Reviews into evidence for sediment enrichment and lake restoration best practice for the Windermere catchment II: in-lake restoration measures review

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1. Introduction

Addressing eutrophication problems, the major cause of poor water quality in lakes, necessitates the effective reduction of nutrients phosphorus (P) and nitrogen (N) to levels which can enable the water body and its biota to return to a near natural state, minimally impacted by human pressures. Water quality and ecological improvements are then expected to result in more fully functional ecosystems, providing wider benefits for biodiversity and ecosystem services to society as well as greater resilience to other stressors. However, long term delays in the recovery of lakes frequently occurs, despite reductions in current external loads, owing to the legacy of P accumulated and stored in catchment soils and lake sediments over previous decades to centuries (Jarvie et al. 2013). Internal loading of nutrients, particularly P, from lake bed sediments, can represent an important part of the P budget of a system, particularly as external loads decline in significance (Jeppesen et al., 2005; Søndergaard et al., 2013; Spears et al., 2012). Control of the internal load of P through management interventions can therefore be used to accelerate ecological recovery of lakes, where applied appropriately (Mehner et al., 2008).

Regulatory drivers for improving water and ecological quality in lakes stem from historic legislation from the European Union, in particular the Urban Waste Water Treatment (European Commission, 1991 (91/271/EEC)), Habitats (European Commission, 1992 (92/43/EEC)) and Water Framework (European Commission, 2000 (2000/60/EC)) Directives and more latterly targets set out in the Defra 25 year plan and UK Environment Act 2021. To date, the continued use of the WFD targets in the UK necessitates the achievement of ‘good ecological status’ across a range of biological quality elements in each water body by dates specified in the legislation. Since natural recovery from eutrophication can be very slow for some lakes, taking many decades, particularly where internal loading is a problem, processes to 'speed up’ recovery by managing the internal load may be necessary to achieve policy deadlines.

Action to improve water and ecological quality in UK lakes is particularly pressing given the current state of lakes. In 2022, 68% of all assessed lakes (n=1058) across the UK failed to meet ‘good ecological status’ under the Water Framework Directive classification (JNCC, 2023). Split by country, based on classification from 2020, this total is 86% (n=598) for England, 30% (n=334) for Scotland, 81% (n=114) for Wales and 86% (n=21) for Northern Ireland. These percentages have changed little since assessments began in 2009 (JNCC, 2023), suggesting that progress towards achieving regulatory targets has stalled. In a previous study, Duethmann et al. (2009) showed that a combination of catchment management and internal loading control would be required to address TP concentrations in a large proportion of lakes in England and Wales that were failing to meet WFD targets at that time. It is unlikely that this situation will have changed substantially since this analysis was carried out, suggesting that integrated management solutions are required to address eutrophication in lakes (Zamparas and Zacharias 2014).

Four main approaches are used to control internal nutrient loading in lakes. The management measures target different aspects of the, usually P, release processes. Three of these target the removal of P: by physical means such as sediment removal (Søndergaard, et al., 2007; Pitt et al., 1997) or chemical treatment of sediment to 'bind' the P within it and minimise P release (Hickey and Gibbs, 2009; Meis et al., 2012) or through removal of P from the water column via hypolimnetic withdrawal, which can eventually exhaust sediment supplies (Nürnberg 2007). The fourth method involves the aeration or oxygenation of the water column directly above the sediment. Low oxygen concentrations are frequently associated with low redox potential and the reduction of metals such as iron (Fe), which liberates P bound to the Fe in the process.

This report provides an overview of the main in-lake management measures adopted to try to address internal phosphorus (P) loading within lake waterbodies. The first part of the report provides a brief description of the measures, including some context on the control of external nutrient sources, which represents an important component of lake nutrient management. The report then goes on to consider each of the in-lake measures in turn. Assessment of the scientific literature returned by systematic searches is then undertaken to evaluate:

1) existing evidence for the success of each measure, and the strength of that evidence
2) the challenges involved in implementing the measures
3) whether the techniques have been used at the field scale
4) whether the technique is appropriate to be applied in a protected area
5) whether the technique is climate proof

Finally, a summary of the lessons learned and best practice approaches to in-lake restoration is made.

1.1 Lake restoration measures targeting catchment nutrient load reduction and internal P loading

Catchment and inflowing nutrient control measures

Although not the focus for this review of measures targeting in-lake nutrients, external nutrient control from catchment sources is an essential component of lake restoration practice. In part, this is because the planning of restoration and nutrient management actions should be carried out using a systems analysis approach (Shortle et al., 2020) considering the specific context of the lake (Lürling et al., 2016). It is also because the reduction and control of external sources of nutrients have critical implications for the potential success of in-lake measures, or the requirement for repeated in-lake treatment to establish sustained improvements in water quality over the long term (Fastner et al., 2016; Spears et al., 2018). Finally, climate change is expected to exacerbate the effects on nutrient enrichment through increased temperatures, longer stratified periods, and more extreme events including heatwaves, storms, high winds, floods, and droughts, which are not within the control of lake managers. Building resilience to these climate impacts in lake ecosystems through the reduction of external nutrient sources is likely to grow in importance over coming decades (Spears et al., 2022). However, even if external reductions have occurred, lakes with eutrophic histories may still be vulnerable to climate impacts. This is evident in the alpine Lake Constance which spans Austria, Germany, and Switzerland and whose significant depth (max. depth 251 metres) makes it vulnerable to temperature changes and increased thermal stability (Müller, 2002; Schranz, 2016). Significant international cooperation and investment (over €5 million) into state-of-the-art wastewater treatment has led to recovery from eutrophication and a return to its oligotrophic state, however, prolonged stratification from increasing temperatures is reducing the oxygen in its hypolimnion and increasing the risk of internal P loads (Müller, 2002; Schranz, 2016). Its depth and large volume complicates the design of its in-lake remediation strategy (Schranz, 2016).

Reduction in phosphorus loads from point sources has been widely acknowledged as a success over the last few decades in Europe and North America. Awareness of the issues around eutrophication from anthropogenic sources has developed since the 1960s and initial controls targeted the removal of phosphates in laundry detergents (Fastner et al., 2016). Examples of subsequent lake-specific interventions have targeted waste water and point sources, either through 1) upgrading treatment processes to remove P, 2) diverting the waste water source, or 3) removing the P from a lake inflow (Fastner et al., 2016). A key European driver for the upgrade of treatment processes has been the wide scale adoption of the European Union Urban Waste Water Treatment Directive, which brought in P emission concentration limits (1 – 2 mg P L^{-1}) for larger sewage treatment plants and those discharging in more sensitive areas. More recently, the EU Water Framework Directive has led to a reduction in these limits to 0.1 -0.5 mg P L^{-1}, to allow the attainment of the Good Ecological Status targets. Good ecological status is the slight deviation from baseline conditions of various biological and supporting elements, adopting a ‘one out, all out’ criteria. At the same time, other point source P emissions, such as those from industry have also been identified and controlled at a catchment scale.

There are a number of examples from lakes in Europe and North America where the three different forms of reduction in point source emissions listed above have resulted in reductions in P concentrations in lake waters. In some cases, this resulted in a return to very low levels of nutrients in these lakes, analogous to near natural conditions. Comparison of eight different lake case studies by Fastner et al. (2016) suggests that load reduction from point sources was generally effective at reducing total phosphorus (TP) and chlorophyll-*a* concentrations, and cyanobacteria biomass in the water. The relative success of these measures was, however, lake-specific and dependent on the residual external P load, either from treated effluent and other non-point sources, any internal P loading, and the water residence time of the lake. The length of time to recovery was also related to how long it took to reach the critical TP load to the lake, calculated from empirical models developed by Vollenweider

(Vollenweider, 1976). Rapid reductions in P loads below the critical level resulted in rapid recovery for lakes where waste waters were diverted, such as Lake Washington, or P stripping of the inflow carried out, such as Schlachtensee. It is important to note that examples which refer to reductions in P loading following diversions of wastewater indicate a transferal of water quality issues elsewhere (unless stated otherwise), and therefore, is not considered a holistic approach to reduction in P sources. Whereas slower reductions in load towards the critical level, through progressive upgrades to treatments works, as in the case of Onondaga Lake, New York, USA and Lake Maggiore, Italy, resulted in much longer response times (>10 years) in the algal and cyanobacteria biomass.

In-lake measures for internal loading control

A range of processes are important for controlling P release from lake sediments, including temperature, redox and pH changes and physical disturbance. Different processes may dominate in any particular lake, requiring lake-specific understanding of these patterns. However, one important factor differentiating the importance of different processes is the distinction between deep and shallow lakes, particularly with respect to whether the lake thermally stratifies. In stratifying lakes, which suffer from seasonal hypolimnetic de-oxygenation, the highest rates of P release from sediment tend to occur under anoxic, reducing (low redox potential) conditions at the sediment-water interface when Fe bound P is released. P concentrations (dissolved and total) tend to increase in the lake hypolimnion over the anoxic period, with the length and extent of anoxia linked to the size of the P pool in the hypolimnion at the end of the summer. Measures which typically target these processes include aeration and oxygenation, hypolimnetic withdrawal and chemical treatment of sediment. In contrast, shallow lakes which either do not or only rarely stratify are usually subject to more physical sediment disturbance from wind waves or bioturbation, causing P release or tend to be warmer and therefore promote more rapid microbially mediated P release. In shallow lakes, sediment removal or chemical treatment of sediments are the most viable options to target the P at source. An overview of these measures and two further considerations for addressing these internal P release processes is provided below.

Aeration and oxygenation

- The principle behind the use of aeration or oxygenation systems is to maintain sufficient oxygen concentrations at the sediment-water interface so that P remains bound to oxidised Fe within the sediment.
- Systems typically add air (aeration), oxygen enriched air or pure oxygen (oxygenation) into the deep water of lakes or reservoirs.
- Three main types of system are used, airlift aerators, down-flow bubble contact or Speece cones, and deep oxygen injection systems or bubble plume diffusers. Speece cones are inverted cones, with a water supply attached to the top of the cone which then flows down (Rogers, 2009). Oxygen is pumped in around the middle of the cone, and which due to the downward pressure of

the water supply at the top, prevents air bubbles from rising, thereby saturating the water at the bottom of the cone with oxygen (Rogers, 2009). It is often deemed a more efficient method to other aerators and plumes (Rogers, 2009).

- All systems are associated with sizeable installation and running costs, depending on the size of the waterbody.
- These technologies have been applied for many decades in efforts to control deep water oxygen concentrations and internal P loading, however, a recent review suggests that the effectiveness of such approaches can be variable (Lürling et al., 2020).

Hypolimnetic withdrawal

- Hypolimnetic withdrawal targets the P enriched water in the hypolimnion and is therefore used at the end of summer or early autumn, which typically coincides with the peak in the P concentration.
- The P enriched water is either siphoned or pumped from the hypolimnion and replaced by a source of low P water, ensuring a water balance.
- The removed water is either discharged downstream of the lake or, more beneficially, treated to remove much of the P content and returned to the lake or discharged downstream.

Sediment removal

- There are two main methods used, depending on the context of the lake, excavation or dredging (Lürling et al., 2020). Excavation can occur only in situations where the entire lake or sections of the lake are drawn down exposing the bed sediment, which is then removed by scraping. In contrast, dredging can be carried out while lakes contain water.
- The method is frequently used to improve navigation, recreation or hydrological capacity, although it has also been adopted to improve water quality, including a reduction in internal nutrient cycling.
- Significant quantities of dredged material are generated, therefore adequate facilities adjacent to the water body are required to safely de-water sediments.
- Careful consideration of the disposal of the sediment, which may be enriched in a range of compounds, is required.

Chemical treatment of sediments

- Sediment oxidation and chemical inactivation are the main processes involved in chemical treatment.

- Sediment oxidation uses the same principles as the aeration or oxygenation techniques listed above, but instead provides alternative electron acceptors in the form of chemical compounds added to sediments.
- The process of chemical inactivation involves application of additional P-binding compounds to sediments to increase sediment P absorption capacity and, over time, potentially transform P into more recalcitrant forms that are less likely to release into the overlying water.

Other considerations

In addition to the implementation of sediment P control measures, the dynamic and interconnected nature of lake ecosystems means that other management measures may be required to ensure that internal cycling of P from bed sediments is reduced. It should be remembered that, typically, lakes face several interacting stressors which contribute to poor water quality outcomes, and that often long-term histories of nutrient enrichment will have transformed other properties of the lake ecosystem into states which may entrench the poor water quality, such as altered fish and macrophyte communities. These can be managed via:

- Biomanipulation of fish communities can involve the removal of planktivorous species, which can supress zooplankton grazing controls on algal growth that can ultimately fuel deoxygenation, or removal of benthic feeding fish species which disturb sediments and can enhance P release into the water column.
- Improvements to littoral communities. Extensive beds of macrophyte species, can play an important role in mitigating internal P release by stabilising sediments to prevent resuspension, competing with phytoplankton for nutrients, and via the direct uptake of sediment P before it can be released. Re-establishing macrophyte beds in lakes where they have been lost due to historic eutrophication may necessitate additional controls on factors such as wind energy and grazing by waterfowl.

2. Literature Review Method

Firstly, a consultation of lake restoration literature and stakeholder discussions with regulatory agencies determined the need for a systematic review on in-lake P mitigation measures, given the minimal experience and investigation into their use and applicability in the Windermere catchment lakes. However, as internal loading remains unquantified for many lakes in the catchment, as well as an absence of nutrient budgets which include sediments (see "Reviews into evidence for sediment enrichment" report), a broader scoping review was selected to determine the successes, challenges, and the suitability of each technique in protected areas, as well as their continued viability under future climate change.

Web of Science searches.

Two recent reviews (Lürling et al. 2020; May et al. 2024) highlighted the four main approaches to internal loading control listed in the previous section. Given the limited time available to the project, four separate searches, one for each in-lake method of internal P remediation, were entered into the search engine Web of Science on 26/03/2024. All searches included the following terms to encompass the **Where** category or in other words the ecosystem of interest:

> "Lake" OR "loch" OR "standing water" OR "reservoir" OR "lough" OR "pond" OR "wetland" OR "still water" OR "mere"

The terms for the **What** category was then modified to search for the internal P load mitigation measures of interest. These were as follows:

1) "hypolimnetic withdrawal" AND "internal nutrient loading" = returned 17 publications.
2) "sediment excavation" OR "dredging" AND "internal nutrient loading" = returned 50 publications.
3) "aeration" OR "oxygenation" AND "internal nutrient loading" = returned 58 publications.
4) "sediment oxidation" OR "chemical inactivation" OR "P bind" OR "Phoslock" AND "internal nutrient loading" = returned 81 publications.

These search terms were selected to constrain returned publications to those which focussed on internal nutrient remediation.

Screening of titles followed by abstracts was then undertaken, with publications removed if they did not contain any of the search terms in the title/abstract or it was clear the study was not undertaken at the lake-scale. This reduced the publications for the full review to the following:

1) “hypolimnetic withdrawal” AND “internal nutrient loading” = title/abstract screening returned 11 publications for full review.
2) “sediment excavation” OR “dredging” AND “internal nutrient loading” = title/abstract screening returned 20 publications for full review.
3) “aeration” OR “oxygenation” AND “internal nutrient loading” = title/abstract screening returned 21 publications for full review.
4) “sediment oxidation” OR “chemical inactivation” OR “P bind” OR “Phoslock” AND “internal nutrient loading” = title/abstract screening returned 21 publications for full review.

An even weight of publications was returned for all searches except hypolimnetic withdrawal which had the lowest number of publications to review (11). Therefore, consideration that this measure will have a lower strength of evidence compared to the others must be noted.

All publications were then reviewed in full, and information extracted which could help address the following:

1) the success of the measure as determined by water quality improvements.
2) the challenges involved in not only the implementation but post-treatment.
3) the strength of evidence for each measure i.e. had studies been conducted at the field-scale and how long had monitoring pre- and post-treatment taken place.
4) The applicability of the technique in a protected area.
5) Is it future proof in the context of climate change?

3. Review of in-lake measures for internal P control

3.1 Hypolimnetic withdrawal

Hypolimnetic withdrawal is the process of withdrawing bottom waters often rich in nutrients such as P and other reduced substances including Fe and manganese (Mn) (Nürnberg, 2007). This approach requires a lake to be strongly stratified but have a low retention time or have a source of clean water to replenish the hypolimnion to reduce water column instability and de-stratification (Nürnberg, 2007). The timing of withdrawal should be based on understanding of a lake's stratification phenology with preferential operation in late-summer and autumn or whenever stratification is at its strongest and nutrients and other substances have accumulated in the hypolimnion (Nürnberg, 2007). Finally, internal P loads must be the biggest contributor to the lake P budget for this measure to be successful. This requires a thorough understanding of the nutrient budget for the lake, specifically addressing external inputs. The costs of this measure are considered low, for instance estimates range from ~ \$10 k to \$400k per treatment (Hickey and Gibbs, 2009), with costs at the upper range reported for operations in US lakes e.g., \$420 k at Lake Ballinger (42 ha) and \$310 k at Devil's Lake (151 ha) (Cooke et al.,2005; Bormans et al., 2016). This cost increases dramatically when withdrawn water is also treated to remove nutrients (Zamparas and Zacharias, 2014).

Evidence for success

One early report of hypolimnetic withdrawal was established at Lake Kortowskie, Poland in 1956 and continues to this day (Dunalska, et al., 2007). Reductions of both P and N concentrations in bottom waters during operation led to reductions in nutrients in the upper sediments of the lake, indicating success of this measure (Dunalska, et al., 2007). Remediation of hypolimnetic oxygen conditions has been less evident, although some reduction in the extent and longevity of anoxia has been documented (Dunalska, et al., 2007).

Successful remediation of hypolimnetic conditions have been documented in North America and Europe (e.g. Nürnberg et al., 1987; Dunalska et al., 2007; Keto et al., 2004; Cooke et al., 2005; Sherman et al., 2001). In a review of 48 North American and European lakes, Nürnberg (2007) found a strong correlation between reductions in summer P concentrations and volume of water removed. These lakes typically had lower retention times, enabling more water to be removed without disturbing stratification, and were typically small (<44 ha) and deep (Nürnberg, 2007). In addition, the lakes which showed the greatest reductions in epilimnetic TP, had the highest concentrations to start with but, for most lakes studied, initial reductions in P tapered off after prolonged operation (Nürnberg, 2007). Evidence for reductions in cyanobacteria blooms were also noted in these reviewed lakes (Nürnberg, 2007).

At Chain Lake, British Columbia, Canada, a small (46 ha), shallow (maximum depth = 9m), eutrophic, polymictic lake with a long retention time (0.5-3 years), hypolimnetic withdrawal was established to address internal loading which accounted for 87% of summer TP concentrations (Macdonald et al., 2004). Water was withdrawn from the deepest depth and at the sediment-water interface at different frequencies following the stratification pattern of the lake (Macdonald et al., 2004). It was estimated that, over 2 years, withdrawal removed 6 times the TP that would have been removed by natural flushing or non-modified lake conditions (Macdonald et al., 2004). However, after the first year of operation, a delayed algal bloom did occur and in the following years no or little improvement in Secchi depth or chlorophyll-*a* concentration was reported. In response to this, recommendations to initiate withdrawal earlier in the year to reduce accumulation and risk of entrainment were suggested (Macdonald et al., 2004).

At Devil's Lake, Wisconsin, USA, an augmented inflow was established to regulate the water level following hypolimnetic withdrawal, resulting in both water quality improvements and flood mitigation (Nürnberg, 2020).

At the man-made reservoir, Ford Lake, Michigan, USA, hypolimnetic withdrawal was used to de-stratify the water column and enhance vertical mixing in this polymictic system (Lehman, 2014). Withdrawal was triggered by declining hypolimnetic DO and when water was available to replenish the discharge (Lehman, 2014). In the four years of operation, suppressed cyanobacterial growth was attributed to improved oxic conditions in the hypolimnion and associated reduction of internal P loading and denitrification (Lehman, 2011).

Hypolimnetic withdrawal at Pławniowice Reservoir, Poland consisted of three pipes removing water from this hyper-eutrophic site (Kostecki & Suschka, 2013). After the first 8 years of operation, P export from hypolimnetic withdrawal was 2.5-3 times greater than P input, hypolimnetic anoxia declined in duration from ~225 days to 77 days, and charophytes returned to the once hydrogen sulphide-rich hypolimnion (Kostecki and Suschka, 2013; Kostecki, 2014).

Evidence of declining efficacy of hypolimnetic withdrawal over time as water quality improves has been recognised in sites such as the Pławniowice Reservoir, Poland (Kostecki and Suschka, 2013) and Piburger See, Austria (Roland Psenner, personnel communication) (Nürnberg, 2020). This can be used as evidence of the success of the treatment. Specifically, decreased efficiency is due to the reduction of the sediment P pool as hypolimnetic waters which are rich in mobile P released from the sediments are removed, this reduces the hypolimnetic and so, sediment P pool over time (Nürnberg, 2020). External load control is key to this success, as is the confidence that internal loads are the key contributor to water quality (Nürnberg, 2020).

Hypolimnetic withdrawal has also been judged a success in terms of reduced cyanobacterial dominance or blooms in lakes. For example, the disappearance of *Planktothrix rubescens* (de Candolle ex Gomont) Anagnostidis et Komárek, 1988 at

Lake Mauensee, Switzerland followed implementation of a withdrawal scheme (Gächter, 1976; Vrhovsek et al., 1985).

Challenges

The biggest challenge to be addressed when adopting this measure is the handling, treatment, or disposal of the withdrawn hypolimnetic water. Without careful consideration, the release of this water often results in nutrient issues being re-located downstream (Nürnberg, 2007; Jilbert et al., 2020). Indeed, the high costs of treatment often result in discharges of nutrient-rich, oxygen-depleted water and reduced compounds such as hydrogen sulphide, resulting in water quality and odour problems elsewhere (Nürnberg, 2007). For instance, in Chain Lake, British Columbia, depleted oxygen, elevated P, Fe, Mn and ammonium (NH_4) was recorded 500m downstream of the discharge point of withdrawn water (Macdonald et al., 2004). In addition, catchment land-use including beaver dams, cattle-grazing and logging further exacerbated elevated TP and anoxic conditions at the downstream site (Macdonald et al., 2004). The withdrawn effluent therefore exacerbated established water quality issues downstream. This demonstrates the need for holistic catchment nutrient management and mitigation design. Interestingly, the stable temperatures downstream of the discharge point from Chain Lake were suggested to have improved fish habitat (Macdonald et al., 2004). In another case, no effect of hypolimnetic discharge was evident in the downstream Havel River from Lake Schlachtensee, Germany. This was credited to the relatively low volume of this input (Schauser and Chorus, 2007). Hypolimnetic water treatment systems have been tested including the use of filters and precipitants such as zeolite and calcium hydroxide to enhance the removal of P from the extruded water (Silvonen et al., 2022). However, these treatments can be influenced by the chemistry of the hypolimnetic water as well as microbiological and phytoplankton communities which can block filters and reduce the efficiency of precipitants (Silvonen et al., 2022). Designing treatment of hypolimnetic water therefore requires lake-specific development and consideration.

Disentangling the direct effects of hypolimnetic withdrawal from other management measures can be difficult at the whole lake scale. For example, water quality improvements were recorded at Lake Verese, Italy, after 2 years of hypolimnetic withdrawal from three points in its main basin, resulting in reduced anoxia and algal bloom proliferation (Premazzi et al., 2005, Nürnberg, 2020). However, after 4 years of treatment, and despite an additional reduction of external loading of 70%, water column TP and P concentrations collected at multiple depths, did not show any improvement (Zaccara et al., 2007) and issues with eutrophication and odour at the discharge outlet resulted in the cessation of this operation (Crosa et al., 2013). Initial success was also suggested to be in part due to concomitant external load abatement (Premazzi et al., 2005, Nürnberg, 2020).

Hypolimnetic withdrawal may require the implementation of additional measures to complement any achieved reductions of P concentrations, particularly due to the complexity of tackling nuisance phytoplankton and cyanobacterial blooms. For example, at Reither See, Austria, Phoslock© treatment and planting of macroalgae was required to tackle the floating benthic cyanobacterial mats which still developed despite internal loading reductions of P following hypolimnetic withdrawal (Nürnberg, 2020). It was hypothesised that the cyanobacteria were able to grow by utilising interstitial nutrients prior to withdrawal and stratification, before floating to the surface in summer (Nürnberg, 2020). Similarly, at Ford Lake, Michigan, USA, hypolimnetic withdrawal eliminated *Aphanizomenon flos-aquae* blooms in early summer but resulted in higher nitrate- nitrogen (NO_3-N) concentrations, an extended diatom bloom and no decrease in *Microcystis aeruginosa* biovolume (Lehman et al., 2009).

Given the potential ecological impacts of this intervention, careful consideration of treatment design is essential. For instance, the engineering design of the pipes should consider the potential for fish to become trapped in these structures (Nürnberg, 2007). In addition, pipes require maintenance after deployment as issues with biofouling may occur (Nürnberg, 2007). The influence of water level change on littoral habitats must also be considered, as extreme fluctuations can erode littoral zones and expose macrophytes to desiccation (Nürnberg, 1987).

An understanding of the long-term dynamics associated with lake P budgets is necessary to mitigate the risk that ongoing and new operations would be unsuccessful. For example, in Lake Schlachtensee, Germany, hypolimnetic withdrawal operated each year over a few weeks, from the 1980s, but a recent re-evaluation of the lake P budget found that the treatment was having a minimal impact on P concentrations, when considering both internal and external P loads (Schauser and Chorus, 2009). Here, external P load reductions were deemed responsible for improvements in water quality, with withdrawal offering additional support by removing internal hypolimnetic P. However, over time, this P source became less important to the lake's P budget compared to external inputs (Schauser and Chorus, 2007).

What is the strength of evidence for measures – has it been applied at field/ lake scale?

Out of the 10 publications returned in this review, all were based on field-scale studies where the measure had taken place. There was, however, one feasibility study and a modelling study which involved collected field data to ground-truth and drive the model (see section 3.1.5, Dresti et al., 2023). The evaluation periods post-mitigation ranged in length from 1 year to decades. The strength of evidence of the studies are greatest as longevity of monitoring both prior and post-treatment increases, for example, in Pławniowice Reservoir, Poland and often coincide with successful remediation (Kostecki and Suschka, 2013). Whilst a more thorough assessment of methods and monitoring protocols would help address the confidence in the findings, the numerous examples at field-level and agreement from many publications on the (1) importance of external load control and dominance of internal loading, (2) knowledge and monitoring of stratification phenology and (3) the issues associated with the disposal and treatment of withdrawn water, point to the significance of this evidence.

Is it appropriate in a protected area?

There are examples of hypolimnetic withdrawal from sites of natural significance including lakes in the European Alps (Piburger See and Reither See, Austria and Lago Avigliana Grande, Italy) and Devil's Lake, Devil's Lake State Park, Wisconsin, USA. At these sites, passive syphoning which requires little maintenance and typically uses gravity rather than an energy-intensive source was selected, thereby reducing disturbance (Nürnberg, 2020). In these examples, there is little consideration of the impact of hypolimnetic withdrawal on specific protected habitats or species.

Is it future proof in the context of climate change?

Drought is problematic due to associated reductions in both water level and renewal, resulting in the need to cease operations. For example, at Lake Varese, Italy, operations were ceased in 2022 due to drought. Furthermore, future climate scenarios predict prolonged drought in this region, which would reduce the future usability of this measure (Dresti et al., 2023). In addition, predictions that drought could lead to changes in stratification, nutrient delivery and algal bloom succession point to this measure becoming less efficient in the future (Dresti et al., 2023). Indeed, at Ford Lake, Michigan, in 2012, a drought led to cessation of hypolimnetic withdrawal and a return of nuisance algal blooms, increased algal biomass and a reduction of hypolimnetic oxygen (Lehman et al., 2013).

Like drought, rainfall events may alter nutrient delivery, physical functioning in lakes, and/or result in this measure becoming redundant (Lehman, 2014; Bormans et al., 2016). For instance, intense rainfall events at Ford Lake, Michigan prior to hypolimnetic withdrawal were predicted to lead to anoxia, fish kills and water quality issues from the flooding of excess water and transport of poor quality hypolimnetic water downstream (Lehman, 2014). This particular site was more at risk of flooding during heavy rainfall due to hydrodam operations and stage height maintenance, pointing to the importance of considering wider engineering and lake processes.

3.2 Sediment removal

Sediment excavation or dredging is the process of removing sediment from a lake, which can be rich in organic matter, nutrients, heavy metals, or other pollutants. The approach has greater success in lakes where internal loading is the main contributor to P inputs and where sediments have high nutrient concentrations (Van der Does et al., 1992; Jeppesen et al., 1999). Understanding vertical sediment geochemistry is essential as part of the design of the scheme, prior to removal, as this will enable the identification of the depth of sediment which needs removing e.g. to ensure the inclusion of sediment depths high in labile P (Fan et al., 2004; Bormans et al., 2016). Lakes have different sedimentation rates, with eutrophic, productive lakes often having higher sedimentation rates from primary production and/or inorganic material inputs from inflows and catchment runoff. Lakes with low sedimentation rates present greater restoration success following dredging as the build-up of material which may potentially contain or adsorb mobile nutrients is reduced (Bormans et al., 2016). The approach is used predominantly in shallow systems or shallow sections of deeper lakes. It is usually an expensive restoration measure, particularly as it includes the need to dispose of and treat the removed sediment, which requires permits and mitigation of any toxicity to surrounding terrestrial and aquatic ecosystems (Cooke et al., 2005; Bormans et al., 2016). The approach, particularly in the case of dredging, can require large-scale specialised machinery (Hickey and Gibbs, 2009). For example, at Lake Finjasjön, Sweden, dredging cost the equivalent of £7 million, whilst cheaper measures such as fish reduction (i.e. biomanipulation) and the development constructed wetlands were considered to be more successful in their remediation effects, costing £0.5 million and £0.6 million, respectively (Annadotter et al., 1999). In some cases, dredged sediments have been recycled and used as fertilisers which can to some degree offset the dredging costs (Bormans et al., 2016).

Evidence for success

Success of the approach appears variable. The small peat Lake Geerplas, in the Netherlands, was first dredged in 1989 due to its high internal load and following consideration that external load abatement alone would be insufficient to improve the water quality (Van der Does et al., 1992). However, this first attempt was followed by redistribution of sediment from an un-dredged area, due to internal wave action (Van der Does et al., 1992). A second dredging attempt in 1990-1991, which involved barriers to separate dredged and un-dredged areas, thereby preventing horizontal redistribution, was deemed successful. This underscores the importance of understanding sediment processes and lake geomorphology (Van der Does et al., 1992). However, a simultaneous reduction (in 1990) of external loading has been suggested as the real cause of improvement here (Bormans et al., 2016). This example shows that it can be difficult to directly attribute the success of restoration measures at the whole lake scale, without investment in appropriate pre- and post-intervention monitoring.

Examples of successful dredging operations typically follow mitigation of external loads. In Lake Trummen, Sweden, dredging in 1970-1971 followed a restriction in external nutrient loads which had brought about little change in condition (Bormans et al., 2016). After dredging, cyanobacteria blooms disappeared and a diverse phytoplankton community developed (Bormans et al., 2016). Similarly, at Cockshoot Broad, UK, diversion of the River Bure inflow, which was contaminated with sewage effluent, occurred prior to dredging and other in-lake measures such as fish removal (Moss et al., 1996). These combined measures resulted in a decrease of phytoplankton chlorophyll-*a* and aquatic plant recolonisation (Moss et al., 1996).

Much like Cockshoot Broad, UK, biomanipulation in urban ponds in the Netherlands to reduce fish biomass, following dredging, resulted in improved water quality (Lürling et al., 2023). The success of using both nutrient load reduction and biomanipulation in shallow systems is related to both, acting along the nutrient-turbidity plane (Scheffer et al., 1993; Lürling et al., 2023). Broadly, this works as reduced nutrients lead to reduced primary production, with removal of planktivorous fish increasing zooplankton grazing and reducing phytoplankton biomass further, thereby resulting in a shift from turbid to clear water lakes. However, biomanipulation is only feasible in small, shallow lakes (Lürling et al., 2023).

Macrophyte planting coupled with dredging has also been successful in remediating poor water quality. For instance, in 2008 in Gonghu Bay, in the north of Lake Taihu, China, cutter suction dredging removed sediment over 10 ha (Gu et al., 2016). This intervention was made following reported taste and odour issues, despite external nutrient abatement (Gu et al., 2016). Immediately following dredging, internal release of P and NH_4 occurred due to the low oxygen content of freshly deposited organic-rich sediment (Gu et al., 2016). Submerged macrophytes were found to enhance remediation success by oxygenating the sediment. However, further consideration of plant densities and the influence of respiration and decomposition are needed (Gu et al., 2016). In contrast, dredging in Bracas Lagoon, Portugal initially reduced transparency and plant growth and resulted in an initial increase in internal loading. However, over time, chlorophyll-*a* concentrations reduced, and zooplankton populations quickly recovered, with new taxa such as *Ceriodaphnia* recorded (Silva et al., 1997).

Challenges

External loads

The reduction of external loads, prior to dredging, is essential to ensure the success of this measure. Despite this, there are many examples of the failure of lake dredging projects which are due to a lack of reduction in external loadings. For example, in the man-made Lake Mustijärv, Estonia, despite dredging removing an estimated 6400 kg of P, the external load remained high, with organic matter decomposition causing severe anoxia and re-release of P into the water column (Kiani et al., 2020). Post-treatment, the TP concentration of surface waters exceeded 100 µg/l, considered as eutrophic-hypereutrophic (Nürnberg, 1996), because of the high diffusive flux of P from the sediments occurring under anoxia, which is typical in hypereutrophic systems (Nürnberg, 1984; Carter and Dzialowski, 2012).

Similarly, in Lake Brabrand, Denmark, the removal of 0.5 million m^3 of P-rich sediment addressed internal loads, but conditions in the lake remained unaltered due to high external inputs with mean inlet TP concentrations >0.2 mg P L^{-1} (Søndergaard et al., 2000).

In Wuli Lake, China, whilst soluble reactive phosphorus (SRP) values decreased following dredging, nitrate (NO_3) and NH_4 concentrations in the water column remained high (Liu et al., 2016). This was attributed to external sources which also delivered SRP to the lake. NH_4 release from the sediments was also recorded up to 8 months post-treatment. It took over three years before external load reductions resulted in a reduction in NH_4 in surface waters, however summer SRP concentrations remained high (Liu et al., 2016).

Site-specific system understanding

Holistic understanding of the system, and site-specific diagnostics, are essential before in-lake mitigation (Lürling and Mucci, 2020). For instance, changes in primary producer communities can indicate when mitigation has not been successful, often due to a failure to address external inputs. In Vajgar fishpond, Czech Republic, an $850 k dredging project led to signs of improvement in the phytoplankton community in the first year after treatment, but by the second year, a return to the pre-dredging community occurred, attributed to external loading (Pokorny and Hauser, 2002). In Lake Dongqian, China, *Cylindrospermopsis raciborskii* cell densities were nearly 30 times higher than before dredging was carried out, despite reduced TN and TP concentrations in the lake water (Li et al., 2020). It was hypothesised that background physiochemical conditions resulted in high pH, causing P release from the sediments, which altered the water column N:P ratios promoting *C. raciborskii* growth, even under sub-optimal temperatures (Li et al., 2020).

Wider lake effects of sediment removal

The disturbance of sediments can result in accumulation of particulates and other contaminants for several months after dredging, with increased turbidity reducing light availability for photosynthesis. For example, in Lake Yuehu, Wuhan, China, suction dredging of this rain fed hypereutrophic lake in 2006 removed an average of 1m of sediment from the whole lake area. This resulted in a decrease in P and chlorophyll-*a* but an increase in N, total dissolved solids, conductivity, turbidity, particulates, and heavy metals (Zhang et al., 2010). Organic matter mineralisation decreased as primary productivity decreased likely due to the change in water clarity, with a loss of the macrophyte community which had been re-established in the lake one year prior to treatment (Zhang et al., 2010). Interestingly, crustacean zooplankton increased following dredging and there was a shift in the community away from a dominance of eutrophic taxa (Zhang et al., 2010). Similarly, dredging of an urban pond in the Netherlands impacted macrophyte development. Specifically, macrophyte colonisation depth was restricted to 4m, as insufficient light reached the lakebed (Lürling et al., 2023).

In addition, this method is extremely disturbing to benthic fauna, significantly reducing diversity and density, and these communities may take years to recover (Lewis et al., 2001; Cooke et al., 2005). This may be due to the physical removal and damage to their habitat, burial from resuspended sediment as well as modification to hydrology and morphology (Wenger et al., 2017; Wilber and Clarke, 2001; Ellery and McCarthy, 1998; Kibuye et al., 2021).

Sediment removal technique

The method of dredging is another important consideration during the design phase of a scheme and can have consequences for the magnitude of impact. For example, different dredging methods were used in schemes in Wuli Lake and Xuanwu Lake, China. At Xuanwu Lake, the method most frequently used in China was selected. This involves the drying and hydraulic purging of the lake, followed by a water blast gun to concentrate the sediment, which is then syphoned by a pump (Fan et al., 2004). Issues with fine silt and turbidity in the water column after the treatment were apparent, likely caused by the indiscriminate disturbance to the lakebed (Fan et al., 2004). Cutterhead suction was used at Wuli Lake; a targeted method which involves the loosening of the sediment by a cutter head before syphoning off by a dredge ship (Fan et al., 2004). This approach causes fewer issues with silt disturbance. Both lakes, however, showed only short-term improvements in water column SRP, before concentrations increased again (Fan et al., 2004; Liu et al., 2016).

What is the strength of evidence for measures – has it been applied at field/ lake scale?

Most studies returned by the search were from field-based applications of the technique. A variety of approaches were used to assess the effectiveness of the measure, including monitoring limnological variables and use of sediment cores to monitor post-dredging conditions of the sediments. The length of evaluation for effectiveness before and after dredging varied from months to over a decade. One recurring finding from many of the studies was a lack of monitoring and understanding of nutrient-mass balances in the lakes prior to dredging (Lürling and Mucci, 2020), perhaps explaining why many of the publications note that the measure did not remediate nutrients as planned. In a report published by the Broads Authority (Phillips et al., 2015), many of the sites which had been dredged had little sediment monitoring following removal. For the sites where this had been undertaken, the reductions in sediment P content where short-lived (<3 years), and for sites which had water column nutrient monitoring, only a few showed evidence of stable reduced P content, with many returning to high nutrient conditions after 5 years with authors citing high levels of confidence in these findings (Phillips et al., 2015). The sites which showed the most successful P remediation following dredging were isolated (i.e. Cockshoot Broad (Moss et al., 1996)) and had lower nutrients prior to dredging (Phillips et al., 2015).

Is it appropriate in a protected area?

The approach has been applied in protected sites. Mogan Lake, Turkey, a Ramsar wetland and Special Environment Protected Area underwent dredging following fish kills, anoxia, algal blooms and extremely high NH_4 concentrations (Topçu and Atlığ, 2023). No consideration for the method of dredging, sediment disposal or impacts on the lake and its surrounding catchment is mentioned by Topçu and Atlığ, (2023). However, a reduction of NH_4 and NO_3 concentrations and of organic matter mineralisation due to sediment-water aerobic conditions maintained by plants was noted post-treatment (Topçu and Atlığ, 2023).

Dredging has also been frequently applied in the Norfolk Broads National Park, as illustrated above. However, it should be noted that the water bodies of the Broads are ecologically quite different to those of the Lake District. The effectiveness of dredging in the Broads was typically lake specific and dependent on the engineering approach, though attributing changes to sediment removal were difficult owing to the intensive multiple management interventions and environmental changes occurring at the same time (Phillips et al., 2015). It was noted that where sediment assessments took place to determine the optimum depth of sediment to be removed, as well as engineering adaptations to minimise damage to littoral margins and other fragile zones such as peat basins, reductions in sediment release potential were evident (Phillips et al., 2015).

Whilst not returned in any publications within this review, lessons from dredging in a culturally important location can be gleaned from the £9 million dredging project at Blenheim Palace, UK, a UNESCO heritage site. Here, 300,000 m^3 of sediment was dredged from Queen Pool, with sediment disposed of on the estate (Personal communication, Blenheim Palace, 2022). Translocation of fish and water voles from the site was carried out prior to the dredging works, with impacts on wildfowl thought to be minimised due to the staged approach of this measure (Personal communication, Blenheim Palace, 2022). The success and impacts from this remediation has yet to be reported.

Is it future proof in the context of climate change?

Increasing temperatures could result in stronger and prolonged stratification and anoxia, as well as enhanced remineralisation processes in shallow systems which, in turn, could increase the risk of internal loading even in dredged systems (Chen et al., 2021). However, Lürling et al., (2023) found that dredged urban ponds subject to external nutrient load reduction showed greater resilience to heatwaves as sediments with lower nutrients or non-mobile fractions, had reduced capacity to undergo internal loading under prolonged stratification.

In addition to the likely implications of changing temperatures, unstable hydrological conditions from rainfall events may increase the risk of floods or internal waves which result in re-distribution and disturbance of sediments (Chen et al., 2021). This has implications for sites which have just been dredged, with weather conditions delaying recovery from sediment disturbance and internal loading. Sites with longer eutrophication histories, or where external loads have not been adequately addressed, are more vulnerable to these climate-related risks (Chen et al., 2021).

3.3 Hypolimnetic oxygenation/aeration

Oxygenation of the hypolimnion differs from the artificial aeration of the whole lake, in that it is likely to require more specialised machinery and be more costly (Cooke et al., 2005). The use of pure oxygen instead of air is also more efficient but more costly (Hickey and Gibbs, 2009). Whole-lake aeration focuses on decreasing lake stability and inducing mixing, whereas hypolimnetic oxygenation/aeration aims to prevent destratification. In both cases, the focus is on reducing internal loading before entrainment of nutrients into the epilimnion occurs, when the lake eventually mixes. The design of the system is very important to the likelihood of success (Cooke et al., 2005; Hickey and Gibbs, 2009). Since the approach treats the symptoms of internal loading, rather than the source, this treatment is often required over the long-term.

For example, in Lake Durowskie, Poland, reductions in internal nutrient loading took four years before improvements in water quality were evident (Kowalczewska-Madura et al., 2018). This treatment has been used on both deep (maximum depth >20 m) (Preece et al., 2019) and shallow lakes (maximum depth= 9-10m) (Gerling et al., 2014; Toffolon et al., 2013).

Evidence for success

In Amisk Lake, Canada, pure oxygen injected into the hypolimnion from 1988-1993, resulted in substantial decreases in epilimnetic P, NH_4 and chlorophyll-*a* because of reductions in previously high internal nutrient loads rather than external load (Prepas and Burke, 1997). In the small, deep Lake Długie, Poland, the implementation of hypolimnetic aeration reduced internal P loading, following reductions in external wastewater P sources (Gawrońska et al., 2003). Interestingly, this system used bubbles to destabilise the water column and encourage mixing (Grochowska and Gawronska, 2004). The subsequent aeration caused increased temperatures in the hypolimnion, which likely encouraged the mineralisation of organic fractions and encouraged denitrification resulting in N_2 loss from the lake to the atmosphere (Gawrońska et al., 2003; Grochowska and Gawronska, 2004).

One of the first sites to use hypolimnetic oxygenation was Camanche Reservoir, California, USA which suffered from anoxia and accumulation of nutrients such as NH_4 and P (Horne and Beutel, 2009). In 1987-1989, hypolimnetic water supplied to a fish hatchery downstream was responsible for large-scale fish kills (Horne and Beutel, 2009). Once the oxygenation system was installed, it took only 4 weeks for reductions in TP and NH_4 to occur, with an 83% decrease in chlorophyll-*a* in the spring bloom (Horne and Beutel, 2009). However, *Daphnia* populations also fell by 63%, likely due to the decline in food availability following treatment, though no wider impacts were noted on other zooplankton grazers (Horne and Beutel, 2009). There were two occasions when the system was not in use, and this resulted in a return to sediment anoxia and increased bioavailable P in the water column. This emphasised the need for prolonged use of this measure. In addition, Horne and Beutel (2009) note the unexpected effect that nitrification of NH_4 in the hypolimnion resulted in low and not high NO_3 concentrations. They hypothesised that bacterial denitrification at the warm (13–15°C) sediment–water interface may have caused this response. This unanticipated effect did not result in negative ecological conditions and was believed to have reduced cyanobacterial dominance in the lake. However, it emphasises that this treatment can modify sediment-water geochemical processes beyond those targeted and careful consideration of these secondary effects should be carried out during the design of the scheme (Horne and Beutel, 2009).

Challenges

Impacts on effectiveness of the measure

As noted for other in-lake measures, the success of this mitigation strategy is often reliant on the abatement of external loads. In Lake Tegel, Germany, several years of hypolimnetic aeration resulted in limited remediation of P concentrations and phytoplankton biomass (Schauser and Chorus, 2007). Lindenschmidt and Hamblin (1997) suggested that aeration may have been destabilising the water column in Lake Tegel and entraining P into the epilimnion. The largest reduction in P in Lake Tegel followed external nutrient mitigation (Schauser and Chorus, 2007). At Lake Sembach, Germany, organic matter sedimentation rates remained high so, despite high oxygen levels, the P retention capacity of the sediments did not increase, and P concentrations remained high (Bormans et al., 2016). In Lake Dalbang, aeration was installed to mitigate the cyanobacterial blooms believed to be responsible for odour issues in the lake, and to promote mixing during the warming period (Heo and Kim, 2004). However, little reduction in external sources were undertaken prior to treatment. When little change in TP concentrations occurred, it was found that internal loading was not a major contributor to the overall lake nutrient budget. In fact, the dominant P source arose from monsoonal rains, which delivered agricultural run-off including soil-adsorbed P to the system (Kim et al., 2000). Aeration resulted in a shift in the phytoplankton community from cyanobacteria to diatoms, however, by promoting mixing which reduced algal losses as a result of sinking under stratification (Heo and Kim, 2004).

There are several examples where oxygenation and aeration did not reduce internal P loading. One potential reason for this is a lack of Fe in sediments with which to bind P (Bormans et al., 2016). At Lake Długie, Poland, despite some remediation of P concentrations, total algal biomass remained high, as P could not be bound and permanently buried in the sediments due to the lack of Fe and Mn in the waters (Grochowska and Gawronska, 2004). An Fe to P mass ratio of >15 is required (Jensen et al. 1992) to prevent P release from aerobic sediments. If the ratio is below this threshold, aeration will have no impact on the P retention of the sediments (Schauser and Chorus, 2007). In addition, sulphur (S) can interfere with P adsorption into the Fe complex and further increase the need for a higher Fe:P ratios (Gächter and Wehrli, 1998, Gächter and Müller, 2003, Christophoridis and Fytianos, 2006; Preece et al., 2019).

Oxygenation of the sediment-water interface can also stimulate oxygen demand, both biological and chemical, as aerobic decomposition of material is encouraged and metals and nutrients are oxidised, resulting in depleted oxygen following treatment (Gerling et al., 2014; Gantzer et al., 2009, Lorenzen and Fast, 1977, Moore et al., 1996). This phenomenon was noted to vary between sites in the USA. At Falling Creek Reservoir, a small, shallow reservoir with a retention time of 200-300 days (Gerling et al., 2014), continued oxygenation was recommended to reduce sediment oxygen demand (Gantzer et al., 2009). This resulted in the oxic-anoxic boundary moving from the water column into the sediments, oxidising sediment Fe and Mn so that soluble Fe and Mn did not accumulate in the water column (Gerling et al., 2014). However, in five eutrophic Swiss lakes and at Falling Creek Reservoir, Moosmann et al., (2006) and Munger et al., (2019) found that sediments could remain anoxic despite well-oxygenated waters from oxygenation/aeration and that the redox-dependent release of nutrients, Fe, and Mn, continued. Rapid hypolimnetic deoxygenation was also found at Lake Długie, Poland followed organic matter deposition in midsummer. Increasing temperatures promoted the degradation of this material in the deepest waters and sediments despite hypolimnetic oxygenation (Grochowska and Gawronska, 2004).

Other ecosystem impacts

Non-target effects of oxygenation and aeration are also an important consideration for the implementation of the technique. Evidence gathered by monitoring oxygenation and aeration shows that, even if destratification is avoided, increased hypolimnetic temperatures can occur due to the entrainment of warmer epilimnetic waters into the hypolimnion, as occurred at Lake Serraia, Italy (Toffolon et al., 2013). Increased hypolimnetic temperatures may also occur because of exothermic reactions in the sediments, due to increased DO concentrations (Toffolon et al., 2013). Similarly, in Amisk Lake, Canada, oxygen was added to the hypolimnion from 1988 to 1993 and whilst DO concentrations increased above 5 mg l^{-1} and summer TP concentrations declined by 50% of pretreatment levels, hypolimnetic temperatures rose by 3.6°C, with no thermocline erosion (Prepas and Burke, 1997). In Lake Vesijärvi, Finland, oxygenation led to slightly destabilised thermal stratification, resulting in higher temperatures in the hypolimnion, resuspension of sediments and increased organic and P mineralisation, regenerating nutrients in the waterbody (Salonen et al., 2020; Niemistö et al., 2020). Whether or not this temperature change is ecologically important was not addressed in any study returned by this review though, as suggested by Kibuye et al., (2021) it may harm aquatic life.

When planning water column aeration, it is important to consider associated impacts on microbial communities, especially the time it takes for them to adapt to aerobic conditions (Gerling et al., 2014). In addition, it is important to consider impacts on sensitive species; oxygen consumption by the sediments following aeration in Swiss lakes was found to reduce whitefish egg survival (Müller and Stadelmann, 2004). Furthermore, incomplete oxidation of sediments in Twin Lakes, Washington, resulted in a release of methylmercury into the water column, with the bubble-plume increasing the delivery of this neurotoxin into the upper layers of the lake, increasing fish exposure (Beutel et al., 2014).

System design

The design of the aeration system is also highly influential to the successful application of the measure (Moore and Christiansen, 2009). For example, partial operation, where oxygenation was applied only at certain times of the year in Newman Lake, USA, was less successful than year-round operation at reducing P internal loads. This was due to increased sediment oxygen demands following partial operation (Moore and Christiansen, 2009). In Lake Wilcox, Canada, hypolimnetic oxygenation was employed to avoid disturbance of stratification during summer, but full water column aeration was used to prolong spring and autumnal mixing

(Nürnberg et al., 2003). This approach followed a bloom of *Planktothrix rubescens* under the ice in the following winter and spring from previously undetectable conditions (Nürnberg et al., 2003).

What is the strength of evidence for measures – has it been applied at field/ lake scale?

Most studies were undertaken on the field-scale or were reviews of field-scale measures. The length of evaluation however, differed, with some noting that treatment was ongoing. Compared to one-off interventions like dredging, for example, repeated oxygenation/aeration spanning years and decades is commonplace and therefore, it is more difficult to monitor and assess the remediation potential of this technique. The different engineering approaches used in the returned studies requires more careful consideration, as each design will be suitable to different systems and sediment conditions. All studies point to the importance of system understanding in the selection of the design, recognising the importance of (1) stratification phenology, (2) internal and external P loads, (3) sediment geochemistry and (4) the ecological context of the lake.

Is it appropriate in a protected area?

The studies returned as part of this review did not reference its applicability to protected areas. However, unlike hypolimnetic withdrawal and dredging, this technique does not require disposal or treatment of retrieved waters or sediments thereby reducing the footprint of impact (Bormans et al., 2016). However, the scale of specialised equipment and machinery is dependent on the intervention design, which could result in notable disturbance to the lake and surrounding catchment, as well as the high capital costs (Hickey and Gibbs, 2009). However, Bormans et al., (2016) note in their review that most studies indicate an improvement to fish habitat and zooplankton densities following hypolimnetic oxygenation/aeration, which could be important for sites with protected fish or insectivorous and piscivorous wetland birds. For example, the expansion of oxygenated bottom waters can result in an expansion of habitat for zooplankton and fish (Bormans et al., 2016). In Lake Sempach, Switzerland, zooplankton migrated to deeper layers following aeration and were found to be more evenly distributed throughout the water column (Bürgi and Stadelmann, 2002; Preece et al., 2019). In Lake St Mary and Amisk Lake, Canada, hypolimnetic aeration increased zooplankton and macroinvertebrate abundance, resulting in an increase in fish profiting from increased prey densities. In the case of Lake St Mary, this effect may have been further facilitated by increased visibility at depth (Rieberger, 1992; Webb et al., 1997). Müller and Stadelmann, (2004) found that improved oxygenation in the hypolimnion increased invertebrate populations and the productivity of the fisheries, even when eutrophic conditions, risk of algal blooms and high epilimnetic pH remained. However, whilst vertical expansion of foraging habitat and additional food resources in oxygenated lakes provide fish better access to prey items, this may not always result in an improvement in fish condition (Cross et al., 2017).

Is it future proof in the context of climate change?

Changes to thermal stratification and retention from rain events and warming could make it difficult to predict mixing and thermal regimes, which may impact the effectiveness of oxygenation and aeration interventions which seek to preserve stratification. For instance, at Falling Creek Reservoir, USA, breakdown of stratification occurred following several large rain events, thereby preventing aeration from being operational (Gerling et al., 2004). In addition, warmer waters cannot hold as much oxygen as cooler waters, so temperature increases may reduce the effectiveness of this measure, and therefore increase the timespan over which treatment is required. The tendency of aeration/oxygenation systems to increase hypolimnetic water temperatures, as noted above, suggests that this technique is unlikely to negate climate impacts on deep waters.

Aeration/oxygenation may also increase the susceptibility of lake ecosystems to mixing caused by weather conditions, due to weakening of stratification. In Lake Carmi, USA, aeration led to a transient stratification regime, and increased sediment oxygen demands in warmer bottom habitats, despite increased DO levels (Kirol et al., 2024). It was suggested that oscillating redox conditions can lead to higher P release compared to sustained reducing conditions (Wilkes, 2019). This resulted in higher summer surface TP concentrations during aeration than those before, due to summer wind-driven mixing events which the lake was now more responsive to due to its weakened stratification (Kirol et al., 2024).

3.4 Chemical treatment of sediments

A whole range of materials and approaches have been suggested and implemented to reduce or completely eliminate P release from sediments and remove P from the water column (Spears et al., 2013; Zamparas and Zacarias 2014; Douglas et al., 2016). Two broad approaches are available in this context, sediment oxidation, where alternative electron acceptors are provided in the surface sediments to prevent P release from Mn and Fe reduction, and chemical inactivation, where materials are added to potentially both flocculate and bind P within sediments (Lürling et al., 2020). The former approach has been used for many decades and generally relies on the addition of compounds that will enhance the provision of electron acceptors at the sediment surface. These are frequently in the form of compounds of NO_3 or Fe, the availability if which then exceeds the microbial oxidation demand and prevents redox potential changes that cause P bound within ferric iron precipitates, Fe(III) to be released due to Fe dissolution (Heinrich et al., 2021). Chemical inactivation approaches to sediment P binding have rapidly grown in the range of compounds available and modes of application over the last decade (Zamparas and Zacarias 2014; Lürling et al., 2020). The approach utilises the propensity of the phosphate anion to readily form a chemical salt by binding with aluminium (Al), calcium (Ca), Fe or other metals (Lürling et al., 2020). These compounds have varying levels of solubility and stability depending on environmental conditions such as pH and redox potential. Through the exploitation of these chemical properties, the identification of materials with more efficient binding capacities and more stable structures under varying environmental conditions has driven the potential for improved P retention in sediments and other co-benefits, such as retention of N, improved oxygen conditions and macrophyte seeding. However, while many materials can demonstrate promise under controlled laboratory conditions, relatively few products have been used at the whole lake scale. Four products for chemical inactivation are the most widely used in the context of eutrophic lake restoration: alum, calcite, lanthanum modified clay (Phoslock™) and modified zeolite (Z2G1, Aqual-P), although even within these products evidence from long-term assessments at lake scale are limited (Zamparas and Zacarias 2014).

Evidence for success

Various lake-scale applications of chemical sediment treatment have been reported in the literature. Early examples of sediment oxidation treatment through the addition of NO_3 in White Lough, Northern Ireland or a combination of calcium NO_3, Fe chloride and slaked lime in Lake Lillesjön, Sweden, proved effective at suppressing P release from sediments over the short- (White Lough) and longer term (Lake Lillesjön) (Foy, 1986; Ripl, 1986). However, a lake scale application of calcium NO_3 at Long Lake, USA did not improve water quality as expected, although this was largely attributed to insufficient control of external nutrient inputs and potential sediment composition issues (Noonan, 1986). Concerns around wider non-target effects such as the increase in N concentrations, promoting eutrophication and potential toxicity effects of NO_3 on freshwater biota (e.g. Carmago and Alonso, 2006) are likely to have limited the wider application of this approach. Although alternative, slow release NO_3 compounds such as DEPOX® have been developed to reduce these toxic effects and has shown short-term effectiveness during mesocosm experiments (Wauer et al., 2005). More recently, NO_3 based oxidation has been superseded by the use of clays modified to include nano oxygen bubbles, which release oxygen into the overlying sediment and enhance oxygen concentrations at the sediment surface. This approach has proved effective in experimental systems (Zhang et al., 2018; Wei et al., 2022) but evidence for its use at whole lake scale is currently lacking (Waters et al., 2022).

Despite the huge number of chemical inactivation materials that have been developed, relatively few have been applied extensively at lake-scale. Perhaps the most widely used material in this context is alum, aluminium sulphate, which has been widely applied in north America and some parts of Europe. The effectiveness of lake-scale applications of alum have also been assessed in an extensive review of 114 lakes by Huser et al., (2016). Their meta-analysis found that the applications improved water quality by 21 years on average in deeper stratified lakes, but only 5.7 years in shallow polymictic lakes (range 0 – 45 years). The variation in response of lakes to the treatment was attributed to Aluminium (Al) dose, relative to sediment P content, the morphology of the lake basin and the catchment area to lake area ratio, likely linked to the lake residence time and importance of internal vs external P load.

Another widely used material for the inactivation of P within sediment are different salts of Fe. Since Fe bound P is frequently an important component of sediment P composition, the addition of Fe can enhance the potential binding capacity of a lake's sediments. In a meta-analysis of Fe addition, Bakker et al,. (2015) found that P retention in sediments varied depending which Fe salts were used (e.g. $FeCl_2$, $FeCl_3$, $FeSO_4$, Fe_2O_3), while additions occurring in the water column or in the sediment were both effective at retaining P. Sediment pore water Fe:P was an important determinant of the effectiveness of the measure (Bakker et al,. 2015). Since Fe binding capacity is also redox sensitive, oxygen conditions at the sediment-water interface can impact on short-term P release, however where sufficient Fe is available (sediment molar ratio >7), P retention can be maintained in the longer term (Hupfer and Lewandowski, 2008). This sediment molar ratio refers to the ratio of the amount of Fe to P in the sediment, where Fe content is increased above a ratio value of 7 it should result in long term retention of P within sediment.

A meta-analysis of 18 lakes treated with lanthanum modified bentonite (LMB) clay (Phoslock™), revealed that on average, water quality improved in the two years post-treatment, when compared to two years pre-treatment (Spears et al., 2016). Median annual TP concentrations decreased, while small changes in chlorophyll *a* concentrations and Secchi depth also occurred and improvements to the aquatic macrophyte community were identified. The authors also noted that responses were very site-specific and inconsistency in monitoring data made longer term analyses of effectiveness challenging. Other whole-lake LMB applications have produced mixed results, particularly when the target effect is a reduction in cyanobacteria. An application to a tropical hyper-eutrophic reservoir in Brazil noted improved non-cyanobacteria phytoplankton species richness, following multiple dosing of the system, although P concentrations were still too high to sustain the improvements over the longer term (Barçante et al., 2020). Similarly in Loch Flemington, Scotland, the continuation of cyanobacteria blooms, at lower concentrations, after a single LMB application was attributed to drier, warmer weather conditions in post-application years that would favour P persistence in the lake (Lang et al., 2016).

Innovation in the area of chemical treatment of sediments is occurring through the combination of different chemical and sometimes non-chemical measures to address water quality issues. Lürling et al., (2020b) conceptualise four different approaches to in-lake treatment, depending on the lake type: 1) 'Floc and Sink' to coagulate and sink cyanobacteria blooms from the water column, 2) 'Floc and Lock' to coagulate cyanobacteria blooms from the water column and bind P within sediments for stratifying lakes, 3) 'Lock' to bind P within sediments of shallow lakes with summer cyanobacteria blooms and 4) 'Kill, Floc and Sink/Lock' for shallow lakes with perennial cyanobacteria blooms. The authors identify a range of potential coagulants such as alum, poly aluminium chloride (PAC) and iron III chloride, along with a range of organic polymers and synthetic organic coagulants and ballast materials that will bind P within sediment, e.g. local soils, modified clay, bauxite, gravels (Lürling et al., 2020b). Selection of material should be based on safety, cost, availability, and efficacy, including adequate testing of materials at different scales to identify the most cost-effective measures and potential site-specific effects on binding capacity, material stability, toxicity, direct and indirect effects on biota, and adequate understanding of the lake system (Lürling et al., 2020b). The measures were considered potentially most beneficial for smaller water bodies and less suited to large lakes with relatively high external loads. Two whole-lake examples of the approach in the Netherlands, used PAC and LMB in Lake Raubrauken and Fe II chloride in Lake De Kuil. Both sites saw improved water quality and a reduction in cyanobacteria blooms for a number of years (Waajen et al., 2016), however repeated dosing over time is likely to be necessary to counter the effects of continued external inputs (Lürling and Mucci, 2020). Other combinations of measures have largely only been trialled at lab or mesocosm scale, however they could show potential for wider water quality benefits. For example, zeolites (aluminosillicate minerals) have been modified with aluminium salts, LMB or combined in application with PAC or LMB to improve P uptake capacity and provide binding capacity for both N and P in core incubation studies and enclosure experiments (Zamparas and Zacharias 2014; Yang et al., 2021; Wei et al., 2022). In addition, combining LMB applications with sediment dredging or macrophyte planting was found to be more effective at reducing sediment P release in core incubation trials in a Chinese lake (Li et al., 2023).

Challenges

Interference of other chemical species and organic matter

A frequent finding of lake-scale applications of sediment chemical treatment is the reduction in binding capacity of the materials used compared to manufacturer claims (Waajen et al., 2016) or laboratory tests. In some cases, this can be linked to lower concentrations of the active binding agent than expected, however, for a range of compounds interference from other chemical species in the water or sediment or the presence of dissolved organic matter can alter P binding capacity. Two whole lake applications of Phoslock at Hatchmere and Mere Mere, were likely to have had slower P adsorption than observed in other lakes due to the high levels of humic compounds present in the water (Spears et al,. 2018). Both Hatchmere and Mere Mere are two small and relatively shallow (<10m max. depth) lakes found in Cheshire and form part of the Midlands Meres and Mosses Ramsar Site due to their support of rare aquatic vegetation and invertebrates (Spears et al., 2018) While a meta-analysis of Phoslock applications found a positive association between P concentrations and dissolved organic carbon concentrations across sites (Spears et al., 2016). For Fe based binders, interference from humic compounds and sulphate reduces the effectiveness of internal P release due to preferential binding of Fe with these compounds (Bakker et al., 2016).

Lake water chemistry changes

Both pH and redox of receiving waters can be important controls on the effectiveness of chemical amendments designed to target P release. In addition, the introduction of compounds can themselves alter the water chemistry of the receiving water. For example, large additions of Fe or aluminium salts can result in pH reductions, potentially affecting aquatic life in the lake and the solubility of other metals without a suitable buffer also included (Zamparas and Zacharias 2014; Bakker et al., 2016). The pH required for optimum P binding effectiveness depends on the compound but is required to be as broad as possible (5-6 up to 10) for natural waters (Douglas et al., 2016). Optimum pH for LMB is reported as 6 – 10, whilst alum performance is optimal between pH 6 – 8 and floc formation is inhibited at high pH (Zamparas and Zacharias 2014; Bessa da Silva et al., 2019). High pH was also found to interfere with the P binding capacity of aluminium-modified zeolite Aqual-P™ during enclosure experiments in a Dutch pond (Kang et al., 2023). Redox state of sediments and overlying waters can also affect material binding capacity. For example, whilst LMB, carbonates, soils or soil-derivative based materials tend to be relatively insensitive to redox, most Fe based materials are likely to release bound P via Fe reduction (Douglas et al., 2016).

Applying the correct dose

A further factor that tends to limit the effectiveness of P binding materials at the lake-scale is assessment of the correct dose required to prevent P release from the sediment under the environmental conditions which occur within the lake, such as those listed above. A number of studies cite dose-based factors in the continuation of P release or a truncation in the expected time for which the treatment was effective. Huser et al., (2016) found that Al dose was one of the key variables explaining the longevity of treatment across 114 lakes, while Waajen et al., (2016) found that dose rate for LMB was critical for determining the long-term success of a treatment, as binding capacity could be exceeded over time. Dosage rates are strongly related to the active sediment depth for P release. This is often assumed to be only a relatively shallow depth of sediment (~4-5cm), while Meis et al., (2012) suggest that while estimates typically vary from 4–10cm, it may be up to 20–25cm in some lakes. The authors suggest that this depth is likely to be site-specific, affected by factors such as bioturbation, macrophyte density, pH and oxygen concentrations.

Impacts of abiotic and biotic resuspension and redistribution

Applications of sediment chemical treatments at the whole lake-scale are frequently affected by sediment mobilisation and reworking through abiotic and biotic processes. The presence of benthivorous fish species such as carp are frequently implicated in sediment bioturbation and the reduced effectiveness of P binding approaches (Huser et al., 2016; Lürling and Mucci, 2020). Whilst other bioturbation processes by sediment invertebrates could also translocate chemical amendments into deeper sediment layers and render them less effective at controlling P release (Meis et al., 2012). Abiotic processes such as wind-driven resuspension of sediments represents an important factor in the movement of sediment lakes. This is particularly the case in shallower lake systems, where a greater proportion of the bed sediments may be affected by wind-waves and currents. These resuspension processes can also redistribute chemical amendments usually designed for relatively uniform distribution across the lake, which can affect the spatial efficacy of P binding (Douglas et al., 2016). Huser et al., (2016) found that metrics associated with lake morphometry were important determinants of alum treatment longevity, which could, in part, be linked to resuspension in shallow lakes. While Lang et al, (2016) and Waajen et al., (2016) suggest sediment resuspension as a potential mechanism to explain lower effectiveness of LMB treatment on water quality and ecological responses over time.

Sufficient reductions in external loads

As is the case for all within-lake measures covered in this report, a frequent cause of chemical treatment measures being ineffective or only partially successful is the inadequate reduction in the external nutrient load prior to in-lake measures being implemented. In meta-analyses of many lakes subject to both Fe and alum applications, inadequate external load control was implicated in the effectiveness of treatment (Bakker et al., 2016; Huser et al., 2016). For LMB treatments, Lang et al., (2016) and Spears et al., (2018) suggested that reasons for the lack of expected ecological responses to treatment in the UK lakes studied were due to insufficient control of catchment nutrient sources. Whilst continued catchment loading to a Brazilian reservoir and two Dutch lakes either resulted in repeated treatments or was expected to require additional treatments to improve or maintain water quality improvements (Barçante et al., 2020; Lürling et al., 2020a). Lürling et al. 2020a advocate for a selection of lake restoration measures and materials based on a systems analysis, which includes water and nutrient budgets, whilst Zamparas and Zacharias (2014) suggest that the longevity of internal P treatment will be reduced, where management of external loads is not adequately addressed, and use of chemical methods applied as part of an integrated management plan.

Strength of evidence - has it been applied at the field/ lake scale?

Chemical treatment of sediments to prevent P release has been used at whole lake scales for decades. However, despite the plethora of products available, relatively few materials have been applied consistently at lake-scale. Alum has been widely used in north America and parts of Europe (Huser et al., 2016), while although the use of LMB is more recent, it has been applied in >200 lakes to date (Lürling et al., 2020b). Fe dosing has also been a widely adopted technique at the lake scale (Bakker et al., 2016). Based on the meta-analyses from these studies, the effectiveness of measures and longevity of effects varies greatly from lake to lake, usually as a result of site-specific factors (Spears et al., 2016). Evidence for other products and particularly newly emerging approaches are largely based on experimental studies at laboratory or mesocosm scale (Lürling et al., 2020b). The key features identified in the Challenges section above reveal the main factors that can influence the effectiveness of the measures. Studies by Douglas et al., (2016), Lürling et al., (2020a;2020b) provide a useful framework for assessment and application of sediment chemical treatments. These include 1) appropriate target setting for the restoration objectives, 2) assessment of lake characterisation and function – a ‘systems analysis’ including a nutrient budget and assessment of the ecology of the site for potential ecotoxological or ecological impacts of the measure, 3) performance testing of materials across differing time periods (days to weeks,

weeks to months, months to years), 4) assessment of properties of materials in comparison to the receiving waters, 5) assessment of ecotoxicity of the materials, 6) evaluation of materials prior to full-scale implementation, such as laboratory testing of water, intact sediment core experiments, mesocosm studies and enclosure experiments.

Is it appropriate in a protected area?

Very few studies returned by the searches specified whether sites were located in national parks or protected areas. However, three of the whole-lake applications of LMB in the UK considered in this review were carried out on sites designated as SSSI: Loch Flemington, Hatchmere and Mere Mere. When assessing the suitability of materials for sediment chemical treatment at protected sites, appropriate consideration of potential target and non-target effects, including ecotoxological impacts on lake biota should be part of any assessment of options for restoration. Consideration of these measures should be made as part of a wider assessment and systems analysis of the site and part of a cost-benefit analysis (Douglas et al., 2016). More broadly, application of these techniques will also increasingly need to consider the wider sustainability agenda of creating circular economies for limited resources such as P. Future developments in chemical treatment and P management more generally may focus more on P recovery and recycling, rather than 'locking' P within sediments (Zamparas and Zacharias, 2014).

Is it future proof in a changing climate?

Warmer water temperatures and longer and more strongly stratified lake ecosystems in future are likely to exacerbate internal P loading (Mackay et al., 2020). Both changes in the physical property of the water column may impact on factors that alter the effectiveness of some of the chemical treatments. Longer stratification is implicated in long term DO decline (Jane et al., 2023), which could prolong low redox conditions, affect the processing and release of organic matter from sediments and impact pH. In addition, warmer water temperatures are likely to enhance microbial processing of organic matter, also affecting oxygen concentrations and the remineralisation of P within sediments and the water column. More redox sensitive materials such as those based on Fe may become less effective at binding P, while very low pH can affect the toxicity of Al and other metals and very high pH can reduce binding capacity for Al and LMB. In addition to longer term impacts on lake ecosystems, greater intensity of storm events could enhance the resuspension and reworking of materials within the lake. Increased catchment nutrient loading via high runoff events may enhance external nutrient supplies and reduce the longevity of the treatment by more rapidly exceeding the binding capacity. Careful consideration of effective doses would be required to try and address some of these issues.

4. Lessons for best practice implementation of in-lake measures to manage internal P loading

The evidence review for in-lake measures to reduce or prevent internal P loading in lakes identified a range of techniques that have been implemented at the whole-lake scale over many decades. The effectiveness of these measures has varied dramatically across sites and over time. In addition, lake responses to treatment are site-specific and dependent on the technique and method of implementation. This indicates that the appropriate choice of measure(s) for in-lake P control is not uniform across all lakes and there is no ‘one size fits all’ option. Despite this system-specific requirement for the design of these measures, there are however, a number of more general lessons that can be applied in the approach to managing internal P loading in lakes. These are provided in this section and draw from the wealth of literature which advocates for the appropriate use of in-lake remediation within wider integrated management approaches (e.g. Mackay et al., 2014; Zamparas and Zacharias, 2014; Lürling et al., 2016; Douglas et al., 2016; Lürling et al., 2020a,b).

4.1 Lessons identified from the systematic review

Clearly defined restoration goals

The identification of clear targets for water quality or restoration outcomes is the key first principle in the process of lake management and restoration (Douglas et al., 2016). This allows for the success of interventions to be assessed against quantifiable criteria. It enables stakeholders to identify and agree on priorities for the lake, which may require negotiation and trade-offs to be considered. It may also provide guidance on the types of measures that could be used to achieve those goals.

Lake system characterisation and pre-intervention monitoring

Critical to the successful implementation of management measures in lake ecosystems is a comprehensive understanding of the individual lake within its catchment context, a ‘systems analysis’ (Lürling et al., 2016; Douglas et al., 2016; Lürling et al,. 2020a,b). In the context of nutrient inputs, this requires a quantification of the nutrient budget for the lake, particularly the partitioning of the relative importance of external vs internal loads. This is crucial where in-lake interventions are being considered, as it will directly impact on the suitability of the measure and the likely longevity of its affect. In-lake measures are most successful when it is clear that internal loading was the major contributor to the P loads of the lake. For example, pure oxygen injected into the hypolimnion of Amisk Lake, Canada led to reductions in epilimnetic nutrients and chlorophyll-*a* with the success of this remediation cited due to the high internal loading at this lake (Prepas and Burke, 1997). Site specific assessments, including mass balance calculations are integral to accurately assess the location, sources, timings, and metabolism of external and internal lake nutrients, and are a priority measure for catchment management (Spears et al., 2018).

The application of in-lake measures necessitate a comprehensive understanding of the physical, chemical and biological components of the lake ecosystem in order to select the most suitable measure(s) to address the nutrient issue and account for likely interferences in the effectiveness of the measure being applied. These may include the potential for physical disturbance or redistribution of P enriched sediments through wind-wave resuspension or bioturbation from large populations of benthivorous fish. The potential for chemical interference with materials used to supress P release due to redox, pH changes or the presence of high concentrations of dissolved organic matter in the water column and sediment. A need to understand the chemical composition of the sediment to identify the level of P enrichment, the active sediment depth for P release or the chemical composition of sediments that could affect the P binding capacity. The potential for fish communities to help or hinder restoration efforts due to the composition and population of piscivourous, planktivorous or benthivorous fish species. The potential for zooplankton communities to offer ‘top-down’ control on phytoplankton populations. The potential for littoral macrophyte communities to support or hinder efforts to control phytoplankton populations via competition for nutrients, providing fish habitat, reducing sediment resuspension and mediating sediment P release. In addition to this system level understanding of the effectiveness of the measure, the same ecosystem components should also be considered from the perspective of potential for non-target effects of the measure, including potential ecotoxological issues.

External load control

One of most common reasons for failure in the management of eutrophication in lakes is the insufficient reduction of catchment P loads (Sondergaard 2007). This is particularly important where internal P load control measures are being considered since it undermines the efficacy of the internal measure (Spears et al., 2016; Spears et al., 2018). Most of the studies reviewed in this report which found in-lake remediation to be unsuccessful cited lack of external load abatement as the primary reason. For example, the lack of water quality improvements in Lake Braband, Denmark and Wuli Lake, China following dredging was attributed to high external inputs (section 3.2.2). Similarly, in lakes where internal and external reductions coincided, it was often the external nutrient management that had the most pronounced effect. For example, in Lake Tegel, Germany, hypolimnetic aeration had little effect until external nutrient control was implemented (Schauser and Chorus, 2007). Therefore, focussing on external load reduction must be the priority for effective long term lake nutrient management.

Intervention selection and cost-benefit analysis

In-lake remediation is costly, although variable depending on the size of the lake, engineering design, need for repeated treatments and the disposal and treatment of wastewater or waste sediment. A cost-benefit analysis of the different options available to address the problem, including consideration of the costs of the degraded ecosystem state and the implications of a ‘do nothing’ approach (Douglas et al., 2016), will enable a prioritisation of measures and inform decision making as part of the management process. To carry this out effectively, system understanding and the application of the measure in the context of the specific lake is key. Therefore, the cost-effectiveness of monitoring, catchment management and in-lake measures also need to be considered when developing the lake management plan (Spears et al., 2018). In addition, small scale trials in laboratory or mesocosm scale settings, using water and sediment from the target lake can be used to build evidence of the effectiveness of a measure and aid the design of the measure implementation to maximise its impact (Lürling et al., 2020).

Post-intervention monitoring and adaptive management

Many lake restoration projects fail to implement effective monitoring plans to adequately capture system behaviour before and after interventions. This makes an assessment of their effectiveness or an understanding of their impact very difficult. Not only does this create a barrier to being able to directly quantify the potential for the measure to be used in future lake management plans, it also results in an inability to effectively adapt the management of the lake in response to unforeseen effects on the ecosystem. For example, on-going monitoring during and following treatment is required to adapt to changing conditions which may inhibit the delivery of the measure. A good example of this is from Chain Lake, Canada, where an alteration in the timing of hypolimnetic withdrawal to earlier in the season were suggested following continued high algal productivity despite several years of withdrawal treatment (Macdonald et al., 2004).

Additional challenges from lake responses in a changing climate

Climate change is likely to make in-lake remediation more challenging particularly for those treatments which require certain physical or physico-chemical conditions to operate, such as for hypolimnetic withdrawal, which requires predictable stratification. Extreme weather events and altered seasonal weather including droughts and heavy rainfall can lead to cessation of operations or compromised water quality. For example, hypolimnetic withdrawal operations have been ceased in Lake Verese, Italy and Ford Lake, Michigan, USA following drought due to lower water levels and reduced water renewal (see section 3.1.5), whereas heavy rainfall caused redistribution of sediments from un-dredged to dredged locations resulting in higher nutrients and turbidity at these locations (Chen et al., 2021, section 3.2.5)). In addition, warmer temperatures may alter the redox conditions and pH which can influence the effectiveness of chemical inactivation agents (see section 3.4.5).

Monitoring of lake conditions and the impact of measures is therefore important to be able to respond and adapt measures to optimise their effectiveness.

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