

Internal phosphorus loading in Iowa lakes:

Causes, consequences, and management recommendations

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OVERVIEW OF DATA SOURCES AND REPORT AIMS

The primary aim of this report is to synthesize the current understanding of lacustrine phosphorus (P) cycling as it applies to Iowa lakes and small impoundments in Part I. The analyses presented here form a critical foundation for making data-based management recommendations in Part II of the report. We have leveraged a variety of data sources to characterize P cycling dynamics in Iowa lakes and reservoirs, identify the dominant pathways for P loading in Iowa waterbodies, and provide context for how these processes operate at continental and global scales. (Figure 1). To provide perspectives on lacustrine P cycling in lakes and reservoirs beyond Iowa, we used data from the EPA National Lakes Assessment (2017) and conclusions from the scientific literature. We conducted a formal synthesis and meta-analysis of internal P loading in lakes and reservoirs worldwide (Albright et al. 2022). In addition to this broad context, we focused our analyses on Iowa-specific data sources including the Ambient Lake and Shallow Lakes Monitoring Programs and more intensive monitoring of seven natural lakes included in the 2018-2020 carp and bigmouth buffalo biomanipulation study (Contract No. 18CRDLWMBALM-0013). We also drew on previous studies of Iowa lakes, including Water Quality Improvement Plans for impaired waterbodies and studies conducted by graduate students and postdoctoral researchers at the Iowa State University (ISU) Limnology Laboratory from 2018-2020. The ISU studies focused on sediment P sources (Albright et al. 2022a), seasonal and spatial variation in internal P loading (Albright and Wilkinson 2022b), macrophyte-hydrodynamic interactions in small waterbodies (Albright et al. in review), and consumer-mediated nutrient cycling (Moody et al. 2022, Butts et al. 2022).

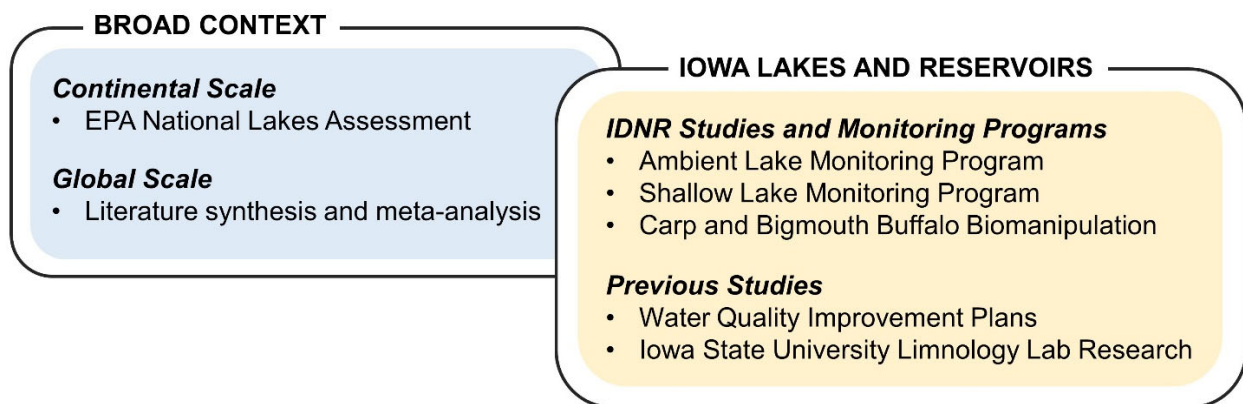


Figure 1. Overview of data sources used to characterize P cycling dynamics at broad spatial scales (left) and data sets focused on Iowa lakes and reservoirs (right).

PART I

Phosphorus-cycling dynamics and dominant pathways in Iowa lakes

SECTION 1 – INTRODUCTION

Phosphorus is one of the elemental building blocks of life. It is a component of DNA (deoxyribonucleic acid), the cellular energy molecules ATP and ADP (adenosine tri- or diphosphate), and the phospholipid bilayer of cell membranes, to name a few examples. Given the importance of P for so many biological structures and processes, the availability of this element has significant consequences for ecosystem function, especially in lakes and reservoirs. P concentrations in lentic ecosystems are tightly coupled to phytoplankton biomass (Dillon and Rigler 1974), which defines the trophic state of the waterbody (Carlson 1977). Due to human land use and climate change, many lakes are becoming enriched in P (eutrophied), which increases the likelihood of harmful algal blooms, deep water oxygen depletion, and a loss of ecosystem services. Phosphorus' essential biological role forms a delicate balance in lakes between limiting ecosystem functioning when there is too little and amplifying a water quality problems when there is too much (Wilkinson and Albright 2022).

1.1 Forms of P in Inland Waters

Elemental P is rare in nature. Instead, P in inland waters usually binds to oxygen in the phosphate molecule (PO_4), which may occur as orthophosphate (PO_4^{3-}) or protonated anions of phosphate (H_2PO_4^- and HPO_4^{2-}). Phosphate is incorporated into many other molecules that are cycled through the biotic and abiotic components of lentic ecosystems. These molecules can be in inorganic forms or as components of organic matter (Figure 2, left panel). The inorganic form is either particulate (i.e., found in minerals or adsorbed to the surface of sediment particles) or dissolved in an aqueous solution. Orthophosphate (PO_4) is a form of dissolved inorganic P that is taken up by plants, algae, and other primary producers. Organic P encompasses a complex set of molecules which can also be dissolved in water or particulate. Some primary producers, such as cyanobacteria, can take up and use dissolved organic phosphorus in addition to orthophosphate (Carey et al. 2012). Particulate organic forms include P contained in the living tissues or necromass of microbes, plants, and animals. These diverse chemical forms of P are found in the water column of lentic ecosystems and in the bottom sediments.

One of the most reported measurements of P in inland waters is the concentration of total P, which is a measure of all particulate, dissolved, organic, and inorganic P contained within a sample (Figure 2, right panel). To measure total P, unfiltered water samples are digested to convert all P forms into orthophosphate. While the total P concentration is a measure of all the P present in a sample, the measurement does not provide specific information on the forms or bioavailability of P present in the ecosystem. Instead, water samples that are filtered to remove particles can be analyzed for dissolved P fractions, such as the commonly reported soluble reactive P (i.e., orthophosphate). Measurements of soluble reactive P reveal more about the abundance of bioavailable P in the environment.

Soluble reactive P concentrations are usually low in the epilimnion of lakes and reservoirs, even though total P concentrations are high. This is due to the P present in the epilimnion being bound in algal and microbial biomass or sorbed to inorganic sediment particles suspended in the water column. Repeated and comprehensive measurements of P concentrations and forms in lacustrine environments can provide insight into the source and dynamic cycling of this element. Ultimately, the source of P in the water column of a lake is from its watershed (external loading) or from the sediments of its ecosystem (internal loading). As the combined magnitude and timing of P loading helps shape aquatic ecosystem dynamics, particularly algal biomass, understanding the sources of P in a lake is key to managing water quality in that ecosystem (Wilkinson and Albright 2022).

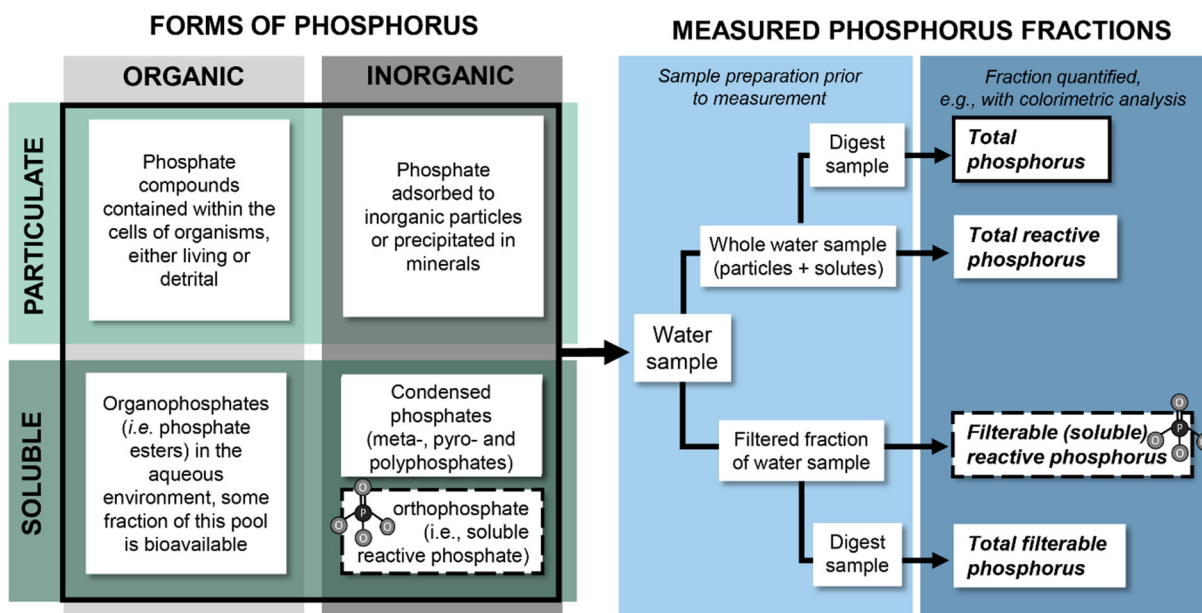


Figure 2. There are many forms of P in aquatic environments that can be functionally separated based on whether the P is in a particulate or soluble form (green rows, left panel) and if it is part of an organic or inorganic molecule (gray columns, left panel). A water sample from the environment can contain one, some, or all these functional forms of P (arrow from left to right panel). The measurement of P concentration is made from either the whole water (particles + solutes) or filtered fraction (solutes + particles smaller than the filter pore size), on either digested or non-digested samples, yielding the P fractions and concentrations measured with colorimetric analysis (right panel). Total P (TP; solid outline), which can contain any number of the functional forms of P, and soluble reactive P (SRP; dashed outline) are noted with boxes in both panels as they are two of the most frequently reported concentrations from lacustrine ecosystems. Figure reproduced from Wilkinson and Albright (2022).

1.2 Sources of P to Waterbodies

External P loading includes particulate and dissolved forms that are transported from the watershed into lentic ecosystems through surface and subsurface flows (Carpenter et al. 1998; Pease et al. 2018). P present in terrestrial soils may be transported to surface waters through groundwater, tile, or overland flows as well as through soil erosion, which transports P bound to soil particles. During snowmelt or precipitation events, eroded soil is transported with surface runoff and can enter nearby waters. The magnitude of external loading from soil erosion is mediated by the magnitude of the precipitation event and soil conditions prior to and during the event (Eghball et al. 2002, Carpenter et al. 2015). In addition to surface transport of P, subsurface flows can also move P, predominantly in dissolved forms, into receiving waters. Subsurface P transport is enhanced in watersheds with artificial tile drainage due to increased infiltration rates and increased hydrologic connectivity of upland soils to nearby surface waters (Gentry et al. 2007, Pease et al. 2018). Reservoirs and drainage lakes are also affected by P mobilization within upstream rivers. Specifically, bed and bank erosion in streams during high energy flows can transport substantial amounts (up to 30%) of P downstream (Sekely et al. 2002, Schilling et al. 2022).

In many areas, including Iowa, human land use and climate change have increased the number of watershed P sources and the magnitude of the flux of P into lakes and reservoirs from the surrounding landscape (Figure 3; Stoddard et al. 2016, Yuan et al. 2018). Mining of P-rich minerals for use in synthetic fertilizers has led to P accumulation in soils globally, particularly agricultural soils, which has

increased P loading to inland waters and oceans through surface runoff and groundwater infiltration (Bennett et al. 2001, Lu et al. 2017, Sabo et al. 2021). Agricultural practices like tilling also lead to higher rates of soil erosion, further exacerbating the magnitude of P loading from the catchment (Alewell et al. 2020). While nonpoint sources of P from agriculture, urbanization, and other land-use change are pervasive (Carpenter et al. 1998), municipal and industrial effluent can also be substantial point sources of nutrient pollution. A consequence of all these human activities is an increasing amount of P stored in the watershed that is susceptible to erosion and leaching into surface waters (Kusmer et al. 2019). This eutrophication threat is further aggravated by changes in precipitation due to climate change. Specifically, the frequency and intensity of precipitation events are forecasted to increase in Iowa and the upper Midwest because of ongoing climate change. More extreme precipitation events will increase soil erosion, subsurface flows, and high stream flow events, all of which will increase external P loading to receiving lakes and reservoirs (Carpenter et al. 2015).

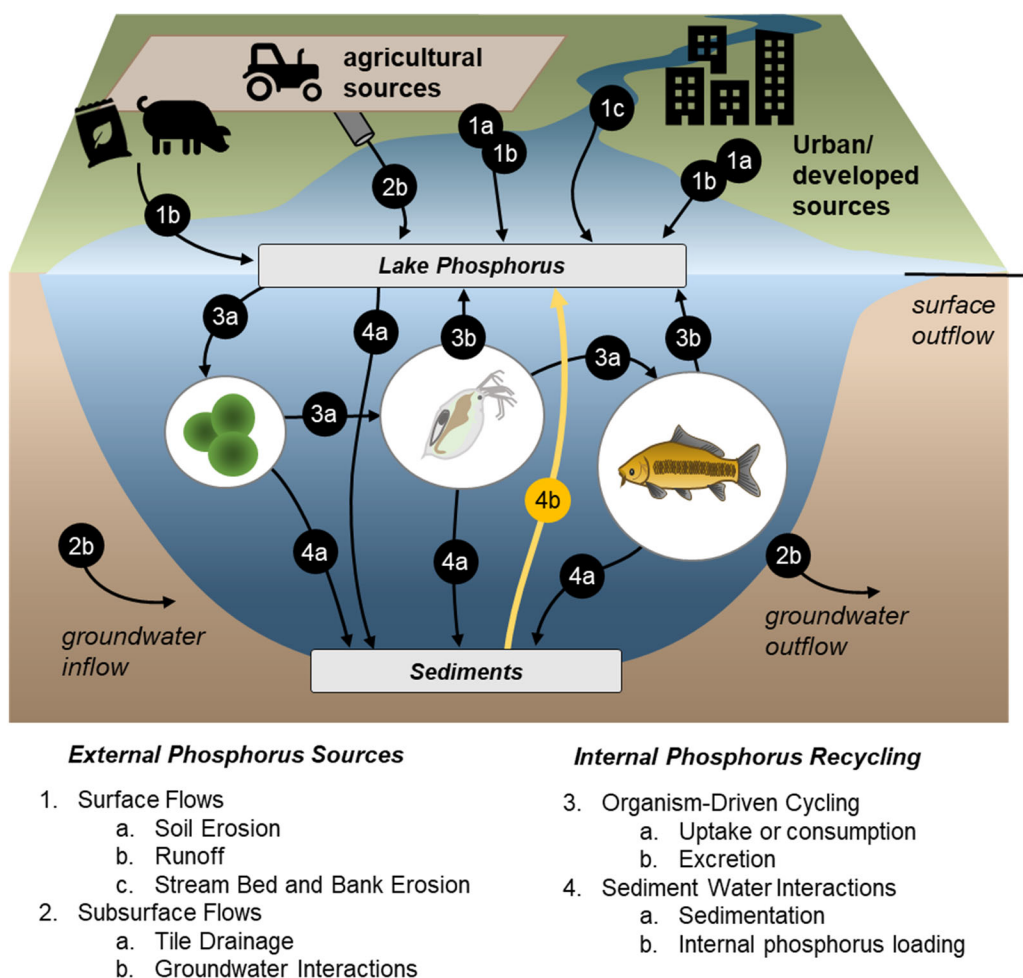


Figure 3. External sources of P to lakes and reservoirs (1 and 2) and internal cycling processes within the food web (3) and between sediments and the overlying water (4). Original illustration G.M. Wilkinson and E. Albright.

Once P has entered a lake from the surrounding watershed, it can be taken up by primary producers and cycle through the food web, be transported further downstream, or settle to the bottom of the waterbody and be stored in the sediments – season, form, and amount of P entering the lake are the primary controls of the initial fate of P when it enters the lake. The same P atom may be recycled repeatedly through the

food web, water column, and sediments of a waterbody (Figure 3). In many lakes and reservoirs, the bottom sediments hold a large pool of P, which is a legacy of past external P loading and subsequent sedimentation (Søndergaard et al. 2003; Walsh et al. 2019). This P may remain in the sediments and eventually be buried, but it can also be mobilized because of disturbance or changes in chemical conditions and then released back into the overlying water (i.e., internal P loading; Orihel et al. 2017). In some lakes, the retention of P in the sediments through immobilization and burial can be greater than the amount released into the water column, causing the sediments to be a net sink of P. Yet in many nutrient-rich lakes, the release of P from the sediments contributes substantially to the bioavailable P pool in the water column (Søndergaard et al. 2003, Orihel et al. 2017). Even with substantial reductions in external P loading, internal loading of legacy P from the sediments can continue to maintain high water column P concentrations and fuel algal production, delaying improvements in water quality (Søndergaard et al., 1999; Jeppesen et al., 2005; Søndergaard et al., 2013). As such, limiting the magnitude and frequency of internal P loading is key for managing water quality in many lakes. However, internal P loading is ultimately controlled by the amount of P available in the surface sediments, which is a legacy of watershed P loading. **Therefore, efforts to control internal P loads will only be effective when coupled with reductions in external P loads.**

1.3 Patterns of P in Iowa Lakes and Reservoirs

Total P concentrations are generally higher in Iowa lakes and reservoirs than lentic ecosystems across the contiguous United States (Figure 4). We compared the recent, annual mean total P concentrations for publicly-significant waterbodies across Iowa (n=137; Ambient Lakes Monitoring Program Data, 2017-2021, IDNR 2022) to over a thousand lakes, ponds, and reservoirs across the country (n=1,186; National Lakes Assessment 2017; US EPA 2022). Both distributions are right-skewed, meaning that there are a few waterbodies with very high total P concentrations compared to the rest of the lakes and therefore the median is the best measure to compare the distributions. The median total P concentration for Iowa lakes is 87.7 $\mu\text{g L}^{-1}$, nearly three times greater than the median value for national lakes (32.2 $\mu\text{g L}^{-1}$).

High total P concentrations indicate most Iowa lakes and reservoirs are eutrophic or hypereutrophic (Carlson and Simpson 1996, Appendix Figure S1). Based on annual mean total P concentrations from 2017-2021, 3.0% of Iowa's publicly-significant lakes and impoundments are mesotrophic (n=4; 12-24 $\mu\text{g L}^{-1}$ total P), 53.8% are eutrophic (n=71; 24-96 $\mu\text{g L}^{-1}$ total P), and 43.2% of these waterbodies are hypereutrophic (n=57; greater than 96 $\mu\text{g L}^{-1}$ total P; IDNR 2022). As such, 96.97% of Iowa's publicly-significant lakes and reservoirs are classified as eutrophic or hypereutrophic. These trophic categories are associated with reduced water clarity; increased frequency and severity of algal blooms, including blooms dominated by cyanobacteria; and increased likelihood of bottom water anoxia and resulting fish kills.

Table 1. Trophic state based on total P in Iowa lakes and impoundments

Trophic State	Total P ($\mu\text{g L}^{-1}$)	Proportion ALM Waterbodies*
Mesotrophic	12-24	3.0%
Eutrophic	24-96	53.8%
Hypereutrophic	Greater than 96	43.2%

* Based on 2017-2021 mean annual total P (n=132 lakes and reservoirs; IDNR 2022)

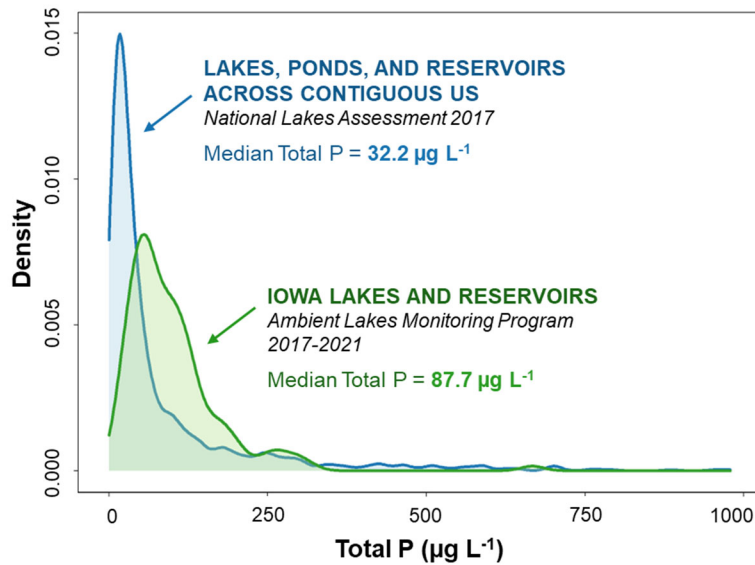


Figure 4. Distribution of total P concentrations in lentic ecosystems in Iowa and across the US. Total P data are drawn from lakes and small impoundments included in the IDNR Ambient Lakes Monitoring Program 2017-2021 ($n=137$) and the US EPA National Lakes Assessment 2017 ($n=1,186$). The median total P concentration for Iowa waterbodies ($87.7 \mu\text{g L}^{-1}$) is 2.7 times greater than the national median ($32.2 \mu\text{g L}^{-1}$). Data sources: IDNR 2022, US EPA 2022.

These data clearly show that Iowa waterbodies are greatly enriched in P. As previously described (section 1.2 Sources of P to Waterbodies), the flux of P into Iowa waters from the surrounding watershed has been enhanced by agricultural land use, livestock production, urban and suburban development, and changing precipitation patterns. However, there are no clear geographic patterns in the concentrations of total P, soluble reactive P, or the proportion of total P that is readily bioavailable (i.e., present as soluble reactive P; Figure 5). A relatively uniform spatial pattern of P concentrations across Iowa reinforces that P enrichment is a statewide water quality problem.

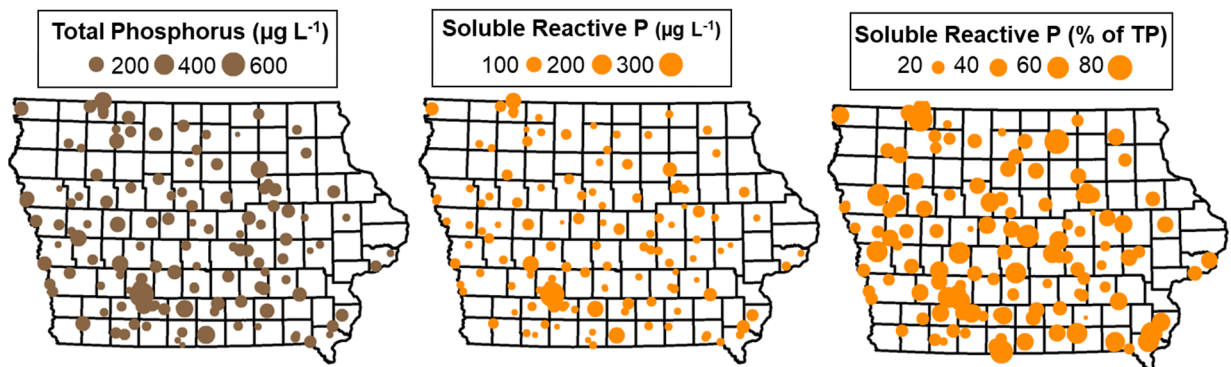


Figure 5. Total P concentrations (TP; left panel), soluble reactive P concentrations (SRP; center panel), and the proportion of the TP pool that is SRP (right panel). Points are sized proportionately to the mean annual P concentration from the past five years (2017-2021). Data Source: IDNR 2022.

1.4 Internal P Loading in Iowa Waterbodies

Many lakes and reservoirs in Iowa are at heightened risk for internal P loading due to their shallow morphometry. Shallow lakes have a large sediment surface area relative to water volume. As a result, sediment-water interactions are highly influential in determining whole-lake water chemistry (Welch & Cooke 1995; Søndergaard et al. 2003). Additionally, shallow basins make many lakes and reservoirs across Iowa polymictic (see section 3.2 for analysis of mixing regimes in Iowa lakes and reservoirs). Frequent water column mixing over the course of the summer can prompt internal P loading and brings any sediment-released P to the surface waters, where it is more available to algae and cyanobacteria.

Quantifying the contributions of external and internal loads can provide management insights for P-impaired waterbodies (Ogdahl et al. 2014). To evaluate the relative magnitude of internal loading in Iowa lakes and reservoirs, we synthesized the results of available model estimates of annual external and internal P loads for impaired waterbodies across the state. These model estimates were originally developed to inform Total Maximum Daily Load (TMDL) analyses included in Water Quality Improvement Plans for lakes and reservoirs included on the 303(d) list of federally impaired waters. We began with all available Water Quality Improvement Plans (n=116; available from <https://programs.iowadnr.gov/adbnr/Docs/TMDL>). We then selected reports developed for lakes and reservoirs and for impairments that might include estimates of P loading (n=51; impairments include algae, nutrients, total P, low dissolved oxygen, organic enrichment).

Of the 51 selected reports, there were a variety of approaches for considering internal P loading. Most reports used a mass balance model to estimate annual internal P loads in addition to watershed sources (n=19). However, many reports either ignored internal loads or did not separate internal and external sources in the load estimates (n=15). Some reports did not quantify internal loads but explicitly recommended that internal P sources be measured in the future (n=6). The remaining reports concluded that internal P loads were likely negligible on an annual-scale due to short water residence times (n=11). An overview of how the reports handled internal P loading is included in Appendix Table S1.

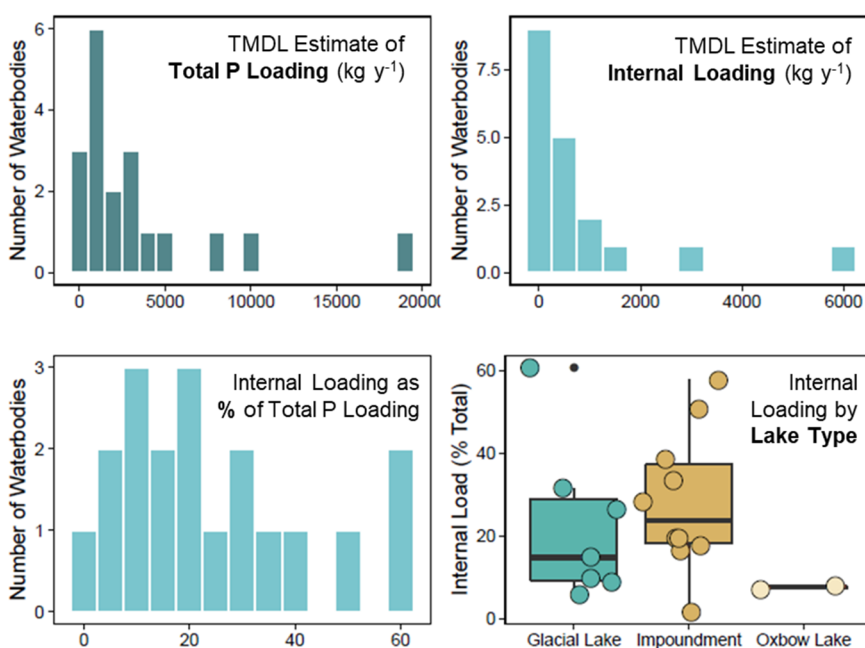


Figure 6. Model estimates of total annual P loads, internal P loads, the relative contribution of internal loads, and the relative proportion of internal P loads by waterbody type (glacial lakes, small impoundments, and oxbow lakes). Lake names and categories are included in Appendix Table S2. Data source: TMDL analyses, Water Quality Improvement Plans.

For the reports that did provide an estimate of annual internal P loads, total P loads varied widely (Figure 6, Appendix Table S2). The distribution of total loads was right-skewed and ranged from 108.8 to 20,328.8 kg P year⁻¹ with a median annual load of 1,564.6 kg P year⁻¹. The distribution of internal P loads for these waterbodies was also right-skewed with a range of 36.3 to 5,986.4 kg P year⁻¹ and a median load of 266 kg P year⁻¹ (Figure 6, Appendix Table S2). The relative contribution of internal P loading to the total annual load ranged from 1.9 to 60.6 percent with a median of 19.5 percent (Figure 6, Appendix Table S2). For lakes and reservoirs with P-associated impairments, internal recycling typically constitutes a fifth of the total P load, although there is variation among different waterbodies. On average, the relative contribution of internal P loads was greatest in small impoundments, followed by glacial lakes and then

oxbow lakes (Figure 6, Appendix Table S2). However, oxbow lakes were not well-represented, and there was considerable variation within the impoundment and glacial lake groups.

These model results likely underestimate the true magnitude of internal loading in impaired waterbodies. For most TMDL calculations, external P loads are estimated using a simple watershed model, which provides an estimate of annual P loads to a given waterbody from its catchment. A mass balance model is then used to calculate the annual internal P load based on the modeled watershed load and observed P concentrations in the lake. More specifically, if the observed concentration of P in the waterbody is greater than would be anticipated given the modeled watershed load, the difference is attributed to internal loading. As such, the accuracy of the estimated internal P load depends on the accuracy of the watershed model and whether the observed data captures P dynamics in the waterbody. P concentrations vary over the course of a year in Iowa lakes and reservoirs (see section 2.7 for analyses of intra-annual variation in internal P loading). If empirical measurements of P in the waterbody do not capture peak concentrations, then the input used for the mass balance model will produce an underestimate of the internal load. For example, model results for Green Valley Lake suggest that the annual internal load is 61.4 kg P y⁻¹. However, empirical measurements of internal P loading rates in the reservoir were highly variable, including loading rates greater than 40 kg P per *day* in the later summer (Box 1).

Even though the reported internal loads are likely underestimates, the modeling results still reveal the significance of internal P recycling in Iowa lakes and reservoirs. Overall, internal loading typically constitutes a fifth of the total P load in these impaired waterbodies, highlighting the importance of including internal recycling in lake restoration plans and TMDL analyses. Additionally, these results suggest that the influence of internal loading cannot be assumed from water residence time. Although some Water Quality Improvement Plans assumed that internal loads were negligible on an annual-scale due to short residence times, plans that did report both internal loads and water residence times (n=13) show that the relative contribution of internal loads can be quite high (>40%) even for waterbodies with short residence times (1-2 months; Figure 7). This pattern reinforces that internal P loads should be estimated whenever possible, even for waterbodies with short residence times. Even if internally loaded P does not remain in the lake or reservoir very long due to the high flushing rate, it can still present water quality challenges for receiving waterbody.

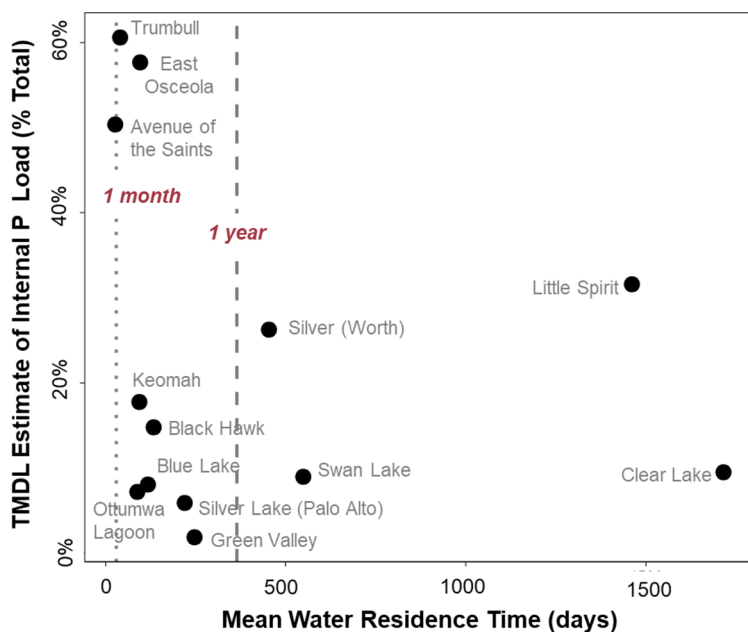
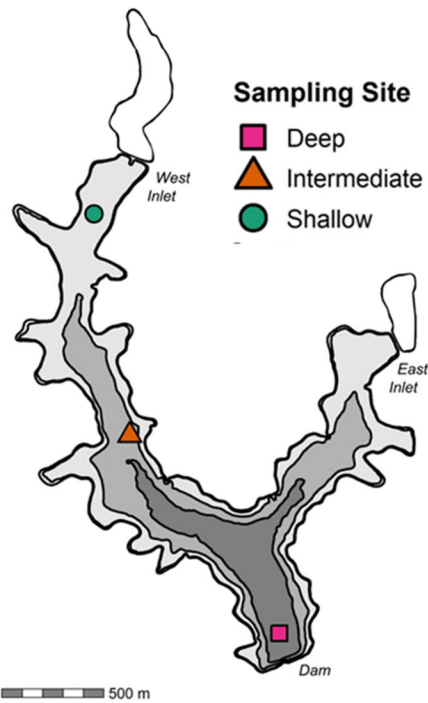


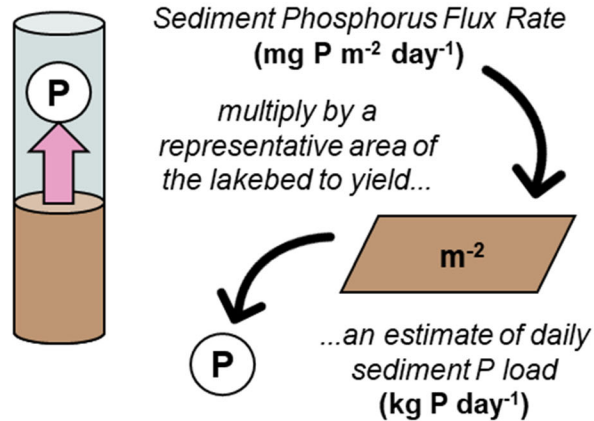
Figure 7. Estimated contribution of internal P loads versus modeled water residence time. Mean water residence times of 1 month and 1 year are noted with dotted and dashed lines, respectively. Data source: TMDL analyses, Water Quality Improvement Plans.

Box 1. Comparison of TMDL estimates and empirical measurements of annual internal phosphorus loads in Green Valley Lake

Model results from the TMDL analysis for Green Valley Lake suggest that the annual internal P load is 61.4 kg P year⁻¹. However, empirical measurements of internal P loading rates in this reservoir reveal high inter-annual variation, including loading rates greater than 40 kg P per day in the later summer.



Using sediment core incubations, we measured sediment P flux rates from sediments collected at three sites from the west inlet of the reservoir to the deep site above the dam. We scaled these flux rate measurements to representative areas of the lakebed to estimate the daily internal P load for each region of the reservoir as well as the total load for the entire waterbody.



Daily internal phosphorus loads estimated from empirical measurements

Internal P loads vary considerably over the course of the year. Loads were greatest in the late summer, but moderate loads (~10 kg P day⁻¹) also occurred under ice cover and in spring. These empirical results suggest that the TMDL estimate of annual internal P loads obscures seasonal variation and underestimates internal loading at key times for phytoplankton growth.

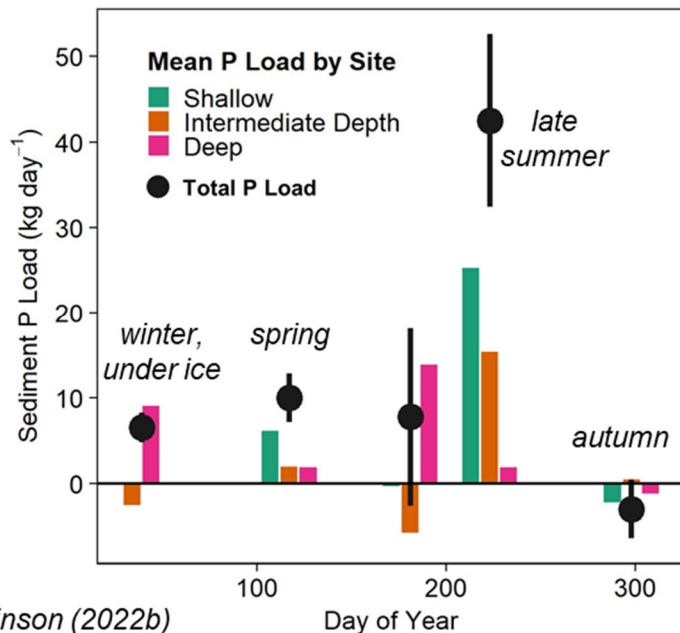


Figure adapted from Albright and Wilkinson (2022b)

1.5 Summary

- P is an essential element for life; however, in aquatic ecosystems excess P enrichment can lead to algal blooms, oxygen depletion, and fish kills.
- P sources to inland waters may be external (i.e., from the surrounding watershed) or internal (i.e., internal recycling between sediments and the overlying water).
- Internal P loading can maintain high water column P concentrations even after watershed loads are reduced.
 - Controlling internal P loading is key for managing water quality in many lakes
 - However, efforts to control internal loads will only be effective when coupled with reductions in external P loads.
- Total P concentrations are generally higher in Iowa waterbodies (median total P for Iowa lakes almost three times US waters, Figure 4), compared to the rest of the contiguous United States.
- 97% of Iowa's publicly-significant waterbodies are classified as eutrophic or hypereutrophic based on high total P concentrations.
- Many lakes and small impoundments across Iowa are at heightened risk for internal P loading due to their shallow morphometry.
- Internal loading typically constitutes a fifth of the total P load in Iowa waterbodies with P-associated impairments, based on a synthesis of TMDL estimates, but can be much greater for some waterbodies.
- Based on the analysis of TMDLs and empirical measurements, current TMDL estimate approaches of P loading are likely underestimating the contribution of internal loading.

SECTION 2 – SEDIMENT FORMS AND MOBILIZATION MECHANISMS

The chemical composition of the sediment P pool is an important mechanism controlling internal loading because some forms of P are more mobile than others. Sediment P is found in a wide variety of minerals and organic materials as well as sorbed to particle surfaces (North et al., 2015). This chemical speciation determines the conditions in which P is mobilized and released as different P forms are vulnerable to different internal loading processes (Orihel et al., 2017; Table 2, Figure 8). Here we review five functional forms of P found in sediments and discuss mobilization mechanisms.

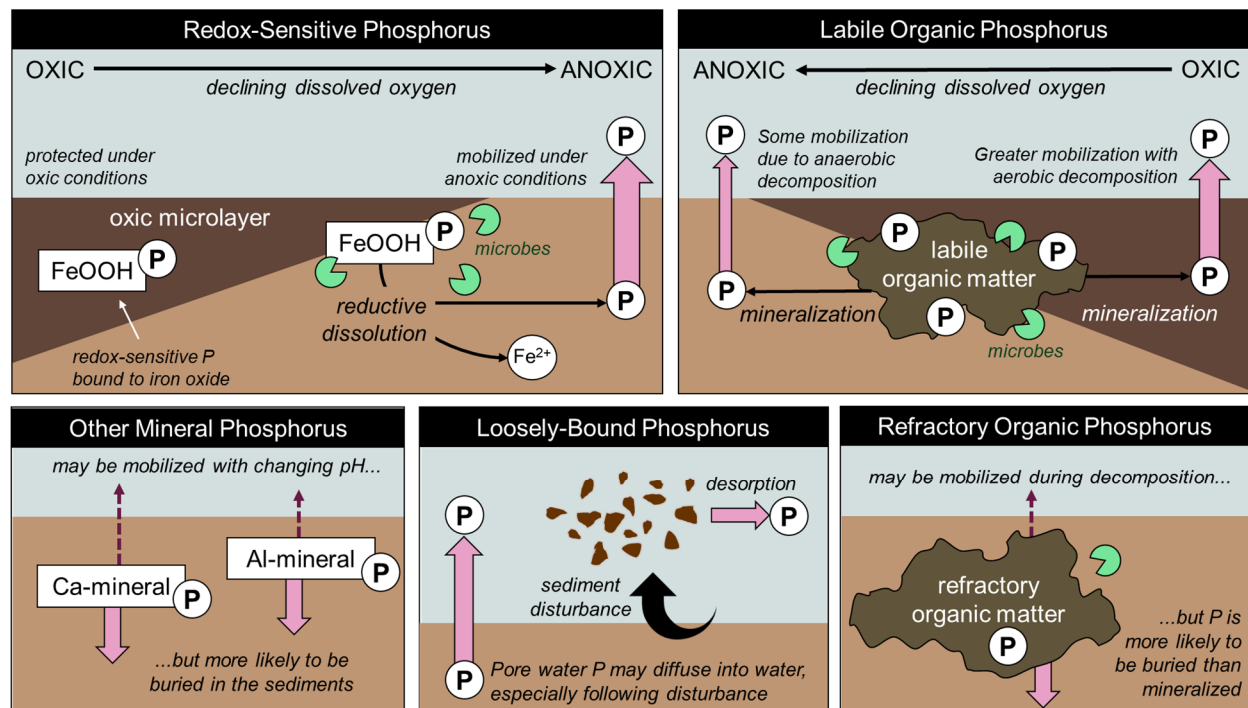


Figure 8. Five functional forms of P found in sediments and the mechanisms that may mobilize them into the overlying water. Original illustration E. Albright 2022.

2.1 Labile Organic P

Labile organic P describes any P compounds built into organic materials that are readily decomposed (Table 2, Figure 8). Common sources of labile organic matter to the sediments include algal detritus and some macrophyte leaf litter (Twilley et al. 1986; Jiang et al. 2022). These labile materials have few structural carbon compounds, such as lignin or cellulose, and high nutrient content. The nutrient content or quality of organic materials can be quantified by the ratio of carbon to nitrogen and P (i.e., C:N:P). Labile organic materials have low C:N:P ratios, which signifies that the material has a high abundance of nitrogen and P compared to the carbon content. As microbes in the sediments decompose organic matter, they take up carbon and nutrients from the material to meet their metabolic needs. However, for materials with low C:N:P ratios, there will be left over P that is not taken up into microbial biomass but is instead released as phosphate (i.e., a bioavailable form of P; Lukkari et al. 2007). Nutrient mobilization during the decomposition of organic matter is called mineralization.

Microbial decomposition of organic matter and subsequent mineralization of P is the primary mechanism that will mobilize labile organic P forms in lakebed sediments. This biological pathway for sediment P mobilization is controlled by temperature, dissolved oxygen availability, and the quality of the organic

matter. There is a direct relationship between temperature and decomposition rates such that warmer water temperatures will be associated with higher decomposition rates, which may in turn enhance P mineralization (Gudas et al. 2011, Jensen and Andersen 1992). Decomposition rates and P mineralization will be enhanced under oxic conditions, but these processes can still occur under anoxic conditions (Joshi et al. 2015, Horppila et al. 2017, Song and Burgin 2017). Aerobic decomposition (i.e., when dissolved oxygen is available) is more efficient than various forms of anaerobic decomposition. However, decomposition will still occur under anoxic conditions provided that alternative electron acceptors are available (e.g., nitrate, manganese IV, iron III, sulfate, and carbon dioxide). Organic matter quality will determine whether microbial decomposition of sediment organic matter results in P mineralization or immobilization. The sources of sedimented organic materials in a lake will influence the lability of sediment organic matter. For example, if phytoplankton detritus dominates organic matter sedimentation, then sediment organic P will be associated with more labile materials (Jiang et al., 2022).

2.2 Refractory Organic P

Refractory organic P describes any P compounds incorporated into organic materials that are difficult to decompose (Table 2, Figure 8). Refractory organic materials resist microbial decomposition due to complex carbon compounds (e.g., lignin) and high C:N:P ratios. High carbon to nutrient ratios slow litter decomposition and limit P mineralization in the sediments. Specifically, relatively low nutrient content means the P available in the organic material will be incorporated into microbial biomass or immobilized, rather than being released through mineralization (Lukkari et al. 2007). Common sources of refractory organic matter to the sediments include leaf litter from emergent vegetation stands, such as cattails (*Typha spp.*). Some research suggests that much of the litter from cattail stands is permanently buried, causing some emergent macrophyte beds to act as P sinks (Davis and van der Valk 1983). P is much less likely to be mobilized from refractory organic matter than from labile materials. However, P mineralization from refractory organic materials is still possible given sufficient nutrient subsidies. Specifically, the decomposition and mineralization of more recalcitrant organic matter can be stimulated by high nutrient availability or the presence of labile organic material (i.e., a priming effect; Guenet et al. 2010, Guillemette et al. 2013).

2.3 Redox-Sensitive P

Redox-sensitive P describes P bound to either iron or manganese oxide minerals (Lukkari et al. 2007; Table 2, Figure 8). These minerals are sensitive to changes in redox potential, which is determined by the abundance of oxygen and alternative electron acceptors present in the sediments. For example, if the hypolimnion of a lake becomes anoxic during summer stratification, and dissolved oxygen is unavailable in the surface sediments, anaerobic microbial metabolic strategies will become favorable. Specifically, iron- and manganese-reducing bacteria will use the iron III and manganese IV found in iron and manganese oxide minerals as alternative electron acceptors, causing the mineral to dissolve. When the host mineral dissolves, the bound P will be released. As such, the primary mechanism that mobilizes redox-sensitive P forms is a change in redox potential, particularly the absence of available dissolved oxygen and nitrate (Mortimer, 1941; Jensen and Andersen 1992; Orihel et al., 2017).

2.4 Other Mineral P

In addition to redox-sensitive minerals, P may be associated with a wide array of other minerals in lakebed sediments, the most common of which are aluminum-bound and calcium-bound P (Lukkari et al. 2007; Table 2, Figure 8). These mineral P forms are considered relatively immobile because substantial changes in pH are required to release P from these minerals, and these pH levels are uncommon in many

lakes and reservoirs. However, under high pH (>10), P bound to aluminum oxides may be released due to ligand exchange with excess hydroxide ions present. Alternatively, acidic conditions can cause some calcium minerals to dissolve, releasing bound P (Orihel et al. 2017). Overall, mineral P forms that are not sensitive to changes in redox potential are more stable and thus more likely to be buried in the sediments rather than released back into the water column; however, P mobilization from these minerals is still possible and dependent on pH.

2.5 Loosely-Bound P

Loosely-bound P is a broad category that includes dissolved P found in sediment pore water as well as P sorbed to the surface of clay minerals or other sediment particles (James 2017a; Table 2, Figure 8). Loosely-bound P may be mobilized when sediments are disturbed due to water column mixing, wave disturbance, or bioturbation (Ekholm et al., 1997). When the sediment surface is disturbed, particles may be resuspended into the water column, where surface-sorbed P is likely to desorb from the sediment particles and be released into the water column. Sediment resuspension can also release pore water P directly and mobilize P from deeper sediments. Resuspension dilutes the pore water near the sediment surface, prompting diffusion of soluble P from deeper within the sediment profile and thus enhancing diffusive P fluxes into the water column (Tammeorg et al. 2016; Tammeorg et al. 2020). Loosely-bound P is likely an important form of sediment P in shallow lakes as these ecosystems experience frequent water column mixing and wind disturbance (Bengtsson and Hellström 1992) as well as in waterbodies with benthivorous fishes such as common carp (*Cyprinus carpio*), which are a significant source of bioturbation (Huser et al. 2016).

Table 2. Functional P (P) forms found in lake and reservoir sediments

Functional Form	Description	Mobilization Mechanisms	Relative Mobility
Labile Organic	P incorporated in organic matter that is easily decomposed (e.g., algal detritus)	Decomposition and mineralization	More likely to be mobilized and released
Redox-Sensitive	P bound to iron or manganese oxide compounds	Dissolution of host mineral under anoxic conditions	
Loosely-Bound	P loosely-sorbed to particle surfaces or in pore water	Sediment disturbance	
Refractory Organic	P in organic material that is difficult to decompose (e.g., cattail litter)	May decompose with nutrient priming	More likely to be buried in sediments
Other Mineral	P bound to aluminum oxides or incorporated in calcium minerals	May be released under extreme changes in pH	

2.6 Sediment P Forms in Iowa Lakes and Reservoirs

2.6.1 Spatial Variation Within and Among Lakes

We measured the chemical composition of the sediment P pool in seven glacial lakes in northwest Iowa (Table 3) and quantified variation in sediment P composition within and among these waterbodies (Albright *et al.* 2022a).

Table 3. Location, basin morphology, and watershed features of the study lakes, 2018

	Center Lake	Five Island	Silver (Dickinson)	Storm Lake	North Twin	South Twin	Swan Lake
Surface area (ha)	104	405.6	431.4	1270.7	187.4	224.6	48.6
Max depth (m)	5.5	8	3	6.2	3.7	1.6	3.8
Mean depth (m)	3.7	1.7	2	2.6	2.7	1.1	1.7
Chlorophyll- <i>a</i> ($\mu\text{g L}^{-1}$) ¹	37.6	47.1	30.6	16.7	53.9	85.2	77.9
% Cropland	14.3	75.5	78.1	60.2	82.5	78.4	45.1
Dominant soil origin	Glacial till	Glacial till	Glacial till	Loess	Glacial till	Glacial till	Loess
Stratification Frequency (%) ²	26.2	26.8	8.1	2.5	9.3	NA	17.4
Thermocline Depth (m)	2.7	3.4	1.75	2.4	NA	NA	1.55

¹Nineteen-year mean (2000-2018) of measurements (n=36-49 measurements per lake) except for South Twin Lake which is the 13-year mean (2006-2018) of measurements made during the ice-free season (n=35).

²Long-term stratification frequency (% of observations with thermal stratification) based on the Ambient Lake Monitoring program profiles from 2006-2018 for Storm and Silver Lakes and 2005-2018 for the other lakes.

Among the study lakes, whole-lake means of sediment total P content ranged from 897.0 to 1,184.9 $\mu\text{g P g}^{-1}$ dry sediment, with a grand mean of 946.2 $\mu\text{g P g}^{-1}$ dry sediment (Table S3). These concentrations align with ranges from other shallow, eutrophic lakes (Søndergaard *et al.* 2013) and other productive waterbodies more generally (Doig *et al.* 2017; Kowalczywska-Madura *et al.* 2019; Tao and Lu, 2020). Sediment P composition also varied among the study lakes; however, redox-sensitive and organic P were consistently the most abundant fractions, while aluminum and calcium-bound P were usually present in the lowest concentrations (Table S3; Figure 9). Other eutrophic waterbodies also have high levels of redox-sensitive P (Song and Burgin 2017; Papera *et al.* 2021) and organic P forms (Nowlin *et al.* 2005; Frost *et al.* 2019).

Variation in sediment P composition among the lakes corresponded with differences in watershed and lake basin characteristics (Figure 10). The North and South Twin Lake and Swan Lake groupings reflect patterns in dominant watershed soil series. However, there was variation in watershed soil characteristics within the Center, Five Island, Silver and Storm Lake group as soils in the Storm Lake catchment were formed partially in loess while the other watershed soils were formed in glacial till. Variation in watershed soils within this group reinforces that aquatic sediment P composition is not a direct reflection of the soil matrix due to selective erosional and transport processes as well as in situ processing (Kerr *et al.* 2011). For example, similarities in P composition within this group were related to maximum depth

and sediment bulk density, suggesting that these waterbodies experience similar sediment transport, deposition, and resuspension processes compared to the other lakes (Mackay et al. 2012).

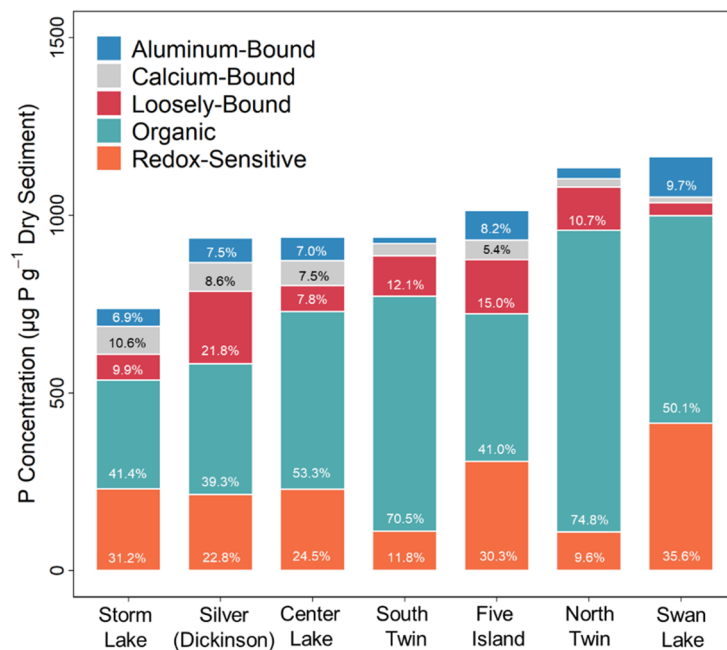


Figure 9. Variation in sediment P composition among seven glacial lakes in northwest Iowa. Data source: Albright et al. 2022a. More details on percentages and total concentration in Table S3.

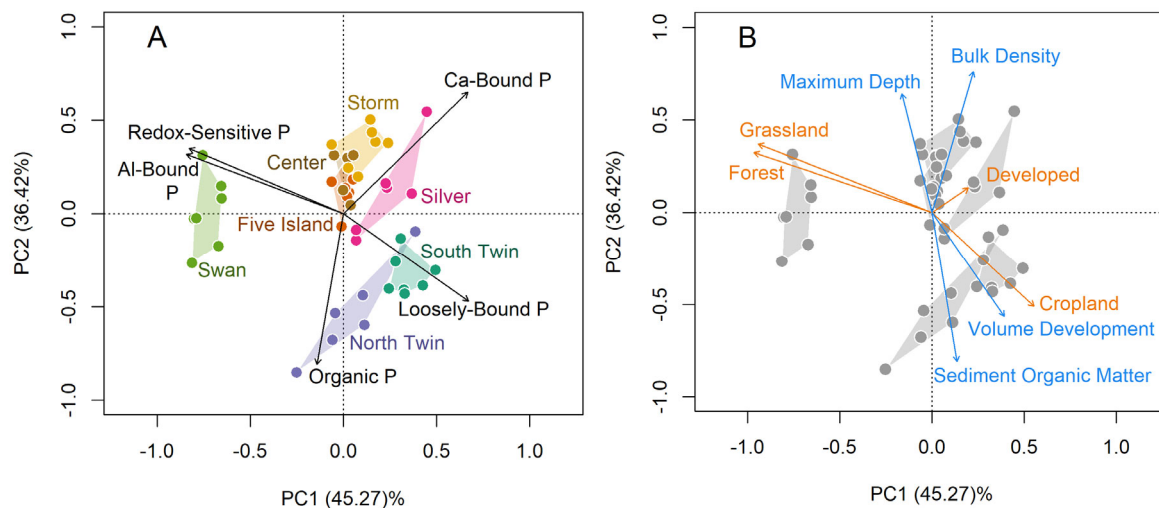


Figure 10. (A) Compositional data analysis of sediment P pools. The compositional analysis was defined by the concentrations of loosely-bound, redox-sensitive, aluminum-bound, calcium-bound, and organic P (including labile and refractory components). Arrows represent center log-ratio transformed concentrations of the P fractions. (B) Interpretation of the analysis with key environmental variables. Watershed land cover classes are shown in orange while lake and sediment characteristics are plotted in blue. Figure reproduced from Albright et al. 2022a.

In addition to the role of watershed and lake basin features in driving differences in sediment P composition across the study lakes, we also found a direct relationship between long-term mean chlorophyll-a concentrations, as an index of phytoplankton biomass, and the average contribution of mobile P species to the total sediment P pool in surface sediments (Figure 11). This relationship suggests a positive feedback loop between mobile sediment P forms, heightened internal P loading rates, and

phytoplankton biomass (Katsev et al. 2006; Song and Burgin 2017). Specifically, internally-loaded P from mobile forms in the sediments supports phytoplankton production, while sedimentation of phytoplankton detritus supplies more mobile P to the sediments. Organic P was the largest fraction of total sediment P in all of the study lakes, driving the among-lake variability in mobile P contributions. Given the prevalence of organic P in the sediments, it is likely that decomposition of sediment organic matter and subsequent P mineralization are key processes for internal P loading in the study lakes (Joshi et al. 2015; Frost et al. 2019). In hypereutrophic waterbodies rampant phytoplankton production and detritus enhance organic matter sedimentation, increasing organic P in lakebed sediments. Large pools of organic P are associated with high extracellular enzyme activities, which mobilize and release sediment P (Song and Burgin 2017). Our study lakes are likely examples of systems in which biologically-mediated cycling of organic P pools drives internal P loading and high phytoplankton production in turn increases mobile P forms in the sediments.

In addition to variation in sediment P content and composition across different lakes, we also quantified spatial heterogeneity in total and loosely-bound P pools within individual lakes and investigated the relationship between variation in sediment P pools and lake basin volume development. Volume development (Dv) is an index of lake basin shape and can be used to infer bathymetric complexity. It is calculated as the ratio of the volume of the lake basin to the volume of a perfect cone with a base area equal to the surface area of the lake and a height equal to the maximum water depth (Hutchinson 1957). Consistent with our expectations, loosely-bound P concentrations were most variable in lakes with low volume development indices, indicative of conical basins with localized deep holes (Figure 12). This bathymetric variability produces both shallow regions where frequent sediment disturbance releases loosely-bound P and deeper areas where loosely-bound P is protected (James 2017b). As a result, loosely-bound P concentrations are more variable in lake basins with more complex bathymetry. In contrast, lakes

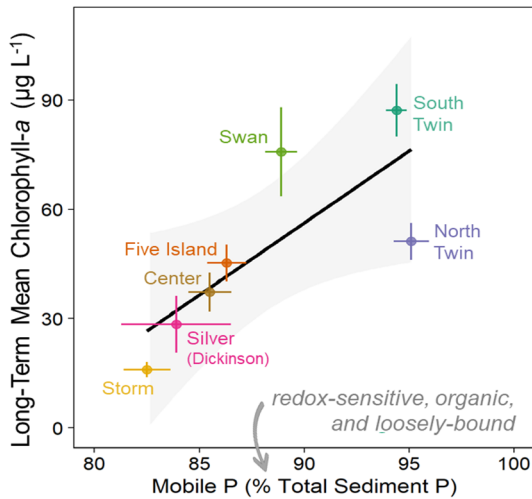


Figure 11. The relationship between the fraction of the total sediment P pool comprised of mobile species (averaged over 0-6 cm at the deep sampling site) and the long-term mean chlorophyll-a concentrations in each lake ($F_{1,5} = 7.6$; $p = 0.04$; $adj R^2 = 0.52$; $b_1 = 4.9$, $CI_{95\%} = [0.33, 9.5]$). The grey shaded area is the 95% confidence interval. Figure reproduced from Albright et al. 2022a.

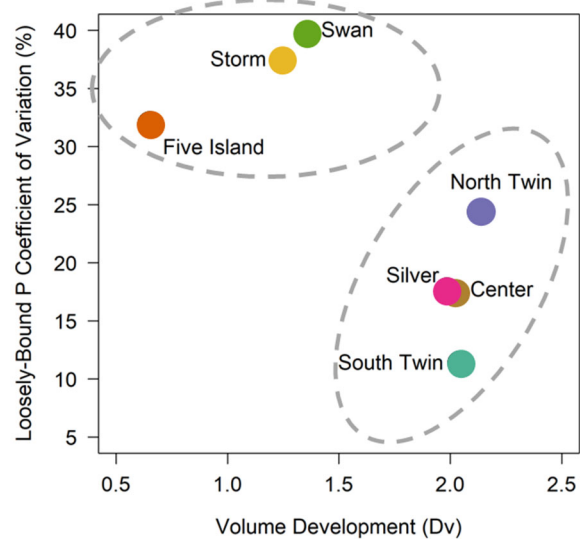


Figure 12. Variation in sediment loosely-bound P with basin volume development. The coefficient of variation for loosely-bound P was highest in lakes with low volume development scores (top left oval), indicating more variable basin shape. Lakes with higher volume development ratios (bottom right oval, “bowl-like” bathymetry) had less intra-lake variation in loosely-bound P. Figure reproduced from Albright et al. 2022a.

with high volume development scores, associated with shallow, bowl-shaped basins, had more consistent loosely-bound P concentrations, as sediment disturbance was likely more uniform across the lakebed.

We also hypothesized that greater water depths would correlate with higher concentrations of total and loosely-bound sediment P across the lakebed, which was supported by the mixed model analysis (Figure 13). In fact, the depth of the water at the sediment sampling location described the vast majority (84.1%) of the variation in loosely-bound P and over half of the variation (63.1%) in total P concentrations. The pattern of higher concentrations at greater depths is likely driven by sediment focusing and greater net sediment deposition in deeper portions of the lake (Hou et al. 2014; Cardoso-Silva et al. 2018). The direct relationship that we observed between water depth and sediment P concentrations is consistent with other studies of spatial variation in sediment P pools (Nowlin et al. 2005; Kowalczywska-Madura et al. 2019). However, water depth did not fully explain the observed variation in sediment total and loosely-bound P, indicating that factors beyond basin morphology contribute to spatial variation in sediment P pools.

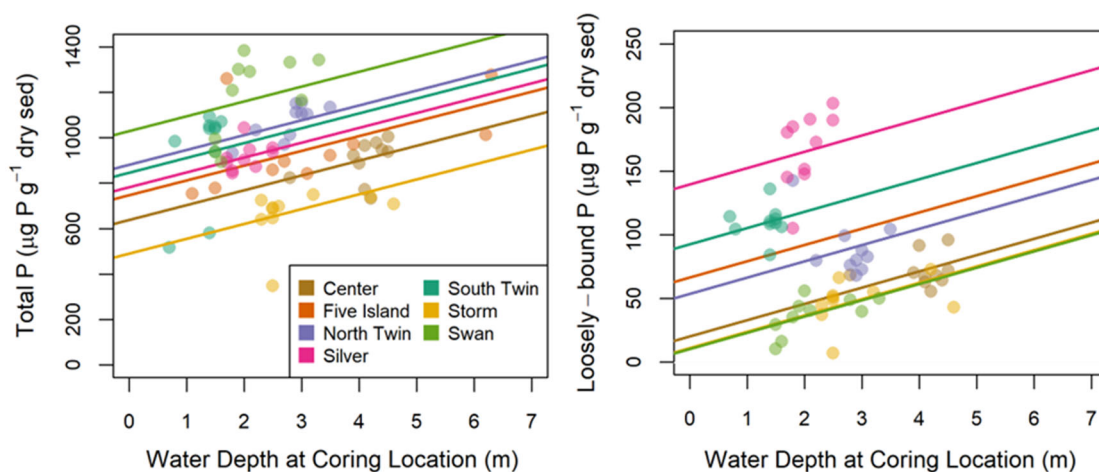


Figure 13. Mixed model regression effects of water depth at the coring location on total sediment P ($b_1 = 65.3$, $CI_{95\%} = [27.6, 102.5]$; top panel) and loosely-bound sediment P ($b_1 = 12.8$, $CI_{95\%} = [7.2, 18.3]$; bottom panel) by lake. Figure reproduced from Albright et al. 2022a.

2.6.2. Seasonal Variation

In addition to spatial variation, sediment P composition can also vary over time within individual waterbodies. To characterize seasonal variation in sediment P pools, we measured sediment P forms five times throughout the course of 2020 in Green Valley Lake (Albright and Wilkinson 2022b). These sampling events included ice cover as well as mixing and stratification events throughout the open water season. We also sampled three sites along the western arm of the reservoir from a shallow region near the inlet to the deep site near the dam outflow. Across the reservoir, redox-sensitive P was the dominant pool of mobile P found in the sediments, constituting an average of 40.9-51.3 percent of the total P pool (Figure 14A). The concentrations of all measured sediment P forms were dynamic over time (Figure 14B-E; see Section 2.7 for more discussion of the mechanisms driving this seasonal variation).

Overall, sediment P forms vary within and among Iowa lakes and reservoirs. However, redox-sensitive and labile organic P forms are consistently the most abundant forms of mobile P found in the sediments. Therefore, we can make inferences regarding influential mechanisms for sediment P mobilization in Iowa waterbodies and test those inferences with available data.

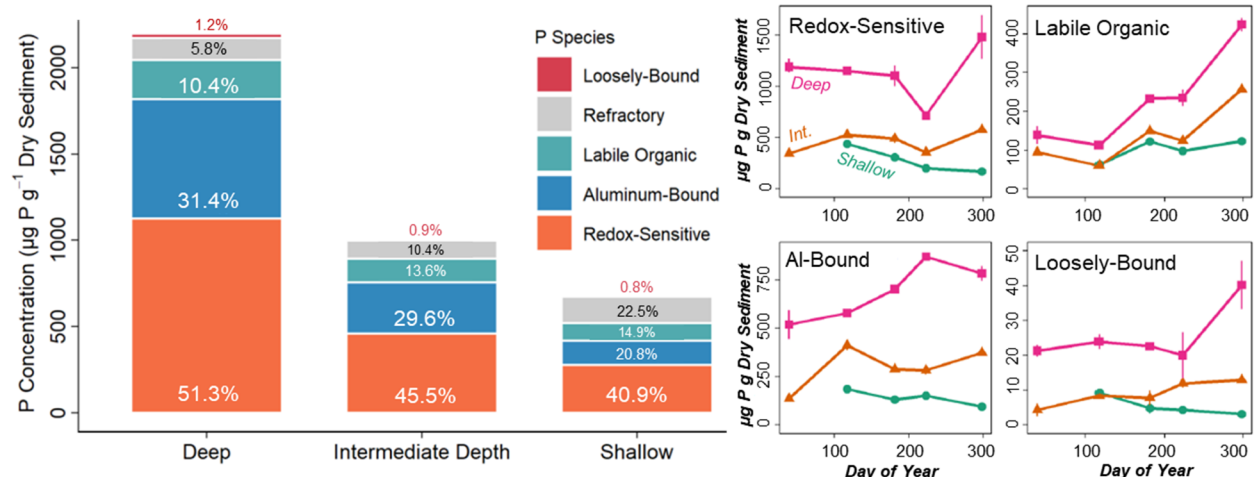


Figure 14. Spatiotemporal variation in sediment P forms. (A) Average, annual sediment P composition by sampling site. Refractory P is estimated as the difference between total P and the sum of the measured P fractions. The relative abundance of each P form as a percent of the total sediment P pool is noted. Time series of mean ($\pm$ standard error) (B) redox-sensitive, (C) labile organic, (D) aluminum-bound, and (E) loosely-bound P concentrations across sampling sites (deep, intermediate, and shallow). Note different y-axis scales across plots depicting more versus less abundant P fractions. Figure reproduced from Albright and Wilkinson 2022b.

2.7 Important Mobilization Processes for Iowa Waterbodies

Given that redox-sensitive and labile organic P are the most abundant forms of mobile P found in the sediments of Iowa lakes and reservoirs, **dissolved oxygen** availability is likely a key mechanism in determining the magnitude and spatiotemporal variation of internal P loading in each waterbody. Internal P loading may occur under a range of dissolved oxygen concentrations based on sediment characteristics (Hupfer and Lewandowski 2008). Specifically, the chemical composition of the sediment P pool mediates the relative importance of anaerobic versus aerobic mechanisms of P mobilization because some sediment P forms are more likely to be mobilized under anoxic conditions while others are more susceptible to release when dissolved oxygen is available (Table 2; Orihel et al. 2017). For example, if redox-sensitive P forms dominate the sediment P pool, then hypolimnetic anoxia will trigger P release due to the dissolution of the host minerals (Section 2.3; Mortimer 1941; Jensen and Andersen 1992). When dissolved oxygen is available at the sediment-water interface, redox-sensitive P forms will be stable. However, if sediments also hold a large pool of labile organic P, then oxic conditions are expected to mobilize and release P via microbial decomposition and mineralization (Section 2.1; Joshi et al. 2015; Horppila et al. 2017). In short, internal P loading likely occurs under both oxic and anoxic conditions in Iowa waterbodies, but different forms of sediment P will be mobilized and released under different dissolved oxygen conditions.

To contextualize the influence of dissolved oxygen on internal P loading for waterbodies beyond Iowa, we used a literature synthesis to collect reported rates of sediment P fluxes for lakes and reservoirs around the world (Albright et al. *in preparation*). These lakes varied in size, depth, trophic state, and stratification frequency. We used a Bayesian hierarchical model to quantify the magnitude of P flux rates based on dissolved oxygen status (i.e., oxic versus anoxic) and sediment location (i.e., profundal versus littoral) as reported in each individual study. Mean P flux rates were nearly twice as high under anoxic (7.02 ± 0.31 mg P m⁻² day⁻¹) versus oxic conditions (3.62 ± 0.22 mg P m⁻² day⁻¹) (Figure 15A). The 95% credible interval of the estimate for mean oxic flux rates did not include zero, indicating that the mean sediment P release rate among lakes under oxic conditions is positive. Further subsetting the meta-analysis data by sediment sampling location, we found that mean P flux rates were lowest in the littoral-oxic group and

highest in the profundal-anoxic group (Figure 15B). This synthesis supports empirical evidence that internal P loading occurs under a range of dissolved oxygen conditions in both the littoral and profundal regions of waterbodies.

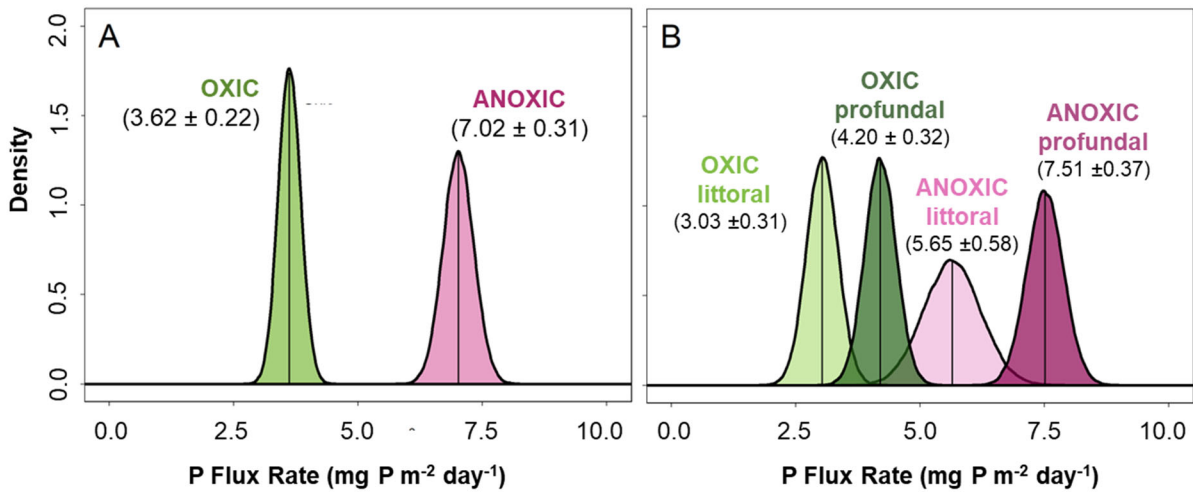


Figure 15. Distribution of the Bayesian posterior estimates of the mean sediment P flux rate (A) by incubation dissolved oxygen conditions and (B) by both sampling location and dissolved oxygen status. Sampling location categories (i.e., littoral vs. profundal) were determined based on the site description for each lake. Density (y-axis) is relative probability (i.e., density distribution) of the mean P flux rate (x-axis) being a certain value. The shorter and wider the density distribution, the more uncertain the estimate of the mean P flux rate based on the population of rates measured across lakes in the studies included in this analysis. Vertical lines mark the mean value of the posterior estimate. Mean $\pm$ standard deviation values are noted in parentheses for each group. Figure adapted from Albright et al. in preparation.

The patterns of sediment P flux rates under different dissolved oxygen conditions revealed in the meta-analysis are also supported in field studies from Iowa lakes and reservoirs. These studies indicate that internal P loading can occur under both oxic and anoxic conditions in Iowa waterbodies, but different forms of sediment P will be mobilized and released under different dissolved oxygen conditions. We evaluated seasonal and spatial variation in sediment P flux rates in Green Valley Lake throughout 2020 (Albright and Wilkinson 2022). Over the course of the year, sediment P release occurred under a broad range of dissolved oxygen concentrations at the sediment-water interface (Figure 16). At the deep site, nearly anoxic conditions were associated with elevated rates of P release in winter and mid-summer (dissolved oxygen 1.1 and 0.3 mg L⁻¹, respectively). However, most of the observed instances of P release occurred when oxygen was available at the sediment-water interface. For example, sediments across all three sampling sites released P under oxic conditions in spring (dissolved oxygen range 5.8-9.4 mg L⁻¹), and high rates of oxic P release occurred at the intermediate and shallow sites in late summer (dissolved oxygen 4.8 and 7.2 mg L⁻¹, respectively). The effect of dissolved oxygen availability on sediment P flux rates differed between the deep site and more shallow sampling sites, and elevated P release rates were observed under both oxic and anoxic conditions.

Although the dominance of aerobic versus anaerobic internal loading in Green Valley Lake shifted over sites and seasons, both processes were important pathways for P recycling between sediments and the overlying water. To understand why sediment P fluxes responded differently to dissolved oxygen conditions across space and time, we measured the chemical composition of the sediment P pool. Spatiotemporal variation in sediment P composition corresponded to variation in flux rates. Specifically, instances of P release under oxic conditions mainly coincided with declines in labile organic P. P release

during hypolimnetic anoxia and the deep site corresponded to decreasing redox-sensitive P concentrations in the sediments (Figure 14B-C).

At the deep site of the reservoir, a change in redox-sensitive P concentrations mirrored P flux rates over time. Redox-sensitive P declined steadily from winter to mid-summer, and then sharply decreased (35% decrease) from mid- to late summer. The mid-summer sampling event had high rates of P release from the profundal sediments that coincided with hypolimnetic anoxia at the deep site. The decline in redox-sensitive P from spring to late summer suggests that the high flux rates in mid-summer were the result of reductive dissolution of redox-sensitive P minerals under anoxic conditions (Figure 14B). The concentration of aluminum-bound P at the deep site followed the inverse pattern of redox-sensitive P. This temporal pattern suggests that a portion of the P mobilized from redox-sensitive minerals may have sorbed to available aluminum oxides rather than being released into the overlying water (Figure 14D). Concentrations of labile organic P in the sediments generally increased over the course of the year at all sampling sites, likely due to organic matter sedimentation. However, declines were measured at the shallow and intermediate sites from mid- to late summer, indicating that the elevated rates of aerobic P release measured at these sites in late summer involved P mineralization following decomposition of labile organic materials (Figure 14C). Temporal coherence between sediment P release and declines in sediment labile organic and redox-sensitive P underscores the complexity of P mobilization and the variety of P species that may be released.

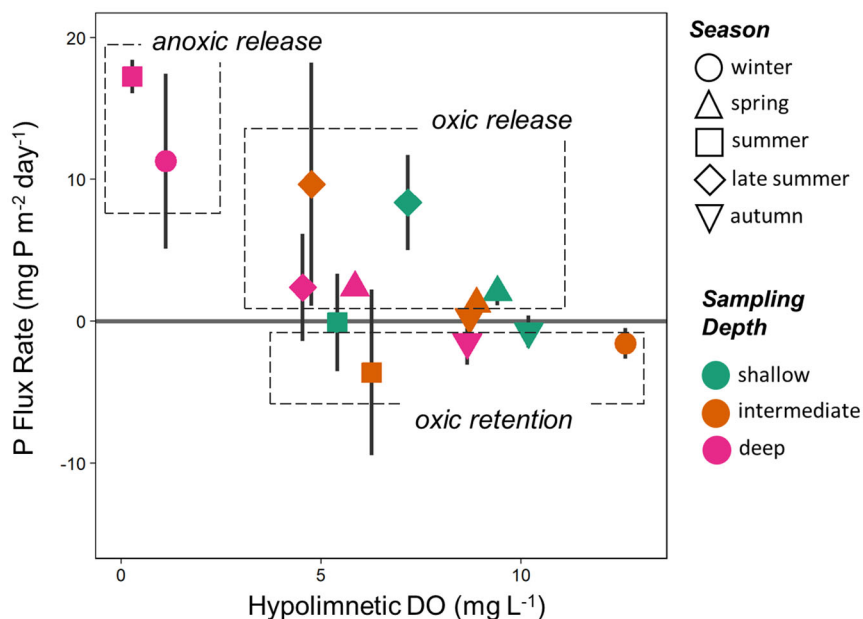


Figure 16. Mean sediment P flux rates ($\pm$ standard error among three replicates) across the range of observed bottom water dissolved oxygen (DO) concentrations in Green Valley Lake. Figure adapted from Albright and Wilkinson 2022b. **See Box 1** for estimates of reservoir-wide loading rates based on these measurements.

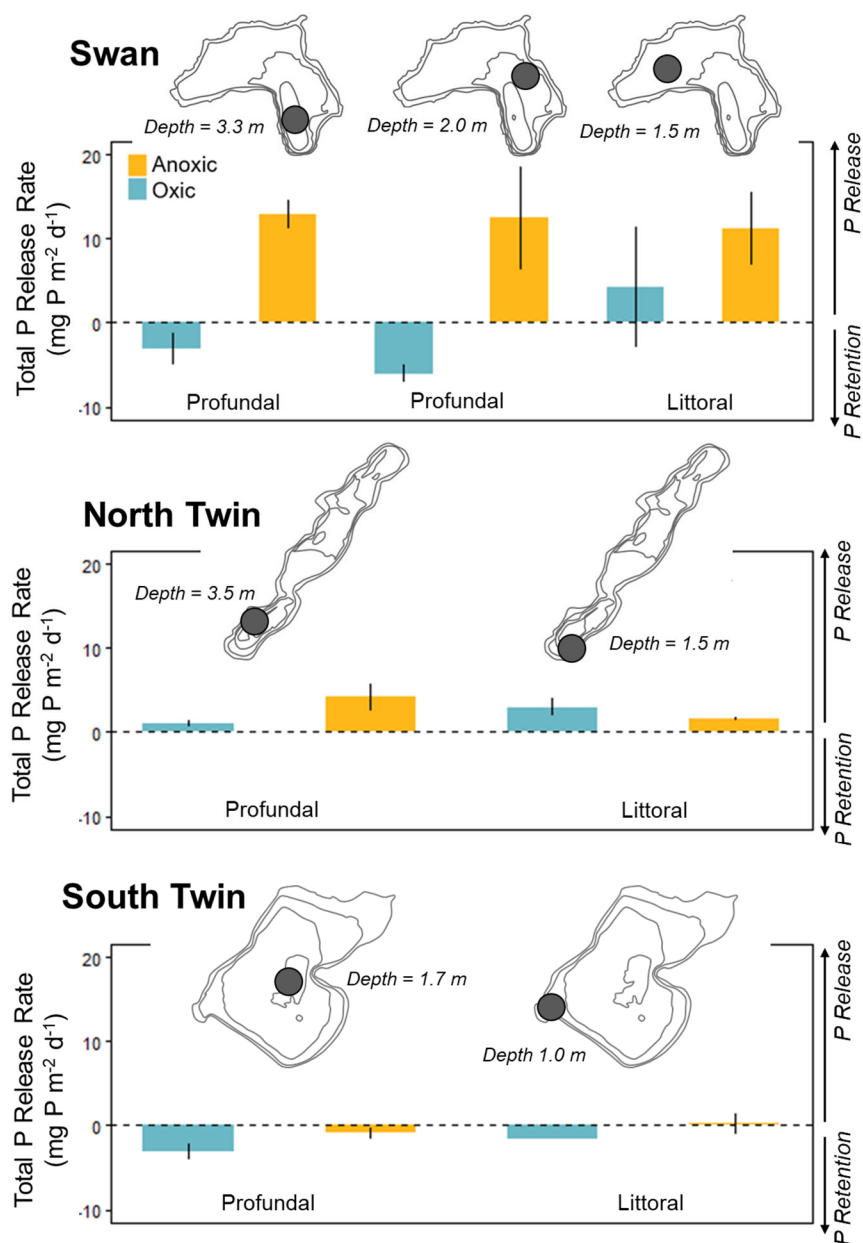


Figure 17. Sediment P flux rates from (A) Swan Lake, (B) North Twin Lake, (C) and South Twin Lake under anoxic and oxic conditions. Bars represent mean values from replicate cores and error bars show one standard error ($n=2$). Sampling site water depth and location are shown on a bathymetric map for each lake above the data for each site. Data source: Carp Biomanipulation Study.

In addition to Green Valley Lake, we also investigated the influence of sediment P composition on anoxic versus oxic sediment P flux rates for three glacial lakes in northwest Iowa (i.e., Swan, North Twin, and South Twin Lakes; Table 3). These lakes differed in their sediment P composition, suggesting that the dominant mechanisms of internal loading might also differ. Redox-sensitive P was the dominant form in Swan Lake while North and South Twin Lakes had a higher prevalence of organic P (Table S3, Figure 9). For Swan Lake, we measured high rates of sediment P release from profundal sediments under anoxic conditions and sediment P retention under oxic conditions (Figure 17A). These patterns suggest that anoxia prompts sediment P release in Swan Lake due to the dissolution of redox-sensitive phosphate minerals. These minerals are stabilized when dissolved oxygen is present, and the sediments retain P. However, both aerobic and anaerobic sediment P release were observed in a shallow, vegetated site of Swan Lake, suggesting that additional P mobilization mechanisms, such as aerobic decomposition of organic matter and subsequent P mineralization, also occur in the lake.

In North Twin Lake we measured sediment P release under both oxic and anoxic conditions, at littoral and profundal sites (Figure 17B). Anaerobic sediment P release in the lake could be due to dissolution of redox-sensitive forms or anaerobic decomposition of organic P. Aerobic sediment P release in the lake is likely due to the high organic P levels, so oxic conditions promoted decomposition and P mineralization. However, there may also be episodic P-release during sediment resuspension events in shallow regions, but this phenomenon is not captured in the sediment core incubations shown in Figure 17.

Differences in sediment P composition between Swan and North Twin Lakes lead to a different internal loading response to dissolved oxygen, with important management implications. For example, bottom water aeration, a management approach to limit anaerobic internal P loading, might protect redox-sensitive minerals and reduce sediment P release in Swan Lake, but the same approach in North Twin Lake could stimulate decomposition and enhance aerobic P release (Horppila et al., 2017, Tammeorg et al., 2017). Inter-lake variation in sediment P composition means that management strategies to control internal P loading are unlikely to translate well across lakes with different sediment P chemistry. Despite similarities in sediment P composition between North and South Twin Lake, South Twin Lake did not exhibit sediment P release. Rather, no change or slight P retention was measured under both oxic and anoxic conditions (Figure 17C). These results suggest that diffusive fluxes may not contribute to sediment P release in South Twin Lake; however, internal loading could still occur due to sediment disturbance.

2.8 Summary

- Sediment P may be found in many different chemical forms, including minerals, organic materials, and sorbed to particle surfaces.
- The chemical composition of the sediment P pool determines what conditions will likely cause internal P loading because different P forms are vulnerable to different mobilization processes.
- Labile organic, redox-sensitive, and loosely-bound P more likely to be mobilized and recycled into the water column while refractory organic and other mineral-associated P forms are more likely to remain in the sediments and be buried.
- The chemical composition of sediment P varies among different waterbodies in Iowa based on watershed soils, lake productivity, and basin morphometry.
- Sediment P also varies within individual waterbodies with concentrations generally increasing with water depth and greater variability found in lakes with complex bathymetry.
- Based on our study of seven glacial lakes and one reservoir, redox-sensitive and labile organic P are consistently the most abundant forms of mobile P in Iowa waterbodies. As such, dissolved oxygen availability is a key mechanism driving internal P loading in Iowa waterbodies.
- Internal P loading can occur under both oxic and anoxic conditions in Iowa waterbodies, but different forms of sediment P will be mobilized and released under different dissolved oxygen conditions. Oxic conditions will primarily mobilize labile organic P whereas anoxic conditions will primarily mobilize redox-sensitive P.
- The importance of aerobic versus anaerobic internal P loading in Iowa lakes and reservoirs may vary among different waterbodies based on differences in sediment P composition.
- The balance of aerobic versus anaerobic processes may also vary across seasons in individual waterbodies as dissolved oxygen and water column mixing are dynamic in Iowa waterbodies.

SECTION 3 – RELEASE FROM SEDIMENTS INTO THE WATER COLUMN

3.1 Physical Processes that Release Mobilized P from the Sediments

Once P has been mobilized from minerals or organic matter within the sediments, it may move into the overlying water due to the physical processes of diffusion and advection. Diffusion is the net movement of particles due to the random motion of individual molecules. Particles will diffuse from areas of high concentration to areas of low concentration until an equilibrium (i.e., uniform concentration) is reached. For example, the concentration of phosphate (PO_4^{3-}) is often higher in the pore water of bottom sediments than in the bottom waters of the lake. In this situation, phosphate will diffuse out of the sediments and into the overlying water along the concentration gradient. Internal P loading may also occur due to advection, which is the movement of particles in fluid due to bulk fluid motion or flow. For example, if the water column of a lake mixes fully, turbulent water flow may transport P in the bottom waters to the surface. Additionally, mixing events may disturb the surface sediments, releasing mobilized P into the water column. Diffusive processes occur quite slowly while advection has the potential to transport P rapidly from the sediments into the water column.

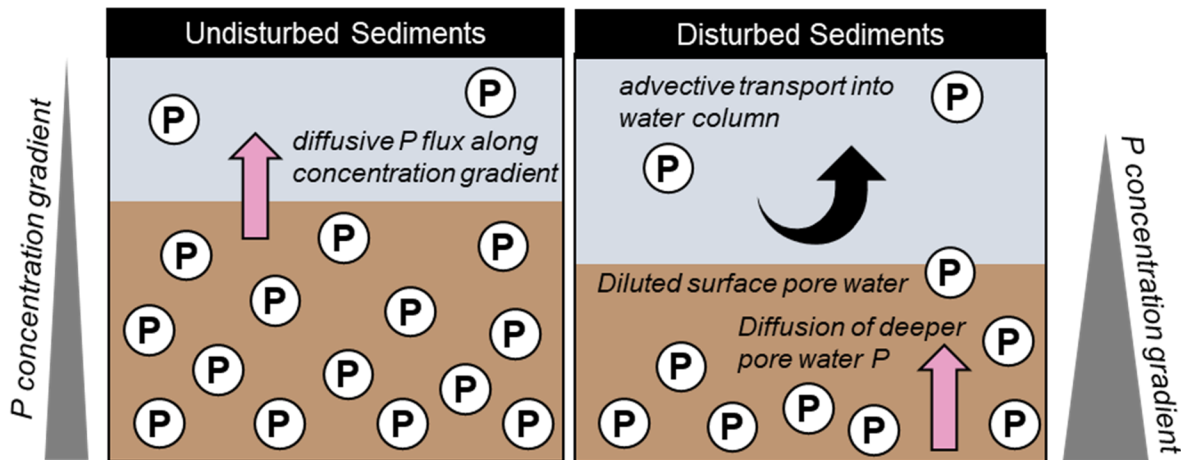
Within a given waterbody, thermal structure is a powerful control on the physical transport of mobilized P from sediments into the overlying water. Thermal stratification versus water column mixing plays a central role in determining the concentration gradient of P across the sediment-water interface, and thus whether internal P loading via diffusion can occur. When a waterbody is stratified, the hypolimnion is relatively isolated from the mixed surface layer such that solutes like phosphate may build up in the bottom waters. If hypolimnetic phosphate concentrations become equal to or greater than sediment pore water phosphate concentrations, diffusive internal P loading will no longer be favorable. However, mixing events will distribute bottom water P throughout the water column, thus lowering the concentration immediately above the sediments and reestablishing a concentration gradient that favors diffusion from the sediments. Mixing events, especially those associated with strong wind events, may also disturb the sediments and transport P into the overlying water via advection (see section 3.2.2 for more discussion of sediment disturbance in Iowa waterbodies). Additionally, the flushing of pore water P in the surface sediments will alter the P concentration gradient within the sediment profile such that P deeper within the sediments will start diffusing towards the sediment-water interface (Tammeorg et al. 2016, Tammeorg et al. 2020; Box 2). The stratification regime of a given waterbody plays a central role in the physical transport of P from sediments into the water column as well as determining the chemical environment at the sediment-water interface and the frequency of sediment disturbance.

3.2 Mixing Dynamics in Iowa Waterbodies

The mixing regimes of lakes and reservoirs in Iowa fall along a spectrum from polymictic to dimictic. Polymictic waterbodies experience frequent water column mixing throughout the open water season while dimictic waterbodies are stratified for the summer months with mixing events in the spring and fall. One method to quantify where a waterbody falls along this spectrum is to measure the frequency at which a thermocline (i.e., an indication of thermal stratification) is observed during the summer. If a thermocline is observed nearly 100% of the summer, then a waterbody is likely dimictic. Conversely, if a thermocline is rarely observed, the waterbody is polymictic. The distribution of thermocline occurrence frequency for Iowa lakes and reservoirs is bimodal, indicating that most waterbodies are either polymictic or dimictic (Figure 18). Summer thermal structure is strongly associated with basin morphometry, especially water depth (Figure 18). In general, polymictic lakes in Iowa have a maximum depth less than 10 meters

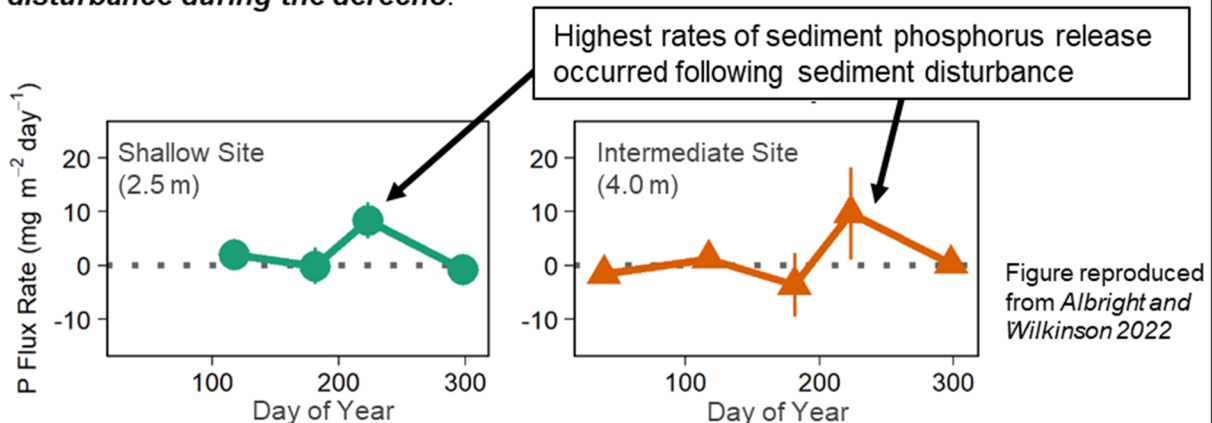
Box 2. Sediment disturbance increases diffusive internal phosphorus loading in Green Valley Lake

Sediment resuspension has been shown to increase diffusive phosphorus fluxes from sediments into the overlying water, even after the sediments have settled following the disturbance. Resuspension dilutes the pore water near the sediment surface, prompting diffusion of soluble phosphorus from deeper within the sediment profile and thus enhancing diffusive phosphorus fluxes into the water column.



We sampled Green Valley Lake immediately following a derecho or intense windstorm that passed over Iowa on 10 August 2020. Although Green Valley Lake lay at the edge of the derecho's path, windspeeds at the reservoir are estimated to have exceeded 65 kph. Following the derecho, the relative contribution of inorganic solids to the total suspended solids pool was much greater than any other point throughout the year in both surface and bottom waters, suggesting that the **storm resulted in sediment disturbance and a well-mixed water column**.

We collected sediment cores from Green Valley and measured the diffusive exchange of phosphorus between the sediments and the overlying water over the next four days. We recorded very high rates of sediment phosphorus release from cores taken along the western arm of the reservoir (water depth 2.5 and 4 m). It is likely that the **high rates of diffusive phosphorus flux into the water column resulted in part from the sediment disturbance during the derecho**.



(Figure 18), and the frequency of thermocline occurrence increases with greater maximum depths. While this not true of every lake, the paucity of profiles throughout the season led to the selection of the conservative 10 m threshold to characterize polymictic lakes. A similar relationship is seen with mean depth. Waterbody surface area does not have a strong relationship with stratification frequency for Iowa waterbodies. In addition to lake basin morphometry, internal structure from aquatic plants or macrophytes can play on role in determining thermal structure (Box 3).

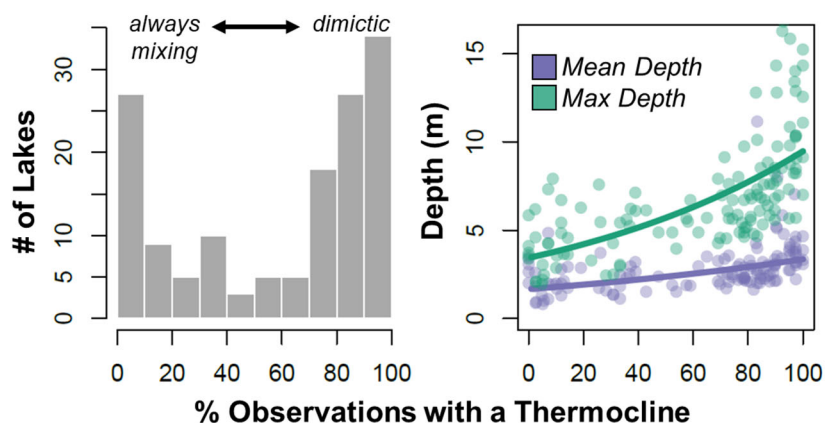


Figure 18. The percentage of times stratification has been observed in a given lake ($n=12-57$ observations per lake) in Iowa based on temperature profiles taken three times every summer from 2007-2020 (left) and the relationship between mean and maximum depth and stratification frequency (right).

High frequency measurements of water column thermal structure across seven waterbodies from northwest to southwest Iowa reveal that deeper lakes stratify more frequently and for longer periods (e.g., Five Island and Swan Lakes) whereas very shallow lakes mix almost continuously throughout the summer (e.g., South Twin Lake; Figure 19). Although these lakes covered a broad geographic area, mixing and stratification events are surprisingly synchronous among the waterbodies, suggesting a strong influence of broad-scale meteorological events. Climate change is expected to alter mixing dynamics in Iowa's waterbodies. Specifically, increased nighttime temperatures and a shortening or loss of ice cover will lead to stronger water column stratification while stronger storm fronts and weather whiplash will make stratification conditions less predictable from year to year.

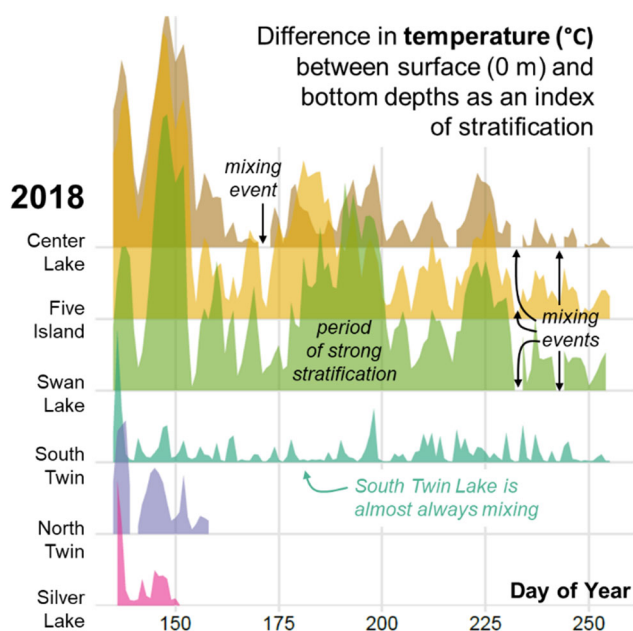
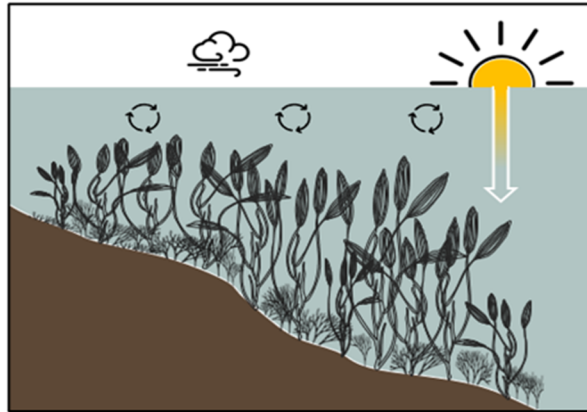


Figure 19. Time series of daily mean temperature difference between the top and bottom waters in six Iowa lakes during summer 2018. Periods with large temperature differences are indicative of stratification, whereas periods with little temperature difference are indicative of water column mixing. These high frequency time series reveal the polymictic nature of most Iowa lakes. The full raw time series of temperature difference for these lakes in 2018 and 2019 are in Figure S3.

Box 3. Macrophytes structure thermal dynamics in shallow ponds

Canopy-forming macrophyte beds attenuate incoming solar radiation and dissipate wind-driven turbulence, thus limiting the depth of heat transport and creating a shallow mixed surface layer separated from cooler bottom waters. In short, **macrophytes induce thermal stratification in small waterbodies.**



We quantified the influence of macrophytes on thermal structure in two shallow, vegetated ponds. We experimentally pulsed one of the ponds with nutrients to simulate a storm-driven external loading event, while the other pond served as a reference. In the reference pond, both submersed and floating-leaf macrophytes followed the expected phenology of spring emergence, summer growth, and then senescence beginning in late summer. In contrast, submersed macrophytes in the nutrient addition pond began to senesce in mid-summer after the first experimental nutrient pulse (day of year 176). This early senescence was dominated by submersed macrophytes in the deeper, central region of pond, suggesting that light limitation from phytoplankton shading was a contributing stressor. Early senescence of submersed macrophytes in the nutrient addition pond altered the thermal environment in relation to the reference pond. **Early macrophyte senescence resulted in destratification and warmer bottom water temperatures.**

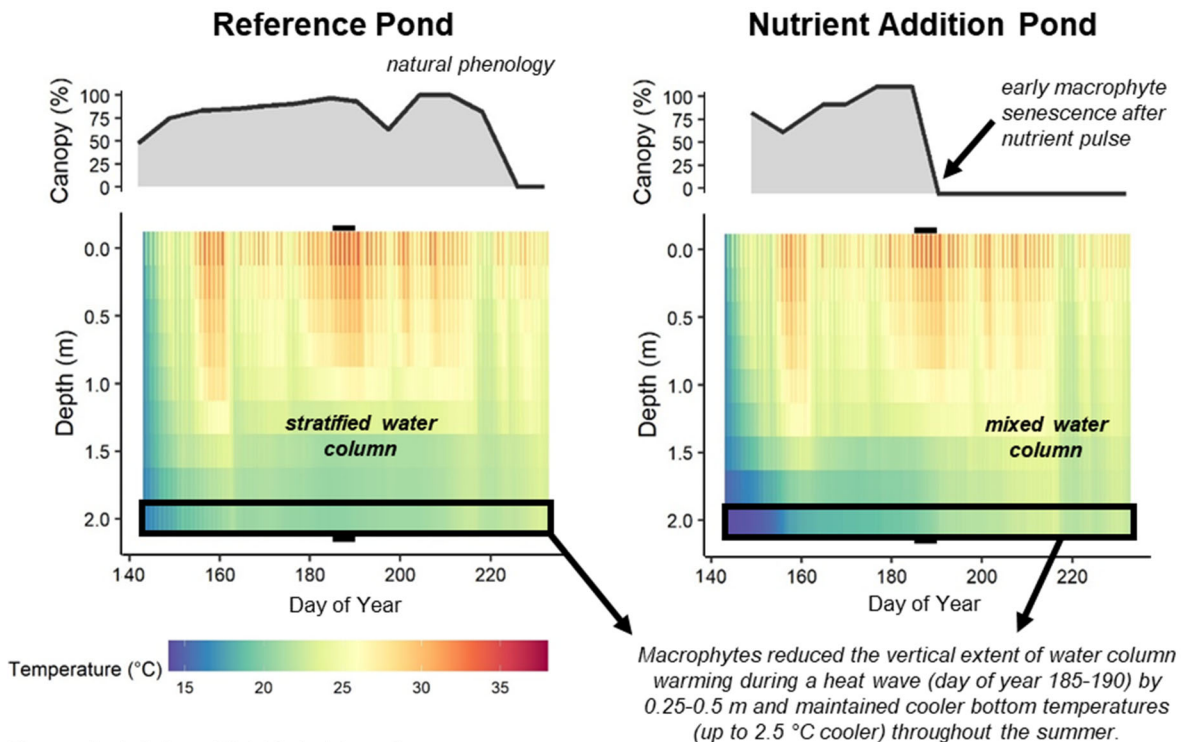


Figure adapted from Albright et al. in review

3.2.1 Mixing and Dissolved Oxygen

The dissolved oxygen saturation at the sediment-water interface in Iowa's lakes varies over the course of the season and at locations within the waterbody. During the late spring and early summer (May-June), approximately half of Iowa's publicly significant lakes are hypoxic (here defined as 20% dissolved oxygen saturation) in the water directly above the sediment surface (Figure 20, left panel). As the summer progresses and surface waters warm, more lakes become stratified and microbial respiration consumes oxygen in the bottom waters, driving hypoxia. However, there is a population of lakes (approximately ¼ of lakes surveyed) that do not have hypoxic bottom waters.

Stratification is the main factor influencing whether the bottom waters of Iowa lakes are hypoxic or oxygenated (Figure 20, right panel). When the temperature difference between the surface water and bottom depth is greater than 2°C (stratified), dissolved oxygen saturation is hypoxic in the hypolimnion. When there is little to no temperature difference vertically in the water column, dissolved oxygen saturation at the sediment-water interface ranges from anoxic to well oxygenated. The population of polymictic lakes in the state (Figure 18) likely oscillate between oxygenated and hypoxic conditions in the bottom waters, depending on the length of time the water column is stratified (Figure 19) and the pool of organic matter decomposing in the sediments. Conversely, dimictic lakes are likely to only be oxygenated in the hypolimnion following spring turnover, progressing towards anoxia during the rest of the summer. Climate-driven changes in the mixing regime will alter hypoxia of bottom waters, likely feeding back to internal loading rates.

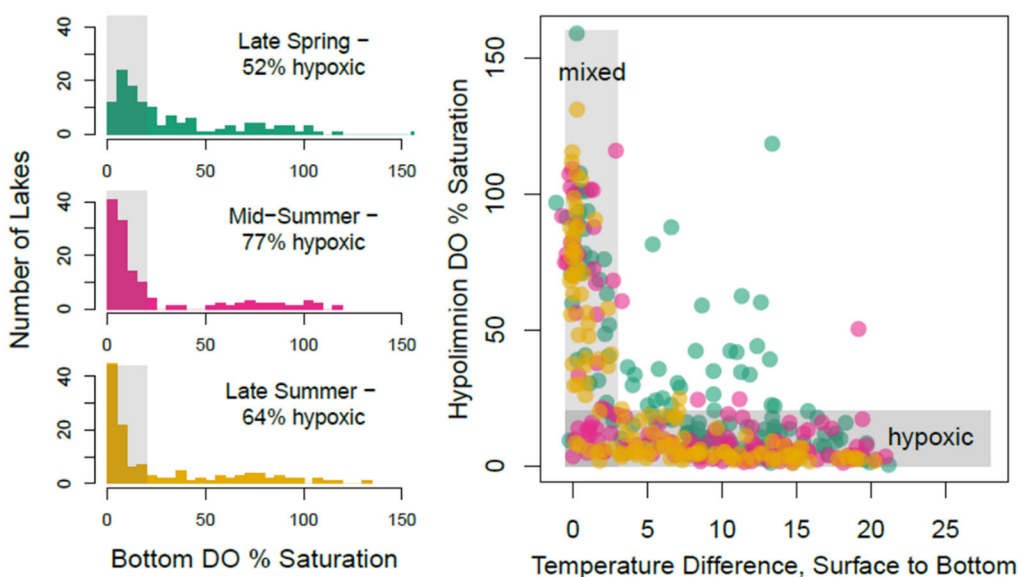


Figure 20. Histogram of dissolved oxygen saturation in Iowa lakes during three sampling rounds in 2018 (left side) and the frequency of hypoxia (<20% saturation) among the population of lakes in the bottom water. Later in the season, when lakes are stratified, the bottom waters are more likely to be hypoxic (right side, colors are the same sampling rounds as left side panels). When lakes are mixed, regardless of time in the open water season, dissolved oxygen saturation is quite variable. In the spring (green points), dissolved oxygen saturation is less related to stratification. Data source: IDNR 2022.

The concentration of dissolved oxygen also varies spatially within a lake. In the surface waters, algal blooms and macrophyte phenology can drive hot spots and hot moments of high dissolved oxygen concentrations (Figure 21; Ortiz and Wilkinson 2021). Similarly, there are regions of persistent hypoxia

in the bottom waters of lakes while other regions maintain oxygenated waters throughout the summer (Albright and Wilkinson 2022b, Figure 21). For example, in Green Valley Lake, the deeper stratified portions of the lake are hypoxic throughout the summer while the upper arms remain mixed and oxygenated. The sites at intermediate depths periodically stratify and become hypoxic (Figure 22).

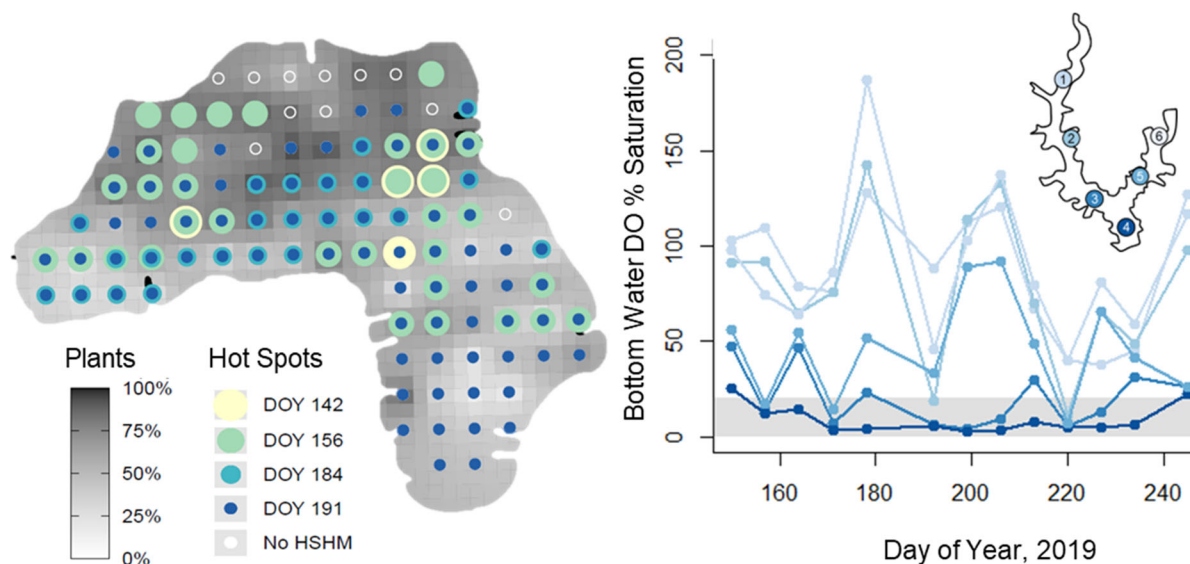


Figure 21. The 98 sampling locations on Swan Lake shown as circles (left panel). The colors correspond to a sampling date in 2018 identified as having a hot spot or hot moment of high dissolved oxygen concentrations in the surface water. Concentric circles indicate that a sampling point was identified as a hot spot observation on multiple sampling dates. The gray scale shading is the macrophyte observation index from 0% (rooted macrophytes never observed at the sampling site) to 100% (rooted macrophytes always observed at the sampling site). Analysis from Walter et al. (2023). Dissolved oxygen percent saturation in the water 0.25-0.5 m above the sediment surface in Green Valley Lake at six sampling locations.

3.2.2 Mixing and Sediment Disturbance

Water column mixing events may disturb the sediments and release P from pore water or resuspended sediment particles (see section 2.5 for an overview of loosely-bound sediment P and Box 2 for an example of increased sediment P release following a mixing event). Whether or not P will be released from resuspended sediment particles is largely determined by the P sorption capacity of the sediments and the concentration of aqueous P (i.e., phosphate or soluble reactive P, SRP) in the water column. Equilibrium P concentration (EPC; James et al. 2017b) is a simple method to quantify sediment P sorption capacity (see Appendix Text S1 for calculations). EPC refers to the aqueous-phase inorganic P concentration at which adsorption and desorption processes are in equilibrium (Figure 22). The EPC may be used to determine whether resuspended sediments are likely to release P. More specifically, the adsorption of P to sediments is favored when hypolimnetic SRP concentrations are greater than EPC. Conversely, sediment P release via desorption is more favorable when hypolimnetic SRP is less than EPC. A higher EPC means that sediments are more likely to act as a source of P because desorption will be favored more often. As a measure of sediment P sorption capacity, EPC is influenced by many factors including sediment particle size, sediment mineralogy, and the amount of exchangeable P already sorbed to sediment surface.

We measured EPC in five glacial lakes in northwest Iowa (Table 3) as well as at six sampling sites across Green Valley Lake (Figure 22). EPC values were highly variable among the six waterbodies (range 49.4 to 235.3 $\mu\text{g L}^{-1}$) and within Green Valley Lake (range 46 to 175 $\mu\text{g L}^{-1}$). There were no significant

relationships between EPC and water depth or sediment physical characteristics such as bulk density or organic matter content. All the EPC values were higher than typical bottom water SRP concentrations measured during the same period in the study lakes (Data Source: *Albright and Wilkinson 2022b, IDNR 2022*). Since bottom water SRP concentrations are often less than the measured EPC values, is likely that resuspended sediments are a source of P to the water column in these waterbodies.

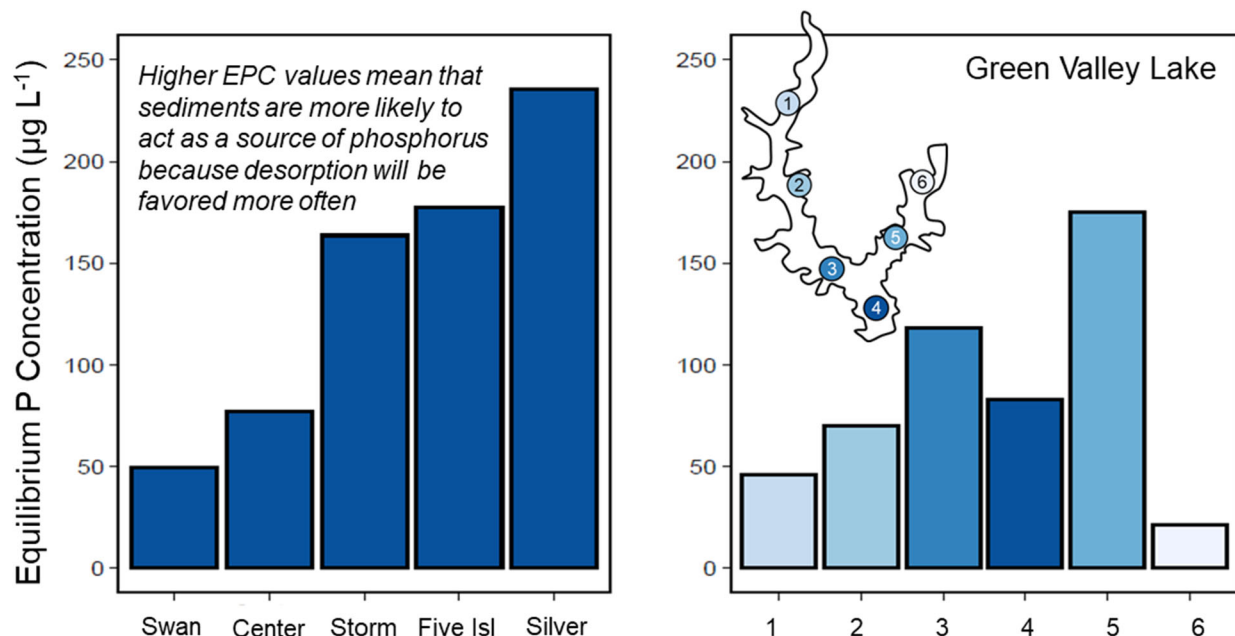


Figure 22. Equilibrium P concentrations at the deep site of five glacial lakes in northwest Iowa (left panel, 2018) and at six sites across Green Valley Lake (right panel, 2019). EPC refers to the aqueous-phase inorganic P concentration at which adsorption and desorption processes are in equilibrium and is a simple method to quantify sediment P sorption capacity. Data source: Carp biomanipulation study.

3.3 Summary

- Diffusion and advection are physical processes that transport mobilized P from the sediments into the overlying water.
 - Diffusion rates are determined by concentration gradients of dissolved P and are relatively slow.
 - Advective transport occurs during bulk water movement and has the potential to move P from the sediments into the water rapidly.
- The thermal structure of the water column is an important control on the physical transport of P from sediments into the water.
- Iowa lakes and reservoirs are either polymictic, with frequent water column mixing throughout the open water season, or dimictic, with mixing events largely limited to the spring and fall.
- In polymictic waterbodies, summer mixing events are primarily driven by air temperature, wind speed, and precipitation, and these events may be synchronous across a broad geographic area.
- Thermal structure interacts with lake basin morphometry and internal water column structure (e.g., macrophytes).

- o Deeper waterbodies are more likely to be stratified for longer periods.
- o Aquatic macrophytes promote thermal stratification, even in shallow waters.
- Thermal structure influences the chemical environment at the sediment water interface, especially dissolved oxygen. In Iowa lakes and reservoirs:
 - o Bottom waters tend to remain well-oxygenated in waterbodies with frequent water column mixing, but bottom water hypoxia may still occur in a mixed waterbody.
 - o Bottom water hypoxia frequently develops in stratified waterbodies, especially in late summer.
 - o Dissolved oxygen is spatially heterogeneous across individual waterbodies and dynamic over time.
- Thermal mixing events influence the frequency of sediment disturbance, which may release P into the water column.
- Iowa waterbodies experience resuspension of bottom sediments.
 - o Based on the P sorption capacity of these sediments, resuspension events likely result in P release into the water column.

SECTION 4 – AQUATIC ORGANISM MEDIATED P-CYCLING

4.1 Zooplankton

As a group, zooplankton have a wide array of feeding strategies including grazing on phytoplankton and preying upon microzooplankton. Nutrient recycling by consumers such as zooplankton and fishes influences aquatic ecosystem function by driving the transport and transformation of nutrients. Zooplankton excrete and egest metabolized and unassimilated organic matter that they consume as waste, recycling nutrients (N and P) back into the ecosystem. Imbalances between zooplankton and phytoplankton elemental composition (i.e. N:P ratio) drive the stoichiometry of nutrients recycled. For example, *Daphnia*, a keystone grazer in lakes, is relatively P-rich compared to copepods (Figure 23A). In low P lakes, *Daphnia* often dominate the zooplankton community, storing P in their tissues (Moody and Wilkinson 2019). However, in hypereutrophic lakes, less P-rich zooplankton such as copepods dominate leading to greater P recycling through the zooplankton community.

While the magnitude of P recycling through zooplankton is small compared to other internal cycling processes on an annual timescale, zooplankton recycling can be an important source of bioavailable P early in the ice-free season. A recent study of hypereutrophic Green Valley Lake estimated that zooplankton excretion of P was approximately 20-60% of the standing stock of soluble reactive P in the surface waters in the early to late spring season (Figure 23B; Butts et al. 2022). Additionally, *Daphnia* in these P-rich lakes have undergone rapid adaptation to eutrophication resulting in a decline in grazing pressure by this keystone species (Appendix Figure S2; Moody et al. 2022). Zooplankton recycling of P and the loss of top-down grazing pressure are both mechanisms by which animals contribute to P cycling in the water column of lakes.

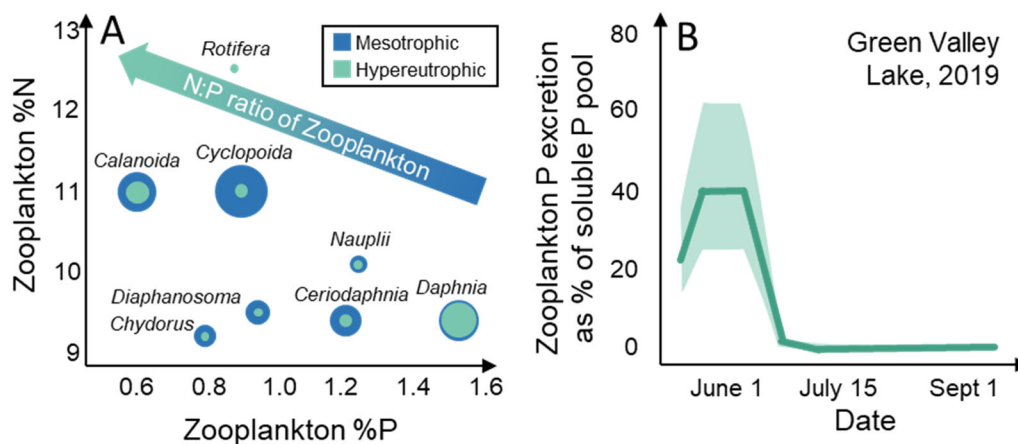


Figure 23. A) The nutrient content and abundance of zooplankton taxa in mesotrophic (blue) and hypereutrophic (green) Iowa lakes. The size of the points scales with the relative abundance of that taxa in each trophic level across all Iowa lakes. The N:P ratio of the zooplankton community increases with trophic state (Moody and Wilkinson 2019), leading to more zooplankton P nutrient recycling in hypereutrophic lakes that make a substantial contribution early in the summer (Butts et al. 2022).

4.2 Common Carp and Bioturbation

Common carp (*Cyprinus carpio*) are a highly influential species for P cycling and may exacerbate internal P loading. As a large benthivore, common carp disturb sediments while foraging, which is a direct

pathway for internal P loading (Huser et al. 2016). Lakes with high carp biomass often have water quality and fisheries concerns (e.g., low water clarity, high nutrient concentrations, reduced macrophyte structure). As such, common carp removal is a restoration tool for controlling internal P loading.

Six glacial lakes in northwest Iowa were involved in a study of the ecosystem effects of incentivized harvest of common carp and bigmouth buffalo from the winter of 2018 through 2020 (Table S4). These fishes were harvested from Center and Five Island lakes in the winter of 2017-2018 and 2018-2019 and from North Twin and Silver Lakes in the winter from 2018-2019 and 2019-2020. Storm and South Twin Lakes served as reference systems. The biomass of common carp removed varied across harvest seasons and lakes (range 10-71 kg ha⁻¹, median 16 kg ha⁻¹, Table S4).

The concentration of soluble reactive P (SRP) in surface and bottom waters was variable over the summer within each study lake (Figure 24). Silver Lake exhibited declines in SRP following the largest carp harvest in winter 2019-2020 (approximately 59% of the summer 2019 population). This decline may have been due to reduced internal loading associated with carp, supported by the increasing or constant SRP concentrations in the reference ecosystems during this time period. The concentration of inorganic suspended solids (ISS) followed a similar pattern to SRP (Figure 25). Moderate declines were seen in Silver Lake following carp harvest. No trends were noted in the other harvested lakes. Since ISS concentrations either increased or stayed constant in the reference systems, declines in ISS in Silver Lake could be due to reduced sediment resuspension due to reduced carp bioturbation. It seems that the large difference in biomass harvest between Silver Lake and the other harvested lakes affected the water chemistry response to the biomanipulation.

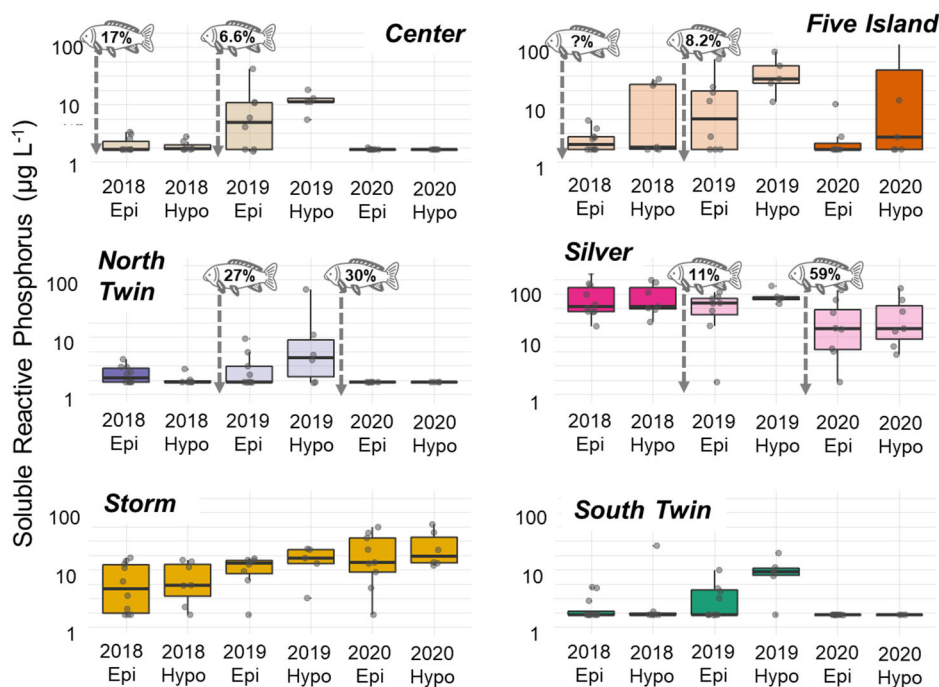


Figure 24. Soluble reactive P in the epilimnion and hypolimnion with % of the common carp population harvested in the winter compared to the median estimate of the previous summer's population (Table S4). There's no population estimate in 2017 for Five Island, but based on the 2017-2018 harvest and 2018 population estimate, <10% of the population was likely harvested.

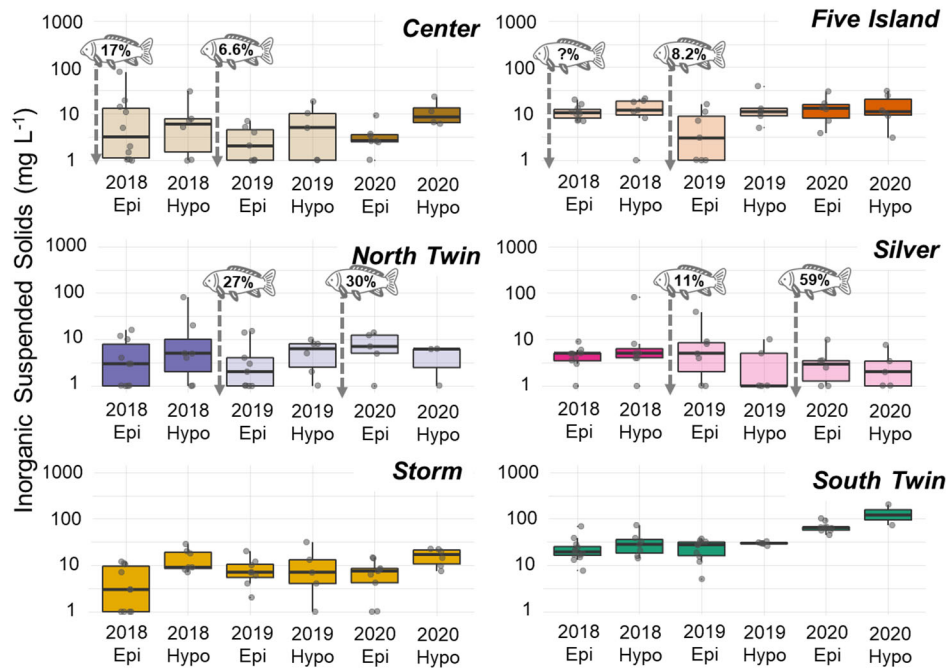


Figure 25. Inorganic suspended solids in the epilimnion and hypolimnion with % of the common carp population harvested in the winter compared to the median estimate of the previous summer's population (Table S4). There's no population estimate in 2017 for Five Island, but based on the 2017-2018 harvest and 2018 population estimate, <10% of the population was likely harvested.

4.3 Macrophytes

Common carp also uproot macrophytes when foraging. Reductions in macrophyte coverage may allow for further sediment P release because macrophyte roots stabilize sediments, reducing resuspension and promoting sediment P retention (Figure 26; Carpenter and Lodge 1986; Granéli and Solander 1988). Removal of common carp could promote macrophyte recovery (Bajer and Sorensen 2015).

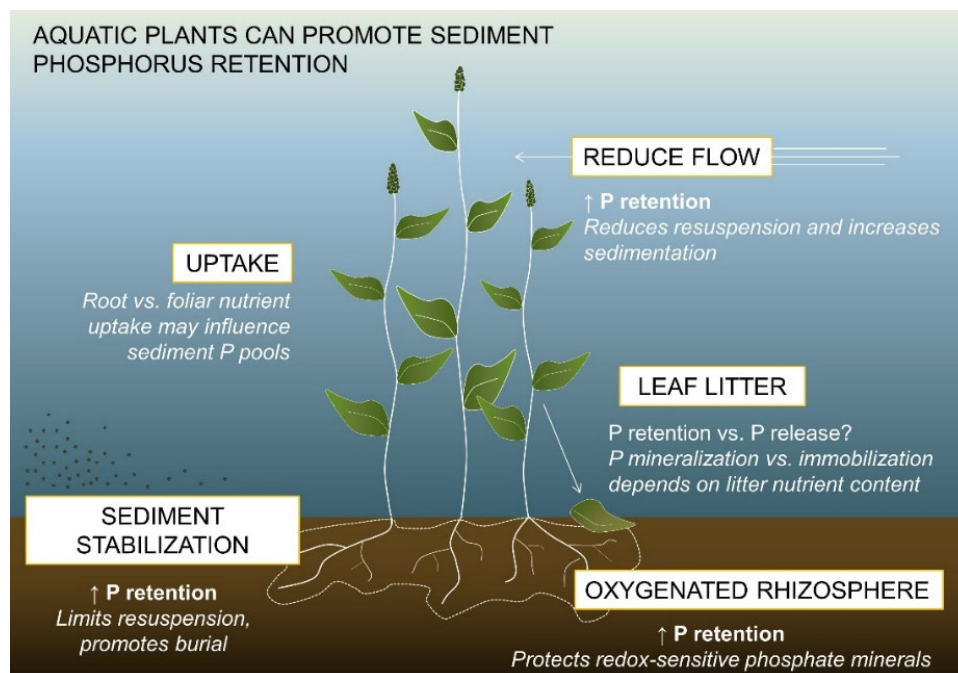


Figure 26. Conceptual overview of the various processes by which aquatic plants (macrophytes) can reduce internal P loading by promoting sediment P retention. Original illustration E. Albright.

Macrophyte coverage and relative biomass were evaluated following carp removal through the incentivized harvest program (Figure 24). Macrophyte coverage was quantified as the proportion of sampling transects where macrophytes were found, either through a visual survey within 2 m of the boat (emergent species) or on a sampling rake (submerged species).

Macrophyte coverage appears to have increased in some lakes following common carp removal (Figure 27). Silver Lake had the largest increase in macrophyte coverage following harvest (48 to 84% between 2019 and 2020), likely due to the high carp biomass removed in winter 2019-2020 (71 kg ha⁻¹). Slight increases in macrophyte coverage occurred following carp removal in Center and North Twin Lakes. No change was observed in Five Island Lake where harvest densities were low (<10% of the population). Macrophyte coverage remained constant in the South Twin (reference lake), where high coverage was due to dense stands of emergent vegetation. Increases in coverage in Storm Lake (reference lake) could be due to increased water clarity following zebra mussel (*Dreissena polymorpha*) establishment.

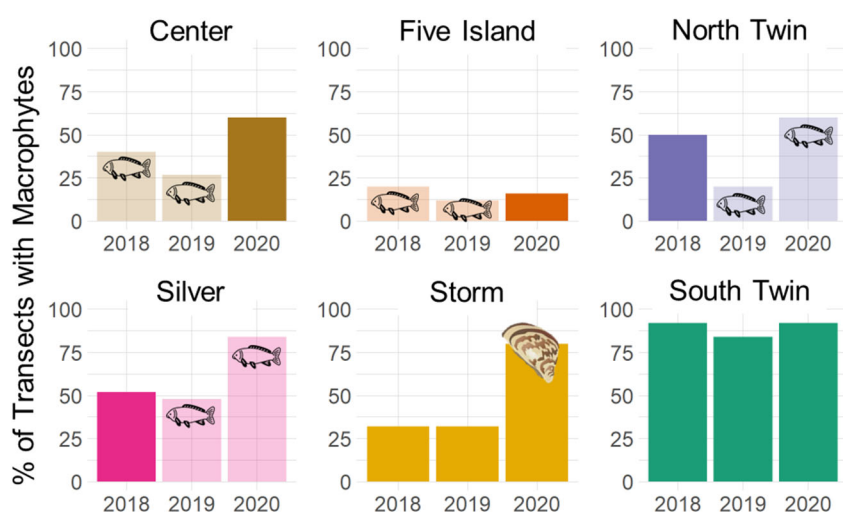


Figure 27. Macrophyte coverage (2018-2020) before (dark color) and after (transparent color) common carp removal and in reference lakes. Coverage was quantified as the percent of sampling transects where macrophytes were found. High coverage in South Twin Lake is due to cattail and bulrush beds surrounding most of the lake.

Relative biomass was determined as a qualitative assessment of plants collected on a sampling rake, which predominantly captures submersed species. Relative biomass followed the same patterns across lakes as submersed species richness (Figure 28). Relative biomass and submersed species richness only increased following carp harvest in Silver Lake. No change over time was noted in the other harvested lakes and South Twin Lake. The increases seen in the Storm Lake could be due to zebra mussels.

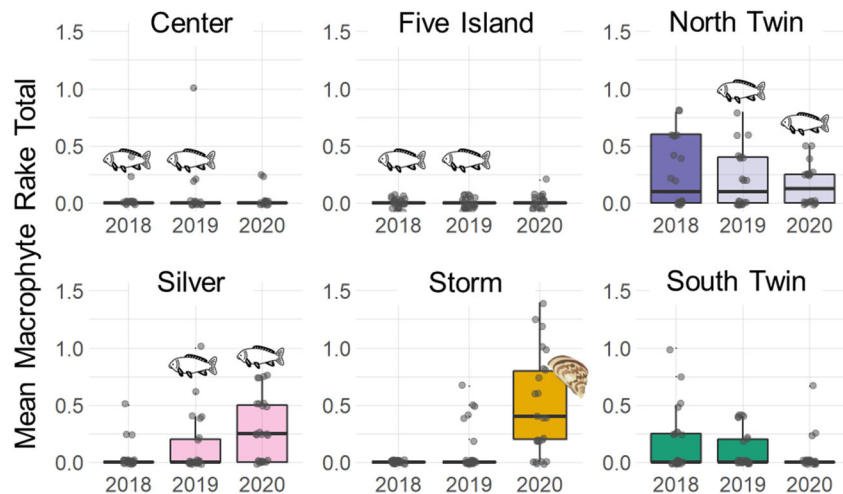


Figure 28. Qualitative estimate of submersed macrophyte biomass (2018-2020) before (dark color) and after (transparent color) common carp removal and in reference lakes. Biomass was estimated on a qualitative scale from 0-3 based on the density of plants pulled up on the sampling rake. Boxplots show variation among sampling transects within each lake..

4.4 Food Web Manipulation through Incentivized Harvest

Bigmouth buffalo are large-bodied planktivores found in eutrophic and turbid shallow lakes. The diet of bigmouth buffalo is dominated by zooplankton that they filter out of the water through their fine gill rakers, mainly consuming copepods and large-bodied cladocerans such as *Daphnia* (Starostka and Applegate 1970). As bigmouth buffalo are planktivores, temporarily lowering densities through commercial harvest is likely to result in cascading top-down trophic interactions (Carpenter et al. 1985). Predation pressure on zooplankton decreases when planktivorous fish are removed in large quantities, resulting in increased predation pressure by zooplankton on phytoplankton. This will, in turn, decrease the standing stock of phytoplankton, an effect which may be amplified in nutrient-rich waterbodies that bigmouth buffalo often inhabit (Carpenter et al. 2001). Lower fish densities due to harvest could also result in a shift in the zooplankton community size structure (Brooks and Dodson 1965, Pace et al. 2013) as bigmouth buffalo are thought to selectively consume large-bodied zooplankton. However, the strength of these food web interactions will be mediated by the magnitude of population changes in bigmouth buffalo with harvest.

We added bigmouth buffalo to a set of fishless experimental ponds at the Iowa State University Horticulture Research Station at densities typical of ambient (450 kg ha^{-1}) and harvested (150 kg ha^{-1}) populations (Table S4), with two additional fishless ponds as a reference (Wilkinson et al. in review). Zooplankton biomass declined in ambient density ponds after the fish addition while chlorophyll-a concentrations increased (Figure 29). In the harvested density treatment chlorophyll-a concentrations were similar to the reference ponds despite a decline in zooplankton biomass, likely due to larger zooplankton community size in the harvested ponds exerting greater grazing pressure. This experimental manipulation revealed density-dependent top-down control by bigmouth buffalo and the potential for cascading trophic interactions with harvest. However, the effects of bigmouth buffalo commercial harvest are likely to be short-lived as the population will quickly rebound once harvest pressure is removed and there are numerous fishes, including other planktivores, in natural lakes.

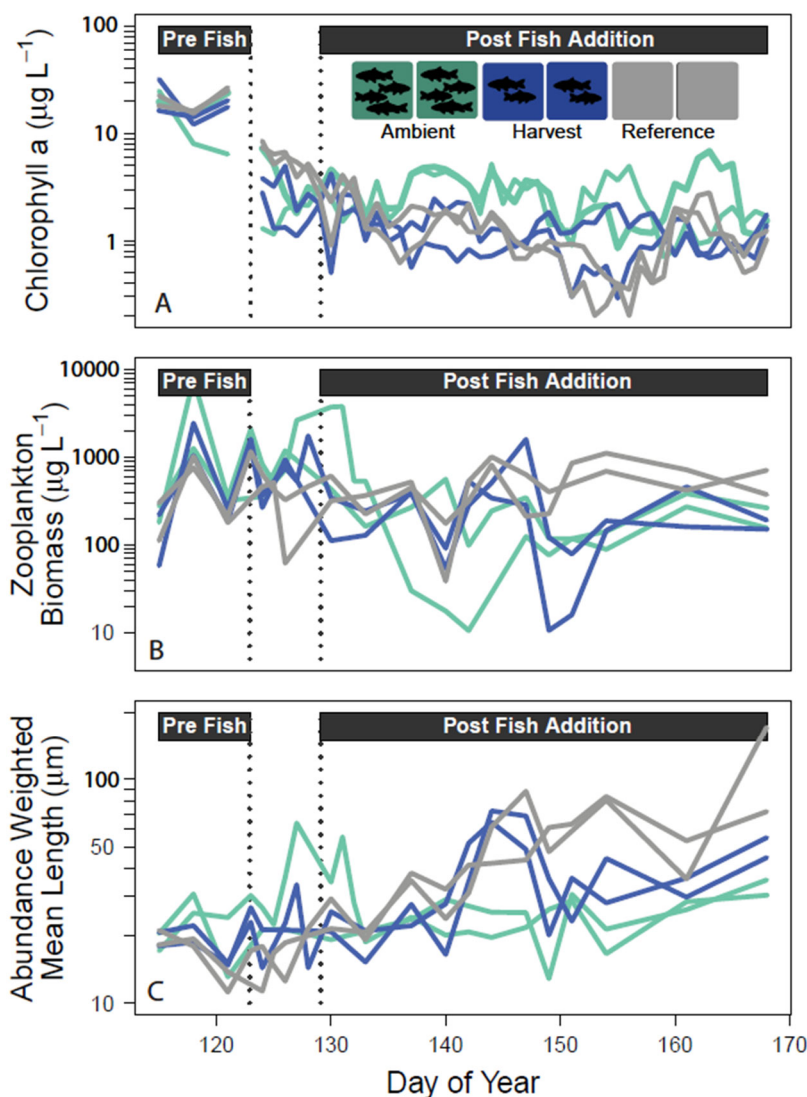


Figure 29. a) Chlorophyll-a concentrations, b) zooplankton biomass, and c) abundance-weighted mean zooplankton length from each of the experimental ponds, color coded by treatment (see legend in panel a). The pre-fish addition and post-fish addition periods are denoted on each graph by the vertical dashed lines. Wilkinson et al. (in review).

We also evaluated changes in the food web structure using size spectra following the incentivized harvest of common carp and bigmouth buffalo in the six study lakes. Size spectra—the relationship between species abundance and body size—can be used to detect changes in food web structure and ecosystem productivity following major disturbances. As such, size spectra may be a powerful tool to investigate the effect magnitude of food web interventions such as biomanipulation. We used measures of zooplankton and macroinvertebrate biomass and length in the harvested and reference lakes to evaluate changes in slope from 2018–2020 (Figure 30). Based on the size spectra slopes, there was no significant changes in trophic transfer efficiency due to incentivized harvest ($p = 0.07$), harvest intensity of common carp ($p = 0.19$), or bigmouth buffalo ($p = 0.07$) (Butts 2023, Figure 30). Through this analysis, we demonstrated that incentivized harvest was unable to significantly shift size spectra slopes as intended, providing evidence that the lack of improvements in water quality observed in the lakes may be due to a failure to induce strong top-down effects through common carp and bigmouth buffalo harvest.

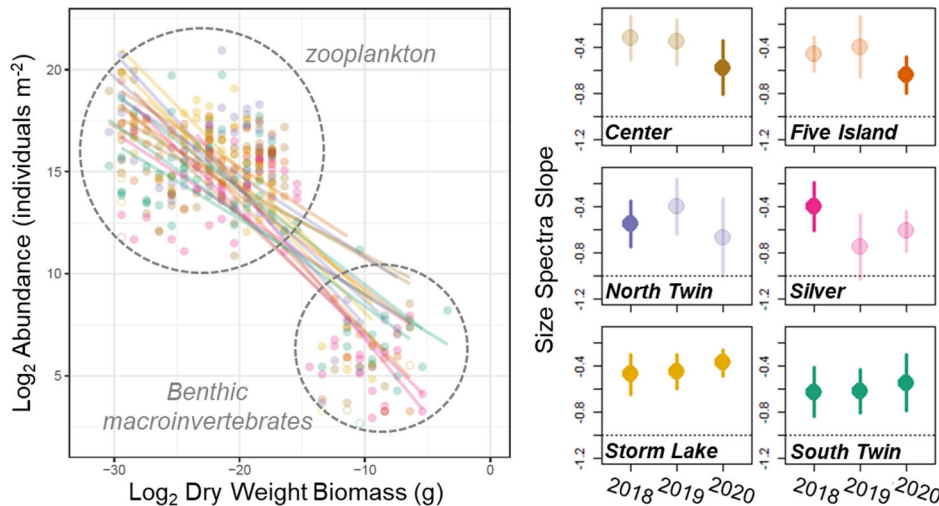


Figure 30. Size spectra for each lake year (left panel) of zooplankton and benthic macroinvertebrates. Size spectra slope (and confidence interval) by lake and year (right panels) are color coded based on harvest. Solid points indicate no commercial harvest in the preceding winter whereas transparent points indicated harvest during the prior winter.

4.5 Summary

- Zooplankton and other consumers such as fishes play an important role in nutrient recycling by excreting and egesting organic matter, thus recycling nutrients like nitrogen (N) and phosphorus (P) back into the ecosystem.
- Zooplankton recycling of P can be an important source of bioavailable P early in the summer, even in hypereutrophic lakes.
- Common carp disrupt sediments while foraging, leading to internal P loading. Carp removal is a restoration method to control internal P loading.
- Removal of common carp can also promote macrophyte recovery by reducing uprooting, thereby promoting sediment P retention.
- Simplified food webs experiments revealed that harvest of bigmouth buffalo can lead to cascading trophic interactions, with increased predation pressure by zooplankton on phytoplankton, potentially reducing phytoplankton standing stock. However, the effects of bigmouth buffalo harvest are likely short-lived, with the population likely to quickly rebound once harvest pressure is removed.

PART II

Management and Restoration Recommendations

SECTION 1 – DIAGNOSING INTERNAL LOADING MECHANISMS

1.1 Introduction to Diagnosis Flow Chart

We developed a tool to aid in building lines of evidence to identify the dominant mechanism(s) of internal P loading for a specific lake or reservoir in Iowa (Figure 31). Internal P loading is a complex biogeochemical process and the mechanisms controlling it vary over space and time within individual waterbodies. However, this diagnosis flow chart provides guidance for isolating which mechanisms are likely responsible for most internal P loads at a given location in a waterbody. The chart uses a series of questions to delineate among five mechanisms of internal P loading:

1. **Carp Bioturbation:** waterbodies with high biomass densities of common carp (*Cyprinus carpio*, a non-native benthivore) such that internal P loading is driven by carp foraging activities, including sediment disturbance and irrigation, and carp excretion of P.
2. **Wind Resuspension:** shallow waterbodies in which sediment disturbance due to wind mixing is the dominant mechanism of internal P loading.
3. **Enhanced Diffusion, Oxic Conditions:** shallow waterbodies in which internal loading is facilitated by a well-mixed water column allowing P to diffuse out of the sediments, but the sediments are protected from resuspension during mixing events (e.g., by aquatic vegetation).
4. **Redox Mediated:** waterbodies in which internal P loading primarily occurs under anoxic conditions, generally in deeper waters, due to the mobilization of P from redox-sensitive minerals.
5. **Perpetual P Cycling:** highly productive, polymictic lakes in which internal P loading occurs under both anoxic and oxic conditions, yielding high internal P loading rates that are difficult to control. This concept was originally described for hypereutrophic plains reservoirs (Song and Burgin 2017).

The diagnosis flow chart was developed for lakes and small impoundments in Iowa, using the available long-term data collected at the deepest point to define thresholds (IDNR 2022). Although the general patterns outlined in this chart may apply to other temperate waterbodies, we do not recommend that it be used to guide internal P management planning beyond Iowa. It is possible to answer the questions in the flow chart using available monitoring data (i.e., from the Shallow and Ambient Lake Monitoring Programs). However, given the coarse spatial and temporal resolution of the monitoring data, we recommend that the long-term record be used to develop hypotheses regarding the dominant mechanism of internal loading and that those hypotheses then be tested with targeted sampling, including spatially resolved sampling in dendritic reservoirs (see Part 2, Section 3.1 for detailed sampling recommendations). Additional sampling is necessary to correctly identify the dominant mechanism, critical for selecting appropriate management actions (Part 2, Section 2).

If spatial variability is high in a waterbody we recommend applying the flow chart separately to hydrodynamically distinct regions of the waterbody. For example, in Green Valley Lake the flow chart could be used at the historical deep point sampled by the Ambient Lake Monitoring program and a site of frequent stratification, and in the shallow arms of the reservoir. Applying the flow chart to different regions of the reservoir would likely reveal different dominant mechanisms (e.g., oxic release and enhanced diffusion in the arms, perpetual cycling at the deep point) and therefore spatially-explicit management actions that could be used to combat internal loading.

The diagnosis flow chart and supporting analyses complement and expand upon previous efforts to model drivers of internal P loading in Iowa waterbodies (Filstrup and Downing 2016, Capstone Project Report).

The Filstrup and Downing (2016) report focused on distinguishing between wind resuspension and redox-mediated internal P loading and did not consider internal loading under oxic conditions that was not resuspension-driven. However, several key studies have been published since 2016 that emphasize the importance of aerobic internal loading, especially in productive waterbodies (Song and Burgin 2017; Horppila et al. 2017; Orihel et al. 2017). We have incorporated this emerging understanding into our analysis by exploring internal P loading under both oxic and anoxic conditions in Iowa waterbodies. This analysis is supported by the current understanding in the scientific literature regarding mechanisms of internal P loading in productive lakes and reservoirs.

1.2 Diagnosis Flow Chart

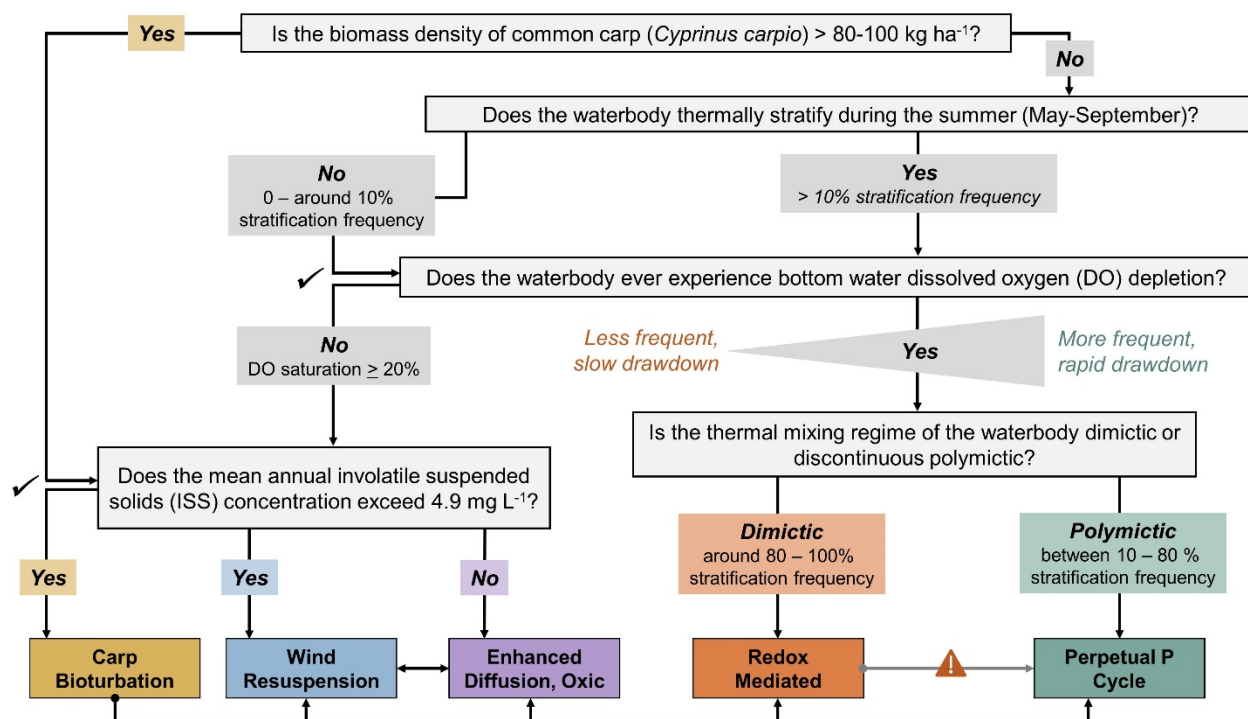


Figure 31. Flow chart of guiding questions to identify likely dominant mechanisms (bottom row) of internal P loading for a given waterbody. Thresholds are based on analyses of long-term data available for Iowa lakes and reservoirs through the IDNR Ambient and Shallow Lake Monitoring Programs as well as the scientific literature. Detailed rationale is provided in the subsequent text. **Arrows** connecting different internal P loading mechanisms indicate that a lake may shift between two mechanisms (e.g., wind resuspension and enhanced diffusion), a lake may experience more than one mechanism (e.g., carp bioturbation may co-occur with any of the other four mechanisms), or that a lake may be at risk of shifting to another mechanism (e.g., redox mediated to perpetual P cycling). Throughout the flow chart, **check marks** indicate optional questions along a decision pathway to provide further evidence to support how the lake is categorized. For one question, a **triangle** is used to visualize the “yes” response, which indicates that lakes meeting this criterion may exhibit a range of bottom water dissolved oxygen conditions.

1.2.1 Common Carp Biomass Density Threshold

The first step in the diagnosis flow chart is to determine whether the water body has a high biomass density of common carp (*Cyprinus carpio*). Carp behavior results in physical disturbance to the sediment surface, resuspending sediments and disturbing macrophyte beds that stabilize sediments. When sediment is disturbed, loosely-bound P and P in pore water gets mixed up into the water column (see Part 1, Section 4.1 for further description of mechanisms). While other fishes can also resuspend sediments during their

feeding and reproductive activities, common carp most often reach densities at which this behavior has ecosystem-level consequences.

At densities of 80-100 kg ha⁻¹ of *C. carpio*, the foraging and spawning behavior of carp are most likely to affect P release on a whole-ecosystem scale. This threshold is based on studies evaluating the effect of changing common carp densities on vegetative cover in shallow lakes, suggesting that common carp populations threaten the ecological integrity of lakes when biomass density exceeds ~100 kg ha⁻¹ (Bajer et al. 2009). This density aligns with the existing IDNR threshold of 112 kg ha⁻¹ (100 lbs acre⁻¹) for recommending fishery renovation to reduce densities below 56 kg ha⁻¹ (50 lbs acre⁻¹). If densities exceed 100 kg ha⁻¹, the long-term mean concentration of involatile suspended solids should be calculated and compared to the breakpoint concentration to confirm carp-driven bioturbation as a dominant mechanism of internal loading (see Part 2 Section 1.2.3 for additional details). However, carp bioturbation is unlikely to be the sole mechanism of internal P loading in an Iowa lake or reservoir and the rest of the flow chart should be evaluated as well using the other monitoring data.

1.2.2 Thermal Stratification Occurrence

For waterbodies in which *C. carpio* are absent or present in low densities, the next step is to determine whether the waterbody thermally stratifies during the summer. Water column thermal structure is a powerful physical control on internal P loading. Stratification versus mixing patterns influence P transport from sediments into the water column as well as the chemical environment at the sediment water interface (e.g., dissolved oxygen concentrations), which may in turn affect sediment P release.

The decision flow chart distinguishes between waterbodies that almost never experience thermal stratification and waterbodies that stratify occasionally throughout the summer months (i.e., May through September; Figure 32). We used stratification frequency, or the proportion of observations in the long-term record when a thermocline was observed, to quantify differences in stratification patterns among different lakes. If the long-term stratification frequency for a given lake or reservoir is less than 10%, that waterbody should be classified as *not* thermally stratifying during the summer. The goal of this threshold is to isolate waterbodies in which the water column is continuously mixed throughout the open water season, with only ephemeral (i.e., sub-daily), weak thermal stratification developing under hot or quiescent conditions (Lewis 1983). We selected 10% as the threshold for stratification frequency because over the long-term record, this means that observing thermal stratification is rare, and not just seasonal (e.g., 33% of the time for a lake monitored through the Ambient Lake Monitoring program). Many of the state's shallow lakes have long-term stratification frequencies below this threshold including:

- | | | |
|------------------------|-----------------------|---------------------------|
| • North Twin Lake | • Lake Manawa | • Little Wall |
| • Clear Lake | • Ottumwa Lagoon | • Silver Lake (Palo Alto) |
| • Ventura Marsh | • Synder Bend Lake | • Little Spirit |
| • Trumbull Lake | • Storm Lake | • Browns |
| • Tuttle Lake | • Spring Lake | • Upper Gar |
| • Coralville Reservoir | • Black Hawk Lake | • Big Spirit |
| • Lost Island Lake | • Lower Gar Lake | • DeSoto Bend |
| • Lizard Lake | • Silver Lake (Worth) | |

However, high frequency measurements of summertime thermal structure are necessary to accurately identify continuous polymictic lakes (See Section 3.1.1 for recommendations). Alternatively, if temporally resolved water clarity and meteorological data are available, stratification dynamics could be studied using the equations from Krillin and Shatwell (2016) (Table 4). In summary, stratification frequency is a useful metric to determine if a lake is well-mixed (stratification frequency ≤ 10%) or demonstrably stratifies during the summer (stratification frequency > 10%).

One approach for calculating stratification frequency was to tally the number of instances when a thermocline was observed in the long-term record (IDNR 2022). However, the coarse temporal sampling resolution of current monitoring programs may not adequately capture stratification and mixing dynamics. For example, high frequency temperature data collected in Center Lake in 2018 revealed that the lake was stratified for 48% of the summer with an average duration of 8.2 days per stratification event (Figure 32). The three sampling dates for the Ambient Lake Monitoring program yielded a similar stratification frequency (33%), but the coarse sampling resolution might lead to the conclusion that stratification was limited to the early spring. Instead, stratification was intermittent during the summer, which could allow for redox sensitive P release followed by mixing events that move the P to the surface waters where it can stimulate blooms.

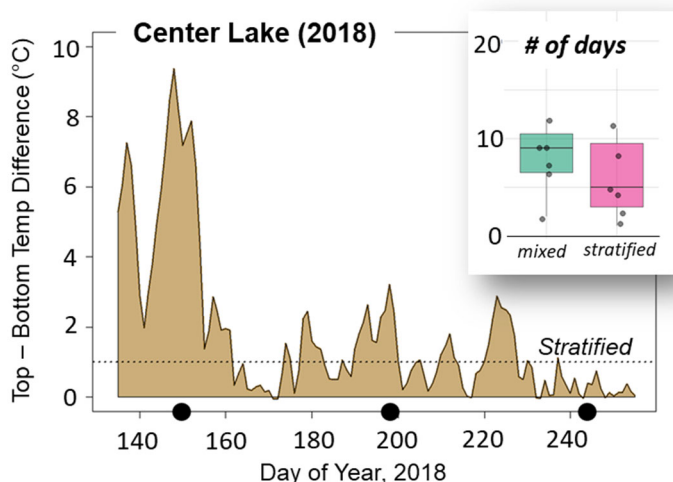


Figure 32. High frequency temperature data in Center Lake. The lake is considered stratified if the temperature difference between the top (0 m) and bottom (3.5 m) sensors is $>1^{\circ}\text{C}$. There were seven periods of stratification lasting a total of 58 days and seven periods of mixing lasting a total of 63 days, with a mean duration of 8.2 and 9 days, respectively. ALM sampling dates are denoted with a black circle. From the ALM data we'd conclude that the lake was stratified 33% of the time in early spring only instead of almost 50% of the time spread throughout the summer based on the high frequency data.

1.2.3 Suspended Solids Concentration Threshold

For waterbodies that do not thermally stratify and remain well-oxygenated during the summer, the next step in the diagnosis flow chart is to determine if there is frequent sediment disturbance not related to bioturbation. We used the mean annual concentration of involatile suspended solids (ISS) to infer the prevalence of sediment resuspension. Sediment disturbance is expected to increase suspended solids in the water column, especially inorganic or involatile particles. As such, higher annual mean ISS concentrations will be associated with more frequent and intense sediment resuspension events in shallow lakes. We explored the relationship between waterbody maximum depth and mean, annual ISS concentrations using a segmented regression. Through this analysis, we identified a statistically significant inflection point in the slope of the relationship at $4.9 \pm 1.1 \text{ mg L}^{-1}$ ISS (Figure 33). This break point corresponds to waterbodies that are between 5.6 and 6.0 meters maximum depth. If the mean, annual ISS concentration is greater than the 4.9 mg L^{-1} threshold, wind-resuspension is likely the dominant mechanism of internal P loading.

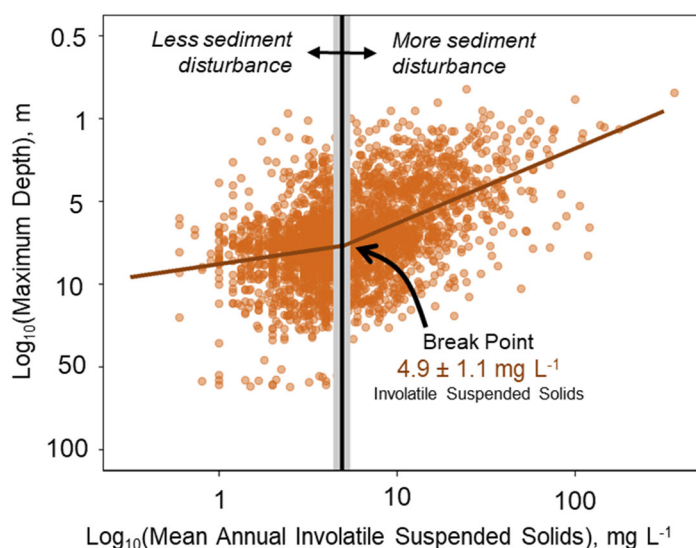


Figure 33. Segmented regression of the relationship between log-transformed annual mean involatile suspended solid (ISS) concentrations and log-transformed waterbody maximum depth (Adjusted- $R^2 = 0.22$). A statistically significant break point was identified at 4.9 mg L^{-1} ISS (standard error = 1.1 mg L^{-1}), which corresponds to a maximum depth of 5.86 meters and divides waterbodies experiencing more versus less sediment disturbance. Data Source: IDNR 2022.

If mean, annual ISS concentrations are below the $4.9 \pm 1.1 \text{ mg L}^{-1}$ ISS threshold, the waterbody might experience internal P loading due to **enhanced diffusion gradients** (or the data resolution may be insufficient to capture sediment resuspension events). Waterbodies with enhanced diffusion gradients under oxic conditions will experience frequent water column mixing; however, these mixing events might not disturb the sediments. For example, a shallow lake with extensive vegetation cover may not experience substantial sediment resuspension during a wind event due to the macrophyte beds dissipating turbulent mixing and stabilizing the sediments with their roots (Madsen et al. 2001; Vilas et al. 2017). Even if the sediments are not resuspended, the disturbance may still promote sediment P release through flushing, altering the concentration gradient of phosphate between sediments and the overlying water.

Whether a waterbody is classified as wind-resuspension or enhanced diffusion as the dominant mechanism of internal P loading may rely on the resolution of the suspended solids data. Infrequent sampling for suspended solids could miss sediment resuspension events, such that the ISS data would erroneously suggest that the waterbody does not experience sediment disturbance during mixing events. Additional sampling may be necessary to determine the prevalence of sediment resuspension in each waterbody, although the method and depth of sample collection are also considerations (Figure 34).

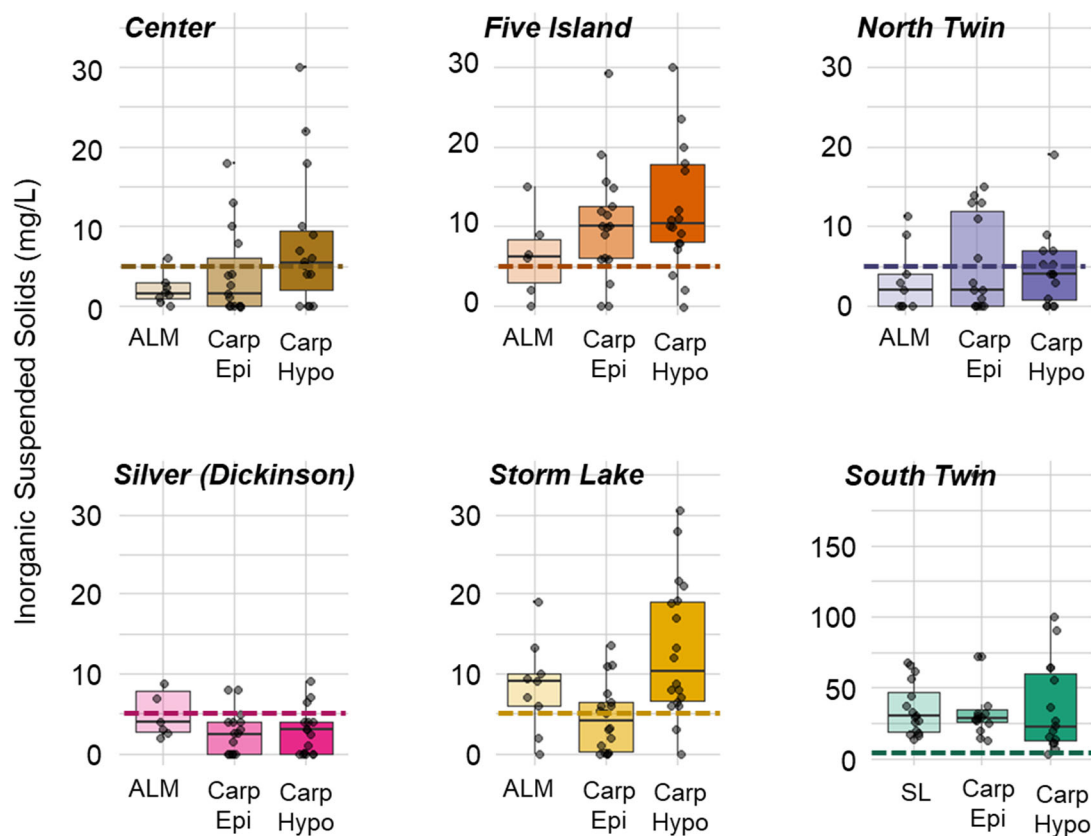


Figure 34. ISS concentrations from 2018-2020 for the study lakes sampled as a part of the Ambient Lake Monitoring Program (ALM) or Shallow Lakes Program (SL) as an integrated column sample to 2 m, the Carp Incentivized Harvest project (Carp) as a grab sample from 0.25 m (Epi) or 0.5 m above the sediments (Hypo). The ALM program sampled three times per summer whereas the Carp project sampled 6-7 times per summer. The horizontal dashed line is the break point ISS concentration (Figure 27). More frequent sampling moved the mean ISS concentration above the break point in Five Island Lake and below the break point in Storm Lake.

1.2.4 Bottom Water Dissolved Oxygen Depletion

Both long-term and seasonal patterns of bottom water dissolved oxygen concentrations are important metrics for categorizing the dominant mechanism of internal P loading in a waterbody. For example, dissolved oxygen data may provide additional evidence that the mechanism should be classified as either *wind resuspension* or *enhanced diffusion* if bottom water anoxia has not been observed in the long-term record. This pattern would provide further support that internal loading is primarily driven by sediment disturbance and water column mixing rather than P mobilization from redox-sensitive minerals. Conversely, instances of bottom water anoxia in the long-term record indicate that a waterbody may experience either *redox-mediated* internal loading under anoxic conditions or a *perpetual P cycle* fueled by both aerobic and anaerobic internal loading.

Although redox-mediated and perpetual P cycling waterbodies are primarily distinguished by mixing regime (section 1.2.5), seasonal dissolved oxygen dynamics may provide further evidence to specify the internal loading mechanism. Specifically, how rapidly dissolved oxygen drawdown occurs in the bottom waters following the onset of thermal stratification is expected to differ between these classifications. For

redox-mediated lakes, dissolved oxygen drawdown is expected to be gradual, perhaps resulting in anoxia after several weeks of stratification. Perpetual P cycling waterbodies are expected to be more productive, resulting in higher inputs of algal detritus to the sediments, resulting in high sediment oxygen demand. As a result, dissolved oxygen drawdowns are expected to occur rapidly during stratification periods (or at night, Figure 35) in perpetual cycling waterbodies.

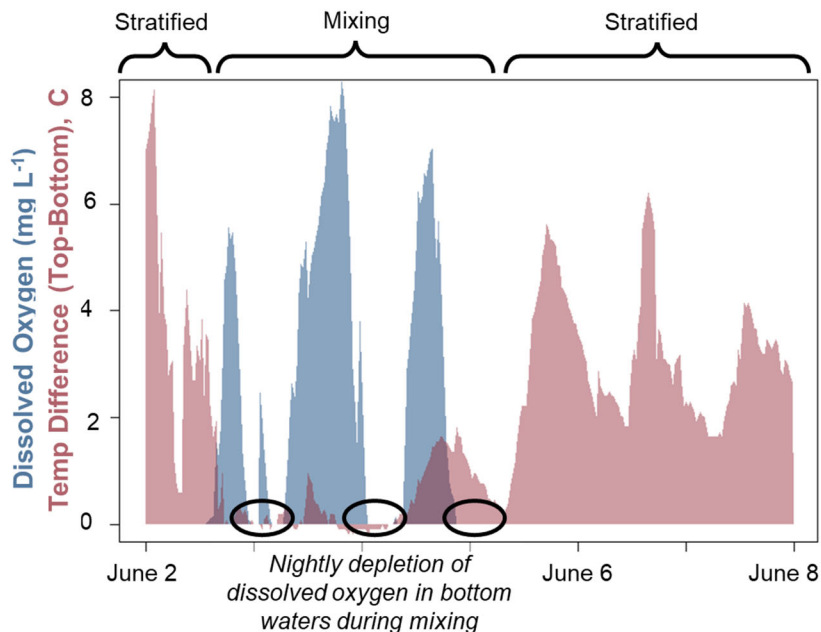


Figure 35. Dissolved oxygen (DO) concentrations in the bottom waters of Swan Lake (blue) and stratification dynamics as the difference in temperature between surface and bottom sensors (pink). On June 2, 2018 stratification broke down and the lake entered a three day period of mixing. Each day during mixing DO concentrations increased in the bottom waters due to primary production and were drawn down to zero at night due to high sediment oxygen demand.

1.2.5 Distinguishing Redox-Mediated Versus Perpetual P Cycle Waterbodies

Waterbodies that are dominated by *redox-mediated* internal loading are distinguished from those with a *perpetual P cycle* by mixing regime and the resultant effects on bottom water anoxia and P transport. Specifically, redox-mediated waterbodies will be dimictic and may develop bottom water anoxia during summer thermal stratification, which will mobilize P from redox-sensitive minerals. However, stratification will largely restrict transport of this mobilized P into the surface waters until fall mixing (except in the case of strong internal seiches), thus limiting how much of the internally loaded P is available for phytoplankton uptake. In contrast, waterbodies with perpetual P cycling will be polymictic with rapid development of anoxia during periods of thermal stratification. Mobilized P will be transported into the surface waters frequently during mixing events, so much of the internally loaded P will be available to phytoplankton.

Based on the strong relationship between maximum lake depth and stratification frequency, we propose that a long-term stratification frequency of ~80% is the threshold dividing dimictic and discontinuous polymictic waterbodies in Iowa. There was not as strong of a relationship between mean depth and stratification frequency (Figure 18). To identify this threshold, we fit the relationship between lake maximum depth and long-term stratification frequency with a segmented regression (Vito and Muggeo 2017), which identified a statistically significant inflection point in the slope of the relationship at $79.8 \pm 4.6\%$ stratification frequency (Figure 36). This break point corresponds to waterbodies that are between 6.5 and 7.9 meters deep. Based on the segmented regression, waterbody maximum depth explains nearly half of the variation in thermal stratification frequency (adjusted $R^2 = 0.42$).

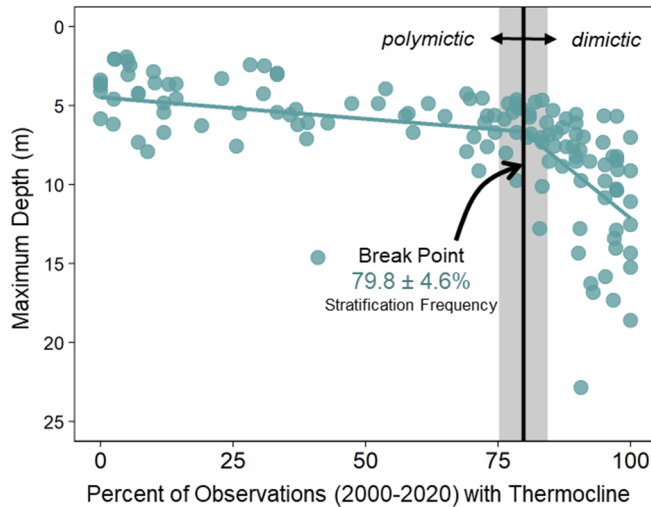


Figure 36. Segmented regression of the relationship between thermal stratification frequency and waterbody maximum depth ($\text{Adjusted-}R^2 = 0.42$). A statistically significant break point was identified at 79.8% stratification frequency (standard error = 4.6%), which corresponds to a maximum depth of 6.6 meters and divides more dimictic waterbodies from polymictic ecosystems. Data Source: IDNR 2022.

In addition to the empirical relationship between lake depth and stratification frequency in Iowa lakes, there are more generalized approaches available to distinguish between dimictic and polymictic waterbodies. For example, Kirillin and Shatwell (2016) present a generalized scaling parameter that may be used to infer the likely stratification regime for a given lake basin. This scaling parameter is a threshold of mean lake depth that divides dimictic and polymictic systems. The calculation considers lake basin length (i.e. fetch), Secchi depth as a proxy for water transparency, and local meteorological conditions, specifically mean windspeed and shortwave radiation. In order to demonstrate how Kirillin and Shatwell's calculation applies to dimictic versus polymictic waterbodies, we calculated the scaling parameter for two contrasting waterbodies in central Iowa: Ada Hayden Lake and Pierce Creek Pond. We used mean summer (May-August, 2020) wind speed, shortwave radiation, and Secchi depth to calculate the depth threshold of polymixis in each waterbody (Table 4).

Table 4. Analysis of mean depth threshold of polymixis in two contrasting waterbodies

Waterbody	Fetch (m)	Mean Depth (m)	Mean Secchi depth (m)	Critical Depth Threshold (m)	Interpretation
Ada Hayden Lake	810	9	1.93	1.449	Mean depth is greater than critical threshold, lake is dimictic
Pierce Creek Pond	1000	1.13	0.5	1.19	Mean depth less than critical threshold, lake is polymictic

This theoretical approach for estimating the mean depth threshold distinguishing dimictic and polymictic waterbodies provides opportunities to evaluate how changing meteorological conditions or water clarity, which are highly variable over time, might influence lake stratification. The critical depth threshold is expected to vary over time based on seasonal or interannual variation in water clarity and weather conditions. Ada Hayden Lake and Pierce Creek Pond are provided as examples of deep and shallow waterbodies, respectively, classified as dimictic and polymictic based on meteorological data for 2020 in central Iowa. For lakes of more intermediate mean depths, the critical depth threshold will likely vary around the mean depth of the waterbody depending on transparency and weather conditions, indicating periods of thermal stratification and mixing, i.e., a discontinuous polymictic waterbody.

Overall, empirical data of thermal stratification from the long-term record or more targeted monitoring (Appendix A4) are the best approaches for determining the mixing regime of a given waterbody at the location monitored. Spatially complex waterbodies may vary in mixing across the basin. Analyzing the long-term record for the average depth of the thermocline will reveal the typical mixing depth. Regions

shallower than this depth could be assumed to remain well-mixed whereas deeper regions experience at least intermittent thermal stratification. However, this assumption should be evaluated with spatially resolved sampling. For restoration planning, empirical temperature and weather data should be combined with hydrodynamic modeling (Appendix A5) to understand the stratification consequences of changing lake depth or water flow.

1.3 Examples of Using the Flowchart for Iowa Waterbodies

1.3.1 Little Wall Lake (Hamilton County, Iowa) – Wind Resuspension

Little Wall Lake is a shallow natural lake (max depth = 4 m) located in central Iowa. The dominant fishes in the lake are channel catfish, white crappie, black crappie, bluegill, and yellow bass. Importantly, there is no record of common carp in Little Wall Lake. From the Ambient Lake Monitoring data, the lake is rarely thermally stratified with stratification observed for only 7% of the sampling events in the long-term record. Based on the bottom water dissolved oxygen concentrations (50-80% saturation) and the low frequency of stratification, this lake is well oxygenated in the bottom waters. The long-term mean concentration of involatile suspended solids is $17 \pm 19 \text{ mg L}^{-1}$. Using the flowchart, we hypothesize that internal loading in Little Wall Lake is dominated by wind-driven sediment resuspension (Figure 37).

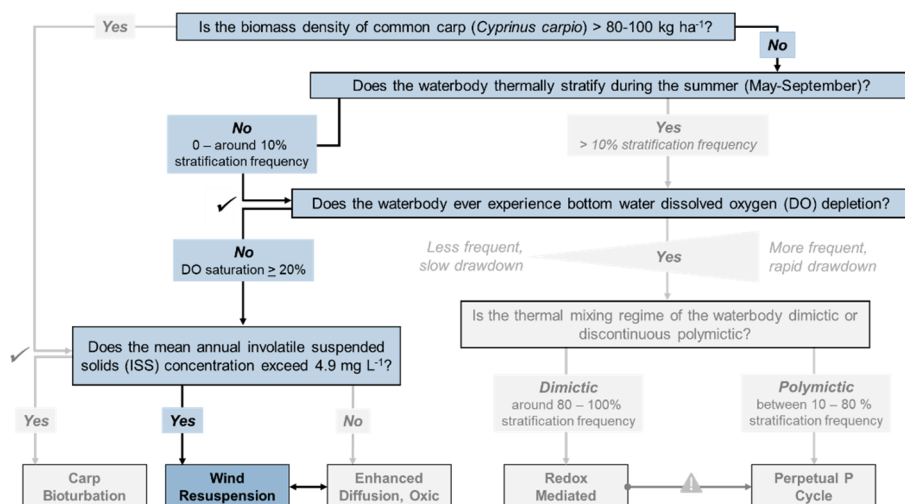


Figure 37. The pathways in the flowchart (Figure 25) highlighted for Little Wall Lake leading to the hypothesis that internal loading is dominated by wind resuspension in this lake.

1.3.2 Lizard Lake (Pocahontas County, Iowa) – Enhanced Diffusion

Lizard Lake underwent an extensive restoration from 2011-2014, which included extirpation of the common carp population. Lizard Lake does not thermally stratify and maintains an oxic environment in the bottom waters of the lake. Following the restoration, soluble reactive P concentrations increased substantially, peaking at $190 \mu\text{g L}^{-1}$ in 2014. Following the restoration, involatile suspended solid concentrations were $2.3 \pm 2.3 \text{ mg L}^{-1}$, which is below the threshold for wind resuspension as the dominant mechanism. Based on the flowchart, we hypothesize that the sharp increase in P concentrations in Lizard Lake was facilitated by enhanced diffusion from the sediments following restoration (Figure 38).

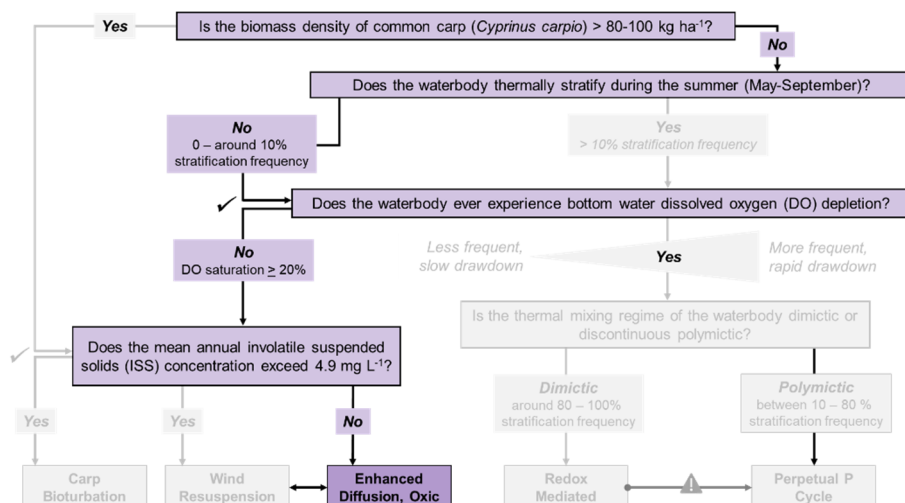


Figure 38. The pathways in the flowchart (Figure 25) highlighted for Lizard Lake after restoration activities, leading to the hypothesis that internal loading is dominated by enhanced diffusion in this lake.

1.3.3 Ada Hayden Reservoir (Story County, Iowa) – Redox Mediated

Ada Hayden reservoir is a deep, constructed waterbody with a small population of common carp. The reservoir thermally stratifies on an annual basis (100% of observations with stratification) and the bottom water dissolved oxygen depletes every summer (1-10% saturation). Given the deep morphometry relative to the surface area and the consistency of stratification, the mixing regime is dimictic. Additionally, the involatile suspended solids concentration is $3.1 \pm 1.2 \text{ mg L}^{-1}$, below the threshold for determining internal loading dominance by wind resuspension. Using the flowchart, we hypothesize that internal loading in Ada Hayden is dominated by redox processes (Figure 39).

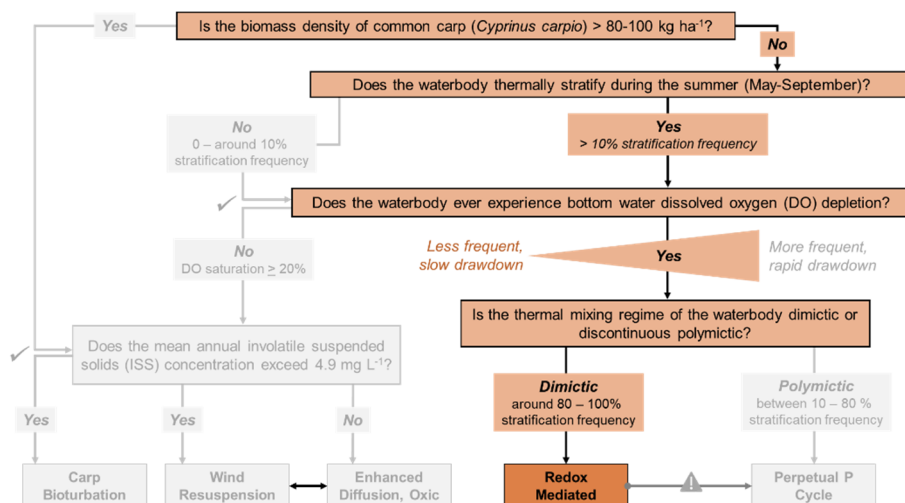


Figure 39. The pathways in the flowchart (Figure 25) highlighted for Ada Hayden Reservoir leading to the hypothesis that internal loading is dominated by redox processes in this lake.

1.3.4 Swan Lake (Carroll County, Iowa) – Perpetual Cycling

Swan Lake is a shallow, hypereutrophic, polymictic waterbody with a small common carp population. The lake stratifies annually, but intermittently throughout the summer (Ortiz and Wilkinson 2021), with only 16% of observations in the long-term record indicating stratification. When stratified, the bottom waters of Swan Lake are often depleted in dissolved oxygen. Involatile suspended solids concentrations

are $16.4 \pm 27 \text{ mg L}^{-1}$. We hypothesize that internal loading in Swan Lake falls into the perpetual cycling process category with contributions from the wind resuspension mechanism (Figure 40).

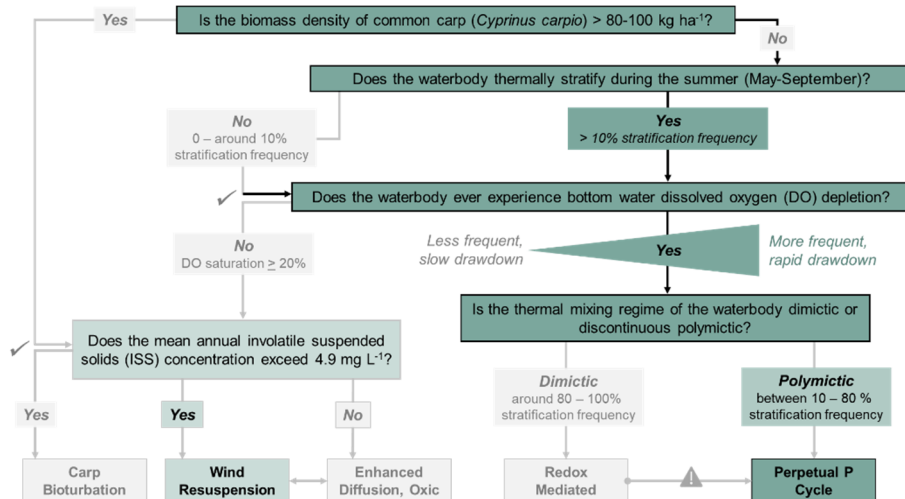


Figure 40. The pathways in the flowchart (Figure 25) highlighted for Swan Lake leading to the hypothesis that internal loading is in the perpetual cycling process, including contributions from wind resuspension.

1.4 Comparison to Previous Classifications of Iowa Waterbodies

The Filstrup and Downing (2016) Capstone Project sought to identify Iowa lakes and reservoirs with high internal P loads and classify the responsible mechanism as either wind resuspension or bottom water anoxia. The authors used statistical modeling approaches to analyze drivers of thermal stability, anoxia frequency and severity, wind-driven resuspension, internal P loading, and “rough fish” influence on internal loading in waterbodies included in either the Ambient Lakes or Shallow Lakes Monitoring Programs. The models for thermal stability and anoxia used available monitoring data and lake morphometric features. However, much of the necessary input data for both response and explanatory variables were unavailable to model wind-driven resuspension and internal P loading. As a result, the authors were forced to make simplified assumptions and estimate many key variables. For example, internal P loads were estimated based on the difference in observed total P in the waterbody and the predicted total P concentration based on annual precipitation, published P export by land cover classes, and an empirical model developed by Canfield and Bachman (1981). This resulted in a coarse estimation of internal P loads with high uncertainty, thus it is difficult to evaluate the validity of the assumptions underlying the estimation approach. As stated by these authors, due to the overfitting and high model error, the resultant models for internal P loading in Iowa waterbodies were not robust.

The diagnosis flow chart and supporting analyses presented here build upon findings included in the 2016 Capstone Project; however, the flow chart takes a more flexible and data-driven approach to classifying internal P loading mechanisms on a lake-by-lake basis. Several questions within the flow chart were informed by Filstrup and Downing’s findings regarding the importance of thermal stratification frequency, hypoxia frequency and severity, and bottom shear stress as drivers of internal P loading. Questions about these waterbody characteristics are used to guide the user towards the most likely dominant mechanism of internal P loading. However, the data and any necessary assumptions that are used to answer each question are left to the user’s discretion. For example, we recommend that long-term monitoring data (i.e., IDNR 2022) be used to answer the flow chart questions and thus develop hypotheses regarding the dominant mechanism of internal loading for a given waterbody but that those hypotheses should then be tested with targeted sampling before any restoration or management action is

taken. However, a user might decide that evidence from the long-term record is compelling enough for a given question that further sampling is not necessary. In this way, the flow chart is designed to meet the user's goals, work with available data, and provide justification for when further data collection is needed. Additionally, the flexibility of the flow chart acknowledges and incorporates the dynamic nature of lake P cycles over time. Rather than providing a static characterization of each Iowa waterbody, the flow chart allows the user to explore how dominant internal loading mechanisms may change over time because of restoration activities, species introductions, and climate change.

Another key difference between the diagnosis flow chart and the Filstrup and Downing (2016) analyses is the consideration of internal P loading under oxic conditions, as included in the “enhanced diffusion, oxic” and “perpetual P cycle” mechanism categories. The importance of aerobic internal P loading was still an emerging idea in the literature at the time of the Filstrup and Downing (2016) analyses. However, at this point there is clear theory to support aerobic sediment P release (Hupfer and Lewandowski 2008; Orihel et al. 2017) as well as evidence of internal loading under both oxic and anoxic conditions, especially in productive waterbodies (Song and Burgin 2017; Albright and Wilkinson 2022b). The diagnosis flow chart considers aerobic sediment P release processes and guides the user to identify if oxic internal loading may occur in a waterbody.

The Filstrup and Downing (2016) report includes a brief analysis of the influence of “rough fish” on internal P loading; however, the results from the statistical model were inconclusive. The authors include a large and diverse group of fishes in their calculation of rough fish biomass: big- and smallmouth buffalo; black, brown, and yellow bullhead; bighead, common, and silver carp; white amur (i.e., grass carp); highfin, quillback, and river carpsucker, long- and shortnose gar, hornyhead chub, shorthead redhorse, and white sucker. These species include benthivores, planktivores, and piscivores as well as both native and introduced species. Some of the fish species included in this analysis are expected to have a strong effect on internal P loading (e.g., common carp; Huser et al. 2016). However, many of these species are native to Iowa, have beneficial ecological roles, and no known links to internal P loading (Scarnecchia 1992; Rypel et al. 2021). As such, it is counterproductive to group all non-game fishes into an analysis of fish influence on internal P loading. For the diagnosis flow chart, we have focused instead on common carp (*C. carpio*) because it is an introduced species with a demonstrated, mechanistic link to internal P loading via sediment disturbance during foraging (Bajer and Sorensen 2015; Huser et al. 2016).

Overall, the flow chart is more adaptable and transparent than the 2016 Capstone results. This tool allows for greater user flexibility in the data and assumptions used to build lines of evidence to identify mechanisms of internal P loading for a given waterbody.

SECTION 2 – MANAGEMENT RECOMMENDATIONS

Management recommendations for each internal loading mechanism identified in the flowchart (Figure 31) are detailed below. We discuss additional benefits beyond internal loading reduction along with considerations for mitigating undesirable side effects of the management intervention. As with many of our management recommendations in this report, **thresholds should be treated as guidelines, not guarantees**. When it comes to strategies such as dredging to increase water depth thereby reducing sediment resuspension, there is not a target depth that can be universally applied. This is because the depth at which sediment resuspension may decrease is a function of fetch whereas the depth to induce seasonal stratification and avoid inducing redox-sensitive or perpetual P-loading through discontinuous stratification is a function of fetch, basin morphometry, meteorology, and water clarity. These two depths are not likely to be the same and the potential for trading one P-loading mechanism for another is high without careful hydrodynamic modeling (Appendix A5) in the planning phase of restorations. This modeling should also consider future climate scenarios to evaluate if planned restoration investments are robust to rapidly changing environmental extremes.

The US National Park Service and other federal agencies have recently adopted the “Resist, Direct, Accept” (RAD) framework for resource conservation decision making. The environment is changing rapidly and beyond the bounds of historic variability (Lynch et al. 2021). The future of ecosystems is unlikely to reflect the past. Managers can choose to respond to the trajectory of a changing ecosystem by:

*“...**resisting** (working to maintain or restore based upon historical or acceptable current ecosystem conditions), **accepting** (allowing an ecosystem to change without intervening), or **directing** (actively shaping ecosystem change toward preferred new conditions).”*

- National Park Service, <https://www.nps.gov/subjects/climatechange/resistacceptdirect.htm>

The internal P loading management strategies detailed below were largely developed and studied by scientists as ways to resist internal P loading. However, the utility of these strategies must be considered in the context of a rapidly changing landscape and climate, and the option to direct or accept the internal P loading trajectory for Iowa’s lakes and reservoirs may be the most prudent.

Another critical consideration in selecting a management strategy is cost, both in terms of initial investment based on the magnitude of the treatment and the longevity of the treatment. The magnitude of the treatment will vary with lake characteristics (e.g., shoreline length, water depth, watershed features) and the severity of the internal loading problem (e.g., treatment targeted at prevention vs. remediation). While evaluating costs and making specific recommendations regarding application rates or installation extent are beyond the scope of this report, we have attempted to provide information on the suite of options available to address internal P loading, feasibility for Iowa’s surface waters, additional benefits that may be realized with implementation, and a discussion of potential risks and known side effects to consider. For all internal loading management, we strongly recommend additional data collection that is spatially and temporally resolved and hydrodynamic modeling of the current conditions, future climate, and proposed changes before committing to a management plan.

2.1 Carp Bioturbation

Table 5. Management strategies to reduce internal loading from carp bioturbation.

Management Action	Internal P Loading Mechanisms Targeted	Additional Benefits	Mitigating Side Effects
Reduce carp biomass density (targeted harvest, renovation) ¹	Reduce sediment P release due to resuspension from carp bioturbation ³	Improved fishery ⁴ , more stable food web ⁴ , increased water clarity through reduced non-algal turbidity ⁵	Regulations on bycatch to protect native species; restocking of game fish
Barriers to limit dispersal and access to spawning wetlands ²			Seasonal barrier placement based on phenology, design barrier spacing to reduce effects on native fish movements ²
Assisted macrophyte re-establishment – planting macrophytes or altering conditions to stimulate existing seedbanks ⁶	Reduce sediment resuspension due to root stabilization ⁷ , reduce shoreline erosion ⁸ , P sink in emergent litter ⁹	Fish habitat ¹⁰ , waterfowl forage and habitat ¹¹	Caution with re-seeding or moving macrophytes to avoid non-native introductions. Pair with invasive species inspections and education.

¹Bajer and Sorensen 2015, ²Piczak et al. 2022, ³Huser et al. 2016, ⁴Weber and Brown 2009, ⁵Parkos et al. 2003, ⁶Lathrop et al. 2013, ⁷Lin and Wu 2013, ⁸Madsen et al. 2001, ⁹Davis and van der Valk 1978, ¹⁰Randall et al. 1996, ¹¹Hansson et al. 2010

Strategies for reducing internal loading induced by carp bioturbation focus on reducing carp biomass and counteracting sediment disturbances caused by carp. Lower carp density will result in less sediment resuspension and accompanying P release caused by carp behavior. Some empirical studies of shallow lakes have suggested that carp biomass above 100 kg ha⁻¹ threatens ecological integrity of the ecosystem (Bajer and Sorensen 2015), so reduction should be targeted well below this threshold (e.g., less than 50 kg ha⁻¹). However, as the incentivized harvest experiment illustrated (Table S4), estimating carp biomass accurately is difficult and seemingly substantial reductions in density may have little immediate influence on water quality or the standing stock of carp biomass.

Reducing carp biomass can be achieved directly through targeted harvesting programs and chemical renovations, or paired with more indirect measures of barrier installation that prevent additional carp from entering the waterbody and movement to spawning locations (Bajer and Sorensen 2015, Piczak et al. 2022). In addition to reduced P loading, reduced carp biomass can result in improved fishery production, more stable food webs, and increased water clarity (Weber and Brown 2009, Parkos et al. 2003). While harvesting and chemical renovations can also decrease other fish populations in the ecosystem, this can be mitigated through implementing regulations on bycatch or restocking with native fishes following renovation. Carp barriers can interfere with the movement of native fishes, but can be designed to allow passage of smaller, desired species or implemented seasonally based on reproductive phenology to reduce harm to native species (Piczak et al. 2022).

Native fishes that are not targeted for sport (i.e., non-game species) are an important part of Iowa's inland water ecosystems. For example, these fishes provide important regulating services influencing water clarity through their support of freshwater mussel reproduction. The larvae of freshwater mussels are obligate ectoparasites that evolved with native fish hosts. They also contribute to nutrient storage, transformation, and exert top-down influence on phytoplankton biomass. Additionally, some native species such as the bigmouth buffalo (*Ictiobus cyprinellus*) are a culturally important species for Native peoples in the Upper Midwest. In Iowa, the term “rough fish” is used to classify these species in a management context along with other non-native fishes such as common carp. When targeting common

carp for biomass reduction, care should be taken to protect native species during management. For example, instituting and enforcing bag limits on native fishes would protect populations of these long-lived, exploited species (Rypel et al. 2021). Restoring habitat for native and game fishes is also a strategy for enhancing both groups of species. Recent whole-lake experiments demonstrated that ecosystem-based management of targeted fish populations (e.g., game species) outperforms stocking-focused approaches for enhancing fish populations (Radinger et al. 2023).

Outside of biomass reduction, the effects of carp bioturbation can also be mitigated through the establishment of macrophyte beds; however, if carp populations are allowed to grow unchecked macrophyte establishment is unlikely. Strategies for establishment include planting of native macrophytes and placing enclosures to prevent carp from disturbing budding and existing macrophyte beds (Lathrop et al. 2013, Part 2 Section 2.1.2 below). Once established, these beds reduce sediment resuspension through root stabilization (Lin and Wu 2013) and shoreline erosion by reducing wave energy (Madsen et al. 2001), while their leaf litter acts as a P sink (Davis and van der Valk 1978). Macrophyte beds are also habitat and forage space for fish and waterfowl (Randall et al. 1996, Hansson et al. 2010). When establishing new macrophyte beds, care should be taken to avoid introducing non-native species.

2.1.1 Case Study: Lost Island Lake (Palo Alto County, IA, USA)

Since 2008, managers of Lost Island Lake have been implementing an innovative plan to reduce carp biomass within the lake. The restoration began with the goal to remove 75% of carp biomass, restocking the lake with predatory fish, and the construction of five fish barriers. The lake was initially dewatered to allow for removal of most carp and create favorable conditions for macrophyte reestablishment. Since 2008, over 1.27 million pounds of carp and bigmouth buffalo have been removed from the lake, reducing density from around 400 lb acre⁻¹ to 55 lb acre⁻¹. Since restoration began, the lake has seen increased Secchi depth and decreased nutrient concentrations and turbidity compared to before the measures were implemented. Overall, Lost Island Lake is an example of effective reduction in internal P loading via carp removal and implementation of carp barriers (IDNR Report 2021).

2.1.2 Case Study: Lake Wingra (Madison, WI, USA)

Lake Wingra is a shallow, eutrophic lake with an introduced carp population originating in the 1800s. In an effort to improve water quality in the 2000s, two studies were initiated to investigate the feasibility and efficacy of carp removal. A carp exclosure was installed in the lake in summer 2005 – 2008, demonstrating how water clarity and macrophyte populations would respond to carp removal. The results showed a dramatic increase in water clarity, as well as increased density and depth of macrophytes. Additionally, a radio telemetry study of carp was conducted to identify areas for seining removal. After the success of these two studies, carp removal in the lake was carried out via under-ice seining in March 2008. In the five years following removal, mean summer Secchi depths increased by approximately 40% (0.53 m) and total P concentrations decreased by approximately 40% (23 µg L⁻¹) when compared with the twelve years prior to restoration (Lathrop et al. 2013).

2.2 Wind Resuspension and Enhanced Diffusion Gradients

Table 6. Management strategies to reduce wind resuspension of sediments.

Management Action	Internal P Loading Mechanisms Targeted	Additional Benefits	Mitigating Side Effects
Assisted macrophyte re-establishment – planting macrophytes or altering conditions to stimulate existing seedbanks ¹	Reduce sediment resuspension due to root stabilization ² , reduce shoreline erosion ³ , P sink in emergent vegetation litter ⁴	Fish habitat ⁵ , waterfowl forage and habitat ⁶	Caution with re-seeding or moving macrophytes to avoid non-native introductions. Pair with invasive species inspections and education.
Shoreline stabilization with armoring, riparian and emergent plants ⁷	Reduce shoreline erosion and nearshore resuspension ^{8,9}	Property value ¹⁰ , fish and wildlife habitat ⁹ , increased water clarity ¹¹	'Hard' armoring structures can erode sediment behind and underneath due to redirected wave energy ¹² , initial installation can result in increased sediment loss ⁹
Planted or constructed windbreaks ⁸			
Dredging to increase lake depth ¹³	Reduce frequency and severity of sediment resuspension and associated P release in dredged areas ¹³	Increased water clarity though reduced non-algal turbidity	High risk of inducing thermal stratification ¹⁴ and therefore other mechanisms of internal P loading
Edge planting of trees or tall shrubs to provide wind sheltering	Reduce resuspension rates ¹⁵ through wind speed reduction ¹⁶	Increased riparian zone	[no side effects identified]

¹Lathrop et al. 2013, ²Lin and Wu 2013, ³Madsen et al. 2001, ⁴Davis and van der Valk 1978, ⁵Randall et al. 1996, ⁶Hansson et al. 2010, ⁷Palinkas et al. 2017, ⁸Severson et al. 2009, ⁹Engel and Pederson 1998, ¹⁰O'Connell 2010, ¹¹Gabriel and Bodensteiner 2011, ¹²Griggs 2005, ¹³Yang et al. 2023, ¹⁴Holgerson et al. 2022, ¹⁵Carper and Bachmann 1984, ¹⁶Markfort et al. 2010

Strategies that target reduction of internal loading caused by wind resuspension and enhanced diffusion gradients focus on limiting sediment resuspension and decreasing wind shear. One strategy for reducing sediment resuspension is the assisted re-establishment of macrophytes (Lathrop et al. 2013). As discussed in Part 2 Section 2.1, planting and promotion of existing native macrophytes reduces sediment resuspension and shoreline erosion while providing additional ecosystem services (Lin and Wu 2013, Madsen et al. 2001). Another way to reduce nearshore resuspension and erosion is the implementation of shoreline stabilization and windbreaks. Shoreline stabilization can take the form of armoring with riprap or planting of riparian and emergent vegetation (Palinkas et al. 2017). This strategy aims to reduce the effect of wave action as it hits the shoreline, reducing sediment loss from banks (Engel and Pederson 1998). Windbreaks can take the form of strategically-placed riprap structures and planted shallow regions within the water body (Severson et al. 2009). These structures aim to reduce wave action before they reach the shore. Both methods can increase property value by protecting land from erosion (O'Connell 2010). Natural plantings can also be used as increased fish and wildlife habitat (Engel and Pederson 1998). Artificial, 'hard' features have the drawback of enhanced erosion from behind and underneath the structure, lowering the effectiveness and lifetime of these installations (Griggs 2005).

Installation of hard structures or plantings can be targeted based on prevailing wind directions and water depth. Hard structures may be best suited for small lengths of shoreline than experience high erosion in the early spring before plants have emerged. Care should be taken to avoid sediment loss and suspension during installation of these structures (Engel and Pederson 1998), and hard shoreline armoring has the potential to move the zone of erosion down current of installation.

While the strategies above focus on limiting resuspension in the shoreline regions of the waterbody, there are other approaches to limit resuspension in the lake's center. One approach is to dredge the lake to increase depth (Yang et al. 2023). With an increased lake depth, wave energy from the surface is less likely to interact with the sediments, reducing the frequency and severity of resuspension and associated P release (Yang et al. 2023). This method can also improve water clarity by reducing non-algal turbidity within the water column. ***However, dredging is a very expensive method for addressing internal P loading and there are few instances of long-term restoration success using this method.***

In addition to the expense, dredging should be approached with extreme caution as there is a risk of inducing intermittent thermal stratification (Holgerson et al. 2022) and triggering other mechanisms of internal loading. A temporally dynamic calculation of the critical mean depth of polymixis vs. dimixis (Krillin and Shatwell 2016), analysis of mean thermocline depth in the long-term record, and more detailed hydrodynamic modeling (Appendix A5) are needed to test the effects of proposed dredging depths on thermal stratification frequency. Thermal stratification is a function of basin morphometry, fetch, meteorological conditions, and water clarity. Therefore, a 'safe' depth for dredging to avoid intermittent stratification cannot be universally prescribed and must be studied on a lake-by-lake basis. In our view, this is a small investment in data collection and modeling compared to the much larger potential investment and risk of dredging and inadvertently triggering redox-sensitive or perpetual P cycling.

Another method of reducing wave energy throughout the water body is by planting native trees or shrubs along the shoreline to provide shelter. Water bodies with greater sheltering from vegetation experience reduced resuspension rates (Carper and Bachmann 1984) through reducing the wind speed above the lake (Markfort et al. 2010). This approach can also provide an increase in riparian zone, potentially reducing external loading of nutrients and sediment to the lake. As such, we recommend this strategy over hard armouring which has little additional benefit for the ecosystem.

2.2.1 Case Study: Lizard Lake (Pocahontas County, Iowa, USA)

Lizard Lake underwent a series of restoration activities in winter 2011 and 2012. The lake was drained to allow for the consolidation of loose sediments, promote macrophyte growth, and remove the high population of common carp. In addition, a water control structure and fish barrier were installed. Prior to restoration, Lizard Lake was one of the most eutrophic waterbodies in the state. The promotion of aquatic vegetation during the restoration aimed to reduce resuspension of sediments, reduce wave energy within the water column, sequester nutrients that could otherwise fuel algal blooms, and provide habitat for zooplankton which act as a top-down control on phytoplankton growth. Following the restoration, water quality has significantly improved in this water body. Secchi depth increased from an average of 0.13 m prior to restoration activities to 0.98 m post-restoration. Total P concentrations decreased from an average of 320 $\mu\text{g L}^{-1}$ to 120 $\mu\text{g L}^{-1}$ (IDNR Report 2021, IDNR 2022). However, the high total P concentrations that remain post-restoration are driven by enhanced diffusion from P-rich sediments (Figure 38).

2.2.2 Case Study: Kincaid Lake (Jackson County, Illinois, USA)

Kincaid Lake is a 1113 ha reservoir which regularly experiences large waves during windy weather. Approximately 20% of the annual sediment input in this lake came from shoreline erosion. To prevent this, managers installed riprap breakwater structures in nearshore and offshore environments to protect the most eroded shorelines. Following the implementation of these breakwaters, average wave height was reduced by 90%. Erosion in protected regions averaged 0.32 cm month^{-1} , whereas unprotected regions had average rates of 2.1 cm month^{-1} . Bank slopes also decreased following the installation of the structures. Finally, a greater diversity of aquatic plants thrived in the protected regions ($n = 112$ species) compared to unprotected regions ($n = 12$ species). Overall, the installation of protective structures was

able to greatly decrease the amount of erosion and likely corresponding P availability in this reservoir (Severson et al 2009).

2.3 Redox-Mediated Internal P Loading

Table 7. Management strategies to reduce redox-mediated internal loading

Management Action	Internal P Loading Mechanisms Targeted	Additional Benefits	Mitigating Side Effects
Dredging to deepen water column and remove redox-sensitive P ¹	Move lake from polymictic to dimictic regime, limiting P transport to surface waters during mixing, remove P-rich surface sediment layer ¹	Increases reservoir lifetime – flood control capacity, removes any additional contaminants in sediments ¹	Caution against dredging waterbodies with complex morphologies; mixing regime needs to be carefully modeled; dredge spoil must leave watershed or be well-contained
ALUM treatment (note, dosage is important for success) ²	Increase redox-insensitive P sorption capacity in the sediment ²	Does not require draining & dredging, still potentially some benefit in trapping P diffusion under oxic conditions	Works best with low sedimentation rates ³ and limited sediment disturbance ⁴ . Best for small lakes with low macrophyte coverage ² . Can result in release of Al to overlying water ⁵ and changes in biogeochemical cycling and benthic community dynamics ⁶ .
Bottom water withdrawals ⁷ (most effective in small surface area, deep lakes ¹)	Flush P-rich bottom waters out of the lake to limit transport to surface waters ¹	[none identified]	Only effective in reservoirs with existing infrastructure for bottom water release ¹ . Relocates nutrients downstream ⁷ . Can result in water column warming and destabilization of thermal regime, critical to avoid extreme water temperature changes ⁷ .
Aeration ⁸	Maintain a thin layer of oxic sediments to increase redox-sensitive sediment P sorption capacity ¹	Maintains oxic habitat for aerobic organisms during periods of aeration ⁸ , decreases stress from diel variability in oxygen concentration	Careful injection of pure O ₂ can yield successes in strongly stratified waterbodies ⁹ , but standard aeration systems can stimulate sediment respiration and disturbance, releasing additional P ¹⁰

¹Bormans et al. 2016, ²Welch and Cooke 1999, ³Lewandowski et al. 2003, ⁴Huser et al. 2016, ⁵Reitzel et al. 2013, ⁶Nogaro et al. 2013, ⁷Nurnberg 2007, ⁸Beutel and Horne 1999, ⁹Gerling et al. 2014, ¹⁰Horppila et al. 2017

Strategies for reducing redox-mediated internal P loading focus on treatment of the sediments and bottom waters. One method is to dredge the lake to deepen the water column (Bormans et al. 2016); however, this strategy is expensive and not guaranteed to be successful for Iowa lakes as it may induce intermittent stratification. See Part 2 Section 2.2 and introduction to Part 2 Section 2 for a detailed discussion of dredging risks. While dredging does not reduce P release itself, the goal is to trap P-rich water in a strongly stratified bottom layer during the summer months, decreasing the fertilization of surface algal blooms with periodic mixing. Detailed hydrodynamic modeling (see Part 2 Section 3.3, Appendix A5) and high frequency data collection (Appendix A4) at multiple locations in the waterbody are needed to

ensure that strong and lasting stratification is achievable under current and future climate scenarios, otherwise dredging is likely to make the internal loading problem worse. Inducing intermittent stratification in a nutrient-rich lake is likely to tip the ecosystem into the perpetual cycling state. Dredge spoil should be removed from the watershed or well contained to limit the risk of it reentering the waterbody.

Another sediment-focused approach is alum treatment. Aluminum compounds increase the sediment sorption capacity, reducing the sensitivity to anoxic P release via reduction (Welch and Cooke 1999). This approach reduces the amount of P reaching the water column directly, and usually does not require equipment installation, dredging, or draining of the waterbody. This strategy works best in lakes with low sedimentation rates (Lewandowski et al. 2003) and limited sediment disturbance (Huser et al. 2016) to avoid burial of the active alum layer. These are not conditions that are likely to be met in most Iowa waters (Downing et al. 2008). Macrophyte presence in a lake can reduce the effectiveness, as the plant material interferes with the alum's ability to form a capping layer (Welch and Cooke 1999). Success of this method is dependent on proper dosing which is guided by sediment P composition, hydrology, and water chemistry (Welch and Cooke 1999). Dosing should be decided carefully, as alum can result in release of aluminum to the overlying water column (Reitzel et al. 2013) and disruptions to other ecosystem functions such as biogeochemical cycling and benthic organisms (Nogaro et al. 2013).

Bottom water withdrawals are another method of managing redox-mediated internal loading (Nurnberg 2007). This strategy flushes P-rich bottom waters out of dimictic lakes, limiting P transport to the overlying surface water and reducing algal growth (Bormans et al. 2016). This action is particularly effective to implement in deep reservoirs with small surface areas where existing infrastructure for bottom water release is already installed (Bormans et al. 2016). Outside of these conditions, bottom water release is unlikely to be a cost-effective strategy and the conditions are not common for Iowa waters. If used, care should be taken to avoid extreme changes in water temperature, as water level and temperature changes resulting from bottom water removal can result in destabilization of the lake's thermal structure (Nurnberg 2007). This strategy can also be viewed as simply moving the nutrient pollution downstream, which presents an issue of environmental justice for downstream communities. This effect could be mitigated by water treatment before release (Nurnberg 2007).

Aeration of the water column is another method of controlling redox-mediated P release (Beutel and Horne 1999). This strategy aims to maintain a thin oxic layer at the sediment-water interface to increase sorption capacity of redox-sensitive P (Bormans et al. 2016). Aeration systems not only provide oxygen to the surface sediment, but also the water column, allowing for maintenance of oxic summertime habitat for organisms in the lake (Beutel and Horne 1999). However, standard aeration systems can stimulate sediment respiration and disturbance (Horppila et al. 2017), releasing more P and decreasing overall oxygen availability. For example, a mass balance of dissolved oxygen in Swan Lake (Carroll County, Iowa) in 2017 revealed that the mass of oxygen in the lake exceeded the expected mass based on equilibration with the atmosphere when the aeration system was turned off (photosynthesis rates were greater than respiration). However, when the aeration system was turned on, the mass of dissolved oxygen was less than expected likely due to stimulated respiration (Figure 41). Careful, continuous injection of pure O₂ to the hypolimnion of a strongly stratified reservoir in Virginia was successful in controlling P release from sediments (Gerling et al. 2014), but this strategy requires the infrastructure and proper hydrodynamic conditions (i.e., stratification) to make implementation worthwhile.

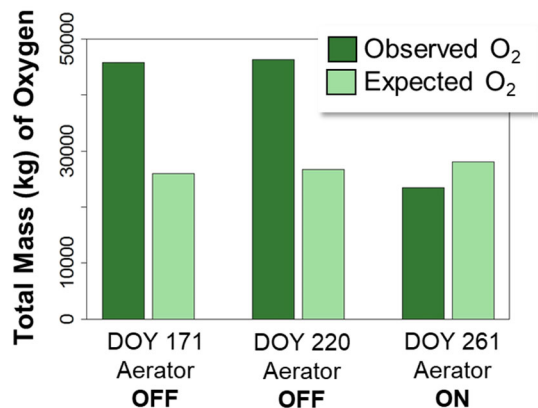


Figure 41. Mass of oxygen in Swan Lake on three dates calculated from the morphometry of the lake and profiles of dissolved oxygen concentration taken at the deepest point. When the aeration system was turned on, the mass of oxygen in the water column was less than expected based on equilibration with the atmosphere. This was likely due to the aerator stimulating aerobic respiration in the organic matter-rich sediments. Analysis provided courtesy of David Ortiz.

2.3.1 Case Study: Carter Lake (Pottawattamie County, Iowa, USA)

Carter Lake has historically had low water clarity, high nutrient concentrations, and frequent algal blooms. To reduce P concentrations, the lake was treated with alum in Spring 2010. This treatment was paired with a fishery renovation and implementation of a no-wake zone in portions of the lake. The lake also underwent hydraulic dredging in summer 2012, with the removal of 80,000 cubic yards of sediment. Following these treatments, water clarity has increased significantly, P and nitrogen concentrations have decreased, and toxic algal blooms declined. The clear water conditions allowed for re-establishment of macrophytes throughout the lake. Carter Lake's average Secchi depth increased from 0.35 m to 1.18 m in the years following restoration, though Secchi disk measurements began to decline again in 2016. Average total P concentrations decreased from 0.19 mg L⁻¹ to 0.08 mg L⁻¹ following restoration. Carter Lake serves as an example of successful use of alum to improve lake conditions (IDNR 2012, 2021).

2.3.2 Case Study: Spring Lake (Michigan, USA)

Spring Lake had high total P concentration, averaging 100 µg L⁻¹ prior to alum treatment. It was estimated that between 55-65% of total P entering the water column was due to internal P loading. In fall 2005, areas of the lake deeper than 15 feet were treated with an alum treatment with concentrations between 10-20 mg L⁻¹. This lake was monitored for 11 years following treatment to assess the longevity of the restoration. In the following 8 months, internal P loading was reduced though water column concentrations remained high. Repeating this monitoring again at 5 years and 11 years post treatment, it was found that water column total P concentrations remained significantly lower than pre-alum conditions (20-80 µg L⁻¹). Though internal P loading rates were beginning to increase as time passed from treatment, they remained an order of magnitude lower than before the alum was applied. Loosely sorbed sediment P concentrations were also very low 11 years following treatment. There were also modest decreases in chlorophyll *a* concentration (Steinman and Hassett 2017).

2.4 Perpetual P Cycling

Table 8. Management strategies for interrupting perpetual P cycling.

Management Action	Internal P Loading Mechanisms Targeted	Additional Benefits	Mitigating Side Effects
Targeted ALUM application at deep site ¹	Interrupt perpetual P cycle, capture some of the chemically mobilized P ²	Does not require dredging	See Table 2.3
Control some organic P loading via waterfowl management ³	Reduce external loading of P ⁴ , though may not be substantial portion of P load	Reduced risk of <i>E. coli</i> ³	Target non-migratory Canada geese, not migratory waterfowl (likely difficult)
Chemical bloom control (e.g., copper sulfate, hydrogen peroxide, barley straw) ⁵	Interrupt production of organic matter contributing to decomposition component of perpetual P cycle ⁶	Reduced risk of cyanobacteria toxins long term (but risk of near-term release with treatment) ⁷	Choice of chemical control can reduce risk of toxin release ⁷ , careful dosing required to avoid acute toxicity for zooplankton, other organisms ⁵
Biomanipulation ⁸	Top-down control on algal production ⁹	Improved fishery ¹⁰ , improved macrophyte populations ¹¹	Can require continuous management and angling restrictions to the food web reverting ⁸

¹Welch et al. 2017, ²Welch and Cooke 1999, ³Swallow et al. 2010, ⁴Manny et al. 1994, ⁵Matthijs et al. 2016, ⁶Song and Burgin 2017, ⁷Kenefick et al. 1993, ⁸Jeppesen et al. 2007, ⁹Moss et al. 1996, ¹⁰Lathrop et al. 2002, ¹¹Lauridsen et al. 2003

When managing lakes that may be susceptible to or are experiencing perpetual loading, there are strategies that should be avoided to prevent tipping into the perpetual cycling pathway for internal loading. One strategy not recommended for lakes susceptible to perpetual cycling is the construction of flow-through sediment retention basins. While these basins are designed to capture incoming sediments and their associated nutrients, as they age, they can instead promote the release of collected P. An assessment of 98 stormwater ponds in Minnesota revealed that 40% of ponds had P concentrations in excess of runoff inputs indicating internal loading in these basins that contributed to downstream eutrophication (Taguchi et al. 2020). This may also be happening in some sediment detention basins upstream of Iowa reservoirs such as Green Valley Lake. During the summer of 2019, total P concentrations were measured at the inflow from the sediment detention basins in the east and west arms of the reservoir and in-lake sites nearby these inflows (Figure 42). Later in the summer, the total P concentration of inflowing water from the basins was higher than in-lake concentrations, especially in the east arm. Internal P loading in the sediment detention basins could be contributing to in-lake P availability in this instance.

Dredging can also induce or exacerbate perpetual cycling by creating larger regions of unstable stratification. This promotes anoxic, redox-mediated P loading when stratified, which then fuels algal growth during periods of mixing. After this algal material dies and sinks to the sediments, decomposition results in release of P under oxic conditions, as observed in Green Valley Lake (Albright and Wilkinson 2022b). Aeration is another strategy mentioned above that can promote increased internal loading in certain lakes. Aeration can promote increased aerobic respiration within the sediments, releasing P previously bound in organic matter. Lakes with high concentrations of organic P in their sediments are more likely to release P when stimulated through aeration. For example, Lake Tuusulanjärvi, a eutrophic lake in southern Finland, was aerated with the goal of disturbing stratification in 1998. Following this

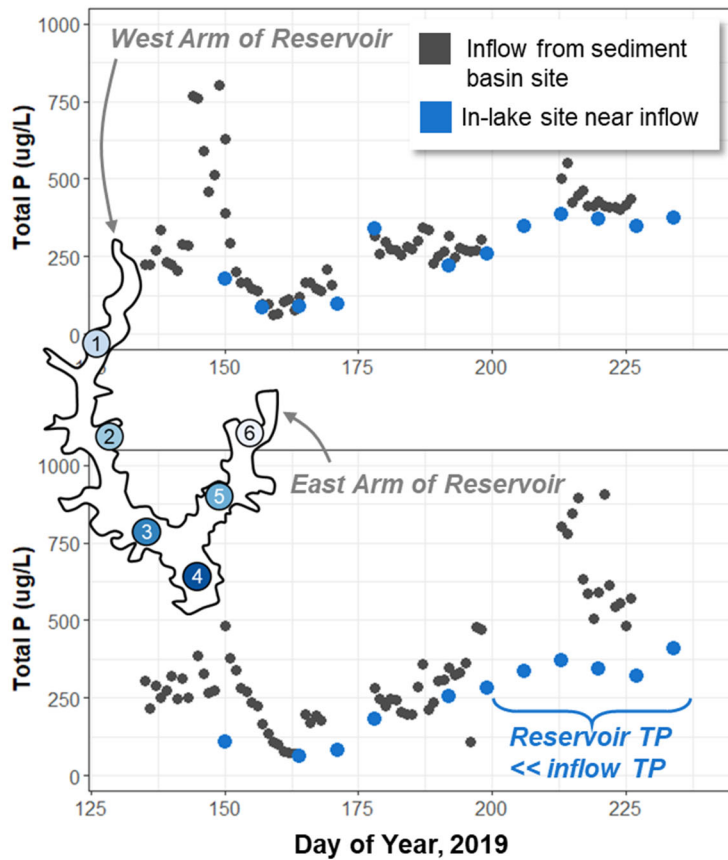


Figure 42. Total P concentrations at the inflow from the sediment detention basins in Green Valley Lake compared to the total P concentrations at sites 1 and 6 nearest to the inlets. The inlet concentrations were sampled with an automated sampler (ISCO) pulling water directly from the culverts connecting the sediment basins and the main reservoir. Later in the summer, TP concentrations are higher at the inlet then compared to the reservoir, potentially indicating P loading from the basins into the reservoir. This pattern is particularly striking in the east arm of the reservoir. Based on observations, the sediment detention basins were productive ecosystems. Perpetual P cycling could be occurring in these basins as well as in the main reservoir, exacerbating the P availability problem in this waterbody.

installation, oxic internal loading increased at least four-fold when compared to rates during other management strategies, averaging $356 \text{ mg P m}^{-2} \text{ y}^{-1}$ (Horpilla et al. 2017).

Strategies for reducing the symptoms of lakes experiencing perpetual P cycling focus on disrupting some part of the cycle. Application of alum to the deep site of the lake could retain some of the chemically mobilized P within the sediments under periods of anoxia (Welch and Cooke 1999). Targeted alum treatment can be a disproportionately effective approach when applied to sediment regions with disproportionately high P release, with percentage of P reduction outpacing the percentage of bottom sediments treated (Welch et al. 2017). However, this approach does have drawbacks, as discussed in Part 2 Section 2.3. Sedimentation rates in perpetual P cycling waterbodies are likely high given the large amount of organic matter being produced in blooms (Downing et al. 2008).

Another method is targeted at reducing external loading of organic P through waterfowl management (Swallow et al. 2010) if there is a large resident population (i.e., thousands of birds). Removing nuisance geese from around and within a water body can reduce substantial amounts of external P loading (Manny et al. 1994), algal growth, and *E. coli* exposure (Swallow et al. 2010). However, it is necessary to target only undesired species such as non-migratory Canada geese to avoid disturbance of desirable migratory waterfowl. On average a goose defecates $\sim 109 \text{ g}$ of dry weight per day with an average P content of 1% of dry weight when feeding on crops in the winter (Dessborn et al. 2016). As a very rough estimate based on these values, a flock of 1000 geese in residence for 30 days will produce approximately 33 kg of P that can either be directly desposited into the waterbody or nearby. A flock of 10,000 geese in residence for 120 days could produce approximately 1308 kg of P.

Other methods of interrupting perpetual cycling focus on reducing primary production. Chemical bloom control, including use of copper sulfate, hydrogen peroxide, barley straw, and other methods (many described in Matthijis et al. 2016), can reduce algal biomass within a lake. Reducing algal biomass should

reduce the amount of organic matter deposition to the sediment, whose decomposition fuels the perpetual P cycle (Song and Burgin 2017). Reducing the stock of cyanobacteria can result in a reduction of cyanobacteria toxins in the long term, though care should be taken if the lake is used as a drinking water source as some chemical methods result in short term toxin release (Kenefick et al. 1993). Dosing of these methods should also be considered carefully, as to avoid toxicity thresholds for zooplankton and other organisms (Metthijs et al. 2016). While some of these remediation strategies have shown promising initial success in certain contexts (mainly small waterbodies, smaller than most Iowa lakes), the long term efficacy and consequences of chemical mitigation are unclear. Therefore, use of chemical mitigation should be done sparingly and approached with skepticism. Similar caution should be employed when considering mechanical mixers and circulators.

Biomanipulation of fish communities is another method to disrupt production. The goal of biomanipulation is to create a fish community that allows for the development of high biomass of grazing zooplankton (Moss et al. 1996, Carpenter et al. 1985). There are many approaches to biomanipulation, some of the most common being the increase in piscivores, removal of zooplanktivores, and complete fishery renovation (Jeppesen et al. 2007). While this method can require maintenance to avoid reversion to the former food web structure (Jeppesen et al. 2007), if successful it can result in an improved fishery (Lathrop et al. 2002) and macrophyte populations (Lauridsen et al. 2003).

2.5 Climate Change and Managing Internal Loading

Iowa lakes are already experiencing the effects of a warming planet with the loss of ice cover, warmer winters, heatwaves, and more frequent and extreme precipitation. These climate dynamics are altering water and sediment temperatures (O'Reilly et al. 2015, Gudas et al. 2011) and stratification and are being compounded by other stressors such as increasing salinity (Dugan et al. 2017, Kushal et al. 2018). All these drivers are likely to alter internal loading mechanisms and magnitude in Iowa waterbodies, which complicates nutrient management.

Increased temperatures alter internal P loading in many ways. Higher water temperatures stimulate respiration and draws down dissolved oxygen concentrations. Aerobic sediment respiration rates in lakes are strongly correlated with temperature across a broad gradient (Gudas et al. 2011, Cardosa et al. 2013), influencing both the redox-mediated and perpetual cycling mechanisms. Warmer waters may also enhance the stratification strength, decreasing bottom water temperatures and the likelihood of mixing events during the summer. This could stimulate redox-mediated loading in a less enriched lake or interrupt perpetual cycling in a hypereutrophic lake. Conversely, increased temperatures may result in stronger wind events or atmospheric stilling in the region. This will alter the magnitude of wind resuspension, stratification strength and depth, and entrainment of P-rich bottom waters.

Warmer waters will also stimulate carp production, contributing to bioturbation as a mechanism of internal loading. If ice cover is lost, the timing of carp production may also shift, and goose occupancy may increase. Heatwaves (discrete extreme temperature increases) are also increasing in frequency during the summer and will have similar effects as warming temperatures but on a faster timescale. Heatwaves may also induce senescence of macrophytes (Albright et al., in review), contributing to wind-driven sediment resuspension or enhanced diffusion from the sediments under oxic conditions.

With warming winters and changing precipitation patterns potentially leading to more ice and snow storms, the application of road salt is likely to continue increasing in Iowa. Erosion enhanced by intensive agriculture on the landscape is also contributing to salinization syndrome (*sensu* Kaushal et al. 2018 in Iowa's waters. Chloride is an anion that can also sorb to sediment particles, similar to loosely-bound P. With increasing chloride pollution, non-specific anion exchange may enhance release of loosely-bound P in the water column. Carp are also salt-tolerant and continuing salinization will favor carp dominance.

Both the magnitude and frequency of summer precipitation events are changing in the Midwest. Rain storms deliver P from the watershed and can stimulate algal production. Subsequently, this organic matter

will be mineralized, releasing the P back into the water column (Stockwell et al. 2020). Through this mechanism, changing precipitation patterns may enhance or induce perpetual P cycling. Storms may also disturb stratification and lead to entrainment of P released through redox processes.

Taken together, the effects of climate change are likely to enhance internal loading in Iowa's lakes and reservoirs. Any management or restoration efforts must consider the interaction of climate change with the dominant mechanism of internal loading to adequately address the problem long-term.

SECTION 3 – PROGRAM RECOMMENDATIONS

3.1 Recommendations for Diagnostic Studies and Other Targeted Sampling

3.1.1 Collecting High Frequency Data

With the ongoing improvements of reliable, affordable sensor technology, monitoring temporal dynamics in surface waters at the scale of minutes to years is a useful method of observation. For lakes targeted for restoration, we recommend that low-cost temperature sensors be deployed in series to monitor stratification dynamics for at least one year. Additionally, dissolved oxygen sensors can be deployed in the surface and bottom waters of lakes to track changes in redox conditions with stratification. Both temperature and dissolved oxygen high frequency data streams are critical for diagnosing the dominant mechanism of internal P loading in a lake (Figure 25). Given the interannual variability of weather patterns in Iowa, multiple years of high frequency data collection are ideal. However, even one summer season of high frequency data will be illuminating if only the low-frequency Ambient Lake Monitoring data are available. Where appropriate, additional high frequency monitoring of algal pigment concentrations (chlorophyll a, phycocyanin), turbidity, pH, and specific conductance with automated sensors is useful for teasing apart bloom dynamics and the biophysical response of the ecosystem to storm events (e.g., Ortiz et al. 2020, Ortiz et al. 2021, Wilkinson et al. 2018). Specific recommendations for instrument deployment, sensor maintenance, and data analysis are presented in Appendix A4.

3.1.2 Sediment Sampling and Core Incubations

Targeted sampling to better understand internal P loading may include chemical analyses of bottom sediments or sediment core incubations to measure P exchange across the sediment-water interface. These direct measurements of sediment P dynamics can prove essential to building an understanding of the magnitude and mechanisms of internal loading. Analyses of sediment P content and composition can reveal the degree of P enrichment as well as the likely mechanisms of internal loading based on the relative abundances of different chemical forms of P (see Part 1 Section 2 for more detailed discussion of sediment P forms and associated mobilization mechanisms). *Ex situ* sediment core incubations are a common method for direct measurement of P exchange between sediments and the overlying water (Orihel et al. 2017; Oghdal et al. 2014). These direct measurements may be used to validate lake P models and understand how sediment P release varies across different sites and seasons within a given waterbody. Understanding this spatiotemporal variation may reveal areas of the lakebed or times of the year with high internal loading rates, which can then allow for targeted management. Core incubations may also be used to test how sediment P release responds to different experimental treatments such as oxic versus anoxic conditions or the addition of alum or other sediment capping agents. See Appendix A2 for a detailed discussion of research considerations for sediment analyses and core incubations including sampling site selection, sampling frequency and timing, and incubation conditions.

3.1.3 Data Archiving

The data collected for commissioned Diagnostic and Feasibility Studies and other targeted sampling efforts have value long after the project is completed. Currently, the Iowa DNR does not have the database infrastructure to capture and archive the numerous data formats used to diagnose the dominant internal loading mechanism (e.g., profile data, high frequency data, sediment data, zooplankton data). At this time, it would not be reasonable for the state to invest in expanded database capacity as the implementation costs for dozens of data formats far outweigh the benefits. However, these are public data and need to be findable, accessible, interoperable, and reusable (FAIR Principles, Wilkinson et al. 2016). Data archiving practices and public accessibility for non-standard AQUiA data sets are not consistent across the water quality programs currently. This has led to the loss of critical raw data as previous partners and contractors have retired, left the state, or declined to share the raw data. This is a lost opportunity to reanalyze and synthesize previously collected data, which would save the state money.

We recommend that the state require contractors to deposit non-AQUiA compatible data sets in a permanent archive with robust metadata standards. Repositories that simply allow investigators to deposit data files without compulsory metadata standards (e.g., Iowa State University's Data Share) are not advisable as the data are unlikely to be reusable by future investigators without sufficient metadata. The ideal repository for Iowa lake data is the **Environmental Data Initiative** (<https://edirepository.org/>). This is the same repository where all data for this report have been deposited (Albright et al. 2022c). The Environmental Data Initiative (EDI) is a free, secure, and persistent archive designed for environmental data with stringent metadata standards. In addition to hosting nearly 10,000 data packages from investigators around the world, it is the data repository for the National Science Foundation's Long-Term Ecological Research Program. The staff at EDI also provide one-on-one support for data curation and submission. Preparation of a data submission to EDI requires no specialized skills beyond the capabilities of a person who is already collecting and analyzing the data. We strongly recommend that Iowa DNR consider adding data and metadata submission to a stable repository that meets the FAIR Principles (i.e., EDI) as a deliverable for future contracts.

3.2 Recommendations for the Ambient Lake Monitoring Program

The Iowa DNR has supported a multi-decadal, comprehensive water quality monitoring program for the state's lakes and reservoirs. This dataset is a valuable resource for initially diagnosing the source of water quality problems such as internal loading and monitoring changes following restoration. Within the structure of the current Ambient Lake Monitoring Program (ALM) and Shallow Lakes program (SL), there are opportunities to enhance sampling efforts on lakes targeted for internal loading restoration to gather evidence of the dominant mechanisms.

3.2.1 Summary statistics for profile data below the thermocline

Currently, profiles are not accessible in the AQUiA database, making it difficult for managers to access critical information about bottom water DO and turbidity concentrations. Requesting summary information (like the information already requested above the thermocline) would allow for easier tracking of bottom water DO and turbidity. Currently, the data submissions include a calculation of mean epilimnetic values for variables taken with the handheld meter (all depths above the thermocline). For minimal extra work by the contract lab, the DNR can request a summary of these values below the thermocline as well. This additional summary should be feasible to produce at little or no additional cost. The same information can also be produced with previously collected profiles; however, this will take a bit more effort for the period prior to 2017 due to the way in which profile data were shared with the DNR by previous contractors.

3.2.2 Filter samples in the field for more accurate P quantification

Reed et al. (2022) performed an analysis of orthophosphate (PO_4^{3-}) concentrations in relation to the time between sample collection and filtering. They found that samples filtered >22 hours after collection (current ALM program standard) varied 480% from field-filtered samples. In Green Valley Lake in 2019, orthophosphate samples were taken less than 24 hours apart in the surface waters as a part of the ALM program (filtered next day) and this study (filtered in the field). Soluble reactive P differed by 123% between these two methods, at $169.4 \mu\text{g L}^{-1}$ for the ALM program and $208.6 \mu\text{g L}^{-1}$ for field filtering. Accurate quantification of P concentrations is critical for diagnosing internal loading mechanisms. Field filtering can be accomplished using 60 mL luer lock syringes and filter capsules (e.g., Fisher #9928061). Field blanks of DI water from the lab filtered in the field could be used to track potential contamination.

Transitioning to field filtering may cause a shift in the long-term P concentration data that is unrelated to environmental drivers. As the data are used for long-term trend analysis and assessments, we recommend a one-year study to document any changes in concentration due to filtering method. Samples filtered in the field and samples handled according to current program practices should be run in parallel and

compared. If shifts are substantial, a new field in the database can be added along the lines of “baseline corrected orthophosphate”.

3.2.3 Collect bottom water samples for total and soluble P quantification

We recommend that water samples from 1 m above the sediment surface be collected for soluble reactive P (SRP) analysis. These data can be compared to concentrations in the surface waters and used to track P build up over time, helping diagnose the dominant mechanism of internal P loading in the waterbody. As illustrated by the data collected in Green Valley Lake in 2019, there was considerable variation in soluble reactive P concentrations between the surface waters and bottom waters at most sites (Figure 42). While not as extreme as Green Valley Lake, other Iowa lakes consistently have large differences in surface and bottom water SRP concentrations, though little difference in nitrate+nitrite concentrations (Figure 43). Additional cost could be limited by restricting bottom water sampling to lakes that are currently stratified, lakes targeted for restoration, or added to all lakes. Samples can be collected using a Van Dorn sampler by crews already sampling for the ALM program. Costs would include an initial investment in Van Dorn samplers (approx. \$400 per Van Dorn) and additional sample analysis costs. The additional time on the lake to take the sample would be minimal (3-5 minutes maximum).

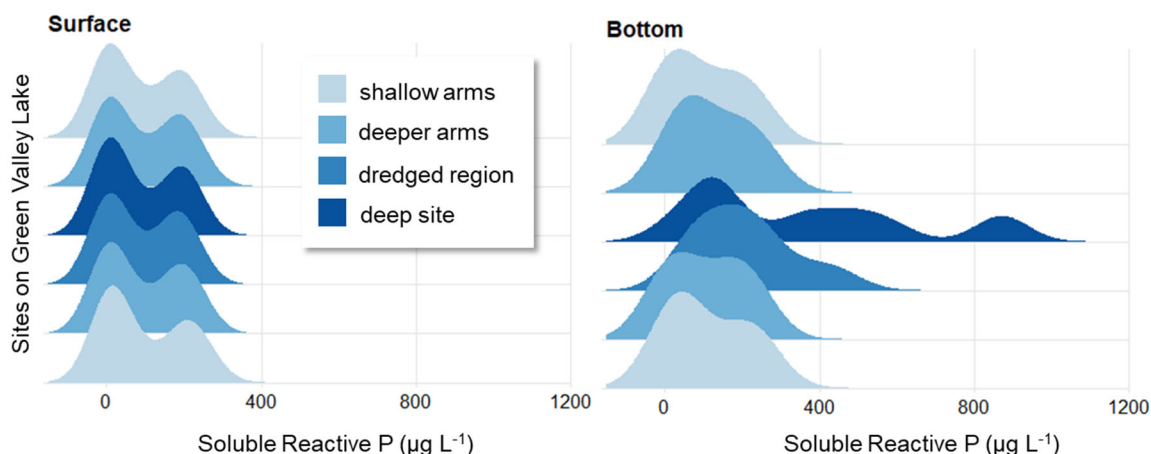


Figure 43. Distribution of soluble reactive P concentrations from weekly sampling from May – September 2019 in the surface and bottom waters of Green Valley Lake. There were 6 sampling sites (Figure 22).

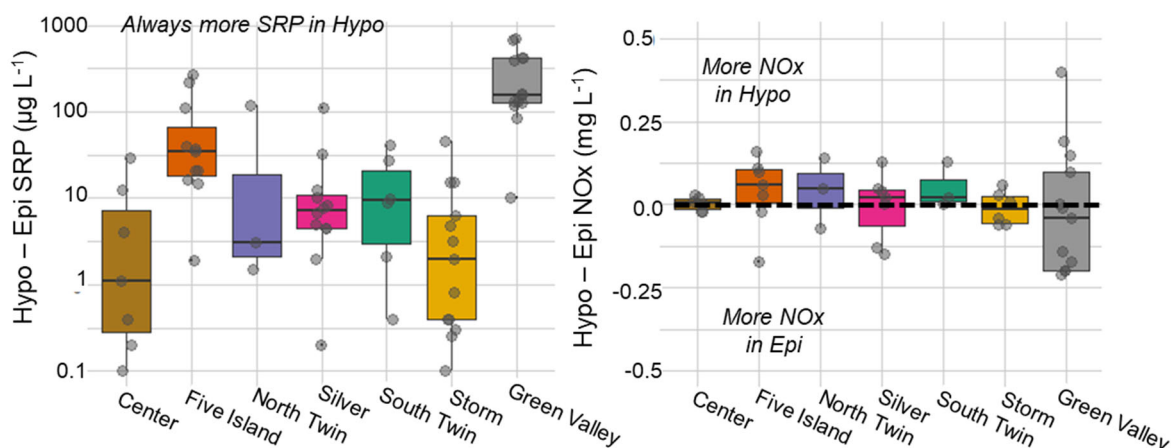


Figure 44. Difference in soluble reactive P (left panel) and nitrate+nitrite (NOx, right panel) concentrations between the hypolimnion and epilimnion of various Iowa lakes sampled from May through September between 2018-2020. There are large differences in SRP concentrations between the two depths, but minimal difference in nitrate concentration.

3.2.4 Augmented Sampling

As illustrated by the intensive study of Green Valley Lake in 2019 (Figures 42) and the lakes in the Carp Incentivized Harvest program, more frequent sampling of nutrients and hydrodynamic conditions (e.g., stratification) are helpful for developing hypotheses regarding the dominant mechanisms of internal loading. Candidate lakes for restoration that are located in interior counties of the state could be efficiently sampled 2-3 times per sampling round to increase the temporal resolution of the data. Some variables such as zooplankton biomass may not be necessary to measure as frequently. For lakes that are not easily added to alternate sampling routes as a part of the Ambient Lake Monitoring program (e.g., lakes in border counties), sampling “kits” with supplies, preservatives, and detailed instructions can be collected by DNR staff on a regular basis. Samples can then be handed off to the Ambient Lake Monitoring field sampling crews periodically or shipped to a contract lab.

Additionally, in spatially complex ecosystems such as Green Valley Lake, a reservoir with two arms draining two subwatersheds, additional sampling outside the historical deep point location is warranted. Sample locations should be selected based on depth, inputs, and data-driven hypotheses of internal loading mechanisms. For example, in Green Valley Lake, additional sampling sites could be positioned at the locations sampled in 2019 (Figure 44) to capture P concentrations at the inlets to each arm (sites 1 and 6), intermediate depth regions that stratify intermittently (sites 2, 3, and 5), and the deep point (site 4).

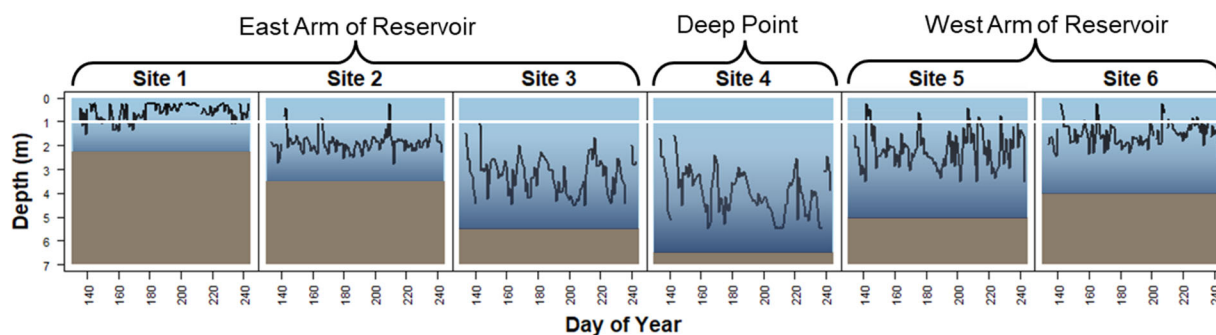


Figure 45. Estimate of thermocline depth at six locations in Green Valley Lake for May-September 2019. The sites numbers are the same as Figure 22. The relative depths of each site are shown as the depth of the blue background in the panel. When the time series (black line) is above 1 m (white horizontal line), mixing is likely occurring.

3.3 Recommendations for TMDL Analyses and Water Quality Improvement Plans

Iowa’s lakes and reservoirs share some common features that make them more susceptible to internal P loading. These features include shallow morphometry which causes sediment-water interactions to be highly influential on overall nutrient availability. All the waterbodies are also embedded in a nutrient-rich landscape with significant P inputs causing legacy storage on the landscape and substantial landcover disturbance (Goyette et al. 2018, Van Meter et al. 2021). Given these common features, internal P loading should be considered in all total maximum daily load (TMDL) models for Iowa lakes and reservoirs with P-associated impairments. Our analysis of 51 TMDL reports from Iowa found that 51% of reports either ignored internal loading, assumed it was negligible based on residence time, or failed to separate internal from external loading in the analysis (Table S2).

We recommend, at minimum, distinguishing between external and internal loads of P in TMDL calculations to estimate the magnitude of each source and guide in-lake restoration efforts. This could be achieved by coupling SWAT and BATHTUB models. To better parameterize the BATHTUB or other lake P model, augmented Ambient Lake Monitoring (Part 2, Section 3.2) or Diagnostic and Feasibility Studies can be used to collect data. These efforts should be paired with empirical analyses of internal P loading dynamics such as measuring bottom water P concentrations during stratification and mixing events with automated sensors. Direct measurements of internal loading rates with core incubations

(Appendix A2) can be used to validate model results (e.g., Box 1). Finally, if in-lake interventions are proposed as a part of the Water Quality Improvement Plan, we recommend using the flow chart (Figure 25) to develop hypotheses regarding the likely mechanisms of internal loading when making recommendations. These hypotheses should then be tested with additional data collection as detailed above.

Beyond these minimum recommendations, we suggest replacing BATHTUB with a more advanced modeling approach that considers lake hydrodynamics and takes a process-based approach to in-lake P cycling, rather than the mass balance approach used in BATHTUB. Specifically, we recommend the General Lake Model (GLM) because it is an open-source, state of the art model that is suitable for most Iowa waterbodies. The GLM is a one-dimensional, process-based, hydrodynamic model for water balance and thermal stratification in lake or reservoir ecosystems. Process-based models are designed such that the relationship between forcing variables (e.g., meteorology) and an ecosystem state variable (e.g., P concentration) is defined based on the physical mechanisms that underpin that relationship. As a vertical, one-dimensional model, GLM considers change in ecosystem conditions (e.g., water temperature) with depth in the water column and assumes horizontal homogeneity across the lake. GLM can be coupled with a water quality model (GLM-AED2) to simulate vertical profiles of nutrients (including P), dissolved oxygen, phytoplankton, and zooplankton as well as sediment processes. GLM is a powerful model with myriad applications including climate change simulations (e.g., effects on stratification, ice cover, oxythermal habitat availability), determining optimal water withdrawal depths in reservoirs, and smart forecasting for impairments in systems used for drinking water supplies. The GLM has extensive documentation and online support available at <https://aed.see.uwa.edu.au/research/models/glm/>.

The output of the GLM hydrodynamic model is a timeseries of water temperature profiles at an hourly or daily timestep, depending on the resolution of the input data. This output allows the user to evaluate expected thermal structure over time, which may be used to calculate the depth, duration, frequency, and timing of stratification events. Such an analysis may help illuminate likely internal P loading mechanisms based on stratification patterns and identify times of the year when internal loading may be more likely. Additionally, the GLM may be used to test how waterbody thermal structure and dynamics may change under different scenarios such as proposed changes in hypsography from dredging. Additionally, the GLM-AED2 is a powerful model addition that may be used to better understand P cycling dynamics in a given waterbody and test different stratification, dissolved oxygen, external P loading, and weather effects on internal P loads. Overall, GLM and GLM-AED2 provide opportunities to understand links between lake thermal structure and internal P loading and scenario testing that will strengthen Water Quality Improvement Plans.

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APPENDICES

A1. SUPPLEMENTAL TABLES AND FIGURES

Table S1. Overview of internal P loading approaches in TMDL reports.

Lake Name	Impairments	Year	Internal Loading in TMDL?
Arrowhead Pond	Algae	2020	Assumed that internal P loads were negligible on an annual scale due to short water residence times, watershed to lake area ratios, or lack of data.
Bob White Lake	Low DO, Algae	2020	
Eldred Sherwood Lake	Algae	2020	
Five Island Lake	Algae, Turbidity	2006	
Iowa Lake	Algae, Total P	2016	
Kent Park Lake	Algae, pH, Total P	2015	
Lake Cornelia	Algae, Turbidity	2006	
Lake of the Hills	Algae, Total P	2018	
Prairie Rose Lake	Algae, Turbidity	2020	
Upper Pine Lake	Algae	2014	
Windmill Lake	Algae, Total P, Turbidity	2018	Did not quantify internal P loads but recommend doing so in the future.
Carter Lake	Algae, Turbidity	2007	
Crysal Lake	Organics	2002	
Easter Lake	Siltation, Nutrients	2008	
Lake McBride	Nutrients, Siltation	2005	
Lake of Three Fires	Nutrients, Siltation	2003	
Mariposa Lake	Algae, Turbidity	2004	
Arbor Lake	Nutrients, Siltation	2003	
Badger Creek Lake	Nutrients, Siltation	2003	
Beaver Lake	Algae, pH	2014	Did not include internal P loading or did not separate internal and external loads
Don Williams Lake	Organic Enrichment	2005	
Ingham Lake	Algae, Turbidity	2005	
Lake Darling	Nutrients, Siltation	2003	
Lake Meyer	Nutrients, Siltation	2005	
Little Wall Lake	Algae, Turbidity	2006	
Littlefield Lake	Algae, Turbidity	2006	
North Twin Lake	Algae, Turbidity	2004	
Rock Creek Lake	Siltation, Nutrients	2002	
Silver Lake (Delaware)	Algae, Ammonia, pH, Turbidity	2009	Mass balance approach to estimate annual internal P loads. Waterbodies used for further analysis of TMDL estimates.
Silver Lake (Delaware)	Nutrients, Siltation	2002	
Tuttle Lake	Algae, Turbidity	2005	
Upper Gar Lake	Algae, Turbidity	2005	
Avenue of the Saints Lake	Algae, Turbidity, pH	2019	
Black Hawk Lake	Algae, Turbidity	2012	
Blue Lake	Algae, Turbidity	2009	
Casey Lake	Algae, pH	2013	
Clear Lake	Algae, Nutrients	2007	
East Lake Osceola	Algae, Turbidity	2008	
Green Valley Lake	Algae, Turbidity, Low DO	2019	
Hannen Lake	Algae, pH	2013	

Indian Lake	Organics, Noxious Aquatics	2005
Lake Keomah	Algae, pH	2012
Little Spirit Lake	Algae, Turbidity	2005
Lower Pine Lake	Organic Enrichment	2003
Otter Creek Lake	Algae	2004
Ottumwa Lagoon	Algae, Chloride, Turbidity	2005
Silver Lake (Palo Alto)	Algae, Turbidity	2005
Silver Lake (Worth)	Algae, Turbidity	2006
Swan Lake	Algae, Turbidity	2004
Trumbull Lake	Algae, Turbidity	2006
Volga Lake	Algae, Non-Algal Turbidity	2014

Table S2. Overview of data extracted from TMDL reports

Lake Name	TMDL Model Estimates			Water Residence Time (days)	Ecosystem Type
	Internal P Load (kg year⁻¹)	Total P Load (kg year⁻¹)	Internal P Load (%)		
Avenue of the Saints Lake	1,376.2	2,730.6	50.4	26	Impoundment
Black Hawk Lake	2,856.7	19,328.8	14.8	133	Glacial Lake
Blue Lake	98.0	1,206.8	8.1	117	Oxbow Lake
Casey Lake	266.0	688.3	38.6	NA	Impoundment
Clear Lake	798.2	8,435.4	9.5	1715	Glacial Lake
East Lake Osceola	192.7	333.8	57.7	96	Impoundment
Green Valley Lake	61.4	3,180.2	1.9	246	Impoundment
Hannen Lake	57.4	294.1	19.5	NA	Impoundment
Indian Lake	36.3	108.8	33.3	NA	Impoundment
Lake Keomah	367.8	2,071.2	17.8	93	Impoundment
Little Spirit Lake	267.6	848.1	31.6	1460	Glacial Lake
Lower Pine Lake	589.6	3,628.1	16.2	NA	Impoundment
Otter Creek Lake	132.4	678.9	19.5	NA	Impoundment
Ottumwa Lagoon	113.4	1,564.6	7.2	88	Oxbow Lake
Silver Lake (Palo Alto)	276.6	4,698.4	5.9	219	Glacial Lake
Silver Lake (Worth)	196.4	745.6	26.3	453	Glacial Lake
Swan Lake	81.6	902.5	9.0	548	Glacial Lake
Trumbull Lake	5,986.4	9,886.6	60.6	40	Glacial Lake
Volga Lake	861.7	3,072.1	28.0	NA	Impoundment

Table S3. Variation in sediment P among lakes and within-lake heterogeneity, 2018 (dashed lines are missing data)

Lake Name	Center	Five Island	Silver (Dickinson)	Storm	North Twin	South Twin	Swan
Total P ($\mu\text{g P g}^{-1}$ dry sediment)	938.3 (± 191.4)	1,013.5 (± 160.9)	936.5 (± 97.0)	738.2 (± 29.5)	1,134.1 (± 288.3)	939.1 (± 113.1)	1,164.7 (± 306.2)
% Redox-sensitive	24.45	30.30	22.81	31.16	9.62	11.81	35.58
% Labile organic	31.45	25.02	21.79	19.06	--	--	--
% Refractory organic	21.85	15.98	17.53	22.35	--	--	--
% Total Organic	53.31	41.00	39.32	41.41	74.81	70.48	50.14
% Loosely-bound	7.75	15.04	21.79	9.88	10.74	12.09	3.16
% Al-bound	7.02	8.21	7.50	6.91	2.79	1.96	9.72
% Ca-bound	7.47	5.44	8.58	10.64	2.04	3.64	1.40
CV Total P (%)	10.2	19.0	6.4	17.5	7.3	22.4	15.2
CV Loosely-bound (%)	17.5	32.0	17.7	37.3	24.4	11.4	39.7

Note. Total P values are the mean ($\pm$ standard deviation) for all depth intervals of a sediment core taken at the deep site of the lake. The percent contribution of each P species to the total sediment P pool is an average value across the deep site core profile. The coefficient of variation (CV) for total and loosely-bound P was calculated for all ten sediment cores collected across each study lake. Table reproduced from *Albright et al. 2022a*.

Table S4. Population density (kg/ha) estimate with the 95% confidence interval (CI) and harvest estimates from the incentivized harvest program on the study lakes (2017-2020). Harvest as a percentage of population density (% pop) was calculated as the harvest total during the fall to spring season compared to the population estimate from the preceding summer. If a population estimate was unavailable for the prior summer (e.g., Five Island Lake in 2017), the harvest as a percentage of the population was not available (NA). Data from Simonson et al. (2022).

Lake	Year	Season	Common Carp				Bigmouth Buffalo			
			Population		Harvest		Population		Harvest	
			kg/ha	95% CI	kg/ha	% Pop	kg/ha	95% CI	kg/ha	% Pop
Center Lake	2017	summer	134.8	(54-214)			164.3	(115-212)		
		fall/spring			23	17.1%			222	135.1%
	2018	summer	182.7	(79-286)			81.3	(35-127)		
		fall/spring			12	6.6%			61	75.0%
	2019	summer	45.4	(3-89)			67.5	(2-242)		
		fall/spring			0	0.0%			0	0.0%
	2020	summer	77.1	(15-139)			105.2	(59-151)		
Five Island	2017	summer	NA	NA			NA	NA		
		fall/spring			15	NA			17	NA
	2018	summer	230.8	(17-285)			20.2	(1-39)		
		fall/spring			19	8.2%			185	915.8%
	2019	summer	126.6	(54-198)			9.7	(0-66)		
		fall/spring			0	0.0%			0	0.0%
	2020	summer	169.9	(101-237)			24.7	(1-52)		
North Twin	2017	summer	23.6	(7-39)			255.6	(77-433)		
		fall/spring			0	0.0%			0	0.0%
	2018	summer	63.2	(32-94)			504.2	(218-790)		
		fall/spring			17	26.9%			356	70.6%
	2019	summer	43.2	(17-68)			474.2	(16-938)		
		fall/spring			13	30.1%			276	58.2%
	2020	summer	11.6	(0-79)			32.0	(2-115)		
Silver Lake	2017	summer	NA	NA			NA	NA		
		fall/spring			0	NA			0	NA
	2018	summer	91.2	(62-119)			86.4	(2-202)		
		fall/spring			10	11.0%			65	75.2%
	2019	summer	119.5	(54-184)			31.5	(0-216)		
		fall/spring			71	59.4%			74	234.9%
	2020	summer	65.5	(43-87)			13.5	(3-23)		
South Twin	2017	summer	101.8	(30-172)			50.3	(38-62)		
		fall/spring			0	0.0%			0	0.0%
	2018	summer	69.0	(42-95)			290.4	(147-433)		
		fall/spring			0	0.0%			0	0.0%
	2019	summer	48.5	(19-77)			100.8	(3-274)		
		fall/spring			0	0.0%			0	0.0%
	2020	summer	84.9	(1-198)			59.8	(1-411)		
Storm Lake	2017	summer	NA	NA			NA	NA		
		fall/spring			0	0.0%			0	0.0%
	2018	summer	13.8	(10-17)			68.2	(1-468)		
		fall/spring			0	0.0%			0	0.0%
	2019	summer	27.6	(13-41)			0.4	(0-2)		
		fall/spring			0	0.0%			0	0.0%
	2020	summer	37.2	(25-49)			5.1	(0-18)		

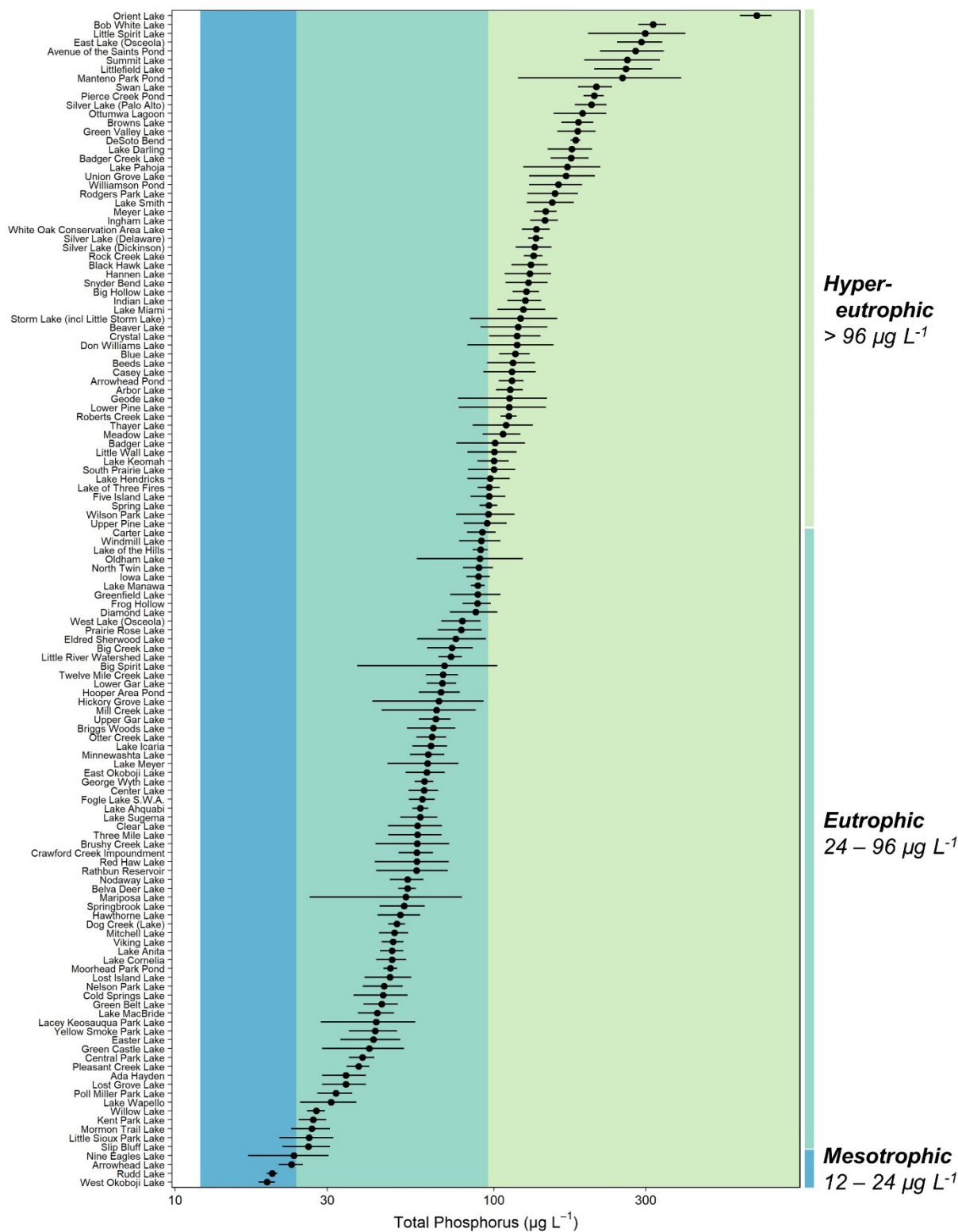


Figure S1. Mean annual total P concentrations in Iowa lakes and reservoirs. Dots are the mean concentration from 2017–2021, and error bars are the standard error of the mean. The x-axis is on a $\log_{10}$ scale, and mean total P concentrations range from 19.4–667.3 $\mu\text{g L}^{-1}$. Data source: IDNR 2022.

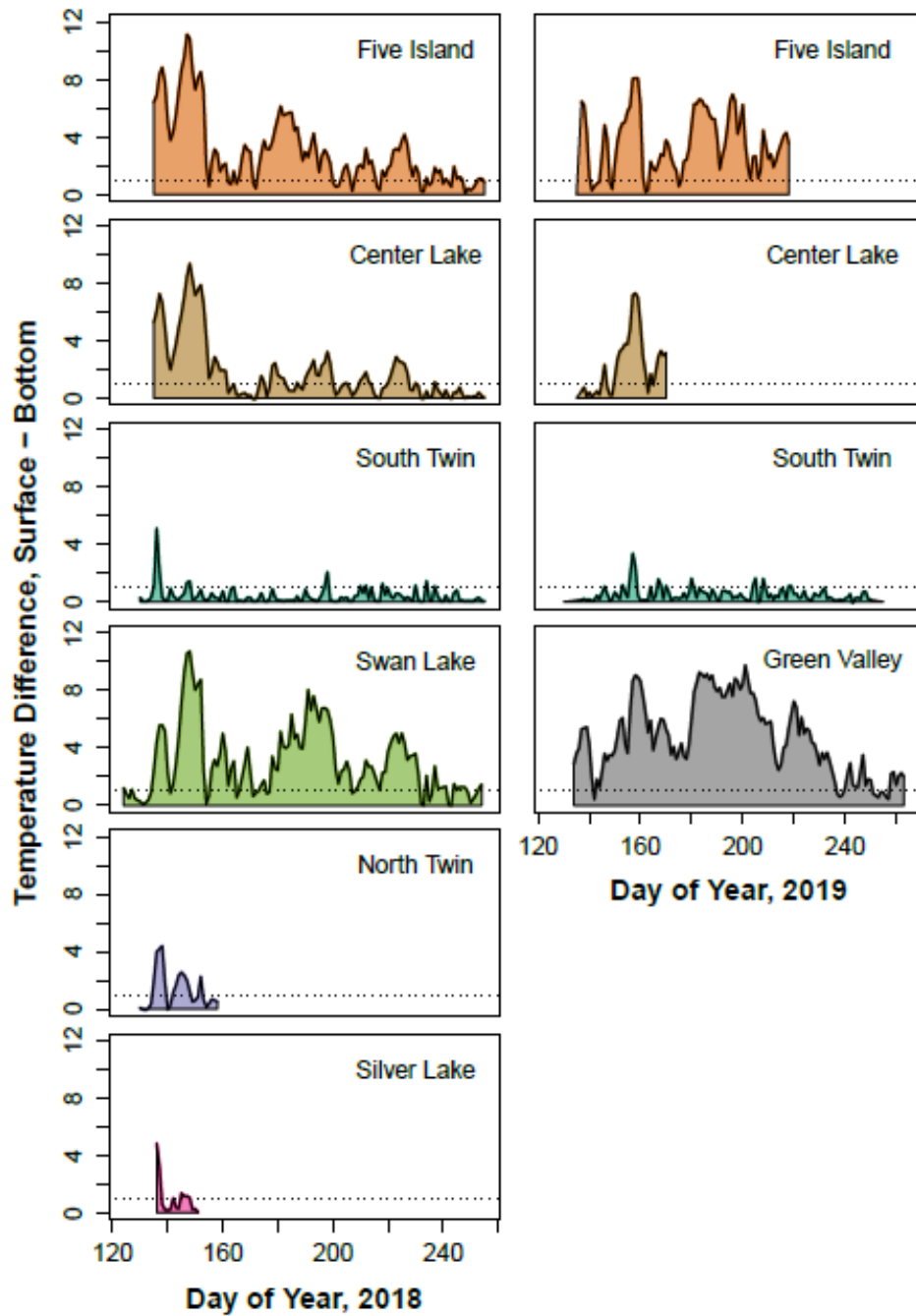


Figure S2. The difference in temperature from the surface (0 m) to 0.5 m from the sediment from thermistors measuring every 30 minutes in 2018 and 2019. The horizontal dashed line in each panel is the 1°C difference, the Iowa DNR definition of stratification. Abbreviated time series were due to sensor theft and sensor malfunctions. Data source: Carp biomanipulation study.

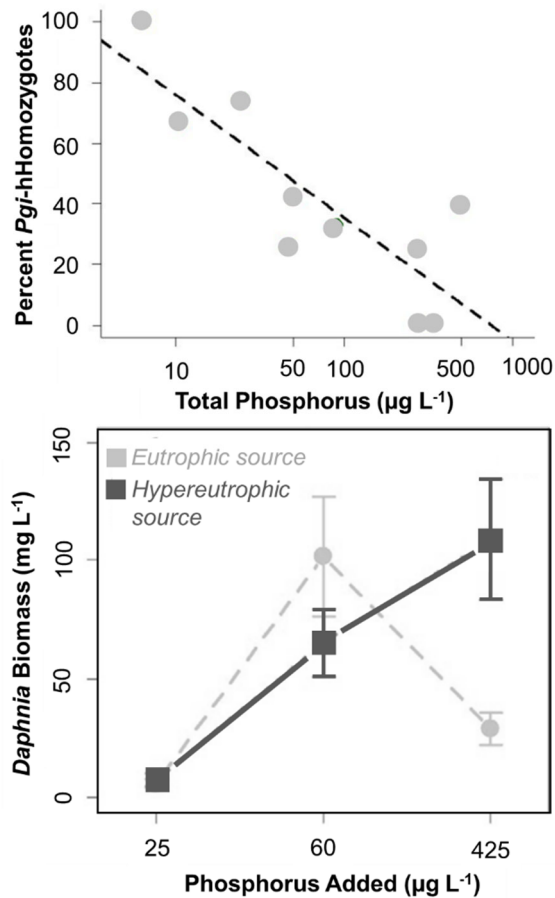


Figure S3. Extreme P enrichment in lakes has led to rapid adaptation of *Daphnia*, a keystone grazer, causing them to be more efficient grazers and decreasing top-down control on phytoplankton.

(top panel) The percent of the *Daphnia* population in Iowa lakes that are Pgi-homozygotes decreases with increasing total P concentrations in lakes. Pgi-homozygotes (low TP conditions) lose more energy during glycolysis, facilitating acquisition of the limited P in the environment (higher P grazing rate). Pgi-heterozygotes (high TP conditions) lose less energy during glycolysis, preventing consumption of excess P from the environment (lower P grazing rate).

(bottom panel) In a common garden experiment, *Daphnia* from a eutrophic Iowa lake have a higher biomass in eutrophic conditions compared to *Daphnia* from a hypereutrophic Iowa lake. However, under hypereutrophic conditions, eutrophic source *Daphnia* have a substantially lower biomass.

Figures reprinted from Moody et al. (2022)

A2. SEDIMENT SAMPLING AND CORE INCUBATIONS

Chemical analyses of sediments or sediment core incubations to measure P exchange across the sediment-water interface can provide important insights into the magnitude of internal P loading in each waterbody and illuminate the underlying mechanisms. However, these analyses are challenging and costly. As such, careful research design is advised to optimize sampling effort and obtain results that can be applied reasonably to the whole waterbody. This appendix outlines key research considerations for sediment analyses and core incubations including sampling site selection, sampling frequency and timing, and different options for incubation conditions.

A2.1 Sampling Site Selection – Sediment Chemistry

The spatial distribution of sediment P content and composition is expected to be heterogenous in Iowa lakes and reservoirs (Albright et al. 2022; Albright and Wilkinson 2022). As such, this spatial variation should be considered in sampling design for sediment chemical analyses. Although the exact sampling procedure will need to be tailored to specific research goals and waterbody features, below are some general guidelines that can be used to begin sampling site selection.

- Sediment P concentrations generally increase with water depth due to sediment focusing processes that concentrate fine grain, P-enriched sediments in the deepest areas of the lake basin. As such, P concentrations at the deep site of a waterbody will not be representative of the entire lakebed. Field evidence from glacial lakes in northwest Iowa suggests that a minimum of six sampling sites is required to produce an accurate estimate of whole-lake mean sediment P concentrations (Albright et al. 2022). Ideally, those six sites would span a range of water depths from the deep site of the lake basin to nearshore environments and cover diverse areas of the lakebed (e.g., vegetated versus bare sediments, a variety of sediment grain sizes from sand to clays, and areas near river inflows). If budgetary or other logistical constraints require fewer than six sampling sites, at minimum the deep site and one nearshore area should be sampled for each basin of a given waterbody.
- There are several options for analyzing P concentrations with depth in the sediment profile. Most gravity coring devices collect a half meter deep sediment core, which can then be sliced into discrete depth intervals for further analysis. For example:
 - All surface sediments can be combined into one sample for analysis. The 0–10-centimeter depth interval is typically used for this type of analysis as the top ten centimeters of sediment can be considered actively exchanging with the overlying water (e.g., Albright et al. 2022; Albright and Wilkinson 2022).
 - The sediment core can be sliced more finely (e.g., every 2 centimeters) to analyze change in P concentrations with depth in the sediments. This analysis can reveal the history of total P enrichment to the sediments as well as the depth of sediment mixing, which can be inferred based on change in loosely-bound P concentrations with depth in the sediment profile.
 - If dredging is planned for a lake, coring depth and subsequent analysis could be informed by the sediment depth of the proposed dredging to determine anticipated P levels in the surface sediments following dredging.

A2.2 Sampling Site Selection – Sediment Core Incubations

Internal P loading rates are expected to vary within Iowa lakes and reservoirs due to spatial variation in controlling mechanisms such as water column thermal mixing, bottom water dissolved oxygen concentrations, and organic matter sedimentation (Albright and Wilkinson 2022). As such, multiple sampling sites are recommended for sediment core incubations to measure P exchange between sediments and the overlying water. Site selection should reflect project goals and waterbody features; however, some general guidelines include:

- Collect sediment cores for incubations at the same sites where other sediment samples have been taken (e.g., for analysis of sediment P chemistry). Pairing sediment analysis and core incubation sampling sites will increase the utility of both data sets.
- Select sampling sites that experience a range of water column stratification frequency. At minimum, the deep site (may experience thermal stratification) and one nearshore area (unlikely to experience thermal stratification) should be sampled per basin of the waterbody.
- Consider spatial variation in other drivers of internal P loading and select sites that capture the range of those drivers. For example:
 - Nearshore sediments will likely remain well-oxygenated all summer while deeper sediments may experience anoxia during stratification events.
 - Organic matter inputs and lability will differ for sites near river inflows (greater terrestrial organic matter inputs) versus more lacustrine regions of the waterbody (greater inputs of *in situ* organic matter production).

A2.3 Sampling Timing and Frequency

Internal P loading rates and the P composition of surficial sediments can vary across seasons in Iowa lakes and reservoirs. Beginning to construct these seasonal dynamics requires sampling under winter ice cover and then during both thermal mixing and stratification events throughout the open water season from spring to autumn (e.g., Albright and Wilkinson 2022). As this intensive sampling effort may be unfeasible for many projects, we recommend focusing on the spring and summer seasons. P that is released from the sediments during spring and summer is most likely to be available to fuel algal blooms. Additionally, sampling in spring and summer will capture different water temperatures, mixing versus stratification dynamics, and dissolved oxygen levels. Specifically, spring sampling may illuminate instances of oxic internal P loading when the water column is well-mixed while summer sampling events might capture anoxic internal loading under thermal stratification (e.g., Albright and Wilkinson 2022). Although we strongly recommend sampling at least two times over the course of a year in spring and summer, if only one sampling event is possible, we recommend targeting early to mid-summer, shortly after the onset of thermal stratification, as internal loading rates are likely to be high at this time. Monitoring water column thermal structure (e.g., with chains of high frequency temperature sensors, Appendix A4) can inform when this sampling should occur.

A2.4 Core Incubation Conditions

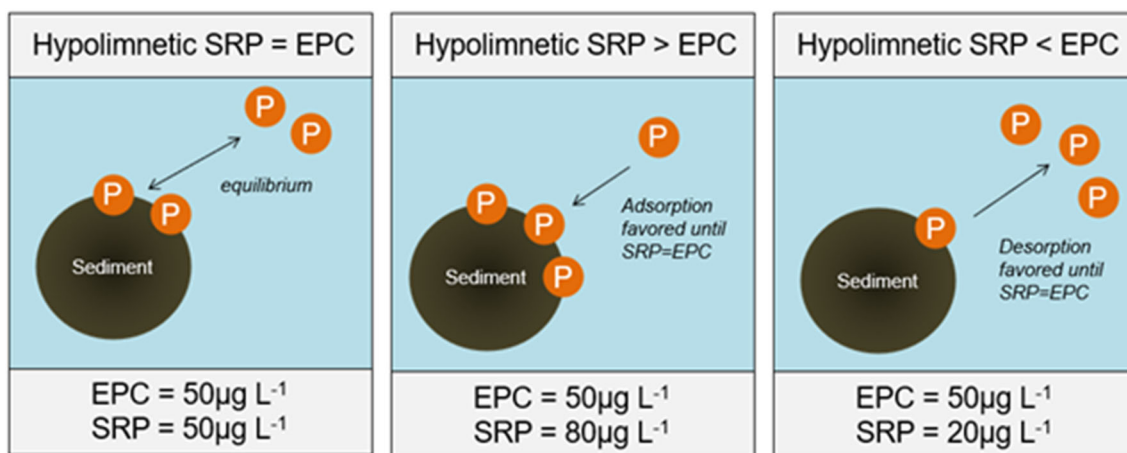
Ex situ sediment core incubations allow for various experimental treatments that may be designed to test how internal loading rates respond to different chemical conditions or management interventions. For example, a common treatment is to expose replicate cores to either oxic or anoxic conditions in the overlying water to measure sediment P release rates under these opposing conditions. These rates may

then be used to calculate expected internal P loads when the oxygen is available at the sediment-water interface versus periods of bottom water anoxia. Although an oxic versus anoxic treatment may be used on replicate cores from the same sampling site and location, we recommend mimicking ambient temperature and dissolved oxygen conditions in the bottom waters at the time of sediment core collection and then sampling during different times of the year to capture oxic and anoxic conditions (e.g., Albright and Wilkinson 2022b). Experimentally manipulating dissolved oxygen concentrations in a core incubation may fail to capture antecedent conditions that influence sediment P release in the ecosystem. Collecting sediment cores at different times and then maintaining ambient conditions during the core incubations may provide a more reliable measurement of sediment P flux rates under anoxic versus oxic conditions. In addition to testing phosphorus flux rates under different dissolved oxygen conditions, sediment core incubations may be used to test ALUM or other capping treatments, including efficacy and dosage requirements. General recommendations for sediment core incubations include using replicate cores and short incubation periods (less than 1 week) to avoid bottle effects (Ogdahl et al. 2014).

A3. DETERMINATION OF EQUILIBRIUM P CONCENTRATION (EPC)

Equilibrium P concentration (EPC) is a simple method to quantify sediment P sorption capacity and the likelihood of sediment P release through desorption processes. EPC refers to the concentration of aqueous-phase inorganic P (i.e., phosphate, measured as soluble reactive P or SRP) at which the rates of adsorption and desorption of phosphate molecules to and from sediment particle surfaces are equal. EPC may be used to estimate whether resuspended sediments are likely to release P into the water via desorption.

The relationship between sediment equilibrium P concentration (EPC) and bottom water or hypolimnetic soluble reactive P (SRP) concentrations is illustrated in the figure below. When EPC and hypolimnetic SRP are equal, P will be in equilibrium between sediment surfaces and the water column, so no P will be released (left panel). When hypolimnetic SRP is greater than EPC, the adsorption of P from the water to the sediment particle will be favored, so resuspended sediments will take up P from the bottom waters (center panel). When hypolimnetic SRP is less than EPC, P will desorb from the sediment surface, so resuspended sediments will likely be a source of P to the water column (right panel).



Original illustration E. Albright 2022.

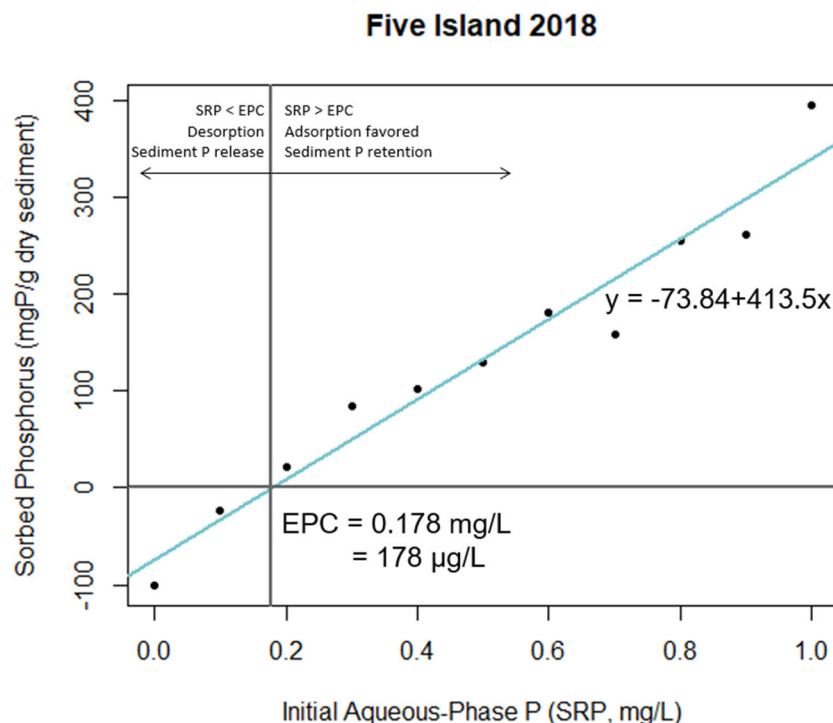
A3.1 Laboratory Procedure

1. For each sediment sample, label 8 clean, 50 mL falcon tubes with the equilibrium solution treatments (soluble reactive P, SRP, concentrations $0\text{--}1000\ \mu\text{g L}^{-1}$)
2. Sediment subsampling for each sediment core:
 - a. Thoroughly homogenize the sediment within the sample bag
 - b. Collect subsamples using a clean scoopula. For very fluid sediment, use a transfer pipette with the tip cut off. If working on multiple sediment cores, rinse the scoopula with DI water and dry with a kim wipe between cores.
 - c. Subsampling – place a small scoop of wet sediment into each of the 8 clean, 50 mL falcon tubes labeled with the concentration treatments
3. Add 40mL of the given equilibrium solution ($0\text{--}1000\ \mu\text{g L}^{-1}$) to each falcon tube using a graduated cylinder. Start with the most dilute solution and rinse three times with DI water between treatment concentrations. Note the volume and concentration of the equilibrium solution added to each tube.
4. Seal the falcon tube with parafilm and then the cap.
5. Shake horizontally on a shaker table overnight. Record the total time the samples were on the shaker table.

6. Centrifuge for 30 minutes at 3000 rpm.
7. Using a syringe, filter a 20 mL aliquot of each sample through a 0.45µm filter into a glass scintillation vial or a clean falcon tube.
8. Add 40 µL concentrated sulfuric acid to preserve the sample.
9. Run samples for SRP.
10. Determine the dry mass of sediment used in each treatment.
 - a. Transfer the remaining sediment pellet from each falcon tube onto a pre-weighed aluminum tin.
 - b. Dry to a constant mass at 105°C.
 - c. Record the dry mass of sediment used.

A3.2 Calculations

1. Change in SRP concentration: Initial SRP solution - final SRP concentration post shaking.
Units=mg/L
2. Calculate change in SRP as mass: Change in SRP concentration (mg/L) / volume of solution used (L). Units=mg
3. Calculate the change in SRP mass per gram of dry sediment. Units=mgP/g dry sediment
= Change in SRP (mg) / dry mass sediment used (g)
4. Plot the change in SRP as mass per gram of dry sediment (calculated in step 5) as a function of the initial SRP concentration in mg/L. The x-intercept of this regression estimates the EPC concentration.



A4. USING AUTOMATED SENSORS TO DIAGNOSE INTERNAL LOADING MECHANISMS

Deployment of high frequency sensors can provide insight into some of the mechanisms controlling internal P loading and the resulting consequences. However, these instruments can be costly. It is important to consider how to optimize sensor arrays to gain the most useful information about internal loading. Here we provide some best practices for sensor deployment and data analysis to ensure collection of the highest quality data and rigorous interpretation.

This document focuses on temperature chains, dissolved oxygen sensors, and multiparameter sondes. Temperature chains consist of temperature sensors (e.g., Onset HOB0 sensors) or thermistors hung in series in the water column, often attached to a rope. This collection of sensors can provide information about lake stratification and mixing dynamics. Dissolved oxygen sensors (e.g., PME Minidots) are useful for documenting dissolved oxygen dynamics within the lake. These sensors can provide information about dissolved oxygen drawdown in bottom waters or deployed in the surface waters to monitor ecosystem productivity. Multiparameter sondes are pieces of equipment that can house multiple sensors (e.g., YSI EXOs), housing two to six sensors and a wiper. Common parameters measured by these sondes include temperature, dissolved oxygen, specific conductance, chlorophyll, phycocyanin, and pH.

A4.1 Sensor Deployment – Temperature Chains

Temperature chains are composed of several temperature sensors attached to the same line at various depths throughout the water column. The number of sensors on the chain depends on lake depth and available resources. At minimum, a temperature chain should contain one thermistor at the water's surface and one near the maximum depth. Additional sensors placed between those depths can be used to monitor the depth of the thermocline in addition to mixing. These chains are usually best placed at the deep site of a water body, though in lakes with complex morphometry, additional chains should be deployed. Sensors should be set to record temperature frequently, usually every 15-60 minutes to capture mixing events and tie them to meteorological events. A high recording frequency is possible (e.g., 1 minute) but can introduce unhelpful noise into the data set and should be done only for specific questions that require higher frequency measurements. A higher measurement frequency will also result in a shorter battery life. If possible, we recommend programming each of your sensors to begin collecting data at the same point in time to ease data collation. Sensors should also be checked and intercalibrated by deploying them submerged in a bucket of water kept at a constant temperature.

To deploy the temperature chain, a simple buoy should be installed in the lake from which the chain can hang. A modest weight should be attached to the bottom of your chain to prevent lateral movement of the chain with wave action. Lake stratification changes throughout the year, so it is recommended that sensors are deployed for the entire open water season to gain an understanding of mixing dynamics.

A4.2 Sensor Deployment – Dissolved Oxygen Sensors

Dissolved oxygen sensors often accompany temperature chains. It is useful to have two dissolved oxygen sensors – one deployed in the surface waters and one in the bottom waters. However, if only one sensor is available and stratification is likely, deploying the dissolved oxygen sensor near the sediment-water interface is preferred when coupled with a temperature chain. Care should be taken to ensure that a bottom water sensor will not reach the sediments in periods of low water level, as this will not provide accurate readings of dissolved oxygen concentrations at the sediment-water interface. Like the temperature chains, these sensors are best deployed at the lake's deep site with measurements occurring

every 15-60 minutes. The concentration of dissolved oxygen in the environment can change significantly throughout the season, so deployment for the full open water period is recommended.

Many dissolved oxygen sensors use an optical sensing mechanism, making it important to keep the sensor face clear of biofouling. Particularly in eutrophic lakes, periphyton buildup can occur quickly. We recommend frequent (weekly to biweekly) cleaning of your sensors using a mild cleaner such as Simple Green to ensure the highest data quality. Wire brushes can be helpful for the cleaning of sensor housing but should not be used on optical sensor faces (instead, use a soft sponge). Some sensor types also come with available wiper attachments to clear the sensor face, which can be helpful in lakes with significant biofouling potential or long deployments without cleaning. The wiper attachments roughly double the cost of the dissolved oxygen sensor.

A4.3 Sensor Deployment – Multiparameter Sondes

Multiparameter sondes are useful tools to collect measurements of many parameters at the same place and time. These instruments are most frequently deployed from a buoy near the surface at the deep site. In addition to information about temperature and dissolved oxygen, these sondes can support an array of sensors measuring total algal pigments, conductivity, pH, and many other parameters that can provide useful information about your study system depending on your interests. Logging measurements less frequently (e.g., every 15-30 minutes) will save memory and battery; however, critical information on algal bloom dynamics can be gleaned from more frequent measurements (e.g., Wilkinson et al. 2018). Like the dissolved oxygen sensors, many of these sensors will need to be cleaned regularly for biofouling. Many sondes can be equipped with a wiper to aid in the cleaning process.

A4.4 Data Management

The amount of time your sensor can be deployed without data download or battery change is dependent on the sensor manufacturer and measurement collection frequency. While some sensors have the internal storage capacity to be deployed for months without downloading, we suggest a biweekly download frequency to limit the risk of data gaps due to sensor failure. The timing of sensor removal and deployment should be recorded so that the data can be trimmed to only include in-lake measurements.

Upon download, data should be stored with useful file names denoting the water body, dates of deployment, and the sensor type. We recommend collating and visualizing data with each download to ensure proper calibration and operation. A singular file with all the sensor data from a lake is most useful for analysis and visualization (therefore, consideration should be given to matching the measurement frequency and timing among instruments). However, always preserve the raw data files from the sensors in case of file corruption or loss.

A4.5 Visualization and Analysis

The purpose of deploying automated sensors is to track changes in the lake over time. As such, plotting the time series of variables is an important first step to understanding the data. It is particularly useful to compare the dynamics of meteorological, physical, chemical, and biological variables to each other to infer drivers of change.

Temperature chain data can be used to understand stratification dynamics. One particularly useful tool is to compare the topmost sensor reading with the bottommost sensor readings. When the difference between top and bottom temperature is more than 1°C, the waterbody can be considered stratified. The difference in top and bottom water temperature can be plotted as a time series and compared to dissolved oxygen dynamics and other variables. This time series can also be used to calculate the number of periods

of stratification and mixing and the duration of each event. This information will determine the mixing regime of the lake.

Once you have determined the mixing regime of your lake, it can be helpful to understand how the oxygen dynamics are changing throughout the lake in times of mixing or stratification. In particular, it is helpful to know how often and to what extent the bottom waters are anoxic. If a bottom water oxygen sensor is deployed, this information is straightforward to track. However, a surface water DO sensor can also provide insights to bottom water dynamics. After identifying when the water column is mixing (difference in top and bottom water temperature is $<1^{\circ}\text{C}$), observe how the surface DO concentrations change during this transition. If surface water concentrations decrease, it is likely that the bottom water DO concentrations were very low. If the surface DO concentrations do not change significantly, the bottom waters were likely oxygenated.

Specific conductivity is a useful index of water movement in most lakes. If specific conductivity decreases following a large precipitation event, this could be evidence that the waterbody was flushed and residence time changed. Increases in specific conductivity with mixing are indicative of ion-rich bottom water mixing to the surface.

A5. GLOSSARY OF MODELING TERMINOLOGY

Hydrodynamics: the flow or movement of a fluid (i.e., water). Temporal and spatial variations in the movement and distribution of a fluid (i.e., water).

1-dimensional model: a model that simulates variation in a given state variable (e.g., water temperature) along one spatial direction (e.g., water depth) as a function of time.

Timestep: the time interval at which the model calculation runs. The output at the end of each timestep is then used as the input for the next run. This modeling resolution depends on the resolution of the input data. For example, if the input data were collected every hour, then the model can run on an hourly time-step. If the input data were collected daily, the model can only run on a daily time-step.

Process-based model: a type of mathematical model that is based on a theoretical understanding of relevant physical, chemical, or biological processes. An example would be a model designed to not allow more dense water to exist above less dense water in a lake profile since we know this is physically impossible.

State variable: an amount of something that can change over time and that describes the state of a dynamic system (i.e., a lake ecosystem). Examples of relevant ecosystem state variables include water temperature; concentrations of nutrients, dissolved oxygen, chlorophyll, or population sizes of a plant or animal

Model calibration: an iterative process of comparing the model to observed data and revising the model if needed until the model error (compared to the observed data) is acceptably low.

Model validation: process of comparing the model output to observed data (not previously used for model calibration) from the system to evaluate the accuracy and validity of the model.