Sedimentary losses of phosphorus in some natural and artificial Iowa lakes

Daniel E. Canfield Jr.,[1] John R. Jones[2] & Roger W. Bachmann
Department of Animal Ecology, Iowa State University, Ames, IA 50011, U.S.A.
[1] *Present address: School of Forest Resources and Conservation, University of Florida, Gainesville, FL 32611, U.S.A.*
[2] *School of Forestry, Fisheries and Wildlife, University of Missouri, Columbia, MO 65211, U.S.A.*

Keywords: sedimentation rates, phosphorus cycle

Abstract

Phosphorus sedimentation in four natural and four artificial Iowa lakes was measured by using sediment traps to determine if sedimentary phosphorus losses were greater in artificial lakes than in natural lakes and the limnological factors influencing phosphorus loss rates. Mean phosphorus sedimentation rates ranged from 13.3 to 218 $mg \cdot m^{-2}\ day^{-1}$. Although phosphorus sedimentation rates for the natural lakes as a group did not differ significantly from the rates for artificial lakes, there were significant differences among individual lakes. Phosphorus sedimentation rates also varied significantly during different seasons at different locations within a lake and at different depths within a location. Despite the variance, phosphorus sedimentation rates were strongly correlated with inorganic sediment concentrations and inorganic matter sedimentation rates, thus suggesting that inorganic sediments influence phosphorus sedimentation rates. When Iowa data were combined with data from published studies, mean sedimentation rates were directly correlated with mean chlorophyll *a* concentrations of the lakes. These data strongly suggest that sedimentation rates as measured by sediment traps are strongly influenced by the trophic status of a lake. Though sedimentation rates were higher in the more productive lakes, it is suggested that these rates represent only gross sedimentation rates rather than net sedimentation rates because of resuspension and resedimentation of bottom sediments.

Introduction

Studies on a broad range of lakes have suggested that phosphorus is the element most likely controlling algal biomass (Sakamoto 1966; Dillon & Rigler 1974a; Jones & Bachmann 1976; Schindler 1978); thus, any factor that removes this element from a lake or causes it to become unavailable to phytoplankton may be an important determinant of lake trophic state. Vollenweider (1969) proposed that the concentration of phosphorus in lakes is determined largely by the rate of phosphorus supply, but is modified by losses through the outlet and to the sediments. Dillon (1975) demonstrated the importance of phosphorus losses through the outlet, but the importance of sedimentary losses has not been directly demonstrated because phosphorus sedimentation is difficult to measure.

Sedimentary phosphorus losses have, however, been estimated from data on phosphorus inputs and outputs. Using data from natural lakes, various authors (Dillon & Rigler 1974b; Vollenweider 1975; Kirchner & Dillon 1975; Chapra 1975; Jones & Bachmann 1976; Larsen & Mercier 1976) have developed empirical relationships to estimate sedimentary phosphorus losses. Using these relationships, they have developed empirical phosphorus loading models that can predict total phosphorus concentrations in natural lakes with reasonable success (Jones & Bachmann 1976; Chapra & Robertson 1977). Studies by Jones & Bachmann (1978), however, on several artificial lakes in central

Hydrobiologia 87, 65–76 (1982). 0018-8158/82/0871-0065/$02.40.

Iowa have shown that the existing models overestimate summer phosphorus concentrations by 3 to 10 times. They suggested that this overestimation resulted because phosphorus sedimentation rates are greater in artificial lakes than in natural lakes. By using sedimentation coefficients 2 orders of magnitude greater than those used for natural lakes, Jones and Bachmann demonstrated that the general phosphorus model proposed by Vollenweider (1969) could be used to calculate summer phosphorus concentrations in artificial Iowa lakes.

Successful prediction of total phosphorus concentrations in natural and artificial lakes will depend upon how well sedimentary phosphorus losses are estimated. This study was designed to measure directly phosphorus sedimentation by using sediment traps in some natural and artificial Iowa lakes. Our objective was to test the hypothesis that sedimentary phosphorus losses are greater in artificial lakes than in natural lakes. We also wished to determine the limnological factors influencing phosphorus sedimentation rates in lakes, because an understanding of these factors might permit the development of better predictive models.

Methods

Between June 1976 and September 1978, sediment traps were used to measure directly phosphorus sedimentation in four natural and four artificial lakes (Table 1). The locations of the lakes sampled are given in Canfield (1979). Preliminary studies were made between June and September 1976 to determine if the location of a sediment trap in a lake could significantly influence estimates of lake sedimentation rates. Paired sediment traps were placed at three locations along the long axis of West Okoboji, East Okoboji, Big Creek, and Don Williams lakes. Traps were also placed at various depths in the deep areas of Big Creek and West Okoboji lakes. The traps were retrieved approximately every 2 weeks.

In January 1977, paired sediment traps were placed at the deep areas of the eight lakes, and winter phosphorus sedimentation rates were measured about every 40 days until March 1977. Winter sedimentation rates were extremely low, so subsequent measurements were limited to the open water period. Sediment traps were retrieved ap-

Table 1. Origin, morphometric data, and average hydrologic and chemical conditions for the surface waters of the natural (N) and artificial (A) Iowa lakes during the open-water seasons of 1977 and 1978; S indicates summer stratification and NS indicates the absence of permanent summer stratification.

	Year	West Okoboji	East Okoboji	Spirit	Lost Island	Big Creek	Don Williams	Beeds	Pine
Origin		N	N	N	N	A	A	A	A
Watershed area (ha)		7698	5903	9962	5180	19453	7995	8236	4118
Surface area (ha)		1540	764	2168	465	358	65	43	25
Mean depth (m)		11.9	2.8	5.2	3.1	5.4	5.1	2.9	2.8
Stratification		S	NS	NS	NS	S	S	S	NS
Hydraulic flushing rate (y^{-1})	1977	0.008	0.05	0.02	0.06	1.8	4.4	5.6	2.6
	1978	0.07	0.44	0.14	0.56	1.9	4.6	21.7	14.6
Phosphorus loadings ($g \cdot m^{-2} y^{-1}$)	1977	0.06	0.08	0.05	0.10	3.3	7.4	3.6	2.4
	1978	0.29	0.44	0.27	0.61	3.4	7.7	21.0	13.6
Total P ($mg \cdot m^{-3}$)	1977	20.8	194.2	40.0	115.6	30.3	57.8	110.9	79.2
	1978	23.5	121.3	40.1	98.5	40.7	43.3	72.8	75.5
Chlorophyll *a* ($mg \cdot m^{-3}$)	1977	3.8	38.5	24.6	92.3	22.1	43.7	75.0	70.2
	1978	6.6	83.6	24.0	61.6	25.9	25.7	44.4	58.0
Total hardness ($mg \cdot l^{-3}$ as $CaCO_3$)	1977	219.0	239.0	240.0	210.0	203.0	221.0	223.0	149.0
	1978	219.0	217.0	233.0	217.0	249.0	283.0	271.0	180.0
Calcium hardness ($mg \cdot l^{-3}$ as $CaCO_3$)	1977	75.0	96.0	72.0	71.0	119.0	120.0	118.0	73.0
	1978	78.0	87.0	74.0	80.0	155.0	166.0	165.0	95.0

Drainage areas for West Okoboji, East Okoboji, and Spirit Lakes are assumed to be independent (Jones 1974).

proximately every 2 weeks during the summer and every 15 to 30 days during spring and fall.

Sediment traps were similar in design to those used by Dudley (1976). Collecting tubes consisted of four PVC cylinders (5.1 cm × 25.4 cm) closed at one end with a rubber stopper. No preservatives were used in the cylinders. Two cylinders were inverted in the holding platform to estimate attached growth (White 1974), which was found to be minimal compared with the quantity of sedimented material. The holding platforms were clamped 1.5 m above the bottom sediments to a nylon rope held in place by an anchor and subsurface float suspended at 2 m. Sediment traps were carefully raised to the surface so that no resuspension of deposited materials occurred during retrieval.

Surface water samples (0.5 m) were collected at four midlake stations near the location of the traps. All water samples were collected in acid-cleaned Nalgene bottles and transported in insulated boxes to the laboratory for chemical analyses. Water clarity was measured by using a Secchi disc, and water temperature was measured with a resistance thermometer.

Chlorophyll *a* was extracted by using the methods of Yentsch & Menzel (1963) and calculated by using the equations of Parsons & Strickland (1963). Values were not corrected for phaeophytin.

Total phosphorus concentrations (Murphy & Riley 1962) were determined after a persulfate oxidation (Menzel & Corwin 1965). After oxidation, samples were centrifuged to remove suspended particles. Dissolved total phosphorus samples were filtered through a 0.45 μm membrane filter and treated as total phosphorus. Particulate phosphorus values were determined by difference.

Total iron concentratins were determined by using the ferrozine method of Hach Chemical Co. (1975). Sample volumes were 50 ml, and all samples were gently boiled for 20 min to ensure that all the iron had reacted. After cooling, sample volumes were returned to 50 ml with distilled water and centrifuged to remove particulate matter.

Concentrations of total, organic, and inorganic suspended matter were determined on measued volumes of lake water filtered through precombusted (550 C), preweighed Gelman Type A-E glass fiber filters. The total weight of particulate matter on a filter was determined after drying the filters at 103 °C for 1 h. The quantity of inorganic matter was determined after combustion at 550 °C for 1 h. Organic weights were estimated by difference. Filters were cooled over desiccant and weighed to ± 0.01 mg with a Cahn Model G Electrobalance (Cahn Instrument Co., Paramont, California). Filter blanks were processed, and all filters were corrected for losses during handling.

The contents of each PVC cylinder from the sediment traps were emptied into acid-cleaned, 1 l glass jars with 10 ml of formaldehyde and refrigerated at 2 °C. Preceding chemical analysis, the sample was brought up to a volume of 600 ml with distilled water. Sediment samples were mixed by using a magnetic stirrer and subsamples for chemical analyses (the volume being dependent upon the concentration of sediments) were withdrawn by using a large bore 10 ml pipette.

To determine the quantity of total phosphorus that could potentially sediment to the bottom in each lake, four 1 l samples of surface water from each lake were returned to the laboratory and allowed to settle in the dark at ambient temperatures in 1 l graduated cylinders. After 24 h, approximately 950 ml of sample were siphoned off and discarded. The sedimented materials were resuspended and analyzed for total phosphorus. Results were expressed as the quantity of phosphorus lost from a cubic meter of lake water.

To determine how well sediment traps estimate net annual phosphorus losses, sedimentation coefficients were estimated from lake phosphorus budgets (Jones & Bachmann 1976) by:

$$\sigma = L/(zTP) - \rho \tag{1}$$

where

TP = concentration of total phosphorus in lake water, $\mathrm{mg \cdot m^{-3}}$
L = annual areal phosphorus loading, $\mathrm{mg \cdot m^{-2}\, y^{-1}}$
z = lake mean depth, m
σ = phosphorus sedimentation coefficient, $\mathrm{y^{-1}}$
ρ = hydraulic flushing rate, $\mathrm{y^{-1}}$.

Water inputs for each lake were calculated from watershed areas and runoff from the nearest United States Geological Survey gaging station (Table 2). Lake evaporation was assumed to equal precipitation, and groundwater influences were ignored.

could not detect any consistent sedimentation patterns among the lakes and thus were unable to determine the factors influencing spatial variations in sedimentation rates. Based on previous studies and our data, we conclude that unless the factors influencing spatial variations in sedimentation rates are known for a given lake an accurate determination of sedimentation rates will require the placement of sedimentation traps at many different depths and locations. We found, however, greater differences in sedimentation rates among lakes than between locations within a lake (Table 3). Variance components calculated from a standard analysis of variance on the 1976 phosphorus sedimentation data (Table 4) showed that 65% of the total observed variance could be attributed to differences among lakes while only 13% of the variance could be attributed to differences between locations within individual lakes. From these data, we conclude that, for an understanding of the limnological factors influencing sedimentation rates, particularly phosphorus, a large number of lakes should be studied rather than intensively studying individual lakes. For this reason, we placed sediment traps at the deepest area in our study lakes and added two natural (Lost Island and Big Spirit) and two artificial (Beeds and Pine) lakes (Table 1) to the study in 1977.

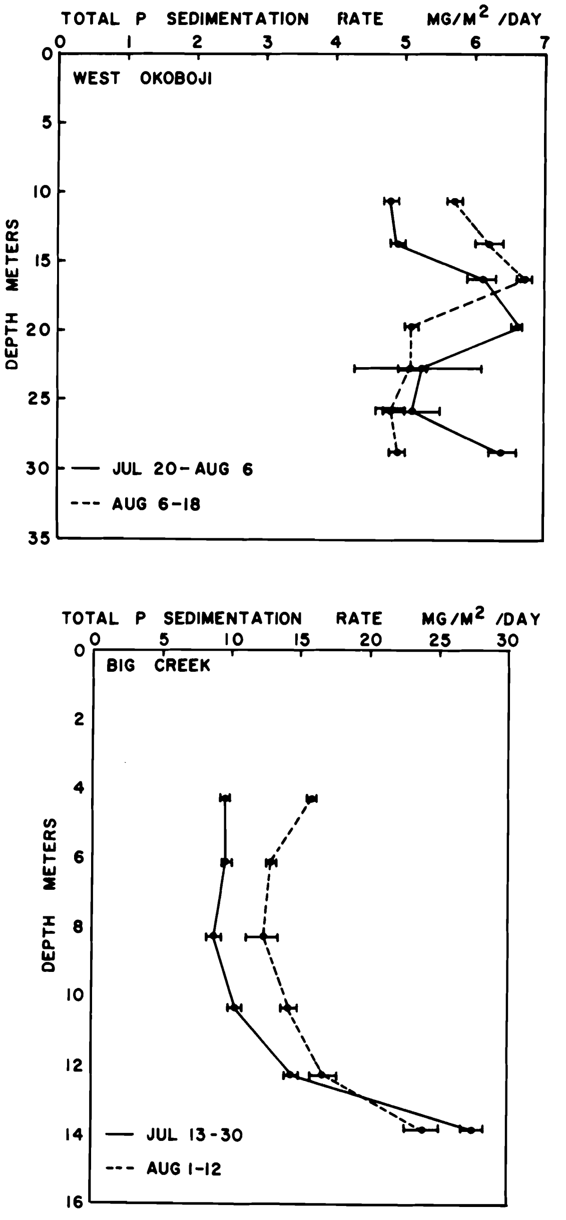

Fig. 1. Variation in phosphorus sedimentation rates (mg·m^{-2} day^{-1}) with depth in Big Creek and West Okoboji; bars represent the standard error of the mean (n = 4).

Studies on natural lakes have shown that sedimentation rates may change over an annual cycle in either a bimodal pattern, with deposition being greatest in the spring and fall and lowest in the summer and winter (Lawcz 1969; White 1974; Lastein 1976), or in a unimodal pattern, with deposition being greatest in the spring or summer (Toyoda *et al.* 1968; Gasith 1976). Variations in sedimentation rates generally have been attributed to inputs of allochthonous sediments (Toyoda *et al.* 1968; Hakala 1977), phytoplankton production (Thomas 1955; Lawcz 1969; Lastein 1976; Gasith 1976; Birch 1976; Hakala 1977), and resuspension and resedimentation of bottom sediments (Tutin 1955; Davis 1968; Pennington 1974; Lastein 1976). In the Iowa lakes, we found sedimentation rates, particularly phosphorus (Fig. 2), were highly variable within and among lakes. Rates were lowest when the lakes were covered by ice and increased with the loss of ice cover and thermal stratification

Table 4. Percent contribution of various sources to the total variance of phosphorus sedimentation rates.

Variance source	%
Lake	65
Trap location	13
Traps within location	9
Sampling	1
Error	12

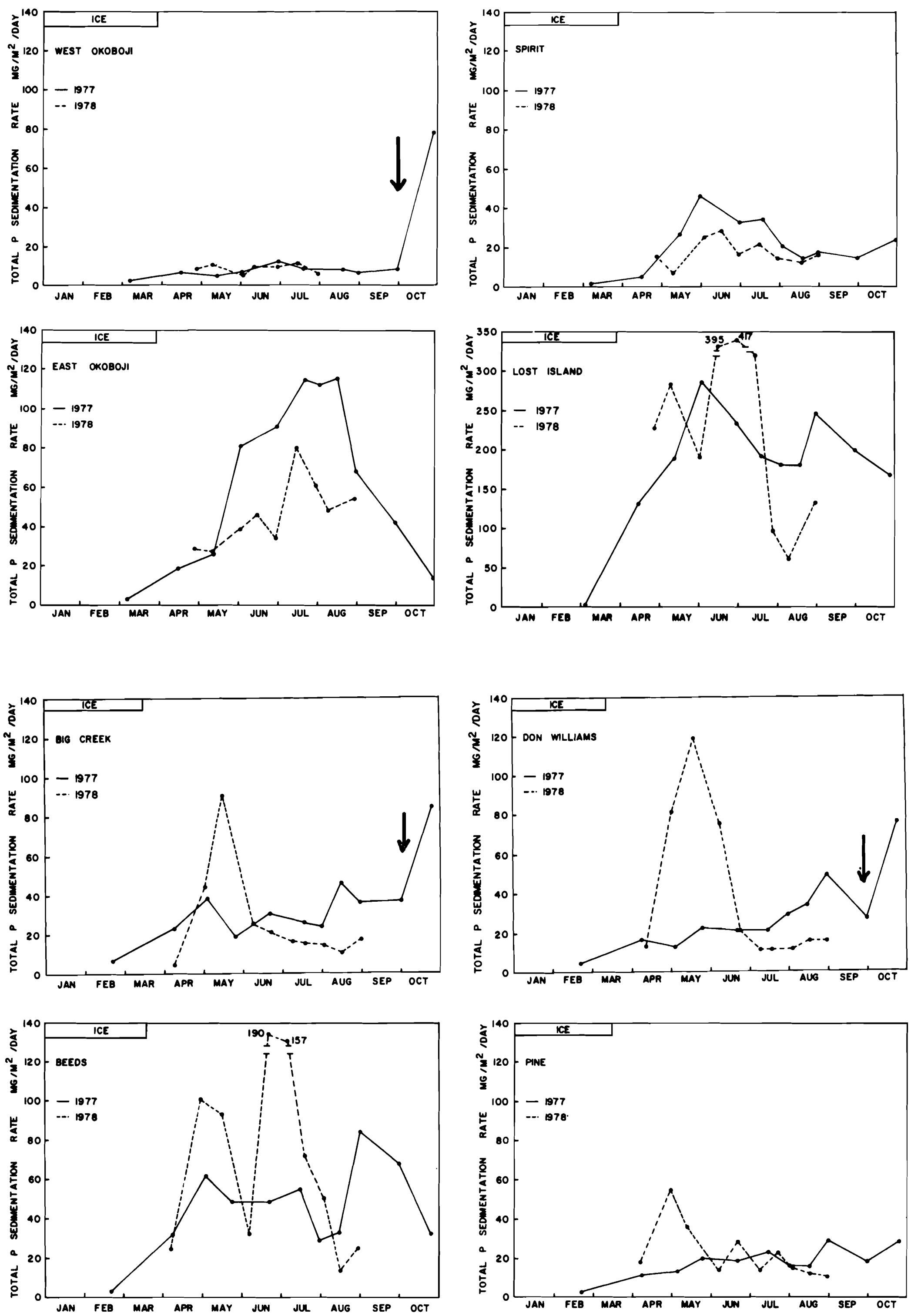

Fig. 2. Seasonal variations in phosphorus sedimentation rates (mg · m^{-2} day^{-1}) in the Iowa lakes; note scale change for Lost Island Lake.

(Fig. 2), thus suggesting the importance of resuspension and resedimentation of bottom sediments. Variations in sedimentation rates at certain times could be attrributed to inputs of allochthonous sediments; in the artificial lakes (Fig. 2, see data for 1978), peak sedimentation rates coincided with inputs of sediments during runoff events. In some lakes such as East Okoboji, sedimentation rates were greatest during the summer months of peak algal populations. We could not, however, consistently relate changes in sedimentation to changes in inputs of allochthonous sediments, phytoplankton production, or resuspension and resedimentation of bottom sediments. We believe that this failure results because these factors, as well as others, operate independently and concurrently, thus making it very difficult to use seasonal data to determine the primary factors influencing sedimentation in lakes. Therefore, generalizations about seasonal sedimentation patterns, particularly open water patterns, should be made with caution and probably be limited to the specific lake under study.

Despite the many factors that can influence sedimentation in a lake during an annual cycle, we found significant differences among lakes (Table 5). This suggests that there may be a factor that influences sedimentation rates more than other factors. In our sample, sedimentation rates in natural and artificial lakes were not different, indicating that lake origin is not an important determinant of sedimentation rates. We did find, however, that mean phosphorus sedimentation rates were directly and significantly correlated with mean inorganic suspended matter concentrations (Table 6; $r = 0.89$; $p = 0.01$) and mean inorganic matter sedimentation rates ($r = 0.97$; $p = 0.01$), which suggests that inorganic sediments resulting from allochthonous inputs and resuspension may be an important determinant of phosphorus sedimentation in the Iowa lakes. Other factors (Table 6) also correlate significantly with phosphorus sedimentation rates, which makes it very difficult to conclusively state that inorganic sediments are the prime determinant of sedimentation rates.

To determine which factors influence sedimentation rates in a broad range of lakes, we expanded our sample by using data from the published literature (Table 7). A comparison of measured rates shows that the Iowa lakes have some of the highest values yet reported, perhaps because they

Table 5. Averages of limnological measurements made on some natural and artificial Iowa lakes during the ice-free periods between June 1976 and September 1978.

Parameter	West Okoboji	Spirit	East Okoboji	Lost Island	Big Creek	Don Williams	Beeds	Pine
Total phosphorus mg·m^{-3}	18.7	40.0	139.4	109.5	30.7	42.0	96.2	77.8
Soluble phosphorus mg·m^{-3}	12.1	13.9	90.9	32.0	13.5	20.3	34.1	22.3
Particulate phosphorus mg·m^{-3}	6.6	26.1	48.5	77.5	17.2	21.7	62.1	55.5
Total iron mg·m^{-3}	18.0	37.0	38.0	43.0	64.0	58.0	135.0	195.0
Total suspended matter g·m^{-3}	2.2	9.6	13.2	33.9	6.5	7.4	16.4	13.7
Organic suspended matter g·m^{-3}	1.4	6.6	8.2	17.6	3.5	4.5	8.3	8.9
Inorganic suspended matter g·m^{-3}	0.8	3.0	5.0	16.3	3.0	2.9	8.0	4.8
Chlorophyll *a* mg·m^{-3}	4.0	24.0	66.0	81.0	27.0	31.0	63.0	66.0
Secchi disc transparency m	3.7	1.3	1.0	0.4	1.6	1.6	0.8	0.7
Total dry matter sedimented g·m^{-2}day^{-1}	11.6	16.9	65.8	202.8	32.0	23.8	72.4	19.5
Organic matter sedimented g·m^{-2}day^{-1}	3.0	7.0	14.8	57.3	5.2	4.7	12.4	4.1
Inorganic matter sedimented g·m^{-2}day^{-1}	8.6	9.9	51.0	145.5	26.8	19.1	60.0	15.4
Total iron sedimented mg·m^{-2}day^{-1}	54.0	53.0	119.0	130.0	351.0	215.0	717.0	281.0
Total phosphorus sedimented mg·m^{-2}day^{-1}	13.3	20.8	60.3	218.1	28.8	29.1	62.9	21.1
Total phosphorus sedimented in laboratory mg·m^{-3}	2.0	7.4	7.9	29.0	4.5	5.9	9.4	5.6

Table 6. Correlations between mean phosphorus sedimentation rates and other measured parameters.

Parameter	r
Total dry matter sedimentation rates	0.99
Organic matter sedimentation rates	0.97
Inorganic matter sedimentation rates	0.97
Inorganic suspended matter	0.89
Total suspended matter	0.84
Organic suspended matter	0.78
Total phosphorus	0.75
Particulate phosphorus	0.74
Chlorophyll *a*	0.70

are shallow and productive. Past studies (Tutin 1955; Davis 1968; Pennington 1974; Lastein 1976) have suggested that resuspension and resedimentation of bottom sediments increase sedimentation rates in shallow lakes, but we found total dry weight sedimentation rates and lake mean depth only weakly correlated ($r = -0.61$; $p = 0.01$). Total dry weight sedimentation rates, however, were strongly correlated with chlorophyll *a* concentrations ($r = 0.94$; $p = 0.01$). Phosphorus sedimentation rates also were strongly correlated with chlorophyll *a* concentrations ($r = 0.94$; $p = 0.01$). From these

Table 7. Total dry weight and total phosphorus sedimentation rates measured by use of sediment traps reported in the published literature and in this study.

Reference	Lake	Mean depth (m)	Chlorophyll *a* ($mg \cdot m^{-3}$)	Dry weight sedimentation ($g \cdot m^{-2}$ day^{-1})	Phosphorus sedimentation ($mg \cdot m^{-2}$ day^{-1})
Toyoda *et al.* (1968)	Biwa	41.2	1.7	0.57	0.79
Lawacz (1969)	Mikolajki	11.0	-	4.10	-
Brunskill (1969)	Fayetteville Green	28.0	-	0.81	0.40
Johnson & Brinkhurst (1971)	Ontario	91.0	-	0.66	-
Hayashi *et al.* (1972)	Suwa	5.0	-	9.0	-
Pennington (1974)	Belham Tarn	6.8	-	2.45	-
	Windermere	21.0	-	0.50	-
	Ennerdale	17.8	-	0.56	-
	Wastwater	39.7	-	0.45	-
Bloesch (1974)	Rotsee	9.0	-	2.4	5.8
	Lake of Lucerne (Horw Bay)	42.6	-	3.5	3.8
White (1974)	Lawrence	5.9	-	0.14	-
Charlton (1975)	Char Lake	10.2	-	1.00	-
Birch (1976)	Findley	8.2	0.88	0.21	0.37
	Chestermorse	19.0	1.4	0.45	0.83
	Sammamish	17.0	5.9	1.8	4.2
	Washington	33.0	6.3	2.4	7.6
Dudley (1976)	Big Creek	5.4	17.4	34.7	34.8
	Don Williams	5.11	30.8	15.6	28.0
Gasith (1976)	Wingra	2.44	-	13.2	-
Lastein (1976)	Ersom	12.3	-	1.7	-
Hakala (1977)	Paajarvi	14.4	-	0.4	-
Kajak & Lawacz (1977)	Smolak	2.4	-	1.0	-
	Piecek	3.4	-	5.7	-
	Dgal Maly	4.3	-	7.8	-
	Czarna Kuta	1.3	-	30.0	-
Ravera & Viola (1977)	Lugona	130.0	-	1.74	-
Serruya (1977)	Kinneret	25.0	-	3.6	8.4
This study	West Okoboji	11.9	4.2	11.6	13.3
	East Okoboji	2.8	66.2	65.7	60.3
	Spirit	5.2	24.3	16.9	20.7
	Lost Island	3.1	81.3	203.0	218.0
	Big Creek	5.4	27.1	32.0	28.8
	Don Williams	5.1	30.8	23.8	29.1
	Beeds	2.9	63.3	72.4	62.9
	Pine	2.8	65.7	19.5	21.1

Table 8. Mean phosphorus sedimentation coefficients (y^{-1}) calculated from sediment trap data and lake phosphorus budgets for the Iowa lakes during 1977 and 1978.

Lake	From sediment trap data	From lake phosphorus budgets
West Okoboji	22	0.61
East Okoboji	56	0.47
Spirit	36	0.72
Lost Island	235	0.86
Big Creek	63	16.0
Don Williams	49	26.0
Beeds	82	43.0
Pine	35	29.0

data, we conclude that, although resuspension and resedimentation of bottom sediments probably occur to a greater extent in shallow lakes, the trophic status of a lake as indicated by chlorophyll *a* concentrations is a more important determinant of sedimentation rates measured by using sediment traps.

Though our analyses strongly suggest that productive lakes have greater sedimentation rates, we do not believe that the high rates measured in productive lakes represent the net loss of materials to the bottom sediments. This is particularly true in Iowa lakes where daily phosphorus losses ranged from 5 to 60% of the total phosphorus in the lake (Table 5). Comparing phosphorus sedimentation coefficients calculated from lake nutrient budgets (Equation 1), we find that sediment traps overestimate phosphorus losses by approximately 4× in the artificial lakes and 200× in the natural lakes (Table 8). This indicates that sediment traps probably are not estimating net losses of material to the bottom sediment. These data, however, raise three questions: 1) What are sediment traps measuring? 2) Why do sediment traps provide a better estimate of phosphorus losses in Iowa's artificial lakes than in Iowa's natural lakes? 3) Why are phosphorus sedimentation coefficients calculated from nutrient budgets nearly 2 orders of magnitude greater in artificial Iowa lakes than in natural Iowa lakes (Table 8)?

Kleerekoper (1952) and Golterman (1973) suggested that sediment traps do not accurately measure sedimentation rates because traps reduce turbulence and thus induce sedimentation. Studies by Kirchner (1975) and Hargrave (1979), which evaluated the performance of different types of traps, however, have provided data that suggest otherwise, and Pennington (1974) has shown that, in some lakes, sediment accumulations in traps are similar to those expected from radioactive dating of bottom sediments.

Laboratory settling experiments showed that from 2.0 to 29.0 $mg \cdot m^{-3}$ of potentially settleable phosphorus (Table 5) were suspended in the surface waters of the Iowa lakes, which represents from 5 to 26% of the phosphorus suspended in the surface water. Although these data are not directly comparable to sediment-trap data, sedimentation rates measured by use of sediment traps parallel the laboratory values (Table 5) thus suggesting the traps are measuring the downward movement of particulate matter. Rates measured by traps should be considered as gross sedimentation rates, however, because nutrient recycling and resuspension and resedimentation of bottom sediments occur to some extent in all lakes.

To explain why the sediment traps provided better estimates of phosphorus losses in the artificial lakes, we initially hypothesized that resuspension and resedimentation of bottom sediments might be significantly greater in the natural lakes. Conditions for resuspension of sediments seemed less favorable in the artificial lakes because these lakes have smaller surface areas (Table 1), channel-type morphometry, and are better protected from the wind by surrounding hills. However, with the loss of ice cover in 1977 (a drought year with minimal surface-water runoff during the spring), we found that sedimentation rates increased by a similar magnitude in both the natural and artificial lakes (Fig. 2). If these increases after ice-out are used as an index of the potential importance of resuspension and resedimentation of bottom sediments in the Iowa lakes, we conclude that the natural and artificial lakes have large quantities of sediments resuspended and that resuspension is not significantly greater in the natural lakes as a group.

We next hypothesized that differences in algal production might cause sediment traps to overestimate net phosphorus loss rates in the natural lakes, but algal population densities (as measured by chlorophyll *a* concentrations) were similar in the natural and artificial lakes (Table 5); thus, we rejected this hypothesis.

Jones and Bachmann (1978) suggested that the input and sedimentation of inorganic particulate materials influence the removal of phosphorus in the artificial lakes. Though our data (Fig. 2), show phosphorus sedimentation rates measured by sediment traps are variable in both the natural and artificial lakes, we noted that rates in the artificial lakes were highest after runoff events brought in large amounts of allochthonous sediments (see peak rates in 1978; Fig. 2). The only exceptions were the peak sedimentation rates associated with the loss of thermal stratification (see 1977 data; Fig. 2). We also found that once surface runoff subsides (based on visual observations), phosphorus sedimentation rates in the artificial lakes decline rapidly (Fig. 2; see peak periods for 1978); thus, we conclude that the sediments are rapidly sedimented to the bottom, and the data indicate that relatively little of this material is resuspended and resedimented over time. Because the sedimentation of allochthonous sediments dominates sedimentation events in the artificial lakes, we conclude that the sedimentation of inflowing sediments dominates both gross and net sedimentation events in the artificial lakes. This is in contrast to the natural lakes where resuspension and resedimentation of bottom sediments dominate gross sedimentation and cause sediment traps to overestimate net phosphorus losses.

The question remains as to why phosphorus sedimentation coefficients calculated from our estimated phosphorus budgets are nearly 2 orders of magnitude greater in the artificial lakes than in the natural lakes (Table 8). We do not believe this difference can be attributed to errors in our estimation of nutrient budgets because the values are in good agreement with those calculated from direct measurements made by Jones & Bachmann (1978). They are also in general agreement with coefficients (Table 9) estimated from a large sample of natural and artificial lakes (Canfield 1979). We believe that the differences are related to the input and sedimentation of allochthonous sediments (Jones & Bachmann 1978), which may be described by three distinct mechanisms. First, tributaries to the natural lakes may carry smaller loads of inorganic particulate sediments relative to their phosphorus load than do tributaries to the artificial lakes; thus, a smaller fraction of the phosphorus inputs to the natural lakes would sediment. This would result in the natural lakes having smaller sedimentation coefficients than the artificial lakes. We have no field data to verify this hypothesis, but the natural Iowa lakes lie in glaciated depressional topography, whereas the artificial lakes are located in valleys of erosional topography (Jones & Bachmann 1978). Second, if the tributary streams of the natural and artificial lakes carry similar loads of inorganic sediments relative to their phosphorus loads, the phosphorus removal capacity of these sediments may depend upon the concentration of these sediments in the lakes. We have observed that the artificial lakes, with their greater hydraulic flushing rates (Table 2), become extremely turbid with inorganic sediments during surface runoff events, whereas, in the natural Iowa lakes, the concentrations of inflowing sediments are diluted by large volumes of lake water, and the lakes show very little response to these events. If phosphorus is bound to the inflowing sediments, physical settling could account for the artificial lakes having higher phosphorus sedimentation coefficients. Third, the artificial lakes (Table 5) have higher iron concentrations (58 to 195 $mg \cdot m^{-3}$) and iron sedimentation rates as measured by use of sediment traps (215 to 717 $mg \cdot m^{-2}\ day^{-1}$) than the natural lakes (18 to 78 $mg \cdot m^{-3}$; 53 to 130 $mg \cdot m^{-2}\ day^{-1}$). Because iron, as well as the clays and aluminium associated with the inflowing sediments, are known to effectively bind phosphorus, we believe that the high concentrations of these materials in the artificial lakes may enhance the removal of the phosphorus during sedimentation. This would be similar to the action of fly ash, iron, and alum used in nutrient inactivation experiments.

Conclusions

Sediment traps have been used by many investigators to estimate the loss of materials from the water column or accumulations of materials in the bottom sediments. Because many different factors can influence sedimentation, loss rates may vary considerably at different locations within a lake, at different depths within a location, and within time over an annual cycle. Despite the many factors that could possibly influence sedimentation and the extreme variability in sedimentation, we are able to show that the trophic status of lakes as measured by chlorophyll *a* concentrations is a major determinant of sedimentation rates measured by sediment traps. However, because nutrient recycling and resuspension and resedimentation of bottom sediments occur to some extent in all lakes, we believe that sedimentation rates measured by sediment traps should be considered only as gross sedimentation rates. The use of sediment traps to determine net loss rates is not recommended unless factors such as resuspension and resedimentation of bottom sediments are minimal. Application of correction factors for resuspended matter as proposed by Gasith (1976) should be used cautiously inasmuch as errors in estimating resuspended matter could be significantly greater than the resulting estimate of net sedimentation rates.

Although we could not use sediment traps to measure net sedimentation rates directly in the natural and artificial Iowa lakes, our study did show that large quantities of particulate matter move through the water column of the Iowa lakes. Resuspension of bottom sediments (Pennington 1974; Lastein 1976; Gasith 1976) seems to be an important source for this particulate matter. There have, however, been no studies that have directly measured this process in a range of natural and artificial lakes; thus there is very little information concerning the limnological significance of resuspended bottom sediments. Information is needed on how often sediments are mixed into the water column and how long they remain there. We also need information concerning the release or absorption of nutrients by resuspended sediments if we are to better understand nutrient dynamics in lakes.

Our studies suggest that inputs of allochthonous sediments and their subsequent sedimentation strongly influence net phosphorus sedimentation rates. Lakes with high flushing rates and large inputs of allochthonous sediments, such as the artificial Iowa lakes, should have higher net phosphorus losses. There have, however, been no studies that have concurrently measured sediment and phosphorus inputs and outputs to lakes having different limnological conditions; thus, it is difficult to determine if differences in sediment inputs are responsible for the greater phosphorus loss rates in artificial lakes. Information is also needed to determine if allochthonous sediments absorb phosphorus from the water column. Information from these studies should provide a better understanding of phosphorus and sediment dynamics in lakes and lead to the development of improved phosphorus loading models.

Acknowledgements

Journal Paper No. J-9989 of the Iowa Agriculture and Home Economics Experiment Station, Ames, Iowa. Project 2051. Supported through a grant from the Office of Water Resources Research and Technology (Project No. A-063-IA), U.S. Department of the Interior, Washington D.C., as authorized by the Water Research and Development Act of 1978.

We thank Mark Hoyer, Terry Noonan, Jackie LaPerriere, Carol Bachmann, David Bachmann, Mark Johnson, Sarah Lindquist, Steve Parris, and Pat Nagy for assistance with sample collection and laboratory analyses. We also thank R. V. Bovbjerg for use of laboratory facilities at Iowa Lakeside Laboratory and R. O. Anderson and R. Carlson for their comments and advice in writing this paper. A microfiche copy of the original data can be obtained for $1.00 from the Photo Service, Iowa State University, Ames, IA 50011, USA; request Supplement to Publication No. 6/7/79 Canfield Ph. D. Thesis.

References

Birch, P. B., 1976. The relationship of sedimentation and nutrient cycling to the trophic status of four lakes in the Lake Washington drainage basin. Ph. D. thesis. Library, University of Washington, Seattle, Washington.

Bloesch, J., 1974. Sedimentation und phosphorhaushalt im Vierwaldstattersee (Hower Bucht) und im Rotsee. Schweiz. Z. Hydrol. 36: 71-186.

Brunskill, G. J., 1969. Fayetteville Green Lake, New York. II. Precipitation and sedimentation of calcite in a meromictic lake with laminated sediments. Limnol. Oceanogr. 14: 830-847.

Canfield, D. E., Jr., 1979. Prediction of total phosphorus concentrations and trophic states in natural and artificial lakes: the importance of phosphorus sedimentation. Ph. D. dissertation, Iowa state University, Ames. Catalog Number 8000120, University Microfilms International, Ann Arbor, Michigan, U.S.A.

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 Resour. Res. 11: 1033-1034.

Chapra, S. C. & Robertson, A., 1977. Great lakes eutrophication: the effect of point source control of total phosphorus. Science 196: 1448-1449.

Charlton, M. N., 1975. Sedimentation: measurements in experimental enclosures. Verh. int. Verein. Limnol. 19: 267-272.

Davis, M. B., 1968. Pollen grains in lake sediment: redeposition caused by seasonal water circulation. Science 162: 796-799.

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

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

Dillon, P. J. & Rigler, F. H., 1974b. A test of a simple nutrient budget model predicting the phosphorus concentration in lake water. J. Fish. Res. Bd Can. 31: 1771-1778.

Dudley, D. R., 1976. Phosphorus sedimentation in some Iowa reservoirs. M. S. thesis, Library, Iowa State University, Ames, Iowa.

Gasith, A., 1976. Seston dynamics and tripton sedimentation in the pelagic zone of a shallow eutrophic lake. Hydrobiologia 51: 225-231.

Golterman, H. L., 1973. Vertical movement of phosphate in freshwater. In: Griffith, E. J. et al. (eds.) Environmental Phosphorus Handbook, pp. 509-538. John Wiley, New York.

Hach Chemical Co., 1975. Water and Wastewater Analysis Procedures, 3rd edn. Hach Chemical Co., Ames, Iowa.

Hakala, I., 1977. Sedimentation energy flow to the profundal of the oligotrophic Lake Paajarvi, southern Finland. Ann. bot. fenn. 14: 157-164.

Hayashi, H., Okino, T. & Aoyama, K. A., 1972. The balance of organic matter in a water column in a lake. I. Examinations into the propriety of the concept of water column using an artificial water column separated by a vinyl sheet. Jap. J. Limnol. 33: 51-59.

Johnson, M. G. & Brinkhurst, R. O., 1971. Benthic community metabolism in Bay of Qunite and Lake Ontario. J. Fish. Res. Bd Can. 28: 1715-1725.

Jones, J. R., 1974. Eutrophication of some northwestern Iowa lakes. Ph. D. dissertation, Iowa State University, Ames. Catalog Number 75-3314, University Microfilms International, Ann Arbor, Michigan, U.S.A.

Jones, J. R. & Bachmann, R. W., 1976. Prediction of phosphorus and chlorophyll levels in lakes. J. Water Pollut. Control Fed. 48: 2176-2182.

Jones, J. R. & Bachmann, R. W., 1978. Phosphorus removal by sedimentation in some Iowa reservoirs. Verh. int. Verein. Limnol. 20: 1576-1580.

Kajak, Z. & Lawacz, W., 1977. Comparison of tripton sedimentation in four small lakes. In: Golterman, H. L. (ed.) Interactions between Sediments and Freshwater, pp. 72-75. Junk, The Hague.

Kirchner, W. B., 1975. An evaluation of sediment trap methodology. Limnol. Oceanogr. 20: 657-660.

Kirchner, W. B. & Dillon, P. J., 1975. An empirical method of estimating the retention of phosphorus in lakes. Water Resour. Res. 11: 182-183.

Kleerekoper, H., 1952. A new apparatus for the study of sedimentation in lakes. Can. J. Zool. 30: 185-192.

Larsen, D. P. & Mercer, H. T., 1976. Phosphorus retention capacity of lakes. J. Fish. Res. Bd Can. 33: 1742-1750.

Lastein, E., 1976. Recent sedimentation and resuspension of organic matter in eutrophic Lake Erson, Denmark. Oikos 27: 44-49.

Lawacz, W., 1969. The characteristics of sinking materials and the formation of bottom deposits in a eutrophic lake. Mitt. int. Verein. Limnol. 17: 319-331.

Menzel, D. W. & Corwin, N., 1965. The measurement of total phosphorus in seawater based on the liberation of organically bound fractions by persulfate oxidation. Limnol. Oceanogr. 10: 280-282.

Murphy, J. & Riley, J. P., 1962. A modified single solution method for the determination of phosphate in natural waters. Analyt. chim. Acta 27: 31-36.

Parsons, T. R. & Strickland, J. D., 1963. Discussion of spectrophotometric determination of marine-plant pigments, with revised equations for ascertaining chlorophylls and carotinoids. J. mar. Res. 21: 155-163.

Pennington, W., 1974. Seston and sediment formation in five lake district lakes. J. Ecol. 62: 215-251.

Ravera, O. & Vicla, M., 1977. Sedimentation-rate in a basin (Agno) of Lake Lugan. In: Golterman, H. L. (ed.) Interactions between Sediments and Freshwater, pp. 174-178. Junk, The Hague.

Richards, F. A. with Thompson, T. G., 1952. The estimation and characterization of plankton populations by pigment analyses; II. A spectrophotometric method for the estimation of plankton pigments. J. mar. Res. 11: 156-171.

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

Schindler, D. W., 1978. Factors regulating phytoproduction and standing crop in the world's freshwaters. Limnol. Oceanogr. 23: 478-486.

Serruya, C., 1977. Rates of sedimentation and resuspension in Lake Kinneret. In: Golterman, H. L. (ed.) Interactions between Sediments and Freshwater, pp. 48-56. Junk, The Hague.

Thomas, E. A., 1955. Stoffhaushalt und Sedimentation im oligotrophen Aegerisee und im eutrophen Pfaffiker- und Greifensee. Mem. Ist. Ital. Idrobiol. 8 (suppl.): 357-465.

Toyoda, Y., Horie, K. & Saija, Y., 1968. Studies on the sedimentation in Lake Biwa from the viewpoint of the lake metabolism. Mitt. int. Verein. Limnol. 14: 243-255.

Tutin, W., 1955. Preliminary observations on a year's cycle of sedimentation in Windermere, England. Mem. Ist. Ital. Idrobiol. 8 (suppl.): 467-484.

Vollenweider, R. A., 1969. Possibilities and limits of elementary models concerning the budget of substances in lakes (in German). Archs. 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.

White, W. S., 1974. Role of calcium carbonate precipitation in lake metabolism. Ph. D. thesis. Library, Michigan State University, East Lansing, Michigan.

Yentsch, C. S. & Menzel, D. W., 1963. A method for the determination of phytoplankton chlorophyll and phaeophytin by fluorescence. Deep Sea Res. 10: 221-231.

Received 14 November 1980.