $[\overline{P}]$ relationship by its non-linearity. This is due to the self-shading effect of the biomass with increasing levels of productivity which can be dealt with by introducing a generalized primary production model. This model assumes that the annual primary production can be expressed with a hyperbolic function similar to that of a daily photosynthesis integral (cf. Vollenweider 1970) i.e.

$$\Sigma\, C\ (g/m^2.y) = K \cdot \frac{[\overline{chl}]}{\acute{\epsilon}_w + n\,[\overline{chl}]}$$

where [chl] is the average yearly chlorophyll concentration of the euphotic zone, ε_w' a characteristic average extinction coefficient (1/m) which includes turbidity, humic substances and other coloured substances, and η the specific vertical extinction coefficient per unit of chlorophyll.

In order to establish the relationship to nutritional conditions, the chlorophyll term in the above equation can be substituted by the corresponding relationships, and the remaining parameters calculated from measured data by least square techniques. Correspondingly, the hypolimnetic oxygen depletion rates should be predictable from primary production. This hypothesis further implies that the relationship between oxygen depletion rates (expressed as areal hypolimnetic oxygen depletion rates) and nutritional conditions, should parallel those for primary production.

Our preliminary results show that this is indeed the case. Yet, the much larger scattering of the data points also shows that factors other than those taken into account in our analysis are involved in determining primary production and hypolimnetic oxygen conditions. The higher uncertainty, e.g., in linking hypolimnetic oxygen depletion rates with loadings, as found in our study, depends undoubtedly on the complex interactions between the epilimnetic and hypolimnetic regime in each individual case. The underlying factors relate to specific lake morphology, length and type of thermal stratification, vertical entrainment and oxygen transfer, and interactions between sediments and overlying waters. It is evident that, in order to reduce the uncertainties, much additional work is required.

<u>How far did these preliminary results meet our expectations?</u>

Considering the large variety of lakes examined, and considering also the unavoidable inequality in the data collected, the results achieved to date probably exceed by far what could be expected from this programme. Admittedly, some of the correlations of factors thought to be interrelated, in part, were found to be poor, yet at least some of the more important correlations turned out to be highly significant (cf. relationship between phosphorus load and in-lake phosphorus concentrations, between this latter and chlorophyll, and between loading and chlorophyll).

Generally speaking, what has been achieved in terms of understanding lake behaviour, lies say, half way between the historic position that each lake is an entity which has to be understood on its own, and solely on its own, and an advanced but not yet attained level of insight which would make it possible to deduce the reaction of bodies of water with a high degree of precision from a few parameters.

The programme, seen in its totality, has provided a unique opportunity to study limnology in a comparative sense. In this respect, it can be considered as a milestone in national and international cooperation, the prospects of which are manyfold and leading into the direction of what Elster outlined as the future of limnology in his memorable 1956 conference (cf. Elster 1958).

However, the programme would have failed if it had not also provided the basis from which it is possible to establish improved loading criteria for practically combating eutrophication of lakes. A synthesis of such criteria is given in Figure 8. These criteria are in logical sequence of the criteria proposed in previous papers by Vollenweider (1968, 1975, 1976), linking average inflow concentrations for phosphorus with expected average lake concentration and average chlorophyll concentrations as a function of the flushing regime of lakes. Division between the main trophic categories is based on the 50% probability of belonging to the indicated class, and the vertical arrows may be read in the sense of "belonging to or better than" the indicated class. With this, management has a tool to establish whatever goal is thought to be desirable to reach, or conversely, to anticipate the level of improvement which can be expected from an established reduction programme. How this should be done in practice, and with what level of uncertainty one has to reckon with, is discussed in the Synthesis Report.

Besides this positive note, however, it must be underlined that many questions remain open, and that a blind and uncritical application of the OECD results can lead to gross error. Limnology, and its application for practical purposes, was and is a complex science, and remains a matter of skill and experience. The establishment of group behaviour of lakes, as was the main objective of the OECD Programme, does not necessarily mean that each single case can be subordinated to one single rule.

Indeed, a more detailed elaboration of the OECD data - a work which still requires considerable time - already indicates that a more selective grouping of lakes having similar limnological properties would reduce some of the uncertainties resulting from an indiscriminate pooling of all data. From here on, one has to find out what the discriminative parameters are for group differences. Factors which lend themselves for further consideration are: type and length of stratification, epi- hypolimnetic ratio, mixing depth, ice coverage, humic substances, N/P ratio, zooplankton and fish population, etc.

An improved approach to discrimination analysis of trophic conditions of lakes is underway by Chapra and Reckhow (1979) who try to avoid some of the pitfalls of the hitherto used prediction models by applying the uncertainty theory. Schaffner and Oglesby (1978) and Oglesby and Schaffner (1978) introduce in their modifications for some of the factors mentioned above.

Last but not least, the next step in the endeavour will be a concerted effort to link experimental with theoretical limnology. Over the last decade or so, theoretical limnologists have made much progress and brought into the open many of the uncertainties in our understanding. This throws the ball back to experimental limnologists who will have to rethink many of their programmes. The extended experimental work of Schindler and his colleagues in the Experimental Lakes Area studies - which cannot be referred to in detail in this review - provides further guides to understanding the complex relationship between nutrient loading and lake reaction.

REFERENCES

(for literature cited consult the following reports)

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- Alpine Lakes, prepared by Hj. Fricker, Swiss Federal Board for Environmental Protection (Bundesamt für Umweltschutz), CH-3003 Bern, Switzerland. 1980.
- The Nordic Project, prepared by S-O. Ryding. Nordforsk. Nordic Cooperative Organization for Applied Research. Secretariat of Environmental Sciences. Folkskolegatan 10A, SF-00100 Helsingfors 10, Finland. 1980.
- Shallow Lakes and Reservoirs, prepared by J. Clasen. The Water Research Centre, Medmenham Laboratory, P.O. Box 16, Medmenham, Marlow, Bucks., England. 1980.
- Summary Analysis of the North American OECD Project (U. S. Portion), prepared by W. Rast and F. Lee. U.S. EPA-600/3-78-008. Ecol. Res. Ser. 1978.
- A Test Case Study of the OECD Programme on Eutrophication (Canadian Portion), prepared by L. L. Janus and R. A. Vollenweider. IWD-National Water Research Institute, CCIW, Burlington, Ontario, 1981.

OECD Eutrophication Programme

- Synthesis Report, prepared by R. A. Vollenweider and J. Kerekes, and members of the Technical Bureau. OECD Secretariat, Environment Directorate, 2, rue André Pascal, 75775 Paris Cedex 16, France. 1981.

- <u>Eutrophication Control</u>. Conclusions of the OECD Cooperative Programme on Eutrophication, prepared by R. A. Vollenweider and the members of the Technical Bureau. (published in UNESCO Nature and Resources 16, 3, 1980).

Table 1. CATEGORIZATION OF PARAMETERS

FOR MEASURING AND MONITORING EUTROPHICATION

Ergodic (Resultant) Variables		Causative Variables
A) Short Term Variability: High	B) Short Term Variability: Moderate to Low	

- Phytoplankton biomass
- Major algal groups and dominant species
- Chlorophyll a and other phytopigments
- Particulate organic carbon and nitrogen
- Daily primary production rates
- Secchi disc visibility
	- Zooplankton standing crop
	- Bottom fauna standing crop
	- Epilimnetic Δ P, Δ N, Δ Si (Δ = difference between winter and summer concentrations)
	- Hypolimnetic O_2 and Δ O_2
	- Annual primary production
		- Nutrient Loadings
		  - Total Phosphorus
		  - Ortho phosphates
		  - Total Nitrogen
		  - Mineral Nitrogen (NO_3 + NH_3)
		  - Kjeldahl Nitrogen
		- Nutrient Concentrations
		  - Same as above
		  - Reactive Silica
		  - Others (e.g. Microelements)

Related Descriptive Parameters

- Morphometric parameters of lake and catchment area
- Flushing regime
- Geological and climatic parameters
- Land Use
- Urbanization and industrialization
- Main nutrient sources
- Temperature and mixing regime
- Conductivity, pH, alkalinity
- Major ion spectra
- Insolation and optical properties
- Others as deemed necessary

Table 2. TROPHIC CHARACTERIZATION OF LAKES

IMPAIRMENT OF VARIOUS USES

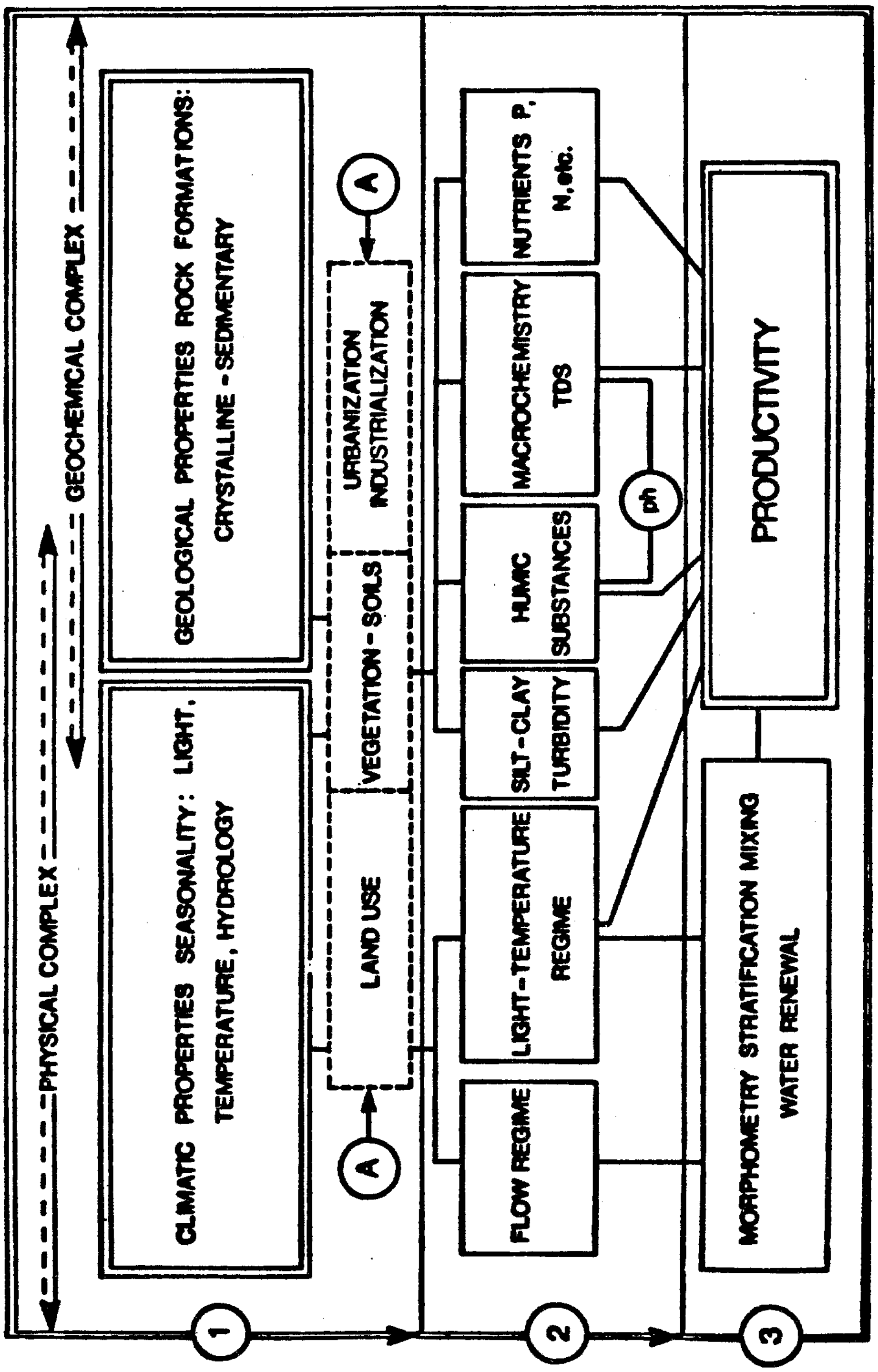
Limnological Characterization	Oligotrophic	Mesotrophic	Eutrophic
General level of production	low	medium	high
Biomass	low	medium	high
Green and/or blue-green algae fractions	low	variable	high
Hypolimnetic oxygen content	high	variable	low
Impairment of multi-purpose use of lake	little	variable	great

Table 3. Preliminary classification of trophic state in the OECD eutrophication programme. Trophic status is assigned based on the opinion of the investigator of each lake. The geometric mean (based on log 10 transformation) was calculated after removing values < or > x 2 SD obtained (where applicable) in the first calculation.

Variable (Annual Mean Values)		Oligotrophic	Mesotrophic	Eutrophic	Hyper-eutrophic
Total Phosphorus mg/m^3	$\bar{x}$	8.0	26.7	84.4	
	$\bar{x} \pm 1$ SD	4.85 - 13.3	14.5 - 49	38 - 189	
	$\bar{x} \pm 2$ SD	2.9 - 22.1	7.9 - 90.8	16.8 - 424	
	Range	3.0 - 17.7	10.9 - 95.6	16.2 386	750 - 1200
	n	21	19 (21)	71 (72)	2
Total Nitrogen mg/m^3	$\bar{x}$	661	753	1875	
	$\bar{x} \pm 1$ SD	371 - 1180	485 - 1170	861 - 4081	
	$\bar{x} \pm 2$ SD	208 - 2103	313 - 1816	395 - 8913	
	Range	307 1630	361 - 1387	393 - 6100	
	n	11	8	37 (38)	
Chlorophyll a mg/m^3	$\bar{x}$	1.7	4.7	14.3	
	$\bar{x} \pm 1$ SD	.8 - 3.4	3. - 7.4	6.7 - 31	
	$\bar{x} \pm 2$ SD	.4 - 7.1	1.9 - 11.6	3.1 - 66	
	Range	0.3 - 4.5	3. - 11	2.7 - 78	100 - 150
	n	22	16 (17)	70 (72)	2
Chlorophyll a Peak Value mg/m^3	$\bar{x}$	4.2	16.1	42.6	
	$\bar{x} \pm 1$ SD	2.6 - 7.6	8.9 - 29	16.9 - 107	
	$\bar{x} \pm 2$ SD	1.5 - 13	4.9 - 52.5	6.7 - 270	
	Range	1.3 - 10.6	4.9 - 49.5	9.5 - 275	
	n	16	12	46	
Secchi Depth m	$\bar{x}$	9.9	4.2	2.45	
	$\bar{x} \pm 1$ SD	5.9 - 16.5	2.4 - 7.4	1.5 - 4.0	
	$\bar{x} \pm 2$ SD	3.6 - 27.5	1.4 - 13	.9 - 6.7	
	Range	5.4 - 28.3	1.5 - 8.1	.8 - 7.0	0.4 - 0.5
	n	13	20	70 (72)	2

$\bar{x}$ = geometric mean
SD = standard deviation
( ) = value in bracket refers to the number of variables (n) employed in the first calculation.

THE THREE LEVELS DETERMINING THE PRODUCTIVITY OF BODIES OF WATER



① BASIN PROPERTIES ② WATER PROPERTIES ③ LIMNOLOGICAL PROPERTIES
Ⓐ ANTHROPOGENIC ALTERATIONS

Fig. 1

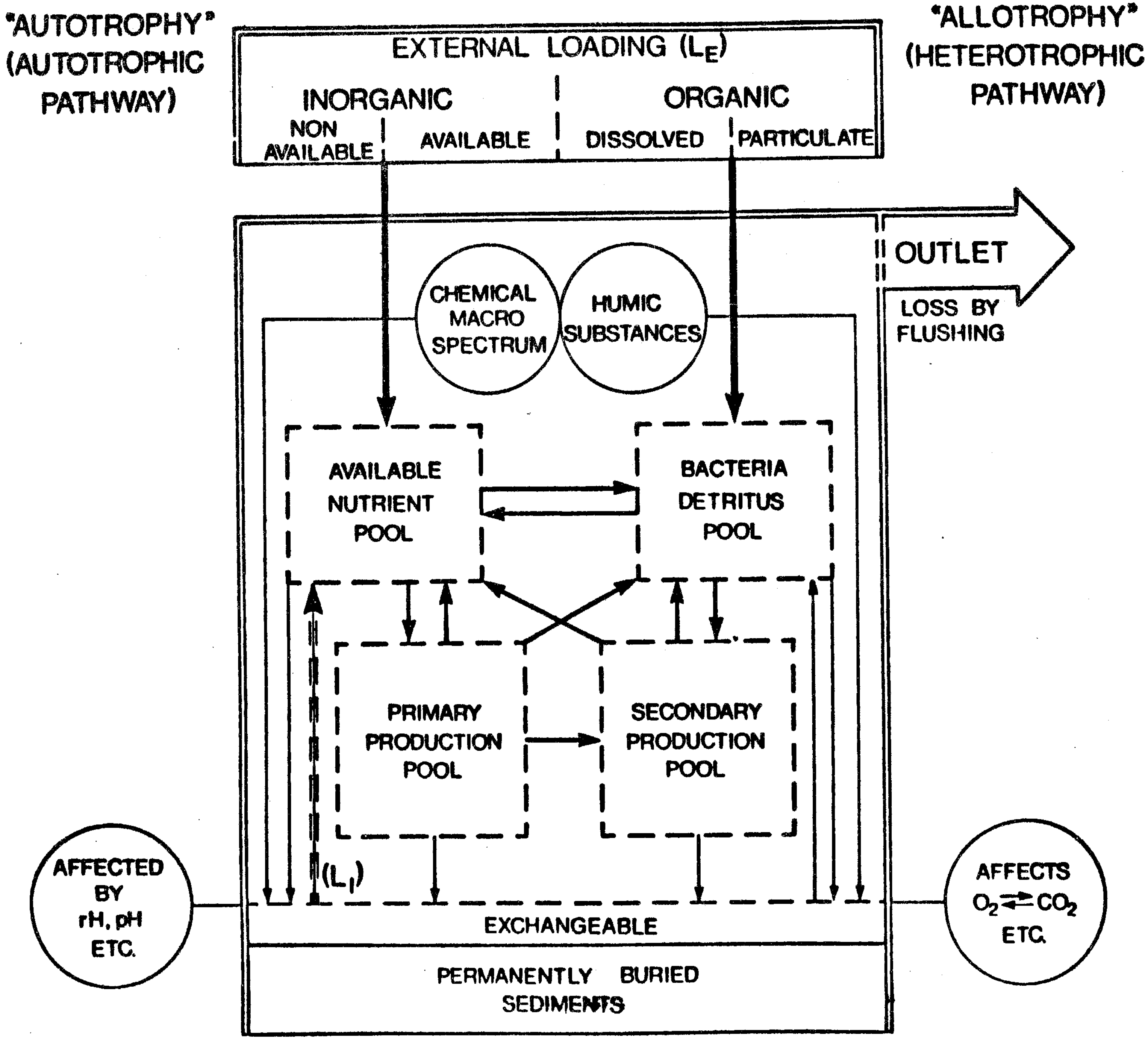
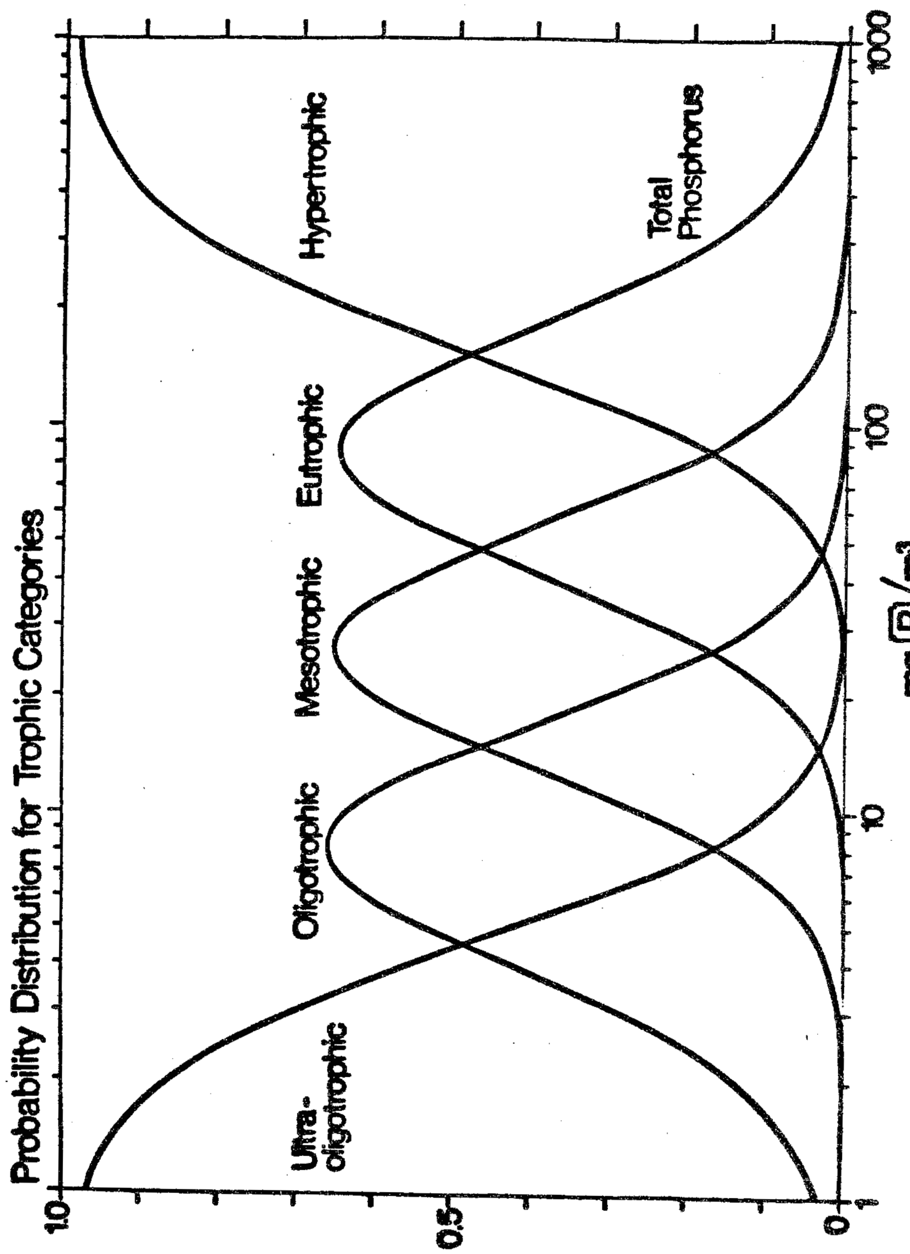

"AUTOTROPHY"
(AUTOTROPHIC PATHWAY)
EXTERNAL LOADING (L_E)
INORGANIC
NON AVAILABLE
AVAILABLE
ORGANIC
DISSOLVED
PARTICULATE
"ALLOTROPHY"
(HETEROTROPHIC PATHWAY)
OUTLET
LOSS BY FLUSHING
CHEMICAL MACRO SPECTRUM
HUMIC SUBSTANCES
AVAILABLE NUTRIENT POOL
BACTERIA DETRITUS POOL
PRIMARY PRODUCTION POOL
SECONDARY PRODUCTION POOL
(L_I)
AFFECTED BY rH, pH ETC.
AFFECTS $O_2 \rightleftarrows CO_2$ ETC.
EXCHANGEABLE
PERMANENTLY BURIED SEDIMENTS

Fig. 2



Probability Distribution for Trophic Categories
Ultra-
oligotrophic
Oligotrophic
Mesotrophic
Eutrophic
Hypertrophic
Total
Phosphorus
1.0
0.5
0
1
10
100
1000
mg [P]/m³

Fig. 3

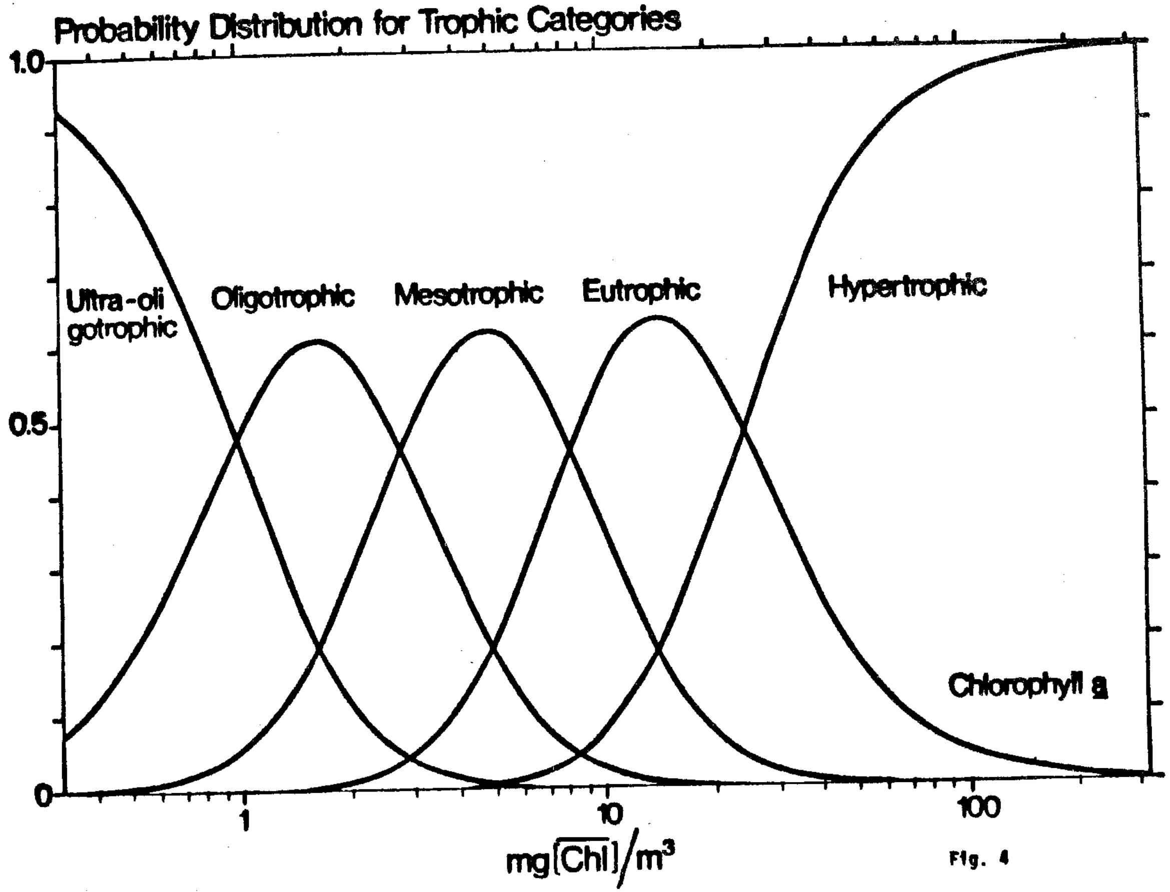

Probability Distribution for Trophic Categories
1.0
0.5
0
Ultra-oli gotrophic
Oligotrophic
Mesotrophic
Eutrophic
Hypertrophic
Chlorophyll a
1
10
100
mg[Chl]/m³

Fig. 4

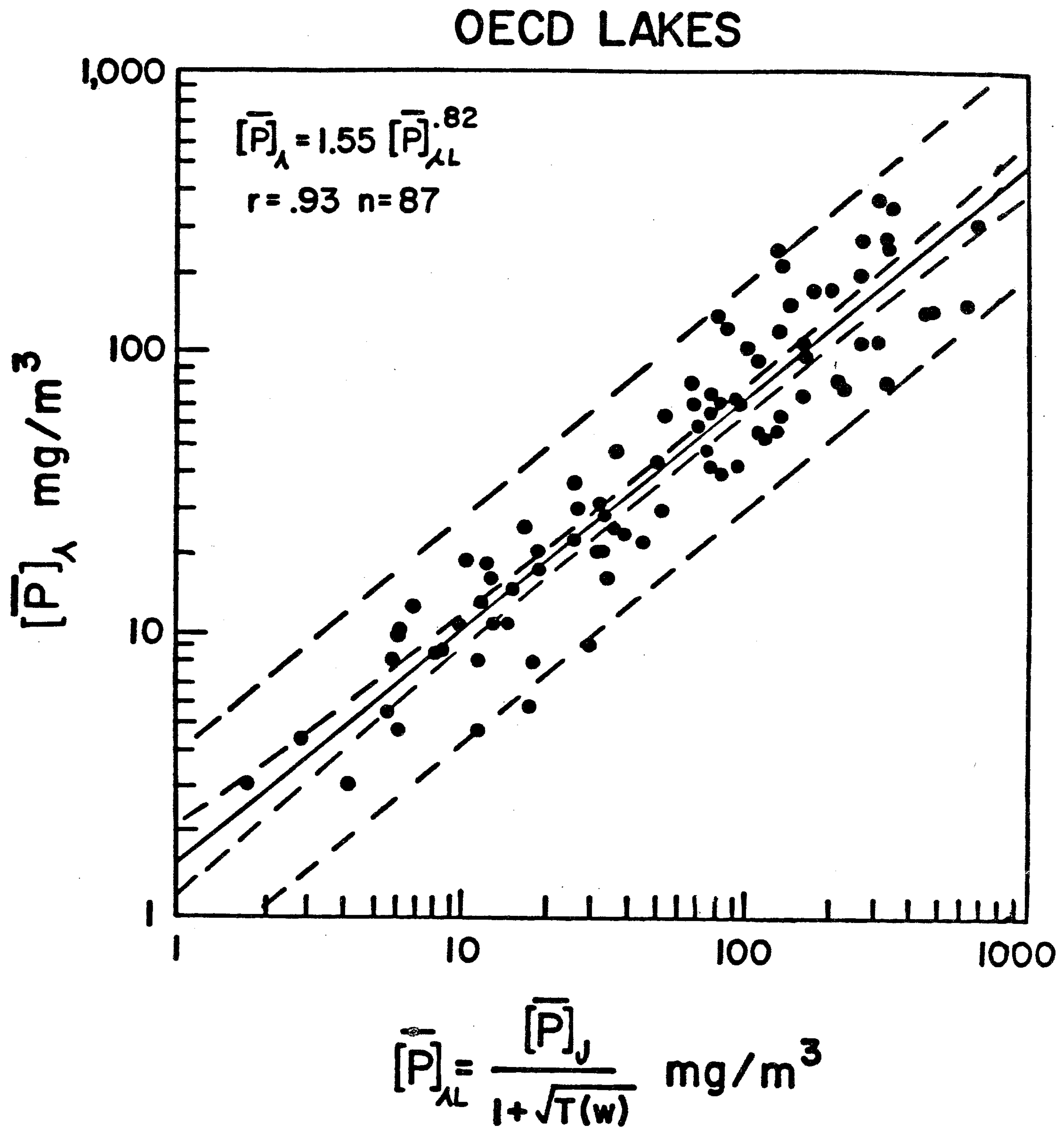

OECD LAKES
$[\bar{P}]_\lambda = 1.55\,[\bar{P}]_{\lambda L}^{.82}$
r = .93 n = 87
1,000
100
10
1
1
10
100
1000
$[\bar{P}]_\lambda$ mg/m^3
$[\bar{P}]_{\lambda L} = \dfrac{[\bar{P}]_J}{1+\sqrt{T(w)}}$ mg/m^3

Fig. 5

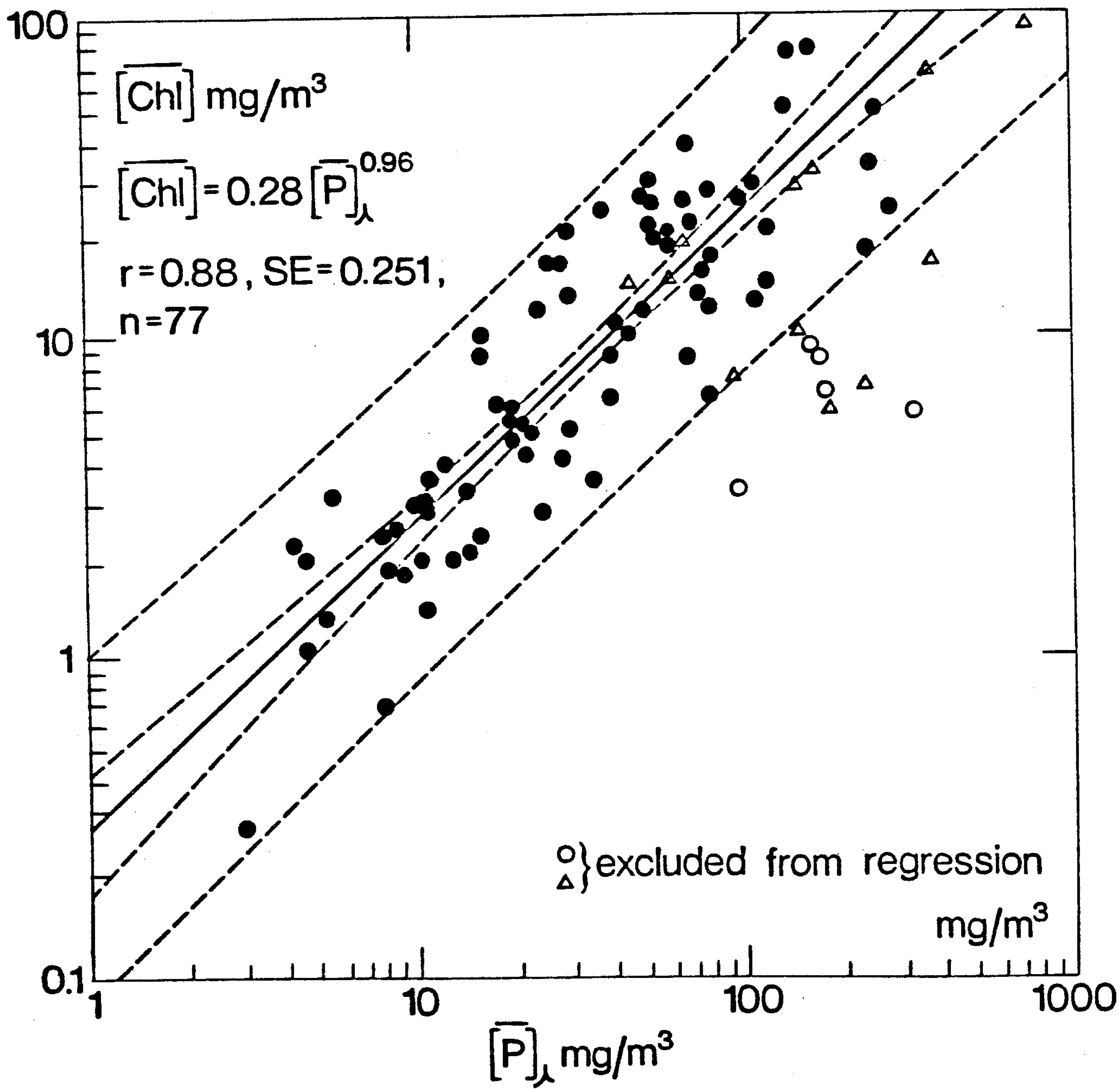

100
10
1
0.1
[Chl] mg/m3
[Chl] = 0.28 [P]λ 0.96
r = 0.88, SE = 0.251,
n = 77
excluded from regression
mg/m3
1
10
100
1000
[P]λ mg/m3

Fig. 6.A

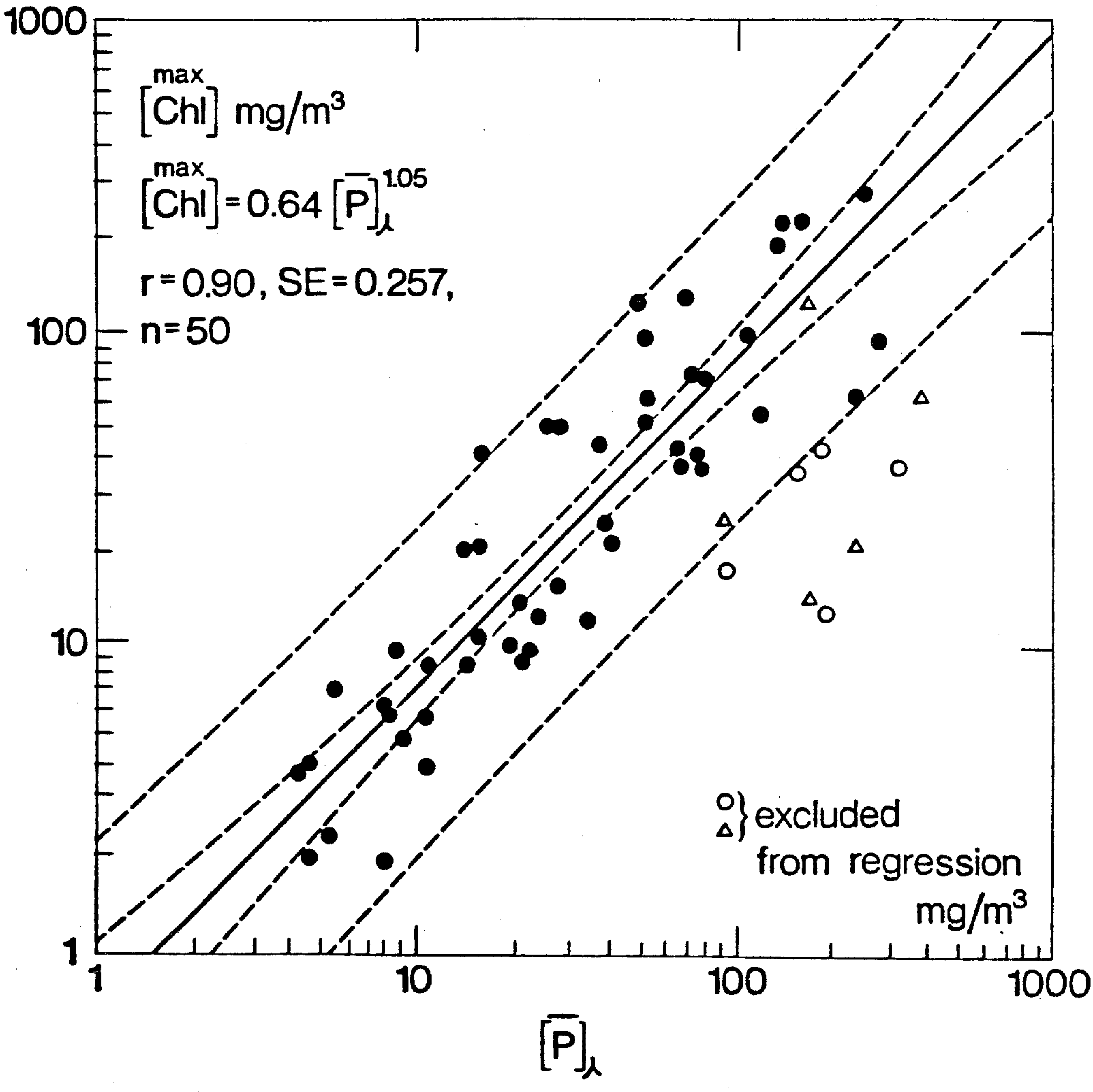

1000
100
10
1
$\overset{max}{[Chl]}$ mg/m^3
$\overset{max}{[Chl]} = 0.64\,[\bar{P}]_{\lambda}^{1.05}$
r = 0.90, SE = 0.257,
n = 50
$\circ$ $\triangle$ } excluded from regression
mg/m^3
1
10
100
1000
$[\bar{P}]_{\lambda}$

Fig. 6. B

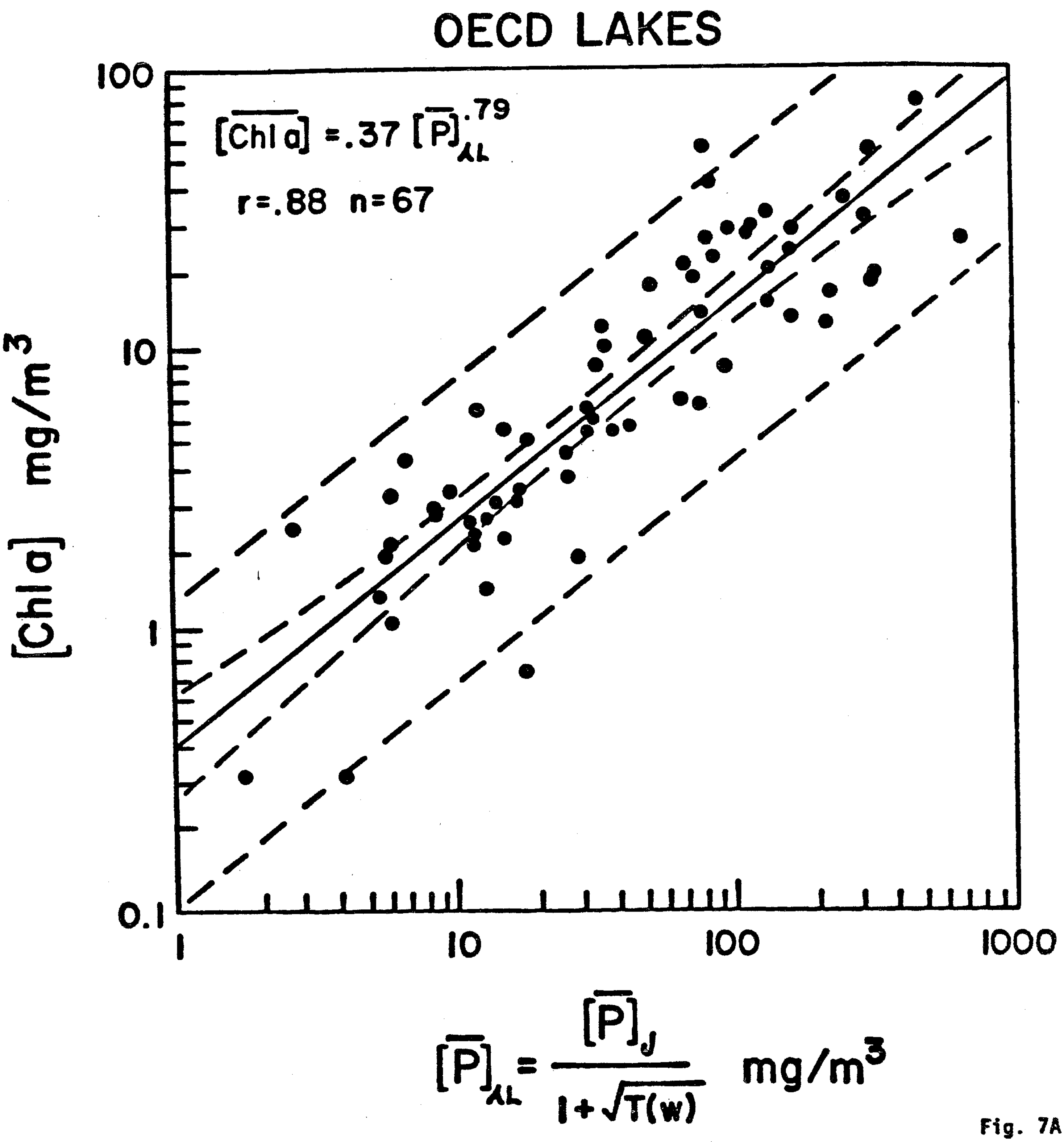

OECD LAKES
$[\overline{Chl\,a}] = .37\,[\overline{P}]_{\lambda L}^{.79}$
r=.88 n=67
100
10
1
0.1
1
10
100
1000
[Chl a] mg/m³
$[\overline{P}]_{\lambda L} = \frac{[\overline{P}]_J}{1+\sqrt{T(w)}}$ mg/m³

Fig. 7A

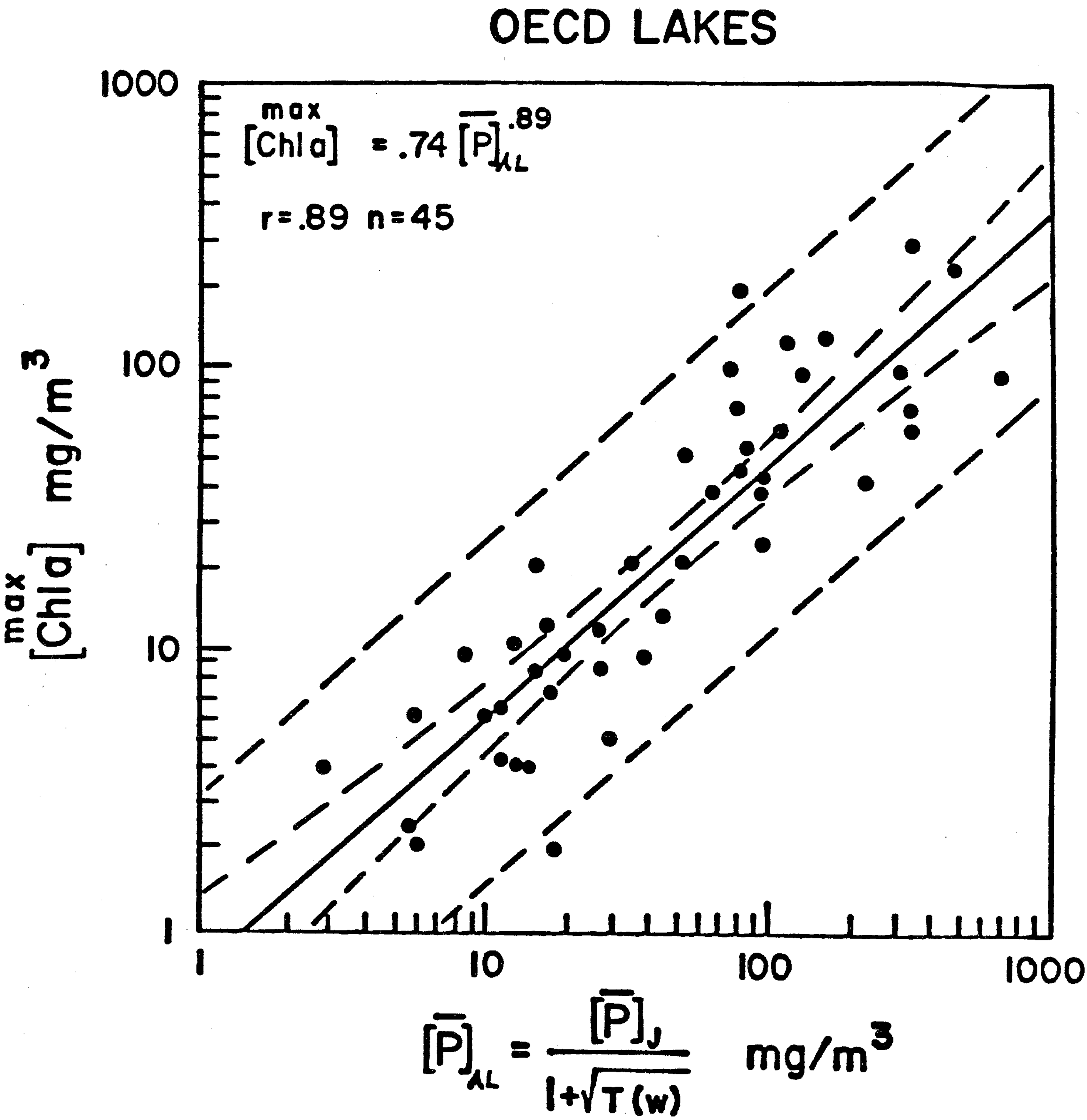
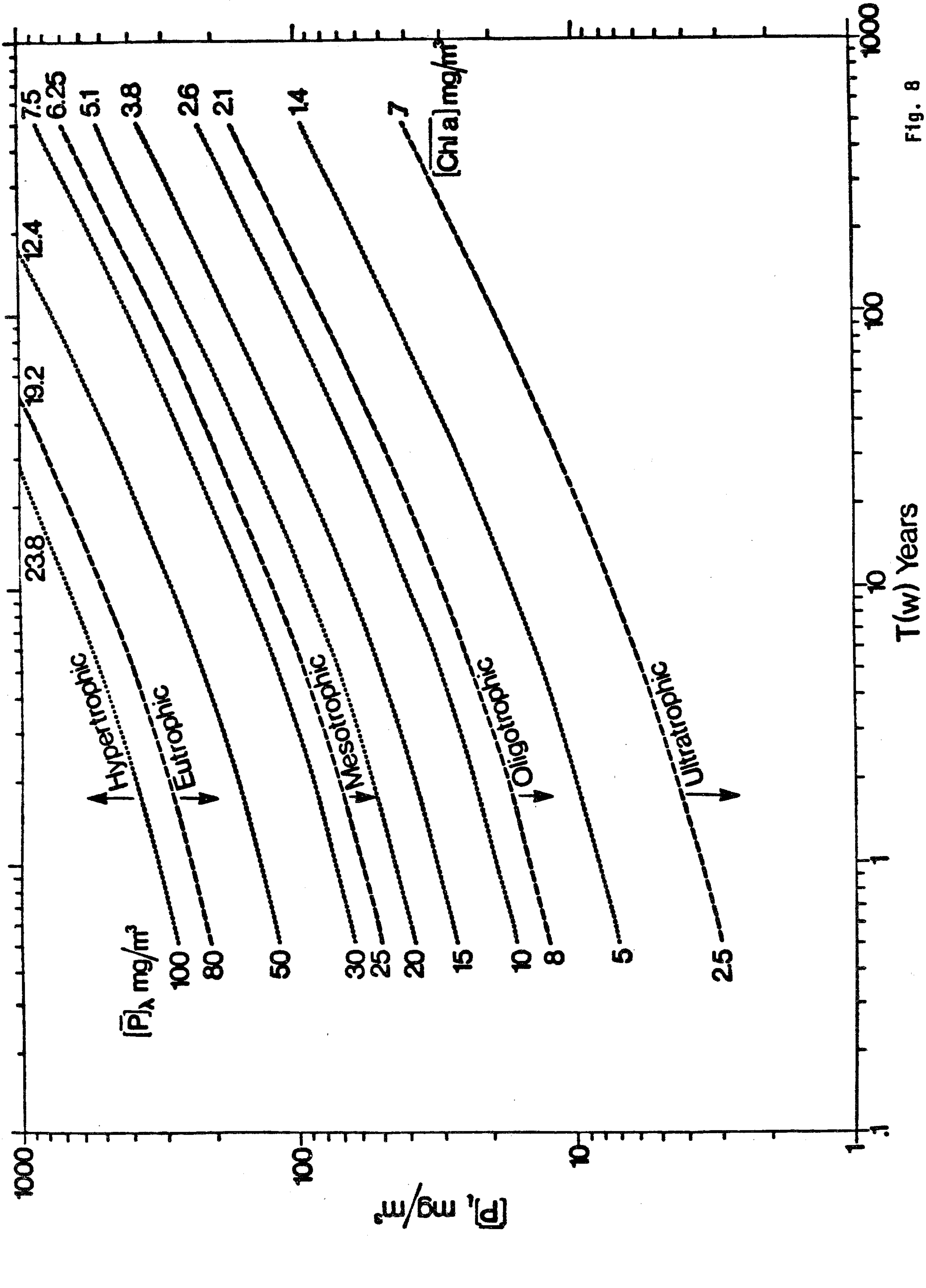

OECD LAKES
$[Chl\,a]^{max} = .74\,[\bar{P}]_{\lambda L}^{.89}$
r=.89 n=45
1000
100
10
1
1
10
100
1000
$[Chl\,a]^{max}$ mg/m^3
$[\bar{P}]_{\lambda L} = \frac{[\bar{P}]_J}{1+\sqrt{T(w)}}$ mg/m^3

Fig. 7B



Hypertrophic
Eutrophic
Mesotrophic
Oligotrophic
Ultratrophic
[P̄]λ mg/m³
100
80
50
30
25
20
15
10
8
5
2.5
[Chl a] mg/m³
23.8
19.2
12.4
7.5
6.25
5.1
3.8
2.6
2.1
1.4
.7
T(w) Years
[P], mg/m³
1000
100
10
1
.1

Fig. 8

Addendum to Appendix 1

The statistical relationships of Table 3 are those calculated from the composite of OECD data as per the Synthesis Report (Vollenweider and Kerekes, 1981) and are referred to throughout the Canadian report as OECD standard relationships (also cf. Introduction). They form the basis of comparison for the Canadian lakes data and are presented graphically in Figures A1.1 to A1.17.

Standard relationships are given in pairs of figures. The first figure shows (a) and (b) lines with an intermediate line. The (a) line, commonly called the OECD line, is the regression resulting from minimizing y against x and the (b) line represents the regression obtained by minimizing x against y. The (b) line results have been inverted (i.e. recalculated for x as reference variable) for presentation on the same axes as the (a) lines. The intermediate line has been calculated from the ratio SE(y)/SE(x). The second figure of each pair gives the confidence intervals of the (a) line for 80 and 95% probability levels.

Table 3. OECD Standard Regressions, $Y = A.X^B$. (See following graphical representations in Figures A1.1 to A1.17).

Fig. A.1...	Y	versus X	n	r	$A^{2)}$	$B^{2)}$	$A^{3)}$	$B^{3)}$	$\bar{A}^{4)}$	$\bar{B}^{4)}$	$SE(Y)^{5)}$	$SE(X)^{6)}$
1, 2	$[\overline{chl}]$	$[\overline{P}]_\lambda$	77	.88	.287	.962	.110	1.242	.183	1.093	.251	.230
3, 4	$\overset{max}{[chl]}$	$[P]_\lambda$	50	.90	.635	1.048	.277	1.294	.429	1.165	.257	.221
5, 6	$[\overline{P}]_\lambda$	$[X]^{1)}$	87	.93	1.566	.821	.948	.950	1.229	.883	.193	.219
7, 8	$[\overline{chl}]$	$[X]^{1)}$	67	.88	.373	.795	.160	1.027	.251	.904	.256	.283
9, 10	$\overset{max}{[chl]}$	$[X]^{1)}$	45	.89	.707	.905	.290	1.142	.464	1.017	.284	.279
11, 12	$\overline{S}$	$[\overline{chl}]$	78	- .75	9.32	- .508	22.0	- .903	13.5	- .677	.198	.292
13, 14	$\overline{S}$	$[\overline{P}]_\lambda$	87	- .47	9.89	- .284	505	- 1.284	34.8	- .603	.255	.423
15, 16	$\overline{S}$	$[X]^{1)}$	67	- .69	14.8	- .387	85.7	- .812	30.3	- .560	.236	.421
17, -	$[N]_\lambda$	$[X]'^{1)}$	42	.92	5.30	.782	2.00	.924	3.33	.850	.157	.182

1) $[X] = [\overline{P}]_1/(1 + \sqrt{T(W)})$; $[X]' = [\overline{N}]_1/(1 + \sqrt{T(W)})$

2) a) - Regression

3) b) - Regression

4) Average between a) and b)

all reported against X as reference variable

5) and 6) Standard Errors referring to ^{10}log-transformed equation

LIST OF FIGURES FOR ADDENDUM TO APPENDIX 1

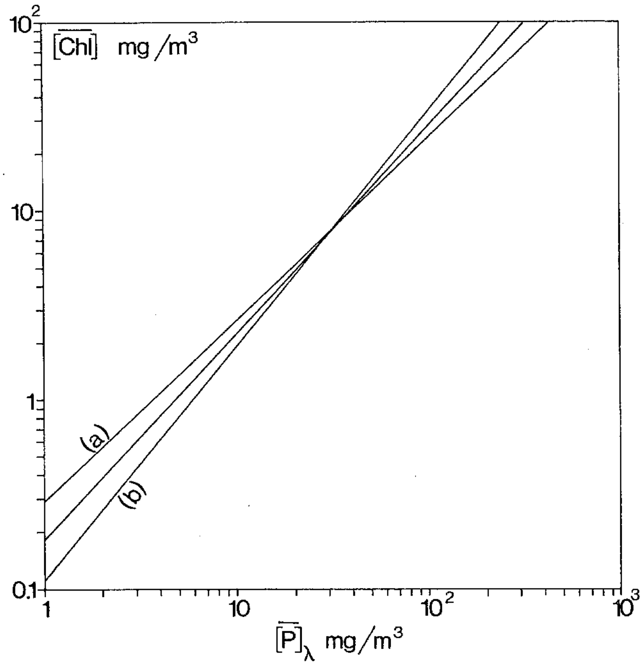


Figure A 1.1 OECD Standard Regression: Annual mean chlorophyll a concentration in relation to annual mean total phosphorus lake concentration:

a-, b- and intermediate lines

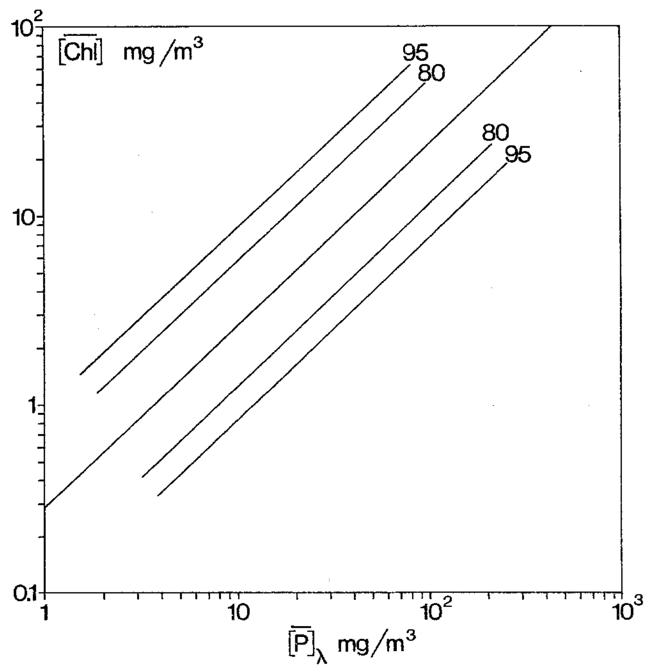


Figure A 1.2 OECD Standard Regression: Annual mean chlorophyll a concentration in relation to annual mean total phosphorus lake concentration:

80 and 95% confidence limits for the a-line

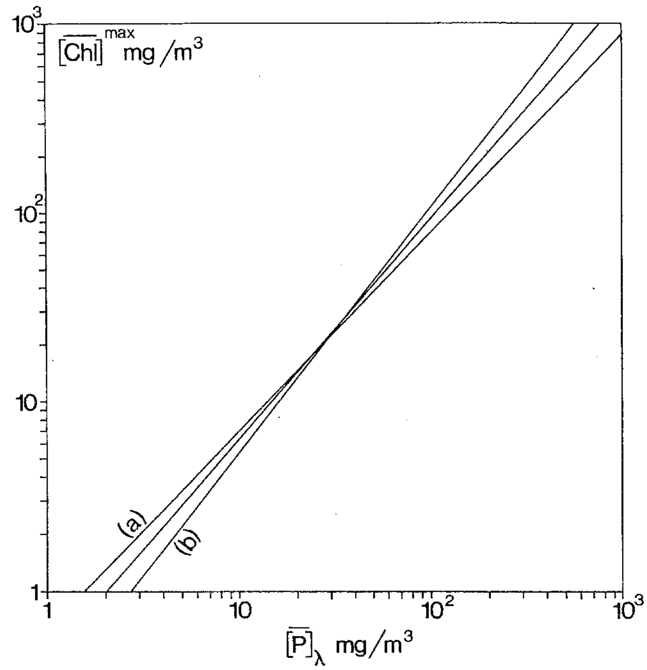


Figure A 1.3 OECD Standard Regression: Maximum chlorophyll $\underline{a}$ concentration in relation to annual mean total phosphorus lake concentration: a-, b- and intermediate lines

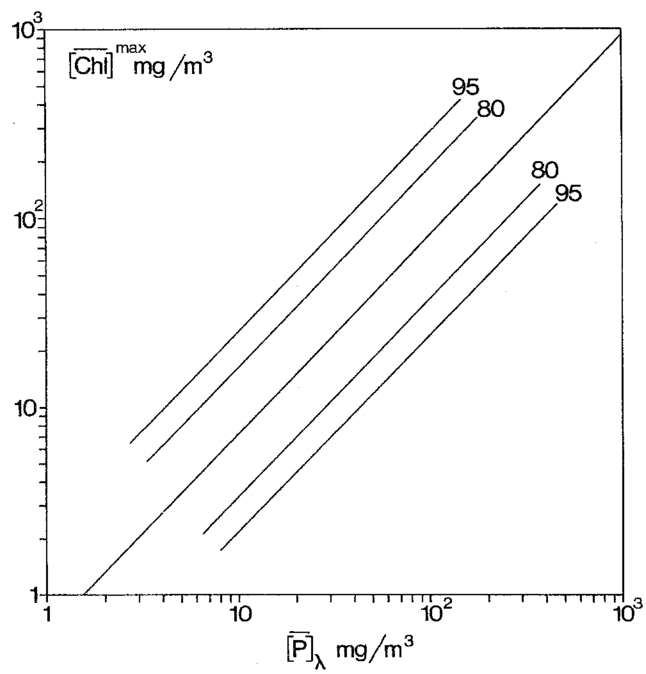


Figure A 1.4 OECD Standard Regression: Maximum chlorophyll $\underline{a}$ concentration in relation to annual mean total phosphorus lake concentration: 80 and 95% confidence limits for the a-line

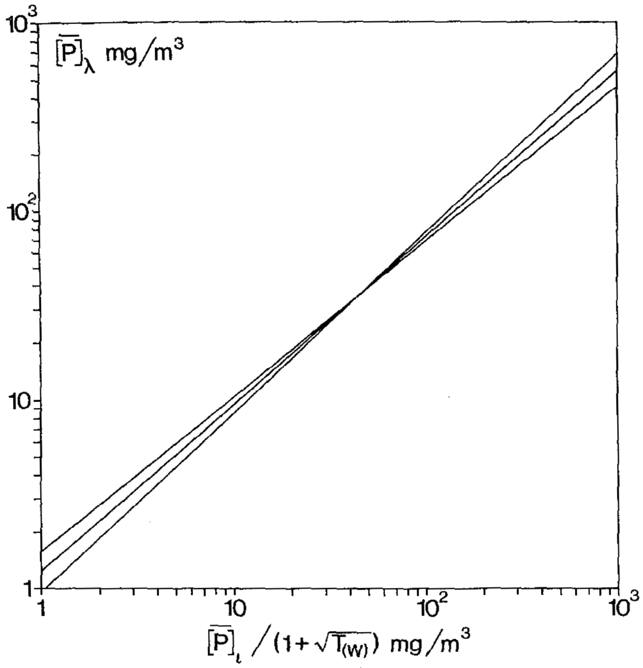


Figure A 1.5 OECD Standard Regression: Annual mean total phosphorus concentration in relation to flushing corrected annual mean total phosphorus inflow concentration: a-, b- and intermediate lines

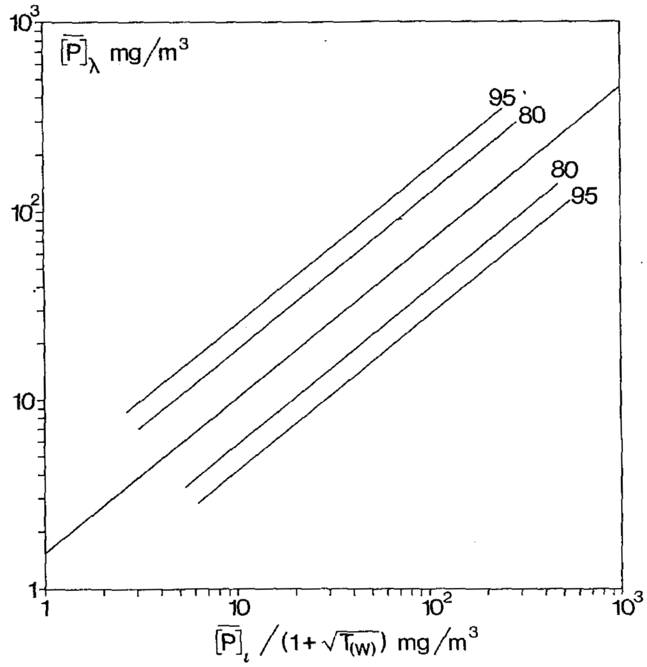


Figure A 1.6 OECD Standard Regression: Annual mean total phosphorus concentration in relation to flushing corrected annual mean total phosphorus inflow concentration: 80 and 95% confidence limits for the a-line

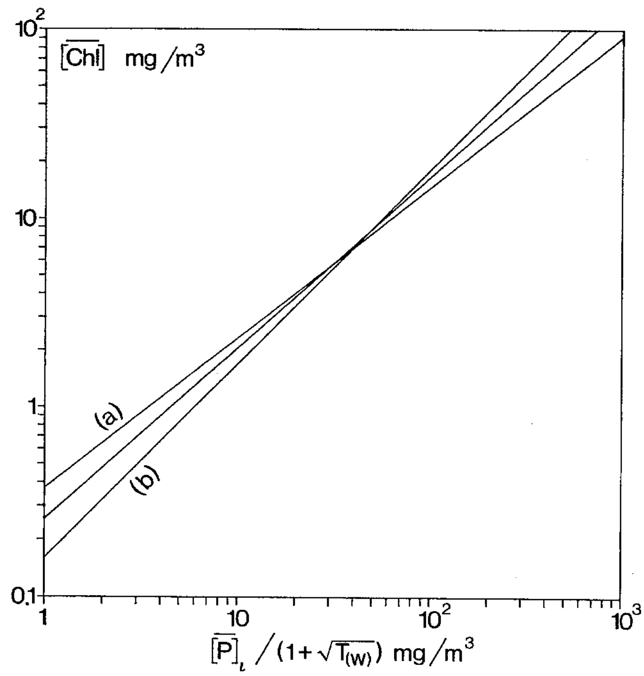


Figure A 1.7 OECD Standard Regression: Annual mean chlorophyll a concentration in relation to flushing corrected annual mean total phosphorus inflow concentration: a-, b- and intermediate lines

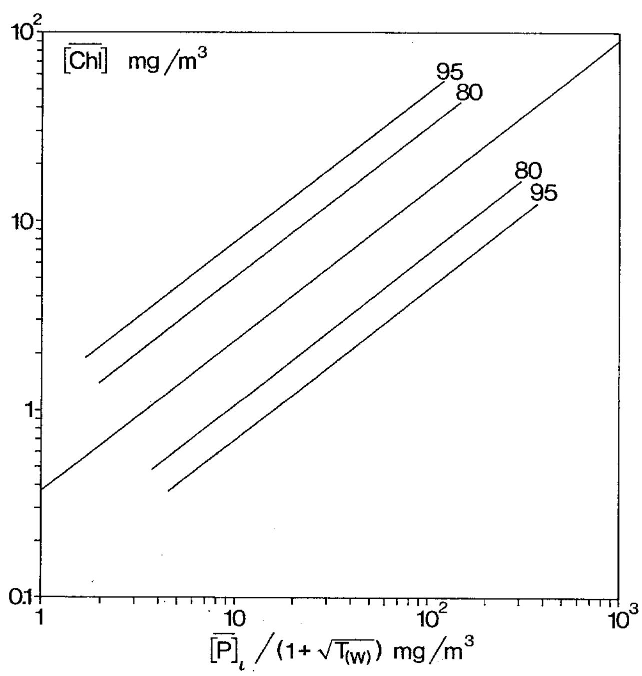


Figure A 1.8 OECD Standard Regression: Annual mean chlorophyll a concentration in relation to flushing corrected annual mean total phosphorus inflow concentration; 80 and 95% confidence limits for the a-line

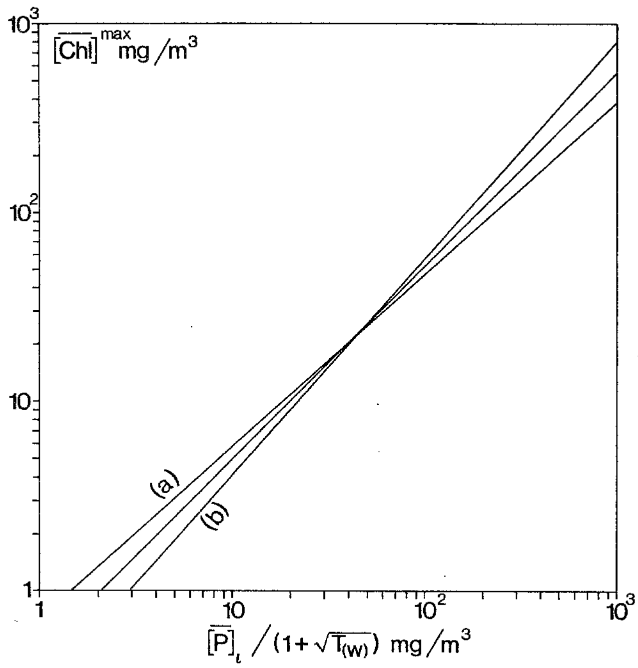


Figure A 1.9 OECD Standard Regression: Maximum chlorophyll a concentration in relation to flushing corrected annual mean total phosphorus inflow concentration: a-, b- and intermediate lines

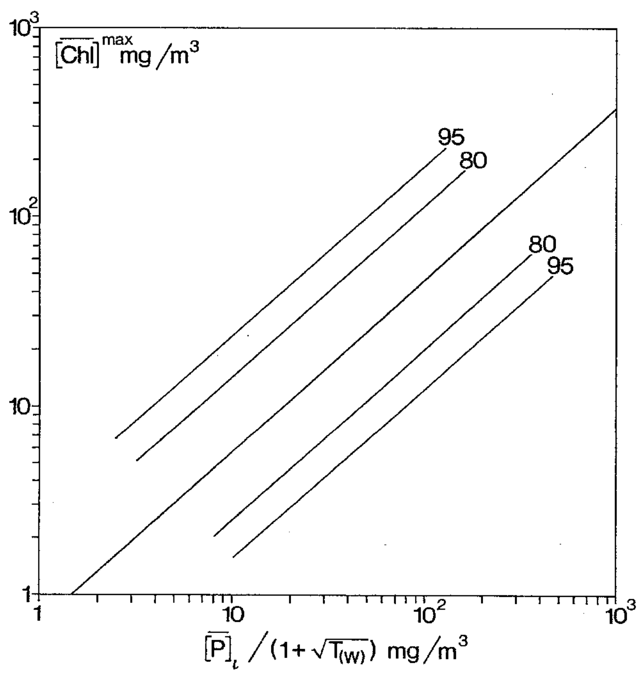


Figure A 1.10 OECD Standard Regression: Maximum chlorophyll a concentration in relation to flushing corrected annual mean total phosphorus inflow conconcentration: 80 and 95% confidence limits for the a-line

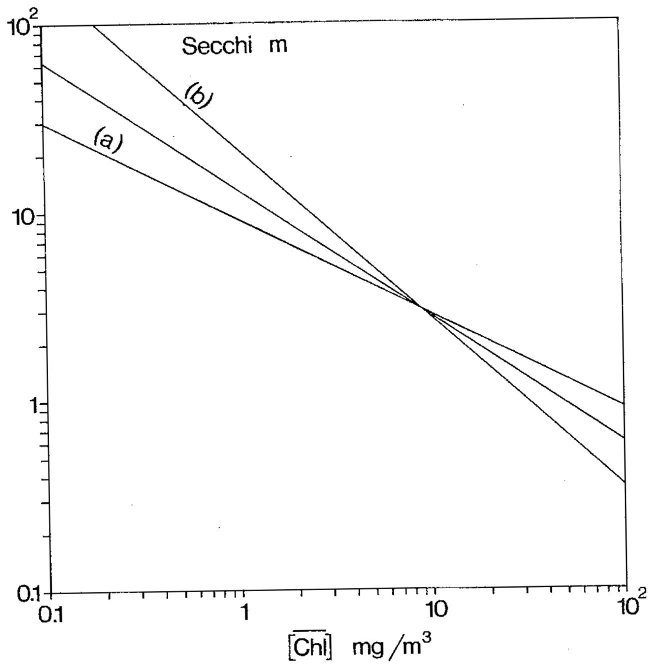


Figure A 1.11 OECD Standard Regression: Annual mean Secchi transparency in relation to annual mean chlorophyll a concentration: a-, b- and intermediate lines

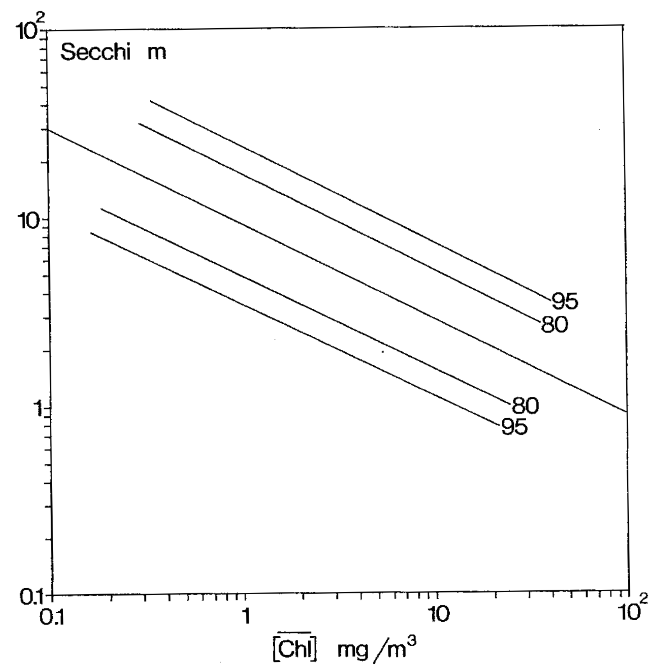


Figure A 1.12 OECD Standard Regression: Annual mean Secchi transparency in relation to annual mean chlorophyll a concentration: 80 and 95% confidence limits for the a-line

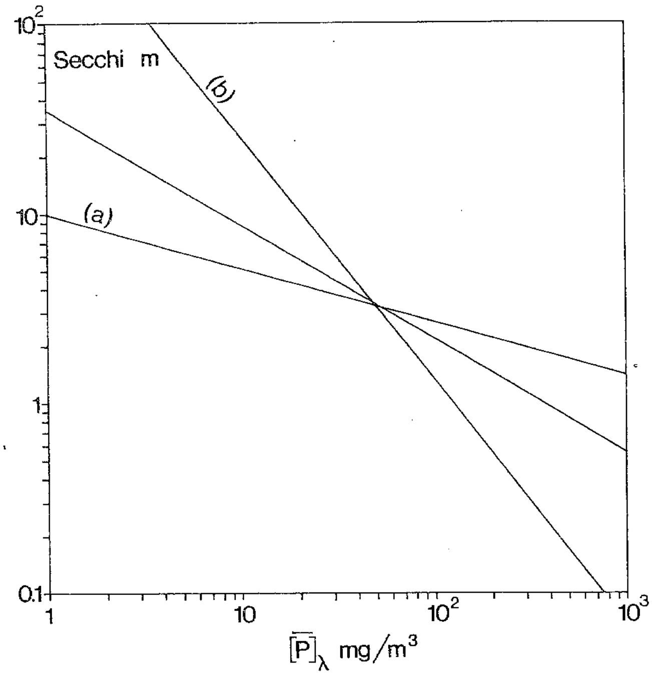


Figure A 1.13 OECD Standard Regression: Annual mean Secchi transparency in relation to annual mean total phosphorus lake concentration:
a-, b- and intermediate lines

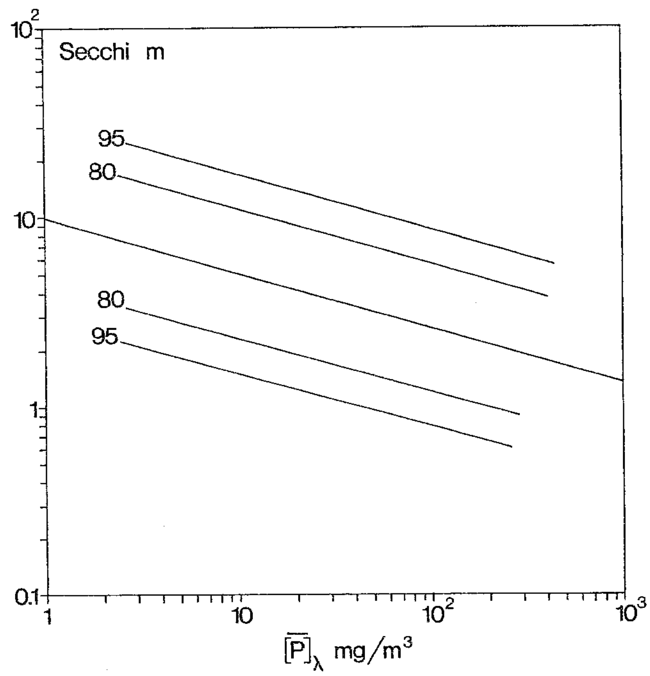


Figure A 1.14 OECD Standard Regression: Annual mean Secchi transparency in relation to annual mean total phosphorus lake concentration:
80 and 95% confidence limits for the a-line

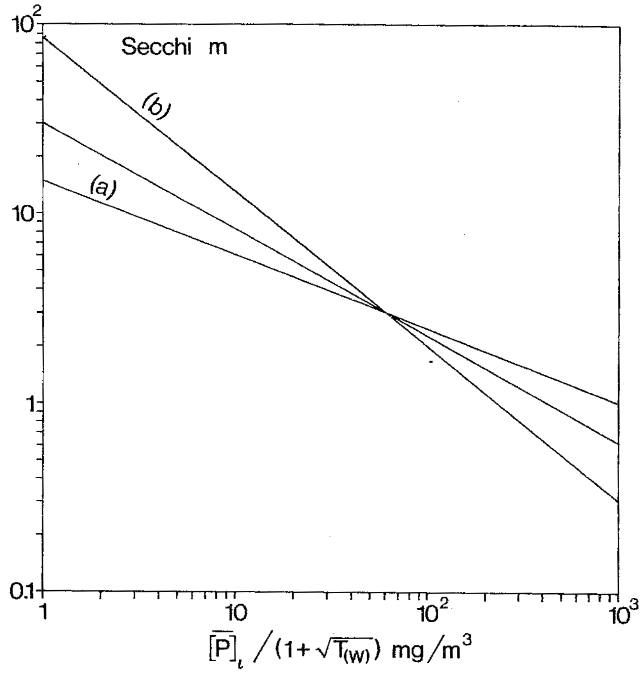


Figure A 1.15 OECD Standard Regression: Annual mean Secchi transparency in relation to flushing corrected annual mean total phosphorus inflow concentration: a-, b- and intermediate lines

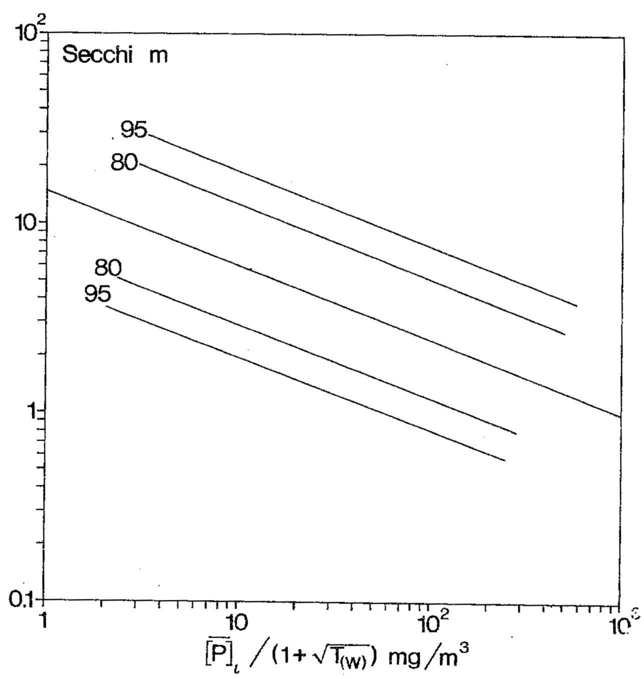


Figure A 1.16 OECD Standard Regression: Annual mean Secchi transparency in relation to flushing corrected annual mean total phosphorus inflow concentration: 80 and 95% confidence limits for the a-line

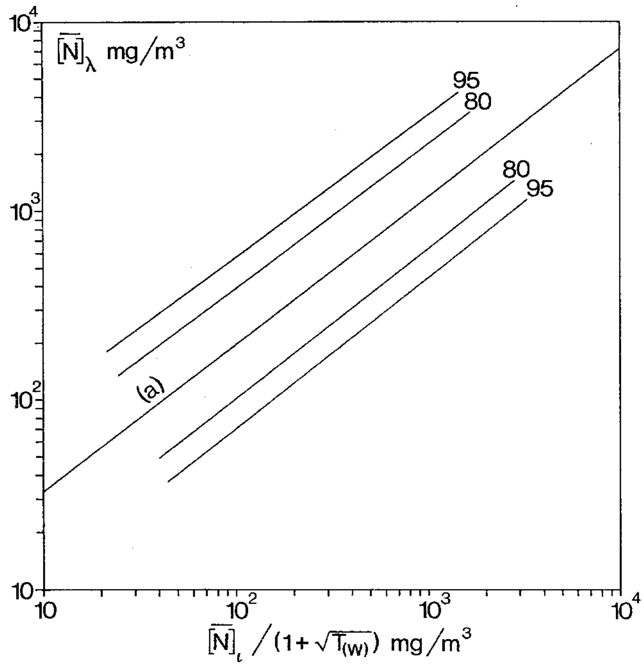


Figure A 1.17 OECD Standard Regression: Annual mean total nitrogen lake concentration in relation to flushing corrected annual mean total nitrogen inflow concentration:
80 and 95% confidence limits for the a-line

APPENDIX 2

Rationale and Critical Considerations for Using $1/(1 + \sqrt{T(w)})$ as Standard Flushing Correction

The following are a few hints in regard to the simplifications made to relate inlake concentration (both for phosphorus and chlorophyll, as well as Secchi transparency, primary production a.o.) to standard corrected average inflow concentrations.

The basis for all considerations is the assumption that lakes, in principle, behave as mixed reactors. This is only true to some extent, but for comparative purposes, experience shows that for a first order approximation, this assumption is justifiable, under the condition that the time resolution is one year. On a seasonal basis, or shorter time period, this assumption is invalid.

The respective mass balance equation requires that the storage change in the lake equals the net load minus the balance of the water-sediment exchange, i.e.

$$V \frac{d[P]_\lambda}{d_t} = Q\,([P]_\iota - [P]_\omega) - A\,(\downarrow \text{Flux}\,(P) - \uparrow \text{Flux}\,(P)) \qquad (1)$$

where

V	=	lake volume (m^3)
A	=	surface area of sediments (m^2) (= approx. surface area of lake)
Q	=	annual outflow (m^3/year)
$[P]_\lambda$	=	average lake concentration (g/m^3)
$[P]_\iota$	=	average inflow concentration (g/m^3)
$[P]_\omega$	=	average outflow concentration (g/m^3)
$F(P)$	=	flux to, or from the sediments (g/m^2 . year)

With the further assumption that

$$A\ (\downarrow F(P) - \uparrow F(P)) = \bar{s}\ A\ [P]_{\lambda}\quad ,$$

where $\bar{s}$ is an apparent sedimentation velocity (m/y) - which can be interpreted as the balance between all downward and upward movements integrated over one year (cf. Chapra 1977) - and setting the outflow concentration equal to lake concentration, the steady state solution for the lake concentration is

$$[P]_{\lambda} = [P]_{\iota}\ \frac{Q}{Q + \bar{s}\ A} \tag{2}$$

Formulated in dimensionless terms, this equation can also be written as

$$[P]_{\lambda}/[P]_{\iota} = \frac{1}{1 + \bar{s}/q_s} = \frac{1}{1 + \bar{s}\ T(w)/\bar{z}} \tag{3}$$

where q_S, $T(w)$ and $\bar{z}$ are the hydraulic load (m/y), the theoretical water residence time (y), and mean depth (m), respectively.

Stepwise regression of the Canadian data shows that the interrelationships of the parameters in Eq. (3) cannot be satisfied by linear assumptions (cf. also Table A2.1). This has been found also in other programmes. Statistically, the term in the denominator results in the form of

$$s'\ T(w)^{\alpha}/\bar{z}^{\beta} \text{ or } s'\ T(w)^{\gamma}/q_S^{\sigma}\quad ,$$

whereby α and γ are < 1 but > 0, whereas β and σ differ only marginally from 0. The water residence time was found to be the most important

Table A2.1 Stepwise Regression. Average lake phosphorus concentration as function of average inflow concentration, mean depth, water residence, and hydraulic load (Canadian lakes).

Dep. Variable	Indep. Variable	Resulting Model	Res. Sq.	St. Err.	r²	N
1 $[P]_\lambda$	$[P]_\iota$	$P_\lambda = 2.78 \, . \, P_\iota^{.492}$	.179	.423	.340	82
2	$[P]_\iota$, τ_W	$P_\lambda = 1.55 \, . \, P_\iota^{.646}/\tau_W^{.304}$	.124	.352	.550	82
3	$[P]_\iota$, τ_W , $\bar{z}$	$P_\lambda = 1.188 \, P_\iota^{.653} \, . \, \bar{z}^{.101}/ \, \tau_W^{.342}$	.124	.352	.555	82
4	$[P]_\iota$, q_S	$P_\lambda = .601 \, . \, P_\iota^{.669} \, . \, q_S^{.373}$	.124	.352	.549	82
5	$[P]_\iota$, q_S , $\bar{z}$	$P_\lambda = .967 \, . \, \underline{P}^{.691} \, . \, q_S^{.360}/\bar{z}^{.222}$	.115	.339	.588	82
6 $[P]_\iota/[P]_\lambda-1$	τ_W	$P_\lambda/P_\iota = 1/(1 + 1.126 \, \tau_W^{.648})$	.304	.552	.493	63
7	τ_W , $\bar{z}$	$P_\lambda/P_\iota = 1/(1 + 1.573 \, \tau_W^{.701}/\bar{z}^{.142})$	.307	.554	.498	63
8	q_S	$P_\lambda/P_\iota = 1(1 + 6.411/q_S^{.747})$	.330	.574	.451*)	63
9	q_S , $\bar{z}$	$P_\lambda/P_\iota = 1/(1 + 1.685 \, . \, \bar{z}^{.504}/q_S^{.690})$	.279	.529	.543	63

*) Most uncertain model.

modifying factor determining the inlake concentration. Though not identical, this finding is in principle in agreement with the approximation made of the RHS of Eq. (3) for the elaboration of the OECD data, using as standard flushing correction term $1/(1 + \sqrt{T(w)})$.

However, there is no reason why the water residence time should play such an overriding role. The internal reasons for this are twofold.

a) Assuming total phosphorus load of equal magnitude ($[P]_\iota \times Q$ = Konst) for two lakes but different hydraulic load, the hydraulic load would affect both the average inflow concentration and the modifier (cf. e.g. 2).

b) There are other internal correlations between parameters, the effects of which tend to smooth out certain variations. The most important of this kind is the high correlation between water residence time and mean depth. Although for any single lake hydraulic load and mean depth are unique and independent features, if expressed in terms of water residence time ($T(w) = \bar{z}/q_S$), $T(w)$ and $\bar{z}$ <u>over a large spectrum of lakes</u> show a high degree of autocorrelation. Interestingly, the resulting regressions are non-linear (cf. Table A2.2) Further, the slopes of the regressions deviate significantly from 1 and 0, but not significantly from .5.

Therefore, substituting the mean depth term in Eq. (3), one obtains the following equations.

Table A2.2 Statistical relationship between mean depth and water residence time.

	No. of Observations	Log transformed regression equation	r	Confidence level for slope to deviate from 0	1	.5
OECD Data	86	$\log \bar{z} = 1.055 + .465 \log T(w)$	.579	sign.	sign.	not sign.
Canadian Data	108	$\log \bar{z} = .9958 + .366 \log T(w)$	.655	sign.	sign.	not sign.
All Data	194	$\log \bar{z} = 1.027 + .405 \log T(w)$	.620	sign.	sign.	not sign.

$$[P]_\lambda/[P]_\iota \approx \frac{1}{1 + (\frac{\bar{s}}{a}) \cdot T(w)^{(1-b)}} , \quad (4)$$

and considering that $\bar{s}/a$ and (1-b) are close to 1*) and .5, respectively, Eq. (4) can be approximated by

$$[P]_\lambda/[P]_\iota \approx \frac{1}{1 + \sqrt{T(w)}} \quad (5)$$

In straightforward statistical terms, the procedure is open to criticism. Apart from the simplifications made in Eq. (5), the problem of interpreting the real significance of whatever formulation is considered, can be shown from the results of the elaboration of the OECD data using Eq. (5). According to the resulting regression equation for predicting inlake concentration, i.e.

$$[P]_\lambda \approx 1.55 \cdot \{[P]_\iota/(1 + \sqrt{T(w)})\}^{.82} \quad (6),$$

it would appear that, if Eq. (6) is expressed as

$$[P]_\lambda/[P]_\iota = 1.55 \cdot \frac{1}{[P]_\iota^{.18}} \cdot (\frac{1}{1 + \sqrt{T(w)}})^{.82} \quad (6a) ,$$

the $[P]_\lambda/[P]_\iota$ ratio depends also on the inflow concentration. The flaw of such an inference can be shown from the Canadian data (cf. Figure A2.1)

*) Vollenweider (1975, 1976) and Chapra (197) have shown that $\bar{s}$, the apparent settling velocity, is normally between 10 to 15 m/y, though the variability can be greater; a, the other constant, results as close to 10 m, so that $\bar{s}/a \approx 1$ to 1.5.

Although the spread of the data lies within the boundaries calculated for inflow concentrations from $[P]_\iota = 1$ to 1000 mg/m^3, scrutiny of the data reveals no relationship to the postulated dependency of the $[P]_\lambda/[P]_\iota$ ratio on $[P]_\iota$.

However, the success of using $1/(1 + \sqrt{T(w)})$ as a standard flushing correction still justifies its use for the near future until the total material now available is re-elaborated to reflect more realistically the statistical connection between the different parameters.

LIST OF FIGURES

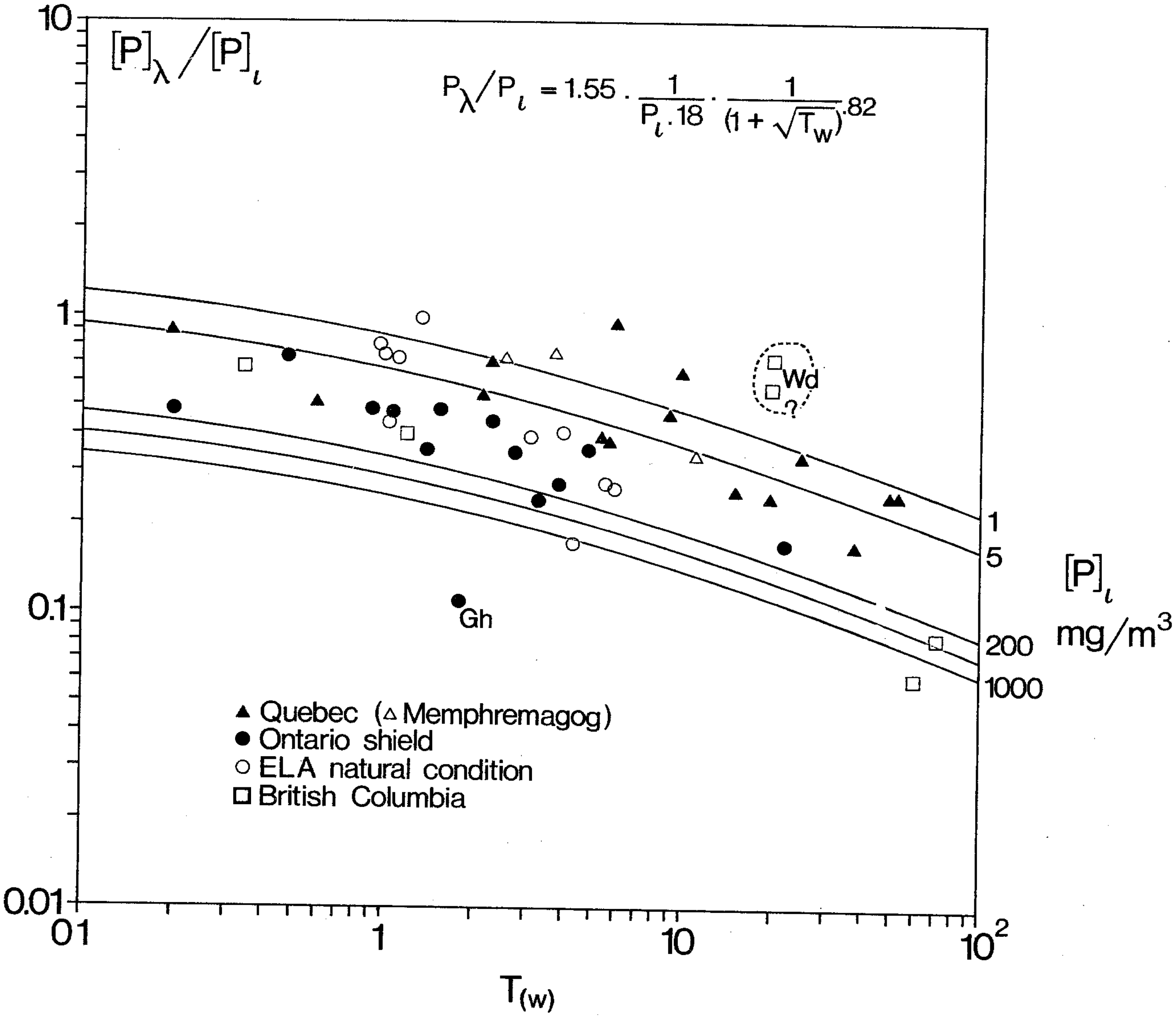


Figure A2.1 Annual mean total phosphorus lake concentration/inflow concentration ratio in relation to theoretical water residence time

APPENDIX 3

Hypolimnetic Oxygen Depletion Models

The following is a condensed version of a study to predict column averaged monthly oxygen depletion rates ($gO_2/m^3.mo$) at the basis of a few easily available parameters. The models have been developed using OECD material including Canadian data. Details of the development will be reported elsewhere.

The essential assumptions made in these models are that oxygen depletion rates depend

a) on biomass, as measured by yearly averages of chlorophyll concentrations of the euphotic zone;

b) on the ratio between euphotic and tropholytic zone;*)

c) on average temperature of the hypolimnion during summer stagnation.

These features are essentially the same as those discussed by Charlton (1980), but in difference to Charlton's approach, depletion rates are expressed here on a volumnar instead of on an areal basis. Volumnar depletion rates are considered as limnologically more meaningful

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*) Note the distinction made here between euphotic and tropholytic zone as opposed to epilimnion and hypolimnion. The former are defined by optical, the latter by thermal gradients.

A/70

than areal rates. In regard to temperature effects, we have adopted the standard correction proposed by Charlton; depletion rates have been normalized to 4°C by dividing measured rates by $2^{0.1\,(\theta-4)}$, θ being the average hypolimnetic temperature over the stagnation period. In practice this was only necessary for Lake Erie Central and Eastern Basin.

From several models of increasing complexity tested, two have been found most useful. In the first, a function of (x) has been defined which accounts for the shape of the curvet of the hypolimnion by multiplying yearly average chlorophyll with a modifier as follows

$$f(x) = [\overline{Chl}] \left(\frac{z_e/4.6}{\bar{z}\,(1-z_e/z_{max})^{\zeta}} \right) \qquad (1)$$

ζ being $z_{max}/\bar{z}$ (which is a measure for the concavity; cf. Hutchinson 1957), and z_e being the mean depth of the euphotic zone defined as $4.6/\bar{\varepsilon}$. The vertical extinction coefficient ($\bar{\varepsilon}$) can be split into two parts, i.e. $\bar{\varepsilon} = \varepsilon_w + \eta\,[chl]$, the basic water extinction coefficient ε_w (which includes all non-phytoplankton components), and a component due to phytoplankton. Referred to chlorophyll, η is in the order of 0.02 ± 0.004 per mg chlorophyll/m^3 per meter. For lack of measured extinction coefficients, ε_w has been assumed to be .3 on average.

In principle, the same modifier has been used in the second model but setting $\zeta = 1$. This is equal to taking just the mean depth z_h of the trophplytic zone instead of a volume function. Accordingly, f(x) becomes

$$f(x) = [chl] \left(\frac{z_e/4.6}{\bar{z}\ (1 - z_e/z_{max})} \right) \qquad (2)$$

which is a simplification of (1), and reduces calculation.

For model calibration, measured monthly depletion rates have been regressed against the functional values $f(x)$ calculated for each lake testing a number of linearized family regression functions. In both cases, three regression models, i.e.

1. $\Delta\ O_2\ (g/m^3.mo) = A + B\ \ f(x)$
2. $\Delta\ O_2 = A + B\sqrt{f(x)}$
3. $\Delta\ O_2 = A\ [f(x)]^B$

gave the highest r^2 and F values, and the lowest standard errors. These, and the corresponding A's and B's are listed in Table A3.1.

Model class I explains up to 95.5% of the variations of the y's; though somewhat less precise, model class II also explains up to 91% of the variations, and is still acceptable. If the model structure were perfect, then the model 1. should give the best output with $A \approx 0$, and B close to 1. However, experience shows that model 3. is a better predictor over the total range covering almost 3 orders of magnitude, and model 2. is intermediate. Model 1, while equally well predicting depletion rates over the upper part of the range, tends to over-estimate depletion rates

Table A3.1 Hypolimnetic oxygen depletion rate model.

Family regression model 1. $= A + B\ f(x)$

2. $= A + B \sqrt{f(x)}$

3. $= A\ [f(x)]^B$

Model class I: $f(x) = \frac{[chl]}{\bar{z}} \left(\frac{z_e/4.6}{(1 - z_e/z_{max})^{\zeta}}\right)$

Model class II: $f(x) = \frac{[chl]}{\bar{z}} \left(\frac{z_e/4.6}{1 - z_e/z_{max}}\right)$

$\bar{\varepsilon} = \varepsilon_w + \eta\ [chl]$; $z_e = 4.6/\bar{\varepsilon}$

$\zeta = z_{max}/\bar{z}$

Model	A	B	r^2	S(y/x)	F
I. 1.	.218	.940	.940	.167	299
2.	- .200	1.488	.955	.145	402
3.	1.232	.672	.936	.172	272
II. 1.	.203	1.132	.907	.204	195.4
2.	- .235	1.659	.909	.202	199.6
3.	1.355	.683	.900	.216	180.9

at low levels, whereas model 2. tends to under-estimate prediction rates at lower levels (cf. Figures A3.1, A3.2 and A3.3). Therefore, it is recommended that all three family models be used, and the average from all predictions taken.

In using these models, it has to be recalled that - if the average hypolimnetic temperature during the stratification period deviates essentially from 4°C, an appropriate temperature correction has to be made by multiplying the oxygen depletion rate predicted by the models with the correction factor given above.

Further, the models are sensitive to ε_w. This is the case when larger amounts of humic substance and/or mineral turbidity are present, and therefore ε_w deviates substantially from 0.3. In this case, appropriate corrections have to be made.

From preliminary testing of the models on lakes other than those included in the data base, it appears that the models should be applied with care if $\bar{z} \leq 8$ m, $z_{max} \leq 15$ m. However, a more stringent criterion seems to be the mean depth of the tropholytic zone which should not be less than 2 m. z_h can be estimated from

$$z_h = \bar{z}\,(1 - z_e/z_{max}) = \bar{z}\,\left(1 - \frac{4.6}{\varepsilon\, z_{max}}\right)$$

LIST OF FIGURES

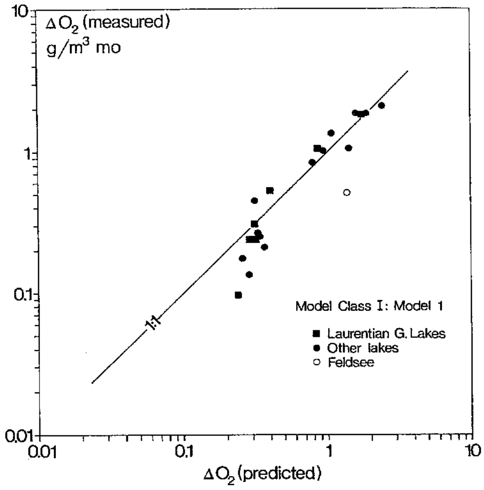

Figure A 3.1 Measured oxygen depletion rate vs predicted oxygen depletion rate; Model Class I: model 1

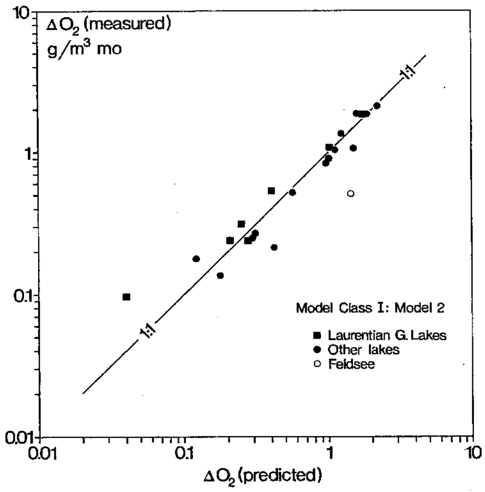

Figure A 3.2 Measured oxygen depletion rate vs predicted oxygen depletion rate; Model Class I: model 2

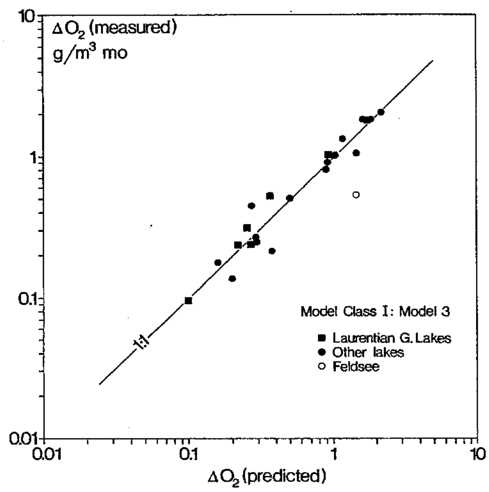

Figure A 3.3 Measured oxygen depletion rate vs predicted oxygen depletion rate; Model Class I: model 3

APPENDIX III 1

BEDROCK GEOLOGY OF ONTARIO SHIELD LAKES

- from P. Dillon, 1981

LIST OF FIGURES

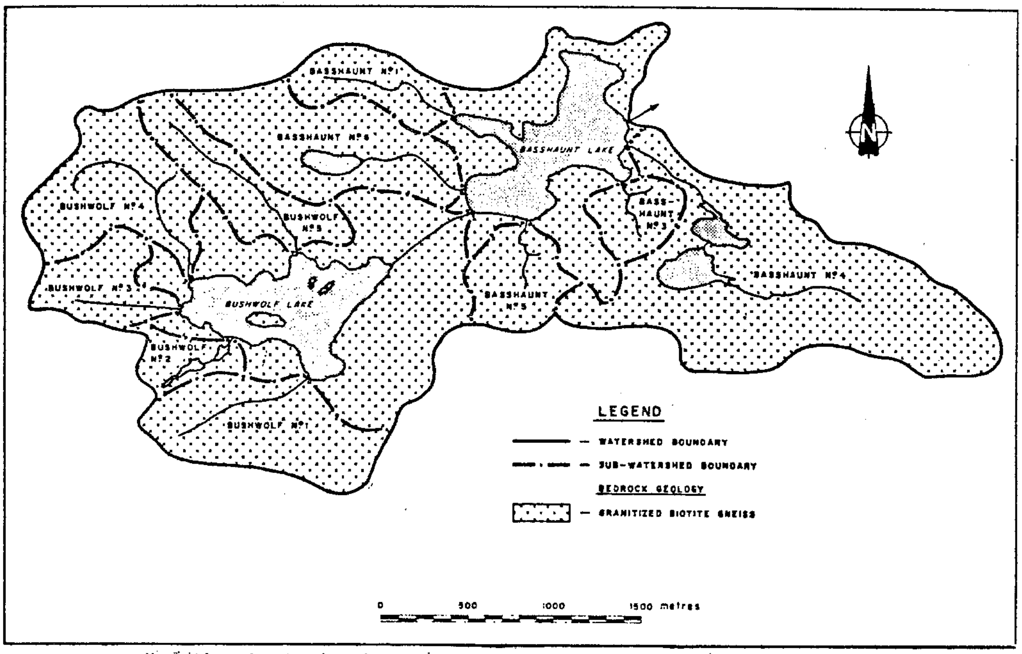

Figure III 1.1 Bedrock geology and basin designation of the Basshaunt Lake watershed

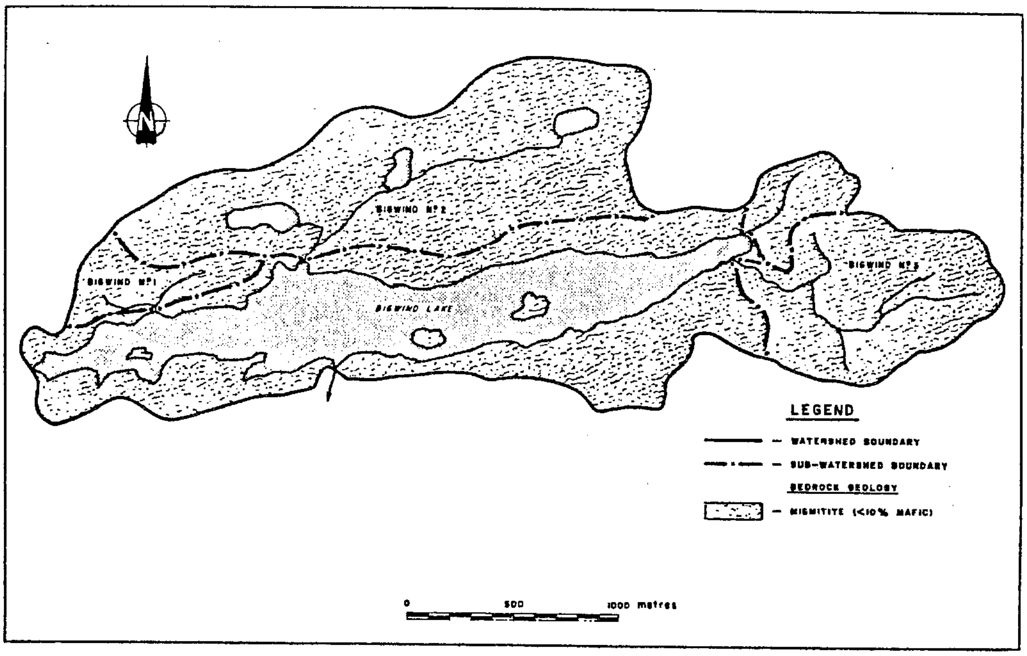

Figure III 1.2 Bedrock geology and basin designation of the Bigwind Lake watershed

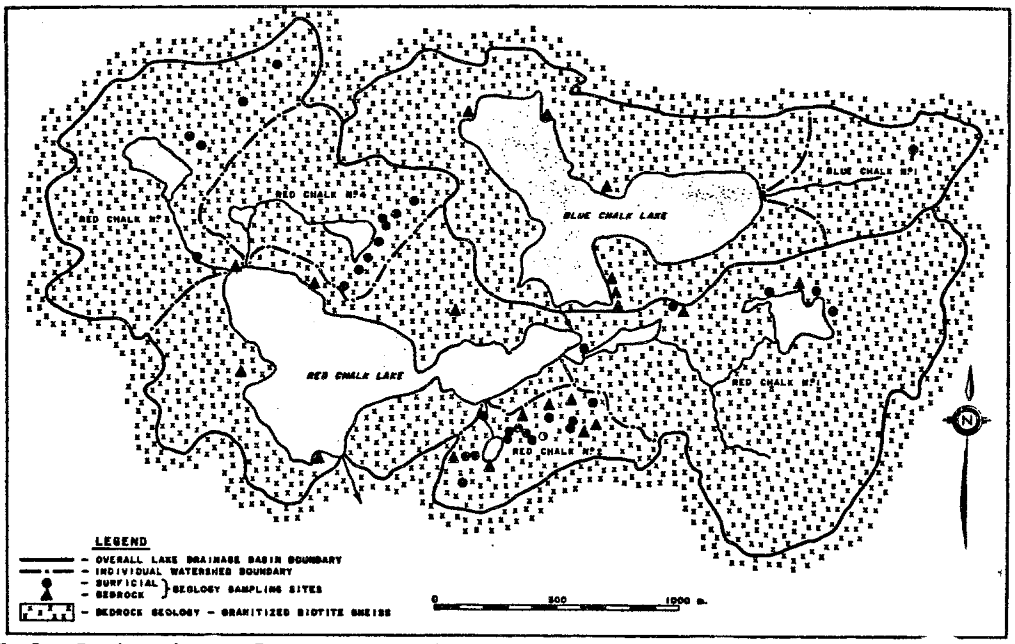

Figure III 1.3 Bedrock geology, sampling sites and basin designation of the Blue Chalk and Red Chalk Lakes drainage basin

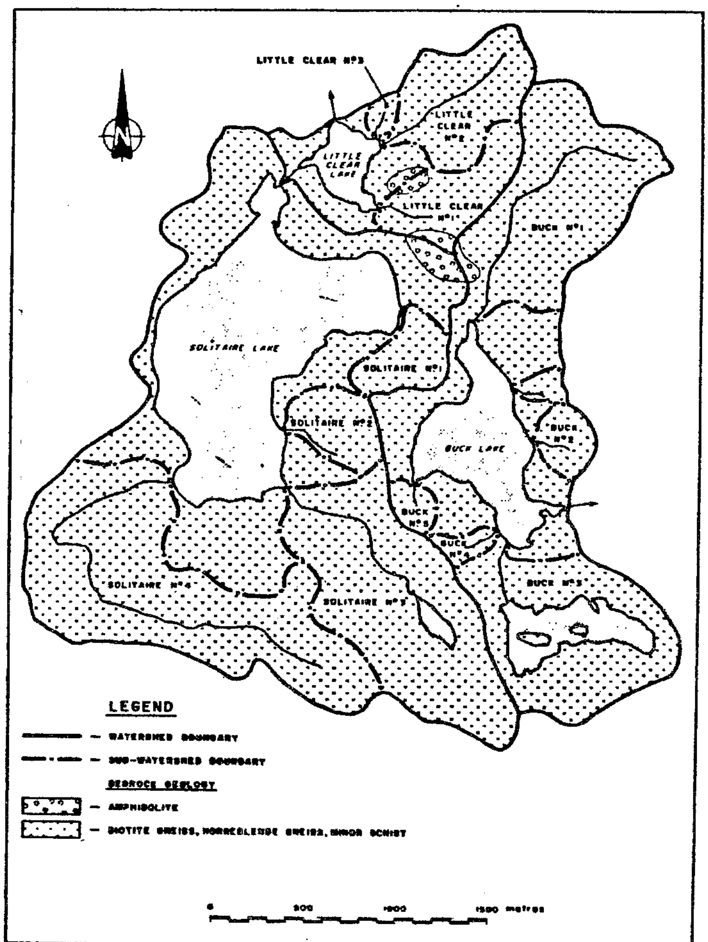

Figure III 1.4 Bedrock geology and basin designation of the Solitaire, Buck and Little Clear Lake watersheds

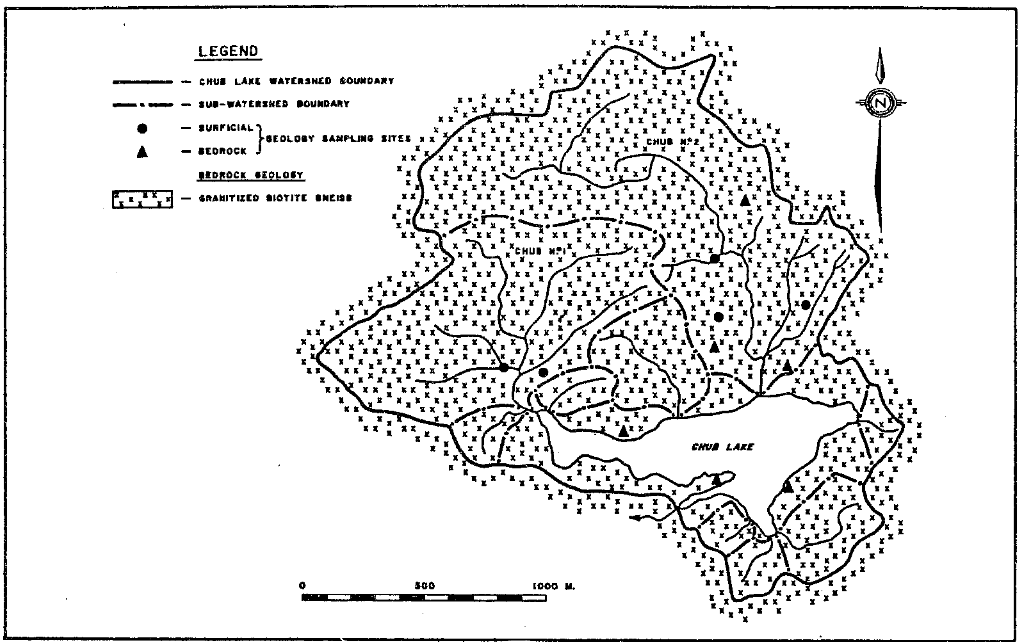

Figure III 1.5 Bedrock geology, sampling sites and basin designation of the Chub Lake watershed

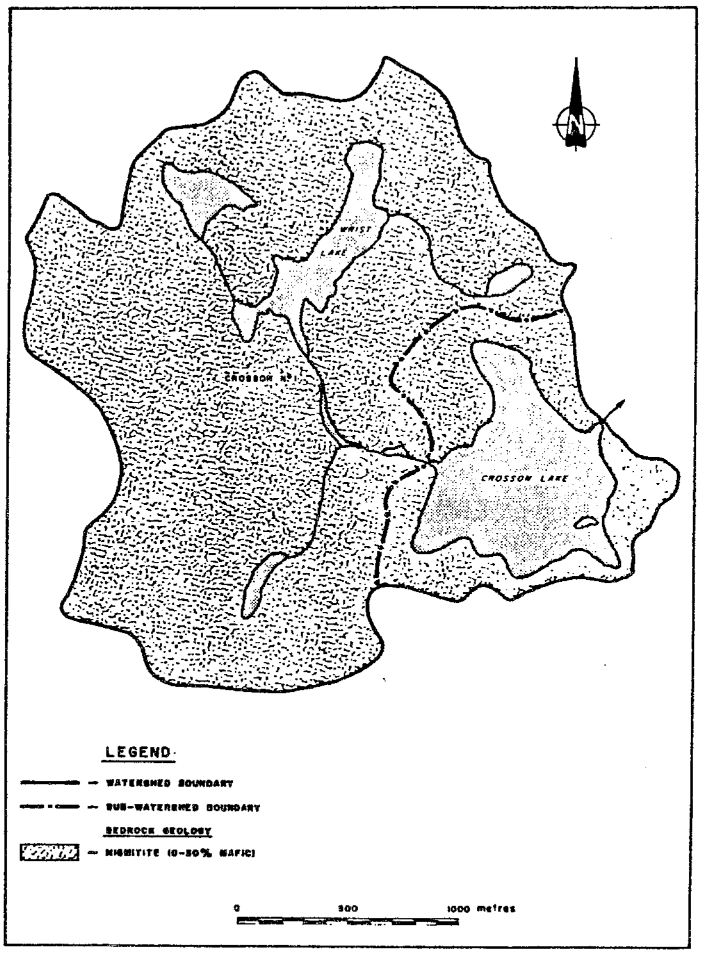

Figure III 1.6 Bedrock geology and basin designation of the Crosson Lake watershed

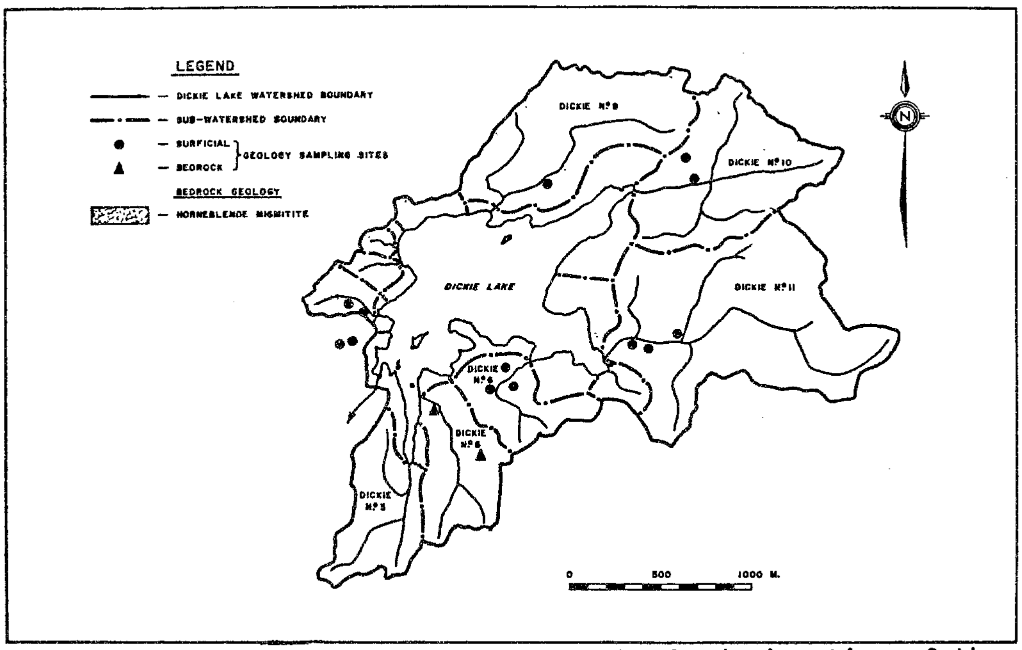

Figure III 1.7 Bedrock geology, sampling sites and basin designation of the Dickie Lake watershed

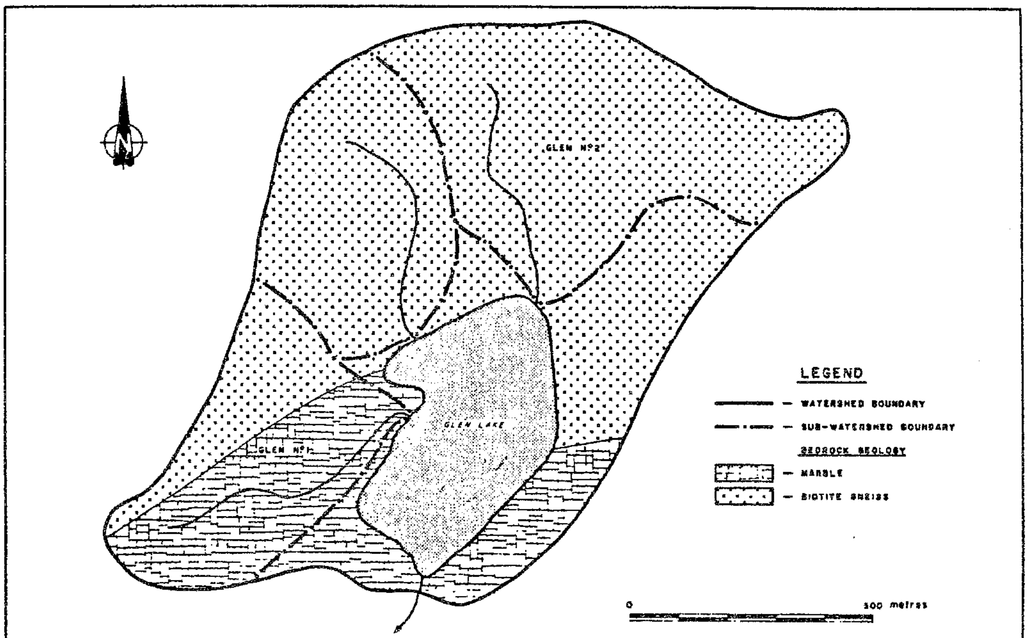

Figure III 1.8 Bedrock geology and basin designations of the Glen Lake watershed

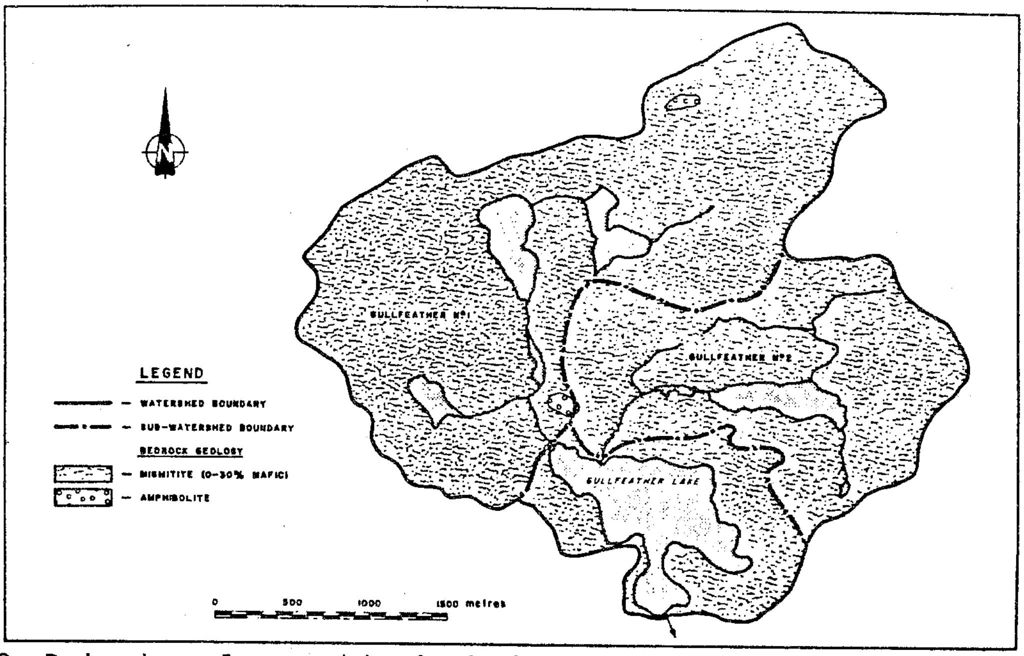

Figure III 1.9 Bedrock geology and basin designation of the Gullfeathers Lake watershed

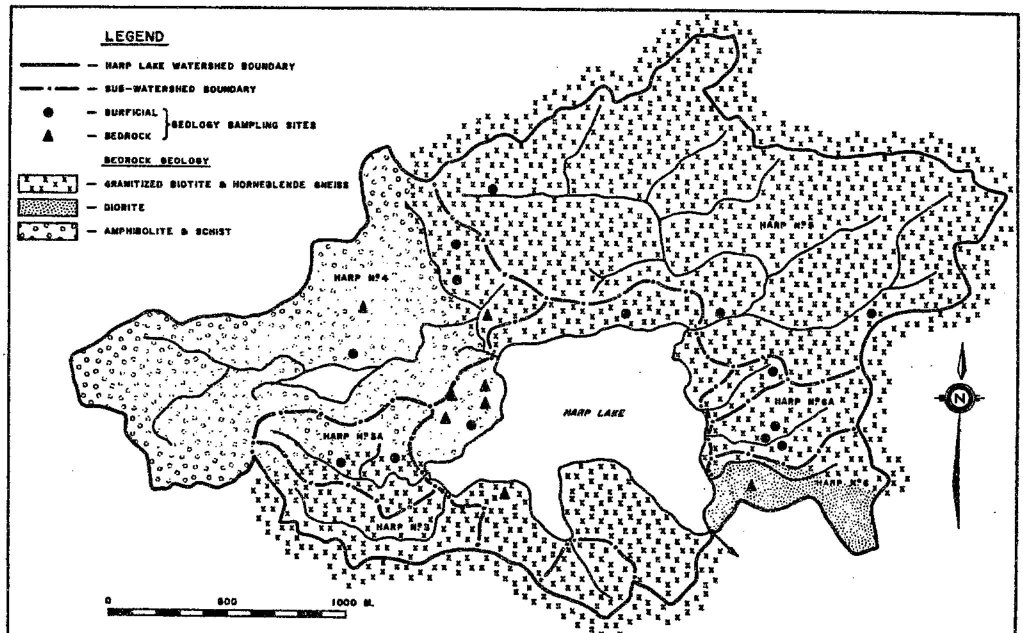

Figure III 1.10 Bedrock geology, sampling sites and basin designation of the Harp Lake watershed

A/83

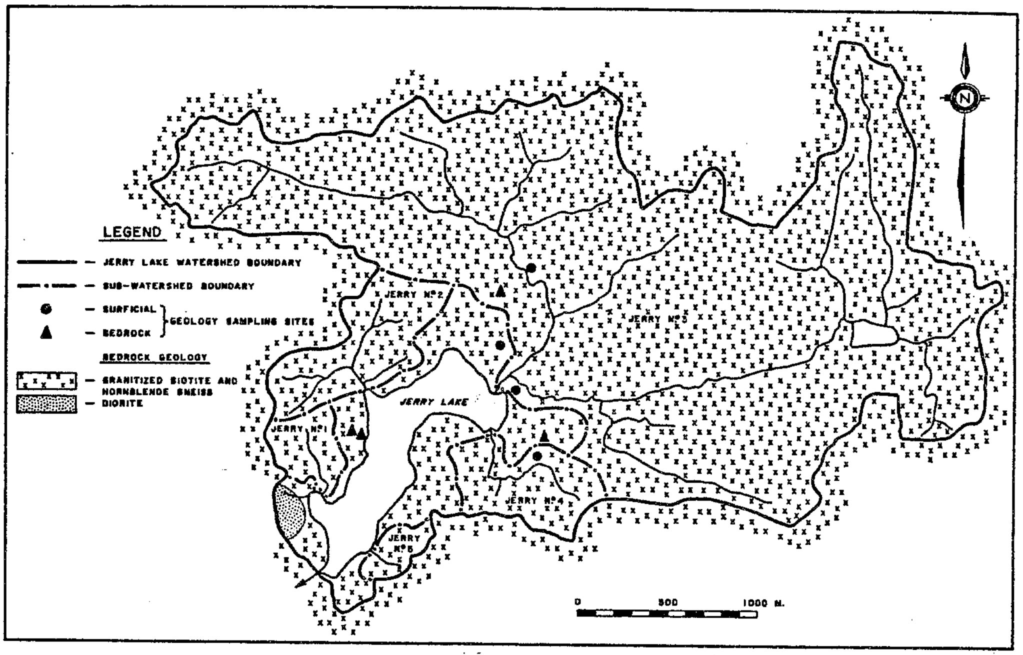

Figure III 1.11 Bedrock geology, sampling sites and basin designation of the Jerry Lake watershed

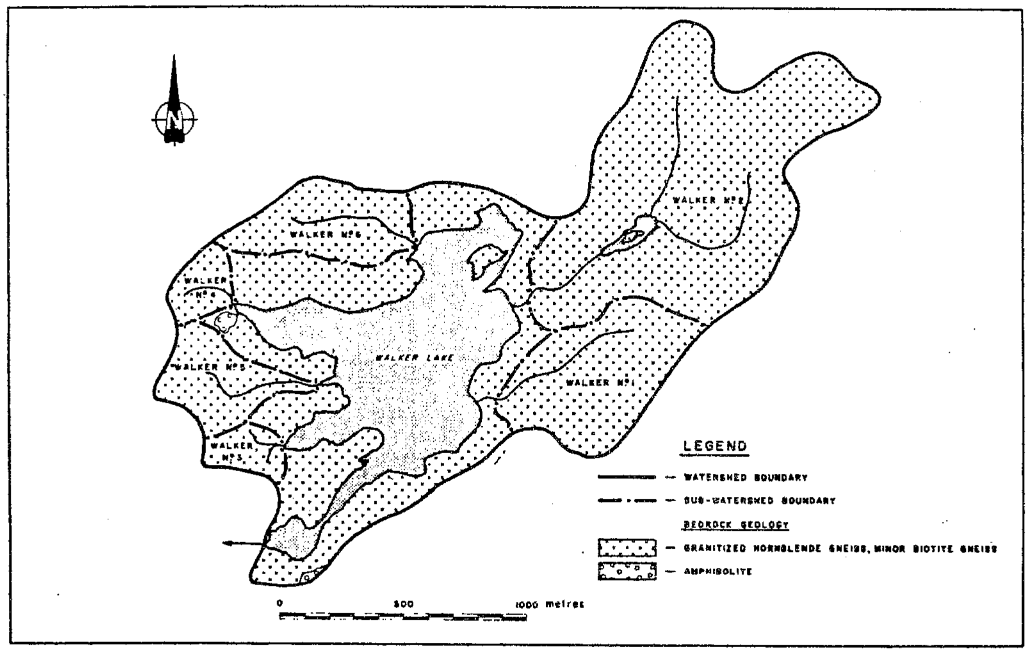

Figure III 1.12 Bedrock geology and basin designation of the Walker Lake watershed

APPENDIX III 2
SURFICIAL GEOLOGY OF ONTARIO SHIELD LAKES

- P. Dillon, 1981

SURFICIAL GEOLOGY KEY TO VEGETATION TYPE

Map Unit	Vegetation
thin till and rock ridges -	yellow birch poplar pockets of pine, hemlock
minor till plains -	maple ironwood beech
peat over sand -	hemlock balsam fir sphagnum some spruce
peat over clay over sand -	black spruce sphagnum
peat over bedrock -	
bedrock -	moss pine
outwash sand and gravel -	hardwood white pine
beach sand -	pine brambles white pine

LIST OF FIGURES

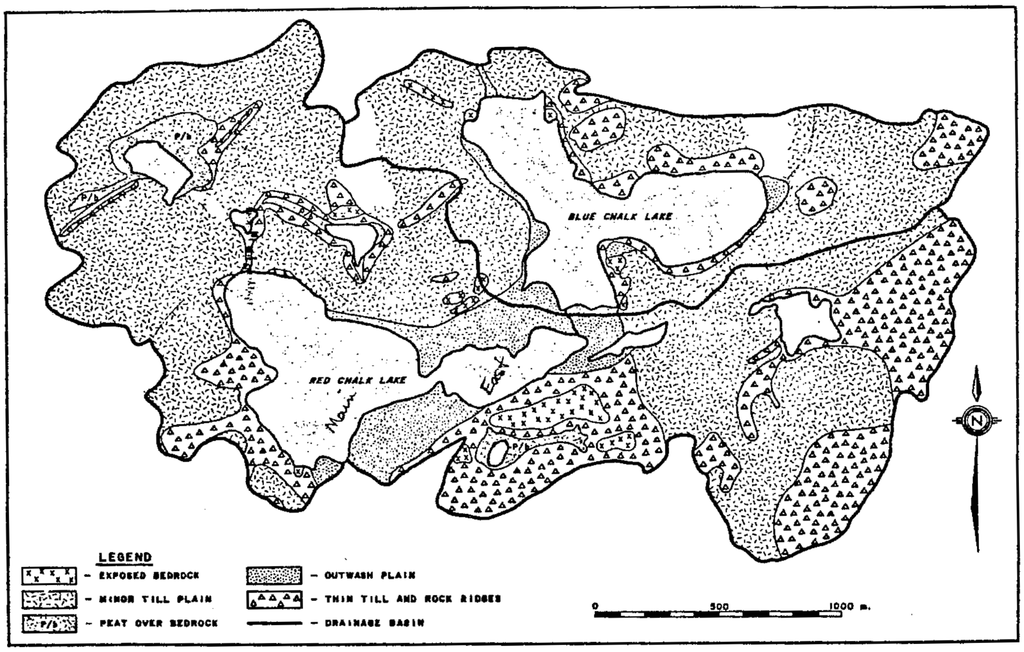

Figure III 2.1 Surficial geology of the Blue Chalk and Red Chalk Lakes watershed

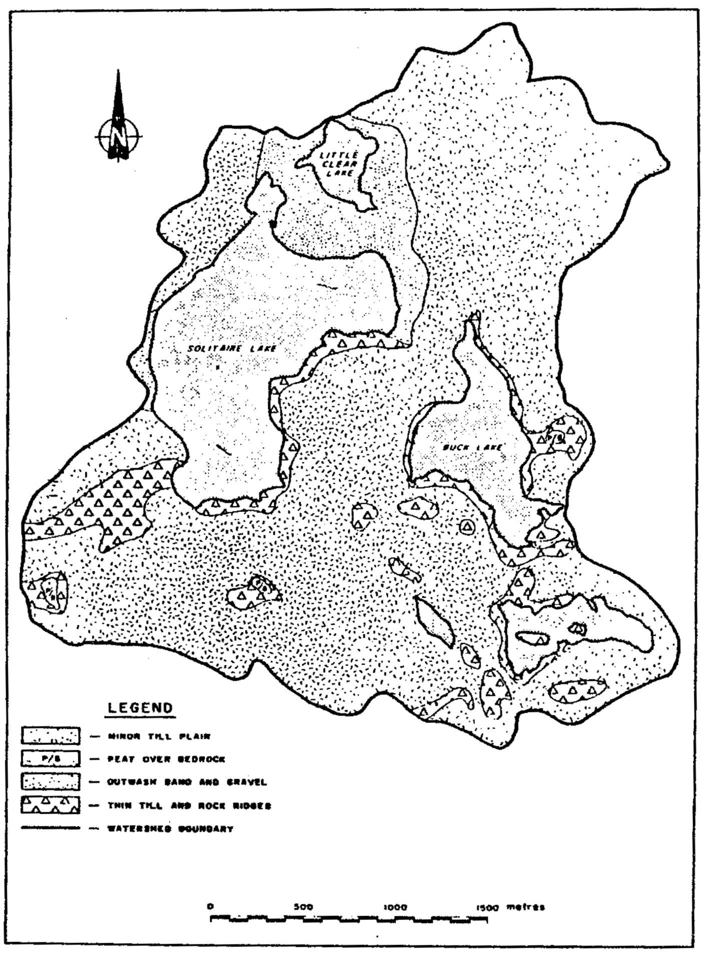

Figure III 2.2 Surficial geology of the Solitaire, Buck and Little Clear Lakes watershed

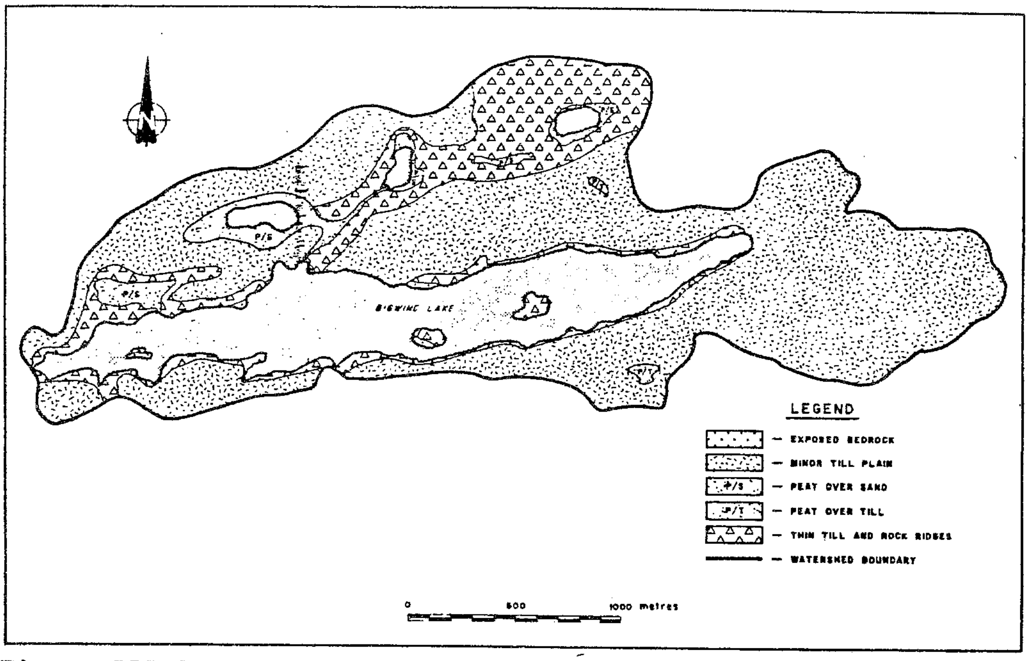

Figure III 2.3 Surficial geology of the Bigwind Lake watershed

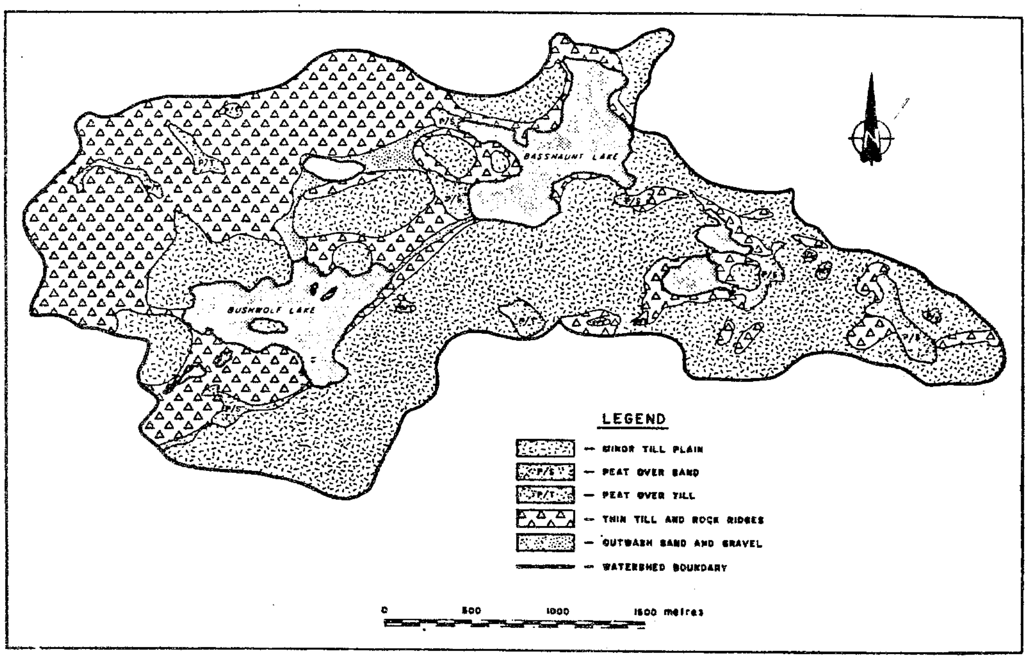

Figure III 2.4 Surficial geology of the Basshaunt Lake watershed

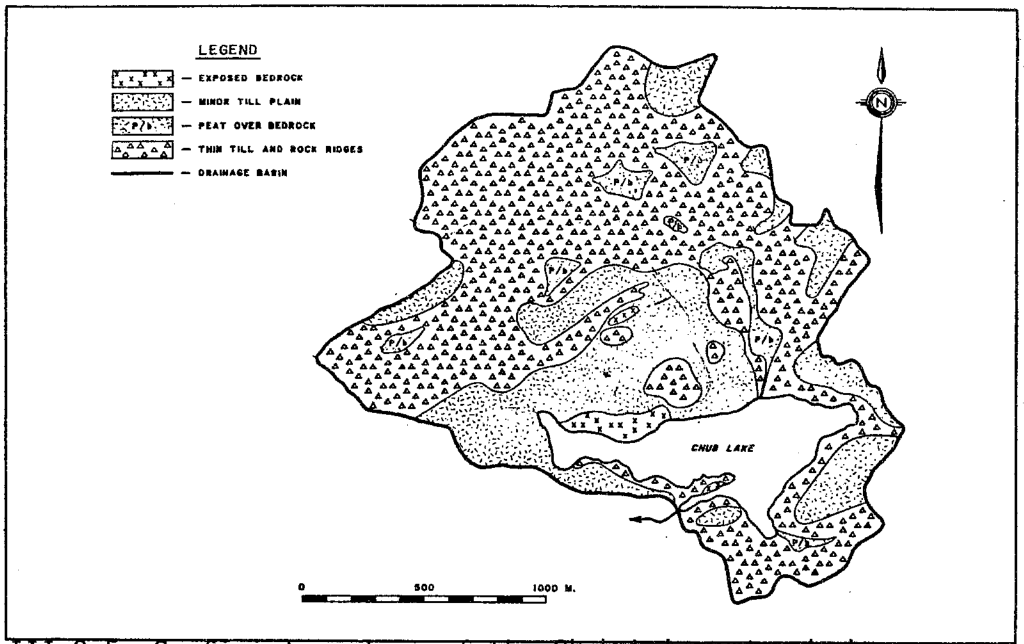

Figure III 2.5 Surficial geology of the Chub Lake watershed

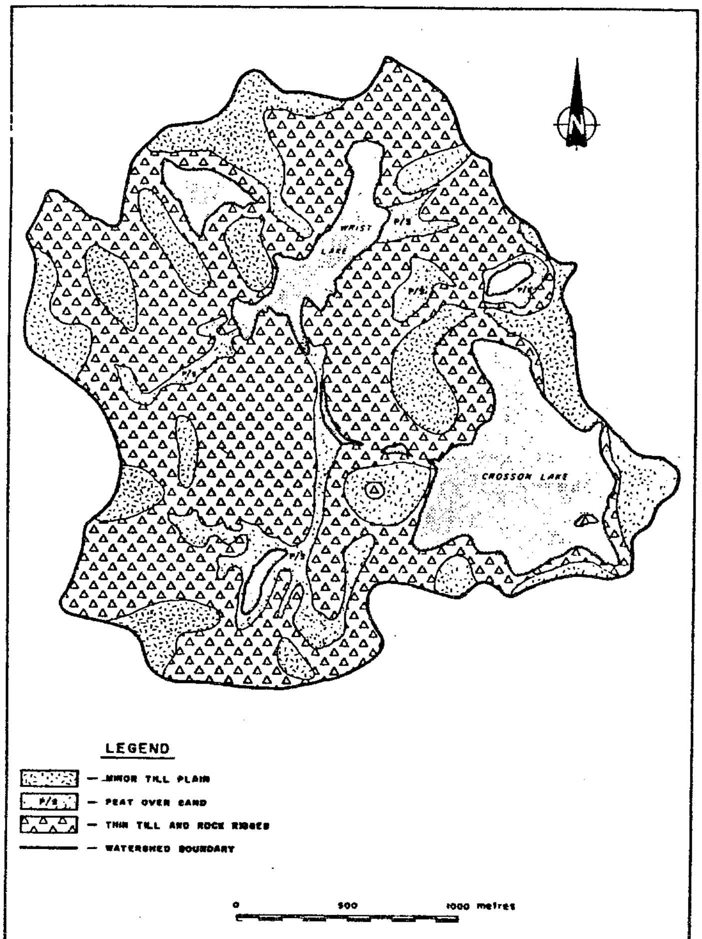

Figure III 2.6 Surficial geology of the Crosson Lake watershed

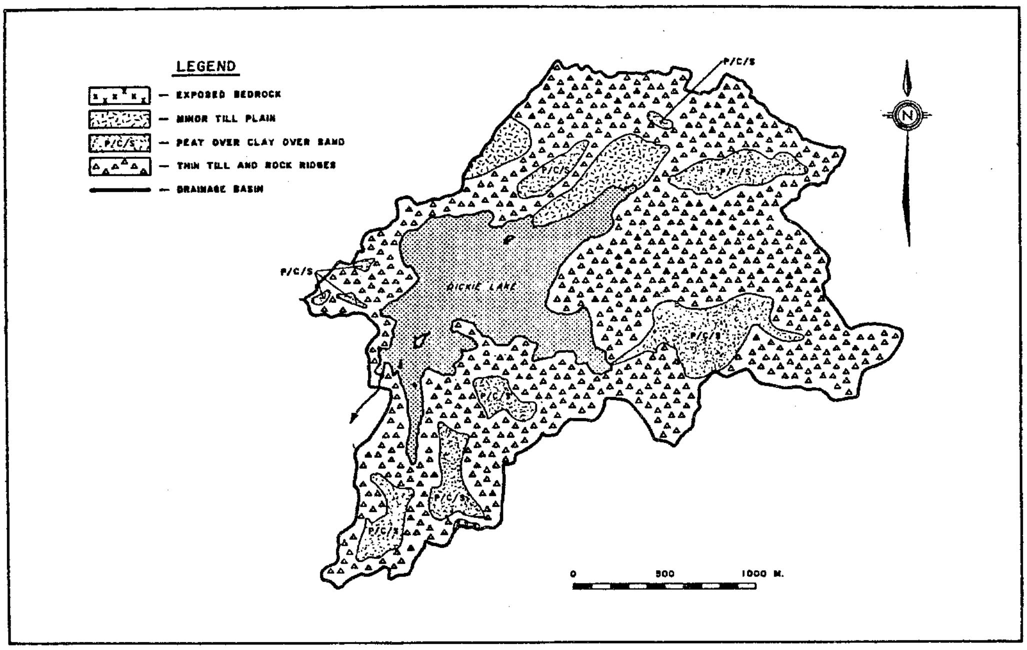

Figure III 2.7 Surficial geology of the Dickie Lake watershed

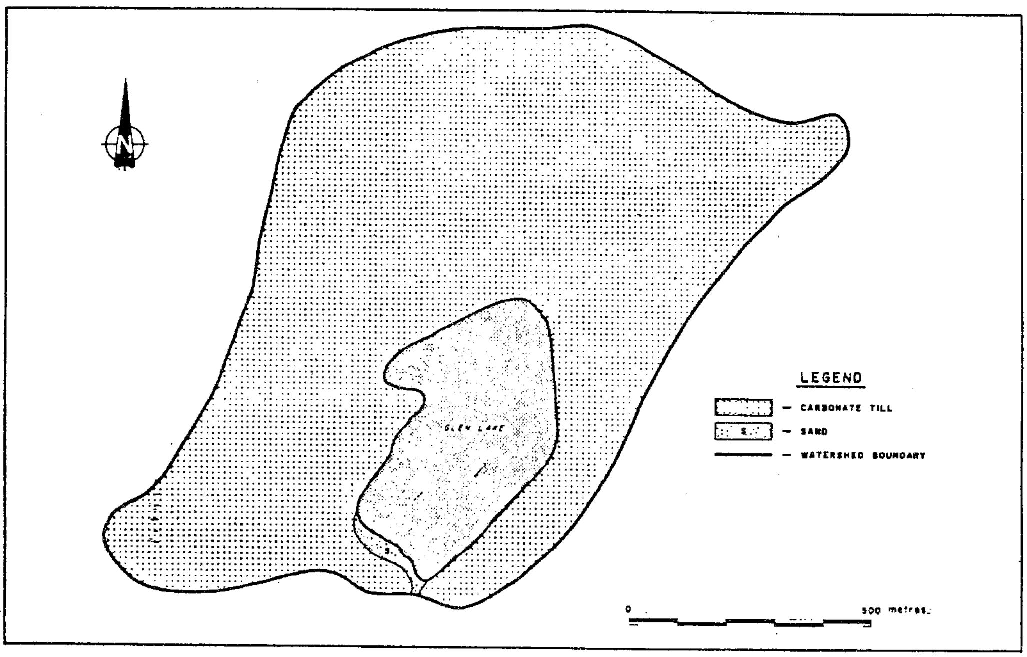

Figure III 2.8 Surficial geology of the Glen Lake watershed

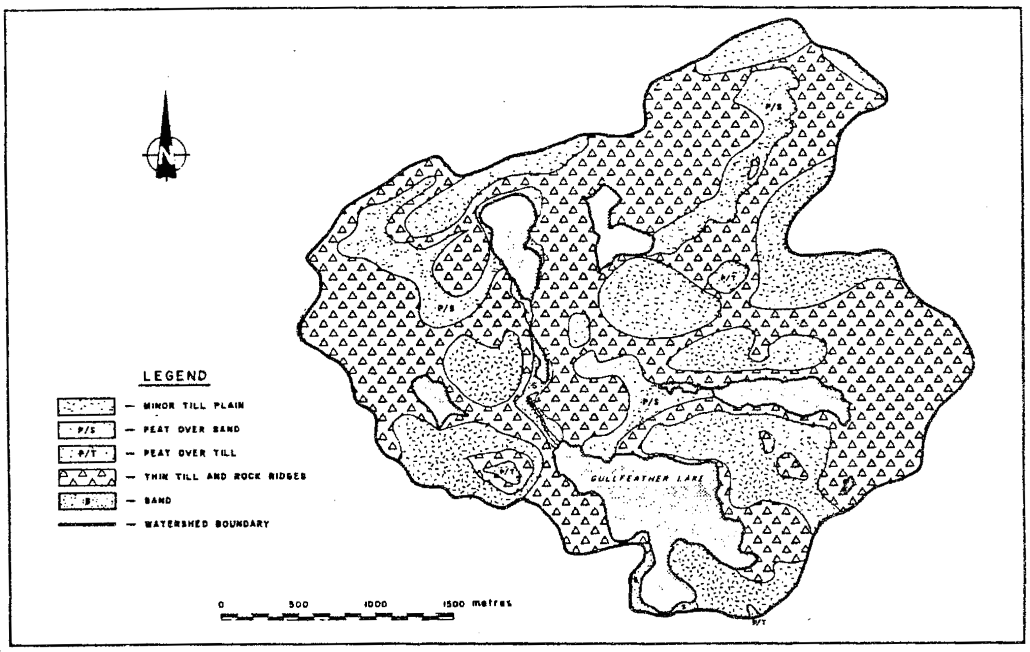

Figure III 2.9 Surficial geology of the Gullfeather Lake watershed

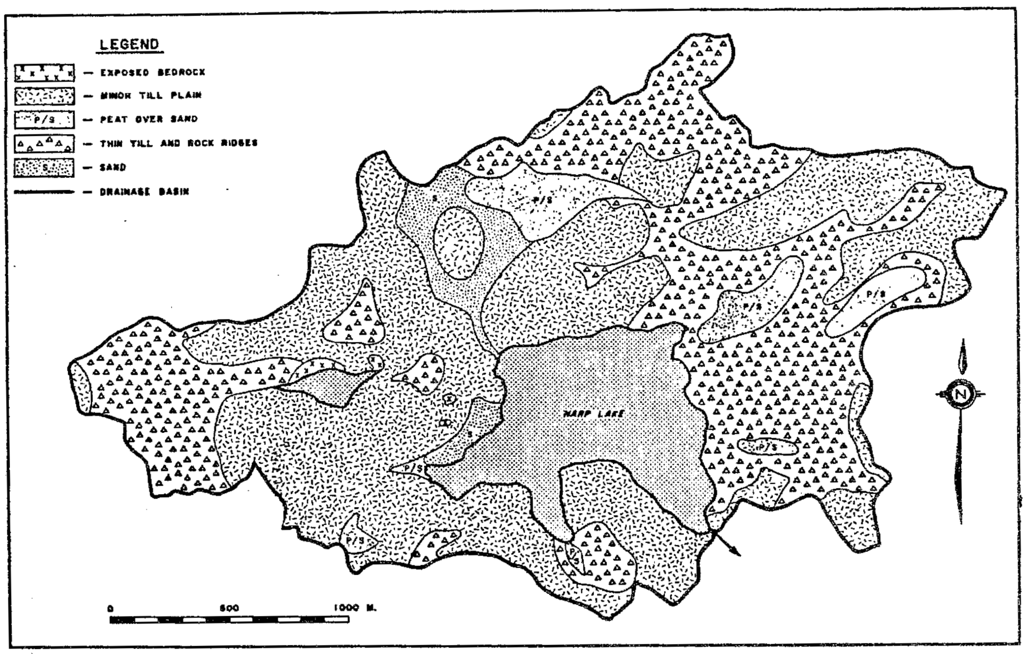

Figure III 2.10 Surficial geology of the Harp Lake watershed

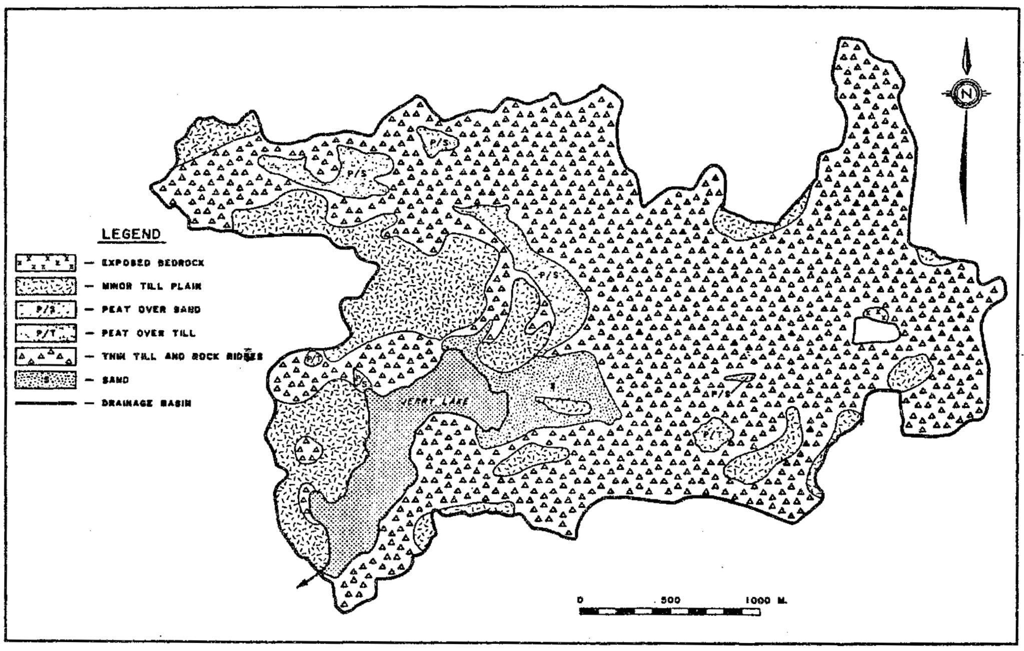

Figure III 2.11 Surficial geology of the Jerry Lake watershed

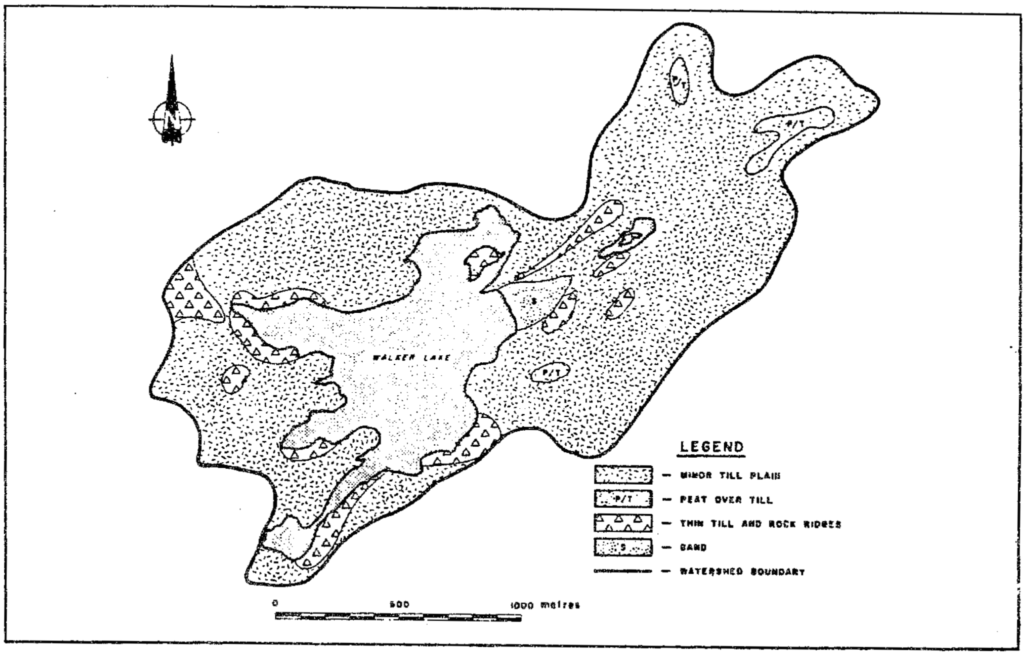

Figure III 2.12 Surficial geology of the Walker Lake watershed

APPENDIX V 1

Table V 1. Theoretical water renewal times for some ELA lakes (individual years).

Lake #	1970	1971	1972	1973	1974	$\bar{\tau}_w$ [1]
114	-	1.49	1.71	1.62	-	1.61
120	-	-	-	4.57	-	4.57
223	-	-	-	-	-	(3.3)
224	-	-	-	-	-	(12.5)
225	-	-	-	-	-	(0.9)
226	-	-	5.17	4.23	3.99	4.46
227	1.43	2.17	2.63	2.63	2.91	2.35
239	4.01	4.39	6.01	5.66	4.45	4.9
240	.97	1.03	1.37	1.16	1.06	1.12
261	-	-	1.34	-	.77	1.06
303	-	.79	1.19	1.09	.94	1.0
304	-	-	-	-	-	(1.8)

1) From Newbury and Beaty 1980 (1970 to 1974 means)

( ) data 1975 - 1977

APPENDIX V 2

Table V 2. Yearly loadings of P and N to ELA lakes ($mg/m^2/yr$).

	1969	1970	1971	1972	1973	1974
Lake #	Lp / Ln	Lp / Ln	Lp / Ln	Lp / Ln	Lp / Ln	Lp / Ln
114						100/1010
120						
223						58/1110
224						40/ 760
225						
226 NE			83/1230		637*/4070*	637*/4070*
226 SW					90/4540*	90/4540*
227	410*/6100*	582*/7900*	585*/7790*	528*/7350*	590*/7220*	590*/7220*
239		62/1210	96/1460	53/1330	46/1050	46/1050
240	82/1770	139/2230	69/1720	61/1820	61/ -	
261			99/1500	57/1170	307*/1070	310*/1080
303		52/1040	85/1270			

* artificial enrichments