# Can reductions in water residence time be used to disrupt seasonal stratification and control internal loading in a eutrophic monomictic lake?

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## Abstract

Anthropogenic eutrophication caused by excess loading of nutrients, especially phosphorus (P), from catchments is a major cause of lake water quality degradation. The release of P from bed sediments to the water column, termed *internal loading*, can exceed catchment P load in eutrophic lakes, especially those that stratify during warm summer periods. Managing internal P loading is challenging, and although a range of approaches have been implemented, long-term success is often limited, requiring lake-specific solutions. Here, we assess the manipulation of lake residence time to inhibit internal loading in Elterwater, a shallow stratifying lake in the English Lake District, UK. Since 2016, additional inflowing water has been diverted into the inner basin of Elterwater to reduce its water residence time, with the intention of limiting the length of the stratified period and reducing internal loading. Combining eight years of field data in a Before-After-Control-Impact study with process-based hydrodynamic modelling enabled the quantification of the residence time intervention effects on stratification length, water column stability, and concentrations of chlorophyll *a* and P.  Annual water residence time was reduced during the study period by around 40% (4.9 days). Despite this change, the lake continued to stratify and developed hypolimnetic anoxia. As a result, there was little significant change in phosphorus (as total or soluble reactive phosphorus) or chlorophyll *a* concentrations. Summer stratification length was 2 days shorter and 7% less stable with the intervention. Our results suggest that the change to water residence time in Elterwater was insufficient to induce large enough physical changes to improve water quality. However, the minor physical changes suggest the management measure had some impact and that larger changes in water residence time may have the potential to induce reductions in internal

loading. Future assessments of management requirements should combine multi-year observations and physical lake modelling to provide improved understanding of the intervention effect size required to alter the physical structure of the lake, leading to increased hypolimnetic oxygen and reduced potential for internal loading.

Keywords: *Lake restoration, lake management, water quality, lake modelling, hypolimnetic anoxia, destratification*

# Introduction

The degradation of fresh waters is a pervasive and persistent problem (Smith, 2003). In Europe alone, 60 % of surface waters failed ecological quality targets set by the European Water Framework Directive in 2018, with little to no improvement in the ecological quality of lakes being reported in over a decade (EEA, 2018). The principal cause of degradation for lakes remains eutrophication (Birk et al., 2020), caused by anthropogenic inputs of phosphorus (P) and nitrogen (N) from catchment sources that result in excess phytoplankton growth, a loss of biodiversity, and low oxygen conditions (Jeppesen et al., 2007; Søndergaard et al., 2005). In 2018, eutrophication impacts in the UK, in particular algal blooms, were estimated to cost £173 million annually, with the potential to rise to £481 million under a 4 °C warming climate (Jones et al., 2020).

External nutrient load reductions are the primary measure to improve in-lake conditions (Lürling and Mucci, 2020; Van Liere and Gulati, 1992). However, problems can persist in lakes decades after reductions (McCrackin et al., 2017). Slow recovery can often be attributed to the release of nutrients accumulated in bed sediments, maintaining water column nutrient concentrations, a process known as *internal loading* (Does et al., 1992; Søndergaard et al., 2003; Van Liere and Gulati, 1992). In temperate zone stratifying lakes, internal loading principally occurs in the summer period. High biological oxygen demand in the isolated hypolimnion depletes oxygen that cannot be replenished as the water column density gradient inhibits mixing. Anoxia in the hypolimnion overlying lake bed sediments promotes redox conditions where Fe-P complexes are reduced and their dissolved components liberated across diffusive concentration gradients to the water column (Mortimer, 1942; Nürnberg, 1984). In order to meet legislative water quality targets (e.g. the European Water Framework Directive and the US Clean Water Act), there is a growing need for in-lake measures to control internal loading (Lürling and Mucci, 2020; Zamparas and Zacharias, 2014).

A range of in-lake measures have been proposed to control internal loading (Lürling et al., 2020). These include sediment dredging (Bormans et al., 2016; Does et al., 1992), chemical inactivation (Mackay et al., 2014; Spears et al., 2016), hypolimnetic aeration and oxygenation (Preece et al., 2019; Toffolon et al., 2013), artificial mixing (Visser et al., 2016), and less frequently, hypolimnetic withdrawal (Nürnberg, 2019). These in-lake measures vary in their approach but generally target the manipulation of hypolimnetic anoxia or sediment-P binding to decrease the intensity of internal loading (Figure 1). Increasing inflow discharge has also been used to promote direct flushing of phytoplankton cells and/or to dilute nutrient concentrations, with moderate short-lived effects (Jagtman et al., 1992; Verspagen et al., 2006; Welch and Patmont, 1980; Zhang et al., 2016). While there has been some success using these existing in-lake methods, restoration outcomes have been inconsistent (Huser et al., 2016), and can incur high capital and running costs (Mackay et al., 2014; Visser et al., 2016). With the pressure to achieve water quality targets, the threat of climate change, and the mixed success of existing measures, there is a need for innovative methods to tackle internal loading.

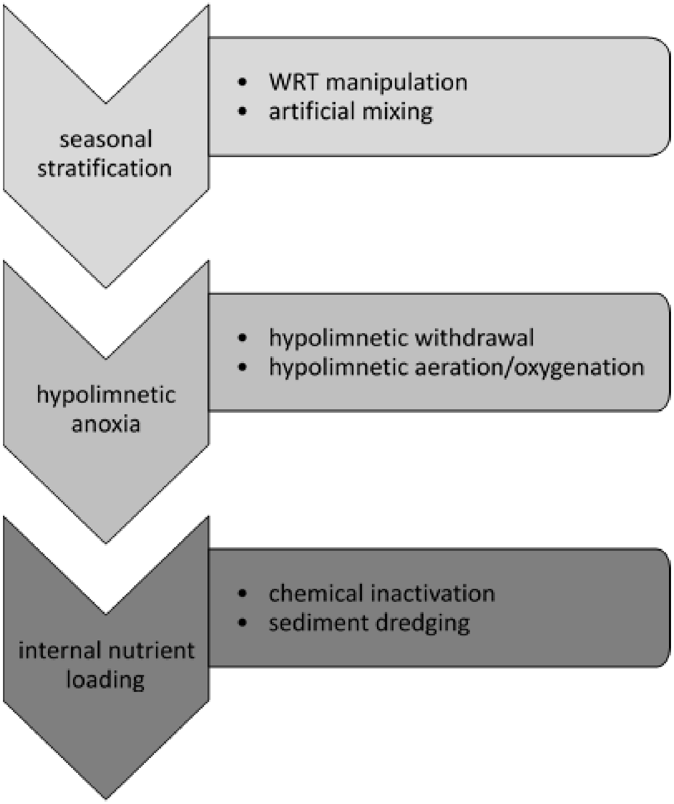


*Figure 1 Different aspects of the stratification → anoxia → internal P loading sequence are targeted by different in-lake restoration methods*.

In some cases, water residence time (WRT) reductions may present an effective method to inhibit stratification, suppressing internal loading and algal blooms in stratifying lakes. Lake inflows can impact the thermal structure of lakes, influencing lake water temperatures (Carmack et al., 1979; Fenocchi et al., 2017). Previous reservoir modelling studies suggest that maintaining flow levels to reduce WRT can modify stratification (Li et al., 2018; Straškraba and Hocking, 2002). In addition, lakes with shorter WRTs or periods of reduced WRT can experience a shorter stratified period and periods of increased mixing (Andersen et al., 2020; Li et al., 2018; Straškraba and Hocking, 2002). Thus, artificial manipulations of water residence time may present another technique to suppress anoxia and internal loading through increased cooling of in-lake temperatures and reduction to stratification length and strength, but effectiveness has yet to be quantitatively assessed.

It is important to assess efficacy of novel management measures using whole-lake case studies alongside robust statistical and process modelling approaches. The Before-After-Control-Impact

(BACI) statistical approach, in which a control system is used alongside an impacted system, has been shown to detect changes not possible using impact lake data only (Christie et al., 2019; Smokorowski and Randall, 2017). Multiple years of pre- and post- intervention data is also needed to allow inter-annual variability to be separated from intervention impacts (Smokorowski and Randall, 2017; Underwood, 1994). High-resolution data can provide valuable insights and increased statistical power in detecting responses to management (Kerr et al., 2019). Moreover, lake modelling can be used to provide process understanding of restoration impacts and refine future applications (Janssen et al., 2015). Despite their synergistic potential, this combination of methods is uncommon in restoration assessments. Here, we combine these approaches to assess the efficacy of WRT management to control stratification and internal loading, in a small eutrophic lake, Elterwater, in the English Lake District, UK.

Using eight years of pre- and post-intervention monitoring data (2012-2019), water quality profiles, and hydrodynamic modelling, we investigated the impact of decreasing the WRT, by means of a diversion of flow from a river through the lake initiated in 2016. We used the BACI approach to assess responses in (1) WRT, (2) the intensity and duration of stratification, (3) development of hypolimnetic anoxia, and (4) nutrient and chlorophyll *a* concentrations in the water column. We discuss indicators of effectiveness and outline approaches that may be used to refine the intervention in Elterwater, and potentially in other lakes. As one in four lakes globally have a short residence time (< 100 days, Messager et al., 2016), this study will provide context for future lake restoration efforts and will begin to explore how this novel method may be applied more widely as in-lake restorations grow in importance.

# Methods and materials

## Impact site

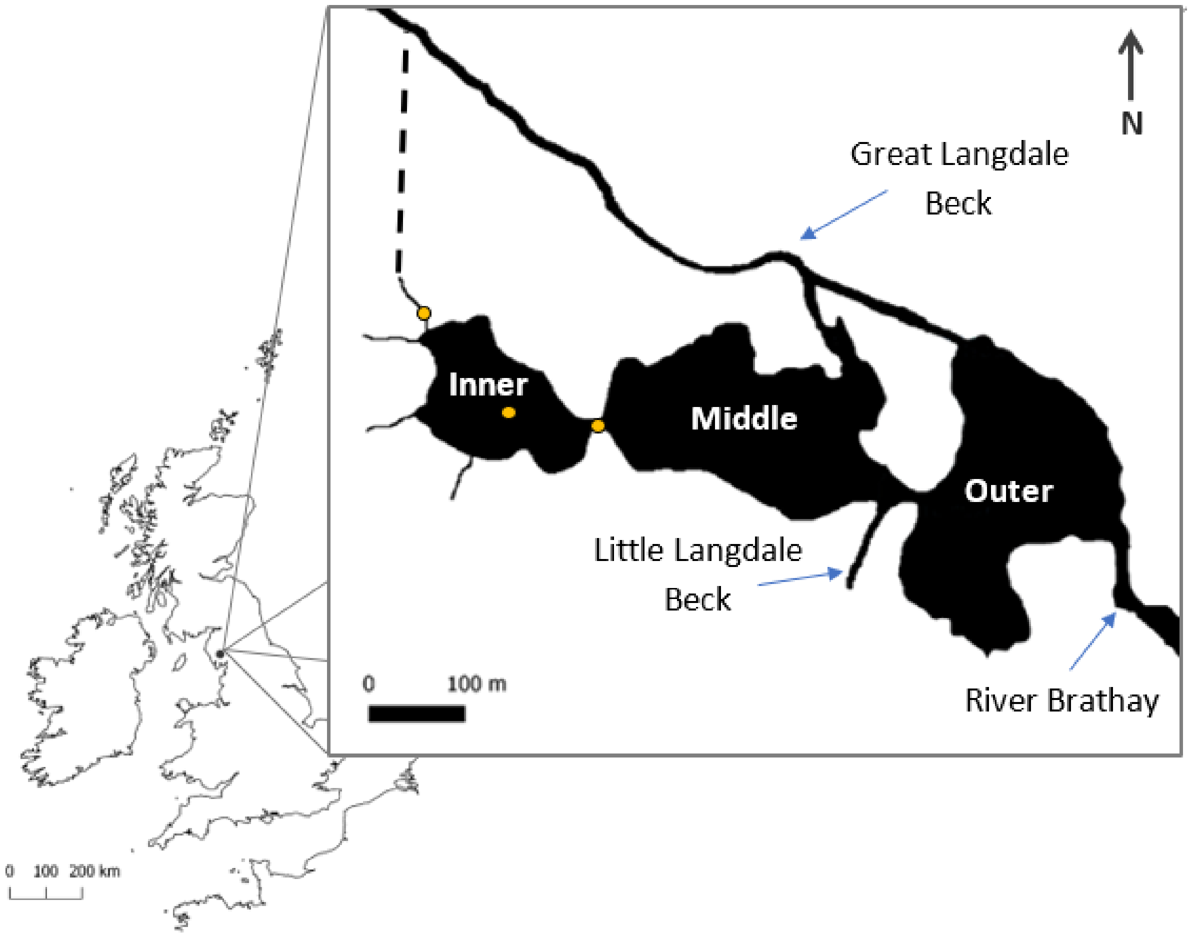


*Figure 2 Map of Elterwater with its major inflow and outflows labelled. Approximate location of the flow diversion, implemented in 2016, is shown with a dashed line and sampling locations with yellow circles.*

Elterwater (impact site) is a small lake located in the English Lake District, UK. It has three distinct basins, inner, middle, and outer (Figure 2), with the main inflow, the Great Langdale Beck (GLB), and outflow, River Brathay, flowing into and out of the outer basin, respectively. Smaller inflows discharge into the inner and middle basins. Due to the system's hydrology, the WRT varies significantly between the basins, with previous studies estimating WRTs of around 15-20 days in the inner and middle basins and as little as 0.5 days in the outer basin (APEM, 2012; Beattie et al., 1996). The inner basin of Elterwater (Elterwater-IB) was the main target of the restoration efforts and is the focus of this study. The inner basin is the smallest of the three basins and the most nutrient-enriched (Supplementary information 1). Elterwater-IB was, historically, the primary discharge point for wastewater treatment effluent (Zinger-Gize et al., 1999) and in-lake concentrations of total phosphorus (as TP) and chlorophyll *a* regularly exceed eutrophic status (see Supplementary information 1; Nürnberg, 1996). Sediment TP concentrations in the deep parts of the inner basin exceed 4500 µg $g^{-1}$, suggesting there is a high potential for internal loading of nutrients from the sediments (Mackay et al., 2020). Internal loading of nutrients under anoxic conditions, during the annual summer stratification period, is suspected to be the source of the persistent water quality problems (APEM, 2012) and we present evidence of persistent summer spikes in TP and chlorophyll *a* (Supplementary Figure S1.1) to support this (Søndergaard et al., 2002).

*Table 1 Hydromorphometric and physiochemical comparison of Impact and Control sites. Data from 2015 Lakes Tour (Maberly et al., 2016) and Haworth et al. (2003).*

| Attribute | Elterwater inner basin (Impact) | Blelham (Control) |
|---|---|---|
| **Location** | Lat: 54.4287<br>Long: -3.0350 | Lat: 54.3959<br>Long: -2.9780 |
| **Elevation (m above ordinance datum)** | 53 | 47 |
| **Surface area (km$^2$)** | 0.031 | 0.1 |
| **Mean depth (m)** | 3.3 | 6.8 |
| **Maximum depth (m)** | 6.5 | 14.5 |
| **Annual mean WRT (days)** | 20 | 50 |
| **Catchment area (km$^2$)** | 1.0 | 4.3 |
| **Annual mean total phosphorus concentration (µg $L^{-1}$)** | 18.3 | 24.5 |
| **Annual mean chlorophyll *a* (µg $L^{-1}$)** | 16 | 23 |
| **Trophic state** | Meso-eutrophic | Meso-eutrophic |
| **Annual mean alkalinity (m equiv $m^{-3}$)** | 285 (low) | 450 (medium) |

## Control site

Blelham Tarn (control site) is a small monomictic lake located approximately 5 km to the southeast of Elterwater (Table 1). Blelham Tarn was selected as the “control” site for the BACI analysis as the sites are close together, thermally stratify during the summer, develop hypolimnetic anoxia (Foley et al., 2012), exhibit internal loading (Gray, 2019), and share relatively similar physio-chemical characteristics pre-restoration (Table 1). While all lakes are unique, the similarities shared between Elterwater and Blelham Tarn allow comparison over the experimental period. Furthermore, the BACI method does not require that the sites to be the same but that the difference between the sites is consistent in the Before period with only the focal impact changing in the After period (Stewart-Oaten et al., 1986). Blelham Tarn is also part of the UKCEH Cumbrian Lakes Monitoring Platform (see https://ukscape.ceh.ac.uk/our-science/projects/cumbrian-lakes-monitoring-platform), with both a fortnightly long-term monitoring programme and a monitoring buoy providing water quality and meteorological data for the study period (2012-2019).

# Data collection

## Flow and water residence times

The inflow to Elterwater-IB, before the restoration, had been estimated as 2% of lake outflow discharge (Environment Agency, 2000), measured at the River Brathay gauging station. Since the intervention in 2016, additional water flow has increased the discharge into the basin via a diversion from GLB. This diversion forms an underground pipeline (approximately 0.7 m diameter) running 350 m from the GLB river channel to one of the small field drains that discharges into the inner basin (see Supplementary Information Figure S2.1). Flow through the pipe is not maintained at a consistent discharge but acts by passively diverting a small proportion of the GLB flow. The pipeline is monitored and maintained by the South Cumbria Rivers Trust, and they provided daily (January 2016-July 2017) and hourly (July 2017-2019) discharge data for the pipeline. There were gaps in the data due to sensor error and maintenance. Small gaps (< 24 hours) were filled using linear interpolation. Larger gaps were filled based on a statistical relationship between flow measurements in the pipeline and the River Brathay gauged flow (Supplementary information 3). This relationship is bounded by legal abstraction limits (0.122 $m^3 s^{-1}$) and a minimum flow requirement in the source river (Q85, 0.383 $m^3 s^{-1}$). A hydro-brake system operates to shut off the pipeline when the flow is outside of these limits. The total flow into Elterwater-IB is the gauged pipeline discharge plus the 2% of the gauged outflow.

Water residence time ($WRT$) was calculated as,

$$\mathrm{WRT} = \frac{V}{Q} \tag{1}$$

Where $V$ is the basin volume and $Q$ is the outflow discharge. By assuming outflow is equal to inflow discharge and constant basin volume (from Haworth et al., 2003), inflow discharge, as determined above, can be used to calculate water residence times at each hourly time step. When calculating monthly, seasonal, or annual WRTs the mean inflow discharge for that period was calculated and used in Equation 1. WRTs were calculated for the post-intervention period, both with and without the additional intervention flow, to isolate the exact change in WRT caused by the intervention.

## Biological and chemical data

### *Water sampling*

In Elterwater-IB, monthly water samples were collected for water chemistry analysis at the deepest point in the basin from 2012-2019 using a 5 m integrated sampling tube. Additionally, in 2018-2019 water samples were collected in Elterwater-IB at 0.5 m and 6 m from the surface, using a Ruttner sampler, and from the basin inflow and outflow (Figure 2). Dissolved oxygen profiles were taken using a Yellow Springs Instruments-Exo2 multi-parameter sonde (Xylem, OH, USA) weekly during stratification and monthly during isothermal conditions between May 2018 and December 2019. Measurements were taken at 0.5 m intervals from 0.5 m to 6.5 m. Oxygen sensors were calibrated monthly according to the manufacturer's specifications. In Blelham Tarn, integrated water chemistry samples of the top 5 m of the water column were taken at the deepest point of the lake, fortnightly from 2012 to 2019.

### *Laboratory methods*

Chlorophyll *a* was measured as a proxy for phytoplankton biomass. A measured volume of the water was filtered onto a Whatman GF/C filter paper from the integrated sample. Filter papers were frozen, and analysis completed within six months. Samples from Elterwater-IB were extracted using a cold acetone extraction. At Blelham, samples were extracted using heated methanol, according to Talling (1974). Although the extraction method differs between sites, consistent methods were used across the entire period, so any difference in values due to the extraction method will be maintained. Total phosphorus (TP) concentrations, from both Elterwater-IB and Blelham, were determined using a potassium persulphate ($K_2S_sO_8$) digestion and colourimetric analysis using the molybdenum blue method from 2014 -2019. Due to methodological differences prior to 2014 only 2014-2019 samples were used for TP data analysis. We determined gross summer internal load estimates using a mass balance approach (Nürnberg, 2009, 1984). Soluble reactive phosphorus (SRP) concentrations were determined from a 50 ml sub-sample filtered using a Sartorius cellulose acetate 0.45 µm filter into an acid-washed polypropylene tube. All SRP concentrations were determined using a colorimetric method according to Stephens (1963) and carried out on the day of collection. Bioavailable phosphorus (BAP), that is, phosphorus which readily assimilates or is already assimilated by biomass, was calculated as the concentration of SRP plus chlorophyll *a* concentration, following Reynolds & Davies (2001).

### *Preparing data for analysis*

Data were linearly interpolated to a daily timestep and then a monthly average calculated to give paired monthly values from 2012 to 2019. TP covered 2014 to 2019. Larger gaps (> 1 month) were not interpolated and left as missing values.

## Statistical modelling and impact assessment

To assess the effects of the intervention on in-lake water chemistry (TP, SRP, chlorophyll *a*) we used Before-After (BA) and BACI analysis for the period before intervention (2012-2015) and after (2016-2019). All statistical analyses on field data were carried out on monthly observations, using R (R Core Team, 2020), with the mgcv (version 1.8; Wood, 2017) and emmeans (version 1.5.0; Lenth, 2020) packages.

## Before-After-Control-Impact (BACI)

This statistical design considers the relative change in the “impact” site compared to the “control” site using statistical analysis of the inter-lake differences. There were no known changes in land-use or catchment management during the study period at Blelham or Elterwater, except for the Elterwater flow diversion work, described above. It is assumed that any variation in the control site (Blelham) will be driven by inter-annual and seasonal variation in weather that would also drive similar variation within the nearby treatment site (Elterwater). Short-lived differences are likely to be masked by the noise contributed by other errors in the data and are unlikely to result in a long-term shift in conditions at the site (Lang et al., 2016).

The difference between lakes was calculated as Elterwater-IB minus Blelham. In the case of chlorophyll *a* and SRP*,* data were log-transformed before the differences were calculated, to account for positive skew and non-additivity in the data. To account for autocorrelation in the time series data, monthly data were used (Stewart-Oaten et al., 1986), and a temporal component (Season) was included in the models. The BACI analysis used two-way ANOVAs, fitted with an interaction between Intervention, before or after, and Season (winter - Dec, Jan, Feb; spring - Mar, Apr, May; summer - Jun, Jul, Aug; or autumn - Sep, Oct, Nov) to account for expected differences in responses between seasons and to minimise non-additivity issues (Stewart-Oaten et al., 1986). The assumptions of the models were checked visually using diagnostic plots of residuals and lag plots of autocorrelation (See Supplementary Information 4).

Statistical coherence of the control and impact sites before the intervention (2012-15) was confirmed using regression analysis of lake differences against date to ensure that the slope did not deviate significantly from zero ($p > 0.05$) (as per McGowan et al., 2005) (see Supplementary Information 5).

## Before-After (BA)

Before-After analysis of Elterwater-IB data was used as a confirmatory method to strengthen the results of the BACI analysis. BA models were fitted using generalised linear models (GLMs) with a gamma distribution and log-link function. The assumptions of the model were checked visually using diagnostic plots of residuals and lag plots of autocorrelation.

## Seasonality changes

Changes in the seasonal pattern of TP, SRP, and chlorophyll *a* following the intervention were assessed by fitting General Additive Models (GAM) using a Gamma distribution, with a log-link function and a lag-1 auto-correlation structure. The GAM included intervention (Before or After) as an ordered factor parametric term, plus an overall smoother for month, and a smoother for the difference between the Before and After periods as predictors of concentration.

## Hydrodynamic modelling

### Model description

Water temperature profiles were not taken in Elterwater-IB before May 2015, so before-after timeseries were not available. Therefore, a process-based physical lake model was used to derive hourly water temperatures at Elterwater-IB with and without the intervention. The lake version of the General Ocean Turbulence Model (GOTM), a one-dimension hydrodynamic model, uses measured meteorological data, specified bathymetry and inflow discharge and temperature to estimate in-lake water temperature profiles (Umlauf et al., 2005). GOTM uses a fixed layer structure and resolves turbulent kinetic energy production and diffusion between these layers to estimate vertical water temperature profiles. GOTM was run at an hourly timestep with 50 vertical layers from 2016-2019. Previous studies have successfully applied GOTM to a range of lake systems (Darko et al., 2019; Mesman et al., 2020; Moras et al., 2019).

The nearby automatic water quality monitoring buoy at Blelham Tarn measures the required input meteorology (i.e. air temperature, wind speed, relative humidity, and short-wave radiation). Gap filling of meteorological data was conducted using linear interpolation for small gaps (< 24 hours or 6 hours for short-wave radiation) and relationships with other local meteorological stations when there were larger gaps (Supplementary information 6). Alongside inflow discharge, as above, inflow temperature was measured on the diversion pipeline since July 2017. Before July 2017, hourly water temperature estimates were made based on a relationship derived between observations of inflow temperature and the previous 12 hours' average air temperature (Supplementary information 3).

### Model calibration and validation

GOTM was calibrated for Elterwater-IB using observed water temperature profiles from 2018 and validated using 2019 profiles. GOTM was calibrated using an auto-calibration tool, ACPy (Bolding & Bruggeman, 2017), which uses a differential evolution method to estimate the best parameter set, based on a maximum-likelihood measure. The parameters estimated were three non-dimensional scaling factors relating to wind speed (wsf), short-wave radiation (swr) and outgoing surface heat flux (shf) plus minimum kinetic turbulence (k-min) and non-visible (g1) and visible light extinction (g2). Model fit was assessed against observations for water column temperatures using the metrics root mean square error (RMSE), Nash-Sutcliffe efficiency (NSE), and mean absolute error (MAE), giving a good fit between the modelled and observed water temperatures in both the calibration and validation periods (Table 2). For a full description of the model parameters, ranges used in calibration, and the validation process, see Supplementary information 7.

*Table 2 The maximum, minimum and final parameters values, optimised during the auto-calibration route. Calibration parameters estimated: short-wave radiation (swr), outgoing surface heat flux (shf), wind speed (wsf), minimum kinetic turbulence (k-min), and visible light extinction (g2). Model performance statistics for the calibration (2018) and validation (2019) periods reported as root mean squared error (RMSE), Nash-Sutcliffe efficiency (NSE) and mean absolute error (MAE).*

| Calibration factor | Max allowable value | Min allowable value | Final parameter value |
|---|---|---|---|
| swr | 1.1 | 0.85 | 0.95 |
| shf | 1.2 | 0.8 | 0.80 |
| wsf | 1.1 | 0.9 | 1.08 |
| k-min | 1.0 $e^{-5}$ | 1.4 $e^{-7}$ | 1.4 $e^{-7}$ |
| g2 | 2.0 | 0.5 | 0.61 |

| | RMSE (°C) | NSE | MAE (°C) |
|---|---|---|---|
| Calibration | 0.93 | 0.97 | 0.72 |
| Validation | 0.97 | 0.92 | 0.75 |

The resulting model was used in two scenarios: 1) a 'with intervention scenario' using observed input data to predict the actual water temperatures in Elterwater-IB from 2016-2019; 2) a 'no intervention scenario' using inflow discharge without the additional inflow from the intervention, to estimate water temperatures if no intervention had occurred (2016-2019), thus isolating the impact of the intervention on the lake's thermal structure. The water temperature profiles were averaged on to a daily timestep.

## Stratification and stability metrics

Metrics of stratification length and water column stability were calculated. A minimum density difference between the top and bottom of the water column of 0.1 kg $m^{-3}$ was used to define stratification occurrence (Wilson et al., 2020). Based on this 0.1 kg $m^{-3}$ threshold, the following stratification metrics were calculated:

i. Total number of stratified hours (as day equivalents)
ii. Length of the longest continuously stratified period
iii. Onset and overturn dates of the longest stratified period

Water column stability during the stratified period, measured as Schmidt stability (Idso, 1973), was calculated using the rLakeAnalyzer R package (version 1.11.4.1; Read et al., 2011). Mixed depth was calculated using a modified version of the rLakeAnalyzer meta.depths function, using a density difference threshold (0.1 kg $m^{-3}$) between the top and bottom water layers and a minimum density gradient of 0.1 kg $m^{-3}$ $m^{-1}$ to define the mixed depth.

# Results

## Changes to WRT

Using measurements of the intervention flow and estimates of the natural flow, we compared WRTs with and without the additional piped water for the period after intervention (2016-2019). Summer average WRT was reduced by 8 ± 5 days (mean ± standard deviation) with the intervention in place. Spring and summer had larger reductions in WRT than winter and autumn. Overall, the mean annual WRT was around 5 ± 1 days shorter with the intervention than without. Within season variability was also large; daily WRTs varied by orders of magnitude within seasons, in summer ranging from almost 700 days to < 2 days without the intervention. With the intervention the variability was reduced as the longest WRTs were suppressed, reducing the maximum daily WRTs from 700 to 462 days.

## Changes to stratification and lake water temperatures

Comparing the two modelled temperature scenarios for 2016-2019, one with the intervention and the other without, shows predicted changes caused by additional flow from the diversion pipe. These changes depend on the season. Overall, with the intervention, average water column temperature was cooler in summer (difference in average temperature -0.7 ± 0.2 °C) and in winter the lake was warmer (change in average +0.5 ± 0.1 °C) (Figure 3a). Spring and autumn changes in average temperatures were smaller than in summer and winter. The change in water temperatures with the intervention varied by depth, with surface water temperatures (SWT) generally showing larger differences than deeper water except in autumn when deeper water cooled more than surface water (Figure 3b).

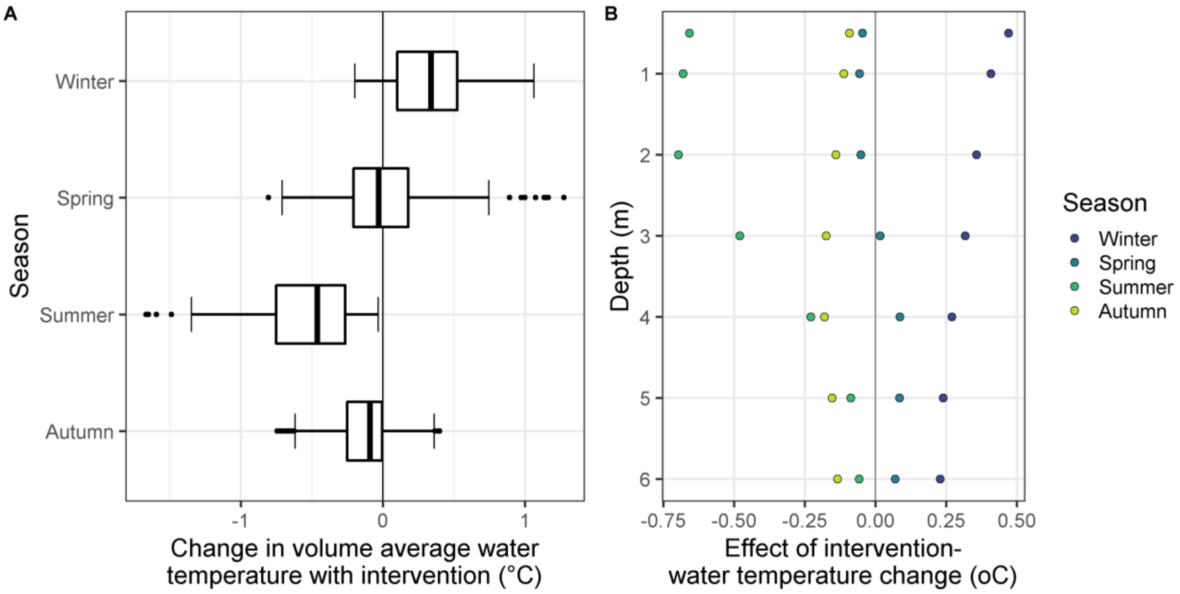

*Figure 3 a) change in volume averaged water column temperature in each season, b) water temperature changes at different depths with the intervention. Positive values indicate warming and negative values cooling, compared to water temperatures without the intervention.*

Overall, increased flow tended to increase stratification in the winter and reduce stratification in the summer. The modelling results show that stratification occurrence increased following the intervention, due to increases in transient stratification during the winter. Without the intervention the average number of stratified days per year would have been 177 days (min =168, max =183), compared to 179 days per year with the intervention (min = 171, max = 185). However, the intervention, on average, shortened the longest continuous period of stratification in the summer by 2 days, from 156 (min = 145, max = 165) days to 154 (min = 145, max = 163). With the intervention, Elterwater-IB's average stratification onset remained the $14^{th}$ April (earliest = $28^{th}$ March, latest = $1^{st}$ May) but overturn was expedited by 2 days from $16^{th}$ to $14^{th}$ September (earliest = $7^{th}$ September, latest = $29^{th}$ September).

Average Schmidt stability for the stratified period ranged from 11.1 J $m^{-2}$ to 20.6 J $m^{-2}$ without the intervention and 9.8 J $m^{-2}$ to 20.0 J $m^{-2}$ with the intervention. The mean value decreased from 15.4 to 14.4 J $m^{-2}$, a reduction of 7%.

## Natural inter-annual variability

The "natural" inter-annual variability of annual and summer WRTs in Elterwater-IB for the 8 years of the study (4 years before plus 4 years after without intervention) was assessed (*Table 3*). Summer WRT varied by 23 days (maximum = 31 days, minimum = 8 days) and annual WRT by 6 days, across the 8 years. This inter-annual variability in summer WRTs was almost three times the reduction estimated to have been caused by the intervention (i.e. 8 days). Modelled mean summer SWTs across the 8 years varied by 5 °C between years and maximum summer SWTs varied by 7 °C. This variability in mean summer SWT was an order of magnitude larger than the 0.7 °C cooling estimated to have been caused by the intervention. Inter-annual variability in mean winter SWTs was 1.6 °C and minimum winter SWTs was 1.9 °C, three to four times the estimated effect of the intervention (0.5 °C). The change in dates of stratification onset and overturn were minor compared to the variation in onset and overturn dates that were estimated to occur naturally in Elterwater-IB. The onset date varied by 49 days and overturn by 25 days.

*Table 3 Summary of the water residence time, water temperatures and stratification metrics for the two modelled scenarios (2016-2019 with and without intervention) compared with the natural variability in parameters (2012-2019 without intervention). Values show the mean and the range (minimum – maximum).*

| | **With (2016-2019)** | **Without (2016-2019)** | **Natural variability (2012-2019)** |
|---|---|---|---|

| | Mean | Range | Mean | Range | Mean | range |
|---|---|---|---|---|---|---|
| **Annual WRT (days)** | 8 | 7 - 9 | 13 | 11-15 | 12 | 9 - 15 |
| **Summer WRT (days)** | 10 | 7-15 | 18 | 13-31 | 19 | 8 - 31 |
| **Mean summer SWT (°C)** | 17.9 | 16.2 - 20.2 | 18.5 | 17.0 - 20.7 | 16.1 | 14.1 - 19.2 |
| **Max summer SWT (°C)** | 23.8 | 22.5 - 26.6 | 24.5 | 23.4 - 26.7 | 24.0 | 20.0 - 27.0 |
| **Mean winter SWT (°C)** | 5.1 | 4.7-5.4 | 4.6 | 4.0 - 5.1 | 4.9 | 4.2 - 5.8 |
| **Minimum winter SWT (°C)** | 2.5 | 2.4 - 2.7 | 2.4 | 2.1 - 2.7 | 2.4 | 1.5 - 3.4 |
| **Days stratified** | 179 | 171 - 185 | 177 | 168 - 183 | 177 | 166 - 192 |
| **Longest period of stratification (days)** | 154 | 145 - 163 | 156 | 145 - 165 | 155 | 121 - 186 |
| **Day of onset** | 14th Apr | 29th Mar - 1st May | 14th Apr | 29th Mar - 1st May | 17th Apr | 29th Mar - 17th May |
| **Day of overturn** | 14th Sep | 7th Sep - 29th Sep | 16th Sep | 7th Sep - 29th Sep | 18th Sep | 7th Sep - 2nd Oct |
| **Stratification stability (J m$^{-2}$)** | 14.4 | 9.8-20.0 | 15.4 | 11.1-20.6 | 14.2 | 10.0 - 20.6 |

## Changes in lake water quality

Annual mean TP concentrations in Elterwater-IB showed a slight change in the “After” period compared to the “Before”, relative to the control site, although this was not significant ($p$ = 0.538). There was a significant interaction of season and period ($p$ = 0.025) in the BACI ANOVA, which indicates the response of TP concentration in each period differed depending on the season. In winter and autumn, the TP concentration increased relative to control, but in spring and summer there was a relative decrease (Figure 4a). However, post-hoc analysis indicated that only the spring intervention effect was significant ($p$ = 0.017) with an average reduction in Elterwater-IB relative to the control (*Table 4*). The BA analysis confirmed these responses (Supplementary information 8), with no significant overall effect of the intervention or an effect in individual seasons ($p$ > 0.05). There was also no significant change in the seasonality of TP concentrations (Supplementary Figure 9.1a), confirmed by the fitted GAM curve for After being no different to Before ($p$ = 0.721).

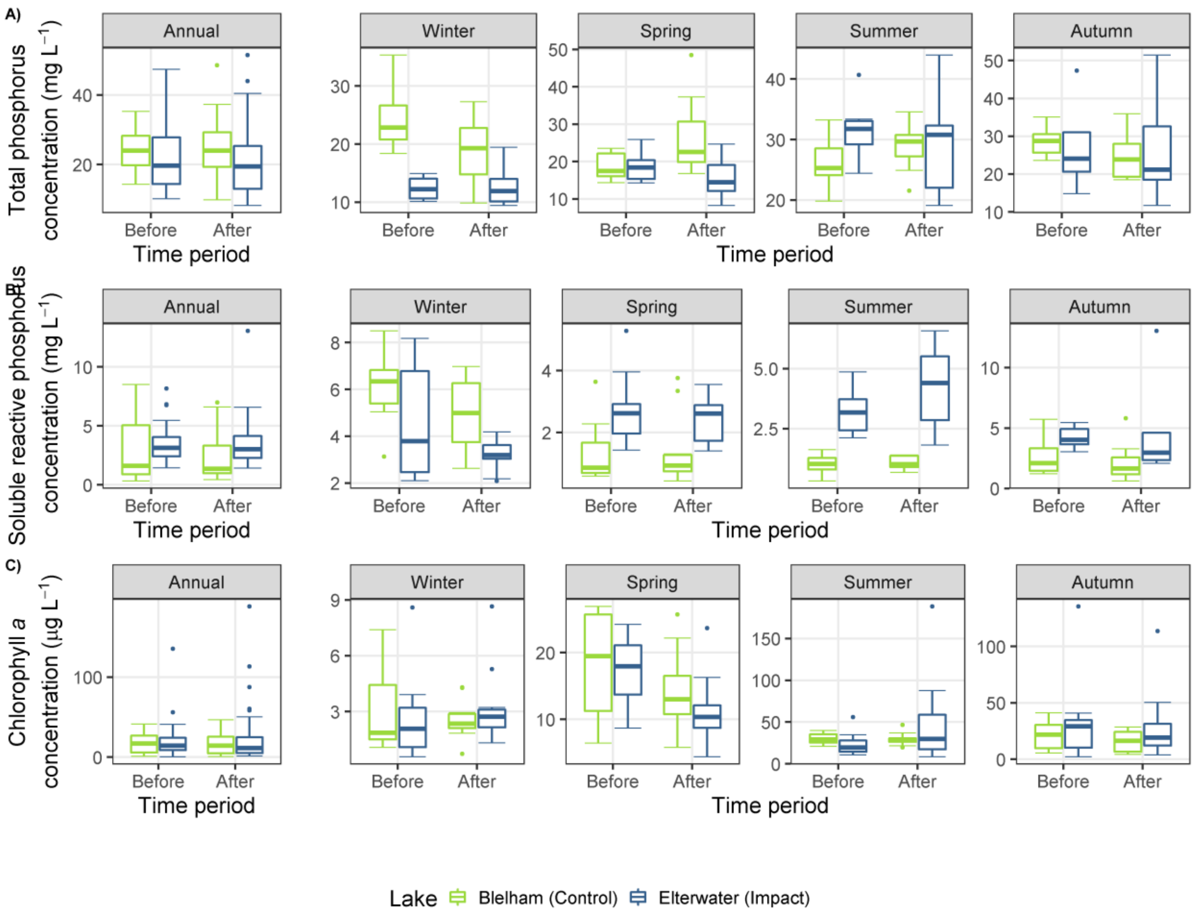


*Figure 4 a) Total phosphorus, b) soluble reactive phosphorus and c) chlorophyll a concentrations in Elterwater-IB (impact) and Blelham (control) sites before and after intervention, annually and grouped by season.*

BACI analysis indicated that the annual mean SRP concentration at Elterwater-IB was unchanged following the onset of the intervention (*Table 4*) and no significant difference in the response between seasons (Figure 4b). The BA analysis confirmed these responses (Supplementary 8). There was also no significant change in the seasonality of SRP concentration (Supplementary figure 9.1b), confirmed by the fitted GAM curve for After being no different to Before ($p$ = 0.164).

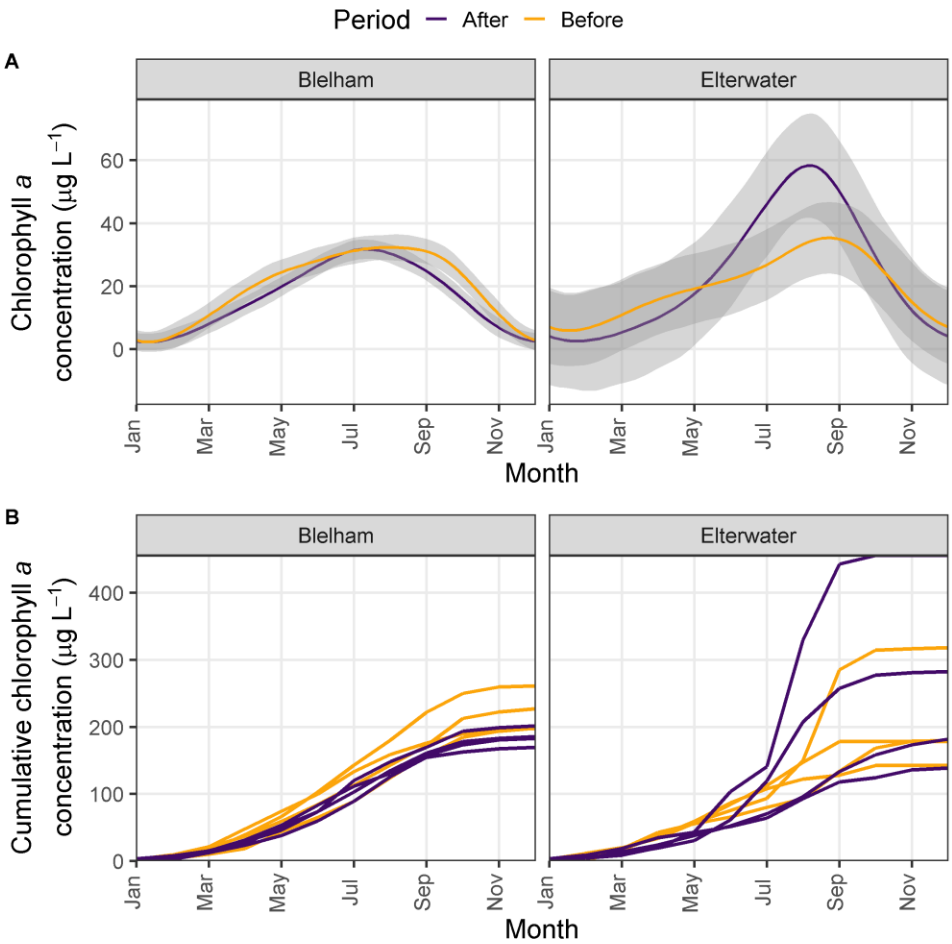


*Figure 5 a) average annual chlorophyll a dynamics (shading shows +/- 1 SE) and b) cumulative chlorophyll accumulation for each year before and after restoration at the control (Blelham) and impact (Elterwater-inner basin) lakes.*

Annual mean chlorophyll *a* concentration in Elterwater-IB increased slightly following the intervention, relative to the control (*Table 4*). However, this increase was not significant ($p$ = 0.702) and there was no significant interaction between season and period (p= 0.093). Post-hoc analysis indicated no significant change in any individual seasons (Figure 4c; $p$ > 0.05). There was also no significant change in the seasonality of the chlorophyll *a* concentration (Figure 5), confirmed by the GAM modelling approach ($p$ = 0.364). However, there was large inter-annual variability in the seasonality of chlorophyll *a* dynamics (Figure 5), including elevated summer chlorophyll concentrations in both the Before and After periods.

*Table 4 Before-After-Control-Impact (BACI) analysis results using a two-way Analysis of Variance including the effect of the interaction of Period (Before and After) and season on mean difference between lakes (Elterwater – Blelham). Where a significant interaction was found, post-hoc analysis of contrasts was done to look at Period effect in individual seasons. Bold face and asterisks denote significant results at the 0.05 (*) level.*

| | Mean difference ± SD | | P-value | |
|---|---|---|---|---|
| | Before | After | Period × Season | Period |

| | | | | |
|---|---|---|---|---|
| TP (annual) | -2.5 ± 11.0 | -3.9 ± 9.3 | **0.025 *** | 0.538 |
| Winter | -12.9 ± 5.4 | -5.9 ± 4.2 | | 0.117 |
| Spring | 0.0 ± 5.5 | -10.4 ± 12.1 | | **0.017 *** |
| Summer | 5.6 ± 7.6 | -0.1 ± 7.36 | | 0.191 |
| Autumn | -2.7 ± 16.9 | 0.8 ± 7.0 | | 0.484 |
| SRP (annual) | 1.1 ± 2.0 | 1.3 ± 2.8 | 0.748 | 0.588 |
| Winter | -1.5 ± 2.2 | -1.8 ± 1.5 | | - |
| Spring | 1.4 ± 1.5 | 1.1 ± 1.1 | | - |
| Summer | 2.2 ± 1.0 | 3.2 ± 1.8 | | - |
| Autumn | 1.8 ± 1.0 | 2.7 ± 3.8 | | - |
| Chlorophyll *a* (non-transformed data) | -0.1 ± 19.5 | 7.3 ± 30.7 | 0.093 | 0.702 |
| Winter | -0.3 ± 2.3 | 0.7 ± 2.4 | | - |
| Spring | -0.8 ± 6.9 | -2.9 ± 5.0 | | - |
| Summer | -6.7 ± 10.7 | 18.3 ± 52.9 | | - |
| Autumn | 9.7 ± 38.7 | 12.6 ± 26.2 | | - |

Profiles from 2018-2019 showed that following the intervention anoxia occurred in up to 20% of the lake volume during the summer (Figure 6). In addition, accumulation of BAP persisted in the deepest part of the lake, indicating persistent internal loading (Figure 6). Concentrations of BAP at 6 m were > 5 times that at 0.5 m during the summer stratified period (Figure 6). Furthermore, the mass-balance results suggest that internal loading contributed three times more to the TP budget than external loads during summer following the intervention (Supplementary Information 10).

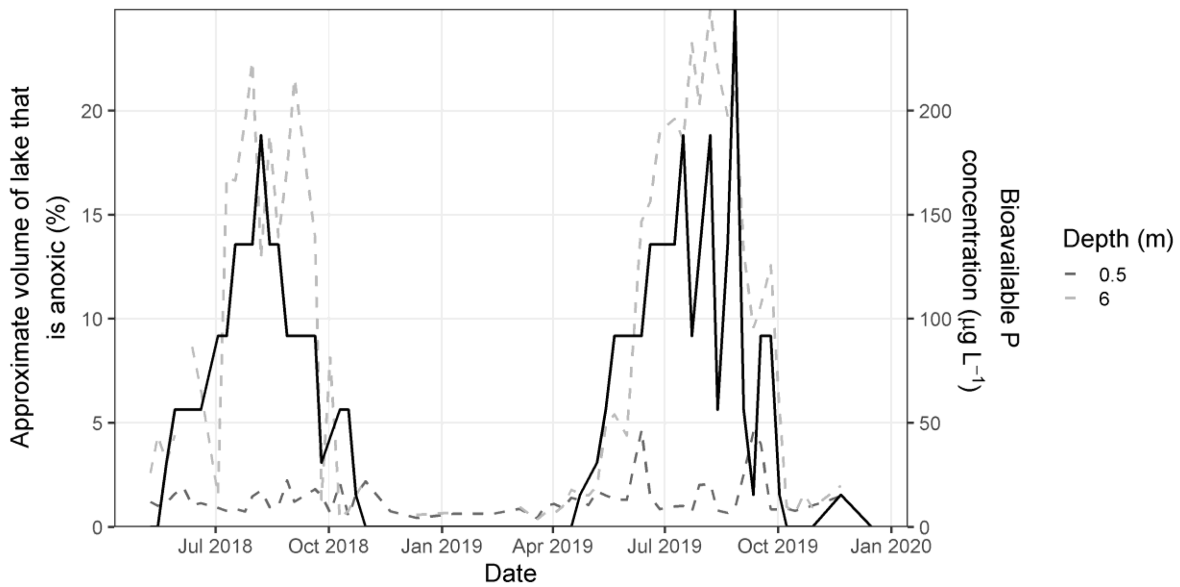


*Figure 6 Approximate volume of Elterwater-IB that is anoxic (< 2 mg L$^{-1}$, solid line) and concentration of biological available phosphorus (dashed lines) in the surface (0.5 m) and hypolimnion (6 m) based on profiles collected from May 2018 - December 2019.*

# Discussion

## Changes to Elterwater-IB WRT

This study aimed to assess the success of the WRT manipulations in affecting lake physical structure and water quality, based on four elements: 1) WRTs, 2) thermal structure, 3) development of hypolimnetic anoxia, and 4) nutrient and chlorophyll *a* concentrations. Although there was some change in WRTs in response to the intervention, it was not sufficient to have a large effect on the thermal structure, with anoxia persisting. No evidence for significant reductions to internal loading or water column nutrient and chlorophyll *a* concentrations were reported. The intervention reduced annual WRT by 40% and seasonal WRTs by 29 – 45%, equivalent to an 8 day reduction in summer, a smaller change than the natural variability in WRT between years. The intervention change in WRT was less than the dilution efforts at Moses Lake, Lake Veluwe and Lake Taihu, which reduced annual WRTs by 63% (Welch et al., 1992), 75% (Cooke et al., 2005; Hosper and Meyer, 1986) and 51% (Hu et al., 2010), respectively. In the years when flushing and dilution occurred in Moses and Veluwe Lakes, both non-stratifying lakes, more than 50% reductions in TP concentrations were reported and chlorophyll *a* concentrations reduced (Cooke et al., 2005; Hosper and Meyer, 1986; Welch et al., 1992). The greater success of these schemes could relate to the larger overall reductions in WRT, the initial WRT, and the rate of flushing being maintained through pumping rather than via the passive flow diversion used in Elterwater. Water quality conditions prior to intervention were also worse than at Elterwater, leaving the potential for greater improvements.

## Changes to thermal structure and stratification

The change in WRT had a small but quantifiable impact on Elterwater-IB's physical structure, partially satisfying element two of the assessment. The summer stratified period was shortened by 2 days and was 7% less stable, and average summer surface water temperatures were 0.7 °C cooler with the intervention, demonstrating the cooling effect of the inflow in summer (Carmack et al., 1979; Richards et al., 2012). However, the cooling effect was not sufficient to break down stratification and considerably larger changes in WRT would be required to modify the heat budget sufficiently to induce the desired changes to thermal structure.

In the summer, lake surface heat fluxes are likely to be considerably larger than the heat flux exerted by the inflows (Livingstone and Imboden, 1989) and are therefore difficult to overcome, despite a 45% decrease in WRT. Previous modelling studies in two reservoirs, considering a range of WRTs, showed that maintained flow rates can affect stratification (Li et al., 2018; Straškraba and Hocking, 2002). Halving reservoir WRT, using a constant inflow rate, resulted in a 30 day shorter stratified period, with both later onset and earlier overturn (Li et al., 2018). However, Straškraba & Hocking

(2002) suggest that below a WRT of 200 days, WRT changes have an effect on stratification stability, but not on stratification length. Field evidence linking shorter WRTs with shorter stratified periods is often gathered at times of concurrent meteorological change (Andersen et al., 2020; Woolway et al., 2018), which both highlights the importance of transient weather effects for stratification, but also partially confounds our ability to discern the specific effects of WRT change. The minor changes in stratification length, following the intervention, suggests larger WRT reductions or WRT reductions targeted at specific times of year may be needed in Elterwater-IB to effect a larger change in the thermal structure of the lake and consequent water quality.

## Prevention of hypolimnetic anoxia and internal loading as a means to reduce nutrient and phytoplankton concentrations

Our results showed evidence of continued hypolimnetic anoxia, and the accumulation of BAP in deep water, suggesting that internal loading continued in Elterwater-IB following the intervention. Previous water diversion interventions provide limited evidence of their effect on internal loading, as the target action was dilution and flushing of nutrients and phytoplankton biomass. However, evidence from Moses Lake suggests internal loading may have increased following increased flushing rate due to greater sediment resuspension (Welch and Patmont, 1980). Our results also indicate that internal loading remains a dominant source of P to the water column. For our study site, the water source of the intervention is lower in nutrients than the lake (APEM, 2012) and, where internal loads exceed external loads, the additional flow may act to dilute nutrient concentration (Elliott et al., 2009; Jones et al., 2011). No significant change in P concentration following the intervention, suggests that there is either no dilution effect or the effect is being compensated for by additional internal loading.

WRT changes can affect phytoplankton concentrations in several ways, including through changes to thermal structure, nutrient loading, and direct flushing. In our study, in-lake chlorophyll *a* concentrations did not change after the intervention, suggesting the change in WRT was insufficient to reduce phytoplankton growth by reducing internal loading or by increasing losses through increased flushing. Reducing WRT has been shown to reduce phytoplankton biomass through flushing in other studies (Hosper and Meyer, 1986; Welch et al., 1992), although drawing direct conclusions on cause-effect in case studies is often complicated as a result of the co-implementation of multiple restoration measures including external load reduction. A modelling study with a 50% increase in discharge only reduced chlorophyll *a* by 12% (Zhang et al., 2016), failing to reduce nutrient concentrations or impact growth rate sufficiently. Although not assessed here, decreases in WRT can also impact losses of zooplankton to a greater extent than phytoplankton where the flushing rates exceeds the growth of the former but not the latter (Obertegger et al., 2007; Rennella and Quirós, 2006). In our study, the lack of response in chlorophyll *a* concentration is not surprising given that nutrient concentrations also did not change, a result that is commonly reported in similar hydrological interventions across other case studies (Hu et al., 2010; McGowan et al., 2005; Zhang et al., 2016). No significant change in chlorophyll *a* concentration with the intervention indicates that either none of the flow-related mechanisms had much obvious effect, were cancelling each other out, or the system was resilient to WRT changes within hydrological and physical conditions naturally occurring in Elterwater. The modest change in WRT might alter species composition, by selecting for taxa with different growth rates (Reynolds et al., 2012) potentially occurring before any noticeable change in total biomass.

## Implications for future restoration in Elterwater and other lakes

The success of lake restoration measures, particularly over the long term, is often limited (Jilbert et al., 2020; Søndergaard et al., 2007). One common reason for failure is insufficient understanding of site-specific aspects of lake functioning, water residence times, and nutrient sources before restoration measures are implemented (Hamilton et al., 2018; Lürling et al., 2016).

Using a hydrodynamic model, we demonstrated that a decrease in WRT in Elterwater-IB had a measurable but minor effect on stratification and water temperature compared to the natural variability of these physical variables. Other research has shown that inter-annual variability can far exceed restoration or management impacts (e.g. Fink et al., 2014), a consequence being that longer time series are needed to confirm, statistically, treatment effects. For example, in this study the large inter-annual variability in chlorophyll *a* concentrations may have masked smaller but genuine impacts that could only be identified statistically with a much longer timeseries of data. Therefore, understanding the inter-annual variability over significant time periods is crucial to design effective schemes and detect “real” and relevant changes. This highlights the need for multi-year studies to fully determine intervention effects (Smokorowski and Randall, 2017), also allowing for adaptive management of protected sites (Tanner-McAllister et al., 2017), crucial for long-term management success.

In addition, single site BA studies can fail to capture changes caused by external factors, co-occurring with management (Stewart-Oaten et al., 1986), and including a “control” site in BACI analysis, partially alleviates this (Smokorowski and Randall, 2017; Stewart-Oaten et al., 1986). Had we not employed the BACI approach, utilising long-term data, we may, for example, have incorrectly concluded that the apparent decrease in spring TP and chlorophyll *a* concentrations in Elterwater-IB (Figure 4) were direct effects of the intervention, whereas, similar changes were observed in our control lake. However, “control” systems at the ecosystem scale may not be considered perfect controls (Kerr et al., 2019), but do provide comparisons to identify variations in response to local climate effects (Schwartz, 2015).

For the hydrological interventions at Elterwater, a simple 1-D hydrodynamic model allowed us to isolate the intervention effects, however small, of modifying inflow rates on lake stratification. This modelling approach is transferable to other sites and could be applied to conduct site-specific assessments to inform the suitability of this approach given specific lake characteristics, heat budgets, and local weather variability. A pre-intervention modelling study of this kind, for example, may have been useful in setting intervention targets, with respect to indicators presented in this paper. We recommend that additional modelling be conducted in this respect to aid the implementation of future WRT management interventions.

## Conclusions

Water residence time was reduced in all seasons in Elterwater-IB and this had quantifiable, if small, effects on lake temperatures. The extent of the change in the lake’s thermal structure was insufficient to induce significant changes in water quality, with summer stratification, seasonal anoxia, and internal loading persisting after the intervention. Greater changes, or WRT manipulations that target particular times of year, would be needed to modify lake physical structure sufficiently to inhibit stratification in this lake, whilst also considering undesirable effects on ecosystem function. Hydrodynamic modelling and a systems approach to lake restoration, including knowledge of nutrient sources and inter-annual variability, would be crucial to determine the magnitude of change required to produce a measurable effect on lake water quality. This study suggests that WRT manipulations can impact lake thermal structure in short-residence time lakes, although the extent of the change is lake dependent, requiring in depth investigations to determine likely effectiveness prior to any intervention as well as long-term targeted monitoring before and after intervention.

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