Best Management Practices Manual

Chapter 4 : Eutrophication

4.1 Introduction

Eutrophication is the process of increasing nutrient enrichment, especially phosphorus and nitrogen, in which the enrichment leads to phytoplankton blooms and deterioration of water quality and causes changes to the ecosystem (NRC 2000). Eutrophication of a reservoir and the resulting increases in phytoplankton growth can have various direct effects on the quality of water within the impoundment. Eutrophication increases phytoplankton, zooplankton, bacteria, fungi, and detritus. Phytoplankton production in the reservoir shifts from green algae to cyanobacteria (Smith 1998), which is a less desirable condition because cyanobacteria can produce undesirable tastes and odors as well as produce toxins (Figure 4.1). Cyanobacteria can out-compete green algae under low nitrogen to phosphorus ratios because of their ability to fix atmospheric nitrogen, they have nitrogen-storing heterocysts that help maintain buoyancy and thereby shade out other genera, and their ability to proliferate in hot and stagnant water, as well as other advantages (Scheffer et al. 1997). However, cyanobacteria may be less available as food for certain organisms (e.g., cladocerans) because of their larger size; also they may not provide sufficient nutrition (Smith 1998). As cyanobacteria blooms subside, the dead and decaying cells can reduce oxygen levels in the water, causing stress or death to aquatic animals and potentially prolonging stratification. Dissolved oxygen concentrations in eutrophic waters become more variable, reaching higher highs and lower lows (Bouvy et al. 1999).

Figure 4.1. Klamath River as it flows into the Copco Lake reservoir, California, and mixes with a cyanobacteria bloom. Photo credit: U.S. Environmental Protection Agency.

Although phosphorus and nitrogen occur in many different chemical forms in aquatic environments, it is the dissolved inorganic forms that are most readily available for assimilation by phytoplank- ton. Analyses of the inorganic species, ammonium (NH_4) and nitrogen oxides (NO_x), give reliable estimates of bioavailable nitrogen. Bioavailable phosphorus is more difficult to measure because of its high affinity to particles and because it is frequently used as soon as it enters the water column by algae and plants. Bioavailability of phosphorus varies depending on the source. Gerdes and Kunst (1998) showed that 72% of the total phosphorus in effluent from sewage treatment plants was bioavailable, but only 30% of the total phosphorus in eroded material entering a river was available. They also showed that this percentage increased to 59% when the soils from which the material was sourced were fertilized, suggesting that fertilizer introduced significant amounts of bioavailable material into the runoff.

Well-defined relationships between phosphorus and phytoplankton biomass (i.e., chlorophyll-*a*) have been identified in reservoirs (e.g., Hoyer and Jones 1983; Jones and Knowlton 1993). Consequently, phosphorus often has been considered the primary nutrient limiting phytoplankton production in reservoirs, and management ef- forts to control eutrophication generally have emphasized control of phosphorus loadings (e.g., Dodd et al. 1988). However, Elser et al. (1990) reviewed various studies and reported that co-limitation by nitrogen and phosphorus was a common response of phytoplankton to nutrient additions. In Kansas reservoirs, Dzialowski et al. (2005) reported that the addition of phosphorus or nitrogen alone rarely increased phytoplankton growth. Instead, growth was co-limited by both nutrients. Generally, reservoirs that were nitrogen limited had total nitrogen to total phosphorus ratios (TN:TP) <18; reservoirs that were co-limited by nitrogen and phosphorus had TN:TP between 20 and 46; and reservoirs that were phosphorus limited had TN:TP >65. Overall, these results suggested that in Kansas reservoirs management efforts might need to focus on both nutrients (Dzialowski et al. 2005).

In reservoirs, eutrophication is accelerated by a large watershed-to-lake-area ratio (Wetzel 1990). Croplands in the watersheds of reservoirs are usually the biggest contributors to eutrophication. The proportion of cropland cover in the watersheds of 126 Missouri reservoirs accounted for 60%–70% of the variance in long-term averages of total phosphorus and total nitrogen (Jones et al. 2004). Even among reservoir watersheds with >80% grass (including pasture) and forest cover, cropland accounted for most of the variation in nutrients. Reservoir nutrients showed a strong negative relation to forest cover. Relations between grass cover and nutrients were positive but weak, and grass had no detectable effect once the effects of croplands was taken into account. In this set of Missouri reservoirs, urban reservoirs had about twice the nutrient levels as reservoirs in forest and grass watersheds.

Back to top

4.2 Trophic State Indices

Indices of trophic state based on readily obtainable water-quality data are used to describe the trophic state of lakes but have been modified for use in reservoirs. Indices assign trophic states according to the phytoplankton biomass present during summer (indexed by chlorophyll-*a* biomass), the concentrations of key nutrients(phosphorus and nitrogen), and water transparency as measured with a Secchi disk (Table 4.1). This trophic classification of reservoirs results from the division of a trophic continuum into categories called trophic states. The trophic state of reservoirs is indicative of their biological productivity, that is, the amount of living material supported within them, primarily in the form of phytoplankton. The least productive reservoirs are classified as oligotrophic. These are typically deep and clear and have relatively low nutrient concentrations. The most productive reservoirs are classified as hypereutrophic and are characterized by high nutrient concentrations and shallow depth, which result in phytoplankton growth, cloudy water, and low dissolved oxygen levels.

Table 4.1. Trophic state classification based on total phosphorus, total nitrogen, chlorophyll-*a*, and Secchi depth visibility for lakes (Forsberg and Ryding 1980) and for reservoirs (Jones and Knowlton 1993) in parentheses.

Trophic state	Total phosphorus (ppb)	Total nitrogen (ppb)	Chlorophyll-*a* (ppb)	Secchi depth (ft)
Oligotrophic	<15 (≤10)	<400 (<350)	<3 (<3)	>13 (≥8.5)
Mesotrophic	15–25 (>10–25)	400–600 (≥350–550)	3 –7 (≥3–9)	8–13 (≥4–8.5)
Eutrophic	>25–100 (>25–100)	>600–1500 (≥550–1200)	≥9–40 (≥9–40)	3–8 (≥1.5–4)
Hypereutrophic	>100 (>100)	>1500 (>1200)	>40 (>40)	<3 (<1.5)

Back to top

4.3 Effects on Fish

Figure 4.2. Oxygen depletions and toxins associated with phytoplankton blooms in hypereutrophic lakes can cause fish kills. Photo illustrates a fish kill at Possum Kingdom Lake reservoir, Texas. Photo credit: NBC 5–KXAS, Dallas–Fort Worth.

Early stages of eutrophication may enhance fish growth and biomass and seem to be desirable from a fisheries perspective (i.e., more nutrients = more fish). However, water-quality changes associated with higher trophic states (e.g., hypoxia, denser phytoplankton blooms, reduced water clarity, and altered fish fauna) usually argue against promoting higher trophic states because of changes in fish food habits, spatial distribution, and community composition. In fact, extreme cases of hypereutrophication promote dense, noxious phytoplankton blooms that can cause fish kills (Figure 4.2). Moreover, phytoplankton communities in eutrophic reservoirs can shift from domination by green algae to potentially noxious cyanobacteria. While this dominance may shift seasonally in many reservoirs, cyanobacteria tend to dominate for an increasingly longer segment of the year in eutrophic and hypereutrophic reservoirs (Smith 1998) and are considered “sentinels” of eutrophication (Stockner et al. 2000). In turn, zooplankton composition is affected by phytoplankton availability because macrofiltrators (usually large-bodied zooplankton) that are more abundant in oligotrophic reservoirs give way to low-efficiency, small-bodied phytoplankton and bacterial feeders as nutrients increase (Taylor and Carter 1997). In hypereutrophic reservoirs, the food supply for zooplankton actually may decrease because of the dominance by cyanobacteria.

Eutrophication can lead to undesirable shifts in fish community composition. Although early stages of eutrophication may enhance fish growth and fishery yield, later stages may force changes in food habits, spatial distribution, and community composition (Larkin and Northcote 1969). In Florida lakes, fish biomass increased with eutrophication status to a maximum in mesotrophic lakes and fluctuated around the maximum value in hypereutrophic lakes (Kautz 1982). In contrast, fish density increased to a maximum in mesotrophic lakes but declined in hypereutrophic lakes. Piscivorous fishes reached maximum biomass and optimum densities in lakes with a total nitrogen concentration of 1,200 ppb and a chlorophyll-*a* concentration of 11 ppb but suffered adverse effects with further enrichment (Bachmann et al. 1996). Nevertheless, planktivorous (Yurk and Ney 1989; Bachmann et al. 1996) and benthivorous fish have been observed to increase with eutrophication status (Persson et al. 1991; Jeppesen et al. 2000).

Trophic state reportedly has a major influence on gizzard shad population characteristics in reservoirs (Power et al. 2004; Gonzalez et al. 2010). Gizzard shad represent an important prey species for many piscivorous fish. In oligomesotrophic reservoirs in Alabama, gizzard shad abundances were relatively low, yet these populations contained faster-growing fish (after age 1) and a higher proportion of older fish (DiCenzo et al. 1996). Furthermore, populations in less productive reservoirs contained fewer gizzard shad but a higher percentage of larger gizzard shad. In eutrophic reservoirs, gizzard shad were more abundant, and the population was characterized by smaller, slower-growing fish that were more vulnerable to predation. Consequently, gizzard shad were more available as forage in eutrophic reservoirs because of their greater abundance, smaller size, and slower growth, which made them vulnerable to predation for a longer period of time.

Concerted efforts by government agencies and private citizens to reverse cultural eutrophication (e.g., promoting or mandating the use of phosphorus-free laundry detergents, building more efficient wastewater treatment plants, agriculture best management practices) have ocassionally led to unwanted consequences. Specifically, nutrient loading rates into some reservoirs were reduced at a time when many reservoirs were experiencing decreased internal nutrient-loading rates and trophic depression in the decades following impoundment. Rates of nutrient loading and trophic states have changed so abruptly in a few systems that a new word entered the lexicon of reservoir and lake managers: *oligotrophication*. Moving from eutrophy to mesotrophy or from mesotrophy to oligotrophy usually results in clearer water because of reduced phytoplankton biomass, which most citizens equate to “cleaner” water. However, the trade-off between “clean water” and productive fisheries began to be discussed by fisheries biologists (e.g., Ney 1996; Stockner et al. 2000). The tight linkage between phytoplankton standing crops or phosphorus concentrations and fish biomass and sport fish harvest means that fisheries can suffer in reservoirs that shift to a lower trophic state.

The trade-offs between cleaner water and popular recreational fisheries were investigated by Maceina et al. (1996). They showed that modest shifts in trophic state could achieve clearer water while still maintaining good fisheries. A reduction in chlorophyll-*a* concentrations in eutrophic reservoirs in Alabama to 10–15 ppb was projected to increase water clarity and improve aesthetics for other recreational users without adverse effects to recreational fisheries. If a shift in trophic state was to occur from eutrophic to oligomesotrophic, catch rates of major recreational species was unlikely to shift, but in some instances smaller fish could result.

Back to top

4.4 Eutrophication Management

4.4.1 Monitoring Program

Identifying clear goals and means to achieve them is important in designing a eutophication monitoring program. Collecting data with vague goals and without a clear idea of how the monitoring data will achieve the goals rarely will produce good information. Possible goals include characterizing nutrient levels, identifying embayments where nutrients are excessive, determining whether management practices have forestalled eutrophication, and tracking water-quality changes or trends.

Monitoring eutophication can be complex and expensive and requires long-term commitments. Nevertheless, monitoring eutrophication may not always require field monitoring. Alternatives may include using existing data, partnering with another agency that is field monitoring, or documenting eutrophication with indicators that are not obtained through in-lake water-quality monitoring.

Whether or not monitoring is implemented, it is useful to gather and examine data from previous water monitoring in the reservoir. Often data are available from the reservoir controlling agency, from state water-quality agencies, from federal agencies such as the U.S. Environmental Protection Agency (USEPA) or U.S. Geological Survey, from local government agencies, or from universities. Existing data may not provide all the information needed but will help make an informed decision on whether it is necessary to implement a monitoring strategy and what information to target with monitoring.

One critical decision is what variables to monitor (Green et al. 2015). For monitoring eutrophication, obvious choices are total phosphorus, total nitrogen, chlorophyll-*a*, and Secchi depth. These variables have already been used to develop lake and reservoir classification schemes relative to trophic state (Table 4.1). Total phosphorus combines organic phosphorus (i.e., phosphorus bound to plant or animal tissue) and orthophosphate (PO4; inorganic form of phosphorus). Although only orthophosphate is readily available to phytoplankton or aquatic plants, other forms of phosphorus can be converted to orthophosphate. Therefore, total phosphorus is the most complete indicator of eutrophication status. Total nitrogen combines nitrate (NO3), nitrite (NO2-), ammonia (NH3), and organic nitrogen. Total nitrogen can be analyzed in one step or calculated from the sum of nitrate + nitrite + total Kjeldahl nitrogen. Chlorophyll-*a* is the photosynthetic pigment that yields the green color in phytoplankton and can be used as an indirect estimate of phytoplankton biomass in water. Chlorophyll-*a* is generally correlated with levels of phosphorus and nitrogen, although the correlation is not always strong because other variables also influence phytoplankton production. Secchi disk may be used to index eutrophication when suspended sediments are not a large component of suspended solids (section 5.4).

These four eutrophication metrics have daily and seasonal cycles that change regionally and locally depending on precipitation, land use, and local effects. Moreover, the peak of cycles may not coincide among these four metrics. Thus, standardizing the best time to conduct measurements over a large geographical scale is ineffective. Nevertheless, if there is a best time it may be when rainfall is low (usually late summer to early fall in the eastern USA).

Eutrophication monitoring of the nation's aquatic resources is conducted routinely by the National Aquatic Resource Surveys (NARS), run by the USEPA's Office of Water. A collaborative program between the USEPA, states, and tribes, NARS is designed to assess the quality of the nation's lakes and reservoirs through statistically-based survey designs. The NARS program provides nationally consistent data on the nation's waters, although the number of lakes and reservoirs included in the surveys are limited. The NARS database (NARS 2016) and the National Lakes Assessment Field Operations Manual (NLA 2012) are useful resources that can provide the foundation for developing local monitoring programs.

Back to top

4.4.2 Watershed Remediation

Most remediation techniques are directed toward reducing phosphorus flowing into reservoirs, moving nutrients through the reservoir with minimal retention, and neutralizing or removing nutrients already accumulated in the reservoir. Eutrophication control measures are aimed at reducing the levels of nutrients reaching a water body rather than treating the water body once a problem has occurred. Reduction of nutrients entering reservoirs from the surrounding watershed is a major emphasis of programs designed to control eutrophication. Reservoir managers can partner with watershed agencies and organizations (section 2) to reduce the amount of nutrients that enter the reservoir from point and nonpoint sources in the watershed.

When topographic conditions are favorable, bypass channels (section 3.7.1.3) may be constructed to route water around the reservoir during high-flow events with excessive nutrient concentrations. Infrequent large-flood events often contribute most to total loadings because of high concentrations and volumes (Morris et al. 2014). By routing nutrients around the reservoir into the tailwater, both nutrients and sediment accumulations are reduced. Conversely, bypass channels may be applicable during low flow when certain pollutants may flow in high concentrations.

Back to top

4.4.2.1 Constructed wetlands

Constructed wetland systems (Figure 4.3) are designed explicitly to incorporate the functions of natural wetlands to aid in nutrient removal from inflowing water (USEPA 2000b). Constructed wetlands also can provide for quantity control of inflows by offering a temporary water storage above the permanent pool elevation. As runoff flows through the wetland, nutrient removal is achieved by settling and by biological uptake. Constructed wetlands are among the most effective practices in terms of pollutant removal and also offer aesthetic value (Moshiri 1993). A sediment forebay can be constructed for removal of coarse sediment that could degrade wetland performance. Construction costs may be relatively higher than the sediment basins described in section 3.7.1.1.

Figure 4.3. Constructed retention cells and wetland complexes in the 4,700-ac watershed of Iron Horse Trail Lake reservoir, Nebraska. The inset in the top left shows Lores Branch approaching the complex. The com- plex slows down flows and allows expansion into wetlands to remove sediment and nutrients approaching the reservoir. 1 = channel training and maintenance access berms; 2 = sediment retention dikes; 3 = islands created from in-lake sediment spoils; 4 = wetlands. The top of the berms are lower than the top of the sediment retention dikes, and are designed to train the normal stream inflows to meander during normal flows and small rain events, but will be overtopped at high flows. Wetland vegetation will grow around the berms. During drawdowns the berms provide access for heavy equipment to remove deposited sedi- ment. Photo credit: M. Porath, Nebraska Game and Parks Commission, Lincoln.

The processes of nitrogen and phosphorus removal in wetlands are different. Plants uptake inorganic nitrogen and phosphorus (e.g., nitrate, ammonia, and soluble reactive phosphate) through their roots, foliage, or both during warm seasons and convert them into organic compounds (USEPA 2000a). The majority of these assimilated nutrients are released back into the water and soils when plants grow old and decompose during the cold season. Roughly 10%–50% of the nutrients remain stored in hard-to-decompose plant litter and becomes incorporated in wetland soils.

Nitrogen removal involves a large suite of bacteria that mediate or conduct numerous chemical reactions (USEPA 2000a). These bacteria are found on solid surfaces such as soil, litter, and submerged plants. The main transformation processes are ammonification (organic nitrogen to ammonia), nitrification (ammonia to nitrate or nitrite), and denitrification, by which nitrate (NO3) is converted to nitrogen gas (N2), which composes 85% of the atmosphere.

Denitrification is the dominant, sustainable removal process in wetlands that receive high nitrate loadings from agricultural runoff (Hammer 1989). Denitrification primarily is performed by bacteria that are heterotrophic, meaning they require a carbon source for growth and energy. Wetland plants are a key source of this carbon. Because denitrification is facilitated by bacteria, the process is temperature dependent. Higher rates of denitrification occur during higher temperatures when the bacteria are more active. Therefore, wetlands designed for nutrient removal work hardest at removing nitrogen during the summer.

Conversely, phosphorus is removed primarily through physical and chemical processes (USEPA 2000a). Phosphorus typically enters wetlands attached to sus- pended material such as small soil particles or as dissolved phosphorus (PO4). Particulate phosphorus is deposited in wetlands during sedimentation. The leaves and stems of vegetation help settle out particles by slowing the passing of water and al- lowing the particles to drop onto the substrates. The dissolved phosphorus accumulates quickly in sediment by sorption (to aluminum and iron oxides and hydroxides) and precipitation (to form aluminum, iron, and calcium phosphates).

There are several design variations of the constructed wetland, each design differing in the relative amounts of shallow and deep water and dry storage above the wetland. These designs include the shallow wetland, the extended-detention shallow wetland, and the pond–wetland system (Schueler 1992; Davis 1995).

In the shallow wetland design, most of the water-quality treatment volume is in the relatively shallow (≤1 ft) marsh depths (Hammer 1997). The only deep portions (3–5 ft) of the shallow wetland design are the forebay at the inlet to the wetland and a small pool at the outlet. One disadvantage of this design is that because the pool is very shallow, a relatively large area is typically needed to store large volumes of water.

The extended-detention shallow wetland design is similar to the shallow wetland (Schueler 1992). However, the wetland is designed to hold deeper water (>1 ft) temporarily so that water can be held for a longer period. This design can treat a greater volume of water in a smaller space than the shallow wetland design. Plants that can tolerate longer and deeper flooding, as well as dry periods, are desirable in this design.

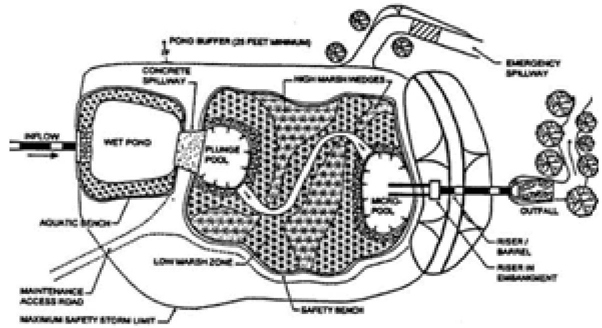

Figure 4.4. Shallow wetland schematic. Image credit: Center of Watershed Protection, Ellicott City, Maryland.

The pond–wetland system has two separate cells (Schueler 1992). These in- clude a sediment basin (section 3.7.1.1) and a shallow marsh (Figure 4.4). The sediment basin traps sediment and reduces runoff velocities prior to the water's entry into the wetland for additional treatment. Less land area is required for a sediment basin– wetland system than for the shallow wetland or the extended-detention shallow wetland systems. Access to the sediment basin is desirable to remove sediment accumulation periodically. According to Schueler (1992) approximately 70% of the volume should be deep storage and 30% marsh.

Constructed wetlands have a high suspended solids removal capability. Most wetland designs are able to remove roughly 50%–80% of the total suspended solids (Hammer 1989; USEPA 2000a, b). Removal of other pollutants is usually lower. Rough estimates of pollutant reductions derived from published data suggest that wetlands may reduce total phosphorus by about 30%–40%, total nitrogen by 20%–30%, fecal coliform by 50%–70% (if no resident waterfowl population present), and heavy metals by 40%–50%.

It may be beneficial to incorporate a cascade ponding system into the wetland layout to take advantage of an existing grade, provide depth diversity, and incorporate flow to provide aeration and increase oxygen levels in the water exiting the wetland. A cascade of wetlands would provide the ability to incorporate more than one wetland type to enhance different aspects of the overall treatment process (Kadlec and Wallace 2009).

Many reservoirs impounded over lowland rivers have extensive upstream floodplains associated with major tributaries. Wagner and Zalewski (2000) considered converting part of the natural floodplain of the Pilica River into constructed wetlands to trap phosphorus loads associated with major floods and normal flows. The river has an average discharge of 650 cfs and varies from 200 to 5,900 cfs. They estimated that for a reduction of 17%–27% of the total phosphorus load transported down to the 5,400-ac Sulejow Reservoir, wetland areas totaling 370 ac would have to be constructed in the floodplain upstream of the reservoir. These wetlands were estimated to average about 3-ft deep and extend about 15 mi upstream of the reservoir. They also predicted that if the wetlands had an area of

1,200 ac and a depth of 5 ft, the total phosphorus reduction achieved would be 21%–34%. In general, one large wetland may be more expensive to construct than many small ones, but a single wetland is easier and less expensive to operate and maintain (Hammer 1997).

Back to top

4.4.2.2 Pre-dams

Pre-dams are small reservoirs having a relatively low retention time, usually just a few days. They are constructed immediately above the main reservoir in one or more of the major tributaries (Figure 4.5). Their objective is to trap nutrients to reduce the load into the main reservoir. Effectiveness depends on retention time. Because nutrient uptake by phytoplankton and sedimentation are the focal processes for nutrient removal in pre-dams, they are relatively shallow, have a surface outlet, and are of a size appropriate to optimize nutrient uptake. Complete draining is required for periodic removal of sediment. Benndorf and Putz (1987a, b) describe a method for estimating optimum size for optimal nutrient removal.

Figure 4.5. Rappbode Reservoir, Germany with two pre-dams (lower left corner). Photo credit: Google Earth.

A pre-dam was constructed above Nielisz Reservoir, Poland, to improve water quality in the reservoir (Mazur 2010). Nielisz Reservoir is an impoundment of the Wieprz River. The reservoir has a watershed of 477 mi2, an area of 2,200 ac, and an average depth of 9 ft. The pre-dam impoundment has an area of 442 ac and an average depth of 2.3 ft. A survey in 2008 revealed seasonal reduction of the majority of water- quality indicators at the outflow of the pre-dam. Within the study period the level of total suspended solids decreased by 78%, phosphates concentration by 47%, ammonia by 37%, nitrates by

34%, total nitrogen by 24%, nitrites by 17%, and potassium by 15% (Mazur 2010). Nevertheless, there is limited information about the value of pre-dams and whether pre-dams may be applicable for very large reservoirs.

Back to top

4.4.3 In-Lake Remediation

Control of external sources may not be sufficient to return reservoirs to a de- sired state. In many cases the changes in the reservoir have been so dramatic—major shifts in biota, loss of habitat, physical changes in bottom sediment—that merely turning off the loadings is not sufficient to improve water quality and ecosystem performance. Therefore, in-lake restoration techniques may need to also be applied.

Back to top

4.4.3.1 Guide curve revision

Operation of reservoirs often is guided by a water management plan that outlines the level at which the reservoir will be maintained on a daily basis, therefore dictating retention and discharge. This plan usually is known as a rule or guide curve (section 7). Depending on the purpose of the dam, the guide curve may permit large annual water-level fluctuations and may have some flexibility for modification.

Operation of a navigation reservoir requires a relatively stable water level, and retention time varies little from that of the river. Conversely, flood control or storage reservoirs usually fluctuate greatly over the year; water is stored in spring, held in summer, released in fall, and allowed to move through the reservoir in winter. This regime results in a large fluctuation of retention times and, therefore, the extent to which nutrients entering the reservoir are allowed to settle in the reservoir. Changing the guide curve to adjust the residence time may reduce long-term eutrophication of the reservoir. One strategy may be to drop the water level during the high-inflow season to allow undesirable inflows entering the reservoir to be flushed through (section 3.7). A few months later the reservoir is refilled to normal pool, when the nutrient concentrations in the inflow water are typically lower. Another strategy may be to maintain a large pool with a greater retention time, which may allow suspended material to settle uplake and thereby reduce nutrients in the main reservoir. Models are available to estimate the effect of discharge rates on nutrient retention (Park et al. 2008).

Back to top

4.4.3.2 Inflow routing

Routing of undesirable major storm inflows is possible through control of discharges (WOTS 2004). Water from major storm events tends to have similar density and can be routed through the reservoir and past the dam to minimize nutrient and sediment settling within the reservoir. The inflow generally seeks and follows a layer of neutral density in a density-stratified reservoir, and thus a density current will develop and proceed through the reservoir. Because of the differences in density, currents can proceed toward the dam without mixing with most of the reservoir water. If the reservoir shape and bathymetry are highly irregular, with projecting features that can break up the flows, density currents may not sustain themselves. However, density currents occur in many reservoirs, and it is often possible to allow such currents to pass through the reservoir toward the dam. Existing outlets in the dam can then be operated to move the density current downstream. Selective withdrawal capability is required if inflow occurs at mid-depth levels. However, depending on the elevation of the inflow, it may be possible to use spillways, sluiceways, or other outlets to release the inflow (WOTS 2004). No structural modification or addition is involved, so costs are associated with only change of operation. This technique is applicable in small or large reservoirs, where other techniques may not be feasible (Kondolf et al. 2014). This technique is most applicable in reservoirs that stratify thermally (WOTS 2004).

Back to top

4.4.3.3 Dilution

Dilution efforts direct a low-nutrient water source into and through a reservoir as a way to dilute and remove nutrients from the high-nutrient impounded water. The additional flow may wash out surface phytoplankton and replace high-nutrient impounded water with lower-nutrient dilution water. A disadvantage is that dilution requires large volumes of low-nutrient water that may not be accessible or available.

Moses Lake reservoir, Washington, was diluted with low-nutrient water from the Columbia River during summer. Annual volumes and timing of dilutions were highly variable depending on water availability. Average turnover rate in the lake was 0.3% per day, and dilutions increased it to 0.4%–2.2% per day (Welch and Weiher 1987). Notable reductions in total phosphorus from about 150 to 50 ppb were recorded in various parts of the lake, particularly in the arm where dilution water was inflowing. The oligotrophication of Moses Lake from hypereutrophic to mesotrophic was accompanied by a marked shift in the fish assemblages (Welch 2009).

Back to top

4.4.3.4 Flushing

Flushing increases flow velocities in a reservoir to the extent that dissolved or suspended nutrients and nutrients concentrated in sediment are transported through low-level outlets in the dam (see section 3.7.3.6).

Back to top

4.4.3.5 Selective withdrawal

This method is applicable to stratified reservoirs, where the highest phosphorus concentrations have accumulated in the hypolimnion resulting from the strong release of phosphorus from sediment during anoxic conditions. The method relies on selective discharge of hypolimnetic waters (low in oxygen and rich in phosphate, iron, and manganese) from a reservoir (WOTS 2004) instead of discharge of water from the epilimnion, which often has lower nutrients. Hypolimnetic withdrawals are most effective if done without affecting stratification and thus avoiding the transport of nutrients and anoxic water from the hypolimnion to the epilimnion. Effectiveness is also increased when the discharged volume can be replaced by sufficient inflow to maintain the lake level relatively constant (Cooke et al. 2005).

The advantage of hypolimnetic withdrawals is the relatively low cost. A disadvantage is the discharge of cold water, nutrients, and other toxic compounds downstream. Whereas cold water may allow development of specialty fisheries, water may require aeration or other treatments. Mixing discharges with epilimnetic water may improve discharge water quality, although increase temperature.

Back to top

4.4.3.6 Hypolimnetic aeration and oxygenation

The basic concept of an aeration system is to maintain oxygen continually at the bottom of the reservoir so that phosphorus release from the sediment to the water column is reduced. The aeration also supports more rapid degradation of organic sediment by aerobic bacteria. Most commonly, aeration is achieved by compressors that introduce air near the bottom of the reservoir through perforated tubes. The rising bubbles push the anoxic water up to the surface where it is re-aerated with atmospheric oxygen. However, this method can break the stratified conditions in the reservoir and bring up nutrient-rich water to the epilimnion, which may trigger even more intensive phytoplankton growth. Additional details about hypolimnetic aeration and oxygenation are given in section 6.11.3.

Hypolimnetic aeration may not operate satisfactorily if the water body's maximum depth is <40–50 ft (Cooke et al. 2005). Aerators are usually turned on after the spring circulation and run throughout the summer until autumn circulation. Aerators also may be turned on during the winter under the ice cover if necessary. Hypolimnetic aeration has to be designed specifically for the conditions existing in a particular reservoir. Because of the need for a power source to operate the equipment, operation costs may be high, although solar systems are becoming available.

Hypolimnetic aeration and oxygenation is not always successful in controlling nutrients. An oxygenated hypolimnion does not necessarily assure that the sediment surface will be oxic enough to decrease phosphorus release sufficiently from the sediment. Also, in some cases diffusion of nutrients to the epilimnion from the hypolimnion has been observed even though stratification was maintained (Steinberg and Arzet 1984). Some side effects of aeration can be beneficial. Aeration allows zooplankton access to deeper, dark water that serves as refuge (McComas 2002). Additionally, the expanded aerobic environment can develop habitat for cold and coolwater fish.

Back to top

4.4.3.7 Sediment removal

Sediment removal through dredging or excavation (section 3.7.3) could be an effective method for reducing nutrient availability in reservoirs. The advantage of this method is that the results are long lasting. Removal of upper layers of the reservoir bottom sediment is most effective in shallow water, and the upper layer is where phosphorus is often most available for plant production. Removal of sediment also may remove cyanobacterial inoculum (Drabkova and Marsalek 2007). Sediment removal is probably most applicable in small reservoirs (<2,000 ac) or when limited to carefully chosen embayments (Peterson 1982; Eiseltová 1994; Cooke et al. 2005). The decision of whether the sediment will be removed or treated and left in place (see 4.4.3.9) depends on local circumstances, including sediment amount and quality, nutrient content, content of toxic compounds, availability of a disposal area, and possibility of re-use. Disposal of the dredged materials can be especially problematic. If the sediment does not contain toxic compounds, it can be used for agricultural purposes as a fertilizer. In special cases, the dredged sediment can be applied directly on agricultural fields (Pokorný and Hauser 2002).

Back to top

4.4.3.8 Sediment drying

Some reservoirs are characterized by a high degree of water-level fluctuations associated with their operational objectives (section 7). Deep portions of the reservoir may remain inundated under all but the most extreme drought conditions. Conversely, some of the shallower parts of the reservoir may be inundated for only a few days or weeks every few years. This wetting and drying can have a profound effect on the processes responsible for nutrient cycling in the regulated zone (i.e., the fraction of the reservoir dewatered in an annual cycle).

As sediment dries out, a decrease in bacterial biomass and activity is expected (Van Gestel et al. 1992; De Groot and Van Wijck 1993). It has been shown that bacterial activity declines linearly with soil water content (Orchard et al. 1992; West et al. 1992). At the extreme end of sediment desiccation a high bacterial mortality and release of nitrogen and phosphorus caused by cell lysis have been reported (De Groot and Van Wijck 1993; Qiu and McComb 1995), resulting in a flush of nitrogen and phosphorus upon rewetting of sediment. Thus, various studies have shown a net release of nutrients from sediment that has been exposed to air and subsequently rewetted.

Back to top

4.4.3.9 Phosphorus precipitation and inactivation

This technique focuses on lowering the reservoir's phosphorus levels by removing phosphorus from the water column and retarding release of phosphorus from sediment (Holdren et al. 2001). This is achieved by application of coagulants. These compounds, when added into the water, precipitate into flocculates. During flocculation, the phosphorus is bounded and converted into a form unavailable to phytoplankton. Some coagulants also can bind small particles, including phytoplankton cells, into the flocculates. The flocculates then settle to the sediment, thus removing phosphate and cyanobacteria from the water column. At the bottom of the reservoir, the coagulum further increases the binding capacity of sediment for phosphorus (Holdren et al. 2001).

Binding of bioavailable phosphate into flocculates is stronger than binding of phosphorus in particulate form (e.g., organic matter). Therefore, this treatment works best when applied to reservoirs with long retention time during late fall to early spring, when free phosphate is highest before it is incorporated into intensively growing phytoplankton (Holdren et al. 2001). Interference with the binding process occurs in shallow reservoirs overgrown by macrophytes and when external loading exceeds the phosphorus binding capacity of the flocculate (Welch and Cooke 1995). Effectiveness of this treatment can be low in shallow reservoirs where wind and waves resuspend phosphorus in the sediment. Effectiveness increases in small reservoirs with long retention time when the major phosphorus input is from the sediment (Cooke et al. 2005).

Various compounds are available to use as coagulants, including aluminum, iron, calcium salts, and clay materials. The compounds vary in their effectiveness and are described below.

Aluminum.-- The most commonly used aluminum coagulant is aluminum sulfate (alum, Al2(SO4)3·14H2O). When added to the water, alum quickly forms large, visible, nontoxic precipitates of aluminum hydroxide that settle to the sediment. Alum is extremely effective in controlling sediment phosphorus release rates, improving water clarity, reducing phytoplankton biomass, shifting population species composition from cyanobacteria dominance toward bacillariophytes and chlorophytes, increasing daphnid (Cladocera) biomass, and increasing usable fish habitat (Jorgensen et al. 2005).

To remove not only dissolved phosphorus successfully but also particulate phosphorus and to provide sufficient inactivation of sediment phosphorus, the goal is to apply as much alum as possible consistent with environmental safety. Several procedures to estimate a proper dose are suggested by Cooke et al. (2005) and are based on determination of mobile inorganic phosphorus in the sediment (Rydin and Welch 1998; Reitzel et al. 2005), estimated rates of phosphorus internal loading from sediment (Kennedy et al. 1987), or lake water alkalinity (Kennedy and Cooke 1982). The inorganic phosphorus is removed more effectively than particulate organic phosphorus (cells, detritus), suggesting that the most effective timing of alum treatment would be in early spring when the content of soluble phosphorus is highest. On the other hand, coagulation is re- duced at low temperatures. Treatments in early summer before cyanobacterial blooms occur are reportedly successful (Cooke et al. 2005). Alum treatments are often admin- istered to only sections of reservoirs (Figure 4.6), but when applied to a whole reser- voir, treatments are spread over several days, allowing organisms that are affected by the doses to escape to untreated areas (Cooke et al. 2005).

Figure 4.6. Applying alum at a Ne- braska reservoir. Photo credit: M. Porath, Nebraska Game and Parks Commission, Lincoln.

Barko et al. (1990) reported on the effects of a hypolimnetic alum treatment on sediment phosphorus availability in the 136-ac Eau Galle Reservoir, Wisconsin. Alum treatments over 5 years at batch doses of 100 lb/ac resulted in a substantial reduction in hypolimnetic total phosphorus and internal total phosphorus loading during the study. However, the frequency of major external total phosphorus loading events during that year (i.e., major precipitation events) negated the effectiveness of alum treatment in reducing epilimnetic total phosphorus mass as it remained essentially unchanged from pretreatment years.

Morency and Belnick (1987) reported on alum treatments at two relatively small (110 and 370 ac) and shallow (6 and 8 ft mean depth) hypereutrophic lakes in Oregon. Both lakes were treated with liquid alum at a dosage of 10.9 ppm based on their similar alkalinities (80 and 88 ppm as $CaCO3$). This strategy allowed the highest alum application without decreasing pH below 6. In the smaller lake this treatment resulted in a dramatic reduction in total phosphorus from a mean summer concentration of 115 to 26 ppb and chlorophyll-*a* concentration from 58 to 5 ppb; water clarity increased from 5.6 to over 8.2 ft. Changes were also noted in the larger lake, although the changes were not as pronounced as in the smaller lake.

Alternative methods for alum applications have been developed (Harper et al. 1998). Continuous alum injection involves a flow-weighted alum dosing system designed to fit inside a storm sewer manhole. This method allows treatment of storm-water runoff (point sources). Continuous alum treatment is typically most applicable in unstratified lakes with short retention times to remove nutrients and sediment from the incoming waters at or near the lake inlets. It is also most applicable in reservoirs for which the locations of all the major stormwater inputs are known. Because of high installation and operation costs, alum injection is best applied to situations in which a large volume of water can be treated. To increase the efficiency and cost effectiveness, alum dosing may be designed to occur only during storm-flow conditions when nutrient and sediment concentrations are elevated to problematic levels. Alum dosing may not be necessary or may be reduced significantly during base-flow conditions when nutrient and sediment inputs are generally low.

There has been debate about the safety of alum to humans and the aquatic environment, particularly within the North American Lake Management Society (NALMS). The current position of NALMS is that alum is a safe and effective lake management tool, but that alum applications should be designed and controlled to avoid concerns with toxicity to aquatic life. Moreover, NALMS considers watershed management as an essential element of protecting and managing lakes. In cases in which watershed phosphorus reductions are neither adequate nor timely, alum is an appropriate tool to accomplish meaningful water-quality objectives (NALMS 2004).

Iron.-- Iron is applied usually in the form of FeCl3 (iron chloride), but FeCl2 or Fe(SO4)3 (iron sulfate) also may be used. In contrast to alum, the stability of iron flocculates is less dependent on pH, and iron does not appear in toxic form. Nevertheless, the sorption to Fe(OH)3 (iron hydroxide) is greatest at pH 5 to 7, which is not common in eutrophic lakes especially if high phytoplankton densities are present. As with alum treatment, hydrogen ions are released, which may lead to a significant decline in pH and toxic effects to fish if pH levels decline below 6 (Søndergaard et al. 2002).

Further, the stability of Fe-P compounds is strongly dependent on changes in the redox state. As the dissolved oxygen in water above sediment drops below 1 ppm, iron is used as an alternate electron acceptor. Reduced ferrous ion (Fe2+) is soluble, and iron-bound phosphorus is released. This change occurs rapidly, so that even brief periods of anoxia at the bottom of the reservoir lead to substantial phosphorus release. To prevent this effect, aeration is usually applied along with Fe application. Continuous Fe application during summer has been used combined with artificial destratification to prevent cyanobacterial blooms (Deppe and Benndorf 2002).

Calcium.-- Calcium carbonate (calcite, CaCO3) or calcium hydroxide (lime, Ca(OH)2) can be added to water bodies as phosphorus precipitants (Neal 2001). Calcite sorbs phosphorus especially when pH exceeds 9.0 and results in significant phosphorus removal from the water column. Phosphate adsorbs at the calcite surface or binds inside a crystal during CaCO3 formation when calcium hydroxide is applied (Kleiner 1988; House 1990). Various calcite forms have been reported for potential use as active barriers in sediment caps designed to reduce phosphorus release from sediment (Hart et al. 2003).

The described application doses of lime are in a range of 25–300 ppm as Ca (Søndergaard et al. 2002). The advantage of lime is its low price and nontoxicity. However, adverse effects to aquatic organisms may occur because application of lime in- creases pH (Miskimmin et al. 1995). In soft-water lakes pH easily may exceed 11 (Zhang and Prepas 1996). The lime treatment also temporarily increases turbidity. As an additional benefit, lime and calcite also may be used to precipitate cyanobacterial cells from the water column (Zhang and Prepas 1996).

Clay materials.-- A range of clay materials can be used to bind phosphate from water, including zeolites, modified clays, and kaolins (Moharami and Jalali 2015). PhoslockTM is a commercially available specially modified clay made from bentonite clay in which the sodium or calcium ions (or both) are exchanged for lanthanum. The addition of this element allows it to bind with phosphates to form rhabdophane and thereby remove phosphorus permanently from the water column. PhoslockTM was reported to bind phosphorus successfully in the Canning and Vasse rivers in Australia (Robb et al. 2003). PhoslockTM applications require no buffer to protect water quality and aquatic life during and after application. Clay substrates with high phosphorus-sorption capacity may improve sustained phosphorus removal in wetlands (Mateus-Dina and Pinho-Henrique 2010).

Back to top

4.4.4 Biomanipulation

4.4.4.1 Fish populations

Excretion of nutrients by benthic-feeding fish assemblages can be a substantial fraction of nutrient inputs, comparable to external loading or nutrient release from sediment (Schaus et al. 1997). Thus, eutrophication control may be more effective if it considers both external inputs of nutrients and the translocation of nutrients from the reservoir sediment to the water.

Biomanipulation refers to employing the service of secondary or tertiary aquatic producers to affect a community structure and ecosystem (Shapiro and Wright 1984). Limnologists traditionally have considered lake systems to consist of components linked through a unidirectional flow of influence from nutrients to phytoplankton and to zooplankton and finally to fish. Biomanipulation represents a shift in paradigm by considering the reverse effects. Thus, a reduction of planktivores and benthivores through predation by piscivorous fishes would be followed by an increase in the abundance of large zooplankton (predominantly cladocerans), an increase in water clarity, and a decrease in nutrient recirculation from sediment. As a consequence, the grazing pressure on phytoplankton by zooplankton is enhanced, bottom stirring is reduced, and bottom nutrients remain less disturbed because of the lack of stirring and the lack of recirculation of nutrients through feces. In theory, the reduction of planktivory or benthivory may be achieved either by managed removal of the zooplanktivorous and benthivorous fishes or by increasing predation by creating an abundant piscivorous fish community via stocking, introductions, or protection regulations.

Thus, manipulation of fish populations, especially through artificial enhancement of piscivore populations, could be a useful method for reduction of phytoplankton levels and eutrophication. Biomanipulation has the potential to combine eutrophication management and sustainable fisheries management. The strategy may be particularly successful in those regions where commercial and recreational fisheries target a broad scope of species. Nevertheless, biomanipulation can be unpredictable as there are many unknowns about community interactions (DeMelo et al. 1992).

As an example of an unintended biomanipulation, increased piscivory by introduced Nile perch caused a shift in the ecosystem of Lake Victoria, Africa (Ochumba and Kibaara 1989). Cichlids grazed on the lake's plant community. After the introduction of Nile perch, cichlid populations were depressed, which reduced grazing, and eventually phytoplankton biomass increased. Concurrently, water quality deteriorated, as measured by increased phytoplankton turbidity and anoxia in deep waters.

In Round Lake, Minnesota, Shapiro and Wright (1984) reported applying rotenone to eliminate the lake's fish community. The lake was then restocked with bluegill and a high population density of largemouth bass and walleye. After restructuring the fish community, water transparency increased and chlorophyll-*a* decreased. Zooplankton densities decreased, but the mean sizes of zooplankton increased. The shift in zooplankton size is important because the filtering rate and the size range of edible phytoplankton increases with zooplankton size. Changes in zooplankton were not only responsible for the decrease in phytoplankton but also appeared to be responsible for a reduction in nutrient concentrations in the epilimnion. Assimilation of nutrients by zooplankton occurs primarily in the epilimnion at night, while nutrient excretion occurs throughout the water column, possibly resulting in a net downward movement of nutrients during diel migration. Although Shapiro and Wright (1984) achieved a reversal of planktivore effects by stocking piscivores, the beneficial effects lasted for only 2 years. After the initial reductions, the water transparency and chlorophyll-*a* concentration began reverting to their pre-biomanipulation condition because of an expanding bluegill population.

Back to top

4.4.4.2 Fish harvesting

In several lakes in Florida (e.g., Apopka, Dora, Griffin), biomanipulation programs have aimed to reduce nutrients by harvesting omnivorous gizzard shad (Schaus et al. 2010, 2013). These large-scale removals have reduced the biomass of harvestable (>12 in) gizzard shad by 40–60 lb/ac via subsidized commercial gillnet fisheries (Figure 4.7). These harvest rates represented about 75% of the harvestable gizzard shad. Given the size selectivity of the gear, the total population biomass of gizzard shad was reduced by <50% from an average pre-manipulation biomass. No major changes in total phosphorus or chlorophyll-*a* concentrations were detected following the biomanipulation. According to research in shallow lakes in the Netherlands, a biomanipulation must remove 75% of planktivorous and benthivorous fish before it can be successful (Hosper and Meijer 1993).

In practice, control of nutrients through biomanipulation and fish harvesting is not usually easy. Significant changes require substantial reductions in planktivores, often unachievable through fishing alone. Large reductions that rely on piscivores may be difficult to achieve in many reservoir communities because the prey communities are dominated by fish that grow beyond the reach of predators. For example, the adults of gizzard shad, perhaps the most abundant

Figure 4.7. Gizzard shad harvest pro- grams can remove nutri- ents from

and common planktivore/benthivore in reservoirs of the eastern USA, grow to a size not available to gape-limited piscivores. Moreover, the effects of biomanipulation do not always last because populations of other fish species with a similar niche, or the same population, may expand. Ecosystem interactions are complex and difficult to predict, so it is also difficult to predict the results of manipulating a biological community (DeMelo et al. 1992). Despite this lack of predictability, the potential benefits of biomanipulation (e.g., low cost, absence of chemicals or machinery, fishery development) make the technique attractive.

reservoirs. Photo credit: St. Johns River Water Man- agement District, Palatka, Florida.

Back to top

4.4.4.1 Macrophytes

Macrophytes in reservoirs can control nutrients and prevent development of phytoplankton blooms. Macrophytes reduce wind and boat-induced resuspension of nutrients. They also absorb some of the nutrients and support periphyton communities, which further remove dissolved phosphorus (McComas 2002; Cooke at al. 2005). However, growth of macrophytes is limited in many reservoirs because of wave action, water-level fluctuations, low water clarity, fish and other vertebrates uprooting or eating plants, and other disturbances (section 11).

Back to top

4.4.4.2 Floating wetland islands

Floating wetland islands are an emerging variant of constructed wetland technology that consist of emergent wetland plants growing hydroponically on structures floating on the surface of a pond-like basin (Headley and Tanner 2008). It is an artificial raft that houses native wetland plants (Figure 4.8). They represent a means of potentially improving the treatment performance of conventional pond systems by integrating the beneficial aspects of emergent macrophytes without being constrained by the requirement for shallow water wetlands. An island consists of emergent wetland vegetation growing on a mat or structure floating on the surface of a pond-like water body. The plant stems remain above the water level, while their roots grow down through the buoyant structure and into the water column. In this way, the plants grow in a hydroponic manner, taking their nutrition directly from the water column in the absence of soil. Beneath the floating mat, a hanging network of roots, rhizomes and attached biofilms is formed. This hanging root–biofilm network provides a biologically active surface area for biochemical processes as well as physical processes such as filtering and entrapment.

Figure 4.8. Floating wetland islands enhance removal of nu- trients, pro- vide habitat for fish and wildlife, and can enhance aesthetics. Photo credit: Cascade Meadow Wetlands and En- vironmental Science Center, Saint Mary's University, Roch- ester, Minnesota.

Research has shown that floating wetland islands can reduce nitrogen and phosphorus levels in ponds (Stewart et al. 2008). One unpublished study found 32% removal of phosphorus and 45% reduction in nitrogen in lake water used in a mesocosm experiment. This is relatively new, not fully developed technology, and mesocosms are small-scale experiments that may not transfer directly into larger water bodies.

Floating wetland islands have the potential to upgrade the water-cleansing qualities of sediment basins (3.7.1.1), although more research is needed. Sediment basins are generally effective at attenuating hydraulics and removing coarse suspended sediments but are less effective at removing nutrients and dissolved contaminants. These floating wetlands are anchored but can rise and fall as the water level changes. The water depth typically has to be a minimum 3 ft to prevent the macrophyte roots from attaching to the benthic substrate.

Back to top

Recent Projects

Improving the Fishery at Lake Nessmuk by the Addition of Critical Fish Habitat

Learn More

Thanks to Our Sponsor

LAKESIDE CITY TEXAS

Sign-up for our Email Newsletter

First Name

Last Name

Email

Sign-up

Donate

Friends of Reservoirs (FOR) is a coalition of local citizen groups dedicated to improving fish habitat in reservoir systems. FOR is a 501(c)(3) non-profit organization.

Donate

Contact Us

Friends of Reservoirs
Jeff Boxrucker, Coordinator
405-659-1797
jboxrucker@sbcglobal.net

Connect with Us