Environmental impacts of floating solar photovoltaics on their host water bodies: opportunities and risks

Giles Sebastian Exley, BSc

Lancaster Environment Centre

Lancaster University

This thesis is submitted for the degree of Doctor of Philosophy

April 2022

Environmental impacts of floating solar photovoltaics on their host water bodies: opportunities and risks

Abstract

Freshwater ecosystems provide services that are essential for human survival. However, as the energy system is decarbonised, the surfaces of inland water bodies are increasingly being transformed to host floating solar photovoltaics (FPV). Water bodies are favoured over conventional ground and rooftop solar PV installations as they conserve limited land resources and provide higher electricity generation efficiencies. However, FPV represents a new stressor to water bodies. The permanent shading and sheltering effects of FPV arrays at the water's surface pose potential impacts to the functioning of the water environment. To date, impacts on the host environment, both the opportunities and risks, are poorly resolved, in the context of present and future climates. This thesis synthesises scientific and stakeholder knowledge from an evidence review and stakeholder engagement to define modelling experiments that investigate the opportunities and threats of FPV installations and aims to inform best practices and future management decisions.

Results reveal the effect of FPV on the water environment scales with coverage extent and siting location. Typically, FPV cools water temperatures, reduces stratification duration, and limits the growth of phytoplankton, with higher coverage leading to greater magnitude changes. Given these physical and biological changes, FPV may have the potential to reduce or offset some of the impacts of climate warming on water bodies, depending on FPV coverage and future emissions concentrations. The results suggest that FPV could be an effective tool for managing water bodies by improving water quality and enhancing ecosystem services. However, host water body response will be highly specific to siting location and coverage of FPV installations. Failing to understand the impacts of a specific FPV installation on the host water body could result in undesirable ecosystem impacts, curtailing this technology's deployment and slowing the net-zero energy transition.

Contents

List of Tables

List of Figures

Acknowledgements

It would not have been possible to complete this PhD without the help and guidance of my supervisory team, Alona Armstrong, Rebecca Hernandez, Andrew Folkard and Ian Jones. Their support and motivation have been invaluable. In addition, Trevor Page and Stephen Thackeray have supported me throughout, albeit without the formal title of supervisor. Also, thanks to my co-collaborator and friend, Alex Cagle.

This PhD would not have been possible without the financial assistance of United Utilities and the NERC Envision DTP.

I want to thank my industry contacts at United Utilities, Affinity Water, Southern Water, South East Water and Thames Water for their guidance on reservoir operations. In particular, special thanks to Carmel Burke, Clare Bullen, Adam Reynolds and Tony O'Brien at United Utilities for facilitating fieldwork at Langthwaite reservoir and to Michael Chipps at Thames Water for his detailed knowledge on the Queen Elizabeth II reservoir.

Fieldwork at Langthwaite reservoir would not have been possible without the help of Hollie, Charlotte, Alex and Sarah, who ably helped with boat operation and data collection.

Of course, my friends and family deserve a lot of credit for keeping me sane over the past four years. Imogen, Naomi, and Liam thank you for all the welcome distractions and trips that stopped me from working all the time. In addition, my housemates Dr. Charlotte Smith and Dr. Freya Olsson, thank you for all the tea breaks and opportunities to moan about PhD life. Thank you to Eden – whilst the endless phone calls often seemed like a distraction, I would not have had it any other way.

Declaration

I declare that this thesis has not been submitted in support of an application for another degree at this or any other university. It is the result of my own work and includes nothing that is the outcome of work done in collaboration except where specifically indicated. Excerpts of this thesis have been published in journals, as indicated within. Many of the ideas in this thesis were the product of discussion with my supervisors; Alona Armstrong, Andrew Folkard, Rebecca Hernandez and Ian Jones, and my co-collaborators; Trevor Page and Stephen Thackeray.

Signed: Giles Exley Date: 11/04/2022

Giles Exley BSc

Lancaster University, UK

Statement of authorship

All the data chapters in this thesis have been written for publication. **Chapter 2** and **Chapter 3** have appeared as peer-reviewed and published papers in *Renewable and Sustainable Energy Reviews* and *Solar Energy*, respectively. **Chapter 4** is in the submission process for the *Journal of Environmental Management*, and **Chapter 5** is being prepared for submission. GE is first author on all chapters, carrying out the data collection and processing, running modelling experiments, completing data analysis, and co-designing the studies.

Chapter 1 – Introduction

1.1 Overview

Global renewable energy demand is proliferating, accelerated by the switch from fossil fuels to low carbon energy sources to mitigate the impacts of climate change (IEA, 2019). Solar photovoltaics (PV) are an increasingly popular choice in the drive to decarbonise, with exponential growth worldwide (World Bank Group et al., 2018). Although PV deployment continues to grow, several pitfalls exist with traditional ground- or building-mounted PV, such as the occupation of scarce land resources, representing an opportunity cost in terms of lost agricultural or industrial land (Sahu et al., 2016; Cazzaniga et al., 2018; Cagle et al., 2020). Floating solar photovoltaics (FPV) help alleviate some of the barriers that may arise with traditional PV energy generation by occupying the surface of water bodies. However, potential concomitant effects of FPV are poorly resolved (Ziar et al., 2021), risking undesirable impacts on hosting water bodies or unidentified opportunities that might improve water body function.

1.2 FPV systems – design and deployment decisions

An FPV system is the ensemble of components required for collecting, converting and transmitting energy to the point of connection (e.g. an electricity grid). The components are comprised of a floating structure, PV modules, a station-keeping system, transmission cables and equipment for balancing the system (e.g. invertors) (Oliveira-Pinto and Stokkermans, 2020; DNV, 2021). The floating structure is typically composed of interconnected individual rigid floats to allow for movement in rough conditions, although single-body membrane style flexible floating structures are also used (Figure 1-1). The floating structures can be constructed from plastic, concrete or steel, depending on the design of the array (Kim et al., 2016; Cazzaniga et al., 2018). PV module choice is determined by FPV array design, with some floating structures requiring smaller PV modules and others supporting more substantial modules. The station-keeping system maintains the FPV array within prescribed limits and typically uses anchoring and mooring (Figure 1-1) (Rosa-Clot and Marco Tina, 2020; DNV, 2021). Anchors may be attached to the bed or the shoreline of the host water body (Friel et al., 2019). A smaller number of FPV arrays use fixed mountings that are permanent structures built on the bed of the host water body (Figure 1-1). Balancing equipment may be installed on-array or on-shore, with current transferred using floating or underwater cabling (Figure 1-1) (World Bank Group et al., 2018; Oliveira-Pinto and Stokkermans, 2020).

Figure 1-1 – FPV array components. a) a multibody FPV array, b) mooring equipment for the station-keeping system, c) a single-body membrane style flexible floating structure © Isifloating, d) floating cabling to transfer current from the FPV array to onshore balancing equipment, e) an FPV array using fixed mountings built on the bed of the host water body ©Munch, f) the underside of an FPV showing the PV module, HDPE float and the surface of the host water body.

Although FPV arrays comprise similar components, designs vary between manufacturers and by deployment decisions (e.g. panel tilt angle). Considerations for the suitability of a water body for an FPV array include; available surface area, current usage (e.g. for recreation or fisheries), accessibility (for installation and maintenance), and the availability of a grid connection or sufficient local energy demand (Zubair et al., 2020; Piana et al., 2021). The available water body surface area limits the maximum potential generation of an FPV array (Cuce et al., 2022). FPV surface coverage, which is the proportion of the host water body covered by an FPV array, can vary substantially (<2% to >80%; (Spencer et al., 2019; Cagle et al., 2020)). Typically, energy demand and the water surface use efficiency of the array (Cagle et al., 2020) determine overall surface coverage.

1.3 The rapid uptake of FPV

A growing body of evidence suggests that FPV enhances the capabilities of traditional PV, with several articles presenting corollaries to the deployment of FPV systems beyond electricity generation. Firstly, FPV has been shown to deliver an enhanced generation efficiency of between 0.79% to 12.5% compared to ground-based PV due to the cooling effect of the hosting water body (Choi et al., 2013; Sacramento et al., 2015; Yadav et al., 2016; Oliveira-Pinto and Stokkermans, 2020). Although the cooling yield has been found to vary across climates, with heat loss dependent on wind speed and the openness of the floating structure (Dörenkämper et al., 2021). FPV also offers scope to improve the power curve when deployed with hydro-electric generation (Liu et al., 2018; Lee et al., 2020). Secondly, FPV averts the need for large areas of land use change (Holm, 2017; Cagle et al., 2020); this is particularly beneficial to land-scarce countries, such as Japan, and areas with high land prices (Abid et al., 2019; Campana et al., 2019). Thirdly, although dependent on system design, FPV has been shown to reduce evaporative losses by 25 to 50% (Choi, 2014; Santafe et al., 2014; Sahu et al., 2016; Taboada et al., 2017); reductions of this magnitude make FPV particularly attractive to drought-stricken areas and would help water body managers preserve water supplies. Additionally, the shading from FPV has been hypothesised as a method to limit the photosynthesis of harmful algae blooms (HAB) (Sahu et al., 2016). With the prevalence of HABs forecast to increase due to climate change (Paerl and Huisman, 2008), water body managers are increasingly considering FPV as a means to prevent any HAB derived taste and odour issues (Rosa-Clot and Tina, 2018).

As a result of these co-benefits, the global growth of FPV has been exponential. FPV has established a foothold in the renewable energy mix, with over 200 FPV systems in operation globally (Solarplaza, 2019) and generating capacity exceeding 2.6 GW (Sanchez et al., 2021). Estimates suggest there is a conservative global potential for 400 GW-peak FPV (World Bank Group et al., 2018), with growth likely to continue at pace. While deployment speed has been rapid, understanding of potential detrimental environmental impacts and the response of the hosting water body is poorly resolved (Lee et al., 2020; Stiubiener et al., 2020; Zhang et al., 2020; Gorjian et al., 2021; Ziar et al., 2021).

1.4 The importance of water bodies

Humans rely on water bodies, such as lakes, reservoirs and irrigation ponds, to provide provisioning (e.g. water for consumption), regulating (e.g. flood control), cultural (e.g. recreation) and supporting (e.g. ecosystem resilience) ecosystem services (Postel and Carpenter, 1997). In the United Kingdom alone, water resources for public consumption have a Natural Capital asset value in excess of £109 billion (Office for National Statistics, 2021). Further, the ecosystem services delivered by water bodies extend beyond the needs of humans. Water bodies are vital habitats (Miranda et al., 2020), supporting biodiversity and large numbers of endemic species (Collen et al., 2014; Tickner et al., 2020). Water bodies face increasing stressors from anthropogenic activities, including eutrophication, pollution and over-abstraction. These stressors threaten the functioning of water bodies, affecting the ecosystem services relied upon by human populations and the natural environment.

Water bodies are facing increasing pressures from climate change, including warming surface water temperatures (O'Reilly et al., 2015), increased evaporation (Wang et al., 2018) and altered mixing regimes (Woolway and Merchant, 2019). Combined with other anthropogenic stressors, such as eutrophication, climate change is a significant threat to water quality and quantity. A water body's response to climate change depends on the individual system (Adrian et al., 2009), although some effects are likely to be widespread. Warmed and nutrient-enriched water bodies face extensive changes to planktonic communities (Woolway et al., 2020), such as increased cyanobacterial blooms that lead to oxygen depletion (Paerl and Huisman, 2009), increased turbidity (Jeppesen et al., 2015) and toxin production (Gallina et al., 2017). Increased cyanobacterial blooms can lead to increased water treatment costs, loss of tourism and recreation and pose a health risk to humans, livestock and pets (Steffensen, 2008). Interactions between climate change and other water body stressors are complex, presenting non-linear ecological responses that pose a challenge to developing effective water body management strategies to mitigate the effects (Woolway et al., 2020).

1.5 The interaction of FPV with the host water body

FPV represents a long-term perturbation to the hosting water body, given FPV has an anticipated lifespan of 20-30 years (Rodrigues et al., 2020; Charles Rajesh Kumar and Majid, 2021; Costa and Silva, 2021). Emerging evidence has shown that FPV systems will have potentially significant consequences on water body process and function with the magnitude and direction of the effect dependent on the scale of reduction to wind speed and solar radiation (Armstrong et al., 2020). However, knowledge is limited, and understanding is not keeping up with the pace of FPV deployment. Evidence gaps remain, including the effects of FPV on water temperature, stratification and phytoplankton biomass and species composition.

The physical presence of the FPV array will shelter the surface of the water from the wind and shade the water column, reducing the attenuation of solar radiation. Potential impacts of FPV on the host water body can be hypothesised based on the current understanding of lake systems (Figure 1-2). FPV will modify the balance between stratifying (solar radiation) and mixing (wind-induced turbulence) forces at the air-water interface, depending on the individual system's shading and sheltering effect. The subsequent modification to the thermal structure of the water column may perturb existing stratification regimes, modifying in-lake processes and making water body management less predictable.

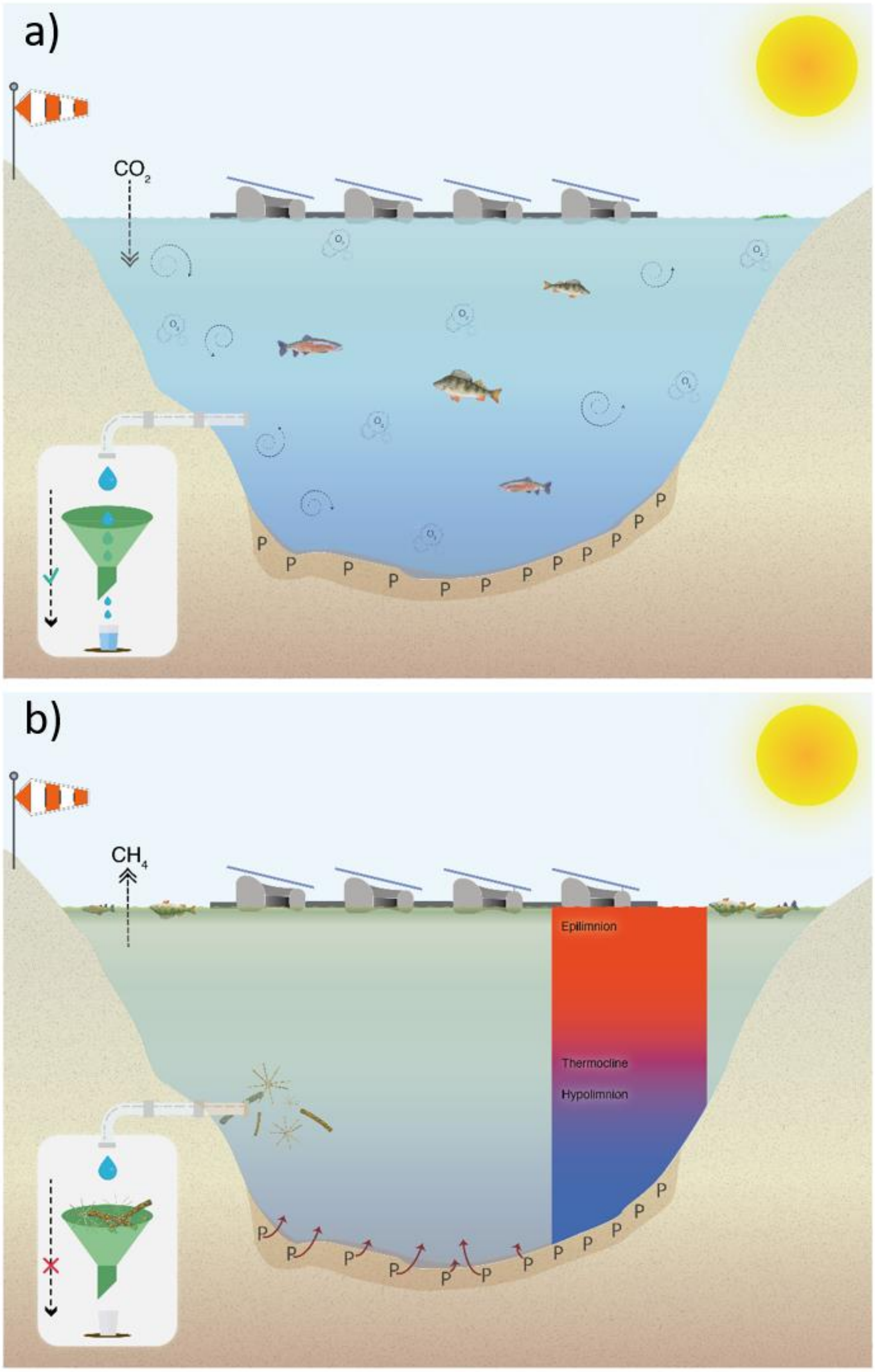


Figure 1-2 – A hypothetical overview of a) a best-case scenario with an FPV array and b) a worst-case scenario with an FPV array.

Water bodies of sufficient depth can become thermally stratified when solar radiation received at the surface creates a temperature-imposed density difference that wind-induced turbulence cannot overcome (Boehrer and Schultze, 2008). During stratification, the water column is separated into three distinct horizontal layers (Wetzel, 2001). The surface layer, or epilimnion, is exposed to the atmosphere and experiences turbulent conditions. The deep-water bottom layer, or hypolimnion, which is in direct contact with the water body bed, is the densest of the three layers and is characterised by non-turbulent conditions. The transition zone between the epilimnion and hypolimnion is the metalimnion, which is characterised by a steep density gradient (thermocline) that restricts the movement of gases and nutrients between the top and bottom layers. Heat loss from the water body can weaken stratification strength until turbulence from wind mixing is sufficient for complete water column mixing (overturn) to occur (Dake and Harleman, 1969).

In extreme cases, if stratification duration was to increase with FPV installations, oxygen depletion may occur in the hypolimnion, given the reduction in wind mixing ability. As redox potential reduces in the bottom sediments, heavy metals and nutrients diffuse into the water column, degrading water quality and facilitating algal growth (Beutel et al., 2008). Further, anoxic conditions in the hypolimnion could lead to fish kills and the subsequent release of methane, a potent greenhouse gas (Tranvik et al., 2009).

Shading from the FPV array may limit the growth of phytoplankton, improving water quality. However, the reduced light concentrations could prompt a switch in phytoplankton species composition. Changes to phytoplankton populations may lead to ecosystem-wide changes at all trophic levels or disrupt anthropogenic activities. For example, a switch to diatom dominance might disrupt water treatment processes due to filter clogging (Joh et al., 2011; Crittenden et al., 2012). The presence of an FPV array will limit other uses of the water's surface, perhaps reducing availability for recreation or causing a reduction in tourism if a water body is considered aesthetically unappealing following a deployment.

1.6 Thesis contributions

The effects of FPV on the host environment has wide-ranging risks and opportunities for industry and society. Advancing knowledge on this novel water surface transformation will inform future deployment designs and decisions, minimising potential negative impacts and maximising potential co-benefits. Therefore, this thesis seeks to understand the interaction between FPV and the host environment. Specifically, a focus on stakeholder perceptions of the strengths and weaknesses of FPV-environment interactions, the compatibility of FPV with ecosystem services and the United Nations Sustainable Development Goals (UN SDGs), the physical and biological response of water bodies to FPV deployments under present-day and future climates and the effectiveness of FPV as a tool for managing water bodies.

All the data chapters in this thesis have been written for publication. **Chapter 2** and **Chapter 3** have appeared as peer-reviewed and published papers in Renewable and Sustainable Energy Reviews and Solar Energy, respectively. **Chapter 4** is in the submission process for the Journal of Environmental Management, and **Chapter 5** is being prepared for submission. As a result, each chapter contains a detailed introduction and motivation. The supplementary information of each data chapter is presented alongside the respective chapter. The content and contribution of the chapters are as follows.

> **Chapter 2** presents scientific evidence from a systematic review and stakeholder expertise, captured through an international survey and a workshop, to assess the interaction of FPV with ecosystem services. Linkages between ecosystem services and the UN SDGs were used to determine the compatibility of FPV with the UN SDGs.
>
> **Chapter 3** developed some of the first understanding on the interaction of FPV and lake thermal structure utilising a one-dimensional process-based model, *MyLak*e. FPV-induced changes in wind speed and solar radiation were modelled to simulate varying FPV coverage and its effect on water temperatures, evaporation rate, ice cover, stratification timing and mixed depth.
>
> **Chapter 4** expands on the work of Chapter 3, detailing upgrades to *MyLake* to permit the simulation of discreet regions of the host water with improved phytoplankton representation. Chapter 4 evaluates the importance of FPV array siting location on the host water body in conjunction with varying FPV coverage.
>
> **Chapter 5** presents the interaction of an FPV array on the host water body under future climates. The chapter develops knowledge on the effectiveness of FPV as a reservoir management intervention and its ability to mitigate the effects of future climates on water body function and process.
>
> **Chapter 6** provides a summary of thesis findings and conclusions.

A chapter consisting of the first in-depth monitoring of an FPV array for environmental impacts was abandoned when access to the field site was prohibited due to the COVID-19 pandemic. Full methods for this monitoring are detailed in Appendix B. A number of months work went into gaining authorisations, planning the experimental design and instrumenting the field site. Data collection took place for approximately seven months (September 2019 to March 2020) before lockdown restrictions commenced. The collected data, albeit a small proportion of the planned monitoring work, were used in Chapter 4 and Chapter 5 as modelling assumptions.

Chapter 2 – Scientific and stakeholder evidence-based assessment: ecosystem response to floating solar photovoltaics and implications for sustainability

Exley, G., Hernandez, R. R., Page, T., Chipps, M., Gambro, S., Hersey, M., Lake, R., Zoannou, K. S. and Armstrong, A. (2021) 'Scientific and stakeholder evidence-based assessment: Ecosystem response to floating solar photovoltaics and implications for sustainability', Renewable and Sustainable Energy Reviews, 152, pp. 111639, doi: https://doi.org/10.1016/j.rser.2021.111639

2.1 Highlights

- Floating solar can be beneficial or detrimental to ecosystem service provision.
- Reduced evaporation is the greatest perceived opportunity of floating solar.
- Detrimental chemical impacts on water quality are the greatest perceived threat.
- Floating solar could interact with eight Sustainable Development Goals.
- Understanding on water body impacts of floating solar needs to be rapidly developed.

2.2 Abstract

Floating solar photovoltaic (FPV) installations are increasing globally. However, their interaction with the hosting water body and implications for ecosystem function is poorly understood. Understanding potential impacts is critical as water bodies provide many ecosystem services on which humans rely and are integral for delivering the United Nations Sustainable Development Goals (SDGs). Here, we used scientific evidence from a systematic review and stakeholder expertise, captured through an international survey and a workshop, alongside existing understanding of the role of water bodies in delivering ecosystem services and the SDGs. We found 22 evidence outcomes that indicated potential physical, chemical and biological impacts of FPV on water bodies. Assessment by stakeholders from across sectors indicated that reduced water evaporation is the greatest opportunity, whilst changes to water chemistry, including nitrification and deoxygenation, are the greatest threat. Despite these findings, FPV operators reported no observed water quality or ecosystem impacts.

However, only 15% of respondents had performed water quality analysis; visual inspection alone cannot ascertain all water quality impacts. Based on the integration of these findings, we determined that FPV could impact nine ecosystem services. Furthermore, established linkages between ecosystem services and SDGs indicate the potential for impacts on eight SDGs, although whether the impact is positive or negative is likely to depend on FPV design and water body type. Our results further the understanding of the effects of FPVs on host water bodies and may help to ensure the anticipated growth in FPVs minimises threats and maximises opportunities, safeguarding overall sustainability.

Keywords: floatovoltaics, floating solar, renewable energy, water quality, natural capital, sustainability, ecosystem impact, knowledge system

2.3 Introduction

In the rush to mitigate the climate crisis, it is critical that new energy developments do not inadvertently hinder, but ideally enhance, other sustainable development goals. The deployment of low carbon, renewable energy technologies is central to achieving the United Nations (UN) Sustainable Development Goal (SDG) adopted by UN member states in 2015 to ensure *access to affordable, reliable, sustainable and modern energy for all* (SDG7) (United Nations, 2015; IEA et al., 2019). Consequently, the increasing demand for low carbon energy has led to the rapid deployment of solar energy infrastructure across the world (IEA, 2019), with technologies including solar photovoltaics (PV), concentrating solar power and solar thermal. PV technology dominates current solar energy infrastructure (Hernández-Callejo et al., 2019) due to its viability across climates (Pogson et al., 2013) and scalability, permitting deployments ranging in capacity from residential- to utility-scale (Hernandez et al., 2014). In comparison to other electricity generation methods, solar PV has a low energy density (~0.25 MW acre^{-1} (Kabir et al., 2018)) and thus exerts a considerable land-use pressure (Sahu et al., 2016; Cazzaniga et al., 2018; Cagle et al., 2020), potentially impacting other SDGs, such as *life on land*, and the provision of ecosystem services on which society relies (Grizzetti et al., 2019). However, the flexible nature of PV has enabled innovative deployments that could be harnessed to incorporate co-benefits for other SDGs and the provision of ecosystem services (Randle-Boggis et al., 2020). For example, efforts to overcome land-use conflict between solar PV and agricultural production (SDG2 – *zero hunger*) led to the first commercial floating photovoltaic (FPV) solar energy installation in 2007 (World Bank Group et al., 2018; Cagle et al., 2020).

The rapid deployment of FPV coupled with the variations in FPV design provide opportunities for positive and negative impacts on the SDGs. FPVs are emerging worldwide as an alternative means of deploying PV (World Bank Group et al., 2018). To date, installed capacity has grown exponentially and is expected to continue (Cazzaniga and Rosa-Clot, 2021), with estimates suggesting a minimum global potential of 400 GW-peak deployment (World Bank Group et al., 2018). Growth has been particularly strong in India and China, accounting for six of the ten largest FPV projects (Power Technology, 2021) (see Solarplaza (2019) for details on global deployment locations). FPV arrays typically consist of five components: the floating support structure, a mooring and anchoring system, inverters, transmission cables and the PV modules (Oliveira-Pinto and Stokkermans, 2020). Although FPVs vary considerably in their design, with manufacturers offering both bespoke and off-the-shelf systems, the majority employ an inter-locking floating pontoon comprised of high-density polyethylene, each supporting a fixed-tilt angle PV module (Oliveira-Pinto and Stokkermans, 2020). Designs can be categorised by their surface coverage density, defined as the proportion of the installation in contact with the water body. In ascending order, 'Freestanding' designs (i.e. those mounted on poles) have the lowest coverage density, followed by 'small footprint' (i.e. where multiple PV panels are mounted on frames supported by floats), 'large footprint' (i.e. where a single PV panel is mounted on an individual float) and 'insulated' (i.e. those where PV panels are mounted on a continuous cover or membrane) designs (see Liu et al. (2018) for full descriptions on FPV design and structure types and Figure S 2-1). FPV systems rarely cover the whole water surface, and most are deployed at a distance from the edge of the water to prevent access or damage by theft and vandalism. Further, this permits variations in water level due to drought or maintenance, with some designs flexible enough to enable the installation to rest and operate on the water body bed if necessary (WWT, 2016). In terms of water body selection, some locations enable a direct supply of power (e.g. to a water treatment works), while others export the power to a centralised electricity network.

FPVs offer several co-benefits, but there are also risks of unintended detrimental impacts, especially for water body function. One notable advantage of FPVs over building- and ground-mounted systems is the potential for greater PV panel efficiencies in response to the water body cooling effect (Choi et al., 2013; Sacramento et al., 2015; Yadav et al., 2016; Oliveira-Pinto and Stokkermans, 2020). They also spare land; regions with land-use conflicts have seen the greatest growth in FPV deployment (Hoffacker et al., 2017). Further, several schemes have been co-located with hydroelectric power generation. FPV-hydro systems take advantage of existing grid connections and infrastructure and improve the power output profile (Silverio et al., 2018; Haas et al., 2020). However, while the economic and technical feasibilities of FPVs are well established, indicating contributions to SDG7, scientific understanding of the water body opportunities and threats of FPVs is very limited. FPVs could benefit or disrupt water body function with implications for ecosystem services, natural capital and SDGs, including the provision of drinking water (SDG6 – *clean water and sanitation*) and carbon stores (SDG13 – *climate action*).

Given the dearth of understanding and the current rate of FPV deployment, there is an urgent need to accelerate understanding rapidly. The water body effects of FPVs will be primarily driven by their physical presence altering wind and solar radiation receipts, two fundamental regulators of water body behaviour, with implications for surface meteorology, air-water fluxes and consequently water body physical, chemical and biological processes and properties (Armstrong et al., 2020). Accordingly, the impacts of FPVs will vary with design, in particular the nature and extent of water surface use, with the response modulated by water body characteristics such as location, morphology and nutrient status (Armstrong et al., 2020). Potential impacts can be determined by utilising emerging knowledge from FPV systems and inferring likely impacts from the established scientific understanding of natural water body covers, such as plants and ice, and artificial covers, such as evaporation suppression systems.

Given the multiple uses of water bodies, including FPV, it is critically important to capture the perspectives and expertise of stakeholders when resolving the potential implications of FPV on water body function (Menzel and Teng, 2010; Lamarque et al., 2011; García-Nieto et al., 2015). As water bodies provide a large range of ecosystem services, the perspectives of a broad range of stakeholder groups and organisations (e.g. water body managers, recreational users, developers, environmentalists and local and national authorities) are required to develop a comprehensive FPV 'knowledge system'. Specifically, knowledge systems collate the expertise of actors (e.g. stakeholders who mobilise knowledge), organisations (e.g. intermediaries between actors), and objects (e.g. data or models) that perform knowledge-related functions (Cash et al., 2003; McCullough and Matson, 2016). Several studies have shown that the coordination and identification of priorities across knowledge systems have contributed towards the transition to low carbon energy (Tawney and Weischer, 2011; Cornell, 2013; Clar and Sautter, 2014). Tapping into the FPV knowledge system helps bridge the knowledge gaps in this upcoming area of research.

The rapid deployment rate of FPV has outpaced understanding of the potential impacts on the host water body. Consequently, developing an understanding of the ecosystem impacts of FPVs is critical to ensure sustainable deployments that avoid concomitant detrimental impacts and maximise co-benefits. Therefore, the overarching aim of this paper is to determine the potential impacts of FPVs on the host ecosystem, the ecosystem services they provide and the potential benefits and trade-offs with other SDGs. To achieve this, we (1) synthesise current evidence on the water body impacts of FPVs; (2) establish ecosystem service opportunities and threats presented by FPVs; and (3) discuss the overall sustainability of FPVs using a generalised framework by linking FPV impacts with SDGs. Finally, we prioritise further research needs and innovation to ensure the design and deployment of future FPVs promote co-benefits across the suite of SDGs, contributing to a sustainable low-carbon energy transition.

2.4 Methods

In order to address objectives one (evidence synthesis) and two (ecosystem opportunities and threats), we conducted an evidence review of the scientific literature, an international stakeholder survey and a stakeholder workshop (Figure 2-1). Finally, outcomes from these were synthesised to address objective three (discuss the overall sustainability of FPVs).

2.4.1 Evidence review

The review of the scientific literature was conducted using the Defra Quick Scoping Review method, a methodology designed to assess the volume and characteristics of an evidence base prior to evidence synthesis (Collins et al., 2015). The scope of the evidence search was constrained by the question; 'What are the potential impacts of FPV on water body function?' Search strings were formulated using the Population, Intervention, Comparison and Outcome (PICO) framework (see supplementary information for full details; section 2.9.2) and were developed by the authors and a steering group comprised of stakeholders from four United Kingdom (UK) water utility companies. The search was limited to studies published in English, while no restriction was imposed based on publication date. All literature returned was subject to pre-defined inclusion and exclusion criteria. Specifically, all literature needed geographical and climatic relevance to temperate regions and to contain evidence of an effect of water body coverage.

Returned articles underwent an initial title screen, followed by an abstract screen. If relevant or inconclusive, the whole article was read (see Figure S 2-2 for an overview of the review process). Evidence (defined here as information and preferably numerical data) suggesting that surface covers impact water body function, was then extracted from each of the articles which passed the screening process. Each article was summarised and categorised by surface cover type: 'Ice', 'Plant' or 'Artificial'. An evidence outcome was allocated to indicate if the effect on water body function was 'negative', 'neutral' or 'positive' (see supplementary information for further details; section 2.9.2). Articles that speculated or hypothesised an effect were excluded from the review. Evidence strength was also assessed to indicate confidence. For example, if the articles were based on simulations of minor relevance to the temperate climatic region or if there were concerns regarding study design and applicability to FPV, the evidence was classified as weak. The remaining studies, which met the search criteria, were graded as strong.

2.4.2 International stakeholder survey

To gather contemporary understanding, which is especially important given the relative immaturity of FPVs and thus the limited studies in the scientific literature, we deployed an online international stakeholder survey. The survey targeted the knowledge system of FPV operators, actors with first-hand experience of FPV system functionality and potential water body impacts. Questions focussed on four categories; FPV characteristics (such as array size and type), water body characteristics (such as depth, surface area and use), sampling and data collection, and FPV array management (such as bird deterrents and cleaning). The full list of questions and further methods, including ethical procedures, can be found in the supplementary information (section 2.9.3).

2.4.3 Stakeholder Workshop

To gather further expert insight on FPVs, specifically on hosting water body types and the relevance and implications of the evidence review findings, we held a free to attend one-day *Floating solar: water quality impacts* workshop in London, UK, in November 2019. The workshop was attended by 27 stakeholders from different interest groups, specifically 11 participants from the water industry, six FPV developers, three from trade associations, four attendees from community-interest parties and three researchers (A.A., G.E. and T.P.). Attendees were predominantly UK-based, although global input was contributed by attendees based in Brazil, France and Norway.

2.4.3.1 Identification of different potential hosting water body types

Workshop attendees were asked to identify as many water body types as possible, including both natural and human-made systems that could conceivably host an FPV array. The ecosystem services provided by each water body type that could be affected by FPV deployment were qualitatively identified post-workshop using a conceptual framework for the integrated assessment of water-related services (Grizzetti et al., 2016), with the list of freshwater provisioning, regulating, cultural and supporting ecosystem services compiled using a selection of established typologies (Costanza et al., 1997; de Groot et al., 2002; Chopra et al., 2005; Kumar, 2010; Maltby et al., 2011; Grizzetti et al., 2016; Wood et al., 2018).

2.4.3.2 Evidence review relevance and implications

To determine relevance and the implications of the evidence review (section 2.4.1), attendees at the *Floating solar: water quality impacts* workshop assessed the findings. Divided into five groups, each comprising a mix of people from different interest groups, workshop attendees were asked to identify if each piece of evidence represented an opportunity or a threat to water quality (i.e. water body physical processes, chemistry, and biology). The opportunity and threat categories were partitioned into 'low', 'medium', 'high' and 'neutral' options, allowing attendees to choose both the direction and magnitude of the potential effect. The responses were pooled to create a stakeholder score to inform areas of greatest knowledge need, allocating positive or negative outcomes to each piece of presented evidence. Scores could range from -15, indicating stakeholders consider the evidence a 'high' level threat, to +15, indicating attendees consider the evidence to present a 'high' level of opportunity.

2.4.4 Overall sustainability of FPV

To address objective three (to contextualise the overall sustainability of FPVs using a generalised framework), evidence gathered during the evidence review, stakeholder survey and stakeholder workshop was combined with established knowledge in the scientific literature. First, we inferred FPV impacts on ecosystem services by identifying relationships between our gathered evidence and our typology of freshwater ecosystem services (section 2.4.3.1). For example, evidence that FPV reduces evaporation could be linked to the freshwater ecosystem service *provisioning of water for consumption*. This was original work and semi-qualitative in that it is based on evidence from stakeholders and scientific knowledge.

We subsequently identified linkages between the potentially impacted ecosystem services and the SDGs using the typology established in Wood et al. (2018) and the dependencies across SDGs in Le Blanc (2015). Specifically, Wood et al. (2018) selected 16 ecosystem services and used expert judgement to identify the magnitude of contributions of ecosystem services to specific SDGs and their targets. For example, Wood et al. (2018) found a strong level of support for a contribution by the ecosystem service *water provision* to all Targets of SDG11 Sustainable Cities, except Target 11.7 (access to green spaces), where only a weak level of support exists between water provision and Sustainable Cities. In this study, we matched the freshwater ecosystem services we identified to be impacted by FPV to the terms used to describe ecosystem services in Wood et al. (2018). We then linked SDGs to individual SDG targets in Le Blanc

(2015), allowing us to build a generalised framework of FPV sustainability. Links defined as weak by Wood et al. (2018) were not included.

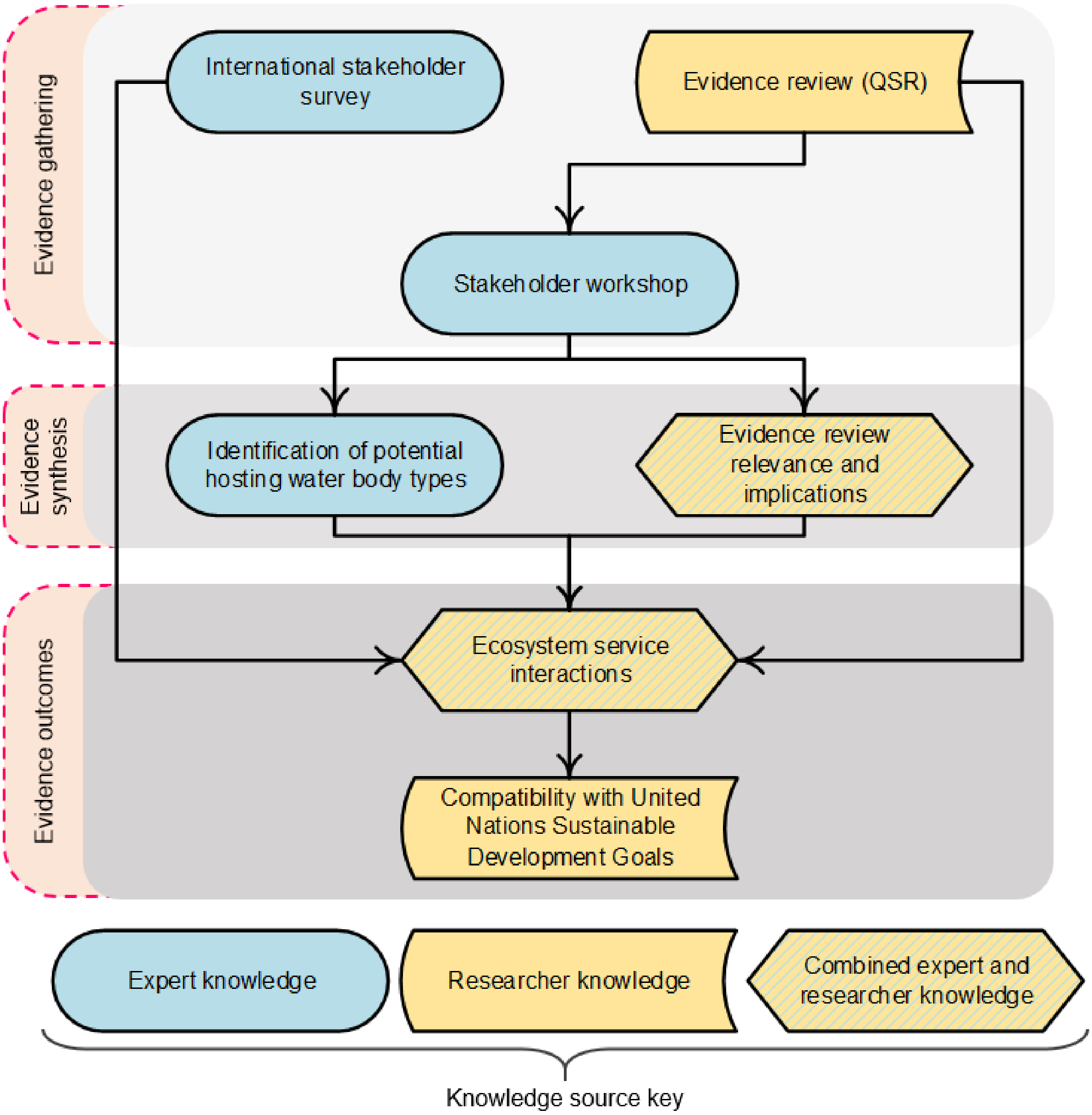


Figure 2-1 – Schematic of knowledge system components and the integration of connecting research activities.

2.5 Results & discussion

Below we provide a synthesis of the impacts of FPVs on water bodies, informed by scientific evidence and actors within the FPV knowledge system (objective 1). Subsequently, we determine the ecosystem service opportunities and threats presented by FPVs (objective 2) and discuss their overall sustainability (objective 3).

2.5.1 Synthesis of FPV impacts on water bodies evidence

Given the relative immaturity of FPV installations, there has been limited scientific study of their interactions with water bodies. Consequently, in the following sections, we share the outcomes of the scientific evidence review and the insight gained from the stakeholder survey. Finally, we discuss the potential beneficial and detrimental implications of FPVs on water body function and ecosystem services. Overall, there is limited evidence; thus, outcomes are indicative, and future research is urgently required.

2.5.1.1 Scientific evidence review

The evidence review of the scientific literature detailing the water body impacts of FPV covers, along with analogue natural and artificial covers as proxies, identified potential impacts on water body physical, chemical and biological behaviour. Over 7000 peer-reviewed scientific articles were initially identified. After evidence screening, 51 articles that detailed the impact of surface covers in temperate environments remained. In total, 29 (one categorised as weak) and 15 (one categorised as weak) pieces of evidence suggested that surface covers had positive and negative outcomes on water quality, respectively (see supplementary information; section 2.9.2; Figure S 2-2). Out of these 51 articles, 45 articles described natural surface covers; 37 articles were studies of ice as a surface cover, and eight were studies of plants – the remaining six evaluated artificial surface covers, including FPVs, shade cloths and floating evaporation suppression devices. Although 14 articles on FPVs met the initial criteria to be read in full, 13 were subsequently rejected as they did not adequately consider, based on the protocol of this review, the effects of FPV coverage on water quality. Instead, these articles typically focussed on the technical or financial aspects of FPVs, often stating the effects on water quality are largely unknown and/or hypothesising impacts. The evidence, across surface cover types, were dominated by articles assessing biological impacts ($n = 27$), with equal numbers of articles ($n = 12$) for physical and chemical properties and processes (Table 2-1, Figure S 2-2).

The impacts of surface covers are summarised below (the evidence is provided in the supplementary information; section 2.9.2). The appropriateness of each analogue cover as a proxy for FPV must be considered when inferring potential impacts of FPV. For example, in the instance of ice cover, the proxy with the most retained articles (*n = 37*), surface cover is likely to be spatially continuous, completely insulating the water body from the air during the winter months. However, FPVs do not extend fully across water surfaces. Moreover, the continuous nature of ice is a better representation of insulating FPV designs, rather than Freestanding and footprint designs (Figure S 2-1). 'Small' and 'large' footprint designs (Figure S 2-1) are better represented, particularly by plant cover (*n = 8*), where coverage may be spatially discontinuous across the water body and a lower density than ice. Only artificial covers (*n = 6*), such as shade cloths and floating evaporation suppression devices, provide a temporally representative proxy for FPVs, with coverage continually present throughout the year. Given these differences between the analogues and FPV, the potential beneficial and detrimental effects may differ from the evidence synthesised below.

2.5.1.1.1 *The effect of surface covers on physical process & properties*

The evidence, across all surface cover types, detailed impacts on physical processes and properties, namely solar radiation receipts, water body temperatures, evaporation and mixing dynamics with implications for sediment suspension (Table 2-1). Surface covers promoted reductions in water temperature (*n = 3*). Whilst the evidence is limited, this trend is likely to pervade across FPV designs as they act as a physical barrier (Pinto et al., 2007; Balayla et al., 2010), attenuating solar radiation and reducing the heating of surface waters, lowering water temperature (Austin and Allen, 2011; Ozkundakci et al., 2016; Haldna and Haberman, 2017). We found that artificial covers tended to reduce the solar radiation reaching water bodies more than natural covers, due to their more extensive nature (typically deployed to cover the full water body surface) and lower transparency (e.g. a suspended shade cloth cover reduced light transmission by 99% (Maestre-Valero et al., 2011) while ice cover reduced transmission by 53 – 82% (Lenard and Wojciechowska, 2013)). For FPVs, the scale of impact will be highly dependent on FPV design. The surface cover colour, specifically black versus white, did not affect surface water temperatures even though black covers reached almost twice the temperature of white covers (Lehmann et al., 2019). Instead, the cover's thermal properties control the transfer of absorbed thermal radiation to the water body

(Lehmann et al., 2019). Consequently, FPV design, including float construction material, should be considered when evaluating potential water body effects.

The water temperature impacts of FPVs will vary with incoming solar radiation, which can fluctuate dynamically across diel and seasonal scales depending on the location. For example, in Taiwan, a country with a tropical climate, temperature effects were quantified for a 'large footprint' FPV array covering 40% of an irrigation pond's surface; it reduced winter water temperatures by 0.77 °C, and summer water temperatures by 1.4 °C (Chateau et al., 2019). Additionally, given that water bodies act as thermal stores, the reduction of solar radiation by FPVs will alter seasonal temperature dynamics. For example, with every 1% increase in winter-averaged ice cover on Lake Superior (MN, USA), average summer (July-September) surface water temperature decreased approximately 0.1 °C due to the impacts of ice thickness on solar radiation receipts and thereby temperature (Austin and Colman, 2007).

Water body surface covers change the thermal dynamics at the air-water interface (Oke, 2002), with significant impacts on evaporation (*n = 2*, Table 2-1). The multiple methods for estimating evaporative losses from water bodies with surface covers can present a challenge when comparing evidence qualitatively (Assouline et al., 2011). Experiments using palm fronds in an arid region suggest that the total area covered by a FPV may be approximately proportional to evaporative losses: palm fronds reduced evaporation by 55% when covering the full surface of a pool, and 26% when covering half (AlHassoun et al., 2011). However, given the importance of wind in determining evaporation rates, the proportional relationship may not hold, especially for larger water bodies (Finch and Hall, 2001; Wüest and Lorke, 2003). Furthermore, FPV design (i.e. the change in roughness and impact on water-air connectivity), may also be an important factor in determining evaporative losses. For example, an evaporation suppression experiment in a laboratory setting found covering 91% of a tank's surface with free-floating spheres and free-floating disks reduced evaporation by 70% and 80%, respectively (Lehmann et al., 2019). Given the large variation in FPV design, a better understanding is required to resolve the impacts of FPVs on evaporation.

Mixing dynamics are an important determinant of water quality, influencing sediment and water chemistry (Kalff, 2002); thus, the implications of FPVs on water body mixing must be resolved. Two studies found that surface covers reduced sedimentation and sediment resuspension, suggesting reductions in vertical mixing ($n = 2$, Table 2-1). For example, ice surface covers lowered gross sedimentation by over 20 times compared to the ice-free period (Niemisto and Horppila, 2007). Devoid of wind stress beneath the ice, resuspension rates fell to 50 to 78% of gross sedimentation, compared to a resuspension rate of 87 to 97% of gross sedimentation for an uncovered water body (Niemisto and Horppila, 2007). In contrast to vertical mixing, horizontal mixing has been observed under ice (Bengtsson, 1996; Kenney, 1996; Salmi et al., 2014b; Pernica et al., 2017) and plant covers (Coates and Ferris, 1994). Consequently, resolving how FPVs alter mixing will be critical to understanding water quality impacts.

2.5.1.1.2 *The effect of surface covers on chemistry*

FPVs could impact several water chemistry properties and processes, including nutrient concentrations and gas exchange, with potential positive and negative consequences (Table 2-1). Reductions in nutrient and contaminant concentrations could occur in response to the reduced evaporation (AlHassoun et al., 2011; Lehmann et al., 2019) caused by FPVs (Taboada et al., 2017). For example, surface covers have reduced the salinity of water bodies due to lower evaporative losses, with one example identifying an 8.2% reduction in soluble salt concentration (Maestre-Valero et al., 2011). Further, water nutrient and contaminant concentrations could be altered given the effect of surface covers on sedimentation and sediment resuspension (Niemisto and Horppila, 2007). For instance, water bodies with less extensive FPV covers, or comprised of lower footprint designs, are more likely to experience higher total phosphorus concentrations as the entrainment of suspended particulate matter can continue for a longer period or over a greater area of a water body's bed (Kleeberg et al., 2013). The responses are also likely to vary with water depth, with the effects of FPVs on sedimentation and sediment resuspension greater in shallower lakes (Bloesch, 1982; Evans, 1994). For example, reduced vertical mixing in response to ice cover was associated with a reduction of phosphorus at the sediment-water interface (Kleeberg et al., 2013).

However, FPVs may also negatively impact water chemistry (*n = 3*, Table 2-1). Surface covers, particularly ice cover (a proxy for 'insulated' FPV designs) due to its spatially continuous nature (Wetzel, 2001), isolates the water from the atmosphere, causing dissolved oxygen depletion (Balayla et al., 2010). A lack of dissolved oxygen can have multiple implications for water quality, including the release of nutrients and contaminants from bed sediments (Hupfer and Lewandowski, 2008). Oxygen depletion increases over time as aerobic processes continually draw on the limited oxygen supply. Eventually, if insufficient oxygen enters the system, the water body becomes anoxic (Bai et al., 2016). The rate of oxygen depletion will depend on FPV design and highly water body-specific, depending on the rate of biological processes, stratification (Bouffard et al., 2013), forced aeration (such as reservoir agitation with mechanical mixers or bubblers), residence time (Nurnberg, 2004) and degree of wind mixing (Lepparanta et al., 2012). For example, under ice cover, nitrification and the activation of anaerobic processes placed the greatest demands on dissolved oxygen supply, although fish contributed minimally to winter oxygen depletion (*n = 3*, Table 2-1). The rates of oxygen depletion also vary seasonally. For instance, sediment-water heating facilitates enhanced microbial respiration during winter (Ellis and Stefan, 1989), speeding up the development of anoxic conditions (Golosov et al., 2007; Chen et al., 2019).

Oxygen depletion activates anaerobic processes that cause detrimental impacts; for example, ice cover on Russian, American and Canadian lakes caused a release of deoxidised gases such as methane, hydrogen sulphide, and ammonia (Golosov et al., 2007). As water bodies help regulate the climate, the release of methane, a potent greenhouse gas, is concerning as it could increase the carbon intensity of the electricity produced by FPVs. Moreover, oxygen-depleted bottom-waters can become enriched in reactive species of manganese, iron and phosphorus, with the concentrations increasing higher in the water column during prolonged periods of cover (Joung et al., 2017). The release of metals such as manganese and iron is detrimental to water quality and constituent aquatic ecology, whereas increased phosphorus concentrations may facilitate phytoplankton growth, including problem blue-green algae, in phosphorus-limited water bodies (Welch and Cooke, 2005).

Processes that consume oxygen can also impact water quality directly. For example, 1 to 25% of the dissolved oxygen depletion rate in seven temperate seasonally frozen lakes in Wisconsin, USA, was attributed to nitrifiers (Powers et al., 2017a). As well as consuming oxygen, nitrification leads to the accumulation of nitrate, which can be used by phytoplankton once the growing season commences (Powers et al., 2017b), potentially leading to problematic blooms (see section 2.5.1.1.3). However, if dissolved oxygen is fully depleted, anaerobic conditions cause denitrification, the process that reduces nitrate to gaseous nitrogen, reducing eutrophication (Kalff, 2002). The likelihood and rate at which denitrification occurs under FPVs will also be linked to the temperature impacts as the rate at which heterotrophic denitrifying bacteria convert nitrate to nitrogen is controlled hierarchically, first by nitrate concentrations, then by temperature (Cavaliere and Baulch, 2018). Consequently, depleted oxygen could lead to phytoplankton blooms, or, if anoxic conditions occur, lower water temperatures and depleted dissolved oxygen associated with FPVs may induce denitrification, potentially improving water quality by reducing eutrophication and phytoplankton recruitment.

Consequently, it is critical to understand the impacts of FPVs in light of the water body and design characteristics (including adaptive strategies for mitigating potential adverse effects; see section 2.7) when resolving potential water quality impacts. Furthermore, water body use informs the significance of the perturbations or enhancements. For example, an FPV installation could cause enhanced denitrification rates and improve water quality, while enhanced internal loading of phosphorus from anoxic bed sediments may promote phytoplankton growth, degrading water quality. Ensuing changes to water quality could require modified chemical water treatment to maintain drinking water quality, either reducing or increasing cost.

2.5.1.1.3 *The effect of surface covers on biology*

The evidence review identified biological effects of surface covers on three trophic levels; phytoplankton, zooplankton and fish (Table 2-1). Resolution of the impacts of FPVs on phytoplankton response is pivotal as they are the food source for all higher trophic levels and some exert considerable influence over water quality (e.g. Henderson et al., 2008; Rolland et al., 2013). All types of surface cover lowered phytoplankton density, biomass and chlorophyll-*a* concentrations, attributable to lowered solar radiation curtailing photosynthesis (Agbeti and Smol, 1995; Seto et al., 2013; Ji et al., 2016) and potentially reduced vertical mixing limiting the release of phosphorus at the water-sediment interface (Kleeberg et al., 2013). The magnitude of impacts varied with water body type, surface cover and coverage extent, but were generally significant. For example, ice cover on a small lake in Poland reduced phytoplankton biomass by 51% (Lenard and Wojciechowska, 2013). Plant cover also reduced phytoplankton biomass, with an 88% reduction observed in an Argentinian mesocosm experiment (Pinto et al., 2007). Further, experiments using a dye that reduced light intensity to 1% of surface light in the photic zone reduced phytoplankton biomass by 60% (Batt et al., 2015).

As well as impacting overall biomass, light suppression caused by FPVs could cause shifts in the timing and occurrence of phytoplankton blooms. Lower phytoplankton growth, and therefore nutrient uptake, will allow the persistence of nutrients in the water column (Agbeti and Smol, 1995; Salmi et al., 2014b; Salmi and Salonen, 2016), increasing the chance of phytoplankton blooms later in the growing season. However, the complexity of phytoplankton and nutrient dynamics curtails the potential to offer universal predictions of timings and abundance (Page et al., 2018). However, overall, reductions in phytoplankton growth are likely to lead to enhanced water quality with improvements for recreational use and potentially reduced water treatment costs.

Reductions in phytoplankton biomass and shifts in the timing and occurrence of phytoplankton blooms are also likely to be accompanied by changes in phytoplankton species composition, given the different physical and chemical conditions imposed by FPVs (Wright, 1964; Danilov and Ekelund, 2001; Lenard and Wojciechowska, 2013). For example, filamentous diatoms that are adapted to darker conditions may increase due to improvements in water clarity and reduced sediment resuspension in very sediment-rich waters (Twiss et al., 2012; Beall et al., 2016). The characteristics, or functional traits, of phytoplankton determine if they will increase or decrease under FPVs, for example, the motility, nutritional mode, ability to form resting stages, organisation, cell shape, and size class were found to be significant predictors of phytoplankton species under ice (Ozkundakci et al., 2016). Generally, reduced light availability and the associated cooler water temperatures under surface covers eliminates large and drifting types of phytoplankton, favouring smaller motile forms capable of mechanical movement

(Campbell and Haase, 1981). Drifting types are also impacted by the lower vertical mixing rates under surface covers, with populations decreasing if the water movement is less than the species' sinking rate (Matthews and Heaney, 1987).

Competition with other species will influence the abundance of each phytoplankton species. For example, as FPVs shift water column irradiance from high-intensity to low-intensity, there is the potential for blue-green algae populations with low critical light intensity to increase, utilising their low light tolerance and the reduced turbulence under FPVs to outcompete other phytoplankton species (Pinto et al., 2007). For example, plant cover resulted in blue-green algae dominating the overall species composition when 50-75% of the water's surface was covered but was less abundant when the surface cover was lower (Stiers and Triest, 2017). Resolving the impacts of FPVs on blue-green algae will be of key importance for water body managers given increased bloom prevalence with climate change (Paerl and Huisman, 2008), implications for recreational activities and aesthetic values (Brooks et al., 2016), and the need for enhanced raw water processing and treatment due to the production of muddy odour metabolites geosmin and 2-methylisoborneol (resulting in taste and odour issues) if FPVs are deployed on reservoirs used for drinking water (Young et al., 1996; Watson et al., 2016).

Although solar radiation and nutrient concentrations are the primary drivers of phytoplankton response, resolving the oxygen and temperature impacts is critical for understanding impacts at higher trophic levels. In ice-covered lakes, oxygen depletion is the most important factor determining the onset of fish mortality (Ellis and Stefan, 1989; Balayla et al., 2010), while non-covered lakes may see fish die-offs during extreme summer conditions which cause a temperature-oxygen squeeze (Till et al., 2019). However, the impact of FPVs on oxygen content is poorly resolved, with both increased and decreased risk of anoxia possible (see section 2.5.1.1.2; (Armstrong et al., 2020)). Temperature, as a regulator of metabolic rate, has significant impacts on higher trophic levels. For example, one study found a 9% decrease in zooplankton abundance with a 1 °C decrease in water temperature in autumn, while in spring, a 1 °C rise in water temperature increased zooplankton abundance by 27% (Haldna and Haberman, 2017). In addition to temperature regulation of metabolic rates, temperature thresholds exist that cause step changes in biological processes. For example, a shift from cold water to warm water zooplankton species occurred at a critical threshold of 10 °C in the spring, with a less conspicuous change occurring in the autumn (Haldna and Haberman, 2017). Comparison of FPV induced temperature changes to those caused by other water body covers suggest that the impacts are likely to be less extreme. However, FPV studies are very limited (Chateau et al., 2019; Haas et al., 2020).

In addition to the direct impacts of FPVs on species, the indirect effects through altered predator-prey relationships are critical to determining the overall impacts on water body biology. Surface covers, such as emergent and floating-leaved macrophytes, may enhance the survival of zooplankton by providing a refuge from predation. For example, the overall density of cladocerans (‘water fleas’) was, on average, over 60 times greater in the presence of plants than in open water (Cazzanelli et al., 2008). Further, zooplankton have been observed to increase their horizontal and vertical migration under surface covers as shading from plants offers a mechanism to avoid predation from fish (Horppila and Nurminen, 2008). However, such impacts do not always occur; evidence from a different study found no significant difference in zooplankton abundance or diversity along a horizontal gradient from the macrophyte-covered littoral to the open pelagic zone of temperate lakes (Spoljar et al., 2018). Fish may also change their behaviour, including by reducing their predator vigilance in the presence of FPVs. For instance, brown trout increased their swimming activity under ice cover, swimming 38% of the time, compared to 21% in the absence of cover (Watz et al., 2015).

A further indirect impact on lower trophic levels is the consequences of fish kills due to anoxia. For example, in a study of 13 European lakes with winter ice cover, summer zooplankton communities were comprised of a significantly greater proportion of larger-bodied taxa, as smaller planktivorous fish populations reduced the predation pressure on zooplankton (Gyllstrom et al., 2005). In turn, these larger-bodied and more abundant zooplankton had stronger grazing impacts on phytoplankton, having a positive cascading effect on water quality (Ellis and Stefan, 1989; Gyllstrom et al., 2005). Such changes in species composition between trophic levels can impact overall ecosystem resilience and may have consequential impacts on the provision of food for human consumption for some water bodies.

Table 2-1 – Summarised outcomes of potential floating photovoltaic solar energy installations ('FPVs') effects on physical, chemical, and biological aspects of water quality from the scientific evidence review. Evidence outcome indicates if the article author('s) identified the outcome as a negative (-), neutral (0) or positive (+) effect on water quality. The cover category refers to the type of natural (i.e. ice, plants) or artificial (i.e. other) surface cover studied. Stakeholder Workshop attendees were asked to identify if each outcome is either an opportunity or a threat to water quality. Opportunities and threats were further prioritised by stakeholders as 'low', 'medium', 'high' or they could choose 'indifferent'. A final stakeholder score was calculated for each evidenced effect. Low negative numbers indicate that the stakeholders considered the evidence as a 'high' level threat (i.e. −15). In contrast, a high positive number (i.e. 15) indicates the attendees considered the evidence to present a 'high' level of opportunity.

a, (AlHassoun et al., 2011; Lehmann et al., 2019); b, (Austin and Colman, 2007; Austin and Allen, 2011; Chateau et al., 2019); c, (Niemisto and Horppila, 2007; Kleeberg et al., 2013); d, (Coates and Ferris, 1994; Bengtsson, 1996; Kenney, 1996; Salmi et al., 2014b; Pernica et al., 2017); e, (Ellis and Stefan, 1989; Lepparanta et al., 2012; Bai et al., 2016); f, (Powers et al., 2017a; Powers et al., 2017b; Cavaliere and Baulch, 2018); g, (Golosov et al., 2007); h, (Golosov et al., 2007; Chen et al., 2019); i, (Golosov et al., 2007); j, (Joung et al., 2017); k, (Maestre-Valero et al., 2011); l, (Agbeti and Smol, 1995; Seto et al., 2013; Batt et al., 2015; Ji et al., 2016); m, (Agbeti and Smol, 1995; Salmi et al., 2014a; Salmi and Salonen, 2016); n, (Wright, 1964; Campbell and Haase, 1981; Matthews and Heaney, 1987; Danilov and Ekelund, 2001; Lenard and Wojciechowska, 2013; Kalinowska and Grabowska, 2016; Ozkundakci et al., 2016); o, (Oveisy et al., 2014; Weirich et al., 2019); p, (Pinto et al., 2007; Stiers and Triest, 2017); q, (Twiss et al., 2012; Beall et al., 2016); r, (Ellis and Stefan, 1989); s, (Gyllstrom et al., 2005; Cazzanelli et al., 2008; Balayla et al., 2010; Haldna and Haberman, 2017; Spoljar et al., 2018); t, (Watz et al., 2015); u, (Horppila and Nurminen, 2008); v, (Maestre-Valero et al., 2013).

Evidence Review									Stakeholder Workshop								
	Evidence from Quick Scoping Review	Evidence Outcome			Total Papers	Cover Category			Threat			Indifferent	Opportunity			Stakeholder Score	Reference
		-	0	+		Ice	Plants	Other	High	Medium	Low		Low	Medium	High		
Physical	Reduced evaporation			2	2			2								**9**	a
	Reduced water temperatures			3	3	2		1								**8**	b
	Reduced sedimentation and reduced sediment resuspension			2	2	2						1				**3**	c
	Horizontal mixing continues		1	4	5	4	1					2				**7**	d
Chemical	Anoxia (as water body isolated from the atmosphere)	2	1		3	3										**-14**	e
	Nitrification continues (substantial oxygen demand)	2	1		3	3										**-15**	f
	Release of: • Methane (CH_4)	1			1	1										**-14**	g
	• Hydrogen sulfide (H_2S)	2			2	2										**-13**	h
	• Ammonia (NH_3)				1	1										**-14**	i
	• Heavy metals from bed sediments	1			1	1										**-14**	j
	Reduced salinity			1	1			1				3				**1**	k
Biological	Reduced algae growth			4	4	1	2	1								**7.5**	l
	Delayed algal biomass peaks	1		2	3	3						1				**-6**	m
	Modified algal community composition	2	2	3	7	7										**-9**	n
	Prolonged cover led to large algal blooms	2			2	2						3				**-4**	o
	Blue-green algae success is improved as competition from other species reduced due to lower light levels	1	1		2		2									**-12**	p
	Reduced mixing and turbidity allowed extensive growth of filamentous diatoms	2			2	2						1				**-7.5**	q
	Fish kills	1			1	1										**-4.5**	r
	Increased zooplankton numbers and enhanced survival		1	4	5	3	2									**4**	s
	Fish reduced their predator vigilance (birds)			1	1	1										**3**	t
	Shading (i.e. darkness) reduced the feeding effectiveness of fish			1	1		1									**-3**	u
	Reduced concentrations of faecal coliforms E. coli			1	1			1				1				**6.5**	v

2.5.1.2 Stakeholder insight

Stakeholder expertise is crucial to capture the potential impacts of FPVs and contextualise findings from the scientific literature. Our stakeholder survey captured responses for approximately 6% (n = 13) of FPV installations globally (based on the total number of FPV systems, n = 229, (Solarplaza, 2019)). All of the FPV installations surveyed were deployed on human-made water bodies (although FPVs have been deployed on natural water bodies); 11 FPV arrays were deployed on irrigation reservoirs, and one on a reservoir supplying raw water to a water treatment works, storm water pond and a sand extraction pit. These deployment locations may reflect the co-benefits of locating FPVs near to energy demand (e.g. water treatment works) and the relative challenge of obtaining permission to deploy FPVs on natural water bodies.

At present, FPV capacity is often limited by water body size and the desire to deploy systems to meet specific power needs. For example, the three largest systems deployed in the UK are on raw water reservoirs and were designed to meet the electrical needs of the adjacent water treatment works. Consequently, the capacities tend to be smaller than ground-mounted systems, with the surveyed FPV's capacities ranging from 26 to 2100 kWp (Figure 2-2), although, globally, systems up to 70 MW have been deployed (Solarplaza, 2019).

Given the implications for solar radiation and wind energy inputs and thus water body response (Haas et al., 2020), percentage cover is the most important determinant for resolving impacts on the hosting water body (Exley et al., 2021a). Percentage cover has been shown to impact physical, chemical and biological water body properties and processes ((Grizzetti et al., 2019; Exley et al., 2021a); section 2.5.1.1), and ranged from 3% to 74% in the survey (Figure 2-2). The optimum FPV percentage coverage needs to balance power demands with potential water quality impacts in light of other water body uses (World Bank Group et al., 2019).

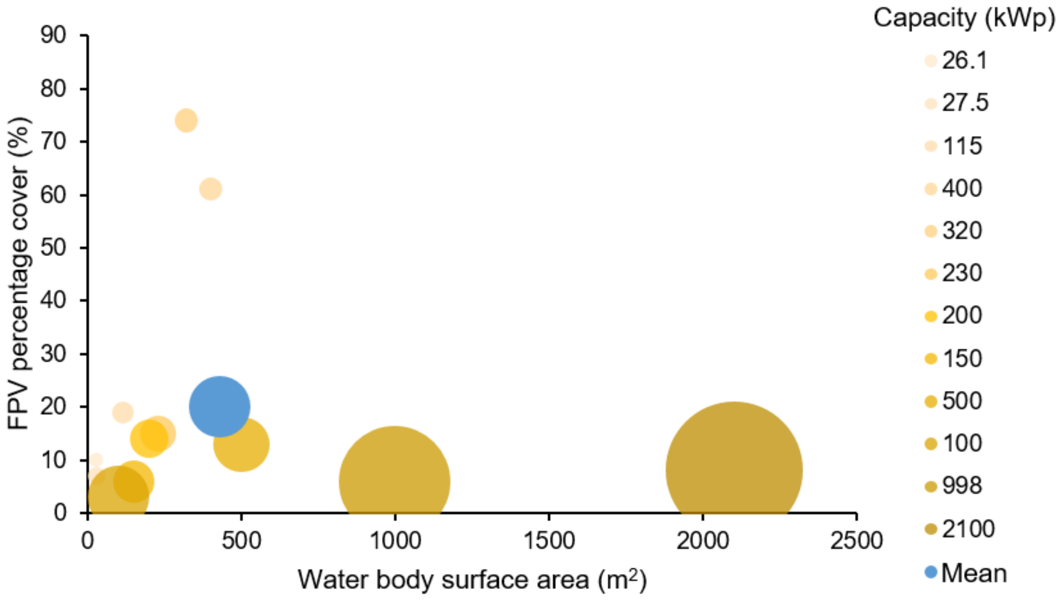


Figure 2-2 – Global stakeholder reported FPV surface coverage as a percentage of water body surface area. Each 'bubble' is proportional in size to the capacity (kWp) of the individual FPV array. The blue bubble indicates the mean FPV percentage coverage, host water body surface area and array capacity.

Scientific evidence of the effects of surface covers on water bodies infers some negative impacts (see section 2.5.1.1), for example, a switch to problematic phytoplankton species. However, the survey respondents did not detail any adverse water body impacts; the negative impacts were predominantly technical, such as issues with operation and maintenance (see supplementary information for further details; section 2.9.3). Nevertheless, water body impacts may have been overlooked as specific monitoring was only undertaken at two of the sites post-deployment, the survey was completed by FPV operators who may have limited environmental expertise, and many impacts may not be identifiable visually. For example, all survey respondents reported birds perching and/or nesting on the PV panels or infrastructure supporting the panels as solely a technical issue, as bird fouling reduces PV performance (Fouad et al., 2017; Said et al., 2018). However, results from our evidence review suggest that bird fouling increases the nutrient loading of phosphorus (Manny et al., 1994; Chaichana et al., 2010) and bacterial pathogens, including campylobacters (Mitchell and Ridgwell, 1971; Alderisio and DeLuca, 1999; Benskin et al., 2009), both of which have detrimental impacts on water quality. For example, bird droppings have been found to account for 25-34% of external phosphorus loading to an urban lake (Scherer et al., 1995), with other studies identifying even greater loading (e.g. Manny et al., 1994). Unlike other perching features, bird droppings will be washed off during panel cleaning, in addition to heavy rainfall (Maghami et al., 2016), releasing pulses of nutrients into the host water body (see supplementary information for further details; section 2.9.3). Moreover, there may be numerous other unseen impacts on water quality, such as changes to thermal stratification (Exley et al., 2021a), phytoplankton populations and lake productivity.

2.5.1.3 Evidence synthesis

The limited scientific evidence and FPV operator knowledge demonstrates a critical need to rapidly develop a more detailed understanding of FPV impacts on water bodies, including the effect of FPV design and water body characteristics (see section 2.5.1.1). Whilst scientific evidence of the water body impacts of FPV is very limited, the consequences of other water body covers suggests significant physical, chemical and biological impacts could occur. The limited stakeholder evidence is underpinned by limited monitoring of existing FPV installations and that many of the potential water body impacts are not visible. Consequently, there is an urgent need to generate FPV specific evidence of water quality impacts through both scientific assessments and by extending stakeholder monitoring beyond minimum statutory obligations (see section 2.7), encapsulating different water body types and FPV designs, along with modelling capabilities.

2.5.2 Potential ecosystem service impacts

Perturbations to ecosystem properties and processes caused by FPVs will influence the provision of ecosystem goods and services upon which society relies. Water bodies provide a range of essential ecosystem services and store vital natural capital (DEFRA, 2018). For example, water bodies are critical for providing drinking water, regulating water quality through natural filtration and supporting essential nutrient cycling (Grizzetti et al., 2016; Reynaud and Lanzanova, 2017). However, one of the challenges of assessing the impact of interventions, such as FPVs, on ecosystem services is correlating beneficial and detrimental changes in properties and processes, which are measurable, to ecosystem services which are commonly estimated using a range of measures (Dick et al., 2014; Birkhofer et al., 2015). Here we use our scientific understanding and stakeholder expertise to infer the potential of FPVs to impact ecosystem services and natural capital.

Water body type is central to estimating the ecosystem services delivered and their associated value (Reynaud and Lanzanova, 2017), and thus the impacts of FPV deployment. Despite only four types of FPV hosting water body being identified in the stakeholder survey (see section 2.5.1.2), stakeholders at the workshop identified an extensive range of potential recipient water bodies, suggesting that as FPV deployments accelerate, hosting water body types may expand. All water body types identified offer additional ecosystem services beyond the supply of low carbon energy. However, there was variation in the number of services, and likely value, between water body types (Table 2-2). FPVs could affect every ecosystem service provided by water bodies except 'buffering of flood flows, erosion control through water/land interactions and flood control infrastructure' (Table 2-2). Of all the ecosystem services, the regulation of water quality is provided by nearly all the human-made water bodies that may host FPVs (Table 2-2). Moreover, even if the delivery of additional ecosystem services were unnecessary, such as food provisioning, many would need to be maintained by default given their synergistic relationship with water quality and the complex dynamic interactions between individual ecosystem services (Rodríguez et al., 2006; Brauman et al., 2007; Bennett et al., 2009; Grizzetti et al., 2019).

Whilst most ecosystem services could be impacted by FPVs, the direction and magnitude of impacts are often unclear due to limited evidence of the effects of FPV and the complexity of water body function (Armstrong et al., 2020). For example, in terms of the provisioning of water for consumptive use, FPVs could enhance the quantity of water available and potentially the quality: reduced phytoplankton biomass (influenced primarily by temperature and light), evaporation (primarily influenced by wind and water temperature) and sediment resuspension rates (primarily influenced by wind mixing) are potential positive consequences of FPVs (see section 2.5.1.1). However, there is a chance that FPV could enhance ecosystem disservices, impacting the quality and quantity of water available for consumptive use. For example, changes in phytoplankton species dynamics to taxa which are suited to the low-light, non-turbulent conditions under FPVs including problematic blue-green algae and filamentous diatoms (see section 2.5.1.1.3). Predicting the consequences of FPV across the full suite of ecosystem services water bodies provide is particularly challenging given the range of ecosystem processes and properties that will influence the outcomes (Bennett and Balvanera, 2007).

On average, natural water bodies identified by workshop attendees as potentially suitable for FPV deployment support double the number of ecosystem services compared to those identified for human-made water bodies. The difference suggests that, on average, deployments of FPVs on human-made water bodies may have fewer adverse impacts on ecosystem service provision (Table 2-2) and ultimately on the SDGs. Unsurprisingly, this reflects the motivation to create water bodies that deliver a specific ecosystem service (Saulnier-Talbot and Lavoie, 2018) compared to natural water bodies that have existed for millennia and provide a range of ecosystem services (Maltby et al., 2011). Given that all the FPV deployments reported in the stakeholder survey were on human-made water bodies, reflecting a global trend (World Bank Group et al., 2019), suggests current FPV deployments may have relatively limited impacts on ecosystem service provision. However, water body ecosystem services and their value are likely to change over time in response to climate change (Nelson et al., 2013; Chang and Bonnette, 2017) and changes in water body use and ecosystem service demand (Saulnier-Talbot and Lavoie, 2018). Consequently, enhancing knowledge of the impacts of FPVs on all water bodies is important.

Table 2-2 – Water body type and potential ecosystem service delivery. Water body types gathered from attendees at the Floating solar: water quality impacts workshop. Ecosystem service typology based on Costanza et al. (1997); de Groot et al. (2002); Chopra et al. (2005); Kumar (2010); Maltby et al. (2011); Grizzetti et al. (2016); Wood et al. (2018). A • indicates an ecosystem service delivered by the water body. ▀service and treated water reservoirs store fully treated potable water in a drinking water network, ◆raw water reservoirs store untreated water, ❖bankside storage holds water abstracted from a river prior to water treatment and treatment reservoirs

		Water body type and potential ecosystem service delivery																						
		Natural										Human-made												
		Ponds	Tarns	Lakes/ Lochs (freshwater)	Rivers	Fjords/sea lochs	Seasonally dry water beds	Marshes and wetlands	Lagoons	Estuaries/tidal areas	Open ocean/sea	Service and treated water reservoirs■	Raw water reservoirs◆	Bankside storage❖	Agricultural ponds	Canals	Aqueducts	Harbours, marinas, docks	Filter beds	Sewage ponds	Contaminated water area	Settlement ponds for mining	Fish farms or aquaculture	**Total services**
Provisioning	Aquatic organisms for food and medicines			•	•	•				•	•												•	6
Provisioning	Water (quantity and quality) for consumptive use (for drinking, domestic use, and agriculture and industrial use)	•	•	•	•				•			•	•	•	•									9
Provisioning	Water for non-consumptive use (e.g. generating power or navigation)	•		•	•	•				•	•		•	•		•	•	•						11
Regulating	Buffering of flood flows, erosion control through water/ land interactions and flood control infrastructure	•	•	•	•		•	•	•				•	•	•	•	•				•	•		14
Regulating	Maintenance of water quality (natural filtration and water treatment)	•	•	•	•			•	•			•	•	•	•		•		•	•	•	•	•	16
Regulating	Climate regulation	•	•	•				•																4
Cultural	Tourism & Recreation (kayaking, hiking, etc.)	•	•	•	•	•		•	•				•	•		•		•					•	11
Cultural	Existence values (e.g. personal satisfaction from seeing water bodies)	•	•	•	•	•		•	•	•			•	•		•								10
Supporting	Role in nutrient cycling (role in maintenance of floodplain fertility), primary production	•	•	•	•	•		•	•	•	•												•	10
Supporting	Predator/prey relationships and ecosystem resilience	•	•	•	•	•		•	•	•	•		•										•	11
	Total by water body	9	8	10	9	6	1	7	7	5	4	2	7	6	3	4	3	2	1	1	2	2	5	
	Total by water body type	Total = 66					Mean = 6.6					Total = 38						Mean = 3.2						

2.5.3 Critical implications for water bodies

Once FPV ecosystem service effects are understood, prioritising particular ecosystem services and trading potential positive and negative impacts of FPVs for specific water bodies will be imperative. Given the common underpinning importance of water quality regardless of water body type or use, and lack of understanding of FPV impacts, we focus on the impacts of FPVs on the physical, chemical and biological properties of water bodies highlighted in the evidence review (section 2.5.1.1).

Overall, stakeholders perceived the enhancement of water body physical processes by FPVs as offering the greatest opportunity in terms of water quality, specifically the potential to reduce evaporation (score +9) – which strongly aligns with the evidence gathered during the review (Table 2-1). Conversely, stakeholders perceived changes to water body chemical properties and processes as representing the greatest potential threat of FPVs in terms of water quality impacts, identifying nitrification and the consequent deoxygenation of the water in particular (score -14, Table 2-1). The scientific evidence mirrored these stakeholder concerns, with the majority of evidence suggesting that water body covers adversely impact water chemical properties and processes (Table 2-1). In terms of biological impacts, the likelihood of reduced phytoplankton growth was perceived as the greatest opportunity of FPV deployment on water bodies (score +7.5, Table 2-1). However, the uncertainty in response, particularly the potential for blue-green algae proliferation (as competition from other species reduces due to lower light levels), was seen as the greatest threat (score -12, Table 2-1). Concern that prolonged periods of cover could lead to large phytoplankton blooms was also highlighted (score -4, Table 2-1). The broad range in stakeholder response for biological impacts emulates the mixed evidence outcomes gathered during the evidence review (Table 2-1).

The diversity of actors in the knowledge system (i.e. stakeholders), and the associated implications for their primary interests, led to variation in assessments of opportunities and threats. For example, reduced planktivorous fish stocks may enhance water quality by lowering nutrient concentrations and improving water clarity (Bernes et al., 2015), a benefit to raw water reservoir managers. However, fish kills suggest poor ecological condition and many water body managers are required to replenish fish stocks for recreational purposes. The largest variation in responses (i.e. responses were spread over four or more threat or opportunity categories) were for the potential of FPVs to reduce water temperatures, lead to fish kills, modify phytoplankton community composition and reduce phytoplankton growth (Table 2-1). In contrast, stakeholders unanimously viewed all chemical responses as a threat, except for salinity impacts (Table 2-1).

The differences in stakeholder-identified relative opportunities and threats of FPVs for water bodies indicates the complexity in resolving deployments for specific water body types and integrating ecosystem service impact with management and design decisions (de Groot et al., 2010). For example, balancing the delivery of ecosystem services beyond the provision of drinking water from a water supply reservoir, such as recreational and leisure opportunities (Saulnier-Talbot and Lavoie, 2018; Meyerhoff et al., 2019) and disservices such as greenhouse gas emissions, which increase the rate of global warming (Tranvik et al., 2009; Deemer et al., 2016). Moreover, understanding the impacts in light of FPV designs, host water body characteristics, and management goals will be critical to maximise the opportunities and minimise the threats posed by FPVs (World Bank Group et al., 2019). For example, minimising water quality impacts on raw water reservoirs will be a priority, but potentially of little consequence for irrigation reservoirs; evidence for this can be seen in the survey results, where stakeholders routinely monitored water quality for raw water reservoirs but not for irrigation reservoirs (see section 2.5.1.2). Moreover, if FPVs are deployed on a reservoir supplying drinking water with no public access, a lack of recreation opportunity cannot be considered an ecosystem disservice. Consequently, identifying the full suite of ecosystem services opportunities and threats posed by FPVs is complex and should be resolved for individual water bodies prior to deployment.

2.6 The overall sustainability of FPV

To determine the overall sustainability of FPV, the impacts of FPV on ecosystem services (Table 2-2) and the links between ecosystem services and the SDGs, including dependencies across SDGs (Le Blanc, 2015), can be placed into a generalised framework based on the UN SDGs. We found FPVs have opportunities and trade-offs with nine water body ecosystem services and may beneficially or detrimentally affect progress towards reaching eight out of the 17 SDGs. Based on the ecosystem service links determined by Wood et al. (2018), we found the SDGs most linked to water bodies (i.e. by seven ecosystem services), and thus potentially most influenced by FPV deployment, are *zero hunger* (SDG2), *sustainable cities and communities* (SDG11) and *climate action* (SDG13) (Wood et al., 2018) (Figure 2-3). *Clean water and sanitation* (SDG6) is linked to six water body ecosystem services; *no poverty* (SDG1) to four; *good health and wellbeing* (SDG3) to three; and *industry, innovation and infrastructure* (SDG9) and *responsible consumption and production* (SDG12) to two (Wood et al., 2018) (Figure 2-3). Out of the ten water body ecosystem services, FPVs are most likely to impact on *water quality provisioning* (Table 2-2), therefore, likely making opportunities and trade-offs with SDGs 1, 3, 6, 11 and 13 the most widespread (Figure 2-3).

Moreover, four of the SDGs, *affordable and clean energy* (SDG7), *decent work and economic growth* (SDG8), *life below water* (SDG14) and *life on land* (SDG15), are partially linked to several other SDGs (Le Blanc, 2015). Thus, FPV deployment could beneficially or detrimentally affect SDGs indirectly (Figure 2-3).

Synthesising multiple components of the knowledge system highlights the complexities and potential extent of the opportunities and trade-offs in FPV sustainability, underscoring the need to accelerate understanding rapidly. To ensure relevance among the wide range of FPV installations identified in our international survey and potential recipient water body types identified by workshop attendees, our framework provides a generalised overview that is non-specific to FPV design or deployment characteristics (e.g. location, water body usage, lake size metrics etc.). Given the compelling evidence gathered, some ecosystem service interactions are more certain than others, regardless of FPV design or deployment characteristics, but this is not universal (see section 2.5.2). As knowledge of the beneficial and detrimental impacts of FPVs evolve, our framework can be populated with evidence beyond our current understanding, improving specificity and strengthening the overall knowledge system. As such, it will be critical to establish the variation in impacts between different FPV designs, host water body characteristics and water body management goals through open sharing of installation-specific data and collaboration between all knowledge system actors and entities.

Figure 2-3 – Generalised framework linking outcomes gathered from the knowledge system (e.g. international survey, evidence review and stakeholder workshop) with the ecosystem services delivered by freshwaters (based on Wood et al. (2018)) and the United Nation's Sustainable Development Goals (SDG). Links between Tier 1 (light grey box) and Tier 2 (dark grey box) SDGs are based on Le Blanc (2015).

2.7 Future research and innovation

The previous sections highlight notable knowledge gaps that impede the sustainable deployment of FPVs. Consequently, we suggest essential priorities for future research and innovation.

The international stakeholder survey and evidence review demonstrated the critical need for more monitoring of FPV installations to elucidate impacts. As stakeholders perceived changes to water chemistry as the greatest threat, work in this area should be prioritised. A concerted research effort is required to enhance fundamental understanding of the processes by which FPVs affect the water body. Moreover, stakeholder sampling protocols must be extended beyond minimum statutory obligations to enable better resolution of impacts. The knowledge generated should be synthesised across FPV deployments to elucidate the influence of FPV design and water body characteristics. Bayesian and fuzzy systems could provide a useful means to synthesise quantitative (e.g. from monitoring and simulations) and qualitative (e.g. expert insight) information from across the FPV knowledge system (Armstrong et al., 2020). The outcomes should be collated and made available to inform industry best-practices and guide future innovations. Moreover, enhanced knowledge will permit the implementation of standards for deployment, ensuring environmental compliance throughout the FPV's life cycle, including manufacturing, deployment, operation and decommissioning.

FPV design is adaptable and versatile (see Figure S 2-1 for examples), so the using a techno-ecological approach should be considered when innovating future systems (Hernandez et al., 2019). Incorporating engineering which is mutually beneficial for technological and ecological systems offers an opportunity to enhance the overall sustainability of FPV. For example, one respondent of the international stakeholder survey used glass-glass PV modules, enabling light to reach the water's surface to minimise ecological impacts. Other adaptations include the addition of an aeration system to manage deoxygenation risks. Such FPV design adaptations must reflect the specific deployment location and anticipated impacts.

Finally, means to produce urgently required low carbon energy should be compared to the counterfactual in order to maximise the overarching sustainability of the energy system. If not FPV here, then where? If not FPV, then what? To make such decisions improved knowledge and better integration of ecosystem services with management practices is required (de Groot et al., 2010). Mapping of ecosystem service and SDG impacts is currently generic, but FPVs are likely to interact with nine ecosystem services and eight SDGs. Resolving the impacts is critical to ensure FPVs are appropriately designed and located.

2.8 Conclusions

FPV deployments are increasing rapidly worldwide, but there is minimal scientific evidence of water body impacts. This is a critical knowledge gap given the potential implications for ecosystem services and ultimately sustainability with this emerging form of low carbon electricity. Here, by drawing on an FPV knowledge system underpinned by scientific evidence and stakeholder expertise, we elucidated the possible impacts. The evidence showed a range of physical, chemical, and biological water body properties and processes could be impacted, predominately driven by changes in light attenuation, water temperature, and water movement. However, the available evidence was limited and shows there is an urgent need for further research. Without this understanding, ecosystem service provision could be at risk, or opportunities for co-benefits missed, with implications for eight SDGs unknown. Ultimately, advancing the state of knowledge on FPVs will provide the framework to maximise environmental benefits, ensuring the preservation or enhancement of water body processes, function, and service delivery.

2.9 Supplementary Information

2.9.1 FPV Design Types

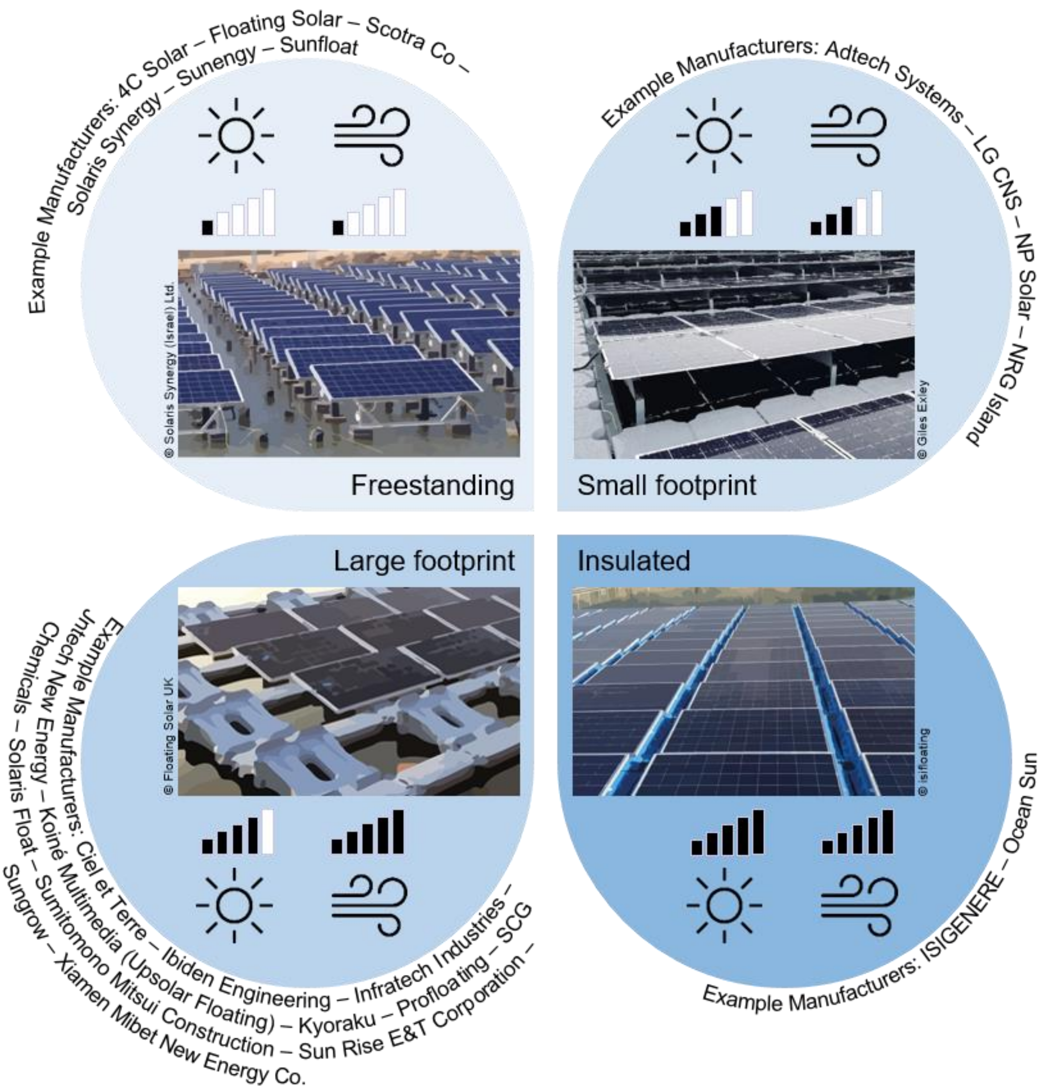


Figure S 2-1 – FPV design types categorised by surface coverage density. 'Freestanding' designs have the lowest coverage density, followed by 'Small footprint', 'Large footprint' and 'Insulated' designs in ascending order. Each design type is accompanied by an approximation of the design's effect on light, wind and interactions at the air-water interface. A single shaded bar represents a minimal effect, while five shaded bars indicated an extreme effect. Lists of example manufacturers are non-exhaustive and are categorised by the article author's, not the manufacturers themselves.

2.9.2 Evidence Review

To address objective one and two (establish ecosystem service opportunities and threats presented by FPV), a review of the scientific literature was conducted using the Defra Quick Scoping Review (QSR) method, a methodology designed to assess the volume and characteristics of an evidence base prior to evidence synthesis (Collins et al., 2015). The scope of the evidence search was constrained by the question; 'What are the potential impacts of floatovoltaics on water body function'? The question structure was defined using the Population, Intervention, Comparison and Outcome (PICO) framework (Table S 2-1). Briefly, the population is the subject of study, defined here as water bodies. The intervention is the exposure of water bodies to covers. In addition to searching for the impacts of FPV coverage on water bodies, and given the current limited level of understanding, natural (e.g. plants or ice) and artificial coverage (e.g. shade covers) disturbances were included as a proxy. The comparator, acting as a control, was defined as an absence of water body cover. The outcomes, or impacts on water quality, were inferred using our basic understanding of water body function and ecosystem services and included terms such as nutrients and stratification.

The PICO framework was finalised at a steering group comprised of project partners (four UK water utility companies), where further terms were added based on their industry knowledge. Search strings of the agreed terms were formed using the Boolean operators 'AND' and 'OR'. Relevant references were identified using Web of Science on 10 November 2019; the database considered most appropriate for ecological research (Randle-Boggis et al., 2020). The search was limited to studies published in English, while no restriction was imposed based on publication date. All literature returned was subject to pre-defined inclusion and exclusion criteria. Specifically, all literature needed geographical and climatic relevance to temperate regions and to contain evidence of an effect of water body coverage. Articles underwent an initial title screen, followed by an abstract screen, and if inconclusive, the whole article was read (Figure S 2-2).

Evidence, defined here as information, and preferably numerical, that surface covers impact water body function was extracted from each of the articles which passed the screening process. The evidence was compiled into a database with six fields, including the reference of the article, a summary, surface cover type, direction of evidence, evidence strength and impact type. Each article was summarised and categorised by surface cover type: 'Ice', 'Plant' or 'Artificial'. An outcome was allocated to indicate if the effect on water body function was 'negative', 'neutral' or 'positive'. Articles that reported neutral effects or no change were retained, while articles that speculated or hypothesised an effect were excluded from the review. Evidence strength was also assessed to indicate confidence: if based on simulations, of minor relevance to the temperate region or if there were concerns regarding study design and applicability to FPV the evidence was classified as weak. The remaining studies which met the search criteria were graded as strong. Further, to summarise the evidence base, evidence from individual studies were grouped by effect and sorted into physical, chemical, and biological categories.

2.9.2.1 Protocol Template

Table S 2-1 PICO elements table for Quick Scoping Review search.

Background for the work:	
The UK water industry has carbon emissions targets and one means of achieving this is through the deployment of low carbon energy technologies, reducing the carbon intensity of power used and benefiting from direct use energy efficiency gains. Solar photovoltaics have been deployed on buildings, land and, more recently, on reservoirs. Deployment on reservoirs avoids land use pressures and averts costs and time delays associated with planning permission as they are categorised as permitted developments. Floating photovoltaic solar systems, floatovoltaics, have been deployed by Thames Water on the Queen Elizabeth II and by United Utilities on Godley and more recently Langthwaite reservoirs. However, to date the potential impacts on the water body, and implications for water quality are unknown, posing risks and missed opportunities for water supply companies.	
Conceptual model: A description of how the policy, practice and science related to the evidence review topic interact and influence each other	
Potential risks to water supply will vary depending on water body characteristics and floatovoltaic extent and design. Development of a preliminary theoretical framework suggests floatovoltaics could have significant impacts on water reservoirs. For example, the hypothesised reduction in light and water temperature could reduce cyanobacterial bloom occurrence, reducing health risks, disruption to recreation and water treatment costs. In contrast, there are also mechanisms by which floatovoltaics could promote the depletion of oxygen in the bottom waters and trigger release of nutrients from bed sediments, increasing water treatment costs and potentially posing problems for water treatability.	
Primary Question: The main question to be addressed by the review	
What are the potential impacts of floatovoltaics on water body function?	
Population:	Water bodies (lakes/reservoirs/ponds etc.)
Impact/intervention:	Exposure; water body covers as an analogue for floatovoltaics
Control:	Absence of a cover (as an analogue to floatovoltaics), non-covered water body
Outcome:	Impact; water quality, nutrients, algae/phytoplankton, stratification, mixing, diatoms, birds
Secondary questions: Additional questions to be addressed by the review that contribute to building up the evidence surrounding the primary question:	
N/A	

Scope of the work: clear limits of the question to be addressed by the review	
Geographical reference:	Relevance to the UK
Climatic conditions:	Relevance to the UK
Language restrictions:	English
Date restrictions:	N/A
Population restrictions:	Exclude brackish water bodies (freshwater only)
Outcome restrictions:	All outcomes with a relevance to the effect of floatovoltaics on water quality considered
Other restrictions:	N/A
Search date:	11/10/19

2.9.2.2 Search Strings

The following strings were searched for on Web of Science on 11/10/19:

Population:

TS =(Lake* OR Reservoir* OR Pond* OR "*Water bod*" OR "Bod* of water" OR Lagoon* OR basin* OR mere OR pool OR creek OR loch OR millpond OR sluice OR tarn OR lakelet*)

AND

Intervention:

TS =(("Ice cov*" OR "Ice layer*" OR "Frozen surface" OR "Vegetation cov*" OR "Plant cov*" OR "macropyhte* cov*" OR "canop* form*" OR "macrophyte canop*" OR "Float* lea* macrophyte*" OR "aquatic plant*" NEAR/5 (surface) OR "float* plant*")

OR

TS=("floating solar" OR floatovoltaic* OR "floating photovoltaic*" OR "floating PV" OR aquavoltaic* OR "suspend* cov*" OR "float* cov*" OR "float* ball*" or "bird ball*" OR "artificial surface cov*" OR "shad* cloth" OR pontoon* or "Reed bed*" NEAR/5■ (float*) OR jett* OR "house boat*" OR "houseboat" OR "float house")

No **Comparator** or **Outcome** strings were used. This was to ensure all potentially relevant articles were returned during the search. ■ NEAR/5 dictates that the specified terms must be within a certain number of words (five words) of each other.

2.9.2.3 QSR Evidence

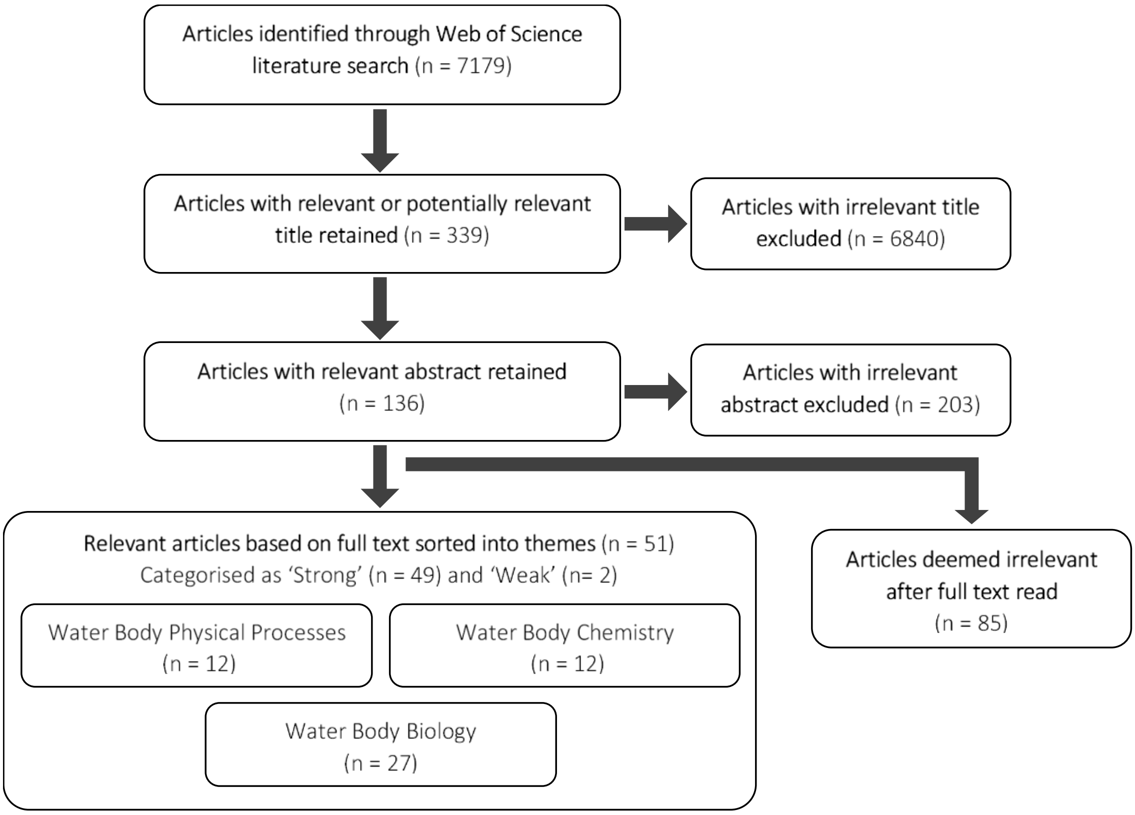


Figure S 2-2 – Flow diagram of the Quick Scoping Review process. The number of articles downloaded, screened and subsequently included or excluded are shown for stage. Articles were sorted into themes based on content.

2.9.3 International Stakeholder Survey

2.9.3.1 Methods

To address objective one (synthesise current evidence on the water body impacts of FPV), an anonymous online survey was distributed to FPV operators to gather their knowledge and expertise. This provides a means to gather contemporary understanding which is especially important given the relative immaturity of FPV and thus the limited studies in the scientific literature. Utilising the *Qualtrics* platform, participants were asked to complete multiple choice and text-entry questions. The survey focussed on four categories; FPV characteristics (such as array size and technology), water body characteristics (such as depth, surface area and use), sampling and data collection, and FPV array management (such as bird deterrents and cleaning). Prior to distribution, the survey was evaluated following standard ethical procedures and a pilot study conducted. The survey was then distributed via a stakeholder database, compiled from a search of the grey literature and of people who had signed up to our FPV website (www.sunnywaters.co.uk). The survey was also widely shared on Twitter and via the *PV Tech* email newsletter, facilitating a snowball technique to increase participation. Participants had six-weeks to complete the survey; any incomplete responses were discarded (n = 5). As an incentive for completion, participants were sent a summary of the survey responses.

2.9.3.2 Survey questions

2.9.3.2.1 *System*

- In which region and country is the floating solar array located?
- What year was the floating solar system installed?
- What is the generating capacity of the floating solar array?
- Please select the provider of the floating system. (Please chose one response, if the provider is not listed then please select 'Other' and type in the provider name).
- Please can you describe the float system (such as the design and the height from the water to the photovoltaic panel)? Please feel free to include a URL to a photo of the float system.
- Have you experienced any issues (e.g. technical, operational) with the floating solar plant?

2.9.3.2.2 *Water Body*

- How is the water body used? (select all that apply)
- Did the plans to deploy the floating solar array face any objections or concerns? Please consider objections and concerns that were voiced from groups including, stakeholders that use the water body, planning or government authorities and environmental organisations.
    - Please select from the list below any objections received about the deployment of the floating solar array.
- What is the average summer surface water temperature?
- What is the maximum depth of the water body?
- What is the surface area of the water body?
- Please indicate the area or percentage covered by the floating solar array.

2.9.3.2.3 *Water Quality and Data Collection*

- Did you or others collect water quality samples before deployment? This includes any sampling or monitoring conducted in preparation for the deployment of the array.

    - What water quality samples did you collect before the deployment of the floating solar array? Please include any specific sampling or monitoring you conducted prior to the installation of the floating solar array.
- Did you or others sample water quality post-deployment of the floating solar array?
    - Please describe the post-deployment water quality sampling you conducted.
- Since the deployment of the floating solar array, has it been necessary to action any additional water quality monitoring?
    - Please describe the additional water quality monitoring you are now conducting.
- Have you observed a change in algae type (e.g. a switch to deep water algae)? Please briefly describe any change.
- Are there any visible changes to the water quality or ecosystem impacts since deployment? Please describe them below.
- Have you or others observed birds landing directly on the floating solar panels or the infrastructure supporting the panels?
- Please select any methods you use to deter the birds from landing directly on the floating solar panels or the infrastructure supporting the panels?
- Please describe any design features of the floating solar system that were included to enhance the aquatic ecology value of the water body (e.g. holding areas for fish, artificial refugia for aquatic invertebrates etc.).
- Please describe the process for cleaning the floating solar array. Please include if water from the host water body is used and if the water from cleaning is washed into the host water body?

2.9.3.2.4 *Other*

- Based on your experience with floating solar, would you consider deploying another system if resources allowed?

2.9.3.3 Stakeholder survey ethics

Prior to distribution, the survey, participant information sheet and supporting resources (Appendix A) underwent ethical approval by the Faculty of Science and Technology Research Ethics Committee (FSTREC) at Lancaster University and a small trial to check survey functionality was conducted. The survey was distributed via a stakeholder database, compiled from a search of the grey literature and of people who had signed up via the www.sunnywaters.co.uk website.

2.9.3.4 Non-water quality effects from stakeholder survey

One in five participants of the stakeholder experience survey experienced technical or operational issues with their FPV system. These issues included; submerged cables, small punctures in the floats, problems with float connectors, higher operation and maintenance costs than expected, a grid outage issue, bird fouling, and bird nesting. However, the survey found all participants either agreed or strongly agreed that they would consider installing further FPV (Agree: 1, Strongly Agree: 13).

Bird fouling and other soiling, such as snowfall, leaves, dust and pollen decrease PV module performance. If individual cells are shaded, they are unable to produce any current. In turn, shaded cells create resistance to current generated by other cells. The shaded cells may heat up causing damage to the PV module (Maghami et al., 2016; Liu et al., 2018). All survey respondents noticed birds landing on the PV modules or the infrastructure supporting them. To overcome bird related issues, one in five survey participants stated they used bird deterrents. Bird deterrent technology included the use of laser and audible systems. Bird deterrents are widespread, with systems frequently being used in applications other than FPV such as airports, greenhouses and soft fruit farms.

Bird fouling and the deposition of dust and other debris on PV modules necessitate periodic cleaning to maintain module performance. Survey participants reported a variety of cleaning methods, including; cleaning with a brush, use of a photovoltaic cleaning robot, and ‘manual cleaning’. Water for the cleaning operations was sourced either from the recipient water body or external sources, with waste water returned to the recipient water body. Discharging waste cleaning water directly into the host water body may create a spike in nutrient concentrations if the soiling is nutrient rich. No respondent reported the use of chemicals for cleaning the PV modules, likely in order to preserve the warranty. Typically, PV modules are washed with non-abrasive soft brushes and ideally de-ionised water only.

The weaknesses discussed here are not necessarily exclusive to FPV, with other energy generation sources susceptible to similar and other issues. Weaknesses may be location or design specific, while strategies to mitigate issues are often implementable. For example, survey participants were asked if designs to maximise ecological value had been chosen. One respondent used glass-glass PV modules, enabling light to reach the water's surface. Another used floats that 'allow the life of living beings under the array'.

Chapter 3 – Floating photovoltaics could mitigate climate change impacts on water body temperature and stratification

Exley, G., Armstrong, A., Page, T. and Jones, I. D. (2021) 'Floating photovoltaics could mitigate climate change impacts on water body temperature and stratification', Solar Energy, 219, pp. 24-33, doi: https://doi.org/10.1016/j.solener.2021.01.076

3.1 Highlights

- Effects of floating solar photovoltaics on lake thermal structure are simulated.
- Low coverages of floating solar have minimal impact and may enhance water quality.
- Impacts can be as, or more influential, than the effects induced by climate change.
- Floating solar could be used as a tool for managing water quality in reservoirs.

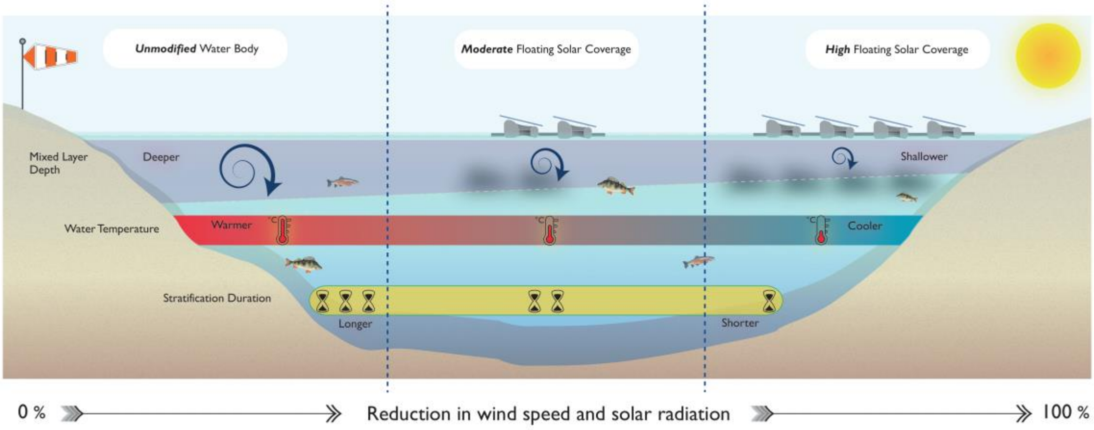
3.2 Abstract

Floating solar photovoltaics, or floatovoltaics (FPV), are a relatively new form of renewable energy, currently experiencing rapid growth in deployment. FPV decarbonises the energy supply while reducing land-use pressures, offers higher electricity generating efficiencies compared to ground-based systems and reduces water body evaporation. However, the effects on lake temperature and stratification of FPV both sheltering the water's surface from the wind and limiting the solar radiation reaching the water column are unresolved, despite temperature and stratification being key drivers of the ecosystem response to FPV deployment. These unresolved impacts present a barrier to further deployment, with water body managers concerned of any deleterious effects. To overcome this knowledge gap, here the effects of FPV-induced changes in wind speed and solar radiation on lake thermal structure were modelled utilising the one-dimensional process-based MyLake model. To resolve the effect of FPV arrays of different sizes and designs, observed wind speed and solar radiation were scaled using a factorial approach from 0% to 100% in 1% intervals. The simulations returned a highly non-linear response, dependent on system design and coverage. The responses could be either positive or negative, and were often highly variable, although,

most commonly, water temperatures reduce, stratification shortens and mixed depths shallow. Modifications to the thermal dynamics of the water body may subsequently drastically alter biogeochemical processes, with fundamental implications for ecosystem service provision and water treatment costs. The extreme nature of response for particular wind speed and solar radiation combinations results in impacts that could be comparable to, or more significant than, climate change. As such, depending on how they are used, FPV have the potential to mitigate some of the impacts of climate change on water bodies and could be a useful tool for water body managers in dealing with changes to water quality, or, conversely, they could induce deleterious impacts on standing water ecosystems. These simulations provide a starting point to inform the design of future systems that maximise ecosystem service and environmental co-benefits from this growing water body change of use.

Keywords: Floating solar, floatovoltaics, renewables, mixed depth, ecosystem impacts, lake management

3.3 Graphical Abstract



3.4 Introduction

Increased energy demands and the urgent need to decarbonise are prompting the rapid deployment of renewable energy technologies. One such technology, solar photovoltaics (PV), has experienced exponential growth over the past 25 years (IEA, 2019) and accounted for 57% of newly installed renewable energy capacity in 2019 (REN21, 2020). While solar PV has traditionally been ground- or rooftop-mounted, water-deployed, floating solar photovoltaics (FPV), known colloquially as floatovoltaics, have emerged in recent years. Global cumulative FPV capacity more than trebled among the top 70 FPV systems from 2018 to 2019 (Solar Asset Management, 2018; Solarplaza, 2019; World Bank Group et al., 2019), with a forecasted annual average growth rate of 22% (Cox, 2019). Conservative estimates suggest that FPV has a global potential of 400 GW-peak (World Bank Group et al., 2018), demonstrating the likely widespread uptake of this renewable energy technology. Although this could be severely hampered by a lack of understanding about the impacts of the technology on the hosting environment (Lee et al., 2020; Stiubiener et al., 2020; Zhang et al., 2020; Gorjian et al., 2021; Ziar et al., 2021).

FPV systems are typically comprised of five main components: a pontoon of floaters, a mooring system, PV modules, cabling, and connectors (Sahu et al., 2016). The specific design of a system can be adapted to suit water body function and application through variations to floater material (Oliveira-Pinto and Stokkermans, 2020), PV module type (Tina et al., 2021; Ziar et al., 2020), orientation (Campana et al., 2019), and surface coverage (Cagle et al., 2020). However, each combination of components will have a unique impact on the atmospheric drivers of lake dynamics, potentially resulting in a large variation in lake function impacts between systems.

A growing body of evidence suggests that FPV has several advantages over conventionally deployed PV. Firstly, FPV averts the need for large areas of land-use change by occupying the surface of water bodies (Holm, 2017; Cagle et al., 2020). This is of particular benefit to land-scarce countries and regions with high land prices (Abid et al., 2019; Campana et al., 2019). Secondly, FPV has been shown to deliver enhanced performance over ground-based PV due to the cooling effect of the hosting water body (Choi et al., 2013; Sacramento et al., 2015; Yadav et al., 2016; Oliveira-Pinto and Stokkermans, 2020). The cooling yield has been found to vary across climates, with heat loss dependent on wind speed and the openness of the floating structure (Dörenkämper et al., 2021). Thirdly, and also dependent on system design, FPV has also been shown to reduce evaporative losses substantially (Choi, 2014; Santafe et al., 2014; Sahu et al., 2016; Taboada et al., 2017), potentially providing vital water savings for drought-stricken areas. Furthermore, studies have shown that hydroelectric dams operating in conjunction with FPV can optimise energy efficiency and improve system reliability

(Stiubiener et al., 2020; Zhou et al., 2020). Integrated hydroelectric-FPV systems may also lessen the environmental and social impacts of stand-alone hydroelectric operation (Sulaeman et al., 2021) providing synergistic benefits to the water-food-energy nexus (Zhou et al., 2020).

Nonetheless, the biological, chemical and physical impacts of FPV on water bodies remain virtually unknown (Ziar et al., 2021), despite the global importance of water bodies for supplying numerous ecosystem goods and services (Maltby et al., 2011; Reynaud and Lanzanova, 2017; Grizzetti et al., 2019). Given the forecasted growth in FPV deployment, it is critical that we increase our understanding of its impact on water bodies. A fundamental starting point to this understanding is recognising the impacts of FPV on the thermal structure of a water body, as this thermal structure will be directly affected by FPV and it has a pervasive influence on most other aspects of the ecosystem (e.g. Macintyre, 1993; Diehl et al., 2002; Huisman et al., 2004; Jäger et al., 2008).

A small number of previous studies have considered the effects of natural or artificial floating elements on lakes (e.g. Maestre-Valero et al., 2013; Ozkundakci et al., 2016). However, their focus has typically been on specific surface coverage ratios (e.g. Aminzadeh et al., 2018) or particular ecological effects such as phytoplankton and zooplankton assemblages (e.g. Pinto et al., 2007; Cazzanelli et al., 2008). Present understanding relating specifically to the ecological impacts of FPV on lake functioning is limited, with studies typically focussed on technological advancements and system implementation (e.g. Liu et al., 2017). Of the limited number of studies with an ecological focus, topics include; the viability of FPV on fish ponds (Chateau et al., 2019); the effect of novel FPV designs on water quality indicators at an FPV pilot site (Ziar et al., 2021) and the potential impact of sunlight reduction on biological processes, such as algal blooms (Haas et al., 2020) and microorganism proliferation in drinking water reservoirs (Mathijssen et al., 2020). Up to now, the impacts of FPV on water body thermal structure remains unexamined.

FPV will both reduce the amount of solar radiation reaching the water and shelter the water from the effects of wind mixing (Armstrong et al., 2020), modifying water body temperature and stratification. Wind speed and solar radiation typically have opposite effects on water body thermal structure. Decreases in wind will tend to increase stratification and surface warming, while reductions in solar radiation will enhance mixing and cooling of surface water (Kalff, 2002). At present, it remains unclear whether FPV-induced changes in wind speed or solar radiation will dominate, as well as the extent of any resulting changes to lake thermal structure. The critical role of temperature and stratification in determining lake biochemical and ecological processes (Elci, 2008; Kraemer et al., 2017) means that without this knowledge, deployment of FPV risks inadvertently altering the provisioning of ecosystem goods and services. This could derail future investment in FPV. Modifications to the processes, function and service delivery of water bodies with an FPV installation must be carefully managed to ensure the pathway to decarbonisation continues with minimal concomitant environmental impacts.

Here we address this knowledge gap by applying simulations from a one-dimensional, process-based model and data from a test lake in North West England. We simulate water temperature, mixed depth and stratification timing to (1) determine the sensitivity of a lake's thermal structure to FPV deployed at varying scale. We then (2) consider the potential ecosystem consequences and implications for lake management in a changing climate.

3.5 Methods

3.5.1 Site description

The impacts of FPV on lake thermal structure were modelled for the south basin of Windermere, a typical monomictic, mesotrophic, deep and temperate lake in the Lake District, North West England. The south basin of Windermere is long and narrow in shape – with a maximum depth of 42 m, a mean depth of 16.8 m and a surface area of approximately 6.7 km^2. As one of the most comprehensively studied lake systems in the world (Rooney and Jones, 2010), the wealth of understanding and availability of high-resolution meteorological and in-lake water temperature data make Windermere an excellent test system for this study (Maberly and Elliott, 2012).

3.5.2 Modelling methodology

3.5.2.1 MyLake

To resolve the effects of FPV on lake physical properties, we simulated lake variables by adapting an existing MATLAB model. *MyLake* v1.2 (Saloranta and Andersen, 2007) is a one-dimensional process-based model, used to simulate the daily vertical distributions of water body temperature, evaporation and instances of ice cover accurately. *MyLake* partitions horizontal layer volumes by exploiting interpolated lake bathymetric data, making it similar to other one-dimensional lake models. The lake water simulation part of the model is based on Ford and Stefan (1980), Riley and Stefan (1988) and Hondzo and Stefan (1993), while the ice simulation component is based on Leppäranta (1993) and Saloranta (2000). In brief, the model initially computes the temperature distribution of the lake for the 24-hour time-step, taking into account diffusive mixing processes and local heat fluxes. A sequential process then accounts for convective mixing, wind-induced mixing, the water-ice heat flux and the effect of river inflow (Saloranta and Andersen, 2007). The model has been successfully applied to various projects as a standalone simulation tool assessing lake thermodynamics and ice regime (e.g. Livingstone and Adrian, 2009; Woolway et al., 2017b). Predominantly, model parameters were kept as per the user manual (Saloranta and Andersen, 2004), with minor adjustments made during calibration (see Section 3.5.4).

3.5.2.2 Input data

Meteorological data, logged at 4-minute intervals using a Campbell Scientific CR10X data logger, were obtained from an Automatic Water Quality Monitoring Station (AWQMS) located at the deepest point of Windermere south basin for 2009 (Jones and Feuchtmayr, 2017). Specifically, air temperature (Skye Instruments SKH2012) was measured with a relative accuracy of ±0.35 °C; relative humidity (HOBO U23-001) with an accuracy of ±3%; incoming short-wave radiation (Kipp & Zonen CMP6) with a relative accuracy of 5%, and wind speed (Vector Instruments A100L2) was measured with an accuracy of 1% for wind speeds >10.3 m s^{-1} and an accuracy of up to 0.1 m s^{-1} for wind speeds <10.3 m s^{-1}. Water temperature profiles were obtained from 12 stainless-steel sheathed platinum resistance thermometers (Labfacility PT100), accurate to within 0.1 °C at the following depths; 1, 2, 4, 7, 10, 13, 16, 19, 22, 25, 30 and 35 m. Data were averaged to daily time steps. Estimates for cloud cover (0-1) were obtained from the R package *insol* (Corripio, 2019), using incoming short-wave radiation data from the AWQMS. As *MyLake* requires air temperature and relative humidity at 2 m, and wind speed at 10 m, corrections for measurement height were applied using a modified version of *Lake Heat Flux Analyser* (Woolway et al., 2015b). An iteration scheme with a smoothing function capable of assessing bulk fluxes at individual time steps allowed the appropriate scheme to be applied for accurate bulk flux simulation.

Daily discharge data from Windermere (River Leven) were used as a proxy for inflow (National River Flow Archive, 2018), following the assumption that inflow was approximately matched by outflow, with negligible change in lake level. Lake morphometry (Ramsbottom, 1976) was interpolated to one-metre intervals. The light attenuation coefficient (K_d, m^{-1}) for Windermere south basin was obtained from Woolway et al. (2015a).

3.5.2.3 Thermal structure simulations

The effect on wind speed and solar radiation (forcing variables) for a given percentage coverage of FPV is unknown and likely to vary substantially depending on the design of the floatovoltaic deployment. While reductions to both forcing variables are likely, the relative proportions of these reductions remain to be determined. Here, the forcing variables were altered using a factorial design, simulating reductions at 1% intervals from 0% to 100%. A factorial design allowed the identification of non-linear changes and thresholds in the output variables; this was of particular importance given the range of FPV designs and surface coverages that exist between different systems. Considering reductions to the forcing variables as a whole lake average, not just in the footprint of the array, maximises transferability between systems with different FPV designs.

3.5.3 Data Analysis

Mixed layer depth and Schmidt stability were subsequently estimated from modelled water temperatures using *Lake Analyzer* (Read et al., 2011), a freely available physical limnological tool (e.g. Read et al., 2012; Kraemer et al., 2015). Mixed layer depth was estimated using the metalimnion extent function, an algorithm that defines the approximate depth of the base of the mixed layer using a density gradient threshold of

0.1 kg m^{-3} m^{-1}. Mean mixed layer depth for the stratified period of each scenario, along with annual mean mixed layer depth were calculated.

The onset of thermal stratification was defined from the depth-resolved temperature simulations as the time when the temperature differential between the surface (0 m) and the bottom (42 m) of the lake exceeded 1 °C (Fee et al., 1996). Alterations to stratification duration were assessed by calculating the longest stratified period, defined here as the greatest number of consecutive days of stratification across the simulated period. This was then compared to the stratified period of the water body without FPV (unmodified system), permitting the calculation of a gain or loss in stratified days. Stratification onset and overturn days were derived from these data, with onset being the first day and overturn being the final day of the longest stratified period.

Three simulation scenarios were considered in further detail. The first being an equal (1:1) reduction to each forcing variable. Given the relative proportions of reductions to forcing variables remain unknown and are likely to vary substantially depending on FPV design (see Section 3.5.2.3), two scenarios with scaled forcing variables were simulated. A ‘wind dominant’ scenario where the wind speed reduction scales as 80% of the solar radiation reduction and a ‘solar dominant’ scenario where the reduction to solar radiation scales as 80% of the wind speed reduction.

3.5.4 Model Calibration

The model was calibrated for a one-year period against observed water body temperatures. Standard calibration procedures were undertaken following Moriasi et al. (2007). Briefly, calibration of the scaling of forcing variables was guided by Monte Carlo sampling of uniform parameter distributions. The Nash-Sutcliffe model efficiency coefficient (NSE) (Nash and Sutcliffe, 1970) and the Root Mean Square Error (RMSE) for metalimnion top, Schmidt stability and volume average temperature (see supplementary information, section 3.9) were used to identify the best simulation. Slight modifications to scale the original driving data were required to achieve the optimum parameter values for the calibration year; these were +2% for wind speed and +13% for solar radiation. These modifications are within the instrumentation error range and help reflect the variation likely experienced in forcing variables across the whole of the water body. Thus, driving the model using 2009 measured meteorological data with a wind speed multiplier of 1.02 and a solar radiation multiplier of 1.13 provided the optimum fit against the observed in-lake temperature data and this then constituted the baseline model simulation.

3.6 Results

After calibration, simulated water temperatures, volume averaged temperatures, mixed layer depth and Schmidt stability compared favourably to the observed data (Figure S 3-1). Model efficiency computed with NSE ranged from 0.93 to 0.97, an encouraging indication of the ability of the model to reproduce the system response (see supplementary information, section 3.9, for full calibration details, Table S 3-1).

3.6.1 Response of water body temperature to FPV

Modelled reductions to the forcing variables generally reduced annual mean surface water temperatures (Figure 3-1a). Surface water temperature reductions were non-linear, with small reductions to the forcing variables having a negligible effect and larger reductions having an increasingly greater effect (Table S 3-2). Increases in surface water temperatures occurred only in scenarios when wind speed was reduced considerably more than solar radiation. Similarly, annual mean bottom temperatures generally decreased, albeit less than surface temperatures (Figure 3-1b). As could be expected, given the reductions in surface and bottom water temperatures, mean annual volume average temperature was reduced for all scenarios (Figure S 3-2).

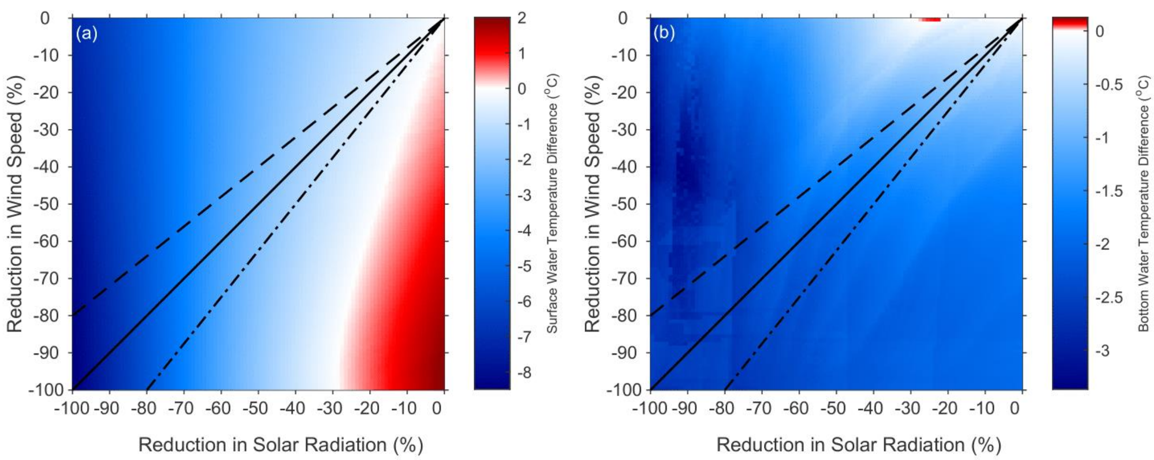


Figure 3-1 – Differences in mean surface and bottom water temperatures. Results are shown for mean annual (a) surface water temperature and (b) bottom water temperature. Water temperatures for the unmodified system were (a) 11.2 °C and (b) 7.0 °C. The solid black line represents an equal wind speed and solar radiation reduction *approximating floating solar coverage (1:1). A wind dominant scenario (solar radiation reduced more than wind speed) is shown with a dashed line. The dot-dash line represents a solar dominant scenario (wind speed reduced more than solar radiation).*

In 2009 there was no ice-cover on the lake and, indeed, ice cover on Windermere is very rare. Nevertheless, simulations with more than a 90% reduction to the forcing variables resulted in sufficiently cold surface water temperatures for ice to form (Figure S 3-3). Ice cover duration increased as the forcing variables were further reduced above 90%. For example, a 90% 1:1 reduction resulted in 22 days of ice cover, while a 98% reduction resulted in 43 days of ice cover.

Each reduction to the forcing variables decreased total annual evaporation in comparison to the baseline (Figure 3-2). At a 74% 1:1 forcing variable reduction, a threshold was crossed where dew formed on the lake surface, providing an annual net gain in water. Wind dominant scenarios (solar reduced by more than wind) saw greater reductions in evaporation than in solar dominant scenarios (Table S 3-2).

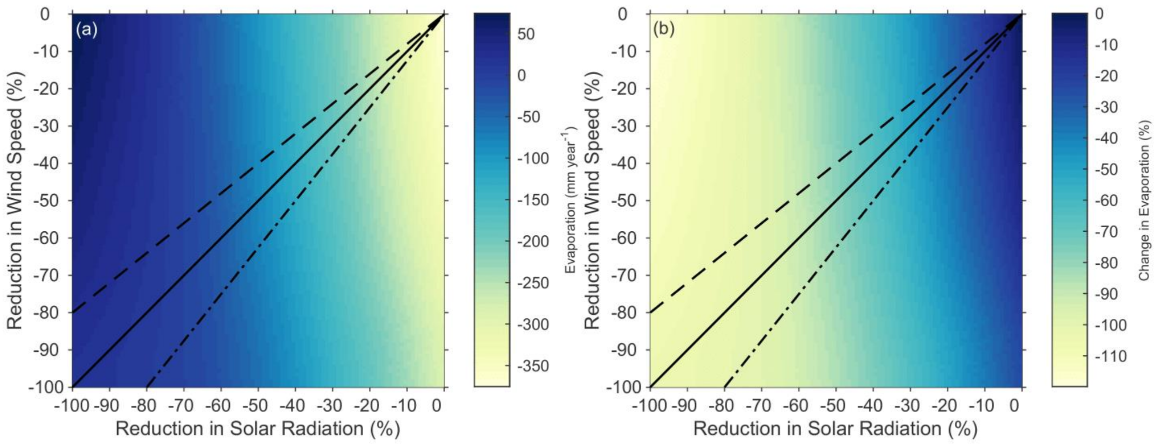


Figure 3-2 – Annual evaporation and change in evaporation. Results are shown for (a) total annual evaporation. A negative value indicates a net loss of water from the lake, while a positive value indicates a net gain in water. (b) The percentage change in evaporation in comparison to the baseline (375.2 mm year⁻¹). The solid black line represents an equal wind speed and solar radiation reduction approximating floating solar coverage (1:1). A wind dominant scenario (solar radiation reduced more than wind speed) is shown with a dashed line. The dot-dash line represents a solar dominant scenario (wind speed reduced more than solar radiation).

3.6.2 Response of stratification duration and strength to FPV

3.6.2.1 Stratification duration

When reductions to the forcing variables were 1:1 and did not exceed 45%, stratification duration was similar (± three days) to that of Windermere without FPV (Figure 3-3). Reductions in excess of this threshold decreased stratification duration by ~39 days for every additional 10% reduction to the forcing variables (Table S 3-3a). However, when the reductions to the forcing variables were not 1:1, stratification duration was modified even with small reductions. A solar dominant scenario, for example, increased stratification duration for all scenarios up to a 52% solar reduction, ranging from 3 to 13 days increase. The opposite was true when wind dominated, with stratification duration decreasing for all scenarios by a minimum of 29 days, up to a maximum of 214 days. Solar radiation reductions tended to dominate over wind speed reductions in determining stratification duration.

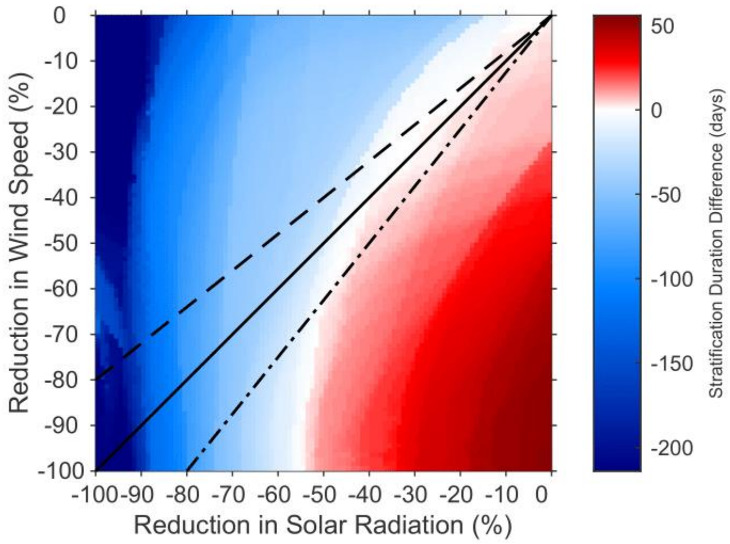


Figure 3-3 – Stratification duration for each scenario. The unmodified system was stratified for 214 days. The solid black line represents an equal wind speed and solar radiation reduction approximating floating solar coverage (1:1). A wind dominant scenario (solar radiation reduced more than wind speed) is shown with a dashed line. The dot-dash line represents a solar dominant scenario (wind speed reduced more than solar radiation).

3.6.2.2 Stratification Onset & Overturn

FPV deployment shifted the stratified period to later in the year, with delayed onset and overturn (Table S 3-3a, b). Wind dominant scenarios typically delayed stratification, where wind speeds remained proportionally higher than solar radiation (dashed-line Figure 3-4a). However, in scenarios where the wind speed was reduced by at least 30%, but solar radiation remained little changed, onset occurred earlier in the year. Overturn was delayed by up to 10 days as a consequence of reduced wind speed when 1:1 forcing variable reductions were less than 72%. Above 72%, the dominant forcing variable switched, with reduced solar radiation advancing overturn timing (Figure 3-4b).

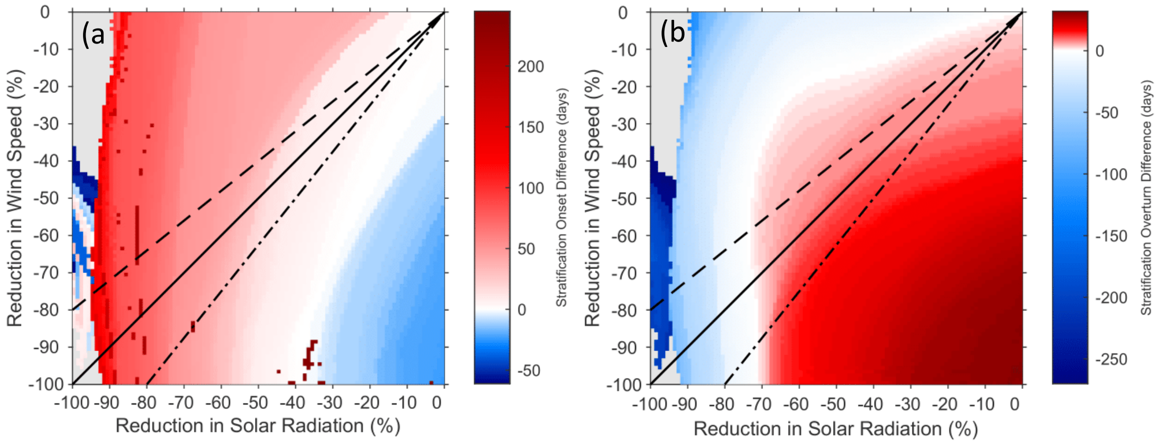


Figure 3-4 – Stratification onset and overturn. Change in day of year shown for (a) onset and (b) overturn of thermal stratification with modified wind speed and solar radiation. A negative value indicates an earlier day of the year (advancement), while a positive value indicates a later day of the year (postponement). Stratification onset and overturn occurred at day 102 and 315 respectively. The solid black line represents an equal wind speed and solar radiation reduction approximating floating solar coverage (1:1). A wind dominant scenario (solar radiation reduced more than wind speed) is shown with a dashed line. The dot-dash line represents a solar dominant scenario (wind speed reduced more than solar radiation).

3.6.2.3 Stability

Forcing variable reductions of up to 13% modified Schmidt stability by a relatively modest ±10 J m^{-2}, within 3% of the unmodified system. Scenarios where FPV reduced forcing variables by more than 13% reduced Schmidt stability substantially (Figure S 3-4). The stability of the water body only increased in instances when wind speed was reduced considerably, with solar radiation reduced by no more than 20%. A 10% solar radiation reduction and a 50% wind speed reduction, for example, increased mean annual Schmidt stability by 59 J m^{-2}. When each forcing variable was reduced by 50%, Schmidt stability was reduced by 126 J m^{-2}. Solar radiation changes were generally the dominant factor determining Schmidt stability, seen by the vertical bands in Figure S 3-4; changing the wind speed had less influence, especially at higher reductions of solar radiation.

3.6.3 Mixed Depth

Annual mean mixed depth shallowed with 1:1 forcing variable reductions of up to 60% (1:1) (Table S 3-4a), indicated by the negative mixed depth difference. Reductions greater than 60% (1:1) deepened the annual mean mixed depth, with the water body remaining mixed all year when reductions exceeded 94% (1:1) (Figure 3-5a, b). Mixed depth was shallowed by 0.58 m for every 10% reduction to the forcing variables up to 40% (1:1).

These changes in annual mixed depth were, in part, caused by the changes in stratification duration. Excluding this effect by focussing only on the stratified period, each scenario demonstrated a shallowing of mean summertime mixed depth for all 1:1 reductions (Figure 3-5c, d). Reductions in excess of 81% were highly non-linear (1:1), while smaller reductions were relatively proportional to the forcing variable reduction. The effect of FPV on mixed depth was considerable, with 85% of all scenarios shallowing for the stratified period (Table S 3-4b). Net summertime deepening occurred for the remaining scenarios, typically when very large changes to solar radiation were coupled with only small changes to wind speed. Mixed depth was at least halved for 29% of all scenarios.

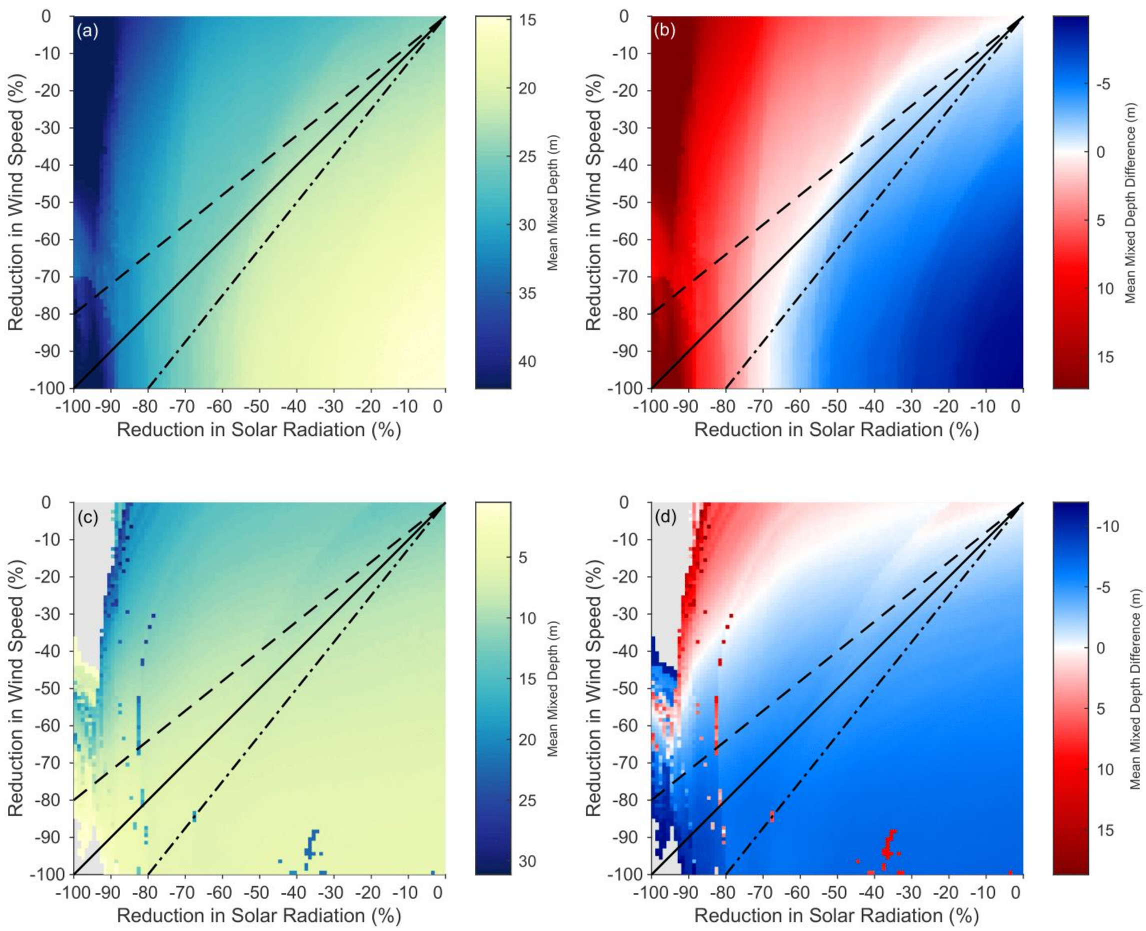


Figure 3-5 – Annual and stratified period mixed depths for each scenario. Results shown for (a) annual mean mixed depth, (b) difference from the baseline for annual mean mixed depth, (c) mean mixed depth for the stratified period and (d) the difference in mean mixed depth for the stratified period of each scenario with modified wind speed and solar radiation. A negative value on (b) or (d) indicates mixed depth has shallowed, i.e. has moved closer to the surface of the water body. A positive value on (b) or (d) indicates a deepening of mixed depth, i.e. mixed depth has shifted towards the bottom of the water body. Annual and stratified period mean mixed layer depth were 24.7 m and 12.4 m, respectively. The solid black line represents an equal wind speed and solar radiation reduction approximating floating *solar coverage (1:1). A wind dominant scenario (solar radiation reduced more than wind speed) is shown with a dashed line. The dot-dash line represents a solar dominant scenario (wind speed reduced more than solar radiation).*

There were strong seasonal dynamics in mixed depth, with progressive deepening throughout the summer months for scenarios where forcing variables were reduced by up to 75% (1:1) (Table S 3-5; Figure 6). Daily mixed depths, for scenarios with forcing variable reductions of 5, 10, 25, 50 and 75% (1:1) were initially closely aligned to the mixed layer depth of the unmodified system (Figure 3-6). At day 175 (24/06/09) the mixed depth of each scenario diverged from the unmodified system before converging again at day 325 (21/11/09). During the diverged period, scenarios with forcing variable reductions of 10% or greater differed substantially from the unmodified system, with mean mixed depths differing by more than 2 m. Although the trend remained consistent, the magnitude did vary. The difference in mixed depth peaked at 15.4 m for the 75% scenario on day 305 (01/11/09). A 100% (1:1) reduction to the forcing variables kept the water body fully mixed throughout the entire year.

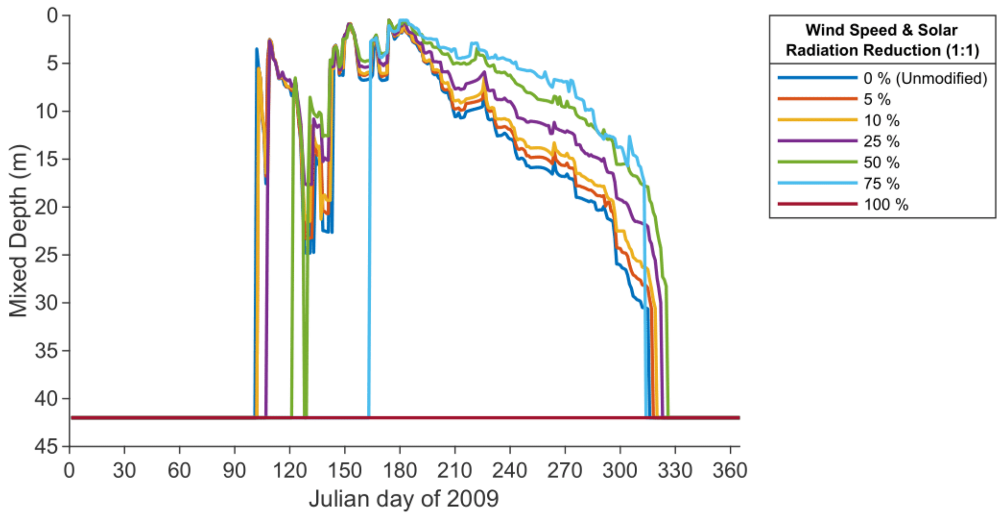


Figure 3-6 – Daily mixed depth. The scenarios shown have equal wind speed and solar radiation reductions approximating floating solar coverage (1:1).

3.7 Discussion

Lake thermal structure is dependent on a range of factors, including weather conditions, lake morphology and geographical location (Kalff, 2002). Although FPV deployments will alter net wind speed and solar radiation at the lake surface, the simulations here did not assume a specific extent of coverage or system design. Instead, we considered the effects of varying the scale of the forcing variables. For this discussion, we use only the assumption that surface coverage is negatively correlated with the forcing variables, i.e. that higher surface coverages cause a greater reduction in wind speed and solar radiation.

Thermal responses to differing reductions in wind speed and solar radiation varied enormously, from the negligible to the very large. Proportionate increases in alteration of driving forces resulted in highly non-linear responses. Both positive and negative responses were possible, depending on the changes to the driving variables, reflecting the opposite effects that wind speed and solar radiation typically have on lake thermal structure. The responses most commonly seen, though, were for temperatures to reduce, stratification to shorten, but mixed depths to become shallower. In the small number of instances when water temperature increased or stratification duration lengthened, an FPV system would need to cause substantial wind speed reductions and minimal solar radiation reductions. Conversely, the rare instance of mixed depth deepening (when considered during the stratified period only) occurred when substantial solar radiation reductions were coupled with minimal wind speed reductions.

3.7.1 The sensitivity of lake thermal structure to FPV

3.7.1.1 Cooling effect on water temperature

Water temperature changes were minor for small coverages of FPV, while more extensive FPV coverages drove major decreases (Figure 3-1). As many metabolic processes are highly temperature-dependent, the deployment of FPV at large coverages has the potential to change the functioning of lentic ecosystems by modifying animal behaviour, food web dynamics, life histories, species interactions and carbon cycling (Tranvik et al., 2009; Kraemer et al., 2017). Reduced water temperatures may also present operational challenges, particularly to networks comprised of cast iron distribution mains. During the colder winter months, increased tensile stresses from reduced water temperatures may lead to pipe fractures and an increased incidence of pipe bursts (Jesson et al., 2010).

Cooler water temperatures and greatly reduced wind speeds permitted the formation of ice at high surface coverages (Figure S 3-3), shifting the lake from a monomictic to a dimictic stratification regime. This considerable temporal shift in ice cover regime may have implications for cyanobacterial community composition (Ozkundakci et al., 2016) and fish behaviour (Jurvelius and Marjomki, 2008) while enhancing cultural ecosystem service provisioning (Knoll et al., 2019). In water bodies where FPV deployment could induce ice-cover, consideration would need to be given in the FPV design to mitigate the possibilities of compression forces and the restriction of array movement due to ice cover.

3.7.1.2 Changes to stratification length

Typically, the interception of incoming solar radiation by FPV extended the period of water column heating required in the spring before a density gradient established, postponing thermal stratification onset (Figure 3-4). Delayed epilimnion formation has been shown to shift the timing of spring phytoplankton blooms to later in the year (Meis et al., 2009), a phenological desynchronization which could lead to trophic mismatch, affecting the wider food web hierarchy (Visser and Both, 2005; Thackeray et al., 2013).

At low to moderate FPV coverages, stratification duration increased a little, and more so when wind reductions were substantially greater than solar radiation reductions (Figure 3-3), increasing the likelihood of hypolimnetic anoxia and the increased regeneration of soluble phosphorus and metals from the lake sediment (Forsberg, 1989; Beutel et al., 2008). The regeneration of heavy metals from lakebed sediment degrades water quality, necessitating enhanced water treatment, although the postponement of overturn may mean extra nutrient releases occur at periods of lower light availability when conditions are less suitable for phytoplankton growth (Butcher et al., 2015). At higher FPV coverages and scenarios with enhanced solar reduction, stratification duration shortened, which would tend to have the opposite effect of reducing anoxia and internal loading of nutrients and metals. The possibility of either outcome, increase or decrease, for such critical components of water quality emphasises the need for astute system design.

3.7.1.3 Alteration of mixed layer depth

While it was more common in the model results that water temperature was lowered, stability reduced and stratification shortened, mixed layers typically were shallowed, not deepened (Figure 3-5). Thus, reductions in solar radiation seemed to be more influential than wind speed reductions on water temperature and stratification, but the reduction in wind speed more influential on the depth of the epilimnion. As a fundamental driver of the chemistry and biology of lake ecosystems, the modification of mixed layer depth by FPV is of considerable importance for water quality (North et al., 2014; Kraemer et al., 2015; Yankova et al., 2017). FPV deployments will reduce photosynthetically active radiation (PAR) directly under array structures as well as mixed depth, so the ratio of epilimnetic depth to euphotic depth will alter, impacting phytoplankton growth (Huisman et al., 1999). Individual phytoplankton species with adaptations well suited to the modified epilimnetic depth to euphotic depth ratio beneath an FPV array will thrive, so changes in biomass and species composition should be expected. Non-continuous FPV deployments that allow a mosaic of light availability will complicate alterations to the phytoplankton community further. In particular, and of concern for water body managers, toxic cyanobacteria are well adapted to such conditions, utilising gas vesicles to regulate their buoyancy (Walsby et al., 1997). Simulations by Haas et al. (2020) found FPV systems that reduced light attenuation by 40%, or more, greatly reduced algal biomass, although they did not consider the effects of reduced wind speed, which may improve conditions for phytoplankton growth. The use of semi-transparent PV modules which provide specific transmittance windows to control light intensities have been proposed as a means to regulate phytoplankton growth (Zhang et al., 2020).

3.7.2 FPV and lake management in the context of a changing climate

The deployment of FPV is a direct response to the need to decarbonise the global energy supply in order to avert catastrophic climate change. Simulations here demonstrate that the effects on lake thermal structure of certain combinations of forcing variable reduction can be as, or more influential, than effects induced by climate change, and could either mitigate or exacerbate the impact. Numerous studies have identified increasing lake temperatures due to climate change, which are predicted to disturb both ecological and biogeochemical processes (e.g. Thackeray et al., 2008; O'Neil et al., 2012; Paerl and Paul, 2012). Woolway et al. (2019) found the average annual minimum surface-warming rate of eight lakes to be 0.35 °C decade^{-1}, while O'Reilly et al. (2015) found 235 globally distributed lakes' summer surface water temperatures were warming at a mean trend of 0.34 °C decade^{-1}. Thus, FPV may provide a useful tool for water body managers in mitigating against lake warming. For example, a decade of lake surface temperature warming could be mitigated with the deployment of an FPV array at a surface coverage that reduces lake-average wind speed and solar radiation by approximately 10% (Figure 3-1).

A further example of climate change mitigation, and of particular relevance to water-scarce locations, is the reduction in evaporation achieved by increasing FPV coverage (Figure 3-2). Cooler surface water temperatures weaken the water-to-air vapour pressure difference (Oke, 2002) while the FPV array intercepts incoming radiative energy, reducing the latent heat flux (Aminzadeh et al., 2018). Although research has previously identified that FPV will reduce evaporative losses (e.g. Ferrer-Gisbert et al., 2013; Redón-Santafé et al., 2014; Taboada et al., 2017), here it is also shown that the cooler surface water under FPV relative to the warmer, moist air above the water body permits dew deposition (Oke, 2002). At coverages greater than 74% (1:1 forcing variable reduction) a tipping point is crossed, resulting in a net gain of water to the lake.

However, while FPV could be an effective tool to mitigate against lake warming, FPV facilitated prolonged stratification duration and delayed overturn for some scenarios simulated in this study, with the potential consequences similar to those of climate warming (e.g. Adrian et al., 1995; Woolway and Merchant, 2019). Foley et al. (2012) examined long-term changes in stratification dynamics for a lake close to Windermere between 1968 and 2008; they found climate warming led to onset occurring 28 days earlier, overturn 18 days later, and the duration of stratification increased by 38 days. While FPV may be able to lessen the earlier onset of stratification brought about by climate change, the simulations show FPV deployment at lower coverages may also exacerbate the effects of climate change, potentially lengthening stratification duration and postponing overturn further.

3.7.3 FPV deployment best practice

These simulations show impacts on water body process and function in response to the deployment of FPV, with results which are relevant for other monomictic and mesotrophic deep lakes in the temperate zone, although variations in local climate may constrain or exacerbate many of the effects identified in this study. Any wider extrapolation of these impacts needs to take into consideration geographical and morphological factors that affect lake-atmosphere interactions. For example, ice cover, which occurred with high FPV coverage rates, would not occur in tropical regions due to higher air temperatures. Lakes in tropical regions also undergo different mixing regimes and tend to have less vertical temperature difference than temperate lakes (Lewis, 1987), so may respond differently to a temperate system. As latitude also influences turbulent surface heat fluxes (Woolway et al., 2018) and atmospheric stability above lakes (Woolway et al., 2017c), geographical location is likely to be a key contributor to the overall effect of FPV on lake thermal structure. The response of lakes with differing morphometric characteristics must also be considered; lake surface area, volume and mean depth are pertinent drivers of lake thermal structure (Lerman et al., 1995; Talling, 2001; Wetzel, 2001; Kraemer et al., 2015). In smaller lakes, convection is the dominant driver of mixed-layer turbulence, while wind shear is the primary driver for larger lakes (Read et al., 2012). Lakes of a smaller surface area have broader diel temperature ranges than larger lake-systems making them more prone to disturbance (Woolway et al., 2016). The temporal variation in these drivers will further modify the response between individual systems.

The number of water bodies hosting FPV arrays will increase with the sustained global drive to decarbonise energy supplies; therefore, we anticipate an urgent need for further understanding on the effects of FPV. Critically the model simulations demonstrate a high sensitivity to extent and design of deployments with highly non-linear thermal responses and both increases or decreases in temperature and stratification being possible. The model simulations suggest only a few percent cover (< 10%) of FPV typically only induces minor changes, but more significant covers (> ~50%) result in large temperature changes and very extensive modifications to stratification timing. The effects of FPV at larger coverages are of a similar magnitude to that of climate change. This considerable variation in possible response provides those deploying FPVs an opportunity to utilise deployments for actively enhancing water quality benefits as well as decarbonising electricity production.

3.8 Conclusion

By simulating the response of a lake to FPV deployed at varying extent, this study has demonstrated patterns of increased impact with increased perturbation, ranging from negligible to very large. Based on these findings, future FPV designs should consider the following to maximise ecosystem co-benefits and limit potential harm:

- Reductions in wind speed and solar radiation as an average across the lake cause a non-linear, complex response with the direction of these effects dependent on FPV array design, including coverage density
- Low FPV surface coverages had a negligible effect on the thermal structure of the test system, while high coverages were a major disruptor of the archetypal thermal structure
- FPV deployments may have impacts that are as, or more, influential than catastrophic climate change, therefore providing an opportunity to manage the effects of climate change on lake systems actively
- Appropriate design and deployment of FPV will be required to mitigate the likelihood of hypolimnetic anoxia and to optimise changes in the composition of phytoplankton communities as FPV modifies lake thermal structure and light climate

FPV is a substantial perturbation to water body process and function. Deployment with minor impact is possible, but the infancy of knowledge on FPV necessitates planning and impact assessment on a system-by-system basis.

3.9 Supplementary information for Chapter 3

3.9.1 Model Calibration

Model calibration was guided by the Nash-Sutcliffe model efficiency coefficient (NSE) (Nash and Sutcliffe, 1970) and the Root Mean Square Error (RMSE) for metalimnion top, Schmidt stability and volume average temperature (Table S 3-1). Simulated output was compared against observed data from the Windermere South Basin instrumented buoy.

3.9.2 Supplementary Figures

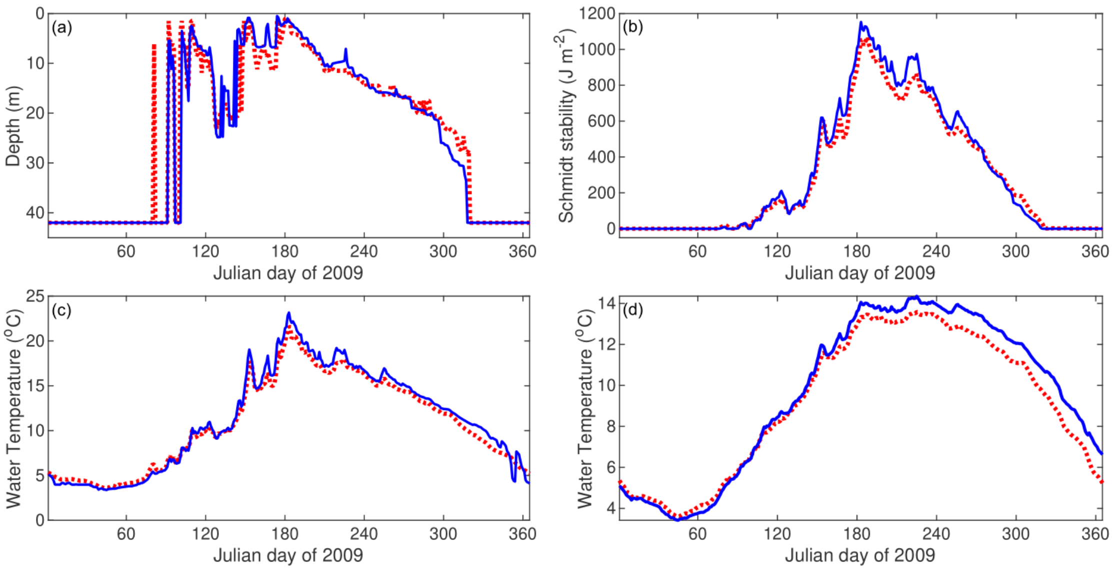


Figure S 3-1 – Annual observed and simulated data. Calibration shown for (a) metalimnion top depth, (b) Schmidt stability, (c) surface water temperature and (d) volume-averaged temperature.

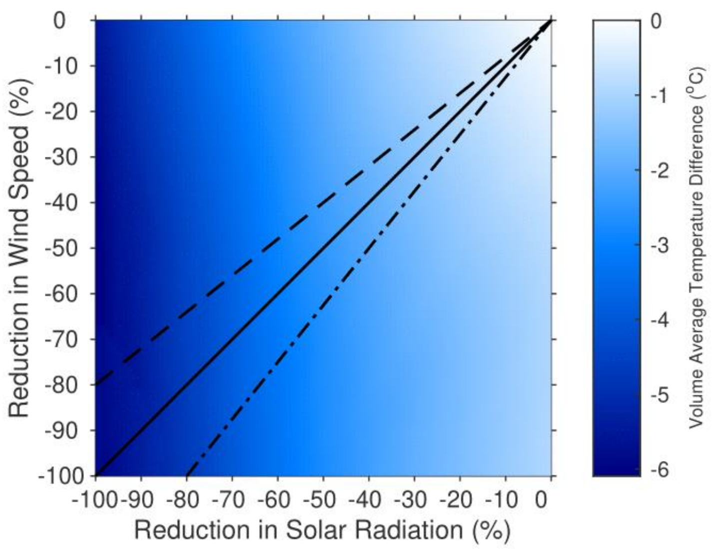


Figure S 3-2 – Mean annual volume average temperature.

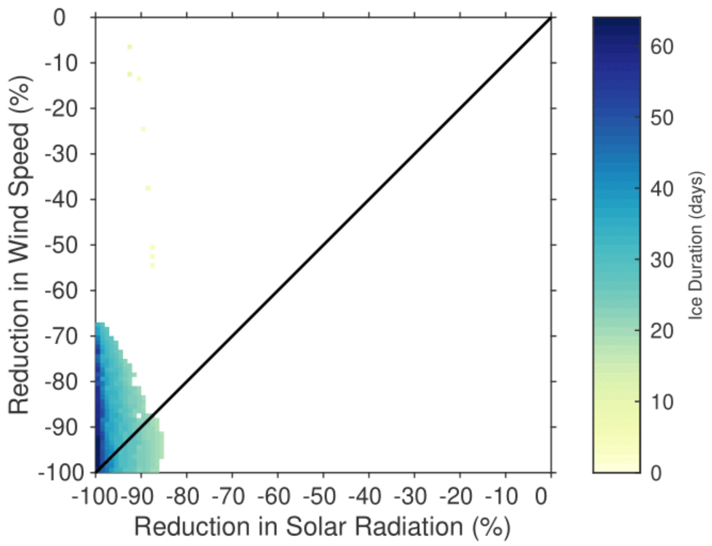


Figure S 3-3 – Ice cover duration. The solid black line represents an equal wind speed and solar radiation reduction (1:1). Ice cover duration for the baseline scenario is 0 days.

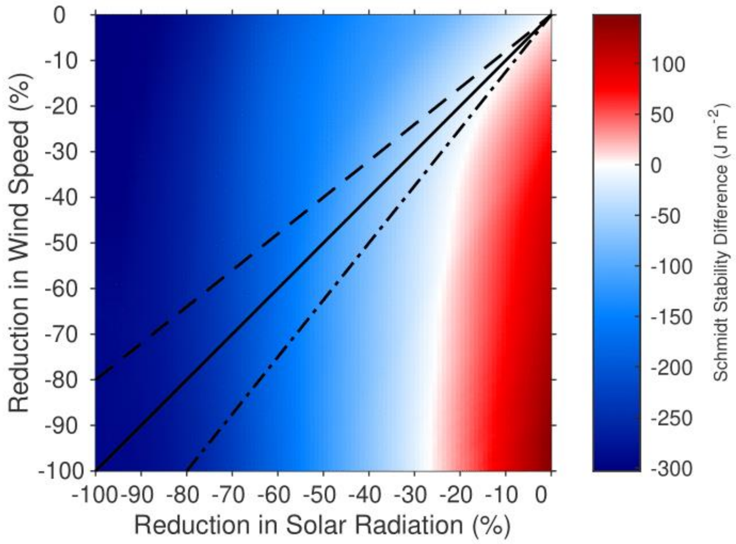


Figure S 3-4 – Mean annual Schmidt stability. Schmidt stability for the unmodified lake was 303.5 J m^{-2}. The solid black line represents an equal wind speed and solar radiation reduction approximating floating solar coverage (1:1). A wind dominant scenario (solar radiation reduced more than wind speed) is shown with a dashed line. The dot-dash line represents a solar dominant scenario (wind speed reduced more than solar radiation).

3.9.3 Supplementary Tables

Table S 3-1 – Likelihood measures for model calibration. Nash-Sutcliffe model efficiency coefficient (NSE) and Root Mean Square Error (RMSE). An NSE score of 1 (NSE = 1) indicates perfect efficiency, where there is an exact match between simulated and observed data.

	NSE	RMSE
Metalimnion top	0.93	3.88
Schmidt stability	0.97	46.83
Volume average temperature	0.95	0.70

Table S 3-2 – (a) mean annual surface water temperatures and (b) annual evaporation rates for a selection of wind speed and solar radiation reductions. The baseline scenario is indicated in bold. 1:1 wind speed and solar radiation reductions are shaded. Positive values in (b) indicate a net gain in water during the simulated scenario.

(a)

		Solar Radiation Reduction (%)						
		0	5	10	25	50	75	99
Wind Speed Reduction (%)	0	11.2	10.9	10.7	10.0	8.6	6.6	4.2
	5	11.3	11.0	10.8	10.1	8.7	6.5	4.1
	10	11.4	11.1	10.9	10.2	8.7	6.5	4.0
	25	11.7	11.4	11.2	10.4	8.8	6.5	3.8
	50	12.2	12.0	11.7	10.8	9.0	6.5	3.4
	75	12.8	12.5	12.2	11.2	9.3	6.4	3.0
	99	13.2	12.9	12.5	11.5	9.4	6.3	2.8

(b)

		Solar Radiation Reduction (%)						
		0	5	10	25	50	75	99
Wind Speed Reduction (%)	0	-375.2	-342.7	-314.7	-226.4	-98.8	7.2	72.6
	5	-373.3	-342.5	-312.3	-224.6	-98.9	6.4	70.4
	10	-371.3	-341.2	-310.4	-224.6	-95.8	6.7	67.9
	25	-362.6	-332.9	-303.2	-216.8	-90.2	5.4	60.5
	50	-343.3	-313.9	-284.9	-201.7	-82.9	3.0	46.8
	75	-312.5	-287.2	-258.4	-181.7	-70.0	1.0	31.0
	99	-286.2	-262.6	-237.1	-161.9	-58.1	-1.1	18.7

Table S 3-3 – (a) longest duration of stratification (days), (b) stratification onset (Julian day) and (c) stratification overturn (Julian day) for a selection of wind speed and solar radiation reductions. The baseline scenario is indicated in bold font. 1:1 wind speed and solar radiation reductions are shaded.

(a)

		Solar Radiation Reduction (%)						
		0	5	10	25	50	75	99
Wind Speed Reduction (%)	0	214	212	205	165	159	101	0
	5	217	215	215	181	162	113	0
	10	219	217	217	209	167	118	0
	25	222	222	221	215	175	132	0
	50	248	245	242	231	196	149	13
	75	265	264	260	248	219	150	38
	99	268	266	265	252	226	149	0

(b)

		Solar Radiation Reduction (%)						
		0	5	10	25	50	75	99
Wind Speed Reduction (%)	0	102	103	108	142	145	175	–
	5	102	103	103	132	145	175	–
	10	102	103	103	109	145	175	–
	25	101	101	102	108	144	173	–
	50	88	90	92	102	130	166	93
	75	80	81	84	92	115	164	87
	99	78	80	81	92	109	163	–

(c)

		Solar Radiation Reduction (%)						
		0	5	10	25	50	75	99
Wind Speed Reduction (%)	0	315	314	312	306	303	275	–
	5	318	317	317	312	306	287	–
	10	320	319	319	317	311	292	–
	25	322	322	322	322	318	304	–
	50	335	334	333	332	325	314	97
	75	344	344	343	339	333	313	124
	99	345	345	345	343	334	311	–

Table S 3-4 – (a) annual mean mixed depth (m) and (b) mean mixed depth during the stratified period (m) for a selection of wind speed and solar radiation reductions. The baseline scenario is indicated in bold font. 1:1 wind speed and solar radiation reductions are shaded.

(a)

		Solar Radiation Reduction (%)						
		0	5	10	25	50	75	99
Wind Speed Reduction (%)	0	24.7	25.0	25.4	27.2	29.6	35.1	42.0*
	5	23.9	24.2	24.4	26.0	29.1	34.2	42.0*
	10	23.2	23.6	23.8	25.0	28.5	33.6	42.0*
	25	21.5	21.7	22.1	23.1	26.9	31.5	42.0*
	50	18.1	18.6	18.9	20.2	23.3	28.6	40.2
	75	15.6	15.8	16.2	17.4	20.5	27.3	38.2
	99	14.8	15.1	15.2	16.7	19.3	27.0	42.0*

*The lake is fully mixed when the mixed depth is 42 m

(b)

		Solar Radiation Reduction (%)						
		0	5	10	25	50	75	99
Wind Speed Reduction (%)	0	12.4	12.8	13.0	12.6	13.6	17.0	42.0*
	5	11.8	11.8	12.2	12.8	12.9	17.0	42.0*
	10	10.9	11.1	11.4	12.3	12.5	16.0	42.0*
	25	8.7	8.9	9.1	9.9	10.5	13.1	42.0*
	50	7.1	7.1	7.2	7.5	8.1	9.2	9.1
	75	5.7	5.8	5.8	5.8	6.2	6.2	5.0
	99	5.0	5.1	5.1	5.3	5.3	5.6	42.0*

*The lake is fully mixed when the mixed depth is 42 m

Table S 3-5 – Monthly mean mixed depth (m) for summer and autumn months with driving factor reductions (1:1) of 10%, 15% and 90% in comparison to the water body with no FPV.

Month	No FPV	10	50	90
June	3.9	3.5	2.7	42.0*
July	5.7	5.0	3.0	32.6
August	11.2	9.7	5.4	2.6
September	16.0	13.9	8.6	16.7
October	21.4	18.8	12.6	40.8

*The lake is fully mixed when the mixed depth is 42 m

Chapter 4 – Floating solar panels on reservoirs impact phytoplankton populations: a modelling experiment

Giles Exley, Trevor Page, Stephen J. Thackeray, Andrew M. Folkard, Raoul-Marie Couture, Rebecca R. Hernandez, Alexander E. Cagle, Kateri R. Salk, Lucie Clous, Peet Whittaker, Michael Chipps, Alona Armstrong

In submission to the Journal of Environmental Management.

4.1 Highlights

- Use of a lake model to simulate floating solar on lakes and reservoirs
- Floating solar coverage influences phytoplankton response
- Reduced phytoplankton biomass offsets changes to species composition
- Within water body deployment location significantly impacts water body response
- Simulations provide valuable insight to inform floating solar deployment decisions

4.2 Abstract

Floating solar photovoltaic (FPV) deployments are increasing globally as the switch to renewable energy intensifies, representing a considerable water surface transformation. FPV installations can potentially impact ecosystem function, either positively or negatively. However, these impacts are poorly resolved given the challenges of collecting empirical data for field or modelling experiments. In particular, there is limited evidence on the response of phytoplankton to changes in water body thermal dynamics and light climate with FPV. Given the importance of understanding phytoplankton biomass and species composition for managing ecosystem services, we use an uncertainty estimation approach to simulate the effect of FPV coverage and array siting location on a UK reservoir. FPV coverage was modified in 10% increments from a baseline with 0% coverage to 100% coverage for three different FPV array siting locations based on reservoir circulation patterns. Results showed that FPV coverage significantly impacted thermal properties, resulting in highly variable impacts on phytoplankton biomass and species composition. The impacts on phytoplankton were

often dependent on array siting location as well as surface coverage. Changes to phytoplankton species composition were offset by the decrease in phytoplankton biomass associated with increasing FPV coverage. We identified that similar phytoplankton biomass reductions could be achieved with less FPV coverage by deploying the FPV array on the water body's faster-flowing area than the central or slower flowing areas. The difference in response dependent on siting location could be used to tailor phytoplankton management in water bodies. Simulation of water body-FPV interactions efficiently using an uncertainty approach is an essential tool to rapidly develop understanding and ultimately inform FPV developers and water body managers looking to minimise negative impacts and maximise co-benefits.

Keywords: floating solar, renewable energy, water quality, phytoplankton, ecosystem impact, *MyLake*

4.3 Introduction

Falling costs and the drive to decarbonise global energy supplies have led to widespread uptake of renewable energy sources, including solar photovoltaic (PV) technology. Solar PV has traditionally been dominated by ground- and rooftop- mounted installations. However, since 2007, water-deployed floating solar photovoltaics (FPV) have emerged as an alternative, especially in land-scarce areas (Cagle et al., 2020). FPV deployment has been rapid, with over 2.6 gigawatts of installed capacity globally (Haugwitz, 2020) and an anticipated annual growth rate of 28.9% between 2020 and 2027. Estimates show that there is technical potential for FPV to produce almost 10% of current national generation in the United States (Spencer et al., 2019), based on a water surface coverage of 27% on suitable water bodies. At a continental scale, FPV covering less than 1% of the surface of African hydropower dams could equal the generation from existing hydropower dams, the largest source of renewable energy across the continent (Sanchez et al., 2021).

FPV is comprised of PV modules attached to a series of floats moored on the surface of a water body (Sahu et al., 2016). Host water bodies tend to be artificial (e.g. raw water reservoirs) and may be used for drinking water provision, irrigation or hydroelectric power generation (Momayez et al., 2009; Lee et al., 2020; Exley et al., 2021b). Deploying

PV panels on water delivers enhanced performance and electricity generation over ground-based PV due to the cooling effect of the hosting water body (Choi et al., 2013; Sacramento et al., 2015; Yadav et al., 2016; Oliveira-Pinto and Stokkermans, 2020) and reduces land use and land-cover change for renewable energy (Cagle et al., 2020). FPV is deployed at a range of coverages, that is, the percentage of the water surface transformed to host FPV relative to the water body area. Coverage depends on the size of the host water body, the FPV design and the rated capacity of the installation (Exley et al., 2021a).

FPV represents an unprecedented change in the use of artificial water bodies. Understanding impacts is critical as water bodies provide numerous essential ecosystem goods and services, including water for consumption, water quality regulation, and supporting biodiversity (Maltby et al., 2011; Reynaud and Lanzanova, 2017; Grizzetti et al., 2019). Impacts on the host water body could be significant, as light intensity and wind shear will be modified by the shading and sheltering effect of an FPV installation (Armstrong et al., 2020; Haas et al., 2020). Consequently, there is a pressing need to understand and predict the effects of FPV on water body processes and functions (Lee et al., 2020; Stiubiener et al., 2020; Zhang et al., 2020; Gorjian et al., 2021; Ziar et al., 2021). In particular, understanding changes to phytoplankton is critical, given their role as primary producers in aquatic ecosystems (Reynolds, 2006), the increased likelihood of harmful algal blooms under climate change (Ho et al., 2019), and the subsequent implications for recreation and potable water supply (Chapra et al., 2017). Moreover, surface cover proxies for FPV (e.g. ice) suggest that deployments could alter physicochemical habitat conditions in a way that would affect phytoplankton biomass and species composition (Wright, 1964; Danilov and Ekelund, 2001; Lenard and Wojciechowska, 2013; Yamamichi et al., 2018; Exley et al., 2021b).

Given the limited understanding of water body response to FPV deployment, investigations that rapidly develop knowledge should be prioritised. *In-situ* monitoring studies have quantified the impact of FPV installations on water temperatures (de Lima et al., 2021) and aquatic plants (Ziar et al., 2021). However, comprehensive empirical studies are resource-intensive and largely impractical when considering multiple deployment scenarios (Meyer et al., 2009; Janssen et al., 2015). Several studies have hypothesised potential effects of FPV, but these are often conflicting given the complexity of water body functioning. For example, it is claimed that water column shading beneath FPV installations will reduce phytoplankton growth (Armstrong et al., 2020; Lee et al., 2020). Yet, an evidence review of natural and human-made water surface covers found that surface cover may cause a shift to low-light adapted nuisance species, rather than a reduction in total biomass (Yamamichi et al., 2018; Exley et al., 2021b). Numerical modelling 'experiments' provide a less time- and resource-demanding alternative for rapidly testing multiple hypotheses on potential FPV interactions without being limited to a single FPV design, sited on a specific part of a single water body for a limited time. However, given the limited empirical observations so far and limited data to parameterise models, conventional modelling approaches may be unsuitable (Page et al., 2018). Therefore, approaches that can account for the uncertainty associated with sparse input parameters or forcing data are necessary.

Our overarching aim was to determine if FPV coverage and siting location, based on areas of differing circulation, influence phytoplankton biomass and species composition in a reservoir. We used an extended version of the *MyLake* model with enhanced phytoplankton representation to simulate FPV water quality impacts across discrete zones of a water body. Moreover, we employed an uncertainty estimation approach, a practical solution to overcome the problems associated with limited input data, model parameterisation and validation of simulated output. We also discuss the implications of our findings for water body management and the application of the expanded model for future FPV deployments.

4.4 Methodology

4.4.1 *MyLake* FPV model

To determine if FPV array siting location affects water body thermal properties, phytoplankton biomass and functional-type dynamics, we extended an existing open-source lake model, *MyLake v2* (Markelov et al., 2019). Full details on the original *MyLake* can be found in Saloranta and Andersen (2007) and the accompanying user manual (Saloranta and Andersen, 2004).

4.4.1.1 MyLake – existing model description

MyLake v2 (Markelov et al., 2019) is a one-dimensional process-based model capable of simulating the daily vertical distributions of water body temperature, phytoplankton, and dissolved and particulate substances, as well as interactions at the sediment-water interface (Saloranta and Andersen, 2007). *MyLake* has been successfully applied to various projects as a standalone simulation tool. For example, assessing ice regime (Livingstone and Adrian, 2009), lake thermodynamics (Woolway et al., 2017a), greenhouse gas emissions (Kiuru et al., 2018), light dynamics (Pilla and Couture, 2021) and predicting cyanobacterial blooms (Moe et al., 2016).

Like many one-dimensional lake models (e.g. PROTECH; Reynolds et al. (2001)), *MyLake* computes horizontal layer volumes from interpolated water body bathymetric data. In the original version of *MyLake*, the model could simulate a maximum of two species or functional groups of phytoplankton, with population dynamics controlled by phosphorus (P) limitation, light requirements, and loss processes (see Table 4-1 for a complete list of modifiable phytoplankton parameters). Nitrogen (N) and silica (Si) species, as state variables, were added in v2 (Markelov et al., 2019). N-limited phytoplankton growth was incorporated in a recent application (Salk et al., 2022).

Table 4-1 – MyLake Parameters describing phytoplankton functional traits. PAR is photosynthetically active radiation.

Parameter	Description
PAR saturation level for growth (mol-quanta m^{-2} s^{-1})	Controls the light-limitation of growth
Optical cross section of chlorophyll-*a* (m^{-2} mg^{-1})	Specifies self-shading contribution
Loss rate at 20 °C (day^{-1})	Overall loss rate (includes death, grazing etc. but not settling losses)
Settling velocity (m day^{-1})	Phytoplankton-specific settling rates
Specific growth rate at 20 °C (day^{-1})	Phytoplankton-specific maximum growth rates – modified by temperature, light and nutrient availability
Half saturation growth P concentration (mg m^{-3})	Controls shape of growth curve based upon P concentration
Half saturation growth N concentration (mg m^{-3})	Controls shape of growth curve based upon N concentration
Half saturation growth Si concentration (mg m^{-3})	Controls shape of growth curve based upon Si concentration
If phytoplankton are N-Limited	Allows specification for N-fixing phytoplankton

If phytoplankton are Si-Limited	Allows specification of Si requirement (e.g. diatoms and chrysophytes)
Scaling factor for inflow concentration of chlorophyll-*a* (-)	Distributes inflow chlorophyll-*a* across functional groups simulated

4.4.1.2 MyLake – updated model description

The assumption of lateral homogeneity in *MyLake*, inherent to most one-dimensional models, limits the model's adaptability for simulating different water column 'zones'. Consequently, in order to model the impacts of FPV we adapted and extended *MyLake* to enable simulation of the effects of varying FPV coverage on water bodies. Moreover, given the importance of phytoplankton on water supply reservoirs where FPV are often located, we enhanced the phytoplankton functionality.

4.4.1.2.1 *Multiple tanks and exchanges between tanks*

To enable the explicit simulation of FPV installations on different types of water bodies and differently functioning 'zones' of water bodies, the *MyLake* model was extended to represent water bodies in a quasi-two-dimensional way, an approach successfully applied with other freshwater models (e.g. de la Fuente and Niño, 2008; Zhang and Rao, 2012; Dimitriou et al., 2017). Specifically, the original one-dimensional (one '*tank*') model structure was replicated into '*n tanks'* (see supplementary information, section 4.8.1).

The quasi-two-dimensional functionality permits each tank to be independent, allowing for variation in water body characteristics, such as depth and flow, and spatial characteristics, such as littoral and pelagic zones. Alternatively, the functionality permits the simulation of covered and uncovered zones of a water body with FPV. Flows and exchanges are specified using an eddy diffusion matrix, which governs the amount of lateral mixing between contiguous tanks and an advection matrix that specifies flows between tanks (e.g. to represent internal water body circulation patterns). While the updated *MyLake* model can simulate an unrestricted number of tanks, the computational burden and availability of data for parameterising appropriate advection and diffusion matrices could be limiting. Consequently, the number of tanks should be as parsimonious as possible given the simulation requirements (see supplementary information, section 4.8.2).

4.4.1.2.2 *Phytoplankton growth module*

To investigate phytoplankton species composition in response to FPV and the risk to water quality, we updated the *MyLake* phytoplankton growth module to simulate an unrestricted number of phytoplankton species (or groups) with different functional behavioural traits. Here, we simulated phytoplankton in functional groups, as widely used in modelling applications, to overcome the difficulty of specifying individual species parameters (Shimoda and Arhonditsis, 2016). Specifically, we modelled diatoms, green algae and cyanobacteria, enabled by generic representations of their non-taxonomical traits known to dictate behaviour (Salmaso et al., 2015), such as growth, loss and nutrient uptake (Reynolds et al., 2002).

As nutrient limitation is a primary determinant of the abundance and species composition of phytoplankton in water bodies (O'Neil et al., 2012) we increased the model growth equations from two to three (Equation 4-1, Equation 4-2, Equation 4-3). Specifically, Si species were linked to the phytoplankton dynamics equations to allow the simulations of diatoms (Harrison et al., 2012), in addition to the original phosphorus uptake module and the recently incorporated N-limited growth module (Salk et al., 2022) (see Table S 4-3 for a complete list of phytoplankton parameters). Consequently, there are now three phytoplankton growth equations:

Equation 4-1

P limited
$$\mu = \mu_{max} . \left(\frac{S_1}{(S_1 + K_{S_1})}\right) . T_f . L_f$$

Equation 4-2

P and N limited
$$\mu = \mu_{max} . \left(\frac{S_1}{(S_1 + K_{S_1})} \cdot \frac{S_2}{(S_2 + K_{S_2})}\right) . T_f . L_f$$

Equation 4-3

P, N and Si limited
$$\mu = \mu_{max} . \left(\frac{S_1}{(S_1 + K_{S_1})} \cdot \frac{S_2}{(S_2 + K_{S_2})} \cdot \frac{S_3}{(S_3 + K_{S_3})}\right) . T_f . L_f$$

where μ = phytoplankton species growth rate on a given day (1/day), μ_{max} is the maximum phytoplankton growth rate at 20 ^oC; $.T_f$ (-) is a water temperature modifier; $.L_f$ (-) is a light modifier; S_1 is phosphorus concentration (mg m^{-3}), S_2 is nitrogen concentration (mg m^{-3}), S_3 is silica concentration (mg m^{-3}), and K_{S_x} (mg m^{-3}) is the half molar saturation level for each nutrient (see Table S 4-3 for full definitions).

4.4.1.2.3 *Initial model testing (functionality)*

Testing examined the functionality of multiple tank configurations and additional phytoplankton functional group representation using data from Lake 227 (Ontario, Canada), Lake Vansjø (Norway) and subsequently Thames Water's Queen Elizabeth II reservoir (outlined below; Section 4.4.2.1). We tested for internal consistencies (e.g. mass-balance conservation), appropriate phytoplankton functional group behaviour and dynamics (e.g. response to nutrient concentrations (Klausmeier and Litchman, 2001) and functional group succession) and the sensitivity of model output to the number and configuration of tanks (see supplementary information, section 4.8.2, for details).

4.4.2 Modelling methodology

We used the expanded model to simulate the effect of FPV on water quality in the Queen Elizabeth II (QEII) reservoir. FPV are typically deployed on raw (untreated) water reservoirs, irrigation ponds and other artificial water bodies (Exley et al., 2021b), which typically have less extensive data than natural water bodies instrumented for research. Consequently, we took an uncertainty approach, specifically the Generalised Likelihood Uncertainty Estimation (GLUE) procedure (Beven and Binley, 1992) to account for the limited data.

4.4.2.1 *Study location*

The QEII reservoir is in south-west London (51° 23′ 27″ N, 0° 23′ 32″ W, surface area: 128 hectares). The raw water reservoir has a maximum depth of 17.8 m and a maximum capacity of 19.6 million cubic meters. The reservoir is supplied with nutrient-rich water from the River Thames (Reynolds et al., 2005), pumped via three inlets on the reservoir bed, one to the west and one in each of the two southern corners. The reservoir outlet is situated in the north-eastern corner (Figure 4-1). During the study year, 2018, the QEII reservoir had a mean hydraulic residence time of 44 days (Ta, 2019). Reservoir volume ranged from > 95% full between January to early May, before being drawn down over the summer and autumn to 73% volume in early November. Reservoir volume then returned to > 95% at the end of 2018. In 2016, a 6.3 MW capacity FPV installation was deployed on the QEII reservoir, covering ~4.5% of the reservoir's surface when full.

4.4.2.2 Data inputs

4.4.2.2.1 *Forcing inputs*

The QEII reservoir was modelled on a daily time step for one-year to demonstrate model application, using data from 2018. As monitoring of the QEII reservoir is conducted at the reservoir outlet, inflow nutrient concentrations were obtained from two monitoring stations on the River Thames situated upstream (Wey tributary; ~5.5 km) and downstream (Teddington Weir; ~11.5 km) of the QEII reservoir inlet (Environment Agency, 2018). Samples were taken approximately monthly and were linearly interpolated to obtain mean daily values throughout 2018. Inflow water temperatures were approximated from observed in-reservoir water temperatures. Daily outflow data provided by the reservoir operator were used as a proxy for inflow volume. In the absence of on-site meteorological measurements, global radiation, cloud cover, wind speed, air temperature, relative humidity, air pressure and rainfall observations from Heathrow Airport (10.5 km to the north) for 2018 were used (Met Office, 2019). Bathymetry of the QEII reservoir was digitised to 1 m intervals from a survey provided by the reservoir operator.

4.4.2.2.2 *Data for evaluation of model performance*

Observed water temperature and total chlorophyll-*a* data provided by the reservoir operator was used for model calibration and uncertainty estimation. Typically, these samples were collected weekly at the reservoir outlet at depths of 1, 3, 5, 7, 9, 11, 13 and 15 m. Weekly phytoplankton speciation, analysed by the reservoir operator, was derived from an integrated sample of the upper 1 m of the reservoir and recorded based on the cell count by ascribing a rating on an ACFOR (Abundant, Common, Frequent, Occasional, Rare) scale (see supplementary information, section 4.8.3, for further details). Six functional groups were simulated to broadly reflect the phytoplankton species composition observed in the QEII reservoir during 2018, separated by grazed and ungrazed groupings. The groups represent the broad functional trait differences, including grazing pressures (represented by loss rate), size, growth rate, their light requirement for growth and settling velocity. The six functional groups were reported only as *diatoms*, *green algae* and *cyanobacteria* in the following analyses (Table 4-2), as the grazed and ungrazed groupings were combined for each group.

Table 4-2 – Nominal phytoplankton functional groups used and descriptive functional traits. See supplementary information, section 4.8.3, for the ranges of each sampled parameter.

Phytoplankton functional group	Nutrient limited			Size	Growth Rate	Light requirement for growth	Settling velocity	Loss rate
	P	N	Si					
Diatoms – *ungrazed*	✓	✓	✓	Large	Slow	Low	High	−
Diatoms – *grazed*	✓	✓	✓	Large	Slow	Low	High	+
Green/other algae – *ungrazed*	✓	✓		Small	Fast	Medium	Low	−
Green/other algae – *grazed*	✓	✓		Small	Fast	Medium	Low	+
Cyanobacteria – *ungrazed*	✓			Small/ medium	Medium	Medium	Very low	−
Cyanobacteria – *grazed*	✓			Small/ medium	Medium	Medium	Very low	+

+: Increased to reflect grazing losses; −: Reduced to reflect no grazing losses

4.4.2.3 *Model geometry and simulations*

4.4.2.3.1 *Tank configuration*

In this study, the new multi-tank functionality of the model was used to represent discrete zones of internal circulation. Tank configuration was based on a detailed study of internal circulation in the QEII reservoir for 2018 (Ta, 2019) and testing of tank configurations (see supplementary information, section 4.8.2). The baseline model was assigned two tanks, one relatively short residence time, *faster-flowing tank* (70% of QEII surface area) and one comparatively longer residence time, s*lower-flowing tank* (30% of QEII surface area). The tanks mimic the hydrologic behaviour of the QEII reservoir, namely the short-circuiting of flow between the reservoir inlets and outlet. The existing FPV array is positioned on the slower-flowing tank (Section 4.4.2.1; Figure 4-1). The inflow and outflow of the reservoir were located in the faster-flowing tank. The distribution matrices described exchanges between the faster-flowing tank and the slower-flowing tank; lateral eddy diffusion (set at 2.5% of tank volume) and advection (set at 2.5% of tank volume) (see supplementary information, section 4.8.2, for further details).

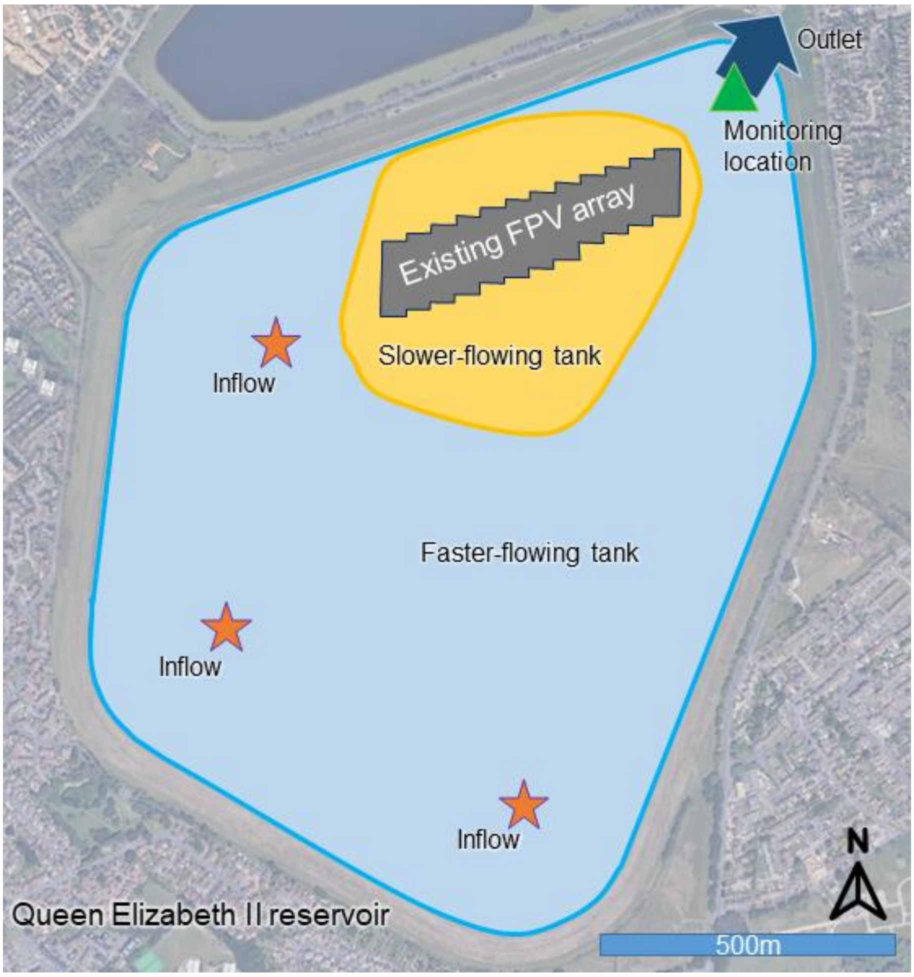


Figure 4-1 – Conceptual Baseline 'tank' structure for QEII during 2018. Satellite image from Google Earth.

4.4.2.3.2 *Identification of baseline model simulations*

Acceptable baseline simulation results and parameter sets were identified by comparing model output from multiple model runs with observed data (total chlorophyll-*a*, surface temperature, stratification pattern and phytoplankton functional group proportions). Parameter ranges, comprised of all physically reasonable values for each parameter (see supplementary information, section 4.8.3), were sampled 8,000 times using a Monte Carlo strategy to limit bias within the parameter sets. Each Monte Carlo sample provided a unique set of parameters to run 8,000 simulations. Each of the simulations underwent the GLUE procedure (Beven and Binley, 1992), where formalised Limits of Acceptability (LoA) were developed and used as acceptance criteria to account for the significant uncertainties associated with modelling environmental systems (see supplementary information, section 4.8.4, for further details).

LoA were applied in the strictest sense for chlorophyll-*a* and mixed depth: any simulations that fell outside the specified limits were rejected and not used in the analyses. The remaining variables were used solely to provide additional confidence weightings. Confidence weightings (L) for accepted simulations were calculated using fuzzy weighting functions and were combined to give an overall weighting for each simulation. Chlorophyll-*a* (L_{Chl}), mixed depth (L_{mxd}) and water temperature (L_{wt}) were equally weighted in the combined overall goodness of fit weighting (Wt). Phytoplankton functional groups, where L_D, L_G and L_C are the weighting for diatoms, green algae, and cyanobacteria, respectively, had a weighting of one-third to prevent over-constraint on functional groups (Equation 4-4).

Equation 4-4

$$Wt = [(L_{Chl} + L_{mxd} + L_{wt} + (L_D.0.33) + (L_G.0.33) + (L_C.0.33))]$$

As all acceptable simulations are deemed to represent the system behaviour (given the available data), they are all used to represent the baseline. However, as each acceptable simulation is associated with a goodness of fit weighting, which is propagated to the final results, each acceptable simulation contributes differently. Using all the acceptable simulations in this way explicitly propagates all known modelling uncertainties to final modelling results. The implementation of FPV deployment in the model took the form of a modification to each of the acceptable parameter sets to represent the solar array associated with the scenario of interest.

4.4.2.3.3 *FPV deployment scenarios*

Three 'deployment scenarios' were run to investigate the impact of array siting location on water body response (Table 4-3). Each scenario was run multiple times to simulate varying degrees of FPV coverage – the 'coverage increments'. The coverage increment represents the percentage of the reservoir's total surface area covered by FPV, accounting for the existing 4.5% coverage of the presently deployed array (see Section 4.4.2.1). In the following scenarios, we use coverage increments of 10% from a baseline of 0% coverage to complete reservoir coverage (100%).

Table 4-3 – Scenarios and a summary of the deployment configuration.

Scenario	Description
Baseline	The reservoir simulated with no additional FPV coverage (includes the existing 4.5% coverage of the presently deployed array) – shown as 0% coverage.
Scenario-Fast	FPV installation initially deployed on the faster-flowing tank, an area of the reservoir with a shorter-residence time. Once the FPV installation exceeds the area of the faster-flowing tank the FPV array is deployed upon the slower-flowing tank (Figure 4-1).
Scenario-Slow	FPV installation initially deployed on the slower-flowing tank, an area of the reservoir with a longer-residence time. Once the FPV installation exceeds the area of the slower-flowing tank the FPV array is deployed upon the faster-flowing tank (Figure 4-1).
Scenario-Central	Central siting of FPV installation. Initially the array is deployed on the faster-flowing tank, as the larger of the two tanks (Figure 4-1). Once the remaining uncovered area of the faster-flowing tank is equal in area to the slower-flowing tank, the deployment of the array is split equally between each tank.

- Each deployment scenario was simulated with a range of FPV ‘coverage increments’ from 0% coverage (baseline) to 100% in 10% increments.

4.4.2.3.4 *Modelling assumptions and sources of uncertainty*

Each model run, in terms of the deployment scenario and coverage increment, was based on a set of assumptions to represent the water body and approximate the effects of FPV coverage. At present, there are no published values for the effect of FPV on air temperature, wind speed and incoming solar radiation at the air-water interface. The effect on each driver is likely modified depending on system design, such as transparency of the PV module, airflow beneath the floating array and orientation of the array (Armstrong et al., 2020; Exley et al., 2021a; Ziar et al., 2021). For this study, the effects of an array were estimated from unpublished observations made at an FPV installation (see supplementary information, section 4.8.5, for methods) and published observations made at a ground-based installation (Armstrong et al., 2016). Based on the results of these preliminary observations, we assumed that between the water’s surface and the underside of the PV module; air temperature is warmed by 8%, incoming solar radiation is reduced by 94%, and wind speed is reduced by 95%. All scenarios are also based on the likely assumptions that the functional phytoplankton groups adequately represent the phytoplankton community observed in the QEII reservoir and that the initial phytoplankton community composition (i.e., relative proportions of taxa) were set to be equal on the first day of each simulation to permit an equal chance of proliferation.

4.4.2.4 Model output analysis

To summarise the impact of varying FPV coverage and siting location on phytoplankton biomass and species composition, we compared model outputs from each scenario against the baseline (Table 4-3). We analysed the output from the faster-flowing tank, as this is the tank that feeds the water treatment works. Given the plethora of data outputted, we focussed on variables influencing phytoplankton biomass and species composition, including surface water temperature at 1 m and stratification metrics. To represent phytoplankton biomass and species composition, we used total chlorophyll-*a* concentration and the proportions of each phytoplankton functional group as a proportion of total chlorophyll-*a*, both at 1 m depth. The proportions of phytoplankton functional groups are presented as relative, not absolute values for visual clarity.

Given the use of the GLUE methodology, each scenario has the outputs from several model simulations. To capture the variability in outcomes, thus representing the uncertainty, we use the median, 2.5th and 97.5th percentiles, thus providing the average outcome and the 95% confidence interval. To explore the impacts on the annual minimum (T_{min}) and maximum (T_{max}) water temperature and maximum total chlorophyll-*a* concentration, we use the mean of each based on a ten-day window defined by the baseline model runs. Stratification was determined using a threshold density gradient of 0.1 kg m^{-3} m^{-1} between adjacent layers (Gray et al., 2020). Two metrics were used to summarise stratification duration. These were continuous stratification, the longest period of stratification in each simulation, and cumulative stratification duration, the total number of stratified days during the one-year simulation period. Stratification onset and overturn were defined as the first and last day of the longest period of continuous stratification, respectively.

4.5 Results

4.5.1 Simulations within the limits of acceptability

Seventy-five parameter sets were within the LoA for all simulations; the remaining 7925 parameter sets were rejected and not used in the subsequent analyses. Given the limited input data and strict inclusion criteria applied, most excluded parameter sets were rejected based on their representation of total chlorophyll-*a* (Figure 4-2), functional groups and mixed depth. The model simulated water temperature within the LoA for most parameter sets (< 95%). The goodness of fit weighting for the accepted parameter sets ranged from 80.62 to 84.01, of a maximum possible weighting of 204 (determined by the number of observations available for the QEII reservoir).

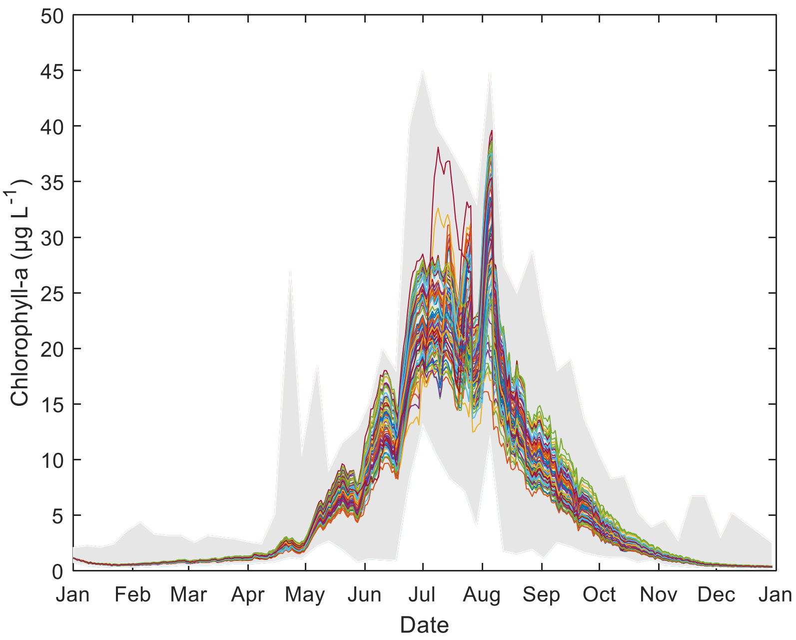


Figure 4-2 – Simulated chlorophyll-a (coloured lines) samples within the limits of acceptability (grey shaded area).

4.5.2 Response of thermal properties

FPV coverage cooled median surface water temperatures throughout the year (Figure S 4-10, Figure S 4-11, Figure S 4-12). However, on a small number of days between mid-spring and early summer, the 10% coverage increment resulted in slightly warmer (< 0.6 °C) surface water temperatures than the baseline (i.e. no additional FPV coverage) in Scenario-Fast and Scenario-Central for nine days. Similarly, at 10 to 30% coverage in Scenario-Slow, there were ten days when FPV coverage warmed median surface water temperatures (< 0.5 °C) compared to the baseline.

For all scenarios, median T_{max} and T_{min} were reduced with increasing FPV coverage, based on the mean of a window (± five days), defined by the baseline model runs. FPV deployment on the fast-flowing tank (Scenario-Fast) saw a comparatively quick decline in T_{max} and T_{min} with increasing FPV coverage. Median T_{max} decreased on average 0.55 ± 0.09 °C (mean difference ± *SD*; hereafter unless stated) per 10% coverage increment for FPV coverages up to 70% (i.e. when the FPV encroached on the slower flowing tank). Median T_{min} decreased by 0.20 ± 0.11 °C per 10% coverage increment up to 70% coverage. The rate was reduced once the array encroached on the slower-flowing tank (FPV coverages greater than 70%). T_{max} decreased by 0.16 ± 0.03 °C per 10% coverage increment and T_{min} decreased by 0.02 ± 0.004 °C per 10% coverage increment (Figure 4-3a and b).

Deployment on the slower-flowing tank (Scenario-Slow) initially caused a slower decline in median T_{max} and T_{min}, 0.15 ± 0.04 °C and 0.02 ± 0.01 °C, respectively, per 10% coverage increment up to 30%, than in Scenario-Fast. After the FPV encroached on the faster-flowing area (above 30% coverage), T_{max} decreased by 0.56 ± 0.15 °C, and T_{min} decreased by 0.20 ± 0.11 °C, per 10% coverage increment. In contrast, median T_{max} declined linearly by 0.44 ± 0.08 °C for each 10% coverage increment when the array was located centrally on the reservoir (Scenario-Central; Figure 4-3a). T_{min} for Scenario-Central reduced by 0.14 ± 0.06 °C for each 10% increase in FPV coverage. There was increasing divergence between the lower (2.5th) and upper (97.5th) percentile at higher FPV coverages (Figure 4-3b). For example, at 10% coverage the range between the lower and upper percentile was 0.65 °C, this increased to 0.70 °C at 50% and 0.94 °C at 90% FPV coverage.

In response to increasing array coverage, continuous and cumulative stratification duration decreased rapidly when the array was deployed on the faster-flowing tank or centrally (Scenario-Fast and Scenario-Central; Figure 4-3c and d). Maximum stratification duration was up to 22 days longer in Scenario-Slow than under Scenario-Fast at 30% FPV coverage (Figure 4-3c). Cumulative stratification duration was up to 75 days longer in Scenario-Slow than under Scenario-Fast at 30% FPV coverage (Figure 4-3d). Significant stratification events did not occur in Scenario-Fast and -Central when array coverage exceeded 50% and in Scenario-Slow when coverage exceeded 70%

(Figure 4-3c and d). See supplementary information, section 4.8.9, for the number of stratified simulations at each FPV coverage.

The relationships between FPV coverage and stratification onset and overturn were weaker than for stratification duration (Figure 4-3e and f). Stratification onset generally shifted to later in the year with FPV coverages of up to 40% for Scenario-Fast and -Central (Figure 4-3e). However, some simulations had an earlier onset of stratification at the 10% FPV coverage increment. In Scenario-Slow, stratification onset showed a weak shift to later in the year with FPV coverages of up to 70% (Figure 4-3e). However, a few Scenario-Slow simulations showed an earlier onset at 10 to 40% FPV coverage than the baseline (Figure 4-3e).

The overturn of stratification did not have a clear trend with increasing FPV coverage. However, overall, there was a tendency for overturn to be slightly later for all three scenarios than the baseline (Figure 4-3f). However, a small number of simulations showed earlier overturn than the baseline (Figure 4-3f). For example, at 10 and 20% FPV coverage, the lower extent of the estimated range was earlier than the lower extent of the baseline range for Scenario-Fast and Scenario-Central (Figure 4-3f). Overturn of stratification in Scenario-Slow did not have a clear trend with increasing coverage, although typically it occurred slightly earlier than in Scenario-Fast and Scenario-Central at FPV coverages 20% or greater (Figure 4-3f). Overturn occurred earlier than the baseline in a small number of simulations, for example, at 10 to 40% FPV coverage, when only the slower-flowing tank was covered.

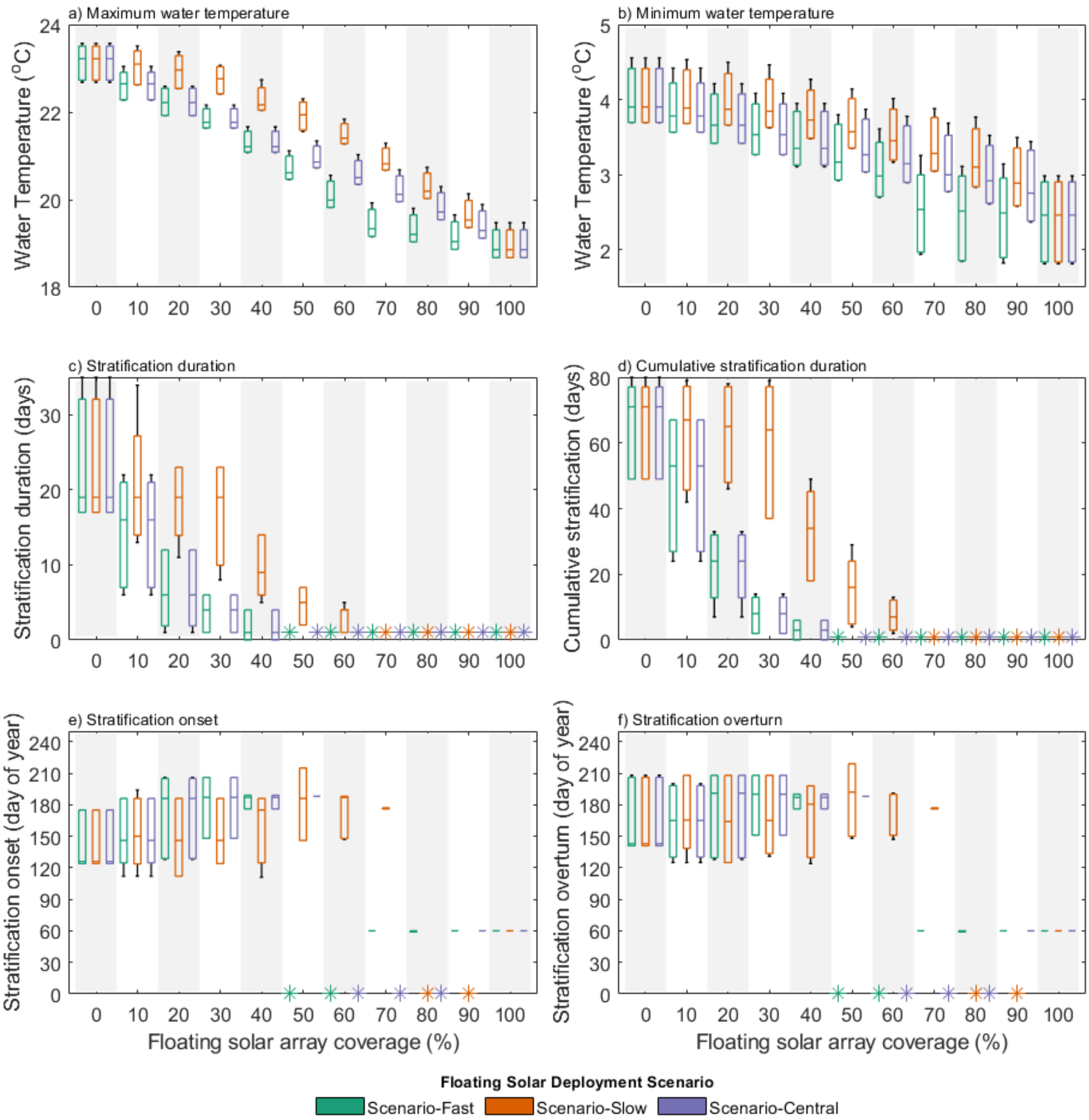


Figure 4-3 – a) Annual maximum and b) minimum water temperature, c) stratification duration, d) cumulative stratification duration, and stratification e) onset and f) overturn day, versus floating solar (FPV) array coverage for each deployment scenario. An asterisk indicates no prolonged stratification event occurred for the simulation. Whiskers represent the minimum and maximum of the simulation results presented. The box represents the 2.5th & 97.5th percentiles, which gives a 95% confidence interval that simulation estimates fall within this range. 0% FPV coverage represents QEII reservoir simulated as a baseline with no additional FPV coverage.

4.5.3 Response of phytoplankton

4.5.3.1 Total chlorophyll-a

In Scenario-Fast and Scenario-Central maximum total chlorophyll-*a* concentration, based on the mean of a window (± five days), defined by the baseline model runs, declined exponentially with increasing FPV coverage (Figure 4-4). For example, in Scenario-Fast, median total chlorophyll-*a* was reduced by 10.21 µg L^{-1} at 10% FPV coverage, 20.40 µg L^{-1} at 50% and 22.09 µg L^{-1} at 90% compared to the baseline scenario.

Each additional 10% coverage increment, up to 60%, reduced median total chlorophyll-*a* by 3.59 ± 1.84 μg L^{-1} on average (Figure 4-4). Coverages exceeding 60% in Scenario-Fast had negligible total chlorophyll-*a* (< 1 μg L^{-1}).

Comparatively, Scenario-Central showed a slightly smaller reduction in median total chlorophyll-*a* concentration than Scenario-Fast. For example, median total chlorophyll-*a* was reduced by 4.01 μg L^{-1} at 10% FPV coverage, 19.69 μg L^{-1} at 50% and 22.01 μg L^{-1} at 90% compared to the baseline scenario. In Scenario-Central, each additional 10% FPV coverage increment, up to 70%, reduced median total chlorophyll-*a* by 3.05 ± 2.11 μg L^{-1} (Figure 4-4). Coverages exceeding 70% in Scenario-Central had negligible total chlorophyll-*a* (< 1 μg L^{-1}).

In Scenario-Slow, total chlorophyll-*a* concentration generally reduced with increasing FPV coverage. However, at lower FPV coverages (10 to 30% coverage) where only the slower-flowing tank was covered, total chlorophyll-*a* simulations showed both increases and decreases from the baseline (Figure 4-4). At 10% FPV coverage, total chlorophyll-*a* was either reduced by up to 5% (0.89 μg L^{-1}; 2.5th percentile) or increased by up to 28% (7.95 μg L^{-1}; 97.5th percentile). At 20% FPV coverage, total chlorophyll-*a* either reduced by up to 15% (2.52 μg L^{-1}; 2.5th percentile) or increased by up to 15% (4.28 μg L^{-1}; 97.5th percentile). At 30% FPV coverage, total chlorophyll-*a* either reduced by up to 19% (3.23 μg L^{-1}; 2.5th percentile) or increased by up to 4% (1.02 μg L^{-1}; 97.5th percentile). Above 30% FPV coverage, when the faster-flowing tank started to be covered, median total chlorophyll-*a* declined on average by 2.87 ± 2.35 μg L^{-1} per 10% additional cover.

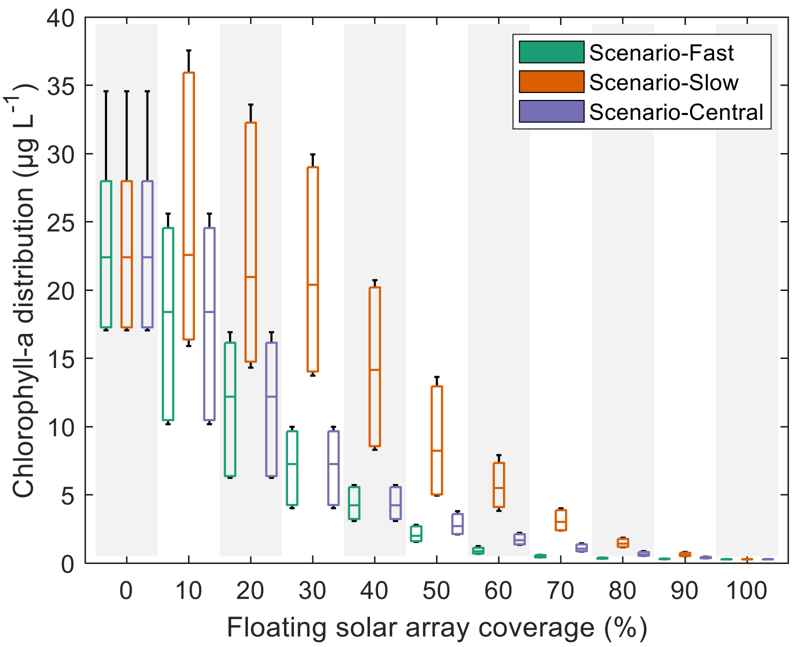


Figure 4-4 – Total chlorophyll-a (based on the mean of a window, ± five days, around the day of maximum total chlorophyll-a) versus floating solar array coverage. Whiskers represent the minimum and maximum of the simulation results presented. The box represents the 2.5th & 97.5th percentiles, which gives a 95% confidence interval that simulation estimates fall within this range. 0% floating solar coverage represents QEII reservoir simulated as a baseline.

4.5.3.2 Annual total chlorophyll-a

Median total chlorophyll-*a* generally reduced with increasing FPV coverage throughout the year for all scenarios (Figure 4-5 and supplementary information, section 4.8.7, for 95% confidence interval). However, on a small number of days between late May and the end of July, median total chlorophyll-*a* was greater than the baseline in Scenario-Slow at 10 to 30% FPV coverage. FPV coverage had the greatest relative impact on median total chlorophyll-*a* at the start of August (Figure S 4-13). For example, at 10% FPV coverage median total chlorophyll-*a* had a relative reduction of 48% (24.38 µg L^{-1}) in Scenario-Fast and Scenario-Central. Whilst in Scenario-Slow, the greatest relative difference for 10% FPV coverage occurred in early June; a 17% (2.09 µg L^{-1}) reduction compared to the baseline scenario. The absolute differences as coverage exceeded 70% in Scenario-Fast and Scenario-Central were relatively small compared to lower coverages when the array was deployed exclusively on the faster-flowing tank (Figure 4-5). The opposite occurred for Scenario-Slow, with coverages up to 30%, the area of the slower-flowing tank, having a small absolute difference with the baseline. The absolute difference increased once the array started to cover the faster-flowing tank (Figure 4-5).

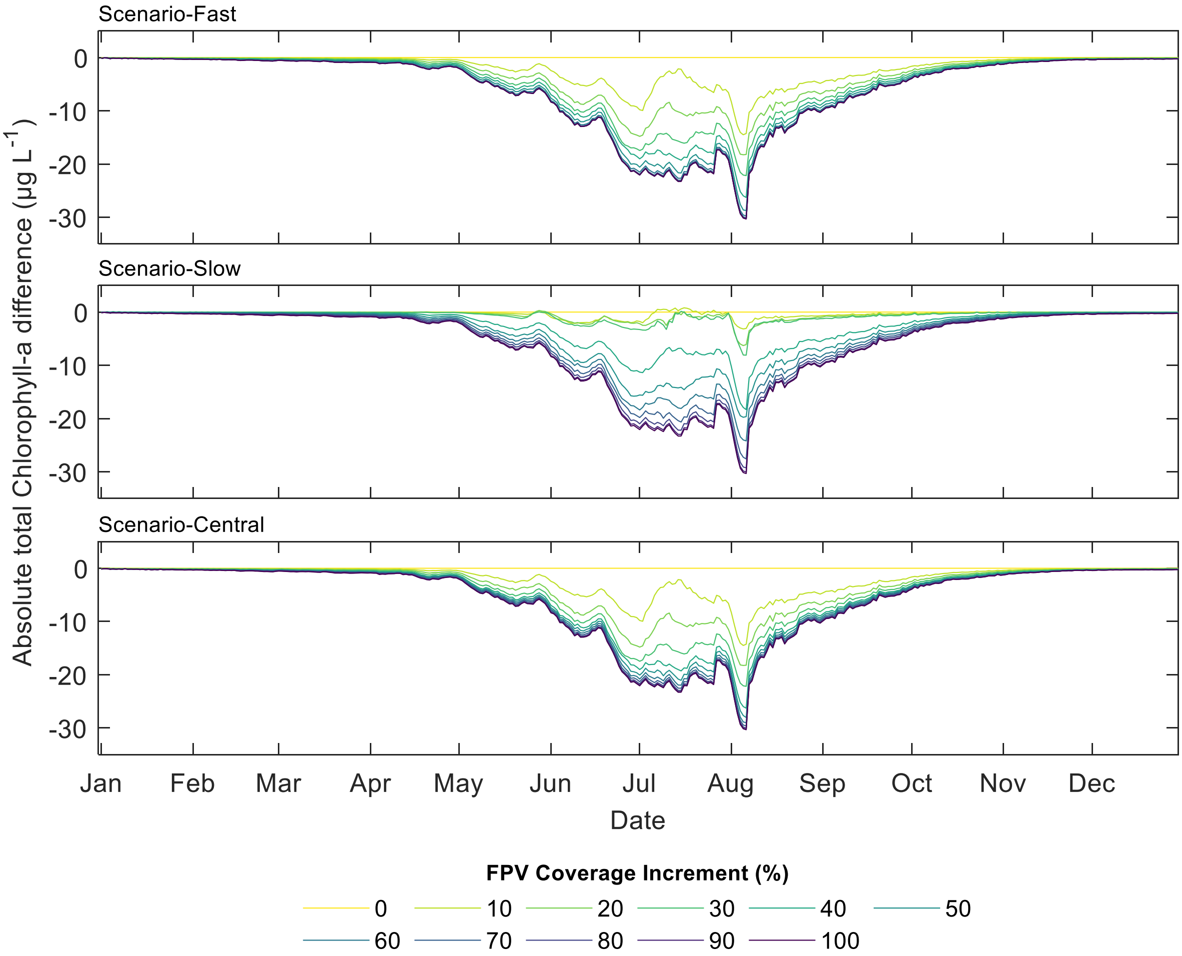


Figure 4-5 – Annual median total chlorophyll-a absolute difference by scenario. 0% floating solar coverage represents QEII reservoir simulated as a baseline with no additional floating solar coverage. The relative difference in chlorophyll-a is shown in Figure S 4-13.

4.5.3.3 Phytoplankton functional group dynamics

While simulated chlorophyll-*a* concentrations declined exponentially with increasing coverage, the relative proportion of phytoplankton functional groups varied. In Scenario-Fast, at FPV coverages of up to 60%, diatoms dominated for most of the year, with their dominance increasing as FPV coverage increased up to 40% (Figure 4-6 and Figure S 4-17). As the coverage increased above 60%, proportions of green algae increased, approaching a similar proportion as diatoms. In some cases, green algae were very similar to, or slightly exceeded, the proportions of diatoms towards the end of summer, as for the baseline scenario (Figure 4-6).

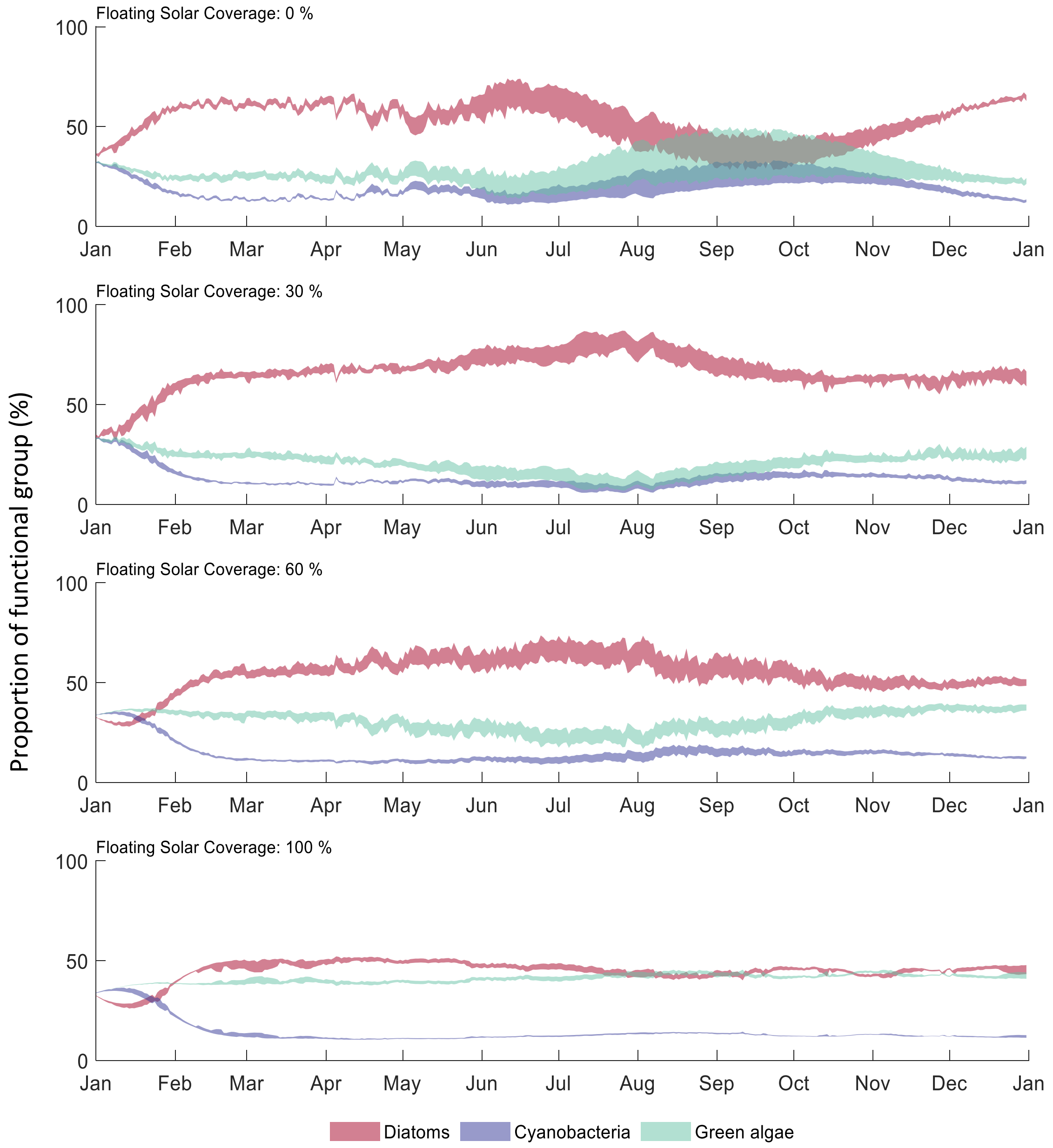


Figure 4-6 – Scenario-Fast: Proportion of phytoplankton functional groups as a percentage of total chlorophyll-a for the simulated period. The initial phytoplankton functional group proportions were set evenly, therefore, the first 30 days of simulations are model run-in time and should be ignored. 0% floating solar coverage represents QEII reservoir simulated as a baseline.

Similarly, diatoms increasingly dominated with FPV coverages of up to 70% in Scenario-Central. Diatom dominance slowly reduced from 70% to 100% coverage, associated with a higher proportion of green algae (Figure S 4-18). In Scenario-Slow, FPV coverages of up to 90% were associated with diatoms dominating for most of the year (Figure S 4-19). Diatom dominance strengthened as FPV coverage increased over 40% but declined again over 70%. In some cases, typically at FPV coverages up to 30%, green algae were very similar to, or slightly exceeded, the proportions of diatoms towards the end of summer, as they did in the baseline scenario (Figure S 4-19). Cyanobacteria did not have a high relative or absolute abundance regardless of FPV coverage.

4.6 Discussion

We found reduced phytoplankton biomass and changes in species composition can be directly attributed to the direct shading effects from reduced solar radiation and indirect mixing effects from wind sheltering of FPV. We also found that the different thermal dynamics associated with each siting location meant phytoplankton in the faster flowing tank appear more sensitive to low FPV coverage than the phytoplankton in the slower flowing tank, as they have to contend with both shading and rapid flushing, resulting in a large cumulative effect. Inflow volume, water temperature and nutrient inputs remained unchanged. In general, increased FPV coverage reduced total chlorophyll-*a*, although the absolute and relative reduction varied between each FPV deployment siting location scenario. There were a small number of simulations where phytoplankton biomass increased when the array was deployed on the slower flowing area of the reservoir. However, these increases were time-limited and only at array coverages of up to 30% in a small number of simulations.

4.6.1 Drivers behind the reduced phytoplankton biomass

We found that minimum, maximum and median surface water temperatures cooled due to the shading effects of FPV, slowing phytoplankton growth by reducing metabolic rates (Kraemer et al., 2017) as FPV coverage increased. As growth rates are species-specific, varying with cell size, each functional group responded uniquely to cooler water temperatures owing to increasing FPV cover (Reynolds, 2006). While deployment location had several complex and interacting effects, the effects of higher flow speed combined with FPV coverage led to an enhanced cooling effect. Given this flushing effect, the faster circulation tank exhibited a greater reduction in total chlorophyll-*a* and a more pronounced change in phytoplankton community structure than for similar coverages of FPV deployed on the slower circulation tank.

The cooler water temperatures associated with increasing FPV coverage reduced continuous and cumulative stratification duration. This indirect effect of FPV on reservoir mixing contributed to lower total chlorophyll-*a* in the reservoir (Exley et al., 2021a). In the absence of stratification or a shorter stratified period, the mixed layer, a fundamental driver of phytoplankton growth (Ross and Sharples, 2008; Longhi and Beisner, 2009), is deeper or fully mixed. The deepening of the mixed layer worsens the effective light climate for phytoplankton, moving them further from the higher light intensity surface waters (Reynolds, 1997). However, non-stratified conditions may allow phytoplankton to access pools of nutrients in the lower water column, favouring those species tolerant of the lower light availability at depth. On the small number of days when total chlorophyll-*a* increased with FPV coverage, the sheltering effect at the air-water interface likely reduced mixing, improving the conditions for phytoplankton growth (Exley et al., 2021a).

4.6.2 The consequences for phytoplankton functional-type dynamics

Modifications to reservoir thermal properties and shading from FPV coverage resulted in changes to phytoplankton functional-type dynamics, with the different siting locations modifying the response. Generally, the relative dominance of diatoms increased in the autumn with moderate FPV coverages as green algae populations reduced. However, these changes were offset by the overall rapid decline in phytoplankton biomass associated with increasing FPV coverage. In the faster circulation scenario, as FPV coverage increased and the reservoir became more mixed, dominance switched from green algae to diatoms, consistent with their affinity to well-mixed water bodies (Jäger et al., 2008). In the slower circulation scenario, which experienced less of a reduction in stratification duration than the faster circulation scenario, species composition remained similar to the baseline conditions.

Importantly, given the implications for water treatment and reservoir recreational use, cyanobacteria dominance did not increase with increasing FPV coverage for any of the deployment scenarios. This is attributable to the shaded conditions and additionally, the more mixed water column reduced the ability of cyanobacteria to regulate their buoyancy and vertical position to obtain favourable light and nutrient conditions (Reynolds et al., 1987; Burkholder, 2009). However, whilst our simulations show a reduction in total cyanobacteria biomass with increasing FPV coverage relative to the baseline, our use of functional-type aggregates may overlook the specific traits, tolerances and sensitivities among cyanobacteria taxa which could allow individual shade-tolerant or lower-optimum temperature species to dominate (Carey et al., 2012; Mantzouki et al., 2016; Armstrong et al., 2020). Studies considering the effects of surface covers have shown a switch to cyanobacteria dominance in some instances (Yamamichi et al., 2018; Exley et al., 2021b). However, the expanded model can simulate an unrestricted number of phytoplankton species, so assuming sufficient input data and observations to constrain the model, this uncertainty could be reduced in future applications.

4.6.3 FPV as a tool for water body management

Our results suggest that water body managers could tailor FPV system design and siting location to achieve the management goals of the host water body. The impact of percentage cover is clear, with opportunities for tailoring reductions to water temperature, mixing dynamics and phytoplankton biomass and species composition. Further, the interaction between the different residence times associated with each scenario and increasing FPV coverage shows that siting location is an important consideration when planning the deployment of an FPV array. Modifying FPV siting location between areas of different circulation can contribute greater water quality co-benefits while using identical FPV coverage. For example, deploying an FPV array covering 40% of the reservoir on the faster-flowing tank reduced total chlorophyll-*a* by up to 2.9 times more than deploying the same size array on the slower-flowing tank. Whilst the primary objective of an FPV installation is to generate renewable electricity, the potential for non-energy water quality co-benefits could offer an additional incentive to water body managers (de Lima et al., 2021; Exley et al., 2021a). However, this should be tested empirically given the simplification of the water body into faster and slower flowing tanks.

Regardless of deployment location, the large, sustained reductions in phytoplankton with FPV deployment may provide an alternative to hydrological manipulation in reservoirs. Typically, reservoirs used for drinking water are managed to limit thermal stability, impeding the development of stratification and subsequent phytoplankton growth, which can be detrimental to water quality and disrupt the water treatment process (Paerl, 2014; Visser et al., 2015; Huisman et al., 2018). Currently, management techniques that attract capital and operational expenditure, including flushing and artificial mixers, are used to change the system's hydrology or light regime for phytoplankton (Visser et al., 2015). Alternatively, FPV provides an opportunity to overcome the growing challenge of managing phytoplankton blooms (Burkholder, 2009; Paerl et al., 2019; Plaas and Paerl, 2021), negating the need for such reservoir management and also generating zero-carbon electricity.

However, there may be undesirable consequences of FPV deployment, especially for reservoirs used for recreation or those supporting aquatic life. Phytoplankton are the primary source of energy in lake food webs (Kalff, 2002) and an important component of global biogeochemical cycles (Falkowski, 1994). Consequently, FPV induced changes could have profound impacts. For example, lake production is a significant driver of zooplankton species richness (Hessen et al., 2006) and the disruption to trophic cascades may cause a significant reduction in planktivorous fish (Jeppesen et al., 2002; Gerdeaux et al., 2006). Therefore, practitioners should undertake careful planning to ensure deployments and their corresponding impact on phytoplankton aligns with the management goals of the host water body, with consideration for all trophic levels.

4.6.4 Expanded model adequacy, application and critical research needs

This study has provided novel model insights into FPV impacts unobtainable through field manipulation. The expanded model allows the explicit simulation of FPV installations on different types of water bodies and differently functioning tanks of water bodies. The expanded model remains computationally efficient, thus allowing multiple runs to capture uncertainty, given the nature of the data commonly available for the water bodies FPV tend to be deployed on. The functionality to simulate discrete zones of water bodies will allow further research questions pertinent to the deployment of FPV to be answered. For example, determining the influence of water body morphometric characteristics (e.g. depth and surface area) and FPV deployment layout (i.e. one continuous array or multiple smaller arrays) on FPV water quality impacts. Moreover, it will allow the implications of geographical location and future climate to be simulated.

Enhanced phytoplankton representation to simulate species composition enables the model to assess phytoplankton response in more detail. Better resolution of phytoplankton impacts is critical given the impacts of climate change and the implications for water supply reservoirs. In particular, the linking of Si species to the phytoplankton dynamics equations allows the representation of diatoms that can adversely affect water treatment as filamentous species block filters.

Application of the GLUE methodology provides insightful model outcomes despite the more sparse data inputs than desirable for water body modelling. High frequency and spatially explicit monitoring of water quality impacts at existing FPV installations are required to constrain the model better and reduce uncertainties in estimated responses. Ideally, studies should consider a BACI (Before, After, Control, Impact) design (Stewart-Oaten et al., 1986), to monitor water body response before and after FPV deployment, using a control to ensure any observed impacts are specific to the intervention. Such observations will provide an empirical assessment of model outcomes and more robust modelling representations of change. Further, given the importance of phytoplankton communities to water body function and the implications for water treatment, detailed quantitative phytoplankton speciation data would be invaluable to constrain the model better and improve phytoplankton functional group representations.

4.7 Conclusion

FPV deployment continues rapidly worldwide, outpacing understanding of any concomitant environmental impacts. Our findings demonstrate that modelling, using an uncertainty framework, can provide useful insight into possible water body response. Specifically, we found that FPV generally promotes cooler water temperatures that, coupled with deteriorated light conditions, slow phytoplankton growth. A less favourable mixing regime with FPV coverage can also lead to substantial phytoplankton biomass reductions, even with only a small percentage of a reservoir covered by FPV. FPV deployment also changes phytoplankton community composition, but any negative consequences were negated by the considerable reductions in total biomass, allaying hypothesised water quality concerns of a switch to undesirable species.

Moreover, our results show that the location of an FPV on the water surface can significantly affect water body thermal dynamics, modifying phytoplankton response beyond the impacts of percentage coverage. This outcome demonstrates the need to consider spatial location within the water body in addition to the total magnitude of FPV coverage for deployment decisions. Modelling approaches present a valuable and resource-efficient tool to explore water body-FPV interactions, enabling the assessment of FPV design and location options without the need for extensive *in-situ* testing. Pre-deployment modelling thus could help FPV developers and water body managers minimise negative impacts and maximise co-benefits of FPV across a range of targeted water bodies worldwide.

4.8 Supplementary Information

4.8.1 Advection and diffusion matrix examples

Table S 4-1 – Example diffusion matrix where 2.5% of Tanks 1 and 2 are mixed.

	Tank 1	Tank 2
Tank 1	0	0.025
Tank 2	0.025	0

Table S 4-2 – Example advection matrix where 2.5% of Tanks 1 and 2 flow between each tank.

	Tank 1	Tank 2
Tank 1	0	0.025
Tank 2	0.025	0

4.8.2 Tank configuration experiments

4.8.2.1 Rationale

Tank configuration experiments are important in determining the sensitivity of model results to any defined tank configuration. The experiments determine how much simulation results change when only the model tank setup is changed and runs are made with the same inputs. This is important as if the results from comparative simulations are significantly different, it is not possible to compare results from different configurations without separate calibration of each. Multiple tank configurations were modelled and compared, but only one illustrative example is provided here.

4.8.2.2 Tank configuration example

4.8.2.2.1 *Description of example configurations*

The Queen Elizabeth II (QEII) reservoir is supplied with nutrient-rich water from the River Thames (Reynolds et al., 2005), pumped via three inlets on the reservoir bed, one in the north-western corner and one in each of the two southern corners. The reservoir outlet is situated in the north-eastern corner. The example undertaken compared a 2-tank (2-parallel) and a 4-tank (3-series, 1-parallel) representation of QEII (Figure S 4-1). Both configurations are qualitatively consistent with a detailed study of circulation within the QEII reservoir during 2018 (Ta, 2019), emulating the flow characteristics of the reservoir, the only difference being the complexity of the representation.

In the case of the *2-tank configuration*, Tank 1 is 70% of the QEII reservoir surface area and Tank 2 is 30%. Tank 2 has the solar array which is present on QEII in the baseline situation (4.5% of the total QEII reservoir area). It is assumed that lateral diffusion between tanks mixes 2.5% of the volume each tank every day and that there is 2.5% of the volume of each tank circulating as advection that is separate from the inflow-outflow advection that are both assumed to be to and from Tank 1 (outflow is from Tank 3 in the 4-tank example).

The *4-tank configuration* has a similar circular advection imposed by lateral flows from tank 1 to 2, tank 2 to 3, tank 3 to 4 and tank 4 to 1 (all set to be 2.5% of volume); diffusion between all contiguous tanks is assumed to be 2.5% of each pair of tanks. In this configuration, Tanks 1 to 3 each had 33.3% of the volume of Tank 1 in the previous 2-tank configuration and Tank 4 had the same volume and location as Tank 2 in the 2-tank configuration.

4.8.2.2.2 *Structure of comparison*

For both tank configurations a *baseline* was simulated using the 75 acceptable simulations identified as specified in Section 3.1 of the manuscript. These baseline simulations are consistent with those of Section 3.1 in that in both cases a solar array was situated on a longer residence time tank having a solar array coverage equal to 4.5% of the entire reservoir surface area. The comparison with the baseline cases was made to the situation where Tank 2 (of the 2-tank Configuration) and Tank 4 of the 4-tank configuration) were 100% covered. Comparison is made between the two configurations for both the baseline and 100% cover over the long-residence time tank cases. Simulation results are shown for all tanks for completeness.

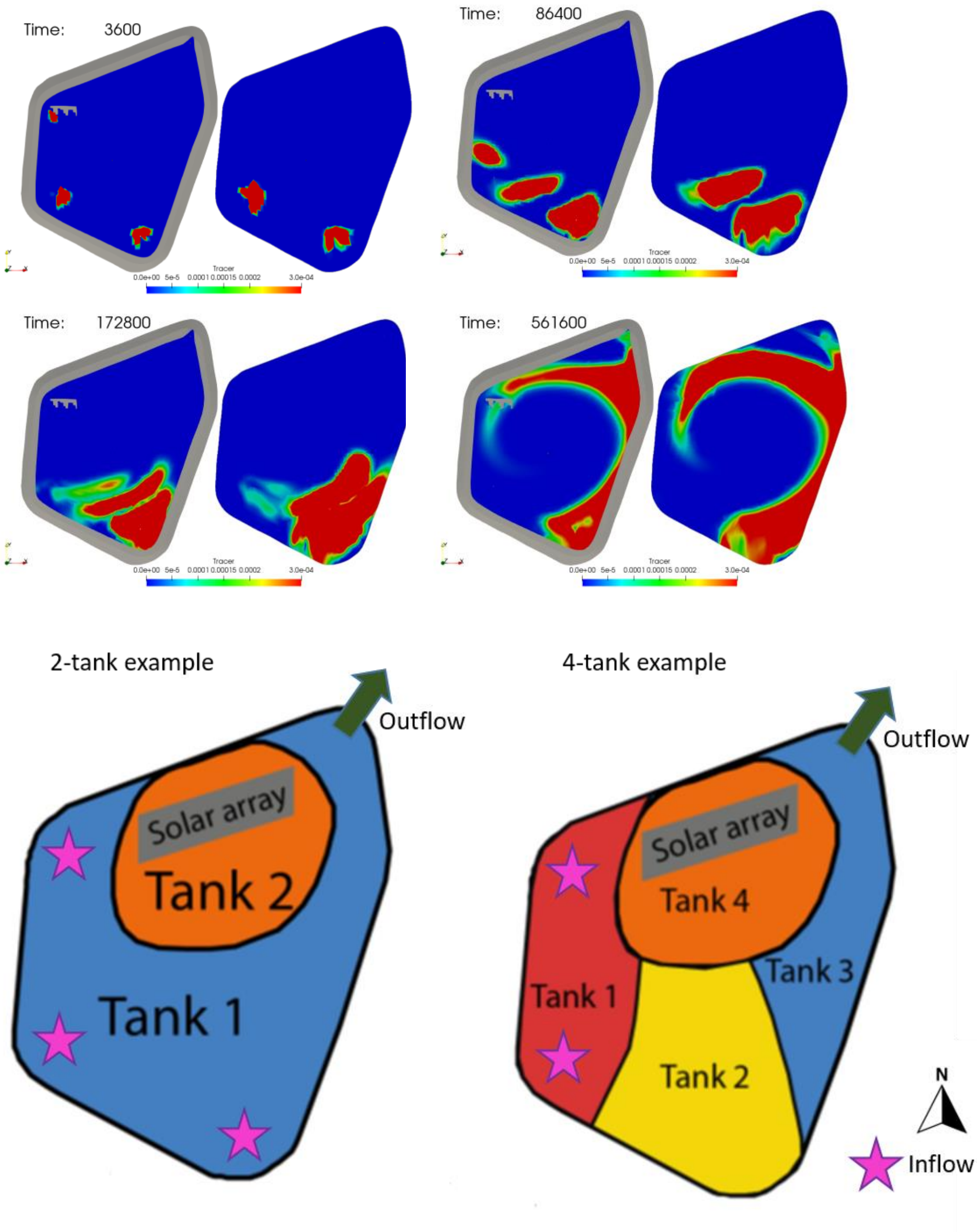


Figure S 4-1 – Top: example of the simulated Computational Fluid Dynamics from a detailed study on QEII circulation. QEII has three inlets, one in the north-western corner and one in each of the southern corners (identifiable by the red colour in the time 3600 graphic). The red colouring depicts the flow of a tracer over a given period. Graphic from (Ta, 2019) as prepared for Thames Water. Bottom: Tank configuration example comparison: 2-parallel- tanks (left) and 3-series-1-parallel tanks (right). The stars indicate approximate inflow locations on the reservoir bed. In the 4-tank example all inflow is directed into Tank 1.

4.8.2.2.3 *Tank configuration results*

Under *baseline conditions (i.e. in the absence of floating solar coverage)*, stratification patterns are altered between the 2-tank (Figure S 4-2) and the 4-tank cases (Figure S 4-3). Stratification for Tank 3 is slightly increased in persistence (Figure S 4-3). Subsequently, the chlorophyll-*a* time series showed slightly reduced concentrations in Tank 3 compared to Tank 1, but this is mainly due to a few short-lived spikes in concentration in the former. The distribution of chlorophyll-*a* in a 10-day window around the maximum chlorophyll-*a* value however showed that this was a decline of approximately 6% (Figure S 4-4). Diatoms had a similar pattern to chlorophyll-*a* but green algae and cyanobacteria had a slight shift upwards (Figure S 4-3).

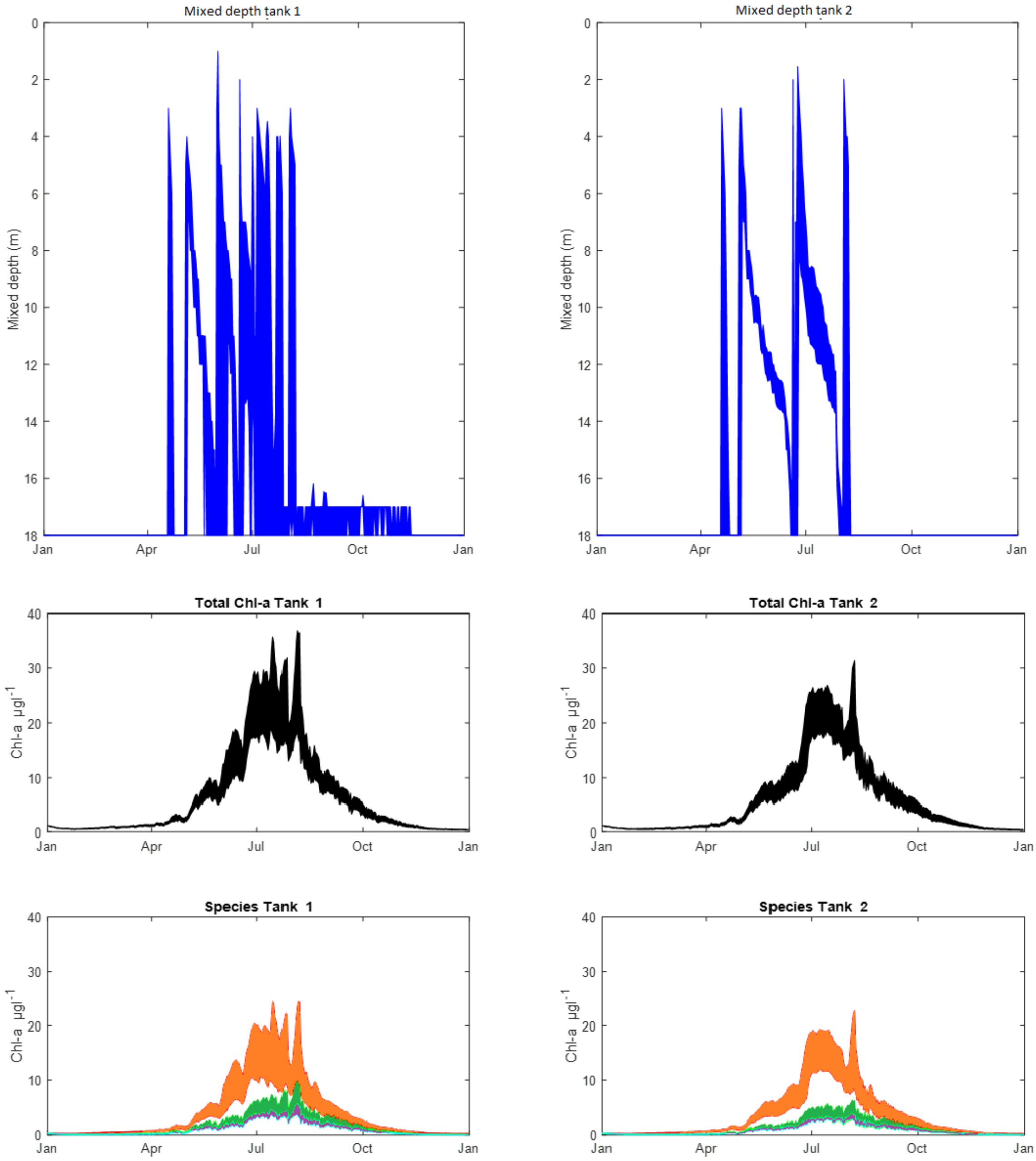


Figure S 4-2 - Baseline model (i.e. no FPV coverage) where the QEII reservoir is simulated with two tanks. In the mixed depth plots, the reservoir can be considered fully mixed when the mixed depth is 18 m. Phytoplankton species are represented as follows: diatoms = orange, green algae = green, cyanobacteria = purple.

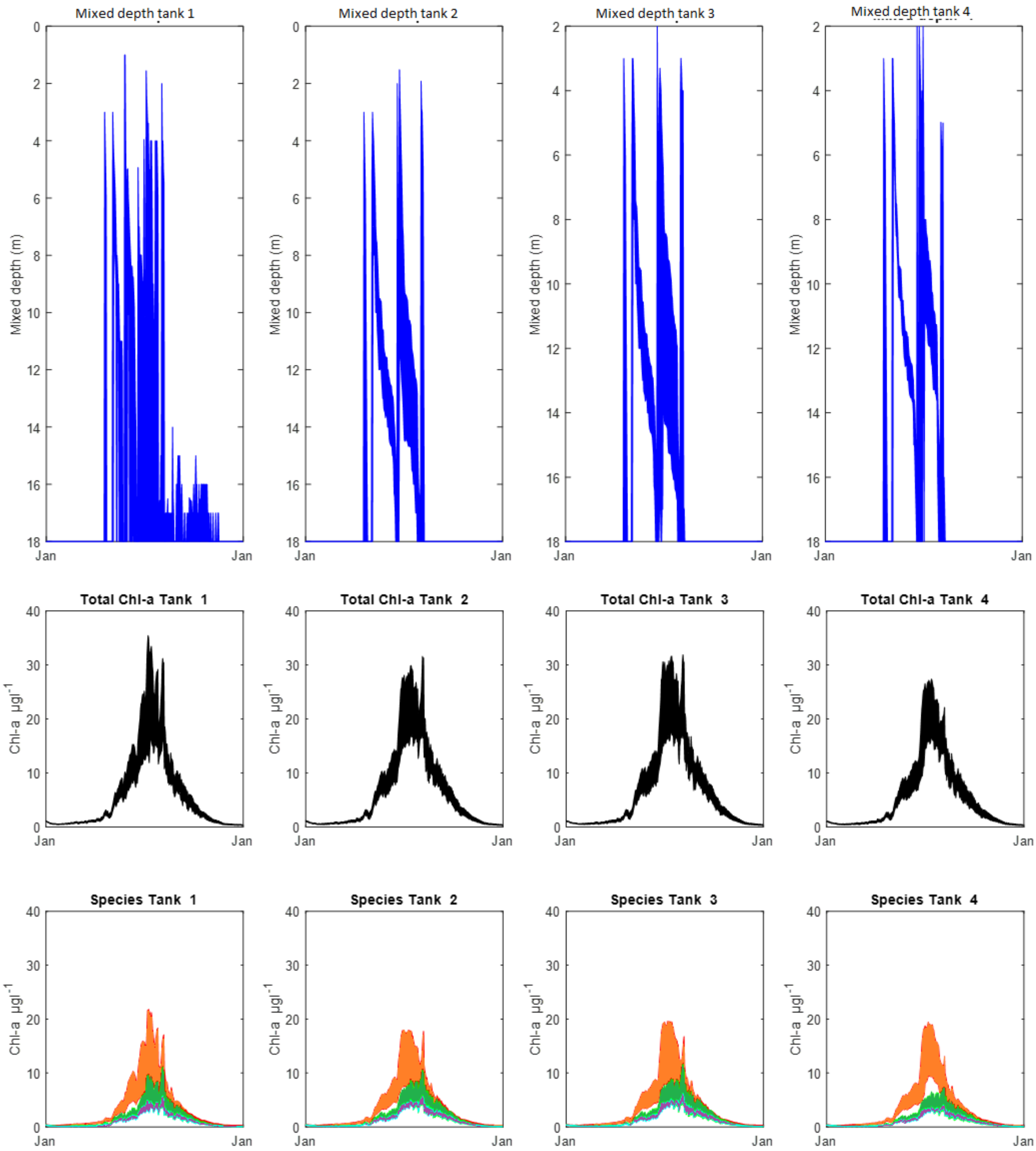


Figure S 4-3 - Baseline model (i.e. no FPV coverage) where the QEII reservoir is simulated with four tanks. In the mixed depth plots, the reservoir can be considered fully mixed when the mixed depth is 18 m. Phytoplankton species are represented as follows: diatoms = orange, green algae = green, cyanobacteria = purple.

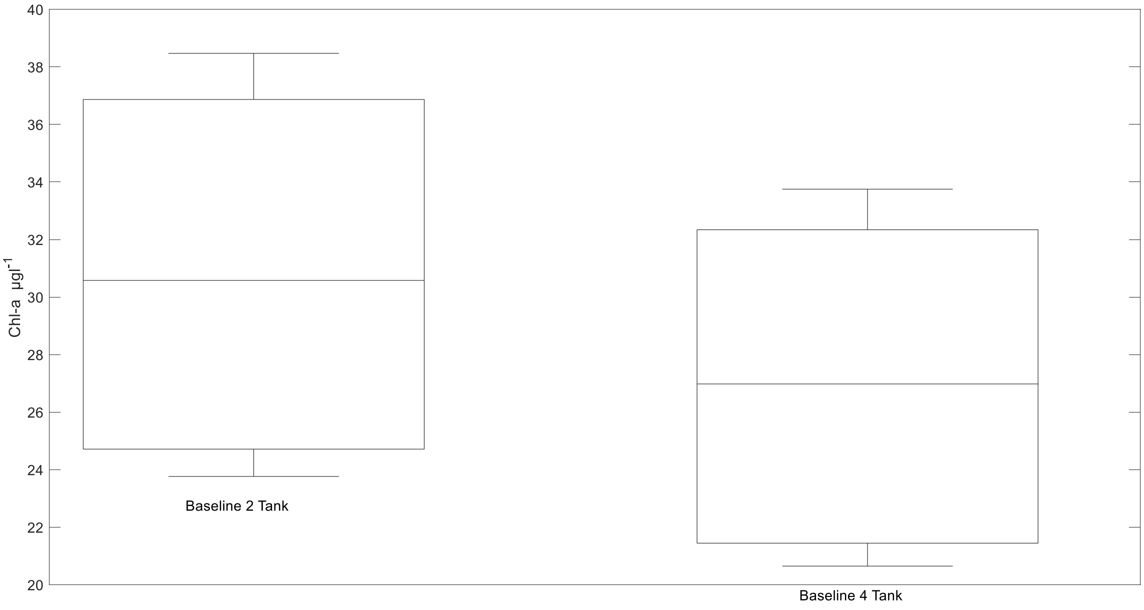


Figure S 4-4 – Total chlorophyll-a (based on the mean of a window, ± five days, around the day of maximum total chlorophyll-a) for the QEII reservoir simulated as a baseline (i.e. no FPV coverage) at the reservoir outflow for a two tank and a four tank model. In the two tank model, outflow is from Tank 1. In the four tank model, outflow is from Tank 3. Whiskers represent the minimum and maximum of the simulation results presented. The box represents the 2.5th & 97.5th percentiles, which gives a 95% confidence interval that simulation estimates fall within this range.

The results for the simulations where *Tanks 2 and 4 are completely covered* show a more pronounced decline in concentrations for chlorophyll-a (Figure S 4-5, Figure S 4-6, Figure S 4-7) and diatoms, little change for green algae and a slight tendency for an increase for cyanobacteria.

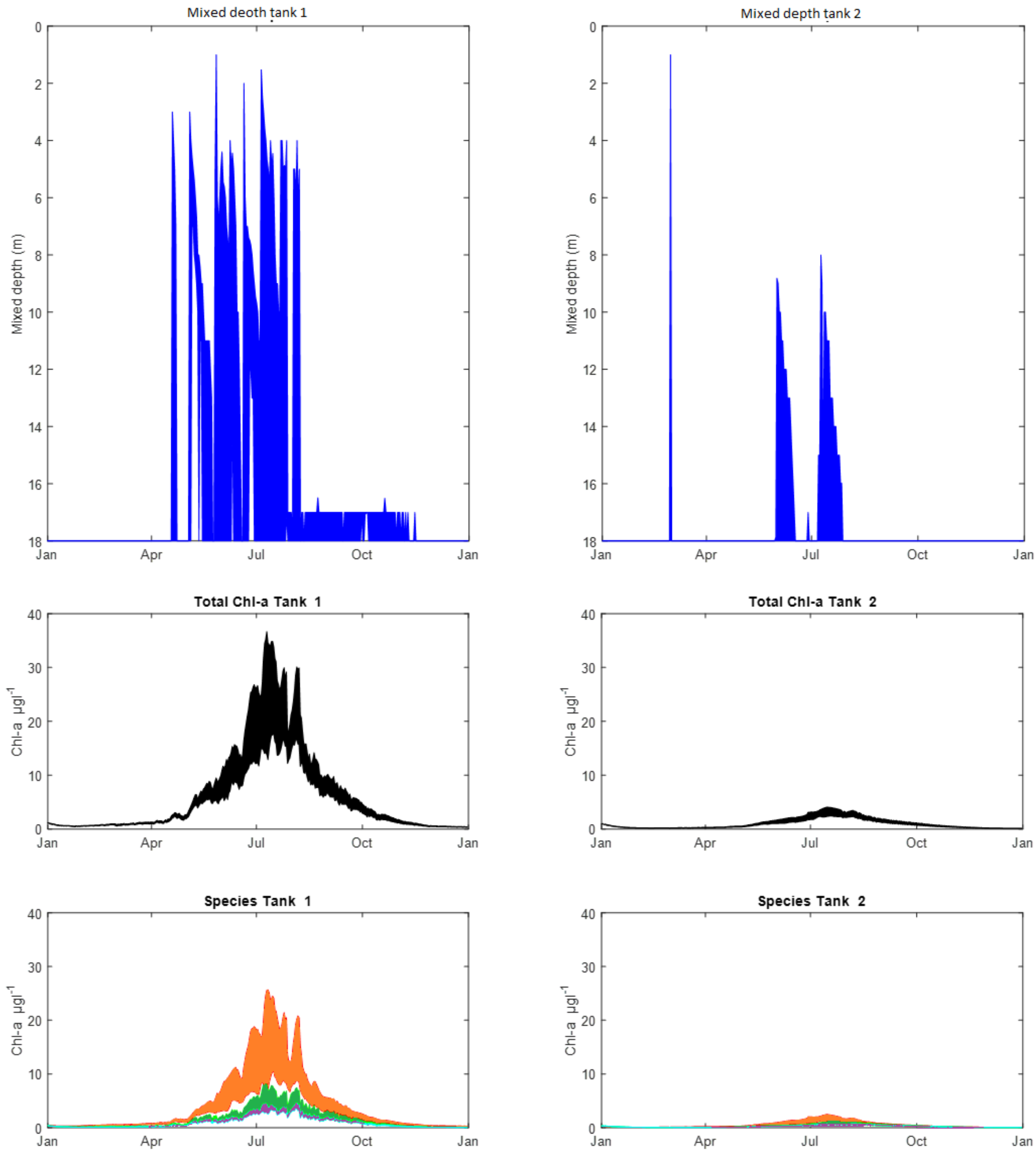


Figure S 4-5 – The QEII reservoir simulated with two tanks. Tank 1 is open (i.e. no FPV coverage) while Tank 2 is simulated with 100% FPV coverage. In the mixed depth plots, the reservoir can be considered fully mixed when the mixed depth is 18 m. Phytoplankton species are represented as follows: diatoms = orange, green algae green, cyanobacteria = purple.

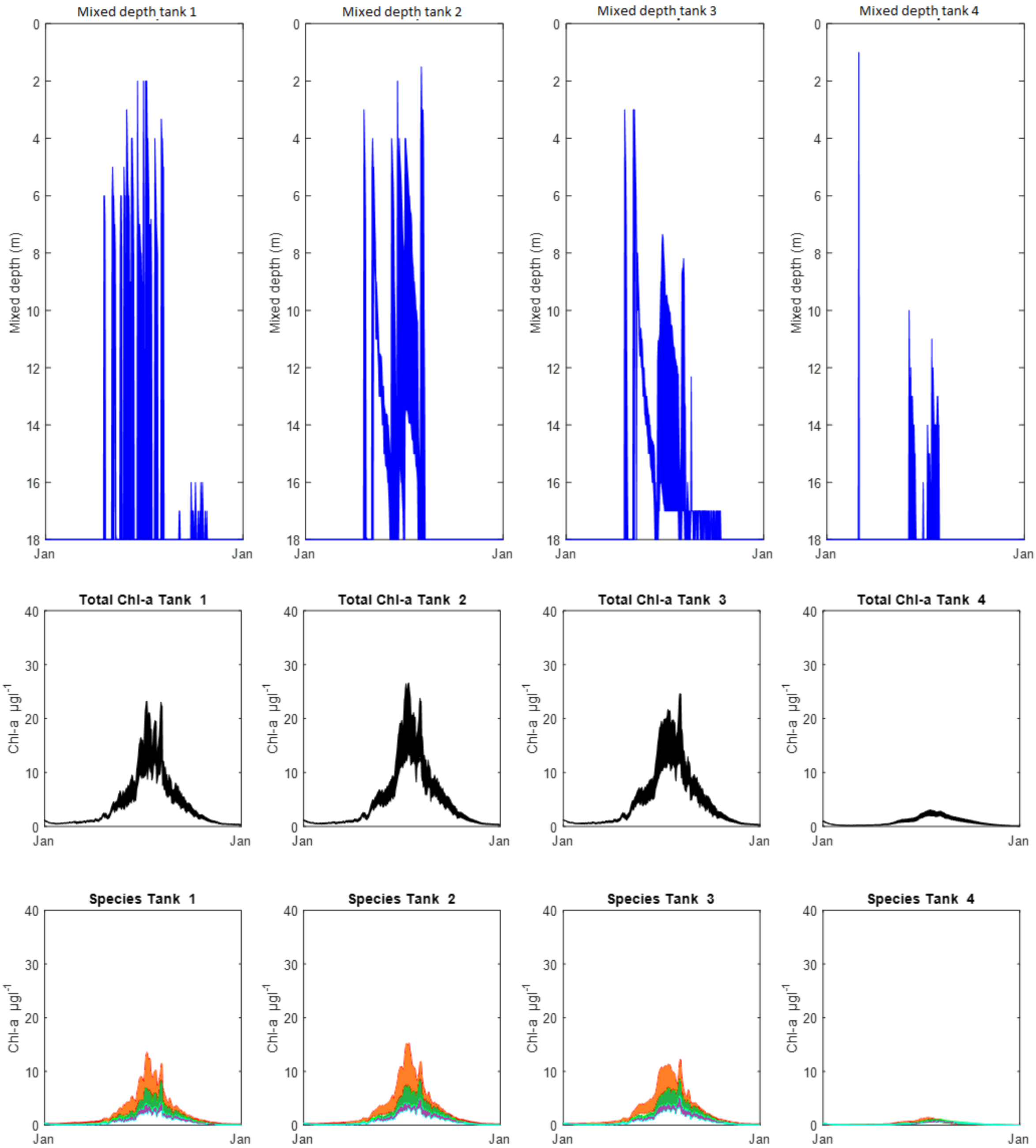


Figure S 4-6 – The QEII reservoir simulated with four tanks. Tanks 1, 2 and 3 are open (i.e. no FPV coverage) while Tank 4 is simulated with 100% FPV coverage. In the mixed depth plots, the reservoir can be considered fully mixed when the mixed depth is 18 m. Phytoplankton species are represented as follows: diatoms = orange, green algae green, cyanobacteria = purple.

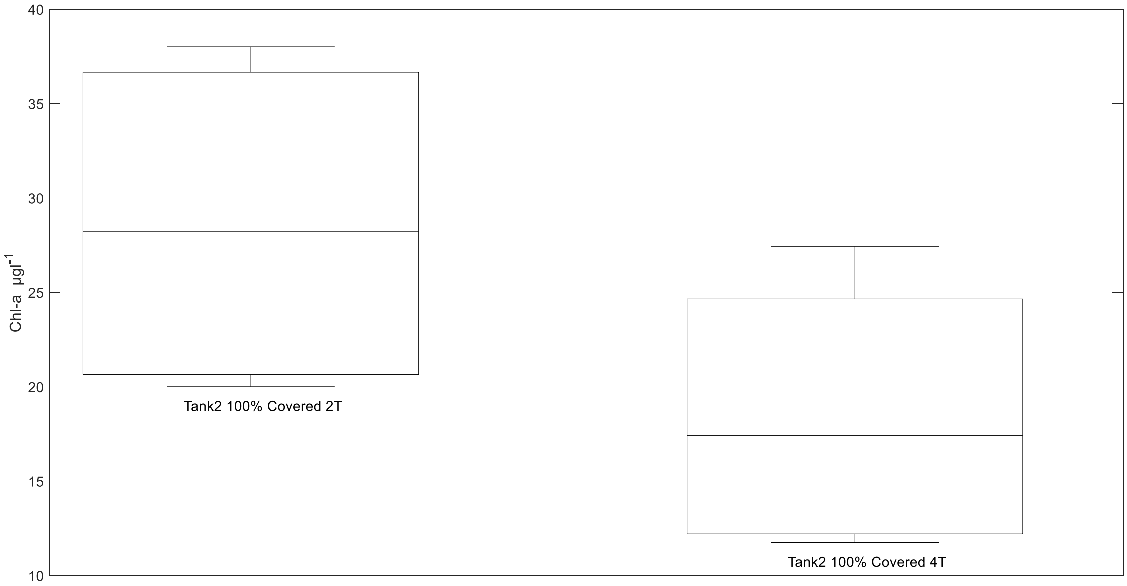


Figure S 4-7 – Total chlorophyll-a (based on the mean of a window, ± five days, around the day of maximum total chlorophyll-a) for the QEII reservoir simulated with partial FPV coverage (Tank 2 is 100% covered with FPV) at the reservoir outflow for a two tank and a four tank model. In the two tank model, outflow is from Tank 1. In the four tank model, outflow is from Tank 3. Whiskers represent the minimum and maximum of the simulation results presented. The box represents the 2.5th & 97.5th percentiles, which gives a 95% confidence interval that simulation estimates fall within this range.

4.8.3 Parameters and ranges sampled

Phytoplankton speciation was derived from an integrated sample of the upper 1 m of the reservoir and recorded based on the cell count by ascribing a rating on an ACFOR (Abundant, Common, Frequent, Occasional, Rare) scale (samples collected and analysed by Thames Water staff). Species with an 'Occasional' or 'Rare' abundance were disregarded. Each species with an 'Abundant', 'Common' or 'Frequent' abundance were assigned a chlorophyll-*a* per cell (picograms per cell) value from Reynolds (1984). If there was no published chlorophyll-*a* per cell value, the cell volume regression was used (Reynolds, 1984). A chlorophyll-*a* estimate for each species was derived from the number of cells per slide reported with each ACFOR category.

Given that the cells per slide count are reported as a range, the total biomass of each species was unknown. As the data available for the QEII reservoir were not sufficiently accurate to constrain phytoplankton behaviour in any more detail, six functional groups were chosen to broadly reflect the phytoplankton species composition observed in the QEII reservoir during 2018 and represent the broad functional trait differences. The relative proportions of functional types were used to partition the observed chlorophyll-*a* concentrations among broad functional groups, reported only as *diatoms*, *green algae* and *cyanobacteria* in the following analyses (Table S 4-3 and Table S 4-4).

Table S 4-3 – Phytoplankton parameters sampled and their ranges (min., max. of uniform distribution or fixed).

Parameter	Range sampled (min., max. or fixed)						
		Diatoms		Greens		Cyanobacteria	
PAR saturation level for growth (mol-quanta m^{-2} s^{-1})	Min	0.000025		0.0005		0.00075	
	Max	0.000045		0.00075		0.0009	
Optical cross section of chlorophyll-*a* (m^{-2} mg^{-1})	Fixed	0.03		0.005		0.01	
Loss rate at 20 °C (day^{-1})		Grazed	Un-grazed	Grazed	Un-grazed	Grazed	Un-grazed
	Fixed	0.18	0.13	0.05	0.025	0.025	0.0125
Settling velocity (m day^{-1})	Fixed	0.3		0.05		0.005	
Specific growth rate at 20 °C (day^{-1})	Min	0.7		1		1.1	
	Max	1.1		2		1.8	
Half saturation growth P level (mg m^{-3})	Fixed	5		5		5	
Half saturation growth N level (mg m^{-3})		Grazed	Un-grazed	Grazed	Un-grazed	Grazed	Un-grazed
	Fixed	80	80	80	80	80	0.1
Half saturation growth Si level (mg m^{-3})	Fixed	550		---		---	
If algae are N-Limited		Grazed	Un-grazed	Grazed	Un-grazed	Grazed	Un-grazed
	Fixed	1	1	1	1	1	1
If algae are Si-Limited	Fixed	1		0		0	
Scaling factor for inflow concentration of chlorophyll-*a* (dimensionless)	Fixed	0.5		0.4		0.1	

Table S 4-4 – Physical parameters sampled and their ranges (min., max. of uniform distribution or fixed). (-) = dimensionless.

	Sampled Ranges			Accepted range	
	Minimum	Maximum	Fixed	Minimum	Maximum
Wind shelter factor (–)	0.6	0.8		0.6016	0.7057
Fraction of PAR in incoming solar radiation (–)			0.45		
Inflow factor (–)			1.0		
Inflow temperature (absolute) (°C)	-1.0	1.0		-0.9986	0.4050
Inflow total phosphorus factor (–)			1.0		
Inflow NH_3 factor (–)			1.0		
Inflow NH_4 factor (–)			1.0		
Inflow Si factor (–)			1.0		
Non-PAR light attenuation coefficient (K_d)			1.0		
PAR light attenuation coefficient (K_d)	0.4	0.7		0.4097	0.6995

4.8.4 Model evaluation: Limits of Acceptability

4.8.4.1 Rationale

We used the extended Generalised Likelihood Uncertainty Estimation Framework (GLUE; Beven and Binley (1992)), where the criteria for acceptance are formalised *Limits of Acceptability* (LoA), for model simulations (see Page et al. (2017)). Simulations are evaluated under this approach where interactions between the uncertainties arising from model structural components, parameters, model inputs and observations used for model constraint are taken into account. Using LoA has the advantages that explicit representation can be made for the variability of errors (e.g. non-stationary/state-dependent errors and correlation of errors) at individual observation times and/or locations and is a natural way to combine different types of observation. This approach is critically important for focussing on how different sources of uncertainty determine model acceptability, affect the assessment of modelling hypotheses and inform strategies used when implementing the model to make predictions.

4.8.4.2 Variables used and their Limits of Acceptability

Limits of Acceptability were estimated from observed data using the same rationale as Page et al. (2017). There was an iterative process of relaxing stringent LoAs because of data and model limitations. For chlorophyll-*a*, the LoA for the observed early spring phytoplankton growth had to be modified to allow some acceptable simulations. This resulted from the fact that the model could not simulate the early observed growth which is thought to be associated with sub-daily stratification events which cannot be simulated by the model and may not be observed (on a given day) where only one temperature profile is taken. This is a problem that has been highlighted before using PROTECH (Page et al., 2017).

4.8.4.3 Fuzzy weightings and functions

For variables where LoA have been defined, simulation estimates at each observation time step are compared to the pre-defined limit of uncertainty. In this case each LoA for each variable is defined by a fuzzy weighting function that returns a relative weighting for each simulation depending on its position within the LoA. For example, for the 3 fuzzy weighting functions in Figure S 4-8, a simulation with a value of the example variable of 3 would give a relative weighting value of 1, 0.5 and 0.33 for the square, trapezoidal and triangular membership functions respectively (see intersections of red line in Figure S 4-8). These individual weightings were combined to provide an overall weighting for each scenario.

Table S 4-5 - Variable and fuzzy function used for each

Variable	Fuzzy weighting function
Water temperature at 1m	Triangular
Total Chlorophyll-*a* 1m	Trapezoidal
Mixed Depth	Trapezoidal
Diatoms (functional group)	Triangular
Green algae (functional group)	Triangular
Cyanobacteria (functional group)	Triangular

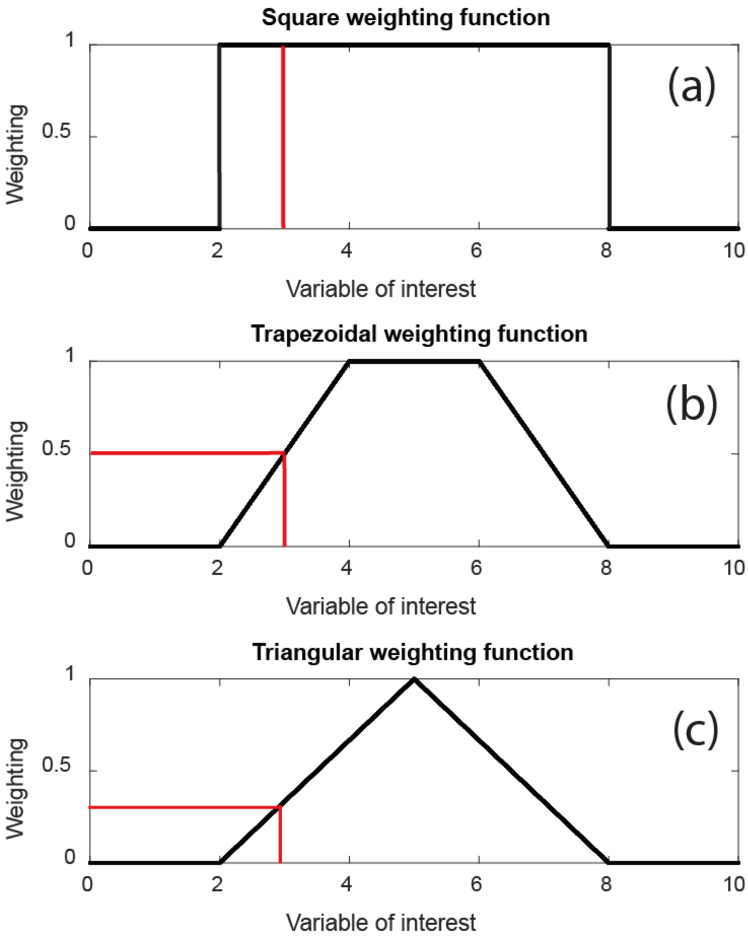


Figure S 4-8 – Fuzzy weighting functions: (a) square, (b) trapezoidal and (c) triangular relative weightings.

4.8.4.4 Combination of fuzzy weightings and model rejection

Model simulations were accepted if they fell within the minimum and maximum of the LoA at observation time steps for total chlorophyll-*a* and mixed depth (i.e. these were used as the primary LoA); all simulations where any estimated value fell outside of the relevant LoA was rejected. Although only these primary variables were used to reject simulations.

The fuzzy weighting functions were used to provide a confidence weighting (Table S 4-5). The variables chlorophyll-*a*, mixed depth and temperature had a weighting of one for the combined overall weighting (Wt) and the phytoplankton functional groups had a weighting of one-third (Equation S 4-1) to prevent over-constraint on functional groups.

Equation S 4-1

$$Wt = [(L_{Chl} + L_{mxd} + L_{wt} + (L_D.0.33) + (L_G.0.33) + (L_C.0.33))]$$

4.8.5 Observations made at an FPV installation – modelling assumptions

The effects of an FPV array on air temperature and incoming solar radiation were estimated from unpublished observations made at a separate FPV installation. Monitoring was undertaken at Langthwaite Impounding Reservoir (IR), UK (54° 1'26''N 02°46'1''W), an 865 megalitre raw water reservoir supplying drinking water to Lancaster and the surrounding area. The majority of water stored in Langthwaite IR is obtained by pumping from the River Lune, while there are also smaller feeds from two reservoirs and upland fell intakes. A 968 kWp FPV array was installed in the south of the reservoir in 2018 using a bespoke design of floating 'tables' to support 3520 PV panels. The electricity generated by the FPV array is used on-site at the water treatment works. The array is a minimum of 30 m from the banked sides of the reservoir, with a footprint covering ~6% of the reservoir's 127000 m^2 surface (Figure S 4-9). Langthwaite IR has a maximum depth of ~11 m, with the water depth under the FPV array ranging from 4.9 to 9 m when the reservoir is at top water level.

Air temperature (50 cm above the surface) was recorded at the centre of the FPV array, in the void between the underside of the PV module and the water's surface. The HOBO UA-001-08 logger was securely fastened to the centre of the underside of the PV panel, ensuring it was positioned at a fixed height from the water's surface and shielded from any direct sunlight. Similarly, a pyranometer suspended in the void between the underside of the PV module and the water's surface recorded incoming solar radiation. A shore based weather station continuously monitored at minute intervals, air temperature and solar radiation, providing a comparison between under array and open condition micrometeorology. The mean difference between the on-array and open condition measurements were used as assumptions for informing the effect of an FPV array.

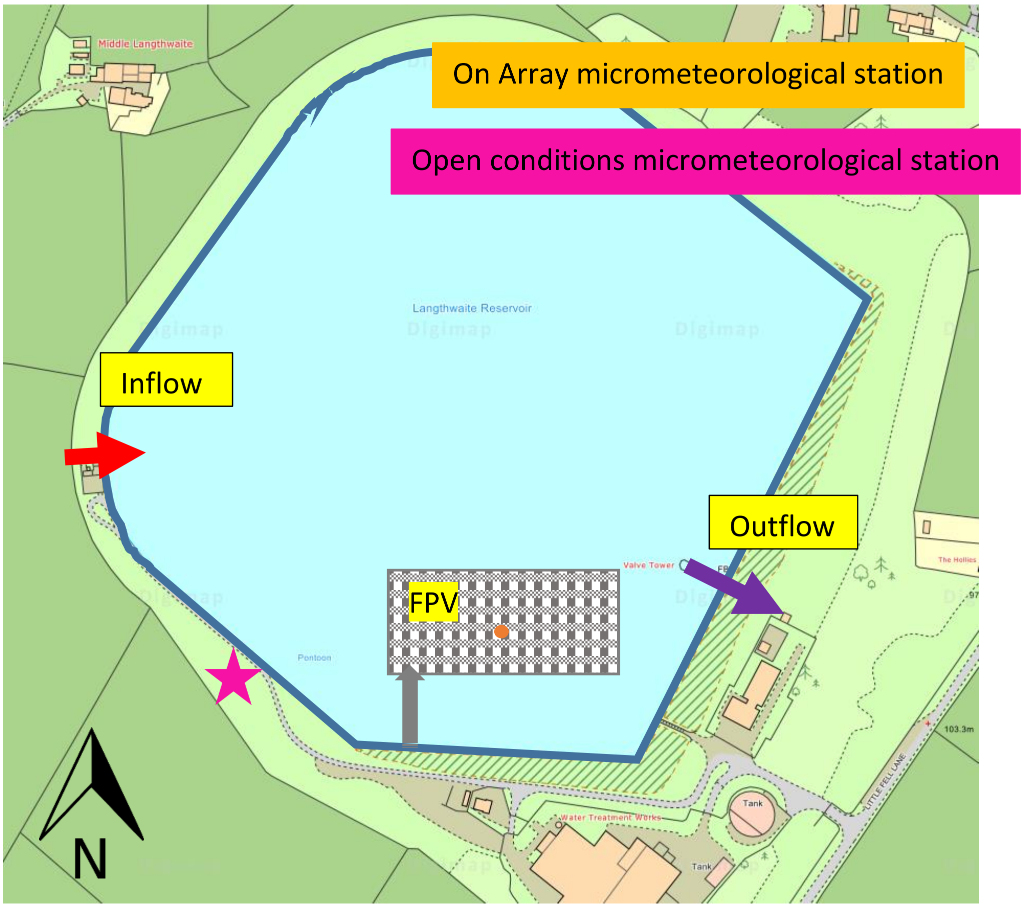


Figure S 4-9 – Sample locations showing the open water and co-located under and on array treatments. The shore based meteorological station is indicated with the star symbol, situated at the southeast corner on Langthwaite IR.

4.8.6 Annual median surface water temperature time series

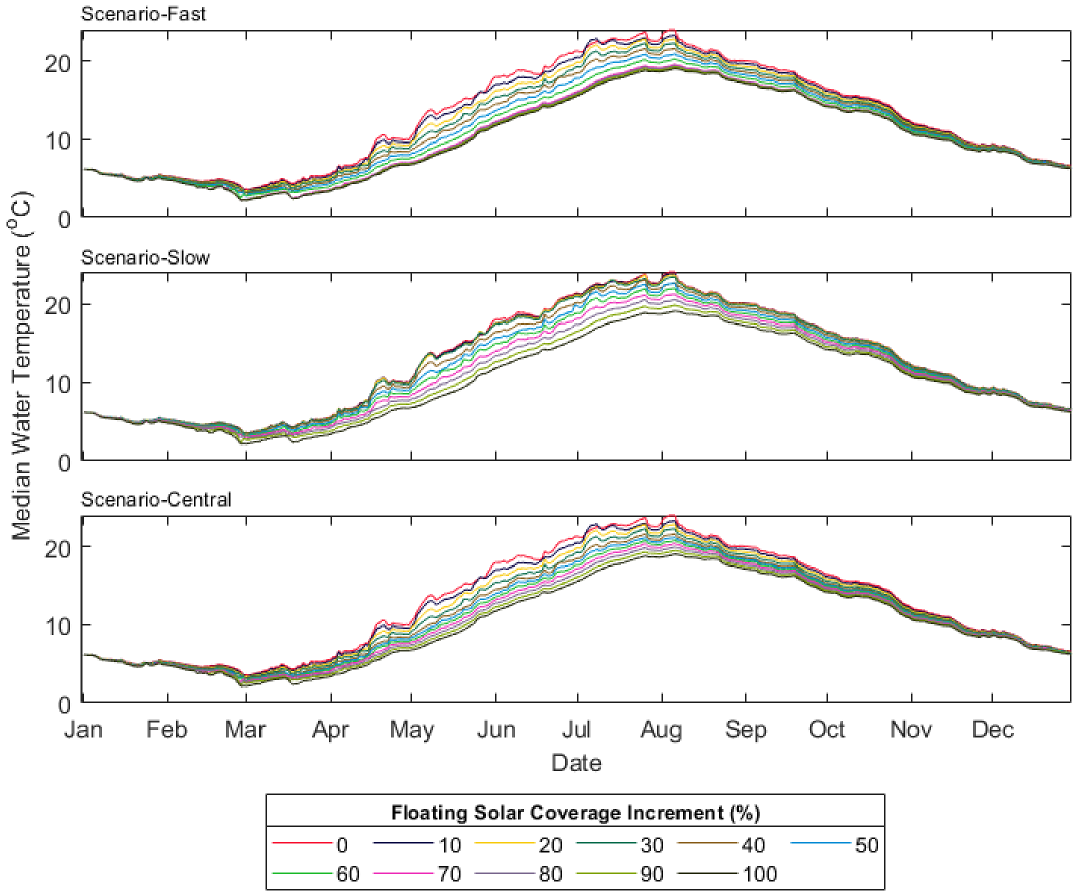


Figure S 4-10 – Annual median surface water temperatures with varying floating solar coverage. 0% floating solar coverage represents QEII reservoir simulated as a baseline. In Scenario-Fast, the FPV array is initially deployed on a short-residence time area of the reservoir, in Scenario-Slow the array is deployed on a longer-residence time area of the reservoir and in Scenario-Central the FPV array is positioned centrally on the reservoir.

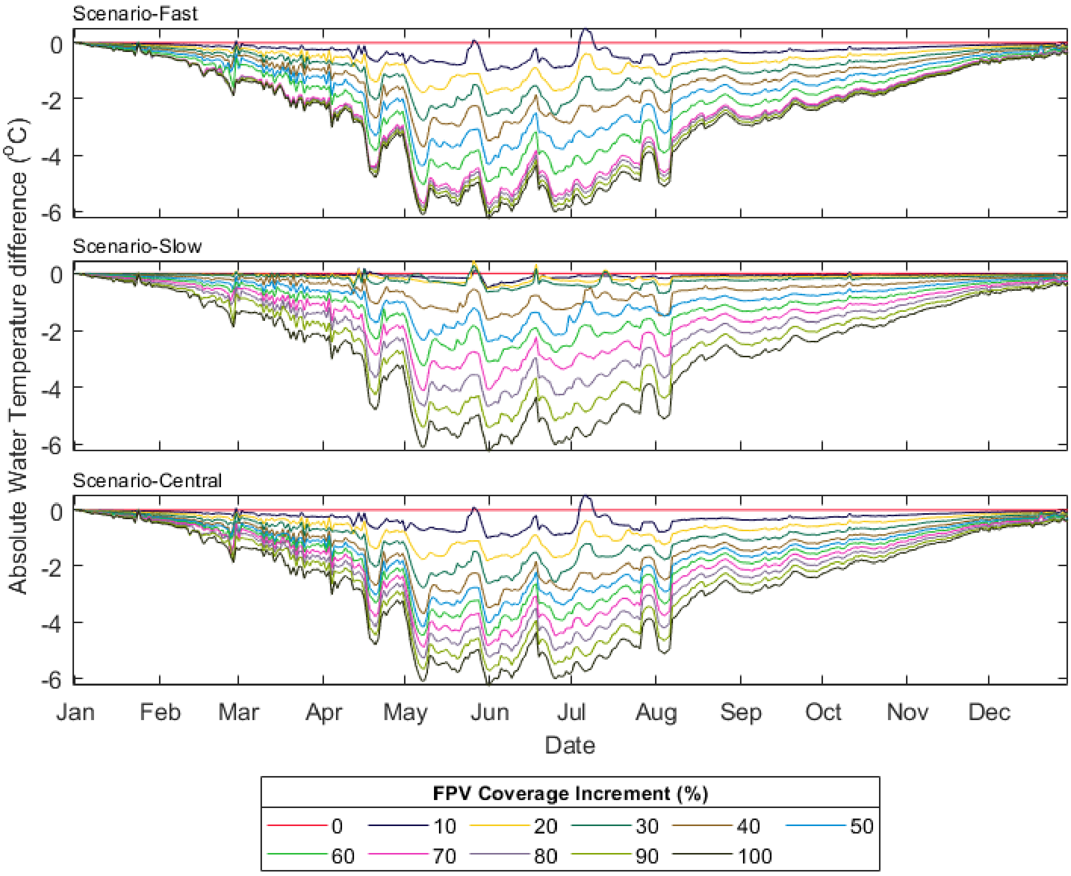


Figure S 4-11 – Annual median absolute surface water temperatures differences. 0% floating solar coverage represents QEII reservoir simulated as a baseline. In Scenario-Fast, the FPV array is initially deployed on a short-residence time area of the reservoir, in Scenario-Slow the array is deployed on a longer-residence time area of the reservoir and in Scenario-Central the FPV array is positioned centrally on the reservoir.

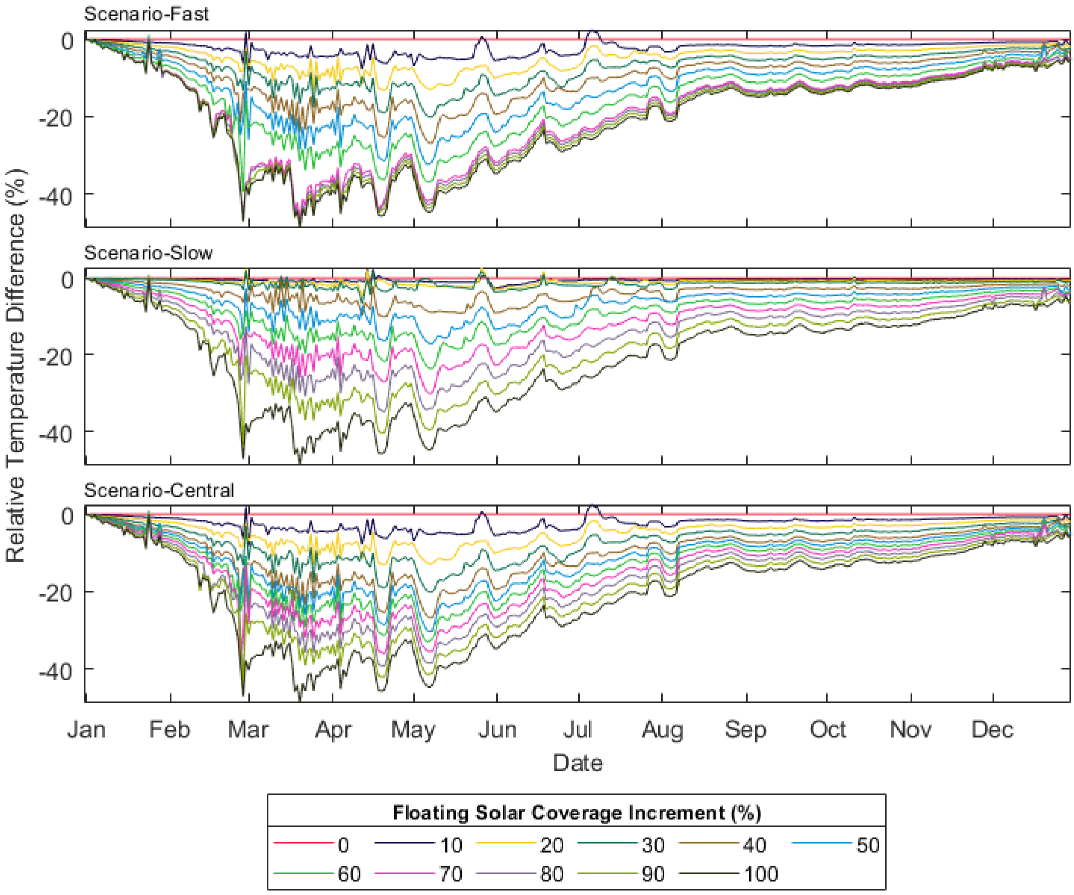
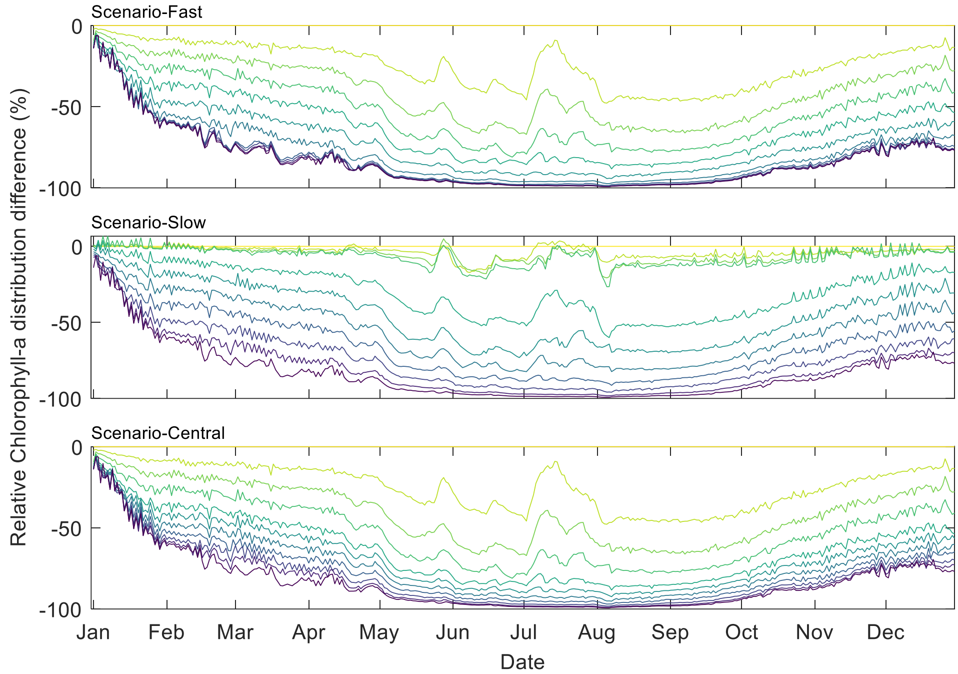
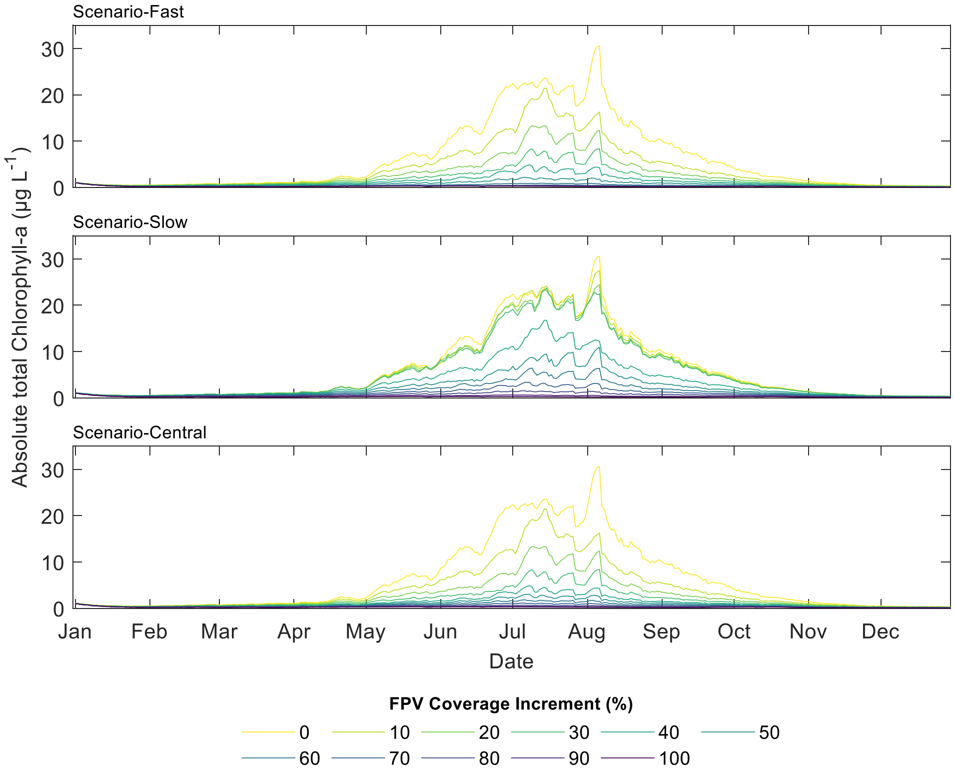


Figure S 4-12 – Annual median relative surface water temperatures percentage difference. 0% floating solar coverage represents QEII reservoir simulated as a baseline. In Scenario-Fast, the FPV array is initially deployed on a short-residence time area of the reservoir, in Scenario-Slow the array is deployed on a longer-residence time area of the reservoir and in Scenario-Central the FPV array is positioned centrally on the reservoir.

4.8.7 Annual chlorophyll-a time series



a)



b)

Figure S 4-13 – a) Annual median total chlorophyll-a percentage difference. b) Absolute median total chlorophyll-a. 0% floating solar coverage represents QEII reservoir simulated as a baseline.

4.8.7.1 Scenario-Fast

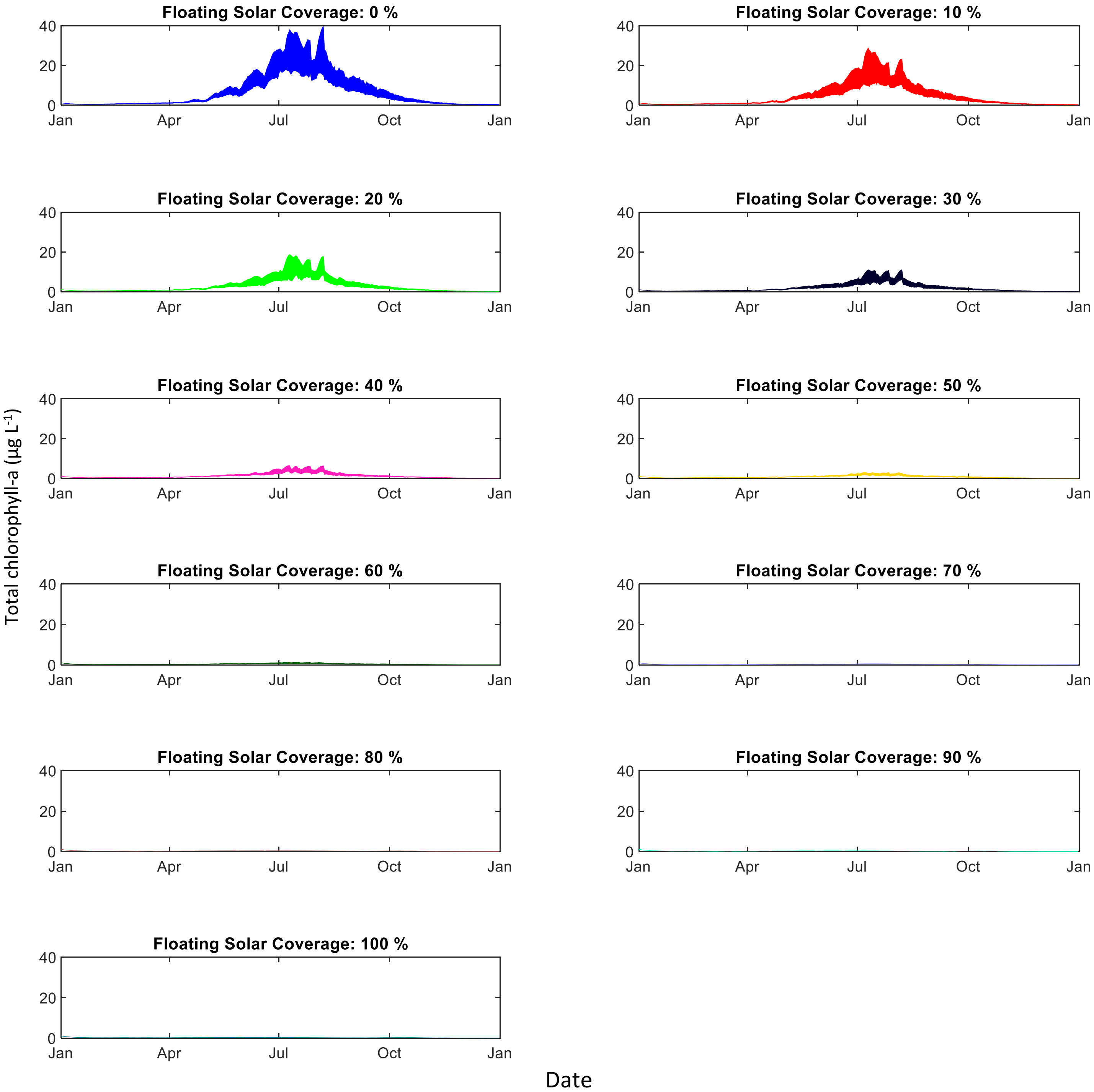


Figure S 4-14 – Scenario-Fast: Total chlorophyll-a for 2018 for increasing floating solar coverage – the coloured envelope represents the 95% confidence interval for each simulation day.

4.8.7.2 Scenario-Slow

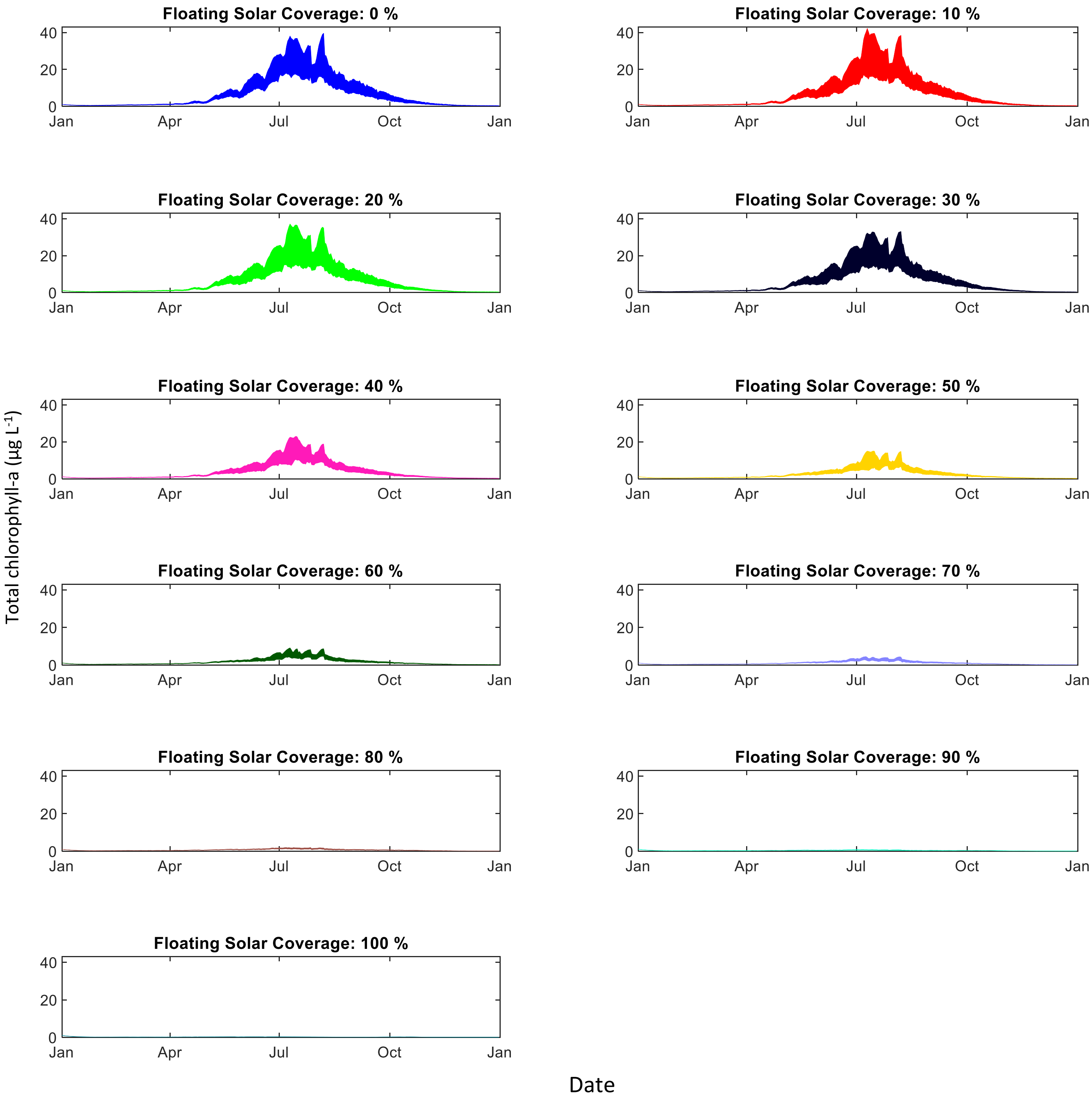


Figure S 4-15 – Scenario-Slow: Total chlorophyll-a for 2018 for increasing floating solar coverage – the coloured envelope represents the 95% confidence interval for each simulation day.

4.8.7.3 Scenario-Central

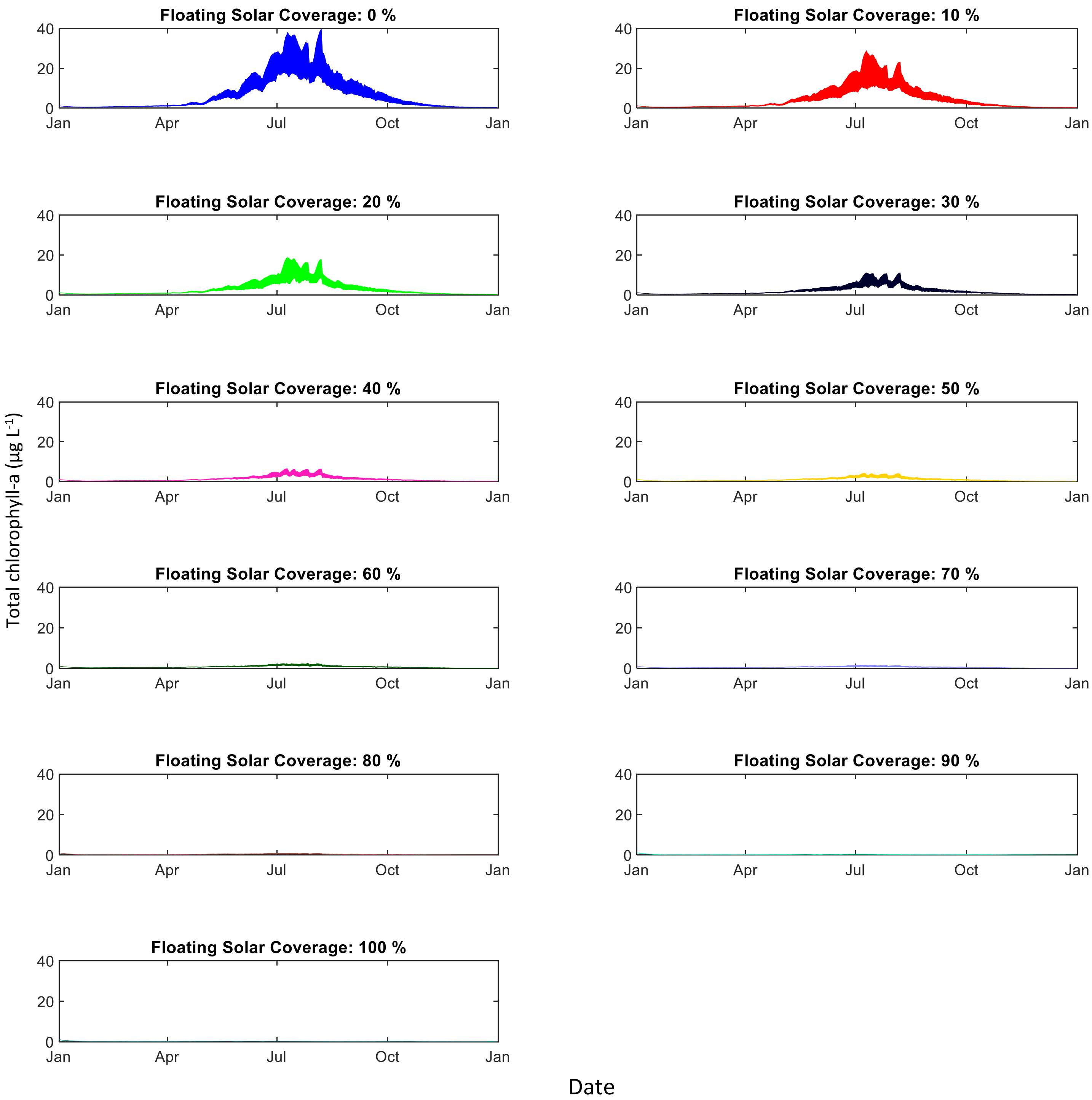


Figure S 4-16 – Scenario-Central: Total chlorophyll-a for 2018 for increasing floating solar coverage – the coloured envelope represents the 95% confidence interval for each simulation day.

4.8.8 Annual chlorophyll-a community composition time series

4.8.8.1 Scenario-Fast Phytoplankton functional groups

Whilst an exponential decline in total chlorophyll-*a* as coverage increased was simulated, the relative proportion of phytoplankton functional groups varied (Figure 4-6). At array coverages of up to 60%, diatoms dominated for most of the year, with their dominance increasing as floating solar coverage increased up to 40%. As the coverage increased above 60%, proportions of green algae increased towards diatom proportions. In some cases green algae were very similar to, or slightly exceeded, the proportions of diatoms towards the end of summer, as for the baseline scenario. The proportion of cyanobacteria remained low for all array coverages. It is important that these phytoplankton functional group proportions are taken in context with the absolute values of chlorophyll-*a* associated with each floating solar coverage increment.

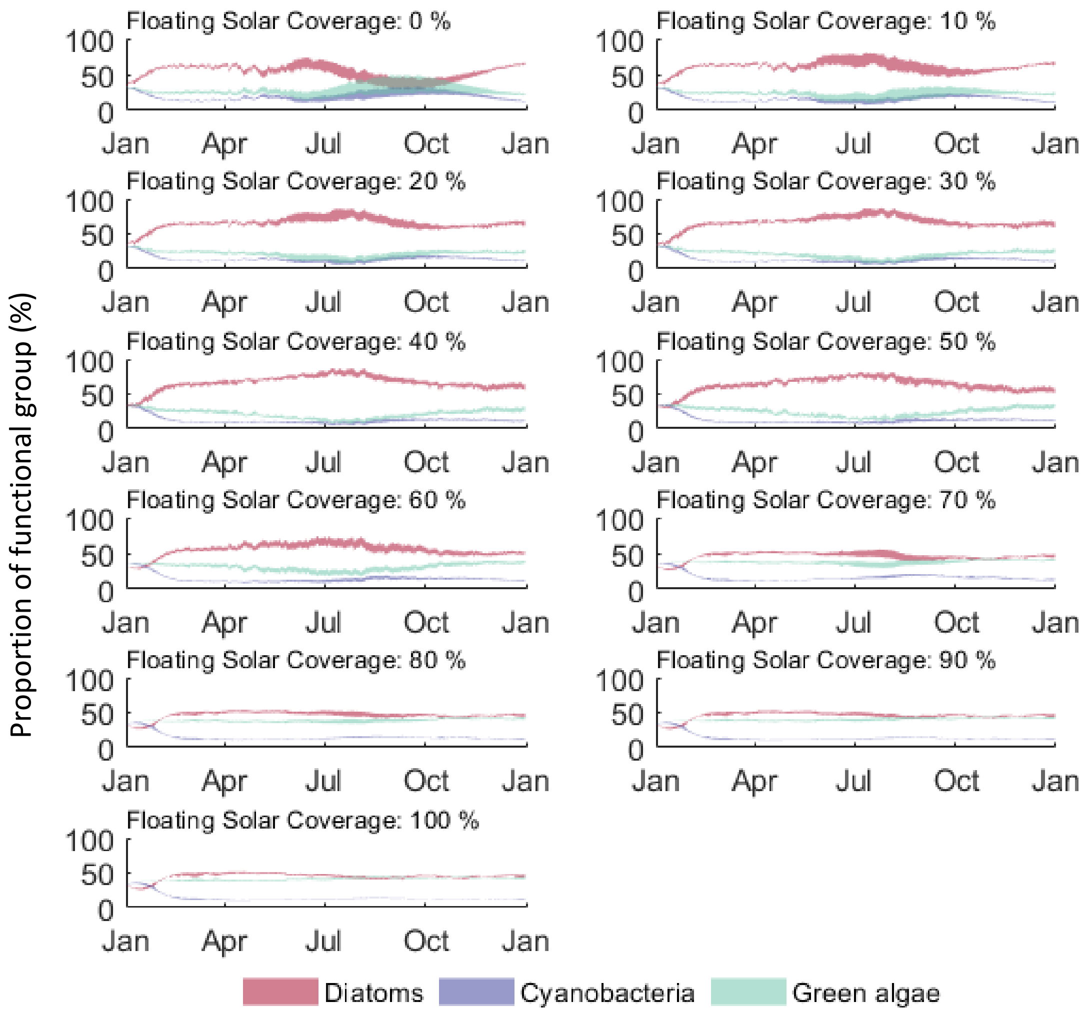


Figure S 4-17 – Scenario-Fast: Proportion of phytoplankton functional groups as a percentage of total Chlorophyll-a for the simulated period. Species inocula (the initial phytoplankton functional group proportions) were set evenly, therefore, the first 30 days of simulations are model run-in time and should be ignored. 0% floating solar coverage represents QEII reservoir simulated as a baseline.

4.8.8.2 Scenario-Central Phytoplankton functional groups

The relative proportion of phytoplankton functional groups for Scenario-Central (Figure S 4-18) varied over the year in a very similar way to the Scenario-Fast results (Figure S 4-18). At array coverages of up to 60%, diatoms increased their dominance for most of the year, with their dominance increasing proportionally as floating solar coverage increased up to 70%. This dominance slowly reduced from 70% to 100% coverage and allowed green algae to achieve a higher proportion. The proportion of cyanobacteria remained low for all array coverages. It is important that these phytoplankton functional group proportions are taken in context with the absolute values of chlorophyll-*a* associated with each floating solar coverage increment.

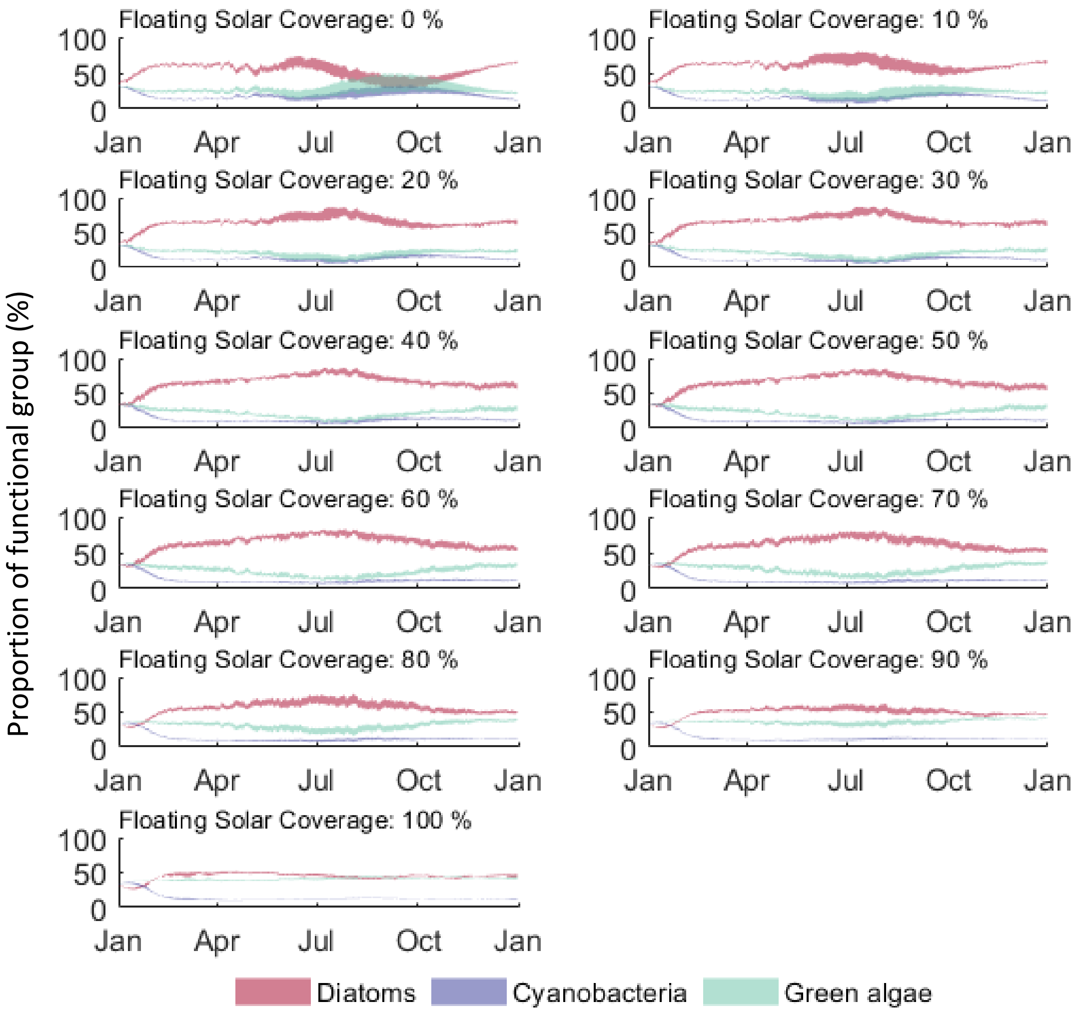


Figure S 4-18 – Scenario-Central: Proportion of phytoplankton functional groups as a percentage of total Chlorophyll-a for the simulated period. Species inocula (the initial phytoplankton functional group proportions) were set evenly, therefore, the first 30 days of simulations are model run-in time and should be ignored. 0% floating solar coverage represents QEII reservoir simulated as a baseline.

4.8.8.3 Scenario-Slow Phytoplankton functional groups

Whilst total chlorophyll-*a* generally reduced as coverage increased, the relative proportion of phytoplankton functional groups varied (Figure S 4-19). At array coverages of up to 90%, diatoms dominated for most of the year: dominance that strengthened as floating solar coverage increased over 40% but declined again over 70%. In some cases (typically at floating solar coverages up to 30%) green algae were very similar to, or slightly exceeded, the proportions of diatoms towards the end of summer, as they did in the baseline scenario. The proportion of cyanobacteria remained low for all array coverages. It is important that these phytoplankton functional group proportions are taken in context with the absolute values of chlorophyll-*a* associated with each floating solar coverage increment.

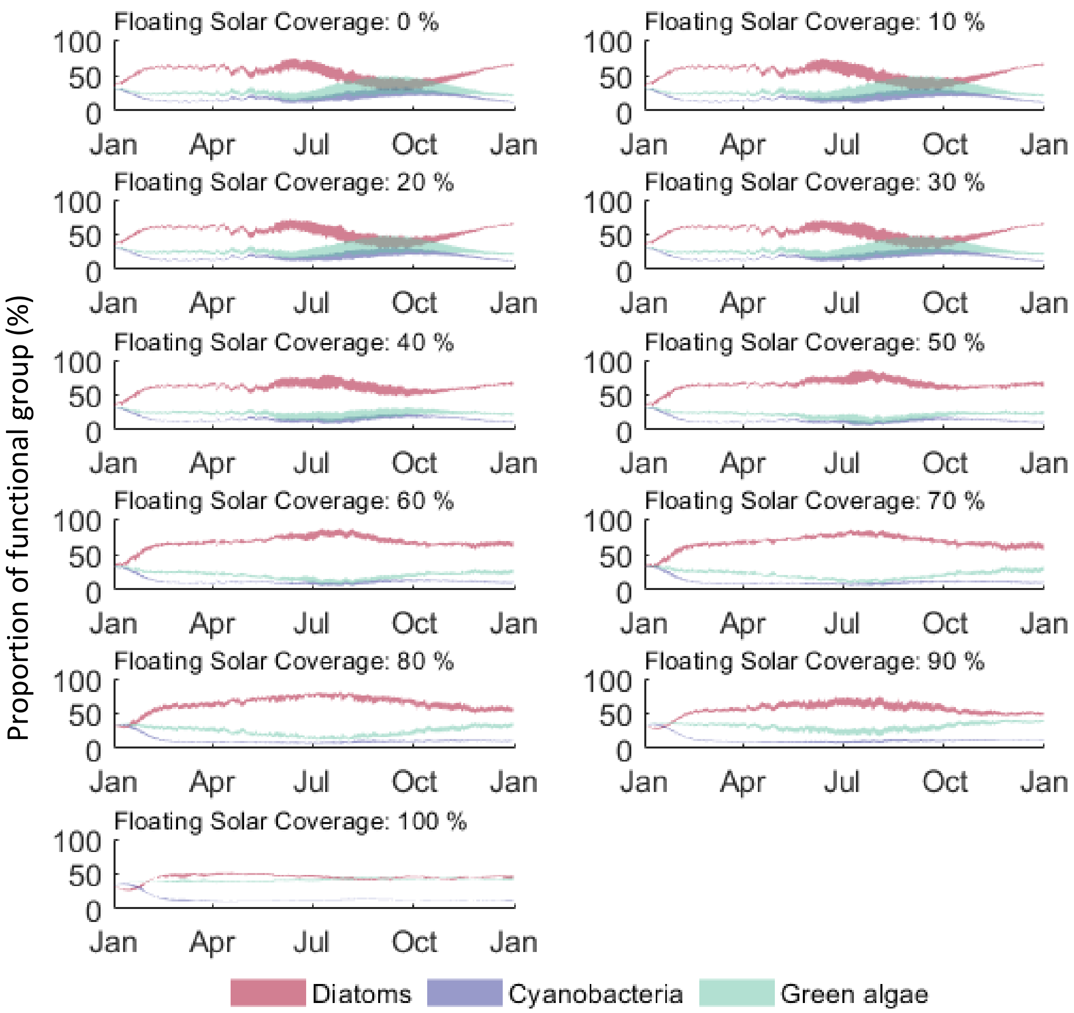


Figure S 4-19 – Scenario-Slow: Proportion of phytoplankton functional groups as a percentage of total Chlorophyll-a for the simulated period. Species inocula (the initial phytoplankton functional group proportions) were set evenly, therefore, the first 30 days of simulations are model run-in time and should be ignored. 0% floating solar coverage represents QEII reservoir simulated as a baseline.

4.8.9 Scenarios with stratification

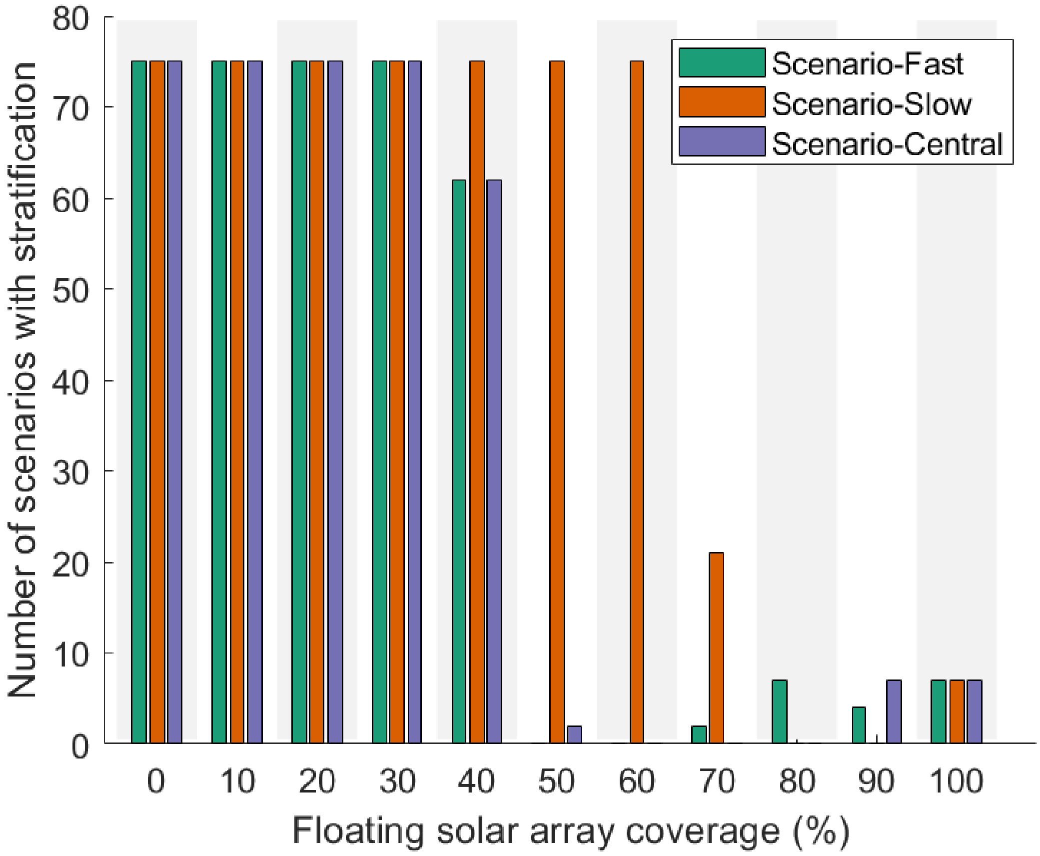


Figure S 4-20 – Total number of deployment scenarios (out of a possible 75) exhibiting significant stratification for each FPV coverage.

Chapter 5 – Mitigating climate change impacts on reservoirs with floating solar photovoltaics

Giles Exley, Trevor Page, Freya Olsson, Rebecca R. Hernandez, Andrew M. Folkard and Alona Armstrong

5.1 Abstract

The deployment of floating solar photovoltaics (FPV) on reservoirs is increasing, and this growth is expected to intensify as the energy transition continues. Based on the understanding of present-day FPV-environment interactions, FPV installations have the potential to reduce the effects of climate change on their host water body. However, the magnitude of the effect under future climates is unclear, and different FPV coverage extents may be required to attain the same offset potential as under present-day climates. FPV deployment represents a long-term change to a water body, so it is critical to understand the potential consequences for both current and future climates to avoid unintended energy-environment interactions. We simulated the effects of varying FPV coverage under four future climate cases and compared them to a present-day baseline case. The simulations suggest reservoir managers may be able to use FPV coverage to compensate partially or fully for changes in reservoir water temperature, stratification duration, phytoplankton biomass, and species composition under future climates. However, the effectiveness, and therefore FPV coverage required, depends on the season, future emissions levels, and desired management goals. Greater FPV coverages were needed to offset water temperature fluctuations in cases with greater emissions. However, lower FPV coverage was sufficient to counteract increases in phytoplankton biomass under future climates. FPV could be a valuable tool for addressing the effects of climate change on reservoirs. Still, stakeholders and practitioners must consider the specifics of each deployment to ensure the compatibility with and protection of water body ecosystem services.

5.2 Introduction

The Earth's aquatic environments are being increasingly affected by direct and indirect anthropogenic stresses, of which the most potentially impactful is climate warming (Adrian et al., 2009; Woolway et al., 2020). Climate warming represents a severe threat to water bodies, including lakes and reservoirs that are essential for providing drinking water and other ecosystem services (Maltby et al., 2011). Climate warming increases both the overall mean water temperature (O'Reilly et al., 2015) and the duration of thermal stratification in lentic waterbodies (Woolway et al., 2017a; Woolway and Merchant, 2019). Both of these have profound impacts on waterbody characteristics, including increased phytoplankton biomass (Winder and Sommer, 2012) and a switch to nuisance species (Paerl and Huisman, 2009). Changes to phytoplankton biomass and species composition are of major concern, given their role in food webs as primary producers and the problems they can cause potable water treatment (Watson et al., 2016). Water bodies are also impacted by other anthropogenic stressors including abstraction (Arnell et al., 2015), eutrophication and pollution (Moss et al., 2011), and more recently, floating photovoltaic solar panels (FPV) for electricity generation (World Bank Group et al., 2018; Cazzaniga and Rosa-Clot, 2021). The way in which these stressors impact water bodies depends on water body usage (e.g., for water supply, fisheries, electricity generation) and will interact, often negatively, with the pressures applied by those usages (Greaver et al., 2016; Collingsworth et al., 2017). As a result, there are widespread concerns about the combined impacts of climate warming and other anthropogenic stressors on water quality and quantity in lakes and reservoirs (Birk et al., 2020; Spears et al., 2021).

The work reported herein sets out to explore the combined effects of the stressors climate warming and FPV installations. FPV are becoming increasingly prevalent on a wide range of water body types (Exley et al., 2021b), especially raw water reservoirs, which have the primary function of storing water prior to treatment but are under increasing stress from climate warming. FPVs are commonly deployed to reduce land use conflicts, sparing land for agriculture, industry and conservation (Cagle et al., 2020). They also offer enhanced generation efficiencies over roof-top and ground-mounted solar panels, because of the cooling effect of the host water body (Choi et al., 2013; Sacramento et al., 2015; Yadav et al., 2016; Oliveira-Pinto and Stokkermans, 2020). In some cases, they have been deployed alongside hydroelectric generation installations, to optimise the use of existing transmission infrastructure and improve the power output profile (Silverio et al., 2018; Haas et al., 2020). FPVs have an expected 20-30 year lifespan, representing a long-term perturbation to the hosting water body (Rodrigues et al., 2020; Charles Rajesh Kumar and Majid, 2021; Costa and Silva, 2021). Therefore, understanding of their impacts must be developed to optimise their deployment, whilst minimising their potential detrimental effects and maximising their potential beneficial effects under both present and future climates.

Since their first commercial deployment in 2007 (Sanchez et al., 2021), knowledge of FPV-environment interactions has been gradually expanding. However, existing predictions of FPV impacts are based on present climate conditions (e.g. Haas et al., 2020; Exley et al., 2021b; Gorjian et al., 2021) and remain to be quantified for future climates. As climates are expected to change during the lifespan of FPV installations, it is important that those responsible for FPV operation consider the effects of the installation on the host water body for a range of plausible future climates. Based on present climates, the effects of FPV on key water body variables generally counteract those of climate change (Chapter 4), offering the potential for FPV to be used as a management tool to mitigate the climatic effects. For example, studies have identified that FPV installations typically cool water temperatures (Ziar et al., 2021), and shorten stratification duration (Exley et al., 2021a), with the magnitude of the effect modulated by the extent of FPV coverage and their deployment location (Chapter 3 and Chapter 4). These changes have consequences for the biological functioning of the host water body, reducing phytoplankton growth and modifying species composition, potential co-benefits that can improve water quality (Chapter 4).

Although FPV installations could improve the conditions of their host water bodies, by dampening the effects of climate warming, the scale of the effect is unknown under future climates and may require different FPV coverage extents to achieve the same offset potential as under present climates. If FPV can offset future climate impacts on reservoirs, it would delay or negate the need for alternative interventions, for example, new infrastructure capable of treating water with large volumes of nuisance phytoplankton species (Whitehead et al., 2009). Moreover, FPV deployment must be compatible with the goals of water body managers in the present, throughout the lifetime of the installation and beyond.

Developing understanding of the impacts of FPVs on their host water bodies under future climates will facilitate their use as a tool to manage the effects of climate change and maintain or enhance vital ecosystem services, at the same time as performing their primary function of providing a low carbon energy source. This study aimed to determine the effectiveness of FPV as a tool to limit three of the likely effects of climate warming on reservoirs: warmer water temperatures; prolonged stratification duration; and increased overall phytoplankton biomass and the proliferation of nuisance phytoplankton species. To achieve this, we used a numerical model to simulate the effects of FPV deployments on a raw water reservoir under plausible future climate scenarios. We explored the effects of varying the percentage coverage of FPV on the water body surface, and tested the hypotheses that increased FPV coverage under future warming climate scenarios would:

1. offset reservoir water temperature warming;
2. reduce the duration of thermal stratification; and
3. limit the growth of phytoplankton and prevent dominance of nuisance phytoplankton species.

We use insights gained from testing these hypotheses within our model to discuss the potential implications for reservoir management of increased use of FPV deployment in the context of warming climates through the 21st century.

5.3 Methods

The methods used in this study extend those of Chapter 4. A summary of the original methods and full details of the additional methods used to simulate future climate impacts are described below.

5.3.1 Site description

The Queen Elizabeth II reservoir (QEII) is a raw water reservoir located in south-west London, UK (51° 23′ 27″ N, 0° 23′ 32″ W; surface area: 128 hectares). The reservoir stores up to 19.6 million m^3 of water and is fed by the adjacent River Thames. Nutrient-rich river water is pumped into the reservoir through three inlets in its bed. Water is abstracted from a single outlet point at its north-eastern corner. Given the internal circulation patterns observed in the reservoir (Ta, 2019), it can be apportioned into two discrete zones based on the reservoir's hydrologic behaviour: one relatively short residence time, *faster-flowing tank* (70% of QEII surface area) and one comparatively longer residence time, s*lower-flowing tank* (30% of QEII surface area; Figure 5-1). Flow in the former zone can be viewed as a 'short-circuiting' between the reservoir's inlets and outlet. An FPV installation with a maximum power generation capacity of 6.3 MW, was installed on the *s*lower-flowing tank of the reservoir in 2016, covering approximately 4.5% of the total reservoir surface.

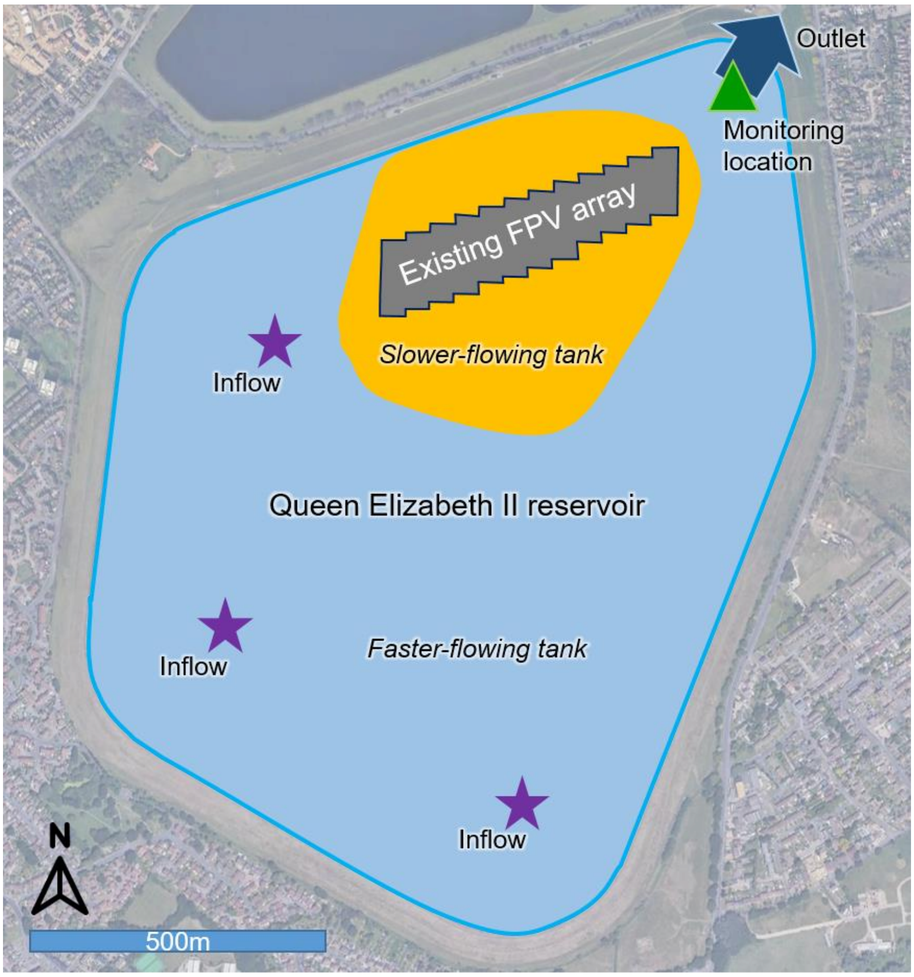


Figure 5-1 – Conceptual zones for the QEII reservoir during 2018 based on hydrologic behaviour. Satellite image from Google Earth.

5.3.2 *MyLake* model description

To determine the effectiveness of FPV installations as tools for managing the effects of climate change on reservoir water temperatures, stratification duration, and phytoplankton biomass and species composition we used a recently expanded version of the open source, one-dimensional (vertical) numerical lake model *MyLake v2* (Markelov et al., 2019). Full details on the original *MyLake* model can be found in

Saloranta and Andersen (2007) and the accompanying user manual (Saloranta and Andersen, 2004). *MyLake v2* (Markelov et al., 2019) operates on a daily time step to simulate 1 m vertical distributions of water temperature, phytoplankton, and dissolved and particulate substances, as well as interactions at the sediment-water interface (Saloranta and Andersen, 2007). *MyLake* is a well-established model and has been successfully used in multiple applications (e.g. Livingstone and Adrian, 2009; Moe et al., 2016; Woolway et al., 2017a; Kiuru et al., 2018; Pilla and Couture, 2021).

Details of recent updates to *MyLake* to enable simulation of differently functioning 'zones' of water bodies and enhanced phytoplankton representation functionality can be found in Chapter 4. To represent the hydrodynamics of QEII, the model was set up to comprise a shorter residence time, *faster-flowing tank* and a longer residence time, s*lower-flowing tank* (Figure 5-1). Whilst in the reservoir, these have a specific spatial configuration defined by bathymetry and the location of the inlets and outlet, in the model they are represented by conveniently and simply shaped domains. An eddy diffusion matrix, which governed the amount of lateral mixing between the two tanks, and an advection matrix, which specified the flows between them, were both set to exchange 2.5% of volume between each tank during each time step.

5.3.3 Future climate scenarios

The climate scenarios used in this analysis are a subset of those generated for the United Kingdom Climate Projections 2018 (UKCP18) daily global (60 km resolution) projections (Met Office Hadley Centre, 2018) for the Thames basin. Two Representative Concentration Pathways (RCPs) were used: RCP2.6 and RCP8.5. RCP2.6 represents a future with significant reductions to greenhouse gas emissions where radiative forcing will increase by 2.6 Wm^{-2} by 2100. RCP8.5 represents a scenario with unabated, very high greenhouse gas emissions, leading to an increase in radiative forcing of 8.5 Wm^{-2} by 2100. Thus, the scenarios envelop a range of plausible future climates.

We simulated four cases, defined by mid-century and late-century conditions for each of the RCP2.6 and RCP8.5 scenarios: $RCP2.6_{2040\text{-}2069}$, $RCP8.5_{2040\text{-}2069}$, $RCP2.6_{2070\text{-}2099}$ and $RCP8.5_{2070\text{-}2099}$. Each case represents one year, derived from the mean of a 30-year window of UKCP18 projections to reduce inter-annual variability. To capture real-world variability specific to QEII, data from 2018 was used to provide a daily variability signature that was superimposed on the smoothed, 30-year mean conditions for each case. To allow comparison between future and present conditions we defined a baseline case using the mean of the RCP2.6 projections for the 30-year window 2003-2033, on which the 2018 daily variability signature was also superimposed.

5.3.3.1 Model forcing inputs

The model requires inputs of data for the meteorological forcing variables global radiation, cloud cover, wind speed, air temperature and relative humidity. The closest site to the reservoir from which measurements of these variables were available was London Heathrow Airport (10.5 km to the north). To define the daily variability signature, raw observations for each of these variables for the whole of 2018 were smoothed using a five-day moving average and the daily difference between the moving average and the raw observation calculated. Daily air pressure and rainfall recorded at Heathrow Airport (Met Office, 2019) were kept the same for all five modelled cases (the baseline case, two mid-century cases and two late century cases).

5.3.3.2 Inflow temperature and volume

A statistically-derived data-based transfer function (TF) model was used for estimating water inflow temperature (Arismendi et al., 2014) from air temperature and global radiation under present and future climate cases. The model is a discrete-time TF derived directly from the available data (Environment Agency, 2018; Met Office, 2019; Findlay, 2022) using the Refined Instrumental Variable (RIV) algorithm (Young, 2015) implemented within the CAPTAIN Toolbox for Matlab™ (Taylor et al., 2007). The resulting model structure has a multi-input single-output model (Equation 5-1).

Equation 5-1

$$T_{R_t} = 1.794.T_{R_{t-1}} + 0.794.T_{R_{t-2}} + 0.1609.T_{a_{t-1}} + 0.0278.R_{g_{t-1}} - 0.1604.T_{a_{t-2}} - 0.0278.R_{g_{t-2}}$$

where: T_{R_t} is river temperature, T_{a_t} is air temperature, R_{g_t} is global radiation and *t* is time.

Future inflow volumes were predicted from projected future river flows at Kingston-upon-Thames (~8 km downstream of QEII) (Haxton et al., 2012; Prudhomme et al., 2013). The use of river flows as a proxy accounts for changes in water available for abstraction from the river, assuming no change to management or water demand (HR

Wallingford, 2020). Mean daily river flow for each day of the year was computed for a 30-year window centred on 2018 (2003-2033), and the difference compared to river flow in 2040-2069 and 2070-2099. A moving average was applied to calculate the percentage change in daily river flow versus current flow. For the mid-century and late-century cases, percentage changes in reservoir inflow volume were assumed to equal the percentage change in river flow, thus inflows that varied daily were synthesized. In the baseline case, we assumed no change to reservoir inflow volumes. Reservoir volume was maintained for all cases.

5.3.3.3 Phytoplankton data

Six functional groups of phytoplankton were simulated to reflect broadly the species composition observed in QEII at the reservoir outlet during 2018 (Figure 5-1): these comprised grazed and ungrazed groups of each of diatoms, green algae and cyanobacteria. Grazing pressures (represented by loss rate), size, growth rate, light requirement for growth, and settling velocity varied between these groups. In the analyses reported herein, the functional groups were only reported as *diatoms*, *green algae*, and *cyanobacteria*, because the grazed and ungrazed groupings were combined for each. Full details on parametrisation of each functional group can be found in Chapter 4.

5.3.3.4 Other model driving data

Two monitoring stations on the River Thames, located upstream (Wey tributary; 5.5 km) and downstream (Teddington Weir; 11.55 km) of the QEII inlet, were used as a proxy for inflow nutrient concentrations (Environment Agency, 2018). Samples (approximately monthly) were linearly interpolated to obtain mean daily values for 2018. The 2018 inflow nutrient concentrations were used for all cases. Bathymetry for QEII was digitised from a survey (2004) provided by the reservoir operator to 1 m intervals.

5.3.4 FPV deployments under future climate simulations

We focused on the impact of FPV surface coverage, which was varied in 10% increments from 0-100%, on wind forcing, solar radiation receipts and air temperature at the reservoir water surface. The simulated FPV installation was deployed centrally on QEII. As the two tanks are unequal in surface area, initially, the array is placed on the faster flow tank (the larger of the two tanks). When the faster flowing tank's remaining exposed area equals that of the slower flowing tank, the array is deployed equally between the two tanks to achieve a central deployment location.

5.3.4.1 Assumptions

We assumed that between the water's surface and the underside of the PV module, air temperature was warmed by 8%, global radiation was decreased by 94%, and wind speed was lowered by 95%. These assumptions are based on the findings of observations made at an FPV installation (Appendix B) and at a ground-based installation (Armstrong et al., 2016). Further details are available in Chapter 4.

5.3.5 Model calibration

MyLake was previously calibrated for use with QEII; see Chapter 4. In brief, we employed the Generalised Likelihood Uncertainty Estimation (GLUE) procedure (Beven and Binley, 1992). We used formalised limits of acceptability to account for the uncertainty associated with modelling environmental systems. We compared model output from multiple model runs with observed data (total chlorophyll-*a*, surface water temperature, stratification pattern and phytoplankton functional group proportions) to identify acceptable baseline simulation results and parameter sets. Acceptable simulations are defined by a fuzzy weighting function that returns a relative confidence weighting for each simulation depending on its position within the formalised limits of acceptability. To limit bias within the parameter sets, the parameter ranges comprised of all physically reasonable values for each parameter and were sampled 8,000 times using a Monte Carlo strategy. Seventy-five parameter sets were within the limits of acceptability for all simulations; the remaining 7,925 parameter sets were rejected and not used in the subsequent analyses.

5.3.6 Model output analysis

To summarise the impact of varying FPV coverage under different predicted future climates, we compared model output from each of the four future cases with the model output from the baseline case. Our analysis focused exclusively on the *faster-flowing tank,* as this zone contains the reservoir outflow and therefore determines the quality of water entering the treatment works and the potential operational implications. Given their importance in determining many biological and chemical processes in reservoirs, we focused on simulated surface water temperature at 1 m and the duration of the period in which the reservoir was thermally stratified. We used two metrics to define stratification duration. These were maximum stratification, the longest period of stratification in each simulation, and cumulative stratification duration, the total number of stratified days during the one-year simulation period. Stratification was defined using a threshold density gradient of 0.1 kg m^{-3} m^{-1} between adjacent 1 m layers (Gray et al., 2020). We also present phytoplankton biomass and species composition, with each group represented as a proportion of total chlorophyll-*a*, both at 1 m depth. The abundances and biomass (as chlorophyll-*a*) of phytoplankton functional groups are presented as proportions of the total for all groups rather than absolute values for visual clarity.

Given our use of the GLUE methodology, each case produced outputs from 75 model simulations. To capture the variability in outcomes, thus representing the uncertainty, we use the median, 2.5th and 97.5th percentiles of the range of values obtained in these multiple outputs for each variable. This provides the average outcome and the 95% confidence interval. For clarity, we present these data for the day of maximum (T_{max}) and minimum (T_{min}) water temperature and maximum total chlorophyll-*a* concentration (Chl-a_{max}) for each season, defined as: winter – December to February; spring – March to May; summer – June to August; autumn – September to November. Maximum and cumulative stratification duration were also presented in this way. Phytoplankton species composition is presented as a time series for the simulated year. Not all FPV coverage extents are shown in the main text but are presented in the supplementary information (section 5.7). The full time series for median water temperature and *chlorophyll-a* simulations, and their differences to the baseline, are presented in the supplementary information (section 5.7).

5.4 Results

5.4.1 The effect of FPV coverage on water temperature

The combined effects of climate warming and FPV coverage on T_{max} for each season are shown in Figure 5-2, highlighting a subset of FPV coverages. Additional FPV coverages and the effect on T_{max} are presented in the supplementary information (section 5.7). In all the future cases, T_{max} was raised relative to the baseline case in every season, with warming increasing between the mid-century and late-century cases for each RCP.

Increasing FPV percentage coverage progressively cooled T_{max}, demonstrating that FPV deployment acts to counter climate-induced warming.

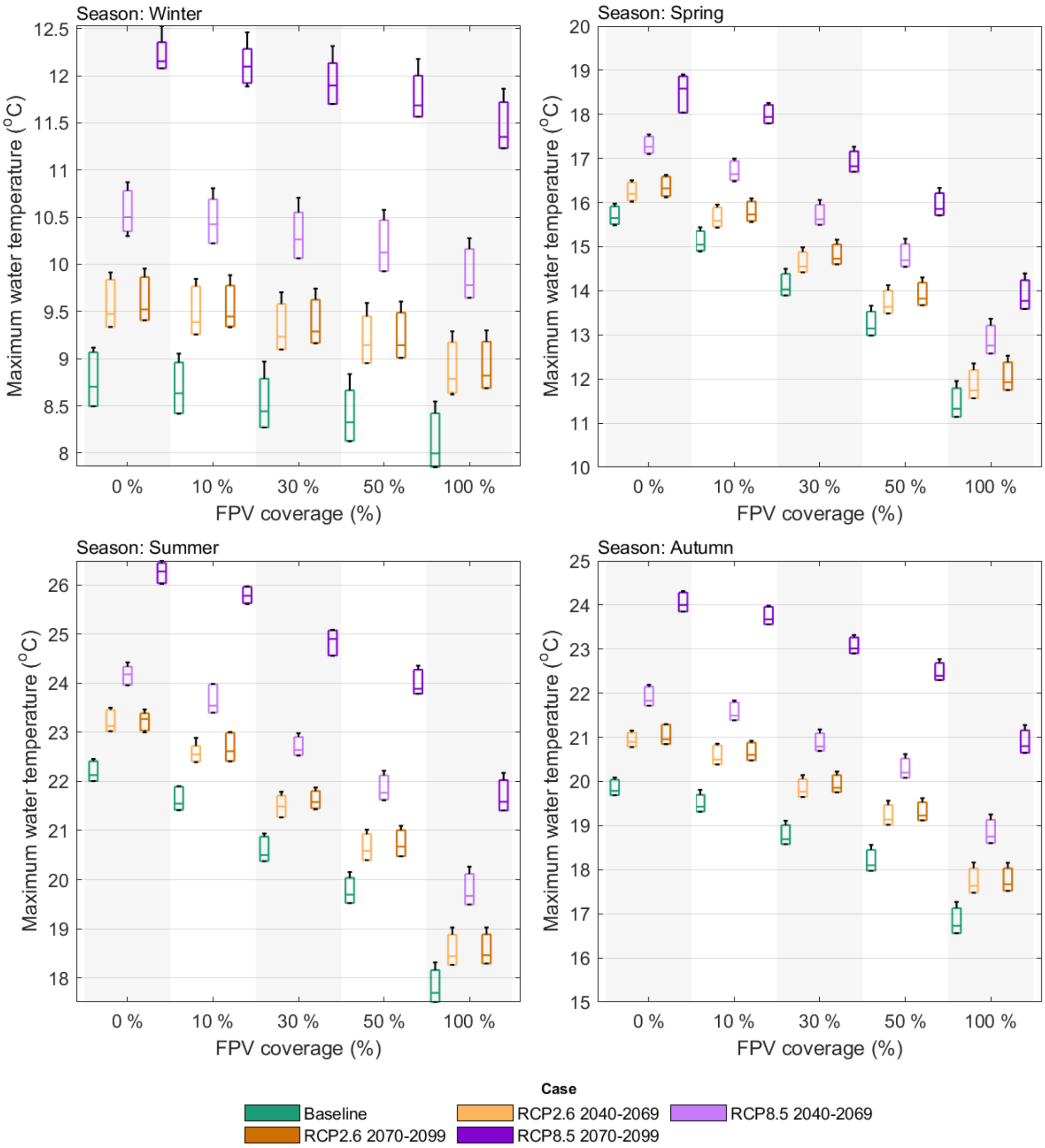


Figure 5-2 – Seasonal maximum water temperatures at 1 m for each case. Whiskers represent the minimum and maximum of the simulation results presented. The box represents the 2.5th & 97.5th percentiles, which gives a 95% confidence interval that simulation estimates fall within this range. 0% FPV coverage represents QEII reservoir simulated as a baseline case with no additional FPV coverage.

FPV coverage was able to fully offset changes to T_{max} in spring and summer for all four future cases, although the amount of coverage required to do this increased from RCP2.6 to RCP8.5 and from mid-century to late-century. In spring, the warming of T_{max} in the RCP2.6 cases was offset by FPV coverages of 10% (for the mid-century case) and 20% (for the late century case). For the RCP8.5 cases, these “offset coverages” were 30% and 60%, respectively. In summer, greater offset coverages were required. For the two

RCP2.6 cases, they were 20% (mid-century) and 30% (late century), and for the RCP8.5 cases, they were 40% and 90%, respectively. In autumn, changes to T_{max} were offset for both RCP2.6 cases and the RCP8.5 mid-century case. Autumn warming was offset by an FPV coverage of 30% (mid-century) and 40% (late-century) for the RCP2.6 cases and 70% for the RCP8.5 mid-century case, while in the RCP8.5 late-century case, even 100% FPV coverage was unable to fully offset the climate warming of T_{max}. During winter, no extent of FPV was able to offset fully the changes to T_{max} in any of the future cases. In all cases, FPV coverages in excess of these values cooled T_{max}, overcompensating for climate warming.

The results for T_{min} are shown in Figure 5-3 and supplementary information (section 5.7). As for T_{max}, and in line with our hypotheses, T_{min} increased in the future cases compared to the present day, with warming increasing between the mid-century and late-century cases. Increasing FPV coverage cooled T_{min} in all cases. FPV coverage was less effective at offsetting climate warming effects on T_{min} than T_{max}. Even 100% FPV coverage was unable to offset T_{min} increases in winter, spring, or autumn in both RCP8.5 cases, or in autumn for both RCP2.6 cases. FPV coverages of 70% and 90% were required to offset winter T_{min} warming in the RCP2.6 mid-century and late-century cases, respectively. In spring, FPV coverages of 50% and 70% offset climate warming of T_{min} for the RCP2.6 mid-century and late-century cases, respectively. Summer T_{min} increases were offset by 20% FPV coverage in both RCP2.6 cases and by 40% and 70% FPV coverage for the RCP8.5 mid-century and late-century cases, respectively.

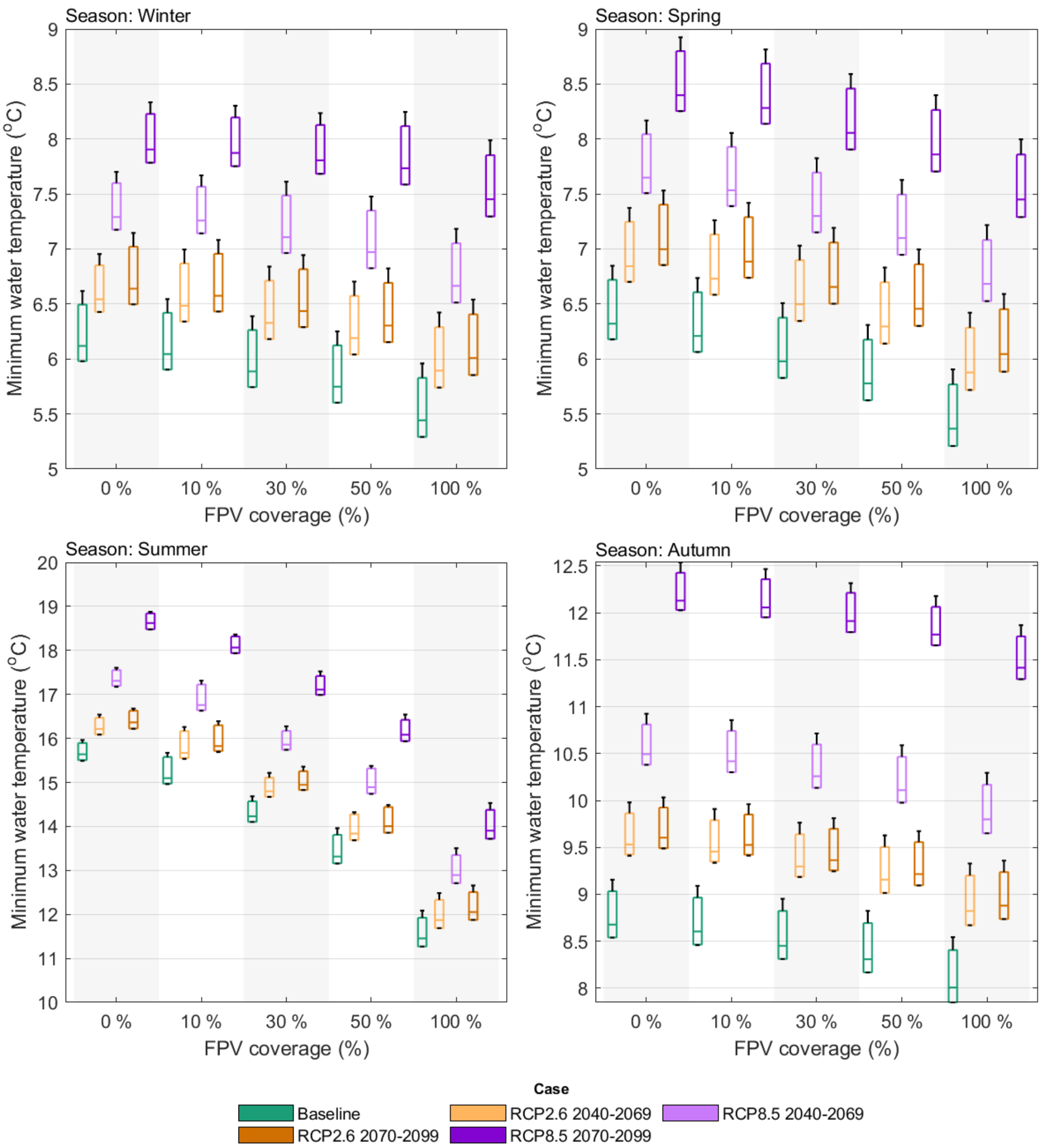


Figure 5-3 – Seasonal minimum water temperatures at 1 m for each case. Whiskers represent the minimum and maximum of the simulation results presented. The box represents the 2.5th & 97.5th percentiles, which gives a 95% confidence interval that simulation estimates fall within this range. 0% FPV coverage represents QEII reservoir simulated as a baseline case with no additional FPV coverage.

5.4.2 The effect of FPV coverage on stratification duration

Figure 5-4 shows the modelling results for stratification duration. With no FPV coverage, maximum stratification duration remained the same as in the baseline case for the two mid-century cases, reduced by 4 days for the RCP2.6 late-century case, and increased by 2 days for the RCP8.5 late-century case. FPV coverage of 10% reduced maximum stratification duration by 11 days for the baseline case, and caused smaller reductions in the future cases: 6 and 5 days by mid-century for the RCP2.6 and RCP8.5 cases, respectively, and 5 and 4 days for the corresponding late-century cases.

Cumulative stratification duration increased by 3 days for the RCP2.6 mid-century case, 8 days for the RCP8.5 mid-century case and 21 days for the RCP8.5 late-century case but reduced by 1 day for RCP2.6 late-century case compared to the baseline case. The increase in cumulative stratification from climate change was offset by 10% FPV coverage for the mid-century cases and 20% coverage for the RCP8.5 late century case. The reservoir experienced no stratification when FPV coverage exceeded 40% for both RCP2.6 cases and the RCP8.5 mid-century case. For the RCP8.5 late-century case, a 60% or greater FPV coverage prevented any thermal stratification.

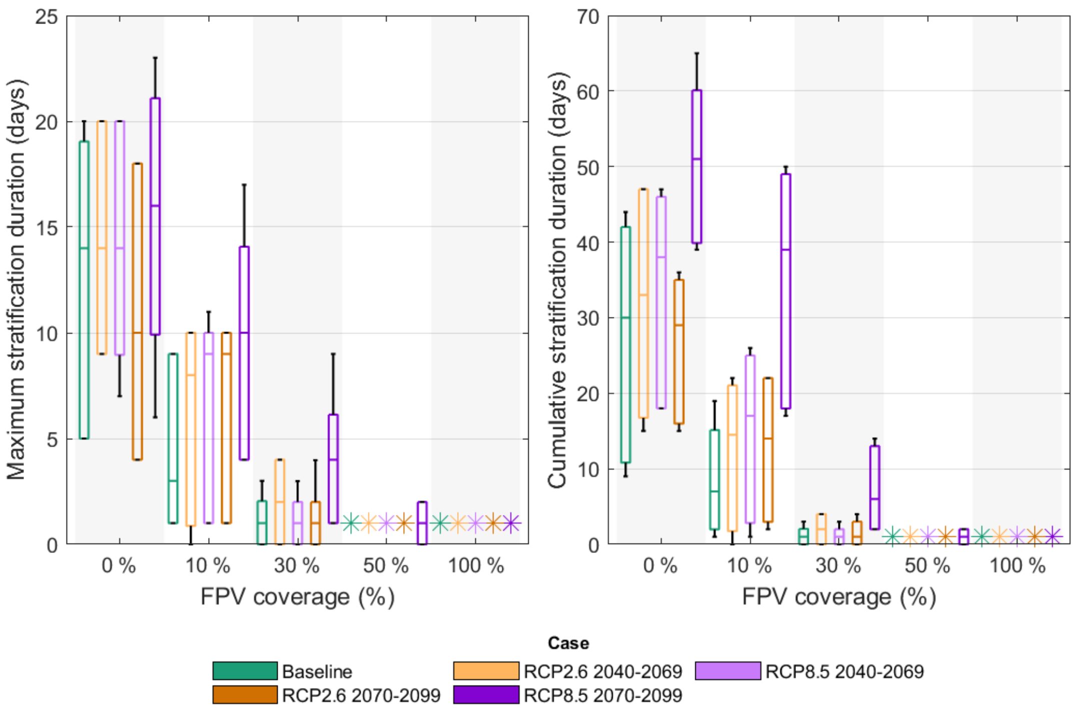


Figure 5-4 – Maximum and cumulative stratification duration for each case at varying FPV coverage. An asterisk indicates no prolonged stratification event occurred for the simulation. Whiskers represent the minimum and maximum of the simulation results presented. The box represents the 2.5th & 97.5th percentiles, which gives a 95% confidence interval that simulation estimates fall within this range. 0% FPV coverage represents QEII reservoir simulated as a baseline case with no additional FPV coverage.

5.4.3 The effect of FPV coverage on phytoplankton biomass and species composition

The modelling results for maximum chlorophyll-*a* concentration (taken as a proxy for phytoplankton biomass) are shown in Figure 5-5. Further FPV coverages and the effect on Chl-a_{max} throughout the full time series are presented in the supplementary information (section 5.7). With no FPV cover, Chl-a_{max} increased in all the future cases in comparison to the baseline case. The largest increases occurred in the summer under the late-century RCP8.5 case (> 24.94 μg L^{-1}). Increases to spring, summer and autumn

Chl-a_{max} compared to the baseline case were offset at 10% FPV coverage for both RCP2.6 cases and the RCP8.5 mid-century case. A greater FPV coverage of 20% was required to offset increases in spring, summer and autumn Chl-a_{max} for the RCP8.5 late-century case. FPV coverage in excess of these thresholds led to substantial reductions in Chl-a_{max} for all future climate cases compared to the baseline case with 0% FPV coverage.

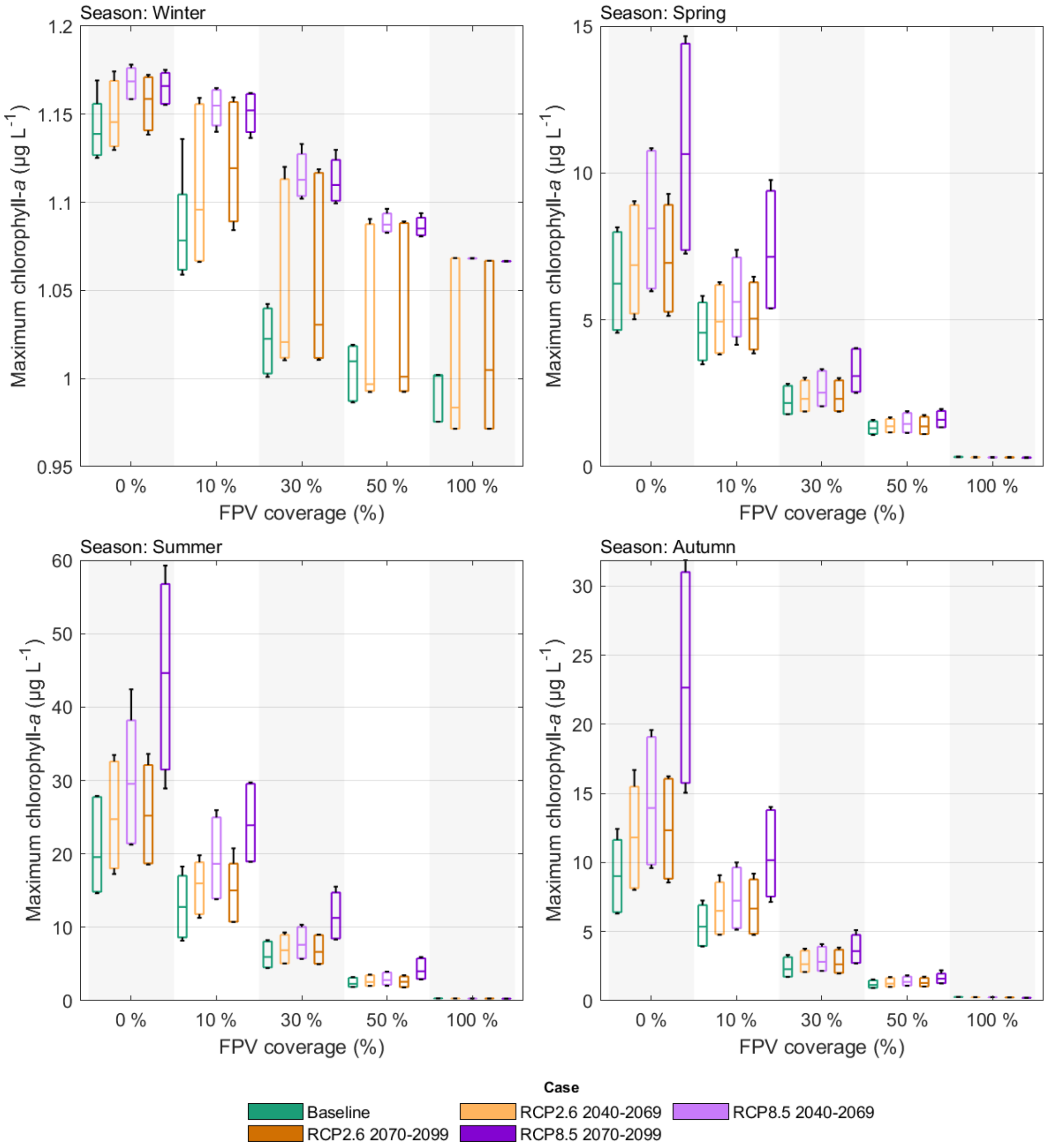


Figure 5-5 – Seasonal maximum chlorophyll-a at 1 m for each case with varying FPV coverage. Whiskers represent the minimum and maximum of the simulation results presented. The box represents the 2.5th & 97.5th percentiles, which gives a 95% confidence interval that simulation estimates fall within this range. 0% FPV coverage represents QEII reservoir simulated as a baseline case with no additional FPV coverage.

Whilst simulated Chl-a_{max} concentrations declined rapidly with increasing coverage (Figure 5-5), the relative proportion of phytoplankton functional groups varied, as shown in Figure 5-6. In the baseline case without FPV coverage, diatoms were the most dominant functional group (> 50% of total chlorophyll-*a*) until the end of July. The proportion of diatoms then reduced throughout August and early September to 37% of total chlorophyll-*a* on 23rd September. Green algae and cyanobacteria peaked at this point, accounting for 36% and 27% of total chlorophyll-*a*, respectively. After this peak, diatoms returned to exceeding 50% of total chlorophyll-*a* from mid-November. In general, FPV coverage typically removed the late-summer/early-autumn peaks in green algae and cyanobacteria. Diatoms remained the dominant functional group throughout the year and accounted for more than 50% of total chlorophyll-*a*.

Under the future climate cases, phytoplankton community composition changed in both proportion and timing. As in the baseline case, the phytoplankton community composition in the future climate cases was comprised primarily of diatoms until July. In the RCP2.6 mid-century case, green algae surpassed diatoms to become the dominant functional group from mid-August until mid-October, peaking at 38% of total chlorophyll-*a* on 20th September. In the RCP8.5 mid-century case, the switch to green algae dominance was earlier, commencing at the start of August. Green algae then peaked at 44% dominance in mid-September before diatoms returned to dominance at the end of October. Phytoplankton species composition followed a similar pattern in both RCP2.6 cases.

In the RCP8.5 late-century case diatoms lose their dominance between the start of July and mid-November, when they are surpassed by green algae, which peak at the end of September, when they account for 51% of total chlorophyll-*a*. Unlike the other future cases with 0% FPV coverage, in the RCP8.5 late-century case, cyanobacteria increase in abundance sufficiently to account for a greater proportion of total chlorophyll-*a* than diatoms between the start of August and mid-November. Cyanobacteria dominance peaks in mid-October, when they account for 33% of total chlorophyll-*a*, compared to 20% for diatoms. Under future climates, the introduction of FPV coverage helped offset the changes to phytoplankton functional groups. Green algae continued to account for a greater proportion of total chlorophyll-*a* than cyanobacteria, as in the baseline case.

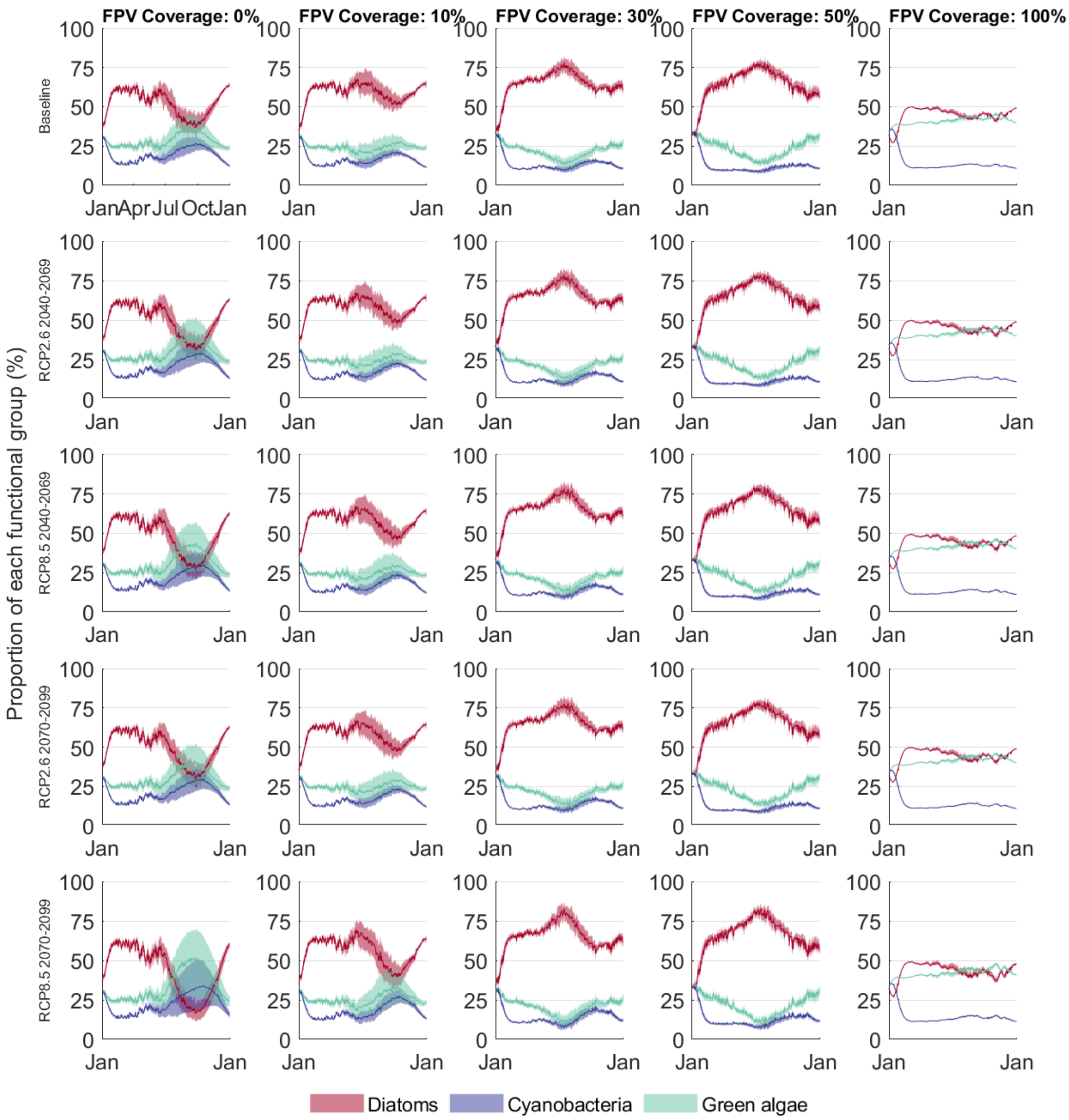


Figure 5-6 – Proportion of total chlorophyll-a represented by each phytoplankton functional groups for the simulated period. Each future climate case and a subset of the simulated FPV coverage is shown. The initial phytoplankton functional group proportions were set evenly, therefore, the first 30 days of simulations are model run-in time and should be ignored. 0% floating solar coverage represents QEII reservoir simulated as a baseline case.

5.5 Discussion

Our results show that FPV coverage can offset future climate change impacts in our modelled simulations, although the extent to which that can be achieved varies between seasons, the extent of future climate warming (RCP case), and the in-lake parameter. Below, we discuss the ability of FPV coverage to offset changes to water temperature, stratification duration, and phytoplankton biomass and species composition. We then discuss the effectiveness of FPV as a tool for mitigating climate change impacts on reservoirs and additional considerations required when assessing individual deployments.

5.5.1 The ability of FPV coverage to offset water temperature warming

Our first hypothesis, that FPV coverage can offset reservoir water temperature warming under future climates was partly supported. FPV coverage was able to offset increases to maximum seasonal temperatures in the spring and summer, although it was only partially effective in autumn (for the lower emissions RCPs) and was not effective in winter. FPV was less effective at offsetting increases to minimum seasonal temperatures, especially during winter, spring and autumn. However, FPV still provided a dampening effect, limiting water temperature warming under climate change. This highlights the value of FPV in buffering the most severe changes to extreme reservoir warming.

In seasons where FPV coverage was unable to offset changes to QEII's simulated future water temperature, the likely cause is the reservoir inflow temperature and volume. In the cooler months of the year, the difference in temperature between reservoir inflow and outflow (the through-flow) leads to net advective heating of the reservoir (Livingstone and Imboden, 1989). Advection of comparatively warmer inflow water from the inlets on the reservoir bed causes in-reservoir warming that cannot be counteracted by the shading and sheltering of an FPV array on the reservoir. Given the short residence time of the reservoir, this advective heat flux is likely to provide an important source of heat, especially during winter when surface heat fluxes are diminished (Livingstone and Imboden, 1989; Fenocchi et al., 2017). However, potential changes in reservoir management and future water demand cause uncertainties in predicting future inflow volumes and water temperature (HR Wallingford, 2020).

Whilst their primary purpose is to store water, reservoirs can become important habitats for aquatic life. Many of the ecosystem services provided by reservoirs, beyond water storage, are highly dependent on the functioning of the ecosystems that have formed within them (Saulnier-Talbot and Lavoie, 2018), including fisheries, tourism, and recreation. However, warmer water temperatures will likely induce heatwaves, increased algal growth, fish die-offs and increased metabolic processes, which have consequences for ecosystem service provision. Consequently, FPV may offer an effective means to reduce the undesirable impacts of climate change on reservoir water temperatures.

Reservoirs under future climates are increasingly threatened by periods of extreme heat, which can disrupt reservoir functioning (Woolway et al., 2021a). Extreme heat events can lead to fish die-off events when species reach their thermal maxima (Till et al., 2019; Miranda et al., 2020). Large-scale fish die-off events can degrade water quality, affecting reservoir operations and lead to concerns from the public about decomposition odours (Thronson and Quigg, 2008; Godinho et al., 2019). FPV coverage limits the magnitude of water temperature warming under future climates, reducing the likelihood of reservoir heatwaves. FPV deployed at a sufficient coverage might help protect fish from reaching their thermal maxima. For example, an FPV coverage of 40% reduced maximum summer temperature by 1.95 °C in the RCP8.5 late-century case compared to no FPV coverage. Further, the increased frequency of reservoir heatwaves are expected to lead to phenological change (Thackeray et al., 2016) and food-web de-synchronisation (Thackeray et al., 2013). Given the fragmented nature of many reservoirs, aquatic organisms will have limited options to disperse (Merritt and Wohl, 2006), resulting in reduced competition in reservoirs and the proliferation of non-native species (Rahel and Olden, 2008; Muhlfeld et al., 2014).

Climate warming is accelerating metabolism in water bodies due to the exponential response of metabolic processes to water temperature (Kraemer et al., 2017). Changes in reservoir metabolism can have substantial impacts on reservoir biota, modifying species interactions, food webs, disrupting life histories and changing animal behaviour (Ficke et al., 2007; Jeppesen et al., 2010; Staehr et al., 2012b). Some of these effects, in some locations, could be advantageous; for example, in raw water reservoirs, reduced planktivorous fish stocks could improve water quality by lowering nutrient concentrations and increasing water clarity (Bernes et al., 2015). However, other consequences of changed reservoir metabolism are detrimental, for example, the effect on greenhouse gas emissions. Accelerated reservoir metabolism can increase methane and carbon dioxide emissions, contributing to further climate warming (Tranvik et al., 2009; Kraemer et al., 2017). Consequently, given the cooling effect of FPV coverage, FPV deployment locations could be selected in order to minimise undesirable and maximise desirable changes to reservoir metabolism under future climates.

Whilst concerns of temperature change impacts on ecological processes primarily focus on increases; conversely, smaller reductions in future minimum water temperatures with FPV coverage in winter may be considered beneficial to reservoir operations. Reduced water temperatures (< 5 °C), as seen with increasing FPV coverage in the baseline case, could lead to increased tensile stresses in the water distribution network, increasing the incidence of iron pipe bursts and fractures (Habibian, 1994; Jesson et al., 2010). Water leaks can cause damage to surrounding infrastructure and reduce water quality with the intrusion of pathogens into the distribution network (Westrell et al., 2003; Mora-Rodríguez et al., 2015). In the winter months, avoiding extreme cold water temperatures could be considered beneficial by reservoir operators.

5.5.2 The ability of FPV coverage to offset stratification duration increases

Our simulations supported our second hypothesis that FPV coverage can reduce stratification duration. Thermal stratification is one of a reservoir's most important physical characteristics, and prolonged stratified periods can degrade water quality by changing biological and chemical processes (Woolway et al., 2021b). Under future climates there was an increase in maximum stratification duration (for the RCP8.5 mid-century case) and increased cumulative stratification duration (for the RCP2.6 mid-century and both RCP8.5 cases), suggesting a potential threat to water quality. However, only modest FPV coverage was required to achieve substantial reductions in stratification duration. For example, in the RCP8.5 late-century case, an FPV coverage of 30% reduced the median cumulative stratification duration by 88%. Such changes could have significant benefits for water body processes, properties, and ultimately ecosystem service delivery.

Periods of thermal stratification can facilitate oxygen depletion from bottom waters (where oxygen is used for biological and chemical processes) by preventing oxygen replenishment from the surface (Jankowski et al., 2006; Boehrer and Schultze, 2008; Foley et al., 2012). Oxygen depletion can have wide-ranging impacts on reservoir function, often acting as a catalyst for water quality problems. For example, hypoxia could lead to fish die-off events in productive reservoirs, degrading water quality and disrupting food webs (Till et al., 2019). Further, deoxygenation of bottom waters due to prolonged stratification duration may facilitate the internal loading of phosphorus from bed sediments, propagating phytoplankton blooms that reduce water quality and reservoir amenity (Hupfer and Lewandowski, 2008; North et al., 2014). Stratification-induced anoxia may also increase methane production from reservoirs (Bastviken et al., 2011; Vachon et al., 2019). However, FPV coverage may limit surface exchanges of dissolved oxygen, given reduced wind shear at the air-water interface (Armstrong et al., 2020; Andini et al., 2021). Therefore, reservoir oxygen levels could still be lower than in the baseline case, particularly at greater FPV coverage, even with reduced stratification duration (Chateau et al., 2019).

Whilst FPV coverage successfully reduced stratification duration, offsetting the effects of future climates, stratification events in QEII were often short regardless. The increases to cumulative stratification duration in QEII under future climate conditions demonstrate a discontinuous polymictic mixing regime, characterised by short irregular stratified periods that are disrupted by frequent mixing events (Kalff, 2002). Even under the higher climate warming scenario (RCP8.5 late-century case), the maximum stratified period lasted for a median of 16 days, a period brief enough for complete anoxia to be unlikely, although partial deoxygenation may occur in sufficiently productive reservoirs (Jane et al., 2021). However, in years with low wind speeds, low inflow or higher air temperatures, the mixing regime could shift to monomictic, characterised by a single period of prolonged stratification (Woolway et al., 2017a), increasing the likelihood of detrimental deoxygenation events. In other water bodies that exhibit prolonged stratification, the ability of FPV coverage to limit increases to stratification duration under future climates may improve reservoir water quality.

5.5.3 The ability of FPV coverage to limit the growth of phytoplankton and proliferation of nuisance phytoplankton groups

Our simulations supported our third hypothesis that FPV coverage limits the growth of phytoplankton and prevents the dominance of nuisance phytoplankton groups under future climates. Increases in Chl-a_{max} concentrations under predicted future climates were offset with low FPV coverage (<20%) in all seasons. For example, in the late-century RCP8.5 case, summer Chl-a_{max} concentrations were simulated to increase by 128% in the

QEII reservoir. However, 10% FPV coverage could limit this increase to just 22%, while a 30% FPV coverage reduced summer Chl-a_{max} concentrations by 42% compared to the baseline case with no FPV coverage. However, future phytoplankton biomass and species composition will also be determined by inflow nutrient concentrations under future climates, which were not modified in this study.

In agreement with our 0% FPV coverage simulations, studies have shown that climate change will lead to an increased prevalence of cyanobacteria (e.g. *Aphanizomenon*), which are considered a nuisance phytoplankton group. Cyanobacterial blooms can cause oxygen depletion (Paerl and Huisman, 2009), increase turbidity (Jeppesen et al., 2015) and release toxins (Gallina et al., 2017). Different water treatment processes may be required to mitigate these impacts, increasing water treatment costs (Dunlap et al., 2015; Watson et al., 2016). Cyanobacterial blooms can also affect tourism and recreation, as they pose a health risk to humans, livestock, and pets (Steffensen, 2008). However, FPV coverage > 20% reduced total phytoplankton biomass sufficiently to offset any net increase in cyanobacteria concentrations, thus presenting FPV as a potential water management tool for eutrophic water bodies.

Whilst FPV are likely to decrease cyanobacteria, they may increase diatoms. Filamentous diatoms (e.g. *Melosira, Asterionella*) and some colonial green algae (e.g. *Scenedesmus*) are also considered nuisance phytoplankton groups as they can disrupt water treatment processes by blocking filters (Henderson et al., 2008). FPV coverage reduced cyanobacteria and green algae peaks between late summer and early autumn, allowing diatoms to proliferate. However, whilst FPV coverage of > 10% increased the dominance of diatoms, the change was compensated by reductions in total chlorophyll-*a* concentration caused by FPV coverage.

5.5.4 The effectiveness of FPV as a tool for offsetting climate change impacts on reservoirs

FPV has the dual benefits of providing low-carbon electricity generation and potentially acting as a management tool for offsetting changes to water temperature, stratification duration, and phytoplankton biomass and species composition under future climates. Although the FPV coverage required to offset future warming varies by season, focusing on the maximum summer temperature, we can elucidate the FPV coverages required to offset predicted changes. Mid-century summer maximum temperature changes could be offset with coverage of 20% (RCP2.6) or 40% (RCP8.5). The late-century cases have considerable variation in the FPV coverage required to offset changes in summer maximum temperature, given the large difference in emissions, with 30% (RCP2.6) and 90% (RCP8.5) required. The FPV coverage required to offset changes in summer Chl-a_{max} and cumulative stratification duration were considerably lower than for maximum summer temperature. Changes in summer Chl-a_{max} and cumulative stratification duration were offset for both mid-century cases and for the late-century RCP2.6 case with 10% FPV coverage, with 20% FPV coverage sufficient to offset changes caused by the late-century RCP8.5 case.

However, FPV coverage should not be the only consideration when designing an FPV array. Firstly, siting location on the host water body can control water body response (Chapter 4). Our simulations considered an FPV array deployed centrally on QEII. However, variations in siting location, i.e., deployment on areas with different characteristics (e.g., flow rates), will modulate response (Chapter 4). Second, our simulations focused exclusively on QEII. Whilst the simulations offer insight into the potential ability of FPV coverage to offset climate change, the impacts will vary between individual water bodies. In particular, our simulations showed the importance of inflow temperature in determining seasonal water temperature response. Reservoirs in other climate zones will be subjected to different inflow conditions that could have a net cooling effect (e.g. Richards et al., 2012). Third, our simulations show that a very high FPV coverage would be required to mitigate projected water temperature warming under the late-century high emissions (RCP8.5) case. However, such high FPV coverages may be incompatible with other water body uses, like recreation and fisheries (Exley et al., 2021b). Further, high FPV coverage may have other impacts on the hosting water body not simulated in this study. For example, reduced sunlight under an FPV array limits the UV degradation of dissolved organic carbon (Armstrong et al., 2020) and pathogens (Mathijssen et al., 2020), an important form of natural water treatment.

Our simulations show that FPV could offer a viable means to manage future climate impacts on reservoirs, limiting or offsetting changes to water temperature, stratification duration and phytoplankton biomass and species composition. Future research should investigate the integration of FPV as a management tool in combination with existing management interventions (e.g. hydrodynamic manipulations, catchment interventions or artificial mixing), where low FPV coverages may generate the same co-benefits as a larger coverage alone.

5.6 Conclusion

The deployment of FPV on reservoirs is accelerating, with growth forecast to continue globally as the energy transition continues. FPV deployment is a long-term perturbation on a water body, so understanding its potential impacts both under present and future climates is vital to minimising concomitant issues. This study shows that reservoir managers can use FPV coverage to partially or wholly offset or even over-compensate for changes in reservoir water temperature, stratification duration, phytoplankton biomass and species composition under future climates. We identified considerable differences in the FPV coverage required, depending on the season, future emissions levels and desired management goals. Higher FPV coverages are needed to offset water temperature changes in cases with higher emissions. However, lower FPV coverages were sufficient at offsetting changes to phytoplankton biomass at all emissions concentrations. FPV could be used as an effective tool for managing climate change impacts on reservoirs, but the specifics of each deployment must be taken into account to ensure suitability and preservation of water body ecosystem service provision.

5.7 Supplementary information

5.7.1 Transfer function model

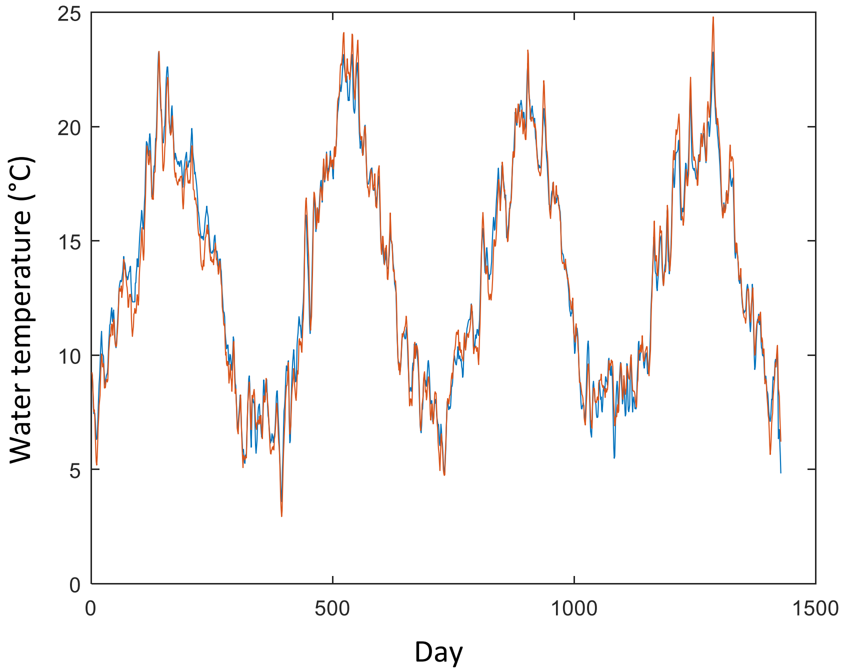


Figure S 5-1 – Performance of transfer function model for estimating water inflow temperature from air temperature and global radiation under present and future climate. Blue = observed, orange = simulated.

Model fit

R^2 = 0.98 RT2

RMSE = 0.63

5.7.2 Water temperature for all FPV coverages

5.7.2.1 Maximum water temperature

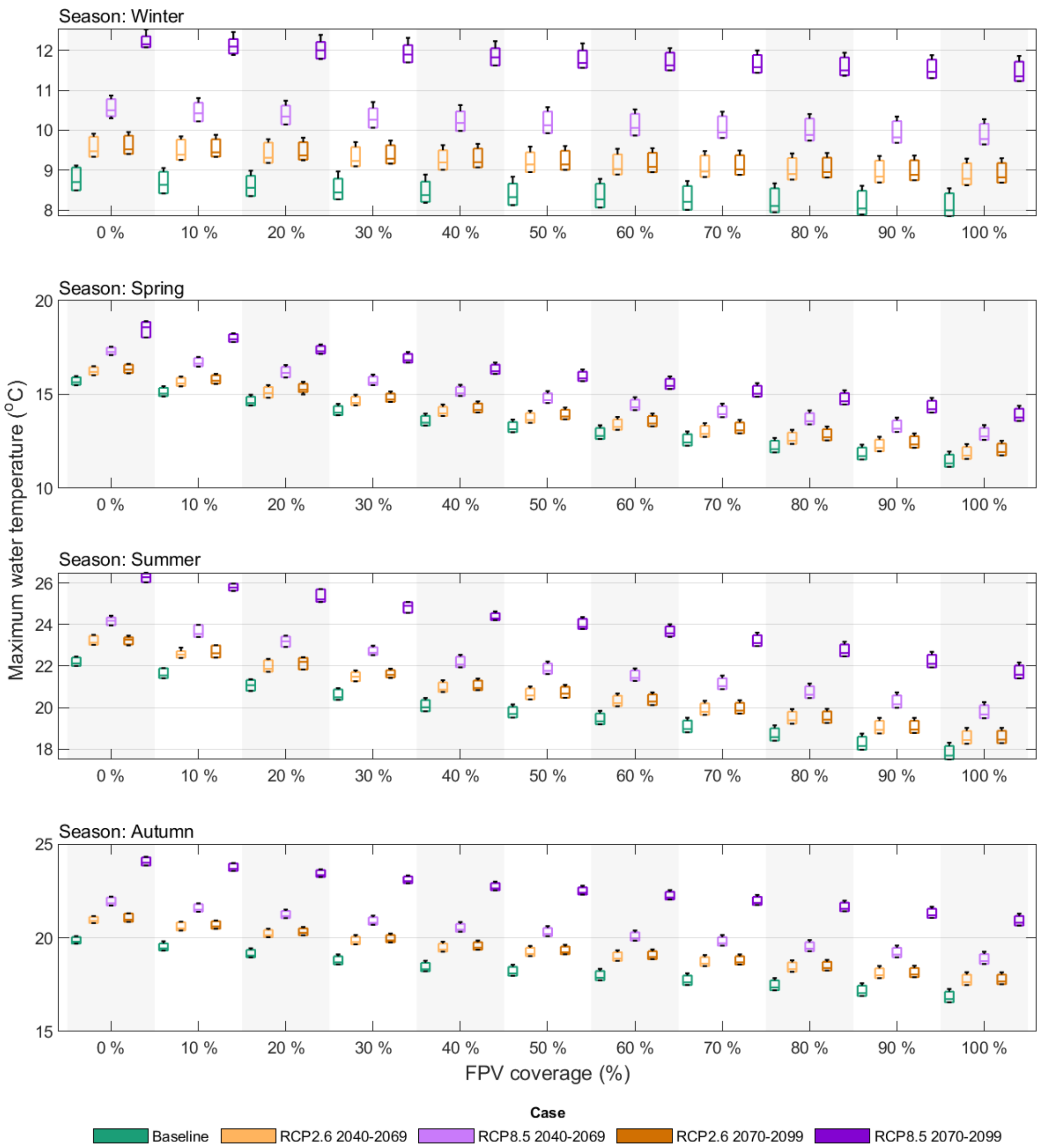


Figure S 5-2 – Seasonal maximum water temperatures at 1 m for each case. Whiskers represent the minimum and maximum of the simulation results presented. The box represents the 2.5th & 97.5th percentiles, which gives a 95% confidence interval that simulation estimates fall within this range. 0% FPV coverage represents QEII reservoir simulated as a baseline with no additional FPV coverage.

5.7.2.2 Minimum water temperature

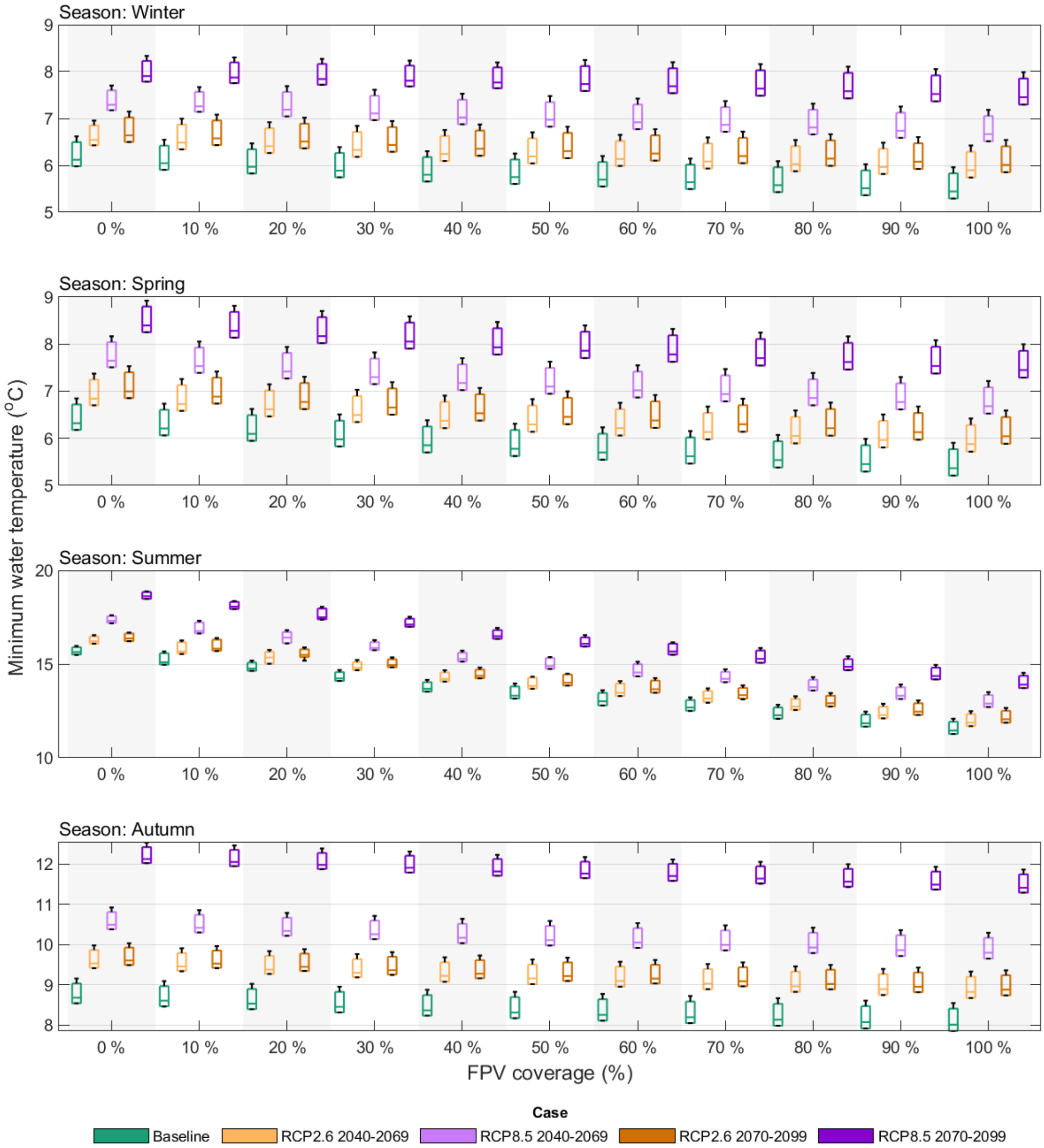


Figure S 5-3 – Seasonal minimum water temperatures at 1 m for each case. Whiskers represent the minimum and maximum of the simulation results presented. The box represents the 2.5th & 97.5th percentiles, which gives a 95% confidence interval that simulation estimates fall within this range. 0% FPV coverage represents QEII reservoir simulated as a baseline with no additional FPV coverage.

5.7.2.3 Annual water temperature (median)

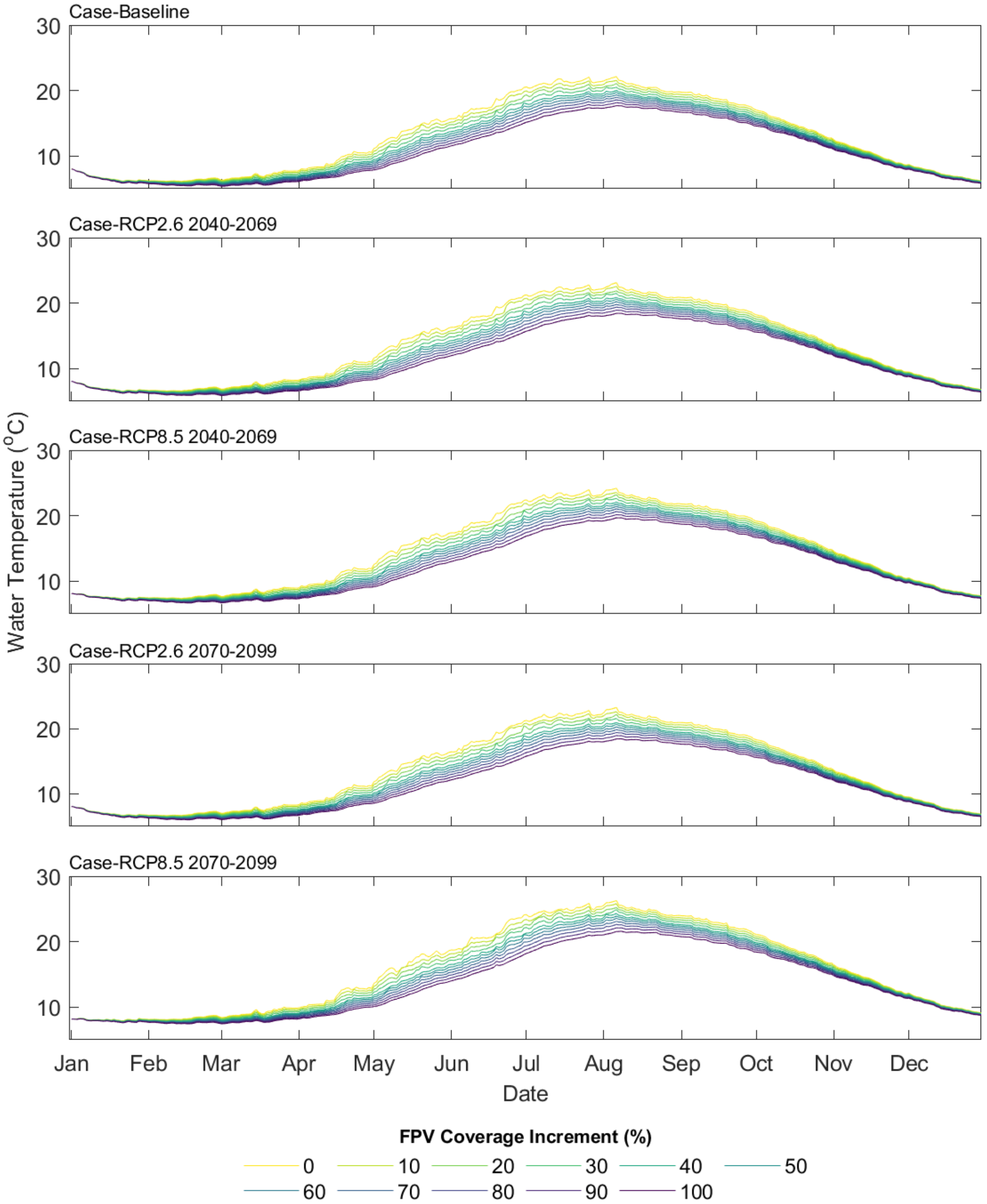


Figure S 5-4 – Median water temperature for 2018 by FPV coverage.

5.7.2.4 Annual water temperature difference (median)

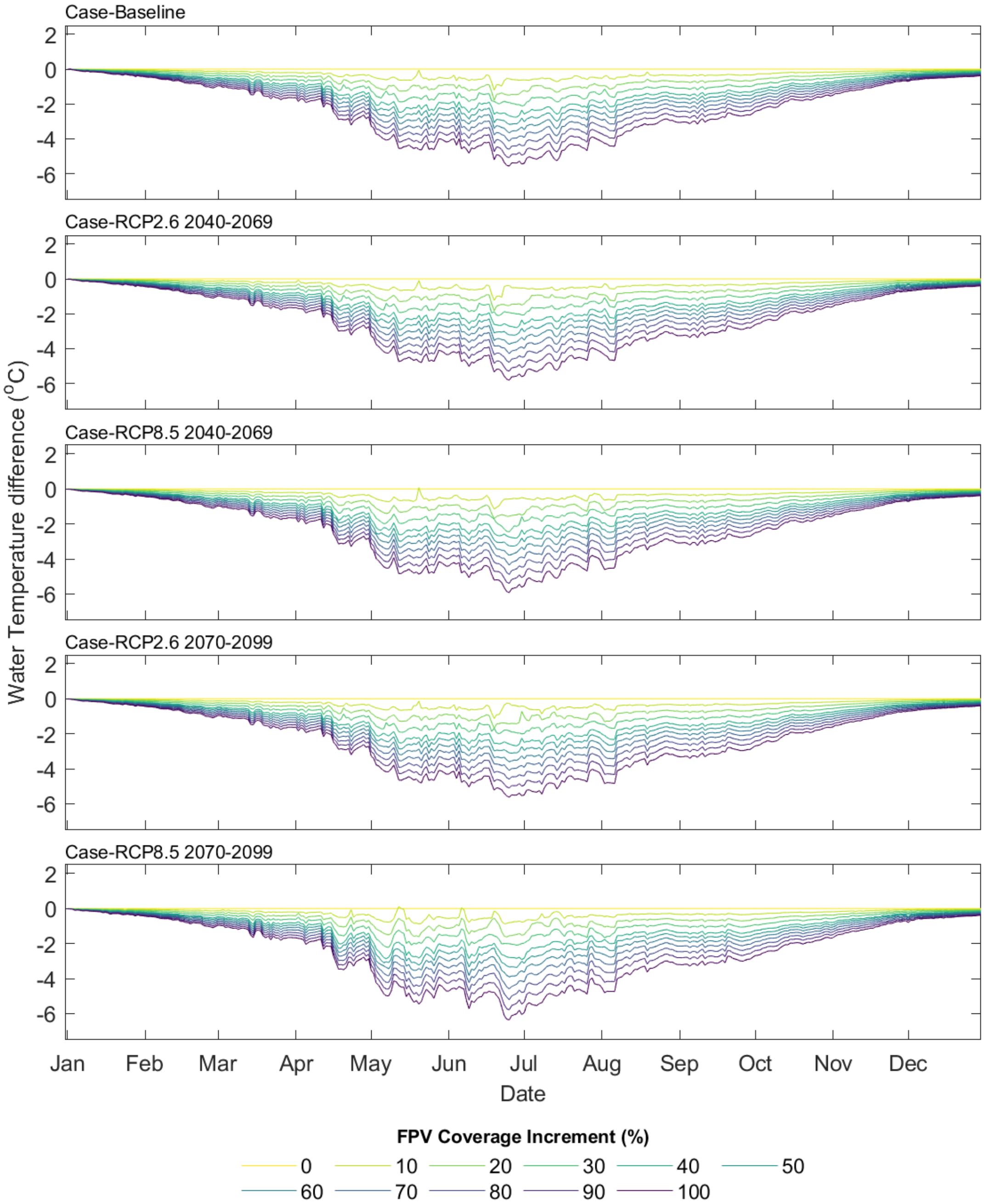


Figure S 5-5 – Median water temperature difference (between simulated and baseline) for 2018 by FPV coverage.

5.7.3 Stratification duration

5.7.3.1 Maximum stratification duration

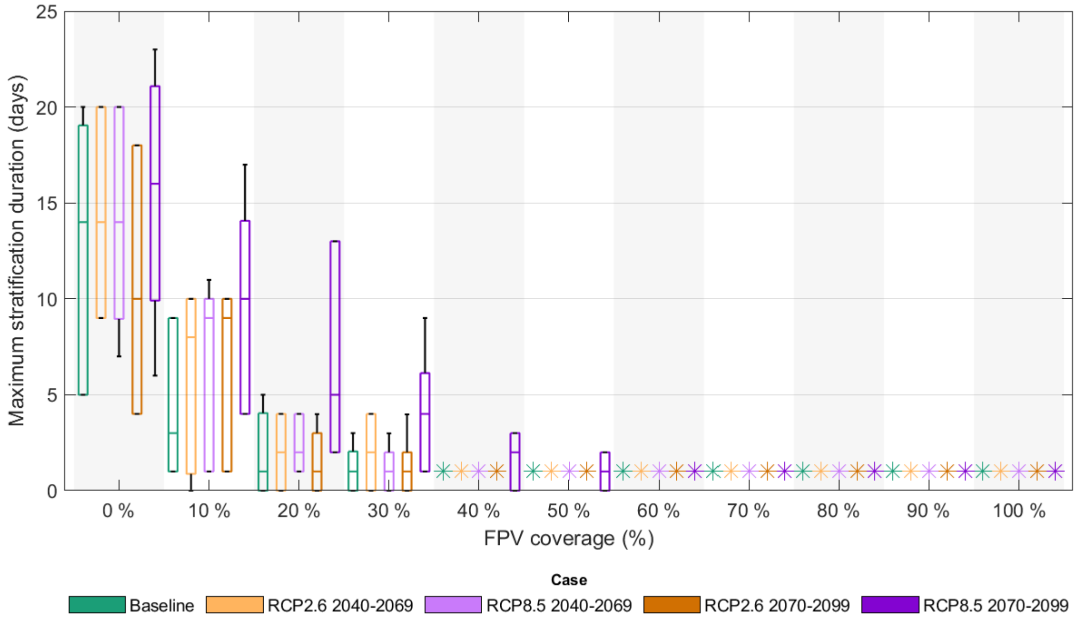


Figure S 5-6 – Maximum stratification duration for each case at varying FPV coverage extent. An asterisk indicates no prolonged stratification event occurred for the simulation. Whiskers represent the minimum and maximum of the simulation results presented. The box represents the 2.5th & 97.5th percentiles, which gives a 95% confidence interval that simulation estimates fall within this range. 0% FPV coverage represents QEII reservoir simulated as a baseline with no additional FPV coverage.

5.7.3.2 Cumulative stratification duration

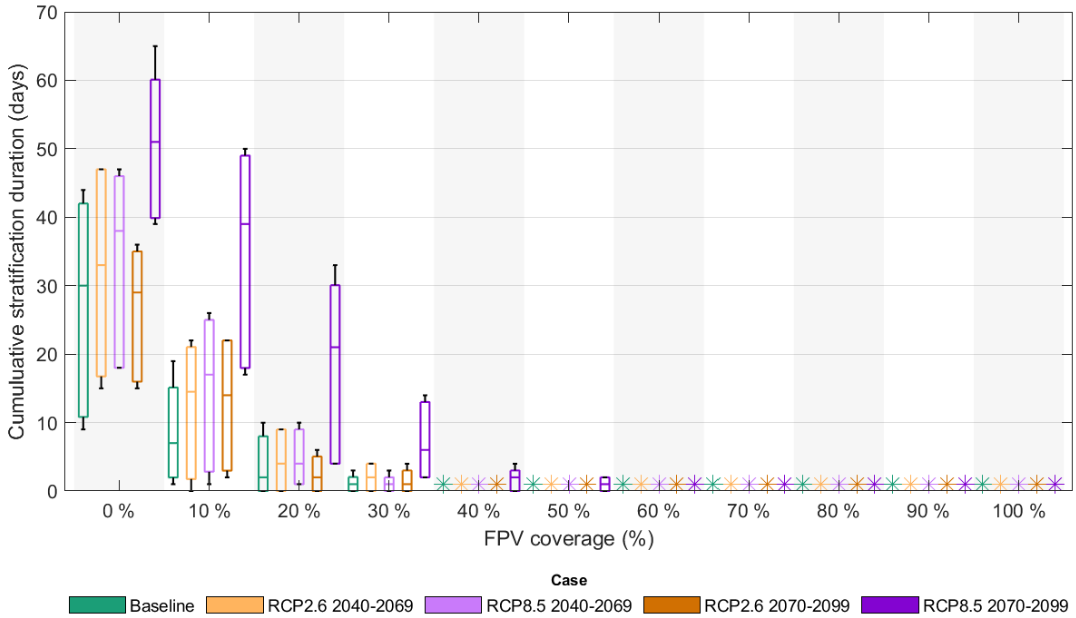


Figure S 5-7 – Cumulative stratification duration for each scenario at varying FPV coverage. An asterisk indicates no prolonged stratification event occurred for the simulation.

5.7.4 Chlorophyll-*a* concentration for all FPV coverages

5.7.4.1 Maximum chlorophyll-a concentration

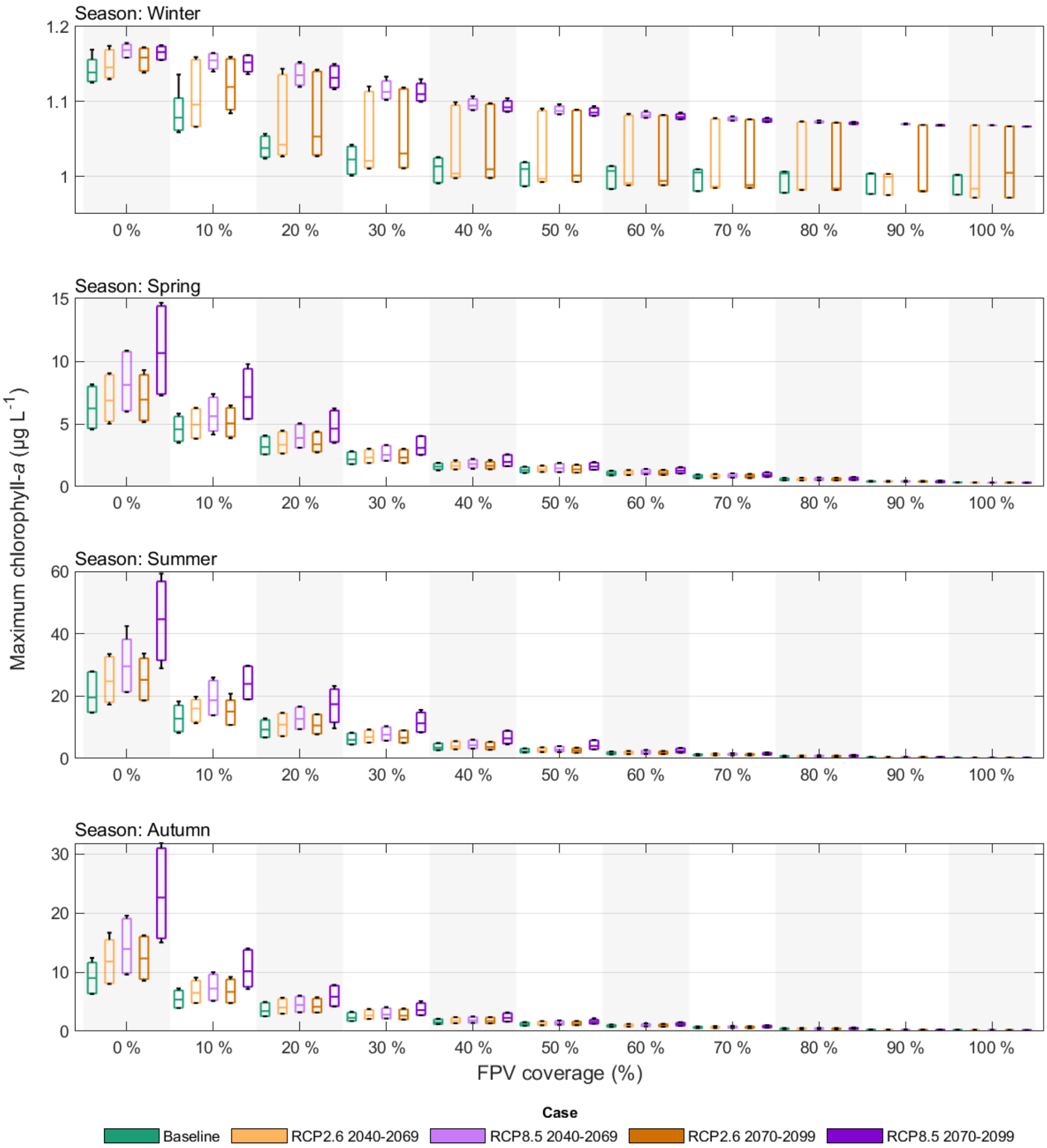


Figure S 5-8 – Seasonal maximum chlorophyll-a at 1 m for each case with varying FPV coverage. Whiskers represent the minimum and maximum of the simulation results presented. The box represents the 2.5th & 97.5th percentiles, which gives a 95% confidence interval that simulation estimates fall within this range. 0% FPV coverage represents QEII reservoir simulated as a baseline with no additional FPV coverage.

5.7.4.2 Annual absolute total chlorophyll-a (median)

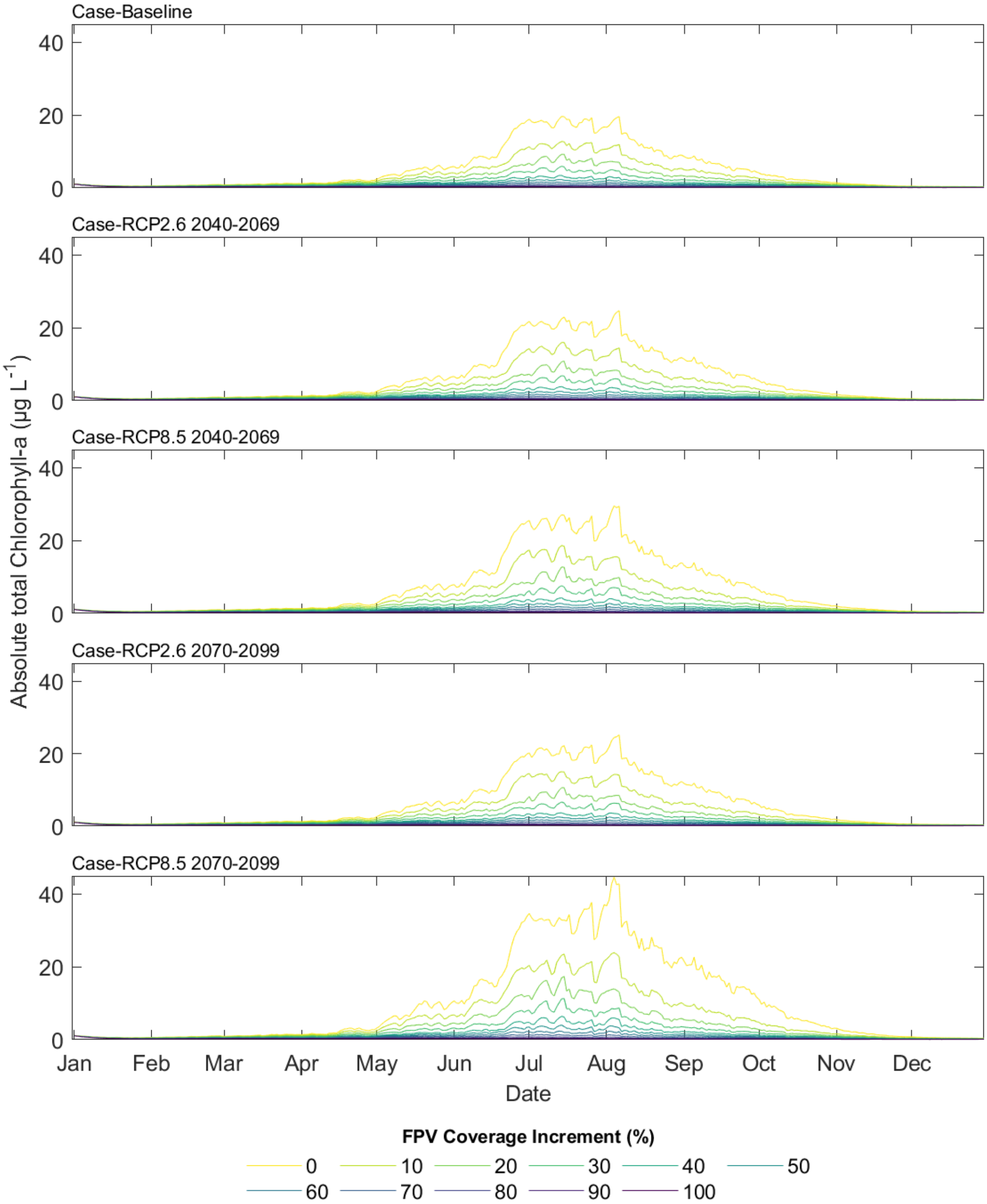


Figure S 5-9 – Median total chlorophyll-a concentration for 2018 by FPV coverage.

5.7.4.3 Annual absolute chlorophyll-a difference (median)

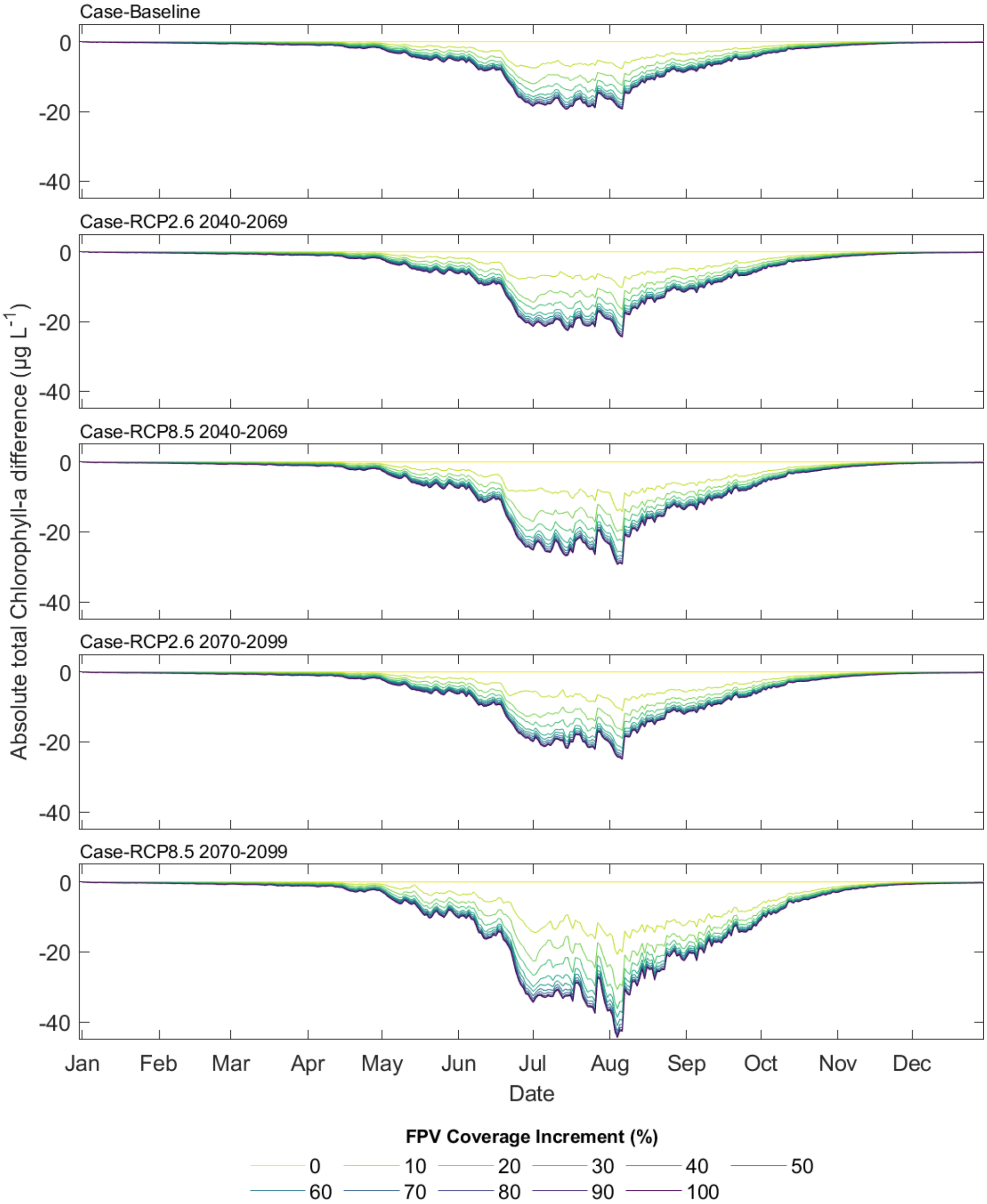
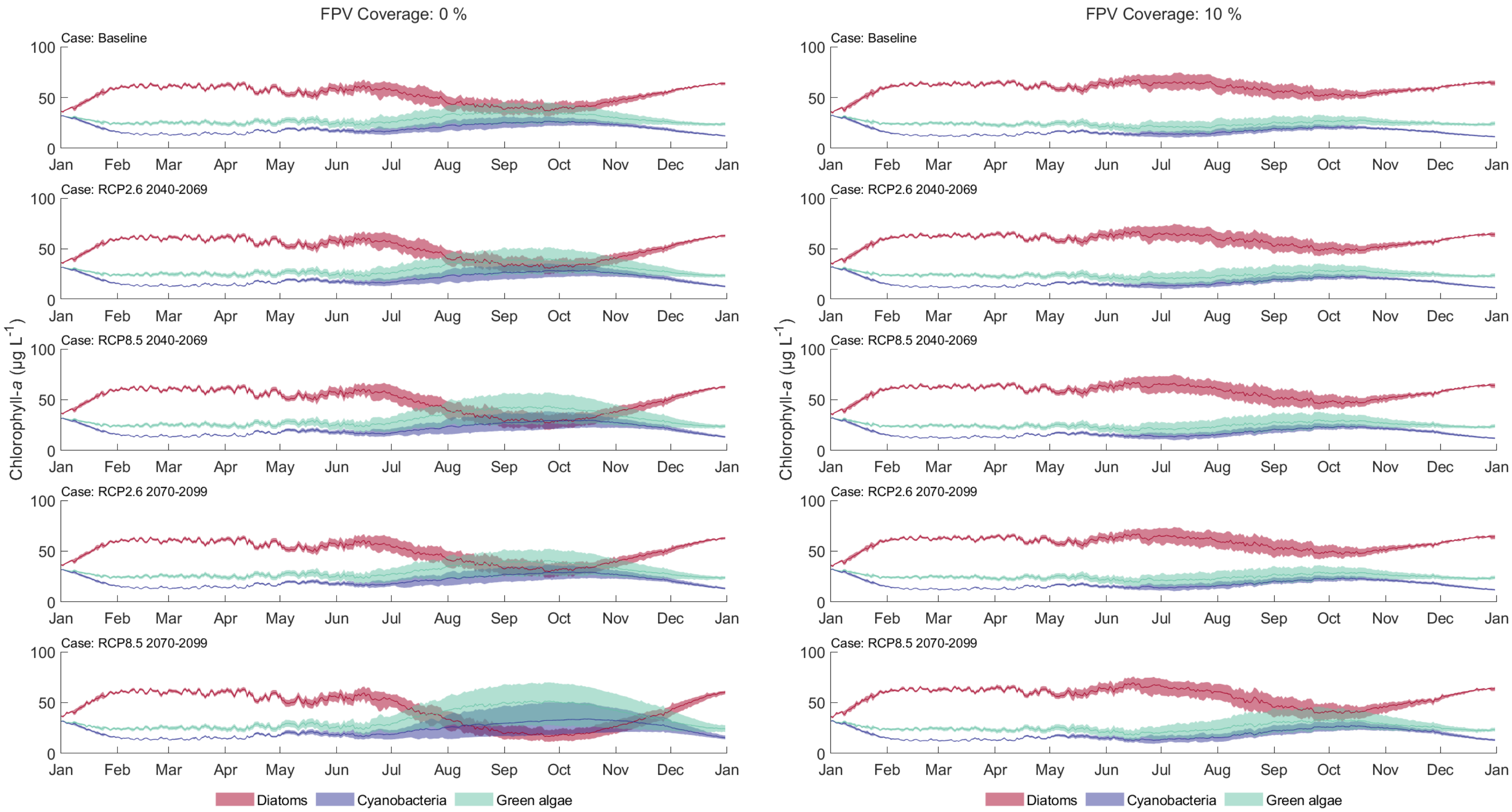
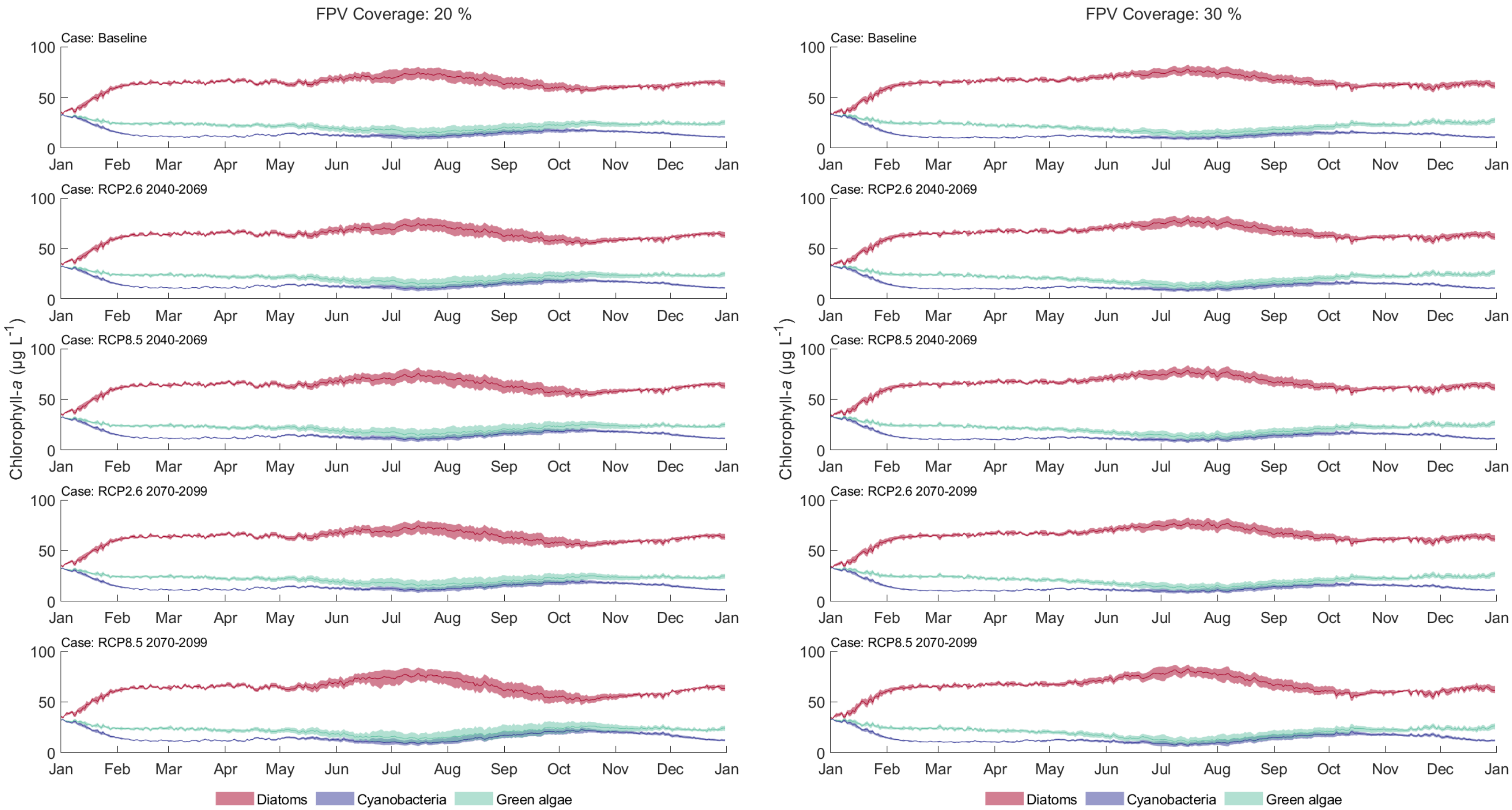
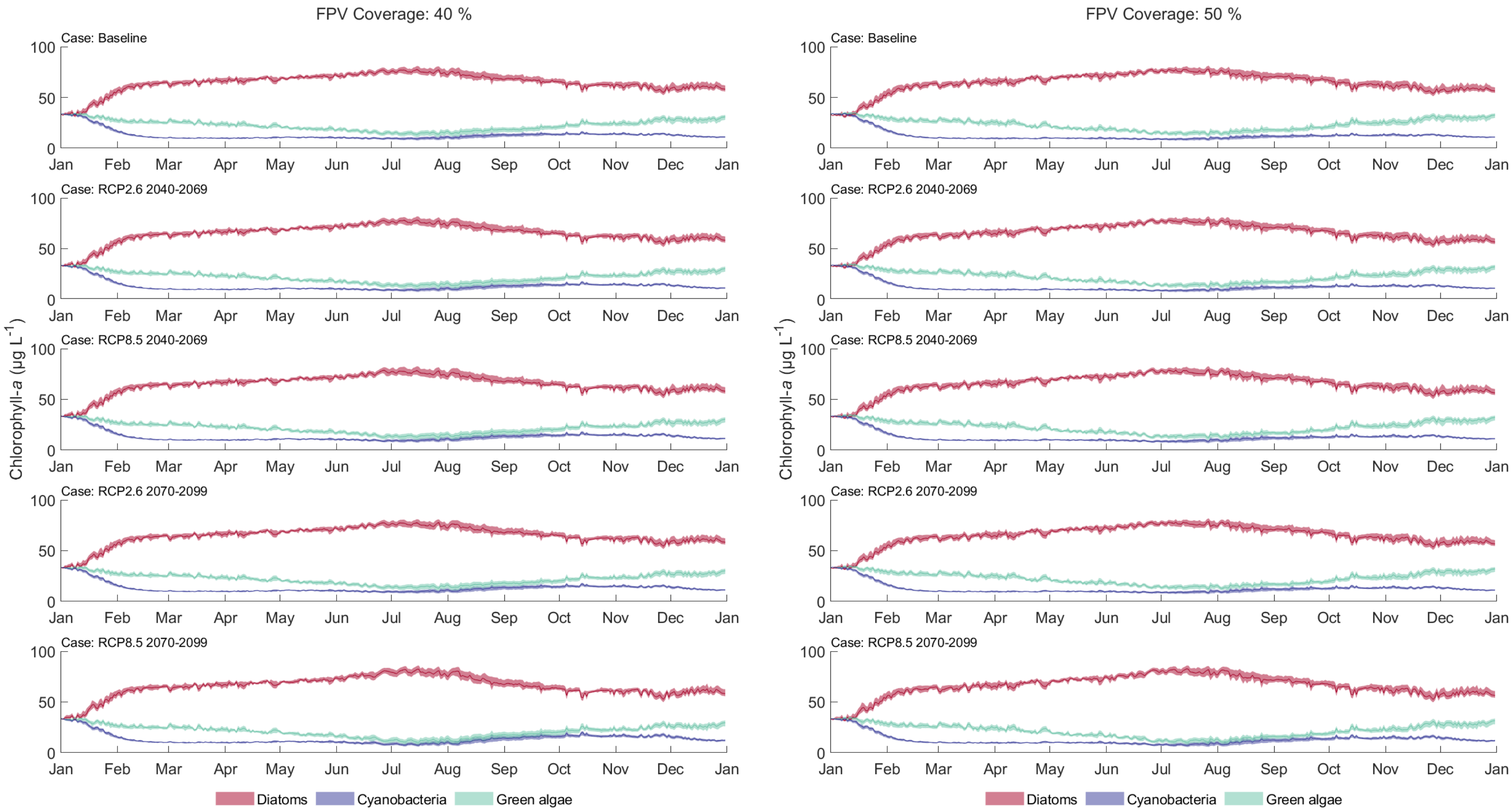


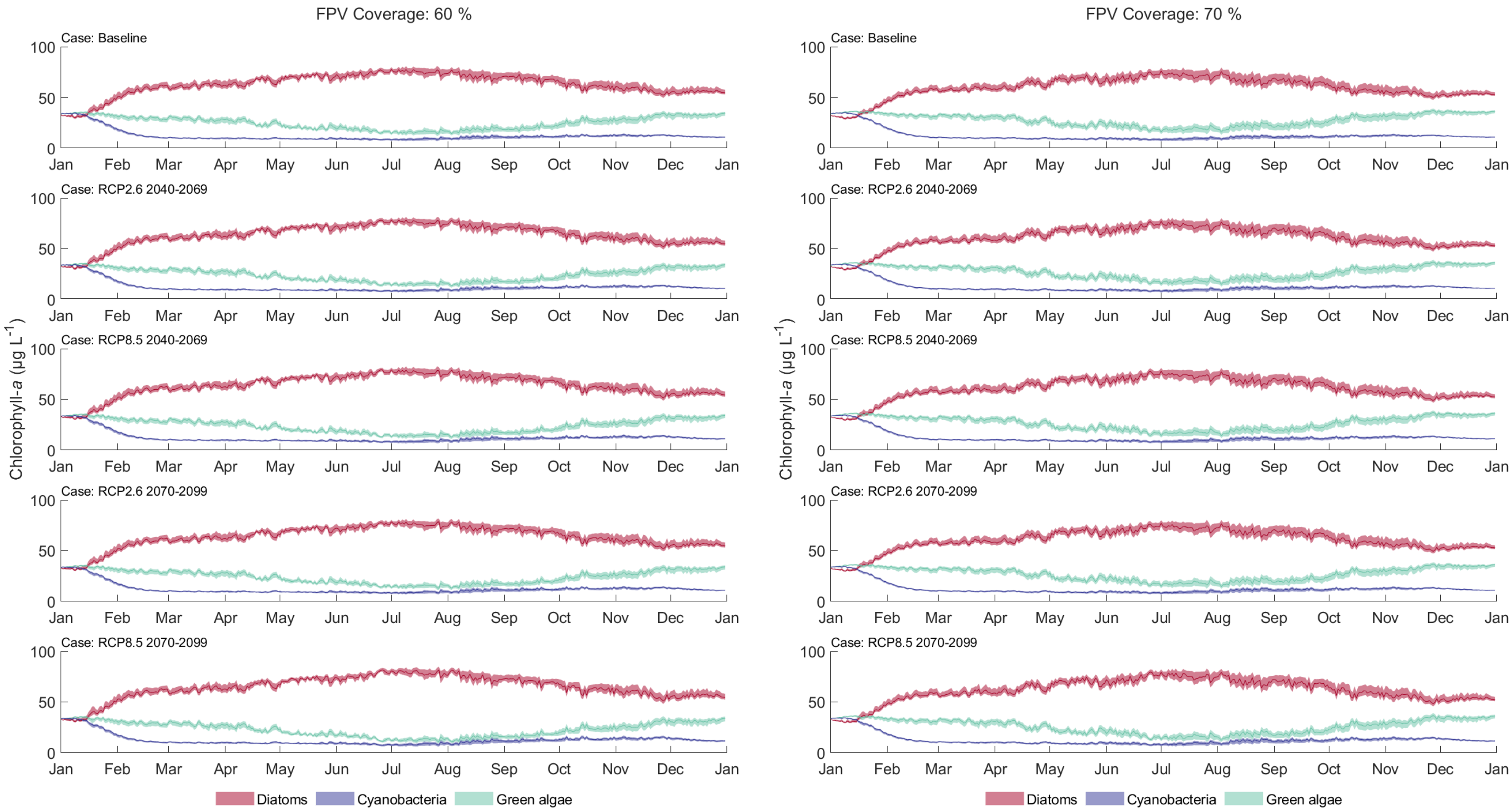
Figure S 5-10 – Median total chlorophyll-a concentration difference (between simulated and baseline) for 2018 by FPV coverage.

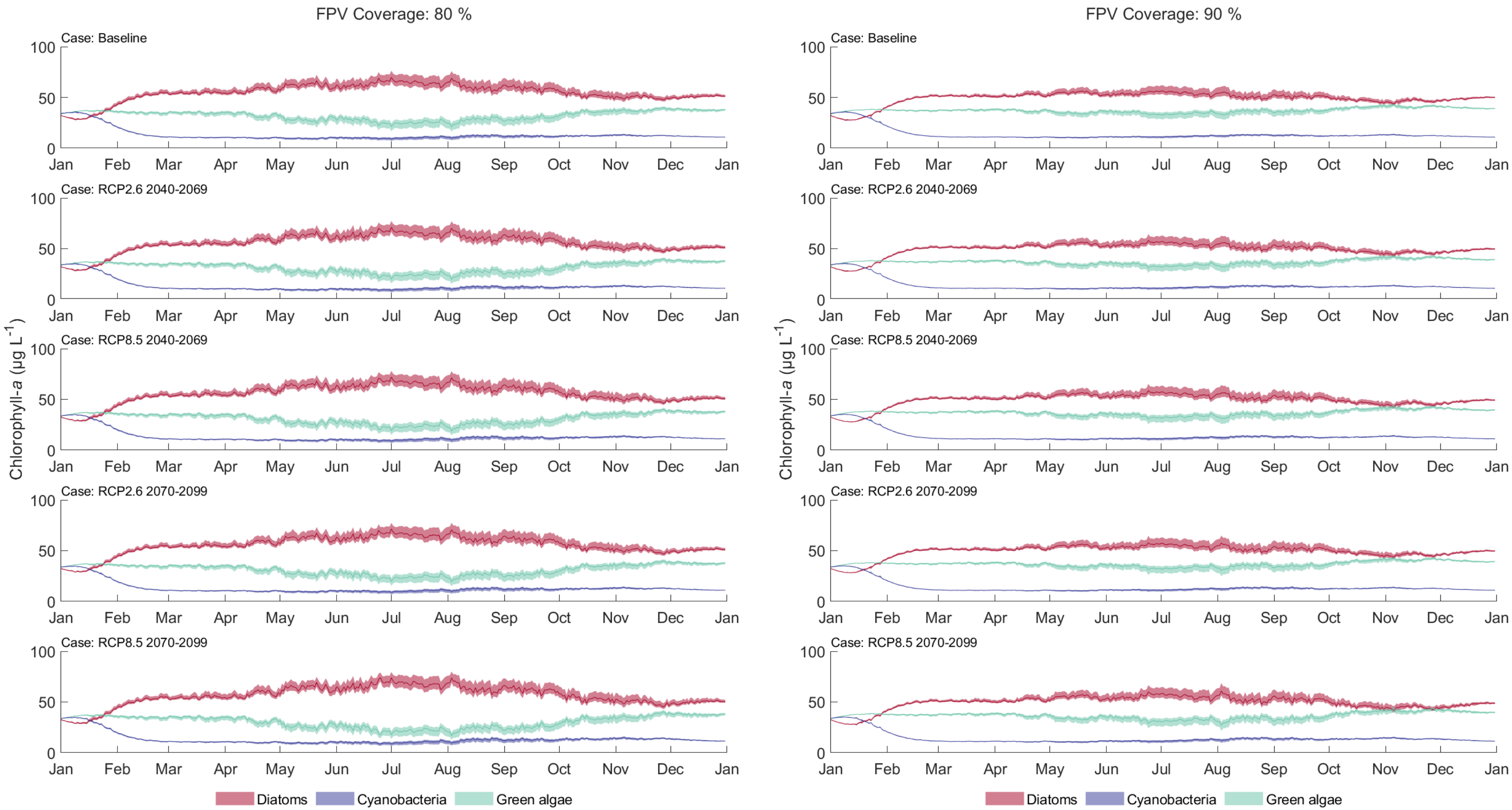
5.7.5 Phytoplankton species composition




FPV Coverage: 20 %
FPV Coverage: 30 %
Case: Baseline
Case: RCP2.6 2040-2069
Case: RCP8.5 2040-2069
Case: RCP2.6 2070-2099
Case: RCP8.5 2070-2099
Chlorophyll-a (µg L-1)
Jan Feb Mar Apr May Jun Jul Aug Sep Oct Nov Dec Jan
Diatoms
Cyanobacteria
Green algae


FPV Coverage: 40 %
FPV Coverage: 50 %
Case: Baseline
Case: RCP2.6 2040-2069
Case: RCP8.5 2040-2069
Case: RCP2.6 2070-2099
Case: RCP8.5 2070-2099
Chlorophyll-a (μg L^{-1})
0
50
100
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
Jan
Diatoms
Cyanobacteria
Green algae


FPV Coverage: 60 %
FPV Coverage: 70 %
Case: Baseline
Case: RCP2.6 2040-2069
Case: RCP8.5 2040-2069
Case: RCP2.6 2070-2099
Case: RCP8.5 2070-2099
Chlorophyll-a (µg L-1)
Jan Feb Mar Apr May Jun Jul Aug Sep Oct Nov Dec Jan
Diatoms
Cyanobacteria
Green algae


FPV Coverage: 80 %
FPV Coverage: 90 %
Case: Baseline
Case: RCP2.6 2040-2069
Case: RCP8.5 2040-2069
Case: RCP2.6 2070-2099
Case: RCP8.5 2070-2099
Chlorophyll-a (µg L-1)
Jan Feb Mar Apr May Jun Jul Aug Sep Oct Nov Dec Jan
Diatoms
Cyanobacteria
Green algae

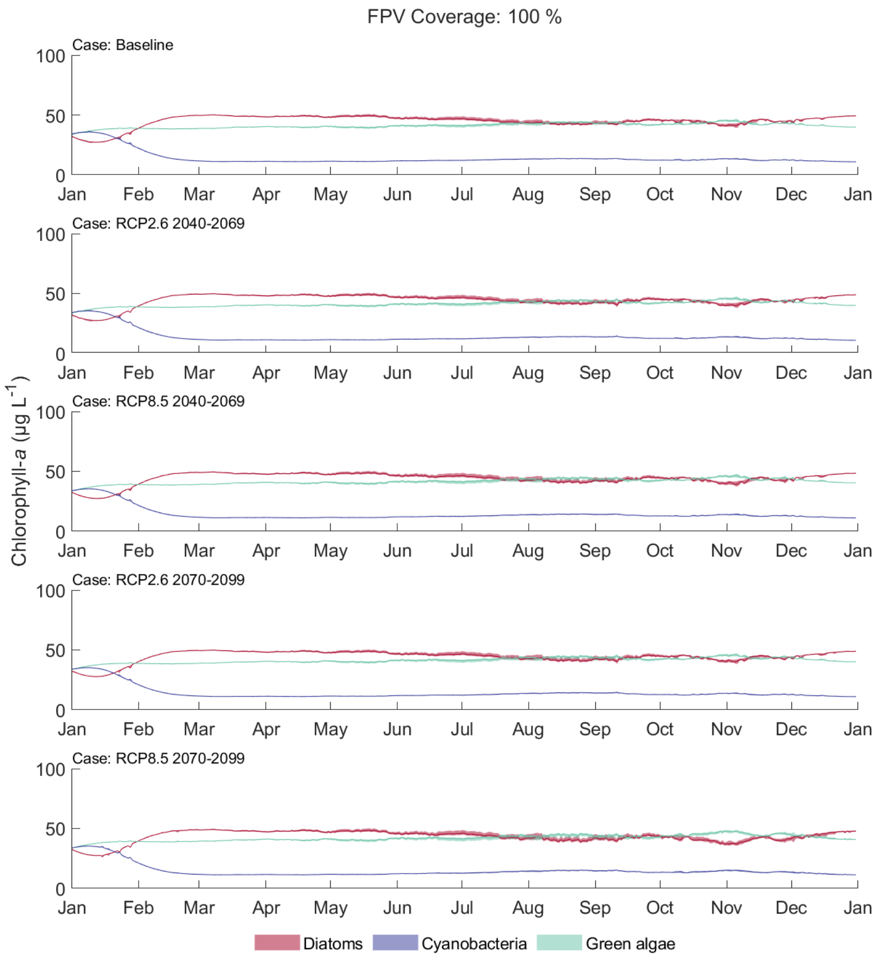


Figure 5-7 – Proportion of phytoplankton functional groups as a percentage of total chlorophyll-a for the simulated period. Each climate case is shown for each simulated FPV coverage (0% - 100%). The initial phytoplankton functional group proportions were set evenly, therefore, the first 30 days of simulations are model run-in time and should be ignored. 0% floating solar coverage represents QEII reservoir simulated as a baseline.

Chapter 6 – Discussion and summary

As the drive to decarbonise intensifies and countries seek to ensure energy security, novel generation technologies like FPV are expanding (Cagle et al., 2020; Haugwitz, 2020). FPV uses the surface of a wide range of water bodies (Chapter 2) to generate electricity for a local demand requirement or for export to a distribution grid. However, FPV can potentially impact its host environment, threatening the ecosystem services provided by water bodies on which humans rely.

This thesis has set out to understand the interaction between FPV and its host environment – the water bodies on which it is sited, which are often reservoirs impounding water for human usage. As one of the first explorations of FPV-environment interactions, this thesis used a range of methods. To increase understanding and evaluate the state of knowledge, Chapter 2 collated stakeholder perceptions of the strengths and weaknesses of FPV-environment interactions. An international survey of FPV operators and a stakeholder workshop (Chapter 2) captured responses, allowing the compatibility of FPV with ecosystem services and the United Nations Sustainable Development Goals (UN SDGs) to be elucidated. Stakeholders identified a need to understand the physical impact of FPV shading and sheltering the host water body. Responding to these needs, Chapter 3 simulated varying FPV coverage on Windermere, English Lake District, as a test case, focussing on the physical effects, including water temperature and stratification dynamics. Given the impact of FPV on lake physics and the likelihood of these extensive perturbations affecting water body biology, Chapter 4 considered the effects of FPV on phytoplankton biomass and species composition. In addition, the chapter evaluated the response of the water body to modifying FPV siting location. The outcomes of Chapter 4 highlighted the ability of FPV to be used as a customisable water body management tool, given the varying response of the water body depending on siting location and FPV coverage. Chapter 3 and Chapter 4 identified the ability of FPV to cool water temperatures, reduce stratification duration and limit phytoplankton growth, the opposite of predicted climate warming impacts, which are one of the most significant threats to global water bodies (Adrian et al., 2009; Woolway et al., 2020). Finally, Chapter 5 simulated FPV under *present*-day and *future* climates, evaluating its effectiveness at compensating for climate change impacts on reservoirs and the potential for FPV to be used as a reservoir management tool through the 21st century.

This chapter examines the connections between the findings of each study chapter and the benefits and drawbacks of the methodological approaches employed in this thesis. The results are reviewed in relation to important areas of water body management and their contribution to FPV design and deployment decisions. Finally, this chapter proposes recommendations for future research based on the remaining knowledge gaps identified by this thesis.

6.1 FPV impacts on water body thermal dynamics

The systematic evidence review (Chapter 2) identified the potential for FPV coverage to cool surface water temperatures by shading the water from incoming solar radiation (Austin and Colman, 2007; Austin and Allen, 2011; Chateau et al., 2019). Numerical modelling simulations of FPV deployments on Windermere's south basin and the QEII reservoir provided evidence to support this theory (Chapter 3, Chapter 4 and Chapter 5). In general, the results indicated that the higher the FPV coverage, the greater the cooling effect on the water body. The response of the QEII reservoir to FPV coverage varied with siting location (Chapter 4); water temperatures cooled more when the FPV array was deployed on areas of faster internal circulation than when deployed on areas with slower circulation. This suggests that understanding water circulation dynamics in water bodies is important for optimal FPV deployment.

Cooler water temperatures were found to reduce stratification duration (Chapter 3, Chapter 4 and Chapter 5), particularly at FPV coverages in excess of 40%. Reductions in stratification duration varied with deployment location (between Windermere south basin and the QEII reservoir) and with siting location (on the QEII reservoir). In the Windermere south basin study, FPV deployment shifted the stratified period to later in the year, with delayed onset and overturn (Chapter 3). However, in the QEII reservoir, stratification onset generally shifted to later in the year, whilst there was no clear trend for stratification overturn with increasing FPV coverage (Chapter 4).

Cooler water temperatures and reduced stratification duration could be considered advantageous to water body managers, given they are opposite to the effects of climate warming impacts (O'Reilly et al., 2015; Woolway et al., 2017a; Woolway and Merchant, 2019). Chapter 5 identified that FPV coverage on the QEII reservoir has the ability to compensate for projected increases in water temperatures and stratification duration due to climate change. The ability to offset projected changes varied by season and according to which *future* emissions prediction was used (Chapter 5). Modest FPV coverages were able to offset increased stratification duration in the QEII reservoir, a potential advantage to water body managers. However, given the relatively short residence time in the QEII reservoir, *future climate* reservoir inflow volume and temperature dampened the ability of FPV coverage to reach the same offset potential as seen in the *present climate* simulations in Chapter 3 and Chapter 4.

6.2 FPV impacts on water body biology

The cooler water temperatures, considerable shading effect and worsened stratification conditions associated with FPV coverage typically led to substantial reductions in phytoplankton biomass in the QEII reservoir (Chapters 4 and Chapter 5). In addition to FPV coverage, reductions to phytoplankton biomass were regulated by FPV siting location (Chapter 4). The area of the QEII reservoir with faster circulation exhibited a greater reduction in total chlorophyll-*a* and a more pronounced change in phytoplankton community structure than for similar coverages of FPV deployed on the slower circulation area (Chapter 4). In a small number of instances, low FPV coverages on the slower circulation area led to an increase in phytoplankton biomass; the opposite outcome of the hypothesised reduction in phytoplankton biomass suggested by early FPV studies (Ferrer-Gisbert et al., 2013; Trapani and Redon Santafe, 2015; Galdino and Olivieri, 2017; Pringle et al., 2017). However, the outcome is uncertain, and whilst an increase in phytoplankton biomass indicates FPV improved conditions for their growth, the increase was short-lived.

Changes in phytoplankton functional-type dynamics were caused by changes in reservoir thermal properties and shading from FPV coverage, with different siting locations modifying the response (Chapter 4). As green algae populations decreased, the relative dominance of diatoms increased in the autumn with moderate FPV coverages. However, the overall decline in phytoplankton biomass associated with increasing FPV coverage offset these changes. Notably, the Chapter 4 simulations allayed concerns identified by the systematic evidence review (Chapter 2) that cyanobacteria dominance may increase with surface coverage (Pinto et al., 2007; Stiers and Triest, 2017). The results suggested that shaded conditions and a more mixed water column would reduce the cyanobacteria's ability to control their buoyancy and vertical position, preventing them from obtaining favourable light and nutrient conditions (Reynolds et al., 1987; Burkholder, 2009).

Based on present climate conditions, FPV appears highly effective at limiting phytoplankton growth (Chapter 4), but *future* climate conditions are likely to improve conditions for growth. Climate change will lead to large increases in cyanobacterial blooms (Paerl and Huisman, 2009; Winder and Sommer, 2012), a nuisance phytoplankton group, which could have implications for water treatment and reservoir recreational use. Additionally, filamentous diatoms (e.g. *Melosira, Asterionella*) and some colonial green algae (e.g. *Scenedesmus*) are also considered nuisance phytoplankton groups as they can disrupt water treatment processes by blocking filters (Henderson et al., 2008). Chapter 5 assessed the effectiveness of FPV at offsetting the increases in phytoplankton biomass and changes to species composition that are forecast under *future* climates. FPV remained effective at limiting phytoplankton growth under *future* climates (~ 10% FPV coverage required), although a slightly greater FPV coverage was required to offset changes in the higher emissions case (~ 20%).

6.3 FPV design and water body considerations

This thesis identified that FPV coverage and siting location are important determinants of host environment response to FPV deployments. Typically, FPV cooled water temperatures, reduced stratification duration (Chapter 3) and reduced phytoplankton growth (Chapter 3, Chapter 4 and Chapter 5). A greater FPV coverage led to a greater reduction in each variable. However, there were instances of a non-linear response when FPV coverage could have the opposite effect, warming water temperatures and increasing stratification duration, depending on the proportions of the reduction to wind speed and solar radiation (Chapter 3). Siting location on the host water body also controls water body response. For example, in Chapter 4, the level of uncertainty in the results implies that the response of the QEII reservoir to increasing FPV coverage could result in either increasing or decreasing total chlorophyll-*a* when the array is deployed on the slower flowing area of the reservoir.

The simulations in Chapter 3 assumed a linear relationship between FPV coverage and wind speed and solar radiation impacts (e.g. a 10% FPV coverage is a 10% reduction in wind speed and solar radiation). Instead, Chapter 4 and Chapter 5 were based on assumptions from monitoring the FPV array at Langthwaite reservoir, Lancaster, UK (Appendix B) and measurements at a ground-mounted solar farm (Armstrong et al., 2016). However, the effect of the FPV array on air temperature, global radiation and wind speed will vary by FPV design (Liu et al., 2018). PV module specification, PV tilt angle, float size, float material and openness of the water's surface will modulate water body response in addition to FPV coverage and siting location. For example, an increased tilt angle or more open design might allow greater wind shear at the air-water interface, increasing mixing (Armstrong et al., 2020). Future users of the revised *MyLake* model (Chapter 4 and section 6.5.2) may use alternative assumptions when parametrising the model to represent individual FPV installations better.

Further, the results of this thesis are likely to be highly water body specific, given the unique meteorology, hydrology and nutrient inputs for each water body. For example, Chapter 3 identified the onset of lake ice in a normally ice-free Windermere, yet ice cover was not simulated in the QEII reservoir (Chapter 4 and Chapter 5). A broad range of water body types in different climates were captured in the international stakeholder survey (Chapter 2) and FPV operators identified no visible impacts of FPV deployment on the host environment.

Indeed, the existing usage of the host water body and the ecosystem services provided determine the importance of individual FPV-environment impacts. For example, FPV arrays have been deployed on mine tailing ponds (Ciel et Terre, 2019) that may be polluted with metals (e.g. arsenic) or radioactive materials (Franks et al., 2011). Given these harsh and toxic conditions, the effect of an FPV installation on biota in a tailings pond may be of little concern considering the existing conditions (Trapani and Millar, 2016). Conversely, potential FPV impacts on raw water reservoir biology would concern water body managers (Chapter 4 and Chapter 5).

6.4 FPV as a tool for managing water bodies

As identified in the international stakeholder survey and at the stakeholder workshop (Chapter 2), FPV has the potential to be deployed on a wide range of water bodies (e.g. lakes, bankside storage reservoirs), each with different functioning and providing different ecosystem services (Maltby et al., 2011). The results of this thesis indicate the potential for FPV to be used as a water body management tool, confirming the hypothesised benefits in the early literature (Sacramento et al., 2015; Sahu et al., 2016). The co-benefits of FPV-environment interactions may alleviate present day concerns but also extend to projected *future* climates. However, it would be naïve to assume a uniform response on all water bodies, so future installations should be evaluated on a case-by-case basis. Further, stakeholders perceive threats of FPV deployment that were not thoroughly examined in this thesis (Chapter 2), for example, impacts on water chemistry (e.g. dissolved oxygen, nutrients and greenhouse gas emissions). Although potential water chemistry impacts were inferred based on the effect of FPV on physical and biological processes. Suggestions for areas of future research are made in section 6.6.

6.5 Methodological approaches to disentangling FPV impacts

This thesis has developed some of the first understanding on the effects of FPV on the host environment. To achieve this, different methodological approaches were used to bring forward knowledge on this nascent topic. This section of the discussion evaluates the different approaches used for benefits and limitations, informing future methodological approaches.

6.5.1 Evidence review and stakeholder insight

Contributing some of the seminal understanding on FPV-environment interactions, Chapter 2 used scientific evidence from a systematic review and stakeholder expertise to assess the compatibility of FPV with ecosystem services and the UN SDGs.

To rapidly accelerate understanding on FPV-environment interactions and evaluate the existing knowledge base, the Defra Quick Scoping Review method, designed to assess the volume and characteristics of an evidence base prior to evidence synthesis (Collins et al., 2015), was used. With the topic of FPV-environment interactions still in early development, using proxies for water body surface covers offered a means to use existing peer-reviewed knowledge to elucidate the potential impacts of FPV. These potential impacts were used as a foundation for forming research questions and guiding the direction of subsequent chapters.

Given the small number of studies at the time considering FPV-environment interactions, Chapter 2 also used stakeholder expertise, captured through an international survey and a workshop to expand the evidence gathered in the systematic review. The international survey and workshop included the broadest possible range of stakeholder groups and organisations (e.g. water body managers, recreational users, developers, environmentalists and local and national authorities) to develop a comprehensive FPV ‘knowledge system’. Knowledge systems collate the expertise of actors (e.g. stakeholders who mobilise knowledge), organisations (e.g. intermediaries between actors), and objects (e.g. data or models) that perform knowledge-related functions (Cash et al., 2003; McCullough and Matson, 2016). Several studies have shown that the coordination and identification of priorities across knowledge systems have contributed towards the transition to low carbon energy (Tawney and Weischer, 2011; Cornell, 2013; Clar and Sautter, 2014). Tapping into the FPV knowledge system helps bridge the knowledge gaps in this upcoming area of research.

Utilising the FPV knowledge system remains a beneficial approach that can complement scientific research on FPV-environment interactions. For example, commercial operators may have access to data that are collected for operational purposes but could also be used to determine potential FPV-environment interactions. Equally, the knowledge system can be reversed. Engaging with stakeholders ensures the knowledge on FPV-environment interactions is communicated to practitioners developing future installations, helping to put research outcomes into practice and maximising impact.

6.5.2 Modelling

Process-based modelling is frequently used to simulate water body functioning. Models are versatile tools and can be used in a wide range of applications, including simulating the impacts of different scenarios (Chapter 3, Chapter 4) and projecting changes under *future* climates (Chapter 5). There are a considerable number of lake ecosystem models available (e.g. PROTECH (Reynolds et al., 2001), PCLake+ (Janssen et al., 2019)), although certain models may be better suited to addressing specific hypotheses or research questions than others, depending on specification.

To gain an initial understanding, this thesis took an existing lake model, *MyLake*, and used it to simulate how FPV installations modify water temperatures and thermal stratification duration (Chapter 3). *MyLake* was identified as the most suitable model for addressing the research questions defined in Chapter 3, as it is open-source, efficient and has accessible architecture (for modifying the source code). The simulated water temperatures were then used to estimate mixed layer depth and Schmidt stability using *Lake Analyzer* (Read et al., 2011). The simulations showed FPV installations cause a highly non-linear response, dependent on system design and coverage. The responses could be either positive or negative, and were often highly variable, although, most commonly, water temperatures reduced, stratification shortened and mixed depths shallowed (Chapter 3). Chapter 3 quantified modifications to lake thermal dynamics would be of sufficient magnitude to alter biological processes, such as phytoplankton biomass and species composition. Therefore, to resolve the potential impact of FPV on phytoplankton biomass and species composition, Chapter 4 details the expansion of *MyLake* to enable the simulation of multiple phytoplankton species. The revised model is better suited to simulating FPV, as it allows the representation of the host water body in multiple vertical cross-sections rather than the original 1-D (horizontal) *MyLake*. The expanded *MyLake* can be used *a priori* to understand the potential impacts of planned FPV installations on the host environment. This provides an opportunity for the planned FPV installation to be modified to limit impacts on the host environment and maximise co-benefits prior to deployment.

However, whilst models are useful tools for determining the potential impacts of FPV on the host environment, there are limitations. Firstly, FPV systems are typically deployed on human-made water bodies (Chapter 2; e.g. mining ponds) which may have limited data available for modelling in comparison to natural lakes that are instrumented for research (e.g. the routine sampling of the English Lake District conducted by the UK Centre for Ecology and Hydrology). In this thesis, Chapter 3 uses high resolution data from Windermere south basin, one of the most comprehensively studied lake systems in the world (Rooney and Jones, 2010). The wealth of understanding and availability of high-resolution meteorological and in-lake water temperature data (Maberly and Elliott, 2012) made Windermere an excellent test system for the first study modelling FPV-environment impacts. In reality, given its world-renowned tourist-appeal, Windermere is unlikely to host FPV.

Secondly, complex biological, chemical and physical processes are often simplified to make a model computationally efficient and relevant to a broad range of applications. For example, assumptions were made to represent the effect of FPV coverage on solar radiation, wind speed and air temperature at the air-water interface (Chapter 4 and Chapter 5). Additionally, modelling studies may have limited input data, as in Chapter 4 and Chapter 5, which simulate an FPV system on a raw water reservoir, a common deployment location for FPV (Chapter 2), but typically with limited high-resolution data available for modelling.

To reduce the uncertainty associated with limited input data and model assumptions, Chapter 4 and Chapter 5 ran the same model multiple times with different parameter sets (i.e. the GLUE method; Beven and Binley (1992)) – a form of ensemble modelling. Ensemble modelling allows the uncertainty in model predictions to be estimated and the likelihood of model predictions to be assessed, improving robustness (Trolle et al., 2014; Kobler and Schmid, 2019). The ensemble runs of *MyLake* take into account the non-uniqueness (*equifinality*, see Beven (2006)) in parameter sets. Alternatively, multi-model ensembles (i.e. using more than one model on the same study site) can be used to identify technical and methodological differences, in addition to weaknesses in the different models (Moore et al., 2021), guiding the development of future models (Janssen et al., 2015; Frassl et al., 2019).

6.5.3 Field monitoring and experiments

There are a small number of studies with in-situ monitoring of FPV installations, focussing, for example, on FPV water temperature impacts (de Lima et al., 2021) and aquatic plant interactions (Ziar et al., 2021). Comprehensive empirical studies of this kind are often resource-intensive or impractical, so available insights are limited (Meyer et al., 2009; Janssen et al., 2015). However, empirical studies are required to improve knowledge of FPV-environment interactions and parameterise models that simulate FPV impacts.

The international stakeholder survey in Chapter 2 identified that only 13% of FPV operators have collected at least some empirical data specifically on FPV-environment interactions post-deployment. However, the sampling interval was often infrequent, or only a small number of parameters were monitored, limiting the utility of the collected data. Alternatively, operators could incorporate sampling on FPV-environment interactions into routine monitoring that already takes place on some human-made water bodies. The type and the scale of this sampling are dependent on water body use; for example, operators of raw water reservoirs are required to conduct regular statutory sampling. This existing sampling could be adapted to monitor FPV-environment interactions.

In the absence of satisfactory existing data sources, bespoke field monitoring or experiments could be used to resolve research questions on FPV-environment interactions. A sampling protocol to monitor an FPV array installed at Langthwaite reservoir, Lancaster, UK was developed, although the COVID-19 pandemic curtailed data collection (see Appendix B). FPV-environment interaction monitoring could employ a BACI (Before, After, Control, Impact) design (Stewart-Oaten et al., 1986), to monitor water body response before and after FPV deployment, using a control to ensure any observed impacts are specific to the intervention.

Water body sampling can comprise traditional methods or more modern high frequency automated monitoring using in situ sensors. Traditional lake sampling involves the manual collection of physical, chemical and biological data and may operate on a weekly, fortnightly or monthly interval (George and Hurley, 2004). Traditional lake sampling is still crucial for capturing detailed samples, such as phytoplankton species and nutrient concentrations (Mantzouki et al., 2018) (Chapter 4 and Chapter 5). However, recent improvements to sensor technology has led to an increase in high frequency lake monitoring (Porter, 2021). High-frequency monitoring has been used for multiple applications given its versatility, for example; DOC load reduction to optimise reservoir operation (Zhan et al., 2021), studying ecosystem metabolism in lakes (Staehr et al., 2012a; Giling et al., 2017), variation of thermal stratification throughout the under-ice season (Bruesewitz et al., 2015) and the response of reservoirs to climate warming and the characterisation of stratification dynamics (Liu et al., 2019). High frequency data from the automated water quality monitoring station on Windermere south basin were used in Chapter 3 for model calibration and validation (Jones and Feuchtmayr, 2017).

Accessing FPV installations can be challenging, particularly if deployed on raw water reservoirs where strict site access protocols apply to protect the asset. Health and safety procedures must be followed by developing safe systems of work that, for example, reduce the risk of drowning and electrocution when using a boat or walking on the FPV array (Charles Rajesh Kumar and Majid, 2021). In the absence of accessible field sites, studies may also use mesocosm experiments to empirically test specific research questions (Andini et al., 2021).

6.5.4 A combined methodological approach

Untangling FPV-environment interactions is a complex undertaking, given the emerging nature of the topic area. Therefore, combining methodological approaches may offer the most effective means to enhance understanding at pace, given the available resources. Complimentary techniques help to overcome the limitations of individual methods, improving robustness. For example, high frequency and traditional lake sampling approaches could be used in conjunction with modelling to infer responses to FPV-environment interactions that might be challenging to test empirically (Chapter 3, Chapter 4, and Chapter 5).

Moreover, advances in sensor technology and modelling approaches are increasingly allowing forecasting of real-time ecological conditions and water quality (Page et al., 2018; Thomas et al., 2020; Shan et al., 2022), a multi-method approach that researchers could adopt with FPV-environment research. Lower cost sensors will allow increased spatial resolution, such as recording regions of different water body characteristics (e.g. littoral and limnetic zones) or with contrasting FPV coverage (e.g. covered or uncovered areas). In the future, in situ sensors may be capable of monitoring key water body properties that are currently manually sampled, including light attention, algal speciation and nutrient concentrations. Accessing high-resolution samples of these variables would greatly assist in model parametrisation and, if affordable, would allow the monitoring of multiple FPV installations across a range of host water bodies.

6.6 Areas of potential future research

This thesis has presented some of the first research on FPV-environment interactions. The outcomes highlight the potential for FPV to have industrial, ecosystem and societal opportunities; although there are potential risks with FPV installations on certain water body types. Other potential risks of FPV-environment interactions remain to be quantified and clarified to facilitate the swift transition to net-zero using this technology.

1. The effect of FPV on dissolved oxygen is poorly resolved. A mesocosm experiment found a lower dissolved oxygen concentration with 100% FPV coverage than no coverage (Andini et al., 2021). The systematic evidence review (Chapter 2) also identified that surface covers can cause anoxia (Ellis and Stefan, 1989; Lepparanta et al., 2012; Bai et al., 2016). However, the outcomes from the simulations in Chapter 3, Chapter 4 and Chapter 5 suggest water body conditions will make anoxia less likely given the typically cooler water temperatures and shortened stratification duration with FPV installations. In order to increase confidence on the impact of FPV on host water body dissolved oxygen concentrations further research should be conducted at a range of FPV coverages on different water body types.
2. Given the large scale reductions to phytoplankton biomass with FPV deployments (Chapter 4 and Chapter 5), the consequences for food webs need to be established to assess the potential impacts on higher trophic levels. The evidence review and international stakeholder survey (Chapter 2) identified the potential for surface covers to impact on zooplankton (Gyllstrom et al., 2005; Haldna and Haberman, 2017), fish populations (Horppila and Nurminen, 2008) and birds. For example, fish reduce their predator vigilance under surface covers (Watz et al., 2015), suggesting there might be a reduced catch for water birds. Changes in species composition across trophic levels can have an impact on overall ecosystem resilience (Downing and Leibold, 2010), as well as the availability of food for human consumption.
3. Future research may focus on the potential for water body pollution from FPV arrays. Microplastics and leaching from plastic FPV float degradation could lead to host water body pollution. Although FPV float materials undergo rigorous

testing procedures, there is scope for float degradation across the lifetime of the FPV installation, given the harsh conditions (water and constant movement) and ultraviolet radiation (Sahu and Sudhalzar, 2019) or due to fires on the array (DNV, 2021). Impacts of this kind would be of particular concern if the host water body supplied drinking water or was used for fisheries.

4. Given the potential for FPV to improve water quality, particularly in managing raw water reservoirs and its use as a tool to reduce water treatment requirements, there is potential for applied or commercial research on this topic area. Studies may have a commercial context or focus on the research and development of improved FPV technologies better suited for managing specific water body management goals, e.g. evaporation suppression. This thesis demonstrates the importance of siting location on the host water body and the choice of the host water body itself. Developers should consider deployments on a case-by-case basis, as the transferability of detailed understanding between different water bodies is uncertain. However, resolving the uniformness of the impact that FPV has on the host environment would improve confidence in making more generic deployment decisions in the knowledge that effects will be similar between different host water bodies. Therefore, enabling operators and policymakers to deploy and approve FPV more widely with likely outcomes understood.

6.7 Final remarks

This thesis has presented some of the first work on FPV-environment interactions, determining the opportunities and risks for water bodies that deliver essential ecosystem services. Critical knowledge gaps on stakeholder perceptions of the strengths and weaknesses of FPV-environment interactions, the compatibility of FPV with ecosystem services and the United Nations Sustainable Development Goals (UN SDGs), the physical and biological response of water bodies to FPV deployments under present-day and future climates and the effectiveness of FPV as a tool for managing water bodies have been addressed. The thesis findings demonstrate the ability of FPV to be used as a complementary tool for water body management. However, the benefits are water body specific and undesirable impacts are possible. This thesis sets the stage for furthering knowledge on this viable and expanding renewable energy technology. As FPV deployments grow globally, increasing the number of water bodies experiencing FPV-environment interactions, knowledge will continue to emerge, further propagating the understanding of this novel topic.

Appendix A – Ethics for international stakeholder survey

A.1 Application for Ethical Approval for Research

Faculty of Science and Technology Research Ethics Committee (FSTREC)

Lancaster University

Application for Ethical Approval for Research

This form should be used for all projects by staff and research students, whether funded or not, which have not been reviewed by any external research ethics committee. If your project is or has been reviewed by another committee (e.g. from another University), please contact the FST research ethics officer for further guidance.

In addition to the completed form, you need to submit **research materials** such as:

i. Participant information sheets
ii. Consent forms
iii. Debriefing sheets
iv. Advertising materials (posters, e-mails)
v. Letters/emails of invitation to participate
vi. Questionnaires, surveys, demographic sheets that are non-standard
vii. Interview schedules, interview question guides, focus group scripts

Please note that **you DO NOT need to submit pre-existing questionnaires or standardized tests** that support your work, but which cannot be amended following ethical review. These should simply be referred to in your application form.

Please submit this form and any relevant materials by email as a **SINGLE** attachment to fst-ethics@lancaster.ac.uk

Section One

Applicant and Project Information

Name of Researcher: Giles Exley

Project Title: SPIRs (Solar Photovoltaic Impacts on Reservoirs) Network Questionnaire

Does your research project involve any of the following?

☐ Human participants (including all types of interviews, questionnaires, focus groups, records relating to humans, use of internet or other secondary data, observation etc.)

☐ Animals - the term animals shall be taken to include any non-human vertebrates or cephalopods.

☐ Risk to members of the research team e.g. lone working, travel to areas where researchers may be at risk, risk of emotional distress

☐ Human cells or tissues other than those established in laboratory cultures

☐ Risk to the environment

☐ Conflict of interest

☐ Research or a funding source that could be considered controversial

☐ Social media and/or data from internet sources that could be considered private

☐ any other ethical considerations

Yes – complete the rest of this form

No – your project does not require ethical review or submission of this form

Section Two

Type of study

☑ Includes direct involvement by human subjects. **Complete all sections apart from Section 3.**

☐ Involves *existing documents/data only*, or the evaluation of an existing project with no direct contact with human participants. ***Complete all sections apart from Section 4.***

If your research involves data from chat rooms and similar online spaces where privacy and anonymity are contentious, please complete all sections

Project details

1. Anticipated project dates (month and year)

Start date: October 2019 End date: March 2020

2. Please briefly describe the background to the research (no more than 150 words, in lay-person's language):

At present there is very little understanding on the effects of floating solar photovoltaics on water quality. Floating solar photovoltaics are a relatively new means of renewable energy generation, with the technology first deployed a little over a decade ago. Solar photovoltaics are typically ground or roof mounted, while floating solar photovoltaics are mounted to plastic floats which are deployed on the surface of water bodies. A group of four UK water companies are working with Lancaster University to further understanding on this technology, enhancing their understanding before potentially deploying their own floating solar photovoltaics. An objective of the research package is to gather the experiences of existing floating solar photovoltaic operators.

3. Please state the aims and objectives of the project (no more than 150 words, in lay-person's language):

The aim of the questionnaire is to gather the experiences of existing floating solar photovoltaic operators from across the world. The experiences shall form part of a SWOT report, accessible for project partners and the questionnaire participants.

4. Methodology and Analysis:

The experiences of existing floating solar photovoltaic operators shall be gathered with an accessible online questionnaire. Responses to each question shall be optional.

Anonymised responses shall be presented in a report to the project partner water companies and shared with questionnaire participants as an incentive to participate.

Section Three – N/A

Section removed for brevity.

Section Four

Participant information

Complete this section if your project includes *direct* involvement by human subjects.

1. Please describe briefly the **intended human participants** (including number, age, gender, and any other relevant characteristics):

The questionnaire is aimed at floating solar photovoltaic operators, this includes individuals in the industry and companies that manufacture or operate systems. There is no targeted respondent based on age, gender or any other characteristic, beyond their involvement in floating solar photovoltaics.

2. How will participants be **recruited** and from where?

Participants will be recruited from a database of floating solar photovoltaic operators that we have collated from publicly available contact details, obtained from internet searches of publicly available press releases with corporate contact details. It is anticipated that there may be some organic sharing of the questionnaire, with recruited participants choosing to share the questionnaire with their relevant colleagues, we do not perceive this to cause an issue, and will allow all responses.

3. Briefly describe your **data collection methods**, drawing particular attention to any potential ethical issues.

A questionnaire using the Lancaster University Qualtrics software shall be emailed to participants. Responses will be stored in the Qualtrics interface for subsequent download to LU Box. The questionnaire will be written in lay person’s language where possible. The questions shall be neutral and non-leading.

4. Consent

4a. Will you take all necessary steps to **obtain the voluntary and informed consent** of the prospective participant(s) or, in the case of individual(s) not capable of giving informed consent, the permission of a legally authorised representative in accordance with applicable law? **YES**

If yes, please go to question 4b. If no, please go to question 4c.

4b. Please explain the procedure you will use for **obtaining consent**?. If applicable, please explain the procedures you intend to use to gain permission on behalf of participants who are unable to give informed consent.

Consent shall be obtained twice, both by the voluntary actions of the participant. The first indication of consent shall be obtained from the participant choosing to follow the link to the questionnaire in their invitation email. The invitation email will clearly state the voluntary nature of participation.

Secondly participants will encounter an information page at the start of the Qualtrics questionnaire, detailing the voluntary nature of the questionnaire. If participants wish to continue and indicate their consent to the survey they shall be directed to continue with the survey by clicking the proceed button.

Participants are free to withdraw their consent at any time during the questionnaire, they can exit the online survey at any point, with incomplete responses deleted. As the survey is anonymous, participants will be told they will NOT be able to withdraw their data/contribution once they have submitted it because it will not be possible to identify it as theirs.

4c. If it will be necessary for participants to take part in the study **without their knowledge and consent at the time**, please explain why (for example covert observations may be necessary in some settings; some experiments require use of deception or partial deception – not telling participants everything about the experiment).

N/A

5. Could participation cause **discomfort** (physical and psychological eg distressing, sensitive or embarrassing topics), **inconvenience or danger beyond the risks encountered in normal life**? Please indicate plans to address these potential risks. State the timescales within which participants may withdraw from the study, noting your reasons.

We fully anticipate the questionnaire to be within the expectations of an industry professionals normal work remit, therefore we do not anticipate any discomfort at the rest of questionnaire participation.

6. How will you protect participants' **confidentiality and/or anonymity** in data collection (e.g. interviews), data storage, data analysis, presentation of findings and publications?

All questionnaire responses are collected anonymously. Responses shall be held on Lancaster University servers, with the data collected being classed as 'Ordinary'. The anonymous responses will ensure the data analysis and report remain confidential. If a participant revealed their identity steps would be taken to conceal this, including the redaction of any distinguishing information. Any instance of this would be handled by the research team to ensure confidentiality.

Questionnaire participants will have the option to record their email address (most likely a corporate account) if they wish to receive a free copy of the final report. This personal data shall be held in accordance with Lancaster University policy, including the use of encryption on LU Box.

7. Do you anticipate any ethical constraints relating to **power imbalances or dependent relationships**, either with participants or with or within the research team? If yes, please explain how you intend to address these?

N/A

8. What potential **risks may exist for the researcher** and/or research team? Please indicate plans to address such risks (for example, noting the support available to you/the researcher; counselling considerations arising from the sensitive or distressing nature of the research/topic; details of the lone worker plan you or any researchers will follow, in particular when working abroad.

The nature of this work mean risks are no different to those encountered with normal life.

9. Whilst there may not be any significant direct **benefits to participants** as a result of this research, please state here any that may result from participation in the study.

Participants will be given the option to receive a copy of the final report. Participants will also be able to contribute their knowledge and experience to a growing body of research of floating solar photovoltaics.

10. Please explain the **rationale for any incentives/payments** (including out-of-pocket expenses) made to participants:

A copy of the final report is an incentive for participation as it shares experiences that are not currently collated into one report. There shall be no payments to questionnaire participants.

11. What are your plans for the **storage of data** (electronic, digital, paper, etc.)? Please ensure that your plans comply with the General Data Protection Regulation (GDPR) and the (UK) Data Protection Act 2018.

Questionnaire response data will be downloaded from the Qualtrics interface and stored on LU Box for the duration of data analysis and report write up, accessible only by the project team. Once the report write up is completed, the data will be held by the project PI (Alona Armstrong) in accordance with the Lancaster University Data Policy for a minimum of 10 years.

Data used in the final report will be aggregated, preventing individuals from being identified. The raw data will not be made publicly available. The aggregated results shall be shared with the project partner water companies (Thames Water, South East Water, Affinity Water & Southern Water) and a summary of the aggregated results shall be shared with participants who want to receive the results of the survey.

12. Please answer the following question *only* if you have not completed a Data Management Plan for an external funder.

12.a How will you make your data available under open access requirements?

N/A

12b. Are there any restrictions on sharing your data for open access purposes?

N/A

13. Will **audio or video recording** take place? ☑ no ☐ audio ☐ video

13a. Please confirm that portable devices (laptop, USB drive etc) will be **encrypted** where they are used for identifiable data. If it is not possible to encrypt your portable devices, please comment on the steps you will take to protect the data.

Portable devices not used in this project.

13b. What arrangements have been made for **audio/video data storage**? At what point in the research will tapes/digital recordings/files be destroyed?

N/A

13c. If your study includes video recordings, what are the implications for participants' anonymity? Can anonymity be guaranteed and if so, how? If participants are identifiable on the recordings, how will you explain to them what you will do with the recordings? How will you seek consent from them?

N/A

14. What are the plans for dissemination of findings from the research? If you are a student, mention here your thesis. Please also include any impact activities and potential ethical issues these may raise.

Findings from the research will be shared with project partners and questionnaire participants via the final report. Results of the research may be submitted for publication in an academic/professional journal separate to the SPIRs project.

15. What particular ethical considerations, not previously noted on this application, do you think there are in the proposed study? Are there any matters about which you wish to seek guidance from the FSTREC?

Questionnaire responses in the final report could reflect badly on floating solar floatovoltaic technology, with a potential to damage industry. We anticipate that the scale of the report will be small, so even with an overall negative conclusion, harm to the industry is unlikely. The questionnaire is part of a wider research project that is objectively assessing the effects of floating solar photovoltaics on water quality.

Section Five

Additional information required by the university insurers

If the research involves either the nuclear industry or an aircraft or the aircraft industry (other than for transport), please provide details below:

Section Six

Declaration and Signatures

I understand that as Principal Investigator/researcher/PhD candidate I have overall responsibility for the ethical management of the project and confirm the following:

- I have read the Code of Practice, Research Ethics at Lancaster: a code of practice and I am willing to abide by it in relation to the current proposal.
- I will manage the project in an ethically appropriate manner according to: (a) the subject matter involved and (b) the Code of Practice and Procedures of the University.

- On behalf of the University I accept responsibility for the project in relation to promoting good research practice and the prevention of misconduct (including plagiarism and fabrication or misrepresentation of results).
- On behalf of the University I accept responsibility for the project in relation to the observance of the rules for the exploitation of intellectual property.
- If applicable, I will give all staff and students involved in the project guidance on the good practice and ethical standards expected in the project in accordance with the University Code of Practice. (Online Research Integrity training is available for staff and students here.)
- If applicable, I will take steps to ensure that no students or staff involved in the project will be exposed to inappropriate situations.
- I confirm that I have completed all risk assessments and other Health and Safety requirements as advised by my departmental Safety Officer.

☑ Confirmed

Please note: If you are not able to confirm the statement above please contact the FST Research Ethics Committee and provide an explanation.

All Staff and Research Students must complete this declaration:

I confirm that I have sent a copy of this application to my Head of Department (or their delegated representative). Tick here to confirm: ☒
Name of Head of Department *(or their delegated representative) Phil Barker*

Applicant electronic signature: Giles Exley Date 16/08/19

A.1.1 Email invitation to participants

Dear [Participant Name]

I am a researcher at the Lancaster Environment Centre at Lancaster University and I would like to invite you to take part in a research questionnaire about floating solar photovoltaics and your experiences of them.

Participation in the questionnaire is voluntary. As a thank you for your time, there is an option at the end of the questionnaire to provide your email address so you can receive a summary of the survey results. We anticipate the questionnaire will take between 5 to 10 minutes to complete, you are welcome to skip questions if you can't provide an answer.

You can access the questionnaire here: https://lancasteruni.eu.qualtrics.com/jfe/form/SV_0d0TlvY2gTHIflj

The deadline for questionnaire responses is **20th November 2019**.

I have attached a participant information sheet for your reference, but please do feel free to get in touch if you have any questions. You can find my contact details at the end of this email.

Kind regards,

Giles Exley

A.1.2 Participant information sheet

SPIRs Project – Solar Photovoltaic Impacts on Reservoirs

For further information about how Lancaster University processes personal data for research purposes and your data rights please visit our webpage: www.lancaster.ac.uk/research/data-protection

What is this study about?

This questionnaire aims to gather the experiences of existing floating solar photovoltaic operators. We want to find out people experiences of deployment, generation and environmental impacts, such as changes to water quality in the host water body. This questionnaire forms part of a wider body of research on floating solar photovoltaics.

Why have I been invited?

I have approached you because of your involvement and knowledge of floating solar photovoltaic systems. I would be very grateful if you would take part in this study.

What will I be asked to do if I take part?

If you decided to take part, you would be asked to complete an online questionnaire. We want to gather the experiences of people who are involved with floating solar photovoltaics, to understand the experiences of existing operators and if they have noticed any water body impacts. The questionnaire will take between 5 to 10 minutes to complete. All your responses will remain anonymous.

What are the possible benefits of taking part?

Taking part in this study will allow you to share your experiences of floating solar photovoltaics. You will also be given the opportunity to receive a copy of the final report, detailing the experiences of existing stakeholder's experiences of floating solar photovoltaics.

Do I have to take part?

No. It is completely up to you to decide whether you take part. Your participation is voluntary and you are free to withdraw at any time, without giving any reason.

What if I change my mind?

Participants are free to withdraw their consent at any time during the questionnaire; they can exit the online survey at any point, with incomplete responses deleted. As the survey is anonymous, you will NOT be able to withdraw your contribution once you have submitted it because it will not be possible to identify it as yours.

Will my data be identifiable?

After you complete and submit the questionnaire, only I, the researcher conducting this study and the overall project lead (Alona Armstrong) will have access to the data you share.

I will keep all personal information about you (e.g. your name and other information about you that can identify you) confidential, that is I will not share it with others.

How will my questionnaire responses be stored?

Your questionnaire responses will be stored securely on the Lancaster University Qualtrics interface, our online questionnaire provider. Once we close the questionnaire to further respondents we shall download your responses and store them with the other responses in an encrypted file, on password-protected computers.

In accordance with University guidelines, the project lead (Alona Armstrong) will keep the responses securely for a minimum of 10 years.

For further information about how Lancaster University processes personal data for research purposes and your data rights please visit our webpage: www.lancaster.ac.uk/research/data-protection

How will we use the information you have shared with us and what will happen to the results of the research study?

I will use the data you have shared with only in the following ways:

- Academic purposes (e.g. publications, such a journal articles)
- A report sharing the experiences of floating solar photovoltaic operators. The report will be available to questionnaire participants free of charge (if you choose to leave your email address at the end of the survey), and to the research project partners.

When writing up the findings from this study, I would like to reproduce some of the views and ideas you provided in response to the questionnaire. When doing so, I will only use anonymised responses, so that although I will use your exact words, you cannot be identified in our publications.

Who has reviewed the project?

This study has been reviewed and approved by the Faculty of Science and Technology Research Ethics Committee.

What if I have a question or concern?

If you have any queries or if you are unhappy with anything that happens concerning your participation in the study, please contact myself.

Giles Exley g.exley@lancaster.ac.uk

If you have any concerns or complaints that you wish to discuss with a person who is not directly involved in the research, you can also contact:

Professor Philip Barker (Director of Lancaster Environment Centre)
Thank you for considering your participation in this project.

Appendix B – Langthwaite IR field monitoring

A chapter consisting of the first in-depth monitoring of an FPV array for environmental impacts was abandoned when access to the field site was prohibited due to the COVID-19 pandemic. Full methods for this monitoring are detailed below. A number of months work went into gaining authorisations, planning the experimental design and instrumenting the field site. Data collection took place for approximately eight months (September 2019 to February 2020) before lockdown restrictions commenced. The collected data, albeit a small proportion of the planned monitoring work, were used in Chapter 4 and Chapter 5 as modelling assumptions.

B.1.1 Planned materials and methods

This research was undertaken at Langthwaite Impounding Reservoir (IR), UK (54° 1'26''N 02°46'1''W), an 865 megalitre reservoir supplying drinking water to Lancaster and the surrounding area. The majority of water stored in Langthwaite IR is obtained by pumping from the River Lune (~50%), while there are also smaller feeds from two reservoirs (Damas Ghyll and Blea Tarn), upland fell intakes and the Thirlmere aqueduct. A 968 kWp FPV array was installed in the south of the reservoir in 2018 using a bespoke design of floating 'tables' to support 3520 PV panels. The electricity generated by the FPV array is used on-site at the water treatment works. The array is a minimum of 30 m from the banked sides of the reservoir, with a footprint covering ~5.8% of the reservoir's 130000 m^2 surface (Figure B-1). Langthwaite IR has a maximum depth of ~10.5 m.

To determine the effects of the FPV array on water quality, measurements were made at eight reservoir sampling points, one on-array location and one shore-side location. Four of the reservoir sampling points were located in each of the two designated treatment areas: in the water under the FPV array and in the open water, referred to as under and open treatments (Figure B-1).

The on array samples were taken in the gap between the water's surface and the rear of the PV panel located in the centre of the FPV array. The under and open sampling points were randomly selected. Further samples were collected along transects from the longest edge of the array which extended northwards into the open water area of the reservoir. The first sampling point on the transect was within 2 m of the edge of the array, with continuous measurements recorded across the reservoir to measure the proximity effect. Each transect was orientated to incorporate the random open water sampling location. At each sampling point, physiochemical, nutrient, biological and micrometeorological metrics were measured. Manual data collection visits occurred monthly during winter and more frequently from April to October.

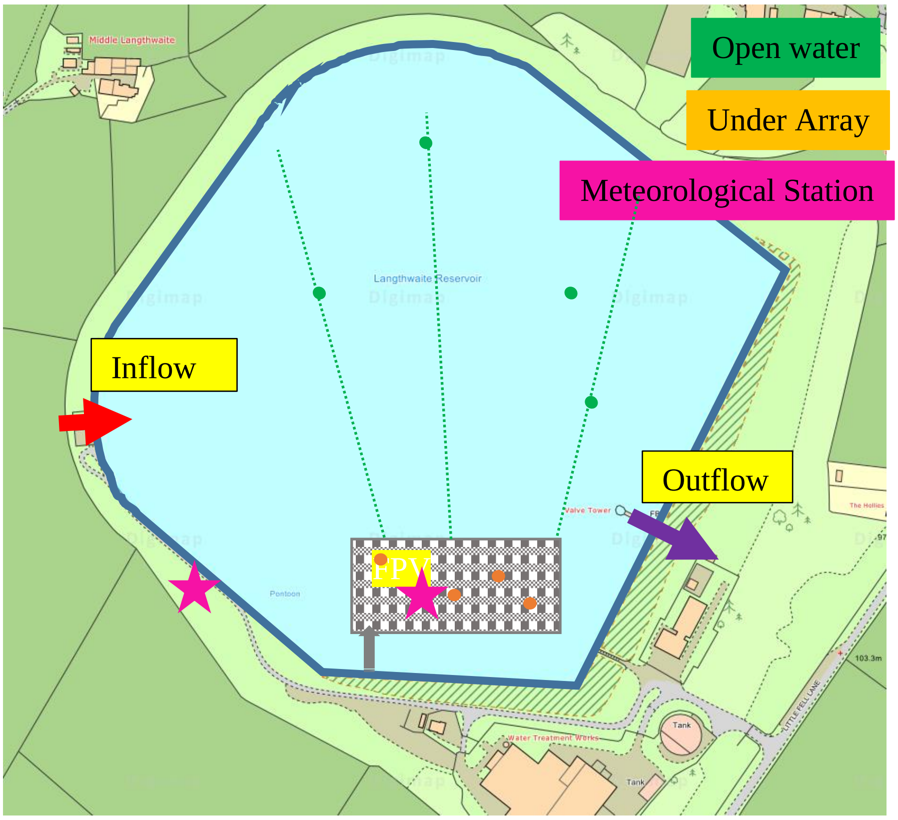


Figure B-1 – Sample locations showing the open water, and co-located under and on array treatments. The shore based meteorological station is indicated with the star symbol, situated at the southeast corner on Langthwaite IR.

B.1.2 Physiochemical

At the under and open treatment sampling points, water temperature was sampled with calibrated data loggers (HOBO Pendant® Temperature 64K) every 5 minutes and the average over each hour recorded. Each under and open sampling point had a string of data loggers distributed down the water column at depths of 0.5, 1, 3 and 5 m (Figure B-2). Water temperature, pH, turbidity and dissolved oxygen were recorded with a multi-parameter Sonde (YSI ProDSS (digital sampling system) handheld multi-parameter meter) at 1 m intervals down the water column. The Sonde was deployed by hand at each open and under sampling point and held at each depth for three minutes, allowing an average to be taken with data logging every one second. In addition to the Sonde-derived turbidity measurements, a Secchi disc was used to measure light attenuation at the under and open treatment sampling points and water depth recorded (Echotest II depth sounder).

Figure B-2 – Water depth being recorded at an under array location. Each under array location had a metal chain attached to the underside of the FPV array, supporting HOBO Pendant® temperature loggers at 0.5, 1, 3 and 5 m.

B.1.3 Nutrients and Biological

A total algae sensor attached to the multi-parameter Sonde was used to measure total algae-phycocyanin and chlorophyll-*a* down the water column at the under and open treatments (Figure B-1), these measurements were conducted at the same frequency and depths as the physiochemical sampling (0.5, 1, 3 and 5 m; Figure B-3). The algae-phycocyanin and chlorophyll-a sensor estimated pigment concentrations at each depth using two excitation beams, an orange beam excited the phycocyanin accessory pigments found in blue-green algae, while a blue beam excited the chlorophyll-*a* molecules. Blue-green algae are of particular concern to water companies, with harmful species causing taste and odour issues that necessitate additional expensive water treatment processes.

Water samples were collected from each under and open treatment location (Figure B-1). Depth integrated samples were collected using a weighted plastic tube (Lund, 1949) from 0 to 5 m. Samples from definite depths (same as physiochemical samples) were collected using a Kemmerer water sampler. Total phosphorus (TP) and chlorophyll-*a* concentrations were determined for each sample according to Mackereth et al. (1978) and Talling (1974), respectively.

Figure B-3 – Sampling within the confines of the FPV array.

B.1.4 Micrometeorological

Air temperature (HOBO Micro Station with S-TMB-M002), photosynthetically active radiation (HOBO Micro Station with S-LIA-M003) and solar radiation (Delta-T BF5 Sunshine Sensor with GP1 data logger) were recorded at one-minute intervals at 0.5 m above the water's surface at the centre of the FPV array (Figure B-1). A shore-based weather station continuously monitored at minute intervals air temperature, relative humidity, solar radiation, rainfall and wind speed and direction, providing a control (HOBO U30 data logger with S-TMB-M002, S-LIA-M003, S-RGB-M002, S-WSA-M003, S-WDA-M003 and Delta-T BF5 Sunshine Sensor with GP1 data logger; Figure B-4).

Figure B-4 – The onshore HOBO Micro Station, with wind speed, wind direction, air temperature, relative humidity, photosynthetically active radiation and rainfall sensors.

Appendix C – Abbreviations

CH_4, methane;

CO_2, carbon dioxide;

Defra, Department for Environment, Food and Rural Affairs;

FPV, floating photovoltaic;

GLUE, General Liklihood Uncertainty Estimation

GW, gigawatt;

HAB, harmful algae bloom;

kWp, kilowatt peak;

MW, megawatt;

N, nitrogen;

NSE, Nash-Sutcliffe model efficiency coefficient;

P, phosphorus;

PICO, Population, Intervention, Comparison and Outcome;

PV, photovoltaic;

QEII, Queen Elizabeth II;

QSR, Quick Scoping Review;

RCP, Representative Concentration Pathway;

SDG, Sustainable Development Goal;

Si, silica;

TF, Transfer function;

UK, United Kingdom;

UKCP, United Kingdom Climate Projections;

UN, United Nations;

USA, United States of America;

Chapter 7 – Bibliography

Abid, M., Abid, Z., Sagin, J., Murtaza, R., Sarbassov, D. and Shabbir, M. (2019) 'Prospects of floating photovoltaic technology and its implementation in Central and South Asian Countries', *International Journal of Environmental Science and Technology,* 16(3), pp. 1755-1762.

Adrian, R., Deneke, R., Mischke, U., Stellmacher, R. and Lederer, P. (1995) 'A long-term study of the Heiligensee (1975-1992). Evidence for effects of climatic change on the dynamics of eutrophied lake ecosystems', *Archiv für Hydrobiologie,* 133(3), pp. 315-337.

Adrian, R., O'Reilly, C. M., Zagarese, H., Baines, S. B., Hessen, D. O., Keller, W., Livingstone, D. M., Sommaruga, R., Straile, D., Van Donk, E., Weyhenmeyer, G. A. and Winder, M. (2009) 'Lakes as sentinels of climate change', *Limnology and Oceanography,* 54(6), pp. 2283-2297.

Agbeti, M. D. and Smol, J. P. (1995) 'Winter Limnology - a Comparison of Physical, Chemical and Biological Characteristics in 2 Temperate Lakes during Ice Cover', *Hydrobiologia,* 304(3), pp. 221-234.

Alderisio, K. A. and DeLuca, N. (1999) 'Seasonal enumeration of fecal coliform bacteria from the feces of ring-billed gulls (Larus delawarensis) and Canada geese (Branta canadensis)', *Appl Environ Microbiol,* 65(12), pp. 5628-30.

AlHassoun, S. A., Al Shaikh, A. A., AlRehaili, A. M. and Misbahuddin, M. (2011) 'Effectiveness of using palm fronds in reducing water evaporation', *Canadian Journal of Civil Engineering,* 38(10), pp. 1170-1174.

Aminzadeh, M., Lehmann, P. and Or, D. (2018) 'Evaporation suppression and energy balance of water reservoirs covered with self-assembling floating elements', *Hydrology and Earth System Sciences,* 22(7), pp. 4015-4032.

Andini, S., Suwartha, N., Setiawan, E. A. and Ma'arif, S. 'Evaluating the Effect of Floating Photovoltaic on Trophic State using Mesocosm Experiments', *7th International Conference on modern approaches in science, technology & engineering*.

Arismendi, I., Safeeq, M., Dunham, J. B. and Johnson, S. L. (2014) 'Can air temperature be used to project influences of climate change on stream temperature?', *Environmental Research Letters,* 9(8), pp. 084015.

Armstrong, A., Ostle, N. J. and Whitaker, J. (2016) 'Solar park microclimate and vegetation management effects on grassland carbon cycling', *Environmental Research Letters,* 11(7).

Armstrong, A., Page, T., Thackeray, S. J., Hernandez, R. R. and Jones, I. D. (2020) 'Integrating environmental understanding into freshwater floatovoltaic deployment using an effects hierarchy and decision trees', *Environmental Research Letters,* 15(11).

Arnell, N. W., Halliday, S. J., Battarbee, R. W., Skeffington, R. A. and Wade, A. J. (2015) 'The implications of climate change for the water environment in England', *Progress in Physical Geography-Earth and Environment,* 39(1), pp. 93-120.

Assouline, S., Narkis, K. and Or, D. (2011) 'Evaporation suppression from water reservoirs: Efficiency considerations of partial covers', *Water Resources Research,* 47(7).

Austin, J. A. and Allen, J. (2011) 'Sensitivity of summer Lake Superior thermal structure to meteorological forcing', *Limnology and Oceanography,* 56(3), pp. 1141-1154.

Austin, J. A. and Colman, S. M. (2007) 'Lake Superior summer water temperatures are increasing more rapidly than regional air temperatures: A positive ice-albedo feedback', *Geophysical Research Letters,* 34(6), pp. 5.

Bai, Q. X., Li, R. L., Li, Z. J., Lepparanta, M., Arvola, L. and Li, M. (2016) 'Time-series analyses of water temperature and dissolved oxygen concentration in Lake Valkea-Kotinen (Finland) during ice season', *Ecological Informatics,* 36, pp. 181-189.

Balayla, D., Lauridsen, T. L., Søndergaard, M. and Jeppesen, E. (2010) 'Larger zooplankton in Danish lakes after cold winters: are winter fish kills of importance?', *Hydrobiologia,* 646(1), pp. 159-172.

Bastviken, D., Tranvik, L. J., Downing, J. A., Crill, P. M. and Enrich-Prast, A. (2011) 'Freshwater methane emissions offset the continental carbon sink', *Science,* 331(6013), pp. 50.

Batt, R. D., Carpenter, S. R., Cole, J. J., Pace, M. L., Johnson, R. A., Kurtzweil, J. T. and Wilkinson, G. M. (2015) 'Altered energy flow in the food web of an experimentally darkened lake', *Ecosphere,* 6(3), pp. 23.

Beall, B. F., Twiss, M. R., Smith, D. E., Oyserman, B. O., Rozmarynowycz, M. J., Binding, C. E., Bourbonniere, R. A., Bullerjahn, G. S., Palmer, M. E., Reavie, E. D., Waters, L. M., Woityra, L. W. and McKay, R. M. (2016) 'Ice cover extent drives phytoplankton and bacterial community structure in a large north-temperate lake: implications for a warming climate', *Environ Microbiol,* 18(6), pp. 1704-19.

Bengtsson, L. (1996) 'Mixing in ice-covered lakes', *Hydrobiologia,* 322(1-3), pp. 91-97.

Bennett, E. M. and Balvanera, P. (2007) 'The future of production systems in a globalized world', *Frontiers in Ecology and the Environment,* 5(4), pp. 191-198.

Bennett, E. M., Peterson, G. D. and Gordon, L. J. (2009) 'Understanding relationships among multiple ecosystem services', *Ecol Lett,* 12(12), pp. 1394-404.

Benskin, C. M., Wilson, K., Jones, K. and Hartley, I. R. (2009) 'Bacterial pathogens in wild birds: a review of the frequency and effects of infection', *Biol Rev Camb Philos Soc,* 84(3), pp. 349-73.

Bernes, C., Carpenter, S. R., Gårdmark, A., Larsson, P., Persson, L., Skov, C., Speed, J. D. M. and Van Donk, E. (2015) 'What is the influence of a reduction of planktivorous and benthivorous fish on water quality in temperate eutrophic lakes? A systematic review', *Environmental Evidence,* 4(1), pp. 7.

Beutel, M. W., Leonard, T. M., Dent, S. R. and Moore, B. C. (2008) 'Effects of aerobic and anaerobic conditions on P, N, Fe, Mn, and Hg accumulation in waters overlaying profundal sediments of an oligo-mesotrophic lake', *Water Res,* 42(8-9), pp. 1953-62.

Beven, K. (2006) 'A manifesto for the equifinality thesis', *Journal of Hydrology,* 320(1-2), pp. 18-36.

Beven, K. and Binley, A. (1992) 'The Future of Distributed Models - Model Calibration and Uncertainty Prediction', *Hydrological Processes,* 6(3), pp. 279-298.

Birk, S., Chapman, D., Carvalho, L., Spears, B. M., Andersen, H. E., Argillier, C., Auer, S., Baattrup-Pedersen, A., Banin, L., Beklioglu, M., Bondar-Kunze, E., Borja, A., Branco, P., Bucak, T., Buijse, A. D., Cardoso, A. C., Couture, R. M., Cremona, F., de Zwart, D., Feld, C. K., Ferreira, M. T., Feuchtmayr, H., Gessner, M. O., Gieswein, A., Globevnik, L., Graeber, D., Graf, W., Gutierrez-Canovas, C., Hanganu, J., Iskin, U., Jarvinen, M., Jeppesen, E., Kotamaki, N., Kuijper, M., Lemm, J. U., Lu, S., Solheim, A. L., Mischke, U., Moe, S. J., Noges, P., Noges, T., Ormerod, S. J., Panagopoulos, Y., Phillips, G., Posthuma, L., Pouso, S., Prudhomme, C., Rankinen, K., Rasmussen, J. J., Richardson, J., Sagouis, A., Santos, J. M., Schafer, R. B., Schinegger, R., Schmutz, S., Schneider, S. C., Schulting, L., Segurado, P., Stefanidis, K., Sures, B., Thackeray, S. J., Turunen, J., Uyarra, M. C., Venohr, M., von der Ohe, P. C., Willby, N. and Hering, D. (2020) 'Impacts of multiple stressors on freshwater biota across spatial scales and ecosystems', *Nat Ecol Evol,* 4(8), pp. 1060-1068.

Birkhofer, K., Diehl, E., Andersson, J., Ekroos, J., Fruh-Muller, A., Machnikowski, F., Mader, V. L., Nilsson, L., Sasaki, K., Rundlof, M., Wolters, V. and Smith, H. G. (2015) 'Ecosystem services-current challenges and opportunities for ecological research', *Frontiers in Ecology and Evolution,* 2(87).

Bloesch, J. (1982) 'Inshore-Offshore Sedimentation Differences Resulting from Resuspension in the Eastern Basin of Lake Erie', *Canadian Journal of Fisheries and Aquatic Sciences,* 39(5), pp. 748-759.

Boehrer, B. and Schultze, M. (2008) 'Stratification of lakes', *Reviews of Geophysics,* 46(2).

Bouffard, D., Ackerman, J. D. and Boegman, L. (2013) 'Factors affecting the development and dynamics of hypoxia in a large shallow stratified lake: Hourly to seasonal patterns', *Water Resources Research,* 49(5), pp. 2380-2394.

Brauman, K. A., Daily, G. C., Duarte, T. K. and Mooney, H. A. (2007) 'The nature and value of ecosystem services: An overview highlighting hydrologic services', *Annual Review of Environment and Resources,* 32(1), pp. 67-98.

Brooks, B. W., Lazorchak, J. M., Howard, M. D., Johnson, M. V., Morton, S. L., Perkins, D. A., Reavie, E. D., Scott, G. I., Smith, S. A. and Steevens, J. A. (2016) 'Are harmful algal blooms becoming the greatest inland water quality threat to public health and aquatic ecosystems?', *Environ Toxicol Chem,* 35(1), pp. 6-13.

Bruesewitz, D. A., Carey, C. C., Richardson, D. C. and Weathers, K. C. (2015) 'Under-ice thermal stratification dynamics of a large, deep lake revealed by high-frequency data', *Limnology and Oceanography,* 60(2), pp. 347-359.

Burkholder, J. M. (2009) 'Harmful Algal Blooms', in Likens, G.E. (ed.) *Encyclopedia of Inland Waters*. Oxford: Academic Press, pp. 264-285.

Butcher, J. B., Nover, D., Johnson, T. E. and Clark, C. M. (2015) 'Sensitivity of lake thermal and mixing dynamics to climate change', *Climatic Change,* 129(1-2), pp. 295-305.

Cagle, A. E., Armstrong, A., Exley, G., Grodsky, S. M., Macknick, J., Sherwin, J. and Hernandez, R. R. (2020) 'The Land Sparing, Water Surface Use Efficiency, and Water Surface Transformation of Floating Photovoltaic Solar Energy Installations', *Sustainability,* 12(19).

Campana, P. E., Wasthage, L., Nookuea, W., Tan, Y. T. and Yan, J. Y. (2019) 'Optimization and assessment of floating and floating-tracking PV systems integrated in on- and off-grid hybrid energy systems', *Solar Energy,* 177, pp. 782-795.

Campbell, J. M. and Haase, B. L. (1981) 'Availability of Suitable Phytoplanktonic Food for Zooplankton in an Ice-Covered Lake', *Hydrobiologia,* 79(2), pp. 113-119.

Carey, C. C., Ibelings, B. W., Hoffmann, E. P., Hamilton, D. P. and Brookes, J. D. (2012) 'Eco-physiological adaptations that favour freshwater cyanobacteria in a changing climate', *Water Res,* 46(5), pp. 1394-407.

Cash, D. W., Clark, W. C., Alcock, F., Dickson, N. M., Eckley, N., Guston, D. H., Jager, J. and Mitchell, R. B. (2003) 'Knowledge systems for sustainable development', *Proc Natl Acad Sci U S A,* 100(14), pp. 8086-91.

Cavaliere, E. and Baulch, H. M. (2018) 'Denitrification under lake ice', *Biogeochemistry,* 137(3), pp. 285-295.

Cazzanelli, M., Warming, T. P. and Christoffersen, K. S. (2008) 'Emergent and floating-leaved macrophytes as refuge for zooplankton in a eutrophic temperate lake without submerged vegetation', *Hydrobiologia,* 605, pp. 113-122.

Cazzaniga, R., Cicu, M., Rosa-Clot, M., Rosa-Clot, P., Tina, G. M. and Ventura, C. (2018) 'Floating photovoltaic plants: Performance analysis and design solutions', *Renewable & Sustainable Energy Reviews,* 81, pp. 1730-1741.

Cazzaniga, R. and Rosa-Clot, M. (2021) 'The booming of floating PV', *Solar Energy,* 219, pp. 3-10.

Chaichana, R., Leah, R. and Moss, B. (2010) 'Birds as eutrophicating agents: a nutrient budget for a small lake in a protected area', *Hydrobiologia,* 646(1), pp. 111-121.

Chang, H. and Bonnette, M. R. (2017) 'Climate change and water-related ecosystem services: impacts of drought in california, usa', *Ecosystem Health and Sustainability,* 2(12), pp. e01254.

Chapra, S. C., Boehlert, B., Fant, C., Bierman, V. J., Jr., Henderson, J., Mills, D., Mas, D. M. L., Rennels, L., Jantarasami, L., Martinich, J., Strzepek, K. M. and Paerl, H. W. (2017) 'Climate Change Impacts on Harmful Algal Blooms in U.S. Freshwaters: A Screening-Level Assessment', *Environ Sci Technol,* 51(16), pp. 8933-8943.

Charles Rajesh Kumar, J. and Majid, M. A. (2021) 'Floating solar photovoltaic plants in India – A rapid transition to a green energy market and sustainable future', *Energy & Environment*.

Chateau, P. A., Wunderlich, R. F., Wang, T. W., Lai, H. T., Chen, C. C. and Chang, F. J. (2019) 'Mathematical modeling suggests high potential for the deployment of floating photovoltaic on fish ponds', *Sci Total Environ,* 687, pp. 654-666.

Chen, L., Delatolla, R., D'Aoust, P. M., Wang, R., Pick, F., Poulain, A. and Rennie, C. D. (2019) 'Hypoxic conditions in stormwater retention ponds: potential for hydrogen sulfide emission', *Environ Technol,* 40(5), pp. 642-653.

Choi, Y.-K. (2014) 'A study on power generation analysis of floating PV system considering environmental impact', *Republic of Korea*.

Choi, Y.-K., Lee, N.-H. and Kim, K.-J. 'Empirical Research on the efficiency of Floating PV systems compared with Overland PV Systems'. *Proceedings, The 3rd International Conference on*

Circuits, Control, Communication, Electricity, Electronics, Energy, System, Signal and Simulation, 284-289.

Chopra, K., Leemans, R., Kumar, P. and Simons, H. (2005) *Ecosystems and human well-being: policy responses.* Island Press.

Ciel et Terre (2019) *Chillie's innovative floating PV project on a tailing ponds*. Available at: https://www.ciel-et-terre.net/chiles-innovative-floating-pv-project-on-a-tailings-pond/ (Accessed: 28/03 2022).

Clar, G. and Sautter, B. (2014) 'Research driven clusters at the heart of (trans-) regional learning and priority-setting processes', *Journal of the Knowledge Economy,* 5(1), pp. 156-180.

Coates, M. and Ferris, J. (1994) 'The Radiatively Driven Natural-Convection beneath a Floating Plant Layer', *Limnology and Oceanography,* 39(5), pp. 1186-1194.

Collen, B., Whitton, F., Dyer, E. E., Baillie, J. E., Cumberlidge, N., Darwall, W. R., Pollock, C., Richman, N. I., Soulsby, A. M. and Bohm, M. (2014) 'Global patterns of freshwater species diversity, threat and endemism', *Glob Ecol Biogeogr,* 23(1), pp. 40-51.

Collingsworth, P. D., Bunnell, D. B., Murray, M. W., Kao, Y. C., Feiner, Z. S., Claramunt, R. M., Lofgren, B. M., Hook, T. O. and Ludsin, S. A. (2017) 'Climate change as a long-term stressor for the fisheries of the Laurentian Great Lakes of North America', *Reviews in Fish Biology and Fisheries,* 27(2), pp. 363-391.

Collins, A., Coughlin, D., Miller, J. and Kirk, S. (2015) 'The production of quick scoping reviews and rapid evidence assessments: a how to guide'.

Cornell, W. 2013. How Data Mining Drives Pharma: Information as a Raw Material and Product.

Corripio, J. G.(2019) *insol: Solar Radiation* (Version 1.2) [R-package]. Available at: https://cran.r-project.org/web/packages/insol/.

Costa, L. C. A. and Silva, G. D. P. (2021) 'Save water and energy: A techno-economic analysis of a floating solar photovoltaic system to power a water integration project in the Brazilian semiarid', *International Journal of Energy Research*.

Costanza, R., dArge, R., deGroot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., ONeill, R. V., Paruelo, J., Raskin, R. G., Sutton, P. and vandenBelt, M. (1997) 'The value of the world's ecosystem services and natural capital', *Nature,* 387(6630), pp. 253-260.

Cox, M. (2019) *Floating solar landscape 2019*.

Crittenden, J. C., Trussell, R. R., Hand, D. W., Howe, K. J. and Tchobanoglous, G. (2012) *MWH's water treatment: principles and design.* John Wiley & Sons.

Cuce, E., Cuce, P. M., Saboor, S., Ghosh, A. and Sheikhnejad, Y. (2022) 'Floating PVs in Terms of Power Generation, Environmental Aspects, Market Potential, and Challenges', *Sustainability,* 14(5).

Dake, J. M. K. and Harleman, D. R. (1969) 'Thermal Stratification in Lakes - Analytical and Laboratory Studies', *Water Resources Research,* 5(2), pp. 484-&.

Danilov, R. A. and Ekelund, N. G. A. (2001) 'Phytoplankton communities at different depths in two eutrophic and two oligotrophic temperate lakes at higher latitude during the period of ice cover', *Acta Protozoologica,* 40(3), pp. 197-201.

de Groot, R. S., Alkemade, R., Braat, L., Hein, L. and Willemen, L. (2010) 'Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making', *Ecological Complexity,* 7(3), pp. 260-272.

de Groot, R. S., Wilson, M. A. and Boumans, R. M. J. (2002) 'A typology for the classification, description and valuation of ecosystem functions, goods and services', *Ecological Economics,* 41(3), pp. 393-408.

de la Fuente, A. and Niño, Y. (2008) 'Pseudo 2D ecosystem model for a dendritic reservoir', *Ecological Modelling,* 213(3-4), pp. 389-401.

de Lima, R. L. P., Paxinou, K., C. Boogaard, F., Akkerman, O. and Lin, F.-Y. (2021) 'In-Situ Water Quality Observations under a Large-Scale Floating Solar Farm Using Sensors and Underwater Drones', *Sustainability,* 13(11).

Deemer, B. R., Harrison, J. A., Li, S., Beaulieu, J. J., DelSontro, T., Barros, N., Bezerra-Neto, J. F., Powers, S. M., Dos Santos, M. A. and Vonk, J. A. (2016) 'Greenhouse Gas Emissions from Reservoir Water Surfaces: A New Global Synthesis', *BioScience,* 66(11), pp. 949-964.

DEFRA, London, H.G. (2018) *A green future: our 25 year plan to improve the environment*.

Dick, J., Al-Assaf, A., Andrews, C., Díaz-Delgado, R., Groner, E., Halada, Ľ., Izakovičová, Z., Kertész, M., Khoury, F., Krasić, D., Krauze, K., Matteucci, G., Melecis, V., Mirtl, M., Orenstein, D. E., Preda, E., Santos-Reis, M., Smith, R. I., Vadineanu, A., Veselić, S. and Vihervaara, P. (2014) 'Ecosytem Services: A Rapid Assessment Method Tested at 35 Sites of the LTER-Europe Network', *Ekologia,* 33(3).

Diehl, S., Berger, S., Ptacnik, R. and Wild, A. (2002) 'Phytoplankton, light, and nutrients in a gradient of mixing depths: Field experiments', *Ecology,* 83(2), pp. 399-411.

Dimitriou, E., Ntoanidis, L., Bellos, V. and Papadaki, C. (2017) 'Comparison of West Balkan adult trout habitat predictions using a Pseudo-2D and a 2D hydrodynamic model', *Hydrology Research,* 48(6), pp. 1697-1709.

DNV (2021) *Design, development and operation of floating solar photovoltaic systems*.

Dörenkämper, M., Wahed, A., Kumar, A., de Jong, M., Kroon, J. and Reindl, T. (2021) 'The cooling effect of floating PV in two different climate zones: A comparison of field test data from the Netherlands and Singapore', *Solar Energy,* 214, pp. 239-247.

Downing, A. L. and Leibold, M. A. (2010) 'Species richness facilitates ecosystem resilience in aquatic food webs', *Freshwater Biology,* 55(10), pp. 2123-2137.

Dunlap, C. R., Sklenar, K. S. and Blake, L. J. (2015) 'A Costly Endeavor: Addressing Algae Problems in a Water Supply', *Journal American Water Works Association,* 107(5), pp. E255-E262.

Elci, S. (2008) 'Effects of thermal stratification and mixing on reservoir water quality', *Limnology,* 9(2), pp. 135-142.

Ellis, C. R. and Stefan, H. G. (1989) 'Oxygen-Demand in Ice Covered Lakes as It Pertains to Winter Aeration', *Water Resources Bulletin,* 25(6), pp. 1169-1176.

Environment Agency (2018) 'Water Quality Archive'. Available at: https://environment.data.gov.uk/water-quality/view/landing (Accessed: 16/02/21).

Evans, R. D. (1994) 'Empirical-Evidence of the Importance of Sediment Resuspension in Lakes', *Hydrobiologia,* 284(1), pp. 5-12.

Exley, G., Armstrong, A., Page, T. and Jones, I. D. (2021a) 'Floating photovoltaics could mitigate climate change impacts on water body temperature and stratification', *Solar Energy,* 219, pp. 24-33.

Exley, G., Hernandez, R. R., Page, T., Chipps, M., Gambro, S., Hersey, M., Lake, R., Zoannou, K. S. and Armstrong, A. (2021b) 'Scientific and stakeholder evidence-based assessment: Ecosystem response to floating solar photovoltaics and implications for sustainability', *Renewable and Sustainable Energy Reviews,* 152, pp. 111639.

Falkowski, P. G. (1994) 'The role of phytoplankton photosynthesis in global biogeochemical cycles', *Photosynth Res,* 39(3), pp. 235-58.

Fee, E. J., Hecky, R. E., Kasian, S. E. M. and Cruikshank, D. R. (1996) 'Effects of lake size, water clarity, and climatic variability on mixing depths in Canadian Shield lakes', *Limnology and Oceanography,* 41(5), pp. 912-920.

Fenocchi, A., Rogora, M., Sibilla, S. and Dresti, C. (2017) 'Relevance of inflows on the thermodynamic structure and on the modeling of a deep subalpine lake (Lake Maggiore, Northern Italy/Southern Switzerland)', *Limnologica,* 63, pp. 42-56.

Ferrer-Gisbert, C., Ferran-Gozalvez, J. J., Redon-Santafe, M., Ferrer-Gisbert, P., Sanchez-Romero, F. J. and Torregrosa-Soler, J. B. (2013) 'A new photovoltaic floating cover system for water reservoirs', *Renewable Energy,* 60, pp. 63-70.

Ficke, A. D., Myrick, C. A. and Hansen, L. J. (2007) 'Potential impacts of global climate change on freshwater fisheries', *Reviews in Fish Biology and Fisheries,* 17(4), pp. 581-613.

Finch, J. and Hall, R. (2001) *Estimation of open water evaporation: A review of methods.* Environment Agency Bristol, UK.

Findlay, A. (2022) 'River Thames data logger above Shiplake Lock'. Available at: https://dl1.findlays.net/rawdata/ (Accessed: 03/03/22).

Foley, B., Jones, I. D., Maberly, S. C. and Rippey, B. (2012) 'Long-term changes in oxygen depletion in a small temperate lake: effects of climate change and eutrophication', *Freshwater Biology,* 57(2), pp. 278-289.

Ford, D. E. and Stefan, H. G. (1980) 'Thermal Predictions Using Integral Energy-Model', *Journal of the Hydraulics Division-Asce,* 106(1), pp. 39-55.

Forsberg, C. (1989) 'Importance of Sediments in Understanding Nutrient Cyclings in Lakes', *Hydrobiologia,* 176(1), pp. 263-277.

Fouad, M. M., Shihata, L. A. and Morgan, E. I. (2017) 'An integrated review of factors influencing the perfomance of photovoltaic panels', *Renewable and Sustainable Energy Reviews,* 80, pp. 1499-1511.

Franks, D. M., Boger, D. V., Côte, C. M. and Mulligan, D. R. (2011) 'Sustainable development principles for the disposal of mining and mineral processing wastes', *Resources Policy,* 36(2), pp. 114-122.

Frassl, M. A., Abell, J. M., Botelho, D. A., Cinque, K., Gibbes, B. R., Johnk, K. D., Muraoka, K., Robson, B. J., Wolski, M., Xiao, M. and Hamilton, D. P. (2019) 'A short review of contemporary developments in aquatic ecosystem modelling of lakes and reservoirs', *Environmental Modelling & Software,* 117, pp. 181-187.

Friel, D., Karimirad, M., Whittaker, T., Doran, J. and Howlin, E. 'A review of floating photovoltaic design concepts and installed variations'. *Proceedings of the 4th International Conference Offshore Renew Energy CORE*, 1-10.

Galdino, M. A. E. and Olivieri, M. M. d. A. (2017) 'Some Remarks about the Deployment of Floating PV Systems in Brazil', *J. of Electrical Engineering,* 5(1).

Gallina, N., Beniston, M. and Jacquet, S. (2017) 'Estimating future cyanobacterial occurrence and importance in lakes: a case study with Planktothrix rubescens in Lake Geneva', *Aquatic Sciences,* 79(2), pp. 249-263.

García-Nieto, A. P., Quintas-Soriano, C., García-Llorente, M., Palomo, I., Montes, C. and Martín-López, B. (2015) 'Collaborative mapping of ecosystem services: The role of stakeholders' profiles', *Ecosystem Services,* 13, pp. 141-152.

George, D. G. and Hurley, M. A. (2004) 'The influence of sampling frequency on the detection of long-term change in three lakes in the English Lake District', *Aquatic Ecosystem Health & Management,* 7(1), pp. 1-14.

Gerdeaux, D., Anneville, O. and Hefti, D. (2006) 'Fishery changes during re-oligotrophication in 11 peri-alpine Swiss and French lakes over the past 30 years', *Acta Oecologica-International Journal of Ecology,* 30(2), pp. 161-167.

Giling, D. P., Nejstgaard, J. C., Berger, S. A., Grossart, H. P., Kirillin, G., Penske, A., Lentz, M., Casper, P., Sareyka, J. and Gessner, M. O. (2017) 'Thermocline deepening boosts ecosystem metabolism: evidence from a large-scale lake enclosure experiment simulating a summer storm', *Glob Chang Biol,* 23(4), pp. 1448-1462.

Godinho, F. N., Segurado, P., Franco, A., Pinheiro, P., Padua, J., Rivaes, R. and Ramos, P. (2019) 'Factors related to fish kill events in Mediterranean reservoirs', *Water Res,* 158, pp. 280-290.

Golosov, S., Maher, O. A., Schipunova, E., Terzhevik, A., Zdorovennova, G. and Kirillin, G. (2007) 'Physical background of the development of oxygen depletion in ice-covered lakes', *Oecologia,* 151(2), pp. 331-40.

Gorjian, S., Sharon, H., Ebadi, H., Kant, K., Scavo, F. B. and Tina, G. M. (2021) 'Recent technical advancements, economics and environmental impacts of floating photovoltaic solar energy conversion systems', *Journal of Cleaner Production,* 278, pp. 124285.

Gray, E., Mackay, E. B., Elliott, J. A., Folkard, A. M. and Jones, I. D. (2020) 'Wide-spread inconsistency in estimation of lake mixed depth impacts interpretation of limnological processes', *Water Res,* 168, pp. 115136.

Greaver, T. L., Clark, C. M., Compton, J. E., Vallano, D., Talhelm, A. F., Weaver, C. P., Band, L. E., Baron, J. S., Davidson, E. A., Tague, C. L., Felker-Quinn, E., Lynch, J. A., Herrick, J. D., Liu, L., Goodale, C. L., Novak, K. J. and Haeuber, R. A. (2016) 'Key ecological responses to nitrogen are altered by climate change', *Nature Climate Change,* 6(9), pp. 836-843.

Grizzetti, B., Lanzanova, D., Liquete, C., Reynaud, A. and Cardoso, A. C. (2016) 'Assessing water ecosystem services for water resource management', *Environmental Science & Policy,* 61, pp. 194-203.

Grizzetti, B., Liquete, C., Pistocchi, A., Vigiak, O., Zulian, G., Bouraoui, F., De Roo, A. and Cardoso, A. C. (2019) 'Relationship between ecological condition and ecosystem services in European rivers, lakes and coastal waters', *Sci Total Environ,* 671, pp. 452-465.

Gyllstrom, M., Hansson, L. A., Jeppesen, E., Garcia-Criado, F., Gross, E., Irvine, K., Kairesalo, T., Kornijow, R., Miracle, M. R., Nykanen, M., Noges, T., Romo, S., Stephen, D., Van Donk, E. and Moss, B. (2005) 'The role of climate in shaping zooplankton communities of shallow lakes', *Limnology and Oceanography,* 50(6), pp. 2008-2021.

Haas, J., Khalighi, J., de la Fuente, A., Gerbersdorf, S. U., Nowak, W. and Chen, P. J. (2020) 'Floating photovoltaic plants: Ecological impacts versus hydropower operation flexibility', *Energy Conversion and Management,* 206, pp. 112414.

Habibian, A. (1994) 'Effect of Temperature-Changes on Water-Main Breaks', *Journal of Transportation Engineering-Asce,* 120(2), pp. 312-321.

Haldna, M. and Haberman, J. (2017) 'How are spring zooplankton and autumn zooplankton influenced by water temperature in a polymictic lake?', *Proceedings of the Estonian Academy of Sciences,* 66(3), pp. 264-278.

Harrison, J. A., Frings, P. J., Beusen, A. H. W., Conley, D. J. and McCrackin, M. L. (2012) 'Global importance, patterns, and controls of dissolved silica retention in lakes and reservoirs', *Global Biogeochemical Cycles,* 26(2).

Haugwitz, F. (2020) *Floating solar PV gains global momentum*: PV Magazine. Available at: https://www.pv-magazine.com/2020/09/22/floating-solar-pv-gains-global-momentum/ (Accessed: 02/05 2021).

Haxton, T., Crooks, S., Jackson, C. R., Barkwith, A. K. A. P., Kelvin, J., Williamson, J., Mackay, J. D., Wang, L., Davies, H., Young, A. and Prudhomme, C. (2012) 'Future flows hydrology data'. Available at: https://doi.org/10.5285/f3723162-4fed-4d9d-92c6-dd17412fa37b (Accessed.

Henderson, R., Chips, M., Cornwell, N., Hitchins, P., Holden, B., Hurley, S., Parsons, S. A., Wetherill, A. and Jefferson, B. (2008) 'Experiences of algae in UK waters: a treatment perspective', *Water and Environment Journal,* 22(3), pp. 184-192.

Hernández-Callejo, L., Gallardo-Saavedra, S. and Alonso-Gómez, V. (2019) 'A review of photovoltaic systems: Design, operation and maintenance', *Solar Energy,* 188, pp. 426-440.

Hernandez, R. R., Armstrong, A., Burney, J., Ryan, G., Moore-O'Leary, K., Diedhiou, I., Grodsky, S. M., Saul-Gershenz, L., Davis, R., Macknick, J., Mulvaney, D., Heath, G. A., Easter, S. B., Hoffacker, M. K., Allen, M. F. and Kammen, D. M. (2019) 'Techno-ecological synergies of solar energy for global sustainability', *Nature Sustainability,* 2(7), pp. 560-568.

Hernandez, R. R., Easter, S. B., Murphy-Mariscal, M. L., Maestre, F. T., Tavassoli, M., Allen, E. B., Barrows, C. W., Belnap, J., Ochoa-Hueso, R., Ravi, S. and Allen, M. F. (2014) 'Environmental impacts of utility-scale solar energy', *Renewable & Sustainable Energy Reviews,* 29, pp. 766-779.

Hessen, D. O., Faafeng, B. A., Smith, V. H., Bakkestuen, V. and Walseng, B. (2006) 'Extrinsic and intrinsic controls of zooplankton diversity in lakes', *Ecology,* 87(2), pp. 433-43.

Ho, J. C., Michalak, A. M. and Pahlevan, N. (2019) 'Widespread global increase in intense lake phytoplankton blooms since the 1980s', *Nature,* 574(7780), pp. 667-670.

Hoffacker, M. K., Allen, M. F. and Hernandez, R. R. (2017) 'Land-Sparing Opportunities for Solar Energy Development in Agricultural Landscapes: A Case Study of the Great Central Valley, CA, United States', *Environ Sci Technol,* 51(24), pp. 14472-14482.

Holm, A. (2017) *Floating Solar Photovoltaics Gaining Ground*: NREL. Available at: https://www.nrel.gov/technical-assistance/blog/posts/floating-solar-photovoltaics-gaining-ground.html (Accessed: 09/11 2017).

Hondzo, M. and Stefan, H. G. (1993) 'Lake Water Temperature Simulation-Model', *Journal of Hydraulic Engineering,* 119(11), pp. 1251-1273.

Horppila, J. and Nurminen, L. (2008) 'Diurnal variation in light intensity underneath floating macrophyte leaves and implications for the behaviour of plant-attached zooplankton', *Biological Rhythm Research,* 39(4), pp. 349-357.

HR Wallingford (2020) *Technical Report, Updated projections of future water availability for the third UK*

Climate Change Risk Assessment (RT002 R05-00.

Huisman, J., Codd, G. A., Paerl, H. W., Ibelings, B. W., Verspagen, J. M. H. and Visser, P. M. (2018) 'Cyanobacterial blooms', *Nat Rev Microbiol,* 16(8), pp. 471-483.

Huisman, J., Sharples, J., Stroom, J. M., Visser, P. M., Kardinaal, W. E. A., Verspagen, J. M. H. and Sommeijer, B. (2004) 'Changes in turbulent mixing shift competition for light between phytoplankton species', *Ecology,* 85(11), pp. 2960-2970.

Huisman, J., van Oostveen, P. and Weissing, F. J. (1999) 'Critical depth and critical turbulence: Two different mechanisms for the development of phytoplankton blooms', *Limnology and Oceanography,* 44(7), pp. 1781-1787.

Hupfer, M. and Lewandowski, J. (2008) 'Oxygen Controls the Phosphorus Release from Lake Sediments - a Long-Lasting Paradigm in Limnology', *International Review of Hydrobiology,* 93(4-5), pp. 415-432.

IEA (2019) *2019 Snapshot of Global PV Markets* IEA-PVPS T1-35:2019). Available at: http://www.iea-pvps.org/fileadmin/dam/public/report/statistics/IEA-PVPS_T1_35_Snapshot2019-Report.pdf.

IEA, IRENA, UNSD, WB and WHO (2019) *Tracking SDG 7: The Energy Progress Report 2019*: Washington: World Bank.

Jäger, C. G., Diehl, S. and Schmidt, G. M. (2008) 'Influence of water-column depth and mixing on phytoplankton biomass, community composition, and nutrients', *Limnology and Oceanography,* 53(6), pp. 2361-2373.

Jane, S. F., Hansen, G. J. A., Kraemer, B. M., Leavitt, P. R., Mincer, J. L., North, R. L., Pilla, R. M., Stetler, J. T., Williamson, C. E., Woolway, R. I., Arvola, L., Chandra, S., DeGasperi, C. L., Diemer, L., Dunalska, J., Erina, O., Flaim, G., Grossart, H. P., Hambright, K. D., Hein, C., Hejzlar, J., Janus, L. L., Jenny, J. P., Jones, J. R., Knoll, L. B., Leoni, B., Mackay, E., Matsuzaki, S. S., McBride, C., Muller-Navarra, D. C., Paterson, A. M., Pierson, D., Rogora, M., Rusak, J. A., Sadro, S., Saulnier-Talbot, E., Schmid, M., Sommaruga, R., Thiery, W., Verburg, P., Weathers, K. C., Weyhenmeyer, G. A., Yokota, K. and Rose, K. C. (2021) 'Widespread deoxygenation of temperate lakes', *Nature,* 594(7861), pp. 66-70.

Jankowski, T., Livingstone, D. M., Bührer, H., Forster, R. and Niederhauser, P. (2006) 'Consequences of the 2003 European heat wave for lake temperature profiles, thermal stability, and hypolimnetic oxygen depletion: Implications for a warmer world', *Limnology and Oceanography,* 51(2), pp. 815-819.

Janssen, A. B. G., Arhonditsis, G. B., Beusen, A., Bolding, K., Bruce, L., Bruggeman, J., Couture, R. M., Downing, A. S., Elliott, J. A., Frassl, M. A., Gal, G., Gerla, D. J., Hipsey, M. R., Hu, F. J., Ives, S. C., Janse, J. H., Jeppesen, E., Johnk, K. D., Kneis, D., Kong, X. Z., Kuiper, J. J., Lehmann, M. K., Lemmen, C., Ozkundakci, D., Petzoldt, T., Rinke, K., Robson, B. J., Sachse, R., Schep, S. A., Schmid, M., Scholten, H., Teurlincx, S., Trolle, D., Troost, T. A., Van Dam, A. A., Van Gerven, L. P. A., Weijerman, M., Wells, S. A. and Mooij, W. M. (2015) 'Exploring, exploiting and evolving diversity of aquatic ecosystem models: a community perspective', *Aquatic Ecology,* 49(4), pp. 513-548.

Janssen, A. B. G., Teurlincx, S., Beusen, A. H. W., Huijbregts, M. A. J., Rost, J., Schipper, A. M., Seelen, L. M. S., Mooij, W. M. and Janse, J. H. (2019) 'PCLake plus : A process-based ecological model to assess the trophic state of stratified and non-stratified freshwater lakes worldwide', *Ecological Modelling,* 396, pp. 23-32.

Jeppesen, E., Brucet, S., Naselli-Flores, L., Papastergiadou, E., Stefanidis, K., Noges, T., Noges, P., Attayde, J. L., Zohary, T., Coppens, J., Bucak, T., Menezes, R. F., Freitas, F. R. S., Kernan, M., Sondergaard, M. and Beklioglu, M. (2015) 'Ecological impacts of global warming and water abstraction on lakes and reservoirs due to changes in water level and related changes in salinity', *Hydrobiologia,* 750(1), pp. 201-227.

Jeppesen, E., Jensen, J. P. and Søndergaard, M. (2002) 'Response of phytoplankton, zooplankton, and fish to re-oligotrophication: An 11 year study of 23 Danish lakes', *Aquatic Ecosystem Health & Management,* 5(1), pp. 31-43.

Jeppesen, E., Meerhoff, M., Holmgren, K., Gonzalez-Bergonzoni, I., Teixeira-de Mello, F., Declerck, S. A. J., De Meester, L., Sondergaard, M., Lauridsen, T. L., Bjerring, R., Conde-Porcuna, J. M., Mazzeo, N., Iglesias, C., Reizenstein, M., Malmquist, H. J., Liu, Z. W., Balayla, D. and Lazzaro, X. (2010) 'Impacts of climate warming on lake fish community structure and potential effects on ecosystem function', *Hydrobiologia,* 646(1), pp. 73-90.

Jesson, D. A., Le Page, B. H., Mulheron, M. J., Smith, P. A., Wallen, A., Cocks, R., Farrow, J. and Whiter, J. T. (2010) 'Thermally induced strains and stresses in cast iron water distribution pipes: an experimental investigation', *Journal of Water Supply Research and Technology-Aqua,* 59(4), pp. 221-229.

Ji, G. H., Xu, Z. X. and Wang, L. Q. (2016) 'Effects of floating-leaved macrophytes on water quality and phytoplankton: an in situ experiment in a Chinese shallow lake', *Desalination and Water Treatment,* 57(57), pp. 27519-27530.

Joh, G., Choi, Y. S., Shin, J.-K. and Lee, J. (2011) 'Problematic algae in the sedimentation and filtration process of water treatment plants', *Journal of Water Supply: Research and Technology-Aqua,* 60(4), pp. 219-230.

Jones, I. and Feuchtmayr, H. (2017) 'Data from automatic water monitoring buoy from Windermere south basin, 2008 to 2011'. Available at: https://doi.org/10.5285/bd37710e-6a53-49f0-9a07-6973408a3342 (Accessed: 01/05/2018).

Joung, D., Leduc, M., Ramcharitar, B., Xu, Y. Y., Isles, P. D. F., Stockwell, J. D., Druschel, G. K., Manley, T. and Schroth, A. W. (2017) 'Winter weather and lake-watershed physical configuration drive phosphorus, iron, and manganese dynamics in water and sediment of ice-covered lakes', *Limnology and Oceanography,* 62(4), pp. 1620-1635.

Jurvelius, J. and Marjomki, T. J. (2008) 'Night, day, sunrise, sunset: do fish under snow and ice recognize the difference?', *Freshwater Biology,* 53(11), pp.???-???

Kabir, E., Kumar, P., Kumar, S., Adelodun, A. A. and Kim, K. H. (2018) 'Solar energy: Potential and future prospects', *Renewable & Sustainable Energy Reviews,* 82, pp. 894-900.

Kalff, J. (2002) *Limnology: inland water ecosystems.* Upper Saddle River, NJ: Prentice Hall.

Kalinowska, K. and Grabowska, M. (2016) 'Autotrophic and heterotrophic plankton under ice in a eutrophic temperate lake', *Hydrobiologia,* 777(1), pp. 111-118.

Kenney, B. C. (1996) 'Physical limnological processes under ice', *Hydrobiologia,* 322(1-3), pp. 85-90.

Kim, S. H., Yoon, S. J., Choi, W. and Choi, K. B. (2016) 'Application of Floating Photovoltaic Energy Generation Systems in South Korea', *Sustainability,* 8(12).

Kiuru, P., Ojala, A., Mammarella, I., Heiskanen, J., Kamarainen, M., Vesala, T. and Huttula, T. (2018) 'Effects of Climate Change on CO2 Concentration and Efflux in a Humic Boreal Lake: A Modeling Study', *Journal of Geophysical Research-Biogeosciences,* 123(7), pp. 2212-2233.

Klausmeier, C. A. and Litchman, E. (2001) 'Algal games: The vertical distribution of phytoplankton in poorly mixed water columns', *Limnology and Oceanography,* 46(8), pp. 1998-2007.

Kleeberg, A., Freidank, A. and Johnk, K. (2013) 'Effects of ice cover on sediment resuspension and phosphorus entrainment in shallow lakes: Combining in situ experiments and wind-wave modeling', *Limnology and Oceanography,* 58(5), pp. 1819-1833.

Knoll, L. B., Sharma, S., Denfeld, B. A., Flaim, G., Hori, Y., Magnuson, J. I., Straile, D. and Weyhenmeyer, G. A. (2019) 'Consequences of lake and river ice loss on cultural ecosystem services', *Limnology and Oceanography Letters,* 4(5), pp. 119-131.

Kobler, U. G. and Schmid, M. (2019) 'Ensemble modelling of ice cover for a reservoir affected by pumped-storage operation and climate change', *Hydrological Processes,* 33(20), pp. 2676-2690.

Kraemer, B. M., Anneville, O., Chandra, S., Dix, M., Kuusisto, E., Livingstone, D. M., Rimmer, A., Schladow, S. G., Silow, E., Sitoki, L. M., Tamatamah, R., Vadeboncoeur, Y. and McIntyre, P. B. (2015) 'Morphometry and average temperature affect lake stratification responses to climate change', *Geophysical Research Letters,* 42(12), pp. 4981-4988.

Kraemer, B. M., Chandra, S., Dell, A. I., Dix, M., Kuusisto, E., Livingstone, D. M., Schladow, S. G., Silow, E., Sitoki, L. M., Tamatamah, R. and McIntyre, P. B. (2017) 'Global patterns in lake ecosystem responses to warming based on the temperature dependence of metabolism', *Glob Chang Biol,* 23(5), pp. 1881-1890.

Kumar, P. (2010) *The economics of ecosystems and biodiversity: ecological and economic foundations.* UNEP/Earthprint.

Lamarque, P., Tappeiner, U., Turner, C., Steinbacher, M., Bardgett, R. D., Szukics, U., Schermer, M. and Lavorel, S. (2011) 'Stakeholder perceptions of grassland ecosystem services in relation to knowledge on soil fertility and biodiversity', *Regional Environmental Change,* 11(4), pp. 791-804.

Le Blanc, D. (2015) 'Towards Integration at Last? The Sustainable Development Goals as a Network of Targets', *Sustainable Development,* 23(3), pp. 176-187.

Lee, N., Grunwald, U., Rosenlieb, E., Mirletz, H., Aznar, A., Spencer, R. and Cox, S. (2020) 'Hybrid floating solar photovoltaics-hydropower systems: Benefits and global assessment of technical potential', *Renewable Energy,* 162, pp. 1415-1427.

Lehmann, P., Aminzadeh, M. and Or, D. (2019) 'Evaporation Suppression From Water Bodies Using Floating Covers: Laboratory Studies of Cover Type, Wind, and Radiation Effects', *Water Resources Research,* 55(6), pp. 4839-4853.

Lenard, T. and Wojciechowska, W. (2013) 'Phytoplankton Diversity and Biomass during Winter with and without Ice Cover in the Context of Climate Change', *Polish Journal of Ecology,* 61(4), pp. 739-748.

Leppäranta, M. (1993) 'A review of analytical models of sea-ice growth', *Atmosphere-Ocean,* 31(1), pp. 123-138.

Lepparanta, M., Heini, A., Jaatinen, E. and Arvola, L. (2012) 'The influence of ice season on the physical and ecological conditions in Lake Vanajanselka, southern Finland', *Water Quality Research Journal of Canada,* 47(3-4), pp. 287-299.

Lerman, A., Imboden, D. M. and Gat, J. R. (1995) *Physics and Chemistry of Lakes.* Berlin: Springer.

Lewis, W. M. (1987) 'Tropical Limnology', *Annual Review of Ecology and Systematics,* 18(1), pp. 159-184.

Liu, H. H., Krishna, V., Leung, J. L., Reindl, T. and Zhao, L. (2018) 'Field experience and performance analysis of floating PV technologies in the tropics', *Progress in Photovoltaics,* 26(12), pp. 957-967.

Liu, L., Wang, Q., Lin, H., Li, H., Sun, Q. and wennersten, R. (2017) 'Power Generation Efficiency and Prospects of Floating Photovoltaic Systems', *Energy Procedia,* 105, pp. 1136-1142.

Liu, M., Zhang, Y., Shi, K., Zhu, G., Wu, Z., Liu, M. and Zhang, Y. (2019) 'Thermal stratification dynamics in a large and deep subtropical reservoir revealed by high-frequency buoy data', *Sci Total Environ,* 651(Pt 1), pp. 614-624.

Livingstone, D. M. and Adrian, R. (2009) 'Modeling the duration of intermittent ice cover on a lake for climate-change studies', *Limnology and Oceanography,* 54(5), pp. 1709-1722.

Livingstone, D. M. and Imboden, D. M. (1989) 'Annual Heat-Balance and Equilibrium Temperature of Lake Aegeri, Switzerland', *Aquatic Sciences,* 51(4), pp. 351-369.

Longhi, M. L. and Beisner, B. E. (2009) 'Environmental factors controlling the vertical distribution of phytoplankton in lakes', *Journal of Plankton Research,* 31(10), pp. 1195-1207.

Lund, J. (1949) 'Studies on Asterionella: I. The origin and nature of the cells producing seasonal maxima', *The Journal of Ecology*, pp. 389-419.

Maberly, S. C. and Elliott, J. A. (2012) 'Insights from long-term studies in the Windermere catchment: external stressors, internal interactions and the structure and function of lake ecosystems', *Freshwater Biology,* 57(2), pp. 233-243.

Macintyre, S. (1993) 'Vertical Mixing in a Shallow, Eutrophic Lake - Possible Consequences for the Light Climate of Phytoplankton', *Limnology and Oceanography,* 38(4), pp. 798-817.

Mackereth, F., Heron, J., Talling, J. and Association, F. B. (1978) 'Water analysis: some revised methods for limnologists'.

Maestre-Valero, J. F., Martinez-Alvarez, V., Gallego-Elvira, B. and Pittaway, P. (2011) 'Effects of a suspended shade cloth cover on water quality of an agricultural reservoir for irrigation', *Agricultural Water Management,* 100(1), pp. 70-75.

Maestre-Valero, J. F., Martinez-Alvarez, V. and Nicolas, E. (2013) 'Physical, chemical and microbiological effects of suspended shade cloth covers on stored water for irrigation', *Agricultural Water Management,* 118, pp. 70-78.

Maghami, M. R., Hizam, H., Gomes, C., Radzi, M. A., Rezadad, M. I. and Hajighorbani, S. (2016) 'Power loss due to soiling on solar panel: A review', *Renewable & Sustainable Energy Reviews,* 59, pp. 1307-1316.

Maltby, E., Ormerod, S., Acreman, M., Dunbar, M., Jenkins, A., Maberly, S., Newman, J., Blackwell, M. and Ward, R. (2011) 'Freshwaters: openwaters, wetlands and floodplains', *UK National Ecosystem Assessment: understanding nature's value to society*. Cambridge, UK: UNEP-WCMC, pp. 295-360.

Manny, B. A., Johnson, W. C. and Wetzel, R. G. 'Nutrient additions by waterfowl to lakes and reservoirs: predicting their effects on productivity and water quality'. *Aquatic Birds in the Trophic Web of Lakes*. Dordrecht: Springer Netherlands, 121-132.

Mantzouki, E., Beklioglu, M., Brookes, J. D., Domis, L. N. S., Dugan, H. A., Doubek, J. P., Grossart, H. P., Nejstgaard, J. C., Pollard, A. I., Ptacnik, R., Rose, K. C., Sadro, S., Seelen, L., Skaff, N. K., Teubner, K., Weyhenmeyer, G. A. and Ibelings, B. W. (2018) 'Snapshot Surveys for Lake Monitoring, More Than a Shot in the Dark', *Front Ecol Evol,* 6.

Mantzouki, E., Visser, P. M., Bormans, M. and Ibelings, B. W. (2016) 'Understanding the key ecological traits of cyanobacteria as a basis for their management and control in changing lakes', *Aquatic Ecology,* 50(3), pp. 333-350.

Markelov, I., Couture, R. M., Fischer, R., Haande, S. and Van Cappellen, P. (2019) 'Coupling Water Column and Sediment Biogeochemical Dynamics: Modeling Internal Phosphorus Loading, Climate Change Responses, and Mitigation Measures in Lake Vansjo, Norway', *Journal of Geophysical Research-Biogeosciences,* 124(12), pp. 3847-3866.

Mathijssen, D., Hofs, B., Spierenburg-Sack, E., van Asperen, R., van der Wal, B., Vreeburg, J. and Ketelaars, H. (2020) 'Potential impact of floating solar panels on water quality in reservoirs; pathogens and leaching', *Water Practice and Technology,* 15(3), pp. 807-811.

Matthews, P. C. and Heaney, S. I. (1987) 'Solar Heating and Its Influence on Mixing in Ice-Covered Lakes', *Freshwater Biology,* 18(1), pp. 135-149.

McCullough, E. B. and Matson, P. A. (2016) 'Evolution of the knowledge system for agricultural development in the Yaqui Valley, Sonora, Mexico', *Proc Natl Acad Sci U S A,* 113(17), pp. 4609-14.

Meis, S., Thackeray, S. J. and Jones, I. D. (2009) 'Effects of recent climate change on phytoplankton phenology in a temperate lake', *Freshwater Biology,* 54(9), pp. 1888-1898.

Menzel, S. and Teng, J. (2010) 'Ecosystem services as a stakeholder-driven concept for conservation science', *Conserv Biol,* 24(3), pp. 907-9.

Merritt, D. M. and Wohl, E. E. (2006) 'Plant dispersal along rivers fragmented by dams', *River Research and Applications,* 22(1), pp. 1-26.

Met Office (2019) 'MIDAS Open: UK daily weather observation data, v201908'. Available at: https://catalogue.ceda.ac.uk/uuid/6ad6792f44c84c228651b01d182d9d73 (Accessed: 16/02/21).

Met Office Hadley Centre (2018) 'UKCP18 Probabilistic Projections by UK River Basins for 1961-2100'. Available at: https://catalogue.ceda.ac.uk/uuid/10538cf7a8d84e5e872883ea09a674f3 (Accessed.

Meyer, K. M., Mooij, W. M., Vos, M., Hol, W. H. G. and van der Putten, W. H. (2009) 'The power of simulating experiments', *Ecological Modelling,* 220(19), pp. 2594-2597.

Meyerhoff, J., Klefoth, T. and Arlinghaus, R. (2019) 'The value artificial lake ecosystems provide to recreational anglers: Implications for management of biodiversity and outdoor recreation', *J Environ Manage,* 252, pp. 109580.

Miranda, L. E., Coppola, G. and Boxrucker, J. (2020) 'Reservoir Fish Habitats: A Perspective on Coping with Climate Change', *Reviews in Fisheries Science & Aquaculture,* 28(4), pp. 478-498.

Mitchell, T. R. and Ridgwell, T. (1971) 'The frequency of salmonellae in wild ducks', *J Med Microbiol,* 4(3), pp. 359-61.

Moe, S. J., Haande, S. and Couture, R. M. (2016) 'Climate change, cyanobacteria blooms and ecological status of lakes: A Bayesian network approach', *Ecological Modelling,* 337, pp. 330-347.

Momayez, M., Wilson, T., Cronin, A., Annavarapu, S. and Conant, B. 'An investigation to use tailing ponds as solar photovoltaic farms'. *26th Annual Meetings of the American Society of Mining and Reclamation and 11th Billings Land Reclamation Symposium 2009*, 817-832.

Moore, T. N., Mesman, J. P., Ladwig, R., Feldbauer, J., Olsson, F., Pilla, R. M., Shatwell, T., Venkiteswaran, J. J., Delany, A. D., Dugan, H., Rose, K. C. and Read, J. S. (2021)

'LakeEnsemblR: An R package that facilitates ensemble modelling of lakes', *Environmental Modelling & Software,* 143.

Mora-Rodríguez, J., Delgado-Galván, X., Ortiz-Medel, J., Ramos, H. M., Fuertes-Miquel, V. S. and López-Jiménez, P. A. (2015) 'Pathogen intrusion flows in water distribution systems: according to orifice equations', *Journal of Water Supply: Research and Technology - Aqua,* 64(8), pp. 857-869.

Moriasi, D. N., Arnold, J. G., Van Liew, M. W., Bingner, R. L., Harmel, R. D. and Veith, T. L. (2007) 'Model evaluation guidelines for systematic quantification of accuracy in watershed simulations', *Transactions of the ASABE,* 50(3), pp. 885-900.

Moss, B., Kosten, S., Meerhoff, M., Battarbee, R. W., Jeppesen, E., Mazzeo, N., Havens, K., Lacerot, G., Liu, Z. W., De Meester, L., Paerl, H. and Scheffer, M. (2011) 'Allied attack: climate change and eutrophication', *Inland Waters,* 1(2), pp. 101-105.

Muhlfeld, C. C., Kovach, R. P., Jones, L. A., Al-Chokhachy, R., Boyer, M. C., Leary, R. F., Lowe, W. H., Luikart, G. and Allendorf, F. W. (2014) 'Invasive hybridization in a threatened species is accelerated by climate change', *Nature Climate Change,* 4(7), pp. 620-624.

Nash, J. E. and Sutcliffe, J. V. (1970) 'River flow forecasting through conceptual models part I — A discussion of principles', *Journal of Hydrology,* 10(3), pp. 282-290.

National River Flow Archive (2018) '73010 - Leven at Newby Bridge'. Available at: https://nrfa.ceh.ac.uk/data/station/info/73010 (Accessed 25/07/2018).

Nelson, E. J., Kareiva, P., Ruckelshaus, M., Arkema, K., Geller, G., Girvetz, E., Goodrich, D., Matzek, V., Pinsky, M., Reid, W., Saunders, M., Semmens, D. and Tallis, H. (2013) 'Climate change's impact on key ecosystem services and the human well-being they support in the US', *Frontiers in Ecology and the Environment,* 11(9), pp. 483-493.

Niemisto, J. P. and Horppila, J. (2007) 'The contribution of ice cover to sediment resuspension in a shallow temperate lake: possible effects of climate change on internal nutrient loading', *J Environ Qual,* 36(5), pp. 1318-23.

North, R. P., North, R. L., Livingstone, D. M., Koster, O. and Kipfer, R. (2014) 'Long-term changes in hypoxia and soluble reactive phosphorus in the hypolimnion of a large temperate lake: consequences of a climate regime shift', *Glob Chang Biol,* 20(3), pp. 811-23.

Nurnberg, G. K. (2004) 'Quantified hypoxia and anoxia in lakes and reservoirs', *ScientificWorldJournal,* 4, pp. 42-54.

O'Neil, J. M., Davis, T. W., Burford, M. A. and Gobler, C. J. (2012) 'The rise of harmful cyanobacteria blooms: The potential roles of eutrophication and climate change', *Harmful Algae,* 14, pp. 313-334.

O'Reilly, C. M., Sharma, S., Gray, D. K., Hampton, S. E., Read, J. S., Rowley, R. J., Schneider, P., Lenters, J. D., McIntyre, P. B., Kraemer, B. M., Weyhenmeyer, G. A., Straile, D., Dong, B., Adrian, R., Allan, M. G., Anneville, O., Arvola, L., Austin, J., Bailey, J. L., Baron, J. S., Brookes, J. D., de Eyto, E., Dokulil, M. T., Hamilton, D. P., Havens, K., Hetherington, A. L., Higgins, S. N., Hook, S., Izmest'eva, L. R., Joehnk, K. D., Kangur, K., Kasprzak, P., Kumagai, M., Kuusisto, E., Leshkevich, G., Livingstone, D. M., MacIntyre, S., May, L., Melack, J. M., Mueller-Navarra, D. C., Naumenko, M., Noges, P., Noges, T., North, R. P., Plisnier, P. D., Rigosi, A., Rimmer, A., Rogora, M., Rudstam, L. G., Rusak, J. A., Salmaso, N., Samal, N. R., Schindler, D. E., Schladow, S. G., Schmid, M., Schmidt, S. R., Silow, E., Soylu, M. E., Teubner, K., Verburg, P., Voutilainen, A., Watkinson, A., Williamson, C. E. and Zhang, G. Q. (2015) 'Rapid and highly variable warming of lake surface waters around the globe', *Geophysical Research Letters,* 42(24), pp. 10773-10781.

Office for National Statistics (2021) *UK natural capital accounts 2021*. Available at: https://www.ons.gov.uk/economy/environmentalaccounts/bulletins/uknaturalcapitalaccounts/2021.

Oke, T. R. (2002) *Boundary layer climates.* Routledge.

Oliveira-Pinto, S. and Stokkermans, J. (2020) 'Assessment of the potential of different floating solar technologies - Overview and analysis of different case studies', *Energy Conversion and Management,* 211, pp. 112747.

Oveisy, A., Rao, Y. R., Leon, L. F. and Bocaniov, S. A. (2014) 'Three-dimensional winter modeling and the effects of ice cover on hydrodynamics, thermal structure and water quality in Lake Erie', *Journal of Great Lakes Research,* 40, pp. 19-28.

Ozkundakci, D., Gsell, A. S., Hintze, T., Tauscher, H. and Adrian, R. (2016) 'Winter severity determines functional trait composition of phytoplankton in seasonally ice-covered lakes', *Glob Chang Biol,* 22(1), pp. 284-98.

Paerl, H. W. (2014) 'Mitigating harmful cyanobacterial blooms in a human- and climatically-impacted world', *Life (Basel),* 4(4), pp. 988-1012.

Paerl, H. W., Havens, K. E., Hall, N. S., Otten, T. G., Zhu, M., Xu, H., Zhu, G. and Qin, B. (2019) 'Mitigating a global expansion of toxic cyanobacterial blooms: confounding effects and challenges posed by climate change', *Marine and Freshwater Research,* 71(5), pp. 579-592.

Paerl, H. W. and Huisman, J. (2008) 'Climate. Blooms like it hot', *Science,* 320(5872), pp. 57-8.

Paerl, H. W. and Huisman, J. (2009) 'Climate change: a catalyst for global expansion of harmful cyanobacterial blooms', *Environ Microbiol Rep,* 1(1), pp. 27-37.

Paerl, H. W. and Paul, V. J. (2012) 'Climate change: links to global expansion of harmful cyanobacteria', *Water Res,* 46(5), pp. 1349-63.

Page, T., Smith, P. J., Beven, K. J., Jones, I. D., Elliott, J. A., Maberly, S. C., Mackay, E. B., De Ville, M. and Feuchtmayr, H. (2017) 'Constraining uncertainty and process-representation in an algal community lake model using high frequency in-lake observations', *Ecological Modelling,* 357, pp. 1-13.

Page, T., Smith, P. J., Beven, K. J., Jones, I. D., Elliott, J. A., Maberly, S. C., Mackay, E. B., De Ville, M. and Feuchtmayr, H. (2018) 'Adaptive forecasting of phytoplankton communities', *Water Res,* 134, pp. 74-85.

Pernica, P., North, R. L. and Baulch, H. M. (2017) 'In the cold light of day: the potential importance of under-ice convective mixed layers to primary producers', *Inland Waters,* 7(2), pp. 138-150.

Piana, V., Kahl, A., Saviozzi, C. and Schumann, R. (2021) 'Floating PV in mountain artificial lakes: a checklist for site assessment', *Renew. Energy Environ. Sustain.,* 6, pp. 4.

Pilla, R. M. and Couture, R. M. (2021) 'Attenuation of photosynthetically active radiation and ultraviolet radiation in response to changing dissolved organic carbon in browning lakes: Modeling and parametrization', *Limnology and Oceanography,* 66(6), pp. 2278-2289.

Pinto, P. D., Allende, L. and O'Farrell, I. (2007) 'Influence of free-floating plants on the structure of a natural phytoplankton assemblage: an experimental approach', *Journal of Plankton Research,* 29(1), pp. 47-56.

Plaas, H. E. and Paerl, H. W. (2021) 'Toxic Cyanobacteria: A Growing Threat to Water and Air Quality', *Environmental Science & Technology,* 55(1), pp. 44-64.

Pogson, M., Hastings, A. and Smith, P. (2013) 'How does bioenergy compare with other land-based renewable energy sources globally?', *GCB Bioenergy,* 5(5), pp. 513-524.

Porter, J. H. (2021) 'Sensors in the landscape', *The Routledge Handbook of Landscape Ecology*, pp. 250-263.

Postel, S. and Carpenter, S. (1997) 'Freshwater ecosystem services', *Nature's services: Societal dependence on natural ecosystems*.

Power Technology (2021) *World's biggest floating solar farms*. Available at: https://www.power-technology.com/features/worlds-biggest-floating-solar-farms/ (Accessed: 09/07/21.

Powers, S. M., Baulch, H. M., Hampton, S. E., Labou, S. G., Lottig, N. R. and Stanley, E. H. (2017a) 'Nitrification contributes to winter oxygen depletion in seasonally frozen forested lakes', *Biogeochemistry,* 136(2), pp. 119-129.

Powers, S. M., Labou, S. G., Baulch, H. M., Hunt, R. J., Lottig, N. R., Hampton, S. E. and Stanley, E. H. (2017b) 'Ice duration drives winter nitrate accumulation in north temperate lakes', *Limnology and Oceanography Letters,* 2(5), pp. 177-186.

Pringle, A. M., Handler, R. M. and Pearce, J. M. (2017) 'Aquavoltaics: Synergies for dual use of water area for solar photovoltaic electricity generation and aquaculture', *Renewable & Sustainable Energy Reviews,* 80, pp. 572-584.

Prudhomme, C., Haxton, T., Crooks, S., Jackson, C., Barkwith, A., Williamson, J., Kelvin, J., Mackay, J., Wang, L., Young, A. and Watts, G. (2013) 'Future Flows Hydrology: an ensemble of daily river flow and monthly groundwater levels for use for climate change impact assessment across Great Britain', *Earth System Science Data,* 5(1), pp. 101-107.

Rahel, F. J. and Olden, J. D. (2008) 'Assessing the effects of climate change on aquatic invasive species', *Conserv Biol,* 22(3), pp. 521-33.

Ramsbottom, A. (1976) 'Depth charts ofthe Cumbrian lakes', *Sci. Publ. Freshwater Biol. Assoc*.

Randle-Boggis, R. J., White, P. C. L., Cruz, J., Parker, G., Montag, H., Scurlock, J. M. O. and Armstrong, A. (2020) 'Realising co-benefits for natural capital and ecosystem services from solar parks: A co-developed, evidence-based approach', *Renewable & Sustainable Energy Reviews,* 125, pp. 109775.

Read, J. S., Hamilton, D. P., Desai, A. R., Rose, K. C., MacIntyre, S., Lenters, J. D., Smyth, R. L., Hanson, P. C., Cole, J. J., Staehr, P. A., Rusak, J. A., Pierson, D. C., Brookes, J. D., Laas, A. and Wu, C. H. (2012) 'Lake-size dependency of wind shear and convection as controls on gas exchange', *Geophysical Research Letters,* 39, pp. 5.

Read, J. S., Hamilton, D. P., Jones, I. D., Muraoka, K., Winslow, L. A., Kroiss, R., Wu, C. H. and Gaiser, E. (2011) 'Derivation of lake mixing and stratification indices from high-resolution lake buoy data', *Environmental Modelling & Software,* 26(11), pp. 1325-1336.

Redón-Santafé, M., Ferrer-Gisbert, P.-S., Sánchez-Romero, F.-J., Torregrosa Soler, J. B., Gozalvez, F., Javier, J. and Ferrer Gisbert, C. M. (2014) 'Implementation of a photovoltaic floating cover for irrigation reservoirs', *Journal of Cleaner Production,* 66, pp. 568-570.

REN21 (2020) *Renewables 2020 Global Status Report*, Paris. Available at: https://www.ren21.net/reports/global-status-report/.

Reynaud, A. and Lanzanova, D. (2017) 'A Global Meta-Analysis of the Value of Ecosystem Services Provided by Lakes', *Ecological Economics,* 137, pp. 184-194.

Reynolds, C., Irish, T. and Elliott, A. (2005) 'A modelling approach to the development of an active management strategy for the Queen Elizabeth II reservoir', *Freshwater Forum,* 23(1).

Reynolds, C. S. (1984) *The ecology of freshwater phytoplankton.* Cambridge university press.

Reynolds, C. S. (1997) *Vegetation processes in the pelagic: a model for ecosystem theory.* Luhe, Hamburg: Ecology Institute Oldendorf.

Reynolds, C. S. (2006) *The Ecology of Phytoplankton. Ecology, Biodiversity and Conservation* Cambridge: Cambridge University Press.

Reynolds, C. S., Huszar, V., Kruk, C., Naselli-Flores, L. and Melo, S. (2002) 'Towards a functional classification of the freshwater phytoplankton', *Journal of Plankton Research,* 24(5), pp. 417-428.

Reynolds, C. S., Irish, A. E. and Elliott, J. A. (2001) 'The ecological basis for simulating phytoplankton responses to environmental change (PROTECH)', *Ecological Modelling,* 140(3), pp. 271-291.

Reynolds, C. S., Oliver, R. L. and Walsby, A. E. (1987) 'Cyanobacterial dominance: the role of buoyancy regulation in dynamic lake environments', *New Zealand Journal of Marine and Freshwater Research,* 21(3), pp. 379-390.

Richards, J., Moore, R. D. and Forrest, A. L. (2012) 'Late-summer thermal regime of a small proglacial lake', *Hydrological Processes,* 26(18), pp. 2687-2695.

Riley, M. J. and Stefan, H. G. (1988) 'Minlake - a Dynamic Lake Water-Quality Simulation-Model', *Ecological Modelling,* 43(3-4), pp. 155-182.

Rodrigues, I. S., Ramalho, G. L. B. and Medeiros, P. H. A. (2020) 'Potential of floating photovoltaic plant in a tropical reservoir in Brazil', *Journal of Environmental Planning and Management,* 63(13), pp. 2334-2356.

Rodríguez, J. P., Beard, T. D., Bennett, E. M., Cumming, G. S., Cork, S. J., Agard, J., Dobson, A. P. and Peterson, G. D. (2006) 'Trade-offs across Space, Time, and Ecosystem Services', *Ecology and Society,* 11(1).

Rolland, D. C., Bourget, S., Warren, A., Laurion, I. and Vincent, A. F. (2013) 'Extreme variability of cyanobacterial blooms in an urban drinking water supply', *Journal of Plankton Research,* 35(4), pp. 744-758.

Rooney, G. G. and Jones, I. D. (2010) 'Coupling the 1-D lake model FLake to the community land-surface model JULES', *Boreal environment research,* 15(5), pp. 501-512.

Rosa-Clot, M. and Marco Tina, G. (2020) *Floating PV Plants.* San Diego, UNITED STATES: Elsevier Science & Technology.

Rosa-Clot, M. and Tina, G. M. (2018) 'Floating Plants and Environmental Aspects', in Rosa-Clot, M. & Tina, G.M. (eds.) *Submerged and Floating Photovoltaic Systems*: Academic Press, pp. 185-212.

Ross, O. N. and Sharples, J. (2008) 'Swimming for survival: A role of phytoplankton motility in a stratified turbulent environment', *Journal of Marine Systems,* 70(3-4), pp. 248-262.

Sacramento, E. M. d., Carvalho, P. C. M., de Araújo, J. C., Riffel, D. B., Corrêa, R. M. d. C. and Pinheiro Neto, J. S. (2015) 'Scenarios for use of floating photovoltaic plants in Brazilian reservoirs', *Iet Renewable Power Generation,* 9(8), pp. 1019-1024.

Sahu, A., Yadav, N. and Sudhakar, K. (2016) 'Floating photovoltaic power plant: A review', *Renewable & Sustainable Energy Reviews,* 66, pp. 815-824.

Sahu, A. K. and Sudhalzar, K. (2019) 'Effect of UV exposure on bimodal HDPE floats for floating solar application', *Journal of Materials Research and Technology-Jmr&T,* 8(1), pp. 147-156.

Said, S. A. M., Hassan, G., Walwil, H. M. and Al-Aqeeli, N. (2018) 'The effect of environmental factors and dust accumulation on photovoltaic modules and dust-accumulation mitigation strategies', *Renewable & Sustainable Energy Reviews,* 82, pp. 743-760.

Salk, K. R., Venkiteswaran, J. J., Couture, R. M., Higgins, S. N., Paterson, M. J. and Schiff, S. L. (2022) 'Warming combined with experimental eutrophication intensifies lake phytoplankton blooms', *Limnology and Oceanography,* 67(1), pp. 147-158.

Salmaso, N., Naselli-Flores, L. and Padisak, J. (2015) 'Functional classifications and their application in phytoplankton ecology', *Freshwater Biology,* 60(4), pp. 603-619.

Salmi, P., Lehmijoki, A. and Salonen, K. (2014a) 'Development of picoplankton during natural and enhanced mixing under late-winter ice', *Journal of Plankton Research,* 36(6), pp. 1501-1511.

Salmi, P., Malin, I. and Salonen, K. (2014b) 'Pumping of epilimnetic water into hypolimnion improves oxygen but not necessarily nutrient conditions in a lake recovering from eutrophication', *Inland Waters,* 4(4), pp. 425-434.

Salmi, P. and Salonen, K. (2016) 'Regular build-up of the spring phytoplankton maximum before ice-break in a boreal lake', *Limnology and Oceanography,* 61(1), pp. 240-253.

Saloranta, T. and Andersen, T. (2004) *MyLake v1.1: technical documentation & user’s guide*: Norwegian Institute for Water Research (NIVA report 4838. Available at: https://niva.brage.unit.no/niva-xmlui/bitstream/handle/11250/212445/4838_200dpi.pdf?sequence=2&isAllowed=y.

Saloranta, T. M. (2000) 'Modeling the evolution of snow, snow ice and ice in the Baltic Sea', *Tellus Series a-Dynamic Meteorology and Oceanography,* 52(1), pp. 93-108.

Saloranta, T. M. and Andersen, T. (2007) 'MyLake - A multi-year lake simulation model code suitable for uncertainty and sensitivity analysis simulations', *Ecological Modelling,* 207(1), pp. 45-60.

Sanchez, R. G., Kougias, I., Moner-Girona, M., Fahl, F. and Jäger-Waldau, A. (2021) 'Assessment of floating solar photovoltaics potential in existing hydropower reservoirs in Africa', *Renewable Energy,* 169, pp. 687-699.

Santafe, M. R., Gisbert, P. S. F., Romero, F. J. S., Soler, J. B. T., Gozalvez, J. J. F. and Gisbert, C. M. F. (2014) 'Implementation of a photovoltaic floating cover for irrigation reservoirs', *Journal of Cleaner Production,* 66, pp. 568-570.

Saulnier-Talbot, E. and Lavoie, I. (2018) 'Uncharted waters: the rise of human-made aquatic environments in the age of the "Anthropocene"', *Anthropocene,* 23, pp. 29-42.

Scherer, N. M., Gibbons, H. L., Stoops, K. B. and Muller, M. (1995) 'Phosphorus Loading of an Urban Lake by Bird Droppings', *Lake and Reservoir Management,* 11(4), pp. 317-327.

Seto, M., Takamura, N. and Iwasa, Y. (2013) 'Individual and combined suppressive effects of submerged and floating-leaved macrophytes on algal blooms', *J Theor Biol,* 319, pp. 122-33.

Shan, K., Ouyang, T., Wang, X., Yang, H., Zhou, B., Wu, Z. and Shang, M. (2022) 'Temporal prediction of algal parameters in Three Gorges Reservoir based on highly time-resolved monitoring and long short-term memory network', *Journal of Hydrology,* 605, pp. 127304.

Shimoda, Y. and Arhonditsis, G. B. (2016) 'Phytoplankton functional type modelling: Running before we can walk? A critical evaluation of the current state of knowledge', *Ecological Modelling,* 320, pp. 29-43.

Silverio, N. M., Barros, R. M., Tiago, G. L., Redon-Santafe, M., dos Santos, I. F. S. and Valerio, V. E. D. (2018) 'Use of floating PV plants for coordinated operation with hydropower plants: Case study of the hydroelectric plants of the Sao Francisco River basin', *Energy Conversion and Management,* 171, pp. 339-349.

Solar Asset Management (2018) *TOP 70 FLOATING SOLAR PV PLANTS*: Solar Asset Management. Available at: https://solarassetmanagement.asia/news/top-70-floating-solar-pv-plants (Accessed: July 2018).

Solarplaza (2019) *A Comprehensive Overview of 200+ Global Floating Solar Plants*. Available at: https://www.solarplaza.com/channels/future-grid/12067/200-global-floating-solar-plants/.

Spears, B. M., Chapman, D., Carvalho, L., Rankinen, K., Stefanidis, K., Ives, S., Vuorio, K. and Birk, S. (2021) 'Assessing multiple stressor effects to inform climate change management responses in three European catchments', *Inland Waters,* 12(1), pp. 94-106.

Spencer, R. S., Macknick, J., Aznar, A., Warren, A. and Reese, M. O. (2019) 'Floating Photovoltaic Systems: Assessing the Technical Potential of Photovoltaic Systems on Man-Made

Water Bodies in the Continental United States', *Environ Sci Technol,* 53(3), pp. 1680-1689.

Spoljar, M., Drazina, T., Lajtner, J., Sertic, M. D., Radanovic, I., Wallace, R. L., Matulic, D. and Tomljanovic, T. (2018) 'Zooplankton assemblage in four temperate shallow waterbodies in association with habitat heterogeneity and alternative states', *Limnologica,* 71, pp. 51-61.

Staehr, P. A., Christensen, J. P. A., Batt, R. D. and Read, J. S. (2012a) 'Ecosystem metabolism in a stratified lake', *Limnology and Oceanography,* 57(5), pp. 1317-1330.

Staehr, P. A., Testa, J. M., Kemp, W. M., Cole, J. J., Sand-Jensen, K. and Smith, S. V. (2012b) 'The metabolism of aquatic ecosystems: history, applications, and future challenges', *Aquatic Sciences,* 74(1), pp. 15-29.

Steffensen, D. A. (2008) 'Economic cost of cyanobacterial blooms', in Hudnell, H.K. (ed.) *Cyanobacterial Harmful Algal Blooms: State of the Science and Research Needs*. New York, NY: Springer New York, pp. 855-865.

Stewart-Oaten, A., Murdoch, W. W. and Parker, K. R. (1986) 'Environmental impact assessment:" Pseudoreplication" in time?', *Ecology,* 67(4), pp. 929-940.

Stiers, I. and Triest, L. (2017) 'Impact of non-native invasive plant species cover on phytoplankton and zooplankton communities in temperate ponds', *Aquatic Invasions,* 12(3), pp. 385-395.

Stiubiener, U., da Silva, T. C., Trigoso, F. B. M., Benedito, R. D. and Teixeira, J. C. (2020) 'PV power generation on hydro dam's reservoirs in Brazil: A way to improve operational flexibility', *Renewable Energy,* 150, pp. 765-776.

Sulaeman, S., Brown, E., Quispe-Abad, R. and Muller, N. (2021) 'Floating PV system as an alternative pathway to the amazon dam underproduction', *Renewable & Sustainable Energy Reviews,* 135, pp. 110082.

Ta, T. (2019) *Queen Elizabeth II (QEII) Reservoir Computational Fluid Dynamics (CFD) Model for Cryptosporidium Residence Time*, Unpublished report prepared for Thames Water.

Taboada, M. E., Caceres, L., Graber, T. A., Galleguillos, H. R., Cabeza, L. F. and Rojas, R. (2017) 'Solar water heating system and photovoltaic floating cover to reduce evaporation: Experimental results and modeling', *Renewable Energy,* 105, pp. 601-615.

Talling, J. 1974. Sampling techniques and methods for estimating quantity and quality of biomass: general outline of spectrophotometric methods. A Manual on Methods for Measuring Primary Productivity in Aquatic Environments, Handbook No. 12. 2 ed.: Blackwell Scientific Publications, Oxford.

Talling, J. F. (2001) 'Environmental controls on the functioning of shallow tropical lakes', *Hydrobiologia,* 458(1), pp. 1-8.

Tawney, L. and Weischer, L. (2011) 'Innovation and technology transfer: supporting low carbon development with climate finance', *World Resources Institute Working Paper (January 2011), http://www.wri.org/publication/innovation-and-technology-transfer.*

Taylor, C. J., Pedregal, D. J., Young, P. C. and Tych, W. (2007) 'Environmental time series analysis and forecasting with the Captain toolbox', *Environmental Modelling & Software,* 22(6), pp. 797-814.

Thackeray, S. J., Henrys, P. A., Feuchtmayr, H., Jones, I. D., Maberly, S. C. and Winfield, I. J. (2013) 'Food web de-synchronization in England's largest lake: an assessment based on multiple phenological metrics', *Glob Chang Biol,* 19(12), pp. 3568-80.

Thackeray, S. J., Henrys, P. A., Hemming, D., Bell, J. R., Botham, M. S., Burthe, S., Helaouet, P., Johns, D. G., Jones, I. D., Leech, D. I., Mackay, E. B., Massimino, D., Atkinson, S., Bacon, P. J., Brereton, T. M., Carvalho, L., Clutton-Brock, T. H., Duck, C., Edwards, M., Elliott, J. M., Hall, S. J., Harrington, R., Pearce-Higgins, J. W., Hoye, T. T., Kruuk, L. E., Pemberton, J. M., Sparks, T. H., Thompson, P. M., White, I., Winfield, I. J. and Wanless, S. (2016) 'Phenological sensitivity to climate across taxa and trophic levels', *Nature,* 535(7611), pp. 241-5.

Thackeray, S. J., Jones, I. D. and Maberly, S. C. (2008) 'Long-term change in the phenology of spring phytoplankton: species-specific responses to nutrient enrichment and climatic change', *Journal of Ecology,* 96(3), pp. 523-535.

Thomas, R. Q., Figueiredo, R. J., Daneshmand, V., Bookout, B. J., Puckett, L. K. and Carey, C. C. (2020) 'A Near-Term Iterative Forecasting System Successfully Predicts Reservoir Hydrodynamics and Partitions Uncertainty in Real Time', *Water Resources Research,* 56(11), pp. e2019WR026138.

Thronson, A. and Quigg, A. (2008) 'Fifty-five years of fish kills in coastal Texas', *Estuaries and coasts,* 31(4), pp. 802-813.

Tickner, D., Opperman, J. J., Abell, R., Acreman, M., Arthington, A. H., Bunn, S. E., Cooke, S. J., Dalton, J., Darwall, W. and Edwards, G. (2020) 'Bending the curve of global freshwater biodiversity loss: an emergency recovery plan', *BioScience,* 70(4), pp. 330-342.

Till, A., Rypel, A. L., Bray, A. and Fey, S. B. (2019) 'Fish die-offs are concurrent with thermal extremes in north temperate lakes', *Nature Climate Change,* 9(8), pp. 637-+.

Tranvik, L. J., Downing, J. A., Cotner, J. B., Loiselle, S. A., Striegl, R. G., Ballatore, T. J., Dillon, P., Finlay, K., Fortino, K., Knoll, L. B., Kortelainen, P. L., Kutser, T., Larsen, S., Laurion, I., Leech, D. M., McCallister, S. L., McKnight, D. M., Melack, J. M., Overholt, E., Porter, J. A., Prairie, Y., Renwick, W. H., Roland, F., Sherman, B. S., Schindler, D. W., Sobek, S., Tremblay, A., Vanni, M. J., Verschoor, A. M., von Wachenfeldt, E. and Weyhenmeyer, G. A. (2009) 'Lakes and reservoirs as regulators of carbon cycling and climate', *Limnology and Oceanography,* 54(6), pp. 2298-2314.

Trapani, K. and Millar, D. L. (2016) 'Floating Photovoltaic Arrays to Power the Mining Industry: A Case Study for the McFaulds Lake (Ring of Fire)', *Environmental Progress & Sustainable Energy,* 35(3), pp. 898-905.

Trapani, K. and Redon Santafe, M. (2015) 'A review of floating photovoltaic installations: 2007-2013', *Progress in Photovoltaics,* 23(4), pp. 524-532.

Trolle, D., Elliott, J. A., Mooij, W. M., Janse, J. H., Bolding, K., Hamilton, D. P. and Jeppesen, E. (2014) 'Advancing projections of phytoplankton responses to climate change through ensemble modelling', *Environmental Modelling & Software,* 61, pp. 371-379.

Twiss, M. R., McKay, R. M. L., Bourbonniere, R. A., Bullerjahn, G. S., Carrick, H. J., Smith, R. E. H., Winter, J. G., D'Souza, N. A., Furey, P. C., Lashaway, A. R., Saxton, M. A. and Wilhelm, S. W. (2012) 'Diatoms abound in ice-covered Lake Erie: An investigation of offshore winter limnology in Lake Erie over the period 2007 to 2010', *Journal of Great Lakes Research,* 38(1), pp. 18-30.

United Nations (2015) *Transforming our World: The 2030 Agenda for Sustainable Development*.

Vachon, D., Langenegger, T., Donis, D. and McGinnis, D. F. (2019) 'Influence of water column stratification and mixing patterns on the fate of methane produced in deep sediments of a small eutrophic lake', *Limnology and Oceanography,* 64(5), pp. 2114-2128.

Visser, M. E. and Both, C. (2005) 'Shifts in phenology due to global climate change: the need for a yardstick', *Proc Biol Sci,* 272(1581), pp. 2561-9.

Visser, P. M., Ibelings, B. W., Bormans, M. and Huisman, J. (2015) 'Artificial mixing to control cyanobacterial blooms: a review', *Aquatic Ecology,* 50(3), pp. 423-441.

Walsby, A. E., Hayes, P. K., Boje, R. and Stal, L. J. (1997) 'The selective advantage of buoyancy provided by gas vesicles for planktonic cyanobacteria in the Baltic Sea', *New phytologist,* 136(3), pp. 407-417.

Wang, W., Lee, X. H., Xiao, W., Liu, S. D., Schultz, N., Wang, Y. W., Zhang, M. and Zhao, L. (2018) 'Global lake evaporation accelerated by changes in surface energy allocation in a warmer climate', *Nature Geoscience,* 11(6), pp. 410-+.

Watson, S. B., Monis, P., Baker, P. and Giglio, S. (2016) 'Biochemistry and genetics of taste- and odor-producing cyanobacteria', *Harmful Algae,* 54, pp. 112-127.

Watz, J., Bergman, E., Calles, O., Enefalk, A., Gustafsson, S., Hagelin, A., Nilsson, P. A., Norrgard, J. R., Nyqvist, D., Osterling, E. M., Piccolo, J. J., Schneider, L. D., Greenberg, L. and

Jonsson, B. (2015) 'Ice cover alters the behavior and stress level of brown trout Salmo trutta', *Behavioral Ecology,* 26(3), pp. 820-827.

Weirich, C. A., Robertson, D. M. and Miller, T. R. (2019) 'Physical, biogeochemical, and meteorological factors responsible for interannual changes in cyanobacterial community composition and biovolume over two decades in a eutrophic lake', *Hydrobiologia,* 828(1), pp. 165-182.

Welch, E. B. and Cooke, G. D. (2005) 'Internal phosphorus loading in shallow lakes: Importance and control', *Lake and Reservoir Management,* 21(2), pp. 209-217.

Westrell, T., Bergstedt, O., Stenstrom, T. A. and Ashbolt, N. J. (2003) 'A theoretical approach to assess microbial risks due to failures in drinking water systems', *Int J Environ Health Res,* 13(2), pp. 181-97.

Wetzel, R. G. (2001) *Limnology: lake and river ecosystems.* 3rd edn. San Diego: Academic Press.

Whitehead, P. G., Wilby, R. L., Battarbee, R. W., Kernan, M. and Wade, A. J. (2009) 'A review of the potential impacts of climate change on surface water quality', *Hydrological Sciences Journal-Journal Des Sciences Hydrologiques,* 54(1), pp. 101-123.

Winder, M. and Sommer, U. (2012) 'Phytoplankton response to a changing climate', *Hydrobiologia,* 698(1), pp. 5-16.

Wood, S. L. R., Jones, S. K., Johnson, J. A., Brauman, K. A., Chaplin-Kramer, R., Fremier, A., Girvetz, E., Gordon, L. J., Kappel, C. V., Mandle, L., Mulligan, M., O'Farrell, P., Smith, W. K., Willemen, L., Zhang, W. and DeClerck, F. A. (2018) 'Distilling the role of ecosystem services in the Sustainable Development Goals', *Ecosystem Services,* 29, pp. 70-82.

Woolway, R. I., Jennings, E., Shatwell, T., Golub, M., Pierson, D. C. and Maberly, S. C. (2021a) 'Lake heatwaves under climate change', *Nature,* 589(7842), pp. 402-407.

Woolway, R. I., Jones, I. D., Feuchtmayr, H. and Maberly, S. C. (2015a) 'A comparison of the diel variability in epilimnetic temperature for five lakes in the English Lake District', *Inland Waters,* 5(2), pp. 139-154.

Woolway, R. I., Jones, I. D., Hamilton, D. P., Maberly, S. C., Muraoka, K., Read, J. S., Smyth, R. L. and Winslow, L. A. (2015b) 'Automated calculation of surface energy fluxes with high-frequency lake buoy data', *Environmental Modelling & Software,* 70, pp. 191-198.

Woolway, R. I., Jones, I. D., Maberly, S. C., French, J. R., Livingstone, D. M., Monteith, D. T., Simpson, G. L., Thackeray, S. J., Andersen, M. R., Battarbee, R. W., DeGasperi, C. L., Evans, C. D., de Eyto, E., Feuchtmayr, H., Hamilton, D. P., Kernan, M., Krokowski, J., Rimmer, A., Rose, K. C., Rusak, J. A., Ryves, D. B., Scott, D. R., Shilland, E. M., Smyth, R. L., Staehr, P. A., Thomas, R., Waldron, S. and Weyhenmeyer, G. A. (2016) 'Diel Surface Temperature Range Scales with Lake Size', *Plos One,* 11(3), pp. e0152466.

Woolway, R. I., Kraemer, B. M., Lenters, J. D., Merchant, C. J., O'Reilly, C. M. and Sharma, S. (2020) 'Global lake responses to climate change', *Nature Reviews Earth & Environment,* 1(8), pp. 388-403.

Woolway, R. I., Meinson, P., Noges, P., Jones, I. D. and Laas, A. (2017a) 'Atmospheric stilling leads to prolonged thermal stratification in a large shallow polymictic lake', *Climatic Change,* 141(4), pp. 759-773.

Woolway, R. I., Meinson, P., Nõges, P., Jones, I. D. and Laas, A. (2017b) 'Atmospheric stilling leads to prolonged thermal stratification in a large shallow polymictic lake', *Climatic Change,* 141(4), pp. 759-773.

Woolway, R. I. and Merchant, C. J. (2019) 'Worldwide alteration of lake mixing regimes in response to climate change', *Nature Geoscience,* 12(4), pp. 271-276.

Woolway, R. I., Sharma, S., Weyhenmeyer, G. A., Debolskiy, A., Golub, M., Mercado-Bettín, D., Perroud, M., Stepanenko, V., Tan, Z., Grant, L., Ladwig, R., Mesman, J., Moore, T. N., Shatwell, T., Vanderkelen, I., Austin, J. A., DeGasperi, C. L., Dokulil, M., La Fuente, S., Mackay, E. B., Schladow, S. G., Watanabe, S., Marcé, R., Pierson, D. C., Thiery, W. and Jennings, E. (2021b) 'Phenological shifts in lake stratification under climate change', *Nature Communications,* 12(1), pp. 2318.

Woolway, R. I., Verburg, P., Lenters, J. D., Merchant, C. J., Hamilton, D. P., Brookes, J., de Eyto, E., Kelly, S., Healey, N. C., Hook, S., Laas, A., Pierson, D., Rusak, J. A., Kuha, J., Karjalainen, J., Kallio, K., Lepisto, A. and Jones, I. D. (2018) 'Geographic and temporal variations in turbulent heat loss from lakes: A global analysis across 45 lakes', *Limnology and Oceanography,* 63(6), pp. 2436-2449.

Woolway, R. I., Verburg, P., Merchant, C. J., Lenters, J. D., Hamilton, D. P., Brookes, J., Kelly, S., Hook, S., Laas, A., Pierson, D., Rimmer, A., Rusak, J. A. and Jones, I. D. (2017c) 'Latitude and lake size are important predictors of over-lake atmospheric stability', *Geophysical Research Letters,* 44(17), pp. 8875-8883.

Woolway, R. I., Weyhenmeyer, G. A., Schmid, M., Dokulil, M. T., de Eyto, E., Maberly, S. C., May, L. and Merchant, C. J. (2019) 'Substantial increase in minimum lake surface temperatures under climate change', *Climatic Change,* 155(1), pp. 81-94.

World Bank Group, ESMAP and SERIS (2018) *Where Sun Meets Water : Floating Solar Market Report*, Washington, D.C. Available at: http://documents.worldbank.org/curated/en/579941540407455831/Where-Sun-Meets-Water-Floating-Solar-Market-Report-Executive-Summary.

World Bank Group, ESMAP and SERIS (2019) *Where Sun Meets Water: Floating Solar Handbook for Practitioners*, Washington, DC.

Wright, R. T. (1964) 'Dynamics of a Phytoplankton Community in an Ice-Covered Lake', *Limnology and Oceanography,* 9(2), pp. 163-178.

Wüest, A. and Lorke, A. (2003) 'Small-Scalehydrodynamics Inlakes', *Annual Review of Fluid Mechanics,* 35(1), pp. 373-412.

WWT (2016) *Project Focus: Floating solar panels for Thames Reservoir*. Available at: https://wwtonline.co.uk/features/project-focus-floating-solar-panels-for-thames-reservoir (Accessed: 27/07/20 2020).

Yadav, N., Gupta, M. and Sudhakar, K. (2016) *Energy Assessment of Floating Photovoltaic System. 2016 International Conference on Electrical Power and Energy Systems*.

Yamamichi, M., Kazama, T., Tokita, K., Katano, I., Doi, H., Yoshida, T., Hairston, N. G., Jr. and Urabe, J. (2018) 'A shady phytoplankton paradox: when phytoplankton increases under low light', *Proc Biol Sci,* 285(1882), pp. 20181067.

Yankova, Y., Neuenschwander, S., Koster, O. and Posch, T. (2017) 'Abrupt stop of deep water turnover with lake warming: Drastic consequences for algal primary producers', *Sci Rep,* 7(1), pp. 13770.

Young, P. C. (2015) 'Refined instrumental variable estimation: maximum likelihood optimization of a unified Box–Jenkins model', *Automatica,* 52, pp. 35-46.

Young, W. F., Horth, H., Crane, R., Ogden, T. and Arnott, M. (1996) 'Taste and odour threshold concentrations of potential potable water contaminants', *Water Research,* 30(2), pp. 331-340.

Zhan, Q., Kong, X. Z. and Rinke, K. (2021) 'High-frequency monitoring enables operational opportunities to reduce the dissolved organic carbon (DOC) load in Germany's largest drinking water reservoir', *Inland Waters*, pp. 1-16.

Zhang, N., Jiang, T., Guo, C., Qiao, L. F., Ji, Q., Yin, L. Q., Yu, L. M., Murto, P. and Xu, X. F. (2020) 'High-performance semitransparent polymer solar cells floating on water: Rational analysis of power generation, water evaporation and algal growth', *Nano Energy,* 77, pp. 105111.

Zhang, W. T. and Rao, Y. R. (2012) 'Application of a eutrophication model for assessing water quality in Lake Winnipeg', *Journal of Great Lakes Research,* 38, pp. 158-173.

Zhou, Y. L., Chang, F. J., Chang, L. C., Lee, W. D., Huang, A., Xu, C. Y. and Guo, S. L. (2020) 'An advanced complementary scheme of floating photovoltaic and hydropower generation flourishing water-food-energy nexus synergies', *Applied Energy,* 275, pp. 115389.

Ziar, H., Prudon, B., Lin, F. Y., Roeffen, B., Heijkoop, D., Stark, T., Teurlincx, S., Domis, L. D., Goma, E. G., Extebarria, J. G., Alavez, I. N., van Tilborg, D., van Laar, H., Santbergen, R. and

Isabella, O. (2021) 'Innovative floating bifacial photovoltaic solutions for inland water areas', *Progress in Photovoltaics,* 29(7), pp. 725-743.

Zubair, M., Bilal Awan, A., Ghuffar, S., Butt, A. D. and Farhan, M. (2020) 'Analysis and Selection Criteria of Lakes and Dams of Pakistan for Floating Photovoltaic Capabilities', *Journal of Solar Energy Engineering,* 142(3).