Manitoba Prairie Lakes: Eutrophication and In-Lake Remediation Treatments

Literature Review

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EXECUTIVE SUMMARY

This report is part of a broader project of the International Institute for Sustainable Development (IISD) working in partnership with the Province of Manitoba. Continuing work includes characterizing and comparing available lake chemistry and physical data for selected Manitoba prairie lakes, and a decision-support framework that includes in-lake remediation approaches in relation to characteristics of the lakes concerned.

Eutrophication causes pronounced deterioration of water quality and is a widespread environmental problem, one which affects the quality of many of Manitoba's prairie lakes. These lakes have deteriorated due to excessive loading of nutrients, organic matter and silt, which in turn cause increased primary producer biomass and reduced water quality. The understanding of eutrophication and its management has evolved from simple control of external nutrient sources to recognition that it is often a cumulative effects issue, requiring protection and restoration of various features of a lake's community and its catchment (Schindler, 2006). Anthropogenic activities are a major source of nutrient enrichment, where cyanobacteria blooms have afflicted many lakes internationally due to increased external nutrient loading causing eutrophication of water bodies. However, eutrophication is also the natural aging process for lakes and reservoirs, which involves an increase in nutrient concentration and sediment thickness. A lake may also be naturally eutrophied when situated in a fertile area with naturally nutrient-enriched soils. High internal loading of phosphorus from lake sediments is frequently reported as an important mechanism delaying lake recovery after a reduction of external loading.

To effectively assess remediation technologies feasibility, analysis of a waterbody's limnological and morphological parameters is necessary. Lakes are generally classified into four categories; oligotrophic, mesotrophic, eutrophic or hypereutrophic. Oligotrophic lakes are lakes with low nutrients and are biologically unproductive (Klapper, 1991). Mesotrophic lakes are an intermediate state of nutrient availability and biological productivity, while eutrophic lakes are characterized as nutrient-rich and highly productive. Hypereutrophic lakes are the extreme conditions of the eutrophic state. These lake trophic states correspond to gradual increases in lake productivity from oligotrophic to eutrophic, and can be classified by Secchi depth, chlorophyll-*a* (Chl-*a*) and total phosphorus (TP).

The success of a management technique varies greatly from lake to lake, where it is generally agreed that these technologies are usually not worth considering unless external nutrient loads can also be reduced and controlled. In-lake remediation techniques can be categorized by limiting and controlling the sediment, or by managing the consequences of lake aging. The techniques described are designed to control nutrients, plankton algae, and other related effects of over production and species composition changes that result from eutrophication. Algal biomass is dependent on the concentration of the limiting nutrient in the lake's photic zone; therefore, appropriate evaluation and modelling can determine the feasibility of controlling the primary sources of the most limiting nutrient. More than one technique may be used at once; however, for most in-lake techniques to be effective, important external loading sources should be evaluated and controlled. This review identifies multiple in-lake biological, physical and chemical treatments to limit and control P-enriched sediments and remediate the effects of eutrophication on lake water quality.

Table ES 1: In-lake remediation and restoration treatments

Biological & Ecological Engineering	Physical Engineering	Chemical Application
Biomanipulation Floating Treatment Wetlands Removal of Macrophytes	Hypolimnetic Withdrawal Dilution and Flushing Hypolimnetic Aeration and Oxygenation Artificial Circulation Dredging and Removal of Sediment	P Inactivation and Capping Sediment Oxidation Algicide

Biomanipulation

Grazing of algae by large zooplankton, particularly Daphnia, can be enhanced by eliminating planktivorous fish through physical removal or increased piscivory. Food-web manipulations have been relatively successful, although treatment longevity is limited.

Floating Treatment Wetlands

Wetlands rely on natural processes to biologically filter water as it passes through shallow areas of dense aquatic vegetation and permeable bottom soils. Floating treatment wetlands (FTW) are comprised of basins and cells to make an artificial platform containing emergent macrophytes. The primary mechanisms for nutrient removal are: microbial transformation and uptake; macrophyte assimilation, absorption into organic and inorganic substrate materials; and volatilization.

Removal of Macrophytes

Removing macrophyte biomass from lakes removes nutrients, which in some lakes can be a significant contribution to internal loading. Thick overstory, as well as decomposition of organic matter, contributes to oxygen deficiency and sediment phosphorus release, which can be alleviated by macrophyte removal.

Hypolimnetic Withdrawal

Nutrient-enriched hypolimnetic waters can be preferentially removed through siphoning, pumping or selective discharge instead of low-nutrient surface waters. Hypolimnetic withdrawal has been shown to accelerate phosphorus export, reduce surface phosphorus concentrations and improve hypolimnetic oxygen content.

Dilution and Flushing

Dilution involves the addition of low-nutrient water to reduce lake nutrient concentration and has been effective where external or internal sources are not controlled. Flushing refers to the removal of algal biomass.

Hypolimnetic Aeration and Oxygenation

Hypolimnetic aeration is highly effective at increasing dissolved oxygen in the hypolimnion without destratifying the lake. However, the treatment has not been demonstrated consistently as a technique to control algae.

Artificial Circulation

Artificial circulation is used to prevent or eliminate thermal stratification. Circulation can improve dissolved oxygen and reduces iron and manganese, cause light to limit algal growth in environments where nutrients are uncontrollable, and neutralize the factors favouring the dominance of blue-green algae.

Dredging and Removal of Sediment

Dredging can control both algae and macrophytes, and can restrict internal nutrient loading by eliminating the enriched sediment layer or sediments contaminated with toxic substances.

Phosphorus Inactivation and Capping

Internal phosphorus release is a significant source that could delay recovery of lake quality and can be controlled by additions of aluminum salts to the water column. The addition of alum results in an aluminum hydroxide floc which settles to the sediment surface, forming a physical barrier for further nutrient release. Additions of iron and calcium have also been an effective technique.

Sediment Oxidation

Sediment oxidation by enhanced denitrification results in improved complexation with iron and has been discussed separately from phosphorus inactivation techniques. The method reduces internal loading in lakes where iron redox reactions control P fluxes between sediment and overlying water.

Algicide

Commonly used in earlier lake remediation management and water supply reservoirs suffering from algal biomass, the addition of copper sulfate and other algicides is not frequently practiced due to significant detrimental aspects associated with the technique on the lake's biological community.

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GLOSSARY OF TERMS

C	Carbon
$CaCO_3$	Calcium carbonate
$CaOH_2$	Lime
CH_4	Methane
Chl-*a*	Chlorophyll-*a*
CO_2	Carbon dioxide
COD	Chemical oxygen demand
Cu	Copper
$CuSO_4$	Copper sulfide
DO	Dissolved oxygen
DS	Dissolved solids
dw	Dry weight
Fe	Iron
$FeCl_3$	Ferric chloride
$FeOH_3$	Ferric hydroxide
ft	Feet
ha	Hectares
H_2O	Water
H_2S	Hydrogen sulfide
kg	Kilogram
L	Litre
lb.	Pound
m	Metres
mg	Milligram
Mn	Manganese
N Nitrogen	(N_2)
N_{org}	Organic nitrogen
NH_4	Ammonium (NH_4-N)
Ni	Nickle
NO_3	Nitrate (NO_3-N)
O_2	Oxygen
OM	Organic matter
P	Phosphorus
Pb	Lead
pH	Hydrogen ion activity, aqueous solution
PO_4	Phosphate (PO_4-P)
SD	Secchi depth
SiO_2	Silicon dioxide
SRP	Soluble reactive phosphorus
SS	Suspended solids
SO_4^{2-}	Sulfate
TIN	Total inorganic nitrogen
TN	Total nitrogen
TOC	Total organic carbon
TP	Total phosphorus
TSS	Total suspended solids
wt	Weight
ww	Wet weight
yr	Year
Zn	Zinc
µg	Microgram

1. INTRODUCTION

Anthropogenic activities are a major source of nutrient enrichment and cause of eutrophication of water bodies. However, eutrophication is also a natural aging process for lakes and reservoirs and involves an increase in nutrient concentration and sediment thickness. A lake may also be naturally eutrophied when situated in a fertile area with naturally nutrient-enriched soils. Various countries have addressed the issue of eutrophication by implementing a range of technologies, legislative and biological measures, where approaches to improve lake quality internationally have been documented extensively during the past several decades.

1.1 Eutrophication

Eutrophication causes pronounced deterioration of water quality and represents a serious threat to the biotic components of the fresh water ecosystem, an issue for many lakes in Manitoba. According to the Organisation for Economic Co-operation and Development (OECD) (1982) eutrophication refers to

> "excessive nutrient enrichment of water which results in the stimulation of an array of undesirable symptomatic changes, such as nuisance production of algae and other aquatic macrophytes, deterioration of water quality, taste and odour problems and fish kills. Each of these changes significantly interferes with human use of water resources."

The main target of lake restoration is controlling the sources of nutrients that result in eutrophication. The excessive enrichment of waters with anthropogenic sources of nutrients leads to the transition of oligotrophic water bodies to mesotrophic, eutrophic and finally hypertrophic conditions. Phosphorus is the most important nutrient in regulating primary productivity in lakes and is a major contributor to eutrophication in aquatic systems. Eutrophication restricts water use for fisheries, recreation, industry and drinking due to increased growth of undesirable algae and oxygen shortage cause by algal decomposition. Eutrophication is still one of the most important problems to solve through lake management.

The success of a lake restoration depends on understanding physical, chemical and biological processes that occur within a lake and in the surrounding watershed. Criteria for assessing the eutrophication status of a waterbody were developed by Ryding & Rast (1989; Table 1.1) and include determination of major taxonomic groups, including phytoplankton.

Table 1.1: Criteria for assessing the eutrophication status of a waterbody

PARAMETER	UNITS[1]
Morphometric conditions	
Lake surface area	km^2
Lake volume (*average condition*)[2]	10^6 m^3
Mean and maximum depth	m
Location of inflow and outflow	-
Hydrodynamic conditions	
Volume of total inflow (including ground water) and outflow for different months	m^3/day
Theoretical mean residence time (*renewal time, retention time*)	y
Thermal stratification (vertical profiles along longitudinal axis, including the deepest point)	-
Flow-through conditions (surface overflow or deep release, and possibility of bypass flow)	-
In-lake nutrient conditions	
Dissolved reactive phosphorus; total dissolved phosphorus; and total phosphorus	µg P/L
Nitrate nitrogen; nitrite nitrogen; ammonia nitrogen; and total nitrogen	mg N/L
Silicate (if diatoms constitute a large proportion of phytoplankton population)	mg SiO_2/L
In-lake eutrophication response parameters	
Chlorophyll-*a*	mg/L
Transparency (Secchi depth)	m
Hypolimnetic oxygen depletion rate (during periods of thermal stratification and ice cover)	g O_2/d
Primary production	g C/m^3 d g C/m^2 d
Diurnal variation in dissolved oxygen[3]	mg/L
Major taxonomic groups and dominant species of phytoplankton, zooplankton and bottom fauna	
Extent of attached algal and macrophyte growth in littoral zone	

[1] The terminology and unit proposed by the International Organization of Standardization.
[2] A bathymetric map and hypsographic curve is necessary in many cases.
[3] Can provide additional information on the trophic condition of a waterbody.

Source: Ryding & Rast (1989).

1.2 Plankton Algae (Phytoplankton)

Eutrophic lakes are typified by the development of dense blooms of phytoplankton, and the goal of most remediation programs is to reduce algal abundance. In order to develop effective algal biomass control techniques, it is important to understand general characteristics of plankton algae and their interaction in the lake biotic community. Phytoplankton are maintained in the water column by wind-caused turbulence and many have evolved adaptations to resist sinking. Most of the phytoplankton are in the taxonomic orders Chlorophyta (green algae), Chrosophyta (diatoms, yellow-green, and golden brown algae), Cyanophyta (blue-green algae or cyanobacteria), Pyrrhophyta (dinoflagellates), Euglenophyta (euglenoids) and Cryptophyra (cryptomonads). In many lakes, maximum phytoplankton growth is limited by the availability of essential nutrients (especially phosphorus), light, and loss mechanisms, including sinking and grazing.

There is a typical seasonal cycle in the phytoplankton that usually begins with a spring diatom bloom in temperate waters. In oligotrophic lakes, the spring bloom consumes much of the available nutrient accumulated during the winter and spring runoff. With sedimentation of algal cells, nutrient content in the photic zone is depleted and phytoplankton growth slows, which is usually the result of nutrient limitation and increased zooplankton grazing. With minimal external loading during low summer stream flow and thermal stratification preventing vertical mixing of enriched hypolimnetic waters, nutrient content in the epilimnion often decreases, causing lower algal productivity and biomass. Green algae, desmids and yellow-green algae are an important part of the summer phytoplankton in oligotrophic lakes, whereas cyanobacteria are usually relatively insignificant.

In eutrophic lakes, the spring diatom bloom is usually succeeded by cyanobacteria to an increasing degree during summer. Their abundance depends on nutrient loading from external or internal sources. Generally, the higher the nutrient load and concentration, the greater the biomass and extent of dominance by cyanobacteria.

Cyanobacterial blooms and dominance are among the least desirable aspects of eutrophication, and the reduction of cyanobacteria blooms is a major goal of many remediation programs. Blooms of cyanobacteria are especially undesirable because they produce unsightly surface scums, generate noxious tastes and odours, and many taxa produce toxins that threaten the health of humans, wildlife, and aquatic biota. The collapse of cyanobacterial blooms often leads to severe oxygen depletion and, occasionally, fish kills. While there is universal agreement that cyanobacterial blooms are most likely to occur in systems with high-nutrient inputs, a complex array of additional factors are known to affect their predominance including the relative availability of nitrogen to phosphorus (Smith, 1983; Schindler et al., 2008), ferrous iron (Molot et al., 2014), dissolved inorganic carbon (DIC) (Shapiro, 1997), turbulence, high temperature, zooplankton grazing, and light (Paerl & Otten, 2013). There is poor understanding of how these factors interact in different lakes and at different times to produce cyanobacterial blooms.

1.3 Internal and External Nutrient Loading

Nutrient loading refers to the total amount of nutrients entering a water body in a given time and is one of the primary determinants of phytoplankton biomass. Nutrients can be derived from both internal and external sources. There are many external sources of nutrients, including natural inputs from rain and runoff, and anthropogenic inputs from diffuse sources such as fertilizer inputs and point sources such as sewage treatment plants. Internal loading usually refers to recycling of nutrients from sediments, primarily through redox reactions at the sediment–water interface and decomposition of settling organic matter (Smolders et al., 2006). Internal recycling is predominant in lakes that undergo stratification and hypolimnetic oxygen depletion.

Remediation of lakes from eutrophication should begin with the reduction of nutrient inputs from external sources. Otherwise, any improvement in the eutrophication of the lake will be short-lived and limited by utilization of nutrients and organic matter entering the lake from these external sources. High internal loading of phosphorus from lake sediments is frequently reported as an important mechanism delaying lake recovery after a reduction of external loading (Marsden, 1989; Phillips et al, 2005; Søndergaard et al., 2005; Welch & Cooke, 2005). A recent survey of long-term data from 35 lakes in Europe and North America concluded that internal release of phosphorus typically occurs for more than 10–15 years after the loading reduction (Jeppesen et al., 2005) and for some lakes, internal release last longer than 20 years (Søndergaard et al, 2003). Sediment has a profound role in the eutrophication of lakes, which is further amplified by the adsorptive and desorptive capacity of a large number of toxic elements and compounds. The ability of lake sediments to retain P depends on the physiochemical characteristics of the sediments and oxidation-reduction conditions at the sediment–water interface (Istvanovics et al., 1989; Bostic & White, 2007). Consequently, remediation of sediments must often be carried out to enhance lake recovery.

2. LAKE CLASSIFICATION

To effectively assess remediation technologies, an analysis of a waterbody's limnological and morphological parameters is necessary. Lakes are generally classified into four categories; oligotrophic, mesotrophic, eutrophic or hypereutrophic. Oligotrophic lakes are typically clear, cold lakes with low nutrients and few macrophytes with P concentrations less than 1 microgram per litre (Klapper, 1991). Mesotrophic lakes are in an intermediate state, while eutrophic lakes are characterized by high nutrient concentrations. Classification of those trophic states determined by Secchi depth, Chl-*a* and TP are defined in Table 2.1. Parameters for determining trophic state of a lake are summarized in Table 2.1; 2.2; 2.3; 2.4.

Table 2.1: Trophic state of surface waters

Status	Secchi Depth (m)	Chl-*a* (µg/L)	TP (µg/L)
Oligotrophic	>5	<2	<10
Mesotrophic	1.6-5	2-10	10-30
Eutrophic	0.7-1.6	10-30	30-60
Hypereutrophic	<0.7	>30	>60

Source: Ghosh & Mondal (2012).

Table 2.2: Parameters for determination of the trophic state of a lake

Parameter	Oligotrophic	Eutrophic
Occurrence of algal bloom	Rare	Frequent
Frequency of green and blue-green algae	Low	High
Daily migration of algae	Considerable	Limited
Characteristic algal groups	*Bacillariophyceae* *Pinnularia, Cymbella* *Chlorophyceae* *Chrysophyceae* *Synura, Chromulina*	Cyanophyceae *Microcystis, Nostoc*
Characteristic zooplankton groups	Represented by small size species: Cladocerans (*Bosmina*) Copepods	Represented by large size species: *Daphnia* (decreases in hypereutrophic)
Density of plankton	Low	High
Characteristics of fish	Finer variety of fish	Coarse fish
Depth	Deep	Shallow
Summer oxygen in hypolimnion	Present	Absent
Algae	High species diversity with low density and productivity often dominated by Chlorophyceae.	Low species diversity with high density and the productivity often dominated by Cyanophyceae.
Blooms	Rare	Frequent
Plant nutrient flux	Low	High
Animal production	Low	High
Fish	Finer variety of flux (e.g. carps)	Coarse fish (e.g. air breathers)

Source: Ghosh & Mondal (2012).

Managing lakes can be facilitated by lake classifications and the OECD Cooperative Programme related the classical limnological trophic terminology of oligotrophic, mesotrophic, eutrophic and hypertrophic, to specific in-lake water-quality parameters (Table 2.3). This open boundary classification system recognized the uncertainty in classifying a lake to a given trophic category by the integration of standard deviation associated with boundary ranges. Furthermore, biotic interactions are tightly coupled with and mediated by abiotic factors, and the OECD also organized trophic criteria and their responses to increased eutrophication, taking into account these response variables (Table 2.4).

Table 2.3: OECD boundary values for open trophic classification system

Parameter		Oligotrophic	Mesotrophic	Eutrophic	Hypertrophic
		(Annual mean values)			
Total phosphorus (µg/L)	x	8	26.7	84.4	
	x±1 SD	4.9–13.3	14.5–49	48–189	
	x±2 SD	2.9–22.1	7.9–90.8	18.8–424	
	range	3–17.7	10.9–95.6	16.2–386	750–1200
	n	21	19(21)	71(72)	2
Total nitrogen (µg/L)	x	661	753	1875	
	x±1 SD	371–1180	485–1170	861–4081	
	x±2 SD	208–2103	313–1816	395–8913	
	range	307–1630	361–1387	393–6100	
	n	11	8	37(38)	
Chlorophyll-*a* (µg/L)	x	1.7	4.7	14.3	
	x±1 SD	0.8–3.4	3–7.4	6.7–31	
	x±2 SD	0.4–7.1	1.9–11.6	3.1–66	
	range	0.3–4.5	3–11	2.7–78	100–150
	n	22	16(17)	70(22)	2
Chlorophyll-*a* peak value (µg/L)	x	4.2	16.1	42.6	
	x±1 SD	2.6–7.6	8.9–29	16.9–107	
	x±2 SD	1.5–13	4.9–52.5	6.7–270	
	range	1.3–10.6	4.9–49.5	9.5–275	
	n	16	12	46	
Secchi depth (m)	x	9.9	4.2	2.45	
	x±1 SD	5.9–16.5	2.4–7.4	1.5–4.0	
	x±2 SD	3.6–27.5	1.4–13	0.9–6.7	
	range	5.4–28.3	1.5–8.1	0.8–7.0	0.4–0.5
	n	13	20	70(72)	

The geometric means (after being transformed to base 10 logarithms) were calculated after removing values which were greater than or less than two times the standard deviation in the first calculation

x = geometric mean; SD = standard deviation; ( ) = the value in brackets refers to the amount of variables (n) used in the first calculation.

Source: Organisation for Economic Co-operation and Development (1982).

Table 2.4: Trophic criteria and responses to increased eutrophication

Physical	Chemical	Biological
Transparency [D]	Nutrient concentration [I]	Algal bloom frequency [I]
Suspended solids [I]	Chlorophyll-*a* [I]	Algal species diversity [D]
	Electrical conductance [I]	Phytoplankton biomass [I]
	Dissolved solids [I]	Littoral vegetation [I]
	Hypolimnetic oxygen deficit [I]	Zooplankton [I]
	Epilimnetic oxygen supersaturation [I]	Fish [I]
		Bottom fauna [I]
		Bottom fauna diversity [D]
		Primary production [I]

1. (I) signifies the value of the parameter generally increases with the degree of eutrophication; (D) signifies the value generally decreases with the degree of eutrophication.
2. The biological criteria have important qualitative (e.g., species) changes as well as quantitative (e.g., biomass) changes, as the degree of eutrophication increases.
3. Aquatic plants in the shallow, near-shore area may decrease in the presence of a high density of phytoplankton.
4. Fish may be decreased in numbers and species in bottom waters (hypolimnion) beyond a certain degree of eutrophication, as a result of hypolimnetic oxygen depletion.
5. Bottom fauna may be decreased in numbers and species in high concentration of hydrogen sulfide (H_2S), methane (CH_4) or carbon dioxide (CO_2), or low concentrations of oxygen (O_2) in hypolimnetic waters.

Source: Organisation for Economic Co-operation and Development (1982).

3. REMEDIATION TREATMENTS AND LAKE MANAGEMENT

Numerous lake restoration techniques have been developed and implemented during past decades to combat eutrophication. Success or failure of restoration technologies involves many factors; and it is accepted that permanent effects of restoration can only be achieved if external nutrient loading is reduced to sufficiently low levels (Sas et al., 1989; Jeppesen et al., 1990; Benndorf et al., 2002; Jeppesen & Sammalkorpi, 2002; Mehner et al., 2002). However, reductions of external nutrient inputs may take many years to be effective, and recovery is often delayed by continued internal loading of nutrients and other factors (Carpenter et al., 1999; Mehner et al., 2008). As a result, it is common to use multiple restoration techniques to speed lake recovery.

Early attempts to manage eutrophication largely involved treating symptoms, using copper sulfate or herbicides, rather than the source of the problem. Until the linkage of algal blooms with increasing nutrient supply due to human activities in the catchment of lakes was made, remediation technologies remained limited. Despite advances in the understanding of eutrophication, it still remains one of the foremost issues in protecting freshwater ecosystems. The understanding of eutrophication and its management have evolved from simple control of external nutrient sources to recognition that it is often a cumulative effects issue, requiring protection and restoration of various features of the lake's community and its catchment (Schindler, 2006). In many cases, liming, aeration, dredging, macrophyte harvesting, biomanipulation and other technologies have been applied to accelerate the recovery process by reducing internal loading or accelerating the removal of phosphorus (Cooke et al., 1993). Over the last several decades, a vast number of different techniques have been developed and tested to reduce eutrophication.

There are many different symptoms of eutrophication, and it is essential to determine management goals at the outset of any remediation project. For example, is the main purpose to reduce toxic algal blooms, increase hypolimnetic oxygen conditions for fish, or improve other conditions? Different lake restoration techniques vary in their effectiveness for achieving different goals and can be categorized based on the primary management objectives and the water-quality problems of concern.

The success of different technologies will vary greatly from lake to lake and will depend on many factors, including lake morphometry, stratification, sediment quality, flushing rates, and other factors. An essential first step is to characterize lakes of concern and to establish cause-and-effect linkages between water-quality concerns and proposed management techniques. This should include determinations of seasonal nutrient loading and concentrations, stratification, and sediment quality (Hickey & Gibbs 2009). Consultation with affected communities about the consequences of different proposed management techniques is also essential. Many remediation techniques will affect different aspects of water quality and aquatic biota, and clear identification of possible unintended effects is required.

Biotic interactions are tightly coupled with and mediated by abiotic factors. Interactions among biotic and abiotic influences need thorough consideration when selecting an appropriate management and remediation technology. Mechanical (physical) and chemical methods are the primary management procedures; however, biological control and watershed management are becoming more influential.

The treatments described are designed to control nutrients, plankton algae, and other related effects of overproduction and species composition changes that result from eutrophication. This review identifies multiple in-lake treatments (biological and ecological engineering, physical engineering and chemical application) to limit and control P-enriched sediments and remediate the effects of eutrophication on lake water quality. Common remediation treatment methods have been reviewed; however, it is not entirely exhaustive. In-lake remediation and restoration treatments included are as follows:

Table 3.1: In-lake remediation and restoration treatments

Biological & Ecological Engineering	Physical Engineering	Chemical Application
Biomanipulation Floating Treatment Wetlands Removal of Macrophytes	Hypolimnetic Withdrawal Dilution and Flushing Hypolimnetic Aeration and Oxygenation Artificial Circulation Dredging and removal of sediment	P Inactivation and Capping Sediment Oxidation Algicide

4. BIOMANIPULATION

Extensive research on biomanipulation (the manipulation of the biotic components and their habitats to facilitate certain interactions and results that are beneficial to ecosystem health) as a restoration technique for eutrophic lakes has developed an understanding of factors regulating distribution, abundance, productivity and species composition of phytoplankton. Biomanipulation was originally based on the concept of cascading trophic interactions in aquatic food webs (Shapiro, 1984; Carpenter & Kitchell, 1985; 1993). Trophic cascades are an important determinant of biomass at planktivore, herbivore and producer trophic levels, and can account for variance in regressions between resources and biomass. The abundance of phytoplankton reflects the balance between gains from growth and immigration, and losses from settling, flushing, and grazing. Hence, increases in grazing pressure (primarily from zooplankton) will often decrease phytoplankton abundance at a given level of nutrient availability. Grazer abundance is, in turn, often controlled by fish predation, and reductions in planktivory by fishing or the introduction of piscivores may be used to indirectly increase grazing pressure. Biomanipulation has been widely used in North America and Europe as a lake restoration technique (Reynolds, 1994; Hansson et al., 1998; Søndergaard et al., 2007; 2008; Jeppeson et al. 2003; 2007; Lathrop et al., 2002).

The use of biomanipulation for lake restoration has been extensively researched and has been the subject of many reviews (DeMelo et al., 1992; Reynolds, 1994; Hansson et al., 1998; Benndorf et al., 2002; Drenner & Hambright, 2002; Bell et al., 2003; Jeppeson et al., 2003; Søndergaard et al. 2007; 2008). It is now widely recognized that biomanipulation can result in a wide range of outcomes and that long-lasting results are rare. Notwithstanding the enormous amount of research on this topic, there is still uncertainty about the most common mechanisms behind successful biomanipulations and the causes of failures, and many explanations are non-exclusive. For example, many successes in European lakes may not be caused by a trophic cascade, but instead by changes in sediment resuspension caused by the removal of bream and other coarse fish (Søndergaard et al., 2008). If this is the case, biomanipulations targeting fish taxa that do not resuspend sediments may be less successful. In addition, reductions in fish planktivory may be offset by increases in invertebrate planktivory (Scheffer, 1999; Blumenshine & Hambright, 2003). Changes in phytoplankton densities may also be related to changes in inshore-offshore transport and nutrient recycling mediated by fish (Findlay et al., 2005).

The most successful biomanipulations have occurred in shallow lakes (Benndorf et al., 2002; Jeppeson et al., 2003). Shallow lakes are categorized by depths of less than 3 m, are usually polymictic and often have significant nutrient recycling affecting the entire water column (Cooke et al., 2005). Compared to deeper lakes, fish biomass per volume is higher, the impacts of fish on turbidity and sediment nutrient release are greater, and the area colonized by macrophytes may be close to 100 per cent (Moss et al., 1996a; Scheffer, 1998; Cooke et al., 2001). Unlike deep lakes, shallow lakes at moderate nutrient levels (30–100 μg/L) may exist in multiple states: clear with rooted plant dominance or turbid with phytoplankton dominance (Scheffer, 1998). Macrophytes (and associated periphyton) compete for nutrients and light in shallow lakes. Under conditions of high phytoplankton, algae outcompete macrophytes for nutrients and control macrophytes through reductions in light (Scheffer, 1999). By increasing light infiltration, macrophytes can become dominant and outcompete phytoplankton by sequestering nutrients, increasing shading, decreasing sediment resuspension and providing refuge for grazers. Macrophyte-dominant, clear water lakes are resistant to development of algal dominance from increased external nutrient loading because rooted plants and associated epiphyton compete for water-column nutrients, increase shading, reduce wind-induced turbulence and resuspension of sediments, provide daytime refuge to algae-grazing *Daphnia*, and some macrophytes release compounds inhibitory to algae. Piscivores thrive in macrophyte-dominated lakes, controlling fish that prey on zooplankton and on periphyton-consuming snails (Brönmark & Weisner, 1996). Successful removal of planktivorous fish is also more likely to be effective in shallow lakes. Fish removals, followed by piscivore stocking and enclosures to protect plants from birds, are among the biomanipulation procedures that often trigger a switch to a clear water state (Cooke et al., 2005). Furthermore, reduction of nutrient concentration through diversion or other methods can increase the probability of the switch.

4.1 Case Studies and Lake Response

Søndergaard et al. (2007) reviewed lake restoration projects carried out in Denmark and the Netherlands since the 1980s. In most, biomanipulation was the primary method (Table 4.1).

Table 4.1: Internal restoration measures and numbers of lakes (> 10 hectares) applied to combat eutrophication in Denmark and the Netherlands

	Denmark	The Netherlands
Fish removal (zoo- and benthivores)	42	14
Stocking of piscivores	34	4
Hypolimnetic oxygenation	6	3
Alum treatment	2	0
Sediment dredging	1	7

Source: Søndergaard et al. (2007).

Changes in summer means of TP, TN, Chl-*a*, Secchi depth and SS were used as key variables to evaluate the success of the restoration projects. For some of the Danish lakes, data on fish, macrophytes and zooplankton were also available. Most of the restored lakes in Denmark were shallow, eutrophic and relatively small. Dutch lakes were generally larger (mean 239 hectares versus 116 hectares in Denmark) and more eutrophic (mean TP 0.37 mg P/L versus 0.18 mg P/L in Denmark). The nutrient loading history of the lakes prior to the 1970s was generally undocumented but it was assumed that loading increased steadily (Gulati & van Donk, 2002; Søndergaard et al., 2003; Søndergaard et al., 2007). Following treatment, external loading to many lakes was reduced via chemical treatment of wastewater or by diversion of point source pollution (Søndergaard et al., 2007). Consequently, mean TP concentrations in rivers decreased in Denmark by 73 per cent from 1978 to 1993 (Jeppesen et al., 1999). From 1989 to 2002, TP and TN discharge from point sources to the Danish aquatic environment declined by 69 per cent (N) and 82 per cent (P) (Kronvang et al., 2005).

Table 4.2: Summary of lakes in Denmark and the Netherlands undergone lake restoration

Country	No. lakes	Min.	Mean	Max.	Min.	Mean	Max.
		Area (ha)			Mean depth (m)		
Denmark	41	2	116	941	0.8	2.6	13.5
Netherlands	28	1.5	239	3022	0.5	1.5	2.2
		Mean TP (mg P/L)			Mean Chl-*a* (µg/L)		
Denmark	41	0.05	0.18	1.25	21	80	208
Netherland	28	0.10	0.37	1.40	35	116	300

Source: Søndergaard et al. (2007).

In the >50 lakes subjected to biomanipulation, roach (*Rutilus rutilus*) and bream (*Abramis brama*) were the species most frequently removed, often supplemented by piscivore stocking (Table 4.1; 4.2). For most lakes, improvements were recorded immediately upon fish removal (Table 4.3). Secchi depth increased by more than 50 per cent in 14 of 20 lakes and Chl-*a* decreased in 8 of 21 lakes. TN and TP remained unchanged in most lakes and decreased in only 5 of 21 lakes. One lake of particular importance was Lake Vaeng, in which post-restoration data covered 18 years and showed significant changes in fish and macrophyte composition (Søndergaard et al., 2007). Immediately following fish removal, Chl-*a* and turbidity decreased and two years after the improved Secchi depth, submerged macrophytes colonized the lake (*Potamogeton crispus* followed by *Elodea canadensis*).

Table 4.3: Changes recorded after biomanipulation by fish removal

Parameter	Denmark				The Netherlands			
	n	nc	∧	∨	n	nc	∧	∨
Secchi depth	20	6	14	0	14	8	6	0
Chl-*a*	21	13	0	8	14	6	1	7
TP	21	15	1	5	14	9	0	5
TN	21	16	0	5	13	9	0	4
SS	17	13	0	4	-	-	-	-
Daphnia/cladoceran, number	14	7	7	0	-	-	-	-
Mean wt cladoceran								
(µg, dw)	13	9	4	0	-	-	-	-
Zoo:phyto (dw:dw)	12	7	5	0	-	-	-	-
Submerged macrophytes, coverage	12	10	2	0	13	8	5	0

Notes: Decrease or increase is defined as more than 50% change in summer means from before to after. Before is the mean of 1-3 year before biomanipulation; after is the mean of 1-3 years after.

N = number of lakes; nc = no changes; ∧ = increase; ∨ = decrease; zoo:phyto = the ratio between zooplankton and phytoplankton biomass; wt = weight; and dw = dry weight.

Source: Søndergaard et al. (2007).

4.1.1 Lake Zwemlust, the Netherlands

Lake Zwemlust (1.5 hectare, 1.5 m mean depth) is an example where issues occurred in maintaining the clear water state after biomanipulation. In 1968, broad-spectrum herbicide (diuron) was applied, eliminating macrophytes and correspondingly causing a shift to a turbid, algae-dominated state. A minimum transparency of 1.0 m in a swimming lake is required in the Netherlands, but blooms of *Microcystis aeruginosa* reduced transparency below this standard. In 1987, the lake was seined, electro-fished and drained to eliminate planktivorous and benthivorous fish. Following, the lake was stocked with pike (*Esox lucius*) and rudd (*Scardinius erythrophthalmus*) and *Daphnia magna* and *D. hyalina* (1 kg wet weight) were introduced (Gulati, 1990; van Donk et al., 1990).

By 1989, *Elodea nuttalli* dominated the lake and phytoplankton became N-limited. Summer Chl-*a* concentrations were low immediately following the treatment, partly attributed to intense *Daphnia* grazing. However, by 1991, small-bodied *Daphnia* became dominant after planktivory resumed and grazing on algae declined (van Donk & Gulati, 1995). Another planktivore removal took place in 1991, followed by a clear water period and slow macrophyte return. It was determined that herbivorous birds and rudd caused the shift back to the turbid water condition. High nutrient loading prevented the establishment of a permanent clear water state; only the turbid state appeared to be stable.

4.1.2 Lake Christina, Minnesota, United States

Lake Christina is a large (1,600 hectares), shallow (mean depth = 1.5 m) lake in west-central Minnesota. High chlorophyll and turbidity limited light diffusion to submersed aquatic plants, where by 1977–1980, plants and waterfowl were largely absent from the lake. Rotenone (3.0 mg/L) was applied in 1987 to eliminate the fish community, dominated by bullhead (*Ictalurus nebulosus*), bigmouth buffalo (*Ictiobus cyprinellus*), yellow perch (*Perca flavescens*) and northern pike (*Esox lucius)*. Largemouth bass (*Micropterus salmoides*) and walleye (*Stizostedion vitreum vitreum*) were then stocked to suppress reinvading rough fish. The lake was monitored from 1985–1998, making it one of the most well documented restoration and biomanipulation projects. The treatment was successful and long-lasting, switching the lake to a stable, clear water, macrophyte-dominated habitat (Hanson & Butler, 1994b; Hobbs et al., 2012).

4.1.3 Lake Mendota, Minnesota, United States

Biomanipulation of Lake Mendota (surface area 4,000 hectares; mean depth 12.7 m; and max. depth 25.3 m) included stocking walleye fingerlings (2.7×10^6) and northern pike fingerlings (1.7×10^5) from 1987 to 1999 to increase piscivory to levels that would reduce planktivore biomass, increase *Daphnia* grazing and consequently reduce algal densities in the lake (Lathrop et al., 2002; Karatayev et al., 2013). This biomanipulation project has been documented as successful due to the continued high densities and dominance of the large-bodied *D.pulicara* for over a decade and improved sport fishing for walleye and northern pike (Lathrop et al., 2002). Furthermore, reduced planktivory in the eutrophic lake noticeably did cascade to lower trophic levels, causing an increase in large *Daphnia*, reduced algal densities and increased water clarity. However, it is uncertain whether walleye and northern pike biomass directly controlled planktivory.

4.2 Effects and Precautions

4.2.1 Beneficial Effects of Biomanipulation on Lake Quality

The beneficial effects of biomanipulation on lake water quality have been described in detail for specific case studies in the previous section. To summarize: for most lakes, lake water-quality improvements included increased Secchi depth, and decreased turbidity, Chl-*a*, TP and TN concentrations due to changes in fish and macrophyte composition and abundance. Overall, the method is inexpensive relative to other remediation options, and does not require complex infrastructure or the use of potentially toxic chemicals (unless rotenone is used to remove fish). The introduction of piscivorous fish may enhance recreational fishing.

An assessment procedure was developed to estimate the probability of success with shallow lake biomanipulation (Hosper & Jagtman, 1990; Hosper & Meijer, 1993). Prior to application, the following criteria should be established (Hosper & Meijer, 1993; Perrow et al., 1997; Hansson et al., 1998):

1. Removal of planktivore and benthivore biomass should exceed 75 per cent in two years or less.
2. Piscivores should be stocked.
3. Immigration of new fish should be prevented.

4.2.2 Undesirable Effects of Biomanipulation on Lake Quality

As noted above, the success of biomanipulation projects has been extremely variable. Overall, there is limited data on the long-term responses of lakes to biomanipulation and most successful examples provide only data from the first 1–3 years (Bell et al., 2003; Benndorf et al., 2002). Biomanipulations have failed for many reasons, including insufficient numbers of fish removed, immigration by planktivores from other systems, increased planktivory by invertebrates, resistance to grazing by larger, toxic cyanobacteria, resuspension of sediments by wind, low piscivory by introduced fish, inedibility of many algae common to eutrophic lakes, replacement of fish predation by invertebrates (*Chaoborus*), overstocking of piscivores and long-term instability of the fish population and many other reasons (Table 4.4). Biomanipulations are often used in conjunction with other remediation measures to enhance lake recovery. Notwithstanding the enormous amount of research on this topic, there is still uncertainty about the most common mechanisms behind successful biomanipulation and the causes of failures.

Table 4.4: Internal reason for failure of biomanipulation in Denmark and the Netherlands

Treatment	Reason
Fish removal	Insufficient number of fish removed.
	Rapid return of strong cohorts of zooplanktivorous fish.
	Invertebrate predators (*Neomysis/Leptodora*) reduce the zooplankton.
	High resuspension rate of loose sediment.
	Internal P loading because of formerly high external loading.
	Instability due to low coverage of submerged macrophytes.
Pike stocking	Low survival of stocked fish due to predation.
	Low pike consumption of young-of-the-year fish.
	Poor timing of pike stocking relative to the hatching of young-of-the-year cyprinids.

Source: Modified from Søndergaard et al. (2007).

Drenner and Hambright (2002) reviewed methods and successes of biomanipulation experiments, where 80 per cent were in Europe and on small (< 25 hectare), shallow (<3 m mean depth) lakes. Success was high, at 61 per cent; however, the least successful was piscivore stocking. Partial removal of the fish population in shallow lakes was most successful but this was confounded by nutrient diversion, which occurred in 60 per cent of cases. A reason for the relatively low success of piscivores stocking is that planktivorous and benthivorous fish may grow to sizes beyond the piscivore mouth gape (Hambright et al., 1991).

There is considerable debate about the potential for successful biomanipulation and enormous variability in the likelihood of positive outcomes. While biomanipulation can be an effective lake management procedure, it is not a long-term substitution for controlling nutrient source and should be enhanced by additional lake management strategies.

4.3 Cost

Minimal cost analysis for biomanipulation has been developed. A recent feasibility study conducted by Barr Engineering for Twin Lake, Minnesota reviewed the cost of biomanipulation (Table 4.5). Due to fish migration between Sweeney and Twin Lake, the fish population was planned to be monitored at an annual cost of $1,000 per year (USD, 2013). Four stocking events were determined to be included to account for failure in initial stocking application, which amounted to $270,000 (four stocking events, monitoring, permitting and contingency [25 per cent]).

Table 4.5: Cost estimation for several remediation technologies for Twin Lake, MN

Treatment	Cost (USD)					
	Capital Cost	Engineering and Design (20%)	Contingency (25%)	Annual Operation	Permit	Total (20 years)
Biomanipulation	$216,000	-	$54,000	-	$3,000	$273,000
Hypolimnetic withdrawal	$400,000	$80,000	$100,000	$40,000	$10,000	$1,330,000
Alum treatment	$100,000	$20,000	$25,000	-	$3,000	$148,000
Aeration	$160,000	$32,000	$40,000	$35,000	$3,000	$935,000
Dredging	$1,700,000	$425,000	$425,000	-	$20,000	$2,570,000
Note: Biomanipulation assumes a total of four stocking events in years 1, 2, 4 and 6; and aeration treatment is a bubbler system design.						

Source: Barr (2013).

Biomanipulation Summary

Grazing of algae by large zooplankton, particularly Daphnia, can be enhanced by eliminating planktivorous fish through physical removal or increased piscivory. Food-web manipulations have been relatively successful; however, treatment longevity is limited. There is enormous variability in the likelihood of a positive outcome and uncertainty about the most common mechanisms behind successful and unsuccessful biomanipulation. Long-lasting results are rare.

Beneficial Effects

- Water-quality improvements included increased transparency, decreased turbidity, decreased Chl-*a*, TP and TN concentration.
- Generally, method is inexpensive.
- Does not require complex infrastructure.
- Does not require potentially toxic chemicals; however, chemicals such as rotenone have been applied.
- The introduction of piscivorous fish may enhance recreational fishing.

Undesirable Effects

- Resistance to grazing by large cyanobacteria.
- Replacement of fish predation by invertebrates (Chaoborus).
- Overstocking of piscivores.
- Long-term unsustainability of the fish populations.
- Nutrient transport by fish.
- Immigration by planktivores from other systems.
- Increased planktivory by invertebrates.
- Resuspension of sediments.

Treatment success is extremely variable and reasons for failure include:

- Poor timing of stocking.
- Inedibility of many algae common to eutrophic lakes (cyanobacteria).
- Insufficient numbers of fish removed.
- Low survival of stocked fish.

Suitable Lake Conditions

Lake size: In theory, there is no restriction on lake size, although lakes <25 hectares in size have had the highest percentage of success. Successful implementation in the literature: 1.5–240 hectares. One of the most famous biomanipulations was in Lake Mendota, WI (4000 hectares).

Depth: Greatest probability to reduce algal biomass, <3 m.
Successful implementation in the literature: 1.5–2.6 m.

P load: 1–14 kg/hectare/year (Olin et al., 2006).
Lakes with external P loadings below 0.6 g P $m^{-2}yr^{-1}$ have a higher probability for biomanipulation to reduce algal densities (Lathrop et al., 2002).

TP: Successful implementation in the literature 0.05–1.4 mg/L.
Recommend lake P conc. < 100 µg/L (Hanson, 1998).

Chl-*a*: 21–300 µg/L.
Successful implementation in the literature 80–116 µg/L.

Secchi depth: 0.9–2.9 m

Longevity: Enormous variability in success and multiple restocking events might be necessary.

Cost

Twin Lake, MN:
Capital cost: $216,000
Total cost (20-year lifespan): $273,000 (Barr, 2013).

Lake Nokomis, MN (walleye):
$12,700 per year, plan for 10 years.
Total cost (10 years): $127,000 (MCWD, 2014).

5. FLOATING TREATMENT WETLANDS

Past lake water-quality treatment and restoration has relied heavily on chemical, physical or mechanical treatment. An alternative to these treatments is the implementation of floating treatment wetlands (FTWs) and islands. Constructed wetlands have been recognized as effective mechanisms for water treatment. Wetlands rely on natural processes to biologically filter water as it passes through shallow areas of dense aquatic vegetation, and through permeable bottom soils (Stewart et al., 2008). FTWs work similarly to constructed wetlands; however, they are composed of basins and cells to make an artificial platform containing emergent macrophytes. The primary mechanisms for nutrient removal from FTWs are microbial transformation and uptake; macrophyte assimilation, absorption into organic and inorganic substrate materials; and volatilization (Stewart et al., 2008). Surface area is necessary for bacterial growth within a wetland, as the greater surface area allows for a larger bacterial community and therefore greater uptake of nutrients from the lake. Floating wetlands and islands have an increased surface area compared to conventional constructed wetlands. Floating wetlands differ in comparison to constructed wetlands, in that the microbes and macrophytes grow on and with the floating platform, and macrophyte roots extend into the water to absorb nutrients hydroponically (Stewart et al., 2008). The extension of the roots into the water column increases contact with nutrient-rich surface-flow water. Additionally, the roots from floating wetland macrophytes also provide an additional surface area to support the growth of microbes.

5.1 Case Studies and Lake Response

Several projects have documented the effectiveness of FTWs and islands (Boutwell & Hutchings, 1999; Hart et al., 2003; Hubbard et al., 2004; Stewart et al., 2008); however, comparison between floating wetlands is difficult due to the variability of application and monitoring. The results from four studies are summarized in Table 5.1. It is important to note that these studies involved tank-scale experiments, which would occur over shorter timelines than full-scale wetland applications.

Table 5.1: Nutrient removal results from selected studies

Nutrient	Removal Rate ($mg/m^2/d$)	Description	Reference
NO_3-N	111,100	Microbes measured	Biohaven TM Floating Islands
NO_3-N	5,597	Macrophytes, microbes and algae	Boutwell & Hutchings, 1999
TN	5,815	Vetiver grass on floating platform	Hart et al., 2003
NH_4-N	2,940	Microbes measured	Biohaven TM Floating Islands
TN	2,906	Total removed (plants, microbes, adsorption), tank scaling	Tanner, 1996
TN	2,745	Macrophyte uptake only.	Hubbard et al., 2004
PO_4-P	4,607	Microbes measured, aerated.	Biohaven TM Floating Islands
PO_4-P	560	Vetiver grass on floating platform (microbes and macrophytes)	Hart et al., 2003
PO_4-P	431	Total removal (plants, microbes, absorptions), tank scaling.	Tanner, 1996
PO_4-P	410	Macrophyte uptake	Hubbard et al., 2004
Note: $mg/m^2/d$ is milligram of nutrient removed per square metre of the island surface per day.			

Source: Stewart et al. (2008).

5.1.1 Eucha Lake, Oklahoma, United States

Eucha Lake, located in Oklahoma, has a surface area of 2,800 acres. The water quality for the lake was listed as impaired for Chl-*a*, TP and dissolved solids (DS) in 2008. At the beginning of the sample period in 2011, the lake's mean TP concentration was 0.04–0.06 mg/L in 2011 and 2012, as compared with Oklahoma Water Quality Standards criteria of 0.017 mg/L. It was determined that 93 per cent of TP loading to Eucha Lake came from the watershed, estimated as 43,314 kg/yr (38 kg/ha/yr). Following the floating wetlands installation, lake TP was reduced due to removal by plant biomass, estimated at 41 kg p/year (Kansas Department of Health and Environment [KDHE], 2013).

5.1.2 Pasco County. Florida, United States

CH2M Hill analyzed the effectiveness of floating wetlands to reduce nutrient levels to assist with meeting total maximum daily loads (TMDL). The area of the retention pond where the floating wetland treatment was installed was 1.6 hectares in size, with a depth of 1.2 m. The results of the study found that the removal of TN and TP was substantially higher with the use of floating wetlands in comparison to the control period. Furthermore, bacterial and plant nutrient removal processes were substantially enhanced during the treatment period. Net nutrient removal rates attributed to floating wetland treatment were calculated by subtracting the control removal from the treatment removal: 27 kg/m^3/year for TN and 08.7 kg/m^3/year for TP (Floating Island International, 2014).

Table 5.2: Floating treatment wetland effects on water quality

Parameter	Floating Wetland Treatment			Control		
	Inflow	Outflow	Removal (%)	Inflow	Outflow	Removal (%)
TN (mg/L)	6.10	2.04	67	4.47	3.44	23
TP (mg/L)	1.96	0.63	68	1.37	1.00	27
pH	-	9.96	-	-	11.25	-

Source: Floating Island International (2014).

5.2 Effects and Precautions

5.2.1 Beneficial Effects of Floating Treatment Wetlands on Lake Quality

Research has shown that constructed wetlands and floating islands significantly reduce aquatic pollutants, especially nutrients, without requiring costly maintenance (Stewart et al., 2008). Therefore, floating wetlands provide high nutrient removal rates and are a strong alternative to conventional treatments for reducing excess phosphorus, ammonia and nitrate concentrations. Floating wetland treatment technology is easy to implement, adaptable to a wide variety of plant species, and applicable in a variety of environments.

Bacteria and algae are important in P cycling in soils, the rhizosphere and water column (Vymazal, 2007). P uptake by microbes in conventional wetlands is rapid; however, little might actually be stored. Therefore, having a greater surface area and consequently higher microbial mass in a floating wetland provides a larger sink of P. Harvesting of floating island plants will increase permanent nutrient removal.

Table 5.3: Removal rates observed by White & Cousins (2013)

Nutrient (mg/m^2/day)	Inflow	Outflow	Removal (%)
TP	37.2	15.4	59
TN	320	106	67
Note: Trough 1.15 m^2 and 3.03 m^2 were used with 100% island coverage and soluble fertilizer added to pond water as inflow.			

In the study conducted by van de Moortel et al. (2010), approximately one 23 square metre FTW can remove roughly 4.5 kg TP per year. One kilogram of TP has the potential to generate up to 1000 kg of algal biomass; therefore, under the assumption that nutrients limit growth, one 220 square metre FTW can reduce algal biomass by approx. 5,000 kg per year. In addition to removing TN and TP, floating wetlands provide ancillary benefits of increasing wildlife habitat, reducing local nuisance insect populations and increasing water body aesthetics.

5.2.2 Undesirable Effects of Floating Treatment Wetlands on Lake Quality

Little to no adverse effects on lake quality have been mentioned in the literature; however, treatment could potentially affect N:P ratio. Traditionally, emergent plants have a higher need for N, which may decrease N:P and potentially promote the growth of cyanobacteria. Aquatic vegetation affect the physical and chemical conditions of the water. Specifically, respiration by plants and associated biota can reduce dissolved oxygen concentration, particularly during warm months. Low dissolved oxygen concentrations produce conditions fish may not tolerate; directly affecting respiratory processes and indirectly by impeding growth and reducing swimming speed (Sculthorpe, 1985). High lake surface coverage can potentially promote anoxia conditions and the percentage of surface area coverage by FTW must be considered in the application of this technology. It is important to note that there are potential limitations to the application of this technology. For maximum nutrient removal efficiency, the platform should be harvested or removed seasonally. FTWs occupy open water surface and can restrict access or reduce available area for recreational use. Minimum water depth should be no lower than 1 metres, and 1.5–2 metres is recommended because plants on the platform can root into sediments in shallow water, causing the floating platform to submerge.

5.3 Cost

Constructed wetlands and floating island systems can be complex, with multiple plant and microbial species occupying specific niches, where the size of the system is an indicator of its effectiveness. Cost estimates vary, dependent on various factors such as macrophyte species chosen, design (harvested or permanent), the size of the platform and the purpose (nutrient management, nursery production, habitat restoration, etc.). The size of the system will depend on the water quality goals and local climatic conditions. In general, larger systems involve higher construction, installation and maintenance costs. The cost of the floating wetland is proportional to the number and sizes of treatment cells or platforms required. Initial cost estimates for FTW platforms range from $11 to $260 per square metre (Virginia State University, 2013). Additionally, harvesting the above-ground vegetation can both enhance phosphorus removal and recovery, but also can serve to recover costs through the sale of biomass material for bioenergy production.

Biohaven Floating Treatment Wetlands (BFTW) was installed at an oxidation pond in St. Gabriel, Louisiana. The system, 145 square metres, was installed into the 2.1 hectare pond, providing 0.7 per cent top surface area pond coverage. Each platform was planted with approximately 70 individual plant plugs consisting of three species: Common Rush, Pickerelweed and Arrowhead. The total cost of the project was less than $40,000, which included installation, plants and monitoring for one year. In comparison, dredging was considered and estimated at a cost of $1,000,000 (Martin Ecosystem, 2013).

Floating Treatment Wetlands Summary

Wetlands rely on natural processes to biologically filter water as it passes through shallow areas of dense aquatic vegetation and permeable bottom soils. Floating treatment wetlands (FTWs) are comprised of basins and cells to make an artificial platform containing emergent macrophytes. The primary mechanisms for nutrient removal are microbial transformation and uptake; macrophyte assimilation; absorption into organic and inorganic substrate materials; and volatilization.

Beneficial Effects

- Relatively inexpensive compared to physical and chemical remedial treatments.
- Rooted macrophytes extract nutrients from both the sediment and the water column.
- Reduce redox potential and anoxic conditions.
- Harvesting platform plant material and the removal of biomass can further reduce nutrient concentration.
- Increase wildlife habitat.
- Reduce local nuisance insect populations.
- Increase waterbody aesthetics.

Undesirable Effects

- Little to no adverse effects on lake quality mentioned in the literature.
- Potential effects on N:P ratio, with effects on cyanobacterial growth.
- Potential to restrict access or reduce available area for recreational use.
- Potential for anoxic conditions with high lake surface coverage.

Suitable Lake Conditions

Floating wetland treatment suitable for a wide range of lake characteristics and water quality conditions. The size of the system is an indicator of effectiveness, where platform characteristics (design, size, macrophyte species) and specific lake characteristics (temperature, pH, TP, TN, Chl-*a*) will determine nutrient reduction/removal.

Lake size: Application successful and suitable to a wide range of lake size; however, most efficient on small lakes, ponds, small reservoirs and retention ponds.

Depth: Minimum water depth >1 m to prevent platform plants from rooting into lake bottom sediment. Ideal depth 1.5-2 m.

Longevity: With relatively low maintenance and secured placement, FTW will continuously sequester nutrient in the plant material. Harvesting material increases nutrient removal and longevity.

Cost

Cost determined by water quality goals and specific lake conditions, e.g., lake size.

FWT platforms range $11-$260 per square metre (Virginia State University, 2013).

Biohaven™ FTW installation in St. Gabriel, LA (2.1 hectare pond, 0.7 per cent surface area coverage)

$40,000: Installation, plants (70 plants) and monitoring for one year (2013 USD).

6. REMOVAL OF MACROPHYTES

Nutrient removal by harvesting macrophytes from a lake is frequently cited to improve water quality (Carpenter & Adams, 1978). Submerged macrophytes and their associated epiphyton use both aqueous and sedimentary nutrient sources, and sites of uptake (roots versus shoots) are related to nutrient availability in sediment versus the overlying water. During periods of active growth, macrophytes and epiphyton may be a net sink for nutrients, but following senescence they may become an important source mobilizing nutrients from the sediments to the water column (Carpenter, 1980; Landers, 1982). Internal nutrient loading from macrophyte senescence in many eutrophic lakes may be greater than external loading, particularly as external loading is reduced. The role aquatic plants play in internal nutrient loading is being increasingly appreciated and macrophyte harvesting may be a way of reducing internal nutrient cycling (Cooke et al., 2005).

Rooted macrophytes extract nutrients from both the sediment and the water column, where removing nutrients in plant biomass by harvesting may complement reductions from external nutrient sources (Carpenter & Adams, 1978). Rooted macrophytes usually achieve their phosphorus and nitrogen requirements directly from sediments, and the role of sediment as a source of P and N is ecologically significant because available forms of these elements are normally low in the open water during growing season (Barko et al., 1986). The parameters involved to calculate the potential for removing nutrients are as follows: the area of the lake covered with macrophytes (m^2), the average biomass of the plants in the area (g/m^2 per year, dry weight) and the nutrient concentration of the plants (g nutrient/ g dry weight of plants) as an estimate of the total nutrient available for removal (Burton et al., 1979). This value is reduced by the percentage of the total area harvested and the efficiency of the harvest. Furthermore, this value can be compared with nutrient loading to the lake to determine the percentage of the net annual loading that might have been or was removed by harvesting. It is also important to consider nutrient content of plant tissue variation by season, waterbody and species. In eutrophic lakes, even where nutrient loading is controlled, it still may take several years for harvesting to have an impact on nutrient concentrations of the lake (Carpenter & Adams, 1978; Burton et al., 1979).

6.1 Case Studies and Lake Response

6.1.1 Lake Sallie, Minnesota, United States

Harvesting of macrophytes in Lake Sallie, Minnesota, occurred each summer from 1970 through to 1972. Harvesting was halted in July 1973 due to low macrophyte yield (kg/ha), suggesting that successive harvests reduced plant biomass from year to year. Table 6.1 summarizes phosphorus removal by macrophyte harvesting for Lower Chemung Lake, Lake Sallie, Lake Wingra and East Twin Lake. Phytoplankton productivity also changed with harvesting, where the year prior to harvesting (1969) phytoplankton production was relatively high and typical of eutrophic conditions.

Table 6.1: Phosphorus removal by macrophyte harvesting

Parameter	Lower Chemung [a]	Sallie [b]	Wingra [c]	East Twin [d]
Surface area cover by macrophytes	430 ha	34%	34%	11.7 ha
Macrophytes harvested	18.7%	100%	100%	50%
Dry weight removed (kg)	2,500 metric tonnes	30,400	130,100	18,720
Mean tissue P conc. (% dw)	0.25	0.25	0.39	0.15
P removed by harvesting (kg)	560	100	580	28.1
Net annual P load (kg)	610	10,360	1,592	8.1-62
Percentage of net annual load removed by harvesting	92%	0.96%	36.4%	46-100%

Sources: [a] Wile et al. (1979); [b] Neel et al., (1973); [c] Carpenter & Adam (1978) (based on estimates of the nutrient pool); [d] Conyers & Cooke (1982).

6.1.2 Shagawa Lake, Minnesota, United States

Macrophyte harvesting failed to improve water quality in Shagawa Lake, Minnesota. Internal loading was greater than external loading, which wasn't taken into account in the initial restoration analysis. The release of phosphorus from lake sediment, which occurs as a part of internal loading, contributed a significant portion of the total P load to a lake. Submerged macrophytes, which use both aqueous and sedimentary nutrient sources, and sites of uptake (root versus shoot) are related to nutrient availability in sediment versus the overlying water. Macrophyte harvesting has been an effective controller of internal P loading; however, in the case of Shagawa Lake, macrophyte removal had negligible results due to high sediment P concentration.

6.2 Effects and Precautions

6.2.1 Beneficial Effects of Macrophyte Removal on Lake Quality

Over the long term, harvesting submerged macrophyte has the potential to affect nutrient cycling between the water column and lake sediments, depress photosynthesis with a decrease in pH and change oxygen levels (Cooke et al., 2005). Macrophyte decay accounted for approximately half of the P loading in Lake Wingra, Wisconsin. Asaeda et al. (2000) estimated that phosphorus released from decaying *Potamogeton pectinatus* could be reduced by at least 75 per cent by harvesting above-ground biomass. In some lakes, macrophytes are considered undesirable because they interfere with swimming, boating, and other recreational uses.

6.2.2 Undesirable Effects of Macrophyte Removal on Lake Quality

The environmental impacts of harvesting include immediate physical, and prolonged physical and chemical effects on biota and ecosystem processes. The treatment may not be sustainable, and the plant community may not be able to maintain the high biomass production needed for extensive nutrient removal over the long term.

Macrophyte harvesting directly removes fish, invertebrates and other species, where the magnitude of impact is variable. Engel (1990) estimated that between 11 per cent and 22 per cent of all plant-dwelling macroinvertebrates and more than 50,000 fish were removed from Halverson Lake (4 hectares) over two years of harvesting. Fish and invertebrates are associated with macrophytes that provide substrate, food, and a refuge from predators. Removal of all but a 2-m marginal macrophyte zone of River Great Ouse, United Kingdom, led to a rapid decline of mean cladoceran densities as a result of increased washout, fish predation and starvation (Garner et al., 1996). Fish common to the littoral zone are often considered desirable for fishing. It is recommended that harvesting should occur in the late summer or fall, at peak growth but prior to macrophyte senescence, when decay may release nutrients back to the water column.

6.3 Cost[1]

The cost of macrophyte removal is variable and dependent upon width of cut and harvesting method, frequency of harvesting, plant species and density, water depth and bottom obstructions. Korth et al. (1997) estimated the cost of harvesting for Browns Lake, Wisconsin. Based on equipment size (6.4 tonnes harvester) and the area harvested (60 hectares), the cost of the project was projected at $28,864 per year or $500 per hectare in 1997. That value equates to approx. $42,000 per year or $728 per hectare (2015). Cost of harvesting has also been cited as $200-$300 per hectare in 1979 (USEPA, 1979), with inflation (~3.25 per cent) would amount to approx. $650-$1000 per hectare.

[1] Treatment cost values that have been calculated to 2015 USD were based on inflation rates obtained by the United States Department of Labor CPI Inflation Calculator (Bureau of Labor Statistics, 2015). However, remediation treatments cost adjusted using inflation rates are estimates and do not incorporate technology cost changes. Remediation technology, methods and materials are moving targets, where various methods become less expensive, others more. Cost estimates adjusted by inflation rate should be considered as an approximation.

Removal of Macrophytes Summary

Removing macrophyte biomass from lakes removes nutrients, which for some lakes can be a significant contribution to internal loading. Thick overstory, as well as decomposition of organic matter, contributes to oxygen deficiency and sediment phosphorus release, which can be alleviated by macrophyte removal.

Beneficial Effects

- Extract nutrients from both the sediment and the water column.
- Long-term, harvesting macrophytes can affect nutrient cycling between the water column and the sediment.
- Increase waterbody aesthetics.

Undesirable Effects

- Immediate physical, and prolonged physical and chemical effects on biota and ecosystem processes.
- Directly and indirectly removes fish, invertebrates and other species from the ecosystem.
- Loss of habitat for grazers.
- Fish common in the littoral zone are often considered desirable for fishing.
- Reducing macrophytes decreases competition with algae and may even promote algal blooms.

Suitable Lake Parameters

Parameters involved to calculate the potential for removing nutrients:

- Area of the lake covered with macrophytes (m^2).
- The average biomass of the plants in the area (g/m^2 per year).
- The nutrient concentration of the plants (g nutrient/g dry weight of plant).

Successful implementation of hypolimnetic withdrawal as reviewed in the literature.

Lake size: 10–5,300 hectares.

Depth: 2.4–5 m (shallow lakes).

Longevity: Harvesting is continuous and a multiyear obligation for maximum affect in the long term.

Cost

Cost is variable and dependent upon width of cut and harvesting method, area harvested, plant species and density, water depth and bottom obstructions.

- $42,000 per year or $728 per hectare (2015, USD) to harvest 60 hectares (Korth et al., 1997).
- $550,000 per year: Chautauqua Lake, New York to harvest 5,300 hectares, 2,348 tonnes removed in 2014; Chautauqua Lake Association, 2015).
- Range in the literature $650–$1000 per hectare.
- Cost of a large system harvester $50,000–$200,000 (Aquamarine, 2014).
- Smaller harvesters (attached to a boat) are significantly less expensive.

7. HYPOLIMNETIC WITHDRAWAL

In stratified lakes, phosphorus, ferrous iron (Fe(II)) and ammonia often accumulate in hypolimnetic bottom waters as a result of the redox cycling of iron (Fe), P under anoxic conditions and decomposition of settling organic matter. Nutrients accumulating in the hypolimnion may become available to phytoplankton through periodic lake mixing or vertical migration of phytoplankton, and this nutrient source is a major component of internal loading. Many remediation methods have been developed to decrease nutrient loading from these sources, including hypolimnetic withdrawal, sediment capping, dredging, and other approaches covered in the remainder of this review. Ultimately, however, internal nutrient loading is a reflection of past nutrient inputs or of naturally high nutrient levels in inflows and catchment soils, and remediation approaches using these methods cannot provide permanent solutions.

Hypolimnetic withdrawal involves the direct removal of P-laden lake bottom waters and changing the depth at which water leaves the lake from the surface to near the maximum depth, allowing nutrient-rich rather than nutrient-poor water to discharge. This is usually accomplished by building a weir at the outlet and displacing outflowing surface water with water drawn from the hypolimnion (Klapper, 2003). Consequently, the hypolimnion retention time is shortened, the chance for anaerobic conditions to develop is decreased and the availability of nutrients to the epilimnion, through entrainment and diffusion, is reduced. The treatment is applied by installing a pipe along the bottom of the lake, from near the deepest point to the outlet. The outlet pipe is usually situated below lake level and acts as a siphon. This treatment is applicable to stratified lakes and small reservoirs where anaerobic hypolimnia restrict habitat for fish and promote the release of P, toxic metals, and hydrogen sulfide from sediments.

There are two important requirements for treatment success: (1) the lake level must remain relatively constant; and (2) thermal stability should not change (Cooke et al., 2005). While stratification may be weakened because epilimnetic water tends to be drawn downward, destratification will not occur provided the removal rate of hypolimnetic water is relatively slow. Furthermore, destratification should be avoided because it increases the transport of hypolimnetic nutrients and anoxic water to the epilimnion. To decrease the chances of destratification, directing inlet water to the metalimnion or hypolimnion may be possible. Preferentially removing hypolimnetic water decreases the residence time of the hypolimnion and decreases the period of anoxia and increases the depth of the anoxic boundary, resulting in decreased internal loading of P (Cooke et al., 2005). Hypolimnetic withdrawal is a proven remediation technology and at relatively low cost to accelerate recovery in stratified lakes with high internal loading. Low dissolved oxygen (DO) content of discharged water has historically been a major problem with deep-discharge impoundments. However, multiple and shallower outlets have been incorporated into reservoir design to counteract low DO discharge. Reducing discharge depth also minimizes nutrient export.

7.1 Case Studies and Lake Response

Hypolimnetic withdrawal installation was documented for 21 lakes, 15 of those in Europe. Results were reported from 17 of those lakes by Nürnberg (Tables 7.1 & 7.2; 1987). Internal loading from anoxic sediments during summer stratification occurred in all lakes prior to withdrawal and in most cases, external loading was reduced simultaneously. Prior to withdrawal, Kleiner Montiggler See had been aerated with liquid oxygen and Reither See was treated with iron chloride to precipitate P followed by dredging. Withdrawal is initiated preferably after stratification but before anoxic conditions occur. The siphon pipe is located usually 1 to 2 metres above the bottom at the greatest depth to maximize P transport. Hypolimnetic and epilimnetic data were available for 12 of these lakes, where maximum hypolimnetic TP concentration decreased in 11 of 12 lakes and epilimnetic TP decreased in 8 of 12 following treatment. The reduction in hypolimnetic TP is a direct effect; however, the epilimnetic reduction in TP is an indirect effect demonstrating that entrainment of P from hypolimnion to epilimnetic was reduced. The effect of withdrawal on epilimnetic TP was most significant as a function of total TP exported over the project life rather than annual export (expressed as total mass or per area). Furthermore, the longer withdrawal operated, the greater the proportional change in epilimnetic TP.

Table 7.1: Characteristics of withdrawal systems

Lake	Pipe Depth (m)	Withdrawal		Diameter (cm)	Pipe outflow (m)	Annual TP export (kg)	Duration (yr)
		Volume (10^3 m^3)	Rate (m^3/min)				
Ballinger	9.0	480	3.4	30.5	-	-	3.0
Bled	-	6,307	12	-	-	-	10
Burgäschi	15	1,000	3.0	33	0.5	147.1	5.0
Chain	6.2	435	4.8	45	1.0	30	9.0
Devil’s	14.3	629	9.1	48	2.2	446	1.0
Germündener Maar	-	-	0.1	-	-	-	-
Hecht	25	843	1.5	18	2.0	50.8	10
Kleiner Montiggler	13	16	-	-	-0.5	16	1.0
Klopeiner	30	-	-	-	-	-	3.0
Kortowo	13	-	-	-	0.5	-	45
Kraiger	-	-	-	20	-	-	4.0
Mauen	6.5	1,000	4.0	30	0.5	617	6.0
Meerfelder Maar	16	190	0.6	30	1.2	40.0	1.5
Paladru	31	-	21	-	-	416	5.0
Piburger	23	284	0.6	8.9	11	7.8	6.0
Pine	10.2	1,140	5.3	53	2.0	153	2.0
Reither	8	126	0.24	10	1.0	-	-
Stubenberg	-	-	-	-	-	-	-
Waramaug	8.5	1,330	6.3	31.8	0.0	131.9	3.0
Wiler	17.5	-	0.6	11	-	-	3.0
Wononscopomuc	15.1	201	0.9	-	0.0	21	5.0

Source: Modified from Nürnberg (1987).

Table 7.2: Morphometric characteristics of lakes treated with hypolimnetic withdrawal

Lake	Watershed area (10^3 m^2)	Lake area (10^3 m^2)	Lake vol. (10^3 m^2)	Water residence time (yr)	Mean depth (m)	Max. depth (m)	Mixis
Ballinger, Washington	11,720	405	1,838	0.26	4.5	10.0	Monomictic
Bled, Yugoslavia	NA	1,438	25,690	3.6	17.9	30.2	Meromictic
Burgäschi, Switzerland	3,190	192	2,483	1.4	12.9	32	Meromictic
Chain, British Columbia	NA	469	2,760	0.5-3.0	6.0	9.0	Polymictic
Devil's, Wisconsin	6,860	1,510	1,390	7.8	9.2	14.3	Dimictic
Germündener Maar, Germany	430	75	1,330	8.0	17.7	39.0	Meromictic
Hecht, Austria	2,221	263	6,428	2.8	24.4	56.5	Meromictic
Kleiner Montiggler, Italy	1,252	52	518	NA	9.9	14.8	Meromictic
Klopeiner, Austria	NA	1106	24,975	1.5	22.6	48.0	NA
Kortowo, Poland	1,020	901	5,293	NA	5.9	17.2	Dimictic
Kraiger, Austria	NA	51	245	2.0	4.8	10.0	Dimictic
Mauen, Switzerland	4,300	510	1,989	0.6	3.9	6.8	Dimictic
Meerfelder Maar, Germany	1,270	248	2,270	4.5	9.2	18.0	Dimictic
De Paladru, France	48,000	3,900	97,000	4.0	25.0	35.0	Dimictic
Piburger, Austria	2,640	134	1,835	1.9	13.7	24.6	Meromictic
Pine, Alberta	157,070	4,125	24,088	9.0	5.3	13.2	Dimictic
Reither, Austria	NA	15	67	0.3	4.5	8.2	Dimictic
Stubenberg, Austria	NA	450	NA	NA	NA	8.0	Polymictic
Waramaug, Connecticut	37,000	2,866	24,758	0.8	8.6	12.8	Dimictic
Wiler, Switzerland	257	31	325	1.0	10.0	20.5	NA
Wononscopomuc, Connecticut	5,994	1,400	15,500	4.0	11.1	32.9	Dimictic

Source: Nürnberg (1987).

7.1.1 Mauen See, Switzerland

Mauen See has been documented as one of the most successful cases of hypolimnetic withdrawal (Gächter, 1976). A pipe was installed in 1968 at a depth of 6.5 metres. Prior to installation, external P loading was reduced from approximately 700 to 300 mg/m^2 per year (Nürnberg, 1987). Improvements in lake quality were noticeable immediately following installation. Hypolimnetic DO and Secchi visibility increased, and hypolimnetic TP decreased by 1,500 µg/L, the most of any lake examined (Nürnberg, 1987). Epilimnetic TP decreased by 60 µg/L and *Oscillatoria* biomass decreased from a before-treatment summer maximum of 152 g/m^2 to 41 g/m^2 seven years following installation. Before installation, internal P loading from lake sediments during June and July was more than 200 times that of external loading (Nürnberg, 1987). After installation, internal loading decreased to only four times that of external loading (Nürnberg, 1987). Sediment P release declined for the six years of observation following installation and during that time, P export exceeded external loading (360 kg/yr) by a total of 3,700 kg, resulting in a decrease in P content of the surficial sediments.

7.1.2 Austrian Lakes: Reither See and Hechtsee

Several Austrian lakes have been described in the literature following the installation of Olszewski tubes, including Reither See and Hechtsee. Reither See, a dimictic lake, improved significantly in quality following the installation of a pipe placed near the maximum depth (8.2 m) in 1972 (Pechlaner, 1979). Tube diameter was 10 cm and water discharged at 0.24 m^3/min from the hypolimnion of the lake, which is 1.5 hectares in size (Cooke et al., 2005). Epilimnetic TP decreased from annual means of 38 and 43 µg/L in 1974 and 1975, to 21 µg/L in 1977 (Cooke et al., 2005). Transparency doubled over the 4-year period following installation and less blue-green algae was recorded (Cooke et al., 2005).

A larger tube (18 cm) was placed in Hechtsee in 1973, however, it was not placed near the maximum depth as in Reither See. The tube was placed at a depth of 25 m (56.6 m maximum depth of the lake) in order to protect the recreational environment around the lake from nuisance odours. Tube discharge from the 26.3 hectare lake varied from 1.2 to 1.8 m^3/min (Cooke et al., 2005). Because monomolimnetic water was not withdrawn, DO remained at zero from 25 m to the bottom of the lake. DO increased significantly above 25 m after installation and P transport from the lake increased despite the tube placement above the monomolimnion. During the first 4 years following installation, P output (203 kg) exceeded input (93 kg) by 110 kg, amounting to an actual decrease in lake TP content. TP above 24.5 m declined by 70-80 per cent from 1973 to 1977, and TP below 25 m change was minimal (Pechlaner, 1979).

7.1.3 Lake Wononscopomuc, Connecticut, United States

Withdrawal systems were installed in the shallower of two basins in Lake Wononscopomuc, Connecticut, in 1980. Hypolimnetic water was discharged from the shallow basin's maximum depth of 15.1 m at 0.9 m^3/min, which was sufficient to replace the hypolimnetic volume in 5.6 months (Kortmann et al., 1983; Nürnberg et al., 1987). Hypolimnetic TP decreased from approximately 400 µg/L before installation to 50 µg/L over five years. Epilimnetic TP decreased from 24–30 µg/L to 10–14 µg/L following the withdrawal (Kortmann et al., 1983). The decrease in TP was apparently due to reductions in internal loading by 79 per cent measured in the sediment release in the shallow basin after two years of withdrawal (Nürnberg et al., 1987).

DO in the hypolimnion also increased and the anoxic factor (days of anoxia) decreased from 50–65 before installation to less than 30 after withdrawal. Transparency remained high and unchanged (>5 m); however, metalimnetic blooms of Oscillatoria rubescens were eliminated by the treatment (Cooke et al., 2005).

7.1.4 Lake Ballinger, Washington, United States

Another U.S. lake treated by withdrawal is Lake Ballinger, Washington. Equipment was installed in 1982 to direct the inlet stream to the hypolimnion through a 276 m, 30.5 cm diameter pipe (Cooke et al., 2005). A control weir was constructed at the outlet to allow for adjustment of hypolimnetic and epilimnetic water discharge fraction. The mean flow-through was 3.4 m^3/min, resulting in a replacement time of the hypolimnion of approximately three months (Kramer et al. [KCM], 1986). Anoxia occurred for only two weeks in 1983, and the hypolimnion remained oxic with 3 to 4 mg/L DO during the stratification period in 1984. Maximum hypolimnetic TP decreased from approx. 450–900 µg/L during 1979–1981 (before installation) to approx. 100–150 µg/L during 1982–1985 (after installation). Internal loading was reduced from 224 kg in 1979 (pre-installation) to 17 kg in 1984. The overall decrease in internal loading was 70 per cent. Unfortunately, a significant increase in external loading during the late 1970s and early 1980s prevented a reduction in epilimnetic TP and therefore improvement in lake quality (KCM, 1986). The lake was treated with alum in 1993.

7.1.5 Lake Waramaug, Connecticut, United States

Two systems were installed in Lake Waramaug, Connecticut in 1983. One withdrew water from a depth of 8.5 m (12.8 m maximum depth) and discharged at 6.3 m^3/min. The other system withdrew water from the hypolimnion at the other end of the lake, returning the water aerated. There were no significant trends recorded in TP, both in the hypolimnion or epilimnion during the first three years following withdrawal.

7.1.6 Devil's Lake, Wisconsin, United States

Devil's Lake installed a 1,677 m hypolimnetic withdrawal pipe in 2002 at the maximum depth, varying from 13.5 to 15.7 m. Hypolimnetic withdrawal was applied due to high internal loading from deep-water sediments causing excessive amounts of planktonic and periphytic algae. The outflow rate was controlled to fluctuate from 6.8–10.3 m^3/min, dependent on lake level. The outflow P concentration average 725 µg/L for 48 days of operation, discharging 446 kg of hypolimnetic TP (Lathrop et al., 2004).

7.1.7 Pine Lake, Alberta, Canada

The restoration of Pine Lake, Alberta began in 1991 to improve water quality and to shift the lake to a mesotrophic state (Sosiak, 2002). Epilimnetic TP concentrations reached a median of 100 µg/L and Chl-*a* median of 20-50 µg/L during the mid-1990s (Sosiak, 2002). Most of the lake's TP originated from internal loading (61 per cent) and the 1,400 m hypolimnetic withdrawal pipe was installed in 1998. Controls on external loading from surface sources (36 per cent) took place during 1996–1998. The withdrawal system produced high rates of P loss and along with external controls, has reduced lake TP and improved water quality (Sosiak, 2002). TP concentration decreased by 44–47 per cent and Chl-*a* by 76-81 per cent during 1996–2000. Since 2000, TP and Chl-*a* concentrations have remained relatively low. Despite the decrease in TP and Chl-*a*, and increased transparency, there have been events of Gloeotrichia blooms that were never observed prior to treatment. There have been no significant adverse water-quality effects in the outlet stream, although temperature and DO were lowered immediately downstream.

7.2 Effects and Precaution

7.2.1 Beneficial Effects of Hypolimnetic Withdrawal on Lake Quality

The advantages of hypolimnetic withdrawal are: (1) relatively low capital and operational costs; (2) evidence of effectiveness in a large portion of cases; and (3) potentially long-term effectiveness. In the majority of cases, hypolimnetic DO increased, resulting in a decrease in the anoxic volume and days of anoxia. Internal P loading decreased and with the elimination of high external loading, epilimnetic TP also decreased. Another potential advantage is that hypolimnetic withdrawal may reduce the accessibility of cyanobacteria to Fe(II), which is now thought to be a precursor to the development of blue-green algae blooms (Molot et al., 2014) Furthermore, increase in hypolimnetic DO will increase habitat quality for fish, which is often a desirable outcome.

The effectiveness of withdrawal depends on magnitude and duration of TP transport from the hypolimnion, and it is important to exchange the hypolimnion volume as frequently as possibly (Cooke et al., 2005). A low rate of replacement may limit the effectiveness of this treatment.

7.2.2 Undesirable Effects of Hypolimnetic Withdrawal on Lake Quality

Discharge of hypolimnetic water containing high concentrations of P, ammonia, hydrogen sulfide, reduced metals and low oxygen may cause water-quality problems downstream. If outflow streams contain important fisheries and are otherwise used for recreation or water supply, then special precautions are necessary to minimize adverse effects. For example, withdrawal water from Lakes Wononscopomuc and Waramaug is aerated and mechanically cleaned before being discharged downstream. The extent to which DO in the outflow water will be reduced can be estimated by comparing existing DO deficit in the lake with the input load of DO (Pechlaner, 1979). If low DO is expected in the outlet, then aeration equipment should be installed (Cooke et al., 2005).

The primary drawbacks of this treatment include:

- Withdrawal followed by treatment and discharge back to the lake is inefficient in removing phosphorus compared to in-lake treatment.
- Potential warming of the lake as bottom waters are exposed to surface temperatures.
- Water removed may have a strong odour.
- Method is restricted to deep, stratified lakes with considerable internal loading.

7.3 Cost

Installation costs for three systems in the United States lakes are summarized in Table 7.3. Relatively low capital and annual operational cost are advantages of hypolimnetic withdrawal.

Table 7.3: Cost comparison and withdrawal characteristics for selected North American lakes

Lake	Size (ha)	Discharge Rate (m^3/min)	Cost (2015 USD)*
Lake Ballinger	41	3.4	$545,500
Lake Waramaug	287	9.1	$80,500
Devil's Lake	151	9.1	$402,500
Pine Lake	412	5.3	$366,000*
Note: Pine Lake's cost does not include labour and equipment; costs do not include treatment of outflow water.			

** The cost in 2002 was $420,000 (Lake Ballinger), USD 62,000 (Lake Waramaug), USD 310,000 (Devil's Lake) and USD 282,000 (Pine Lake).*
Source: Cooke et al. (2005).

A feasibility study conducted by Barr Engineering for remediation technologies in Twin Lake included the cost analysis of hypolimnetic withdrawal. Construction costs were estimated at $400,000, and annual operation and maintenance cost (including electricity, chemical cost and settle flocculent disposal) were estimated at $40,000 (Barr, 2013). Engineering and design cost were estimated at 20 per cent of total capital costs. For a 20-year treatment life, the total cost of the project would total $1,330,000.

Hypolimnetic Withdrawal Summary

Nutrient-enriched hypolimnetic waters can be preferentially removed through siphoning, pumping or selective discharge, where the direct removal of P-laden bottom water and changing the depth at which water leaves the lake from the surface to near the maximum depth, allowing nutrient-rich rather than nutrient-poor water to discharge. Consequently, hypolimnetic withdrawal shortens hypolimnetic retention time, decreases the chance for anaerobic conditions to develop, accelerates phosphorus export, reduces surface phosphorus concentrations, and improves hypolimnetic oxygen content.

Beneficial Effects

- Relatively low capital and operational costs.
- Potentially long-term effectiveness.
- Hypolimnetic DO increase, which can result in a decrease in the anoxic volume and days of anoxia.
- Reduce the accessibility of cyanobacteria to Fe(II), now thought to be a precursor to the development of blue-green blooms.
- Increase in hypolimnetic DO can improve fish habitat.

Undesirable Effects

- Potential for water quality issues downstream if hypolimnetic water contains high concentrations of P, ammonia, hydrogen sulfide and low oxygen.
- Withdrawal followed by treatment and discharge back to the lake is inefficient in removing phosphorus compared to in-lake treatment.
- Potential warming of the lake as bottom waters are exposed to surface temperatures.
- Destabilization of the thermocline and enable nutrients from the hypolimnion to become available for phytoplankton growth in the epilimnion.
- Water removed may have strong odour.

Suitable Lake Parameters

There is evidence of the effectiveness of hypolimnetic withdrawal in a large portion of cases, however, successful implementation of this method is restricted to deeper, stratified lakes with considerable internal loading.

Lake size: 1.5–400 hectares.

Depth (mean): 3.0–48.0 m.

Depth (max.): 6.8–56 m.

Residence time: 0.26–9.0 years.

Longevity: Effectiveness of treatment depends on magnitude and duration of TP transport from the hypolimnion, and it is important to exchange the hypolimnion volume as frequently as possible. A low rate of replacement may limit the effectiveness and longevity of treatment.

Important to understand natural refilling rate and if it's high enough to reduce lake drawdown resulting from hypolimnion discharge. Smaller lakes may refill too slowly to be effective.

Cost

Relatively low capital and annual operation cost are advantages of hypolimentic withdrawal.

Cost (range from literature): $80,000–$600,000.

Twin Lake, MN (8 hectares in size, Barr, 2013):

Construction cost: $400,000.

Annual operation $40,000.

20-year treatment life, total cost $1.3 million.

8. DILUTION AND FLUSHING

Dilution and flushing can achieve improved quality in eutrophic lakes by reducing the concentration of the limiting nutrient (dilution) and by increasing the water exchange rate (flushing). Both processes can reduce plankton algae biomass by reducing the inflow concentration of the limiting nutrient, resulting in a decreased lake concentration on which maximum algal biomass depends. By increasing water input, loss rates of planktonic algae from the lake are also increased. Other effects of dilution are also possible, such as increased vertical mixing and decreased concentration of algal excretory products that can influence the abundances and species of algae (Keating, 1977). Dilution is usually feasible where large quantities of low-nutrient water are available and treatment effectiveness is greatest when dilution water is low in limiting nutrient concentration relative to that of the lake and its natural inflow (Cooke et al., 2005). Furthermore, lake nutrient concentration can be more effectively lowered if dilution water is the dominant inflow.

The concentration and loading of limiting nutrients is a critical determinant of potential algal biomass. In-lake nutrient concentrations are usually lower than inflow concentrations because sedimentation is greater than internal loading. Adding more water with lower nutrient content will decrease nutrient concentrations and increase nutrient and plankton losses via the outflow. Overall nutrient loading is increased with this strategy; however, nutrient loss through sedimentation is potentially decreased (Uttormark & Hutchins, 1980). A large increase in the flushing rate achieved by adding low-nutrient water (40 per cent of the normal inflow nutrient content) could theoretically increase lake nutrient concentrations if the original flushing rate is low enough (e.g., 0.1/year). If the flushing rate is relatively large (≥ 1.0/year) initially, the effect of reduced sedimentation is minimized and a reduction in lake concentration will result. The amount of water needed to achieve a given reduction in inflow concentration is a function of the concentration difference between the normal inflow and dilution-water source.

8.1 Case Studies and Lake Response

Dilution and flushing have been successfully applied in several lakes, including Green Lake (Washington), Moses Lake (Washington), Clear Lake (California), Snake Lake (Wisconsin), Lake Bled (Yugoslavia) and Lake Veluwe (the Netherlands). Lakes where the dilution and flushing technique was implemented can be used as a guide for application: Rotsee Lake, Moses Lake and Green Lake, Seattle.

8.1.1 Rotsee Lake, Switzerland

The first recorded lake flushing experiment was diverting water from Switzerland's Reuss River to Rotsee from 1921 to 1922 in an attempt to alleviate eutrophic conditions (Stadelman, 1980). The flushing rate was increased from 0.33 to 2.5/year (0.1 to 0.7 per cent per day) and the lake was 460 ha in size. Despite flushing, the lake's state did not improve due to high concentrations of nutrients in the river water used for flushing, where the nutrients originated in sewage effluent from the upstream city of Lucerne. In 1933, the diversion of direct sewage effluent input also had no considerable improvement of the lake quality. Despite past failed attempts, in 1970 the removal of nutrients from Lucerne's wastewater significantly affected the lake quality, resulting in a tenfold P reduction in the Reuss River inflow water.

8.1.2 Moses Lake, Washington, United States

Moses Lake, located in eastern Washington, has a surface area of 2,753 hectares and a mean depth of 5.6 metres. Dilution water from the Columbia River has been added to one arm of the lake during the spring and early summer since 1977. Transport to a previously undiluted portion of the lake by pumping began in 1982 and sewage effluent was diverted from Pelican Horn in 1984. The effects of dilution treatment on lake quality were only evaluated for Parker Horn.[2]

The average amount of diluted water added from 1977 to 1988 represented a flushing rate of 17 per cent per day for the 971 days of actual inflow. The average input for April to September was 5.8 per cent per day (Cooke et al., 1993). With dilution water plus the normal input, the flushing rate averaged 7.8 per cent per day for Parker Horn. For the entire lake, these inputs represented a flushing rate of less than 1 per cent per day.

Table 8.1: Nutrient concentration in inflow water to Parker Horn (May to September, 1977-78)

	TP (µg/L)	TN (µg/L)	SRP (µg/L)	NO_3-N (µg/L)
Crab Creek inflow without dilution	148	1331	90	1096
East Low Canal dilution water	25	305	8	19
TP = total phosphorus; TN = total nitrogen; SRP = soluble reactive phosphorus; NO_3-N = nitrate nitrogen.				

Source: Cooke et al. (1993).

Dilution water addition continued at a slightly higher rate during the 1990s, resulting in a 30 per cent increase over the previous 12 years. P and N concentrations in Crab Creek, Parker Horn's natural inflow, were high due to irrigation and fertilization practices in the watershed; therefore, relatively large quantities of Columbus River water were needed to significantly lower the composite inflow concentration (Table 8.1). Diverting Crab Creek was not economically feasible in this case, but a similar manipulation could be considered for other lakes to obtain greater efficiency from dilution water.

While algal biomass was substantially reduced by dilution, algal composition did not change during the 12 years of dilution and blue-green algae dominated throughout the summer (Table 8.2). Blue-green algae decreased initially but did not persist, which was unexpected since decreased blue-green dominance has accompanied decreased TP content in most cases (Sas et al., 1989). It was determined that the principal effects in Pelican Horn were from sewage diversion.

[2] The lake is complex in shape, with several sections or "arms," which are called horns.

Table 8.2: Nutrient concentration of three lakes: Parker Horn, South Lake and Pelican Horn

Years	Dilution rate (%/day)	TP (µg/L)	SRP (µg/L)	Chl-*a* (µg/L)	Secchi depth (m)
Parker Horn					
1969–1970	1.6	152	28	71	0.6
1977–1979	7.8	68	15	26	1.3
1986–1988	8.0	47	6	21	1.5
South Lake					
1969–1970	1.1	156	48	42	1.0
1977–1979	3.5	86	35	21	1.7
1986–1988	3.6	41	7	12	1.7
Pelican Horn					
1969–1970	0.0	920	634	48	0.40
1977–1979	0.0	624	441	39	0.45
1986–1988	7.7	77	6	12	0.65
Note: Samples were taken from 0.5 m depth transects.					

Source: Cooke et al. (1993).

8.1.3 Green Lake, Washington, United States

Green Lake, located in Seattle, Washington, has an area of 104 hectares and mean depth of 3.8 metres. The lake received diluted water from the city domestic supply at relatively high rates from 1962 through to the mid-1970s. However, variable water inputs resulted in worsening lake quality. Dilution was proposed as the primary treatment in 1960 and applied in 1962 (Sylvester & Anderson, 1964). In contrast to Moses Lake's Parker Horn, dilution of Green Lake represented a much lower rate of flushing, 2–3 per cent per day. The addition of dilution water to the lake from 1965 to 1978 produced a flushing rate ranging from 0.88-2.4 per year (0.24-0.65 per cent per day) (Perkins, 1983).

Improvements in Chl-*a*, TP and Secchi transparency were observed during the first few years of dilution. Water transparency during the summer increased nearly fourfold to an average of 4 m (mean lake depth is 3.8 m) where most of the lake bottom was visible. Chl-*a* decreased more than 90 per cent from 45 to 3 µg/L. Summer mean TP decreased from 65 to 20 µg/L and a substantial decrease in the cyanobacterial fraction was observed. However, during the late 1970s, lake quality had degraded significantly primarily due to declining dilution water inputs. The limitation in the availability of Seattle domestic water required developing a long-term solution for the lake, and mass balance analysis in the 1980s determined the major contributing factor was internal P loading. Following the recognition of the effect of internal loading, dilution still remained the principal option to improve and maintain Green Lake quality in achieving mean summer TP concentration of 28 µg/L. Water from the city supply was desirable due to the delivery facility in place and the city water concentration of 10 µg P/L. However, city water cost approx. \$0.13/m^3 (2002 USD) and the supply during summer was no longer routinely available (URS, 1983). The cost and effectiveness of other, more reliable dilution-water sources and other controls on internal loading were evaluated and alum additions were implemented in 1991 (URS, 1983, 1987; Jacoby et al., 1994). City water is still added occasionally when excessive algal blooms occur.

8.2 Effects and Precautions

8.2.1 Beneficial Effects of Dilution and Flushing on Lake Quality

Dilution and flushing has been noted to improve lake water quality due to the reduction in nutrient concentration. The advantages of using dilution water include: relatively low cost if water is available in high quantity; an immediate and proven effectiveness if the limiting nutrient can be decreased; and moderate success even if only moderate-to high nutrient water is available, through physical limitations to large algal concentrations.

8.2.2 Undesirable Effects of Dilution and Flushing on Lake Quality

The principal limitation for the use of this technology is the availability of low-nutrient dilution water. Furthermore, dilution is frequently used synonymously with flushing. For dilution to be cost effective, the inflow water must be substantially lower in concentration than the lake, where effectiveness increases as the difference between inflow and lake concentrations increases.

For washout to be an effective control on algal biomass, the flushing rate must be a large fraction of the algal growth rate. For the cases previously described, facilities for transporting water existed and associated costs applied. Dilution and flushing is limited to relatively small lakes, where there is a sufficiently large amount of low-nutrient water to effect a decrease in nutrient concentration. If water is derived from a source outside of the catchment, there may be a risk of introducing undesirable taxa and undesirable concentrations of other chemical constituents of water.

8.3 Cost

Costs are highly variable, dependent upon the presence of a facility to deliver the water, and the quantity and proximity of available water. If the lake is in an urban setting and domestic water is available, then improvement may be possible for less than $195,000 (2015 USD) for construction, water cost and the first year of maintenance and operation (Cooke et al., 2005). In the case of Moses Lake, the primary project cost had been the pumping facility for Pelican Horn at $750,000 (2015 USD) plus planning, administrative, and monitoring and research cost (Cooke et al., 2005). If the lake is in close proximity to a free-flowing river and diversion of a portion of the river flow through the lake during the summer is feasible, then the costs involves facilities, pumps, pipes, operation and prevention of side effects, for example entraining fish. One of the main advantages to using dilution water is the relatively low cost associated if water is available in high quantity.

Dilution and Flushing Summary

Dilution and flushing can achieve improved quality in eutrophic lakes by reducing the concentration of the limiting nutrient (dilution) and by increasing the water exchange rate (flushing). Dilution involves the addition of low-nutrient water to reduce lake nutrient concentration and has been effective where external or internal sources are not controlled. Flushing refers to the removal of algal biomass.

Beneficial Effects

- Relatively low cost if water is available in high quantity.
- An immediate and proven effectiveness if the limiting nutrient can be decreased.
- Moderate success even if only moderate-to high-nutrient water is available.

Undesirable Effects

- If dilution water is derived from a source outside of the catchment, there may be a risk of introducing undesirable taxa.
- Potential impacts on the diverted water source.

Suitable Lake Parameters

Generally limited to relatively small lakes where there is sufficiently large amounts of low-nutrient water to effect a decrease in nutrient concentration.

Successful implementation of dilution and flushing as reviewed in the literature.

Lake size: 104–490 hectares.
However, Moses Lake (WA) successful decreased TP concentration with the application of dilution/flushing, and the lake is 2,753 hectares in size.

Depth (mean): 3.8–5.6 m.

P load: In-lake nutrient conc. are usually lower than inflow conc. because sedimentation is greater than internal loading. Nutrient load is usually increased with this strategy, however, nutrient loss through sedimentation is potentially decreased.

Residence time: 0.26–9.0 years.

Chl-*a*: 71–102 µg/L.

Flushing rate: 5.8 - 17% per day or ≥ 1.0/year large enough initially to reduce in-lake concentration.

The amount of water needed to achieve a given reduction in inflow concentration is a function of the concentration difference between the normal inflow and dilution water source.

Cost

High variability and dependent upon the presence of a facility to deliver water, and the quantity and proximity of available water.

Cost range from literature: $100,000–$800,000.

Moses Lake, WA: Primary cost was the pumping facility $750,000 (USD, 2015).

9. HYPOLIMNETIC AERATION AND OXYGENATION

The depletion of dissolved oxygen (DO) in the hypolimnia of stratified eutrophic lakes in one of the first signs of eutrophication (also occurs in oligotrophic lakes). Anoxia occurs if respiration of organic matter in hypolimnetic water and sediments is sufficient to exhaust hypolimnetic DO before autumn destratification (Cooke et al., 2005). Anoxia produces undesirable changes in lake quality, including accelerated internal recycling of nutrients, solubilization of undesirable metals, and changes in the distribution of fish and other biota, particularly coldwater species. Hypolimnetic aeration was first used in Lake Bret, Switzerland, a lake management technique designed to counteract hypolimnetic anoxia and its associated effects on lake quality (Mercier & Perret, 1949). Hypolimnetic aeration is usually accomplished by the injection of pure oxygen or air in fine bubbles to the hypolimnion so that oxygen dissolves without disturbing stratification (Singleton & Little, 2006). It is typically used in lakes where it is desirable to maintain coldwater habitat and where whole-lake circulation is undesirable. The specific objectives of hypolimnetic aeration are:

1. To raise the oxygen content of the hypolimnion without destratifying the water column or warming the hypolimnion.
2. To provide an increased habitat and food supply for coldwater fish species (dependent on the previous objective).
3. If sediment-to-water exchange of P is controlled by iron redox, to reduce sediment P release by establishing undesirable conditions at the sediment–water interface.

Aerator design

There are several designs for hypolimnetic aerators. Fast and Lorenzen (1976) reviewed 21 designs and grouped them into three categories: (1) mechanical agitation, involving removal, treatment and return of the hypolimnetic water; (2) injection of pure oxygen; and (3) injection of air, either full or partial airlift design or through a down-flow injection design. The most commonly used designs in current use are the airlift aerator, Speece Cone, and bubble plume diffuser (Singleton & Little, 2006)

Mechanical agitation involves drawing water from the hypolimnion, aeration of that water and returning the water to depth with minimal increase in temperature. This technique is not encouraged due to poor gas exchange efficiency (Pastorak et al., 1982). To improve gas exchange efficiency, pure oxygen has been used to aerate hypolimnetic water drawn down from depth, treated at the shore or on the surface, with or without pressure, and returned. This is referred to as a deep oxygen injection system (DOIS).

The injection of air via airlift systems has been more prevalent for hypolimnetic aeration (Singleton & Little, 2006). Full airlift brings bottom water to the surface by forcing compressed air into the bottom of an outer cylinder. The rising bubbles drive the air–water mixture to the surface, exposing the water to the atmosphere, and then returns to the hypolimnion via an inner cylinder, after venting the air bubbles. Partial airlift units aerate hypolimnetic water in place with water and air bubble separated at depth and excess air discharged at the surface. In both airlift systems, the contained air forces the oxygenated water to distribute horizontally into the hypolimnion. Pure O_2, instead of air, will increase gas transfer efficiency but provides less distribution force than air. Fast and Lorenzen (1976) found that the full airlift design is least costly and more efficient at delivering oxygen than other systems. However, the partial airlift design is used more frequently due to commercial availability (Cooke et al., 2005).

Unit sizing

The effectiveness of aerators to oxygenate a hypolimnion depends on the DO transport efficiency between air bubbles and pumped water (Ashley, 1985; Ashley et al., 1992). Ashley (1987) developed and tested an empirical sizing method:

1. Determination of the maximum hypolimnetic volume.
2. Estimation of the hypolimnetic oxygen consumption rate in kg/day by calculating the slope for the relationship between average hypolimnetic DO and time.
3. Calculation of water flow necessary to meet DO demand rate, which depends on the aerator input DO concentration.
4. Calculation of aerator flow and outflow tube size (radius) from water flow and velocity estimation. Tube length should be as long as possible to optimize oxygen transfer efficiency.
5. Determination of friction losses at the aerator entrance and exit, as well as in the line to estimate total head loss.
6. Determination of air–water mixture density, assuming the theoretical head generated is due to the density difference between ambient water and the air–water mixture.
7. Calculation of the airflow rate to transport the required water flow to satisfy the oxygen demand.
8. Estimation of pressure requirements at the compressor and power demand.

Some important questions to consider when reviewing aeration feasibility for a specific lake are:

- Does the hypolimnion become anoxic, and how long is the duration of this anoxia? Aeration will have greater benefits for lakes with longer periods of anoxia. Some lakes may continuously be mixed and never experience hypolimnetic anoxia, others may have diurnal cycle of anoxia and oxygenation (Carlton & Wetzel, 1988). Aeration will have greater benefits for lakes with longer periods of anoxia.
- Hypolimnion size: Hypolimnetic aerators tend to be more efficient and less likely to disrupt stratification if the hypolimnion is large.
- Is there sufficient iron in the sediment to bind available P under oxygenated conditions?
- What is the sulphide concentration at the sediment–water interface? High sediment sulphide concentration can result in P release from iron even in oxic conditions (Gächter & Müller, 2003).
- If external loading exceeds the retention capacity of the sediment, aeration will not affect the amount of P released.
- Are blue-green algae a nuisance species? In some cases where hypolimnetic aeration mixes the whole water column, it may produce enough turbulence to reduce the buoyancy and dominance of blue-green algae (Westwood & Ganf, 2004).

9.1 Case Studies and Lake Response

Pastorak, et al. (1981) reviewed multiple lakes treated by hypolimnetic aeration or oxygenation technologies. Table 9.1 defines the characteristics of the individual aeration projects, including depth of treatment, volume and area treated, as well as water flow (m^3/min). The main benefit of this treatment is that anoxic hypolimnia can be readily switched to an oxic state by effective aeration, while maintaining a normal coldwater environment and potentially decreasing internal loading of P, Fe, Mn, ammonium and hydrogen sulfide.

However, hypolimnetic aeration has both improved and worsened trophic states of the lakes examined. With the application of aeration in Medical Lake, Washington, hypolimnetic P declined with aeration; however, there was no effect on epilimnetic Chl-*a* (Soltero et al., 1994).

Table 9.1: Lakes treated by hypolimnetic aeration or oxygenation and associated characteristics

Lake	Depth			Vol. (10^6 m^3)	Area (ha)	Q air/gas (m^3/min)	Reference
	Max.	Mean	Device				
Brunsviken, Sweden	-	-	13 [1]	-	100	15.5	Atlas Copco, 1976
Caldonazzo, Italy	50	-	11 [1]	-	700	44	Atlas Copco, 1980
Jarlasjön, Sweden	24	9.3	24 [1]	7.8	84	22.8	Bengtsson & Gelin, 1975
Kolbontnvatn, Norway	18.5	10.3	18.5 [1]	3.1	30.3	5.5	Atlas Copco, 1980
Larson, Wisconsin	11.9	4.0	11.9 [1]	0.188	4.8	0.45	Smith et al., 1975
Mirror, Wisconsin	13.1	7.6	12.8 [1]	0.4	5.3	0.45	Smith et al., 1975
Ottoville Quarry, Ohio	18	8.6	18 [2]	0.063	0.73	0.11	Fast, 1973; Overholtz et al., 1977
Spruce Run, New Jersey	13.1	-	12.2 [2]	-	-	0.15	Whipple et al., 1975
Waccabuc, New York	13	7.5	13 [1]	4.053	53.6	7.93	Garrell et al., 1977
Medical, Washington	18.3	9.8	18 [1]	6.2	63	4.5	Soltero et al., 1994
Camanche, California	31	17	30 [2]	511	3,000	4.6	Beutel & Horne, 1999
Amisk, Alberta	34	10.8	33 [2]	25.1	233	0.3-0.6	Prepas et al., 1997
Fenwick, Washington	7.9	4.0	6.7 1	0.42	10.4	-	Pastorak et al., 1981
Baldegg, Switzerland	65	32.7	~60 [2]	170	520	5.6	Gächter & Wehrli, 1998
Sempach, Switzerland	85	21.5	~80 [2]	670	1,440	3.3	Gächter & Wehrli, 1998
Hallwil, Switzerland	45	28.4	~40 [2]	290	1,020	-	Jungo, 1993
Stevens, Washington	46	20.5	43 [1]	194	421	44	Pastorak et al., 1981
Black, British Columbia	9	-	7 [1]	0.18	-	1.13	Ashley & Hall, 1990
Newman, Washington	9.1	5.8	9 [2]	28.4	490	1.15	Beutel & Horne, 1999
Gross-Glienicker, Germany	11	6.5	11	4.2	68	-	Wolter, 1994
Vadnais, Minnesota	16.5	8.1	~16 [1]	12.5	155	-	Walker et al., 1989
Spruce Knob, Virginia	5.7	2.1	5.2 [1]	0.224	10.5	1.3	Hess, 1977; LaBaugh, 1980
Ghirla, Italy	14	8	14 [2]	2	24.5	-	Bianucci & Bianucci, 1979
Tory, Ontario	10	4.5	9.0 [1]	0.055	1.23	3.45	Taggart & McQueen, 1981
Irondequoit, New York	23.7	3.5	14-23 [2]	23.4	679	0.6	Babin et al., 1999

[1] Hypolimnetic aeration; [2] oxygenation.

Source: Cooke et al. (2005)

9.2 Effects and Precautions

9.2.1 Beneficial Effects of Hypolimnetic Aeration and Oxygenation

The principal benefit of hypolimnetic aeration is that anoxic hypolimnia can switch to an oxic state while still maintaining a coldwater environment and potentially decrease internal loading of P, Fe, Mn, ammonium, hydrogen sulfide, and methyl mercury. By changing the sediment–water interface from reducing to oxidizing, the release of dissolved forms of P, Fe and Mn often decreases, and the release of ammonium may decrease due to increased nitrification (McQueen & Lean, 1984). Based on the case studies reviewed, the response was usually a 30–50 per cent reduction in hypolimnetic P. Hypolimnetic aeration has both improved and worsened trophic state; however, aeration may improve habitat quality for coldwater fish, even if improvements in epilimnetic water quality are not achieved.

9.2.2 Undesirable Effects of Hypolimnetic Aeration and Oxygenation

Aeration primarily reduces internal P loading by disrupting Fe-P redox reactions at the sediment–water interface. Interactions between Fe and P primarily affect only the short-term cycling of P and do not result in the permanent storage of P in lake sediments (Gächter & Müller, 2003; Gächter & Wehrli, 1998; Golterman, 2001; Katsev, 2006). As a result, long-term reductions can be achieved primarily only through decreases in watershed inputs. Phosphorus improvements do not always occur with aeration. Lakes where internal P recycling is driven by processes unrelated to Fe-P interactions may not show any positive effects of aeration on nutrient loading. Supersaturation of hypolimnetic water with nitrogen gas (N_2) was suggested as a possible effect that might lead to gas bubble disease in fish. Cooke et al., (2005) determined that this has not occurred as a result of hypolimnetic aeration; however, N_2 content can reach potentially damaging concentrations with the application of this technology. As well, 150 per cent saturation relative to surface temperature and pressure occurred after 80 days of aeration in Lake Waccabuc (Fast et al., 1975; Cooke et al., 2005) and has also occurred in other lakes (Bernhardt, 1974; McQueen & Lean, 1984). Kortmann et al. (1994) suggested that increased N_2 levels resulting from hypolimnetic aeration might be a concern but only in the deepest of treated lakes. Liquid oxygen aeration can reduce this problem.

Hypolimnetic aeration may also increase eddy diffusion of nutrients into the epilimnion even though stratification is maintained. Metalimnetic P content increased by a factor of 2 in Lake Wesslinger during hypolimnetic aeration, which doubled cyanobacterial biomass (Steinberg & Arzet, 1984). Similar entrainment of hypolimnetic water also increased blooms in Lake Tegel (Cooke et al., 2005). One of the principal reasons for maintaining stratification during aeration is to prevent the recirculation of sediment-released P and maintain habitat for coldwater biota. If this does not occur, hypolimnetic aeration can be regarded as less efficient in comparison to complete circulation as an enhancement to fisheries-usable habitat. Metalimnetic DO minima could also negatively affect fish communities, restricting fish movement (Bengtsson & Gelin, 1975). These potential problems with hypolimnetic aeration may be eliminated with layer aeration (Kortmann et al., 1988, 1994), a double bubble contact system (DBCS) or a deep oxygen injection system (DOIS) (Beutel & Horne, 1999).

Hypolimnetic aeration may not be effective if the water body is too shallow. Although stratification may exist, the density gradient may not be sufficient to resist thermocline erosion. In slow circulation conditions and if the water body destratifies, this may result in low DO throughout the water column and the introduction of toxic chemicals (e.g. H_2S) into the epilimnion (McQueen & Lean, 1986; Cooke & Carlson, 1989). This would be of particular concern if epilimnetic water is being used as a drinking source. Hypolimnetic aeration is not recommended if maximum depth is less than 12 to 15 m and/or the hypolimnetic volume is relatively small (Cooke & Carlson, 1989). While hypolimnetic aeration may restore oxygen conditions for fish and other biota, other toxic elements may not be sufficiently reduced to allow survival.

9.3 Cost

The cost of hypolimnetic aeration depends primarily on hypolimnetic O_2 demand; however, the amount of air required (compressor size) to deliver O_2 to the hypolimnion depends on the distance from the compressor to the discharge site and depth of the unit. Large variations in energy efficiency and project cost occur due to friction loss associated with pipe length, size and head loss corresponded to depth. Energy efficiency can be optimized with compressor size. General Environmental Systems provided project costs for seven partial airlift systems installed in the 1970s and 1980s, where the average energy efficiency in operation was 1.4 kg O_2/kWh. Cooke et al., 2005 examined these projects and used \$0.12/kWh (2002 USD) to represent an average cost efficiency of \$0.86 (± 36%)/kg O_2. The operating cost plus the average installed cost over a 10-year longevity at 160 days per year, was estimated at \$0.39/kg O_2 per day. In 2015, this project would cost \$0.5/ kg O_2 per day. Using the cost efficiency and the average air flow reported for the 15 aeration projects listed in Table 10.1, which is 15.3 m^3/min or 3.7 kg O_2/min, amounts to approximately \$340,000 for 160 days per year for installation and operation cost. Based on the average area of the 15 lakes, the annual cost is approx. \$4,000/ha per year (2015 USD).[3]

In the case of Lake Steven, two custom units delivered 15.5 metric tonnes O_2/day to the lake's hypolimnion, a volume of 17 x 10^6 m^3 for over 10 years. The operating and capital costs of the project for 160 days/year were \$0.27/kg O_2 per day or \$1,610/ha per year (2015 USD).[4] Additionally, 15 units installed in Lake Tegel delivered 4.5 tonnes O_2/day starting in 1980 (Verner, 1984). The initial cost of that system was \$3,770,160 (2002 USD). The operating cost over the 10 years of operation was \$0.09/kg O_2 per day and 160 days/year, amounting to \$64,000. In 2015 dollars that would amount to \$575,000 or \$1,370/ha per year.

[3] 2002 USD cost approx. \$3,000/ha per year (Cooke et al., 2005)

[4] 2002 USD cost approx. \$0.21/kg O_2 per day or \$1,240/ha per year (Cooke et al., 2005).

Hypolimnetic Aeration and Oxygenation Summary

Hypolimnetic aeration is usually accomplished by the injection of pure oxygen or air into the hypolimnion, without disturbing stratification. The specific objectives of hypolimnetic aeration are:

- To raise the oxygen content of the hypolimnion without destratifying the water column or warming the hypolimnion.
- To provide an increased habitat and food supply for coldwater fish species (dependent on the previous objective).
- If sediment-to-water exchange of P is controlled by iron redox, to reduce sediment P release by establishing undesirable conditions at the sediment–water interface.

Beneficial Effects

- Anoxic hypolimnia can switch to an oxic state while still maintaining a coldwater environment.
- Potentially decrease internal loading of P, Fe, Mn, ammonium, hydrogen sulfide and methyl mercury.
- Aeration may improve habitat quality for coldwater fish, even if improvements in epilimnetic water quality are not achieved.

Undesirable Effects

- Interactions between Fe and P primarily affect only the short-term cycling of P, and do not result in the permanent storage of P in lake sediment.
- Phosphorus improvements do not always occur with aeration.
- Lakes where internal P recycling is driven by processes unrelated to Fe-P interactions may not show any positive effects on nutrient loading.
- Potential supersaturation hypolimnetic water with N_2 can lead to gas bubble disease in fish.
- Potential to increase eddy diffusion of nutrients into the epilimnion, even though stratification is maintained.
- Slow circulation conditions and destratification may result in low DO throughout the water column and introduce toxic chemicals (H_2S) into the epilimnion.

Suitable Lake Parameters

Hypolimnetic aeration will not be effective if the waterbody is too shallow. Although stratification may exist, the density gradient may not be sufficient to resist thermocline erosion. While hypolimnetic aeration may restore oxygen conditions for fish and other biota, other toxic elements may not be sufficiently reduced to allow survival. Successful implementation as reviewed in the literature:

Lake size: 5.3–3,000 hectares.

Depth (mean): 3.5–28.4 m and lake must be stratified.

Depth (max.): 5.7–85 m. Not recommended if max. depth is less than 12–15 m and/or hypolimnetic volume is relatively small.

Device depth: 5.2–33 m.

Longevity: Continual treatment. For example, Lake Steven and Lake Tegel 10 years of operation.

Cost

Less cost effective than other treatments for phosphorus control, example alum; however, there are other reason for aeration, such as creating an aerobic environment.

$4,000 per hectare per year (2015 USD) (based on mean areas of 15 lakes).

Lake Steven (operating 160 days per year): $340,000 ($0.27/kg O_2 or $1,610 per hectare).

10. ARTIFICIAL CIRCULATION AND AERATION

Artificial circulation, similar in concept to hypolimnetic aeration, is achieved by pumps, jets and diffused air; however, circulation of the entire lake is generally intended rather than a select region or depth. Unlike hypolimnetic aeration, the temperature of the whole lake will increase with complete circulation if mixing includes water that was previously part of the cooler hypolimnion. The principal improvements in water quality caused by complete circulation are oxygenation and chemical oxidation of substances in the entire water column, as well as enlarging the suitable habitat for aerobic warm-water species (Pastorak et al., 1981, 1982).

Complete circulation may reduce internal loading of P if the principal P-release mechanism is iron reduction in anoxic profundal sediments (Pastorak et al., 1981). Complete circulation may also reduce algal biomass by increasing the mixed depth, reducing available light and subjecting mixed algal cells to rapid changes in hydrostatic pressure (Lorenzen & Mitchell, 1975; Fast, 1979; Forsberg & Shapiro, 1980). Many cyanobacteria are vacuolated, which allows them to migrate vertically and access nutrients from the meta- and hypolimnion. This competitive advantage in quiescent, stratified lakes may be lost with complete circulation, enhancing conditions for more desirable algal species. This management technique has been employed since the early 1950s (Hooper et al., 1953) and was initially used to prevent winter fish kills in shallow, ice-covered lakes (Halsey, 1968).

The principal effect of circulation is to raise the dissolved oxygen (DO) content throughout the lake over time. If the lake is destratified, the DO content in what was the hypolimnion (bottom layer of the water column) will increase and the DO content in the upper layer of the water column) will initially decrease. Toxic substances such as ammonia and H_2S that may build up in anoxic hypolimnia will also be decreased. DO will continue to increase as circulation is maintained because water undersaturated with oxygen is continually brought into contact with the atmosphere. Internal loading of P can be decreased through increased circulation when the dominant mechanism of P release is from iron-bound P in anoxic hypolimnetic sediments. By aerating the sediment–water interface of lakes where iron is controlling P solubility, P will be sorbed from solution by ferric-hydroxy complexes (Mortimer, 1941, 1971; Stumm & Leckie, 1971). Whole-lake mixing may be used seasonally during autumn and winter to ensure more complete lake oxygenation and prevent fish winterkills (Ashley, 1987). Artificial circulation may also be used in unstratified shallow lakes. Internal loading of P may also be high in unstratified, shallow, eutrophic lakes in which the sediment–water interface is usually oxic (Søndergaard et al., 1999). Internal loading and whole-lake TP may decrease in shallow, stratified lakes following circulation; however, the concentration available for growth in the photic zone may increase (Osgood & Stiegler, 1990). Therefore, depth is an important criterion in determining the application of complete circulation to shallow lakes, for not only will phytoplankton production be related to available light, but so too will internal P loading. Unless oxic conditions will substantially reduce P internal loading, maintaining stratified conditions may be preferable for limiting P availability in the photic zone.

10.1 Case Studies and Lake Response

Multiple lakes were treated by artificial circulation and reviewed by Pastorak et al. (1981): their associated characteristics are summarized in Table 10.1.

Table 10.1: Lakes treated by artificial circulation with associated characteristics

Lake	Depth (m)			Vol. ($10^6/m^3$)	Area (ha)	Q Air/m^3/min	Q Air/m^3 x 10^6	Q Air/km^2	Reference
	Max.	Mean	Device						
Canada									
Corbett, BC	19.5	7.0	19.5	1.689	24.2	4.5 [a]	2.66	18.52	Halsey, 1968.
Buchanan, ON	13	4.9	13.0	0.42	8.9	0.28 [a]	0.67	3.17	Cooke et al., 2005.
Heart, ON	10.4	2.7	10.0	0.382	14.5	0.23 [a]	0.58	1.56	Nicholls et al., 1980
United States									
Casitas, CA	82.0	26.8	39.0	308.0	1,100	17.84 [b]	0.06	1.62	Barnett, 1975.
Parvin, CO	10.0	4.4	10.0	0.849	19.0	2.1 [a]	2.5	11.18	Lackey, 1972.
Altoona, GA	46.0	9.4	42.7	453	4,800	21.6 [b]	0.05	0.45	USAE, 1973.
Wahiawa, HI	26.0	8.0	2.7	1.7	20.0	2.4 [b]	1.4	12.0	Devick, 1972.
Kezar, NH	8.2	2.8	8.2	2.008	73.0	2.83 [a]	1.41	3.88	Haynes, 1973.
Roberts, NM	9.1	4.4	9.1	1.233	28.3	3.54 [a]	2.87	12.5	USEPA, 1970.
Silver, OH	12.0	4.22	10.0	1.68	38.44	3.37 [b]	2.01	8.77	Brosnan, 1983.
Clines Pond, OR	4/9	2.5	4.9	0.003	0.13	0.028 [a]	10.2	21.6	Malueg et al., 1973.
Prompton, PA	10.7	3.7	10.7	0.193	112.0	4.53 [a]	1.08	4.04	McCullough, 1974.
Hyrum, UT	23.0	11.9	15.2	23.1	190.0	2.83 [b]	0.17	1.49	Drury et al., 1975.
Cox Hollow, WI	8.8	3.8	8.8	1.480	38.8	4.08 [a]	2.76	10.53	Wirth et al., 1970.
Mirror, WI	13.1	7.6	12.8	0.40	5.3	0.45 [a]	1.13	8.55	Smith et al., 1975.
Europe									
Wahnbach, Germany	43.0	19.2	43.0	41.618	38.8	4.08 [b]	2.76	10.53	Bernhardt, 1967.
Starodworskie, Poland	23.0	-	23.0	1	7.0	0.27 [a]	-	3.81	Lossow et al., 1975.
Växjosjön, Sweden	6.5	3.5	6.0	3.1	87.0	7.2 [a]	2.32	8.28	Bengtsson & Gelin, 1975.
Pfaffikersee, Switerland	35.0	18.0	28.8	56.5	325.0	6.0 [c]	0.11	1.85	Thomas, 1966.
Maarsseveen, U.K.	29.9	14.0	19.0	8.018	60.7	2.49 [a]	0.31	4.10	Knoppert et al.., 1970.
Australia									
Tarago, Australia	23.0	10.5	14.0	27.6	360	3.0 [c]	0.08	0.83	Bowles et al., 1979.

[a] Flow rate produced destratification.
[b] Partly mixed.
[c] Flow rate inadequate to destratify.
Notes: Max. and mean depth= depth of the lake; Device= device depth; volume= volume of the lake; Q= flow rate.

Source: Modified from Pastorak et al. (1981)

10.1.1 Squaw Lake, Wisconsin, United States

Squaw Lake, Wisconsin did not respond to artificial circulation. The lake is naturally divided into two basins; south (9.1 ha, 2.55 m mean depth) and north (16.8 ha, 2.92 m mean depth). In the summer of 1993, the south basin was artificially circulated and enriched with CO_2. The pH exceeded 10 in the north basin but remained around 7 in the enriched south basin during circulation. Despite the contrasting pH/CO_2 condition, populations of *Aphanizomenon* and *Anabaena* reached levels exceeding 300 µg/L Chl-*a* in both basins (Shapiro, 1997). Increased circulation favoured non-buoyant algae that would otherwise tend to sink under stable conditions. Mixing rate and neutralization of buoyancy regulation may have been more important than CO_2/pH to alter blue-green algae dominance in this lake.

10.1.2 Lake Nieuwe Meer, the Netherlands

A whole-lake mixing application in Lake Nieuwe Meer, the Netherlands (132 hectares, 18 m mean depth, 230 m max. depth) resulted in decreased dominance of *Microcystis*, a toxic cyanobacteria, and a shift in algal dominance to a mixed community of flagellates and diatoms during two summers of complete circulation (1993–1994) (Spencer & King, 1987). *Microcystis* decreased from 90 per cent to <5 per cent of the biomass. This decrease was caused by buoyancy loss and entrainment of *Microcystis* through the water column, which increased with greater distance from the diffuser plumes. Lake Nieuwe Meer's system design and mixing rate (~1 m/h) was an important factor to control cyanobacteria because the rate exceeded the mean flotation velocity of *Microcystis* (0.11 m/h). The velocity was achieved with an overall airflow rate of 9.9 m^3/km^2 per min, where nutrient content was high before and during mixing (TP 420–450 µg/L and SRP 350–380 µg/L). The high TP and SRP before and during mixing showed that circulation can restrict the abundance of buoyant cyanobacteria despite nutrient level. Lake Nieuwe Meer is relatively deep, and mixing can effectively reduce cyanobacteria in much shallower lakes (Spencer & King, 1987).

10.1.3 Crystal Lake, Minnesota

A worst-case result from inadequate airflow rate can be understood from the destratification of Crystal Lake, Minnesota (3 m mean depth, 10.4 max depth). Internal P loading significantly increased 2–3 fold (TP, TN and Chl-*a*) following resumed circulation, with a proportional decrease in transparency (Osgood & Stiegler, 1990). Airflow rate (4.7 m^3/km^2 per min) was inadequate to provide continuous oxic conditions at the sediment–water interface. A similar experience occurred in East Sydney Lake, New York.

10.1.4 Lake Wilcox, Ontario

Lake Wilcox, located in southern Ontario, also experienced worsened lake quality due to complete circulation. Circulation promoted cyanophyte *Planktothrix rubescens* blooms, which previously did not occur (Nürnberg & LaZerte, 2003). Circulation produced increased blooms from continued entrainment of P and algae throughout the water column, and exposure to higher light conditions.

10.2 Effects and Precautions

10.2.1 Beneficial Effects of Artificial Circulation on Lake Quality

Four indicators of the effects of artificial circulation are DO, ammonium, epilimnetic pH and the trace metals iron and manganese (Table 10.2). DO increased and trace metal decreased in a high percentage of cases studied by Pastorak et al. (1982). Positive changes in ammonium and pH were less frequent. Changes in all variables were statistically significant and are a result of increased contact of a mixed water column with the atmosphere.

Circulation can reduce phytoplankton biomass by increasing the depth of mixing of plankton cells and increasing light limitation. This is known as the "critical depth"[5] concept and can be determined using information on

[5] The Critical Depth Hypothesis formalized by Sverdrup in 1953 suggests that "phytoplankton blooms occur when surface mixing shoals to a depth shallower than a critical depth horizon, defining the point where phytoplankton growth exceed losses"(Behrenfeld, 2010).

light at the surface, the algal photosynthetic compensation depth and the extinction coefficient. Increased circulation also usually results in the complexation and precipitation of Fe and Mn, reducing trace elements and P internal loading, and therefore algal biomass. If sediments are disturbed by mixing, algal biomass may also decrease due to decreased light availability (Cooke et al., 2005).

Furthermore, aeration is widely used in Canada and around the world to reduce O_2 depletion with the goal of improving habitat conditions for warm-water fish. This speaks to the importance of clearly identifying the goals of a restoration project from the outset. Aeration may not reduce algae, but still enhance fish production and prevent winterkill.

Table 10.2: Summary of lake responses to artificial circulation

Parameter	n		Lake Response			X2
			+	-	0	
Δt after	45	No. %	15 33	30 67	-	5.0 [a]
SD	19	No. %	4 21	10 53	-	6.5 [a]
DO	41	No. %	33 80	1 1	2 5	55.2 [c]
Phosphate	17	No. %	3 18	5 29	7 41	1.6
TP	20	No. %	5 23	6 30	8 40	0.74
Nitrate	20	No. %	7 35	8 40	3 15	2.33
Ammonium	20	No. %	3 15	13 65	3 15	10.5 [b]
Iron/manganese	22	No. %	-	20 91	2 9	33.1 [c]
Epilimnetic pH	21	No. %	1 5	9 43	8 38	6.33 [a]
Algal density	33	No %	6 18	14 42	8 24	3.71
Biomass Chl-*a*	23	No %	5 22	6 26	6 26	0.12
Green algae	18	No %	7 39	4 22	4 22	1.0
Blue-green algae	25	No %	5 20	13 52	13 52	5.57
Ratio green: blue-green algae	21	No %	11 52	3 14	3 14	4.9

Note: Diffused-air systems only.
Δt after = temperature difference between surface and bottom water during artificial mixing:
+ means Δt > 3 °C; - mean Δt <3 °C.
[a] p<0.05. Goodness-of-fit test to uniform frequency distribution for +,-, 0 responses: [b] <0.0: [c] <0.001

Source: Pastorak et al. (1982).

10.2.2 Undesirable Effects of Artificial Circulation on Lake Quality

Artificial circulation has potential adverse effects on lake quality. If circulation increases the suspension of particulate material, associated P may mineralize and become available to phytoplankton (Cooke et al., 2005). Mixing of sediment may increase inorganic turbidity, and care must be taken to avoid this problem.

Furthermore, transparency may decrease following treatment, if (1) the photic zone is initially nutrient-limited such that phytoplankton increase following upward circulation of nutrients previously entrained in the hypolimnion; (2) circulation is too weak, resulting in microstratification that favours buoyant blue-green algae with a more favourable light climate for productivity; or (3) circulation is so intense that particulate matter becomes resuspended.

Whole-lake circulation will result in the loss of deep coldwater habitat for fish in stratified lakes, and overall lake temperatures typically increase following treatment. Temperature increase from 15 to 20°C in the hypolimnetic waters, as a result of complete circulation, may have adverse effects on coldwater fish and other biota (Pastorak et al., 1981). If mixing is initiated during stratification, care must be taken not to introduce large amounts of low-oxygen water containing toxic substances (NH_4, H_2S) with potentially catastrophic impacts to biota. If the lake is used for drinking water, introduction of these substances must be undertaken with special care to avid toxic effects or odour problems. Lake mixing may affect ice formation, and precautions must be taken to protect the public.

10.3 Cost

Lorenzen & Fast (1977) referred to an annual cost of $262,400 (2015 USD) for artificial circulation application. The cost included two air compressors, producing an airflow rate of 34.3 m^3/min at standard conditions, pipes and air diffusers. At the recommended rate of 9.2 m^3/km^2 per min, this represents $700/ha for the first year of operation (2015 USD) (Cooke et al., 2005). Dierberg & Williams (1989) reviewed 13 projects in Florida, where cost ranged from $520 to $6,100/ha for initial costs, and $150 to $2,940/ha for annual costs (2015 USD). Median values for initial and annual costs, respectively, were $1,287 and $575/ha (2015 USD). Cost increases with lake size, although cost per hectare declines, demonstrating economies of scale (Ashley 1987; Ashley & Nordin, 1999).

For a more complete analysis, costs for a compressor, pipe, installation and 1-year of operation for a system at a rate of 6 m^3/min were approx. $73,000 or $610/ha (2015 USD) (Davis, 1980). Other similar systems installed were approx. $95,000 ($830/ha or greater) (2015 USD) (Cooke et al., 2005). Moreover, there is usually an economy of scale for circulation projects. The cost for 33 projects installed by General Environmental Systems during 1991-2002 were averaged for water bodies of a particular size (Table 10.3)

Table 10.3: Project cost (n=33) installed by General Environmental System (1991-2002)

Water Body Size (ha)	No. Projects	Cost *
> 53 hectares	17	$760/ha
23-25 hectares	4	$1,680/ha
<10 hectares	12	$7,743/ha

** Cost (2002) $588/ha (>53 hectares); $1,295/ha (23-25 hectares); and $5,960/ha (<10 hectares)*
Source: Data obtained from Cooke et al., 2005.

Aeration was also examined for Twin Lake, Minnesota in 2013. The 20-year lifetime cost was determined to amount to $935,000 (installation cost $160,000; $400/meter; and $35,000 yearly maintenance) (Barr, 2013). The most frequently used method currently is the SolarBee, and depending on the model, costs approximately $40,000 per circulator (ICLEI, 2006).

Artificial Circulation Summary

Artificial circulation, similar in concept to hypolimnetic aeration, is achieved by pumps, jets and diffused air; however, circulation of the entire lake is generally intended rather than a select region or depth. Unlike with hypolimnetic aeration, the temperature of the whole lake will increase with complete circulation if mixing includes water that was previously part of the cooler hypolimnion. The principal improvements in water quality caused by complete circulation are oxygenation and chemical oxidation of substances in the entire water column, as well as enlarging the suitable habitat for aerobic warm-water species. Circulation improves dissolved oxygen and reduces iron and manganese, as well as causes light to limit algal growth in environments where nutrients are uncontrollable and neutralize the factors favouring the dominance of blue-green algae.

Beneficial Effects

- Circulation can reduce phytoplankton biomass by increased depth of mixing of plankton cells and increased light limitations.
- Increased circulation usually results in the complexation and precipitation of Fe and Mn, reducing trace elements and P internal loading, therefore algal biomass.
- If sediments are distributed by mixing, algal biomass may also decrease due to decreased light availability.
- Improvement of warm-water fisheries.

Undesirable Effects

- If circulation increases the suspension of particulate material, associated P may mineralize and become available to phytoplankton.
- Mixing of sediment may increase inorganic turbidity.
- Whole lake circulation will result in the loss of deep coldwater habitat for fish in stratified lakes.
- Overall lake temperatures typically increase following treatment.

Suitable Lake Conditions

Four indicators of the effects of artificial circulation are DO, ammonium, epilimnetic pH and the trace metals iron and manganese.

Successful implementation of artificial circulation as reviewed in the literature.

Lake size: 9.1–18 hectares.

However, successful implementation in a large lake, Lake Nieuwe Neer (Netherlands) 132 hectares in size.

Depth (mean): 3.5–28.4 m and lake must be stratified.

Depth (max.): 2.6–3 m.

Longevity: Continual treatment and management.

Cost

Cost increases with lake size, although costs per hectare decline, demonstrating economies of scale.

Lakes > 53 hectares: $760/ha.

Lakes 23–25 hectares: $1,680/ha.

Lakes <10 hectares: $7,743/ha.

Twin Lake, MN:

20-year lifetime cost: $935,000 (USD, 2013).

Maintenance cost: $35,000 per year.

Initial cost: $520–$6100 per hectare

Annual cost: $150–$2940 per hectare.

11. DREDGING AND REMOVAL OF SURFICIAL SEDIMENT

Many shallow, eutrophic lakes do not stratify thermally, making them susceptible to continual or periodic nutrient inputs from sediment. For lakes where significant nutrient loading from sediment occurs, removal of nutrient-rich surficial sediments has the potential to reduce the rate of internal nutrient recycling, improving overall lake water quality. In addition to removing nutrients in bottom sediments, removal may also decrease cyanobacterial inocula. There are two major techniques for sediment removal from freshwater lakes and reservoirs: lake drawdown followed by bulldozer and scraper excavation; and dredging. Lake drawdown followed by excavation has limited application and has been used more successfully in small reservoirs due to the limitation that water must be drained or pumped from the basin (Born et al., 1973). When considering whether dredging is a feasible remediation technology, multiple factors must be considered, including equipment used, area dredged and depth, and impoundment location. Multiple dredges have been reviewed in the literature and can be categorized in two groups: mechanical and hydraulic. As with many of the other techniques discussed in this review, sediment removal will be effective for reducing the symptoms of eutrophication only if nutrient loading to the system can be significantly reduced. Otherwise, dredged sediment will only be replaced with newly settling sediment with high nutrients. Dredged sediments must also have higher nutrient content than underlying sediments for this technique to be effective.

Mechanical Dredges

Grab mechanical dredges are commonly used in lake restoration. A disadvantage of grab-bucket dredges is the discharge of material in the immediate vicinity of the removal area or into barges or trucks for transportation to the disposal area. Grab buckets commonly create turbid water conditions and reduced pickup efficiency because much lake sediment is highly flocculent. Advantages of grab-bucket dredges are ease of transport from one location to another and the ability to work in relatively confined areas. A grab bucket operates most efficiently in near-shore areas that contain soft to stiff mud, where efficiency declines rapidly with depth due to the time-consuming operating cycle.

Hydraulic Dredges

Several variations of hydraulic dredges exist, including the suction dredge, the hopper, the dustpan and the cutterhead suction dredge. Inland lake sediment removal is most commonly accomplished with a cutterhead hydraulic pipeline dredge (Cooke et al., 2005). Hydraulic dredges have the advantage of fewer problems related to sediment resuspension.

Disposal Area Selection

Once dredging equipment is selected, another concern is disposal area and the possible treatment for toxic substances contaminating the sediment. A challenge to disposal area is containment design of an appropriate size and retention time to hold the dredged material volume. Upland disposal is common and guidance for design, operation and management is available (USACOE, 1987). It is important to characterize the dredged sediment, including water content, organic content, metal concentration, as well as grain size to determine sedimentation rate. Monitoring requirements for the disposal site (water flowing from the site) include analysis of TSS, ammonium-nitrogen, oil and grease, chlordane, pH, temperature, nitrate nitrogen, TP and DO.

11.1 Case Studies and Lake Response

11.1.1 Lake Trummen, Sweden

Prior to 1959, Lake Trummen (Sweden) received sewage discharge from domestic and industrial sources; however, despite wastewater diversion, the lake showed little sign of recovery. In 1970 and 1971, 50 cm of surficial sediment was dredged from the main lake basin, increasing the mean depth of the lake from 1.1 to 1.75 m, and the maximum depth from 2.1 to 2.5 m (Andersson, 1984). The total volume of sediment removed was approx. 30 x 10^5 m^3, and the sediment skimming treatment reduced the total P content in the surface layer from approximately 0.78 mg/kg to 0.03 mg/kg (Sjön Trummen I Växjö, 1977). TP in the surface water of the lake after dredging decreased 90% (600 µg/L to a range of 70-100 µg/L) and TN concentrations by 80% (6.3 to 1.3 mg/L) (Andersson, 1984). Due to a temporary increase of phosphorus in 1975, which was associated with a large influx of planktivorous cyprinid fish, approx. 2 metric tonnes (30 kg/ha) of fish were removed from the lake in 1976. Biomanipulation discontinued in 1979.

The nutrient reductions correspondingly resulted in biological changes. The Shannon diversity index for phytoplankton increased from 1.6 (1968) to 3.0 in 1973 (Cronberg et al., 1975) and SD transparency increased from 23 to 75 cm. Cyanobacterial biomass decreased significantly and nuisance species *Oscillatoria agardhii* were no longer present (Cronberg et al., 1975). Phytoplankton productivity declined from 370 g C/m^3 in 1968–1969 to 225 g C/m^3 in 1972–1973. The effect of dredging on the benthic community of Lake Trummen was determined as negligible due to a consistency in the number of benthic organisms before and after the treatment (Andersson et al., 1975).

Some of the dredged sediment was disposed of in shallow, diked-off bays and the remainder in upland diked ponds, where return flow was treated with aluminum sulfate to reduce TP concentration from 1 mg/L to 30 µg/L. Dried dredge material was also sold as topsoil. There is a strong indication that the effects of dredging at Lake Trummen continue today and have lasted over 40 years. However, there are two key factors that are recognized in the success of this lake treatment. First, there was continual fish management following dredging (the removal of rough fish such as roach and bream) from 1976 through to 1978. Secondly, the dredge was modified with a special suction nozzle that allowed precision selection of soft sediment removal.

11.1.2 Lilly Lake, Wisconsin, United States

Lilly Lake, located in southeastern Wisconsin, is 37 hectares in size and its watershed is 155 hectares (Dunst, 1981). The lake suffered years of infilling by aquatic plants and by 1977, shoaling had reduced the lake to a maximum depth of 1.8 m and a mean depth of 1.4 m. The basin contained more than 10 m of partially decomposed plant material and infilling by plant material reached 0.5 cm/year (Dunst, 1981). Chemical eradication and restocking of centrarchids and northern pike failed due to severe winterkill (Wisconsin Department of Natural Resources [WDNR], 1969).

Restoration of Lilly Lake for fish management was recommended by the Wisconsin Department of Natural Resources (WDNR), and deepening at least 10 per cent of the basin to a depth of 6 cm was proposed in 1969.

Hydraulic suction dredging was undertaken from 1978 to 1980, and by 1981 TP concentrations were significantly lower than concentrations prior to dredging (Table 11.1). Chl-*a* concentrations were slightly higher in 1980 and 1981 than in the pre-dredging year of 1977, however, the increase was relatively minor (approx. 3–4 µg/L) (Dunst, 1981). The most significant result was in water storage capacity of the lake, basin volume, with an increase by 128 per cent. A study conducted in 1989 showed further reductions in Chl-*a* ranged from 0.9 to 5.3 μg/L (mean = 2.6 μg/L) (Garrison, 1989). While the lake never demonstrated algal issues and was remediated due to macrophyte domination, low Chl-*a* continued in combination with a diverse macrophyte community.

Table 11.1: Limnological parameters immediately following dredging and a decade later in Lilly Lake, WI

Parameter	Year	
	1981	1989
TP (μg/L)	14	9
TN (mg/L)	1.1	0.8
Total inorganic N (TIN) (mg/L)	0.02	0.02
Chl-*a* (μg/L)	<5	3.3
SD (m)	2.2	4.0
Note: Chl-*a* values are the summer mean for 1981 (June- Sept) and annual mean for 1989. Important to note that summer means would be higher than annual means.		

Source: Garrison & Ihm (1991).

11.2 Effects and Precautions

There are multiple reasons for removing sediment from a lake, including deepening, limiting nutrient recycling, reduction of macrophyte nuisances by subsequent light limitation and removing toxic sediments. Environmental concerns associated with lake sediment removal are often less serious than with other technologies, such as sediment capping. Case studies reveal that in-lake adverse impacts are often short-lived; however, disposal of dredged material can be problematic.

11.2.1 Suitable Lake Conditions

Dredging is generally limited to shallow lakes; however, lake area is not a constraint (Peterson, 1981; 1982a). Peterson (1979) examined 64 lake dredging projects, showing that lake sizes ranged from less than 2 hectare to over 1,050 hectares. Sediment volume removed ranged from a few hundred to over 7 million cubic meters.

A factor that can limit dredging of a large inland lake is the requirement for a large disposal area. Restoration most frequently is sought for lakes in high use areas, where sediment disposal space can be restricted. To overcome this limit, various productive uses of dredged material have been proposed (Lunz et al., 1978; Spaine et al., 1978; Walsh & Malkasian, 1978). With Nutting Lake, MA, 153 x 10^3 m^3 of sediment was sold as soil at $1.40/$m^3$, reducing the total dredging cost by $215,000 and per-unit dredging cost to approximately $1/$m^3$ (Worth, 1981).

Depth, size, disposal area, watershed area and sedimentation rates are important physical variables that affect treatment feasibility. Dredging will only be effective for reducing eutrophication in lakes with highly nutrient-enriched surface sediments relative to underlying sediment (e.g. Lake Trummen) (Andersson et al., 1975; Bengtsson et al., 1975). Dredging depth depends on the purpose of the treatment and the sediment characteristics of the lake. Sediment depth may vary considerably, depending on the basin configuration at the time of the lake formation or the transport of sediment to the lake via stream inlets. Several variables, cited previously, determine the suitability of a lake for dredging; however, generally the most suitable lakes have shallow depths, low sedimentation rates, organically rich sediments, relatively small water-to-surface ratios

(10:1), long hydraulic residence time and the potential for extensive use following dredging (Cooke et al., 2005). Furthermore, Kleeberg & Kohl (1999) determined at least a 50 per cent reduction in external nutrient load is required.

11.2.2 Beneficial Effects of Dredging on Lake Quality

The reasons for removing sediment from a lake include deepening, limiting nutrient recycling, reducing macrophyte nuisance, and removing toxic sediments. Many projects designed to control internal loading cycling show improvements in lake quality.

11.2.3 Undesirable Effects of Dredging on Lake Quality

Sediment resuspension during dredging is a concern when evaluating the feasibility of this technology. Resuspension of sediments has negative impacts on many aquatic organisms including clogging of filtering apparatus of benthos and zooplankton and reductions of light for primary producers. Many fish species cannot tolerate high sediment loads. An issue associated with resuspension is nutrient liberation from disturbed sediments and porewaters. Another concern is the release of toxic substances that are often associated with fine particles in some systems. Extra consideration of these issues is required if the lake is used for drinking water.

Another concern is the destruction of benthic fish-food organisms and its effect on a lake's food web. However, reestablishment of benthos can be promoted if portions of the bottom are left undredged. Complete removal of an entire basin's surficial sediment may require two to three years to reestablish (Carline & Brynildson, 1977). Drawdown of the entire lake to expose littoral sediments or the entire basin is more destructive to the benthic community than dredging. An obvious drawback of this technique is that the basin must be allowed to dewater sufficiently before equipment can operate. Lake draining will result in the mortality of most of the native aquatic biota.

11.3 Cost

The main objection to dredging in many cases is the high cost. Project-to-project cost comparison for sediment removal is difficult due to a large number of variables that affect dredging costs. These variables include equipment, project size or volume of material removed, disposal site availability, density of material being removed, distance to disposal area, and use of the removed material. Generally, the per-unit volume cost of sediment removal is inversely related to the total volume of material removed (Cooke et al., 2005). Data assembled by Peterson (1981) from 64 sediment removal projects in the United States indicate a cost range from \$0.47/m^3 to \$27/m^3 (2015 USD). Blazquez et al. (2001) estimated costs between \$34–\$1,409/m^3 for removal of contaminated sediments.

In 1996, a bid for hydraulic removal of muck from Lake Macy in Lake Helen, Florida quoted \$4.00/yd^3 (\$5.23 m^3) for removal of up to 20,000 yd^3 (15,292 m^3). Removal beyond 20,000 yd^3 was quoted at a rate of \$3.50 yd^3 (\$4.58 m^3). Additional costs to this bid included \$57,000 to remove 7.69 hectares of aquatic vegetation and \$60,000 for mobilization and installation of a temporary pipeline (McDougal Construction, 1996). An additional cost can also be sediment contamination with toxic substances, where special dredges or treatment methods are required and dredging costs may exceed \$65/m^3 (2015 USD)(Barnard & Hand, 1978; Koba et al., 1975; Matsubara, 1979). Any time dredged material can be used a potting soil or topsoil, as in the case at Lake Trummen, Sweden (Sjön Trummen Väjöm 1977), Lilly Lake, Wisconsin (Dunst et al., 1984) and Paradise Lake, Illinois (Lembke et al., 1983), the overall project costs could be reduced significantly.

Dredging was reviewed for Twin Lake, Minnesota at a total cost of $2,570,000 (Barr, 2013). Removing 15 cm of sediment would have yielded approx. 13,500 cubic metres of dredge material for a total capital cost of $1.7 million. However, due to space limitations surrounding the lake, on-site disposal of the dredged material and high cost determined the treatment to be unfeasible.

For Lake Trummen, Sweden, the 1971 cost of the entire project was $572,222, approximately $5,722/ha. If the cost is amortized over the post-treatment years the lake showed benefit (over 35 years), the costs are reduced to approximately $163/ha per year ($940/ha, 2015 USD). At today's cost, the project would cost $3.3 million (2015 USD).

Peterson (1981) compared dredging and P inactivation costs by assuming that both treatments are aimed at controlling in-lake P cycling (Table 11.2). Therefore, treatment costs reflect the amount of P removed (dredging) or bound into the lake system (P inactivation) to prevent excessive internal cycling. However, direct comparisons of the two treatments should not be made since P inactivation costs are usually based on the materials and labour required to treat a hectare of lake surface (area), while dredging costs are based on the cost of removing a cubic meter of sediment (volume). The costs in Table 11.2 were calculated as end-of-treatment costs.

Table 11.2: Per-hectare cost comparison of dredging and alum treatment to control in-lake P dynamics

Lake	Treatment	Physical and Chemical Data		Chemical and Dose	Sediment removed (m^3)	Treatment cost ($/ha)
Horseshoe Lake, Wisconsin	Liquid alum	AO = 8.9 ha V = 3.6×10^5 m^3 Z_{max} = 16.7 m Z = 4.0 m	Alk = 218-278 mg/L pH = 6.8-8.9 Dimictic	2.6 g Al/m^3	-	150
Lake San Marcos, California	Liquid alum	AO = 18.2 ha V = 4.3×10^5 m^3 Z_{max} = 2.3 m	Z = 2.3 m Alk = 190-268 mg/L pH = 7.3-9.1	6.0 g Al/m^3	-	189
Welland Canal, Ontario	Liquid alum, surface application	AO = 74 ha V = 6.2×10^6 m^3 Z_{max} = 9.0 m	Z = 7.8 m Alk = 109 mg/L Dimictic	2.5 g Al/m^3	-	306
Mirror Lake, Wisconsin	Liquid alum and aeration	AO = 5.1 ha V = 4×10^5 m^3 Z_{max} = 13.1 m Z = 7.8 m	Alk = 222 mg/L pH = 7.6 Monomictic	6.6 g Al/m^3	-	600[a]
Shadow Lake, Wisconsin	Liquid alum	AO = 5.1 ha V = 9.1×10^5 m^3 Z_{max} = 12.4 m Z = 5.3 m	Alk = 188 mg/L pH = 7.4 Dimictic	5.7 g Al/m^3	-	600[a]
Cline's Pond, Oregon	Liquid sodium aluminate and HCl	AO = 0.4 ha V = 9600 m^3 Z_{max} = 4.9 m Z = 2.4 m	Alk = 30-50 mg/L pH = 7.0-7.7 Monomictic	10.0 g Al/m^3	-	630
West Twin Lake, Ohio	Liquid alum	AO = 34 ha V = 14.2×10^4 m^3 Z_{max} = 11.5 m	Z = 4.4 m Alk = 102-149 mg/L Dimictic	26.0 g Al/m^3	-	638
Dollar Lake, Ohio	Liquid alum	AO = 2.2 ha V = 0.86×10^5 m^3 Z_{max} = 7.5 m Z = 3.9 m	Alk = 101-127 mg/L pH = 6.7-8.6 Dimictic	20.9 g Al/m^3	-	756
Medical Lake, Washington	Liquid alum	AO = 64 ha V = 6.4×10^6 m^3 Z_{max} = 18 m Z = 10 m	Alk = 750 mg/L pH = 8.5-9.5 Dimictic	12.2 g Al/m^3	-	2,610[b]
Half Moon Lake, Wisconsin	Dredging (~ 30% of basin)	AO = 64 ha V = 8.9×10^5 m^3	Z_{max} = 2.7 m Z = 1.7 m	-	25×10^3	3,205
Lilly Lake, Wisconsin	Dredging (100% of basin)	AO = 35.6 ha V = 5.3×10^8 m^3	Z_{max} = 1.8 m Z = 1.4 m	-	680×10^3	6,876
Commonwealth Lake, Oregon	Dredging (100% of basin)	AO = 2.6 ha	Z = 0.9 m	-	19×10^3	8,653
Steinmetz, New York	Draining and bulldozing (75% of basin)	AO = 1.2 ha V = 18.1×10^5 m^3	Z_{max} = 2.1 m Z = 1.5 m	-	2×10^3	10,849
Carnegie Lake, New York	Dredging (75% of basin)	AO = 110 ha V = 765×10^3 m^3	Z_{max} = 3.0 m	-	-	-

Lake	Treatment	Physical and Chemical Data		Chemical and Dose	Sediment removed (m^3)	Treatment cost ($/ha)
Lenox Lake, Iowa	Dredging (100% of basin)	AO = 13.4 ha Z_{max} = 3.4 m	Z = 0.9 m	-	76 x 10^3	19,992
Sunshine Springs, Wisconsin	Dredging (100% of basin)	AO = 0.4 ha V = 1.7 x 10^3 m^3	Z_{max} = 1.2 m Z = 0.5 m	-	5.1 x 10^3	40,248
Krause Springs, Wisconsin	Dredging (100% of basin)	AO = 0.3 ha V = 0.98 x 103 m^3	Z_{max} = 1.0 m Z = 0.34	-	4.9 x 10^3	47,631
Collin Park Lake, New York	Dredging (15% of basin)	AO = 24.3 ha V = 6.5 x 10^5 m^3	Z_{max} = 9.75 m Z = 2.7 m	-	52 x 10^3	58,767

Note: Costs were calculated from data of Peterson (1981) and Cooke & Kennedy (1981), and adjusted to June 1991 costs using inflation factors (1.28 x table costs = 2002 USD). Labour cost for P inactivation assumed 8 hour workday at $5/hour (1982 cost).

[a] Cost per hectare, spread over Mirror Lake and Shadow Lake as a unity (5.1 ha + 17.1 ha = 22.2 ha).

[b] Source: Gasperino et al., 1981.

Source: Peterson (1981); Cooke & Kennedy (1981).

Dredging and Removal of Sediment Summary

For lakes where significant nutrient loading from sediment occurs, removal of nutrient-rich surficial sediments has the potential to reduce the rate of internal nutrient recycling, improving overall lake water quality. In addition to removing nutrients in bottom sediments, removal may also decrease cyanobacterial innocula.

Beneficial Effects	Undesirable Effects
• Lake deepening. • Expand habitat. • Limit nutrient recycling. • Reduce macrophyte nuisance. • Remove toxic sediment.	• Resuspension of sediments on aquatic organisms including clogging filtering apparatus of benthos and zooplankton, and reduction of light. • Many fish species cannot tolerate high sediment loads. • Nutrient liberation from disturbed sediments and porewaters. • Potential release of toxic substances associated with fine particulars (polluted). • Destruction of benthic fish-food organisms and its effect on a lake's food web. • Lake draining will result in the mortality of most native aquatic biota.

Suitable Lake Conditions

Dredging is generally limited to shallow lakes (<3 m) but lake area is not a constraint. Depth, size, disposal area, watershed area and sedimentation rates are important physical variables that affect treatment feasibility. Successful implementation of dredging as reviewed in the literature:

Lake size: Area is not a constraint.

Successful implementation reviewed in the literature ranges from 2–1,000 hectares.

Lake depth: Highest success <3 m.

Depth as reviewed in the literature: 0.5–9.75 m (max. depth).

Sediment depth: Dredging will only be effective in lakes with high nutrient-enriched surface sediments relative to underlying sediment. Sediments are the source of internal loading and the bulk of nutrients are located in the top 0.3–0.5 m of a sediment core, then removal of that layers by dredging should provide reliable and permanent solution, although costly (Cooke et al., 1993).

Sedimentation rate: Low

Water-to-surface ratio: Small, 10:1.

Hydraulic retention time: Long.

Watershed sourced loading: Requirement is a reduction in external nutrient load of at least 50%.

Longevity: Long-term benefit of removing the nutrient source.

Cost

Main objection to dredging is the high cost. Project-to-project cost comparison for sediment removal is difficult due to a large number of variables that affect dredging cost, for example equipment, volume of material removed, and density of material.

\$34–\$1,409 per m^3 for removal of contaminated sediments (Balzquez et al., 2001).

\$1–\$30 per m^3; \$3,200–\$60,000 per hectare (as reviewed in the literature).

Twin Lake, MN: Total cost \$2,570,000 (Barr, 2013).

12. PHOSPHORUS INACTIVATION AND SEDIMENT CAPPING

Sediments play an important role in the phosphorus cycle in lakes, and elements present in lake sediment provide sorption sites for P including iron (Fe), aluminum (Al) and calcium (Ca) (Huser & Pilgrim, 2014). In most lakes, P is permanently buried in lake sediments over time, but temporary P release can occur from reductant-soluble elements and mineralization of P containing organic compounds (Boström & Pettersson, 1982; Nürnberg, 1988; Hupfer et al., 1995; Huser & Pilgrim, 2014). If additional adsorbate, such as Al, Fe, or calcite is added to sediment, recycling of mobile P from sediment can be reduced (Huser & Pilgrim, 2014). Alternatively, sediments may be physically capped to prevent diffusion of nutrients to the overlying water column. Without reductions of external loads, however, these approaches will have only a short-term effect on nutrient recycling from sediments because the adsorptive capacity of added materials will ultimately be overwhelmed and new organic matter rich in nutrients will settle and bury capping materials. Following external load reductions, internal P release to the water column may prolong a lake's enriched state and support continued algal blooms (Cullen & Forsberg, 1988; Sas et al., 1989; Jeppesen et al., 1991; Welch & Cooke, 1995; Scheffer, 1998).

Many different methods of sediment capping and P inactivation have been used in lake restoration projects including physical (mechanical or passive) capping and active capping using alum, calcium, zeolite, Phoslock™, and iron. Passive capping using sand, gravel, or clay is used to decrease diffusion of nutrients and contaminants to the overlying water column and bury them deeper in the sediments (Wang et al., 1994). Diffusion rates are decreased with the use of finer materials and thicker caps. Capping thickness typically exceeds 5 cm, which limits the approach to small ponds or reservoirs because of the large volume of material required and difficulties depositing a uniform layer (Hickey & Gibbs, 2009). A disadvantage of the approach is that it smothers benthos and capping materials often provide poor benthic habitat. As a result of these limitations, physical capping agents are rarely used, and active capping is more common. Active capping methods are designed to increase the binding capacity of sediments to retain P and other contaminants. Below, we review the most commonly used P-inactivation approaches.

12.1 Alum

Aluminum salt, either aluminum sulfate (alum), sodium aluminate, or both, can be added to the water column to form aluminum phosphate and a colloidal aluminum hydroxide floc to which certain P fractions bind. The aluminum hydroxide floc settles to the sediments and continues to sorb and retain P within the lattice of the molecule, even under reducing conditions (Cooke et al., 2005). Alum has been used for coagulation in water treatment for over 200 years and is the most commonly used drinking water treatment in the world (Søndergaard et al., 1990). Despite the increasing use of aluminum (Al) additions for restoring eutrophic lakes, the solubility and chemistry of Al is still not completely understood (Welch & Cooke, 1999; Cooke et al., 2005). However, generally, alum is not suitable in shallow, energetic lakes because it is flocculent and easily resuspends (Egemose et al., 2010).

Cooke et al. (2005) determined that P inactivation longevity does not typically exceed 15 years and will depend on PO_4 release rates and application dose (Gibbs et al., 2011). Welch & Cooke (1999) documented treatment longevity between 4–21 years in stratified lakes, and from 1–11 years in shallow lakes. A study conducted in Twin Lake (8 hectares in size) determined that longevity could range from 10–20 years and the treatment reduced internal P loading by 110 kg per year (Barr, 2013).

12.1.1 Alum Case Studies and Lake Response

Research over the past decades has shown that Al permanently binds phosphorus in sediment (Rydin et al., 2000; Lewandowski et al., 2003; Huser et al., 2011). In time, however, binding sites become saturated and added Al is buried, requiring continual additions in the absence of external load reduction. To achieve desired internal P loading reductions, Al is typically added in excess of surficial mobile sediment P content (Rydin & Welch, 1999) and excess binding capacity is then available to capture mobile P migration to the treated layer or P released during organic matter mineralization (Lewandowski et al., 2003).

Lake characteristics and alum doses for multiple project lakes were reviewed by Welch & Cooke (Table 12.1; 1999), relating dosage and application depth to lake area, maximum and mean depth, and lake alkalinity.

Table 12.1: Characteristics and alum doses of selected project lakes

Lake	Chemical	Dose (gm Al/m³)	Application Depth	Lake area (km²)	Max. depth (m)	Mean depth (m)	Alkalinity (mg/L $CaCO_3$)	Mixis
Annabessacook, ME [1]	AS:SA 1:1.6	25	Hypolimnion	5.75	12.0	5.4	20	Dimictic
Kezar, NH [2]	AS:SA 2:1	30	Hypolimnion	0.74	8.2	2.7	3–10	Dimictic
Morey, VT [3]	AS:SA 1.4:1	11.7	Hypolimnion	2.20	13.0	8.4	35–54	Dimictic
Irondoquoit Bay, NY [4]	AS	28.7	Hypolimnion	6.79	23.7	6.9	170	Dimictic
Dollar, OH [5]	As	20.9	90% hypo. 10% surface	0.02	7.5	3.9	101–127	Dimictic
West Twin, OH [6]	AS	26	Hypolimnion	0.34	11.5	4.4	102–149	Dimictic
Mirror, WI [7]	AS	6.6	Hypolimnion	0.05	13.1	7.8	222	Dimictic
Shadow, WI [8]	AS	5.7	Hypolimnion	0.17	12.4	5.3	188	Dimictic
Eau Galle, WI [9]	AS	4.5	Hypolimnion	0.60	9.0	3.2	144	Dimictic
Long, Port Orchard, WA [10]	AS	5.5	Surface	1.40	3.7	2.0	10–40	Polymictic
Long, Tumwater, WA [11]	AS	7.7	Surface	1.30	6.4	3.6	45	Polymictic
Erie, WA [12]	AS	10.9	Surface	0.45	3.7	1.8	80–90	Polymictic
Campbell, WA [13]	AS	10.9	Surface	1.50	6.0	2.4	80–90	Polymictic
Pattison, WA [14]	AS	7.7	Surface	1.10	6.7	4.0	45	Polymictic
Wapato, WA [15]	AS	7.8	Surface	0.12	3.5	2.5	–	Polymictic

[1] Treatment date 8/1978 (Dominiw, 1980); [2] Treatment date 6/1984 (Connor & Martin, 1989); [3] Treatment date 5-6/1986 (Smeltzer, 1990); [4] Treatment date 7-9/1986 (Spittal & Burton, 1991); [5] Treatment date 7/1974 (Cooke et al., 1978); [6] Treatment date 7/1975 (Cooke et al., 1978); [7] Treatment date 5/1975 (Cooke et al., 1978); [8] Treatment date 5/1978 (Garrison & Ihm, 1991); [9] Treatment date 5/1986 (Barko et al., 1990); [10] Treatment date 9/1980 (Welch et al., 1982); [11] Treatment date 9/1983 (Entranco, 1987b); [12] Treatment date 9/1985 (Entranco, 1983); [13] Treatment date 10/1985 (Entranco, 1987a); [14] Treatment date 9/1983 (Entranco, 1987b); [15] Treatment date 7/1984 (Entranco, 1986).

Source: Welch & Cooke (1999).

Furthermore, sediment was collected from 20 lakes in the upper Midwest United States and Al was applied to hundreds of individual samples to elucidate the binding relationship between Al and sediment P across a broad range of mobile sediment P compositions (Table 12.2; Huser & Pilgrim, 2014). Consistent relationships between Al, initial mobile sediment P content, and sediment Al-P formed were observed.

Table 12.2: Lake characteristics, sediment properties, depth of Al additions (depth) and sediment binding coefficients

Lake	Area (ha)	Z_{max}	Z_{mean}	Depth (cm)	OM (%)	H_2O (%)	Mobile P	Sm	k	Lake type	Trophic status
		(m)					g m^{-2}cm-1				
Mitchel (SD)	271	8.8	3.7	0–8	1.2–8.4	47–89	0.84–1.4	0.87–1.36	0.15–0.19	Polymictic	Hypereutrophic
Minnetonka	119	7.9	3.4	0–10	16–18	81–86	0.34–2.2	0.35–2.2	0.05–0.10	Polymictic	Hypereutrophic
Nokomis	83	10.1	4.3	0–1	27	93	0.13	0.13	0.08	Polymictic	Eutrophic
McCarons	33	17.4	6.6	0–4	23	90	0.11	0.11	0.08	Dimictic	Eutrophic
Calhoun	170	27.4	10.6	0–4	26–28	92–93	0.13–0.41	0.13–0.41	0.05–0.06	Dimictic	Eutrophic
Harriet	143	25.0	8.7	0–4	27–29	92–94	0.22–0.24	0.22–0.24	0.03–0.06	Dimictic	Mesotrophic
Gervais	111	14.6	1.9	0–4	212–31	89–94	0.27–0.34	0.27–0.32	0.03–0.04	Dimictic	Mesotrophic
Powderhorn	4	6.1	1.2	0–8	9.1–17	76–86	0.46–0.96	0.46–0.91	0.08–0.15	Polymictic	Eutrophic
Como	29	4.9	7.5	0–1	31	94	1.8	1.6	0.13	Dimictic	Hypereutrophic
Beaver	29	3.4	7.4	0–6	32	95	0.24	0.23	0.11	Polymictic	Eutrophic
Kohlman	30	2.7	7.2	0–10	6.7	72	1.2	1.2	0.05	Polymictic	Hypereutrophic
Riley	116	12.8	1.9	0–8	21–25	89–91	0.19–0.44	0.19–0.44	0.05–0.07	Dimictic	Eutrophic
Lotus	97	8.8	2.2	0–6	15–22	84–88	0.14–0.71	0.14–0.71	0.04–0.07	Dimictic	Eutrophic
Red Rock	29	4.9	7.5	0–8	29	92	0.70	0.70	0.11	Polymictic	Eutrophic
Staring	61	4.9	3.5	0–2	19	87	0.31	0.31	0.04	Polymictic	Hypereutrophic
Silver	34	4.0	6.3	0–4	39	95	0.10	0.10	0.02	Polymictic	Hypereutrophic
Rice Marsh	33	3.0	6.6	0–6	48	95	0.04	0.04	0.01	Polymictic	Eutrophic
Duck	15	3.0	14.0	0–6	32	96	0.14	0.20	0.01	Polymictic	Hypereutrophic
Mitchell (MN)	50	5.5	4.3	0–8	30	94	0.41	0.41	0.08	Polymictic	Hypereutrophic
Round	13	11.0	16.6	0–6	25–29	90–91	0.95–2.8	0.91–2.9	0.11–0.24	Dimictic	Eutrophic

Notes: Z_{max} = lake max. depth; Z_{mean} = lake mean depth; depth (cm)= depth of Al additions; OM = organic matter; Sm = maximum sorption; k= adsorption constant (unitless).

Source: Huser & Pilgrim (2014).

Welch & Cooke (1999) examined the reduction in mean summer epilimnetic TP and Chl-*a* in seven treated and one untreated lakes (Table 12.3). P inactivation treatment resulted in a reduction of TP and Chl-*a* in many of those examined lakes.

Table 12.3: Reduction in mean summer epilimnetic TP and chl-*a* in seven treated and one untreated stratified lakes

Lake	Pre-treatment (µg/L)		Initial (yr)	% Reduced	Latest (yr)	% Reduced
	TP	Chl-*a*	TP	Chl-*a*	TP	Chl-*a*
E. Twin (untreated) -	48 (4)	57 (4)	51 (1–5)	75 (1–5)	59 (15–18)	81 (17–18)
W. Twin (treated) -	45 (4)	42 (4)	52 (1–5)	66 (1–5)	66 (15–18)	49 (17–18)
Dollar +	82 (1)	41 (1)	65 (1–7)	61 (1–7)	68 (16–18)	29 (17–18)
Annabessacook +	32 (2)	13 (3)	34 (1)	39 (1–2)	41 (9–13)	0 (8–13)
Morey +	13 (1)	13 (6)	30 (1–4)	72 (1–3)	60 (5–8)	93 (5–8)
Kezar +	24 (4)	17 (4)	34 (1–3)	65 (1–3)	37 (4–9)	45 (4–8)
Cochewagon -	15 (5)	5 (5)	28 (1–3)	67 (1–3)	0 (5–6)	47 (5–6)
Irondoquoit Bay +	47 (4)	23 (4)	13 (1–3)	28 (1–3)	24 (4–5)	47 (4–5)
Mean	-	-	37	57	42	42
Note: Years of observation in parentheses and lakes showing availability of hypolimnetic TP to the epilimnion are indicated with + and those that did not with -.						

Source: Welch & Cooke (1999).

Effectiveness (% reduction) and treatment longevity was examined by Welch & Cooke (1999), including reduction in whole-lake TP, release rate and Chl-*a* for multiple lakes (Tables 12.4; 12.5). All lakes, except Pattison South, experienced improved lake water quality.

Table 12.4: Effectiveness (% reduction) and longevity of P inactivation based on summer, whole-lake TP conc. and observed P release rate

Lake	TP (µg/L)	Reduction Whole-lake TP		Reduction Release Rate		Longevity (yr)
		Initial (%)	Latest (%)	Initial (%)	Latest (%)	
Erie [a]	115 (2)	77 (1)	75 (5–8)	79 (1)	82 (5–6)	>8
Campbell [a]	49 (2)	43 (1)	46 (5–8)	57 (1)	64 (5–6)	>8
Long (T)						
North [a]	42 (3)	60 (1–2)	56 (7–8)	84 (1–2)	79 (7–8)	8
South [a]	31 (3)	32 (1–2)	50 (4–5)	Macrophytes		5
Pattison (T)						
North [a]	28 (3)	43 (1–2)	29 (5–7)	81 (1–2)	73 (5–7)	7
South	30 (3)	-7 (1–2)	-	Macrophytes		<1
Long (K) [a]	63 (3)	48 (1–4) 68 (1)b	30 (7–11)	62 (1–4)	40 (7–10)	11
Wapato	46 (2)	-24 (1–2)	-	Macrophytes		<1
Pickerel	35 (1)	-26 (1)	-	-	-	<1
Mean of 6 [a]	55	51	48	73	68	5–11
Note: Years of observation in parentheses. T = Thurston County; K = Kitsap County. [a] six successful treatments; [b] second treatment in 1991 at same does as 1980.						

Source: Welch & Cooke (1999).

Table 12.5: Effectiveness of P inactivation in reducing TP and chl-*a* in unstratified lakes

Lake	Longevity (yr)	% Reduction Initial		% Reduction Latest Year	
		TP	Chl-*a*	TP	Chl-*a*
Erie	>8	77	91	75	83
Campbell	>8	43	44	46	28
Long (T)					
North	>8	60	89	56	39
South	5	32	68	50	0
Pattison					
North	7	43	40	29	-
South	<1	-7	6	-	-
Long (K)	>11	48	65	30	49
Mean of 6	-	51	66	48	40
Note: Unsuccessful treatment of Pattison South excluded from mean calculation.					

Source: Welch & Cooke (1999).

12.1.2 Effects and Precautions of Alum on Lake Quality

Alum forms a floc and will not be effective in shallow lakes with high wave energy (Table 12.6; Egemose et al., 2010). The P-removal effectiveness of alum is restricted to a narrow pH range and a $CaCO_3$ buffer may be required to keep the lake in the appropriate range. Additions of unbuffered alum to low-alkalinity lakes can result in acidification and the release of toxic $Al^{3}+$. There are many concerns about the toxicity of aluminum added as alum to aquatic biota, and restrictions may apply if the lake is used for drinking water (Hickey & Gibbs, 2009). Risks to biota can be minimized by separating treatments over subsequent years, reducing the potential effects on pH. Despite Al naturally occurring in sediment, the element can be toxic, particularly under acidified conditions, and is toxic to fish and macroinvertebrates at concentrations as low as 0.1–0.2 mg Al/L at a pH of 4.5–5.5 (Baker, 1982). Al can bioaccumulate in fish tissue, magnifying through a food web. Careful examination of biotic communities in lakes before and after treatment, with continued long-term monitoring, is needed.

Table 12.6: Recommended treatment based on lake characteristics

Lake Characteristics	Alum	Phoslock™	Modified Zeolite
Wind exposed lakes	NR	R	R
Deep lakes	R	R	R
Highly turbid lakes	NR	R	R
High sedimentation	R	NR	NR
Low alkalinity and poor buffering capacity	NR	R	R
Long period of stratification and anoxia	NR	R	R
High ammonium concentration	NR	NR	R
Note: NR=Not recommended; R=recommended.			

Source: Zamparas & Zacharias (2014).

12.1.3 Alum Cost

The cost of treatment varied with application technique. An order of magnitude decrease in labour per hectare occurred with the introduction of the modified harvester design (Connor & Smith, 1986). The cost to treat Three Mile Pond (266 hectares) with the high-speed system was $1,000/ha (2015 USD) (Connor & Martin, 1989), whereas the cost to treat Cochnewagon Lake (Maine) with the modified harvester was $1,500/ha and Medical Lake (Washington) with the old barge system was $1,975/ha (2015 USD). Most treatments in the United States during the 1990s have employed a large, high-speed barge that is capable of holding over 11,250 kg of liquid alum, where one person can apply approx. 115 m^3 of liquid alum per day along 15-metre-wide paths (Table 12.7; Cooke et al., 2005).

Table 12.7: Dose, treated area and worker-days per hectare for P inactivation treatments

Lake	Date	Application	Area (ha)	Dose (g Al/m^3)	Worker days/ ha
Horseshoe, WI [a]	1970	Barge system	9	2.1 [d]	1.33
Welland Canal, NY [a]	1973	Barge system	74	2.5 [d]	1.35
Dollar, OH [a]	1974	Barge system	1.4	20.9 [d]	4.30
West Twin, OH [a]	1975	Barge system	16	26.1 [d]	4.61
Medical, WA [a]	1977	Barge system	227	12.2 [d]	2.03
Annabessacook, ME [a]	1978	Barge system	121	25.0 [d]	1.12
Kezar, NH [b]	1984	Modified harvester	48	40.0 [e]	0.50
Morey, VT [b]	1986	Modified harvester	133	45.0 [e]	0.57
Cochnewagon, ME [b]	1986	Modified harvester	97	18.0 [e]	0.41
Sluice Pond, MA [b]	1987	Modified harvester	6	20.0 [e]	0.67
Three Mile Pond, ME [c]	1988	Computerized does and navigation system	266	20.0 [e]	0.06

[a] Barge system (Kennedy & Cooke, 1982); [b] Modified harvester (Connor & Smith, 1986); [c] Computerized dose and navigation system (Eberhardt, 1990); [d] Aluminum sulfate; [e] Aluminum sulfate and sodium aluminate.

Note: One person working 8 hour = 1 worker-day. Most treatments involved 12 to 14 days.

Source: Cooke & Kennedy, 1981; Connor & Martin, 1989; Cooke et al., 1993a; Cooke et al., 2005.

A more recent study was on the feasibility of alum application for Twin Lake. The treatment was estimated to cost $148,000, including engineering, design (20 per cent), a contingency (25 per cent) and permitting (Barr, 2013). However, the cost does not include monitoring.

Alum treatment, hypolimnetic aeration and hypolimnetic withdrawal were reviewed to remediation eutrophication issues for Jessie Lake, Minnesota (Table 12.8). In comparison to other remedial treatments, sediment removal is the only other operation technique that could be used to eliminate internal P loading. Phosphorus inactivation is more economical and effective, however, sediment removal does have the long-term benefit of removing the nutrient source.

Table 12.8: Estimated costs associated with internal load reduction for Jessie Lake, MN

Treatment	Initial Capital Cost	Annual O & M	Reapplication Costs	Annualized Cost
Alum treatment (40%)	$508,000	$0	$508,000/15 years	$44,100
Alum treatment (60%)	$754,000	$0	$754,000/15 years	$65,500
Alum treatment (100%)	$1,250,000	$0	$1,250,000/15 years	$109,000
Hypolimnetic aeration	$2,290,000	$174,000	$897,000/10 years	$357,000
Hypolimnetic withdrawal	$1,580,000	$79,300	$16,000/10 years	$166,000
Hypolimnetic withdrawal with alum injections	$1,620,000	$86,800	$16,000/10 years	$176,000

Source: Itasca Soil and Water Conservation District (2011).

12.2 Calcium Carbonate ($CaCO_3$) and $Ca(OH)_2$

Calcium carbonate precipitation in hard-water lakes is an effective treatment for eutrophication (Dittrich & Koschle, 2002). In the epilimnion of hard-water lakes, pH often increases during algal blooms, resulting in $CaCO_3$ precipitation. Additions of lime ($CaCO_3$) or calcium hydroxide ($Ca(OH)_2$) have been made to lakes to induce the precipitation of calcium carbonate, flocculation of algae, and to reduce P cycling from bottom sediments (Prepas et al., 1990; Prepas et al., 2000; Hart et al., 2003). Phosphorus is bound to the surface of the calcium and over time becomes incorporated into apatite, where it is permanently removed from the system.

Artificial hypolimnetic calcite precipitation may be induced by $Ca(OH)_2$ injection combined with hypolimnetic aeration (Table 12.9). The aeration system is used to mix the added $Ca(OH)_2$ into the hypolimnion, rather than as a measure for sustainable restoration itself.

Table 12.9: **Morphometric and limnological data of the Basin Carwitz, Lake Schaler Luzin; additions of $Ca(OH)_2$ in Basin Carwitz**

	Lake Schmaler	Luzin Basin Carwitz
	Morphometric parameters	
Surface area (km^2)	1.34	0.58
Volume (106 km^3)	20.6	0.5
Mean depth (m)	14.7	18.1
Max. depth (m)	34	33
Mean hydraulic residence time	4.4	-
	Limnological parameters	
TP (µg/L)	37	38
Chl-*a* (µg/L)	4.0	4.0
Secchi depth (m)	5.2	5.2
$CaCO_3$ (mg/L)	0.20	0.22
pH	8.5	8.5
Ca^2+ (mg/L)	41.1	41.1
O_2 hypolimnion conc. (mg/L)	0	0
	$Ca(OH)_2$ additions to Basin Carwitz	
Year	Air addition (m^3)	$Ca(OH)_2$ addition (tonne)
1996	416,000	140
1997	532,000	91
1998	510,000	72

Source: Dittrich, Gabriel, Rutzen, & Koschel (2011).

12.2.1 Calcium Carbonate ($CaCO_3$) and $Ca(OH)_2$ Case Studies and Lake Response

Calcium additions have been extensively used in hard-water lakes in western Canada and the United States as a remediation technique. Prepas et al. (1990) describe the results of $CaCO_3$ and Ca(OH)2 additions to hypereutrophic Figure Eight Lake, Alberta in 1986 and 1987. In the first year of addition, Chl-*a* decreased within two weeks due to flocculation with added $CaCO_3$, but returned to former levels 35 days later. Under-ice oxygen conditions and P-release from bottom sediments was markedly improved in subsequent years, with declines in Chl-*a* being observed in summer.

12.2.2 Effects and Precautions of Calcium Additions

Calcium additions to hard-water lakes have fewer toxic impacts than alum. The main issues of concern surround increases in turbidity that often accompany calcium additions and potential smothering of benthos by $CaCO_3$ (Hickey & Gibbs, 2009). Some calcium products have low P-binding efficiency, and this efficiency may be affected by changes in pH.

12.2.3 Calcium Cost

Two calcium carbonate active barrier materials were assessed to reduce eutrophication problems in Australian lakes: SoCal estimated application cost was approximately $3,800 per tonne, and ESCal was estimated at $2,000 per tonne (Hart et al., 2002).

12.3 Phoslock™ and P-sorbing Products (PSPs)

Of growing interest is the use of solid phase P-sorbing products (PSPs) including industrial by-products and naturally occurring or modified mineral complexes designed to remove soluble P through sorption into solid state material (Hickey & Gibbs, 2009). Generally, PSPs have high contents of Fe, Al or Ca, or a combination, and all elements form mineral complexes with a high sorption capacity under specific physiochemical conditions (Hickey & Gibbs, 2009). Effective PSPs have high P-sorption removal properties and the ability to strip soluble P to low concentrations (<3 μg/L) and produce relatively refractory product-P complexes (Reynolds, 1999; Reynolds & Davis, 2011). Phoslock™ is a lanthanum (La) modified bentonite clay that has been increasingly used as treatments to control legacy P release from lake bed sediments to overlying waters. Other PSPs include industrial waste products (red ochre, black ochre), waste products from building practices (gypsum, sander dust, mag dust and vermiculite) (Spears et al., 2013a). Modified zeolite (often marketed as Z_2G_1) is especially effective for reducing recycling of both nitrogen (N) and P (Wen et al., 2006). Although the phytoplankton communities of most lakes are P-limited (Schindler, 2012), there may be occasional need to also limit N recycling.

12.3.1 Phoslock™ Case Studies and Lake Response

The application of PSPs to lakes and reservoirs in the United Kingdom has been restricted to modified La-bentonite (Phoslock™ Meis et al., 2012). Spears et al. (2013a) compared the elemental properties and P-sorption properties of six industrial waste products (red ochre, black ochre), waste products from building practices (gypsum, sander dust, mag dust and vermiculite) and the modified La-bentonite product, Phoslock™. The comparison was made between treatment technologies because all have been proposed or used in U.K. lakes. It was concluded that the P-sorption capacity of each PSP is dependent on the specific physiochemical conditions in the receiving lake (pH, redox conditions and alkalinity) as well as lake morphology (Huser, 2012). All products, except gypsum, were capable of reducing P concentrations.

Spears et al. (2013b) examined 16 lakes following the application of Phoslock™. Table 12.10 reviews location of the lake, maximum fetch, mean depth, maximum depth, surface area, annual mean alkalinity, conductivity in the year following treatment application, as well as Phoslock® dose quantities.

Table 12.10: Application of Phoslock™ to 16 case study lakes

Lake	Surface Area (ha)	Mean Depth (m)	Max. Depth (m)	Fetch (km)	Date, Mass Applied (tonnes)	Phoslock™ Load (tons/ha/mg/L)
Clatto Reservoir, U.K.	9.0	2.8	7.0	0.4	04/03/2009 (24.0)	2.67/96.97
Loch Flemington, U.K.	15.7	1.0	2.5	0.7	15/03/2010 (25.0)	1.59/159.24
Somerset Reservoir, U.K.	2.2	4.5	9.0	-	27/03/2007 (6.6)	3.00/66.67
Lake Rauwbraken, NL	4.0	8.8	16.0	0.2	21/04/2008 (18.0)	4.50/51.43
Lake De Kuil, NL	7.0	4.0	10.0	-	18/05/2009 (41.5)	5.93/148.21
Lake Silbersee, G	7.0	5.0	9.0	0.3	08/11/2006 (21.5)	3.07/61.43
Lake Ottersteder See, G	4.5	5.0	11.0	0.3	30/10/2006 (11.0)	2.44/48.89
Lake Behlendorfer, G	64.0	6.2	16.0	2.0	02/12/2009 (230.0)	3.59/57.96
Lake Blankensee, G	22.5	1.6	2.5	0.5	16/11/2009 (66.0)	2.93/183.33
Lake Baerensee, G	6.0	2.6	3.8	0.1	11/06/2007 (11.5)	1.92/73.72
Lake Kleiner See, G	0.9	2.0	5.0	0.2	25/06/2010 (6.0)	6.67/333.33
Lake Eichbaumsee, G	23.2	6.5	16.0	0.9	17/11/2010 (148.0)	6.38/92.14
Lake Ladillensee, G	1.0	2.1	5.0	0.1	03/03/2009 (4.7)	4.65/221.43
Lake Völlen, G	2.0	2.5	5.5	0.1	19/03/2008 (10.0)	5.00/200.00
Neidersachsen Lake	4.2	2.5	6.0	0.1	19/03/2008 (6.0)	1.43/57.14
Lake Okareka, NZ	340.0	2.0	34.0	2.8	16/08/2005 (20.0)	0.06/0.29
Note: U.K. = United Kingdom; G = Germany; NZ = New Zealand.						

Source: Spears et al. (2013b).

12.3.2 Effects and Precaution of Phoslock™ on Lake Quality

The advantages of Phoslock™ are that it is effective over a wider range of pH than other P-inactive methods, it precipitates polyphosphates as well as orthophosphates, and the solubility product of La-phosphate is very low (Hickey & Gibbs, 2009). Phoslock™ is also less flocculent than alum, allowing for more effective additions in shallow lakes (Gibbs et al., 2011).

The greatest concern with Phoslock™ is the potential toxicity of lanthanum (La). La concentrations may remain elevated in surface and bottom water for several months following Phoslock™ applications, with decrease in total La concentrations varying with site-specific physiochemical factors (Roskocsh et al., 2011; Spears et al., 2013b). Following treatment, Spears (2013b) found that total La and filterable La concentrations in surface waters were between 0.026–2.30 mg/L and 0.002–0.14 mg/L, respectively. In comparison, the Netherlands has a legal maximum permissible concentration of filterable La (0.01 mg/L) and total La in surface water (0.15 mg/L)[6]. Drinking water restrictions will typically apply for periods following Phoslock™ applications.

Spears et al. (2013b) examined the potential for negative ecological impacts from elevated La concentrations associated with the use of Phoslock™. The potential for negative ecological impacts indicated that the concentrations of La^{3+} ions were very low (<0.0004 mg/L) in lakes of moderately low to high alkalinity (>0.8 mEq/L), but high (up to 0.12 mg/L) in lakes characterized by very low alkalinity. Phoslock™ is a relatively new treatment method and its negative effects on the environment over the long-term are not fully understood.

[6] Standard is based on studies specific to the assessment of reproduction rates in *Daphnia magna*.

Landman et al. (2007) report bioaccumulation of La in trout. Following treatment of Phoslock™ to Lake Okareka, New Zealand, they found accumulation of La in the liver and hepatopancreas of fish and crayfish. Phoslock™ has the potential to disrupt feeding mechanisms of various taxa due to Phoslock™ particles in the water column and to smother benthos.

12.3.3 Phoslock™ Cost

The cost of Phoslock™ treatment depends on application dosage, the amount of monitoring, data analysis, permitting, and labour and equipment costs associated with lake size. Project costs typical for Florida lakes were between \$440–\$880 per kg of biologically available P immobilized (Morrison, 2012). Furthermore, Ackerman & Cicek examined algal removal followed by sediment capping and the associated cost for Killarney Lake, Manitoba (2014) (Table 12.11).

Table 12.11: Cost comparison of clay-modified chitosan, chitosan + clay cap, and Phoslock™

Cost of algal bloom flocculation with clay-modified chitosan (dosage 25 mg/L)	
Materials cost per m^3	\$0.04
Mean lake depth 3 m	
Cost per m^2	\$0.13
Chitosan + clay cap (1 cm depth)	
Clay for 1 cm cover	\$0.15
Chitosan dosed to 2 mg/75 mg clay	\$4.00
Material cost per m^2	\$4.15
Phoslock™ cap material cost per m^2	
At dosage of 100 g/g mobile sediment P	
If 50 mg mobile P/kg in top 10 cm	\$0.75
If 400 mg mobile P/kg in top 10 cm	\$6.02

Source: Ackerman & Cicek (2014).

12.4 Modified Clays

Chitosan (deacerylated chitin), is a biopolymer derived from chitin and is one of the most abundant organic materials produced annually by biosynthesis. Due to its unique polycationic nature, chitosan has been used an active material that is antifungal, antibacterial and antitumor in nature. In regards to eutrophication and remediation, chitosan is primarily used as a flocculent, as opposed to P inactivation. Recent research has shown clays can remove algae from the water column—a study with 26 clays, ranking their ability to remove *Microcystis aeruginosa* cells, was performed with 700 mg/L of talc, kaolinite and sepiolite (Pan et al., 2006). Clays were able to remove 94 to 98 per cent of cells in eight hours. Over 90 per cent algal cell removal was achieved with chitosan-modified clay doses of 11 mg/L, where increased dosage removed cells faster. With doses of 51 mg/L, 80 per cent of removal was achieved in 10 minutes, whereas 6 mg/L moved 80 per cent in 200 minutes (Zou et al., 2006).

A pilot-scale experiment was conducted using chitosan-modified sepiolite. Field experiments were conducted in two non-permeable enclosures in Meiliang Bay of Taihu Lake. Each enclosure was 32 m^2 in area and approximately 2 m deep (Pan et al., 2006). An average of 1.5 kg of chitosan-modified local soils were added to the treatment enclosure (contained 10 per cent chitosan) and the other enclosure was used as a control. Both chl-*a* and P concentrations in the enclosures were monitored throughout the experiment (Table 12.12).

Table 12.12: **Application of chitosan-modified clay in a eutrophic lake**

	Clay Treatment Enclosure		Control Enclosure	
	Pre-Treatment	Post-treatment	Pre-Treatment	Post-Treatment
Depth (m)	1.9	1.9	1.9	1.9
pH	8.22	7.85	8.22	8.22
Secchi depth (m)	15	86	15	13
Dissolved P (mg/L)	0.020	0.014	0.016	0.018
Dissolved TP (mg/L)	0.052	0.034	0.046	0.040
Chl-*a* (mg/L)	2.05	0.016	1.96	1.95
Chl-*a* removed (%)	99	-	-	-

Source: Pan et al. (2006).

The cost of algal bloom flocculation with modified clay (chitosan, dosage 25 mg/L) was evaluated by Ackerman & Cicek (2014) for Killarney Lake (Manitoba); \$0.04/m^3 (mean lake depth 3 m) or \$0.13/m^2. Chitosan and a clay cap (1 cm depth) were estimated at \$4.15/m^2 (Table 12.11).

Phosphorus Inactivation Summary

Many different methods of sediment capping and P inactivation have been used in lake restoration projects, including physical (mechanical or passive) capping and active capping using alum, calcium, zeolite, Phoslock™, iron and modified clays. Passive capping with sand, gravel, or clay is used to decrease diffusion of nutrients and contaminants to the overlying water column and bury them deeper in the sediments.

Beneficial Effects

Alum

- Proven effective control.

Calcium

- Extensively used in hard-water lakes.
- Calcium additions to hard-water lakes have fewer toxic impacts than alum.

Phoslock™

- Proven effective control, non-toxic under a wide range of environmental conditions; effective under a wide range of pH values and alkalinities; does not affect pH levels following treatment (advantages over alum).

Undesirable Effects

Alum

- Restricted to a narrow pH range; additions to low-alkalinity lakes can result in acidification.
- Toxicity bioaccumulates in fish tissue.

Calcium

- With increased turbidity, potential smothering of benthos by $CaCO_3$.

Phoslock™

- Potential toxicity of La.
- Long-term negative ecological impacts not well understood.

Suitable Lake Conditions

Capping thickness usually exceeds 5 cm, which limits the approach to small lakes or reservoirs due to the large volume of material required and difficulties depositing a uniform layer.

Successful implementation of P inactivation as reviewed in the literature:

Alum

Lake size: 9–600 hectares.

Mean depth: 1.8–8.4 m

Max depth: 3.5–23.7 m

pH: 6–8 throughout treatment

Alkalinity: <50 mg $CaCO_3$/L; will lower pH if lake has low alkalinity

Calcium

Lake size: 58–240 hectares.

Mean depth: 10–18 m

Max. depth: 30–42 m.

Hydraulic res. time: 4.4 yrs. (Lake Schmaler).

Chl-*a*: 4.0

Secchi depth: 5 m

pH: 8.5

Phoslock™

Lake size: 0.9–64 hectares.

Mean depth: 1.6–8.8 m.

Max. depth: 2.5–34 m.

Longevity: P inactivation longevity does not typically exceed 15 years and will depend on PO_4 release rates and application dose. Alum longevity typically 4–21 years (stratified); 1–11 years (shallow lakes).

Lake characteristic	Alum	Phoslock™
Wind-exposed lakes	N	R
Deep lakes	R	R
Highly turbid lakes	NR	R
High sedimentation	R	NR
Low alkalinity and poor buffering capacity	NR	R
Long period of stratification and anoxia	NR	R
High ammonium concentration	NR	NR

Cost

Alum: Twin Lake, MN: $148,000 (USD, 2013).

Jessie Lake, MN: alum (40%) $508,000; alum (60%) $754, 000.

Phoslock™: $440–$880 per k for biologically available P immobilized (Morrison, 2012).

If 50 mg mobile P/kg in top 10 cm: $0.75 (Ackerman & Cicek, 2014).

If 400 mg mobile P/kg in top 10 cm: $6.02 (Ackerman & Cicek, 2014).

13. SEDIMENT OXIDATION

A restoration technique to oxidize the top 15 to 20 cm of anaerobic lake sediment was developed by Ripl (1976) and has been promoted by Atlas Copco Co. and then Aquatec Inc., under the name Riplox. The Riplox method reduces internal loading in lakes where iron redox reactions control P fluxes between sediment and overlying water (Cooke et al., 2005). Nitrate, as $Ca(NO_3)_2$, iron (as $FeCl_3$), and lime ($CaCO_3$) are injected into lake sediments using a discing harrow. Nitrate acts as an alternate electron acceptor to oxygen, preventing the development of ferrous iron and subsequent P release. Nitrate is preferred as the electron acceptor because of its liquid state: the solution penetrates into the sediment more readily and is therefore more efficient than adding oxygen to the hypolimnion. Sulfate reduction is also prevented, therefore decreasing the formation of iron sulfide and leaving iron available to complex P (Cooke et al., 2005). To deplete organic matter in the sediments and restore an oxidized state, a solution of $Ca(NO_3)_2$ is injected into the sediment to stimulate denitrification. Ferric chloride ($FeCl_3$) is added to increase the binding capacity of sediments and to remove hydrogen sulfide (H_2S) (Cooke et al., 2005). Lime ($CaOH_2$) is then added to raise the pH and encourage microbial denitrification. The redox potential for nitrate reduction is higher than for iron reduction, therefore the latter is inhibited and P remains complexed with ferric iron compounds (Foy, 1986). Ferric chloride and lime additions have been determined as unnecessary in some cases, where pH may be sufficiently high to promote denitrification and sediment iron content adequate (30–50 mg/g) for P binding (Cooke et al., 2005).

The chemical solutions are applied by direct injection into the sediment with a "harrow" device, which is approx. 6 to 10 m wide and equipped with flexible tubes that penetrate the sediment. Sediment is disrupted to a depth of approx. 20 cm as the harrow is dragged along the lake bottom at a rate of 4 to 5 m/min and the chemical solutions are injected through tubes at the end of the device.

13.1 Case Studies and Lake Response

13.1.1 Lake Lillesjön, Sweden

An example of dosages used is from the treatment of Lake Lillesjön, a small (4.2 ha), shallow (2 m) lake in Sweden (Ripl, 1976). To treat the lake, 13 tonnes of ferric chloride (146 g Fe/m^2), 5 tonnes of lime (180 g Ca/m^3) and 12 tonnes of calcium nitrate (141 g N/m^2). The dose of calcium nitrate for Long Lake, MN sediments was found to be the same as for Lake Lillesjön.

Following the treatment of Lake Lillesjön sediment in 1975, interstitial P content in the top 20 cm decreased by 70 to 85 per cent in comparison to the 1974 levels and levels continued to 1977 (Ripl, 1981). Despite high loading of NO_3 to the sediments, nitrogen was lost through evolution of N_2 gas, and the concentration of ammonia decreased (Ripl, 1981). Denitrification of added NO_3 was completed after 1.5 months, and the oxygen demand of the sediment also decreased by approximately 30 per cent (Ripl, 1981). Recycling of P and N to the water overlying treated sediment was reduced between 10 per cent and 20 per cent of the rate observed prior to restoration (Ripl & Lindmark, 1978; Ripl, 1981). Oxygen demand of sediment and sediment P release continued to remain low 10 years following treatment application.

13.1.2 Lake Trekanten, Sweden

To treat Lake Trekanten, Sweden, below the 3 m contour (49 ha) required 160 tonnes of $Ca(NO_3)_2$ (56 g N/m^2) in a 50 per cent solution (Cooke et al., 2005). Following treatment, the iron naturally present in this lake's sediments was considered adequate. The interstitial P content of sediments in Lake Trekanten decreased from 2 to 4 mg/L (prior to treatment, May, 1980) to 0.01 to 0.3 mg/L (July, 1980) (Cooke et al., 2005). Although P release from the sediment decreased, lake P content remained unchanged because of continued P inputs from external sources and possibly by other mechanisms. Predicted external loading was found to be twice the calculated rate, and the continued algal production supplied an energy source for sulfate reduction, leading to iron complexation with sulfide and a recurrence of sediment P release (Cooke et al., 2005).

13.1.3 Long Lake, Minnesota, United States

Sediment P release decreased by 50 to 80 per cent following calcium nitrate treatment to Long Lake, MN; however, less than predicted from laboratory experiments (Noonan, 1986). There was no change in lake P content and the failure of Riplox to reduce P was attributed to continued high external loading (Ripl, 1986).

13.1.4 White Lough, Ireland

Following calcium nitrate treatment to the sediments of White Lough, Ireland, P concentrations decreased and P release from sediment was delayed by one month. Maximum hypolimnetic P concentration decreased 30 per cent following the treatment (Foy, 1986). Although expected results were observed in White Lough, the lake was under-dosed: an optimum dose of 30 to 60 g N/m^2 instead of 24 g N/m^2 would have had greater reductions (Foy, 1986).

13.2 Effects and Precautions

Although higher in cost, this treatment is an effective alternative to alum treatment to inactivate sediment P. Moreover, the chemicals added are found naturally in high concentrations in unpolluted sediments, and toxicity to animals is perceived as a lesser issue than for other P inactivation methods. It is possible that potential effects may be more permanent than alum, which initially covers the sediment and then settles and distributes through the sediment column. Because the method requires direct injection of chemicals into sediments, it can typically be used only in shallow lakes with relatively flat bottoms (Hickey & Gibbs, 2009).

An important precaution in using sediment oxidation is that it can be expected to succeed only if the internal loading of P is controlled by iron redox reactions. However, if the lake is shallow and internal P loading is controlled to a large extent by high pH and temperature in the sediment–water interface during summer, then sediment oxidation may not significantly decrease P loading (Pettersson & Böstrom, 1981). Another concern with the selection of Riplox over other inactivation treatment (e.g., alum) is the lack of documented successes. Lake Lillesjön appears to be the only documented case where lake P significantly declined and the oxidized state of the sediment persisted. External P loading was the cause of lake quality recovery failure in Trekanten and Long Lakes. As with other sediment remediation methods, the benefits of Riplox will be short-term and permanent reductions in nutrient concentrations can only be obtained with reductions of external loadings.

13.3 Cost

Riplox is a comparatively expensive remediation method. The total cost for the Lake Lillesjön treatment was approx. $232,500 (2015 USD), 40 per cent of which was for development of the application device and lake investigations (Foy, 1986). The chemicals applied to the 1.2 hectare lake area represented only 6 per cent of the total cost and 28 per cent of the total cost went to equipment installation.

The total cost for Lake Trekanten treatments, excluding preliminary investigations, was approximately $609,000 (2015 USD). The total area of the lake, 87 hectares, was treated once (1980), and then an area of 49 hectares and greater than 3 m in depth was treated a second time.

The total costs for calcium nitrate and iron/alum were $43,000 and $11,500, respectively (2015 USD) for the treatment of White Lough, Ireland (Foy, 1986). The costs per treated hectare were $9,350 and $2,500, respectively (2015 USD) (Foy, 1986). The greater cost for Riplox in Lake Lillesjön in comparison to White Lough (approximately 5.4 times) was attributed to the use of compressed air in Lake Lillesjön.

Sediment Oxidation Summary

Sediment oxidation by enhanced denitrification results in improved complexation with iron (Riplox). The method reduces internal loading in lakes where iron redox reactions control P fluxes between sediment and overlying water. Nitrate, as $Ca(NO_3)_2$, iron (as $FeCl_3$), and lime ($CaCO_3$) are injected into lake sediments using a discing harrow. Nitrate acts as an alternate electron acceptor to oxygen, preventing the development of ferrous iron and subsequent P release.

Beneficial Effects

- Although greater in cost, treatment is an effective alternative to alum to inactivate sediment P.
- Chemicals added are found in high conc. naturally in unpolluted sediments.
- Toxicity to animals is perceived as a lesser issue than for other P inactivation methods.
- Potentially more permanent than alum due to direct injection into the sediment column.

Undesirable Effects

- Expected to succeed only if internal loading of P is controlled by iron redox reaction.
- Concerns over the lack of documented successful applications.

Suitable Lake Conditions

Because the method requires direct injection of chemicals into sediments, it can typically be used in shallow lakes with relatively flat bottoms.

Successful implementation of hypolimnetic withdrawal as reviewed in the literature.

Lake size: 4.2–49 hectares.

Depth: Suitable for shallow lakes; 0.7–2.3 m.

Injection depth: 0.2–3 m.

pH: Ferric chloride and lime additions have been determined as unnecessary in some cases, where pH may be sufficiently high to promote denitrification and sediment iron content adequate (30–50 mg/g) for P binding.

Longevity: Continued low sediment P release 10 years following treatment was observed for Lake Lillesjön, Sweden.

Cost

Comparatively expensive remediation method.

Lake Lillesjön, Sweden (applied to 1.2 hectares lake area):

$232,500 (USD, 2015).

Lake Trekanten, Sweden (87 hectares):

$609,000 (USD, 2015).

White Lough, Ireland:

Nitrate: $43,000; $9,350 per hectare (USD, 2015).

Iron/alum: $11,500; $2,500 per hectare (USD, 2015).

14. ALGICIDE

Algicides alleviate issues associated with algae by directly killing standing biomass. A wide range of chemicals have been used, including copper sulfate ($CuSO_4$), silver nitrate ($AgNO_3$), potassium permanganate ($KMnO_4$), sodium hypochlorite (NaOCl) and several different organic herbicides (Lam et al. 1995). In particular, various copper salts (e.g., copper sulfate) have been used widely in relatively small lakes in the past. Today, algicides are rarely used due to concerns about toxicity to non-target aquatic biota and, in some cases, human health. In particular, copper accumulates in sediments and can cause long-term contamination. Copper effects on eutrophication are temporary and annual treatment costs are high, but more importantly, the treatment has major negative impacts on non-target organisms. Several U.S. states have restricted, phased out or lowered permissible doses due to the negative effects on the environment.

The primary toxic form of copper to algae is the cupric ion (Cu^{2+}), however other forms such as copper-hydroxy complexes may also be toxic (McKnight et al., 1981; Erickson et al., 1996). The effects on algae vary with species and include inhibitions of photosynthesis, phosphorus (P) uptake and nitrogen fixation (Havens, 1994). Cyanobacteria are particularly sensitive, where concentrations as low as 5–10 µg Cu/L can suppress activity (Horne & Goldman, 1974; Cooke et al., 2005).

Guidelines for $CuSO_4$ treatments for planktonic algae were developed by Mackenthun (1961). For lakes with a methyl orange alkalinity > 40 mg/L as $CaCO_3$, the dose for planktonic algae is 1.0 mg $CuSO_4 \cdot 5\ H_2O$ per litre, as copper sulfate crystals, for the upper 0.3 m depth, regardless of actual depth. If alkalinity is < 40 mg/L, the dose is 0.3 mg $CuSO_4 \cdot 5\ H_2O$ per litre. For water of this alkalinity, 0.3 m is considered the maximum effective depth range after which copper is rapidly lost to complexation. Moreover, copper sulfate is more effective at water temperatures >15 °C. Nor (1987) determined that doses at these concentrations will be toxic to many species of algae and to some non-target organisms. The control of *Chara* and *Nitella* require a dose of 1.5 mg/L or higher, and must be applied early in the season prior to algae becoming encrusted with marl (Cooke et al., 2005). It is important to note that all of these concentrations greatly exceed existing Canadian limits for the protection of aquatic life (Canadian Council of Ministers of the Environment, 2003) and, consequently, cannot be added legally to lakes in most jurisdictions.

14.1 Case Studies and Lake Response

The Fairmont Lakes in southern Minnesota were treated with copper sulfate for 58 years to reduce excessive algal growth. Copper sulfate was applied to five lakes at aggregate rates and increased upwards to 1647 kg/ha. Application over the years totaled 1.5 million kilograms in total (Hanson & Stefan, 1984). The Fairmont Lakes are located in southern Minnesota and form a string of interconnected shallow water bodies with surface areas ranging from 34 to 224 hectares (Hanson & Stefan, 1984). The lake water hardness was approximately 200 to 300 mg/L $CaCO_3$ and total alkalinity 150 to 200 mg/L $CaCO_3$. Blue-green algae were the predominant taxa.

Copper sulfate treatment began in 1921 and data collected to measure the effects of the algicide produced distressing insight into lake response to sustained application of this chemical treatment.

Short-term effects included:

1. The intended, however temporary, suppression of algae.
2. Dissolved oxygen depletion by decomposition of dead algae.

3. Accelerated phosphorus recycling from the lake bed and recovery of the algal population within 7 to 21 days.
4. Occasional fish kills due to oxygen depletion and/or copper toxicity.

Furthermore, long-term effects include:

1. Copper accumulation in the sediment.
2. Tolerance adjustments of certain species of algae to higher copper sulfate dosages.
3. Shift of species from green to blue-green algae and from game fish to rough fish.
4. Disappearance of macrophytes.
5. Reductions in benthic macroinvertebrates.

While copper sulfate treatments may remove algae almost instantaneously, other immediate and cumulative side effects have been determined harmful to many other aquatic organisms.

14.2 Effects and Precaution

14.2.1 Beneficial Effects of Copper Sulfate on Lake Quality

Copper sulfate application was the standard treatment for algal problems for many decades due to its short-term effectiveness, particularly in water supply reservoirs. As described previously, short-term, intended beneficial effects include suppression of algae. However, the negative, undesirable effects far outweigh this positive.

14.2.2 Undesirable Effects of Copper Sulfate on Lake Quality

Copper sulfate negatively impacts aquatic communities and can create human health problems (Cooke et al., 2005). Mortality of toxic algae from $CuSO_4$ and other algicides may result in the release of cellular toxins such as microcystin, with potentially serious negative health effects on wildlife and humans drinking the water (Lam et al., 1995). Resistance may also develop in target algae and algae grazing by zooplankton and benthic invertebrates may be eliminated by indirect toxic impacts. Dissolved oxygen depletion can occur when large volumes of dead algal cells decompose, creating conditions which increase ferrous iron, P, manganese, hydrogen sulfide and ammonia concentrations. Accumulation of sulfate in lake sediments following long-term $CuSO_4$ additions may bind Fe as FeS, permanently reducing the P-binding of capacity of lake sediments. Hence, algicide applications may make conditions more favourable for the development of eutrophic conditions, in exact opposition to the desired effect.

Laboratory toxicity tests have demonstrated lethal effects of copper on bluegills (*Lepomis macrochirus*), where locomotor activity was impaired at concentrations of 40 µg Cu/L (Ellgaard & Guillot, 1988). Hatchability and survival of four-day old larvae were affected by concentrations greater than 77 µg Cu/L (Benoit, 1975). The risk of direct bluegill mortality was determined as low; however, sublethal effects on behaviour and reproduction, and feeding behaviour, could lead to reduced growth and occur at concentrations more than an order of magnitude less than those recommended for algae treatment (Sandheinrich & Atchison, 1989). Other species, such as trout, may be more copper-sensitive (Cooke et al., 2005). Furthermore, Anderson et al. (2001) compared the hepatic concentrations of copper in largemouth bass (*Micropterus salmoides*) and common carp (*Cyprinus carpio*) in Lake Mathews and Copper Basin Reservoir, California. Lake Mathews, a water supply reservoir, received more than 2,000 tonnes of granular copper sulfate over a 20-year period. The lake retained 80 per cent of the applied copper, primarily associated with oxidizable and carbonate-bound phases (Haughey et al., 2000). Copper Basin Reservoir was untreated. Sediment copper in Lake Mathew averaged 290 mg Cu/kg dry weight and Copper Basin averaged 8 mg Cu/kg dry weight (Haughey et al., 2000). Hepatic accumulation of copper was found in small bass (< 41 cm length) and in all carp in the treated lake.

Copper sulfate is highly toxic to *Daphnia*, a common and effective grazer of planktonic algae. Copper concentrations 100 times less than needed for algae control inhibit reproduction are lethal to zooplankton (Blaylock et al., 1985; Winner et al., 1990). *Daphnia magna, D. pulex, D. parvula* and *D ambigua*, tested in waters with an alkalinity of 100–119 mg/L and $CaCO_3$ additions exceeding concentrations of 8 µg Cu/L exhibited reductions in survival and reproduction (Winner & Farrell, 1976).

Copper stress also impairs food web functions. When planktonic communities in *in situ* mesocosoms were exposed to 140 µg Cu/L for 14 days, not only were *Daphnia*, phytoplankton and Protozoa (ciliates, flagellates) greatly reduced in abundance, but carbon flow through the food web was also impaired (Havens, 1994). Bacteria increased significantly and there was minimal energy transfer via the microbial loop to higher trophic levels (Havens, 1994).

14.3 Cost

As with other methods of lake remediation, applications of algicides provide only temporary relief for algal problems and will require continued reapplications if external nutrient loads are not reduced. The costs for $CuSO_4$ use in algae management are dictated by dose, frequency of reapplication, area to be treated, species of algae and other lake-specific factors. The more costly chelated or complex forms may be needed in hard-water situations, but may be longer-lasting and more effective. In four Minnesota recreational lakes and a water supply lake, treatment occurred for over 58 years of $CuSO_4$: 1.5 million kg of $CuSO_4$ were applied at an estimated cost of $5.25 million (2015 USD, including labour and operating costs) (Cooke et al., 2005).[7] During the summer months, 35 per cent of the chemical costs for the treatment plant were for $CuSO_4$. The treatments were not sufficiently cost effective, given that benefits were temporary and there were long-term environmentally harmful effects. Ultimately, $CuSO_4$ applications were terminated (Hanson & Stefan, 1984).

The variation of single treatment costs with copper formulation is demonstrated by the treatments at Casitas Reservoir, California (Table 14.1). Application costs vary greatly, where granular copper sulfate costs approx. $2/kg and liquid Cutrine Plus cost approx. $10/litre (McComas, 2003).

Table 14.1: Copper sulfate treatments cost, Casitas Reservoir, California (445 hectares)

Treatment	Cost (2015, USD)
$CuSO_4$ solution	$220–$650 ($169–$499, 2002 USD)
$CuSO_4$ crystals	$197–$1,185 ($152–$913, 2002 USD)
$CuSO_4$ citric acid solution	$127–$1,500 ($98–$1106m 2002 USD)
Copper-ethanolamine granular	$710–$3,000 ($547–$2263, 2002 USD)

Source: Modified from AWWARK (1987).

[7] $4.04 million was cited by Cooke et al. 2005 (2002 USD).

As mentioned previously, copper sulfate application was the standard treatment for algal problems due to its short-term effectiveness. However, there is substantial evidence against the use of this compound, as well as the fact that the method is largely illegal. There are other longer-term and more permanent options to manage algae, where there is a higher margin of safety on non-target organisms.

Algicide Summary

Commonly used in earlier lake remediation management and water supply reservoirs suffering from algal biomass, algicide is not frequently practiced due to significant detrimental aspects associated with the technique on the lake's biological community.

It is important to note that dose application concentrations for copper sulfate (*Chara* and *Nitella* require a dose of 1.5 mg/L or higher) greatly exceed existing Canadian limits for the protection of aquatic life (Canadian Council of Ministers of the Environment, 2003) and, consequently, cannot be legally added to lakes in most jurisdictions.

Beneficial Effects

- Copper sulfate application used to be standard treatment for algal problems for many decades due to its short-term effectiveness.
- Short-term beneficial effects include suppression of algae; however, the negative effects outweigh this positive.

Undesirable Effects

- Ineffective for long-term treatment.
- Potential human health problems.
- Mortality of toxic algae from copper sulfate may result in the release of cellular toxins such as microcystin.
- Resistance may develop in target algae.
- DO depletion can occur when large volumes of dead algal cells decompose.
- Reduce potential binding capacity of lake sediment.
- Negatively impacts aquatic communities.
- Copper stress impairs food-web functions.
- Accelerated phosphorus recycling from lake bed.
- Copper accumulation in the sediment.
- Disappearance of macrophytes.
- Reductions in benthic macroinvertebrates.

Suitable Lake Conditions

Successful implementation of algicide as reviewed in the literature.

Lake size: 84–224 hectares (58 years of copper sulfate additions in the Fairmont Lakes, MN).

Depth: For lakes with a methyl orange alkalinity > 40 mg/L as $CaCO_3$, the dose for planktonic algae is 1.0 mg $CuSO_4 \cdot 5$ H2O per litre, as copper sulfate crystals, for the upper 0.3 m depth, regardless of actual depth.

Temperature: More effective >15 °C.

Alkalinity: For lakes with a methyl orange alkalinity >40 mg/L ($CaCO_3$), dose is 1.0 mg $CuSO_4 \cdot 5\, H_2O$ per litre. If alkalinity is < 40 mg/L, the dose is 0.3 mg $CuSO_4 \cdot 5\, H_2O$ per litre.

Longevity: Applications of algicides provide only temporary relief for algal problems and will require continued reapplication.

Cost

Single treatment costs, Casitas Reservoir, CA (AWWARK, 1987).

$CuSO_4$ solution: $220–$650.

$CuSO_4$ crystals: $197–$1,185.

$CuSO_4$ citric acid solution: $127–$1,500.

Copper-ethanolamine granular: $710–$3,000.

Application costs vary greatly: granular copper sulfate ~$2/kg; liquid Cutrine Plus ~$10/litre.

15. WATERSHED MANAGEMENT AND PROTECTION FROM EXTERNAL LOADING

It is generally agreed that in-lake remediation technologies will usually fail to provide long-term relief unless external loads of nutrients are reduced or controlled. The primary aim of in-lake remediation methods is to decrease factors that slow lake recovery following reductions of external loads. In addition to point source controls, reductions in agriculture runoff of nutrients, reestablishment of wetlands and littoral zones, and restoration of channelized streambeds have been shown to restore many lakes (Jeppesen et al., 1999). Land-use modifications can be used to control nutrient loss from the watershed and therefore improve lake water quality. Following are a few watershed management techniques to control and reduce nutrient loading to water bodies.

15.1 Wetland Rehabilitation and Constructed Treatment Wetlands

Natural wetlands retain water and store nutrients and other materials. However, many wetlands have been drained or filled, thus eliminating these important functions. Restoration of wetlands can return function to the landscape, thereby protecting lakes and streams, as well as reducing the volume and frequency of floods (Cairns et al., 1992). The construction of new wetlands is another approach to mitigate losses or to treat nutrient loading from diffuse sources. Retention of runoff P by constructed wetlands is influenced by P loading, amount of P attached to solids and P settling velocity (Braskerud, 2002). Constructed treatment wetlands are generally built on uplands and outside floodplains or floodways to avoid damage to natural wetlands and waterbodies. Wetlands can be constructed by excavating, backfilling, grading, diking and installing water control structures to established hydraulic flow patterns (USEPA, 2000). Wetland flora are then planted or allowed to establish naturally.

Potentially adverse environmental impacts include alteration of natural hydrology, introduction of invasive species and disruption of natural flora and fauna communities. Impacts can be mitigated by thorough design and planning.

15.1.1 Case Studies and Effects on Lake Quality

Agricultural field drains have been successfully treated with constructed wetlands (Kovacic et al., 2000). In one study, the application on a potato field was monitored over three years, and approximately 48 per cent of TP was retained over three years as pond soil in the sedimentation basin (Higgin et al., 1993).

A large constructed wetland (14 km^2) was used in the rehabilitation of Lake Apopka, Florida. The lake is 125 km^2 in size, with a mean depth of 1.6 m, and became hypereutrophic due to agricultural runoff. Results from the pilot-scale treatment (2.1 km^2) over 29 months indicated a reduction in TSS and TP of 85 per cent and 30 per cent, respectively (Coveney et al., 2002). Costs for the full-scale project were estimated at $1.6 million (2002 USD) (Coveney et al., 2002).

15.1.2 Cost

Daily flow design and treatment objectives determine the size, and therefore the cost of constructed treatment wetlands. The University of Guelph estimated the cost for various water volume and size designs (Table 15.1). Cost estimates were sourced from Aqua Treatment Technologies.

Table 15.1: Cost estimates for three treatment designs based on size and water volume

Volume of Water (litre per day)	Size (m^2)	Construction Cost	Cost (per L daily flow)	No. of Wetland cells
10,000–25,000	400–1,000	$65,000–$100,000	$6.50–$4	4
25,000–75,000	1,000–3,000	$100,000–$250,000	$4–$3.30	4
75,000–150,000	3,000–6,000	$250,000– $450,000	$3.30–< $3	4

Source: University of Guelph, 2012.

15.2 Lakeshore Rehabilitation and Shoreline Vegetation

The riparian zone is the community on the edges of a stream or lake and the surrounding land, and has a major influence on water quality. The riparian zone functions to reduce surface and sub-surface runoff volume, protect banks from erosion and lower pollutant concentrations in runoff (Dosskey, 2001). Furthermore, the zone is often characterized by a high gradient of submerged, emergent, semi-terrestrial and terrestrial vegetation, high species diversity, high biomass and productivity, high retention of materials, and periods of significant export of dissolved and particulate organic material that subsidize aquatic food webs (Wall et al., 2001). Creating a vegetative buffer zone between land development and streams or lakes provides protection by intercepting nutrients and sediments, and assists in restoring lost biodiversity (Wall et al., 2001; Dosskey, 2001).

15.2.1 Cost

The restoration of Jessie Lake, Minnesota incorporated a shore buffer zone and had an estimated installation cost of $285,000 to $475,000 and an individual buffer cost ranging from $30–$50 per lineal foot. A typical lake or stream buffer zone ranges from 15 to 100 feet with corresponding removal efficiencies for phosphorus at approx. 50–70 per cent (Itasca Conservation District, 2011). To achieve water quality goals, installation of native buffers along 30 per cent of the lake's shore (9,472 feet) would remove 60 lbs. P/year. Removal of phosphorus was calculated based on 30-foot buffers with a design standard 60 per cent P removal.

Lakeshore property owners also play an important role in lake protection and restoration. Fertilized lawns are sources of nutrients and the removal of aquatic vegetation along the shoreline causes sediment resuspension, adding nutrients to the water column. Land rehabilitation for private owners involves eliminating the use of fertilizers and low maintenance vegetation. Landscaping costs involve the installation of turf and shrubs, estimated at $5,000 to $40,000 per hectare (2015 USD).

15.3 Retention Ponds

Retention ponds, or stormwater retention basins, are water control structures that provide both retention and treatment of contaminated water runoff and overflow from rainfall and snowmelt. Natural wetlands on agricultural landscapes act as sponges, absorbing large amounts of snow melt runoff. Retention ponds respond in a similar manner; they have a higher capacity to store runoff during period of high flow, and filter and capture nutrients. The delay of water release also reduces downstream effects. Retention ponds also provide the opportunity to recapture and reuse nutrients, such as phosphorous, by harvesting the biomass material grown in these areas. Retaining storm water or flood-event water for more than 24 hours can reduce particulate loads up to 90 per cent; however, minimal soluble nutrients will be removed. Nutrient retention, upwards to 50 per cent of TP, can be increased by a two-stage design: the top section is dry between events and a smaller permanent wet pond remains at the outlet. An additional benefit to detention ponds comes from reducing peak stream velocity, protecting stream banks and riparian zones, and therefore reducing the silt load.

Ponds should be sized to hold runoff from the mean storm flow, preferably the volume of a 2.5 cm storm (Przepiora et al., 1998). The most useful pond size indicator is the ratio of pond volume to mean storm runoff volume (VB/VR). A VB/VR of 2.5 is expected to remove 75 per cent of suspended solids and 55 per cent of TP (Schueler, 1987). Furthermore, the National Urban Runoff Program recommended a wet pond with a surface area outlet, a mean depth of 1.0 m and a surface area equal to or greater than 1 per cent of the watershed area (Athayde et al., 1983). Deepening the ponds will increase the area for P removal; however, very deep ponds can also thermally stratify and cause P recycling. Furthermore, increased nutrient removal can occur with biomass harvesting. If ponds are located on native streams, modifications to pond design may be required to avoid disruption of fish passage.

15.3.1 Case Study and Effect on Lake Quality

Retention ponds decrease the potential for downstream flooding and stream bank erosion, and improve water quality by the removal of suspended soil, metals and particulate nutrients. Retention pond application at Lake Sammamish, United States, demonstrated the importance of design and size, and its effect on nutrient removal. Two ponds were created to protect Lake Sammamish from drainage impacts. Pond C, constructed in a horseshoe shape to minimize short-cutting where stormwater passes through the pond with no displacement of pond water, had a detention time of one week and an area that was 5 per cent of its watershed (40-hectare watershed) (Comings et al., 2000). Pond A was designed with three cells, but allowed for short-cutting through the first two cells. Pond C removed 81 per cent of TSS, 46 per cent of TP, 62 per cent of soluble P and 54 per cent of bioavailable P. Pond A removed 61 per cent of TSS, but only 19 per cent of TP, 3 per cent of soluble P and 19 per cent of bioavailable P (Comings et al., 2000).

15.3.2 Cost

The USEPA estimated the typical costs for wet retention ponds to range from \$17.50–\$35.00 per cubic metre (Center for Watershed Protection [CWP], 1998), approximately \$20–\$50 (2015 USD). The total cost of the treatment includes permitting, design and construction, and maintenance costs. Furthermore, the construction cost of retrofitting a wet retention pond into a developed area may be 5 to 10 times that cost of constructing the same size pond in an underdeveloped area (USEPA, 1999). Annual maintenance costs are estimated at 3 to 5 per cent of construction costs (Schueler, 1992).

15.4 Nutrient Diversion and Advance Wastewater Treatment

Diversion and advance wastewater treatment (AWT) are two techniques used to reduce external loading. Diversion of treated sewage or industrial wastewater involves installing interceptor lines to convey the wastewater away from the degraded water body to waters that have greater assimilative capacity. The wastewater may already be collected in a sewer system and represent a point source, which requires only a connecting pipe for diversion. Where individual household septic tank drainfields or stormwater runoff constitutes nonpoint sources, a collection system may be a necessary part of the diversion project. Diversion may require large pipes to transport wastewater long distances at relatively high cost. Diversion is expected to have similar effects to P removal through advanced wastewater treatment (AWT). The rate of recovery depends on several factors. Lakes usually return to near previous trophic state or improved water quality, after reduction in external P loading, although the rate of recovery may be slowed by internal loading.

15.4.1 Case Studies, Lake Response and Effects on Lake Quality

There are several reviews of lake response to external nutrient load reduction (Uttormark & Hutchins, 1980; Cullen & Forsberg, 1988; Marsden, 1989; Sas et al., 1989; Jeppesen et al., 2002). These projects indicate that while lakes respond to external load reduction, the response can be slow. Cullen and Forsberg (1988) reviewed the response of 42 lakes to external load reduction (Table 15.2). The response varied among three groups: "sufficient to change trophic category" (Type I), "reduction in lake P and chlorophyll-*a*, insufficient to change trophic category" (Type II) and "small or no obvious improvement or reduction in lake P and with little e reduction in chlorophyll-*a*" (Type III). The magnitude of external loading (inflow concentration) reduction averaged from approximately two-thirds to three-fourths of the pre-treatment loading. Lake P concentration decreased, but was often considerably less than load reductions. Chl-*a* decreased on average only in the first two categories. The criterion used for trophic state change was 25 µg/L TP for the eutrophic-mesotrophic boundary.

Table 15.2: Results of diversion and advance treatment of nutrient inputs to 42 world lakes

	TP_1	TP_2		Chl-*a*	
	% change	% change	Conc.	% change	Conc.
Type 1 (n=15)	-74 ± 18	-38 ± 14	28 ± 23	-37 ± 18	5.4 ± 5.4
Type II (n=9)	-76 ± 10	-51 ± 14	118 ± 118	-57 ± 17	26 ± 29
Type III (n=18)	-64 ± 22	-67 ± 14	100 ± 152	+216 ± 394	44 ± 49

Type I = sufficient to change trophic.
Type II = reduction in lake P and chlorophyll-*a*, insufficient to change trophic.
Type III= small or no obvious improvement or reduction in lake P and with little e reduction in chl-*a*.
TP_1 = equilibrium concentration; TP_2 = average inflow concentration (µg/L ± 1 SD).

Source: Cullen & Forsberg (1988).

Decline in lake P can occur following external load reduction, even if internal loading is high (Cooke et al., 2005). However, the difficultly in forecasting the extent of recovery is in predicting equilibrium concentrations in lakes with substantial internal loading. This is particularly an issue in shallow lakes where several mechanisms of internal loading may be operating. All lakes had reduced annual mean lake TP concentration, however, the percent reduction in TP was less than the reduction in loading. Inflow TP decreased 82 per cent on average, while in-lake TP decreased 73 per cent. These means were values based on the highest before and lowest after treatment concentrations.

16. EUTROPHICATION AND MANITOBA PRAIRIE LAKES

Lake eutrophication is a natural process resulting from the gradual accumulation of nutrients, increased productivity and slow filling in of the basin with accumulated sediments, silt and organic matter from the watershed. The original shape of the basin and the relative stability of watershed soil strongly influence the lifespan of a lake. Many lakes would be eutrophic despite development in the watershed due to Manitoba's regional soil fertility, runoff patterns and geology, which encourage eutrophic natural conditions. However, human-induced cultural eutrophication occurs when nutrient, soil or organic matter loads to the lake dramatically increase, significantly shortening a lake's lifespan. The success of remediation treatments varies greatly from lake to lake, and it is generally agreed that these treatments are usually not worth considering unless external nutrient loads can also be reduced and controlled.

Lakes are highly dynamic and interactive systems, and it is impossible to alter one characteristic without affecting other aspects of the system. A complex set of physical, chemical and biological factors influences lake ecosystems and affects their responsiveness to remediation and management efforts. These factors vary with lake origin, the regional setting and the watershed, and include hydrology, climate, watershed geology, watershed-to-lake ratio, soil fertility, hydraulic residence time, lake basin shape, lake biota, the presence or absence of thermal stratification, and external and internal nutrient loading sources and rates. Appropriate evaluation can determine the feasibility of controlling the primary sources of the most limiting nutrient.

This report identified multiple in-lake remediation treatments to limit and control P-enriched sediments and remediate the effects of eutrophication on lake water quality (Table 16.1). Common remediation treatment methods have been reviewed; however, it is not entirely exhaustive. To assess remediation techniques effectively, however, requires an analysis of a waterbody's limnological and morphological parameters. This report is part of a broader project of the International Institute for Sustainable Development (IISD) working in partnership with the Province of Manitoba. Continuing work includes characterizing and comparing available lake chemistry and physical data for selected Manitoba prairie lakes, and a decision-support framework that includes in-lake remediation approaches in relation to characteristics of the lakes concerned.

Table 16.1: Summary of selected in-lake remediation treatments

Remediation Technique		Appropriate Lake Conditions	Beneficial Impacts	Potential Adverse Ecological Effects	Cost
Biomanipulation	Grazing of algae by large zooplankton can be enhanced by eliminating planktivorous fish through physical removal or increased piscivory.	In theory, there are no restrictions on lake size. Lake size: Lakes <25 hectares in size have had the highest percentage of success. Depth: Greatest probability to reduce algal biomass <3 m. P load: 1-14 kg $ha^{-1}year^{-1}$. Lakes with external P loading <0.6 g $Pm^{-2}yr^{-1}$. TP: Successful application 0.05-1.4 mg/L. Recommended P conc. <100 µg/L. Chl-*a*: 21-300 µg/L. Longevity: Enormous variability in success and multiple restocking events might be necessary. Long-lasting results are rare.	Water-quality improvements include increased transparency, decreased turbidity, decreased Chl-*a*, TP and TN conc. Technique does not require complex infrastructure or potentially toxic chemicals.	Resistance to grazing by large cyanobacteria. Replacement of fish predation by invertebrates (*Chaoborus*). Long-term unsustainability of the fish population. Nutrient transport by fish. Increased planktivory by invertebrates. Resuspension of sediments.	Generally, method is inexpensive relative to other techniques. $273,000 (Twin Lake, MN total cost, 20-year lifetime). $127,000 (Lake Nokomis, MN total cost, 10-year lifetime).
Floating Treatment Wetlands	Floating treatment wetlands (FTWs) are comprised of basins and cells to make an artificial platform containing emergent macrophytes. The primary mechanisms for nutrient removal are microbial transformation and uptake; macrophyte assimilation; absorption into organic and inorganic substrate materials; and volatilization.	Floating wetland treatment is suitable for a wide range of lake characteristics and water-quality conditions. Lake size: Most efficient on small lakes, ponds, small reservoirs and retention ponds. Depth: Minimum water depth >1 m to prevent platform plants from rooting into lake bottom sediment. Ideal depth 1.5–2 m. Longevity: With relatively low maintenance and secured placement, FTW will continuously sequester nutrients in the plant material. Harvesting material increases nutrient removal and longevity.	Rooted macrophytes extract nutrients from both the sediment and the water column. Reduce redox potential and anoxic conditions. Harvesting platform plant material and the removal of biomass can further reduce nutrient concentration. Increase wildlife habitat and reduce local nuisance insect populations. Increase waterbody aesthetics.	Little to no adverse effects on lake quality mentioned in the literature. Potential effects on N:P ratio, with effects on cyanobacterial growth. Potential to restrict access or reduce available area for recreational use. Potential for anoxic conditions with high lake surface coverage.	Relatively inexpensive compared to physical and chemical remedial treatments. Cost is determined by lake characteristics and condition. FWT platforms range $11-$260 per square metre. Biohaven™ FTW installation in a 2.1 hectare pond, 0.7 per cent surface area coverage: $40,000 (installation, 70 plants and monitoring for one year).
Removal of Macrophytes	Removing macrophyte biomass from lakes removes nutrients, which for some lakes can be a significant contribution to internal loading. Thick overstory, as well as decomposition of organic matter, contributes to oxygen deficiency and sediment phosphorus release, which can be alleviated by macrophyte removal.	Parameters involved to calculate the potential for removing nutrients: • Area of the lake covered with macrophytes. • The average biomass of the plants in the area. • Nutrient concentration of the plants. Successful implementation of as reviewed in the literature. Lake size: 10–5,300 hectares. Depth: 2.4–5 m (shallow lakes). P load: 1,890 tonnes of N and 296 tonnes of P were removed by harvesting 60 x 10^4 tonnes of macrophytes, which corresponded to 28 per cent and 57 per cent, respectively, of external loading. Longevity: Harvesting is continuous and multiyear obligation for maximum affect in the long term.	Extract nutrients from both the sediment and the water column. Long-term, harvesting macrophytes can affect nutrient cycling between the water column and the sediment. Increase waterbody aesthetics.	Immediate physical, and prolonged physical and chemical effects on biota and ecosystem processes. Directly and indirectly removes fish, invertebrates and other species from the ecosystem. Loss of habitat for grazers. Fish common in the littoral zone are often considered desirable for fishing. Reducing macrophytes decreases competition with algae and may even promote algal blooms.	Cost is variable upon harvesting method, area harvested, plant species and density and water depth. Range in the literature $650–$1000 per hectare. $42,000 per year or $728 per hectare to harvest 60 hectares. $550,000 per year: Chautauqua Lake, New York to harvest 5,300 hectares, 2,348 tonnes removed in 2014. Cost of a large system harvester $50,000–$200,000. Smaller harvesters (attached to a boat) are significantly less expensive.

Remediation Technique		Appropriate Lake Conditions	Beneficial Impacts	Potential Adverse Ecological Effects	Cost
Hypolimnetic Withdrawal	Nutrient-enriched hypolimnetic waters can be preferentially removed through selective discharge, where the direct removal of P-laden bottom water and changing the depth at which water leaves the lake from the surface to near the maximum depth, allowing nutrient-rich rather than nutrient-poor water to discharge. Consequently, hypolimnetic withdrawal shortens hypolimnetic retention time, decreases the chance for anaerobic conditions to develop, accelerates phosphorus export, reduces surface phosphorus concentrations, and improves hypolimnetic oxygen content.	Successful implementation is restricted to deeper, stratified lakes with considerable internal loading. Successful implementation as reviewed in the literature. Lake size: 1.5–400 hectares. Depth (mean): 3.0–48.0 m. Depth (max.): 6.8–56 m. Residence time: 0.26–9.0 years. Longevity: Effectiveness of treatment depends on magnitude and duration of TP transport from the hypolimnion, and it is important to exchange the hypolimnion volume as frequently as possible. A low rate of replacement may limit the effectiveness and longevity of treatment. Important to understand natural refilling rate and if it is high enough to reduce lake drawdown resulting from hypolimnion discharge. Smaller lakes may refill too slowly to be effective.	Relatively low capital and operational costs, and potentially long-term effectiveness. Hypolimnetic DO increase, which can result in a decrease in the anoxic volume and days of anoxia. Reduce the accessibility of cyanobacteria to Fe(II), now thought to be a precursor to the development of blue-green blooms. Increase in hypolimnetic DO can improve fish habitat.	Potential for water quality issues downstream if hypolimnetic water contains high conc. of P, ammonia, hydrogen sulfide and low oxygen. Withdrawal followed by treatment and discharge back to the lake is inefficient in removing phosphorus compared to in-lake treatment. Potential warming of the lake as bottom waters are exposed to surface temperatures. Destabilization of the thermocline and enable nutrients from the hypolimnion to become available for phytoplankton growth in the epilimnion.	Relatively low capital and annual operation cost. Cost (range from literature): $80,000–$600,000. Twin Lake, MN (8 ha) $1.3 million (total cost, 20-year treatment life) $400,000 (construction cost) $40,000 (annual operation)
Dilution and Flushing	Improved quality in eutrophic lakes by reducing the concentration of the limiting nutrient (dilution) and by increasing the water exchange rate (flushing). Dilution involves the addition of low-nutrient water to reduce lake nutrient concentration and has been effective where external or internal sources are not controlled. Flushing refers to the removal of algal biomass.	Generally limited to relatively small lakes where there is sufficiently large amounts of low-nutrient water to effect a decrease in nutrient concentration. Successful implementation as reviewed in the literature. Lake size: 104–490 ha. Depth (mean): 3.8–5.6 m. P load: In-lake nutrient conc. are usually lower than inflow conc. because sedimentation is greater than internal loading. Nutrient load is usually increased with this strategy; however, nutrient loss through sedimentation is potentially decreased. Chl-*a*: 71–102 μg/L. Flushing rate: 5.8–17% per day or ≥ 1.0/year large enough initially to reduce in-lake concentration.	Relatively low cost if water is available in high quantity. An immediate and proven effectiveness if the limiting nutrient can be decreased. Moderate success even if only moderate-to high-nutrient water is available.	If dilution water is derived from a source outside of the catchment, there may be a risk of introducing undesirable taxa. Potential impacts on the diverted water source.	High variability and dependent upon the presence of a facility to deliver water, and the quantity and proximity of available water. Cost range from literature: $100,000–$800,000. Moses Lake, WA: Primary cost was the pumping facility $750,000.

Remediation Technique		Appropriate Lake Conditions	Beneficial Impacts	Potential Adverse Ecological Effects	Cost
Hypolimnetic Aeration and Oxygenation	Hypolimnetic aeration is usually accomplished by the injection of pure oxygen or air into the hypolimnion, without disturbing stratification. Hypolimnetic aeration is applied to raise the oxygen content of the hypolimnion without destratifying the water column or warming the hypolimnion. If sediment-to-water exchange of P is controlled by iron redox, aeration reduces sediment P release by establishing undesirable conditions at the sediment–water interface.	Hypolimnetic aeration will not be effective if the waterbody is too shallow. Although stratification may exist, the density gradient may not be sufficient to resist thermocline erosion. While hypolimnetic aeration may restore oxygen conditions for fish and other biota, other toxic elements may not be sufficiently reduced to allow survival. Successful implementation as reviewed in the literature: Lake size: 5.3–3,000 ha. Depth (mean): 3.5–28.4 m and lake must be stratified. Depth (max.): 5.7–85 m. Not recommended if max. depth is less than 12–15 m and/or hypolimnetic volume is relatively small. Device depth: 5.2–33 m. Longevity: Continual treatment.	Anoxic hypolimnion can switch to an oxic state while still maintaining a coldwater environment. Potentially decrease internal loading of P, Fe, Mn, ammonium, hydrogen sulfide and methyl mercury. Aeration may improve habitat quality for coldwater fish, even if improvements in epilimnetic water quality are not achieved.	Interactions between Fe and P primarily affect only the short-term cycling of P, and do not result in the permanent storage of P in lake sediment. Phosphorus improvements do not always occur with aeration. Lakes where internal P recycling is driven by processes unrelated to Fe-P interactions may not show any positive effects on nutrient loading. Potential supersaturation hypolimnetic water with N_2 can lead to gas bubble disease in fish. Potential to increase eddy diffusion of nutrients into the epilimnion, even though stratification is maintained. Slow circulation conditions and destratification may result in low DO throughout the water column and introduce toxic chemicals (H_2S) into the epilimnion.	Less cost-effective than other treatments for phosphorus control (e.g., alum); however, there are other reason for aeration, such as creating an aerobic environment. $4,000 /ha/year (mean areas of 15 lakes). Lake Steven: $340,000 (operating 160 days per year) $0.27/kg O_2 or $1,610 per ha
Artificial Circulation and Aeration	Circulation of the entire lake rather than a select region or depth, and the temperature of the whole lake will increase with complete circulation if mixing includes water that was previously part of the cooler hypolimnion. The principal improvements in water quality are oxygenation and chemical oxidation of substances in the entire water column, as well as enlarging the suitable habitat for aerobic warm-water species. Circulation improves dissolved oxygen and reduces iron and manganese, as well as causes light to limit algal growth in environments where nutrients are uncontrollable and neutralize the factors favouring the dominance of blue-green algae.	Successful implementation as reviewed in the literature. Lake size: 9.1–18 hectares. However, successful implementation Lake Nieuwe Neer (Netherlands) 132 ha in size. Depth (mean): 2.6–3 m. Longevity: Continual treatment and management.	Circulation can reduce phytoplankton biomass by increased depth of mixing of plankton cells and increased light limitations. Increased circulation usually results in the complexation and precipitation of Fe and Mn, reducing trace elements and P internal loading, therefore algal biomass. If sediments are distributed by mixing, algal biomass may also decrease due to decreased light availability. Improvement of warm-water fisheries.	If circulation increases the suspension of particulate material, associated P may mineralize and become available to phytoplankton. Mixing of sediment may increase inorganic turbidity. Whole lake circulation will result in the loss of deep coldwater habitat for fish in stratified lakes. Overall lake temperatures typically increase following treatment.	Cost increases with lake size, although costs per hectare decline, demonstrating economies of scale. Lake size: > 53 ha = $760/ha. 23–25 ha = $1,680/ha. <10 ha = $7,743/ha. Twin Lake, MN: $935,000 (20-year lifetime) $35,000 per year (maintenance cost) Initial cost: $520–$6100 per ha. Annual cost: $150–$2940 per ha.

Remediation Technique		Appropriate Lake Conditions	Beneficial Impacts	Potential Adverse Ecological Effects	Cost
Dredging and Removal of Surficial Sediment	For lakes where significant nutrient loading from sediment occurs, removal of nutrient-rich surficial sediments has the potential to reduce the rate of internal nutrient recycling, improving overall lake water quality. In addition to removing nutrients in bottom sediments, removal may also decrease cyanobacterial innocula.	Dredging is generally limited to shallow lakes (<3 m) but lake area is not a constraint. Depth, size, disposal area, watershed area and sedimentation rates are important physical variables that affect treatment feasibility. Successful implementation as reviewed in the literature: Lake depth: Highest success <3 m. Depth as reviewed in the literature: 0.5–9.75 m (max. depth). Sediment depth: Dredging will only be effective in lakes with high nutrient-enriched surface sediments relative to underlying sediment. Top 0.3–0.5 m of a sediment core, then removal of the layers by dredging should provide reliable and permanent solution, although costly. Sedimentation rate: Low Water-to-surface ratio: Small, 10:1. Hydraulic retention time: Long. Watershed sourced loading: Requirement is a reduction in external loading of at least 50%. Longevity: Long-term benefit of removing the nutrient source.	Lake deepening. Expand habitat. Limit nutrient recycling. Reduce macrophyte nuisance. Remove toxic sediment.	Resuspension of sediments on aquatic organisms including clogging filtering apparatus of benthos and zooplankton, and reduction of light. Many fish species cannot tolerate high sediment loads. Nutrient liberation from disturbed sediments and porewaters. Potential release of toxic substances associated with fine particulars (polluted). Destruction of benthic fish-food organisms and its effect on a lake's food web. Lake draining will result in the mortality of most native aquatic biota.	Main objection to dredging is the high cost. Project-to-project cost comparison for sediment removal is difficult due to a large number of variables that affect dredging cost. \$34–\$1,409 per m^3 (removal of contaminated sediments). \$1–\$30 per m^3; \$3,200–\$60,000 per hectare (as reviewed in the literature). Twin Lake, MN: \$2,570,000
Phosphorous Inactivation and Sediment Capping	Many different methods of sediment capping and P inactivation have been used in lake restoration projects including physical (mechanical or passive) capping and active capping using alum, calcium, zeolite, Phoslock™, iron and modified clays. Passive capping using sand, gravel, or clay is used to decrease diffusion of nutrients and contaminants to the overlying water column and bury them deeper in the sediments.	Capping thickness usually exceeds 5 cm, which limits the approach to small lakes or reservoirs due to the large volume of material required and difficulties of depositing a uniform layer. Longevity: P inactivation longevity does not typically exceed 15 years and will depend on PO_4 release rates and application dose. Alum longevity typically 4–21 years (stratified); 1–11 years (shallow lakes). **Alum** Lake size: 9–600 hectares. Mean depth: 1.8–8.4 m Max depth: 3.5–23.7 m pH: 6–8 throughout treatment Alkalinity: <50 mg $CaCO_3$/L; will lower pH if lake has low alkalinity. **Calcium** Lake size: 58–240 hectares. Mean depth: 10–18 m Max. depth: 30–42 m. Hydraulic res. time: 4.4 yrs. (Lake Schmaler). Chl-*a*: 4.0 Secchi depth: 5 m pH: 8.5 **Phoslock™** Lake size: 0.9–64 hectares. Mean depth: 1.6–8.8 m. Max depth: 2.5–34 m.	**Alum** Proven effective control. Calcium Extensively used in hard-water lakes. Calcium additions to hard-water lakes have fewer toxic impacts than alum. **Phoslock™** Proven effective control, non-toxic under a wide range of environmental conditions; effective under a wide range of pH values and alkalinities; does not affect pH levels following treatment (advantages over alum).	**Alum** Restricted to a narrow pH range; additions to low-alkalinity lakes can result in acidification. Toxicity Bioaccumulates in fish tissue. **Calcium** With increased turbidity, potential smothering of benthos by $CaCO_3$. **Phoslock™** Potential toxicity of La. Long-term negative ecological impacts not well understood.	**Alum** Twin Lake, MN: \$148,000. Jessie Lake, MN: \$508,000 (40% alum) \$754, 000 (60% alum) **Phoslock™** \$440–\$880 per k for biologically available P immobilized. If 50 mg mobile P/kg in top 10 cm: \$0.75. If 400 mg mobile P/kg in top 10 cm: \$6.02.

Remediation Technique		Appropriate Lake Conditions	Beneficial Impacts	Potential Adverse Ecological Effects	Cost
Sediment Oxidation	Sediment oxidation by enhanced denitrification results in improved complexation with iron (Riplox). The method reduces internal loading in lakes where iron redox reactions control P fluxes between sediment and overlying water. Nitrate, as Ca $(NO_3)_2$, iron (as $FeCl_3$), and lime ($CaCO_3$) are injected into lake sediments using a discing harrow. Nitrate acts as an alternate electron acceptor to oxygen, preventing the development of ferrous iron and subsequent P release.	Method requires direct injection of chemicals into sediments, it can typically be used in shallow lakes with relatively flat bottoms. Successful implementation as reviewed in the literature. Lake size: 4.2–49 hectares. Depth: Suitable for shallow lakes; 0.7–2.3 m. Injection depth: 0.2–3 m. pH: Ferric chloride and lime additions have been determined as unnecessary in some cases, where pH may be sufficiently high to promote denitrification and sediment iron content adequate (30–50 mg/g) for P binding. Longevity: Continued low sediment P release 10 years following treatment was observed for Lake Lillesjön, Sweden.	Although greater in cost, treatment is an effective alternative to alum to inactivate sediment P. Chemicals added are found in high conc. naturally in unpolluted sediments. Toxicity to animals is perceived as a lesser issue than for other P inactivation methods. Potentially more permanent than alum due to direct injection into the sediment column.	Expected to succeed only if internal loading of P is controlled by iron redox reaction. Concerns over the lack of documented successful applications.	Comparatively expensive remediation method. Lake Lillesjön, Sweden (applied to 1.2 hectares lake area): $232,500 Lake Trekanten, Sweden (87 hectares): $609,000. White Lough, Ireland: Nitrate: $43,000 ($9,350 per ha) Iron/alum: $11,500 ($2,500 per ha)
Algicide	Commonly used in earlier lake remediation management and water supply reservoirs suffering from algal biomass, algicide is not frequently practiced due to significant detrimental aspects associated with the technique on the lake's biological community. It is important to note that dose application concentrations for copper sulfate (Chara and Nitella require a dose of 1.5 mg/L or higher) greatly exceed existing Canadian limits for the protection of aquatic life (Canadian Council of Ministers of the Environment, 2003) and, consequently, cannot be legally added to lakes in most jurisdictions.	Successful implementation as reviewed in the literature: Lake size: 84–224 hectares (58 years of copper sulfate additions in the Fairmont Lakes, MN). Depth: For lakes with a methyl orange alkalinity > 40 mg/L as $CaCO_3$, the dose for planktonic algae is 1.0 mg $CuSO_4 \cdot 5\ H_2O$ per litre, as copper sulfate crystals, for the upper 0.3 m depth, regardless of actual depth. Temp: More effective >15 °C. Alkalinity: For lakes with a methyl orange alkalinity >40 mg/L ($CaCO_3$), dose is 1.0 mg $CuSO_4 \cdot 5\ H_2O$ per litre. If alkalinity is < 40 mg/L, the dose is 0.3 mg $CuSO_4 \cdot 5\ H_2O$ per litre. Longevity: Applications of algicides provide only temporary relief for algal problems and will require continued reapplication.	Copper sulfate application used to be standard treatment for algal problems for many decades due to its short-term effectiveness. Short-term beneficial effects include suppression of algae; however, the negative effects outweigh this positive.	Ineffective for long-term treatment. Potential human health problems. Mortality of toxic algae from copper sulfate may result in the release of cellular toxins such as microcystin. Resistance may develop in target algae. DO depletion can occur when large volumes of dead algal cells decompose. Reduce potential binding capacity of lake sediment. Negatively impacts aquatic communities. Copper stress impairs food-web functions. Accelerated phosphorus recycling from lake bed. Copper accumulation in the sediment. Disappearance of macrophytes. Reductions in benthic macroinvertebrates.	Application costs vary greatly: granular copper sulfate ~$2/kg; liquid Cutrine Plus ~$10/litre. Single treatment costs, Casitas Reservoir, CA (AWWARK, 1987). $CuSO_4$ solution: $220–$650. $CuSO_4$ crystals: $197–$1,185. $CuSO_4$ citric acid solution: $127–$1,500. Copper-ethanolamine granular: $710–$3,000.

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