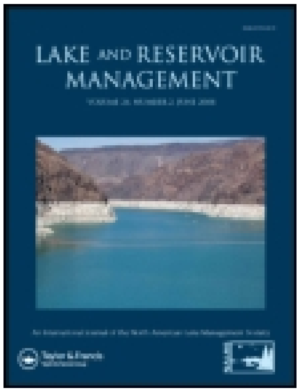

Lake and Reservoir Management

Publication details, including instructions for authors and subscription information:
http://www.tandfonline.com/loi/ulrm20

Assessing internal phosphorus load – Problems to be solved

Gertrud K. Nürnberg [a]

[a] Freshwater Research , 3421 Hwy 117, RR 1, Baysville , Ontario , P0B 1A0 , Canada

Published online: 14 Dec 2009.

Lake and Reservoir Management, 25:419–432, 2009

ISSN: 0743-8141 print / 1040-2381 online
DOI: 10.1080/00357520903458848

Assessing internal phosphorus load – Problems to be solved

Gertrud K. Nürnberg*
Freshwater Research, 3421 Hwy 117, RR 1, Baysville, Ontario P0B 1A0, Canada

Abstract

Nürnberg, G. K. 2009. Assessing internal phosphorus load – Problems to be solved. Lake Reserv. Manage. 25: 419–432.

Internal loading as phosphorus (P) released from anoxic sediment surfaces often represents the main summer P load to lakes and reservoirs and can have an immense effect on their water quality. Many difficulties in internal load assessment exist, however, including ignoring internal load altogether, ambiguity about the origin of sediment released P and inexact definitions. Most of these problems are due to the difficulty in distinguishing internal from external P sources, which is particularly challenging in polymictic lakes. To prevent misconceptions and facilitate its evaluation, internal load in stratified and polymictic lakes should be expressed in a similar way to external loads: as annual, gross and areal load of total phosphorus (TP). Possible approaches to internal load quantification are: *in situ* determination from hypolimnetic P increases, mass balance approaches, and estimates from anoxic active area and P release. Further suggestions to facilitate the study of internal loading include: (a) the differentiation between polymictic and stratified lakes, sections of lakes, and time periods when evaluating indicators and impact of internal load; (b) the separation of internal load (upward flux) from sedimentation (downward flux) of external and internal loads, and (c) the consideration of the downward flux of both external (L_{ext}, mg/m^2/yr) and internal (L_{int}, mg/m^2/yr) loads by a retention model (R_{sed}) when predicting lake TP averages in a mass balance model of the form (q_s = annual areal water load in m/yr):

$$TP = \frac{L_{ext} + L_{int}}{q_s} \times (1 - R_{sed})$$

Key words: internal phosphorus load, retention, stratified and polymictic lakes and reservoirs, TP mass balance modeling

After more than 70 years (Einsele 1936, Mortimer 1941) of knowledge about phosphorus (P) release from sediments, assessment of internal P load is still one of the most challenging subjects in lake and reservoir eutrophication and restoration. We know that internal loading as P released from sediment surfaces often represents the main summer P load to lakes. Because of its high biological availability, lack of dilution and timing, it can have an immense effect on the water quality of a lake, reservoir or pond; however, the following difficulties in internal load assessment still remain:

1. Undetected internal load.
2. Controversy about the ultimate origin and ambiguity about the form of sediment released P.
3. Unclear definitions and inconsistent units and dimensions.
4. Inadequate quantification and modeling of internal load by confusing downward with upward fluxes and net with gross estimates; and
5. Inadequate determination of its contribution to lake P concentration.

Most of these problems are due to the difficulty in tracking lake water P, in particular distinguishing internal from external sources. Possible approaches have been developed including regression analysis, mass balance and time-dynamic modeling (e.g., Nürnberg 1998, 2005, Håkanson 2004), but many pitfalls remain and are discussed here to help prevent further misconceptions and facilitate the evaluation of internal load in all lakes and reservoirs.

In this paper I first identify and discuss problems 1–3. Next, while addressing problems 4 and 5, I present three different methods for the quantification of internal load and describe the mass balance modeling of lake P concentration as

*Corresponding author: gkn@fwr.on.ca

affected by the concept of P retention. Examples from case studies are presented throughout to illuminate the problems and their possible solutions.

Unnoticed internal load

Internal P load is generally attributed to reductive dissolution of P adsorbed to iron oxyhydroxides in the sediments and subsequent release from the anoxic sediment surfaces according to the classic model of Mortimer (1941) and Einsele (1936), or to release from organic compounds (Gächter and Meyer 1993) and poly-phosphates in very eutrophic sediments (Hupfer *et al*. 2007) according to more recent models. Controversies exist about the origin of P derived from sediments and its release mechanism (Prairie *et al*. 2001, Hupfer and Lewandowski 2008), but these questions are less important for the quantification and management of internal load. It is generally accepted that most internal loading is released in the form of ortho-phosphate, which is fully biologically available and can potentially be used by phyto- and bacterio-plankton (Cooke *et al*. 2005).

While external P sources are generally recognized as contributors to lake P concentration, internal sources are often overlooked. In stratified lakes, the epilimnion can seem to be untouched by internal load during the summer, and a monitoring program could easily miss its occurrence if sampling occurs only in the mixed epilimnion. Nevertheless, certain seasonal and spatial patterns indicate the occurrence of internal load (Table 1A). Typically, total P (TP) and dissolved reactive P (DRP, analysis for ortho-phosphate, often also called soluble reactive P or SRP) concentrations increase in the hypolimnion during summer stratification so that profiles show increasing concentrations below the thermocline toward the sediment (Fig. 1A). If such elevated P concentrations are associated with anoxia and the prevalence of some reduced substances including ferrous iron, manganeous manganese, ammonium and gases of hydrogen sulfide and methane, the occurrence of internal P load is certain (Fig. 1B). In addition to hypolimnetic P increases, epilimnetic increases due to entrainment and diffusion during the period of thermocline erosion later in the summer have been documented (Mataraza and Cooke 1997) and were included in a seasonal mass balance model (Auer *et al*. 1997). Further, conspicuous increases during and after fall turnover are a definite sign of internal P load when it becomes mixed into the surface water (Nürnberg and Peters 1984b, Nürnberg 1985).

In shallow, polymictic water bodies it is more difficult to distinguish internal from external loads because their water is usually vertically mixed, in addition to the horizontal exchange as happens in all lakes (Søndergaard *et al*. 2005); therefore, internal load indicators are different in

Table 1.-Indicators of internal load in stratified (A) and polymictic lakes (B)

A. Stratified, mono- or dimictic (deep) lakes
Severe hypolimnetic anoxia
Profiles: increasing TP and DRP with depth
Seasonal: increasing hypolimnetic TP and DRP throughout summer
Concomitant iron, manganese or reduced gas development
Fall turnover: blooms, increased turbidity
Mass balance:
• More TP leaving the lake than entering (negative retention)
• Less TP retained than predicted (from q_s)
• Higher TP concentration than predicted
B. Polymictic (shallow) lakes
Seasonal: increasing TP and DRP throughout summer, even in upper water layers
Turnover events during summer: blooms, increased turbidity
Thin oxic sediment layer; occasional anoxia in weed beds and open water during quiescent conditions (early morning)
Occasional iron, manganese or reduced gas development during quiescent conditions
Mass balance:
• More TP leaving the lake than entering (negative retention)
• Less TP retained than predicted (from q_s)
• Higher TP concentration than predicted

polymictic lakes (Table 1B). For example, some P is released from bottom sediments into the mixed overlaying water so that it is taken up by phytoplankton and may foster algal and cyanobacterial blooms in shallow lakes, while a large proportion remains as DRP in the stagnant summer hypolimnion of stratified lakes (Fig. 2). Consequently, internal load affects surface water quality in shallow polymictic lakes even in summer, but mostly during thermocline erosion and turnover in the fall in stratified lakes.

The effect of internal load is obvious when TP increases during a summer drought where all external inputs cease. In the shallow, polymictic reservoir Lake Mitchell, South Dakota (Table 2), TP concentrations typically increase throughout the summer. Because in summer 2001 the inflow, which on average contributes 92% of the annual external TP load, had ceased, all increases had to derive from internal sources and can be used to quantify internal load (after correction for changes in lake level; Fig. 3). Similarly, evidence of internal load was observed when TP concentrations greatly exceeded inflow concentrations in western Washington lakes (Welch and Jacoby 2001).

Differences due to the mixing state are less clear in lakes that are polymictic in some summers but stratified in

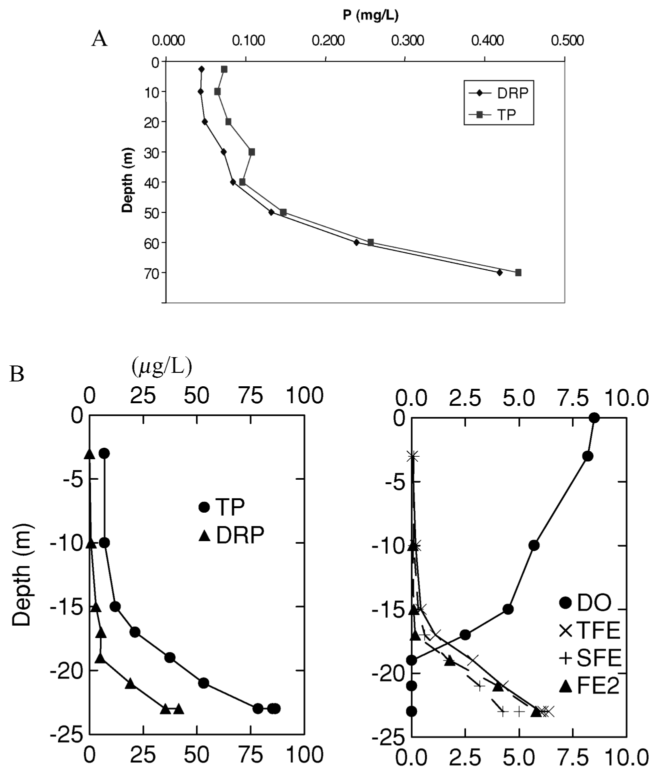

Figure 1.-(A) P and DRP profiles at the dam of eutrophic Brownlee Reservoir, Snake River, CO, 11 August 1999; (B) P, dissolved oxygen (DO) and iron profiles (TFE = total; SFe = soluble; FE2 = ferrous) in oligotrophic Chub Lake, Ontario, Sep 13, 1982 (note lake characteristics in Table 2).

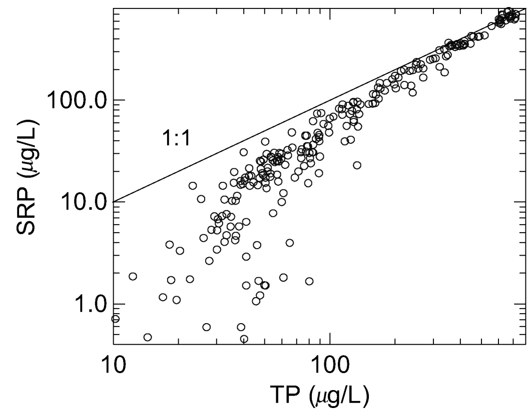

Figure 2.-Comparison of DRP with TP from individual measurements in the anoxic hypolimnion of eight stratified Canadian lakes (with data from Nürnberg and Peters 1984b).

others, as in Brome Lake, Quebec (Table 2). In this lake, TP increases were followed by chlorophyll increases throughout two growing seasons, while in one year (1996) increases happened only in late September (Fig. 4). Thus, the mixing state in polymictic lakes can be variable between years, sometimes more resembling the summer stratification of dimictic lakes, like in 1996 in Brome Lake.

Differences are also less pronounced in water bodies with both stratified and polymictic areas. This is particularly important for deep, run-of-the river reservoirs that deepen gradually from the shallow and polymictic section at the inflow to a deep stratified section at the dam. Such differences also exist in lakes where a large shallow area and a deeper area exist in the same lake, as in Lake Pyhäjärvi, Finland (Table 2) and Mona Lake, Michigan (Steinman and Ogdahl 2008). In these lakes a high proportion of DRP in the stratified part indicates that internal loading is the P source, while a low proportion in the shallow riverine section may be the consequence of immediate transformation of orthophosphate into algae biomass and adsorption onto silt particles after its release from the sediments, in addition to sediment resuspension.

These examples reveal many clues to internal load in lakes and reservoirs despite its seeming invisibility. Such signs are especially obvious when the stratification regime is considered so that polymixis and stratification are differentiated with respect to the whole or partial sections of the lake or reservoir and different time periods.

Table 2.-Characteristics of lakes and reservoirs used as examples and case studies (Nürnberg, unpublished data; avg = average, max = maximum).

Name, Location	Area (km^2)	Mean Depth (m)	Mixing Regime*	Trophic State**	Summer Avg Epilimnetic TP (μg/L)	Max Hypolimnetic TP (μg/L)
Brome Lake, Southern Quebec	14.6	5.7	s/m	m	15	150
Brownlee Reservoir, Snake River, ID	47.5	32.3	s/m	e-h	80-130	500
Cherry Creek Reservior, Denver, CO	4	3.2	m	e	75	200
Chub Lake, Muskoka, Ontario	0.34	8.9	s	o	9.3	86
Lake Mitchell (reservoir), SD	3.1	3.7	s/m	h	320	320
Pyhäjärvi (Lake), Finland	155	5.5	s/m	m	18	100

*s = stratified; m = mixed; s/m = depending on location or year: s or m.
**Based on classification of Nürnberg (1996). o = oligotrophic; m = mesotrophic; e = eutrophic; h = hyper-eutrophic.

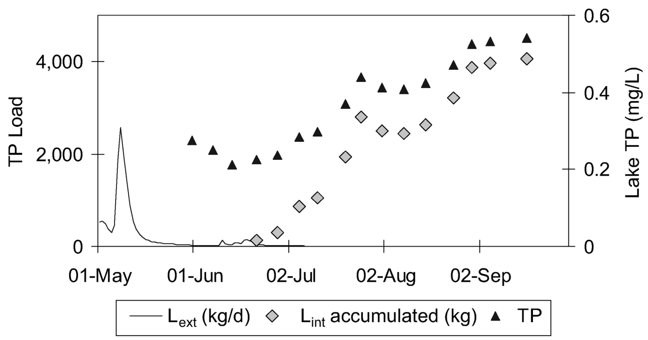

Figure 3.-L_{int_1} estimate from *in situ* TP increases in polymictic Lake Mitchell Reservoir, SD, for summer 2001.

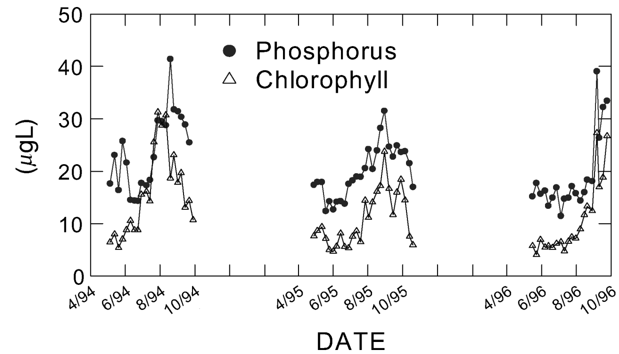

Figure 4.-TP and chlorophyll concentration for three growing seasons in polymictic Brome Lake, Quebec.

Phosphorus release is not restricted to and does not require anoxia in the overlaying water. It suffices to have a mechanism that conveys the P released into the porewater from sediment iron-oxyhydroxides to the sediment surface. In addition to low redox potential at the sediment surfaces, P release can be enhanced by bioturbation, when Chironomids effectively pump the phosphate-rich porewater into the overlying water (Holdren and Armstrong 1980). Persistent P release occurs especially from highly eutrophic sulfuric sediments where the formation of iron sulfides effectively removes iron from the P-Fe cycle and liberates P from vivianite. A result of such mechanisms is that artificial hypolimnetic oxygenation by pure oxygen in eutrophic Swiss lake Sempachersee did not decrease internal P loading during more than 15 years of this elaborate and expensive restoration treatment (Gächter and Müller 2003).

Chemical forms of sediment released P

Although internal load is typically ortho-phosphate release, P speciation, such as the analytical analysis of orthophosphate as DRP, does not necessarily lead to a reliable quantification of internal load. Only in anoxic hypolimnia with elevated TP concentrations can correctly conducted DRP determination be used to quantify internal load, because under these conditions most of the sediment-released P remains DRP, as discussed earlier (Fig. 1 and 2). In anoxic P-rich hypolimnia, a large proportion of DRP was biologically available when mixed with epilimnetic plankton in short-term bioassays, so that DRP also quantifies the potential bioavailability of internal load in anoxic hypolimnia (Nürnberg 1984a; Nürnberg and Peters 1984a).

However, analytical errors are abundant when determining DRP in anoxic waters (Nürnberg 1984a), especially underestimation, for several reasons: (a) accidental aeration during the sampling and filtration procedure can lead to phosphate adsorption onto ferric iron; (b) conversely, neglecting to degas the sample by aeration before analysis implies interference by the hydrogen sulfide gas with the molybdenum blue analysis in more reduced hypolimnetic water; (c) delay in analysis immediately after sample collection allows P uptake by phytoplankton and bacteria. In addition to possible underestimation, overestimation can easily occur due to contamination when dealing with low concentrations of DRP.

More important, P compounds are altered after release in most circumstances except in anoxic hypolimnia. Generally, changes of the released P depend on conditions such as trophic state, chemistry and mixing regime of the lake water and abundance and nutrient state of the plankton (Søndergaard *et al*. 2001). The following pathways have been described for sediment-released P (Nürnberg 1984a, Nürnberg 1985): (1) it remains ortho-phosphate (and is analyzed as DRP); (2) upon aeration it may become adsorbed onto ferric iron hydroxide particles that are larger than 0.45 μm and are not analyzed as DRP; (3) it is analyzed as DRP in the presence of organic acids, in particular humic and fulvic acids, which can keep adsorbed P small ($<$0.45 μm) and suspended (in this case, DRP would overestimate phosphate); (4) it becomes incorporated into algae biomass that outcompetes aeration processes described in cases 2 and 3 due to high dilution during entrainment of hypolimnetic water and mixing events.

Because all these processes may have taken place after the release of sediment-derived ortho-phosphate, determination of DRP mass alone will not adequately quantify internal load in many circumstances. Furthermore, because analytical determination of DRP is problematic in anoxic waters, I suggest quantifying internal load as mass of TP (molybdenum blue analysis after complete digestion; e.g., Murphy and Riley 1962) for routine determination. This method analyzes all P compounds in a water sample as TP, is easy to carry out and relatively free of interference, if contamination is avoided.

Inexact definitions

Internal load estimates are sometimes ambiguous because studies relate internal load to various time periods, including year, growing season, summer, month, anoxic period or stratified period. Some studies contemplate the question about the original source of internal load, which in eutrophic lakes most often is external. Considering the long-term, such load is not actually internal (Hupfer and Lewandowski 2008; Michael Coveney, St Johns River Water Management District, Department of Water Resources, Palatka, FL, U.S.A., 2006, pers. comm.). Other studies relate internal load to varying areas, such as total lake surface and anoxic or hypolimnetic area. For example, the Lake Mendota research group refers only to the portion of the sediment-released P that is entrained into the epilimnion throughout the growing season and has an immediate effect on phytoplankton (e.g., Lathrop *et al*. 1999).

However, calculations that do not consider gross release from the sediments during the whole year under investigation, underestimate internal load and its contribution to lake P concentration; the origin (internal vs. past external) is irrelevant for quantification unless the load originates from external sources of the current study year. In some highly eutrophic systems a large portion of the present external load possibly settles and becomes recycled within the same year, resembling sediment-released P (Coveney *et al*. 2005; Hupfer and Lewandowski 2005). In such cases the differentiation of internal from external sources is ambiguous, and the direct impact of external on the apparent internal load is obvious. If significant, the portion of the present internal load that stems from such a current external source is best distinguished from a former external source (e.g., 30% of the sediment released TP is recycled from this year's external load, the source of the present internal load stems mostly from the years before sewage diversion or it was created during 10 years of aquaculture).

In general, the expression of gross internal load as P mass released per lake surface area and year (L_{int}, mg/m^2/yr) is comparable and analogous to the definition of external P load (L_{ext}, mg/m^2/yr). In comparison, and to prevent ambiguity, an anoxic P release rate (RR, mg/m^2/d) can be defined as a rate of P release from actively releasing sediments, with units of P mass released per anoxic sediment surface area and day.

Confusing net with gross estimates of internal load

The terms gross and net with respect to internal load refer to whether the proportion that settles annually to the lake bottom has been considered in the calculations. For example, external loading is usually expressed as gross because it is calculated from individual external P sources such as precipitation, runoff and point sources before it enters the lake; it has not been subjected to changes within the lake such as biological and physical uptake and sedimentation. In contrast, internal loading originates from the lake bottom in any given year, and its measurement may include settling events of the summer season that have taken place before its final quantification.

Considering theses processes, Nürnberg and LaZerte (2001) and Nürnberg *et al*. (in prep.) proposed three principally different procedures to determine internal load depending on available information:

1. A partially net estimate from ***in situ* summer increases**
2. Net (and gross) estimates from complete P budgets (**mass balance approach**)
3. A gross estimate from P release (anoxic areal P release rate, RR in mg/m^2/d) and measured anoxia (as the anoxic factor, AF_{meas}, in days per summer) in stratified lakes and predicted anoxia (AF_{pred}) in polymictic lakes (Nürnberg 2005; **AF × RR approach**)

While Methods 1 and 2 are independent of theoretical considerations with respect to the origin of internal load, Method 3 assumes that the release depends on and is enhanced when sediment surfaces become anoxic. This is expected during microbial mediated anoxia for the classic release from ferrous oxy-hydroxides as well as from polyphosphates of organic compounds and does not require anoxia in the overlaying water (Holdren and Armstrong 1980). Approaches, underlying assumptions, models and formulas in these methods are sometimes different for stratified and shallow lakes or water bodies with unusual features such as hard water, bottom water outlet, and for fast flushed reservoirs.

Method 1: In situ *P increases* (L_{int_1})

When using summer *in situ* TP increase to estimate internal load according to equation 1, some sedimentation has already happened before quantification, yielding a partially net estimate.

$$L_{int\_1} = \frac{TP\_t_2 \times V\_t_2}{A_o\_t_2} - \frac{TP\_t_1 \times V\_t_1}{A_o\_t_1} \quad (1)$$

where, t_1 is initial date and t_2 is date at end of period or at maximum summer TP concentration (Julian day); TP_t is the corresponding water column average TP concentration and V_t is the corresponding lake volume; and A_o_t is lake surface area (if necessary, surface area changes have to be considered).

Table 3.-Internal load estimates (L_{int_1}, equation 1) for 11 summers (summer, su = May to "End-Date") from *in situ* TP concentration changes in polymictic reservoir Lake Mitchell, SD.

			TP (mg/L)		Lake Volume	Internal TP Load	
	End-Date	Days	TP at Date	Change	10^6 m^3	(kg)	($mg/m^2/su$)
1991	19-Aug	60	0.208	0.058	11.43	662	226
1992	26-Aug	42	0.282	0.117	12.43	1455	476
1993	18-Aug	67	0.620	0.421	13.70	5768	1775
1994	13-Aug	63	0.372	0.212	12.83	2719	875
1995	11-Aug	59	0.149	−0.121	14.20	0	0
1996	21-Sep	76	0.320	0.055	13.70	1507*	464
1997	14-Aug	70	0.340	0.165	13.49	2227	696
1998	20-Aug	126	0.488	0.140	13.49	2675	591
1999	25-Sep	121	0.320	0.060	13.20	792	249
2000	27-Aug	97	0.770	0.560	12.43	6963	2279
2001	17-Sep	96	0.540	0.327	12.43	4066	1331
Average		**80**	**0.401**	**0.181**	**13.03**	**2621**	**815**
w/o 1995		**82**	**0.426**	**0.211**	**12.91**	**2883**	**896**

Note: The days of release are days between sampling efforts; when the available efforts were not spread to the beginning (1996), adjustments of internal load (kg) were made based on the daily rates (*). In some years, TP concentration dropped in September; therefore estimates recorded only until August were not adjusted and may underestimate internal load of the whole summer.

In strongly stratified lakes, L_{int_1} is the difference between the end-of-summer hypolimnetic or whole water column TP mass ($P\_t_2 \times V\_t_2$) and the respective TP mass at the beginning of the anoxic period ($P\_t_1 \times V\_t_1$), divided by lake area. Any winter internal load under ice could be determined in a similar fashion.

In polymictic and weakly stratified lakes with early thermocline erosion, or where anoxia approaches the thermocline, the maximum summer whole water column mass of the anoxic period may be more appropriate to use as final mass ($P\_t_2 \times V\_t_2$), rather than a mass from the end of summer, to account for entrainment of sediment released P into the surface water and subsequent loss though the outflow. Such calculations are presented for polymictic Lake Mitchell, where the inflows were low or not existent in the summer (Fig. 3; Table 3).

When external water and TP inputs and exports are variable and high during the period of computation, the unsettled portion of external load should be included:

$$L_{int\_1} = \frac{TP\_t_2 \times V\_t_2}{A_o\_t_2} - \frac{TP\_t_1 \times V\_t_1}{A_o\_t_1} - L_{ext\_t1-2} \times (1 - R_{sed}) + L_{out\_1-2} \qquad (2)$$

where, L_{ext_t1-2} is external load for the specified period; L_{out_t1-2} is export for the specified period; and R_{sed} is proportion of settling load as defined in the following section.

In a related approach (Moore and Christensen 2009), partial P budgets throughout the growing season were calculated for each sampling event, and those indicating an internal source were summed. In this way, several internal loading occasions were documented and quantified and provided some evidence of the extent of internal loading. Such an estimate is also partially net because it does not take any previous settling into account.

As in any lake characteristic calculated from seasonally and spatially variable data, the involved monitoring frequency, quality and quantity all contribute to the accuracy of the *in situ* internal load assessment. This contributes to the commonly high variability observed in L_{int_1}.

Method 2: Annual TP mass balance (L_{int_2})

A net estimate of internal load can be obtained from an annual TP mass balance in stratified and polymictic lakes (Nürnberg 1998, Nürnberg and LaZerte 2001). Because it is based on an annual mass balance, L_{int_2} includes summer and winter loads. This process is data intensive because it requires annual input and export data for both TP and water. When winter data are not available, L_{int_2} can be approximately estimated from summer mass balances alone (Welch and Jacoby 2001).

The main complication is that upward (internal load) and downward (settling) fluxes are combined in a lake budget.

This means that the proportion of P that is retained and can be determined from mass balances as measured retention (R_{meas}, equation 3) yields net estimates of retention:

$$R_{meas} = (\text{in} - \text{out})/\text{in} \quad or$$
$$R_{meas} = (L_{ext} - L_{out})/L_{ext} \tag{3}$$

where L_{ext} is annual areal external TP load, and L_{out} is annual areal TP export via the outflow, both as mg/m^2/yr. According to equation 3, $R_{meas} \times L_{ext}$ equals the net amount of TP retained by the lake ($L_{ext} - L_{out}$), or the difference between sedimentation to and release from the sediments.

Accordingly, only if sedimentation is known can internal load be determined from mass balance studies because sedimentation occurs in all but fast flushing reservoirs and is often quite high. For example, in approximately 100 of 305 lakes at least 60% of the incoming P mass was retained (Brett and Benjamin 2008). Even when P export exceeds input by a certain amount on an annual basis (in lakes with high internal loading after sewage diversion; Sas 1989) it would be wrong to suppose that only this marginal amount is due to sediment release. Such an assumption would underestimate internal load because a certain fraction (the exact amount can be modeled by R_{sed} depending on hydrology and morphometry) of both external and internal load has settled over the year.

Average P settling or sedimentation rates are extremely hard to determine in lakes and therefore are almost always modeled instead. Consequently, Method 2 largely depends on a retention model that determines the part that settles (R_{sed}) so that downward fluxes can be separated from upward fluxes of TP. Such models are described in detail in Problematic retention.

Because it only represents the (downward) sedimentation, predicted retention is larger than measured retention in lakes with internal loads (Fig. 5), approximately by the amount of P released from the sediments, so that the difference of the retentions quantifies the net upward flux out of the sediment or net L_{int_2} (Nürnberg 1984b):

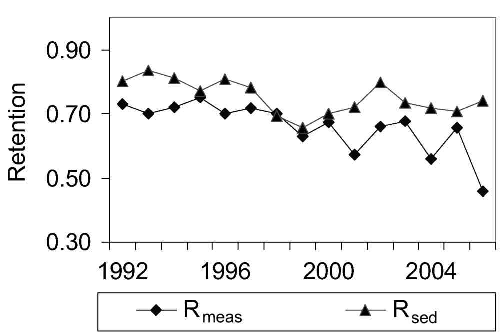

Figure 5.-Comparison of measured net P retention (R_{meas} from mass balance) and predicted gross retention (R_{sed}) for Cherry Creek Reservoir, Denver, CO (Table 2). Differences are due to internal load.

Table 4.-Comparison of long-term median net and gross internal load estimates (equation 5) according to Method 2 in several water bodies. Years indicates the sample size as the number of years with mass balance estimates.

Lake or Reservoir	Mixing State	Gross (mg/m^2/yr)	Net (mg/m^2/yr)	Years (#)
Lake Wilcox	Dimictic	280	62	9*
Cherry Creek Reservoir	Polymictic	255	59	15
Lake Pyhäjärvi	Partially polymictic	66	13	26

*Based on modeled external load.

$$\text{Net } L_{int\_2} = L_{ext} \times (R_{sed} - R_{meas}) \tag{4}$$

The conversion of net to gross estimates is possible when the proportion of internal load that has settled within the year is known. The annual retained (settled) proportion of internal load is not dissimilar to that of external load according to Nürnberg (1998), so that gross estimates of L_{int_2} can be computed:

$$\text{Gross } L_{int\_2} = \text{Net } L_{int\_2}/(1 - R_{sed}) \tag{5}$$

Note that values of net and gross loads are extremely different (Table 4), and mixing net, partially net and gross loads severely compromises the quantification of internal load.

Method 3: Product of active (anoxic) sediment area and P release rate (L_{int_3})

A more direct method of determining gross internal load is based on its components, the P releasing area and rate. Anoxic release rates (RR) have been shown to yield gross internal load when multiplied by an expression for the releasing sediment area in stratified (Nürnberg 1987) and polymictic lakes (Nürnberg 2005). In this method, gross internal load is determined separately for summer and winter and then summed for an annual estimate:

$$L_{int\_3} = L_{int\_summer} + L_{int\_winter} \tag{6}$$

In stratified lakes, summer internal load is determined as the product of release rate and anoxic factor according to (Nürnberg 1987):

$$L_{int\_summer} = RR \times AF_{summer} \tag{7}$$

where RR represents the areal release rate of P from the anoxic sediment surface and AF_{summer} is the anoxic factor computed from dissolved oxygen profiles throughout the stratification period (Nürnberg 2004).

In polymictic lakes summer internal load (equation 8) is determined as the product of release rate and the representation of the actively releasing area modeled as anoxic factor, AF_{pred} (equation 9, Nürnberg 1996):

$$L_{int\_summer} = RR \times AF_{pred} \tag{8}$$

$$AF_{pred} = -36.2 + 50.2 \log(TP_{summer}) + 0.762\ z/A_o^{0.5} \tag{9}$$

where TP_{summer} is average summer TP, and $z/A_o^{0.5}$ is a morphometric factor (with z, mean depth in m and A_o, lake surface area in km^2). Equation 9 is based on summer TP averages (rather than annual TP, as used in Nürnberg 2005) because often only summer TP values are available. The actively releasing area has to be modeled because the P-releasing areas are not restricted to those overlain by anoxic water in polymictic lakes as discussed earlier (Unnoticed internal load).

A base (or average) areal release rate for stratified or polymictic lakes can be determined in the lab, or predicted from sediment TP concentration, reductant-soluble P fractions, or lake trophic state (Nürnberg 1988) to yield a long-term average internal load.

P release rates and releasing area are influenced by temperature (Jensen and Andersen 1992), and variability in internal load due to varying temperature between years is obvious in polymictic lakes (Nürnberg, unpubl. data). To include this variability, both AF_{pred} and RR can be adjusted with respect to the summer mixed layer water temperature to follow the Q10 rule of Van Holst:

$$RR_i = RR \times c^{(t\_i - t\_avg)/10} \tag{10}$$

where RR_i is the average daily release rate for $year_i$ based on a given summer average lake temperature t_i; t_{avg} is the average summer temperature of all study years; and *c* is the Van Holst constant or Q10 value, usually close to c = 3, that can be calibrated (Nürnberg *et al*. in prep.).

Winter internal load is typically small because the low temperature decreases oxygen consumption and P release. If necessary, it can be determined in a manner similar to summer internal load from the product of the winter anoxic factor and a winter release rate (equation 11), although the RR and AF models do not apply to winter temperatures. The AF_{winter} can be computed from oxygen profiles under ice, and winter release rates (RR_{winter}) can be either measured in the lab at *in situ* temperature or predicted from the summer release rate average according to the Q10 rule.

$$L_{int\_winter} = AF_{winter} \times RR_{winter} \tag{11}$$

Different from the other two methods, submodels are used that were originally developed for P release from anoxic sediments. Therefore, if other internal sources exist, for example those of senescing macrophytes (Graneli and Solander 1988), littoral P release (Cyr *et al*. 2009), benthivorous fish (Sereda *et al*. 2008), and groundwater, these sources may have to be quantified independently. However, they are usually small and negligible compared to the sediment derived portion (Sereda *et al*. 2008).

Comparison

Quite often only one of the three methods presented here can be applied, and sometimes only for an overall long-term average because detailed data are seldom available. In such cases the large uncertainty of any internal load estimate has to be considered. In addition, long-term annual estimates produced by the three methods can be expected to differ from each other because of the various uncertainties discussed previously. However, if the overall long-term averages are similar, while considering differences in net versus gross estimates, it lends credibility to the estimates derived from the different methods.

In particular, L_{int_1} may be less exact in polymictic lakes and can always be expected to be lower than the other two methods because it is partially net. The method of choice may be L_{int_2} in cases where in- and outflows have been monitored intensely, as in important reservoirs. L_{int_3} does not capture much annual variability when its components are largely predicted from average lake TP concentrations, and the other methods are more useful if estimates of long-term trends and annual variability are required.

All three methods were used in the internal load assessment of polymictic Cherry Creek Reservoir near Denver, Colorado, where hard water and a bottom outlet influenced predictions of sedimentation (Fig. 6, Table 2, and discussed in Modeling lake TP concentration) and of polymictic Lake Pyhäjärvi, where the TP cycle was complicated by biomanipulation efforts (Fig. 7 and Nürnberg *et al*. in prep.). While differences between estimates are due to errors in the input data as well as model structure, differences with respect to L_{int_1} may be due to gross versus net estimates to a varying degree. It is therefore important to determine whether a load

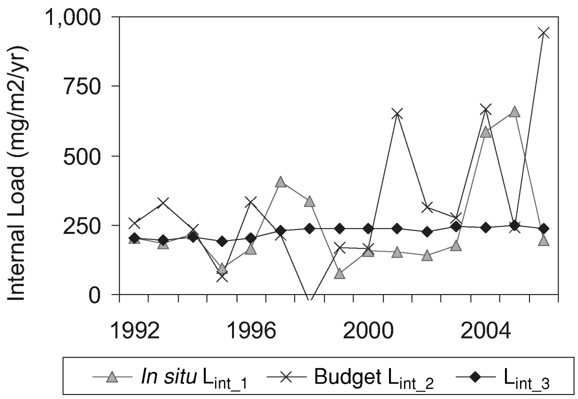

Figure 6.-Comparison of different internal load estimates for polymictic Cherry Creek Reservoir, Denver, CO. L_{int_3} estimates are based on a constant RR and moderately variable active areas.

estimate represents a gross, net, or partially net estimate; only then can it be appropriately used for TP predictions.

Modeling lake TP concentration in lakes with internal load

The main reason for quantifying internal load is to determine its effect on the trophic state and water quality of a lake. In such an attempt, internal load is best compared to external load and explicitly included in a model that predicts lake TP concentration.

Empirical P mass balance modeling approaches are the most often used and are readily available. In a generic mass balance (equation 12, areal external P load divided by areal

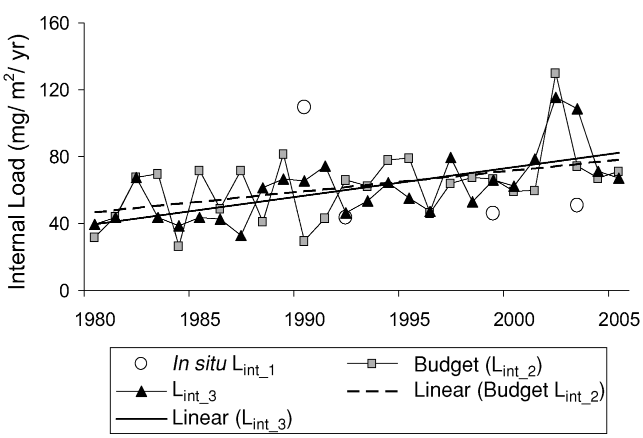

Figure 7.-Comparison of different internal load estimates in partially polymictic Lake Pyhäjärvi, Finland. L_{int_3} estimates include annual changes in summer temperature. Note that there was a variable biomanipulation effort between years with fish catch exporting as much TP as the outflow. An increasing trend with time is detectable for both estimates ($R^2 = 0.21$ for L_{int_2}; $R^2 = 0.43$ for L_{int_3}).

waterload of the outflow, corrected for retention, R_{meas} as in equation 3; Dillon and Rigler 1974), annual average outflow TP concentrations, which are deemed to be similar to annual surface concentrations, are computed from the measured inputs and outputs:

$$TP = \frac{L_{ext}}{q_s} \times (1 - R_{meas}) \tag{12}$$

However, models of this structure do not always predict TP concentrations in lakes with internal load accurately because of un-met assumptions and the need of modeling retention.

Problems using outflow

Being based on P export, the mass balance is solved for annual average outflow P concentration, which is assumed to be similar to annual average lake P concentration; however, lake P concentrations during the growing season are the most important and determine the trophic state. In a lake with seasonal P inputs like internal load, summer P concentrations would be underestimated because they are usually higher than annual average (surface) outflow concentration, especially in polymictic lakes. Also, the mass balance does not apply to lakes and reservoirs with a mid-depth or bottom outlet. In these cases P concentration of the surface mixed layer would be overestimated because much of the P released from sediments at greater depths is lost through the outflow.

Problematic retention

Even if the assumptions of the mass balance in equation (12) are fulfilled, this method is rarely applicable because detailed P budgets including retention are not often available. Consequently, retention models (R-models) have been developed, and their performance has been discussed since their explicit initial definition (Dillon and Rigler 1974) because of their uncertainty in lake and reservoir P modeling (Søndergaard *et al*. 2001, Hejzlar *et al*. 2006, Brett and Benjamin 2008).

Retention measured from mass balances as R_{meas} (equation 3) is a net estimate and includes upward and downward fluxes. Accordingly, R-models developed on mass balances from these lakes also represent net estimates. When models were developed with data that included lakes with any internal loading, the upward flux partially canceled out the downward flux so that flux issues were confused, and the separation between sedimentation and internal load was prevented. Regrettably, many R-models include internal load to an unspecific degree, depending on the original lakes that were used in model development.

As a result, TP concentration tends to be underestimated if such an R-model is applied to a lake with internal load; however, TP is overestimated if the model is applied to a lake without internal load. These R-models have been widely used for a number of years (see review by Brett and Benjamin 2008), despite the known problem (Nürnberg 1984b). To avoid the most severe values of internal load, sometimes water bodies with negative R (more export than input) were excluded when developing a model (Hejzlar *et al*. 2006) or only oligotrophic and mesotrophic lakes were used (Larsen and Mercier 1976). But lakes with less apparent internal loading were still included.

Low predictability of previous R-models has spawned a number of attempts to improve them by expanding the dataset, often combining the lakes of former studies on R-models (Brett and Benjamin 2008), by using different and more elaborate statistical approaches (Prairie 1989), by investigating the importance of specific variables in a hierarchical analysis (Cheng *et al*. 2009), by distinguishing between reservoirs and lakes (Straskraba 1998, Hejzlar *et al*. 2006) and by including additional variables that address the outflow problem (e.g., ratio of outflow to annual average lake P concentration; Schauser and Chorus 2007).

One general approach with some mechanistic appeal is based on the annual average settling velocity, v (m/yr), that can be constant or variable, and annual water load, q_s (equation 13). I describe it here in more detail because it makes theoretical sense to use an expression of v for the prediction of sedimentation (without upward fluxes) in an R-model:

$$R_v = v/(v + q_s) \tag{13}$$

Constant values between 10 and 30 m/yr for v were found empirically to fit annual mass balances in lakes and reservoirs that included some lakes with internal loads and hypolimnetic anoxia (reviewed in Nürnberg 1984b). In a study where more than half the lakes were eutrophic (and likely releasing P from their sediments), optimization of v yielded a small value of 5.1 m/yr, as can be expected because downward fluxes are partially canceled by upward fluxes in such lakes (Brett and Benjamin 2008).

A specifically designed R-model that predicts only sedimentation (and not release) was developed from a more general form of equation 13, using two different constants. The constants were calibrated with a dataset of lakes with no or low potential of sediment P release because there was no evidence of hypolimnetic anoxia (Nürnberg 1984b). In such lakes, R_{meas} (equation 3) essentially describes the proportion that settles because it can be assumed that internal load does not occur or is only marginal. The specific R-model (R_{pred}, equation 14) developed for these lakes therefore represents the downward flux of TP due to settling and sedimentation:

$$R_{pred} = 15/(18 + q_s) \tag{14}$$

While the parameters in R_{pred} (equation 14) seem to predict sedimentation well in many lakes (Nürnberg 1984b), it underestimates P sedimentation in certain lake types. For example, settling is enhanced in hardwater lakes experiencing P-calcite co-precipitation. Retention averaged 39% higher than predicted by R_{pred} in stratified, calcium-rich alpine lakes with 11–50 mg/L Ca (Nürnberg 1998). In some of these cases, calibration to find a more appropriate value of v can produce an adequate R-model. For example, average annual settling velocity was 25.5 m/yr in a study on 28 stormwater detention ponds and urban lakes designed to settle solids; it was also higher in run-of-the-river reservoirs (Nürnberg, unpublished studies).

In some cases where R-models based on a constant v have not been successful, those based on a variable v were more useful. For example, models especially designed for shallow lakes and reservoirs relate v to annual water detention time, τ ("tau," also called water residence time, with units in year, measured as lake volume divided by annual outflow volume) and q_s, along with a constant k:

$$v = k \times q_s \times \sqrt{\tau} \tag{15}$$

Inserting this v relationship (equation 15) into equation 14 with simplification (equation 16) leads to the following retention model, R_τ (equation 17):

$$R_\tau = \frac{1}{1 + \frac{1}{k\sqrt{\tau}}} \tag{16}$$

$$R_\tau = \frac{k\sqrt{\tau}}{1 + k\sqrt{\tau}} \tag{17}$$

The original relationship was developed for natural lakes with k = 1 (Larsen and Mercier 1976). An Organisation for Economic Co-operation and Development (OECD) study of 43 European, Australian and Japanese man-made reservoirs and some shallow lakes determined k = 2 (Clasen 1980). A study with 119 records of European and North American reservoirs that ranged from deep to shallow and oligotrophic to eutrophic determined k = 1.84 (Hejzlar *et al*. 2006). This study also found, like previous studies (Straskraba 1998, Nürnberg, unpubl. data), that retention of reservoirs is far higher than retention in natural lakes.

As in most other discussed models, there was no provision in these models to accommodate internal load separately from sedimentation, although, for example, more than 70% of the lakes and reservoirs in the OECD dataset were eutrophic with lake TP concentrations between 30 and 100 μg/L. Furthermore, sediment P release was deemed to occur in the more eutrophic OECD systems (Clasen 1980) as well as in the Hejzlar *et al.* (2006) study, although they excluded reservoirs with obvious and large amounts of internal load (determined from negative net retention in the mass balance). Consequently, it can again be argued that retention, and in particular k, is underestimated in most of these models because of the omission of sediment-released P, and that their computed retention is actually a net estimate that includes upward fluxes.

The R-models discussed here were developed on long-term averages for several years so that they do not necessarily apply to individual lakes and even less to specific years. By using models that include a term that relies on lake-specific hydrology, like q_s (equation 14) or τ (equation 17), annual variances in hydrology and climate can be captured and modeled even for a given year of a specific lake.

Therefore, a lake-specific calibration of R_τ (equation 17), for example, can create a valid R-model as long as internal load is considered. Eutrophic, hardwater Cherry Creek Reservoir near Denver, Colorado (Table 2), has a bottom outlet, and therefore average summer TP (Jul–Sep) could not be predicted from available R-models because the simple R_{pred} model (equation 14) did not apply. Using 15 years of TP budgets, the most appropriate R-model was determined to be of the R_τ form (equation 17) with a *k* value of 2.7. Calibration was done simultaneously with independent estimates of internal load (L_{int_1}; Fig. 6) and seasonal TP values (see next section).

For the reasons discussed here, any R-model must be tested and, if necessary, calibrated to accurately predict TP concentrations in a specific lake or reservoir. I recommend testing the simple R_{pred} model first and if results are considered inaccurate, to calibrate v in the model of the type R_v (equation 13) or R_τ (equation 17). In the end, and despite theoretical considerations, the model that fits the data is the best or most appropriate model in any particular case (Chapra 1997) and may have to be determined by trial and error (testing). Consequently, model testing (and hopefully verification) with several seasons of lake TP concentration data is essential.

Application of R_{sed} in P model

The prediction of lake TP concentration, the ultimate goal for the development of R-models, can be accomplished by adding an internal loading term and replacing R_{meas} with R_{sed} in equation 12, where R_{sed} is any R-model specific to downward fluxes, such as R_{pred} (equation 14). In this way, both upward and downward fluxes are considered, and specific seasonal or annual TP concentrations can be predicted in stratified and polymictic lakes according to Nürnberg (1998, 2005).

The model for average TP concentration is based on the same term of retention for both external and internal loads and TP is predicted using:

$$TP = \frac{L_{ext} + L_{int}}{q_s} \times (1 - R_{sed}) \tag{18}$$

This model predicts annual average TP in stratified lakes with and without internal loading (Fig. 8) and growing season concentrations of polymictic lakes (Nürnberg 1998). This formula also seems to be the most useful for the calibration of a lake specific R-model as described in the previous section (Fig. 9).

Because of the seasonality of sediment release, early summer TP minimum can be predicted without considering internal load (equation 19) and a late-fall maximum that does consider the entire gross internal load (equation 20). However, these two models usually deliver extreme values (too small or too large) and are mainly used to indicate ranges (Nürnberg 1998):

$$TP_{min} = \frac{L_{ext}}{q_s} \times (1 - R_{sed}) \tag{19}$$

$$TP_{max} = \frac{L_{ext}}{q_s} \times (1 - R_{sed}) + \frac{L_{int}}{q_s} \tag{20}$$

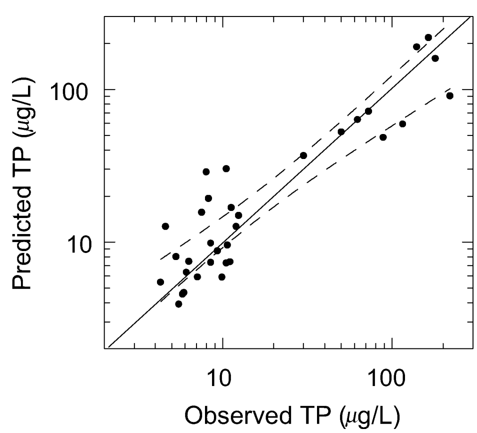

Figure 8.-Comparison of observed summer TP averages with those predicted from the TP model (equation 18 with $R_{sed} = R_{pred}$ of equation 14 and L_{int}) for 33 non-alpine lakes with data from Nürnberg (1998; $R^2 = 0.84$, $p < 0.0001$, slope is not significantly different from one and t-test on differences is not significant, $p = 0.383$). The line of perfect prediction (solid) and the 95% confidence band around the regression line (broken) are shown.

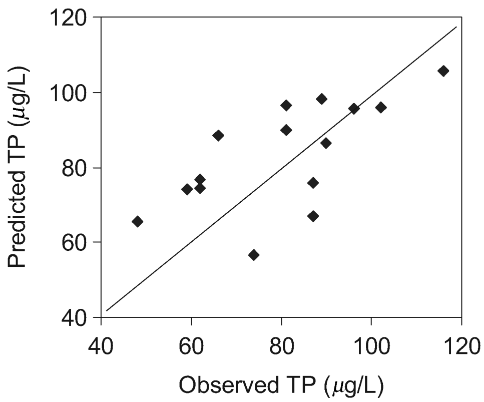

Figure 9.-Comparison of observed Jul-Sep TP averages with those predicted from the TP model (equation 18 with $R_{sed} = R_\tau$ of equation 17 and k = 2.7 and L_{int_1}) for 15 years of polymictic Cherry Creek Reservoir, Denver, CO ($p < 0.01$, $R^2 = 0.38$, n = 15). The line of perfect prediction is shown.

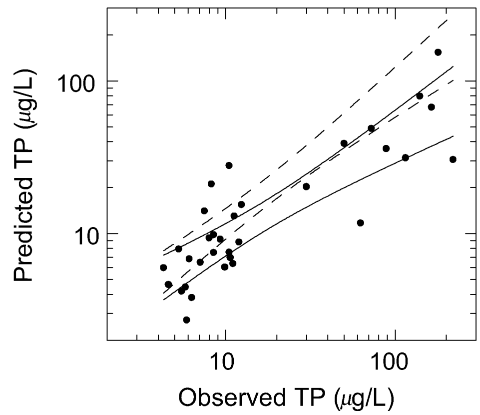

Figure 10.-Comparison of observed summer TP averages with those predicted from the Vollenweider model (with R_v of equation 13 for v = 10, and no provision for internal load) for 44 stratified, non-alpine lakes (including 33 of Fig. 8, computed with data from Nürnberg, 1998. $R^2 = 0.72$, p = 0.0001, n = 44, slope = 0.697, SE = 0.068 significantly different from one, and t-test on differences significant, $p < 0.05$). The 95% confidence bands are shown (solid). Bands of the TP model of Fig. 8, are shown for comparison (broken).

As with the R models, TP models that do not consider internal load are not applicable to eutrophic lakes (with internal load). For example, when the original and most influential TP model is used (that does not consider internal load specifically; Vollenweider 1976), TP concentrations are underestimated in lakes where internal load is large (Fig. 10; Welch and Jacoby 2001). This is a well-known and acknowledged problem (Vollenweider, Canadian Centre of Inland Waters, pers. comm., 1982; Larsen and Mercier 1976, Hejzlar *et al.* 2006, Brett and Benjamin 2008), discrediting all P mass balance modeling in lakes and reservoirs. Of course the main reason for using models that include downward and upward fluxes in one expression is the difficulty of quantifying internal load. If the different methods (L_{int_1}, L_{int_2}, L_{int_3}) provided here are applied, only the downward flux has to be predicted by R-models.

Recommendations (summary and conclusion)

Remediation and restoration of lakes with unsatisfactory water quality is challenging. The first and only pretreatment investigation in eutrophic lakes usually includes the determination of the different P sources. While external sources are generally more obvious, internal sources are harder to quantify and are often ignored. In particular, internal P loading as sediment-released P can be large but undetected. Internal load seems to be more effective and harmful in shallow polymictic lakes than in deep stratified lakes, even if more difficult to identify. Destratification and aeration treatments that effectively turn stratified lakes into polymictic ones and mix sediment-released P into the trophic zone often increase TP concentration and algae biomass, even favoring cyanobacteria (Nürnberg *et al.* 2003; reviewed in Cooke *et al.* 2005). But internal load also negatively affects the trophic state in stratified lakes. When internal load was moderated with the restoration treatment of hypolimnetic withdrawal, water quality improved as long as stratification was maintained. As expected, however, this treatment was not beneficial in polymictic lakes where sediment-released P was mixed throughout the water column (Nürnberg 2007).

The following suggestions may simplify studies on lakes with internal loading:

- Distinguish between stratified (i.e., mono- and dimictic lakes) and shallow polymictic lakes.
- Use TP for the quantification of internal load to avoid analytical problems inherent to the analysis of DRP and other P fractions.
- Reserve the term "internal load" for the gross P mass released from sediment surfaces and express it per lake surface area and per year (L_{int}, $mg/m^2/yr$) to render it comparable and analogous to the definition of external P load (L_{ext}, $mg/m^2/yr$). It can be quantified from mass balance after correcting for its sedimentation (conversion of net into gross) or from the active area and release rate (AF × RR). Determination from *in situ* summer increases delivers a partially net estimate. Internal load should correspond to the specific study year or average of a period of years; this definition does not consider the previous contribution of external load to the current internal load.
- Use internal load estimates in TP mass balance models, so that downward fluxes and separated from

upward fluxes and only the downward flux has to be predicted.

- Predict downward fluxes with a retention model (R_{sed}), test and calibrate R_{sed}, if necessary, with independent internal load estimates and seasonal lake TP concentration.

Glossary – definition of terms (note that all terms refer to lakes and reservoirs)

V (10^6 m^3) : lake *volume*

A_o (10^6 m^2) : lake *surface area*

q_s (m/yr) : *annual areal water load*, annual outflow volume (Q, m^3) per surface area (A_o, m^2), where $q_s = Q/A_o$.

τ(yr) : *annual water detention time* or *annual water residence time*, lake volume (V) divided by annual outflow volume (Q), where $\tau = V/Q$.

v (m/yr) : *settling velocity*, average distance that TP settles downward within one year.

L_{ext} ($mg/m^2/yr$) : *external* (gross) *load.*

L_{int} ($mg/m^2/yr$) : *internal* (gross) *load.*

R_{meas} : *measured* or observed *retention*, proportion of external load (measured as *in–out/in*, equation 3)

R_{sed} : *predicted retention,* proportion of settling loads, generic expression that is predicted by models for specific lakes (e.g., equation 14: $R_{pred} = 15/(18 + q_s)$.

Acknowledgment

This manuscript is based on workshops and various presentations (SIL and NALMS 2007, 2009) and the discussion with participants and audience is greatly appreciated. The critical review and helpful comments from my colleagues, Chris Holdren, Bruce LaZerte, Barry Moore, Eugene Welch, and one anonymous reviewer greatly improved the manuscript. Data provision from various contract work is acknowledged for the case studies listed in Table 2.

References

Auer, M. T., S. M. Doeer, S.W. Effler, and E. M. Owens. 1997. A zero degree of freedom total phosphorus model; 1. Development for Onondaga Lake, New York. Lake Reserv. Manage. 13:118-130.

Brett, M. T. and M. M. Benjamin. 2008. A review and reassessment of lake phosphorus retention and the nutrient loading concept. Freshwater Biol. 53:194-211.

Chapra, S. C. 1997. Surface water-quality modeling. McGraw-Hill Companies, Inc., Boston. 844 p.

Cheng, V., G. Arhonditsis, and M. Brett. 2009. A revaluation of lake-phosphorus loading models using a Bayesian hierarchical framework. Ecol Res. 0912-3814 (Print) 1440-1703 (Online).

Clasen, J. 1980. Shallow lakes and reservoirs. Oganisation for Economic Co-operation and Development. OECD:246.

Cooke, G. D., E. B. Welch, S. A. Peterson, and S. A. Nichols. 2005. Restoration and management of lakes and reservoirs. CRC, Boca Raton, FL, USA. 576 p.

Coveney, M. F., E. F. Lowe, L. E. Battoe, E. R. Marzolf, and R. Conrow. 2005. Response of a eutrophic, shallow subtropical lake to reduced nutrient loading. Freshwater Biol. 50:1718-1730.

Cyr, H., S. K. McCabe, and G. K. Nürnberg. 2009. Phosphorus sorption experiments and the potential for internal phosphorus loading in littoral areas of a stratified lake. Water Res. 43:1654-1666.

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

Einsele, W. 1936. Über die Beziehungen des Eisenkreislaufs zum Phosphatkreislauf im eutrophen See. Arch. Hydrobiol. 29:664-686.

Gächter, R. and J. S. Meyer. 1993. The role of microorganisms in mobilization and fixation of phosphorus in sediments. Hydrobiologia 253:103-121.

Gächter, R. and B. Müller. 2003. Why the phosphorus retention of lakes does not necessarily depend on the oxygen supply to their sediment surface. Limnol. Oceanogr. 48:929-933.

Graneli, W. and D. Solander. 1988. Influence of aquatic macrophytes on phosphorus cycling in lakes. Hydrobiologia 170:245-266.

Håkanson, L. 2004. Break-through in predictive modelling opens new possibilities for aquatic ecology and management – a review. Hydrobiologia 518:135-157.

Hejzlar, J., K. Šamalova, P. Boers, and B. Kronvang. 2006. Modelling phosphorus retention in lakes and reservoirs. Water Air Soil Pollut. 6:487-494.

Holdren, G. C. and D. E. Armstrong. 1980. Factors affecting phosphorus release from intact lake sediment cores. Environ. Sci. Technol. 14:79-87.

Hupfer, M., S. Gloess, and H.-P. Grossart. 2007. Polyphosphate-accumulating microorganisms in aquatic sediments. Aquat. Microb. Ecol. 47:299-311.

Hupfer, M. and J. Lewandowski. 2005. Retention and early diagenetic transformation of phosphorus in Lake Arendsee (Germany) – consequences for management strategies. Arch. Hydrobiol. 164:143-167.

Hupfer, M. and J. Lewandowski. 2008. Oxygen controls the phosphorus release from lake sediments – a long-lasting paradigm in limnology. Int. Rev. Hydrobiol. 93:415-432.

Jensen, H. S. and F. O. Andersen. 1992. Importance of temperature, nitrate, and pH for phosphate release from aerobic sediments of four shallow, eutrophic lakes. Limnol. Oceanogr. 37:577-589.

Larsen, D. P. and H. T. Mercier. 1976. Phosphorus retention capacity of lakes. J. Fish. Res. Board Can. 33:1742-1750.

Lathrop, R. C., S. R. Carpenter, and D. M. Robertson. 1999. Summer water clarity responses to phosphorus, Daphnia grazing, and internal mixing in Lake Mendota. Limnol. Oceanogr. 44:137-146.

Mataraza, L. K. and G. D. Cooke. 1997. A test of a morphometric index to predict vertical phosphorus transport in lakes. Lake Reserv. Manage. 13:328-337.

Moore, B. C. and D. Christensen. 2009. Newman Lake restoration: A case study. Part I. Chemical and biological responses to phosphorus control. Lake Reserv. Manage. 25:337-350.

Mortimer, C. H. 1941. The exchange of dissolved substances between mud and water in lakes. J. Ecol. 29:280-329.

Murphy, J. and J. P. Riley. 1962. A single-solution method for the determination of phosphate in natural waters. Anal. Chim. Acta 27:31-36.

Nürnberg, G. K. 1984a. Iron and hydrogen sulfide interference in the analysis of soluble reactive phosphorus in anoxic waters. Water Res. 18:369-377.

Nürnberg, G. K. 1984b. The prediction of internal phosphorus load in lakes with anoxic hypolimnia. Limnol. Oceanogr. 29:111-124.

Nürnberg, G. K. 1985. Availability of phosphorus upwelling from iron-rich anoxic hypolimnia. Arch. Hydrobiol. 104:459-476.

Nürnberg, G. K. 1987. A comparison of internal phosphorus loads in lakes with anoxic hypolimnia: laboratory incubations versus hypolimnetic phosphorus accumulation. Limnol. Oceanogr. 32:1160-1164.

Nürnberg, G. K. 1988. Prediction of phosphorus release rates from total and reductant-soluble phosphorus in anoxic lake sediments. Can. J. Fish. Aquat. Sci. 45:453-462.

Nürnberg, G. K. 1996. Trophic state of clear and colored, soft- and hardwater lakes with special consideration of nutrients, anoxia, phytoplankton and fish. Lake Reserv. Manage. 12:432-447.

Nürnberg, G. K. 1998. Prediction of annual and seasonal phosphorus concentrations in stratified and polymictic lakes. Limnol. Oceanogr. 43:1544-1552.

Nürnberg, G. K. 2004. Quantified hypoxia and anoxia in lakes and reservoirs. TheScientificWorldJo 4:42-54.

Nürnberg, G. K. 2005. Quantification of internal phosphorus loading in polymictic lakes. Verh. Internat. Verein. Limnol. 29:623-626.

Nürnberg, G. K. 2007. Lake responses to long-term hypolimnetic withdrawal treatments. Lake Reserv. Manage. 23:388-409.

Nürnberg, G. K. and B. D. LaZerte. 2001. Predicting lake water quality. P. 139-163. *In* C. Holdren, W. Jones and J. Taggart J (eds.). Managing lakes and reservoirs. North American Lake Management Society, Terrene Institute in cooperation with Office Water Assessment Watershed Protection Division U.S. Environ. Prot. Agency, Madison, WI.

Nürnberg, G. K, B. D. LaZerte, and D. D. Olding. 2003. An artificially induced Planktothrix rubescens surface bloom in a small kettle lake in southern Ontario compared to blooms world-wide. Lake Reserv. Manage. 19:307-322.

Nürnberg, G. K. and R. H. Peters. 1984a. Biological availability of soluble reactive phosphorus in anoxic and oxic freshwater. Can. J. Fish. Aquat. Sci. 41:757-765.

Nürnberg, G. K. and R. H. Peters. 1984b. The importance of internal phosphorus load to the eutrophication of lakes with anoxic hypolimnia. Verh. Internat. Verein. Limnol. 22:190-194.

Prairie, Y. T. 1989. Statistical models for the estimation of net phosphorus sedimentation in lakes. Aquat. Sci. 51:192-210.

Prairie, Y., C. de Montigny, and P. del Giorgio. 2001. Anaerobic phosphorus release from sediments: a paradigm revisited. Verh. Internat. Verein. Limnol. 27:4013-4020.

Sas, H. 1989. Lake restoration by reduction of nutrient loading: expectations, experiences, extrapolations. Hans Richarz Publikations-Service, Sankt Augustin, Germany. 497 p.

Schauser, I. and I. Chorus. 2007. Assessment of internal and external lake restoration measures for two Berlin lakes. Lake Reserv. Manage. 23:366-376.

Sereda, J. M., J. J. Hudson, and P. D. McLoughlin. 2008. General empirical models for predicting the release of nutrients by fish, with a comparison between detritivores and non-detritivores. Freshwater Biol. 53:2133-2144.

Søndergaard, M., J. P. Jensen, and E. Jeppesen. 2001. Retention and internal loading of phosphorus in shallow, eutrophic lakes. TheScientificWorldJo 1:427-442.

Søndergaard, M., J. P. Jensen and E. Jeppesen. 2005. Seasonal response of nutrients to reduced phosphorus loading in 12 Danish lakes. Freshwater Biol. 50:1605-1615.

Steinman, A. D. and M. Ogdahl. 2008. Ecological effects after an alum treatment in Spring Lake, Michigan. J. Environ. Qual. 37:22-29.

Straskraba, M. 1998. Limnological differences between deep valley reservoirs and deep lakes. Int. Rev. Gesamt. Hydrobiol. 83:1-12.

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

Welch, E. B. and J. M. Jacoby. 2001. On determining the principal source of phosphorus causing summer algal blooms in western Washington Lakes. Lake Reserv. Manage. 17:55-65.