Science of the Total Environment 967 (2025) 178738

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Review

Artificial destratification options for reservoir management

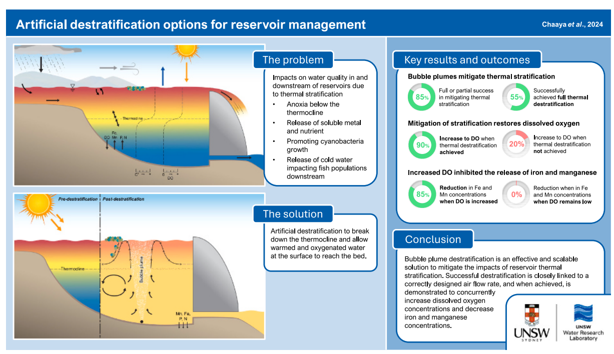
Fred Chaaya [a,*], Brett Miller [a], Matthew Gordos [b], Bojan Tamburic [c], Stefan Felder [a]

[a] Water Research Laboratory, School of Civil and Environmental Engineering, UNSW Sydney, Manly Vale 2093, NSW, Australia
[b] Freshwater Environment Branch, NSW Department of Primary Industries and Regional Development, Wollongbar 2477, NSW, Australia
[c] Water Research Centre, School of Civil and Environmental Engineering, UNSW Sydney, 2052, NSW, Australia

HIGHLIGHTS

- Comprehensive review to identify effectiveness of artificial destratification systems
- Bubble plumes are the most effective and scalable destratification solution.
- Successful thermal destratification leads to mitigating other water quality impacts.
- Correctly designed air flow rates are key to successful thermal destratification.
- Quantitative data for design guidelines and optimised operation needed.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Jay Gan

Keywords:
Aeration
Algae mitigation
Bubble plume destratification
Cold water pollution
Dams and reservoirs
Mechanical mixers
Thermal stratification
Water quality

ABSTRACT

Reservoir stratification impacts reservoir and downstream water quality, creating complex management challenges driven by interactions between hydrodynamics, weather patterns, and nutrient dynamics. Artificial destratification is one technique used to ameliorate the impacts of stratified reservoirs, with bubble plumes or mechanical mixers being the primary methods employed. This global review assessed 138 bubble plume and mechanical mixer artificial destratification systems installed in 114 reservoirs to evaluate the comparative effectiveness of each method. Destratification systems were assessed in terms of their effectiveness in breaking thermal stratification and consequently mitigating cold water pollution, increasing dissolved oxygen concentrations throughout the water column, reducing the concentration of soluble metals, and reducing (potentially toxic) cyanobacteria populations.

Bubble plume destratification was found to be more effective than mechanical mixing at mitigating the impacts of thermal stratification. Successful thermal destratification was closely linked to subsequent increases in dissolved oxygen concentrations and decreases in manganese and iron concentrations. Mixed results were observed for the reduction of cyanobacteria populations from artificial destratification; however, a correlation was observed between cyanobacteria control and successful thermal destratification in deeper reservoirs.

Achieving thermal destratification was closely linked to the ratio of the reservoir capacity to the air flowrate used for destratification (the "volumetric destratification coefficient"). Failed thermal destratification was observed in reservoirs where the volumetric destratification coefficient was less than approximately 0.005 L/s/

* Corresponding author.
E-mail addresses: f.chaaya@wrl.unsw.edu.au (F. Chaaya), b.miller@wrl.unsw.edu.au (B. Miller), matthew.gordos@dpi.nsw.gov.au (M. Gordos), b.tamburic@unsw.edu.au (B. Tamburic), s.felder@unsw.edu.au (S. Felder).

https://doi.org/10.1016/j.scitotenv.2025.178738
Received 4 October 2024; Received in revised form 23 December 2024; Accepted 3 February 2025
Available online 12 February 2025

ML. This review identified the potential for scalability of bubble plume destratification across different reservoirs, but future research needs to provide more quantitative data that can be used to develop holistic design guidelines for bubble plume destratification systems for a wide range of reservoirs and operational conditions.

1. Introduction

Thermal stratification in reservoirs typically occurs during the warmer summer months, when external environmental factors, such as higher solar radiation and elevated air temperature, cause the surface layer of the reservoir to heat up (MacIntyre and Hamilton, 2024). This stratification leads to the formation of distinct layers within the water column, characterized by differences in density and temperature (Fig. 1). These layers are known as the epilimnion (surface), the hypolimnion (bottom), and the metalimnion (middle), which features a pronounced temperature and density gradient called the thermocline. The thermocline, in particular, serves as a barrier that hinders the natural mixing of the warmer epilimnion and the cooler hypolimnion at the reservoir's bottom (Patterson and Imberger, 1989; Huttula, 2012; Li et al., 2020; Shi et al., 2022; MacIntyre and Hamilton, 2024). Global warming is expected to further intensify stratification effects, increasing temperatures at the epilimnion and thermocline at a greater rate than the hypolimnion, leading to enhanced vertical stratification for longer periods (Zhang et al., 2015; Woolway et al., 2021; Wang et al., 2024).

While stratification can occur in reservoirs with depths of <5 m (Pastorok et al., 1981; Müller and Mitrovic, 2015), impacts, and the complexity of managing these impacts, become more significant as the reservoir's depth and capacity increase. Negative impacts on the in-reservoir environment include dissolved oxygen (DO) depletion below the thermocline (Steichen et al., 1979; Zhang et al., 2015; Shi et al., 2022), release of soluble metals and nutrients from the reservoir bed (Davison, 1993; Beutel et al., 2008; Li et al., 2019; Shi et al., 2022), promotion of (potentially toxic) cyanobacteria blooms (Toetz, 1977; Burns, 1994; Visser et al., 2016; Smucker et al., 2021), as well as the release of cold water pollution to downstream receiving waterways when water is discharged from deep-level offtakes in the hypolimnion (Sherman, 2000; Ryan et al., 2001; Preece and Jones, 2002; Preece, 2004) (Fig. 1). Reservoir stratification has a direct impact on aquatic biota both in-reservoir, by way of the hypolimnion being uninhabitable (Preece et al., 2019), and downstream, through poor-quality, cold water releases (Preece and Jones, 2002; Michie et al., 2020b, 2020a, 2023).

Fig. 1 illustrates the impacts thermal stratification can have on water quality in the reservoir including the complex interplay between various biochemical processes. Dissolved oxygen is an essential property of the reservoir environment and supports all aerobically respirating organisms within. Additionally, low DO affects the solubility of many inorganic compounds (Davison, 1993), which can adversely affect water quality both within the reservoir and downstream. In a thermally stratified reservoir, the strong density gradient at the thermocline prevents oxygenated waters from the epilimnion from reaching the hypolimnion, inhibiting its reaeration (Steichen et al., 1979; Miles and West, 2011; Liu et al., 2019; Shi et al., 2022). At the sediment-water interface, high sediment oxygen demand drives aerobic biogeochemical processes that consume oxygen, leading to hypoxic or anoxic conditions (Haynes, 1975) and the subsequent release of undesirable soluble metals and nutrients (Beutel et al., 2008).

To sustain biological oxygen demand under hypoxic or anoxic conditions, bed sediments release soluble iron and manganese through the reduction of metal oxides (Beutel et al., 2008; Bryant et al., 2011; Li et al., 2019) further degrading water quality. The presence of iron and manganese can increase turbidity, darken water colour and result in unpleasant tastes and odours in drinking water (Munger et al., 2016; World Health Organization, 2017). These soluble metals can also negatively impact fish causing oxidative stress (Vieira et al., 2012) and gill damage (Hedayati et al., 2014). Nutrients such as phosphorus and nitrogen are also released under hypoxic and anoxic conditions (Beutel et al., 2008; Shi et al., 2022), promoting algae growth and occasionally

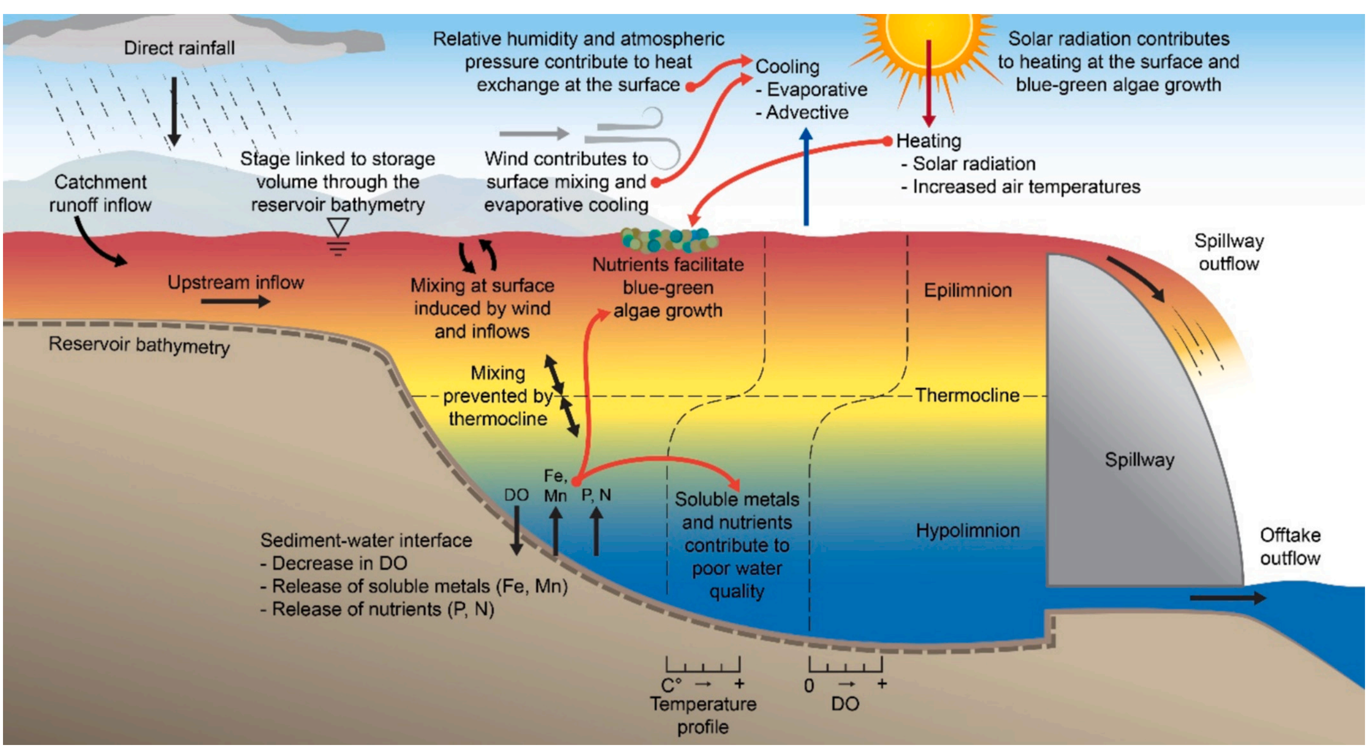


Fig. 1. Natural stratification processes and their impacts on the in-reservoir and downstream environment.

leading to blooms of toxic cyanobacteria (commonly known as blue-green algae).

Large thermally stratified reservoirs are particularly conducive to cyanobacterial growth due to long residence times, reduced mixing, increased nutrient availability caused by stratification (Spigel and Imberger, 1987; Tilzer, 1987) and anthropogenic nutrient loading (Hamilton et al., 2016) (Fig. 1). Anthropogenic nutrient loading from catchments is often a primary driver of cyanobacterial blooms and can be managed by limiting external nitrogen and phosphorus supply. However, this approach may not be economically or politically feasible compared to in-reservoir management methods (Hamilton et al., 2016). Cyanobacteria often outcompete other algal species in reservoirs, especially during periods of stratification (Burns, 1994). Cyanobacteria are able to regulate their buoyancy through gas vesicles (Wallace and Hamilton, 1999), enabling them to migrate across density gradients in a stratified reservoir. This provides them a competitive advantage over other species, allowing them to remain at the surface during periods of low wind and high light availability (Visser et al., 1996), increasing their daily light dose and reducing sedimentation losses (Visser et al., 2016).

Cyanobacteria blooms are recognised globally as an issue of concern for reservoir management (Hamilton et al., 2016) due to the cyanotoxins that some species can produce which impact both human and aquatic health (Howard, 2012; Visser et al., 2016). Cyanotoxins can attack the liver (hepatotoxins), nervous system (neurotoxins) and skin (dermatoxins) (Merel et al., 2013; Hamilton et al., 2014). Certain compounds produced by cyanobacteria, such as geosmin, 2-methylisoborneol and many others, can also introduce an unpleasant taste and odour in drinking water (Falconer, 1999; Zhu et al., 2022). Overall, cyanobacteria blooms can result in significant economic impacts, including increased water treatment costs, lost tourism, and reduced fishing revenues, in particular through summer periods when recreational and irrigation use of reservoirs increases (Hamilton et al., 2014).

Besides negative impacts inside the reservoir, water released from stratified reservoirs can also negatively impact the downstream river system. As water is typically extracted from offtakes located at or near the reservoir bottom in the hypolimnion (Ryan et al., 2001), this can impact the water quality in the downstream receiving waterway due to reduced DO and increased nutrient, iron, and manganese concentrations (Fig. 1). Water releases taken from below the thermocline are significantly colder (commonly 8–12 °C colder) than the reservoir's surface waters (Preece and Jones, 2002; Preece, 2004; Boys et al., 2009; Miles and West, 2011) (Fig. 1), impacting the downstream aquatic environment for hundreds of kilometres (so called cold water pollution (CWP)) (Todd et al., 2005; Lugg and Copeland, 2014). These unnatural thermal variations can have significant impacts on native fish populations (Todd et al., 2005; Michie et al., 2020b, 2020a, 2023) due to the sensitivity of fish physiology and behaviours to temperature changes (Fuiman and Batty, 1997), including fish reproduction (Boys et al., 2009), fish growth and metabolism (Clarke and Johnston, 1999) as well as survival and development of eggs and larvae (Koehn et al., 1995; Todd et al., 2005).

Considering the negative impacts thermal stratification can have and the complex interplay between the dynamics of reservoir hydrodynamics and water quality, management methods must be considered. Herein, this article critically reviews one of the most widely used management method, artificial destratification, with a focus on the performance of the most commonly implemented destratification technologies in reservoirs around the world – bubble plumes and mechanical mixers. Despite these technologies being widely implemented, their performance for mitigating the outlined impacts of reservoir stratification has been varied. This is especially true for larger reservoirs, where the costs of water treatment, loss of recreational value and water release impact on the downstream environment is exacerbated. Reporting on the outcomes of destratification operation is rarely standardised, making comparison between case studies difficult. Design guidelines for both bubble plumes and mechanical mixers are limited and outdated based on more recently available technology and experience. There is a need to synthesize and assess the available information relating to the design, operation and performance of current or previously operated artificial destratification system. Identifying the gaps in this knowledge will promote improved design, optimised performance and highlight the achievable water quality outcomes of effectively operated artificial destratification.

Available performance data from the destratification systems reviewed were recorded in a comprehensive database (see "Destratification Library" in supplementary materials) and analysed in terms of their success in mitigating the key negative impacts of reservoir stratification. The review concludes with a critical discussion on the success of destratification technologies and provides key considerations for designers and practitioners when implementing artificial destratification as a management strategy for their reservoirs.

2. Artificial destratification methods

Artificial destratification is a low capital cost, high operational cost method (relative to alternative selective withdrawal methods) that involves artificially mixing the reservoir to break down the thermocline. This allows heated, oxygenated surface water to reach the bottom of the reservoir, mitigating both the in-reservoir and downstream impacts of natural stratification (Pastorok et al., 1981; Ashby and Kennedy, 1993; Visser et al., 1996; Sherman, 2000; Sahoo and Luketina, 2006; Miles and West, 2011). Artificial destratification is generally accomplished using either rising bubble plumes or mechanical mixers at the surface or bed of the reservoir. These mitigation methods for reservoir stratification are the primary focus of this review.

Other novel methods of destratification such as the Gradual Entrainment Lake Inverter (GELI) (Read et al., 2011; Smith et al., 2018) have not been considered in this review due to the comparably smaller amount of available literature detailing design and outcomes. Water-lifting aerators (WLA) are an emerging hybrid technology (Huang et al., 2016), which, depending on their operation, serve to achieve either artificial destratification or hypolimnetic oxygenation. These systems have been shown to be capable of decreasing thermal stratification (Li et al., 2018), increasing DO concentrations (Ma et al., 2015; Li et al., 2018), mitigating the release of soluble metals (Li et al., 2019) and managing algae growth (Zhang et al., 2020). From the limited literature available, their destratification effects appear to be localised, as suggested in Huang et al. (2016). Due to the relatively small amount of literature surrounding this method, and the hypolimnetic oxygenation operations, WLAs were not further considered in this review of artificial destratification methods.

Selective withdrawal methods can be effective for mitigation of CWP and water quality issues downstream of reservoir (US Army Corps of Engineers, 1986; Sherman, 2000; Preece, 2004), but they have minimal benefit for the reservoir itself. Additionally, selective withdrawal methods are inherently limited to mitigating either the release of potentially toxic cyanobacteria or poor-quality hypolimnion water, as these require withdrawal from two different sections of a stratified water column. Key considerations in regard to selective withdrawal methods can be found in Supplementary Material S1.

Rising bubble plumes are the most common employed technique for artificial destratification. Bubble plumes are created by transporting compressed air through a pipe network to diffusers typically located in the deepest part of a reservoir (Lewis et al., 1991; Ashby and Kennedy, 1993; Visser et al., 1996; Antenucci et al., 2005) (Fig. 2). The air is diffused through small nozzles or holes into plumes of buoyant bubbles, which rise through the water column to the surface. As the plumes rise through the water column, they entrain cold, dense water from the hypolimnion, which then detrains from the rising plume as it reaches neutral buoyancy or the surface of the reservoir (Fig. 2). When water is detrained before reaching the surface of the reservoir, bubbles continue to rise and initiate another plume. Detrained water sinks back through the density field to a depth where it is neutrally buoyant and propagates

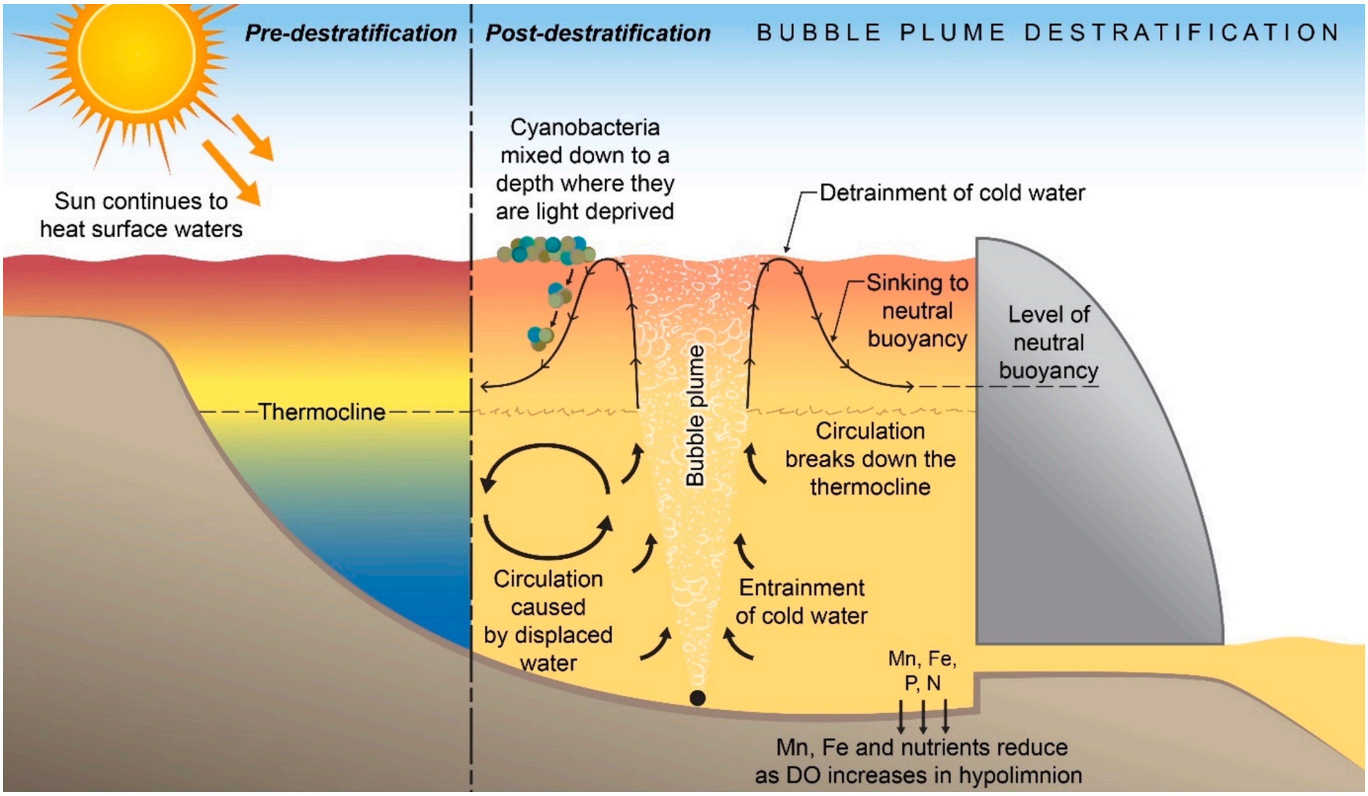


Fig. 2. Conceptual diagram showing bubble plume artificial destratification in a reservoir.

away from the centre of the plume (Mcdougall, 1978; Lewis et al., 1991; Schladow, 1993). This movement of water causes local mixing in the vicinity of the bubble plume, dismantling the water density structure and breaking thermal stratification (Patterson and Imberger, 1989; Ashby and Kennedy, 1993). The distinction between bubble plume destratification and hypolimnetic oxygenation through bubble plumes

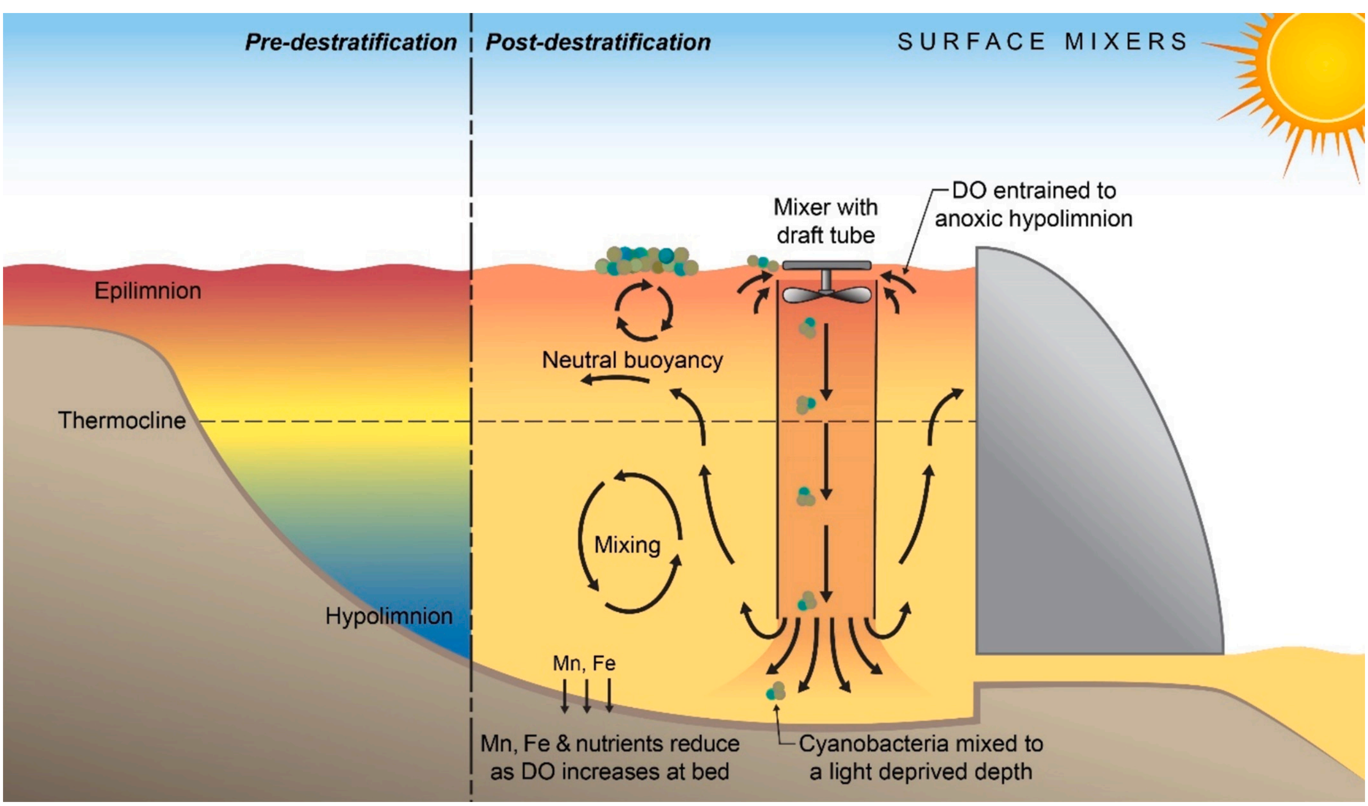


Fig. 3. Conceptual diagram showing artificial destratification with mechanical mixers installed at the surface.

(Gantzer et al., 2009; Liboriussen et al., 2009; Bryant et al., 2011; Munger et al., 2016; Preece et al., 2019) should be noted, as bubble plume destratification is commonly referred to as "aeration" in literature. Hypolimnetic oxygenation aims to oxygenate low DO hypolimnion water without breaking stratification by injecting bubbles small enough to dissolve before reaching the surface. This may be beneficial in reservoirs or lakes where the hypolimnion provides habitat for fish species that prefer cooler water (Bryant et al., 2024).

Artificial destratification can also be achieved using mixers mounted near the surface of the reservoir (Symons et al., 1970; Toetz, 1977; Suter and Kilmore, 1990; Stephens and Imberger, 1993; Lewis, 2004; Hill et al., 2008; Visser et al., 2016) (Fig. 3). These systems generally use a raft-mounted impellor located just below the surface of the reservoir to either jet surface water into deeper reservoir layers (Toetz, 1977), or to draw-up water towards the surface (Symons et al., 1970). The impellors are driven by a motor powered directly on the raft or nearby on shore. Mechanical mixers can be accompanied by a draft tube that extends below the raft to direct the water jet to a desired depth (Brookes et al., 2008). Much like bubble plumes, the entrained water responds to the water density variations and rises or sinks to a depth of neutral buoyancy, creating a mixing effect and destratifying temperature gradients in the vicinity of the system (Fig. 3).

3. Methodology for reviewing the effectiveness of artificial destratification techniques

3.1. Considered reservoirs

An extensive global literature review was conducted of reservoirs where artificial destratification infrastructure comprising bubble plumes or mechanical mixers were used. The review considered information from journal papers, conference papers, reports, previous systematic reviews as well as communications with reservoir managers. As part of this review, 138 different artificial destratification systems in 114 different reservoirs worldwide were identified. Most of the identified destratification systems were installed in reservoirs in Australia (53 %) and the United States (31 %). This large representation is a symptom of previous reviews undertaken in Australia and the United States (Pastorok et al., 1981; McAuliffe and Rosich, 1989). While it is recognised that these systems are used globally, literature pertaining to these systems was not as readily available.

The effectiveness of destratification systems, in terms of their ability to mitigate the impacts of reservoir stratification, was extracted and collated into a "Destratification Library" (see supplementary materials). The library contains key information about each destratification system (e.g., air flow rate) and the reservoir in which it was implemented (e.g., capacity, depth) as well as their effectiveness in influencing thermal stratification, DO concentrations, iron and manganese occurrence, and potentially toxic cyanobacteria. This library is intended to be updated as additional information on destratification systems becomes available.

In this review, reservoirs were categorised as either large or small based on reservoir capacity (total volume of the reservoir at full storage), and deep or shallow based on reservoir depth. These key physical characteristics were considered independently due to their theoretical influence on effectively mitigating the impacts of stratification. A reservoir could be considered as shallow in terms of mean depth, and as large in terms of the reservoir capacity (and vice versa).

Mean depth is considered a key physical characteristic due to its impact on artificial destratification in terms of the energy required to destratify (Schladow, 1993). Deeper reservoirs may increase the effectiveness of destratification methods for mitigating cyanobacterial growth, by mixing cyanobacteria to depths at which they are deprived of light (Visser et al., 2016). Considering the large dam definition by (ICOLD, 2011) in terms of dam height, deep reservoirs were classified as those with a mean depth >15 m, and shallow reservoirs with a mean depth <15 m.

Reservoir capacity is another key physical characteristic due to its direct impact on the amount of energy required to break stratification (Schladow, 1993). ICOLD (2011) defines large dams as being >3000 ML; however, this was found to be too small for the purposes of this review. Based on the observed effectiveness of artificial destratification for addressing thermal stratification from this review (see Section 4), a threshold of 100,000 ML was found to be more appropriate for defining large and small reservoirs, respectively.

3.2. Criteria for destratification success

This section discusses generalised definitions of success based on qualitative information reported in the literature, as well as quantitative metrics when available. In each case, success of the desired outcomes was subjectively stated by the reservoir managers or assigned by the authors.

3.2.1. Thermal stratification mitigation

Thermal stratification mitigation is the core measure of success of an artificial destratification system. Thermal destratification would ideally result in isothermal conditions in the reservoir water column (i.e., the same temperature at the surface and bed); however, in practice, this has been rarely achieved. Fluctuations in surface water temperatures caused by natural diurnal variations in heating and cooling make it difficult to achieve a constant temperature gradient across the reservoir depth at any one time. Additionally, there is often a lag between heating at the surface and heating at the bed. In this review, and where quantitative data was available, near-isothermal conditions of <2 °C temperature difference between the surface and reservoir bed were considered to represent a destratified reservoir (e.g., Haynes, 1975). In cases where quantitative data was not available, the qualitative reported success of destratification was used.

A previous method for defining successful thermal destratification considered the variation between the temperature in the river system upstream and downstream of a reservoir (Sherman, 2000; New South Wales Office of Water, 2011). The upstream temperature was considered a surrogate for the "natural" temperature that should be targeted. However, this approach may not be appropriate due to variability and time lags between upstream and downstream river temperatures resulting from the large reservoir water body acting as a heat sink between them. This approach was therefore not considered in this review.

Thermal stratification can be assessed in terms of the stability of a reservoir, based on the density structure through the water column. The Schmidt stability index (Schmidt, 1928; Idso, 1973) is a common method of quantifying this, representing the work required to transform the stratified density distribution into a uniform distribution without any removal of heat. Bubble plume destratification models (Patterson and Imberger, 1989; Schladow, 1993) make use of this concept to quantify the mechanical energy input required over a period of time (i.e., work) to create a uniform distribution in a stratified reservoir. This approach requires knowledge of the stage-storage relationship of a reservoir, which was not readily available in the literature. The goal of this review was to synthesize the available information into comparable outcomes, and, as such, this approach was not considered for this review.

CWP was considered as to have been mitigated at sites displaying effective thermal destratification. This approach does not address the full complexity of CWP, and the definition of successful CWP mitigation remains an area of research that needs to consider both practical capabilities and realistic ecological targets that is beyond the scope of this review (Preece and Jones, 2002; Astles et al., 2003; Michie et al., 2020a, 2020b, 2023).

3.2.2. Increase in dissolved oxygen and decrease in soluble metals

Dissolved oxygen as well as iron and manganese concentrations are indicative of in-reservoir water quality. Generally, an increase in DO and decrease in iron and manganese concentrations associated with artificial destratification can be considered a measure of success of reservoir water quality mitigation. DO, iron and manganese concentrations are closely linked in a reservoir environment, due to the elevated redox potential in a stratified environment (Pastorok et al., 1981; Müller and Mitrovic, 2015; Müller et al., 2016). Successful actions to increase DO generally result in a decrease of iron and manganese concentrations.

In this review, mitigating poor water quality as a result of reservoir stratification was considered as successful if DO levels were raised consistently above 5 mg/L throughout the water column (homogenous conditions) (Visser et al., 1996). DO of <5 mg/L at any depth was found to negatively affect fish (Gibbs and Howard-Williams, 2018). Ashby and Kennedy (1993) considered a similar threshold for successful water quality improvements with DO between 4 and 6 mg/L at the bed and a decrease in iron and manganese concentrations, as manganese is released at the sediment-water interface when DO < 5 mg/L and soluble iron is released when DO < 2 mg/L.

Nutrient dynamics, in particular those of nitrogen and phosphorus, play a key role in reservoir water quality. Water quality is directly impacted by internal nutrient loading occurring under hypoxic or anoxic conditions (Beutel et al., 2008; Müller et al., 2016) as a result of stratification. Due to the complexity of nutrient cycling in these environments, and the influence of external loading (Jeppesen et al., 2009; Hamilton et al., 2016; Biswas et al., 2021), the direct outcomes of destratification on nutrient dynamics were not considered in this review. However, effectively increasing DO concentrations in a reservoir can be considered a surrogate measure of the potential for reducing internal nutrient loading. Similarly, the impacts of DO on the release of mercury under highly reduced conditions (Beutel et al., 2008) is acknowledged but not directly addressed in review.

3.2.3. Reduction in cyanobacterial populations

Defining successful cyanobacteria mitigation solely because of reservoir destratification is difficult due to the number of external factors that impact cyanobacterial growth, including nutrient input and available sunlight. Nevertheless, total cyanobacterial biovolume or cell density should be measured at both the offtake level and throughout the water column.

The Australian Guidelines for Managing Risks in Recreational Water (NHMRC, 2008) outlines quantitative thresholds for potentially harmful levels of cyanobacteria. "Green" level alert represents surveillance mode, where levels do not exceed a threshold of 5000 cells/mL of *Microcystis aeruginosa* or a total cyanobacteria biovolume of 0.4 mm^3/L. Successful operation of a destratification system should maintain conditions below these recreational thresholds.

Consideration must be given to concentrations at varying depths as artificial mixing distributes the normally surface-dominant cyanobacteria throughout the water column. To reduce cyanobacterial populations, cyanobacteria should be mixed to a depth in the water column at which they become light deprived. Below this depth, respiratory losses overcome the photosynthetic growth, theoretically reducing the potential for net growth (Visser et al., 2016; Gibbs and Howard-Williams, 2018; Huisman et al., 2018). For artificial destratification to be successful, the system should be capable of mixing cyanobacteria deeper than the euphotic depth (depth at which only 1 % of incident light remains). As a rule, the thermocline (or the depth of the reservoir where thermal destratification is achieved) should be more than three times the euphotic depth to mitigate cyanobacterial growth (Sherman, 2000; Gibbs and Howard-Williams, 2018).

Artificial mixing can be used to control cyanobacterial populations by several other mechanisms. Bubble plumes or mechanical mixers can break surface scums of potentially toxic cyanobacteria, such as *Microcystis aeruginosa* and *Dolichospermum circinale*, which allows other phototrophs to access sunlight. Once the water column is well-mixed, cyanobacteria lose their competitive motility advantage that was achieved by buoyancy control. Artificial mixing also reduces the sedimentation rate of larger benign algae, such as diatoms, and allows them to compete more effectively with cyanobacteria for sunlight. Finally, successful increase in bottom DO concentrations can limit algal growth in general by reducing the rate of nutrient resuspension from the sediment (Huisman et al., 2018).

Where a reservoir has an active destratification system, two common metrics can be considered to define successful mitigation of potentially toxic cyanobacteria. The first is an overall reduction in algal biomass (generally determined by measuring chlorophyll-a concentrations or cells densities) and the second is a shift in dominance from potentially toxic cyanobacteria to more benign species such as green algae and diatoms (Visser et al., 1996). More generally though, a destratification system can be considered successful if it results in a reduction of cyanobacterial biovolume, and unsuccessful if it does not decrease, or even increase, cyanobacterial biovolume because of operational activities. Given that cyanobacterial growth is impacted by many factors other than thermal stratification, it is necessary to monitor cyanobacteria for several years following the implementation of destratification to determine whether this intervention was successful at reducing cyanobacterial populations.

3.3. Effectiveness criteria for reservoir destratification

A lack of consistent definitions and qualitative discussion used throughout the reviewed literature made it difficult to directly compare successful mitigation of reservoir stratification impacts among the reservoirs reviewed. Data collection for each reservoir varied, so quantitative measures for direct comparison were not consistently available. This review therefore used more generalised definitions of successful mitigation of the impacts of reservoir stratification that could be applied to most cases reviewed and that were overall consistent with criteria identified in Section 3.2.

Success, or lack thereof, was considered as four categorical outcomes pertaining to the impacts of reservoir stratification:

- Mitigating thermal stratification (and mitigating CWP)
- Mitigating in-reservoir DO concentrations
- Mitigating the release of soluble metals
- Reducing potentially toxic cyanobacterial populations

For each of the above listed outcomes, each reviewed destratification system was considered as successful, partially successful or unsuccessful in mitigating their respective impacts as per the criteria listed in Table 1. Net negative outcomes were considered for DO, soluble metals and cyanobacteria, where destratification operations were observed to further degrade water quality (with respect to each specific impact, as detailed in Table 1). No net negative effect was considered for mitigating thermal stratification, as it is not possible for destratification to increase existing stratification.

4. Assessing the effectiveness of bubble plume destratification

All background information and data presented in this section is also provided in the Destratification Library (supplementary materials). Bubble plume systems were found to be the most utilised artificial destratification systems, comprising 84 % (116 cases) of the systems reviewed. The preference for these systems is typically associated with their perceived low costs (compared to retrofitting selective withdrawal infrastructure); however, the successful design and operation of these systems has varied. Most bubble plume systems reviewed were installed in small reservoirs by capacity (80 %) and reservoir depth (74 %).

Fig. 4 shows the overall outcomes for each of the impacts assessed for bubble plume destratification.

Table 1
Criteria for effective mitigation of the impacts of reservoir stratification.

Outcome	Impact			
	Thermal stratification	Dissolved oxygen	Soluble metals	Cyanobacteria
Successful	System capable of disrupting stratification and/or maintaining isothermal conditions throughout the reservoir and water column	System disrupted DO stratification, increased DO in previously hypoxic or anoxic water and created near-homogenous DO conditions throughout the water column	System successfully reduced high concentrations of iron and/or manganese in the reservoir	System successfully reduced potentially toxic cyanobacterial populations and did not lead to a significant increase in other algal populations
Partially successful	System capable of reducing thermal stratification, but incapable of creating isothermal conditions throughout the reservoir, or system was able to partially mitigate CWP downstream	System increased DO close to the bed, however failed to either create homogenous DO conditions throughout the water column or to increase DO to a level deemed 'acceptable' for water quality purposes	System reduced iron and/or manganese concentrations but not to an 'acceptable' level for water quality purposes	System reduced potentially toxic cyanobacterial populations but led to an increase in other algal populations, or system reduced potentially toxic cyanobacteria populations but didn't eliminate the issues it caused
Unsuccessful	System incapable of disrupting stratification and/or maintaining isothermal conditions throughout the reservoir and water column	System did not increase DO concentrations near the bed and throughout the water column	System did not reduce iron and/or manganese concentrations in the reservoir	System did not reduce potentially toxic cyanobacterial populations in the reservoir
Net negative impact		System operation resulted in a decrease in DO in the reservoir	System operation resulted in an increase in iron and/or manganese concentrations in the reservoir	System led to an increase in potentially toxic cyanobacterial populations, or a significant increase in overall algal populations, due to increased nutrient supply

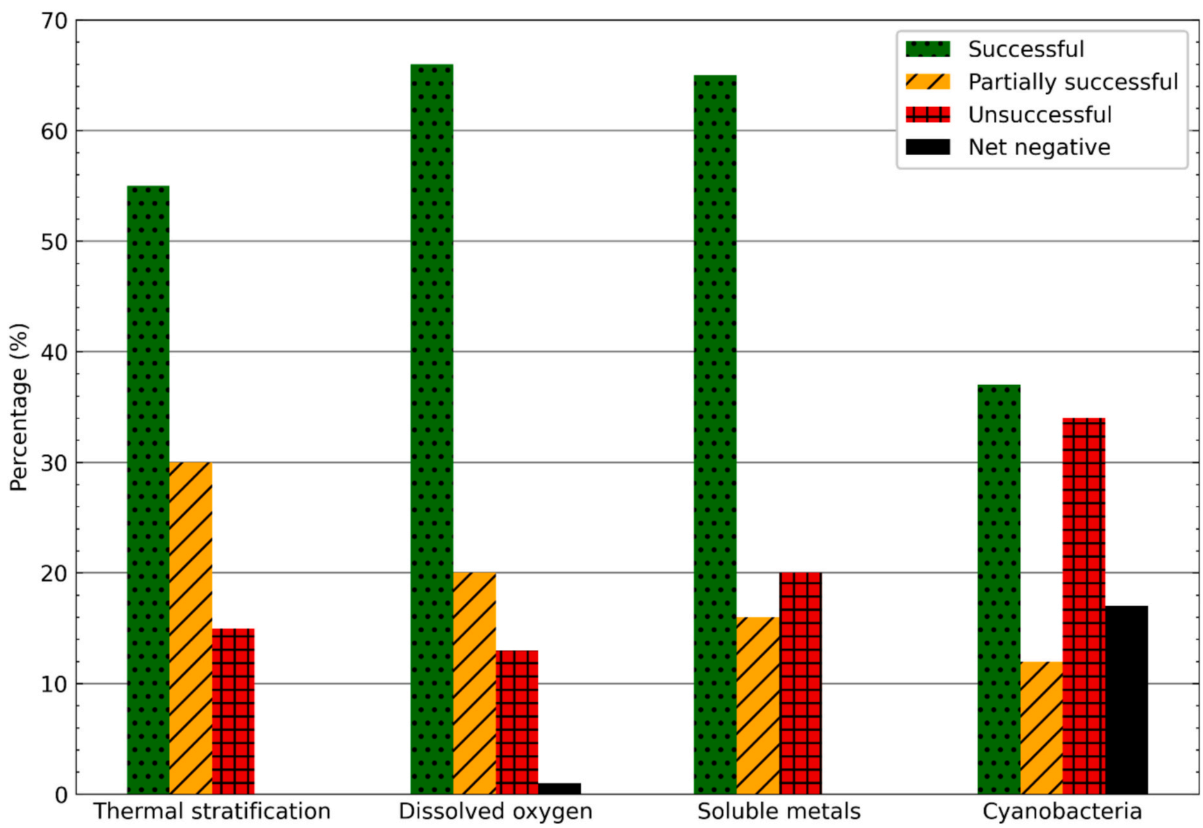


Fig. 4. Percentage of successful, partially successful, unsuccessful and net negative outcomes on the impacts of stratification as a result of bubble plume destratification.

4.1. Mitigation of thermal stratification

Bubble plume systems were found to be an effective way of mitigating thermal stratification and CWP. Where the effects on thermal destratification were reported (94 cases), it was found that 85 % of bubble plume systems either fully or partially mitigated thermal stratification, with 55 % resulting in full destratification and maintaining near-isothermal conditions. Conversely, 15 % of systems did not result in a change in thermal stratification (Fig. 4).

Reservoir capacity was found to be an important factor in the effectiveness of bubble plume systems for mitigating thermal stratification. In small reservoirs with a capacity of <100,000 ML, 91 % of systems were reported to either fully or partially mitigate thermal stratification with 61 % of systems achieving full destratification. In large reservoirs with a capacity >100,000 ML, only 65 % of systems resulted in full or partial mitigation of thermal stratification and only 35 % of systems were capable of full destratification. The primary design parameter for effective bubble plume destratification was the air flow rate used to destratify the reservoir. This is highlighted in Fig. 5a, which shows successful, partially successful and unsuccessful thermal destratification cases in terms of the reservoir capacity of each system as a function of the ratio between the air flow rate (Q_d, as L/s) and volume of the reservoir (V_r, as ML), herein referred to as the volumetric destratification coefficient (*VDC*):

$$VDC = \frac{Q_d}{V_r} \quad (1)$$

For the reviewed reservoirs, as reservoir capacity increases, the volumetric destratification coefficient tends to decrease. As the volumetric destratification coefficient decreases, there are an increasing number of cases where thermal destratification was unsuccessful. This effectively means that bubble plume systems were often under designed for larger reservoirs, i.e., air flow rates appeared to be too low for successful destratification (Fig. 5a).

Fig. 5b shows the effectiveness for destratification with bubble plumes for the reviewed reservoirs as a function of mean depth, and ratio of air flow rate and mean depth (referred to as the depth destratification coefficient). Variations in reservoir mean depth did not appear to impact effective thermal destratification as much as the reservoir capacity. Thermal destratification with bubble plume systems in shallow (mean depth < 15 m) and deep (mean depth > 15 m) reservoirs resulted in similar results with either full or partial mitigation in 88 % and 73 % of cases, respectively, with full destratification reported in 54 % and 55 % of cases, respectively. While the reservoir depth is an important design parameter, results suggest that the effects of water depth are partially offset by the nature of rising bubbles continuing to rise and destratify, regardless of the depth.

4.2. Increase in dissolved oxygen and decrease in soluble metals

Bubble plume destratification systems were found to be an effective way of increasing DO in reservoirs. Out of the 79 cases that reported DO observations after destratification, 86 % reported increased DO concentrations and 66 % reported full restoration of DO concentrations to an acceptable level throughout the water column. Only 13 % of cases reported that bubble plume destratification resulted in no positive effect on DO concentrations, with one case reporting an undesirable decrease in DO concentrations (Fig. 4).

As with thermal destratification, reservoir capacity was found to impact the success of destratification for increasing DO concentrations. For large reservoirs, only 36 % of cases reported full restoration of DO concentrations to an acceptable level, and 64 % reported partial effectiveness. These numbers increased to 72 % and 91 % respectively for small reservoirs. Reservoir mean depth was found to have a less significant impact on the successful increase of DO concentrations, compared to reservoir capacity. For deep reservoirs, 53 % reported full restoration of DO concentrations and 76 % reported partial or full effectiveness, while this increased to 68 % and 88 % respectively for shallow reservoirs.

Similar success was reported for reducing iron or manganese concentrations through bubble plume destratification, albeit in a smaller number of cases. Where these effects were measured (51 cases), 65 % of systems were reported to reduce iron or manganese concentrations to an acceptable level for water quality. A further 16 % reported at least some decrease in soluble metal concentrations. Conversely, 20 % of cases reported no decrease in iron or manganese concentrations (Fig. 4). The results confirm a key link between increasing DO and successfully decreasing iron and manganese concentrations. For cases where DO was successfully increased, 85 % of reported bubble plume destratification systems also successfully decreased iron and manganese concentrations.

Trends in successful increase in DO concentrations and reduction in iron and manganese concentrations (Fig. 6) are similar to those of successful thermal destratification (see Fig. 5), with regards to reservoir capacity and the volumetric destratification coefficient. This highlights the link between successful thermal destratification and successful mitigation of DO, iron and manganese-related water quality impacts. For systems where full thermal destratification was achieved, 77 % reported full restoration of DO concentrations and 90 % reported at least some increase in DO concentrations. This result demonstrates the capability of bubble plume destratification to mitigate concurrent impacts of reservoir stratification.

The overall trends between successful increases in DO concentration with increasing volumetric destratification coefficient would be even stronger without three unsuccessful cases in reservoirs of <5000 ML capacity (Fig. 6a). In two of these reservoirs, low DO levels were attributed to high biological oxygen demand (BOD) (Pastorok et al., 1981; Osgood and Stiegler, 1990). In such small capacity reservoirs with high BOD, the presence of stratification might limit hypoxia or anoxia to the area below the thermocline, while mixing might have an undesired effect, allowing hypoxic or anoxic conditions to form throughout the

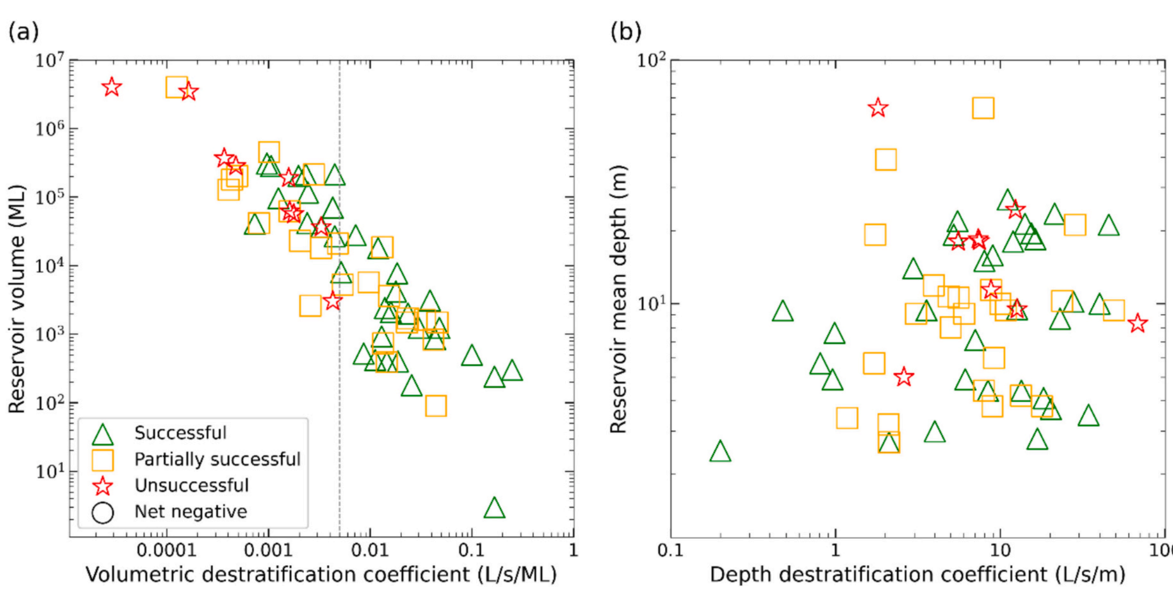


Fig. 5. Success of bubble plume thermal destratification: (a) as a function of reservoir capacity and the volumetric destratification coefficient, with a dashed line indicating a volumetric destratification coefficient of 0.005 L/s/ML; (b) as a function of reservoir mean depth and the depth destratification coefficient.

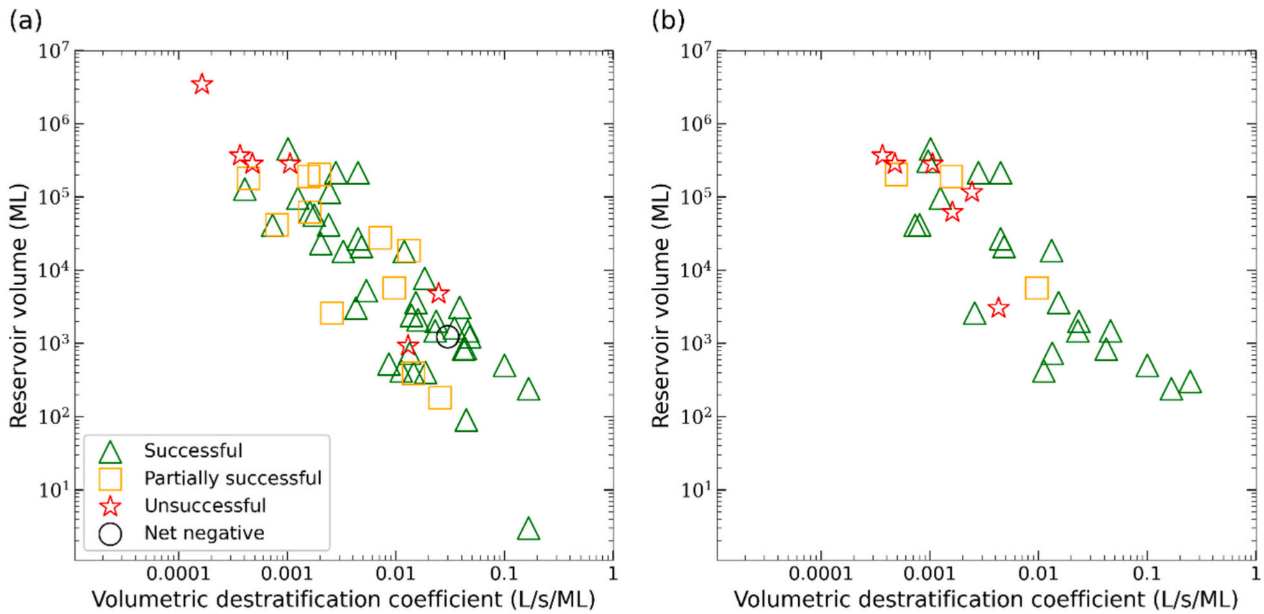


Fig. 6. Success of bubble plume destratification to improve water quality as a function of the reservoir volume and volumetric destratification coefficient in terms of (a) increasing DO concentrations; and (b) decreasing iron or manganese concentrations.

entire reservoir. In the third case, the lack of success in achieving thermal destratification was attributed to the ineffectiveness at increasing DO (McAuliffe and Rosich, 1989).

4.3. Reduction in cyanobacterial populations

The effectiveness of bubble plume destratification for reducing potentially toxic cyanobacterial populations (this outcome herein referred to as "reducing cyanobacterial populations") was found to be significantly more varied compared to the other outcomes previously discussed. Where the effects on cyanobacteria populations were reported (59 cases), only 37 % reported successful reduction in populations, while 12 % reported partial success. A further 34 % reported no reduction of cyanobacterial populations, and 17 % found a net negative effect with increases to cyanobacterial populations (Fig. 4).

Physical reservoir characteristics on their own did not appear to have a significant impact on successful reduction of cyanobacterial populations. Mean reservoir depth has theoretically the greatest impact on reduction of cyanobacterial populations, as a deeper reservoir provides a greater water column depth over which the normally buoyant cyanobacteria can be mixed, limiting light and therefore growth. For reservoirs deeper than 15 m, 54 % reported successful or partially successful reduction of cyanobacterial populations and 46 % reported no reduction or a net negative effect. For reservoirs shallower than 15 m, these outcomes were reported in 47 % and 53 % of cases, respectively. For reservoirs with a capacity >100,000 ML, 30 % reported successful or partially successful reduction of cyanobacterial populations and 70 % reported no reduction or a net negative effect. For reservoirs with a capacity <100,000 ML, these outcomes were reported in 53 % and 47 % of cases, respectively.

Successful thermal destratification should theoretically promote successful reduction of cyanobacterial populations. In cases where thermal destratification was successfully achieved, 63 % of cases reported successful or partially successful reduction of cyanobacterial populations and 37 % reported no reduction or a net negative effect. Conversely, in cases where thermal destratification was not achieved, 100 % of cases (n = six) reported no reduction or a net negative effect on cyanobacteria. While success was varied, these results demonstrate the importance of achieving thermal destratification for improving cyanobacteria outcomes, as well as the risks associated with unsuccessful destratification. Fig. 7 further demonstrates this trend, showing varied success over a range of reservoir sizes, trending towards unsuccessful or net negative impact outcomes as the volumetric destratification coefficient decreases, which is consistent with decreasing thermal destratification success (Fig. 7a). Similarly, when only considering cases where thermal destratification was successful (Fig. 7b), the number of cases of successful or partially successful reduction of cyanobacterial populations increased.

Theoretically, cases where thermal destratification has been achieved and the reservoir is relatively deep should result in the highest chance of cyanobacteria reduction. These theoretical considerations were confirmed by the analysed data for cases where thermal destratification was successfully achieved. For the reservoirs with depth < 15 m, 57 % of cases reported successful or partially successful reduction of cyanobacteria populations and 43 % reported no reduction or a net negative effect. Conversely, in reservoirs with depth > 15 m, 75 % of cases reported successful or partially successful reduction of cyanobacteria populations and only 25 % reported no reduction or a net negative effect (Fig. 7c and d). Successful reduction of cyanobacteria populations was most likely for cases where thermal destratification was successful and reservoir mean depth was >15 m (Fig. 7c), highlighting the benefits of deeper reservoirs for mitigation of cyanobacteria when thermal destratification is successfully achieved.

5. Assessing the effectiveness of mechanical mixer destratification

Mechanical mixers (primarily surface-based) have been used in a much smaller sample of reservoirs compared to bubble plume systems, comprising 16 % of the systems reviewed (22 cases) (see "Destratification Library" in supplementary material). Mechanical mixers have typically been used in smaller reservoirs, with 86 % and 77 % of the systems being installed in small reservoirs by capacity and mean depth, respectively. All but one of the reviewed mechanical mixer systems were installed in reservoirs with a capacity of 124,000 ML or less, while for the much larger Douglas Dam (~1,730,000 ML), the surface mixers were specifically used to promote local mixing near the offtake and were not designed for destratification (Mobley et al., 1995).

Fig. 8 shows the overall outcomes for each of the impacts assessed for mechanical mixer destratification.

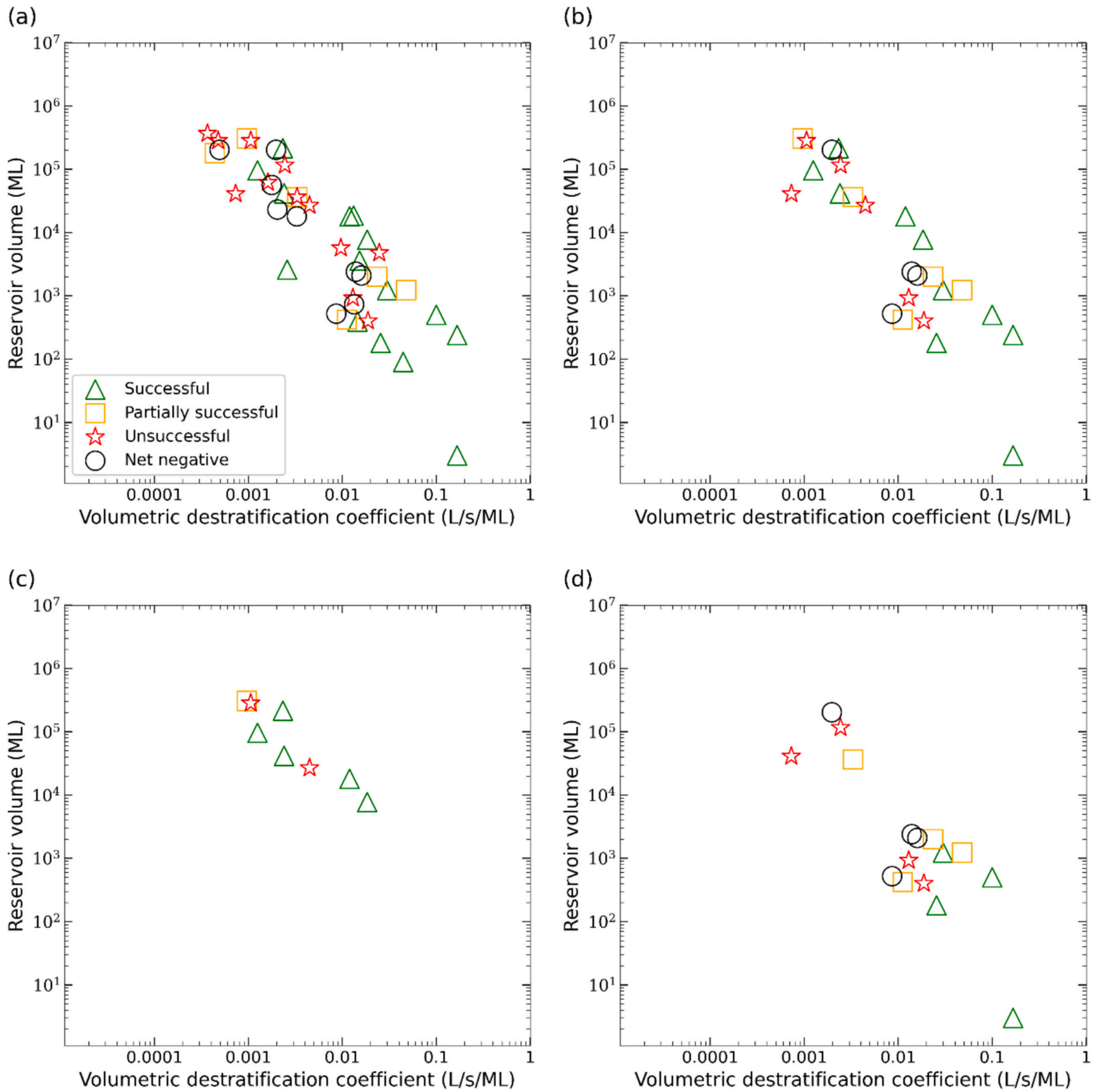


Fig. 7. Success of bubble plume destratification for suppressing cyanobacteria as a function of reservoir capacity and the volumetric destratification coefficient: (a) all reported cases; (b) all cases where successful thermal destratification has been achieved; (c) cases where successful thermal destratification has been achieved and reservoir mean depth is >15 m; and (d) cases where successful thermal destratification has been achieved and reservoir mean depth is <15 m.

Given the significantly smaller number of cases available for mechanical mixers and the low representation in large reservoirs by capacity and deep reservoirs by mean depth, this section does not differentiate the reservoir size for the effectiveness of mechanical mixers. As presented in the following sub-sections, it was difficult to draw key links between system design, reservoir physical characteristics and concurrent outcomes such as thermal destratification and increases in DO concentrations as was found in the bubble plume assessment (Section 4). For the small number of cases (12) that reported design jet flow rates, Fig. 9 summarises the key information on the effectiveness of mechanical mixers to address reservoir issues of thermal stratification (Fig. 9a), DO concentrations (Fig. 9b), manganese and iron concentrations (Fig. 9c) and cyanobacteria (Fig. 9d). These outcomes are plotted as a function of the reservoir capacity and volumetric destratification coefficient for mechanical mixers that used the water flow rate instead of the air flow rate for bubble plume systems. For all these parameters, there was strong variation in effectiveness with mixer performances, different mixer designs and reservoir characteristics, making it difficult to draw definitive conclusions. Some mixers worked well to mitigate some thermal stratification issues, while simultaneously not addressing parallel issues. Despite the inconclusive results in Fig. 9, key information for each parameter is summarised and critically assessed in the following sub-sections.

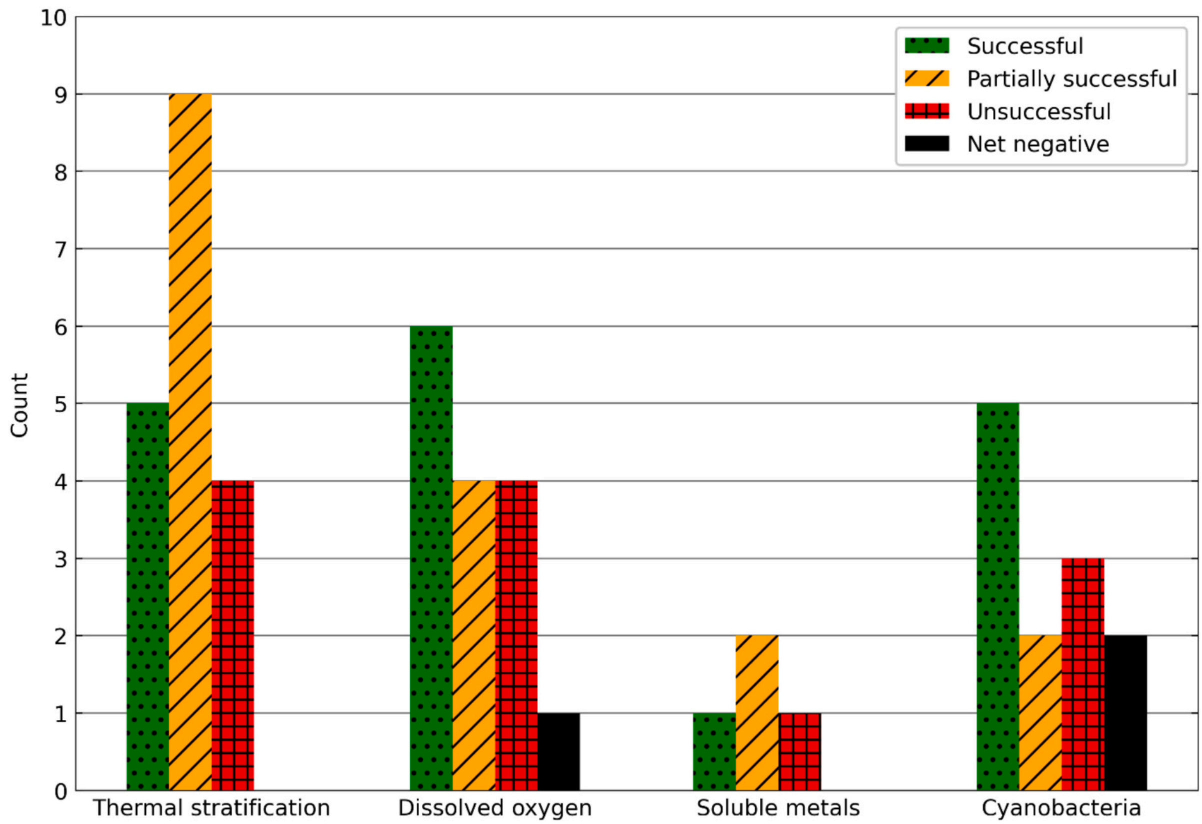


Fig. 8. Percentage of successful, partially successful, unsuccessful and net negative outcomes on the impacts of stratification as a result of mechanical mixer destratification.

5.1. *Mitigation of thermal stratification*

Mechanical mixers had varied success in mitigating thermal stratification (Fig. 9a). Of the 18 cases that reported the effects of mechanical mixers on thermal stratification, five reported full thermal destratification, nine reported partial thermal destratification, and four reported unsuccessful thermal destratification (Fig. 8). The largest reservoir in which full thermal destratification was achieved had a capacity of approximately 30,000 ML.

Four of the five systems that reported full thermal destratification utilised draft tubes. The other successful system utilised an unconfined jet (i.e., no draft tube) in a small reservoir (Ham's Lake) of 1150 ML with a mean depth of 2.9 m (McClintock and Wilhm, 1977; Toetz, 1977). These results, albeit from a small number of samples, suggest that a draft tube is needed to mitigate thermal stratification, except in very shallow reservoirs.

5.2. *Increase in dissolved oxygen (and decrease in soluble metals)*

Varied success in operating mechanicals mixers was also found in terms of increasing DO concentrations (Fig. 9b) and reducing manganese and iron concentrations (Fig. 9c). In the 14 studies that reported the effects on DO concentrations, six systems increased DO concentrations to an acceptable level within the reservoir, four were found to either increase DO by a smaller than acceptable amount or improve DO concentrations downstream, four were reported to have no effect on DO concentrations, while one system reported a decrease in DO concentrations following operations (Fig. 8).

There was no clear link between reservoir size and the effectiveness of mechanical mixers for successfully increasing DO concentrations for reservoirs ranging from 2.9 to 14 m mean depth, and from 89 to 30,140 ML capacity. Similarly, there was no clear trend for cases that reported a lack of success in increasing DO concentrations for reservoirs ranging from 2.1 to 7.6 m mean depth and with 121 to 111,720 ML capacity. There was also no clear link between mitigating thermal stratification and DO concentrations. Of the five cases that reported full thermal destratification, two cases reported no changes to DO concentrations, two reported successful increases in DO concentrations, and one case reported minor changes but not to an acceptable level. For the four systems that were unable to achieve any thermal destratification, two reported no changes to DO concentrations and two reported full restoration of DO concentrations to an acceptable level.

Similar inconclusive results were found for reducing iron and manganese concentrations with only four cases reporting outcomes (Fig. 8). Two of these four cases reported outcomes aligned with the effects on DO concentrations, while there was no link for the expected outcomes of the other two cases.

5.3. *Reduction in cyanobacterial populations*

Mechanical mixers were found to have varied success in reducing cyanobacteria populations in reservoirs (Fig. 9d). Of the twelve cases that reported on cyanobacteria populations, only five reported successful reduction of cyanobacteria populations, two reported partial success, three reported no improvements, and two reported a net negative effect (Fig. 8). There were no clear links between the reservoir sizes and depth and cyanobacteria mitigation.

However, a link seems to exist between the effectiveness of thermal destratification and the effectiveness in reducing cyanobacterial populations. Two of the five cases that reported successful reduction of cyanobacteria populations reported full thermal destratification and one of these cases reporting partial thermal destratification. These three cases were all in relatively small reservoirs with capacity ≤1554 ML. The link between unsuccessful or partially successful thermal

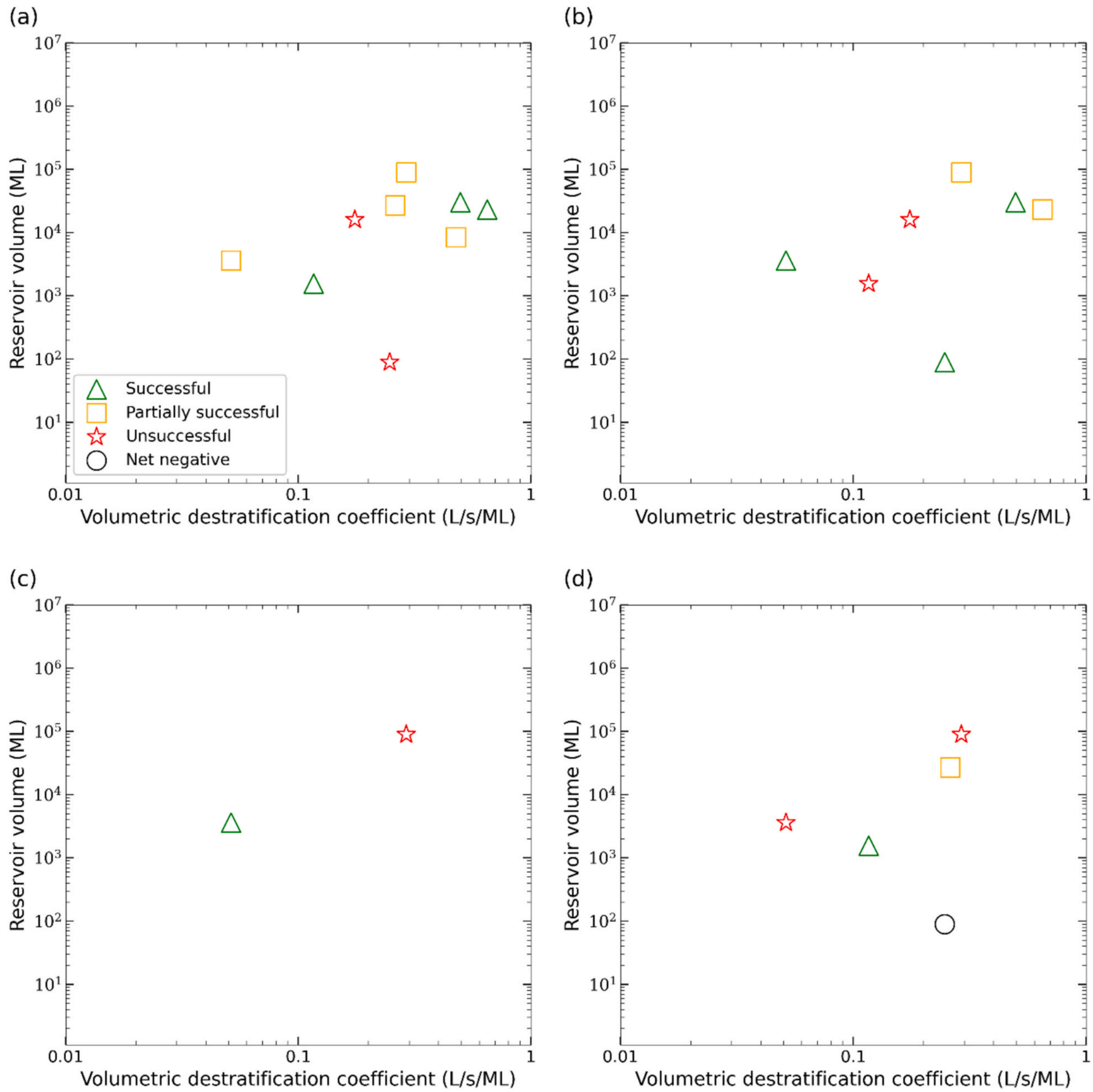


Fig. 9. Success of mechanical mixer destratification as a function of reservoir capacity and volumetric destratification coefficient: (a): Success for thermal destratification; (b) Success for increasing DO concentrations; (c) Success for decreasing manganese or iron concentrations; and (d) Success for suppressing cyanobacteria.

destratification and unsuccessful cyanobacteria reduction was also observed for mechanical mixer systems. The three cases that reported no cyanobacteria reduction reported partial thermal destratification and the two cases that reported an increase in cyanobacteria populations reported unsuccessful thermal destratification.

The effects of draft tubes were also pronounced, with all five cases where cyanobacteria were successfully suppressed utilising a draft tube. For the three cases where a draft tube was used and cyanobacteria remained an issue, thermal destratification was not achieved. In two of these cases, water was pumped from the bed to the surface, likely drawing nutrients from the hypoxic or anoxic section near the bed of the reservoir and promoting cyanobacterial growth.

6. Discussion

The results of this review on artificial destratification systems confirm their general effectiveness as a means of mitigating the impacts of reservoir stratification. While there were cases of the systems not being effective, the ineffectiveness may be attributed to under-designing the systems. Additionally, the review highlighted important design and operation considerations that could improve the effectiveness of newly implemented systems, particularly for bubble plume destratification. Lessons learned on the effectiveness of artificial destratification systems and design recommendations are discussed in the following subsections. Further details on the advantages and disadvantages of artificial destratification systems are summarised in the Supplementary material.

6.1. *Effectiveness of artificial destratification systems*

6.1.1. *Bubble plumes vs. mechanical mixers*

The review demonstrated that, overall, bubble plume destratification (Fig. 4) was more effective as a solution than mechanical mixers (Fig. 8), in particular in larger capacity reservoirs. There was limited demonstrated success of mechanical mixers to thermally destratify larger reservoirs, likely due to the limited localised mixing created by the mixer jet. Draft tubes were key to successful outcomes of mechanical mixers in terms of both thermal destratification and improvements to water quality. The need for draft tubes highlights an inherent flaw with mechanical mixers in large reservoirs, especially those with significant depths. While the draft tube can confine the jet and ensure it reaches the bed, it requires a more complicated installation in deeper reservoirs where internal currents place a large load on the draft tube. Additionally, the draft tube must be able to account for variations in reservoir water levels. Considering these limitations, mechanical mixers appear better suited for localised mixing and not destratification of the whole reservoir. The system at Douglas Dam (Mobley et al., 1995) provides an example of this, where the system is only operated near the dam wall to mix surface waters to the depth of the offtake for release events (a pseudo form of selective withdrawal). This, however, is not a practical solution if cyanobacterial blooms are present, as the cyanobacteria could be released downstream. Further consideration on the use design, installation, operation and cost of mechanical mixers is provided in the Supplementary Material S3.

By comparison, bubble plumes were found to be more effective at artificial destratification in both large and small reservoirs, highlighting the scalability of bubble plume systems to match reservoir characteristics. The scalability is closely linked with the physical processes of rising bubbles, which can entrain and detrain water more than once while rising through the water column, making it effective even for deep reservoirs (Fig. 2). Effective reservoir thermal destratification requires sufficient air flow rate (Fig. 5a), such that the energy input (air flow rate) can overcome the stratifying effects of surface water heating.

6.1.2. Bubble plume artificial destratification success

Bubble plume artificial destratification was shown to be able to remediate the impacts of stratification within the reservoir environment. Moreover, a correctly designed bubble plume destratification system is capable of concurrently mitigating thermal stratification (and consequently, mitigating CWP), increasing DO concentrations, lowering iron and manganese concentrations and reducing potentially toxic cyanobacterial populations. These effects are not only beneficial for the reservoir, but also for the downstream river system as full thermal destratification and the reduction of cyanobacteria populations in the reservoir can effectively mitigate cold, poor water quality hypolimnetic releases and the transport of cyanobacteria downstream.

The review demonstrated the link between thermal destratification and mitigating issues associated with anoxia and hypoxia due to a lack of water mixing below the thermocline. In systems that were able to break the thermocline and stratification or inhibit the onset of stratification entirely, DO concentrations were commonly restored to an acceptable level throughout the water column. Conversely, DO concentrations often remained low when thermal destratification was not achieved due to a lack of mixing through the water column.

By restoring DO concentrations throughout the water column including next to the bed, other water quality issues were often mitigated. Iron and manganese concentrations were commonly reported to reduce with increased DO concentrations, highlighting the concurrent benefits of mitigating thermal stratification in a reservoir as well as the additional benefits to water suppliers due to reduced water treatment costs.

While theoretically achievable, the success of cyanobacterial population reduction through artificial destratification was found to be mixed (Sections 4.3 and 5.3). Successful thermal destratification was found to be an important factor in the success of reducing cyanobacterial populations. This is likely due to both the removal of the competitive advantage buoyant cyanobacteria have over other algae species in a stratified environment and the physical mixing of cyanobacteria to a depth at which it becomes light deprived (Visser et al., 2016). Where thermal destratification was not achieved, increased cyanobacteria concentrations were sometimes observed. In these cases, the artificial destratification systems may have been unable to mix the water column and hypoxic conditions remained near the bed. Under this scenario, nutrients would continue to be released from bed sediments and, while these nutrients would otherwise be contained below the thermocline, the destratification system might facilitate increased cyanobacterial growth by transporting some of these nutrients to the reservoir surface. Additionally, the importance of reservoir depth in conjunction with thermal destratification was highlighted by the increasing success with suppressing cyanobacteria when thermal destratification was achieved in deeper reservoirs. This observation is likely due to the greater depth over which cyanobacteria can be deprived of light.

6.2. Design, operation and maintenance considerations for bubble plume destratification

While the review found that bubble plume destratification was an effective method for mitigating reservoir stratification impacts, limitations and failed cases were noted. The following discussion provides a summary of considerations for both the design and operation of bubble plume systems for reservoir destratification with the intention of assisting future use cases. The discussion provides important recommendations that can aid in the application of bubble plume systems, with further research on bubble plume systems installed in reservoirs recommended for the purpose of establishing general design guidelines.

6.2.1. Design considerations

Air flow rate is a key design parameter for bubble plume systems to achieve full thermal destratification. Importantly, the maximum design air flow rate influences bubble plume design parameters including compressor sizing, pipe sizing, and diffuser nozzle configuration. As shown in this review, a volumetric destratification coefficient (i.e., the ratio of air flow rate in L/s to reservoir capacity in ML) of above 0.005 L/s/ML appears to be sufficient to achieve destratification (Fig. 5a), based on there being no unsuccessful destratification attempts above this threshold. The threshold of 0.005 L/s/ML appears to be adequate for full or partial success in increasing DO concentrations (Fig. 6a; omitting the three specifically discussed cases in Section 4.2), and in reducing iron and manganese concentrations (Fig. 6b). Further, the threshold of 0.005 L/s/ML is also adequate for suppressing cyanobacteria concentrations or blooms in reservoirs with a mean depth > 15 m where thermal destratification is achieved (Fig. 8c) (noting that this is based on a small sample size of n = eight). For the practical design of a bubble plume system, theoretical bubble plume models can be used as a first pass to determine flow rates under static conditions (Mcdougall, 1978; Schladow, 1993). Dynamic numerical models can then be used to refine air flow rates based on the response of a reservoir to temporally varying hydrodynamic and meteorological conditions (Yu et al., 2022, 2024).

Compressor sizing, pipeline sizes, and diffuser nozzle sizes and configurations should consider the pneumatics of bubble plume systems, including pressure losses along the pipeline, hydrostatic pressure of the water above the diffuser, and the pressure required to purge the diffuser if full of water. Additionally, nozzles should be designed to equally distribute air flow rates across the length of the diffuser (Lewis et al., 1991). Consideration should also be given to the costs to install at remote sites, especially large capacity reservoirs that require a large compressor and diffuser setup. Additional power infrastructure may be required to facilitate operations, unless local renewable energy solutions can offset compressor energy usage.

Historically, high operating costs have likely limited the application and success of thermal destratification in large reservoirs, leading to under-designed systems and ineffective destratification. Bubble plume systems should consider the use of renewable energy procurement to offset the long-term operational and environmental costs of reservoir destratification. While renewables come at a high capital cost, there are long-term operational savings, particularly in remote locations. The effects of renewable-only procured energy on operational procedures should also be considered. For example, if solar energy is used, operations may only be available during sunlight hours unless significant battery storage is employed. Further research is required to understand whether intermittent operations could be a less or more efficient means of artificial destratification.

Diffusers situated too close to the reservoir bed may scour bottom sediments, which then entrain in the rising plume and affect water quality by supplying nutrients to cyanobacteria at the surface while also increasing turbidity (Brosnan and Cooke, 1987; Ryan et al., 2001). For larger reservoirs where offtakes may not be located close to the bed, strategic placement of diffusers further above the bed to partially destratify the reservoir may be an effective way of reducing costs, as demonstrated previously (Pastorok et al., 1981; Becker et al., 2006). While mixing to the depth of the diffusers may be beneficial for CWP mitigation, poor quality water may remain an issue in the water below the diffuser (Bryant et al., 2024).

6.2.2. Operation and maintenance considerations

Artificial destratification should be initiated before the onset of stratification in spring, as breaking strong thermal stratification requires a significantly larger amount of energy than simply maintaining a destratified reservoir. Bubble plume systems, however, should be designed with the capability to overcome the strongest expected stratification. In the event of a diffuser line or compressor malfunction, stratification may quickly re-establish, and thus the system must be able to break a stronger thermal stratification than it would in maintenance operation.

Variable speed drive (VSD) compressors should be considered for bubble plume destratification systems. While these come at a higher capital cost compared to fixed speed compressors, they allow for the control of air flow rates, hence reducing long-term operational costs. The energy (i.e., air flow rate) required to destratify a reservoir varies based on the density structure in the reservoir (Schladow, 1993). A VSD compressor would allow for optimised operations by changing air flow rates to maintain a destratified reservoir, but relevant data on density, temperature, and water quality conditions need to be routinely monitored across the depth of the reservoir to guide optimised operation.

While warming the hypolimnion may improve CWP issues downstream, it can negatively impact water quality in the reservoir. At the reservoir bed sediment-water interface, warmer water can increase biogeochemical reactions and subsequently biological oxygen demand (Sherman et al., 2001). These conditions may deplete DO throughout the water column faster than in stratified conditions, resulting in the ongoing release of soluble metals and nutrients. This emphasises the need for adequate destratification, to ensure the re-oxygenation at the surface can compensate for the increased oxygen demand at the bed under warmer conditions.

Intermittent operations may be an effective way of reducing the high operational costs of bubble plume destratification. Some historical examples of intermittent operations have reported negative impacts, particular around the reduction of cyanobacteria populations (McAuliffe and Rosich, 1989; Visser et al., 1996). In these cases, intermittent operations led to an increase in cyanobacterial populations, potentially due to the inconsistent light limitation that continuous operations can provide or the intermittent periods which allowed cyanobacteria to reestablish at the surface. Conversely, some studies suggested that varied operations of bubble plumes, including intermittent operations, can prevent cyanobacteria from acclimating to the new density regime (Steinberg and Zimmermann, 1988). Further research is required to assess the effectiveness of non-continuous operations and to identify optimised operation.

Installation procedures should consider long-term maintenance of diffusers and pipelines. In deeper reservoirs especially, maintenance would preferably be performed at or close to the reservoir surface (as opposed to divers performing maintenance at the bed). A system for raising the installed pipeline and the diffusers from the bed of the reservoir should be considered.

Depending on the bathymetry of a reservoir, it may be nearly impossible to control cyanobacteria with bubble plume destratification, as the euphotic depth may be limited and therefore still provide favourable conditions for growth (Gibbs and Howard-Williams, 2018). Where the system may effectively mitigate thermal destratification, broad expanses of shallow water may allow cyanobacteria to continue dominating surface waters, and even become more of an issue due to entrained nutrients that can increase growth rates (Sherman et al., 2001). If cyanobacteria populations are not controlled through bubble plume destratification, the system may cause more harm than good by mixing potentially toxic cyanobacteria down to a deep offtake that would have otherwise avoided downstream discharging. Similarly, bubble plume destratification may increase the negative effects of an inflow event, by lifting nutrients to the surface that would otherwise settle to the bed, thereby promoting cyanobacterial growth.

7. Conclusion

A critical review of artificial destratification systems in reservoirs has shown that such systems can mitigate the impacts of reservoir stratification by artificially mixing the water column and breaking the thermocline to allow heating and oxygenation at the surface to reach the reservoir bed. This usually inhibits the release of soluble metals and nutrients, improving water quality in and downstream of the reservoir, and mitigates the release of cold water, which can negatively impact riverine ecosystems downstream. Artificial destratification was also found to be capable of suppressing the growth of potentially toxic cyanobacteria, by both removing the buoyancy advantage they have in the stratified environment and physically mixing the normally surface-dominant cyanobacteria to a depth at which they become light deprived.

The two most common artificial destratification systems were evaluated, with mechanical mixers being utilised considerably less frequently (16 %) than bubble plume systems (84 %). Mechanical mixers had varying effectiveness at thermal destratification and mitigation of reservoir water quality impacts, were generally installed in smaller reservoirs and, through this review, deemed to not be viable for scaled destratification operations in large capacity reservoirs. The most effective use for mechanical mixers was in localised operations, where the systems were operated to jet warm water down to the depth of an offtake during water release events.

Bubble plume systems have been effectively used across a range of reservoir sizes, highlighting their feasibility for concurrently mitigating all the impacts of reservoir stratification. Their successful mitigation of thermal stratification was closely linked with correctly designed air flow rates based on reservoir capacity, which was represented by a ratio of the air flow rate (in L/s) to reservoir capacity (in ML), referred to as the volumetric destratification coefficient. Unsuccessful mitigation of thermal stratification was primarily reported in large reservoirs, where the volumetric destratification coefficient was insufficient to effectively destratify the reservoir. Failed thermal destratification was observed for volumetric destratification coefficients of <0.005 L/s/ML, which could be considered a first-pass design check for the required air flowrates.

Where bubble plume systems were effective in terms of thermal destratification, they were commonly reported to mitigate other impacts of reservoir stratification, with 90 % of reservoirs reporting dissolved oxygen restoration in cases where thermal destratification was achieved. Where dissolved oxygen concentrations were restored, iron and manganese concentrations were reduced in 85 % of reservoirs. While less success was reported in terms of reducing (potentially toxic) cyanobacterial populations, success was more likely where thermal destratification was achieved, particularly in reservoirs deeper than 15 m.

Bubble plumes represent a scalable solution, which could theoretically be applied in a reservoir of any size. These systems are primarily limited by the operational costs required to run compressors with an adequate air flow rate, especially in large reservoirs; however, there is potential for these operational costs to be offset through optimised operations and the use of local renewable energy. While the present review provided a comprehensive data base and key recommendations for artificial destratification, further research is needed to provide more quantitative data for the development of detailed design guidelines and optimised operations.

CRediT authorship contribution statement

Fred Chaaya: Writing – original draft, Visualization, Methodology, Investigation, Data curation, Conceptualization. **Brett Miller:** Writing – review & editing, Supervision, Project administration, Methodology, Conceptualization. **Matthew Gordos:** Writing – review & editing, Supervision, Resources, Funding acquisition, Conceptualization. **Bojan Tamburic:** Writing – review & editing, Visualization. **Stefan Felder:** Writing – review & editing, Visualization, Supervision, Conceptualization.

Funding sources

This research was funded by the NSW Department of Primary Industries – Fisheries.

Declaration of competing interest

The authors declare no conflicts of interest.

Acknowledgements

The authors wish to thank Anna Blacka for producing the figures in this paper, Dr. Jamie Ruprecht and Katie Jacka for reviewing the original report produced through this study, and Prof David Hamilton for providing guidance and additional literature for this study.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2025.178738.

Data availability

Data will be made available on request.

References

Antenucci, J.P., Ghadouani, A., Burford, M.A., Romero, J.R., 2005. The long-term effect of artificial destratification on phytoplankton species composition in a subtropical reservoir. Freshwater Biology 50 (6), 1081–1093. https://doi.org/10.1111/j.1365-2427.2005.01374.x.

Ashby, S.L., Kennedy, R.H., 1993. Effects of Artificial Destratification on Water Quality at East Sidney Lake, New York. US Army Engineer Waterways Experiment Station, Vicksburg, Miss. https://apps.dtic.mil/sti/citations/ADA272716.

Astles, K.L., Winstanley, R.K., Harris, J.H., Gehrke, P.C., 2003. Regulated Rivers and Fisheries Restoration Project-Experimental study of the effects of cold water pollution on native fish. NSW Fisheries Final Report Series 44, 1440–3544.

Becker, A., Herschel, A., Wilhelm, C., 2006. Biological effects of incomplete destratification of hypertrophic freshwater reservoir. Hydrobiologia 559 (1), 85–100. https://doi.org/10.1007/s10750-005-4428-3.

Beutel, M.W., Leonard, T.M., Dent, S.R., Moore, B.C., 2008. Effects of aerobic and anaerobic conditions on P, N, Fe, Mn, and Hg accumulation in waters overlaying profundal sediments of an oligo-mesotrophic lake. Water Res. 42 (8–9), 1953–1962. https://doi.org/10.1016/j.watres.2007.11.027.

Biswas, T.K., Karim, F., Kumar, A., Wilkinson, S., Guerschman, J., Rees, G., McInerney, P., Zampatti, B., Sullivan, A., Nyman, P., Sheridan, G.J., Joehnk, K., 2021. 2019–2020 bushfire impacts on sediment and contaminant transport following rainfall in the Upper Murray River catchment. Integr. Environ. Assess. Manag. 17 (6), 1203–1214. https://doi.org/10.1002/ieam.4492.

Boys, C., Miles, N., Rayner, T., 2009. Scoping Options for the Ecological Assessment of Cold Water Pollution Mitigation Downstream of Keepit Dam, Namoi River. Murray–Darling Basin Authority, NSW Department of Primary Industries.

Brookes, J., Burch, M., Lewis, D., Regel, R., Linden, L., Sherman, B., 2008. Artificial Mixing for Destratification and Control of Cyanobacterial Growth in Reservoirs.

Brosnan, T.M., Cooke, G.D., 1987. Response of silver lake trophic state to artificial circulation. Lake and Reservoir Management 3 (1), 66–75. https://doi.org/10.1080/07438148709354761.

Bryant, L.D., Hsu-Kim, H., Gantzer, P.A., Little, J.C., 2011. Solving the problem at the source: controlling Mn release at the sediment-water interface via hypolimnetic oxygenation. Water Res. 45 (19), 6381–6392. https://doi.org/10.1016/j.watres.2011.09.030.

Bryant, L.D., Brockbank, N., Austin, D., 2024. To mix or not to mix? A holistic approach to stratification-preserving and destratification aeration of drinking-water supply reservoirs. Water Res. 261, 121974. https://doi.org/10.1016/j.watres.2024.121974.

Burns, F.L., 1994. Case study: blue-green algal control in Australia by year-round automatic aeration. Lake and Reservoir Management 10 (1), 61–67. https://doi.org/10.1080/07438149409354175.

Clarke, A., Johnston, N.M., 1999. Scaling of metabolic rate with body mass and temperature in teleost fish. J. Anim. Ecol. 68 (5), 893–905. https://doi.org/10.1046/j.1365-2656.1999.00337.x.

Davison, W., 1993. Iron and manganese in lakes. Earth Sci. Rev. 34 (2), 119–163. https://doi.org/10.1016/0012-8252(93)90029-7.

Falconer, I.R., 1999. An overview of problems caused by toxic blue-green algae (cyanobacteria) in drinking and recreational water. Environ. Toxicol. 14 (1), 5–12. https://doi.org/10.1002/(SICI)1522-7278(199902)14:1<5::AID-TOX3>3.0.CO;2-0.

Fuiman, L.A., Batty, R.S., 1997. What a drag it is getting cold: partitioning the physical and physiological effects of temperature on fish swimming. J. Exp. Biol. 200 (12), 1745–1755. https://doi.org/10.1242/jeb.200.12.1745.

Gantzer, P.A., Bryant, L.D., Little, J.C., 2009. Controlling soluble iron and manganese in a water-supply reservoir using hypolimnetic oxygenation. Water Res. 43 (5), 1285–1294. https://doi.org/10.1016/j.watres.2008.12.019.

Gibbs, M.M., Howard-Williams, C., 2018. Physical processes for in-lake restoration: destratification and mixing. In: Hamilton, D., Collier, K., Quinn, J., Howard-Williams, C. (Eds.), Lake Restoration Handbook. Springer, Cham. https://doi.org/10.1007/978-3-319-93043-5_6.

Hamilton, D.P., Wood, S.A., Dietrich, D.R., Puddick, J., 2014. Costs of harmful blooms of freshwater cyanobacteria. In: Cyanobacteria: An Economic Perspective, pp. 245–256. https://doi.org/10.1002/9781118402238.ch15.

Hamilton, D.P., Salmaso, N., Paerl, H.W., 2016. Mitigating harmful cyanobacterial blooms: strategies for control of nitrogen and phosphorus loads. Aquatic Ecology 50 (3), 351–366. https://doi.org/10.1007/s10452-016-9594-z.

Haynes, R.C., 1975. Some ecological effects of artificial circulation on a small eutrophic lake with particular emphasis on phytoplankton II. Kezar lake experiment, 1969. Hydrobiologia 46 (1), 141–170. https://doi.org/10.1007/BF00038729.

Hedayati, A., Hoseini, S.M., Ghelichpour, M., 2014. Acute toxicity of waterborne manganese to Rutilus caspicus (Yakovlev, 1870) – gill histopathology, immune indices, oxidative condition, and saltwater resistance. Toxicol. Environ. Chem. 96 (10), 1535–1545. https://doi.org/10.1080/02772248.2015.1028408.

Hill, D.F., Vergara, A.M., Parra, E.J., 2008. Destratification by mechanical mixers: mixing efficiency and flow scaling. J. Hydraul. Eng. 134 (12), 1772–1777. https://doi.org/10.1061/(ASCE)0733-9429(2008)134:12(1772).

Howard, A., 2012. Cyanobacteria (blue-green algae). In: Bengtsson, L., Herschy, R.W., Fairbridge, R.W. (Eds.), Encyclopedia of Lakes and Reservoirs. Encyclopedia of Earth Sciences Series. Springer, Dordrecht. https://doi.org/10.1007/978-1-4020-4410-6_60.

Huang, T., Sun, X., Li, X., Cong, H., Cheng, Y., 2016. Water quality improvement using water-lifting aeration technology. In: Water Pollution and Water Quality Control of Selected Chinese Reservoir Basins, pp. 279–329. https://doi.org/10.1007/978-3-319-20391-1_9.

Huisman, J., et al., 2018. Cyanobacterial blooms. Nat. Rev. Microbiol. 16 (8), 471–483. https://doi.org/10.1038/s41579-018-0040-1.

Huttula, T., 2012. Stratification in lakes. In: Bengtsson, L., Herschy, R.W., Fairbridge, R.W. (Eds.), Encyclopedia of Lakes and Reservoirs. Encyclopedia of Earth Sciences Series. Springer, Dordrecht. https://doi.org/10.1007/978-1-4020-4410-6_12.

ICOLD, 2011. CIGB ICOLD Constitution. Available at: https://www.icold-cigb.org/userfiles/files/CIGB/INSTITUTIONAL_FILES/Constitution2011.pdf (Accessed July 2024).

Idso, S.B., 1973. On the concept of lake stability. Limnol. Oceanogr. 18 (4), 681–683. https://doi.org/10.4319/lo.1973.18.4.0681.

Jeppesen, E., Kronvang, B., Olesen, J.E., Meerhoff, M., Søndergaard, M., Hansen, K.M., Andersen, H.E., Lauridsen, T.L., Liboriussen, L., Beklioglu, M., Özen, A., 2009. Climate change effects on runoff, catchment phosphorus loading and lake ecological state, and potential adaptations. J. Environ. Qual. 38 (5), 1930–1941. https://doi.org/10.2134/jeq2008.0113.

Koehn, J.D., Doeg, T.J., Harrington, D.J., Milledge, G.A., 1995. The effects of Dartmouth Dam on the aquatic fauna of the Mitta Mitta River. In: Report to the Murray-Darling Basin Commission, Canberra.

Lewis, D.M., 2004. Surface Mixers for Destratification and Management of Anabaena circinalis. Doctoral dissertation. https://digital.library.adelaide.edu.au/dspace/bitstream/2440/22047/2/02whole.pdf.

Lewis, D.P., Patterson, J.C., Imberger, J., Wright, R.P., Schladow, S.G., 1991. Modelling and Design of Reservoir Aeration Destratification Systems, Research Report No. 23. Urban Water Research Association of Australia. https://water360.com.au/wp-content/uploads/2023/01/1919871.pdf.

Li, Y., Huang, T., Tan, X., Zhou, Z., Ma, W., 2018. Destratification and oxygenation efficiency of a water-lifting aerator system in a deep reservoir: implications for optimal operation. J. Environ. Sci. 73, 9–19. https://doi.org/10.1016/j.jes.2017.12.021.

Li, N., Huang, T., Mao, X., Zhang, H., Li, K., Wen, G., Deng, L., 2019. Controlling reduced iron and manganese in a drinking water reservoir by hypolimnetic aeration and

artificial destratification. Sci. Total Environ. 685, 497–507. https://doi.org/10.1016/j.scitotenv.2019.05.445.
Li, N., Huang, T., Li, Y., Si, F., Zhang, H., Wen, G., 2020. Inducing an extended naturally complete mixing period in a stratified reservoir via artificial destratification. Sci. Total Environ. 745, 140958. https://doi.org/10.1016/j.scitotenv.2020.140958.
Liboriussen, L., Søndergaard, M., Jeppesen, E., Thorsgaard, I., Grünfeld, S., Jakobsen, T. S., Hansen, K., 2009. Effects of hypolimnetic oxygenation on water quality: results from five Danish lakes. Hydrobiologia 625 (1), 157–172. https://doi.org/10.1007/s10750-009-9705-0.
Liu, M., Zhang, Y., Shi, K., Zhu, G., Wu, Z., Liu, M., Zhang, Y., 2019. Thermal stratification dynamics in a large and deep subtropical reservoir revealed by high-frequency buoy data. Sci. Total Environ. 651, 614–624. https://doi.org/10.1016/j.scitotenv.2018.09.215.
Lugg, A., Copeland, C., 2014. Review of cold water pollution in the Murray–Darling Basin and the impacts on fish communities. Ecol. Manage. Restor. 15 (1), 71–79. https://doi.org/10.1111/emr.12074.
Ma, W.X., Huang, T.L., Li, X., 2015. Study of the application of the water-lifting aerators to improve the water quality of a stratified, eutrophicated reservoir. Ecol. Eng. 83, 281–290. https://doi.org/10.1016/j.ecoleng.2015.06.022.
MacIntyre, S., Hamilton, D.P., 2024. Fate of heat. In: Wetzels Limnology. Elsevier, pp. 95–153. https://doi.org/10.1016/B978-0-12-822701-5.00007-0.
McAuliffe, T.F., Rosich, R.S., 1989. Review of Artificial Destratification of Water Storages in Australia. Urban Water Research Association of Australia.
McClintock, N.L., Wilhm, J., 1977. Effects of artificial destratification on zooplankton of two Oklahoma reservoirs. Hydrobiologia 54 (3), 233–239. https://doi.org/10.1007/BF00014290.
Mcdougall, T.J., 1978. Bubble plumes in stratified environments. J. Fluid Mech. 85 (4), 655–672. https://doi.org/10.1017/S0022112078000841.
Merel, S., Walker, D., Chicana, R., Snyder, S., Baurès, E., Thomas, O., 2013. State of knowledge and concerns on cyanobacterial blooms and cyanotoxins. Environ. Int. 59, 303–327. https://doi.org/10.1016/j.envint.2013.06.013.
Michie, L.E., Thiem, J.D., Facey, J.A., Boys, C.A., Crook, D.A., Mitrovic, S.M., 2020a. Effects of suboptimal temperatures on larval and juvenile development and otolith morphology in three freshwater fishes: implications for cold water pollution in rivers. Environ. Biol. Fishes 103 (12), 1527–1540. https://doi.org/10.1007/s10641-020-01041-z.
Michie, L.E., Thiem, J.D., Boys, C.A., Mitrovic, S.M., 2020b. The effects of cold shock on freshwater fish larvae and early-stage juveniles: implications for river management. Conservation Physiology 8 (1), coaa092. https://doi.org/10.1093/conphys/coaa092.
Michie, L., Gordos, M., Carruthers, C., Boys, C., Thiem, J., Townsend, A., 2023. Review of Ecological Thresholds and Requirements to Inform Guidelines for Cold Water Pollution Mitigation and Water Release Operations. NSW Department of Primary Industries - Fisheries.
Miles, N.G., West, R.J., 2011. The use of an aeration system to prevent thermal stratification of a freshwater impoundment and its effect on downstream fish assemblages. J. Fish Biol. 78 (3), 945–952. https://doi.org/10.1111/j.1095-8649.2010.02896.x.
Mobley, M., Tyson, W., Webb, J., Brock, G., 1995. Surface water pumps to improve dissolved oxygen content of hydropower releases. In: Waterpower, vol. 1. ASCE.
Müller, S., Mitrovic, S.M., 2015. Phytoplankton co-limitation by nitrogen and phosphorus in a shallow reservoir: progressing from the phosphorus limitation paradigm. Hydrobiologia 744 (1), 255–269. https://doi.org/10.1007/s10750-014-2082-3.
Müller, S., Mitrovic, S.M., Baldwin, D.S., 2016. Oxygen and dissolved organic carbon control release of N, P, and Fe from the sediments of a shallow, polymictic lake. J. Soil. Sediment. 16 (3), 1109–1120. https://doi.org/10.1007/s11368-015-1298-9.
Munger, Z.W., Carey, C.C., Gerling, A.B., Hamre, K.D., Doubek, J.P., Klepatzki, S.D., Schreiber, M.E., 2016. Effectiveness of hypolimnetic oxygenation for preventing accumulation of Fe and Mn in a drinking water reservoir. Water Res. 106, 1–14. https://doi.org/10.1016/j.watres.2016.09.038.
New South Wales Office of Water, 2011. NSW Cold Water Pollution Strategy: guidelines for managing cold water releases from high priority dams. https://water.dpie.nsw.gov.au/__data/assets/pdf_file/0008/456911/NSW-Cold-water-pollution-strategy.pdf.
NHMRC, 2008. Guidelines for Managing Risks in Recreational Water. Australian Government - National Health and Medical Research Council.
Osgood, R.A., Stiegler, J.E., 1990. The effects of artificial circulation on a hypereutrophic lake. JAWRA Journal of the American Water Resources Association 26 (2), 209–217. https://doi.org/10.1111/j.1752-1688.1990.tb01363.x.
Pastorok, R.A., Ginn, T.C., Lorenzen, M.W., 1981. Evaluation of Aeration/Circulation as a Lake Restoration Technique. Environmental Research Laboratory, Office of Research and Development, US Environmental Protection Agency.
Patterson, J.C., Imberger, J., 1989. Simulation of bubble plume destratification systems in reservoirs. Aquat. Sci. 51 (1), 3–18. https://doi.org/10.1007/BF00877777.
Preece, R., 2004. Cold Water Pollution Below Dams in New South Wales: A Desktop Assessment. Water management Division, Dept. of Infrastructure, Planning and Natural Resources.
Preece, R.M., Jones, H.A., 2002. The effect of Keepit Dam on the temperature regime of the Namoi River, Australia. River Research and Applications 18 (4), 397–414. https://doi.org/10.1002/rra.686.
Preece, E.P., Lewis, W.M., Smith, R.A., Anderson, J.T., Thompson, K.L., 2019. A review of the biological and chemical effects of hypolimnetic oxygenation. Lake and Reservoir Management 35 (3), 229–246. https://doi.org/10.1080/10402381.2019.1580325.
Read, J.S., Shade, A., Wu, C.H., Gorzalski, A., McMahon, K.D., 2011. "Gradual Entrainment Lake Inverter" (GELI): a novel device for experimental lake mixing. Limnol. Oceanogr. Methods 9 (1), 14–28. https://doi.org/10.4319/lom.2011.9.14.
Ryan, T., Webb, A., Lennie, R., Lyon, J., 2001. Status of Cold Water Releases From Victorian Dams. Arthur Rylah Institute, Department of Natural Resources and Environment, Melbourne.
Sahoo, G.B., Luketina, D., 2006. Response of a tropical reservoir to bubbler destratification. J. Environ. Eng. 132 (7), 736–746. https://doi.org/10.1061/(asce)0733-9372(2006)132:7(736).
Schladow, S.G., 1993. Lake destratification by bubble-plume systems: design methodology. J. Hydraul. Eng. 119 (3), 350–368. https://doi.org/10.1061/(asce)0733-9429(1993)119:3(350).
Schmidt, W., 1928. Über die Temperatur- und Stabilitätsverhältnisse von Seen. Geogr. Ann. 10 (1–2), 145–177. https://doi.org/10.1080/20014422.1928.11880475.
Sherman, B.S., 2000. Scoping Options for Mitigating Cold Water Discharges From Dams: Report to Agriculture Fisheries and Forestry - Australia, NSW Fisheries, CRC for Freshwater Ecology, and NSW Department of Land and Water Conservation as part of the NHT Murray Darling FishRehab Program. CSIRO Land & Water, Canberra, ACT. https://doi.org/10.4225/08/594eb796470f2, 2000-05. legacy:1957.
Sherman, B., Ford, P., Hatton, P., Whittington, J., Green, D., Baldwin, D., Oliver, R., Shiel, R., van Berkel, J., Beckett, R., Grey, L., Maher, B., 2001. The Chaffey dam story. In: Final Report for CRCFE Projects B, 202.
Shi, J., Wang, L., Yang, Y., Huang, T., 2022. A case study of thermal and chemical stratification in a drinking water reservoir. Sci. Total Environ. 848, 157787. https://doi.org/10.1016/j.scitotenv.2022.157787.
Smith, C.A., Read, J.S., Vander Zanden, M.J., 2018. Evaluating the "Gradual Entrainment Lake Inverter" (GELI) artificial mixing technology for lake and reservoir management. Lake and Reservoir Management 34 (3), 232–243. https://doi.org/10.1080/10402381.2018.1423586.
Smucker, N.J., Beaulieu, J.J., Nietch, C.T., Young, J.L., 2021. Increasingly severe cyanobacterial blooms and deep water hypoxia coincide with warming water temperatures in reservoirs. Glob. Chang. Biol. 27 (11), 2507–2519. https://doi.org/10.1111/gcb.15618.
Spigel, R.H., Imberger, J., 1987. Mixing processes relevant to phytoplankton dynamics in lakes. N. Z. J. Mar. Freshw. Res. 21 (3), 361–377. https://doi.org/10.1080/00288330.1987.9516233.
Steichen, J.M., Garton, J.E., Rice, C.E., 1979. The effect of lake destratification on water quality. J. Am. Water Works Ass. 71 (4), 219–225.
Steinberg, C., Zimmermann, G.M., 1988. Intermittent destratification: a therapy measure against cyanobacteria in lakes. Environ. Technol. 9 (4), 337–350. https://doi.org/10.1080/09593338809384575.
Stephens, R., Imberger, J., 1993. Reservoir destratification via mechanical mixers. J. Hydraul. Eng. 119 (4), 438–457. https://doi.org/10.1061/(ASCE)0733-9429(1993)119:4(438).
Suter, P.J., Kilmore, G., 1990. Mechanical mixers: an alternative destratification technique, Myponga Reservoir, South Australia. J. Australian Water and Wastewater Assoc 32–35.
Symons, J.M., Carswell, J.K., Robeck, G.G., 1970. Mixing of water supply reservoirs for quality control. J. Am. Water Works Ass. 62 (5), 322–334. https://doi.org/10.1002/j.1551-8833.1970.tb03913.x.
Tilzer, M.M., 1987. Light-dependence of photosynthesis and growth in cyanobacteria: implications for their dominance in eutrophic lakes. N. Z. J. Mar. Freshw. Res. 21 (3), 401–412. https://doi.org/10.1080/00288330.1987.9516236.
Todd, C.R., Ryan, T., Nicol, S.J., Bearlin, A.R., 2005. The impact of cold water releases on the critical period of post-spawning survival and its implications for Murray cod (Maccullochella peelii peelii): a case study of the Mitta Mitta River, southeastern Australia. River Res. Appl. 21 (9), 1035–1052. https://doi.org/10.1002/rra.873.
Toetz, D.W., 1977. Effects of lake mixing with an axial flow pump on water chemistry and phytoplankton. Hydrobiologia 55 (2), 129–138. https://doi.org/10.1007/BF00021054.
US Army Corps of Engineers, 1986. Proceedings: CE (Corps of Engineers) Workshop on Design and Operation of Selective Withdrawal Intake Structures. U.S. Army Engineer Waterways Experiment Station, Hydraulics Laboratory.
Vieira, M.C., Torronteras, R., Córdoba, F., Canalejo, A., 2012. Acute toxicity of manganese in goldfish Carassius auratus is associated with oxidative stress and organ specific antioxidant responses. Ecotoxicol. Environ. Saf. 78, 212–217. https://doi.org/10.1016/j.ecoenv.2011.11.015.
Visser, P., Ibelings, B.A.S., Van Der Veer, B., Koedood, J.A.N., Mur, R., 1996. Artificial mixing prevents nuisance blooms of the cyanobacterium Microcystis in Lake Nieuwe Meer, the Netherlands. Freshw. Biol. 36 (2), 435–450. https://doi.org/10.1046/j.1365-2427.1996.00093.x.
Visser, P.M., Ibelings, B.W., Bormans, M., Huisman, J., 2016. Artificial mixing to control cyanobacterial blooms: a review. Aquat. Ecol. 50, 423–441. https://doi.org/10.1007/s10452-015-9537-0.
Wallace, B.B., Hamilton, D.P., 1999. The effect of variations in irradiance on buoyancy regulation in Microcystis aeruginosa. Limnol. Oceanogr. 44 (2), 273–281. https://doi.org/10.4319/lo.1999.44.2.0273.
Wang, H., Deng, Y., Yang, Y., Chen, M., Wang, X., Tuo, Y., 2024. Future projections of thermal regimes and mixing characteristics in a monomictic reservoir under climate change. Sci. Total Environ. 906, 167527. https://doi.org/10.1016/j.scitotenv.2023.167527.
Woolway, R.I., Sharma, S., Weyhenmeyer, G.A., Debolskiy, A., Golub, M., Mercado-Bettín, D., Perroud, M., Stepanenko, V., Tan, Z., Grant, L., Ladwig, R., Mesman, J., Moore, T.N., Shatwell, T., Vanderkelen, I., Austin, J.A., DeGasperi, C.L., Dokulil, M., La Fuente, S., Jennings, E., 2021. Phenological shifts in lake stratification under

climate change. Nat. Commun. 12 (1), 2318. https://doi.org/10.1038/s41467-021-22657-4.

World Health Organization, 2017. Guidelines for Drinking-Water Quality: Fourth Edition Incorporating First Addendum (4th ed + 1st add). World Health Organization. https://iris.who.int/handle/10665/254637.

Yu, S., Sturm, K., Gibbes, B., Kennard, M.J., Veal, C.J., Middleton, D., Fisher, P.L., Rotherham, S., Hamilton, D.P., 2022. Developing best practice guidelines for lake modelling to inform quantitative microbial risk assessment. Environ. Model. Software 150, 105334. https://doi.org/10.1016/j.envsoft.2022.105334.

Yu, S., Hamilton, D.P., Kennard, M.J., Cinque, K., Ollivier, Q., Van Der Sterren, M., Rogers, Z., Veal, C., Gibbes, B., 2024. Best practice guidelines for lake modelling to support quantitative microbial risk assessment (QMRA). In: WaterRA Project 1143. https://www.waterra.com.au/project/best-practice-guidance-for-lake-modelling-to-assist-qmra/.

Zhang, Y., Wu, Z., Liu, M., He, J., Shi, K., Zhou, Y., Liu, X., 2015. Dissolved oxygen stratification and response to thermal structure and long-term climate change in a large and deep subtropical reservoir (Lake Qiandaohu, China). Water Res. 75, 249–258. https://doi.org/10.1016/j.watres.2015.02.052.

Zhang, H., Yan, M., Huang, T., Huang, X., Yang, S., Li, N., Wang, N., 2020. Water-lifting aerator reduces algal growth in stratified drinking water reservoir: novel insights into algal metabolic profiling and engineering applications. Environ. Pollut. 266, 115384. https://doi.org/10.1016/j.envpol.2020.115384.

Zhu, J., Stuetz, R.M., Hamilton, L., Power, K., Crosbie, N.D., Tamburic, B., 2022. Management of biogenic taste and odour: from source water, through treatment processes and distribution systems, to consumers. J. Environ. Manage. 323, 116225. https://doi.org/10.1016/j.jenvman.2022.116225.