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# Nutrient futures for Windermere

Chief Scientist's Group report

May 2026

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Dr Robert Bradburne  
**Chief Scientist**

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## Executive summary

Human activities are increasing the amount of nutrients entering freshwater environments. Microscopic organisms called phytoplankton use nutrients to grow. These organisms naturally occur in rivers and lakes, but their excessive growth can reduce the light and oxygen available to other plants and animals. Some phytoplankton can produce toxins, which can be harmful to aquatic life and human health, and limits the use of freshwaters for drinking water, food production, recreation and tourism.

Climate change is expected to worsen this problem in lakes worldwide. Increasing temperatures and changing rainfall patterns can influence how nutrients move across catchments and into lakes. In addition, increasing lake water temperatures can accelerate phytoplankton growth rates. Understanding how these processes could affect lakes will help the Environment Agency and others adapt nutrient regulation and management, to better protect lake ecosystems in a changing climate.

This study investigated how future climate change will impact lake phytoplankton communities in the Windermere catchment, in the English Lake District. Windermere is facing changing nutrient pressures that have impacts of concern to local and visiting communities. This study considered the two basins of Windermere as well as the smaller connected Esthwaite Water. We combined a catchment water quality model with a phytoplankton community model. We then subjected these models to 5 future climate and nutrient management scenarios.

The results suggest that if current nutrient management (2015 to 2020) is continued, then climate change will have a small impact on the load of nutrients entering Windermere and Esthwaite Water from upstream in the far-future (2076 to 2079). However, these nutrients will be more concentrated. Relative to baseline values, modelled concentration increases for the two main types of nutrients were about 33% for phosphate and 13% for nitrate in Windermere North Basin and about 4% for phosphate and 10% for nitrate in Windermere South Basin by 2076 to 2079. Importantly, these concentration increases were greatest in the summer months, when phytoplankton are best able to use these nutrients for growth.

Climate change also increases the amount (biomass) of phytoplankton in Windermere. The phytoplankton community will become dominated by bacteria called cyanobacteria. Also known as blue-green algae, some types of cyanobacteria produce harmful toxins. The subsequent risk posed to human health was quantified by the number of days per year that cyanobacteria biomass exceeds World Health Organisation thresholds (WHO10). These thresholds indicate when cyanobacteria growth reaches unsafe levels in drinking and recreational waters, with adverse effects on human health.

We found that climate change warms lake water temperatures by up to 2.5°C by 2076-2079. In addition, the time for which lake waters layer up (stratify) in summer extends by about a month. This matters because layering separates warmer, oxygen rich surface waters from cooler, denser bottom waters. This can favour cyanobacteria growth, as some

migrate vertically through the water column and as they move closer to the surface, they use resources that other organisms need to survive.

All the modelled nutrient management scenarios reduce phytoplankton biomass and WHO10 exceedances in Windermere and Esthwaite Water in the period 2076 to 2079.

**Table: Number of days per year where cyanobacteria biomass exceeds the WHO10 threshold (WHO10) for each lake in each modelling scenario**

Modelling scenario	Windermere North Basin	Windermere South Basin	Esthwaite Water
<b>Baseline</b>	8	50	159
<b>Climate change 2076-2079</b>	36	92	167
<b>Scheduled water quality improvements</b> climate change with reductions in phosphate from water company assets scheduled for 2030 (AMP8)	24	3	151
<b>Low agricultural inputs</b> with climate change and AMP8	14	0	136
<b>No sewage</b> climate change with no nutrient inputs from water company or privately owned wastewater treatment facilities	0	0	105

The modelling showed that nutrient reductions lead to improvement in the ecological quality of the lakes as defined by Water Framework Directive (WFD) class. Complete removal of wastewater inputs resulted in the greatest reductions in future phytoplankton biomass, including cyanobacteria, and the highest projected WFD class.

Despite experiencing the greatest reductions in phytoplankton biomass, Esthwaite Water continues to exceed the WHO10 threshold in each nutrient management scenario. In effect, none of the modelled nutrient reductions were sufficient to prevent future cyanobacteria biomass exceeding levels considered harmful to human health. This may be because as a shallow lake, Esthwaite Water is more sensitive than the deeper Windermere basins to the impact of nutrients released from its lakebed sediments and catchment. It also has a different nutrient history related to aquaculture.

The results demonstrate that local nutrient management can help to reduce some of the pressures placed on lake environments by global climate change.

# 1. Introduction

Inputs of nutrients are harming the ecosystems of freshwater lakes globally (Costa and others, 2023; Suresh and others, 2023). Increasing nutrient transport to lakes fuels the prolific growth of naturally occurring phytoplankton, photosynthetic microorganisms, with negative consequences for aquatic food webs and human health (Reynolds, 2006; Codd and others, 2020; Amorim and Moura, 2021). Climate change is expected to exacerbate this process by altering the movement of nutrients across lake catchments and increasing water temperatures which further amplify phytoplankton growth rates (Woolway and others, 2022; Costa and others, 2023; Dory and others, 2024; Tigli and others, 2025). In England, this poses a challenge to future water quality and restricts the provision of lake ecosystem services including drinking water, food production, tourism and recreation (EA, 2022).

This case study investigated climate-nutrient-phytoplankton interactions in the Windermere catchment of the English Lake District. Long-term water quality monitoring ensures these lake systems are well characterised in the past and present (Maberly and Elliott, 2012) but less is understood of how they will respond to anthropogenic pressures in the future. Quantifying the impact of climate change on these lake phytoplankton communities will inform adaptive regulation and effective nutrient management in a warming world.

## 1.1 Aim, objectives and approach

The aim of this study was to better understand how future climate change will impact catchment nutrient supply and lake phytoplankton community structure. Implications for future management of nutrient inputs to protect lake ecosystems were explored. We addressed the following objectives to:

1. Determine how nutrient dynamics (the transport of nutrients throughout a catchment) are influenced by climate change in the Windermere catchment
2. Identify how changes in climate and upstream nutrient dynamics influence phytoplankton community structure in Windermere North Basin, Windermere South Basin and Esthwaite Water
3. Test nutrient management scenarios to explore their ability to counteract the impacts of climate change in Windermere and Esthwaite Water.

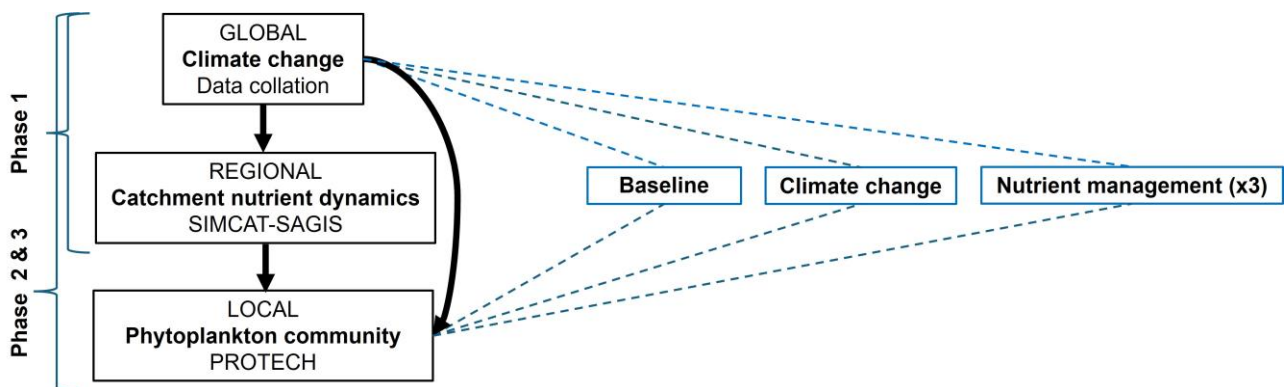
Models were used to investigate how climate, hydrology and water quality parameters in the upstream Windermere catchment influenced the response of lake phytoplankton communities in three sequential phases which aligned to the three research objectives:

**Phase 1: Climate change impacts on catchment nutrient dynamics:** The catchment water quality model SIMCAT-SAGIS (SIMulations of water quality in CATchments - Source Apportionment Geographical Information System) was used to characterise nutrient dynamics in the Windermere catchment in baseline and future climate change (2076-2079) scenarios.

**Phase 2: Climate change impacts on lake phytoplankton communities:** The phytoplankton community model PROTECH (Phytoplankton RespOnses To Environmental CHange) was used to determine how phytoplankton biomass and community composition responded in baseline and future climate change (2076-2079) scenarios.

**Phase 3: Nutrient management for lake catchments in a changing climate:** The impact of three nutrient management scenarios on catchment nutrient inputs to the lakes were determined using SIMCAT-SAGIS, and their influence on phytoplankton communities explored using PROTECH. Model outputs were used to identify if these management scenarios addressed the alterations to lake phytoplankton communities expected to result from climate change.

The three phases of work considered how the relationships between global climate change, regional nutrient dynamics, and local lake phytoplankton communities will respond to changes in climate and nutrient management in future (Figure1).



**Figure 1: Schematic representation of the study structure.**

## 2. Lake catchments in a changing climate

### 2.1 Catchment nutrient dynamics and lake phytoplankton communities

Phosphorus and nitrogen are distributed throughout catchments in particulate and dissolved forms (Edwards and Withers, 2008). In England, they originate from point sources – such as inputs from sewage treatment works and septic tanks, and diffuse sources – dispersed contributions including livestock manures that are washed from the soil, alongside runoff from roads and urban and industrial centres (Edwards and Withers, 2008; Brownlie and others, 2022). These anthropogenic sources are accompanied by natural contributions from rock weathering and atmospheric deposition, which can be exacerbated by human activity (Whitehead and Crossman, 2012).

Nutrients are transported from the land, to downstream running waters and lakes, through to oceans (Li and others, 2024). They move via hydrological pathways including groundwater and aquifer flows, leaching through soils, and surface water transfer (Costa and others, 2023). The mechanisms and rates of dissolved nutrient mobilisation and retention are determined by a range of catchment characteristics including precipitation patterns, topography, hydrological connectivity, soil type, vegetation cover and land use (Brownlie and others, 2022). As nutrients can accumulate in catchments over time, waterbodies may receive legacy inputs that predate current land management practices (Martin and others, 2021).

Lake systems integrate nutrient inputs from atmospheric deposition, terrestrial sources and freshwater inflows (Costa and others, 2023). This nutrient supply critically underpins the growth of aquatic plants, which form the base of freshwater food webs and so support whole lake ecosystems (Carpenter, 2005; Amorim and Moura, 2021; Brownlie and others, 2022). However, anthropogenic activity has significantly modified the cycling of nutrients through catchments worldwide (Steffen and others, 2015; Tu and others, 2023), with a six-fold increase in annual inputs of phosphorus to European lakes identified from pre-industrial to present times (Tu and others, 2023). In England, 73% of assessed lakes currently fail to meet the phosphorus standards required by the Water Framework Directive (WFD) (EA, 2019). These increased nutrient inputs can lead to anthropogenic eutrophication - the proliferation and accumulation of aquatic plant biomass to a point where it becomes harmful to lake ecosystems and human health (EA, 2022; Pannard and others, 2024).

Eutrophic lakes are often characterised by algal blooms. These are rapid surface accumulations of phytoplankton, a diverse group of photosynthetic microscopic organisms, that can be harmful to other living things (Reynolds, 2006). Bloom formation can be predicated on factors including high nutrient levels, water temperature and light availability, alongside a stable water column with limited vertical mixing (Sharma and others, 2011). Algal blooms often include cyanobacteria, phytoplankton also known as blue-green algae,

which can produce cyanotoxins. These are harmful to wild and farmed aquatic species, and terrestrial animals, including livestock and pets, that feed upon cyanobacteria, or use the lake for drinking water (Igwaran and others, 2024). The World Health Organisation (WHO) provides guideline thresholds for cyanobacteria biomass in waters used for drinking and recreation (Chorus and Bartram, 1999) as cyanotoxin exposure can fatally impact human health, and incur costly remediation (Codd and others, 2020; Igwaran and others, 2024). This impinges upon the ecosystem services provided by lakes, including food and drinking water, recreation, wellbeing and tourism (Jenny and others, 2020).

## 2.2 Climate change and catchment nutrient dynamics

Climate change is driving worldwide changes to catchment nutrient dynamics (Costa and others, 2023). Increasing near-surface temperatures will cause alterations to global precipitation patterns and increase the frequency of extreme rainfall events and floods (Costa and others, 2023; IPCC, 2007). These events are expected to acutely increase rates of nutrient mobilisation, leaching and runoff from soils to watercourses, and move particulate phosphorus bound to sediments originating from both historic and current sources (Ockenden and others, 2016; Wagena and others, 2018). This will increase the loading of diffuse nutrients to freshwater systems across temperate regions (IPCC, 2007), with soil type, texture and management practices locally mediating these impacts (Ezzati and others, 2023).

Climate change is also expected to increase the intensity of summer droughts and subsequent low or zero flows in rivers in temperate regions (IPCC, 2007). This will reduce the volume of water available to dilute nutrient inputs to downstream lakes (Ockenden and others, 2016). Reductions in summer flow may be exacerbated as warmer air temperatures potentially increase evapotranspiration rates (Wagena and others, 2018). Heavy rainfall and rewetting of riverbeds can flush nutrients that have accumulated during the dry period as a pulse to downstream waters (Van Vliet and others, 2023). This effect has been found to increase seasonal phosphorus and nitrogen loads in agricultural catchments (Ezzati and others, 2023).

Global changes to climate will alter regional nutrient dynamics but the impact on catchments and lake waterbodies will be site specific, given the differences in catchment and lake characteristics, seasonal variability, and land use and management which may itself alter in response to climatic changes (Moorhouse and others, 2014; Costa and others, 2023; Grigorieva and others, 2023; Suresh and others, 2023). In addition, the impacts of climate change will interact with other anthropogenic activities that alter the cycling of nutrients across catchments, including population growth, urbanisation, deforestation and food production (IPCC, 2007; Van Vliet and others, 2023).

## 2.3 Climate change and lake phytoplankton

Climate change is altering patterns of nutrient cycling across catchments, impacting the input of phosphorus and nitrogen to lake systems (Costa and others, 2023). Across Europe, the expected increase in winter precipitation and extreme rainfall events may increase the runoff of nutrients to lakes (Nöges and others, 2011; Wagena and others, 2018; Costa and others, 2023). The expected increase in drought frequency in summer could increase hydraulic residence times - the length of time that water remains in a lake (Nöges and others, 2011; Paerl and Barnard, 2020). Together these processes increase the loading of nutrients to lakes, and the period for which these nutrients are available to the phytoplankton community, facilitating their growth (Nöges and others, 2011). Climate change is already driving phytoplankton blooms, which are projected to increase in future (Ho and others, 2019; Paerl and Barnard, 2020; Woolway and others, 2022; Tigli and others, 2025).

Climate change is altering patterns of nutrient cycling within lake waterbodies (Me and others, 2018). Heating of lake surface layers during spring and summer causes them to become less dense than cooler, deeper waters, and this difference in density prevents them from mixing, a process termed thermal stratification (Thackeray and Mackay, 2023). Within stratified lakes, surface waters remain homogeneously mixed by wind, wave action and heat fluxes, but these influences are constrained below the density difference (Wells and Troy, 2022). As oxygen is used in deeper waters, by processes including respiration, and the absence of mixing prevents oxygen replenishment from surface waters, this threatens bottom dwelling aquatic life and species dependent on oxygen-rich water (Woolway and others, 2021).

Thermal stratification already occurs in many lakes but globally synchronous increases in lake surface temperatures (Woolway and others, 2020) can facilitate earlier and prolonged thermal separation, extending the duration of low or no oxygen conditions at the lakebed (Woolway and others, 2021). Anoxic conditions - those completely lacking oxygen, enable the movement of dissolved nutrients from lakebed sediments into the overlying water column, a process known as internal loading (Me and others, 2018; Farrell and others, 2020; Woolway and others, 2021; Nürnberg, 2025). Positive feedback may occur if phytoplankton use these nutrients for growth, as this increases the biomass available for decomposition after they die (Woolway and others, 2021). As decomposition reduces oxygen concentrations (Paerl, 2014), it can lead to further nutrient release from bed sediments (Woolway and others, 2021). Therefore, climatic control on lake stratification regimes can increase lake nutrient concentrations through internal loading, even if catchment inputs are reduced (Me and others, 2018).

Climate change is expected to impact lake phytoplankton communities directly through increased mean water temperatures (Dory and others, 2024). Although responses will be type specific, warmer waters can accelerate phytoplankton metabolism and growth rates, and subsequently increase lake phytoplankton biomass (Reynolds, 2006; Me and others, 2018; Dory and others, 2024). Lakes may therefore experience an increase in

phytoplankton growth despite consistent or reduced nutrient contributions (Hutchins and others, 2013), a trend also noted for river systems (Jarvie and others, 2025).

Increasing water temperatures may change the composition of lake phytoplankton communities (Dory and others, 2024). As phytoplankton form the base of lake food webs, this stands to influence zooplankton grazing and transfer of biomass and energy to higher trophic levels (Amorim and Moura, 2021). Phytoplankton communities are expected to become dominated by cyanobacteria, which have specialised physiological traits that provide a competitive advantage in warmer, nutrient-rich waters (Dory and others, 2024). These include their ability to grow prolifically at increased water temperatures; sequester and store phosphorus for growth more efficiently than other phytoplankton types; and control their buoyancy to preferentially migrate to areas with high light and nutrient concentrations (Carey and others, 2012; Huisman and others, 2018). The presence of cyanobacteria indicates cyanotoxin production is possible, but not certain (Ilgwaran and others, 2024).

For many lakes, the impacts of climate change are already evident. For example, comparison of current (2017-2021) and historical (1981-2010) data for the Windermere catchment, England, identified that lake surface waters have warmed by approximately 0.8°C in Windermere, and 0.7°C in Esthwaite Water (Thackeray and Mackay, 2023). This period saw minimum oxygen concentrations reducing at the base of each lake during spring and summer, and small increases in algal biomass for Windermere North Basin and Esthwaite Water, despite reductions in dissolved phosphorus concentrations (Thackeray and Mackay, 2023). Phytoplankton community modelling projects that continued warming of surface waters will further increase annual mean phytoplankton biomass in Windermere, and cause the spring increase in phytoplankton biomass to occur earlier in the year (Elliott and others, 2006), by approximately 2-3 days per 1°C of warming (Elliott, 2012). Warming waters, together with increasing nutrient loads, are expected to reduce the biodiversity of summer phytoplankton blooms as they become dominated by cyanobacteria (Elliott and others, 2006; Elliott, 2012). Thermal stratification is expected to begin earlier and last longer in Windermere (Maberly and McGowan, 2022). Future climate scenarios have been found to exacerbate cyanobacteria blooms for both lakes, even when nutrient loading was held constant (Hutchins and others, 2013).

## 2.4 Nutrient management strategies for lakes

Lake restoration and management strategies often prioritise nutrient reduction. To reduce point source nutrient pollution to waterbodies, regulation of the wastewater industry in England has intensified over recent decades. Examples of this include more stringent discharge permits, tertiary treatment requirements and combined sewer overflow discharge limits (EA, 2019). The Water Industry National Environment Programme (WINEP) details actions taken by water companies to meet the Environment Agency's regulatory standards, with £25 billion of investment since 1989 (DEFRA, 2022). In the Windermere catchment, £75 million has been spent since 2000, with a further £200 million confirmed for 2025-2030 (United Utilities, 2026).

To reduce diffuse nutrient pollution to waterbodies, agricultural catchments aim to balance food security with reducing nutrient inputs to soils (Spiertz, 2010). Examples include precision agriculture, and slowing nutrient transport to freshwaters through mechanisms including riparian buffer strips, constructed wetlands, catch crops and sustainable tillage (Jeppesen and others, 2009; Brownlie and others, 2022). Across England, these practices have been encouraged by statutory mechanisms, including WFD regulations and Farming Rules for Water (DEFRA, 2018), alongside voluntary measures such as agri-environment schemes and the Catchment Sensitive Farming project (EA, 2019).

In-lake measures to reduce nutrient inputs have included artificially increasing flushing rates, oxygenating anoxic deep waters, removing nutrient rich bottom waters or lakebed sediments, chemically treating sediments to reduce phosphorus mobilisation, and altering trophic interactions (Carpenter, 2005; Paerl and Barnard, 2020; Moorhouse and others, 2024a). These approaches have had limited success with many only feasible for small lakes, and it is widely recognised that management strategies will be most effective if nutrient inputs from the catchment are reduced first (Paerl, 2014; Moorhouse and others, 2024a).

Despite multi-sector efforts to reduce nutrient pollution to freshwaters, and a reduction in total phosphorus (TP) concentrations observed in European lakes since the early 1990s (EEA, 2024), lake eutrophication and algal blooms continue to intensify worldwide (Ho and others, 2019). As climate change may be contributing to this discrepancy, it could hinder strategies to improve water quality in eutrophic lakes (Trolle and others, 2011; Me and others, 2018; Brownlie and others, 2022). It is frequently advocated that lake management strategies should consider climate change impacts and tailor more intensive nutrient reductions to specific catchment and lake conditions to effectively protect lake ecosystems in a warming world (Trolle and others, 2011; Moorhouse and others, 2014; Woolway and others, 2020; Costa and others, 2023; Ezzati and others, 2023; Suresh and others, 2023; Dory and others, 2024).

## 3. Approach and methodology

### 3.1 Study site

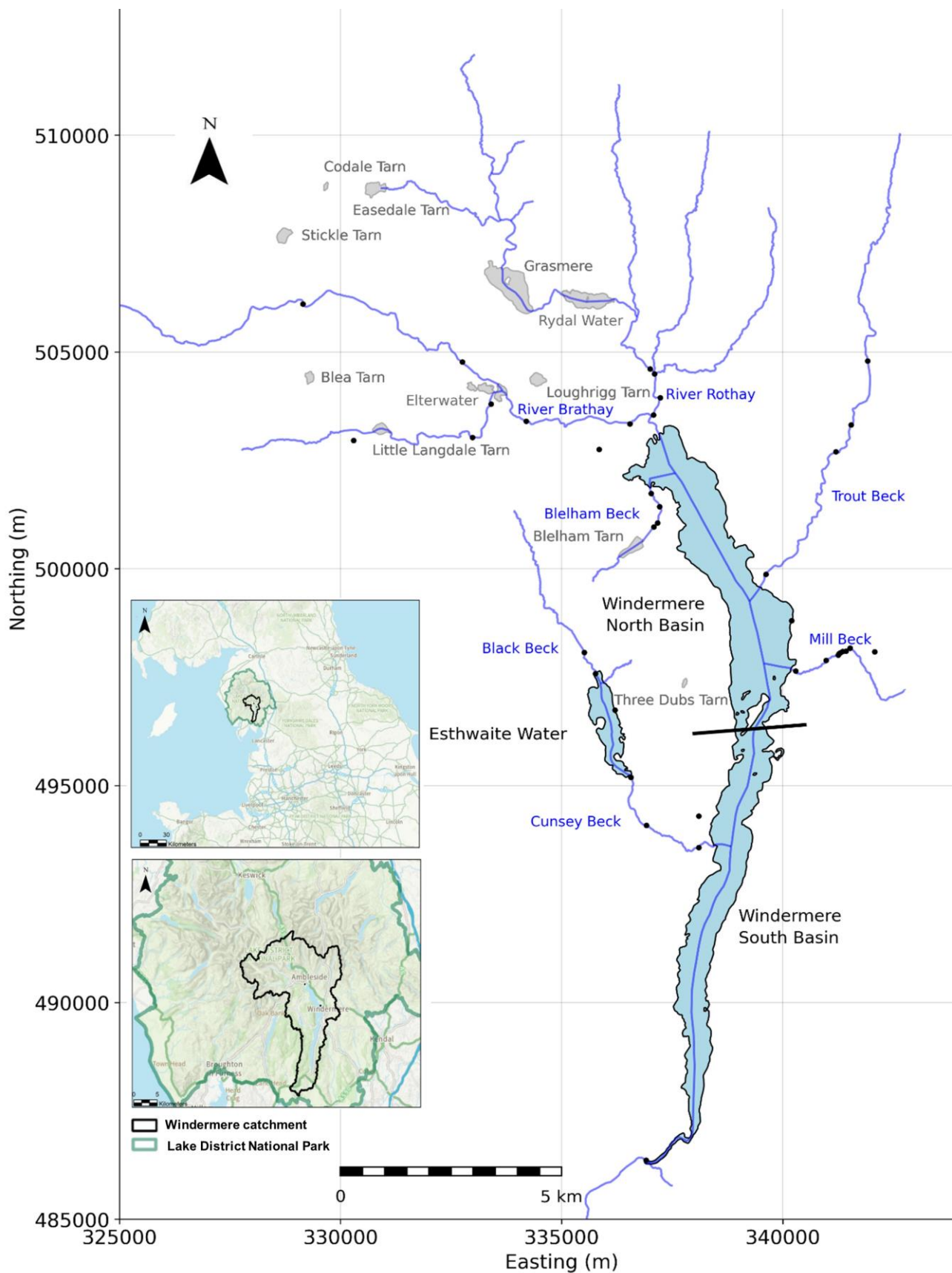
The lakes of the Windermere catchment are some of the best studied in the world, with long-term water quality monitoring datasets originating in the late 1930s (Maberly and Elliott, 2012). Anthropogenic nutrient inputs have influenced these lakes through changes to land management, aquaculture, population density and sewage treatment since approximately 1800–1850 (Maberly and others, 2011; McGowan and others, 2012; Moorhouse and others, 2018). The impacts of climate change, including increasing water temperatures, are already evident (Thackeray and Mackay, 2023). These pressures, together with continued occurrence of algal blooms despite efforts to reduce nutrient inputs, are of concern to the local community as evidenced by recent media and campaign activities. In turn, the Windermere catchment is an important site for testing and tailoring nutrient management.

The Windermere catchment (54°180 N; 2°540 E) lies in the English Lake District, a United Nations Educational Scientific and Cultural Organisation (UNESCO) World Heritage site in Cumbria, in north-west England (Figure 2). Detailed historical and contemporary catchment descriptions are provided by Maberly and Elliott (2012), McGowan and others (2012), Moorhouse and others (2018), Maberly and McGowan (2022), Thackeray and MacKay (2023) and Love Windermere (2025). Within this catchment, this study focused on Windermere and Esthwaite Water (Table 1, Figure 2).

Windermere is the largest lake in England and functions as two distinct basins, North and South, separated by a narrow shallow region (shown as a black line in Figure 2; Moorhouse and others, 2018). It is the most downstream lake and so integrates inputs from the whole catchment (Figure 2). It is arguably subject to the most intensive socio-economic activity of all the Lake District lakes, hosting approximately 7 million visitors annually (Love Windermere, 2025; Tate, 2021). Water quality in Windermere South Basin is intrinsically influenced by Esthwaite Water which enters via Cunsey Beck (Figure 2). As a smaller, shallower, more nutrient-rich system with an aquaculture legacy and greater internal phosphorus loading from bed sediments (Maberly and others, 2011), studying Esthwaite Water provided insight into the impacts of climate change on a contrasting lake system.

**Table 1: Catchment and lake characteristics of Windermere North Basin, Windermere South Basin and Esthwaite Water. Information sourced from McGowan and others (2012) (catchment area), Thackery and MacKay (2023) (trophic status), the PROTECH model (max depth), and Moorhouse and others (2018) (all remaining characteristics).**

<b>Lake characteristic</b>	<b>Windermere North Basin</b>	<b>Windermere South Basin</b>	<b>Esthwaite Water</b>
<b>Catchment area (km<sup>2</sup>)</b>	187	63	17
<b>Altitude (m.a.s.l)</b>	39	39	65
<b>Geology</b>	Silurian flags and slates	Silurian flags and slates	Silurian flags and slates
<b>Max depth (m)</b>	62.0	41.0	15.2
<b>Length (km)</b>	7.0	9.8	2.5
<b>Area (km<sup>2</sup>)</b>	8.1	6.7	1.0
<b>Approximate mean retention time (days)</b>	180	100	100
<b>Trophic status</b>	Mesotrophic	Mesotrophic	Eutrophic



**Figure 2: Map of Windermere and Esthwaite Water. Blue lines represent inflow streams. Individual black dots represent Environment Agency sample points, where water quality measurements are taken. Inset maps show the location of the Windermere catchment within the Lake District National Park and north-west England.**

## 3.2 Model selection

A modelling approach cannot provide an exact representation of future lake structure and function, but can be used as a tool to better understand the trends of future responses to change, and the processes which underpin them (Vinçon-Leite and Casenave, 2019). All projections of future impacts inherently hold uncertainty as they are based upon assumptions informed by current understanding and representation of complex processes (Trolle and others, 2011). They are dependent upon the quality and quantity of data available to support and validate them (Trolle and others, 2011; Costa and others, 2023). Linking multiple models can accumulate the uncertainties associated with each. However, bringing together catchment water quality and phytoplankton community models with climate data, as this study did, can enable design of future nutrient management to inform current decision making (Me and others, 2018; Vinçon-Leite and Casenave, 2019). A brief overview of the models used in this study is provided below.

### 3.2.1 SIMCAT-SAGIS

SIMCAT-SAGIS is a catchment water quality model used by all environmental regulators in the UK, including the Environment Agency, to determine what proportion of nutrients within a catchment are contributed by different sectors (e.g., wastewater treatment infrastructure, arable and livestock agriculture). This source apportionment information can be used to support permitting and planning decisions.

SIMCAT-SAGIS was selected for this study because of its use as a current regulatory tool, meaning recently updated models exist, that are widely applied and tested. The model can be altered to test different nutrient management scenarios, and unlike many water quality models, source apportionment was available at a sectoral and single asset level. In addition, it treated Windermere as two discrete waterbodies, North and South Basin, which is a requirement of the PROTECH model. Further information is provided in Appendix A.

SIMCAT is open-source software which uses a Monte Carlo approach to simulate riverine and lake chemical concentrations and loads. It is a statistical model, in that inputs are simulated using a probability distribution, and this allows for uncertainty to be evaluated. SAGIS is a geographical information system of interconnected tools and databases which builds the input files required to run SIMCAT. The diffuse sector source apportionment is provided by the PSYCHIC model (Phosphorus and Sediment Yield Characterisation In Catchments) for phosphate (Davison and others, 2008), and the Neap-N model for nitrate (Anthony and others, 1996).

Together SIMCAT-SAGIS can determine catchment-scale source apportionment of phosphate and nitrate loads and concentrations in rivers and lakes (Appendix B2.3). It provides water quality conditions within the bounds of uncertainty, and estimates source apportionment, but cannot predict the exact conditions which occurred each day of a given period.

### 3.2.2 PROTECH

PROTECH is a phytoplankton community model that simulates net growth of the phytoplankton community. It models biomass of diatoms, cyanobacteria and other algae, expressed as the concentration of chlorophyll *a*, a pigment used in photosynthesis. Phytoplankton growth is influenced by factors such as temperature, light levels and nutrient availability.

PROTECH was selected for this study as over the past two decades it has been applied successfully to over 30 lakes and reservoirs worldwide (Elliott and others, 2010; Elliott, 2021). It was developed using observational data from Windermere and has previously been calibrated to North Basin, South Basin and Esthwaite Water, furthering its suitability for this study.

PROTECH is a 1-D process-based phytoplankton community model that simulates the development of multiple algal populations and types in lakes and reservoirs. Algae are a broad group of photosynthetic organisms, of which phytoplankton are a subset.

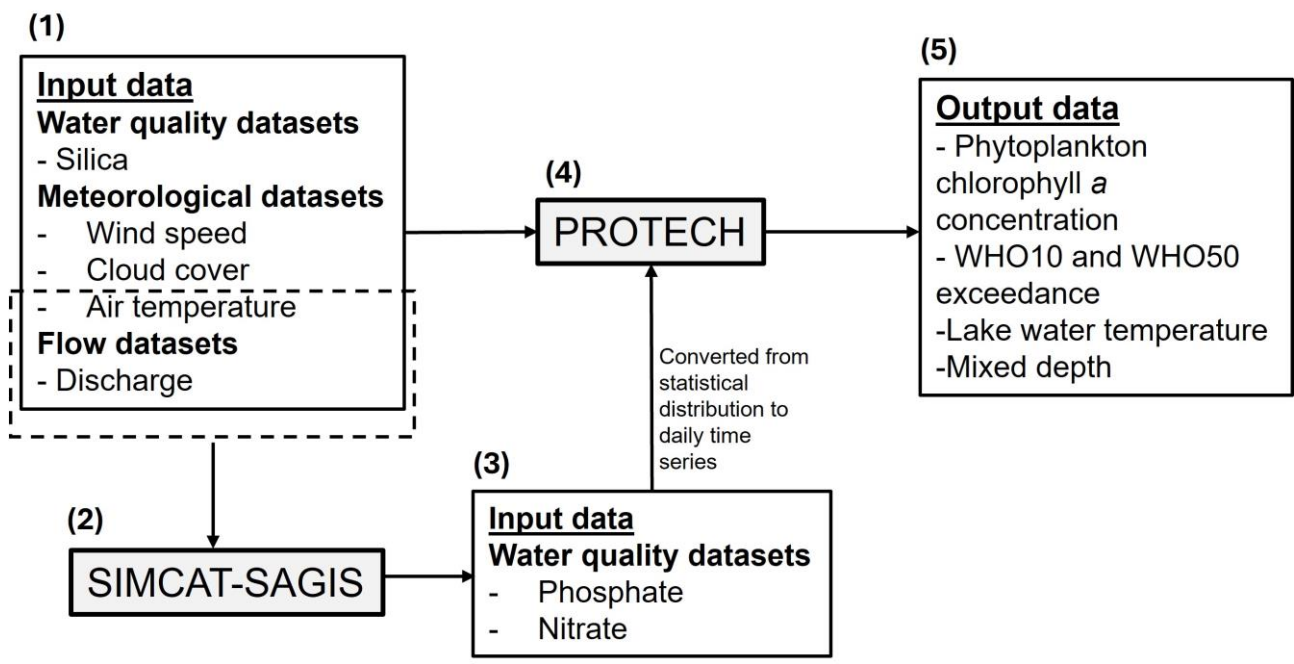
PROTECH models how an environmental simulation will influence the behaviour of algae, with their growth, grazing and movement represented as a series of biological response equations. The model takes air temperature, cloud cover, wind speed at the lake surface, inflow and outflow discharge, and inflow nutrient concentrations (soluble reactive phosphorus (SRP), nitrate corrected for the molecular mass of N, dissolved silica) on daily time steps, and calculates day length, the temperature of the water column, the underwater light regime and mixed depth. In PROTECH, mixed depth is defined as the depth of the homogeneously mixed surface layer within a stratified lake – the distance from the lake surface to the point at which the warmer, less dense surface waters no longer mix with cooler, denser, deeper waters. The impact of these parameters on the algal community is then determined using three equations which relate algal morphology to growth rate, to determine the potential net growth for eight types of algae which are pre-selected from a library of more than 100 types, to represent those most abundant in the study waterbody. Algae present in one time step are carried forward to the next, to capture changes in biomass over time. The model checks there is sufficient phosphorus, nitrogen and silicon available before allowing each alga to increase in biomass and then deducts this from the waterbody resource base.

PROTECH is owned and operated by the UK Centre for Ecology & Hydrology (UKCEH) who conducted all PROTECH modelling for this study.

## 3.3 Scenario development

The SIMCAT-SAGIS and PROTECH models were linked together with climate data to model five scenarios. These scenarios represented the phytoplankton communities of Windermere North Basin, Windermere South Basin and Esthwaite Water in (1) baseline conditions (baseline scenario), (2) future climate change (climate change scenario) and (3)

three nutrient management regimes. The management scenarios represented scheduled and theoretical nutrient reductions, but their feasibility was not considered. The following sections describe these five scenarios and the data collation, transformation and modelling that underpinned them. The modelling workflow is summarised in Figure 3.



**Figure 3: Modelling workflow. Grey boxes represent a catchment water quality model (SIMCAT-SAGIS) and a phytoplankton community model (PROTECH). For definitions of WHO10, WHO50 and mixed depth see Section 4.3.1.3.**

Meteorological, flow and water quality datasets were input directly to the phytoplankton community model (PROTECH), or for some water quality parameters, derived from a catchment water quality model (SIMCAT-SAGIS). SIMCAT-SAGIS was run using the same meteorological and flow datasets as PROTECH. The PROTECH model produced outputs which were used to assess the impact of each scenario on the lake phytoplankton community.

For the baseline scenario, steps 1-5 (Figure 3) were followed. For the climate change scenario steps 1-5 were repeated with input data adjusted to represent the impact of climate change (Figure 3). For the nutrient management scenarios, the SIMCAT-SAGIS model was adjusted to represent alterations to nutrient inputs and steps 2-5 were followed, with step 1 consistent with the climate change scenario (Figure 3). All scenarios assumed wastewater infrastructure would be adapted to ensure that nutrient permit limits were not exceeded despite any future increase in resident or visitor populations.

### Modelling scenario summary

**Baseline:** Response of the phytoplankton community to past climatic conditions, catchment nutrient dynamics and nutrient management.

**Climate change:** Projected the baseline scenario into the far future, 2076-2079, simulating climate change and its impacts on catchment nutrient dynamics, while holding nutrient management constant.

**AMP8:** Includes the climate change scenario, assuming completion of water company improvements to phosphate inputs scheduled in Asset Management Plan 8 (AMP8).

**Low agricultural diffuse:** Includes the climate change scenario, assuming completion of AMP8 and significant reductions in diffuse nutrient inputs from the agricultural sector.

**No sewage:** Includes the climate change scenario, assuming complete cessation of all nutrient inputs from water company and privately owned wastewater treatment infrastructure.

The following sections provide further detail of the modelling underpinning each scenario.

#### 3.3.1 Baseline scenario

The baseline scenario represented the lake phytoplankton communities in baseline conditions. PROTECH was used to simulate the response of the phytoplankton community to environmental conditions for a continuous four-year period from 2015-2018. This period was chosen as it contained meteorological sonde data on the lakes themselves and was a recent period where monitoring data were continuous with few gaps. This modelling required meteorological and hydrological input information, alongside nutrient data supplied by SIMCAT-SAGIS. These nutrient data were obtained using a SIMCAT-SAGIS model that was calibrated using data from 2014-2018 but also captured improvements to nutrient inputs up to and including AMP6 (2015-2020), to more accurately represent recent nutrient reductions (Appendix A1.3). The following sections describe how these input data were collated, transformed and modelled in PROTECH. Detailed information pertaining to each sub-section is provided in Appendix B.

##### 4.3.1.1 Input data collation

PROTECH required meteorological, hydrological and nutrient input data at a daily resolution for 2015-2018 (Figure 3 (1)). Data sources are described in Table 2.

**Table 2: Data sources collated for PROTECH for the baseline scenario. Water quality data pertained to the rivers flowing into each lake.**

	Data	Units	Temporal resolution	Source of information
<b>Meteorological data</b>	Air temperature	°C	Daily	Meteorological sondes on the lake surface collected by UKCEH
	Wind speed	m/s	Daily	Meteorological sondes on the lake surface collected by UKCEH
	Cloud cover	Oktas	Daily	MIDAS weather station sunlight hours
<b>River flow data</b>	Discharge	m <sup>3</sup> /s	Daily	Qube hydrological model
<b>Water quality data</b>	Phosphate (Orthophosphate, reactive as P)	mg/L	Monthly	SIMCAT-SAGIS validated using observed monitoring data
	Nitrate (Nitrate as N)	mg/L	Monthly	SIMCAT-SAGIS validated using observed monitoring data
	Dissolved silica (SiO <sub>2</sub> )	mg/L	Monthly	Observational data

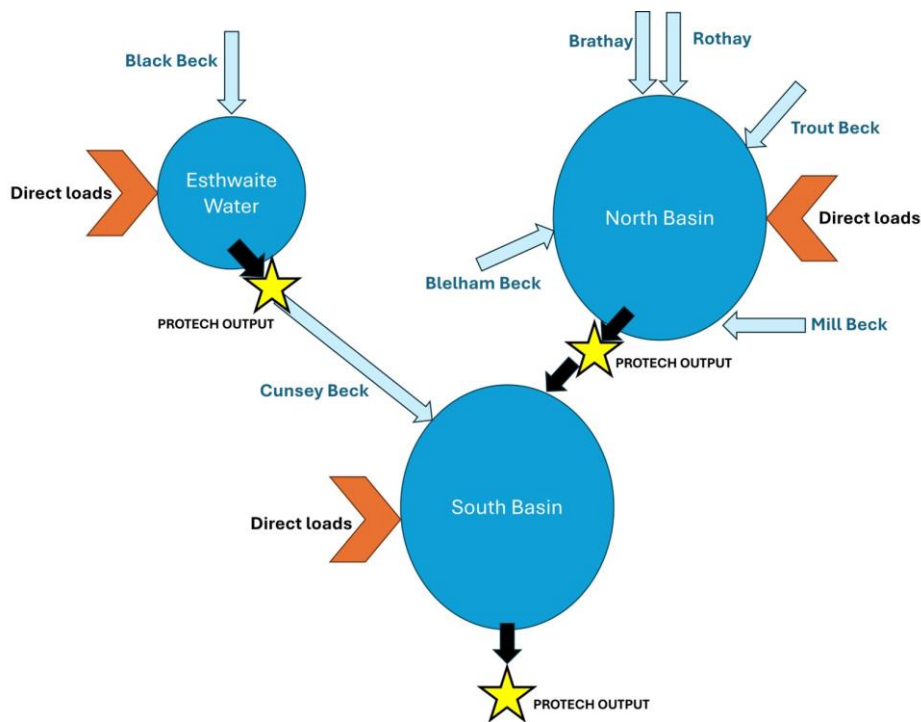
Meteorological inputs to PROTECH were air temperature, wind speed at the lake surface, and cloud cover. Air temperature and wind speed daily time series were obtained from meteorological sondes in Windermere South Basin and Esthwaite Water between 2012-2018 (Feuchtmayr and others, 2021; Jones and others, 2017a, 2017b). A linear regression between the cotemporaneous measurements recorded at the two lakes was used to fill any temporal gaps in the data. Percentage cloud cover was determined by comparing sunlight hours measured at a nearby Met Office Integrated Data Archive System (MIDAS) station (Morecambe Bay No. 2) to the theoretical maximum as predicted by the National Oceanic and Atmospheric Administration (NOAA) calculators (NOAA, 2025). This parameter was converted to oktas using the classification boundaries provided in Boers and others (2010). Further details are provided in Appendix B2.1.

Time-series flow data for the baseline period were extracted from the Qube hydrological modelling platform (Wallingford HydroSolutions, 2025), which provides spatially consistent estimates of river discharge across Great Britain. Qube integrates rainfall-runoff modelling with catchment characteristics to generate daily flow time series for both gauged and ungauged catchments, enabling robust representation of river flows where direct measurements are limited.

Within Qube, river flow estimates are based on outputs from the Continuous Estimation of River Flows (CERF) rainfall-runoff model (Griffiths and others, 2008). Using runoff and soil properties, CERF provided daily time-series data for 11,000 catchments. These were scaled to a flow duration curve from a catchment most analogous to the ungauged catchment of interest. Discharge time series were specified for inflows and outflows for Windermere North Basin, Windermere South Basin and Esthwaite Water. Further details are provided in Appendix B2.2.

The PROTECH model used daily time series of SRP, nitrate and dissolved silica data as nutrient inputs. Existing calibrated SIMCAT-SAGIS catchment model outputs were used to simulate input concentrations of phosphate (see Appendix B2.3) and nitrate for Windermere and Esthwaite Water. As the input for PROTECH was SRP but the form of phosphorus modelled in the river inflows to each lake in the upstream catchment model (SIMCAT-SAGIS) was phosphate, this study assumed the two to be equivalent despite potential for minor differences between them (Appendix B2.3). These outputs accounted for nutrient reductions up to and including AMP6 (2015-2020)(Appendix A1.3). For Windermere North Basin and Windermere South Basin, monthly attribution of load was constrained through inflow river specific Environment Agency monitoring data (Appendix B2.3.2). For Esthwaite Water, SIMCAT-SAGIS predictions were validated with a bespoke study (Anderson, 2018). For dissolved silica, in both lakes, a polynomial regression between the day of the year and the observed dissolved silica concentration was used to create daily predictions of dissolved silica concentration for each input river. Further details are provided in Appendix B2.3.

#### 4.3.1.2 SIMCAT-SAGIS modelling



**SIMCAT-SAGIS was used to provide water quality inputs to three different PROTECH models, one each for Windermere North Basin, Windermere South Basin and Esthwaite Water (Figure 3 (2) & (3)). Below, a description is provided of how SIMCAT-SAGIS represented inputs to each lake, the available observational data, and how this modelling connected to the PROTECH model runs (Figure 4). Figure 4: Perceptual model of nutrient inputs to Windermere North Basin, Windermere South Basin and Esthwaite Water. Light blue arrows indicate a modelled riverine source of nutrient from SIMCAT-SAGIS. Black arrows indicate a modelled source of nutrient from PROTECH model outputs (yellow stars). Orange chevrons indicate an integrated input of all other direct loads.**

- Windermere North Basin: Five riverine inputs (River Brathay, River Rothay, Blelham Beck, Trout Beck and Mill Beck) were added alongside the direct loads to Windermere North Basin. Direct loads are defined as a nutrient load entering a lake from unmodelled river reaches, plus load entering directly from the land rather than being transported via a river or stream. The direct load included the Windermere WwTW storm overflow, which is the only overflow to contribute directly to the lake. While Windermere WwTW discharges into Windermere South Basin, the storm overflow occurs at Glebe Road pumping station and discharges into Windermere North Basin. This was accounted for in the model. The output flow and nutrient concentrations from PROTECH (Figure 4, yellow star) were then used as an input to the Windermere South Basin model. There was at least one Environment Agency monitoring point with nutrient data on each of the input rivers and substantial data from the Big Windermere Survey (FBA, 2025).
- Windermere South Basin: There were four inputs to Windermere South Basin. The outputs from the Esthwaite Water and Windermere North Basin PROTECH models were added to any additional load contributed from Cunsey Beck (taken as the load difference between the point of entry to the Windermere South Basin and the exit point from Esthwaite Water) and any direct loads to Windermere South Basin (of which Windermere WwTW was the primary component). Direct loads are defined as a nutrient load entering a lake from unmodelled river reaches, plus load entering directly from the land rather than being transported via a river or stream.
- Esthwaite Water: This lake was modelled using a two-input approach. The riverine load from Black Beck and the direct loads from all other inflows were added to form the total input of nutrients. The output flows and nutrient concentrations from PROTECH (Figure 4, yellow star) were then used as an input to the Windermere South Basin model. Monitoring data collected by the Environment Agency and Big Windermere Survey only existed for the Black Beck input, but a PhD thesis provided one year's worth of monitoring data for each riverine input into Esthwaite Water (Anderson, 2018). Internal loading could not be directly modelled for this lake (but see Section 4.3.1.3).

#### 4.3.1.2.1 Phosphate and nitrate daily time-series data

SIMCAT-SAGIS provided a statistical load and concentration output, but PROTECH required input data as a daily time series. A script was written in Python to convert phosphate and nitrate data from a statistical load to a daily time series (Figure 3 (3)) (Appendix B2.3). The total riverine load to each lake was calculated by multiplying nutrient concentration by discharge at the closest modelled riverine point to the lake perimeter. This was added to any direct loads. If the mean annual load was split evenly across each day, this would inaccurately reflect variations in load. To account for this, the following adjustments were made:

- Observed concentration and modelled discharge measurements were used to determine the variation in load for each month throughout the year, for each input river and direct load.

- The intermittent component of load was then separated from the total load. Based on storm overflow spill frequency and load data supplied by United Utilities (UU), the intermittent load was applied only on days where river flow exceeded the percentile corresponding to spill frequency.
- The diffuse component of the load was then scaled for a given day based on the flow conditions. Scaling was based on the relationships between coincident load and discharge data reported by Anderson (2018). The effect of this correction was that more diffuse load was delivered on days where river flow was higher, which loosely corresponded to wetter conditions.

Daily concentration was calculated from the daily apportioned load divided by the daily discharge values obtained from Qube.

These adjustments increased the suitability of SIMCAT-SAGIS for investigating the movement of nutrients across catchments in response to varying climate and flow parameters (Appendix B2.3). While this was not the purpose for which it was designed, the requirement to make such adaptations is a challenge currently facing catchment water quality modelling (Gascuel-Oudoux and others, 2023).

#### 4.3.1.2.2 Dissolved silica data

There was no SIMCAT-SAGIS model for dissolved silica, but this is required to run the PROTECH model. Therefore, input concentrations for the models were determined from observational data recorded in each input stream (Figure 3 (1)). Dissolved silica concentrations sampled between 01/01/1950 and 27/11/2024 at monitoring points on input streams were extracted from the Environment Agency's Water Information Management System (WIMS) database. To establish a daily prediction of dissolved silica concentration throughout the four-year baseline period, a polynomial regression was determined between dissolved silica concentration and day of the year. Each lake was treated differently, with further information provided in Appendix B2.3.3:

- Windermere North Basin: A daily weighted concentration was applied based on the fractional flow input.
- Windermere South Basin: The regression established for Cunsey Beck was applied to all additional discharges coming into the lake.
- Esthwaite Water: The regression established for Black Beck was applied to the entire discharge coming into the lake.

#### 4.3.1.3 PROTECH modelling

PROTECH was used to model the phytoplankton community response to environmental variables in Windermere North Basin, Windermere South Basin and Esthwaite Water (Figure 3 (4) & (5)) (Appendix B2.4). Previous PROTECH modelling of Windermere illustrated that the lake was best modelled as two separate basins, North and South, to reflect their contrasting bathymetry (Table 1; Elliott, 2012). Esthwaite Water was represented as a single waterbody.

Within each lake, eight phytoplankton types were selected to reflect the most dominant types recorded in UKCEH long-term monitoring phytoplankton count data for 2015-2018. For both Windermere basins these were *Asterionella*, *Aulacoseira*, *Cryptomonas*, *Monoraphidium*, *Paulschulzia*, *Dolichospermum*, *Aphanizomenon* and *Planktothrix*; the latter three types are cyanobacteria. For Esthwaite Water these were *Asterionella*, *Chlamydomonas*, *Chlorella*, *Plagioselmis*, *Dolichospermum*, *Aphanizomenon*, *Aphanothece clathrata* and *Planktothrix*; the latter four types are cyanobacteria. The chlorophyll *a* concentration of each of these phytoplankton types was considered a proxy for their biomass. Chlorophyll *a* is a pigment that absorbs light energy for use in photosynthesis.

The number of days when the World Health Organisation (WHO) derived risk thresholds of 10 µg/L and 50 µg/L of cyanobacteria chlorophyll *a* were exceeded were recorded to quantify changes in risk posed by cyanobacteria (Chorus and Bartram, 1999). These thresholds signify low (10 µg/L) and moderate (50 µg/L) probability of adverse effects on human health (Chorus and Bartram, 1999). This report refers to these thresholds as WHO10 and WHO50.

Initial comparison of modelled and observed chlorophyll *a* concentration suggested that Windermere North Basin and Windermere South Basin were adequately simulated. However, for Esthwaite Water, there was a lack of nutrients in the summer period of the model runs. This could have been because the internal nutrient load has been shown to contribute a significant part of the lake's overall nutrient budget, especially in summer (Mackay and others, 2014; Anderson, 2018), but this could not be modelled directly in PROTECH. Therefore, between 1<sup>st</sup> June - 31<sup>st</sup> July, an additional 0.5 µg/L SRP was added to the mixed surface layer in PROTECH. In terms of additional modelled phosphorus load, this equated to approximately 50% more nutrient. For the summer, this was 33% of the TP load, when compared to the external load. Mackay and others (2014) estimated similar levels of internal phosphorus contribution (50% in 2008, 34% in 2009).

PROTECH chlorophyll outputs were validated by comparing modelled mean chlorophyll *a* concentration to observed equivalents for 2015-2018 (Appendix B2.4, Table B4). To create an estimate for observed cyanobacteria chlorophyll *a* concentration, the algal count data were converted to estimate biovolumes, and the total chlorophyll *a* concentration was multiplied by the proportion of the biovolume comprising cyanobacteria. This provided an estimate to compare to the WHO threshold values. Both the modelled and observed total chlorophyll *a* data were positively skewed and so were transformed by log<sub>10</sub> to approximate a normal distribution. Given the large sample sizes of the modelled and observed data for the lakes (modelled *n*= 1461, observed *n*= 99-102), the means were tested using a Z-test and none were found to be significantly different (*P* > 0.05). PROTECH also calculated mixed depth, which is analogous to stratification. Mixed depth is influenced by (and calculated using) wind speed at the lake surface, water density and water temperature. A mixed depth of 0.1 m from the lakebed represented a fully mixed lake, while a mixed depth shallower than this (> 0.1 m from the lakebed) illustrated PROTECH model projections of stratification. Annual stratification duration was defined as

the number of days for which a lake was not fully mixed – the continuous duration for which no mixed depth values of 0.1 m from the lakebed were present.

The PROTECH model was initially run for five-year periods, but the first year of output data were discounted to give the model time to adjust to the initial seeding conditions.

### **3.3.2 Climate change scenario**

The climate change scenario represented the lake phytoplankton community under future climate change. Construction of this scenario was identical to that for the baseline scenario, but input data were altered to simulate climate change projected for 2076-2079 (Figure 3). Climate uplift factors for a known time series of data, the 2015-2018 baseline period, were applied using UK Climate Projections (UKCP18). These projections simulated a high emission scenario (Representative Concentration Pathway RCP8.5). While the uplift-factor approach showed how long-term climatic change will influence baseline conditions, it did not effectively capture changes to extreme events and sequencing changes. The climate change scenario assumed continuation of the nutrient management occurring during the baseline scenario period. The following sections describe how the input data were adjusted to represent climate change. Detailed information pertaining to each sub-section is provided in Appendix C.

#### **3.3.2.1 Input data collation**

The meteorological, hydrological and nutrient input data required at a daily resolution for PROTECH were adjusted to represent projected climatic changes for 2076-2079.

##### **3.3.2.1.1 Meteorological datasets**

Using the JASMIN Jupyter notebook system, the 2.2 km gridded UKCP18 daily mean air temperature projection data were extracted for the catchment upstream of Windermere and Esthwaite Water. The median temperature from all 12 ensembles was used. This study used the delta change approach to bias correct air temperature data. This approach calculated a monthly factor difference between two modelled ten-year periods, one covering the baseline scenario period and one covering the climate change scenario period. This monthly factor was then applied to each date in the time series, assigned to the month (Appendix C3.1.1).

Future wind speed and cloud cover conditions were simulated by quantile mapping the future projections to present observed data (Appendix C3.1.3, C3.1.4). This approach was necessary because climate model projections for the present did not accurately capture the statistical distribution of the observed data, overestimating certain percentiles and underestimating others. By quantile mapping, values at specific percentiles could be uplifted to account for the future modelled changes whilst maintaining the broad characteristics of the baseline data. In contrast to air temperature projections, a single ensemble member was chosen based on how well the ensemble projected baseline conditions. This was required as averaging would remove the high and low-end conditions which can occur in a day, for example 100% cloud cover days (Appendix C3.1.4).

### 3.3.2.1.2 Flow datasets

To construct future discharge datasets, the monthly flow factor changes expected for each model ensemble in the RCP8.5 climate scenario using UKCP18 projections were extracted from the Qube platform. Each baseline time-series datapoint, at daily resolution, was then adjusted with the monthly factor derived from the mean of the ensemble members, providing future “adjusted” time-series discharge data for 2076-2079. The uplift was only applied to the natural portion of the flow; the influenced portion, including Windermere WwTW discharge, was assumed to remain unchanged at each time horizon, hence was added directly to the uplifted natural flow. This approach only adjusted to the mean values of each month and therefore could not be used to project changes to the high and low percentile flow events which are anticipated to occur in climate change scenarios (Kendon and others, 2023; Fowler and others, 2024).

### 3.3.2.1.3 Water quality datasets

There were three ways in which climate change can influence the concentration of nutrients using the SIMCAT-SAGIS approach. All were incorporated to generate future nutrient concentrations that align with the future flow time series.

- River temperature: The mean river temperature change in the SIMCAT model could alter the decay rate of nutrients. This was a factor which can decrease nutrient concentrations during the simulation of a river and was considered a proxy for biological uptake or removal to sediment. The mean river temperature applied was taken from the national river temperature projections generated by the Environment Agency, which were based on the same UKCP18 climate data that underpins the air temperature projections in this study. This effect could alter the load provided to the lake, with higher temperature equalling lower load.
- Diffuse scaling: The monthly variation in flow could alter how the diffuse load was distributed throughout a month. This was due to the “scaling factor” used in the baseline scenario. This was automatically adjusted with the updated flow time series. This alteration did not change the summed load which was provided over a month but could change how much load was supplied each day of a month.
- Intermittent thresholds: Variations in the river flow meant that intermittent discharges triggered more or less frequently in the future. The trigger for a threshold was based on the baseline flow percentile. Climate scenarios could have a greater or lesser number of days which exceed this flow percentile, hence there may have been more or fewer spill events in future. For this study, it was assumed that each spill contributed a uniform amount of nutrient load, but in reality, there will be spills with lesser and greater loads.

The intermittent effect was observed as times of significantly altered load, whereas scaling of the diffuse load was seen as smaller scale adjustments to concentration throughout each month (Appendix C3.1.5).

The dissolved silica concentration time series was assumed to stay the same as the baseline scenario. In the absence of a catchment model and lack of a meaningful

catchment-specific relationship to draw upon, this approach was assumed to be the most conservative.

### **3.3.3 Nutrient management scenario 1: AMP8**

The AMP8 scenario simulated how water company improvements to water quality, up to and including those scheduled in Asset Management Plan 8 (AMP8), will influence the lake phytoplankton communities of Windermere North Basin, Windermere South Basin and Esthwaite Water in future (2076-2079), assuming the climatic changes modelled by the climate change scenario were also in effect.

Construction of this scenario was identical to that of the climate change scenario, but changes to catchment phosphate load anticipated to result from completed AMP8 actions were included in the SIMCAT-SAGIS model (Table 3A). This subsequently altered phosphate inputs to the PROTECH model run for each lake (Figure 3).

AMP8 runs from April 2025–March 2030 and is the period in which water companies are required to implement the environmental obligations noted in the WINEP (EA, 2025b). For the Windermere catchment, UU propose to reduce phosphorus outputs at nine wastewater treatment works (Ambleside, Far Sawrey, Ferry House, Grasmere, Hawkshead, Langdale, Near Sawrey, Outgate, Troutbeck). UU also intend to reduce average annual storm overflow spills to 10 or fewer at six sites (Ambleside storm tanks, Elterwater pumping station storm tanks, Glebe Road combined sewer overflow, Grasmere storm tanks, Hawkshead storm tanks, Near Sawrey storm tanks) (UU, 2026). Appendix D4.1 details how these changes were implemented within SIMCAT-SAGIS.

Nitrate loads in the AMP8 scenario are unchanged from the climate change scenario for Windermere North Basin and Esthwaite Water as AMP8 did not alter catchment derived nitrate inputs (Table 3B). AMP8 scenario nitrate loads in inflows to Windermere South Basin increased from the climate change scenario (Table 3B), as the reductions in phosphate inputs to upstream Windermere North Basin and Esthwaite Water driven by AMP8 reduced phytoplankton growth, and so the demand for nitrate, increasing its availability to downstream Windermere South Basin. PROTECH model outputs for Windermere North Basin were an input to Windermere South Basin (see Section 4.3.1.2)

Predicted changes to nutrient contributions were stated in the WINEP for TP, modelled in SIMCAT-SAGIS as phosphate, and modelled in PROTECH as SRP (Appendix B2.3). Although these forms of phosphorus are not equivalent, they were assumed as such to meet the requirements of each model, with differences thought to be minor.

### **3.3.4 Nutrient management scenario 2: Low agricultural diffuse**

The low agricultural diffuse scenario simulated how theoretical maximum potential reductions in phosphate and nitrate inputs from the agricultural sector will influence the lake phytoplankton communities of Windermere North Basin, Windermere South Basin and Esthwaite Water in future (2076-2079), assuming water quality improvements

modelled in the AMP8 scenario and climatic changes modelled in the climate change scenario were also in effect.

Construction of this scenario was identical to that of the AMP8 scenario, but changes to catchment phosphate and nitrate load anticipated to result from reductions in diffuse agricultural nutrient contributions were included in the SIMCAT-SAGIS model (Table 3A, B). This altered nutrient inputs to the PROTECH model and subsequently the projected impacts on each lake phytoplankton community (Figure 3).

FARM SScale Optimisation of Pollutant Emission Reductions (Farmscoper) simulations conducted in a separate internal Environment Agency study indicated that if all regulatory and voluntary land management measures were fully adopted within the Leven catchment (which includes the Windermere catchment), then TP could be reduced by ~40% and nitrate by ~15%, relative to a 2019 baseline. This modelling did not account for specific changes to land use or livestock numbers across the catchment. To model maximum potential reductions in diffuse nutrient inputs from agriculture, a 40% reduction in phosphate and 15% reduction in nitrate were modelled in SIMCAT-SAGIS, and the resulting nutrient data inputted to PROTECH. Appendix D4.2 provides further detail of this process.

### **3.3.5 Nutrient management scenario 3: No sewage**

The no sewage scenario simulated how removal of all phosphate and nitrate inputs from both water company and privately owned wastewater treatment assets will influence the lake phytoplankton communities of Windermere North Basin, Windermere South Basin and Esthwaite Water in future (2076-2079), assuming the climatic changes modelled in the climate change scenario were in effect.

Construction of this scenario was identical to that of the climate change scenario, but all point source nutrient inputs to each lake were removed, including sewage treatment works discharges, septic tank inputs and intermittent discharges, leaving only diffuse nutrient contributions (Table 3A, B). Appendix D4.3 illustrates how this impact was modelled in SIMCAT-SAGIS and PROTECH (Figure 3).

### **3.3.6 The nutrient management scenarios**

The impact of the three nutrient management scenarios on modelled nutrient loads entering each lake are presented in Table 3A and 3B. Equivalent information for nutrient concentrations is provided in Appendix D4.4 (Table D2A, D2B).

**Table 3A: Annual average phosphate loads of modelled input streams (combined) to Windermere North Basin, Windermere South Basin and Esthwaite Water, in the baseline, climate change and three nutrient management scenarios.**

	<b>Phosphate load of lake inflows (kg/yr)</b>				
	<b>Baseline</b>	<b>Climate change</b>	<b>AMP8</b>	<b>Low agricultural diffuse</b>	<b>No sewage</b>
<b>Windermere North Basin</b>	4364	4359	3780	3382	1382
<b>Windermere South Basin</b>	4139	4034	2727	2063	520
<b>Esthwaite Water</b>	513	549	381	331	189

**Table 3B: Annual average nitrate loads of modelled input streams (combined) to Windermere North Basin, Windermere South Basin and Esthwaite Water, in the baseline, climate change and three nutrient management scenarios.**

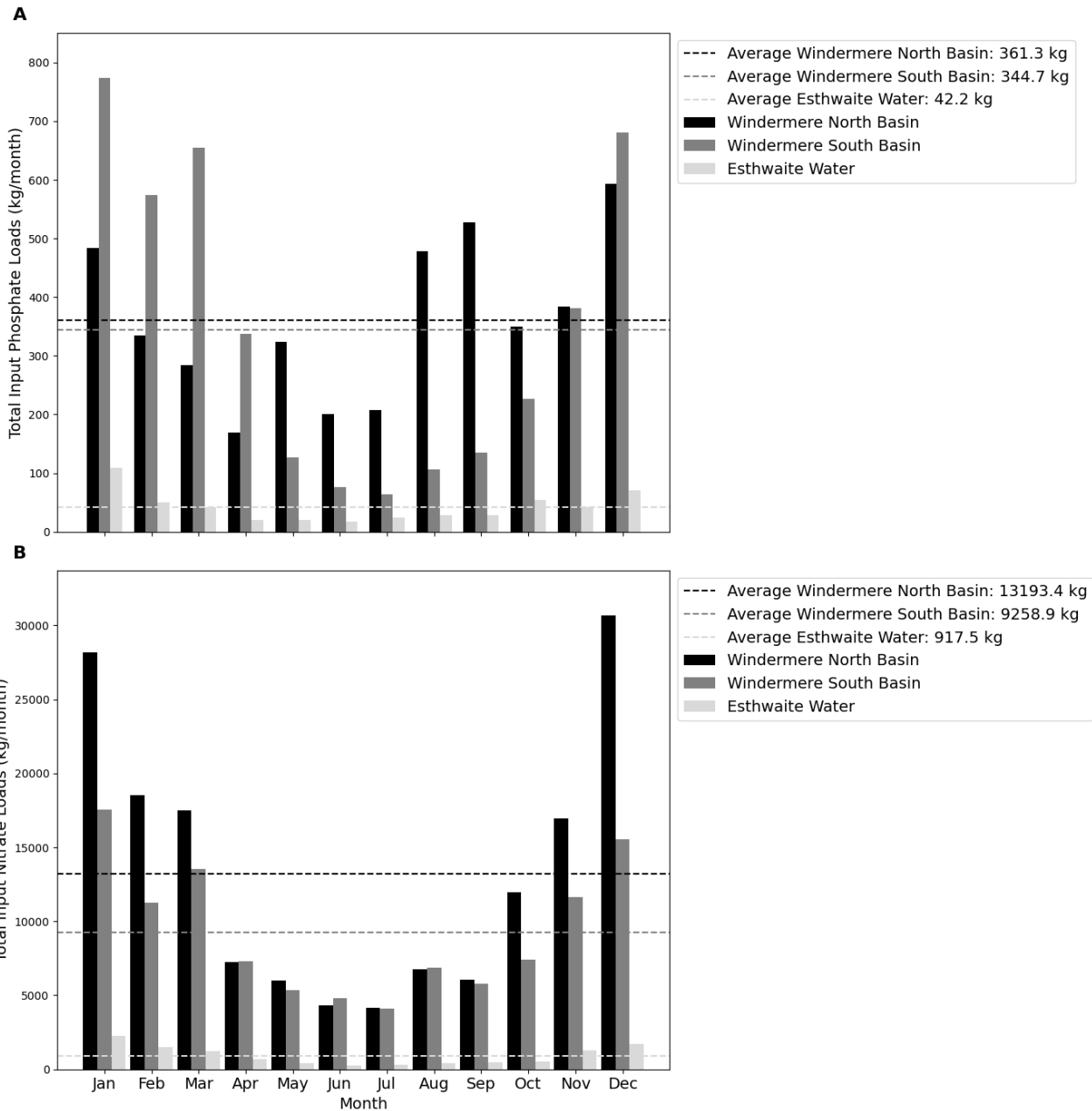
	<b>Nitrate load of lake inflows (kg/yr)</b>				
	<b>Baseline</b>	<b>Climate change</b>	<b>AMP8</b>	<b>Low agricultural diffuse</b>	<b>No sewage</b>
<b>Windermere North Basin</b>	158014	155493	155493	151367	124050
<b>Windermere South Basin</b>	111131	115340	119946	119069	110549
<b>Esthwaite Water</b>	11022	10897	10897	10546	8330

## **4. Results**

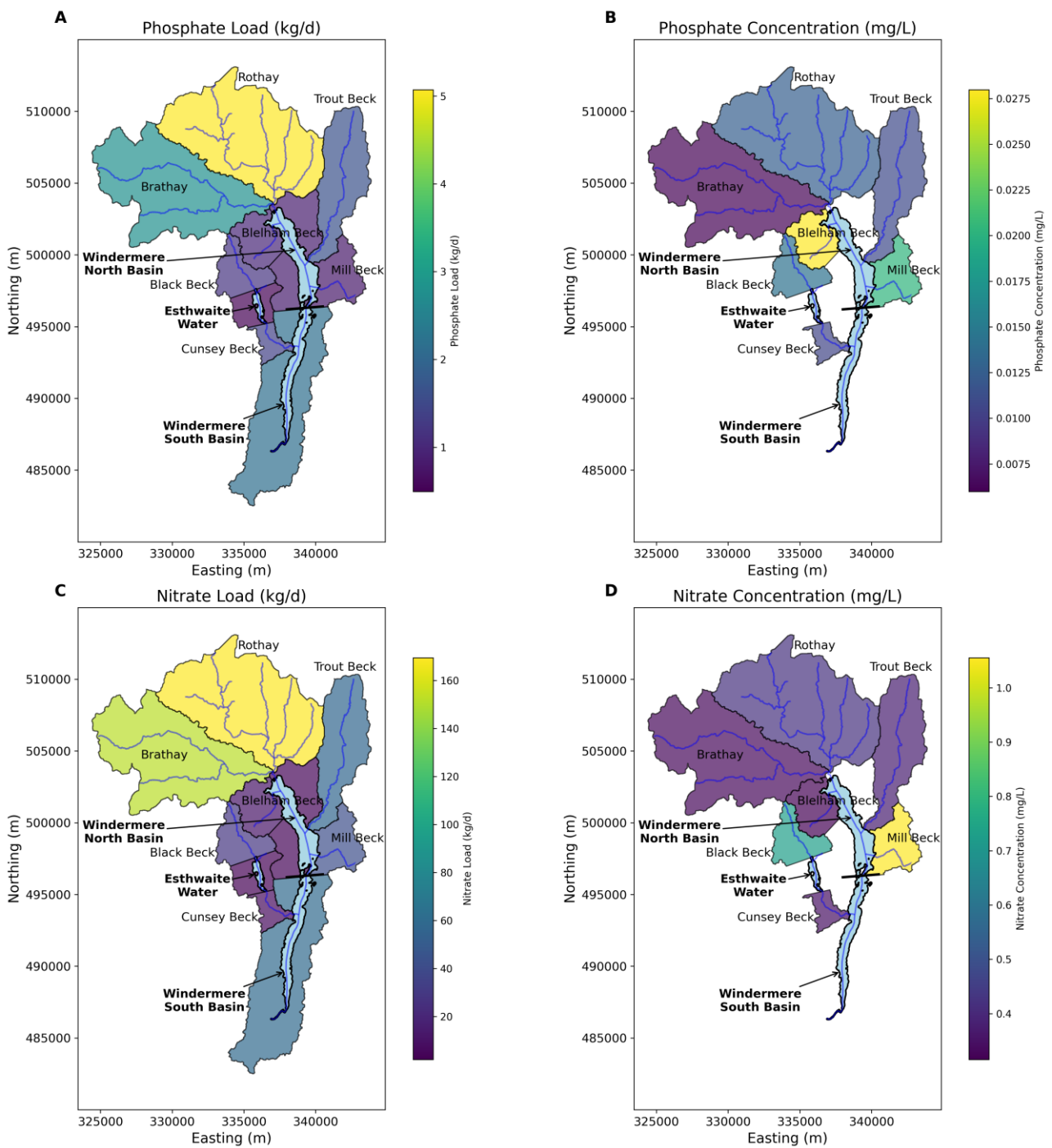
### **4.1 Climate change impacts on nutrient dynamics**

#### **4.1.1 Catchment nutrient dynamics in the baseline scenario**

This section presents the load and concentration of phosphate and nitrate entering Windermere North Basin, Windermere South Basin and Esthwaite Water from their upstream catchments (Figure 5). The contribution of different sub-catchments to the total nutrients delivered to each lake are also illustrated (Figure 6). The source apportionment – the contribution of nutrients from different sources, including diffuse, sewage, septic tanks and intermittent discharges, is then presented to show where the nutrients originated from (Table 4). Nutrient inputs are discussed within the context of those entering the lakes from inflowing rivers or direct loads, rather than nutrients present within each lake.



**Figure 5: Total monthly (A) phosphate and (B) nitrate loads entering Windermere North Basin, Windermere South Basin and Esthwaite Water in the baseline scenario. Dashed lines indicate the average total monthly load for each lake.**



**Figure 6: Nutrient contributions from the lake sub-catchments. Summed annual average nutrient loads (A & C) and concentrations (B & D) for rivers at the point of entry into Windermere North Basin, Windermere South Basin and Esthwaite Water. Annual averages are calculated over the baseline scenario period (2014-2018). There are no concentration measurements for direct loads entering the lakes.**

	Source apportionment (%)				
	Diffuse	Wastewater			
		Total	Sewage	Intermittent discharges	Septic tanks
<b>PHOSPHATE</b>					
Windermere North Basin	37	63	53	8	1
Windermere South Basin (not including inputs from Windermere North Basin and Esthwaite Water)	8	92	85	6	1
Esthwaite Water	62	38	20	15	4
<b>NITRATE</b>					
Windermere North Basin	72	28	22	6	0
Windermere South Basin (not including inputs from Windermere North Basin and Esthwaite Water)	32	68	68	0	0
Esthwaite Water	87	13	12	0	0
<b>TOTAL NUTRIENT LOAD (phosphate + nitrate)</b>					
Windermere (North and South Basins)	37	63	57	5	1
Esthwaite Water	74	26	16	7	2

**Table 4: Source apportionment information. Percentage contributions of each modelled nutrient source to the total phosphate and nitrate load in river inputs and direct loads that drain to each lake. Wastewater total represents the summed contributions of sewage, intermittent discharge and septic tank inputs.**

The presented source apportionment information (Table 4) differs from values published by the Environment Agency for Windermere in 2023 (EA, 2023), for three reasons. First, the studies report source apportionment based upon different forms of phosphorus. The earlier study (EA, 2023) reported apportionment of TP, as this is the form of phosphorus monitored for WFD compliance in lakes across England. This Nutrient Futures for Windermere study modelled phosphate only, which is one component of TP. PROTECH required SRP data for rivers flowing into Windermere, but this is not modelled in SIMCAT-SAGIS so phosphate was used, as this is the closest modelled form to SRP, with difference between the two thought to be minimal. SRP and phosphate are the focus of this study as they are bioavailable forms of phosphorus that can be used by phytoplankton for growth. As a high proportion of some nutrient sources, such as discharges from septic tanks, are often comprised of particulate forms of phosphorus, they will contribute more TP than phosphate to the lakes, explaining in part the difference in values between these reports.

Second, this study used novel modelling of storm tank spills (intermittent flows) in SIMCAT-SAGIS to ascertain more accurate loadings to individual lakes (Appendix A1.3). Third, source apportionment for Windermere South Basin in this study did not include inputs from upstream Windermere North Basin and Esthwaite Water. This was to focus on catchment derived nutrient supply (before this supply was utilised by the lake phytoplankton communities) to address the first objective of this study (page 9). However, these upstream inputs were included in PROTECH modelling of Windermere South Basin.

### **Windermere North Basin**

Windermere North Basin received the highest average monthly loads of phosphate and nitrate of the study lakes (Figure 5). These inputs were higher in winter and lower in summer, although phosphate loads showed greater variability, with additional high values in spring (May) and late summer (August/September) (Figure 5).

The greatest daily loads of phosphate and nitrate were delivered to Windermere North Basin by the Rothay and Brathay rivers (Figure 6). The highest concentrations of phosphate were inputted from Blelham Beck and Mill Beck, and the highest concentrations of nitrate from Mill Beck (Figure 6).

In inflows to Windermere North Basin, phosphate primarily came from wastewater infrastructure, and nitrate from diffuse sources (Table 4). Across all input streams and direct loads, 63% of phosphate and 28% of nitrate were derived from wastewater infrastructure, with 37% phosphate and 72% nitrate from diffuse sources (Table 4).

### **Windermere South Basin**

Inputs of phosphate and nitrate to Windermere South Basin followed a seasonal trend, with loads higher in winter (November-March) and lower in summer (April-October) (Figure 5).

Nutrients entering Windermere South Basin came primarily from wastewater infrastructure, particularly sewage (Table 4). Across all input streams and direct loads, 92% of phosphate

and 68% of nitrate were derived from wastewater infrastructure, with 8% of phosphate and 32% of nitrate from diffuse sources (Table 4).

### **Esthwaite Water**

Esthwaite Water received much lower phosphate and nitrate loading from its surrounding catchment than the Windermere basins (Figure 5), with Esthwaite Water receiving approximately 8x less phosphate, and 12x less nitrate, than Windermere (Figure 5). As for Windermere South Basin, delivery of both nutrients was lowest in summer and highest in winter and expressed less monthly variability than Windermere North Basin (Figure 5).

Modelled catchment nutrients were delivered to Esthwaite Water from Black Beck and direct loads. Black Beck provided comparatively higher loads and concentrations of phosphate and nitrate to the lake (Figure 6).

For Esthwaite Water, modelled inflowing nutrients primarily originated from diffuse sources (Table 4). Across all input streams and direct loads, 38% of phosphate and 13% of nitrate were modelled to originate from wastewater infrastructure, with 62% of phosphate and 87% of nitrate from diffuse sources (Table 4).

### **The study lakes**

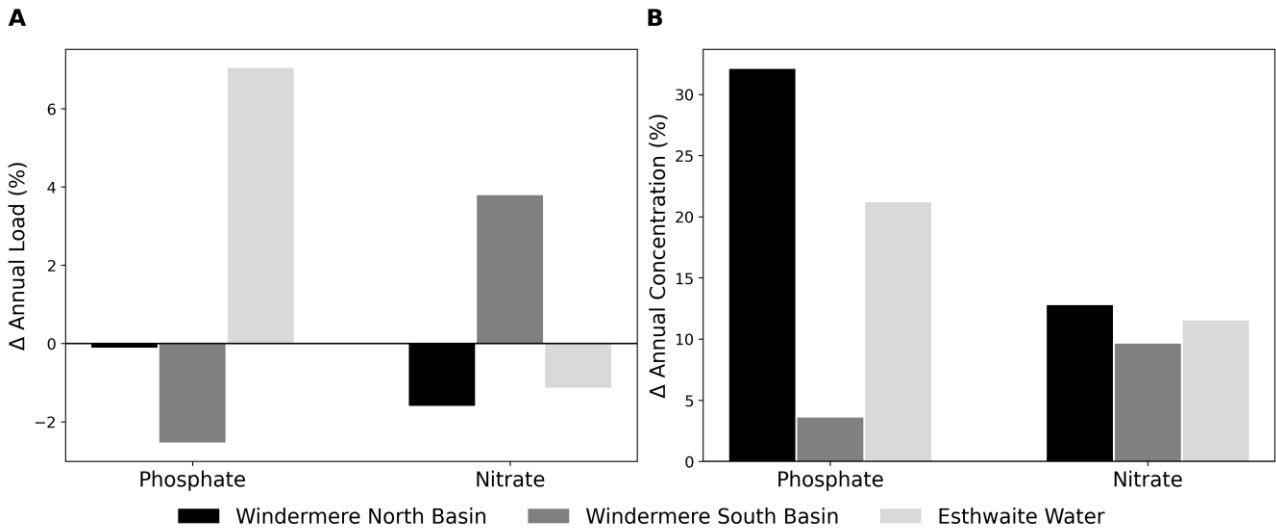
Modelling indicated that for Windermere, nutrients were derived from wastewater infrastructure (63%) and diffuse sources (37%). For Esthwaite Water, nutrient inputs were dominated by contributions from diffuse sources (diffuse = 74%, wastewater = 26%).

Across Windermere North Basin, Windermere South Basin and Esthwaite Water, combined phosphate and nitrate inputs to modelled inflows from wastewater infrastructure were dominated by contributions from sewage treatment works (Windermere = 57%, Esthwaite Water = 16%). Intermittent discharges (Windermere = 5%, Esthwaite Water = 7%) and septic tanks (Windermere = 1%, Esthwaite Water = 2%) provided smaller proportions of the total combined nutrient input to Windermere and Esthwaite Water (Table 4).

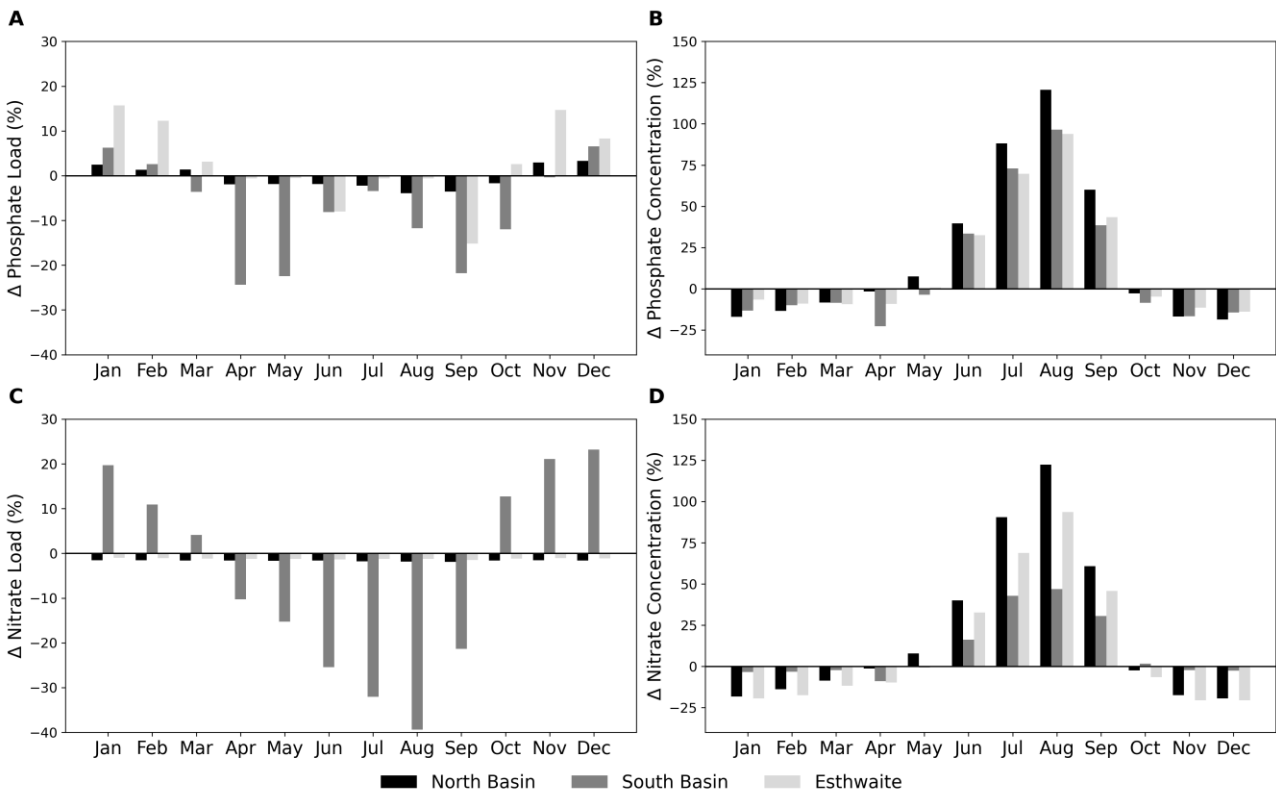
For both lakes, total inflow nitrate loads were consistently greater than phosphate loads (Windermere North Basin = 36x greater, Windermere South Basin = 27x greater, Esthwaite Water = 21x greater).

#### **4.1.2 Catchment nutrient dynamics in the climate change scenario**

To quantify the impact of climate change on the nutrient dynamics of the Windermere catchment, the load, concentration and timing of phosphate and nitrate inputs to Windermere North Basin, Windermere South Basin and Esthwaite Water in the climate change scenario (2076-2079) were compared to those of the baseline scenario, as illustrated in Figure 7 and Figure 8.



**Figure 7: Climate change impacts on annual lake nutrient inputs. Change between the climate change (2076-2079) and baseline scenarios, expressed as a percentage of the baseline scenario for phosphate and nitrate loads (A) and concentrations (B).**



**Figure 8: Climate change impacts on monthly lake nutrient inputs. Change between the climate change (2076-2079) and baseline scenarios expressed as a percentage of the baseline scenario, for phosphate and nitrate loads (A, C) and concentrations (B, D).**

### Windermere North Basin

Modelled outputs for the climate change scenario showed minor decreases in the annual load of phosphate and nitrate entering Windermere North Basin, compared to the baseline scenario (Figure 7A).

The climate change scenario slightly altered the timing of the delivery of phosphate load (Figure 8A), with monthly loads projected to increase in winter (November-March) and reduce in spring/summer/autumn (April-October), relative to baseline scenario values (Figure 8A). Monthly inputs of nitrate load to Windermere North Basin showed less variability, with minor reductions (~ 1%) in all months (Figure 8C).

Annual concentrations of both phosphate and nitrate were projected to increase in the climate change scenario, by 33.3% and 12.6% respectively, relative to the baseline scenario (Figure 7B). These were the greatest increases in phosphate concentrations projected across this study (Figure 7B).

Nutrient concentrations followed a seasonal pattern for both phosphate and nitrate, with increases projected for summer (May-September) and smaller decreases projected from autumn to spring (October-April), relative to the baseline scenario (Figure 8B, D).

### **Windermere South Basin**

For inflows to Windermere South Basin, the climate change scenario resulted in a small reduction to the annual nutrient loading of phosphate (-2.5%), and an increase of 3.8% in the loading of nitrate, in comparison to the baseline scenario (Figure 7A).

Phosphate and nitrate loading was projected to increase in winter (November-February) and decrease in spring to autumn (March-October), relative to the baseline scenario (Figure 8A, C). For phosphate loading, this seasonal trend mirrored that of Windermere North Basin, although Windermere South Basin was projected to experience greater reductions in nutrient loading from spring to autumn (Figure 8A). For nitrate loading, this seasonal trend was observed only for Windermere South Basin (Figure 8C).

Annual concentrations of phosphate and nitrate were projected to increase in the climate change scenario relative to the baseline scenario (Figure 7B), a trend also identified for Windermere North Basin and Esthwaite Water.

Phosphate and nitrate concentrations were projected to follow a seasonal trend when compared to baseline values, with decreases in autumn to spring and larger increases in summer, which were also observed in Windermere North Basin and Esthwaite Water (Figure 8B, D). This was the opposite trend to that displayed for phosphate load in all three lakes, and for nitrate load in Windermere South Basin (Figure 8). The models suggest a general trend of decreasing nutrient load but increasing nutrient concentration in summer, and increasing nutrient load but decreasing nutrient concentration, in winter (Figure 8).

### **Esthwaite Water**

The climate change scenario led to small reductions in annual nitrate loading to Esthwaite Water when compared to baseline values, as identified for Windermere North Basin (Figure 7A). Annual phosphate load increased 7.0% between scenarios, the greatest increase observed across the study lakes (Figure 7A).

The increased phosphate load projected between baseline and climate change scenarios was delivered to Esthwaite Water in winter (Figure 8A). There were minimal changes to the monthly distribution of nitrate loading to Esthwaite Water, as also projected for Windermere North Basin (Figure 8C).

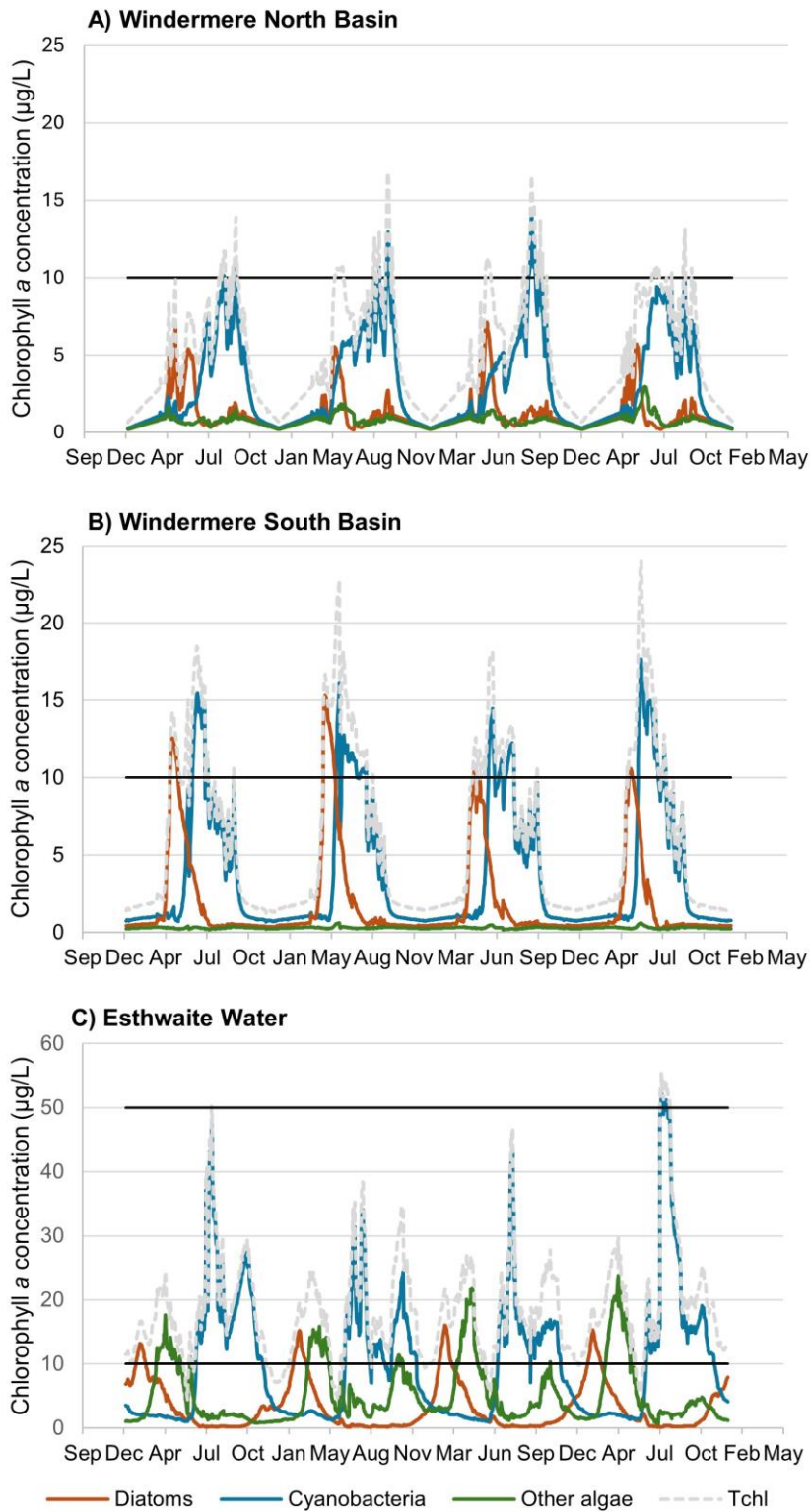
Annual concentrations of both phosphate and nitrate were projected to increase in the climate change scenario (Figure 7B), by 22.2% and 11.3% respectively, relative to the baseline scenario.

For Esthwaite Water, the concentration of monthly nutrient inputs increased in summer, and decreased slightly in winter (Figure 8B, D), relative to baseline values. This pattern aligned to changes in the monthly distribution of nutrient inputs projected to characterise the Windermere basins in the climate change scenario (Figure 8B, D).

## **4.2 Climate change impacts on lake phytoplankton**

### **4.2.1 Lake phytoplankton in the baseline scenario**

In the baseline scenario, the phytoplankton of Windermere North Basin, Windermere South Basin and Esthwaite Water exhibited seasonal patterns in their biomass, measured as changes in chlorophyll *a* concentration (Figure 9). The modelled phytoplankton assemblages were comprised of diatoms, cyanobacteria and other algae (Appendix E5.1, Table E1). Changes in biomass of cyanobacteria influenced the number of days for which the WHO10 and WHO50 thresholds were exceeded (Figure 9).



**Figure 9: Seasonal patterns in lake phytoplankton biomass. Mean chlorophyll a concentration of diatoms, cyanobacteria and other algae in the baseline scenario. Black lines denote WHO10 and WHO50 thresholds.**

#### **4.2.1.1 Windermere North Basin**

Total chlorophyll *a* remained low in winter months, before a diatom dominated peak in spring (April-June), which was followed by a cyanobacteria dominated peak in summer (June-October) (Figure 9A). Other algae had lower biomass than the other phytoplankton groups, which increased in summer and decreased in winter (Figure 9A).

Annual mean total cyanobacteria chlorophyll *a* exceeded the WHO10 threshold on an annual average of 8 days (max: 18 days, min: 0 days), in August/September (Figure 9A). These exceedances were often consecutive, with one event in 2017 lasting 16 days.

#### **4.2.1.2 Windermere South Basin**

The seasonal pattern of phytoplankton biomass identified for Windermere North Basin, was also observed in Windermere South Basin (Figure 9B). However, annual concentrations of total chlorophyll *a* were 17% higher in Windermere South Basin, with peak concentrations occurring in May/June, approximately three months earlier than in Windermere North Basin (Figure 9B).

The annual average number of days for which the WHO10 threshold was exceeded was 50 days (max: 64 days, min: 34 days), from May-August, with a single event in May/June of 2018 lasting 49 days (Figure 9B).

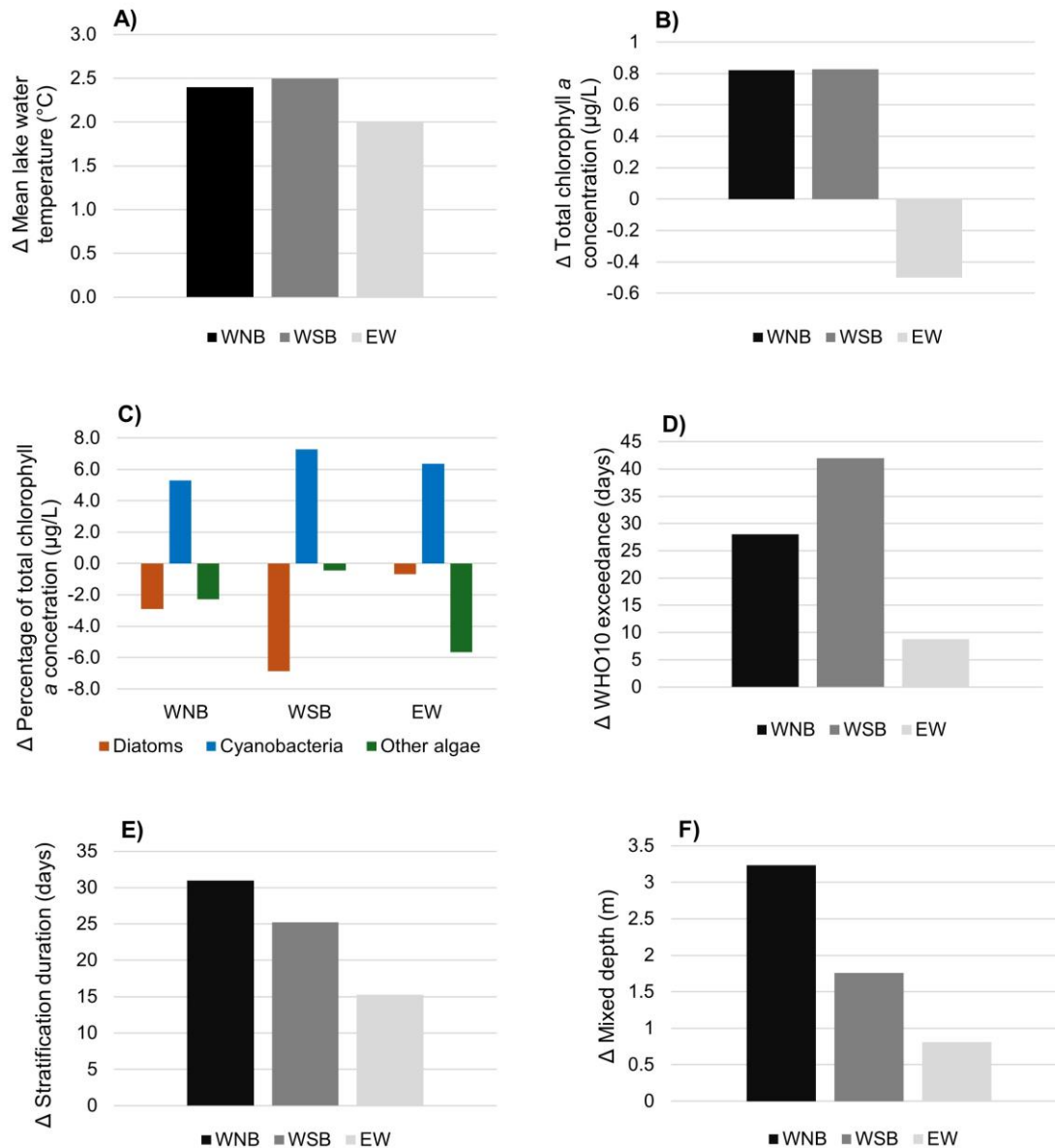
#### **4.2.1.3 Esthwaite Water**

Esthwaite Water experienced an increase in chlorophyll *a* concentration attributed to other algae, occurring from March-May (Figure 9C). The mean total chlorophyll *a* concentration in Esthwaite Water was higher than for both Windermere North Basin (3.1x higher) and Windermere South Basin (2.7x higher) (Figure 9). Phytoplankton biomass remained high for an extended period of the year, with the diatom dominated peak in chlorophyll *a* concentration occurring earlier than for the Windermere basins (February) and the cyanobacteria dominated peak lasting approximately two months longer (November/December) (Figure 9C).

The WHO10 threshold was exceeded on an annual average of 159 days (max: 166 days, min: 152 days) from June-December, a greater number of exceedances than for both Windermere basins. Esthwaite Water was the only lake to exceed the WHO50 threshold (annual average of 3 days (max: 12 days, min: 0 days), July/August) (Figure 9C).

## 4.2.2 Lake phytoplankton in the climate change scenario

To quantify the impact of climate change on the lake phytoplankton communities, outputs from the climate change scenario (2076-2079) were compared to those of the baseline scenario. For each lake, the baseline seasonal patterns of phytoplankton biomass were maintained (Appendix E5.1, Figure E1), but there was variation in chlorophyll *a* concentration, WHO10 exceedances, and the duration and depth of thermal stratification (Figure 10).



**Figure 10: Climate change impacts on lake characteristics. Change between the climate change (2076-2079) and baseline scenarios. WNB = Windermere North Basin, WSB = Windermere South Basin and EW = Esthwaite Water.**

#### **4.2.2.1 Windermere North Basin**

Mean lake water temperature increased 2.4°C between the baseline and climate change scenarios (Figure 10A). Total chlorophyll *a* concentration in the Windermere North Basin increased by 16% between scenario periods (Figure 10B), with this increase in phytoplankton biomass primarily caused by a greater increase in cyanobacteria in comparison to other phytoplankton groups (Figure 10C), across the summer months. Paired t-tests identified that the modelled increase in monthly mean chlorophyll *a* concentration between the scenarios was significant for both cyanobacteria and diatoms (Appendix E, Table E2).

Between the baseline and climate change scenarios, there was a 4.5x increase in the annual average number of WHO10 exceedances (36 days, max: 57 days, min: 20 days), which occurred later in the summer (August-October) (Figure 10D, Figure 11, Table 5).

Lake stratification began on average 19 days earlier and lasted 31 days longer, while the average mixed depth was 3.3 m shallower (Figure 10E, F; Figure 12A) when compared to the baseline scenario.

#### **4.2.2.2 Windermere South Basin**

Between the baseline and climate change scenario periods, mean lake water temperature increased 2.5°C (Figure 10A) in Windermere South Basin. There was also a 14% increase in total chlorophyll *a* concentration relative to the baseline scenario (Figure 10B). The chlorophyll *a* concentration attributed to cyanobacteria increased 27% between scenarios across both the spring (May) and summer (August-September) peak biomass events, with this group contributing over 90% of total chlorophyll *a* during the summer. Differences between monthly mean chlorophyll *a* concentration for baseline and climate change scenarios were significant for all three phytoplankton groups (Appendix E, Table E2).

For Windermere South Basin, there was a 54% increase in WHO10 annual average exceedances (92 days, max: 125 days, min: 89 days), which occurred during an additional month of the year, from May-September (Figure 10D; Figure 11, Table 5) compared to the baseline scenario.

Between scenario periods, lake stratification lasted on average 25 days longer (Figure 10E; Figure 12B). The average mixed depth became shallower, as for Windermere North Basin (Figure 10F; Figure 12B).

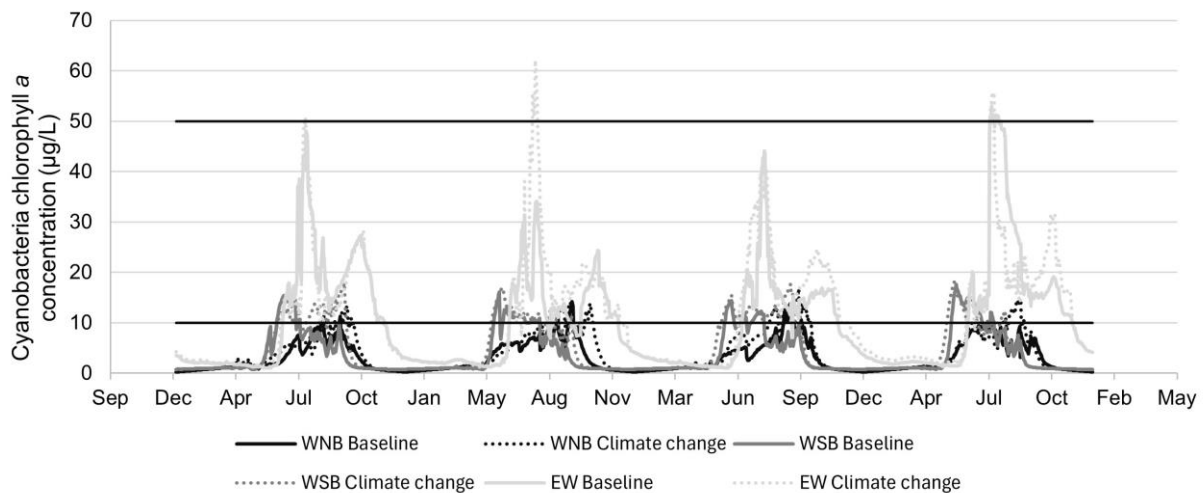
#### **4.2.2.3 Esthwaite Water**

Esthwaite Water's mean lake water temperature increased by 2.0°C between baseline and climate change scenarios (Figure 10A). In contrast to the two Windermere basins, a small decrease (-2.5%) in total chlorophyll *a* concentration was projected for Esthwaite Water between the two scenarios (Figure 10B). This resulted from significant reductions in the monthly mean biomass of other algae (-22.5%) and smaller reductions in diatom biomass (-6.3%) (Figure 10C). In contrast, the summer increase in cyanobacteria biomass identified in the two Windermere basins was also present in Esthwaite Water (Figure 10C), and this

group contributed over 90% of average total chlorophyll a concentration in July, August and September (Figure 11).

The projected increase in cyanobacteria biomass underpinned an annual average increase of 9 additional WHO10 (June-December), and 1 additional WHO50 (July) exceedances (Figure 10D; Figure 11, Table 5) between scenarios. Cyanobacteria chlorophyll a concentration remained above these thresholds in an additional two months of the year (November/December) compared to the Windermere basins (Figure 11).

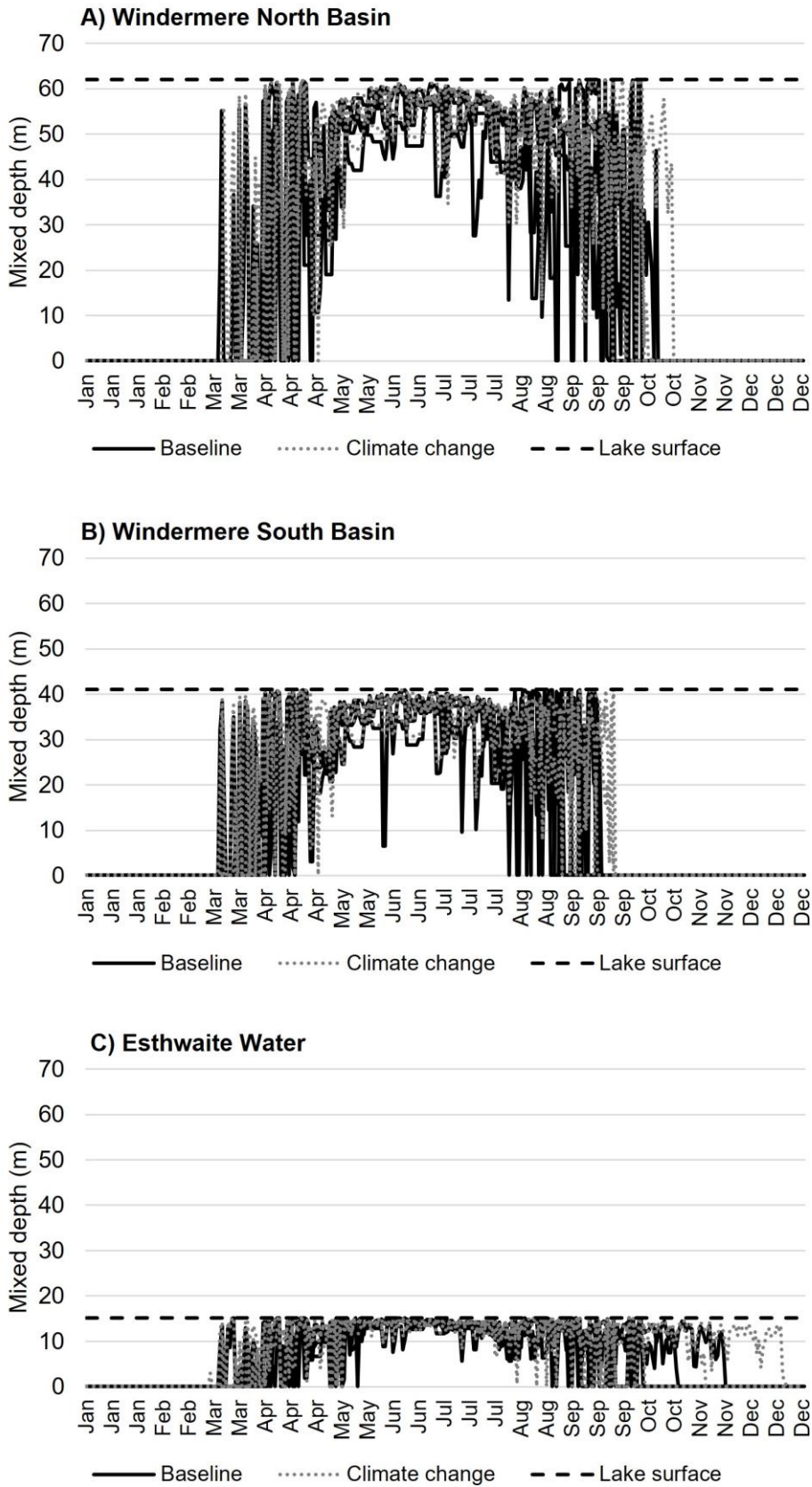
Lake stratification began on average 14 days earlier and the average stratification duration increased by 15 days between scenarios (Figure 12C). Stratification was projected to extend into November/December in some modelled years, in contrast to the Windermere basins where mixing resumed in September/October (Figure 10E; Figure 12C). The average mixed depth became slightly shallower (0.8 m) (Figure 10F; Figure 12C).



**Figure 11: Climate change scenario impacts on WHO10 and WHO50 exceedance. Black lines denote the WHO10 and WHO50 thresholds. WNB = Windermere North Basin, WSB = Windermere South Basin and EW = Esthwaite Water.**

**Table 5: WHO10 exceedance. Annual average number of days the WHO10 threshold was exceeded in the baseline and climate change scenarios for Windermere North Basin, Windermere South Basin and Esthwaite Water.**

Scenario	Windermere North Basin	Windermere South Basin	Esthwaite Water
Baseline	8	50	159
Climate change	36	92	167
Difference	+28	+42	+8



**Figure 12: Climate change impacts on mixed depth and lake stratification duration. The distance between the lake surface and each data line represents the zone of mixing.**

## 4.3 Nutrient management impacts on phytoplankton

This study modelled three nutrient management scenarios. All reduced nutrient inputs to Windermere North Basin, Windermere South Basin and Esthwaite Water by differing amounts. This altered seasonal patterns of phytoplankton biomass, chlorophyll *a* concentration and WHO10 exceedances in each lake (Figure 13). Further details are provided in Appendix E5.2 (Table E3, Figure E2).

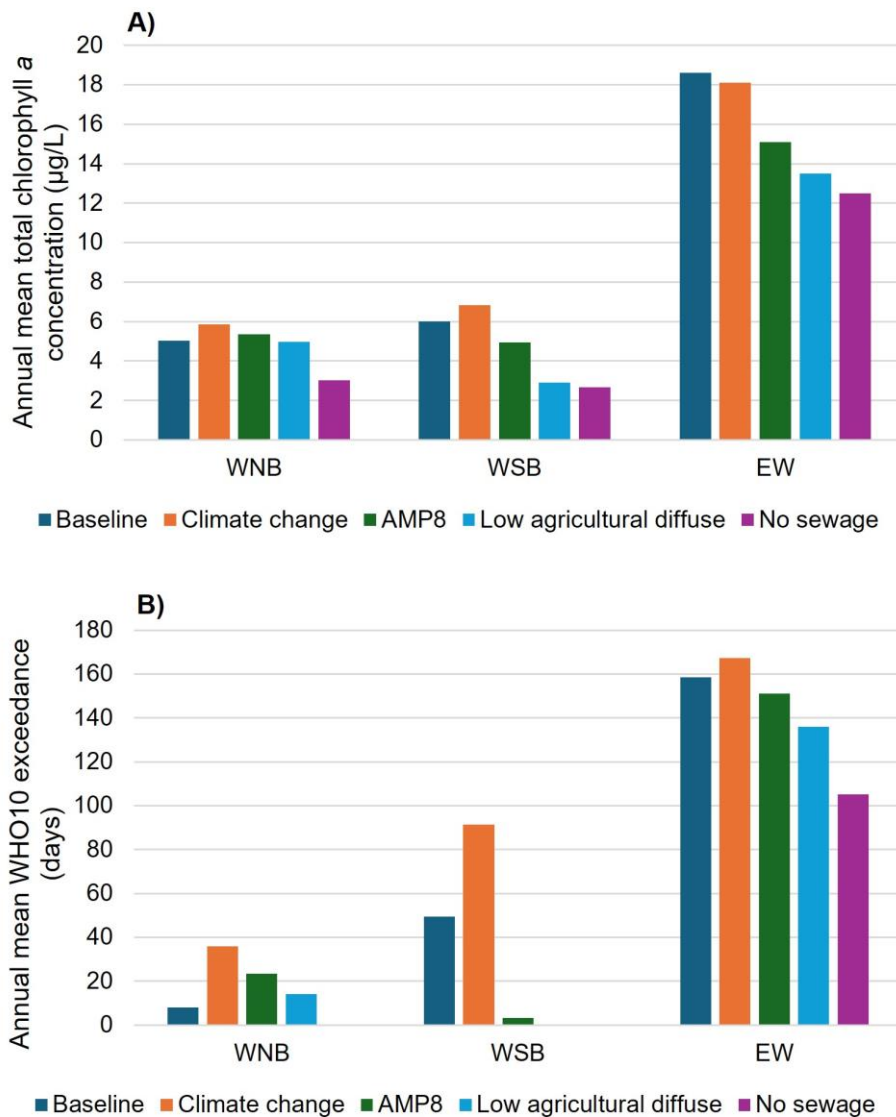
The impact of each nutrient management scenario on phytoplankton biomass and WHO10 exceedance was compared to the modelled outputs from the climate change scenario. This was to indicate the extent to which nutrient management could compensate for the projected increase in phytoplankton biomass and WHO10 exceedance caused by climate change. All three nutrient management scenarios included modelled future climate change. The climate change scenario assumed continuation of the nutrient management in place during the baseline scenario period.

### 4.3.1 Nutrient management scenario 1: AMP8

This nutrient management scenario modelled the reductions in phosphate that were expected to result from water company improvements up to and including AMP8. This scenario reduced the average total chlorophyll *a* concentration in each lake (Figure 13A). For Windermere North Basin, these reductions were less than the increase in this parameter observed in the climate change scenario (Figure 13A). WHO10 exceedances (annual average: 24 days, max: 45 days, min: 4 days) remained 2.9x higher than baseline scenario values (Figure 13B).

In contrast, for Windermere South Basin the AMP8 scenario reduced both average total chlorophyll *a* concentration and annual average WHO10 exceedances below the values of both the climate change and baseline scenarios (Figure 13A, B). WHO10 exceedances (annual average: 3 days, max: 13 days, min: 0 days) were decreased by 93% compared to the baseline scenario (Figure 13B). Windermere South Basin also had the most notable changes to its seasonal patterns of phytoplankton biomass, with cyanobacteria chlorophyll *a* concentration reduced by 49% compared to the climate change scenario, diminishing its summer biomass peak (Appendix E5.2, Figure E2).

For Esthwaite Water, the AMP8 scenario reduced average total chlorophyll *a* concentration by 72%, relative to the climate change scenario. The WHO10 threshold was surpassed on an annual average of 151 days (max: 160 days, min: 143 days) but the WHO50 threshold was no longer exceeded (Figure 13A, B).



**Figure 13: Nutrient management scenario impacts on lake parameters. WNB = Windermere North Basin, WSB = Windermere South Basin, EW = Esthwaite Water.**

### 4.3.2 Nutrient management scenario 2: Low agricultural diffuse

The low agricultural diffuse scenario reduced phosphate by 40% and nitrate by 15% to model the theoretical maximum reductions in diffuse pollution from agriculture for the Windermere catchment. For both Windermere basins, this scenario was sufficient to reduce average total chlorophyll a concentration below both climate change and baseline scenario values (Figure 13A). For Windermere North Basin, despite total phytoplankton chlorophyll a concentration approximating baseline scenario values, annual average WHO10 exceedances (annual average: 14 days, max: 25 days, min: 0 days) remained 57% higher than in baseline conditions (Figure 13A, B).

For Windermere South Basin, this management scenario resulted in no exceedance of the WHO10 threshold (Figure 13B). The summer cyanobacteria biomass peak was effectively removed in this low agricultural diffuse input scenario, with low chlorophyll a concentration between spring diatom (*Asterionella*) dominated peaks (Appendix E5.2, Figure E2).

Esthwaite Water, as for the AMP8 scenario, was projected to experience greater reductions in average total chlorophyll *a* concentration than Windermere North Basin and Windermere South Basin, yet annual average WHO10 exceedances (annual average: 136 days, max: 152 days, min: 114 days) remained higher than for Windermere, despite being reduced by 31 days relative to the climate change scenario (Figure 13A, B).

### **4.3.3 Nutrient management scenario 3: No sewage**

The no sewage scenario modelled how lake phytoplankton communities would respond to complete cessation of phosphate and nitrate inputs from wastewater infrastructure. Average total chlorophyll *a* concentration was the lowest reported of all scenarios for each lake, with values reduced below the concentrations projected for both the climate change and baseline scenarios (Figure 13A). Relative to the climate change scenario, the no sewage scenario reduced total chlorophyll *a* concentration by 48% in Windermere North Basin, 61% in Windermere South Basin and 69% in Esthwaite Water (Figure 13A). The WHO10 threshold was no longer exceeded in either Windermere basin (Figure 13B). For Esthwaite Water, this threshold was exceeded on an annual average of 105 days (max: 141 days, min: 69 days), the lowest number recorded in all modelled scenarios. The WHO50 threshold was surpassed on 4 days in July 2076.

In the no sewage scenario, the seasonal patterns of phytoplankton biomass identified in the baseline scenario were maintained with reduced cyanobacteria chlorophyll *a* concentrations and absence of a cyanobacteria dominated summer peak for Windermere South Basin, as projected in the low agricultural diffuse scenario (Appendix E5.2, Figure E2). Diatom biomass was reduced to a lesser extent, with a spring diatom biomass peak maintained in all nutrient management scenarios (Appendix E5.2, Figure E2).

### **4.3.4 Water Framework Directive (WFD) class**

PROTECH provided modelled total chlorophyll *a* concentration for Windermere North Basin, Windermere South Basin and Esthwaite Water. Following the chlorophyll component of the PhytoPLankton classification with Uncertainty TOol (PLUTO) assessment method, this metric was used to calculate an Ecology Quality Ratio (EQR) to determine the WFD class of each lake in each scenario, as detailed in Appendix E5.2.2. While the climate change scenario reduced the EQR values in each lake, resulting in a transition from Good to Moderate status in Windermere North Basin, the nutrient management scenarios were projected to drive a trajectory of improvement in EQR for the lakes. The no sewage scenario was sufficient to move Windermere South Basin to High status, albeit close to the class boundary (Table 6). Despite improvement in EQR values across the nutrient management scenarios in Esthwaite Water, its WFD status was not better than Moderate by 2076-2079.

**Table 6: Projected site-specific WFD class for the lakes in each study scenario, based upon chlorophyll a concentration ( $\mu\text{g/L}$ ). WFD status: poor = orange, moderate = yellow, good = green, high = blue. Values show down-weighted normalised chlorophyll EQR values, with calculations described in Appendix E5.2.2.**

	Baseline	Climate change	AMP8	Low agricultural diffuse	No sewage
<b>Windermere North Basin</b>	0.646	0.587	0.614	0.638	0.729
<b>Windermere South Basin</b>	0.659	0.603	0.682	0.785	0.803
<b>Esthwaite Water</b>	0.328	0.330	0.365	0.387	0.428

## 5. Discussion

### 5.1 Catchment nutrient dynamics

Our results indicate that future climate change will alter the load, concentration and timing of nutrient inputs to Windermere and Esthwaite Water in the future.

In baseline conditions, catchment derived nutrient loads entering the study lakes were highest for Windermere North Basin, with decreasing inputs delivered to Windermere South Basin and Esthwaite Water. This is most likely a function of contributing catchment size and number of inflows. In addition, Windermere North Basin receives inflows transporting the highest loads (rivers Rothay and Brathay) and concentrations (Blelham Beck, Mill Beck) of phosphate and nitrate modelled in the Windermere catchment. Nutrients entering Windermere South Basin from upstream Windermere North Basin will have been subject to processes including phytoplankton uptake and their loss to sediment, potentially reducing the amount of nutrients available to be transported downstream. This is particularly likely in lakes such as Windermere North Basin, which have long residence times (~180 days) (Grefe and others, 2024; Vigiak and others, 2025). Although catchment nutrient loading to Esthwaite Water was more than eight times lower than to Windermere, its chlorophyll *a* concentration was over three times higher, suggesting input of additional nutrients potentially from internal loading, which has previously been identified in this lake (Mackay and others, 2014; Anderson, 2018; Moorhouse and others, 2024b).

Sources of nutrients varied across the lake sub-catchments in baseline conditions. For inflows to Windermere North Basin, phosphate primarily came from wastewater infrastructure (63%) (sewage treatment works, intermittent discharges, septic tanks) and nitrate from diffuse sources (72%), with variability across some input streams. Direct catchment derived nutrients entering Windermere South Basin (excluding inputs from Esthwaite Water and Windermere North Basin) were predominantly from sewage (phosphate = 85%, nitrate = 68%). Catchment inputs for Esthwaite Water were from diffuse sources (phosphate = 62%, nitrate = 87%). Intermittent discharges and septic tanks made comparatively minor contributions to combined phosphate and nitrate inputs to the combined basins of Windermere (intermittent = 5%, septic tank = 1%) and Esthwaite Water (intermittent = 7%, septic tank = 2%). While their proportional contribution to nutrients entering the lakes was low, septic tanks may impact the localised areas of the lake to which they drain, particularly where multiple tanks are positioned close together. In addition, as for all wastewater inputs to freshwaters, septic tank discharge contains more than nutrients, with other potential impacts on receiving waters (e.g., Wilschnack and others, 2024; Gyimah and others, 2024). These findings reflect the impact of catchment specific characteristics on nutrient dynamics and align to studies highlighting the need to tailor nutrient management to local conditions (Costa and others, 2023; Grigorieva and others, 2023; Suresh and others, 2023).

This modelling study suggests that climate change will increase the annual load of catchment derived phosphate entering Esthwaite Water (7%), and nitrate entering

Windermere South Basin (3.8%) in 2076-2079. These findings align with European studies which document the increased transport of nutrients along a continuum from catchment soils and sediments to freshwater systems in response to future increases in the volume and intensity of winter rainfall and river flows (Ockenden and others, 2016; Costa and others, 2023; Ezzati and others, 2023). Changes in rainfall and flow can subsequently fuel increases in lake phytoplankton biomass (Jeppesen and others, 2009; Nöges and others, 2011). Other nutrient inputs across the study lakes responded differently to climate change (annual loads decreased  $\leq 2.5\%$  relative to baseline values). Climate impacts on nutrient mobilisation and hydrological connectivity between nutrient sources and freshwaters can be modified by catchment specific variability in soil type, land cover and local land management (Brownlie and others, 2022; Ezzati and others, 2023).

Nutrient inputs to Windermere and Esthwaite Water exhibited seasonal variability, with phosphate and nitrate loading higher in winter (October-March) than summer (April-September). This was potentially due to higher rainfall rates increasing the mobilisation and transport of dissolved nutrients across the catchment in winter and periods of lower river flow reducing the delivery of nutrient loads to the lakes in summer (Jeppesen and others, 2009; Costa and others, 2023; Fowler and others, 2024). In addition, nutrient uptake can be highest in rivers during summer when conditions for plant growth are optimised (Clarke, 2002), reducing the load available to downstream lakes. Our modelling study suggests that climate change will exacerbate this seasonal pattern in Windermere and Esthwaite Water, with future phosphate loading projected to increase in winter and reduce in summer. This potentially reflects the wetter winters, and hotter, drier summers expected to characterise temperate regions in a changing climate, and their subsequent seasonal impact on nutrient dynamics in lake catchments (IPCC, 2007; Jeppesen and others, 2009; IPCC, 2023; Costa and others, 2023). This seasonal trend was also identified for future nitrate loading to Windermere South Basin, but this may have been driven by changes to nutrient dynamics within the lake rather than its catchment. As modelled climate change led to an increase in phytoplankton growth in upstream Windermere North Basin in the summer months, more nitrate was consumed in this lake, reducing the load available to flow into Windermere South Basin between May and September.

An exception to this seasonal trend of nutrient delivery to the lakes was the high modelled phosphate loads entering Windermere North Basin in May, August and September in baseline conditions. The Windermere catchment receives approximately 7 million visitors annually which peak from July to September (Love Windermere, 2025), and as Windermere North Basin has the largest sub-catchment of the study lakes, high inflowing phosphate loads in spring and summer could reflect the impact of a greater tourist footfall (Delowar and others, 2024; EA, 2025a). Our study found 63% of phosphate to originate from wastewater infrastructure (of which 53% derived from sewage) in this sub-catchment. These high summer phosphate inputs occurred within the modelled diatom (April-June) and cyanobacteria (June-October) dominated peaks in phytoplankton biomass in Windermere North Basin, ensuring nutrients enter the lake when in greatest demand by the phytoplankton community for growth.

Modelled climatic change also drove seasonal variability in the concentration of catchment derived phosphate and nitrate entering the study lakes, with increases projected for summer and decreases for winter, effectively reversing the seasonal trend identified for future nutrient loading. This is potentially because the lower river flows expected to reduce lake nutrient loading in hotter, drier summers may concentrate inflow nutrients (Nöges and others, 2011; Ockenden and others, 2016; Ezzati and others, 2023), while the higher river flows expected to increase lake nutrient loading in wetter winters may dilute them (IPCC, 2007; Ockenden and others, 2016). Relative to baseline conditions, modelled nutrient concentrations increased for inflows to all study lakes under climate change, with the greatest increases projected from June to September. This could increase nutrient supply to the phytoplankton communities when their additional requirements for growth were optimised (e.g., warmer water temperatures, longer day lengths) (Nöges and others, 2011; Huisman and others, 2018; Dory and others, 2024), potentially driving the modelled summer increases in future phytoplankton biomass for each lake. As modelled climate change led to a greater percentage increase in phosphate concentrations relative to nitrate concentrations for inflows to each study lake, the resulting shifts in the ratio of available nutrients may further influence phytoplankton growth (Graeber and others, 2024).

## 5.2 Lake phytoplankton communities

Baseline mean lake water temperatures increased by approximately 2.3°C across all three lakes due to the climatic changes modelled for 2076-2079. Similar rates of warming have been documented for Windermere and Esthwaite Water (Thackeray and Mackay, 2023), and align to global rates of lake warming of approximately 0.34°C per decade (Woolway and others, 2020).

Warming lake waters may have contributed to the modelled increase in total phytoplankton biomass (chlorophyll *a* concentration) projected for both Windermere basins (2076-2079). These increases are sufficient to reduce projected WFD status (based on chlorophyll only) from Good to Moderate in Windermere North Basin by 2076-2079. Previous studies identified a positive relationship between water temperature and rates of metabolism and growth for many phytoplankton types (Reynolds, 2006; Paerl and Huisman, 2009; Dory and others, 2024). This climate change induced increase in total phytoplankton biomass was not evident in Esthwaite Water. This may be because chlorophyll *a* concentration was already high in baseline conditions. Despite this, the disproportionate increase in the biomass of cyanobacteria relative to other phytoplankton types was identified for both Windermere and Esthwaite Water, and may reflect their physiological adaptations to warmer water temperatures which provide them with a competitive advantage over other phytoplankton types (Carey and others, 2012; Huisman and others, 2018; Dory and others, 2024).

In Esthwaite Water, mean monthly cyanobacteria biomass comprised over 90% of the phytoplankton biomass from July to September, under modelled climatic changes (2076-2079). This reduced diversity of the phytoplankton community (Dory and others, 2024) could impact energy transfer to higher trophic levels (Amorim and Moura, 2021). This

projected restructuring of lake phytoplankton communities aligned with previous modelling of Windermere (Elliott, 2012; Elliott and others, 2006) and evidence of increasing cyanobacteria bloom frequency worldwide (Huisman and others, 2018).

This study found climate change induced increases in cyanobacteria biomass increased the number of days for which the WHO10 threshold was exceeded in all three lakes. Exceedances occurred in an additional summer month in both Windermere basins and continued to occur for half of the year (June-December) in Esthwaite Water. As these peak cyanobacteria events host the potential for concentrated cyanotoxin production (Igwaran and others, 2024), the risk they pose to human and ecosystem health (EA, 2022) is projected to increase in frequency and duration for both lakes. As these exceedances occur predominantly in the summer months when lakes are most frequently used for recreation (EA, 2025a), such as the Great North Swim and other water-based events in Windermere, then an increase in their occurrence amplifies their associated risk.

Alongside directly impacting the phytoplankton community, modelled climatic changes extended the duration of thermal stratification in each lake, with an earlier onset in Windermere North Basin (19 days) and Esthwaite Water (14 days). The average stratification extensions expected by 2076-2079 were 31 days for Windermere North Basin, 25 days for Windermere South Basin and 15 days for Esthwaite Water. Across the Northern Hemisphere, climate warming is projected to extend the summer lake stratification period by 33.3 days ( $\pm 11.7$  days) by the end of the century (Woolway and others, 2021). This aligns closely with the findings of this study, particularly for Windermere. Thermal stratification can be prolonged by mechanisms that reduce vertical mixing, including climate warming of surface waters that exacerbate their density difference from cooler bottom waters, and reduced flushing caused by low rainfall in periods of low river flow or drought (Saber and others, 2020; Thackeray and Mackay, 2023).

Prolonged thermal stratification can extend the period for which anoxic conditions occur at the lakebed (Woolway and others, 2022). This can lead to phosphorus release from bed sediments which become available to phytoplankton for growth, independent of catchment nutrient supply (Me and others, 2018; Farrell and others, 2020; Woolway and others, 2021). While these processes were not measured directly in this study, they may occur in Windermere and particularly shallower Esthwaite Water during the extended stratification periods projected for 2076-2079. Modelled climatic changes also caused mixed depth to become shallower for each study lake. This aligns to the shallower stratification depth identified for Windermere, particularly the North Basin, in response to modelled increases in air temperature (Elliott, 2012). A shallowing mixed depth means the layer of buoyant, oxygen enriched water at the surface of each lake will reduce in volume, limiting oxygen availability to a greater proportion of the lake (Stetler and others, 2021). While these conditions may negatively impact many organisms, it could favour the growth of phytoplankton with the capacity to regulate their buoyancy and migrate vertically through the water column, including many cyanobacteria, as they can continue to access resources in the mixed surface layer (Carey and others, 2012; Huisman and others, 2018).

## 5.3 Nutrient management scenarios

The three modelled nutrient management scenarios illustrated that greater reductions in nutrient inputs led to greater reductions in total phytoplankton biomass and WHO10 exceedances for Windermere North Basin, Windermere South Basin and Esthwaite Water.

The complete removal of phosphate and nitrate inputs from modelled wastewater infrastructure was found sufficient to prevent the increases in total phytoplankton biomass projected to occur in Windermere under future climate change, and to reduce this biomass below baseline values in both Windermere and Esthwaite Water. Under this modelled absence of sewage inputs, the biomass of summer cyanobacteria dominated phytoplankton peaks in each lake diminished and was effectively removed from Windermere South Basin. As a result, WHO10 exceedances were reduced in Esthwaite Water and prevented completely in Windermere, reducing the risk posed to the health of water users in future.

Theoretical maximum reductions in nutrient contributions from the agricultural sector were sufficient to reduce total phytoplankton biomass below values expected in future climate change and baseline conditions for Windermere and Esthwaite Water. However, this nutrient management scenario led to smaller reductions in WHO10 exceedances in Windermere North Basin and Esthwaite Water, when compared to the removal of sewage inputs.

Completion of works scheduled in AMP8 to reduce phosphate inputs prevented the increase in total phytoplankton biomass expected to occur under climate change in Windermere South Basin. This lake continued to experience WHO10 exceedances, but fewer than in baseline conditions. These findings aligned with published studies that illustrate the need for current management strategies to further reduce catchment nutrient inputs to lakes, to accommodate the expected impacts of climate change on lake phytoplankton communities (Jeppesen and others, 2009; Trolle and others, 2011; Paerl, 2014; Moorhouse and others, 2024b).

Windermere North Basin, Windermere South Basin and Esthwaite Water responded differently to the three nutrient management scenarios. Windermere South Basin required the smallest reductions in nutrient inputs to address the impacts of climate change on its phytoplankton community. Its increased sensitivity to the AMP8 phosphate reductions relative to Windermere North Basin, may be due to a greater proportion of its catchment derived nutrients originating from wastewater infrastructure.

For Esthwaite Water, all three nutrient management scenarios reduced total phytoplankton biomass and the number of WHO10 exceedances below baseline conditions. However, total phytoplankton biomass and WHO10 exceedances remained high. This may be because Esthwaite Water has a different nutrient legacy, with historic aquaculture practices (Maberly and others, 2011). The fish farm that is now closed has contributed to the enrichment of lake bed sediments with nutrients that are then released into the water column independent of current catchment inputs (Mackay and others, 2014; Anderson, 2018). It is also likely that, as Esthwaite Water is a much shallower lake than Windermere,

it will be more sensitive to this internal loading as a result of its morphology (Gorham and Boyce, 1989). When stratified, the volume of water below the mixed depth will be smaller in Esthwaite Water than in the deeper Windermere basins, reducing the water available to contain and dilute internal loads, making them more accessible to the phytoplankton community for growth (Gorham and Boyce, 1989). In turn, even when all modelled sewage inputs were removed, future cyanobacteria biomass continued to exceed the WHO10 or WHO50 thresholds from June to December in Esthwaite Water. Esthwaite Water is less used for recreation than Windermere so direct risks will be lower but this highlights the importance and benefit of site-specific modelling and management of lakes in a changing climate (Richardson and others, 2018).

Modelled reductions in nutrient inputs led to a trajectory of improvement in chlorophyll EQR and WFD class for each lake. Windermere South Basin achieved High class when all sewage inputs were removed. Despite the nutrient management scenarios leading to greater reductions in chlorophyll *a* concentration than for Windermere, this only improved Esthwaite Water's status from Poor to Moderate. Esthwaite Water may require time beyond the duration of this study (2079) to achieve a higher WFD class.

## 5.4 Study limitations

Modelling studies inherently involve assumptions. This section details how these may have influenced the main study findings.

1. There was a disparity between observed and modelled chlorophyll *a* concentration in Esthwaite Water, suggesting that a source of nutrients was not captured in the PROTECH model. This source may have been a nutrient input not currently represented in the catchment water quality model, or could reflect the release of nutrients from lakebed sediments, a biogeochemical process previously identified as an important contributor to the nutrient budget of Esthwaite Water (Mackay and others, 2014; Moorhouse, Thackeray, and others, 2024b). It is not currently possible to model this complex mechanism in PROTECH.
2. SRP derived from internal loading in Esthwaite Water may have been overestimated. This study corrected for the disparity between observed and modelled phytoplankton biomass in baseline conditions for Esthwaite Water by adding SRP into the lake surface layers of PROTECH model runs during the summer months (Section 4.3.1.3). However, the model's ability to accurately represent this nutrient input in future was less certain, particularly if the disparity was caused by internal phosphorus loading. This is because although the rate of internal loading changes over time, the addition of SRP was held constant in PROTECH in the future scenarios, because of the complexity of quantifying this variability. Release of sediment nutrients can be influenced by factors including wind disturbance, periods of high pH, and sustained anoxia, which can be exacerbated by climate change induced alterations to water temperature and thermal stratification dynamics (Mackay and others, 2012, 2014; Moorhouse and others, 2024b; Nürnberg, 2025). In addition, if nutrient inputs from the catchment

were constrained then the store of lakebed nutrients may be depleted by release over time. This may have led to an overestimation of the SRP entering downstream Windermere South Basin.

3. The modelled transfer of nutrients from Windermere North Basin to Windermere South Basin may have been underestimated. PROTECH model runs of Windermere North Basin comprise one of the inputs when modelling downstream Windermere South Basin, with nutrients not assimilated into phytoplankton biomass or their loss to sediment assumed to be transferred from one basin to the other. As PROTECH is a 1D model, it does not capture detailed 3D water movements within lakes. For example, nutrients that enter Windermere North Basin close to the narrow shallow region which demarks the two basins (e.g., from Mill Beck or direct loads) may pass into Windermere South Basin before these nutrients are converted into phytoplankton biomass. Therefore, a slightly greater nutrient load may enter Windermere South Basin than is represented in the PROTECH model runs.
4. Modelled estimates of WHO10 and WHO50 exceedance may have been conservative, and the lived experience of water users could differ from modelled projections. PROTECH is a 1D model that provides chlorophyll *a* concentration values that are homogenised across the top 7 m of the lake surface water. This means the model outputs do not reflect the spatial variability in algal distributions that can result when prevailing winds and upwelling cause accumulations of phytoplankton in downwind bays and specific areas of the lake (Cyr, 2017). While the average phytoplankton conditions may not have exceeded WHO10 thresholds for cyanobacteria biomass, localised exceedances could occur. Wind speed was an input to PROTECH, but it is difficult to accurately forecast this parameter into the far future. In effect, PROTECH was diagnostic of the whole lake phytoplankton community, rather than spatial aggregations within this.
5. Catchment nutrient loading to the lakes resulting from climate change driven changes to rainfall intensity may be inaccurate and could be conservative. Extreme flow events may disproportionately transport catchment nutrients and sediment in a changing climate (Ockenden and others, 2016). However, as monthly uplift factors were used to adjust baseline time-series discharge data to reflect the climatic changes projected in RCP8.5, flow data inputs to the future scenarios could not be used to project changes to high and low percentile flow events. In effect, the models could not capture the extreme episodic rainfall events that lead to high flows, or periods of drought that lead to low or no flows, that are projected to characterise a warming world (IPCC, 2023; Costa and others, 2023; Kendon and others, 2023; Fowler and others, 2024). Despite this, this approach was taken so that future projections of flow were grounded in observed, catchment specific trends. The movement of sediments could not be parameterised in these models.
6. Climate change may influence lake ecosystems through mechanisms in addition to the altered nutrient dynamics considered in this study. For example, increasing lake water temperatures may alter trophic interactions. Increased abundance of fish species that preference warmer waters, such as roach, could lead to increased predation of zooplankton and in turn, reduced grazing of phytoplankton (Maberly and others, 2008). This could lead to the proliferation of phytoplankton without alteration of nutrient concentrations, as previously hypothesised for Windermere

(Maberly and others, 2008). In addition, the response of these systems to climate change could be compounded by other challenges to water quality, including chemical pollutants, invasive species and physical habitat degradation (Birk and others, 2020; Tickner and others, 2020).

7. Our modelling explored a limited range of management scenarios. We deliberately chose an extreme option (removing all wastewater inputs) and already planned nutrient reductions to illustrate the range of possible outcomes. Further modelling would be needed to understand the effectiveness of other management interventions. In addition, the impacts of climate change will continue beyond the 21st century, so the effectiveness of some of the tested management scenarios may not extend beyond 2080.

## 8. Conclusions

Climate change is altering lake ecosystems globally. This study combined climate data with a catchment water quality model and phytoplankton community model. In doing so, it illustrated the impact of future climate change on catchment nutrient supply and lake phytoplankton communities in three lakes in the English Lake District's Windermere catchment: Windermere North Basin, Windermere South Basin and Esthwaite Water. It tested three future nutrient management scenarios to determine their efficacy when addressing these projected climate impacts.

The climate change projected for 2076 to 2079 in a high greenhouse gas emission future (RCP8.5) is expected to alter the catchment nutrient dynamics and lake phytoplankton communities of the Windermere catchment in the following ways, relative to baseline conditions.

### **Climate change impacts on catchment nutrient dynamics:**

- Increased phosphate loading to Esthwaite Water (by 7%)
- Increased nitrate loading to Windermere South Basin (by 3.8%)
- Increased concentration of phosphate by 33% and nitrate by 13% in inflows to Windermere North Basin.
- Increased concentration of phosphate by 4% and nitrate by 10% in Windermere South Basin
- Increased concentration of phosphate by 22% and nitrate by 12% in Esthwaite Water
- The greatest increases in phosphate and nitrate concentrations entering each lake occurred in the summer months

### **Climate change impacts on lake phytoplankton communities:**

- Increased total phytoplankton biomass in both Windermere basins
- Increased summer cyanobacteria dominance in all three lakes
- Increased number of days per year in which cyanobacteria biomass exceeded the World Health Organisation thresholds for low risk of adverse effects on human health in drinking and recreational waters (WHO10) in all three lakes
- WHO10 exceedance increased from an annual average of 8 to 36 days (Windermere North Basin), from 50 to 92 days (Windermere South Basin) and from 159 to 167 days (Esthwaite Water)
- WHO10 exceedances primarily occurred in the summer months when the risk to human health is amplified by increased recreational activity

### **Climate change impacts on lake physical properties:**

- Increased mean lake water temperatures by approximately 2.4 to 2.5°C in Windermere, and 2°C in Esthwaite Water

- Shallower mixed depth and increased duration of stratification in all three lakes, with earlier onset of stratification identified in Windermere North Basin (19 days) and Esthwaite Water (14 days)

The nutrient management scenarios would help to reduce these projected impacts of climate change on each lake phytoplankton community in the following ways:

### **AMP8 impacts on lake phytoplankton**

The AMP8 scenario (2076 to 2079) assumed climate change was in effect and simulated the phosphate reductions detailed in Asset Management Plan 8, which are scheduled for completion by 2030. The following impacts were shown:

- Greater reductions in total phytoplankton biomass in Windermere South Basin than Windermere North Basin. This parameter was reduced 18% relative to baseline conditions in Windermere South Basin
- In Windermere South Basin, WHO10 exceedance reduced from an average of 50 days per year in baseline conditions, to 3 days (2076 to 2079)
- In Windermere North Basin, the WHO10 threshold was exceeded on an annual average of 24 days, only partly addressing the increase in WHO10 exceedances projected between baseline (8 days) and climate change (36 days) conditions
- In Esthwaite Water, the WHO threshold was exceeded on average 151 days a year, despite this lake having the greatest reductions in phytoplankton biomass of the three lakes

### **Low agricultural diffuse scenario impacts on lake phytoplankton**

The low agricultural diffuse scenario (2076 to 2079) assumed climate change and AMP8 were in effect but with reduced phosphate (40%) and nitrate (15%) inputs to represent theoretical maximum reductions in diffuse pollution from agriculture. The following impacts were shown:

- In Windermere South Basin and Esthwaite Water, total phytoplankton biomass and WHO10 exceedances reduced below baseline values, with reductions greater than in the AMP8 scenario
- The WHO10 threshold was exceeded on average on 14 days a year in Windermere North Basin and 136 days in Esthwaite Water
- In Windermere South Basin, the summer peaks in cyanobacteria biomass were effectively removed and the WHO10 threshold was no longer exceeded
- In Windermere North Basin, this scenario could not fully prevent the increases in total phytoplankton biomass projected to be driven by climate change

### **No sewage scenario impacts on lake phytoplankton**

The no sewage scenario (2076 to 2079) assumed climate change was in effect and removed all phosphate and nitrate inputs from water company and private wastewater

treatment works, septic tanks and intermittent discharges. The following impacts were shown:

- Total phytoplankton biomass and WHO10 exceedances were the lowest reported of all modelling scenarios. This scenario prevented the projected increases in these parameters under climate change (Windermere) and reduced them below baseline values (all three study lakes)
- Total phytoplankton biomass reduced by 48% in Windermere North Basin, 61% in Windermere South Basin and 69% in Esthwaite Water, relative to the climate change scenario
- In Windermere North Basin, this was the only modelled nutrient management scenario sufficient to completely prevent the increases in phytoplankton biomass projected under climate change
- In Windermere South Basin, the summer peak in cyanobacteria dominated phytoplankton biomass was absent
- In Esthwaite Water, the WHO10 threshold was exceeded on an annual average of 105 days, the lowest values modelled for this lake.
- Reducing nutrients led to a trajectory of improvement in WFD class, with this scenario projected to move Esthwaite Water from Poor to Moderate class, and Windermere South Basin from Good to High class.

### **Implications for future lake management**

Windermere North Basin, Windermere South Basin and Esthwaite Water varied in their response to the three nutrient management scenarios, reflecting the role of lake morphology, catchment characteristics, size and nature of nutrient sources, and land use history. This variation highlighted the importance of site-specific factors and consideration of climate change impacts in shaping future lake management strategies.

Climate change and changes to nutrient dynamics will continue to impact lakes of the Windermere catchment, and those facing similar pressures, in future. However, this study demonstrated that local nutrient management could reduce some of the pressures placed on lake environments by global climate change.

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## 10. List of abbreviations

AMP	Asset Management Period
CERF	Continuous Estimation of River Flows
DEFRA	Department for Environment, Food and Rural Affairs
EEA	European Environment Agency
EQR	Ecology Quality Ratio
Farmscoper	FARM SScale Optimisation of Pollutant Emission Reductions (model)
FBA	Freshwater Biological Association
IPCC	Intergovernmental Panel on Climate Change
JASMIN	
m.a.s.l	meters above sea level
MIDAS	Met Office Integrated Data Archive System
NEAP-N	
NOAA	National Oceanic and Atmospheric Administration
NRFA	National River Flow Archive
pH	Potential of Hydrogen
PLUTO	PhytoPLankton classification with Uncertainty TOol
PROTECH	Phytoplankton RespOnses To Environmental CHange
PSYCHIC	Phosphorus and Sediment Yield CHaracterisation In Catchments
Python	
Qube	
RCP8.5	Representative Concentration Pathway 8.5
SAGIS	Source Apportionment Geographical Information System
SIMCAT	SIMulations of water quality in CATchments
SRP	Soluble reactive phosphorus

Tchl	Total chlorophyll a concentration (µg/L)
TP	Total phosphorus
UKCEH	UK Centre for Ecology & Hydrology
UKCP18	United Kingdom Climate Projections 2018
UNESCO	United Nations Educational, Scientific and Cultural Organization
UU	United Utilities
WFD	Water Framework Directive
WHO	World Health Organisation
WHO10	World Health Organisation (10 µg/L of cyanobacteria chlorophyll a threshold) (this study only)
WHO50	World Health Organisation (50 µg/L of cyanobacteria chlorophyll a threshold) (this study only)
WIMS	Water Information Management System
WINEP	Water Industry National Environment Programme
WwTW	Wastewater Treatment Works

# 11. Glossary

Algae	A broad group of photosynthetic organisms, many of which are classed as phytoplankton.
Algal blooms	Rapid surface accumulations of phytoplankton, a functionally and morphologically diverse group of microscopic primary producers.
Anoxic	Completely lacking oxygen.
Anthropogenic	Caused by human activity.
Aquatic ecosystems	The physical, chemical and ecological components of water environments and the interactions between these. These environments include freshwater (rivers, streams, lakes, ponds), marine (oceans, seas) and estuarine (estuaries) waterbodies.
Basin (lake)	In this report, lake basin refers to a water filled depression, rather than a lake's wider drainage area.
Biodiversity	The variety of plants and animals living together in a particular place.
Biomass	The organic material that makes up living animals and plants.
Catchment	The area over which water drains, to reach a specific point.
Chlorophyll <i>a</i>	A pigment that absorbs light energy for use in photosynthesis. The concentration of chlorophyll <i>a</i> is used as a measure of phytoplankton biomass.
Climate change	Long-term shifts in meteorological parameters, including mean weather patterns and air temperatures.
Community structure	The number of different species found within an ecological community, the relative abundance of these species and how they interact with each other.
Cyanobacteria	Sometimes called blue-green algae, these are bacteria. They can photosynthesise and some forms can produce toxins (cyanotoxins). They form part of the phytoplankton community.

Decay rate	The rate at which phosphate and nitrate concentrations are lost from a river reach, to simulate removal to sediment or through biological uptake.
Decomposition	The breakdown of organic material into its simpler constituent organic and inorganic components.
Diatom	A diverse group of microalgae, found in water and soil ecosystems. Their cell wall is made of silica. In lakes, they form part of the phytoplankton community.
Diffuse sources	Dispersed outlets of nutrients, including those from fertilisers and livestock manures washed from the soil, alongside runoff from roads, construction sites, forestry operations and urban centres.
Direct load	A nutrient load entering a lake from unmodelled river reaches, plus load entering directly, rather than being transported there via a river or stream.
Ecosystem (services)	The physical, chemical and ecological components of an environment that interact and operate as a system. These systems provide direct and indirect benefits to human society, through provisioning, regulating, supporting and cultural services.
Evapotranspiration	The movement of water from the Earth's surface into the atmosphere. This occurs through both transpiration from plants and evaporation of water from waterbodies, ice and soil.
Eutrophication (eutrophic)	The proliferation and accumulation of aquatic plant biomass in response to increasing phosphorus and nitrate inputs, with harmful consequences for aquatic ecosystems and human health.
Farmscoper	A decision support tool based in Excel, which calculates the diffuse pollution load from farms and assesses the impacts of different methods for reducing this pollution from the agricultural sector.
Flushing	When the total volume of water within a lake is moved downstream and replaced by inflowing inputs and precipitation. The flushing rate is the time taken for this to occur and is effectively the inverse of hydraulic residence time.

Mixed depth	In PROTECH, mixed depth is defined as the depth of the homogeneously mixed surface layer within a stratified lake – the distance from the lake surface to the point at which the warmer, less dense surface waters no longer mix with cooler, denser, deeper waters. Mixed depth is influenced by (and calculated using) wind speed at the lake surface, water density, and water temperature.
Hydraulic residence	Period of time for which water remains in a lake.
Hydrological connectivity	Movement of water across landscapes, the pathways taken by flowing waters.
Inflows	Rivers or streams flowing into a lake.
Intermittent discharges	Discharges that release, stop, and then potentially release again. This includes discharges from storm overflows and combined sewer overflows.
Leaching	The dissolution of substances into water, that then percolates through the soil, or other media, transporting the solutes to watercourses and waterbodies.
Monte Carlo	A technique that randomly samples a population, repeatedly. It can also be called a shot-based approach and can be used to quantify the uncertainty in a model. It provides a range of outcomes and the probability of each.
Natural flow	Within Qube, natural flows are estimations of river flow that would occur in a particular river reach in the absence of anthropogenic influence, such as abstraction, impoundment or discharges.
Nutrient budget	Quantification of the nutrients entering and leaving a particular ecosystem over a specified time.
Nutrient concentration	The amount of a particular nutrient (e.g., phosphate, nitrate) within a volume of water.
Nutrient dynamics	The movement, transformation and cycling of nutrients (e.g., phosphorus and nitrogen) through landscapes and within aquatic systems over time.
Nutrient load	The total amount of a particular nutrient (e.g., phosphate, nitrate) entering a waterbody during a specified period.

Oktas	A unit of measurement for cloud cover. The sky is divided into eighths, with 0 oktas meaning a completely clear sky, and 8 oktas representing full cloud cover.
Other algae	Of the eight phytoplankton types modelled by PROTECH for this study, this refers to the types which cannot be classified as cyanobacteria or diatoms (e.g., green algae).
Photosynthesis	The biological process through which plants create nutrients from water and carbon dioxide, using sunlight. This process provides energy for all other levels of the food web.
Phytoplankton	A functionally and morphologically diverse group of photosynthetic microscopic organisms which form the base of aquatic food webs. This community includes diatoms, cyanobacteria and other algae.
Point sources	Discrete outlets of nutrients, including those from sewage treatment works and septic tanks.
Qube	A national hydrological modelling platform developed by Wallingford HydroSolutions that provides spatially consistent estimates of river flows for gauged and ungauged catchments.
Respiration	The biological process through which living organisms take in oxygen and release carbon dioxide.
Runoff	Draining of water and its solutes across the surface of the land into watercourses and waterbodies.
Sonde	A scientific instrument used to measure environmental parameters that automatically transmits the information to remote receivers, often in real-time. They can be used in locations that are difficult to access, including on lakes.
Source apportionment	Quantification of the contribution of different sectors or sources (e.g., wastewater treatment works, intermittent discharges, septic tanks, industrial inputs, agriculture, mine water drainage, soil erosion) to the nutrients found at a particular site.
Storm overflows	Release of excess water (often a combination of rainfall and wastewater) into downstream rivers during storm

events to prevent the sewer system from becoming overwhelmed and backing up.

Stratification (thermal)	When heating of the surface layers of lake water during spring and summer causes them to become less dense than cooler deeper waters, and the difference in density prevents them from mixing. Defined in this study as the number of days for which there was consistent absence of full lake mixing.
Tchl	Total chlorophyll <i>a</i> concentration ( $\mu\text{g/L}$ ) of the whole phytoplankton biomass averaged across the four-year modelling periods.
Time series	A dataset that records variability of a parameter through time, often at regular, standardised intervals.
Trophic	Feeding relationships between organisms. Often used to describe different levels within a food chain or web.
Uplift factors	The uplift factor approach is a method used in climate science to estimate changes in a particular parameter under climate change using scaling factors.
Water Framework Directive	European legislation designed to protect and improve freshwater environments.
Water quality	The physical and chemical characteristics of water that influence aquatic ecosystems and the services they provide.
WHO10	The number of days when the World Health Organisation derived risk threshold of $10 \mu\text{g/L}$ of cyanobacteria chlorophyll <i>a</i> is exceeded, signifying low potential for adverse human health impacts.
WHO50	The number of days when the World Health Organisation derived risk threshold of $50 \mu\text{g/L}$ of cyanobacteria chlorophyll <i>a</i> is exceeded, signifying moderate potential for adverse human health impacts.
Zooplankton	Small organisms that float, drift or weakly swim in the water column. They are not photosynthetic but feed upon primary producers, including phytoplankton.

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