

IMPACT OF PHOSPHORUS REMOVAL AT THE DANBURY, CONNECTICUT SEWAGE TREATMENT PLANT ON WATER QUALITY IN LAKE LILLINONAH

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Abstract. Lake Lillinonah is a highly eutrophic impoundment on the Housatonic River receiving 20 to 30% of its P load from domestic wastewaters from Danbury, Connecticut. In 1977 P removal from the domestic wastewater treatment plant was initiated, reducing the P load from the Danbury sewage treatment plant from 13 700 to 6700 kg/yr in 1976 and 1977, respectively. The flows of the Housatonic River in 1976 and 1977 were also dissimilar; in 1977 about twice as much water entered Lake Lillinonah as in the former year.

The OECD eutrophication study results were used to estimate the impact of flow and altered P loads on Lake Lillinonah's eutrophication-related water quality. It was found that the average planktonic algal chlorophyll concentration would have been reduced from 35 $\mu\text{g/L}$ to 30 $\mu\text{g/L}$ if the flows of the Housatonic River would have been the same in the two years of the study.

1. Introduction

In the summer of 1977, P removal by iron precipitation was initiated at the Danbury, Connecticut wastewater treatment plant (STP), a trickling filter plant serving about 35,000 people (Cervera, 1977). The primary purpose of this activity was to demonstrate the technical and economic feasibility of this approach in Connecticut.

The wastewaters from the Danbury sewage treatment plant are discharged into a brook which enters the Still River. Within a few kilometers, the Still joins the Housatonic River about 0.2 km above Lake Lillinonah (see Figure 1). Lake Lillinonah is a long, narrow (16 x 0.5-1 km) impoundment, created for electric power generation on the Housatonic River in western Connecticut. Some of its general physical characteristics are presented in Table I. This lake is generally considered to be highly eutrophic with frequent obnoxious blooms of blue-green algae occurring during the summer months. From data available, it appears that algal growth in the lake is generally P-limited during that period. There is, however, some evidence for N or other limitation of algal growth at some times (State of Connecticut, 1975). Since the Danbury STP has been estimated to contribute 24 to 30% of the total P load to Lake Lillinonah (US EPA, 1975 and State of Connecticut, 1975, respectively), it is of interest to determine the impact of the P removal at the Danbury STP on the water quality in Lake Lillinonah.

The FMC Corporation conducted limnological surveys of Lake Lillinonah and Lake Zoar, an impoundment just downstream from Lake Lillinonah, and their major tributaries during the summers of 1976 and 1977. These surveys provided data that should be useful to evaluate the effect of removal at the Danbury STP on water quality in Lakes

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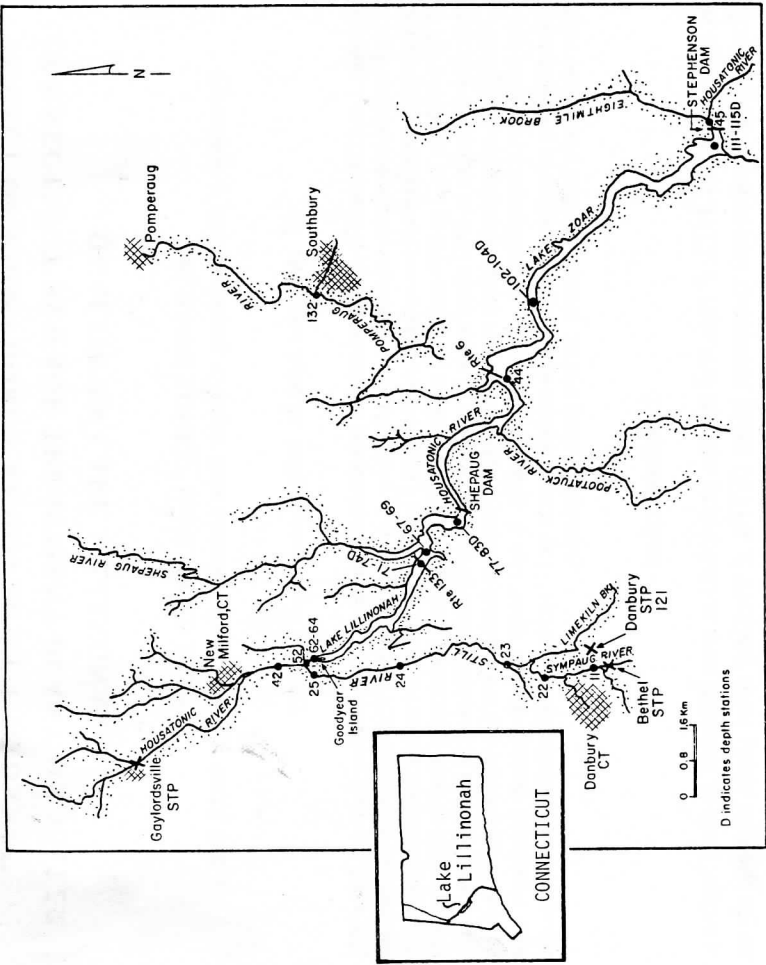


Fig. 1. Study area and sampling sites (After Smith and Brown, 19768).

Lillimonah and Zoar. Examination of these data shows that there was an apparent improvement in Lake Lillimonah water quality between the two years which could possibly be related to P removal at the Danbury STP.

This paper presents an independent assessment of the results of the FMC study (Smith and Brown, 1978) employing some of the recently developed Vollenweider-OECD eutrophication modeling techniques to predict and evaluate changes in water quality that have occurred or would have been expected to occur as a result of altering the P load to the lake. The majority of this work considers the time period April 28 through September 30, because corresponding concentration and flow data were available for this period for both 1976 and 1977. Due to the complex system and insufficient data for some parameters, the water quality in only Lake Lillimonah will be discussed. The impact of the P removal at Danbury would be expected to be greater in this lake than in Lake Zoar.

2. OECD Eutrophication Modeling Approach

About ten years ago, the Organization for Economic Cooperation and Development (OECD) initiated a eutrophication study program which had as its primary purpose, the

TABLE I
General characteristics of Lake Lillimonah for study period June 1–September 15, 1976 and 1977

Characteristic	1976	1977
Mean depth (m) ^a	11.9	11.9
Volume (m ³) (acre-feet)	9.13 × 10 ⁷ 74 000	9.13 × 10 ⁷ 74 000
Surface area ^a (m ²) (acres)	7.69 × 10 ⁶ 1900	7.69 × 10 ⁶ 1900
Mean flow ^b (m ³ s ⁻¹) (cfs)	36.4 1287	20.2 714
Hydraulic residence time ^c (τ ₀) (yr)	0.079	0.14

^a US EPA (1975).

^b Based on USGS (1976, 1977).

^c Calculated as a quotient of the volume divided by the mean daily flow (June 1–September 15).

formulation of algal nutrient load-lake and impoundment response relationships. This study was conducted in Western Europe, Japan, Australia, Canada, and the United States, where a total of approximately 200 waterbodies was being investigated. Within the U.S., the US EPA was the lead agency for the OECD eutrophication studies. Under contract with the US EPA, selected researchers compiled, using a standardized format, the results of their previous studies on approximately 40 U.S. waterbodies or parts thereof. These investigators prepared a series of reports which were submitted to the US EPA, covering nutrient load-response relationships for their respective waterbodies. These reports have been published (Seyb and Randolph, 1977). Rast and Lee (1978) prepared a summary report in which they critically examined each of the individual investigators' reports for their data reliability and comparability.

Rast and Lee (1978), and Lee *et al.* (1978) found that the US OECD eutrophication study waterbodies' nutrient load-response data can be formulated into relationships similar to those developed by Vollenweider (1975, 1976). Vollenweider found for a group of lakes, that the average summer chlorophyll *a* concentration was related to the P load normalized by mean depth and hydraulic residence time. Rast and Lee (1978), Lee *et al.* (1978), and Jones and Lee (1981) have shown that this relationship is applicable to a wide variety of U.S. waterbodies as well. They have also been able to extend Vollenweider's relationship to include formulation of a P load-average summer Secchi depth relationship, and a P load-average hypolimnetic oxygen depletion rate relationship. Jones

et al. (1979) have extended this work and formulated a maximum summer chlorophyll *a* concentration-average summer chlorophyll *a* concentration relationship. Based on these efforts, it is now possible to predict, for many waterbodies, the impact of altering a P load on the water quality of a lake or impoundment.

The Vollenweider-OECD eutrophication modeling approaches provide the means to estimate the eutrophication response of a lake no matter what its hydrologic characteristics. This is especially important in the case of Lake Lillinonah where the hydrologic characteristics of the Housatonic were markedly different during the two study periods. This difference in itself should have had a significant effect on water quality in Lake Lillinonah through an alteration of the hydraulic residence times of the lake. The modeling approaches applied here normalize the P loads so that the eutrophication response of Lillinonah to the altered Danbury STP P load alone can be estimated.

2.1. CALCULATION OF PHOSPHORUS LOADS

In order to estimate the impact of removal of P from the Danbury STP effluent on the water quality in Lake Lillinonah, it is necessary to calculate the total P load to the lake as well as the percentage load from the Danbury STP. To do this, the input flows and concentrations must be determined.

It would appear from Figure 1 that the P concentrations and flows at Station 52 could be used to derive the total P load to Lake Lillinonah. One means of checking this estimated load would be to sum the loads at Stations 25 and 42 (Figure 1) and compare this load to that at Station 52. Although the Still River was, at one time, gaged at Station 25, it was not gaged during the study periods. According to Smith (1977), its flow was similar to that of the Pomperaug River which is located in the same geographic area and gaged at Southbury, Connecticut (Figure 1). The Pomperaug River flows, therefore, were used where Still River flows were required. The Housatonic is gaged at Gaylordsville, Connecticut, about three miles upstream from Lake Lillinonah. These flows were used in calculating the P loads at Stations 42 and 52.

2.1.1. River Discharges

The mean daily discharges at the Gaylordsville and Southbury, Connecticut gaging stations for 1976 and 1977 were plotted over time (Figures 2 and 3). These figures indicate that the Still River makes a minor contribution to the flow of the Housatonic. Mean daily flows of the Housatonic River ranged from about 11 to 280 $\text{m}^3 \text{s}^{-1}$ (400 to over 10 000 cfs), while that estimated for the Still River generally ranged from 0.1 to 20 $\text{m}^3 \text{s}^{-1}$ (5 to about 700 cfs). At both gaging stations, during both years' studies, the discharges were highly variable, especially during the spring months. The flows were higher during the spring than in the summer. This could have had an effect on the water quality in the lake and is discussed further in a subsequent section.

The summer discharges in 1977 were markedly lower than in 1976 in both the Housatonic and Pomperaug Rivers (Figures 2 and 3). This is also demonstrated by the monthly rainfall patterns presented in Figure 4 (USGS 1976, 1977). For May, July, and

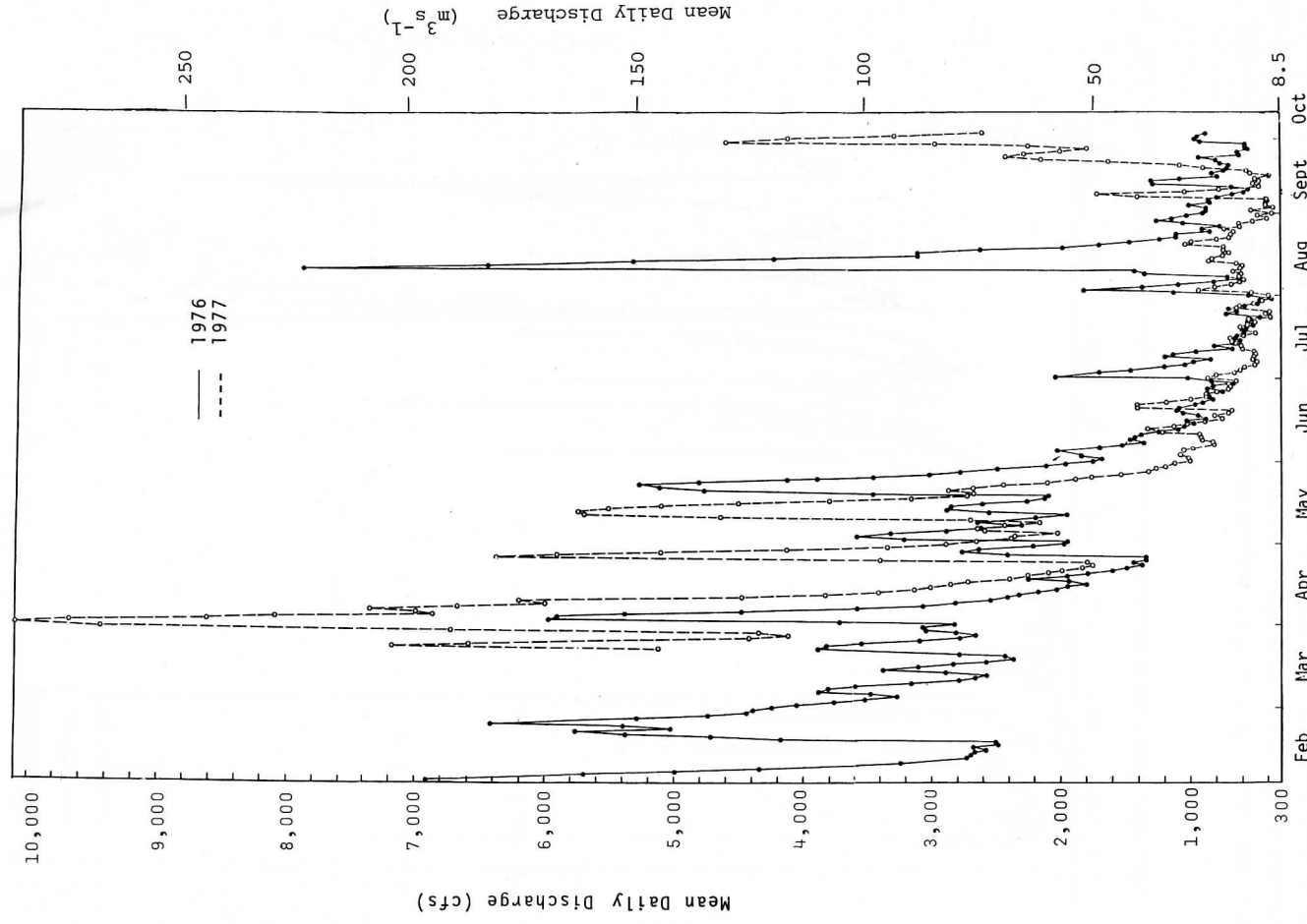


Fig. 2. Mean daily discharge — Housatonic River at Gaylordsville, CT (After USGS, 1976, 1977).

August, not only was the 1977 rainfall considerably below that of 1976, but also was considerably below 'normal'.

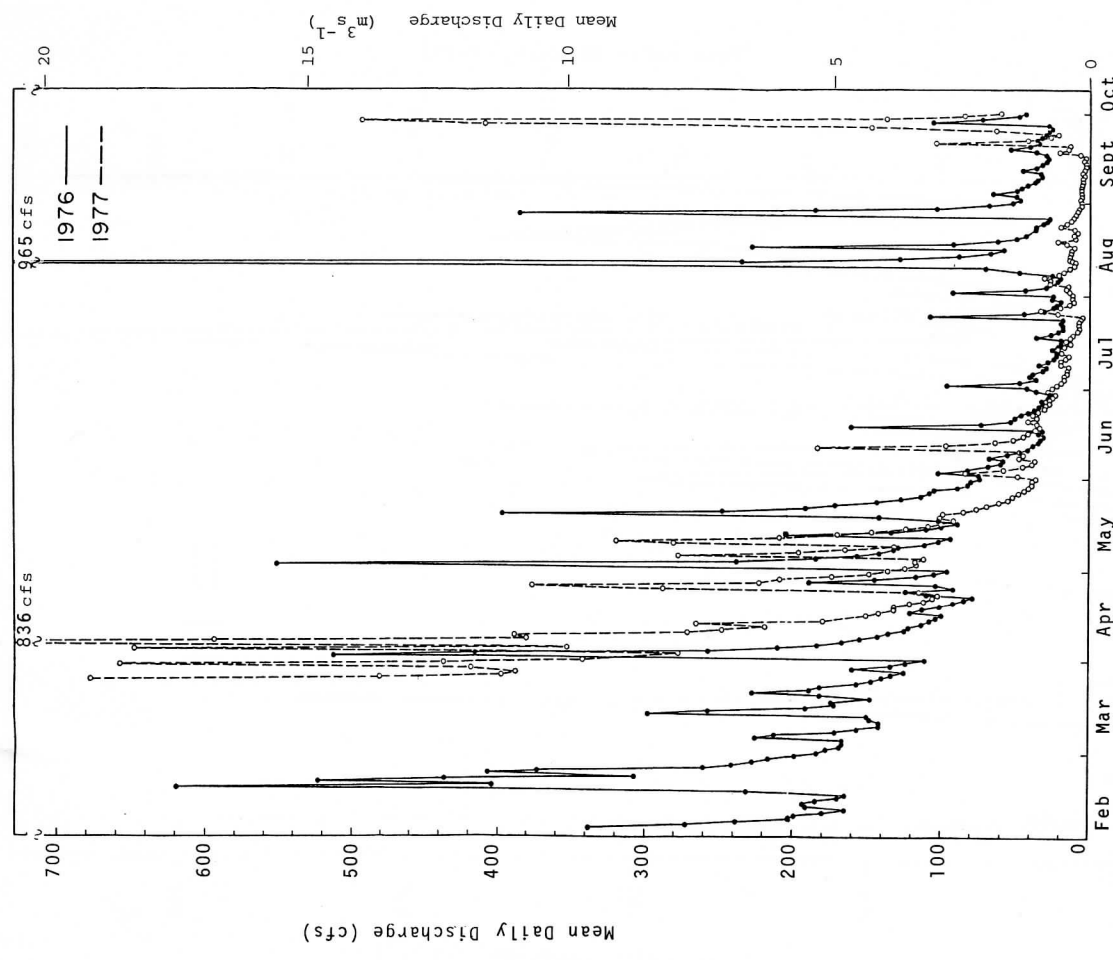


Fig. 3. Mean daily discharge - Pomperaug River at Southbury, CT (After USGS, 1976, 1977).

Figure 5 presents a comparison of the mean monthly 1976 and 1977 discharges with the overall monthly discharges found for 1966-71 and 1975 (with one standard deviation of the means). For the study period, it appears that the 1976 and 1977 flows are typically within one standard deviation of the overall mean found for previous years. May flows were higher than this average as was the August 1976 discharge.

In mid-August 1976, there was a period of high flow in the Housatonic River (Figure 3). This high flow period yielded about $4 \times 10^8 \text{ m}^3$ ($1.4 \times 10^{10} \text{ ft}^3$) of water.

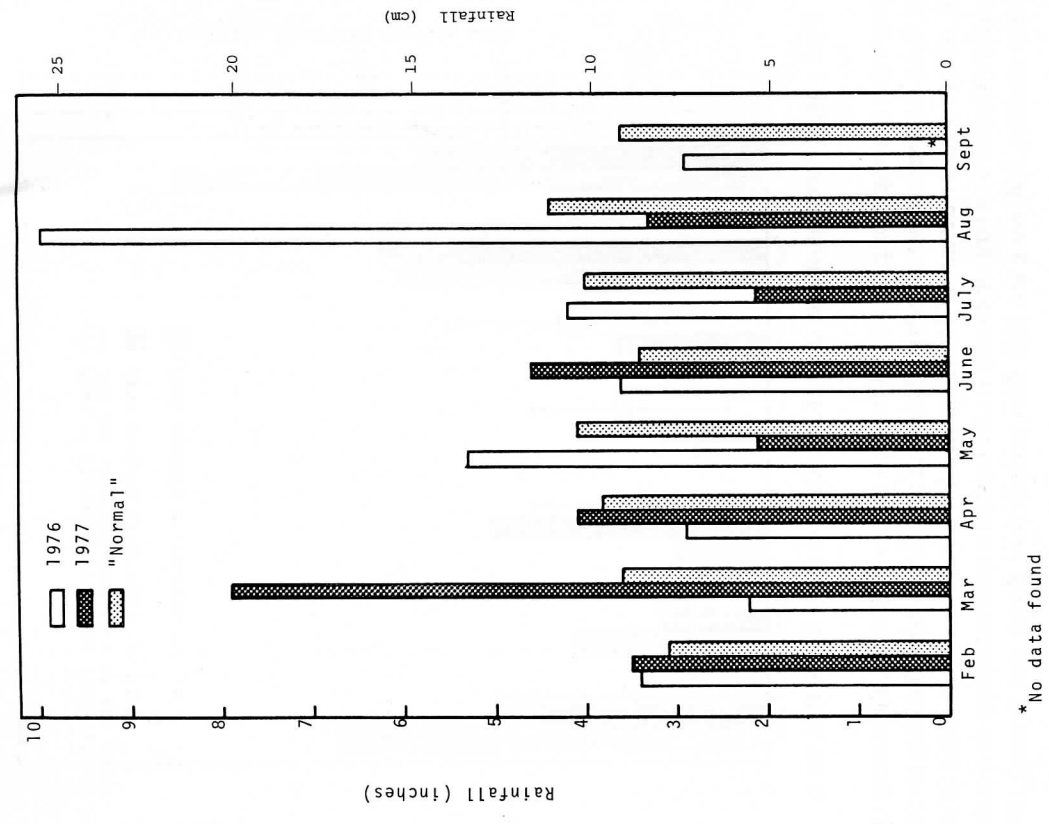


Fig. 4. 1976-77 Connecticut Rainfall (After USGS, 1976, 1977).

which was over four times the volume of Lake Lillinah. The impact of this flow can be seen in the dissolved oxygen (DO) profiles in the lake (see Smith and Brown, 1978). At midlake, the water column profiles indicated that in early August 1976, before the high flow, the DO ranged from about 8.5 mg l^{-1} in the surface waters to about 2 mg l^{-1} between 8 and 11 m depth. At the Shepaug Dam, at the lower end of the lake, the DO ranged from about 8.5 mg l^{-1} at the surface to less than 1 mg l^{-1} below 7 m depth. During the high flow period, concentrations were increased throughout the water column at these locations; DO then appeared to remain in the bottom waters through the rest of the 1976 study. A similar situation occurred in mid to late August 1977 when the flows appeared to be somewhat elevated. This could have resulted from the increased flow of

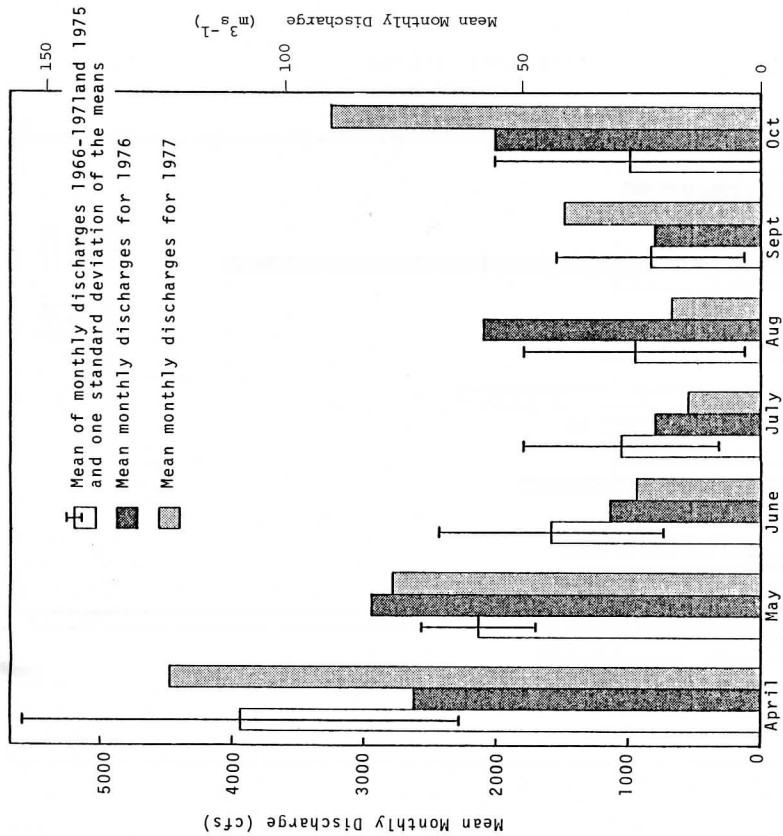


Fig. 5. Mean monthly discharges of the Housatonic River (After USGS, 1976, 1977).

incoming water moving along the lake bottom as a density current, increasing the DO content there. As seen at several in-lake stations, the high flow period during the summer of 1976 and, to a lesser extent, of 1977, appeared to result in a large influx of total P. This is also reflected in a large increase in chlorophyll *a* concentrations at a number of the in-lake stations, following the high flow periods (see Smith and Brown, 1978).

2.1.2. Phosphorus Loads

Since the flows were quite variable over time, it was inappropriate to use average tributary flow and average tributary concentrations for the study period in calculating the P loads at the various stations. Rather, daily loads should be calculated. In order to derive daily P concentrations at Stations 25, 42, and 52, the biweekly total P values were plotted over time; daily concentrations were then interpolated (Figures 6 and 7). In order to maintain the same study period for these stations for both years and, at the same time, include some of the spring flow characteristics, several values for concentrations at Station 52 were estimated by the authors. They were chosen to follow the relative patterns observed at the stations over the study period. The P load to Lake Lillinonah, based on the load at Station 52, was obtained by multiplying the interpolated, estimated, or measured total P concentration by the mean daily flow for each day during the study period

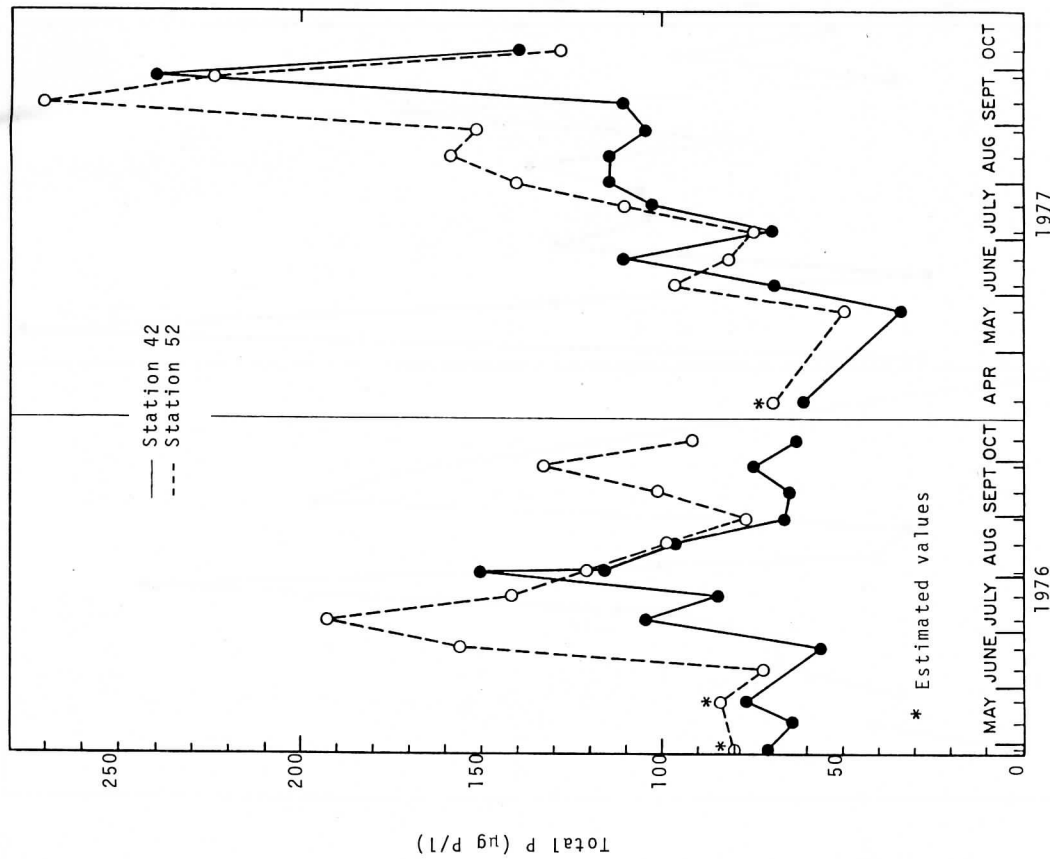


Fig. 6. Total P concentrations over time at Stations 42 and 52 - Housatonic River (After Smith and Brown, 1978).

(April 28-September 30). These daily values were summed to yield a total P load for the 156 day period. It was apparent from the short hydraulic residence times (the derivation of which is to be discussed) and the patterns of flow that the P load contributed during the spring and summer is the primary source of P which stimulates algal growth in the lake during the summer months. Phosphorus loads from fall, winter, and early spring would likely be flushed through the lake by late spring flows and, therefore, have little or no impact on summer algal growth. To account for this, and in light of the fact that the Vollenweider modeling approach requires yearly loads, the average daily load for the study period was assumed to be applicable for the entire year. The same method was

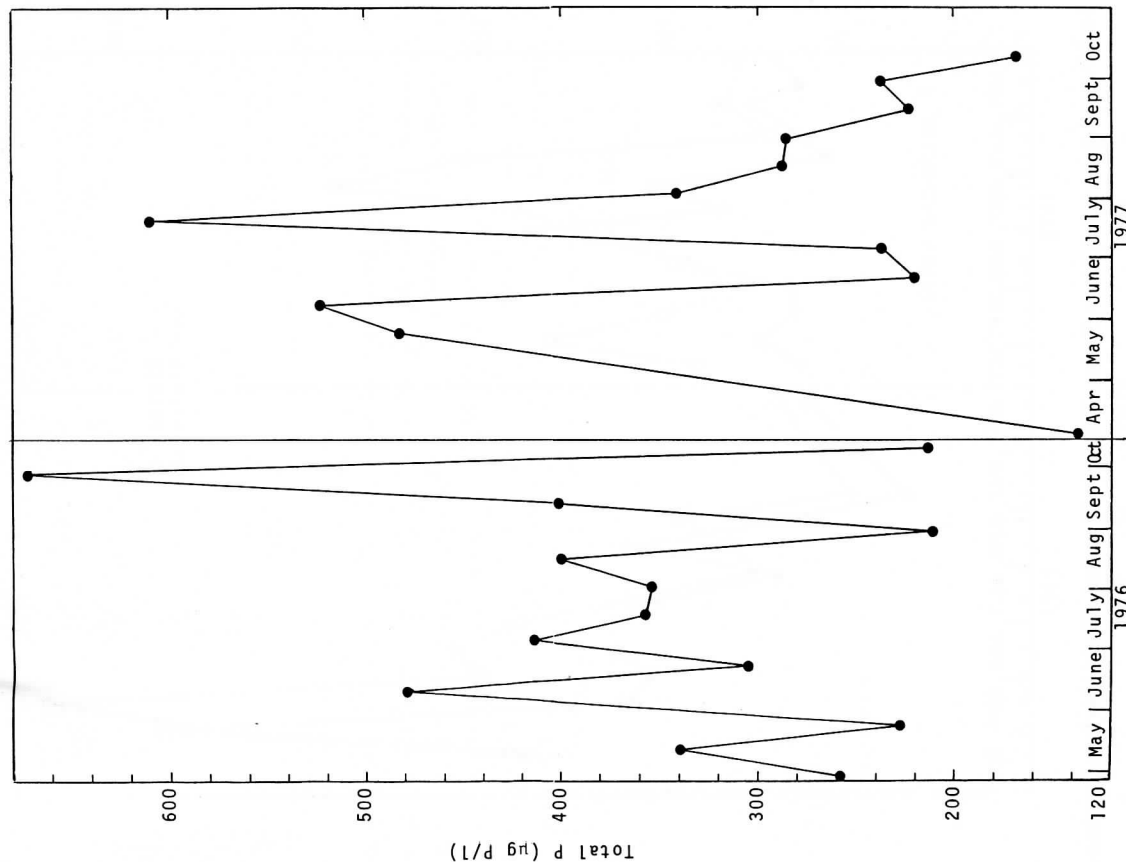


Fig. 7. Total P concentrations over time at Station 25 – Still River (After Smith and Brown, 1978).

used to calculate the P loads at Stations 25 and 42. The values for these loads are presented in Table II.

The mean monthly flows from the Danbury wastewater treatment plant for spring-summer 1976 and 1977 were made available to the authors by Cervera (1977). The biweekly values for the phosphorus concentration in the Danbury effluent, as measured as part of the FMC study, were used to calculate monthly mean effluent P concentrations. By multiplying the mean monthly flow by the mean monthly P concentration

TABLE II

Summary of Total P loads to Lake Lillinonah

Location	kg P/156 day (4/28 – 9/30)	
	1976	1977
Station 42	48 807	46 088
Station 52	61 798	56 865
Danbury STP	13 709	6 692
Station 25	10 282	7 102

by the number of days in the month, a monthly P load value was derived. The three days in April included in the study period, but for which there were no STP P data, were assumed to contribute the same load as was contributed in three days in May. A summary of these computations is presented in Table III. As indicated by comparison of the monthly P load values for 1976 and 1977, after the Danbury STP P removal was initiated, there appeared to have been an average 70% reduction in total P load from the plant. The total loads from the Danbury plant are also included in Table II for comparison with other loads.

TABLE III

Total P load from Danbury wastewater treatment plant (April 28–September 30)

Month	Mean total P ($\mu\text{g P l}^{-1}$)	Mean flow		Mean P load (kg P day $^{-1}$)	Total P load for month (kg P)
		(mgd)	(m 3 day $^{-1}$)		
1976					
April ^a	3140	7.5	2.8×10^4	88	264
May	3140	7.5	2.8×10^4	88	2 728
June	4298	7.1	2.7×10^4	116	3 480
July	3211	6.2	2.3×10^4	74	2 294
August	2870	7.7	2.9×10^4	83	2 573
September	3142	6.6	2.5×10^4	79	2 370
Total P load for study period					13 709
1977					
April ^a	4405	6.12	2.3×10^4	101	303
May	4405	6.12	2.3×10^4	101	3131
June	1880	5.65	2.1×10^4	39	1170
July	1307	5.2	2.0×10^4	26	806
August	1232	4.8	1.8×10^4	22	682
September	1163	4.4	1.7×10^4	20	600
Total P load for study period					6692

^a Daily load for the three days in April included in this study period was assumed to be the same as the mean daily load in May.

Based on Smith and Brown (1978) and Cervera (1977).

Examination of Table II shows that between 1976 and 1977 there appears to have been a substantial removal of the total P from the Danbury wastewater treatment plant effluent, in the Still River. Disregarding other loads to the river, more than 3400 kg P were removed in the river between the Danbury STP outfall and Station 25 during the study period. Actual removal was likely much greater than this because of other inputs of P to that region of the river. The 1976 load from the Danbury STP effluent was about 22% of the Housatonic load at Station 52, whereas the Still was estimated to contribute about 17% of the Station 52 load.

Removal of some portion of total P load from the Danbury plant in the Still River during the 1977 study period cannot be demonstrated by the calculated loads at Station 25 and the Danbury effluent during the study period. It is likely, however, that some part of the Danbury P load was removed in the Still River in 1977 also. It may be expected that a greater amount of the total P load from the wastewater treatment plant would have been removed in the river during the summer of 1977 because of the apparently markedly lower flow than in 1976.

A comparison of the load calculated using Station 52 values with that calculated using the sum of values at Stations 25 and 42 shows that, based on the sampling program used and the discharge information available, there is reasonable agreement of these load estimates. For the purpose of nutrient load-response relationship development, the P load, as calculated for Station 52, was used for the load to Lake Lillingtonah.

2.2 EVALUATION OF LOADING ESTIMATES

In order to determine if the loads calculated for Lake Lillingtonah were reasonable with reference to how lake systems have been found to typically behave, the 1976 and 1977 P loads were plotted on a Vollenweider mean P/influent P-mean depth/hydraulic residence time diagram (Figure 8). Table IV presents the values from which these points were plotted. The mean P concentration [P] in the lake was determined by taking the arithmetic mean of the measured biweekly P concentrations at all stations in the lake

TABLE IV

Evaluation of loading estimates		
Parameter	1976	1977
P load (kg P day^{-1}) ^a	397	364
[P] ($\mu\text{g P l}^{-1}$)	65	68
[P̄] ($\mu\text{g l}^{-1}$)	119	136
τ_w (yr) ^b	0.08	0.14

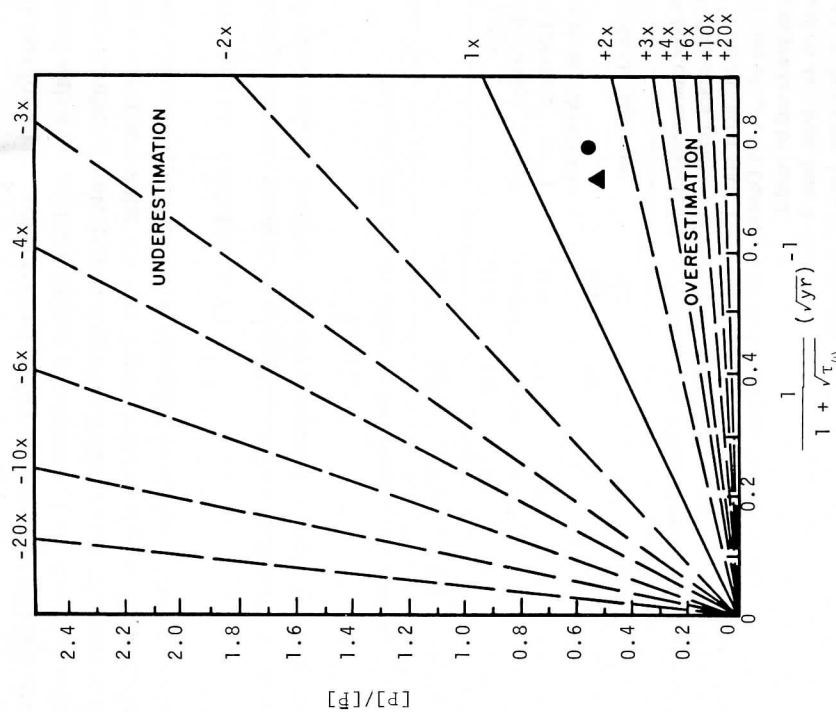
[P] = mean total P concentration in the lake (excluding unrealistic values) (April 28–September 30).

[P̄] = mean influent total P concentration (April 28–September 30).

τ_w = hydraulic residence time.

^a Based on period April 28–September 30.

^b Based on period June 1–September 15.



● 1976 load calculated using Station 52 P loads
▲ 1977 load calculated using Station 52 P loads
Fig. 8. Evaluation of estimates of Lake Lillingtonah phosphorus loadings: Vollenweider mean P/influent P and hydraulic residence.

(Smith and Brown, 1978). Influent P concentrations were derived by dividing the arithmetic mean of the concentrations at Station 52 by the mean daily flow into the lake as measured on the Housatonic for the 156 day study period.

By dividing the lake volume, which was assumed to be constant, by the mean inflow during the period June 1 through September 15, the hydraulic residence time (τ_w) was determined. This time period (June 1–September 15) was used for calculation of the hydraulic residence time because the hydraulic residence time during this period is the one which is applicable to algal growth during the time of greatest recreational use of the waterbody. Algal growth, however, is influenced by the P load entering the lake in the late spring as well as that entering during the summer. The residence time was about 29 days in the summer of 1976 and about 51 days in the summer of 1977.

The '1X' line on the diagram (Figure 8) indicates where a lake with a given hydraulic residence time should theoretically plot. It was found by Rast and Lee (1978) and Lee

et al. (1978) that for a lake which plotted within ± 2 times the $[P]/[\bar{P}]$, the P load has been estimated with an acceptable degree of accuracy. This finding was based on the fact that a large number of waterbodies obeys this Vollenweider relationship and typically falls within a factor of two of the '1X' line. Figure 8 shows that the Station 52 total P load gives a reasonable estimate of the total phosphorus load to Lake Lillinonah.

2.3. PHOSPHORUS LOADING DIAGRAM

The general, physical and chemical characteristics of Lake Lillinonah needed to plot this lake on the Vollenweider P loading curve are presented in Table V. The position of Lake

TABLE V
Characteristics of Lake Lillinonah for eutrophication modeling

Parameter	1976	1976a	1976b	1977	1977a	1977b
Mean depth, $\bar{z}$ (m)	11.9	11.9	11.9	11.9	11.9	11.9
Hydraulic residence time, τ_o (yr) ^b	0.079	0.079	0.079	0.14	0.14	0.14
Areal P loads, L(P) (g m ⁻² yr ⁻¹) ^c	18.9	15.3	15.9	17.3	19.4	19.6
Mean summer in-lake chlorophyll <i>a</i> (µg l ⁻¹) ^b	34.6	29 ^a	30 ^a	32.6	36 ^a	36.2 ^a
Mean summer Secchi depth (m)	1.1	1.2 ^a	1.2 ^a	1.6	1.5 ^a	1.5 ^a

1975a = 1976 flow characteristics; 85% of Danbury STP P load subtracted from load to lake.
1976b = 1976 flow characteristics; 70% of Danbury STP P load subtracted from load to lake.
1977a = 1977 flow characteristics; 1976 load from Danbury STP used instead of 1977 load.
1977b = 1977 load using 1976 Danbury STP loads for June–September only.
^a = Values predicted by model.
^b = Based on data from June 1–September 15.
^c = Based on data from April 28–September 30.

Lillinonah on the Vollenweider P loading curve (Figure 9) was determined for each of the following loading and hydrologic conditions: (1) 1976 P load and hydraulic residence time; (2) 1976 Housatonic P load with only 15% of the Danbury STP P load, i.e., 85% of the Danbury STP load, a percent removal which is readily achievable, was subtracted from the Station 52 load; (3) 1976 Housatonic P load with only 30% of the Danbury STP load; (4) 1977 P load and the hydraulic residence time; (5) 1977 P load substituting the 1976 load from the Danbury STP for the 1977 Danbury STP load; and (6) 1977 P load substituting the 1976 Danbury STP P loads for those months (June–September) when some P was being removed at the Danbury treatment plant. Figure 9 shows that for both 1976 and 1977, Lake Lillinonah plots above the 'excessive loading' line, in the area where many lakes classified as eutrophic plot. It also shows that a dramatic change in water quality would not be expected to result from initiation of P removal at the Danbury treatment plant. As discussed by Rast and Lee (1978) water quality is related to the vertical distance above the permissible loading line. Those lakes which plot at the greatest vertical distance above the line would have the poorest water quality. In order to interpret the differences in P load shown in Figure 9, it is necessary to utilize the approach developed by Vollenweider and evaluated by Rast and Lee (1978) of relating P load to mean summer chlorophyll *a* concentrations.

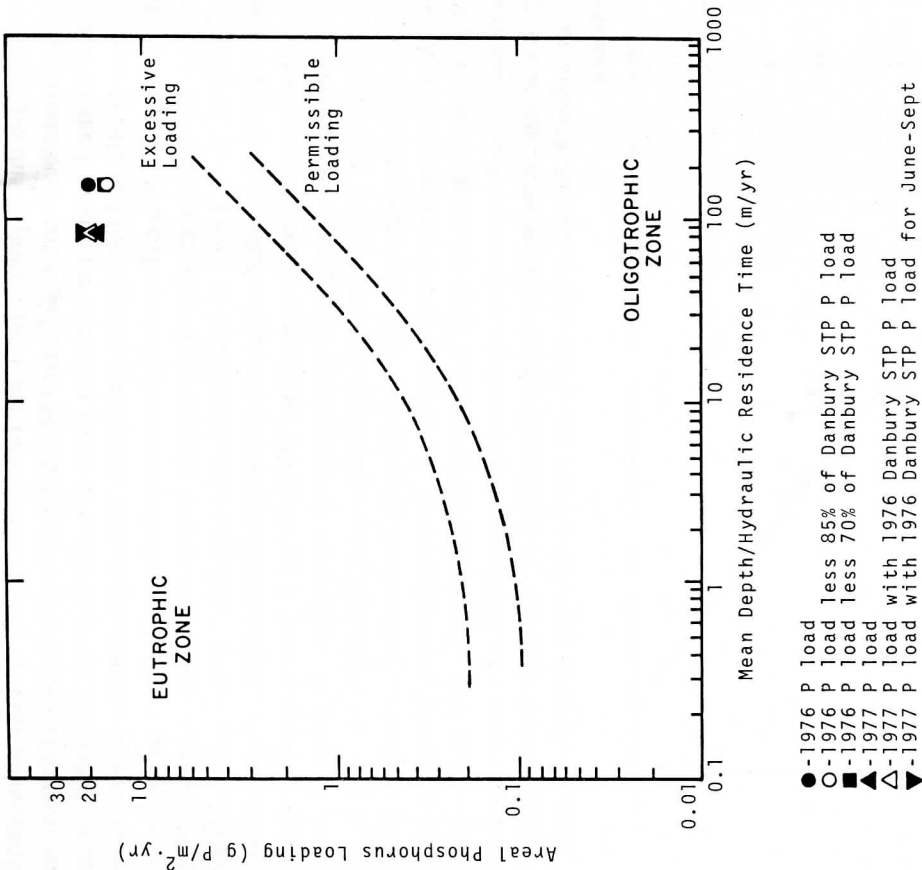


Fig. 9. Vollenweider P loading relationship - summers 1976-1977.

2.4. MODELING LAKE WATER QUALITY RESPONSE TO ALTERED P LOAD

Based on their work on about 40 US OECD waterbodies, Rast and Lee (1978) have defined a relationship between the mean summer chlorophyll *a* concentration in a lake in which planktonic algal growth is P limited, and the P load to the lake normalized by the lake's mean depth and hydraulic residence time. This relationship and approach allow the mean summer chlorophyll *a* concentration of a lake to be predicted based on the lake's P load, mean depth ($\bar{z}$), and hydraulic residence time (τ_o). (Note: In the subsequent discussion, $\bar{z}/\tau_o$ is defined as q_s .) They also can be used to predict the change in water quality, as measured by mean summer chlorophyll *a* concentration, that would be expected to result from altered P load or hydraulic residence time. It has been repeatedly demonstrated that this approach can be applied to many lakes throughout the world (Jones and Lee, 1981).

In order to use this approach for Lake Lillinnonah, a mean summer chlorophyll *a* concentration must be determined. In the FMC study, chlorophyll *a* concentrations were determined for the lake surface waters at about biweekly intervals between June and October in both 1976 and 1977 (Smith and Brown, 1978). Eight stations were monitored in 1976; four in 1977. Since the period of concern with respect to water quality in Lake Lillinnonah is the summer months when the lake is used for recreation, chlorophyll *a* values during the period early or mid-June to mid-September were used in modeling changes in water quality. Values apparently resulting from the analysis of a scum of algae on the lake surface (with levels ranging from 150 to 900 $\mu\text{g l}^{-1}$) were excluded when the arithmetic mean of the chlorophyll *a* concentrations were taken. Justification for this approach stems from the fact that the algal scum is not representative of living algal growth; it is generally composed of dead or dying algal cells which accumulate in areas of a lake depending on wind direction and speed. These scums usually last only a short time. It is not generally understood by those in the water quality field how to use the results of such analyses on surface algal scum, in a meaningful way, in assessing water quality. The arithmetic mean of the summer chlorophyll *a* concentrations in surface (1 m) waters, excluding those of the algal scum, was used as the mean summer chlorophyll *a* concentration.

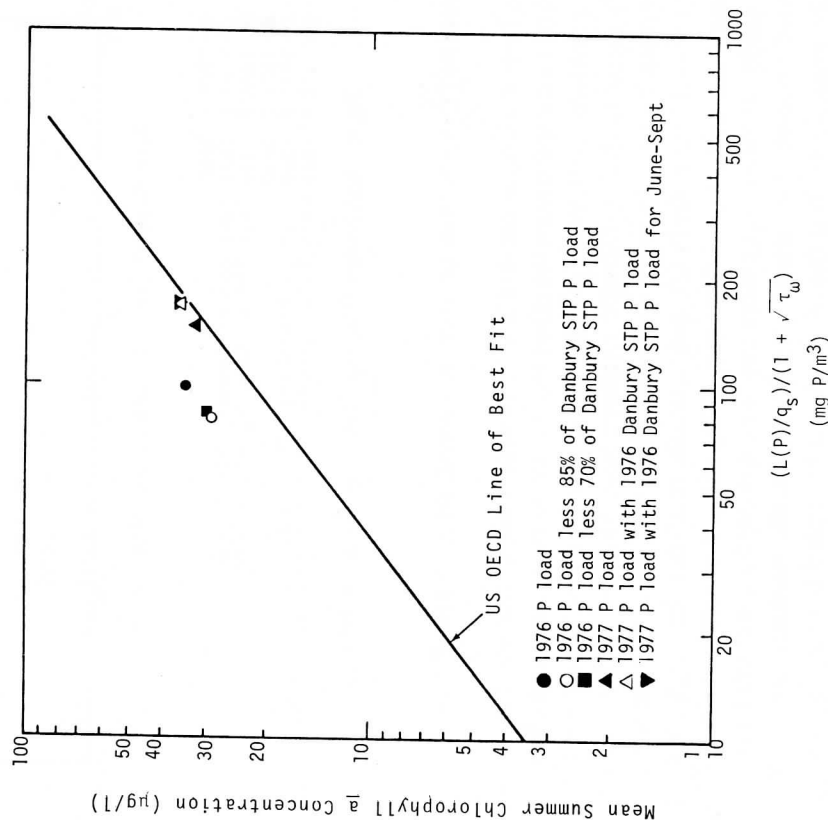


Fig. 10. Relationship between P load and mean summer chlorophyll *a* concentrations in Lake Lillinnonah.

Table V presents the P loading and lake characteristics necessary to apply the Rast and Lee chlorophyll *a* - P load relationship to Lake Lillinnonah under the various loading and hydraulic conditions given or assumed. Figure 10 shows the positions of the lake in 1976 and 1977, relative to the US OECD line of best fit for this relationship and also expected changes in the mean summer chlorophyll *a* concentration resulting from altered P loads. This figure shows that for both 1976 and 1977, the relationship for Lake Lillinnonah is similar to that determined for other waterbodies having the same P load - mean depth/hydraulic residence time characteristics, i.e., the lake plots near the US OECD line of best fit. This finding gives considerable support to the validity of the

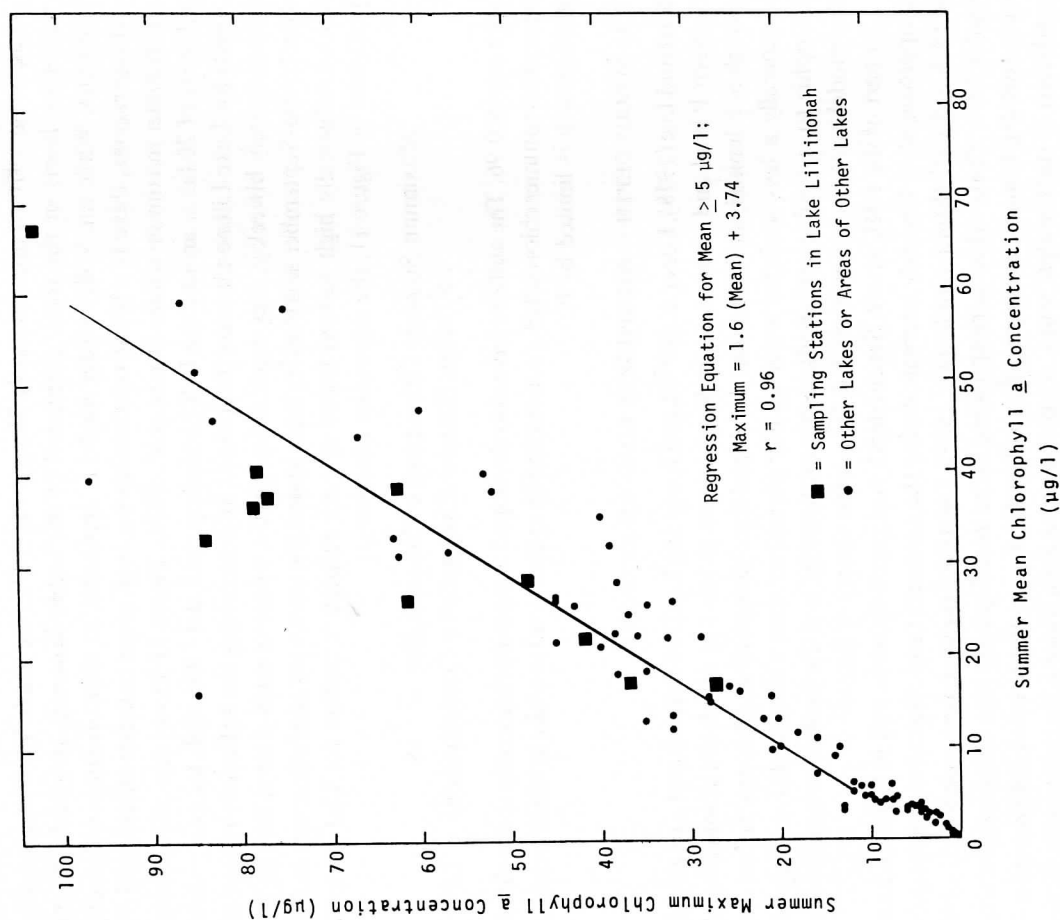


Fig. 11. Relationship between mean summer chlorophyll *a* and maximum chlorophyll *a* concentrations (After Jones *et al.*, 1979).

assumptions made with respect to the characteristics of the lake that were used to evaluate nutrient load-response relationships.

Figure 10 shows that there was a reduction in the mean summer chlorophyll *a* concentration between 1976 and 1977. Further, it predicts that had the total phosphorus load to the lake been reduced in 1976 by an amount equivalent to 70 to 85% of the Danbury wastewater treatment plant P load, a reduction in the mean summer chlorophyll *a* concentrations would have been from approximately $35 \mu\text{g l}^{-1}$ (actually found in 1976) to about $30 \mu\text{g l}^{-1}$. This model also predicted that if the 70% of the Danbury STP P load that had apparently been removed at the plant had not been removed in 1977, the mean summer chlorophyll *a* concentration would have increased from about $35 \mu\text{g l}^{-1}$ in 1976 to approximately $36 \mu\text{g l}^{-1}$ in 1977 (see Table V). It is generally the maximum levels or the total 'greenness' of the water, however, to which the public responds. While the Vollenweider model is applicable to mean summer chlorophyll *a* concentrations, there is a linear relationship between mean summer chlorophyll *a* and the summer maximum values (Jones *et al.*, 1979). Figure 11 shows this relationship for 62 areas of 20 lakes around the U.S., including the Lake Lillinnonah sampling stations. Points for Lake Lillinnonah were determined by determining the arithmetic means of the approximately biweekly, 1 m depth chlorophyll *a* values at each station for the period June to mid-September and plotting them against the maximum concentration (excluding the unrealistically high values) found at each respective station for that period. As presented in Figure 11, the regression equation is:

$$\begin{aligned} \text{Maximum Summer Chlorophyll } a \text{ Concentration} = \\ 1.6 (\text{Mean Summer Chlorophyll } a \text{ Concentration}) + 3.74 \end{aligned}$$

with an *r* of 0.96. The authors have also developed a general relationship that relates the maximum summer chlorophyll *a* concentration to the P load to a lake in which planktonic algal growth is limited by P.

2.5. SECCHI DEPTH - PHOSPHORUS LOAD RELATIONSHIP

Rast and Lee (1978), Lee *et al.* (1978), and Jones and Lee (1981) developed a relationship between P load and mean summer Secchi depth in lakes in which the planktonic algal growth is P limited, similar to that developed for chlorophyll *a*. Although Secchi depth is generally a less sensitive water quality parameter than chlorophyll for moderately eutrophic to eutrophic waterbodies, this relationship and the predictive capabilities of this model have been demonstrated for a wide variety of lakes.

As part of the FMC study, Secchi depth was generally measured in Lake Lillinnonah at approximately biweekly intervals during the summers (mid-June to mid-September) of 1976 and 1977 (Smith and Brown, 1978). Eight lake stations were monitored in 1976; four in 1977. Those values corresponding to the unrealistically high chlorophyll *a* values were excluded from the averaging since values for Secchi depth, when determined under conditions of surface algal scum, are not interpretable. Since light penetration in water is a logarithmic function, the arithmetic mean of the logs of the individual Secchi depth values was taken by the authors; the antilog of this mean was then used as the mean

summer Secchi depth in this modeling. Table V presents the mean summer Secchi depths for 1976 and 1977 as well as those predicted for various alterations of P load, based on the Rast and Lee model.

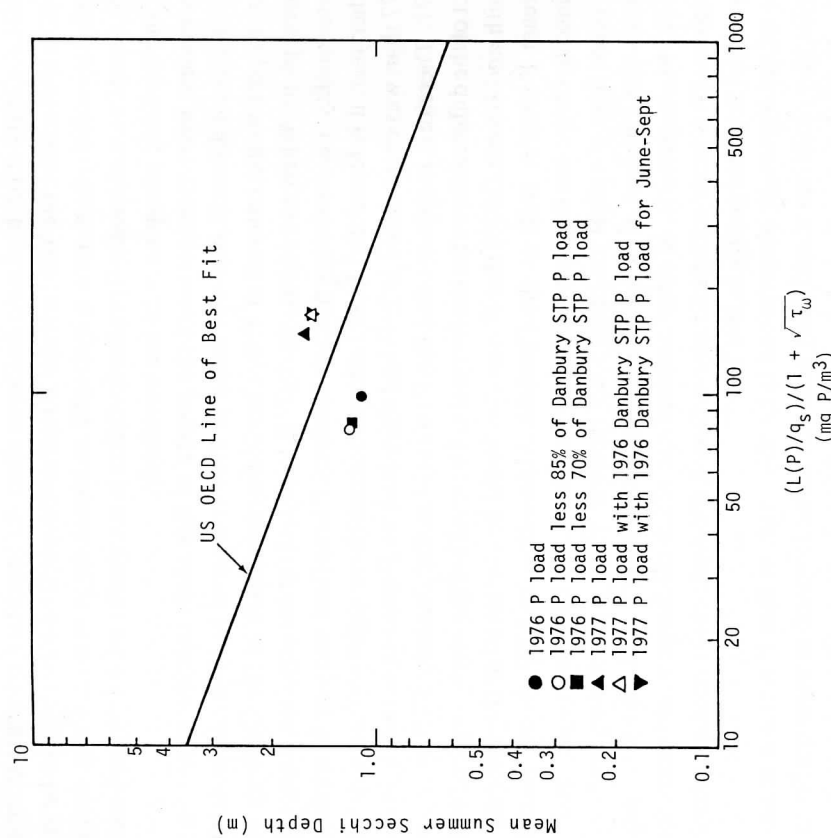


Fig. 12. Relationship between P load and mean summer Secchi depth for Lake Lillinnonah.

Figure 12 shows the positions that Lake Lillinnonah assumes in 1976 and 1977 relative to the US OECD line of best fit for the Secchi depth - P load relationship. The lake, in both 1976 and 1977, appears to have a relationship between mean summer Secchi depth and P load normalized by mean depth and hydraulic residence time similar to that found by Rast and Lee (1978) and Lee *et al.* (1978) for many U.S. waterbodies. This model predicted that if the P load to the lake was reduced in 1976 by an amount equivalent to the removal of 70 to 85% of the Danbury STP P load, the mean summer Secchi depth would have increased by 0.1 m to 1.2 m (see Table V). If the proportion of the Danbury P load that can be estimated to have been removed as a result of Danbury's advanced wastewater treatment, were added to the 1977 phosphorus load to the lake, the model predicted that the mean summer (1977) Secchi depth would have been 1.5 m.

3. Discussion

The US EPA National Eutrophication Survey report on Lake Lillinnah indicated that the Vollenweider nutrient load-lake response relationship may not be applicable to this lake because of its short hydraulic residence time. An important conclusion from this study is, however, that the Vollenweider P load-chlorophyll relationship can be applied to waterbodies with short annual hydraulic residence times provided that during the period of maximum chlorophyll production there is adequate time for phytoplankton to utilize the available phosphorus in the waterbody.

It is evident from examination of the results of this study that the improved water quality in Lake Lillinnah that was found in 1977 compared to 1976 can be attributed at least in part to the removal of P by chemical precipitation at the Danbury sewage treatment plant. When one adjusts the 1977 P load data for the expected impact of flow on water quality, i.e., assuming that the Danbury P load from one year is applicable to the other year, it is found that Lake Lillinnah should have slightly worse water quality in 1977 than was experienced in 1976 had there been no P removal at the Danbury STP in 1977. Therefore, it appears that the P removal at the Danbury STP overcame the impact of the different flows for the two years. The results of this study support the results of Smith and Brown (1978) that P removal should be continued at the Danbury STP. Upstream P sources such as the Pittsfield, Massachusetts STP should be critically examined to determine the potential improvement in Lake Lillinnah water quality that could result from chemical treatment for P removal. From a preliminary point of view, it appears that initiation of chemical treatment for P removal at Danbury, Connecticut, Pittsfield, Massachusetts, or other communities which discharge wastewaters into the Housatonic River or its tributaries above Lake Lillinnah, could result in a significant improvement in water quality in this lake.

One of the important observations from this study is that P removal based on a detergent phosphate ban would likely have very limited impact on water quality in Lake Lillinnah. As noted earlier, about 35% of the P in domestic wastewaters can be attributed to the use of phosphate as a builder in synthetic detergents. Therefore, the maximum removal of P that could be achieved at the Danbury STP by a detergent phosphate ban, would be on the order of half of that actually achieved during the summer of 1977. Actually, it is expected that with extended experience and good maintenance and operation practices, upwards of 90% removal of the P in the Danbury domestic wastewaters can be achieved using chemical precipitation at a cost to the residents of Danbury of less than a cent per person per day (Smith and Brown, 1978). Therefore, even greater reductions in the mean and maximum summer chlorophyll concentrations in Lake Lillinnah would be expected in the future as a result of P removal from domestic wastewaters by chemical precipitation. It should also be noted that in addition to providing P removal at a small cost, other benefits such as further removal of BOD, suspended solids, heavy metals and trace organics would be expected concomitant with P removal by chemical precipitation.

Results of this study support the proposal of Smith and Brown (1978) of removing

P only during the late spring and summer. It does not appear that there is a need to practice P removal during fall, winter, and early spring since P contributed to Lake Lillinnah during this period is likely flushed through the lake during high flows in the early spring. This practice would further reduce the cost of P removal by chemical precipitation at the treatment plants that discharge to the Housatonic or one of its tributaries above Lake Lillinnah.

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