FUTURELAKES

For Nature, Climate and People

Innovation in Lake Restoration:

Literature Review

Deliverable D1.1

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Summary

The FutureLakes project aims to deliver innovative solutions for the protection and restoration of European lakes. This report introduces the range and potential of technical innovations available for lake restoration with a special emphasis on nature-based solutions (NbS), circular blue-economy solutions (CBS) and biodiversity-focused solutions (BfS) based on published literature. A structured literature review and complementary scoping interviews for selected scientific/practitioner communities were conducted to address the FutureLakes' specific objective of:

- Reviewing the technical efficiency and added value of innovative lake restoration solutions to protect and restore biodiversity and to reduce pollution.

In total 208 publications dealing with 78 individual types of technical water protection measures or in-lake restoration innovations were reviewed. Information was searched on the implementation, applicability, costs and impacts of preventative external solutions for improved water protection and internal, in-lake measures for enhanced lake restoration. The review included consideration of their potential contribution in relation to several policy goals (e.g., Biodiversity Net Gain, Pollution Reduction, Water Framework Directive Good Status, Climate resilience).

Most of the innovations reviewed were not entirely new but they were novel applications of existing, well-established methods, in which excess nutrients and biomass were reframed as recoverable resources, providing innovative links to circular economies. A majority (55%) of innovations were in-lake measures with most of them representing either NbS, CBS or other solutions that are innovative but do not directly fall within either of these two categories. Additionally, several measures could be labelled as multifunctional, i.e. measures combining elements from NbS, CBS or BfS. As an example, biomanipulation is a NbS that can also be considered as a BfS, if it is used to primarily benefit biodiversity in terms of, for instance, waterfowl, and as a CBS with possibilities for the circular reuse of fish catch in food and feed production. Similarly, some of the methods for nutrient recovery were labelled as CBS, but also BfS when proven to support natural functioning of a lake after intervention, for instance by enabling the recolonization of submerged macrophytes.

From the reviewed innovations, 40% dealt with external solutions for improved water protection on catchment scales. Most of the external measures were labelled as NbS with multifunctional measures also common. Noteworthily, only a minority of studies (5%) dealt with combinations of measures on both catchment and in-lake scales. A strategy combining these approaches has in many lake restoration programmes proven to provide best results.

The following approaches were concluded to provide the highest potential in transforming lake restoration by improving lake ecological status in the long term while at the same time providing multiple co-benefits:

- **Nature-based land use practices and approaches in catchment water management** to prevent nutrient loading and reduce erosion risk. These provide contribute to pollution reduction and greenhouse gas emission reductions and provide co-benefits for biodiversity
- **Nature-based and circular blue economy solutions to improve water and nutrient retention within catchment areas**, with benefits for biodiversity, flood and drought risk management, and reducing pollution
- **Nature-based and circular economy solutions for the recovery of legacy nutrients and excess biomass from lakes**, with benefits for biodiversity, greenhouse gas emission reductions, food and feed production, alternative fertilisers and materials for construction, health and well-being
- **Nature-based and biodiversity-focused solutions to restore morphological conditions and improve the structural complexity of lakes** to support their natural functioning, with co-benefits for biodiversity
- **Combined measures on both catchment and in-lake scales** to simultaneously manage external loading within tolerable boundaries and tackle multiple in-lake processes maintaining eutrophication

The report is structured to be a useful catalogue of measures for EU Member States for the development of their National Restoration Plans under the relevant articles of the EU Nature Restoration Regulation (NRR). For this purpose, the external catchment measures are considered for three landscape contexts: agricultural, forest and urban landscapes.

Many of the approaches mentioned above already have a high scalability potential, together with high technical readiness level, with efficiency proven at operational scales. A pre-requisite for them to become mainstream approaches in lake protection and restoration are supportive policies, incentives and financing schemes to support their implementation more broadly. Additionally, coherent water governance, together with comprehensive cross-sectoral stakeholder engagement, is needed to ensure wide uptake of these innovative practices and approaches.

The Deliverable underpins the complexity of successful lake restoration, the necessity for novel, multifunctional approaches, and the need to prevent lake degradation in lakes less impacted by anthropogenic activities (e.g. to prevent deterioration of lakes in good and high ecological status or favourable conservation condition). A take home message of the deliverable is that no magic pill for lake restoration exists. Achieving sustainable lake restoration, with high longevity, should initially involve a thorough lake-specific diagnosis of water and nutrient sources and stressors, together with an understanding of the biology and functions of the lake, to evaluate the most appropriate restoration measures for that individual lake context. This deliverable contributes to an integrated framework for lake protection and restoration (D4.4, FutureLakes Blueprint).

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Abbreviations

AES – Agri-Environmental Schemes
Al – Aluminium
BfS – Biodiversity-focused Solution
BGS – Bottom grid structure
C – Carbon
$CaCO_3$ – Calcium carbonate
CAP – Common Agricultural Policy
CBS – Circular Blue Economy Solution
CCF – Continuous cover forestry
Cd – Cadmium
Chl *a* – Chlorophyll *a*
COD – Chemical oxygen demand
CO_2 – Carbon dioxide
Cr – Chromium
CSO – Combined sewer overflow
Cu – Copper
CW – Constructed wetland
DAF – Dissolved air flotation
dFe – Dissolved iron
DMPP – Dimethylpyrazole phosphate
DNM – Ditch network maintenance
DWTR – Drinking water treatment residual
eFLOAT – Efficient flotation of Algae Technology
EU – European Union
FA – Fly ash
FAO – Food and Agriculture Organization of the United Nations
Fe – Iron
$FeCl_3$ – Ferric chloride
f-MB – Iron-modified benthonite
FTW – Floating Treatment Wetland
GCC – Ground calcium carbonate
GHG – Greenhouse gas
GLEAMS – Groundwater Loading Agricultural Management Systems
H_2O_2 – Hydrogen peroxide
HAB – Harmful algal bloom
HRAP – High-rate algae pond
HW – Hypolimnetic withdrawal
HWTS – Hypolimnetic Withdrawal and Treatment Systems
La – Lanthanum
LMB – Lanthanum-modified bentonite
MBR – Membrane bioreactor
MCM – Magnetic chitosan microsphere
MFS – Microbial fuel cells
MgO – Magnesium oxide
MP – Magnetic particle
N – Nitrogen
NaCl – Sodium chloride
NaOH – Sodium hydroxide
NbS – Nature-based Solution

NO_3 – Nitrate
NRR – Nature Restoration Regulation
P – Phosphorus
PAC – Polyaluminum chloride
PMB – Permeable reactive barriers
POC – Particulate organic carbon
P-PO_4 – Phosphate phosphorus
RFM – Rotation forest management
S – Sulphur
Se – Selenium
SRP – Soluble reactive phosphorus
SS – Suspended solids
TP – Total phosphorus
TRL – Technology Readiness Level
TSC – Two-Stage Channel
TSC – Two-stage channel
TSS – Total suspended solids
USD – US dollar
UWWTD – Urban Wastewater Treatment Directive
VDD – Vegetated drainage ditches
WFD – Water Framework Directive
Zn – Zinc

Disclaimer

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1 Introduction

Lake ecosystems are highly valued by citizens and many businesses due to their importance for biodiversity, recreation and tourism, the provision of clean water for drinking and irrigation, fisheries and energy production. Despite these recognised values, degradation of lake environments across Europe is severe (EEA 2024). The main pressures weakening the ecological status of lakes are diffuse sources of pollution and hydromorphological alterations, with point source pollution and abstraction also common (EEA 2024). Degraded lake systems not only impact biodiversity and ecosystem health but also weaken the ecosystem services provided by lakes, impacting the social and economic benefits we receive from them.

In European Union (EU), the Water Framework Directive (2000/60/EC, WFD) calls for member states to achieve a good ecological and chemical status of surface waters and requires that human activities do not deteriorate them further. Through the implementation of WFD and other European policies, such as Urban Wastewater Treatment Directive (UWWTD), Nitrates Directive, and the Agri-Environmental Schemes (AES) of EU's Common Agricultural Policy (CAP), thousands of measures have been undertaken across Europe to reduce pollution and restore lakes. Additionally, the newly established Nature Restoration Regulation (2024/1991/EU, NRR) represents an unprecedented opportunity for freshwater habitat restoration and, consequently, freshwater biodiversity protection (Stoffers et al. 2024). NRR will provide explicit obligations on Member States for implementing restoration measures to support and enforce reaching the favourable conservation status of the Nature Directives and to improve the quality ecosystems and habitats for people, climate and the planet.

Despite that the implementation of the UWWTD has successfully reduced external nutrient loading to lake ecosystems (e.g., Jeppesen et al. 2005), ecological improvements have, however, been less evident and the nutrient concentrations for many lakes in Europe have not declined sufficiently to achieve improvements to good ecological status (EEA 2024). One of the identified reasons for not achieving a good ecological state are measures ineffective at tackling diffuse sources of pollution and internal phosphorus loading (Carvalho et al. 2019; Poikane et al. 2024). Hence, there can be long time lags before the effects of nutrient loading reductions become apparent. The scale of protective measures in lake catchments are widely acknowledged to be insufficient, particularly to tackle nutrient loading from diffuse sources and there is a tension between "source-oriented" measures (reduction of nutrient use, less crop yield, less livestock) and "effect-oriented" measures (cleaning up the nutrients from lakes) (Wiering et al. 2020). While the implementation of source-oriented measures relies largely on individual member states, the WFD lacks integration of other policy areas, tools or sanctions to enforce effective measures to reduce agricultural diffuse nutrient pollution (Boezeman et al. 2020; Wiering et al. 2023). Therefore, most focus has been paid on effect-oriented measures for either intercepting leached diffuse-source nutrients or tackling them when already in the receiving surface waters. However, even if nutrient use in the catchment was adequately reduced, lakes will not recover rapidly because nutrients will either keep on leaching from saturated soils or are mobilised from the lake sediments. Subsequently, there is a need for more effective mitigation and lake restoration measures to deliver successful water and habitat quality improvements, enhance biodiversity and secure the ecosystem services that lakes provide to society and the economy.

FutureLakes is a Horizon Europe Framework project funded under the call for European natural lakes (HORIZON-MISS-2023-OCEAN-01-04). The project has a specific objective (SO1) to demonstrate innovative nature-based (NbS), circular blue economy (CBS) or biodiversity-focused solutions (BfS) that are effective at restoring biodiversity, reducing pollution from catchments and dealing with legacy pollutants from lake sediments. To meet this objective, we aim to discover and document the range and potential of technical restoration innovations available to support and transform lake restoration by reviewing existing literature and surveying practitioners. Here, we define innovations as measures either bringing new approaches (inventions) or developing the performance and sustainability of existing solutions. Our target was to evaluate at least 50 individual NbS, CBS and BfS restoration measures in a structured literature review, of which results are summarized in this document (D1.1, KPI 1.1).

2 Methods

Innovations were mapped by reviewing existing peer-reviewed literature and by surveying practitioners to discover complementary grey literature. The search engines Web of Science (https://apps.webofknowledge.com), Bielefeld Academic Search Engine (https://www.base-search.net/), Scopus (https://www.scopus.com), Aquatic Sciences and Fisheries Abstracts (https://www.fao.org/fishery/en/openasfa) and Google Scholar (https://scholar.google.com) were used by applying the following search string in December 2024.

- “lake restor* OR lake protect* OR lake manage* AND innovati* OR nature?base* OR circular*”

The searches were complemented with structured online interviews for selected scientific/practitioner communities on methods still at a trial stage. During the review, information on the implementation, applicability, costs, and impacts of water protection or restoration methods used were searched from all the studies included. Where applicable, i.e. enough information for evaluation was provided, impacts and additional value of measures described were assessed in relation to several policy relevant indicators (e.g., lake ecological or chemical status, biodiversity net gain, greenhouse gas emission reductions, flood and drought resilience, human health and wellbeing). Additionally, the technology readiness level (TRL) of innovations from scale TRL 1 (basic principles observed) to TRL 9 (actual system proven in operational environment) as defined in Horizon 2020 framework program were assessed based on the information provided in the publications.

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3 Innovations available for lake restoration

The searches yielded a total of 208 publications dealing with 78 individual innovations available for lake protection and restoration that were combined in applicable clusters in the following sections. Here, we describe the range and potential of innovative solutions separated into **preventative external solutions** for improved water protection at catchment scales (Section 3.1) and **restorative in-lake solutions** (Section 3.2) for either reducing the symptoms of eutrophication and/or tackling legacy nutrients. A total of 41 publications dealt with different **combinations of measures** either on internal, external, or combined scales, that are also briefly described and summarized (Section 3.3). The following symbols and definitions are used in describing the different categories of innovations:

= NbS = CBS = BfS = Other = Multifunctional

- **Nature-based Solutions (NbS):** Actions to protect, sustainably manage and restore natural or modified ecosystems, that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits (Cohen-Shacham et al. 2016).
- **Biodiversity-focused Solutions (BfS):** Solutions focused primarily on biodiversity restoration that may include established restoration measures implemented in innovative ways.
- **Circular Blue economy Solutions (CBS):** Actions that include efficient recovery of resources by integrating circular economy principles into the bioeconomy.
- **Other innovative solutions (Other):** Actions that are innovative in relation to traditional lake restoration approaches but do not directly fall within any of the above-mentioned categories.
- **Multifunctional solutions (Multifunctional):** External and/or internal solutions or their combinations for improved water protection and lake restoration that could be labelled as more than one type of innovation.

A majority of reviewed publications dealt with innovative in-lake measures with most of them representing solutions that could be considered as NbS or other innovative solutions (Figure 1). Likewise, most reviewed papers dealing with external solutions for improved water protection were categorised as NbS. Interestingly, only a fraction of reviewed publications dealt with a combination of external and internal measures, although both approaches are most often needed for successful lake restoration.

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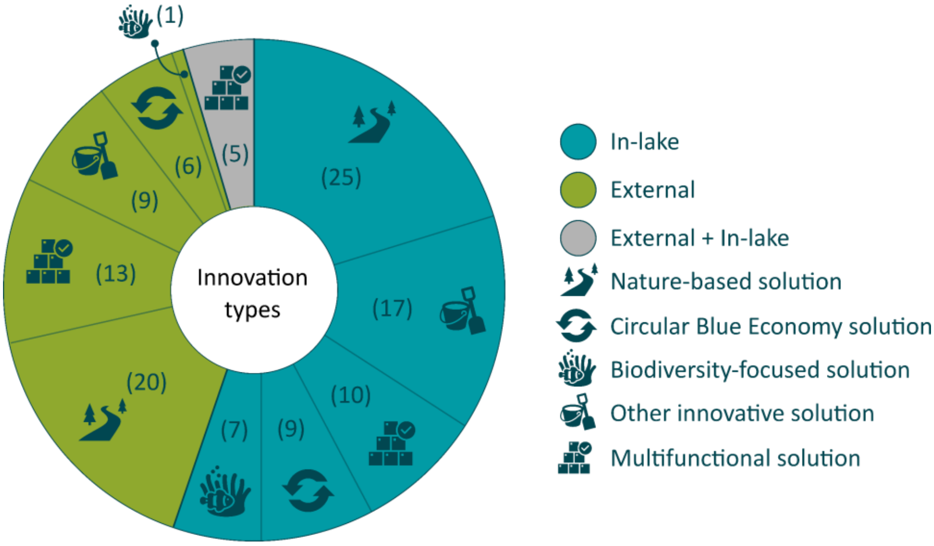

Figure 1 – Distribution of reviewed innovations to external and in-lake measures, and to different innovation types.

3.1 External solutions for improved water protection and reduced pressures

Intensive agriculture remains the largest source of nutrient loading to European surface waters in countries with advanced treatment of municipal and industrial wastewaters (Grizzetti et al. 2021). However, sources of loading naturally vary depending on the catchment characteristics and land use. For instance, forestry or urban stormwaters can represent a majority of loading to surface waters in catchment areas with limited proportion of cultivation (e.g., Finér et al. 2021; Müller et al. 2020).

Sufficient reduction of external loading is a necessity for sustainable restoration of lakes (Tammeorg et al. 2024) and thus, efficient preventative measures are needed to both prevent future loading and to trap pollutants that have already leached. In the following sections, we describe the applicability and efficiency of external measures supporting lake protection. In line with the NRR, the innovative external solutions reviewed are categorised under measures related to agricultural and forestry practices and urban habitats. Water protection structures are mainly described under the section for urban measures (3.1.3), as most of the reviewed solutions have originally been developed to improve wastewater treatment. However, it must be noted that many of these measures (e.g. constructed wetlands (3.1.3.1)) are also applicable and frequently applied in agriculture and forestry as preventative measures for improved water protection.

The applicability, benefits, possible disadvantages, costs and scalability potential of innovations available are summarized at the end of each section separately for agricultural practices (Table 1), forestry practices (Table 2) and urban habitats (Table 3).

3.1.1 Measures related to agricultural practices and agriculture-associated habitats

3.1.1.1 Catchment water management

Catchment water management has long been practiced mainly to ensure suitable soil water status for crops through field drainage, flood management, and irrigation in agriculture. Recently, more integrated actions at the catchment-scale are increasingly being adopted to respond to the harmful water quality impacts of drainage as well as to ensure sustainable water availability for agriculture in drought-prone areas. Catchment water management is typically a combination of technical and nature-based solutions, with the latter referred to as natural flood management or natural water retention measures (https://www.nwrm.eu/), aiming at improving water quality and quantity mainly in the receiving rivers and lakes, but also locally at the field scale (Nikraftar et al. 2021; Robotham et al. 2023; Zhang et al. 2011). This is achieved mostly by regulating the flow of water in the catchment by controlled drainage and taking advantage of different structures for improved water retention, but also in some cases by regulating the use of water by agriculture. Catchment water management is an important tool for nutrient load conservation practices that can be implemented at the source (avoiding strategy), during transport (controlling strategy), or at the edge of water bodies (trapping strategy) (Osmond et al. 2019).

Catchment water management in general is mainly targeted for catchments with large agricultural share of land use. In, e.g., Finland, where 25% of tree stand production originates from peatlands drained for forestry, catchment water management has received increasing attention as a tool for mitigating loading also from forested catchments. Measures for catchment water management exist for different scales being adaptable for most geographical and climatic regions. At the plot/field scale, the measures are mostly technical, such as improving irrigation efficiency or delaying/controlling the drainage (Nikraftar et al. 2021; Salla et al. 2021; Zhang et al. 2011), potentially combined with changes in cropping patterns. In addition to technical water management measures, other field-scale NbS management approaches for improved nutrient retention and load prevention are needed. Improved nutrient management by reduced and accurately targeted fertilization, utilisation of winter cover crops and perennials in the cropping system can improve discharge water quality by reducing the nutrient source, increasing plant nutrient uptake ad reducing the amount of water entering the streams and channels (Withers and Jarvis 1998; Withers and Lord 2002).

At the sub- and catchment scales, various technical and nature-based measures, such as (natural) flood management and water protections structures such as constructed wetlands (Section 3.1.3.1) can be applied (Robotham et al. 2023). Drainage for forestry can be avoided taking advantage of tree stand evaporation using continuous cover forestry (Section 3.1.2.1). In peatland-dominated forested catchments, other possibilities for catchment water management are restoration of drained peatlands (Bonn et al. 2016) and wetland re-establishment (Section 3.1.2.2).

Field studies show that catchment water management by controlled drainage and utilisation of different water retention practices in general can substantially decrease the suspended sediment, TP and particulate organic carbon (POC) loads (Robotham et al. 2023; Zhang et al. 2011), improve flood and drought resilience (Janatrostami 2024; Lamsodis et al. 2006; Robotham et al. 2023; Salla et al. 2021), increase water levels in lakes suffering from prolonged droughts (Nikraftar et al. 2021), and sustain crop yields (Zhang et al. 2011). Typically, multiple benefits, both economical, and ecological, can be obtained, although careful technical design and extensive catchment-scale application of measures is needed to obtain measurable impacts in the receiving water bodies (Nikraftar et al. 2021; Robotham et al. 2023). In terms of drinking water treatment, a recent cost-effectiveness analyses has identified a hybrid strategy combining catchment management and capital-intensive water treatment approaches as a profitable solution due to savings provided by catchment management practices tackling pollutants at source (Glass and Burgess 2025). However, catchment water management alone cannot reduce harmful substances enough for achieving drinking water compliance, for which also technical drinking water treatment measures are needed (Glass and Burgess 2025).

The impacts of catchment water management are expected to be long-term if the measures are adequately maintained and adjusted to changes in climate. Different models and tools are often available for the tracking of nutrient export control with respect to land use practices on catchment-scales. For instance, a management model Groundwater Loading Agricultural Management Systems (GLEAMS) has been successfully extended to the basin scale in Lake Vico, Italy (Ripa et al. 2006). Enhanced coherence in water governance and improved stakeholder engagement is crucial to determine the sustainable allocation of water for various agricultural users especially in drought-prone areas. Likewise, supportive policies and economic incentives are crucial in mainstreaming catchment water management practices and other load reduction approaches to everyday agriculture.

3.1.1.2 Soil amendments

Soil amendments have been increasingly developed since around 2010 mainly to decrease the leaching of P from agricultural fields. Soil amendments are reactive materials that can be either spread on the entire treated field area (Ekholm et al. 2024; Uusi-Kämppä et al. 2022) or applied as distinct permeable reactive barriers that intercept the runoff (e.g., Bus et al. 2019) (Section 3.1.3.3). Utilisation of soil amendments are to improve the soil structure, that in a good condition can support crop productivity, enhance flood resilience, improve soil biodiversity and reduce water pollution (e.g., Brassard et al. 2019; Norberg and Aronsson 2022). Applied materials include structure lime (mixture of CaO or $Ca(OH)_2$ and $CaCO_3$), pulp and paper mill sludge, gypsum, autoclaved aerated concrete, Polonite, zeolite, biochar, and limestone, preferably from recycled sources and can thus be considered as a CBS for improved lake protection.

Some of the soil amendments (structure lime, pulp and paper mill sludge, gypsum (Figure 2)) have been rather extensively studied in Finland, where these materials have decreased the loads of phosphate phosphorus ($P\text{-}PO_4$), total phosphorus (TP), and suspended solids (SS) from the treated field plots by ~25-60% (Ekholm et al. 2024; Ollikainen et al. 2024; Uusi-Kämppä et al. 2022). Similar reductions using structure lime have been observed in Sweden with 38-45% lower leaching of TP compared to the non-limed control plots (Norberg and Aronsson 2022). Structural liming has also significantly reduced P leaching from subsurface drains suggesting that soil structure improvement can remarkably benefit eutrophication control (Svanbäck et al. 2014). For gypsum, the reductions are typically larger in the particulate than in the dissolved P fractions. Substantial decreases in the loads at the catchment-scale require treating at least several tens of % of the total agricultural area with soil amendments. At the catchment scale, reductions of ~15-25% in particulate P and SS have been reported over a 5-y period when around half of the agricultural fields (23% of the entire catchment area) were treated with gypsum (Ekholm et al. 2024). Simultaneous reductions have been also reported for dissolved organic carbon (DOC). Pulp and paper mill sludges have also significantly reduced SS and TP concentrations in percolation water by reducing soil erodibility. The effect declined with time, but the reduction for both was still >25% 4 yr after application (Rasa et al. 2021). The result longevity of soil amendment varies depending on the material and type of application. The performance of e.g. gypsum decreases after ~5 years (Ekholm et al. 2024).

A co-benefit of spreading soil amendments that decrease the SS loads and erosion is the conservation of the fertile soil on the fields. Some soil amendments also enhance the structure of the soil and have slight, typically positive, effects on crop yields (Uusi-Kämppä et al. 2022).

With a TRL 9, structure liming is a prioritised measure for reducing P losses from arable land in Sweden (Norberg and Aronsson 2022). Efficiency of pulp and paper mill sludges has also been tested and demonstrated on operational scales (Rasa et al. 2021). Gypsum treatment has also been successfully piloted at a rather large scale particularly in Finland (Ekholm et al. 2024; Ollikainen et al. 2020). However, it is noteworthy that gypsum treatment is not recommended for use in lake basins as the material contains sulphate that could increase internal loading (Ollikainen et al. 2019). Costs for other soil amendments apart from gypsum treatment have not been reported. For gypsum, it was estimated by Ollikainen et al. (2019) that the total cost per treatment could be 220 €/ha. The authors concluded that soil amendments can be a cost-efficient and attractive measure for pollution control as they do not reduce crop yields or arable area and hence, do not result in loss of income for farmers (Ollikainen et al. 2019; Ollikainen et al. 2020).

Figure 2. Different soil amendments ("maanparannusaineet") such as pulp and paper mill sludge ("kuitu"), structure lime ("rakennekalkki"), and gypsum ("kipsi") studied in Southern Finland. © Laura Härkönen.

3.1.1.3 Two-stage channels

Two-stage channels (TSCs) are a nature-based solution aiming at ensuring agricultural drainage and flood management with less ecological impacts compared to the conventional dredging of channels (Damphousse et al. 2024; Västilä et al. 2021). Dredging within channels decreases the in-stream habitat quality, biodiversity and retention of nitrogen and phosphorus, increasing the nutrient loads transported downstream (Pierce et al. 2012). Because of excessive sediment deposition on the widened channel bed, conventionally dredged channels typically need re-dredging after every few decades.

TSCs mimic the natural geometry of lowland streams as they consist of a confined floodplain excavated on one or both sides of the existing agricultural ditch, stream or small river (Figure 3, Figure 4). The TSC design aims at creating a self-cleansing low-flow (main) channel and thus at decreasing the required frequency of maintenance. Vegetated floodplains of TSCs retain suspended sediment and phosphorus (Västilä et al. 2021) and reduce nitrogen loads mainly through denitrification (Hallberg et al. 2024b; Roley et al. 2012). In case of vegetated two-stage channel, this NbS is also mentioned in literature as vegetated drainage ditches (VDD, (Nsenga Kumwimba et al. 2018)). If the groundwater table is shallow and close to the ditch, the floodplain can be planted with trees acting also as buffer strips targeting groundwater interception (Gumiero and Boz 2017).

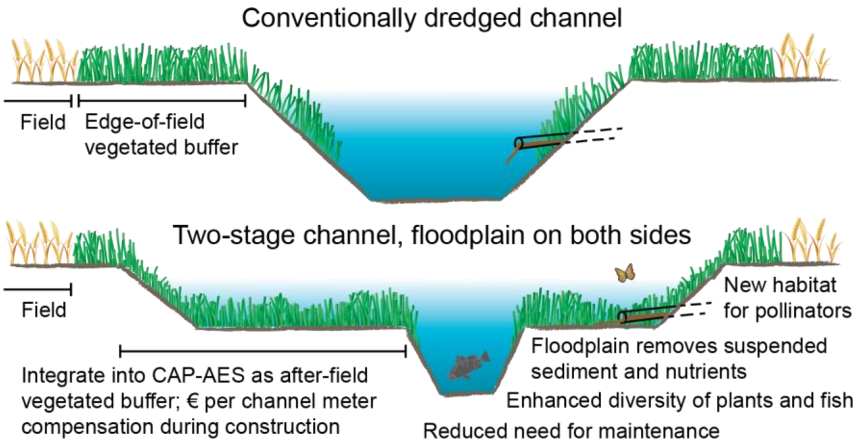

Figure 3. Schematic presentation of conventionally dredged channel and a two-stage channel with floodplains on both sides. Figure from Västilä et al. (2021).

Several scientific case studies and reviews show partly contrasting impacts of TSCs on the total SS and P loads. The few representative field studies covering the entire annual hydrograph have shown changes in TP loads ranging from 20% reductions (Hallberg et al. 2024a) to no changes (Mahl et al. 2015) and to 2% increases (Västilä and Jilbert 2025). A TSC with self-cleansing low-flow channel has resulted in a 9% increase in SS load (Västilä and Jilbert 2025). Studies measuring only concentrations have shown increased SS concentrations (Davis et al. 2015) and decreased (Mahl et al. 2015) or varied results on turbidity (Kindervater and Steinman 2019). Modelling investigations suggest TSCs have high potential for soluble reactive phosphorus (SRP) load reductions (Trentman et al. 2020). By damming the low-flow channel, higher removal efficiency of 38% for nitrogen (N), 40% of P and 67% of SS were registered (Vymazal and Dvořáková Březinová 2018), as the channel's behaviour started to resemble to that of CWs (3.1.3.1). In case of favourable groundwater table location, the inclusion of trees as buffer strips could buffer up to 75% of nitrate pollution from groundwater (Gumiero and Boz 2017). Uncertainties result from different monitoring methods and time frames of the studies. Additionally, the performance of TSCs depends on the channel design, length of TSC, channel history and sediment properties.

It appears that there are some trade-offs in the environmental performance of TSCs as excessive deposition on main channel bed substantially decreases nutrient loads (Hallberg et al. 2024a) but is likely harmful for local aquatic biodiversity (Västilä and Jilbert 2025). Limited evidence suggests that TSCs with self-cleansing low-flow channels have positive effects on biodiversity (Västilä et al. 2021), such as greater species richness of fish (DeZiel et al. 2019), and could thus be also considered as BfS. Converting part of the conventionally dredged channels into two-stage channels increases the local catchment-scale richness of stream invertebrates, diatoms and plants, and riparian beetles and plants (Huttunen et al. 2024; Västilä et al. 2021). The required maintenance of TSCs is less frequent and disruptive than for conventional dredging, as selective/partial mowing of the floodplain vegetation can be used as a maintenance measure instead of dredging to maintain the conveyance capacity.

Overall, TSCs appear to have benefits for both long-term agricultural water management, and thus crop yields, and for the environment. However, the financial compensation for the landowners for establishing TSCs through European Union agri-environmental subsidy scheme (CAP-AES) is lower than the 2-3 fold additional costs of TSCs compared to conventionally dredged channels (Västilä et al. 2021), at least in Finland, which is a shortcoming not encouraging the application of TSCs.

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A B

Figure 4 –Southern Finnish two-stage channels at A) stream Ritobäcken during August 2021, © Laura Härkönen and B) river Perniönjoki in June 2025, © Harri Aulaskari. In both applications, there is a vegetated floodplain at the edge of field (to the left) and a low-flow channel (obliquely to the right).

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Table 1 – Applicability and scalability potential of reviewed innovative solutions related to agricultural practices.

Solution	Applicability	Benefits	Disadvantages	Costs	Scalability potential
Catchment water management (NbS)	Drained catchments	Improved flood and drought resilience; decreased suspended sediment, TP and POC loads; improved crop yields; reduced costs of drinking water treatment	Careful technical design required to maintain high agricultural productivity while managing drainage system	Not provided	High, TRL up to 9. Frequently implemented in agriculture but supportive policies and incentives crucial
Soil amendments (CBS)	Agricultural fields	Improved soil structure; improved crop productivity; reduced erosion risk; reductions in P-PO_4, TP, and suspended sediment loads; circular economic co-benefits when using recycled materials	Performance may decrease over time. Gypsum not applicable to lake catchments as it contains sulphate	Only provided for gypsum, for which total costs ~220 €/ha (Ollikainen et al. 2019)	High, TRL 7-9 depending on amendment used
Two-stage channels (NbS)	Lowland drainage ditches with mild slope	May reduce suspended sediment and nutrient loads; likely improves aquatic and riparian diversity; likely enhances flood resilience in the long term	May increase suspended sediment loads if there are abundant prior deposits in the main channel bed; losses of field area	Construction 21 €/1 m per channel length; ~60 000 €/ha per floodplain (Västilä et al. 2021)	High, TRL 7-9 with some aspects of performance proven in operational environment. However, insufficient support from policies hinders the applicability.

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3.1.2 Measures related to forestry practices and forest and wetland related habitats

3.1.2.1 Continuous cover forestry

In conventional rotation forest management (RFM) most commonly applied in Europe (Mason et al. 2021), the most substantial nutrient, suspended solid and organic carbon loads tend to occur following regeneration cuttings, particularly clearcuttings with soil preparation, ditch network maintenance (DNM), and forest fertilisation activities (e.g., Kaila et al. 2015; Nieminen 2004; Nieminen et al. 2015; Piirainen et al. 2013).

Application of continuous cover forestry (CCF) is a promising nature-based strategy for both peatland and mineral soil forests, in which large-scale clearcuttings are avoided. Instead, selection cutting, strip cutting, shelterwood cutting and small gap cuttings are used, leaving sufficient volume of growing vegetation to sustain evapotranspiration (Sarkkola et al. 2025). CCF usually involves the use of natural regeneration, selective harvesting, and clear-fellings with a gap size below 0.25 ha (Schütz et al. 2012) and it is suggested to reduce loading to surface waters by diminishing the need for DNM and soil preparation and by supporting natural regeneration (Dawson and Smith 2007; Härkönen et al. 2023; Nieminen et al. 2018a; Palviainen et al. 2022; Sarkkola et al. 2025). Additionally, CCF can reduce wind damage to stand (Pukkala et al. 2016) and increase the recreational value and forest biodiversity by increasing the forest structural complexity and reducing the physical disruption to the site (Mason et al. 2021; Sarkkola et al. 2025). When applied in peatland forests, CCF could potentially reduce greenhouse gas (GHG) emissions by reducing the water level fluctuation and subsequent peat decomposition (Nieminen et al. 2018a). However, the benefits of CCF are so far mostly postulates and the evidence base is weak due to decades of stand rotation and hence, lack of longer-term studies. Thus far, few results have been published on e.g., the impacts of CCF on discharge water quality, and further research is needed on the processes controlling element exports from CCF sites with varying attributes (Palviainen et al. 2022; Sarkkola et al. 2025). Additionally, the risks associated with CCF require further examination. CCF may for instance pose a higher spreading risk of root pathogens than RFM, as found for Norway spruce (*Picea abies*) due to efficient secondary infection from overstory trees to dense regeneration (Nevalainen 2017; Piri and Valkonen 2013). More frequent harvesting in CCF sites and consequent risk of damage to tree caused by heavy off-road forest machinery is also one of the major concerns (Sarkkola et al. 2025). However, little experimental research is yet available about possible damages and methods to prevent them (Ahtikoski et al. 2025).

Despite these knowledge gaps, we highlight that CCF is a promising forest management practice reducing the need for measures producing heavy loading and as such, has potential for providing broader range of ecosystem services compared to RFM and being more efficient in diminishing negative water quality impacts than most end-of-pipe methods (Hertog et al. 2022; Härkönen et al. 2023). In the beginning of 2020s, 22-30% of European forests were estimated to be managed through CCF (Mason et al. 2021), although it has a high scalability potential to both peatland and mineral soil sites. However, forestry culture and education, industrial networks and timber markets promoting RFM are identified as barriers preventing CCF from becoming mainstream (Hertog et al. 2022). Improved competence in CCF within the forestry profession (Hertog et al. 2022; Mason et al. 2021) together with supportive policies, incentives and legislation are essential to accommodate CCF from niche to regime approach in European forestry. While RFM may provide higher private net revenue than CCF, the latter is an environmentally effective alternative bringing significant water quality benefits from the society’s point of view (Miettinen et al. 2025).

3.1.2.2 Peatland restoration and wetland re-establishment

Peatland restoration and wetland re-establishment include rewetting formerly dried by human activities in the past. Peatland restoration is often conducted by drain blocking and, in case of drained forested peatlands, accompanied with harvesting (Figure 5). Although the restoration of drained peatlands may pose a risk of increased initial export of nutrients and organic carbon (Koskinen et al. 2017), it can be a cost-effective option eventually putting an end to water emissions from drained peatlands (Juutinen et al. 2020). Additionally, peatland restoration has estimated to bring significant co-benefits for flood and drought risk management, GHG emission reduction and biodiversity (e.g., Bonn et al. 2016).

As a lighter version of peatland restoration, ditch diversion for overflow to partially dried peatlands affected by surrounding ditch network has recently received attention in Finland as a potential measure improving water retention in the catchment areas (Granqvist 2024). However, information on the water quality benefits of ditch diversion is still lacking, although overland flow areas and wetland buffers in general are currently considered as the most efficient water protection measures in forested catchments (e.g., Nieminen et al. 2018b). Drained areas could also be re-wetted using reintroduction NbS ecosystem engineers such as beavers, as Law et al. (2017) described from a Scottish wetland re-establishment study. Authors found that beaver re-introduction remarkably benefitted plants associated with high moisture with increased mean plant species richness and the number of species recorded (Law et al. 2017). However, such approach is strongly restricted to specific conditions and comes with a limited scalability potential.

Wetland re-establishment aims to bring back the lost landscape functions as the habitat for aquatic flora and fauna. For successful wetlands re-establishment, restoration of past hydrological conditions and sometimes modifying geomorphological conditions to favour sufficient water retention in the area. For instance Macquarie and Etra (2001) described a massive restoration project of a historical wetland in the Tahoe Lake basin in US by soil excavation, revegetating native wetland plants, improving flood control by expanding culverts and enhancing fish passage to reconnect L. Tahoe and upstream Snow Dreek tributaries. This action brought back the wetland function of the area, benefited biodiversity and increased flood risk management capacity (Macquarie and Etra 2001). Indeed, restored wetlands play a crucial role in the flood and drought resilience management (Hoffmann and Baattrup-Pedersen 2007) (see also https://www.epa.gov/wetlands/wetlands-restoration-definitions-and-distinctions), are anticipated to benefit carbon sequestration abilities and reduce nutrient export from the drainage basin (Hoffmann and Baattrup-Pedersen 2007).

Figure 5 – A restored, formerly drained Finnish peatland. Former drains can be identified by their lighter colour. © Maarit Similä

Table 2 – Applicability and scalability potential of reviewed innovative solutions related to forestry practices and forest related habitats.

Solution	Applicability	Benefits	Disadvantages	Costs	Scalability potential
Continuous-cover forestry (NbS)	Mineral to peatland forest sites	May reduce nutrient, suspended solid and organic carbon export to surface waters; can potentially reduce GHG emissions from peatland forests by reducing soil water level fluctuation	Requires successful natural regeneration of trees, potentially posing a risk of shade-tolerant conifer species outcompeting others; potentially increased risk of damage to stand caused by forestry machinery due to more frequent harvesting	Numbers not provided. However, economical model shows lower private net revenue compared to RFM (Miettinen et al. 2025)	High, TRL 8-9. Increasingly common management practice in Europe. However, lack of knowledge together with controversial public perceptions hinder the upscaling
Peatland restoration and wetland re-establishment (NbS)	Forested or peatland-dominated catchments with intensive drainage	Habitat restoration; increased biodiversity; potentially reduced nutrient export in longer term; improved carbon sequestration and climate protection	Risk of temporarily increased nutrient and OC release from the re-wetted land	Costs not provided for peatland restoration. For wetland re-establishment, e.g., 33,000 USD (~29,000 €)(Comoss et al. 2002). Vary case-specifically.	High, TRL 9.

3.1.3 Measures related to urban habitats

This section describes the reviewed innovations originally developed and applied in urban environments. However, we highlight that many of the water protection measures described here are applicable, frequently utilized and scalable to agricultural and forestry practices and can be considered as more general solutions potentially trapping pollution.

3.1.3.1 Constructed wetlands

Constructed wetlands (CWs) are a well-known NbS for wastewater treatment, developed since the 1960's (Vymazal 2022). The idea of CWs lies in the biological processes of pollutants transformation by microorganisms and hydrophytes as well as in the filtration and sedimentation of particulate matter. Hydrophytes create a base for microorganisms active in the biogeochemical cycling of nutrients (carbon (C), N, P, sulphur (S)) (Nowak et al. 2013; Skrzypiec and Gajewska 2017) and slow down the flow, thus enhancing sedimentation (Kill et al. 2022) and biological removal of pollutants.

CWs can be established either as free water surface wetlands with sedimentation basins, channels and macrophyte zones; horizontal sub-surface flow fields with a porous medium under the surface of the bed; or vertical flow wetlands comprising of a flat bed of graded gravel topped with sand planted with macrophytes (Mostafa et al. 2022; Vymazal 2022). Advanced CWs have been also recently developed (Wu et al. 2014), including aerated wetlands (Nivala et al. 2020) and French reed beds for the treatment of raw wastewater (Morvannou et al. 2015). These are targeted to increase of treatment performance, as well the reduction of the areal footprint (Dotro et al. 2017) and of the operational and maintenance costs (Rizzo et al. 2018). Increase of treatment performances have been tested also with innovative media adsorbing and reactive media; typical alternative media have been proposed for increasing P removal (Vohla et al. 2011), such as apatite or lanthanum-modified bentonite. Interestingly, innovative media in CW also includes the use of biochar (Nguyen et al. 2020), creating an interesting link with CBS solutions.

CWs with a TRL of 9 are frequently used for domestic, storm and industrial wastewater treatment (Gunes and Tuncsiper 2009; Ijff et al. 2021; Irvine et al. 2023; Kupiec et al. 2022; Nguyen et al. 2021; Sánchez-Almodóvar et al. 2022; Tao and Xiong 2021; Xie et al. 2012) and can be considered as green infrastructure urban water management (Stefanakis 2019). Additionally, CWs are also applicable and commonly used for water protection in agriculture (Kill et al. 2022; Koskiaho and Puustinen 2019) and also forestry (Finér et al. 2020)(Figure 6). CWs are able to effectively remove nutrients, organic compounds, heavy metals and suspended sediment (Irvine et al. 2023; Kasak et al. 2018; Vymazal 2014; Vymazal 2022), although contrasting results for e.g., N retention are also presented and the ability of retaining dissolved substances is often limited (Kasak et al. 2018). However, retention of dissolved nutrients may increase when CWs are aging (Koskiaho and Puustinen 2019). Also the phytoremediation by vegetation can significantly improve the purification efficiency of CWs (Zamora et al. 2019). In terms of diffuse pollution, free water surface CWs can be considered a suitable NbS at catchment scale too, especially for removing nutrients (Rizzo et al. 2023) and pesticides (Vymazal and Březinová 2015), as also organic micropollutants and microplastics (Koukoura et al. 2024; Sarti et al. 2024).

A strong advantage of CWs is much lower costs of construction and operation compared to traditional wastewater treatment methods (Rizzo et al. 2018; Vymazal 2022). A limit of this solution is temperature, in which the wetland is operating, as for most of the biological treatment systems. CWs work more effectively in the areas with higher annual temperatures as their effectiveness in the temperate zone is much lower during colder seasons (Vymazal 2022) and the acclimatisation of CWs in northern countries can take several years (Kasak et al. 2018). However, as the physical sedimentation and filtration processes are independent of temperature, good decrease in SS and total P loads has been observed in agricultural CWs also in Northern Europe (Kill et al. 2022) and recent works reviewed the capability of delivering an efficient treatment also in cold climate, especially if properly insulated (Ji et al. 2020). Another limit of CWs is their footprint, which is often competing with other land users in the basins, but this factor can be compensated by the higher creation of side-benefits when compared to other approaches (Liquete et al. 2016).

The use of CWs to complement pollutant control in the catchment areas not only has potential for positively influencing water quality of lake tributaries but also create a link to BfS by creating habitats for fish and birds (Kupiec et al. 2022; Mostafa et al. 2022). CWs can also play a role in flood and drought risk mitigation via increasing the water storage capacity within the catchment areas (Sánchez-Almodóvar et al. 2022; Tao and Xiong 2021). The scalability potential of CWs is high, but often a prerequisite for improved performance is a sufficient size of CW in relation to the size of its catchment area. For instance, Koskiaho and Puustinen (2019) reported higher retentions in CWs with larger CW-to-catchment area ratio allowing for longer residence time of water and, subsequently, longer period for water-purification processes within the CW.

Figure 6 – A constructed wetland in an agricultural, forested landscape in Finland. © Laura Härkönen

3.1.3.2 Floating treatment wetlands

Floating Treatment Wetlands (FTWs), also referred to as floating beds or vegetated rafts mimic the functions of natural wetlands to improve water quality and support aquatic biodiversity. These systems consist of buoyant platforms plant ed with emergent macrophytes, such as *Iris pseudacorus*, *Phragmites australis*, and *Juncus effusus*, whose roots extend into the water column, facilitating nutrient uptake and microbial pollutant degradation. FTWs are designed to reduce eutrophication and pollution from nutrients and heavy metals while also contributing to landscape aesthetics and habitat provision. Tested in a wide range of environments, including urban retention ponds, stormwater basins, eutrophic lakes, and industrial discharge channels, FTWs have been studied in countries such as Finland, Poland, Sri Lanka, China, and the USA (Hartshorn et al. 2016; He et al. 2022; Ozan and Yilmazer 2020; Rodrigues et al. 2022; Tao et al. 2017; Weragoda et al. 2010) and are assessed to be a promising technology for the remediation of stormwater and other effluents (Batista et al. 2025).

The systems are typically constructed from polyethylene or biodegradable mats and range in size from a few square meters to several thousand. They are anchored in place and require maintenance such as periodic biomass harvesting. While generally considered low- to medium-cost solutions, detailed cost data are rarely reported. FTWs have demonstrated reductions in total phosphorus, nitrogen, chlorophyll-a, and heavy metals like zinc and copper, while also providing habitat for birds and aquatic invertebrates (Colares et al. 2020). They contribute to urban green-blue infrastructure and may also support climate mitigation goals through carbon uptake and greenhouse gas reduction. Despite these benefits, the evidence base consists mostly of short-term studies (often under one year), and longer-term performance data under variable seasonal conditions are limited. Implementation challenges include plant vulnerability to harsh weather, anchoring stability, and site-specific design requirements. Nonetheless, their modular design allows for flexible deployment, and their benefits align well with the objectives of the WFD and broader sustainability policies.

The estimated TRL of FTWs ranges from 4 to 9, based on a synthesis of nine peer-reviewed studies. These range from experimental-scale setups (e.g., Henny and Kurniawan 2019; Henny et al. 2019) to full-scale, operational installations (e.g., Hartshorn et al. 2016; He et al. 2022). Overall, FTWs may represent a scalable and adaptable restoration innovation that can enhance water quality and biodiversity of pond-like water protection structures particularly in settings where conventional approaches are not feasible or sufficient. However, the effectiveness of FTWs is still debated and the size, coverage and position of floating modules for their best performance still require further investigations.

3.1.3.3 Permeable Reactive Barriers

Permeable reactive barriers (PRB) are underground technological solutions, installed *in situ*, used to pre-treat contaminated groundwater (Blowes et al. 1997; Borden et al. 1997). The principle working of PRB technology involves the emplacement of a reactive wall filling with reactive material perpendicular to the potential trajectory of the contaminated groundwater (Song et al. 2021). Depending on the type of contamination, an appropriate reactive material is selected to capture specific contaminants. The reactive material may be activated carbon, bentonite, limestone or zerovalent iron (Gillham et al. 2010) but also metal oxides/sulphides, mineral material, industrial waste, ion exchange resin, organic polymers, and carbonaceous materials for the remediation of heavy metal contaminated groundwater have been investigated and reviewed (Song et al. 2021). Possible use of recycled materials makes the PRB a sustainable remediation option (Phillips 2009).

Pollutants react with active material, and they can be captured inside the barrier (via adsorption and precipitation) while the non-harmful reaction products pass the barrier. The installed barriers may also be composed of a mixture of reactive materials with soil formations. Native soil may also be removed, and reactive material is introduced in its place. In some cases, washable filters are also installed. In some technologies, biologically active compounds (e.g. active microorganisms) are also added to the barrier (Song et al. 2021), stimulating biological transformation of pollutants. Such solutions were implemented for example in Poland for protection water bodies Jelonek Lake and Sulejów Reservoir (Bednarek et al. 2010; Frątczak et al. 2019; Izydorczyk et al. 2013). The active barriers in these applications were constructed using sawdust or other waste materials (e.g. barley straw, corn cobs, cotton, flax straw, culm), limestone with addition of denitrifying microorganisms, giving high efficiency (higher than 77%) N compounds (NO_3^-, NH_4^+ and TN) removal.

PRBs may eventually need to be decommissioned for the completion of remediation, due to which a careful consideration of the ease of removal of the PRB should be considered as part of the design prior to installation (Phillips 2009). PRBs can also become clogged, and the reactive material coated causing them to become less effective. According to Phillips (2009), a funnel-and-gate design with the reactive material placed in single or sequenced containers is probably the most cost-effective design, as it uses less reactive material than a continuous trench.

Permeable reactive barriers can also be used in agricultural landscapes to reduce P loading as demonstrated by (Bus et al. 2019). Laboratory tests have yielded P reductions of up to 65-99% depending on the material (Bus et al. 2019), but the sorption is typically lower in the long term and in field conditions. However, the usage of PRBs in agricultural landscapes have to our knowledge only been studied in laboratory scale, yielding to a low TRL of 3-4.

3.1.3.4 Sediment and nutrient interceptors

Sediment interceptors, also known as sediment traps, can be compact engineered systems designed to reduce sediment and pollutant loads from surface runoff before reaching lakes, ponds, or other receiving water bodies. These systems are typically installed at inflow points and function by trapping suspended solids through gravity-based separation or settling mechanisms.

In the reviewed studies, one system involved an on-site treatment unit targeting polluted pond sediments (Chang et al. 2010), while another employed a bottom-grid structure (BGS) designed to reduce sediment loading from land use runoff (He et al. 2014). Additionally, a novel sedimentation-biofiltration (SED-BIO) system consisting of different segments with various water purification processes was described and implemented in a stream draining to L. Jelonek, Poland (Kupiec et al. 2022). SED-BIO system included a sedimentation segment with gradually releasing micro-organisms, filtration by willow gabions and a plant biofilter, i.e. a vegetated CW to take up nutrients (Kupiec et al. 2022). Studies on these engineering solutions evaluated performance under field conditions and demonstrated moderate effectiveness in reducing suspended solids and particulate nutrients (Chang et al. 2010; He et al. 2014; Kupiec et al. 2022). These systems may be particularly suitable for urban and agricultural catchments where sediment transport is high and full-scale retention basins may not be feasible. Due to their modular design and small footprint, sediment interceptors could be scaled to fit various drainage structures. However, their effectiveness depends on the interceptor's volume in relation to the size of the catchment area and can also be affected by the sediment size distribution (He et al. 2014). The performance is typically lower at high flows and sediment interceptors require regular maintenance to remove accumulated material to ensure longer-term performance. While cost data were not reported, their localized scale suggests potential cost-efficiency.

Other solutions for retaining sediment flushed from agricultural fields were recently revised by Smith and Muirhead (2024) suggesting that sediment interceptors can also be less engineered, such as geotextile silt fences, or classified as NbS, including silt traps and decanting earth buds, detainment bunds, modified drainage ditches or farm ponds. Novel products developed to intercept P in streams, drainage systems, or agricultural fields to reduce leakage into lakes also include examples such as Eutrosorb F (Sepro), Kylin Pellets (Eastern Node), BioPhree (see https://www.balticwaterhub.net/innovation/aquacare-biophreer-capturing-phosphate-through-absorption), and porous filters (Kumar et al. 2019). These materials can either be applied as fertilizer or phosphate can be freed from it for recycling while the carrier can be re-used for new binding. Also a P removal structure containing 20 m^3 spent lime drinking water treatment residual (DWTR) that was constructed at an existing stormwater outfall into Wakefield Lake (Maplewood, Minnesota, USA) reduced total P loading by 70.9%, and dissolved reactive P by 78.5% linking sediment interceptors to CBS.

TRL were estimated at level 5 for engineering solutions, indicating some successful field trials but limited scaling. Further tests are required to evaluate their net benefit at the landscape scale and to propose guidance for their maintenance. However, TRL of up to 9 can be considered for some sediment interceptor solutions that were successfully implemented in real conditions since decades (Smith and

Muirhead 2024). These interceptors serve as useful tools for reducing external loading into lakes and may complement broader catchment-level water protection strategies.

3.1.3.5 Stormwater basins

Stormwater basins are engineered surface water retention or detention systems that capture and treat urban runoff before it enters receiving water bodies. These basins function by providing temporary storage of stormwater, allowing sediment and pollutants to settle, and biological and chemical processes to reduce contaminant loads. While not wetlands themselves, stormwater basins are often classified as hybrid NbS/engineered solutions when designed with vegetated edges, infiltration zones, or ecological enhancement features. They address a range of catchment-scale pressures including nutrient and metal pollution, hydrological alteration, and urban runoff surges (Adhikari et al. 2023; Honour et al. 2013).

In the reviewed studies, stormwater basins have been tested in contexts involving heavy metals (e.g., copper (Cu), zinc (Zn)) and base cation contamination from urban or industrial land uses. Adhikari et al. (2023) investigated basin performance in Finland, focusing on the influence of storm intensity and water residence time on treatment efficiency. Results indicated that pollutant removal was strongly correlated with rainfall conditions, suggesting that design features such as retention volume and inflow control are key to ensuring consistent treatment performance. These systems also demonstrated the capacity to trap fine sediments and attached pollutants even in compact urban landscapes (Honour et al. 2013).

Stormwater basins are generally suited for urban and peri-urban catchments where impervious surface cover is high, and the volume of runoff is substantial. They can be designed to accommodate a range of sizes, from small neighbourhood-scale basins to large municipal systems, and can be retrofitted into existing drainage infrastructure. Some systems are purely functional and hardscaped, while others incorporate vegetation and habitat features, thus providing co-benefits for biodiversity and public amenity (Liebl 2010).

Performance outcomes from the reviewed studies included moderate to high reductions in total suspended solids (TSS), particulate-bound metals, and occasionally nutrients, though the latter is more variable. Challenges include seasonal performance shifts, especially in cold climates, and inconsistent removal under high-intensity storm events. Long-term sustainability depends on regular sediment removal, vegetation management, and adaptation to changing storm patterns (Adhikari et al. 2023).

TRLs for stormwater basins are high, typically TRL 8–9, as they are widely implemented across Europe and globally. However, innovation lies in improving their design for multifunctionality, resilience to climate extremes, and integration with other green infrastructures, such as rainwater gardens that are also demonstrated in one of the FutureLakes' Innovation sites, Loch Leven (UK). No cost data for stormwater basins were reported in the reviewed studies, though stormwater basins are generally considered moderate-cost interventions relative to their scale and land requirements. Their modularity and proven performance make them a foundational component of urban water protection strategies and a candidate for inclusion in lake restoration frameworks, particularly where external loading from stormwater is a dominant pressure.

3.1.3.6 Wastewater treatment

Wastewater treatment is the combination of physical, biological and chemical processes for pollutants removal from sewage which otherwise would end up in the receiving water bodies. The process is conducted in artificial conditions as a sequential process often including a primary physical treatment (screening, filtration, sedimentation, grit removal) followed by secondary (biological processes using active sludge technology or biofilters) and tertiary treatments (advanced physical, biological and chemical methods for nutrients removal or specific pollutants removal). Industrial sewage treatment is more complicated, and treatment techniques are depended on the specific features of the wastewater (Dubey et al. 2024; Osmani et al. 2023; Sonune and Ghate 2004). If suitable area at an affordable cost is available, wastewater treatment plant can also be established as a NbS, i.e. using a CW (Section 3.1.3.1). NbS can be efficient in treating e.g. combined sewer overflow (CSO) pollution when optimized for dealing with stochastic conditions of combined sewers (Rizzo et al. 2020). A system consisting of a rotating belt filtration, activated carbon filtration and UV disinfection for treating CSO pollution has also been tested in L. Garda's catchment in Italy, where the pilot-scale installation showed promising results with the treatment cost per 1 m^3 of sewage amounted to 1.2 € (Botturi et al. 2020).

Today, wastewater treatment faces new challenges, such as new types of pollutants or specific compounds and thus we are forced to seek modern, innovative solutions for water and sewage treatment. Recent advances in wastewater treatment based on the reviewed innovations are linked to utilisation of different nature-based agents for the removal of pollutants (e.g., Li et al. 2021) and possible reuse of treated wastewater (Dubey et al. 2024; Gukelberger et al. 2020). Li et al. (2021), for instance, tested microalgal growth, nitrogen uptake and storage, and dissolved oxygen production in a polyculture based-open pond fed with municipal wastewaters in northern Sweden. Their results suggested that a local consortium of microalgae is efficient in N and P removal from municipal wastewater in boreal zone installation and may potentially assists in GHG emissions by carbon dioxide (CO_2) sequestration (Lage et al. 2021). Li et al. (2021) piloted high-rate algae ponds (HRAPs) as a technical solution to treat domestic wastewater for selenium (Se) removal and investigated the consequent production of Se-enriched microalgae as potential feed supplement. The HRAP system achieved removal efficiencies of 43% for selenium, 93% for ammonium nitrogen, 77% for total phosphorus, and 70% for chemical oxygen demand (COD) (Li et al. 2021). On average 49–63% of Se in the Se-enriched microalgae was bioaccessible for animals and was concluded to potentially offer a promising alternative for upgrading low-value resources into high-value feed supplements (Li et al. 2021).

Domestic wastewater treatment in general has substantial nutrient recovery potential in both urban and peri-urban context if source separation systems for either black water or urine diversion can be utilized (Malila et al. 2019; Wielemaker et al. 2018). It has been estimated that by source separation, a significant increase in the recovery rate of phosphorus (41–81%) and nitrogen (689–864%) compared to the conventional system could be achieved (Lehtoranta et al. 2022). Improved nutrient recovery from wastewaters could potentially reduce the need for mineral fertilizers in agriculture (Lehtoranta et al. 2022; Malila et al. 2019; Wielemaker et al. 2018). However, utilisation of human urine in fertilisation poses a risk of field acidification (Malila et al. 2019) that should be overcome to allow for wider application. Moreover, despite of the great potential source separation systems have in fostering circular economies, the implementation of these systems in urban areas would to a large extent require an entire system change of the wastewater treatment sector (Lehtoranta et al. 2022). Consequently, utilisation of source separation systems could especially benefit rural areas, in which source separation would also benefit eutrophication control (Lehtoranta 2022).

Table 3 – Applicability and scalability potential of reviewed innovative solutions related to urban environments (continued, 1/2).

Solution	Applicability	Benefits	Disadvantages	Costs	Scalability potential
Constructed wetlands (NbS)	Urban environments, also agriculture and forestry. From warm to temperate and boreal zones.	Cost-effective solution compared to traditional wastewater treatment plants; reduced nutrient and SS loads; improved flood and drought risk management; local biodiversity co-benefits	Effectiveness reduced during cold season; requires sufficiently large areas for construction with optimal spatial targeting	Vary case-specifically (Djodjic et al. 2022). In e.g., Italy, examples from construction 100-200 €/m² for subsurface flow, 50-70 €/m² for surface flow.	High, TRL 7-9 with mature technology and wide implementation on operational scales
Floating treatment wetlands (NbS, BfS)	Stormwater ponds; urban discharge zones; adaptable to deep or shallow conditions and potentially applicable in small lakes	Reduced water nutrient and metal concentrations via phytoremediation; local biodiversity support; urban amenity enhancement	Small-scale units with effectiveness limited to smaller-scales and application to lakes most probably supports local biodiversity but has negligible impact on water quality; effectiveness reduced during cold seasons; challenges with wave exposure if applied in larger units	Not provided.	Intermediate to high, TRL 4-9, varying technologies implemented on operational scales.
Permeable reactive barriers (NbS, CBS)	Industrial or agricultural catchments to allow for polluted groundwater control	Pollutant removal from groundwater, benefits for downstream waterbodies; possibilities for reusing natural plant waste materials as a filling material for reactive barriers	Complicated management; short service time	Not provided.	Intermediate, TRL 7
Sediment and nutrient interceptors (NbS, CBS, Other)	Urban and agricultural catchments with high sediment transport; applicable to small and large lake inflows	Effective at reducing suspended solids and sediment-bound nutrients; Low footprint	May require frequent maintenance and sediment removal; effectiveness can vary with flow regime	Not provided.	Intermediate to high, TRL 5-9. From varying to mature technologies implemented on operational scales.

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Table 3 – Applicability and scalability potential of reviewed innovative solutions related to urban environments (continued, 2/2).

Solution	Applicability	Benefits	Disadvantages	Costs	Scalability potential
Stormwater basins (NbS, Other)	Urban and peri-urban catchments with high impervious surface cover and runoff volume; suitable for pre-treatment of stormwater before entering lakes or rivers	Reduces particulate-bound pollutants (e.g. heavy metals), total suspended solids, and in some cases nutrients; can enhance local biodiversity and public amenity if vegetated; scalable and adaptable to different land-use intensities	Efficiency can vary with rainfall intensity and season; requires regular sediment and vegetation maintenance; nutrient removal may be inconsistent; performance may decline without proper design or upkeep	Not provided.	High, TRL 8-9. Mature technology demonstrated on operational scales.
Wastewater treatment (NbS, CBS, Other)	Wastewater treatment plants	Lake protection from pollutants and nutrients; decreased external loading of nutrients; removal of specific pollutants; potential source of reusable materials, nutrients and energy	High energy and space demand; generated sludge needs special utilization (e.g. composting or burning) if treated with grey infrastructure	Not provided. Highly case- and pollutant-specific.	From low to high, TRL 1-9 depending on technologies

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3.2 In-lake solutions for advanced lake restoration

NRR sets out restoration targets for restoring freshwater ecosystems to improve at least 30% of listed habitat types (HD Annex I) to good condition by 2030. Here, we categorize in-lake measures as 1) engineering solutions targeted to physically, mechanically of chemically impact excess nutrients and symptoms of eutrophication; and 2) biological solutions to support ecological functioning of lake ecosystem. Additionally, a couple of in-lake measures with no proven efficiency were identified as measures to be avoided and are briefly described and grounded in Annex 1.

3.2.1 Engineering solutions

3.2.1.1 Aeration and oxygenation

Aeration of water in a lake is targeted to improve oxygen conditions in water column of temperature in stratified lakes. It can be run using two variants – with destratification (artificial mixing) where the whole water column is mixed or with protection of thermal stratification (hypolimnetic aeration) where only hypolimnetic waters are affected. Aeration of hypolimnetic water is a well-established restoration method that introduces oxygen or air into the deeper layers of stratified lakes (hypolimnion) often without disrupting thermal stratification (Cooke et al. 2005). It has widely been applied in deep stratifying lakes and basins, where seasonal or permanent stratification leads to oxygen depletion in the lower layers, triggering the release of phosphorus from sediments, which is a key driver of internal nutrient loading and eutrophication (Nürnberg 2024). By increasing the oxygen concentrations in the hypolimnion, aeration suppresses the internal redox dependent phosphorus release from the sediment, thereby potentially also limiting the nutrient availability for harmful algal blooms (HABs). In shallow lakes, which do not stratify or do so only weakly or temporarily, aeration is usually not needed although it has been used to ensure additional water circulation in ponds to reduce cyanobacterial blooms (Chmiel et al. 2024). The choice of specific aeration technology depends on lake depth and stratifying dynamics, but typically include air compressors or oxygen diffusers, piping systems and a power source (solar, wind or fuel-based). Reported installation costs range from approximately €4,000 per ha (Łopata et al. 2023) to €27,000 per ha (Chmiel et al. 2024) with annual maintenance costs of €150,000 – 200,000 (Łopata et al. 2023).

Multiple studies have demonstrated that aeration or oxygenation increases dissolved oxygen levels, and potentially reduces internal phosphorous loading and preventing the occurrence of HABs (Chmiel et al. 2024; Dixit et al. 2007; Dondajewska et al. 2019a; Łopata et al. 2023; Mehdizadeh et al. 2023; Osuch et al. 2020; Podsiadłowski et al. 2018; Shi et al. 2021). Multi-annual artificial mixing can also be effective in nitrogen removal from the lake, promoting coupled nitrification-denitrification processes in the water-sediment interface (Brzozowska and Gawrońska 2009; Grochowska et al. 2017; Liboriussen et al. 2009). Additional benefits include prevention of hypoxia leading to improved fish habitats (Mehdizadeh et al. 2023). Recovery of submerged macrophytes has also been found after applying aeration (Łopata et al. 2023).

However, although hypolimnetic oxygenation may prevent anoxia and reduces the hypolimnetic accumulation of phosphorus, long-term oxygenation is required. It is uncertain whether the overall lake water quality can be improved by oxygenation (Gächter and Wehrli 1998; Tammeorg et al. 2024) and it is still essential to reduce the external nutrient loading to improve lake water quality (Liboriussen et al. 2009). Indeed, several studies have demonstrated limited success of aeration/oxygenation on lake water quality management due to continued P release from shallow, non-aerated areas and sub-oxic sediments (Gächter and Wehrli 1998; Horppila et al. 2017; Tammeorg et al. 2017), and limited ability of the sediment to bind additional P even in oxic conditions (Gächter and Müller 2003).

While aeration can support aquatic biodiversity, barriers include high costs, energy requirements and technical complexity i.e. in deeper lakes stratification is a prerequisite for the success of aeration. Artificial aeration and increased mineralization of organic bound phosphorus in the sediment increases the risk of more labile inorganic bound phosphorus. Furthermore, the effectiveness of the restoration method is temporary, lasting only while the aeration system operates (McQueen et al. 1986). Aeration can therefore not be considered as a sustainable restoration method. But using renewable sources of energy (e.g., solar) can be alternative for classical energy sources. Using wind energy for aeration also was tested (Lossow et al. 1998), but the effectiveness is naturally limited during windless time.

3.2.1.2 Algaecides

Using algaecides is a well-established and commonly used measure to control algae blooms (Jančula and Maršálek 2011). All algaecides are developed with the specific aim of killing algae cells (Lürling and Mucci 2020). Generally, algaecides are applied in eutrophic lakes, reservoirs or ponds as short-term and rapid intervention. Although these acute measures do not solve the problem of eutrophication, they are relatively cheap, easy to apply and an effective way to protect humans rapidly against risks from massive cyanobacterial blooms.

Copper-based algaecides are extremely cheap (less than 4 € per kg of copper sulphate) and commonly used in the USA (Bishop et al. 2018), with an increasing number of copper-based products being developed nowadays, e.g. Cutrine® -Plus, Cutrine® Ultra Captain-XTR and others (Bishop et al. 2018; Jančula and Maršálek 2011; Kang et al. 2022b). Copper algaecide formulations differ significantly in terms of copper partitioning. Testing the new products in the lab and in situ before application is crucial. Copper leads to cell damage and the release of cyanobacterial toxins into water (Kang et al. 2022b) and can accumulate in sediments. Meanwhile, they may have negative effects on non-target organisms; novel copper-based products and traditional copper sulphate decreased the growth of the zooplankton grazer *Daphnia magna* (Kang et al. 2022a). Due to their non-specificity, copper-based products are no longer used in the Netherlands (Jančula and Maršálek 2011), but they are still acceptable in countries across the world.

Hydrogen peroxide is a strong oxidizing agent and leaves no traces in the environment (Jančula and Maršálek 2011; Matthijs et al. 2012). It has been applied at least 20 times to Dutch surface waters, varying from a small 0.2 ha pond to 100 ha lakes (https://www.stowa.nl/sites/default/files/assets/DIGITALE%20DIENSTEN/Beating%20the%20Blues/FSctrl-biomassa_verwijderen-waterstofperoxide_2019_NOL.pdf). Cyanobacteria have drastically declined after treatments, while eukaryotic phytoplankton and copepods seem much less sensitive to H_2O_2 (Piel et al. 2024). A starting concentration of 5 mg/L H_2O_2 has been suggested with a consequent 2 mg/L remaining concentration for at least 5 hours in the water column for successful algal control in natural environments (Matthijs et al. 2016). However, careful balance is required between a H_2O_2 dosage high enough to effectively suppress the cyanobacteria bloom and a sufficiently low H_2O_2 dosage to minimize impacts on non-target species, such as zooplankton (Weenink et al. 2022). Although hydrogen peroxide is registered as oxidising agent in the European Union (EC number 231-765-0), it is currently not registered as an algaecide. The advantage of a relatively rapid breakdown of H_2O_2 into water and oxygen is also a drawback, as it caps the effectiveness. To lengthen the duration of H_2O_2 release, novel formulations have been developed including liquids that consist of H_2O_2 and peroxyacetic acid and granular forms, such as calcium peroxide, and sodium carbonate peroxyhydrate formulations that more gradually release H2O2 (Calomeni et al. 2015; Geer et al. 2016; Sinha et al. 2018; Sukenik and Kaplan 2021; Trainic et al. 2021; Yun et al. 2024). The solid granular formulations have, besides an extended duration of H_2O_2 release, also advantages of safer storage, ease of transport and dispersion (Sinha et al. 2018).

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3.2.1.3 Hypolimnetic withdrawal and treatment systems

Hypolimnetic withdrawal (HW) is a well-established restoration method for eutrophied, thermally stratifying lakes that is based on diverting nutrient-rich water from the hypolimnion during stratification (Bormans et al. 2016; Dunalska et al. 2007; Nürnberg 2020). HW is traditionally implemented by passive syphoning, and it can lead to long-term reductions in lake nutrient concentrations (Dunalska et al. 2007; Nürnberg 1987). However, diversion downstream often causes eutrophication and nuisance problems in receiving waterbodies (Nürnberg 2020), in addition to which water levels can be affected and an environmental permit may be needed.

Hypolimnetic withdrawal and treatment systems (HWTS) represent a novel application of HW in which the withdrawn water is diverted to a nutrient-capturing purification system before diversion downstream (Łożyńska et al. 2021) or to the same lake (Nürnberg 2020; Silvonen et al. 2021; Silvonen et al. 2022; Silvonen et al. 2023; Tammeorg et al. 2024)(Figure 7). Similarly to traditional HW, HWTS takes advantage of hypolimnetic, anoxia-promoted internal P loading. By targeting legacy P, HWTS is expected to gradually decrease the P concentration in the lake with increased annual P output from the system (Silvonen et al. 2021).

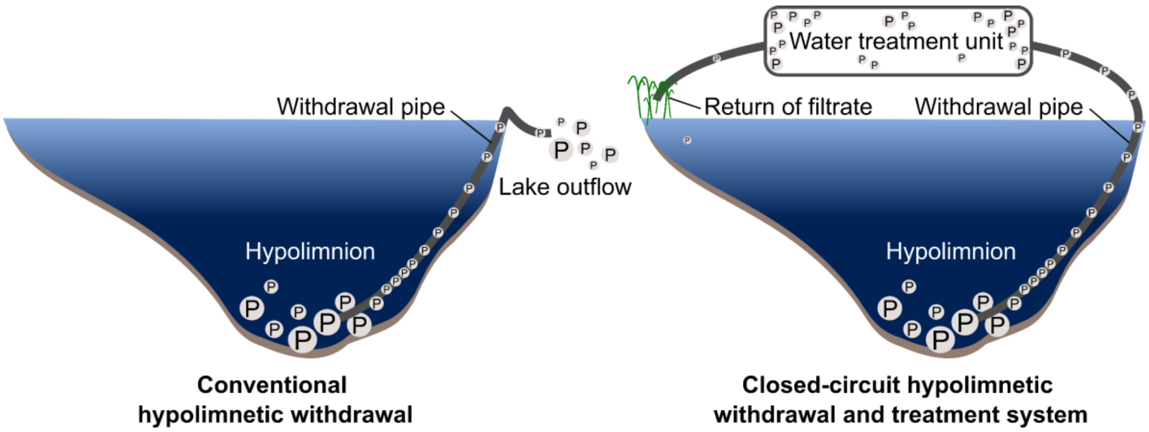

Figure 7 – Schematic presentation of the traditional hypolimnetic withdrawal and the closed-circuit hypolimnetic withdrawal and treatment system (HWTS). Modified from Nurminen & Härkönen 2024.

In a Finnish field-scale application (L. Kymijärvi/Myllypohja basin; 90 ha, 10.1 m max. depth; TRL 9), the withdrawn hypolimnetic water is first aerated, then treated in a simple 200-m^2 quartz sand filter (Figure 8) where P is precipitated by amorphous iron (Fe) oxides formed in the hypolimnetic water upon aeration (Silvonen et al. 2023) and the filtrate is diverted to a wetland trapping remaining nutrients and other elements before the treated water is returned to the lake epilimnion (Silvonen et al. 2022; Silvonen et al. 2023). The aeration-sand filter treatment has resulted in 91-95% removal of the dissolved (d)Fe and 71-91% removal of P-PO_4 (Silvonen et al. 2022). Addition of treatment chemicals such as $Ca(OH)_2$ and biopolymer have enhanced Fe flocculation, leading to more effective removal of dFe (99-100%) and P-PO_4 (88-95%) (Silvonen et al. 2022). Another, a 1-mo short-term Finnish field-scale trial (TRL 8) at Lake Linkullasjön (60 ha; 7.1 m max depth) utilizing movable water treatment technology (aeration/sand filter/micro-screen filter treatment with synthetic polymer) (Figure 8) resulted in an average of 63% dP reduction (Härkönen et al. 2024). In a Polish laboratory study by Łożyńska et al. (2021), filtration of hypolimnetic water through lightweight expanded clay aggregate (LECA) and crushed limestone was applied with an observed 50% dP reduction. In Sweden, closed-circuit hypolimnetic withdrawal is also being implemented on operational scale (TRL 9) at 6.6-km^2 L. Bornsjön, a backup drinking water supply for the City of Stockholm, where a HWTS has been established with a direct link to a drinking water treatment facility (Nürnberg 2020).

The potential for recycling the precipitate obtained with HWTS is a potential positive addition of the method (Łożyńska et al. 2021). The reject water could potentially be used for irrigation if e.g. there are

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no harmful substances present in the hypolimnetic water, possible flocculant used does not prevent the reuse in agriculture or the risk of Fe accumulation to irrigated fields can be accepted (Härkönen et al. 2024). However, HWTS may represent a rather inefficient means for harvesting P for reuse if concentrations in the treated water and the resulting precipitate remain low (Łożyńska et al. 2021; Silvonen et al. 2023). Additionally, the recovery of P requires its separation from the reactive material (Łożyńska et al. 2021) and the tendency of P to bind with Fe in the process weakens the bioavailability of P (Silvonen et al. 2023). Indeed, a fractionation study suggested that only a minor portion (12-14%) from the concentration of 4327 mg P/kg initially bound in L. Linkullasjön's Fe-precipitate was easily available for plants (Härkönen et al. 2024). Subsequently, it was concluded that the utilization of HWTS precipitate would not significantly increase the growth of plants and possible reuse potential is always to be site-specifically assessed.

Stratification is a prerequisite for the success of HWTS in general, due to which the method is limited to eutrophied, deep lakes. Additionally, the withdrawal rate needs to be adjusted with P diffusive fluxes from the sediment and to a level not compromising the stratification (Silvonen et al. 2021). Coupling of HWTS with a biotic system such as wetland (section 3.1.3.1) or a vegetated channel is recommended to improve purification efficiency (Härkönen et al. 2024; Łożyńska et al. 2021; Silvonen et al. 2023). This also reduces the risk of epilimnetic P increment caused by HWTS effluent, although the risk in general has been evaluated to be low (Silvonen et al. 2023).

The reported capital costs of establishing the Finnish HWTS have varied between €50,000–100,000 depending on the system applied (Härkönen et al. 2024). Evaluation of cost-effectiveness of Finnish applications is thus far not possible, as the long-term impacts of HWTS on lake scale have not been evaluated. No information on the costs of Polish or Swedish applications have been provided.

Figure 8 – Different HWTS tested in Finland on operational scales. In panel A), an established sand filter field on which aerated hypolimnetic water is being diverted for P precipitation, © Leena Nurminen; and B) movable pilot-scale water treatment technology HWTS combining aeration, sand filter and micro-screen filter, © Laura Härkönen.

3.2.1.4 Phosphorus inactivation

Phosphorus inactivation using different coagulants is a well-established method for reducing the symptoms of eutrophication and providing a potential emergency measure for inhibiting HABs. Different coagulants and their combinations have been tested for decades in laboratory, mesocosm and in-situ conditions. The efficiency of several coagulants, such as polyaluminium chloride (PAC), ferric chloride ($FeCl_3$), and Phoslock® (lanthanum-modified bentonite (LMB)) in rapidly reducing water column P and chl *a* concentrations together with reducing bioavailability of P in lake sediments have been proved on both laboratory and operational scales with both empirical studies, reviews and meta-analyses on chemical treatments available (Huser et al. 2016; Lin et al. 2021; Liu et al. 2022; Sarvala and Helminen 2023; Waajen et al. 2016; Zamparas and Zacharias 2014; Zamparas and Kyriakopoulos 2021).

The reviewed, recent advances in chemical precipitation treatments are related to advanced materials and approaches to support the lake recovery. For instance, LMBs with higher La content than the well-studied and widely applied LMB Phoslock (~ 5% La) have recently been developed, tested and commercialised (Wang et al. 2024; Zhang et al. 2024). The novel lanthanum-based compounds that have entered the market recently are Eutrosorb G (a LMB with 10% La, https://www.sepro.com/aquatics/eutrosorb-g), Eutrosorb WC (a liquid La formulation, https://eutrosorb.com/eutrosorb-wc.html), and the Zeofixer formulations, which are LMBs that come in 5%, 10%, 15% and 20% La variants (https://www.easternnode.com/). All of these compounds are being applied to lakes already (TRL 9) (Figure 9). Sequential application of different coagulants in the littoral and profundal zones with different oxygen regimes has recently been in the focus Polish studies (Grochowska et al. 2023). In this application, iron chloride (PIX 111) has first been introduced in the oxidised littoral areas and polyaluminium chloride (PAX 18) to profundal areas experiencing anoxia with an 80% reduction observed in water column P content over short-term (Grochowska et al. 2023). Another example of sequential application is the novel combination “Floc & Lock” using LMB first to inactivate water column P, PAC thereafter to remove phytoplankton and finally modified clay for sediment capping (see section 3.2.1.6). The combination has reduced sediment P release and decreased Chl *a* concentration at a 4-ha L. Rauwbraken (Netherlands) (Van Oosterhout and Lürling 2011). Based on 2 years pre- and 10 years post-treatment monitoring, the water quality in the lake improved strongly; total phosphorus was reduced from on average 134 µg/L to 14 µg/L, chlorophyll-a from 16.5 to 5.5 µg/L, and the contribution of cyanobacteria from 64% to 17% of Chl *a* (van Oosterhout et al. 2022).

Phosphorus inactivation may come with ecological risks and an environmental permit is country-specifically required for the intervention. For instance, a temporal collapse of crustacean zooplankton, such as *Daphnia*, spanning from weeks to few months could occur in whole-lake chemical treatments (Sarvala et al. 2020; Van Oosterhout and Lürling 2011), but zooplankton may also develop normally after treatment (Waajen et al. 2016). Agents such as aluminium salts that, upon hydrolysis in water, may lower pH should be applied carefully as they could cause fish kill (Sarvala and Helminen 2023) and products may temporarily colour (Sarvala and Helminen 2023) or turbidify the water (Van Oosterhout and Lürling 2011). The reproduction and larval development of mussels could also be affected by $FeCl_3$ (Drewek et al. 2022), and the growth and stoichiometry of *Chara hispida* disturbed by iron sulphate ($Fe_2(SO_4)_3$) (Rybak et al. 2020), whereas the charophyte establishment has not hampered by the introduction of $FeCl_3$ (Immers et al. 2013). LMB application, in turn, has in laboratory-studies decreased the total biomass and aboveground growth of *Myriophyllum spicatum*, while belowground biomass (e.g., root length and biomass) has increased suggesting that the reduced sediment P availability can alter plant resource allocation (Lin et al. 2021). However, a meta-analysis on whole-lake LMB applications did not find support for a negative effect on macrophytes, but, in contrast, found an increase in species number and colonisation depth, albeit lake-specific (Spears et al. 2016). Relatedly, several treatments of ponds and lakes that included LMB have resulted in higher macrophyte abundances (Berthelsen et al. 2024; van Oosterhout et al. 2022; Waajen et al. 2016).

The increased concentrations of dissolved P and N in sediments after cyanobacterial biomass flocculation could also pose a potential risk for increased internal nutrient loading (Liu et al. 2022). However, this could easily be overcome by adding a strong solid phase P binder (e.g., LMB) to the flocculant, a procedure coined 'Floc & Lock' (Lürling et al. 2020a), or by better planning of the intervention, i.e., in winter when phytoplankton biomass in lakes in temperate regions is usually low. Indeed, in cases where potential negative environmental consequences can be overcome, P inactivation could be considered as a BfS potentially providing longer-term ecosystem-level benefits via e.g. re-establishment of natural flora (van Oosterhout et al. 2022).

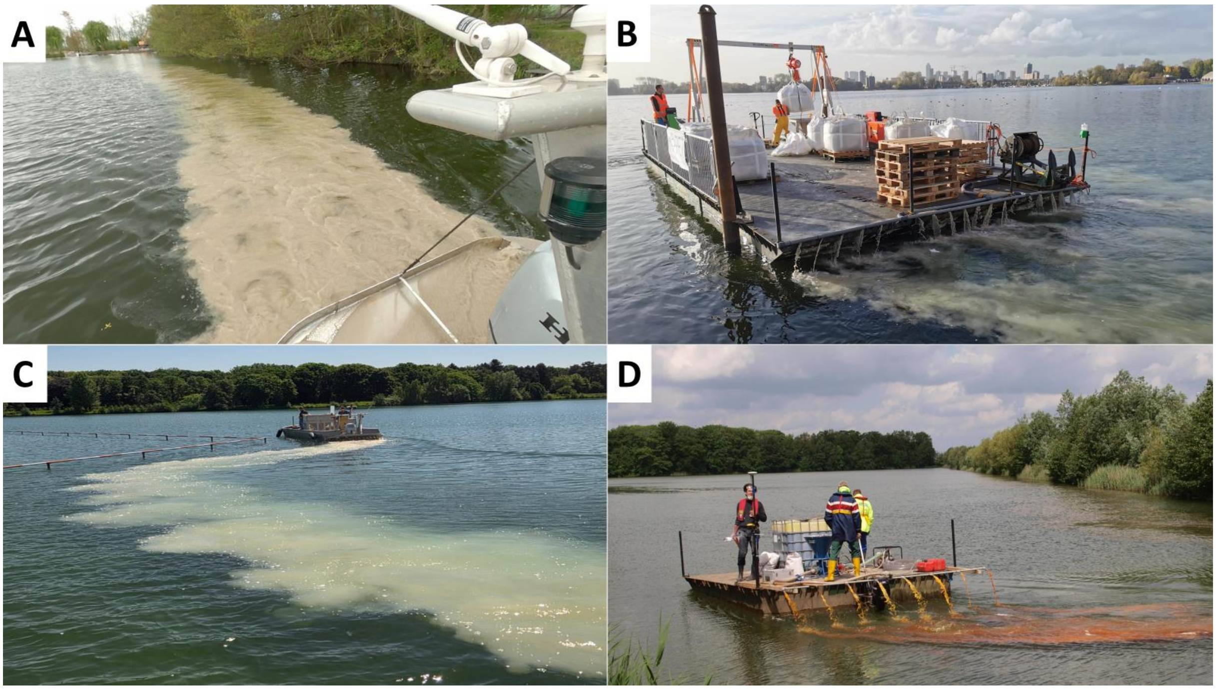

Figure 9 – Application of the novel LMB Limnoplus (Zeofixer Origin, 10% La) to Lake Alonco (Belgium, 15-04-2025; panel A © Paul Knights, application of the LMB Phoslock (5% La) to Kralingse Plas (The Netherlands, 08-11-2021; panel B © Miquel Lürling) and to Lake De Ster (Belgium, 13-05-2025 © Miquel Lürling), and application of iron(III)chloride to Lake De Kuil (The Netherlands, 18-05-2009, © Guido Waajen.

In general, the result longevity of e.g. aluminium salt treatments in terms of water column P reduction has spanned from 0-45 yr depending on the doses and inherent characteristics of the lake treated (Egemose et al. 2010). La in LMB will precipitate with phosphate forming a stable mineral rendering P no longer bioavailable (Copetti et al. 2016), which is strongly supported by increased La-P pools in LMB treated sediments (e.g., Kang et al. 2023; Meis et al. 2013; Yin et al. 2021) and formation of La-P minerals in sediments of LMB treated lakes (Dithmer et al. 2016b). Result longevity varies depending on the chemical used (Egemose et al. 2010), the dose applied, lake characteristics, and on the efficiency of external load control.

In studies reporting the costs of chemical treatments, purchase cost per hectare have varied between €1,000-1,900 depending on the doses and chemicals used (Dunalska et al. 2018; Grochowska et al. 2023; Sarvala et al. 2020). The doses of flocculants used are always determined site-specifically. High rates of algal flocculation may inhibit the performance of P inactivation agents, subsequently impacting the required chemical doses (Kong et al. 2022). High alkalinity together with humic substances also interfere with the P binding capacity of LMBs in the short-term (Dithmer et al. 2016a; Funes et al. 2018), but do not affect the dose-calculation and therewith the costs. The treatment costs of 4-ha Lake Rauwbraken (April 2008) and 7-ha Lake De Kuil (May 2009) were €50,000 and €140,000, respectively, and included both materials used as well as application. Longevity, however, varied; while Lake Rauwbraken remains devoid of cyanobacteria blooms (with longevity of at least 17 years), Lake De Kuil relapsed within 7 years. This boils down to annual costs of ~ €750 per hectare for Lake Rauwbraken and of ~ €3000 per hectare for Lake De Kuil.

Recently, recycled industrial side-stream materials such as drinking water treatment residue (DWTR) which is a waste product derived from coagulation using an aluminium (Al) and/or Fe salt in drinking water preparation (Kuster et al. 2023), and/or use of spent lime , have been investigated to reduce costs and the need for raw materials (Kuster et al. 2021; Wang et al. 2016). DWTR has potential for P inactivation and algal flocculation (Kuster et al. 2021; Wang et al. 2016) and could be a promising technology for the control of internal P loading in eutrophic systems (Xia et al. 2023), but the efficacy of Fe-rich DWTR may be affected by low redox conditions near the sediment (Zhan et al. 2022). DWTR also requires pre-treatment due to potential risk for e.g. heavy metal leaching (Kuster et al. 2021). The different compositions and properties of DWTR obtained from various water treatment plants imply variability in recycling potential. Hence, guidelines should that include risk assessment, sorption capacity, and environmental application procedures have been proposed to aid water managers (Wang 2025). The TRL of side-streams is still low (3) and feasibility in natural environments requires further investigation.

3.2.1.5 Phosphorus adsorbents

Phosphorus adsorbents to recover P from lake water column have recently received attention as a modification of chemical treatments allowing for more permanent removal of P, potential for P reuse, elimination of undesirable effects of sorption material on the aquatic environment and reducing the risk of desorption (Pryputniewicz-Flis et al. 2021).

Magnetic nano- or micron-sized particles (MPs) with magnetite or Fe are used to adsorb P from aqueous solutions, after which P loaded MPs can be separated from the solution by sodium hydroxide (NaOH) washing or using high gradient magnetic separation process and subsequent desorption of P providing potential for its reuse (Álvarez-Manzaneda Salcedo et al. 2016; de Vicente et al. 2010; Funes et al. 2018). The use and separation of different magnetic microparticles (carbonyl iron (800-900 nm), magnetite (90 nm), hydrous lanthanum oxide loaded silica-coated magnetite (Fe-Si-La) and commercial zero-valent iron particles (FeHQ)) have been tested under laboratory conditions and microcosms (TRL 3-5) to assess their P adsorption capacity (de Vicente et al. 2010; Funes et al. 2016; Funes et al. 2018; Merino-Martos 2014). The adsorption capacity of magnetite appears somewhat higher than of carbonyl iron (de Vicente et al. 2010; Merino-Martos 2014) but the adsorption efficiency of MPs seems in general high (exceeds 80%) (Merino-Martos 2014). However, it decreases with increasing pH and increased concentration of other salts or dissolved organic carbon (de Vicente et al. 2011; Funes et al. 2018; Merino-Martos 2014). Additionally, mesocosm experiments have indicated that Fe-based MPs are efficient under oxic conditions whereas Fe can expectedly be released to the water under anoxia (Funes et al. 2016). MPs can be reused after P separation but the reuse reduces the adsorption capacity by 20% (Merino-Martos 2014). Treating the surface of MPs with amino silane groups may counteract magnetic and van der Waals attractive interactions and promote kinetic stability (de Vicente et al. 2010).

Some short-term laboratory and mesocosm studies have suggested only negligible or minor lethal and sublethal impacts of MPs on zooplankton (Álvarez-Manzaneda et al. 2019; Álvarez-Manzaneda and De Vicente 2017; Alvarez-Manzaneda et al. 2019) but the evidence is mixed and MPs removal especially may cause drastic effects on zooplankton abundance if MPs are e.g. ingested or attached to plankton (Del Arco et al. 2018). To counteract potential toxicity, preparation of magnetic chitosan microspheres (MCMs) using a reverse-phase suspension cross-linking technique has also been studied (Funes et al. 2018). Their P adsorption capacity in laboratory scale studies has been 4.84 mg g^{-1} (Funes et al. 2018).

Other applications studied in laboratory scales (TRL 3-5) are ground calcium carbonate ($CaCO_3$, GCC) that could be used as laminates, cassettes or bags for P adsorbing units (Pryputniewicz-Flis et al. 2021), nano magnesium oxides (MgO) (Xia et al. 2020), La-modified adsorbents using solid waste coal fly ash (La-FA) (Xu et al. 2022), water-permeable nonwovens as P sorbent carriers (Burska et al. 2019) and composite materials such as La hydrogels that remove both phosphate and nitrate (Chen et al. 2025) or aluminium hydroxide-coated nanoscale zero-valent iron that adsorbs phosphate which subsequently can be harvested as hydroxy-apatite, while nanoscale zero-valent iron and dissolved aluminium oxides (Al_2O_{3-}) can be recovered for recycling (Tan et al. 2025). Additionally, Zamparas et al. (2020) tested combining LMB and Fe-modified bentonite (f-MB; Bephos™) for P adsorption under laboratory conditions. However, the latter are not applicable to field-scale due to unrealistic material demands in production and unsuitability of LMB as a filtering agent. First, the preparation of Bephos™

requires both chemicals, excessive amounts of water and energy for heating and freeze-drying, which makes the production process commercially unfeasible. Second, using LMB in a bag would either clog the bag, or leach out causing increment in turbidity if upscaled from laboratory- to field-scale, loss of material to harvest, and have large diffusion issues inside the bag limiting its P binding capacity.

The upscaling potential of P adsorbing agents in general remains understudied with earlier research mainly focused on laboratory scales. However, an upscaling trial in the Netherlands with big-bags of iron-rich sand yielded far less P adsorption than was predicted from small-scale lab trials, due to the strongly hampered diffusion from the outer thin layer into the material (Koomen et al. 2023).

3.2.1.6 Sediment capping

Sediment capping inserts an artificial barrier between the sediment and the water which aims to prevent or minimize internal loading and diffusion of harmful substances from the sediment into the overlying lake water. Several capping materials have been used or piloted. Both passive (e.g. sand, silt, clay) and active capping (e.g. calcite, activated carbon, biochar) can be used over sediments (Gibbs and Hickey 2018; Jersak et al. 2016; Pan et al. 2012; Vlassopoulos et al. 2017) with some of the materials considered as CBS.

Batch, mesocosm and lake level experiments have shown a 54-99 % nutrient reduction during 10–300 days of capping (Sunaryani et al. 2023). A multi-layer amended cap using sand with activated carbon for chemical isolation and siderite for pH control has been used in Onondaga Lake, USA. This cap blocks contaminants, fixes high pH levels, and helps restore the lake's habitat (Vlassopoulos et al. 2017). The lifespan of this cap system may exceed 1,000 years, but it has high initial cost and technical expertise requirements for design, modelling, and construction.

In Lake Taihu and other shallow lakes in China, sediment capping has been used to reduce the recruitment of sediment algae and harmful cyanobacteria HABs and eutrophication in natural shallow waters (Pan et al. 2012). A chitosan modified local soil/sand suspension (MLS-IER) has been sprayed to the cyanobacteria bloom water to flocculate and sink algal blooms together with their excessive nutrients into the sediment, with the aim to subsequently convert the algae biomass into submerged vegetations in shallow lakes (Pan et al. 2012).

A novel salt cap closure technology has been developed for contaminated wastewater evaporation ponds at a brine mining operation near Great Salt Lake, USA (Lundmark et al. 2023). Salt cap formation via controlled brine evaporation creates a natural barrier against sediment contamination. Pilot testing over four years demonstrated that brine evaporation from solar ponds precipitated sodium chloride (NaCl), forming a protective salt crust over waste sediments. Slow implementation timeline (10–19 years) limits short-term benefits. The method is applicable to other brine mining and salt-rich industrial sites, but it is vulnerable to seasonal precipitation events, and requires hyper-saline conditions, limiting broader application.

3.2.1.7 Sediment removal and reuse

Sediment removal is a well-established method that has potential for providing long-term water quality improvements by directly targeting sediment legacy nutrients and other pollutants (Cooke et al. 2005; Härkönen et al. 2025; Lürling et al. 2020b; Peterson 1982; Peterson 1983; Pierce 1970). A recent global meta-analysis underpinned dredging is a powerful mitigation measure that can achieve long-term effectiveness on ecological restoration, as sediment removal is effective in decreasing pollutants, mitigating algal blooms, and minimising sediment nutrient fluxes (Wan et al. 2025). However, sediment removal is also associated with temporal environmental risks such as harmful water quality impacts, resuspension of legacy contaminants, and disturbances on flora and fauna (e.g., Abell et al. 2022; Bormans et al. 2016).

Due to high costs, application of sediment removal is perhaps more common in relatively small lakes with high recreational or environmental value (Björk et al. 2010), but it is also conducted to improve water quality in larger lakes with few millions of m^3 of sediment removed (Lürling et al. 2020b). For instance, up to 42 million m^3 sediment was removed in Zhushan Bay and Meiliang Bay (Lake Taihu, China) equalling a removal of nutrient inputs of around 20 years (Zhang et al. 2023). In Europe, around 300 million tons of freshwater dredged sediments are generated annually (Fořt et al. 2025) although it is unclear how much is being produced from water quality improvement projects (Lürling et al. 2020b). Management of these dredged sediments is a multifaceted process that involves careful consideration of environmental, regulatory, and economic factors to ensure sustainable and responsible handling (Fořt et al. 2025).

Sediment removal can be considered as a CBS as it provides potential for the reuse of both mass and P recovered. Sediment dredged from small waters such as ditches can be deposited on the adjacent pastures (Harmsen et al. 2012). Dredged sediment has also been reused for e.g. island creation or dike construction, floodplain restoration and agricultural soil enhancement (Marlin 1999; Sittoni et al. 2019). At Hickling Broad (UK), 20,000 m^3 of sediment was removed to reduce internal loading and increase the lake depth for recreation. Subsequently, the dredged sediment were used in construction of reed beds to create an artificial lagoon to support waterfowl (Figure 10) (D. Hoare, pers. comm. April 2025, see also Phillips et al. (2016)). In this application, geotextile tubes were filled with the dredged sediment and used for the construction of the artificial reefs that could also allow for avoiding possible CO_2 emissions from the drying sediment (D. Hoare, pers. comm. April 2025). However, installation of geotubes in the lake may potentially pose a risk for the spread of nanoplastics and and environmental permit was required (D. Hoare, pers. comm. April 2025). Hence, alternative, more sustainable materials than polypropylene should be preferred in such applications. For instance, jute as an alternative material for geotextile tubes has been investigated (Ghosh et al. 2014; Kiffle et al. 2017).

An important aspect of sediment removal is securing the lake from nutrient rich water leakage from dredged sediments when and if handled and stored on the lake shoreline. Improper handling and storage may result in failure of the intervention. In FutureLakes' Kartuzy Demo Site, an innovative hydrotransport pipeline system was designed for the diversion of extracted sediments to a special technological line located in Kartuzy wastewater treament plant (J. Grochowska, pers. comm, May 2025) (Figure 11). Usage of geotubes with flocculating agents also provides possibilities for handling and dewatering dredged sediment on-site (Simoni et al. 2024).

Figure 10 – Artificial reed beds and a central lagoon after construction at Hickling Broad, UK. Dredged sediment was diverted into geotextile tubes that were used in the construction of the artificial reed beds. © Mike Page

Dredged 29,000 m^3 of sediment from the lake restoration project of Lake Groote Melanen (Lürling et al. 2024)) was processed on site; sand was separated from the fine silt and dewatered yielding reusable materials where sand was used in construction and the dewatered silt as soil amendment in agriculture. The addition of sediment to the soil has potential to increase crop Chl *a* content, photosynthetic rate and leave growth with biomass production of the same order as with chemical fertilisers (Braga et al. 2024). Laboratory studies have indicated that fresh sediment has P fertilizer potential with amorphous Fe-P as a significant contributor, and fertilization with fresh sediment and Fe-P can increase soil adsorptive capacities potentially reducing P leaching from soils but also creating dependency of plant P bioavailability on plant-soil interactive mechanisms (Haasler et al. 2024). However, sediments tend to accumulate not only phosphorus and organic matter, but also microcystins and pollutants, such as pesticides, industrial chemicals, and heavy metals (typically cadmium (Cd), Cu, or Zn) of which bioaccumulation to plants and subsequent introduction into the food chain is of great concern and may limit the suitability of sediments for agricultural reuse (Cao et al. 2024; Fořt et al. 2025; Haasler et al. 2024). Validation of sediment reuse in agriculture on operational scales is needed with the TRL still remaining low (4). However, accumulated contaminants may still allow reuse in building material (Fořt et al. 2025).

Treatment of dredged sediments with composite materials such as Fe-biochar, clinoptilolite, wheat straw, and nitrification indicator dimethylpyrazole phosphate (DMPP) have recently been investigated in laboratory conditions to improve reuse potential of contaminated sediments in greenfield and agricultural land (Huo et al. 2024). Bioremediation, which is a strategy capitalizing the ability of e.g. aquatic macrophytes (Newete and Byrne 2016) or other trapping agents such as nanodiamonds (Yotinov et al. 2022) to sequester pollutants from water column, sediments and contaminated soils could also be effective in reducing organic pollutants but may take years before levels have reduced below safety thresholds (Harmsen and Rietra 2018). For instance, the effectiveness of *Cannabis Sativa* (var. Carmagnola) in the phytoextraction and/or phytostabilisation of heavy metals in contaminated soils and subsequent possibility of valorising the contaminated biomass into biochar has been assessed in laboratory studies with a high uptake of especially Ni and Cu to plant biomass (G. Picchi, pers. comm. April 2025). However, in regards of using *Cannabis* sp. as a bioremediating agent, there may be country-specific legislative barriers regulating its cultivation (G. Picchi, pers. comm. April 2025). Additionally, the biogeochemistry and potential toxicology of bioremediating agents must be fully understood and overstatement of practical value avoided before wide application to lake restoration (Zhang 2012). Also, it must be stressed that while cultivation of e.g. water hyacinth (*Eichhornia crassipes*) as a biological agent for removing excess nutrients from eutrophic waterbodies has been recently studied (Thanappan 2016; Yan et al. 2017), utilization of invasive species is not sustainable and should not be considered as bioremediating agents for natural lakes to avoid ecological and socio-economic harm by uncontrolled invasive species outbreaks.

Country-specific regulations can affect the scalability of both sediment removal, and its reuse as environmental permits may often be needed, and deposition of sediment can be restricted. Dredging also incurs relatively high costs; for example, in the Netherlands, dredging costs vary from a few euros per m^3 to up to €50 per m^3 (https://edepot.wur.nl/400395); and in Finland around €30 per m^3 if all the costs of permitting, dredging, monitoring, and disposal are being considered (Härkönen et al. 2025). For the 4-ha L. Rauwbraken and 7-ha L. de Kuil (presented in section 3.2.1.4), dredging was not considered an alternative option to P inactivation due to high costs (~ €500,000 for Lake Rauwbraken and ~ €1,200,000 for Lake De Kuil). Potential environmental risks associated with sediment removal are also to be overcome to improve the methods feasibility and scalability, for which there are a number of recent projects developing more environmentally friendly and automated implementation (https://repairprojektet.dk/, https://www.richwaters.se/category/en/, https://kalmar.se/bygga-bo-och-miljo/life-sure.html). Automated implementation also has potential for reducing the costs of sediment removal as is shown by the case of 10-ha L. Ormstrup (Denmark)(Figure 12), where the implementation costs with an autonomous dredger developed in RePAIR-project have gone remarkably down to approx. €10.000/ha (O. Wolff, pers. comm. May 2025, https://repairprojektet.dk/). These estimated costs are to be qualified by dredging of three other Danish lakes with different sizes during autumn/winter 2025/2026 (O. Wolff, pers. comm. May 2025).

Figure 11 – Dredging at FutureLakes' Kartuzy Demo Site and treatment of sediment at the Kartuzy Wastewater Treatment Plant. *The sediment from Karczemne Lake was withdrawn by a dredger and transported directly to Kartuzy WWTP via pressure pipeline (length 4.5 km) in order to avoid leaks from the sediment storage on the shore of Karczemne Lake. Working sector in the lake was separated from the rest of the lake by geomembrane curtains. Kartuzy WWTP constructed full technological line for lake sediment treatment (several sedimentation tanks, screen grit chamber, centrifugation station and storage shed for final product of lake sediment treatment. © Kartuzy WWTP and Mike Lürling*

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Figure 12 – Dredging at FutureLakes' Innovation Site Lake Ormstrup, Denmark and reuse of sediment on agricultural areas. A: location of Lake Ormstrup, B: satellite image of Lake Ormstrup where the dredger can be seen in upper left corner of the lake, C: the dredger, D: geotubes where the sediment is left for about ½ year for dewatering, F: pipe for sediment from dredger to land, F: removal and reuse of dried sediment from geotubes. © Martin Søndergaard

3.2.2 Biological solutions

3.2.2.1 Algal harvesting

Effective utilization of harmful algal biomass from eutrophic lakes can be viewed as a potential circular blue economy solution (Macário et al. 2023). Algal harvesting is the way to separate or detach algae from their growth medium or natural water body. Algae have economic importance and many benefits and can be the source of food for animals or humans, antibiotics and medicines, biofuel, fertilizer and purifiers of wastewater (Singh and Patidar 2018; Vijayakumar and Menakha 2015). Surface bloom-forming cyanobacteria are low-quality food for zooplankton due to their low content of essential biomolecules (Taipale et al. 2019), that could implicate low nutritional value for human too. Nevertheless, harmful algal biomass (*Microcystis aeruginosa*) can serve as an alternative substrate for microbial fuel cells (MFC) for bioelectricity generation and waste treatment (Ali et al. 2020) and has the potential for sourcing of higher value products.

The term 'algal harvesting' appears mostly related to industrial wastewater or drinking water treatment and as a method of controlling external nutrients. The reason is that industries work within closed systems, where conditions can be manipulated to optimize harvesting efficiency (Pandhal et al. 2017). Sedimentation and flotation, including dissolved air flotation (DAF), are among the most cost-effective harvesting techniques and are considered the most cost-effective harvesting methods across many industries (Srinivasan et al. 2011). However, the energy costs are relatively high in eutrophic lakes or retention ponds, making these applications harder. As an alternative, a combination of chitosan-induced flocculation and efficient flotation of Algae Technology (eFLOAT) has the potential to harvest microalgal biomass and remove P from eutrophic water systems (Pandhal et al. 2017).

Microcystin removal using recycled-membrane biofilm reactors (R-MBfR) that leftovers from desalination by reverse osmosis have also been tested in laboratory-scale as a circular solution potentially suitable for treating surface water for e.g. irrigation purposes (Morón-López and Molina 2020).

Currently, algae harvesting includes mechanical, chemical, and biological methods. Mechanical methods are considered the most reliable and commonly used methods for harvesting microalgal biomass (Grima et al. 2003). For example, a three-level system combined with soil filtration was able to remove 99% of cyanobacterial cells, toxins, and persistent organic pollutants from Lake Taihu water samples (Chen et al. 2017). Additionally, lake water can be subjected to micro-sieving as a purity improvement method with 93-100% removal efficiency, which could lead to a decrease in internal P loading. It also should be noted that micro-sieving for lake restoration should ensure high hydraulic capacity (Napiórkowska-Krzebietke and Łuczyński 2022). The integration of coagulation and flocculation with mechanical methods can further enhance harvesting efficiency while lowering maintenance costs (Singh and Patidar 2018). Harvesting biomass from cyanobacterial blooms can be used for the retrieval of high-value compounds (Prabha et al. 2022), where some of the revenues generated from the extracted high-value products can be used for measures to mitigate cyanobacterial blooms (Macario et al. 2021). Overall, although algal harvesting should also be listed in the lake restoration toolbox, the associated high energy costs and the complexity of the set-up system on site should not be ignored.

3.2.2.2 Artificial reefs

Artificial reefs and substrate enhancement are increasingly used as ecological restoration methods to rehabilitate degraded freshwater systems, particularly lakes and reservoirs to reduce wave erosion and support local biodiversity. While the deployment of artificial reefs has been a common practice in marine environments since the 17th century, (originating from Japanese fishing practices that recognized fish aggregated around natural structures such as rocks and submerged branches (Bolding et al. 2004; Yamamoto et al. 2014), their application in freshwater ecosystems is more recent. One of the first examples was Lake Constance in Germany where artificial reefs were built to reduce wave erosion and help support re-development of reedbeds (Ostendorp 2008). Floating artificial rafts are also frequently applied in waterfowl lakes to support the nesting of birds (e.g., Nummi et al. 2013) (Figure 13).

Figure 13. Common Terns (Sterna hirundo) on a floating artificial raft at a shallow Lake Kanteleenjärvi, Finland. © William Velmala

Artificial reefs aim to restore habitat complexity, enhance biodiversity, and support the recovery of both benthic and pelagic communities, moving beyond fisheries enhancement to broader ecological restoration goals. Artificial reefs can be constructed from various materials, including concrete modules, rocks, and recycled structures (e.g., tires or plastics), designed to mimic natural habitats and provide shelter, feeding, and breeding sites for fish and invertebrates (Figure 14). In reservoirs with high water-level fluctuations, artificial macrophytes have proven effective in improving fish richness and abundance (Santos et al. 2011). FTWs (Section 3.1.3.2) could also be considered as a type of artificial reefs supporting local biodiversity if applied in lakes and could be suitable for both shallow and deep-water bodies or urban settings where traditional rooted vegetation cannot thrive. The use of artificial reefs in general is particularly applicable in artificial reservoirs (where water level fluctuates) or eutrophic water bodies where natural substrate structure and aquatic vegetation have been lost due to shoreline modification, sedimentation, or nutrient enrichment.

Recent innovations focus on using eco-friendly materials, such as biomass-derived fillers from power generation byproducts (Zhu et al. 2021) or natural wood (Yamamoto et al. 2014). Studies have shown that such artificial reefs not only increase fish species richness but also can reduce phytoplankton biomass and promote a shift in species composition from cyanobacteria to chlorophytes (Zhu et al. 2021) (Yamamoto et al. 2014). Despite the promising results, long-term ecological impacts of these interventions remain understudied.

Figure 14 – Different examples of artificial reefs from the Netherlands.

3.2.2.3 Fish manipulation

Removal of either planktivorous or benthivorous fish, or both, from lakes (i.e., biomanipulation) is a well-established, nature-based solution to increase water transparency by increasing the top-down control of phytoplankton by zooplankton grazing and reducing sediment resuspension (Hansson et al. 1998; Triest et al. 2016). The method is particularly suitable for the restoration of relatively shallow lakes with high internal P loading (Benndorf et al. 2002; Ventelä and Lathrop 2005) but has also been successfully implemented in larger, deeper lakes such as FutureLakes' 109-km^2 Finnish Demo Site L. Vesijärvi (Anttila et al. 2013; Salonen et al. 2020b).

Figure 15 – Carp removal from Lumen Pond, Wageningen (The Netherlands, April 12th 2018) © Miquel Lürling

Biomanipulation is based on the assumption that the removal of planktivorous fish (e.g. roach) increases the abundance of filter-feeding cladoceran zooplankton (top-down control), such as large-bodied *Daphnia* species efficient in grazing phytoplankton (e.g., Urrutia-Cordero et al. 2016). Reduction of benthivorous fish (e.g. bream and carp), in turn, whose feeding behaviour resuspend fine sediment and P into the water column subsequently increasing lake water turbidity, and potentially increasing internal P load (bottom-up control). With the fish, significant amounts of P should also be removed from the water body (Jeppesen et al. 2012). Usually, the caught piscivorous fish are returned into the lake and biomanipulation can be accompanied with stocking of piscivorous fish fry to support their recruitment (Anttila et al. 2013). Different fish passage structures have also been developed in an attempt to support the reproduction of migratory fish by restoring the connection within the hydrological continuum and improving longitudinal continuity (e.g. https://theafsluitdijk.com/projects/fishmigrationriver/), and also to selectively manage invasive species (French et al. 1999).

Experiences have shown that lake water quality can respond to biomanipulation within a few years of intensive execution. For instance, in L. Vesijärvi, the water column TP content dropped within three years after the initial start of intensive fish removal (*c*. 50-90 kg fish/ha/yr removed) and has remained at lower levels ever since as a smaller-scale biomanipulation (*c*. 20-40 kg fish/ha/yr) has continued (Salonen et al. 2020a). Indeed, the initial, intense fish removal should be conducted rapidly, within 1-3 years (Hansson et al. 1998; Søndergaard et al. 2007) after which continued efforts are often needed. The prerequisite of the success of the method is a sufficient reduction of planktivorous fish. Based on a Danish experience, it has been recommended that at least 80% of the planktivorous fish stock is removed (Søndergaard et al. 2000). Longevity of the results is naturally also dependent on sufficient reduction of both external and internal loads that have occasionally prevented regime shifts in a restored lake despite of biomanipulation (Qin et al. 2019).

Most often the main goal of biomanipulation is to reduce proliferation of HABs and improve the water quality from human perspective. However, biomanipulation by fish removal can also be considered as a BfS providing means for improving the ecosystem conditions for breeding waterfowl communities (Fox et al. 2020) and to restore natural fish populations (Perrin et al. 2006). Recent advances in biomanipulation also include a link to CBS, as the recovered fish biomass can potentially be reused as e.g., food, feed or biogas production (Tammeorg et al. 2024). In Finland, various food products for human consumption have been developed from the removed, formerly underutilized cyprinid biomass, such as minced roach meet and various canned products (Figure 16). Commercial biomanipulation also provides possibilities for cost reimbursement, if part of the costs can be compensated by the reuse of catch. In a 150-ha Finnish Lake Littoistenjärvi, the reported costs for biomanipulation have been c. €10,000 per year (Sarvala et al. 2020). However, the costs of biomanipulation vary greatly and depend e.g., on the fishing method, the size of the lake, the duration of biomanipulation, and whether professional fishermen or volunteers are used.

Figure 16 – Canned, smoked roach from FutureLakes' Demo Site L. Vesijärvi, Finland, where c. 20% of the annual catch from biomanipulation is being produced for food. © Kristiina Vuorio

3.2.2.4 Littoral and shoreline protection and habitat creation

The primary cause of declining worldwide biodiversity is habitat loss in freshwater ecosystems. Lake habitats are complex and three major habitats are recognized -- the littoral zone, the pelagic and the benthic (Meerhoff and de los Ángeles González-Sagrario 2022). Habitat restoration can return degraded habitats to their pristine conditions (Carroll 1994). As the diversity of habitat types within lakes, the habitat restoration is inherently complex.

In freshwater ecosystems, the littoral zone is the key affected habitat type because of shoreline development. As shoreline development is expected to intensify in the future, restoring the littoral zone is increasingly important. Littoral zone is a highly productive and species-rich habitat type with gradual land-water transitions, where vegetation establishes, and nutrients between terrestrial and aquatic systems exchange; it increases aquatic food web functioning (Meerhoff and de los Ángeles González-Sagrario 2022).

A novel CBS and BfS restoration approach in the Netherlands was adopted in one of the FutureLakes' Demo Site, a 70,000-ha turbid lake Markermeer, lacking littoral habitat (Jin 2021; Stouten et al. 2022; van Leeuwen et al. 2023; van Leeuwen et al. 2021). The project created a 1000 ha archipelago 'Marker Wadden', consisting of five islands constructed from local sediments, such as sand dikes, minimal riprap fortification on the most exposed side, and several basins filled with fine nutrient-rich clays. Water depths between the islands were reduced from the lake's original 4 m to 1 ~ 2 m, creating a gradual transition towards the shorelines. The idea is to create new littoral zones and stimulate bottom-up trophic levels in aquatic food webs simultaneously, including vegetation, zooplankton and macroinvertebrates and young fish. This project has successfully enhanced natural processes and attracted birds and fish, without conflicting with existing ecosystem services. Total costs for Marker Wadden's habitat creation were estimated up to M€78.

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Among seven German lowland lakes with natural shorelines, retaining walls, ripraps and recreational beaches, macroinvertebrate density increased with increasing proportion of developed shoreline (Brauns et al. 2007). The study concluded that water management should prioritize conserving littoral habitat complexity and heterogeneity (Brauns et al. 2007). Similarly, reconstructed Lake Karla in Greece is also considered a vital multiple-use aquatic ecosystem in terms of improving biodiversity and conserving natural habitats. The reconstruction included three man-made islands and a shallow wetland area of 0.45 km^2 for bird nesting and the reproduction of fish (Panagopoulos and Dimitriou 2020). In Lake Erie, a semi-open structure of embankments in the shoreline zone was proposed to use sediments from deepening mouth sections of rivers to create in-lake features that support biodiversity and recreation (Theresa RUSWICK 2021). According to Comoss et al. (2002), protection against shoreline erosion can help to establish of vegetated riparian area, for which a combination of natural and "engineered" erosion protection measures can be used. These measures were implemented In Presque Isle State Park on the shore of Lake Erie. Natural shoreline erosion protection was made using geotextiles, wattles of woody branches, groins of downed trees. Indigenous riparian and wetland plants were planted in addition to using dewatered dredged sand and stone ripraps. Also, mechanical and chemical removal of invasive plant species was made. The cost of implementation was 33,000 USD (~30,000 €) (Comoss et al. 2002).

In summary, creating habitats through lake restoration is a crucial strategy for biodiversity conservation. Approaches such as building islands and wetlands to protect littoral zones are widely used. However, the high cost of these interventions highlights the importance of evaluating their cost-effectiveness to ensure sustainable restoration practices.

3.2.2.5 Macrophyte manipulation

Macrophyte manipulation involves either the replanting of native aquatic vegetation or harvesting of excessive plant biomass, which is a restoration method aimed to improve ecological function in degraded lakes suffering from macrophyte overgrowth.

During lake eutrophication, submerged macrophytes often more or less disappear or the original species are replaced by more nutrient tolerant species (Sand-Jensen et al. 2017; Søndergaard et al. 2022). Macrophytes have an important role for both lake ecosystem structure and functioning, as their presence stabilizes the sediment, nutrient cycling, improves water clarity and provides habitat and food for other aquatic organisms (Jeppesen et al. 1998; Søndergaard et al. 1998; Søndergaard and Moss 1998). Macrophyte replanting is used to reestablish lost submerged vegetation, restore habitat complexity and improve ecological stability (water clarity and enhance biodiversity), while harvesting aims to manage dense plants that contribute to oxygen depletion (e.g. floating plants as water hyacinth) and interfere with recreational activities (Thiemer et al. 2023). Innovation in macrophyte manipulation has significantly advanced the method’s effectiveness and adaptability. Modern replanting involves the strategic use of native species with functional traits (e.g., high oxygen production, phosphorus uptake). For instance, replanting with the native submerged macrophyte *Vallisneria spiralis* in lake Como in Italy, which translocate substantial amount of oxygen to its roots helps creating oxic conditions in sediment (Castelnuovo et al. 2024). This oxygenation can drive redox sensitive processes such as phosphorus binding and nitrogen cycling. Other replanting innovations focus more on planting in specialized substrates, where macrophytes were transplanted in 3-D printed biodegradable substrates showed a 85.7% higher survival rate compared to plants replanted directly in the lake sediment (Castelnuovo et al. 2024).

Macrophyte harvesting is not only used for control of nuisance species, but the recovered biomass also provides potential support for circular economies. For instance, the biomass of emergent macrophytes (e.g., *Phragmites australis* and *Typha* spp.) could be used as building and insulation materials (Bajwa et al. 2015; Colbers et al. 2017), and for renewable energy production (Komulainen et al. 2008). Submerged invasive species, such as *Elodea* sp., in turn, could be used for biogas production (Muñoz Escobar et al. 2011; Zoppi et al. 2024), may be suited as an additional feedstock, and have potential in replacing mineral fertilisers, although the enrichment of potential phytotoxic elements to plant biomass poses a potential risk if used in agriculture (Zoppi et al. 2024). Recent innovations in macrophyte harvesting also include new technologies for a more cost-effective way of harvesting biomass, by developing, for instance, autonomous suction harvesting (ASH) techniques replacing the driver with a remotely operated vehicle (Olden 2024).

The longevity of macrophyte manipulation depends on water quality, due to which a sufficiently low external nutrient loading must be ensured. When conditions are favourable, i.e. water turbidity does not hinder the recolonisation of submerged macrophytes, replanting can deliver long-lasting ecological benefits. The durability of harvesting, in turn, may be limited in cases where macrophytes rapidly regrow after harvesting, or where nutrient pressures remain unresolved. In particular, the removal of floating and submerged species can trigger algal blooms if external nutrient loading is not simultaneously reduced (Harpenslager et al. 2022). Additionally, it must be noted that submerged macrophytes are crucial for both biodiversity and supportive ecosystem services in shallow lakes (Janssen et al. 2021), and their high abundance is associated with good ecological status (Poikane et al. 2018). Hence, the removal of submerged macrophytes at any scale larger than necessary should be avoided (Härkönen et al. 2025).

Figure 17. Dense Phragmites australis stands at FutureLakes' Demo site, L. Vesijärvi. © Laura Härkönen

Table 4 – Applicability and scalability potential of reviewed innovative in-lake restoration solutions (continued, 1/4).

	Solution	Applicability	Benefits	Disadvantages	Costs	Scalability potential
Engineering solutions	Aeration and oxygenation (BfS, Other)	Eutrophied, thermally stratifying lakes or lake basins; temporarily stratifying shallow lakes and ponds	Support fish survival when risk of anoxia under ice; anticipated to reduce internal phosphorous loading and suppress the development of HABs	High operational costs and energy consumption; impacts only during intervention and continuous maintenance required; risk of increased decomposition of organic matter in the sediment; potentially increasing pool of mobile phosphorus with a risk of release when oxygenation is stopped; increased hypolimnetic temperature with subsequent implications for increased hypolimnetic microbial productivity, vertical zooplankton refugia and cold-stenothermic fish; requires constant repetition without providing long-term results	Implementation costs between 4,000-27,000 € per ha, maintenance reported to 10,000-200,000 per year (Chmiel et al. 2024; Łopata et al. 2023; Sarvala et al. 2020)	Intermediate to high, TRL 6-9 depending on technology
	Algaecides (Other)	Surface waters with HABs	Rather selective against cyanobacteria, relatively rapid decay and no residue	Not tackling the root cause of the problem; possible negative impacts for non-target planktonic organisms still possible with some of the chemicals used; regular repeated treatments necessary	Several thousands of EUR per ha	High, TRL 8-9 with mature technology demonstrated in operational environment
	Hypolimnetic withdrawal and treatment systems (CBS, Other)	Eutrophied, thermally stratifying lakes or lake basins	Can potentially improve lake ecological status by permanent nutrient removal; prevents negative downstream water quality impacts associated with conventional HW; possibilities for P reuse if it can be extracted from the precipitate	High energy demand; consumes more time and resources than traditional HW; requires periodic replacement of sorption beds and regular maintenance of the treatment unit	50,000-100,000 € initial costs for the construction (Härkönen et al. 2024); maintenance costs required but not provided	Intermediate to high, TRL 4-9 depending on the treatment systems and sorption materials used. Mature technology available.

Table 4 – Applicability and scalability potential of reviewed innovative in-lake restoration solutions (continued, 2/4).

	Solution	Applicability	Benefits	Disadvantages	Costs	Scalability potential
Engineering solutions	Phosphorus inactivation (CBS, BfS, Other)	From shallow to deep lakes	Provides quick reductions in water column P concentration and has potential for pushing the equilibrium towards a clear, macrophyte-dominated state of the ecosystem	Result longevity highly dependent on the site characteristics; negative temporal impacts to biota	Capex 1000-1900 € depending on the doses and chemicals used (Dunalska et al. 2018; Grochowska et al. 2023; Sarvala et al. 2020)	High, TRL 9 for P inactivation itself with mature technology demonstrated in operational environments. TRL for the reuse of DWTR and other side-streams as P inactivating agents remains low, 3
	Phoshorus adsorbents (CBS)	Studies thus far restricted to laboratory scale	Potentially provides possibilities for P recovery and P reuse after separation from adsorbents	Neutral pH, low alkalinity and conductivity and limited concentration of humic substances may be required for improved P-binding capacity; Fe-based MPs require oxic conditions; high material demands; adsorbent-specific risk of increased turbidity; lack of field-scale studies due to which the scalability potential cannot be assessed	Not provided	Low, TRL 3-5 with technology mainly demonstrated in laboratory
	Sediment capping (CBS, Other)	Best applicable to small-sized lakes or ponds	Prevents nutrient and/or contaminant release from eutrophic or contaminated lake sediments	Temporal negative impacts for biota	Depends on the scale (lake area) of capping, and capping material; from low to high initial cost and technical expertise requirements for design, modelling, and construction	High, TRL 7-8 with mature technology demonstrated in operational environment; typically requires extensive coordination with regulatory agencies and stakeholders ensured design approval, risk compliance, and project execution

Table 4 – Applicability and scalability potential of reviewed innovative in-lake restoration solutions, (continued, 3/4).

	Solution	Applicability	Benefits	Disadvantages	Costs	Scalability potential
Engineering solutions	Sediment removal and reuse (CBS, Other)	Shallow lakes with high internal loading	Can provide long-term water quality benefits if external loading has been sufficiently reduced and the amount and depth of sediment removed is sufficient; can create favourable conditions for the recolonization of submerged macrophytes if turbidity is reduced after the intervention; provides circular economic co-benefits if the sediment can be reused	Negative temporal impacts to biota; laborious handling of sediment	High, vary from a few EUR per m³ to up to 50 € per m³. However, novel automated sediment removal technologies provide potential for remarkably reducing the costs	High, TRL 9 for the sediment removal itself with mature technology demonstrated in operational environments. TRL for the sediment reuse 4-9
Biological solutions	Algal harvesting (CBS)	Wastewater treatment; eutrophic lakes or retention ponds	Reduced algal biomass, toxins and other contaminants; harvested algae could serve as the sources of food, biofuels or alternative substrate in biotechnological processes or pharmaceutical applications; harvesting may removes nutrients from the system along with algae	Potentially high energy demand and complicated to establish systems in natural lakes; potential negative implications for planktonic non-target organisms; for safety reasons, possible food applications require toxin removal	Not provided	Intermediate to high, TRL 7-8 with mature technology widely applied in industrial work; potential for HAB removal if harvesting energy costs can be reduced (e.g. using solar energy)
	Artificial reefs (NbS, BfS, CBS)	In lakes where natural substrate structure and/or aquatic vegetation have been lost	Mimic natural habitats and provide shelter, feeding, and breeding sites for fish, invertebrates and waterfowl	Negligible impacts on water quality	Not provided	Intermediate, TRL 3-7

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Table 4 – Applicability and scalability potential of reviewed innovative in-lake restoration solutions (continued, 4/4).

	Solution	Applicability	Benefits	Disadvantages	Costs	Scalability potential
Biological solutions	Fish manipulation (NbS, CBS, BfS)	Eutrophied, shallow lakes with dense planktivorous and benthivorous fish populations	Can potentially improve the ecological status of lakes and reduce cyanobacterial blooms by reducing zooplankton grazing by planktivorous and sediment resuspension (internal P loading) by benthivorous fish; removed fish can be used both as feed or as human nutrition as well as in biogas production	Consumes resources as the effort has to be high and may require repeated measures to achieve long-term results	Operational costs 10,000 € per year (Sarvala et al. 2020)	High, TRL 9. However, implementation in shallow lakes with dense macrophyte cover may sometimes be difficult.
	Littoral and shoreline protection and habitat creation (NbS, BfS, CBS)	Suitable for large freshwater lakes suffering from habitat degradation, especially loss of littoral zones due to shoreline development	Restores biodiversity; enhances aquatic food web dynamics; supports fish reproduction and bird nesting; improves nutrient cycling between land and water	High financial and resource costs; Requires long-term planning and large-scale engineering	No specific financial data provided, but projects are energy-intensive and resource-demanding (Lilith: the costs for the Marker Wadden were 78 million euro (this is mentioned in the conclusion) for 900 ha)	High, TRL 7-8
	Macrophyte manipulation (NbS, BfS, CBS)	Mainly replanting and harvesting of macrophytes. Possible to harvest free-floating plants in deeper lakes	Replanting: nutrient retention, increases in water clarity. Harvesting: Combatting invasive species; improved recreational amenities; improved reproduction success of waterfowl via increased share of open water areas in lakes suffering from emergent macrophyte overgrowth	Requires repetition, thus increasing the operational costs for both replanting and harvesting; removal of submerged macrophytes from shallow lakes potentially poses a risk of shifting from a clear water to turbid, algal dominated state.	Not provided for macrophyte re-establishment. For harvesting, *c.* €30-45/tkg plant biomass removed have been reported (Sarvala et al. 2020)	High for harvesting, with TRL 9. Intermediate for re-establishment with a TRL of 3-5

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3.3 Combinations of measures

3.3.1 Multiple external and internal measures

Sufficient reduction of external loading is in most cases a prerequisite for achieving long-term water quality improvements in lakes. Combining preventative catchment measures with restorative in-lake solutions are often considered to provide best restoration performance. Intensive catchment management combined with in-lake restoration approach with a CW, protection of riparian margins in the streams, introduction of agricultural nutrient management practices and a phosphorus-adsorbent sediment capping was effective in managing the nutrient load and P concentration in L. Oharo (New Zealand) (Özkundakci et al. 2010). Biomanipulation combined with preventative measures for external load reduction has also greatly improved the status of FutureLakes' Demo Site L. Vesijärvi (Salonen et al. 2020a). Likewise, the water quality in FutureLakes' Innovation site Lake Groote Melanen improved greatly when in-lake measures such as fish removal, dredging, capping of peat rich sediment with sand and an active barrier (lanthanum-modified bentonite), reconstruction of banks, and planting macrophytes were combined with external nutrient-load control, which was the diversion of two inlet streams (Lürling et al. 2024). Also, in another FutureLakes' Innovation site – Lake Bleiswijkse Zoom – the external nutrient load was reduced by stopping water inlet from the nutrient-rich Rotte River, creating a run-off interception ditch, and removing overhanging trees and shrubs. External load reduction was accompanied by in-lake measures, such as fish removal, shoreline reconstruction, dredging and LMB addition, yielding strongly improved water quality and ecosystem recovery (Meier et al. 2024).

Figure 18 – Multiple measures were combined in restoring Lake Groote Melanen (The Netherlands). © Guido Waajen, © Visserijbedrijf Kalkman, © Kurstjens B.V.

Importance of combining external and internal solutions is supported with several cases displaying negligible impact of extensive in-lake restoration efforts in re-oligotrophication if external loading has continued, underscoring the need for applying integrated restoration strategies (Waajen et al. 2019). For instance, in Lake Kleine Melanen, a combination of sediment removal, biomanipulation, P inactivation and shoreline reconstruction first failed to result in a projected clear-water state because of insufficient reduction of the external loading (Waajen et al. 2019). Once the required bypass of external loads was constructed in January-March 2018, right after which biomanipulation and addition of PAC and LMB to bind excess phosphate were conducted, the water quality in the lake strongly improved (Huitema 2020) and has remained till present (L. Seelen and G. Waajen, pers. comm. May 2025). Also, a restoration of three urban ponds in the Netherlands in which a package of measures was implemented including dredging, creating soft banks, planting macrophytes, and fish stock manipulation, was only successful in the two ponds that also had adequate external load control (Lürling et al. 2023).

3.3.2 Multiple internal measures

Especially in shallow lakes, the impacts of single restoration efforts can be hampered with nonlinear response trajectories mediated by e.g. interactions with biota (Abell et al. 2022; Gulati and van Donk 2002; Jeppesen et al. 2012; Sarvala et al. 2020; Scheffer 2004). Consequently, in most cases, a combination of in-lake measures may produce best result longevity if several processes maintaining eutrophied status can be tackled simultaneously. For instance, aeration (Section 3.2.1.1), that often has negligible impact on lake nutrient concentrations itself, is best applied in combination with other in-lake restoration measures to reduce internal nutrient cycling such as P inactivation (Section 3.2.1.4) or biomanipulation (Section 3.2.2.3) to provide longer-term water quality improvements (Grochowska et al. 2017). In e.g., Lake Długie (Poland), multiannual artificial mixing followed by sequential P inactivation using PAC has brought positive effects lasting for more than 20 years (R. Augustyniak, J. Grochowska pers. comm, May 2025). In the US, a whole-lake aluminium sulphate (alum) treatment followed by longer-term hypolimnetic aeration incorporated with microfloc alum injection has resulted in decreased water P concentration, decreased proliferation of HABs and changes in phytoplankton community composition (Moore et al. 2012; Moore et al. 2009). A combination of biomanipulation with LMB treatment in the Danish Lake Lyngsø has resulted in decreased nutrient levels, increased water clarity, increased coverage of submerged macrophytes and a shift from littoral benthic to more pelagic food resources by the dominant fish species (Berthelsen et al. 2024); combined biomanipulation and P inactivation using $FeCl_3$ together with LMB in Lake De Kuil, the shallow pond Dongen and pond Eindhoven (The Netherlands) improved water quality and supported macrophyte re-establishment (Waajen et al. 2016; Waajen et al. 2017); and combined biomanipulation, macrophyte re-establishment (Section 3.2.2.5) and LMB application have strongly improved water quality in the tropical Lake Yanglan (China) (Li et al. 2025).

In Uzarzewskie Lake (Poland) a combination of P inactivation (using $Fe_2(SO_4)_3$) and a novel approach of nitrate (NO_3) rich groundwater diversion into hypolimnion has also been demonstrated on operational scale (Dondajewska et al. 2018; Kowalczewska-Madura et al. 2017; Kowalczewska-Madura et al. 2024). The method could be effective in managing both main nutrients (N, P) in lake. Multiannual analysis showed promising results in water quality improvement. Also, (Kozak et al. 2020; Kozak and Gołdyn 2016) analysed effects of such treatment on macrophytes and phytoplankton. They noted an increase of share of eloeids and charophytes which confirmed an improved ecological status of lakes, as well as shift from cyanobacterial dominance into green algae, diatoms and mixotrophic cryptophytes. The benefits of those methods' combination were also assumed by (Gołdyn et al. 2014). However, as highlighted by Tammeorg et al. (2024),

All in all, a successful lake restoration requires understanding both external and internal nutrient fluxes together with biological traits and functions maintaining the eutrophied status. In several cases, a system analysis guided selection of cohesive measures has led to strongly improved water quality (e.g., Lake Rauwbraken, Lake Bleiswijkse Zoom, Lake Groote Melanen, pond Dongen, Pond Heesch). Cases in which essential buttons identified by the system analysis (external load sources) had not been pressed failed to meet water quality improvements (e.g., Lake Kleine Melanen, pond Eindhoven).

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4 Conclusions

Innovations available for improved water protection and lake restoration were reviewed using a structured literature review that was complemented by scoping interviews with selected scientific/practitioner communities. A majority (55%) of innovations reviewed were in-lake measures with most of them representing NbS or other solutions that are innovative but do not directly fall within NbS, CBS of BfS categories. Novel solutions for in-lake measures have mainly concentrated on developing well-established measures to better provide circular economic or biodiversity co-benefits (Table 4, Figure 19). Well-established engineering solutions to tackle internal loading without P recovery have experienced progress in terms of more environmentally friendly materials and applications for P inactivation, which **better secures and supports biodiversity**. Other engineering solutions, such as sediment removal and HWTS for P recovery have also been developed to **reduce internal sources of pollutants** from lake sediments and to simultaneously provide possibilities for P reuse, for instance, as an alternative for mineral P fertilisers. Additionally, different P adsorbents for P recovery from the water column have been in the focus of experimental research. While the efficacy and long-term water quality impacts of sediment removal have been well-reported, HWTS still lacks longer-term studies evaluating its restoration performance and the scalability potential of P adsorbents to whole-lake scale remains questionable.

Several in-lake measures could be labelled as multifunctional, i.e. measures combining elements from NbS, CBS or BfS. For instance, biomanipulation has earlier been denoted as a NbS (Triest et al. 2016) but it can also be considered as a BfS when implemented on waterfowl lakes to primarily support biodiversity and as a CBS when the catch is utilised in circular reuse for e.g., food and feed production. Indeed, most biological in-lake solutions tend to have a link to circular economies as harvested biomass (fish, algae, macrophytes) from the lakes can be re-framed as opportunities for recovering valuable, alternative resources and materials. Despite being potentially categorisable as NbS, most of the biological solutions were categorised as BfS as they often focus on **restoring the biological communities and natural functioning of lakes** by, for instance, rebalancing fish communities and supporting the recolonization of submerged macrophytes. Protection and restoration of shoreline habitats together with installation of artificial reefs and different fish passage structures can be used to **restore a more natural morphology of lakes** that also supports biota. **Creation of new habitats** such as littoral areas or islands to support waterfowl breeding were also among the reviewed innovations to benefit biodiversity. However, it should be emphasised that to sustainably restore lakes it is critically important to reduce external pressures too while implementing in-lake measures. Despite advances in in-lake restoration methods, no quick fixes for lake restoration exist and often the best restoration performance is achieved with combinations of measures in both catchments and in lakes.

From the reviewed innovations, 40% dealt with external solutions for improved water protection, pressure reduction and management actions on catchment scales. Most of the external measures were labelled as NbS with multifunctional measures also common. Following the objectives of the Zero Pollution Action Plan, the most effective solution to restore lakes remains to **prevent contaminants from entering the water cycle**. Hence, such catchment water management practices aimed at pressure reduction together with other cropping and forest management strategies that prevent nutrient losses from land are the most effective approaches in pollution control. These practices can provide multiple co-benefits for several policy goals such as the WFD, Nature Restoration Regulation and Biodiversity Net Gain, Flood and Drought Risk Management, and wider societal and economic policy goals within the European Green Deal by tackling water scarcity, diffuse pollution and higher frequencies of extreme weather events (Table 1, Table 2). Indeed, the uptake of improved catchment management practices, such as controlled drainage, no tillage, continuous cover crops, and CCF may have potential for providing higher net revenue from a societal point of view when compared to traditional practices (e.g., Miettinen et al. 2025). Peatland restoration and wetland re-establishment are also important tools for increasing the water retention capacity in catchment areas, that not only benefit pollution control but also deliver resilience to floods and droughts, benefit biodiversity and reduce GHG

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emissions. NRR targets to increase the presence of landscape features on intensively used agricultural land explicitly support the wider adoption of such NbS that incorporate wet, woody and grassy features, including buffer zones and pond creation. Further, the EU's initiative of planting "3 billion trees" could offer ways to increase the presence of woody riparian corridors and edge-of-field buffers in agricultural areas, thus enhancing the efficiency of external water protection measures while at the same time improving the carbon capture.

Different NbS and CBS water protection structures are available to complement water protection with many of them originating from wastewater treatment but also applicable in agricultural or forested landscapes (Table 3). Recent advances in water protection are different innovative CBS adsorbents and reactive media for **improved pollution control**, in addition to which, the treatment performance of well-established measures such as CWs has been improved and their areal footprint and operational costs have been reduced. Also, BfS **structures mimicking the functions of natural habitats** such as FTWs to improve water quality and support aquatic biodiversity have been developed as part of promoting urban green-blue infrastructure. Different soil amendments as CBS to improve soil quality have been tested on operational scales, suggesting significant reductions in soil erodibility, subsequent benefits for SS and P load control, and attraction for farmers due to not resulting in loss of income for farmers in terms of land requirements and due to quick load reductions. Different sediment and nutrient interceptors installed at lake inflow points also have potential in complementing broader catchment-scale water protection strategies. However, all these edge-of-field or end-of-pipe methods have limited ability to trap nutrients and substances especially in dissolved forms, due to which it must be emphasised that it is critically important to reduce external pressures first with such **land use choices that reduce the pollution losses from land**. Although the scalability potential of multi-functional, NbS-type catchment management approaches to treat pollution on-site in both agriculture and forestry is high, a prerequisite for these preventative measures to become mainstream approaches in lake protection are uptake of such policies, regulative instruments, economic incentives and financing schemes that support the transition. For instance, it has been recognised that to better comply with the WFD, there is a need to more thoroughly apply and enforce the Polluter Pays principle in general (European Court of Auditors 2021), and especially to address diffuse and legacy pollution (Sanchez Trancon and Leflaive 2024; Wiering et al. 2020). Similarly, education and awareness raising of stakeholders are essential to be able to adopt new measures and approaches for improved water and nutrient retention.

Noteworthy, a minority of studies (5%) dealt with combinations of measures on catchment and in-lake scales, although a strategy combining these approaches has in many studies proven to provide the best results with the highest longevity. Although some of the reviewed measures were clearly multifunctional, the continuum from the external measures in the catchment to their impacts in the receiving lakes was not well covered by the reviewed literature. Nutrient load reduction practices should in general consider all pathways from source to transport and destination with controlling practices utilized at all scales (Osmond et al. 2019). Consequently, in all cases where a water quality issue in a lake has been identified, a search for the underlying causation is decisive to address the problem adequately. This diagnosis of the water quality issue, the **lake system analysis serves as a blueprint for lake protection and restoration**. It determines the magnitude of external and internal loads together with lake's biological function(s) and boundary conditions to identify which measures are best to pursue to have the highest chance for restoration success. A lake system analysis can also sometimes provide the underpinning evidence for a do-nothing scenario. Preferably, a system analysis should also include a cost-benefit analysis to evaluate best applicable measures.

Our review revealed that most of the scientific studies do not provide any information on costs, and even fewer evaluate them in relation to monetarised estimates of the environmental benefits. Moreover, post-intervention monitoring of measures is often limited and, at best, often concentrated on only few parameters without comprehensive addressing of biota or other policy relevant indicators, such as GHG emissions. This makes it difficult to fully evaluate the benefits of lake protection and restoration efforts, and to compare the cost-efficiency of alternative measures that also require considering the result longevity. In general, restoration efforts that mainly treat the symptoms of eutrophication often need regular repetition. In turn, measures aiming at more permanent nutrient removal often come with higher capital costs. However, they have potential in providing longer term results reducing the need for repeated interventions, thus requiring less financial investment over the long-term. New funding schemes and mechanisms are needed to allow for sufficient resources in these nutrient reductive in-lake methods. At the same time, their profitability still needs to be developed for them to become mainstream approaches in lake restoration. As concluded by Spence et al. (2023), the substantial costs of restoring lakes are outweighed by the received, significant economic benefits to society following restoration.

This review underpins the difficulties for adequate lake restoration, the necessity for novel approaches, and the need to prevent lake degradation in areas still less impacted by anthropogenic activities. It also requires less decentralised policies, properly aligned, coherent water governance, political willingness to address diffuse nutrient pollution and financial investments to measures providing long-term water quality benefits (Figure 19). This document is intended to act as a useful source of information for EU Member States for the development of their National Restoration Plans for the NRR.

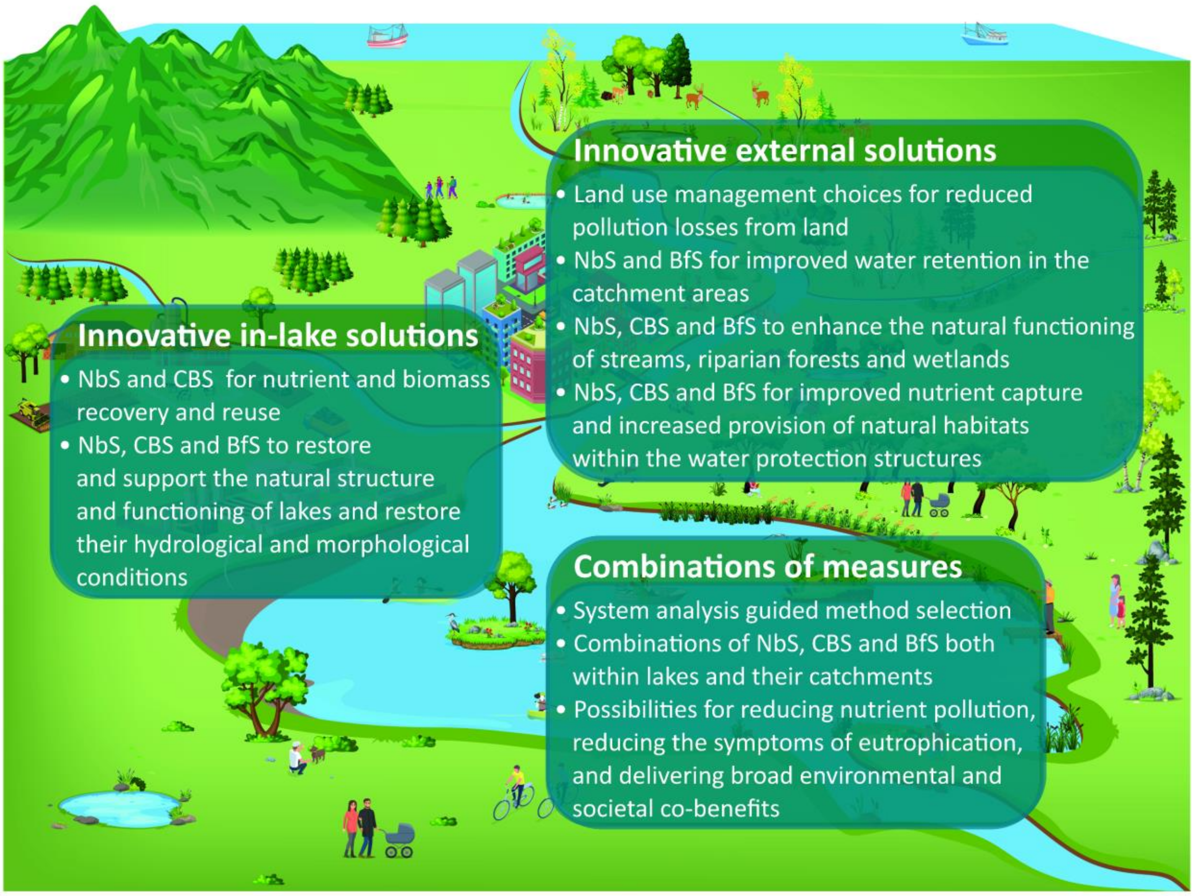

Figure 19 - Innovations to support the protection and restoration of European lakes in a multifaceted, co-beneficial context. Prerequisites for their wider adoption are supportive policies, new financing schemes and mechanisms, cohesion in water governance and increased awareness and knowledge among relevant stakeholders.

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Annex 1

In-lake measures to be avoided

Restoration measures should be effective, easy to make and apply, affordable and safe. Following upon the heatwave years of 2003 and 2006, numerous opportunists entered the market claiming they had THE solution against cyanobacteria blooms that had manifested during these heatwave years. In this section, we briefly describe measures that should not be applied in lake restoration as no benefits have been proven and the measures are likely to have no impacts on the symptoms of eutrophication with harmful implications for eutrophication control or non-target organisms instead.

Effective microorganism technology

Effective microorganism (EM) technology has been advocated to be an end-all solution to water quality and sanitation problems. EM is also referred to as 'mudballs' or 'Bokashi balls', which can be thrown in water bodies directly as an environmentally friendly and cost-effective in-lake restoration method (Zakaria et al. 2010). The concept of EM was initially developed by Higa (1998) suggesting that adding EM-1® changes the microbial community and thereby inhibits harmful bacteria through competitive exclusion. EM-1® is said to contain about 80 species of microorganisms, including photosynthetic bacteria, lactic acid bacteria, actinomycetes, yeasts, and fermenting fungi. Yet, this statement could not be confirmed by analysis of EM-1® (Van Vliet et al. 2006). The efficacy of EM is considerably controversial. There is limited scientific evidence on using EM to control nutrients or algae. Instead, they can be found on various webpages and are based on anecdotal evidence. EM Probiotyk™ was applied in Muchawka Reservoir (surface area 40 ha) in Poland in 2012-2013 and 2013-2015 (16,000 L in two stages) by motor pump. The results showed that cyanobacterial scums or pollution at bathing sites are invisible, and the bacteriological water quality was described as 'excellent' (Sitarek et al. 2017). However, the elevated nutrient content from the River Muchawka might have a further impact on water quality. A similar case occurred in Lake Konin, Poland, where EM was applied in 2014 by applying mudballs in Spring (35 thousand), and a liquid solution of probiotics in summer and autumn (2-5.5 L/ha). Additionally, barley-straw floating bales were established along the shoreline in summer 2014 and removed in autumn of the same year. The results indicated that EM was insufficient to reduce both nutrient content and phytoplankton abundance within Lake Konin (Dondajewska et al. 2019b).

In August 2021, 1500 EM mudballs were added to a 750 m^2 enclosure in Turawa Reservoir (Poland) along with 7500 L of 10 times diluted EM solution (Tomczyk et al. 2024). The authors claim that EM decomposed organic matter, leading to less P and N, which, from a mass balance perspective, especially for P, requires more research into where the P went. Furthermore, a comparison of just before with a one and two-week post-treatment, without proper control, should be met with care.

To further clarify the effectiveness of EM, a controlled experiment was conducted to clear the water of cyanobacteria (Lurling et al. 2010). The results indicated that within the 4-6-week experimental periods, EM was ineffective, and the cyanobacterial chlorophyll-a concentrations increased within 4 weeks from 120 to 325-435 μg/L in all controls and EM treatments. Due to a lack of evidence of removing nutrients and algae blooms from EM, and they could be an extra source of nutrients, (Lurling et al. 2010) consider EM products to be ineffective. In 2015, the Regional Water Authority De Dommel (the Netherlands) added 500 EM mudballs to an urban pond (vijver Jan van Galenweg, Vught), however, without any success of EM to control eutrophication and cyanobacteria overgrowth (Lürling and Mucci 2020). Overall, the claim that EM reduces nutrient levels and prevents the growth of algae is not supported by scientific evidence. Moreover, the persistence and long-term impact of EM applications remain unclear. Rigorous, peer-reviewed research is required to validate any purported benefits of EM in aquatic environmental management.

Ultrasonication

During recent decades, low frequency ultrasonication has also emerged as a potential measure to reduce HABs (Kibuye et al. 2021; Rajasekhar et al. 2012). High energy ultrasonication –causes cavitation that will damage all organisms in its power beam. Because of high-energy costs and undesirable effects on non-target organisms, low-energy ultrasound has been promoted claiming that it controls cyanobacteria by bringing their gas vesicles into resonance. This resonance is claimed to mechanically rupture the cyanobacterial gas vacuoles causing loss of buoyancy control and reducing the cyanobacterial abundance (González-Fernández et al. 2012; Jong Lee et al. 2000; Rajasekhar et al. 2012; Wu et al. 2011; Wu et al. 2012). Several commercialized ultrasonic devices claiming to control algae are available on the market, but the scientific validation on their efficacy is lacking (Wu et al. 2011).

Laboratory studies in small water volumes, limited power and varying ultrasonic frequencies or intensities have shown suppressed cyanobacterial growth, collapse of gas vesicles, cell wall disruption and disturbance of the cyanobacterial photosynthetic activity (Rajasekhar et al. 2012; Wu et al. 2011). However, contrasting effects of ultrasonication are also reported (Purcell et al. 2013a; Wu et al. 2012) and the impact of ultrasonication on cyanobacteria is variable and dependent on both frequency and intensity (Joyce et al. 2010; Lürling et al. 2016). Additionally, although ultrasonication has been claimed to selectively control cyanobacteria with gas vacuoles over other species (Ahn et al. 2007; Rajasekhar et al. 2012; Tang et al. 2004), several controlled laboratory studies have shown that the effects of ultrasonication are not limited to cyanobacteria (González-Fernández et al. 2012; Lürling and Tolman 2014; Wang et al. 2014). Ultrasonication has been demonstrated to destroy the bonds between the cell walls and the cell contents of green algae (González-Fernández et al. 2012; Wang et al. 2014), reduce diatom abundance (Purcell et al. 2013b), and have acute lethal impacts on crustacean zooplankton via destruction of eyes, eggs, intestines and carapaxes (Lürling and Tolman 2014). Application of the method in natural water bodies has been limited (Kibuye et al. 2021; Park et al. 2017) and the impacts of ultrasonication on natural biota remain poorly understood.

In the Netherlands, in 2007 three field trials were conducted – in two basins (a control and an ultrasound treatment) at a former wastewater treatment plant, in a lake (De Gouden Ham) and in an enclosed harbour area (Tholen), but in all the three cases ultrasound was not able to control cyanobacteria (Kardinaal et al. 2008). Similarly, a trial conducted in Lake Zoetermeer (The Netherlands) showed that the four ultrasound buoys installed could not prevent cyanobacteria growth, in the lake cyanobacteria developed in the two years with ultrasound similar to the two years without ultrasound (Lürling and Mucci 2020). Likewise, field trials in Germany (Lessmann and Nixdorf 2015), Belgium (Van Wichelen et al. 2025) and Finland (Härkönen et al. 2022) did not yield any proof that cyanobacteria could be controlled by ultrasound. Instead, during a 7-mo ultrasound trial in Loweswater, UK, cyanobacteria counts were higher than the median of no-ultrasound years, which led to the conclusion that ultrasound "had no discernible effect at any time in terms of the numbers of algal species or the length of algal filaments" (Webb et al. 2017). An 18-mo trial in Reservoir C (Australia) did not find an effect of ultrasound on cyanobacteria, and "The supplier has requested to remain anonymous within this paper" (Vaughan et al. 2023). Recently, another field trial reported no effect of ultrasound on cyanobacteria, concluding that the transducers used in the field trial did not cause cavitation and produced pressure of two orders of magnitude lower than needed to collapse gas vesicles (Tischer et al. 2025). Moreover, the frequencies used are in the kilohertz range, orders of magnitude lower than the megahertz frequencies needed to bring gas vesicles into resonance of which penetration depth in water is very limited (Lürling et al. 2016; Tischer et al. 2025). These physics explain why in the above-mentioned field trials no effect of ultrasound on cyanobacteria was found.

Since ultrasonic devices are advertised as environmentally friendly and easy solution to control HABs, they may be an attractive option for mitigating cyanobacterial blooms in natural lakes. However, low-frequency, low-energy ultrasound is not effective and may exert potential negative impacts on non-target organisms, while high-power ultrasound will be expensive and kill everything in the vicinity of the transducers, not only cyanobacteria, which is in direct conflict with the claim of ultrasonication's environmental safety (Lürling et al. 2016). As ultrasonication has no ability to influence nutrients, unexpected impacts of ultrasonication on ecosystem functioning are also possible in natural lakes where both bottom-up and top-down regulative forces for cyanobacterial proliferation are present (Härkönen et al. 2022). Consequently, ultrasonication should not be considered as a quick fix for combating algal blooms in natural lakes.

- **Applicability**: NA
- **Benefits**: **no proven benefits**
- **Disadvantages**: Can't reduce nutrients and cyanobacterial nuisance, adds nutrients (EM), possible negative impacts for non-target organisms (US), poor manufacturing control (EM).
- **TRL**: 0-1
- **Costs**: Not provided

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