



Environment
Agency



Nutrient futures for Windermere: appendices

Chief Scientist's Group report

May 2026

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1. Appendix A: SIMCAT-SAGIS

This section explains selection of the SIMCAT (SIMulation of water quality in CATchments)-SAGIS (Source Apportionment Geographical Information System) model, details its limitations and describes the version used for the Nutrient Futures for Windermere study.

1.1 Selection of the SIMCAT-SAGIS model

SIMCAT and SAGIS are the main water quality catchment modelling tools used by environmental regulators in the UK, including the Environment Agency. SIMCAT utilises mass-balance approaches to predict water quality and has been used for over 40 years. The code for SIMCAT has recently become available in an open source GitHub repository (DEFRA, 2025). SAGIS has been jointly developed with the UK Water Industry Research group (UKWIR) since 2010 to better describe and constrain diffuse sources of pollution. When combined, the primary use of the models has been to drive water company investment, but they are increasingly being implemented to help understand the significance of other sources of pollution (EA, 2023) and to test future scenarios (EA, 2024a).

Careful consideration was given to the assumptions made when using this statistical regulatory model in a simulation setting for this study. Other modelling approaches were considered, including the Catchment Sensitive Farming HYdrological Predictions for the Environment (CSF-HYPE) model. SIMCAT-SAGIS was chosen as it (1) could test changes to the nutrient contributions of different sectors into the future, (2) modelled Windermere North Basin and Windermere South Basin separately, as required by the Phytoplankton RespOnses To Environmental Change model (PROTECH), and (3) contained the most up-to-date information on sectoral loads across the Windermere catchment, including amendments proposed in the most recent Water Industry National Environment Programme (WINEP).

The SIMCAT-SAGIS model outputs were validated against in-river observational data from the Windermere catchment where available. Event duration monitoring (EDM) data provided by United Utilities (UU) was quality assured by the Environment Agency prior to its inclusion in the models (EA, 2025).

1.2 Limitations of SIMCAT-SAGIS

Despite being the most suitable option for this study, the SIMCAT-SAGIS model has the following limitations:

- The model assumes steady state conditions, using a Monte Carlo shot-based approach to addressing uncertainty. Therefore, the output was provided as an annual or monthly average over a given period in which the model was validated against. For this reason, it could not provide a precise estimate of water quality

parameters for a given day and instead gave a statistical distribution of likelihood for each water quality parameter. This became particularly problematic when considering intermittent inputs such as combined sewer overflows (CSOs), which may either spill or not spill on a given day. This also limited the ability of the model to simulate temporally dynamic events, such as first flush events, where inputs may be concentrated in the first onset of rainfall (Lee and others, 2002).

- Observed data was not available for every sector at every location. In some places, historic data or catchment defaults were used to supplement existing data.
- Diffuse sectoral source apportionment was applied on a waterbody scale. This was derived from a spatial interpolation of 1 km gridded apportionment predictions from the Phosphorus and Sediment Yield CHaracterisation In Catchments model (PSYCHIC) and NEAP-N model (nitrate) (Davison and others, 2008) that feed into SIMCAT-SAGIS. This spatial resolution meant certain features were aggregated and fine detail may have been lost, as resolving smaller variations in sectoral apportionment was not possible.
- As a regulatory tool, SIMCAT-SAGIS uses the precautionary principle. For certain sewage treatment works, particularly those with non-numeric permit limits, a conservative estimate was simulated. During calibration to observed data, this could have effectively reduced the diffuse setting. Therefore, SIMCAT-SAGIS typically overestimated sewage and intermittent assets in the absence of numeric discharge data. However, for the Windermere catchment, these estimates were supported by additional data collected from the largest of these discharges that illustrated the estimates used in the absence of monitoring data were not unduly conservative.

1.3 SIMCAT-SAGIS in the Windermere catchment

There are 17 different SIMCAT-SAGIS models for regions of England. The *Lakes model* is the regional catchment model which contains Windermere and Esthwaite Water. This model was constructed as part of the Price Review 2024 (PR24) process for water quality planning. During this process, a 2014-2018 baseline period of water quality and flow data was used to build and calibrate the model. Bespoke amendments to the model have been developed to explore the effects of other changes in the catchment since the baseline model was established. One of these was the creation of *Wind2020.dat* (13/09/2023). This model contains the calibrated Windermere model but amended with any improvements to nutrient inputs that were confirmed to be implemented between 2015 and 2020 in the sixth Asset Management Period (AMP6). It was developed as part of work commissioned for the Love Windermere Partnership to represent current water quality in Windermere, and was the model used for this study.

The model script was amended to ensure storm tank inputs were more accurately simulated. In previous iterations, UU sewer model data were used to provide a load in the model. This was input as a log-normal distribution into the SIMCAT model, which led to slight overestimations of the storm overflow component. The correct representation, to provide the desired average, is to assign a uniform loading of the storm tank loading,

resulting in an effective average which is equal to the desired average. This approach reduced the predicted loads from storm tanks (particularly in Black Beck and the River Rothay). This could not provide the load variability which may occur for different spills but suited the need of this study to ascertain accurate loadings to each lake.

For this study, all simulations were produced using SIMCAT version 15.7 and SAGIS version 3 (Add-In Release versions up to 1.0.8.1112.21765). The SIMCAT .dat files were amended, rather than directly altering the SAGIS model itself. Each SIMCAT .dat file was trimmed to the Windermere and Esthwaite Water sub-catchment reaches to reduce SIMCAT processing times. These reaches are described in Table A1 and visualised in Figure A1. Scripts written in Python were used to rapidly analyse scenarios for this study.

SIMCAT-SAGIS modelling only considered riverine nutrient inputs to the lakes and did not model processes occurring within the lakes themselves. For this study, this element was covered by PROTECH modelling.

Table A1: Reach descriptions for the trimmed SIMCAT file.

Reach Number	Waterbody ID	Name	Length (km)	Upstream area (km ²)
532	GB112073071420	Leven	1.04	2.72
533	GB112073071400	Cunsey Beck/Black Beck	5.14	1.70
534	GB112073071400	Cunsey Beck/Black Beck	1.06	1.70
535	GB112073071400	Cunsey Beck/Black Beck	6.17	5.10
536	GB112073071420	Leven	4.53	2.72
537	GB112073071130	Trout Beck (Kent/Leven)	13.50	1.95
538	GB112073071140	Rothay	6.32	1.59
539	GB112073071140	Rothay	3.40	1.59
540	GB112073071140	Rothay	1.26	1.59
541	GB112073071140	Rothay	1.24	4.78
542	GB112073071140	Rothay	3.61	1.59
543	GB112073071140	Rothay	0.24	7.96
544	GB112073071140	Rothay	0.76	1.59
545	GB112073071140	Rothay	5.80	11.14
546	GB112073071140	Rothay	6.15	1.59
547	GB112073071140	Rothay	1.88	14.33
548	GB112073071140	Rothay	6.72	1.59
549	GB112073071140	Rothay	0.05	17.51
550	GB112073071140	Rothay	1.17	20.69
551	GB112073071450	Brathay	7.85	2.14
552	GB112073071120	Great Langdale Beck	11.41	2.73
553	GB112073071450	Brathay	4.30	7.01
554	GB112073071420	Leven	1.53	30.43

Reach Number	Waterbody ID	Name	Length (km)	Upstream area (km²)
555	GB112073071420	Leven	3.90	2.72
556	GB112073071420	Leven	3.40	35.87
557	GB112073071420	Leven	1.47	40.54
558	GB112073071420	Leven	4.41	45.98
559	GB112073071420	Leven	8.26	53.80
560	GB112073071420	Leven	4.65	59.25

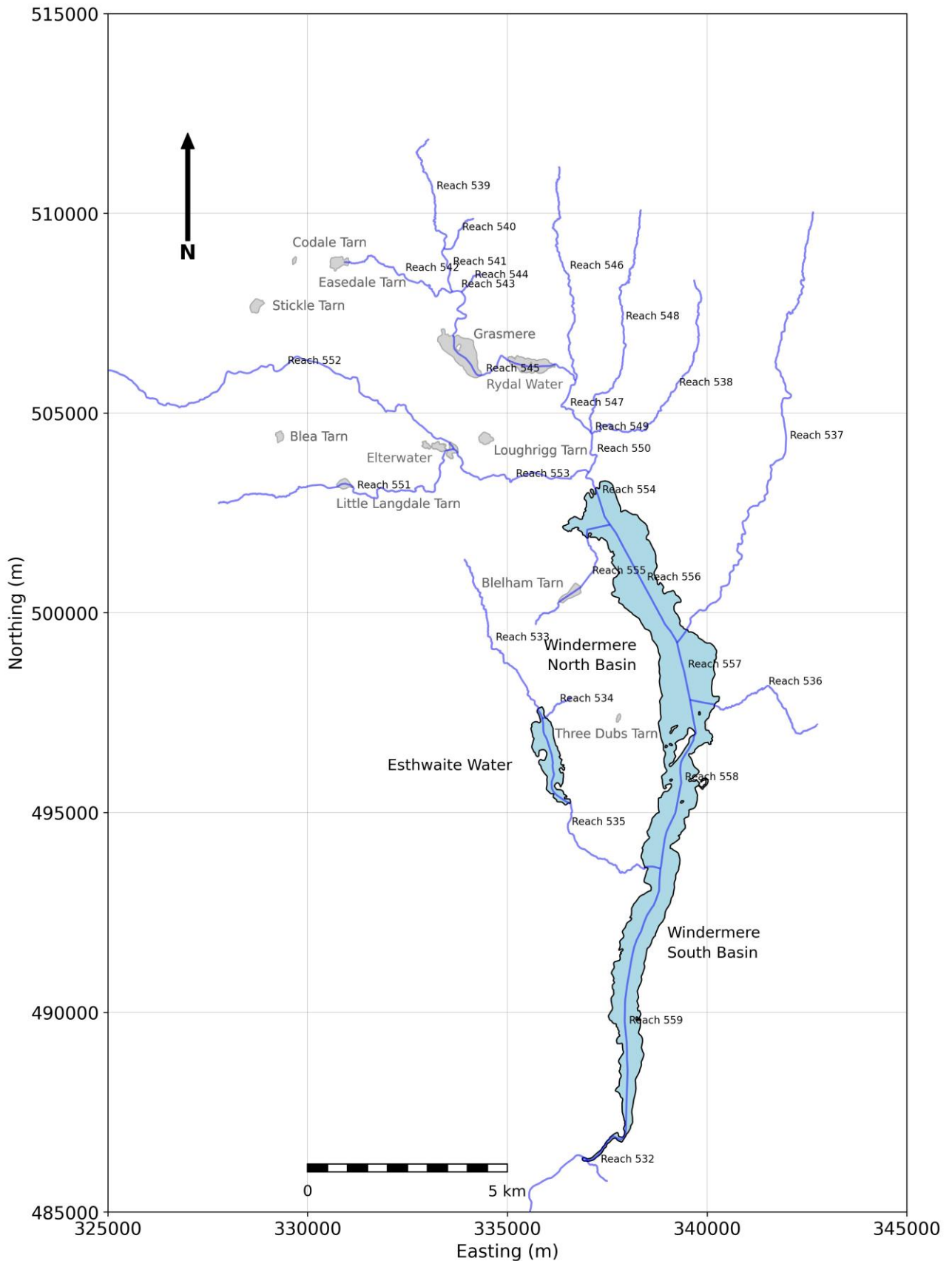


Figure A1: Visual representation of the reach structure in the trimmed SIMCAT file. In a rivers only model run, lakes are simulated as reaches. This did not affect the results of this study, as only the riverine inputs are provided by SIMCAT-SAGIS.

2. Appendix B: Baseline scenario

The following sections outline the collation, transformation, validation and modelling of the meteorological, flow and water quality data that underpinned the baseline scenario.

2.1 Meteorological datasets

2.1.1 Air temperature and wind speed

The three meteorological inputs to PROTECH were air temperature, wind speed (at the lake surface), and cloud cover. For air temperature and wind speed, data from meteorological sondes placed on the lake surface at Windermere and Esthwaite Water by the UK Centre for Ecology & Hydrology (UKCEH) were chosen as the best representation of local conditions, as they were sited at the exact location the PROTECH model was simulating. Alternative data sources included the HadUK-Grid dataset (HadUK) – Met Office gridded weather observations, or Met Office Integrated Data Archive System (MIDAS) weather observation stations, but they were less spatially resolved and so less able to capture local variations in conditions. In addition, the sonde datasets provided solar radiation data alongside air temperature and wind speed, to act as a validation for cloud cover measurements.

- Windermere: Air temperature, wind speed, and solar radiation (measured by pyranometer) were recorded hourly at a water monitoring sonde located on Windermere South Basin. Two datasets were provided which covered two separate periods: between 2012–2015 (Jones and others, 2017a) and between 2016–2018 (Feuchtmayr and others, 2021). There was a large gap in the dataset for wind speed at the start of the latter record (01/01/2016 to 14/03/2016). It is assumed that conditions for Windermere South Basin were true for Windermere North Basin.
- Esthwaite Water: Air temperature, wind speed, and solar radiation (measured by pyranometer) were recorded hourly at a water monitoring sonde located on Esthwaite Water between 2012-2015 (Jones and others, 2017b). There was one notable gap (14/07/2015 to 20/07/2015) where no data for any variable was recorded. In addition, air temperature was not recorded between 14/12/2015 and 31/12/2015 and on the 03/06/2014. Wind speed was not recorded on the 27/11/2014. There was no data available for the 2016–2018 period.

A daily mean value was determined for each parameter from hourly measurements taken by each sonde (Figure B1). To prevent biasing during this calculation, if data was not available for the full 24-hour cycle, data from that day was omitted. The only apparent lack of consistency between the two periods of available data for the Windermere South Basin sonde was for wind speed, where there were many near-zero values towards the end of 2015 (Figure B1). Besides this, there were no large systematic differences between the two datasets.

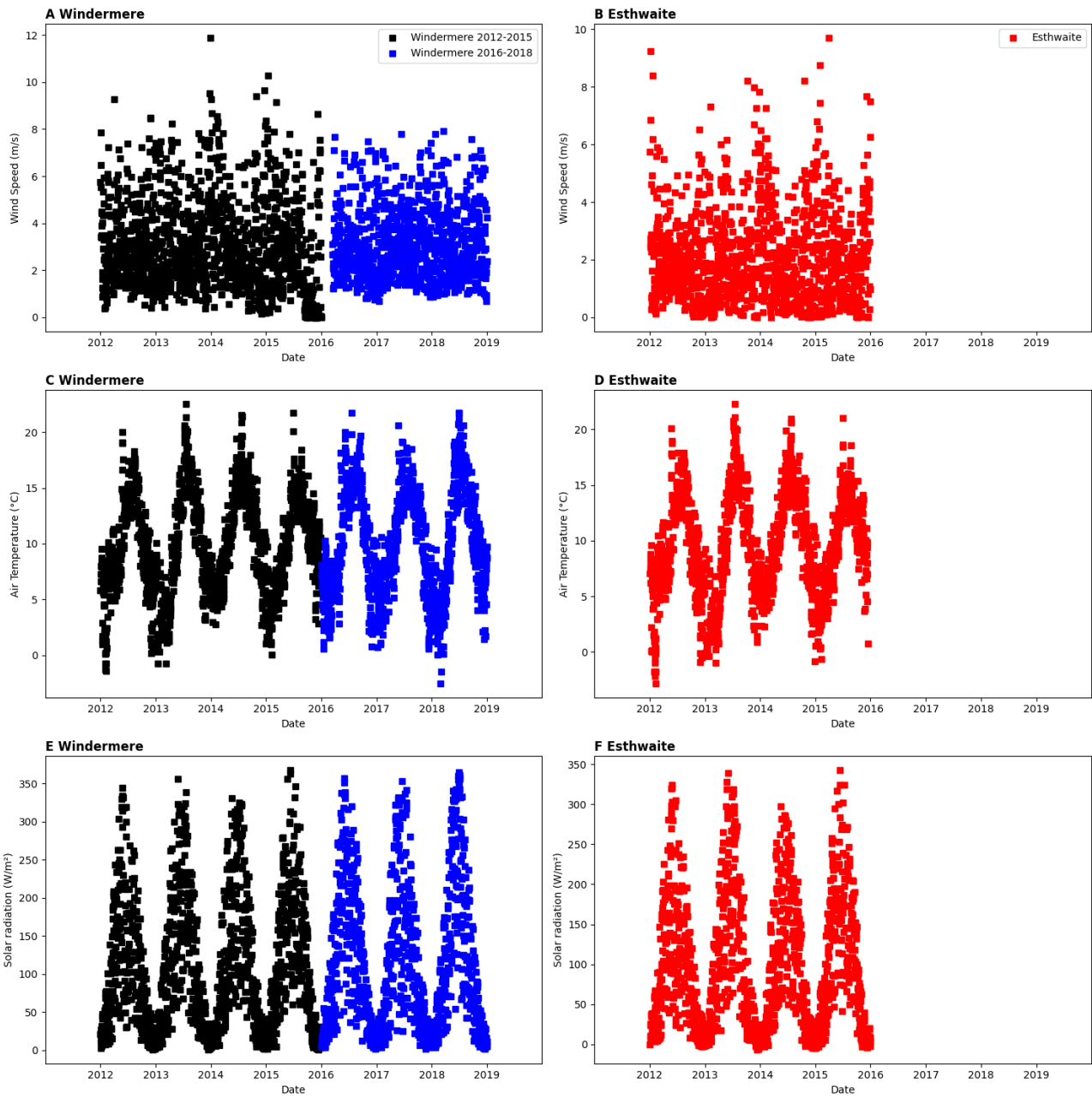


Figure B1: Meteorological data recorded by sondes situated on Windermere South Basin (Windermere) and Esthwaite Water (Esthwaite). Windermere South Basin data is split into the earlier (2012-2015) (black) and later (2016-2018) (blue) datasets.

There was no data available from the Esthwaite Water sonde between 2016-2018. Given the proximity of Windermere South Basin and Esthwaite Water (Figure A1) and the contemporarily sampled datasets in the earlier period, the later Windermere South Basin dataset was used to project meteorological conditions at Esthwaite Water from 2016-2018. Linear regression models were used to define the relationship between measurements recorded at the same time from each sonde at Windermere South Basin and Esthwaite Water between 2012-2016 (Figure B2).

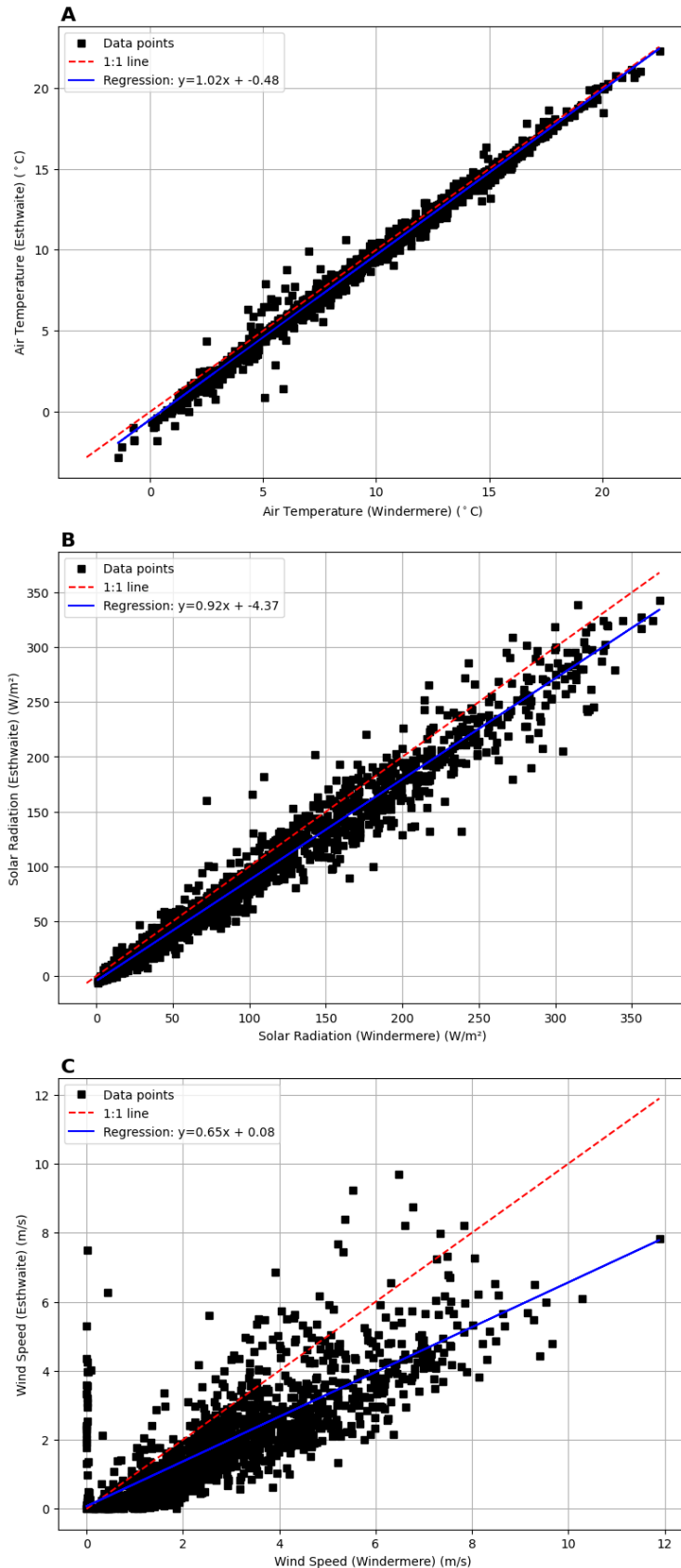


Figure B2: Relationship between coincident measurements recorded by meteorological sondes at Windermere South Basin (Windermere) and Esthwaite Water (Esthwaite) for (A) air temperature, (B) solar radiation, and (C) wind speed. Blue lines indicate the linear regression relationship and red lines indicate the 1:1 line.

Linear relationships were well correlated for air temperature ($r^2 = 0.99$) (Figure B2, A) and solar radiation ($r^2 = 0.96$) (Figure B2, B) and the offset could be determined. Air temperatures at Windermere South Basin were systematically ~ 0.5 °C higher, and solar radiation was typically greater on a given day, than at Esthwaite Water. The correlation for wind speed between coincident measurements was weaker ($r^2 = 0.60$) (Figure B2, C). Windier days appeared to have greater absolute differences between the lakes than stiller days and Esthwaite Water experienced, on average, lower wind speeds than Windermere South Basin. In addition, defining a linear regression was complicated by several samples with high wind speed at Esthwaite Water, but near-zero wind speed at Windermere South Basin. To address this, Density-Based Spatial Clustering of Applications with Noise (DBSCAN) clustering was used to identify outliers in the data (Figure B3, A). When removed, the r^2 of the correlation increased to 0.74 and more accurately reflected the lower end of measurements (Figure B3, B).

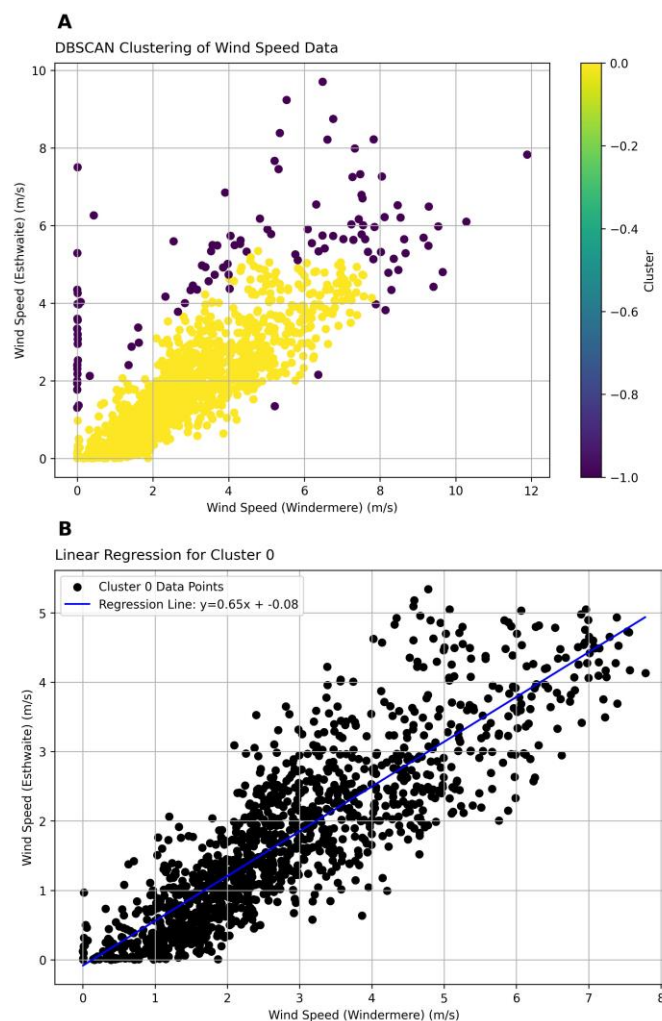
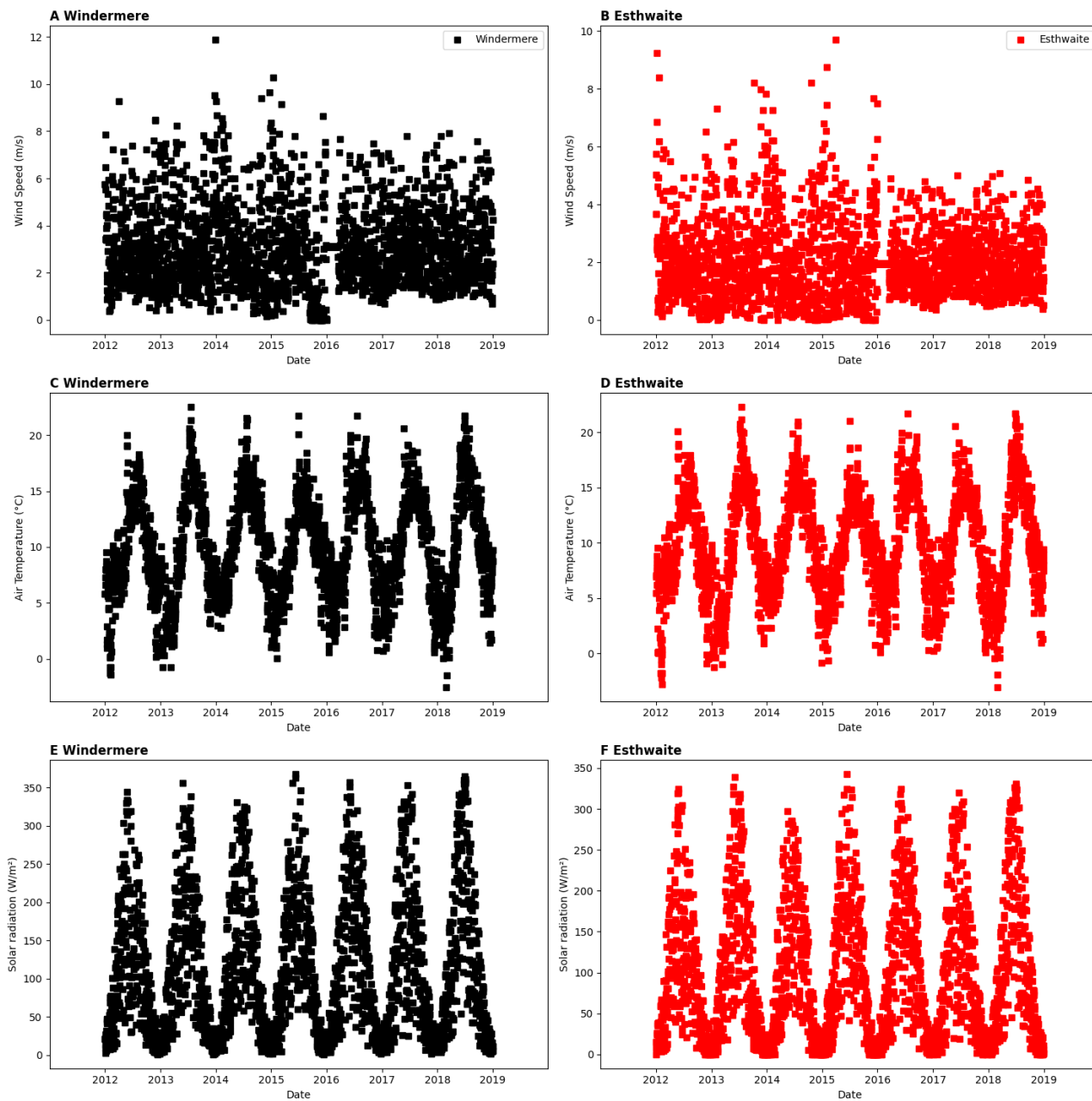


Figure B3: Wind speed data for the study sub-catchments. (A) DBSCAN clustering of the wind speed relationship (epsilon = 0.30; min samples = 15) to identify outliers in the data for Windermere South Basin (Windermere) and Esthwaite Water (Esthwaite). Clustering was optimised to pick up the left-hand side line of outliers. (B) Regression model of cluster 0 to identify the regression relationship when outliers were removed. The blue line provides the regression relationship.

These relationships (Figure B3) were used to project Esthwaite Water conditions from 2016 onwards based on Windermere South Basin measurements for that period. The final corrected measurements are presented in Figure B4. Backwards gap-filling was applied to fill any temporal gaps in the dataset. One notable issue with this approach can be seen in wind speed measurements at the start of 2016, where static wind speeds were recorded for ~2.5 months due to no observed wind speed data (Figure B4).

Figure B4: Corrected meteorological measurements following the data processing steps for Windermere South Basin (Windermere) and Esthwaite Water (Esthwaite).



2.1.2 Cloud cover

Cloud cover was determined using data extracted from a nearby open MIDAS station. Neither meteorological sondes at Windermere South Basin or Esthwaite Water, nor the meteorological station at Esthwaite Water (owned by UKCEH) recorded cloud cover, although both sondes recorded solar radiation (W/m^2). HadUK gridded data only provided number of sunshine hours for a monthly, not daily, period.

Of the currently active MIDAS stations (all others closed prior to 2008), the Morecambe No 2 and Hazelrigg stations both had sunshine hours recorded for the 2014-2018 period, a variable from which cloud cover can be estimated (Table B1). As the closest station where data were available, the sunshine hours recorded (measured by instrumentation other than Campbell Stokes recorders) from Morecambe No 2 were taken to be representative of Windermere and Esthwaite Water conditions. This station, situated on the southern side of Morecambe Bay, is still ~25 km away from the Windermere catchment, hence local variability in cloud cover was a source of uncertainty in these projections.

Table B1: Location of currently active MIDAS stations, their open data and its availability for this study. Sites ordered in descending direct distance from the Windermere catchment.

Open MIDAS station name	Latitude	Longitude	Data available
Levens Hall	54.0138	-2.7737	Data existed on the Centre for Environmental Data Analysis (CEDA) only up to 2011. Could not be used.
Walney Island	54.2587	-2.7771	Data existed on CEDA, but all sunlight hour data was not available. Could not be used.
Morecambe No 2	54.0762	-2.8583	Data existed on CEDA for the baseline period (2014-2018). Could be used.
Hazelrigg	54.1247	-3.2566	Data existed on CEDA for the baseline period (2014-2018). Could be used.

An estimate of cloud cover was projected by comparing the 24-hour sunlight duration to the maximum daylight hours as projected by the National Oceanic and Atmospheric Administration (NOAA) calculators (NOAA, 2025) (accurate to 1 minute at latitudes less than 72°) (Figure B5).

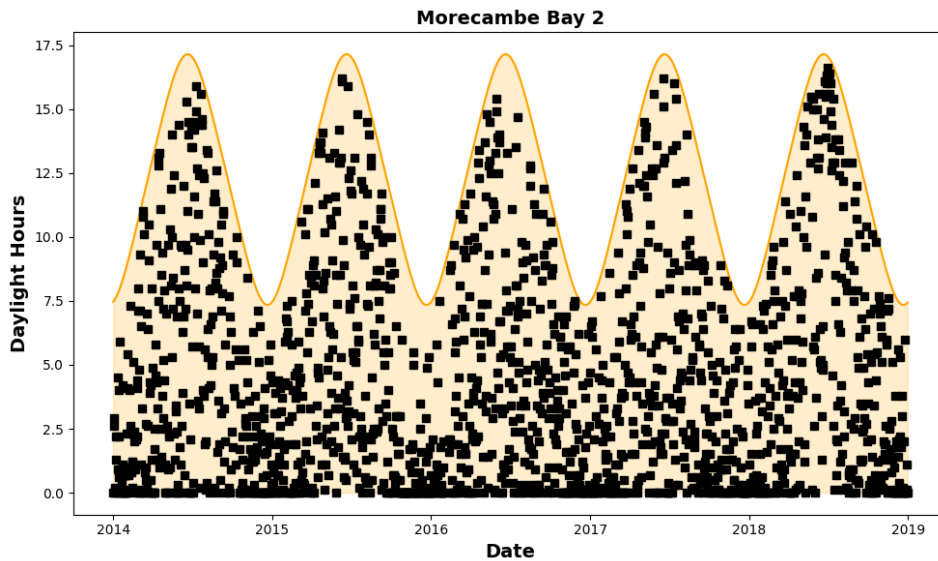


Figure B5: Total daylight hours. Calculated by the Astral library on Python, built upon the NOAA calculators to determine sunrise and sunset times for the latitude and longitude at Morecambe No 2 MIDAS station (orange line). This is overlaid with the duration of sunlight hours recorded at the site (black squares), and the maximum daylight hours (orange line) (see Equation B1).

The implicit assumption from estimating fractional cloud cover from sunlight duration was that over a daily period, cloud cover will have inhibited sunlight for time proportional to the fraction of clouds in the sky. For days where there was 100% cloud cover and no sunlight hours, this was likely to be well predicted. At other times, cloud cover may have been poorly estimated using this method for the following reasons:

- Thin clouds (e.g., cirrus) may permit sunlight to pass through
- The sun position may permit sunlight to reach the sensor even with clouds overhead
- Clouds may be at different atmospheric heights, impacting sunlight hours differently

Percentage cloud cover (CC) was estimated using sunlight hours via Equation B1.

$$CC = \left(1 - \left(\frac{\text{Sunlight hours}}{\text{Max daylight hours}} \right) \right) * 100 \quad \text{Equation B1}$$

This was then converted to oktas using the classification boundaries provided in Boers and others (2010). In this classification, okta 0 and okta 8 should correspond to a

completely clear day (0% cloud cover) and completely overcast day (100% cloud cover) respectively, so oktas 1 and 7 had slightly larger bin sizes to account for this. To test the assumptions in this conversion, solar radiation as measured by a pyranometer on a meteorological sonde at Windermere South Basin was compared with the estimated cloud cover for each date in the baseline period (Figure B6). Lower cloud cover was associated with higher solar radiation, as would be expected, but there was significant variation between different cloud oktas and the amount of solar radiation recorded, which may be due to some of the assumptions listed above. There were no measurements of 0% cloud cover. Whilst a single cloud would negate a day being classed as okta 0, a completely clear day would be expected over a 5-year period, and so this was likely due to this methodology (i.e., daily lengths as projected by the NOAA calculators may have been marginally too high).

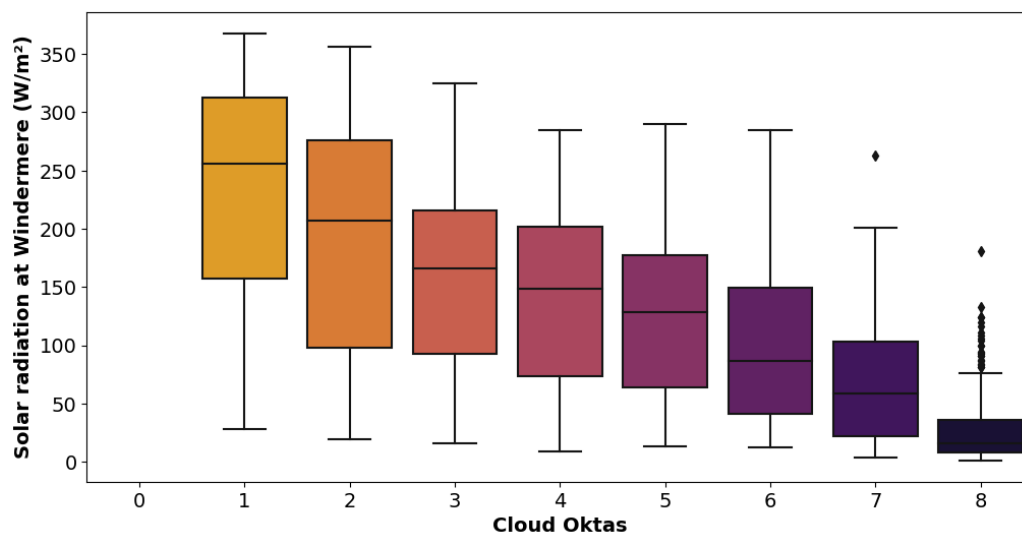


Figure B6: Comparison between the co-temporal cloud cover categories (oktas) from Morecambe No 2 MIDAS station and the solar radiation measurements recorded from the sonde at Windermere South Basin (Windermere).

2.2 Flow datasets

Time-series flow data for the baseline period were extracted from the Qube platform (Wallingford HydroSolutions, 2025b). The location of the defined input catchments for Windermere North Basin, Windermere South Basin and Esthwaite Water, and how these were combined to form a PROTECH discharge input, are described below.

2.2.1 Windermere North Basin

There are five specified input sub-catchments which flow into Windermere North Basin (Figure B7). These are the River Brathay, River Rothay, Blelham Beck, Mill Beck and Trout Beck. National River Flow Archive (NRFA) gauging stations existed on three of these (River Rothay, River Brathay, and Trout Beck), so flow estimates could be compared to observed discharge data (Figure B8). There was a close fit between gauged discharge

data and estimates for the River Brathay and River Rothay. Qube overestimated flows for Trout Beck by ~ 35% (particularly at the higher end of flow values). The River Rothay and River Brathay meet at a confluence before entering Windermere North Basin, but for apportioning of flow and water quality, the two rivers were separated to determine sub-catchment inputs.

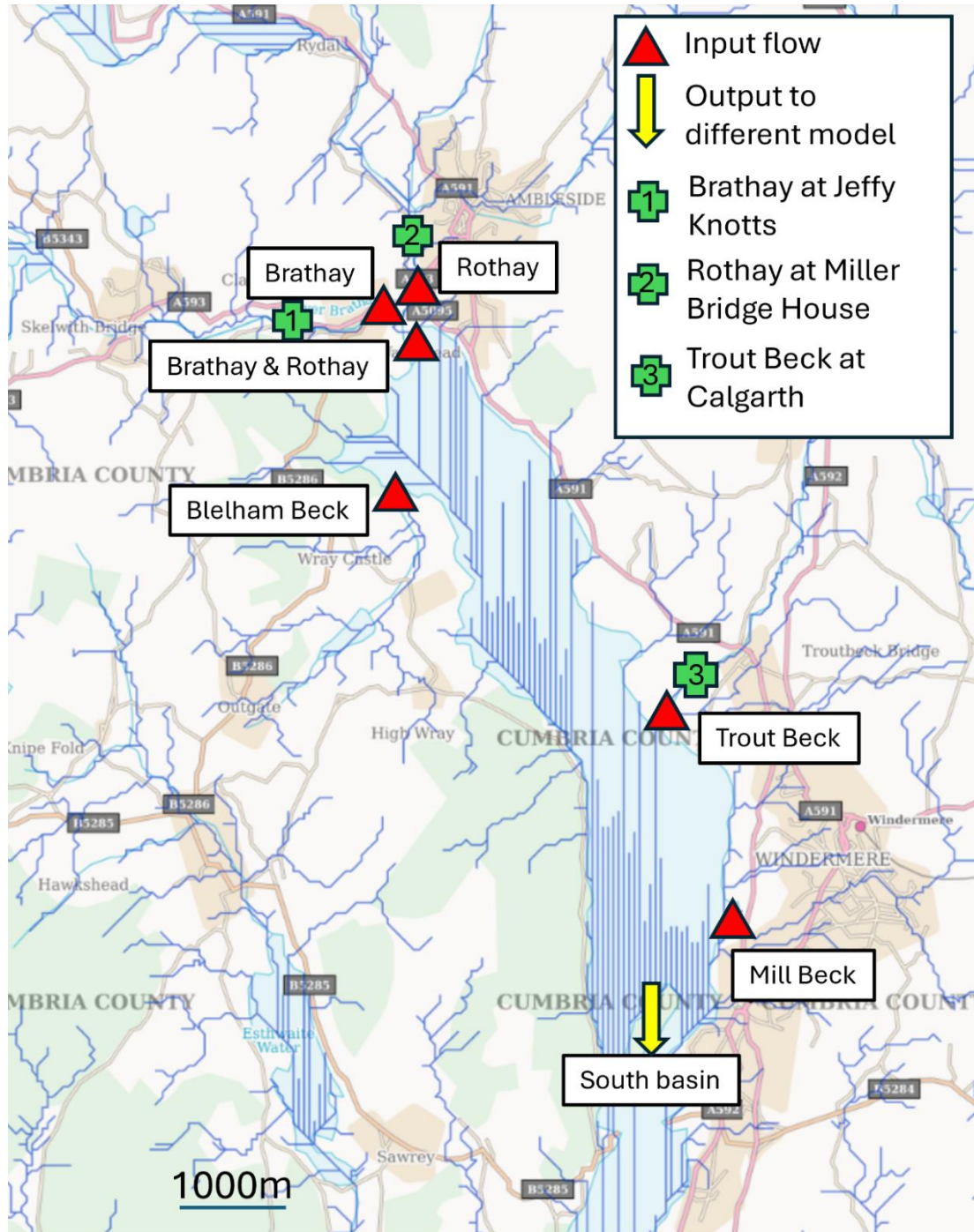


Figure B7: Map of Windermere North Basin as modelled by Qube. Blue lines represent the drainage network using the Continuous Estimation of River Flows (CERF) model. Red triangles indicate the location of the input sources, from which the sub-catchment above is used to estimate the flow characteristics. The yellow arrow indicates the output to the Windermere South Basin model. Green pluses indicate NRFA flow gauges.

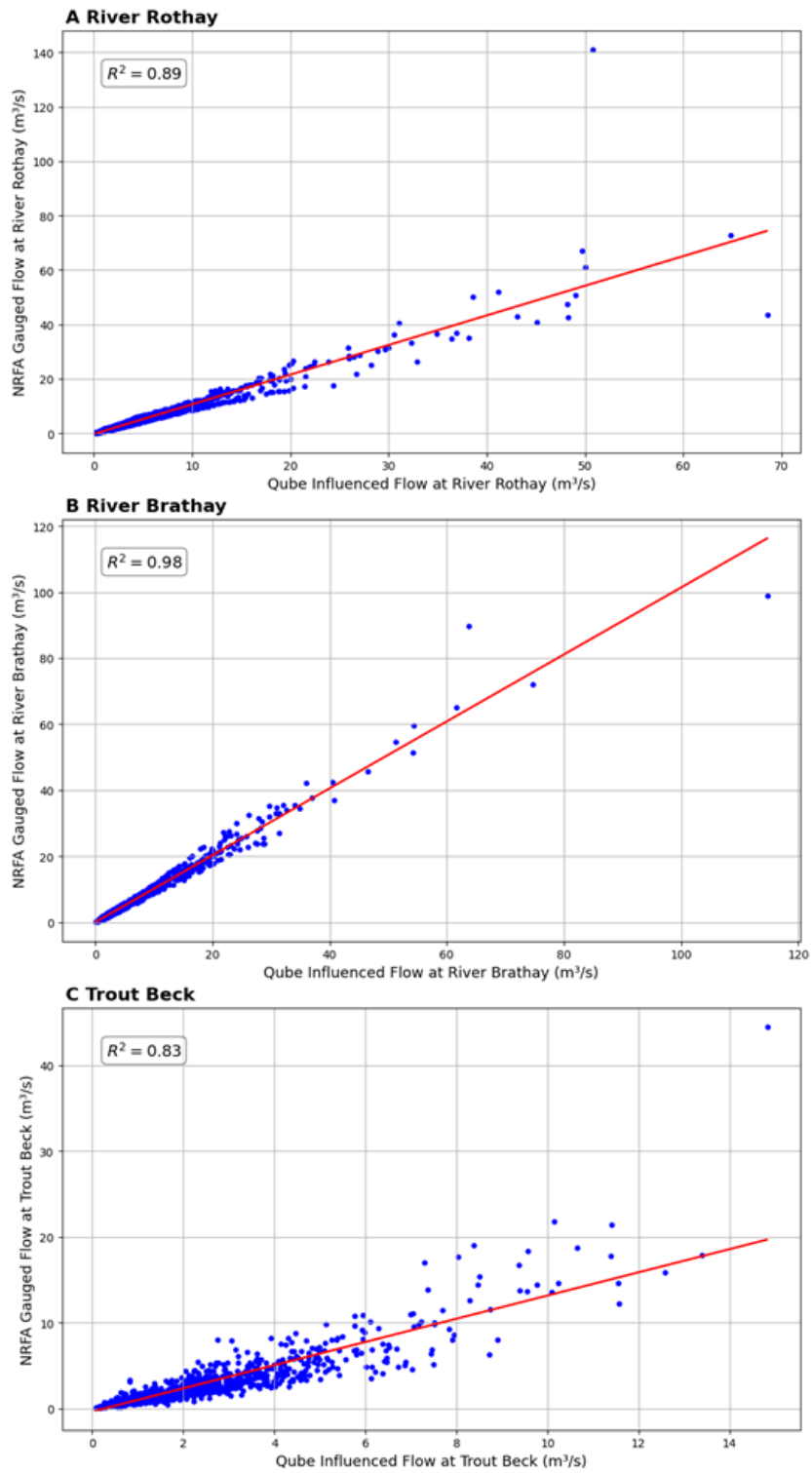


Figure B8: Comparison of flow gauge discharge data (NRFA) with Qube estimates between 2014 and 2018 for (A) the River Rothay, (B) the River Brathay, and (C) Trout Beck. Red lines represent best fit linear regressions.

The River Rothay and River Brathay (Windermere North Basin) provide most of the flow to Windermere, followed by Trout Beck (Windermere North Basin) and Cunsey Beck (Windermere South Basin) (Figure B9). Outflows can sometimes be greater than inflows for a given month. This is due to a residence time of ~280 days for Windermere (Moorhouse and others, 2018). This, combined with lack of a flow gauge estimate for the Windermere North Basin exit, mean an outflow equals inflow assumption cannot be applied, and so each of the Windermere North Basin riverine influent discharge data were combined to produce the Windermere North Basin PROTECH model flow input.

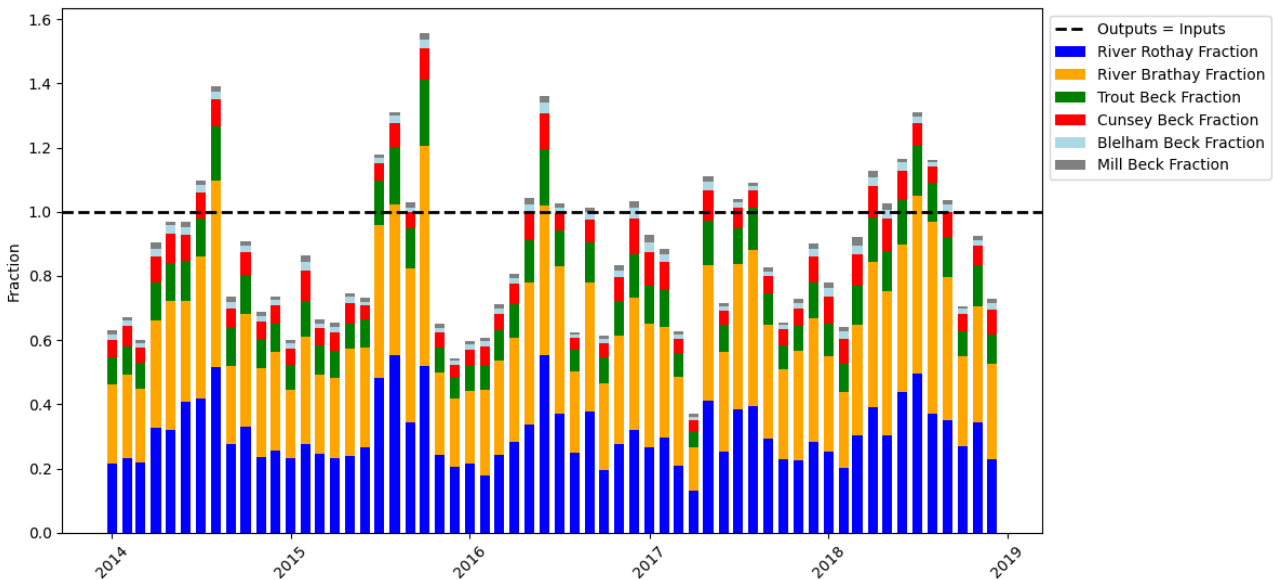


Figure B9: River fractions of input flows into Windermere relative to the total outflow, averaged over each month. The dashed line indicates a situation where the inputs = outputs.

2.2.2 Windermere South Basin

The discharge from Esthwaite Water and Windermere North Basin PROTECH models provided the two largest flow inputs into Windermere South Basin. These were added to two additional discharges, (1) the discharge acquired by Cunsey Beck downstream of the Esthwaite Water outflow, and (2) the discharge provided by the Windermere Wastewater Treatment Works (WwTW) (Figure B10).

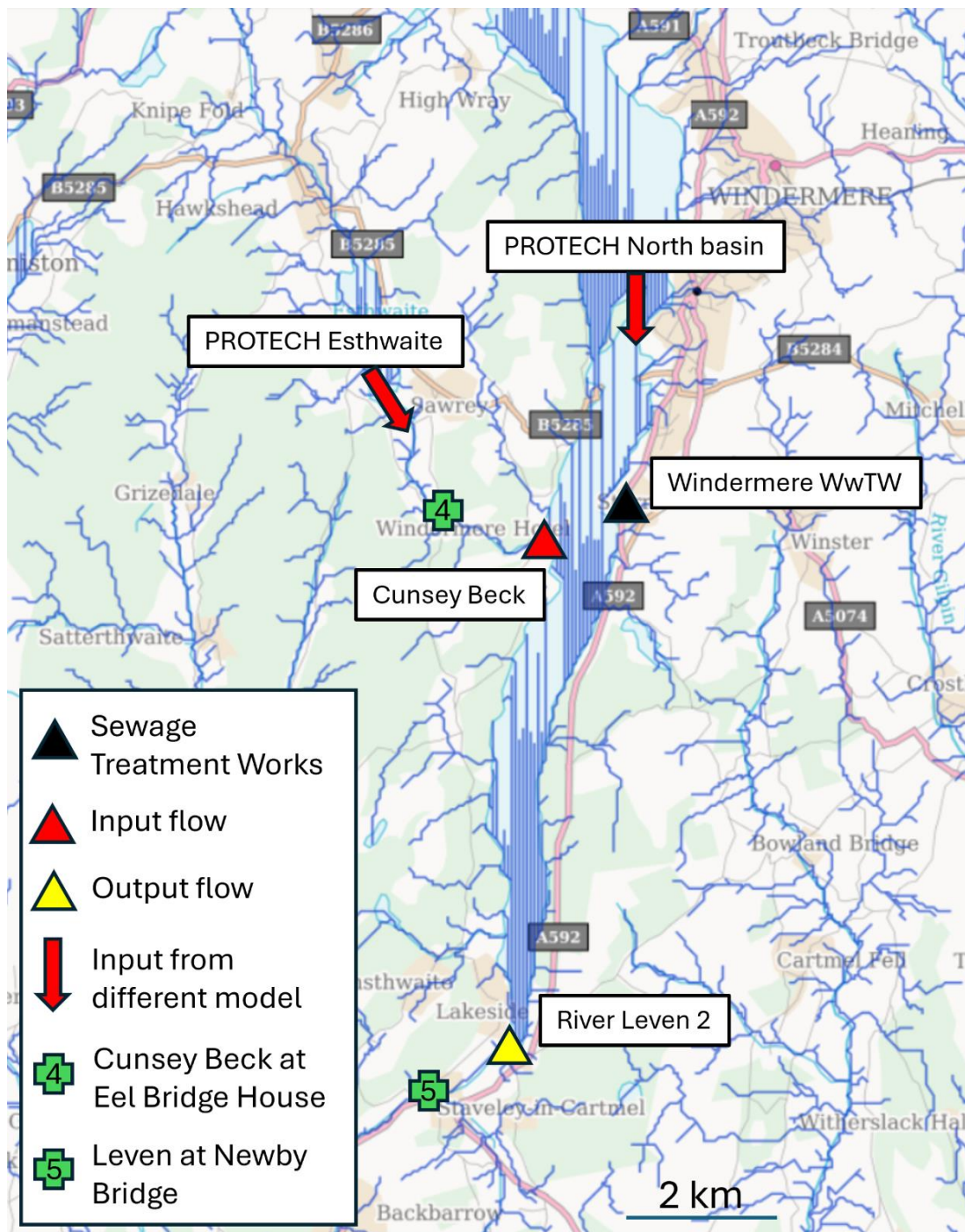


Figure B10: Map of Windermere South Basin as modelled by Qube. Blue lines represent the drainage network using the CERF model. The red triangle and arrows represent input Qube discharge data and PROTECH model output discharge data respectively. The yellow triangle indicates the output flow. Green plusses indicate NRFA flow gauges. The black triangle shows the addition of the Windermere WwTW.

To isolate the Cunsey Beck component, the contemporaneous discharge measurements at the Esthwaite Water outflow were subtracted from the discharge measurements at the input point to Windermere South Basin (Figure B10, red triangle). This addition contributed between 10% and 20% of the total discharge from Cunsey Beck, the other fraction coming

from Esthwaite Water. On a daily timescale, there could be significant variation in this difference, with the discharge difference over Cunsey Beck sometimes being negative for a given day (Figure B11). This occurred due to the time taken for upstream weather events to flow down Cunsey Beck, which may have temporarily led to a higher discharge upstream than downstream. The average flow difference over the 5-year period was positive (0.12 m³/s). This additional Cunsey Beck discharge value was added to the Windermere WwTW daily time series. This is the largest WwTW in the area with a dry weather flow permit limit of 5559 m³/d.

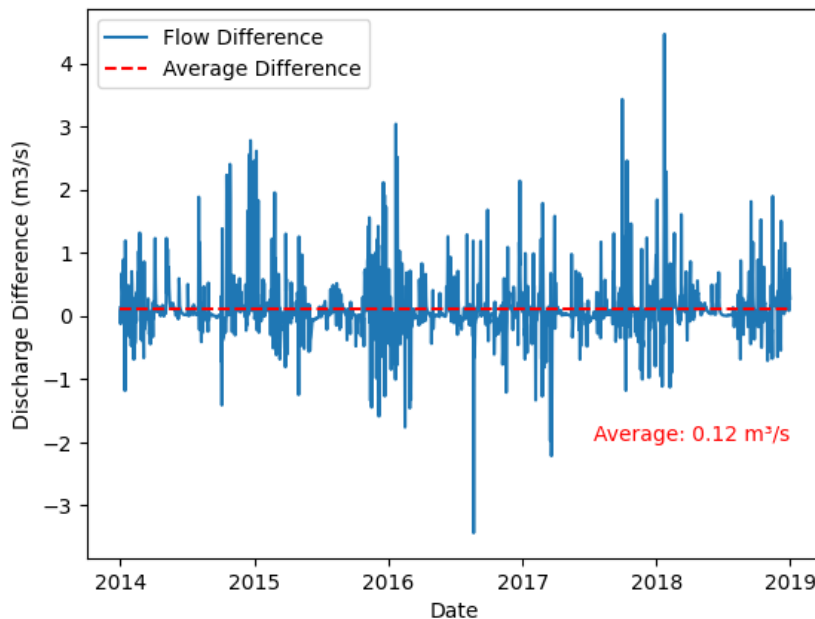


Figure B11: Daily discharge difference between the point of entry to Windermere South Basin and the outlet point from Esthwaite Water, for Cunsey Beck. The average value (red) is positive, but there are significant daily variations around this mean.

The discharge estimates extracted from Qube did not accurately replicate discharge measured at Eel Bridge House (NRFA gauging station) on Cunsey Beck, broadly overestimating lower flows and underestimating higher flows (Figure B12). In addition, smaller tributaries were not well characterised for this sub-catchment. This may explain why the outflow discharge from the lake (Figure B10, River Leven 2) as estimated by Qube is considerably lower than the observed discharge data (Figure B12).

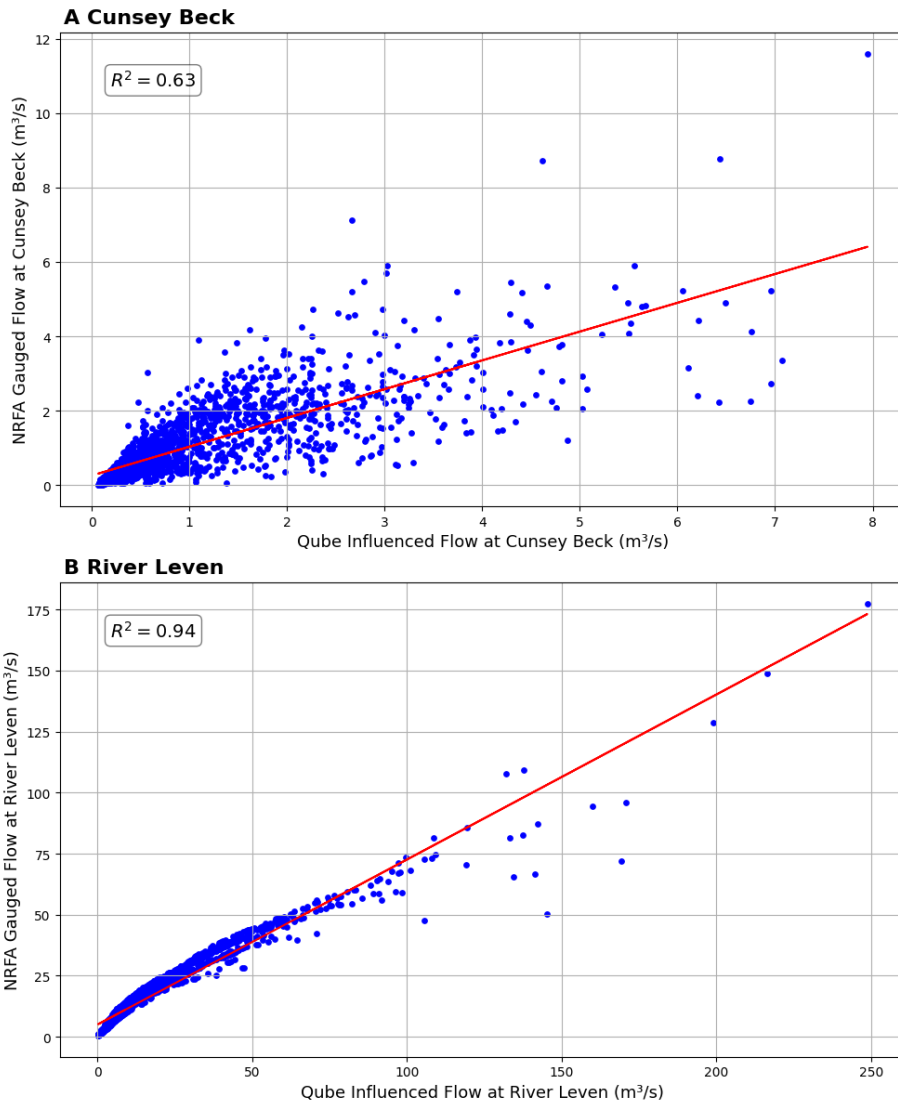


Figure B12: Comparison of flow gauge discharge data (NRFA) with Qube estimates between 2014 and 2018 for (A) Cunsey Beck and (B) the River Leven (Newby House) downstream of the Windermere outflow. Red lines represent best fit linear regressions.

2.2.3 Esthwaite Water

Five input sub-catchments were selected for Esthwaite Water in Qube (Figure B13). Black Beck contributed the largest fraction ($52 \pm 10\%$) of the inflow each month (Figure B14). This aligned with Maberly and others (2011) who reported a 55% contribution. The other four specified inputs contributed smaller flow fractions, less than 10% each.

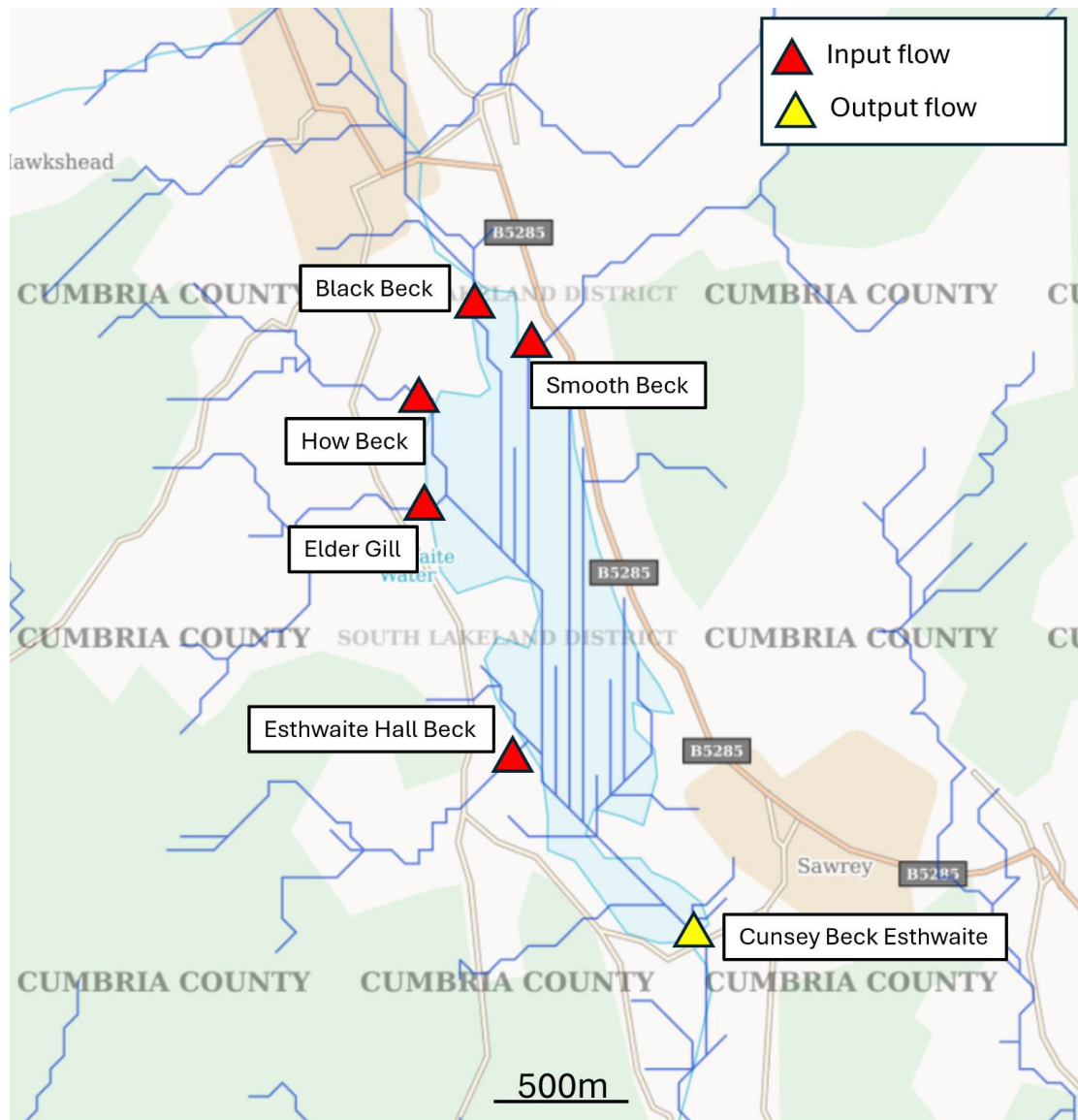


Figure B13: Map of Esthwaite Water as modelled by Qube. Blue lines represent the drainage network using the CERF model. Red triangles indicate the location of the input sources, from which the sub-catchment above is used to estimate the flow characteristics. The yellow triangle indicates the output flow.

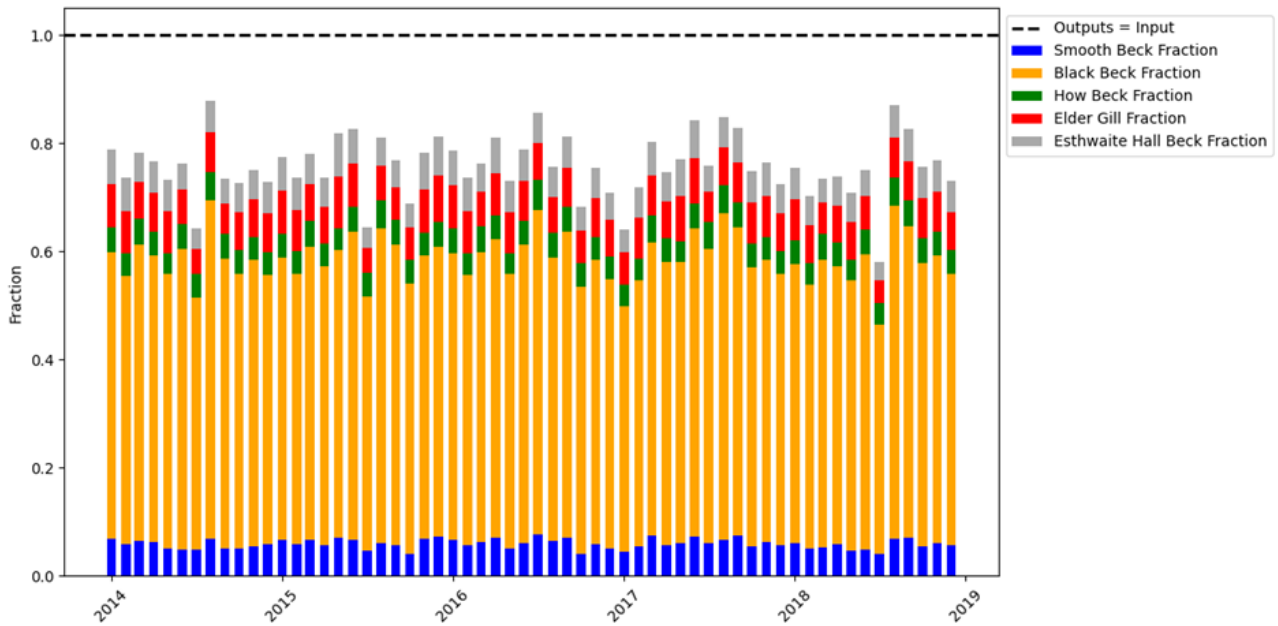


Figure B14: River fractions of input flows into Esthwaite Water relative to the total outflow, averaged over each month. The dashed line indicates a situation where the inputs = outputs.

Comparison of the Qube estimates for dates where comparable *in situ* measurements were taken by Anderson (2018) highlighted the capability of the model to reproduce inputs satisfactorily (Figure B15). Two major outliers in the winter of 2013/2014 where *in situ* measurements were much greater than the Qube estimates may reflect the difficulty in modelling very localised rainfall events which produce high river flows. There was no NRFA gauge on Black Beck to further validate inflow data.

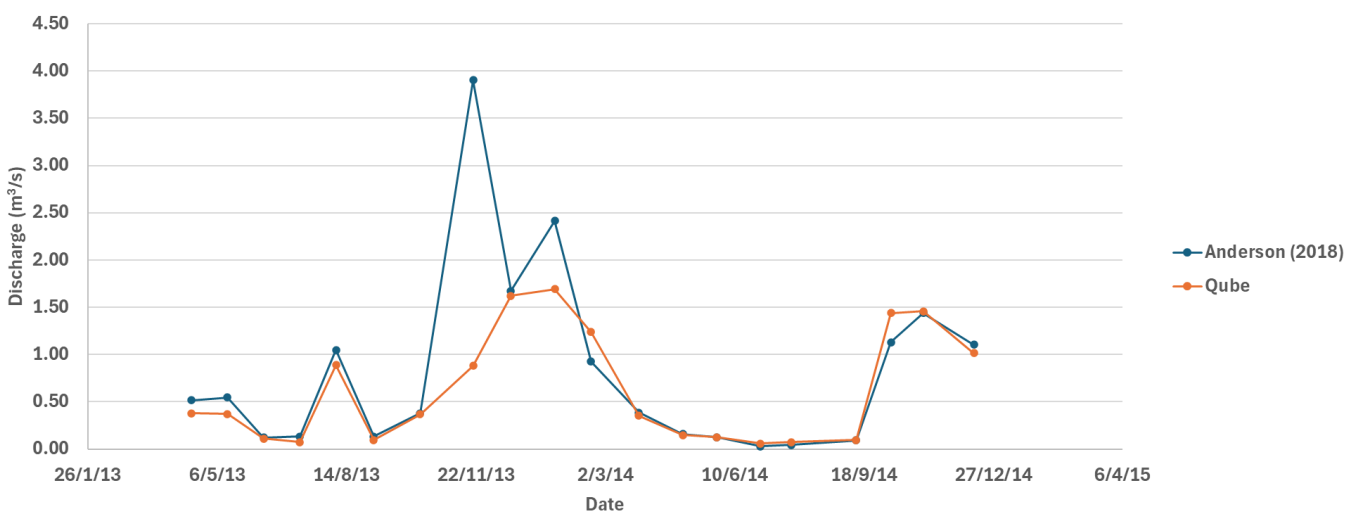


Figure B15: Comparison between Qube estimates of inflow to Esthwaite Water with *in situ* measurements reported in Anderson (2018).

The combined inflows systematically correspond to less than the outflow discharge recorded at Cunsey Beck (by roughly 20% - 40%) across all months, suggesting that there were other sources of flow (e.g., non-riverine runoff, groundwater flow, effluent, or direct precipitation) that were not captured. To model the total inflow in PROTECH, the long-term assumption was that outflow equals inflow (or the lake would change volume with time) and so discharge at the outflow was used as the integrated input discharge. This was a valid assumption given that Esthwaite Water has a short residence time of ~86 days, based on a volume of $6.7 \times 10^6 \text{ m}^3$ (Mackay and others, 2012) and a mean outflow discharge of $0.898 \text{ m}^3/\text{s}$ (2014-2018 Qube estimate).

2.2.4 Final discharge plots

The input discharge data used in the baseline PROTECH model runs for Windermere North Basin flows, Windermere South Basin (Cunsey Beck) flows, and Esthwaite Water output flows are provided in Figure B16, Figure B17 and Figure B18, respectively.

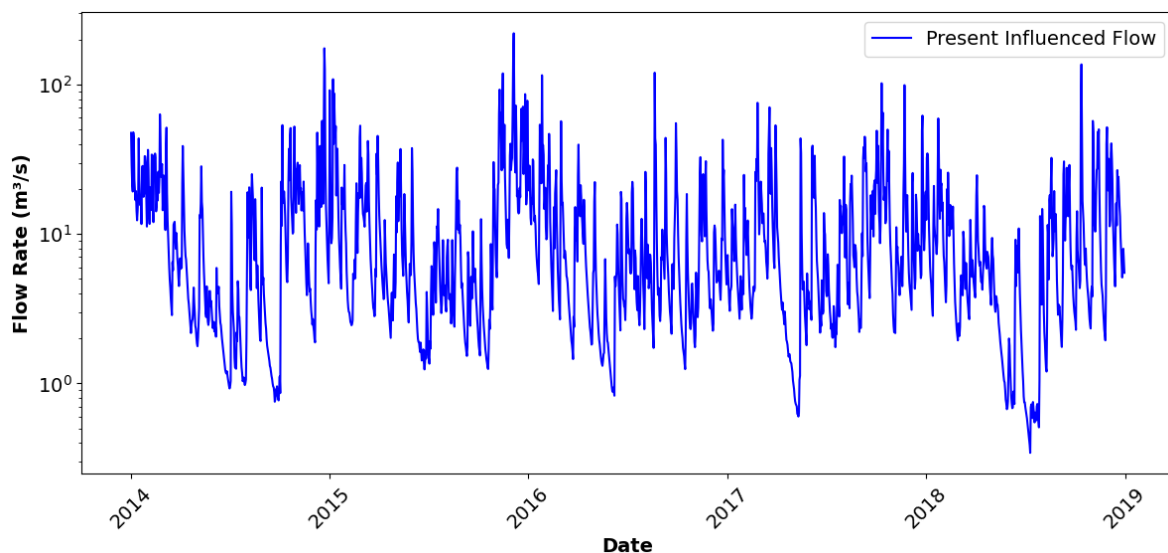


Figure B16: Windermere North Basin inflow discharge extracted from Qube. This was the Windermere North Basin input discharge data for PROTECH. Note the log scale.

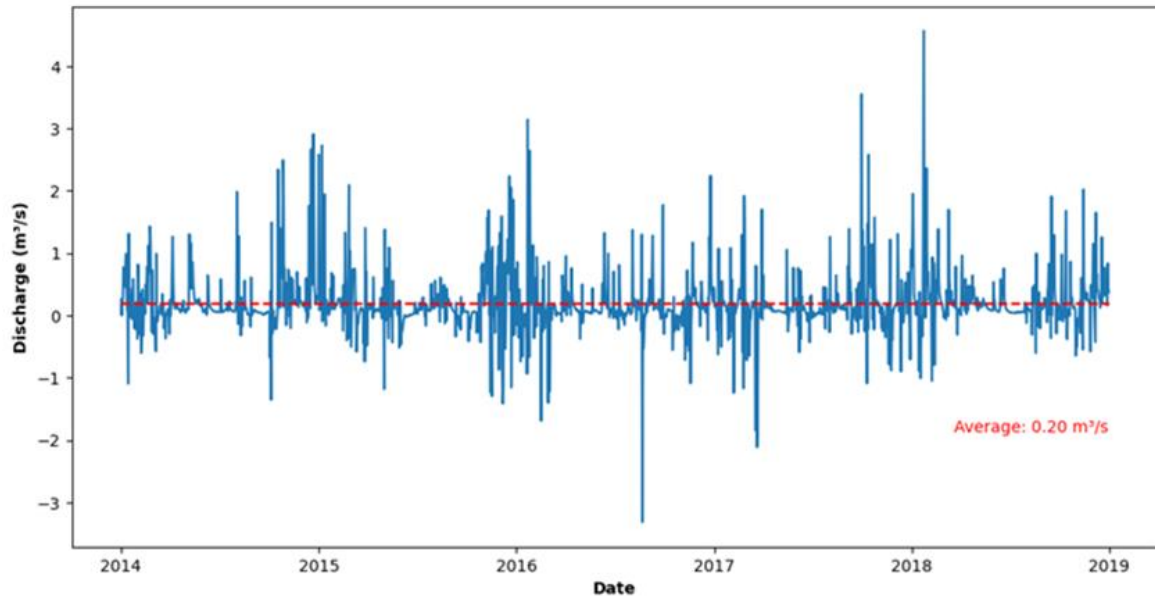


Figure B17: Windermere South Basin inflow discharge extracted from Qube. This was the Windermere South Basin input discharge data alongside any flow output from the Windermere North Basin PROTECH model.

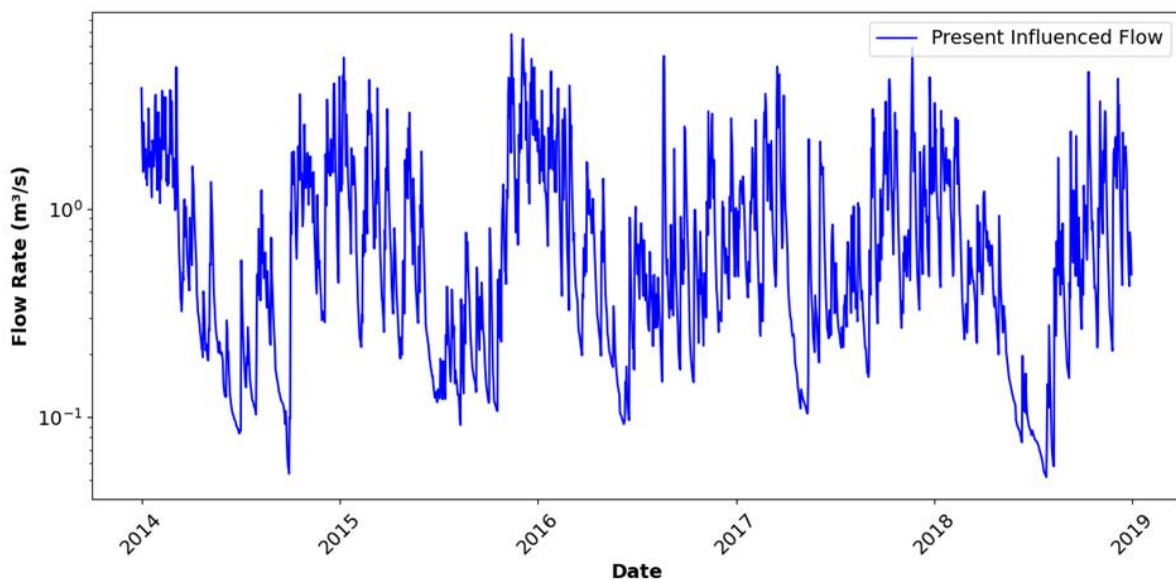


Figure B18: Esthwaite Water outflow discharge extracted from Qube. This was the Esthwaite Water input discharge data to PROTECH. Note the log scale.

2.3 Water quality datasets

Nutrient inputs required for PROTECH were soluble reactive phosphorus (SRP), nitrate as N (hereafter referred to as nitrate), and dissolved SiO₂ (hereafter dissolved silica). As comparison between nutrient definitions can be complex, a brief description of each of these nutrient inputs and the forms used in this study is provided below:

There are numerous forms of phosphorus which are considered in environmental monitoring. Three of the most used in regulation are:

- **Total phosphorus (TP)**: A measure of all dissolved and insoluble phosphorus in the water sample. This includes inorganically and organically bound phosphorus. This is the form of phosphorus that is part of the Water Framework Directive (WFD) classification for English lakes.
- **Orthophosphate**: Orthophosphate is the ionic form of phosphate (PO_4^{3-}) as dissolved in water. In eutrophication science, only the phosphorus component of the ion is relevant and therefore measures of phosphate are corrected for the molecular mass of the P atom and referred to as phosphate-phosphorus (PO_4^{3-} -P).
- **Soluble reactive phosphorus (SRP)**: A measure of phosphate-phosphorus using the phosphomolybdenum blue colorimetric method. It is measured on a filtered sample to ensure P in a particulate form is excluded from the analysis.
- **Reactive phosphorus**: The term used to describe the WFD classification for phosphate-phosphorus in English rivers. This is a reasonable approximation of immediately bioavailable phosphorus (UKTAG, 2013). Reactive phosphorus is determined using the same phosphomolybdenum blue colorimetric method as for SRP but carried out on an unfiltered sample. It is the form of phosphorus most often measured in Environment Agency river sampling programmes and when analysed by the Environment Agency National Laboratory Service this analysis has a determinand code of 0180 Orthophosphate, reactive as P. In turn, reactive phosphorus is often referred to as orthophosphate. Throughout the Nutrient Futures for Windermere report and these appendices we refer to reactive phosphorus (determinand 0180) as phosphate. .

The input required for PROTECH was SRP, whilst the forms of phosphorus modelled in the river inflows to each lake in the upstream catchment model (SIMCAT-SAGIS) were reactive phosphorus and TP. For this study, reactive phosphorus (termed phosphate throughout) was used as it is more like SRP than TP, and the assumption was that SRP and phosphate were equivalent. There are likely to be minor differences between these measures of phosphorus (Mainstone and others, 2000).

Nitrate as N: The ionic form of nitrogen (NO_3^-) where nitrogen is in the +5-oxidation state, corrected for the molecular mass of the N atom. It is typically only present in oxygen-rich waters, or else the nitrogen is liable to reduction. Nitrate is normally not considered (in regulatory situations) to be limiting in freshwater environments, although a growing body of evidence is highlighting the previously underestimated importance of this nutrient in lake phytoplankton growth (Helmholtz Centre for Environmental Research, 2024). The same metric was used in both the PROTECH model and upstream catchment model (SIMCAT-SAGIS).

Dissolved silica: This study used the concentration of dissolved silica (SiO_2) in a sample, the primary form of silicon (S) in water. Dissolved silica in rivers is sourced from the weathering of silicate rocks and dissolves in solution as a function of temperature, in addition to pH conditions greater than ~9 (Neal and others, 2005). Silica comprises the cell

walls of diatoms, a component of lake phytoplankton communities (Martin-Jézéquel and others, 2000). In PROTECH, silica, alongside SRP and nitrate, is an important nutrient for diatom growth. The same metric was used in both the PROTECH model and upstream catchment model (SIMCAT-SAGIS).

2.3.1 Modelling methodology for phosphate and nitrate

Nutrient input information for PROTECH was derived from SIMCAT-SAGIS modelling. The following sections describe (1) how monthly input data was extracted for each lake, (2) how monthly loads were converted to daily concentrations, (3) and then to daily time series, for input to PROTECH.

2.3.1.1 Extracting monthly input loads from SIMCAT-SAGIS models

SIMCAT-SAGIS nutrient output concentrations were provided as annual statistical distributions rather than the time series format required for PROTECH modelling. Concentration, load, and river flow conditions were determined at any point where there was a feature (e.g., a water quality monitoring station, flow gauge or WwTW) or every 1 km, whichever was less (Figure B19). To determine the total riverine inputs to each lake, data from the closest modelled point to the lake perimeter was extracted and treated as that riverine input, the coordinates of which are provided in Table B2. As SIMCAT-SAGIS is primarily a water quality planning tool, not all rivers which enter each lake were simulated explicitly. To account for loads in unmodelled reaches and runoff directly to lakes, a separate direct load input was provided for each lake.

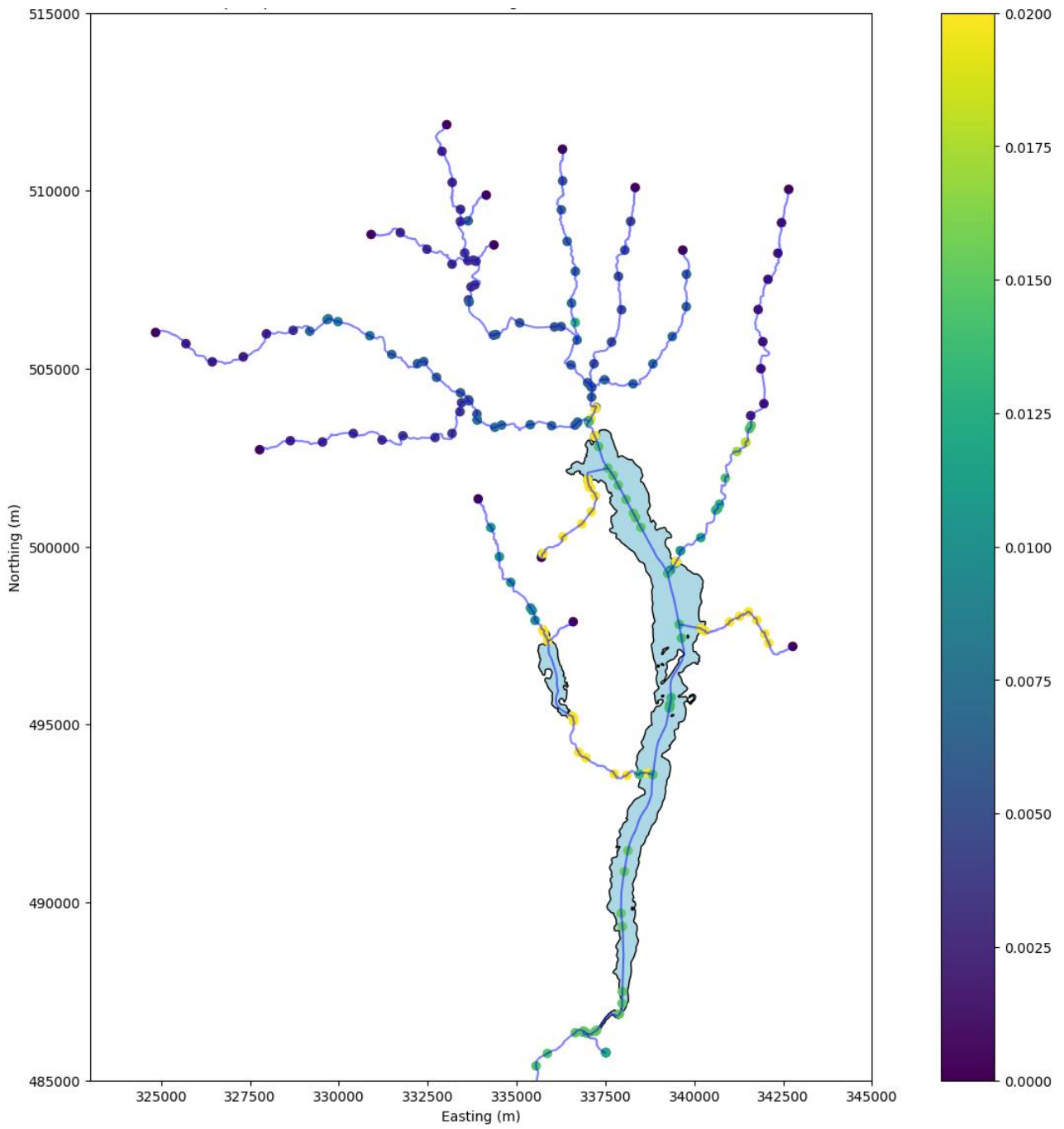


Figure B19: Representation of the SIMCAT-SAGIS outputs for mean phosphate concentrations (mg/L) in the sub-catchments that drain to Windermere North Basin, Windermere South Basin and Esthwaite Water. Each coloured point represents a model prediction along a reach (blue line).

Table B2: Location of modelled SIMCAT-SAGIS coordinates where lake inputs were extracted from. These are the closest riverine points to each lake.

Model input	River name	Easting	Northing
Windermere North Basin	Blelham Beck	337027	501739
	Mill Beck	340295	497635
	Trout Beck	339608	499886
	Rothay	337049	503524
	Brathay	337018	503527
Windermere South Basin	Cunsey Beck	338104	493576
Esthwaite Water	Black Beck	335769	497599

An annual load contributed from each riverine source was calculated from the product of river concentration and discharge. These were added to any direct loads which entered each lake to form annual lake specific input loads. Dividing the annual load equally throughout the year was unlikely to adequately capture the seasonal dynamics in each lake as there can be significant variation in the phosphate or nitrate load that enters from one month to the next. The total annual load (kg/yr) was split into individual input loads for each month based on observed concentration and modelled discharge measurements.

The input nutrient load is the product of the discharge (volume of water flowing through a stretch of river channel over a defined period) and the concentration of that discharge. Therefore, a scaling factor was needed to account for both discharge and concentration. To determine how discharge varies across the year, the ratio between average monthly discharge and average annual discharge was used to define the approximate load increase that occurs in winter due to a larger volume of water entering the lakes. In all inputs, there was a clearly defined seasonal discharge variation, with lower discharge in summer and higher discharge in winter. To determine how variations in concentration affect load throughout the year, any observed measurements on the input river between 2010 and 2023 (a longer period to include enough samples) were grouped as an average by month. The ratio between the average monthly concentration and the average annual concentration was used to provide a concentration scaling factor. Multiplying and normalising these scaling factors provided an approximate distribution of load throughout the year. Normalising was necessary to ensure the total annual load was not altered but apportioned by month.

Example plots of this process are shown for the River Brathay (Figure B20). Higher discharges occurred in winter and lower discharges in summer (blue bars). Nitrate concentrations followed the same seasonal pattern whereas phosphate concentrations were more constant throughout the year (green bars). For the River Brathay, the resultant normalised load scaling (red bars) indicated that winter introduction of nitrate was nearly seven times greater than that of summer, since both concentration and discharge were greater in winter. Seasonal variability in phosphate load was lower than nitrate, driven mostly by variation in discharge.

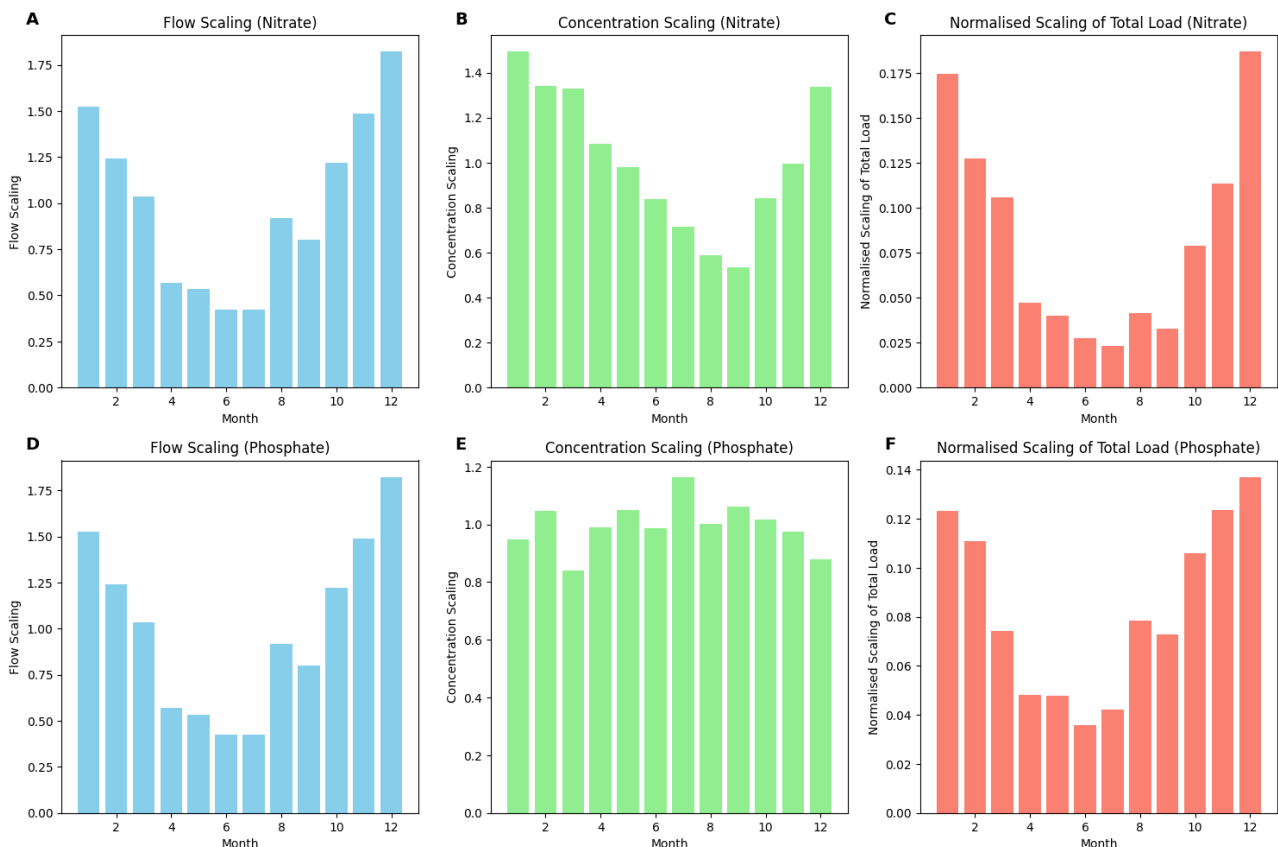


Figure B20: Discharge (blue) and concentration (green) scaling based on average monthly values compared to average annual values. These are multiplied and normalised to form a total load scaling (red), the value of which is used to define the monthly partitioning of load from the SIMCAT-SAGIS models. This example is for the River Brathay, but the process was conducted for every input river.

There were some exceptions and amendments to this approach, particularly when calculating direct loads to the lakes, where no observational data existed:

- Windermere North Basin – The average monthly scaling from each of the riverine inputs was used to constrain the direct load (Figure B21). This was normalised to ensure the total annual load was not changed.
- Windermere South Basin – There was no statistically significant change in concentration throughout the year for either the input from Cunsey Beck or for Windermere WwTW (following improvements in 2020). As such, both the riverine and direct loads to the lake were scaled only using variations in discharge.

- Esthwaite Water: The only river for which monthly apportionment of load could be constrained was Black Beck. It is assumed that the direct loads follow the same yearly distribution.

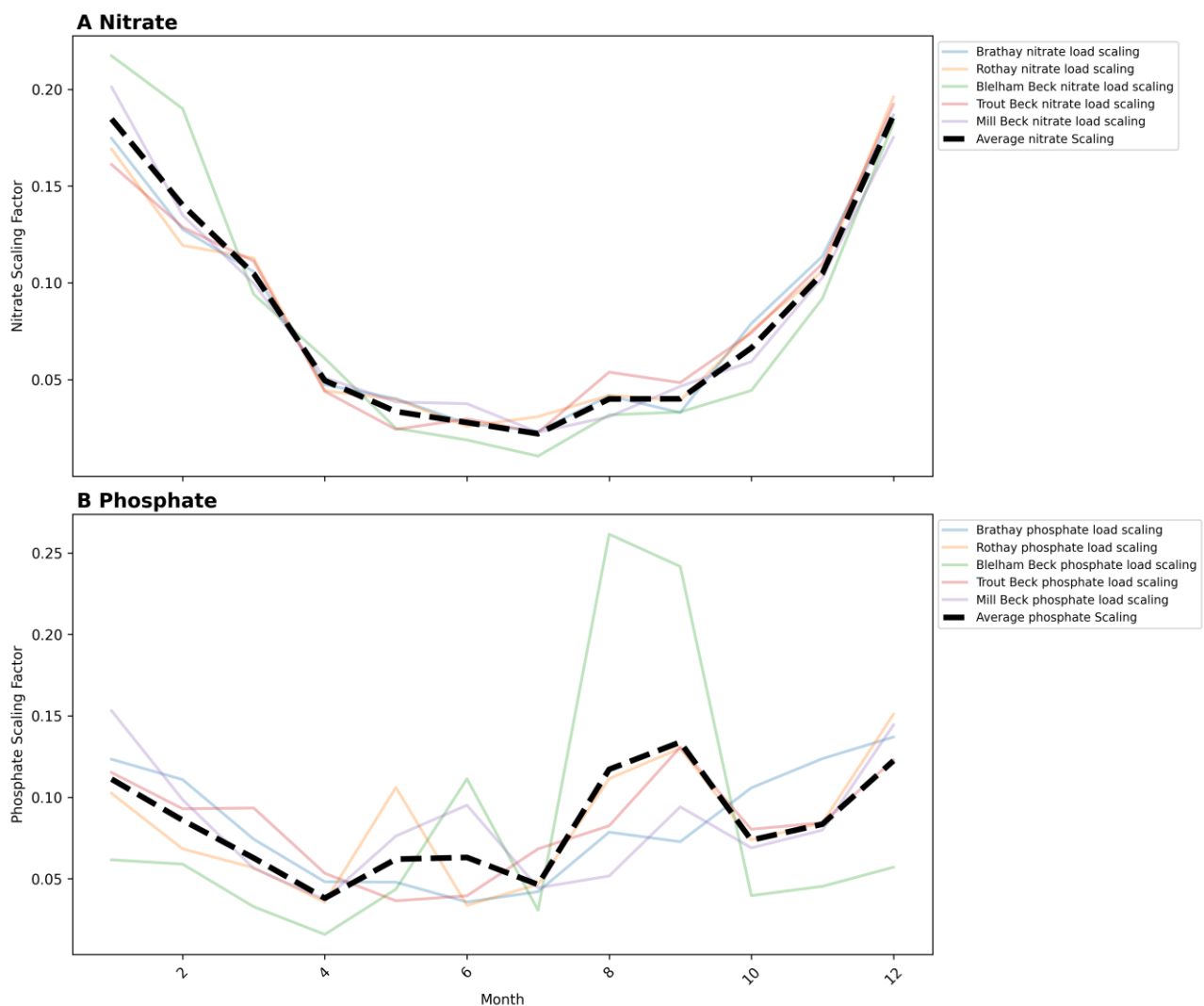


Figure B21: Load scaling factors. Average (dashed lines) of normalised load scaling factors by month for each of the riverine inputs (coloured lines) into Windermere North Basin for (A) nitrate and (B) phosphate.

This process provided a total monthly load to each lake. The total phosphate load to Windermere was estimated to be 5516 kg/yr, in line with estimates from other studies, albeit for TP (Viridian Logic, 2022).

2.3.1.1.1 Validating monthly input loads

As SIMCAT-SAGIS did not accurately capture the temporal variations in load throughout the year, a monthly scaling factor was applied to the annual load based on the observed concentration and discharge information for each input river. Figure B22 illustrates the difference between the monthly modelled concentrations and the average concentration recorded over the period 2010-2023. This longer period was chosen as there was typically

only one measurement per month, and there were significant gaps in the data record. Observed mismatches included Blelham Beck, which had variable phosphate concentrations between 2020 and 2024, with high concentrations observed in late summer months; Mill Beck for which there were highly variable concentrations in phosphate throughout 2010 to 2023; and the River Rothay, for which numerous AMP6 improvements have reduced average phosphate concentrations in recent years. Overall, the monthly distribution was well captured, and the total load entering each lake each month was well simulated.

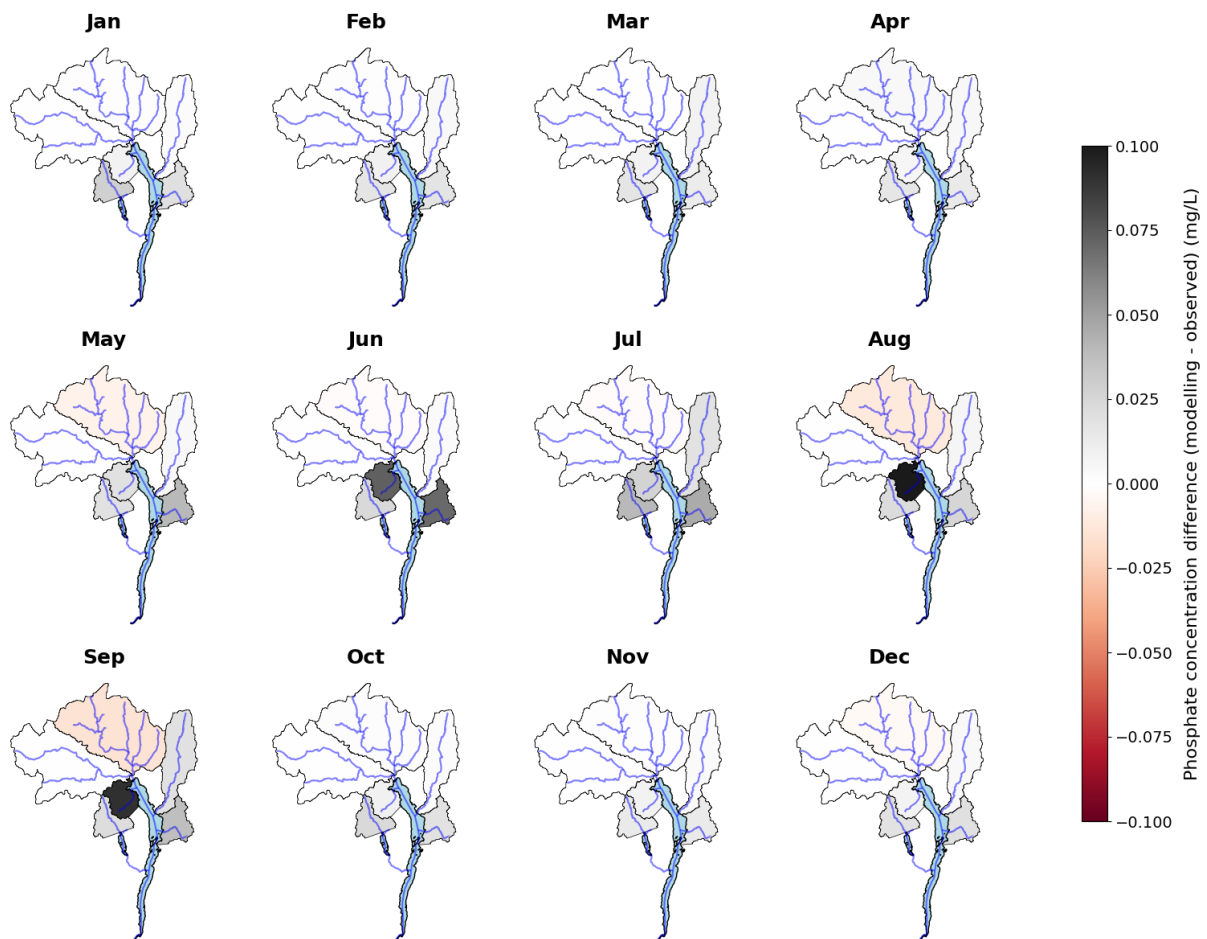


Figure B22: Absolute concentration difference between modelled monthly data and average observed data (over the period 2010-2023) for phosphate in the study sub-catchments of the Windermere catchment.

2.3.1.2 Converting the monthly load to a daily concentration

SIMCAT-SAGIS utilised a calibrated integrated flow dataset (Wallingford HydroSolutions, 2025a) to estimate hydrology across the Windermere catchment. In this study, river flows for the baseline and future scenarios were determined using the Qube platform (Wallingford HydroSolutions, 2025b). The dependency on internal hydrology can be removed for the most part by utilising load output estimations, instead of concentrations, from the SIMCAT-SAGIS model. The total load to each lake determined in the SIMCAT-SAGIS model (provided in kg/day) and the average monthly flows (aggregated from Qube

data (m³/second)) to each model were used to determine the average concentration for each month in milligrams per litre (Equation B2).

$$\sum \text{Concentration} = \frac{\sum \text{Load}}{\sum \text{Flow}} \quad \text{Equation B2}$$

The resultant monthly concentration for Windermere North Basin is presented in Figure B23. Since SIMCAT-SAGIS provided monthly average loads (rather than for a specified period), the monthly signal is invariant year on year. However, changes to monthly flow can considerably affect the resultant concentration of the input sources.

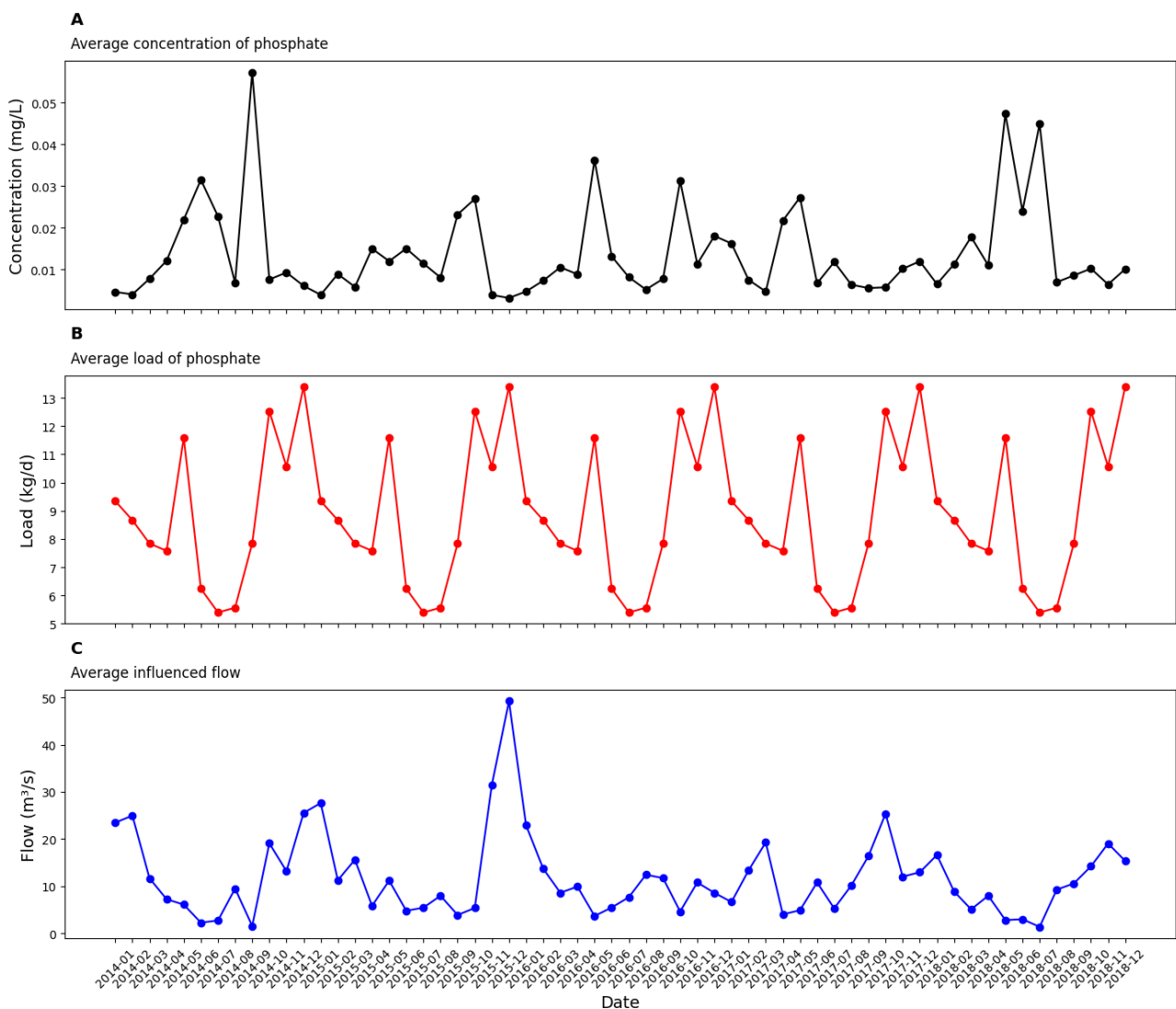


Figure B23: Average monthly (A) concentration, (B) load, and (C) flow inputs for Windermere North Basin. Flow is a monthly average of the data shown in Figure B16.

2.3.1.3 Conversion of monthly outputs to a daily time series

PROTECH simulates lake response at daily resolution, whilst the SIMCAT-SAGIS model provides outputs at monthly resolution. Previous studies using PROTECH have used a linear interpolation between measurements. While this is a reasonable assumption, it may lose some of the variability which occurs across a month and is particularly problematic when considering how SIMCAT-SAGIS simulates intermittent loads. In turn, the SIMCAT-SAGIS monthly loads were translated to concentration at daily resolution. There were two motivations for doing this: (1) to understand daily variations in concentration that may occur due to changes in river flow and associated dilution of load and (2) to understand how different treatment of intermittent loads may alter the daily concentration estimates.

The following sections explain three different components which could impact phosphate concentration at daily resolution. Scenario 1 considers changes in river flow, assuming a constant load for every day of a month. Scenario 2 considers how apportioning intermittent load, during days where intermittent discharges would expect to trigger, impacts daily concentration. Scenario 3 examines the assumption that the load input stays the same throughout a month, and tests how load and discharge relationships may alter daily concentrations. The final input nutrient concentrations incorporated all these components.

2.3.1.3.1 Scenario 1: Using daily river flow to estimate concentration changes (constant load throughout a month)

The monthly load was assumed to be equal each day of the month (Figure B24, B). Therefore, concentration of the nutrient (Figure B24, A) for a given day solely depended on river flow (Figure B24, C) and an equal split of the monthly load. A day of higher river flow would lead to greater dilution; hence concentration decreased under high flow conditions. The converse, very low river flows, resulted in spikes of phosphate concentration during certain periods (e.g., up to 0.2 mg/L in summer months). As discussed in the next two sections, there are issues with using this approach alone, which may lead to an overestimation of daily concentrations, for large portions of the year.

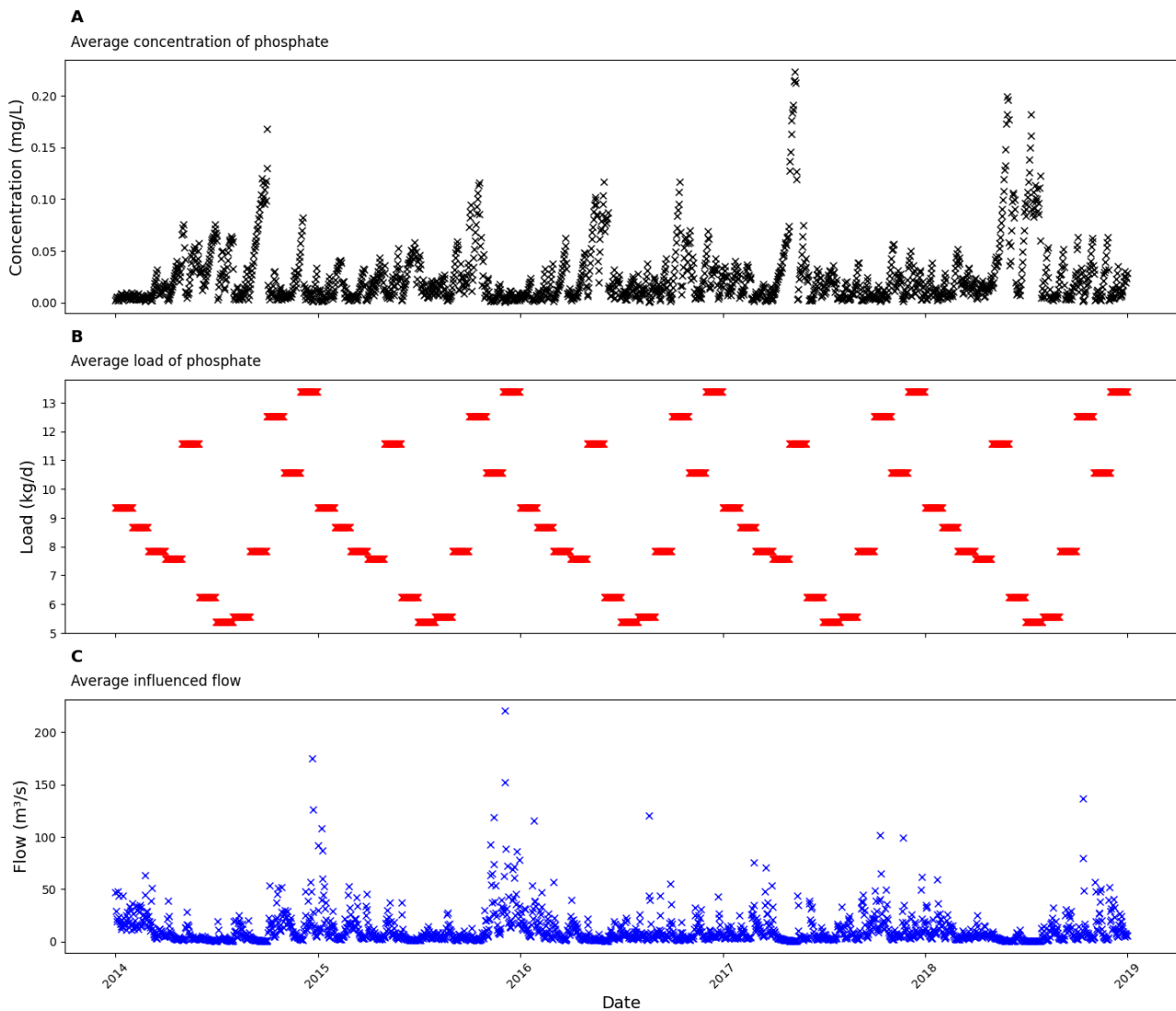


Figure B24: Phosphate concentrations. (A) Daily phosphate concentration as calculated by assuming (B) a constant monthly load, over (C) a varying daily river flow, for Windermere North Basin.

2.3.1.3.2 Scenario 2: Apportioning intermittent loads to individual days

Rather than drawing from a normal or log-normal distribution during the Monte Carlo process, the spill or non-spill nature of an intermittent input was captured using a non-parametric distribution (.npd) file, defining the number of times an intermittent discharge will be active. When a single shot of the Monte Carlo simulation was taken, there may be no intermittent input. Since SIMCAT-SAGIS provided a mean concentration, this nuance was lost in the final statistics. The intermittent input was applied equally over every month, and when downscaled to a daily timestep, was also applied at this resolution. This approach likely overestimated concentrations on drier days, when intermittent discharges should not occur.

To better apportion intermittent discharge loading to times when an intermittent should activate, the total intermittent load was separated from the load from all other sectors. A spill frequency (percentile) was defined for each intermittent separately based on EDM data and loads supplied by sewage modelling data (provided by UU). This total load was

evenly spread across the number of days when a spill would occur according to the percentile. By factoring this approach in, the summer peaks in concentration (driven by lack of dilution) were reduced (Figure B25). Concentration increases occurred during the periods of flow above the percentile threshold for intermittent load to be applied. Since these were periods where river flow was high, the concentration increases caused by higher intermittent load were less than the concentration decreases seen in periods of lower flow. Treating each intermittent asset independently meant that on days with particularly high flow conditions, more intermittent discharges spilt.

An improvement on this approach would be to test the assumption that all days which spill contribute the same load. Given the design of sewage overflows and storm tanks, more load is released in higher flows. In addition, storage capacity may be filled more at certain times of the year, hence more load may be released. As more data becomes available from these intermittent discharges, this could be tested in future.

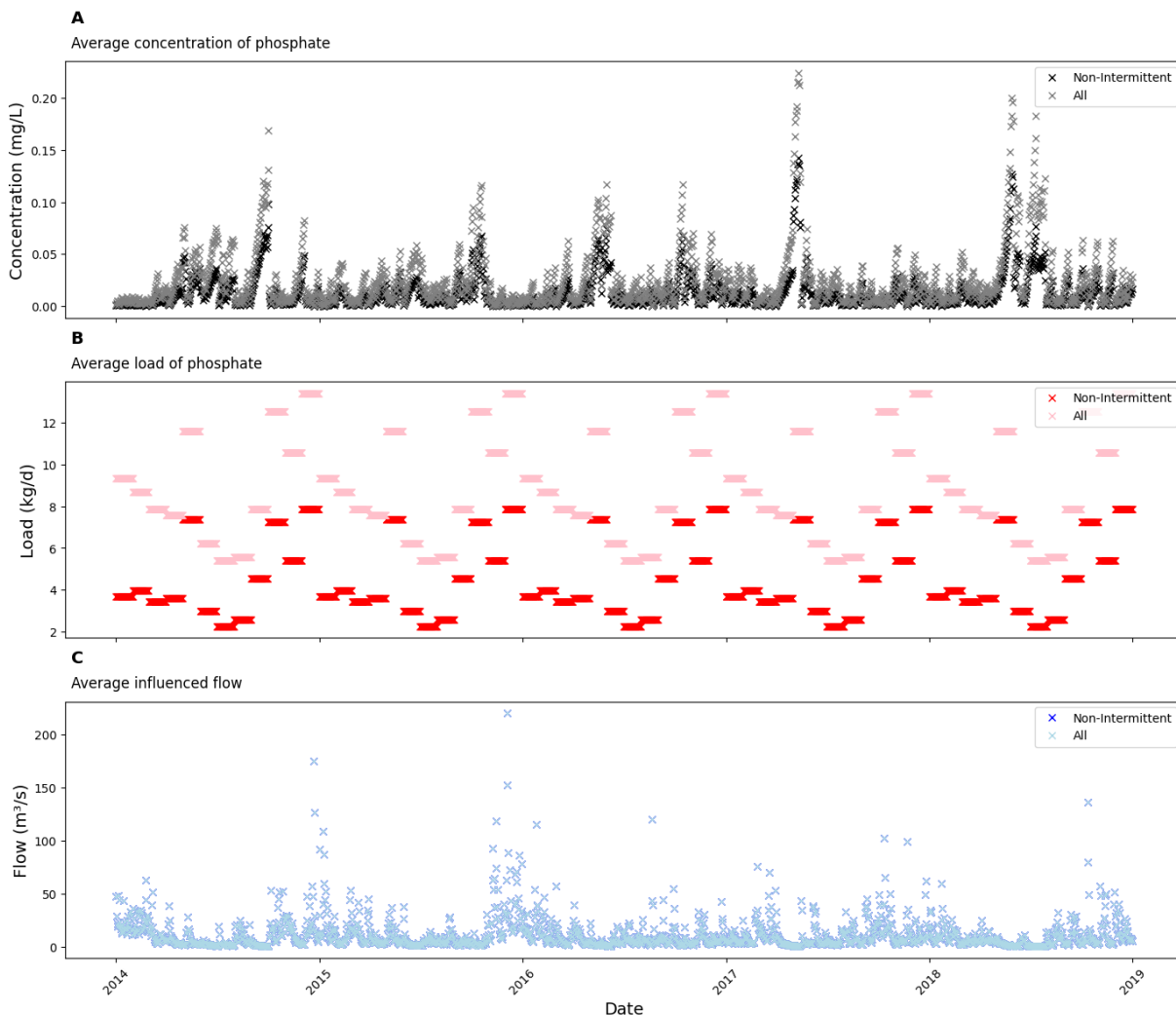


Figure B25: Revised phosphate concentrations. (A) Black crosses represent the revised daily concentration for Windermere North Basin in comparison to grey crosses using the method in Scenario 1 (Figure B24). (B) Pink crosses represent the values determined previously, whilst red crosses represent the revised method. Periods of high load represent periods where different intermittents are active. (C) Daily river flow estimates.

2.3.1.3.3 Scenario 3: Testing the assumption of constant load each day of a month

In Scenario 1, the monthly load provided by SIMCAT-SAGIS was assumed to be equally split for each day of the month. This assumption may not be correct if the load of phosphate is greater during periods of higher river flow. In a catchment, this may occur if more of the diffuse nutrient load is mobilised during periods of high rainfall (and so higher river flow). To scale the diffuse load based on the daily discharge, the following relationship was used (Equation B3):

Equation B3

$$F_{scaled} = \begin{cases} (a + (f - a) \times s) * l & \text{if } f > A \\ (a - (f - a) \times s) * l & \text{if } f \leq A \end{cases}$$

Here a is an average value (in this case, 1 under the assumption that this is the mean discharge), f is a scaled discharge value (where $\text{discharge}/\text{discharge}_{\text{mean}}$), s is a scaling factor, and l is the monthly diffuse load. A scaling factor of 0 (or no adjustment) would mean equal loading on each day of a month regardless of discharge, and a scaling factor of 1 would mean double the discharge would result in double the load. To test for suitable scaling factors based on observed data, the load and discharge data in Anderson (2018) was used to determine fit to a series of scaling factors. The root mean squared error between the observed data and a prediction (for both phosphate and nitrate) based on a scaling factor was lowest at a scaling factor of 0.9 (Figure B26), which was chosen as the optimal scaling factor. This scaling factor indicated that there was a strong relationship between load and discharge in the Windermere catchment, with higher river flow containing more nutrient load.

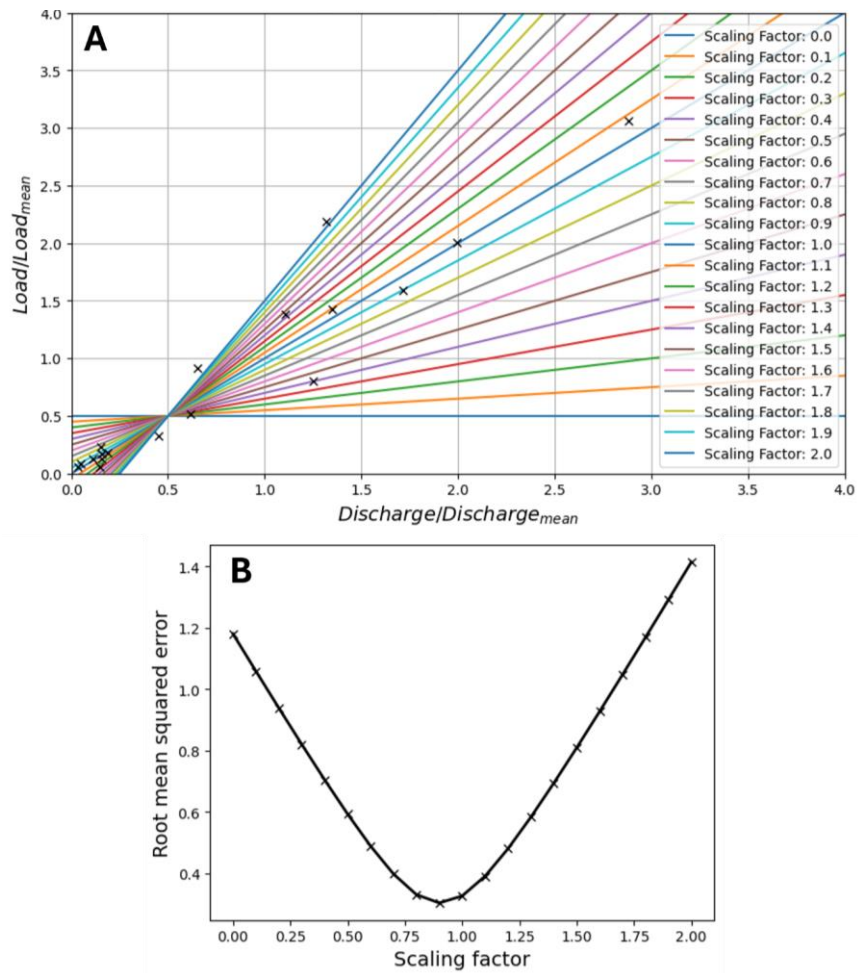


Figure B26: Testing different scaling factors between load and discharge based on the dataset in Anderson (2018). (A) The relationship between discharge and load anomaly for various scaling factors is plotted alongside observed data. (B) The minima of the root mean squared error of the fit of the observed points to various scaling factors provides the optimal scaling factor to use.

With these adjusted daily loads, the resultant daily predicted concentrations were more standardised and less susceptible to the concentration spikes on particularly dry days. Therefore, this scenario provided the representation of daily concentration input which best reflects a reasonable set of assumptions (Figure B27).

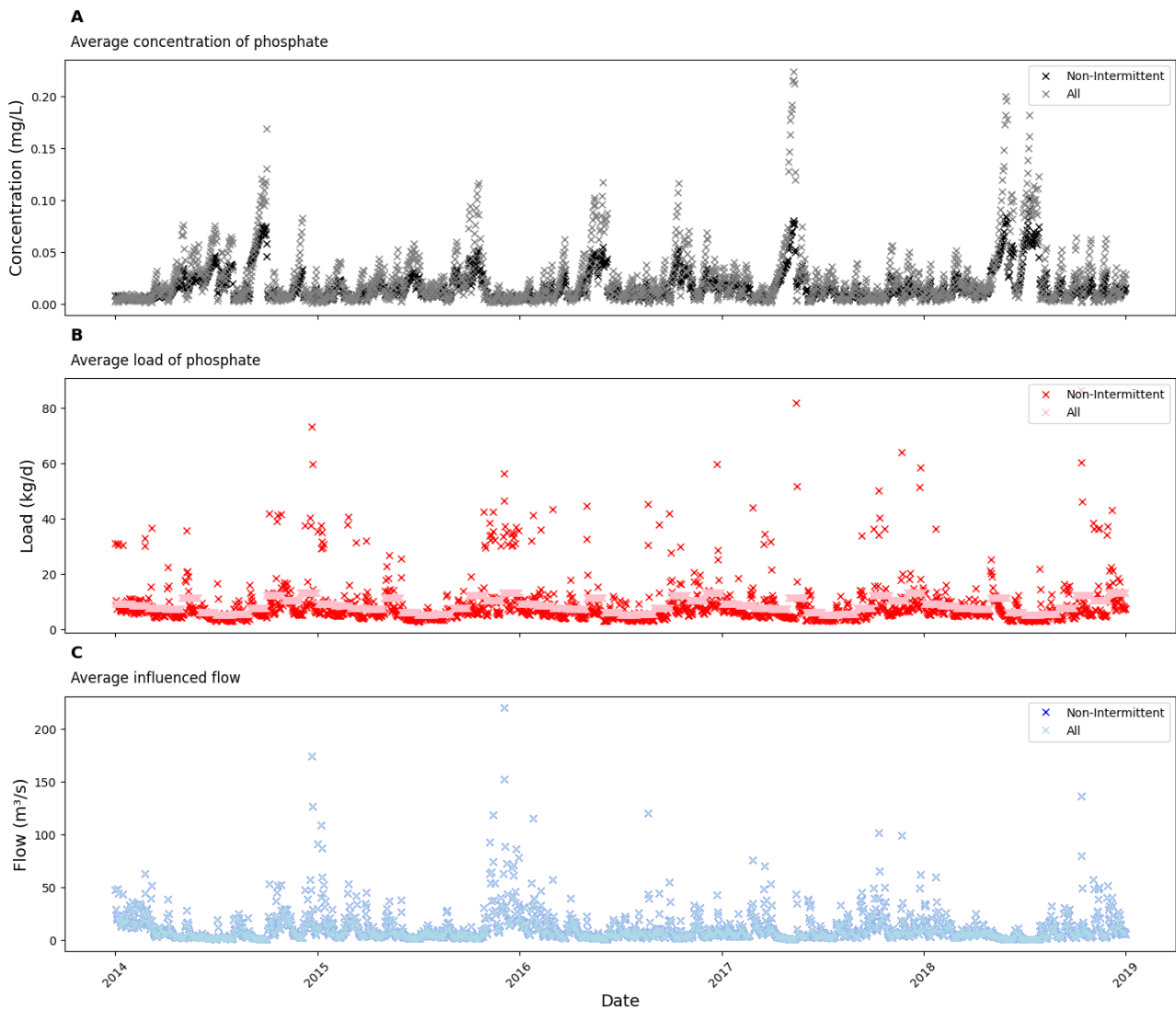


Figure B27: Further revised phosphate concentrations. (A) Black crosses represent the revised daily concentration for Windermere North Basin in comparison to grey crosses using the method in Scenario 1 (Figure B24). (B) Pink crosses represent the values determined previously, whilst red crosses represent the revised method in Scenario 3. Periods of high load represent periods where different intermittents are active. (C) Daily river flow estimates.

2.3.2 Validation of phosphate and nitrate data

To effectively estimate the accuracy of the SIMCAT-SAGIS model performance, comparison was required between observed and modelled concentrations of nutrients, over the same period. The following sections provide further detail specific to modelled input streams.

2.3.2.1 Windermere North Basin

River Brathay

Measurements from WQ88004530 on the River Brathay had a data gap between 2014 and 2018 (Figure B28). The range of nitrate concentrations were well captured by the SIMCAT-SAGIS model. Phosphate concentrations prior to 2014 were higher than

SIMCAT-SAGIS predicted but many of these were recorded at detection limit and so may have been much lower. Samples from the Big Windermere Survey (BWS) aligned with those measured by the Environment Agency and SIMCAT-SAGIS predictions.

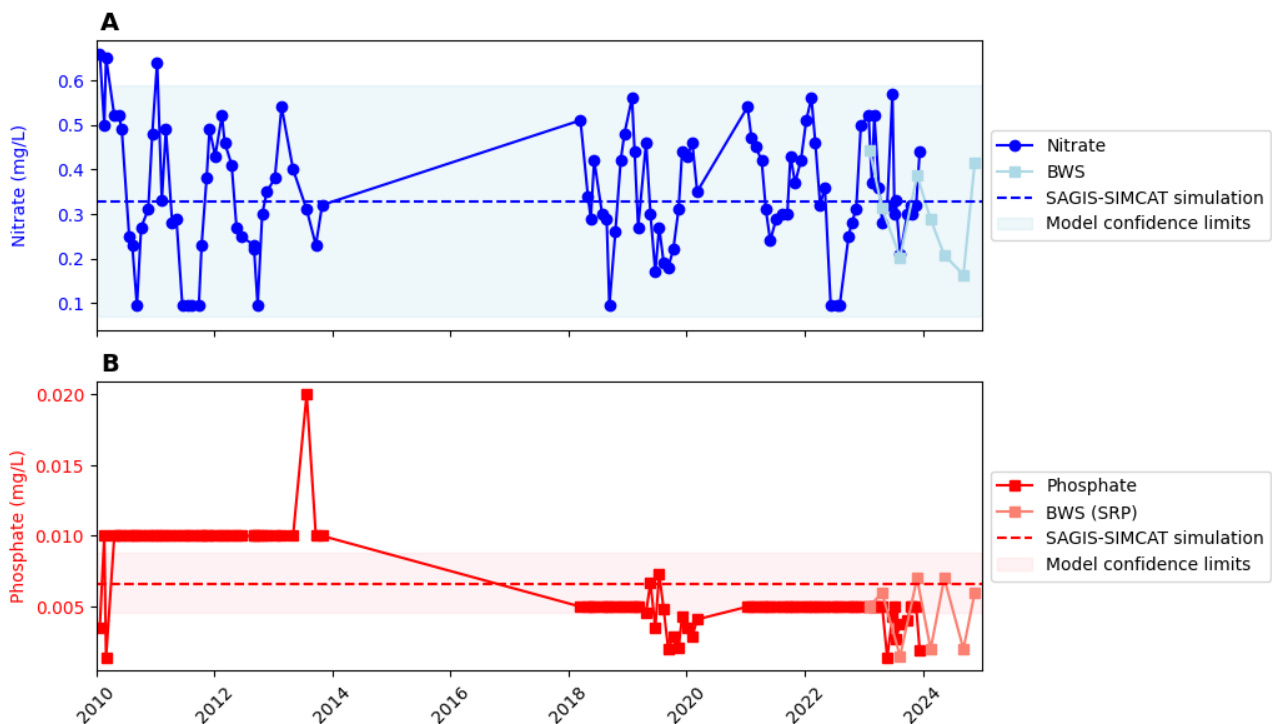


Figure B28: Comparison of (A) nitrate and (B) phosphate concentrations at a co-located sample point on the River Brathay. The dashed line represents the average SIMCAT-SAGIS modelled concentration, and the model confidence limits are denoted by the shaded area. BWS = Big Windermere Survey (note that SRP is reported for the BWS, and phosphate for Environment Agency monitoring points and SIMCAT-SAGIS predictions).

River Rothay

For the River Rothay, mean nitrate concentrations were well predicted by SIMCAT-SAGIS, but the range of nitrate concentrations (Figure B29), and high peaks in phosphate that occurred in 2014 and 2016 were not well captured.

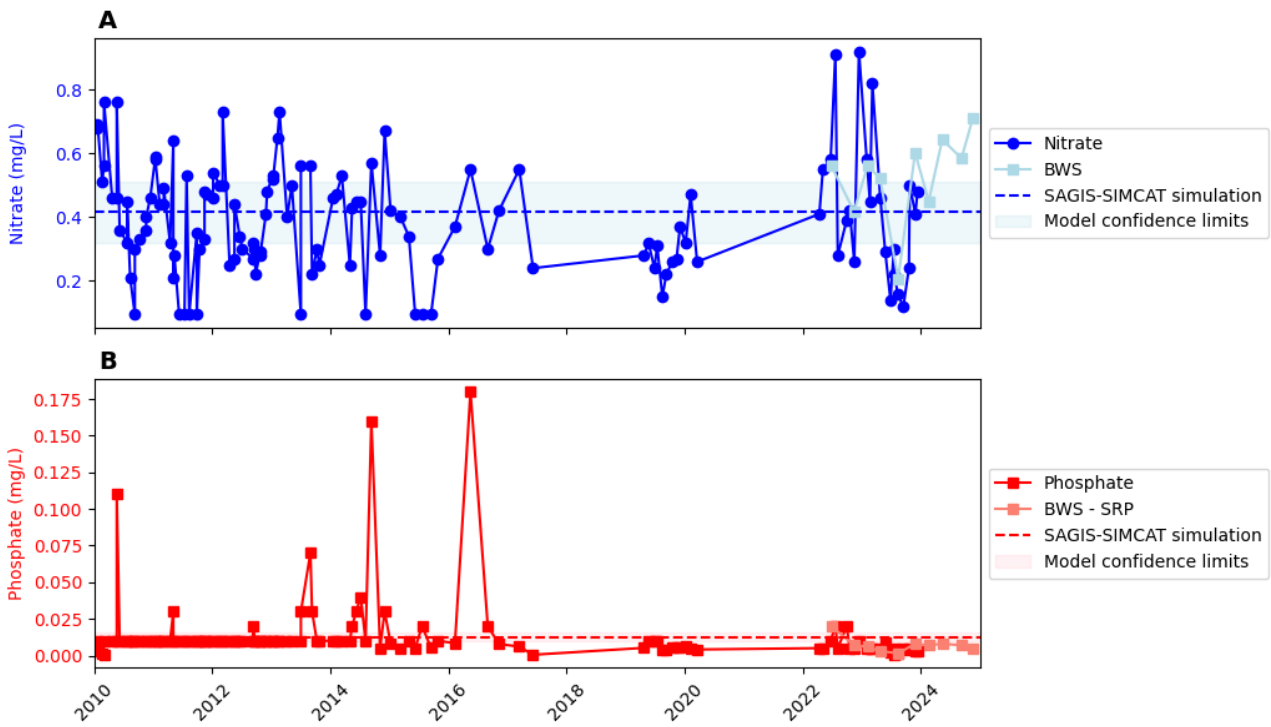


Figure B29: Comparison of (A) nitrate and (B) phosphate concentrations at a co-located sample point on the River Rothay. The dashed line represents the average SIMCAT-SAGIS modelled concentration, and the model confidence limits are denoted by the shaded area. BWS = Big Windermere Survey (note that SRP is reported for BWS, and phosphate for Environment Agency monitoring points and SIMCAT-SAGIS predictions).

Blelham Beck

Mean nitrate concentrations and the range of concentrations were well predicted by the SIMCAT-SAGIS model (Figure B30). There were numerous spikes in phosphate concentration (typically occurring in August and September) which were not captured, from currently unidentified sources.

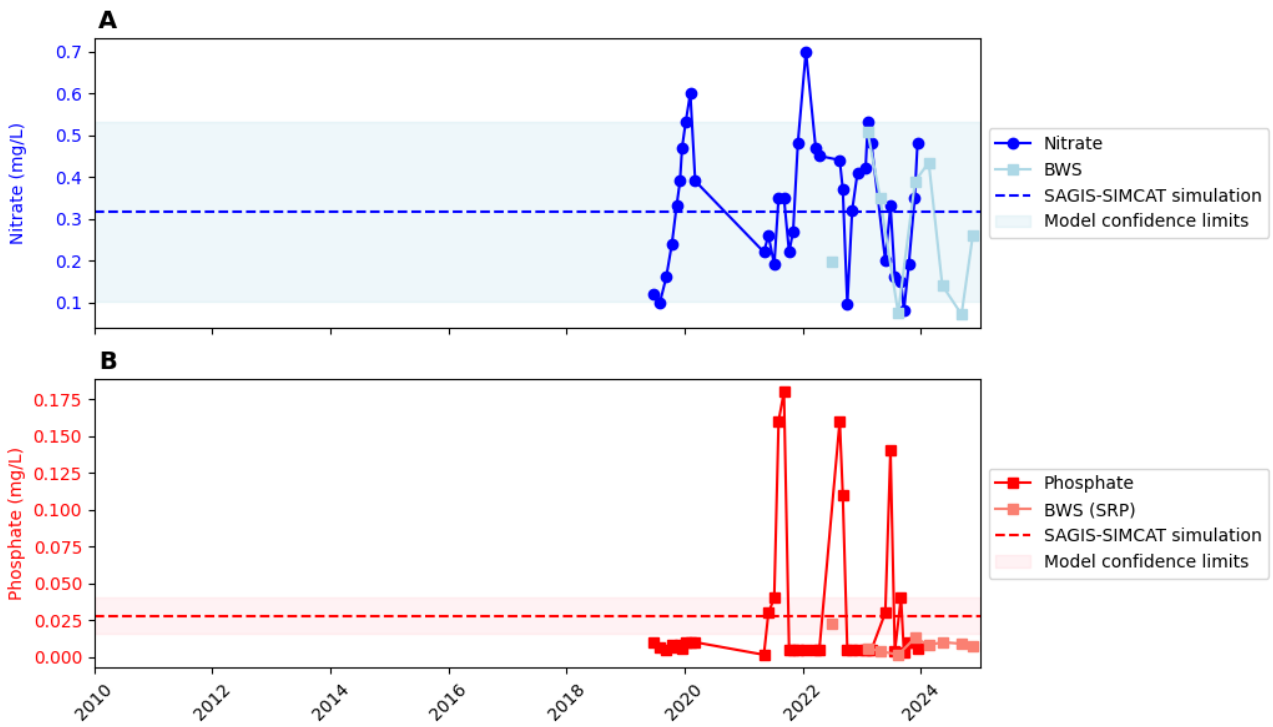


Figure B30: Comparison of (A) nitrate and (B) phosphate concentrations at a co-located sample point on Blelham Beck. The dashed line represents the average SIMCAT-SAGIS modelled concentration, and the model confidence limits are denoted by the shaded area. BWS = Big Windermere Survey (note that SRP is reported for BWS, and phosphate for Environment Agency monitoring points and SIMCAT-SAGIS predictions).

Trout Beck

The mean and range of nitrate concentrations were well captured by the SIMCAT-SAGIS predictions in Trout Beck (Figure B31). Phosphate concentrations were well projected aside from a small number of spikes in 2022 and 2023.

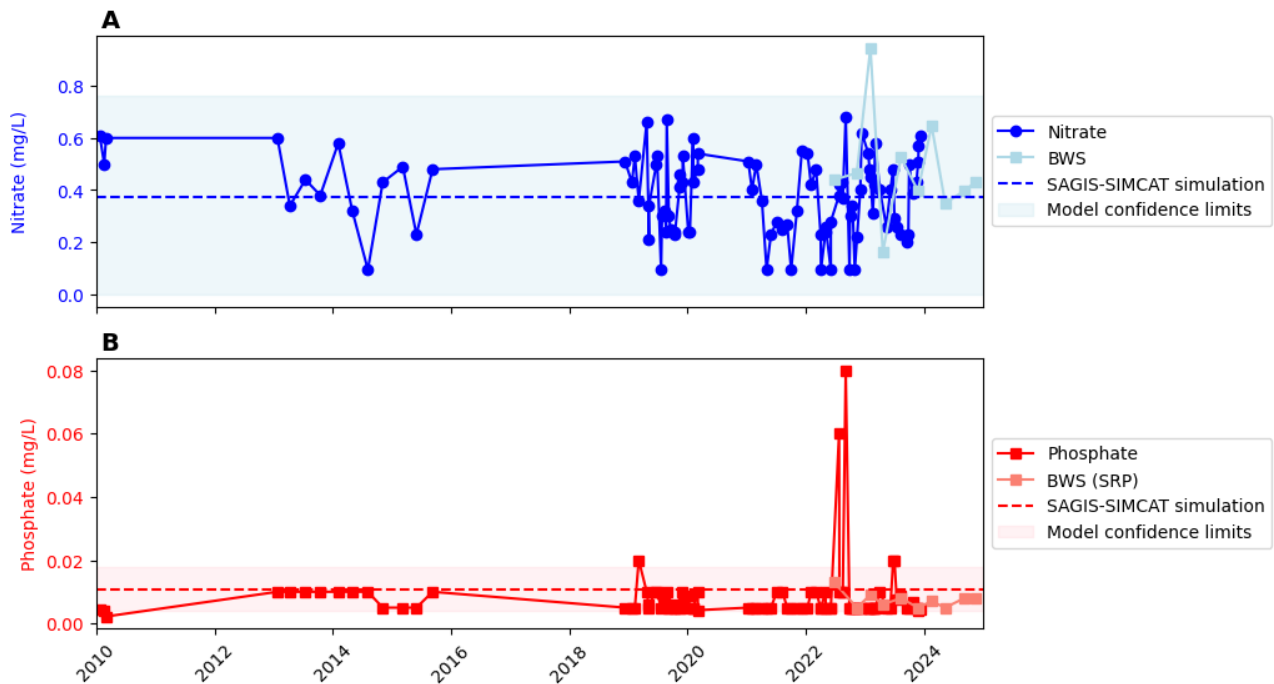


Figure B31: Comparison of (A) nitrate and (B) phosphate concentrations at a co-located sample point on Trout Beck. The dashed line represents the average SIMCAT-SAGIS modelled concentration, and the model confidence limits are denoted by the shaded area. BWS = Big Windermere Survey (note that SRP is reported for BWS, and phosphate for Environment Agency monitoring points and SIMCAT-SAGIS predictions).

Mill Beck

Mean nitrate and phosphate concentrations were higher in Mill Beck than in any other riverine inputs to Windermere North Basin (Figure B32). The range of nitrate concentrations were well captured by the SIMCAT-SAGIS model. The peaks in phosphate concentration were not well captured by the SIMCAT-SAGIS model.

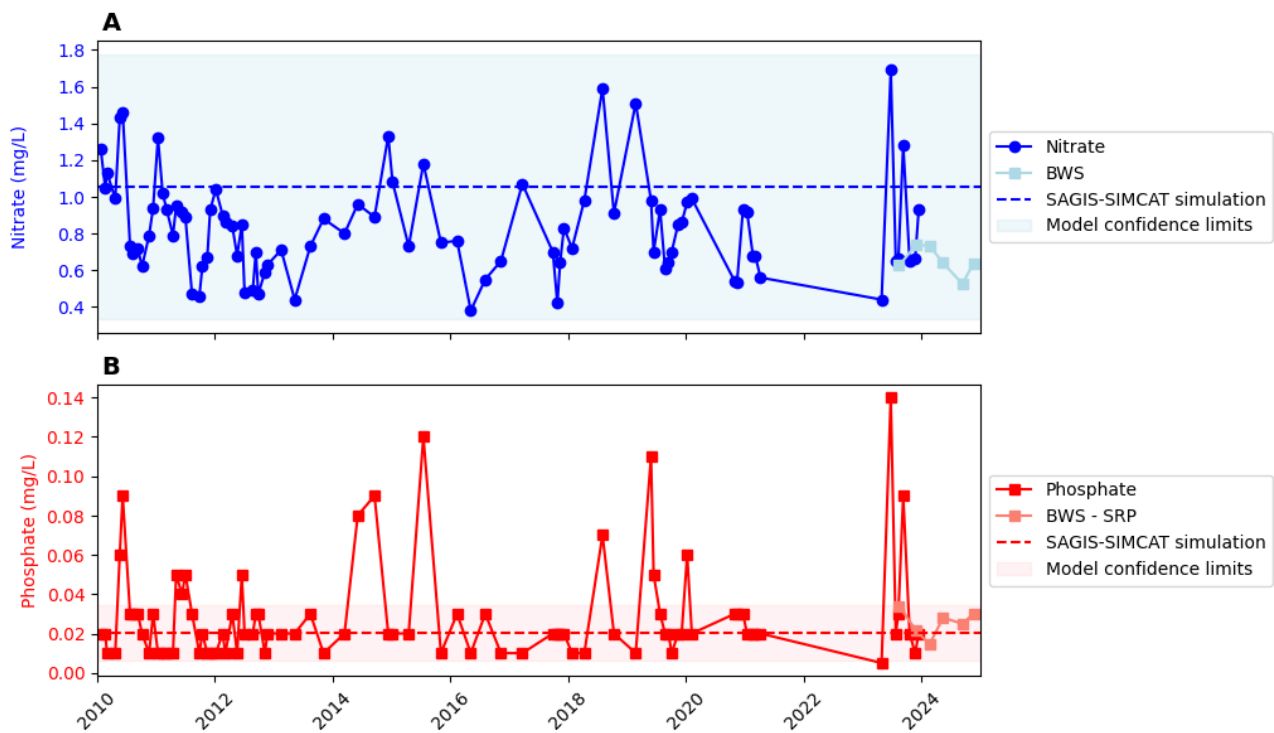


Figure B32: Comparison of (A) nitrate and (B) phosphate concentrations at a co-located sample point on Mill Beck. The dashed line represents the average SIMCAT-SAGIS modelled concentration, and the model confidence limits are denoted by the shaded area. BWS = Big Windermere Survey (note that SRP is reported for BWS, and phosphate for Environment Agency monitoring points and SIMCAT-SAGIS predictions).

2.3.2.2 Esthwaite Water

Whilst Esthwaite Water was supplied by numerous input streams, Black Beck was larger than the others and contained most of the expected nutrient load. Environment Agency monitoring data upstream of Esthwaite Water existed only on the Black Beck input, upstream of Hawkshead WwTW (WQ88004549) (Figure B33).

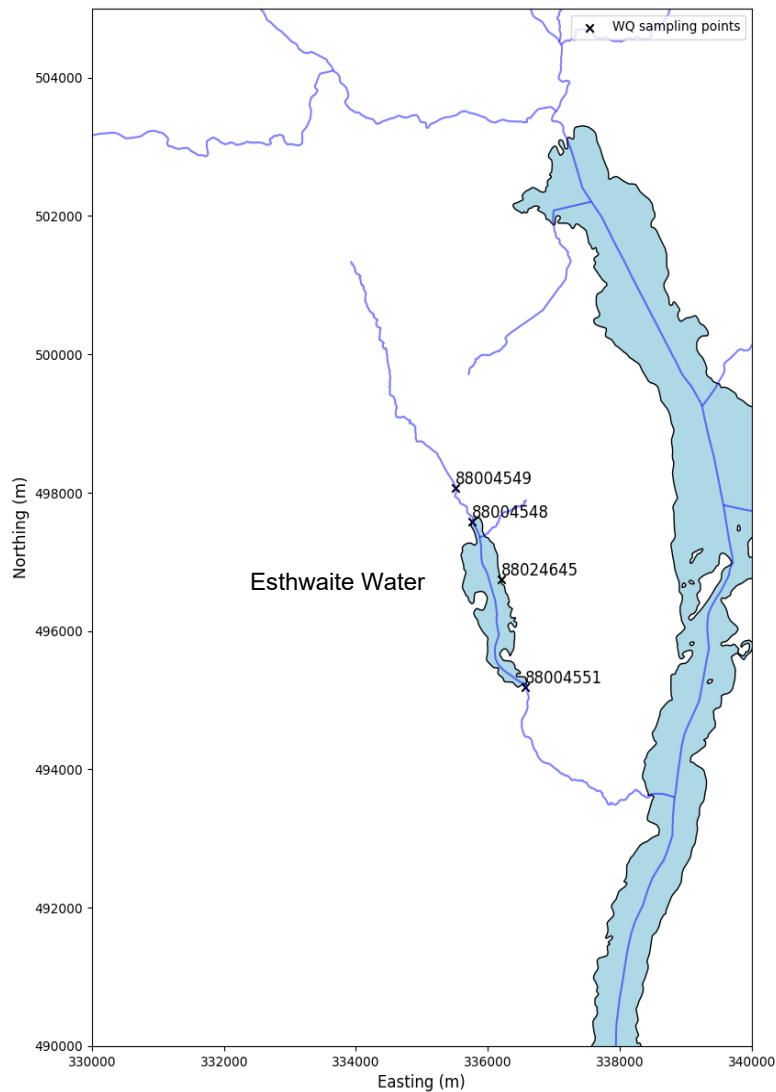


Figure B33: Location of Environment Agency water quality monitoring points in Esthwaite Water. WQ88004549 is the only upstream point of Windermere with nutrient data available.

Comparison of this water quality monitoring point with co-located data from other sources (Anderson, 2018; FBA, 2025) showed relatively close alignment of phosphate and nitrate values (Figure B34). The SIMCAT-SAGIS representation of this point (upstream of Hawkshead WwTW) was also in line with these data sources, although the uncertainty range did not capture some of the highest phosphate concentration events observed earlier in the record.

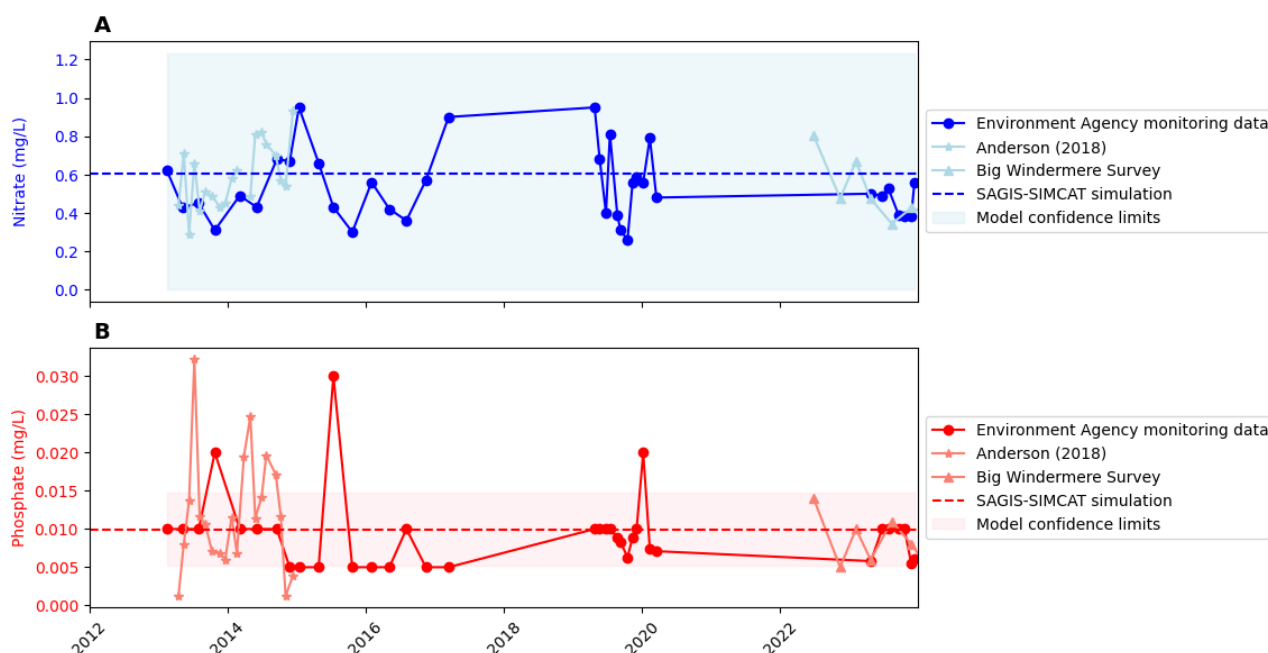


Figure B34: Comparison of (A) nitrate and (B) phosphate concentrations at a co-located sample point upstream of Hawkshead WwTW on Black Beck. The dashed line represents the average SIMCAT-SAGIS modelled concentration, and the model confidence limits are denoted by the shaded area. Note that SRP is reported for the Big Windermere Survey, and phosphate for Environment Agency monitoring points and SIMCAT-SAGIS predictions.

2.3.3 Validation of dissolved silica data

Silica concentrations sampled between 01/01/1950 and 27/11/2024 at monitoring points on input streams were extracted from the Water Information Management System (WIMS) database. The determinand name of these measurements was “SiO₂, reactive as silica” (determinand code = 0182) which reflects the stoichiometric concentration of SiO₂ in solution and was the form of silica used in PROTECH. At least one sampling point on each input river had silica data available (Table B3). Data from the sampling points was filtered to only include measurements taken for monitoring programme purposes (i.e., excluding investigation data). The dissolved silica concentration for any sample below the instrumental detection limit was halved (common practice in water quality). One outlier from Cunsey Beck (SiO₂ > 10 mg/L) was excluded as it was over 5 times the standard deviation from the mean.

Table B3: Availability of dissolved silica data for input rivers to Windermere and Esthwaite Water.

Point code	Input river	Silica information available?
88004530	Brathay	Yes
88019894	Brathay	No (investigation only)
88004520	Rothay	Yes
88004519	Rothay	Yes
88023840	Blelham Beck	No
88023839	Blelham Beck	No
88023838	Blelham Beck	No
88023837	Blelham Beck	No
88004536	Blelham Beck	Yes
88007083	Blelham Beck	No (investigation only)
88023296	Blelham Beck	No
88023835	Blelham Beck	No
88023836	Blelham Beck	No
88004537	Trout Beck	Yes
88022167	Trout Beck	No
88004545	Mill Beck	Yes
88023724	Mill Beck	No
88024250	Mill Beck	No
88004548	Black Beck	No (investigation only)
88004549	Black Beck	Yes
88004551	Cunsey Beck	Yes (but only one)
88004552	Cunsey Beck	Yes
88025805	Cunsey Beck	No

Silica concentrations were variable across the Windermere catchment. Mill Beck and Black Beck contained the highest silica and the River Rothay, Cunsey Beck and River Brathay contained the lowest. Silica concentrations in Blelham Beck and Black Beck additionally aligned with measurements collected by UKCEH. Low concentrations were recorded in the River Rothay and Cunsey Beck in spring months, potentially indicating in-river blooms of silica-containing organisms (e.g., diatoms) (Martin-Jézéquel and others, 2000).

To establish a daily prediction of dissolved silica concentrations throughout the five-year baseline period (2014-2018), a polynomial regression was determined between dissolved silica concentration and day of the year (Figure B35). An appropriate number of degrees of freedom for the polynomial fit (either 2 or 3), and whether the day of year was modelled using a sine and cosine break down, was independently chosen for each input river to ensure best fit.

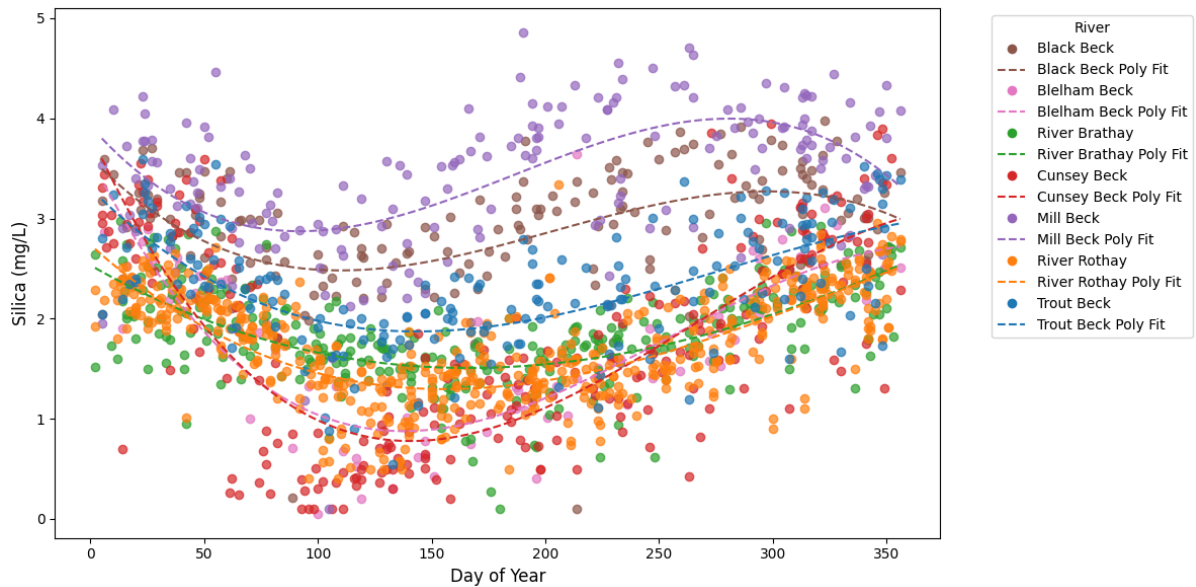


Figure B35: Dissolved silica measurements (silica reactive as SiO_2) from input rivers to Windermere and Esthwaite Water plotted against day of the year. A polynomial fitted line (dashed line) shows the broad relationship throughout the year.

2.3.3.1 Windermere North Basin

For Windermere North basin, where there were multiple inputs, the combined input dissolved silica concentration was defined using the polynomial relationship for each of the inputs weighted by the flow contribution each day in the 5-year period (Figure B36). The resultant input concentration followed a u-shaped curve throughout the year, with higher concentrations in winter than summer (Figure B37). Mill Beck, which contained high dissolved silica concentrations, contributed a small amount of the combined flow.

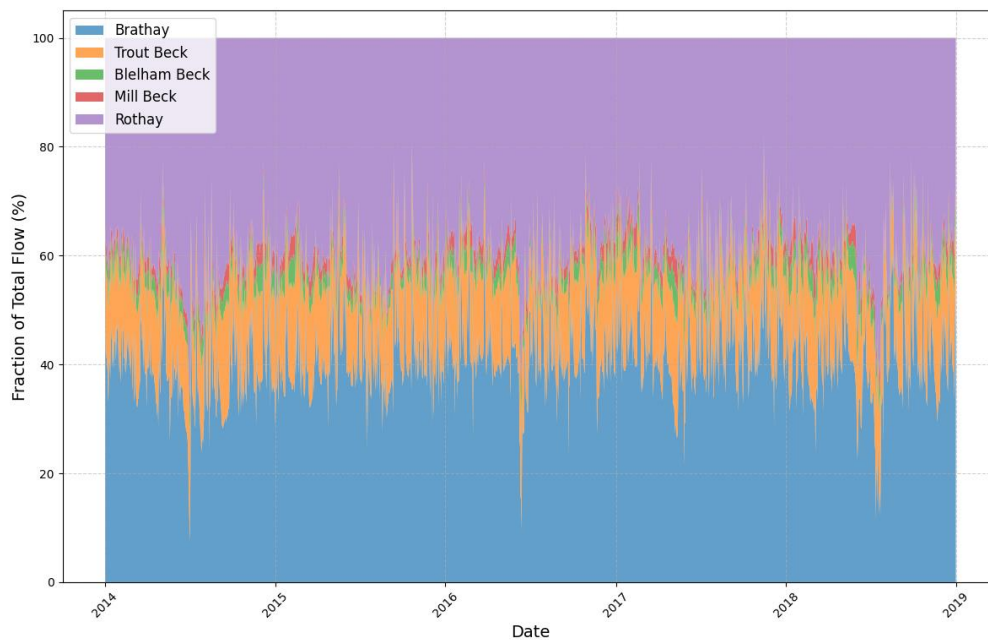


Figure B36: Fraction that each input river contributed to the total input load of Windermere North Basin, each day over the 5-year baseline period (2014-2018). Data obtained from Qube.

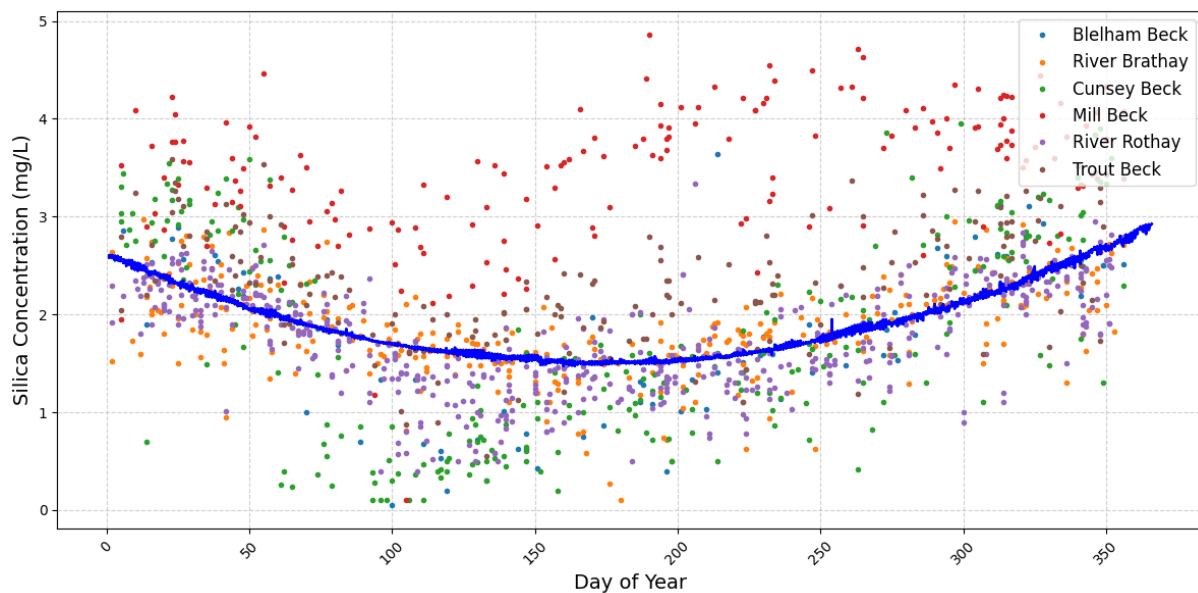


Figure B37: Individual dissolved silica measurements in each of the input rivers to Windermere North Basin plotted against the day of the year. The fitted polynomial relationship is defined by the blue line and provides a daily prediction of dissolved silica concentration. Variation is caused by the differing contribution of flow each day.

2.3.3.2 Windermere South Basin

Simulating dissolved silica load for Windermere South Basin included calculating the concentration of silica contributed by Cunsey Beck, as shown in Figure B38. There was a noticeable decrease in dissolved silica between day 50 and day 150 which is partly

captured by the polynomial regression. This decline may have been caused by diatom formation in either the river itself or the upstream lake (Esthwaite Water).

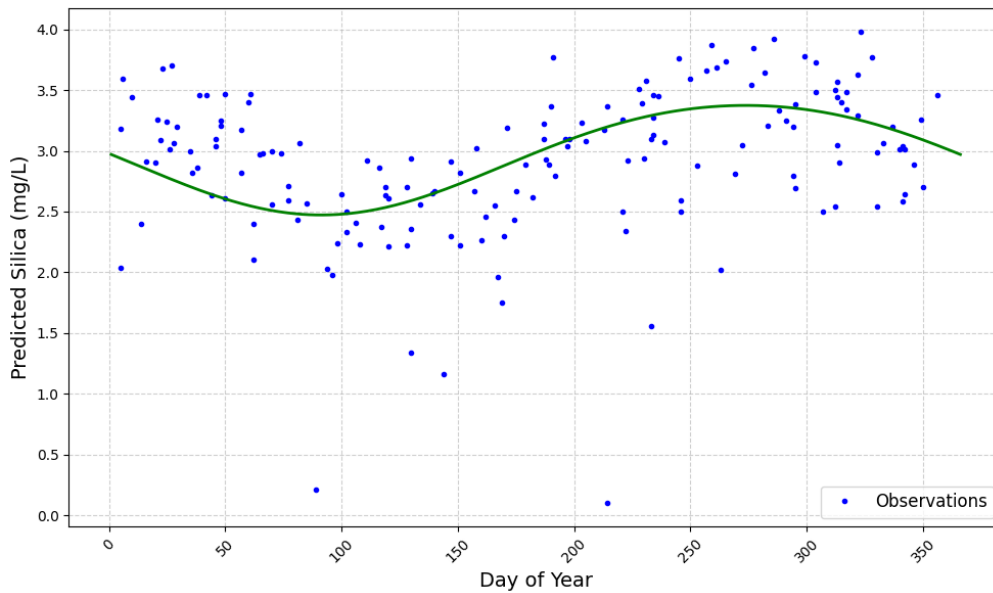


Figure B38: Individual dissolved silica measurements in Cunsey Beck plotted against the day of the year. The fitted polynomial relationship is defined by the green line and provides a daily prediction of dissolved silica concentration.

2.3.3.3 Esthwaite Water

For Esthwaite Water, the regression for Black Beck was used to simulate silica concentrations of all other inputs to the lake. Silica concentration was predicted to follow a sinusoidal pattern throughout the year (Figure B39) which aligned well with observations.

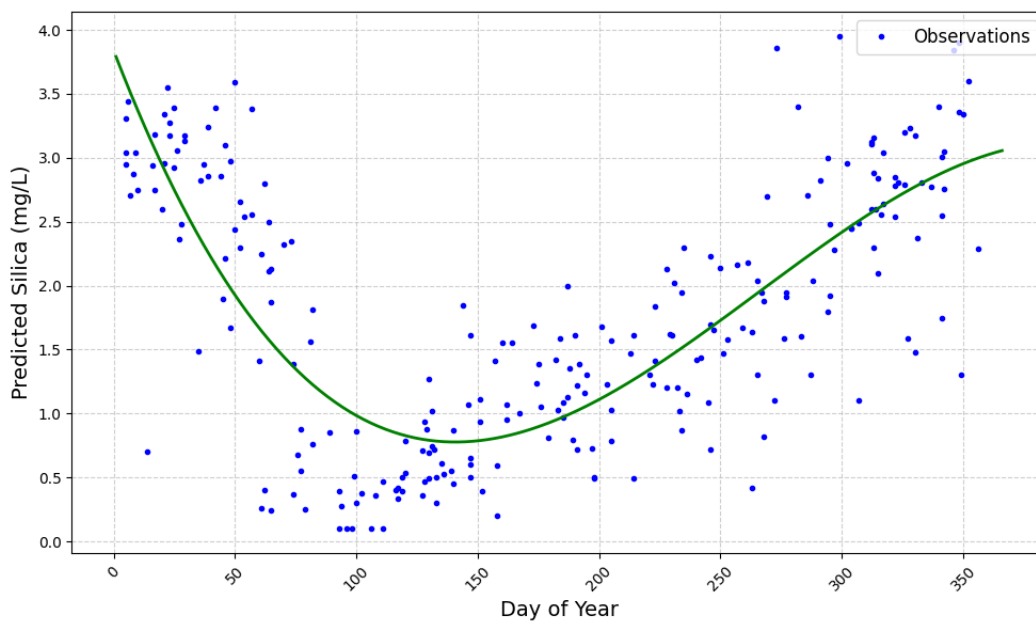


Figure B39: Individual dissolved silica measurements in Black Beck plotted against the day of the year. The fitted sinusoidal relationship is defined by the green line and provides a daily prediction of dissolved silica concentration.

2.4 PROTECH modelling

Meteorological, flow and water quality datasets were inputted to the PROTECH model. PROTECH used this information to simulate the response of the phytoplankton communities in each lake to the environmental conditions associated with the baseline, climate change and nutrient management scenarios.

PROTECH was run for five-year scenario periods, but the Nutrient Futures for Windermere report presented results pertaining to four-year scenario periods, as the first year of outputs were discounted to enable the model to adjust to its initial seeding conditions.

One of the principal PROTECH outputs was total chlorophyll *a* concentration, a proxy for phytoplankton biomass. From this, the chlorophyll *a* attributed specifically to cyanobacteria could be determined, and used to identify the number of days when the World Health Organisation (WHO) derived risk thresholds of 10 µg/L (WHO10) and 50 µg/L (WHO50) were exceeded (Chorus and Bartram, 1999). This enabled quantification of the risk associated with cyanobacteria biomass. Total chlorophyll *a* concentration was also used to determine the WFD status for each lake, as described in Section 5.2.2.

To validate the total chlorophyll *a* concentration outputs from PROTECH, the 2015-2018 modelled means were compared to their observed equivalent, using UKCEH long-term monitoring data (Table B4). To create an estimate for observed cyanobacteria chlorophyll *a* concentration, the algal count data were converted to estimated biovolumes and the total chlorophyll *a* concentration was multiplied by the proportion of the biovolume that was made of cyanobacteria. This provided an estimate that could be compared to the WHO10 and WHO50 thresholds. As the observed samples were taken every two weeks, the annual mean number of sample days where the cyanobacteria chlorophyll *a* concentration was greater than the threshold values was multiplied by 14. This gave a value more comparable to PROTECH's daily cyanobacteria chlorophyll *a* concentration estimates.

Both the modelled and observed total chlorophyll *a* concentration data were positively skewed and so were transformed by \log_{10} to approximate to 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 ($P > 0.05$).

Further outputs from the PROTECH model included lake water temperature (calculated from air temperature input data) 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. Mixed depth is influenced by (and calculated using) wind speed at the lake surface, water density and water temperature. Mixed depth was analogous with duration of stratification in each lake (absence of full lake mixing).

Table B4: Comparison of total chlorophyll a concentration ($\mu\text{g/L}$) of the whole phytoplankton biomass averaged across the four-year baseline modelling period and the mean annual number of days cyanobacteria chlorophyll a concentration exceeded the WHO10 threshold ($10 \mu\text{g/L}$) from 2015-2018 for both observed (UKCEH long-term monitoring) and modelled (PROTECH) data. Values are provided for Windermere North Basin, Windermere South Basin and Esthwaite Water.

	Data source	Mean total chlorophyll a concentration ($\mu\text{g/L}$)	Days per year > WHO10 ($\mu\text{g/L}$)
Windermere North Basin	Observed	6.1	14.0
	Modelled	5.0	8.0
Windermere South Basin	Observed	7.9	42.0
	Modelled	6.0	49.5
Esthwaite Water	Observed	18.3	126.0
	Modelled	18.6	158.5

3. Appendix C: Climate change scenario

This appendix describes the approach taken to simulate the climate change scenario.

While the Nutrient Futures for Windermere report presents findings pertaining to a climate change scenario representative of environmental conditions in the far future (2076-2079), a set of model runs were also produced using the same methodology, but to show how climate change will impact lake phytoplankton communities in the near future (2035-2039). Table C1 illustrates total phytoplankton biomass across the baseline, near-future and far-future scenarios. The nutrient management scenarios were only run for the far-future period (2076-2079), to illustrate the potential worst-case scenario regarding changes to the lake phytoplankton communities, and to determine the efficacy of the nutrient reduction scenarios in addressing this. Therefore, the near-future modelling is discussed in this appendix for contextual information but not included in the Nutrient Futures for Windermere report. The scenario termed 'far future' in this appendix refers to the climate change scenario presented in the Nutrient Futures for Windermere report.

Table C1: Comparison of PROTECH modelling output parameters for Windermere North Basin, Windermere South Basin and Esthwaite Water in the baseline, near-future (2035-2039) and far-future (climate change) (2076-2079) scenarios. Tchl = total chlorophyll a concentration (µg/L) of the whole phytoplankton biomass averaged across each modelling period, WHO10 = mean annual number of days the WHO10 threshold was exceeded.

		Windermere North Basin	Windermere South Basin	Esthwaite Water
Tchl	Baseline	5.02	6.01	18.60
	Near future	5.92	6.54	17.50
	Far Future	5.85	6.84	18.10
WHO10	Baseline	8	50	159
	Near future	34	74	149
	Far Future	36	92	167

3.1 Description of the approach taken to simulate climate change

This study used PROTECH to generate simulations of future lake phytoplankton communities. The impact of climate change was not modelled in PROTECH itself but was instead provided in the model inputs. Climate change will impact the meteorological conditions which are experienced in each catchment, and the flow and water quality of inputs that drain to each lake. To capture the uncertainty of climate projections, different Global Climate Models (GCMs) were used, different ensemble runs of the same model conducted, and different climate scenarios simulated. Downscaling global climate model data to a single location adds additional complexity to projections for that location.

This study considered three main aspects of change when adopting an approach to creating future input conditions:

1. There may be changes to absolute thresholds of conditions (e.g., more days where a particular absolute temperature threshold may be reached).
2. There may be changes in the number of extreme events (e.g., change in the frequency and magnitude of flooding or drought events).
3. There may be changes to the sequencing of environmental conditions that occur each year (e.g., warmer conditions may be experienced earlier in the year).

These three aspects all impact PROTECH projections, which build an ecological community based on time-series data, with each timestep building upon preceding timestep calculations.

Two approaches to improving our understanding of how environmental conditions might change in the future were considered. The first approach was to project future conditions with all three of the aspects above already factored in (i.e., directly use downscaled GCM projections for meteorological and hydrological conditions). This would provide an actual projection of what would be expected in the future, incorporating all three aspects of change. However, climate projections are uncertain, and selecting a single set of conditions to input to PROTECH would only provide information about the response to that specific scenario. The second approach was to target one aspect of climate change and understand what that does to lake processes. For example, by using monthly climate uplift factors for a known time series of data, valuable information could be gained on how long-term climate change would impact baseline conditions, although changes to extreme events and event sequencing would be discounted.

The second approach was chosen because (1) it formed a result that could be interpreted to explain why a change may happen, (2) future conditions could not be projected accurately enough to validate using a single climate scenario, and (3) there was too much uncertainty in how water quality will change into the future to attempt to simulate conditions in a specific scenario. Explanation of process was chosen over a more precise projection.

3.1.1 Air temperature

United Kingdom Climate Projections 2018 (UKCP18) provided data on air temperature, wind speed, and cloud cover projections at 2.2 km resolution. These were also re-gridded at 5 km resolution and aggregated at country, administrative and river basin level.

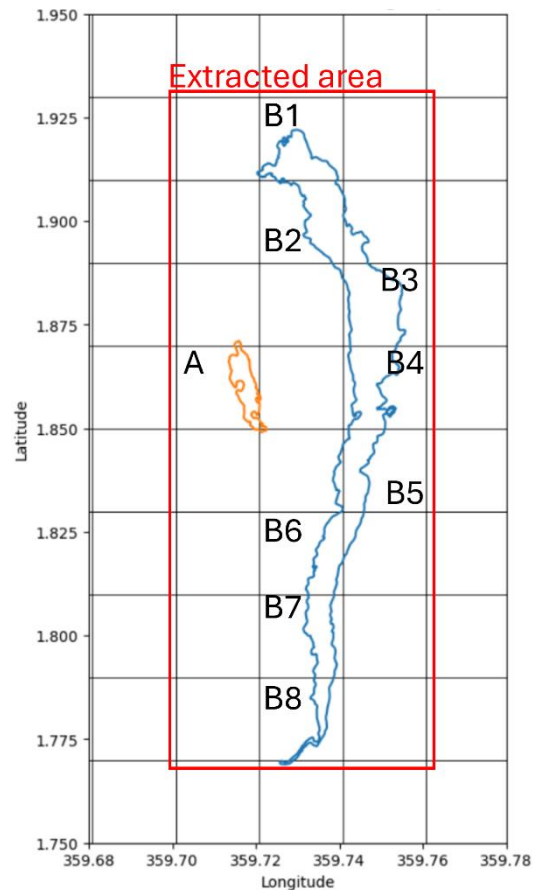


Figure C1: Location of Windermere (blue) and Esthwaite Water (orange) shapefiles on the rotated pole grid of the 2.2 km projections.

To resolve Windermere and Esthwaite Water differences adequately, either the 2.2 km or 5 km gridded data was necessary. This spatial resolution was particularly important in the high relief around the study area. The 2.2 km gridded data was provided on a rotated pole grid (pole longitude = 177.5°, pole latitude = 37.5°) and clearly separated Windermere and Esthwaite Water (Figure C1). In comparison, the 5 km gridded data was reprojected to the British National Grid. Whilst able to capture the Windermere catchment well, there was not a clear separation for Esthwaite Water from this dataset. The effect this had on the temperature projections is tested below. Both gridded records contained the full dataset from December 1980 to November 2080.

Modelled projections contain systematic differences from observations due to differences in how environmental processes are simulated. Numerous bias correction methods exist (Met Office, 2018) to correct future conditions based on how well the model predicts the time periods for which observations exist. This study used one of the simplest of the bias

correction methods for air temperature, the delta change approach. This approach (Equation C1) calculated a monthly factor difference between two modelled ten-year periods, one covering the baseline period and one covering the future period of interest (2076-2079). This monthly factor was then applied to each date in the time series (assigned to the month).

Equation C1
$$T_{Future} = T_{ObservedPresent} + (T_{ModelledMean(Future)} - T_{ModelledMean(Present)})$$

This method did not provide any bias correction for extreme events or the frequency of certain events in the time series; it simply took the mean monthly change expected between the two time periods and applied it to the time series. The main advantage of this as a PROTECH model input was that the sequencing of conditions throughout the year was based on an observed sequence.

Using the JASMIN Jupyter notebook system, the daily 2.2 km and 5 km gridded climate data was extracted for the red areas highlighted in Figure C1 for the time periods shown in Table C2. This was done by iteratively looping through the larger netCDF files with a mask over the latitude 1.765–1.930° and longitude 359.70-359.76°. Data from all 12 UKCP18 ensembles were extracted. For each ensemble, the mean value was determined for each month. The UKCP18 projections were in 360 day-year format, so the impact of leap-years on averages did not need to be accounted for.

Table C2: Target dates for baseline, near-future and far-future (climate change) scenario periods and the extraction dates for UKCP18 data for those periods.

Period	Target dates	Start date of extraction	End date of extraction
Baseline	2014-2018	01/12/2009	30/11/2019
Near future	2035-2039	01/12/2029	30/11/2039
Far future	2075-2079	01/12/2069	30/11/2079

For Windermere, numerous 2.2 km grid squares could be taken to represent the lake climate conditions. Figure C2 highlights how the median value changed between each of these grid squares. The variation was smaller than between different model ensembles, hence a representative grid square could be used as a proxy for the lake in its entirety.

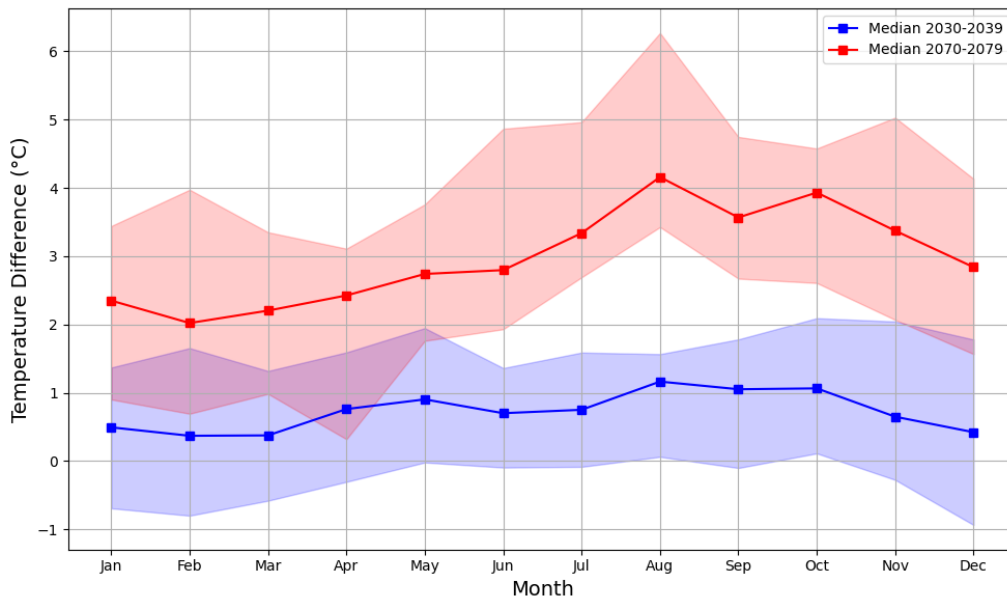


Figure C2: Variability in median ensemble temperature differences between near-future (blue line) and far-future (climate change) (red line) scenarios and the baseline scenario for each Windermere grid cell in Figure C1.

For Esthwaite Water (Figure C1, Cell A), the results are shown in Figure C3.

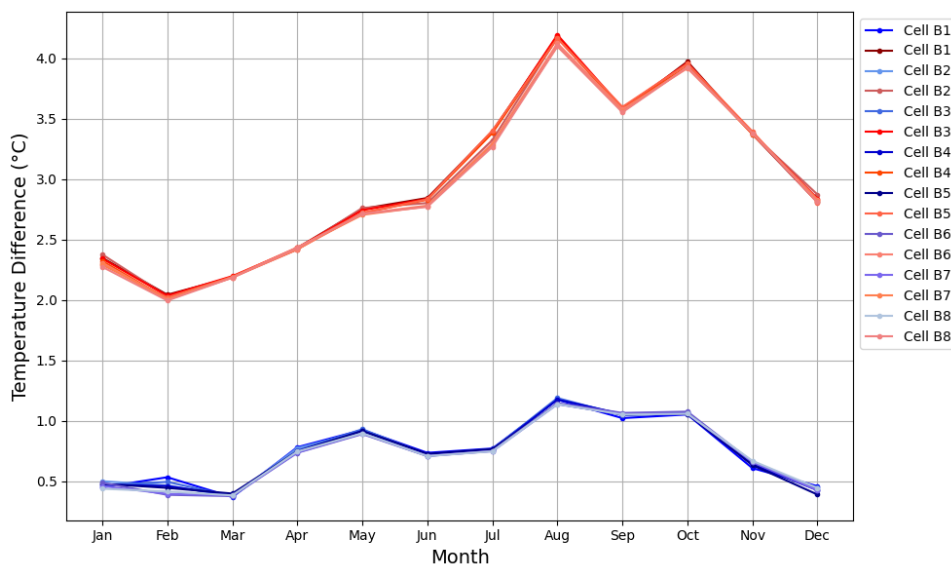


Figure C3: Temperature difference for the near-future (blue line) and far-future (climate change) (red line) scenario periods compared to the baseline scenario period, aggregated by month, for Esthwaite Water. The solid lines indicate the median ensemble value, and the shaded area reflects the ensemble range.

To additionally ensure correct transformation of the rotated pole coordinate system and to test for the level of variability across climate projections, the 5 km grid square projections were additionally extracted for UKCP18 data. The 5 km data was in ten-year increments, so an extra filtering step was needed to obtain the data between the start and end dates. The data were essentially incomparable for a 5 km and 2.2 km grid square, as would be expected from a correctly transformed grid surface (Figure C4).

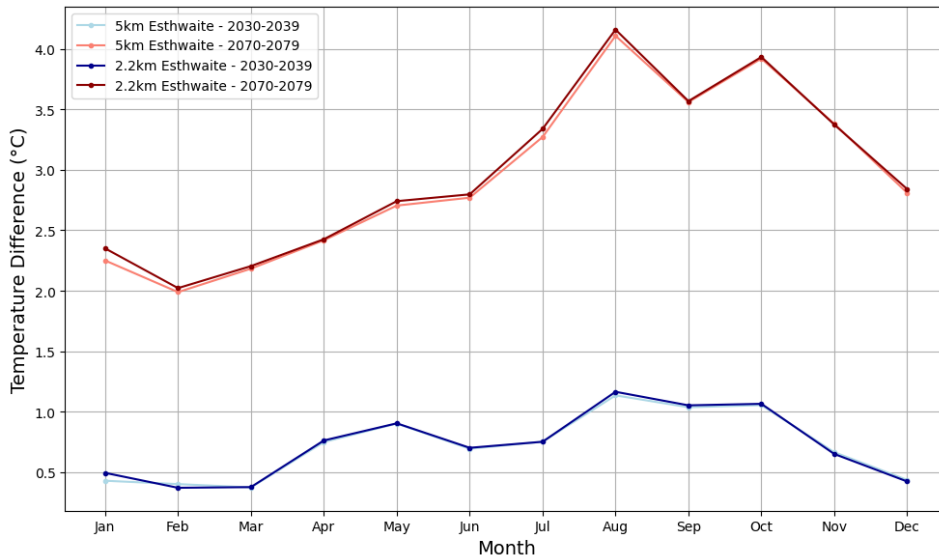


Figure C4: Variability in median ensemble temperature differences between near-future (blue lines) and far-future (climate change) (red lines) scenarios and the baseline scenario for the 2.2 km and 5 km grid cells which cover Esthwaite Water (Esthwaite).

The monthly factors generated from grid cell A and B5 (taken from the middle of Windermere) (Figure C1) were applied to the meteorological sonde data for Esthwaite Water and Windermere respectively for both the near-future and far-future (climate change) scenario periods (Figure C5). Temperature increases were observed in both time periods, with anticipated changes larger from August to October.

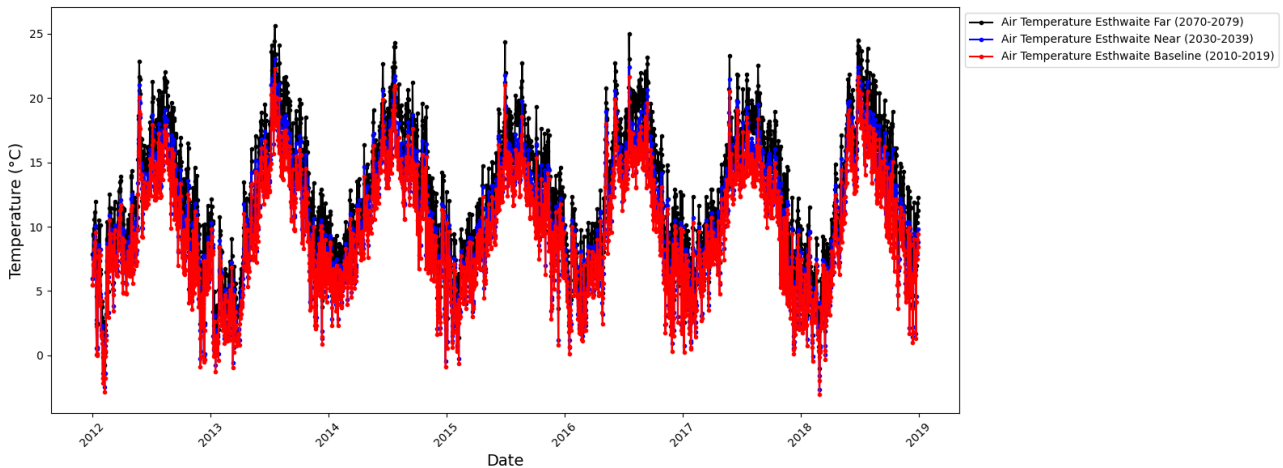


Figure C5: Air temperature time series amended for the near-future and far-future (climate change) scenarios using the delta change method. These are the PROTECH inputs for the baseline, near-future and far-future (climate change) scenarios.

3.1.2 River temperature

While not a direct input to PROTECH, river temperature was a global input variable in the SIMCAT-SAGIS model, controlling the rate at which nutrients decay during simulation

down all rivers within a catchment. The best estimates of future river water temperatures were provided by the Environment Agency (EA, 2024b), utilising the same UKCP18 ensemble approach for the high emissions Representative Concentrations Pathway 8.5 (RCP8.5) scenario (EA, 2024b).

Future river water temperature projections were provided at numerous monitoring points on rivers across England. A catchment average temperature from all monitoring points that feed into Windermere and Esthwaite Water (Figure C6) for each period (based off the periods defined in Table C2) was calculated (Table C3). The variation in projected temperature was minor between the individual points within the region (Figure C7).

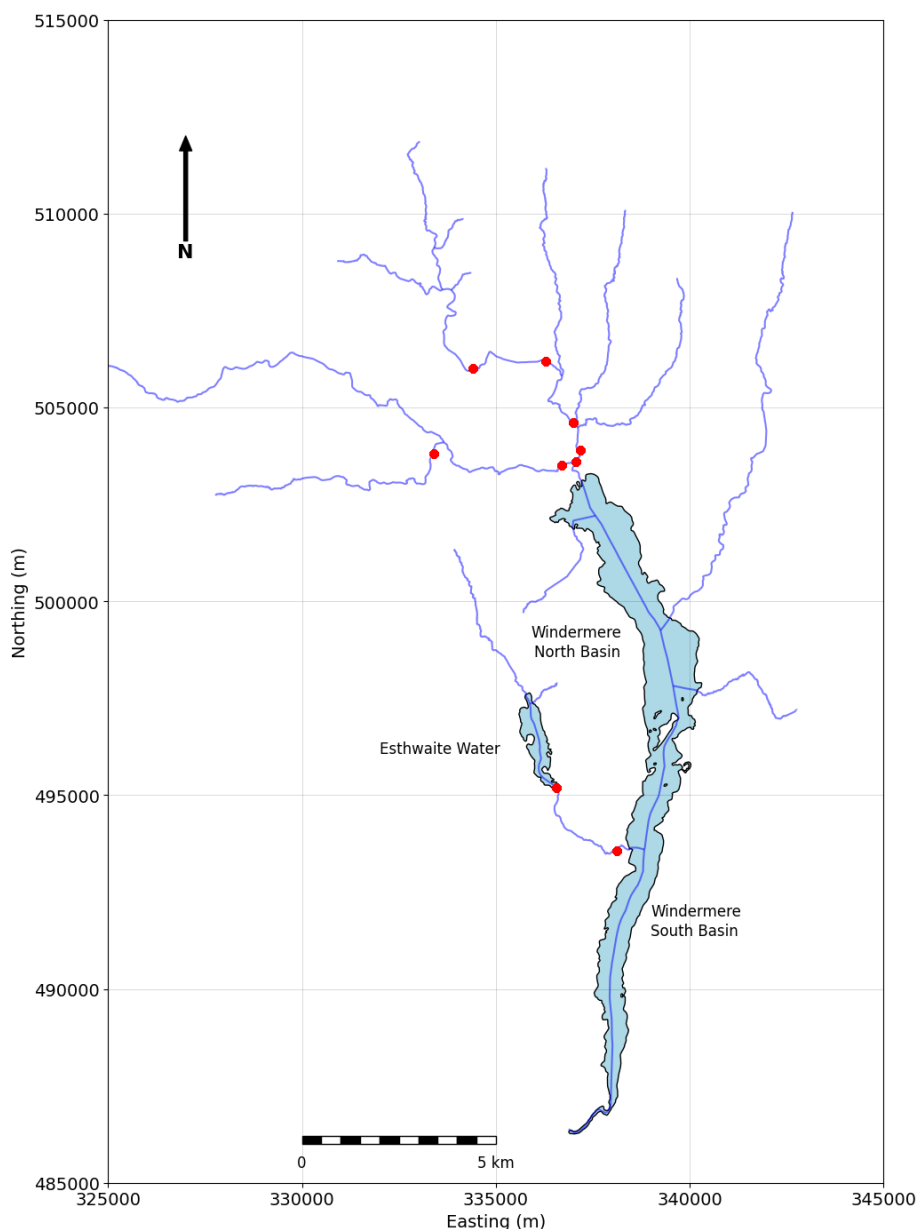


Figure C6: Location of monitoring points (red dots) where river water temperature projections are provided from an Environment Agency study (EA, 2024b).

Table C3: Mean river water temperatures projected for the baseline, near-future and far-future (climate change) scenarios on an annual and monthly basis.

Time period	Baseline (°C)	Near future (°C)	Far future (°C)
Annual	10.46	11.02 (+0.56)	12.78 (+2.32)
January	5.64	5.88	7.21
February	5.64	5.93	7.03
March	7.73	7.94	9.33
April	9.34	9.94	11.16
May	12.10	12.72	14.07
June	14.55	15.03	16.91
July	16.04	16.91	19.15
August	16.15	16.96	19.87
September	13.53	14.20	16.35
October	10.74	11.53	13.63
November	8.03	8.71	10.68
December	5.99	6.46	7.93

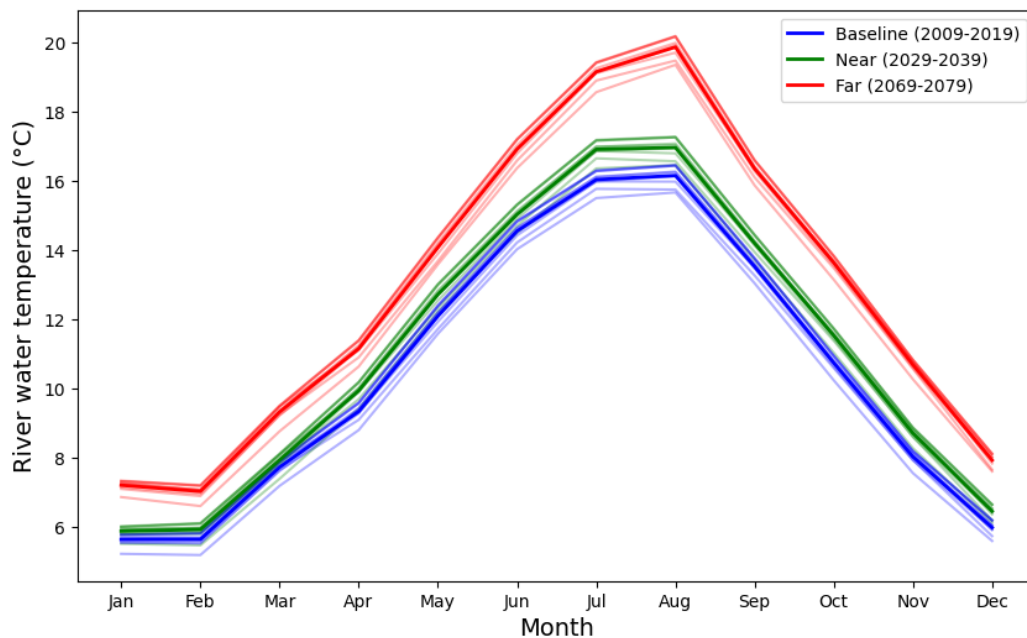


Figure C7: River water temperature projections for baseline, near-future, and far-future (climate change) scenarios as a monthly median value (solid lines) for inflowing rivers to Windermere and Esthwaite Water.

3.1.3 Wind speed

Wind speed was too variable for the delta change method to be appropriate. Comparing the probability distributions between baseline observations and baseline projections, this approach tended to overpredict in the 2-4 m/s range and underpredict in the 5-8 m/s range (Figure C8).

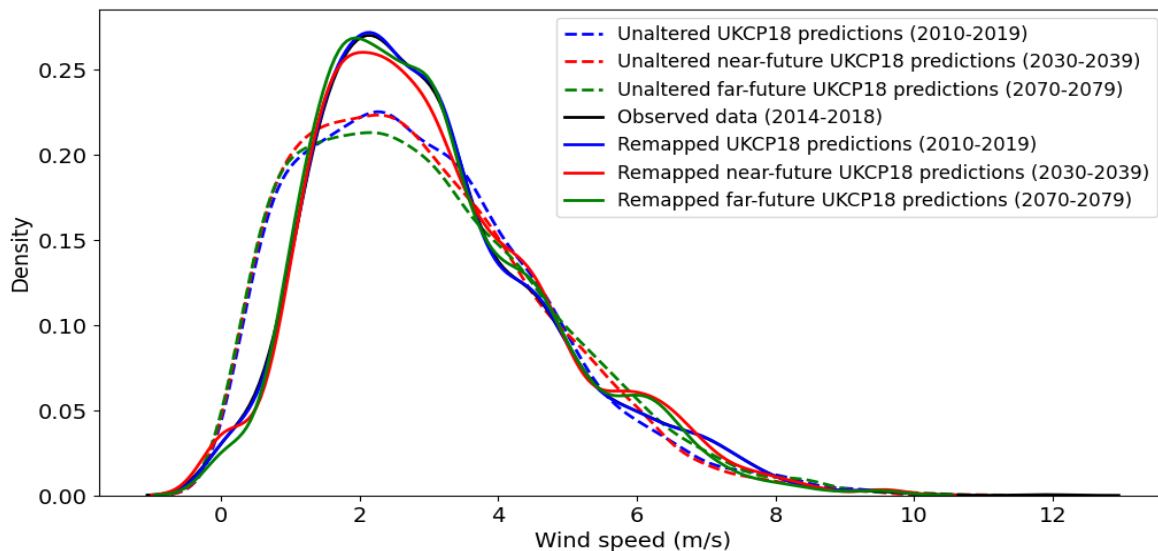


Figure C8: Histogram of wind speed at the (A) Windermere South Basin sonde (observed data) and the UKCP18 projections at the grid cell B5 (as a central marker for Windermere) (Figure C1), for periods covering the (B) baseline, (C) near-future and (D) far-future (climate change) scenarios.

The distribution of wind speed was skewed towards the high end of values. Therefore, the quantile mapping was scaled so that the high end was split at higher resolution (50 quantiles between 0.95 and 1.00 in the probability distribution). In addition, whilst an interpolation method was used for values within percentiles, a linear extrapolation method was used for the uppermost percentile, to allow for events beyond the baseline maximum. The results of this remapping are shown in Figure C9.

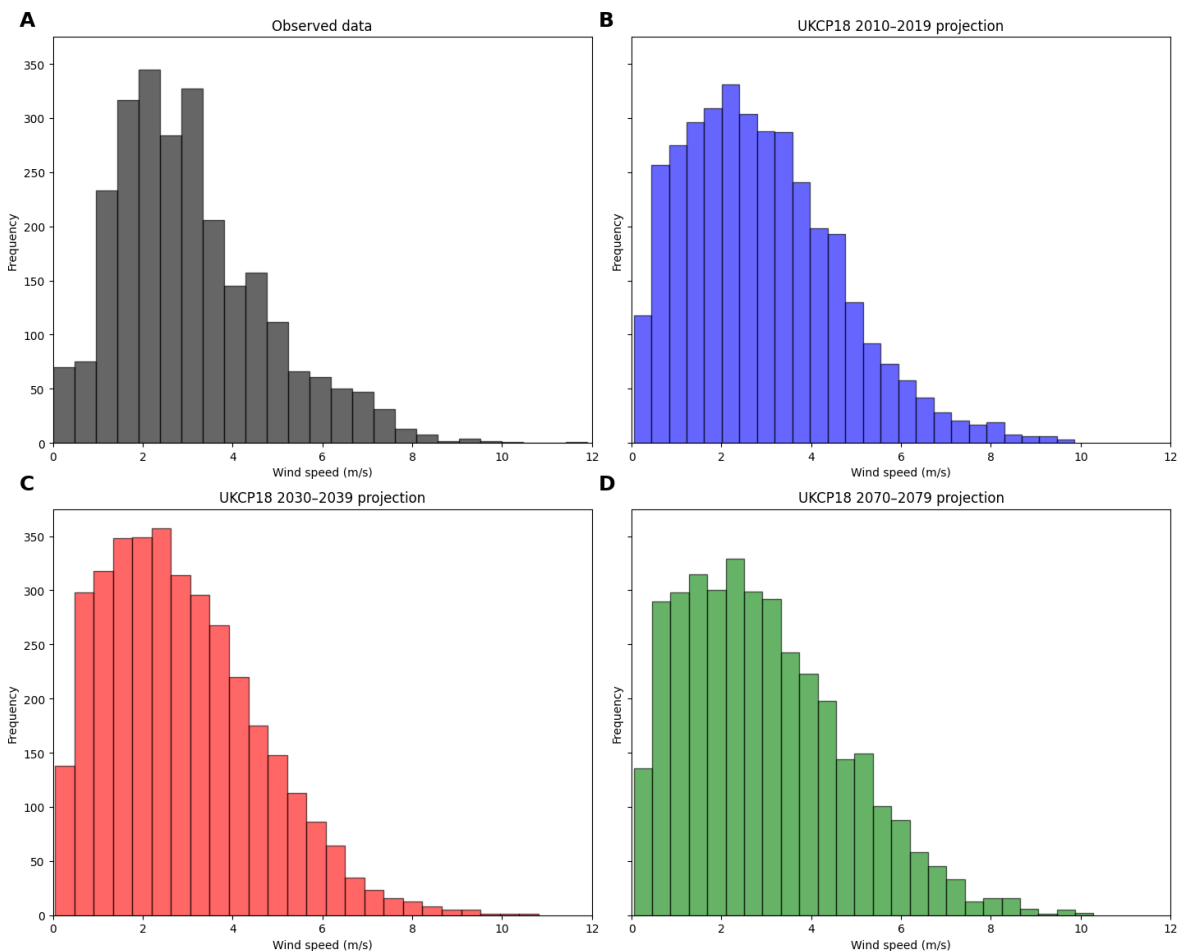
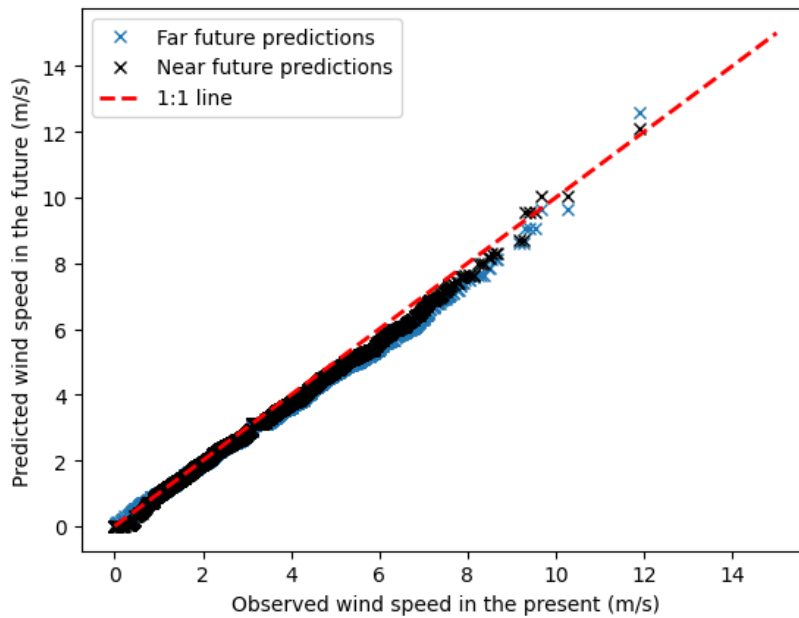


Figure C9: Kernel density estimation plot showing the distribution of wind speed data for unaltered UKCP18 projections, observed data, and remapped UKCP18 projections, for periods covering the baseline, near-future and far-future (climate change) scenarios.

Ensemble members could not be averaged as this would have remove some of the intrinsic variability within ensembles. An ensemble was chosen (RCM05) based on the similarity of the median, mean, interquartile ranges and the high endmembers (which had a very similar range to baseline observations following remapping) (Figure C10).

Figure C10: Box and whisker plots to compare the remapped baseline, near-future, and far-future (climate change) scenario wind speed datasets for the RCM05 ensemble. The ensemble with the best fit of the remapped baseline to the observed data was chosen.



The future projections were then quantile mapped on each tenth of a percentile onto the current time series to ensure sequencing remained the same. Noticeable differences due to climate change were a decrease in the middle-to-high wind speed events but a minor increase in the intensity of the most extreme events (Figure C11).

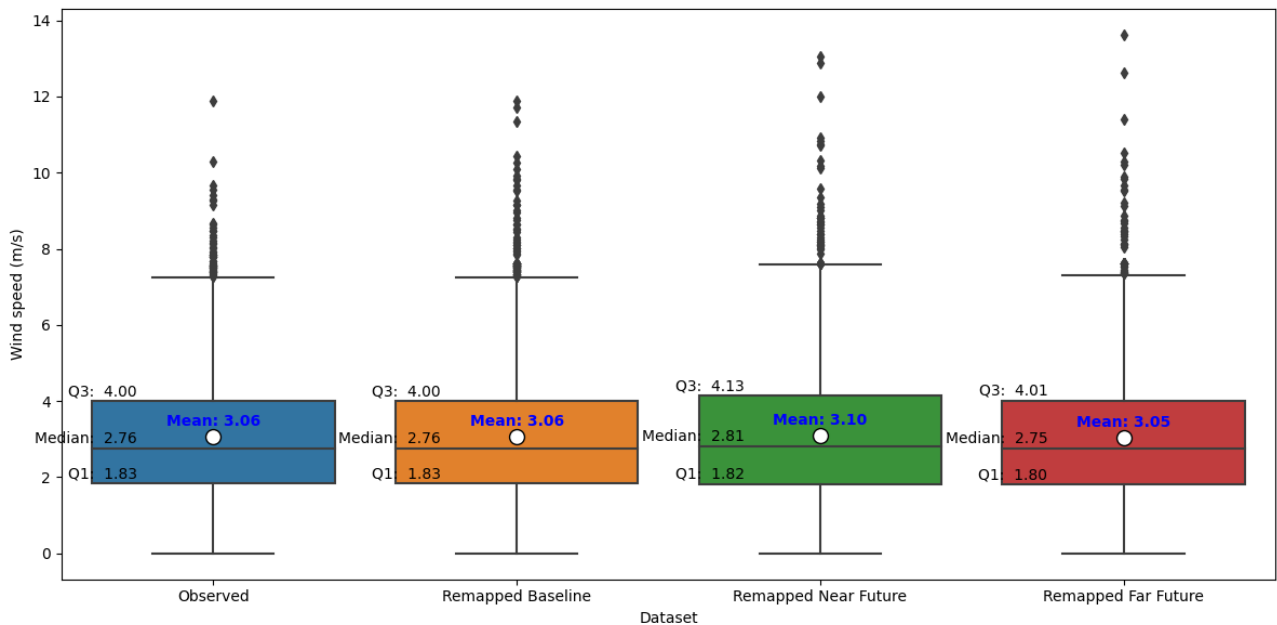


Figure C11: Comparison of near-future and far-future (climate change) scenario wind speed projections across the range of wind speed values. Baseline scenario values are represented by the 1:1 red line.

3.1.4 Cloud cover

The delta change method would have been ineffective for the cloud cover parameter because there was unlikely to be a consistent bias throughout the year. In addition, cloud cover was non-gaussian, with more events occurring at high (100%) and low (0%) cloud cover.

The statistical distributions of observed data from the Morecambe No 2 MIDAS station and contemporary model projections were sufficiently different to hinder direct use of the UKCP18 projections (Figure C12). UKCP18 tended to overestimate the middle percentiles and underestimate the 100% cloud cover days. There were more significant differences between baseline observations and baseline model projections, than between baseline model projections and future model projections. Therefore, unbiased use of climate projections may have reflected modelling artefacts more than long-term change.

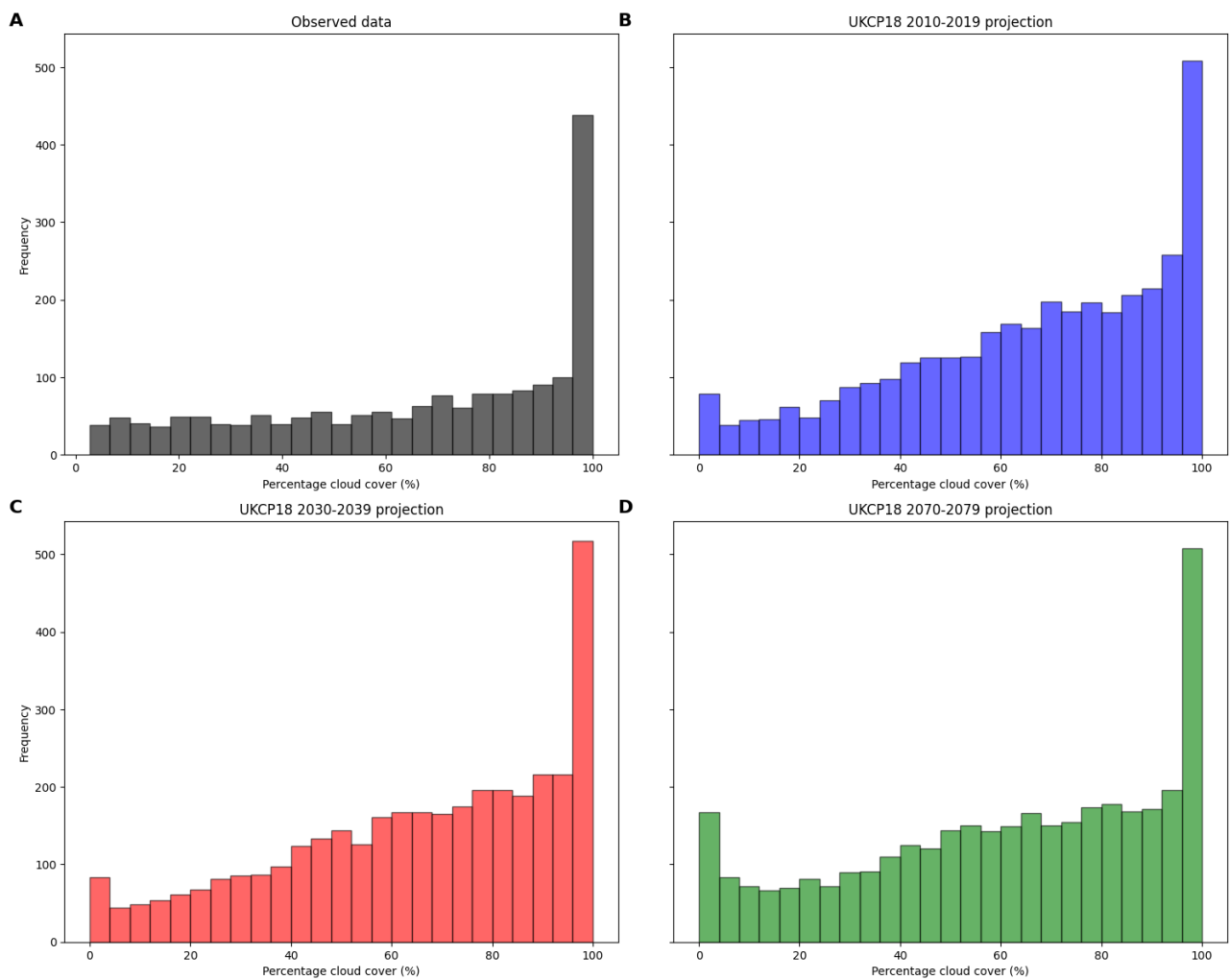


Figure C12: Histogram of percentage cloud cover at the (A) meteorological station (Morecambe No 2) (observed data) and the UKCP18 projections at the grid cell B4 (as a central marker in the region) (Figure C1), for periods covering the (B) baseline, (C) near-future and (D) far-future (climate change) scenarios.

Quantile mapping provided a means of bias-correction by adjusting the statistical distribution of the modelled data to the observed data at specific quantiles. Deciles were determined for both the modelled baseline data (2014-2018) and observed baseline data (2010-2019). The statistical distribution of the modelled data was then remapped onto the statistical distribution of the observed data, providing a bias-corrected model projection. The same transformation was then applied to the near-future and far-future (climate change scenario) model projections.

The results of this remapping for a single ensemble (RCM05) are shown in Figure C13. Whilst previous model projections (dashed lines) often over project middle-percentage cloud cover days, the bias corrected baseline projections were closely aligned with observed data, only deviating significantly at the highest end. Climate impacts in the far future (a decrease in high cloud cover days and an increase in low cloud cover days) were correctly transferred to the remapped climate projections.

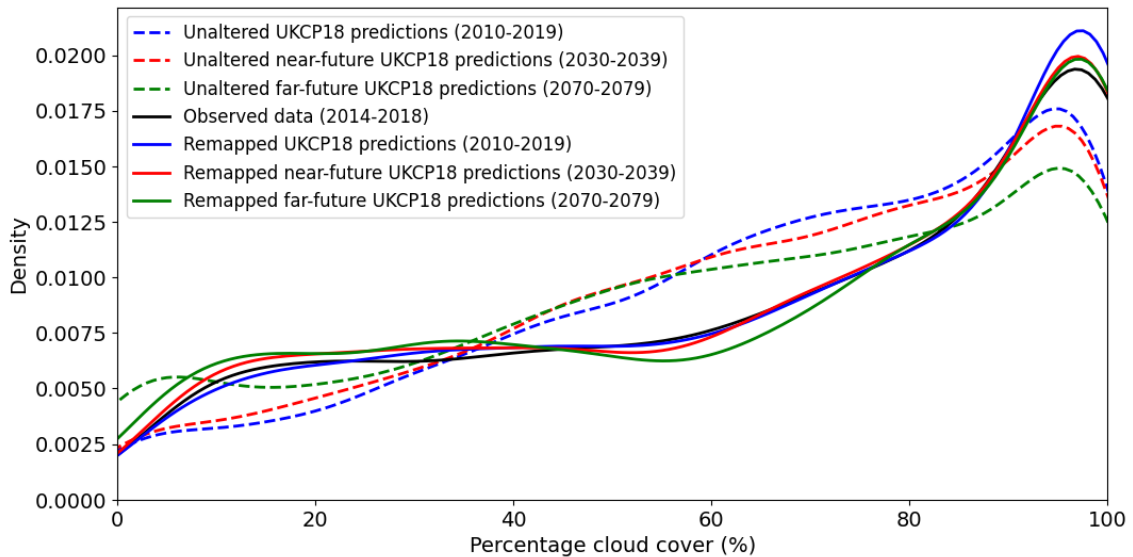


Figure C13: Kernel density estimation plot showing the distribution of percentage cloud cover data for unaltered UKCP18 projections, observed data, and remapped UKCP18 projections, for periods covering the baseline, near-future and far-future (climate change) scenarios.

UKCP18 uses a 12-member ensemble approach. For many metrics, a mean or median of the ensemble projections can be used to provide a representative pathway. However, for cloud cover, averaging ensemble projections removed the high and low ends of the dataset (removing 100% cloud cover days altogether). Therefore, a single ensemble was chosen (RCM05) based on closeness of the modelled and observed datasets following remapping (Figure C14). This assumed that a model which performs well in the baseline continues to perform well in future.

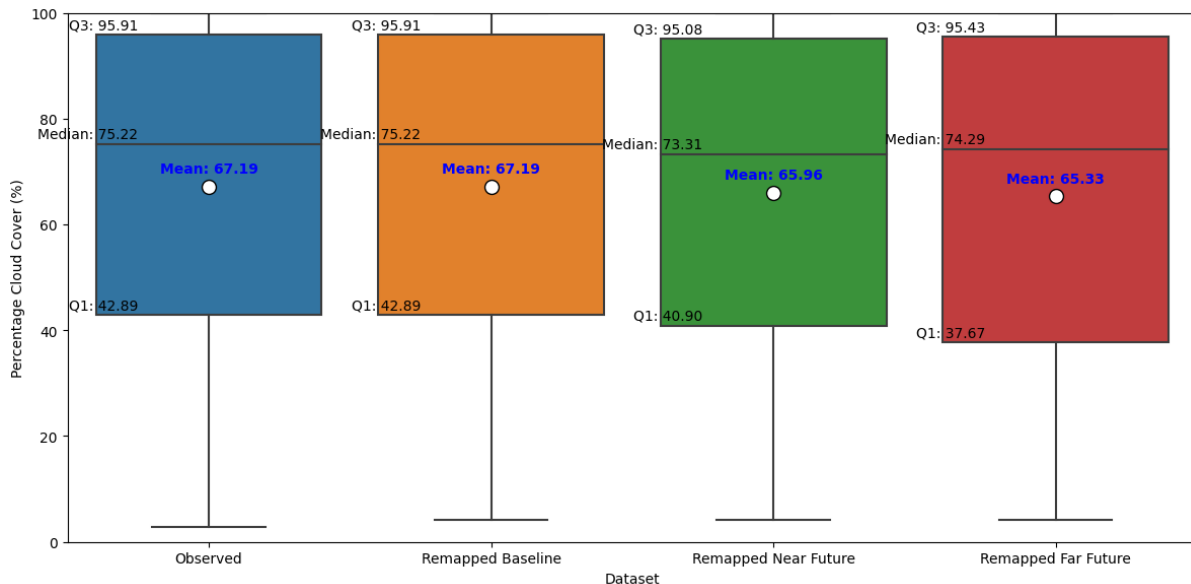


Figure C14: Box and whisker plots to compare the remapped baseline, near-future, and far-future (climate change) scenario cloud cover datasets for the RCM05 ensemble. The ensemble with the best fit of the remapped baseline to the observed data was chosen.

The future projections were then quantile mapped on each individual percentile onto the current time series to ensure sequencing remained the same. There were differences in 100% cloud cover events in the baseline and future time periods, with 100% cloud cover days anticipated to occur 42% less of the time in 2070. To account for these differences, a random chance, based on this proportional decrease, was assigned to each baseline 100% cloud cover event as to whether it was decreased to 99%. This changed the event from okta 8 to okta 7 classification, so the decreases in complete cloud-cover events could be accounted for. The result was a corrected time-series dataset from 2014-2018. The sequencing remained the same as the 2014-2018 MIDAS station data, but the data were corrected based on the expected changes between the baseline and future UKCP18 model projections. Observed cloud cover was projected to be lower than baseline in both near-future and far-future (climate change) scenarios and the number of 100% cloud cover days decreased in both scenarios (Figure C15). For input into the PROTECH model, percentage cover was converted into oktas.

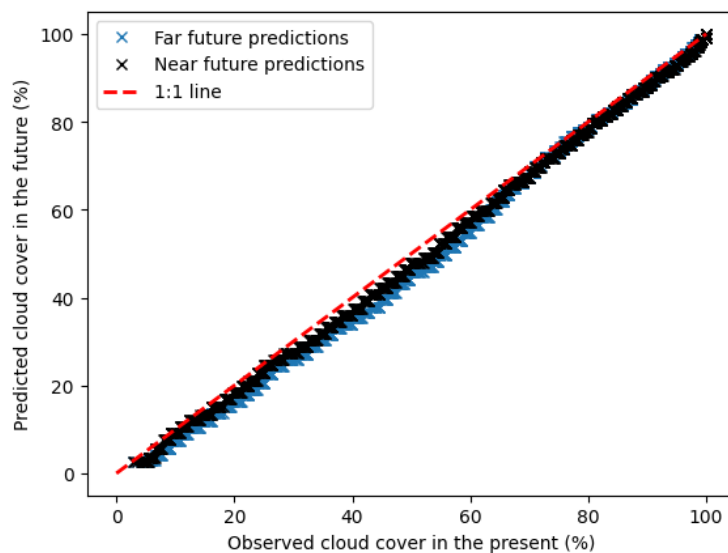


Figure C15: Comparison of near-future and far-future (climate change) scenario cloud cover percentage projections across the range of cloud cover percentage values. Baseline values are represented by the 1:1 red line.

3.1.5 Water quality

3.1.5.1 Phosphate and nitrate

There were three ways in which climate change could impact the concentration of nutrients using SIMCAT-SAGIS.

1. River temperature: The mean river temperature change in the SIMCAT model can impact the decay rate of nutrients. This factor can decrease nutrients during the

simulation of a river and is 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 (EA, 2024b), which is based on the same UKCP18 climate data that underpins the air temperature projections in this study. This effect can alter the load which is provided to the lake (higher temperature = lower load). The river temperature used for the baseline scenario run was 10.46 °C, for the near-future run was 11.02 °C and for the far-future (climate change scenario) run was 12.78 °C.

2. Diffuse scaling: The monthly variation in flow can alter how the diffuse nutrient load is distributed throughout a month. This is due to the scaling factor used in the baseline scenario. This is automatically adjusted with the updated flow time series. This change does not alter the summed load which is provided over a month but can change how much load is supplied each day of a month.
3. Intermittent thresholds: Variations in river flow mean that intermittent discharges will trigger more or less frequently in future in the model scenarios. The threshold for a trigger is based on the baseline flow percentile. Climate scenarios can have a greater or lesser number of days which exceed this flow percentile, so there may be more or fewer spill events in future. It is assumed that each spill contributed a uniform amount of nutrient load, while in reality this will vary.

Each of these three effects were incorporated to generate future nutrient concentrations that aligned with the future flow time series.

3.1.5.2 Dissolved silica

Dissolved silica was assumed to stay the same in future scenarios. In the absence of a catchment model and lack of a meaningful catchment-specific relationship to draw upon, this approach was considered the most conservative.

4. Appendix D: Nutrient management scenarios

The following information describes how the three nutrient management scenarios were modelled for this study. These scenarios amended the climate change scenario and represented lake conditions expected for 2076-2079. Amendments made in SIMCAT-SAGIS altered the water quality outputs (phosphate and sometimes nitrate) that were then inputted to PROTECH, to determine the impact of these changes on the lake phytoplankton community.

4.1 Nutrient management scenario 1: AMP8

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

Planned AMP8 improvements to TP concentrations for the Windermere catchment are detailed in the WINEP (DEFRA, 2022). These were included in the SIMCAT-SAGIS model as phosphate and subsequently inputted to PROTECH as SRP. 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.

4.1.1 Amendment of direct discharges

The direct discharges for Ambleside, Langdale, Outgate, Near Sawrey and Hawkshead WwTW were amended so that the mean concentration equalled their permit limits (Table D1) (with a coefficient of variation of half the mean). This assumed a worst-case scenario: that each WwTW operated at the maximum allowable discharge concentration. As these permit limits referred to a TP limit, a conservative approach where the phosphate concentration equals that of TP was adopted. As Grasmere WwTW was already operating below the improved permit limit in the baseline scenario, this remained unchanged.

For Far Sawrey WwTW there was no current permit limit for dry weather flow nor any form of phosphorus (Table D1). The permit documents stated that Far Sawrey WwTW served a population of less than 250 people. Assuming usage of 142 litres per day per person and multiplying by the permit limit concentration of 0.5 mg/L, this resulted in a maximum loading of 0.0178 kg/day of phosphorus, 16% of the previous loading simulated in the model (0.1093 kg/day). This was amended in the Direct Loads table for Windermere South Basin.

For Ferry House WwTW, the current maximum dry weather flow was 0.025 megalitres per day. A conservative approach was to assume that the mean flow value corresponded to the dry weather flow value (or the flow at the 20th percentile or Q80). This provided a

phosphorus loading of 0.0125 kg/day, 17% of the previous loading simulated in the model (0.0756 kg/day). This was amended in the Direct Loads table for Windermere North Basin.

For Troutbeck WwTW the current maximum dry weather flow was 0.017 megalitres per day. A conservative approach was to assume that the mean flow value corresponded to the dry weather flow value (or the flow at the 20th percentile or Q80). This provided a phosphorus loading of 0.0085 kg/day, 7% of the previous loading simulated in the model (0.128 kg/day). This was amended in the Direct Loads table for Windermere North Basin.

Table D1: AMP8 changes to TP permit limits for sewage treatment works in the Windermere catchment.

Sewage treatment works	Pre-AMP8 TP limit (mg/L)	AMP8 TP limit (mg/L)
Ambleside	0.50	0.25
Far Sawrey	None	0.50
Ferry House	None	0.50
Grasmere	0.30	0.25
Hawkshead	1.00	0.25
Langdale	2.00	0.25
Near Sawrey	None	0.50
Outgate	2.00	0.50
Troutbeck	None	0.50

4.1.2 Amendment of storm overflows

Storm overflows were amended in post-processing of the SIMCAT-SAGIS files. Each spill event from an individual storm overflow was assumed to contain the same amount of load. The spill frequency in the AMP8 scenario was set to ten per year (a spill frequency of 2.7%) for the following assets:

- Elterwater pumping station storm tanks
- Grasmere storm tanks
- Ambleside storm tanks
- Glebe Road CSO
- Hawkshead storm tanks
- Near Sawrey storm tanks

The spill rate at Glebe Road was modelled below this threshold in baseline modelling, so this input remained unchanged. Each asset was assumed to trigger when the river flow exceeded the percentile of spill frequency (in this case, days where flow was in the top 2.7% of flows) over the scenario period (an average of ten spills a year).

4.2 Nutrient management scenario 2: Low agricultural diffuse

This scenario simulated the impact of maximum potential reductions in phosphate and nitrate inputs from agriculture on each lake phytoplankton community in future (2076-2079), assuming water quality improvements detailed in the AMP8 scenario and changes projected in the climate change scenario were in effect.

FARM SScale Optimisation of Pollutant Emission Reductions (Farmscoper) simulations indicated that if all regulatory and voluntary land management measures were fully adopted within the Leven catchment, then TP could be reduced by ~40% and nitrate by ~15%, relative to a 2019 baseline. These TP values align to the Environment Act's statutory targets to reduce the load of TP, TN and sediment from agriculture in England by 40% by 2038, relative to a 2018 baseline (Environment Act, 2022).

The Farmscoper decision support tool was selected for its capability to model diffuse agricultural pollution mitigation measures at a catchment scale and estimate theoretical maximum reductions in agricultural nutrient inputs. The tool is built upon a suite of well-established national nutrient models (e.g., PSYCHIC, NEAP-N) that have been peer-reviewed and used for government policy support across the UK (Willows and Whitehead, 2015). Farmscoper has several limitations, including the lack of site-specific detail (e.g., soil texture, slope, daily rainfall intensity) and its use of generalised assumptions for many farm practices (e.g., timing of fertiliser application and duration of animal grazing). While model uncertainty is influenced by factors such as the availability of local data (e.g., agricultural surveys and soils data), the Leven catchment is well parameterised, increasing confidence in its outputs.

To simulate this scenario, direct local loads to the lakes for arable and livestock components in SIMCAT-SAGIS were reduced for the Windermere catchment. To do so, the SIMCAT data file was directly amended. For each reach (or river), the headwater load (i.e. the nutrient load that was delivered throughout the river) was reduced by 40% for phosphate and 15% for nitrate. It was assumed there would be equivalent proportional reductions in TP and phosphate. Python scripts were used to produce the daily time-series nutrient information required by PROTECH. All other diffuse sectors were left unchanged from the AMP8 scenario.

4.3 Nutrient management scenario 3: No sewage

This scenario simulated the impact of removing all point source nutrient inputs to the phytoplankton communities of Windermere North Basin, Windermere South Basin and

Esthwaite Water in future (2076-2079), assuming the climatic changes projected in the climate change scenario were in effect. Inputs of phosphate and nitrate were removed from both water company and privately owned sewage treatment works, intermittent discharges and septic tanks, meaning the lakes only received diffuse nutrient inputs.

The point source inputs were removed in the post-processing steps of the SIMCAT-SAGIS workflow. A new SIMCAT file was not required as specific nutrient sources were isolated using the apportioned nutrient data, and a daily time series created from these, for each nutrient.

Turning off sewage or intermittent inputs to each lake would cause a small reduction in river discharge. To simulate this effect, only the natural flow outputs from Qube were used as hydrological inputs, and these same natural flows were used to determine the nutrient concentration time-series dataset. For direct inputs to Windermere South Basin, the additional discharge from Windermere WwTW was removed. This was added to the flow component in the baseline, climate change and AMP8 scenarios.

The sewage treatment work, intermittent discharge and septic tank components were removed from the riverine and direct load data. For this scenario, the nutrient load from Cunsey Beck (specifically the difference between the outflow of Esthwaite Water and the input point of Cunsey Beck to Windermere South Basin) was negative and had to be removed. It was negative because the decay factors on nutrients simulated by SIMCAT-SAGIS outweighed any simulated additional diffuse load. The decay factor typically reflects the loss of phosphate and nitrate to sediment along a river reach. Therefore, any input to Windermere South Basin was solely from direct loadings.

4.4 The nutrient management scenarios

The impact of the three nutrient management scenarios on modelled nutrient loads entering each lake are presented in the Nutrient Futures for Windermere report (Table 3A, 3B). Table D2A and D2B provide the equivalent information for nutrient concentrations.

Table D2A: Annual average phosphate concentrations 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.

	Phosphate concentration of lake inflows (mg/L)				
	Baseline	Climate change	AMP8	Low agricultural diffuse	No sewage
Windermere North Basin	0.026	0.034	0.030	0.028	0.009
Windermere South Basin	0.013	0.014	0.009	0.008	0.001
Esthwaite Water	0.024	0.029	0.026	0.023	0.013

Table D2B: Annual average nitrate concentrations 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.

	Nitrate concentration of lake inflows (mg/L)				
	Baseline	Climate change	AMP8	Low agricultural diffuse	No sewage
Windermere North Basin	0.632	0.713	0.731	0.697	0.511
Windermere South Basin	0.361	0.395	0.400	0.402	0.272
Esthwaite Water	0.534	0.595	0.595	0.579	0.440

Nitrate concentrations in the AMP8 scenario were unchanged from the climate change scenario for Windermere North Basin and Esthwaite Water as AMP8 did not alter catchment derived nitrate inputs (Table D2B). AMP8 scenario nitrate concentrations in inflows to Windermere South Basin increased from the climate change scenario (Table D2B), 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.

5. Appendix E: Results

This appendix provides information to support the results section of the Nutrient Futures for Windermere report.

5.1 Climate change impacts on lake phytoplankton communities

Table E1: The biomass (chlorophyll a concentration) of lake phytoplankton averaged across the baseline (Base) and climate change (2076-2079) (Climate) scenario periods for Windermere North Basin, Windermere South Basin and Esthwaite Water. The eight most dominant phytoplankton types recorded in UKCEH long-term monitoring phytoplankton count data for 2015-2018 were used to represent the phytoplankton community of each lake in PROTECH.

Phytoplankton group	Phytoplankton type	Phytoplankton biomass (mean chlorophyll a concentration $\mu\text{g/L}$) per lake and modelling scenario					
		Windermere North Basin		Windermere South Basin		Esthwaite Water	
		Base	Climate	Base	Climate	Base	Climate
Cyanobacteria					CC		
	Aphanizomenon	1.26	1.86	2.14	2.96	2.01	2.23
	Aphanothece clathrata					0.66	1.07
	Dolichospermum	0.47	0.57	0.86	1.09	6.63	6.78
	Planktothrix	1.37	1.50	0.78	0.76	0.89	1.02
Diatoms	Asterionella	0.39	0.36	1.87	1.65	3.27	3.06
	Aulacoseira	0.80	0.87	0.05	0.07		
Other algae	Chlamydomonas					1.58	1.31

Phytoplankton group	Phytoplankton type	Phytoplankton biomass (mean chlorophyll a concentration $\mu\text{g/L}$) per lake and modelling scenario					
	Chlorella					0.95	0.68
	Cryptomonas	0.21	0.19	0.01	0.01		
	Monoraphidium	0.16	0.16	0.18	0.19		
	Paulschulzia	0.36	0.35	0.11	0.11		
	Plagioselmis					2.61	1.99

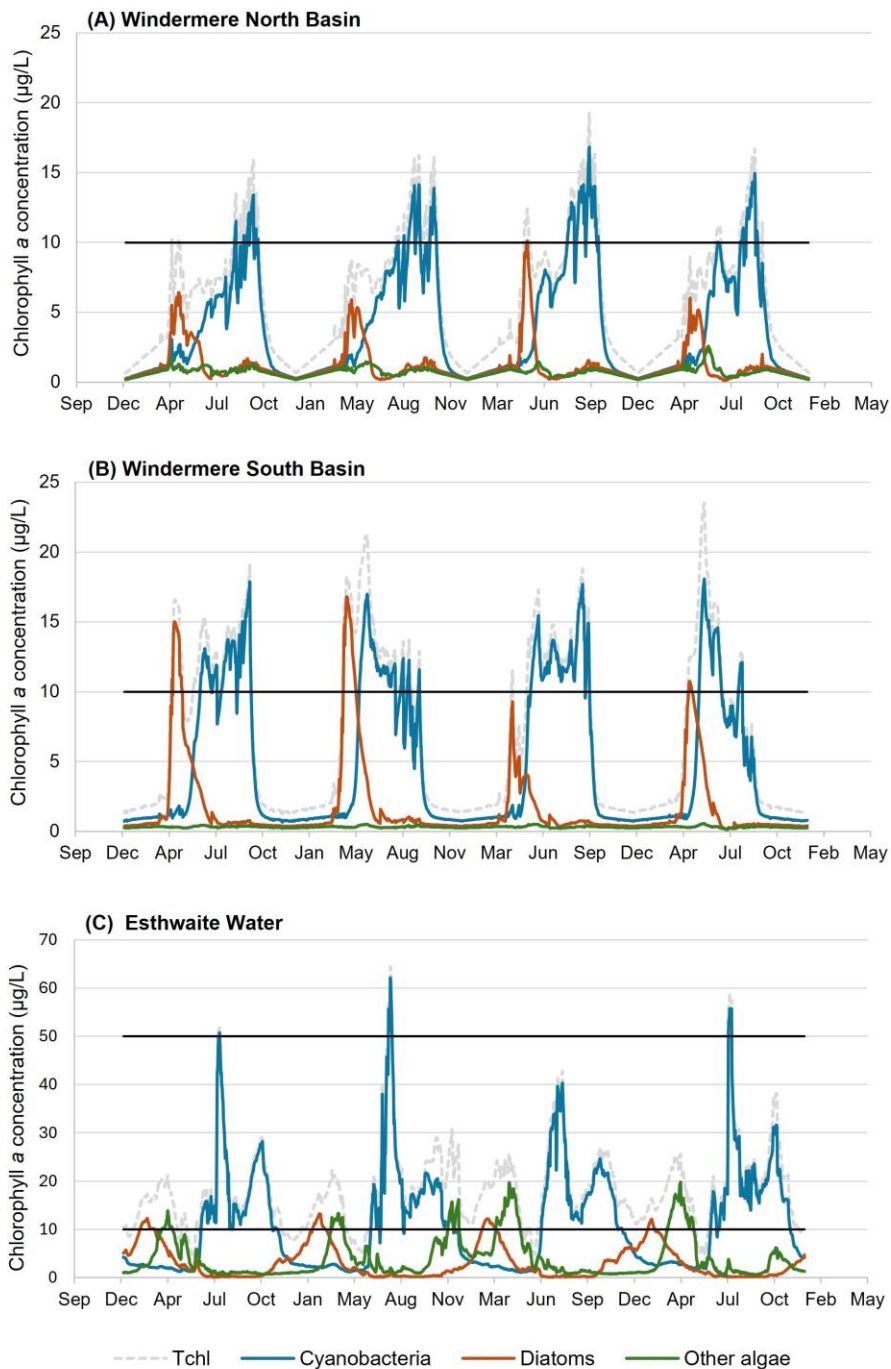


Figure E1: Mean chlorophyll a concentration of diatoms, cyanobacteria and other algae in (A) Windermere North Basin, (B) Windermere South Basin and (C) Esthwaite Water in the climate change scenario (2076-2079). Tchl = total chlorophyll a concentration ($\mu\text{g/L}$) of the whole phytoplankton biomass averaged across the four-year modelling period. Black lines denote WHO10 and WHO50. The WHO thresholds pertain to cyanobacteria chlorophyll a only.

Table E2: Paired two-tailed t-test results for differences in monthly mean biomass (chlorophyll a concentration) of cyanobacteria, diatoms and other algae between the baseline and climate change scenarios. * Denote significant ($P<0.05$) results.

	Phytoplankton group	Paired two-tailed t-test
Windermere North Basin	Cyanobacteria	0.02*
	Diatoms	0.01*
	Other algae	0.65
Windermere South Basin	Cyanobacteria	0.05*
	Diatoms	0.00*
	Other algae	0.02*
Esthwaite Water	Cyanobacteria	0.12
	Diatoms	0.12
	Other algae	0.02*

5.2 Nutrient management for lake catchments in a changing climate

5.2.1 Phytoplankton communities

This section provides supporting information to illustrate the impacts of the three nutrient management scenarios on the phytoplankton communities of Windermere North Basin, Windermere South Basin and Esthwaite Water.

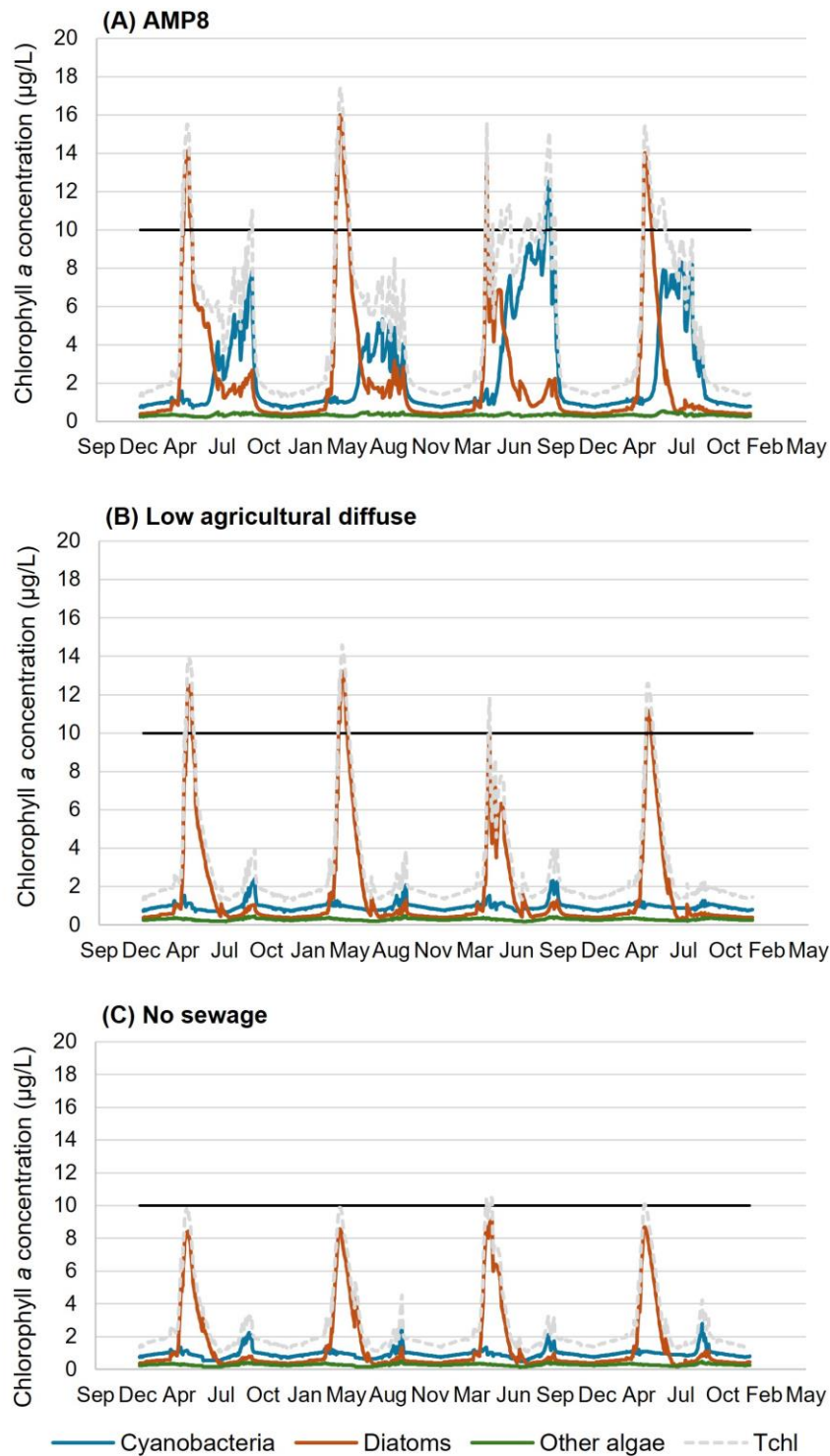


Figure E2: Mean chlorophyll a concentration of cyanobacteria, diatoms and other algae in Windermere South Basin in three nutrient management scenarios: (A) AMP8, (B) low agricultural diffuse and (C) no sewage, each representative of 2076-2079. Tchl = total chlorophyll a concentration (µg/L) of the whole phytoplankton biomass averaged across the four-year modelling period. Black lines denote WHO10 and WHO50. The WHO thresholds pertain to cyanobacteria chlorophyll a only. Windermere South Basin is shown as an example as its response to the nutrient management scenarios was the most pronounced of the lakes modelled.

Table E3: Median summer (July-September) cyanobacteria biomass ($\mu\text{g/L}$) for each lake in each modelling scenario.

Modelling scenario	Windermere North Basin	Windermere South Basin	Esthwaite Water
Baseline	6.94	6.72	17.00
Climate change	8.04	10.95	17.30
AMP8	7.44	4.66	15.05
Low agricultural diffuse	7.44	0.90	12.70
No sewage	2.31	0.90	15.80

5.2.2 Calculating Water Framework Directive (WFD) status

The chlorophyll component of the PhytoPLankton classification with Uncertainty TOol (PLUTO) assessment method was used to determine the WFD status for each lake, in each scenario (UKTAG, 2014a). Modelled daily total chlorophyll *a* concentration ($\mu\text{g/L}$) data was obtained for each four-year scenario period from PROTECH (Appendix B2.4). Monthly averages were calculated for the first 36 months of each scenario to comply with requirements of the PLUTO single site calculator (UKTAG, 2014b). The calculator used this data, together with the site information detailed in Table E4, to provide a normalised chlorophyll Ecology Quality Ratio (EQR) for each lake in each scenario.

Chlorophyll *a* is one component metric of an overall WFD phytoplankton classification, which also requires data on taxonomic composition (UKTAG, 2014a). We were unable to include taxonomic composition as PROTECH only provided information on the eight most abundant phytoplankton types in each lake. The chlorophyll *a* component can be calculated in isolation and provides a useful indication of trajectory of change in the condition of the lake phytoplankton biomass. Normalised EQR values were down weighted ($\times 0.83$, $+0.004$) to account for sole use of chlorophyll *a* concentration data. This down-weighted adjustment is used by the Environment Agency where full taxonomic information is unavailable. It is a pragmatic adjustment based on the known relationship between the chlorophyll EQR and the full PLUTO EQR. These adjusted EQR values were compared to the classification boundaries in Table E5 to determine WFD status.

Table E4: Lake characteristics inputted to the PLUTO single site calculator.

	Waterbody ID	Mean depth (m)	Alkalinity (mEq/L)	Humic type
Windermere North Basin	47007	25.10	0.234	C
Windermere South Basin	47008	16.70	0.273	C
Esthwaite Water	29328	6.42	0.438	C

Table E5: WFD classification boundaries for normalised phytoplankton EQR values, sourced from UKTAG (2014a).

Normalised phytoplankton EQR	WFD status classification
>0.8	High
>0.6 ≤0.8	Good
>0.4 ≤0.6	Moderate
>0.2 ≤0.4	Poor
≤0.2	Bad

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7. List of abbreviations

AMP	Asset Management Period
BWS	Big Windermere Survey
CC	Percentage Cloud Cover
CEDA	Centre for Environmental Data Analysis
CERF	Continuous Estimation of River Flows model
CSF	Catchment Sensitive Farming
CSO	Combined Sewer Overflow
.dat	Data file
DBSCAN	Density-Based Spatial Clustering of Applications with Noise
EDM	Event Duration Monitoring
EQR	Ecology Quality Ratio
Farmscoper	FARM SScale Optimisation of Pollutant Emission Reductions (model)
GCM	Global Climate Model
HadUK	HadUK-Grid dataset
HYPE	Hydrological Predictions for the Environment
JASMIN	
MIDAS	Met Office Integrated Data Archive System
N	Nitrogen
netCDF	Network Common Data Form file
NEAP-N	
NOAA	National Oceanic and Atmospheric Administration
NRFA	National River Flow Archive
pH	Potential of Hydrogen
PLUTO	PhytoPLankton classification with Uncertainty TOol

PR24	Price Review 2024
PROTECH	Phytoplankton RespOnses To Environmental CHange
PSYCHIC	Phosphorus and Sediment Yield CHaracterisation In Catchments
Python	
Qube	
RCP8.5	Representative Concentration Pathway 8.5
S	Silicon
SAGIS	Source Apportionment Geographical Information System
SIMCAT	SIMulations of water quality in CATchments
SRP	Soluble Reactive Phosphorus
Tchl	Total chlorophyll <i>a</i> concentration (µg/L)
TP	Total Phosphorus
UKCEH	UK Centre for Ecology & Hydrology
UKCP18	United Kingdom Climate Projections 2018
UKWIR	UK Water Industry Research group
UU	United Utilities
WFD	Water Framework Directive
WHO	World Health Organisation
WHO10	World Health Organisation (10 µg/L of cyanobacteria chlorophyll <i>a</i> threshold) (this study only)
WHO50	World Health Organisation (50 µg/L of cyanobacteria chlorophyll <i>a</i> threshold) (this study only)
WIMS	Water Information Management System
WINEP	Water Industry National Environment Programme
WwTW	Wastewater Treatment Works

8. Glossary

Astral library	A package in the coding language Python which calculates times for various positions of the sun and moon.
Basin (lake)	In this report, lake basin refers to a water filled depression, rather than a lake's wider drainage area.
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.
Cyanobacteria	Sometimes called blue-green algae, these are actually bacteria. They can photosynthesise and some forms can produce toxins (cyanotoxins). They form part of the phytoplankton community.
Decay factor	The rate of loss of phosphorus and nitrate to sediment along a river reach, when modelled in SIMCAT-SAGIS.
Deciles	A method for quantitatively splitting ranked data into 10 subsections of equal size.
Determinand	A specific parameter that is being identified or measured
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	Dispersed over an area. In relation to nutrient sources, diffuse outputs include 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.
Discharge	The volume of water flowing through a stretch of river channel over a defined period.

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.
First flush	The first pulse of water through a stretch of river that has been experiencing low or no flow conditions, resulting from the onset of rainfall.
GitHub	A web-based repository to enable the storage, management and sharing of open-source code.
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.
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.
Mass-balance approach	A model that considers the input, accumulation and output of a substance or energy within a particular system.
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 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).
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.
Precautionary principle	An approach to risk management or decision making where proactive measures are used to prevent potential environmental harm, even if there remains some scientific uncertainty around the extent to which harm may occur. Effectively this principle acts on the premise of being safe rather than sorry.
Q80	In a hydrological flow record, the Q80 is a value which the flow is equal to or exceeds 80% of the time.
Reduction	If a nutrient undergoes reduction, it loses oxygen. This is the opposite process to oxidation.
Residence time (hydraulic)	Period of time for which water remains in a lake.
Runoff	Draining of water and its solutes across the surface of the land into watercourses and waterbodies.
Sectorial (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.
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.
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
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.
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.

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