Mitigating Climate Change Impacts on the Water Quality of Scottish Standing Waters

Linda May, Miriam Glendell, Kerr Adams, Zisis Gagkas, Lily Gouldsbrough, Iain Gunn, Megan Hannah, Michaela Roberts, Bryan Spears, Philip Taylor, Stephen Thackeray, Mads Troldborg, Erica Zaja

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Report and Appendices

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CENTRE OF EXPERTISE FOR WATERS

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Glossary/Acronyms

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.

Algal bloom: High density of phytoplankton (algae plus cyanobacteria) in, or floating on the surface of, a standing water.

Alkalinity: Buffering capacity of a water body; a measure of its ability to maintain a fairly stable pH level.

BBN: Bayesian Belief Networks are probabilistic models that can define relationships between variables and be used to calculate probabilities.

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, colloquially known as blue-green algae (BGA), which may form visible surface blooms in the water column or shoreline scums when present in high concentrations.

Eutrophication: Process of nutrient enrichment in aquatic ecosystems.

Evapo-transpiration: 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.

INNS: Invasive Non-Native Species

Lake/Loch: Area of standing water surrounded by land; for the purposes of this project, this includes lochs, reservoirs and locally important standing waters greater than 1 ha in area.

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.

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.

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 indicates 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 or soluble reactive phosphorus; 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.

SSPs: Stakeholder-elaborated Shared Socioeconomic Pathways representing alternative socio-economic trajectories). Within this, SSP1 represents a sustainable and co-operative society with a low carbon economy and high capacity to adapt to climate change; in contrast SSP3 represents a future where food production dominates land use and social and economic barriers lead to a highly fragmented society with limited capacity to adapt to climate change.

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

Thermal stratification: Vertical change in water temperature (and density) with depth in a standing water; this creates three distinct layers – the epilimnion (upper warm layer); the metalimnion (middle layer); and the hypolimnion (cool bottom layer).

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

TP: Total phosphorus is the total amount of phosphorus in an environmental system or one of its components.

TRL: Technology Readiness Levels are a measurement system used to assess the maturity level of a technology or innovation.

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

WHO: World Health Organisation.

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

Executive Summary

Purpose of research

The aim of this project was to inform fit for purpose strategies to mitigate the effects of climate change on Scottish standing waters. The study aimed to answer the following questions:

- To what extent can changes in catchment management practices and in-lake processes successfully mitigate climate change impacts on Scottish standing waters?
- Does existing water policy, and its implementation, sufficiently account for climate change impacts on the quality of Scottish standing waters?
- What changes are required under current and projected climate change scenarios for adaptive management responses to be put in place?
- What recommendations, priorities for action and practical mitigation measures/solutions can we implement in the short term (≤ 5yrs) and long term (> 5yrs)?

Background

There is a policy focus at national and international levels on mitigating climate change impacts by reducing carbon emissions and increasing carbon sequestration. However, even if we can slow climate change down, we cannot prevent or reverse it. So, alternative approaches must be used to lessen its effects. These include adaptive interventions that increase the resilience, and reduce the vulnerability, of people and nature to weather extremes and other climate change impacts. A recent CREW report by May *et al*. (2022a,b) found that Scottish standing waters are already warming at an alarming rate and are projected to continue warming into the future. This is likely to cause more frequent and/or more intense algal blooms unless measures to reduce their growth are put in place. Since we cannot cool our water bodies, and increasing their flushing rates is unlikely to be a widely applicable solution, this study has been exploring what else can be done to reduce the likelihood of algal blooms worsening under future climate change.

Key findings

- The main causes of poor water quality, especially algal blooms, in Scottish standing waters are high phosphorus concentrations and periods of low rainfall and warm weather.
- The cost of algal blooms to the Scottish economy, based on very limited data, was estimated to be at least £16.5 million per year, excluding the medical and veterinary costs incurred when they affect the health of people, pets and livestock; however, a detailed site specific study at Loch Leven (£2 million per year) suggests that this figure is likely to be much higher.
- Less than one percent of the 6,836 standing waters included in this study receive effluent from waste water treatment works; runoff from land is the main source of phosphorus entering these systems.
- The equivalent cost to treat water to reduce phosphorus run off before it enters our lochs is estimated to be about £56.4 million per year.
- Phosphorus laden runoff into standing waters can be reduced by adopting more sustainable land use practices, such as using less fertiliser and maintaining soil nutrient status at or below the agronomic optimum. This would almost halve phosphorus losses from land to water, whereas increasing the extent of buffer strips would only reduce these losses by about 1%.
- Adopting sustainable land use practices and achieving low greenhouse gas emissions would result in a c. 20% reduction in phosphorus runoff by 2080 compared to present-day conditions; this would improve water quality in c. 85% of Scotland's standing waters.
- In contrast, adopting high intensity farming practices and not achieving low greenhouse gas emissions will more than double phosphorus runoff, resulting in only about ⅓ of our standing waters having high quality water by 2080.
- Within lake measures can contribute to nutrient reduction. However, evidence on the effectiveness of implementing in-lake measures to directly reduce the impacts of climate change, including those related to warming and extreme weather events, is limited. To avoid unintended consequences, in-lake measures

should be selected based on a detailed site-specific assessment.

- Where future habitat degradation is projected to exceed the tolerances of species of high conservation concern, in-lake habitat improvement or species translocation programmes may be necessary to avoid local or national extinctions.
- Most (93%) of stakeholders interviewed for this project expressed concern about the impacts of climate change on Scottish standing waters, with increases in algal blooms (79%), eutrophication caused by nutrient runoff (71%), increased storm events and flooding (43%) and high soil erosion rates (43%) being the main issues mentioned.
- In the short-term (≤ 5 years), reducing diffuse pollution from farmlands (43%), engaging with farmers to enable change to more sustainable practices (36%), incentivising and monitoring actions by farmers (29%), engaging with land managers and landowners (29%), and increasing the number and/or the width of buffer strips (29%) were identified as being the main measures that need to be put in place.
- In the long-term (> 5 years), geoengineering or innovative solutions (29%), re-evaluating the suitability of existing controls (29%), reducing runoff from land (29%), introducing more prescriptive land management guidance (21%) and having an agricultural reform (21%) were identified as being the most important mitigation measures to put in place.
- Most interviewees agreed that changes to land and water policy would be needed to ensure effective implementation of mitigation measures (86%) and many (29%) mentioned the need for an holistic and balanced approach to tackling climate change impacts on Scottish standing waters.

Recommendations

1. There is a need to improve estimates of the financial impacts of algal blooms on Scotland by collecting better datasets at national scale; algal blooms affect the health and welfare of people, and their pets and livestock, so the costs of any associated medical or veterinary treatment should be included. More accurate water treatment costs, and costs of removing phosphorus from runoff, also need to be added.
2. Successful mitigation of climate change impacts on Scottish standing waters should be focused on adaptive land management aimed at reducing nutrient inputs, especially of phosphorus.
3. Sensitivity to changes in climate and land use varies regionally and by waterbody type; River Basin Management Planning needs to reflect this.
4. Where land use changes cannot fully mitigate the effects of climate change on standing waters, within waterbody measures should be considered; these need to be assessed carefully before they are implemented to check their viability and reduce any risk of unintended consequences.
5. Site specific monitoring programmes need to be modified to inform the development of comprehensive management plans within a robust restoration/adaptation framework; monitoring of nutrient inputs to standing waters should be included.
6. A national roadmap for adapting to, or mitigating, the effects of climate change on standing waters needs to be included in Scotland's National Adaptation Planning.
7. Where future habitat degradation is projected to exceed the tolerances of species of high conservation concern, in-lake habitat improvements or species translocation programmes will be needed to avoid local or national extinctions.
8. Changes to land and water policy will be needed to ensure effective implementation of mitigation measures to reduce diffuse pollution as part of a holistic and balanced approach to tackling climate change impacts on Scottish standing waters.
9. Future research should be focused on developing a readily accessible Toolkit to help water managers and landowners make evidence based decisions on how best to reduce climate change impacts on standing waters; this should be in the form of a decision support tool.

1 Introduction

1.1. Background and scope

There is a policy focus at national and international levels on mitigating climate change impacts by reducing carbon emissions and increasing carbon sequestration. However, even if we can slow climate change down, we cannot prevent or reverse it. So, alternative approaches must be used to lessen its effects. These include adaptive interventions that increase the resilience, and reduce the vulnerability, of people and nature to weather extremes and other climate change impacts (Scottish Government, 2018).

In their report on the most up-to-date evidence of climate change trends observed in the UK, the UK Climate Change Committee (2021) indicated that the most likely changes to the UK climate by 2050 would be warmer and wetter winters, and hotter and drier summers. Amongst other impacts, these increases in temperature and changes in hydrology are likely to have adverse effects on the quality of Scotland's standing waters. In a recent CREW report by May *et al*. (2022a,b), these were summarised as:

- Increased risk of phytoplankton (algal) blooms, driven by increases in air temperatures and changes in rainfall patterns.
- An associated increase in the risk of potentially harmful toxins being released into the water by cyanobacteria – often known as harmful algal blooms (HABs).

May *et al*. (2022a) concluded that water policy and existing monitoring networks would need to be adapted as part of Scotland's strategic and coordinated response to the climate crisis. They suggested:

- Closing the policy gap between global and national understanding of the impacts of climate change on air temperatures and rainfall patterns.
- Adapting water policy and management practices (e.g. River Basin Management Planning; Third Land Use Strategy) to enable national-scale climate driven risks to be taken into account at regional to catchment scale.
- Combining current nutrient status criteria for Scottish standing waters with other policy-based and nature-based solutions to support required legislative outcomes under different climate scenarios.
- Monitoring key indicators of climate-related risks to inform decisions on adaptive water policy and management practices.

Of particular interest are the effects of climate change on catchment and in-lake processes, such as nutrient inputs and biological uptake, temperature regimes and flushing rates. Individually, or in combination, changes in these factors are likely to increase the risk of algal blooms under climate change conditions, thereby reducing Scotland's ability to meet statutory goals and regulatory targets within the timelines required. However, secondary impacts may also be important. For example, Moore and Cole (2022) suggested that the Scottish Environment Protection Agency (SEPA) should integrate river flow forecasting into reservoir operating procedures to reduce flood risks. However, this could increase the water retention time of reservoirs, causing unintended consequences up- and down-stream, such as increased algal blooms. Other climate change effects that will affect the water retention time of standing waters, thereby increasing the risks of Harmful Algal Blooms (HABs), include more direct changes linked to variations in patterns of rainfall. For example, Boca, White and Bertram (2022) noted that river flows are likely to fall in spring and summer, and increase in autumn and winter, with these changes varying regionally and over time. Some areas – such as eastern and southern Scotland – are predicted to see an overall reduction in flows.

1.2 Project objectives

There is now an urgent need for fit for purpose mitigation/adaptation strategies to be developed and implemented to safeguard the integrity, biodiversity, and sustainable use of our standing waters. To do this, further research was required to identify catchment interventions and within waterbody measures that, separately or in combination, can be used to mitigate the effects of climate change on these water bodies.

In Phase 1 of this research, May *et al*. (2022a,b) showed that climate change will increase the temperatures of standing waters across Scotland over the next 40 years, which will increase the risk of algal blooms, and reduce amenity value and wildlife habitat. This study builds on the outcomes of that report by identifying and prioritising changes in land management practices, and reviewing potential in-lake options, that will reduce water quality problems and maximise the success of climate mitigation.

1.3 Structure of the report

This project addresses the following questions:

- To what extent can changes in catchment management practices and in-lake processes successfully mitigate climate change impacts on the quality of Scottish standing waters?
- Does existing water policy, and its implementation, sufficiently account for climate change impacts on the water quality of Scottish standing waters?
- What changes are required under current and projected climate change scenarios for the required adaptive management responses to be put in place?
- What recommendations, priorities for action and practical mitigation measures/solutions can we implement in the short term (≤ 5yrs) and in the longer term (> 5yrs)?

2 Research undertaken

This report focuses on the main drivers of water quality problems in Scottish standing waters, how these will be affected by climate change and how those impacts can be reduced. These main drivers have been identified as total phosphorus (TP) inputs, flushing rates and water temperature (Carvalho *et al*., 2013). The waterbody response to these changes that causes the most damaging degradations in water quality, either directly or indirectly, is the amount of phytoplankton in the water. High levels of phytoplankton are commonly referred to as ‘algal blooms’ even though they are a mixture of algae and blue-green (photosynthesising) bacteria. We use the terms ‘algae’ and ‘algal blooms’ to mean phytoplankton more generally throughout this report.

Climate change related projections of water quality were derived from linked catchment input and waterbody response models. These have been used to explore how different management scenarios could, potentially, reduce the projected increase in ‘algal blooms’ that would otherwise occur by 2080, under climate change.

A number of climate change, socioeconomic and land use scenarios were tested for their potential impact on in-lake total P (TP) concentrations and on the likelihood of cyanobacterial blooms exceeding World Health Organisation (WHO) thresholds for safe use (WHO 2003, 2004, 2021). Models developed from SEPA Water Framework Directive (WFD) data for 142 monitored lochs were used to project water quality for 6,836 standing waters across Scotland under the different combinations of change scenarios outlined below. Risk of failing WFD water quality status targets was also assessed based on the 317 standing waters for which these regulatory targets were available.

2.1 Climate change scenarios

There are many climate warming projections available, each based on different future levels of greenhouse gas emissions (Figure 1). In this study, we focused on the potential effects of two of these scenarios, i.e. RCP2.6 (about 1.5 degrees centigrade of warming by 2080) and RCP6.0 (about 3 degrees centigrade of warming by 2080). Given that we already have an average global rise in temperature of almost 1.5°C, RCP6.0 probably reflects a future that is close to a status quo or ‘do nothing’ scenario.

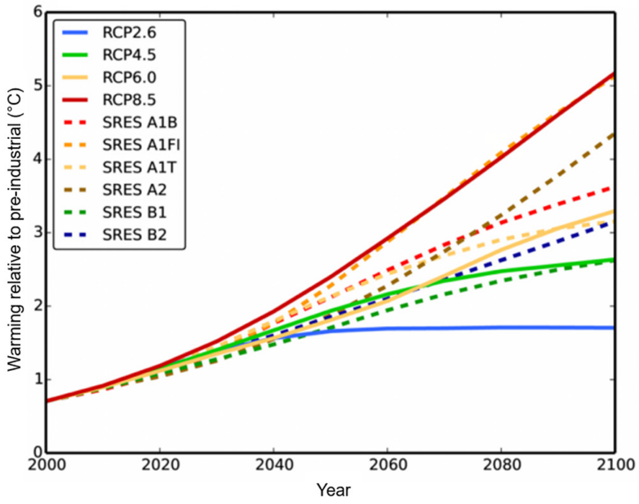

Figure 1: Global mean temperature (°C) projections from a climate model (MAGICC6) relative to a pre-industrial average (1850-1900) for RCP2.6 (blue), RCP4.5 (green), RCP6.0 (yellow) and RCP8.5 (red) (IPCC Special Report on Emissions Scenarios (SRES)); the older SRES emissionsscenarios (dashed coloured lines) are also shown (after MET Office, 2018).

2.2 Socioeconomic scenarios

Future land use within lake catchments was derived from the CRAFTY-GB land use scenarios published by Brown *et al*. (2022) for 2040, 2060 and 2080. These scenarios were used to investigate how terrestrial P losses are influenced by future land use change based on the following Shared Socioeconomic Pathways (SSPs):

- **SSP1 – Sustainability**, which represents a sustainable and co-operative society with a low carbon economy and high capacity to adapt to climate change. From a land use perspective, this scenario predicts areas of intensive agriculture decreasing over time.
- **SSP3 – Regional rivalry**, which represents increasing social and economic barriers that may trigger international tensions, nationalisation of key economic sectors, job losses and, eventually, a highly fragmented society. From a land use perspective, this scenario predicts that food production will dominate land uses, with other ecosystem services being achieved as by-products of enforced low-intensity management.

2.3 Mitigation measure scenarios

Several potential land use scenarios were run using the models developed to explore their potential to reduce TP inputs to standing waters, thereby improving water quality and reducing the risk of cyanobacterial blooms exceeding WHO thresholds for safe use (WHO 2003, 2004, 2021: Carvalho *et al*. 2013). These were compared using different land management scenarios applied to 2020, only:

- S1: Baseline in 2020 based on observed values
- S2: Fertiliser application rate below agronomic optimum
- S3: Fertiliser application rate at agronomic optimum
- S4: Increase in extent of buffer strips
- S5: Maximum mitigation (fertilisers below agronomic optimum and increased buffer strips)

2.4 Catchment delivery model

Catchment nutrient delivery models were used to examine the effects of current levels of nutrient runoff on standing water quality, and the effects of future combinations of climate and land use change scenarios (Table 1). The models focused on the effects of varying TP inputs from land to standing waters, because TP has been identified as the main factor affecting the likelihood of 'algal blooms' developing in these systems. The models were also used to investigate the potential effectiveness of mitigation strategies listed in Section 2.3 in terms of reducing P runoff and, consequently, the risk of algal blooms developing under a range of climate change scenarios.

Notably, only 95 of the 6,836 standing waters included in this study were affected by discharges from wastewater treatment works (Figure 2). Most standing waters that had high TP concentrations were not within these areas, indicating that most high TP inputs were coming from diffuse sources such as agricultural runoff.

Table 1 Combined climate change and shared socio-economic pathway scenarios tested.

Shared Socioeconomic Pathways	Climate change scenarios	
	RCP2.6	RCP6.0
SSP1: Sustainability	2020, 2040, 2060, 2080	2020, 2040, 2060, 2080
SSP3: Regional rivalry	2020, 2040, 2060, 2080	2020, 2040, 2060, 2080

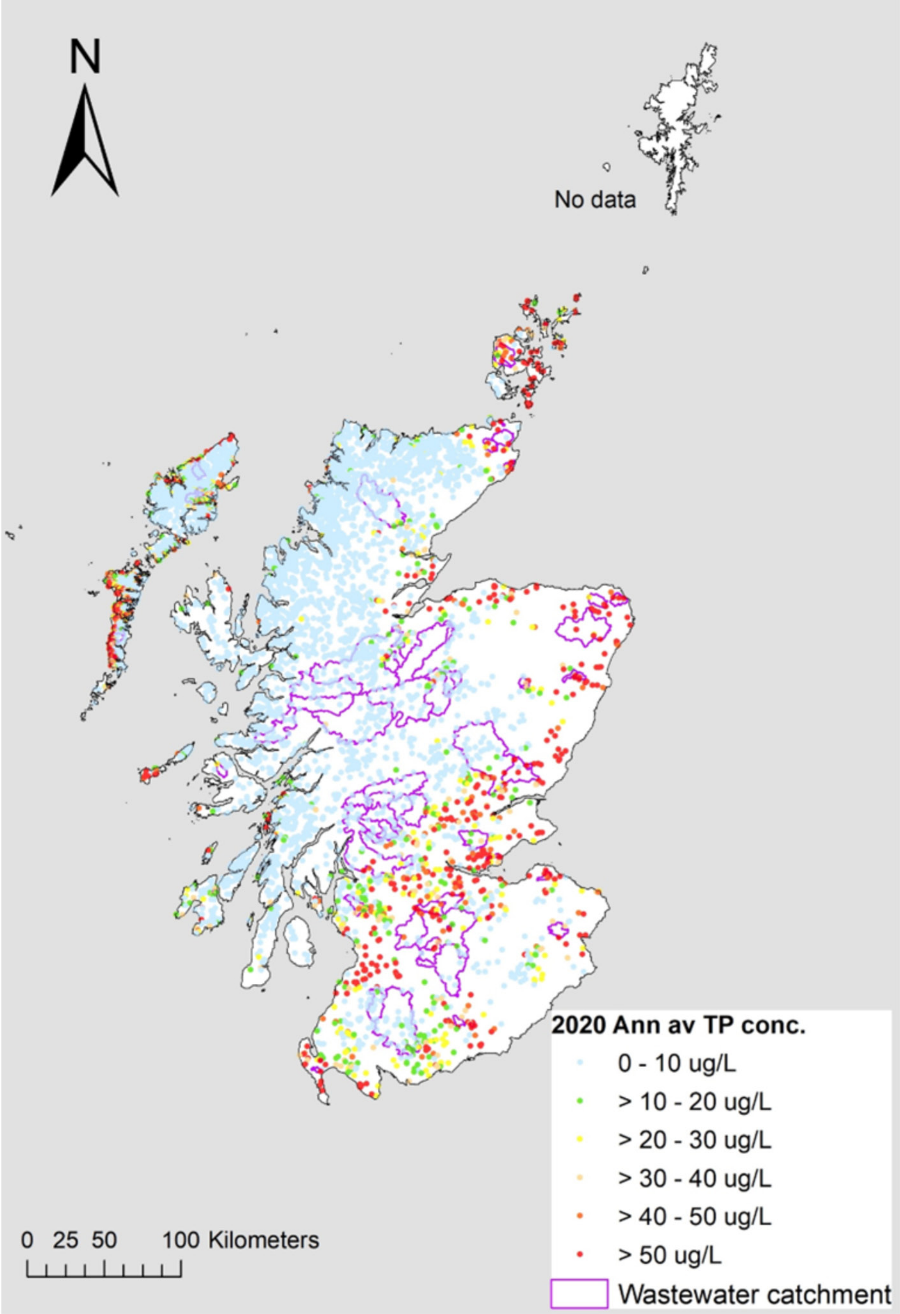

Figure 2 Annual average TP concentrations in standing waters in 2020 showing catchments affected by discharges from wastewater treatment works.

2.5 Climate change scenario testing

The key drivers of climate change impacts on standing waters, such as changes in air temperature and hydrologically effective rainfall (runoff), were incorporated into the waterbody and catchment models. Once complete, the linked catchment and standing water models were run under different climate change scenarios for the period 2020 to 2080, and under different land use change scenarios, to determine which of these would be most effective at reducing algal blooms in the future.

2.6 The costs of algal blooms

Within Scotland, standing freshwater provides people and nature with water supply, recreational facilities, and habitat for wildlife. We used a rapid evidence review approach to investigate the potential costs of algal blooms to the Scottish economy. Where no robust data could be sourced for Scotland specifically, we used results for England and Wales published by Jones *et al*. (2020). Data from a more detailed study at Loch Leven was also reviewed. All costs are presented as 2023 equivalent values.

2.7 Assessment of the potential for using in-lake mitigation measures to reduce climate change impacts

A systematic review of available literature and expert knowledge was used to investigate whether within waterbody measures can be used to control the release and recycling of P from the sediments of standing waters, because this can delay recovery when external inputs of nutrients are reduced. Evidence on the efficacy of such approaches was summarised for each type of in-lake mitigation measure available.

2.8 Stakeholder engagement

Individual stakeholder engagement interviews were held between December 2023 and January 2024 to obtain interviewee responses to the following questions:

Q1: Are you concerned about the impacts of climate change on Scottish standing waters? Why?

Q2: What do you perceive to be the main water quality issues caused by climat change?

Q3: Who or what are the water quality issues likely to affect?

Q4: What are the main measures that could be implemented to mitigate water quality issues in the short term (≤ 5yrs) and in the long term (> 5yrs)?

Q5: Would changes to land and water policy be needed to ensure the effective implementation of these measures? What changes would you suggest?

Q6: Do you have any other suggestions to mitigate climate change impacts on Scottish standing waters?

Of the 24 stakeholders contacted, we received 14 replies; this equated to a response rate of 58%. During the interview, participants were invited to suggest how the guidance for land managers could be modified to ensure the future resilience of Scottish standing waters to climate change.

3 Findings

'Algal blooms' degrade the water quality, and reduce the amenity and habitat value, of Scottish standing waters. Algal blooms are expected to increase in frequency, size and severity under climate change unless measures are put in place to control them. In this section, we explore potential solutions to this problem.

3.1 What are the current costs of 'algal blooms' in standing waters?

The World Health Organisation (WHO) has set water quality thresholds for safe use (WHO 2003, 2004, 2021) that are based on levels of potentially toxic cyanobacteria in water. When these are exceeded, water supplies need additional treatment to prevent filters clogging and to remove toxins. Also, recreational facilities must be closed to users until the problem has been resolved (Codd *et al*. 2005). Increased water treatment costs incurred, and income lost due to restrictions on recreational use, have economic consequences for local businesses. There are also wider impacts, including detrimental effects on property values, and human and animal health implications (illness; death) associated with cyanobacterial concentrations that exceed WHO thresholds for safe use (WHO 203, 2004, 2021).

The current overall cost of algal blooms in Scotland, in terms of water treatment costs, impacts on house prices and reductions in visitor numbers, was estimated to be about £16.5 million per year. However, this should be taken as a lower estimate because many elements could not be costed. For example, this figure excludes many unknown factors, such as medical and veterinary costs for people and animals whose health is affected by these blooms. Also, given that the cost of an algal bloom at Loch Leven alone has been estimated to be about £2 million per year, the total figure for all lochs in Scotland is likely to be much higher than the estimate above. The cost of removing the phosphorus that causes algal blooms from runoff is estimated to be about £56.4 million per year.

'Algal blooms' also lead to increases in greenhouse gases emissions from affected waterbodies. The costs of these emissions to England and Wales were estimated at £8.9m – £14m per year, based on the combined impacts of climate, health, sea-level rise, water availability, biodiversity and natural disasters (Pretty *et al*., 2003). When extrapolated to Scotland, these costs were estimated at £0.6m per year (Jones *et al*., 2020). Based on estimates obtained from Biodiversity Action Plans in England and Wales, the costs of 'algal blooms' to biodiversity management in Scotland have been estimated to be about £0.7m per year (Jones *et al*., 2020).

However, many of these values do not estimate the actual impacts of algal blooms in Scotland; for greater accuracy in these estimates, these costs need to be updated with Scottish data when they become available. Under climate change, 'algal blooms' are expected to become more common with associated increases in costs to the Scottish economy unless effective mitigation strategies are put in place.

3.2 Can catchment interventions reduce nutrient inputs to standing waters?

Targeted and extensive land-based mitigation measures focused on maintaining soil nutrient status at, or below, the agronomic optimum were found to reduce TP inputs to standing waters from their catchments by up to 46%. This shows that holistic management of soils to maximise soil organic matter content and nutrient use efficiency would reduce TP losses. To implement these changes, regular soil testing and optimisation of fertiliser application rates is essential. In contrast, smaller-scale interventions, such as buffer strips, did not affect TP losses to water very much at a catchment scale.

In terms of projecting the impacts of climate and land use change into the future, our study shows that lifestyle choices will greatly affect the impact of climate change on our standing waters by 2080. For example, following a low emissions strategy that leads to a 1.5°C rise in temperature (i.e. greenhouse gas emissions going to net zero by 2100) combined with a Shared Socioeconomic Pathway focused on sustainable land use (SSP1; Sustainability) would lead to TP losses from land to water decreasing by c. 20% between 2020 and 2080. In contrast, a temperature increase of 3°C (RCP6.0) combined with a Shared Socioeconomic Pathway focused on the expansion of arable land and intensification of agriculture (SSP3; Regional rivalry) would result in projected TP losses from land to water increasing by *c.* 138% between 2020 and 2080. The latter would greatly increase the risk of algal blooms. These results demonstrate that extensive adoption of sustainable agronomic practices, such as those promoted under the Net Zero targets, coupled with future lower emissions pathways are

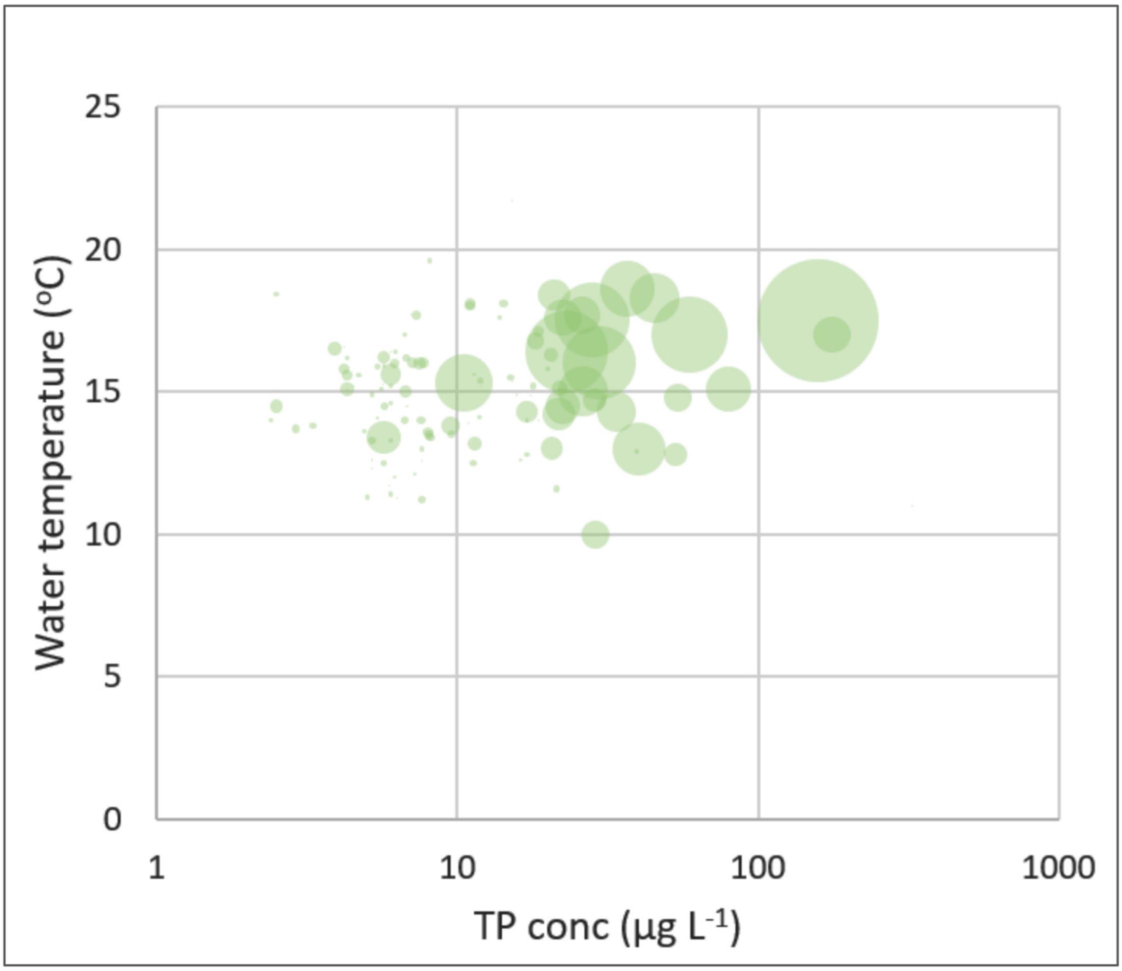

Figure 3 Relationship between water temperature, TP concentration and the amount (biovolume) of cyanobacteria is proportional to area of bubble in Scottish standing waters, 2009 – 2012. Scale: Maximum value shown = 0.09 mm^3 L^{-1}

needed to reduce eutrophication risks to Scottish standing waters in the future. In contrast, pursuing higher emissions pathways coupled with land use intensification will lead to a significant increase in TP inputs to standing waters with an associated increase in eutrophication problems.

3.3 Will reducing nutrient losses to standing waters maintain or improve their quality?

Under certain conditions, phytoplankton grow very quickly in standing waters and accumulate to form what is commonly known as an 'algal bloom'. These blooms are not just algae; they also contain cyanobacteria, also known as blue-green algae. Together, the quantities of algae and cyanobacteria in the water can be estimated by the amount of chlorophyll-*a* they contain.

'Algal blooms' reduce the amenity value of standing waters and the quality of freshwater habitats for wildlife, so it is important that they are kept to a minimum. To mitigate the impacts of climate change on Scottish standing waters, we need to understand the links between TP availability, water temperature and flushing rate, which are key drivers of 'algal blooms' in these systems. For example, Figure 3 illustrates how, for a given water temperature, the amount of cyanobacteria in standing waters increases with higher TP concentrations.

In this study, we explored the potential impacts of two contrasting future climate and land use change projections for the 6,836 standing waters across Scotland for which we had sufficient data. Of the climate change x land use scenarios tested (Table 1), the most extreme scenario (RCP6 x SSP3) generated a much higher risk of cyanobacterial blooms in a larger number of standing waters than the less extreme scenario (RCP2.6 x SSP1). In contrast, reducing TP losses by keeping fertiliser application rates below the agronomic optimum led to lower chlorophyll-*a* concentrations in many standing waters.

A summary of changes in TP concentrations, water temperature, water retention time and risk of

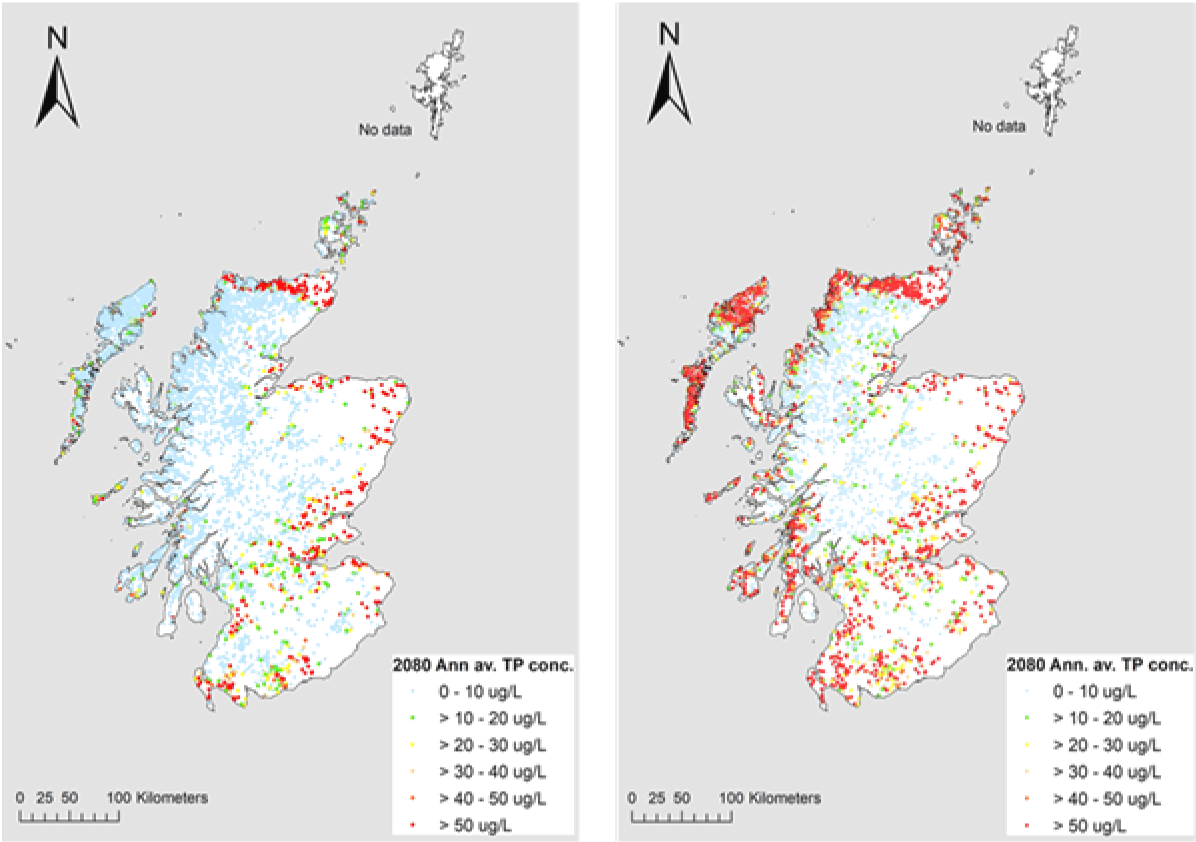

Figure 4 Projected TP concentrations of Scottish standing waters by 2080 under a low GHG emissions and sustainable land use future (RCP2.6 x SSP1; left panel) compared to a future characterised by high GHG emissions and an increase is intensive agriculture (RCP6.0 x SSP3; right panel).

Table 2 Percentage of standing waters showing an increase in various indicators of water quality status between 2020 and 2080.

Waterbody response	RCP2.6 x SSP1	RCP6.0 x SSP3
Increase in TP concentration	74%	87%
Increase in water temperature	99%	99%
Increase in water retention time	95%	37%
Increase in number of sites with >40% chance of exceeding WHO water quality thresholds for safe use	0%	16%
Increase in failures to meet Good WFD status or above	-7%	36%
Increase in failures to meet High WFD status	-11%	44%

exceedance of WHO water quality thresholds for safe use (WHO 2003, 2004, 2021) that are projected to occur between 2020 and 2080 are shown in Table 2. Changes in the number of standing waters that are likely to fail WFD targets for High status, or Good status and above, are also shown.

The projected changes in these values across Scotland under the two different climate change projections and shared socio-economic pathways tested indicated an improvement in water quality across the central belt under RCP2.6 x SSP1, but a noticeable degradation in water quality across the north of mainland Scotland, between 2020 and 2080 (Figure 4). In contrast, a marked increase in projected TP concentrations in standing waters across the south and west of the country was found under the RCP6 x SSP3 scenario over this timescale. The high TP concentrations in lochs across the north of mainland Scotland and the western isles is caused by the projected change from peatland to arable land associated under scenario SSP3, which includes a projected change towards food production dominated land uses. The overall change in TP concentration, water temperature or water retention time between 2020 and 2080, in terms of percentage of waterbodies affected, is shown in Table 2.

In terms of amenity value, these projected TP concentrations were converted to likelihood of exceedance of WHO water quality thresholds

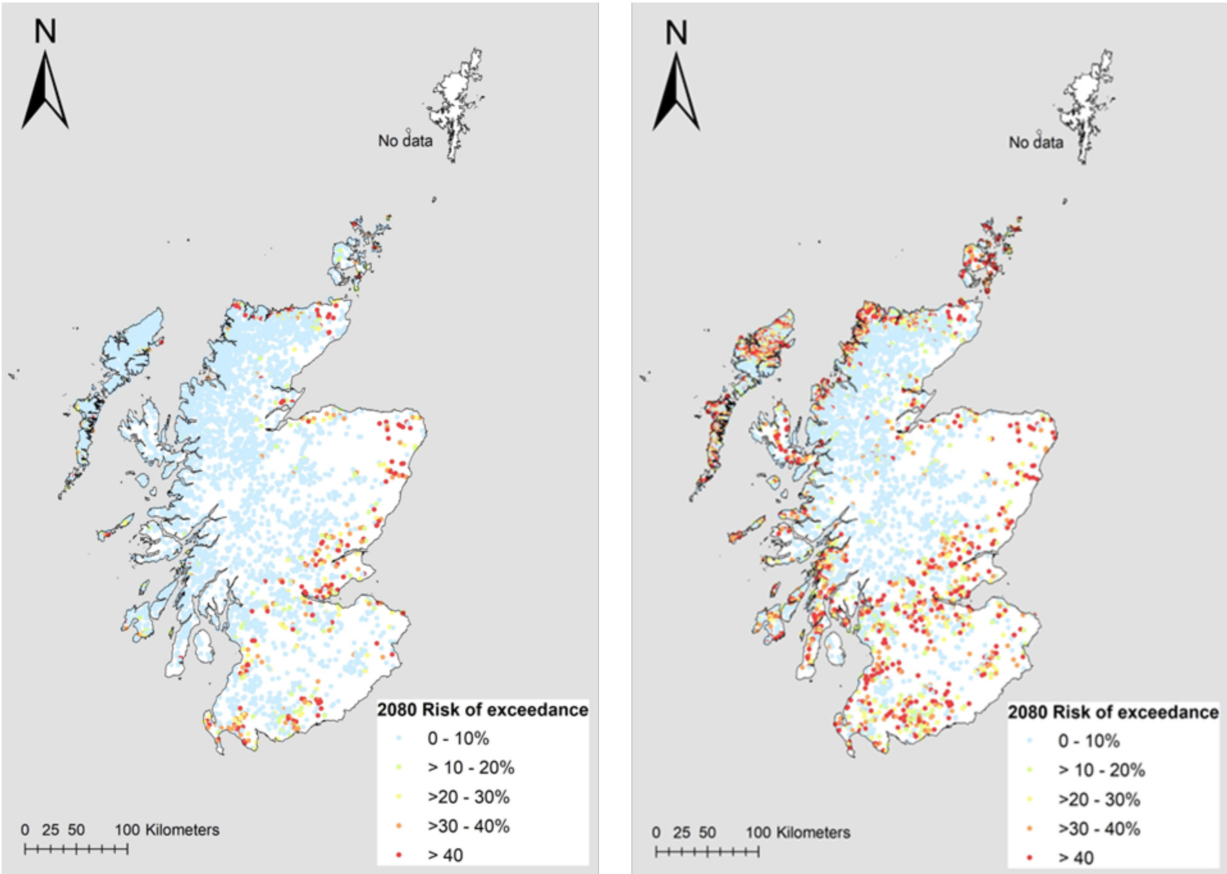

Figure 5 Comparison of the number and distribution of Scottish standing waters projected to exceed WHO water quality targets for safe use by 2080 under a low GHG emissions and sustainable land use future (RCP2.6 x SSP1; left panel) compared to a future characterised by high GHG emissions and an increase is intensive agriculture (RCP6.0 x SSP3; right panel).

Table 3 Number and percentage of standing waters projected to achieve WFD High status or WFD Good status or higher under the two climate change and land use scenarios tested; results based on 317 standing waters with sufficient data.

Change scenario	WFD High status				WFD Good status or higher			
	2020	**2040**	**2060**	**2080**	**2020**	**2040**	**2060**	**2080**
RCP2.6 x SSP1	233 (74%)	252 (80%)	254 (80%)	269 (85%)	262 (83%)	271 (85%)	275 (87%)	285 (90%)
RCP6.0 x SSP3	230 (73%)	150 (47%)	142 (45%)	92 (29%)	260 (82%)	187 (59%)	184 (58%)	145 (46%)

for safe use, following the method developed by Carvalho *et al*. (2013). Under the worst-case scenario tested (RCP6 x SSP3), many more standing waters are projected to fail WHO water quality standards for safe use for water supply or recreation than under the best-case scenario tested (RCP2.6 x SSP1) (Figure 5). This demonstrates that, although the evidence suggests that we can reduce the impacts of climate change in this way, an urgent move towards more sustainable pathways for socioeconomic development and changes to land management practices are needed.

When the number of standing waters that are likely to meet Water Framework Directive (WFD) water quality objectives are considered, it was found that, under change scenario RCP2.6 x SSP1, there was a slight increase in the number of standing waters likely to fail WFD Good status for TP along the northern part of the Scottish mainland between 2020 and 2080, but there was little change elsewhere. In contrast, under change scenario RCP6.0 x SSP3, the number of failures to meet WFD Good status for TP increased rapidly across all parts of Scotland over that period.

More specifically, under change scenario RCP2.6 x SSP1, the number and percentage of standing waters that were projected to meet High quality status, or Good quality status or higher, for TP concentration increased from about 233 (74%) to about 269 (85%), and from about 262 (83%) to about 285 (90%), respectively, between 2020 and 2080 (Table 3). In contrast, under change scenario RCP6.0 x SSP3, the the number and percentage of standing waters that were projected to meet High

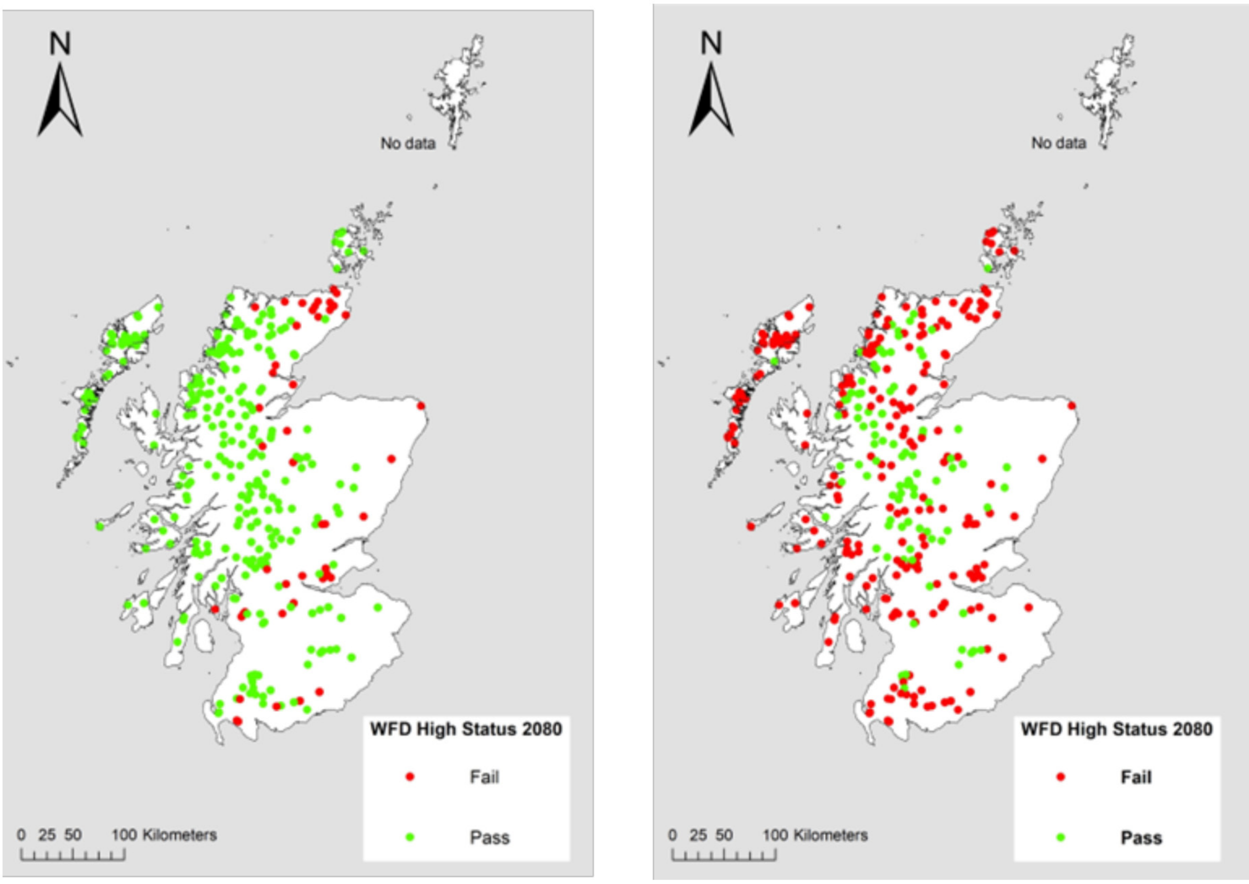

Figure 6 Comparison of the number and distribution of Scottish standing waters projected to fail WFD targets for High water quality status by 2080 under a low GHG emissions and sustainable land use future (RCP2.6 x SSP1; left panel) compared to a future characterised by high GHG emissions and an increase is intensive agriculture (RCP6.0 x SSP3; right panel).

quality status for TP concentration, or Good quality status or higher, decreased from about 230 (73%) to about 92 (29%), and from about 260 (82%) to about 145 (46%), respectively, between 2020 and 2080 (Table 3). These estimates are based on the 317 standing waters for which there are Water Framework Directive targets.

These results indicate that the future quality of Scottish standing waters is very dependent upon the climate scenario and socio-economic pathway that we follow, with water quality changing little or sometimes improving under RCP2.6 x SSP1 and worsening under RCP6.0 x SSP3. Overall our ability to meet WFD water quality targets will have reduced markedly by 2080 unless a pathway of low emissions and sustainable land use is followed. In terms of WFD compliance, the consequences of not following this more sustainable future scenario are visualised in Figure 6. Given that we already have an average global rise in temperature of almost 1.5°C, RCP6.0 probably represents a future that is close to a *status quo* or 'do nothing' scenario.

3.4 Can in-lake mitigation measures improve the quality of Scottish standing waters?

Many in-lake measures have been proposed to manage the adverse effects of nutrient enrichment on standing waters. However, few studies have considered how useful these established techniques would be for climate change adaptation and/or mitigation. An overview of available methods is given below. Although these are, for the most part, expensive options that are not sustainable solutions in the long term, they can be useful in solving short term problems. Examples of locations in Scotland where some of these approaches have been used successfully include Strathclyde Loch – to improve water quality for recreation, Loch Flemington – to reduce fish kills, and the James Hamilton Heritage Park loch, Lanarkshire – to control of toxic algal blooms.

3.4.1 Sediment removal

Sediment removal can improve water quality by removing the nutrient rich bed sediments that keep water column P concentrations high long after catchment inputs have been reduced. It can also improve habitat for aquatic plants and expose viable seeds to encourage re-growth of native species (Peterson, 1982). However, large-scale disposal of sediment requires large areas of land (Peterson, 1982; Cooke *et al*., 2005), unless it can be re-reused as a fertiliser. In general, sediment removal is expensive, especially if sediments need de-watering (Oldenburg and Steinman, 2019) or contain toxic contaminants, such as heavy metals. Case studies have reported variable success globally (Lürling, Smolders and Douglas, 2020), with effects being short term unless catchment P inputs are reduced sufficiently (Phillips *et al*., 2020).

3.4.2 Chemical amendments

Chemical amendments, including coagulants, oxidisers/P binders and algaecides/peroxides (Lürling, Smolders and Douglas, 2020), either clump cyanobacterial biomass and remove it rapidly to the lake bed, alter the chemical composition of the bed sediment to control P release into the water column, or kill the algae. Targeting chemical amendments towards areas where sediment P release is most likely to occur can be disrupted by internal mixing processes, making effective dosage difficult to assess (Douglas *et al*., 2016). Although some longer-term positive effects on ecological communities have been reported (Lürling, Smolders and Douglas, 2020), the release of cyanotoxins into the water during algae breakdown is a risk (Pei *et al*., 2014). Few of the materials available have been comprehensively tested for use in lakes or reservoirs and, in some cases, there is little evidence that they are suitable for large-scale application.

3.4.3 Physical alterations

Ultrasound has been suggested for controlling algal biomass by physically disrupting cells with sound waves (Purcell *et al*. 2013; Wu, Joyce and Mason, 2011, 2012; Rajasekhar *et al*., 2012). However, most ultrasound devices have a limited range (e.g. 10-12 m) and are ineffective for use at whole lake scales (Lürling and Tolman, 2014). Also, they can adversely affect non-target species, such as Daphnia, which provide an important nature-based solution that removes algal biomass from lakes through grazing.

Aeration and physical mixing can be used to oxidise bed sediments, thereby reducing the release of nutrients (e.g. P) into the water (Smolders *et al*., 2006). Aeration can also break down stratification in deeper waterbodies, stopping algae accumulating at the surface and reducing their growth through light limitation. The widespread use of hypolimnetic oxygen enrichment to control P release from bed sediments has had limited success (Lürling, Smolders and Douglas, 2020), although this approach has been more effective in lakes where high frequency monitoring systems have enabled this intervention to be targeted better (Carey *et al*., 2022).

Hypolimnetic withdrawal of nutrient rich water from the deep waters (hypolimnion) of lakes or reservoirs aims to exhaust sediment nutrient pools and reduce the period over which internal release of nutrients slows the recovery process (Nürnberg, 1987, 2007, 2019). However, to be effective, the withdrawn water needs to be replaced with low nutrient water (Nürnberg, 2007). This method also provides a way in which cooler water can be introduced to lower water temperature (Olsson *et al*., 2022).

Changes in lake flushing rates have been considered to reduce algal biomass, with May and Elliott (2019) showing that relatively minor changes to the flushing rate of Loch Leven could reduce algal blooms by up to 40%. Although other studies have suggested that this approach is an efficient restoration technique in stratified lakes (Nürnberg 2007), Olsson *et al*. (2022) found that reducing the water residence time in Elterwater, in England, by 40% had little effect on P concentrations or the amount of algae in the water. From a practical point of view, difficulties in finding sufficient low nutrient water for dilution often makes this technique unsuitable (Welch, 1981). However, where sources are available, this measure can be very effective, especially for shallow lakes (Hosper and Meyer, 1986).

3.4.4 Habitat and biodiversity management

Evidence from case studies suggests that, once nutrient inputs to lakes have been reduced, biodiversity recovery may be slowed by constraints on biological dispersal. This is true especially of aquatic plant communities and species that rely on them for habitat (Jeppesen *et al*., 2012). A range of options can be used to *manage the recovery process*, including seed dispersal, species translocation, removal of competing plants, reduction of herbivorous fish species and nutrient control (Blindow, Carlsson and van deWeyer, 2021; Orsenigo, 2018; Waters-Hart, 2019).

The ingress of non-native species can limit the recovery of native biodiversity once nutrient inputs are reduced. *Biomanipulation* involving the eradication of non-native fish species has been shown to increase the grazing of algae by zooplankton and reduce sediment disturbance. This improves water clarity and supports recolonisation by aquatic plants. However, control of invasive species across other trophic levels, including aquatic plants and macroinvertebrates, is more difficult. Even for fish, reductions in numbers or stocking levels can only be maintained through repeated interventions and their effect is reduced if nutrient inputs remain high (Jeppesen *et al*., 2012; Mehner *et al*., 2002; Skeate *et al*., 2022).

3.5 What do stakeholders think?

Stakeholders from the Scottish freshwater sector were asked their opinions on (1) current water quality issues caused by climate change on Scottish standing waters and who these are likely to affect; (2) potential mitigation measures that could be employed to help reduce the impacts of climate change on Scotland's standing waters and (3) potential policy changes required to implement mitigation measures. Replies were received from environmental regulators (n=5; 36%), government agency officers (n=3; 21%), charity officers (n=2; 14%), local council members (n=1; 7%), farmers union members (n=1; 7%), environmental consultants (n=1; 7%), and policy makers (n=1; 7%). Our key findings are summarised below. More details can be found in Appendix 8.

3.5.1 Current water quality issues caused by climate change

Most interviewees (n= 13; 93%) were concerned about the impacts of climate change on Scottish standing waters although a small minority (n=1; 7%) said that it was not their highest priority. Most respondents were concerned about the impacts of climate change on standing waters in terms of impacts on biodiversity (n=9; 64%), increases in algal blooms (n=8; 57%), although impacts on drinking water reservoirs, and therefore on the security of drinking water supply (n=5; 36%) and impacts on public health (n=4; 29%) were also seen as important. Interviewees perceived increases in algal blooms (n=11; 79%), eutrophication problems caused by nutrient runoff (n=10; 71%), increased storm events and flooding (n=6; 43%) and higher soil erosion rates (n=6; 43%) as the main water quality issues caused by climate change. When asked who or what water quality would be affected by water quality issues, and how, most interviewees mentioned biodiversity (n=11; 79%), the general public in terms of recreational use (n=11; 79%) and drinking water supply (n=10; 71%), fish communities and anglers (n=10; 71%), as well as businesses (n=5; 36%) and tourism (n=5; 36%).

3.5.2 Possible mitigation measures

As short-term measures (≤ 5 years) to mitigate water quality issues, more interviewees mentioned reducing diffuse pollution from farmlands (43%), engaging with farmers to enable change to more sustainable practices (36%), incentivising and monitoring actions by farmers (29%), engaging with land managers and landowners (29%), and increasing the number and/or the width of buffer strips (29%) than other potential interventions. As long-term measures (> 5 years) to mitigate water quality issues, most interviewees mentioned using geoengineering or innovative solutions (29%), re-evaluating the suitability of existing controls (29%), reducing runoff from land (29%), introducing more prescriptive land management guidance (21%), and having an agricultural reform (21%) than other potential measures. In general, the top suggestions of measures to be implemented in the short-term belonged to the engagement and funding, and landscape management categories. On the other hand, the top suggestions of measures to be implemented in the short-term belonged to the in-lake interventions (including geo-engineering) and policy and enforcement categories.

3.5.3 Possible policy changes required

Most respondents agreed that changes to land and water policy would be needed to ensure the effective implementation of mitigation measures, most interviewees responded yes (n12; 86%), with only a few leaving the response unclear (n=2; 14%), as shown in Figure 46. When asked what policy changes they would suggest, more respondents suggested increasing engagement and education with farmers and landowners (n=4; 29%), incentivising sustainable practices (n=4; 29%), changing the general binding rules (n=3; 21%), and generally implementing stricter policy (n=2; 14%) than any other changes. Finally, more interviewees (n=4; 29%) mentioned the need for a holistic and balanced approach to tackling climate change impacts on Scottish standing waters than any other approach.

3.6 Which policies are relevant to solving this problem?

Climate change is currently affecting, and projected to further affect, standing water quality in Scotland (May *et al*., 2022a,b). This policy review has shown that adaptations to current water policy and existing monitoring networks will need to be included in Scotland's strategic and coordinated response to reducing climate change impacts on these waterbodies. This conclusion is supported by the recent Climate Change Committee (2022). Is Scotland climate ready? – 2022 report to Scottish Parliament, which highlighted that, for Scotland's adaption plans (e.g. the SCCAP2 programme) to be effective, Scotland needs to improve its monitoring and evaluation systems urgently to assess changes in climate-related risks and impacts.

Policy recommendations based on this review, and those suggested by May *et al*. (2022a), are given below. These are reported according to the global, national and regional impacts that they aim to address.

3.6.1 Global climate change impacts – adaptive national water policy perspectives

There is a policy gap between global and national understanding of the impacts of climate change on water temperatures and changing rainfall patterns that needs to be closed. Failure to address this issue and monitor key indications of climate-related risks effectively will undermine the development and implementation of adaptive water policy and any management practices intended to mitigate the complex interactions that affect water use and nutrient run off at regional and local scales.

3.6.2 National climate change impacts – adaptive regional water policy perspectives

Changes to national scale water policies and land management practices will be required to limit climate change impacts on the quality of Scottish standing waters in the future. These impacts will be mediated through shifts in catchment and in-lake processes associated with changes in nutrient runoff, flushing rates and water temperatures. In combination, these changes will exacerbate the future risk of algal blooms and may compromise Scotland's ability to meet statutory goals and regulatory targets within given timelines.

As it is likely that climate change and its effects cannot be addressed in the short-term, it is important to identify the main factors that limit algal growth and accumulation that can be controlled at national scale. For example, better control of nutrient losses to water from agricultural, industrial and sewage related sources may be required to reduce the likelihood of potentially harmful algal blooms (Scottish Government, 2012; May *et al*., 2019). In the past, many of these interventions have required a licence issued by SEPA under the Controlled Activities Regulation (CAR) (Water Environment (Controlled Activities) (Scotland) Regulations 2011 (as amended)). CAR has now been superseded by the Environmental Authorisations (Scotland) Regulations 2018, which aims to bring permitting across all regimes under a single integrated authorisation framework. To be effective, future licensing criteria will need to take account of climate change and the need for adaptation to reduce its impacts.

Other national water policy and land use management practices (e.g. River Basin Management Plan for Scotland 2021-2027, Scotland's Third Land Use Strategy 2021-2026. Getting the best from our land. Scottish Government (2021)) will also need to be take into consideration how national-scale climate driven risks affect the quality of standing waters at regional to catchment scales. The Climate Change Committee (2022). Is Scotland climate ready? – 2022 report to Scottish Parliament highlighted that the current River Basin Management Plan for Scotland does not include any specific actions or adaptions for countering changing climatic conditions. For example, it does not take increasing river temperatures into account. Also, although the Third Land Use Strategy highlights the need for sustainable land-use to help in climate change mitigation and adaption, like the RBMP it does not detail specific actions to achieve those objectives.

The future agriculture bill in Scotland will provide an opportunity to encourage sustainable and regenerative agricultural practices. Encouraging improved nutrient use efficiency through regular soil testing would likely be an effective means to improve water quality.

Revision of current nutrient status criteria for Scottish standing waters 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. For example, in relation to meeting EU Water Framework Directive (WFD) (EU Water Framework Directive (2000)) targets for Scottish standing waters, mitigation/adaptation strategies will need to be implemented to achieve good ecological status, prevent its further deterioration and guide restorative action. It may be necessary to recast WFD objectives to achieve this, given that baseline conditions that underpin the WFD concept will have changed under current climate change conditions. In addition, the recast Drinking Water Directive (EU Drinking Water Directive - Recast (2020)) will require Catchment Risk Assessments to be created for all drinking water catchments to increase the level of source control for pollutants (referred to as 'Hazards and Hazardous Events'). This will encourage a prevention-led approach to addressing climate change interactions with these catchment factors, instead of reactively managing potential impacts (e.g. of algal blooms) on public health with expensive water treatment processes.

3.6.3 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 – Revised Guidance. Scottish Government (2012), especially in relation to climate change impacts, by capturing new evidence that emerged from the May *et al*. (2022a,b) report. This would help protect the amenity value of locally important still waters (e.g. for recreational use, water supply and wellbeing purposes) and reduce climate-driven water quality risks to public and animal health, in addition to meeting climate change mitigation/adaptation needs through other policy routes.

3.6.4 Future monitoring

The recent Climate Change Committee (2022). Is Scotland climate ready? – 2022 report to Scottish Parliament made it clear that "Scotland lacks an effective monitoring and evaluation systems meaning, that changes in aspects of many climate-related risks are largely unknown". In response to this, the existing monitoring network for Scottish standing waters needs to be reviewed, urgently, with a focus on developing an integrated approach for detecting climate change impacts whilst focusing on the use of new scientific innovations and resource capabilities. May *et al*. (2022) made the following recommendations for monitoring key indicators of climate-related risks to inform adaptive water policy and management practices.

- 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 total algae and cyanobacteria, separately.
- Measure nutrient inputs from catchments, including high temporal resolution gauging of inflows and nutrients 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; low flushing rates due to droughts).
- 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" can be integrated to improve our ability to detect and forecast change.

In addition, the incorporation of modelling into waterbody assessment processes would enable lessons learned from site specific monitoring to be extended to un-monitored sites. That said, there is still a need for frequent, regular monitoring of key sites to provide a meaningful assessment of change and provide early warning of potential impacts that may require mitigation.

4 Recommendations

4.1 Estimating the financial impacts of algal blooms

- This study has estimated the cost of algal blooms to the Scottish economy to be more than £16.5 million per year, but many of the values are based on figures from England and Wales. More accurate figures are needed for Scotland.
- The costs of medical and veterinary expenses due to algal blooms affecting the health of people and animals are not included. Better recording of incidents requiring such interventions would provide more accurate information.

4.2 Reducing nutrient runoff into water

- Runoff from land, especially farmland, is the main source of nutrient inputs to water. Soil testing to optimise nutrient use efficiency would reduce nutrient losses from land to water at catchment scale.
- Following a low emissions pathway for greenhouse gas (GHG) emissions, combined with sustainable land use, could reduce phosphorus runoff by c. 20% by 2080; intensifying agriculture under a high GHG emissions scenario almost double phosphorus runoff by c. 138%.
- Sustainable land management is more effective than increasing buffer strips at reducing phosphorus runoff, with their estimated levels of phosphorus reduction being c.40% and c.1%, respectively.

4.3 Reducing the risk of algal blooms

- Harmful algal blooms are projected to increase under climate change, with consequent detrimental effects on water use, water supply, biodiversity, and health and wellbeing. Sustainable use of phosphorus to reduce losses from land to water is key to reducing the threat from algal blooms.
- National monitoring data should be used to identify lake typologies or geographical regions that are likely to be most sensitive to changes in nutrient inputs and climate. This information should be incorporated into river basin planning to focus mitigation strategies into areas where they will be most effective.
- Site-specific monitoring is needed to develop comprehensive lake restoration/adaptation plans within a robust assessment framework.

4.4 Identifying effective methods for in-lake mitigation

- The trialling of in-lake measures needs to be accelerated, perhaps through Scotland's Hydro Nation Research and Innovation Programme, to identify measures that are suitable for Scottish standing waters.
- Methods for using new *in situ* and remote sensing base monitoring techniques should be developed to identify where interventions would work best; this would improve cost effectiveness.
- It is vital that pre- and post-mitigation monitoring data are collected/made available over several years, so that success can be assessed properly.

4.5 General recommendations

- A national roadmap for climate change adaptation (and mitigation) in Scottish standing waters should be developed as part of National Adaptation Planning; there is considerable practical experience within the international community that is transferrable to the Scottish context.
- Knowledge exchange through engagement with activities related to the UN Decade on Ecosystem Restoration, the UN Sustainable Development Goals (especially SDG 6), the Convention on Biological Diversity Global Biodiversity Framework 2030 (especially Target 2), and the United Nations Environment Assembly Resolution 5/4 on Sustainable Lake Management (UNEP/EA.5/Res.4) would be beneficial.

5 Conclusions

Climate change is already affecting the quality of Scotland's lochs, reservoirs and other standing waters (May *et al*., 2022a,b). For example, standing waters are already warming and their flushing rates are already falling, especially in summer. Together, these changes will increase the sensitivity of Scottish standing waters to nutrient inputs, putting them at higher risk of developing water quality issues. These effects will degrade water quality and adversely affect the sustainable use of waterbodies for recreation, tourism, habitat for wildlife, and water supply. This, in turn, will have negative impacts on wellbeing, water treatment costs and economic growth.

As climate change effects cannot be addressed in the short-term, it is important to identify the main factors limiting the growth and accumulation of phytoplankton so that this can be controlled. Phosphorus is one of the key factors that affects the likelihood of algal blooms developing, so this needs to be reduced. Land-based mitigation measures focused on the better management of soil nutrient status to at or below the agronomic optimum can reduce TP inputs to standing waters by up to 40%. However, these measures need to include the holistic management of soils to maximise soil organic matter content and to improve nutrient use efficiency through regular soil testing to optimise the scale, timing and location of fertiliser applications. Smaller-scale interventions, such as buffer strips, are less effective at reducing TP losses from land to water.

Under future change scenarios, the sustainable land use reconfiguration under SSP1 combined with the lower climate change projected to occur under RCP2.6 was projected to reduce TP losses from land to standing waters by up to 20% compared to the current baseline. In contrast, expansion of arable land and intensification of agriculture (SSP3) alongside the higher emissions scenarios reflected within RCP6.0 was found to almost double TP inputs to standing waters, greatly increasing the risk of algal blooms. Changes in land and water management policies will be required to limit these effects in the future. For example, nutrient inputs from agricultural, industrial and sewage sources may need to be reduced (Scottish Government, 2012; May *et al*., 2019). As many of these interventions will require a licence to be issued by SEPA, or an existing licence updated, under the controlled activities regulation (CAR) (Scottish Government, 2005) or relevant revisions of this legislation, the relevant licensing criteria will need to take climate change effects into account.

In terms of amenity value, projected TP concentrations were converted to likelihood of exceedance of WHO water quality thresholds for safe use (WHO 2003, 2004, 2021). This showed that, by 2080 and under the worst case scenario tested, many more standing waters will fail WHO water quality standards for safe use than under the best case scenario tested. In addition, a marked increase in the number of standing waters failing to meet WFD targets for Good and High ecological status can be expected.

Understanding the processes operating in standing waters and the extent to which they will respond to changes in climate and nutrient inputs (reduction or increase) will be key to informing within waterbody management options as well as catchment management approaches. However, national monitoring programmes are designed to inform regulatory reporting of state and are insufficient to allow such assessments to be made. So, although collecting and screening national monitoring data can indicate types of waterbodies and regions that are likely to be sensitive to changes in nutrient inputs and climate, site specific monitoring programmes need to be modified to inform the development of comprehensive management plans within a robust restoration/adaptation focused monitoring and assessment framework (e.g. an adaptation of the current River Basin Management Approach).

Although measures are available that allow adaptation to minimise the effects of climate change in standing waters, there are no processes in place to inform water managers on how to select the most appropriate measure for their particular waterbody. There are no examples of long-term and effective in-lake management for eutrophication control or climate change adaptation (or both) within Scotland, but evidence from other countries (e.g. England, the Netherlands, Denmark, USA, China) where water quality issues have already necessitated the development and implementation of in-lake measures provide a source of knowledge on a range of measures that could inform decision making within Scotland. Given the habitat quality decline projected in this study, in-lake habitat improvement measures or species translocation programmes may be important in the future if biodiversity and conservation targets are to be met.

6 References

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Appendix 1 Potential costs of 'algal blooms' to the Scottish economy.

Background

The World Health Organization (WHO) has set water quality thresholds for safe use (WHO 2003, 2004, 2021) that are based on levels of potentially toxic cyanobacteria in the water. When these are exceeded, water supplies need additional treatment to prevent filters clogging and to remove toxins. Also, recreational facilities must be closed to users until the problem has been resolved (Codd *et al*. 2005). Increased water treatment costs incurred, and income lost due to restrictions on recreational use, have economic consequences for local businesses. There are also wider impacts, including detrimental effects on property values, and human and animal health implications (illness; death) associated with cyanobacterial concentrations that exceed WHO thresholds for safe use (WHO 2003, 2004, 2021).

There is limited economic data available on the costs of standing freshwater eutrophication problems and impacts in Scotland. Most previous cost estimates have been based on research carried out in England and Wales (Pretty *et al*., 2003). These costs have been extrapolated to Scotland to estimate costs of eutrophication of £8.3 million, rising to £38.1 million under 4°C warming (Jones *et al*., 2020). However, this estimate does not use any Scottish data; instead costs are based on population size.

Aims and objectives

In this study, we aimed to update national and international estimates of the costs of algal blooms and potential mitigation measures using Scottish data, where available. Where Scottish data were not available, we have summarised or adapted data from Jones *et al*. (2020).

Methods

Standing freshwater provides a number of ecosystem services within Scotland, including drinking water, recreation, fishing and cultural and landscape values. Estimating the costs of degradation of freshwater due to eutrophication requires an understanding of the change in the value of the services provided.

We used a rapid evidence review approach to identify available data on the costs of degradation of Scottish standing freshwater. We used the following approach:

Stage one – Identification of key papers by the project team.

Stage two – Searches of the papers cited in, and citing, the key papers identified in stage one. Stage two was considered complete once no new papers were identified from reference lists or citing new papers identified.

Stage three – Targeted searches for specific services not identified through stages one and two.

Data extracted from papers included details on: study location, time period, methods, population and results. Scottish data, collected over the largest area and time period and with a representative population was prioritised for inclusion.

Where no new robust data could be sourced for Scotland, we have presented the results from Jones *et al*. (2020). However, because these are based in a scaling up of England and Wales data, we suggest that these values should be re-estimated when Scottish data becomes available.

All costs are presented as 2023 equivalent values.

Results

There are a number of costs associated with water quality degradation. Most, but not all, are linked to the increase in occurrence of algal blooms associated with climate change. Although there is limited information on this, the data that are available are summarised below.

Estimated costs associated with levels of nutrients delivered to water due to soil erosion.

Phosphorus

Algal blooms are caused, primarily, by excess levels of phosphorus entering standing waters from their catchments; this is exacerbated by climate change, which makes these water bodies more sensitive to phosphorus inputs (May *et al*., 2022a). The cost of reducing phosphorus run-off into Scottish lochs was estimated at £4.5 million/year by Rickson *et al*. (2019). This figure was based on cost estimates from England and Wales (Pretty *et al*., 2000) and modelled phosphorus run off values in Scotland; it does not include loss of agricultural productivity.

In addition, the cost of removal of phosphorus when run-off cannot be mitigated is estimated at £273/kg (Vinten *et al*., 2012). In Appendix 3, we have estimated that about 200 tonnes of phosphorus per year is entering Scottish standing waters in the form of runoff; this suggests that removal of phosphorus from this runoff would, therefore, cost about £56.4 million per year.

Nitrogen

Nitrates are routinely removed from water supplies by water companies to enable them to meet the UK drinking water standard of ≤50 mg L^{-1}, which is based on WHO guidelines. According to Pretty *et al*. (2000), £38.3 million was spent per year in removing nitrates from water supplies in England and Wales between 1990 and 1997 (costs updated to 2023 values). Based on these costs and modelled levels of run off, the costs of removing nitrates from drinking water in Scotland have been estimated at £2.5 million per year by Rickson *et al*. (2019). The proportion of this that is attributable to standing freshwaters, alone, is unknown. Estimating this value will require measurements of run off into reservoirs and the percentage of water abstracted from those reservoirs for water supply to be estimated. This is beyond the scope of this project.

Sediment

The removal of sediments due to soil erosion from rivers and canals across Scotland has been estimated to £4.7 million per year (Rickson *et al*., 2019). These costs were estimated on the basis of estimated costs of £7.40 per tonne of sediment removed. However, sediment removal is not a carried out as routinely from standing waters as from rivers and other channels (SEPA, 2010) potentially leading to higher costs per tonne removed. Also, sediment build up in standing waters is a slower process than in rivers, and removal is not recommended in most cases (SEPA, 2010). So, total removal costs may be lower. Costs of removing sediment from drinking water was estimated at £202 million per year, and £22 per tonne (Rickson *et al*., 2019). However, as with nitrates, this includes data from rivers and there are no data from which accurate apportionment of this cost can be estimate for standing freshwaters.

Estimated costs associated with algal blooms

Recreation

Water quality affects how often people visit water (often known as blue spaces) for recreational purposes, with fewer visits being made to sites that people perceive to have poorer water quality (Börger *et al*., 2021). Given the highly visible nature of algal blooms, it would be expected that waterbodies undergoing, or with a history of, algal blooms would be perceived as having lower water quality than those without. A European scale study of bluespace visits found that 21% fewer visits were made to water bodies with a lower perceived water quality, with an estimated loss of £44 per visit for UK visitors (Börger *et al*., 2021). The same value was also recorded in a specific survey of visitors to Loch Leven in Perth and Kinross (Donnell, 2015; ScotInform, 2010). Using visitor data from Loch Lomond and the Trossachs and Cairngorms National parks, we estimated the minimum loss of recreational value for loch visits at £5.7 million per year (Table 1).

This value is a significant underestimate of the value of standing freshwater for recreation in Scotland. In the Cairngorms National Park, for example, visitor data are limited to the most significant lochs and excludes Loch Garten, because visits there could not be ascribed to the presence of freshwater,

Table 1 Impact of decline in water quality on visitors to Scotland's National parks, and associated loss of value as estimated in Börger *et al*. (2021); values shown as percentage and actual reductions in visits.

	Reduction in number of visits to lochs per year	Number of visits per year	Number of visits expected at reduced water quality/year	Value of visit	Cost of water quality reduction/ year
Cairngorms – Total visitor numbers 1.92 million					
Loch Morlich	23%	44,000	34,760	£44	£1.5 million
Loch an Eileen	16%	31,000	24,490	£44	£1.1 million
Loch Muick	8%	15,000	11,850	£44	£0.5 million
Total Cairngorms - £3.1 million					
Loch Lomond and the Trossachs – Total visitor numbers 4 million					
All lochs	19%	76,000	60,000	£44	£2.6 million
Total Scottish National Parks - £5.7 million per year					

as some may have been to the Osprey Centre. In Loch Lomond and the Trossachs, visits to all lochs are recorded in one data of entry, with visitors likely to have visited more than one loch during their stay in the park. The cost estimate of £44 per visit accounts only for local travel and did not include tourists with higher transport and accommodation costs.

Human health and welfare

Algal blooms have the potential to cause significant damage to human health through direct contact (e.g., swimming, water sports) and indirect contact (e.g. spray), and the ingestion of infected seafood (Kouakou and Poder, 2019). Associated costs could be estimated from medical costs, lost days of work and reductions in Quality of Life Years. However, in general, the literature focuses on the effects of marine algal blooms (Kouakou and Poder, 2019) and we were not able to find data for fresh waters from the UK or Scotland. Nevertheless, the closure of affected standing waters reduces the number of visits made, which incurs a cost to local businesses.

Animal health and welfare

Health risks to dogs and other animals, such as livestock, from bathing in and/or ingesting harmful blue-green algae have been well documented. This has led to dog walkers being advised that they should avoid letting their animals enter affected water during the summer months (British Veterinary Association, 2022). There are no data available on the number of dogs impacted, but the average cost of insurance claims for medium sized dogs affected by vomiting, a common consequence of blue-green algae ingestion, is estimated to be £849 per incident per dog (Animal Friends, 2021). This is likely to be at the lower end of estimates for dogs affected by algal blooms, given that the complications that often arise from blue-green algae being ingested by dogs often necessitate hospitalisation and may lead to death (British Veterinary Association, 2022).

Fisheries

Algal blooms can cause damage to fish farms and wild fisheries. The majority of the current research in Scotland is focused on marine algal blooms and their impacts on shellfish industries. However, it is also likely that wild salmon fisheries, which undertake part of their lifecycle in lochs, also suffer damage from algal blooms. Algal blooms can cause damage to fisheries. The majority of the current research in Scotland is focused on marine algal blooms and their impacts on shellfish industries. However, it is also likely that salmon fisheries which undertake part of their growing in lochs also suffer damage from algal blooms. There has been a recent injection of funding of £335,000 to develop a training plan to deal with algal blooms impacting salmon farming (Editorial Team, Fish Farming Expert, 2023). Part of this cost is likely to be related to freshwater salmon farming, although the exact proportion cannot be determined from available data. The consequences of an algal bloom on the fishery at Loch Leven has been well documented and provides a good example of the types of impact involved, in terms of cost and duration. An algal bloom event at the Loch Leven in 1992 had large and long lasting impact on the trout fishery, including long lasting and persistent reputational damage (May and Spears, 2012). The initial cost was estimated at £110k for the three months immediately following the algal bloom (unpublished data) but the full cost was much higher. This is illustrated in Figure 1, which shows how the number of boat hires declined after the algal blooms of the early 1990s and did not begin to recover until about 2009. Restoration activities from the late 1980s to the mid- 1990s cost approximately £4.1M at present day values, with the total loss in revenue to the fishery being estimated to be £320k per year when comparing 1975 to 2005. Post recovery (2007 onwards) income from boat hires increased by £58.8k per year.

Additional costs

House prices

Algal blooms can lead a decline in the value of houses that have views of, or are within short travelling distances of, an impacted water body. Most of the research into this has been undertaken within North America, with large spatial variation in the economic impacts reported. It has been suggested that the impacts of a 1µg L^{-1} increase in algal bloom levels decreased nearby property values by 1.7% in Lake Erie (Wolf, Gopalakrishnan and Klaiber, 2022), while the same increase in Ohio was found to have reduced house prices by 22% for lakefront properties, and 11% in properties within 300 metres of the impacted lake (Wolf and Klaiber, 2017). We found only one paper looking at the relationship between house prices and water quality in the UK. This was limited to the Mersey basin, where estate agents estimated a fall in house prices of up to 20% in the event of poor water quality (Wood and Handley, 1999). We have no accurate data for Scotland, but it is likely that house prices here are similarly affected by poor water quality in nearby standing waters, too.

Water treatment – algal removal

Similar to nitrates, water companies need to remove algae and related toxins from drinking water. In England and Wales, the costs of removing algae that originates from algal blooms was estimated to be 10% of the total cost of algal control, i.e. £33m per year (Pretty *et al*., 2003), with £1.4m per year being estimated for Scotland by Jones *et al*. (2020). As these data do not include values for Scotland, specifically, and do not estimate the amount of algal blooms occurring in standing freshwater used for drinking water, further work is needed to improve this estimate. Atmospheric pollution

Algal blooms lead to increases in production of nitrous oxide, ammonia and methane. The costs of these greenhouse gas emissions to England and Wales were estimated at £8.9 – £14m per year, based on impacts on climate, health, sea-level rise, water availability, biodiversity and natural disasters (Pretty *et al*., 2003). When extrapolated to Scotland, these costs were estimated at £0.6m per year (Jones *et al*., 2020). Data from Scotland would greatly improve this estimate.

Ecological damage

Based on estimates obtained from biodiversity action plans, the costs of algal blooms to biodiversity management in England and Wales was estimated at £12.8m-£17.6m per year (Pretty *et al*., 2003), and £0.7m per year for Scotland (Jones *et al*., 2020). This value does not estimate the actual impacts of algal blooms in Scotland, and these costs should be updated with Scottish data.

Table 2 Minimum costs per year of eutrophication and air pollution associated with standing water degradation in Scotland. Data in italics have been estimated from Jones *et al*. (2020) (England and Wales only) and would benefit from being updated with Scottish values when available. N/A = Not Applicable.

Impact	Spatial scale of costs	Costs per year	Notes	References
Costs associated with soil erosion				
Phosphorus	National	£4.5 million	Based on current erosion rates	Rickson et al. (2019)
Nitrates (drinking water)	National	Under £2.5 million	Data is only available for all water treatment, not only standing freshwater	Rickson et al. (2019)
Sediment	N/A	Tens to hundreds of thousands	Data not available for lochs.	N/A
Costs associated with algal blooms				
Recreation (inc. tourism)	National parks	£5.7 million	Estimates from visitor numbers to national parks. To scale to the national level national scale data would be needed on visits to freshwater.	Börger et al. (2021)
Human health	N/A	Unknown	Data not available.	N/A
Animal health	Per dog	Minimum £849	No data on number of dogs treated for blue-green algae.	Animal Friends (2021)
Fisheries	National	£100,000	Scaled from England and Wales	Jones et al. (2020)
Value waterside properties	National	£0.72 million	Based on data from rivers, scaled from England and Wales to Scotland	Jones et al. (2020)
Water treatment – algal removal	National	£1.4 million	Scaled from England and Wales	Jones et al. (2020)
Atmospheric pollution	National	£0.6 million	Scaled from England and Wales	Jones et al. (2020)
Ecological damage costs	National	£0.7 million	Scaled from England and Wales	Jones et al. (2020)
Estimated total costs for Scotland		**£16.5 million**	**Excluding medical and veterinary costs**	

Loch Leven case study

There is no national scale data on visits to standing freshwaters in Scotland, so it is not possible to scale up the above estimates from the National Parks to the whole of Scotland. However, more detailed data are available for Loch Leven, and these can be compared with the national scale estimates summarised in Table 2, above. In 2015, it was estimated that the loch has 208,572 visits per year from recreational users, specifically for the heritage trail which circuits the loch (Donnell, 2015). This study estimated that each visitor spends on average £24.07 per visit, excluding travel. Travel was estimated in a previous study at £20.37 (ScotInform, 2010). We therefore estimate a total value of £44.44 for each visit to Loch Leven. Making the assumption that algal blooms would lead to a reduction in visitor numbers of 21% (Börger *et al*., 2021), we predict that an algal bloom in any given year at this one site alone would reduce visitor numbers to 164,772, with an associated loss of £1.9 million.

In another example, a contingent valuation study was carried out with residents in Kinross and Milnathort, the two local towns to Loch Leven, when an algal bloom occurred. Although direct health impacts could not be estimated, this study estimated willingness of local residents to pay for a reduction in the number of cyanotoxin risk days at the site. Willingness to pay was estimated at between £15.48 per household per year and £18.95 per household, or £232,216 - £284,285 per year when aggregated across the whole population (Hunter *et al*., 2012). In a third study, the Loch Leven Area Management Group (LLAMAG) surveyed the costs to local businesses in the three months following and algal bloom in the summer of 1992. Losses to local businesses were estimated to be about £1,421,600 to shops, hotels, guest houses, etc. and £232,360 to the fishery, with increased water treatment costs to local industries being estimated at £337,970 (at present day values; original data from LLAMAG (1993) cited in LLCMP (1999)) – a total of £2 million.

Given these very detailed studies at Loch Leven, which estimated losses of about £2 million per year due to algal blooms, it seems unlikely that the cost of algal blooms across the whole of Scotland is only £16.5 million. This illustrates the need for a more detailed study across the whole of Scotland of the economic impacts associated with these events.

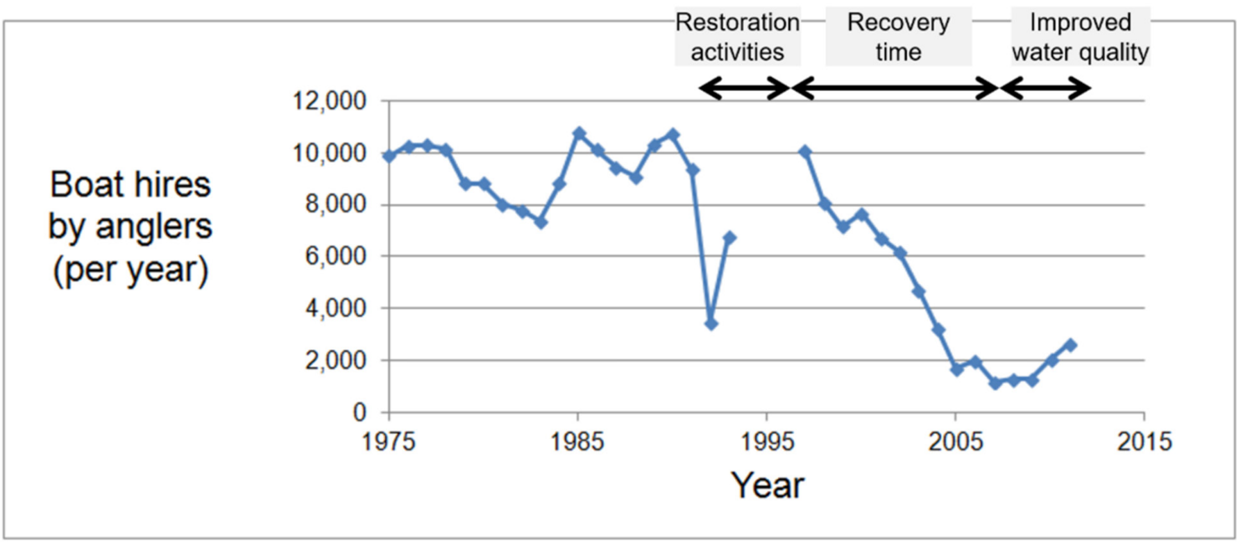

Figure 1 Impact of algal bloom, restoration activities and improvements in water quality on numbers of boats hired by anglers at Loch Leven; changes in revenue were estimated at £40 per boat hire based on based on present day values. Data: Kinross Estate Company.

Conclusions

A summary of the overall costs of algal blooms in Scotland, in terms of water treatment, house prices and visitor numbers is shown in Table 1. These values exclude the costs of treatment for people and animals whose health is affected by these blooms.

The cost estimates from Jones *et al*. (2020) would benefit from further research to provide more figures for Scotland. However, on the basis of existing information, we estimate the costs of eutrophication to Scottish standing freshwater to be about £11.3 million per year, excluding data from Jones *et al*. 2020, or £14.8 million including costs from Jones *et al*. (2020). This should be taken as a lower estimate as many elements could not be costed. These include costs related to human and animal health impacts, which are not recorded at present. However, we note that these could be measured in terms of number of incidents reported, costs to the National Health Service [NHS] and veterinary treatment costs to the owners for pets and livestock affected, in future.

When algal blooms occur, water companies need to remove algae and related toxins from drinking water supplies. In England and Wales, these costs have been estimated to be about 10% of the total cost of algal control, i.e. £33m per year (Pretty *et al*., 2003), with £1.4m per year being estimated for Scotland by Jones *et al*. (2020). As these data do not include values for Scotland, specifically, and do not estimate the amount of algal blooms in standing waters used for drinking water supply, further work is needed to improve this estimate.

Algal blooms lead to increases in the emissions of the greenhouse gases nitrous oxide, ammonia and methane from affected waterbodies. The costs of these emissions to England and Wales were estimated at £8.9–£14m per year, based on the combined impacts on climate, health, sea-level rise, water availability, biodiversity and natural disasters (Pretty *et al*., 2003). When extrapolated to Scotland, these costs were estimated at £0.6m per year (Jones *et al*., 2020). Data collected from within Scotland would greatly improve this estimate.

Based on estimates obtained from biodiversity action plans, the costs of algal blooms to biodiversity management in England and Wales was estimated at £12.8m–£17.6m per year (Pretty *et al*., 2003), and £0.7m per year for Scotland (Jones *et al*., 2020). However, these values do not estimate the actual impacts of algal blooms in Scotland; for greater accuracy in these estimates, these costs need to be updated with Scottish data. These data are not available at present. In addition, the cost of removing phosphorus from runoff to prevent algal blooms has been estimated to be about £56.4 million per year.

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Appendix 2 Data preparation

Background

May *et al*. (2022a,b) showed that climate change is causing Scottish standing waters to warm, thereby increasing the risk of them developing algal blooms that reduce their amenity value. In this current study, there is a focus on the effects of climate change on the quality of these standing waters and the potential mitigation of these impacts. This requires linked catchment and lake response models to be developed that can be used to project future TP inputs to lakes and their future responses in terms of TP and chlorophyll-*a* concentrations. High chlorophyll-*a* levels are indicative of algal blooms, some of which can be toxic to people, their domestic animals and livestock, and to wildlife.

Aims and objectives

The aim of this part of the project was to gather and prepare all relevant data for modelling the behaviour of catchments and lakes under climate change. Data were processed to enable waterbody specific TP inputs to the lochs (loads) and in-loch chlorophyll-*a* concentrations to be predicted under different climate change and land management scenarios at 10-year intervals between 2020 and 2080.

Methods

Data processing

First, Scottish standing waters with delineated catchments of greater than 1 ha in area were assembled and intersected with SEPA monitored waterbodies. Waterbody and catchment polygon files for these lochs were then outputted. These formed the basis of further data processing in preparation for modelling catchment based TP loads to water bodies and the consequent prediction of within water body TP and chlorophyll-*a* concentrations. All data were processed using RStudio version 4.3.1.

Inflow identification and sub-catchment generation

SEPA river monitoring stations were assessed for proximity to each standing water and converted to spatial output files (.gpkg). The SEPA inflow data were then matched to downstream lochs manually using maps and a 50m digital elevation model (DEM). To classify as a match, monitoring station locations needed to be (a) on a direct inflow into the lake (not a tributary), (b) within the lake catchment, and (c) at a higher altitude than the loch itself. The SEPA river monitoring data (1997–2022) were then filtered to correspond to sites in the matched inflow dataset. Values below the limit of detection (<LOD) were halved, whereas values over the limit of detection (>LOD) were left unchanged.

Figure 2 Example of sub-catchments generated using SEPA inflow monitoring station locations.

Hydrological catchments were delineated for all of the 44 SEPA monitoring stations that were found to be on loch inflows, using the Institute of Hydrology Digital Terrain Model (IHDTM) dataset (see example shown in Figure 2). Use of these sub-catchments allowed hydrologically effective rainfall (*EffRain*) values to be calculated and further models to use an accurately delineated catchment to predict in-stream TP concentrations for comparison with the equivalent SEPA monitoring data.

Deriving variables for modelling

Effective rainfall calculation

In Phase 1 of this project (May *et al*., 2022a,b), hydrologically effective rainfall (*EffRain*) and average water temperatures were calculated for all catchments and waterbodies and summarised as monthly values for 1981 to 2080. Although this approach provided the data necessary for the first phase of this project (May *et al*., 2022a,b), in this study spatial data for *EffRain* were required for the whole of Scotland to ground truth the TP runoff model. Monthly *EffRain* values were calculated for 2015-2019 based on the following interpolated monitoring data: (a) CHESS-met, gridded daily meteorological variables over Great Britain for the years 1961-2017 at 1 km resolution (Robinson *et al*., 2020) and (b) CHESS-PE, gridded potential evapotranspiration over Great Britain for the years 1961-2017 at 1 km resolution (Robinson *et al*., 2020). Monthly data for precipitation (*pr*) and potential evapotranspiration (*peti*) were used to calculate *EffRain*, which was then extracted as mean values per catchment.

To calculate *EffRain*, the potential evapotranspiration data (*peti*) was converted into predicted 'actual evapotranspiration (*AET*)' using the method described by May *et al*. (2022b). The amount of evapotranspiration varies with land cover type, so the 1 km Land Cover Maps (LCM) 2015 data were used to make this adjustment. Each land cover class was assigned a 'crop coefficient' (*Kc*), with seasonal values taken from, or approximated to, FAO Crop Evapotranspiration data and data from Nistor *et al*. (2015) – informed by Corbari *et al*. (2017). The overall approach was adapted from Richardson *et al*. (2018) and the final values are shown in Table 3. Seasonal *Kc* values were then converted into an annual mean value, the 21 LCM classes were converted into the 1 km aggregate classes, and the *Kc* values were applied to the *peti* data using Equation 1.

$$ActualET\ (AET) = peti\ x\ Kc$$

Equation 1

Monthly loch water retention time calculations

Water retention times ($T\omega$, years) for all lochs greater than 1 ha in surface area were calculated for use in the chlorophyll-*a* models using Equation 2, where *EffRain* (m) is the mean annual hydrologically effective rainfall across each catchment.

$$T\omega = \text{Vollake}(m^3)/\text{Area}_{catch}(m^2) * EffRain(m)$$

Equation 2

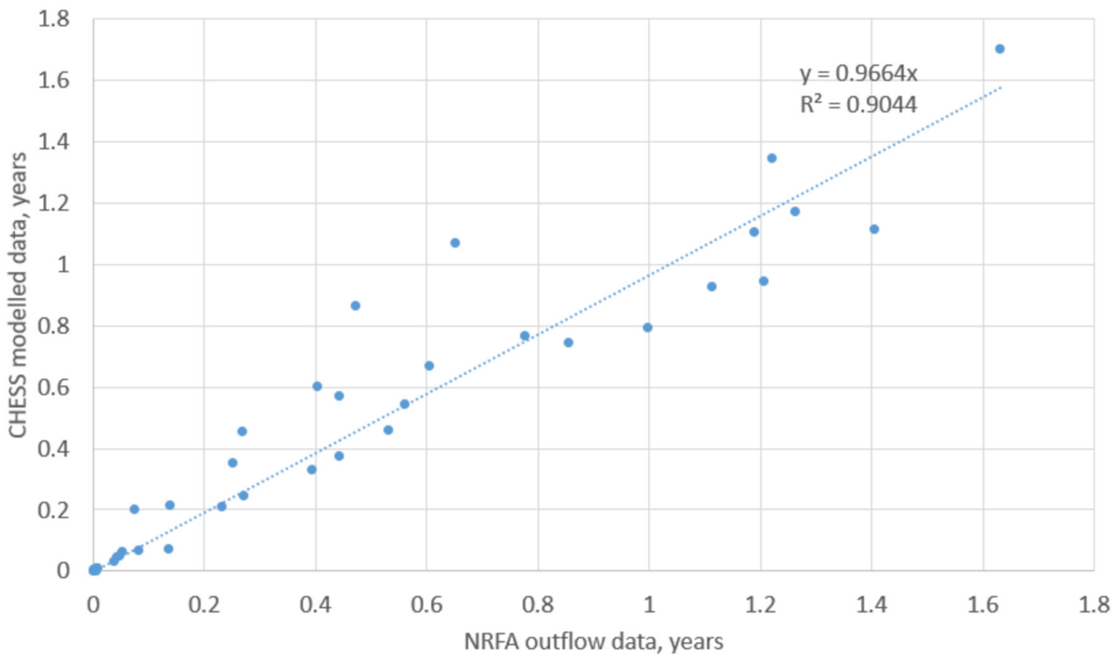

Figure 3 Relationship between annual average lake water retention times, 2004-2013, based on NRFA measured outflow data and CHESS modelled outflow data. Scatter plot shows raw data points and line of best fit.

Retention times were then calibrated using the relationship derived by May *et al*. (2022b), which compared predicted values to measured outflow discharges at relevant National River Flow Archive (NRFA) gauging stations (Figure 3).

Climate change scenarios variable extraction

The modelling work and climate scenarios focused on four time-points: 2020, 2040, 2060 and 2080. The TP model used the five years of prior data to define each time-point. For example, the value for 2020 (baseline period) was defined using monitoring data from 2015-2019. Similarly, future climate data derived from projected data for 2035-2039 were used for 2040, 2055-2059 for 2060 and 2075-2079 for 2080.

Monthly water retention times and air temperatures were taken for these five-year time periods, giving 60 monthly values. Due to the issue of retention times heading towards infinity when EffRain approached 0, the median value was chosen to represent monthly retention times for each time period. For air temperature, a mean value was used. Total annual volume of *EffRain* falling over the catchment was also calculated for modelling purposes, using the same five time periods. A mean monthly value for *EffRain* was calculated for each of the 60 values, then these were converted into annual means. Finally, the annual means were converted to volume expressed in cubic metres (m^3) for consistency with other model parameters.

Table 3 Seasonal and annual crop coefficient (Kc) values used for each land cover class (lc) showing the source of each value. Values adapted from FAO (1998) and Nistorand Porumb (2015).

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 & Porumb
2	Coniferous Woodland	1	1	1	1	1.00	Nistor and Porumb
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 and Porumb
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 and Porumb
13	Saltwater	0.3	0.7	1.3		0.77	Nistor and Porumb
14	Freshwater	0.25	0.65	1.25		0.72	Nistor and Porumb
15	Supra-littoral Rock	0.15	0.2	0.05		0.13	Nistor and Porumb
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 and Porumb
18	Littoral sediment	0.15	0.15	0.15		0.15	FAO
19	Saltmarsh	0.1	0.45	0.8		0.45	Nistor and Porumb
20	Urban	0.2	0.4	0.25		0.28	Nistor and Porumb
21	Suburban	0.1	0.3	0.2		0.20	Nistor and Porumb

Climate change scenarios specifications

Climate change scenarios were used to project loch conditions (in terms of TP and chlorophyll-*a* concentrations) into the future. Representative Concentration Pathways (RCPs) are a method for capturing climate assumptions within a set of change scenarios (van Vuuren et al., 2011). Four RCPs are available within UKCP18/CHESS-SCAPE (2.6, 4.5, 6.0, 8.5), with each number representing the radiative forcing targets for 2100 in watts per square metre (W m^{-2}). Figure 4 shows differences in relative warming among these RCPs compared to the older Special Report on Emissions Scenarios (SRES) reported in UKCP09.

Alternative futures in the British land use system were taken into account when projecting TP levels using the CRAFTY-GB model (Brown *et al.*, 2022). This model uses five Shared Socioeconomic Pathways (SSPs) to predict different land use change scenarios (Figure 5). Each of these SSPs maps onto an individual RCP so predicted *EffRain* and temperature values were taken from the appropriate CHESS-SCAPE RCP to create a combined dataset.

Preparation of variables for phosphorus and chlorophyll-*a* modelling

To predict in-lake TP concentrations, two of the 13 scenarios mentioned above (based mainly on CRAFTY-GB SSPs) were used. These were RCP2.6 x SSP1 and RCP6.0 x SSP3. The outputted spatial data were then extracted using the loch catchments, providing an annual TP load (kg) for each loch and scenario. As the raster values were in kg TP

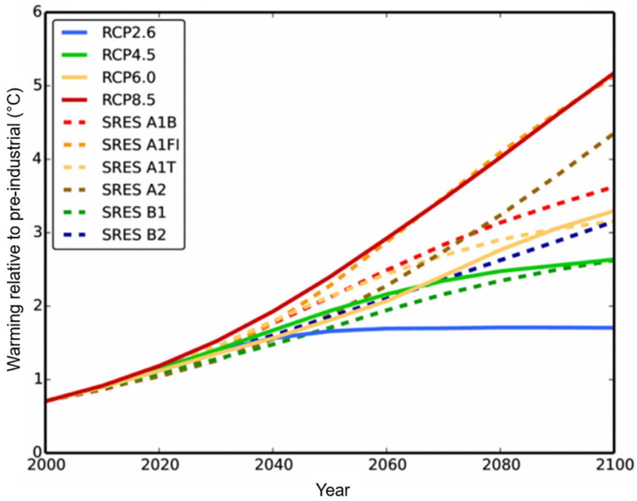

Figure 4 Global mean temperature (°C) projections from a climate model (MAGICC6) relative to a pre-industrial average (1850-1900) for RCP2.6 (blue), RCP4.5 (green), RCP6.0 (yellow) and RCP8.5 (red); the older SRES scenarios (dashed coloured lines) are also shown (after MET Office, 2018). See text for details.

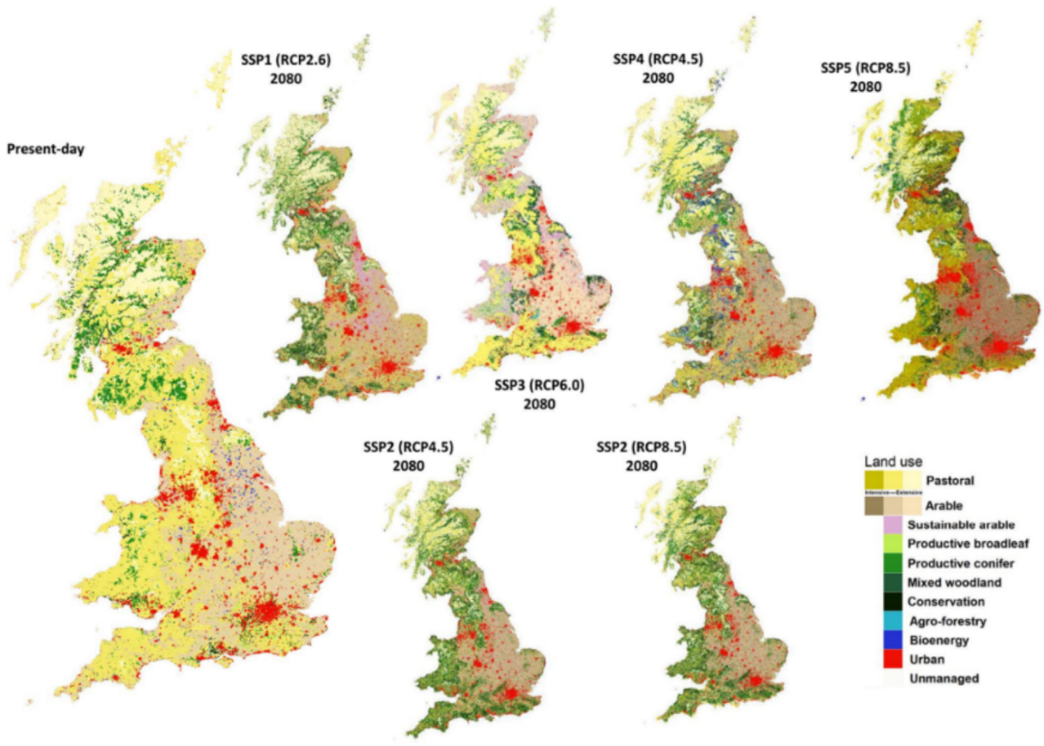

Figure 5 UK land use projections for 2080 across different combinations of Shared Socioeconomic Pathways (SSPs) and CHESS-SCAPE Representative Concentration Pathways (RCPs) (after Brown *et al.*, 2022).

ha^{-1} year^{-1}, and each cell represented one hectare, the values within the catchment were summed to provide estimated TP loads (kg) to each waterbody.

To create outputs ready for the chlorophyll-*a* modelling, each of the TP scenarios was matched to the relevant RCP to obtain corresponding values for *EffRain* and air temperature (e.g. SSP3 was matched to RCP6.0, and so on). Other waterbody characteristics relevant to each loch were obtained from the UK Lakes Portal. Projected air temperature was used to predict loch temperature using the relationship established by May *et al*. (2022b; Figure 6).

The chlorophyll-*a* model was built from SEPA monitoring data, i.e. using in-loch concentrations of TP from 142 monitored lochs. To convert catchment load into in-loch TP concentration, the Combined Organization for Economic Cooperation and Development (OECD) equation was used from Vollenweider and Kerekes (1982).

This predicted in-lake TP concentration (mg m^{-3}) for all scenarios using the data generated from the appropriate RCP x SSP scenarios. The baseline model outputs were then compared to the SEPA monitoring data to assess the viability of using the OECD model to predict TP concentrations in Scottish lochs. Due to consistent under-predicting of the measured loch TP concentrations, a new coefficient was developed for Scottish standing waters, as shown in Equation 3.

$$[P]_\lambda = 4.65X^{.82}$$

where $X = \frac{[P]_j}{1+\sqrt{T_w}}$

and $[P]_j$ = annual mean inflow of total phosphorus (mg/m³)

Equation 3

The difference between this coefficient for Scotland and the lower values given in most of the OECD equations published by Vollenweider and Kerekes (1982), may be due to the in the fact that the OECD equations were developed on monitoring data that were (a) from different climatic zones, and (b) had been pre-filtered to remove all waterbodies with evidence of internal phosphorus release from sediments and/or coloured water. When the combined OECD model was applied to the Scottish monitoring data, the measured phosphorus data were approximately three times higher than the predicted values.

The datasets prepared for running the chlorophyll-*a* model for each scenario contained the following variables:

- WBID – Waterbody ID from the UK Lakes Portal
- NAME – Waterbody name
- CTAREA – Catchment area (ha)
- Predicted in-lake TP concentration (mg m^{-3})
- Mean water temperature (°C) for associated time-point/RCP
- Median retention time (years) for associated time-point/RCP
- Annual effective rainfall volume (m^3) for associated time-point/RCP
- WBLAT – Latitude of lake centroid
- WBLONG – Longitude of lake centroid
- MNDP – Mean depth (m)
- WBALT – Altitude of lake centroid (m)
- WBSAREA – Waterbody area (ha)
- UK_GEOL_TYPE_ – Catchment geology type (categorical)
- UK_HUMIC_TYPE – Catchment humic type (categorical)

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Appendix 3 Effects of climate change and catchment based mitigation measures on P loads to lakes

Background

Total phosphorus (TP) pollution remains an important cause of water quality impairment and eutrophication worldwide. Diffuse pollution from agriculture is the second most important cause of the failure of freshwater bodies to achieve Good Ecological Status (GES) under the European Water Framework Directive (WFD).

Soil type, climate, landscape characteristics and land management contribute to diffuse P water pollution, with surface runoff and erosion being the principal source of P loss in cultivated, drier soils while P loss through drains is the dominant pathway in improved grasslands on wetter soils (Cloy *et al*., 2021).

Aims and objectives

The aim of this part of the study was to simulate the size of the terrestrial losses of total P (TP) within the study area likely to be delivered to standing waters via surface and sub-surface pathways. Also, the effect of land-based mitigation measures on simulated TP loads was explored under a range of combined climatic and land cover change scenarios.

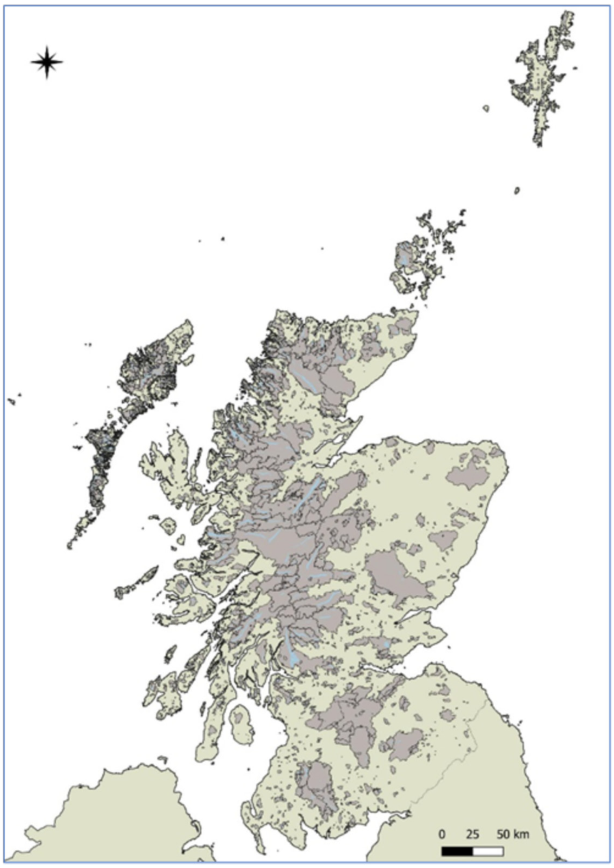

Figure 6 Catchment boundaries of Scottish standing waters included in this study.

Methods

BBN Model description

The study area comprises of the catchment boundaries of the lakes used in May *et al*. (2022a,b) and covers 28,781 km^2 (Figure 6).

Total phosphorus (TP) loads delivered from land to standing waters were simulated using a spatial, hybrid Bayesian Belief Network (BBN) developed previously by Glendel *et al*. (2022); this includes discrete and continuous variables and integrates the available understanding of key processes related to soluble reactive phosphorus (SRP) pollution risk along the full P transfer continuum from source to impact along surface and sub-surface pathways. The BBN model comprises five sub-modules: a) hydrology; and four sources sub-modules simulating losses from b) diffuse sources - both through drains and by soil erosion; c) incidental losses from farmyards; d) sewage treatment works (STWs); and e) septic tanks (STs). Modules a-c and e are conceptualised to simulate the risk of TP losses from spatially distributed 100 × 100 m raster cells in kg^{-1} ha^{-1} yr^{-1}, while module d simulates losses for each individual STW lying within the study area. A detailed description of the structure and numerical and spatial data requirements of the BBN model is provided by Glendel *et al*. (2022) and the accompanying Supplementary Material.

The BBN model was initially developed to simulate SRP losses from cultivated land; for the purpose of this project, we extended model capabilities to simulate TP losses in cultivated land and areas covered by seminatural vegetation. To this end, we amended the representation of TP losses from drains, soil erosion and STWs. TP losses from septic tanks and farmyards were not changed from the original model, because the former was already based on expected TP concentrations and there was not sufficient information to represent TP losses from farmyards. Hence, we assumed that SRP and TP losses from these two sources fell within similar concentration ranges.

Scaling of predicted SRP losses from drains and soil erosion to TP involved using P species fractions identified by Stutter *et al*. (2008) for catchments with different land cover types. Firstly, we classified the 'Crops' categories included in the original BBN model into two groups: the 'arable' group included intensive grassland and the intensive and extensive arable crop categories. All other crop categories were classified as 'seminatural'. Secondly, we used data on all P fractions available for TP loss from erosion and for TP loss from drains of Stutter *et al*. (2008; Table 5 and Table 6) to calculate the proportion of SRP as part of TP in catchments dominated by the two land cover types. This allowed us to account for dissolved organic P (DOP) and particulate P (PP) not included in the original model predictions. The Tarland Burn catchment studied in Stutter *et al*. (2008) was considered to be representative of agricultural catchments, whilst the River Dee catchment was considered to be representative of semi-natural catchments. The regression equation calculating SRP release from soils, based on Morgan P status (Glendell *et al*., 2022), was then adjusted to predict TP concentration by dividing the predicted SRP concentration by a uniform distribution representing the likely minimum to maximum fraction of SRP to TP in catchments with different land cover types (Table 4 below).

Losses of TP from STWs for different treatment types were calculated using data from Yates *et al*. (2019). This provides mean and standard deviation of TP concentrations for STW primary (11.38 µg L^{-1}, 2.65 µg L^{-1}), secondary (4.71 µg L^{-1}, 1.94 µg L^{-1}) and tertiary treatment (0.68 µg L^{-1}, 0.56 µg L^{-1}) types.

All spatial data layers were processed and harmonised to a common 100 m grid cell resolution using the open-source software QGIS 3.22 (QGIS.org, 2023). Losses from diffuse sources (soil erosion and leaching to drains) were driven by land use composition and soil characteristics, using published soil erosion rates by land use (Table 6) and soil erosion risk classes (Table 7). TP loss from soil erosion is based on soil erosion rates available by land use class and adjusted by the probabilities of soil erosion occurring as determined by soil type and calculated soil erosion risk. Land cover composition for the current (baseline) condition was determined using two data layers generated by UKCEH for 2020, which was the most recent, common available year: a) Land Cover® plus: Crops for 2020 (https://www.ceh.ac.uk/data/ceh-land-cover-plus-crops-2015#product), available in ESRI shapefile (polygon) format with a minimum mapping unit of 2ha, which was used to determine crop type within cultivated land and b) Land Cover Map (LCM) for 2020 (Morton *et al*., 2021), available in raster 10m gridded data at UK Biodiversity Action Plan (BAP) Broad Habitat (BH) level; this was used to determine land cover composition for the remainder of the study area. Crop and BH categories were aggregated and harmonised into the eight land use classes shown in Table 5.

Most of the study area was covered by Wildscape (~42%), followed by Rough grasslands (~26%), Improved grasslands (~12%) and Forestry (~11%) (Table 5). Land use composition reflects the dominance of seminatural vegetation (Wildscape and Rough grasslands) in the study area, which reflects the remote nature of many of the lake catchments. Only ~5% of the study area was arable land (extensive and intensive farming).

TP losses to drains were simulated for cultivated land (arable and pastures), only. Records of where artificial field drains have been installed were not available for cultivated areas of Scotland, so their location and distribution needed to be inferred. We used the approach of Lilly *et al*. (2012), who estimated that almost all soils in Scotland that were under cultivation and had inhibited natural drainage (i.e. imperfect, poor or very poor drainage classes) had artificial drainage systems. To identify which soils within the study area were likely to have artificial soil drainage, we overlayed areas of

Table 4 Minimum to maximum fractions of SRP to TP used to scale predicted SRP losses in different catchment types based on Tables 4 and 5 of Stutter *et al*. (2008).

Example catchment	Erosion losses	Leaching to drains
Arable catchment (Tarland Burn)	0.16-0.22	0.13-0.6
Semi-natural catchment (River Dee)	0.03-0.12	0.44-0.52

Table 5 Area (km^2) and respective coverage (%) of land cover classes based on UKCEH LCM 2020 and Land cover plus: Crops within the study area.

Land cover class	Area (km2)	Cover (%)
Rough grassland	7,347	25.5
Grassland unimproved	17	0.1
Grassland improved	3,474	12.1
Arable extensive	1,382	4.8
Arable intensive	70	0.2
Forestry	3,079	10.7
Woodland	441	1.5
Wildscape	12,056	41.9
Other (coastal and built up areas)	916	3.2

imperfect, poor and very poor soil drainage with cultivated areas based on the aggregated land use classes of improved grassland and extensive and intensive arable shown in Figure 8.

Required soils information was defined using an available digitally-derived map of soil type (series) at a 50 m grid cell resolution, which had been generated previously by disaggregating the National Soil Map units (Soil Survey of Scotland Staff, 1981) using a predictive soil modelling technique to derive a digital soil map (Gagkas and Lilly, 2019). This disaggregated soil has recently been used to map areas of likely wetlands in Scotland (Hare *et al*., 2022). The map was upscaled to 100 m grid cell resolution and then linked to soil series information from the Scottish Soil Database to produce the required soil property maps, namely soil erosion risk class, soil drainage class and Hydrology of Soil Types (HOST) classes maps.

For the baseline 2020 simulations, we used mean annual hydrologically effective rainfall (*EffRain*, mm) for 2015-2019 using 1km^2 gridded monthly data (Robinson *et al*., 2020). These values had been calculated as the difference between observed precipitation and potential evapotranspiration (*PET*) by May *et al*. (2022b).

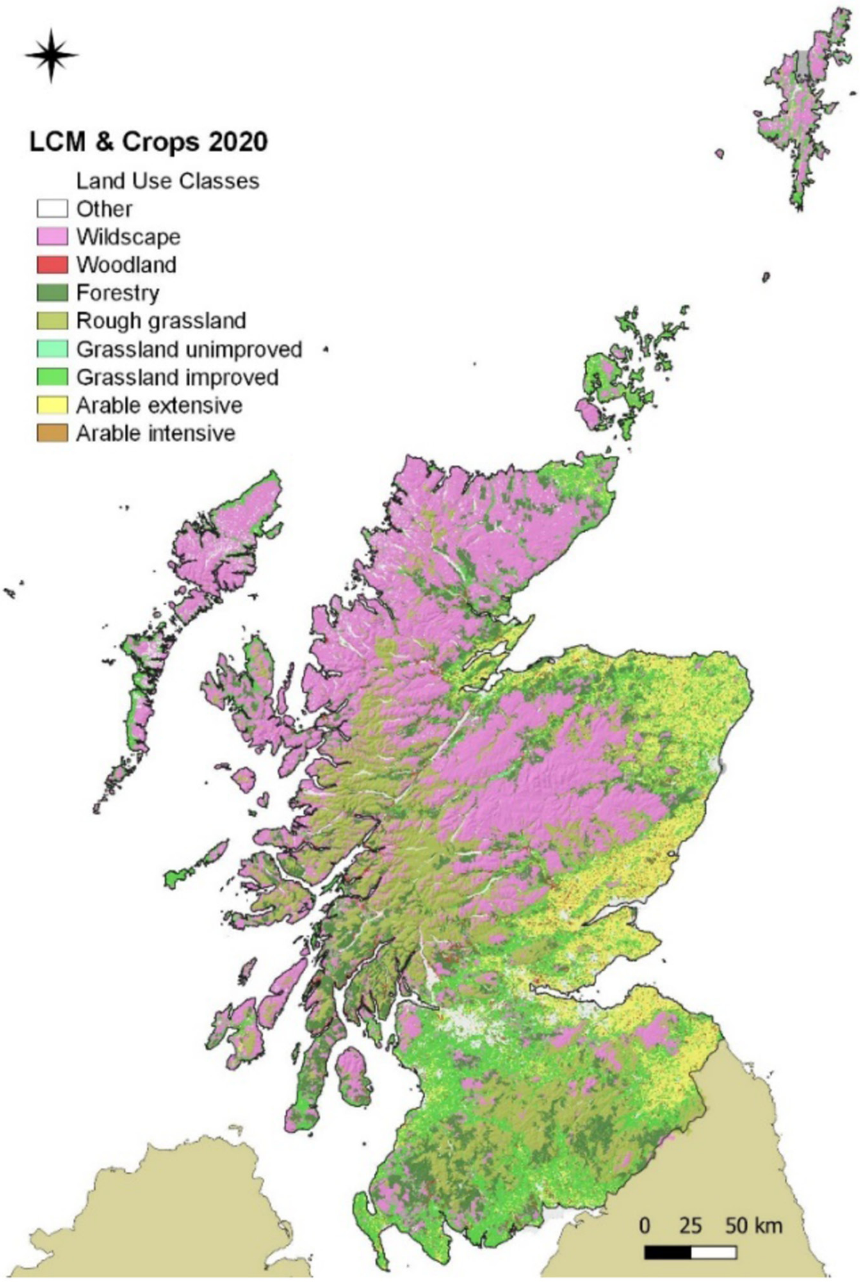

Figure 7 Map of BBN model land use classes for 2020 generated from UKCEH Land Cover Map and Land Cover Plus: Crops © and database right UKCEH, © and database right RSAC. All rights reserved. © Crown copyright and/or database right 2007. Licence number 100017572. © Crown copyright and database right (2023) Ordnance Survey Licence Number AC0000812928

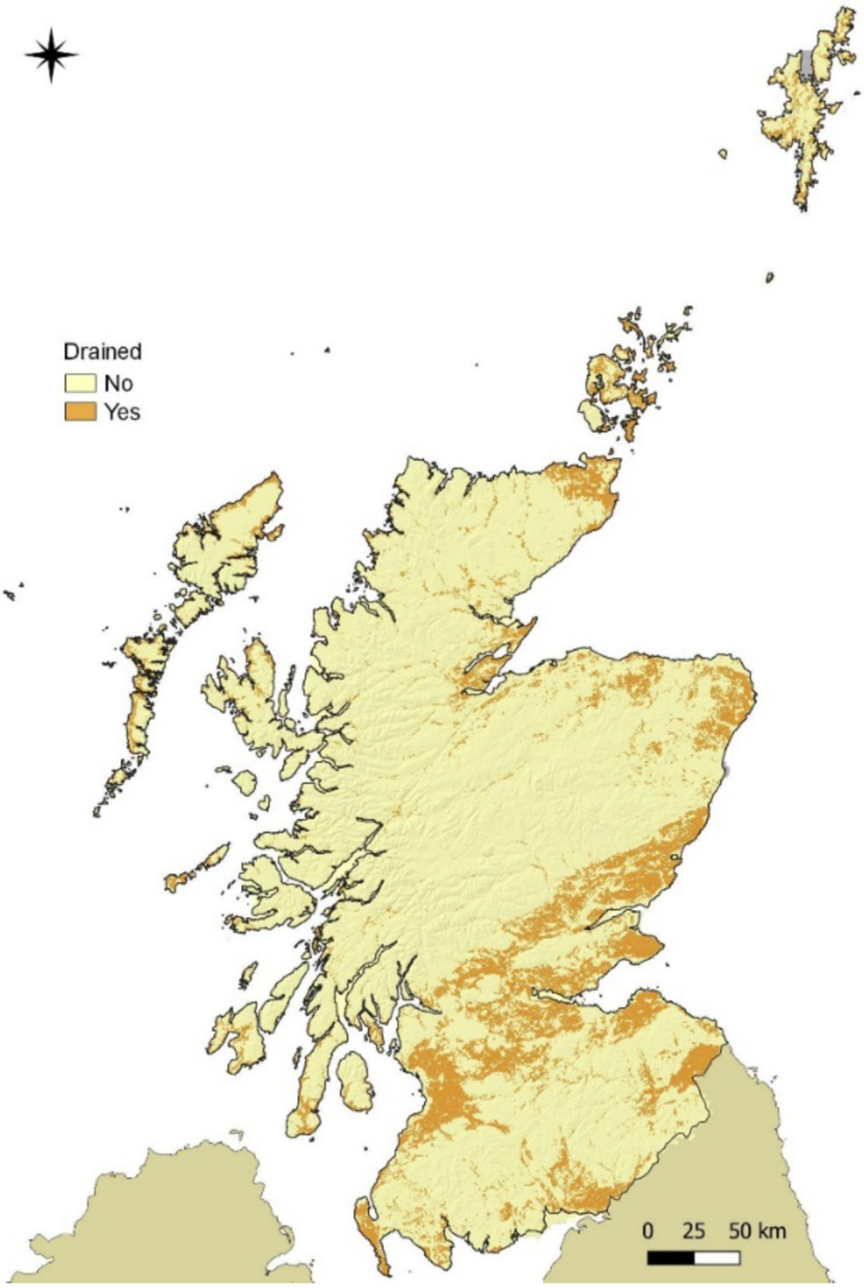

Figure 8 Land cultivated and likely to have been drained based on UKCEH Land Cover Map and Land Cover Plus: Crops information for 2020 and information on soil drainage class. LCM Plus Crops© and database right UKCEH, © and database right RSAC. All rights reserved. © Crown copyright and/or database right 2007. Licence number 100017572. © Crown copyright and database right (2023) Ordnance Survey Licence Number AC0000812928.

Table 6 Soil erosion rates (t ha^{-1} yr^{-1}) by Land Use class and soil type (Rickson *et al*., 2019).

Land Use class	Mineral soils	Organo-mineral soils and Peat
Rough grassland	0.75	0.39
Grassland improved	3	1
Grassland unimproved	2.07	0.39
Arable intensive	4.3	10
Arable extensive	2.4	5
Forestry	0.6	0.13
Woodland	0.6	0.13
Wildscape	0.6	0.13

Table 7 Probabilities of soil erosion by soil erosion risk class (Rickson *et al*., 2019).

Erosion risk class	Mineral soils	Organo-mineral soils and Peat
Low	2%	12%
Moderate	13%	12%
High	24%	31%

CRAFTY-GB

We used modelled land cover in the study area for 2040, 2060 and 2080 from CRAFTY-GB (Brown *et al*., 2022) to simulate the impact of future land use change on TP losses. The CRAFTY-GB land use scenarios were simulated in a globally-embedded agent-based modelling framework that used paired socio-economic and climatic scenarios in the GB land system. These were in the form of stakeholder-elaborated Shared Socioeconomic Pathways (SSPs, representing alternative socio-economic trajectories) and Representative Concentration Pathways (RCPs, representing alternative greenhouse concentration trajectories).

We selected two RCP-SSP combinations from the CRAFTY-GB modelling, RCP2.6 x SSP1 and RCP6.0 x SSP3 to represent the future alternative scenarios. These were, respectively:

- **SSP1 – Sustainability:** UK-SSP1 shows the UK transitioning to a fully functional circular economy as society quickly becomes more egalitarian leading to healthier lifestyles, improved well-being, sustainable use of natural resources, and more stable and fair international relations. It represents a sustainable and co-operative society with a low carbon economy and high capacity to adapt to climate change. Distinguishing features in CRAFTY-GB include the adoption of novel forms of sustainable agriculture with strong societal support, low demand levels for livestock products, with a preference for grass-fed production and for native tree species in forestry. From a land use perspective at a UK-scale, these scenarios lead to a decreasing area of intensive agriculture, with a move away from livestock production and a decrease in pastoral area.
- **SSP3 – Regional rivalry:** UK-SSP3 shows how increasing social and economic barriers may trigger international tensions, nationalisation of key economic sectors, job losses and, eventually, a highly fragmented society. It represents a society where rivalry between regions and barriers to trade entrench reliance on fossil fuels and limit capacity to adapt to climate change. Distinguishing features in CRAFTY-GB include large increases in all capitals and trade barriers that reduce food imports. From a land use perspective at UK-scale, food production dominates land uses with other ecosystem services being by-products of enforced low-intensity management.

Table 8 shows the correspondence between the Land Use classes used in the BBN model (Table 5) and the CRAFTY-GB Land Use agents, many of which are based on the LCM version of 2015 so they can be aligned well with the LCM and Crops typology used in the BBN model. Note that there is no CRAFTY-GB Land Use agent corresponding to the “Grassland Unimproved” Land Use class. Using this correspondence, we translated the CRAFTY-GB Land Use agent raster layers at 1 km grid cell resolution to the Land Use classes of Table 8, downscaled the translated layers to 100 m grid cell resolution and used them in the BBN model for running the TP simulations for the two RCP x SSP combinations for 2040, 2060 and 2080.

For the 2040, 2060 and 2080 simulations, we calculated mean annual EffRain for 2035-2039, 2055-2059 and 2075-2079 using 1km^2 gridded monthly data (Robinson *et al*., 2022), which have been calculated as the difference between projected precipitation and potential evapotranspiration from CHESS-SCAPE (Robinson *et al*., 2022) for both RCP2.6 and RCP6.0. For the baseline 2020 simulation, land cover from 2020 was used, but for consistency with CRAFTY-GB simulations, we used CHESS-SCAPE projected climatic information for RCP2.6 and RCP6.0 to calculate *EffRain* for 2015-2019.

Table 8 Correspondence of CRAFTY-GB Land Use Intensity agents with land use classes used in the BBN model.

CRAFTY-GB Land Use agents	Land Use classes
Extensive pastoral	Rough grassland
Intensive pastoral	Grassland improved
Extensive arable	Arable extensive
Sustainable arable	Arable extensive
Bioenergy	Arable intensive
Intensive arable (fodder)	Arable intensive
Intensive arable (food)	Arable intensive
Productive native conifer	Forestry
Productive non-native conifer	Forestry
Agroforestry	Woodland
Multifunctional mixed woodland	Woodland
Native woodland (conservation)	Woodland
Productive native broadleaf	Woodland
Productive non-native broadleaf	Woodland
Unmanaged	Wildscape
Very extensive pastoral	Wildscape
Urban	Other

Results

Calibration

Simulated TP losses for the baseline scenario for 2020 were calibrated against observed (measured) TP in-stream water samples collected by SEPA from ten locations upstream of six Scottish loch systems. Seminatural land, consisting mainly of upland moorlands and forestry, dominated the catchment areas of four of these catchments, while cultivated land (arable and improved grasslands) was the dominant land cover in the remaining two. Mean and median summary statistics of TP concentrations (in mg L^{-1}) were calculated from stream water samples collected between 2015 and 2019, to match the period of EffRain used in the BBN model simulations for 2020. Simulated TP losses within each of the six lake catchments were summed and converted to flow-weighted concentrations using runoff represented by *EffRain* for 2015-2019. Results showed that simulated TP matched the observed TP concentrations relatively well in the cultivated catchments; for example, simulated catchment and mean observed TP concentrations were the same (0.04 mg L^{-1}), while simulated TP in Strathclyde Loch was ~0.04 mg L^{-1} compared to a mean observed concentration of 0.17 mg L^{-1}. However, the BBN model underestimated TP losses from seminatural land because simulated TP concentrations were an order of magnitude lower than observed TP in streams located downstream of the catchment areas in more seminatural loch systems. This is due to the fact that the regression equation estimating the loss of SRP on the basis of its association with Morgan P is derived from mineral soils in the agricultural Lunan catchment and data to characterise the association between SRP and Morgan P in organic soils is not available. It is expected, that in organic soils, a greater proportion of TP will comprise of organic P forms. These may be under-estimated in semi-natural catchments in our model, even with the additional scaling that was applied. However, greater TP losses are expected from cultivated land, and these are simulated adequately in our model (Glendell *et al*., 2022). In summary, the BBN model outputs provide a plausible gradient of relative magnitude of terrestrial TP losses between different simulated scenarios, needed as input to the lakes modelling work, even if the absolute values in semi-natural catchments are under-estimated.

Mitigation scenarios

The following mitigation scenarios were tested for their impact on TP inputs to standing waters and the effects of those inputs on standing water quality:

- **S1:** Baseline data for 2020 based on observed values
- **S2**: Fertiliser application rate below agronomic optimum level
- **S3**: Fertiliser application rate at agronomic optimum level
- **S4**: Increase in extent of buffer strips
- **S5**: Total losses under maximum mitigation (fertilisers below agronomic optimum and increased in buffer strips)

Figure 9 shows the spatial distribution of TP losses in the study area for the baseline and mitigation scenarios, while Table 9 gives TP losses by land use class for each scenario. For the baseline scenario, TP losses due to soil erosion and leaching to drains accounted for 37% and 36%, respectively, of total TP losses, followed by 14% for combined losses from STs and STWs. TP losses due to soil erosion accounted for 19%, 25%, 36% and 18% of total losses for Scenarios S2, S3, S4 and S5, respectively; the respective figures for TP losses to drains are 30%, 40%, 36% and 30%. For the baseline scenario, losses from improved grasslands and extensive arable land accounted for ~81% of TP losses from all land (60.5% and 20.3%, respectively) (Table 9). The greatest reductions in TP losses are achieved by the mitigation measures of Scenarios S5 (-46.0%) and S2 (-45.7%), whereas Scenario S3 results in a reduction of –19.5% in TP losses, and Scenario S4 leads to a small reduction of only –1.5% in TP losses (Table 9). As expected, reductions in TP losses based on Scenarios S2 and S5 are driven by reductions in TP lost from improved grasslands, whereas most TP losses to drains occur, and from extensive arable land, where soil erosion is driving most of TP loss.

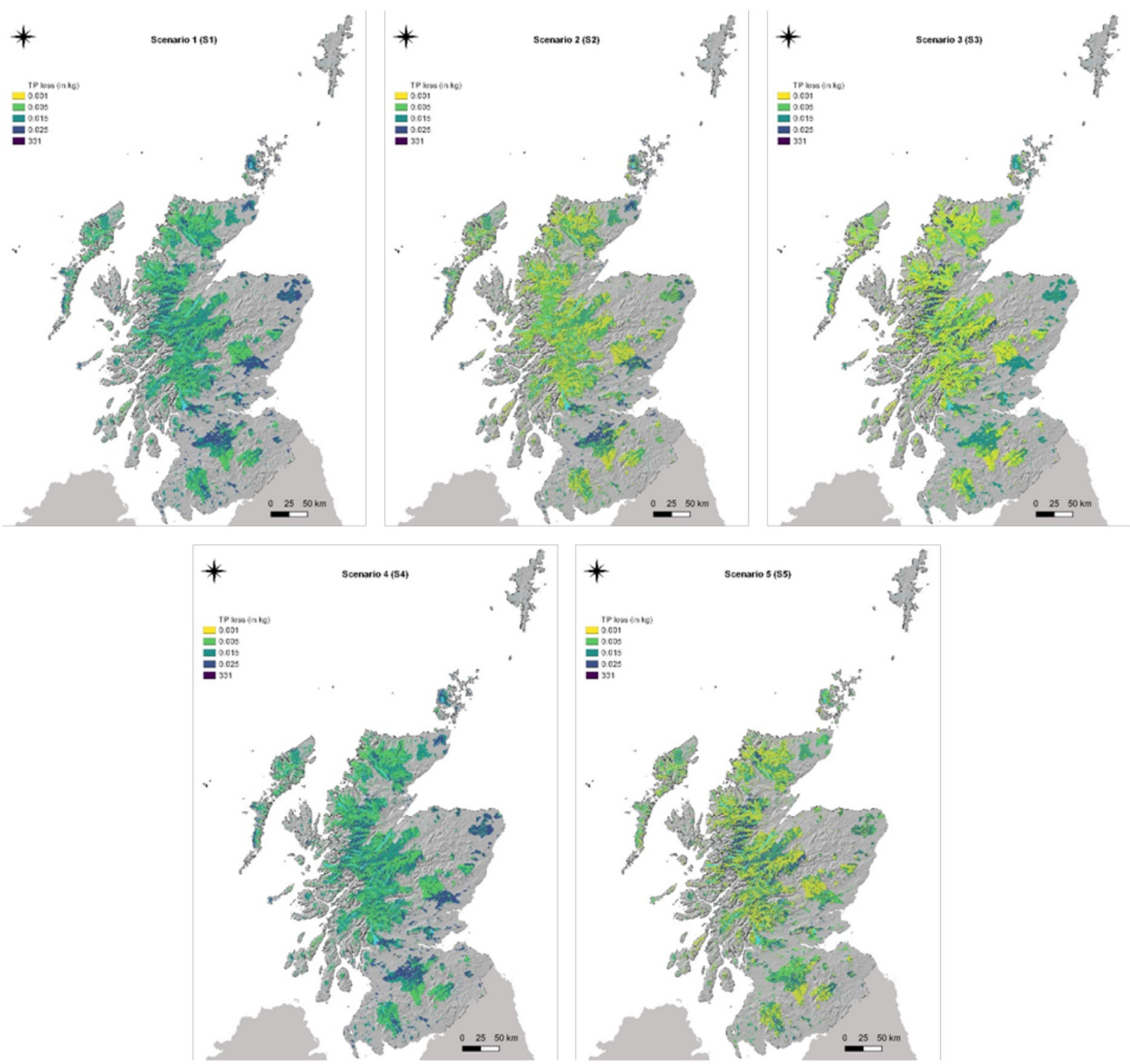

Figure 9 Maps at 100 m grid cell resolution of TP losses (kg yr-1) in the study area for the baseline and mitigation scenarios. Background map is OS Terrain 50 (OS OpenData Plan).

Table 9 Total TP annual losses (kg) by Land Use class for baseline and mitigation scenarios S1-S5. Changes in TP loads are compared to the baseline scenario (see % values shown in brackets).

Land Use class	**Mitigation scenarios**				
	S1	**S2**	**S3**	**S4**	**S5**
Rough grassland	10,373	6,232 (-40%)	6,232 (-40%)	10,373 (0%)	6,232 (-40%)
Grassland unimproved	472	397 (-16%)	397 (-16%)	472 (0%)	397 (-16%)
Grassland improved	120,334	62,441 (-48%)	100,433 (-17%)	120,334 (0%)	62,441 (-48%)
Arable extensive	40,343	22,694 (-44%)	35,612 (-12%)	37,589 (-7%)	22,100 (-45%)
Arable intensive	3,198	1,990 (-38%)	2,716 (-15%)	3,008 (-6%)	1,951 (-39%)
Forestry	3,457	1,467 (-58%)	1,986 (-43%)	3,457 (0%)	1,467 (-58%)
Woodland	2,040	1,844 (-10%)	1,844 (-10%)	2,040 (0%)	1,844 (-10%)
Wildscape	13,724	6,090 (-56%)	6,090 (-56%)	13,724 (0%)	6,090 (-56%)
Other	5,098	4,940 (-3%)	4,994 (-2%)	5,098 (0%)	4,940 (-3%)
Total TP loss	**199,037**	**108,097 (-46%)**	**160,305 (-19%)**	**196,094 (-1%)**	**107,464 (-46%)**

Table 10 Coverages (in %) of each Land Use class within the study area based on Land Cover Map (LCM) and Crops 2020 and CRAFTY-GB Land Use Agents for RCP2.6 x SSP1 and RCP6.0 x SSP3 combinations for 2040, 2060 and 2080.							
Land Use class	Land Cover 2020 (%)	CRAFTY-GB 2040		CRAFTY-GB 2060		CRAFTY-GB 2080	
		SSP1	SSP3	SSP1	SSP3	SSP1	SSP3
Rough grassland	25.5	26.0	25.5	15.8	25.4	13.2	23.8
Grassland unimproved	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Grassland improved	12.1	5.8	3.6	3.8	4.1	3.2	6.2
Arable extensive	4.8	0.8	10.8	1.3	10.7	1.8	8.8
Arable intensive	0.2	3.7	6.2	4.4	6.1	3.6	11.7
Forestry	10.7	11.8	8.6	12.0	8.5	12.8	8.0
Woodland	1.5	5.1	4.8	9.2	4.8	12.1	4.4
Wildscape	41.9	46.5	40.1	53.1	39.8	52.8	36.5
Other	3.2	0.4	0.5	0.4	0.6	0.5	0.7

CRAFTY-GB scenarios

The sustainable scenario of RCP2.6 x SSP1 compared to land use composition in the study area based on land cover information for 2020, shows a decrease in pasture land used for grazing and livestock production (Rough and Improved grasslands) but an increase in intensive farming, and increases in the areal extent of wildscape, forests and woodlands (Table 10 and Figure 10). In contrast, scenario RCP6.0 x SSP3 shows increases in the areal extent of extensive and intensive arable land and a decrease of Improved grasslands compared to land cover in 2020. However, by 2080, the area of Improved grassland under the RCP6.0 x SSP3 is projected to be twice that of the RCP2.6 x SSP1 scenario, while arable land is projected to be almost four times greater under the RCP6.0 x SSP3 scenario compared to under the RCP2.6 x SSP1 scenario (Table 8, Figure 10).

Compared to the baseline scenario for 2020, the RC2.6 x SSP1 scenario led to c. 20% reduction of total P losses in the study area by 2080 that was mainly driven by a significant reduction in TP lost, mainly to drains, from Improved grasslands (Table 11). Overall, TP loss in Improved grasslands decreased from 60% of TP loss in the baseline 2020 scenario to around 16% in 2080, while there was also a small reduction from 20% to 16% in TP losses from extensive arable land between 2020 and 2080. There was also a significant increase in TP lost from intensive arable land, from just 2% in the baseline 2020 scenario to 29% of total P in 2080, but these additional TP loads were offset by the reduced TP loads from Improved grasslands.

Regarding the RCP6.0 x SSP3 scenario, simulated total TP losses more than doubled by 2080 (+138%) compared to the TP losses of the baseline 2020 scenario (Table 11). This significant increase in TP losses was mainly driven by the increase in the areal extent of intensive and extensive agricultural land in the study area, partially at the expense of seminatural land on wet soils, meaning that these areas were likely to be drained to be suitable for agricultural use (Figure 8); this eventually led to eight times more TP lost via erosion and leaching to drains in arable land in 2080 than in 2020. Overall, losses from arable land in 2080 accounted for 50% of all TP losses, more than doubling the 22% estimated from the baseline 2020 scenario. In contrast, TP losses from Improved grasslands accounted for only 13% in 2080 compared to 60% in the baseline 2020 scenario.

Figure 10 Maps of Land Use classes (100m grid cell resolution) generated from CRAFTY-GB Land Use Intensity agents (Brown *et al*., 2022) for RCP2.6 x SPP1 and RCP6.0 x SSP3 scenarios and 2040, 2060 and 2080. Coverages of land use classes for each scenario are given in Table 7.

Table 11 TP losses (kg y^{-1}) by Land Use class based on land cover for 2020 and land cover based on the RCP2.6 x SSP1 and RCP6.0 x SPP3 scenarios for years 2040, 2060 and 2080. Values in brackets show percentage contribution of TP loss from each land use class to total TP losses for each scenario, while values in brackets in the last row give percentage change in total TP losses for each of the scenarios for 2040, 2060 and 2080 compared to TP losses for 2020.

Land Use class	Baseline 2020		CRAFTY-GB 2040		CRAFTY-GB 2060		CRAFTY-GB 2080	
	RCP2.6 x LCM	RCP6.0 x LCM	RCP2.6 x SSP1	RCP6.0 x SSP3	RCP2.6 x SSP1	RCP6.0 x SSP3	RCP2.6 x SSP1	RCP6.0 x SSP3
Rough grassland	10,390 (5%)	10,349 (5%)	16,268 (10%)	13,813 (4%)	10,271 (6%)	13,536 (4%)	6,466 (4%)	12,683 (3%)
Grassland unimproved	473 (0%)	463 (0%)	0 (0%)	0 (0%)	0 (0%)	0 (0%)	0 (0%)	0 (0%)
Grassland improved	121,575 (61%)	117,024 (60%)	50,086 (30%)	33,624 (10%)	32,358 (19%)	37,871 (11%)	25,690 (16%)	58,789 (13%)
Arable extensive	40,561 (20%)	40,046 (20%)	11,071 (7%)	154,985 (44%)	19,607 (12%)	151,371 (44%)	25,148 (16%)	130,417 (28%)
Arable intensive	3,204 (2%)	3,193 (2%)	41,554 (25%)	114,420 (33%)	52,856 (31%)	111,193 (32%)	45,810 (29%)	233,165 (50%)
Forestry	3,459 (2%)	3,455 (2%)	10,120 (6%)	4,822 (1%)	12,135 (7%)	4,754 (1%)	11,574 (7%)	4,501 (1%)
Woodland	2,049 (1%)	2,033 (1%)	13,507 (8%)	9,335 (3%)	15,470 (9%)	9,212 (3%)	18,022 (11%)	8,603 (2%)
Wildscape	13,727 (7%)	13,722 (7%)	20,034 (12%)	15,296 (4%)	23,388 (14%)	14,941 (4%)	24,535 (15%)	13,292 (3%)
Other	5,115 (3%)	5,086 (3%)	1,965 (1%)	2,506 (1%)	2,040 (1%)	2,642 (1%)	2,095 (1%)	2,691 (1%)
Total	**200,552 (100%)**	**195,371 (100%)**	**164,606 (-18%)**	**348,801 (+79%)**	**168,124 (-16%)**	**345,521 (+77%)**	**159,340 (-20%)**	**464,141 (+138%)**

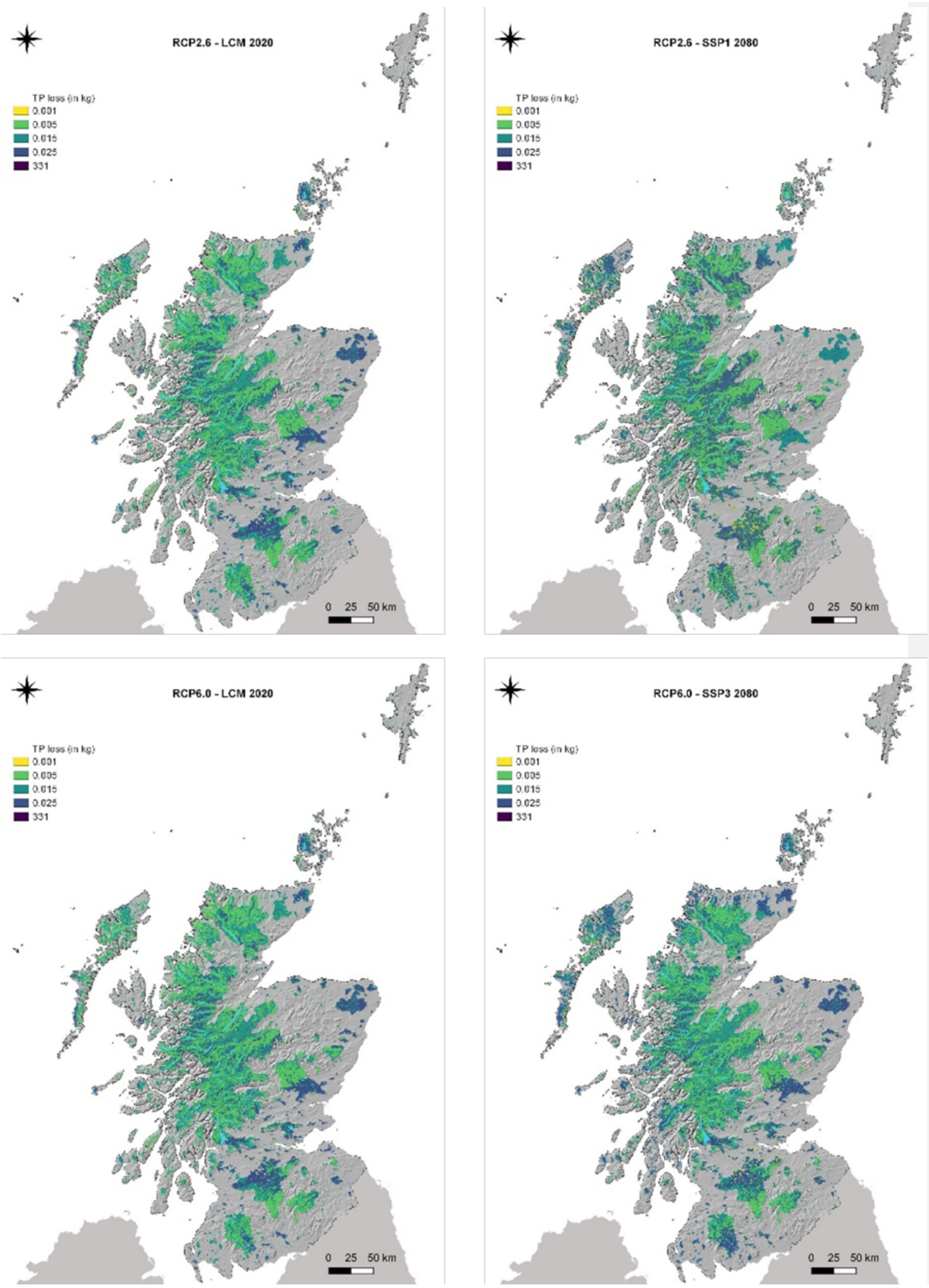

Figure 11 Maps (100 m grid cell resolution) of TP loss (kg yr^{-1}) in the study area for the baseline 2020 and CRAFTY-GB RCP2.6 x SSP1 and RCP6.0 x SSP3 scenarios for 2080.

Conclusions

Targeted extensive land-based mitigation measures focused on the management of soil nutrient status at or below agronomic optimum can help to significantly reduce TP inputs to standing waters from their catchments, in some case by more than 40%. This should include holistic management of soils to maximise soil organic matter content and nutrient use efficiency through regular soil testing and optimising of nutrient inputs. Smaller-scale interventions, such as buffer strips, did not affect TP losses to water significantly at a catchment scale.

Under future scenarios, the sustainable land use reconfiguration under SSP1 associated with the lower emissions under RCP2.6 could reduce TP losses from land to standing waters by up to 20% compared to the current baseline. In contrast, expansion of arable land and intensification of agriculture under the higher emissions scenarios of RCP6.0 and linked to the unfavourable land use changes predicted under SSP3 could more than double TP inputs to standing waters, hence greatly increasing the risks of eutrophication impacts.

Extensive adoption of sustainable agronomic practices, such as those promoted under the Net Zero targets, coupled with future lower emissions pathways and sustainable land use scenarios are the most sustainable adaptation options for reducing future eutrophication risks to Scottish lochs. Conversely, pursuing higher emissions pathways coupled with land use intensification is likely to lead to doubling of current TP inputs to standing waters, leading to potentially irreversible eutrophication impacts such as algal blooms.

More laboratory analyses to allow soil-specific understanding of the association between Morgan P and water extractable SRP and TP in organic soils would help to improve the accuracy of model predictions in catchments dominated by organic soils and seminatural vegetation.

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Appendix 4 Effects of climate change on nitrogen leaching to groundwater

Background

NIRAMS (Nitrogen Risk Assessment Model for Scotland) is a national scale model for predicting nitrate leaching to groundwater (Sample and Dunn, 2014). NIRAMS modelling has in the past been used to support SEPA and Scottish Government in defining and reviewing Nitrate Vulnerable Zones (NVZ) in Scotland. The methodology for designating and reviewing NVZ is mainly based on data from SEPA's nitrate monitoring network but also include additional lines of evidence, including NIRAMS modelling of nitrate leaching, to address variability in results and assess confidence in the outcome. In line with the Nitrate Directive (ND), the review of NVZs takes place every 4 years.

Aims and objectives

The aim of this work is firstly to run NIRAMS using the most up to date climate and land use data to simulate nitrate leaching to groundwater across all of Scotland. Secondly, NIRAMS will be run using climate change projections as input to generate future change scenarios of groundwater nitrate pollution and recharge rates.

Methods

NIRAMS is spatially distributed model that simulates the daily nitrate leaching and loads to groundwater (i.e., the amount of nitrate leaving the bottom of the soil profile) at national scale on a 1 km grid scale basis. NIRAMS and the input data processing is only briefly described in the following. A more detailed description can be found in Sample and Dunn (2014) and Dunn *et al*. (2004a, b).

The model consists of two modules: a water balance module and a nitrate leaching module (Figure 12). The water balance module uses information representing climate, soil properties and land use patterns to predict the daily runoff and drainage. Each 1 km grid cell is considered a 'soil reservoir' for which the water level is calculated on a daily basis. The water storage in the soil reservoir will change depending on the excess water input (i.e., the difference between precipitation and evapotranspiration) and the amount of water leaching from the soil reservoir as either shallow subsurface flow (SSF) or baseflow (GW). The amount of water that leaches from the soil reservoir depends on the 'water level' in the soil reservoir, the field capacity of the soil, and

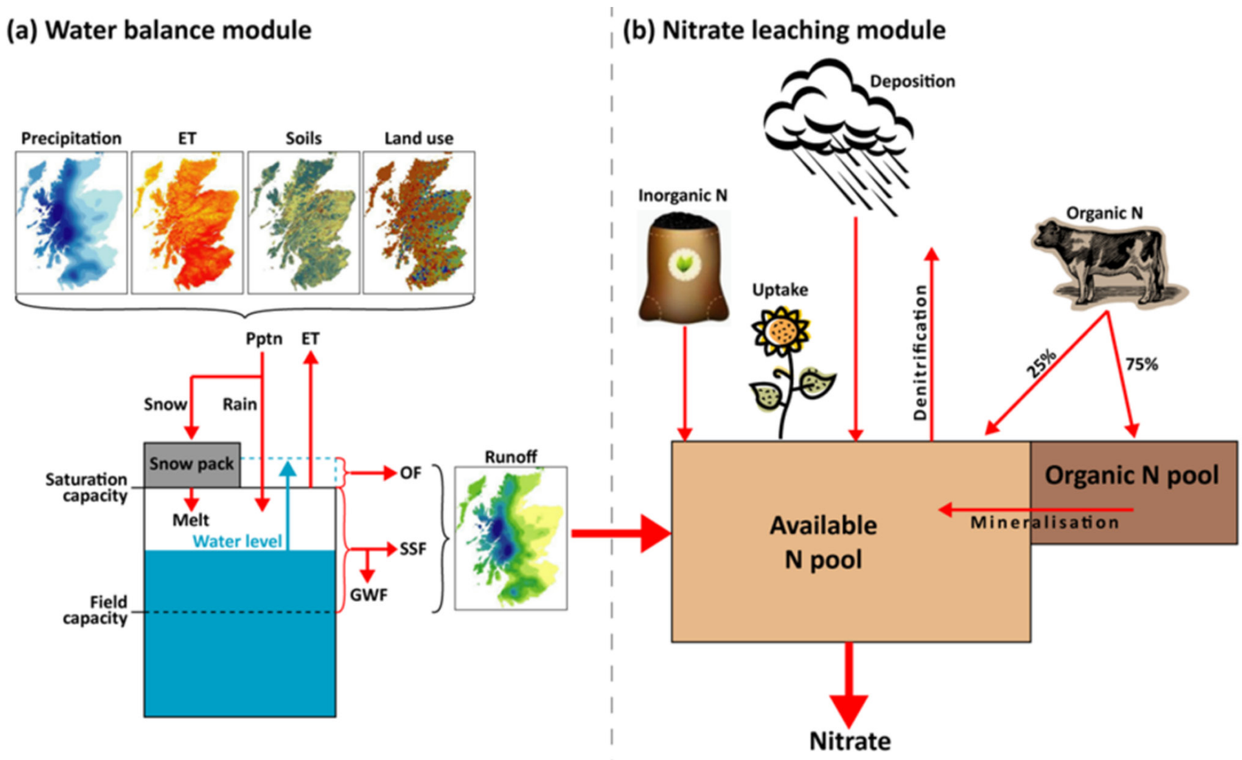

Figure 12 Illustration of the NIRAMS II model structure (after Sample and Dunn, 2014).

some calibration drainage constants that are based on HOST classes. If the water level in the reservoir exceeds the saturation capacity of the soil, this excess water is lost via overland flow (OF).

The nitrate leaching module combines the water balance results with detailed information representing land use and agricultural activities to estimate the daily amount of nitrate leaching from the soil reservoir. This estimation is essentially a simple mass balance, which accounts for daily inputs of organic (from livestock and manures), inorganic (fertilisers), and atmospheric deposition of nitrate, and for daily losses of nitrate due to uptake by crops and vegetation and due to drainage. The mass balance also includes some simple nitrogen fate processes (mineralisation and denitrification) that depend on climate and the soil water storage, and hence are linked to the water balance model.

The key outputs from NIRAMS are 1 km^2 resolution grids representing the amount of water and nitrate following each flow pathway at each time step. These grids can be aggregated both spatially and temporally to give estimates of the total runoff and amount of nitrate leached within particular areas and time periods of interest.

It should be noted that NIRAMS is not a groundwater model, i.e., it does not contain any physical representations of groundwater mixing or the aquifer systems and it also does not consider any routing; it is therefore not appropriate to calibrate the model directly to borehole data. However, by combining the water following the overland, shallow subsurface and groundwater flow pathways, it is possible to use the model to simulate annual average surface water nitrate concentrations. The model was therefore first calibrated and tested against surface water data, and the most promising parameter sets were then compared to the groundwater data to investigate the extent to which groundwater concentrations reflect nitrate inputs from the surface.

Table 12 shows an overview of the spatial inputs required by NIRAMS. The meteorological data was derived from Had-UK (2011-2021) and were all at 5km grid scale, which were then ‘resampled’ to 1km grid scale using a nearest neighbour approach. Monthly potential evapotranspiration (PET) was calculated using the FAO modified Penman-Monteith methodology (Allen *et al*., 1998) using monthly temperature, wind speed, relative humidity, and sunshine hours from Had-UK. The monthly PET data were converted to daily by dividing the monthly PET with the number of days in the month.

The total annual N deposition (2011-2020) was derived from EMEP (European Monitoring and Evaluation Programme) and projected to 1 km resolution. The annual deposition data were converted to daily by dividing by the numbers of days in the year.

The annual estimates of organic and inorganic N inputs as well as N uptake by plants were derived from AGCensus data (2011-2019) and Land Cover Map (LCM) (2011-2021). AGCensus contain gridded data on the distribution of land uses/crop types as well as the number, age, and type of livestock. The amount of organic nitrogen excreted annually by each animal class was taken from manure planning documentation issued to farmers within Scotland's nitrate vulnerable zones (NVZs), and these figures were used to estimate the total amount of organic nitrogen produced each year. These annual organic N estimates were then distributed spatially at 1 km resolution based on appropriate land classes from LCM, using a rule set designed to be broadly compatible with the application limits currently in force within the NVZs.

The LCM and the AGCensus crop distribution data were used together with the results of the British Survey of Fertiliser Practice to obtain annual estimates for the application rate of inorganic nitrogen fertiliser. Table 13 shows the annual rates

Table 12 Overview of spatial inputs required by NIRAMS.

Input	Spatial resolution	Temporal resolution	Source
Rainfall	5 km	Daily	Had-UK
Min & max temperature	5 km	Daily	Had-UK
PET	5 km	Monthly	Calculated from Penman-Monteith and Had-UK data
Total N deposition	0.1°	Annual	EMEP
Organic N input	1-2 km	Annual	Livestock numbers from AGCensus (2 km) and LCM (1 km)
Inorganic N input	1-2 km	Annual	Crop & land use data from AGCensus (2 km) and LCM (1 km)
N uptake	1-2 km	Annual	Crop & land use data from AGCensus (2 km) and LCM (1 km)
Soil field capacity & saturation	Saturation 1 km	–	

Table 13 Inorganic N application and N uptake rates associated with each crop and land use class in NIRAMS.		
Land use class	**Inorganic N ($kg\ yr^{-1}$)**	**N uptake ($kg\ yr^{-1}$)**
Spring barley	130	110
Winter barley	180	140
Spring wheat	150	110
Winter wheat	200	170
Spring oil seed rape	130	100
Winter oil seed rape	190	130
Spring oats	120	105
Winter oats	140	120
Seed potatoes	90	100
Ware potatoes	110	130
Fruit	120	100
Vegetables	100	75
Other arable	130	100
Set aside	0	20
Rough grassland	0	35
Improved grassland	59	105
Woodland and forest	0	20
Short rotation coppice	50	90
Bare and built up	0	0
Water	0	0
Grazed woodland	0	20
Fen, marsh and swamp	0	100
Heathland	0	20
Bog	0	100
Montane	0	20
Saltmarsh	0	100
Other	0	0

of inorganic N application and N uptake associated with each land use class in NIRAMS. The annual estimates of the organic and inorganic nitrogen applied as well as the nitrogen uptake are distributed temporally using a set of idealised time series that define, for a variety of crop classes, the length of the growing season, the amount of nitrogen uptake and the timing of fertiliser application.

For the future scenarios, NIRAMS was run using daily climate change projection data from the CHESS-SCAPE RCP6.0 scenario. The land use and land cover were assumed not to change in the future scenarios, and average land use and land cover conditions for the period 2015-2020 was used for the simulations. Hence the future simulations of the nitrate leaching only accounts for changes in climate.

Results

Figure 13 shows the average annual drainage (mm y^{1}) and nitrate leached (kg N yr^{-1}) over the period 2015-2019. This was assumed to be the equivalent to the average annual drainage (mm y^{1}) and nitrate leached (kg N yr^{-1}) for the 2020 baseline.

Figure 14 shows the simulated future average annual drainage (mm yr^{-1}) and nitrate leached (kg N yr^{-1}) for 2040 (average of the years 2035-2039), 2060 (average of the years 2055-2059) and 2080 (average of the years 2075-2079). Overall, the results of the future scenarios show that the projected climate change has limited effect on the annual amount of N leaching, and that this is more dependent on the land use and potential land use changes. The effect of future land use changes will be explored in future work.

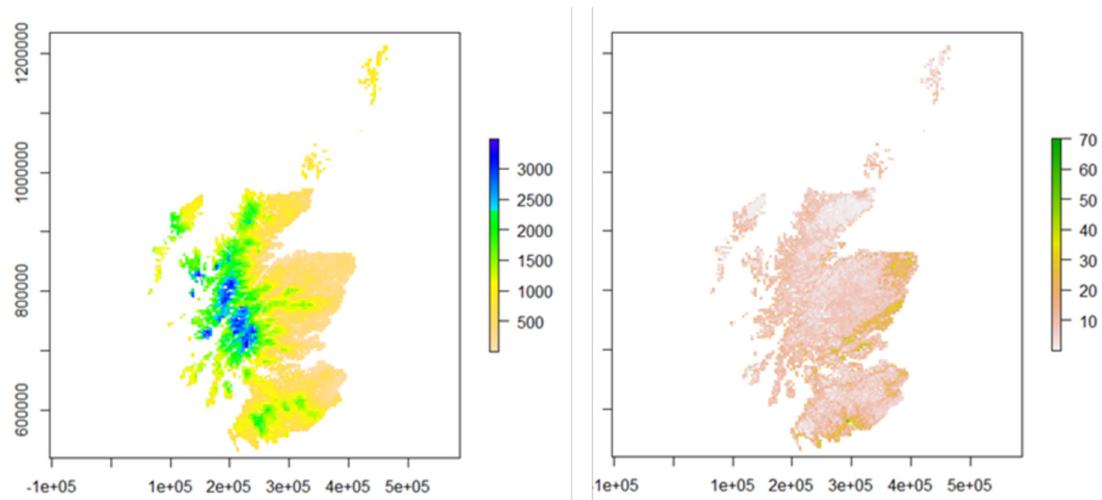

Figure 13 Simulated average annual drainage (mm y^{-1}) and leached N (kg ha^{-1}) for the period 2015-2019.

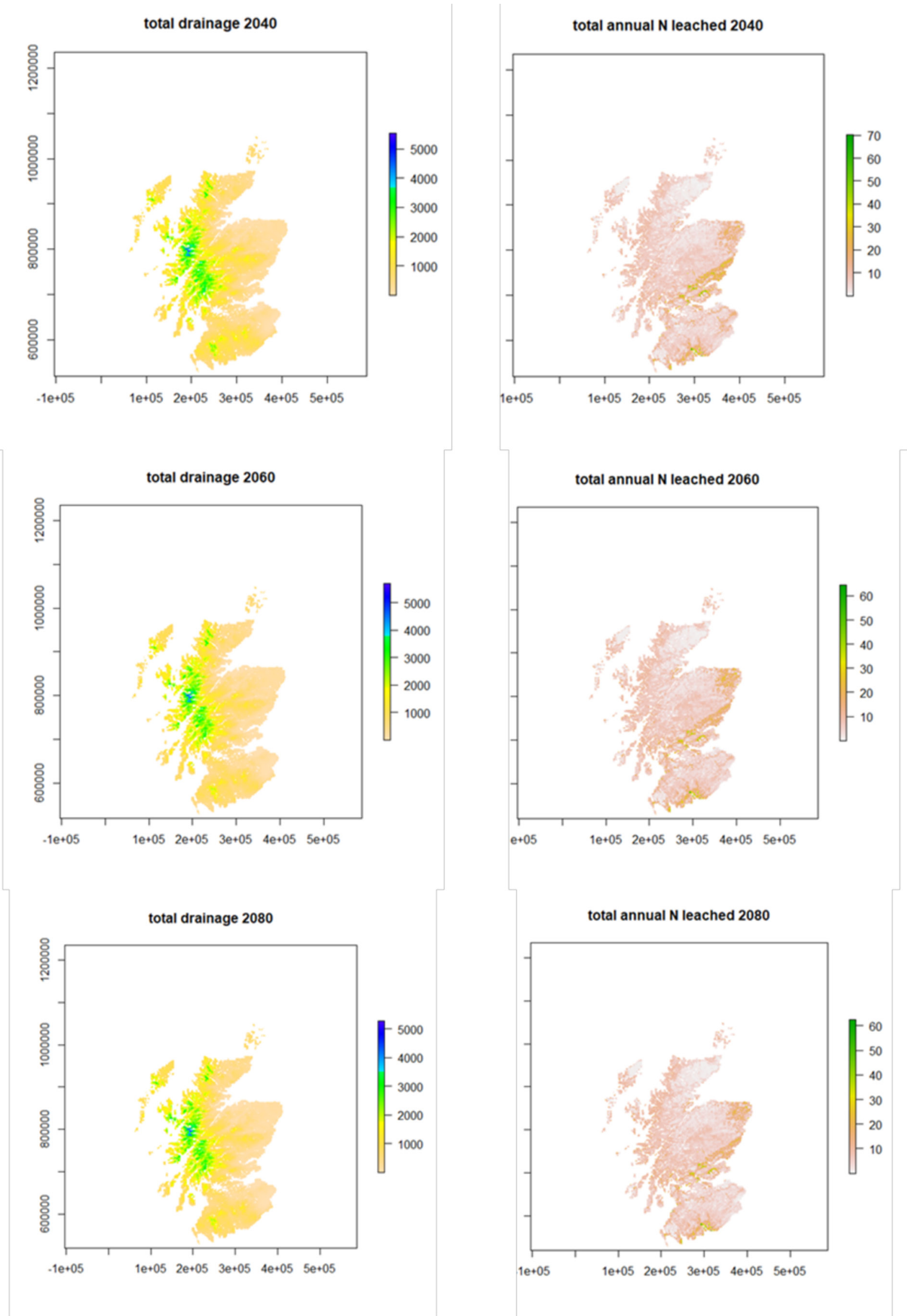

Figure 14 Simulated future annual average drainage (mm yr[-1]) and nitrate leaching (kg N yr[-1]) for 2040, 2060, and 2080 using future climate projections from CHESS-SCAPE RCP6.0 as input.

Conclusions

The modelling suggests that future climate change will have a limited effect on nitrate leaching and, as such, no further analyses were undertaken on this aspect of diffuse pollution. Although climate change may not affect the delivery of nitrogen to water, increasing loch water temperatures are affecting its utilisation with these waterbodies becoming more nitrogen limited in summer (May *et al*., 2022a,b). Nitrogen limitation can increase the likelihood of harmful algal blooms because this favours the growth of cyanobacteria, many of which can use atmospheric nitrogen as a supply of this nutrient.

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Appendix 5 Main drivers of changes in water quality

Background

Phytoplankton ('algal') biomass is a key indicator of how productive freshwater ecosystems are and so it is often used as a biological indicator of water quality. The growth of phytoplankton is determined by a complex mixture of factors that include nutrient concentrations (e.g. phosphorus, nitrogen, and silica), physical habitat characteristics (water temperature, flushing rate, mixing, and transparency), zooplankton grazing rates, waterbody characteristics (e.g. depth), and geographical location (e.g. latitude, longitude, and altitude which can determine prevailing climatic conditions). To understand the potential impacts of climate on the accumulation of phytoplankton biomass ('blooms'), it is important to take the influences of these additional factors into account.

Aims and objectives

The aim of this element of the project was to build a bespoke lake response model that could be used to infer the importance of these factors on phytoplankton growth so that the effects of climate change on water quality could be projected to 2080 across Scottish standing waters. Once developed, the outputs from these models were incorporated into several scenarios of future climate and land management/nutrient inputs to make projections of likely changes in phytoplankton concentrations in Scottish standing waters over space and time.

Methods

Data processing

Available data from SEPA monitoring and the UK Lakes Portal (https://eip.ceh.ac.uk/apps/lakes) were used to develop the response models. A total of 328,422 rows of data (sites x sampling dates x determinands) were available from 142 lochs covering the geographical extent of Scotland (Figure 15). Most of the data were collected post-2000, and fewer measurements were available for December compared to other months of the year (Figure 16).

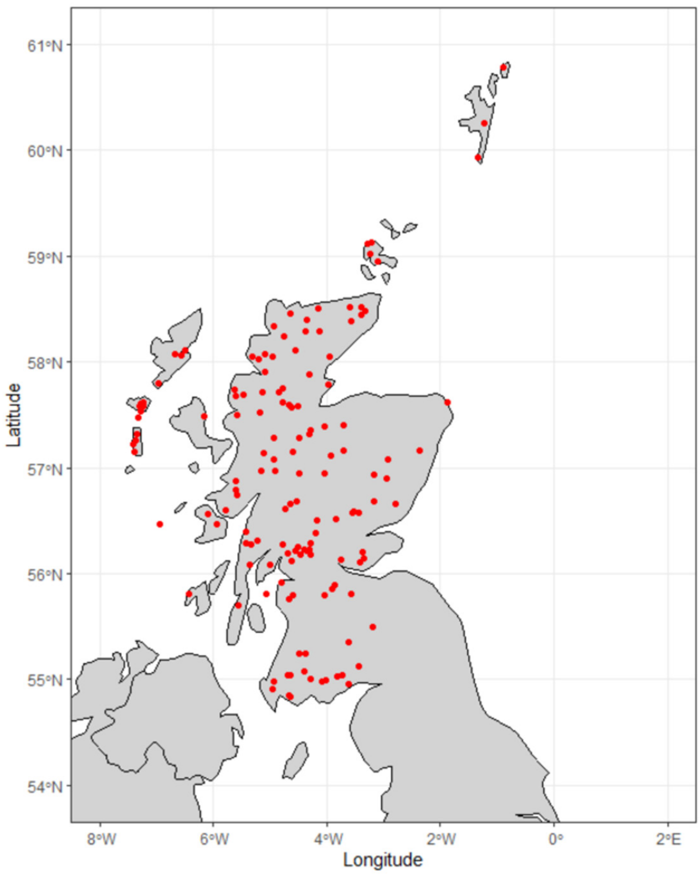

Figure 15 Spatial distribution of SEPA monitoring sites contributing data to the current modelling exercise.

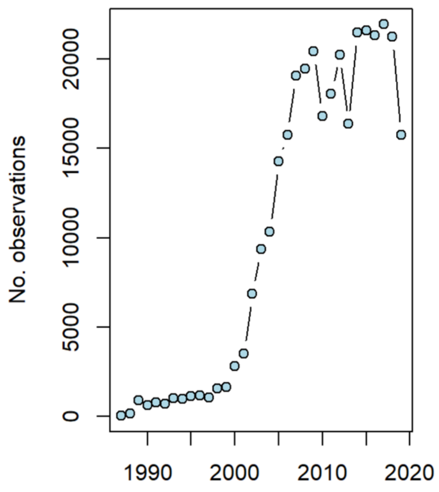

Figure 16 Temporal frequency distribution of SEPA monitoring data, by year (left) and by month of year (right).

Phytoplankton growth was quantified using concentration of chlorophyll-*a* (µg L^{-1}), which is a widely used proxy for phytoplankton biomass. Prior to modelling, the data were filtered to remove variables that were relatively poorly recorded. This was defined as <10% of site-sample combinations having available data. Specifically, this step removed the data for colour (Hazen), concentrations of dissolved organic carbon (DOC<1.2µm, mg L^{-1}), dissolved reactive phosphorus (DRP as P <0.45µm, mg L^{-1}), dissolved total phosphorus (DTP as P <0.45µm, µg L^{-1}), and Secchi depth (m). Furthermore, we removed variables that were unlikely to be causal drivers of phytoplankton growth, based upon ecological understanding (oxygen saturation and concentration, pH). Following this processing step, we had retained data on the following potential predictors of phytoplankton growth:

- Total phosphorus as P (mg L^{-1})
- Reactive phosphorus as P (mg L^{-1})
- Particulate phosphorus as P (mg L^{-1}), estimated as total phosphorus minus reactive phosphorus
- Total Oxidised Nitrogen (TON) as N (mg L^{-1}), calculated as NO_3-N + NO_2-N where TON data not available
- Silicate as SiO_2 (mg L^{-1})
- Flushing rate (the reciprocal of retention time)
- Water temperature (°C)
- Latitude (decimal degrees)
- Longitude (decimal degrees)
- Loch altitude (m.a.s.l.), surface area (ha), mean depth (m), and volume (m^3)
- Catchment area (ha)
- Loch type attributes– altitude, size, depth, geology, humic type

Initial model runs at the original, monthly, data resolution were computationally intensive whereas the catchment modelling produced time-averaged scenario data for key drivers. For consistency, the standing waters data were time-averaged prior to modelling. Nutrient drivers were averaged (mean) across late winter/early spring (January – March) for each loch and year, to estimate nutrient resource availability prior to significant biological uptake. Phytoplankton response (chlorophyll-*a* concentration) was averaged over spring/early autumn (April – September) to capture the main period of growth, as were physical variables (water temperature, flushing rate).

Random Forest and General Additive Model analyses

Two approaches were used to construct phytoplankton response models based upon these data. The first adopted a Machine Learning (ML) approach called Random Forest (RF) analysis to build a predictive model for phytoplankton growth. RF modelling considered all possible predictors of change in chlorophyll-*a* concentrations and worked by iteratively sampling subsets of the available data and predictor variables to produce a "forest" of models, each fitted to a subset of the data, using a subset of the available predictors of chlorophyll-*a*. These models were then aggregated to determine which predictors of chlorophyll-*a* were most important. The conditional importance of each predictor of chlorophyll-*a* was assessed by randomly scrambling the values of that variable (to "break" the correlation with chlorophyll-*a*) and then calculating the difference in the predictive performance of the model. If a predictor variable was important, this procedure would greatly reduce model performance (i.e. the extent to which it statistically "explains" chlorophyll-*a* variation) and yield a high importance value. RF models were implemented using *party* (v1.3.13) in the R (v4.3.0) programming environment (Hothorn *et al*., 2006a,b; Strobl *et al*., 2007, 2008, 2009; Zeileis *et al*., 2008).

RF approaches are potentially powerful in the sense that they can build predictive models that capture complex associations between ecological responses and the factors that influence them. However, these models are not readily visualised. Therefore, in a second modelling step, simpler empirical models were fitted to the data to allow visualisation of the relationships between phytoplankton biomass and key drivers. For this stage of the analysis General Additive Models (GAMs) were used, and implemented in *mgcv* (v1.8.42, Wood 2003, 2004, 2017). GAMs allow the exploration of smooth, non-linear associations between ecological responses and possible drivers of change. GAMs are more affected by skew in observational data than RF, so (apart from temperature, latitude and longitude) all variables were log transformed prior to analysis. In the GAMs, latitude and longitude were fitted as a 2-dimensional spatial smooth.

Two GAMs were fitted at this stage of the analysis. To explore a wide range of potential drivers of phytoplankton growth, we fitted a GAM that included all predictor variables that were assigned a non-zero importance in the RF step. For subsequent scenario modelling, we fitted a simplified GAM that contained only the driving variables that were available in the climate/land management scenarios and compared its performance to the "full" model in the previous step. All models were built/trained on a randomly selected 80% of the seasonally averaged data. They were then tested by using them to predict chlorophyll-*a* concentrations based upon the remaining 20% of the original dataset.

The fitted RFs and GAMs were then applied to the scenario datasets to generate tentative predictions of future chlorophyll-*a* concentrations. The results of these scenario runs were plotted as probability distributions showing the relative likelihood of different chlorophyll-*a* concentrations under each scenario. As a means of assessing their sensitivity to climate and land management change, we compared these distributions among scenarios to assess the relative distributions of lochs along the chlorophyll-*a* gradient for each scenario.

Results

Random Forest analysis

The RF analysis (Figure 17) suggested that total and particulate phosphorus (TP_P, part_P), oxidised nitrogen concentrations (TON), latitude (WBLAT), temperature (Temp), longitude (WBLONG), and UK geological type (UK_GEOL_TYPE) were the most important predictors of chlorophyll-*a* concentrations in Scottish lochs. Lake size type (UK_SIZE_TYPE), reactive phosphorus (RP_P) concentrations and silica concentrations (SiO_2) were assigned zero-to-negative importance as predictors.

Comparison of predicted and observed chlorophyll-*a* concentrations (Figure 18) suggested that the RF could make useful predictions, albeit accepting that there remained unexplained variation in phytoplankton biomass (correlation between predicted and observed values for test data $r = 0.64$, $R^2 = 0.41$; Random Forest RMSE = 7.5).

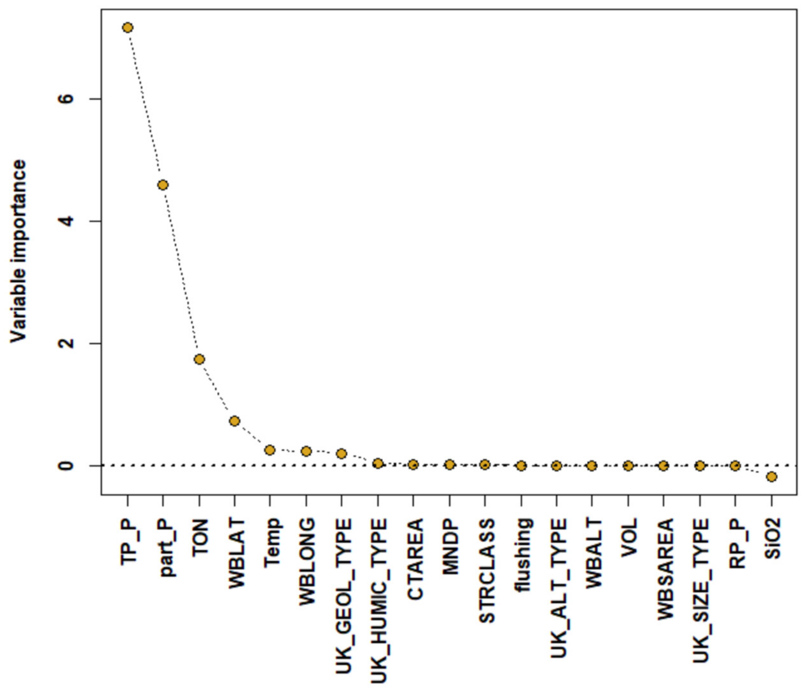

Figure 17 Measures of variable importance for the predictors of chlorophyll-*a* concentration included in the Random Forest analysis.

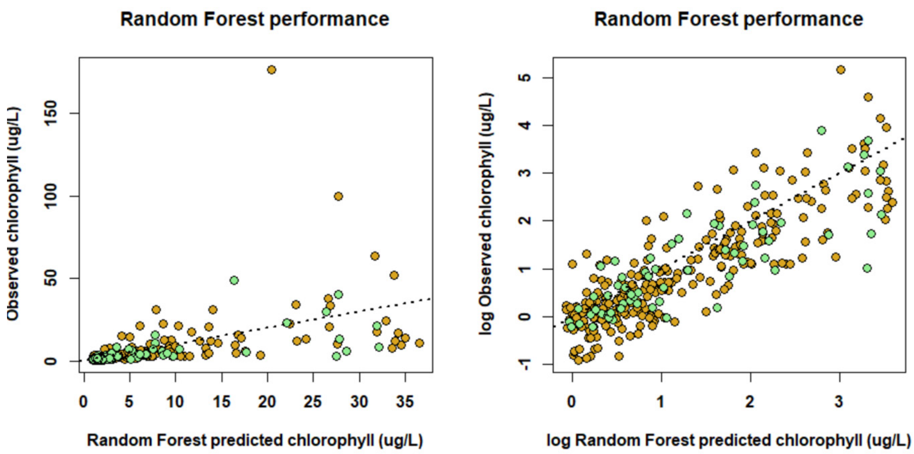

Figure 18 Random Forest predicted cf. observed chlorophyll-a concentrations on the original measurement scale (left) and logged (right). 80% training dataset plotted in gold, 20% testing dataset plotted in green.

General Additive Modelling

A General Additive Model including all predictors assigned a non-zero and positive importance score by the Random Forest, explained much of the variability in log chlorophyll-*a* concentration in the 80% training dataset (adjusted R^2 = 0.78, Deviance explained = 79.6%). This model contained statistically significant effects of geological typology, latitude and longitude, temperature, and logged values of total oxidised nitrogen, total phosphorus, altitude, and flushing rate. There was some evidence of weaker effects of humic type and catchment area (Table 14).

Visualisation of the statistically significant terms within this model (Figure 19) showed that log chlorophyll-*a* concentrations increased with higher (log) concentrations of total oxidised nitrogen and total phosphorus, and at intermediate-to-higher temperatures and altitudes. There was a tendency for log chlorophyll-*a* concentrations to be higher in medium and high alkalinity lochs, than in low alkalinity and marl waterbodies, and in clear waters. Concentrations also tended to be higher in small catchments, and at intermediate flushing rates. Finally, in addition to these effects, there remained a spatial gradient in chlorophyll concentrations. The fitted GAM was able to predict much of the variability in log chlorophyll-*a* concentration (correlation between model predicted and observed log chlorophyll-*a* in the test dataset r = 0.91, R^2= 0.83, RMSE = 0.49 (Figure 20).

Table 14 Summary results from the General Additive Model of log chlorophyll-*a* concentration, including predictors highlighted through prior Random Forest analysis. Model fitted to 80% training dataset. Statistically significant predictors are highlighted in bold. The prefix *s* denotes a one-dimensional smooth function of the predictor in question, while *te* denotes a two-dimensional tensor product smooth term.

Predictor	edf	*F* statistic	*P* value
UK geological type	n/a	3.6	0.01
Stratification class	n/a	0.2	0.82
UK humic type	n/a	2.4	0.07
te(latitude, longitude)	9.7	2.3	$<2.0e^{-16}$
s(log total oxidised nitrogen)	$7.2e^{-01}$	1.4	0.04
s(log TP)	$9.9e^{-01}$	37.6	$<2.0e^{-16}$
s(log mean depth)	$1.9e^{-04}$	0.0	0.31
s(log catchment area)	$7.1e^{-01}$	1.2	0.06
s(log altitude)	$7.7e^{-01}$	1.6	0.03
s(log surface area)	$5.3e^{-05}$	0.0	0.37
s(temperature)	1.6	4.4	$3.6e^{-03}$
s(log flushing)	$7.8e^{-01}$	1.8	0.03

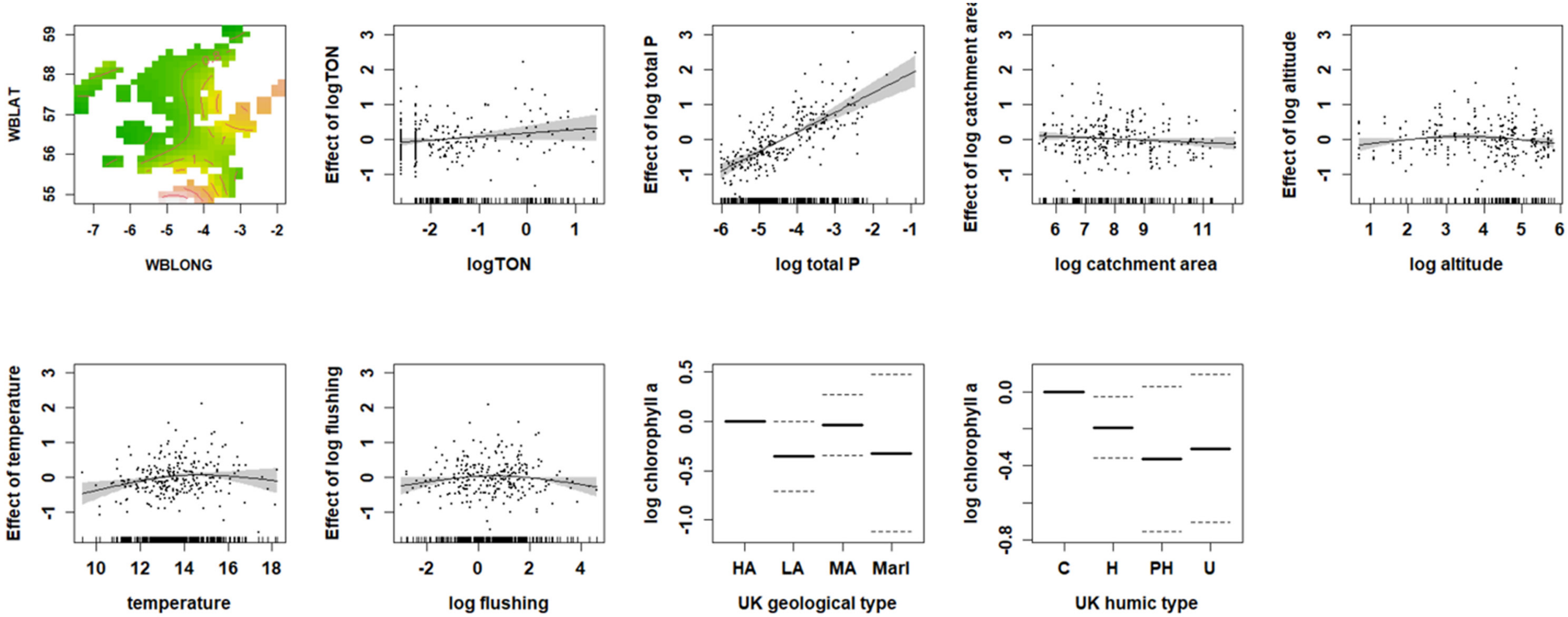

Figure 19 Statistically significant terms from the General Additive Model of log chlorophyll-*a* concentration, based upon the 80% model training dataset.

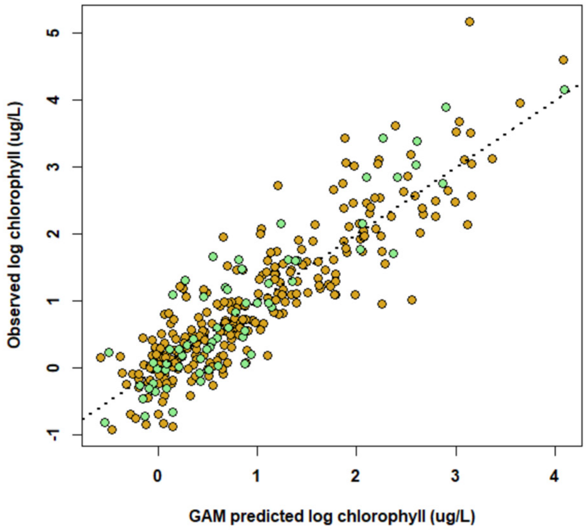

Figure 20 General Additive Model performance expressed as predicted vs observed chlorophyll-*a* concentrations on a log scale. 80% training dataset plotted in gold, 20% testing dataset plotted in green.

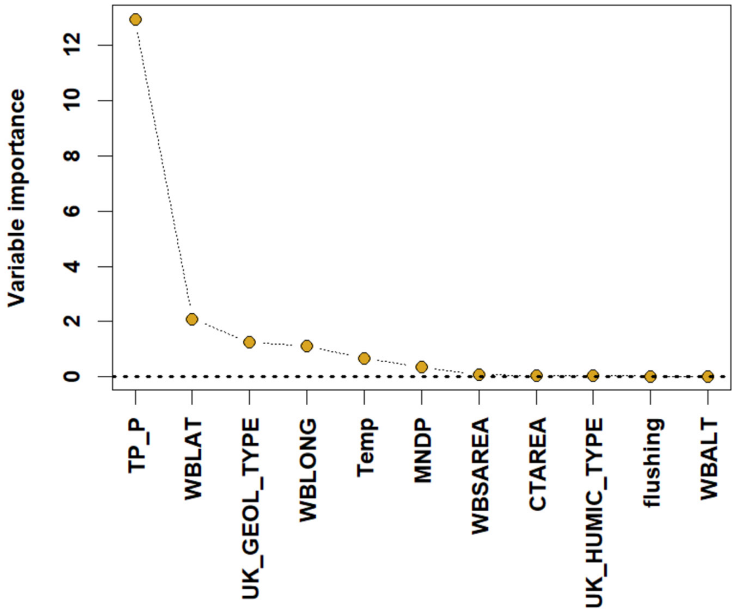

Figure 21 Variable importance measures for the predictors included in the simplified random forest model.

Adapting the models for scenario testing

Fewer predictor variables were available for scenario analysis and so the above RFs and GAMs were simplified by including only the variables that were available, namely:

- Total phosphorus as P (mg L^{-1})
- Flushing rate (the reciprocal of retention time)
- Water temperature (°C)
- atitude (decimal degrees)
- Longitude (decimal degrees)
- Loch altitude (m.a.s.l.), surface area (ha), and mean depth (m)
- Catchment area (ha)
- Loch attributes – geology, humic typologies

RF analysis of the reduced dataset (Figure 21) assigned the highest relative importance scores to TP concentration (TP_P), latitude (WBLAT), geological type (UK_GEOL_TYPE), longitude (WBLONG), temperature (Temp), and mean depth (MNDP). Loch surface area (WBSAREA), catchment area (CTAREA), humic type (UK_HUMIC_TYPE), flushing, and altitude (WBALT) were assigned lower importance as predictors of chlorophyll-*a* concentrations.

Comparison of predicted and observed chlorophyll-*a* concentrations for the simpler model

(Figure 22) suggested that, while the RF model could be used to make predictions, there would be great uncertainty due to the extensive unexplained variation in phytoplankton biomass (correlation between predicted and observed values for test data $r = 0.58$, $R^2 = 0.34$; Random Forest RMSE = 7.7).

A General Additive Model including only predictors available in the scenario data sets explained much of the variability in log chlorophyll-*a* concentration in the 80% training dataset (adjusted $R^2 = 0.78$, Deviance explained = 79.4%). This model contained statistically significant effects of geological and humic typologies, latitude and longitude, log TP concentration, temperature, and logged values altitude and flushing rate (Table 15).

The results of this simplified model were consistent with those from the more complex model. Specifically, chlorophyll-*a* concentrations were predicted to be higher with higher TP concentrations, and at intermediate-to-higher temperatures and altitudes (Figure 23). Concentrations were also predicted to be higher in high and moderate alkalinity and clear waterbodies. As in the more complex model, phytoplankton biomass also showed additional spatial variation across Scotland.

The fitted GAM, including only the variables available in the scenario datasets, was able to predict much of the variability in log chlorophyll-*a* concentration, and performed very similarly to the more complex model (correlation between model predicted and observed log chlorophyll-*a* in the test dataset $r = 0.91$, $R^2 = 0.82$, RMSE = 0.51, Figure 24).

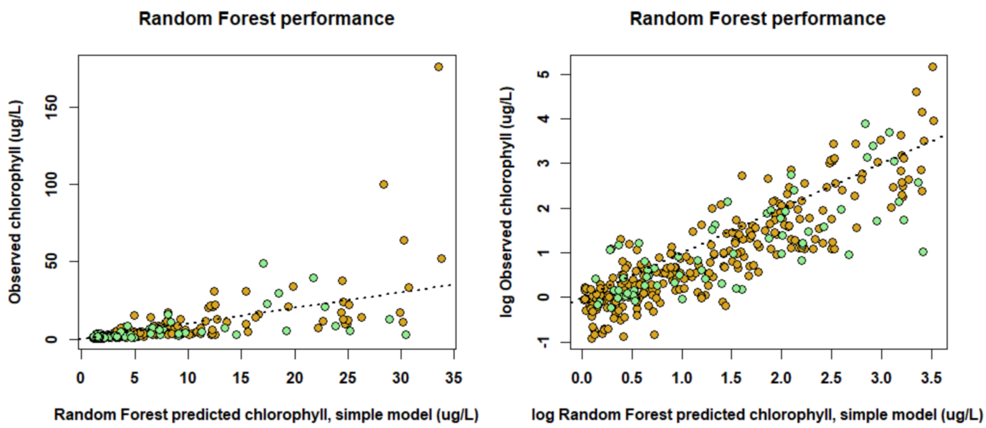

Figure 22 Random Forest predicted cf. observed chlorophyll-*a* concentrations on the original measurement scale (left) and logged (right). 80% training dataset plotted in gold, 20% testing dataset plotted in green. Results are from simplified RF model, with fewer predictors.

Table 15 Summary results from the General Additive Model of log chlorophyll-*a* concentration, showing predictors available in scenario data only. Model fitted to 80% training dataset. Statistically significant predictors are highlighted in bold. The prefix *s* denotes a one-dimensional smooth function of the predictor in question, while *te* denotes a two-dimensional tensor product smooth term.

Predictor	edf	*F*statistic	*P*value
UK geological type	n/a	5.5	$1.1e^{-03}$
UK humic type	n/a	4.4	$5.0e^{-03}$
te(latitude, longitude)	10.3	3.2	$<2.0e^{-16}$
s(total P)	$9.9e^{-01}$	49.1	$<2.0e^{-16}$
s(log mean depth)	0.4	0.4	0.14
s(log catchment area)	0.4	0.3	0.17
s(log altitude)	$8.3e^{-01}$	2.5	0.01
s(log surface area)	0.2	0.1	0.23
s(temperature)	1.5	4.1	$4.2e^{-03}$
s(log flushing)	$7.3e^{-01}$	1.4	0.05

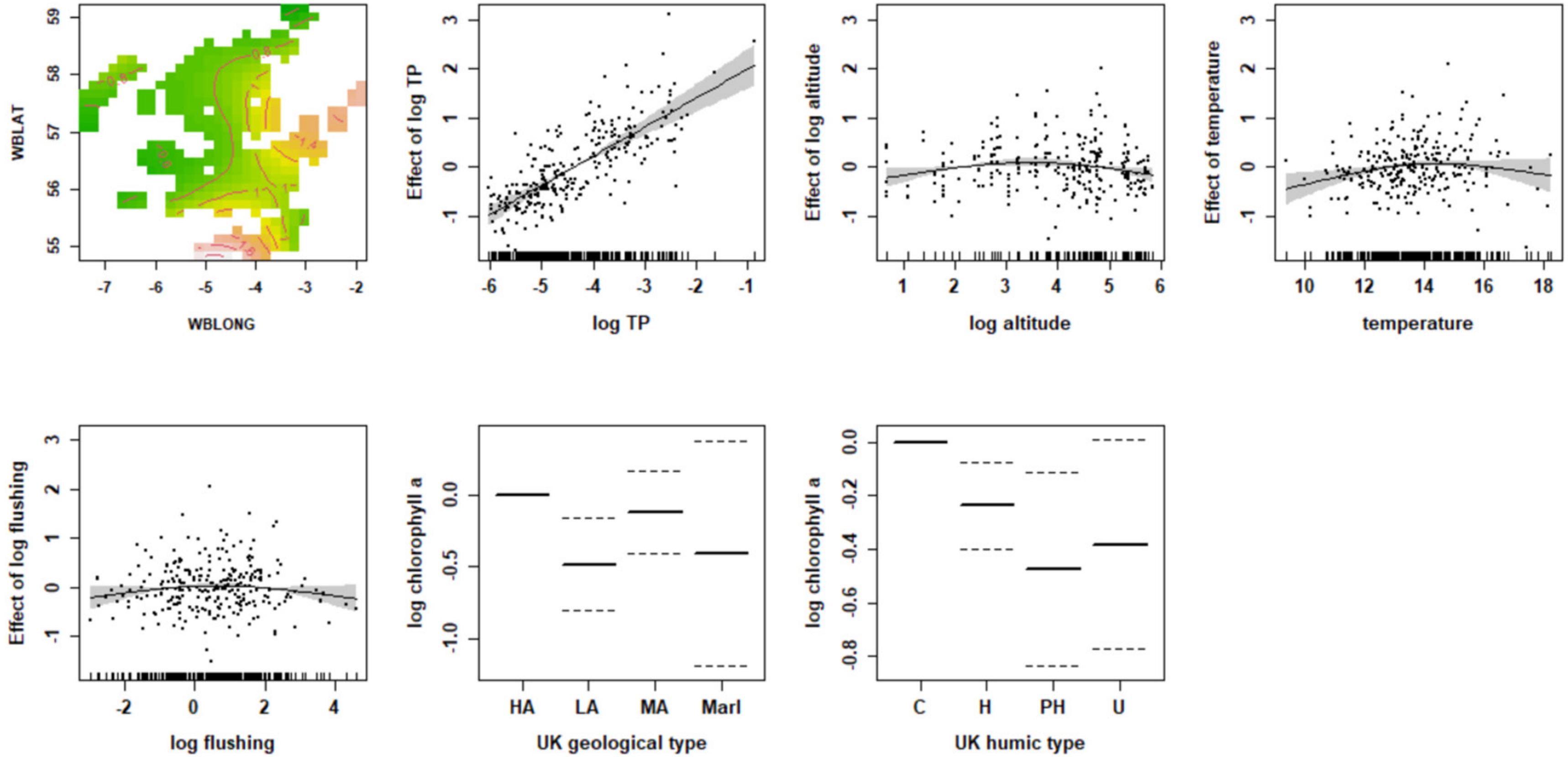

Figure 23 Statistically significant terms from the simplified General Additive Model of log chlorophyll-a concentration, based upon the 80% model training dataset.

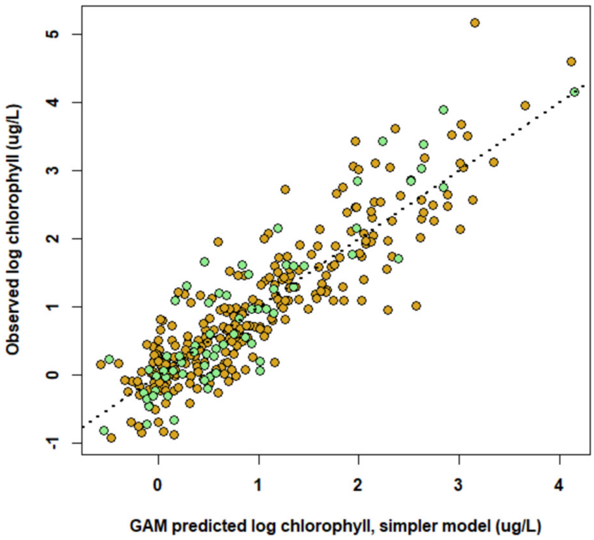

Figure 24 General Additive Model predicted vs observed chlorophyll-*a* concentrations on a log scale. 80% training dataset plotted in gold, 20% testing dataset plotted in green. Model fitted using only predictors that are available in the scenario datasets.

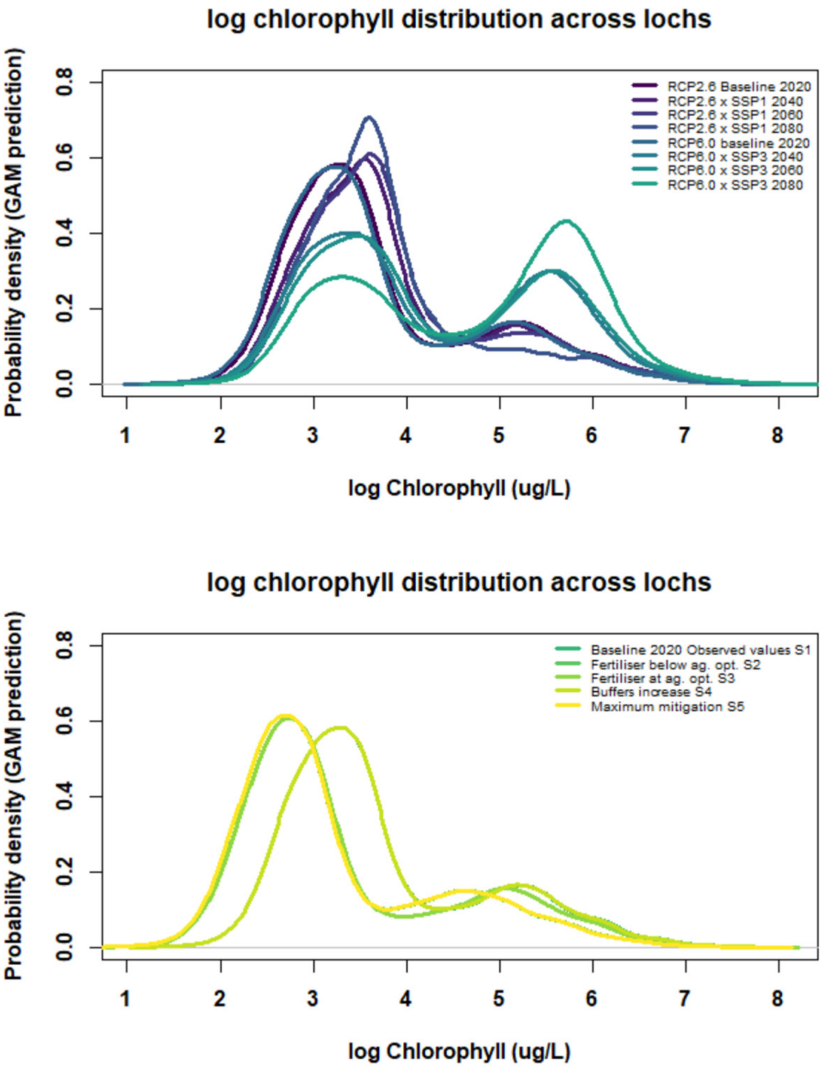

Figure 25 Predicted statistical distributions of log chlorophyll-*a* concentrations across lochs derived by applying the simplified general additive model to the scenario datasets.

Scenario analysis

Chlorophyll-*a* responses to different scenarios were based upon applying the simplified general additive model described above. Summarising the results of these model runs as probability density plots indicates the relative probability values for lochs with different log chlorophyll-*a* concentrations under each scenario (Figure 25).

For most scenarios, the model predicts a skewed distribution of log chlorophyll-*a* concentrations, with most lochs having concentrations at the lower end of the scale. This pattern was typical of the observed SEPA monitoring data used to build the model. Notably, the most extreme scenario (RCP6.0 x SSP3) yielded a sub-population of lochs with higher predicted log chlorophyll-*a* concentrations (Figure 25, upper panel), while most land management scenarios, especially the fertiliser application rate at below agronomic optimum scenario, led to lower predicted chlorophyll-*a* concentrations in many lochs (Figure 25, lower panel). Here, the scenario predictions are used to indicate possible broad scale patterns of change within the multi-loch chlorophyll-*a* concentrations, since the statistical model on which they are based represents an overall multi-loch average response and is not "tuned" to any specific water body.

Conclusions

Further consideration should be given to the multimodal distribution inherent in the chlorophyll-*a* concentration data, by creating subsets of lochs with lower and higher concentrations, and by using modelling methods suitable for multimodal data (e.g. Gaussian mixture models) to improve predictive power. Additionally, incorporating temporal algal dynamics (trends, interannual variation, seasonality) through time series analysis and more finely resolved land use driving data could help determine how the risk of algal blooms changes over time, and when the highest risk periods occur.

By adopting two different statistical modelling approaches, it was shown that a large proportion of the variability in phytoplankton biomass among Scottish standing waters (using the proxy of chlorophyll-*a* concentration) is associated with nutrient concentrations (phosphorus and nitrogen fractions), temperature, flushing, waterbody typology (alkalinity and humic categorisation), and catchment or geographical features (catchment area, altitude, latitude and longitude). By applying a general additive model, built on these observed data, to thirteen land use and climate change scenarios, we show that the relative numbers of lochs with lower and higher phytoplankton biomass are likely to change in the future, in a scenario-dependent manner. Specifically, the "worst case" scenario (RCP6.0 x SSP3) could result in a sub-population of lochs with higher concentrations of chlorophyll-*a*, whereas fertiliser application rates below agronomic optimum could help to reduce chlorophyll-*a* concentrations in many of the lochs with higher chlorophyll-*a* values.

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Appendix 6 Climate change impacts on standing waters

Background

There is a policy focus at national and international levels on mitigating climate change by reducing carbon emissions and increasing carbon sequestration. However, even if we can slow climate change down, it is unlikely that it can be prevented or reversed in this way. So, alternative approaches must be used to lessen its effects. These include adaptive interventions that increase the resilience, and reduce the vulnerability, of people and nature to weather extremes and other climate change impacts (Scottish Government, 2018).

In their report on the most up-to-date evidence of climate change trends observed in the UK, the UK Climate Change Committee (2021) indicated that the most likely changes to the UK climate in the future would be warmer and wetter winters, and hotter and drier summers. These changes are likely to have adverse effects on the quality of Scotland's standing waters. In a recent report by May *et al*. (2022a,b), these were summarised as:

- Increased risk of phytoplankton (algal) blooms, driven by increases in air temperatures and changes in rainfall patterns.
- An associated increase in the risk of potentially harmful toxins being released into the water by cyanobacteria – often known as harmful algal blooms (HABs).

In this study we have used the catchment and lake models and future change scenarios developed in Appendices 2 to 5 to explore the extent to which changes in the way changes in catchment management could help to mitigate these effects.

Aims and objectives

The data and models from work described in Appendices 2 to 5 have been used, here, to predict the impacts of different climate change predictions and different Shared Socioeconomic Pathways (SSP) scenarios on the water quality of standing waters across Scotland. Impacts are expressed changes in TP concentrations, likelihood of cyanobacteria levels exceeding WHO alert levels for unsafe use and in WFD water quality status. Of the SSPs considered, SSP1 represents a sustainable and co-operative society with a low carbon economy and high capacity to adapt to climate change. In contrast, SSP3 represents a future where food production dominates land use and social and economic barriers lead to a highly fragmented society with limited capacity to adapt to climate change.

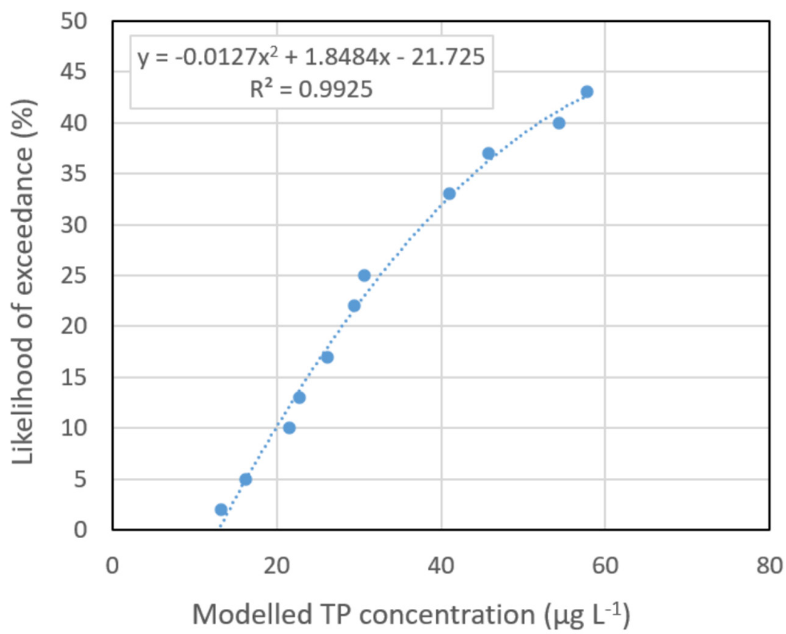

Figure 26 Likelihood of cyanobacteria levels exceeding WHO guidelines for safe us based on P concentrations (after Carvalho *et al*., 2013).

Methods

Total phosphorus (TP) concentrations under the different climate change and land use scenarios were calculated for 6836 standing waters across Scotland using the Equation 3 (see Appendix 4). In addition, the likelihood of these waterbodies exceeding WHO alert levels indicating that cyanobacterial levels are too high for safe use were calculated for the equation shown in Figure 26, which was derived from data published by Carvalho *et al*. (2013).

Of the change scenarios considered in this study, RCP2.6 x SSP1 represented the best case scenario for the future whereas RCP6.0 x SSP3 represented the worst case scenario.

Results

Figure 27 shows the annual average TP concentrations in Scottish standing waters derived from the SEPA WFD monitoring data for 2015-18. These levels have been taken to represent the baseline conditions for 2020. The data show high levels of TP in waterbodies across the central belt and in lowland areas around the east coast of the mainland, the Shetland Isles and some of the larger islands off the west coast. Figure 28 and Figure 29 shows projected changes in these values across Scotland under two different climate projections and two different socio-economic pathways. Visual comparison of the maps in Figure 28 show an improvement in water quality across the central belt between 2020 and 2080 under RCP2.6 and SSP1, but a noticeable degradation in water quality across the north of mainland Scotland. In contrast, the maps in Figure 29 show a marked increase in TP concentrations in standing waters in the south and west of the country between 2020 and 2080 under RCP6.0 and socio-economic scenario SSP3.

Notably, only 95 of the 6,836 standing waters included in the study were affected by discharges from waste water treatment works (Figure 27). The majority of standing waters affected by high TP concentrations are not within these areas. This indicated that the high TP inputs that were affecting the in-loch TP concentrations were coming mostly from diffuse sources, such as agricultural runoff. In addition, inputs from other sewage sources such as septic tanks are shown in Table 16. On average, diffuse pollution, associated with soil erosion, leaching to drains and runoff from farmyards, accounted for 66% to 88% of the TP input to the standing waters included in this study.

Table 16 TP losses (kg y^{-1}) by source and total TP losses based on land cover for 2020 and land cover based on the RCP2.6 x SSP1 and RCP6.0 x SPP3 scenarios for years 2040, 2060 and 2080. Values in brackets show percentage contribution of TP loss from each source to total TP losses for each scenario.

Source of TP	Baseline 2020		CRAFTY-GB 2040		CRAFTY-GB 2060		CRAFTY-GB 2080	
	RCP2.6 x LCM	RCP6.0 x LCM	RCP2.6 x SSP1	RCP6.0 x SSP3	RCP2.6 x SSP1	RCP6.0 x SSP3	RCP2.6 x SSP1	RCP6.0 x SSP3
Soil erosion	72,865 (36%)	72,788 (37%)	70,290 (43%)	184,303 (53%)	75,054 (45%)	183,638 (53%)	70,814 (44%)	256,134 (55%)
Leaching to drains	72,228 (36%)	68,028 (35%)	39,736 (24%)	109,095 (31%)	38,323 (23%)	107,030 (31%)	33,913 (22%)	153,152 (33%)
Farmyards	26 (0%)	26 (0%)	25 (0%)	26 (0%)	25 (0%)	25 (0%)	25 (0%)	25 (0%)
Septic Tanks	28,294 (14%)	27,734 (14%)	27,416 (17%)	28,238 (8%)	27,582 (16%)	27,688 (8%)	27,449 (17%)	27,690 (6%)
Sewage Treatment Works	27,139 (14%)	26,796 (14%)	27,139 (16%)	27,139 (8%)	27,139 (16%)	27,139 (8%)	27,139 (17%)	27,139 (6%)
Total	**200,552**	**195,371**	**164,606**	**348,801**	**168,124**	**345,521**	**159,340**	**464,141**

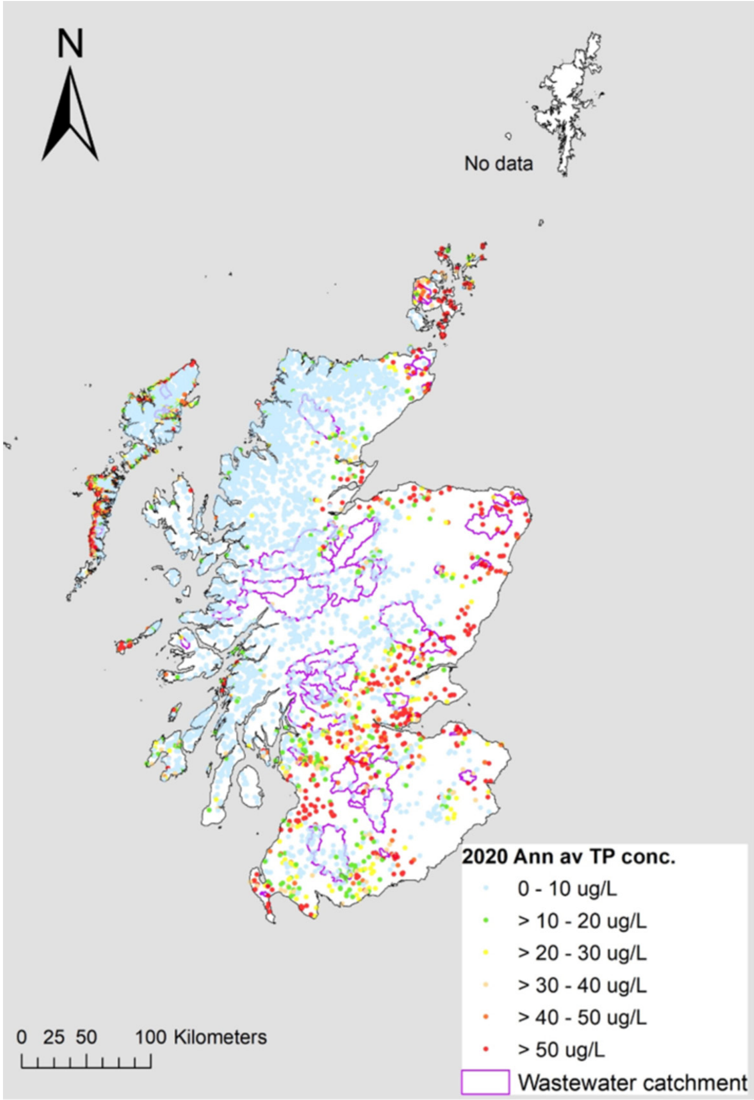

Figure 27 Annual average TP concentrations in standing waters in 2020 showing those catchments affected by discharges from wastewater treatment works.

The projected change in TP concentrations under a range of climate and land use change scenarios between 2020 and 2080 are shown in Figure 28 and Figure 29. Under the least damaging scenario tested (RCP2.6 x SSP1), site specific TP concentrations changed very little between 2020 and 2080 in most areas of Scotland. However, there was a notable reduction in TP levels in standing waters across the central belt whereas there was a notable increase in TP concentrations across the more northern parts of the Scottish mainland. In contrast, under the most damaging scenario tested (RCP6.0 x SSP3), TP concentrations remained similar to 2020 levels in the east of the country and across the central below whilst increasing markedly in the north west and south west of the country.

Under the least damaging scenario tested, RCP2.6 x SSP1, there were 5,211 (76%) standing waters across Scotland with TP concentrations of less than 10 µg L^{-1} in 2020, and this number had risen to 5699 water bodies (83%) by 2080 (Table 17; Table 18). In contrast, under the more damaging scenario tested, RCP6.0 x SSP3, there were 5,178 standing waters (76%) across Scotland with TP concentrations of less than 10 µg L^{-1} in 2020, but this number had fallen to 2813 (41%) by 2080 (Table 19, Table 20).

Overall, some improvement in water quality was projected to occur between 2020 and 2080 under the RCP2.6 x SSP1 change scenario, whereas a marked decline in water quality was projected to occur under the RCP6.0 x SSP3 change scenario.

Table 17 Projected changes in the percentage of standing waters within each total phosphorus concentration category under scenario RCP2.6 x SSP1, 2020 – 2080.				
Phosphorus concentration	2020	2040	2060	2080
0 - 10 µg P L^{-1}	76%	77%	79%	83%
>10 - 20 µg P L^{-1}	6%	4%	5%	5%
>20 - 30 µg P L^{-1}	4%	3%	3%	3%
>30 - 40 µg P L^{-1}	3%	2%	2%	2%
>40 - 50 µg P L^{-1}	3%	3%	2%	1%
>50 µg P L^{-1}	8%	10%	9%	7%

Table 18 Projected changes in the number of standing waters within each total phosphorus concentration category under scenario RCP2.6 x SSP1, 2020 – 2080; Δlochs indicates the number of standing waters that changed from one total phosphorus concentration category to another between 2020 and 2080, and the direction of change.					
Phosphorus concentration	2020	2040	2060	2080	Δlochs
0 - 10 µg P L^{-1}	5211	5274	5375	5699	488
>10 - 20 µg P L^{-1}	377	300	327	320	-57
>20 - 30 µg P L^{-1}	280	220	209	171	-109
>30 - 40 µg P L^{-1}	214	153	154	107	-107
>40 - 50 µg P L^{-1}	208	192	162	84	-124
>50 µg P L^{-1}	546	697	609	455	-91

Table 19 Projected changes in the percentage of standing waters within each total phosphorus concentration category under scenario RCP6.0 x SSP3, 2020 – 2080.				
Phosphorus concentration	2020	2040	2060	2080
0 - 10 µg P L^{-1}	76%	58%	57%	41%
>10 - 20 µg P L^{-1}	5%	6%	6%	6%
>20 - 30 µg P L^{-1}	4%	5%	5%	5%
>30 - 40 µg P L^{-1}	3%	4%	4%	5%
>40 - 50 µg P L^{-1}	3%	4%	4%	4%
>50 µg P L^{-1}	8%	25%	25%	38%

Table 20 Projected changes in the number of standing waters within each total phosphorus concentration category under scenario RCP6.0 x SSP3, 2020 – 2080; Δlochs indicates the number of standing waters that changed from one total phosphorus concentration category to another between 2020 and 2080, and the direction of change.					
Phosphorus concentration	2020	2040	2060	2080	Δlochs
0 - 10 µg P L^{-1}	5178	3936	3877	2813	-2365
>10 - 20 µg P L^{-1}	372	377	387	433	61
>20 - 30 µg P L^{-1}	281	330	342	374	93
>30 - 40 µg P L^{-1}	223	253	261	327	104
>40 - 50 µg P L^{-1}	209	254	257	301	92
>50 µg P L^{-1}	573	1686	1712	2588	2015

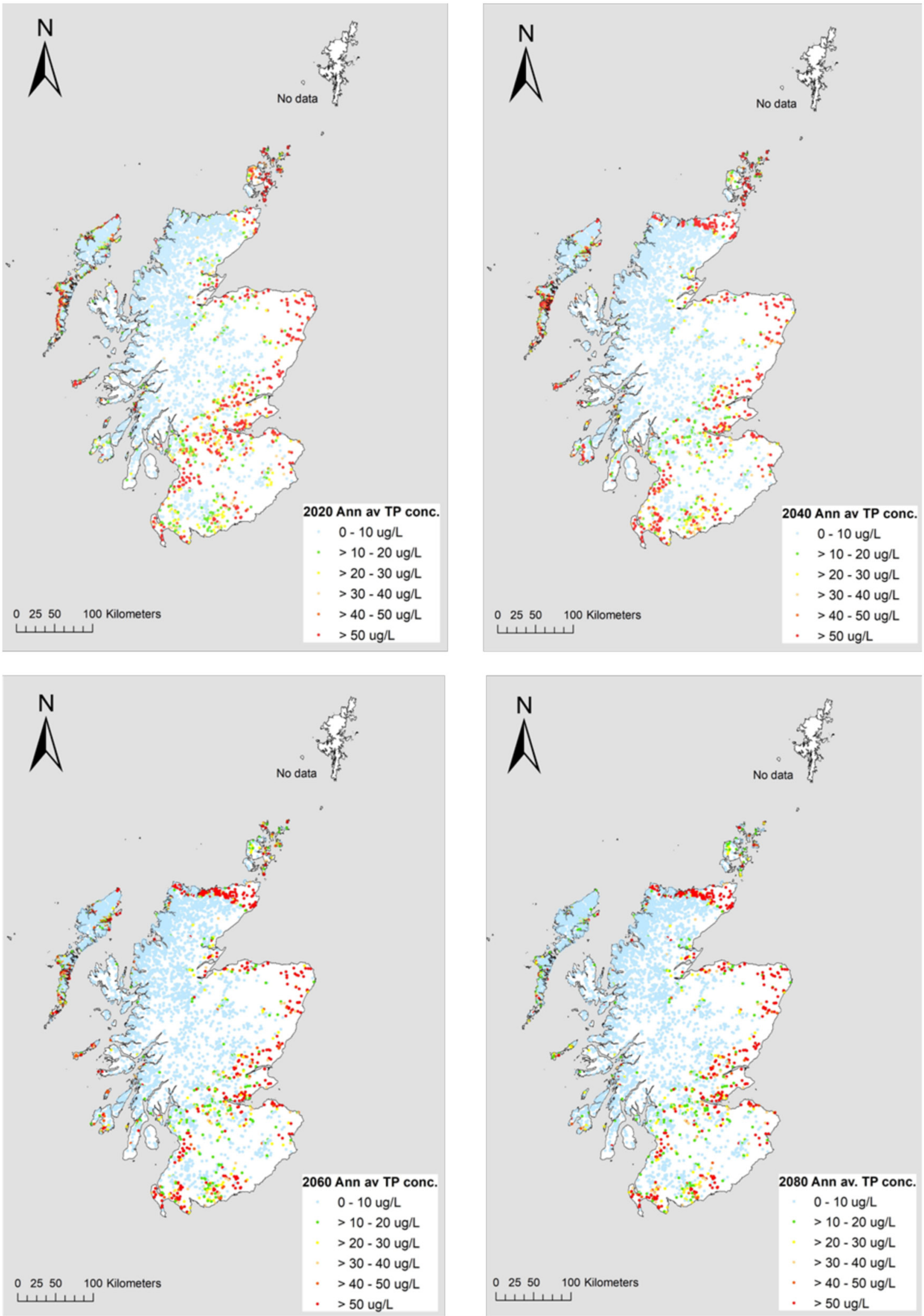

Figure 28 Changes in the TP concentration of Scottish standing waters under climate change scenario RCP2.6 and Shared Socioeconomic Pathway SSP1, 2020 to 2080.

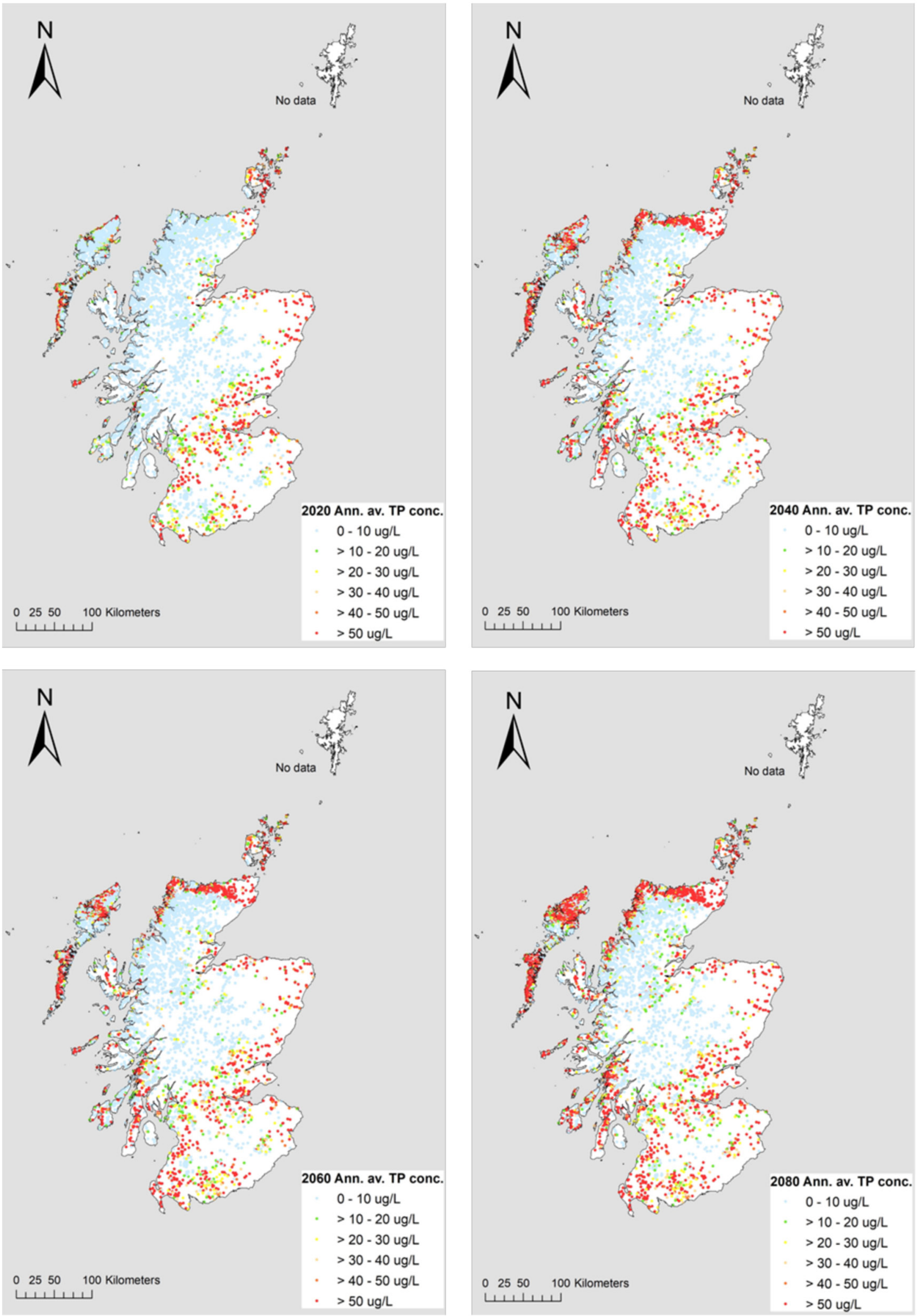

Figure 29 Changes in the TP concentration of Scottish standing waters under climate change scenario RCP6.0 and Shared Socioeconomic Pathway SSP3, 2020 to 2080.

The effects of the future climate and land use change scenarios tested on the likelihood of cyanobacterial levels exceeding WHO limits for safe use of these waterbodies (WHO 2004, 2021) is shown in Figure 31 and Figure 32. As these likelihoods are based on TP concentrations (Figure 26), the pattern of change is very similar to that of TP. Under scenario RCP2.6 x SSP1, the increase in the number and distribution of standing waters that were projected to develop troublesome algal blooms (assumed for mapping purposes to be a likelihood of greater than 40%) by 2080 was relatively low. In contrast, under scenario RCP6.0 x SSP3, the likelihood of such cyanobacterial blooms increased more in the north west and south west of the country than elesewhere.

Under the least damaging scenario tested, RCP2.6 x SSP1, 5,607 standing waters (82%) across Scotland were projected to have likelihoods of less than 10% of failing WHO water quality criteria for safe use of less in 2020 (Table 21), and this number was projected to rise to 6,132 water bodies (90%)

Table 21 Changes in the percentage of standing waters projected to fail WHO water quality criteria for safe use between 2020 and 2080 under the RCP2.6 x SSP1 change scenario.

Likelihood of exceedance	2020	2040	2060	2080
0-10%	82%	83%	85%	90%
>10-20%	3%	3%	3%	3%
>20-30%	4%	3%	3%	2%
>30-40%	5%	4%	4%	2%
>40-50%	6%	7%	5%	3%
>50%	0%	0%	0%	0%

Table 22 Changes in the number of standing waters projected to fail WHO water quality criteria for safe use between 2020 and 2080 under the RCP2.6 x SSP1 change scenario; Δlochs indicates the number of standing waters that changed from one category to another, and the direction of change.

Likelihood of exceedance	2020	2040	2060	2080	Δlochs
0-10%	5607	5670	5818	6132	525
>10-20%	235	209	208	175	-60
>20-30%	245	176	185	149	-96
>30-40%	324	289	259	162	-162
>40-50%	425	492	366	218	-207
>50%	0	0	0	0	0

Table 23 Changes in the percentage of standing waters projected to exceed WHO criteria for safe use between 2020 and 2080 under the RCP6.0 x SSP3 change scenario.

Likelihood of exceedance	2020	2040	2060	2080
0-10%	82%	67%	66%	55%
>10-20%	3%	5%	5%	6%
>20-30%	4%	5%	6%	8%
>30-40%	5%	9%	9%	12%
>40-50%	7%	14%	14%	19%
>50%	0%	0%	0%	0%

Table 24 Changes in the number of standing waters projected to exceed WHO criteria for safe use between 2020 and 2080 under the RCP6.0 x SSP3 change scenario; Δlochs indicates the number of standing waters that changed from one category to another between 2020 and 2080, and the direction of change.

Likelihood of exceedance	2020	2040	2060	2080	Δlochs
0-10%	5579	4562	4540	3757	-1822
>10-20%	236	364	355	442	206
>20-30%	254	375	401	572	318
>30-40%	322	598	599	795	473
>40-50%	445	937	941	1270	825
>50%	0	0	0	0	0

by 2080 (Table 22). In contrast, under the most damaging scenario tested, RCP6.0 x SSP3, there were 5,579 standing waters (82%) across Scotland with likelihoods of WHO water quality criteria of less than 10% in 2020 (Table 23), but this number had fallen to 3,757 (55%) by 2080 (Table 24).

Figure 30 illustrates how the cyanobacterial concentrations in Scottish lochs increase with increasing water temperature and TP concentrations. This relationship suggests that TP concentrations need to be reduced to prevent cyanobacterial blooms worsening under the rising temperatures associated with climate change.

The effects of the future climate and land use change scenarios tested on future WFD water quality status is shown in Figure 35 and Figure 36. Because these values are relative to site specific phophorus targets, which vary depending on the type of standing water, the pattern of change is different to that of TP concentrations alone. This allows for impacts on more oligotrophic waters to be determined, including level of compliance with statutory water quality requirements.

In general, under change scenario RCP2.6 x SSP1, there was a slight increase in the number of standing waters likely to fail WFD Good status for TP along the northern part of the Scottish mainland between 2020 and 2080 (Figure 35 and Figure 36). However, there was little change elsewhere. In contrast, under change scenario RCP6.0 x SSP3, the number of failures to meet WFD Good status for TP increased rapidly across all parts of Scotland over that period (Figure 35 and Figure 36).

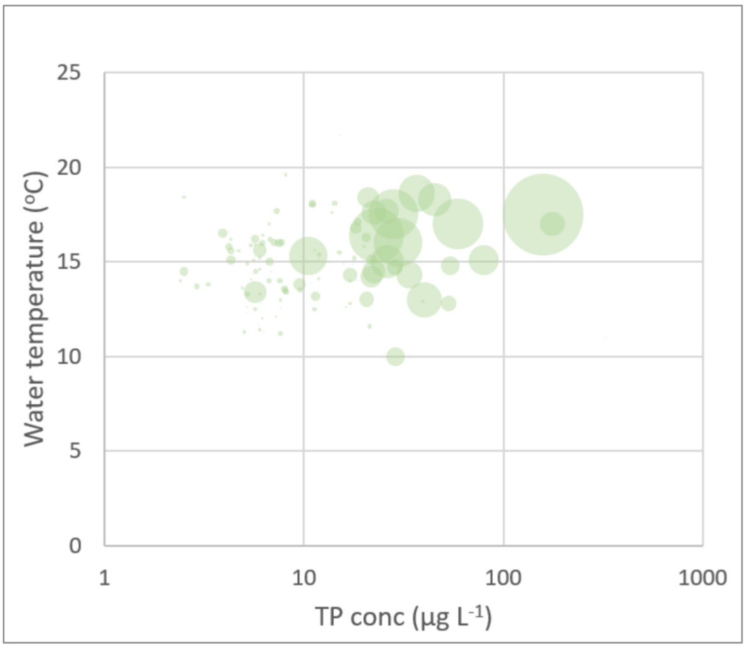

Figure 30 Relationship between water temperature, TP concentration and the amount (biovolume) of cyanobacteria (proportional to area of bubble) in Scottish standing waters, 2009 – 2012. Scale: Maximum value shown = 0.09 mm3 L^{-1}.

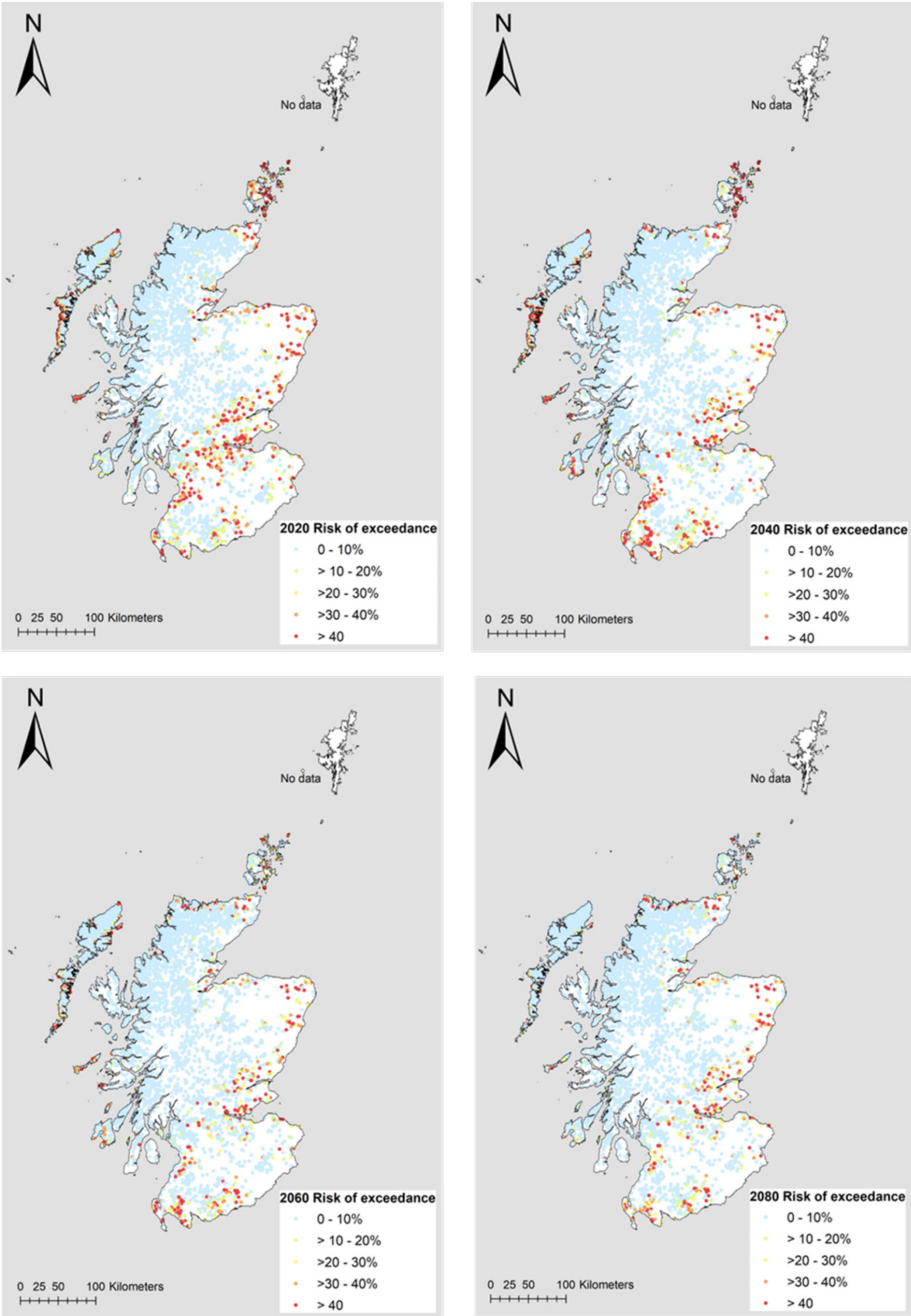

Figure 31 Changes in the level of risk of exceeding WHO thresholds for safe use of Scottish standing waters under climate change scenario RCP2.6 and Shared Socioeconomic Pathway SSP1, 2020 to 2080.

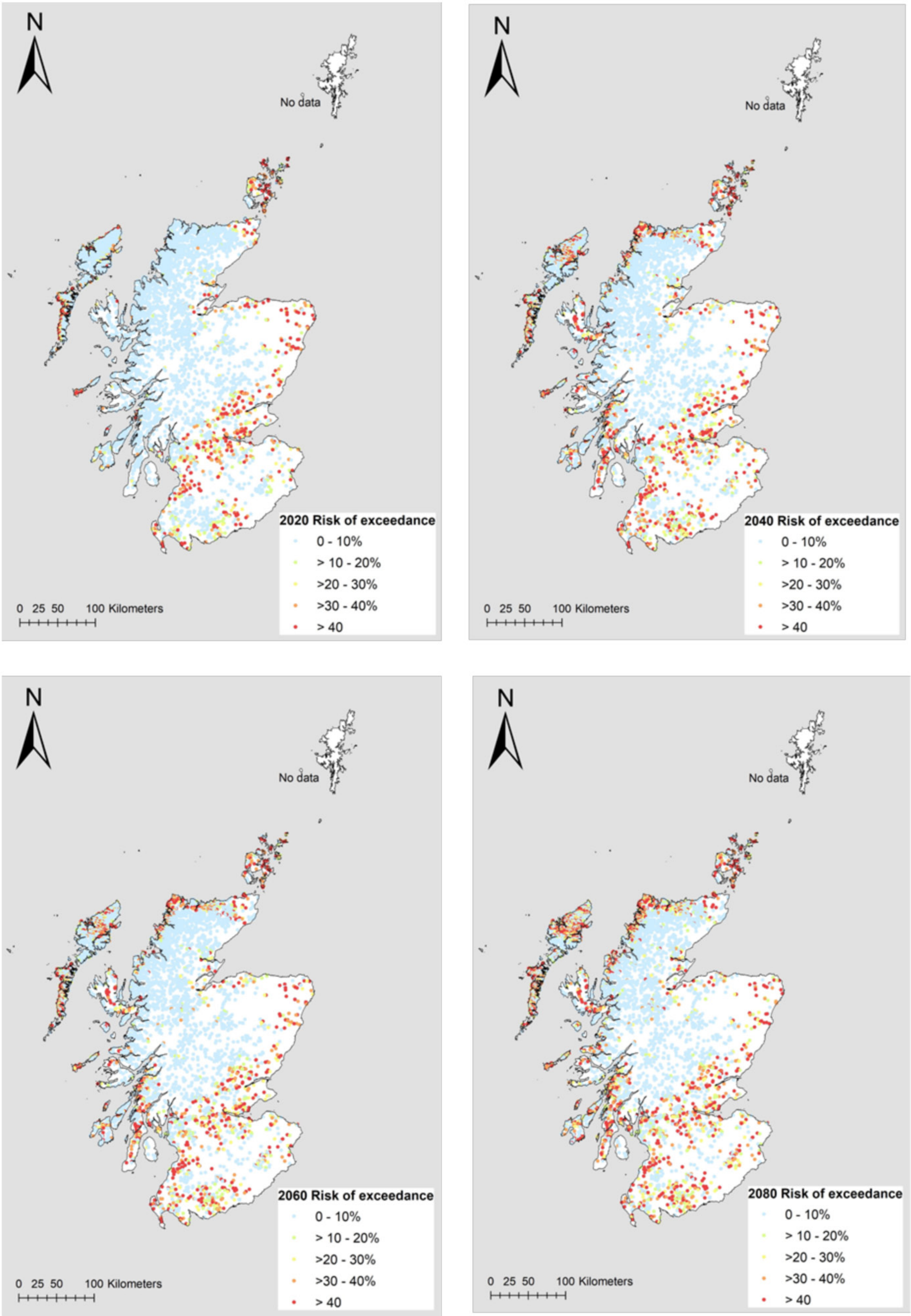

Figure 32 Changes in the level of risk of exceeding WHO thresholds for safe use of Scottish standing waters under climate change scenario RCP6.0 and Shared Socioeconomic Pathway SSP3, 2020 to 2080.

Table 25 Number and percentage of standing waters projected to achieve WFD High status or WFD Good status or higher under the two climate change and land use scenarios tested.								
Change scenario	WFD High status				WFD Good status or higher			
	2020	2040	2060	2080	2020	2040	2060	2080
RCP2.6 x SSP1	233 (74%)	252 (80%)	254 (80%)	269 (85%)	262 (83%)	271 (85%)	275 (87%)	285 (90%)
RCP6.0 x SSP3	230 (73%)	150 (47%)	142 (45%)	92 (29%)	260 (82%)	187 (59%)	184 (58%)	145 (46%)

Table 26 Land cover types within the catchments of lochs in the north and north west of Scotland, comparing percentage cover by area in 2020 and 2080, under scenario RCP6.0 x SSP3.		
Land cover type	Percentage areal coverage 2020	Percentage areal coverage 2080
Other	17.7	0.0
Wildscape	65.2	33.0
Woodland	0.1	0.4
Forestry	3.1	0.6
Rough grassland	0.1	1.5
Grassland improved	12.8	5.3
Arable extensive	0.9	23.2
Arable intensive	0.0	36.0

More specifically, under change scenario RCP2.6 x SSP1, the the number and percentage of standing waters that were projected to meet WFD High status or WFD Good status or higher for phophorus increased from 233 (74%) to 269 (85%), and from 262 (83%) to 285 (90%), respectively, between 2020 and 2080 (Table 25). In contrast, under change scenario RCP6.0 x SSP3, the the number and percentage of standing waters that were projected to meet WFD High status or WFD Good status or higher decreased from 230 (73%) to 92 (29%), and from 260 (82%) to 145 (45%), respectively, between 2020 and 2080 (Table 25).

Most the outputs from the RCP6.0 x SSP3 scenarios showed a marked decrease in water quality in lochs in the north of Scotland and in the Western Isles. The reason for this was explored by selecting catchment data for lochs where TP>40 ug L^{-1}; these covered a combined land area of 4,434 km^2.

By comparing baseline land cover from LCM 2020 with CRAFTY-GB land cover for 2020 under scenario RCP6.0 x SSP3. Between 2020 and 2080, there is a very large projected increase in arable land with at the expense of Wildscape (shrublands, acid grassland and peatlands) much of which would have to be drained to increase land available for food production (Table 26). There is also a huge projected increase in losses due to soil erosion, from 11,490 kg yr^{-1} in 2020 to 176,770 kg yr^{-1} in these areas. Together, these changes are projected to increase TO inputs to these lochs greatly. In addition, lower rainfall under RCP6.0 will reduce flushing rates making the lochs more susceptible to algal blooms. So, in summary, these high lake TP concentrations are driven, mainly, by the extreme simulated land use change imposed by CRAFTY-GB simulations for RCP6.0 x SSP3.

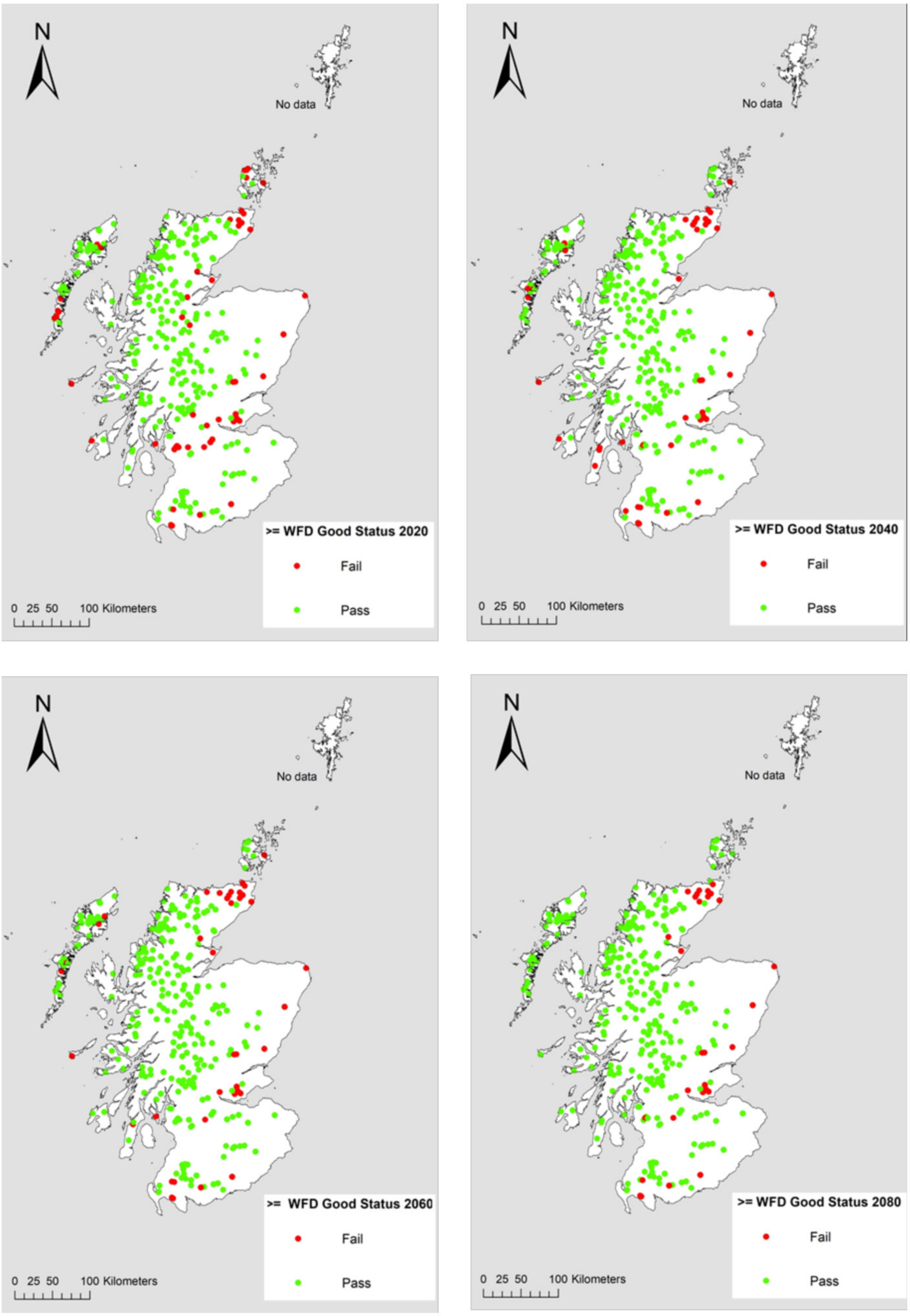

Figure 33 Changes in the number and location of Scottish standing waters passing/failing to achieve WFD Good or higher water quality status for total phosphorus concentration under climate change scenario RCP2.6 and Shared Socioeconomic Pathway SSP1, 2020 to 2080.

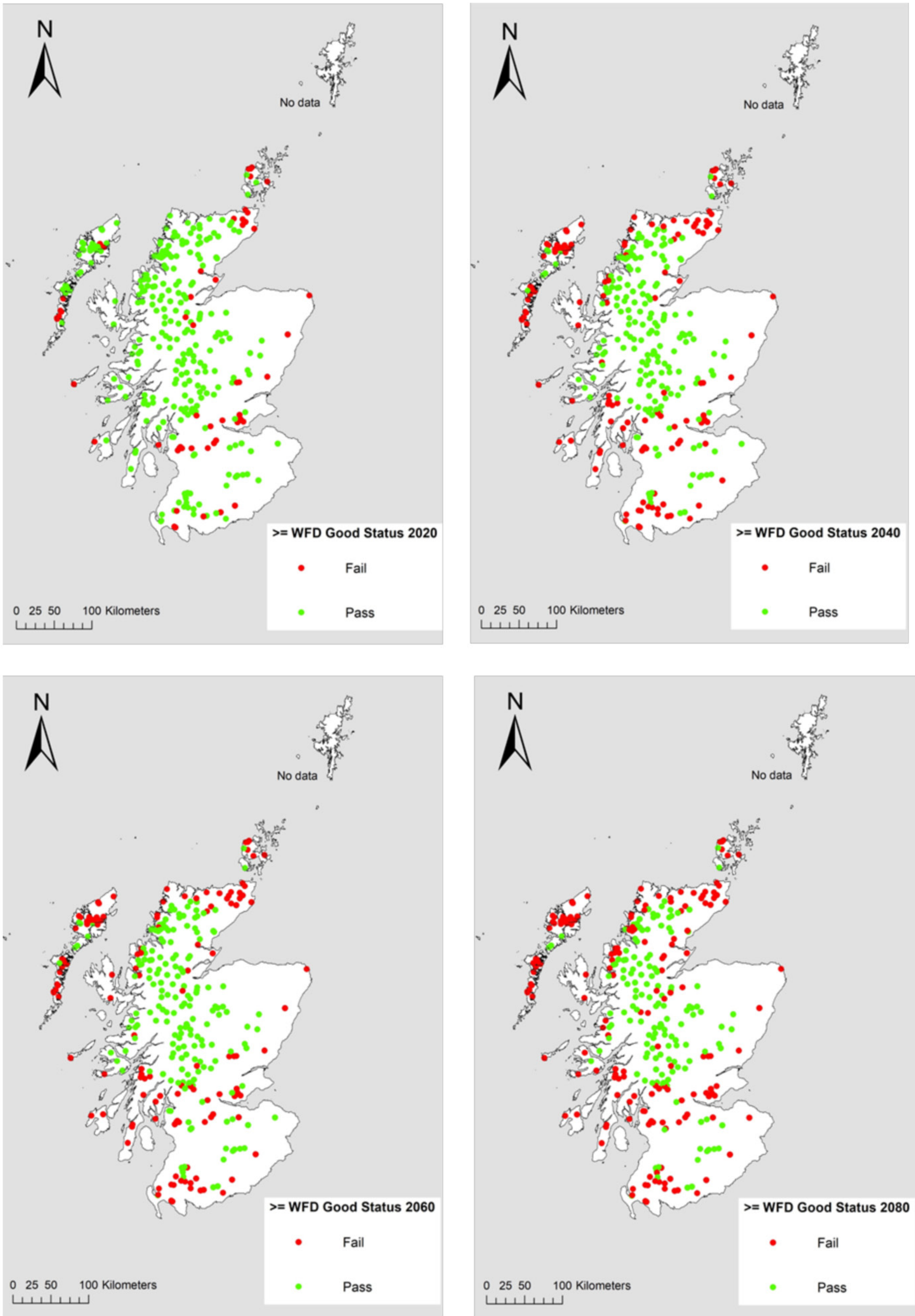

Figure 34 Changes in the number and location of Scottish standing waters passing/failing to achieve WFD Good or higher water quality status for total phosphorus concentration under climate change scenario RCP6.0 and Shared Socioeconomic Pathway SSP3, 2020 to 2080.

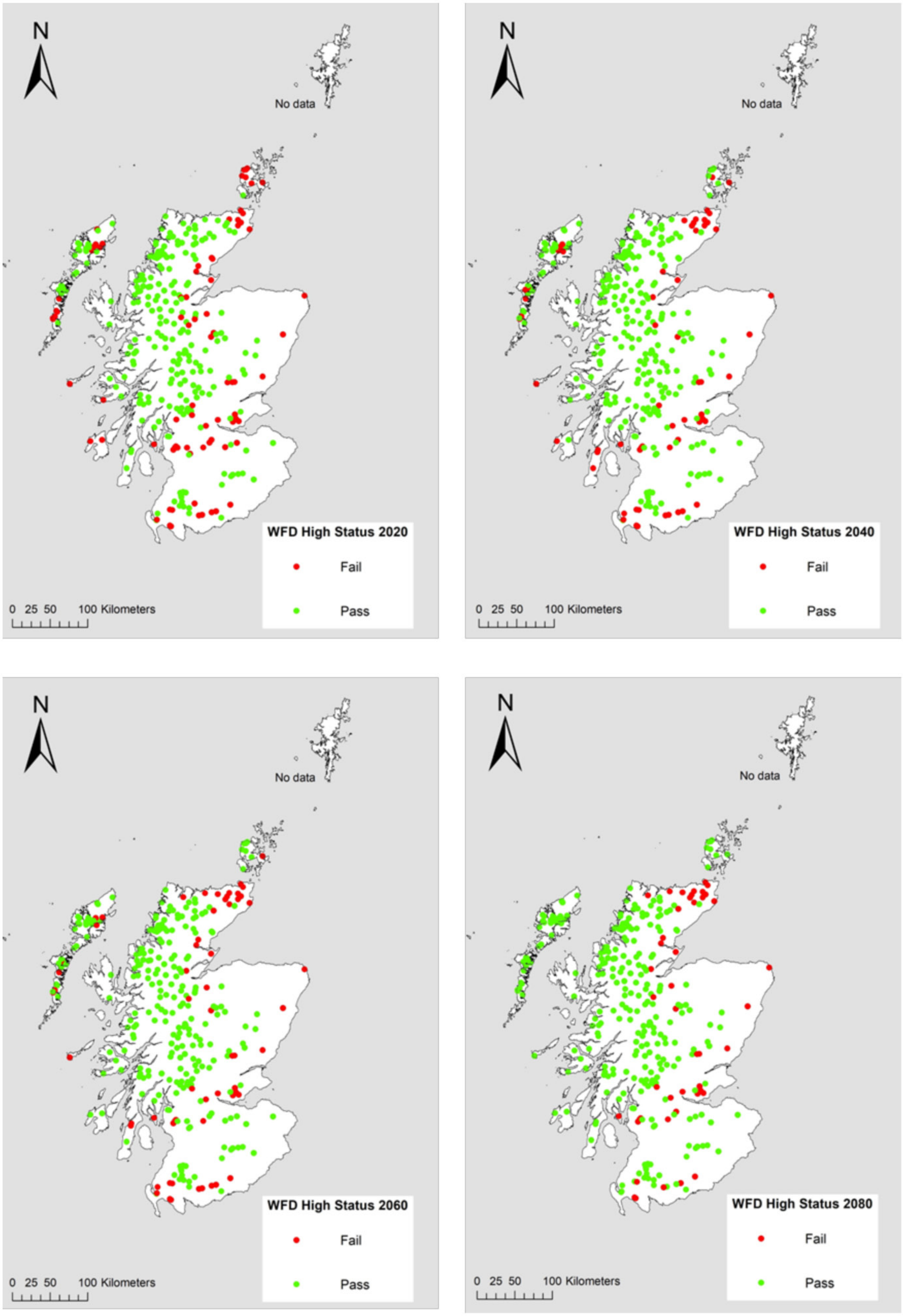

Figure 35 Changes in the number and locations of Scottish standing waters passing/failing to achieve WFD High status for total phosphorus concentration under climate change scenario RCP 2.6 and Shared Socioeconomic Pathway SSP1, 2020 to 2080.

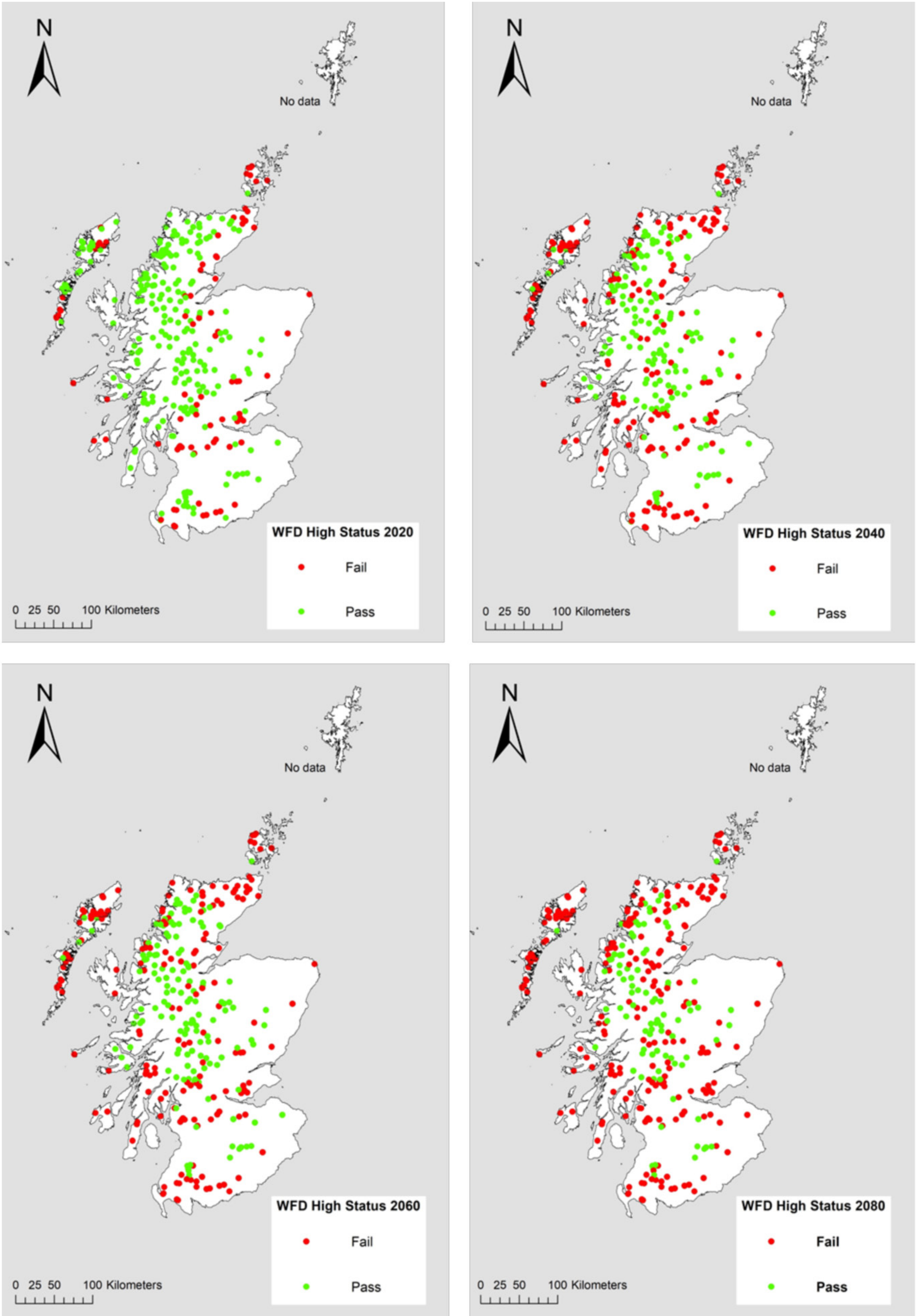

Figure 36 Changes in the number of Scottish standing waters passing/failing to achieve WFD High water quality status for total phosphorus concentration under climate change scenario RCP 6.0 and Shared Socioeconomic Pathway SSP3, 2020 to 2080.

Conclusions

Under certain conditions, phytoplankton grow very quickly in standing waters and accumulate to form what is commonly known as an 'algal bloom'. These blooms are not just algae; they also contain cyanobacteria, also known as blue green algae. Together, the quantities of algae and cyanobacteria in the water can be estimated by the amount of chlorophyll-a that they contain.

'Algal blooms' reduce the amenity value of standing waters and the quality of freshwater habitats for wildlife, so it important that they are kept to a minimum, but they are increasing with climate change. To mitigate the impacts of climate change on Scottish standing waters, we need to understand the links between TP availability, water temperature and flushing rate, because these are key drivers of 'algal blooms' in these systems. For example, for a given amount of phosphorus in the water, the amount of cyanobacteria increases with higher temperatures.

We explored the potential impacts of two contrasting future climate and land use change projections for the 6,836 standing waters across Scotland for which we had sufficient data. Of the climate change x land use scenarios tested we found that the most extreme scenario (RCP6 x SSP3) generated a much higher risk of cyanobacterial blooms in a larger number of standing waters than the less extreme scenario (RCP2.6 x SSP1). In contrast, we found that reducing drain losses and keeping fertiliser application rates below the agronomic optimum tended to lead to lower chlorophyll-*a* concentrations in many standing waters.

Total phosphorus concentrations, water temperature, water retention time and risk of exceedance of WHO water quality thresholds for safe use (WHO 2003, 2004, 2021) were projected to 2080. The projected changes in these values across Scotland under the two different climate change projections and shared socio-economic pathways tested showed an improvement in water quality across the central belt of Scotland under RCP2.6 x SSP1, with a noticeable degradation in water quality across the north of mainland Scotland, between 2020 and 2080. In contrast, a marked increase in projected TP concentrations in standing waters across the south and west of the country was found under the RCP6.0 x SSP3 scenario over the same timescale.

In terms of amenity value, these projected TP concentrations were converted to likelihood of exceedance of WHO water quality thresholds for safe use, following the method developed by Carvalho *et al*. (2013). Under the worst case scenario tested (RCP6 x SSP3), many more standing waters are projected to fail WHO water quality standards for safe use for water supply or recreation than under the best case scenario tested (RCP2.6 x SSP1). This demonstrates that, although the evidence suggests that we can reduce the impacts of climate change in this way, the move towards more sustainable pathways for socioeconomic development and changes to land management practices is now urgent.

When the number of standing waters that are likely to meet Water Framework Directive (WFD) water quality objectives are considered, it was found that, under change scenario RCP2.6 x SSP1, there was a slight increase in the number of standing waters likely to fail WFD Good status for TP along the northern part of the Scottish mainland between 2020 and 2080. However, there was little change elsewhere. In contrast, under the more extreme change scenario (RCP6.0 x SSP3), the number of failures to meet WFD Good status for TP increased rapidly across all parts of Scotland over the same period. More specifically, under change scenario RCP2.6 x SSP1, the the number and percentage of standing waters that were projected to meet WFD High status or WFD Good status or higher for TP increased from 233 (74%) to 269 (85%), and from 262 (83%) to 285 (90%), between 2020 and 2080, respectively. In contrast, under change scenario RCP6.0 x SSP3, the the number and percentage of standing waters that were projected to meet WFD High status or WFD Good status or higher decreased from 230 (73%) to 92 (29%), and from 260 (82%) to 145 (45%), between 2020 and 2080 , respectively.

These results indicate that the future quality of Scottish standing waters is very dependent upon the socio-economic pathway that we follow, with water quality changing little or sometimes inproving under RCP2.6 x SSP1 and worsening under RCP6.0 x SSP3. Overall our ability to meet WFD water quality targets will have reduced markedly by 2080 unless a pathway of low emissions and sustainable land use is followed.

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Appendix 7 Review of in-lake measures for the management of eutrophication impacts caused by climate change

Background

Standing waters can be adversely affected by anthropogenic eutrophication where enrichment with phosphorus (P) and nitrogen (N) can promote algal growth favouring cyanobacteria dominance (Smith and Schindler, 2009; Hering *et al*., 2010). In addition, warming and changes in retention time associated with a changing climate may affect ecological responses to nutrient enrichment directly, reinforcing poor water quality conditions and biodiversity loss through a reduction in habitat quality and extent. For example, Phase 1 of this project highlighted that under high nutrient, warm and dry conditions, algal-related problems may be more likely to occur in lakes. High algal biomass has been associated with increased carbon burial in lakes (Anderson *et al*., 2020) and methane emissions from lakes to the atmosphere (Beaulieu, Del Sontro and Downing, 2019), where methane emissions are expected to increase with lake productivity, and, therefore, nutrient loading. These issues occur as a result of changes in in-lake processes, including interconnected cycling of nitrogen, phosphorus and carbon (Figure 37) and physical processes including thermal stratification, biological oxygen demand, and habitat disturbance related to extreme weather events. In addition, warming and other practices (e.g. fish stocking) may combine to increase the likelihood of nuisance and invasive non-native species spread and establishment (e.g. common carp, Skeate *et al*., 2022).

A range of in-lake management techniques have been proposed and, in some cases trialled, to manage nutrients and/or the effects of eutrophication and other stressors including harmful algal blooms,

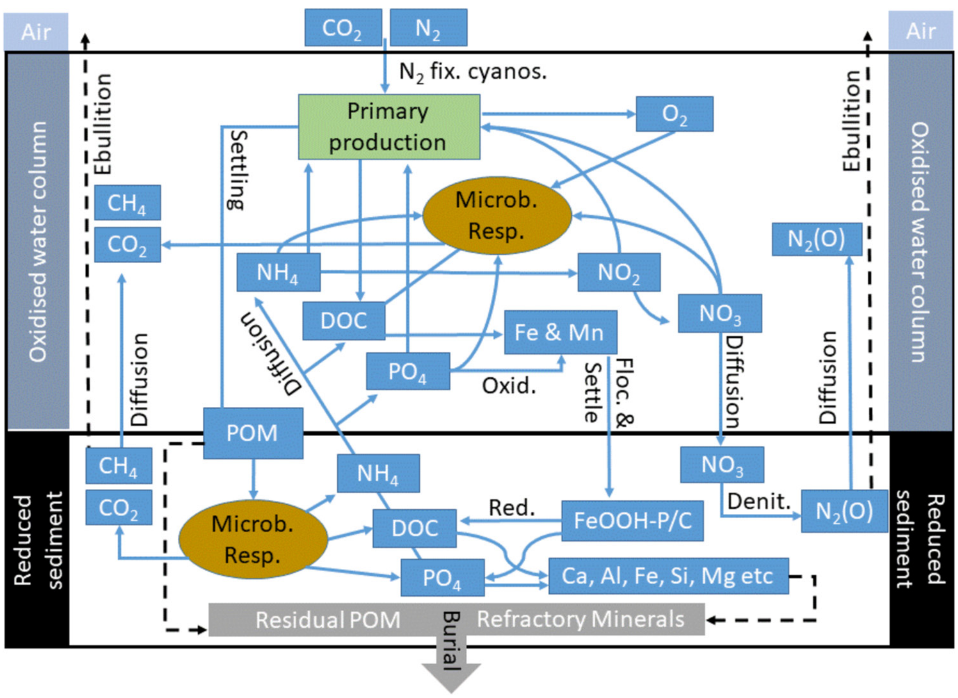

Figure 37 Coupled nutrient cycles operating across the sediment-water interface of lakes and reservoirs.

The figure aims to demonstrate that redox conditions at the sediment water interface is a key driver of the coupled nutrient cycles. Redox conditions may be sensitive to climate change, warmer drier conditions are expected to promote reducing conditions in bottom waters. When selecting suitable measures for the control of eutrophication, or for climate change adaptation, it is important that these processes and others (e.g. effects of warming and changes in retention time on physical structure of the lake; trophic interactions resulting from food web structure, and catchment nutrient loading) be examined for each site to reduce the risk of management failure. Many in-lake management measures are designed to reduce PO_4 in the water column and/or maintain oxidising conditions in the sediment surface. Where successfully applied, such measures should result in an overall reduction in DOC, PO_4, Fe, Mn, NH_4, and CH_4 fluxes from bed sediments to the water column, and a reduction in primary production through the feedbacks indicated in our simplified model.

anoxia/hypoxia, habitat creation and improvement and species control. These measures have the potential to be considered as climate change adaptation measures. For example, where a lake is sensitive to the effects of warming and nutrient enrichment, additively, then reducing one stressor may counteract the effects of another (Spears *et al*., 2021). Using this rationale, recent focus has been given to the potential for climate change adaptation (i.e. managing the lake to lessen the severity of climate change stressors; Spears *et al*., 2022) and mitigation (i.e. reduction of greenhouse gas emissions; Beaulieu, Del Sontro and Downing, 2019) through management in lakes. However, while a large body of evidence exists documenting successes and failures of in-lake measures (e.g. Cooke *et al*., 2005; Spears and Steinman, 2020), few studies have considered the utility of well established in-lake techniques for climate change adaptation and/or mitigation. In addition, novel approaches are emerging and need to be assessed fully, prior to their consideration for wide scale application.

We present a synthesis of available evidence on in-lake management techniques focused primarily on eutrophication control but also including other measures in the context of wider climate change adaptation options.

Aims and objectives

We draw on evidence produced from recent literature reviews to summarise management measures proposed for use in-lakes. We highlight the intended application and evidence of effectiveness of these techniques. Recommendations for developing site-specific plans including these measures is provided.

Methods

Literature review methodology

Results presented here are a summary taken from a systematic review covering the period 1998 to 2023 This review aimed to collate and synthesise available existing information on potential measures for controlling internal P loading in lakes and reservoirs, using standard literature review techniques (e.g. Collins *et al*., 2015).

In the selection of evidence, priority was given to studies conducted in the UK but the wider European and international literature in English was also consulted across primary evidence sources (i.e. grey literature, peer-reviewed literature (using Web of Science and Google Scholar) supplemented by expert knowledge of literature published more recently than 2019). The final WoS search used the following key search terms: *(reservoir* OR lake*)* AND *(sediment*)* AND *(removal OR control* OR method* OR mitigation OR technique* OR material* OR restoration OR Phoslock) AND (DOC OR nutrient* OR phosphorus OR taste OR odour OR redox sensitive metal*)*. This yielded 4,371 records, the titles and abstracts of which were manually checked to confirm relevance. An additional literature search was carried out on Google Scholar to check for any peer-reviewed articles and grey literature not picked up in the above Web of Science search. This list was supplemented by citations within both the already selected peer-reviewed and 'grey' literature; and from our own knowledge of relevant published and non-published reports in this particular field of research.

The evidence drawn from the above literature searches and expert knowledge was then compiled, providing the evidence base for this review on potential methods for the management of in-lake nutrient processes, as a means of controlling eutrophication. For each measure, consideration is given to the (i) intended mode of action, (ii) a description of operation of the measure, and (iii) examples of field scale application including limitations or recommendations from these studies. Consideration is given on the application of measures in the context of climate change adaptation. We propose that selection of measures is conducted at a site-specific level within a probabilistic framework to inform the development of adaptation scenarios.

Results

In this section, we provide a high-level review of a range of measures proposed in the literature for in-lake application. These measures have been proposed generally for the management of eutrophication symptoms and for habitat improvement and biodiversity enhancement. Here, we extend the scope of their application to include adaptation to climate change stressors including warming and changes to precipitation patterns, including floods and droughts. The review has been focussed on assessing evidence on (i) the mode of action of the measure; (ii) a description of the operation; and (iii) evidence of efficacy at the whole lake scale. We do not consider here all measures that have been proposed for use in lakes. Instead, we focus our review on those measures that are most commonly proposed or that have been trialled, including those that have been trialled and where evidence does not necessarily support their application at the whole lakes scale. This could be, for example, as a result of operational issues at the field scale relating to non-target effects (e.g. ultrasound). A range of measures not included here have been assessed in other studies and discounted for wide scale application (e.g. effective microorganisms, golden algae, plant/tree extracts, Bokashi balls; Lürling *et al*., 2016) and many measures proposed for use have received insufficient assessments to warrant inclusion.

Sediment excavation (redistribution and removal)

Mode of Action. This approach has been applied to deepen water bodies for navigation and to improve water quality by removing nutrient rich bed sediments shown to be responsible maintaining high water column P concentrations. The specific objectives of the approach extend beyond the reduction of internal loading and vary across case studies. Peterson (1982) reviewed these objectives and, with modifications, these include (1) to deepen the waterbody for improved recreational use or navigation, (2) to prevent or reduce internal P loading through the removal of P-laden sediments, (3) to deepen the lake to improve growing conditions for macrophytes, and (4) to expose viable macrophyte seed banks in deeper sediment layers to encourage growth of native species.

Description of Operation. Cooke *et al*. (2005) describe two main methods of sediment removal: dredging and excavation. Dredging involves the removal of sediment using either hydraulic or mechanical dredges (Lürling *et al*., 2020). Excavation, in contrast, requires that the overlying water is removed before the upper sediment layer is mechanically removed (Lürling *et al*., 2020). Large-scale sediment removal may require access to extensive areas of land for disposing the removed sediment (Peterson, 1982; Cooke *et al*., 2005), although in Denmark trials are underway to re-reuse lakebed sediments as an agricultural fertiliser. Dewatering of sediments prior to re-use or disposal should also be considered (Oldenburg and Steinman, 2019). If the sediment removed by either dredging or excavation is contaminated (e.g. with heavy metals), there may be yet further costs associated with its safe disposal.

Evidence of efficacy at whole lake scale. Lürling *et al*. (2020) review a range of sediment removal case studies reporting variable successes globally, noting short term effects (i.e. a few years) where catchment nutrient loading had not been sufficiently reduced. In a review of some 49 case studies, Pierce (1970) reported that it was impossible to draw conclusions on the effectiveness of the sediment removal activities due mainly to data paucity or concurrent catchment or in-lake management events. Although some short-term studies do show that sediment removal can result in reduced sediment P release rates (e.g. Kleeberg and Kohl, 1999, Reddy *et al*., 2007), there are fewer long-term studies to judge whether such confidence is justified. The Broads Authority published a Dredging Disposal Strategy (Broads Authority, 2009) as part of a wider Sediment Management Strategy (Broads Authority, 2010). This addressed disposal issues in detail and considered nutrient recycling opportunities, for example, through use in creating habitat for wetlands or terrestrial biodiversity. Phillips *et al*. (2014) reviewed sediment removal activities in the Norfolk Broads and reported an improvement in sediment P concentrations, generally, although improvements were limited as a result of persistent catchment P loading (i.e. improvements lasting a few years). In a detailed assessment of the combined effects of catchment P load reduction and sediment dredging in Barton Broad, Phillips *et al*. (2020) conclude that sediment dredging may have reduced the recovery time of the lake following catchment nutrient load reduction. The current Hoveton Great Restoration Programme will produce an assessment of multiple restoration techniques, including sediment management to create littoral habitats (Bringing life back to The Broads - GOV.UK (www.gov.uk)).

Chemical amendments

Mode of Action. There are three main categories of materials that will be considered here; coagulants; oxidisers/P binders; and algaecides/peroxides (Lürling *et al*., 2020). Coagulants (e.g. aluminium salts or organic polymers include Chitosan) are used to aggregate cyanobacterial biomass (and in some cases dissolved P), and, commonly with the use of a ballast (e.g. sand or clay), rapidly remove them to the lakebed. Oxidisers and P binders are groups of compounds applied to control either redox processes directly (i.e. Oxidisers; e.g. oxygen releasing materials; nitrate containing compounds; direct O_2 injection to sediments) or to alter the chemical composition of the bed sediment to control the release of phosphorus to the water column (i.e. P binders; aluminium sulphate; lanthanum-modified bentonite; aluminiummodified zeolite; the latter may be effective for phosphorus and ammonium control). Combined application of coagulants and P-binders have been trialled to simultaneously control algal blooms and sediment P release in some lakes, with the P-binder acting also as ballast. Copper-based algaecides including copper sulphate and copper citrate have been used although peroxide based algaecides including hydrogen peroxide are gaining popularity.

Description of Operation. Chemical amendments are applied to lakes using a variety of techniques. In general, amendments are added either directly to the surface of the water column or to deeper waters using hoses or dispersal units. For solid phase P binders and some liquid coagulants, materials are mixed with water from the lake to create a slurry prior to application. For coagulant application, ballasts are typically added prior to coagulant application. For direct sediment treatment, for example, in the application of oxidisers, materials may be applied through bottom landers. Application of amendments can be planned to allow for variation in sediment chemistry or in-lake physical structure to target applications towards areas where sediment P release is most likely to occur (e.g. in high sediment P, deeper water, anoxic zones). However, natural internal mixing processes may limit the extent to which spatial targeting may be effective over the longer-term. Of particular note is the need to estimate effective dose accurately. For all materials, this requires comprehensive understanding of the chemistry of the receiving waters and the behaviour of proposed materials within them to avoid non-target effects (Douglas *et al*., 2016).

Evidence of efficacy at whole lake scale. A wide range of whole lake case studies are available in the literature with which to assess the effectiveness of chemical amendments (reviewed by Lürling *et al*., 2020). For example, Huser *et al*. (2016) review the effects of alum (i.e. aluminium sulphate) application on water quality and ecology in 114 treated lakes across the USA, Germany, Denmark and Sweden, covering nearly a 50-year period of monitoring data. On average, reductions in TP concentrations related to the effective control of internal loading lasted 11 to 15 years (range 0-40 years), although effect size and longevity varied greatly among the case studies. Importantly, lake morphology was a key determining factor, where deeper lakes exhibited longer effect periods (i.e. 21 years on average) when compared to shallow lakes (5.7 years on average). For example, an important caveat when considering alum application for internal loading control is its interactions with pH, especially in low alkalinity low DOC waterbodies, where Al_3^+ (pH < 4.5) and $Al(OH_4)^-$ (> pH 8.5) ion formation can represent a significant ecotoxicological risk (Tempero, 2015). The use of buffering agents are critical in determining the correct alum dose to reduce potential ecotoxicological effects in such situations. Case studies exist in the literature for other materials with lanthanum bentonite and iron chloride applications well documented in the literature (e.g. including UK lakes; Spears *et al*., 2018; Perkins *et al*., 2002). The findings of Huser *et al*. (2016) are generally applicable to many amendment types targeting a reduction in sediment P release in that efficacy is dependent upon lake chemistry and physical conditions (Spears *et al*. 2016). However, where applications have been effective, longer-term positive effects on ecological communities have been reported including a reduction in cyanobacterial biomass, an improvement in water clarity and recovery of the aquatic plant communities. In contrast, the application of algaecides, including peroxides, may provide rapid reduction in algal biomass but the effect will be short-lived (i.e. days to weeks; Matthijs *et al*., 2012). Nevertheless, a number of studies have identified potential negative ecological effects of the use of chemical amendments including the release of cyanotoxins during algae breakdown (algaecides, hydrogen peroxide, and coagulants; Pei *et al*., 2014), non-target ecotoxicological effects (e.g. peroxides, herbicides; Jancula and Marsalek, 2011; Weenink *et al*., 2022) and the potential for persistence of potentially toxic elements in the water column, sediments and food webs of treated lakes (potentially, all products; Spears *et al*., 2013; Spears *et al*., 2018; van Oosterhaut *et al*., 2020).

UKCEH maintain an archive of materials that have been proposed for use in the control of P in fresh water ecosystems for eutrophication management. This has been compiled over the period 2007 to present and currently consists of over 65 records with physical samples held for 27 materials, only 15 of which are considered 'established' in that a commercial supplier was identified and product information was available including material safety data sheets and data on operational performance. However, very few materials have been comprehensively assessed for use in either lakes or drinking water reservoirs. In some instances, suppliers hold insufficient evidence on their specific materials to support large-scale application.

Physical alterations

Ultrasound

Mode of Action. The use of ultrasound devices has been proposed by industry for the control of algae biomass targeting the physical disruption of algal cells through the propagation of sound waves (Purcell *et al*., 2013). Laboratory studies have confirmed that ultrasound application can affect cyanobacterial growth under laboratory conditions through the collapse of gas vesicles and through the disturbance of cell walls (Wu *et al*., 2011 and 2012; Rajasekhar *et al*., 2012).

Description of Operation. A range of ultrasound emitting devices are available commercially for deployment to the water column of lakes with devices designed to emit across a range of frequencies. Lürling *et al*. (2014) assessed the operational performance of three commercially available devices and report a measured range of 'block' or 'square' waves of 20 to 44 kHz with an acoustic power of 0.7 (±0.2, 1 SD) W, giving an intensity of 8.5×10^{-4} W mL^{-1} tested in 800 mL solutions. The intensity of a device increases as a function of device power and decreases with distance form source, yet cell disruption increases with intensity. Ultrasound is used at high power and intensity in disinfection processes, and so at these operating conditions will be effective at reducing algal biomass. However, moderating device power to intensities capable of algal cell disruption while avoiding damage to other organisms at the whole lake scale is problematic (Rajasekhar *et al*., 2012). Manufacturers of the devices tested by Lürling *et al*. (2014) propose an effective range of 10-12 m. Lürling *et al*. (2014) reported that the commercially available devices trialled in their laboratory study killed the zooplankton *Daphnia magna* within 15 minutes of operation under their test conditions.

Evidence of efficacy at whole lake scale. Lürling *et al*. (2014) review evidence on effectiveness of ultrasound across five field trials. Although these trials are not scientifically comparable, Lürling *et al*. (2014) conclude that the operation of the devices resulted in no significant reductions in cyanobacteria or chlorophyll-*a* concentration at the field scale and note that potential non-target effects must be considered prior to use in lakes.

Aeration and Physical Mixing

Mode of Action. This management strategy is based on the principle that by promoting oxidising conditions in bed sediments (i.e., well mixed and aerated water column) that the release of redox sensitive elements (e.g. phosphorus, iron and manganese) common under anoxic stratified conditions, can be controlled (Smolders *et al*., 2006). Aeration has also been used to drive artificial destratification in an attempt to alter the phytoplankton community directly, and by limiting the competitive advantage of surface forming species through light limitation achieved by continual mixing.

Description of Operation. There are three main types of oxygen enriching devices commonly employed: airlift aerators; 'double bubble' contact or Speece cones; and deep oxygen injection systems or bubble plume diffusers (Cooke *et al*., 2005; Singleton and Little, 2006). Scottish Water use the ResMix system in several of their managed reservoirs. This aeration system involves the use of a floating axial propeller system that skims large volumes of water from the surface and pushes it down to break up the layers caused by thermal stratification.

Evidence of efficacy at whole lake scale. Despite the widespread use of hypolimnetic oxygen enrichment as a lake restoration measure to control internal P loading over many decades, its effects are considered to be variable (Lürling *et al*., 2020). However, effective control of redox sensitive element release from bed sediments using aeration systems has been documented when coupled to high frequency monitoring systems in drinking water reservoirs allowing targeted operation and adaptive management (Carey *et al*., 2022). Bailey-Watts, Wise and Kinka (1987a, 1987b) reported no reduction in the release of redox sensitive elements as a result of artificial aeration in Coldingham Loch, UK. However, in Coldingham Loch, aeration resulted in the prevention of surface blooms of *Aphanizomenon*, reduced late summer chlorophyll-*a* levels by about half (*ca*. 20 µg L^{-1}) and increased water clarity (*ca* 3.0 m) during two years of operation.

Hypolimnetic withdrawal and hydraulic recharge

Mode of Action. Hypolimnetic withdrawal involves the removal of nutrient rich water from the hypolimnion of lakes or reservoirs in an attempt to exhaust sediment nutrient pools to reduce the period of internal loading (reviewed in detail by Nürnberg, 1987, 2007, 2019). The approach is most suited to stratifying lakes where the release of redox sensitive elements (predominantly targeting phosphorus) has been confirmed as a driver of surface water quality impairment (Zamparas and Zacharias, 2014). The replacement of withdrawn nutrient rich water to compensate for the volume of water removed is a prerequisite (Nürnberg, 2007) and allows potential for controlling water temperature using input waters of cooler temperature to that of source water.

The assessment of hydraulic recharge has gained attention in recent years where low nutrient input waters are introduced to the impacted lake as a diluent. This approach has also been considered with respect to water temperature management and stratification disruption in the context of climate change adaptation (Olsson, 2022). In addition, the management of lake flushing rates has been considered to reduce algal biomass in Loch Leven (May and Elliott, 2019).

Description of Operation. A wide range of hypolymentic withdrawal facilities have been designed and are in operation in Europe and North America, providing a mature engineering base for the approach. Facilities typically rely on mechanical or passive syphoning of bottom waters through glass fibre or high-density polyethylene pipes to an onshore treatment works or discharge location (Klapper 1985, Scharf *et al*. 1992, Macdonald *et al*. 2004). The former is suited for remote areas and allows a reduction in energy costs for long-term operation. Nürnberg (2007) reports on the engineering design of 29 installations including pipe diameters (range, 8.9 to 100 cm), pipe length (range, 50 m to 2856 m), withdrawal water depth (range, 5.4 to 48.5 m), and total phosphorus export (range, 2 to 810 kg TP yr^{-1}). Passive withdrawal is proposed by Nürnberg (2007) as a low cost energy efficient application of hypolymentic withdrawal. Hydraulic recharge relies on a source of low nutrient input water with which to moderate thermal balance, nutrient concentrations, and flushing rate of the lake (Ollson *et al*., 2022), or the presence of a managed outflow with which water can be held back and flushing rate increased during peak algal growth periods (May *et al*., 2019).

Evidence of efficacy at whole lake scale. No known case studies of hypolimnetic withdrawal or hydraulic recharge exist in the UK, to our knowledge. However, hypolimnetic withdrawal has been applied and documented outside of the UK. In a review of effectiveness in about 40 European and 8 North American lakes, Nürnberg (2007) concludes the approach is an efficient restoration technique in stratified lakes and documents significant reductions in bottom water total phosphorus concentrations following application. Evidence across these case studies for improved surface water quality is variable, although reductions in chlorophyll-*a* concentrations are reported for 4 of 5 lakes and improvement in water clarity is reported for 19 of 21 lakes, for which data were available. Olsson *et al*. (2022) review the alteration of retention time in Elterwater, Cumbria. Despite a successful; reduction in water residence time by 40%, little change was reported in thermal stratification, hypolimnetic anoxia, P concentrations and algal biomass, suggesting that the treatment intensity was insufficient. These authors have developed decision support recommendations for application of hydraulic recharge for lakes in the context of effective climate change adaptation and internal P loading control. Although sources of sufficiently low nutrient water for dilution may be an insurmountable issue in many cases (Welsh, 1981), where sources are available dilution using alternative inputs can be an effective measure, especially in shallow lakes (Hosper and Meyer, 1986). Treatment of high P input water using P amendments prior to introduction to lakes may improve the capacity for dilution. Elliott and Defew (2012) highlight, using modelling, the potential for climate change to drive changes in flushing rates increasing the magnitude of cyanobacterial blooms in lochs and reservoirs, using Loch Leven as a case study. May and Elliott (2008 and 2019) develop this modelling approach further to demonstrate that relatively minor changes to the flushing rate of Loch Leven through management of a sluice gate could reduce algal blooms by up to 40%.

Habitat and Biodiversity Management

The improvement and creation of habitat coupled with species reintroduction and eradication to enhance biodiversity recovery has received attention in the literature. Studies in the literature generally include accounts of measures proposed to (i) increase habitat and species connectedness, (ii) decrease habitat disturbance and provide refugia for littoral biodiversity, and (iii) eradicate or control non-native or nuisance species. In some instances, combinations of these measures may be considered, for example in implementing measures designed to restore native macrophyte communities (Figure 38).

Increase habitat and species connectedness

Mode of Action. Evidence from nutrient load reduction case studies indicates that biodiversity recovery may be slow following water quality improvement if there are constraints on biological dispersal. This may be especially important for the recovery of macrophyte communities but also for species which rely on macrophyte community structure for habitat, and, those species that have become locally or nationally extinct (Jeppesen *et al*., 2012). Such constraints may be related to poor ecosystem connectivity (e.g. hydromorphological alterations), local species extinctions, or historical habitat repurposing (e.g. reclaimed shoreline land for agriculture). However, it should be noted that there may be benefits of decreased connectivity, too, because it reduces the likelihood of invasion by non-native species (INNS)

Description of Operation. A range of measures may be considered to increase connectedness for the recovery of macrophyte communities in lakes, including translocation. Bennion *et al*. (2023) consider measures to enhance connectedness of communities in the landscape to enhance species dispersal and also connectedness to dormant macrophyte seed banks in deeper layers of the bed sediments, including an assessment of lakes of the Greater Glasgow area. In general, two restoration approaches are proposed by Bennion *et al*. (2023), natural recolonization and active reintroduction with measures including seed dispersal (Waters-Hart, 2019) and species translocation (Orsenigo, 2018), and, increasing connectedness to extant populations across lake districts. National assessment of threatened species, including macrophytes can inform wider scale conservation priorities. For example, Taylor *et al*. (2021) assess the national habitat conditions for *Najas flexilis* and identify a shortlist of sites with favourable conditions towards which conservation measures may be focused. In addition, seed collection from extant seed banks and germination procedures have been developed in another CREW project for *Najas flexilis* (Gunn and Carvalho, 2020) to support potential future recolonization and/or re-introduction programmes, the seed archive being held at the UKCEH laboratory in Penicuik. In work considering species protection and the prioritisation of conservation translocations consideration was given to issues such as the loss of low lying coastal sites to climate change related issues.

Examples of efficacy at whole lake scale. Blindow *et al*. (2021) review over 25 case studies on macrophyte transplantation (of charophytes) from 8 countries (no case studies from the UK) reporting at least partial success in 17 typically shallow lake case studies. Additional measures were typically employed across these case studies to support macrophyte transplantation. These included cutting competing macrophyte species, fish reduction (biomanipulation, see later), and nutrient reduction. Reasons for failure or partial success included competition with other species, persistent nutrient loading, and herbivory. Natural, or passive, revegetation case studies have also been documented. Choi *et al*. (2021) compare community establishment in ponds with either passive revegetation or initial planted communities and conclude that planting effects were minimal over a three-year colonisation period and that invasive species control may be more important than planting. As an example of active revegetation, the Norfolk Ponds Project has demonstrated recovery of historical plant communities (inc. the rare *Lythrum hyssopifolia L.*) following exposure of dormant seed banks following sediment removal in 15 farm ponds, so called 'ghost ponds' (Sayer and Parmenter, 2020; Sayer *et al*. 2022).

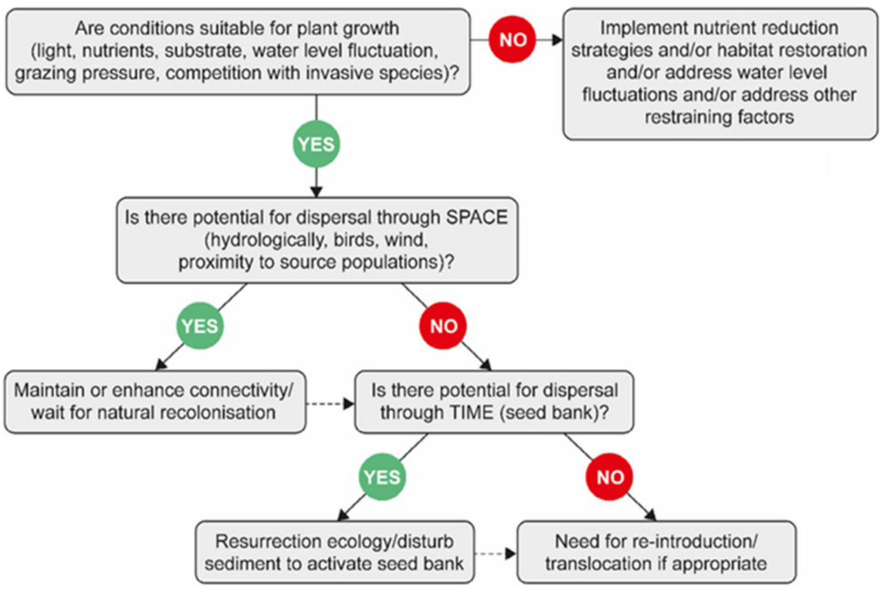

Figure 38 Flowchart showing a framework to guide conservation decisions for restoration of macrophytes in lakes including increasing habitat and species connectivity, water quality improvement, seed bank exposure, and non-native species control (dashed arrows indicate further options; after Bennion *et al*., 2023).

Decrease habitat disturbance and increased refugia for littoral biodiversity.

Mode of Action. There is increasing recognition that lakes, especially shallow ones, will experience an increased frequency and intensity of extreme weather events including increased storminess, which may limit biodiversity recovery. The renaturalisation or protection of shoreline habitats may therefore be necessary to reduce bank erosion and to improve habitat for wetland birds, nursery habitat for fish, and refugia for zooplankton and macroinvertebrates. Barriers including islands can act to dissipate energy across a lake's surface whilst providing refugia for littoral species, especially in shallow water areas. In some cases, riparian vegetation (i.e. 'bluff topography'; Markfort *et al.*, 2010) and artificial structures (e.g. dissipation reefs; Becker *et al.*, 2022) have been proposed to reduce wave power and habitat disturbance. It should be noted that, historically, shoreline habitat and water levels of Scotland's lochs have been heavily modified in many cases, as reviewed by Stratigos (2016). For example, the lowering of water levels through drainage to expose land for agriculture is documented for Loch Leven by May and Spears (2012). For shallow lakes, reducing water depth can increase exposure to wind induced mixing of bed sediments (Spears and Jones, 2011), although sediment dredging and hydrological alterations (both reviewed in a different context above) to deepen water bodies to their historical and natural bathymetries is not considered further in this review, due to social acceptability considerations.

Description of Operation. In the Netherlands, a pioneering restoration programme has led to the construction of the Marker Wadden Island system, composed of five islands in an archipelago, in Markermeer, a large shallow lake with European importance for waterfowl conservation (Marker Wadden | Rijkswaterstaat). The construction and biodiversity recovery effects of the project are described in detail by van Leeuwen *et al.* (2021). Island constriction was initiated 2016 through excavation and replacement of the lake's own bed materials, and designed specifically to improve biodiversity in response to a long-term downward trend in water bird populations and water quality. Between 2016 and 2020, five islands were constructed. First ring dikes were formed utilising deep sands extracted from 8 to 35m deep bed sediment layers. These dykes were then filled with clays and silts from the top 5–8m of bed sediment. These structures were shored up in 2017 by the installation of stone harbour allowing further construction of a long sand bank through which the archipelago could be extended to create a diversity of habitats including littoral and terrestrial vegetation, marshland, and gradual transition zones to open water. Examples of large-scale shoreline wetland creation and riparian habitat management are also available in Scotland. For example, vegetation management and the exclusion of human visitors (on mainland habitats) and predators (on island habitats) is in practice at Loch Leven Nature Reserve, for the protection of water bird nesting sites. At the same lake, the Royal Society for the Protection of birds and NatureScot have led on the construction of a wetland system on the south shore. Artificial reefs have been trialled, especially in coastal lakes and lagoons including in Lake Macquarie, Australia. Here, artificial reefs (Reef Balls®; Designed and restored reefs and aquatic ecosystems: Reef Ball Foundation) consist of a shallow water (4 6 m water depth) network of 180 small dome concrete structures (0.5 m high and 0.7 m in diameter), weighing approximately 80 kg each. These structures and others are designed to enhance fish community colonisation, for example, including holes to allow fish refugia (Maclean *et al.*, 2015; Becker *et al.*, 2022).

Examples of efficacy at whole lake scale. We note that a comprehensive scientific assessment of the effects of contemporary management approaches of riparian and shoreline wetland habitats on lake ecology and water quality in Scotland is lacking in the scientific literature. This is despite the apparent common application of these approaches in contemporary and historical land and lake management. Shoreline wetland habitat creation has been demonstrated to increase biodiversity including for water birds in other studies. For example, Zhang *et al*. (2021) report on the case study of Dongting Wetland Restoration Programme, China, in which habitat diversity and quality were improved between 2000 and 2006) through control of human access, the installation of protective dyke allowing control of drainage to the shoreline wetland, and planting of vegetation (e.g. reed, carex, phalarius and submerged plants). During a monitoring assessment (2012 to 2020) of the restored wetland relative to an unrestored control, these measures resulted in an increase in a range of ecological and biodiversity indices relating to water birds, also indicating improvements in abundance of other supporting trophic levels (e.g. species richness, density, foraging guilds, and recovery of target species). In the Marker Wadden project, van Leeuwen *et al*. (2021) review the responses in biodiversity and water quality in the area of the constructed islands within four years of construction. These authors report improvements in biodiversity across marginal (i.e. dominated by natural colonisation of *Tephroseris palustris* and managed colonisation of P*hragmites australis* and *Typha latifolia*) and aquatic vegetation (e.g. natural colonisation of eight charophyte species, *Potamogeton pectinatus, Nitellopsis obtuse, Zannichellia palustris*), an increase in the abundance of large bodies cladoceran zooplankton, and 19 fish species were recorded including evidence of high larval densities in the littoral habitats. Perhaps the greatest success for the Marker Wadden project was the response in the water bird community, the islands attracting > 20,000 sand martine, 43000 northern shovelers, 100 pied avocets and 100 black terns, the islands attracting >10% of the national population of common ringed plover and common terns within four years of construction. The extent to which this local benefit will impact national trends and abundances is not yet clear. The long term effectiveness of artificial reefs for application in freshwater lakes has received little scientific attention, although in a review of such structures in the Laurentian Great Lakes benefits to recreational fish and spawning were noted (Maclean *et al*., 2015). Efforts to transfer knowledge from coastal applications to freshwater lakes should be considered for this measure (e.g. Brochier *et al*., 2021).

Eradicate or control non-native or nuisance species.

Mode of Action. The ingress of non-native species (e.g. common carp, roach, signal cray fish, a wide range of aquatic macrophytes) either as contemporary colonisers following water quality improvement, or historical well-established communities can limit native biodiversity recovery through competition. Biomanipulation has been demonstrated across a range of sites where the eradication of non-native fish species can reinforce zooplankton grazing of the phytoplankton community and reduce sediment disturbance, improving water clarity and promoting aquatic macrophyte colonisation. The control of invasive species across other trophic levels including aquatic macrophytes and macroinvertebrates is challenging, although measures have been proposed and trialled, at least for macrophytes. Skeate *et al*. (2022) highlight the importance stocked sports fish for the poor conditions of Sites of Special Scientific Interest (SSSI) designated lakes in England and highlight that warming will exacerbate this issue through onset of successful recruitment, for example, of common carp in the British Isles. These authors highlight the importance of biomanipulatoin to control nuisance fish species in SSSI sites and go further to propose “A ban on fish stocking in SSSI lakes is recommended, unless fish are introduced to restore natural communities” (Skeate *et al*., 2022).

Description of Operation. Jeppesen *et al*. (2012) review various biomanipulation studies targeting enforcement of food web interactions designed to improve water quality and enhance ecosystem recovery following nutrient reduction. Measures, include stocking with predatory fish (e.g. pike) and removal of benthivorous omnivores (e.g. common carp), and zooplanktivore species (e.g. roach). The method of removal is variable but can include active measures such as netting or manual collection following drawdown and passive measures such as selective exclusion barriers on inflows for migratory species of concern. The use of piscicides (e.g. rotenone) has been considered but has the potential for non-target ecotoxicological effects. Regardless of the measure, repeated application may be necessary to ensure sufficient control levels. Hussner *et al*. (2017) review approaches and limitations for the control of invasive macrophyte species stressing the need for comprehensive knowledge of the ecology of the target species to inform measures selection. The restoration goal is also important to consider when selecting any biomanipulation measure, be it eradication, containment, or reduction in species abundance. Measures for invasive macrophyte control may include mechanical harvesting and excavation by boat or by hand, suction dredging, shading of infested areas with materials, water level drawdown, nutrient load reduction, the application of a range of herbicides (e.g. glyphosate and diquat), and biological controls including planting of free-floating or rooted shading species (Hussner *et al*., 2017).

Examples of efficacy at whole lake scale. A range of fish biomanipulation case studies are reviewed by Mehner *et al*. (2002) and Jeppesen *et al*. (2012). These studies stress the need to ensure fish reduction or stocking levels are maintained through repeated measures and that persistent nutrient loading will reduce efficacy. Jeppesen *et al*. (2012) provide synthesis across a wide range of field studies to propose recommendations for fish biomanipulation in temperate lakes. These include that (i) planktivorous and benthivorous fish stock be reduced over a 1-2 year period by 75-80 %; (ii) stocking of 0+ pike to densities of > 1000 ha^{-1}; and (iii) success of biomanipulation measures may be greatest when total phosphorus concentrations are reduced to below 0.05-0.1 mg P L^{-1}. Søndergaard *et al*. (2017) report on two biomanipulation events in a long-term study in Lake Væng, Denmark, reporting temporary improvements in water quality (e.g. reduction in phytoplankton biomass and increased water clarity) and ecology (e.g. increased colonisation of macrophytes and increased abundance of large bodied zooplankton). These authors conclude that, for this case study the control of roach and bream led to improved water quality (including a reduction in nutrients in the lake) for a number of years but that repeated fish removal may be necessary to sustain these effects, with lower effort required to deliver the same effects in subsequent biomanipulation events.Simberloff (2021) reviews the available literature on aquatic species reduction and eradication programmes. Of note for this review is advances in the use of eDNA techniques to detect invasion spread and to inform the need for eradication measures and their effectiveness (e.g. topmouth gudgeon control in ponds in England), and the use of biocides to control an isolated American signal crayfish population in a small waterbody, Ballachulish Pond, Scotland (Ballantyne *et al*., 2019). For aquatic plants, Simberloff (2021) highlights that the target of management can dictate the expected outcome, where control of small areas of a larger population (e.g. a specific bay within a lake) is unlikely to lead to eradication but may deliver effective local control necessitating a continuous management approach. In addition, some measures (e.g. mechanical harvesting and shading) may be considered as maintenance measures and will be less likely to result in eradication. Where herbicide application is regulated against, ecosystem scale eradication of established aquatic macrophyte species in lakes may be extremely difficult.

Measures for the management of lakes in the context of Climate Change Mitigation

Lakes and reservoirs have recently been identified as an important potential source of anthropogenic greenhouse gas emissions (predominantly as methane) to the atmosphere, and these emissions are expected to increase with eutrophication (Beaulieu, Del Sontro and Downing, 2019) and warming (Jansen *et al*., 2022). The importance of eutrophication management has therefore been proposed as an important consideration in reducing anthropogenic methane emissions from lakes and surface waters. For example, net global social costs of emissions reductions through nutrient management are expected to far exceed local cost benefits (e.g. through tourism and recreation) for Lake Erie, North America (Downing *et al*., 2021). This evidence base has led to early studies exploring measures (some included above) for in-lake management, which may lead to reductions in methane emissions from eutrophic lakes. It is important to note that lakes play an important role in the carbon balance, including carbon burial in sediments (Mendonca *et al*., 2017) as well as through the production of greenhouse gases (Figure 38). At the same time, lakes are being considered as 'hot-spots' or 'bright-spots' of renewable energy production, including floating solar installations (e.g. Exley *et al*., 2021) and water source heat generation (e.g. Gaudard *et al*., 2019). We review some notable case studies from the literature on recent advances in these fields.

Measures for the reduction of greenhouse gas emissions

The following studies provide at least some context to the potential effects of in-lake management measures in the context of methane emissions. These studies and others demonstrate a complex set of processes that must be considered when assessing the efficacy of in-lake measures for the control of methane emissions. We expect that, in time, this growing evidence base will be reviewed to produce guidance for measures selection and application in the context of climate change mitigation options for lake managers; as yet, this information is not available. One significant barrier is in the collection of high frequency data on greenhouse gas emissions and in-lake processes with which to underpin potential management measures. The UKCEH is currently leading on a national observing network across lakes and reservoirs to produce such data – the GHG Aqua Project (https://www.ceh.ac.uk/our-science/projects/ghg-aqua), including Loch Leven as a sentinel site. This will place the UK scientific community in a strong position to conduct experimental studies to assess the efficacy of in-lake, or in-reservoir, measures targeting a reduction in anthropogenic emissions.

- Nijman *et al*. (2022) in an 18-month mesocosm experiment demonstrate that the phosphorus binding material lanthanum-bentonite acted to decrease phosphorus and increase oxygen concentrations in bed sediment which reduced ammonium and methane production. In the same study, sediment dredging also reduced methane production, although here the process was likely due to the removal of organic matter from the sediment surface and a reduction in macrophyte cover.

- Pajala *et al*. (2023) report on a lake experiment to test the effects of artificial aeration of hypolimnetic waters in a stratified lake in Sweden. They report that although methane accumulation in aerated hypolimnetic waters was reduced, relative to a control site, whole-lake methane emissions during lake turnover were not. This indicates that the control of methane emissions in stratifying lakes should target key processes, including those driving emissions on turnover.
- Davidson *et al*. (2018) report on a mesocosm study to demonstrate that eutrophication can increase methane ebullition (and total diffusive + ebullitive flux) in shallow lakes and that warming acts synergistically with nutrient enrichment.
- Devlin *et al*. (2015) report in a whole lake experiment where they demonstrate that the presence of predatory fish (European perch) alter methane emissions through a reduction in zooplankton abundance related to an increase in methanotrophic bacteria in the water column.
- Van Doorn *et al*. (2023) report on a greenhouse experiment using organic rich sediment where isoetids (L*obelia dortmanna* and *Littorella uniflora*) significantly reduced methane production through radial oxygen losses from roots, which controlled redox associated processes (e.g. Figure 1).
- Zhu *et al*. (2023) assessed the role of assisted macrophyte community recovery in West Lake (Hangzhou, China) using planting and confirm that this approach can reduce methane production where increased macrophyte diversity is linked to the promotion of the benthic methanotroph community. So, the promotion of macrophyte community diversity may be important in limiting methane production from littoral sediments relative to low diversity communities.

Measures for renewable energy generation in lakes

The water environment is increasingly being considered as a source of renewable energy. Although Scotland has a long and well documented history of hydropower generation (Sample *et al*., 2015), considerations of surface water as a source of renewable energy through water source heat pumps and solar energy installations are less well developed. There is a water Source Heat Pump at Queens Quay on the Clyde and there have been enquiries about their installation on lochs. SEPA have produced regulatory guidance for the installation of these systems Some recent studies in this field have considered the potential effects of such climate change mitigation measures. These studies also assess the effects of renewable energy installations on lakes to counter the effects of climate change, for example, assessing the effects of operational design on water quality and physical conditions of the target waterbody. While these approaches are not yet widely applied on lakes, they appear to offer the potential to deliver both climate change adaptation and mitigation. In addition, there is potential to combine them with other in-lake approaches reviewed above, for example, the in combining water source heat extraction with hypolimnetic withdrawal for nutrient management.

- Exley *et al*. (2021) report a modelling assessment of floating photovoltaic cells using Windermere as a model lake. The authors highlight that installations on lakes may address the need for land pressures for large-scale solar farms (e.g. Thames Water's Queen Elizabeth II Reservoir). The model assessment provides data with that inform design of installations whereby reductions in water temperature and stratification and the shallowing of mixed depths could be moderated. Effects sizes reported were comparable to predicted counter-effects of climate change for the model system.
- Exley *et al*. (2022) assess using modelling the siting of a floating photovoltaic system on a UK reservoir highlighting that the positioning and construction of the installation could potentially be optimised (e.g. light climate, water temperature and stability effects) to reduce phytoplankton biomass. The authors stress the potential for undesirable ecological effects of incorrect design of floating photovoltaic installations and provide an approach to optimising co-benefits.
- Fink *et al*. (2014) report on a modelling study designed to assess the effects of heat demand form water source heat installations versus those of climate change in a typical large, deep, temperate lake. They conclude that heat extraction demand will have minor effects on lake water temperature, stratification, and seasonal mixing up to +/- 2 W m^{-2} and that those influences are insignificant relative to the effects expected due to climate change. Water

extraction and discharge could be tailored to reduce or enhance heating and cooling effects in the lake.

- Gaudard *et al*. (2018) report on an assessment of heat demand and potential from surface waters in Switzerland. They assess the effects of installations in the large upper Lake Constance (low population heat demand) and the smaller Lake Zurich (high population heat demand). They report potential mean temperature effects of the installations of -0.5°C to +0.02°C for Lake Constance and -0.6°C to +0.2°C for Lake Zurich. Again, these authors indicate that operational design and performance can be optimised to yield greater or lesser heating or cooling effects on the water body.

A synthesis of the potential effects of selected in-lake measures proposed for lake management is shown in Table 27.

Table 27 Synthesis of potential effects of selected in-lake measures proposed for lake management, with a focus on physical, chemical and biological responses with relevance to eutrophication management and climate change adaptation.

	Physical					Chemical					Biological				
Measure	**Relative cost**	**Water temp.**	**Wind exposure**	**Retention time**	**Thermal stratification**	**Dissolved Oxygen**	**PO_4**	**NO_3**	**NH_4**	**Metals & pollutants**	**HABs**	**Aquatic plants**	**Invertebrates**	**Fish**	**Birds**
Sediment excavation															
Sediment redistribution	H	↓	↓		↑	↑↓	↑	↓	↑	↑	↑↓	↑↓	↓		
Sediment removal	H	↓	↓	↑	↑	↑↓	↓↓	↓	↑	↓	↓↓	↑↓	↓↓		
Chemical amendments															
Coagulants	M					↓	↓			↑	↓↓↓				
Active nutrient binders	M					↑	↓↓↓		↓	↑	↓↓↓	↑↑↑			
Algaecides/peroxides	L					↓	↑			↑	↓↓↓	↓↓	↓	↓	
Physical alterations															
Aeration and physical mixing	H	↑			↓	↑↑	↓	↑	↓	↓	↓				
Hypolimnetic withdrawal	M	↓↑			↓	↑↑	↓↓	↑	↓	↓	↓				
Hydraulic recharge	L	↓↑		↑↑	↓	↑↑	↓				↓↓				
Habitat & biodiversity management															
Increased habitat/ecosystem connectedness	H											↑	↑	↑	↑
Island creation to decrease habitat disturbance	H		↓↓↓									↑	↑		↑↑
Re-introduction of aquatic plant species	L											↑			
Biomanipulation of fish (e.g. carp or pike)	H					↑↑	↓↓		↓		↓↓	↑↑	↑↑	↑↑	
Control of non-native aquatic macrophytes	M											↑			

Synthesis of potential effects of a selection of in-lake measures proposed for lake management, with a focus on physical, chemical and biological responses with relevance to eutrophication management and climate change adaptation. For chemical responses, we consider only the potential effects of the measure on in-lake processes. For biological responses, we consider biodiversity as a general indicator for aquatic plants, invertebrates (macroinvertebrates and zooplankton), fish and birds. Based on available evidence and expert judgement the following criteria have been developed: direction of arrow indicates likely effect direction with number of arrows indicating strength of effect. This assessment assumes the target effect has been achieved, considers reports of unintended effects reported or implied in the literature, and assumes that catchment nutrient load reduction is achieved as part of a comprehensive management programme. HABs – harmful algal blooms typically dominated by toxin producing cyanobacteria. Relative cost estimates adapted form Burford et al. (2019).

Conclusions

The predicted effects of climate change on lakes are expected to exacerbate the effects of nutrient pollution, at least in temperate lakes. This will probably be manifest through warming and changes in precipitation resulting in higher oxygen consumption in bottom waters and an increase in the release of legacy phosphorus and other redox sensitive elements (e.g. Fe, Mn) from bed sediments, fuelling algal blooms. The sensitivity of these processes to climate change will be highly lake-type specific (e.g. related to fetch, depth, food-web structure, nutrient loading, invasive species ingress and establishment, connectivity to local biodiversity inocula), yet the relative responses of different lake types across Scottish lochs is not, yet, well quantified.

In very general terms, where anthropogenic nutrient loading is high leading to hypereutrophic conditions, then the in-lake effects of climate change may be masked – the lake is so bad that climate change cannot make it any worse. However, where nutrient loading has been reduced, or is naturally low, the effects of climate change are likely to be more pronounced (Birk *et al*., 2020; Spears *et al*., 2021). So, the question of whether to consider in-lake measures to adapt to climate change effects will be related to whether or not local sensitivities are deemed unacceptable, for example, in the context of site-specific water quality, conservation, or recreation targets. Clearly, there is a need to revise existing targets to account for the effects of climate change.

Understanding the processes operating in lakes and the extent to which they will respond to climate change and nutrient loading (reduction and increase) will be key to informing in-lake management options, as well as catchment management approaches. However, national monitoring programmes are designed to inform regulatory reporting and are insufficient to allow such assessment to be made, with the notable exception of a few comprehensive long-term monitoring programmes. So, where national monitoring data and data screening exercises can indicate lake typologies or regions likely to exhibit high sensitivity to nutrient loading and climate change, site specific monitoring programmes should be tailored to inform the development of comprehensive lake management plans within a robust restoration/adaptation monitoring and assessment framework (e.g. an adaptation of the River Basin Management Approach).

On the question of whether measures are available to allow adaptation to climate change effects in lakes, the answer is 'yes', although no process is in place to inform water managers on their selection. In this Appendix, we review a range of in-lake measures setting out specific modes of action and application techniques reported across case studies. Arguably, no examples of long-term effective in-lake management for eutrophication control or climate change adaptation (or both) exist in Scotland. So, the evidence base draws on experiences from other countries (e.g. England, the Netherlands, Denmark, USA, China) where contemporary water quality has necessitated the development and implementation of in-lake measures. These experiences have been well captured in a growing literature base providing transferable knowledge on a range of measures with utility for Scottish lochs.

Collectively, the measures reviewed target changes in lake chemistry, and the ecological and physical structure of the water column and lake bottom habitats. Evidence from case studies including modelling assessments, laboratory and mesocosm trials and whole lake applications reported in the literature confirm that when both mode of action and application procedures are carefully designed, some of the identified measures may be useful in the control of eutrophication (e.g Table 27). In some cases, measures have been proposed for climate change adaptation and mitigation. However, many of these measures may be considered to be at a low Technology Readiness Level (TRL) with respect to their use in both eutrophication control and climate change adaptation in Scotland. The risk of management failure or of unintended consequences increases with decreasing TRL. Indeed, the lack of a robust framework to guide water managers on measures selection in Scotland risks the market being flooded with measures at a low TRL level (Spears *et al*., 2013).

There is a requirement now to accelerate testing and trialling of in-lake measures, perhaps through Scotland's Hydro Nation Research and Innovation Programme. In addition, the use of new in situ and remote monitoring techniques should be explored in the context of in-lake management. There is an opportunity to prime the market to develop and test new and emerging measures in collaboration with regulatory, industry, academic and policy partners. This should aim to deliver a suite of measures that can be applied through a portfolio approach at a high Technology Readiness Level. An opportunity exists to engage multiple co-beneficiaries in this process within a blended finance model. Co-benefit sectors may include sustainable nutrient management and nutrient recovery, biodiversity enhancement, human health risk reduction, and enhancing climate change resilience

There is a requirement to establish a regulatory framework for in-lake measures underpinning comprehensive and standardised validation. Here, experiences could be drawn from existing regulatory processes including for hydropower, solar and wind installation frameworks. A national roadmap for climate change adaptation (and mitigation) in lakes could be developed as part of the National Adaptation Planning process.

Going beyond the peer reviewed literature, there is a need to draw on practical experiences of the international community and to translate these into the Scottish context. The Scottish scientific community contributes to relevant international initiatives in this field providing scientific evidence to inform the development of sustainability policies on lakes (WWQA, 2023). Knowledge exchange with international partners could be strengthened, for example, through engagement with activities related to the UN Decade on Ecosystem Restoration, the UN Sustainable Development Goals (especially SDG 6), the Convention on Biological Diversity Global Biodiversity Framework 2030 (especially Target 2), and the United Nations Environment Assembly Resolution 5/4 on Sustainable Lake Management (UNEP/EA.5/Res.4). It should be noted that new monitoring and assessment processes are being developed across these initiatives as reported recently by the UN Food and Agricultural Organisation and the CBD joint meeting on Developing a Roadmap for the Kunming to Montreal Global Biodiversity Framework - Target 2 (Rome, 22-24 November, 2023), with relevance to lakes and climate change adaptation within the Scottish Biodiversity Strategy.

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Appendix 8 Stakeholder engagement

Background

Several policies aim to mitigate climate change by reducing carbon emissions and increasing carbon sequestration. However, even if climate change can be slowed down, it cannot be prevented or reversed. So, alternative approaches must be used to increase the resilience, and reduce the vulnerabilities, of people and nature to weather extremes and other impacts of climate change (Scottish Government, 2018).

In a recent project, May et al. (2022) explored the potential impacts of climate change on Scottish standing waters. They found that:

- Climate change is already affecting the quality of Scottish standing waters, especially in relation to increasing the risk of phytoplankton (algal) blooms.
- The main drivers of these changes will be increases in air temperatures and changes in rainfall patterns.
- More algal blooms will increase the risk of potentially harmful toxins being released into the water by cyanobacteria; this will reduce amenity value and wildlife habitat.
- The report concluded that adaptations to land and water policy will be needed, as part of Scotland's strategic and coordinated response to the climate crisis, to mitigate these effects.

Aims and objectives

Stakeholders from the Scottish freshwater sector were interviewed based on questions agreed with the project steering group (PSG). Opinions of participants were sought on the following issues:

- Current water quality issues caused by climate change on Scottish standing waters and who these are likely to affect.
- Potential mitigation measures that could be employed to help reduce the impacts of climate change on Scotland's standing waters.
- Potential policy changes needed to implement mitigation measures.

Methods

Between December 2023 and January 2024, the project carried out online interviews with individual stakeholders to gain an understanding of what stakeholders perceived to be the main effects of climate change on Scottish standing waters and what potential mitigation measures they suggested.

Invitees were recommended by members of the project research team and the project steering group as being key stakeholders within the freshwater sector in Scotland. Their areas of expertise ranged from farming, environmental regulation, consulting, conservation, and policy.

The following questions were asked during the interviews:

Q1: Are you concerned about the impacts of climate change on the quality of Scottish standing waters? Why?

Q2: What do you perceive to be the main water quality issues caused by climate change?

Q3: Who are the water quality issues likely to affect, and how?

Q4: What are the main measures that could be implemented to mitigate water quality issues in the short term (≤ 5 years) and in the longer term (> 5 years)?

Q5: Would changes to land and water policy be needed to ensure the effective implementation of these measures? What changes would you suggest?

Q6: Do you have any other suggestions to mitigate climate change impacts on Scottish standing waters?

The information collected was summarised and anonymised prior to reporting.

Results

Of the 24 stakeholders contacted, we received 14 replies, equating to a response rate of approximately 58%. Replies were received from environmental regulators (36%), government agency officers (21%), charity officers (14%), local council members (7%), farmers union members (7%), environmental consultants (7%), and policy makers (7%) (Figure 39).

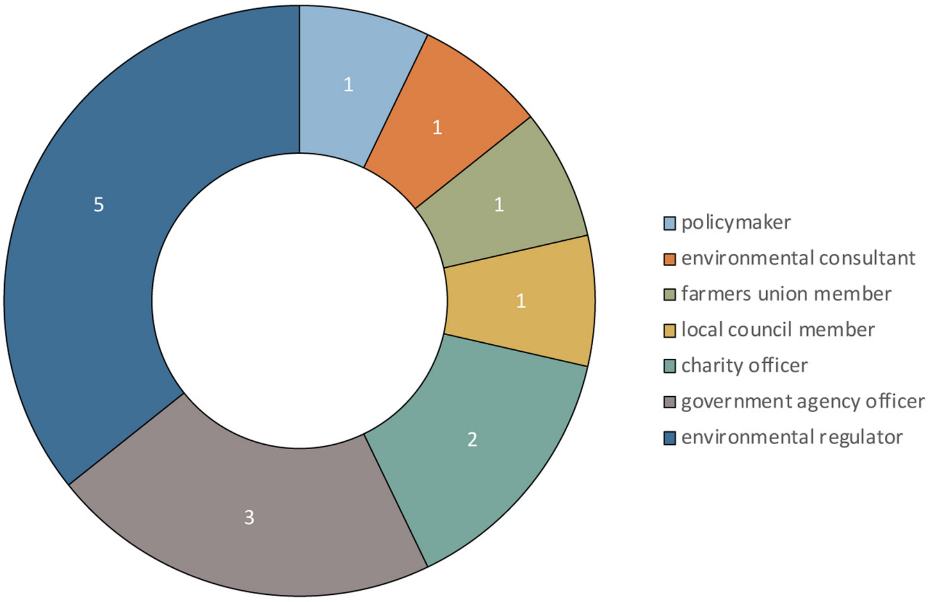

Figure 39. Associated bodies of the interviewees. Doughnut plot showing number of interviewees associated to a specific body (n=14).

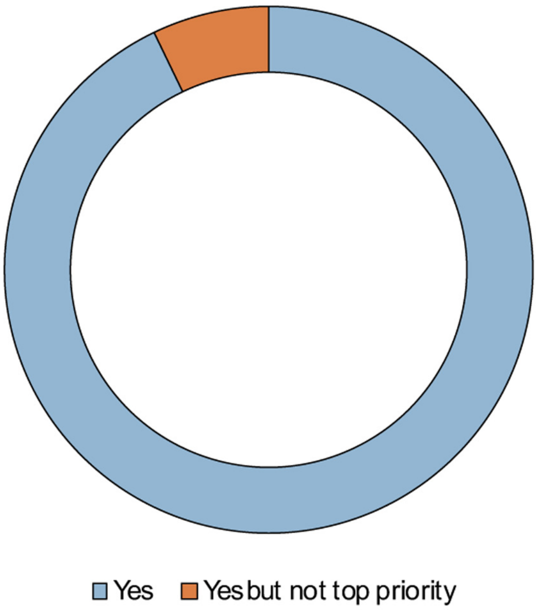

Figure 40. Interview responses to Question 1, part 1: Are you concerned about the impacts of climate change on Scottish standing waters? Doughnut plot showing responses and number of interviewees selecting a specific response (n=14).

Q1: Are you concerned about the impacts of climate change on the quality of Scottish standing waters?

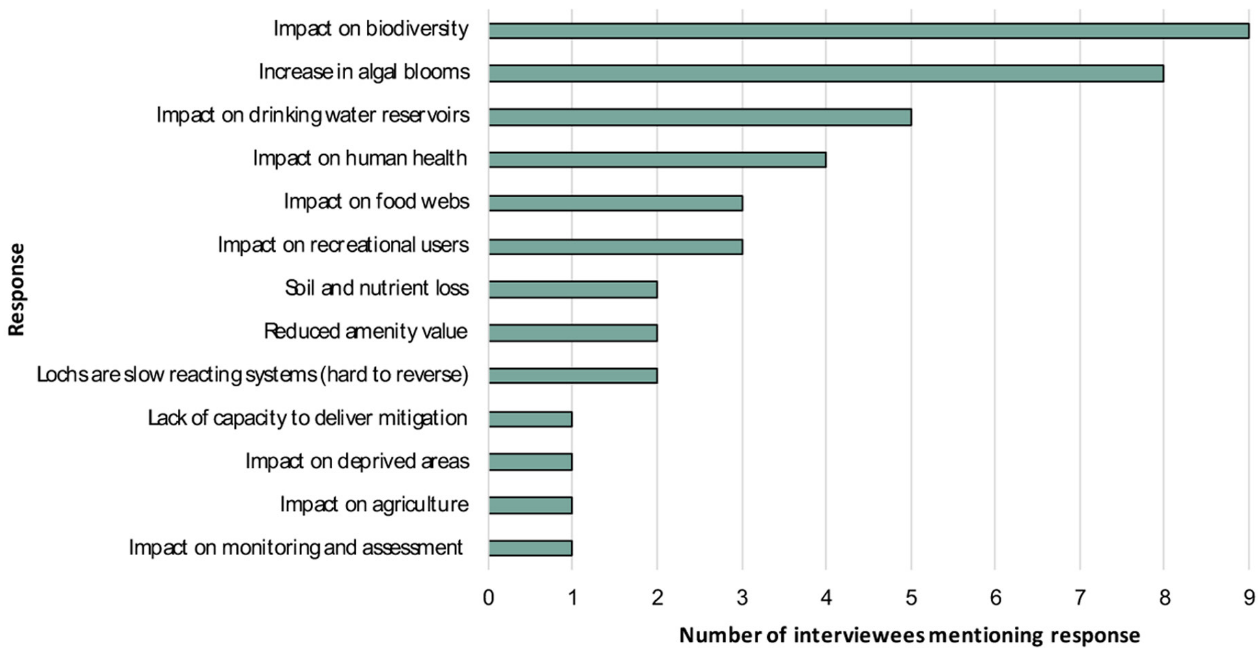

Figure 41. Interview responses to Question 1, part 2: Why are you concerned about the impacts of climate change on Scottish standing waters. Bar plot showing responses and the number of interviewees mentioning a specific response (n=14).

Q2: What do you perceive to be the main water quality issues caused by climate change?

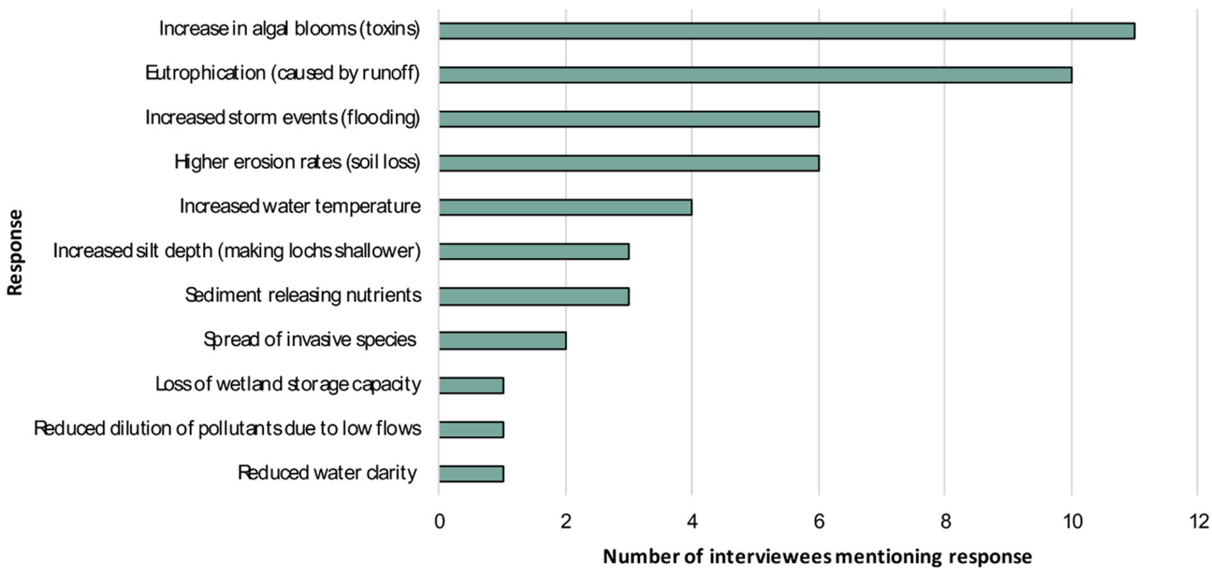

Figure 42. Interview responses to Question 2: What do you perceive to be the main water quality issues caused by climate change? Bar plot showing responses and the number of interviewees mentioning a specific response (n=14).

When asked whether they were concerned about the impacts of climate change on Scottish standing waters, the majority of interviewees (n=13; 93%) said yes, and a small minority (n=1; 7%) said yes but indicated that it was not their top priority (Figure 40). When asked to specify the reasons for their concern, most respondents mentioned impacts on biodiversity (n=9; 64%), increases in algal blooms (n=8; 57%), impacts on drinking water supply reservoirs (n=5; 36%) and impacts on public health (n=4; 29%). Other concerns expressed are shown in Figure 41.

When asked what they perceived to be the main water quality issues caused by climate change, most respondents mentioned increases in algal blooms (n=11; 79%), eutrophication problems caused by nutrient runoff (n=10; 71%), increased storm events and flooding (n=6; 43%) and higher soil erosion rates (n=6; 43%). Other issues were mentioned, as shown in Figure 42.

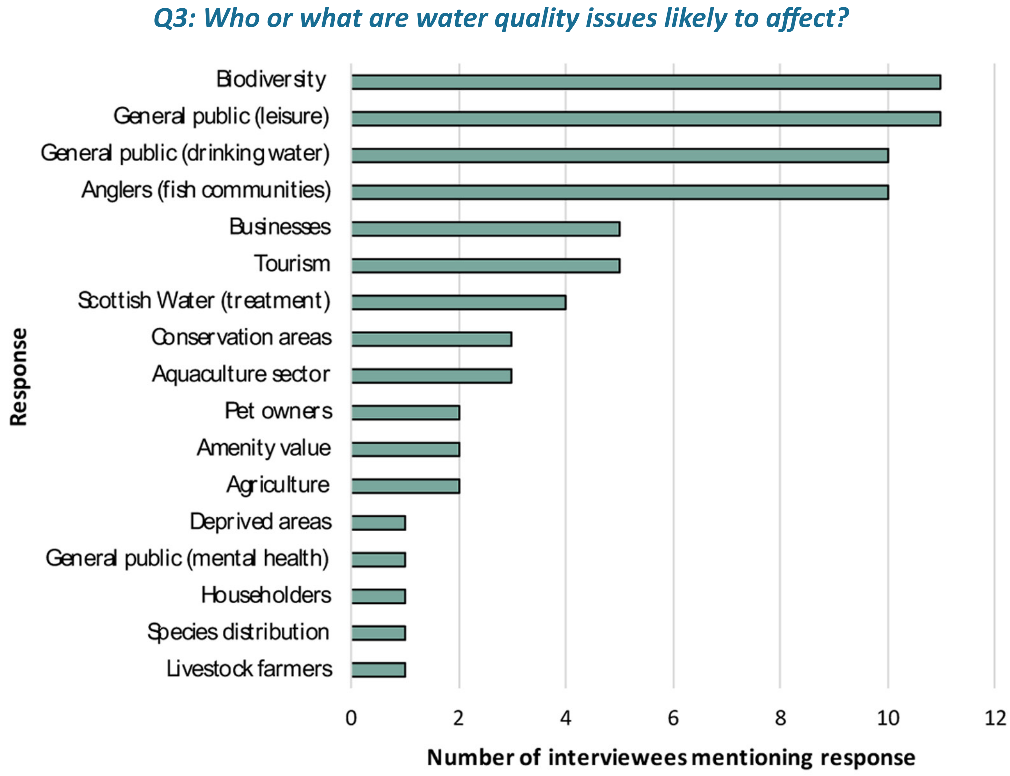

Figure 43. Interview responses to Question 3: Who or what are water quality issues likely to affect? Bar plot showing responses and the number of interviewees mentioning a specific response (n=14).

When asked who or what these water quality issues would be likely to affect, and how, most interviewees mentioned biodiversity (n=11; 79%), the general public in terms of recreational use (n=11; 79%) and drinking water supply (n=10; 71%), fish communities and anglers (n=10; 71%), as well as businesses (n=5; 36%) and tourism (n=5; 36%). Other entities likely to be affected by water quality issues were mentioned, as shown in Figure 43.

When asked what the main measures to mitigate water quality issues could be in the short term (≤ 5 years), more interviewees mentioned reducing diffuse pollution from farmlands (43%), engaging with farmers to enable change to more sustainable practices (36%), incentivising and monitoring actions by farmers (29%), engaging with land managers and landowners (29%), and increasing the number and/or the width of buffer strips (29%) than other potential interventions. The other possible short-term measures suggested are shown in Figure 44. The top suggestions for measures to be implemented in the short-term belonged to the engagement and funding and landscape management categories (Figure 44).

When asked what the main measures to mitigate water quality issues could be in the long term (> 5 years), more interviewees mentioned using geoengineering or innovative solutions (29%), re-evaluating the suitability of existing controls (29%), reducing runoff from land (29%), introducing more prescriptive land management guidance (21%), and having an agricultural reform (21%) than other potential measures. Other possible long-term measures suggested are shown in Figure 45. The top suggestions of measures to be implemented in the short-term belonged to the in-lake interventions (including geo-engineering) and policy and enforcement categories (Figure 45).

When asked if changes to land and water policy would be needed to ensure the effective implementation of mitigation measures, most interviewees responded yes (n12; 86%), with only a few leaving the response unclear (n=2; 14%), as shown in Figure 46. When asked what policy changes they would suggest, more respondents suggested increasing engagement and education with farmers and landowners (n=4; 29%), incentivising sustainable practices (n=4; 29%), changing the general binding rules (n=3; 21%), and generally implementing stricter policy (n=2; 14%) than any other changes. Other possible policy changes were suggested, as shown in Figure 47 .

Finally, when asked if they had any other suggestions to mitigate climate change effects on Scottish standing waters, most interviewees mentioned the need for a holistic and balanced approach (29%), along with other suggestions as shown in Figure 48.

Q4: What are the main measures that could be implemented to mitigate water quality issues in the short term (≤ 5yrs)?

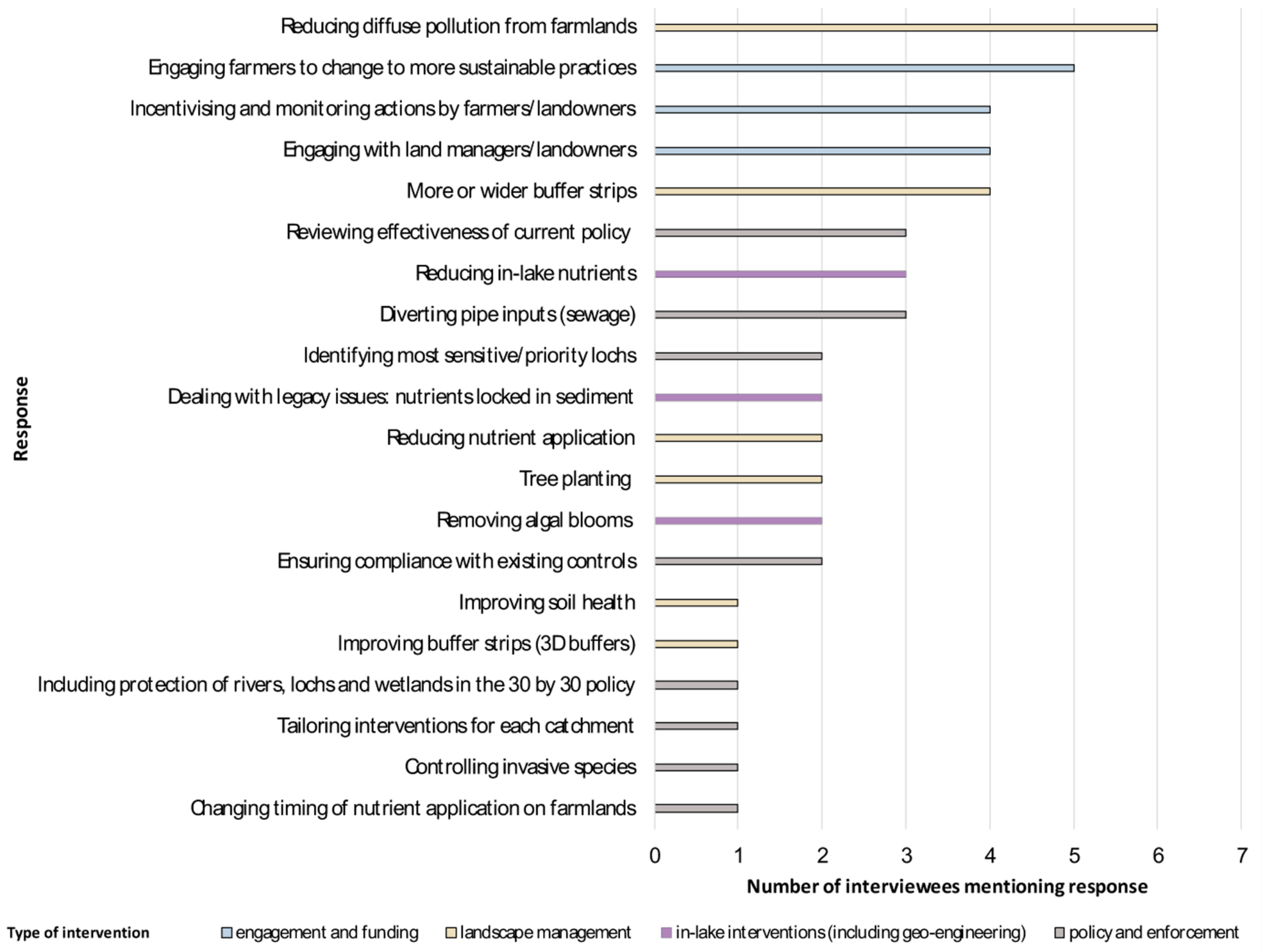

Figure 44. Interview responses to Question 4, part 1: What are the main measures that could be implemented to mitigate water quality issues in the short term (≤ 5yrs)? Bar plot showing number of interviewees mentioning a specific response (n=14).

Q4: What are the main measures that could be implemented to mitigate water quality issues in the long term (>5yrs)?

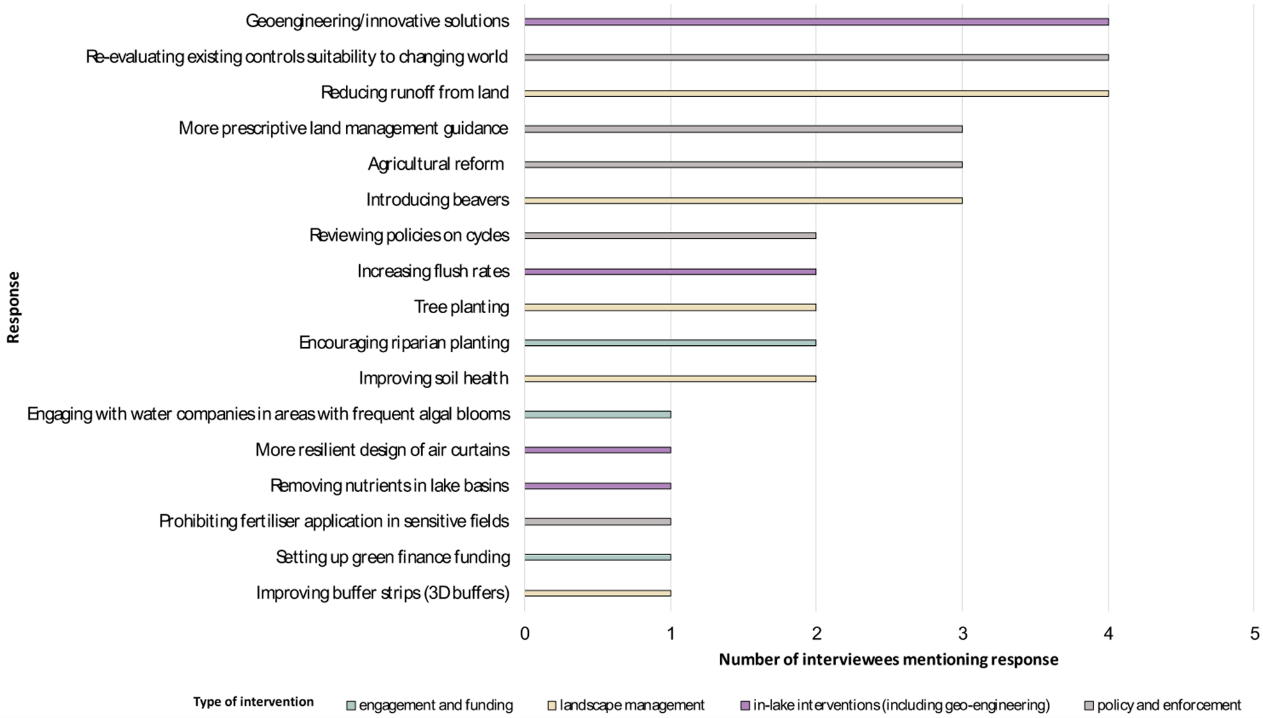

Figure 45. Interview responses to Question 4, part 2: What are the main measures that could be implemented to mitigate water quality issues in the long term (> 5yrs)? Bar plot showing responses and the number of interviewees mentioning a specific response (n=14).

Q5: Would changes to land and water policy be needed to ensure the effective implementation of these measures?

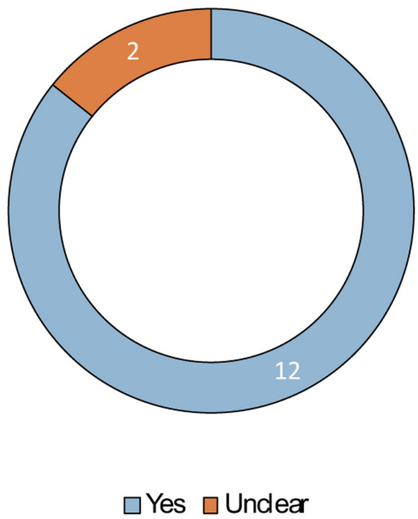

Figure 46. Interview responses to Question 5, part 1: Would changes to land and water policy be needed to ensure the effective implementation of these measures? Doughnut plot showing responses and number of interviewees selecting a specific response (n=14).

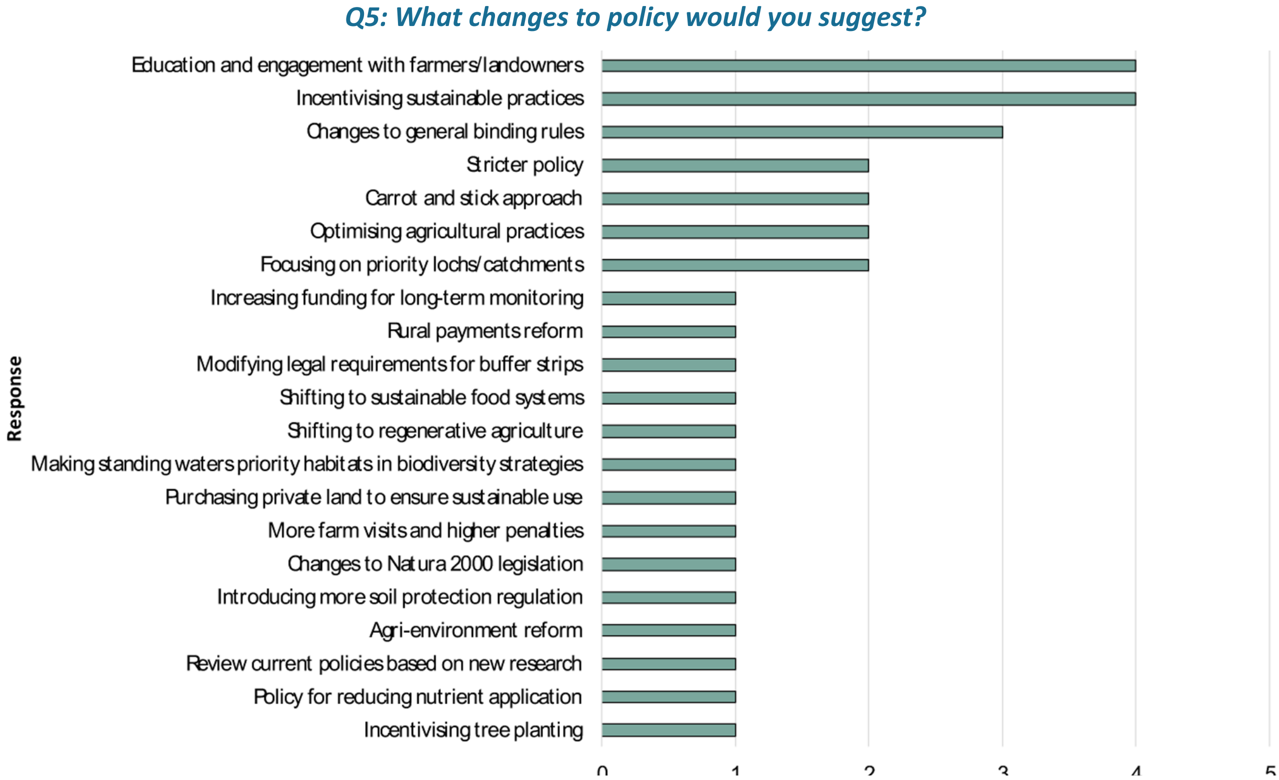

Figure 47. Interview responses to Question 5, part 2: What changes to policy would you suggest? Bar plot showing responses and the number of interviewees mentioning a specific response (n=14).

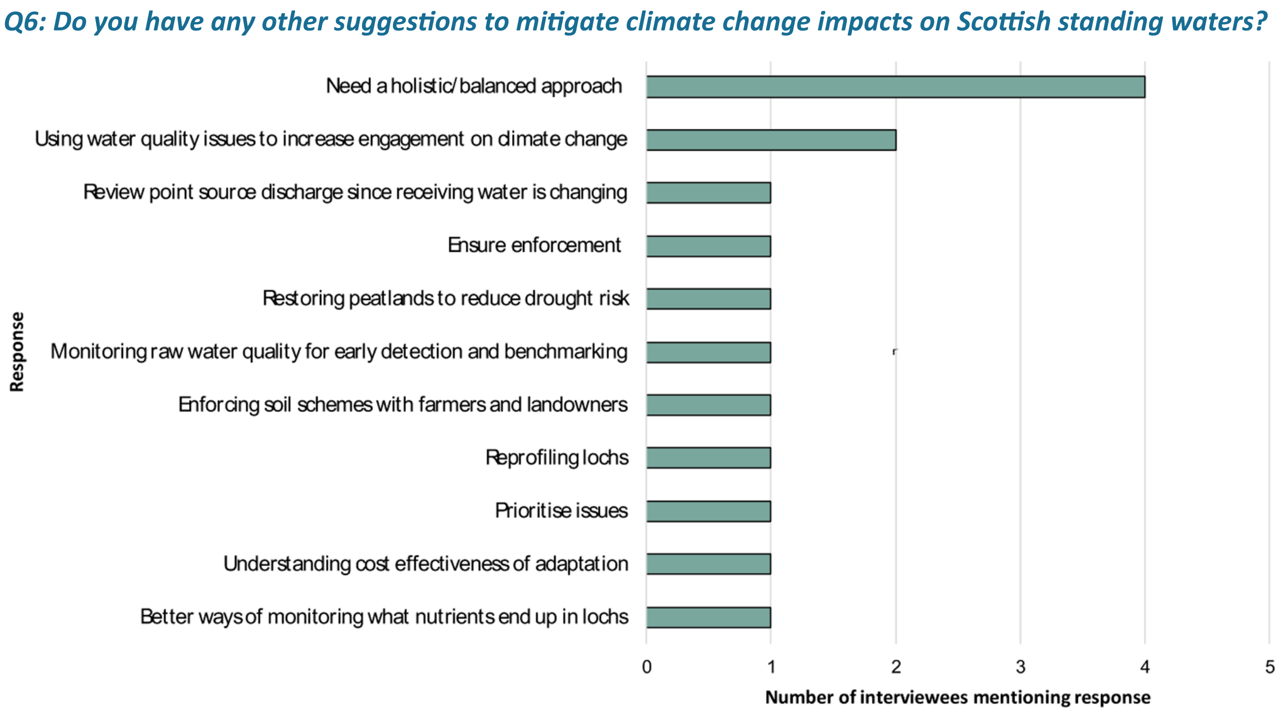

Figure 48. Interview responses to Question 6: Do you have any other suggestions to mitigate climate change impacts on Scottish standing waters? Bar plot showing responses and the number of interviewees mentioning a specific response (n=14).

Conclusions

Most (93%) of stakeholders interviewed for this project expressed concern about the impacts of climate change on Scottish standing waters, with increases in algal blooms (79%), eutrophication caused by nutrient runoff (71%), increased storm events and flooding (43%) and high soil erosion rates (43%) being the main issues mentioned.

In the short-term (≤ 5 years), interviewees mentioned reducing diffuse pollution from land (12%), engaging with farmers to enable more sustainable practices (10%), incentivising and monitoring actions by farmers (8%), engaging with land managers and landowners (8%), and increasing the number and/or width of buffer strips (8%) as being the main measures that need to be put in place.

In the long-term (> 5 years), geoengineering or innovative solutions (29%), re-evaluating the suitability of existing controls (29%), reducing runoff from land (29%), introducing more prescriptive land management guidance (21%) and having an agricultural reform (21%) were identified as being the most important mitigation measures to put in place.

Most interviewees agreed that changes to land and water policy would be needed to ensure effective implementation of mitigation measures (86%) and many (29%) mentioned the need for an holistic and balanced approach to tackling climate change impacts on Scottish standing waters.

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May, L., Taylor, P., Gunn, I.D.M., Thackeray, S.J., Carvalho, L.R., Hunter, P., Corr, M., Dobel, A.J., Grant, A., Nash, G., Robinson, E. and Spears, B.M. (2022) *Assessing climate change impacts on the water quality of Scottish standing waters.* CREW Report CRW2020-01.

Scottish Government (2018) Climate Change Plan: The Third report on Proposals and Policies 2018- 2032

Appendix 9 Policy relevance

Background

In their summary of the most up-to-date evidence of observed climate change trends across the UK (Kendon et al., 2023), the UK Climate Change Committee (CCC) noted that the following patterns of change had occurred within the UK in recent decades:

- Air temperatures were increasing, with 2022 being the warmest year on record since 1884 and the first year that the UK annual mean temperature had been above 10°C.
- Extremes of temperature are changing much faster than average temperatures.
- Rainfall extremes are higher and lower than before.

The CCC also summarised the major changes expected in the UK climate by 2050. These include:

- Warmer and wetter winters
- Hotter and drier summers

If global warming reaches 4 degrees centigrade above pre-industrial levels by 2100, then further significant changes in the UK climate would be expected by 2050. Like the rest of the world, Scotland is facing an unprecedented climate change crisis. Amongst other impacts, this will affect the quality of its standing waters.

In the first phase of this project, May *et al*. (2022) showed that:

- All types of standing waters in all areas of Scotland are at high risk of climate change impacts, although different types of standing waters are likely to respond differently.
- Initially, standing waters are projected to get warmer in the south and east of Scotland; however, this warming will reach all parts of Scotland by 2040.
- Increases in the temperatures of standing waters are closely related to changes in air temperatures; rapid and extensive warming of these standing waters has already occurred in recent years and is expected to continue.
- Climate change, mediated through increases in water temperatures and changes in rainfall patterns, will increase the risk of algal and cyanobacterial blooms developing in Scottish standing waters.
- The likelihood of cyanobacterial blooms, often associated with an increased risk of potentially harmful toxins being released into the water, will increase under warmer conditions and lower flushing rates.

To try to minimise climate change effects, the Scottish Government has enacted climate change legislation that aims to achieve net zero emissions of greenhouse gases by 2045. This includes an Update to the Climate Change Plan 2018-2032: Securing a Green Recovery on a Path to Net Zero. Scottish Government (2020) that reflects the ambitious new emission reduction targets set out by the Climate Change (Emissions Reduction Targets) (Scotland) Act (2019), which amended the Climate Change (Scotland) Act (2009). However, it has now been recognised that, even if climate change can be slowed down, it cannot be prevented or reversed, so alternative approaches are needed to mitigate its effects. Scotland's devolved statutory framework on climate change, established through the Climate Change (Scotland) Act (2009), includes provision for strategic planning to implement such climate change adaptions. It was envisaged in the Scottish Government's Climate Change Plan: The Third Report on Proposals and Policies 2018 – 2032 (RPP3) that this framework would include adaptive interventions to increase the resilience, and reduce the vulnerabilities, of people and nature to weather extremes and other climate change effects through successful adoption of appropriate mitigation actions and/or adaption strategies.

Aims and objectives

In the specific case of climate change impacts on Scottish standing waters, May *et al*. (2022a,b) recommended a whole system, catchment-based approach to mitigate future climate change impacts involving the setting appropriate water quality targets and planning interventions. This second phase of the project aims to identify and prioritise key changes in management practices that will be required to maximise the success of climate mitigation actions and/or adaptation strategies. In particular, this review addresses the following questions:

- Does existing water policy, and its implementation, sufficiently account for climate change impacts on the water quality of Scottish standing waters?

- In this specific policy context, what changes may be required and applied under current and projected climate change scenarios for adaptive management responses, monitoring and prioritisation of mitigation measures/ solutions?
- What are the recommendations, priorities for action and practical mitigation measures/ solutions that can be implemented in the short term (≤ 5yrs) and longer term (> 5yrs)?

Methods

We conducted a policy review to explore whether existing water policy, and its implementation, sufficiently accounts for climate change impacts on the water quality of Scottish standing waters. Then, we considered what needs to be changed in terms of policy responses to enable adaptive management responses, and the monitoring and prioritisation of mitigation measures/solutions. All relevant policy documents are included in this review.

Results

The increasing importance of climate change impacts on Scottish standing waters is reflected in a plethora of revised and recent legislation, policy goals, statutory commitments, and policy decisions at, global, European, UK, national, regional and local scales. These include a range of published documents, the key points from which are summarised below.

Global policies

The IPCC 2023 Sixth Assessment Report summarises the current knowledge of global climate change, impacts and risks, and climate change mitigation and adaptation. The report concludes that human activities, principally greenhouse gas emissions, have caused global warming and that global surface temperatures had reached 1.1 degrees centigrade above 1850–1900 levels by 2011–2020. Global greenhouse gas emissions continue to increase due to unsustainable energy use, and changes in land use, lifestyles and patterns of consumption and production, across regions, within countries and among individuals.

The Intergovernmental Science-Platform on Biodiversity and Ecosystem Services (IPBES) 2019 global assessment report on biodiversity and ecosystem services highlighted that climate change is increasingly exacerbating the impact of other pressures on nature and human well-being. In addition, the 2030 Agenda for Sustainable Development, adopted by all United Nation Member States in 2015, offers a global blueprint for peace and prosperity for people and the planet, now and into the future. Central to this agenda of sustainable development are 17 Sustainable Development Goals (SDGs), two of which are pertinent to this policy review: Goal 6 – Ensure availability and sustainable management of water and sanitation for all, and Goal 13 – Take urgent action to combat climate change and its impacts.

The fourth edition of the World Health Organization's WHO (2021). Guidelines for Drinking Water Quality builds on over 50 years of guidance by WHO on drinking-water quality, which has formed an authoritative basis for the setting of national regulations and standards for water safety in support of public health.

The primary aim of the World Health Organization's WHO (2003) Guidelines for Safe Recreational Water Environments is the protection of public health. The guidelines describes the present state of knowledge regarding the impact of recreational use of coastal and freshwater environments upon the health of users – specifically drowning and injury, exposure to cold, heat and sunlight, water quality (especially exposure to water contaminated by sewage, but also exposure to free-living pathogenic microorganisms in recreational water), contamination of beach sand, exposure to algae (including cyanobacteria) and their products, exposure to chemical and physical agents, and dangerous aquatic organisms. Also control and monitoring of the hazards associated with these environments are discussed.

European policies

The EU Drinking Water Directive – Recast (2020) is the EU's main drinking water legislation and covers access to, and the quality of, water intended for human consumption with a view to protecting human health.

EU Habitats Directive (1992) is the short name for European Union Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. This Directive led to the establishment of European sites and sets out how they should be protected. It also extends to other topics such as European protected species. In Scotland, the Habitats Directive was translated into specific legal obligations through the Conservation (Natural

Habitats, &c.) Regulations 1994. These cover:

- protecting sites that are internationally important for threatened habitats and species.
- a legal framework for species requiring strict protection.

The Habitats Regulations were amended in Scotland in 2019 when the UK left the EU. These amendments commit the Scottish Government to complying with the requirements of the Habitats Directives.

EU Water Framework Directive (2000) (WFD) aimed to protect European water environments by preventing their deterioration and improving their quality. The WFD requires river basin management plans (RBMPs) to be developed and then reviewed on a six-yearly basis, specifying the actions required to achieve environmental quality objectives. In Scotland, statutory objectives are set for Scottish waters through RBMPs, which are produced by SEPA on behalf of Scottish Government. These objectives are based on ecological assessments and economic judgments. The plans cover all types of water body, including rivers, lochs, estuaries, coastal waters and groundwater. For each river basin, each RBMP describes:

- the current condition of the water bodies.
- where historic or ongoing activities are reducing the quality of those water bodies.
- actions required to ensure that 'Protected Areas' meet required standards.
- actions needed to deliver environmental improvements to water bodies over the following six years and in the longer-term (up until 2027).

Two RBMPs, The River Basin Management Plan for Scotland 2021-2027 and the River Basin Management Plan for the Solway Tweed River Basin District 2021 Update, currently cover Scotland.

UK climate change risk assessment and biodiversity reports

The third five-year UK Climate Change Risk Assessment (CCRA3) report (UK Climate Change Risk Assessment 2022) to Parliament, as required by the Climate Change Act 2008, outlines the UK government and devolved administrations' position on the key climate change risks and opportunities faced by the UK, today. This is based on the technical report for the CCRA3 (Climate Change Committee (2021). Independent Assessment of UK Climate Risk), the statutory advice provided by the CCC. In this report, based on extensive new evidence, the CCC outlined:

- the UK's changing climate;
- priority risks for urgent further action;
- principles for effective risk management and adaption planning; and
- the benefits of adaption action.

The State of Nature UK Report 2019 presented an overview of how the UK's wildlife has changed over nearly 50 years of monitoring. This focused, especially, on what has changed over the last decade, and whether the situation for nature has improved or worsened. In addition, the report assessed the pressures on nature and the responses being made, collectively, to counter these pressures. The report demonstrated that the abundance and distribution of the UK's species has, on average, declined since 1970, with many metrics suggesting that this decline has continued into the most recent decade. Climate change was highlighted as one of the main pressures causing a net loss of freshwater biodiversity over recent decades.

Scottish national climate change related documents

Scotland's devolved statutory framework on climate change was established through the Climate Change (Scotland) Act (2009). This was subsequently amended by the Climate Change (Emissions Reduction Targets) (Scotland) Act (2019), which set out ambitious new emission reduction targets for net zero carbon by 2045. These amendments were enacted following a Scottish Government Climate Emergency Response Statement to the Scottish Parliament in May 2019, and in the light of advice from the UK CCC in recognition of growing evidence of a global climate emergency. In 2018, the Scottish Government published its third report detailing its proposals and policies to achieve decarbonisation targets by 2032 (Climate Change Plan: The Third Report on Proposals and Policies 2018 – 2032 (RPP3). Scottish Government (2018)). These were subsequently updated in 2020 (Update to the Climate Change Plan 2018-2032: Securing a Green Recovery on a Path to Net Zero) to include further actions to achieve this ambition.

The Climate Change (Scotland) Act 2009 included provision for the strategic planning of climate change adaptions. The second programme under the 2009 Act was the Climate Ready Scotland: Second Climate Change Adaption Programme

(SCCAP2) 2019-2024, which was published in 2019 in response to the second UK Climate Change Risk Assessment (CCRA2) published in 2017. This adaption programme aimed to build resilience to a changing climate within Scotland by adopting an outcomes-based framework of 170 policies. This aimed to take a people-centric approach to climate change adaption over the period to 2024 by promoting co-benefits and integrating adaption into wider Scottish Government policy, with a number of high level outcomes derived from UN Sustainable Development Goals (2030 Agenda for Sustainable Development) and Scotland's National Performance Framework. However, an independent assessment by the CCC of progress towards this target, while welcoming this vision of a well-adapted nation, highlighted that much work remained to be done to translate this ambition into actions that match the scale of the climate change challenge (Climate Change Committee (2022). Is Scotland climate ready? - 2022 report to Scottish Parliament). In particular, the CCC report highlighted that, for Scotland's adaption plans to be effective, there was a need to improve its monitoring and evaluation systems urgently to properly assess changes in climate-related risks. The scale of the challenge in adapting to climate change is set out in the Independent Assessment (CCC, 2021) used to inform the third UK Climate Change Risk Assessment (CCRA3) (UK Climate Change Risk Assessment 2022) and its underling reports, including the evidence compiled for the Scotland summary Evidence for the third UK Climate Change Risk Assessment (CCRA3): Summary for Scotland. A third Scottish statutory Adaption Programme is expected to be published in 2024 in response to CCRA3.

In 2020, the Scottish Government, recognising the intrinsic links between climate change and biodiversity loss, published a new Environment Strategy for Scotland. This is also reflected in the Scottish Environment Protection Agency (SEPA) regulatory strategy (Our Planet Prosperity – Our Regulatory Strategy). Under this strategy, SEPA aimed to deliver environmental protection and improvement by helping communities and businesses thrive within the environmental resources available by taking a co-ordinated sector planning approach across all sectors responsible for regulating. The overall purpose of this strategy was to create an overarching framework for all of Scotland's strategies and plans that concern climate change and the environment, with the ultimate aim of creating a net zero carbon, circular economy that reduces the global impact of consumption and helps ameliorate the biodiversity crisis. In 2013, the Scottish Government set out its view of the steps needed to improve the state of nature in Scotland (2020 Challenge for Scotland's Biodiversity). However, when it published its Scottish Biodiversity Strategy Post-2020: A Statement of Intent, it realised that the work needed to deliver this objective was even more complex and challenging than originally conceived, as evidenced by the State of Nature Scotland Report 2019 in which climate change was highlighted as one of the main pressures causing a net loss of biodiversity. Hence, the Scottish Government is currently consulting on a new, more ambitious 25-year Scottish Biodiversity Strategy (SBS) (Scotland's Biodiversity Strategy 2022-2045 (draft)) that will supersede the 2020 Challenge strategy. This new SBS aims to halt biodiversity loss by 2030 and reverse it with large-scale restoration by 2045.

Linked to the Environment Strategy for Scotland was the publication, under the remit of the Climate Change (Scotland) Act 2009, of the Scottish Government's Third Land Use Strategy 2021 (Scotland's Third Land Use Strategy 2021-2026.Getting the best from our land. Scottish Government (2021)). This outlines actions that Scottish Government believes would achieve sustainable land use while addressing climate change and biodiversity loss. For fresh waters, it talks about delivering 'healthy water, healthy land' by facilitating climate change resilience in catchments through habitat restoration measures and improved catchment management, e.g. of agricultural or industrial discharges. All of these improved land use management goals would be underpinned by the regulatory environmental actions undertaken by the responsible authority, SEPA. Key to achieving these environmental targets is the recently published National Planning Framework 4 (Scottish Government 2023), which was given enhanced status under the Planning (Scotland) Act (2019), and which sets out a new planning framework for Scotland in 2050 that replaces the previous National Planning Framework 3 and Scottish Planning Policy. The 2021 Third Land Use Strategy also indicated a commitment to developing Regional Land Use Partnerships (RLUPs) in which each a number of pilot RLUPs will be established to develop their own Regional Land Use Frameworks by 2023. Such frameworks will take a natural capital/ecosystem based approach at the landscape level for achieving positive land use changes that mitigate climate and environmental change impacts.

Specific Scottish national sectoral guidance, plans and regulations

River Basin Management Plans (RBMPs) form the main regulatory tools for improving water quality in Scotland. Arising from the EU Water Framework Directive (2000), RBMPs provide a framework for protecting and improving the water environment. Two Scottish RBMPs have been established, one for Scotland (River Basin Management Plan for Scotland 2021-2027) and a second for the Solway Tweed River Basin District (River Basin Management Plan for the Solway Tweed River Basin District 2021 update). Because Solway Tweed River Basin District is cross border, the management of this RBMP is jointly coordinated between the Environment Agency (EA) and SEPA. The RBMPs need to be reviewed every six years and the third cycle of Scottish RBMPs were published in 2021, setting out revised objectives for the period 2021 to 2027 and a programme of actions for achieving those objectives.

RBMPs form part of a range of environmental strategies and plans (e.g. Climate Change Plan, Biodiversity Strategy) within the overarching Environment Strategy for Scotland framework, described above, to ameliorate the impacts of both climate change and biodiversity loss. For example, work carried out for the Climate Change Plan is likely to create new Regional Land Use Partnerships, which will identify actions that can be taken to help achieve climate change targets. Underpinning all of Scotland's environmental strategies and plans is a drive towards achieving more sustainable and resilient land use and management through various partnerships and initiatives. In particular, SEPA are keen to continue to focus on controlling diffuse pollution within priority catchments that have been highlighted for water quality improvements within the third Scottish RBMP and Solway Tweed RBMP cycles. In addition, SEPA plan to identify areas where actions can best achieve improvements in water quality.

A key mechanism in river basin management planning for SEPA is the Water Environment (Controlled Activities) (Scotland) Regulations 2011 (as amended), more commonly known as the Controlled Activity Regulations (CAR). CAR was intended to regulate activities that may have an adverse effect on the quality of Scotland's water environment through an authorisation process, implemented by SEPA. The type of authorisation depended on the environmental risk posed by a proposed activity. CAR provided for three levels of authorisation: General Binding Rules; Registrations; and Licences. Through this authorisation process, conditions were set to protect the water environment, including standing waters. Other relevant environmental regulations in relation to the quality of Scottish water bodies are the Urban Waste Water Treatment (Scotland) Regulations 1994 (amended in 2003) (Urban Waste Water Treatment (Scotland) Regulations 1994; Urban Waste Water Treatment (Scotland) Amendment Regulations 2003). Under these regulations, Scottish Ministers are required to publicise, in accordance with Regulation 2, any decision taken on the identification of "sensitive areas" and "high natural dispersion areas", which are areas of water defined in accordance with specified criteria. Regulation 3 (amended in 2003) places a duty on Scottish Ministers and SEPA to ensure that their respective websites show maps of current sensitive areas and high natural dispersion areas with the dates that any such areas of water were identified or ceased to be identified. It also places a duty on SEPA to ensure that maps and other information are available for public inspection at its principal offices.

Of particular concern for standing waters in a changing climate are blooms – especially of cyanobacteria (blue-green algae). In recent years, these have become more common due to the combined effects of nutrient enrichment and climate change, and their complex interactions (e.g. warmer waters, shifts in seasonality, prevailing weather conditions, changes in flushing rate, and hydrological extremes such as floods and droughts), which may exacerbate the risk of water quality issues. Cyanobacterial blooms can:

- reduce the amenity value of standing waters
- increase public health risks
- increase water treatment costs
- prevent statutory environmental objectives from being met within policy/regulatory relevant timescales
- affect biodiversity
- reduce the capacity of water managers to deliver water quality improvement targets or maintain effective measures to prevent further deterioration.

Because of the potential risks to public health associated with cyanobacterial blooms in inland and inshore waters, in 2012 the Scottish Government updated their guidance to Directors of Public Health, to Heads of Environmental Health in Local Authorities and to others in Scotland

Cyanobacteria (Blue-Green Algae) in Inland and Inshore waters: Assessment and Minimisation of Risks to Public Health – Revised Guidance. Scottish Government (2012). This updated guidance has not been published, yet.

Sources of help and support

In addition to the above, Scottish Government's Vision for Scottish Agriculture focuses on sustainability and a just transition to a support framework that includes climate mitigation and adaptation. In February 2023, Scottish Government released a list of measures that are being considered to help agriculture plan for future change. The consultation on the new Agriculture Bill included proposals for its content aimed at delivering key outcomes; these include climate mitigation and adaptation. Preparing for sustainable farming is already helping businesses by providing support for carbon audits and soil sampling. There are also measures available under the Agri-environment Climate Scheme to help mitigate the impacts of climate change, with some measures to reduce the impacts of climate change being funded under the Forestry Grant Scheme. Peatland Action supports on-the-ground peatland restoration activities and is open to eligible land managers who have peatlands that would benefit from restoration. The Nature Restoration Fund, a competitive fund launched in July 2021, specifically aims to fund projects that are aiming to restore wildlife and habitats while addressing the twin crises of biodiversity loss and climate change. Farming for a Better Climate, which is run by Scotland's Rural College (SRUC) on behalf of the Scottish Government, combines ideas trialled by volunteer Climate Change Focus Farms and information from scientific research to offer practical advice that helps businesses choose the most relevant measures to improve farm performance and increases resilience to future climate change effects.

Conclusions

Climate change is currently affecting, and projected to further affect, standing water quality in Scotland (May et al., 2022a,b). This policy review has shown that adaptations to current water policy and existing monitoring networks will need to be included in Scotland's strategic and coordinated response to reducing climate change impacts on these waterbodies. This conclusion is supported by the recent Climate Change Committee (2022). Is Scotland climate ready? – 2022 report to Scottish Parliament, which highlighted that, for Scotland's adaption plans (e.g. the SCCAP2 programme) to be effective, Scotland needs to improve its monitoring and evaluation systems urgently to assess changes in climate-related risks and impacts.

Policy recommendations based on this review, and those suggested by May et al. (2022a) are given below. These are reported according to the global, national and regional impacts that they aim to address.

Global climate change impacts – adaptive national water policy perspectives

There is a policy gap between global and national understanding of the impacts of climate change on water temperatures and changing rainfall patterns that needs to be closed. Failure to address this issue and monitor key indications of climate-related risks effectively will undermine the development and implementation of adaptive water policy and any management practices intended to mitigate the complex interactions that affect water use and nutrient run off at regional and local scales.

National climate change impacts – adaptive regional water policy perspectives

Changes to national scale water policies and land management practices will be required to limit climate change impacts on the quality of Scottish standing waters in the future. These impacts will be mediated through shifts in catchment and in-lake processes associated with changes in nutrient runoff, flushing rates and water temperatures. In combination, these changes will exacerbate the future risk of algal blooms and may compromise Scotland's ability to meet statutory goals and regulatory targets within given timelines.

As it is likely that climate change and its effects cannot be addressed in the short-term, it is important to identify the main factors that limit algal growth and accumulation that can be controlled at national scale. For example, better control of nutrient losses to water from agricultural, industrial and sewage related sources may be required to reduce the likelihood of potentially harmful algal blooms (Scottish Government, 2012; May et al., 2019). In the past, many of these interventions have required a licence issued by SEPA under the Controlled Activities Regulation (CAR) (Water Environment (Controlled Activities) (Scotland) Regulations 2011 (as amended)). CAR has now been superseded by the Environmental Authorisations (Scotland) Regulations 2018, which aims to bring permitting across all regimes under a single integrated authorisation framework. To be effective, future licensing criteria will need to take account of climate change and the need for adaptation to reduce its impacts.

Other national water policy and land use management practices (e.g. River Basin Management Plan for Scotland 2021-2027, Scotland's Third Land Use Strategy 2021-2026. Getting the best from our land. Scottish Government (2021)) will also need to be take into consideration how national-scale climate driven risks affect the quality of standing waters at regional to catchment scales. The Climate Change Committee (2022). Is Scotland climate ready? – 2022 report to Scottish Parliament highlighted that the current River Basin Management Plan for Scotland does not include any specific actions or adaptions for countering changing climatic conditions. For example, it does not take increasing river temperatures into account. Also, although the Third Land Use Strategy highlights the need for sustainable land-use to help in climate change mitigation and adaption, like the RBMP it does not detail specific actions to achieve those objectives.

Revision of current nutrient status criteria for Scottish standing waters 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. For example, in relation to meeting EU Water Framework Directive (WFD) (EU Water Framework Directive (2000)) targets for Scottish standing waters, mitigation/adaptation strategies will need to be implemented to achieve good ecological status, prevent its further deterioration and guide restorative action. In addition, the recast Drinking Water Directive (EU Drinking Water Directive – Recast (2020)) will require Catchment Risk Assessments to be created for all drinking water catchments to increase the level of source control for pollutants (referred to as 'Hazards and Hazardous Events'). This will encourage a prevention-led approach to addressing climate change interactions with these catchment factors, instead of reactively managing potential impacts (e.g. of algal blooms) on public health with expensive water treatment processes.

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 – Revised Guidance. Scottish Government (2012), especially in relation to climate change impacts, by capturing new evidence that emerged from the May et al. (2022a,b) report. This would help protect the amenity value of locally important still waters (e.g. for recreational use, water supply and wellbeing purposes) and reduce climate-driven water quality risks to public and animal health, in addition to meeting climate change mitigation/adaptation needs through other policy routes.

Future monitoring

The recent Climate Change Committee (2022). Is Scotland climate ready? - 2022 report to Scottish Parliament made it clear that "Scotland lacks an effective monitoring and evaluation systems meaning that changes in aspects of many climate-related risks are largely unknown". In response to this, the existing monitoring network for Scottish standing waters needs to be reviewed, urgently, with a focus on developing an integrated approach for detecting climate change impacts whilst focusing on the use of new scientific innovations and resource capabilities. May et al. (2022) made the following recommendations for the future monitoring of key indicators of climate-related risks to inform adaptive water policy and management practices.

- 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 total algae and cyanobacteria, separately.

- 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; low flushing rates due to droughts).
- 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” can be integrated to improve our ability to detect and forecast change.

In addition. the incorporation of modelling into waterbody assessment processes would enable lessons learned from site specific monitoring to be extended to un-monitored sites.

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