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NUTRIENT LOADING - TROPIC STATE RELATIONSHIPS IN FLORIDA LAKES

by

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ABSTRACT

Quantitative relationships among important lake trophic state indicators and watershed enrichment factors were examined using a data base of 101 Florida lakes. Trophic state information was obtained from three previous surveys on Florida lakes, from which data were compiled into a uniform format. Watershed nutrient export data and lake nutrient loading rate data were obtained from a comprehensive review of the literature and from the Florida portion of the National Eutrophication Survey (NES) conducted by the U.S. EPA. This data base was used to determine relationships between non-point source (NPS) nutrient loading rates and land use characteristics of Florida watersheds; to evaluate interrelationships among trophic state indicators in Florida lakes; and to revise nutrient loading models and develop appropriate nutrient loading criteria for these lakes.

The magnitude of NPS nutrient loadings was estimated from published export coefficients and by a statistical analysis of the NES watersheds. The literature-based approach produced a wide range of export coefficients for Florida watersheds. 0.2-0.7 kg P/ha-yr and 1.5-6.1 kg N/ha-yr for forests; 0.4-2.4 kg P/ha-yr and 2-50 kg N/ha-yr for cropland; 0.2-4.7 kg P/ha-yr and 1.5-7.4 kg N/ha-yr for residential areas; and 0.3-7.5 kg P/ha-yr and 3-10 kg N/ha-yr for urban areas. NPS nutrient loading (dependent variables) and land use characteristics (independent variables) for 41 NES watersheds were analyzed by stepwise multiple regression to improve the predictive capability of the land use-nutrient loading approach. For phosphorus, a model using three land terms (cropland, forest & rangeland) explained 72% of the variance in NPS loading (vs 21% for a model with drainage area as the sole independent variable). Models to predict NPS nitrogen loading and hydraulic flow had high levels of predictability using drainage area as the sole independent variable ($r^2 = 0.84$ and 0.91, respectively); and inclusion of land use data resulted in little predictive improvement.

Evaluation of the limnological characteristics of 101 Florida lakes indicated that most of these lakes are shallow and well-mixed; few have stable thermoclines or anoxic hypolimnia. The lakes are highly variable in alkalinity (0-16 mg/L as $CaCO_3$), pH (4.7 - >10) and amount of color (2-54 CPU), reflecting differences in geological origin and watershed characteristics. The majority of the lakes in the data base are eutrophic, and unlike most temperate lakes, tend to be nitrogen-limited (46% had SIN:SRP ratios of <10:1). For a given level of total phosphorus, Florida lakes have less chlorophyll a than do temperate lakes; this is true even for phosphorus-limited Florida lakes.

Carlson's trophic status index (TSI) was modified for application to Florida lakes by inclusion of a nitrogen index to reflect the importance of nitrogen as a limiting nutrient. A composite TSI was developed by averaging the TSI's based on Secchi disk, chlorophyll *a* and nutrient concentration (the smaller of the nitrogen or phosphorus index) to produce an index that reflects the multidimensionality of the eutrophication concept.

Various nutrient loading models were evaluated statistically for their ability to predict mean chlorophyll *a*, nitrogen and phosphorus concentrations in Florida lakes. The best predictions for nitrogen and phosphorus were made using a modified Dillon and Rigler-type model, while the best predictions of chlorophyll *a* were obtained using a Jones and Bachman-type model.

Existing phosphorus loading criteria were evaluated for the Florida NES lakes. The 1975 Vollenweider criteria and the 1975 Dillon criteria were revised to improve their predictive capabilities; in both cases the revisions resulted in higher critical values (minimum mesotrophic and minimum eutrophic loading rates) than the original criteria. Both sets of criteria were equally successful in delineating eutrophic lakes from mesotrophic lakes in the NES data base. Nitrogen loading criteria were developed using loading terms analogous to those used for phosphorus. The most successful nitrogen criteria were based on a Dillon-type model. In evaluating the impact of a proposed management strategy it is suggested that both nitrogen and phosphorus loading criteria be used; the correct response of the lake will be obtained with the criterion that predicts the lower trophic status.

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CHAPTER I. INTRODUCTION

Inland lakes are an important natural resource for Florida, and they are particularly valuable recreational assets. Unfortunately, the potential beneficial use of many Florida lakes has been impaired by accelerated eutrophication. Although the process of eutrophication is natural, the addition of plant nutrients from municipal sewage, septic tanks, urban and agricultural runoff, livestock operations, and industrial effluents accelerates the process by stimulating the growth of algae and macrophytes. Excessive levels of aquatic production result in general impairment of water uses. Under highly enriched conditions, blue-green algae tend to dominate the algal flora and may form dense, unsightly surface blooms that impart unpleasant tastes and odors to the water. In lakes that stratify, the decomposition of algae and macrophytes results in the depletion of dissolved oxygen in the bottom waters, and this in turn limits the diversity of benthic organisms and benthic feeding fish. The quality of fishing eventually decreases, as populations of game fish, such as bass and sunfish, are replaced by populations of rough fish (bullheads, shad, carp). Recreational boating may become restricted by thick beds of weeds. Contact sports are diminished by the reduced transparency of the water, the growth of water weeds, and the occurrence of infections in swimmers.

One of the most notable examples of cultural eutrophication in Florida is Lake Apopka, a 12,000 hectare (ha) lake once recognized for its exceptional bass fishery. Since the late 1940's, water quality in the lake has deteriorated as the result of discharges of municipal wastewater, citrus processing plant wastes, and muck farm irrigation water to the lake, in conjunction with ill-fated attempts to control the growth of water hyacinth and rough fish. As the result of these perturbations, populations of important game fish have been drastically reduced, and the dominant fish in the lake today are two species of shad (USEPA 1978). Cultural eutrophication has affected other lakes in the Oklawaha chain (Brezonik and Shannon 1971), Lake Okeechobee (MacGill et al. 1976), and numerous smaller lakes throughout the state. In order to manage Florida's lakes and control eutrophication, planners and regulatory agencies must be able to quantify nutrient loadings to lakes from their watersheds and to predict the response of individual lakes to changes in nutrient loading.

CONCEPTUAL FRAMEWORK FOR PREDICTIONS OF LAKE TROPHIC STATUS

The most widely used approach in predicting the trophic status of lakes has involved the use of simplified input-output (I/O) models (Dillon and Rigler 1974a; Chapra and Tarapchak 1976; Vollenweider 1968, 1969, 1975, 1976; Kratzer 1979). The I/O models are based on the assumption that a lake is a continuously-stirred tank reactor (CSTR) in which all nutrient fluxes are at steady state. Since phosphorus is the most common limiting nutrient in temperate zone lakes, most of these models have been developed to predict mean lake phosphorus concentrations in a lake. The utility of these models is enhanced by their modest data requirements, which include only morphologic and hydrologic parameters and areal phosphorus loading rates.

Phosphorus loading models have been further advanced by the recognition of statistically significant relationships between concentrations of total phosphorus and the concentration of chlorophyll a in lakes (Sakamoto 1965; Dillon and Rigler 1974). This development has led to the formulation of I/O models in which phosphorus loading can be used to predict the mean chlorophyll a concentration directly (Vollenweider 1976; Chapra and Tarapchak 1976; Kratzer 1979; Uttormark and Hutchins 1978). Finally, several authors have found strong correlations between Secchi disk transparency and chlorophyll a concentration (Carlson 1977, Brezonik 1978), enabling predictions of lake transparency from chlorophyll a data. A conceptual framework for prediction of lake trophic status based on I/O models is outlined in Figure I-1.

The need for nutrient loading data in the I/O models has prompted efforts to predict the non-point source (NPS) loading of nutrients from tributary watersheds based on watershed characteristics. The simplest approach for estimating NPS nutrient loading is based on relationships between land use and nutrient loading. Numerous studies have been conducted to determine the areal loading of nutrients from agricultural, forested and urban watersheds. Omernik (1976, 1977) used a statistical approach to evaluate nutrient export from watersheds as a function of land use. Reviews of areal nutrient export rates for various land use categories have been assembled by Loehr (1974), Uttormark et al. (1974) and Reckhow et al. (1980).

SCOPE AND OBJECTIVES OF THIS REPORT

This report builds on and updates the earlier work of Brezonik and Shannon (1971) in assessing nutrient loading-trophic response relationships and developing critical loading rate guidelines for Florida lakes. Since that early report, a large volume of data has been collected on several aspects of eutrophication in Florida. Several studies of non-point source nutrient loading have provided data on nutrient export rates for watersheds in various land uses, and additional data are available on the trophic status, nutrient budgets and water budgets for many Florida lakes.

During the period in which this additional data has become available the I/O models for predicting lake trophic status have been developed. While these models have been found to be quite useful for the temperate zone lakes in which they were developed, the applicability of the models and the nutrient loading criteria developed from them has not been evaluated for the warm temperate and subtropical Florida lakes. Thus, the objectives of this study are:

1) to examine the relationships between land use and nutrient export in Florida watersheds;

2) to examine the general limnological characteristics of Florida lakes, particularly with respect to relationships among trophic status indicators and to compare these relationships to those found in temperate lakes;

3) to assess the applicability of existing I/O models to Florida lakes, and

4) to develop critical nutrient loading guidelines for Florida lakes.

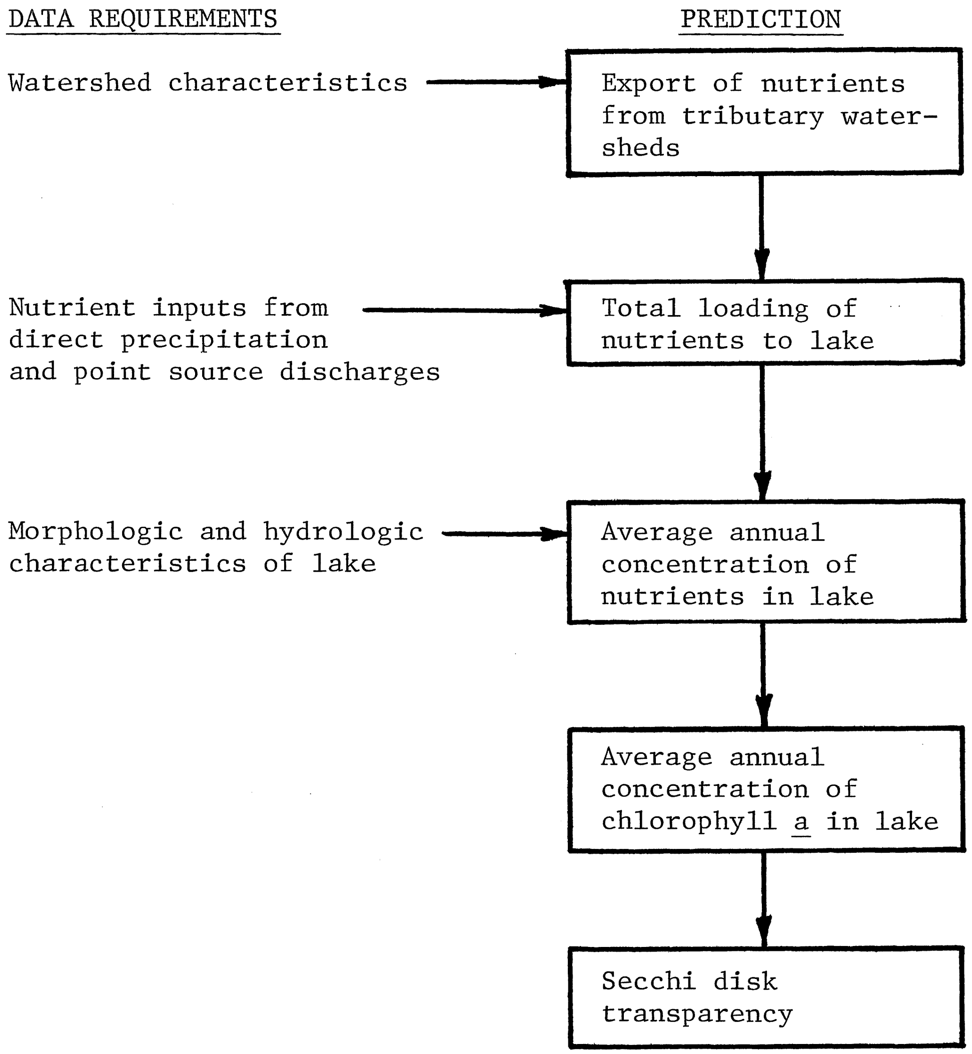


Figure I-1. Conceptual framework for predictions of lake trophic status.

CHAPTER II. DATA SOURCES AND METHODS

DATA SOURCES

In order to accomplish the objectives of this study, limnological data from three synoptic studies were compiled into a uniform format for computer analyses. The first of these studies (Brezonik and Shannon 1971) involved an evaluation of trophic conditions of 55 lakes in north and central Florida. Three groups of lakes were included in this study: 1) 16 primarily oligotrophic lakes in the Trail Ridge Region of Putnam and Clay counties, 2) 33 lakes and ponds throughout Alachua County and 3) six lakes of the Oklawaha chain. Samples were collected four times over a one-year period from 36 of the lakes, while the remaining 19 were sampled at two-month intervals to obtain more detailed data on seasonal trends.

The second study from which lake data were obtained was the Florida National Eutrophication Survey (NES), conducted by the EPA during the mid-1970's. Lakes included in this survey were selected on the basis of three criteria: 1) lakes impacted by one or more sewage treatment plant outfalls within 40 km (25 miles) of the lake waters; 2) lakes having a surface area greater than 40 ha (100 acres); and 3) lakes with residence times of greater than 30 days. In addition to the lakes that met these criteria, five lakes of special interest (South Lake, Lake Yale, Glenada Lake and Horseshoe Lake) were included in the Florida NES, for a total of 40 lakes. Most of the lakes are located in central Florida, particularly in Polk, Orange, Lake, Osceola, and Seminole Counties. Each lake was sampled from two to four times (usually three times) during 1973.

The third set of data is from a recent study of the effects of acid rain on softwater lakes in Florida (Brezonik et al. 1981). Twelve of the lakes included in this study are located in the Trail Ridge Region of north Florida and eight are located on the Isotokpoka Ridge in Highlands County (south-central Florida). Samples were collected every three months during 1979-80.

Since several lakes were included in more than one survey, only the more complete data set was used in this report. The resulting data base contains 101 lakes (Table II-1) whose locations are shown in Figure II-1. The variables measured in each survey are listed in Table II-2. In compiling the limnological data set, we used mean values for the entire water column. This approach is considered valid for Florida lakes, most of which are generally shallow and do not exhibit seasonal thermal stratification. Although the sampling methods differed slightly among the three studies from which data were obtained, these differences are not likely to be important with respect to the applications of the data in this report. The limnological data (annual means) compiled for the study lakes are presented in Appendix A-1; morphological data are presented in Appendix A-2.

Table II-1. Florida study lakes.

Lake Code	Lake Name	County
1	Minneola	Lake
2	East Lake Tohopekaliga	Osceola
3	Minnehaha	Orange
4	Weohyakapka	Polk
5	Tarpon	Pinellas
6	Istokpoga	Highlands
7	Yale	Lake
8	Kissimmee	Osceola
9	Jessie	Polk
10	Horseshoe	Seminole
11	Haines	Polk
12	South	Brevard
13	Okeechobee	Glades, Hendry Okeechobee
14	Marion	Polk
15	Crescent	Flagler, Putnam
16	Poinsett	Brevard, Orange
17	Doctors	Clay
18	Reedy	Polk
19	Gibson	Polk
20	Dora	Lake
21	Talquin	Gadsden, Leon
22	Apopka	Lake, Orange
23	Griffin	Lake
24	Glenada	Highlands
25	Thonotosassa	Hillsborough
26	Seminole	Pinellas
27	George	Putnam, Volusia
28	Tohopekaliga	Osceola
29	Monroe	Seminole, Volusia
30	Hancock	Polk
31	Eloise	Polk
32	Howell	Orange, Seminole
33	Banana	Polk
34	Jessup	Seminole
35	Alligator	Columbia
36	Trout	Lake
37	Lawne	Orange
38	Munson	Leon
39	Effie	Polk
40	Lulu	Polk
41	Santa Fe	Alachua
42	Little Santa Fe	Alachua
43	Hickory Pond	Alachua
44	Altho (Alto)	Alchua
45	Cooter Pond	Alachua
46	Elizabeth	Alachua
47	Clearwater	Alachua
48	Hawthorne	Alachua
49	Little Orange	Alachua
50	Unnamed 10	Alachua
51	Moss	Alachua
52	Jeggord	Alachua
53	Still Pond	Alachua
54	Lochloosa	Alachua
55	Orange	Alachua
56	Palatka Pond	Alachua
57	Newnan's	Alachua
58	Mize	Alachua
59	Calf Pond	Alachua
60	Unnamed 20	Alachua
61	Meta	Alachua
62	Alice	Alachua
63	Bivens Arm	Alachua
64	Clear	Alachua
65	Unnamed 25	Alachua
66	Beville's Pond	Alachua
67	Unnamed 27	Alachua
68	Kanapaha	Alachua
69	Watermelon Pond	Alachua
70	Long Pond	Alachua
71	Burnt Pond	Alachua
72	Wauberg	Alachua
73	Tuscawilla	Alachua
74	Harris	Lake
75	Eustis	Lake
76	Weir	Marion
77	Kingsley	Putnam
78	Sand Hill (Lowry)	Clay
79	Magnolia	Clay
80	Brooklyn	Clay
81	Geneva	Clay
82	Swan	Putnam
83	Wall	Putnam
84	Santa Rosa	Putnam
85	Adaho	Putnam
86	McCloud	Putnam
87	Anderson Cue	Putnam
88	Suggs	Putnam
89	Long	Putnam
90	Winnot	Putnam
91	Cowpen	Putnam
92	Gallilee	Putnam
93	Annie	Highlands
94	Clay	Highlands
95	Francis	Highlands
96	Johnson	Clay
97	Josephine	Highlands
98	June	Highlands
99	Letta	Highlands
100	Placid	Highlands
101	Sheeler	Clay

(1) Lake codes: 1-40 = NES lakes, 41-92 = 55 lakes study (Brezonik & Shannon 1971), 93-101 = Brezonik et al. (1981).

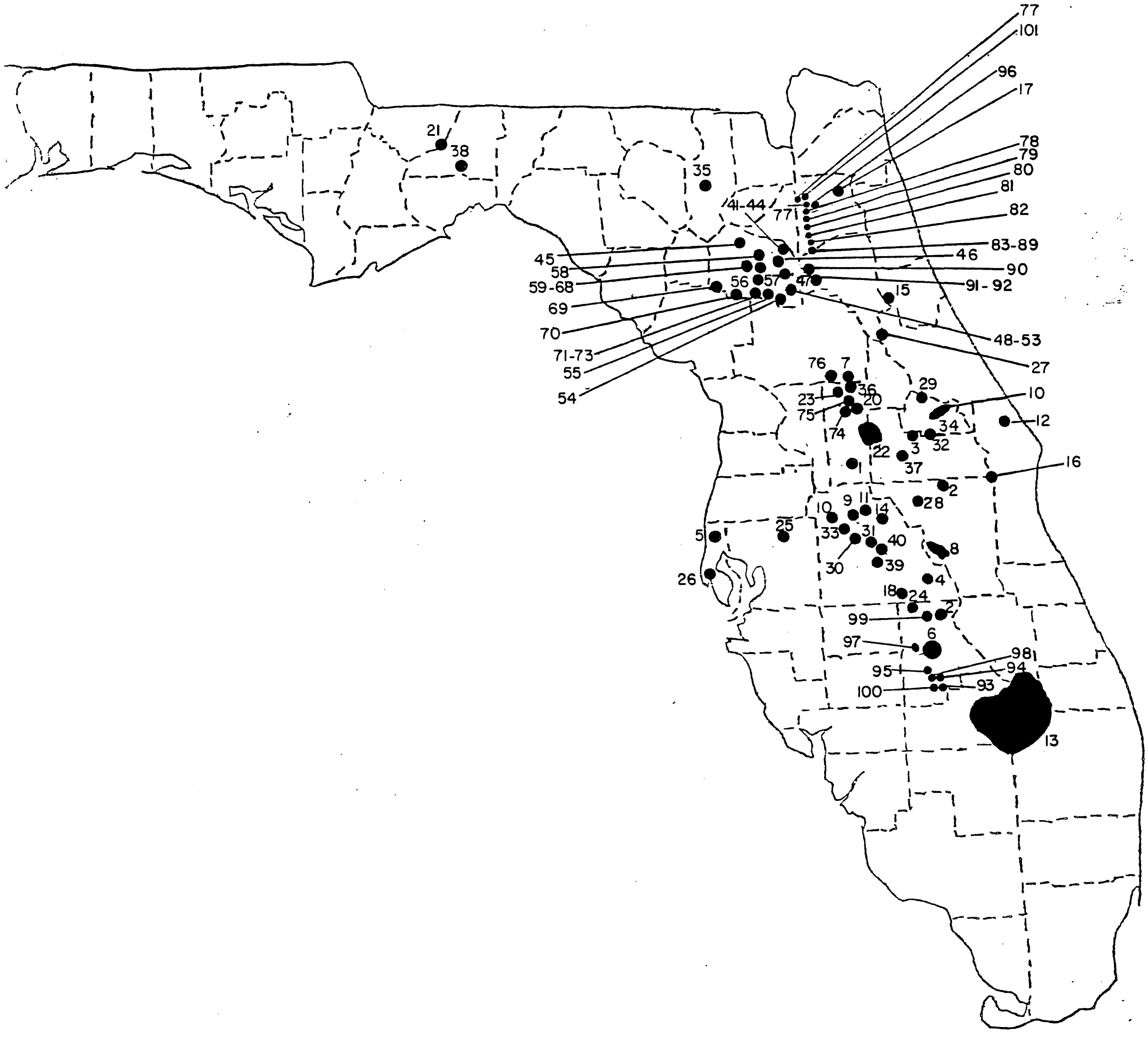


Fig. II-1. Location of 101 study lakes.

Table II-2. Data collected in synoptic studies of Florida lakes.

	Brezonik & Shannon, 1971	Nation Eutrophication Survey	Brezonik, et al., 1980 (1)
Watershed			
Drainage area	X	X	X
Land use characteristics	X	X	
Hydrologic & morphologic			
Water budget		X	
Volume	X	X	X
Surface area	X	X	X
Max. depth	X	X	X
Mean depth	X	X	X
Nutrient budget	X$^{(1)}$	X	
Chemical & physical			
Total nitrogen	X	X	X
NH$_3^+$	X	X	X
NO$_4^-$	X	X	X
NO$_3^-$	X	X	X
Organic N	X	X	X
Total phosphorus	X	X	X
Orthophosphate	X	X	X
COD	X		
pH	X	X	X
Alkalinity	X	X	X
Acidity	X		
Color	X		X
Dissolved oxygen	X	X	X
Specific conductance			
Major ions (Ca, Mg, K, Na, Fe, SO$_4$, Cl, Si)	X		X
Total organic carbon			X
Total inorganic carbon			X
Trace metals	X$^{(2)}$		X$^{(3)}$
Secchi disk transparency	X	X	X
Temperature	X	X	X
Suspended solids	X		
Total solids	X		
Turbidity	X		X
Biological			
Chlorophyll a	X	X	X
Carotenoids	X		
Algal identification & counts	X	X	X
Primary productivity	X		
Zooplankton identification & counts			X
Limiting nutrient bioassays		X	
Sediments			
Sediment type (visual classfication)	X		
Benthic organisms	X		X
Chlorophyll derivatives	X		
Total carotenoids	X		
Volatile solids	X		
Organic nitrogen	X		
NH$_4^+$	X		
Total phosphate	X		
Iron	X		
Manganese	X		

(1) Input of nutrients computed from land use and population characteristics.

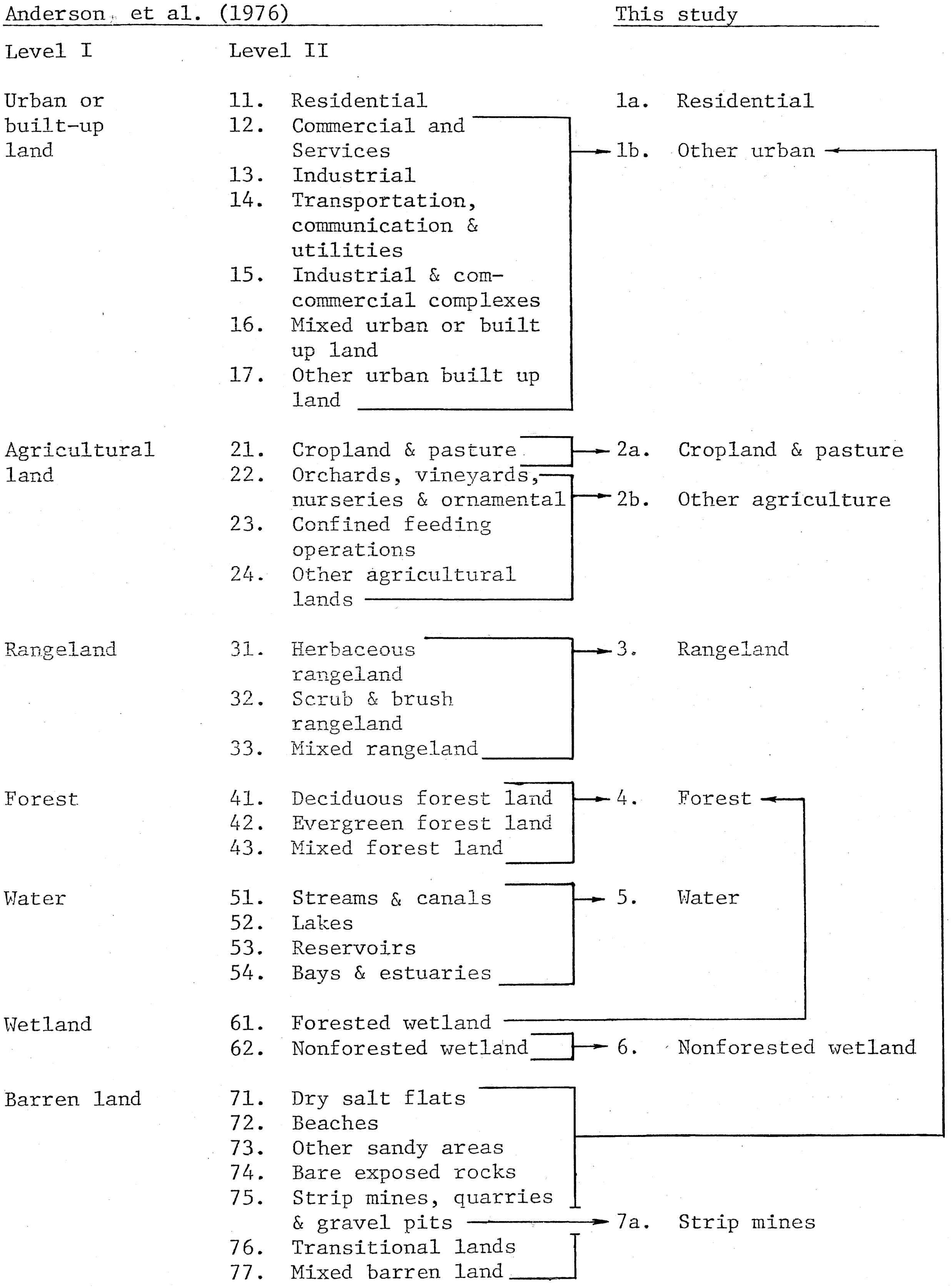
(2) Includes Mn, Cu, Zn, Fe and Sr

(3) Al only.

In addition to the limnological data, the EPA also computed nutrient budgets for 34 of the 40 Florida NES lakes. Methods used to compute nutrient budgets are described in NES Working Paper No. 175 (NES 1975). In this study, non-point source (NPS) loadings of nitrogen and phosphorus for each tributary were computed by subtracting the reported point source loadings from the total tributary loadings. Runoff for each tributary was computed by dividing the total streamflow by the watershed area. Data on the nutrient and hydrologic budgets for the Florida NES lakes are presented in Appendix A-3.

Determination of land uses in the NES watersheds was made using Mark Hurd photoquads and USGS 7.5 minute quadrangles (both 1:24,000). Photos from the Agricultural Stabilization and Conservation Service and the State of Florida CITRUS survey were used to provide additional resolution. The land use classification scheme used was a modification of the system developed by Anderson et al. (1976). The modifications involve classification of agricultural land, forests and barren land (Fig. II-2). Agricultural land was divided into two categories: "cropland and pasture" and "other agriculture". The category "other agriculture" was formed to assess the effects of different types of agricultural land use on NPS nutrient loading. This category is comprised largely of citrus orchards in the NES watersheds. The second modification was a combining of "deciduous forest land", "evergreen forest land" and "mixed forest land" (categories 41 to 43 in Anderson's Level II scheme) with "forested wetland" (category 61) into a composite "forest" category. This was done to facilitate identification of forests in the air photos. Finally, "salt flats", "barren land", "beaches", "other sandy areas", "transitional areas" and "mixed barren land" (categories 71-74 and 76-77) in Anderson's Level II scheme were grouped together with non-residential urban land uses into an "other urban" land use category. This modification was made because these barren land subgroups were largely comprised of "transitional areas" associated with urban areas.

The watersheds used in the evaluation of land use-nutrient loading relationships are listed in Table II-3. Several NES watersheds were not included in the analyses because runoff from them was found to be abnormally high (> 70% of total precipitation in the watershed). Such high values could result from groundwater inflows from other watersheds or backpumping in agricultural areas; or alternatively, they may simply reflect errors in flow measurement. Since it was impossible to establish the cause of high runoff for individual watersheds in this study, all watersheds in which runoff was $\geq$ 70% of precipitation were excluded from the analyses. This resulted in the deletion of watersheds 13B1, 18B1, 22C1, 37D1, 32B1 and 34G1. The 70% criterion was considered a reasonable upper limit for natural runoff from Florida watersheds (W. C. Huber, per. comm., 1980). Several other watersheds were excluded because flow data were not available (31A1, 40B1) or because the measured drainage basin area differed substantially from that reported by the NES (17B1). Thus, data from 41 watersheds were used in the analyses of land use-nutrient loading relationships. Land use data and data for non-point source nitrogen and phosphorus loadings for these watersheds are presented in Appendix A-4.



Anderson et al. (1976) Level I	Level II	This study
Urban or built-up land	11. Residential	1a. Residential
	12. Commercial and Services	1b. Other urban
	13. Industrial	
	14. Transportation, communication & utilities	
	15. Industrial & commercial complexes	
	16. Mixed urban or built up land	
	17. Other urban built up land	
Agricultural land	21. Cropland & pasture	2a. Cropland & pasture
	22. Orchards, vineyards, nurseries & ornamental	2b. Other agriculture
	23. Confined feeding operations	
	24. Other agricultural lands	
Rangeland	31. Herbaceous rangeland	3. Rangeland
	32. Scrub & brush rangeland	
	33. Mixed rangeland	
Forest	41. Deciduous forest land	4. Forest
	42. Evergreen forest land	
	43. Mixed forest land	
Water	51. Streams & canals	5. Water
	52. Lakes	
	53. Reservoirs	
	54. Bays & estuaries	
Wetland	61. Forested wetland	
	62. Nonforested wetland	6. Nonforested wetland
Barren land	71. Dry salt flats	
	72. Beaches	
	73. Other sandy areas	
	74. Bare exposed rocks	
	75. Strip mines, quarries & gravel pits	7a. Strip mines
	76. Transitional lands	
	77. Mixed barren land	

Figure II-2. Land-use classification system.

Table II-3. NES watersheds used in analysis of land use-nutrient loading relationship.

Tributary Code	Tributary	Lake
01 B1	Unnamed	Minneola
02 A1	Boggy Cr.	East L. Tohopekaliga
02 B1	Unnamed	
03 B1	Unnamed	Minnehaha
04 A1	Tiger Cr.	Weohyakapka
05 A1	South Cr.	Tarpon
06 B1	Arbuckle Cr.	Istokpoga
08 D1	Jackson Canal	Kissimmee
09 A1	Unnamed	Jessie
13 B1*	Unnamed	Okeechobee
13 C1	Taylor Creek	Okeechobee
13 D1	Lembin Creek	Okeechobee
13 F1	Indian Prarie Canal	Okeechobee
13 G1	Harney Pond Canal	Okeechobee
14 D1	Unnamed Cr.	Marion
15 B1	Haw Creek	Crescent
15 C1	Unnamed	Crescent
17 B1*	Swimming Cr.	Doctors
18 B1*	Unnamed	Reedy
19 B1	Unnamed	Gibson
20 A1	Dora Canal	Dora
20 B1	Unnamed	Dora
21 B1	Ockawaha Cr.	Talquin
21 D1	Bear Cr.	Talquin
22 C1*	Unnamed	Apopka
23 A1	Dead River	Griffin
25 A1	Baker Creek	Thonotosassa
26 B1	Unnamed	Seminole
26 C1	Bayou Cr.	Seminole
28 A1	Shingle Cr.	Tohopekaliga
28 B1	Partin Canal	Tohopekaliga
29 B1	Bethyl Cr.	Monroe
30 A2	Saddle Cr.	Hancock
30 B1	Unnamed	Hancock
30 C1	Unnamed	Hancock
31 A1*	Unnamed	Eloise
32 A1	Howell Cr.	Howell
32 B1*	Unnamed	Howell
34 A1	Gee Cr.	Jessup
34 B1	Soldier Cr.	Jessup
34 C1	Unnamed	Jessup
34 D1	Howell Cr.	Jessup
34 E1	Salt Cr.	Jessup
34 G1*	Sweetwater Cr.	Jessup
35 A1	Unnamed Cr.	Alligator
35 B1	Unnamed Cr.	Alligator
35 C1	Unnamed Cr.	Alligator
37 B1	Unnamed Canal	Lawne
37 C1	Unnamed Canal	Lawne
37 D1*	Unnamed Canal	Lawne
38 B1	Unnamed	Munson
40 B1*	Unnamed	Lulu

*Watershed dropped from analyses. See text.

STATISTICAL METHODS

Statistical analyses were performed using the Statistical Analysis System (SAS) (SAS Institute, 1979). Where multiple independent variables were used in regression analyses, the STEPWISE (backward) procedure was used to select the significant independent variables. The GLM procedure was used to compute predicted values and confidence limits for final regression analyses. Confidence limits for the mean (CLM) were used when the objective of the regression analysis was to predict the mean response of the dependent variable. Confidence limits for individual predictions (CLI) were used when the objective of the regression model was to predict values of the dependent variables from individual values of the independent variable (Snedecor and Cochran 1967).

In evaluating the nutrient loading models, several lakes were discarded as the result of suspected errors in the data or on the basis of a statistical outlier test. The test procedure used was the criterion T_n, defined as:

$$T_n = |Y_{obs} - Y_{pred}|/s$$

where Y_{obs} = observed y value,

Y_{pred} = predicted y value, and

s = population standard deviation excluding the outlier.

If the resulting T_n value of a suspected outlier was greater than the 5% T_n value from a table of critical T_n values (Grubbs 1969), then the value was considered an outlier. Although the method is recommended for removing only a single outlier, several outliers can be removed by reevaluating s following the removal of each outlier. In this manner, up to three outliers were removed from any one prediction equation. The removal of a value as an outlier in one equation did not automatically result in its removal from other equations.

CHAPTER III. PREDICTIONS OF NUTRIENT LOADING FOR FLORIDA LAKES

INTRODUCTION

In studies of lake eutrophication it is often necessary to know the loadings of plant nutrients, particularly nitrogen and phosphorus, into a lake. Ideally, these loadings are determined by obtaining data on water fluxes and nutrient concentrations for all sources (tributary inflows, direct point source discharges and direct precipitation). Unfortunately, the acquisition of these data is expensive and time-consuming, usually requiring at least one year of study. It is therefore desirable to be able to estimate the loadings of nutrients from various sources indirectly.

The most difficult component of a lake's total nutrient loading to determine is the non-point source loadings from the surrounding watershed. In this chapter the major emphasis will be to develop a method of estimating non-point source nutrient loadings from tributary watersheds using land use data. The approach used, originally conceived by Lee et al. (1966), is based on the assumption that nutrient export from a portion of a watershed can be computed as the product of the drainage area and an export coefficient that is determined by land use. Thus for a watershed composed of multiple land uses:

$$L_i = \sum_{j=1}^{n} \beta_{ij} A_j \qquad (3\text{-}1)$$

where L_i = loading of constituent i, kg/yr,

β_{ij} = export coefficient for constituent i from land use j, kg/ha-yr, and

A_j = area of watershed in land use j, ha.

Since this concept was first developed, dozens of studies have been conducted to determine export coefficients for various land uses. These have been compiled by Uttormark et al. (1974), Loehr (1974) and, most recently, by Reckhow et al. (1980). Unfortunately, the range of reported export coefficients for a given land use varies widely. This variability is expected since the approach does not take into account variations in precipitation, soil type, length of the growing season and other parameters that influence nutrient export. Furthermore, most of the studies included in these compilations have been conducted in temperate climates, limiting the validity of these coefficients for Florida watersheds.

In order to determine suitable land-use export coefficients for Florida, two methods were used. First, data from studies of non-point source nutrient loading conducted in Florida were compiled by land use. By examining only studies that have been conducted in Florida, variability in export coefficients caused by differences in soil type, climate and terrain should be reduced, resulting in a narrower range of export coefficients for each land use. For studies in which export coefficients were not computed, the reported loading data were used to perform the appropriate calculations. In a few cases, data are included for several studies conducted in nearby states, where land uses and other conditions (soils; climate) were judged to be representative of conditions in Florida.

The second method used to compute export coefficients was a statistical approach similar to that used by Omernik (1976). In this approach, export coefficients are determined by multiple regression techniques using data on nutrient loading (dependent variable) and land uses (independent variables). For this analysis, data on nutrient loading, nutrient concentration, runoff and land use were compiled for 41 Florida NES watersheds (Chapter 2).

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Point Source Loadings. The results of a nationwide survey of nutrient loadings from 809 municipal wastewater treatment plants (Gakstatter et al. 1978) may be used to estimate treatment plants loadings of these nutrients for preliminary studies on the basis of population served and treatment type (primary, trickling filter, activated sludge, stabilization pond). As seen in Table III-1, the type of treatment has little effect on the phosphorus loading from wastewater treatment plants: median loadings ranged from 0.9 kg/cap-yr for stabilization ponds to 1.1 kg/cap-yr for primary treatment facilities. These results compare favorably with those of Vollenweider (1968), who computed a mean phosphorus loading of 0.8 kg/cap-yr for municipal wastewater from the results of 15 studies reported in the literature (treated and untreated wastewaters were included). Although the type of treatment had little effect on phosphorus loadings, phosphorus loadings in the compilation of Gakstatter et al. were significantly lower for a group of 33 plants that included tertiary phosphorus removal processes (median loading = 0.4 kg/cap-yr) and for a group of 25 convential treatment plants located in communities having phosphorus detergent bans.

The results of Gakstatter et al. indicate that nitrogen loading from wastewater treatment plants is affected by the type of treatment process used. Loadings ranged from 2.0 kg/capita-yr for stabilization pond effluents to 4.2 kg/cap-yr for primary treatment plant effluents. In comparison, Vollenweider (1968) reported a nitrogen loading rate of 3.9 kg/cap-yr. For all four treatment processes, the median effluent TN:TP ratios were less than 5:1, indicating that sewage effluents typically are nitrogen limited.

Although the standard errors for loading estimates in Table III-1 indicate that loadings from municipal wastewater treatment plants can usually be estimated with reasonable accuracy, actual loadings for a given plant may differ significantly from predicted values because of (1) modifications in the design process, (2) hydraulic overloading resulting from stormwater inflows, (3) excessive infiltration or overuse, or (4) impairment of the treatment process by toxic wastes or improper operation. Thus, while the values presented in Table III-1 may be used for preliminary estimates of nutrient

Table III-1. Median and mean phosphorus and nitrogen concentrations and median loads in wastewater effluents following four conventional treatment practices(1)

		Treatment Type			
		Primary	Trickling Filter	Activated Sludge	Stabilization Pond
Number of Sampled Plants		55	244	244	119
Total Population Served		1,086,784	3,459,983	4,357,138	270,287
Ortho-P Conc. (mg/l)	Median	3.5 ± 0.29*	5.1 ± 0.21	4.6 ± 0.24	3.9 ± 0.34
	Mean	4.0 ± 0.62	5.4 ± 0.38	5.3 ± 0.40	4.8 ± 0.62
Total-P Conc. (mg/l)	Median	6.6 ± 0.66	6.9 ± 0.28	5.8 ± 0.29	5.2 ± 0.45
	Mean	7.7 ± 1.19	7.2 ± 0.50	6.8 ± 0.51	6.6 ± 0.81
Total-P Load (kg/cap-y)	Median	1.1 ± 0.10	1.2 ± 0.05	1.0 ± 0.06	0.9 ± 0.10
Inorganic-N Conc. (mg/l)	Median	6.4 ± 1.00	7.1 ± 0.38	6.5 ± 0.45	1.3 ± 0.29
	Mean	8.3 ± 1.40	8.2 ± 0.60	8.4 ± 0.69	5.5 ± 1.95
Total-N Conc. (mg/l)	Median	22.4 ± 1.30	16.4 ± 0.54	13.6 ± 0.62	11.5 ± 0.84
	Mean	23.8 ± 3.48	17.9 ± 1.23	15.8 ± 1.16	17.1 ± 3.59
Total-N Load	Median	4.2 ± 0.40	2.9 ± 0.17	2.2 ± 0.15	2.0 ± 0.26
TN:TP Ratio	Median	3.4	2.4	2.4	2.2
Per Capita Flow (l/cap·d)	Median	473 ± 72	439 ± 19	394 ± 26	378 ± 38

* Value ± 1 standard error.

(1) From Gakstatter et al. (1978).

loadings from wastewater treatment plants, they should not be regarded as substitutes for actual measurements when expensive management decisions are made.

Precipitation Inputs. For many lakes, particularly seepage lakes with long detention times, bulk precipitation (wetfall + dryfall) may be a major source of nutrients. A recent study on the chemical composition of bulk precipitation in Florida (Brezonik et al. 1981) includes data on nitrogen and phosphorus loadings for 24 sites throughout Florida, providing a data base that can be used to estimate precipitation loadings to lake surfaces (Table III-2).

The mean deposition rate for nitrogen was 0.76 g N/m^2-yr, but the deposition rates at individual stations varied considerably. The lowest nitrogen deposition rate, 0.32 g N/m^2-yr, occurred at Bahia Honda Key, while the highest rate, 1.13 g N/m^2-yr, occurred at Belle Glade in the intensively cultivated Everglades Agricultural Area. When sampling stations were grouped according to location and local land use, deposition rates were found to be lowest at the coastal and non-agricultural sites and highest at the agricultural sites (Table III-2). On a statewide basis, 69% of the total nitrogen deposited was in the form of inorganic species, and speciation generally followed the sequence $NH_4^+ > NO_3^-$ > organic N.

Phosphorus deposition was also site dependent and ranged from 17 mg P/m^2-yr at Bahia Honda to 111 mg P/m^2-yr at Jasper, with a mean of 51 mg P/m^2-yr. As with nitrogen deposition, phosphorus deposition was generally lowest at the coastal and non-agricultural rural sites and highest at agricultural sites. Soluble reactive phosphorus was the dominant species, accounting for 68% of the total phosphorus in bulk precipitation.

The significance of nutrient inputs from precipitation with respect to lake eutrophication can be evaluated by comparing the magnitude of these loadings with critical loading values. For example, using Vollenweider's (1968) original critical loading criteria, a lake with a mean depth of 3 m has a critical loading of 97 mg P/m^2-yr and 1.45 g N/m^2-yr. According to these criteria, the mean precipitation loading for Florida corresponds to 53% of the critical loading for phosphorus and 52% of the critical loading for nitrogen. Thus, precipitation inputs of nutrients may be a significant component of the total nutrient budget for a lake.

Non-point source loadings from Florida watersheds. Export coefficients for nitrogen and phosphorus determined for Florida watersheds are presented by land-use category (urban residential, agricultural, forest) in Tables III-3 to III-6. For comparison, the results of a literature review of export coefficients for studies conducted throughout the U.S. and Canada (Reckhow et al. 1980) are presented.

Before discussing the magnitude of these export coefficients, several comments are in order concerning the nature of these studies. First, the methods used to determine nutrient loading are highly variable among investigators, and some are based on rather limited data. For example, Lamonds (1974) based his estimates of N and P export from a residential area in Eustis, Florida, on data collected during only seven storms, while Wanielista

Table III-2. Atmospheric deposition (via bulk precipitation) of total nitrogen and total phosphorus at stations grouped according to dominant land use in the area.(1)

	TN kg/ha-yr	TP kg/ha-yr
Coastal	5.8	0.31
Urban	7.6	0.50
Rural (non-agricultural)	6.2	0.27
Rural (agricultural)	8.8	0.66
State average	7.5	0.51

(1) From Brezonik et al. (1981)

Table III-3. Nutrient export from urban areas.

Source	Location	Loading (kg/ha-yr) Total P	Ortho P	Total N	Org. N	NO_3^--N	NH_4^+-N	Comments
Burton and Turner (1977)	Nr. L. Jackson Fla.	7.49	0.19 (25%)	0.37 (inorg.)		0.18 (0.02 NO_2^-)	0.17	80% residential & commercial 423 samples analyzed
Wanielista (1977)	Orlando, Fla.	3.5	2.0 57%	10 (TKN + NO_3^-)		6		Commercial area. Only two storms sampled - results extrapolated.
Miller et al. (1979)	Ft. Lauderdale, Fla.	0.26	0.11 42%	2.88	2.12 (74%)	0.55 (.042 NO_2^-) (19.1%)	0.10 (3.5%)	97.9% impervious area 31 storms sampled. Loadings calc. from raw data.

Table III-4. Nutrient export from residential areas.

		Loading (kg/ha-yr)						
Source	Location	Total P	Ortho-P	Total N	Organic N	NO_3^--N	NH_4^+-N	Comments
Burton and Turner (1977)	Vicinity of L. Jackson, Fla.	4.74	0.09 (1.9%)			0.58 (.03 NO_2)	0.16	Area is lightly developed but includes 10% under highway eenst. Stream receives package plant eff. from school.
Wanielista et al. (1977)	Near Orlando, Fla.	2.24	0.80 (35.7%)	3.98 (TKN + NO_3)		2.17		Loading est. extrapolated from data on two storms.
Bedient et al. (1978)	Houston, Texas	0.745				0.29		Residential development included new construction. Clayey soils.
Mattraw & Sherwood (1977)	Broward Co., Fla.	0.21		1.48				Uniform single family dwellings. Only 5-10% of rainwater collected as runoff.
Lamonds (1974)	Eustis, Fla.	0.82		7.36				Flow estimated only 7 storms sampled.

Table III-5. Nutrient export from agricultural areas.

Source	Location	Loading (kg/ha-yr) Total P	Ortho-P	Total N	Org. N	NO_3^--N	NH_4^+-N	Comments
Campbell (1978)	Alachua Co.							
	1975-76	1.34	1.21	6.36	5.30	0.37	0.68	Land in intensive crop production w/some pasture near stream. 3-8% slope, sandy soil w/claypan at 1-2 m.
	1976-77	0.86	0.63	2.10	1.92	0.09	0.09	
Burton & Turner (1977)	Near L. Jackson, Fla.	0.51	0.14	0.21 (inorg.)		0.07 (.004 NO_2)	0.14	48% agricultural (52% forested). 371 samples analyzed.
Stewart et al. (1978)	Upper Taylor Creek							1% cropland 59" improved pasture" 30% range & forest 10% misc.
	W-3 1972		1.48			0.58		
	1974		2.80			0.44		
	1975		0.65			0.16		
	W-5 1972		0.38			0.49		70% improved pasture 20% range & forest 10% misc.
	1974		1.61			0.42		
	1975		0.53			0.20		
	W-13 1972		6.11			1.36		78% improved pasture 21% dairy operations
	1974		12.73			3.16		
	1975		1.94			0.30		
Ritter et al. (1979)	Delaware Coastal Plain Stockley Branch	0.68	0.083	20.6	7.57	12.41	0.63	1359 ha. watershed 45% crop (soybean, corn, grain) 47% Forest 4% urban. Sandy loam soils, low slope.
	Blackwater Creek	0.48	0.078	18.2	6.96	10.78	10.07	1456 ha. 57% cropland (as above). 37% forest, 2% urban sandy loam soils.

Table III-5. Nutrient export from agricultural areas (cont'd)

Source	Location	Loading (kg/ha-yr)						Comments
		Total P	Ortho-P	Total N	Org. N	NO_3^--N	NH_4^+-N	
Asmussen, et al. (1979)	Little River, Georgia							
	1975		0.141			0.24		Cropland = 36.8%. Forest, swamp, etc.
	1976		0.145			0.14		$\sim$60%
CH2MHill (1979)	Everglades Ag. Area Sugarcane	0.65		27.18				All areas backpumped & irrigated. Organic soils mean of 3 sites.
	Vegetable farm	2.37		38.86				Mean of 3 sites.
	Cattle ranch	0.55		12.36				Mean of 2 sites.
Lutz (1977)	Everglades Ag. Area. S-5A	0.77		50.1				Note: drainage areas poorly defined in EAA 88% agricultural (71.2% truck crops, 8.9% pasture)
	S-6	0.58		27.0				93% agricultural (36% sugarcane, 15% truck farming, 45% pasture.
	S-7	0.41		30.3				78% agricultural (69% sugarcane, 9% pasture). 22% forest & wetland.
	S-8	0.75		32.3				30.5% agricultural (26% sugarcane). 69.3% forest and wetlands.

Table III-6. Nutrient export from forested areas.

Source	Location	Loading (kg/ha-yr)						Comments
		Total P	Ortho -P	Total N	Org. N	NO_3^--N	NH_4^+-N	
Campbell (1978)	Alachua Co.							Native forest w/small amount of cropland. Sandy soils w/claypan, 0-3% slope.
	1975-76	0.33	0.30	1.43	1.21	0.12	0.11	
	1976-77	0.68	0.52	1.65	1.49	0.09	0.07	
Bedient et al. (1978)	Houston, Texas	0.21				0.29		Heavily forested, clayey soils, 0.1% slope. 25 storms sampled.
Reikerk, et al. (1978)	Bradford Co., Fla.	0.4	0.2	6.1	5.5	0.1 (inc. NO_2)	0.5	Data collected from 3 coastal plain flatwoods areas for 1 water year. Mean values shown here.
Duffy et al. (1978)	Northern Mississippi	0.30	0.029					Data collected for <u>storm events</u> only (no base flow between storms). Five watersheds sampled. Pine (mix) forest; loess soils.

et al. (1977) computed export coefficients for several watersheds in the Orlando area using data collected during only two storms. In the most careful studies (including Riekerk et al. 1978; Campbell 1979; Ritter et al. 1979; Burton et al. 1977) data on baseflow concentrations were collected on a regular basis (usually weekly), and stormflow events were sampled at frequent intervals using an automated sampler. Concentration data were then combined with continuous flow measurement data to produce relatively accurate estimates of loading. In applying export coefficients reported in the literature, the reader should consider the quality of the methods used to compute nutrient loadings.

A second weakness of many of these studies is that water budgets were not constructed. Because of this, it cannot be certain that the streamflow at the sampling station represents the sole outlet of water (and nutrients) from the watershed, although this is generally assumed. Conversely, unless a water budget is completed, it cannot be assumed that all of the water coming from a watershed results from precipitation falling on that watershed rather than from seeps and springs whose flow may originate from outside the boundaries of the watershed in question. In order to develop reliable nutrient export coefficients as a function of land-use, it is important that all inflows and outflows to the watershed are determined; this is particularly true in Florida where the groundwater table is fairly shallow and springs are common.

Finally, only a few of the studies reported here involved more than one annual cycle, and therefore they did not evaluate annual variations in loading. The data of Campbell (1978), Stewart et al. (1978) and Asmussen et al. (1979) illustrate the extent of variation that may occur from year to year (Table III-5). Campbell observed decreases of 36% and 67% for total P and total N loads, respectively, between the 1975-76 sampling period and the 1976-77 sampling period. Most of the difference in loading between the two annual periods occurred as the result of a decrease in streamflow rather than changes in the concentrations of constituents. For the Upper Taylor Creek watershed, the highest annual export rate of orthophosphate was 5.5 times the lowest export rate, and the highest export rate of NO_3^--N was 6.1 times the lowest export rate over a three year period (Stewart et al. 1978). Changes in management practices, as well as variations in precipitation, were considered to be responsible for the observed fluctuations in loading. Asmussen et al. (1979) reported relatively constant orthophosphate loads but a 2X variation in the nitrate load over a two year period for the Little River watershed in southern Georgia. As can be seen from these examples, the magnitude of the variations in loadings can be substantial, at least for agricultural watersheds. Studies cited by Reckhow et al. (1980) indicate that there are also substantial annual variations in the loadings from forested watersheds. The extent of variations in nutrient loadings observed in multiple-year studies suggests that for critical applications, calculations of nutrient loadings should be based on data acquired over the period of several years. Furthermore, it can be concluded that the variation in export coefficients reported for a given land use is attributed in part to temporal variability.

The range of export coefficients for N and P reported in literature for various land uses is shown in Figures III-1 and III-2, together with export coefficients determined from the statistical analysis (to be discussed in the following section). The range of phosphorus export coefficients reported for Florida watersheds falls within the range of coefficients found by Reckhow et al. (1980) for all three major land uses, although the range of phosphorus export coefficients for urban areas in Florida is near the lower end of the range reported by Reckhow and coworkers. The range of nitrogen export coefficients found for Florida watersheds also falls within the range of values reported by Reckhow et al. (1980) for all three major land uses. Limited data on speciation of nitrogen in runoff from agricultural and forested watersheds (Fig. III-3) indicates that while the majority of nitrogen in runoff from forested watersheds is in the organic form, both nitrate and organic nitrogen are important fractions in the runoff from agricultural wathersheds.

It is tempting to conclude that the narrower range of each land-use export coefficient for Florida watersheds compared to the range reported by Reckhow et al. (1980) reflects the greater similarities in climate, soils, and topography among the Florida watersheds. However, it is also possible that the narrower ranges of export coefficients for Florida watersheds simply reflects the fact that fewer studies have been conducted for Florida watersheds. Unfortunately, a statistical analysis of the variation in export coefficients reported for Florida watersheds cannot be conducted because only a few values have been reported for each land use.

The values of export coefficients reported for each land use in Florida watersheds may vary by more than an order of magnitude for most land uses. Thus, phosphorus export coefficients range from 0.26 to 7.49 kg/ha-yr for urban areas, 0.21 to 4.74 kg/ha-yr for residential areas, 0.41 to 2.37 kg/ha-yr for agricultural areas, and 0.21 to 0.68 kg/ha-yr for forested watersheds. Nitrogen export coefficients exhibit similar variability: 0.37 to 2.88 kg/ha-yr for urban areas, 1.48 to 7.36 kg/ha-yr for residential area, 2.1 to 50.1 kg/ha-yr for agricultural areas and 1.43 to 6.1 kg/ha-yr for forested areas. These broad ranges of values limit the accuracy of loading estimates that can be made using a literature-base approach. A further disadvantage of this approach is that the accuracy of predictions cannot be evaluated using statistical analysis.

STATISTICAL ANALYSIS OF NUTRIENT EXPORT FROM THE NES WATERSHEDS.

An alternative approach in predicting NPS nutrient export from watershed land-use characteristics is to use multiple regression techniques whereby land-use characteristics (the independent variables) are used to predict nutrient export (the dependent variable). In this study, data on nutrient export, flow-weighted nutrient concentrations, flow, and land-use characteristics were compiled for 41 NES watersheds, as described in Chapter II. These data were used to develop statistically significant regression relationships between land use and NPS nutrient loading, land use and flow-weighted nutrient concentration, and land use and flow.

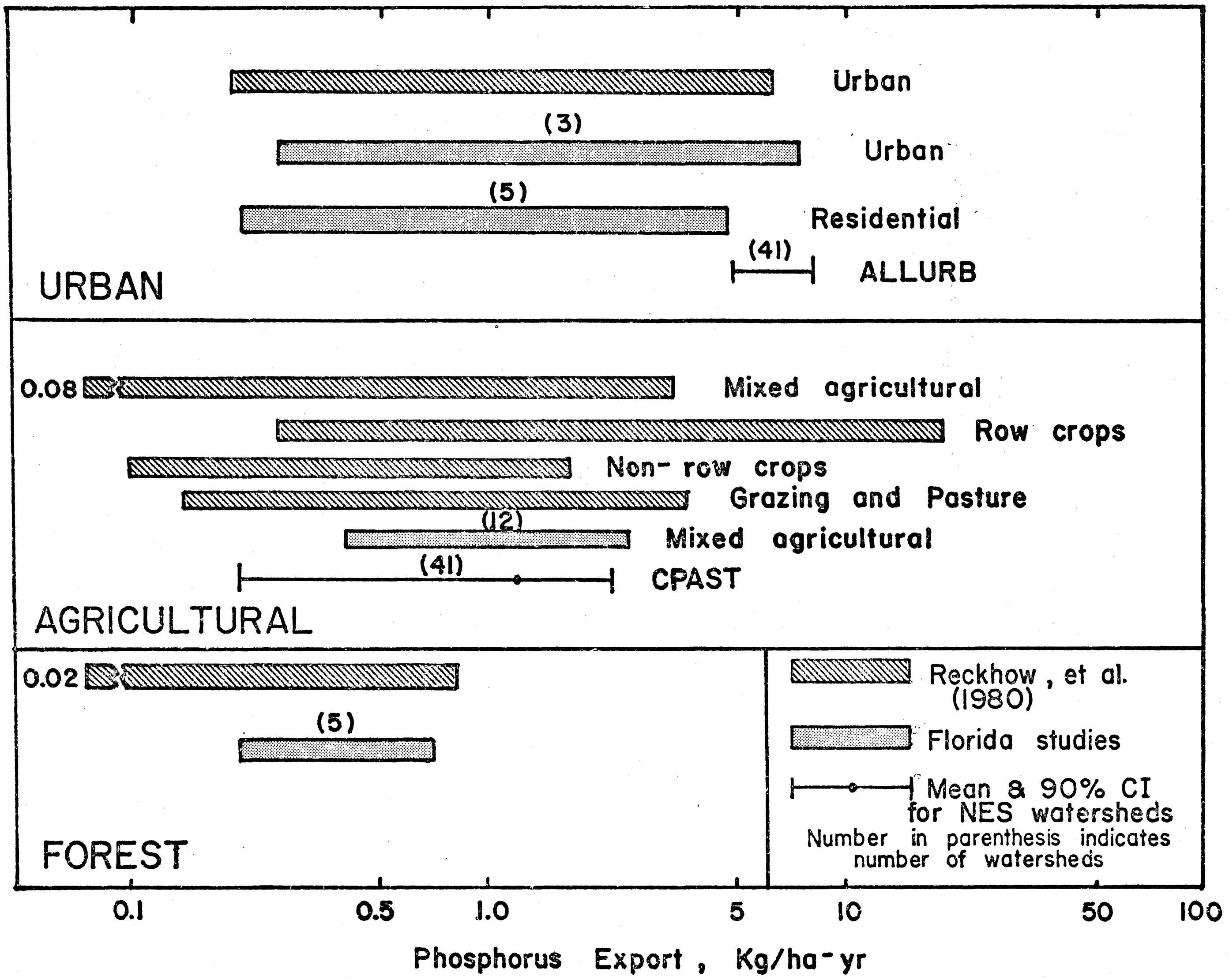

Figure III-1. Phosphorus export from urban, agricultural, and forested watersheds.

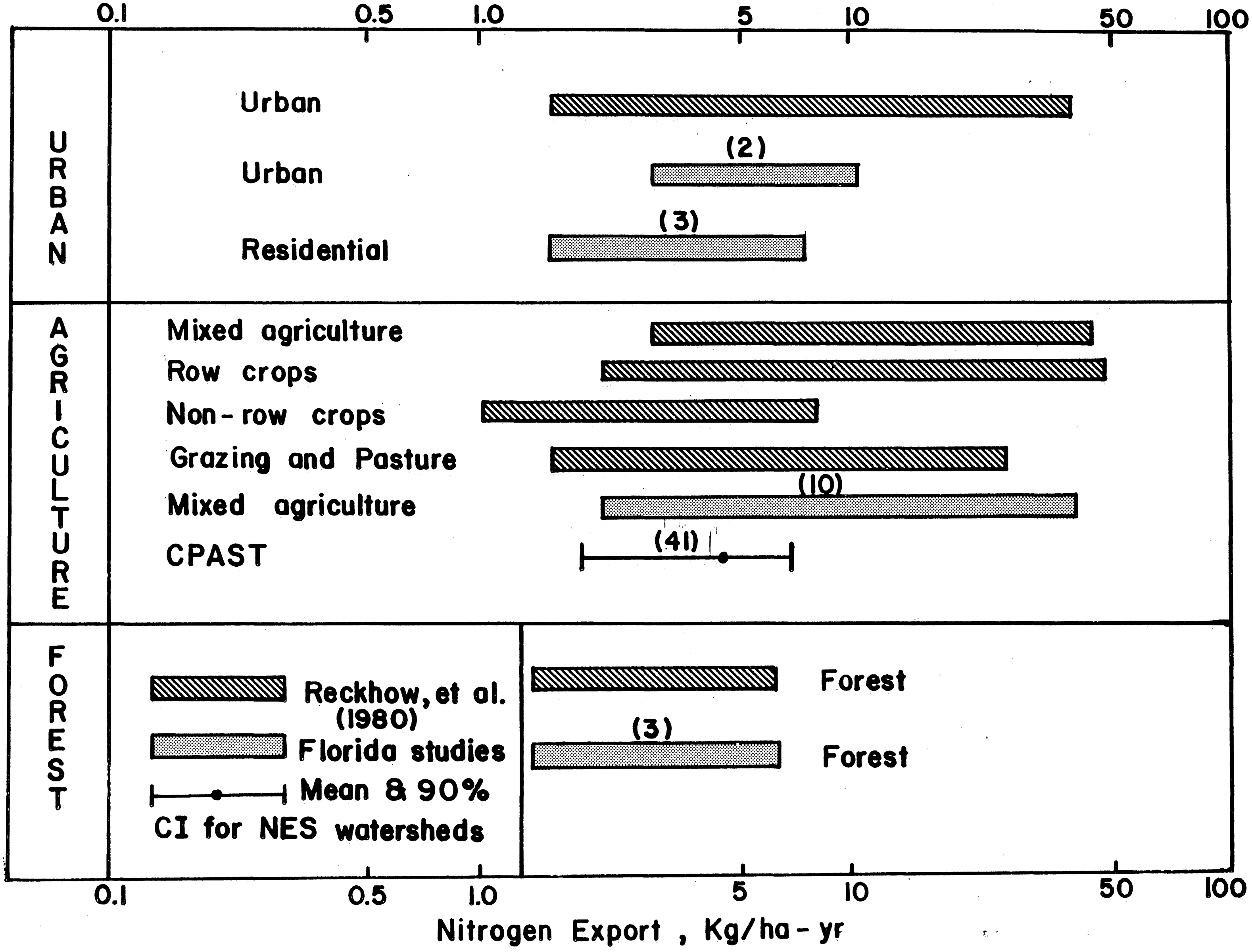

Figure III-2. Nitrogen export from urban, agricultural, and forested watersheds.

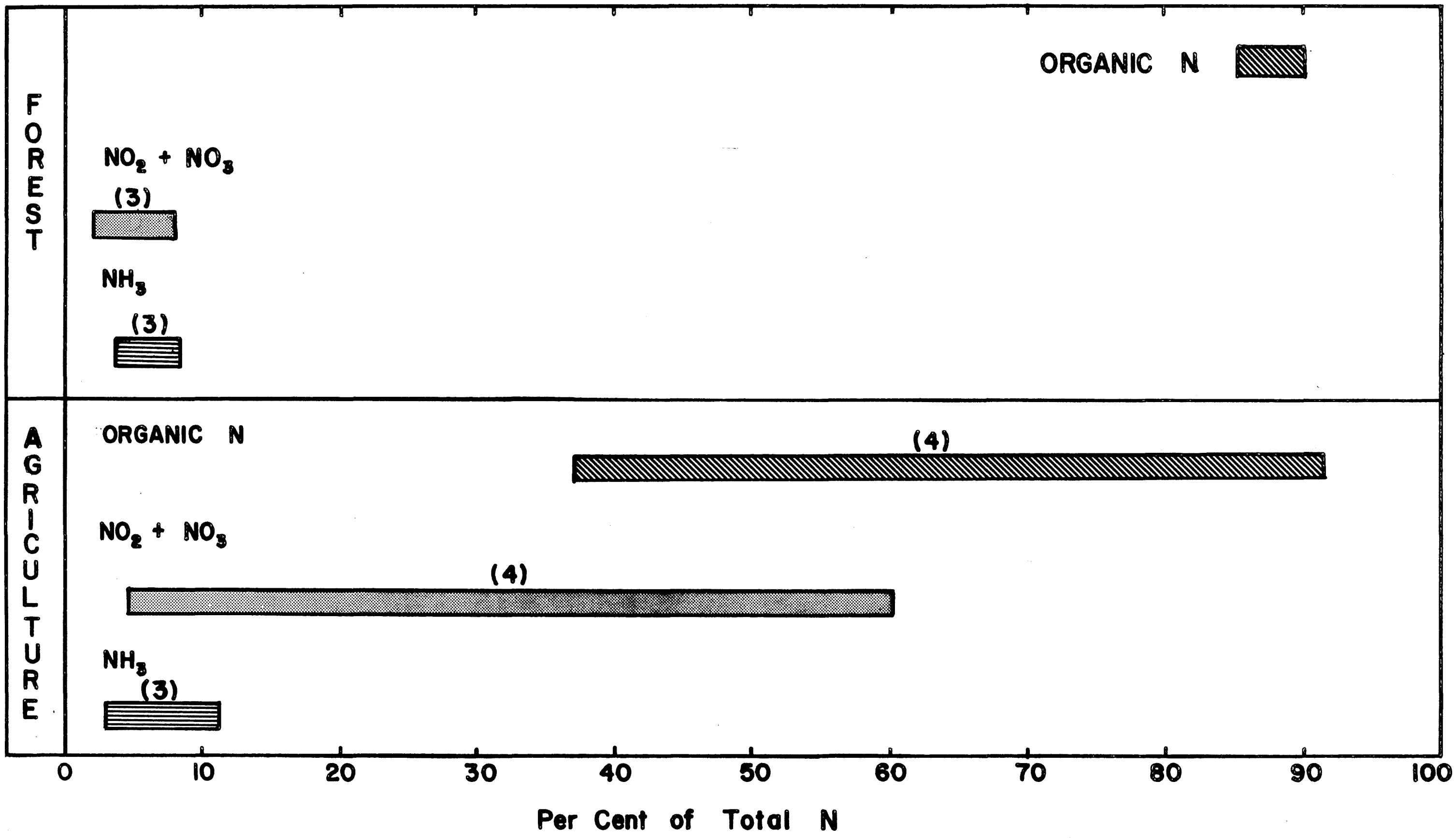

Figure III-3. Range of nitrogen fractions in runoff from forested and agricultural watersheds.

In order to determine which land uses were statistically significant as independent variables in these regression equations, the backward elimination procedure (STEPWISE/B) of SAS was used. In this procedure, all potentially significant independent variables (in this case, area in each land use) are entered in the regression equation. The least significant variables are deleted in sequential steps until all remaining independent variables are found to be statistically significant in contributing to the variability of the dependent variable. For these analyses, the 0.05 level of significance was used as the criterion for selecting independent variables for the final regression equations. The 95% CLI for the final regression equations were determined using the GLM procedure of SAS.

Prior to performing the regression analyses, a correlation matrix of the land use categories was computed, and it indicated that the categories "other urban" (OURB) and "residential" (RES) were significantly correlated ($r^2 = 0.25$). In order to avoid the problem of multicollinearity among independent variables (Neter and Wasserman 1974), these two categories were combined into an "all urban" (ALLURB) category. Thus, eight land use categories were initially included as independent variables in the regression equations: ALLURB, "crops and pastureland" (CPAST), "other agriculture" (OAG), "forest" (FOR), "rangeland" (RA), "non-forested wetland" (FOR), "open water" (WA) and "strip mine" (SMINE) (see Figure II-2). The land-use characteristics of the 41 NES watersheds used in these analyses are shown in Table III-7.

The utility of using land-use areas as independent variables in these equations was evaluated by also using total drainage area (DA) as the sole independent variable in equations to predict NPS nutrient loading, nutrient concentration, and flow. Regression results are summarized in Table III-8.

Equations to Predict Phosphorus Loading (TPL). The best equation to predict TPL, eq. 3-3, has an r^2 of 0.71 (P >0.0001) and includes three land use areas as statistically significant independent variables (ALLURB, CPAST, and RA). The relationship between DA and TPL (eq. 3-2) was much weaker ($r^2 = 0.21$), indicating that the use of individual land use areas results in much better predictions of TPL than does the use of DA alone (Table III-8).

The β value (phosphorus export coefficient) of ALLURB in eq. 3-3, 6.0 ± 1.4 kg/ha-yr, is high compared with phosphorus export coefficients reported for urban watersheds in other studies (Figure III-1). One likely explanation for this is that ALLURB is a well-defined and fairly restrictive land-use category (See Figure II-1), while the "urban" watersheds in other studies often included areas of forest and other land uses that tend to decrease the overall phosphorus export. The β value for CPAST in eq. 3-3, 1.2 ± 1.0 kg/ha-yr, falls within the range of phosphorus export coefficients reported for other agricultural watersheds in Florida and throughout North America (Figure III-1).

The β value for RA in eq. 3-3, (-1.4 kg/ha-yr) is a statistical anomoly that reflects an inherent weakness of the regression approach in evaluating nutrient export. However, since RA comprised little of the average watershed area (8%), the magnitude of its coefficient has relatively little effect on the predictions produced using eq. 3-3. In contrast, CPAST comprised

Table III-7. Land use characteristics of study watersheds.

Land Use (%)(1)	Computer Code	Mean	Minimum	Maximum
Other urban	OURB	8.2	0.0	32.5
Residential	RES	18.5	0.0	62.8
Total urban (2)	ALLURB	26.7	0.0	78.5
Crop and pasturaland	CPAST	19.6	0.0	70.0
Other agriculture	OAG	11.2	0.0	60.5
Forest	FOR	19.2	0.0	93.5
Range	RA	8.3	0.0	50.6
Non-forested wetland	NFWET	7.6	0.0	46.1
Open water	WA	5.7	0.0	33.3
Strip mine	SMINE	0.9	0.0	25.4
Drainage Area (km^2)	DA	135	0.4	978

(1) Land uses defined in Chapter II.

(2) ALLURB = OURB + RES.

Table III-8. Regression equations to predict nutrient loading, nutrient concentration and flow for 41 NES watersheds.

Equation	Dependent variable	Intercept		Independent variables	β		n	r^2	P > F
		Estimate	Std. Error		Estimate	Std. Error			
3-2	TPL	7123	3663	DA	0.48	14.6	41	0.21	0.002
	(kg/yr)			(ha)					
3-3	TPL	1705	2373	ALLURB	6.0	0.7	41	0.71	0.0001
	(kg/yr)			CPAST	1.2	0.5			
				RA	-1.4	0.7			
				(ha)	(kg/ha-yr)				
3-4	TNL	8235	8897	DA	5.1	0.4	41	0.84	0.001
	(kg/yr)			(ha)	(kg/ha-yr)				
3-5	TNL	15145	8179	CPAST	4.2	1.2	41	0.87	0.001
	(kg/yr)			NFWET	16.1	1.8			
				WA	16.0	3.4			
				(ha)	(kg/ha-yr)				
3-6	TOTAL P	0.36	0.06	ALLURB	5.6×10^{-5}	1.8×10^{-5}	41	0.24	0.005
	(mg/L)			RA	-2.6×10^{-5}	1.2×10^{-5}			
				(ha)					
3-7	FLOW	1,314,000	3,794,000	DA	0.31	0.02	41	0.91	0.0001
	(m^3/yr)			(m^2)	(m/yr)				
3-8	FLOW	2,765,000	2,978,000	ALLURB	0.25	0.09	41	0.96	0.0001
	(m^3/yr)			FOR	0.55	0.04			
				RA	0.49	0.08			
				CPAST	0.15	0.06			
				OAG	0.75	0.09			
				(m^2)	(m/yr)				

an average of 19.6% of the total watershed area, while ALLURB comprised an average of 26.7% of the total watershed area.

The best predictive equation found to relate the concentration of total phosphorus (TOTAL P) to land use is eq. 3-6, which includes only ALLURB and RA as independent variables. The relatively low r^2 for this equation (0.24, P> .005) suggests that TOTAL P is not greatly affected by land use in these watersheds. The strong correlation found between TPL and land use thus suggests that variations in flow rather than concentration are responsible for variations in TPL among watersheds.

The mean non-point source loading of phosphorus for all 41 watersheds is 1.0 kg/ha-yr. Since precipitation contributes only 0.3 to 0.7 kg P/ha-yr to Florida watersheds (Table III-2), it can reasonably be concluded that there is a net addition of phosphorus to the runoff water from within the watersheds.

Predictions of Nitrogen Loading (TNL). Unlike the situation for TPL, there is a strong correlation between TNL and DA (r^2 = 0.84, P> 0.001), indicating that drainage area alone is a reasonably good predictor of non-point source nitrogen loading (eq. 3-4). When the areas in individual land uses were used as independent variables, three terms (CPAST, NFWET and WA) were found to be significant at the 0.05 level, producing an equation (eq. 3-5) with an r^2 of 0.87 (P> 0.001). Thus, the use of individual land use areas as independent variables rather than total watershed area alone contributes little toward improving predictions of TNL.

Two of the significant terms in eq. 3-5 are NFWET and WA, both of which have nitrogen coefficients near 16 kg/ha-yr. The magnitude of these export coefficients is high compared to the mean input of nitrogen from precipitation (7.5 kg/ha-yr). The high loadings for open water and non-forested wetland may be caused by inputs from surrounding land uses, such as loadings from lawn fertilization and septic tanks associated with shoreline development. Alternatively, the high reported non-point source loadings for these land uses may reflect inaccuracies in the computation of non-point source loadings.

The only other significant term in eq. 3-5 is CPAST, which has a nitrogen export coefficient of 4.2 ± 2.4 kg/ha-yr. It can be seen (Figure III-2) that this 95% C.I. for nitrogen loading from agricultural areas is within the lower end of the range of values reported for agricultural watersheds in Florida (Table III-5) and throughout North America.

The mean non-point source loading for nitrogen in all 41 study watersheds is 5.7 kg/ha-yr., which is somewhat lower than the mean precipitation loading of 7.5 kg/ha-yr reported by Hendry et al. (1981) and Brezonik et al. (1981). Thus it appears that there is a net accumulation of nitrogen in these watersheds.

Equations to Predict Flow (FLOW). As shown by eq. 3-7, FLOW is highly correlated with DA (r^2 = 0.91, P> 0.0001). The β value for DA in eq. 3-7, 0.31 m/yr, is about 24% of the mean annual precipitation for the NES watersheds. The use of individual land use areas rather than DA improves the prediction of flow (eq. 3-8) only slightly ($r^2 = 0.96$, $P > 0.0001$). The β values in eq. 3-8 are runoff coefficients (m/yr), which are significant at the 0.05 level for five land uses.

APPLICATION

The accuracy of predictions for nutrient loading and flow can be evaluated using the 90% and 95% CLI's shown in Figures III-4 to III-9. The CLI is the confidence limit for individual predictions, and represents the degree of confidence with which a regression equation can be used to produce new predictions. Since the width of the confidence interval remains approximately constant, predictions of high values are relatively more accurate than are predictions of low values. For example, the 90% CLI for predictions of TPL using equation 3-3 is approximately ± 21,000 kg P/yr. Thus, a watershed having a predicted phosphorus export of 13,600 kg P/yr (the mean value) has a 90% CLI of ± 21,000 kg P/yr, or ± 154% of the predicted value. However, a watershed having a predicted phosphorus export of 40,000 kg P/yr also has a 90% CLI of around 21,000 kg P/yr., so the relative accuracy is improved to ± 53% of the predicted value. Predictions of TNL and FLOW are relatively more accurate than are predictions of TPL. For predictions of TNL the 90% CLI is approximately ± 80,000 kg N/yr, or ± 104% of the predicted value for a watershed having than mean value of nitrogen export (77,000 kg/yr), using either equation 3-5 (Figure III-6) or equation 3-6 (Fig. III-7). The 90% CLI for predictions of FLOW using equation 3-8, is ± 32.5×10^6 m^3/yr, or ± 75% of the predicted value for a watershed having the mean flow of 43.1×10^6 m^3/yr (Figure III-9). However, as seen in Figure III-8, the confidence intervals for equation 3-7, in which DA is the only independent variable, are comparable.

Since the use of individual land use areas rather than drainage area alone does little to improve predictions of TNL and FLOW, it is suggested that equations 3-5 and 3-7, in which DA is the sole independent variable, be used for predictions of these parameters. For predictions of TPL, the use of equation 3-3, in which land use areas are used as independent variables is recommended. In applying these equations, confidence intervals should be used to assesses the degree of reliability associated with each prediction.

In using these equations to predict nutrient loadings, nutrient concentration or runoff in other Florida watersheds, several points should be emphasized:

1) These predictive equations were developed using a relatively small group of watersheds that are not necessarily representative of all Florida watersheds. Predictions made using these equations are valid for only those watersheds that are from the same population as the test watersheds. Thus, in evaluating the applicability of these equations for a new situation, the user is urged to compare the land use characteristics of the new watershed(s) with those of the watersheds used in these analyses (See Table III-7). For example, it would be inappropriate to apply the predictive equations developed for urbanized watersheds to a new watershed that is composed

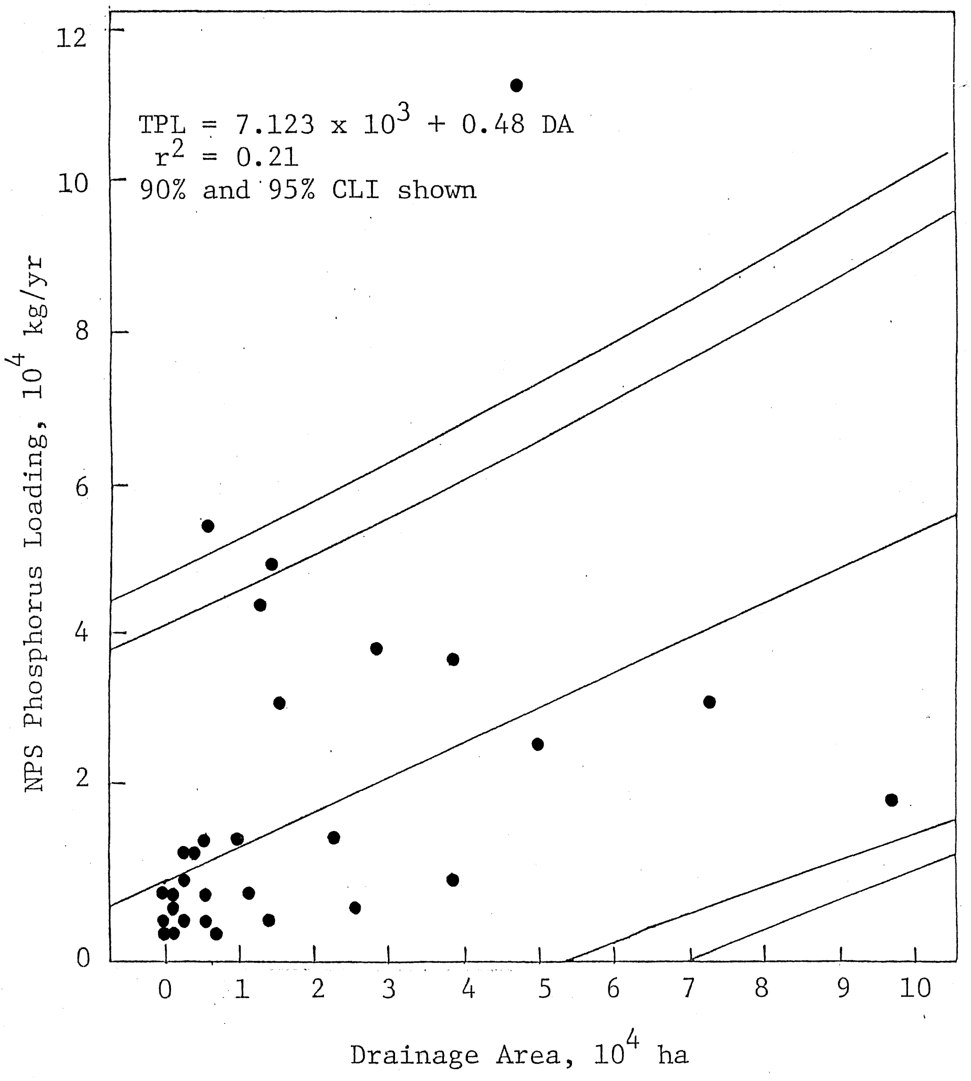

Figure III-4. Non-point source phosphorus loading vs. drainage area for 41 NES watersheds.

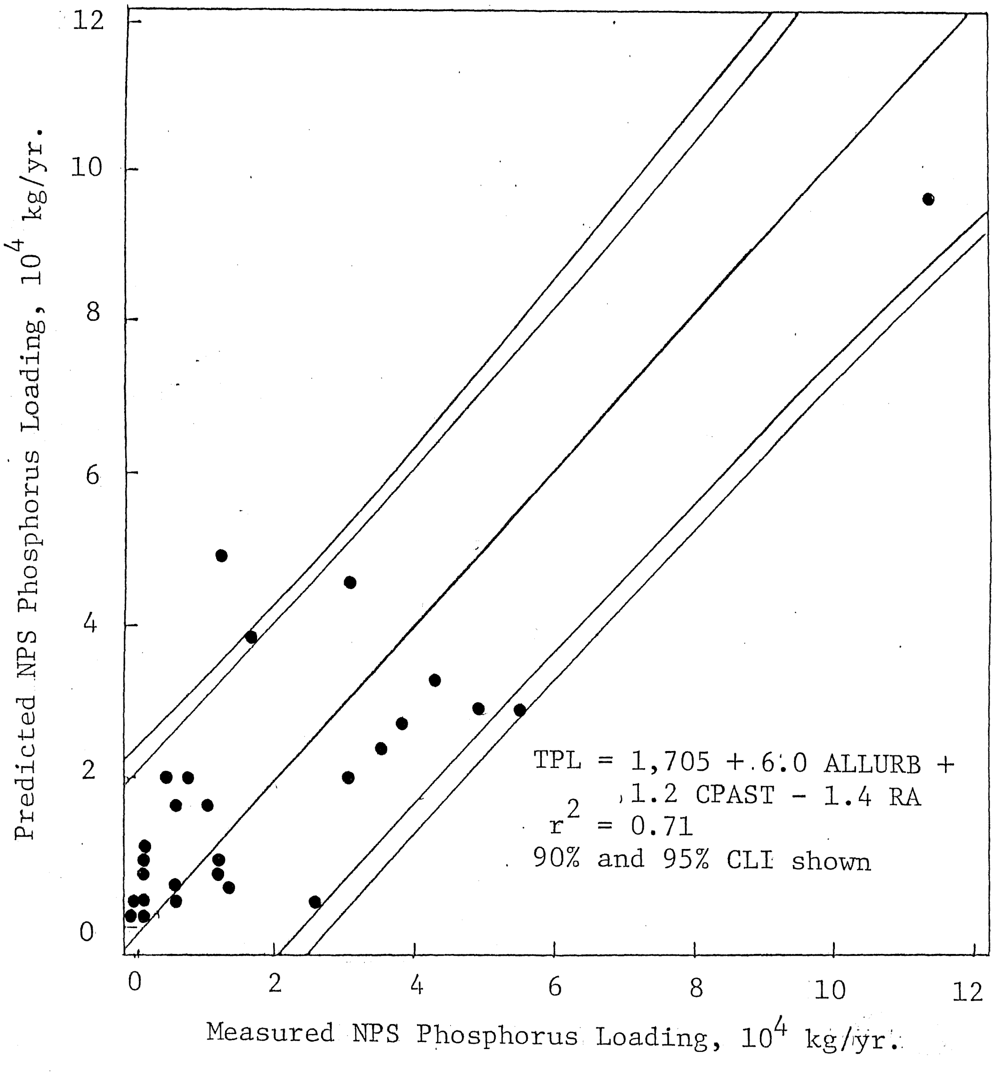

Figure III-5. Predicted vs. measured NPS phosphorus loading using Equation 3-3.

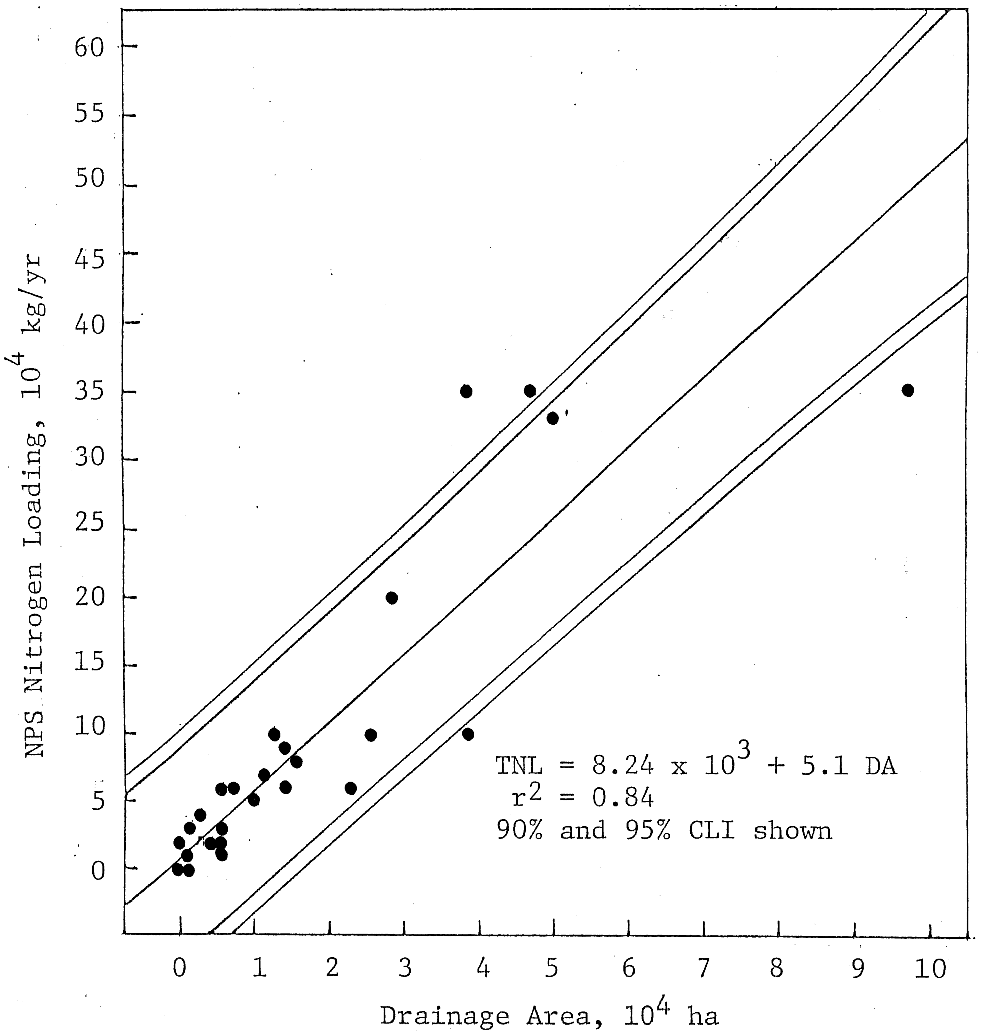

Figure III-6. Non-point source nitrogen loading vs. drainage area for 41 NES watersheds.

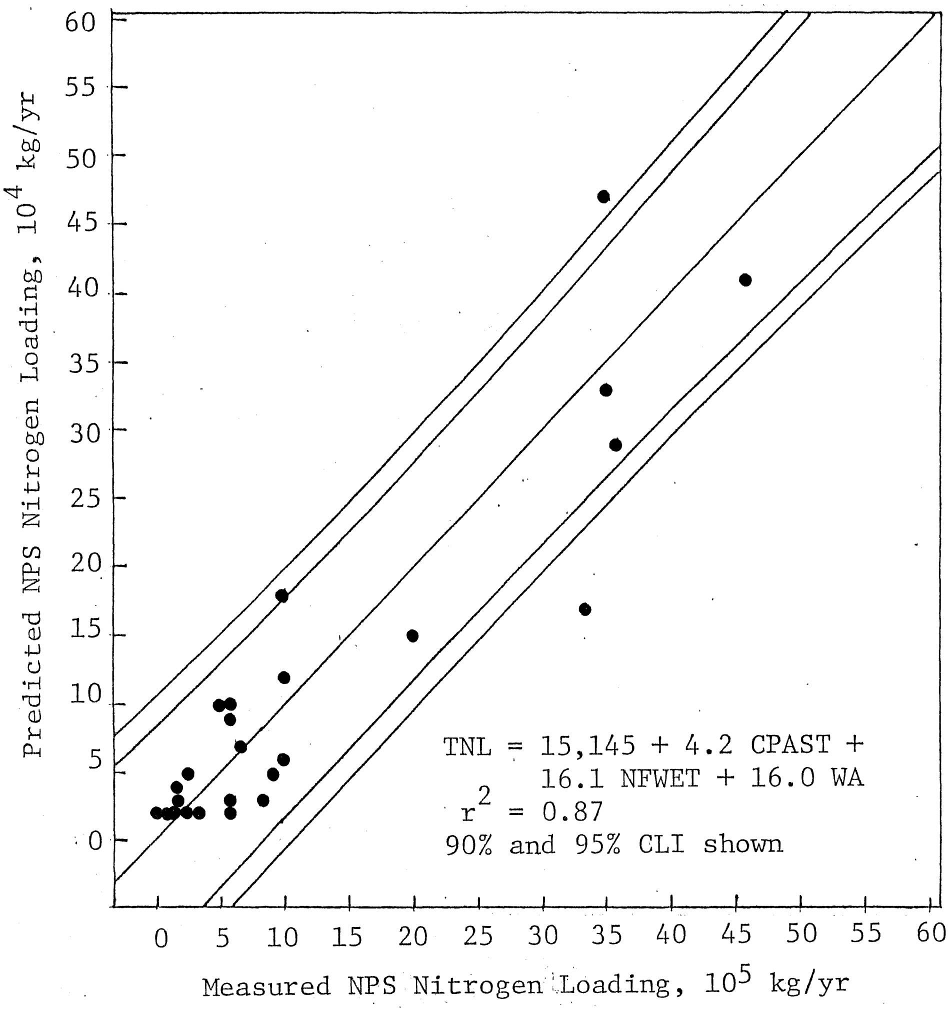

Figure III-7. Predicted vs. measured NPS nitrogen loading using Equation 3-5.

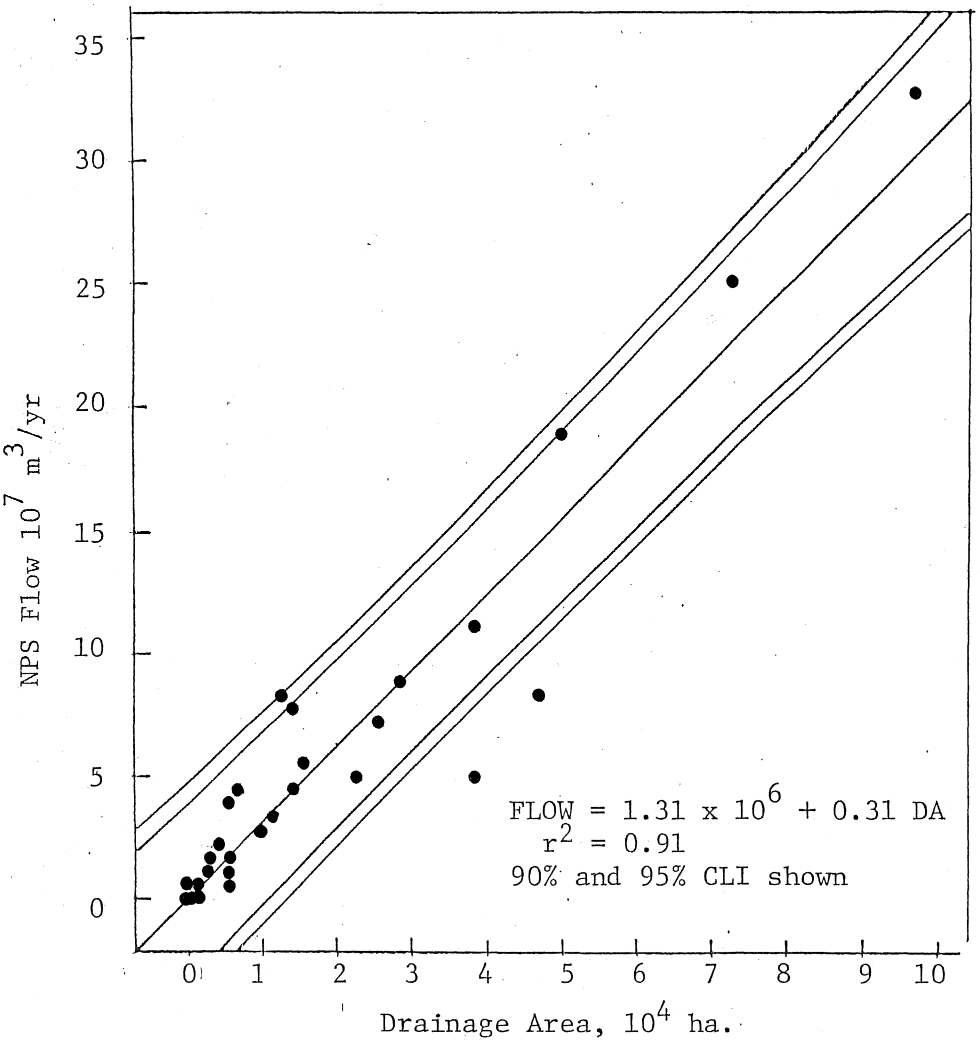

Figure III-8. Non-point flow vs. drainage area for 41 NES watersheds.

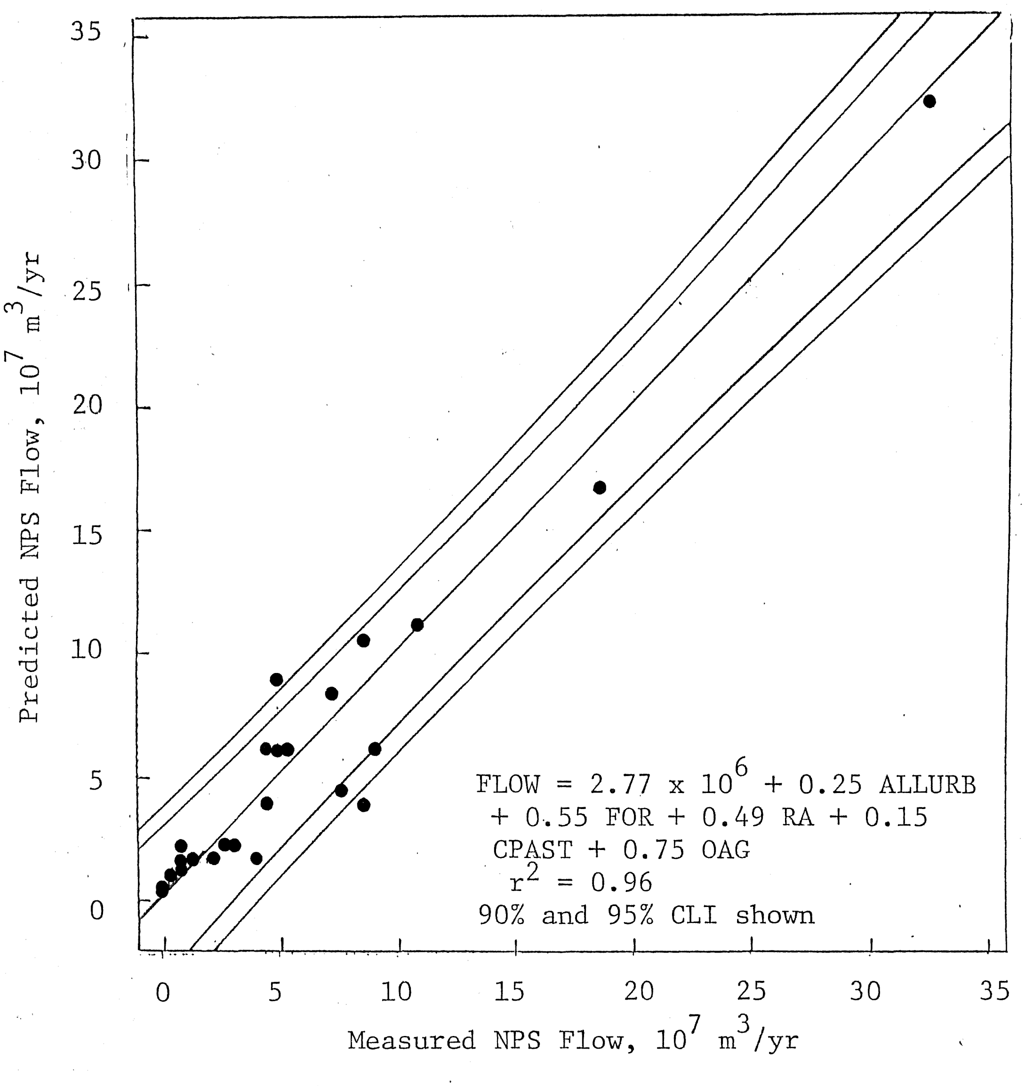

Figure III-9. Predicted vs. measured NPS flow using Eq. 3-8.

of 90% ALLURB, since the largest fraction of any of the watersheds used in developing these equations covered by ALLURB is 78.5% (Table III-7).

2) It would be inappropriate to use these predictive equations to estimate the change in loading that would occur if a portion of a watershed is converted from one land use to another (i.e., forest to residential). The reason for this is that regression analysis does not necessarily imply a causal relationship between the independent variables (i.e., nutrient loading) and the dependent variables (land uses). Underlying factors, such as soil type, drainage and slope may be related to both land use and nutrient loading. Thus, the difference in loading between forest and urban areas reflects, in part, differences in edaphic features of the landscape that render certain areas suitable for urban development and others less so. However, as development pressure increases, there will be a tendency to urbanize areas that are not currently considered suitable for urban development (i.e. under the conditions in which these predictive equations were developed). Because of this, the loading of nutrients from urban areas developed in the future may be different from the loading from urban areas currently in existence. In a similar vein, it should be realized that changes in management practices of existing land uses may alter the rate of nutrient export from some areas. For example, in the past 10 years, the practice of nitrogen fertilization in agricultural areas has changed considerably. These trends include 1) decreased application rates, 2) the improvement of tillage methods, 3) the use of fertilizers that are better retained in the soil column (i.e., urea and ammonium instead of nitrate), and 4) the application of nitrification inhibitors (Calvert 1975; Terman and Allen 1970). These trends should act to conserve fertilizers and reduce the export of nitrogen from agricultural areas.

3) The predictive equations developed here are based on <u>mean</u> annual loadings. As mentioned earlier, there may be considerable year-to-year variation in the annual nutrient loading from non-point sources. For some applications, such as the assessment of lake restoration techniques, it is important to be able to determine actual annual loadings of nutrients into a lake for a period of several years. For this type of study the predictive equations developed here would not be suitable.

4) In the development of these predictive equations it was assumed that all of the flow in a tributary other than that emanating from known point sources (i.e., sewage treatment plants), was derived from precipitation falling on the watershed. Several watersheds were excluded from the analyses because it was believed that the flows in the tributaries were too high to be the result of natural precipitation. In applying these predictive equations, the user should also be reasonably certain that the flow in a particular tributary is derived from natural precipitation and runoff.

For some watersheds, the use of these equations would not be suitable, either because of the limitations cited above or because their predictive capability is not adequate for a particular purpose. In some cases, the use of export coefficients obtained from the literature (Tables III-3 to III-6) may be more appropriate in generating preliminary estimates of NPS nutrient loading. This is particularly true for small watersheds in which one land use is predominant. When the literature-based approach is used, we suggest that original studies be consulted and that reported export coefficients be used only when the watersheds in which they were determined are similar, with respect to topography, soils and other characteristics, to the watershed under investigation.

When more accurate results are required or when the effects of proposed management strategies on nutrient loadings are being evaluated, the use of simulation models should be considered. Examples of these include a number of agricultural models (reviewed by Haith 1980) and the Storm Water Management Model, designed for use in urban areas (Heaney et al. 1976).

CHAPTER IV. LIMNOLOGICAL CHARACTERISTICS OF FLORIDA LAKES

The data compiled in this study enable a broad characterization of the limnological conditions of Florida's lakes. Data were collected for most major limnological parameters in all three studies (Table II-2), with the exception of color and turbidity, which were not collected by the National Eutrophication Survey. It should be noted that the sampling intensity for most of the lakes in the three surveys was low; most lakes were sampled only 3-4 times during an annual period. Considering the large variations (often order of magnitude) that may occur in algal standing crop and in the concentrations of major nutrient species during an annual period, computed means for any single lake must be accepted with caution. However, considering the large number of lakes in the data base, use of the data to make general inferences on limnological relationships in the lakes is justified.

MORPHOLOGICAL CHARACTERISTICS

Although Florida's lakes vary considerably in size and morphometry, most are quite shallow. Of the 101 lakes included in this study, only ten have maximum depths greater than 10 m, and only three (Annie, Kingsley and Mize) are over 20 m deep. The study lakes have a considerable range in surface area (Table IV-1). Many of the lakes in Alachua and Putnam counties have surface areas of only a few hectares, but several of Florida's largest lakes are included in this study. By far the largest is Lake Okeechobee (1890 km^2), which after the Laurentian Great Lakes is the largest freshwater lake (in surface area) entirely in the United States.

The morphometry of most Florida lakes has been affected by limestone solution processes and many have been formed in sinkhole depressions. Some of these lakes, like Lake Santa Rosa (Putnam County) are nearly circular, while others are complex dolines that have been formed in adjoining solution basins (e.g. Cowpen Lake in Putnam County).

The smaller lakes are often hydraulically connected to perched water tables that are separated from the main (Floridan) limestone aquifer by an clayey aquiclude. Many lakes are in seepage basins and lack distinct inflows or outflows. The water level in these lakes may fluctuate by as much as several meters between wet and dry periods. Water levels in the larger lakes are usually structurally controlled to minimize natural variations.

Because of their shallowness and the mild climate, most Florida lakes do not exhibit stable thermal stratification. Of the 55 lakes studied by Brezonik and Shannon (1971), only eight developed stable thermal stratification during the warm season. Approximately eight additional lakes showed evidence of temporary thermal stratification lasting for periods of a few weeks to a few months. The shallowness of Florida's lakes also encourages resuspension of sediments to the overlying water, particularly in large lakes and lakes with loose, flocculent sediments. Studies on Lake Apopka

Table IV-1. General characteristics of study lakes.

Morphological	Mean	Minimum	Maximum
Surface area, km^2	23.0	0.01	1890.7
Volume, $m^3 \times 10^6$	61.4	0.03	2494.0
Mean depth ($\bar{z}$), m	2.9	0.7	8.3
Maximum depth ($\bar{z}_{max}$), m	5.4	0.9	25.3
Chemical & Biological (annual means)			
Color, units	116	2	539
pH	7.1	4.7	10.4
Alkalinity, mg/L as $CaCO_3$	32	0	163
Chlorophyll *a* (chl *a*), μg/L	29.1	0.9	276.6
Total nitrogen ($\bar{N}$), mg/L	1.51	0.19	5.56
Total phosphorus ($\bar{P}$), μg/L	231	7	2,750

(Pollman and Brezonik 1981) indicate that phosphorus is released to the water column during wind events as the result of desorption from suspended sediments. This phenomenon undoubtedly contributes to eutrophication problems in such lakes.

CHEMICAL AND PHYSICAL CHARACTERISTICS.

The majority of Florida's lakes are poorly buffered (mean alkalinity = 32 mg/L as $CaCO_3$), even though they are underlain by limestone. This apparently contradictory situation occurs because many lakes are not hydraulically connected to the underlying limestone formations, but receive the bulk of their water either directly from precipitation or by surface and subsurface runoff from the sandy, low-calcareous soils. Many are also highly colored; the mean color for the study lakes is 116 CPU (chloroplatinate units). The frequency and intensity of color reflects the abundance of watersheds composed of pine forests and wetlands (swamps).

Relationship Between pH and Alkalinity. The pH of Florida lakewaters is strongly correlated with alkalinity (Figure IV-1). The solid line in the figure shows the relationship between pH and alkalinity in a water system where the carbonate buffering system controls pH at atmospheric pressure ($P_{CO_2} = 10^{-3.5}$ atm). The data suggest that except for the most acidic lakes the pH is near the equilibrium value defined by the CO_2-bicarbonate system. However, for many of the poorly buffered lakes in the Trail Ridge region, the observed pH is considerably lower than that expected for pure water in equilibrium with atmospheric CO_2 (pH 5.7). Brezonik et al. (1981) found that the pH of some of these lakes has decreased by as much as 0.5 pH units over the past 20 years, apparently as the result of inputs of acid precipitation. This trend is expected to continue as Florida increases its production of electricity by coal-fired electric plants.

Factors Affecting Transparency. Seechi disk transparency is one of the most commonly measured parameters in the study of lake eutrophication. In addition to being of scientific interest, Secchi disk transparency is readily comprehended by the public as a measure of water clarity. Several investigators (Bachman and Jones 1974; Carlson 1977; Brezonik 1978) have found a hyperbolic relationship between the concentration of chlorophyll *a* and Seechi disk transparency. Brezonik (1978) reviewed the theoretical relationship between light attenuation and Secchi disk transparency (SD) and concluded that for many lakes the equation:

$$1/SD = a\,[\text{color}] + b\,[\text{turbidity}] \qquad (4\text{-}1)$$

can be used to describe the variation in Secchi disk transparency. Since turbidity is often closely related to chlorophyll *a*, it was hypothesized that

$$1/SD \cong a\,[\text{color}] + b\,[\text{chl } a] \qquad (4\text{-}2)$$

The non-linearity between inverse Secchi disk and chlorophyll *a* observed by Carlson (1977) and Brezonik (1978) may be explained by 1) an increase in the amount of chlorophyll per cell in more eutrophic situations, 2) a

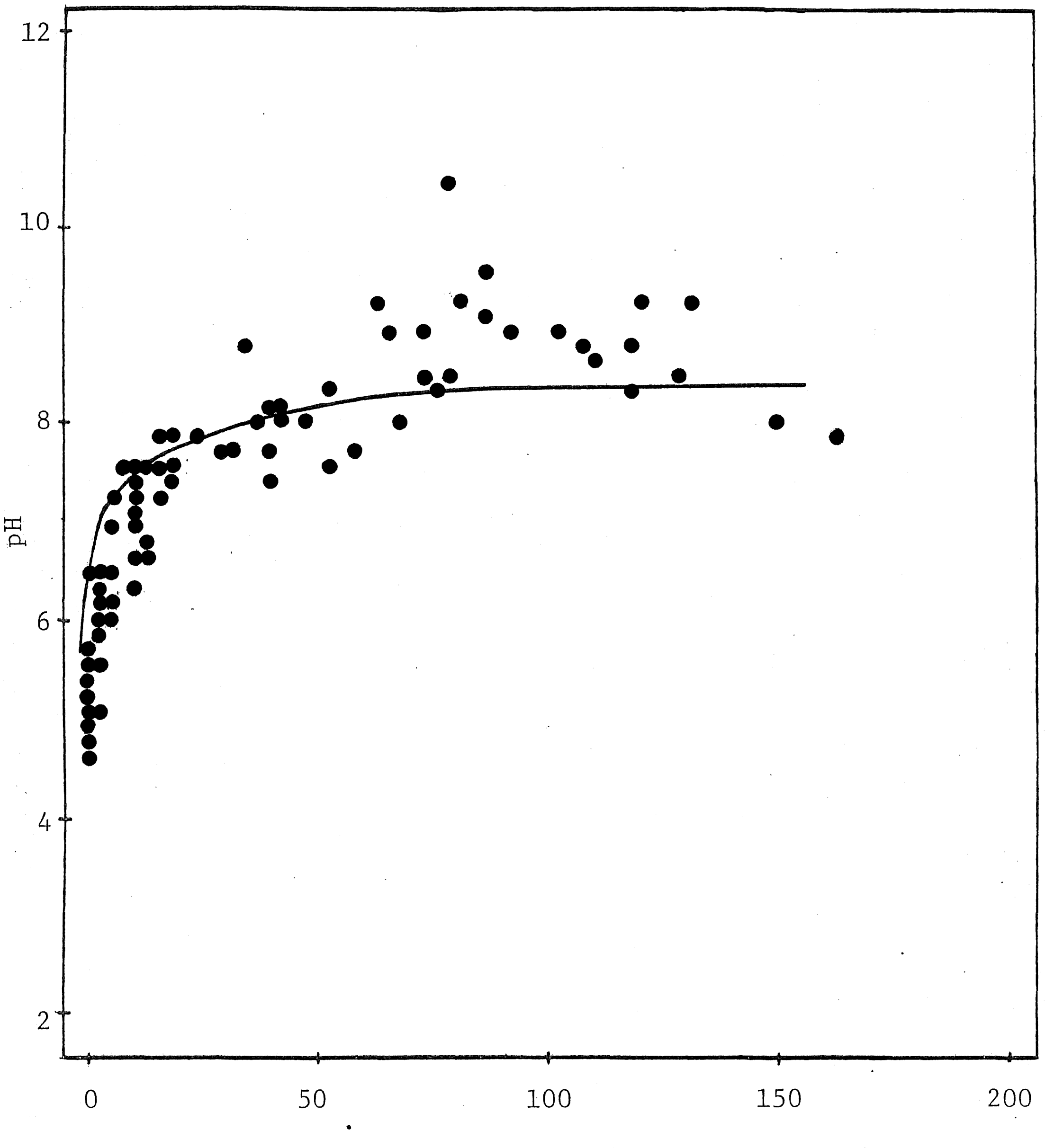

Figure IV-1. Mean pH vs. mean alkalinity for 101 study lakes. Solid line shows equilibrium relationship between pH and alkalinity at $pCO_2 = 10^{-3.5}$ atm.

change in the size distribution of the seston and in the morphology of algae with increasing eutrophication, or 3) light attentuation due to color (Brezonik 1978).

A statistical analysis of the relationship between inverse Secchi disk and color, turbidity, and chlorophyll *a* was conducted by Brezonik (1978) using data from the 55 lake study. A similar analysis was conducted for this report using the additional data from the two other lake surveys (NES 1977 and Brezonik et al. 1981). Results of the statistical analyses (Table (IV-2) are similar to those of Brezonik (1978), although the coefficients differ slightly, and the coefficients of determination (r^2 values) are slightly lower. The lower r^2 values may result from the more diverse group of lakes used in the present analysis. Differences in sampling and analytical procedures among the studies also may account for the slightly lower r^2 values.

When chlorophyll *a* is the only independent variable, a log-log model yields better predictions of SD transparency than does an inverse relationship (cf. eq. 4-2 and eq. 4-8). This result is consistent with the findings of Carlson (1977), Bachman and Jones (1974), and Brezonik (1978). The coefficient of determination for the log SD vs. log (chl *a*) relationship is lower for Florida lakes ($r^2 = 0.70$) than reported by Carlson (1977) and Bachman and Jones (1974) for temperate zone lakes (r^2 0.86 and 0.90, respectively) because color affects SD transparency more in Florida lakes than in temperate zone lakes. As seen by equations 4-4 and 4-9, color alone is reasonably good predictor of SD transparency.

The best equation for predicting 1/SD (eq. 4-6) uses turbidity and color as independent variables (equation 4-6). Apparently, the use of turbidity overcomes some of the problems mentioned above that are encountered when (chl *a*) is used to represent light attenuation. Despite the apparent good fit (Figure IV-2), this equation does not predict SD transparency accurately in some of the clear, oligotrophic lakes in the Trail Ridge group. The intercept value of eq. 4-6 ($0.17 m^{-1}$) corresponds to a SD of only 5.9 m at zero color and turbidity in the water column. This transparency is far lower than that expected in a water column devoid of algae and color. Hutchinson (1957) reported that the maximum SD transparency ever measured in a lake is about 40 m, which would give an intercept term in eq. 4-6 of 0.025. The failure of this equation to predict high SD values accurately reflects the fact that high values of SD have very low inverse values that do not affect the fit of the regression equation as much as do lower SD values. Furthermore, the data base does not include any lakes with very high transparency values; the maximum mean SD in the data set is 7.9 m ($1/SD = 0.013 m^{-1}$) for Lake Sheeler. However, eq. 4-6 does produce reasonable estimates of SD transparency for predicted SD values less than 3 m. This situation accounts for most of the lakes in the data set, and probably most lakes in Florida. For lakes with a predicted SD >3m., eq. 4-6 underestimates the actual SD transparency.

BIOLOGICAL CHARACTERISTICS.

Phytoplankton Communities. The composition and standing crop of phytoplankton communities in the study lakes is highly variable. The most pris-

Table IV-2. Regression equations to predict Secchi disk transparency. (1)

Inverse relationships

$$1/SD = 0.80 + 0.01\ (\text{chl}\ a) \tag{4-2}$$
$$r^2 = 0.33 \quad n = 100$$

$$1/SD = 0.49 + 0.12\ (T) \tag{4-3}$$
$$r^2 = 0.62 \quad n = 63$$

$$1/SD = 0.70 + 0.002\ (C) \tag{4-4}$$
$$r^2 = 0.70 \quad n = 63$$

$$1/SD = 0.37 + 0.03\ (\text{chl}\ a) + 0.001\ (C) \tag{4-5}$$
$$r^2 = .62 \quad n = 63$$

$$1/SD = 0.17 + 0.11\ (T) + 0.002\ (C) \tag{4-6}$$
$$r^2 = 0.82 \quad n = 63$$

Log-log relationships

$$\log\ (SD) = 0.49 = 0.76 \log\ (T) \tag{4-7}$$
$$r^2 = 0.55 \quad n = 63$$

$$\log\ (SD) = 0.55 - 0.47 \log\ (\text{chl}\ a) \tag{4-8}$$
$$r^2 = 0.70 \quad n = 100$$

$$\log\ (SD) = 0.78 - 0.39 \log\ (C) \tag{4-9}$$
$$r^2 = 0.53 \quad n = 63$$

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(1) SD = mean Secchi disk transparency, m; C = mean color, Pt units; T = mean turbidity, FTU, (chl *a*) = mean chlorophyll *a* concentration, µg/L.

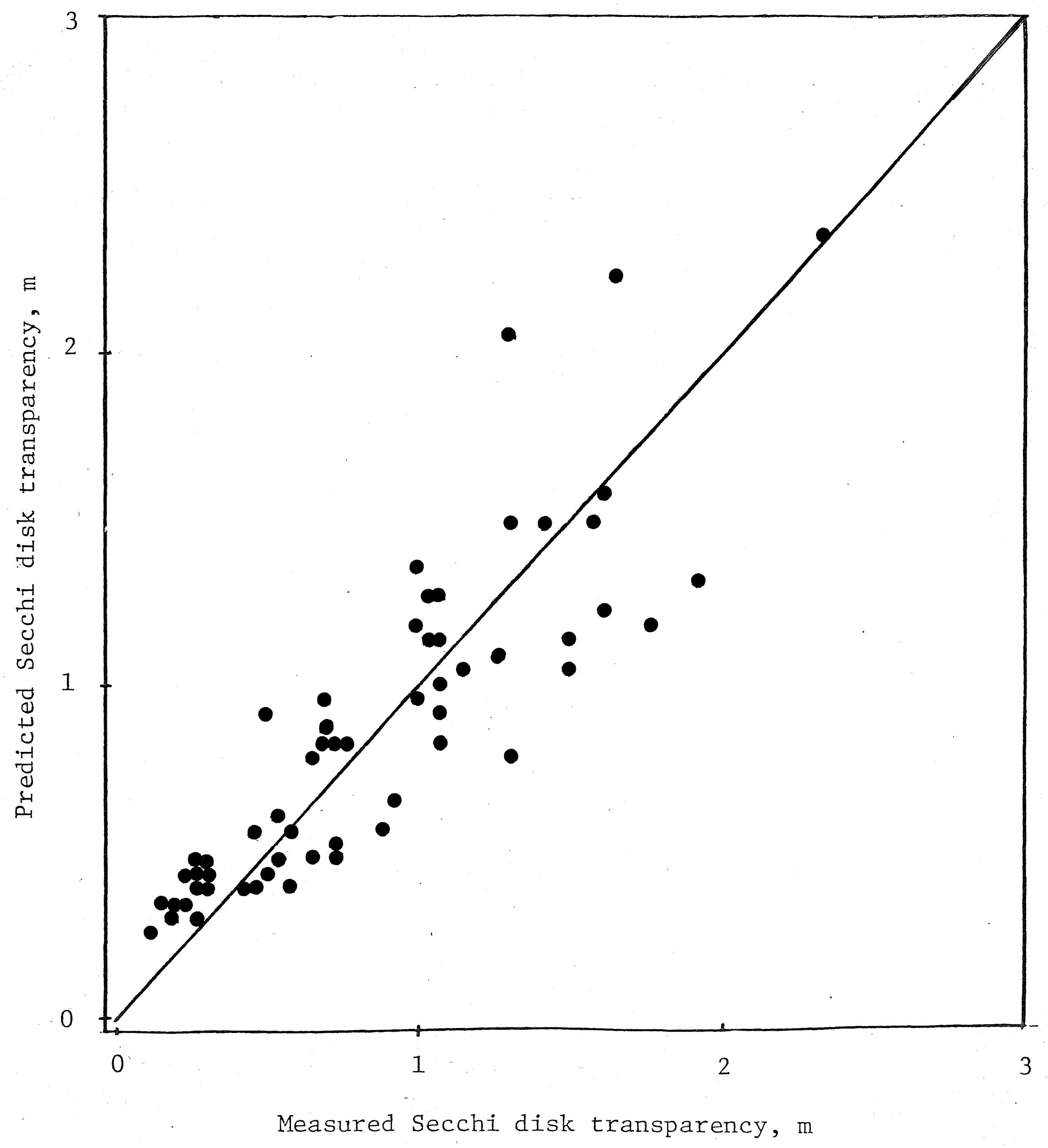

Figure IV-2. Predicted vs. measured Secchi disk transparency using Eq. 4-6.

tine are those in the Trail Ridge region. Most of these lakes (except Altho, Geneva and Kingsley) have mean pH values of <5.6 and phytoplankton communities typical of acidic, oligotrophic lakes (Schulze 1980). These communities are dominated by Staurastrum sp., Scenedesmus sp., Ankistrodesmus falcatus, Peridinium inconspicuum and several small green coccoids. Blue-green algae occur in low abundance in these lakes and are represented mainly by Oscillatoria limnetica and Anacystis incerta. At the other extreme, many of the study lakes are highly eutrophic (e.g. Biven's Arm, Newnan's, Wauberg, Apopka and Kanapaha). These lakes may exhibit dense algal blooms dominated by blue green and green algae; in some of the most fertile these blooms are virtually continuous.

Fish Populations. A recent survey of 22 Florida lakes describes the changes that occur in fish communities with increasing eutrophication (Kautz 1981). The oligotrophic lakes, characterized by well-developed communities of littoral vegetation, limited planktonic production, and sandy bottoms covered with a thin layer of detritus, have fish communities dominated by populations of sport fishes (largemouth bass, bluegill and other sunfish, striped bass, and pickerel) and forage fishes (Fig. IV-3). Populations of rough fishes (gar, gizzard shad, bowfin, and tilapia) and commercial fishes (primarily catfishes) are limited and the total biomass and species diversity of the fish communities is low.

The mesotrophic-eutrophic lakes have the best developed populations of sport and forage fishes (Fig. IV-3). These lakes are characterized by well-developed communities of littoral vegetation, large populations of benthic invertebrates and planktonic organisms, and high habitat diversity. The total density and species diversity of the fish communities was higher in these lakes than in either the oligotrophic lakes or the hypereutrophic lakes. Total fish biomass reached a maximum in the mesotrophic-eutrophic lakes and fluctuated about this maximum in the hypereutrophic lakes.

The fish communities of the hypereutrophic lakes are dominated by filter-feeding rough fish such as the gizzard shad. Populations of sport and forage fishes are limited by the reduced littoral vegetation, blooms of blue-green algae, accumulations of detritus on the bottom, and periods of reduced oxygen concentration. Commercial fishes reached their highest levels in the hypereutrophic lakes, accounting for 10% of the total biomass.

Nutrient Limitation. An assessment of the limiting nutrient(s) in a lake is important because the control of inputs of a limiting nutrient to a lake may be used as a means by which to control eutrophication. In the majority of temperate zone lakes, phosphorus is the limiting nutrient for algal growth, while nitrogen is limiting in most of the others. For example, Miller et al. (1974) found that phosphorus was limiting algal productivity in 71% of the lakes and N was limiting in 16% of the 49 temperate zone lakes they examined using the algal assay procedure. Other nutrients than nitrogen or phosphorus were limiting in the remaining lakes. A variety of other micronutrients, including iron, several other trace metals (Goldman 1972), and carbon (King 1972) have also been identified as limiting nutrients in freshwater lakes.

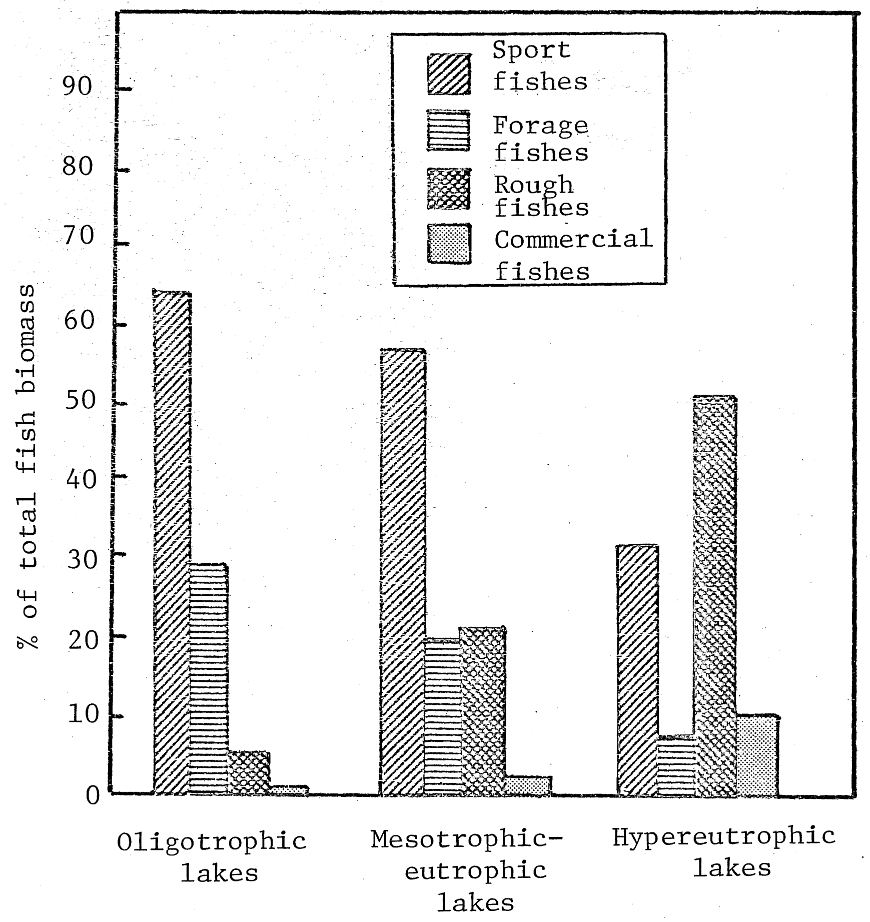

Figure IV-3. Structure of fish communities in Florida lakes. Data from Kautz (1981).

Of the methods used to evaluate nutrient limitation, the most common are (1) enrichment bioassays and (2) the computation of nutrient ratios, usually the ratio of total soluble inorganic N to soluble reactive phosphate (SIN:SRP). In enrichment bioassays, various nutrients are added to aliquots of the test water. Either a test species of algal (usually Selenastrum capricornutum) or an indigenous mixed culture, are added to each treatment, and the limiting nutrient is determined by comparing the growth of algae in the nutrient-spiked aliquots with a control. The procedure has become highly standardized in the Algal Assay Procedure: Bottle Test (Miller et al. 1978) and is widely used. The second method involves the measurement of nutrient concentrations and computation of the ratio SIN:SRP. this ratio has been compared with the results of nutrient enrichment bioassays (cf. Chiaudini and Vighi 1974; Miller et al. 1975). The "critical ratio" usually falls between 10:1 and 20:1. Porcella and Bishop (1975) stated that a SIN:SRP ratio less than 10:1 clearly indicates N-limitation, a ratio greater than 20:1 indicates P-limitation, and an intermediate ratio indicates mixed nutrient limitation.

Algal bioassays (AAP:BT) conducted for 31 of the NES lakes (NES 1977) indicated that 23 (74%) were nitrogen limited, seven (23%) were phosphorus limited and one had mixed nutrient limitation. When the results of nutrient bioassays are compared with the criteria of Porcella and Bishop for nutrient limitation (Figure IV-4), few misclassifications occur. For the 27 lakes having an SIN:SRP ratio of less than 10:1, 23 were found to be nitrogen limited in the AAP:BT and four were found to be phosphorus limited or have mixed nutrient limitation. All four lakes with SIN:SRP ratios >10:1 in which biassays were conducted were phosphorus limited. Thus, the criteria of Porcella and Bishop seem to be reasonably valid, although more data are needed for lakes having high SIN:SRP ratios to make conclusive remarks concerning the application of these criteria in Florida lakes.

A frequency distribution of mean SIN:SRP ratio for all 101 study lakes (Figure IV-5) shows that 46% have SIN:SRP ratios <10:1, 21% have ratios between 10:1 and 20:1, and 33% have ratios >20:1. Thus, if the proposed criteria of nutrient limitation are valid, nearly half of the study lakes are nitrogen-limited while only a third are phosphorus-limited.

A plot of the mean SIN:SRP ratio versus the mean chlorophyll *a* for each lake (Figure IV-6) shows an inverse relationship. Lakes with high algal standing crops (mean chl *a* >50 μg/L) have values of SIN:SRP <10, whereas lakes with low algal standing crops show more variability but often have mean SIN:SRP ratios >>20. This observation is consistent with that of Miller et al. (1974), who concluded that highly productive lakes are more likely to be nitrogen limited than are less productive lakes. Several explanations may account for this observation. First, nutrient inputs to eutrophic lakes are relatively more enriched with phosphorus than are the inputs to less productive lakes. Sewage effluents, for example, have low ratios of both SIN:SRP (<2:1) and total N: total P (<3:1) and are thus strongly nitrogen-limited (Gakstatter et al. 1978). Furthermore, the presence of phosphate-rich deposits and sandy soils with a low phosphorus retention capacity in Florida may result in relatively high orthophosphate concentrations in many streams. Odum (1953) reported that the concentration of orthophosphate in

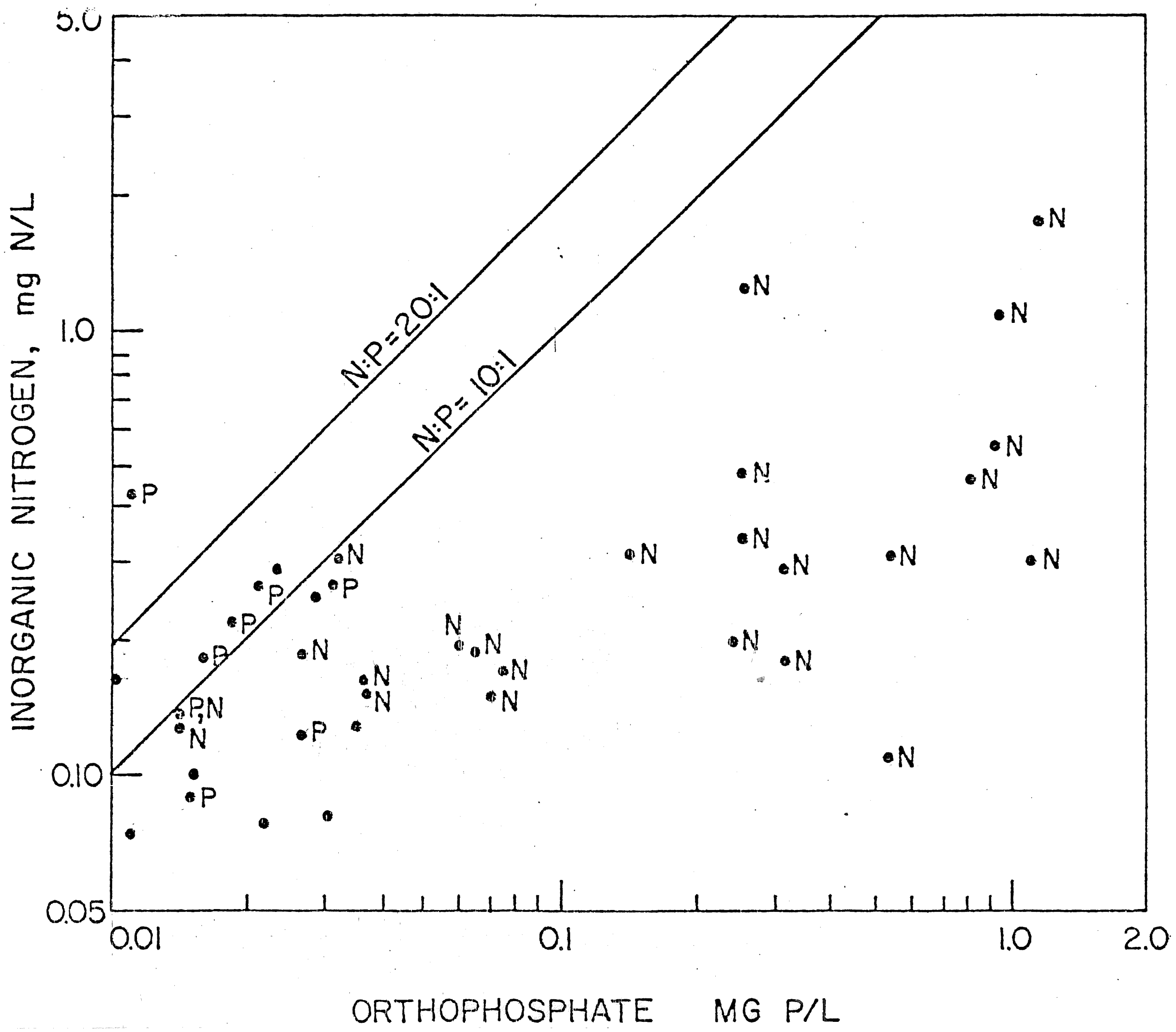

Figure IV-4. Total inorganic nitrogen vs. orthophosphate concentrations (mean values) in the Florida NES lakes. N and P next to data points indicate nutrient found limiting in lake by Algal Assay Procedure.

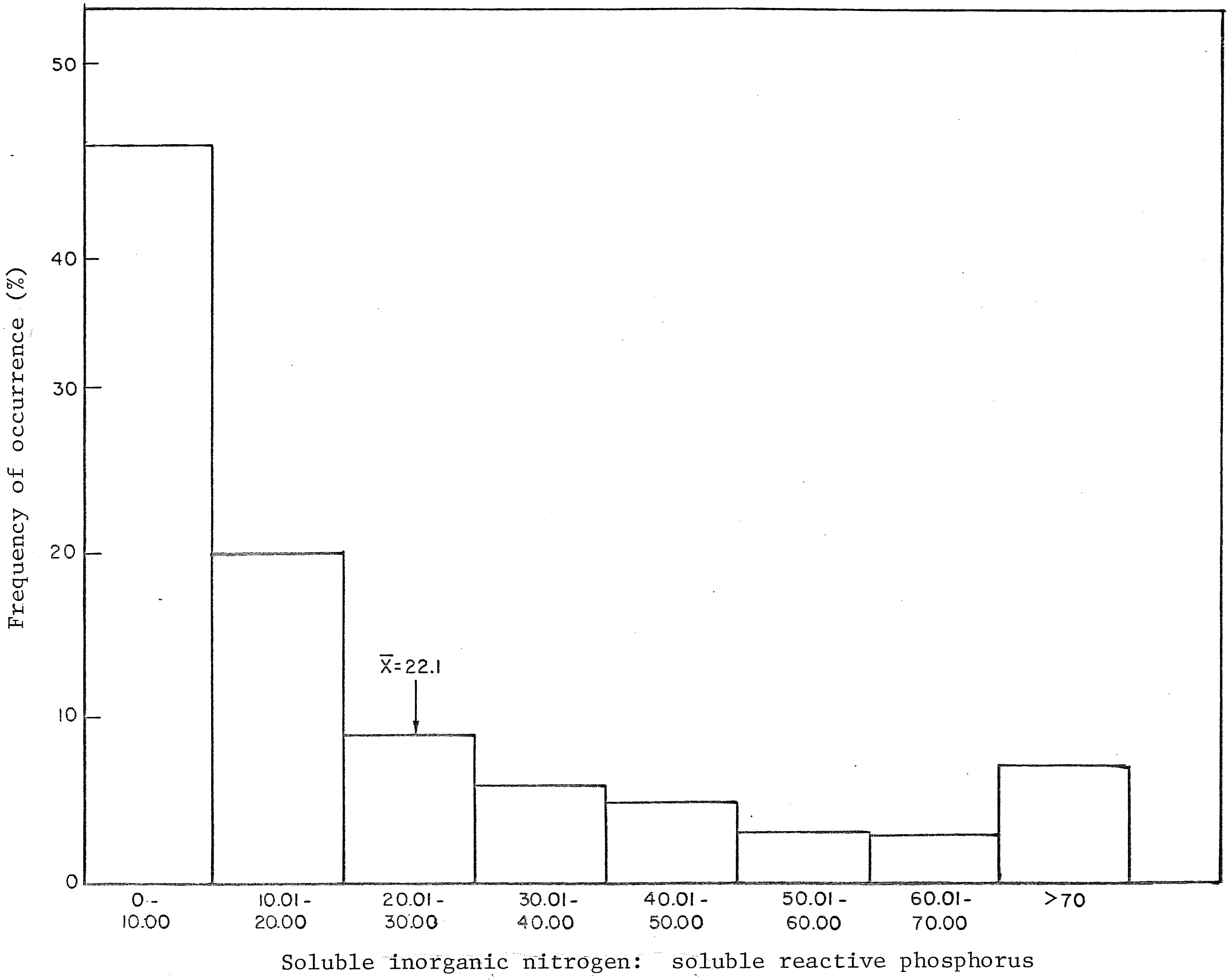

Figure IV-5. Frequency distribution of mean SIN:SRP ratios in the 101 study lakes.

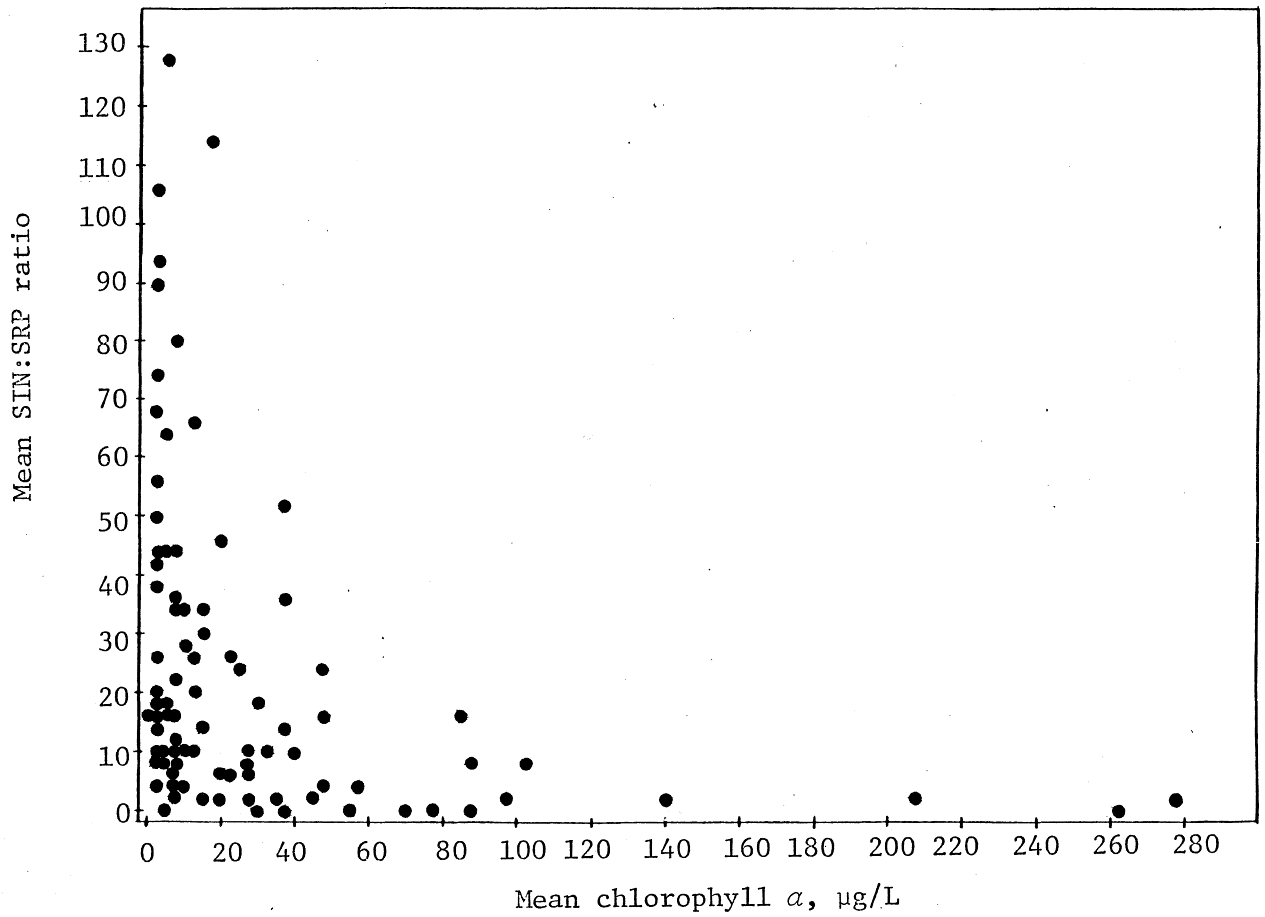

Figure IV-6. Relationship between soluble nitrogen: soluble reactive phosphorus (SIN:SRP) and mean chlorophyll *a*, µg/L.

Florida streams is related to their proximity to phosphate deposits, and an analysis of the SIN:SRP ratios in the NES tributaries indicates that all the streams in the vicinity of the phosphate deposits had ratios less than 5 (n = 6). However, about 30% of the streams not in the vicinity of these deposits also had such low ratio, indicating that factors other than proximity to phosphate deposits are responsible for the low SIN:SRP ratios in many of Florida's streams.

In-lake mechanisms, particularly denitrification, also may result in a decrease in the ratio of nitrogen to phosphorus with increasing eutrophication. In many lakes, loss of nitrogen via denitrification can represent a large fraction of the input of nitrogen. For example, Messer et al. (1979) concluded that up to 26% of the nitrogen input into Lake Okeechobee is removed by denitrification; losses of over 10% have been reported in the literature for a number of other lakes. Although no direct correlation has been demonstrated between trophic state and denitrification rates, conditions existing in eutrophic lakes encourage denitrification. These conditions include a supply of organic matter to supply energy, anoxic conditions near the bottom, and high levels of nitrate.

The Relationship Between Nutrients and Chlorophyll a Standing Crops. Several investigators (Dillon and Rigler 1974b Sakamato 1966; Jones and Bachman 1976) have demonstrated a strong log-log relationship between the mean concentration of phosphorus in the water column during turnover and the mean epilimnetic chlorophyll *a* concentration during the summer (Table IV-3). This relationship has become the basis for constructing input/output (I/O) models that can be used to predict chlorophyll concentrations in lakes using only data on phosphorus loading and hydraulic characteristics (Chapter 5). However, these relationships were developed in temperate zone lakes that are usually phosphorus limited (Miller et al. 1974). The lakes in this study differ from these temperate zone lakes in that 1) a dimictic pattern of stratification is usually not observed, and 2) most are N-limited rather than P-limited.

Thus we examined the relationship between chlorophyll *a* and total N and P for Florida lakes. Since there are no distinct periods of turnover and stratification in these lakes, annual means usually were used, although the relationship between mean spring phosphorus concentration (spring being defined as March through May) and mean summer chlorophyll *a* (summer being defined as June through September) also was evaluated.

Log-log plots of $(P)_1$ and $(N)_1$ versus (chl *a*) (annual means) are shown in Figures IV-7 and IV-8, together with the lines of the regression equations that describe the best fit (equations 4-13 and 4-16). The regression line describing the relationship between $(P)_1$ and (chl *a*) (eq. 4-13) for the study lakes has a much lower slope that do the regression lines of Dillon and Rigler (1974b)or Jones and Bachman (1976) (see Figure IV-9). A second regression line, determined using only spring total P and summer chlorophyll *a* values, has a similar but slightly lower slope (eq. 4-15) than that for the annual means. These two lines demonstrate that the amount of chlorophyll *a* associated with a given level of total P is lower in Florida lakes than for most temperate zone lakes. This is the situation that one would expect for a group of lakes that are largely N-limited. In order to test the hypothesis

Table IV-3. Regression equations of total phosphorus & total nitrogen vs. chlorophyll *a*.(1)

Phosphorus

Jones & Bachman (1976):

$$\log (\text{chl } a)_s = -1.09 + 1.46 \log (P) \qquad (4\text{-}10)$$

$r^2 = 0.98$ $n = 143$

Dillon and Rigler (1974b):

Using data from Sakamoto (1966):

$$\log (\text{chl } a)_s = -1.134 + 1.583 \log (P)_{sp} \qquad (4\text{-}11)$$

$r^2 = 0.95$ $n = 30$

Using data from North American lakes:

$$\log (\text{chl } a)_s = -1.136 + 1.449 \log (P)_{sp} \qquad (4\text{-}12)$$

$r^2 = 0.90$ $n = 46$

This study:

Entire data set, annual mean (P) vs. annual mean (chl *a*)

$$\log (\text{chl } a) = -0.41 + 0.79 \log (P) \qquad (4\text{-}13)$$

$r^2 = 0.72$ $n = 100$

P-limited lakes only:

$$\log (\text{chl } a) = -0.71 + 1.03 \log (P) \qquad (4\text{-}14)$$

$r^2 = 0.53$ $n = 33$

Entire data set, spring P vs. summer chl *a*:

$$\log (\text{chl } a)_s = -0.16 + 0.71 \log (P)_{sp} \qquad (4\text{-}15)$$

$r^2 = 0.57$ $n = 63$

Nitrogen

Entire data set:

$$\log (\text{chl } a) = 1.03 + 1.46 \log (N) \qquad (4\text{-}16)$$

$r^2 = 0.77$ $n = 100$

N-limited lakes only:

$$\log (\text{chl } a) = 1.12 + 1.53 \log (N) \qquad (4\text{-}17)$$

$r^2 = 0.77$ $n = 44$

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(1) chl *a* = mean chlorophyll a, µg/L., P = mean total phosphorus, µg/L. N = mean total nitrogen, mg/L. Subscripts: s = summer mean, sp = spring mean, none = annual mean.

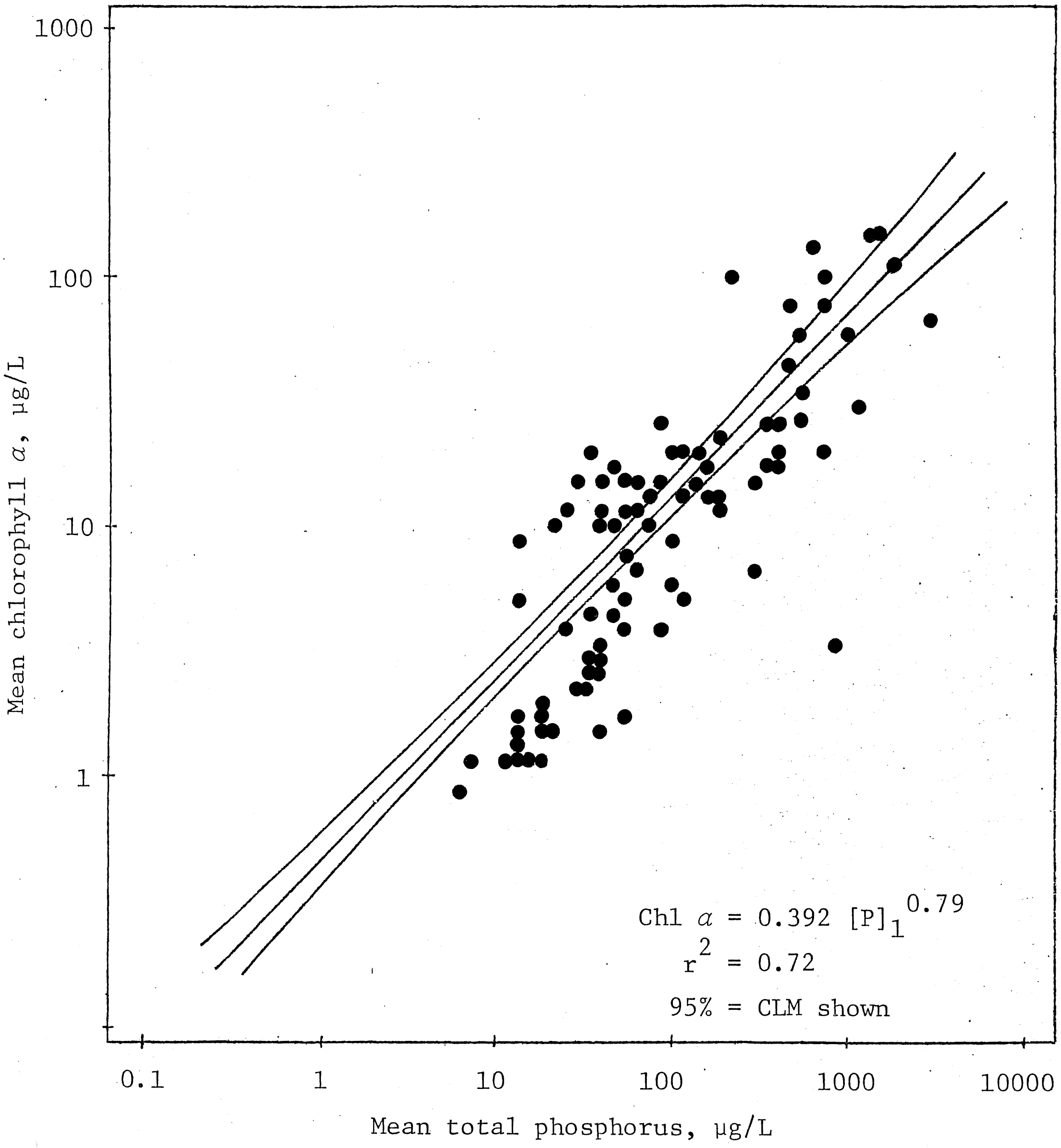


Figure IV-7. Relationship between mean total phosphorus and mean chlorophyll *a* for 101 study lakes.

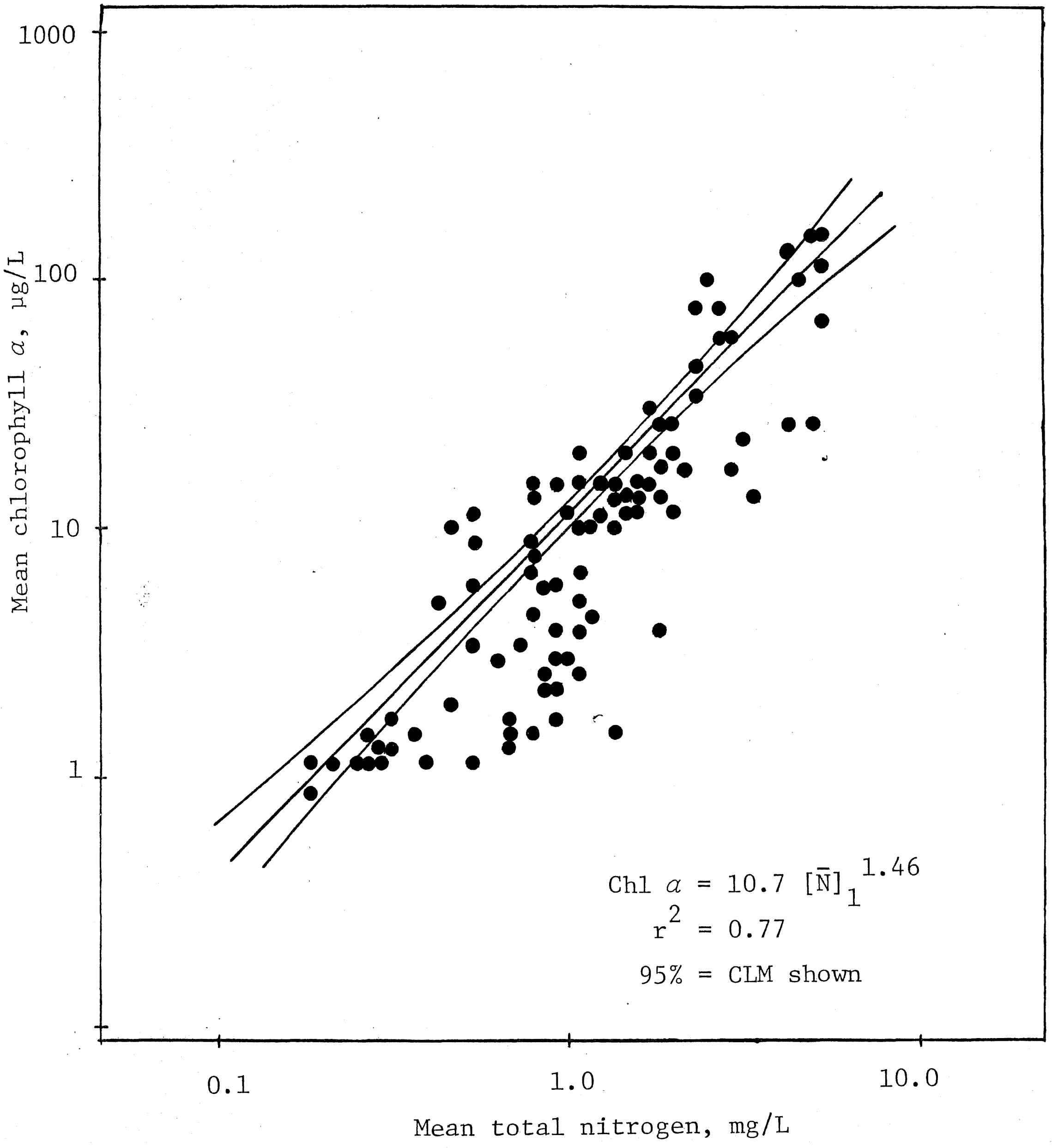


Figure IV-8. Relationship between mean total nitrogen and mean chlorophyll a in 101 study lakes.

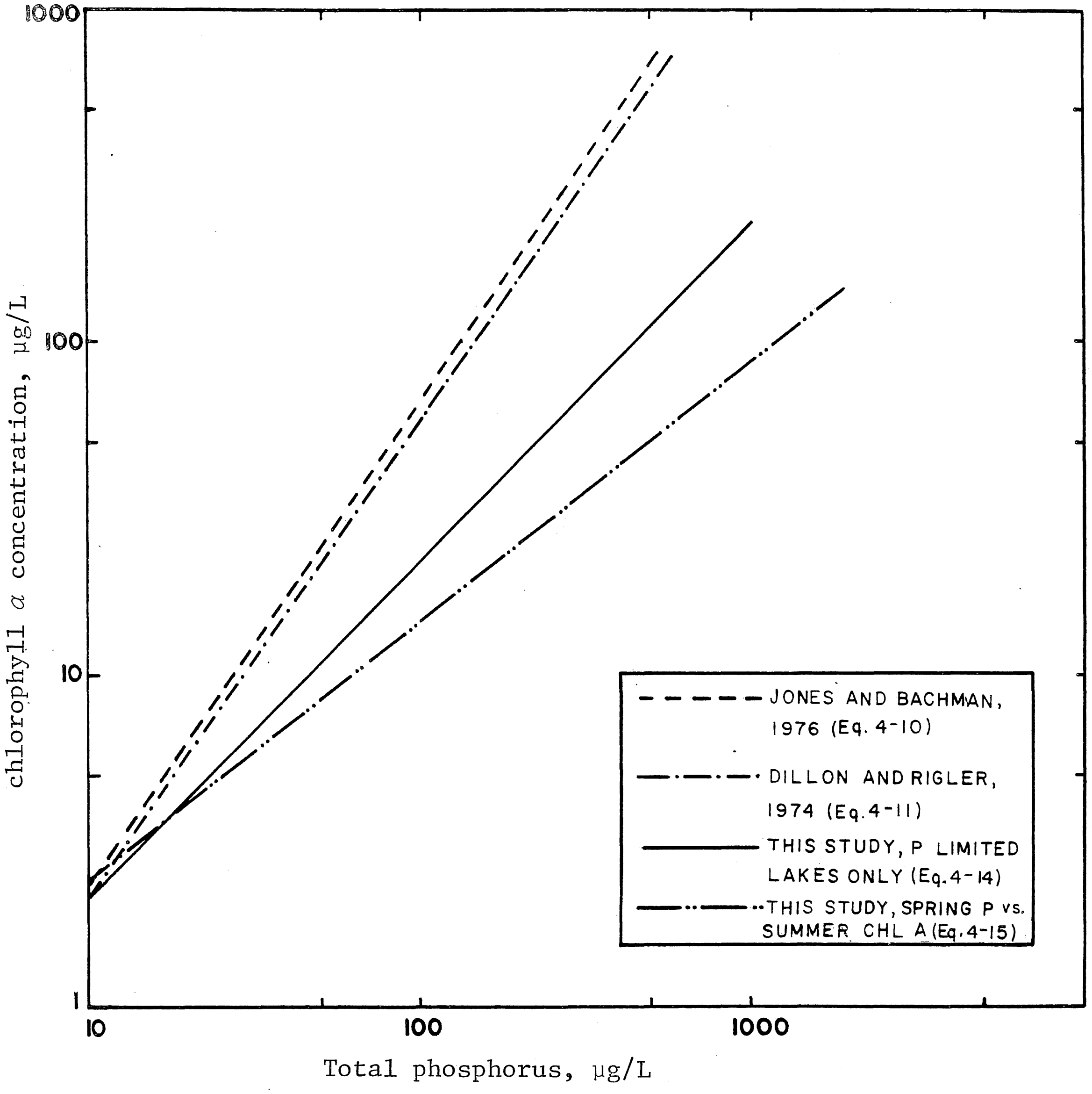


Figure IV-9. Regression lines of total phosphorus vs. chlorophyll *a* determined by several investigators.

that nitrogen limitation is actually the factor that results in the low slope for the Florida lakes, a regression of $(P)_1$ vs. (chl *a*) was determined for a group of P-limited lakes (eq. 4-14). In this analysis, phosphorus limitation was defined by a ratio of SIN:SRP of >20:1. The slope of this line is steeper than that of the regression line determined for the entire group of Florida lakes (Figure IV-9), but it still is lower than the slopes of the lines determined for temperate zone lakes.

There are several factors that could account for the lower slope for phosphorus-limited Florida lakes than for the temperate zone lakes. First, it is possible that a ratio of 20:1 for SIN:SRP is not an accurate criterion of phosphorus limitation; that is, some higher ratio would be more realistic. This is unlikely, since a critical ratio of 20:1 is quite conservative.

Second, factors other than phosphorus concentration may limit algal standing crop even in lakes that are considered phosphorus-limited on the basis of nutrient ratios. For example, nutrients other than nitrogen or phosphorus may limit algal productivity in some lakes, as may toxic substances. Miller et al. (1974) found that constituents other than N or P were limiting to algal productivity in 6 of the 49 temperate zone lakes included in their survey. Additional bioassay data are needed to determine to what extent other nutrients or toxic constituents may be limiting for algae in Florida lakes.

Third, biological interactions may limit the standing crop of algae in Florida's lakes below the level in temperate zone lakes containing the same level of phosphorus. Grazing by herbivorous grazing fish (e.q., shad) or other structural differences in the food chain may control the algal standing crop more effectively in Florida's lakes than in temperate lakes, as may interactions with macrophytes. A detailed discussion of ecological interactions that affect the algal standing crops in lakes is presented by Shapiro (1979).

Finally, the much longer growing season (essentially year-round in Florida, compared to the compressed growing season in temperate lakes) may also affect the chlorophyll *a*-total phosphorus relationship. The long period of ice cover and dormancy in temperate lakes leads to a build-up of inorganic nutrients, culminating in the "spring maximum". This in turn leads to the spring and early summer pulses of algal blooms. In contrast, inorganic nutrient levels exhibit less seasonal variations in warm temperate lakes, and algal growth is more evenly distributed throughout the year (see following section). Coupled with the biological interactions (e.g. grazing) mentioned above, this may result in a lower standing crop of algae (hence chlorophyll _a_) for a given total phosphorus level than is found in most pulsed systems.

The relationship between $(N)_1$ and (chl *a*) in the study lakes is shown by Eq. 4-16. The r^2 for this relationship (0.77) is slightly higher than the relationship between $(P)_1$ and (chl *a*) ($r^2 = 0.72$), as one would expect for a group of lakes that is largely nitrogen-limited. When only nitrogen-limited lakes were considered (according to the criterion of SIN:SRP<10:1), the regression equation was only slightly altered (eq. 4-17).

Analysis of Seasonal Trends. In temperate zone lakes, pronounced seasonal variations occur in the standing crop of algae and in the concentrations of major nutrients. Major nutrients tend to reach peak concentrations during spring and fall turnover, and the algal standing crop tends to reach maximum levels either following the turnover periods or during the summer stratification period. In Florida, where most lakes do not undergo stable thermal stratification and seasonal variations in temperature are less pronounced than in the temperate zone, it is reasonable to hypothesize that seasonal variations in algal standing crop and in the concentrations of major nutrients will be minimal. To test this hypothesis for the study lakes, normalized parameter values were computed for each lakes:

$$N_{ij} = \frac{X_{ij}}{\bar{X}_{ij}} \qquad (4\text{-}18)$$

where N_{ij} = the normalized value for variable i and lake j;
X_{ij} = the variable value during a sampling period;
$\bar{X}_{ij}$ = the annual mean value.

An overall normalized mean can be calculated:

$$\overline{N_i} = \sum_{j=1}^{n} \frac{N_{i,j}}{n} \qquad (4\text{-}19)$$

where n = the number of lakes in the study group.

Thus, if a study lake had a value for a variable during a particular sampling period equal to the annual mean, N_{ij} would be 1.0. Values greater than 1.0 indicate a positive seasonal trend while values below 1.0 indicate a negative seasonal trend.

Seasonal trends of chlorophyll *a*, total phosphorus and total nitrogen were analyzed for the two geographical subgroups of lakes included in acid lake study: the 13 Trail Ridge lakes in northern Florida and the seven Highlands Ridge lakes in southern Florida. These data are particularly well-suited for an analysis of seasonal trends for several reasons. First, these two groups of lakes are located at opposite ends of the state and therefore represent the extremes in climatic conditions in Florida. For the northern group, the difference in mean daily air temperature between January and July is 22°C; for the southern group the difference is only 16°C. Second, most of the lakes in both groups are relatively undisturbed by human activity and pulses in algal productivity are not likely to reflect man's activities (e.g., nutrient-rich runoff during agricultural fertilization). Finally, since all 20 lakes were studied by one group of investigators using standarized methods, differences between the two groups are not likely to be the result of variations in methodology.

A plot of the mean normalized values for chlorophyll *a* throughout the year (Figure IV-10), shown with 95% confidence intervals, indicates that there are no statistically significant seasonal trends in chlorophyll *a* values for

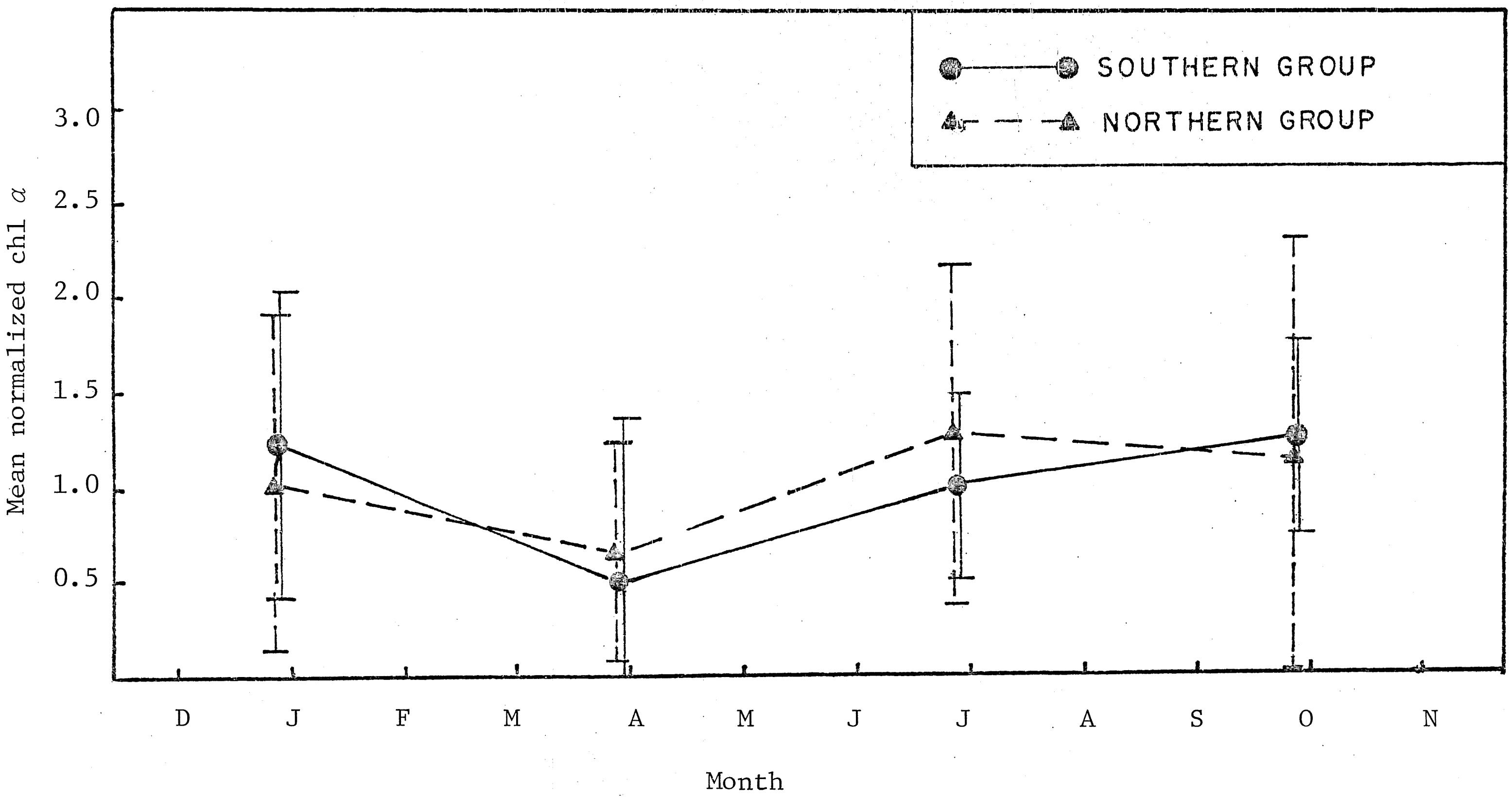


Figure IV-10. Seasonal trends in chlorophyll *a* concentration for 10 northern Florida lakes and 10 southern Florida lakes. See text for calculations of mean normalized parameter values.

either group of lakes. Similar results were obtained when total phosphorus and total nitrogen data were analyzed.

Data compiled for Lake Apopka and the other Oklawaha lakes (Brezonik et al. 1978; Tuschall et al. 1979; Pollman et al. 1980) also indicate that while seasonal trends in chlorophyll *a* do occur, the trends are not consistent among lakes within a given year or in any one lake throughout the three-year study period. A plot of chlorophyll *a* data for Lake Apopka (Figure IV-11) shows that while there are major oscillations in the chlorophyll *a* standing crop, the timing of periods of bloom and scenescence vary from year to year. For example, in 1977 peak chlorophyll *a* concentrations occurred in the fall, while in 1978 peaks occurred in May and August and in the first half of 1979 a peak occurred in April. Periods of scenescence show a similar lack of temporal regularity: minimum chlorophyll *a* values occurred in March and May of 1977 and in February and June of 1978.

This analysis suggests that there are no (or only small) regular seasonal trends in nutrient and chlorophyll *a* levels in Florida lakes. This conclusion serves as a basis for using annual means for nutrient and chlorophyll *a* concentrations, rather than seasonal means (e.g., mean spring total phosphorus; mean summer chlorophyll *a*) in the refinement of nutrient loading models for Florida lakes (Chapter V).

DEVELOPMENT OF A TROPHIC STATE INDEX

A trophic state index (TSI) that allows the ranking of lakes along a linear gradient is useful for several reasons: 1) a linear ranking system facilitates comparisons of trophic state within a group of lakes, 2) use of a TSI obviates the need to place a lake into a discrete trophic class (oligotrophic, mesotrophic, eutrophic), 3) a TSI can be used to quantify historical changes in trophic state and thereby assess the impacts of cultural perturbations, and 4) a numerical index can be comprehended by the public. Trophic state indices have been developed based on both single measures of trophic state (univariate indices) and on a composite of several trophic state indicators (multivariate indices). Trophic state variables that have been used in indices include dissolved oxygen, total and inorganic phosphorus and nitrogen, Secchi disk transparency, chlorophyll *a*, primary production, and the relative abundance of major ions). Concepts and applications of trophic state indices and the relative merits of univariate versus multivariate indices have been reviewed and discussed by Shapiro (1976), Brezonik (1976) and Carlson (1977).

Trophic state indices that might be applied to Florida lakes include those of Shannon and Brezonik (1972), the National Eutrophication Survey (1975), and Carlson (1977). The National Eutrophication Survey developed a "water quality index" (WQI) based on six parameters related to trophic state. The WQI has two major shortcomings in ranking Florida's lakes: 1) its use of a dissolved oxygen parameter and 2) its use of two phosphorus parameters (median total phosphorus and median dissolved phosphorus). The dissolved oxygen parameter (a constant value minus the dissolved oxygen concentration in the bottom waters) is inappropriate because most of Florida's

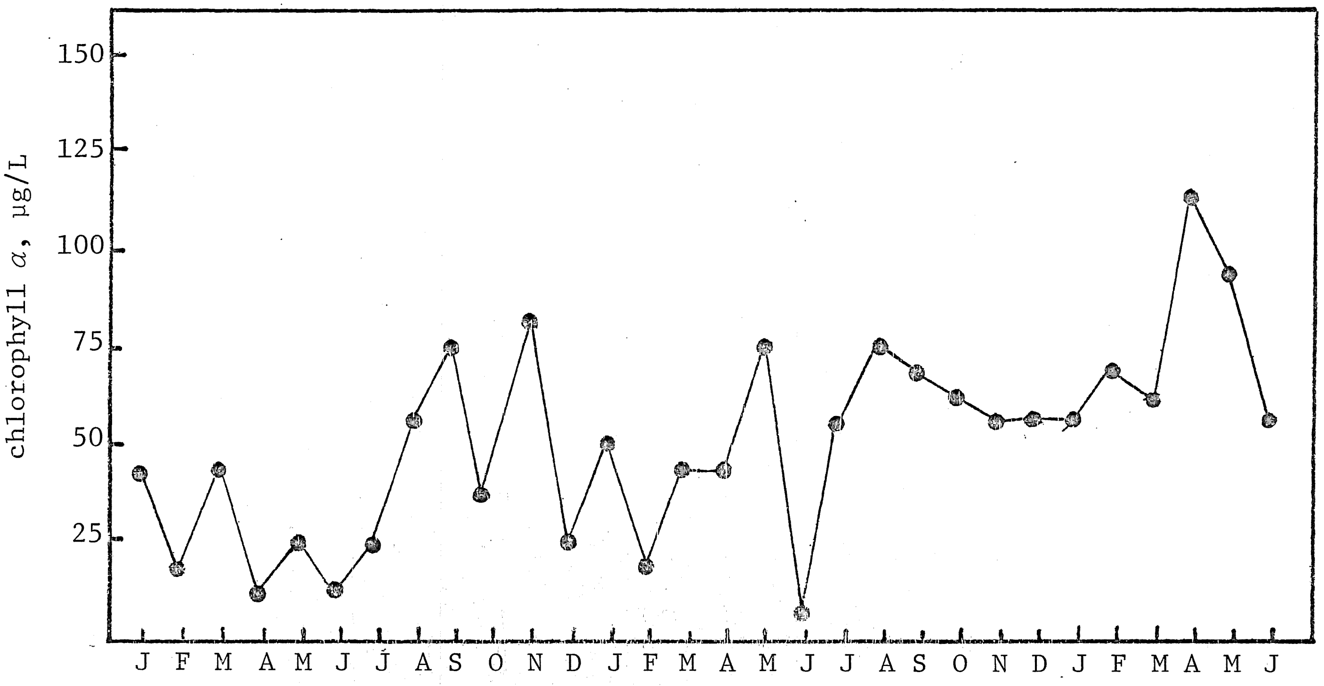


Figure IV-11. Chlorophyll *a* concentrations in Lake Apopka, January, 1977 to June, 1979.

lakes are not stratified. Rankings made using the D.O. parameter for the Florida NES lakes were widely scattered and not well-correlated with other trophic state indicators. The use of two phosphorus parameters in the WQI overemphasizes the importance of phosphorus as a trophic state indicator, particularly for Florida's many nitrogen-limited lakes.

The index developed by Brezonik and Shannon (1971) using data from 55 Florida lakes could reasonably be applied to other lakes in the present data set except that data on primary production and cation composition required for the computation of this TSI were not collected in the Florida NES and the acid lake survey. Furthermore, values of chlorophyll *a*, total organic nitrogen and specific conductivity for several lakes lie outside the range of values in the original data base used to construct the TSI.

Carlson (1977) developed three separate univariate indices of trophic state, based respectively on Secchi disk transparency, total phosphorus and chlorophyll *a*. Carlson's index was initially based on Secchi disk transparency, with values scaled so that the zero point corresponded to a Secchi disk value greater than any value yet reported (64 m):

$$\mathrm{TSI(SD)} = 10(6 - \ln(\mathrm{SD})/\ln 2), \qquad (4\text{-}20)$$

where SD = Secchi disk transparency (m).

Trophic state indices for phosphorus and chlorophyll *a* were developed using the relationships between these parameters and Secchi disk transparency. Thus, for a group of 147 lakes the relationship between Secchi disk transparency and chlorophyll *a* concentration was best expressed (Carlson 1977) using the equation:

$$\ln \mathrm{SD} = 2.04 - 0.68 \ln (\mathrm{chl}\ a);\ r^2 = 0.86. \qquad (4\text{-}21)$$

A TSI based on chlorophyll *a* was computed by combining equations 4-20 and 4-21:

$$\mathrm{TSI(CHA)} = 10(6 -(2.04 - 0.68 \ln(\mathrm{chl}\ a)/\ln 2)). \qquad (4\text{-}22)$$

A TSI based on phosphorus was based on an observed inverse relationship between total phosphorus and Secchi disk transparency:

$$\mathrm{SD} = 64.9/(\mathrm{P})_1. \qquad (4\text{-}23)$$

Combining eqs. 4-21 and 4-23 results in a TSI based on phosphorus:

$$\mathrm{TSI(TP)} = 10(6 - \ln(48/(\mathrm{P})_1)/\ln 2). \qquad (4\text{-}24)$$

The indices developed by Carlson have several advantages, including small data requirements, objectivity, and reliance upon commonly measured and understood indicators of trophic state. However, linear regressions of Carlson's TSI(TP) against both TSI(SD) and TSI(CHA) showed that the TSI(TP) values were significantly different (at the 95% confidence level) from the other two TSI values for the Florida NES lakes. This is not surprising, considering the predominance of nitrogen-limited lakes in the NES data set. Consequently, we developed a nitrogen-based TSI using a computation method analogous to that used by Carlson in developing the phosphorus-based TSI (Kratzer and Brezonik 1981). For the NES lakes, the relationship between chlorophyll *a* and total nitrogen is expressed by the equation:

$$\ln(\text{chl } a) = 2.44 + 1.6 \ln(N)_1; \; r^2 = 0.89 \; (n = 39). \qquad (4\text{-}25)$$

Combining equations 4-21 and 4-25 yields:

$$SD \simeq \frac{1.46}{(N)_1} \qquad (4\text{-}26)$$

A TSI(TN) can be calculated by substituting eq. 4-26 into eq. 4-20:

$$TSI(TN) = 10(6 - \ln(1.46/(N)_1)/\ln 2). \qquad (4\text{-}27)$$

Ideally, when both TSI(TP) and TSI(TN) are computed, the smaller of the two TSI's should represent the limiting nutrient for any given lake. This hypothesis is generally supported by the Florida NES data. The five lakes (Yale, Kissimmee, Marion, Reedy and Apopka) with TSI(TP) values considerably lower than TSI(TN) all were phosphorus-limited according to the NES algal bioassay results.

For this study the lesser of TSI(TP) and TSI(TN) was averaged with the corresponding TSI(SD) and TSI(CHA) to compute a TSI(AVE). The values of the four univariate TSI's plus TSI(AVE) are listed for the 101 study lakes in Table IV-4. The TSI(AVE) values correspond well with assessments made by the NES limnologists (Lakes 1 to 40) and by Brezonik and Shannon (1971) (lakes 41 to 92). In addition, the TSI(AVE) values for the NES lakes agree reasonably well with the values of the EPA's Water Quality Index ($r^2 = 0.64$). The use of a TSI(AVE) has the advantage over the use individual TSI's in that the use of a single index of trophic status can be more easily used for comparative and management purposes. Furthermore, the compositing of physical, biological and chemical components of trophic state into one index reflects the multidimensional nature of the eutrophication phenomenon, since it is generally agreed that no single trophic indicator adequately measures the underlying concept. Combining the major physical, chemical and biological indicators of trophic state into a single index smooths out the variability associated with individual indicators and provides a reasonable composite measure of trophic conditions in a lake.

Table IV-4. Trophic Status Index for study lakes.

Lake	TSI(CHL)	TSI(TP)	TSI(TN)	TSI(SD)	TSI(AVE)	Assessment*
101	30	31	31	30	30	-
84	35	44	33	34	34	U
89	34	40	30	39	34	U
93	35	34	35	35	35	-
78	34	41	36	42	38	U
77	36	42	41	38	38	U
91	38	41	37	40	38	U
79	37	41	38	40	38	U
82	35	43	37	43	38	U
86	39	45	40	37	39	U
92	36	46	42	43	40	U
81	34	45	46	43	41	U
80	41	41	53	41	41	U
87	43	46	43	42	43	U
90	40	49	49	42	43	M
96	40	42	35	55	43	-
53	39	41	49	54	44	U
1	42	47	49	47	46	O-M
100	50	41	42	47	46	-
47	40	50	59	48	46	O
99	42	63	38	58	46	-
65	41	56	51	51	48	M
41	47	56	48	49	48	O
98	52	43	46	50	49	-
5	49	56	51	47	49	M
62	48	102	46	54	49	H
76	49	50	55	50	50	O
61	45	57	56	51	50	M
95	54	48	43	55	51	-
43	49	54	51	52	51	O
94	55	50	45	52	51	-
42	45	56	53	55	51	O
83	45	52	53	56	51	O
44	48	58	50	56	51	O
2	47	57	55	54	52	M
3	52	61	52	53	52	M
52	51	71	46	60	52	O
88	44	56	53	61	53	O
51	51	59	54	54	53	O
66	60	74	52	50	54	M
4	51	59	52	59	54	M
50	54	66	55	55	55	M
85	47	56	53	64	55	U
21	53	70	51	60	55	E
70	50	61	56	60	55	U
49	50	74	56	61	56	M
73	52	87	56	60	56	M
6	49	59	57	62	56	M-E
67	63	86	51	54	56	M
46	49	62	53	66	56	M

Table IV-4. continued...

Lake	TSICHL)	TSI(TP)	TSI(TN)	TSI(SD)	TSI(AVE)	Assessment*
97	52	65	51	66	56	-
7	62	52	59	54	56	E
10	55	60	55	60	57	E
56	54	57	57	59	57	M
69	56	61	55	60	57	M
9	63	61	54	57	58	E
74	57	57	61	60	58	M
55	56	64	58	61	59	M
16	49	68	63	64	59	E
15	53	66	56	66	60	E
24	63	74	58	58	60	E
11	63	63	56	60	60	E
8	62	56	62	62	60	E
54	59	63	60	62	60	M
45	60	78	60	61	60	M
19	61	79	59	64	61	E
12	61	66	61	62	61	E
18	66	55	64	64	62	E
17	63	68	61	62	62	E
13	57	65	64	65	62	E
14	64	60	66	63	62	E
29	57	80	64	67	63	E
25	66	99	61	62	63	H
64	65	90	60	63	63	E
59	60	79	64	65	63	E
58	66	74	56	67	63	M
72	65	78	64	61	63	E
27	65	76	62	64	64	E
32	70	107	62	62	64	E
28	64	88	64	65	64	E
48	69	68	63	64	65	E
75	61	79	72	64	66	H
57	66	71	63	68	66	E
63	68	94	64	66	66	H
31	72	93	67	62	67	E
68	64	92	70	69	68	H
71	70	95	67	67	68	E
23	67	81	71	67	68	E
36	73	105	70	64	69	H
35	74	100	69	66	70	H
26	76	83	68	66	70	E
60	74	93	66	72	71	H
20	69	90	76	74	73	H
34	73	94	69	77	73	H
30	76	98	77	73	75	E
22	68	89	78	81	76	H
33	83	97	76	71	77	H
38	79	113	79	74	78	H
40	86	109	79	72	79	E
39	85	109	78	77	80	H
37	74	118	79	89	81	E

* U = ultraoligotrophic, O = oligotrophic, M = mesotrophic, E = eutrophic, H = hyperutrophic. Assessment by NES (lakes 1 to 40) or Shannon and Brezonik, 1972 (lakes 41-92).

CHAPTER V. APPLICATION OF NUTRIENT LOADING MODELS TO THE FLORIDA NES LAKES

INTRODUCTION

In the past decade considerable efforts have been made to quantify the relationships between nutrient loading and lake trophic status using simple input-output (I/O) models. Nutrient loading models have been widely used to predict the effects of changes in nutrient loading on lake trophic status (Vollenweider 1969, 1975, 1976; Patalas and Salki 1973; Dillon and Rigler 1975; and many others) and to predict the trophic status of new reservoirs (e.g. Bradford and Maiero 1978; Baker et al. 1978; Huber and Brezonik 1980). The utility of these models is greatly enhanced by their modest data requirements and computational simplicity.

This chapter reviews the developments made in mass balance nutrient models for lakes over the past decade and the resulting advances in our ability to predict trophic conditions in lakes from information on phosphorus and hydraulic loadings and basic lake morphometry. Nearly all these predictive models have been developed using data on temperate lakes. This chapter describes the application of these models to warm-temperate and subtropical lakes in Florida and evaluates their usefulness in predicting trophic conditions in Florida lakes.

HISTORICAL DEVELOPMENT

Phosphorus Input/Output Models

The relationship between nutrient concentrations and algal productivity in lakes has long been recognized (see review by Vollenweider 1968). Since phosphorus has been identified as the most common limiting nutrient in temperate lakes, the development of nutrient loading models has focused entirely on phosphorus, although Vollenweider (1969) noted that the principles involved could be applied to other nutrients.

In the development of his phosphorus loading model, Vollenweider (1969) expressed the change in the mass of phosphorus in a simplified form:

$$dP/dt = J - L_{out} - S \qquad (5\text{-}1)$$

where dP/dt = rate of change of the mass of P in a lake.

J = flux of P into the lake,

L_{out} = flux of P from the lake via its outflow, and

S = the rate of loss of P via sedimentation and other mechanisms other than loss through the outlet.

In formulating his model, Vollenweider made several simplifying assumptions:

(1) A lake behaves like a continuously stirred tank reactor. That is, any substance entering a lake becomes completely mixed as soon as it enters.

(2) The rate of sedimentation is proportional to the amount of phosphorus in the lake, i.e., $S = \sigma_p P$, where P = the mass of phosphorus in the lake and σ_p is a first order sedimentation coefficient with units of yr^{-1}.

(3) The concentration of phosphorus in the outflow is equal to the concentration of phosphorus in the lake. Thus, $L_{out} = \rho_w (P)_1$, where ρ_w is the hydraulic flushing coefficient (Q/V) in yr^{-1} and $(P)_1$ = mean lake phosphorus concentration.

(4) There is no seasonal fluctuation in loading.

Although none of these assumptions is entirely valid, they allow the development of a simple expression to compute $(P)_1$:

$$dP/dt = J - \sigma_p P - \rho_w P, \tag{5-2}$$

If it is further assumed that the system is at steady state, i.e., $dP/dt = 0$:

$$J = \rho_w P + \sigma_p P; \text{ or } P = \frac{J}{\sigma_p + \rho_w}. \tag{5-3}$$

Dividing through by lake volume (V) yields

$$\frac{P}{V} = \frac{J/V}{\rho_w + \sigma_p}, \tag{5-4}$$

and since $P/V = [P]_1$ and $J/V = L_v$ (the volumetric loading rate),

$$[P]_1 = \frac{L_v}{\sigma_p + \rho_w}, \tag{5-5a}$$

or

$$L_v = \sigma_p [P]_1 + \rho_w [P]_1. \tag{5-5b}$$

Phosphorus loading usually is expressed on an areal basis (in g/m^2-yr), as $L_p = L_v/\bar{z}$, where $\bar{z}$ = mean depth:

$$[P]_1 = \frac{L_p}{z\ (\rho_w + \sigma_p)} = \frac{L_p}{z\ \sigma_p + q_s} \tag{5-6}$$

where q_s = hydraulic loading (m/yr) = $\bar{z}\,\rho_w$. Since L_a, $\bar{z}$ and ρ_w can readily be measured, the major difficulty in using this model is the estimation of the sedimentation coefficient, σ_p. Vollenweider (1975) found that for a group of 25 temperate zone lakes, $\sigma_p = 10/\bar{z}$ Substituting this expression into eq. (5-6) yields

$$[P]_1 = \frac{L_p}{10 + q_s}. \tag{5-7}$$

This relationship is tantamount to stating that the apparent deposition or settling velocity for total phosphorus in lakes is constant. If $\sigma_p = 10/\bar{z}$, then $\sigma_p \cdot \bar{z}$ = constant; $\sigma_p \cdot \bar{z}$ has units of m/yr, which dimensionally is a velocity. This term can be interpreted as the settling velocity for total P and is given the symbol v_p. It should be noted that both v_p and σ_p are not subject to strict physical interpretation and measurement, since there is more than a single mechanism (and a single form of phosphorus) involved in deposition to sediments. Moreover, sediment deposition is not the sole internal sink for phosphorus, although on a long term basis it is by far the most important. Phosphorus also can be lost from the water column via uptake by macrophytes or incorporation in fish biomass; neither of these reservoirs is measured in typical phosphorus budgets. Thus the basic mass balance model for phosphorus is a simplification of reality, and the sedimentation term is a composite of all internal sink processes, including deposition (settling) of detritus, adsorption of orthophosphate by sediments, and uptake by macrophytes followed by direct incorporation of dead macrophyte tissue into the sediments. Consequently, it is not possible to measure σ_p or v_p directly, although sedimentation traps may provide good approximations under certain limited conditions.

Jones and Bachman (1976) found that for a group of 16 Iowa lakes, the best fit for eq. (5-6) was obtained using a constant value of 0.65 for σ_p. Thus for their data sedimentation rate was independent of depth, and eq. (5-6) becomes

$$[P]_1 = \frac{0.84\, L_p}{q_s + 0.65\bar{z}} . \tag{5-8}$$

More recently, Vollenweider (1976) proposed that $\sigma_p \cong 1/\sqrt{\tau_w}$, where τ_w is the hydraulic retention time ($\tau_w = \rho_w^{-1}$; hence $\tau_w = \bar{z}/q_s$). Eq. 5-6 then becomes:

$$[P]_1 = \frac{L_p}{q_s(1 + \tau_w^{-\frac{1}{2}})} . \tag{5-9}$$

It is to be noted that in developing eq. 5-9, Vollenweider has gone beyond the dimensionally- and theoretically-correct (albeit perhaps simplistic) mass balance model and has interjected an element of empiricism into the phosphorus predictive equation.

An alternative approach to that requiring a determination of σ_p was proposed by Dillon and Rigler (1974). These workers proposed that an easily measured retention coefficient, R_p, can be used to replace σ_p in Vollenweider's model:

$$R_p = 1 - \frac{Q_{out}[P]_{out}}{Q_{in}\ [P]_{in}}, \tag{5-10}$$

which is simply the fraction of the input phosphorus that is retained within a lake. If the lake is at steady state, it follows that the phosphorus retained must be added to the sediment. The sedimentation coefficient can be expressed in terms of R_p. Since $L_v = \sigma_p[P]_1 + \rho_w[P]_1$ (eq. 5-5b),

$$\frac{L_v}{[P]_1} = \sigma_p + \rho_w \tag{5-11}$$

From eq. 5-5b and the nature of steady state systems, it also is clear that $R_p L_v = \sigma_p [P]_1$. Substituting this relationship into eq. 5-11, we obtain:

$$\sigma_p \quad \frac{R_p \rho_w}{1 - R_p}, \tag{5-12}$$

and $$R_p = \frac{\sigma_p}{\sigma_p + \rho_w}. \tag{5-13}$$

Substituting eq. 5-12 into eq. 5-6 and noting that $q_s = \bar{z}\ \rho_w$, we obtain

$$[P]_1 = \frac{L_p\ (1 - R_p)}{\bar{z}\ \rho_w} = \frac{L_p(1 - R_p)}{q_s}. \tag{5-14}$$

Dillon and Rigler (1974a) developed loading criteria plots based on eq. 5-14. Plots of $L_p(1 - R_p)/\rho_w$ vs. $\bar{z}$ have a slope of equivalent to the steady state concentration of total phosphorus ($[P]_1$) in a lake and can be used to predict trophic conditions (see following section).

Larsen and Mercier (1975) presented an alternative approach that is derived directly from Vollenweider's model but emphasizes the importance of the influent concentration of phosphorus, $[P]_i$, rather than the total loading. Since $L_p = J_i/A$, where J_i is the total input of phosphorus (g/yr), and since $q_s = Q_i/A$, it is clear that $L_p/q_s = J_i/Q_i = [P]_i$. Substitution of this relationship into the Dillon-Rigler relationship (eq. 5-14) results in:

$$[P]_1 = [P]_i\ (1 - R_p). \tag{5-15}$$

Finally, Chapra (1975) introduced the concept of an "apparent settling velocity", v_p, where $v_p = \sigma_p\ \bar{z}$, as discussed earlier. Substituting this term into eq. 5-13 results in

$$R_p = \frac{v_p}{v_p + q_s}. \tag{5-16}$$

Combining equations 5-14 and 5-15 yields

$$[P]_1 = \frac{L_p}{v_p + q_s}. \tag{5-17}$$

The apparent settling velocity can be determined by rearranging eq. 5-16 and solving. If v_p is assumed to be constant, R_p can then be calculated directly knowing only q_s. This relationship is conceptually sound in that when $q_s = 0$, R_p must be 0, regardless of the magnitude of v_p. Chapra (1975) found a value of 16 m/yr for v_p in a group of Ontario lakes, whereas for a somewhat larger group of lakes Kirchner and Dillon (1975) found that v_p was 13.3 m/yr (using eq. 5-16 and measured values of R_p and q_s). As noted earlier, Vollenweider's (1969, 1975) formulation, $\sigma_p = 10/\bar{z}$, corresponds to a v_p of 10 m/yr.

In addition to the fundamental I/O approach to modeling lake phosphorus dynamics, several empirical models have been developed. Two such models recently were proposed as statistical improvements on the previous predictive models for total phosphorus concentration (Reckhow 1977, Walker 1977). The Reckhow (1977) model was developed by using a nonlinear regression to account for those variables that produce a nonlinear response in the total phosphorus concentration. The resulting equation, based on 33 north temperate lakes with $q_s < 50$ m/yr, was:

$$(P)_1 = \frac{L_p}{\frac{18\,\bar{z}}{10 + \bar{z}} + 1.05\, q_s \exp(0.012\, q_s)} \qquad (5\text{-}18)$$

Walker (1977) developed a model from a data base of 105 north temperate lakes and came up with the predictive equation:

$$(P)_1 = \frac{L_p}{q_s\,(1 + 0.824\,\tau_w^{\,0.454})} \qquad (5\text{-}19)$$

Imboden (1974) and Snodgrass and O'Melia (1975) have developed more complicated models of phosphorus that divide lakes vertically into two compartments (the epilimnion or upper, mixed layer, and the hypolimnion, or lower stagnant layer). Their models require the solution of four coupled differential equations for the summer (stratification) period (ortho-P and particulate-P for the epilimnion and hypolimnion). Only two differential equations need to be solved for the winter (circulation) period (ortho-P and particulate-P for the entire lake). Data requirements for these models are more complicated and include: (1) phosphorus exchange coefficients between the epilimnion and the hypolimnion and between the sediment and overlying water; (2) rate coefficients for photosynthesis and mineralization; (3) settling coefficients for particulate phosphorus; (4) loading rates for both forms of phosphorus; (5) water flow rates; and (6) depths of the epilimnion, thermocline, and hypolimnion. These models produced one surprising result: deep lakes were predicted to have higher phosphorus concentrations in their euphotic zones than shallow lakes, a result that is contradictory to the observations of several investigators. Snodgrass and O'Melia (1975) proposed to resolve this discrepancy by assigning a depth dependence to the exchange coefficient across the thermocline and to the effective settling velocity of particulate phosphorus. Since most Florida lakes are thermally unstratified at all times, the complications introduced by these two-compartment models are unnecessary.

Prediction of R_p

An advantage of the Dillon-Rigler and Larsen-Mercier models is that R_p can be determined experimentally (eq. 5-10) from phosphorus budget measurements. However, in some important applications, R_p is not known. For example, in predictive studies on proposed reservoirs, R_p obviously cannot be measured directly. Also, it is often desirable to predict R_p for an existing lake for which a complete phosphorus budget is not available or for which proposed management activities may change hydraulic and or nutrient loading characteristics. These needs have led to several empirical attempts to predict R_p from other easily measured limnological variables, particularly hydraulic parameters such as q_s and ρ_w. Kirchner and Dillon (1975) derived a double exponential equation to predict R_p from q_s using a data base of 15 southern Ontario lakes:

$$R_p = 0.426 \exp(-0.271\, q_s) + 0.574 \exp(-0.00949\, q_s). \qquad (5\text{-}20)$$

The correlation coefficient for this relationship based on 15 lakes in southern Ontario was 0.94; furthermore, the relationship is reasonable in that it gives an R_p of 1.0 when $q_s = 0$ (i.e. when there is no flow from the lake).

Larsen and Mercier (1975) evaluated a variety of empirical formulations to estimate R_p using a data base of 20 temperate lakes, for example:

$$R_p = 0.854 - 0.142\, q_s, \qquad (5\text{-}21)$$

and $$R_p = 0.482 - 0.112 \ln \rho_w. \qquad (5\text{-}22)$$

Equation 5-21 yielded a correlation coefficient of 0.92 using all 20 lakes in the data set. When the two shallowest lakes were excluded, eq. 5-22 gave nearly as good a fit. Larsen and Mercier noted that these equations do not provide theoretically correct predictions for lakes with extreme values of q_s or ρ_w, in which cases unreasonable predictions of R_p (>1 or <0) could occur. A relationship that was more theoretically sound was obtained by Larsen and Mercier from the finding that σ_p (as estimated by mass balance models) was related to the flushing coefficient (ρ_w) for the 20 lakes:

$$\ln \sigma_p = 0.472 \ln \rho_w - 0.273, \quad (r = 0.84). \qquad (5\text{-}23)$$

Substituting this term into eq. 5-13 and simplifying resulted in the approximate relationship:

$$R_p \cong \frac{1}{1 + \rho_w^{\frac{1}{2}}} \qquad (5\text{-}24)$$

Prediction of Chlorophyll *a* Concentration

A major advance in the development of nutrient loading models came with the recognition of a relationship between the concentration of phosphorus during spring turnover $(P)_{sp}$ and the mean summer concentration of chlorophyll *a* (Dillon and Rigler 1974a; Sakamoto 1966; Jones and Bachman 1976; see Chapter 4). These correlations have led to the development of models that can predict the chlorophyll *a* concentration directly from data on nutrient loading, morphological features and hydraulic parameters (Vollenweider, 1976; Jones and Bachman 1976; Chapra and Tarapchak 1976). For example, Vollenweider (1976) regressed the right side term of eq. 5-9 against mean summer chlorophyll *a* $(\text{chl } a)_s$ to obtain the relationship:

$$[\text{chl } a]_s = 0.367 \left[\frac{L_p/q_s}{1+\sqrt{\tau_w}}\right]^{0.91}. \qquad (5\text{-}25)$$

In a similar manner, Chapra and Tarapchak (1976) combined Dillon and Rigler's equation (5-14) and Chapra's equation (5-16), setting $v_p = 12.4$, and regressing the resulting term against $[\text{chl } a]_s$ to obtain

$$[\text{chl } a] = 1866 \left[\frac{L_p}{q_s + 12.4}\right]^{1.449} \qquad (5\text{-}26)$$

Finally, Hand (1975) used an empirical approach to predict $[\text{chl } a]$ for Florida lakes using a shape factor (5) and the outflow phosphorus concentration (P_{out}/Q_{out}):

$$[\text{chl } a] = 114\ (N_{out}/3 + P_{out})/Q_{out}\ \sqrt{5\bar{z}}. \qquad (5\text{-}27)$$

Nutrient Loading Criteria: Graphical Approaches

Parallelling the development of models to predict $[P]_1$, and later $(\text{chl } a)_s$ is the development of critical loading plots that depict critical loadings (i.e., permissible loadings that maintain oligotrophic conditions and excessive rates above which eutrophic conditions occur). These criteria are plotted as functions of lake morphometry and/or hydraulic conditions. The first such plot was developed by Vollenweider (1968) and was empirical in nature. This graph (Figure V-1) depicts trophic status as a function of areal phosphorus loading (L_p) and mean depth $(\bar{z})$. Brezonik and Shannon (1971) developed a similar graph showing the relationships between nitrogen and phosphorus loading and a trophic state index for Florida lakes. The permissible and excessive loading rates they developed were higher than Vollenweider's corresponding loading criteria, implying that Florida lakes are capable of assimilating more nutrients than are the temperate lakes used by Vollenweider.

Although Vollenweider's 1968 loading plots were widely used following their introduction, Vollenweider and others realized that the failure of the plots to account for hydraulic characteristics limited their usefulness. Thus Vollenweider (1975) introduced a second critical loading plot based on his phosphorus loading model (eq. 5-6). Critical phosphorus levels were considered to be 10 and 20 µg/L, respectively, as lower limits for mesotrophic and eutrophic conditions. The resulting plot of L_p vs. q_s (Figure V-2) has three distinct segments:

1) $q_s < 4$ m/yr. $L_{crit} \cong$ constant (dependent only on σ_p).

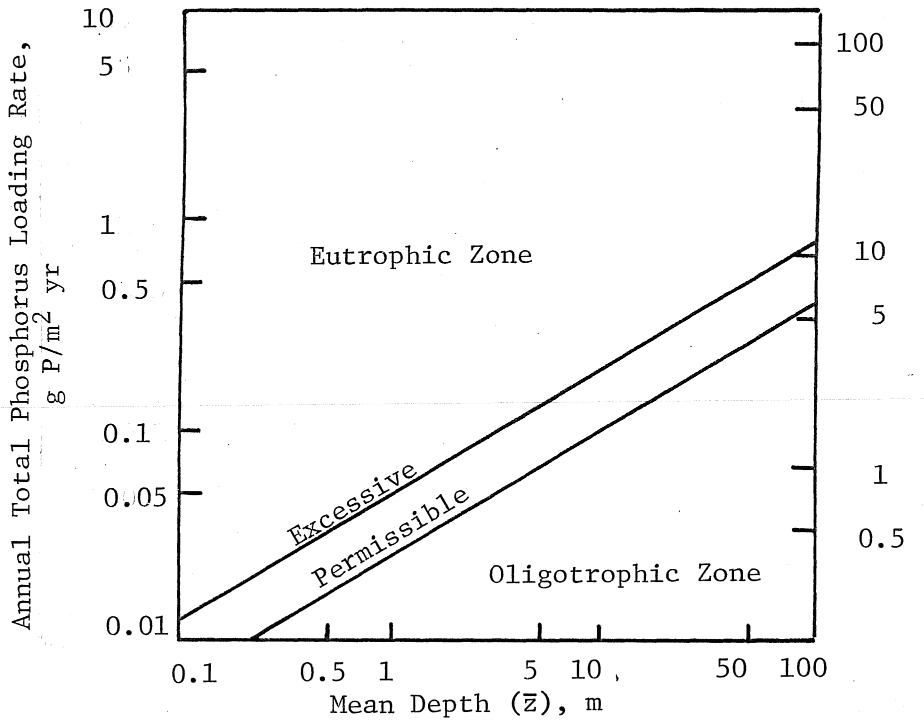


Figure V-1. Vollenweider's phosphorus and nitrogen loading criteria (1968), L versus $\bar{z}$.

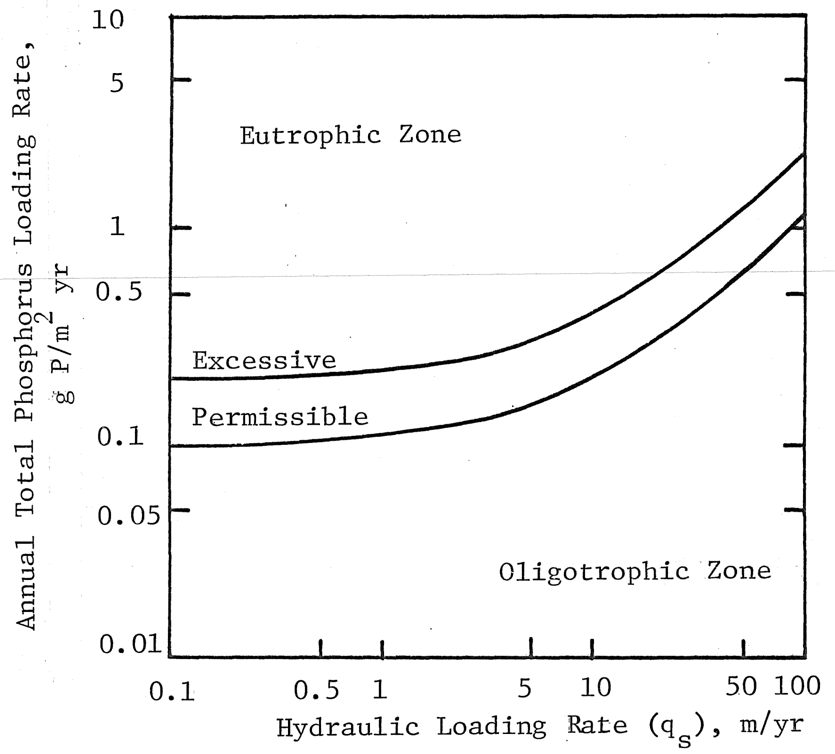


Figure V-2. Vollenweider's phosphorus loading criteria (1975), L_p versus q_s.

2) 40 m/yr > q_s > 4 m/yr. L_{crit} proportional to q_s (and σ_p).

3) q_s > 40 m/yr. L_{crit} proportional to q_s and σ_p, q_s being dominant.

This model is an improvement over Vollenweider's early depth-loading plots in that the critical loading is recognized to be a function of the areal water.

Dillon (1975) introduced a critical loading plot based on the model developed by Dillon and Rigler (eq. 5-14), again using 10 and 20 μg/L for critical levels of $[P]_1$. In this case, $L_p\ (1 - R_p)\ \tau_w$ is plotted vs. $\bar{z}$; the slopes of the critical lines are 10 and 20 μg/L, respectively (Figure V-3).

Finally, Larsen and Mercier (1975) used eq. 5-15 to construct a plot relating $(P)_i$ to R_p (Figure V-4). This plot delineates zones of oligotrophic, mesotrophic, and eutrophic lakes on the basis of R_p and average influence concentrations of total phosphorus.

APPLICATION OF NUTRIENT LOADING MODELS TO FLORIDA LAKES

The nutrient loading models described above were developed using data from temperate lakes, and their validity has not been evaluated for subtropical lakes. As demonstrated in Chapter IV, Florida lakes differ considerably from temperate zone lakes in their limnological characteristics. Unlike temperate zone lakes, Florida lakes do not undergo a dimictic pattern of stratification, as do most temperate zone lakes. Many are quite shallow and harbor large beds of macrophytes. Overall, they are considerably more colored than most temperate zone lakes. Finally, nitrogen tends to be the limiting nutrient for many Florida lakes, raising the question of whether loading models based on phosphorus are applicable.

Thus, the objective of this phase of the study was to evaluate the use of I/O models using the Florida NES data base. Models to predict $(P)_1$, $(N)_1$, (chl *a*) and phosphorus-and nitrogen-retained coefficients (R_p and R_n) were analyzed using regression analyses. The GLM procedure of SAS was used to select the best models and to determine the 95% CLI for these models. Since seasonal variability is less pronounced in Florida lakes compared to temperate zone lakes (Chapter IV), annual mean values of $(P)_1$, $(N)_1$ and (chl *a*) were used in these analyses. Available phosphorus loading plots (Vollenweider 1968, 1975; Dillon 1975; Larsen and Mercier 1975) were examined and modified to fit Florida lakes. Analogous loading plots for nitrogen also were developed since many Florida lakes are nitrogen-limited.

Total Phosphorus Concentration. Most of the predictive equations for total phosphorus ($[P]_1$) analyzed here are based on equations developed by previous investigators. Coefficients for the equations were modified using regression analysis to improve their predictive capability for the Florida NES lakes. Log-log transformations of both $(P)_1$ and the predictive terms in the equations were used because of the wide ranges of encountered values.

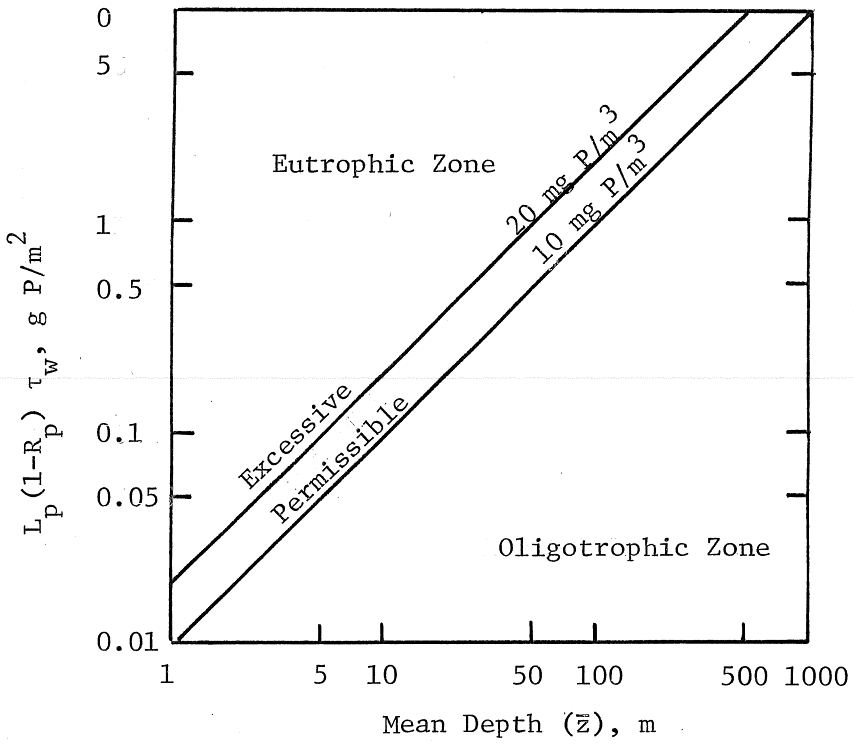


Figure V-3. Dillon's phosphorus loading criteria (1975), with lines of constant phosphorus concentration distinguishing trophic states.

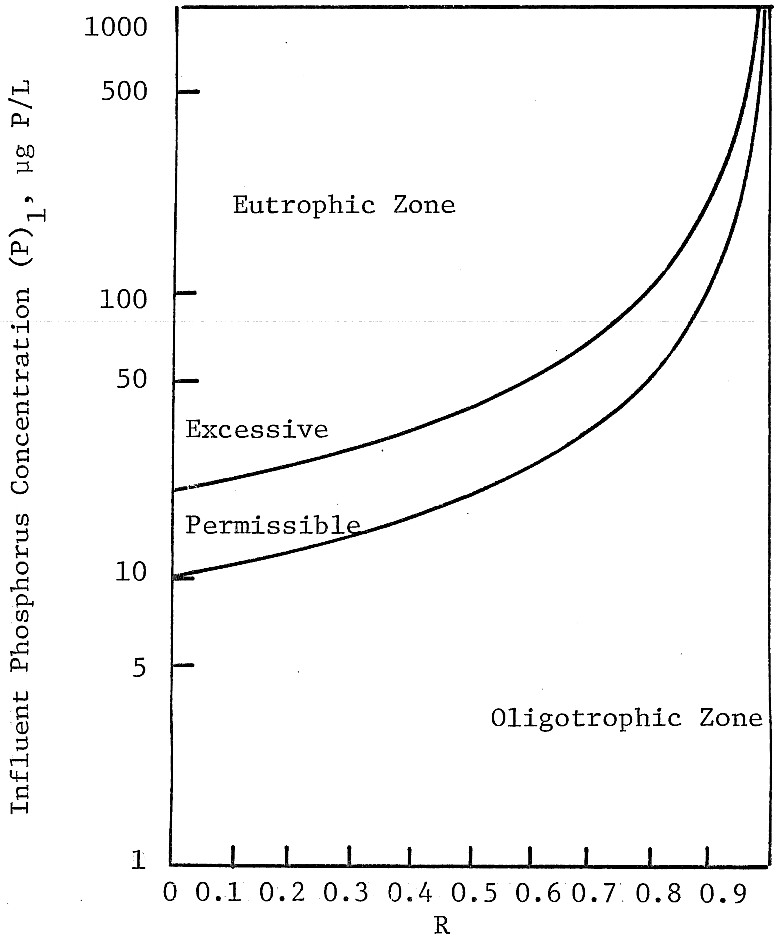


Figure V-4. The Larsen and Mercier phosphorus loading criteria (1975), Equation TP5 is based on the concept of a "mean apparent settling velocity" for phosphorus (Chapra 1975); v_p was calculated for each of the Florida NES lakes by rearranging equation 5-16:

$$v_p = \frac{q_s R_p}{1 - R_p} \qquad (5\text{-}28)$$

The mean value of v_p for the Florida NES lakes, 8.53 m/yr, was then substituted into equation 5-17 to produce TP5.

Eight equations were evaluated (Table V-1), and the r^2 values ranged from 0.77 to 0.91. The best predictive equation, TP2, is based on Dillon and Rigler's (1975) model. Prediction of total phosphorus using TP2 requires data on areal phosphorus loading, hydraulic loading, and the phosphorus retention coefficient (R_p). To obtain the best estimates of lake phosphorus concentration the measured phosphorus retention coefficient $[R_p = (\Sigma P_{in} - \Sigma P_{out})/\Sigma P_{in}]$ should be used in TP2. However, since the outflow phosphorus loading may not be known or economically determined, it may be necessary to predict R_p from morphometric and hydrological parameters. The ability to predict R_p in Florida lakes by equations involving such parameters is limited (see p. 80), and estimates of $(P)_1$ made using predicted values of R_p in TP2 will not be very accurate. Several other equations that do not require data on outflow phosphorus loading (TP3 to TP6) have r^2 values between 0.82 and 0.84, but these equations all have C.V. values higher than TP2. The 95% confidence limits for individual predictions shown in Figure V-5 can be used to evaluate the accuracy of predictions made using TP2. For example, a lake whose predicted phosphorus concentration is 0.100 mg/L has a 95% CLI of 0.057 to 0.116 mg/L.

Total Nitrogen Concentrations. The nitrogen balance in a lake can be expressed by the following equation:

$$dN/dt = N_{in} - N_{out} + N_{fix} - N_{sed} - N_{den} \qquad (5\text{-}29)$$

where dN/dt = change in the mass of nitrogen in a lake,

N_{in} = flux of nitrogen into lake,
N_{out} = flux of nitrogen throughout outflow,
N_{fix} = rate of nitrogen fixation,
N_{den} = rate of denitrification, and
N_{sed} = rate of nitrogen loss by sedimentation.

Since N_{fix} and N_{den} were not determined for the NES lakes, these fluxes are grouped together with N_{sed} as parts of a composite loss term, σ_N N. The nitrogen mass balance equation thus simplifies to:

$$dN/dt = N_{in} - \rho_w N - \sigma_N N, \qquad (5\text{-}30)$$

which is analogous to the phosphorus mass balance (eq. 5-2). At steady state:

$$\frac{dN}{dt} = 0 = N_{in} - N(\rho_w + \sigma_N) \qquad (5\text{-}31)$$

or

$$N = \frac{N_{in}}{\rho_w + \sigma_N} \qquad (5\text{-}32)$$

Table V-1 Predictive equations for total phosphorus concentration

	Original Equation			Modified Equation		
Predictive Equation*	Investigator	Eqn. no. in text	r^2	n**	r^2	C.V.
$TP1=0.682[L_p/q_s(1+\sqrt{\tau_w})]^{0.934}$	Vollenweider (1976)	5- 9	-	29	0.79	31.3
$TP2=0.748[L_p(1-R_p)/q_s]^{0.862}$	Dillon and Rigler (1975)	5-14	-	25	0.91	14.8
$TP3=0.984[L_p(1-R_{(K+D)})/q_s]^{0.851}$	Dillon and Rigler (1975)	5-14	-	28	0.84	23.6
$TP4=0.706[0.84L_p/(0.65\bar{z}+q_s)]^{0.964}$	Jones and Bachman (1976)	5- 8	0.84	29	0.82	29.3
$TP5=0.952[L_p/(8.53+q_s)]^{.860}$	Chapra (1975)	5-17	-	29	0.83	28.1
$TP6=0.885\left[\frac{L_p}{\frac{18\bar{z}}{10+\bar{z}}+1.05q_s\exp(0.012q_s)}\right]^{0.968}$	Reckhow (1977)	5-18	0.88	29	0.82	29.1
$TP7=0.643\left[\frac{L_p}{q_s(1+0.824\tau_w^{0.454})}\right]^{0.932}$	Walker (1977)	5-19	0.91	29	0.78	32.6
$TP8=0.416[L_p/q_s]^{0.873}$	This study	-	-	26	0.77	30.0

* For each predictive equation (except TP10), TP_i = a [Original Equation]^b where a and b are constants determined by the regression.

** n is the number of lakes included in the regression.

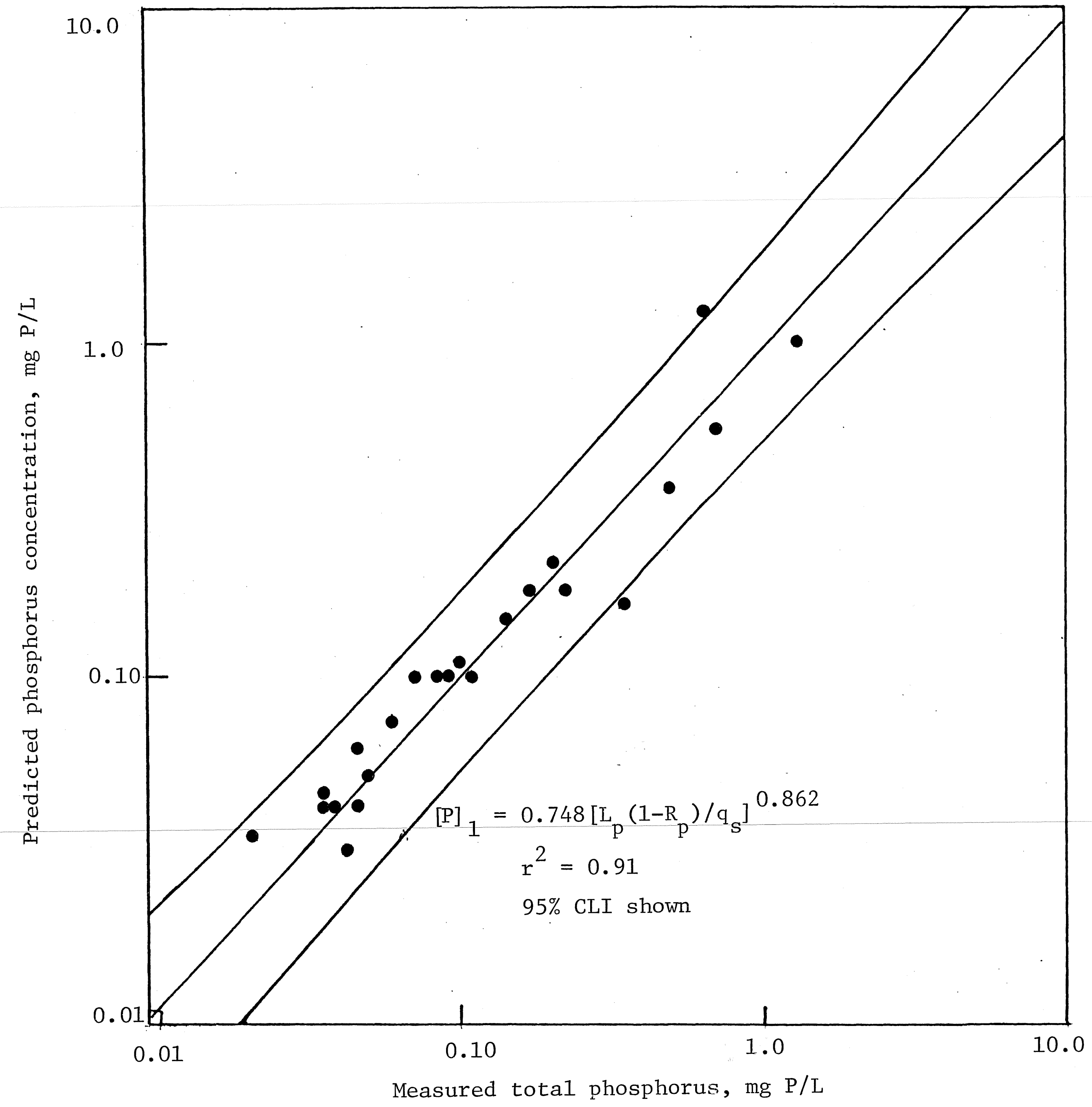


Figure V-5. Predicted vs. measured total phosphorus concentration using equation TP2.

Dividing by volume (V), we obtain a predictive equation for the steady-state concentration of total nitrogen:

$$(N)_1 = \frac{N_{in}}{V(\rho_w + \sigma_N)} = \frac{L_n}{\bar{z}(\rho_w + \sigma_N)} \quad . \tag{5-33}$$

Since there are no existing equations to predict total nitrogen concentration, the equations evaluated here were based on modifications of phosphorus loading equations. For example, TN1 was derived by setting $\sigma_N = 0.65\bar{z}$ and substituting L_n for L_p; TN2 - TN3 were derived by substituting L_n for L_p and R_n for R_p. Equation TN5 was derived by computing a mean apparent settling velocity for nitrogen (v_n) in Florida NES lakes in a manner analogous to that used above to compute v_p.

The predictive equations (TN1-TN4) have r^2 values ranging from 0.52 to 0.77 (Table V-2). The best predictive equation, TN2, is based on Dillon and Rigler's (1974a) model, with L_n and R_n substituted for L_p and R_p. TN2 requires data on areal nitrogen loading, hydraulic loading and the nitrogen retention coefficient. The use of measured retention coefficients will produce the most accurate estimates of mean lake nitrogen concentration. However, in some cases measured values of R_n may not be available; it may therefore be necessary to use predicted values of R_n. Equations to predict R_n, evaluated below, are unfortunately not highly accurate. Thus, when it is necessary to use predicted values of R_n in TN2, the resulting predictions of $(N)_1$ will be of limited usefulness. Although the other three predictive equations do not require data on the outflow nitrogen loading, their predictive capacity is considerably lower than that of TN2. The small 95% confidence limits for individual predictions shown in Figure V-6, indicates that good accuracy can be achieved in predicting $[N]_1$ using TN2.

Phosphorus and Nitrogen Retention Coefficients. Since the best equations to predict total phosphorus and total nitrogen require the use of retention coefficients, it is desirable to be able to predict R_p and R_n using data on the hydrologic and morphologic characteristics of a lake. Several investigators (Kirchner and Dillon 1975; Larsen and Mercier 1975) have shown that this approach can be successful in predicting R_p. The seven R_p predictive equations examined here (Table V-3) were based on modifications of previous equations using the parameters $\bar{z}$, τ_w, and q_s. Unfortunately, none of the equations is very successful in predicting R_p.
The r^2 values for the equations ranged from 0.41 to 0.53. A plot of the measured phosphorus retention R_p versus the values predicted by the best predictive equation (RP6) shown in Figure V-7 illustrates the substantial scatter in the relationship, and the 95% CLI for new predictions shows that at mean value of R_p (0.48) the predicted values is approximately ± 0.4 of the actual value. The best predictive equation (RP6) was developed here by combining mean depth ($\bar{z}$) and water residence time (τ_w) by multiplication, rather than by division as is done to define the areal hydraulic loading rate. The physical meaning of this factor ($\bar{z} \cdot \tau_w$) is uncertain, and the difference in predictive capability between RP6 and the other equations is too small to warrant conclusions about the relative importance of mean depth and water residence time as predictive parameters of phosphorus retention.

Table V-2. Predictive equations for total nitrogen concentration

Predictive Equations*	Original Basis for Equation	n	r^2	C.V.
$TN1 = 1.08\ [L_n/(0.65\bar{z} + q_s)]^{0.859}$	Jones and Bachmann (1976)	27	0.53	69.7
$TN2 = 0.899\ [L_n/(1 - R_n)/q_s]^{0.976}$	Dillon and Rigler (1975)	24	0.77	47.6
$TN3 = 0.841\ [L_n/q_s]^{0.877}$	This study	27	0.52	70.2
$TN4 = 1.29\ [L_n/q_s(1 + \sqrt{\tau_w})]^{0.858}$	Vollenweider (1976)	27	0.55	67.9
$TN5 = 1.69\ [L_n/(5.49 + q_s)]^{0.341}$	Chapra (1975)	29	0.19	91.5

* For each predictive equation (except TN3), the original equation was a total phosphate predictive equation with L_p and R_p replaced by L_n and R_n. Thus, TN_i = a[Transformed Original Equation]^b where a and b are constants determined by the regression.

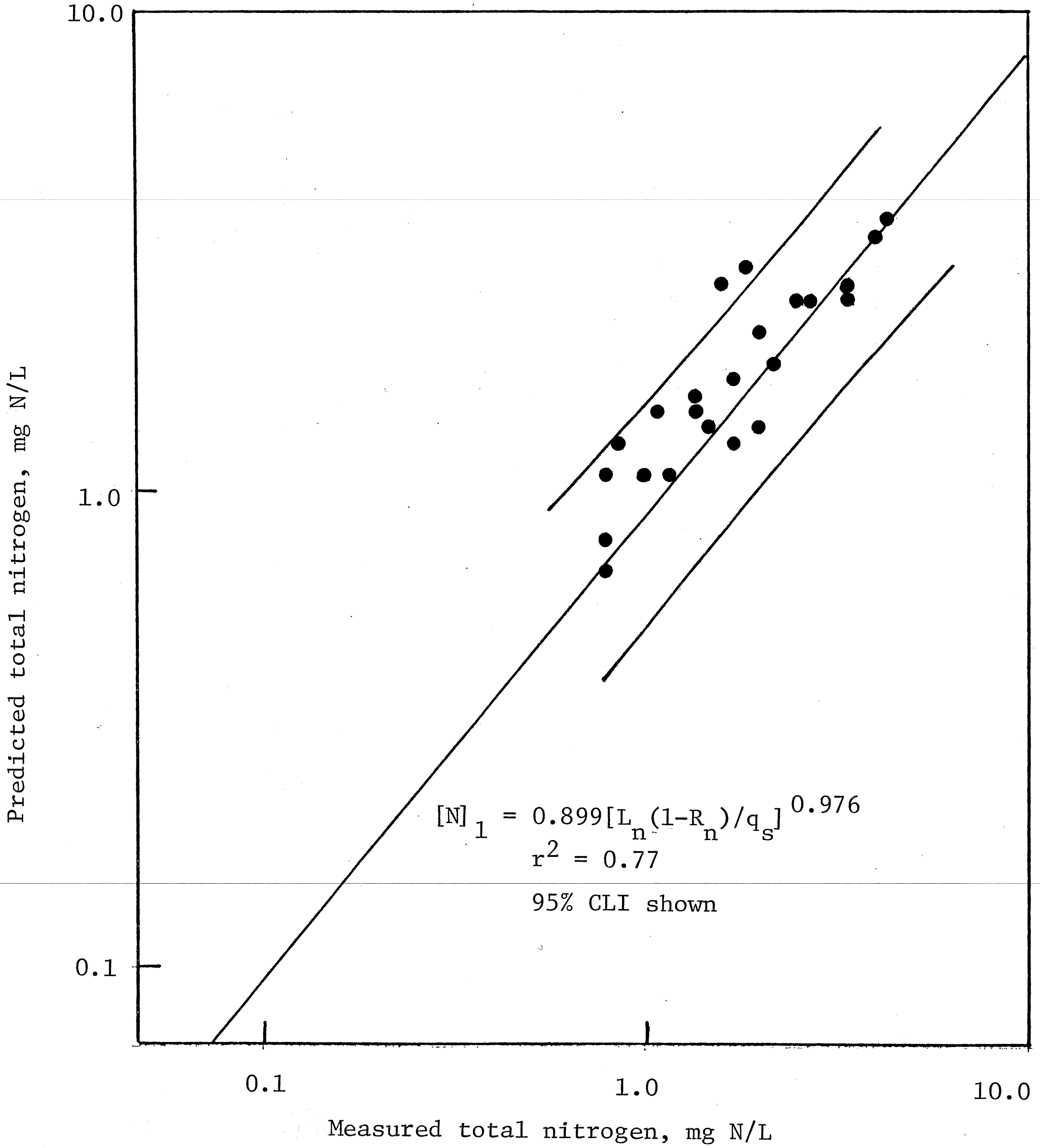


Figure V-6. Predicted vs. measured total nitrogen concentration using equation TN2.

Table V-3. Predictive equations for phosphorus retention coefficient

Predictive Equation*	Original Equation			Modified Equation		
	Investigator	Eqn. no. in text	r^2	n	r^2	C.V.
$RP1 = 0.767 - 0.367 \log (q_s)$	Larsen and Mercier (1975)	5-21	0.88	27	0.46	42.2
$RP2 = 0.639 + 0.355 \log (\tau_w)$	Larsen and Mercier (1975)	5-22	0.86	27	0.46	41.9
$RP3 = -0.056 + 1.40/(1+\sqrt{\rho_w})$	Larsen and Mercier (1975)	5-24	0.88	27	0.47	41.6
$RP4 = -0.009 + 8.17/(10+q_s)$	Larsen and Mercier (1975)	-	0.86	27	0.46	41.9
$RP5 = -0.249 + 0.487 \exp(-0.271\, q_s) + 0.656 \exp(-0.00949\, q_s)$	Kirchner and Dillon (1975)	5-20	0.88	27	0.46	41.9
$RP6 = 0.500 + 0.353 \log (\bar{z} \cdot \tau_w)$	This study	-	-	26	0.53	40.3
$RP7 = -0.131 + 1.07\, R_{(K+D)} - 0.172(N_{out}/Q_{out})(R_{(K+D)})\sqrt{shape/\bar{z}}$	Hand (1975)	-	0.50	26	0.41	44.4
$RP8 = 0.734[8.53/(8.53 + q_s)]^{0.91}$	Chapra (1975)	5-16	-	26	0.34	67.2

* For each predictive equation (except RP6) the variables used were the same as those in the original equation, but the equation is altered by new regression constants a and b, such that $RP_i = a + b$ [Original Variable].

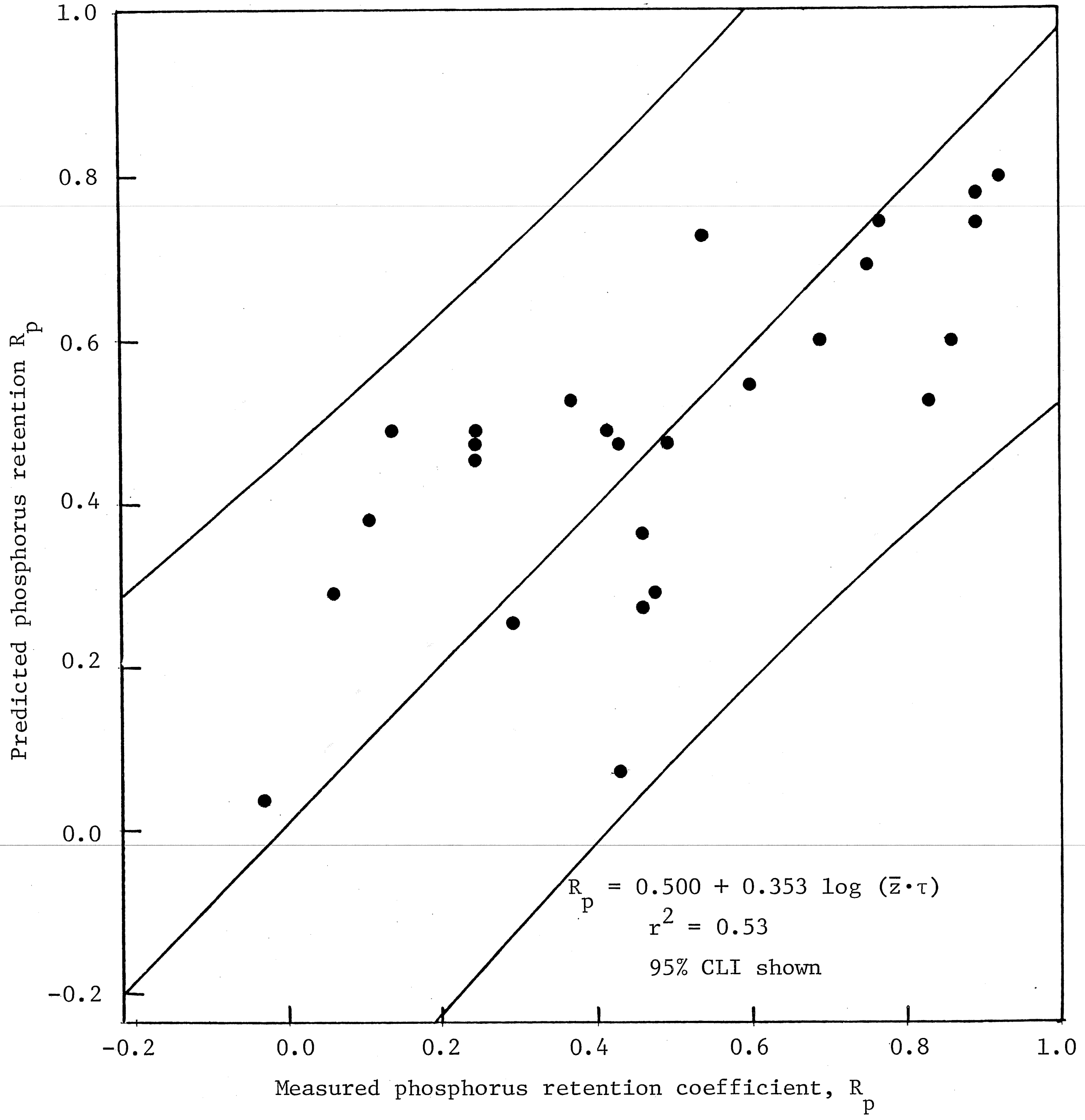


Figure V-7. Predicted vs. measured phosphorus retention coefficients using Equation RP6.

The predictive equations examined here to estimate R_n similarly were based on hydrologic and morphologic characteristics (RN1 to RN5). These equations (Table V-4), like those examined to predict R_p, also were not very successful in predicting nitrogen retention coefficients for Florida's NES lakes. The r^2 values of the equations ranged from 0.12 to 0.51, and C.V. values ranged from 67.0 to 89.6. The width of the 95% confidence intervals shown for individual predictions in a plot of measured nitrogen retention (R_n) vs. values predicted by the best equation (RN4) illustrates its poor predictive ability (Figure V-8). Equation RN4 predicts R_n as a function of the mean inflow nitrogen concentration (L_n/q_s). The inclusion of the areal nitrogen loading L_n in the predictive equation resulted in a vast improvement over the other equations that were based entirely on hydrologic and morphologic data.

The poor correlations between observed and predicted values of R_n and R_p using predictive equations RN1-RN5 and RP1-RP8 could be due to several factors. First, it is likely there are substantial errors in many of the NES water and nutrient budgets, particularly for many lakes where portions of the nutrient and water budgets were estimated rather than measured directly. In particular, nutrient inputs were estimated for ungauged tributaries and for some municipal wastewater treatment plants. It is also possible that morphologic and hydrologic factors are simply not good predictors of nutrient retention in Florida's lakes. Unlike temperate zone lakes, most of the Florida NES lakes are shallow and do not undergo thermal stratification. Nutrients lost via sedimentation thus may re-enter the water column more readily in Florida lakes than in temperate zone lakes. Pollman (unpublished data) has shown that resuspension of sediments in Lake Apopka during storms causes a significant, temporary increase in the concentration of soluble reactive phosphorus from sediment particles. Although the effect of this phenomenon on the long-term retention of phosphorus is not known, the data suggest that long-term retention may be affected. Biological processes also may be more important in affecting nutrient retention in Florida lakes than in temperate zone lakes because of the warm climate, long growing season and generally nutrient-enriched conditions. For example, release of sediment-derived phosphorus to the water column by rooted aquatic macrophytes has been shown to be an important mechanism affecting the concentration of phosphorus in several lakes (Smith 1978, Lie unpublished ms). It seems reasonable that macrophytes may have important effects on nutrient retention in many Florida lakes, although this has not been studied.

Prediction of chlorophyll a. The predictive models for chlorophyll *a* evaluated here (Table V-5) generally were based on loading expressions developed by previous investigators, although regressions also were determined for relationships between chlorophyll *a* levels and lake nutrient concentrations. CHA12 and CHA13 are regression equations that describe the relationship between chlorophyll *a* and the concentrations of phosphorus and nitrogen, respectively, in the Florida NES lakes. As would be expected for a group of primarily nitrogen-limited lakes, nitrogen is better correlated with chl *a* ($r^2 = 0.79$) than is phosphorus ($r^2 = 0.63$). The coefficients for CHA13 are nearly identical to those of eq. 4-17, which describes the relationship between (chl *a*) and $(N)_1$ for the 44 nitrogen-limited lakes in the entire set of 101 study lakes.

Table V-4. Predictive equations for nitrogen retention coefficient

Predictive Equation*	Original Basis for Equation	n	r^2	C.V.
$RN1 = 0.445 - 0.189 \log(q_s)$	Larsen and Mercier (1975)	25	0.12	89.6
$RN2 = 0.391 + 0.210 \log(\tau_w)$	Larsen and Mercier (1975)	25	0.16	87.5
$RN3 = 0.322 + 0.198 \log(\bar{z}\tau_w)$	This study	25	0.18	86.6
$RN4 = 0.010 + 0.597 \log(L_N/q_s)$	This study	25	0.51	67.0
$RN5 = 0.105 + 0.775\, R_{(K+D)} - 0.124\, (N_{out}/Q_{out})(R_{(K+D)})\sqrt{shape/\bar{z}}$	Hand (1975)	24	0.20	88.3

* For RN1, RN2, and RN5 the variables used in the predictive equation were the same as those used in equations RP1, RP2, and RP7, respectively.

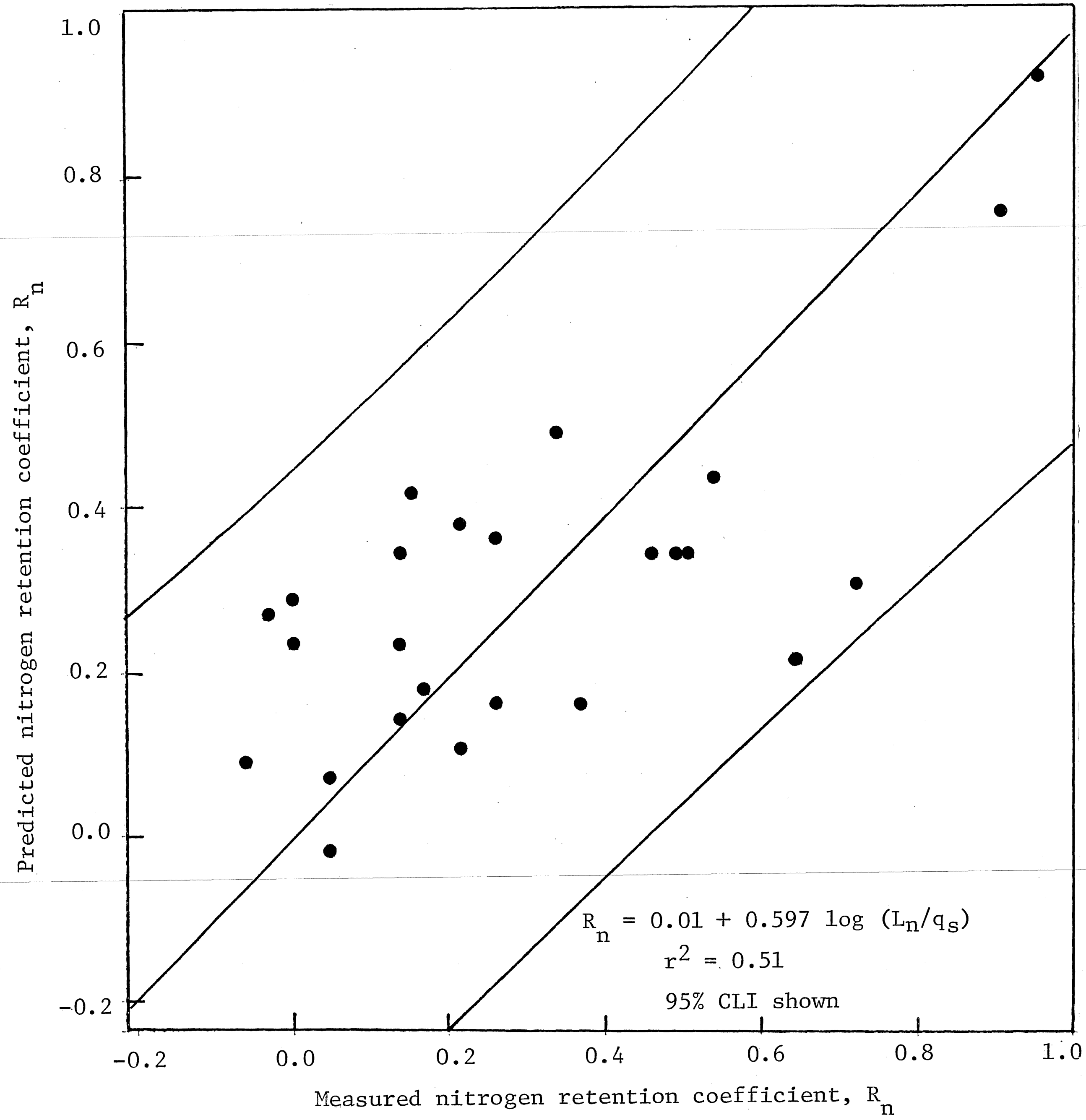


Figure V-8. Predicted vs. measured nitrogen retention coefficients using equation RN4.

Table V-5. Predictive equations for chlorophyll *a* concentration.

	Original Equation			Modified Equation		
Predictive Equations*	Investigator	Eqn. no. in text	r^2	n	r^2	C.V.
$CHA1=81.3[L_p/q_s(1+\sqrt{\tau_w})]^{0.652}$	Vollenweider (1976)	24	0.75	29	0.59	21.1
$CHA2=91.6\left[\frac{L_p}{8.53+q_s}\right]^{0.549}$	Vollenweider (1975)	5-30d	-	29	0.52	22.8
$CHA3=21.9\left[\frac{L_n}{5.49+q_s}\right]^{0.786}$		5-34a	-	29	0.32	27.1
$CHA4=9.82[L_n(1-R_n)/q_s]^{1.58}$	Dillon and Rigler (1974a)	-	-	26	0.60	19.0
$CHA5=89.3[L_p(1-R_p)/q_s]^{0.604}$	Dillon and Rigler (1974a)	-	-	27	0.52	22.7
$CHA6=8.30[L_n/(0.65\bar{z}+q_s)]^{1.71}$	Jones and Bachmann (1976)	-	-	26	0.69	19.2
$CHA7=73.4[L_p/(0.65\bar{z}+q_s)]^{0.667}$	Jones and Bachmann (1976)	-	-	29	0.60	20.9
$CHA8=97.5\left[\frac{L_p}{\frac{18\bar{z}}{10+\bar{z}}+1.05q_s\exp(0.012q_s)}\right]^{0.676}$	Reckhow (1977)	-	-	29	0.61	20.5
$CHA9=78.7\left[\frac{L_p}{q_s(1+0.824\tau_w^{0.454})}\right]^{0.657}$	Walker (1977	-	-	29	0.59	21.1
$CHA10=59.2[L_p/q_s]^{0.663}$	This study	-	-	29	0.57	21.7
$CHA11=5.62[L_n/q_s]^{1.61}$	This study	-	-	26	0.58	22.5
$CHA12=101(P)_l^{0.640}$	This study	-	-	40	0.63	20.0
$CHA13=11.5(N)_l^{1.60}$	This study	-	-	39	0.79	14.7
$CHA14=97.8[N_{out}/3+P_{out})\sqrt{shape/\bar{z}}/Q_{out}]^{1.04}$	Hand (1975)	25	0.94	26	0.59	20.1

* For CHA1 - CHA9, and CHA14, CHA_i = a [Original Equation]^b where a and b are constants determined by the regression. However, in the case of CHA1, CHA2, and CHA14, a and b are the result of multiplication by the original regression constants also.

For predictive models based on loading terms, CHA6 gave the best results ($r^2 = 0.69$). This equation is based on the equation of Jones and Bachman (1976) with L_n substituted for L_p. A plot of predicted versus measured values of chlorophyll *a* using CHA6 (Figure V-9) shows the 95% CLI for this equation. Thus, for a lake having a predicted chlorophyll *a* concentration of 20 μg/L, the 95% CLI is 5 to 74 μg/L. Although this level of predictibility is useful when considering the range of chlorophyll *a* values in the entire data set (3 to 208 μg/L), further refinement of models to predict chlorophyll *a* concentrations from nutrient loading data is needed.

NUTRIENT LOADING CRITERIA FOR FLORIDA LAKES

One of the major objectives of the project this report summarizes was to develop nutrient loading criteria for Florida lakes. In this section, existing nutrient loading criteria (Vollenweider 1968; Shannon and Brezonik 1972; Vollenweider 1975; Dillon 1975) are analyzed for their ability to predict trophic status in the Florida NES lakes. The phosphorus loading criteria based on mass balance models are then modified, using the predictive equations developed earlier in the previous section, to improve their predictive ability for the Florida NES lakes. Since many of Florida's lakes are nitrogen-limited, nitrogen loading criteria have been developed using I/O models analogous to those used for the development of phosphorus loading criteria.

The loading criteria are evaluated according to their ability to predict the trophic status of the Florida NES lakes. In addition, a "trophic ratio", defined as the ratio of a lake's nutrient loading to the minimum eutrophic loading for that lake, is used to evaluate the degree of eutrophication predicted by each model.

In earlier developments of nutrient loading criteria, the terms "excessive" and "permissible" have been used to describe the minimum loading levels that result in eutrophic and mesotrophic conditions, respectively, (Vollenweider 1968, 1975; Dillon 1975; Larsen and Mercier 1975). These terms invoke a value judgement that many aquatic scientists now regard as unnecessary and unjustified. Thus, in this report we have used "minimum eutrophic loading" (MEL) to designate the minimum loading required to cause eutrophic conditions and "minimum mesotrophic loading" (MML) to designate the minimum loading required to cause mesotrophic conditions. These terms correspond respectively to the "excessive" and "permissible" loadings of earlier investigators.

Phosphorus Loading Models.

Vollenweider (1968) Loading Model. The Vollenweider (1968) loading model delineates trophic state as a function of mean lake depth. The minimum eutrophic loading and the minimum mesotrophic loadings are:

$$MEL_p = 0.05\bar{z}^{0.6} \quad (5\text{-}34a)$$

$$MEL_p = 0.025\bar{z}^{0.6} \quad (5\text{-}34b)$$

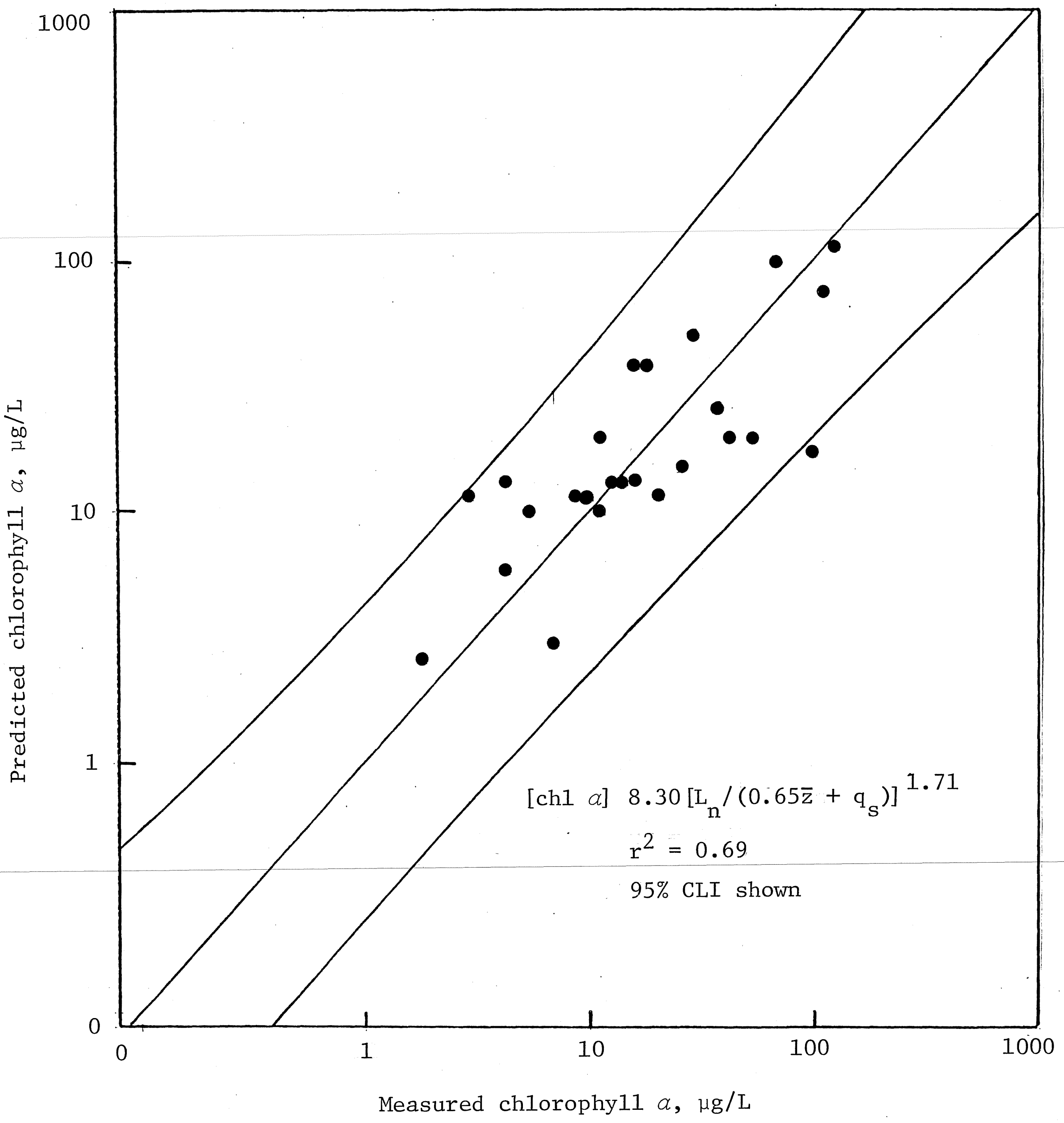


Figure V-9. Predicted vs. measured chlorophyll *a* using equation CHA6.

According to the Vollenweider (1968) criteria (Figure V-10), all of the Florida NES lakes are eutrophic. Thus, these criteria are conservative with respect to the classification of the mesotrophic lakes, since all of the mesotrophic lakes lie above the minimum eutrophic loading line (MEP_p). The failure of the Vollenweider (1968) model is not surprising since this model does not include any hydrologic variables. The Florida NES lakes are extremely diverse with respect to hydrologic conditions ($0.03 \text{ yr} \leq \tau \leq 2.90 \text{ yr}$), and hydrologic conditions are an important factor affecting trophic state in these lakes.

Shannon and Brezonik (1972) Model. Phosphorus loading criteria were developed by Shannon and Brezonik (1972) for Florida lakes using a data base of 55 lakes in the northern part of the state. Their criteria were based on volumetric loading rates (0.22 and 0.12 g P/m^3-yr, respectively, for excessive and permissible loading rates), and like the original Vollenweider criteria, they ignore hydrologic conditions. Although these criteria are more successful in predicting the trophic status of mesotrophic NES lakes than are Vollenweider's criteria, the extent of eutrophication expressed is excessive for many lakes (Table V-6), particularly those with high flushing rates. For the lakes in which $\tau_w < 0.10$ years (Monroe, LC 29; Howell LC 32; Banana, LC 33; Trout, LC 36; Lawne, LC 37 and Munson, LC 38) the degree of eutrophication expressed by the Shannon and Brezonik criteria is greater than that expressed by the models that incorporate hydrologic variables.

Vollenweider (1975) Model. The criteria proposed by Vollenweider (1975) are based on the equation:

$$[P]_1 = \frac{L_p}{\sigma_p \bar{z} + q_s} \qquad (5\text{-}35a)$$

Vollenweider found $\sigma_p \bar{z} \simeq 10$ m/yr for his study lakes. He also considered "permissible" (i.e., minimum mesotrophic) and "excessive" (i.e., minimum eutrophic) levels of total phosphorus to be 0.01 mg/L and 0.02 mg/L, respectively. Substituting these values into eq. 5-30a produces

$$MEL_p = 0.20 + 0.02\, q_s \qquad (5\text{-}35b)$$

$$MML_p = 0.10 + 0.01\, q_s \qquad (5\text{-}35c)$$

These criteria (Figure V-11) are an improvement over Vollenweider's 1968 criteria, although most of the mesotrophic lakes still appear in the eutrophic zone.

In order to improve this model for application to Florida lakes, two modifications were made. First, the concentration criteria for phosphorus were revised to account for the higher level of phosphorus associated with a given concentration of chlorophyll *a* in Florida lakes than found in most temperate lakes. For the phosphorus-limited lakes in the study set (n = 33), the relationship between phosphorus and chlorophyll *a* is given by equation 4-14:

$$(\text{chl } a) = 0.195\, [P]_1 \qquad (4\text{-}14)$$

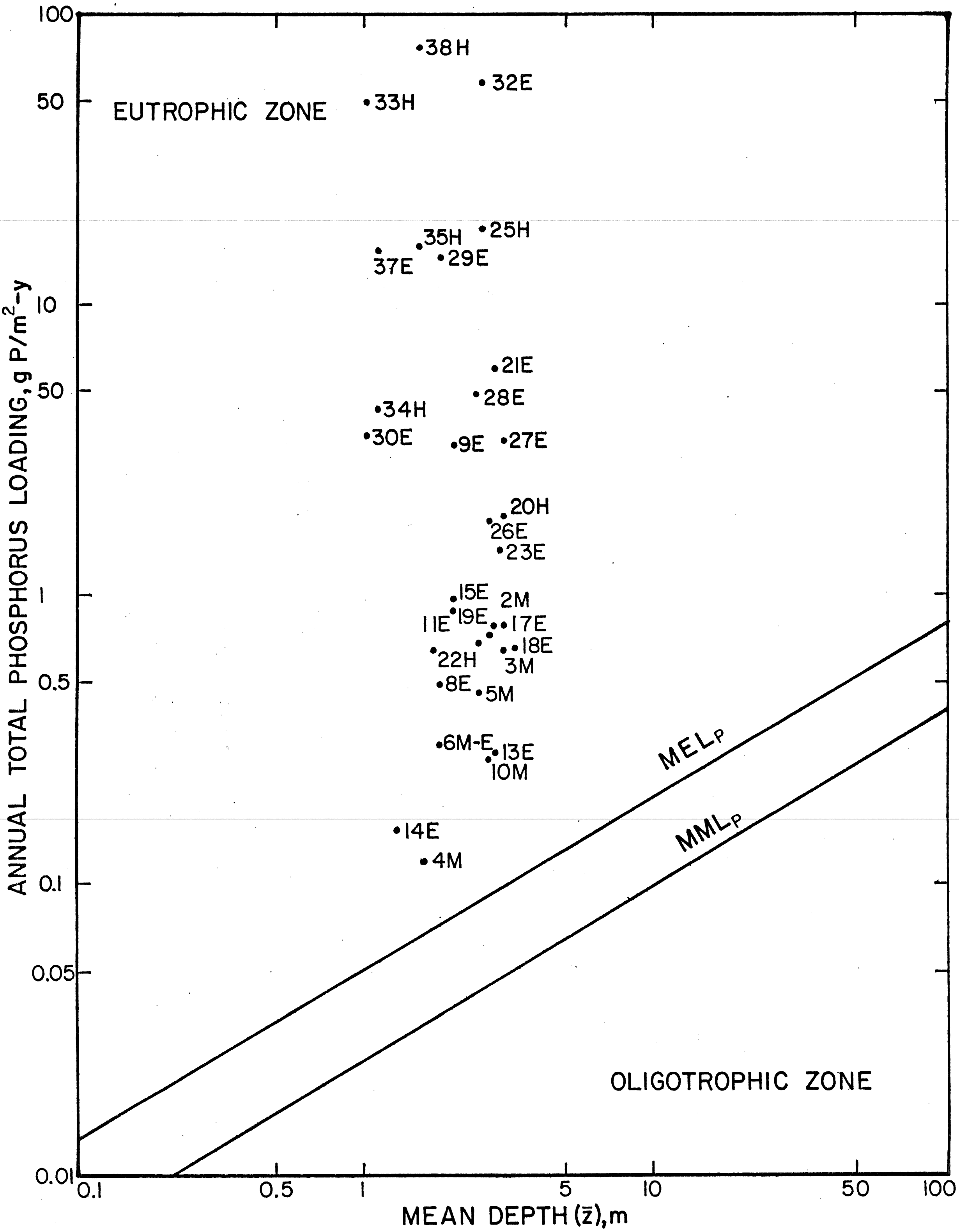


Figure V-10. Trophic state delineation of the Florida NES lakes by the Vollenweider (1968) phosphorus model.

Table V-6. Comparison of phosphorus models by trophic ratios and trophic state classifications.

LC	EPA-NES Assessment	TSI(AVG)	Vollenweider	Shannon and Brezonik (1972a)	Vollenweider (1975)		Dillon (1975)	
					Original	Modified	Original	Modified
1	O-M	46 (O-M)	2.98 (E)	0.45 (O)	0.76 (M)	0.51 (M)	1.31 (E)	0.61 (M)
2	M	52 (M)	7.93 (E)	1.21 (E)	2.57 (E)	1.75 (E)	1.28 (E)	0.59 (M)
3	M	52 (M)	6.52 (E)	0.95 (E)	1.48 (E)	0.97 (M)	2.15 (E)	0.98 (M)
4	M	54 (M-E)	1.81 (E)	0.34 (O)	0.49 (O)	0.34 (O)	1.69 (E)	0.80 (M)
5	M	49 (M)	5.19 (E)	0.82 (M)	1.52 (E)	1.02 (E)	1.82 (E)	0.85 (M)
6	M-E	56 (E)	4.22 (E)	0.76 (M)	1.05 (E)	0.71 (M)	2.67 (E)	1.25 (E)
7	E	56 (E)	--	--	--	--	--	--
8	E	60 (E)	6.89 (E)	1.24 (E)	1.40 (E)	0.93 (M)	1.78 (E)	0.82 (M)
9	E	58 (E)	42.9 (E)	7.39 (E)	--	--	--	--
10	E	57 (E)	--	--	--	--	--	--
11	E	60 (E)	7.85 (E)	1.24 (E)	3.13 (E)	2.0 (E)	3.16 (E)	1.53 (E)
12	E	61 (E)	--	--	--	--	--	--
13	E	62 (E)	3.02 (E)	0.45 (O)	1.25 (E)	0.88 (M)	1.29 (E)	0.61 (M)
14	E	62 (E)	2.56 (E)	0.52 (M)	0.63 (M)	0.43 (O)	2.12 (E)	0.99 (E)
15	E	60 (E)	12.5 (E)	2.16 (E)	3.22 (E)	2.17 (E)	5.00 (E)	2.32 (E)
16	E	59 (E)	--	--	--	--	--	--
17	E	62 (E)	7.97 (E)	1.17 (E)	3.28 (E)	2.28 (E)	5.02 (E)	2.35 (E)
18	E	62 (E)	6.35 (E)	0.90 (M)	2.66 (E)	1.84 (E)	1.62 (E)	0.75 (M)
19	E	61 (E)	11.5 (E)	1.32 (E)	3.88 (E)	2.73 (E)	9.25 (E)	4.33 (E)
20	H	73 (H)	18.9 (E)	2.77 (E)	6.12 (E)	4.13 (E)	5.58 (E)	2.61 (E)
21	E	55 (E)	64.8 (E)	9.76 (E)	6.82 (E)	4.28 (E)	4.55 (E)	2.18 (E)
22	H	76 (H)	9.16 (E)	1.68 (E)	2.94 (E)	2.08 (E)	21.2 (E)	9.73 (E)
23	E	68 (H)	14.9 (E)	2.21 (E)	3.80 (E)	2.51 (E)	4.79 (E)	2.23 (E)
24	E	60 (E)	--	--	--	--	--	--
25	H	63 (E)	208 (E)	32.7 (E)	27.0 (E)	17.11 (E)	36.8 (E)	16.68 (E)
26	E	71 (H)	19.7 (E)	3.01 (E)	5.06 (E)	3.35 (E)	9.87 (E)	4.62 (E)
27	E	64 (E)	34.8 (E)	5.09 (E)	5.81 (E)	3.72 (E)	8.07 (E)	3.72 (E)
28	E	64 (E)	57.8 (E)	9.26 (E)	16.2 (E)	10.91 (E)	8.15 (E)	3.80 (E)
29	E	63 (E)	208 (E)	37.3 (E)	9.77 (E)	6.04 (E)	12.8 (E)	5.42 (E)
30	E	75 (E)	68.8 (E)	15.6 (E)	12.5 (E)	8.48 (E)	--	--
31	E	67 (E)	--	--	--	--	--	--
32	E	64 (E)	675 (E)	106 (E)	72.4 (E)	45.58 (E)	66.4 (E)	31.72 (E)
33	H	77 (H)	987 (E)	224 (E)	95.4 (E)	61.39 (E)	86.0 (E)	41.60 (E)
34	H	73 (H)	80.1 (E)	17.5 (E)	14.2 (E)	9.60 (E)	22.5 (E)	10.66 (E)
35	H	70 (E-H)	245 (E)	47.4 (E)	--	--	--	--
36	H	69 (E-H)	--	--	--	--	--	--
37	E	81 (H)	284 (E)	62.1 (E)	35.5 (E)	23.18 (E)	20.5 (E)	9.37 (E)
38	H	78 (H)	1195 (E)	231 (E)	63.8 (E)	39.64 (E)	--	--
39	H	80 (H)	--	--	--	--	--	--
40	E	79 (H)	--	--	--	--	--	--

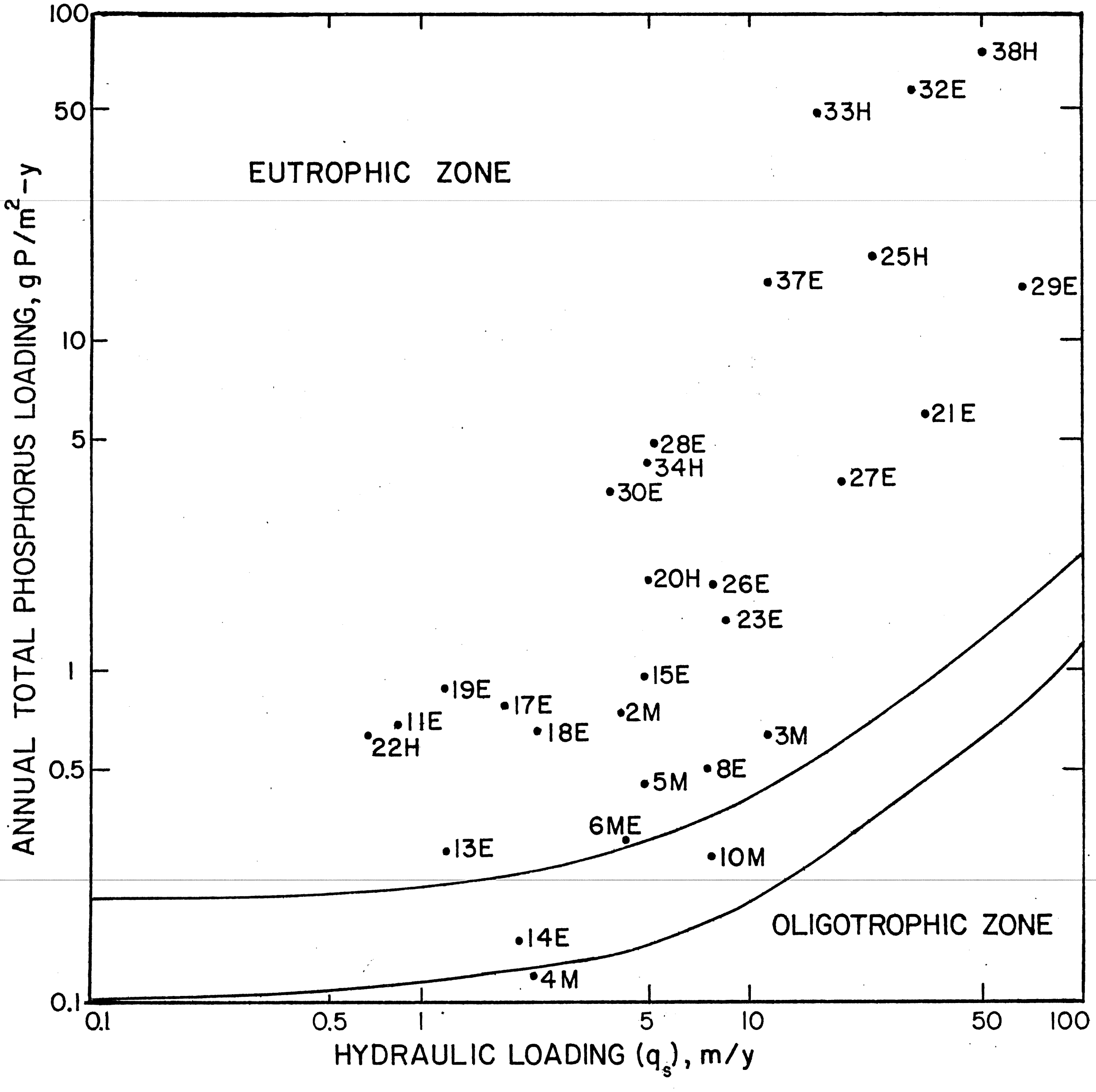


Figure V-11. Trophic state delineation of the Florida NES lakes by the Vollenweider (1975) phosphorus model.

The lower limit of chlorophyll *a* concentrations for mesotrophic conditions is commonly considered to be 5 µg/L, and 10 µg/L is usually considered the lower limit for eutrophic conditions. The corresponding phosphorus concentrations using equation 4-14 are 23 and 46 µg/L. Thus, in revising the phosphorus loading criteria, the minimum mesotrophic and eutrophic phosphorus concentrations were considered to be 25 and 50 µg/L, respectively.

The model was further modified by using the statistically fitted form of equation 5-35a (TP5) determined in the previous section:

$$[P]_1 = 0.952\ [L_p/(8.53 + q_s)]^{0.860}, \tag{5-35d}$$

where 8.53 is the mean value of the apparent settling velocity for phosphorus (v_p), as determined earlier (p.77) using eq. 5-28. When values of 0.050 and 0.025 mg/L phosphorus are used with this equation, the following loading criteria result:

$$MEL_p = 0.033q_s + 0.28 \tag{5-35e}$$

$$MML_p = 0.015q_s + 0.12 \tag{5-35f}$$

The revised criteria, shown in Figure V-12, are an improvement over the original criteria, particularly with respect to the classification of mesotrophic lakes. While the original criteria place three mesotrophic lakes in the eutrophic zone, one of these lakes, plus one classified as mesotrophic-eutrophic are placed in the mesotrophic zone, although two moderately eutrophic lakes (Okeechobee, LC 13 and Kissimmee, LC 8) are now placed in the mesotrophic zone.

Dillon (1975) Model. Dillon (1975) proposed loading criteria for phosphorus based on the predictive equation for $[P]_1$ of Dillon and Rigler (1974a) based on phosphorus loading and hydrologic data (eq. 5-14). setting $[P]_1$ in eq. 5-14 to 0.010 and 0.020 mg P/L to present minimum mesotrophic and eutrophic levels of phosphorus and substituting $q_s = \bar{z}/\tau_w$ resulted in the criteria:

$$(MEL_p)\cdot(1-R_p)\tau_w = 0.020\bar{z} \tag{5-36a}$$

$$(MML_p)\cdot(1-R_p)\tau_w = 0.010\bar{z} \tag{5-36b}$$

Like the phosphorus loading criteria developed by Vollenweider (1968 and 1975), these criteria appear conservative when applied to the Florida NES lakes (Figure V-13).

To improve these criteria for Florida lakes, the minimum mesotrophic and eutrophic levels of phosphorus were adjusted upward to 0.025 and 0.050 mg/L, as was done with the Vollenweider (1975) model. Furthermore, equation TP2, a modification of the original Dillon and Rigler (1974a) predictive equation, was used to express the relationship between phosphorus loading and phosphorus concentration in the revised criteria. Thus, the minimum eutrophic loading of phosphorus becomes:

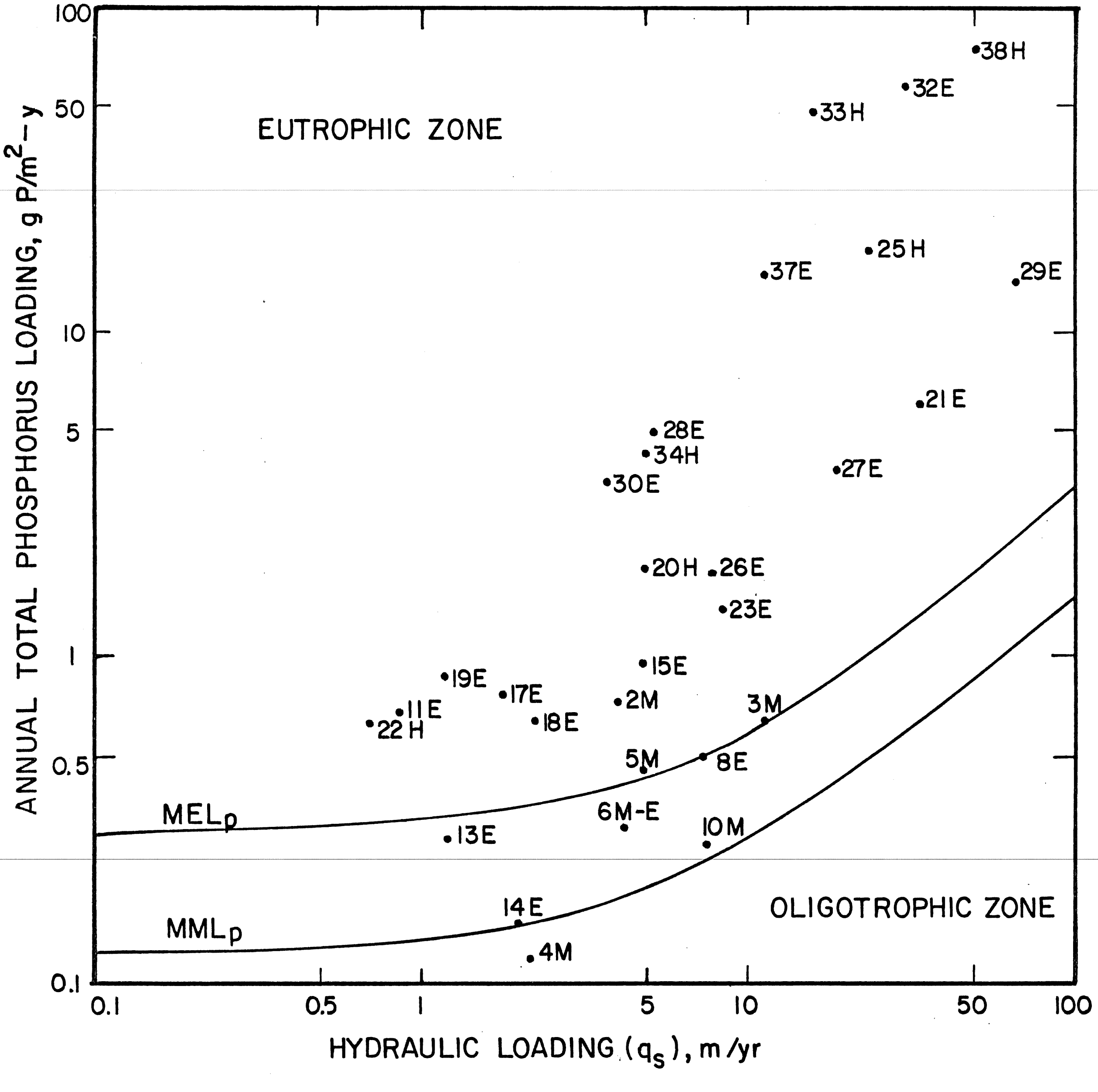


Figure V-12. Trophic state delineation of the Florida NES lakes by the modified Vollenweider (1975) model using loading criteria developed in this study.

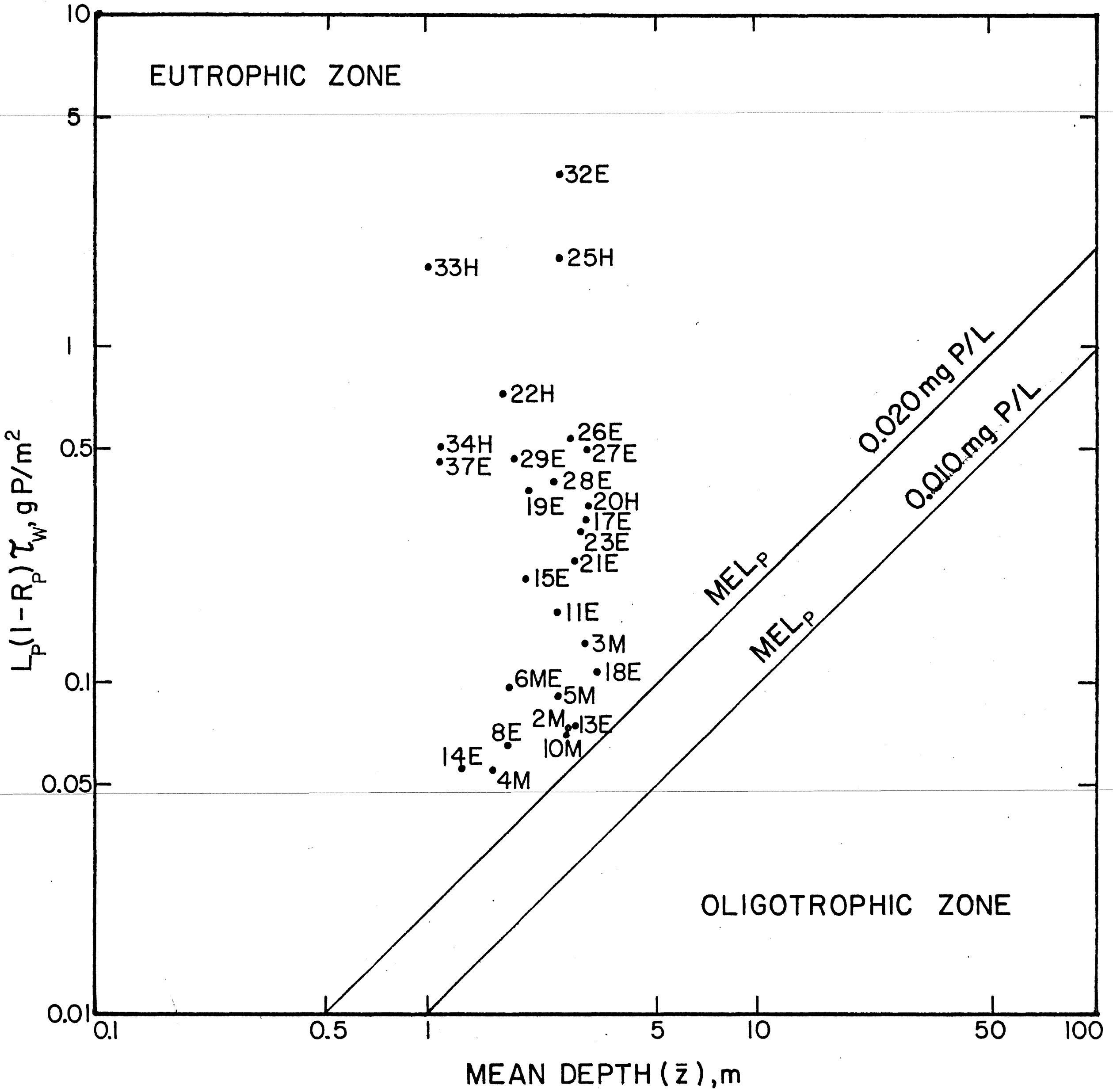


Figure V-13. Trophic state delineation of the Florida NES lakes by the Dillon (1975) phosphorus model.

$$TP2 = 0.748[(MEL_p)\cdot(1-R_p)/q_s]^{0.862} \qquad (5\text{-}36c)$$

$$0.050 = 0.748[(MEL_p)\cdot(1-R_p)/q_s]^{0.862} \qquad (5\text{-}36d)$$

$$(MEL_p)\cdot(1-R_p)\tau_w = 0.043\bar{z} \qquad (5\text{-}36e)$$

and the minimum mesotrophic loading is:

$$0.025 = 0.748[(MML_p)\cdot(1-R_p)/q_s]^{0.862} \qquad (5\text{-}36f)$$

$$(MML_p)\cdot(1-R_p)\tau_w = 0.019\bar{z} \qquad (5\text{-}36g)$$

These criteria, shown in Figure V-14, appear more reasonable than the original criteria, particularly with respect to the classification of mesotrophic lakes. Thus, while the original criteria placed all of the mesotrophic lakes in the eutrophic zone, the revised criteria place all of the mesotrophic lakes in the mesotrophic zone, with only three eutrophic lakes classified incorrectly (all three in the mesotrophic zone).

Although the modified loading criteria improve the predictive ability of the Dillon model, the resulting plot can be further modified in order to spread the points out along the abscissa and improve the clarity of the plot. To accomplish this, each side of equations 5-31e and g was multiplied by τ_w:

$$L_p\ (1 - R_p) = 0.043\ q_s \qquad (5\text{-}36h)$$

$$L_p\ (1 - R_p) = 0.019\ q_s \qquad (5\text{-}36i)$$

The resulting plot, shown in Figure V-15, places each lake in the same zone as does Figure V-14 but has the graphical advantages noted above.

Nitrogen Loading Models

Vollenweider (1968) Model. Vollenweider (1968) developed loading criteria based on his phosphorus loading criteria and an N/P ratio of 15. The minimum eutrophic and mesotrophic areal nitrogen loading rates were:

$$MEL_N = 0.750\bar{z}^{0.6} \qquad (5\text{-}37a)$$

$$MML_N = 0.375\bar{z}^{0.6} \qquad (5\text{-}37b)$$

As with the Vollenweider (1968) phosphorus criteria, the nitrogen criteria classify all of the Florida NES lakes as eutrophic (Figure V-16); these criteria are therefore unacceptable for Florida lakes.

Shannon and Brezonik (1972) Model. Based on their study of Florida lakes, Shannon and Brezonik (1972) proposed criteria of 1.51 g/m^3-yr and 0.86 g m^2/yr as minimum eutrophic and minimum mesotrophic volumetric nitrogen loading rates. These criteria were developed using lakes having relatively long hydraulic residence times and are not successful in classifying the Florida NES lakes since all of these lakes except Lake Okeechobee (LC 13)

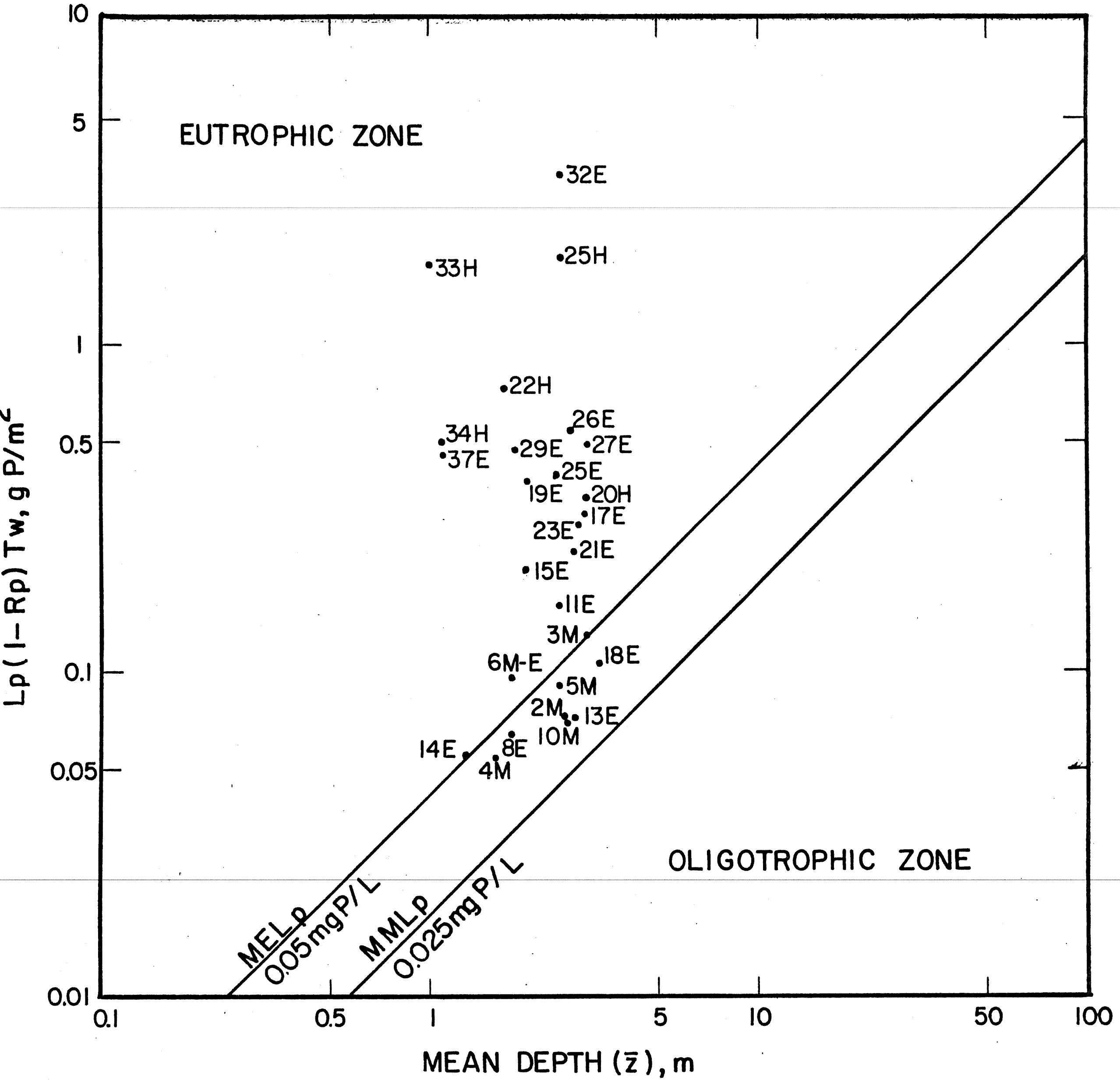


Figure V-14. Trophic state delineation of the Florida NES lakes by the Dillon (1975) phosphorus model using the loading criteria developed in this study.

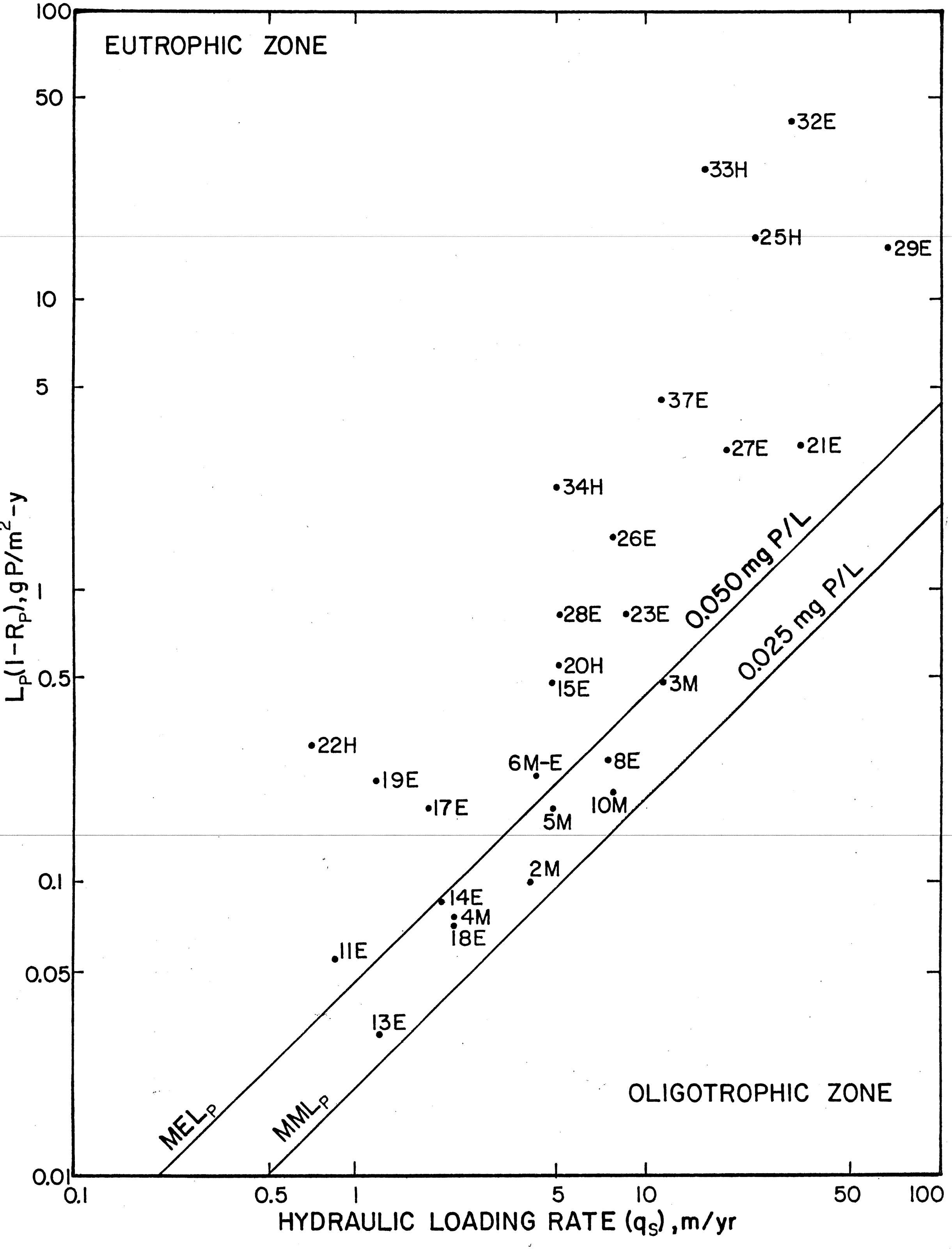


Figure V-15. Trophic state delineation of the Florida NES lakes of the Dillon (1975) phosphorus model using loading criteria developed in this study.

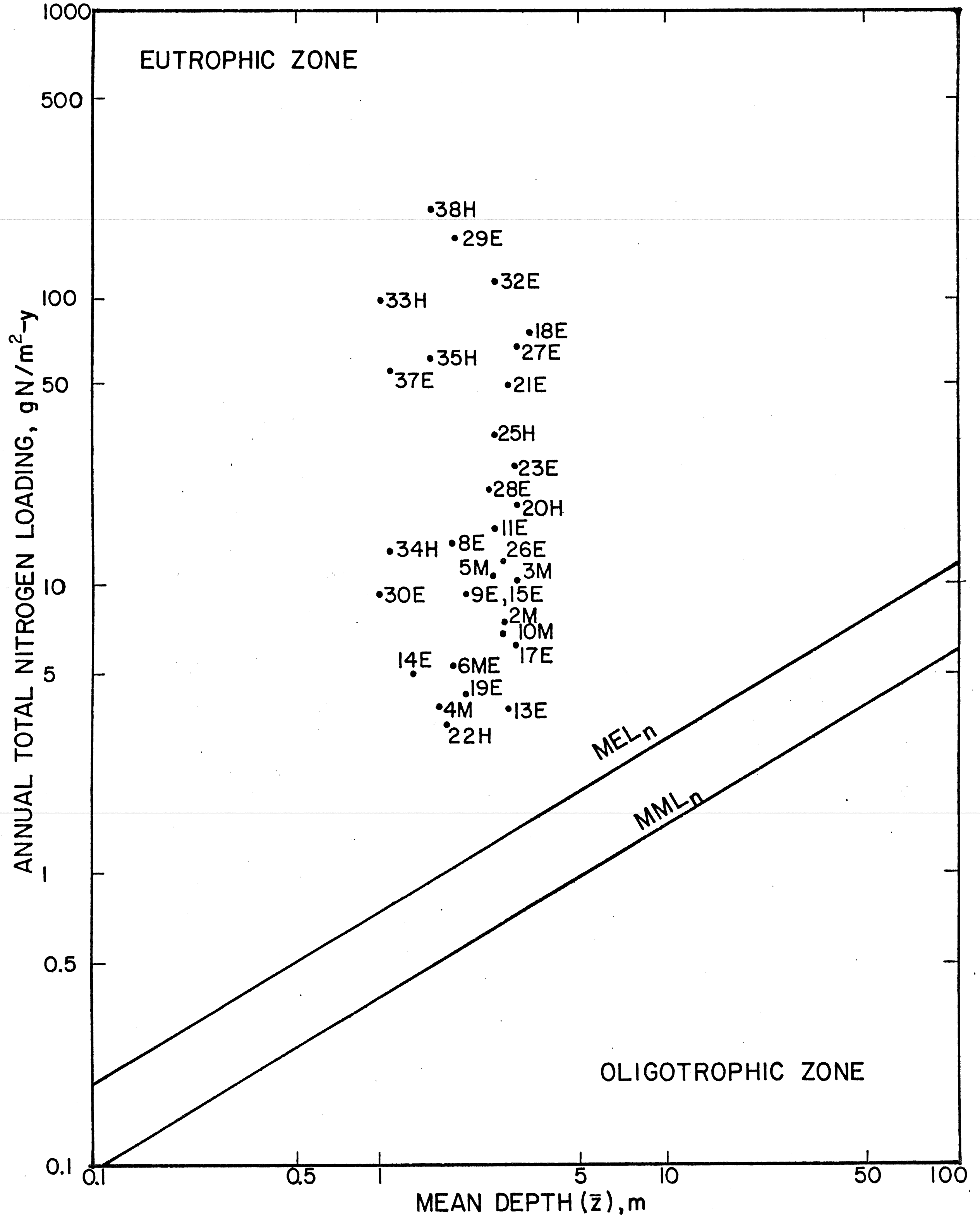


Figure V-16. Trophic state delineation of the Florida NES lakes by the Vollenweider (1968) nitrogen model.

are classified as eutrophic. Furthermore, the degree of eutrophication expressed by these criteria is excessive for lakes with short hydraulic residence times (Table V-7). This fact again illustrates the need to include hydrologic parameters in models to predict lake trophic status.

Vollenweider (1975)-Type Nitrogen Model. A nitrogen-based analog of Vollenweider's (1975) phosphorus loading model was described earlier (equation TN5, p. 80) that predicts $[N]_1$ as a function of nitrogen loading and q_s. This model was used to develop nitrogen loading criteria for Florida's lakes. The first step in developing these criteria was to determine minimum concentrations of $[N]_1$ for mesotrophic and eutrophic conditions. Since most of the Florida NES lakes were nitrogen limited, equation CHA 13 was used to establish the relationship between nitrogen and chlorophyll *a* concentrations:

$$(\text{chl } a) = 11.5(N)_1^{1.60} \qquad (5\text{-}38)$$

For chlorophyll *a* concentrations of 5 and 10 µg/L, the corresponding nitrogen concentrations are 0.59 and 0.92 mg/L. Thus, reasonable criteria for nitrogen concentrations are 1.0 mg/L-N for minimum eutrophic conditions and 0.5 mg/L-N for minimum mesotrophic conditions.

Equation TN5 was then used to compute the minimum eutrophic nitrogen loading:

$$(N)_1 = 1.69\ [L_n/(5.49 + q_s]^{0.341} \qquad (5\text{-}39a)$$

$$1.0 = 1.69\ [L_n/(5.49 + q_s]^{0.341} \qquad (5\text{-}39b)$$

$$MEL_N = 1.18 + 0.21\ q_s \qquad (5\text{-}39c)$$

The minimum mesotrophic loading was determined in the same way:

$$0.50 = 1.69\ [L_n/(5.49 + q_s)]^{0.341} \qquad (5\text{-}39d)$$

$$MML_N = 0.154 + 0.03\ q_s \qquad (5\text{-}39e)$$

The value 5.49 represents the mean for the NES lakes of the apparent settling velocity (v_n) for nitrogen (in m/yr), as determined from an equation analogous to the equation used for v_p (eq. 5-28). The criteria given as eqs. 5-39-c and e are plotted as dashed lines in Figure V-17, and they result in placement of all the NES lakes in the eutrophic zone. Thus on this basis they are not satisfactory. It should be noted that the r^2 value for the equation on which these criteria were based (TN5) was very low (0.18), and its predictive capability is limited. Moreover, the low fractional exponent in TN5 (and hence in eqs. 5-39 b and d) renders these equation unsatisfactory as semi-theoretical (mass balance) models of the behavior of nitrogen in aquatic systems. Thus the regression (TN5) yielded poor predictions at the expense of increased empiricism and decreased realism (i.e. the fractional exponent). Consequently, we developed a set of loading criteria for nitrogen based on the unaltered equation (i.e. without the statistically determined coefficients). This approach at least has the merit of maintaining the model's theoretical appeal and realism. The derived equations for MML_N and MEL_N criteria are:

Table V-7. Comparison of nitrogen models by trophic ratios and trophic state classifications.

LC	EPA-NES Assess-ment	TSI(AVG)	Vollenweider (1968)	Shannon & Brezonik (1972)	Vollenweider (1975) type	Dillon (1975) type
1	O-M	46 (M)	5.07 (E)	1.69 (E)	0.52 (M)	1.93 (E)
2	M	52 (M)	5.44 (E)	1.82 (E)	0.78 (M)	1.05 (E)
3	M	52 (M)	7.10 (E)	2.27 (E)	0.61 (M)	0.79 (M)
4	M	54 (M-E)	3.82 (E)	0.70 (M)	0.49 (O)	1.32 (E)
5	M	49 (M)	1.30 (E)	2.86 (E)	1.04 (E)	0.70 (M)
6	M-E	56 (E)	4.97 (E)	1.95 (E)	0.54 (M)	1.07 (E)
7	E	56 (E)	-	-	-	-
8	E	60 (E)	13.1 (E)	5.15 (E)	1.08 (E)	1.26 (E)
9	E	58 (E)	8.27 (E)	3.11 (E)	-	-
10	E	57 (E)	-	-	-	-
11	E	60 (E)	15.7 (E)	4.13 (E)	2.46 (E)	1.47 (E)
12	E	61 (E)	-	-	-	-
13	E	62 (E)	2.66 (E)	0.88 (M)	0.55 (M)	0.78 (M)
14	E	62 (E)	5.70 (E)	2.55 (E)	0.67 (M)	1.91 (E)
15	E	60 (E)	8.27 (E)	3.11 (E)	0.92 (M)	1.47 (E)
16	E	59 (E)	-	-	-	-
17	E	62 (E)	4.35 (E)	1.39 (E)	0.87 (M)	2.74 (E)
18	E	62 (E)	49.1 (E)	15.2 (E)	9.83 (E)	1.33 (E)
19	E	61 (E)	3.69 (E)	1.39 (E)	0.63 (M)	1.59 (E)
20	H	73 (H)	13.0 (E)	4.17 (E)	1.81 (E)	2.53 (E)
21	E	55 (E)	36.3 (E)	11.9 (E)	1.28 (E)	1.05 (E)
22	H	76 (H)	3.20 (E)	1.29 (E)	0.53 (M)	3.68 (E)
23	E	68 (H)	17.9 (E)	5.82 (E)	1.81 (E)	2.71 (E)
24	E	60 (E)	-	-	-	-
25	H	63 (E)	33.1 (E)	8.72 (E)	1.14 (E)	1.35 (E)
26	E	71 (H)	8.67 (E)	2.89 (E)	0.89 (M)	2.64 (E)
27	E	64 (E)	47.0 (E)	15.0 (E)	2.79 (E)	1.76 (E)
28	E	64 (E)	16.9 (E)	5.91 (E)	2.02 (E)	2.94 (E)
29	E	63 (E)	151 (E)	59.3 (E)	2.26 (E)	2.22 (E)
30	E	75 (H)	12.4 (E)	2.05 (E)	1.00 (E)	-
31	E	67 (E)	- (E)	- (E)	-	-
32	E	64 (E)	115 (E)	30.4 (E)	3.20 (E)	1.71 (E)
33	H	77 (H)	132 (E)	65.7 (E)	4.64 (E)	3.74 (E)
34	H	73 (H)	16.7 (E)	8.01 (E)	1.28 (E)	2.51 (E)
35	H	70 (E-H)	63.6 (E)	26.8 (E)	-	-
36	H	69 (E-H)	-	-	-	-
37	E	81 (H)	70.4 (E)	33.7 (E)	3.36 (E)	2.08 (E)
38	H	78 (H)	217 (E)	91.6 (E)	3.75 (E)	-
39	H	80 (H)	-	-		-
40	E	79 (H)	-	-		-

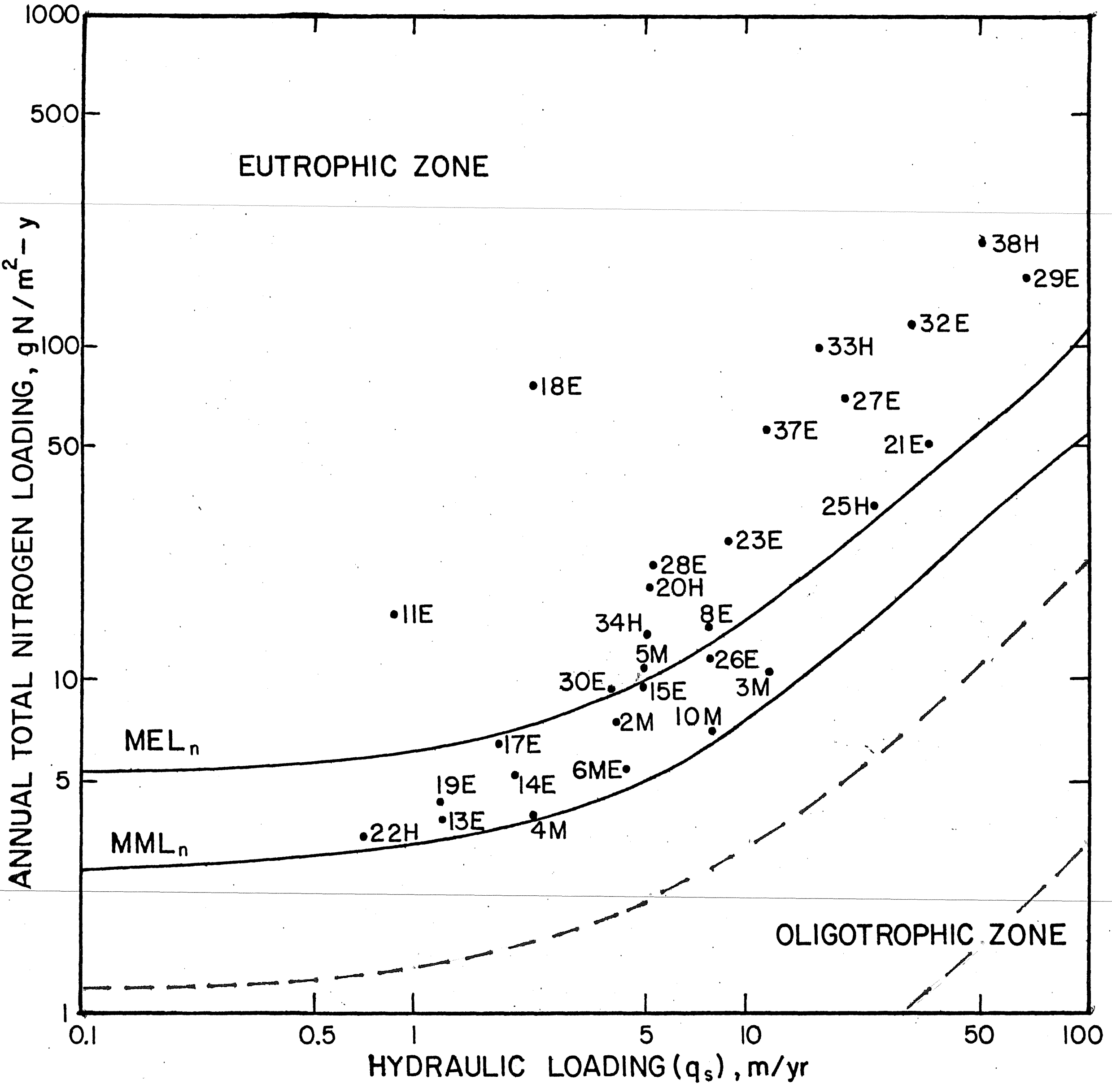


Figure V-17. Trophic state delineation of the Florida NES lakes by a modified Vollenweider (1975) model for nitrogen using loading criteria developed in this study. Dotted lines represent MEL_n and MML_n according to equations 5-39c and e (See text).

$$(N)_1 = L_n/(5.49 + q_s) \quad (5\text{-}40a)$$

$$1.0 = MEL_N/(5.49 + q_s) \quad (5\text{-}40b)$$

$$MEL_N = 5.49 + q_s \quad (5\text{-}40c)$$

$$0.5 = MML_N/(5.49 + q_s) \quad (5\text{-}35d)$$

$$MML_N = 2.75 + 0.5\ q_s \quad (5\text{-}40e)$$

These criteria, also plotted in Figure V-17, place nearly all of the mesotrophic lakes in the mesotrophic zone. However, they also result in placement of many eutrophic lakes in the mesotrophic zone. Particularly affected are lakes with low areal water loading rates (i.e., $q_s < 5$ m/yr). Since most of these eutrophic lakes are indeed nitrogen-limited (except Apopka; LC 22), the incorrect classification of these lakes seems to reflect a general weakness in the model. Specifically, the use of a uniform value for the apparent settling velocity (v_n) as a basis for estimating nutrient losses appears to have limited validity for nitrogen, as reflected by the large standard deviation associated with v_n for the NES lakes (5.5 ± 11.2 m/yr). By contrast, the estimate of v_p (although still large) was more precise (8.5 ± 8.8), and the fitted equation to predict $(P)_1$ using v_p had an r^2 value of 0.83. The weakness of this approach for nitrogen may reflect the more complex biogeochemical cycle for this element than for phosphorus.

Dillon (1975)-Type Nitrogen Model. The relationship between nitrogen loading rate and nitrogen concentration can be expressed in a form analogous to that used by Dillon (1975) to express the relationship between phosphorus loading rate and phosphorus concentration:

$$(N)_1 = L_n(1 - R_n)/q_s \quad (5\text{-}41a)$$

This equation was modified statistically in an earlier section (p. 80) to fit the Florida NES data set. The resulting equation (TN2) can be used as a basis for constructing a plot to show the relationship between nitrogen loading rate and trophic status:

$$(N)_1 = 0.899\ [L_n(1 - R_n)/q_s]^{0.976} \quad (5\text{-}41b)$$

To construct the loading plot, values of 1.0 and 0.5 mg/L were used as minimum eutrophic and mesotrophic concentrations of total nitrogen, and the term $\bar{z}/\tau_w$ was substituted for q_s. The minimum eutrophic and mesotrophic loadings can be expressed as follows:

$$(MEL_N)\cdot(1-R_n)\tau_w = 1.12\ \bar{z} \quad (5\text{-}41c)$$

$$(MML_N)\cdot(1-R_n)\tau_w = 0.55\ \bar{z} \quad (5\text{-}41d)$$

The resulting plot (Figure V-18) shows the positions of the Florida NES lakes with respect to the lines of minimum eutrophic and minimum mesotrophic nitrogen concentration. This model works well, since nearly all of

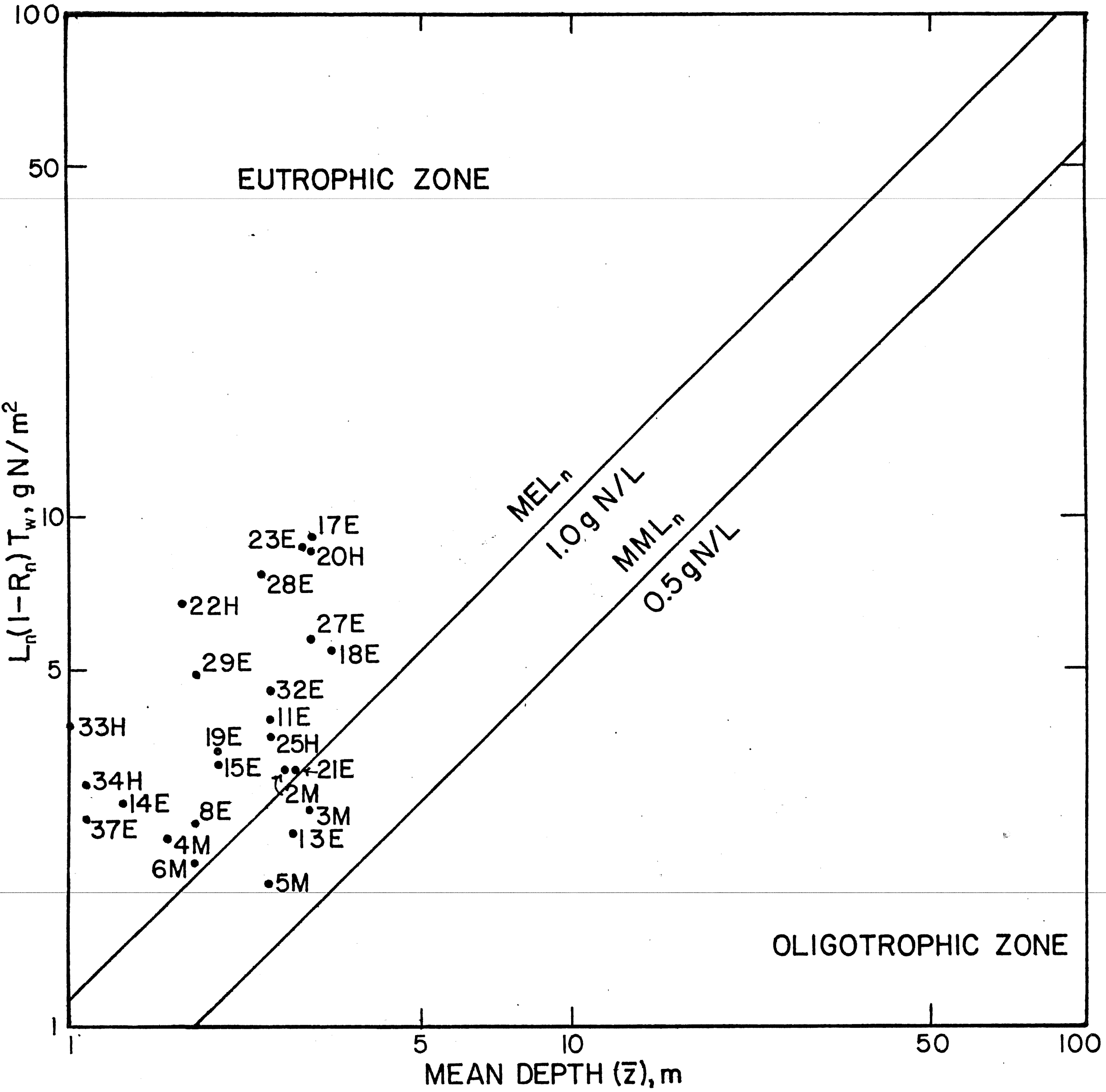


Figure V-18. Trophic state delination of the Florida NES lakes by a modified Dillon (1975) model for nitrogen using loading criteria developed in this study.

the mesotrophic lakes, but only two eutrophic lakes (both moderately eutrophic) are located in the mesotrophic zone. This model is an improvement over the Vollenweider (1975)-type nitrogen model (Figure V-17) in its placement of eutrophic lakes with low areal hydraulic loading rates ($q_s < 5$ m/yr). Several lakes this type that were located in the mesotrophic zone using the Vollenweider-type model including Marion (LC 14), Doctors (LC 17), Gibson (LC 19), and Apopka (LC 22) are located in the eutrophic zone using the Dillon-type model.

The clarity of the Dillon-type graph for nitrogen loading can be improved by dividing both sides of equations 5-36c and d by τ_w:

$$(MEL_N) \cdot (1-R_n) = 1.12\ q_s \qquad (5\text{-}41e)$$

$$(MML_N) \cdot (1-R_n) = 0.55\ q_s \qquad (5\text{-}41f)$$

The resulting graph (Figure V-19) spreads the lakes out along the abscissa but does not change the relative position of the lakes with respect to trophic state zone (cf. Figures V-18 and V-19).

APPLICATION

Predictive equations were developed in a preceding section for total nitrogen, total phosphorus and chlorophyll *a* that enable reasonably reliable estimates of these parameters from nutrient loading and hydrologic data. The best equations to predict total nitrogen (TN2) total phosphorus (TN2) are both based on Dillon and Rigler's (1974a) model, and they require data on both the inputs and the outputs of these nutrients to the lake to compute the retention coefficients. The best equation to predict chlorophyll *a* is CHA6, which is a nitrogen-based analog of the Jones and Bachman (1976) model.

Equations to predict R_n and R_p were not very successful, and even the best equations (RP6 and RN4) do not produce accurate predictions of R_p and R_n. The failure of these equations to predict R_p and R_n may reflect errors in the nutrient or water budgets or may indicate an inherently poor correlation between hydrologic and morphologic parameters and nutrient retention in Florida lakes.

The best equations for $(P)_1$, $(N)_1$, (chl *a*), R_n and R_p were plotted in Figures V-5 to V-9 together with the 95% confidence limits for individual predictions. New predictions made using equations TP2, TN2, CHA6, RP6 and RN4 should be accompanied by the confidence intervals shown in these figures to express the degree of confidence associated with those predictions.

The predictive equations described above have been developed for a group of lakes that are primarily nitrogen-limited, and their application to phosphorus-limited lakes should be considered with caution. Unfortunately there are few data on nutrient loadings for phosphorus-limited lakes in Florida, and predictive equations that are directly applicable to these lakes have not been developed.

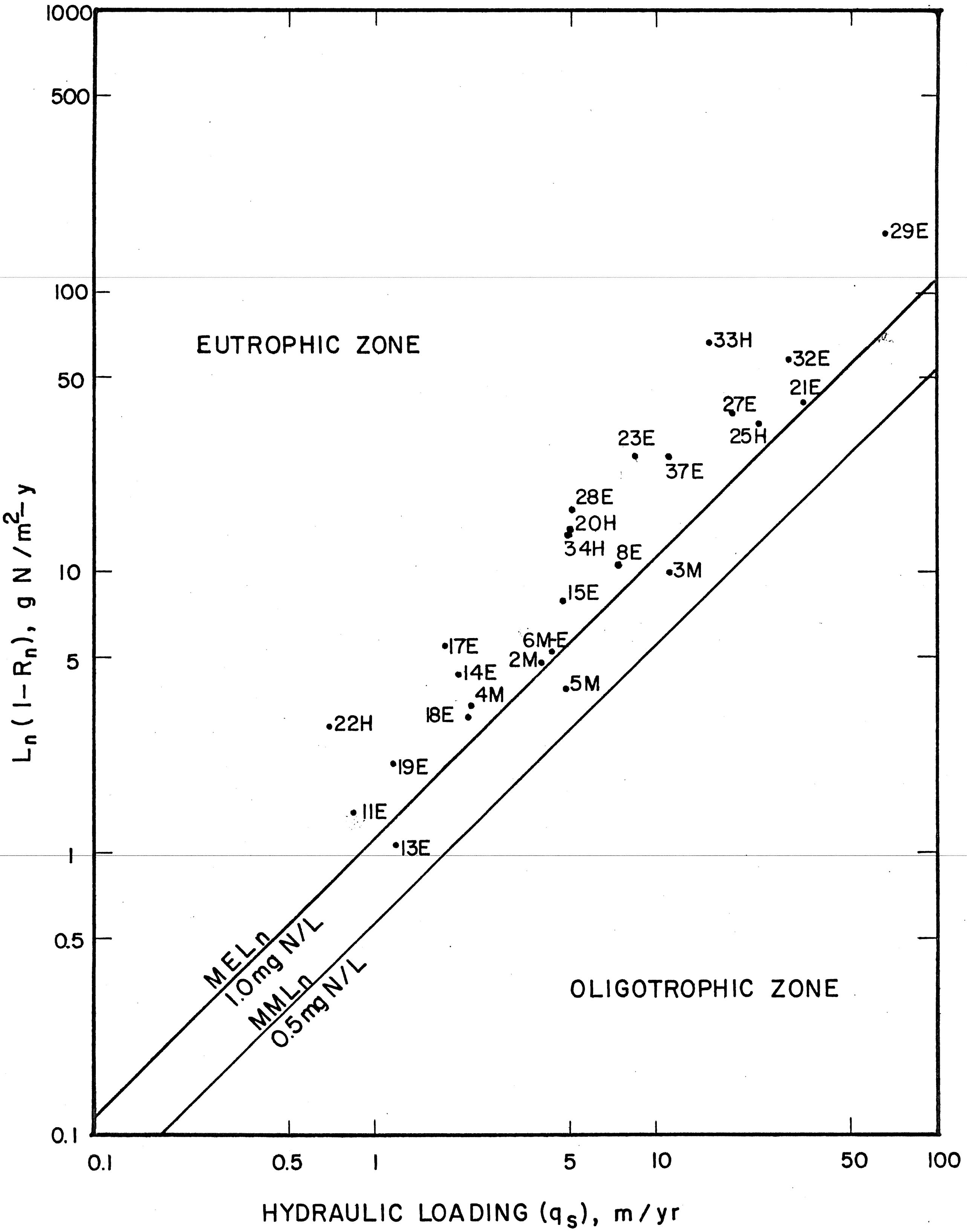


Figure V-19. Trophic state delineation of the Florida NES lakes using a further transformation of Dillon-type nitrogen model.

The nutrient loading criteria developed for Florida lakes can be used to evaluate the effects of management strategies on lake trophic status. Either the modified Vollenweider (1975) criteria (Figure V-12) or the modified Dillon (1975) criteria (Figures V-13 and V-14) can be used to evaluate phosphorus loadings, but the Vollenweider (1975) model does not require data on R_p and thus can be used when outflow phosphorus loadings are not known. For nitrogen loading, the Dillon-type loading criteria (Figures V-18 and V-19) are more valid than the Vollenweider-type criteria and should be used.

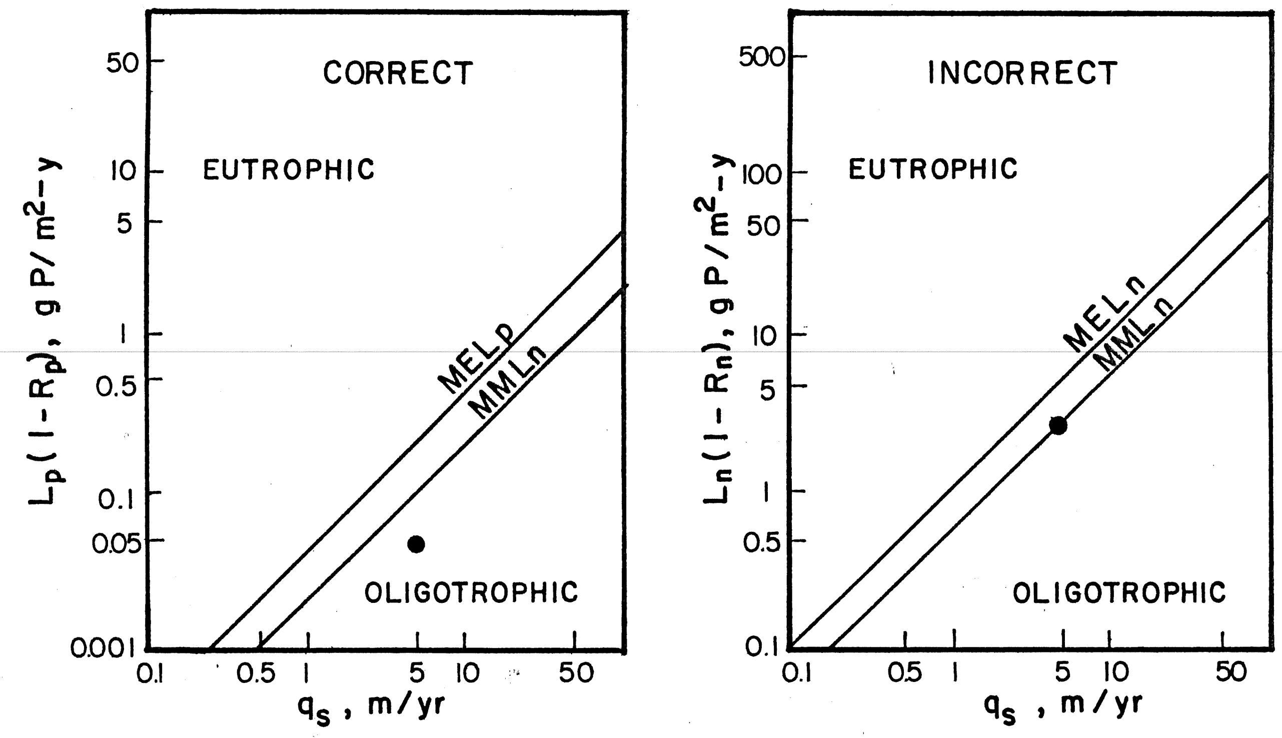
In using these loading plots, the user must decide whether to use nitrogen or phosphorus loading criteria. This should be done on the basis of the observed nutrient limitation of the lake; phosphorus loading criteria should be used for phosphorus-limited lakes and nitrogen loading criteria should be used for nitrogen-limited lakes. For lakes with mixed nutrient limitation, both nitrogen and phosphorus loading criteria should be used, since the addition of either nutrient could result in enhanced productivity.

The need to use appropriate loading criteria can be demonstrated by applying both nitrogen and phosphorus loading criteria to lakes with large nutrient imbalances, as shown in Table V-8. For the one lake with a high SIN:SRP ratio (52.0, LC 18), the trophic ratios based on nitrogen criteria are higher than the trophic ratios based on phosphorus criteria using both the modified Vollenweider (1975) model and the modified Dillon (1975) model. Conversely, for the seven lakes with very low SIN:SRP ratios, the trophic ratios based on phosphorus criteria are always higher than the trophic ratios based on nitrogen.

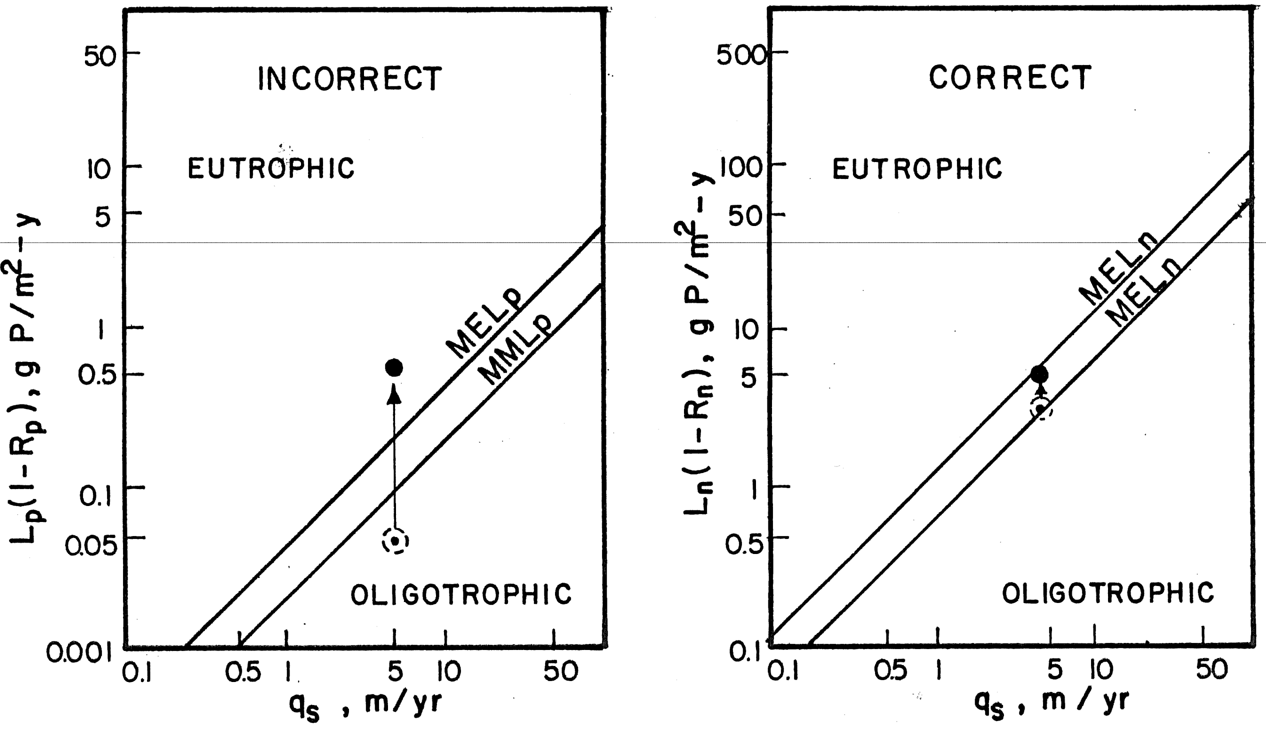
When large increases or reductions in nutrient loadings occur, the limiting nutrient may change, particularly when the altered inflow has a large nutrient imbalance. For example, treated municipal sewage is usually nitrogen-limited (See Table III-1), and the addition of treated municipal sewage to an initially phosphorus-limited lake may tend to shift the nutrient limitation towards nitrogen. In this case, phosphorus loading criteria are applicable before the addition of sewage, and nitrogen loading criteria may be applicable after the addition of sewage (see Figure V-20). In such a case, the use of phosphorus loading criteria to evaluate the effect of sewage addition would result in a prediction of greatly enhanced productivity (Figure V-20c). However, if the lake had shifted to nitrogen-limitation following this perturbation, its actual trophic status would be predicted more accurately by nitrogen loading criteria (Figure V-20d). Thus, when evaluating the effects of altered loading, both nitrogen and phosphorus loading criteria should be used, and the correct prediction should be based on the criteria that produces the <u>lower</u> trophic ratio.

Table V-8. Trophic ratios of lakes with large nutrient imbalances

LC	Average Annual Inorg-N/ SRP	Trophic Ratios: Modified Vollenweider (1975) type models		Modified Dillon (1975) type models	
		P	N	P	N
18	52.0	1.84	9.83	0.75	1.33
25	0.2	17.11	1.14	16.68	1.35
28	0.9	10.91	2.02	3.80	2.94
30	2.2	8.48	1.00	-	-
32	0.2	45.58	3.20	31.72	1.71
33	1.8	61.39	4.64	41.60	3.74
34	1.0	9.60	1.28	10.66	2.51
38	1.3	39.64	3.75	-	-



A. Prior to addition of sewage (Lp = 0.2 g P/m^2-yr.; L_n = 5.0 g N/m^2-yr). Correct trophic state is predicted in phosphorus loading plot.



B. Following addition of 5 x 10^4 kg P and 15 x 10^4 kg N from sewage (N:P =3:1), lake is still mesotrophic due to shift towards N limitation.

Figure V-20. Application of loading criteria developed in this study to a hypothetical lake before and after addition of municipal sewage. T_w = 1 yr, ≃ = 5m, q_s = 5 m/yr, R_p = 0.5, R_n = 0.4.

CHAPTER VI: SUMMARY AND CONCLUSIONS

Nonpoint source loadings of nitrogen and phosphorus from Florida watersheds can be estimated from land use characteristics using export coeffiencts obtained from the literature or using a multiple regression approach. The literature-based approach produced a wide range of export coefficients for each land use: 0.2-0.7 kg P/ha-yr and 1.5-6.1 kg N/ha-yr for forests; 0.4-2.4 kg P/ha-yr and 2-50 kg N/ha-yr for cropland; 0.2-4.7 kg P/ha-yr and 1.5-7.4 kg N/ha-yr for residential areas; 0.3-7.5 kg P/ha-yr and 3-10 kg N/ha-yr for urban areas. NPS nutrient loadings (dependent variables) and land use characteristics (indpendent variables) for 41 NES watersheds were analyzed using multiple regression analysis (stepwise deletion procedure) to improve the predictive capability of the land use-nutrient loading approach. For NPS phosphorus loading, a model that includes three land use terms (cropland, rangeland and forest) has better predictive ability than does a model that includes total drainage area as the sole independent variable ($r^2 = 0.71$ vs 0.21). Both NPS nitrogen loadings and flow were highly correlated with total drainage area ($r^2 = 0.84$ and 0.91, respectively) and the inclusion of specific land use terms as independent variables results in a very modest improvement in predictive ability. Although the use of literature-based export coefficients may be appropriate for small watersheds with a single dominant land use, the regression approach produces reasonably precise estimates of NPS loadings for larger watersheds and has the advantage that the reliability of predictions can be statistically evaluated using confidence bands.

An evaluation of the limnological characteristics of 101 Florida lakes indicates that Florida lakes are different in several important respects from temperate lakes. Most of the study lakes are shallow and well-mixed; few exhibit stable seasonal stratification or have anoxic hypolimnia. Furthermore, there is no evidence of seasonal variation in chlorophyll *a* standing crops or nutrient concentrations. Unlike temperate lakes which are usually phosphorus limited, many of the study lakes are nitrogen-limited (46% had SRP: SIN ratios < 10:1). Finally, for a given concentration of phosphorus, Florida's lakes have less chlorophyll *a* than do temperate lakes; this is true even for phosphorus-limited Florida lakes.

Carlson's trophic status index (TSI) was modified for Florida lakes by inclusion of a nitrogen index to reflect the importance of nitrogen as a limiting nutrient. A composite TSI was developed by averaging the TSI's based on Secchi disk transparency, chlorophyll *a* and nutrient concentration (the smaller of the nitrogen or phosphorus index).

Various nutrient loading models were evaluated statistically for their ability to predict chlorophyll *a*, nitrogen and phosphorus in the Florida NES lakes. Models based on a Dillon and Rigler-type loading term gave the best predictions of total nitrogen and total phosphorus ($r^2 = 0.77$ and 0.91, respectively). The best predictions of chlorophyll *a* were obtained using a Jones and Bachman-type loading term with areal nitrogen loading substituted for areal phosphorus loading ($r^2 = 0.69$). The confidence band associated with these models indicate that they produce reasonably reliable predictions of phosphorus, nitrogen and chlorophyll *a*. None of the models evaluated to predict nitrogen and phosphorus retention coefficients produce reliable estimates of these parameters.

Existing phosphorus loading criteria (Vollenweider 1968, 1975; Dillon, 1975; Shannon and Brezonik, 1972) tend to overestimate the trophic status of the Florida NES lakes. The 1975 Vollenweider criteria and the 1975 Dillon criteria were modified to improve their predictive capabilities; in both cases the revisions result in higher critical loading rates than the original criteria. The two sets of revised criteria are equally successful in delineating eutrophic lakes from mesotrophic lakes. Nitrogen loading criteria were developed using concepts analogous to those used in the development of phosphorus loading criteria. The most successful nitrogen criteria were based on a Dillon-type model; these criteria allow adequate discrimination between mesotrophic and eutrophic lakes.

In evaluating the impact of a proposed management strategy (alteration of nutrient inputs) on lake trophic status both nitrogen and phosphorus loading criteria should be used. The correct response of the lake will be obtained using the criteria that predicts the lower trophic status.

REFERENCES

Asmussen, L. E., J. M. Sheriden and C. V. Boorman, Jr. 1979: Nutrient movement from agricultural watersheds in the Georgia coastal plain. Trans. ASAE, p. 809-821.

Bachmann, R. W. and J. R. Jones. 1974. Phosphorus inputs and algal blooms in lakes. Iowa State J. Res. 49: 155-160.

Baker, L. A., V. D. Adams, J. S. Fifield, L. G. Terry, and D. Sorenson. 1981. Predicted limnology of the Ridges Basin reservoir. Utah Water Research Lab. Pub., Logan UT (in press).

Bedient, P. B., D. A. Harned and W. G. Characklis. 1978. Storm-water analysis and prediction in Houston. J. Env. Eng., ASCE 104: 1087-1100.

Bradford, W. L. and D. J. Maiero. 1978. Lake process models applied to reservoir management. J. Env. Div., ASCE 104, No. EE5: 981-986.

Brezonik, P. L. 1978. Effect of organic color and turbidity on Secchi disk transparency. J. Fish. Res. Bd. Canada. 35: 1410-1416.

Brezonik, P. L., C. D. Hendry, Jr., E. S. Edgerton, R. L. Schulze, and T. L. Crisman. 1981. Acidity, nutrients, and minerals in atmospheric precipitation over Florida: deposition patterns, mechanisms, and ecological effects. Completion rept. on grant no. 805560, U.S. EPA, Corvallis, Oregon (in press).

Brezonik, P. L., W. H. Morgan, E. E. Shannon, and H. D. Putnam. 1969. Eutrophication factors in north central Florida lakes. Publ. No. 5, Univ. Florida Water Resources Center, Gainesville, Fl., 101 p.

Brezonik, P. L., C. D. Pollman, T. L. Crisman, J. N. Allison, and J. L. Fox. 1978. Limnological studies on Lake Apopka and the Oklawaha chain of lakes. 1: Water quality in 1977. Rpt. No. ENV-07-78-01, Dept. of Env. Eng. Sci., Univ. of Florida, Gainesville, 282 p.

Brezonik, P. L. and E. E. Shannon. 1971. Trophic state of lakes in north central Florida. Publ. No. 13, Univ. of Florida Water Resources Res. Cent., Gainesville, Fl., 101 p.

Burton, T. M., R. Turner, and R. C. Harriss. 1977. Nutrient export from three north Florida watersheds in contrasting land use, pp. 323-342. In: Correll, P. L. (ed.). Watershed Research in Eastern North America, Vol. I. Tidemark Printing Co., Edgewater, Md.

Calvert, D. V. 1975. Nitrate, phosphate, and potassium movement into drainage lines under three soil management systems. J. Environ. Qual. 4: 183-186.

Campbell, K. L. 1978. Pollution in runoff from nonpoint sources. Publ. 42, Florida Water Resources Res. Cent., Univ. Florida, Gainesville, 49 p.

Carlson, R. E. 1977. A trophic state index for lakes. Limnol. Oceanogr. 22: 361-369.

Chapra, S. C. 1975. Comment on 'An empirical method of estimating the retention of phosphorus in lakes' by W. B. Kirchner and P. J. Dillon. Water Resources Res. 11: 1033-1036.

Chapra, S. C. and S. T. Tarapchak. 1976. A chlorophyll *a* model and its relationship to phosphorus loading plots for lakes. Water Resources Res. 12: 1260-1264.

Chiaudani, G. and M. Vighi. 1974. The N:P ratio and tests with *Selenastrum* to predict eutrophication in lakes. Water Research 8: 1063-1069.

CH2M Hill. 1978. Water quality studies in the Everglades Agriculture Area of Florida. Report submitted to the Florida Sugar Cane League, Clewiston, Fl.

Dillon, P. J. and F. H. Rigler. 1974a. A test of a simple nutrient budget model predicting the phosphorus content of lake waters. J. Fish. Res. Bd. Canada 31: 1771-1778.

Dillon, P. J. and F. H. Rigler. 1974b. The phosphorus-chlorophyll relationship in lakes. Limnol. Oceanogr. 19: 767-773.

Dillon, P. J. 1975. The PO_4 budget of Cameron Lake, Ontario: The importance of flushing rate to the degree of eutrophy of lakes. Limnol. Oceanogr. 20: 28-39.

Duffy, P. D., J. D. Schreiber, D. C. McClurkin and L. L. McDowell. 1978. Aqueous and sediment-phase phosphorus yields from five southern pine watersheds. J. of Environ. Qual. 7: 45-50.

Gakstatter, J. H., M. O. Allum, S. E. Dominiquez, and M. R. Crouse. 1978. A survey of phosphorus and nitrogen levels in treated municipal wastewater. J.W.P.C.F. 50: 718-722.

Goldman, C. R. 1972. The role of minor nutrients in limiting the productivity of aquatic ecosystems, p. 21-33. In: Likens, G. E. (ed.), Nutrients and eutrophication: the limiting nutrient controversy, American Society of Limnology and Oceanography special symposium, Allen Press, Lawrence, Kansas.

Grubbs, F. E. 1969. Procedures for detecting outlying observations in samples. Technometrics 11: 1-21.

Haith, D. A. 1980. Models for the analysis of agricultural nonpoint source pollution. Collaborative Paper CP-80-27, Internat. Inst. Appl. Systems Analysis, Laxenburg, Austria, 39 p.

Hand, J. G. 1975. Water quality modeling of the Middle St. Johns River System. M.S. thesis, Univ. of Florida, Gainesville, 125 p.

Heaney, J. P., W. C. Huber, and S. J. Nix. 1976. Storm water management model level I: preliminary screening procedure. Tech. Rept. EPA-600/2-76-275. U.S. EPA, Washington, D.C., 77 p.

Hendry, C. D., P. L. Brezonik, and E. S. Edgerton. 1981. Atmospheric deposition of nitrogen and phosphorus in Florida. In Atmospheric pollutants in natural waters, S. J. Eisenreich (ed.) Ann Arbor Science Publ., Inc., Ann Arbor, Mich. (in press).

Huber, W. C. and P. L. Brezonik. 1980. Assessment of projected water quality in man-made lakes and waterways of unit 24 and unit 30 developments near Marco Island, Florida. Rept. to Deltona Corp., Miami, Fl. 91 p.

Hutchinson, G. E. 1957. Treatise on limnology, vol 1. J. Wiley, New York, 1013 p.

Imboden, D. M. 1974. Phosphorus model of lake eutrophication. Limnol. Oceanogr. 19: 297-304.

Jones, J. R. and R. Bachman. 1976. Prediction of phosphorus and chlorophyll levels in lakes. J.W.P.C.F. 48: 2176-2182.

Kautz, R. S. 1981. Effects of eutrophication on fish communities of Florida lakes. Unpublished MS, Florida Game and Fresh Water Fish Commission, Tallahassee, 24 p.

King, D. L. 1972. Carbon limitation in sewage lagoons, p. 98-105. In: Likens, G. E. (ed.), Nutrients and eutrophication: the limiting nutrient controversy, American Society of Limnology and Oceanography Special Symposium, Allen Press, Lawrence, Kansas.

Kirchner, W. B. and P. J. Dillon. 1975. An empirical method of estimating the retention of phosphate in lakes. Water Resources Res. 11: 182-84.

Kratzer, C. R. 1979. Application of input-output models to Florida lakes. MS thesis, Dept. Env. Eng. Sci., Univ. of Florida, Gainesville, 169 p.

Kratzer, C. R. and P. L. Brezonik. 1981. A Carlson-type trophic state index for nitrogen in Florida lakes. Water Res. Bull. 17(4): (in press).

Lamonds, A. G. 1974. Chemical and biological quality of Lake Dice at Eustis, Florida, with emphasis on the effects of storm runoff. Rept. Water Resources Invest. 36-74. U.S.G.S., Washington, D.C.

Larson, D. P. and H. T. Mercier. 1975. Lake phosphorus loading graphs: an alternative. National Eutrophication Survey Working Paper # 174. U.S. EPA, Corvallis, Oregon.

Lee, G. F. (chairman). 1966. Report of the Nutrient Sources of Lake Mendota. Report by the Nutrient Sources Subcommittee, Technical Committee of the Lake Mendota Problems Committee. Univ. Wisconsin, Madison, 41 p.

Lie, G. B. Phosphorus cycling by freshwater macrophytes--the case of Shagawa lake. Unpublished manuscript, Limnol. Res. Center, Univ. Minnesota, Minneapolis. 56 p.

Loehr, R. C. 1974. Characteristics and comparative magnitude of non-point sources. J.W.P.C.F. 46: 1849-1872.

Lutz, J. R. 1977. Water quality and nutrient loadings of the major inflows from the Everglades Agricultural Area to the conservation areas, southeast Florida. Tech. Publ. 77-6, South Florida Water Manag. Dist., West Palm Beach, Fl., 40 p. + appendices.

MacGill, R. A., S. E. Gatewood, C. Hutchinson, and D. D. Walker. 1976. Final report on the special project to prevent eutrophication of Lake Okeechobee. Fla. Dept. of State Planning DSP-BCP-36-76, Tallahassee, Fl.

Mattraw, H. C. and C. B. Sherwood. 1977. Quality of stormwater runoff from a residential area, Broward County, Florida. J. Res. U.S.G.S. 5: 823.

Messer, J. J., P. L. Brezonik and B. R. Snyder. 1979. Denitrification in Lake Okeechobee. Rept. No. 07-79-03 to Fla. Wat. Manag. Dist.; Dept. Env. Eng. Sci., Univ. of Florida, Gainesville, 135 p.

Miller, R. A., H. C. Mattraw, Jr., and J. Hardee. 1979. Stormwater-runoff data for a commercial area, Broward County, Florida. U.S.G.S. Open File Rept. 79-982, 127 p.

Miller, W. E., T. E. Maloney, and J. C. Greene. 1974. Algal productivity in 49 lakes as determined by algal assays. Water Research 8: 667-679.

Miller, W. E., J. C. Greene, and T. Shiromaya. 1975. Application of algal bioassays to define the effects of wastewater effluents upon algal growth in multiple use river systems, pp. 71-92. _In_: Middlebrooks, E. J., D. H. Falkenborg and T. E. Maloney (eds.) Biostimulation and Nutrient Assessment. Water Research Lab PRWG 168-1. Utah State Univ., Logan, UT.

Miller, W. E., J. C. Greene, and T. Shiromaya. 1978. The _Selenastrum capricornutum_ Printz Algal Assay Bottle Test: experimental design, application and data interpretation protocol. EPA-600/9-78-018. U.S. EPA, Corvallis, Oregon, 126 p.

National Eutrophication Survey. 1975. National Eutrophication Survey methods, 1973-1976. Working Paper No. 175. U.S. EPA, Corvallis, Oregon.

National Eutrophication Survey. 1977. National Eutrophication Survey Working Papers No. 224, 225 and 243-280. U.S. EPA, Corvallis, Oregon.

Neter, J. and W. Wasserman. 1974. Applied linear statistical methods. Richard D. Irwin, Inc., Homewood, Ill. 842 p.

Odum, H. T. 1953. Dissolved phosphorus in Florida waters. Rept. Invest. 9 Pt. I. Fla. Geol. Surv. Tallahassee, 39 p.

Omernik, J. M. 1976. The influence of land use on stream nutrient levels. Ecol. Res. Ser. EPA-600/3-76-014. U.S. EPA, Corvallis, Oregon, 106 p.

Omernik, J. M. 1977. Non-point source-stream nutrient level relationships: a nationwide study. Ecol. Res. Ser. EPA-600/3-77-105. U.S. EPA, Corvallis, Oregon, 151 p.

Patalas, K. and A. Salki. 1973. Crustacean plankton and eutrophication of lakes in the Okanagan Valley, British Colombia. J. Fish. Res. Bd. Canada 30: 519-542.

Pollman, C. D. and P. L. Brezonik. 1981. A model for nutrient recycling from sediments in shallow lakes. Paper presented at 44th ann. meeting Amer. Soc. Limnol. Oceanogr., Milwaukee, Wis.

Pollman, C. D., T. L. Crisman, P. L. Brezonik, and P. Sacco. 1980 Limnological studies on Lake Apopka and the Oklawaha chain of lakes 3: Water quality in 1979. Rept. No. ENV-07-80-02, Dept. Env. Eng. Sci., Univ. Florida, Gainesville, 109 p.

Porcella, D. B. and A. B. Bishop. 1975. Comprehensive management of phosphorus water pollution. Ann Arbor Science, Ann Arbor, Mich., 303 p.

Reckhow, K. H. 1977. Phosphorus models for lake management. Ph.D. dissertation, Harvard U., Cambridge, Mass.

Reckhow, K. H., N. Beaulac, and J. T. Simpson. 1980. Modeling phosphorus loading and lake response under uncertainty: a manual and compilation of export coefficients. (Draft). Dept. Resource Development, Michigan State U., East Lansing, Mich., 214 p.

Riekerk, H., S. A. Jones, L. A. Morris, and D. A. Pratt. 1978. Hydrology and water quality of three small lower coastal plain forested watersheds. Proc. Soil Crop Sci. Soc. Fla. 38: 105-111.

Ritter, W. F., R. P. Eastburn and J. P. Jones. 1979. Nonpoint source pollution from coastal plain soils in Deleware. Trans. A.S.A.E. 22: 1044-1053.

Sakamoto, M. 1966. Primary production by phytoplankton community in some Japanese lakes and its dependence on depth. Arch. Hydrobiol. 62: 1-28.

SAS Institute. 1979. Statistical Analysis System (SAS) users manual.

Schulze, R. L. 1980. The biotic response to acid precipitation in Florida lakes. M.S. thesis Dept. Env. Eng. Sci., Univ. Florida, Gainesville. 146 p.

Shannon, E. E. and P. L. Brezonik. 1972a. Limnological characterstics of north and central Florida lakes. Limnol. Oceanogr. 17: 97-110.

Shannon, E. E. and P. L. Brezonik. 1972b. Eutrophication analysis: a multivariate approach. J. San. Eng. Div., A.S.C.E. 98: 37-57.

Shannon, E. E. and P. L. Brezonik. 1972c. Relationships between lake trophic state and N and P loading rates. Environ. Sci. Technol. 6: 719-725.

Shapiro, J. 1975. The current status of lake trophic indices-a review. Interim rept. No. 15, Limnol. Res. Center, Univ. Minnesota, Minneapolis.

Shapiro, J. 1979. The importance of trophic level interactions to the abundance and species composition of algae in lakes. Contribution No. 218, Limnological Research Center, U. of Minnesota, Minneapolis, 25 p.

Smith, C. S. 1978. Phosphorus Uptake by Roots and Shoots of *Myriophyllum spicatum L.* Ph.D. Dissertation, Dept. of Botany, U. of Wisconsin, Madison, 113 p.

Snedecor, G. W. and W. G. Cochran. 1967. Statistical methods (6th ed.), Iowa State U. Press, Iowa, 593 p.

Snodgrass, W. J. and C. R. O'Melia. 1975. Predictive model for phosphorus in lakes. Environ. Sci. Technol. 9: 937-944.

Stewart, E. H., L. H. Allen, Jr., and D. V. Calvert. 1978. Water quality of streams on the Upper Taylor Creek watershed, Okeechobee County, Fla. Proc. Soil and Crop Sci. Soc. Fla. 37: 117-120.

Terman, G.L. and L. H. Allen. 1979. Leaching of soluble and slow-release N and K fertilizers from Lakeland sand under grass and fallow. Proc. Soil Crop Sci. Soc. Fla. 30: 130-140.

Tuschall, J. R., T. L. Crisman, P. L. Brezonik, and J. N. Allison. 1979. Limnological studies on Lake Apopka and the Oklawaha chain of lakes 2: water quality in 1978. Rept. No. Env. Eng. Sci., Univ. Florida, Gainesville, Fl., 114 p.

U.S. Environmental Protection Agency. 1978. *Draft Environmental Impact Statement: Lake Apopka Restoration Project, Lake and Orange Counties, Florida.* Prepared by the USEPA Region IV office, Atlanta. 191 p.

Uttormark, P. D., J. D. Chapin, and K. M. Green. 1974. Estimating nutrient loadings to lakes from non-point sources. Ecol. Res. Ser. EPA-660/3-74-020. U.S. EPA, Corvallis, Oregon, 112 p.

Uttormark, P. D. and M. L. Hutchins. 1978. Input/output models as decision criteria for lake restoration. Tech. Compl. Rept. C-7232, Land and Water Resources Institute, Univ. Maine, Orono, 61 p.

Vollenweider, R. A. 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters with particular reference to nitrogen and phosphorus as factors in eutrophication. Tech. Rept. DAS/CSI/68.27. OECD, Paris, 159 p. (Revised 1971).

Vollenweider, R. A. 1969. Possibilities and limits of elementary models concerning the budget of substances in lakes. *Arch. Hydrobiol.* 66: 1-36

Vollenweider, R. A. 1975. Input-output models with special reference to the phosphorus loading concept in limnology. Schweiz. Z. Hydrol. 37: 53-84.

Vollenweider, R. A. 1976. Advances in defining critical loading levels for phosphorus in lake eutrophication. Mem. Ist. Ital. Idrobiol. 33: 53-83.

Walker, W. W., Jr. 1977. Some analytical methods applied to lake water quality problems. Ph.D. dissertation, Harvard U., Cambridge, Mass.

Wanielista, M. P., Y. A. Yousef, and W. M. McLellon. 1977. Nonpoint source effects on water quality. J.W.P.C.F. 49: 441-451.

APPENDICES

Table A-1. Limnological data for study lakes (mean values).

Lake Code	[chl *a*] µg/L	S.D. m	Turbidity N.T.U.	pH	Alk. mg/L as $CaCO_3$	D.O. mg/L	Cond. µmho/cm	Color C.P.U.	Total P mg P/L	SRP mg P/L	TKN mg N/L	NH_4^+ mg N/L	NO_3^- + NO_2^- mg N/L
1	3.3	2.4	.	8.00	7.17	10.0	85.3	.	0.020	0.011	0.687	0.046	0.029
2	5.2	1.5	.	7.33	6.63	12.0	119.3	.	0.040	0.015	1.007	0.049	0.040
3	8.7	1.6	.	8.27	8.07	41.0	182.7	.	0.051	0.031	0.813	0.043	0.041
4	7.8	1.1	.	7.67	7.00	10.0	73.7	.	0.045	0.011	0.848	0.063	0.050
5	6.9	2.5	.	7.60	6.33	10.0	1597.0	.	0.035	0.022	0.794	0.047	0.032
6	6.6	0.9	.	7.50	7.43	10.0	114.0	.	0.044	0.014	1.207	0.068	0.069
7	25.4	1.5	.	8.77	8.63	110.0	301.7	.	0.028	0.016	1.369	0.089	0.098
8	24.1	0.9	.	7.60	7.50	19.3	127.7	.	0.037	0.008	1.652	0.085	0.081
9	26.3	1.2	.	7.70	8.20	38.3	189.3	.	0.053	0.015	0.955	0.053	0.049
10	12.1	1.0	.	6.43	7.50	10.0	96.7	.	0.049	0.027	1.068	0.073	0.052
11	26.6	1.0	.	6.40	7.47	13.7	207.7	.	0.061	0.014	1.122	0.067	0.061
12	23.2	0.9	.	6.00	8.53	78.3	1085.0	.	0.072	0.035	1.557	0.073	0.057
13	14.4	0.7	.	7.83	8.50	129.0	621.0	.	0.066	0.018	1.948	0.072	0.146
14	29.9	0.8	.	8.77	8.80	34.7	127.7	.	0.048	0.021	2.231	0.126	0.142
15	10.2	0.7	.	6.33	7.40	18.7	372.3	.	0.073	0.037	1.411	0.077	0.077
16	6.5	0.8	.	5.53	7.73	40.0	483.7	.	0.083	0.060	1.858	0.088	0.108
17	27.1	0.9	.	6.73	7.97	48.3	2022.3	.	0.085	0.037	1.593	0.064	0.097
18	36.7	0.8	.	7.87	8.23	51.3	215.3	.	0.035	0.011	1.937	0.258	0.176
19	21.2	0.8	.	7.60	7.90	19.0	149.3	.	0.175	0.070	1.378	0.072	0.076
20	48.7	0.4	25.8	9.94	9.17	120.5	315.5	87	0.386	0.131	4.403	0.167	0.063
21	9.5	1.0	.	4.83	6.70	13.7	64.3	.	0.094	0.033	0.767	0.159	0.143
22	47.0	0.2	26.1	10.04	9.20	130.6	307.2	63	0.358	0.051	4.939	0.222	0.099
23	39.8	0.6	18.0	9.12	8.90	102.2	255.8	27	0.200	0.022	3.124	0.154	0.057
24	27.7	1.2	.	4.67	7.30	16.0	116.3	.	0.128	0.064	1.297	0.120	0.073
25	37.7	0.9	.	6.70	8.30	77.0	266.0	.	0.711	0.530	1.512	0.060	0.053
26	102.0	0.7	.	8.17	8.80	117.3	500.7	.	0.232	0.027	2.498	0.087	0.098
27	34.6	0.7	.	6.20	7.53	52.3	1003.0	.	0.142	0.075	1.662	0.091	0.079
28	30.6	0.7	.	6.60	7.90	24.0	168.0	.	0.342	0.245	1.917	0.094	0.106
29	14.8	0.6	.	5.60	7.43	39.7	953.7	.	0.199	0.143	1.952	0.187	0.125
30	97.9	0.4	.	10.15	9.47	86.0	373.0	.	0.692	0.251	4.682	0.143	0.198
31	70.2	0.9	.	8.23	9.23	81.3	349.0	.	0.484	0.318	2.403	0.089	0.089
32	54.1	0.9	.	9.60	8.87	66.0	252.3	.	1.258	1.139	1.664	0.082	0.215
33	208.6	0.5	.	14.93	10.40	80.0	334.7	.	0.631	0.256	4.457	0.093	0.390
34	76.5	0.3	.	9.70	8.87	73.3	792.7	.	0.505	0.322	2.720	0.133	0.153
35	87.7	0.7	.	7.33	8.37	51.3	170.3	.	0.753	0.554	2.718	0.168	0.151
36	77.0	0.7	.	5.67	7.67	30.3	198.3	.	1.072	0.933	2.833	0.230	0.317
37	84.9	0.1	.	5.50	7.93	42.7	231.0	.	2.750	0.254	5.463	0.200	1.063
38	140.3	0.4	.	12.03	9.30	64.3	226.7	.	1.907	1.157	5.433	1.257	0.483
39	261.4	0.3	.	1.33	7.93	151.0	340.3	.	1.447	0.816	4.997	0.260	0.197
40	276.6	0.4	.	5.60	9.13	87.0	348.3	.	1.483	0.943	5.562	0.238	0.845
41	5.4	2.2	2.4	8.65	6.16	1.5	52.3	65	0.036	0.004	0.646	0.107	0.042
42	4.5	1.4	3.0	8.14	5.48	0.8	52.4	208	0.036	0.006	0.880	0.156	0.095
43	6.7	1.8	2.0	7.70	6.16	1.6	43.4	90	0.033	0.010	0.808	0.078	0.040
44	5.8	1.4	2.7	8.34	6.16	2.0	50.2	162	0.041	0.010	0.726	0.104	0.021
45	20.9	0.9	5.1	7.98	7.24	5.2	60.6	133	0.171	0.024	1.463	0.216	0.033
46	6.5	0.7	4.4	7.14	5.92	2.0	44.0	216	0.054	0.006	0.921	0.110	0.096
47	2.7	2.2	1.7	7.24	5.22	0.2	38.8	19	0.023	0.007	1.407	0.070	0.017
48	47.9	0.8	4.2	8.05	8.06	68.5	165.8	62	0.081	0.007	1.839	0.076	0.011
49	7.3	0.9	3.1	7.98	6.30	3.0	49.6	289	0.125	0.060	1.120	0.166	0.090
50	10.7	1.4	2.8	7.72	6.11	3.0	48.5	162	0.071	0.030	1.053	0.116	0.080
51	7.8	1.5	3.3	8.26	5.98	1.8	42.4	163	0.044	0.013	0.948	0.056	0.116

Table A-1 (continued).

Lake Code	[chl *a*] µg/L	S.D. m	Turbidity N.T.U.	pH	Alk. mg/L as $CaCO_3$	D.O. mg/L	Cond. µmho/cm	Color C.P.U.	Total P mg P/L	SRP mg/L	TKN mg N/L	NH_4^+ mg N/L	NO_3^- + NO_2^- mg N/L
52	7.7	1.0	5.1	8.00	5.24	0.6	54.0	241	0.104	0.051	0.551	0.090	0.019
53	2.3	1.6	2.3	8.52	6.40	2.0	37.8	39	0.013	0.005	0.666	0.048	0.011
54	17.8	0.9	5.4	9.18	7.90	17.8	84.8	130	0.059	0.002	1.511	0.138	0.013
55	14.0	0.9	4.5	8.61	7.55	15.1	74.4	120	0.062	0.007	1.270	0.130	0.065
56	11.5	1.1	2.1	5.78	4.76	0.0	25.2	125	0.039	0.003	1.180	0.074	0.000
57	38.2	0.6	4.8	8.32	7.50	7.0	59.3	235	0.104	0.011	1.743	0.305	0.060
58	38.3	0.6	2.1	2.05	5.10	1.6	52.0	390	0.123	0.045	1.112	0.296	0.116
59	20.3	0.7	1.5	3.97	5.57	1.0	43.0	539	0.178	0.098	1.918	0.410	0.060
60	87.7	0.4	18.0	7.42	8.25	119.6	305.1	94	0.461	0.150	2.245	0.369	0.012
61	4.5	1.9	3.3	8.32	7.74	28.4	99.2	45	0.039	0.009	1.070	0.182	0.055
62	5.8	1.5	2.1	6.81	7.86	163.0	541.3	46	0.857	0.439	0.561	0.057	0.021
63	44.8	0.6	11.0	9.59	8.84	107.8	256.1	48	0.521	0.225	1.987	0.149	0.044
64	34.0	0.8	5.9	7.50	8.04	37.0	131.4	137	0.374	0.123	1.481	0.174	0.018
65	2.8	1.8	1.8	8.86	8.10	42.4	99.8	42	0.038	0.006	0.795	0.042	0.018
66	20.7	2.0	1.0	3.58	5.50	1.8	41.2	304	0.124	0.059	0.820	0.090	0.070
67	27.0	1.5	3.1	2.92	6.06	5.8	50.0	135	0.297	0.116	0.793	0.082	0.131
68	29.6	0.5	7.5	5.96	7.72	58.2	133.8	139	0.431	0.099	2.915	0.246	0.030
69	13.6	1.0	5.9	8.46	4.72	0.0	36.6	156	0.051	0.012	1.014	0.086	0.069
70	7.6	1.0	2.4	5.15	5.05	0.0	21.0	254	0.052	0.003	1.080	0.075	0.010
71	57.1	0.6	5.5	5.48	6.86	11.6	61.2	370	0.535	0.186	2.324	0.360	0.181
72	32.9	0.9	4.6	8.41	7.79	15.0	64.9	63	0.167	0.034	1.899	0.189	0.025
73	8.5	1.0	2.5	4.40	6.60	11.0	49.5	435	0.315	0.127	1.110	0.035	0.030
74	14.5	1.0	8.8	9.20	8.48	73.4	203.6	15	0.039	0.009	1.535	0.150	0.042
75	23.0	0.8	16.4	9.14	8.90	92.6	252.6	36	0.184	0.018	3.296	0.144	0.026
76	6.3	2.0	2.2	8.77	7.14	10.6	134.4	8	0.024	0.007	1.054	0.171	0.031
77	1.7	4.5	1.4	8.22	6.90	5.8	52.9	17	0.014	0.002	0.399	0.101	0.058
78	1.5	3.4	2.3	9.24	5.72	1.0	28.0	7	0.013	0.003	0.271	0.026	0.081
79	1.9	3.9	2.1	8.20	5.50	0.6	25.2	13	0.013	0.003	0.318	0.042	0.086
80	3.0	3.7	2.5	9.06	5.76	0.6	30.2	6	0.013	0.004	0.915	0.096	0.060
81	1.5	3.3	2.4	9.15	6.42	1.0	48.1	8	0.017	0.003	0.554	0.039	0.179
82	1.6	3.2	1.9	8.74	5.92	1.5	44.4	3	0.014	0.005	0.295	0.060	0.015
83	4.3	1.3	2.4	6.76	5.36	0.6	40.4	178	0.028	0.008	0.916	0.122	0.046
84	1.6	6.2	1.7	8.78	5.26	0.2	35.0	8	0.016	0.006	0.224	0.070	0.063
85	5.1	0.8	1.7	6.32	4.68	0.0	37.4	538	0.035	0.011	0.916	0.116	0.058
86	2.4	4.8	1.1	7.74	5.06	0.0	35.9	2	0.017	0.005	0.373	0.051	0.023
87	3.6	3.5	1.2	8.51	4.81	0.0	38.0	2	0.018	0.003	0.449	0.150	0.093
88	4.0	0.9	2.2	7.11	4.89	0.1	44.1	397	0.036	0.007	0.872	0.100	0.051
89	1.4	4.3	2.2	9.02	5.00	0.0	50.0	9	0.012	0.005	0.185	0.028	0.040
90	2.6	3.6	1.8	7.70	6.40	2.2	39.6	21	0.023	0.006	0.663	0.086	0.068
91	2.1	3.9	2.3	7.20	5.44	0.4	46.4	5	0.013	0.007	0.295	0.046	0.005
92	1.8	3.2	2.3	8.78	5.76	1.0	38.6	21	0.018	0.005	0.411	0.034	0.011
93	1.5	5.5	1.4	8.14	5.75	0.0	34.5	7	0.008	0.004	0.260	0.014	0.012
94	12.7	1.8	1.8	7.72	6.71	12.4	146.5	19	0.024	0.006	0.532	0.038	0.017
95	10.7	1.4	2.4	8.32	6.79	11.9	125.0	32	0.021	0.008	0.458	0.030	0.020
96	2.5	1.4	2.4	8.21	5.24	0.0	23.2	31	0.014	0.007	0.262	0.045	0.007
97	8.5	0.7	4.9	9.20	6.14	4.4	93.3	154	0.068	0.027	0.781	0.070	0.026
98	9.3	2.0	2.0	7.75	6.67	11.5	129.0	21	0.014	0.007	0.539	0.020	0.026
99	3.2	1.1	2.9	8.55	6.50	5.4	142.8	28	0.058	0.015	0.326	0.037	0.009
100	7.2	2.4	1.7	8.47	6.53	5.4	69.8	15	0.013	0.005	0.431	0.090	0.016
101	0.9	7.9	0.6	8.71	5.14	0.0	18.6	5	0.007	0.004	0.189	0.043	0.006

Table A-2. Morphometric data for study lakes.

Lake Code	Volume $\times 10^6 m^3$	Area, Km^2	Z_{max}, m	$\bar{Z}$, m
1	20.63	7.64	2.7	4.3
2	133.11	49.30	2.7	4.5
3	1.17	0.39	3.0	3.9
4	48.80	30.50	1.6	3.0
5	25.63	10.25	2.5	4.6
6	201.73	112.07	1.8	2.7
7	58.29	15.95	3.7	4.6
8	254.52	141.40	1.8	2.4
9	1.54	0.77	2.0	3.3
10	0.46	0.38	1.2	2.4
11	7.25	2.90	2.5	4.3
12	4.01	4.46	0.9	1.0
13	5293.00	1890.71	2.8	4.7
14	16.67	12.82	1.3	3.0
15	141.22	70.61	2.0	3.4
16	26.40	17.60	1.5	2.5
17	38.88	12.96	3.0	4.0
18	46.56	14.11	3.3	4.3
19	3.84	1.92	2.0	2.7
20	67.11	22.37	3.0	4.9
21	122.36	43.70	2.8	15.0
22	161.36	124.12	1.3	3.4
23	95.47	39.78	2.4	4.3
24	3.24	0.72	4.5	8.4
25	8.35	3.34	2.5	4.3
26	4.99	1.85	2.7	
27	558.48	186.16	3.0	4.5
28	182.88	76.20	2.4	4.0
29	63.90	35.50	1.8	5.8
30	18.29	18.29	1.0	1.5
31	14.07	4.69	3.0	5.2
32	4.00	1.60	2.5	4.0
33	1.36	1.36	1.0	1.5
34	35.20	32.00	1.1	2.7
35	2.05	1.37	1.5	2.5
36	0.61	0.41	1.5	2.7
37	0.69	0.63	1.1	1.8
38	1.54	1.03	1.5	2.0
39	0.41	0.41	1.0	1.5
40	1.83	1.22	1.5	2.7
42	22.42	4.67	4.8	7.0
43	0.92	0.27	3.4	5.2
44	7.96	2.21	3.6	5.8
45	1.89	0.86	2.2	3.4
46	0.40	0.27	1.5	2.4
47	0.07	0.05	1.5	3.0
48	0.56	0.20	2.8	4.3
49	9.27	3.31	2.8	4.3
50	0.93	0.29	3.2	4.6
51	1.87	0.52	3.6	5.8
52	1.92	0.64	3.0	4.9
53	0.05	0.05	1.1	1.5
54	73.75	25.43	2.9	3.7
55	80.96	44.98	1.8	3.0
56	0.03	0.04	0.8	1.2
57	44.83	29.89	1.5	4.0
58	0.04	0.01	4.0	25.3
59	0.13	0.11	1.2	2.1
60	0.08	0.04	1.9	3.4
61	0.03	0.02	1.6	2.4
62	0.26	0.29	0.9	1.5
63	1.14	0.76	1.5	1.9
64	0.06	0.04	1.6	2.7
65	0.06	0.06	1.0	1.8
66	0.06	0.02	3.1	6.1
67	0.19	0.05	3.8	7.0
68	0.57	0.82	0.7	0.9
69	3.19	2.13	1.5	3.4
70	0.06	0.05	1.2	1.8
71	0.48	0.22	2.2	2.7
72	3.84	1.01	3.8	5.2
73	3.22	2.48	1.3	1.8
74	234.36	55.80	4.2	7.6
75	123.61	30.15	4.1	5.2
76	144.96	23.01	6.3	9.8
77	48.62	6.66	7.3	22.9
78	24.384	5.08	4.8	9.1
79	6.640	0.83	8.0	14.0
80	14.478	2.54	5.7	10.4
81	28.372	6.92	4.1	8.8
82	10.896	2.27	4.8	9.4
83	3.591	1.71	2.1	2.7
84	3.402	0.42	8.1	13.4
85	1.435	0.41	3.5	5.2
86	0.120	0.06	2.0	3.7
87	0.100	0.05	2.0	4.6
88	1.175	0.47	2.5	3.7
89	3.536	1.04	3.4	6.4
90	4.420	0.85	5.2	8.8
91	8.880	2.40	3.7	8.8
92	1.190	0.34	3.5	5.8
93	2.905	0.35	8.3	20.0
94	5.880	1.47	4.0	8.0
95	6.264	2.16	2.9	8.0
96		1.40		4.0
97	9.880	4.94	2.0	3.0
98	84.180	14.03	6.0	12.0
99		1.91		3.0
100	106.320	13.29	8.0	17.0
101	0.399	0.07	5.7	17.0

Table A-3. Hydrologic and nutrient loading data for the Florida NES lakes.

Lake Code	τ_w yr	q_s m/yr	P_{in} kg/yr	P_{out} kg/yr	R_p	N_{in} kg/yr	N_{out} kg/yr	R_n
1	0.351	7.70	2050	1540	0.25	52345	125420	-1.40
2	0.677	3.99	35730	5025	0.86	363650	228820	0.37
3	0.266	11.29	245	185	0.24	4020	3855	0.04
4	0.721	2.22	3560	2260	0.37	116980	100050	0.14
5	0.518	4.83	4645	1830	0.61	110820	38520	0.65
6	0.425	4.24	33615	25570	0.24	595380	567575	0.05
7	1.501	2.43	.	.	.	.	.	.
8	0.241	7.47	68810	36865	0.46	1983505	1478835	0.25
9	.	.	2500	.	.	7265	.	.
10	.	.	.	.	.	.	.	.
11	2.901	0.86	1980	165	0.92	45155	4080	0.91
12	.	.	.	.	.	.	.	.
13	2.329	1.20	517210	58010	0.89	6970160	1953590	0.72
14	0.644	2.02	1860	1065	0.43	63540	54375	0.14
15	0.419	4.77	67325	33775	0.50	661810	549230	0.17
16	.	.	.	.	.	.	.	.
17	1.699	1.77	9915	2295	0.77	82085	70110	0.15
18	1.499	2.20	9240	1010	0.89	1066300	45680	0.96
19	1.699	1.18	1665	420	0.75	8040	3970	0.51
20	0.605	4.95	41015	12450	0.70	423890	312365	0.26
21	0.082	34.07	262440	139420	0.47	2205015	1735985	0.21
22	2.477	0.69	78810	35925	0.54	417730	354670	0.15
23	0.340	8.54	50315	29225	0.42	908380	913825	-0.01
24	.	.	.	.	.	.	.	.
25	0.107	23.40	60110	56055	0.07	110035	117245	-0.07
26	0.351	7.70	3310	2830	0.15	21810	41635	-0.91
27	0.159	18.88	624870	561005	0.10	12675785	6865460	0.46
28	0.471	5.09	372265	63410	0.83	1631585	1269745	0.22
29	0.027	65.70	525075	543740	-0.04	5723790	5760010	-0.01
30	0.263	3.80	62945	.	.	170365	.	.
31	.	.	.	.	.	.	.	.
32	0.082	30.42	93610	66375	0.29	183600	92470	0.50
33	0.063	15.87	67100	38605	0.42	134930	89725	0.34
34	0.225	4.90	135590	71750	0.47	427090	438290	-0.03
35	.	.	21425	.	.	83235	.	.
36	0.036	42.12	.	17910	.	.	59790	.
37	0.099	11.15	9460	2830	0.70	35210	16220	0.54
38	0.030	49.77	78500	.	.	213655	.	.
39	.	.	.	.	.	.	.	.
40	.	.	.	.	.	.	.	.

Table A-4. Data used in the development of land-use/nutrient loading relationships for the Florida NES watersheds.

Land use characteristics

Tributary Code	ALLURB	CPAST	OAG	FOR	RA	NFWET	WA	SMINE	DA, Km^2	TPL, kg/yr	TNL, kg/yr	FLOW, m^3/sec
02A1	0.356	0.225	0.048	0.006	0.196	0.103	0.063	0.003	223.5	12695	60375	1.60
03B1	0.622	0.000	0.087	0.084	0.117	0.000	0.076	0.000	2.6	115	1750	0.06
04A1	0.057	0.065	0.465	0.050	0.012	0.284	0.040	0.027	143.4	1185	62365	1.39
05A1	0.090	0.046	0.493	0.221	0.083	0.000	0.050	0.000	8.6	210	6215	0.08
06B1	0.057	0.257	0.166	0.104	0.191	0.147	0.078	0.000	978.1	17510	352090	10.39
08D1	0.022	0.264	0.001	0.028	0.506	0.088	0.091	0.000	254.3	2900	96105	2.30
09A1	0.318	0.069	0.215	0.039	0.049	0.000	0.295	0.000	7.1	1525	4195	0.07
13C1	0.037	0.700	0.020	0.008	0.130	0.104	0.000	0.000	289.0	38560	196570	2.87
13D1	0.101	0.694	0.000	0.023	0.081	0.101	0.000	0.000	115.7	5640	69625	1.01
13F1	0.000	0.663	0.017	0.027	0.203	0.089	0.000	0.000	386.3	7190	103345	1.52
13G1	0.021	0.386	0.064	0.047	0.395	0.086	0.003	0.000	496.5	26315	331740	5.96
14D1	0.004	0.343	0.192	0.295	0.143	0.000	0.007	0.000	8.5	130	5625	0.08
15B1	0.024	0.082	0.015	0.563	0.000	0.299	0.016	0.000	726.5	30680	455320	7.90
15C1	0.002	0.286	0.000	0.247	0.000	0.461	0.004	0.000	100.1	13340	50410	0.80
19B1	0.048	0.296	0.605	0.024	0.000	0.000	0.000	0.000	1.6	205	1045	0.02
20A1	0.055	0.223	0.295	0.041	0.000	0.052	0.333	0.001	381.7	35920	354670	3.50
20B1	0.748	0.000	0.018	0.162	0.000	0.000	0.010	0.000	1.1	20	445	0.02
21B1	0.000	0.000	0.000	0.935	0.000	0.065	0.000	0.000	74.1	615	55395	1.40
21D1	0.003	0.075	0.000	0.896	0.000	0.027	0.000	0.000	26.6	1435	17730	0.50
23A1	0.159	0.302	0.198	0.103	0.000	0.198	0.040	0.000	12.9	215	4640	0.06
25A1	0.243	0.388	0.216	0.065	0.000	0.054	0.018	0.017	139.5	49940	89850	2.40
26B1	0.516	0.072	0.196	0.062	0.103	0.000	0.010	0.000	1.0	115	1245	0.01
26C1	0.626	0.102	0.046	0.059	0.138	0.000	0.014	0.000	18.5	895	12090	0.24
28A1	0.330	0.098	0.095	0.025	0.068	0.325	0.058	0.000	472.9	113285	351980	2.67
28B1	0.125	0.621	0.055	0.000	0.125	0.043	0.031	0.000	26.1	6405	19945	0.31
29B1	0.020	0.146	0.001	0.446	0.333	0.000	0.075	0.000	15.8	375	11735	0.20
30A2	0.122	0.217	0.076	0.166	0.062	0.075	0.222	0.060	56.0	12705	25540	0.60
30B1	0.314	0.064	0.189	0.100	0.000	0.027	0.054	0.253	41.6	11800	24510	0.70
30C1	0.573	0.037	0.063	0.031	0.002	0.028	0.261	0.005	53.9	5230	14015	0.30
32A1	0.785	0.000	0.037	0.000	0.000	0.025	0.153	0.000	57.2	55110	58885	1.30
34A1	0.695	0.009	0.036	0.119	0.024	0.054	0.061	0.000	33.8	10990	35510	0.35
34B1	0.294	0.057	0.311	0.148	0.000	0.098	0.085	0.008	55.0	1470	18710	0.22
34C1	0.214	0.189	0.100	0.264	0.000	0.179	0.055	0.000	20.5	5725	26360	0.25
34D1	0.385	0.150	0.171	0.117	0.000	0.077	0.100	0.001	124.0	43435	100320	2.67
34E1	0.016	0.132	0.003	0.721	0.127	0.000	0.001	0.000	10.6	3750	14290	0.20
35A1	0.339	0.242	0.109	0.230	0.055	0.000	0.012	0.000	1.7	205	2215	0.02
35B1	0.765	0.000	0.000	0.179	0.007	0.000	0.000	0.000	1.5	470	2005	0.02
35C1	0.415	0.016	0.000	0.420	0.131	0.000	0.000	0.000	3.8	755	4725	0.10
37B1	0.445	0.049	0.000	0.366	0.105	0.000	0.004	0.000	5.4	6100	18225	0.12
37C1	0.578	0.421	0.000	0.000	0.000	0.000	0.000	0.000	0.4	150	1195	0.01
38B1	0.458	0.051	0.000	0.441	0.000	0.029	0.008	0.012	156.5	30440	81545	1.75

*ALLURB = urban + residential, CPAST = cropland and pasture, OAG = other agriculture, FOR = forest, RA = rangeland, NFWET = non-forested wetlands, WA = open water, SMINE = strip mines and other barren land. (See Chapter II.) All land use data as fraction of total watershed area. DA = drainage area, TPL and TNL = non-point source phosphorus and nitrogen loading.