CENTRE OF
EXPERTISE
FOR WATERS

Assessing climate change impacts on the water quality of Scottish standing waters

Combined Technical Appendices

CENTRE OF EXPERTISE FOR WATERS

Assessing climate change impacts on the water quality of Scottish standing waters

Combined Technical Appendices

Linda May, Philip Taylor, Iain D. M. Gunn, Stephen J. Thackeray, Laurence R. Carvalho, Peter Hunter, Mairéad Corr, Anne J. Dobel, Alanna Grant, Gemma Nash, Emma Robinson and Bryan M. Spears.

Hydro Nation International Centre
The James Hutton Institute
Scottish Government
Riaghaltas na h-Alba
gov.scot
University of Stirling
UK Centre for Ecology & Hydrology

CREW CENTRE OF EXPERTISE FOR WATERS

Published by CREW – Scotland's Centre of Expertise for Waters. CREW connects research and policy, delivering objective and robust research and expert opinion to support the development and implementation of water policy in Scotland. CREW is a partnership between the James Hutton Institute and all Scottish Higher Education Institutes supported by MASTS. The Centre is funded by the Scottish Government.

Authors: Linda May[1], Philip Taylor[1], Iain D. M. Gunn[1], Stephen J. Thackeray[2], Laurence R. Carvalho[1], Peter Hunter[4], Mairéad Corr[1], Anne J. Dobel[1], Alanna Grant[1], Gemma Nash[1], Emma Robinson[3] and Bryan M. Spears[1].

[1]UK Centre for Ecology & Hydrology (UK CEH), Bush Estate, Penicuik, Midlothian, EH26 0QB.

[2]UK Centre for Ecology & Hydrology (UK CEH), Lancaster Environment Centre, Library Avenue, Bailrigg, Lancaster, LA1 4AP.

[3]UK Centre for Ecology & Hydrology (UK CEH), MacLean Building, Benson Lane, Crowmarsh Gifford, Wallingford, OX10 8BB.

[4]Scotland's International Environment Centre (SIEC), University of Stirling, Stirling, United Kingdom, FK9 4LA

Project Manager: Pauline Lang (2021–2022).

Please reference this report as follows: Linda May, Philip Taylor, Iain D. M. Gunn, Stephen J. Thackeray, Laurence R. Carvalho, Peter Hunter, Mairéad Corr, Anne J. Dobel, Alanna Grant, Gemma Nash, Emma Robinson and Bryan M. Spears (2022). Assessing climate change impacts on the water quality of Scottish standing waters. CRW2020_01. Scotland's Centre of Expertise for Waters (CREW). Available online with Main Report at: crew.ac.uk/publications

ISBN Number: 978-0-902701-03-8

Dissemination status: Unrestricted

Copyright: All rights reserved. No part of this publication may be reproduced, modified or stored in a retrieval system without the prior written permission of CREW management. While every effort is made to ensure that the information given here is accurate, no legal responsibility is accepted for any errors, omissions or misleading statements. All statements, views and opinions expressed in this paper are attributable to the author(s) who contribute to the activities of CREW and do not necessarily represent those of the host institutions or funders.

Acknowledgements: The authors would like to thank the project steering group: Karen Mitchell, Ewan Lawrie and Iain Sime (NatureScot), Ian Milne, Jan Krokowski and Claire Neil (Scottish Environment Protection Agency; SEPA), Jared Stewart (Scottish Water), Rosemary Greenhill (Drinking Water Quality Regulator for Scotland; DWQR), and David Lister (Scottish Government) for their expertise and support. We also thank Nathan Critchlow-Watton (SEPA) and Bob Ferrier (CREW) for reviewing earlier versions of this work, including the Main Report and/or Combined Technical Appendices published separately. The authors are grateful to SEPA staff providing the Scottish lochs and reservoirs monitoring data for this CREW project, and to all project stakeholders who participated in the survey of expert opinion held during November 2021. The CHESS-Met climate change scenario data and the UKCEH monitoring data from Loch Leven were funded by award number NE/R016429/1 as part of the UK-SCAPE programme, which is delivering UK National Capability.

Cover photographs courtesy of:
Bottom-left image: Loch Achray in the Trossachs, Stirlingshire (Linda May, UK CEH);
Bottom-right image: Evidence of algal bloom on Loch Lubnaig in the Trossachs, Stirlingshire (Pauline Lang, CREW); **Top-right image**: Gartmorn Dam in Clackmannanshire (Iain Gunn, UK CEH).

Preface

This CREW Combined Technical Appendices document informed the basis of the Main Report on ‘Assessing climate change impacts on the water quality of Scottish standing waters’ (ISBN 978-0-902701-03-8) that were commissioned by the Centre of Expertise for Waters (CREW).

This combined document consists of seven appendices:

- **Appendix 1** – Literature Review
- **Appendix 2** – Survey of Expert Opinion
- **Appendix 3** – Development of climate change scenarios from CHESS-SCAPE Future Climate dataset
- **Appendix 4** – Exploration of SEPA monitoring data
- **Appendix 5** – Relationship between chlorophyll-*a* concentrations and environmental factors that area likely to be affected by climate
- **Appendix 6** – Site-specific effects of total phosphorus, air temperature and retention time on chlorophyll-*a* concentration in Loch Leven
- **Appendix 7** – The potential for using remote sensing to monitor climate change impacts on water quality in Scottish standing waters

Contents

Glossary

Adaptation:	Taking action to prepare for, or adjust to, the current and/or future effects of climate change
Algae:	Group of mostly aquatic, microscopic plants that have no true roots, stems, leaves or multicellular reproductive structures
Alkalinity:	Buffering capacity of a water body; a measure of its ability to maintain a fairly stable pH level
Bathymetry:	Depth characteristics of lochs
Benthic:	Zone at the bottom of a waterbody
Bloom:	Proliferation of algal or cyanobacterial cells, often seen as a surface scum
Blue-green algae:	Commonly used synonym for cyanobacteria (see below) which can produce chemicals harmful to animal and human health
Catchment:	Area of land from which water drains into a waterbody; also known as a watershed or drainage basin
CHESS-SCAPE	Future climate data set derived from UK Climate Projections 2018 (UKCP18) that provides projections of several climate variables to 2080 at 1 km spatial resolution and time steps ranging from daily to decadal averages
Chlorophyll-*a*:	Green pigment in plants that converts light energy into chemical energy (photosynthesis); often used as a surrogate measure of algal abundance in standing waters
Climate change:	Changes to the local or global climate, usually attributed to increased levels of greenhouse gases in the atmosphere
Cyanobacteria:	Microscopic photosynthetic bacteria, or blue-green algae (BGA), which may form visible surface blooms in the water column or shoreline scums when present in high concentrations
Diatoms:	Microscopic algae with a silica cell wall, commonly found in standing waters
Epilimnion:	See thermal stratification
Eutrophic:	Description of water enriched with nutrients
Eutrophication:	Process of nutrient enrichment in aquatic ecosystems
Evapotranspiration:	Combined term used to describe water lost as vapour from a soil or open water surface (evaporation) and that lost from the surface of a plant, mainly *via* its stomata (transpiration)
Flushing rate:	Time taken to replace the entire volume of a standing water; the inverse of water retention time
Humic loch/reservoir:	Description of a standing body of water with brown, acidic water
Hydrological extremes:	Extreme hydrological conditions such as droughts and floods
Hypolimnion:	See thermal stratification
Lake/Loch:	Area of standing water surrounded by land; for the purposes of this project, this includes lochs, reservoirs and locally important still waters greater than 1 ha in area
Limnology:	Study of inland waters
Littoral zone:	Shallow water around the shores of a body of standing water that rooted plants can colonise
Nitrogen:	Chemical element required by biological organisms for growth; often referred to as a nutrient
Nutrient limitation:	A process through which a biological process, such as algal growth, is controlled by a lack of nutrient availability
Oligotrophic:	Description of water containing a low concentration of nutrients
Palaeolimnology:	Study of the history and development of freshwater ecosystems, especially lochs, by reconstructing past environments from chemical, physical or biological properties of their sediments

Phenology: Study of the seasonal occurrence of animals and plants in relation to their environment

Phosphorus: Chemical element required by biological organisms for growth; often referred to as a nutrient

Photosynthesis: Process by which green plants use chlorophyll to convert sunlight, water and carbon dioxide into oxygen and chemical energy

Phytoplankton: Plant plankton that form the basis of many aquatic food webs

Plankton: Small or microscopic aquatic organisms that drift in the water column

Pond: A small body of still or standing water formed naturally or by artificial means; smaller than a loch or reservoir

RCP: Representative Concentration Pathways (RCPs) are a method of capturing assumptions about economic, social and physical changes to our environment that will influence climate change within a set of future change scenarios; there are 4 RCPs available within the UKCP18/CHESS-SCAPE datasets used in this project. These are RCP2.6, RCP4.5, RCP6.0 and RCP8.5, where the number represents the radiative forcing targets for 2100 in watts per square metre (W m^{-2}); each RCP pathway gives a good indication of the overall level of warming that is likely to occur under each scenario

RCP ensemble member: Each RCP contains four ensemble members (01, 02, 03, 04); member 01 is the default parameterisation of the Hadley Centre Climate model and the others provide an estimate of climate model uncertainty

Reactive phosphorus: Also known as orthophosphate, a soluble form of phosphorus that can be taken up by algae

Redox: A chemical reaction in which the oxidation state of atoms is changed

Reservoir: Enlarged natural or artificial standing waterbody created using a dam

Residence time: The average time that water (or a dissolved substance) spends in a particular standing water; also called retention time

Retention time: See residence time

Silica: A chemical element required for growth by diatoms

Standing water: Stationary or relatively still inland fresh waters, i.e., lochs, reservoirs or ponds

Thermal stratification: Change in the temperature at different depths in a standing water due to the change in water density with temperature; this creates three distinct layers – the epilimnion (upper warm layer); the metalimnion (middle layer); and the hypolimnion (cool bottom layer)

Thermocline: See thermal stratification

Total oxidisable nitrogen: The sum of nitrate and nitrite concentrations in water

Trophic asynchrony: Phenological mismatch occurs when interacting species change the timing of phases in their life cycles at different rates

Turbidity: A measure of suspended material in a waterbody that affects water clarity

Zooplankton: Animal plankton (including small crustaceans and rotifers) that typically feed on algae

Executive Summary

Background

In their summary of the up-to-date evidence of observed climate change trends in the UK over recent decades, the UK Climate Change Committee (2021) highlighted the following patterns of change:

- *Warmer average air temperatures*. The UK's annual average air temperature has risen by about 0.6°C over the period 1981-2000. This equates to a rise of almost 0.3°C per decade since the 1980s.
- *Changed temperature extremes*. The average duration of heatwaves (periods with more than three days in excess of 25°C) has increased over time, with summers as hot as in 2018 (the warmest summer on record) expected to occur in one year in four in the future.
- *Changed rainfall extremes*. Heavy rainfall events have increased across the UK, although the association with human-induced climate change is still difficult to distinguish from inter-annual variability within the observational record.

The UK Climate Change Committee (2021) has also summarised the major changes expected in the UK's climate by 2050. These include the following projected scenarios:

- *Warmer and wetter winters*: By 2050, the UK's average winter is expected to be c. 1°C warmer (0.5°C cooler – 2.5°C warmer uncertainty range) and c. 5% wetter (10% drier – 20% wetter uncertainty range) than the averages recorded between 1981 and 2000.
- *Hotter and drier summers*: By 2050, the UK's average summer could be c. 1.5°C warmer (0°C – 3°C uncertainty range) than between 1981 and 2000 and around 10% drier (30% drier – 5% wetter uncertainty range). The intensity of summer rainfall (when it occurs) is expected to increase, and changes in evapotranspiration and water demand are expected to result in water supplies becoming less reliable (Afazal et al., 2015).

If global warming reaches 4°C above pre-industrial levels by 2100, then further significant changes in UK climate would be expected by 2050.

Scotland, like the rest of the world and UK as a whole, is facing an unprecedented climate change crisis. Amongst other impacts, this is affecting the quality of its standing waters[1]. There is now an urgent need for evidence (UK Climate Change Committee, 2022) to evaluate climate-related risks and inform fit for purpose climate change adaptation strategies that can be created and implemented in Scotland without delay. These will safeguard the integrity, biodiversity and sustainable use of the water environment, for people and for nature.

Currently, we have focused on compiling the key information needed to assess and understand climate change impacts on the water quality of Scottish standing waters. However, outputs from this project may help steer direction of future work aimed at strengthening evidence to inform the prioritisation, collaborative development, and implementation of climate change mitigation and/or adaptation strategies urgently needing to be embedded and delivered in Scotland.

Aim, Research Questions and Key Findings

Against this background of climate change scenarios, the overall aim of this project was to compile and assess the key evidence required to improve our understanding of climate change impacts on the water quality of Scottish standing waters at national, regional and local scales. The project focussed on the interactions between climate change, the drivers of eutrophication problems and their impacts. We synthesised information from the literature, expert opinion and monitoring data, using statistical analyses and visualisation (mainly mapping) combined with climate change scenario modelling to meet project objectives. Six strategic water research questions (RQs) have been addressed. Our key findings (KFs) are summarised below:

RQ1: Is there evidence of a causal link between climate change impacts and water quality issues in Scottish standing waters at national, regional and local scales?

- **KF1.1: Climate change is affecting the water quality of Scottish standing waters, specifically in relation to algal blooms, at multiple scales; mostly through increases in air temperatures and changes in rainfall patterns.**

[1]For the purpose of this project, these are defined as lochs, reservoirs and locally important still waters, excluding those that are temporary, of less than 1 ha in area or less than 1 m in depth, using the criteria in Annex 1 of the SSSI site selection guidelines (2018).

- Increases in rainfall, especially high rainfall events, will increase the delivery of pollutants (such as sediment and nutrient run-off) to standing waters from their catchments; this will cause nutrient enrichment (eutrophication) problems such as algal blooms – especially of cyanobacteria.
- Decreases in rainfall, including an increase in droughts, reduce the flushing rates of lochs and reservoirs and, potentially, their water levels; this will encourage algal blooms and result in habitat degradation and loss of biodiversity.
- Increases in air temperature leads to an increase in water temperature that favours the development of algal (particularly of cyanobacteria) blooms in standing waters during the typical growing season of April to September; a reduction in the availability of nitrogen and silica as temperatures rise will allow cyanobacteria to outcompete other types of phytoplankton.

- **KF1.2: Increases in Scottish loch and reservoir temperatures are closely related to changes in air temperatures; rapid and extensive climate change-driven warming of these standing waters has already occurred in recent years and is expected to continue increasing.**
    - Between 2010 and 2019, average water temperatures of Scottish lochs and reservoirs, between April and September, increased 1.2 times faster than corresponding air temperatures.
    - Based on monitoring data collected between 2015 and 2019, 97% of Scottish lochs and reservoirs experienced an increase in temperature over this period, with most (88%) warming by between 0.25°C and 1.0°C per year and a small number (9%) increasing by 1.0°C to 1.3°C per year.
    - Standing waters are more likely to experience blooms of algae and cyanobacteria as water temperatures increase if there are sufficient nutrients (mainly nitrogen and phosphorus) available to support their growth.
- **KF1.3: Water temperature increases in many lochs and reservoirs have already been recorded; standing waters are projected to get warmer in the south and east of Scotland but this climate-related risk will spread further and reach all parts of Scotland by 2040.**
    - Climate-driven temperature changes are already occurring over multiple timescales, and are evident in decadal scale trends, seasonal changes and shorter-lived extreme events.
    - Short periods of extremely high water temperatures ('lake heatwaves') are likely to increase in occurrence, exacerbating the adverse effects of long-term warming; however these are expected be less intense in deeper lakes than shallower standing waters.
    - Lake heatwaves are likely to push aquatic ecosystems beyond the limits of their resilience, posing a threat to their biodiversity and related benefits they provide to society; this is especially true where low connectivity to other freshwaters mean that species will need to adapt within.

RQ2: What are the main types of climate-driven water quality impacts identified in Scottish standing waters under current and projected climate change scenarios?

- **KF2.1: Climate change will increase the risk of algal blooms developing in Scottish lochs and reservoirs – especially potentially harmful cyanobacteria.**
    - Increases in nutrient inputs and reductions in flushing rates, combined with warmer water temperatures, will increase the likelihood that blooms of algae and potentially harmful cyanobacteria will occur; their duration of occurrence may also expand.
    - Low flushing rates associated with higher water temperatures will increase the risk of nutrients being released from the sediments, fuelling sudden increases in algal growth.
    - High flushing rates will deliver more nutrients to standing waters from their catchment but, at the same time, limit the rate of accumulation of algae and cyanobacteria in the water due to an increase in losses from the outflow.
- **KF2.2: Increases in algal blooms are often associated with a higher risk of potentially harmful toxins from cyanobacteria being released into the water; the likelihood of this occurring will increase with warmer temperatures and lower flushing rates.**
    - Algal blooms, especially of cyanobacteria, reduce the amenity value of standing waters by increasing the risk of people and animals experiencing adverse effects on their health and welfare when visiting affected water bodies.

- Increases in harmful algal blooms (likelihood and duration of occurrences) would prevent water quality targets being met for water supply and/or safe recreational use, leading to higher water treatment costs and/or restrictions on visitor access.
- Further challenges in protecting public and animal health (e.g., recast Drinking Water Directive; revised Scottish Government guidance on cyanobacteria also known as blue-green algae in inland and inshore waters), as well as preventing failure to meet or restore water quality targets (e.g., EU Water Framework Directive), will increase if cyanobacterial blooms become more common.
- Increases in algal blooms will impede statutory environmental objectives being met within policy/regulatory relevant timescales, have an adverse impact on biodiversity and reduce the capacity of water managers to deliver water quality improvements or maintain effective compliance measures that prevent further deterioration.

RQ3: Which areas, locations and types of Scottish standing waters are currently most to least at risk of developing water quality issues due to climate change impacts at national, regional and local scales?

- **KF3.1: Currently, all types of Scottish standing waters in all areas and locations are at high risk of climate change impacts.**
  - The average April to September surface water temperatures of lochs and reservoirs across Scotland showed that, between 2015 and 2019, 97% of these waterbodies had warmed year on year.
  - Maps of average April to September water temperatures between 2015 and 2019 showed a general increase in temperatures across the whole of Scotland over this period.
  - Most lochs and reservoirs increased by 0.25 to 1.0°C per year between 2015 and 2019, but at four sites (i.e., Loch Achray, Loch Lubnaig, Loch of Girlsta, Loch Sgamhain) water temperatures increased by between 1.0 and 1.3°C per year.
- **KF3.2: Different types of lochs and reservoirs will respond differently to climate change impacts, with some more likely to develop water quality issues than others.**
  - Although all lochs and reservoirs are warming, shallow and very shallow systems are likely to be more sensitive to climate extremes than deeper waterbodies because of their higher surface area to volume ratio.
  - High concentrations of cyanobacteria were found to be rare in deep lochs and reservoirs, and those with 'humic' (coloured) water, although humic lochs tended to have a higher number of algal blooms than clear water lochs.
  - As water temperatures increased, high concentrations of cyanobacteria occurred across the whole range of alkalinity types; however, these were more likely to occur in lochs and reservoirs with medium alkalinity.
  - Reservoirs are more likely to be affected by climate change than waterbodies with a more natural hydrological regime because higher levels of abstraction under low rainfall conditions will exacerbate the combined effects of less water coming in from the catchment and higher evaporation rates.

RQ4: Which areas, locations and types of Scottish standing waters are likely to experience exacerbated water quality risks under projected climate change scenarios?

- **KF4.1: Water temperatures across different types of lochs and reservoirs are already warming in most places; this climate-driven trend is projected to further increase from south to north, with an exacerbated water temperature situation expanding to all parts of Scotland by 2040.**
  - Maps of projected water temperatures indicate that climate change impacts will be seen in lochs and reservoirs across the whole of Scotland.
  - Average April to September air temperatures are projected to rise by about 2.5°C between 2020 and 2080; because loch and reservoir temperatures appear to be increasing by 1.2 times the rate of increase in air temperature, this equates to a corresponding increase of about 3°C in Scottish standing waters by 2080.
  - As water temperatures increase, deeper lochs and reservoirs are likely to experience changes in the depth and duration of thermal stratification, with earlier onset and longer periods of stratification causing changes in oxygen concentrations.
  - Decreases in oxygen concentrations will cause an increase in the release of sediment bound nutrients and other contaminants (such as manganese) into the overlying water; the increase in nutrients, especially phosphorus, will fuel sudden increases in potentially toxic algae and cyanobacteria.

- It should be noted, however, that it is not possible to predict these climate change impacts precisely due to the widely recognised uncertainties surrounding climate change predictions; so, these results should be viewed with caution even though the relationships between the climate change data and the loch and reservoir temperature data have been validated for 2010 – 2019.
- Further research is necessary to establish the relationships between climate change and water quality as the response of standing waters is complex and will be determined by the interaction of multiple factors. A key gap in our current knowledge is how climate change will affect the delivery of nutrients to a water body form its catchment – even with warmer temperatures, algal blooms cannot develop if there are insufficient nutrients available to support their growth.

RQ5: What factors contribute to the risk of water quality issues from climate change impacts in Scottish standing waters at national, regional, and local scales?

- **KF5.1: Climate change driven increases in water temperature and nutrient availability, and reductions in flushing rates, will increase the risk of water quality issues developing in Scottish lochs and reservoirs**.
  - The likelihood of algal blooms, especially of cyanobacteria, will increase as mean monthly water temperatures rise above 17°C; this temperature threshold for water quality impacts has been recognised across Europe and is not unique to Scotland, although the underlying reason for this is uncertain at present.
  - Maps showing changes in the frequency with which average monthly water temperatures are likely to exceed 17°C over time suggest that climate change will result in all Scottish lochs and reservoirs experiencing algal blooms by 2080, unless algal growth is nutrient limited.
  - Nutrient releases from the bottom sediments of lochs and reservoirs will be more likely to occur under climate change due to more frequent deoxygenation at the sediment/water interface; in combination with higher water temperatures, these increases in nutrient availability will lead to an increase in algal blooms.
  - Changes in zooplankton community composition are likely to occur when mean monthly water temperatures exceed 14°C; this will affect aquatic biodiversity – especially in relation to species, such as fish, that depend on these organisms as a source of food.
  - Shifts in the seasonal timing of biological communities will occur under the projected climate change scenarios; where this causes a mismatch in the timing between algal communities and their zooplankton grazers, algal blooms will be more likely to occur.
  - The lack of zooplankton data for Scottish standing waters, especially in the spring/early summer when the zooplankton induced clear water phase is most likely to be adversely affected, makes this difficult to evidence.
  - A key gap in our understanding is the time that biological communities will take to undergo an evolutionary adaption to climate change, and whether this will be fast enough to avoid the catastrophic loss of key species – especially of zooplankton, which play a pivotal role in maintaining good water quality in standing waters.
- **KF5.2: Scottish loch and reservoir sensitivity factors will affect the risk of water quality issues developing due to climate change impacts.**
  - The statistical modelling of the loch and reservoir monitoring data showed that, at national scale, the causal relationships between chlorophyll-*a* concentrations and total phosphorus concentrations or water temperature are masked by variations among lochs and reservoirs in terms of their patterns of response.
  - Responses are likely to be more complex when climate change interacts with other pressures, especially where impacts cascade through connected ecological systems.
  - High levels of cyanobacteria are most common in shallow and very shallow lochs, especially those with medium alkalinity levels; these should be prioritised for mitigation purposes.
  - Humic lochs appear to be more prone to high biomass events than clear lochs; the reason for this is unclear.

RQ6: What factors need to be considered for mitigating climate-driven risks to water quality under current and projected climate change scenarios?

- **KF6.1: A whole system approach needs to be taken to mitigate future climate change impacts on standing waters.**
    - Mitigation of climate-driven risks to water quality under current and projected climate change scenarios needs to take a whole system approach, focusing on improved catchment management and sustainable use of water resources; this is a key policy requirement of, for example, the recast Drinking Water Directive (rDWD) and EU Water Framework Directive (WFD).
    - Where climate change impacts cannot be controlled by mitigation, adaptation may need to be considered.
    - Standing waters are subject to multiple interacting pressures that need to be taken into account when interventions are planned.
    - Site-specific analyses may be needed to inform management interventions aimed at reducing, or adapting to, climate change impacts.
    - Categorising lochs into typologies on the basis of depth or other physical characteristics may be insufficient to inform choices about management interventions; shallow and very shallow systems with high nutrient content should be prioritised first in terms of restorative or preventative management, because these are at highest risk of developing water quality problems.
- **KF6.2: An integrated catchment-based approach needs to be taken for setting water quality targets and planning interventions.**
    - Statistical analysis of long-term monitoring data should be used to determine which site-specific combinations of TP concentration, flushing rate and water temperature would enable chlorophyll-*a* targets to be met.
    - Where long term data are not available, site or type specific modelling should be used to identify where interventions can be targeted most cost effectively to achieve water quality targets into the future in un-monitored standing waters.
    - Catchment based interventions, such as sustainable nature-based (or nature-inspired) solutions, should be considered.

Key Recommendations

As a result of this current project analysis, our key recommendations are outlined as follows:

Policy recommendations

The global envelope of climate change is currently affecting, and is projected to further impact on, standing water quality, especially in relation to increasing water temperatures and algal blooms. As part of an informed strategic and coordinated response to the climate crisis, Scotland needs to consider developing, revising, operationalising, and implementing a combination of broad, dynamic, and targeted policy changes for embedding a proportionate response to climate-driven impacts on people, policy, and the water environment across multiple scales, now and into the future. This is made clear from the recent UK Climate Change Committee (2022) Report to Scottish Parliament.

- **Global Climate Change Impacts – Adaptive National Water Policy Perspectives**: The policy gap between global and national understanding of the impacts of systemic climate change on water temperatures and changing rainfall patterns needs to be closed. Failure to address this issue and monitor for key indications of climate-related risks will undermine the development and implementation of adaptive water policy and management practices intended to mitigate complex interactions that affect water use and nutrient run off at regional and local scales.
- **National Climate Change Impacts – Adaptive Regional and Catchment Water Policy Perspectives:** Water policy and management practices need to be adapted to take into consideration national climate-driven risks on the quality of standing waters at regional and catchment scale in Scotland (e.g., River Basin Management Planning (RBMP); Third Land Use Strategy). These climate change impacts will be mediated through shifts in catchment and in-lake processes such as flushing rates, water levels, and nutrient inputs. In combination, these which exacerbate the future risk of algal blooms and may compromise Scotland's ability to meet statutory goals and regulatory targets within given timelines. Revision of current nutrient criteria for Scottish lochs and reservoirs may need to be considered, in conjunction with other policy-based and nature-based solutions, as a potential climate change mitigation/adaptation strategy, to support

desirable legislative outcomes under different climate scenarios. For example, EU Water Framework Directive (WFD) targets for Scottish standing waters may need to be reviewed, and mitigation/adaptation climate strategies mobilised so that good ecological status can be achieved, prevent further deterioration and guide restorative action. It is anticipated that revision of nutrient standards and regulatory compliance permitting/assessment may need to consider climate-related risk in the future. This current analysis of climate-related water quality issues illustrates links between the twin climate and biodiversity crises colliding and being interwoven. This could form a significant contribution to the forthcoming Scottish Biodiversity Strategy (SBS; expected during 2022). This work could also lead to a re-assessment of the Scottish Government's favourable condition targets for protected sites – particularly Special Areas of Conservation and Sites of Special Scientific Interest. It is also noted that the recast Drinking Water Directive (rDWD) will require the creation of Catchment Risk Assessments for all drinking water catchments to encourage greater source control of pollutants (known in the Directive as Hazards and Hazardous Events) i.e., a prevention-led approach for addressing climate change interactions with these catchment factors than reactively managing potential impacts (e.g., algal blooms) on public health with expensive treatment.

- **Regional Climate Change Impacts – Adaptive Local Water Policy Perspectives:** There is an urgent need to update the publication 'Cyanobacteria (Blue-Green Algae) in Inland and Inshore waters: Assessment and Minimisation of Risks to Public Health – Scottish Government Revised Guidance (2012)' in relation to climate change impacts by capturing new evidence that has emerged from this current analysis. This policy review would help protect the amenity value of locally important still waters (e.g., for recreational and wellbeing purposes), and reduce climate-driven water quality risks to public and animal health, whilst climate change mitigation/adaptation needs are being met through other policy routes.

Future monitoring recommendations

The recent UK Climate Change Committee (2022) Report to Scottish Parliament makes clear that 'Scotland lacks effective monitoring and evaluation systems meaning that changes in aspects of many climate-related risks are largely unknown'. Therefore, the existing monitoring network for Scottish lochs and reservoirs urgently needs to be reviewed with a focus on developing an integrated approach for detecting climate change impacts, at pace and scale, whilst including focussed use of new scientific innovations and adaptive resource capabilities. For example:

- Monitor water temperatures in Scottish standing waters at an accuracy of approximately 0.1°C to provide early warning that water quality issues are likely to develop.
- Monitor total and cyanobacterial chlorophyll-*a* concentrations using handheld devices that provide instantaneous data on accumulation of algal blooms, especially cyanobacteria.
- Measure nutrient inputs from catchments, including high temporal resolution gauging of inflows where site specific problems need to be addressed.
- Collect data on precipitation and wind speed to better represent the multi-faceted nature of climate change drivers and their impacts (e.g., storm-driven mixing events, "pulses" of polluted run-off during high rainfall events).
- Develop and monitor indicators of climate change impacts on ecosystem state, processes, and services.
- Explore the potential role of diverse monitoring approaches (e.g., earth observation, in-situ sensors, molecular techniques) for detecting and understanding climate change impacts.
- Consider how different data "streams", especially earth observation data, can be integrated to improve our ability to detect and forecast change.
- Support citizen science initiatives which can provide useful surveillance monitoring data (e.g., Bloomin' Algae app) for assessing climate change impacts on Scotland's water resources.

Further research recommendations

We have assessed that climate change is currently affecting the water quality of Scottish standing waters. We have also projected it will continue happening without urgent intervention to establish pace and scale of mitigation/adaptation strategies needed to course-correct an exacerbated warming situation in the future. This new evidence offers a significant contribution towards strategic climate change needs identified by the recent UK Climate Change Committee (2022) Report to Scottish Parliament. Yet there is still much to be learned about the extent, complexity, rate, and interactions of climate-related risks to water resources in Scotland. We have initially scratched the surface through this current analysis, and as such, there are potential opportunities to explore and understand this evidence in more depth through further analysis. The following research activities are recommended to make best use of the new information that is now available:

- A more in-depth analysis of the SEPA lochs and reservoirs monitoring dataset should be undertaken to improve our understanding the current impacts of climate change on standing waters, how these have developed over time and how they relate to waterbody structure and function.
- More research is needed to provide a better understanding of the links between climate change impacts at the catchment scale and their effects on downstream waterbodies, especially in relation to the propagation of the impacts of extreme climatic events such as storms, heatwaves, floods, and droughts. Existing lake and catchment models would need to be upgraded and linked together to achieve this.
- Further research is needed to understand the extent of climate change impacts on the ecological functioning of Scottish standing waters, especially when ecosystems and biodiversity are likely to reach the point of no return (or tipping point), and how we can mitigate for this sort of potentially catastrophic collapse.
- There is a precise need to better understand why, and specifically what, ecosystem changes are generally triggered across Scotland and in wider Europe when mean monthly water temperatures exceed 17^{o}C and favour the development algal blooms, especially cyanobacteria; as well as the important role of zooplankton as naturally occurring ‘ecosystem engineers’ or ‘nature-based solutions’ that maintain the good water quality of standing waters for example through grazing pressure. We also need to understand the capacity of zooplankton communities to undergo evolutionary adaptation fast enough to prevent a climate change induced ecological crisis.
- Probing connectivity in Scottish standing waters between the climate and biodiversity crises with modelled outputs examined against a broader context of biodiversity typology and classification system-based approaches (e.g., Duigan et al., 2006).

Potential future phase(s) of work recommendations

- **Delivery Purpose and Implementation Needs:** Potential future phase(s) of work should specifically consider addressing the factors that drive algal blooms and engaging sustainable system-based approaches for mitigation of, or adaptation to, current and future climate change impacts on the water quality of Scottish standing waters. This is in the interest of (1) delivering maximal water-related benefits for people and the environment; (2) co-creating effective solutions and supporting intended climate change adaptation outcomes; and (3) strengthening the evidence needed to inform coordinated adaptive management and strategic delivery responses by key stakeholders. Such responses may involve revising existing or developing new and innovative policy-based changes to, management strategies including the integration of nature-based solutions where feasible. These may involve targeting intervention by effective (e.g., response types or site-specific) approaches in a less data-intensive and more readily deployable way.
- **Prioritisation and Engagement Needs:** If required then further analysis should focus on identifying which lochs, reservoirs, and locally important still waters should be prioritised within a climate change mitigation and/or adaptation strategy. By actively engaging a broad range of individuals and organisations in knowledge-exchange opportunities, it should develop tailored approaches and management practices capable of reducing the risk of climate-driven impacts on the water quality of Scottish standing waters across multiple scales. Such outreach activities will need to engage with the primary beneficiaries, such as staff from strategic to operational levels across key stakeholder organisations including the Scottish Government and its environmental conservation and regulatory agencies (e.g., NatureScot, SEPA, DWQR), and water managers (e.g., Scottish Water). Outreach and engagement activities with representatives of the wider water community (e.g., public interest, local authorities, national park authorities, Fisheries Trusts and District Salmon Fishery Boards, anglers, Scottish Freshwater Group members) could include the creation of data visualisations, infographics, and storyboards to illustrate climate change impacts in an accessible way to empower others to become involved in shaping climate action.
- **Evidence-based Needs**: Given additional evidence needs, mapping and modelling tools to project the risk of algal blooms into the future need to be developed to help forecast which areas, locations and types of Scottish standing waters are most vulnerable or sensitive to bloom formation and the impacts of climate change at the national scale. This should be complemented by a more in-depth exploration of climate-driven risk (e.g., in relation to different water uses). Potentially, site-specific studies would help to identify effective management strategies for reducing or reversing climate change impacts on waterbodies at local to catchment scales. Scenario-based modelling approaches should be employed to examine the potential effectiveness of management interventions aimed at mitigating climate change impacts, especially catchment-based solutions such as land use change and nutrient neutrality-based approaches.

1 Appendix 1: Literature Review

Background

The aim of this literature review was to collate and synthesise all available information in the published and grey literature (including reports and relevant policy and summary documents) on the potential impacts of climate change on standing waters, especially in relation to Scotland. Following best practice established by Collins et al. (2015), this evidence review was undertaken in three main stages:

- Setting clear review questions and developing a protocol (setting scope and methods)
- Evidence search
- Extracting, appraising and synthesising evidence

Methods

A rapid scoping review was used to identify and summarise the available evidence, rather than conducting a full systematic review. Relevant peer-reviewed literature and grey literature were identified using web-based sources of information such as Web of Science and Google Scholar. Priority was given to studies conducted within the UK, especially Scotland, but international literature from similar climatic zones was also consulted.

Web of Science (WoS)

Evidence within peer-reviewed publications were gathered primarily using three separate Web of Science (WoS) literature searches, conducted on the 27th and 28th October 2021. These used the following search terms and covered the period 2000 to 2021.

The first WoS search focused on climate change drivers, using the search terms

> *(lake* OR loch* OR "standing water" OR pond* OR reservoir* OR lough*) AND (Scotland OR UK OR "northern Europe") AND ("air temperature" OR "rainfall OR wind")*

It yielded 1,456 'hits'. When the search was defined more narrowly by adding the term "climate change", the number of 'hits' was reduced to 390. All of the titles and abstracts from these studies (with and without the "climate change" search term) were assessed manually to determine whether or not they were pertinent to the project research questions. This process yielded 48 potentially useful references (including six highly cited peer-reviewed papers) that merited further examination.

The second WoS search focused on the primary impacts of climate change on standing waters, using the search terms:

> *(lake* OR loch* OR "standing water" OR pond* OR reservoir* OR lough*) AND (Scotland OR UK OR "northern Europe") AND (flood* OR drought* OR "flushing rate" OR "water level" OR "ice cover")*

This search yielded 1,608 'hits', which were reduced to 413 'hits' by adding the search term "climate change". All the titles and abstracts of these studies (with and without the "climate change" search term) were assessed manually to determine whether or not they were pertinent to the project research questions. This process yielded 26 references that were potentially useful and merited further examination.

The third search focussed on the secondary impacts of climate change on standing waters, using the search term:

> *(lake* OR loch* OR "standing water" OR pond* OR reservoir* OR lough*) AND (Scotland OR UK OR "northern Europe") AND (eutrophication OR "algal bloom*" OR "invasive species" OR "oxygen level*" OR stratification OR turnover)*

This search yielded 1,279 'hits', which were reduced to 254 'hits' when the search term *"climate change"* was added. All the titles and abstracts from these studies (with and without the *"climate change"* search term) were assessed manually to determine whether or not they were pertinent to the project research questions. This process yielded 83 references (including 11 review articles) that merited further examination, although there was overlap with the list produced from the primary impacts list of references. This also suggested that there was a greater volume of evidence for primary impacts than for either climate change drivers or secondary impacts.

Google Scholar/Grey literature

An additional literature search was carried out on Google Scholar using the simple search terms "climate change" and "Scottish lochs" to check for any peer-reviewed articles not found by the Web of Science searches outlined above, as well as any relevant 'grey literature' that would not have been listed by Web of Science. This search, limited to the years 2000-2021, yielded 399 results. All of these records were then checked manually for any useful additional sources of information relevant to the issue of climate change and impact on Scottish standing waters. Nearly all the references already identified in the WoS searches were also picked up by Google Scholar, and some key reports published in the 'grey literature' were also found.

These initial lists of references were supplemented by information from the following sources: citations within these publications; our own knowledge of relevant published and unpublished reports; and additional information provided by the project steering group (PSG) members and other stakeholders.

Evidence from the above literature review was compiled and summarised in an Excel spreadsheet that provides a framework for addressing the questions outlined above.

Climate change drivers

Observed climate change trends

The Climate Change Committee (2021) summarised the up-to-date evidence of observed climate change trends in the UK over recent decades. These included the following climatic patterns:

Warmer average air temperatures. The UK's annual average air temperature has risen by c. 0.6°C over the period 1981-2000. This equates to a rise of almost 0.3°C per decade since the 1980s.

*Changed temperature extremes***.** The average duration of heatwaves (periods with more than three days in excess of 25°C) has increased over time. For the UK as a whole, summers that are as hot as in 2018 (the warmest summer on record) are currently expected to occur one year in four, whereas one year in ten would have been expected only a few decades ago. In contrast, cold temperature extremes have decreased in frequency and intensity.

Changed rainfall extremes. Metrics for heavy rainfall, generally, have shown an increase in very wet days across the UK. However, the expected signal associated with human-induced climate change is still difficult to distinguish from the large inter-annual variability within the observational record that occurs at a UK scale. Extreme event attribution studies indicate that human-induced climate change has increased the likelihood of some UK rainfall extremes that have been linked to significant flooding impacts.

The evidence of these effects of global climate change are expected to grow over the coming years as human-induced warming continues to increase and as observational records get longer (Climate Change Committee, 2021).

Projected future changes in climate

The Climate Change Committee (2021) have also summarised the major expected changes in the UK's climate by 2050. These include the following projected scenarios:

Warmer and wetter winters: By 2050, the UK's average winter could be c. 1°C warmer (0.5°C cooler – 2.5°C warmer uncertainty range) and c. 5% wetter (10% drier – 20% wetter uncertainty range) than it was on average between 1981 and 2000. An increase in both the intensity of winter rainfall and the number of wet days is expected.

Hotter and drier summers: By 2050, the UK's average summer could be c. 1.5°C warmer (0°C – 3°C uncertainty range) than it was on average over 1981-2000 and around 10% drier (30% drier – 5% wetter uncertainty range). A summer as hot as in 2018 (the joint hottest summer on record) for the UK as whole could be normal summer conditions by 2050. The temperature of the hottest days each year are expected to increase more than the average summer temperatures. The intensity of summer rainfall (when it occurs) is expected to increase. Afazal et al. (2015) found that changes in rainfall variability coupled with several other factors, namely changes in evapotranspiration and water demand, and water supply to six storage reservoirs and two intake schemes in Scotland, were likely to become less reliable.

Using 2009 UK Climate Projection data, Muir et al. (2012) mapped the likely spatial distribution of such projected changes in mean summer and winter temperatures and rainfall across Scotland, based on a 50% probability and mid-range emissions scenario. This showed that, on a national scale, there would be a south-east/north-west gradient across Scotland with most areas likely to encounter warmer annual mean temperatures, with wetter winters and drier summers. However, some areas (for example the Cairngorms) were projected to become drier throughout the year. They also demonstrated projected changes in these climate change drivers at the catchment scale, using Kingside Loch in the Scottish Borders as an example (Muir et al., 2012).

If global warming reaches 4°C above pre-industrial levels by 2100, then further significant changes in UK climate would be expected by 2050. Predicted impacts would include much warmer and wetter winters, and much drier and hotter summers with frequent and intense heatwaves (Climate Change Committee, 2021).

Climate extremes

The Climate Change Committee (2021) also highlighted that climate variability, climate extremes and low probability outcomes are important in assessing future UK weather and climate risks. For example, they indicated that the latest high resolution modelling projections for the UK are providing new insights into possible extremes (rather than average changes) under future high global emissions. These include significantly larger increases in winter rainfall, larger increases in intense summer rainfall and more intense summer temperature extremes (e.g., lake heatwaves) (Climate Change Committee, 2021). It is also likely that similar changes will occur in other weather parameters, for example increases in the frequency and intensity of extreme wind speed events, such as storms (Jennings et al., 2021; Jones et al., 2013; Stockwell et al., 2020).

Sensitivity of Scottish standing waters to climate change forcing/drivers

Globally, despite their limited spatial extent of <1% of the Earth's surface, freshwater ecosystems (including standing waters) host one-third of all vertebrate species and 10% of all species (Tickner et al., 2020). As well as their high biodiversity importance, fresh waters provide many essential services to humanity in terms of water supply (for agriculture, drinking water, industry, etc.), pollution removal and recreation (e.g., fishing and swimming). In Scotland, standing freshwaters are particularly important as there are 25,159 individual waterbodies with surface areas greater than 0.1 ha (UK Lakes Portal https://eip.ceh.ac.uk/apps/lakes/). Their relative importance, on a national scale, is emphasised by the fact that Scotland possesses 69% of the standing waters found in Great Britain, and 79% by total surface area and 91% by volume (Smith and Lyle, 1979). These Scottish standing waters include a diverse range of natural waterbody types, such as peaty pools, mountain corrie lochs, expansive shallow lowland basins and large deep valley lochs, as well as many artificially created reservoirs (Lyle and Smith, 1994). In total, standing waters cover about 2% of the land area of Scotland, with the largest concentration being in the West and North East Highlands (Lyle and Smith, 1994).

Standing waters, in general, can be influenced by direct climatic forcing or variation at the air-water interface, but are also affected by indirect climatic effects on catchments, which are then transferred to lakes (Schindler, 2001). Climate change drivers have been conceptualised as energy transfers (e.g., heat, irradiance and wind mixing) and mass transfers (e.g., rainfall, solutes and suspended solids) to lakes (Leavitt et al., 2009). Climate change, together with expanding human population and global economies, are all factors that are likely to control the influx of energy to lake ecosystems in the future (Leavitt et al, 2009; Williamson et al., 2009). Muir et al. (2012) state that in Scotland, climate change is most likely to affect standing water hydrological cycles through the alteration of water temperature and rainfall patterns, including intensities and extremities. The ecological response of individual lakes to these climate change impacts will be dependent on their relative sensitivity or resilience; this in turn will be determined by their individual lake-landscape characteristics.

Water quality and biological responses to climate change impacts on the physical environment of Scottish standing waters

Woolway et al. (2020a) reviewed the global response of physical lake variables to climate change. These included decreases in winter ice cover and increases in lake surface temperatures, leading to modifications in lake mixing regimes and accelerated lake evaporation which, if not balanced by increased rainfall or inflows, will in turn cause decreases in lake levels and surface area. Together with a probable increase in extreme rainfall events, these lake responses will have knock on effects on water quantity and quality, as well as the ecosystem services they provide (Woolway et al., 2020a). Table A1.1 summarises the main physical and environmental aspects of Scottish standing waters that will be potentially affected by the above projected climate change drivers and the likely consequences that will result from them (Gunn et al., 2021). These

Table A1.1. Summary of potential impacts of climate change drivers on Scottish standing waters (adapted from Gunn et al., 2021)

Change	Affected by	Consequences
Flushing rate	rainfall/runoff/evaporation rates/abstraction	• changes in water quality, e.g., eutrophication/blue-green algal blooms • increased sensitivity to other (multiple) pressures including acidification, abstraction & invasive species • changes in lake connectivity
Water level	rainfall/runoff/evaporation rates/abstraction/bathymetry/rate & duration	• changes in lake/reservoir morphology • changes in ecosystem functioning • changes in habitat diversity, e.g., loss of littoral plants, invertebrate and fish spawning & nursery habitats • changes in levels of greenhouse gas emissions • changes in species diversity and increased vulnerability to invasive species
Nutrient delivery	runoff (diffuse sources)/dilution (point sources)/internal release (sediment sources)	• changes in nutrient concentrations and ecological responses
Sediment delivery	runoff (diffuse sources)/sediment disturbance	• changes to water clarity leading to changes in nutrient concentrations, habitat quality, effects on fish egg survival and fish
Water temperature	rainfall/flushing rate/water level/temperature/bathymetry	• changes to water temperatures, e.g., higher water temperatures leading to increased likelihood of algal blooms (in combination with increased nutrients) • changes in stratification pattern, e.g., affect flushing rate because epilimnion behaves like a shallow lake and hypolimnion is not flushed • reduced habitat volume for fish and changes in zooplankton species composition. • changes in seasonal timing of population peaks and densities • impacts on biogeochemical cycling • heat stress to biota/low oxygen conditions • general degradation of biota
Deoxygenation	flushing rate/temperature/organic decomposition & respiration/bathymetry	• changes to water temperatures/stratification, e.g., higher water temperatures leading to increased likelihood of algal blooms (in combination with increased nutrients), reduced habitat volume for fish and changes in zooplankton species composition. • changes in seasonal timing of population peaks and densities. Impacts on biogeochemical cycling. Heat stress to biota/low oxygen conditions • general degradation of biota

climate change impacts are reviewed in more detail below. Water quality (i.e., physico-chemical) and biological responses (both direct and indirect) to climate change impacts are included, especially in relation to nutrient concentrations and algal blooms.

Impacts of flushing rate

The speed at which influent water passes through a lake is referred to as its flushing rate (calculated as lake volume ÷ inflow volume, per year) or its inverse, water retention time (Winter, 2003). Lake flushing rate and water retention time critically affect a lake's sensitivity to environmental change, such as that caused by climate change. Flushing rate is a key component of the lake models that are widely used to predict lake nutrient concentrations and algal standing crop from nutrient inputs (e.g., Dillon and Rigler, 1974; OECD, 1979; Vollenweider, 1975). The flushing rate of lakes fluctuates naturally in relation to variation in rainfall. Short-term hydrological disturbances associated with very high or very low rainfall are often referred to as 'floods and droughts', respectively. These can affect the flushing rate of a lake, its water level, the level of inputs of nutrients and sediments from the catchment, and its responses to those inputs. Flooding increases the rate at which water flows through a lake. This, in turn, leads to increases in surface area and volume (especially in shallow lakes), changes in habitat availability in the littoral zone, and reductions in the sensitivity of lakes to other pressures such as nutrient enrichment. In contrast, droughts reduce the amount of water flowing into a lake from its catchment and this is likely to be combined with increases in evaporative losses from the surface of the lake, due to the lower levels of air humidity and increases in air temperatures that are often associated with drought conditions. In combination, these factors cause lake water levels and volumes to fall. With decreasing amounts of water entering standing waters, flushing rates will be reduced, increasing their sensitivity to other pressures such as abstraction, acidification, nutrient enrichment and invasive species (Jones et al., 2013; Whitehead et al., 2009). Shallow lakes are particularly susceptible to changes in residence times (George et al., 2007). During severe droughts, lakes and reservoirs may become disconnected from surrounding waterbodies, causing a loss of connectivity and a decrease in water quality and amenity value (Dobel et al., 2020).

Globally, the effects of changing climate on water flow through catchments are predicted to be highly variable, geographically (Milly, 2005) and seasonally dependent (Nijssen et al., 2001). Laizé et al. (2017) computed likely future flows in pan-European rivers for baseline conditions (period 1961-1990) and for different combinations of climate and socio-economic scenarios (2040-2069). They showed that climate change is likely to alter flow regimes, leading to new types of river with the implication that this will have a knock-on effect on water flow through downstream standing water habitats. Within Scotland, such a depletion of river flow in future summers would result in a decrease in the flushing rate of standing waters, suggesting that a further reduction in nutrient inputs would be needed to avoid consequent increases in algal blooms. Muir et al. (2012) illustrate this with some Scottish lake types. For example, small shallow lakes within a large catchment, which are more typical of the south-east of Scotland, are likely to be more sensitive to reduced summer rainfall with lower runoff reducing the flushing of the system and increasing water residence times. In the case of Loch Leven, such lower flushing rates can result in a greater accumulation of phosphorus in the lake sediments (Spears et al., 2012), which is then, periodically, released from storage, fuelling problematical cyanobacterial blooms (Carvalho et al., 2011; Elliott, 2010). Carvalho et al. (2012) showed that, from the long-term water quality monitoring at Loch Leven, there was a clear climate impact arising from a negative relationship between summer rainfall and chlorophyll-*a* concentrations, a measure of phytoplankton abundance. In extreme weather years, with very high summer rainfall, Loch Leven was found to have very low chlorophyll-*a* concentrations as result of enhanced flushing (Bailey-Watts et al., 1990). Increased flushing in the summer months can also help the long-term restoration of standing waters, as in Loch Leven, by exporting phosphorus released from internal sediment sources via the outflows whereas, in the driest summers with low flushing rates, high concentrations of chlorophyll-*a* have been recorded in Loch Leven (Spears, 2007a). Such low flushing rates can indirectly affect (e.g., via changes in temperature regime and nutrient availability) algal species composition and succession (Bailey-Watts et al., 1990; Carvalho et al., 2011; Elliott, 2010; Jones et al., 2011; Reynolds et al., 2012). In contrast, large deep lakes, which are more characteristic of the north-west of Scotland, are less likely to respond to changes in flushing rate but may be more sensitive to other climate induced changes, such as longer periods of thermal stratification reaching greater depths that can lead to the deoxygenation of the hypolimnion and stress in sensitive fish populations (Arvola et al., 2010).

Increases in cyanobacterial populations resulting from reduced flushing rates may be less significant if growth is limited by other factors such as light and nutrient availability (Elliott, 2012a). An increase in the nutrient enrichment of standing waters with cyanobacterial blooms can also cause an increased risk to public health and loss of amenity value (e.g., Cox et al., 2018; Facciponte et al., 2018) by modifying the relationship between nutrient availability and biomass accumulation (e.g., Carvalho et al., 2013). In general, cyanobacterial populations tend to increase during warm dry summers, when flushing rates decrease, but their populations can fall rapidly if a high rainfall event then flushes them out of the lake. These processes are all natural phenomena that occur in response to variations in rainfall and runoff. The underlying mechanisms responsible for the observed link between algal biomass and flushing rate have been investigated using PROTECH, a numerical model that predicts the accumulation of algal biomass in lakes in response to inputs of nutrients, light and thermal energy (Elliott et al., 2010; Reynolds et al., 2001). Using PROTECH, Elliott and Defew (2012) modelled the 2005 Loch Leven phytoplankton community in response to a combination of water temperatures and flushing rates and found that that some bloom-forming algal taxa were negatively affected by increased water flow (e.g., *Aphanocapsa*; a type of cyanobacteria), some were enhanced (e.g., *Stephanodiscus*; a type of diatom) and that other genera responded more to changes in water temperature (e.g., *Aulacoiseira*; a type of diatom that responded positively to increased temperatures and *Asterionella*; a diatom that responded negatively to increased temperatures).

Impacts on water level

Changes in standing water levels are caused by a variety of factors, including floods and droughts, water abstraction and changes in evaporation rates (Woolway et al., 2020a). These may occur seasonally, weekly, or even daily under some circumstances (Smith et al., 1987). The overall water balance of a standing water (i.e., inflow volume minus abstraction, evaporative losses and outflow) affects the water level of the standing water. Where losses are greater than inflow volumes, lake shores can become exposed; the reverse situation can result in high water levels and flooding of marginal areas. Muir et al. (2012) highlight that, in Scotland, climate change will tend to increase variability in rainfall patterns, which will result in more extreme flood and drought events thus affecting both surface and groundwater flows. This will, in turn, change standing water flow regimes, affect lake-landscape connectivity and cause changes in both shoreline complexity and habitat structure (Wantzen et al., 2008). As lakes are relatively closed systems, changes in water level will affect biological communities across the lake, especially around the shoreline. Under lower water level scenarios, previously inundated areas can become dry and exposed and under higher water level scenarios previously dry and exposed areas can become inundated. Shallow standing waters are more vulnerable than deeper standing waters, because small changes in water level represent a much larger proportion of their total surface area and volume (George et al., 2007). In deep, seasonally stratified lakes, the impacts of water level fluctuations are likely to be restricted mainly to changes in the littoral zone (Smith et al., 1987).

Although lake or reservoir biota have evolved life cycles that accommodate natural water level fluctuations, extreme or unusually timed fluctuations in water levels are likely to affect the biota and impair ecosystem functioning. Such changes in water levels, especially those associated with a significant lowering of lake level and consequent loss of volume, can have serious detrimental impacts on plant and invertebrate communities, especially around the shoreline. Littoral areas can become exposed and desiccated leading to significant losses amongst the littoral macroinvertebrate community (e.g., Arvoviita and Heiki, 2008; Baumgartner et al., 2008; White et al., 2008). This, in turn, affects species that depend on this food supply (e.g., aquatic birds and fish). Exposure of such littoral areas during droughts can also prevent their use as spawning and nursery areas by fish (Winfield, 2004). In general, lake biota with relatively short lifespans and generation times (days to weeks), such as open water (planktonic) communities of algae and invertebrates, tend to be least affected by water level changes because their populations can respond rapidly to disturbance. In contrast, lake biota with longer lifespans and generation times (months to years), such as plants and fish, tend to be more affected because they respond over longer timescales. Excessive or prolonged drawdown or flooding of lakes (beyond natural level fluctuations) may cause significant losses in these biota leading to a reduction in structural diversity, or even the complete destruction of some of these communities if physiological limits are exceeded. This may cause a regime shift in the functioning of standing waters, from an aquatic plant-dominated to an algal-dominated system (Reynolds et al., 2012). Examples of such impacts are often seen in lakes managed for water supply (e.g., Loch Doon, Thirlmere) (Jones et al., 2013). In reservoirs, drought impacts may be amplified by the need for higher levels of abstraction to meet increasing demands for water under drought conditions from growing populations.

In standing waters, water level decline and elevated evapo-concentrations are universal driving processes, however water chemistry responses can be site-specific (Webster et al., 1996) with groundwater-fed streams having a greater impact on their water chemistry than from the catchment geology (Webster et al., 2000). Mosley (2015) observed that, in European lakes and reservoirs, water quality responses to drought conditions can be exhibited by increases in dissolved organic carbon, inorganic nutrients, pH, salinity, turbidity and redox metals, and decreases in dissolved oxygen concentrations in the water column, resulting in lower habitat and recreational value. In stratifying, deeper waterbodies these water chemistry responses can be more prominent than in shallow waters, where it only manifests during post-drought re-filling (Baldwin et al., 2008). The effects of such post-drought re-filling are greatly determined by the land-use and geology of the catchment, including reconnection with polluting point sources. In lakes predominantly served by surface water, hydrological disconnection during droughts can result in increased evapo-concentrations, increasing salinity and nutrients. Changes in the shoreline habitat may also result in an increase in greenhouse gas emissions, because emissions of carbon dioxide and methane from exposed sediments increase during drying and re-wetting (Kosten et al., 2018). In shallow standing waters, reduced water levels can also promote wind induced sediment disturbance, leading to increased turbidity in the water (Mosley, 2015).

Impacts on nutrient and sediment delivery

Changes in the hydrological inputs to standing waters also affect the delivery of water-borne substances from the catchment. Nutrient responses may also vary with relative loading from the catchment. Inputs of nutrients and sediments, for example, increase markedly with higher levels of runoff. In general, more than 80% of the nutrient inputs to a lake are delivered in just a small number of high rainfall events with very little being delivered under low rainfall conditions (Jordan et al., 2012; Sharpley, 2008). Variations in rainfall affect not only the magnitude of delivery, but also the timing. This, in turn, will affect the response of the standing water. However, because of the close connection between nutrient and sediment delivery, and hydrological inputs, it is very difficult to differentiate between responses caused by increases or decreases in nutrient inputs and those caused by increases or decreases in waterbody retention time. For example, Defew et al. (2013) found that, in the Loch Leven catchment, if nutrients from external sources were supplied diffusely, the level of input (load) tended to reflect the variation in inflow (and thus flushing) rates, especially during high rainfall events. However, in contrast, if nutrients were from point or internal sources, their load would remain more or less constant as inflow rates changed, with nutrient concentrations within the lake tending to increase as flushing rates decreased (Elliott et al., 2009).

A warmer climate may also enhance both acidification and nutrient enrichment by increasing the release of nitrogen from soil organic matter to runoff (Wright and Schindler, 1995). Futter et al. (2009) modelled the effects of changing climate (and nitrogen deposition) on nitrogen dynamics in Lochnagar and found that, as a result of a warmer and drier climate, there would be less runoff and a much reduced snow pack. Futter et al. (2009) projected that surface water nitrate concentrations would be expected to increase under climate change, but were not likely to return to or exceed historical levels, even if nitrogen deposition levels remained constant.

Although not the primary focus of this literature review, it should be noted that concentrations of dissolved organic matter (DOM) in UK upland drinking waters have been rising over recent decades, largely as a consequence of soil organic matter becoming more soluble as soils recover from the effects of acid rain (Monteith et al., 2021). With predicted future warming, any trend towards drier summers is likely to result in significantly higher DOM concentrations (particularly in water draining peat-dominated catchments), while DOM concentrations in catchments with a greater coverage of freely draining organic soils are likely to be more sensitive to fluctuations in rainfall, with levels increasing with more frequent and intense rainfall events (Monteith et al., 2021). Hence, there are concerns that this could result in even higher concentrations of DOM reaching UK water treatment works, which will affect treatment processes and increase treatment costs (Ritson et al., 2014).

Impacts on water temperature

Worldwide, lake surface water temperatures are warming, with Pilla et al. (2020) reporting an average rate of increase of +0.37°C decade-1 from their study of 102 lakes. However, temperature trends appear to vary over space and time, with some evidence suggesting that surface water temperatures are rising more rapidly than air temperatures (e.g., Pilla et al., 2020). From a smaller study of 10 European lakes (which included Loch Leven), evaluation of 50 years of observational data from 1966 to 2015 showed that annual maximum lake surface temperatures had increased at an average rate of +0.58°C decade^{-1} which was more similar to the observed increase in annual air temperature of +0.42°C decade^{-1} (Dokulil et al., 2021). In contrast to surface water temperatures, Pilla et al. (2020) found little change in average deep water lake temperatures, although there was a high degree of variability across the 102 lakes studied. This variability could not be explained by surface water temperatures or thermal stability of lakes, so it is likely that long-term trends in the thermal structure of deep-water lakes are driven by external factors such as local to regional climate patterns or additional external anthropogenic influences. Normally the temperature of standing waters, other than very small and shallow lakes, follows seasonal trends with perhaps small, short-term variations that can be related to significant or extreme weather events – for example, heat waves or storm events (Climate Change Committee, 2021). Inflows, rainfall and mixing by winds are all variably influential, depending on relative hydrology and morphometry of the lake concerned. Shallow lakes, and lakes with shallow thermoclines (regions of rapid temperature change) are the most susceptible to warming (George et al., 2007), while in deeper waterbodies the higher water temperatures tend to lengthen the period of thermal stratification and deepen the thermocline (Hassan et al., 1998). In the UK there are two distinct types of lake thermal regime: warm monomictic and dimictic lakes. Warm monomictic lakes are exemplified by large deep lakes, such as Loch Ness, that never fall below the temperature of maximum water density (4°C). Dimictic lakes, on the other hand, are characterised by water temperatures passing through 4°C twice a year (in spring and autumn) and, if deep enough, may undergo thermal stratification in the summer. Battarbee et al. (2002) indicated that it is easier to detect evidence of climate change in arctic or alpine lakes where the extent of warming is more pronounced and where its effects are not masked by other types of human activity, e.g., pollution. Identifying the influence of climate change on standing waters affected by nutrient enrichment is more difficult where increasing water temperatures tend to produce similar symptoms to those created by cultural eutrophication, such as increased algal productivity/oxygen stress in the hypolimnion and accelerated nutrient recycling (Moss et al., 2011). In addition, nutrient loading can be enhanced by climate mediated changes in catchment hydrology and soil biochemistry (e.g., Moore et al, 2010; Pierson et al., 2010).

Through its effect on the density of water, temperature change also influences when within the year a lake mixes or becomes stable, essentially altering the length of the growing season (e.g., earlier onset and longer periods of thermal stratification), causing profound effects throughout the lake ecosystem (Woolway et al., 2021a). Thermal stratification can drastically alter the structure and function of lakes by affecting biogeochemical processes between the sediments and the open water (Spears et al., 2007b), and the biology of phytoplankton (i.e., increasing likelihood of cyanobacterial blooms), zooplankton and fish. Increasing water temperature and stratification can lead to heat stress on biota and hypoxia in deeper waters due to increased respiration, leading to a general degradation in the biota. In the case of droughts, an extreme lowering of water levels may reduce the volume of the hypolimnion. This may particularly affect fish requiring relatively low water temperatures and could lead to fish kills due to anoxic conditions and a loss of spawning habitats over time. Some of the UK's rarest fish, for example the vendace (*Coregonus albula*) and the Arctic charr (*Salvelinus alpinus*), are likely to be most affected (Elliott and Bell, 2011; Jones et al., 2008). When the rates of oxygen transport across the thermocline are low compared to rates of decomposition and respiration, oxygen consumption can lead to anoxia at depth. This process may be stimulated by nutrient enrichment and climate change (Foley et al., 2012). Woolway et al. (2021a) investigated changes in lake stratification phenology across the Northern Hemisphere (including lakes from the English Lake District) from 1901 to 2099, using a lake-climate model ensemble and long-term observational data. Under high-greenhouse gas-emission scenarios, it was predicted that stratification will be prolonged by 33.3 ± 11.7 days by the end of the current century (Woolway et al., 2021a). This will inevitably have knock on effects on nutrient mineralisation and phosphorus releases from lake sediments, with a likely misalignment of lifecycle events possibly causing irreversible changes in lake ecosystems. Stratification may also be affected by other larger-scale processes, such as those driven by the Gulf Stream (George and Taylor, 1995), the North Atlantic Oscillation (George et al., 2004) and, potentially, the jet-stream (Strong and Maberly, 2010). Energy, surface heating and lake transparency can also determine the patterns of vertical mixing that occur within lakes and the extent of stratification (mediating diffusive exchanges between sediments and overlying water), and impact upon many biogeochemical processes that are known to regulate key lake functions (Jones et al., 2005; Persson and Jones, 2008). These coupled processes can drive water quality and the ecological structure and function of lakes (Spears et al., 2012), regulate the delivery of nutrients and other pollutants to downstream systems (Spears et al., 2007a), and contribute to global scale climate regulating processes (Maberly et al., 2013). Drivers of these processes are related to a mosaic of interacting physicochemical and biogeochemical drivers that vary in space and time (Spears et al., 2007b), most of which are sensitive to various combinations of external and in-lake hydrological processes, as outlined above.

Impacts on water chemistry

Although the solubility of dissolved oxygen is known to decrease with increasing water temperatures, long-term lake trajectories have, hitherto, proven difficult to predict (Jane at al., 2021). The concentration of dissolved oxygen in standing waters is important as it helps to regulate biodiversity, nutrient biogeochemistry, greenhouse gas emissions and the quality of drinking water (Jane at al., 2021). Increasing temperatures can strengthen stratification, which can lead to a reduction in the volume of the hypolimnion, thereby increasing the risk of hypoxia in bottom waters (Baldwin et al., 2008). The resultant biogeochemical processes in bed sediments can influence water chemistry, resulting in elevated concentrations of dissolved iron, manganese, phosphorus, and ammonium, and reduced concentrations of nitrates. Similarly, an increase in other redox sensitive metals and metalloids are likely, leading to a degradation of water quality and recreational value of standing waters (Jirsa et al., 2013). The production of methane and nitrous dioxide greenhouse gases may also be elevated under reducing bed sediment conditions (Tranvik et al., 2009). Oxygen concentrations and water temperatures near the sediments are also strongly affected by in-lake hydrological processes and are, in turn, important drivers of the biological and chemical processes that are responsible for liberating nutrients from lake sediments. During stratification, surface and deep waters are largely isolated from each other, although some vertical mixing continues to occur through vertical diffusion across the thermocline and/or the mixing of hypolimnetic waters, as the epilimnion deepens. Mackay et al. (2014) have shown that the rate of vertical diffusion and the number and size of entrainment events are affected by atmospheric forcing, causing changes to deep-water anoxia, phosphorus accumulation in the hypolimnion, and subsequent fluxes of phosphorus into the epilimnion.

In a recent analysis of temperature and dissolved oxygen profiles from 393 temperate lakes (over period 1941-2017), Jane et al. (2021) found that there was a widespread trend towards lower dissolved oxygen concentrations in both surface and deep-water standing water habitats. They concluded that oxygen losses in warming lakes may be amplified by enhanced decomposition and stronger thermal stratification, or by oxygen increasing as a result of enhanced primary production. The decline in oxygen concentrations of surface waters was primarily associated with reduced solubility under warmer water temperatures, although dissolved oxygen concentrations in surface waters increased in a subset of highly productive warming lakes, probably owing to increasing production of phytoplankton. By contrast, Jane et al. (2021) associated the oxygen decline in deep waters to stronger thermal stratification (hindering oxygen replenishment and leading to longer periods of oxygen depletion) and loss of water clarity, but not with changes in gas solubility, per se.

Effects of water temperature on algal blooms

With climate change modelling predicting increases in summer water temperatures in line with increasing air temperature, there is likely to be an associated prolongation of periods of thermal stratification, particularly in relatively deep standing waters, leading to increases in the biomass, frequency and intensity of cyanobacterial blooms (Moss et al., 2003). In one such deep Scottish lake system, Loch Lomond, Krokowski (2007) indicated that surface water temperatures had increased from 9°C in 1987 to 13-14°C in 2005 and that the phytoplankton biomass and abundance were most strongly correlated to water temperatures. Through modelling the 2005 Loch Leven phytoplankton community in response to a combination of water temperatures and flushing rates, Elliott and Defew (2012) found that some genera responded positively to increased temperature (e.g., *Aulacoseira*) and some negatively (e.g., *Asterionella*), while other species were more affected by increased water flows. Bennion et al. (2012) also showed for Loch Leven that there had been several changes in the recent fossil record, indicating that shifts in the lake diatom species assemblages could be attributed to climatic controls, e.g., the presence of *Aulacoseira granulata* and *Aulacoseira granulata var. angustissma* seemed to show seasonality and coincided with warmer temperatures. Krokowski et al. (2012), in reviewing the potential effects of climate change on the incidence of cyanobacteria in Scottish lochs, indicated that such temperature rises in combination with increased nutrient runoff from surrounding catchments (associated with predicted increased rainfall) would result in increases in phytoplankton biomass. Modelling has also predicted that, with water temperature warming, there is an increased likelihood of the phytoplankton being dominated by cyanobacteria with more intense and frequent cyanobacterial blooms (Wagner and Adrian, 2009). Krokowski et al. (2012) also highlighted the potential threat of increased summer water temperatures leading to warm-water invasive cyanobacterial species spreading into Scottish standing waters from the tropics and displacing native cold-water species, as has already occurred in some northern European water bodies (Wiedner et al., 2007). Jones et al. (2020), in a national screening assessment of threshold-based climate change impacts on UK fresh waters, identified such water temperature effects on algal blooms as being one of the key impacts of climate change on lakes. Jones et al. (2020) highlighted that above a mean monthly water temperature of 17°C, and in combination with elevated nutrient levels, harmful algal blooms are more likely to form, leading to a decrease in water quality and adverse effects on the range of ecosystem services that are dependent on that water quality (Figure A1.1). The threshold and impacts are summarised in the figure below.

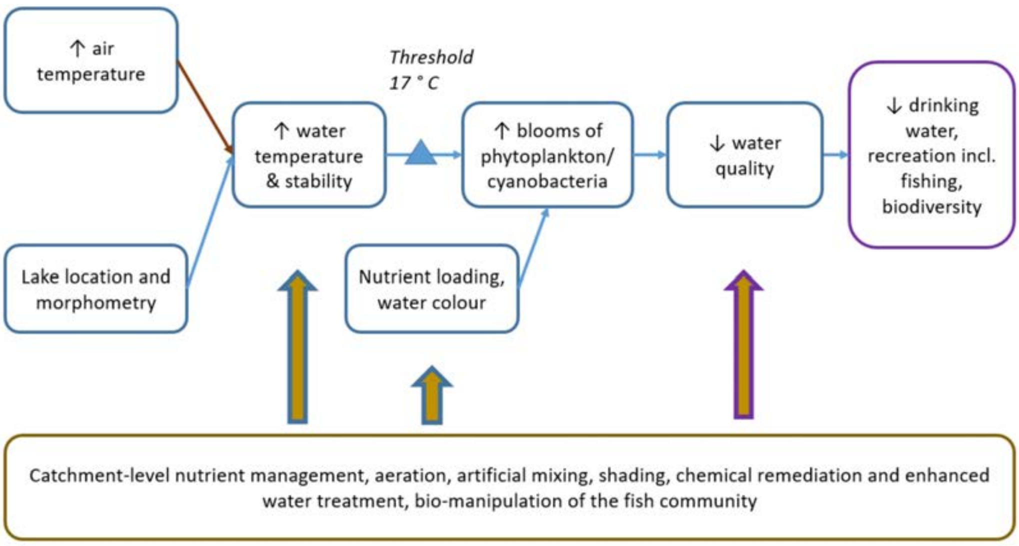


Figure A1.1 Impact chain for temperature effects on phytoplankton blooms in lakes. Purple box shows socio-economic or biodiversity endpoint; brown box shows potential adaptation measures (Source: Jones et al., 2020).

Jones et al. (2020) justified the use of such a temperature threshold on the basis that lake temperatures are likely to warm in line with air temperature, with an associated increase in water column stability due to stronger thermal stratification. Such warming stimulates the growth of phytoplankton species capable of forming blooms, particularly favouring harmful cyanobacterial blooms, as does the improved underwater light climate that results from increased water column stability (Elliott, 2012a, 2012b,; Elliott et al., 2010; Ho et al. 2019; Paerl and Huisman 2008; Richardson et al. 2018). Cyanobacterial growth rates frequently reach their maximum, or exceed those of other phytoplankton, above a water temperature of 25°C (Jöhnk et al., 2008; Paerl and Huisman, 2008). However, the temperature at which bloom formation occurs is often much lower due to other factors such as nutrient availability (total inputs and resultant concentrations), and lake morphology. Recent studies by UKCEH (Carvalho et al., 2013; Van der Spoel, 2019) have suggested a mean monthly water temperature of 17°C as a threshold above which blooms are more likely to occur in the UK, assuming that sufficient nutrients are available to support primary production. An increased availability of nutrients due to eutrophication can further enhance phytoplankton growth as temperatures rise. Changing water colour can also moderate the temperature effect in different ways. It is indicative of organic inputs that that can supply additional nutrients, but also limits the availability of underwater light. Nutrients and water colour are frequently quantified as concentrations, but it is important to recognise that both total inputs (amount) of material entering standing waters and the flow volumes in which they are delivered are also important.

Effects of changes in water temperature on zooplankton

Increases in water temperature, due to rising air temperatures, can also lead to changes in the species composition of the planktonic herbivore community (Bruel et al., 2018). In Loch Leven, warmer spring temperatures have been noted to have had a positive impact on water quality by increasing the abundance of Daphnia grazing on phytoplankton. This results in a decrease in spring chlorophyll-*a* concentrations and an associated improvement in water quality during May and June (Carvalho et al., 2012). Jones et al. (2020), in a national screening assessment of threshold-based climate change impacts on UK fresh waters, identified such water temperature effects on zooplankton composition as being one of the key impacts of climate change on lakes. Jones et al. (2020) derived a summer mean monthly water temperature threshold of 14°C for impacts on zooplankton composition (Bruel et al., 2018. This threshold and impacts are summarised Figure A1.2.

Warming water can stimulate the growth of filamentous phytoplankton, including cyanobacteria (Paerl and Huisman, 2008). These affect food quality and, thereby, favour different species of zooplankton grazers. Furthermore, warming may enhance fish predation on zooplankton and change which species dominate the community (Gyllström et al., 2005). Changing nutrient loading can also interact with these effects by influencing the quantity and quality of food available to grazers. Such changes in zooplankton community composition will have knock-on effects on biodiversity, potentially affecting the grazing pressure exerted on phytoplankton and, thus, water quality. Climate change may also alter the seasonal timing of phytoplankton and zooplankton blooms, leading to desynchronised species interactions with potential implications for food-web interactions (Thackeray et al., 2013; Samplonius et al., 2021).

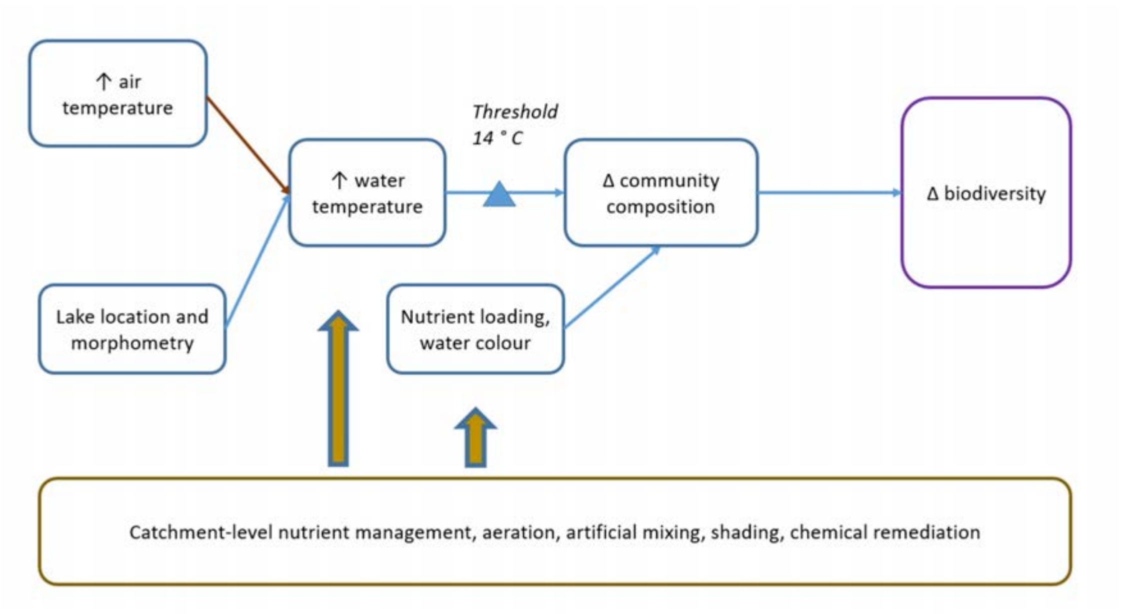


Figure A1.2 Impact chain for temperature effects on zooplankton species composition in lakes. Purple box shows biodiversity endpoint; brown box shows potential adaptation measures (Source: Jones et al., 2020).

Heatwaves

As well as projected long-term increases in global average air temperatures, climate models are indicating an increase in the frequency and severity of extreme temperature conditions under a future climate (Climate Change Committee, 2021). Such temperature extremes include lake heatwaves - periods of extremely warm lake surface water temperatures. One recent example of the impact of lake heatwaves on lake surface water temperature was demonstrated for European lakes for the period May-October 2018, in which, using a validated model, the mean and maximum lake surface temperatures were found to be 1.5 and 2.4°C warmer than the base-period average (1981-2010) (Woolway et al., 2020b). Further analysis, using satellite data and modelling of hundreds of lakes worldwide from 1901 to 2099, showed that globally lake heatwaves are likely to increase in intensity and duration (Woolway et al., 2021b). This modelling suggested that surface heatwaves will be longer-lasting but less intense in deeper lakes (up to 60 metres deep) than in shallower lakes, and predicted that as lakes warm during the twenty-first century heatwaves will begin to extend across multiple seasons, with some lakes reaching a permanent heatwave state. Woolway et al. (2021b) indicated that such heatwaves are likely to exacerbate the adverse effects of long-term warming in lakes and exert a widespread influence on their physical structure and chemical properties, and suggest that these lake heatwaves could alter the lake biota by pushing aquatic species and ecosystems to the limits of their resilience. This, in turn, could threaten biodiversity and the key ecological and economic benefits that lakes provide to society.

Storm events

Severe storms are another type of extreme event that can have large effects on standing waters (Jennings et al., 2021). Such storms principally influence lakes by increasing inputs of terrestrial material derived from catchments during heavy rainfall, and by bringing about physico-chemical extremes due to the increased mixing of the water column by high winds (Woolway et al., 2018). Woolway et al. (2018) studied the impacts of one such event, Storm Ophelia, on Windermere in 2017. This storm caused a great upwelling of deep, cold, oxygen poor water in Windermere, which then flowed into the outflow. The study indicated that the response of standing waters to an extreme weather event of this type may cause important downstream effects that are not immediately apparent at the lake surface – suggesting that climate change effects may be propagated through the drainage network as a result. The impacts of storm events are contingent on many things, including the features of the storm itself and the attributes of the water bodies and catchments affected. In the case of lake phytoplankton, such extreme events can restructure phytoplankton communities and affect their dynamics, resulting in altered ecological functioning (e.g., carbon, nutrient and energy cycling) in the short- and long-term, but this will be dependent on how resilient a particular standing water ecosystem is to such impacts (Stockwell et al., 2020). For example, lake phytoplankton communities may be more vulnerable to an extreme wind event if it comes shortly after another storm event or if periods of drought alternate with periods of intense rainfall. Conversely, an extreme wind event may not have much impact on a lake phytoplankton community if the lake was already fully mixed (Stockwell et al., 2020).

Changes in seasonal timing of biological communities and species range shifts

There is evidence that phenological shifts in response to seasonal warming are potentially leading to trophic asynchrony (e.g., Winder and Schindler, 2004). Such phenological changes have been observed globally for marine, freshwater and terrestrial species, and are an important element of the global biological 'fingerprint' of climate change (Thackeray et al., 2010). Phenological shifts in plankton populations are considered indicative of environmental factors acting upon population-level processes, specifically the balance between rates of replication/reproduction and loss/mortality (Thackeray et al., 2008). Differences in rates of change could desynchronize seasonal species interactions within a food web, threatening ecosystem functioning (Thackeray et al, 2013). The effects of climate change are not restricted to single species; multiple co-occurring species are probably affected in different ways and this can influence ecological processes. In the case of Windermere, the seasonal timing of phytoplankton and zooplankton blooms, and of perch spawning, are now earlier in the year – with those shifts happening at different rates for different species (Thackeray et al., 2013). Such differences in climate change responses among species can mean that predatory species are now reproducing at a time of year when essential food resources for their offspring are no longer highly available; this is known as a trophic mismatch. This has implications for food-web interactions and species fitness, although it is not yet understood what these will be (Samplonius et al., 2021).

Climate change impacts have also been manifested in space as well as time. It has been speculated that increases in air temperature will cause shifts in species ranges, with warm water species expected to expand the northern limits of their range under warming scenarios. Elliott et al. (2015) tested this hypothesis for roach *(Rutilus rutilus)* in the UK – a warm water fish species that is not native outside of the south-east of England. By combining species niche modelling with climate scenarios, Elliott et al. (2015) predicted a range expansion for roach within Great Britain, including the species moving northwards into Scotland by the end of the century.

Climate change complexities

Muir et al. (2012) pointed out it that it is not always easy to attribute changes solely to climate change, because Scottish standing water systems are affected by multiple interacting stressors. Spears et al. (2022a) also asserted that the interactions among facets of climate change, and between climate change and other drivers such as nutrient enrichment, are currently difficult to detect and manage. For example, Battarbee et al. (2012) looked at limnological and palaeolimnological diatom data from seven lakes across Europe, including Loch Leven, to disentangle the effects of nutrient pollution and climate change on lake ecosystems. The evidence suggested that nutrient enrichment remains the dominant influence in Loch Leven, although some of the inter-annual variability in diatom seasonality may be attributed to changes in winter rainfall and temperature (Bennion et al, 2012). Although it is difficult to fully disentangle the relative importance of climate change impacts, there is some evidence that climate change may be offsetting some of the expected recovery of standing waters from a reduction in nutrient inputs. If so, algal growth may remain relatively high, following a recovery trajectory that is deflected away from the reference condition towards a new endpoint. A combination of palaeoecological and contemporary diatom data has also been used to examine the recovery of UK upland lakes (including six Scottish sites) from acidification (Battarbee et al, 2014). In a few cases, the diatom species composition of recent samples is different from those recorded before and during the acidification phase. The reasons were unclear, but nutrient enrichment from nitrogen deposition (e.g., Round Loch of Glenhead and Lochnagar) and climate change (e.g., Lochnagar) were beginning to play a greater role in driving water quality as acidity decreases. Again, it was found to be difficult to disentangle the warming effects of climate change from the effects of nitrogen enrichment, as both pressures are likely to cause symptoms of eutrophication (Moss et al, 2011). Spears et al. (2021a) applied a statistical approach to time series data from three European catchments to assess phytoplankton responses to the relative and interactive effects of climate change and nutrient enrichment. In all three cases, nutrient enrichment was identified as the primary stressor with indicators of climate change (such as water temperature) causing secondary effects.

It is clear from the above discussion that more evidence is needed to support and improve climate change resilience at the catchment scale and help restore impacted standing water systems. Understanding change in standing waters better could also include developing and diversifying indicators of climate change impacts to enable more effective monitoring (e.g., Defra, 2018; Hayhow et al., 2019). Other research challenges, as outlined by Spears (2021a), include:

- Exploring whether the effects of a large-scale driver, such as climate change, can be managed through local action such as better nutrient management
- Investigating whether there is potential for climate change adaptation options to have unintended consequences
- Gaining a better understanding of whether the various effects of climate change on ecosystems are greater than the sum of their parts
- Determining whether a large-scale driver such as climate change synchronises population fluctuations at regional scale and increases the likelihood of regional scale extinction

Thackeray and Hampton (2020) argue that, to increase our understanding of standing water ecosystem responses to global change (such as harmful cyanobacterial blooms) there is a need to investigate processes at multiple spatial, temporal and biological scales by integrating freshwater ecosystem data using both traditional (e.g., taxonomic) and newer (e.g., eDNA, remote sensing) research approaches. These processes could include identifying the sources and cycling of nutrients at catchment scale, the factors that affect delivery of those nutrients to waterbodies over different timescales, regional climate and local weather effects on the physical condition of standing waters, in-lake biological interactions, the evolutionary ecology of toxin production and functional genomics.

The main factors that need to be considered for mitigating climate change impacts on water quality of Scottish standing waters are summarised in Table A1.2.

Table A1.2 Summary of main factors that need to be considered for mitigating climate change impacts on water quality of Scottish standing water

Factor	Affected by	Considerations for mitigation of climate change impacts on water quality of Scottish standing waters
Flushing rate	rainfall/runoff/evaporation rates/abstraction	• cyanobacterial blooms will tend to increase during warm dry summers with reduced flushing rates and decrease during periods of high rainfall • increased storminess and resultant heavy rainfall will lead to increased inputs of terrestrial material derived from catchments and may restructure phytoplankton communities depending on resilience of standing water
Water level	rainfall/runoff/evaporation rates/abstraction/ bathymetry/rate & duration	• shallow standing waters are more vulnerable to climate change than deeper standing waters as larger proportion of water surface area and volume affected • shallow standing waters more sensitive to lower rainfall and reduced flushing rates with increased likelihood of cyanobacterial blooms • deeper standing waters likely to be more sensitive than shallow standing waters to the earlier onset and longer periods of thermal stratification
Nutrient/ sediment delivery	runoff (diffuse sources)/dilution (point sources)/internal release (sediment sources)/sediment disturbance	• timing of high rainfall events very important as will deliver majority of nutrients and sediment inputs into standing waters with very little delivered under low flow conditions – will affect response of standing waters • increased storminess and resultant high winds will lead to physico-chemical extremes in standing waters due to increased mixing of the water column
Water temperature change	rainfall/flushing rate/water level/temperature/ bathymetry	• above mean monthly temperature threshold of 17°C algal blooms, especially of cyanobacteria, are likely to increase in frequency (if sufficient nutrients are available) • - changes in zooplankton community likely to occur above mean monthly temperature threshold of 140C with knock-on effects on grazing pressure on phytoplankton and water quality • - strengthened thermal stratification will lead to trends of lower dissolved oxygen concentrations, resulting in increased releases of sediment-bound nutrients and contaminants into the overlying water and declines in water quality • - the increasing frequency of heatwaves will likely exacerbate the adverse effects of long-term climate change
Changes in seasonal timing of biological communities	changes in seasonal warming patterns	• increased likelihood of trophic mismatches leading to changes in plankton population interactions, threatening ecosystem functioning • shifts in species ranges predicted
Climate change complexities	multiple interacting stressors	• need to consider multiple interacting stressors other than climate change alone in order to determine the climate change resilience of individual standing waters • need to improve climate change resilience at catchment scale by developing and diversifying indicators of climate change impacts

References

Afazal, M., Gagnon, A. S. and Mansell, M. G. (2015). The impact of projected changes in climate variability on the reliability of surface water supply in Scotland. *Water Science & Technology: Water Supply*, 15, 736-745.

Arvola, L., George, G., Livingstone, D. M., Järvinen, M., Blenkner, T., Dokuil, M. T., Jennings, E., Nic Aonghusa, C., Nõges, P., Nõges, T. and Weyhenmeyer, G. A. (2010). The impact of climate change on the thermal characteristics of lakes, pages 85-101, in George, G. (ed.) *The Impact of Climate Change on European Lakes.* Springer, Dordrecht,

Arvoviita, J. and Heikki, H. (2008). The impact of water-level regulation on littoral macroinvertebrate assemblages in boreal lakes. *Hydrobiologia*, 613, 45-56.

Bailey-Watts, A. E., Kirika, A., May, L. and Jones, D. H. (1990). Changes in phytoplankton over various time scales in a shallow, eutrophic loch; the Loch Leven experience with special reference to the influence of flushing rate. *Freshwater Biology*, 23, 85-111.

Baldwin, D. S., Gigney, H., Wilson, J. S., Watson, G. and Boulding, A. N. (2008). Drivers of water quality in a large water storage reservoir during a period of extreme drawdown. *Water Research*, 42, 4711-4724.

Baumgaertner, D., Moertl, M. and Rothhaupt, K. O. (2008). Effects of water-depth and water-level fluctuations on the macroinvertebrate community structure in the littoral zone of Lake Constance. *Hydrobiologia*, 613, 97-107.

Battarbee, R. W., Grytnes, J. A., Thompson, R., Appleby, P. G., Catalan, J. Korhola, A., Birks, H. J. B., Heegaard, E. and Lami, A. (2002). Comparing palaeolimnological and instrumental evidence of climate change for remote mountain lakes over the last 200 years. *Journal of Paleolimnology*, 28, 161-179.

Battarbee, R. W., Anderson, N., Bennion, H. and Simpson, G. L. (2012). Combining limnological and palaeolimnoligical data to disentangle the effects of nutrient pollution and climate change on lake ecosystems: potential problems and potential. *Freshwater Biology*, 57, 209-2106.

Battarbee, R. W., Simpson, G. L., Shilland, E. M., Flower, R. J., Kreiser, A., Yang, H. and Clarke, G. (2014). Recovery of UK lakes from acidification: An assessment using combined paleoecological and contemporary diatom assemblage data. *Ecological Indicators*, 37, 365-380.

Bennion, H., Carvalho, L., Sayer, C., Simpson, G. L. and Wischenewski, J. (2012). Identifying from recent sediment records the effects of nutrients and climate on diatom dynamics in Loch Leven. *Freshwater Biology*, 57, 2015-2029.

Bruel, R., Marchetto, Bernard, A., Lami, A., Sabatier, P., Frossard, V. and Perga, M. E. (2018). Seeking alternative stable states in a deep lake. *Freshwater Biology*, 63, 553-568.

Carvalho, L., McDonald, C., de Hoyos, C., Mischke, U., Phillips, G., Borics, G., Poikane, S., Skjelbred, B., Lyche Solheim, A., Van Wichelen, J. and Cardoso, A. C. (2013). Sustaining recreational quality of European lakes: minimising the health risks from algal blooms through phosphorus control. *Journal of Applied Ecology*, 50, 315-323.

Carvalho, L., Miller, C. A., Scott, E. M., Codd, G. A., Davies, P. S. and Tyler, A .N. (2011). Cyanobacterial blooms: Statistical models describing risk factors for national-scale lake assessment and lake management. *Science of the Total Environment*, 409, 5353–5358.

Carvalho, L., Miller, C., Spears, B. M., Gunn, I. D. M., Bennion, H., Kirika, A. and May, L. (2012). Water quality of Loch Leven: responses to enrichment, restoration and climate change. *Hydrobiologia*, 681, 35-47.

Climate Change Committee (2021). *Independent Assessment of UK Climate Risk: Advice to Government for the Third Climate Risk Assessment (CCRA3)*. CCC, June 2021.

Climate Change Committee (2022). *Is Scotland climate ready? – 2022 Report to Scottish Parliament.* CCC, March 2022.

Collins, A. M. Coughlin, D., Miller, J. and Kirk, S. (2015). *The Production of Quick Scoping Reviews and Rapid Evidence Assessments: A How to Guide.*

Cox, P. A., Kostrzewa, R. M. and Guillemin, G. J. (2018). BMAA and Neurodegenerative Illness. *Neurotoxicity Research*, 33, 178-183.

Defew, L. H., May, L. and Heal, K. (2013). Uncertainties in estimated phosphorus loads as a function of different sampling frequencies and common calculation methods. *Marine and Freshwater Science*, 64, 373-386.

Defra (2018). *A Green Future: Our 25 Year Plan to Improve the Environment.* Defra report, January 2018.

Dillon, P. J. and Rigler, F. H. (1974.) The phosphorus-chlorophyll relationship in lakes. *Limnology and Oceanography*, 19, 767-773.

Dobel, A. J., May, L., Gunn, I., Spears, B. and Edwards, F. (2020). *Lakes and Reservoirs Report Card 2020.* About Drought-UK's Drought and Water Scarcity Research Programme. https://aboutdrought.info/wp-content/uploads/2020/08/AboutDrought-ReportCard-Lakes-Reservoirs-Final.pdf

Dokulil, M. T., de Eyto, E., Maberly, S. C., May, L., Weyhenmeyer, G. A. and Woolway, R. I. (2021). Increasing maximum lake surface temperature under climate change. *Climatic Change*, 165, 56.

Duigan, C. A., Kovach, W. L. and Palmer, M. (2006). Vegetation communities of British lakes: a revised classification. JNCC, Peterborough.

Elliott, J. A. (2010). The seasonal sensitivity of Cyanobacteria and other phytoplankton to changes in flushing rate and water temperature. *Global Change Biology,* 16, 864-87.

Elliott, J. A. (2012a). Is the future blue-green? A review of the current model predictions of how climate change could affect pelagic freshwater cyanobacteria. *Water Research*, 46, 1364-1371.

Elliott, J. A. (2012b). Predicting the impact of changing nutrient load and temperature on the phytoplankton of England's largest lake. *Freshwater Biology*, 57, 400-413.

Elliott, J. A. and Bell, V. A. (2011). Predicting the potential long-term influence of climate change on vendace (Coregonus albula) habitat in Bassenthwaite Lake, U.K. *Freshwater Biology*, 56, 395-405.

Elliott, J. A. and Defew, L. (2012). Modelling the response of phytoplankton in a shallow lake (Loch Leven, UK) to changes in lake retention time and water temperature. *Hydrobiologia*, 681, 105-116.

Elliott, J. A., Henrys, P., Tanguy, M., Cooper, J. and Maberly, S. C. (2015). Predicting the habitat expansion of the invasive roach Rutilus rutilis (Actinopterygii, Cyprinidae), in Great Britain. *Hydrobiologia*, 751, 127-134.

Elliott, J. A., Irish, A. E. and Reynolds, C. S. (2010) Modelling Phytoplankton Dynamics in Fresh Waters: Affirmation of the PROTECH Approach to Simulation. *Freshwater Reviews*, 3, 75-96.

Elliott, J. A., Jones, I. D. and Page, T. (2009.) The importance of nutrient source in determining the influence of retention time on phytoplankton: an explorative modelling study of a naturally well-flushed lake. *Hydrobiologia*, 627, 129-142.

Facciponte, D. N., Bough, M. W., Seidler, D., Carroll, J. L., Ashare, A., Andrew, A. S., Tsongalis, G. J., Vaickus, L. J., Henegan, P. L., Butt, T. H. and Stommel, E. W. (2018). Identifying aerosolized cyanobacteria in the human respiratory tract: A proposed mechanism for cyanotoxin-associated diseases. *Science of the Total Environment*, 645, 1003-1013.

Foley, B., Jones I. D., Maberly S. C. and Rippey B. (2012). Long-term changes in oxygen depletion within a small temperate lake: Effects of climate change and eutrophication. *Freshwater Biology*, 57, 278-289.

Futter, M. N., Helliwell, R. C., Hutchins, M. and Aherne, E. (2009). Modelling the effects of changing climate and nitrogen deposition on nitrate dynamics in a Scottish mountain catchment. *Hydrology Research*, 40, 153-166.

George, G., Hurley, M. and Hewitt, D. (2007). The impact of climate change on the physical characteristics of the larger lakes in the English Lake District. *Freshwater Biology*, 52, 1647–1666.

George, D. G., Maberly, S. C. and Hewitt, D. P. (2004).The influence of the North Atlantic Oscillation on the physics, chemistry and biology of four lakes in the English Lake District. *Freshwater Biology*, 49, 760-774.

George, D. G. and Taylor, A. H. (1995). U.K. lake plankton and the Gulf Stream. *Nature*, 378, 139.

Gunn, I. D. M., Dobel, A. J. and May, L. (2021). *Task 1 – Environmental issues in lakes affected by changes in environmental flows (eflows): Review of issues.* Report to Defra.

Gyllström, M., Hansson, L. A., Jeppesen, E., Garcia-Criado, F., Gross, E., Irvine, K., Kairesalo, T., Kornijow, R., Miracle, M. R., Nykanen, M., Nõges, T., Romo, S., Stephen, D., Van Donk, E. and Moss, B. (2005). The role of climate in shaping zooplankton communities of shallow lakes. *Limnology & Oceanography*, 50, 2008-2021.

Hassan, H., Aramaki, T., Hanaki, K., Matsuo, T. and Wilby, R. L. (1998). Lake stratification and temperature profiles simulated using downscaled GCM output. *Water Science & Technology*, 38, 217–226.

Hayhow, D. B., Eaton, M. A., Stanbury, A. J., Burns, F., Kirby, W. B., Bailey, N., Beckmann, B., Bedford, J., Boersch-Supan, P. H., Coomber, F., Dennis, E. B., Dolman, S. J., Dunn, E., Hall, J., Harrower, C., Hatfield, J. H., Hawley, J., Haysom, K., Hughes, J., Johns, D. G., Mathews, F., McQuatters-Gollop, A., Noble, D. G., Outhwaite, C. L., Pearce-Higgins, J. W., Pescott, O. L., Powney, G. D. and Symes. N. (2019). *The State of Nature 2019*. The State of Nature partnership.

Ho, J. C., Michalak, A. M. and Pahlevan, N. (2019). Widespread global increase in intense lake phytoplankton blooms since the 1980s. *Nature*, 574, 667-670.

Jane, S. F., J. A. Hansen, G. J. A., Kraemer, B. M., Leavitt, P. R., Mincer, J. L., North, R. L., Pilla, R. M., Stetler, J. T., Williamson, C. E., Woolway, R. I., Lauri Arvola, L., Chandra, S., DeGasperi, C. L., Diemer, L., Dunalska, J., Erina, O., Flaim, G., Grossart, H-P., Hambright, K. D., Hein, C., Hejzlar, J., Janus, L. L., Jenny, J-P., Jones, J. R., Knoll, L. B., Leoni, B., Mackay, E., Matsuzaki, S-I. S., McBride, C., Müller-Navarra, D. C., Paterson, A. M, Pierson, D., Rogora, M., Rusak, J. A., Sadro, S., Saulnier-Talbot, E., Schmid, M., Sommaruga, R., Thiery, W., Verburg, P., Weathers, K. C., Weyhenmeyer, G. A., Yokota, K. and Rose, K. C. (2021). Widespread deoxygenation in temperate lakes. *Nature*, 594, 66-70.

Jennings, E., de Eyto, E., Jones, I., Ibelings, B., Adrian, R. and Woolway, R. I. (2021). Ecological consequences of Climate Extremes in Lakes. In *Encyclopedia of Inland Waters*, (2nd Edition). Elsevier.

Jirsa, F., Gruber, M., Stojanovic, A., Omondi, S. O., Mader, D., Körner, W. and Schagerl, M. (2013). Major and trace element geochemistry of Lake Bogoria and Lake Nakuru, Kenya, during extreme drought. *Chemie der Erde-Geochemistry*, 73, 275-282.

Jones, I., Abrahams, C., Brown, L., Dale, K., Edwards, F., Jeffries, M. Klaar, M., Ledger, M., May, L., Milner, A., Murphy, J., Robertson, A. and Woodward, G. (2013). *The impact of extreme events on freshwater ecosystems*. London, British Ecological Society, 67pp. (Ecological Issues, 12).

Jones, I., George, G. and Reynolds, C. (2005). Quantifying the Effects of Phytoplankton on the Summer Heat Budget of Large Limnetic Enclosures. *Freshwater Biology*, 50, 1239–1247.

Jones, L., Gorst, A., Elliott, J., Fitch, A., Illman, H., Evans, C., Thackeray, S., Spears, B., Gunn, I., Carvalho, L., May, L., Schonrogge, K., Clilverd, H., Mitchell, Z., Garbutt, A., Taylor, P., Fletcher, D., Giam, G., Aron, J., Ray, D., Berenice-Wilmes, S., King, N., Malham, S., Fung, F., Tinker, J., Wright, P. and Smale, R. (2020). *Climate driven threshold effects in the natural environment.* Report to the Climate Change Committee, May 2020.

Jones, I. D., Page, T., Elliott, J. A., Thackeray, S. J. and Heathwaite, A. L. (2011). Increases in lake phytoplankton biomass caused by future climate-driven changes to seasonal river flow. *Global Change Biology*, 17, 1809-1820.

Jones, I. D., Winfield, I. J. and Carse, F. (2008). Assessment of long-term changes in habitat availability for Arctic charr (Salvelinus alpinus) in a temperate lake using oxygen profiles and hydroacoustic surveys. *Freshwater Biology*, 53, 393-402.

Jöhnk, K. D., Huisman, J., Sharples, J., Sommeijer, B., Visser, P. M. and Stroom, J. M. (2008). Summer heatwaves promote blooms of harmful cyanobacteria. *Global Change Biology*, 14, 495-512.

Philip Jordan, P., Melland, A. R., Mellander, P.-E., Shortle, G. and Wall, D. (2012). The seasonality of phosphorus transfers from land to water: Implications for trophic impacts and policy evaluation. *Science of the Total Environment*, 434, 101-109.

Kosten, S., van den Berg, S., Mendonca, R., Paranaiba, J. R., Roland, F., Sobek, S., Van Den Hoek, J. and Barros, B. (2018). Extreme drought boosts CO2 and CH4 emissions from reservoir drawdown areas. *Inland Waters*, 8, 329-340.

Krokowski, J. (2007). Changes in the trophic state and phytoplankton composition and abundance in Loch Lomond, Scotland, UK. *Oceanographical and Hydrobiological Studies*, 36.

Krokowski, J. T., Lang. P., Bell, A., Broad, N., Clayton, J., Milne, I., Nicolson, M., Ross, A. and Ross, N. (2012). A review of the incidence of cyanobacteria (blue-green algae) in surface waters in Scotland including potential effects of climate change, with a list of the common species and new records from the Scottish Environment Protection Agency. *The Glasgow Naturalist*, 25, 99-104.

Laizé, C., Acreman, M. and Overton, I. (2017). Projected novel eco-hydrological river types for Europe. *Ecohydrology & Hydrobiology*, 17, 73-83.

Leavitt, P. R., Fritz, S. C., Anderson, N. J., Baker, P. A., Blencker, T., Bunting, L., Catalan, J., Conley, D. J., Hobbs, W. O., Jeppesen, E., Korhola, A., McGowan, S., Rühland, K., Rusak, J. A., Simpson, G. L., Solovieva, N. and Werne, J. (2009). Paleolimnological evidence of the effects on lakes of energy and mass transfer from climate and humans. *Limnology and Oceanography*, 546, 2330-2348.

Lyle, A. A. and Smith, I. R. (1994). Standing Waters in Maitland, P. S., Boon, P. J. and McLusky, D. S. eds *The Fresh Waters of Scotland: A National Resource of International Significance.* Wiley, London, pp 35-50.

Maberly, S. C., Barker, P. A., Stott, A. W. and De Ville, M. M. (2013). Catchment productivity controls CO2 emissions from lakes. *Nature Climate Change*, 3, 391-394.

Mackay, E., Folkhard, A. M. and Jones, I. D. (2014). Interannual variations in atmospheric forcing determine trajectories of hypolimnetic soluble reactive phosphorus supply in a eutrophic lake. *Freshwater Biology*, 59, 1646-1658.

Milly, P. C. D., Dunne, K. A. and Vecchia, A. V. (2005) Global pattern of trends in streamflow and water availability in a changing climate. *Nature*, 438, 347–350.

Monteith, D., Pickard, A. E., Spears, B. M. and Fuechtmayr, H. (2021). *How will climate change influence levels of dissolved organic matter in upland drinking water sources?* FREEDOM-BCRR briefing note 5 to the water industry. UKRI SPF UK Climate Resilience programme - Project no. NE/S016937/2.

Moore, K., Jennings, E., Allot, N., May, L., Järvinen, M. Arvola, L., Tamm, T., Järvet, A., Nõges, T., Pierson, D. and Schneiderman, E. (2010). Modelling the effects of climate change on the supply of inorganic nitrogen. In: *The Impact of Climate Change on European Lakes* (Ed. G. George), pp. 179-197. Springer, Dordrecht.

Mosley, L. M. (2015). Drought impacts on the water quality of freshwater systems: review and integration. *Earth-Science Reviews*, 140, 203-2014.

Moss, B., Kosten, S., Meerhoff, M., Battarbee, R. W., Jeppesen, E., Mazzezo, N. Havens, K. G., Liu, L., De Meester, Z., Paerl, L. and Schaefer, H. M. (2011). Allied attack, climate change and eutrophication. *Inland Waters*, 1, 101-105.

Moss, B., McKee, D., Atkinson, D., Collings, S. E., Eaton, J. W., Gill, A. B., Hatton, H. K., Heyes, T. and Wilson, D. (2003), How important is climate? Effects of warming, nutrient addition and fish on phytoplankton in shallow lake mesocosms. Journal of Applied Ecology, 40, 782-792.

Muir, M. C. A., Spray, C. J. and Rowan, J. S. (2012). Climate change and standing freshwaters: informing adaption strategies for conservation at multiple scales. *Area*, 44, 411-422.

Nijssen, B., O'Donnell, G. M., Hamlet, A.F. and Lettenmaier, D. P. (2001) Hydrologic sensitivity of global rivers to climate change. *Climatic Change*, 50, 143–175.

OECD (1979). *Shallow Lakes and Reservoirs*. Final Report Vol. 1 & 2 to OECD Cooperative Programme for Monitoring of Inland waters (Eutrophication Control).

O'Reilly, C. M., Sharma, S., Gray, D. K., Hampton, S. E., Read, J. S., Rowley, R. J., Schneider, P., Lenters, J. D., McIntyre, P. B., Kraemer, B. M., Weyhenmeyer, G. A., Straile, D., Dong, B., Adrian, R., Allan, M. G., Anneville, O., Arvola, L., Austin, J., Bailey, J. L., Baron, J. S., Brookes, J. D., de Eyto, E., Dokulil, M. T., Hamilton, D. P., Havens, K., Hetherington, A. L., Higgins, S. N., Hook, S., Izmest'eva, L. R., Joehnk, K. D., Kangur, K., Kasprzak, P., Kumagai, M., Kuusisto, E., Leshkevich, G., Livingstone, D. M., MacIntyre, S., May, L., Melack, J. M., Mueller-Navarra, D. C., Naumenko, M., Noges, P., Noges, T., North, R. P., Plisnier, P., Rigosi, A., Rimmer, A., Rogora, M., Rudstam, L. G., Rusak, J. A., Salmaso, N., Samal, N. R., Schindler, D. E., Schladow, S. G., Schmid, M., Schmidt, S. R., Silow, E., Evren Soylu, M., Teubner, K., Verburg, P., Voutilainen, A., Watkinson, A., Williamson, C. E., and Zhang, G. (2015). Rapid and highly variable warming of lake surface waters around the globe. *Geophysical Research Letters,* 10773-10781

Paerl, H. W. and Huisman, J. (2008). Blooms like it hot. *Science*, 320, 57-58.

Persson, I. and Jones, I. D. (2008). The effect of lake colour on lake hydrodynamics: a modelling study. *Freshwater Biology*, 53, 2345–2355.

Pierson, D., Arvola, L., Allott, N., Jarvinen, M., Jennings, E., May, L., Moore, K. and Schneiderman, E. (2010). Modelling the effects of climate change on the supply of phosphate-phosphorus. In: *The Impact of Climate Change on European Lakes* (Ed. G. George), pp. 121-137. Springer, Dordrecht.

Pilla, R. M., Williamson, C. E., Adamovich, B. V., Adrian, R., Anneville, O., Chandra, S., Colom-Montero, W., Devlin, S. P., Dix, M. A., Dokulil, M. T., Gaiser, E. E., Girdner, S. F., Hambright, K. D., Hamilton, D. P., Havens, K., Hessen, D. O., Higgins, S. N., Huttula, T. H., Huuskonen, H., Isles, P. D. F., Joehnk, K. D., Jones, I. D., Keller, W. B., Knoll, L. B., Korhonen, J., Kraemer, B. M., Leavitt, P. R., Lepori, F., Luger, M. S., Maberly, S. C., Melack, J. M., Melles, S. J., Müller-Navarra, D. C., Pierson, D. C., Pislegina, H. V., Plisnier, P.-D., Richardson, D. C., Rimmer, A., Rogora, M., Rusak, J. A., Sadro, S., Salmaso, N., Saros, J. E., Saulnier-Talbot, É., Schindler, D. E., Schmid, M., Shimaraeva, S. V., Silow, E. A., Sitoki, L. M., Sommaruga, R., Straile, D., Strock, K. E., Thiery, W., Timofeyev, M. A., Verburg, P., Vinebrooke, R. D., Weyhenmeyer, G. A., and Zadereev, E. (2020). Deeper waters are changing less consistently than surface waters in a global analysis of 102 lakes. Scientific Reports, 10, 20514. https://doi.org/10.1038/s41598-020-76873-x

Reynolds, C. S., Irish, A. E. and Elliott, J. A. (2001). The ecological basis for simulating phytoplankton responses to environmental change (PROTECH). *Ecological Modelling*, 140, 271–291.

Reynolds, C. S., Maberly, S. C., Parker, J. E. and De Ville, M. M. (2012). Forty years of monitoring water quality in Grasmere (English Lake District): separating the effects of enrichment by treated sewage and hydraulic flushing on phytoplankton ecology. *Freshwater Biology*, 57, 384-399.

Richardson, J., Miller, C., Maberly, S. C., Taylor, P., Globevnik, L., Hunter, P., Jeppesen, E., Mischke, U., Moe, J., Pasztaleniec, A., Søndergaard, M. and Carvalho, L. (2018). Effects of multiple stressors on cyanobacteria abundance vary with lake type. *Global Change Biology*, 24, 5044-5055.

Ritson, J. P., Graham, N. J. D., Templeton, M. R., Clark, J. M., Gough, R. and Freeeman, C. (2014). The impact of climate change on the treatability of dissolved organic matter (DOM) in upland water supplies: A UK perspective. *Science of the Total Environment*, 473-474, 714-730.

Samplonius, J. M., Atkinson, A., Hassall, C., Keogan, K., Thackeray, S. J., Assmann, J. J., Burgess, M. D., Johansson, J., Macphie, K. H., Pearce-Higgins, J. W., Simmonds, E. G., Varpe, Ø., Weir, J. C., Childs, D. Z., Cole, E. F., Daunt, F., Hart, T., Lewis, O. T., Pettorelli, N., Sheldon, B. C., and Phillimore, A. B. (2021). Strengthening the evidence base for temperature-mediated phenological sensitivity asynchrony and its impacts. *Nature Ecology & Evolution*, 5, 155-164. https://doi.org/10.1038/s41559-020-01357-0

Schindler, D. W. (2001). The cumulative effects of climate warming and other human stresses on Canadian freshwaters in the new millennium. *Canadian Journal of Fisheries and Aquatic Sciences*, 65, 878-889.

Sharpley, A. N. (2008). Phosphorus loads from an agricultural watershed as a function of storm size. *Journal of Environmental Quality*, 37, 362-368.

Smith, B. D., Maitland, P. S. and Pennock, S. M. (1987). A comparative study of water level regimes and littoral benthic communities in Scottish lochs. *Biological Conservation*, 39, 291–316.

Smith, I. R. and Lyle, A. A. (1979). Distribution of Freshwaters in Great Britain. *Institute of Terrestrial Ecology*, Cambridge.

Spears, B. M., Carvalho, L. and Paterson, D. M. (2007a). Phosphorus partitioning in a shallow lake: implications for water quality management. *Water Environment Journal*, 21, 47–53.

Spears, B. M., Carvalho, L., Perkins, R., Kirika, A. and Paterson, D. M. (2007b). Sediment phosphorus cycling in a large shallow lake: spatio-temporal variation in phosphorus pools and release. *Hydrobiologia,* 584: 37–48.

Spears, B. M., Carvalho, L., Perkins, R., Kirika, A. and Paterson, D. M. (2012). Long-term variation and regulation of internal phosphorus loading in Loch Leven. *Hydrobiologia*, 681, 23-33.

Spears, B. M., Chapman, D. S., Carvalho, L., Feld, C. K., Gessner, M. O., Piggott, J. J., Banin, L. F., Gutiérrez-Cánovas, C., Lyche Solheim, A., Richardson, J. A., Schinegger, R. Segurado, P., Thackeray, S. J. and Birk, S. (2021a). Making waves. Bridging theory and practice towards multiple stressor management in freshwater ecosystems. *Water Research*, 196, 116981.

Spears, B. M. Chapman, D., Carvalho, L., Rankinen, K., Stefanidis, K., Ives, S., Vuorio, K. and Birk, S. (2022a). Assessing Multiple Stressor Effects to Inform Climate Change Management Responses in Three European Catchments. Inland Waters. DOI:10.1080/20442041.2020.1827891

Spears, B. M., Hamilton, D. P., Pan, Y., Zhaosheng, C. and May, L. (2022b). Lake management: is prevention better than cure? *Inland Waters*. DOI: 10.1080/20442041.2021.1895646.

Stockwell, J. D., Doubek, J. P., Adrian, R., Anneville, O., Carey, C. C, Carvalho, L., De Senerpont Domis, L. N., Dur, G., Frassl, M. A., Grossart, H.-P., Ibelings, B. W., Lajeunesse, M. J., Lewandowska, A. M., Llames, M. E., Matsuzaki, S.-I. S., Nodine, E. R., Nõges, P., Patil, V. P., Pomati, F., Rinke, K., Rudstam, L. G., Rusak, J. A., Salmaso, N., Seltmann, C. T., Straile, D., Thackeray, S. J., Thiery, W., Urrutia-Cordero, P., Venail, P., Verburg, P., Woolway, R. I., Zohary, T., Andersen, M. R., Bhattacharya, R., Hejzlar, J., Janatian, N., Kpodonu, A. T. N. K., Williamson, T. J., and Wilson, H. L. (2020). Storm impacts on phytoplankton community dynamics in lakes. *Global Change Biology*, 26, 2756-2784.

Strong, C. and Maberly, S. C. (2011). The influence of atmospheric wave dynamics on the surface temperature of lakes in the English Lake District. *Global Change Biology*, 17, 2013-2022.

Thackeray, S. J. and Hampton, S. E. (2020). The case for research integration, from genomics to remote sensing, to understand biodiversity change and functional dynamics in the world's lakes. *Global Change Biology*, 26, 3230-3240.

Thackeray, S. J, Jones, I, D and Maberly, S. C. (2008). Long-term change in the phenology of spring phytoplankton: species-specific responses to nutrient enrichment and climatic change. *Journal of Ecology*, 96, 523– 535.

Thackeray, S. J., Sparks, T. H., Frederiksen, M., Burthe, S., Bacon, P. J., Bell, J. R., Botham, M. S., Brereton, T. M., Bright, P. W., Carvalho, L., Clutton-Brock, T., Dawson, A., Edwards, M., Elliott, J. M., Harrington, R., Johns, D., Jones, I. D., Jones, J. T., Leech, D. I., Roy, D. B., Scott, W. A., Smith, M., Smithers, R. J., Winfield, I. J., and Wanless, S. (2010). Trophic level asynchrony in rates of phenological change for marine, freshwater and terrestrial environments. *Global Change Biology*, 16, 3304– 3313.

Thackeray, S. J., Henrys, P. A., Feuchtmayr, H., Jones, I. D., Maberly, S. C., and Winfield, I. J. (2013). Food web de-synchronization in England's largest lake: an assessment based on multiple phenological metrics. *Global Change Biology*, 19, 3568-3580.

Tickner, D., Opperman, J. J., Abell, A., Acreman, A., Arthington, A. H., Bunn, S. E., Cooke, S. J., Dalton, J., Darwall, W., Edwards, G., Harrison, I., Hughes, K., Jones, T., Leclère, D., Lynch, A. J., Leonard, P., McClain, M. E., Muruven, D., Olden, J. D., Ormerod, S J., Robinson, J., Tharme, R. E., Thieme, M., Tockner, K., Wright, M. and Young, L. (2020). Bending the Curve of Global Freshwater Biodiversity Loss: An Emergency Recovery Plan. *BioScience*, 70, 330-342.

Tranvik, L. J. et, al. (2009). Lakes and reservoirs as regulators of carbon cycling and climate. *Limnology and Oceanography*, 54, 2298-2314.

Van der Spoel, M. (2019). *Identifying niche requirements of cylindrospermopsis and nine other common European cyanobacteria genera*. University of Edinburgh, GeoSciences, Dissertation, April 2019.

Vollenweider, R. A. (1975). Input-output models; with special reference to the phosphate loading concept in limnology. *Schweizerische Zeitschrift für Hydrologie*, 37, 53-84.

Wagner, C. and Adrian, R. (2009). Cyanobacteria dominance – quantifying the effects of climate change. *Limnology and Oceanography,* 54, 2460-2468.

Wantzen, K. M., Rothhaupt, K. O., Mörtl, M., Cantonati, M. T., Tóth, L. G. and Fischer, P. (2008). Ecological effects of water-level fluctuations in lakes: an urgent issue. *Hydrobiologia*, 613, 1-4.

Webster, K. E., Kratz, T. K., Bowser, C. J., Magnuson, J. J. and Rose, W. J. (1996). The Influence of Landscape Position on Lake Chemical Responses to Drought in Northern Wiscosin Lakes. *Limnology and Oceanography*, 41, 977-984.

Webster, K. E., Soranno, P. A., Baines, S. B., Kratz, T. K., Bowser, C. J., Dillon, P. J., Campbell, P., Fee, F. J. and Hecky, R. E. (2000). Structuring features of lake districts: landscape controls on lake chemical responses to drought. *Freshwater Biology*, 43, 499-515.

White, M. S., Xenopoulos, M. A., Hodgsen, K., Metcalfe, R. A. and Dillon, P. J. (2008). Natural lake level fluctuation and associated concordance with water quality and aquatic communities within small lakes of the Laurentian Great Lakes region. *Hydrobiologia*, 613, 21-31.

Whitehead, P. G., Wilby, R. L., Battarbee, R. W., Kernan, M. and Wade, A. J. (2009). A review of the potential impacts of climate change on surface water quality. *Hydrological Sciences Journal*, 54, 101-123.

Wiedner, C. Rucker, J. Fastner, J., Chorus, I. and Nixdorf, B. (2008). Seasonal dynamics of cylindrospermopsin and cyanobacteria in two German lakes. *Toxicon*, 52, 677-686.

Williamson, C. E., Saros, J. E., Vincent, W. F. and Smol, J. P. (2009). Lakes and reservoirs as sentinels, integrators and regulators of climate change. *Limnology and Oceanography*, 54, 2273-2282.

Winder, M. and Schindler, D. E. (2004) Climate change uncouples trophic interactions in an aquatic ecosystem. *Ecology*, 85, 2100– 2106.

Winfield, I. J. (2004). Fish in the littoral zone: ecology, threats and management. Limnologica, 34, 124-131.

Winter, T. C. (2003). The hydrology of lakes. O'Sullivan, P. E. and Reynolds, C. S. eds. *In The Lakes Handbook, Volume 1.* Wiley-Blackwell, pp 61-78.

Woolway, R. I., Kraemer, B. M., Lenters, J. D., Merchant, C. J., O'Reilly, C. M. and Sharma, S. (2020a). Global lake responses to climate change. *Nature Reviews Earth & Environment,* 1, 388-403.

Woolway, R. I., Jennings, E. and Carrea, L. (2020b). Impact of the 2018 European heatwave on lake surface temperature. *Inland Waters*, 10, 322-332.

Iestyn Woolway, R. I., Sharma, S., Weyhenmeyer, G. A., Debolskiy, A., Golub, M., Mercado-Bettín, D., Perroud, M., Stepanenko, V., Tan, Z., Grant, L., Ladwig, R., Mesman, J., Moore, T. N., Shatwell, T., Vanderkelen, I., Austin, J. A., DeGasperi, C. L., Martin Dokulil, M., La Fuente, S., Mackay, E. B., Schladow, S.G., Watanabe, S., Marcé, R., Pierson, D. C., Thiery, W. and Eleanor Jennings, E. (2021a). Phenological shifts in lake stratification under climate change. *Nature Communications*, 12.

Woolway, R. I., Simpson, J. H., Spilby, D. Feuchtmayer, H., Powell, B. and Maberly, S. C. (2018). Physical and chemical impacts if a major storm on a temperate lake. A taste of things to come? *Climatic Change*, 15, 333-347.

Woolway, R. I., Jennings, E., Shatwell, T., Golub, M., Pierson, D. C. and Maberly, S. C. (2021b). Lake heatwaves under climate change. *Nature*, 589, 402-407.

Wright, R. F. and Schindler, D. W. (1995). Interaction of acid rain and global changes: effects of terrestrial and aquatic ecosystems. *Water Air and Soil Pollution*, 85, 89-99.

2 Appendix 2: Survey of expert opinion

Expert opinion on the perceived impacts of climate change on Scottish standing waters was gathered from Scotland, other parts of the UK and beyond by setting up an on-line survey based on questions that were agreed with the CREW Facilitation Team (CFT).

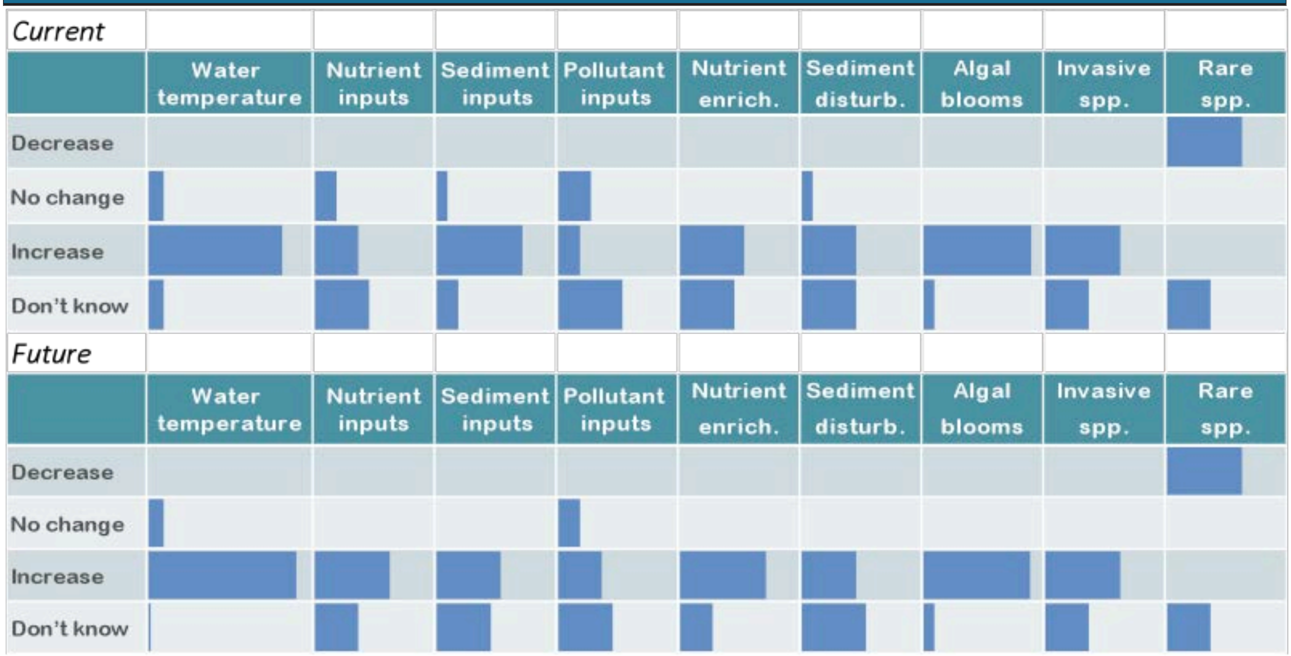
The opinions of participants were sought on the following issues:

- Current and future impacts of climate change (rainfall, temperature, wind) on standing water ecosystems, especially in terms of nutrients, algal blooms and other ecological responses.
- Current and future effects of climate change on water use (recreation, water supply etc.) and on Scotland's ability to meet regulatory and policy requirements.

Survey data

In November 2021, the project undertook a survey of expert opinion to gain an understanding of the perceived effects of climate change on Scottish standing waters. Eleven replies were received, equating to a response rate of 44 per cent. Replies were received from scientists (50%), policy makers (30%) and environmental regulators (20%) and respondents expressed concern about a wide range of standing waters. These included lochs (100%), reservoirs (55%), locally important still waters (73%), ponds and other small waterbodies (18%).

Table A4.1 Expert opinions on types and direction of change in environmental and ecological water quality caused by climate change so far (Current) and likely to be caused by climate change in the future (Future). *Blue bars indicate percentage of respondents selecting each category; maximum 100%.*



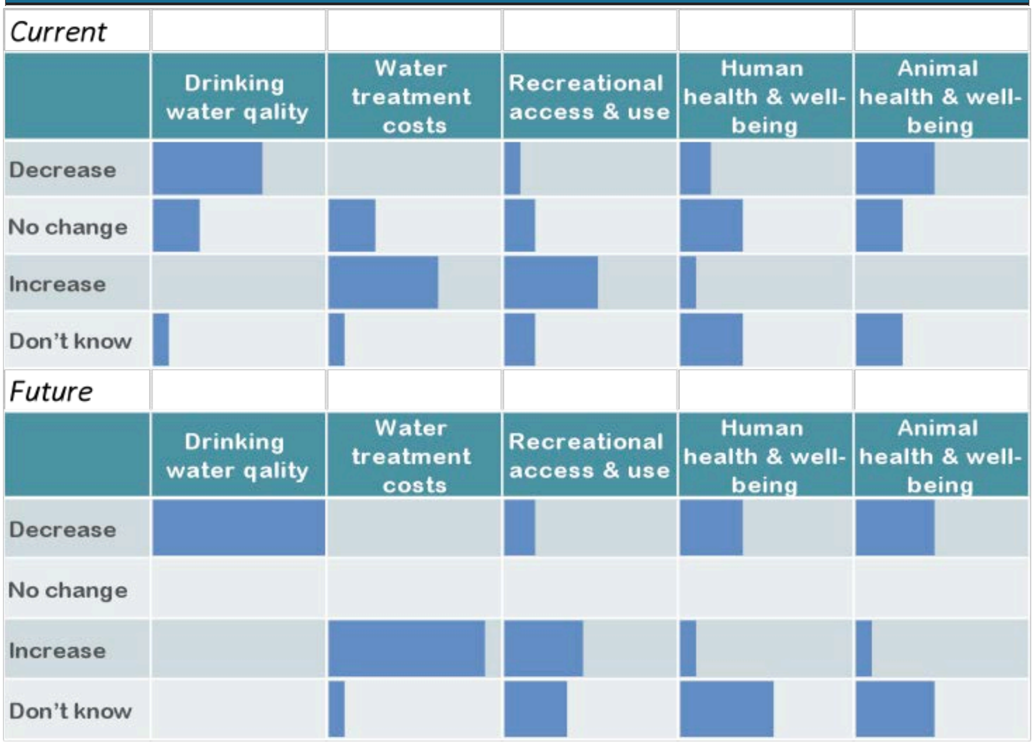
When asked what they believed were the main types and directions of change in environmental and ecological parameters under climate change at the moment, most respondents thought that temperature would increase (82%) but fewer respondents expected an increase in inputs of nutrients (36%), sediments (73%) or other forms of pollution (18%) from catchments (Table A4.1). In terms of within waterbody responses, concerns were expressed about increases in nutrient enrichment (54%), sediment disturbance (46%), algal blooms (91%) and invasive species (64%) and decreases in rare or sensitive species. In addition, many respondents were concerned about a decrease in biodiversity (64%), especially in relation to rare or sensitive species (64%). When asked about their expectations of future climate change impacts, the responses received were very similar.

Concerns about other types of water quality issues were also raised. These included:

- Increases in water colour
- Longer duration of stratification
- Less resilience to introduced or invasive non-native species
- Less ice cover
- Greater variation in water levels, especially in reservoirs

In terms of the impacts of climate change on the use of Scottish standing waters, most respondents expected water quality to decrease now (64%) and in the future (100%), leading to higher water treatment costs now (64%) and in the future (91%) (Table A4.2). Although recreational use of these waterbodies was thought to have increased already (55%) a lower number of respondents (46%) expected this to continue into the future. In addition, few believed that this was already affecting human health and well-being (18%) or expected this to affect human health and well-being in the future (18%). In contrast, 46% of respondents expected animal health and well-being to be adversely affected. Overall, there was a general feeling that recreational visits to standing waters would increase under climate change, but that water quality would decrease. A concern was raised that the risk of contact with potentially harmful algal blooms had increased due to blooms having become more common under climate change.

Table A4.2 Expert opinions on types and direction of change in factors that are likely to have been caused by climate change so far (Current) and likely to be caused by climate change in the future (Future). *Blue bars indicate percentage of respondents selecting each category; maximum 100%.*



When asked whether climate change would change Scotland's capacity to meet water quality and water management objectives for standing waters, 64% of respondents thought that climate change impacts would decrease our capacity to meet statutory environmental objectives within regulatory relevant timescales (Table A4.3). Concern was raised by 46% of respondents that the capacity for water managers to achieve water quality improvements or prevent further deterioration would also be reduced.

Table A4.3 Expert opinions on types and direction of change in our capacity to meet water policy and management objectives that are likely to have been caused by climate change so far (Current) and likely to be caused by climate change in the future (Future). *Blue bars indicate percentage of respondents selecting each category; maximum 100%..*

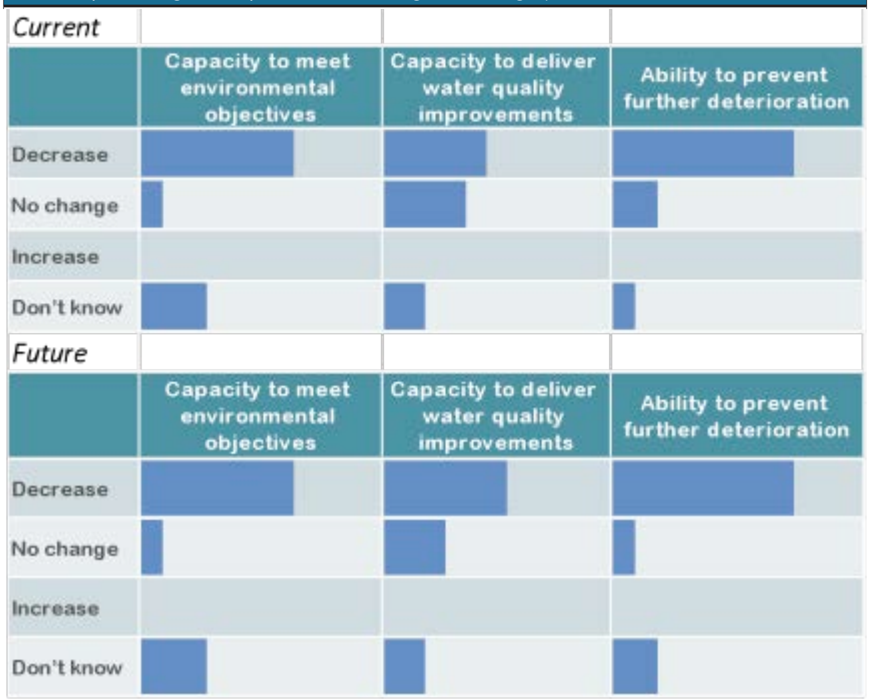
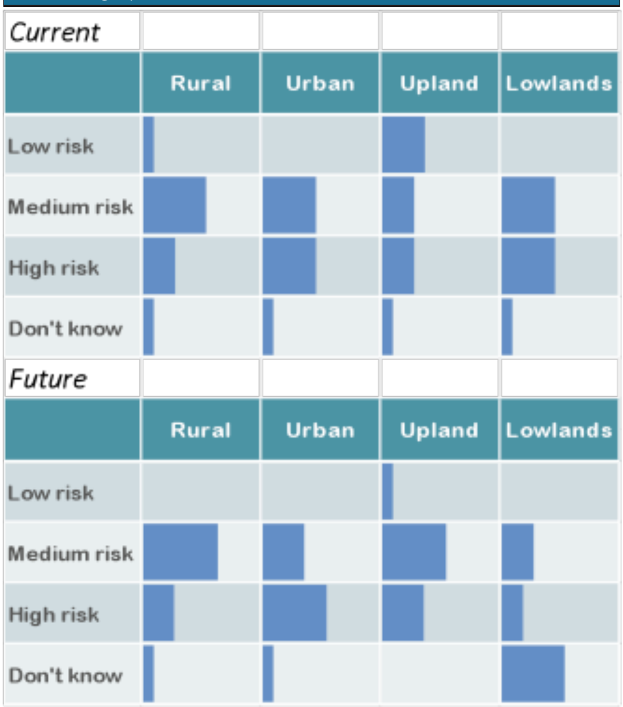
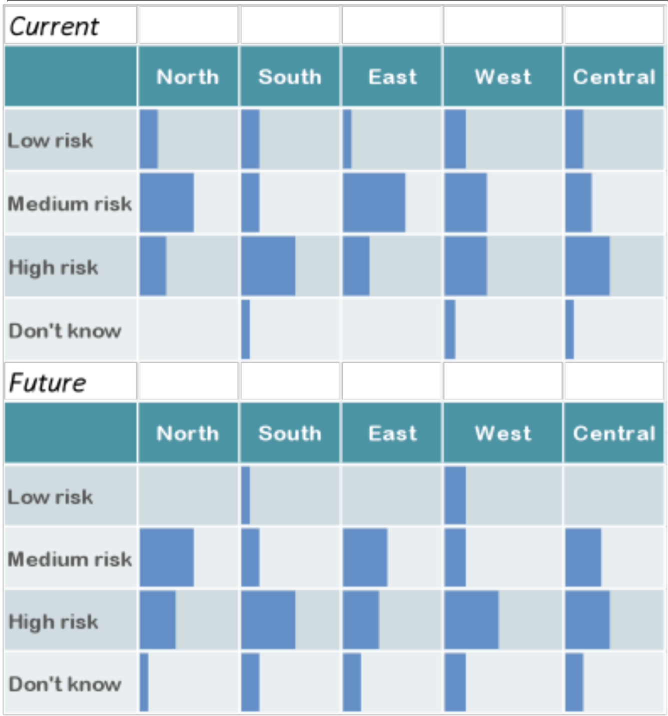


Table A4.4 Expert perceptions of the level of risk of standing waters in different areas of Scotland developing water quality issues due to climate change so far (Current) and in the future (Future). *Blue bars indicate percentage of respondents selecting each category; maximum 100%.*



When asked to indicate their perceived level of risk of standing waters in different areas of Scotland developing water quality issues due to climate change, the responses suggested that rural (55%; 27%), urban (46%; 46%), upland (27%; 27%) and lowland (46%; 46%) standing waters were already at medium to high risk, respectively (Table A4.4). Only upland standing waters were considered by respondents to have a low risk of climate change impacts (36%). In contrast, when considering the future impacts of climate change, the perceived level of risk to standing waters in rural and upland areas was considered to be higher than the current situation, while risks to urban and lowland standing waters were considered to be lower.

Table A4.5 Expert perceptions of the level of risk of standing waters in different parts of Scotland developing water quality issues due to climate change so far (Current) and in the future (Future). *Blue bars indicate percentage of respondents selecting each category; maximum 100%.*



Regionally, most respondents considered standing waters in the southern, western and central parts of Scotland to be at highest risk of developing water quality issues under current climate change conditions, with those in the north and east being at medium risk (Table A4.5). The level of risk across all areas of Scotland was perceived to be greater under future climate change scenarios.

Survey participants were asked to consider the level of risk of different types of standing waters having already developed water quality problems due to climate change (Table A4.6, page 25). Only large and deep waters were considered to be at relatively low risk by some respondents; the majority considered the level of risk across all types of standing waters to be medium to high at present and into the future.

Participants were asked to indicate which, if any, relevant policies their organisations would find it difficult to comply with in the future due to the impacts on climate change on standing waters. Compliance with the EU Habitats and Water Framework Directives were considered to be the most at risk of non-compliance, with 55% and 64% of respondents, respectively, raising concerns (Table A4.7). A further 46% of respondents were concerned about compliance with Scottish government guidance on cyanobacteria, and 27% of respondents raised concerns about compliance with wastewater treatment regulations, the EU Drinking Water Directive and the Scottish Government's Climate Emergency Response statement.

Table A4.7 Concerns about climate change impacts on Scottish standing waters affecting policy compliance. *Blue bars indicate percentage of respondents selecting each category; maximum 100%.*

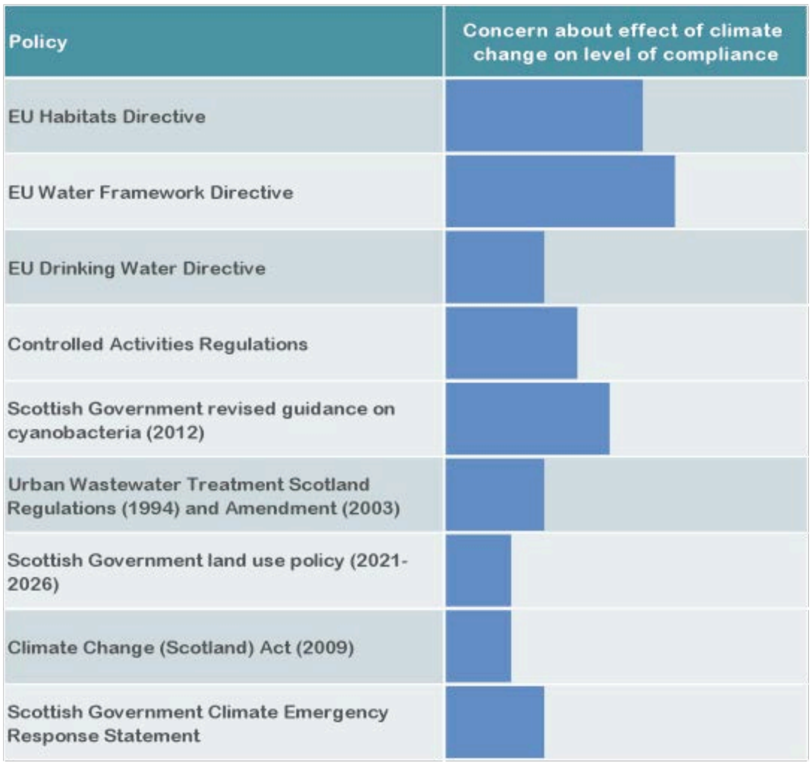


Table A4.6 Expert perceptions of the level of risk of different types of standing waters in Scotland developing water quality issues due to climate change so far (Current) and in the future (Future).
Blue bars indicate percentage of respondents selecting each category; maximum 100%.

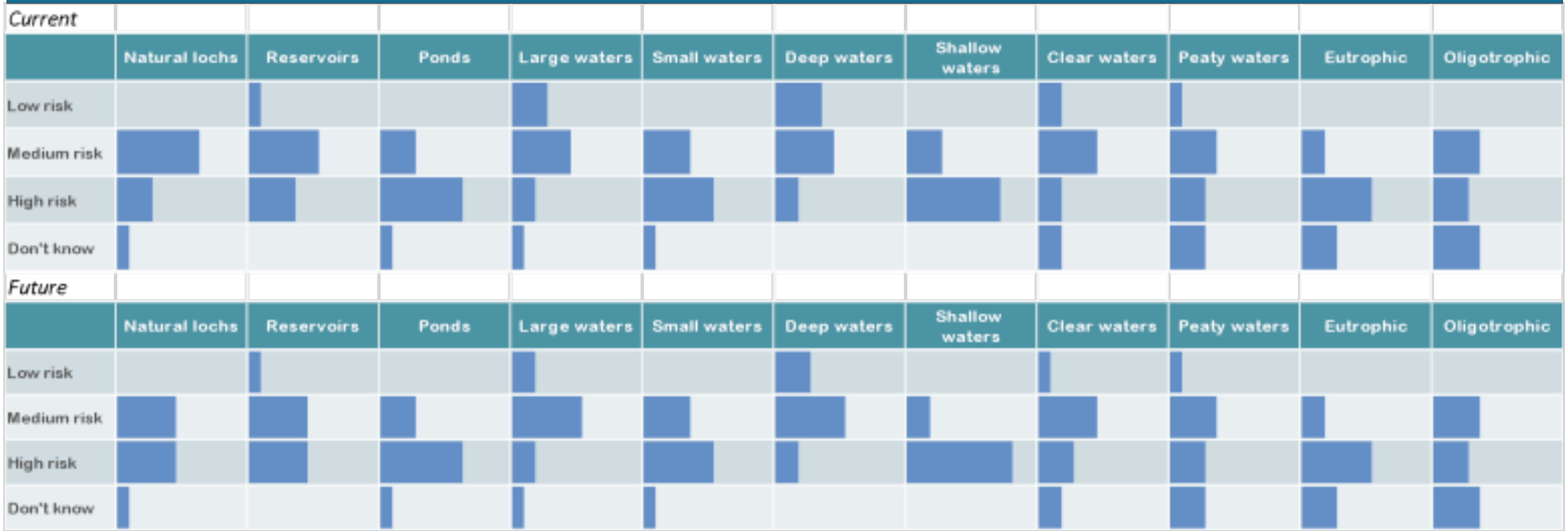


Table A4.8 Expert perceptions of environmental factors related to climate change impacts that are likely to contribute to the risk of water quality issues developing in Scottish standing waters. *Blue bars indicate percentage of respondents selecting each category; maximum 100%.*

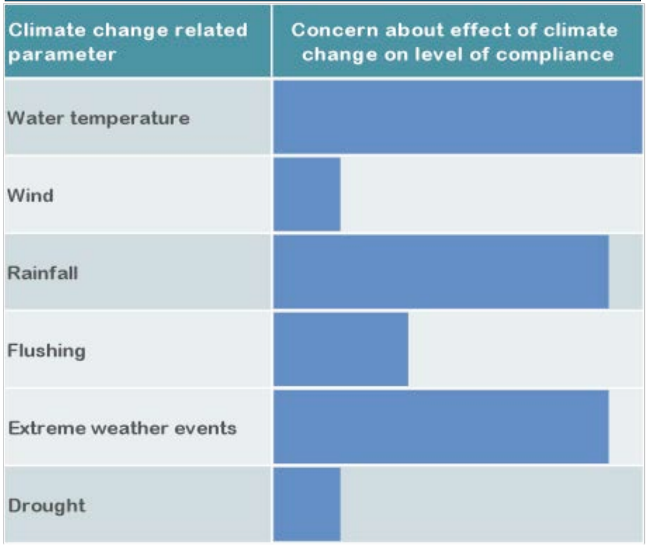


Table A4.9 Potential actions that need to be considered for mitigating climate change impacts on Scottish standing waters, or adapting to them. *Blue bars indicate percentage of respondents selecting each category; maximum 100%.*

Only 18% of respondents were concerned about compliance with the Scottish Government's land use strategy and the Climate Change (Scotland) Act 2009. When participants were asked which environmental factors relating to climate change impacts were likely to contribute to the risk of water quality issues developing in Scottish standing waters, 100% thought that changes in water temperature would cause problems and 91% indicated concerns about changes in level of rainfall and extreme events (Table A4.8). Changes in levels of flushing rates and wind were deemed to be less of a concern, having been selected as important by only 36% and 18% of respondents, respectively.

Of the three potential approaches suggested as possible adaptation or adaption options to reduce the impacts of climate change, nature-based solutions were selected by 100% of participants, with policy based solutions (91%) and coordination of planning and management options (73%) being a little less popular (Table A4.9)

3 Appendix 3: Development of climate change scenarios from CHESS-SCAPE Future Climate dataset

Background

Scottish standing waters are likely to be affected by climate change in the future. To examine this, a future climate change dataset for air temperature and hydrologically effective rainfall (or runoff) was developed for all Scottish standing waters and their catchments, for which relevant information is held within the database that underpins the UKLakes portal. These data were used to create climate change scenarios for Scottish standing waters.

Hydrologically effective precipitation for each catchment and air temperature over each standing water were generated from CHESS-SCAPE climate change data, which forms part of the CHESS (Climate, Hydrological and Ecological research Support System) suite of climate change related datasets. CHESS-SCAPE data have been derived directly from climate model output provided by UK Climate Projections 2018 (UKCP18), which have been extended by:

- downscaling to 1 km resolution based on physical and empirical relationships
- bias-correcting to the CHESS-met observation-based meteorological dataset (Robinson et al., 2017a)

Methods

The datasets used in this project were 1km resolution and at a monthly time-step. They run from December 1980 - November 2080, inclusive, and the first 30 years of the data are hindcast. To give an idea of likely future changes in temperature and precipitation, the dataset was initially mapped to show April to September ('growing season') means of air temperature and precipitation for 2000 and 2080 (Figure A3.1).

Potential Evapotranspiration data for the CHESS-SCAPE dataset were calculated following the method of Robinson et al. (2017b). For the current project, Potential Evapotranspiration with interception correction (PETI) data were used, which include an interception correction that accounts for the evaporation of water from the surface of the plant canopy after rain and better represents the processes that occur before rainwater reaches the soil. All values are estimations of the evaporative demand of the atmosphere for a short grass. Like the CHESS-SCAPE data they are based on, they run from December 1980 - November 2080 inclusive and are 1km resolution.

Representative Concentration Pathways

To model and project future climate it is necessary to make assumptions about the economic, social and physical changes to our environment that will influence climate change. Representative Concentration Pathways (RCPs) are a method of capturing those assumptions within a set of future change scenarios (Met Office, 2018; van Vuuren, et al. 2011).

There are 4 RCPs available within the UKCP18/CHESS-SCAPE datasets. These are RCP2.6, RCP4.5, RCP6.0 and RCP8.5, where the number represents the radiative forcing targets for 2100 in watts per square metre (W m^{-2}), e.g., in RCP2.6, this has been set at 2.6 W m^{-2}. Each pathway results in a different range of global mean temperature increases over the 21st century. The increase in global mean surface temperatures (averaged over 2081-2100), which can be estimated for each RCP pathway, give an good indication of the overall level of warming that is likely to occur under each scenario: 1.6°C for RCP2.6; 2.4°C for RCP4.5; 2.8°C for RCP6.0; 4.3°C for RCP 8.5 (IPCC, 2018). In contrast to the Special Report on Emission Scenarios (SRES) data used in the UKCP09 projections, the UKCP18 predictions include climate change adaptation options.

RCP Selection

Although data processing was performed across all RCPs, modelling was undertaken using a single and consistent 'scenario' (RCP) that provides a simpler message and is rooted in wider policy and current likelihoods. RCPs 4.5 and 6.0 end up with quite similar levels warming (~3°C by 2080), but RCP4.5 has higher emissions than RCP6.0 earlier in the century and stronger adaptation later so the pathway towards reaching this 3°C rise in air temperature by 2080 differs between the two scenarios. Any differences between RCP4.5 and RCP6.0 are better resolved by 2100, after the CHESS-SCAPE dataset ends. RCP2.6 is probably not very useful on its own because, despite recent agreements at the 26th UN Climate Change Conference of the Parties (COP26), it probably still predicts an unlikely future.

There are some arguments that RCP8.5 is too extreme (Hausfather & Peters, 2020) and that it should not be used on its own as a default scenario, although it can a useful tool for quantifying risk over near to mid-term time horizons (Schwalm et al., 2020). However, the advantage of RCP8.5 is that it a) provides a worst case scenario, and b) passes through other temperature levels on the way, so it can provide an intermediate picture of the latter stages of other RCPs, depending on the metrics used.

Either RCP4.5 or RCP6.0 probably offer the most realistic scenarios and Hausfather & Peters (2020) showed that RCP6.0 might be considered the more likely of the two. For these reasons, RCP6.0 was chosen as the main warming scenario for use in this project.

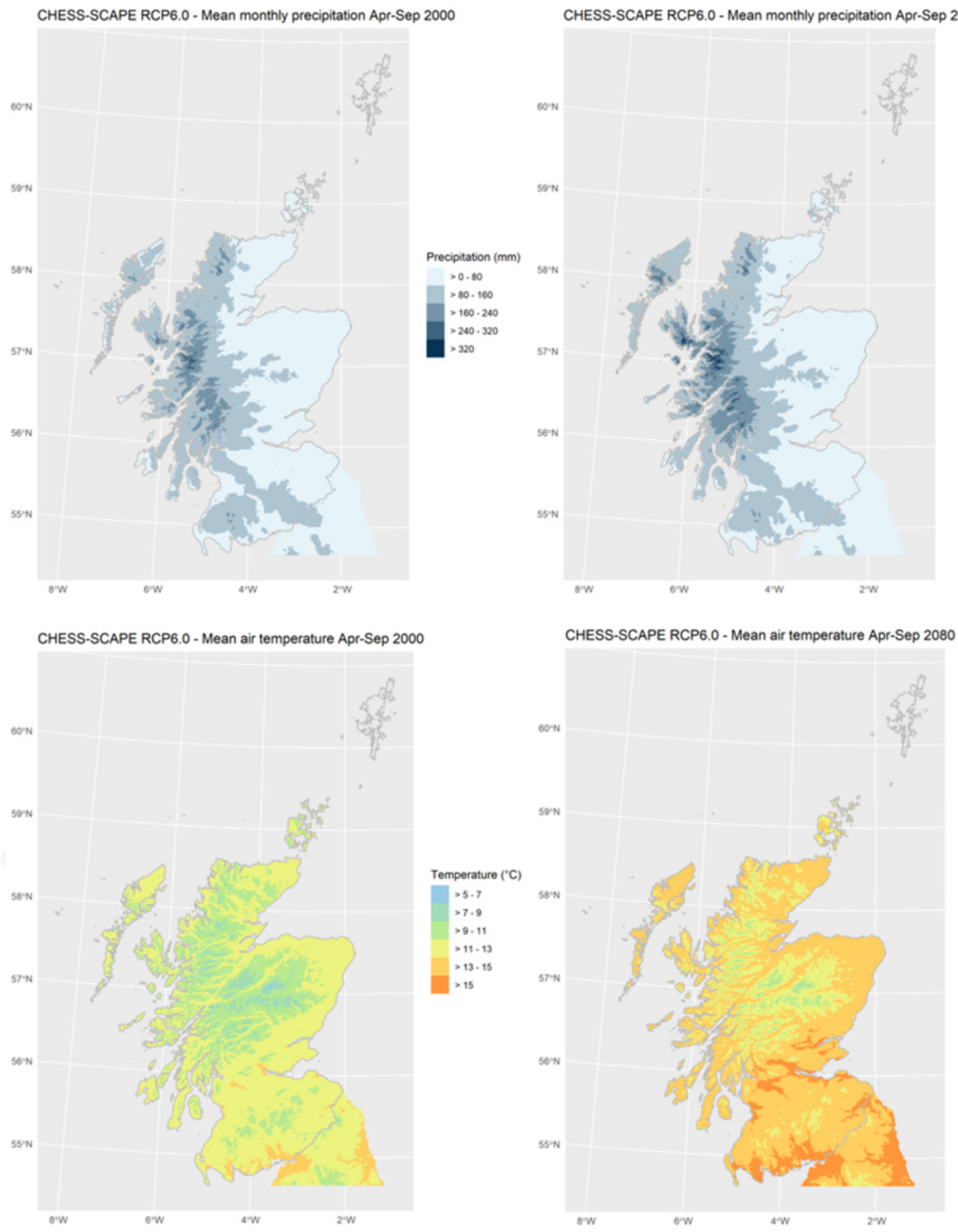


Figure A3.1. Comparison of average precipitation levels (mm) and air temperatures (°C) across Scotland for April to September in 2000 (left) and 2080 (right), as predicted for UKCIP18 climate change scenario RCP6.0, ensemble 1.0

Ensembles

Each RCP includes four ensemble members (01, 02, 03, 04). Ensemble member 01 is the default parameterisation of the Hadley Centre Climate model and closely matches the scenario used for the UK's entry to Coupled Model Intercomparison Project 6 (CMIP6). The rest of the ensemble is a 'perturbed parameter ensemble', where a few parameters within the same climate models have been adjusted within realistic bounds. Ensemble member 01 sits close to the ensemble mean and, as such, is reasonably representative of the whole ensemble. For this reason, ensemble 01 was used across all RCPs in the current project.

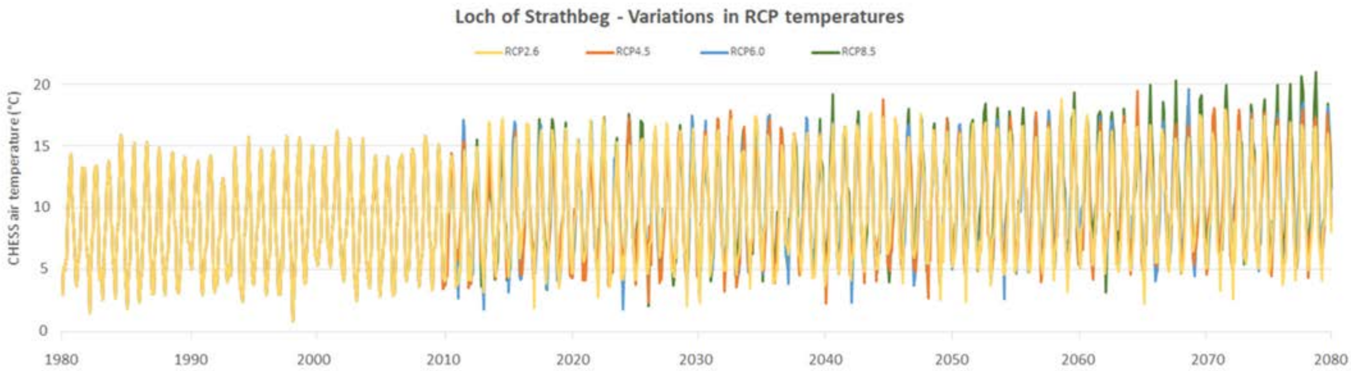


Figure A3.2. Variation in modelled average air temperatures for Loch of Strathbeg, between 1981 and 2080 across different RCPs illustrating the frequency of data that are available for all Scottish lochs and reservoirs. Data 1980 to 2009 are more consistent across RCPs because they are hindcast. Data for 2010-2080 vary across RCPs because they are projections of different emissions scenarios. RCP6.0, considered to be the most likely future scenario, was used for climate change projections in this study. © UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.

Temperature data extraction

Monthly (air) temperature data were extracted for the centre points (centroids) of all Scottish standing waters for 1981-2080. It should be noted, however, that two lakes had their centroids removed manually to prevent areas with missing data being selected.

The air temperature data generated by this process for each standing water were graphed for all of the four RCPs for each waterbody, allowing the data to be checked visually before any further analysis. The data showed a consistent pattern of warming in line with what would be expected from each RCP before 2010, when the data have been matched to measured values, and variation across the RCPs from 2010 onwards, when the data have not been matched to measured values (Figure A3.2).

A further check was performed to see how seasonal values compared across the 100 year period. Again, the results were as expected, building further confidence in the dataset and the processing technique used (Figure A3.3).

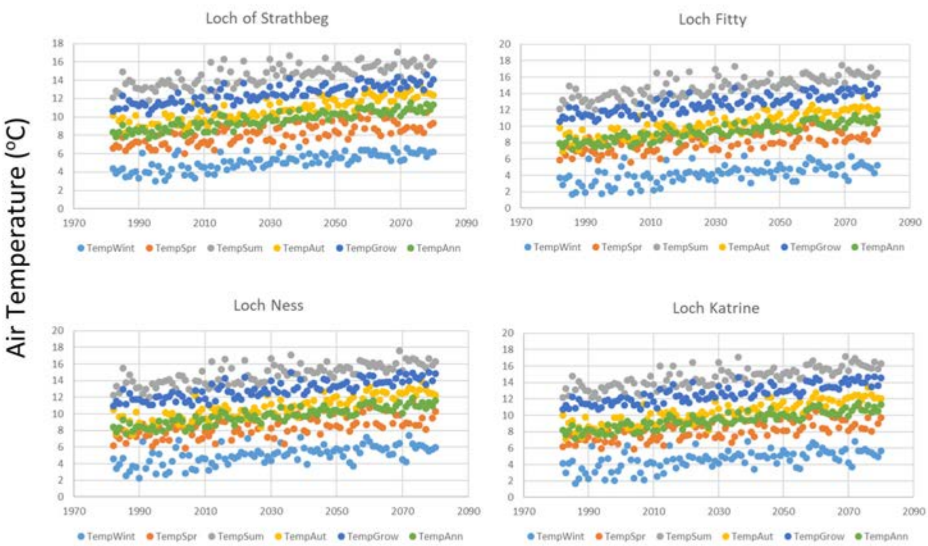


Figure A3.3. Examples of long-term UKCP18 RCP6.0 projections for seasonal air temperature (Loch of Strathbeg, Loch Katrine, Loch Ness, Loch Katrine), 1981 to 2080. TempWin – average of winter months (Dec - Feb); TempSpr - spring (Mar - May); TempSum - summer (Jun - Aug); TempAut - autumn (Sep - Nov); TempAnn - annual (all months); TempGrow - growing season (May - Sep).

Effective Rainfall Calculation

Monthly data for precipitation *(PR)* and potential evapotranspiration *(PETI)* were used to calculate hydrologically effective rainfall *(EffRain)* for the whole dataset, as follows:

EffRain = PR - PETI

This was then extracted as mean values for each catchment. To calculate hydrologically effective rainfall, potential evapotranspiration data (PETI) was first converted into a measure of 'actual evapotranspiration'. Use of a simplified method (not using a water budget or soil moisture deficit) was necessary due to availability of monthly data, only. However, this approximation, especially in a relatively wet country such as Scotland, was considered to be appropriate for the purpose.

Level of evapotranspiration varies with land cover type and, for this analysis, 1km resolution LCM 2015 data were used to provide this information. Each land cover class was assigned a 'crop coefficient' (Kc), with seasonal values taken from, or approximated from, three main sources of information (FAO, 1998; Nistor et al., 2015; Corbari et al., 2017) using an approach adapted from Richardson et al., 2018. The seasonal Kc values were then converted into an annual mean value (Table A1.1). The 21 LCM 2015 land cover classes were then converted into 1km resolution aggregate classes and the Kc values applied to the PETI data using the equation:

*AET = PETI * Kc*

where AET = Actual evapotranspiration.

Table A1.1. Seasonal and annual Kc values for each land cover class (Kclc) showing the source of each value (either Nistor et al., 2015 or FAO 1998)

LCM 2015 Class ID	LCM 2015 Class	Kclc (ini season)	Kclc (mid season)	Kclc (end season)	Kclc (cold season)	Kclc (mean all seasons)	Source
1	Broadleaved woodland	1.3	1.6	1.5	0.6	1.25	Nistor
2	Coniferous Woodland	1	1	1	1	1.00	Nistor
3	Arable and Horticulture	0.7	1.15	0.325		0.73	FAO
4	Improved Grassland	0.3	0.75	0.75		0.60	FAO
5	Neutral Grassland	0.9	0.95	0.95		0.93	FAO
6	Calcareous Grassland	0.9	0.95	0.95		0.93	FAO
7	Acid grassland	0.9	0.95	0.95		0.93	FAO
8	Fen, Marsh and Swamp	0.15	0.45	0.8		0.47	Nistor
9	Heather	0.9	0.95	0.95		0.93	FAO
10	Heather grassland	0.9	0.95	0.95		0.93	FAO
11	Bog	0.15	0.15	0.15		0.15	FAO
12	Inland Rock	0.15	0.2	0.05		0.13	Nistor
13	Saltwater	0.3	0.7	1.3		0.77	Nistor
14	Freshwater	0.25	0.65	1.25		0.72	Nistor
15	Supra-littoral Rock	0.15	0.2	0.05		0.13	Nistor
16	Supra-littoral Sediment	0.15	0.15	0.15		0.15	FAO
17	Littoral Rock	0.15	0.2	0.05		0.13	Nistor
18	Littoral sediment	0.15	0.15	0.15		0.15	FAO
19	Saltmarsh	0.1	0.45	0.8		0.45	Nistor
20	Urban	0.2	0.4	0.25		0.28	Nistor
21	Suburban	0.1	0.3	0.2		0.20	Nistor

Retention Times

Waterbody retention times were calculated for all standing waters as follows, where *EffRain* is the average effective rainfall across the catchment (m), *VolLake* is waterbody volume (m^3) and *CatchArea* is the area of the catchment (m^2):

*Retention time = VolLake (m^3) / CatchArea (m^2) * EffRain (m)*

For every lake, this gave a predicted retention time for each month from 1981 to 2080. The data were validated by comparison with the results obtained using flow data from the National River Flow Archive (NRFA) collected at sites situated on the outflow of a lake, before any tributaries join the river. Thirty-eight suitable locations were identified and flow data were gathered and averaged over the 10-period, 2004-13.

Retention times over the same period were calculated using the CHESS-SCAPE data, giving a comparable mean value for 2004-13. For each RCP, these two sets of values were compared and a linear relationships were established. A correction factor was then applied to the retention times estimated for each RCP to create validated monthly retention time data for each waterbody.

All relationships between measured and modelled retention times, across all RCPs, had an R^2 of >0.9 and slopes >0.96, with the intercept set at 0, showing a good linear relationship that could be used to adjust the climate change data against known values. The lines were also close to 1:1, showing a very good fit between modelled and measured data. The graph for RCP2.6 is shown as an example in Figure A3.4.

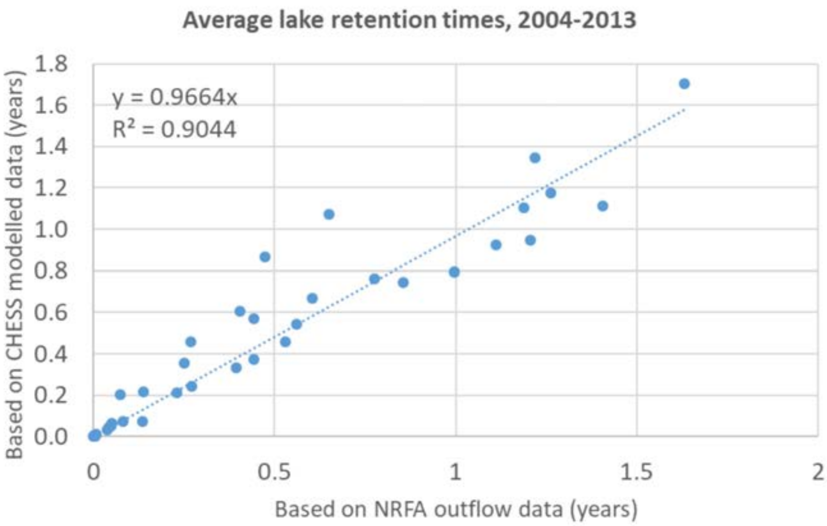


Figure A3.4. Relationship between lake retention times estimated from NRFA flow data and those derived from CHESS rainfall data.
© UKCEH. Contains data from the UK National River Flow Archive. All rights reserved.

Relationship between lake and air temperatures

To derive a relationship between modelled air temperature derived from the CHESS-SCAPE data and lake temperatures measured by SEPA, the two sets of data were compared for two 5-year periods, 2010-14 and 2015-19. These periods were split because the latter included some notable loch and reservoir warming events. The relationships between these variables were strong (R^2 >0.7) over both time periods (Figure A3.5; Figures A3.6), indicating the suitability of this approach for predicting loch and reservoir surface temperatures from the future air temperature data contained within the CHESS-SCAPE data.

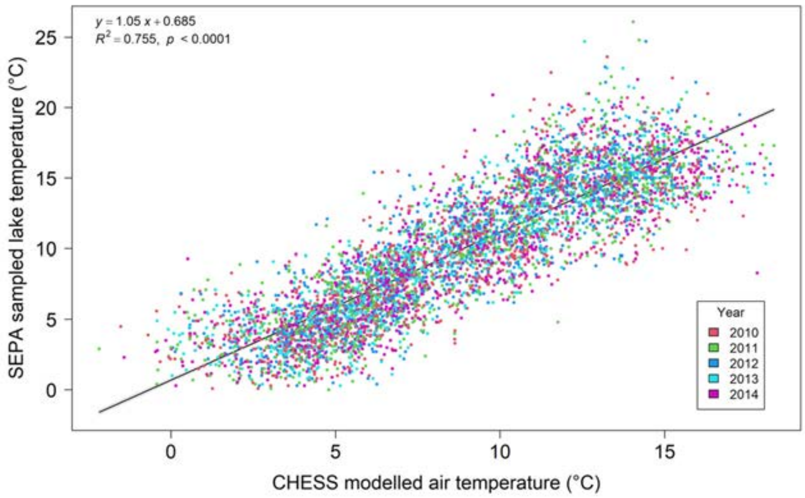


Figure A3.5. Comparison of air temperatures derived from CHESS-SCAPE data for 2010-2014 and loch and reservoir temperatures measured by SEPA over the same period. *© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.*

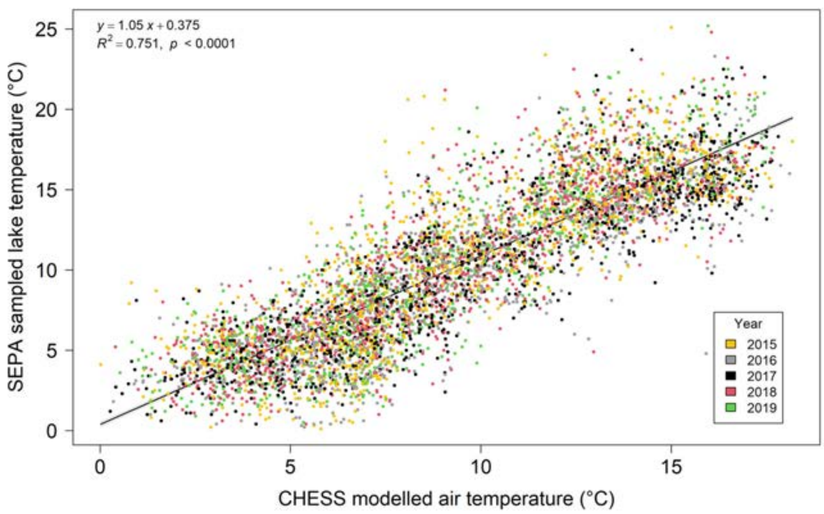


Figure A3.6. Comparison of air temperatures derived from CHESS-SCAPE data for 2015-2019 and loch and reservoir temperatures measured by SEPA over the same period. *© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.*

References

Corbari, C., Ravazzani, G., Galvagno, M., Cremonese, E. and Mancini, M. (2017). Assessing Crop Coefficients for Natural Vegetated Areas Using Satellite Data and Eddy Covariance Stations. Sensors, 17, 2664. DOI: 10.3390/s17112664

FAO (1998). *Crop evapotranspiration - Guidelines for computing crop water requirements*, Chapter 6. https://www.fao.org/3/X0490E/X0490E00.htm

Hausfather, Z. and Peters, G. P. (2020). Emissions – the 'business as usual' story is misleading. Nature, 577, 618-620.

IPCC (2013). *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.* Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 1535 pp.

IPCC (2018) Global Warming of 1.5°C.An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty. Masson-Delmotte, V., P. Zhai, H.-O. Pörtner, D. Roberts, J. Skea, P.R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J.B.R. Matthews, Y. Chen, X. Zhou, M.I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, and T. Waterfield (eds.).

Met Office (2018). UKCP18 Guidance: Representative Concentration Pathways. https://www.metoffice.gov.uk/binaries/content/assets/metofficegovuk/pdf/research/ukcp/ukcp18-guidance---representative-concentration-pathways.pdf

Nistor, M. and Porumb-Ghiurco, C. (2015). How to compute the land cover evapotranspiration at regional scale? A spatial approach of Emilia-Romagna region. Scientific Annals of Stefan cel Mare University of Suceava Geography Series 25(1). DOI:10.4316/GEOREVIEW.2015.25.1.268

Richardson, J., Miller, C., Maberly, S. C., Taylor, P., Globevnik, L., Hunter, P., Jeppesen, E., Mischke, U., Moe, J., Pasztaleniec, A., Søndergaard, M. and Carvalho, L. (2018). Effects of multiple stressors on cyanobacteria abundance vary with lake type. *Glob Change Biol*., 24, 5044– 5055. DOI:10.1111/gcb.14396

Robinson, E. L., Blyth, E. M., Clark, D. B., Comyn-Platt, E., Finch, J., and Rudd, A. C. (2017a). *Climate hydrology and ecology research support system meteorology dataset for Great Britain (1961-2015) [CHESS-met] v1.2*. NERC Environmental Information Data Centre. DOI:10.5285/b745e7b1-626c-4ccc-ac27-56582e77b900

Robinson, E. L., Blyth, E. M., Clark, D. B., Finch, J., and Rudd, A. C. (2017b). Trends in atmospheric evaporative demand in Great Britain using high-resolution meteorological data, *Hydrol*. Earth Syst. Sci., 21, 1189–1224, DOI:10.5194/hess-21-1189-2017

Schwalm, C. R., Glendon, S., Duffy. P. B. (2020). RCP8.5 tracks cumulative CO_2 emissions. Proceedings of the National Academy of Sciences Aug 2020, 117 (33) 19656-19657; DOI: 10.1073/pnas.2007117117

van Vuuren, D. P., Edmonds, J., Kainuma, M. et al. (2011). The representative concentration pathways: an overview. *Climatic Change* 109, 5. DOI:10.1007/s10584-011-0148-

4 Appendix 4: Exploration of SEPA monitoring data

Monitoring data for Scottish standing waters were provided by the Scottish Environment Protection Agency (SEPA). The SEPA data comprised loch and reservoir monitoring data collected between 1989 and 2019, although the earlier years had many missing values which restricted their use in some of the analyses. Although the data comprised records of physical and chemical determinands from 175 lochs and reservoirs, amounting to 629,303 values, only data for total phosphorus (TP), total oxidisable nitrogen (TON), silicate, chlorophyll-*a*, pH, oxygen saturation and surface temperature were included in this study (142 sites; Table A4.1). It should be noted that the measured data were for surface temperatures, only; no depth profile data were included within the SEPA dataset.

Figure A4.2 shows the key determinands within the SEPA data records that demonstrate a positive or negative correlation with water temperature, using three of the 142 lochs examined as examples. The first is silicate, which is an essential nutrient for diatoms and a small number of other algae. The inverse correlation between silica concentrations and water temperature in the three examples shown suggests that diatoms will not be able to sustain viable populations at temperatures above 15°C. Taking 1 mg Si L^{-1} as being the level at which lack of silica starts to limit diatom growth, the SEPA data suggest that about 67% of monitored lochs are silica limited in summer. With climate change, the level and temporal extent of Si limitation. Dissolved nitrogen (TON) availability also becomes severely reduced in many lochs at temperatures above about 15°C; the data suggest that, if a TON:RP (reactive phosphorus) ratio of <10 is taken to indicate N limitation (Maberly et al., 2020), about 90% of lochs are N-limited when the temperature rises above 15°C. With higher temperatures associated with climate change, the period over which N-limitation occurs is likely to lengthen. This will tend to limit the growth of all algae apart from some species of cyanobacteria that can fix nitrogen from the atmosphere. Increases in both N and Si limitation as temperatures rise would be expected to encourage cyanobacteria to outcompete other algal species as long as there is sufficient P available to support growth. These graphs also show that dissolved oxygen concentrations (expressed as percent saturation) and pH increase with increasing water temperature. It is likely that all of the situations outlined above would become more common over longer periods under climate change.

Surface water temperature data with a limit of resolution of at least 0.5°C were available for 142 (Table A4.1) lochs and reservoirs, but there were many gaps in the data especially during the winter months and in July and/or August. For the loch and reservoir temperature mapping, data were selected where there were at least four monthly data values between April and September, with no two missing values in consecutive months. The data were then summarised as April to September average water temperatures for each year between 2005 and 2019; any April to September average water temperature based on less than five monthly values was excluded from the analyses.

The April to September water temperature values were averaged over the four 5 year periods 2005-2009, 2010-2014 and 2015-2019 and the 2010-2019 and the derived values were plotted against similarly summarised climate change data for RCP 6.0 air temperatures. A relationship was derived between the air temperature values derived from CHESS-SCAPE and the water temperature values from the monitoring data, which was then used to predict loch and reservoir water temperature (*LochTemp;* °C) from air temperature (*AirTemp*; °C) into the future (Figure A4.1). This equation was:

LochTemp = 1.2033 x Air temp – 0.7697; $R^2 = 0.3874$

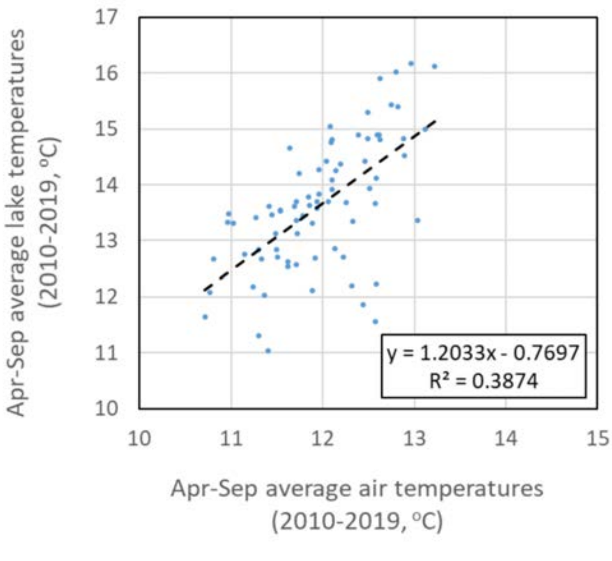


Figure A4.1 Relationship between RCP 6.0 modelled air temperatures and measured loch and reservoir water temperatures, averaged over April-September 2010-2019. © UKCEH. Contains SEPA data © *Scottish Environment Protection Agency and database right 2021. All rights reserved.*

Table 1 List of lochs and reservoirs for which SEPA monitoring data were available; those designated as Heavily Modified Waterbodies (⁺HMWBs) for water supply highlighted in blue, hydropower in green, navigation in grey, flood protection in pink, biodiversity purpose in amber. ⁺HMWBs where the impoundment/abstraction is a significant consideration for this specific project. There are a few other examples that have a water supply abstraction, but they are not HMWBs, and their hydrology status is high, so impacts are minimal (I. Milne, SEPA, *pers. comm.*).

SEPA WBID	Name	SEPA WBID	Loch name	SEPA WBID	Name
100342	Butterstone Loch	100278	Loch Fitty	100003	Loch of Girlsta
100322	Castle Loch	100242	Loch Freuchie	100008	Loch of Harray
100294	Castle Semple Loch	100229	Loch Frisa	100012	Loch of Kirbister
100295	Cobbinshaw Reservoir	100303	Loch Garasdale	100225	Loch of Lintrathen
100313	Daer Reservoir	100190	Loch Garry	100235	Loch of Lowes
100276	Gartmorn Dam	100134	Loch Garve	100185	Loch of Skene
100287	Hillend Reservoir	100113	Loch Glascarnoch	100004	Loch of Spiggie
100300	Kilbirnie Loch	100275	Loch Glashan	100136	Loch of Strathbeg
100271	Lake of Menteith	100288	Loch Gorm	100005	Loch of Swannay
100112	Loch a' Bhraoin	100331	Loch Grannoch	100027	Loch of Toftingall
100238	Loch a' Phuill	100034	Loch Hempriggs	100141	Loch Olabhat
100097	Loch Achall	100019	Loch Hope	100162	Loch Olaidh Meadhanach
100140	Loch Achilty	100187	Loch Insh	100274	Loch Ore
100264	Loch Achray	100261	Loch Katrine	100084	Loch Osgaig
100199	Loch an t-Seilich	100326	Loch Ken	100167	Loch Ruthven
100270	Loch Ard	100334	Loch Kinder	100121	Loch Scadabhagh
100227	Loch Arienas	100192	Loch Kinord	100020	Loch Scarmclate
100197	Loch Arkaig	100198	Loch Laggan	100144	Loch Sgamhain
100259	Loch Avich	100223	Loch Laidon	100078	Loch Sgiobacleit
100585	Loch Awe	100099	Loch Langabhat	100208	Loch Shiel
100241	Loch Ba	100145	Loch Leathan	100065	Loch Shin
100228	Loch Bà	100209	Loch Lee	100260	Loch Sloy
100118	Loch Bad an Sgalaig	100269	Loch Leven	100039	Loch Stack
100042	Loch Badanloch	100194	Loch Lochy	100074	Loch Stranndabhat
100168	Loch Beinn a' Mheadhoin	100257	Loch Lomond	100180	Loch Tarff
20553	Loch Borralan	100029	Loch Loyal	100233	Loch Tay
100316	Loch Bradan	100258	Loch Lubnaig	100284	Loch Thom
100092	Loch Brora	100131	Loch Luichart	100110	Loch Tollaidh
100017	Loch Calder	100091	Loch Lurgainn	100256	Loch Tralaig
100133	Loch Carabhat	100327	Loch Maberry	100033	Loch Urghag
100166	Loch Chill Donnain Uarach	100109	Loch Maree	100139	Loch Ussie
100265	Loch Chon	100018	Loch Meadie	100108	Loch Vaich
100178	Loch Cluanie	100048	Loch Merkland	100266	Loch Venachar
100143	Loch Damh	100100	Loch Migdale	100079	Loch Veyatie
100324	Loch Dee	100030	Loch Mòr Bharabhais	100022	Loch Watten
100215	Loch Doilet	100035	Loch More	100157	Loch Lochindorb
100314	Loch Doon	100182	Loch Morlich	100329	Lochrutton Loch
100155	Loch Druidibeag	100160	Loch Moy	100330	Milton Loch
100268	Loch Drunkie	100202	Loch Muick	100338	Mochrum Loch
100176	Loch Dùn na Cille	100125	Loch nan Eun	100332	Penwhirn Reservoir
100161	Loch Duntelchaig	100070	Loch nan Ritheanan	100226	Rescobie Loch
100251	Loch Earn	100043	Loch Naver	100290	Roughrigg Reservoir
100272	Loch Eck	100246	Loch Nell	100307	St Mary's Loch
100224	Loch Eigheach	100156	Loch Ness	100296	Strathclyde Loch
100206	Loch Eilt	100328	Loch Ochiltree	100336	White Loch
100107	Loch Eye	100007	Loch of Boardhouse	100333	Woodhall Loch
100292	Loch Fad	100000	Loch of Cliff		
100119	Loch Fada	100236	Loch of Clunie		

To estimate the rate at which lochs and reservoirs have been warming over the period 2015-2019, a linear correlation was applied to the annual average April to September values for each loch or reservoir over this period. As above, lochs and reservoirs were selected only if they had at least four monthly data values between April and September in any given year, with no two missing values in consecutive months, and annual mean April to September values for at least four of the five years. Slope of the line was used to estimate the rate of warming (°C per year) of each loch, and values were considered useable only if the associated regression line had a $R^2 > 0.3$.

Relationships between cyanobacterial biomass, total phosphorus concentrations and water temperature were explored for different loch and reservoir types (Table A4.2). Analyses were based on SEPA monitoring data from 62 lochs sampled between June and October, and from 2009 to 2012 (237 data points). Most of these data (229 data points) were collected between July and September. The data were explored visually, using bubble plots.

Analysis of the relationships between cyanobacterial biovolume (biomass), total phosphorus concentrations and water temperature for different loch and reservoir types were based on SEPA monitoring data from 62 lochs that were sampled from June to October 2009 - 2012 (237 data points). Almost all these data (229 data points) were collected between July and September. In general, the highest cyanobacterial biovolumes occurred at higher TP concentrations (>20 µg L^{-1}, noting that Log_e (20) = 3) and at temperatures above 14°C (Figure A4.3).

Examining these data by loch and reservoir type (Table A4.2), the following patterns emerge:

1. high concentrations of cyanobacteria were most common in shallow and very shallow lochs
2. no high concentrations of cyanobacteria were recorded in deep lochs or very humic lochs
3. humic lochs tended to have more high biomass events than clear lochs
4. high concentrations of cyanobacteria were observed across all alkalinity types, and were especially common in medium alkalinity lochs

It should be noted, however, that that many deep lochs are still prone to surface blooms, as has been observed in the Bloomin' Algae records. This is due to the typically large area of water over which buoyant algae cells accumulate down wind.

Table A4.2 Loch and reservoir depth, alkalinity and water colour types

Depth type	**Mean depth (m)**
Very shallow (VS)	< 3m
Shallow (S)	3 – 15
Deep (D)	> 15
Alkalinity type	**Value (µEq L^{-1})**
High (HA)	> 1000
Moderate (MA)	200 - 1000
Low (LA)	< 200
Marl (M)	Limestone catchment
Water colour type	**Value (mg L^{-1} Pt)**
Clear (C)	< 30
Humic (H)	≥ 30
Polyhumic (PH)	≥ 30
Unknown (U)	N/A

References

Maberly, S. C., Pitt, J-A, Davies, P. S. and Carvalho, L.R. (2020) Nitrogen and phosphorus limitation and the management of small productive lakes. Inland Waters 10, 159-172. https://doi.org/10.1080/20442041.2020.1714384

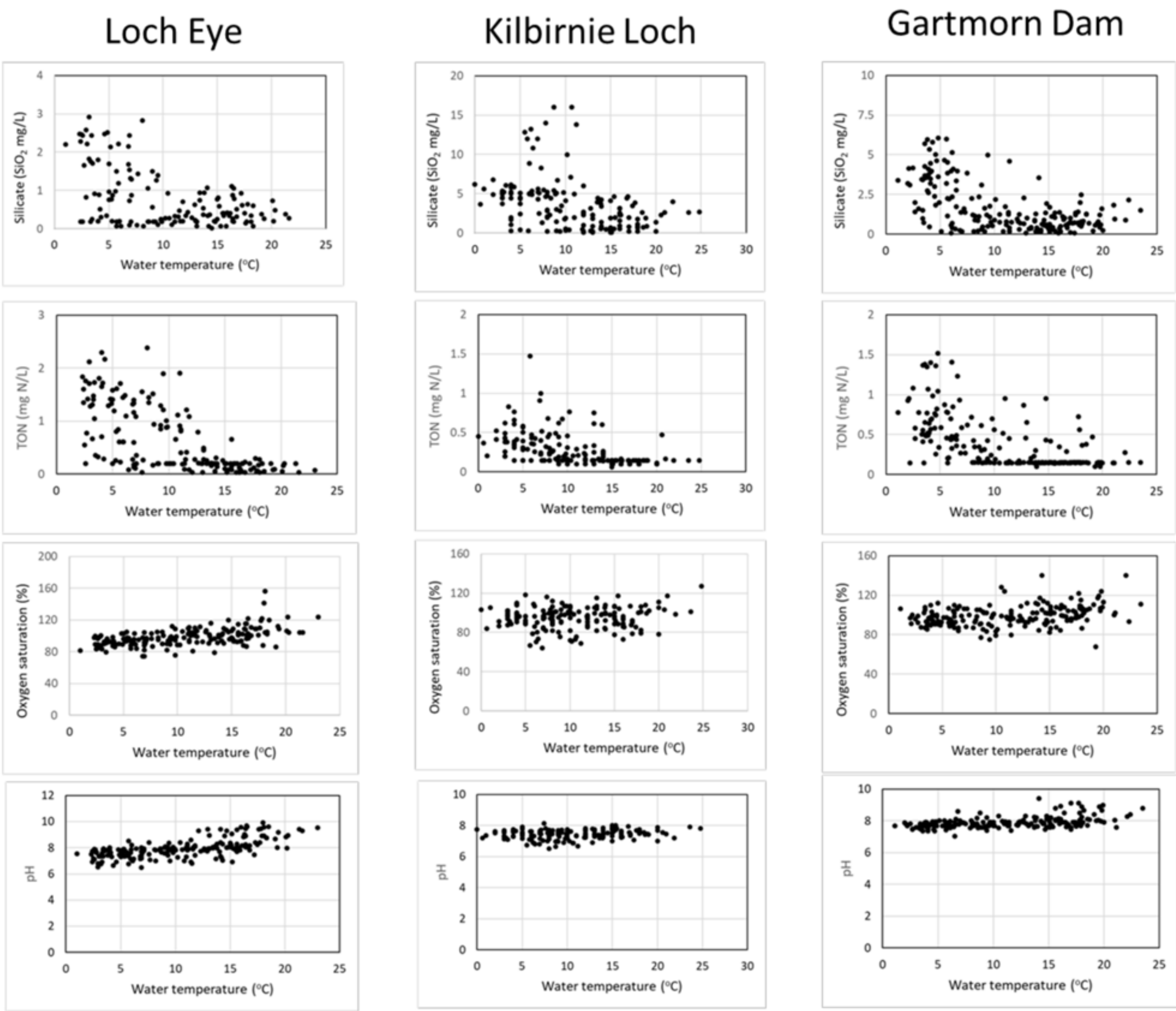


Figure A4.2 Examples of chemical water quality parameters that show a potential relationship with water temperature.
© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.

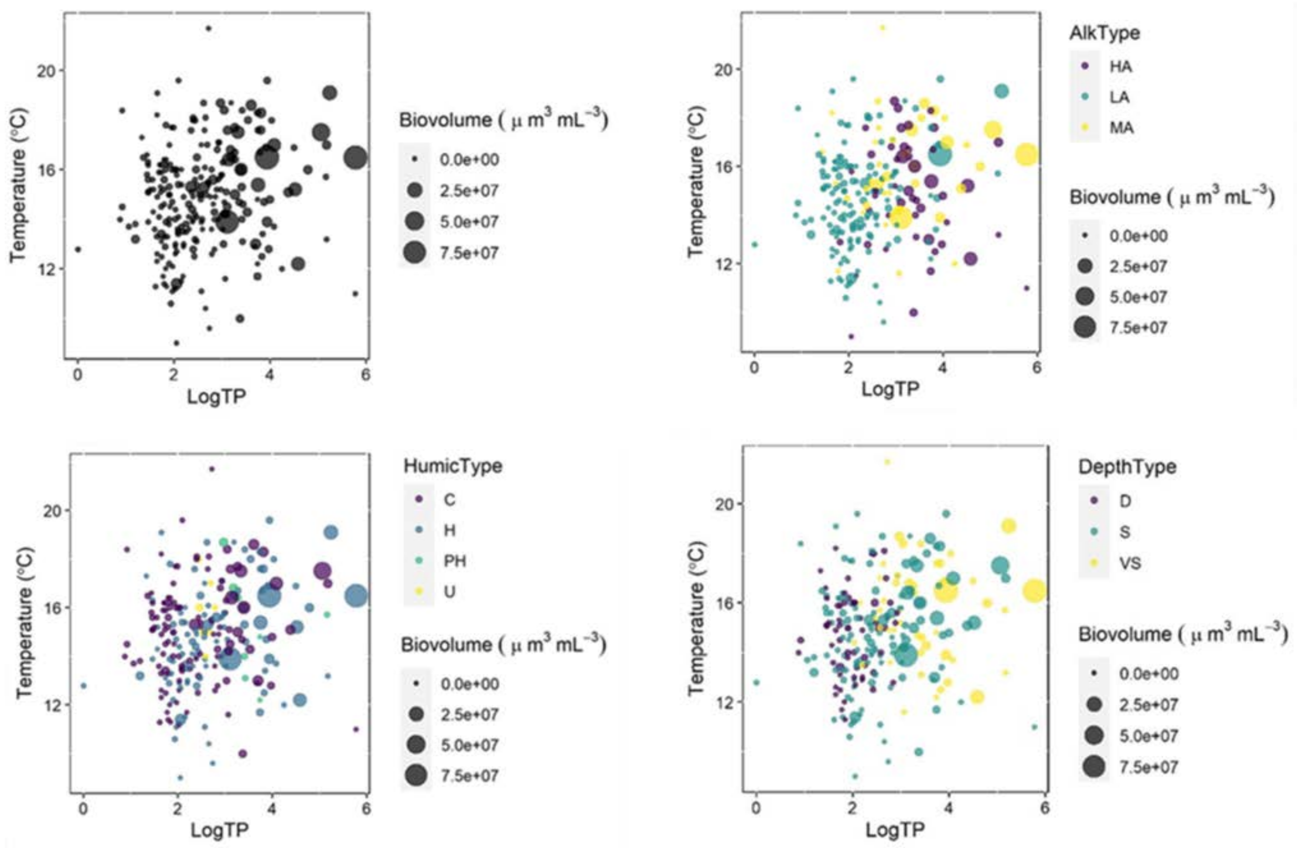


Figure A4.3 Relationship between cyanobacterial biovolume, log_e total phosphorus (LogTP) concentrations and water temperature in all Scottish lochs with sufficient data (top left) and in lochs with different typologies. Alkalinity type [AlkType) – top right; Humic type – bottom left; Depth type – bottom right). Abbreviations are: HA = High Alkalinity; LA = Low Alkalinity; MA = Medium Alkalinity; C = clear; H = Humic; PH = Polyhumic; Humic; U = Unknown; D = Deep; S = Shallow; VS = Very Shallow.
© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.

5 Appendix 5: Relationship between chlorophyll-*a* concentrations and environmental factors that are likely to be affected by climate

Universal model

SEPA lochs monitoring data with information on all of the variables required for these analyses were available for 133 lochs. Changes in monthly chlorophyll-*a* concentrations (summer months, July-September) in response to TP, water temperature and retention time were modelled.

Chlorophyll-*a* and TP concentrations, and estimated retention time, were 'transformed' by taking the natural logarithm of the original values ($\log_e$) and 'standardised' by subtracting the mean and dividing the result by the standard deviation. Water temperature was standardised, only. All retention time values of >100 years were capped at 100 years. Transformation and standardisation greatly reduced skew in the variables prior to modelling, and ensured that model residuals were approximately normally distributed and showed constant variance. Histograms showing the distribution of the data after transformation and standardisation are shown in Figure A5.1.

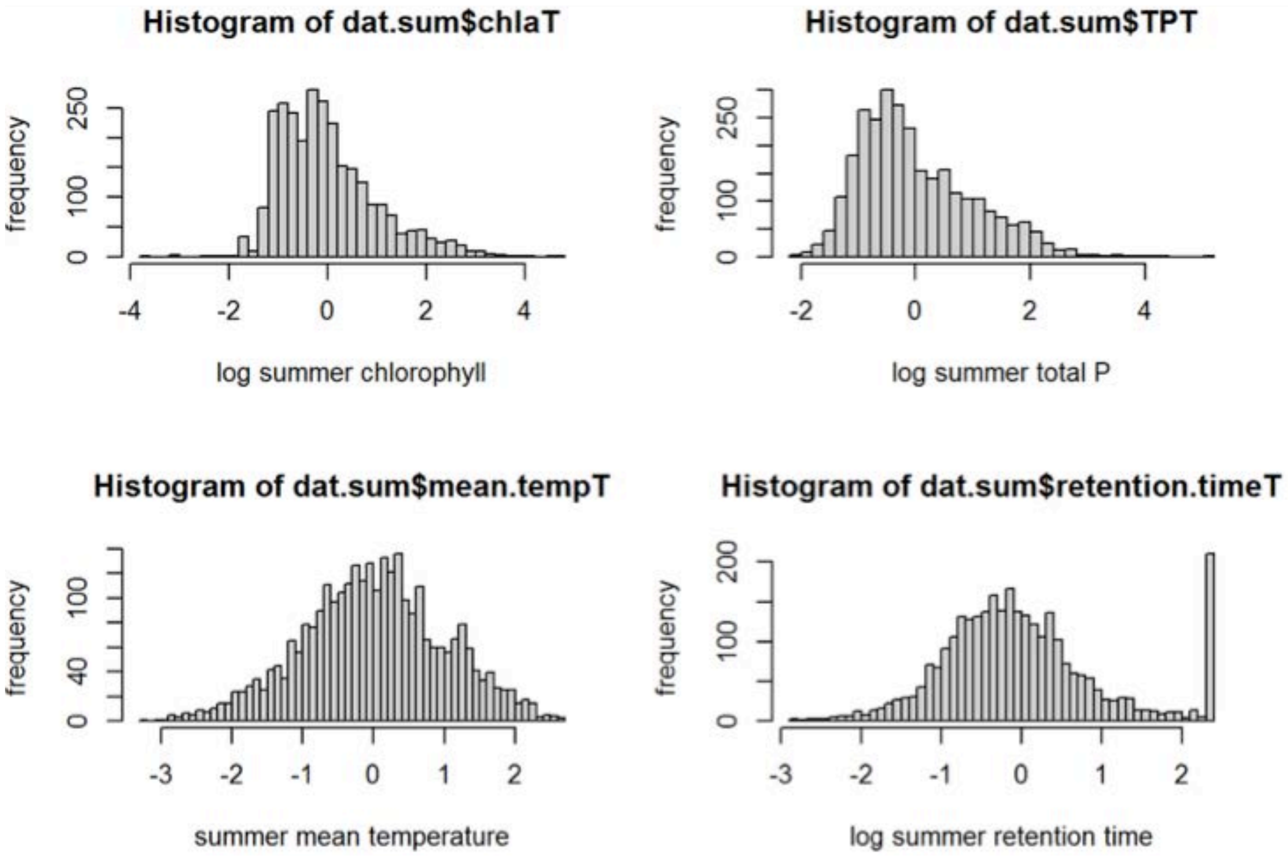


Figure A5.1. Histograms of Scottish loch and reservoir data used in the study. All data except water temperature were $\log_e$ transformed; all data were standardised. *© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.*

Prior to modelling, extensive data exploration was undertaken. During this phase, boxplots of the chlorophyll-*a* data (Figure A5.2) were constructed. These indicated that higher concentrations of chlorophyll-*a* were found mostly in very shallow, high alkalinity lochs.

Initially a null model that had no environmental explanatory variables was examined, to investigate the effects of different lochs (LakeID), different summer months (July, August or September) and different years (2012-2019) on transformed and standardised chlorophyll-*a* concentrations. This revealed that loch and reservoir identity explained considerably more variability in chlorophyll-*a* concentrations than month or year and that there was a lot of unexplained residual variability, potentially related to environmental conditions that had not been included in the analyses (Table A5.1).

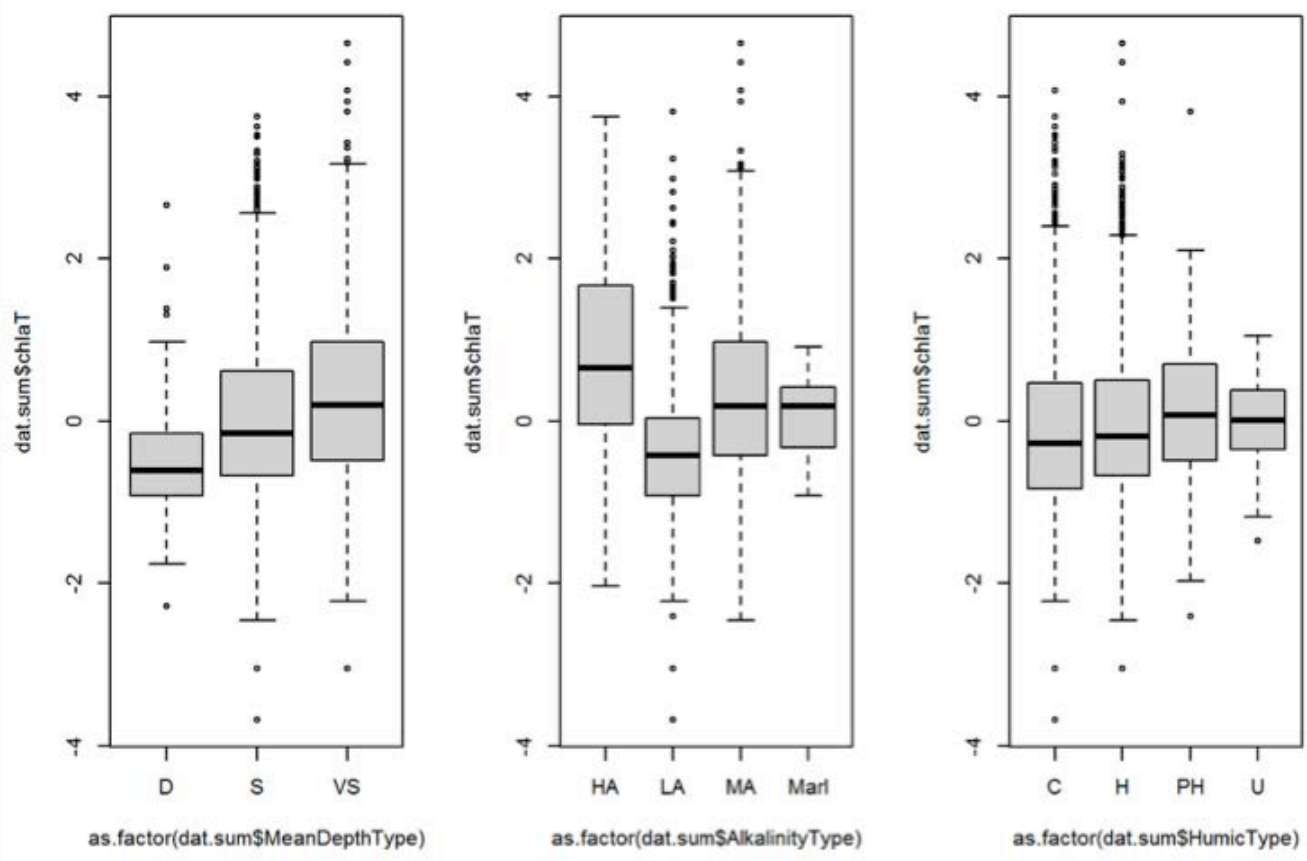


Figure A5.2. Boxplots of the chlorophyll-*a* data showing that higher concentrations occur in very shallow, high alkalinity lochs.
© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.

Transformed and standardised mean summer TP concentrations, water temperatures and retention times for each month were then added to the model, with interaction terms, to fit the following model:

$$\gamma=\beta_0 + \beta_1\chi_{TP} + \beta_2\chi_{Temp} + \beta_3\chi_{Retention} + \beta_4\chi_{TP\ x\ Temp} + \beta_5\chi_{TP\ x\ Retention} + \beta_6\chi_{Temp\ x\ Retention} + \beta_7\chi_{TP\ x\ Temp\ x\ Retention} + \delta_{LakeID} + \delta_{year} + \varepsilon_{,}\ \ \gamma \sim (0,\sigma_l^2),\ \ \varepsilon \sim (0,\sigma_r^2)$$

where γ is the response of interest (chlorophyll-*a*), β_0 is the intercept term, β_1, β_2, and β_3 are model parameters for the TP concentration, water temperature and retention time terms, respectively. The model parameters for the interactions are β_4 (TP and temperature), β_5 (TP and retention time),β_6 (temperature and retention time) and β_7 (TP, temperature and retention time). δ_{LakeID} and δ_{year} are the random effect terms for lake ID and year that allow the response to vary on the intercept for individual lakes and years. These terms accommodate the "nested" structure of the dataset, i.e., where there are multiple observations "within" specific years and waterbodies. The random effects allow mean chlorophyll-*a* concentration to vary systematically among lochs and years, on top of the effects of the fitted explanatory

Table A5.1: Null model showing the relative effects of time (month and year) and space (LakeID) on variability in chlorophyll-*a* concentrations in Scottish lochs. *© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.*

Variable	Variance Explained	Std.Dev.
LakeID	0.56	0.75
Year	0.02	0.16
Month	0.00	0.05
Residual	0.44	0.67

Observations: 2643; Waterbody IDs: 132; Years: 10; Months: 3 variables. Finally, ε is the overall error term with a mean of zero and unknown variance (estimated during model fitting). Due to convergence issues, the random effect of month was excluded from the models.

Manual backwards selection based on Akaike Information Criterion (AIC) values was then used to remove "unimportant variables" and arrive at the most parsimonious model, i.e., the simplest model that explains the most variability statistically. This resulted in four fixed effects (mean.temp, TP, retention.time, mean.temp*retention.time) and two random effects (Lake.ID, Year) being selected for the final model:

*Chlorophyll-a = mean.temp + TP + retention.time + mean.temp*retention.time + (1/Lake.ID) + (1|Year)*

The fixed effects in the final model explained 32% of the variation in the transformed and standardised chlorophyll-*a* concentration data (marginal R^2) and 54% the variation was explained when the random effects of Lake.ID and year were included (conditional R^2). The model coefficients are shown in Table A5.2.

Table A5.2 Linear mixed effects model coefficients explaining chlorophyll-*a* concentrations in Scottish lochs in relation to mean monthly total phosphorus (TP) concentration, air temperature and retention time. All variables, except mean temperature, were Log_e transformed before being standardised (see text for details). *© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.*

All Lakes	Estimate	Std.Error	df	t-value	*p*	
(Intercept)	0.0516	0.0656	17.8	0.79	0.442	
mean.temp	0.0102	0.0179	2117.7	0.57	0.569	
TP	0.5364	0.0239	939.5	22.43	0.000	***
retention.time	0.0472	0.0177	2583.6	2.67	0.008	**
mean.temp*retention.time	-0.0576	0.0134	2566.2	-4.29	0.000	***

The model indicates a highly significant positive relationship between TP and chlorophyll-*a* concentrations (Figure A5.3) and a highly significant interaction between temperature and retention time in their relationship with chlorophyll-*a* concentrations – indicating that the "effect" of one of these variables on chlorophyll-*a* concentrations depends upon the value of the other. At high retention times (low flushing rates), mean water temperature had a negative relationship with chlorophyll-*a*, whereas at low retention times (high flushing rates) a positive relationship was observed between temperature and chlorophyll-*a* (Figure A5.4).

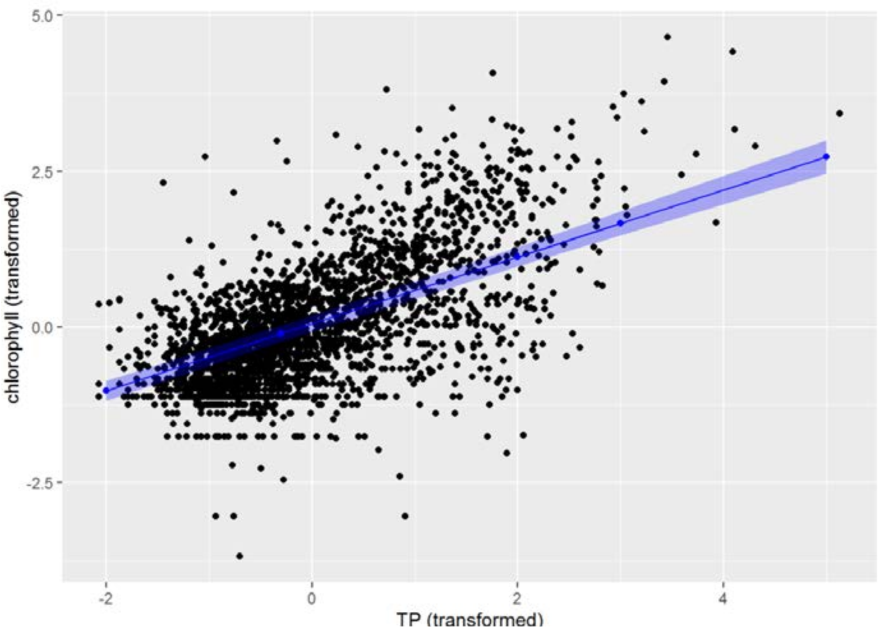


Figure A5.3: Relationship between transformed TP and chlorophyll-*a* concentrations in Scottish lochs showing the modelled best fit relationship (blue line) and confidence bands (blue shaded area). *© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.*

Examining the correlations between chlorophyll-*a* and temperature and retention time, separately, for different lake types, the strongest relationships were found between different depth types, especially positive relationships for shallow and very shallow lochs (Figures A5.5 and A5.6). In contrast, no strong relationships were observed among humic or alkalinity types.

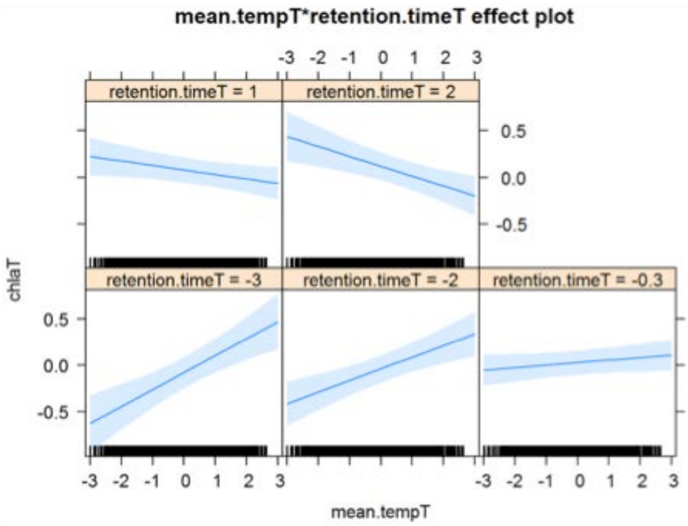


Figure A5.4: Relationship between standardised mean temperature and transformed chlorophyll-*a* concentrations in Scottish lochs, contingent on retention time, showing the fitted modelled relationship (blue line) and confidence bands (blue shaded area).
© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved

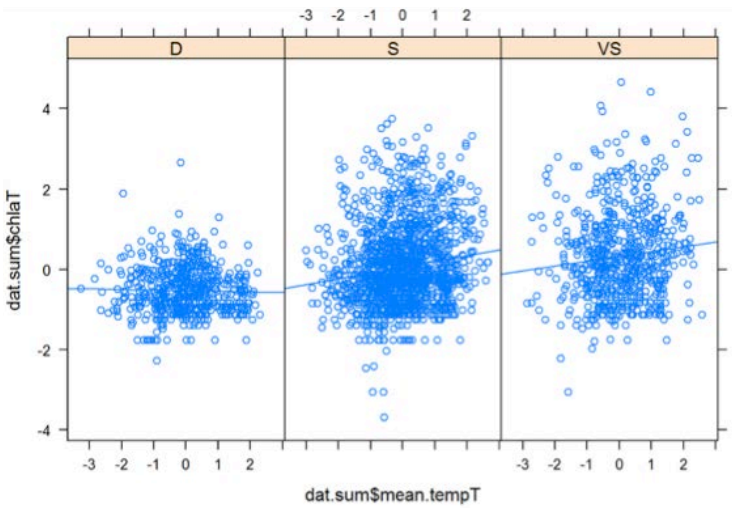


Figure A5.5 Relationship between standardised mean temperature and transformed chlorophyll-*a* concentrations in Scottish lochs by depth type (D = Deep; S = Shallow; VS = Very Shallow).
© UKCEH. Contains SEPA data
© Scottish Environment Protection Agency and database right 2021. All rights reserved.

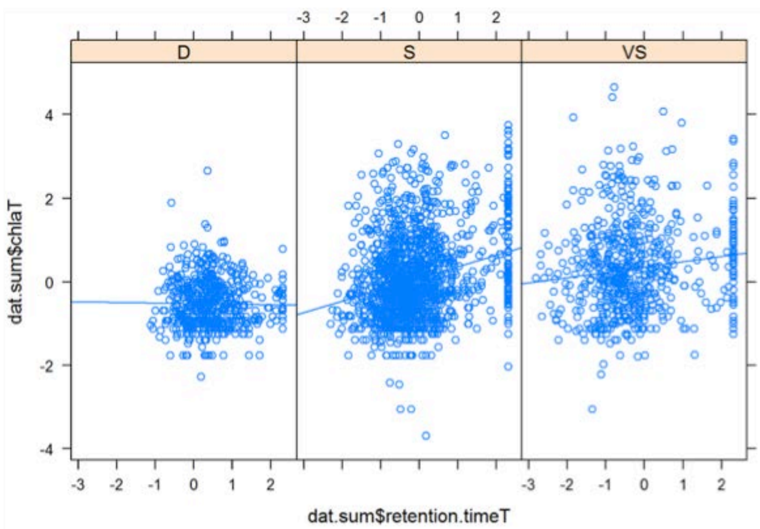


Figure A5.6. Relationship between transformed retention time and chlorophyll-*a* concentrations in Scottish lochs by depth type (D = Deep; S = Shallow; VS = Very Shallow**.**
© UKCEH. Contains SEPA data
© Scottish Environment Protection Agency and database right 2021. All rights reserved.

Type specific models

The distribution of data by loch and reservoir types in relation to Water Framework Directive (WFD) typology classes for mean depth and alkalinity is shown in Table A5.3. There were too few sites to consider specific lake-type models for depth and alkalinity gradients combined (e.g., High alkalinity deep lakes), so separate type-specific models were developed for different depth types, only (i.e., 29 very shallow, 75 shallow and 28 deep lochs). Models for specific depth types were chosen because exploratory analysis had shown that this variable had the greatest effect on chlorophyll-*a* responses to temperature and retention time. A previous analysis of European lakes (Richardson et al., 2018) also showed that chlorophyll-*a* concentrations varied most by depth types.

Table A5.3. Number of lochs with sufficient data available for them to be assigned to WFD mean depth and alkalinity typologies. *© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.*

	Very shallow	Shallow	Deep	Total
Low alkalinity	12	41	21	74
Medium alkalinity	7	21	7	35
High alkalinity	10	12		22
Marl		1		1
Total	**29**	**75**	**28**	**132**

Very shallow lochs (mean depth < 0.3m)

Calculations were based on 611 summer months of data from 29 lochs and collected between 2010 and 2019. For very shallow lakes, the "best" model included only a (positive) fixed effect of TP, which explained about 37% of the variation in transformed and standardised chlorophyll-*a* values (marginal R^2). In addition, 55% of the variation was explained when the random effects of *LakeID* and year were included (conditional R^2). The model coefficients are shown in Table A5.4. There were no significant effects of the climate change related variables on chlorophyll-*a* concentrations, making it impossible to use this approach to predict the impacts on chlorophyll-*a* concentrations of future climate change scenarios.

Table A5.4. Linear mixed effects model coefficients explaining chlorophyll-*a* concentrations in very shallow Scottish lochs. All variables were Log_e transformed and standardised (see text for details). *© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.*

Very Shallow Lochs	Estimate	Std.Error	df	t-value	*p*
(Intercept)	-0.03202	0.10177	31.7	-0.315	0.755
TP	0.64001	0.04669	337.1	13.707	0.000***

Shallow lochs (mean depth 3-15m)

Calculations were based on 1465 summer months of data collected from 75 lochs between 2010 and 2019. For shallow lochs, the most complex model produced the best fit to the measured data. Essentially, all of the effects interacted (were dependent on each other) and, collectively, they explained 26% of the variability in the transformed and standardised chlorophyll-*a* data (Table A5.5). These significant interaction effects make it difficult to visualise the effects of each explanatory variable individually.

Table A5.5. Linear mixed effects model coefficients explaining chlorophyll-*a* concentrations in shallow Scottish lochs. All variables, except temperature, were log_e transformed and all variables were standardised. *© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.*

Shallow Lochs	Estimate	Std.Error	df	t-value	*p*
(Intercept)	1.01E-01	7.77E-02	3.07E+01	1.301	0.203
mean.temp	-1.07E-02	2.42E-02	1.10E+03	-0.441	0.659
TP	4.84E-01	3.59E-02	6.87E+02	13.514	0.000 ***
retention.time	6.54E-02	2.63E-02	1.45E+03	2.49	0.013 *
mean.temp*TP	-5.35E-03	2.11E-02	1.41E+03	-0.254	0.800
mean.temp*retention.time	-9.31E-02	2.14E-02	1.37E+03	-4.344	0.000 ***
TP*retention.time	7.86E-03	2.09E-02	1.44E+03	0.376	0.707
mean.temp*TP*retention.time	4.56E-02	1.70E-02	1.38E+03	2.683	0.007 **

Level of significance: *** p<0.001; ** p<0.01; * p<0.05

Deep lochs (mean depth >15m)

Calculations were based on 567 summer months of data collected from 28 lochs between 2010 and 2019. For deep lochs, TP, mean temperature and retention time were selected in the best model as well as the interaction between temperature and retention time (Table A5.6). The model indicates a highly significant positive relationship between TP and chlorophyll-*a* (Figure A5.7). Lakes with shorter retention times (highly flushed) showed a positive effect of temperature on chlorophyll-*a* concentrations, whereas lakes with long retention times (less flushed) showed a negative effect (Figure A5.8). The fixed effects in the best model explained 5% of the variation in transformed chlorophyll-*a* concentrations, only, with 29% of the variation explained by the fixed and additional random effects (Lake ID and year). This indicated that a large proportion of the variability in chlorophyll-*a* remains relatively unexplained and the model is likely to have low power in terms of predicting the effects of climate change on future chlorophyll-*a* concentrations.

Table A5.6. Linear mixed effects model coefficients explaining chlorophyll-*a* concentrations in deep Scottish lochs. All variables apart from temperature were $\log_e$ transformed and all variables were standardised. *© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.*

Deep Lochs	Estimate	Std.Error	df	t-value	*p*
(Intercept)	-0.3205	0.0978	21.5	-3.276	0.004 **
mean.temp	0.0650	0.0311	528.4	2.088	0.037 *
TP	0.2274	0.0564	376.9	4.032	0.000 ***
retention.time	0.0480	0.0406	326.5	1.184	0.237
mean.temp*retention.time	-0.0977	0.0343	550.0	-2.849	0.005 **

Level of significance: *** p<0.001; ** p<0.01; * p<0.05

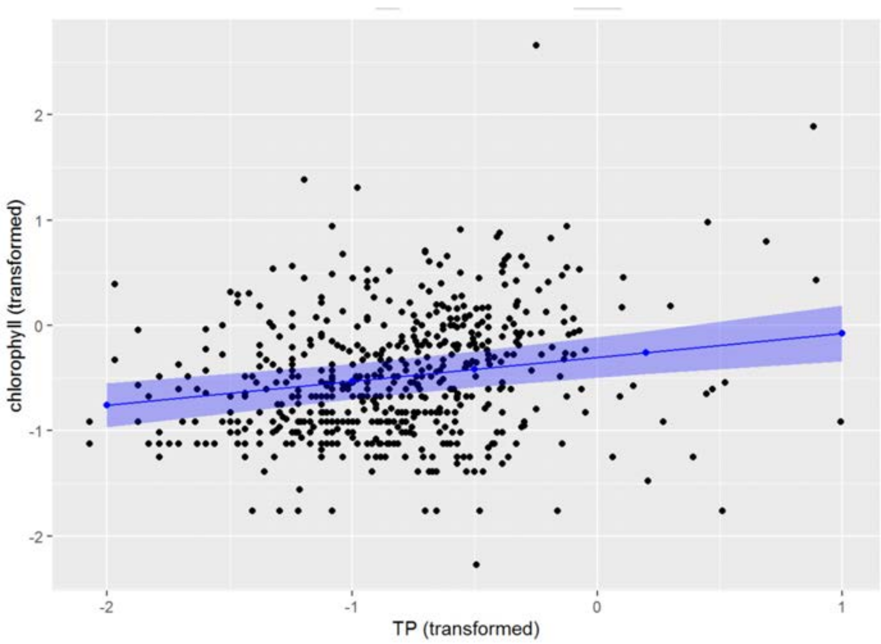


Figure A5.7. Relationship between Loge transformed TP and chlorophyll-*a* concentrations in deep lochs showing the modelled best fit. *© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.*

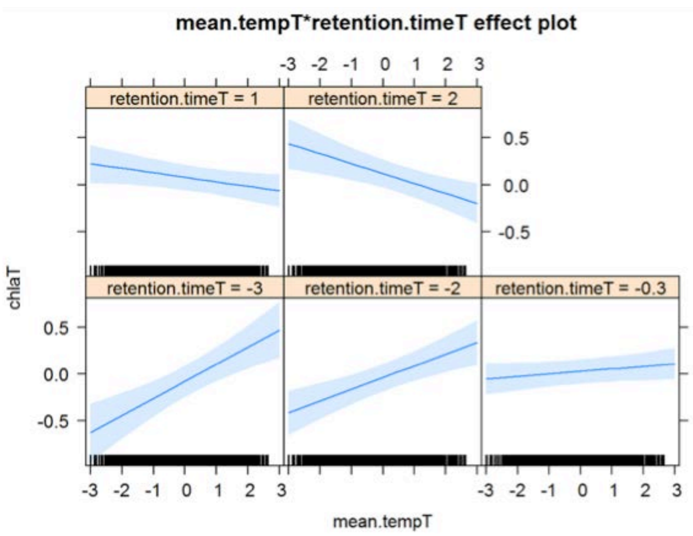


Figure A5.8. Modelled relationships between mean water temperature (mean.tempT) and chlorophyll-*a* (chlaT) concentrations, illustrating the interactive effect of mean temperature and retention time on chlorophyll-*a* concentrations in deep lochs. *© UKCEH. Contains SEPA data © Scottish Environment Protection Agency and database right 2021. All rights reserved.*

Conclusions and Recommendations

Variability in mean monthly chlorophyll-*a* concentrations in Scottish lochs in summer is best explained by monthly mean TP concentrations. Variability in chlorophyll-*a* concentrations in relation to environmental factors is best explained by splitting the data into depth-types and developing type-specific models. These show the following:

1. **Very shallow lochs**: Chlorophyll-*a* concentrations were best explained by TP concentrations; climatic variables did not have a significant effect. This fits with general ecological theory and process understanding, which indicates that internal food web interactions, including aquatic plants, have a strong role in structuring ecosystems in very shallow lochs
2. **Shallow lochs:** All of the variables explored significantly affected chlorophyll-*a* concentrations, but they also showed significant interactions. This made it difficult to visualise or describe, in simple terms, the relationships between chlorophyll-*a* concentrations and individual environmental factors. In general, TP and retention time had strong, positive relationships with chlorophyll-*a* concentrations, and temperature had a weaker, negative relationship.
3. **Deep lochs:** TP and the interaction between temperature and retention time significantly affected chlorophyll-*a* concentrations. In general, TP had a significant positive relationship with chlorophyll-*a* concentrations and mean water temperature had a positive relationship in lochs with shorter retention times (highly flushed) but this relationship became negative at lower flushing rates.

The results obtained may be affected by the relatively small number of examples of very shallow and deep loch and reservoir types included in the analyses.

It is recommended that modelling of the future impacts of climate change is best evaluated for shallow lochs, only, using the model developed. The deep lochs model should be applied with caution because much of the variability in chlorophyll-*a* concentrations could not be explained by the variables considered. It is likely that the performance of this model could be improved by including data on other variables in the modelling process. Factors such as water colour, alkalinity, nitrogen availability, and aquatic plant, fish and zooplankton population densities are all likely to have important effects on chlorophyll-*a* concentrations. Development of individual loch and reservoir models, such as those developed for Loch Leven, would overcome some of these issues as water colour and alkalinity do not vary greatly at this particular site, reducing the number of confounding effects obscuring the impacts of the climatic drivers.

6 Appendix 6: Site-specific effects of total phosphorus, air temperature and retention time on chlorophyll-*a* concentration in Loch Leven

Assessing multiple stressor effects in lakes

The effects of multiple stressors on phytoplankton biomass in lakes, typically measured as chlorophyll-*a* concentration, have been explored using a range of statistical approaches (Feld et al., 2016; Richardson et al., 2019), including linear models (Spears et al., 2022a). These methods enable the effects of two or more stressor variables on a single response variable to be examined, as conceptualised in Figure A6.1. This approach has been used to assess interactions between nutrient concentrations and weather variables, as proxies for eutrophication and climate change, in European catchments (Birk et al., 2020). In general, these previous studies have indicated that nutrient stressors usually exert the strongest effects on chlorophyll-*a* concentrations, but changes in temperature and/or rainfall (expressed as flushing rate, hydraulic retention time, or precipitation) may exacerbate or dampen the effects of nutrients, depending on the form of interaction. The results reported in Appendix 5 indicate that Scottish lochs are sensitive to changes in weather and that interactions between temperature, retention time and nutrient concentrations may be lake-type specific and that an interaction between temperature and retention time is operating at the national scale.

Both temperature and retention time may interact with nutrients, and potentially with one another, to affect phytoplankton biomass. As indicated in the literature review (Appendix 1), the processes underlying these interactions are likely to vary among lakes and seasons depending on trophic structure, nutrient conditions, and lake morphology. So, it is expected that site-specific interactions may vary from those identified at lake type or national scales. For example, phytoplankton productivity may increase with temperature under high nutrient conditions (Elliott et al., 2006), but Scheffer et al. (2001) highlights the importance of warming in driving (through promotion of zooplankton grazing) the onset and duration of the spring clear water phase (i.e., lower chlorophyll-*a* concentrations) in shallow, eutrophic and temperate lakes. Jones et al. (2010) discuss the complexities of the chlorophyll-*a* – retention time relationship for lakes in general and demonstrate that, for Bassenthwaite Lake (annual retention time of about 20 days), phytoplankton biomass responds seasonally to changes in retention time dependent upon nutrient availability. Where the main nutrient source was ‘flow-independent’, summer biomass was projected to increase by about 70 per cent in response to future reductions in precipitation. However, where the nutrient source was flow-dependent, then biomass was predicted to fall. Taken collectively, these studies indicate that responses of phytoplankton community biomass (chlorophyll-*a* concentrations) to changes in retention time and water temperature, related to climate change, may be highly site-specific. It is, therefore, important, where data are available, to confirm empirically or use process models operating at the site specific scale to assess the relationship between weather stressors, nutrient concentrations, and ecological response indicators to inform the testing of future climate change scenarios (Spears et al., 2021).

Here, we demonstrate such an approach using simple linear models to systematically assess historical variation in seasonal and annual chlorophyll-*a* concentrations in Loch Leven as a result of potential interactions between measured total

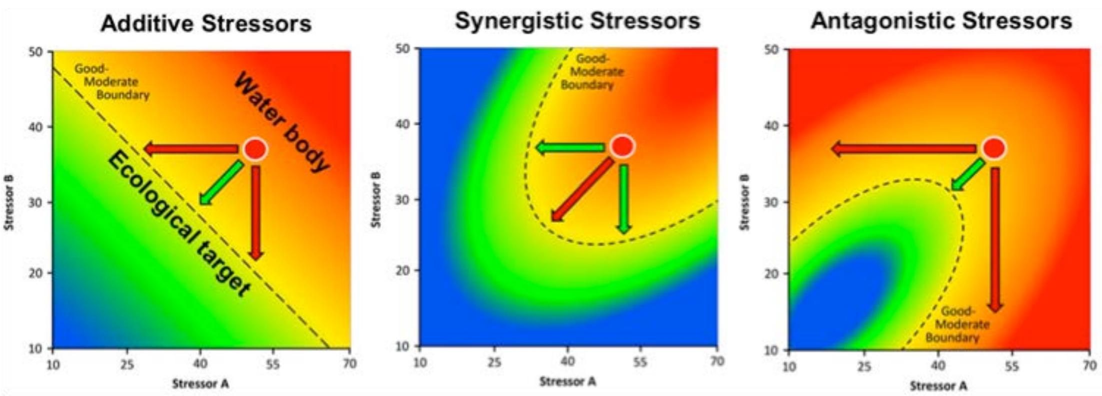


Figure A6.1. Conceptual description of paired-stressor models demonstrating hypothetical stressor interactions and abatement scenarios relative to an ecological target, for example, as set by the ‘Good-Moderate Boundary’ as defined in the European Water Framework Directive (WFD). The most effective stressor abatement option is coloured green.

phosphorus (TP) concentrations and hindcast (i.e., up to 2019; UKCP18 RCP6.0) estimates of air temperature and retention time. As applied here, this approach provides pairwise visualisations of the effects, on chlorophyll-*a* concentration, of total phosphorus (TP) concentration compared to either air temperature or retention time.

Loch Leven case study and evidence of climate change sensitivity

Loch Leven is internationally recognised as a conservation area for waterfowl and macrophytes (Sites of Special Scientific Interest; Ramsar; Special Areas of Conservation, Natura network 2000) and is a source of water supply to downstream industry (May & Spears, 2012). The lake is a nationally important destination for eco-tourism and recreation, attracting more than 200,000 visitors per year (ScotInform, 2015). The lake is also an important focus of income generation for many local businesses, and the current management plan has been designed to balance biodiversity protection with controlled visitor access to support these socio-economic benefits (SNH, 2016). Its catchment is farmed intensively for livestock, poultry, and crop production (Castle et al. 1999, LLCMP 1999) and supports a population of about 11,000 people (*pers. com*., Perth and Kinross Council, Feb. 2018).

Historically, Loch Leven has received nutrient pollution from industry, wastewater treatment works and farm runoff (May et al., 2012). These stressors caused an increase in in-lake TP and chlorophyll-*a* concentrations, a decrease in Secchi disk transparency and deterioration in the extent and diversity of aquatic macrophyte and bird communities (Dudley et al. 2012, May and Carvalho 2010, Carss et al. 2012). In the 1980s and 1990s, measures were introduced to reduce nutrient inputs to the loch. The focus was mainly on point sources, achieving an estimated 60% reduction in the TP input to the lake. Total phosphorus concentrations in the loch declined thereafter and the water quality targets set by the Loch Leven Catchment Management Plan (LLCMP 1999) of <40 µg L^{-1} for TP, <15 µg L^{-1} for chlorophyll-*a*, and >2.5 m for Secchi depth, had been met by 2010 (Carvalho et al., 2012). Currently, the target value for annual mean chlorophyll-*a* concentration as defined by the EU Water Framework Directive (WFD) is 11 µg L^{-1} (Carvalho, et al., 2009). Although the water quality and ecology of Loch Leven improved sufficiently to meet the restoration targets set by the Loch Leven Catchment Management Plan, this took decades to achieve most likely due to the internal cycling of P between the bed sediments and the overlying water column, a process that drives TP concentrations in late-summer and autumn (Spears et al. 2012).

Warming related to climate change has been proposed as a factor that may have confounded recovery (Carvalho et al., 2012; O'Reilly et al., 2015). Carvalho et al. (2012) associated a significant increase in spring Daphnia densities in recent years with increases in water temperature that coincided with lower chlorophyll-*a* concentrations and higher water clarity in spring and early summer. At the same time, high rainfall appears to have resulted in low chlorophyll-*a* concentrations in some years, probably as a result of increased flushing rates (Carvalho et al. 2012). In contrast, May et al. (2017) observed that intense periods of rainfall, which may also be reflected as low retention times in this study, resulted in an increased P load from the catchment that could, potentially, lead to higher phytoplankton biomass. Spears et al. (2022a) assessed the effects of water temperature, local precipitation and TP concentrations on annual chlorophyll-*a* concentrations and reported that, in general, TP varied positively, precipitation varied negatively, and water temperature varied negatively (growing season mean water temperature) and positively (winter mean water temperature) with annual mean chlorophyll-*a* concentration. Finally, in recent years, the perceived recurrence of algal blooms at Loch Leven has renewed attention on the relative importance of weather extremes compared to nutrient enrichment as drivers of phytoplankton biomass, especially in the summer and autumn 'bloom periods'.

Data processing

Loch Leven (Table A6.1), a shallow lake in the UK, offers a unique time series of data from a single sampling site with fortnightly sampling frequency between 1967 and 2017. In this study, chlorophyll-*a* concentration in the water column has been considered as a proxy for phytoplankton biomass and TP concentration in the water column as a proxy for nutrient enrichment. Both chlorophyll-*a* and TP concentration data for the period 1989 to 2019 were extracted from the UKCEH Loch Leven Long-Term Monitoring Programme, part of the UKSCAPE project (https://uk-scape.ceh.ac.uk/our-science/projects/loch-leven). These data were processed to produce monthly average concentrations. Data for air temperature and retention time at Loch Leven were extracted from the UKCP 18 RCP 6.0 Scenario data (see Appendix 3) producing monthly values for the period 1989 to 2019 (and also to 2080 for stressor forecasts). For the linear modelling, monthly retention time values in excess of 100 years were removed (n = 33 removed from 372 monthly values). Values in excess of 100 years occurred typically during the summer and autumn months at Loch Leven. Data for each variable were processed to produce average spring (March to May), summer (June to August), autumn (September to November) and annual (January to December) values for each year between 1989 and 2019. This data processing method produced values for all seasons in all years for all variables, with the exception of summer 2007 for retention time.

Pairwise linear modelling approach

The approach used here generally follows that described in Spears et al. (2022a). However, rather than selecting the best-fit model using a model screening and ranking method with multiple predictor variables, here model selection has been forced systematically to predict the response variable, chlorophyll-*a* concentration, as a function of TP and either temperature or retention time and their interactions for each season. Individual pairwise models were constructed to aid visualisation of the interactions described below. In this way, the hypothesis that climate stressors will interact with nutrient stressors to moderate chlorophyll-*a* concentration, and that these interactions will vary seasonally, was tested. All response variables were modelled with Gaussian errors. All models were fitted by maximum likelihood using the R lm function. Prior to model fitting, response variables and covariates were transformed using Box-Cox transformations, offset by a small value to ensure that all values were greater than zero.

The approach above was applied systematically for each season, and for annual conditions, to produce models with combinations of predictor variables, such as TP and air temperature and TP and retention time, and their interactions. Model residuals were checked for normality and constant variance. The predicted variation in chlorophyll-*a* concentration for each stressor pair were visualised using 'heat maps,' which depict the strength of the response along a blue (low) to red (high) gradient. The gradients of response and predictor variables included in the heat maps are constrained to the model input data. A reference value of 30 µg L^{-1} chlorophyll-*a* concentration is included in each heat map; this is an arbitrary value included for reference, only. These models should not be used to extrapolate beyond the range of the data used to fit them.

Interpretation of linear models for Loch Leven

The results of the systematic linear modelling analysis are presented in Table A6.1 and Figures A6.2 (annual model) and A6.3 (seasonal models). Total phosphorus concentration explained the greatest variation in chlorophyll-*a* concentration across all models and pairwise models were statistically significant for TP vs. temperature and TP vs. retention time for annual, spring and summer models, but not for autumn.

Effects of total phosphorus and temperature on chlorophyll-*a* vary with season

Expected chlorophyll-*a* concentration increases towards the right hand side of each heat map, confirming that TP is an important predictor of this response in all models (Table A6.1). However, the change in chlorophyll-*a* concentration with temperature varies with season along the TP axis. For TP vs. temperature in spring and autumn, the effect of TP on chlorophyll-*a* level is predicted to reduce at higher temperatures (i.e., from 6°C to >10°C in spring and from about 7°C to 11°C in autumn). This suggests an interaction effect in autumn whereby the effect of temperature appears to reverse under low TP concentrations. In contrast to spring and autumn, summer chlorophyll-*a* concentrations are highest at high TP concentrations and high temperatures, especially above about 80 µg L^{-1} and 14°C. As introduced above, the effects of temperature on chlorophyll-*a* concentration may be both positive and negative and this can be explained by increasing grazing rates (i.e., negative relationship observed previously in spring) or increasing phytoplankton production (positive relationship in summer) under warmer conditions. Our analysis suggests that both processes play an important role in moderating the dominant TP vs. chlorophyll-*a* relationships in Loch Leven. The annual heat map (Fig A6.2) represents the net effects of both processes (and perhaps other temperature sensitive processes as yet unidentified) where chlorophyll-*a* concentrations are both lowest and highest at the upper end of the temperature gradient, and, at opposite ends of the TP gradient, on an annual scale.

Effects of total phosphorus and retention time on chlorophyll-*a* vary with season

Higher retention times appeared to dampen the effects of TP on chlorophyll-*a* concentrations (at high TP) in spring and summer. This effect appears to be strongest at >50 µg L^{-1} TP and at retention times of less than about 3 years. In contrast, high retention times were predicted to exacerbate the effects of higher TP concentrations in autumn, especially above 60 µg L^{-1} TP and 0.5 years retention time. As with the contrasting seasonal effects of temperature, retention time may also exhibit negative and positive effects on the phytoplankton community. As introduced above, positive effects of retention time on phytoplankton biomass (i.e., increasing chlorophyll-*a* concentration with decreasing flushing rate/increasing retention time) may indicate a decrease in the removal of nutrients and phytoplankton from the lake. In contrast, a negative effect (e.g., increasing chlorophyll-*a* with increasing flushing rate/decreasing retention time) may indicate increasing nutrient loading from the catchment. Again, the annual heat map (Figure A6.2) represents the net effects of these seasonal conditions and suggests an increase in chlorophyll-*a* concentration at high retention times, but only when annual mean TP concentrations are high.

Table A6.1. Summary of loch typology and linear model coefficients for Loch Leven for the 1989 to 2019. maod – metres above ordinance datum; WFD – European Water Framework Directive; HA – high alkalinity; LA – low alkalinity; VS – very shallow; S – shallow; D – deep. All data extracted from UK Lakes Portal 9th Sep. 2022 (https://eip.ceh.ac.uk/apps/lakes/). * to the left and right of season indicates model p values for TP x Temperature and TP x Retention time, respectively. *** p < 0.001; ** p > 0.001 < 0.05; * p > 0.05 < 0.1. Underscored values indicate p values for individual effects of <0.05. Chl – mean chlorophyll-*a* concentration of the season indicated. TP – mean total phosphorus for the season indicated; T – mean air temperature for the season indicated; Ret – mean retention time for the season indicated. Model coefficients are based on transformed data. MRsq – multiple R squared value; ARsq – adjusted R squared value.

Name	**Mean depth (m)**	**Grid Reference**	**Elevation (maod)**	**Surface area (ha)**	**WFD Lake type**	
Loch Leven	4.5	NO14720146	106	1371	HAS	
Pairwise Linear Model Coefficients						
Chl	TP x T			**TP x Ret**		
Chl	TP	T	TPxT	**TP**	**Ret**	**TPxRet**
Ann	***0.65	-0.20	0.15	0.74***	0.07	0.07
	MRsq: 0.61 ARsq: 0.57			MRsq: 0.57 ARsq: 0.52		
Spr	**0.63	-0.17	-0.17	0.70***	0.06	-0.13
	MRsq: 0.52 ARsq: 0.46			MRsq: 0.49 ARsq: 0.43		
Sum	***0.80	-0.05	0.09	0.79***	-0.17	-0.03
	MRsq: 0.67 ARsq: 0.62			MRsq: 0.67 ARsq: 0.64		
Aut	0.36*	-0.05	-0.11	0.39**	0.05	0.14
	MRsq: 0.17 ARsq: 0.07			MRsq: 0.18 ARsq: 0.09		

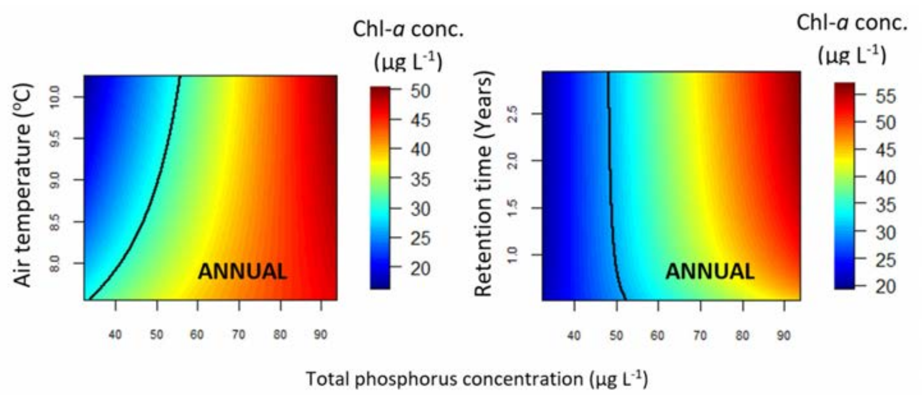


Figure A6.2. Heat maps showing the effects predicted by linear models of annual mean total phosphorus (TP) concentration and air temperature (left panel), and annual mean TP concentration and retention time (right panel), on the expected response in annual mean chlorophyll-*a* concentration in Loch Leven between 1989 and 2019. The black line is an arbitrary reference value of 30 µg L^{-1} chlorophyll-*a* concentration.

Implications of results from a climate change management perspective

The approach demonstrated here offers visualisation of the effects of paired nutrient and climate change stressors on chlorophyll-*a* concentration in Loch Leven. The analysis targeted temperature and retention time, only, although UKCEP18 RCP outputs for wind speed may come on-line in the future, to expand the list of climate change stressors. Additionally, indicators of zooplankton grazing rates, inorganic nitrogen concentrations and modelled catchment nutrient loading would strengthen the modelling approach for Loch Leven. It is likely that a combination of these stressors beyond simple pairwise comparisons, for example, using the Linear Mixed Effects (LME) Modelling framework, will significantly improve model performance, thereby strengthening predictive capacity.

Addressing limitations in the approach for application across other Scottish Lochs.

A similar analysis to that conducted here for Loch Leven could be conducted for other Scottish Lochs where sufficient water quality monitoring data are available. We used a data span between 1989 and 2019 for Loch Leven during a period when analytical and sample collection methodologies were consistent and data quality high. On exploration of the SEPA Water Chemistry data for other lakes it was apparent that quality issues, specifically for TP concentration, were a concern preceding 2002, potentially constraining the period of observation for other lochs. Specifically, SEPA laboratory methodologies were

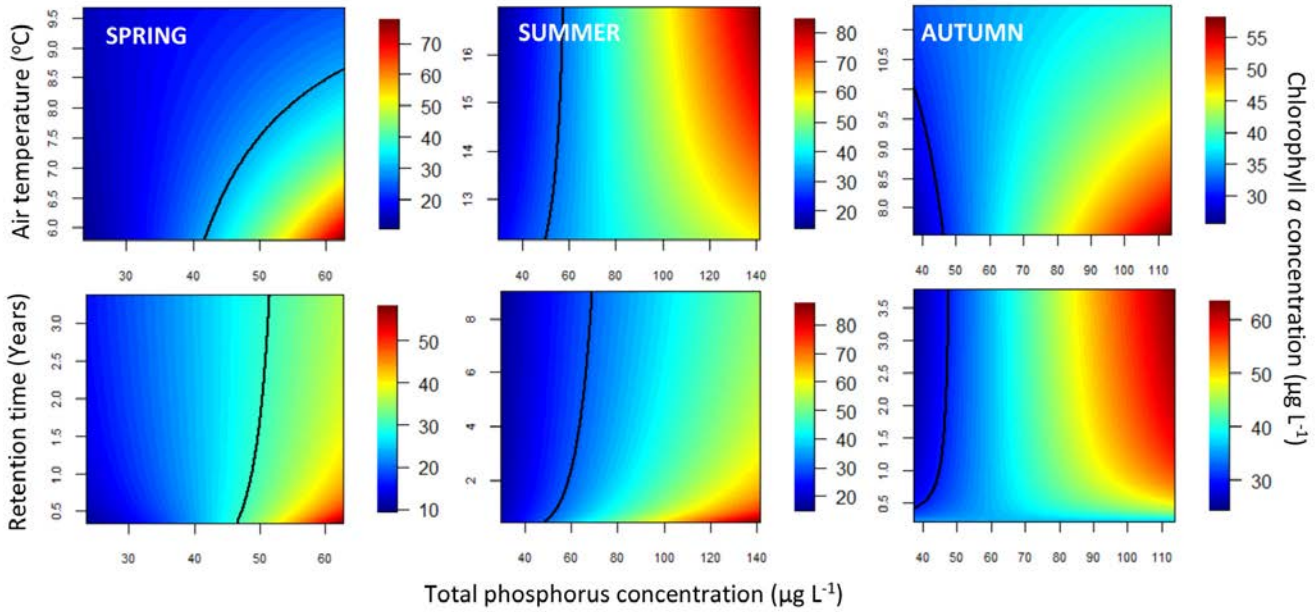


Figure A6.2. Heat maps showing the effects predicted by linear models of annual mean total phosphorus (TP) concentration and air temperature (left panel), and annual mean TP concentration and retention time (right panel), on the expected response in annual mean chlorophyll-*a* concentration in Loch Leven between 1989 and 2019. The black line is an arbitrary reference value of 30 µg L^{-1} chlorophyll-*a* concentration.

inconsistent across lakes prior to 2002 (SEPA, *pers comm*.). In addition, monitoring frequency was variable across lakes both before and after 2002, with a lower frequency of data in August and during winter months. This may further constrain application of the approach for other lochs to include assessments of growing seasons (e.g., May to September), only, to ensure consistency of data processing across sites. It was for these reasons that we focussed our analysis on demonstrating the approach using the longer and more consistent Loch Leven data, in the first instance. On a positive note, the climate change stressor data produced within this project represents a consistent data resource with which similar analyses could be conducted across other lakes. First, there is a need to develop the statistical approach to address data quality issues required to ensure comparability of stressor effects between lakes.

The need to capture the effects of extreme events in climate change scenarios

Spears et al. (2021 and 2022a) propose that the LMEs can be used to estimate the probability of failing a given chlorophyll-*a* target to inform future management interventions. An important caveat to this proposal is that future projections using LMEs, or similar multi-variate empirical modelling approaches, should theoretically be constrained to the gradient of the measured data. Climate change scenarios for Loch Leven using UKCP18 RCP Scenario 6.0 (Fig A6.4) indicate that future temperatures are at least in the range of measured temperatures in our models, and so future projections are feasible at least at this site, for this scenario, and for temperature. However, retention time projections indicate an increase in the occurrence of extreme events (e.g., drought conditions, indicated by very high retention times; Fig A6.4). For our models, we removed values in excess of 100 years. So, our models should represent extreme wet conditions but will underrepresent extreme dry conditions. An alternative modelling approach will be necessary to specifically capture the responses of chlorophyll-*a* concentration to extreme events, especially where they are rare in historical data. Consideration should be given to climate change scenario development for lakes, including indicators of intensity and duration of extreme events beyond threshold values indicated in the literature review, and including temperature, wind and retention time as stressors.

Combining process and empirical modelling to inform climate change management

A combination of process and empirical modelling is likely to be necessary to unpick the complex processes moderating interactions between climate change stressors and nutrient stressors in Scottish lochs, exemplified above for Loch Leven. For example, the lake phytoplankton models PROTECH and PCLake have been applied previously to Loch Leven allowing predictions of ecological responses beyond simple indicators of phytoplankton community biomass.

Figure A6.4. Long-term UKCP18 RCP Scenario 6.0 projections for seasonal air temperature (upper panel) and retention time (middle panel; monthly values >100 years removed for consistency with linear modelling approach) at Loch Leven between 2001 and 2080 for winter (Dec preceding year, Jan, Feb), spring (Mar, Apr, May), summer (Jun, Jul, Aug), autumn (Sep, Oct, Nov), annual (all calendar months), growing season (May to Sep). Monthly distributions of retention time across each decade from the 1980s to the 2070s from UKCP 18 RCP Scenario 6.0 for Loch Leven (lower panel; no correction of values > 100 years with the exception of 'infinity' values which were set to 1000).

Using PROTECH, Elliott et al. (2008) predicted that chlorophyll-*a* concentration in Loch Leven is controlled by P load from the catchment, but that the relative abundance of the dominant cyanobacteria species in Loch Leven, the nitrogen fixing *Anabaena* sp., will be highest when N load is low in the model. Elliott and Defew (2012) predicted that cyanobacteria abundance should increase with nutrient loading during storm events, but not in response to warming, whereas chlorophyll-*a* was predicted to be lowest at low retention time and highest at high retention time. The results on retention time are in general agreement with our linear models for spring and summer, discussed above. However, we also detected temperature effects that were not predicted in the PROTECH model. The PCLake model offers broader scope with respect to ecosystem scale responses across the food web, and with respect to sediment nutrient processes that are likely to be important in Loch Leven. Assessment of empirical models may inform the development of scenarios for process models to explore revised target setting, for example, to revise nutrient targets to avoid undesirable, or promote desirable, effects of climate change stressors. This approach should be considered in the context of a Climate Change Management Planning Framework for Scottish Lochs to inform the selection of potential management interventions based on the probability of their effectiveness, and based on agreed future climate change scenarios.

References

Birk, S., Chapman, D., Carvalho, L., Spears, B. M., Andersen. H.E., Argillier, C., Auer, S., Baattrup-Pedersen, A., Banin, L., Beklioglu, M., Bondar-Kunze, E., Borja, A., Branco, P., Tuba Bucak, T., Buijse, A.D., Cardoso, A. C., Couture, R-M., Cremona, F., de Zwart, D., Feld, C. K., Ferreira, M. T., Feuchtmayr, H., Gessner, M.O., Gieswein, A., Globevnik, L., Graeber, D., Graf, W., Gutiérrez-Cánovas, C., Hanganu, J., Iskın, U., Järvinen, M., Jeppesen, E., Kotamäki, N., Kuijper, M., Lemm, J. U., Lu, S., Lyche Solheim, A., Mischke, U., Moe, S. J., Nõges, P., Nõges, T., Ormerod, S. J., Panagopoulos, Y., Phillips, G., Posthuma, L., Pouso, S., Prudhomme, C., Rankinen, K., Rasmussen, J. J., Richardson, J., Sagouis, A., Santos, J. M., Schäfer, R. B., Rafaela Schinegger, Schmutz, S., C. Schneider, S. C., Schülting, L., Segurado, P., Stefanidis, K., Sures, B., Thackeray, S. J., Jarno Turunen, J., Uyarra, M. C., Venohr, M., von der Ohe, P. C., Willby, N. and Hering, D. (2020). Impacts of multiple stressors on freshwater biota across scales and ecosystems. *Nature Ecology and Evolution*. https://doi.org/10.1038/s41559-020-1216-4.

Carss, D., Spears, B. M., Quinn, L. and Cooper, R. (2012). Long-term variations in waterfowl populations in Loch Leven: identifying discontinuities between local and national trends. *Hydrobiologia*, 681, 85-104.

Carvalho, L., Miller, C., Spears, B. M., Gunn, I. D. M., Bennion, H., Kirika, A. and May, L. (2012). Water quality of Loch Leven: responses to enrichment, restoration and climate change. *Hydrobiologia*, 681, 35-47.

Carvalho, L., Solimini, A. G., Phillips, G., Pietiläinen, O-P., Moe, J., Cardosa, A. C., Solheim, A. L., Ott, O., Sondergaard, M., Tartari, G. and Rekolainen, S. (2009). Site-specific chlorophyll reference conditions for lakes in Northern and Western Europe. *Hydrobiologia*, 633, 59-66.

Castle, K., Frost, C. A. and Flint, D. F. (1999). The Loch Leven Project – Buffer strips in practice on a catchment scale. *Aspects of Applied Biology*, 54, 71-78.

Dudley, B., Gunn, I. D. M., Carvalho, L., Proctor, I., O'Hare, M. T., Murphy, K. J. and Milligan, A. (2012). Changes in aquatic macrophyte communities in Loch Leven: evidence of recovery from eutrophication? *Hydrobiologia*, 681, 49-57.

Elliott, J. A. and Defew, L. (2012). Modelling the response of phytoplankton in a shallow lake (Loch Leven, UK) to changes in lake retention time and water temperature. *Hydrobiologia*, 681, 105-116.

Elliott, J. A., Jones, I. D. and Thackeray, S. J. (2006). Testing the sensitivity of phytoplankton communities to changes. *Hydrobiologia*, 559, 401–411.

Elliott, J. A. and May, L. (2008). The sensitivity of phytoplankton in Loch Leven (UK) to changes in nutrient load and water temperature. *Freshwater Biology*, 53, 32-41.

Feld, C. K., Segurado, P. and Gutiérrez-Cánovas, C. (2016). Analysing the impact of multiple stressors in aquatic biomonitoring data: A 'cookbook' with applications in R. *Science of the Total Environment*, 573, 1320-1339.

Jones, I. D., Page, T., Elliott, J. A., Thackeray, S. J. T. and Heathwaite, A. L. (2011). Increases in lake phytoplankton biomass caused by future climate-driven changes to seasonal river flow. *Global Change Biology*, 17, 1809–1820.

LLCMP. (1999). Loch Leven Catchment Management Plan. Report of the Loch Leven Area Management Advisory Group; 93 pp.

May, L. and Carvalho, L. (2010). Maximum growing depth of macrophytes in Loch Leven, Scotland, United Kingdom, in relation to historical changes in estimated phosphorus loading. *Hydrobiologia*, 646, 123-131.

May, L., Defew, L., Bennion, H. and Spears, B. (2012). Historical changes (1905-2005) in the external phosphorus loads to Loch Leven, Scotland, UK. *Hydrobiologia*, 681, 11-12.

May, L., Moore, A., Woods, H., Bowes, M., Watt, J., Taylor, P. and Pickard, A. (2017). Loch Leven nutrient load and source apportionment study. Scottish Natural Heritage Commissioned Report No. 962; 65 pp.

May, L. and Spears, B. (2012). Loch Leven: 40 years of scientific research. Understanding the links between pollution, climate change and ecological response. *Development für Hydrobiologie*, 218, 130 pp.

O'Reilly, C. M., Sharma, S., Gray, D. K., Hampton, S. E., Read, J. S., Rowley, R. J., Schneider, P., Lenters, J. D., McIntyre, P. B., Kraemer, B. M., Weyhenmeyer, G. A., Straile, D., Dong, B., Adrian, R., Allan, M. G., Anneville, O., Arvola, L., Austin, J., Bailey, J. L., Baron, J. S., Brookes, J. D., de Eyto, E., Dokulil, M. T., Hamilton, D. P., Havens, K., Hetherington, A. L., Higgins, S. N., Hook, S., Izmest'eva, L. R., Joehnk, K. D., Kangur, K., Kasprzak, P., Kumagai, M., Kuusisto, E., Leshkevich, G., Livingstone, D. M., MacIntyre, S., May, L., Melack, J. M., Mueller-Navarra, D. C., Naumenko, M., Noges, P., Noges, T., North, R. P., Plisnier, P., Rigosi, A., Rimmer, A., Rogora, M., Rudstam, L. G., Rusak, J. A., Salmaso, N., Samal, N. R., Schindler, D. E., Schladow, S. G., Schmid, M., Schmidt, S. R., Silow, E., Evren Soylu, M., Teubner, K., Verburg, P., Voutilainen, A., Watkinson, A., Williamson, C. E., and Zhang, G. (2015). Rapid and highly variable warming of lake surface waters around the globe. *Geophysical Research Letters*, 42, 10773-10781.

Richardson, J., Feuchtmayr, H., Miller, C., Hunter, P. D., Maberly, S. C. and Carvalho, L. (2019). The response of cyanobacteria and phytoplankton abundance to warming, extreme rainfall events and nutrient enrichment. *Global Change Biology*, 25, 3365-3380.

Scheffer, M., Straile, D., van Nes, E. H. and Hosper, H. (2001). Climatic warming causes regime shifts in lake food webs. *Limnology & Oceanography*, 46(7), 1780–1783.

ScotInform. (2015). *Loch Leven Heritage Trail. Visitor Survey 2014/15*. Edinburgh (UK). ScotInform Ltd, 57 pp.

Scottish Natural Heritage. (2016). *The management plan for Loch Leven National Nature Reserve, 2016-2026*. Draft for Consultation. Scottish Natural Heritage, The Pier, Loch Leven, Kinross, UK.

Spears, B. M., Carvalho, L., Perkins, R., Kirika, A. and Paterson, D. M. (2012). Long-term variation and regulation of internal phosphorus loading in Loch Leven. *Hydrobiologia*. 681:23-33.

Spears, B. M., Chapman, D. S., Carvalho, L., Feld, C. K., Gessner, M. O., Piggott, J. J., Banin, L. F., Gutiérrez-Cánovas, C., Lyche Solheim, A., Richardson, J. A., Schinegger, R. Segurado, P., Thackeray, S. J. and Birk, S. (2021a). Making Waves. Bridging theory and practice towards multiple stressor management in freshwater ecosystems. Water Research, 196, 116981.

Spears, B. M., Chapman, D., Carvalho, L., Rankinen, K., Stefanidis, K., Ives, S., Vuorio, K. and Birk, S. (2022a). Assessing multiple stressor effects to inform climate change management responses in three European catchments. *Inland Waters*, DOI: 10.1080/20442041.2020.1827891.

Spears, B. M., Hamilton, D. P., Pan, Y., Zhaosheng, C.and May, L. (2022b). Lake management: is prevention better than cure? *Inland Waters*, DOI: 10.1080/20442041.2021.1895646.

7 Appendix 7: The potential for using remote sensing to monitor climate change impacts on water quality in Scottish standing waters

Remote sensing of inland water quality has evolved significantly over recent years to the point where the technology is now starting to be used operationally to monitor the status of waterbodies in the UK (i.e., UK Lakes Observatory: www.eo4ukwater.stir.ac.uk) and internationally (e.g., Copernicus Global Land Service: https://land.copernicus.eu/global/products/lwq). This progress towards operational use has been driven by a combination of factors, including innovations in satellite technology (i.e., Copernicus Sentinel satellites), the development of more sophisticated data processing routines, and advancements in data computation, storage, and web-based visualisation capabilities.

Historically, remote sensing of inland water quality has been hampered by the lack of satellite sensors with suitable technical specifications (i.e., spatial, spectral, radiometric and temporal resolutions). While ocean colour satellites such as Aqua-MODIS, Envisat MERIS, and its successor Sentinel-3 OLCI, can be used to monitor inland water quality, their coarse spatial resolutions (>250 m) mean they can only be used to observe the largest of waterbodies in Scotland (e.g., Loch Lomond). Terrestrial satellite missions such as the long-running Landsat programme provide data at a higher spatial resolution (30 m), but their spectral, radiometric, and temporal resolutions are generally unsuited to aquatic applications.

The MultiSpectral Imager (MSI) onboard the Copernicus Sentinel-2a satellite, launched in 2015, was arguably the first satellite sensor with technical specifications suitable for the operational monitoring of water quality in Scottish standing waters. Sentinel-2a was joined by its sister satellite Sentinel-2b in 2016, and these two platforms now operate in a constellation providing full coverage of the UK approximately every 3 days (cloud cover permitting) and at a spatial (pixel) resolution of 10-60 m. The Copernicus programme has an open and free data policy and, critically for climate applications, a commitment to provide long-term continuity of service beyond the current generation of satellites.

These technological developments have been accompanied by advancements in data processing routines including better methods for correcting atmospheric (Pahlevan et al., 2021) and land-adjacency effects (De Keukelaere et al., 2018) and new approaches for the intelligent selection of algorithms for retrieving water quality parameters such as chlorophyll-*a* (Neil et al., 2019; Spyrakos et al., 2018). The high spatial and temporal resolution of the Sentinel-2 MSI sensors means the mission produces very large volumes of data, but rapid advancements in high-performance computing (HPC) mean it is now feasible to process data at a national (or global) scale and provide the processed outputs to users with less than one day latency.

Sentinel-2 MSI, and comparable missions, can be used to estimate water quality parameters including turbidity (or Secchi depth) and chlorophyll-*a* concentration. These methods are now relatively mature, although there are issues still to be resolved with the estimation of chlorophyll concentrations in clearer waterbodies or those with high concentrations of coloured dissolved organic matter (CDOM). The retrieval of CDOM itself is also possible, but these methods are generally less reliable and transferable than for other parameters.

Lake surface temperature can also be derived from thermal satellite observations, albeit at high frequency only for very large lakes, and infrequently for smaller waterbodies. Satellite radar altimetry can also be used to estimate lake and river levels, although the approach is limited by the spatial and temporal coverage of the satellite ground tracks.

One of the main advantages of using satellite data to monitor water quality parameters such as chlorophyll-*a* is the ability to rapidly assess many hundreds of waterbodies and repeat that every few days. This means remote sensing can provide more frequent data (cloud cover permitting), for a far larger number of waterbodies, than can be achieved through *in situ* sampling alone. It also enables data to be collected for waterbodies that are rarely, if ever, sampled in the field. Remote sensing comes with the added advantage of allowing the same method(s) to be applied consistently and repeatedly across space and time, thus eliminating some potential sources of bias that can affect *in situ* sampling methods.

By means of example, the UK Lakes Observatory (UKLO) produces chlorophyll-*a* data from Sentinel-2a/b MSI data on a weekly timestep for approximately 900 lakes in the UK using a data processing chain adapted from Neil et al. (2019). The data are published on a web-based data visualisation platform. Also, the processing chain has been adapted and optimised for running on cloud-based HPC infrastructures, meaning it can optionally provide national-scale data at a daily timestep within 24 h of the satellite overpass.

Figure A7.1 shows an example of the data produced by the UKLO for Esthwaite Water in the Lake District illustrating how the increased frequency of data collection provided by the Sentinel-2 constellation allows for improved monitoring of the occurrence and duration of algal blooms. Importantly, this example also illustrates how remote sensing can be used to obtain data during periods when sampling in the field is not possible due to exceptional events, such as the Covid-19 pandemic.

The ability to generate internally consistent, and potentially long-term, water quality time-series from remote sensing holds significant potential for monitoring the effects of climate change on water quality in Scotland. The utility of satellite data for climate studies will continue to increase as the length of the observation record grows. Current data archives from the Sentinel-2 mission (5 years with two satellites) are arguably too short for the reliable detection of long-term climate change effects on water quality. But the data can already be used to explore, for example, more recent intra- or inter-annual trends, or the effects of climate extremes such as heatwaves and drought on the proliferation of algal blooms. The recurrence of algal blooms (rather than the quantitative estimation of chlorophyll-*a*) can also be identified using Landsat data which does permit analyses to be extended back to the mid-1980s (e.g., Ho et al., 2017).

The ability to proactively monitor water quality in near real-time, and at scale, can provide useful triggers for additional confirmatory in situ sampling and, potentially, for early intervention measures. In this respect, when used in conjunction with more conventional *in situ* sampling, not only can remote sensing provide data to improve our understanding of climate change effects on water quality, but it can also improve our to resilience to the effects of climate change particularly where risks are posed to drinking water, fisheries and human health.

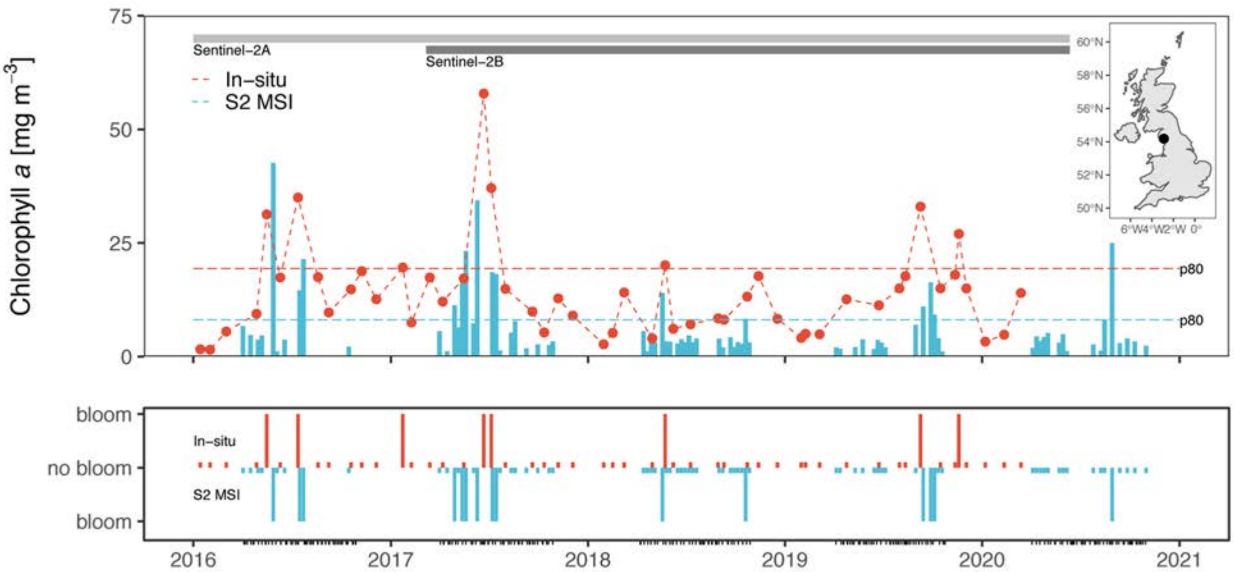


Figure A7.1. Time-series of chlorophyll-*a* concentrations for Esthwaite Water (UK WBID: 29328) derived from Sentinel-2a and Sentinel-2b observations processed through the UK Lakes Observatory compared to monthly *in situ* monitoring data (obtained from the UK Lakes Portal). Data collected between December and February have been removed. The lower panel shows the number and timing of bloom events (defined by the exceedance of the 80th percentile) identified in each time series. *© UKCEH. Contains Environment Agency water quality data from the Water Quality Archive (Beta). All rights reserved.*

References

De Keukelaere, L., Sterckx, S., Adriaensen, S., Knaeps, E., Reusen, I., Giardino, C., Bresciani, M., Hunter, P., Neil, C., Van Der Zande, D. and Vaiciute, D. (2018). Atmospheric correction of Landsat-8/OLI and Sentinel-2/MSI data using iCOR algorithm: validation for coastal and inland waters. *European Journal of Remote Sensing*, 51, 525–542. doi:10.1080/22797254.2018.1457937.

Ho, J. C., Stumpf, R. P., Bridgeman, T. B. and Michalak, A. M. (2017). Using Landsat to extend the historical record of lacustrine phytoplankton blooms: A Lake Erie case study. *Remote Sensing of Environment*, 191, 273–285. doi:10.1016/j.rse.2016.12.013

Neil, C., Spyrakos, E., Hunter, P. D. and Tyler, A. N. (2019). A global approach for chlorophyll-*a* retrieval across optically complex inland waters based on optical water types. *Remote Sensing of Environment*, 229, 159–178. doi:10.1016/j.rse.2019.04.027

Pahlevan, N., Mangin, A., Balasubramanian, S. V., Smith, B., Alikas, K., Arai, K., Barbosa, C., Bélanger, S., Binding, C., Bresciani, M., Giardino, C., Gurlin, D., Fan, Y., Harmel, T., Hunter, P., Ishikaza, J., Kratzer, S., Lehmann, M. K., Ligi, M., Ma, R., Martin-Lauzer, F. R., Olmanson, L., Oppelt, N., Pan, Y., Peters, S., Reynaud, N., Sander De Carvalho, L. A., Simis, S., Spyrakos, E., Steinmetz, F., Stelzer, K., Sterckx, S., Tormos, T., Tyler, A., Vanhellemont, Q. and Warren, M. (2021). ACIX-Aqua: A global assessment of atmospheric correction methods for Landsat-8 and Sentinel-2 over lakes, rivers, and coastal waters. *Remote Sensing of Environment*, 258, 112366. doi:10.1016/j.rse.2021.112366.

Spyrakos, E., O'Donnell, R., Hunter, P. D., Miller, C., Scott, M., Simis, S. G. H., Neil, C., Barbosa, C. C. F., Binding, C. E., Bradt, S., Bresciani, M., Dall'Olmo, G., Giardino, C., Gitelson, A. A., Kutser, T., Li, L., Matsushita, B., Martinez-Vicente, V., Matthews, M. W., Ogashawara, I., Ruiz-Verdú, A., Schalles, J. F., Tebbs, E., Zhang, Y. and Tyler, A. N. (2018). Optical types of inland and coastal waters. *Limnology and Oceanography*, 63, 846–870. doi:10.1002/lno.10674

CREW CENTRE OF EXPERTISE FOR WATERS

CREW Facilitation Team

Hydro Nation International Centre

James Hutton Institute

Craigiebuckler

Aberdeen AB15 8QH

Scotland UK

Tel: +44 (0)344 928 5428

Email: enquiries@crew.ac.uk

www.crew.ac.uk

Hydro Nation International Centre

The James Hutton Institute

Scottish Government
Riaghaltas na h-Alba
gov.scot

CREW is a partnership between the James Hutton Institute and all Scottish Higher Education Institutes and Research Institutes. The Centre is funded by the Scottish Government.