IMPACTS OF DAMS AND THEIR REMOVAL ON CARBON EMISSIONS AND STORAGE

by

LAURA CATHERINE NASLUND

(Under the Direction of Amy Rosemond & Seth Wenger)

ABSTRACT

Dams and their reservoirs play a central role in freshwater carbon cycling as control points for carbon burial and emissions in inland waters. While dams are intensively managed to provide infrastructure services and mitigate their environmental harms, considering carbon impacts in dam management decisions is currently constrained by limited methods for quantifying and incorporating carbon costs and benefits in decision processes. I addressed three major aims to reduce this constraint: 1) evaluate spatiotemporal variability of emissions from small reservoirs to support accurate emissions estimates from these abundant, yet understudied waterbodies, 2) estimate the impacts of dam removals on carbon emissions and storage, and 3) develop a structured decision tool to facilitate the consideration of carbon impacts with other objectives of dam removal. To address the first aim, I intensively sampled carbon emissions from four small reservoirs in Athens, GA over 24-hr periods in the late summer. I found that common practices for sampling small reservoir carbon emissions which fail to account for spatiotemporal variability and measure ebullition can lead to misestimation of total emissions between -89 to 366%. To address the second aim, I combined literature values, statistical, and mechanistic models to estimate carbon fluxes before, during, and after dam removal to determine its net impact on carbon balance in the reservoir footprint. I found that the removal of two large dams decreased the sink strength of the reservoir footprints, but more work is needed to distinguish changes in flux magnitude from changes in flux timing or location due to dam removal. By conducting a systematic review of existing dam removal decision-support tools for aim three, I found that these tools frequently omit common objectives of dam removal. To facilitate structured decision-making inclusive of diverse objectives, I designed a web application to guide users to relevant objectives, metrics, methods, data, and tools for their removal decisions. Dams sit at the intersection of the interacting crises of global biodiversity loss, infrastructure deterioration, and climate change. Dam management decisions can align efforts to address these crises given tools to appropriately estimate and weigh the consequences of those decisions.

INDEX WORDS: methane, carbon dioxide, carbon storage, infrastructure, structured decision-making, reservoirs, dam management, dam decommissioning

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DEDICATION

To my teachers, especially Karen Carroll and Marshall Johnson.

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

The streams, rivers, wetlands, lakes, and reservoirs which constitute inland waters were once considered so inconsequential to the global carbon (C) cycle as to be excluded from its depiction in seminal papers and global syntheses (see Cole et al. 2007). As limnological research moved from imagining inland waters as isolated microcosms to studying their integration with the land, ocean, and atmosphere, the significant role of inland waters in global carbon cycling became clearer (Tranvik et al. 2018). Globally, inland waters receive 1.7 to 2.7 Pg C yr^{-1} from terrestrial inputs (Ciais et al. 2013), with individual waterbodies receiving C inputs approximately equivalent to net ecosystem production in their watersheds (Aufdenkampe et al. 2011). Inland waters do not simply transport this C load from land to ocean (Cole et al. 2007). They off-gas a substantial amount of C (0.8 to 1.2 Pg C yr^{-1}) to the atmosphere (Ciais et al. 2013), acting as both chimneys for CO_2 produced in terrestrial soils and reactors producing CO_2 from the mineralization of terrestrially-derived organic carbon (Hotchkiss et al. 2015). Inland waters also retain 0.2 to 0.6 Pg C yr^{-1} by burying C in their sediments, and transport their remaining carbon loads to coastal oceans (Ciais et al. 2013). While the pools of carbon in inland waters are small, these fluxes are of a similar magnitude as other major global C fluxes including the annual terrestrial C sink and ocean carbon dioxide (CO_2) uptake, which occur over much larger areas, making inland waters important transformers and transporters of C (Cole et al. 2007; Ciais et al. 2013; Tranvik et al. 2018).

Better quantification of the role of inland waters in the global carbon cycle has also revealed the impacts of human activities. Humans have created bottlenecks in connected hydrological systems by constructing dams, which substantially alter the transport and transformation of carbon. As a result, the reservoirs which dams impound play an outsized role in C cycling in inland waters (Maavara et al. 2020). Long residence times and low sediment oxygen conditions in reservoirs promote both high rates of carbon burial and emissions of the greenhouse gas methane (CH_4) (Sobek et al. 2012; Maavara et al. 2020). Reservoirs account for 40% of organic carbon burial in lentic ecosystems, despite accounting for only 8.6% of their cumulative area (Mendonça et al. 2017), and CH_4 emissions from reservoirs are 16% of total inland water CH_4 emissions (Rosentreter et al. 2021). Reducing CH_4 emissions has been identified as a key climate mitigation strategy because CH_4 has a higher global warming potential and a lower atmospheric residence time than CO_2 (Canadell et al. 2021). Reservoirs can also enhance CO_2 emissions by flooding soil organic matter, enhancing its mineralization (Prairie et al. 2018). Both CO_2 and CH_4 can be emitted by diffusion across the air-water interface, and CH_4 can also be emitted via ebullition, a pathway in which CH_4 bubbles form in the sediment and rise through the water column to the reservoir surface (Deemer et al. 2016). Carbon burial in reservoir sediments can be construed as an offset to their emissions; however, the portion of that burial which would have occurred regardless of the dam's existence is better thought of as "early burial" rather than additional burial (Mendonça et al. 2012; Prairie et al. 2018). Understanding the changes in carbon fluxes which can be attributed to the dam and its reservoir is critical for considering carbon in management decisions (Prairie et al. 2018).

As key elements of infrastructure systems, reservoirs are often subject to intense management to deliver services such as hydropower generation and to eliminate or mitigate their negative impacts such as degrading fisheries (Opperman et al. 2011), implying that they could also be managed to reduce emissions (Jager et al. 2023). Initial attempts to integrate carbon into dam management decisions have focused on dams which generate hydropower, because measurements of reservoir emissions have indicated that some hydropower dams may not be low-carbon energy sources. Trade-offs between energy generation and carbon emissions have been estimated, for example, to optimize the locations of new hydropower dam construction (Almeida et al. 2019). However, most existing dams do not generate hydropower. In the U.S., less than 2% of dams in the national inventory produce energy (Gonzales and Walls 2020), and the majority of U.S. dams are too small to be included in the inventory. Incorporating carbon considerations into dam management decisions thus requires contending with the diversity of dam sizes, locations, and constructed purposes. To support this goal, I addressed three issues. In Ch 2, I characterized the fine-scale spatiotemporal variability of emissions from small (< 0.01 km^2) reservoirs to facilitate accurate emissions estimates from an abundant but understudied size-class of reservoirs. In Ch 3, I modeled the impacts of dam removal on carbon emissions and storage, and in Ch 4, I developed a tool to integrate carbon management with other diverse objectives of dam removal decisions.

Chapter 2: Toward more accurate estimates of carbon emissions from small reservoirs

Most efforts to characterize reservoir carbon emissions have focused on the reservoirs > 0.01 km^2 in area. Emissions from reservoirs < 0.01 km^2, which vastly outnumber larger reservoirs, are poorly characterized (Ollivier et al. 2019). Estimates of emissions from these small reservoirs typically rely on measurements which do not capture potential variation in the timing, location, and pathways of emissions (Wang et al. 2017; Ollivier et al. 2019; Webb et al. 2019), which have been shown to impact emissions estimates from larger reservoirs (Beaulieu et al. 2016; Sieczko et al. 2020; Hounshell et al. 2023). In this study, I characterized the variation in CO_2 diffusion, CH_4 diffusion, and CH_4 ebullition over the reservoir surface and 24-hour periods in four small reservoirs in Athens, GA, USA. I then used these data to evaluate different sampling schemes for their ability to characterize daily emissions efficiently and accurately. Managing emissions from these abundant, anthropogenic ecosystems relies on accurate emissions estimates. I support this aim by characterizing the sampling effort required to generate these estimates.

Chapter 3: Consequences of dam removal for carbon storage and emissions

Dam removal has been proposed as a management action to decrease carbon emissions from reservoirs. By reducing residence times and increasing sediment oxygen conditions, dam removal is hypothesized to decrease emissions from the water surface (U.S. EPA 2016; Johnson 2017). However, dam removal can facilitate other pathways of carbon emissions like the mineralization of carbon in dewatered sediments, and removal can alter pathways of carbon storage (Amani et al. 2022; Liang et al. 2024). Ultimately, whether dam removal reduces carbon emissions and enhances carbon storage in the former reservoir depends on the relative magnitude of the carbon fluxes before, during, and after dam removal, which is unknown. In this chapter, I outline a framework for estimating the net impact of dam removal on carbon emissions and storage and apply it to two case studies of removed dams with contrasting features. This framework can be applied to other dam removals to support the consideration of carbon impacts in removal decisions.

Chapter 4: Facilitating dam removal decisions with multiple objectives

Dam removal decisions frequently involve weighing multiple disparate services and disservices of dams (Habel et al. 2020). Removal decisions can involve one or multiple dams which range in size, spatial relationship, ownership, and legal authorities (USSD 2015; McKay et al. 2020). To facilitate structured approaches to these complex decisions, several organizations have developed decision-support tools. However, these tools may guide users to suboptimal decisions if they do not include all relevant objectives. Early elimination of important objectives is a common cause of suboptimal outcomes of structured decision processes (Gregory and Keeney 2002). I reviewed existing dam removal decision-support tools (n = 41) to determine which objectives of dam removal they included. To facilitate decisions inclusive of the diverse objectives I encountered in this search, I developed a web application that links objectives of dam removal to relevant metrics, methods, data sources, and decision-support tools, which can be used to determine the achievement of stated objectives by different alternative actions. This application supports the consideration of carbon impacts in dam removal decisions by facilitating the integration of this objective into decision processes with multiple objectives.

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CHAPTER 2

TOWARD MORE ACCURATE ESTIMATES OF CARBON EMISSIONS FROM SMALL RESERVOIRS[1]

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Abstract

Because of their abundance and high emissions rates, small reservoirs (< 0.01 km^2) can be important emitters of the greenhouse gases carbon dioxide and methane. However, emissions estimates from small reservoirs have lagged those of larger ones, and efforts to characterize small reservoir emissions have largely overlooked variation in emissions pathways, times, and locations. We intensively sampled four small reservoirs in Georgia, U.S.A. during the summer to quantify the contribution and spatiotemporal variability of different emissions pathways (CO_2 and CH_4 diffusion, CH_4 ebullition). We used these data to evaluate the efficiency and accuracy of different sampling schemes. Every emissions pathway was dominant in one reservoir on one sampling day, and excluding ebullition caused misestimation between -89% to -15% of the total flux. Sampling only once daily caused misestimation between -78% to 45%, but sampling twice or just after dawn (7:00) reduced error. Sampling four or fewer locations caused misestimation between -85% to 366%, and our results indicated that six to twenty sampling locations may be needed for reasonable accuracy. The floating aquatic macrophyte *Wolffia sp.* (duckweed) appeared to exert control over emissions variability, and the consequences of not accounting for variability were greater in a duckweed-covered reservoir. Our results indicate that sampling only at 10:00 (modal sampling time of prior efforts) may lead to the erroneous conclusion that reservoirs with high photosynthetic biomass are CO_2 sinks rather than sources. Improving estimation accuracy by accounting for within-reservoir variation in emissions will facilitate more strategic management of these abundant, anthropogenic ecosystems.

Introduction

Of the millions of reservoirs that exist globally, most have surface areas < 0.01 km^2 (Downing 2010). As a consequence of their numeric abundance and positions in river networks as recipients of substantial terrestrially-derived carbon inputs (Harvey and Schmadel 2021), these reservoirs have the capacity to cumulatively emit carbon dioxide (CO_2) and methane (CH_4) in comparable magnitudes to larger reservoirs (Grinham et al. 2018; Ollivier et al. 2019a), which have been the primary focus of research for the past two decades. Variation in the dominant pathways, times, and locations of emissions in natural lakes and larger reservoirs has been shown to impact inferences about the total magnitude of their emissions (Beaulieu et al. 2016; Sieczko et al. 2020; Hounshell et al. 2023); however, this variation has been poorly characterized in small reservoirs, and prior efforts to estimate their emissions have frequently relied on measurements of one pathway of emissions, at one time during the day, and in few locations in the reservoir (Wang et al. 2017; Ollivier et al. 2019a; Webb et al. 2019). Unaccounted-for variation in emissions may lead to misestimation of the contribution of small reservoirs to landscape greenhouse gas (GHG) emissions, which may be reduced or mitigated through management actions.

Reservoirs can yield high CO_2 and CH_4 emissions by concentrating substantial volumes of organic matter from both terrestrial sources and in situ primary production, leading to high rates of decomposition and respiration, depletion of dissolved oxygen (DO), and facilitation of anaerobic metabolic pathways including methanogenesis (Friedl and Wüest 2002). The resulting CO_2 and CH_4 can be emitted from reservoirs by the diffusion of gas from high to low concentration across the water-air interface (Deemer et al. 2016). Methane can also be emitted by bubbles which rise through the water column from the sediment in a process known as ebullition.

These bubbles form when the partial pressure of the accumulated gas surpasses the pressure on the sediments and overcomes water surface tension (Harrison et al. 2017). Ebullition can contribute substantially to radiative forcing from reservoirs because much of the CH_4 transported in bubbles can escape oxidation in the water column (Bastviken et al. 2008), and one molecule of CH_4 has the same warming potential as 27 molecules of CO_2 (Forster et al. 2021). A global synthesis of GHG emissions from mostly larger reservoirs found that ebullition contributed on average 65% of total CH_4 flux (Deemer et al. 2016), yet ebullition has not been included in many past efforts to characterize emissions from small reservoirs (Wang et al. 2017; Ollivier et al. 2019a; Peacock et al. 2019; Webb et al. 2019). If ebullition is a dominant pathway of CO_2-equivalent (CO_2-eq) emissions from small reservoirs, sampling efforts that fail to include it will underestimate total emissions.

Ebullition can exhibit spatial variation associated with depth and distance from the reservoir inlet, which may also impact total emissions estimates (Natchimuthu et al. 2016; Linkhorst et al. 2021). At shallower depths, CH_4 bubbles can grow and escape the sediment more easily because there is lower hydrostatic pressure to overcome (Boudreau 2012). There is also diminished opportunity for CH_4 to be oxidized during the transport of bubbles through the shorter water column to the water surface (Bastviken et al. 2008). At the reservoir inlet, ebullition can be higher due to greater accumulation of watershed-derived carbon as particles in transport settle (Natchimuthu et al. 2016). This greater organic carbon availability can result in higher CH_4 production near the reservoir inlet (Maeck et al. 2013). The influence of depth and inlet distance on rates of CH_4 ebullition can create longitudinal patterns of declining ebullition from the inlet to the dam, which has been observed in larger reservoirs (Beaulieu et al. 2016; McClure et al. 2020; Linkhorst et al. 2021). Patterns of CH_4 and CO_2 diffusion have also been observed in larger reservoirs arising from spatial heterogeneity in gas production and in rates of gas transfer (Paranaíba et al. 2018). If small reservoirs exhibit substantial spatial heterogeneity in emissions, failing to account for this variability may lead to misestimation of emissions.

Carbon dioxide and CH_4 diffusion, like many biogeochemical processes, may exhibit daily temporal patterns due to the overriding influence of the sun on the light, wind, pressure, and temperature conditions that can influence the magnitude of emissions (Nimick et al. 2011; Natchimuthu et al. 2014; Sieczko et al. 2020). If small reservoirs exhibit diel patterns of diffusion, these patterns may be important to incorporate into the estimation of daily emissions from discrete measurements in time. Diffusion is a product of both the supply and transfer of gases to the atmosphere, and both factors have drivers which can exhibit diel patterns, potentially yielding diel fluctuations in emissions (Cole and Caraco 1998). First, CO_2 supply can be altered by photosynthetic activity. When light is available for photosynthesis, autotrophs can take up CO_2, reducing its supply, yielding lower daytime CO_2 emissions (Natchimuthu et al. 2014; Gómez-Gener et al. 2021). Autotrophic activity may also reduce daytime diffusive CH_4 emissions by increasing surface DO concentrations, promoting higher rates of CH_4 oxidation and lower CH_4 supply (Ford et al. 2002).

When transfer rather than supply processes dominate, however, the pattern of lower daytime CO_2 and CH_4 diffusion may be reversed. Wind is a major driver of surface turbulence and gas exchange in reservoirs (Crusius and Wanninkhof 2003). Windspeeds are generally higher during the day, which can force higher rates of gas exchange and thus higher daytime diffusive emissions (Sieczko et al. 2020; Hounshell et al. 2023). Transfer processes can also impact patterns of CH_4 ebullition, with drops in hydrostatic pressure yielding high bubbling rates; however, few studies have captured sub-daily measurements of CH_4 emissions to characterize the periodicity of bubbling events (but see Grinham et al. 2011; Varadharajan and Hemond 2012; Sieczko et al. 2020). If either supply or transfer processes generate diel contrasts in CO_2 and CH_4 emissions, failing to account for these temporal patterns—for example by only sampling during daytime hours—may also lead to misestimation of total emissions.

To support accurate estimation of small reservoir greenhouse gas emissions, we quantified 1) the contribution of different pathways of emissions, 2) temporal variation in emissions, and 3) spatial variation in emissions from four small reservoirs during the summer in the southeastern U.S. Specifically, we measured CH_4 and CO_2 diffusion at 12 sampling stations and CH_4 ebullition at 25 sampling stations every three hours over at least one 24 h cycle in each of the small reservoirs. We used data from this intensive sampling to simulate the consequences of different sampling schemes to identify those that efficiently yielded accurate estimates of total daily emissions.

Materials and Methods

We sampled four small reservoirs within a 3 km^2 area in Athens, GA, USA in August-September 2022 (Figure A1). August and September are among the hottest months of the year, and studies in similar climatic zones have found that they contribute large proportions of total annual emissions (van Bergen et al. 2019; McClure et al. 2020). The reservoirs ranged in area from 0.0012 – 0.0077 km^2 and in mean depth from 0.80 – 2.03 m (Table 2.1). We estimated their water residence time to be 19 to 46 days based on discharge estimates from a regional regression equation with average annual precipitation and watershed area (Gotvald 2017). At the time of sampling, a floating macrophyte in the genus *Wolffia*, an angiosperm in the family Lemnaceae hereon referred to as duckweed, extensively covered the surface of one reservoir (Blue Herron). The other three sites had minimal or no macrophyte coverage and no visual evidence of a phytoplankton bloom (Figure A1). Although not an initial objective of our study, we took this opportunity to evaluate the potential role of duckweed on the magnitude and spatiotemporal patterns of emissions in addition to the objectives outlined above. Every three hours, we measured CO_2 and CH_4 diffusion at 12 locations and CH_4 ebullition at 25 locations for one to two 24-h cycles in each reservoir to evaluate patterns in the pathways, times, and locations of emissions (Table 2.1). Because we observed a high CH_4 concentration at the surface of Blue Herron near the intake for the reservoir outlet, we additionally calculated the flux of CH_4 from degassing as water is discharged over the reservoir outlet, which can be a major pathway of CO_2-eq emissions in larger reservoirs (Kemenes et al. 2007; Maeck et al. 2013).

Measuring emissions pathways: We measured ebullition in each reservoir every three hours over 24-hr periods at 25 sampling stations (Figure A2a). The sampling stations were positioned at different distances from the reservoir perimeter and divided among five transects, which we installed using rope fixed diagonally across the width of the reservoir to capture variation in sampling station depth and distance to the inlet. At each sampling station, we installed an ebullition trap by affixing it to the rope with a zip tie. This method of trap installation permitted us to sample ebullition without disturbing the sediment. The ebullition traps consisted of inverted funnels with 18 cm diameters fastened to 60 mL polypropylene syringes with silicone sealant. At the start of the sampling, we filled the traps with water by evacuating air through a three-way stopcock, fastened polyethylene foam to the syringe for buoyancy, and weighted the funnel so that only the syringe was above the water. As bubbles rose from the sediment, they displaced the water and collected at the top of the syringe where we emptied the gas through the stopcock.

Every three hours we recorded the volume of gas accumulated in the ebullition trap syringe. If the volume exceeded 18 mL, we emptied the trap and injected the contents into a pre-evacuated 12-mL glass vial with a chlorobutyl and PTFE/silicon septum. At the end of the 24-h period, we emptied all traps, combining gas from traps as necessary to obtain 18 mL. After all other measurements had been taken, we physically disturbed the sediment in three locations along the edges of the reservoir to collect fresh bubbles. We compared concentrations of the fresh bubbles to those which had been left in the ebullition traps during the sampling period to evaluate whether oxidation of CH_4 bubbles in the traps impacted the concentrations of the collected gas. On return to the lab, we analyzed the CH_4 concentrations of the collected gas using an SRI Instruments 8610C Gas Chromatograph (Torrance, CA, USA) equipped with a methanizer and Flame Ionization Detector (GC-FID). We did not find a significant difference between the concentrations of gas collected in the traps and fresh bubbles, so we used concentrations from both collection methods to estimate total ebullition.

During each sampling period, we also measured the diffusive flux of CO_2 and CH_4 at 12 of the sampling stations using a portable greenhouse gas analyzer attached to a floating acrylic chamber (Figure A2b,c). We measured diffusion directly adjacent to the ebullition traps at 2-3 of the sampling stations in each transect. The analyzer measured the headspace gas concentration in the chamber every 1 s with a minimum contact time of 60 s (Off-Axis Integrated Cavity Output Spectroscopy Analyzer, GLA131 series, ABB, San Jose, CA, USA). We calculated diffusive flux using the change in headspace gas concentration over the contact period using the following equation in which s is the rate of change in headspace gas concentration over time (ppm s-1), V is the combined volume of the chamber and tubing internal and external to the analyzer (m3), SA is the surface area of the chamber (m^2), P is pressure (atm), T is temperature (K), and R is the universal gas constant (m^3 atm K^{-1} moL^{-1}).

$$\text{flux} = \frac{s}{R * T * \frac{1}{P}} * \frac{V}{SA}$$

Ebullition and changing light conditions during the diffusive flux measurements resulted in portions of the headspace concentration time series that were non-monotonic and non-linear. We therefore estimated the most frequently occurring slope in the time series and used this value for our flux calculations. Briefly, we calculated the slope of every 10 sequential points in the time series and computed the maximum of the probability density function calculated using kernel density estimation. We checked these slopes visually to ensure they appropriately captured rates of diffusion.

To evaluate the daily contribution of the degassing pathway of emissions from our small reservoirs, we calculated the flux by taking the difference in CH_4 concentration of water collected near the dam intake (0.25 m below the water surface) and water collected from the dam outlet. We multiplied this difference by the dam discharge, which we estimated by measuring the time required to fill a known volume (Maeck et al. 2013). We divided this value by reservoir area to compare the contribution of degassing to those of other emissions pathways.

To measure dissolved gas concentrations to calculate degassing emissions, we followed the headspace equilibration protocol in Goodman (2019), but modified the purge gas to use ultra-zero air rather than ambient air. We also measured the dissolved gas concentrations in the inlet stream and at the bottom of the water column, 0.1 m from the sediment (Table A1). Briefly, we took three replicate, 60 mL bubble-free water samples using a Van Dorn bottle 0.25 m below the surface and 0.1 m from the sediment as well as from the inlet and outlet streams to estimate dissolved gas concentrations. On return to the lab, we introduced a 20 mL headspace of ultra-zero air and equilibrated the headspace by vigorously shaking the syringe for five minutes. We then injected 18 mL of the headspace into an evacuated 12 mL vial for gas analysis. At the start of the sample run, we confirmed that both CO_2 and CH_4 concentrations in the purge gas were below detection limit. We measured gas concentrations on an SRI Instruments 8610C Gas Chromatograph (Torrance, CA, USA) as described above and calculated the original dissolved gas concentration from the measured headspace gas concentrations.

Scaling point estimates to the reservoir surface: To estimate diffusion and ebullition from the entire surface of the reservoirs, we used sequential gaussian simulation with simple kriging of our measured fluxes to generate equally probable realizations of emissions in unsampled areas from which to characterize the global uncertainty in our scaled emissions estimates (Delbari et al. 2009). For every sampling period in every reservoir, we generated 500 simulations of emissions across a 1 m^2 grid. We discarded the top and bottom 2.5% of realizations to estimate a 95% confidence interval. To generate the realizations, we used the krige function in the gstat package (Pebesma 2004; Gräler et al. 2016). Because we did not have a CH_4 concentration corresponding to every measurement of gas volume from ebullition, we randomly assigned every volume measurement to a concentration measurement from the sampling event and simulated 25 realizations of ebullition using those concentrations. We repeated this procedure 20 times for a total of 500 simulations of ebullition per sampling event. To calculate rates of emissions from our interpolated estimates, we summed the values in each grid cell at a time point, multiplied that value by the time elapsed until the next sampling interval, summed across the sampling intervals, and then divided by reservoir area.

Drivers of diel patterns of emissions: To characterize diel patterns in CO_2 diffusion, CH_4 diffusion, and CH_4 ebullition, we plotted and visually inspected the fluxes interpolated with sequential gaussian simulation across the sampling intervals. We then ran models explaining the measured point estimates of diffusive fluxes as a function of environmental variables we expected to relate to diel variation in diffusion including DO at the top and bottom of the water column, windspeed, light, and temperature (Natchimuthu et al. 2016; Sieczko et al. 2020; Rudberg et al. 2021; Hounshell et al. 2023). We recorded DO concentration 0.25 m below the water surface and 0.1 m from the sediment using a handheld meter (YSI Pro Plus, Yellow Springs, OH, USA) at two locations near the edge of the reservoir and two in the center during every sampling period. We took two 15 second integrated wind speed measurements using a handheld anemometer (HoldPeak 866, Guangdong, China) at approximately 1 m in height adjacent to the chamber during every diffusive flux measurement. We quantified light availability in two ways: (1) we measured illuminance every 15 min using a light logger installed on the bank of the reservoir where there was no tree cover (UA-002-64 Onset, Bourne, MA, USA) and (2) we measured photosynthetically active radiation 0.5 m below the reservoir surface (Odyssey, Christchurch, New Zealand). We recorded temperature approximately every 0.5 m of depth using a chain of temperature loggers installed at the deepest location in the reservoir (Onset, Bourn, MA, USA). At this location, we also recorded DO every 15 minutes at 0.25 m below the water surface using a miniDOT (PME, Vista, CA, USA) to develop a continuous record of DO during our study period. We ran all subsets of a global linear mixed effects model predicting the diffusive flux of CO_2 or CH_4 using these variables (windspeed, surface water temperature, illuminance, and DO at the top and bottom of the water column averaged across the four sampling locations). In each of these models, we scaled the predictors and used a random intercept for sampling event, which was the site and date sampled. For the top models, we report unscaled parameter estimates (Table A2). We were unable to recover data from the PAR logger installed in Deans, so we ran separate models evaluating the explanatory power of PAR on CO_2 and CH_4 diffusion using data from the other three sites.

Effects of sampling locations: To evaluate the spatial patterns in CH_4 ebullition, we ran a mixed effects gamma regression model with a log link function to predict ebullition from (1) depth and (2) distance to inlet, with a random intercept for sampling event. Because we did not identify a temporal pattern in CH_4 ebullition, we aggregated ebullition across all sampling intervals in the 24-h sampling period for this analysis.

Simulating the consequences of sampling scheme: We conducted simulations using our measured flux data to evaluate the consequences of sampling emissions with less spatial and temporal intensity. We simulated the consequences of sampling one to 11 locations for diffusion and one to 24 locations for ebullition, while sampling the maximum number of times in the 24-h period. For every possible number of sampling locations, we generated 100 bootstrapped replicates for each pathway and calculated the number of replicates which fell within the 95% confidence interval of our interpolated emissions. We used 80% of simulations falling within the 95% confidence interval as an arbitrary threshold of estimation accuracy, and calculated the number of sampling locations required to reach this threshold for each pathway of emissions (Robison et al. 2021). To calculate the misestimation associated with sampling fewer sites, we simulated every combination of one to four sampling locations where we measured both ebullition and diffusion. We took the estimation error to be the difference between the flux estimates generated from simulations using these locations and estimates generated using all of the sampled locations. To evaluate the consequences of sampling with less temporal intensity, we simulated sampling fluxes at every combination of one to two times during the day at the maximum number of sampling locations. To calculate the estimation error, we took the difference between these estimates and the fluxes estimated using all of the sampled times. To find the combinations of times which minimized estimation error across sites, we calculated the cumulative estimation error as the sum of the errors from individual sites. Because Blue Herron had substantially different diel patterns from the other three sites, we determined the times which minimized cumulative estimation error separately for Blue Herron.

We conducted all analyses in R version 4.2.1 and produced figures using ggplot2 version 3.4.2 (Wickham 2016; R Core Team 2022). We ran the mixed models using the lmer and glmer functions in the lme4 package version 1.1-32 (Bates et al. 2015).

Results

Contributions of emissions pathways: Total interpolated emissions from the four reservoirs across the six sampling events ranged from 2.10 to 17.8 g CO_2-eq m^{-2} d^{-1} (Table A3). Each pathway of emissions (CO_2 diffusion, CH_4 diffusion, or CH_4 ebullition) contributed the most CO_2-eq in at least one reservoir on one sampling day (Figure 2.1). Carbon dioxide diffusion contributed the most CO_2-eq emissions from Deans and the 18-19 Sept Catfish sampling (50 to 77% of total CO_2-eq). During the 6-7 Sept Catfish sampling, CO_2 diffusion and CH_4 ebullition contributed approximately equally to total CO_2-eq emissions (46 and 49%, respectively). Methane ebullition contributed most from Sister (89% of total CO_2-eq) and CH_4 diffusion contributed most from Blue Herron (45% of total CO_2-eq), which was the only reservoir in which rates of CH_4 diffusion exceeded CH_4 ebullition (Figure 2.1). The high rate of CH_4 diffusion in Blue Herron (the reservoir covered in duckweed) was consistent with its high surface CH_4 concentration of 98.7 μmol L^{-1}. The surface CH_4 concentrations for the other three sites were two orders of magnitude lower, ranging from 0.51 to 0.98 μmol L^{-1} (Table A1). Despite the high concentration of dissolved CH_4 at Blue Herron, degassing emissions at this site contributed only 0.0003 g CH_4 m^{-2} d^{-1} or 0.008 g CO_2-eq m^{-2} d^{-1}, less than 0.05% of total CO_2-eq emissions, due to the low discharge from the reservoir on the day of sampling (1.6 x 10-5 m^3 s^{-1}). The concentration of CH_4 in the outlet exceeded the concentration near the dam intake on all but one other sampling date: Deans on the 16-17 Aug. Degassing emissions from this site were an order of magnitude lower than those from Blue Herron at 0.00003 g CH_4 m^{-2} d^{-1} or 0.0007 g CO_2-eq m^{-2} d^{-1}, which was less than 0.001% of total CO_2-eq emissions from Deans on 16-17 Aug.

The high surface CH_4 concentration at Blue Herron is consistent with its persistent anoxia. During the sampling period, surface DO at the continuous sampling station near the center of the pond never exceeded 0.2 mg L^{-1} (Figure A3, A4). In contrast, the minimum surface DO concentration at the other three sites was 4.9 mg L^{-1} in Deans, 7.2 mg L^{-1} in Sister, and briefly 0.4 mg L^{-1} in Catfish before rising to an average of 5.5 mg L^{-1} (Figure A3). Across all reservoirs, CO_2 diffusion ranged on average from -0.07 to 3.62 g CO_2 m^{-2} d^{-1}, CH_4 diffusion ranged from 0.01 to 0.29 g CH_4 m^{-2} d^{-1}, and CH_4 ebullition from 0.02 to 0.23 g CH_4 m^{-2} d^{-1} (Table A3).

Diel patterns and drivers: We found no evidence of a diel pattern in CH_4 diffusion at Catfish, Deans, or Sister. At Blue Herron (the reservoir with duckweed), there was a distinct day-night pattern in CH_4 diffusion (g m^{-2} h^{-1}), with the highest rates of diffusion at midday and declining rates in the late afternoon and night (Figure 2.2). None of the models explaining CH_4 diffusion using environmental variables were more parsimonious than the intercept-only model. The next most parsimonious model (ΔAICc = 8.54) included DO at the top of the water column, which was negatively associated with CH_4 diffusion (Figure A5, $\beta = -0.001 \pm 0.0003$, $p = 0.003$, $t = -3.21$, n = 573, $R^2 = 0.17$).

Blue Herron also demonstrated a diel pattern of CO_2 diffusion distinct from those of the other reservoirs and from its diel pattern of CH_4 diffusion. The reservoir was a net sink for CO_2 in the late morning from 10:00 to 13:00, after which it switched to being a net source (Fig. 2A). The rate of emissions increased through the night until it declined in the early morning hours. In contrast, in the other sites, there was only a slight dip in emissions at night (Figure 2.2, A6). Because of the contrasting diel patterns of CH_4 and CO_2 diffusion, Blue Herron did not exhibit a diel pattern in total CO_2-eq emissions. In the other reservoirs, the pattern of lower CO_2 diffusion in the late afternoon and night did not result in a clear diel pattern in CO_2-eq emissions because of the lack of diel patterns in CH_4 diffusion and ebullition; however, all reservoirs had variable CO_2-eq emissions throughout the day (Figure 2.2).

The most parsimonious model explaining CO_2 diffusion included only DO at the top of the water column, which was negatively associated with CO_2 diffusion (Figure A7, $\beta = -0.028 \pm 0.008$, $p = 0.008$, $t = -3.50$, n = 573 , $R^2 = 0.14$) (Table A2). The next most parsimonious models were the intercept-only model (ΔAICc = 3.16) and the model including water temperature at the top of the water column (ΔAICc = 5.00). Temperature was negatively associated with CO_2 diffusion (Figure A8, $\beta = -0.024 \pm 0.01$, $p = 0.033$, n = 573, $t = -2.49$, $R^2 = 0.12$), and this relationship strengthened when Blue Herron was excluded ($\beta = -0.036 \pm 0.006$, $p < 0.001$, n = 477, $t = -6.06$, $R^2 = 0.37$) (Table A2). Blue Herron had the highest CO_2 concentration (560 μmol L^{-1}) of the reservoirs, four times greater than the next highest concentration, which occurred in Catfish (138 μmol L^{-1}), and fourteen times greater than the lowest concentration, which occurred in Sister (40.2 μmol L^{-1}) (Table A1).

Spatial variation: As predicted, both depth and distance to the inlet had negative associations with ebullition; however, only depth had a parameter estimate that did not overlap zero (inlet distance: -0.001 ± 0.003, n = 142, t = -0.41, p = 0.68; depth: -0.30 ± 0.12, t= -2.4, p = 0.02) and even combined these predictors explained little variation in ebullition (R^2 = 0.03). However, when Blue Herron was excluded, both predictors had greater negative associations with ebullition (inlet distance: -0.019 ± 0.0034, n = 118, t = -5.48, p < 0.001; depth: -0.49 ± 0.11, n =118, t = -4.35, p < 0.001) and the model explained more of the total variation in ebullition (R^2 = 0.44). These parameter estimates correspond to a 39% decline in ebullition with every 1 m increase in depth and a 2% decline with every 1 m increase in distance from the reservoir inlet. Plots of CO_2 diffusion indicated a spatial pattern in Blue Herron, where every diffusive flux measurement was taken over duckweed (Figure A2b). To confirm this spatial pattern, we calculated Moran's I to test for spatial autocorrelation for every sampling period and found that during 6 of 8 time periods, measurements of CO_2 diffusion were spatially autocorrelated, indicating that locations closer together in the reservoir had more similar fluxes than those far apart (Moran's I: 0.15 to 0.43) (Figure A9). None of the other sites exhibited spatial patterns in CO_2 diffusion or had significant periods of spatial autocorrelation (Figure A2b, A9).

Sampling scheme efficiency and accuracy: All sites except Blue Herron reached the arbitrary accuracy threshold (80% of simulations falling within the 95% confidence interval of the estimate calculated using all of the data) when six locations were sampled for CO_2 diffusion and eight for CH_4 diffusion. Catfish and Deans reached the accuracy threshold for CH_4 ebullition at nine sampling locations, Sister at 17, and Blue Herron at 20 locations (Figure 2.3). The estimation errors for sampling one to four locations in the reservoir, which is the difference between the fluxes calculated using a limited number of sampling locations and fluxes calculated using all of the locations, ranged from -10.43 to 13.47 g CO_2-eq m^{-2} d^{-1} and -85% to 366% of the total CO_2-eq flux.

Estimation errors for sampling only once during the day ranged from -5.61 to 2.56 g CO_2-eq m^{-2} d^{-1} or -78% to 45% of the total flux, but sampling at certain times or combinations of time substantially reduced estimation error. Sampling at 10:00 and 22:00 minimized the cumulative estimation error for CO_2 diffusion for the duckweed-free reservoirs: Catfish, Deans, and Sister. Other combinations of daytime and nighttime sampling also had low cumulative estimation error (Figure 2.4). The median estimation error from these individual reservoirs was 0.31 g CO_2 m^{-2} d^{-1}, and the maximum estimation error was -1.05 g CO_2 m^{-2} d^{-1}, 33% of the total diffusive CO_2 flux. Although there was no distinct diel pattern of CH_4 diffusion in these reservoirs (Figure A6), there was variation in rates of emissions throughout the day. Diel variation in these reservoirs ranged from 0.001 g CH_4 m^{-2} h^{-1} to 0.008 g CH_4 m^{-2} h^{-1}. As a result, median estimation error from these individual reservoirs was 0.002 g CH_4 m^{-2} d^{-1} and maximum estimation error was 0.01 g CH_4 m^{-2} d^{-1}, 87% of the total diffusive CH_4 flux. Cumulative estimation error for CH_4 diffusion was minimized by sampling at 13:00 and 22:00 (Figure 2.4).

For the duckweed-covered reservoir, Blue Herron, estimation error was minimized by sampling at 13:00 and 04:00 (0.77 g CO_2 m^{-2} d^{-1}, 21% of the total diffusive CO_2 flux); however, sampling just once at 07:00 was among the combinations of times with the lowest estimation error (0.89 g CO_2 m^{-2} d^{-1}, 24% of the total diffusive CO_2 flux). In contrast, sampling at 10:00, which is the modal sampling hour for several greenhouse gas sampling efforts (Gómez-Gener et al. 2021; NEON 2023), had the highest estimation error for CO_2 diffusion (-7.60 g CO_2 m^{-2} d^{-1}, 210% of the total diffusive CO_2 flux) (Figure 2.4). For CH_4 diffusion, sampling at 10:00 and 22:00 minimized estimation error (-0.0005 g CH_4 m^{-2} d^{-1}, 0.2% of the total diffusive CH_4 flux); however, like with CO_2 diffusion, sampling once at 07:00 yielded low estimation error for CH_4 diffusion (0.004 g CH_4 m^{-2} d^{-1}, 1.3% of the total diffusive CH_4 flux) (Figure 2.4).

Discussion

Our results indicate that common practices of sampling small reservoir GHG emissions (i.e., not measuring ebullition, sampling at one time during the day, and sampling few locations in the reservoir) can lead to substantial misestimations of total CO_2-eq flux from these ecosystems. Excluding ebullition from our estimates led to underestimation of the total flux by -0.63 to -6.19 g CO_2-eq m^{-2} d^{-1} or -15 to -89% of the total flux. Sampling only once during the day led to misestimation from -5.61 to 2.56 g CO_2-eq m^{-2} d^{-1} or -78% to 45% of the total flux. Sampling few locations in the reservoir (four or fewer) led to misestimation between -10.43 to 13.47 g CO_2-eq m^{-2} d^{-1} or -366% to 85% of the total flux. Our results indicated that six to twenty sampling locations may be required for reasonable estimation accuracy, depending on the characteristics of a reservoir and the emissions pathway considered. We observed a distinct magnitude, spatial, and diel pattern of CH_4 and CO_2 emissions in the duckweed-covered reservoir compared to the other reservoirs, and the consequences of limited sampling were more severe in the duckweed reservoir. Our results can inform efforts to scale point measurements of emissions in space and in time in smaller waterbodies (<0.01 km^2), for which eddy covariance and other high-temporal-resolution methods used for large reservoirs have limited effectiveness (Zhao et al. 2019).

Magnitude of emissions

Summer emissions from the four small reservoirs we sampled were within the range of values observed previously for reservoirs. Mean CO_2 emissions were higher (2.25 g CO_2 m^{-2} d^{-1}) and mean CH_4 emissions (0.145 g CH_4 m^{-2} d^{-1}) were slightly lower than the mean emissions from a global synthesis of reservoir emissions (1.21 g CO_2 m^{-2} d^{-1} and 0.161 g CH_4 m^{-2} d^{-1}); however, both gases were firmly within the range of previous observations (-1.30 to 9.66 g CO_2

m^{-2} d^{-1} and 0 to 5.26 g CH_4 m^{-2} d^{-1}) (Deemer et al. 2016). Notably, only two of the 144 reservoirs with diffusive CH_4 fluxes measurements in Deemer et al. (2016) had a higher diffusive CH_4 flux than the duckweed reservoir (0.29 g CH_4 m^{-2} d^{-1}), and only 8 of 54 had a higher ebullitive flux of CH_4 than the duckweed reservoir (0.23 g CH_4 m^{-2} d^{-1}). However, we only measured emissions during the summer, and prior evidence of seasonal patterns in emissions suggests that annual diffusive and ebullitive fluxes of CH_4 may be lower than the values reported here (van Bergen et al. 2019). Comparing our findings to waterbodies in the same size class that were not formed by dams, we found both a higher mean CO_2 flux (2.25 g CO_2 m^{-2} d^{-1} vs. 0.933 g CO_2 m^{-2} d^{-1}) and CH_4 flux (0.145 g CH_4 m^{-2} d^{-1} vs. 0.010 g CH_4 m^{-2} d^{-1}) from our small reservoirs (Holgerson and Raymond 2016).

A limitation of our ability to ascribe the patterns in gas fluxes we observed to the presence of duckweed is that duckweed was only abundant in one of our reservoirs. However, the patterns we observed were consistent with the overriding control of emissions by floating macrophytes which has been observed in previous studies (Bastviken et al. 2010; Rabaey and Cotner 2022). The rate of CO_2-eq emissions (g m^{-2} d^{-1}) in the duckweed reservoir was on average 5.5 times greater than the other three reservoirs, and methane emissions from this site accounted for 80% of total CO_2-eq emissions. Duckweed likely elevated CH_4 emissions by limiting O_2 exchange across the air-water interface and loading labile organic carbon to the sediments, generating persistent anoxia and fueling methanogenesis (Morris and Barker 1976; Kosten et al. 2016; Rabaey and Cotner 2022).

Although small reservoirs have the capacity to emit large quantities of GHGs, they also have the capacity to bury large quantities of carbon. One study found that organic carbon burial rates in eutrophic, midwestern US reservoirs were three orders of magnitude higher than mean burial rates in temperate forests (Downing et al. 2008). These burial rates far exceeded rates of CO_2 diffusion from these reservoirs (Pacheco et al. 2014). In contrast, in a temperate, eutrophic pond, carbon burial was only 7.4% of the annual emissions (van Bergen et al. 2019). The carbon balance of our small reservoirs is unknown. While duckweed is very labile, its breakdown in the sediments of the duckweed reservoir may be slow. Our duckweed-dominated reservoir was anoxic for much of the year (unpublished data), potentially resulting in slow breakdown of duckweed biomass and subsequent carbon storage. Additional work on carbon burial is needed to understand the role of small reservoirs in landscape carbon balance (Holgerson et al. 2023).

Diel patterns

The duckweed-free reservoirs demonstrated slight declines in CO_2 diffusion at night. In contrast to studies in larger reservoirs, this pattern did not appear to be associated with higher daytime wind speeds; however, we observed a relatively small range of wind speeds (0 to 7.2 m s^{-1}, mean: 0.4 m s^{-1}) and it is possible that a positive relationship between wind speed and CO_2 diffusion would be apparent with windier conditions (Crusius and Wanninkhof 2003). Wind may also be a less important driver of CO_2 diffusion in these reservoirs because of their small area and insulation by trees (Vachon and Prairie 2013). In the duckweed reservoir, the net uptake of CO_2 from 10:00-13:00 is consistent with a mid-morning peak in CO_2 fixation, which was previously observed in a small lake covered in a different duckweed species (*Lemna minor*) (Filbin and Hough 1985). Although solar radiation may be more intense later in the day, an afternoon depression in rates of CO_2 fixation may occur due to photoinhibition and photorespiration (Filbin and Hough 1985), consistent with the increase in CO_2 emissions we observed in the afternoon in the duckweed reservoir. In the duckweed-free reservoirs, the dominant primary producers were submerged below the water surface. Attenuation of light through the water column may have reduced the strength of photoinhibition, resulting in a later peak in autotrophic CO_2 uptake and minimum in CO_2 emissions (Walsby 1997). Although we did not observe a relationship between CO_2 emissions and PAR measured at a fixed station in the reservoir, we did observe a negative relationship between CO_2 emissions and surface DO, which is consistent with autotrophs driving diel patterns of CO_2 emissions, as in the mechanism proposed above. Our simulations indicated that sampling once during the day and once at night in the duckweed-free reservoirs resulted in low estimation errors, with the smallest errors resulting from sampling at 10:00 and 22:00. Estimation errors in the duckweed reservoir were much larger but were minimized by sampling at 13:00 and 04:00 or at 07:00 alone. If we had sampled at 10:00 only, which is the modal sampling time of prior efforts (Gómez-Gener et al. 2021; NEON 2023), we could have concluded that the duckweed reservoir was a sink for CO_2 when it was a source, emphasizing the importance of accounting for diel patterns of emissions.

The duckweed reservoir was the only site that exhibited a diel pattern of CH_4 diffusion. It had high rates of diffusion in the late morning and early afternoon (10:00-13:00), followed by a sharp decline in the late afternoon and persistently low rates through the night (16:00 – 01:00), and rising again in the early morning (04:00 – 07:00). In contrast to findings in other lentic waterbodies, diel variations in wind speed did not explain the higher daytime diffusive CH_4 fluxes we observed (Liu et al. 2017; Sieczko et al. 2020). In this reservoir, the peak in light coincided with the peak in surface DO, indicating a possible contribution by duckweed to surface DO. Methane oxidation during these periods of elevated DO may have decreased CH_4 diffusion from the reservoir surface (Kosten et al. 2016). Consistent with this mechanism, the strongest association between CH_4 diffusion and an environmental variable across all reservoirs was the negative association with surface DO; however, the model including surface DO explained little variation (R^2 = 0.17), indicating that other mechanisms may be generating diel patterns in CH_4 diffusion, alone or in conjunction with CH_4 oxidation. Estimation errors for CH_4 diffusion in the duckweed reservoir were minimized with sampling at 10:00 and 22:00 or at 07:00 alone.

The amplitude of diel CO_2 emissions in the duckweed reservoir (0.55 g CO_2 m^{-2} h^{-1}) was at the high end of the range observed in previous studies of sub-daily emissions from lentic inland waters. A similar amplitude (0.56 g CO_2 m^{-2} h^{-1}) was observed in ponds in the subarctic wetland region of the Hudson Bay Lowlands, Canada, where CO_2 emissions were attributed to degrading peat (Hamilton et al. 1994). The duckweed reservoir amplitude was 2.6 times higher than that observed in a larger eutrophic reservoir (0.119 km^2) (Hounshell et al. 2023). We observed the lowest amplitude in Sister, which was an order of magnitude lower (0.042 g CO_2 m^{-2} h^{-1}) than the duckweed reservoir. The diel variation in CO_2 emissions from Sister was similar to the minimum amplitude reported in a synthesis of CO_2 emissions from 13 northern latitude lakes and reservoirs, which ranged widely in size, nutrient, and humic states (Golub et al. 2021). For CH_4 diffusion, the duckweed reservoir was the only reservoir that exhibited a diel pattern. Like CO_2 diffusion, the amplitude of diel CH_4 diffusion from the duckweed reservoir (0.011 g CH_4 m^{-2} h^{-1}) was at the high end of the range of previous observations. It was equal to the amplitude of CH_4 emissions measured over dense patches of water hyacinth (*Pontederia sp.*) in shallow lakes in the Pantanal, Brazil (Bastviken et al. 2010). The high amplitude of variation for CO_2 and CH_4 diffusion in the duckweed reservoir highlights the special consideration that sub-daily, temporal variation may merit in characterizing emissions from smaller, floating macrophyte-covered reservoirs. The diel patterns we observed are consistent with substantial autotrophic control of emissions, and were consequential for estimating total emissions from the duckweed reservoir. However, we only sampled emissions during the summer, and lower primary producer biomass and temperature in the winter may result in lower diel fluctuations in diffusion (Ollivier et al. 2019b). Sub-daily variation, therefore, may be less important for emissions estimates at other times of the year. Further investigation of seasonal changes in diel patterns of emissions from small reservoirs merits further study.

Spatial patterns

In the duckweed-free reservoirs, we observed a decline in daily rates of CH_4 ebullition with increasing distance from the inlet and depth, consistent with patterns observed in larger reservoirs (Beaulieu et al. 2016; McClure et al. 2020). Terrestrial organic matter delivered by the inlet streams may have dominated organic matter inputs, creating a pattern of higher sediment organic matter availability near the reservoir inlet. In these reservoirs, CH_4 ebullition declined by 2% for every 1 m increase in distance from the reservoir inlet and 39% with every 1 m increase in depth. However, in the duckweed reservoir, the extensive macrophyte coverage may have more evenly distributed organic matter across the reservoir sediments and persistent water column anoxia may have reduced rates of CH_4 oxidation, eliminating the decline in ebullition with increasing inlet distance and depth that we observed in the duckweed-free reservoirs. In general, more sampling locations were required to accurately estimate fluxes in the duckweed reservoir than in the duckweed-free reservoirs. A high degree of estimation accuracy was reached in the duckweed-free reservoirs when at least 6 locations were sampled for CO_2 diffusion, 8 locations for CH_4 diffusion, and 17 locations were sampled for ebullition. In the duckweed reservoir, 12 or more sampling locations were required to estimate CO_2 diffusion, 8 locations for CH_4 diffusion, and 20 locations for CH_4 ebullition.

Extrapolation to other small reservoirs

To interpret the implications of our findings, it is important to consider whether the differences we observed between the duckweed-dominated and duckweed-free reservoirs can be extrapolated to small reservoirs with high biomass of other dominant primary producers. Elevated diel variation in CO_2 emissions has been linked to photosynthetic uptake by large stands of submerged and emergent macrophytes, as well as blooms of phytoplankton (Maberly 1996; Kragh et al. 2017; Golub et al. 2021). However, the exact diel patterns of emissions may differ by dominant primary producer. For example, primary producers differ in their susceptibility to photoinhibition (Wetzel 2001), which could impact diel patterns of photosynthesis, CO_2 supply, and flux. Because photosynthesis can control CO_2 fluxes, we may also expect greater spatial patterns of CO_2 emissions in small reservoirs with high primary producer biomass, as observed in the duckweed reservoir. However, primary production may not be a major driver of CO_2 emissions if a reservoir receives large external inputs of CO_2 (e.g., in groundwater), even when primary producer biomass is high; in this case, diel and spatial patterns of CO_2 diffusion may not be apparent (van Bergen et al. 2019).

Diel patterns of CH_4 diffusion can arise in reservoirs dominated by primary producers other than duckweed, although the patterns and mechanisms may differ (Hamilton et al. 1994; Bastviken et al. 2010). We expect that other floating macrophytes are likely to behave similarly to duckweed in enhancing CH_4 oxidation, potentially generating a pattern of lower CH_4 emissions when surface DO is high due to photosynthesis (Kosten et al. 2016; Iguchi et al. 2019). In reservoirs dominated by rooted macrophytes, vegetation-mediated emissions could modify these temporal patterns by transporting CH_4 from the water column or sediment pore water to the atmosphere (Whiting and Chanton 1996; Bolpagni et al. 2007). The impact of high phytoplankton biomass on diel patterns of CH_4 fluxes is uncertain; previous studies have identified both the presence and absence of a diel pattern of CH_4 fluxes in lentic ecosystems with high phytoplankton biomass (van Bergen et al. 2019; Waldo et al. 2021). Primary producer identity may be an important factor in determining diel patterns of CH_4 diffusion, and floating macrophyte-dominated small reservoirs may exhibit distinct patterns compared to those dominated by other types of macrophytes and algae.

Conclusion

Several recent inventories of greenhouse gas emissions from inland waters have specifically identified the need for emissions estimates from abundant small reservoirs (Deemer and Holgerson 2021; Pilla et al. 2022; Lauerwald et al. 2023). By intensively sampling small reservoirs in space and time, we identified key considerations for efficiently estimating emissions from point measurements. Our results indicate that 07:00, just after dawn, may be an efficient time to sample while sampling later in the morning or early afternoon may result in greater estimation errors due to diel patterns of CO_2 and CH_4 diffusion. Sampling six to twenty locations in the reservoir may be required for reasonable estimation accuracy. Our results suggest that more sampling locations may be required to characterize emissions from duckweed-covered reservoirs than duckweed-free reservoirs. Selecting sampling locations for ebullition with varying depth and distance to the reservoir inlet may also improve estimation accuracy, as our results indicate that ebullition from all sites declined with increasing depth and declined with increasing distance to the inlet in all sites except for the duckweed-covered reservoir. These results can inform efforts to characterize emissions from small reservoirs and include them in regional and global inventories of GHG emissions from inland waters.

Accurate emissions estimates can also facilitate the identification of drivers of high GHG emissions, which can reveal management strategies to reduce emissions from these systems (Malerba et al. 2022; Nijman et al. 2022). For example, our results highlight management methods to reduce duckweed coverage as potential strategies to reduce GHG emissions from small reservoirs. While the impact of duckweed management on small reservoir GHG emissions has not, to our knowledge, been evaluated, duckweed removal from small ponds and reservoirs is a common practice with the potential to enhance DO content and reduce organic matter loading (Lembi 2009), decreasing rates of methanogenesis. Duckweed harvest from high nutrient ponds has even been recommended as a method to generate feed and fertilizer while reducing stream nitrogen loads from farms (Femeena et al. 2022), highlighting a potential synergy between nutrient and GHG management. With over 2 million small reservoirs in the continental United States alone, opportunities to reduce greenhouse gas emissions from small reservoirs are abundant.

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Table 2.1. Reservoir sampling dates, physical, and chemical characteristics. All sites were sampled in 2022. Temperature, dissolved oxygen, and pH values reported here are spot measurements taken at four locations in each reservoir during every diffusive sampling flux period. We also recorded dissolved oxygen using a continuous sensor at the top of the water column in the deepest location in the reservoir (Figure A4).

Site Name	Area (km^2)	Max depth (m)	Mean depth (m)	Residence time (days)	Dates sampled	Temperature (°C)	Dissolved Oxygen (mg L^{-1})	pH
Sister	0.0012	2.27	1.25	20	22-23 Aug	Top: 27.4 ± 0.6	Top: 7.0 ± 0.5	Top: 8.0 ± 0.4
						Bottom: 26.7 ± 0.4	Bottom: 5.5 ± 1.4	Bottom: 7.6 ± 0.4
Catfish	0.0018	1.98	0.80	19	06-07 Sept	Top: 26.0 ± 0.6	Top: 4.6 ± 0.4	Top: 7.2 ± 0.1
						Bottom: 25.5 ± 0.3	Bottom: 4.0 ± 1.2	Bottom: 7.2 ± 0.1
					18-19 Sept	Top: 22.0 ± 0.6	Top: 5.4 ± 0.6	Top: 7.2 ± 0.1
						Bottom: 21.6 ± 0.3	Bottom: 4.8 ± 1.2	Bottom: 7.2 ± 0.1
Deans	0.0042	3.52	2.03	46	16-17 Aug	Top: 28.6 ± 0.6	Top: 6.2 ± 0.5	Not available
						Bottom: 28.0 ± 0.4	Bottom: 4.4 ± 2.1	
					30-31 Aug	Top: 28.6 ± 0.9	Top: 6.0 ± 0.6	Top: 7.5 ± 0.1

						Bottom: 27.6 ± 0.6	Bottom: 3.4 ± 2.5	Bottom: 7.2 ± 0.3
Blue Herron	0.0077	3.8	1.58	45	13-14 Sept (Duckweed coverage: 100%)	Top: 24.4 ± 1.1 Bottom: 23.2 ± 1.1	Top: 0.7 ± 0.8 Bottom: 0.3 ± 0.4	Top: 6.9 ± 0.3 Bottom: 6.8 ± 0.2

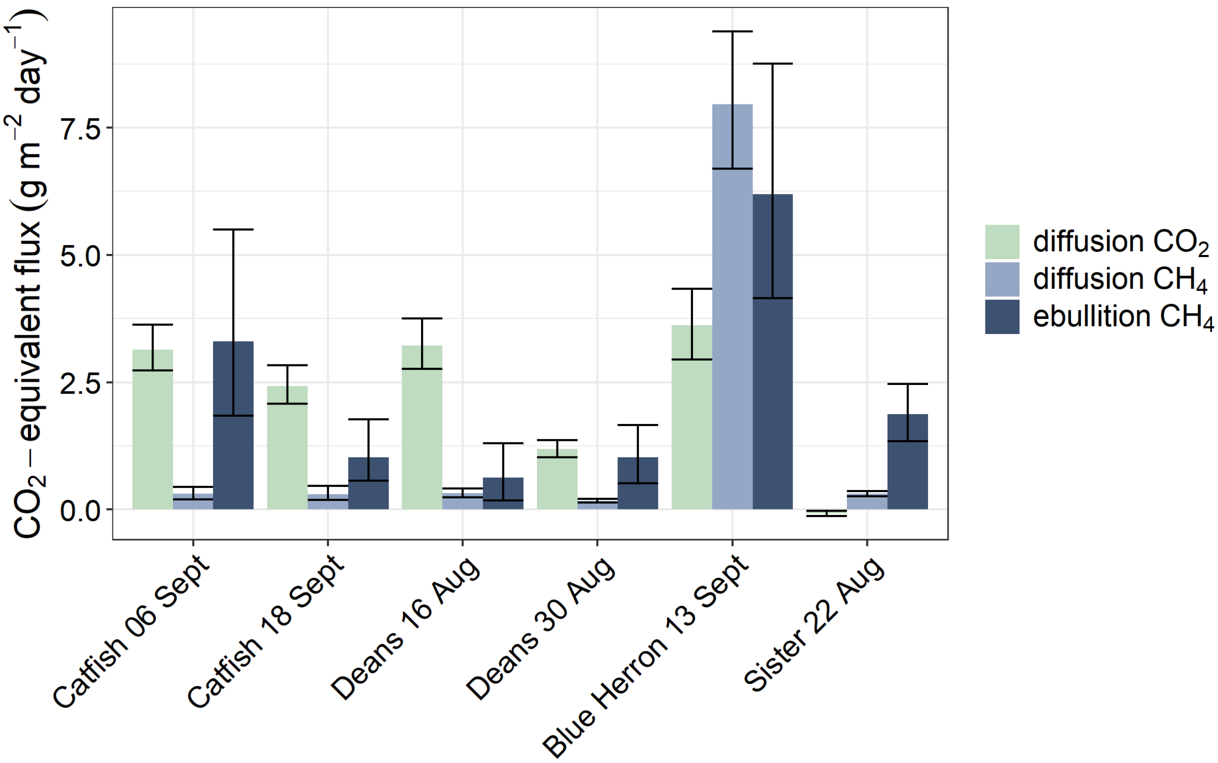

Figure 2.1. CO_2-eq flux rates by emission pathway from estimates interpolated using sequential gaussian simulation. Bars represent average values and error bars represent the simulated 95% confidence interval. The date listed is the start date of the 24-hr sampling period.

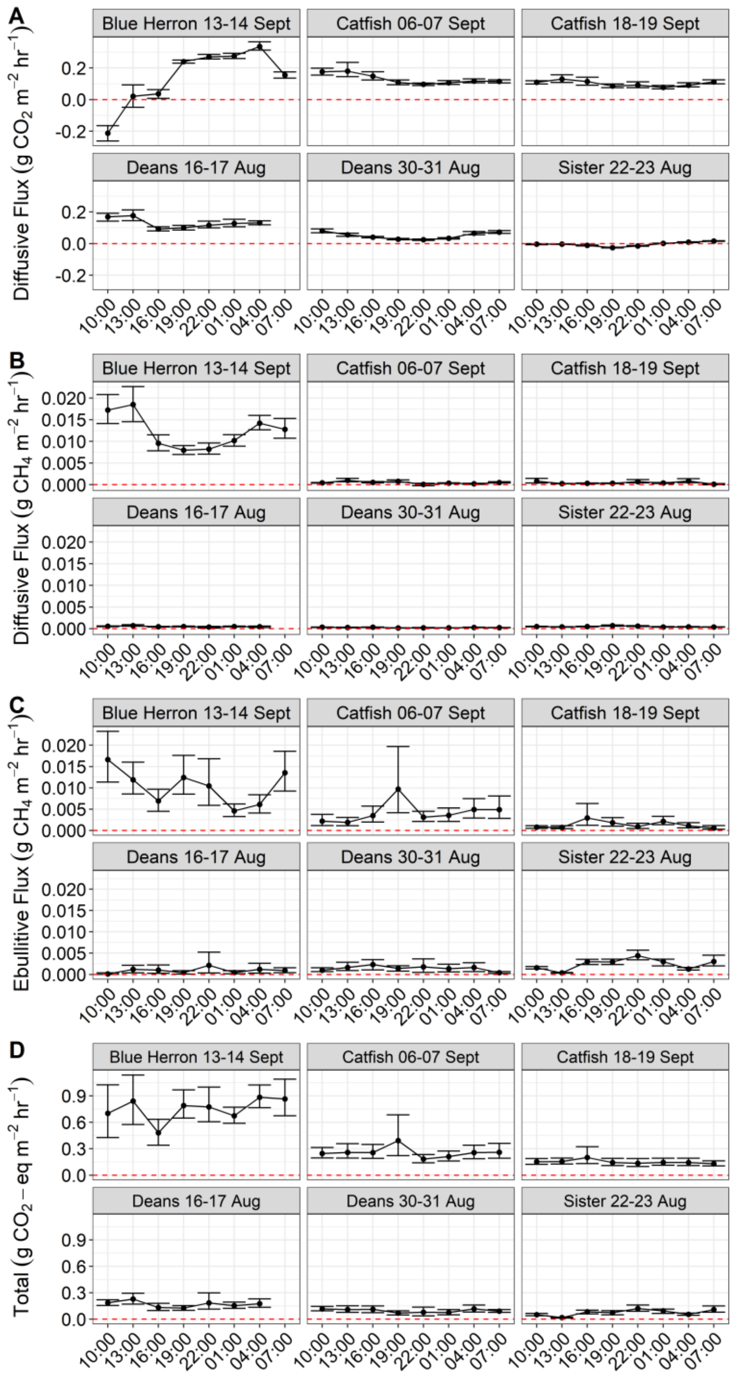

Figure 2.2. Diel patterns in interpolated A) CO_2 diffusion, B) CH_4 diffusion, C) CH_4 ebullition, and D) total CO_2-eq emissions. Points represent average estimates and error bars represent the simulated 95% confidence interval. The times depicted for CH_4 ebullition are the end of the ebullition measurement period (i.e., when the gas volume was recorded). The 07:00 diffusive

flux sampling for Deans on 16 Aug was delayed until 09:00 due to rain. These values were used to calculate total CO_2-eq flux over a 24-h period, but are not depicted here. See Figure A6 for values depicted on free y-axis scales.

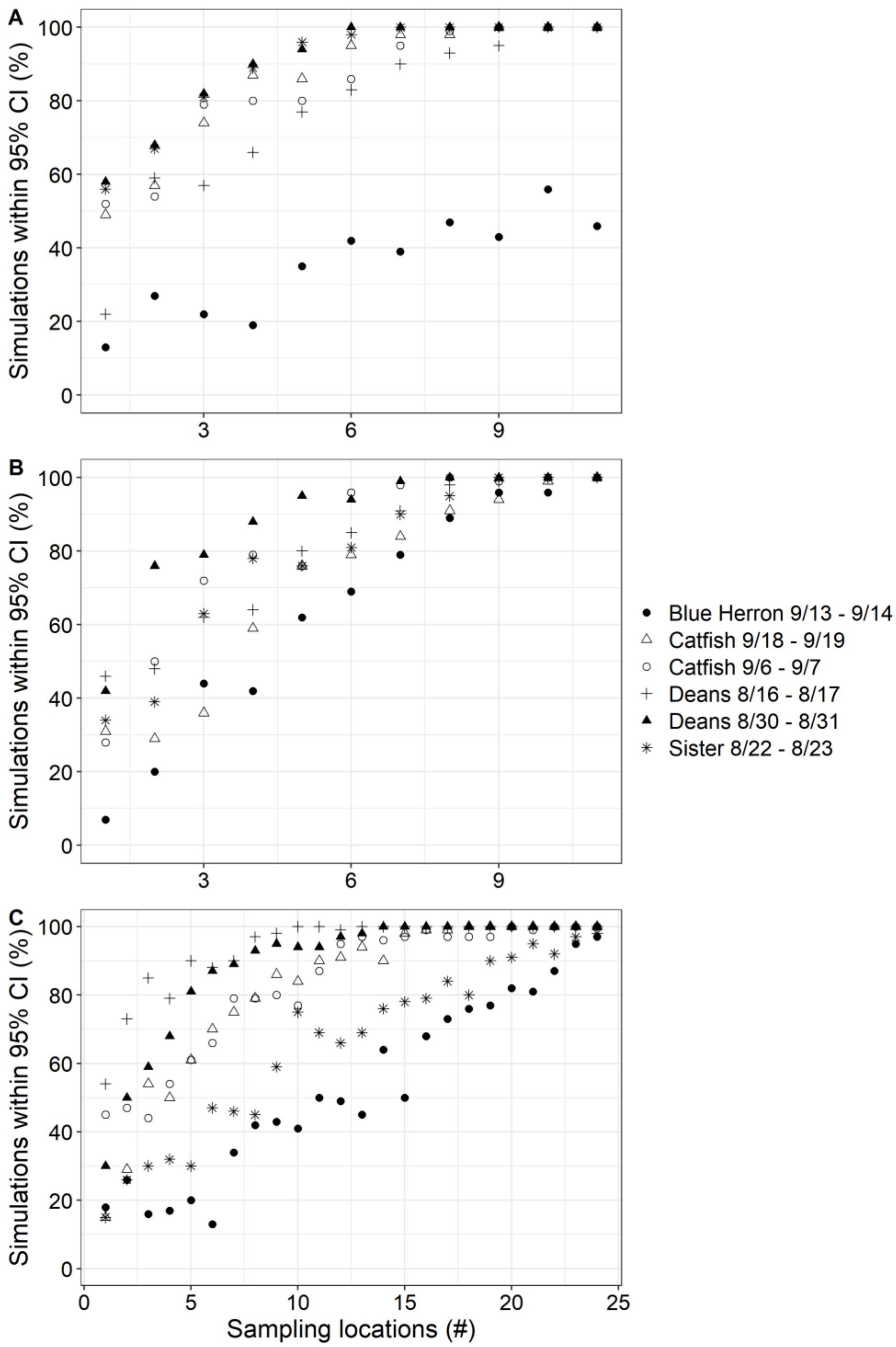

Figure 2.3. The percentage of simulated samplings within the 95% confidence interval of the interpolated fluxes at iterative numbers of random sampling locations for A) CO_2 diffusion, B) CH_4 diffusion, C) CH_4 ebullition.

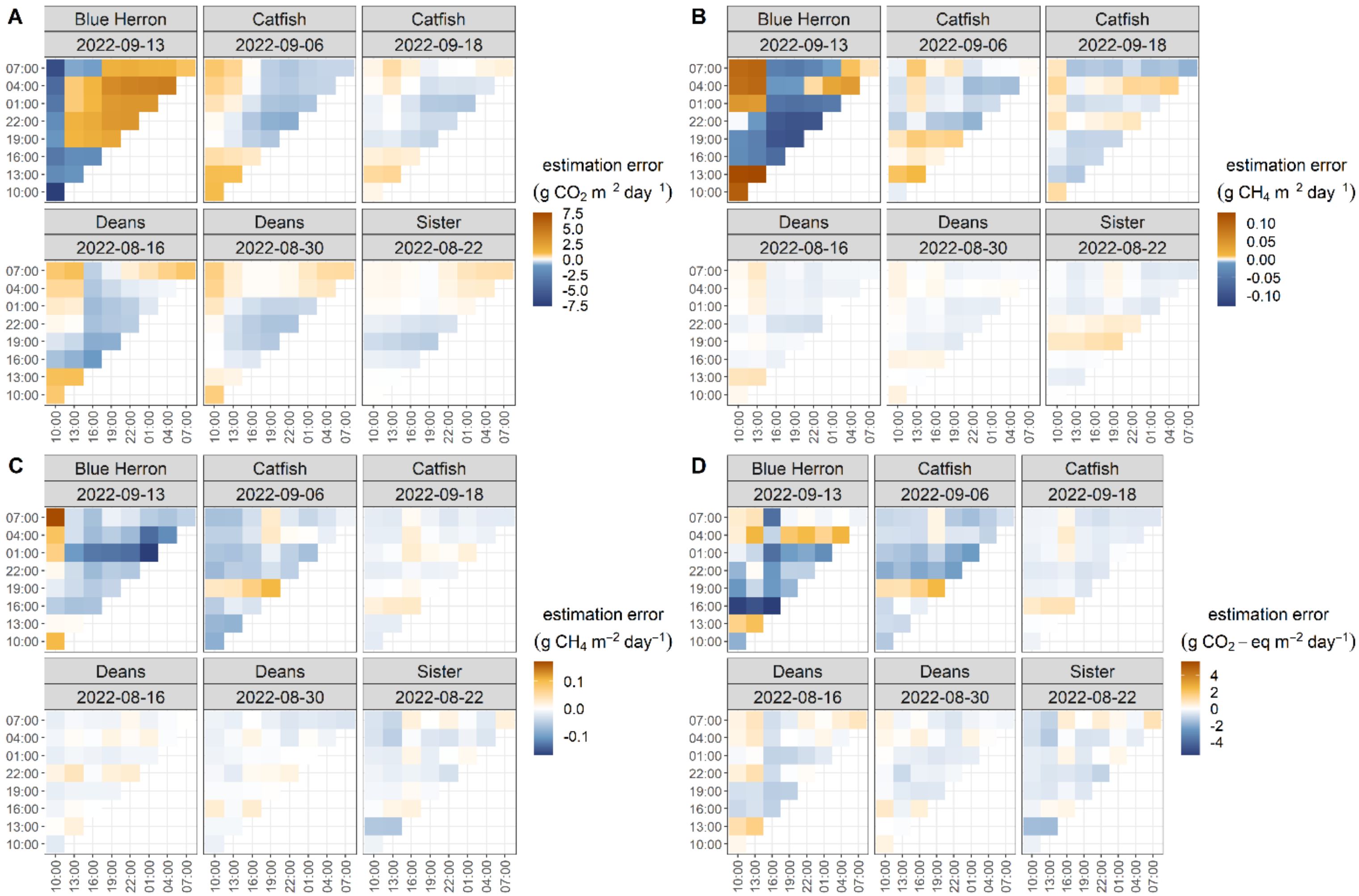

Figure 2.4. A) CO_2 diffusion, B) CH_4 diffusion, C) CH_4 ebullition, and D) CO_2-eq flux estimation error associated with different combinations of one to two sampling times throughout the day. The values along the diagonal represent sampling at only one time during the day.

CHAPTER 3

CONSEQUENCES OF DAM REMOVAL FOR CARBON STORAGE AND EMISSIONS[2]

[2] Naslund, L.C., Mehring, A., Tomczyk, N., Rosemond, A., McKay, S. K., Bernhardt, E., Wenger, S. To be submitted to *Environmental Science & Technology*.

Abstract

Dams can facilitate high carbon dioxide and methane emissions from their reservoirs by flooding terrestrial environments, prolonging water residence times, and concentrating organic matter in oxygen-depleted environments. Reduced emissions has been proposed as a benefit of dam removal; however, the impact of removal on carbon balance depends on several pathways of gas exchange before, during, and after removal. For example, reservoir drawdown can expose previously buried sediment organic matter to oxygen, leading to rapid mineralization, while vegetation regrowth on these same newly exposed sediments can sequester biomass carbon. We developed an analytical approach to account for the carbon emissions and sequestration that determine carbon balance in the reservoir footprint before, during, and after dam removal. We applied this framework to estimate the carbon consequences of removing the Glines Canyon Dam (Elwha River, WA) and the Veazie Dam (Penobscot River, ME), using literature values, statistical, and mechanistic models to estimate carbon balance. We estimated that before removal the Veazie reservoir footprint was likely a small source for CO_2-eq (54.3 Mg CO_2-eq yr^{-1} (-317, 151) while the Glines Canyon reservoir footprint was a large sink (-11500 Mg CO_2-eq yr^{-1} (-33600, -2500)). Over the 100 years after dam removal, we estimated that the Veazie reservoir footprint became a larger source for CO_2-eq (5140 Mg CO_2-eq yr^{-1} (44.4, 24700)) while the Glines Canyon reservoir footprint became a weaker sink (-479 Mg CO_2-eq yr^{-1} (-3220, 3690)), indicating that the carbon balance after removal likely depends on the dam's geomorphic and ecological setting, but the removal of both focal dams reduced landscape scale carbon sequestration. Discerning the implications of these results for the net costs and benefits of dam removal, however, is complex; the estimated net impact of removal on carbon depends on both the counterfactual and the spatial scale of the analysis. Better understanding the changes in

carbon fluxes that can be attributed to dams and their removal is a critical research need for incorporating carbon impacts into decision-making about dams.

Introduction

Over 16 million dams have been constructed globally to control floods, supply water, produce hydropower, facilitate navigation, and create recreational opportunities (Lehner et al. 2011). However, dams can also degrade fisheries, cause extirpations, elevate safety risks, decrease delivery of sediment to coastal zones, and facilitate high emissions of the greenhouse gases carbon dioxide (CO_2) and methane (CH_4) (Pringle et al. 2000; Deemer et al. 2016; Hansen et al. 2020; Dethier et al. 2022). Dam construction in North America, Europe, and Australia has declined since the 1970s (Zhang and Gu 2023), and some dams in these regions are being removed to eliminate their negative impacts (Habel et al. 2020). Ideally, dam removal decisions should consider the full range of services and disservices of a dam and its reservoir, but methods for assessing many services and disservices are not readily available (Naslund et al. 2024). A case in point is carbon emissions; reducing emissions is a proposed motivation for dam removal (U.S. EPA 2016; Johnson 2017), yet methods for calculating the net carbon consequences of removal are lacking (Deemer et al. 2016). In this paper, we outline an approach for estimating the consequences of dam removal for carbon emissions and storage and apply it to two case studies of past dam removals with contrasting characteristics.

Dams facilitate CO_2 and CH_4 emissions from their reservoirs by inundating terrestrial soils and vegetation and delivering organic matter to low oxygen environments (Friedl and Wüest 2002). Because water residence time is a key control on the biogeochemical states and processes which facilitate carbon emissions from dams (Maavara et al. 2020), dam removal could reduce water surface emissions by decreasing residence times. However, other pathways of carbon emissions and burial are impacted by dam removal (Amani et al. 2022; Liang et al. 2024) (Figure 3.1). The relative magnitudes of these fluxes determine the net carbon balance, yet many have never been measured empirically, hindering our ability to assess the overall effect of dam removal on landscape carbon storage and emissions. We address this knowledge gap by developing an analytical approach to estimate carbon emissions and storage before, during, and after dam removal.

Before the dam is removed, the two major contributors to carbon balance in the reservoir footprint are gas exchange from the water surface and carbon burial in reservoir sediments (Prairie et al. 2018). Surface gas exchange includes the diffusion of CO_2 and CH_4 from the reservoir surface and the bubbling of CH_4 from sediments (CH_4 ebullition). Methane can also be emitted from water which is degassed as it is discharged from the reservoir outlet, increasing total reservoir CH_4 emissions. Because CH_4 has a higher global warming potential than CO_2, CH_4 tends to dominate radiative forcing from reservoirs (Deemer et al. 2016). However, reservoirs can also efficiently bury carbon by creating conditions which promote high organic matter sedimentation rates and slow breakdown (Mendonça et al. 2017). Some of this burial may be considered an offset of emissions from the reservoir surface. In attributing carbon emissions or carbon sequestration through burial to the dam, it is important to acknowledge that some portion of total emissions and burial would have happened regardless of the dam's existence; free-flowing rivers emit CO_2 and CH_4 loaded from soils and from in-situ metabolism, and organic carbon transported in flow can be buried elsewhere in river floodplains, terminal lakes, or in the coastal ocean (Wohl et al. 2012; Prairie et al. 2018).

During the removal process as reservoirs are dewatered, previously buried carbon in sediments can be exposed to the air, accelerating its mineralization and subsequent CO_2

emissions (Keller et al. 2021). Breakdown of organic matter in anoxic sites and CH_4 release through drying macropores can also yield CH_4 emissions from the exposed sediment surface (Kosten et al. 2018; Paranaíba et al. 2022). Prior to sediment exposure, drawdown of reservoir water levels can enhance the bubbling of CH_4 from sediments. Drawdown decreases hydrostatic pressure, which is a major driver of CH_4 bubbling, facilitating CH_4 release in a process known as drawdown ebullition. Methane in bubbles can escape oxidation in the epilimnion as they rise quickly to the reservoir surface (Harrison et al. 2017). Because sediment organic matter eroded from the reservoir after dam removal could be transported to higher-oxygen environments, mineralization of that previously buried organic matter could generate additional emissions. We define exposed sediment emissions as originating from sediments which remained in the reservoir footprint after removal and eroded sediment emissions as originating from sediments which were transported out of the reservoir footprint. Together exposed sediment CO_2 and CH_4 emissions, drawdown ebullition, and eroded sediment emissions may contribute to a short-term release, or "burp", of carbon during dam removal.

After dam removal, the now free-flowing river, terrestrial vegetation, and soils influence the carbon balance through exchange of CO_2 and CH_4 with the atmosphere (Hotchkiss et al. 2015; Besnard et al. 2018; Covey and Megonigal 2019; Gatica et al. 2020; Stanley et al. 2023). Photosynthesis and respiration in the terrestrial environment facilitate CO_2 exchange. When this net ecosystem production (NEP) yields greater CO_2 uptake than emissions, it can offset at least some of the "burp" emissions. Upland soils are generally CH_4 sinks while wetland soils are generally CH_4 sources (Gatica et al. 2020). Trees can serve as conduits for soil CH_4 emissions; however, the magnitude of tree CH_4 flux varies greatly among different environments (Covey and Megonigal 2019).

To understand the net impact of dam removal on carbon balance, we modeled each of the above-described fluxes in the reservoir footprint for two dam removals during three critical periods: before dam removal, during the removal carbon "burp," and after removal (Figure 3.1). With these examples, we aimed to 1) characterize the potential impacts of dam removals on carbon balance and 2) identify key uncertainties in our approach to resolve with future study. We then discuss our results in the context of the alternative fates of the emitted and stored carbon in a scenario without dam removal and a scenario with no dam at all.

Methods

Focal dam removals

We selected the Glines Canyon Dam and the Veazie Dam as the focal dam removals for our analyses. We selected these removals because they are among the best studied, which allowed us to parameterize our models with published information. The Glines Canyon Dam was an 87-year old, 64 m tall, 46 m long concrete arch dam removed from the Elwha River, Washington, USA in 2014. The Veazie Dam was a 100-year old, 10 m tall, 253 m long concrete buttress dam removed from the Penobscot River, Maine, USA in 2013 (Wieferich et al. 2016). Both dams were constructed for hydropower and both were removed to restore anadromous fish passage (Randle et al. 2015; Stratton and Grant 2019; Collins et al. 2020). The reservoirs of these dams were a similar size (Glines Canyon: 1.5 km^2; Veazie: 1.6 km^2) and had similar watershed forest cover (Glines Canyon: 80%; Veazie: 67%) and trophic state (oligotrophic) (Munn et al. 1999; Cronan 2012), which are all factors important for reservoir carbon emissions (Deemer et al. 2016; Beaulieu et al. 2020). We expected greater sedimentation rates in the Glines Canyon reservoir because the dam was located in a glaciated landscape with no major upstream dams while the Veazie was located in a post-glacial landscape with six upstream dams. The removal of the Glines Canyon Dam exposed a large area of reservoir sediments, and the resulting water surface area in the reservoir footprint was much smaller than that prior to removal. The removal of the Veazie Dam exposed a much smaller area of reservoir sediments, and the resulting water surface area in the reservoir footprint was similar to that prior to removal.

General approach

To understand the net impact of dam removal on carbon balance, we estimated twelve major carbon fluxes (Table 3.1). Our approach for modeling each flux depended on the data available to inform our estimates. If a flux had been estimated in the literature for a particular reservoir, we used the literature value for that flux (e.g., reservoir carbon burial before the removal of the Glines Canyon Dam (Stratton et al. 2019). If an estimate was not available, we searched the literature for a statistical model describing the relationship between the flux of interest and predictors we could readily calculate for our focal dams (e.g., watershed landcover). If we could not find an appropriate statistical model in the literature, we developed a mechanistic model of the flux and parameterized it using first principles, literature values, or statistical models. If we could reasonably constrain the parameter distribution in a mechanistic model, we sampled the distribution to generate many plausible flux values. When we could not reasonably constrain a parameter distribution, we selected a few contrasting values to generate scenarios of plausible fluxes bracketed by our selection of parameter values.

A major aim of this modeling effort was to quantify the uncertainty in our estimates of fluxes to inform future research priorities. There are three major types of uncertainty quantified in our calculations. Measurement uncertainty—the uncertainty due to the limitations of measurement devices, limited sampling effort, and bias—is the dominant form of uncertainty in our reports of literature-derived flux estimates from a particular reservoir. We report this measurement uncertainty along with the mean flux estimates from the literature. In deriving our estimates from statistical or mechanistic models, we encounter two other types of uncertainty. One type of uncertainty comes from the variability of fluxes among reservoirs and our incomplete ability to describe that variability using predictors. We capture this uncertainty by reporting the prediction intervals of our estimates from the statistical models describing the relationship between predictors and our flux of interest. The other type of uncertainty derives from a lack of data about a flux or component of a flux. We deal with this uncertainty by assigning distributions to these values constrained by our knowledge of the system, and simulating confidence intervals by generating 10,000 realizations of plausible flux values and discarding the top and bottom 2.5% of realizations. We report all fluxes in mass of carbon dioxide equivalents (CO_2-eq) and mass of carbon (Appendix B). For our calculations, we assumed that the global warming potential of CH_4 is 34 for consistency with the model of reservoir surface emissions we used (Prairie et al. 2017b).

Specific fluxes

Before: To estimate reservoir surface emissions, we used the G-res tool, which employs statistical models to describe the relationship between reservoir GHG fluxes and reservoir/environmental characteristics, parameterized using literature values (Prairie et al. 2017a). This tool returns estimates of CO_2 diffusion, CH_4 diffusion, CH_4 ebullition, and CH_4 degassing as well as a simulated 95% confidence interval for total CO_2-eq emissions. We used the 95% confidence interval returned by G-res to describe uncertainty in our emissions estimates.

We used available estimates of reservoir carbon burial from the Glines Canyon Dam (Stratton et al. 2019). For the Veazie Dam, we estimated carbon burial using a statistical model (Clow et al. 2015). This model consisted of four regressions describing the relationship between

1) reservoir sedimentation rate and reservoir/landscape characteristics, 2) sediment organic carbon content and reservoir/landscape characteristics, 3) dry bulk density and sediment organic carbon content, and 4) burial efficiency, the proportion of sediment organic carbon deposited in reservoir sediments which is eventually buried, and sedimentation rate. We first estimated reservoir sedimentation rate and sediment organic carbon content from their respective regression models. We then generated 10,000 realizations of carbon burial by sampling a normal distribution of each parameter given by the model estimate and prediction interval. We fit the dry bulk density and burial efficiency models and sampled a normal distribution of each parameter. We combined our samples of sedimentation rate, organic carbon content, dry bulk density, and burial efficiency (Eq 1) and simulated the 95% confidence interval by discarding the top and bottom 2.5% of realizations:

Eq 1. Carbon burial (g C m^{-2} yr^{-1}) = sedimentation rate (m^3 sediment yr^{-1}) * dry bulk density (g sediment m^{-3}) * organic carbon content of sediment (g C g^{-1} sediment) * burial efficiency (%) / area (m^2)

Burp: To our knowledge, drawdown ebullition has not been measured during the complete drawdown of a reservoir. For this reason, we employed a mechanistic model in which drawdown ebullition was equal to our estimate of annual ebullition from G-res multiplied by a constant. We assumed that drawdown ebullition would be positively related to total annual ebullition, because we reasoned that a reservoir with higher ebullition would have more CH_4 bubbles stored in the sediment to be released upon drawdown. We bracketed the constant as 0, 5, 10, 25, and 100 times annual ebullition and report drawdown ebullition for each scenario (Appendix B).

There have been several measurements of CO_2 emissions from exposed reservoir sediments after drawdown (Keller et al. 2021); however, the amount of time since drawdown is infrequently reported. We assumed that exposed sediment CO_2 emissions decline exponentially over time (Marcé et al. 2019), similar to trajectories of detritus decomposition (Cebrian and Lartigue 2004); thus, a measurement of CO_2 flux from exposed reservoir sediments represents a point-in-time measurement which could poorly represent total emissions via this pathway. To account for the decay in emissions over time, we sampled exposed sediment CO_2 fluxes from the literature and set them equal to the initial rate of emissions in exponential decay functions which represented a near-cessation of emissions after one month, one year, five years, ten years, and thirty years. We then integrated under this curve from 0 to 100 years to represent the total exposed sediment CO_2 emissions.

Methane fluxes from exposed sediments have also been measured in several reservoirs, with some studies reporting CH_4 emissions and some reporting CH_4 uptake from exposed sediments. Like sediment CO_2 emissions, the time since sediment exposure is rarely reported with estimates of sediment CH_4 fluxes. To account for the change in sediment CH_4 flux over time, we set samples of a distribution of measured CH_4 fluxes (Paranaíba et al. 2022) equal to the initial rate of emissions in exponential decay functions as described above for sediment CO_2 emissions. A laboratory study of sediment gas flux upon drawdown suggested that these in situ measurements of sediment CH_4 flux may miss a substantial release of CH_4 as sediment drying creates macropores through which stored CH_4 can escape (Kosten et al. 2018). We refer to this flux as pore CH_4 emissions and report sediment CH_4 flux with and without the pore CH_4 emissions observed in Kosten et al. (2018).

We employed a mechanistic model to estimate CO_2 fluxes from the mineralization of organic matter in eroded sediment (Eq 2). This model assumes that a constant proportion of the organic matter in sediment will be mineralized during the erosion process. To estimate sediment organic carbon, we derived distributions of eroded sediment mass from the literature (Table B2) and combined these distributions with distributions of sediment organic matter content from the literature or from the regression models used to estimate reservoir carbon burial (Clow et al. 2015). We assigned the amount of OC oxidized in transport to a uniform distribution between 0 and 100%, based on the range of fractions of soil organic carbon oxidized in transport compiled by Lal (2003).

Eq 2. Eroded sediment emissions (g C) = eroded sediment mass (g sediment) * % organic matter * % OC oxidized in transport

After: We estimated river CO_2 and CH_4 emissions after removal by sampling fluxes from U.S. rivers binned by discharge. For river CO_2 emissions, we assigned gamma distributions to fluxes in the discharge bins corresponding to our focal reservoirs based on summary statistics reported in Hotchkiss et al. (2015). For river CH_4 emissions, we randomly sampled measured CH_4 fluxes in the corresponding discharge bins in Stanley et al. (2023). We assume that carbon burial in the newly free-flowing river is negligible (Cole et al. 2007), and thus only model surface gas exchange from the river after dam removal.

We estimated soil CH_4 flux using a statistical model relating soil CH_4 flux to environmental and soil characteristics (Gatica et al. 2020) (Table B1). We report our uncertainty as the prediction interval of our estimates derived from this model.

We estimated tree CH_4 fluxes using a mechanistic model which combined estimates of tree trunk surface area and measured tree CH_4 fluxes. We modeled three scenarios based on patterns of tree CH_4 fluxes observed in the literature (Warner et al. 2017; Pitz et al. 2018; Putkinen et al. 2021): 1) tree CH_4 from the trunk is entirely offset by oxidation of CH_4 in the canopy, thus tree CH_4 flux is zero, 2) tree CH_4 emissions only occur in the first 3 m of the trunk height, 3) tree CH_4 emissions occur constantly throughout the trunk. To estimate tree surface area over time, we used the Forest Vegetation Simulator (FVS), which is an individual tree growth and yield model capable of "growing" trees from bare ground (Dixon 2002). Our initial exploration of the FVS model determined that the location (nearest national forest), tree species, and density at the end of the first cycle (10 years) were the most influential parameters for the bare ground simulations. We selected tree species for our focal reservoirs based on the dominant riparian trees in the surrounding area reported in the literature. We parameterized tree density using the densities measured in two ten year old plots in a chronosequence of floodplain tree density along the Queets River in Olympic National Park, Washington, USA (Van Pelt et al. 2006). We assumed that trees in the floodplain would exhibit CH_4 flux dynamics more similar to wetland plots than upland plots, so we used a distribution of wetland tree CH_4 emissions, unless we had a strong reason to suspect that an area of the reservoir would more closely resemble upland plots (see Appendix B). For each realization, we randomly selected one of the two parameterizations of tree surface area per area of ground and multiplied it by a CH_4 flux rate sampled from the distributions described above. We repeated this process 10,000 times and reported uncertainty as the simulated 95% confidence interval from discarding the upper and lower 2.5% tree CH_4 fluxes.

We estimated net ecosystem production using a statistical model describing NEP using age, an exponential decay relationship between NEP and age, mean annual temperature, and average total N deposition. This model is a modification of one presented in Besnard et al.

(2018) to explain variation in NEP measured at FLUXNET sites. The Besnard et al. (2018) model also included a proxy for gross primary production (GPP) derived from the vapor pressure deficit and latent heat flux, which we excluded because we did not feel that we could estimate future vapor pressure deficit or latent heat flux at our sites with confidence. Our revised model explained variation in NEP at the FLUXNET sites reasonably well ($R^2 = 0.63$). Our median predictions of NEP from our revised model implied substantial emissions early in forest development. We assumed that these emissions represent the mineralization of carbon remaining in the FLUXNET footprint after a site was burned or clear cut. Because we modeled the mineralization of carbon remaining in the reservoir footprint after drawdown in the exposed sediment CO_2 and CH_4 emissions terms, we integrated under the curve of NEP vs. age starting when GPP > ER (~6 years) to calculate NEP in our reservoir footprints.

Results

Prior to dam removal, the Veazie (135 Mg CO_2-eq yr^{-1} (117, 153), Figure 3.2) and the Glines had similar emissions (with degassing: 121 Mg CO_2-eq yr^{-1} (106, 136); without degassing: 110 Mg CO_2-eq yr^{-1} (95.9, 123), Figure 3.2). The Veazie was a small net source of CO_2-eq emissions (54.3 Mg CO_2-eq yr^{-1} (-317, 151), owing to its low rate of burial (-80.9 Mg CO_2-eq yr^{-1} (-434, -2.03), Figure 3.2), and the Glines was a large net sink of CO_2-eq (-11500 Mg CO_2-eq yr^{-1} (-33600, -2500)), owing to its high rate of burial (-11700 Mg CO_2-eq yr^{-1} (-33800, -2630)) (Figure 3.2) (results in g C reported in Appendix B).

We report burp emissions here from three modeled emissions scenarios (Figure 3.3) and report results in terms of specific parameterizations in Appendix B (Figure B1-B3). In the least emissions scenario, drawdown ebullition is zero, exposed sediment CH_4 emissions last only a month with no additional pore CH_4 emissions, and exposed sediment CO_2 emissions last only a month. In the moderate emissions scenario, drawdown is 10 times annual ebullition, exposed sediment CH_4 emissions last five years and include pore CH_4 emissions, and exposed sediment CO_2 emissions last five years. In the most emissions scenario, drawdown is 100 times annual ebullition, exposed sediment CH_4 last 30 years and include pore CH_4 emissions, and exposed sediment CO_2 last 30 years.

The total magnitude of the burp and the contributions of individual fluxes depended primarily on the amount of time exposed sediment CO_2 and CH_4 fluxes persisted and the proportion of annual CH_4 ebullition emitted during drawdown (Figure 3.3, Figure B2). In the least emissions scenario, the Veazie removal “burped” a total of 4.38 Mg CO_2-eq (-0.221, 8.58), 407 Mg CO_2-eq (-53.9, 844) in the moderate emissions scenario, and 3010 Mg CO_2-eq (-476, 6360) in the most emissions scenario. “Burp” emissions from the Veazie removal were equivalent to 11 days, 3 years, and 22 years of average pre-removal emissions from the Veazie reservoir in the least, moderate, and most emissions scenarios, respectively. The Glines removal “burped” 187000 Mg CO_2-eq (7300, 841000) in the least emissions scenario, 190000 Mg CO_2-eq (7660, 847000), and 203000 Mg CO_2-eq (7090, 873000) in the most emissions scenario. “Burp” emissions from the Glines Canyon removal were equivalent to 1542, 1567, and 1679 years of average pre-removal emissions from the Glines Canyon reservoir in the least, moderate, and most emissions scenarios, respectively. The flux which contributed the most CO_2-eq during the burp depended on the selection of values for the time of exposed sediment fluxes and the proportion of ebullition emitted during drawdown for the Veazie removal (Figure 3.3). Exposed sediment emissions were the dominant contributor to CO_2-eq emissions in every scenario for the Glines Canyon removal, although this parameter was highly uncertain (Figure 3.3).

After removal, the Veazie reservoir footprint became a larger source of CO_2-eq (5140 Mg CO_2-eq yr^{-1} (44.4, 24700), Figure 3.4) and the Glines Canyon reservoir footprint became a smaller sink of CO_2-eq (-479 Mg CO_2-eq yr^{-1} (-3220, 3690), Figure 3.4). River CO_2 emissions were a dominant though uncertain contributor to CO_2-eq flux from the Veazie reservoir footprint (5040 Mg CO_2-eq yr^{-1} (333, 22800), Figure 3.2). Owing to the small area of exposed sediment, terrestrial NEP contributed little to carbon balance of the Veazie footprint after removal (-111 Mg CO_2-eq yr^{-1} (-345, 124), Figure 3.2). Because of the larger exposed sediment area in the Glines Canyon footprint, terrestrial NEP was the dominant contributor to carbon balance in the reservoir footprint (-1040 Mg CO_2-eq yr^{-1} (-3200, 1130), Figure 3.2) after dam removal.

If the dams had not been removed, the Veazie reservoir footprint would have stored less CO_2-eq and the Glines Canyon reservoir footprint would have stored much more CO_2-eq than it did after dam removal (Figure 3.5). We estimate that the Glines Canyon reservoir would be entirely sedimented and therefore reach its maximum cumulative carbon storage at approximately 331 years after dam construction (Figure 3.5). Because sedimentation rates were low in the Veazie reservoir, this reservoir footprint could continue to accumulate carbon well beyond this time period. Trajectories of carbon emissions were similar for both dams prior to removal, but emissions from the reservoir footprints diverged after removal (Figure 3.5), owing to the substantially different river areas over which water surface CO_2 and CH_4 emissions occurred in the Veazie and Glines Canyon reservoir footprints.

Discussion

This study represents a first attempt to estimate all major pathways of carbon emissions and burial in the reservoir footprint before, during, and after dam removal. Other studies of the impact of dam removal on carbon have excluded biogenic fluxes entirely (Martinez et al. 2018), omitted important pathways of biogenic fluxes (Liang et al. 2024), or neglected key time periods in the dam removal process such as the burp of carbon during removal or the establishment of terrestrial vegetation and soils after dam removal (Pacca 2007; Amani et al. 2022; Gómez-Gener et al. 2023). Our estimates for the Glines Canyon and Veazie Dam removals indicate that although the relative magnitudes of fluxes from their reservoir footprints varied substantially, both removals likely reduced the sink strength of the landscape, which we take to be the tendency of a landscape to take up rather than emit CO_2-eq. The Veazie reservoir footprint was a small net source of CO_2-eq before removal and became a larger source after removal, owing to high river CO_2 emissions. The Glines Canyon reservoir was a large net sink of CO_2-eq before removal due to high rates of C burial and became a smaller sink after removal, primarily because the accumulation of C from terrestrial NEP was smaller than the accumulation from sediment burial. This comparison suggests that the amount of sediment accumulation in the reservoir before removal and the fate of that sediment after dam removal may be key determinants of the impact of removal on carbon balance.

Comparison of Glines Canyon and Veazie Dam removals to other dam removals

To understand the implications of our results, it is important to contextualize the Glines Canyon and Veazie Dam removals with other dam removals in the United States. Of the U.S. dam removals with reported heights and lengths, the Veazie Dam was taller than 90% and longer than 96% of removed dams. The Glines Canyon was taller than all removed dams and longer than 57% of removed dams (American Rivers 2024). These statistics indicate that most dams removed in the U.S. are smaller than the Glines Canyon and Veazie Dams (Figure B4). Thus, our focal dam removals had the capacity to yield relatively large changes in mass flows due to their size. Both reservoirs were likely oligotrophic, resulting in low CH_4 emissions before removal

(Deemer et al. 2016). While the trophic statuses of other removed reservoirs are not readily accessible, the fact that almost half of lakes and reservoirs in the U.S. are impaired due to excess nutrients (U.S. EPA 2022) suggests that the general population of removed reservoirs may be more eutrophic and thus emit more CH_4 on average than the Veazie and Glines Canyon reservoirs. Reservoir emissions also tend to decline with age (Prairie et al. 2017b), and the Glines Canyon and Veazie Dams were in the middle of the distribution of dam age at the time of removal (Figure B4). These dams also likely represented extremes for carbon burial before removal. Median carbon burial in the Veazie reservoir was an order of magnitude lower and the Glines Canyon reservoir an order of magnitude higher than the reported range of U.S. reservoir carbon burial rates (Clow et al. 2015).

Characteristics of a dam removal which may increase CO_2-eq sink

Although the removal of both the Glines Canyon and Veazie dams reduced the sink strength of the reservoir footprints in our models, our findings point to some characteristics of dam removals which could increase sink strength. The reservoirs of both focal dams had low CH_4 emissions (Deemer et al. 2016). A dam with high CH_4 emissions and low rates of carbon burial would be a larger source of CO_2-eq before removal; thus, its removal may increase the sink strength of the reservoir footprint (Figure 3.6). A eutrophic, cascade reservoir may fit these characteristics, as eutrophic reservoirs are generally high CH_4 emitters, and reservoirs at the downstream end of a cascade can receive relatively low inputs of terrestrial organic carbon (Jager et al. 2022). Low inputs of terrestrially-derived organic matter could potentially yield low rates of carbon burial, provided that burial of carbon fixed within the reservoir does not compensate for the low terrestrially-derived inputs (Downing et al. 2008). The removal of a dam which would expose a large area of sediment could also increase the sink strength of the landscape as terrestrial soils and vegetation develop in the exposed sediment and contribute to CO_2-eq uptake (Figure 3.6). However, high terrestrial NEP with a large, exposed sediment area could also trade off with high exposed sediment emissions after reservoir dewatering. Notably, a recent study of the dewatering of the Wolongquan Reservoir (Inner Mongolia, China) found that emissions from the surface of the reservoir footprint declined during and after removal, despite high exposed sediment CO_2 emissions, because reservoir CH_4 emissions before removal were high (Liang et al. 2024).

Attribution of mass flows to dam removal

To adequately evaluate the impact of dams and their removal on carbon emissions and storage we need to distinguish changes in fluxes at two spatial scales: in the reservoir footprint and in the catchment. We calculated the impact of dam removal on carbon balance in the reservoir footprint because empirical flux measurements are available at this scale. However, changes in fluxes in the reservoir footprint may not result in changes "experienced" by the atmosphere if equal and opposite changes in those fluxes occur elsewhere in the catchment; in this case, carbon emissions or storage are simply displaced in space. We also need to evaluate whether fluxes are displaced in time (i.e., whether they would have occurred regardless of dam removal but over a different time period). While emissions displaced in time may not be considered new emissions attributable to dam removal, some have argued that deferring emissions should be counted as a benefit of dams due to the greater value of reducing current versus future emissions (Jager 2022).

If without dam removal, the sediment organic matter which produces exposed sediment CO_2 and CH_4 emissions is buried in anoxic reservoir sediments, these exposed sediment emissions can largely be considered attributable to dam removal. Eroded sediment emissions may also be attributable to removal, as the alternative fate of this sediment carbon is burial in anoxic sediments. If the enhanced bubbling of CH_4 during drawdown allows more CH_4 to escape oxidation in the epilimnion than if drawdown never happened (Harrison et al. 2017), a portion of this flux may be considered emissions attributable to dam removal. Terrestrial net ecosystem production in the former reservoir footprint as well as tree and soil CH_4 exchange can also be considered fluxes attributable to removal. We modeled higher river CO_2 emissions after dam removal than reservoir emissions; however, these higher emissions in our model do not represent removal-attributable emissions for the catchment. Only the emissions resulting from enhanced production of CO_2 in the reservoir footprint after removal can be attributable to removal, and our model does not capture these emissions as it does not employ data from river reaches flowing through former reservoir footprints. We attribute the differences in river and reservoir CO_2 emissions in our model to higher rates of gas exchange in the free-flowing than the impounded river (Hall and Ulseth 2020).

Whether the appropriate counterfactual for CO_2-eq emissions due to dam removal is that 1) the dam remains in place or that 2) the dam never existed depends on the application of this information. The first counterfactual implies that the dam can be maintained throughout the timescale of the analysis with emissions comparable to present emissions, while the second counterfactual allows us to incorporate removal emissions into a lifecycle assessment of the dam. If we assume that the dam remains in place, it is important to recognize that reservoir carbon storage is limited by reservoir storage capacity; when reservoirs are entirely sedimented, they cannot continue to accumulate carbon (Figure 3.5). Additionally, if significant repairs are required to maintain the dam, the process of dam repair may also emit additional emissions (e.g., if repair requires reservoir drawdown, which could result in exposed sediment emissions).

If we compare removal emissions to a scenario in which the dam never existed, we might assume that a portion of the sediment CO_2 and CH_4 emissions during dam removal would not have occurred if the dam never existed. Some of the organic matter that these emissions originated from likely would have remained inundated throughout its transport to its final burial location, resulting in lower total breakdown prior to burial. If deposited elsewhere in the river network or in the coastal ocean, this organic matter may have also been exposed to higher oxygen and/or sulfate conditions compared to anoxic reservoir sediments, reducing overall CH_4 production and emissions. Eroded sediment emissions, in contrast, can be considered emissions displaced in time. In the scenario without the dam's existence, breakdown of organic matter transported in the river network would have happened over time rather than occurring during the relatively brief period of sediment erosion from the reservoir after dam removal. For this reason, eroded sediment emissions may not be considered emissions attributable to dam construction and removal. Assuming that higher sediment oxygen conditions without the dam limited CH_4 production, we would also assume that most of the drawdown ebullition can be attributed to the dam and included in a lifecycle assessment.

Priority future inquiries

The Glines Canyon Dam example illustrates that buried carbon behind dams can be vulnerable to mineralization at dam removal. Our results support the exploration of strategies to protect this buried carbon during the removal process. Strategies to slow the erosion of sediment from the reservoir and accelerate the development of terrestrial biomass, measures which are frequently employed to mitigate water quality impacts of removal (Chenoweth et al. 2022), may also function to reduce the loss of accumulated carbon from the reservoir footprint. If removal methods alter the total volume of sediment ultimately eroded from the reservoir (Major et al. 2017), removal method may also impact the carbon consequences of dam removal.

Our findings also support the development of more complete lifecycle assessments of carbon emissions attributable to dams, including for the thousands of large hydropower dams currently being planned or constructed in countries with emerging economies (Zarfl et al. 2015; Flecker et al. 2022). To date most lifecycle assessments of dams do not include emissions during dam decommissioning, and of those which do, few consider biogenic emissions attributable to removal (Song et al. 2018). Our results indicate that biogenic emissions during removal can constitute a substantial portion of a dam's lifetime carbon emissions and should be considered in the carbon costs of dams. Assuming that most dams will eventually be removed intentionally or by structural failures, emissions attributable to dam removal should be considered a carbon cost incurred at dam construction. The framework outlined here can be used to understand the magnitude of these "sunken" carbon costs.

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Table 3.1. Summary of approaches to estimate carbon storage and emissions from reservoirs before, during, and after dam removal. Additional details on data sources used for model parameterization can be found in Table B1 and B2.

Period	Flux	Approach	Source(s)
Before	Reservoir surface emissions	Statistical model	G-res (Prairie et al. 2017a)
	Reservoir carbon burial	Measured value or statistical model	Measured (Elwha and Glines Canyon dams): (Stratton et al. 2019) Statistical model: (Clow et al. 2015)
Burp	Drawdown ebullition	Mechanistic model	Annual ebullition: G-res output Drawdown ebullition proportion of annual ebullition: bracketed values
	Reservoir sediment CO_2 emissions	Mechanistic model	Initial sediment CO_2 emissions: values compiled in (Keller et al. 2020) and additional values in (Deshmukh et al. 2014; Gómez-Gener et al. 2015; Li et al. 2016; Jin et al. 2016; Almeida et al. 2019; Marcé et al. 2019; Pozzo-Pirotta et al. 2022)
	Reservoir sediment CH_4 emissions	Mechanistic model	Initial sediment CH_4 emissions: (Paranaíba et al. 2022)

			Pore CH_4 emissions: (Kosten et al. 2018)
	Eroded sediment CO_2 emissions	Mechanistic model	Mass of eroded sediment: literature values Organic carbon content of sediment: literature values or (Clow et al. 2015)
After	River CO_2 emissions	Statistical model	(Hotchkiss et al. 2015)
	Reservoir CH_4 emissions	Statistical model	(Stanley et al. 2023)
	Soil CH_4 flux	Statistical model	(Gatica et al. 2020)
	Tree CH_4 flux	Mechanistic model	Tree surface area: Forest Vegetation Simulator (FVS) (U.S. Forest Service) Angiosperm CH_4 emissions: (Pitz et al. 2018) Gymnosperm CH_4 emissions: (Machacova et al. 2016)
	Net ecosystem production	Statistical model	Modified from (Besnard et al. 2018)

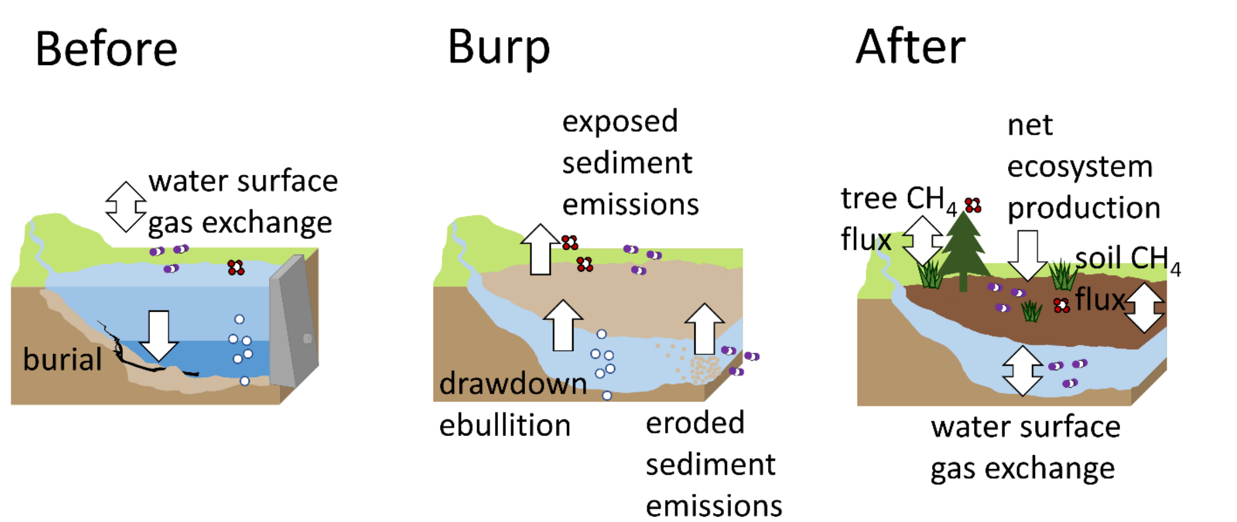

Figure 3.1. Pathways of carbon emissions and storage before, during, and after dam removal. Note that the arrows are the same size for every pathway, so as not to imply a priori hypotheses about the relative magnitudes of carbon fluxes.

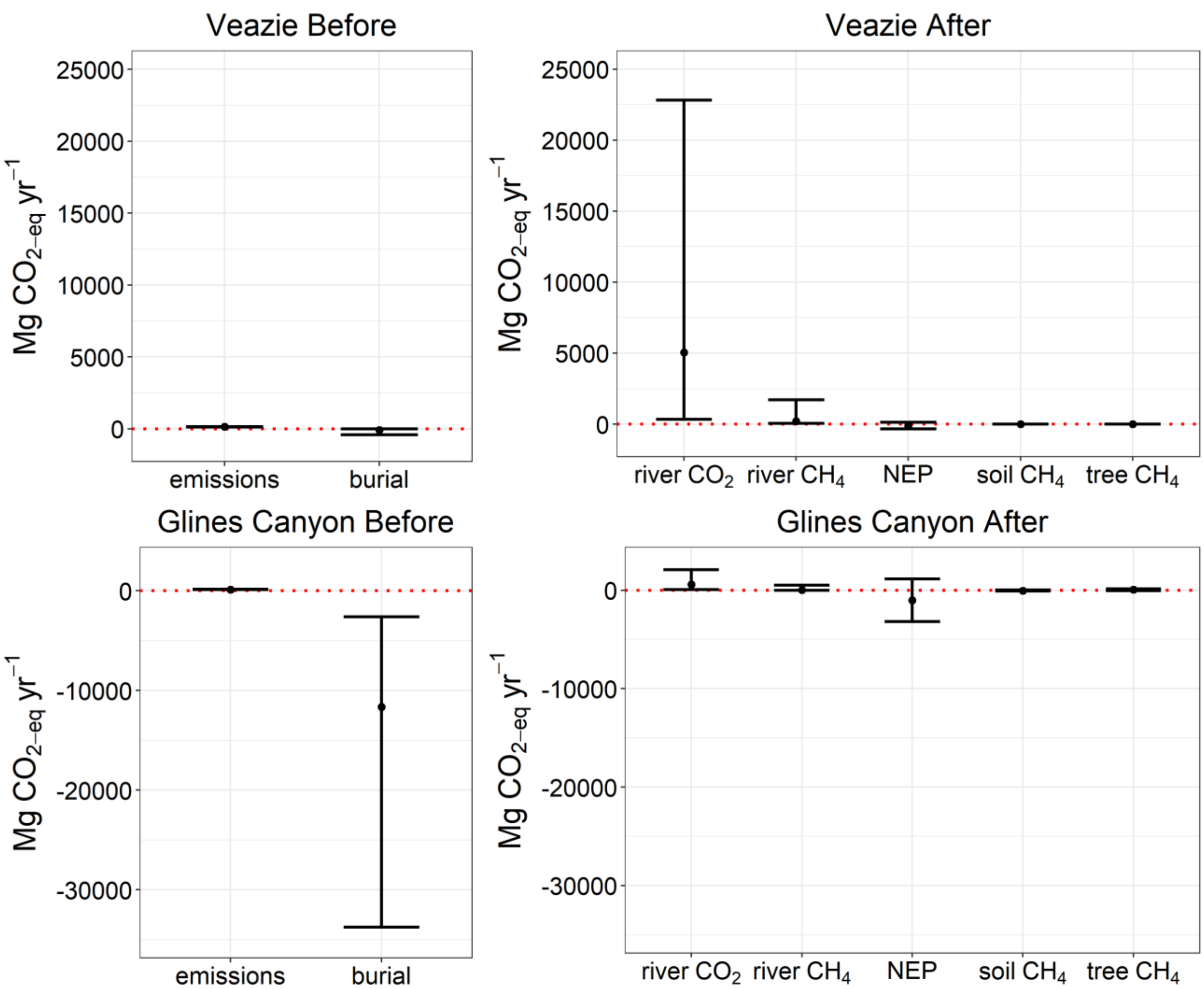

Figure 3.2. Mass flows before and after removal of the Glines Canyon and Veazie Dams, including degassing emissions for the Glines Canyon Dam. The red dotted line indicates 0.

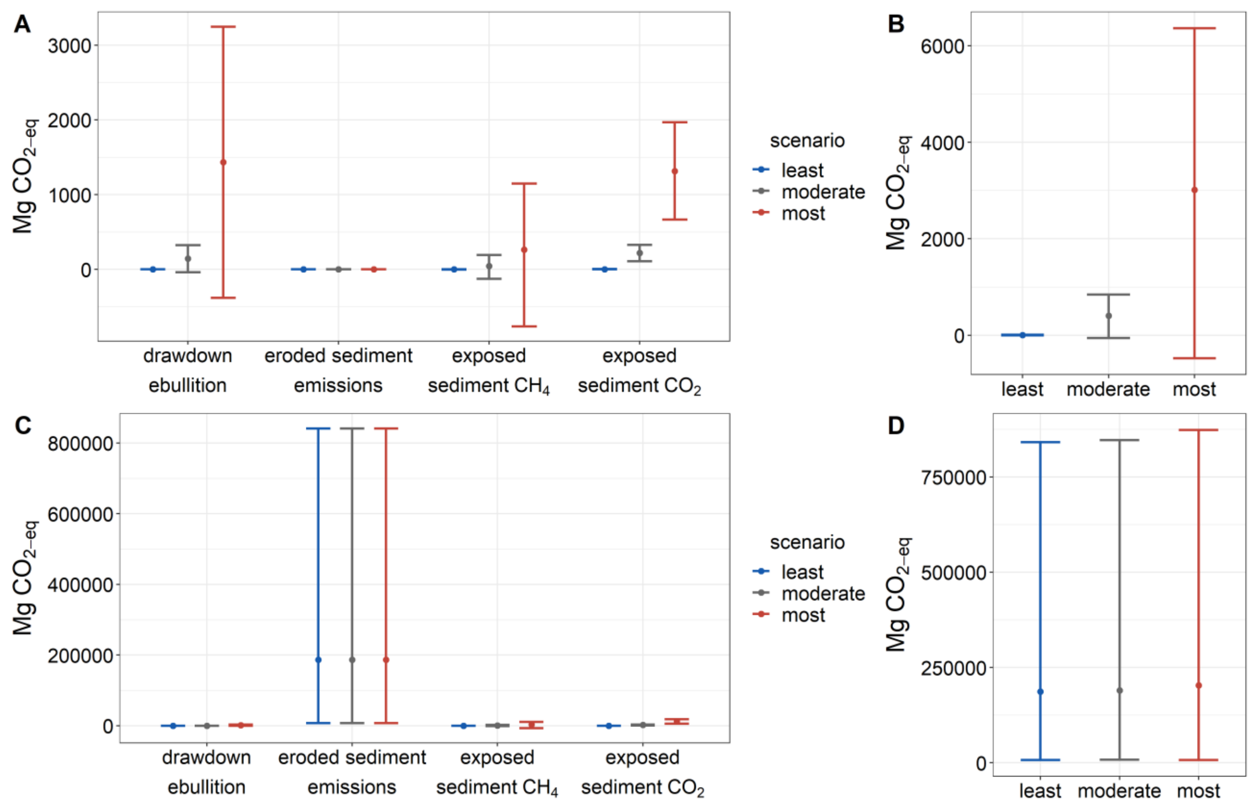

Figure 3.3. Pathways of "burp" emissions for the A) Veazie Dam and C) Glines Canyon Dam removals and the cumulative "burp" emissions for the B) Veazie Dam and D) Glines Canyon Dam removals under three modeled scenarios of emissions (least, moderate, and most emissions). Eroded sediment emissions do not vary among scenarios.

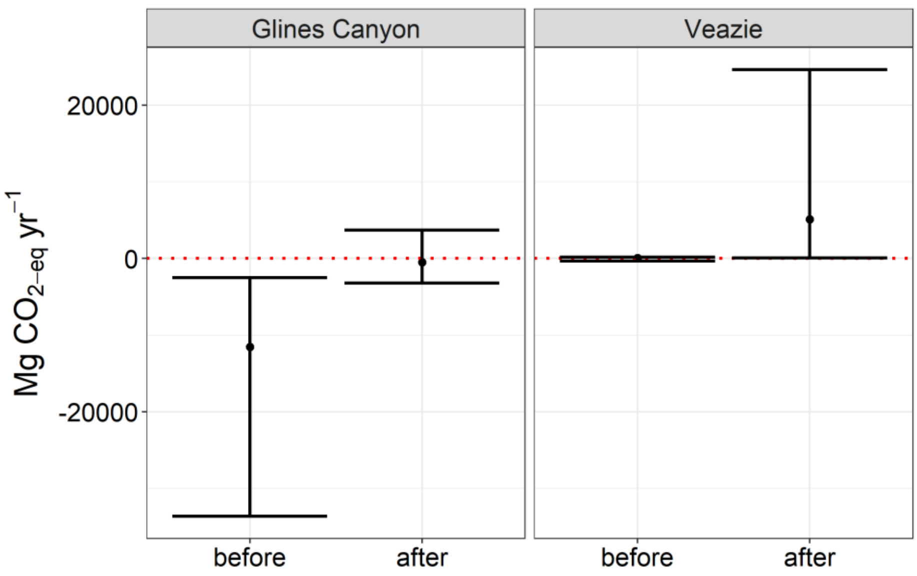

Figure 3.4. Net CO_2-eq mass flow before and after dam removal, assuming trees in both dam footprints are angiosperms emitting CH_4 only over the first three meters of the trunk.

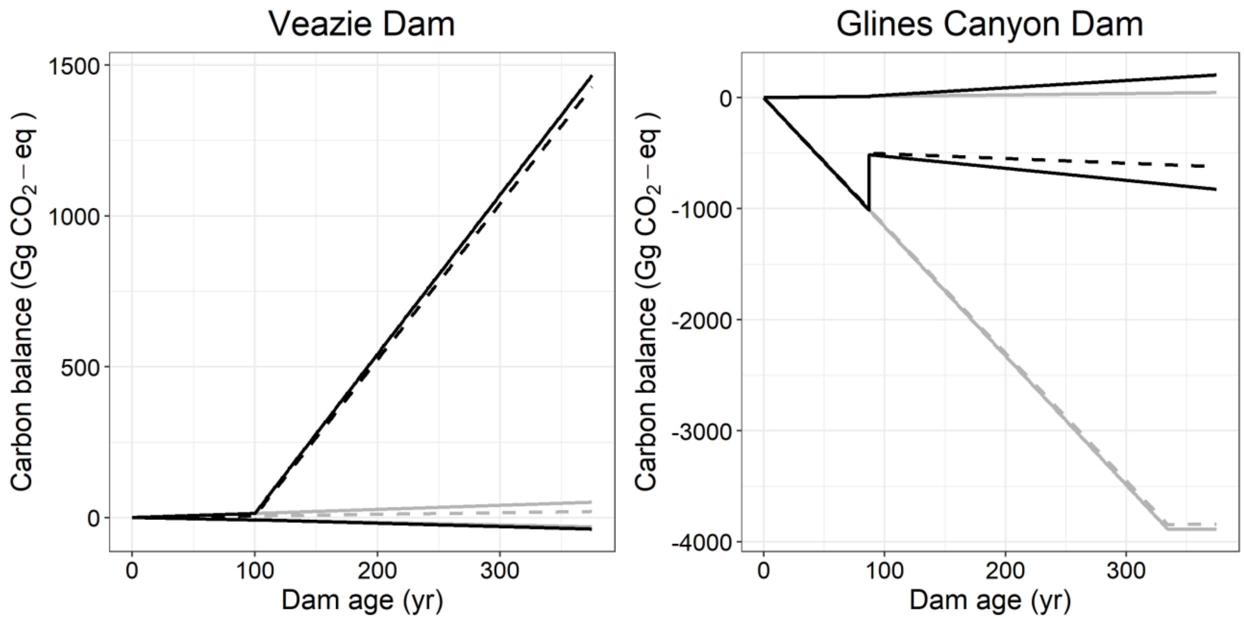

Figure 3.5. Simplified trajectories of emissions and storage (solid lines above and below the x-axis) and net carbon balance (dashed lines) in the Veazie and Glines Canyon reservoir footprints with dam removal (black lines) and without dam removal (grey lines). This figure represents values derived from median fluxes only, and only accounts for annualized rates of carbon burial and emissions integrated over 100 years. It does not, for example, account for changes in NEP over time in the regrowing reservoir. The burp emissions modelled here are from the moderate emissions scenarios. Eroded sediment emissions are not depicted in cumulative carbon emissions, as these emissions occur outside of the reservoir footprint, but eroded sediment contributes to the loss of cumulative carbon storage.

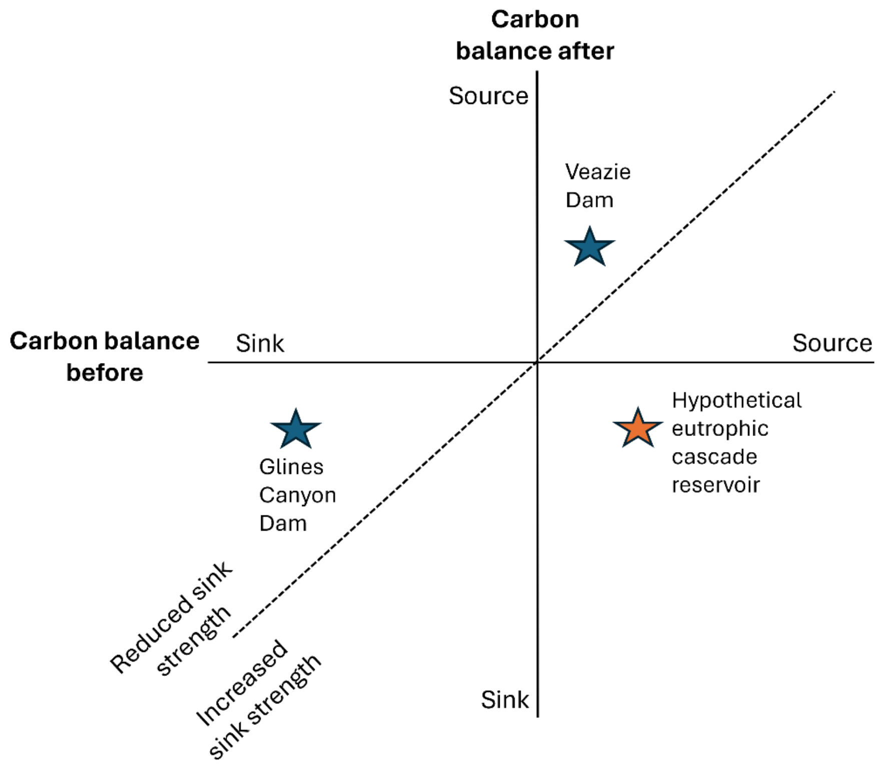

Figure 3.6. Carbon balance before and after the removal of the Glines Canyon Dam, Veazie Dam, and a hypothetical dam with a eutrophic, cascade reservoir. The location of points relative to the dashed 1:1 line indicates whether removals are modeled (Veazie Dam and Glines Canyon Dam) or hypothesized (hypothetical, eutrophic, cascade reservoir) to reduce or increase the sink strength of the reservoir footprint.

CHAPTER 4

FACILITATING DAM REMOVAL DECISIONS WITH MULTIPLE OBJECTIVES[3]

[3] Naslund, L.C., Buhr, D., Chambers, M., McKay, S. K., Jumani, S., Bledsoe, B., Rosemond, A., Wenger, S. Submitted to *River Research and Applications*, *5/16/24.*

Abstract

Proactive and transparent decision-making about the long-term management of dams, including rehabilitation, retrofit, and removal, is critical for successfully managing this aging infrastructure and presents an opportunity to weigh the many services and disservices dams provide. Existing tools to support dam removal decisions often constrain decision processes by considering only a limited set of user objectives. We identified 18 dam removal objectives in the literature and found that most of the 41 dam removal decision-support tools we evaluated included only two or three objectives. Common objectives of previous dam removals like reducing safety hazard and expanding recreational opportunities were included in few decision-support tools. To facilitate dam removal decisions with diverse objectives, we created a new web application which supports decisions with the 18 objectives that we identified in the literature. The application guides users to select appropriate objectives, choose metrics and methods to evaluate management alternatives, and identify additional decision-support tools to weigh alternatives relative to selected objectives. We demonstrate this web application as a resource for the dam management community of practice with a case study of a dam removal decision in Athens, Georgia, USA. More broadly, we propose the process outlined here as a model for aligning diverse objectives in other types of river infrastructure decisions. Given the contribution of this infrastructure to declining biodiversity, intensifying climate, and development needs, failing to align multiple objectives in river infrastructure decisions can represent consequential, missed opportunities.

Introduction

Dam infrastructure has been critical to global economic development, but many dams and their reservoirs have exceeded their design lifetimes and are no longer fulfilling their constructed purposes (Gonzales & Walls, 2020; Perera et al., 2021). In the United States, decades of disinvestment in maintenance has also led to significant deficiencies, such that the American Society of Civil Engineers rated dams as one of the worst-performing categories of infrastructure (ASCE, 2021). The estimated cost to repair all known deficient, non-federal dams in the US National Inventory is $157.5 billion (ASDSO, 2023); however, this estimate includes only a fraction of the total number of dams in the US. For many deficient dams, removal may be a more economically efficient alternative to repair, particularly when the dam no longer serves its constructed purpose (Doyle, Stanley, et al., 2003; Grabowski et al., 2018; IEC, 2015). Even for functioning dams, removal may be the best long-term management strategy due to benefits for wildlife, water quality, recreation, safety, flood risk mitigation, and other services (Hansen et al., 2020).

Dam management, including the decision of whether to remove a dam or which dams to remove, requires weighing the many services and disservices provided by a dammed versus a free-flowing stream, which can vary greatly according to local context (Habel et al., 2020). However, management decisions may not be made according to formal cost-benefit analyses, and the definition of the benefits provided by alternative actions may be disputed among interested parties (Grabowski et al., 2017; Habel et al., 2024). Additionally, dam decommissioning decisions encompass multiple scales of complexity including variation in the number and spatial relationship of dams considered for removal, dam size and impounded volume, diverse ownership, relevant legal authorities, and administrative frameworks (McKay et al., 2020; USSD, 2015). In part due to this complexity, most dams have been removed in an ad hoc manner despite calls for coordinated and structured decision processes (Doyle, Harbor, et al., 2003; Neeson et al., 2015). However, recent large infrastructure investments, such as the $3 billion allocated for dam decommissioning, rehabilitation, and retrofit in the 2021 U.S. Infrastructure Investment and Jobs Act (Pub. L. 117-58) (ASDSO, 2022), may open a policy window for more coordinated dam decision-making.

Several organizations have developed tools to support structured decision-making about dams, but existing tools may constrain options or lead to suboptimal outcomes if they do not include all relevant objectives of concern. We propose that decision-support tools could introduce automation bias in objectives selection: a tendency to favor outputs from automated over non-automated systems in decision-making. Identifying objectives is a critical early step in structured decision-making (Gregory & Keeney, 2002), and automation bias induced by a decision-support tool could lead to the premature elimination of relevant objectives. To facilitate decision-making inclusive of diverse objectives, we (1) compiled a list of objectives relevant to dam removal decisions, (2) examined their representation in existing dam removal decision-support tools which we identified in a literature search (n = 41, see below), (3) developed a new web application that links objectives to relevant metrics, methods, data sources, and decision-support tools, and (4) applied this web application to a case study about whether to remove a dam. We emphasize that the goal of this effort is not to contribute another multi-objective dam removal optimization tool, but rather to advocate for dam management decision processes that center clearly defined objectives. Our web application facilitates this process by aiding in the selection and application of dam management objectives in diverse deliberation contexts.

Dam removal objectives and their representation in decision-support tools

To identify common objectives in dam removal decisions and evaluate their inclusion in existing decision-support tools, we performed a two-stage search of the literature. We first conducted a preliminary literature review to develop an initial list of objectives of dam removal decisions. These objectives included motivations for dam removal (e.g., maximize swift water recreational opportunities) as well as services to be maintained regardless of the removal decision (e.g., meet navigational demands). We then refined this list as we conducted a second, structured review of existing decision-support tools to determine how frequently our identified objectives were included. For the structured review of existing dam removal decision-support tools, we searched Web of Science on April 11, 2022, using the search strings "(multiobjective OR multi-objective OR multicriteria OR multi-criteria) AND dam removal" as well as "prioritiz* AND (dam removal OR dam decommissioning)") and reviewed all results. Although the terms used for our search emphasize decisions to remove dams, we assert that decisions to retrofit or repair dams are the other side of the same coin, and generally encompass the same objectives. If dams are removed because their perceived costs outweigh their perceived benefits, it stands to reason that dams are retrofitted or repaired because they do currently or could provide perceived benefits in excess of their perceived costs after modifications (Parent et al., 2024). Therefore, we propose that the objectives which generate perceived costs and benefits are not materially different between these decision alternatives.

From the 86 results from our search, we identified 9 additional, relevant papers that were not captured in our original search. We also reviewed 13 river barrier prioritization web applications identified by American Rivers (American Rivers, 2022). We included papers and applications (hereon referred to collectively as decision-support tools) returned by these search parameters which concerned optimal siting for dam construction, as many criteria are shared between dam construction and removal decisions. We did not include tools in our analysis if they only included metrics associated with a single objective (16% of evaluated literature, e.g., Kocovsky et al., 2009) or solely concerned water infrastructure other than dams (13% of evaluated literature, e.g., González-Zeas et al., 2019). Of the 108 evaluated tools and papers, 41 met our inclusion criteria. In total, we identified 18 common objectives of dam removal, which we organized into five categories (Table 4.1).

As stated above, the terms used for our structured search of decision-support tools emphasized dam removal rather than retrofit or repair. It is possible that tools explicitly designed for dam retrofit and repair decisions not captured by our search represent objectives differently than the results reported below. Out of 41 dam removal decision-support tools, only ten included more than three objectives. The tool with the greatest number of objectives included 9 of the 18 we identified (Brown et al., 2009). *Supporting/maintaining a population or community of focal taxa* was the most frequently included objective (33/41 tools), followed by *minimizing implementation cost* (29/41 tools) (Figure 4.1). Unsurprisingly, these objectives were the most frequently co-occurring, followed by the combination of *supporting/maintaining a population or community of focal taxa* and *meeting power generation needs* (Figure 4.1). No tool included the objective *reduce stream geomorphic degradation* (Schmidt & Wilcock, 2008). The objectives *reduce personal safety risk, maximize swift water recreational opportunities, maximize flat water recreational opportunities, minimize adverse impacts to sites judged by the community, state or nation to be historically significant places* were each represented in only one tool. Because these objectives were poorly represented in existing decision-support tools, we identified them as high priorities for metric development. We anticipated that developing metrics for these objectives would facilitate their inclusion in structured decision-making for evaluating management alternatives.

Identifying metrics and methods for quantifying outcomes

Metrics operationalize objectives by describing the extent to which a decision alternative achieves an objective (Keeney & Gregory, 2005; McKay et al., 2012). Metrics can be quantified using different methods. For example, the accessibility of the river network to an organism, often called network connectivity (a metric associated with Objective 5a), can be calculated assuming that dams are impassable barriers. Alternatively, connectivity can be calculated assuming that a proportion of an aquatic organism's population (0-100%) can traverse the dam (McKay et al., 2017). In addition to assumptions about barrier navigability, connectivity can also be estimated using different indices (Jumani et al., 2020). To varying degrees, metrics abstract the processes which generate an outcome (i.e., the achievement of an objective) from an action (e.g., dam removal). If a metric substantially mischaracterizes this process, using it may lead to suboptimal decisions (Keeney & Gregory, 2005). For example, assigning higher scores to alternatives that yield greater river network connectivity assumes that dam removal will have a positive impact on the focal population by providing greater access to the river network. Connectivity metrics will fail to capture this objective if the organism is more constrained by factors other than connectivity, such as unsuitable water temperature or nutrient pollution (Reid et al., 2019).

We compiled metrics and methods associated with our 18 identified objectives from the literature and developed additional metrics and methods where we identified gaps (Supporting Information). We organized these linked objectives, metrics, and methods along with their required data sources in an interactive web application called the Dam Objectives & Metrics Selector Application (https://lnaslund.shinyapps.io/MCDA/) (Figure 4.2, more details in

Supporting Information). This web application can be used to populate different frameworks for evaluating tradeoffs among proposed alternatives, and the application automatically produces a list of decision support tools that include the user's selected objectives. We see the application not as a stand-alone tool, but as a complementary application to support critical thinking about the objectives of dam management and to guide users to resources that align with their objectives. For example, an analyst who wishes to employ optimization methods could use the tool to identify computationally efficient metrics to parameterize their selected objectives and constraints. An analyst who wishes to employ scoring and ranking methods could use the tool to identify metrics for parameterizing objectives that are difficult to quantify like minimizing interruption to community sense of place (Objective 4d).

Intended use cases of the application

The Dam Objectives & Metrics Selector Application is intended to support decisions with a diversity of decision alternatives including dam removal, partial removal, repair, retrofit, or no action. Our primary aim in developing this application is to promote critical thinking about the objectives of dam management decisions to reduce the likelihood of premature elimination of relevant objectives due to automation bias. The application, therefore, intentionally does not specify suitable decision alternatives. Additionally, the application is intended to support decisions about a single as well as multiple dams. We provide an example for a single dam decision and guidance for multiple dam decisions below. In a decision involving multiple dams, the decision process likely will differ depending on whether the dams are owned by a single or multiple entities. Single owner decisions, which we call "portfolio decisions," are most applicable to infrastructure utilities and government agencies. In a portfolio decision, the decision-maker may know a great deal about their dams and primary objectives. In this case, the

Dam Objectives and Metrics Selector Application may be useful in quantifying the co-benefits of a decision, for example to assess the effects of decision alternatives on other stakeholders or gauge the likely level of public support for alternatives. In an initial prioritization of removals in a dam portfolio, the analyst may wish to use readily-assessed indirect metrics to provide a first-order approximation. For example, for Objective 4a (maximizing swift water recreation), the decision analyst may choose to estimate stream length accessible to river recreators under different removal scenarios using connectivity methods to identify a few desirable removal scenarios. Once a few scenarios have been identified, the analyst may wish to refine their estimates by assessing the annualized monetary value of recreation under each scenario using stated or revealed preference methods with primary survey data (Loomis, 2002; Platt, 2003). The Dam Objectives and Metrics Selector Application provides multiple metrics for most objectives to accommodate both stages of decisions.

Prioritizations for the removal of dams with different owners are typically initiated by a non-owner entity whose mission is related to specific objectives, such as fish passage. This entity may serve in a facilitation role to identify potentially beneficial removals and bring together various stakeholders. We envision that the Dam Removal and Objective Selector Application will be particularly valuable in identifying priority objectives in such a decision context, because collaborative decision efforts can stall at the stage of defining objectives when stakeholders are unable to articulate objectives from their values or recognize the coherence among priorities stated in slightly different ways (Gregory et al., 2012). This application may streamline the objectives definition process by providing a common language for objectives and their categories that stakeholders can use to locate their priorities and ensure that important objectives are not excluded. As in portfolio decisions, the application can also be useful for identifying metrics appropriate to the decision stage. The identified objectives and metrics can then be used within existing dam removal decision-support frameworks (Jumani et al., 2023) or optimization tools (Kemp & O'Hanley, 2010; Kuby et al., 2005) to facilitate strategic dam removal or restoration planning.

Decision support in a single dam removal context (White Dam, Athens, Georgia, USA)

We illustrate the utility of the Dam Objectives and Metrics Selector Application for evaluating decisions about a single dam using a post-hoc evaluation of the White Dam in Athens, Georgia, USA, which was partially removed in 2018 (Figure 4.3). Our intention is not to reconstruct the decision process that led to the partial removal of White Dam, but to use this example to illustrate the utility of the application. The White Dam was a concrete gravity dam which spanned the Middle Oconee River. It was constructed in 1913 for hydropower and was in operation until the 1950s. In 1978, the dam was acquired by the University of Georgia as part of a land donation (The Georgia Aquatic Connectivity Team, 2020). In 2014, university staff proposed the removal of White Dam to reduce liability from hazards to recreational safety (Objective 3a) caused by large wood jams that occasionally formed at the structure (Figure 4.3) (The Georgia Aquatic Connectivity Team, 2020). The project team considered four management alternatives: 1) do nothing, 2) completely remove the dam, 3) leave the dam in place and construct a bypass channel, 4) remove the center section of the dam wall but leave the remaining structure in place (partial removal) (The Georgia Aquatic Connectivity Team, 2020).

The web application provides a platform to systematically consider the relevance of objectives to a removal decision (Table 4.2). When the relevance of an objective is unknown, the metrics and methods in the application may be used to identify ways to determine its relevance (Table 4.2). Using the application, we identified six relevant objectives and selected appropriate metrics and methods (Table 4.2). For the four objectives with unknown relevance, we identified metrics and methods from the application to evaluate their inclusion in the decision process (Table 4.2). As objectives are determined to be relevant by this evaluation, the process of considering tradeoffs among alternatives may be iterated. We note that none of the decision-support tools identified in the structured review included all six of the relevant objectives identified for the White Dam, illustrating the utility of the web application for minimizing the risk of premature elimination of objectives.

Outlook

Decision analyses often result in the selection of a suboptimal alternative early in the decision process due to failure to define all objectives or to select appropriate metrics and methods to parameterize those objectives (Hemming et al., 2022). The purpose of the Dam Removal Objectives and Metrics Selector Application is to facilitate critical thinking to ensure that appropriate management objectives, metrics, and methods are considered in the decision process. Using this application to identify objectives and select metrics/methods may reveal key uncertainties that require additional analysis. The application may also be useful to inform dam owners about the diverse objectives of dam removal, to identify consistent categories of benefits to compile and report the consequences of multiple projects, and to identify opportunities to align benefits and diverse project funding sources. For example, the U.S. Infrastructure Investment and Jobs Act allocated dam-related funding to over a dozen programs in nine different federal agencies. Approximately $900 M was allocated to improve existing dam safety, $800 M for dam removal, $800 M for hydropower dam safety and retrofit, and several hundred million to programs that can fund additional dam safety projects. The Dam Removal Objectives and Metrics Selector Application may be useful in organizing discussions of shared and distinct objectives across programs and agency mandates to use these funds efficiently and effectively.

More broadly, we propose this process—identifying common objectives, metrics, methods, and decision frameworks—as a model to facilitate structured decision-making for other types of river infrastructure decisions (e.g., decisions about levees and navigation dredging). Historically, many forms of infrastructure were constructed to meet a single, primary objective (National Academies of Science, Engineering, and Medicine, 2022). In a world facing the interacting crises of biodiversity loss, climate change, and infrastructure deterioration, we maintain that single objective decision-making leads to less efficient outcomes, including missed opportunities to align important objectives.

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Table 4.1. Common objectives of dam removal identified by a literature review.

Category	**Objective**
1. Account for monetary costs and feasibility	1a. Minimize implementation costs
	1b. Minimize maintenance costs
2. Meet demands for infrastructure services	2a. Meet water demands
	2b. Meet power generation demands
	2c. Meet navigation demands
	2d. Reduce flood risk
3. Reduce safety hazard	3a. Reduce personal safety risk
	3b. Mitigate risk of failure
4. Meet community desires for use of rivers for recreation, historic preservation, and sense of place	4a. Maximize swift water recreational opportunities
	4b. Maximize flat water recreational opportunities
	4c. Minimize adverse impacts to sites judged by the community, state, or Nation to be historically significant places
	4d. Minimize interruption to community sense of place
5. Maintain and restore the physical, chemical, and biological integrity of the Nation's waters	5a. Promote/maintain a population or community of focal taxa
	5b. Promote/maintain biodiversity
	5c. Prevent the spread of invasive species, disease, or undesirable hybridization
	5d. Reduce greenhouse gas emissions
	5e. Maintain or improve water quality
	5f. Reduce stream geomorphic degradation

Table 4.2. Systematic consideration of the relevance of the objectives in the Dam Objectives and Metrics Selector Application to the White Dam removal, Middle Oconee River, Athens, Georgia, USA. For relevant objectives, a metric/method is provided to evaluate alternatives according to their achievement of the objective. For objectives with unknown relevance, a metric/method is provided to help determine their relevance.

Objective	**Relevance**	**Justification**	**Metric: Method**
1a. Minimize implementation costs	Relevant	The dam owners had additional funding priorities.	Cost of removing the dam and appurtenant structures or constructing the by-pass channel: consult relevant experts
1b. Minimize maintenance costs	Relevant	Although the dam provided no infrastructure services to supplant upon removal, maintaining the dam required the owner to regularly clear the debris which accumulated behind the dam.	Dam maintenance cost: consult relevant experts
2a. Meet water demands	Not relevant	No water intakes were served by the dam.	
2b. Meet power generation demands	Not relevant	Power production potential was determined to be uneconomical in the 1980s.	
2c. Meet navigation demands	Not relevant	The river does not support commercial navigation and the dam did not support terrestrial navigation.	
2d. Reduce flood risk	Not relevant	The dam did not have the capacity to impound	

		substantial volumes of water.	
3a. Reduce personal safety risk	Relevant	The debris build-up behind the dam could entangle recreators.	Relative hazard potential: use dam characteristics and recreational use information
3b. Mitigate risk of failure	Not relevant	The dam did not have a hazard potential rating.	
4a. Maximize swift water recreational opportunities	Relevant	The dam impeded the passage of swift water recreators during some conditions.	Accessible stream length : use connectivity methods
4b. Maximize flat water recreational opportunities	Not relevant	The dam did not impound a sufficient volume of water to support flat water recreation.	
4c. Minimize adverse impacts to sites judged by the community, state, or Nation to be historically significant places	Relevant	The dam was over 50 years old, retained its original setting, location, and many of the original materials (including the original machinery in the powerhouse). These characteristics met criteria for listing in the National Register of Historic Places.	Register of historic places listing: determine eligibility for listing
4d. Minimize interruption to community sense of place	Unknown relevance	Although there was not a clear public access point to the dam, it was accessible to the university community and recreators traveling from other access points.	Community preference: survey community
5a. Promote/maintain a population or	Relevant	Consultation with other stakeholders revealed that the dam also posed a	Accessible stream length: use connectivity methods

community of focal taxa		barrier to movement of the Altamaha shiner (*Cyprinella xaenura*), which is listed as threatened by the Georgia Department of Natural Resources.	
5b. Promote/ maintain biodiversity	Unknown relevance	The extent to which loss of connectivity or other environmental conditions caused by the dam constrained local biodiversity is unknown.	Current watershed, riparian, instream physical and/or chemical condition: assess severity of other stressors
5c. Prevent the spread of invasive species, disease, or undesirable hybridization	Not relevant	Removal had negligible chance of increasing risks of invasive species, diseases, or undesirable hybridization	
5d. Reduce greenhouse gas emissions	Unknown relevance	The effect of the dam on greenhouse gas emissions was unknown.	Estimate current contribution of the dam to greenhouse gases emissions: measure CO_2 and CH_4 emissions along a transect upstream to downstream of the dam to determine if there are elevated emissions in the impounded area
5e. Maintain or improve water quality	Unknown relevance	The effect of the dam on water quality was unknown.	Current water quality: measure relevant parameters (e.g., temperature, dissolved oxygen) upstream to downstream of the dam to evaluate the impact of the dam on water quality

5f. Reduce stream geomorphic degradation	Not relevant	Negligible sediment storage and high background sediment transport rates indicated that dam removal had minimal potential effect on geomorphology

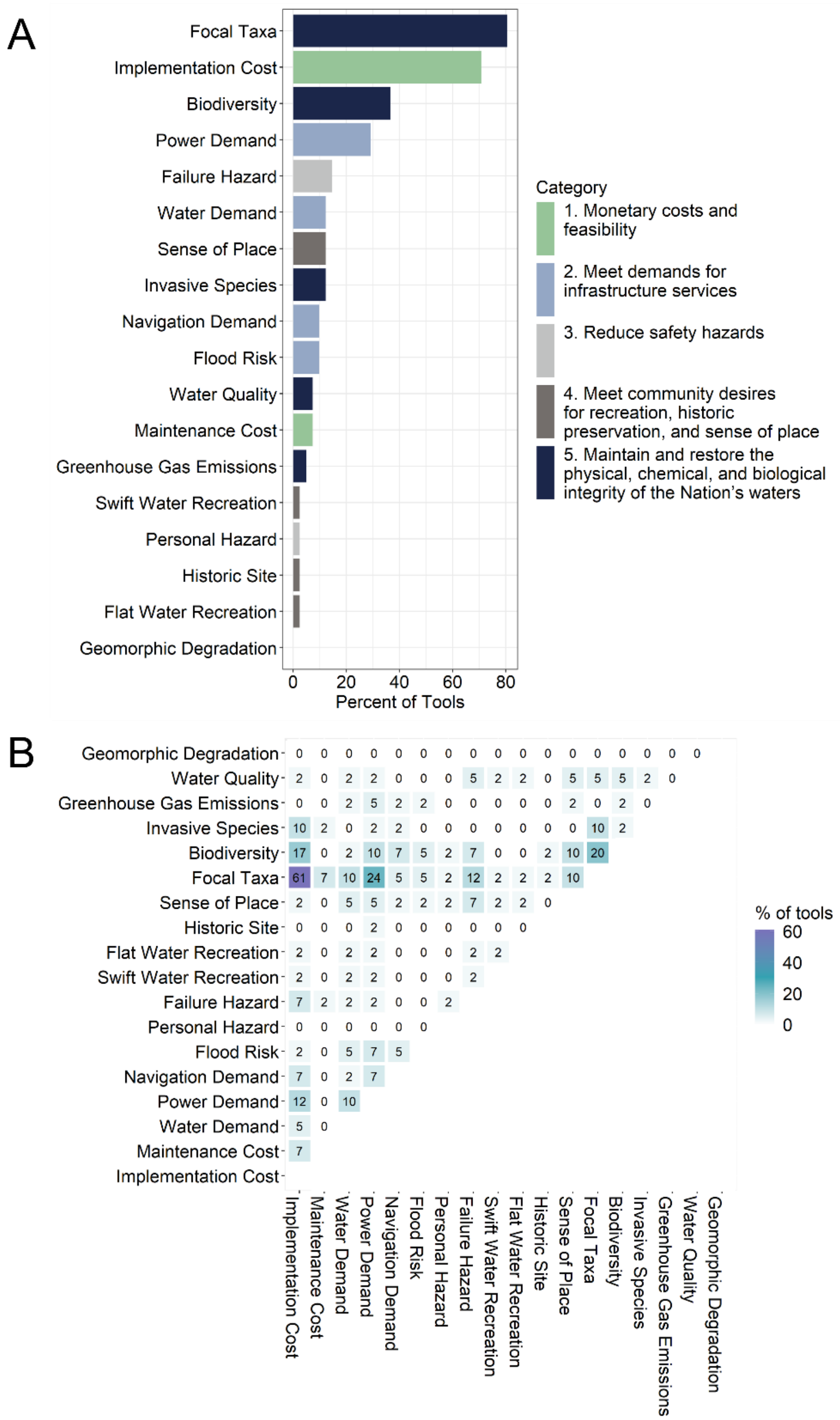

Figure 4.1. Frequency of A) occurrence and B) co-occurrence of identified objectives among existing dam removal decision-support tools (n = 41).

Figure 4.2. The Dam Objectives & Metrics Selector web application. A) The user selects objectives of dam removal from the objectives tab using the information provided in the grey

buttons organized by objective category. B) The metrics tab displays metrics and methods associated with the selected objectives and provides additional information, including citations and data sources in the information buttons. C) The tools tab displays the decision-support tools in the literature review (41 tools total) which include the user's selected objectives.

Figure 4.3. Photograph of White Dam, Athens, GA, USA. The star on the inset map represents the dam location.

CHAPTER 5

CONCLUSIONS

Millions of dams have been constructed globally for power, transportation, recreation, water supply, and flood control (Lehner et al. 2011). Thousands of dams have been removed to restore fisheries, decrease safety risks, and reduce costs (American Rivers 2024). Some regions of the world are now several decades past "peak dam" construction while some are continuing to build dams at a rapid pace (Zhang and Gu 2023). These statistics highlight the continued importance of supporting decision-making about dam construction, operational management, and ultimately removal. Currently, considering the consequences of dam construction, management, and removal for carbon is hindered by the lack of accessible methods for estimating the carbon impacts of these actions and weighing them against other concerns.

This dissertation addressed three issues to support the consideration of carbon impacts in dam management decisions. In Ch 2, I evaluated fine-scale spatiotemporal variability in carbon dioxide (CO_2) and methane (CH_4) emissions from small reservoirs to support accurate emissions estimates. In Ch 3, I developed and applied a framework for estimating the impact of dam removal on carbon emissions and storage from completed dam removals to facilitate their inclusion in removal decisions. In Ch 4, I constructed a web application to guide decision-makers to metrics, methods, data, and decision-support tools for dam removal objectives I identified in a literature review.

Chapter 2: Toward more accurate estimates of carbon emissions from small reservoirs

I sampled CO_2 and CH_4 emissions from four small reservoirs (< 0.01 km^2) in Athens, GA, USA at several locations in each reservoir and over the course of 24 hrs to identify the efficiency and accuracy of different sampling schemes. I found that common practices for sampling small reservoir carbon emissions can lead to substantial misestimation of total carbon dioxide equivalent (CO_2-eq) fluxes. Sampling only once during the day led to misestimations between -78% to 45%, sampling four or fewer locations led to misestimations between -85% to 366%, and excluding ebullition led to misestimations of -89% to 15% of total CO_2-eq flux. In the most extreme case, sampling one of the reservoirs only once at the modal time of prior greenhouse gas sampling efforts would lead to the conclusion that it was a large CO_2 sink when it was in fact a CO_2 source.

Better constraining CH_4 emissions from small waterbodies, including small reservoirs, has been specifically highlighted as a priority for estimating methane emissions from inland waters (Lauerwald et al. 2023), the most uncertain part of the global methane budget (Canadell et al. 2021). These results indicate that fine-scale spatiotemporal variability can impact our inferences about small reservoir emissions. Accurate emissions estimates are required to develop effective strategies to reduce small reservoir emissions. For example, in this study, I found that the floating aquatic macrophyte *Wolffia sp.* (duckweed) appeared to increase the magnitude and alter the spatiotemporal patterns of emissions. These findings point to duckweed removal as a potential strategy to reduce reservoir carbon emissions.

Chapter 3: Consequences of dam removal for carbon storage and emissions

To comprehensively quantify the effects of dam removal on carbon emissions and storage, I identified nine major carbon fluxes impacted by dam removal, and modeled these fluxes for the removal of two large dams. Before removal, emissions from the reservoir surface and burial of carbon in reservoir sediments contribute to carbon balance in the reservoir footprint. When the reservoir is dewatered for dam removal, the exposure of previously buried carbon in the sediments to air facilitates its mineralization and enhances CH_4 and CO_2 emissions in a "burp" of carbon. Erosion of previously buried carbon in reservoir sediments transported out of the reservoir to higher oxygen environments also facilitates CO_2-eq emissions, as does enhanced bubbling of CH_4 from the sediments with water level drawdown. After dam removal, as terrestrial vegetation and soils begin to develop in the reservoir footprint, terrestrial net ecosystem production, tree- and soil-derived CH_4 flux contribute to carbon balance along with emissions from the now free-flowing river. I modeled each of these pathways of carbon emissions and storage for the removal of the Glines Canyon Dam, Elwha River, Washington, USA and Veazie Dam, Penobscot River, Maine, USA to illustrate the potential carbon consequences of dam removal.

I found that prior to removal, the Glines Canyon reservoir was a large net sink for CO_2-eq due to its high rate of sediment carbon burial, and the Veazie reservoir was a small net source for CO_2-eq. Removal of the Glines Canyon Dam facilitated the "burp" of several orders of magnitude more CO_2-eq than the removal of the Veazie Dam. After dam removal, the Veazie reservoir footprint was a larger source of CO_2-eq, and the Glines Canyon reservoir footprint was a weaker sink. The implications of these results for the costs and benefits of dam removals depend on how much of the CO_2-eq emissions enhanced by dam removal can be attributed to the removal as well as the analyzed counterfactual (i.e., whether a comparison is made to a scenario without dam removal or without the dam existing). For some pathways, changes in fluxes due to dam removal can be better considered fluxes displaced in time (i.e., they would have occurred regardless of dam removal but during a different time period), and some pathways are better considered displaced in space (i.e., they would have occurred regardless of dam removal but in a different location). Better constraining the portion of the fluxes that can be attributed to the dam and its removal is a critical research need to facilitate the incorporation of carbon consequences into dam removal decisions.

The focal dam removals I evaluated indicated that dams can vary with respect to their impact and the impact of their removal on carbon. The amount of accumulated sediment behind the dam and its fate upon removal appear to be key determinants of that impact. Constraining the considerable uncertainty in sediment erosion from reservoirs after dam removal presents a substantial challenge to estimating the impact of dam removal on carbon balance; however, the likely importance of sediment fate indicates that factors which have been used to qualitatively predict the volume of eroded sediment following dam removal, like reservoir aspect ratio and accumulated sediment volume to annual river sediment load ratio (Major et al. 2017), may also be useful in anticipating the carbon impacts of dam removals. Additional factors like size and reservoir trophic state, which were similar in the focal dam removals, may also determine the impact of removal on carbon balance, leaving open the possibility that some dam removals may yield net carbon benefits rather than costs.

Chapter 4: Facilitating dam removal decisions with multiple objectives

To determine whether existing dam removal decision support tools potentially constrain decisions by excluding common dam removal objectives, I conducted a review of dam removal literature and decision support tools. In this review, I identified 18 common objectives of dam removal decisions which could be divided into five categories: 1) account for monetary costs and feasibility, 2) meet demands for infrastructure services, 3) reduce safety hazard, 4) meet community desires for use of rivers for recreation, historic preservation, and sense of place, and

5) maintain and restore the physical, chemical, and biological integrity of the Nation's waters. In a systematic review of 41 existing dam removal decision support tools, I found that some of these 18 objectives were included in nearly every tool, while some were rarely included. The majority of tools supported decisions to promote or maintain a population or community of a focal taxa (frequently fish), but common objectives like reducing safety hazard and expanding recreational opportunities were infrequently included. The tool with the most objectives included only 9 of 18, and most tools included fewer than three objectives.

To support the consideration of the 18 objectives of dam removal, I developed a web application linking objectives to metrics, methods, data, and decision support tools to evaluate and deliberate among management alternatives. In a post-hoc evaluation of a completed dam removal in Athens, GA, USA using the application, I identified six relevant objectives. These six objectives were not included in a singular prior dam removal decision support tool, illustrating the utility of the web application for facilitating the consideration of diverse objectives in dam removal decisions.

Synthesis

A unifying theme of this dissertation is evaluating and contending with the consequences of shortcuts in decision-making. In Ch 2, I estimated the consequences of taking a shortcut in sampling small reservoir emissions by limiting sampling times to when and where it is convenient. In Ch 3, I evaluated the logical shortcut that dam removal will reduce landscape carbon emissions through changes in water surface emissions. In Ch 4, I assessed whether the tools for structuring dam removal decisions embed consequential shortcuts by excluding common objectives of dam removals. Shortcuts are not inherently harmful. Cognitive science research indicates that shortcuts can be critical for managing the mental load of the varied, complex challenges which humans face. However, some shortcuts can lead to biased decisions with suboptimal outcomes (Gregory et al. 2012).

Shortcuts typically cause decision-makers to focus on the most apparent or accessible features of a problem (Gregory et al. 2012). For example, for investigators accustomed to conducting field work during typical waking hours, variability in emissions among small reservoirs may be more apparent than diel emissions variability in a single reservoir. My modeling (Ch 2) illustrates that reasonable accuracy of small reservoir emissions estimates can be achieved with strategic sampling during waking hours, but without first evaluating the spatiotemporal patterns of emissions to identify those strategies, an investigator may be led to erroneous conclusions (e.g., that a reservoir which is in fact a large net emitter of CO_2 is a sink). By making spatiotemporal emissions variability more apparent, my findings support accurate estimation of small reservoir carbon emissions.

Changes in emissions from the water surface after dam removal may be more apparent to investigators studying reservoir emissions than other pathways of emissions and storage facilitated by dam removal. This perspective could lead to the hypothesis that dam removal reduces carbon emissions. The framework I presented (Ch 3) indicates that pathways of carbon emissions and storage other than water surface emissions can be critical determinants of the net impact of dam removal. The examples I presented illustrate that removal may not reduce carbon emissions in the reservoir footprint. This work may facilitate better hypotheses about the carbon consequences of dam removals by making key pathways of carbon emissions and storage impacted by removal more apparent.

Decision-makers are often driven by seemingly strong impressions about what matters—the objectives of a decision—but these impressions are malleable given different contexts

(Gregory et al. 2012). Decision support tools can alter the decision context by making included objectives more apparent than excluded ones. The dam removal objectives and metrics web application I developed (Ch 3) aims to provide decision-makers an opportunity to confront their biases about decision objectives and find metrics, data, and additional decision support tools which can challenge and provide alternatives to pre-conceived (and perhaps limited) decision alternatives.

Conclusion

The world currently faces the interacting crises of biodiversity loss, climate change, and deteriorating infrastructure (van Rees et al. 2023). Dams sit at the intersection of these crises as agents of biodiversity loss (Pringle et al. 2000), tools for climate mitigation and adaptation which enhance greenhouse gas emissions from inland waters (Deemer et al. 2016; Almeida et al. 2019), and aging infrastructure which is often no longer functioning as intended (Hansen et al. 2020). In decisions about dam construction, management, and removal, we have the capacity to align seemingly disparate priorities. Realizing this alignment will require interrogating our shortcuts to better assess and weigh the benefits and costs of decisions about dams.

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APPENDIX A

SUPPLEMENTARY METHODS AND DATA: CHAPTER 2

Calculation of the modal sampling time for NEON dissolved gas sampling efforts

We calculated the modal time of dissolved gas sampling from lakes in the National Ecological Observatory Network (NEON) to compare the diel flux patterns we observed to sampling times for efforts to characterize greenhouse gases in inland waters. We used the neonUtilities package in R to download dissolved gas data from six core lake sites (CRAM, BARC, SUGG, LIRO, PRLA, and TOOK) sampled from January 2016 to April 2023 (NEON 2023). We extracted sampling time from each observation and calculated the modal time which was 10:40.

Table A1. Dissolved gas concentrations collected from 0.25m below the surface at the top of the water column and 0.1m from the sediment. Dissolved gas samples from the Catfish sampling on 18-19 Sept could not be analyzed due to handling errors.

Site	**Collection Date and Time**	**Location**	**CH_4 (µmol L^{-1})**	**CO_2 (µmol L^{-1})**
Blue Herron	2022/09/14 10:20	Top	98.7 ± 3.09	560 ± 8.69
Deans	2022/08/31 10:00	Top	0.979 ± 1.32	62.3 ± 7.88
Sister	2022/08/23 10:45	Top	0.849 ± 0.011	40.3 ± 4.55
Catfish	2022/09/07 09:25	Top	0.586 ± 0.002	138 ± 5.69
Deans	2022/08/17 08:10	Top	0.583 ± 0.048	107 ± 3.11
Blue Herron	2022/09/14 10:20	Bottom	126 ± 60.7	634 ± 161
Sister	2022/08/23 10:45	Bottom	2.89 ± 0.985	70.0 ±15.2
Deans	2022/08/17 08:10	Bottom	0.883 ± 0.054	88.9 ± 16.4
Catfish	2022/09/07 09:25	Bottom	0.557 ± 0.069	164 ± 49.1

Deans	2022/08/31 10:00	Bottom	0.290 ± 0.060	36.6 ± 6.79
Blue Herron	2022/09/14 11:25	Inlet	9.98 ± 1.34	167 ± 9.29
Deans	2022/08/31 10:30	Inlet	16.9 ± 4.77	377 ± 99.1
Sister	2022/08/23 11:15	Inlet	0.0229 ± 0.00006	75.2 ± 7.54
Catfish	2022/09/07 9:55	Inlet	0.598 ± 0.197	141 ± 6.10
Deans	2022/08/17 10:45	Inlet	9.95 ± 2.86	234 ± 46.0
Blue Herron	2022/09/14 11:40	Outlet	0.365 ± 0.170	160 ± 25.6
Deans	2022/08/31 10:45	Outlet	4.31 ± 0.940	153 ± 44.1
Sister	2022/08/23 11:25	Outlet	2.62 ± 0.072	81.8 ± 8.42
Catfish	2022/09/07 10:15	Outlet	1.47 ± 0.410	142 ± 9.15
Deans	2022/08/17 11:10	Outlet	0.311 ± 0.106	50.2 ± 5.51

Table A2. Top models explaining diel patterns of CO_2 and CH_4 diffusion from environmental variables. All models included a random intercept for sampling (1 | sampling), which was the site and date sampled. Model selection was performed using scaled predictors; however, the parameter estimates for the top models reported here are unscaled.

Model	**β**	**Marginal R^2**	**ΔAICc**
CO_2 diffusion ~ intercept + surface DO + (1 | sampling)	intercept: 0.23 surface DO: -0.028	0.14	0
CO_2 diffusion ~ intercept + (1 | sampling)	intercept: 0.095	N/A	3.16
CO_2 diffusion ~ intercept + temperature + (1 | sampling)	intercept: 0.71 temperature: -0.024	0.12	5.00
CO_2 diffusion ~ intercept + surface DO + temperature + (1 | sampling)	intercept: 0.35 surface DO: -0.027 temperature: -0.005	0.15	7.54
CO_2 diffusion ~ intercept + bottom DO + surface DO + (1 | sampling)	intercept: 0.23 bottom DO: 0.006 top DO: -0.032	0.14	7.99
CH_4 diffusion ~ intercept + (1 | sampling)	intercept: 0.002	N/A	0
CH_4 diffusion ~ intercept + surface DO + (1 | sampling)	intercept: 0.008 surface DO: -0.001	0.17	8.54
CH_4 diffusion ~ intercept + temperature + (1 | sampling)	intercept: 0.02 temperature: -0.0007	0.068	11.03

Table A3. Interpolated emissions rates and simulated 95% confidence intervals.

Site	**Date**	**CO_2 diffusion ($g\ CO_2\ m^{-2}\ d^{-1}$)**	**CH_4 diffusion ($g\ CH_4\ m^{-2}\ d^{-1}$)**	**CH_4 ebullition ($g\ CH_4\ m^{-2}\ d^{-}$)**	**CO_2-eq emissions ($g\ CO_2$-eq $m^{-2}\ d^{-1}$)**
Catfish	06 Sept 2022	3.1 [2.7, 3.6]	0.012 [0.0075, 0.016]	0.12 [0.068, 0.20]	6.8 [4.8, 9.6]
Catfish	18 Sept 2022	2.4 [2.1, 2.8]	0.011 [0.0068, 0.017]	0.038 [0.021, 0.066]	3.8 [2.8, 5.1]
Deans	16 Aug 2022	3.2 [2.8, 3.8]	0.012 [0.0090, 0.015]	0.023 [0.0065, 0.048]	4.2 [3.2, 5.5]
Deans	30 Aug 2022	1.2 [1.0, 1.4]	0.0064 [0.0052, 0.0077]	0.038 [0.019, 0.062]	2.4 [1.7, 3.2]
Blue Herron	13 Sept 2022	3.6 [2.9, 4.3]	0.29 [0.25, 0.35]	0.23 [0.15, 0.32]	18 [14, 23]
Sister	22 Aug 2022	-0.074 [-0.13, -0.022]	0.011 [0.0098, 0.013]	0.069 [0.050, 0.092]	2.1 [1.5, 2.8]

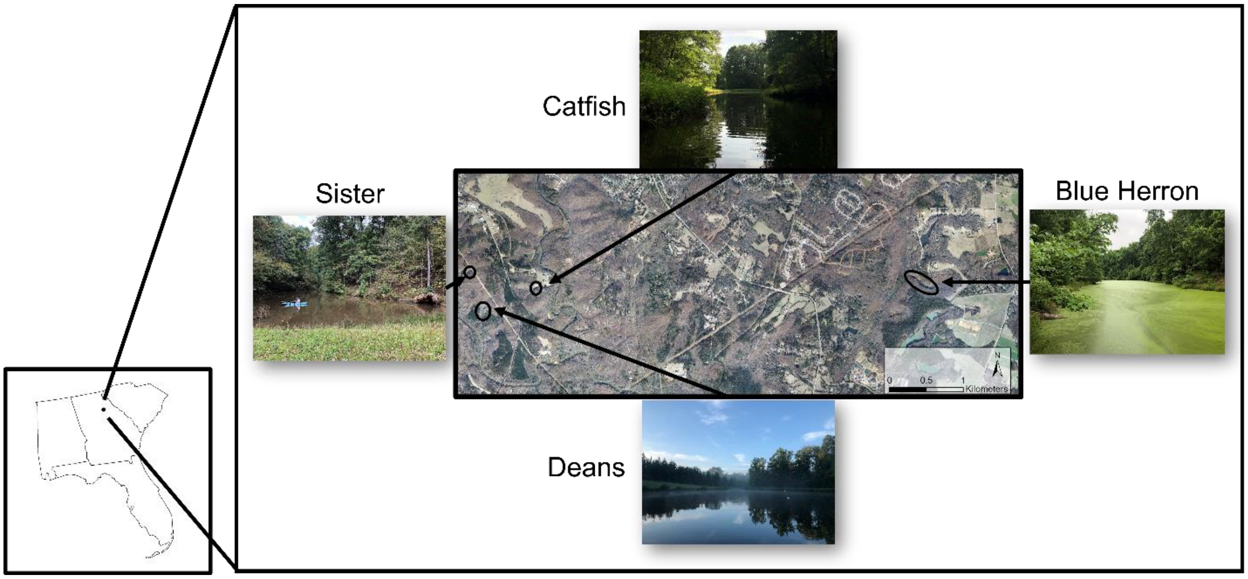

Figure A1. Map of sampled reservoirs in Athens. Site base map from high resolution orthoimagery captured 27 Jan 2013 and published by the NOAA's Ocean Service, Coastal Services Center.

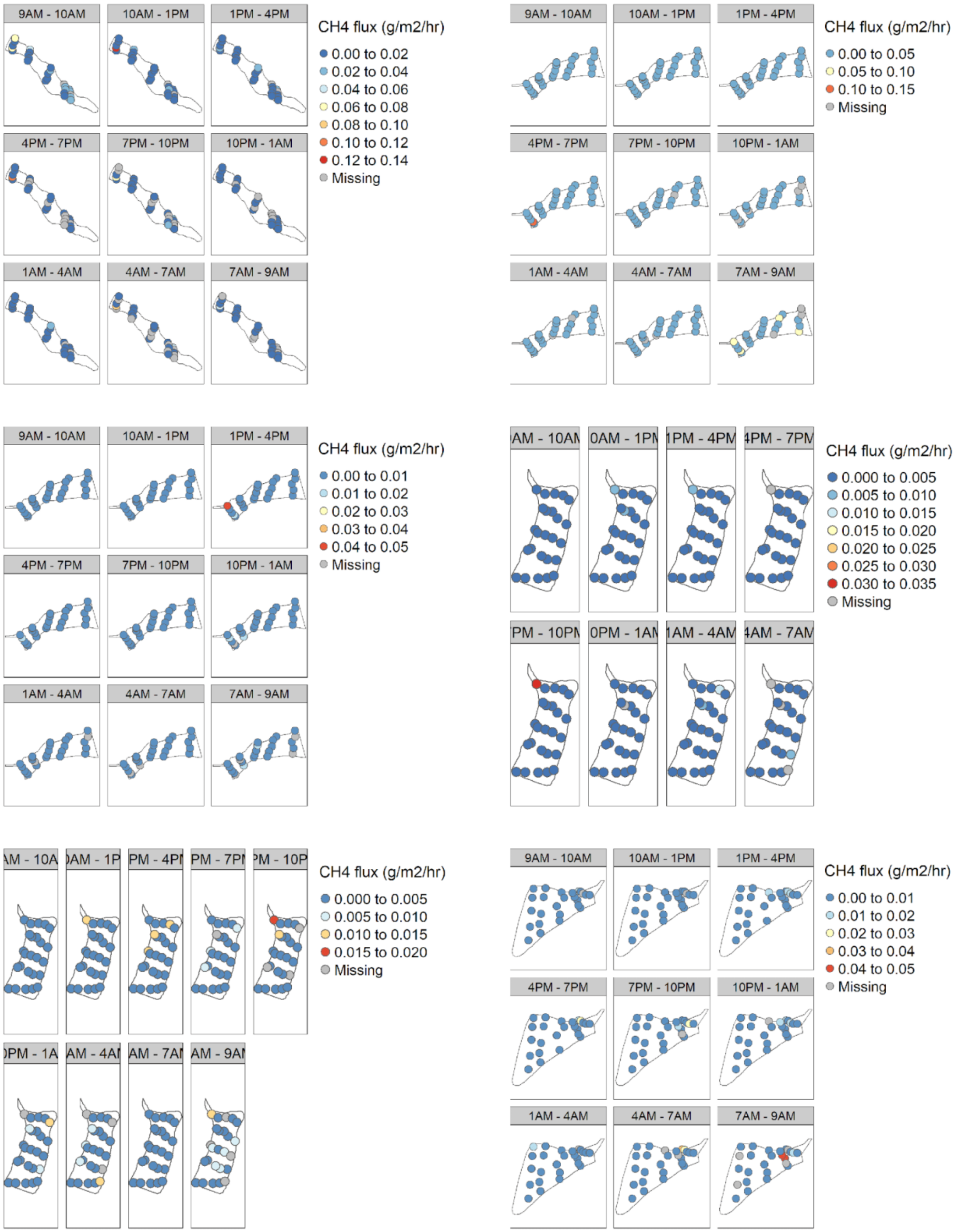

Figure A2a. Spatial patterns of measured CH_4 ebullitive fluxes.

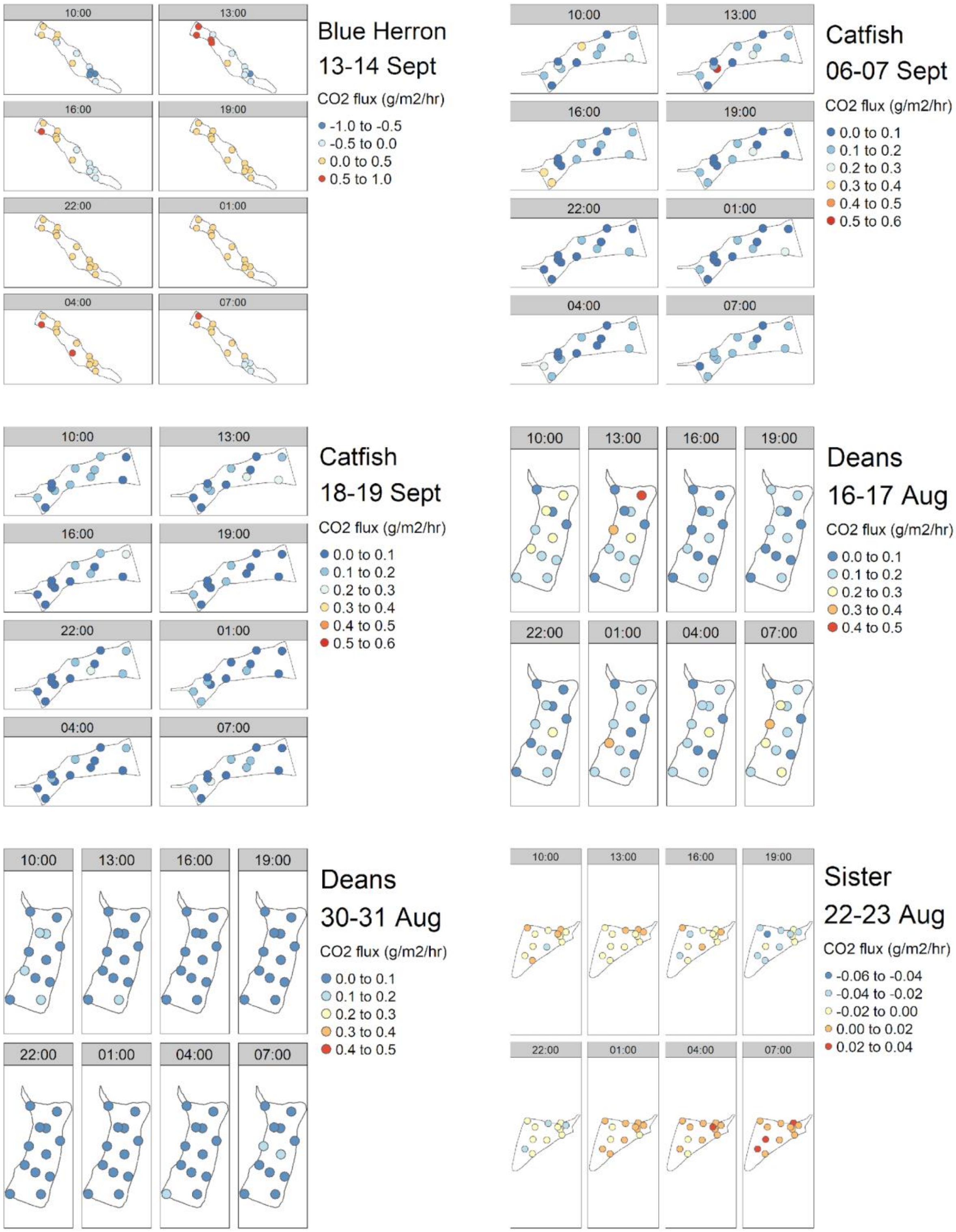

Figure A2b. Spatial patterns of measured CO_2 diffusive fluxes.

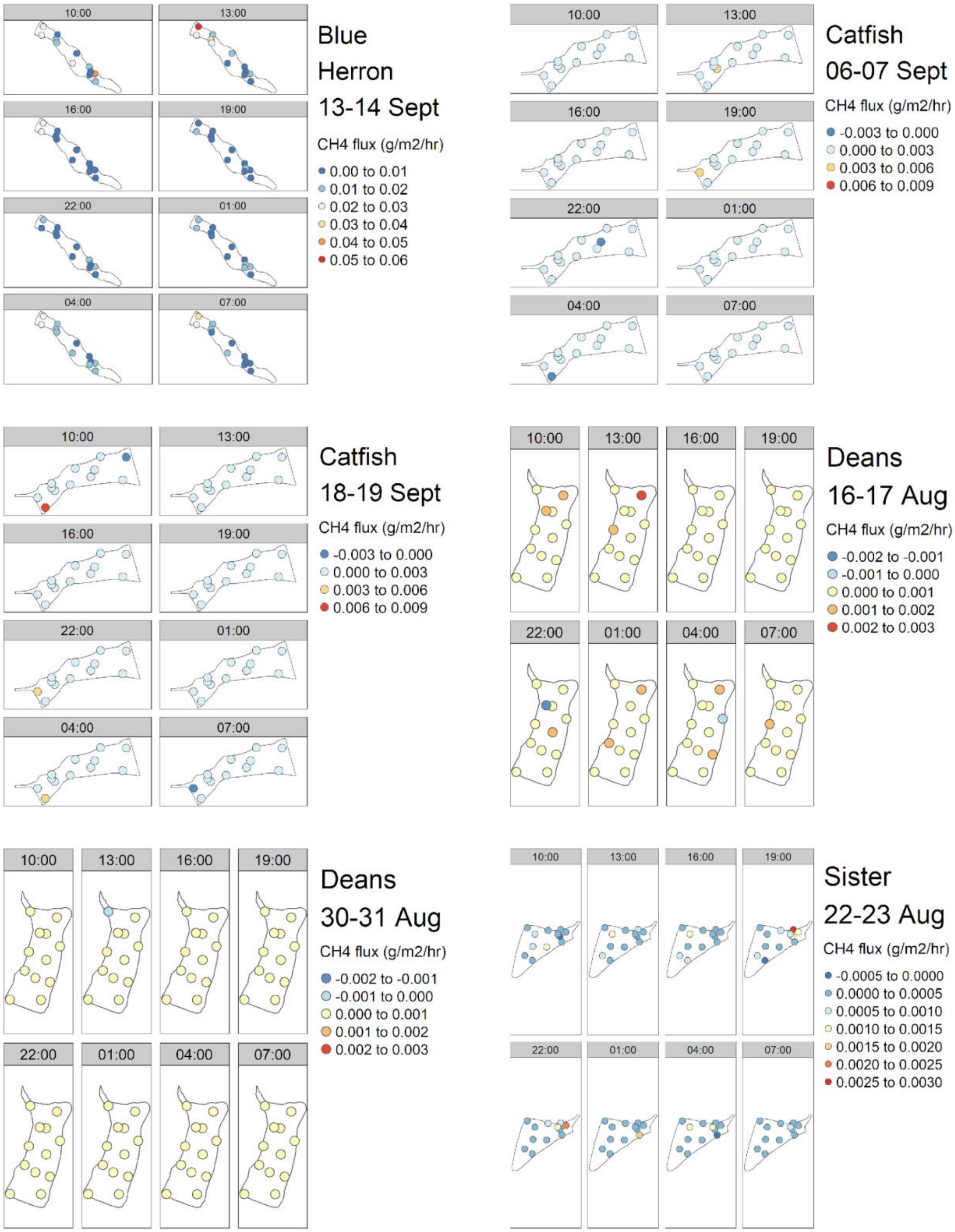

Figure A2c. Spatial patterns of measured CH_4 diffusive fluxes.

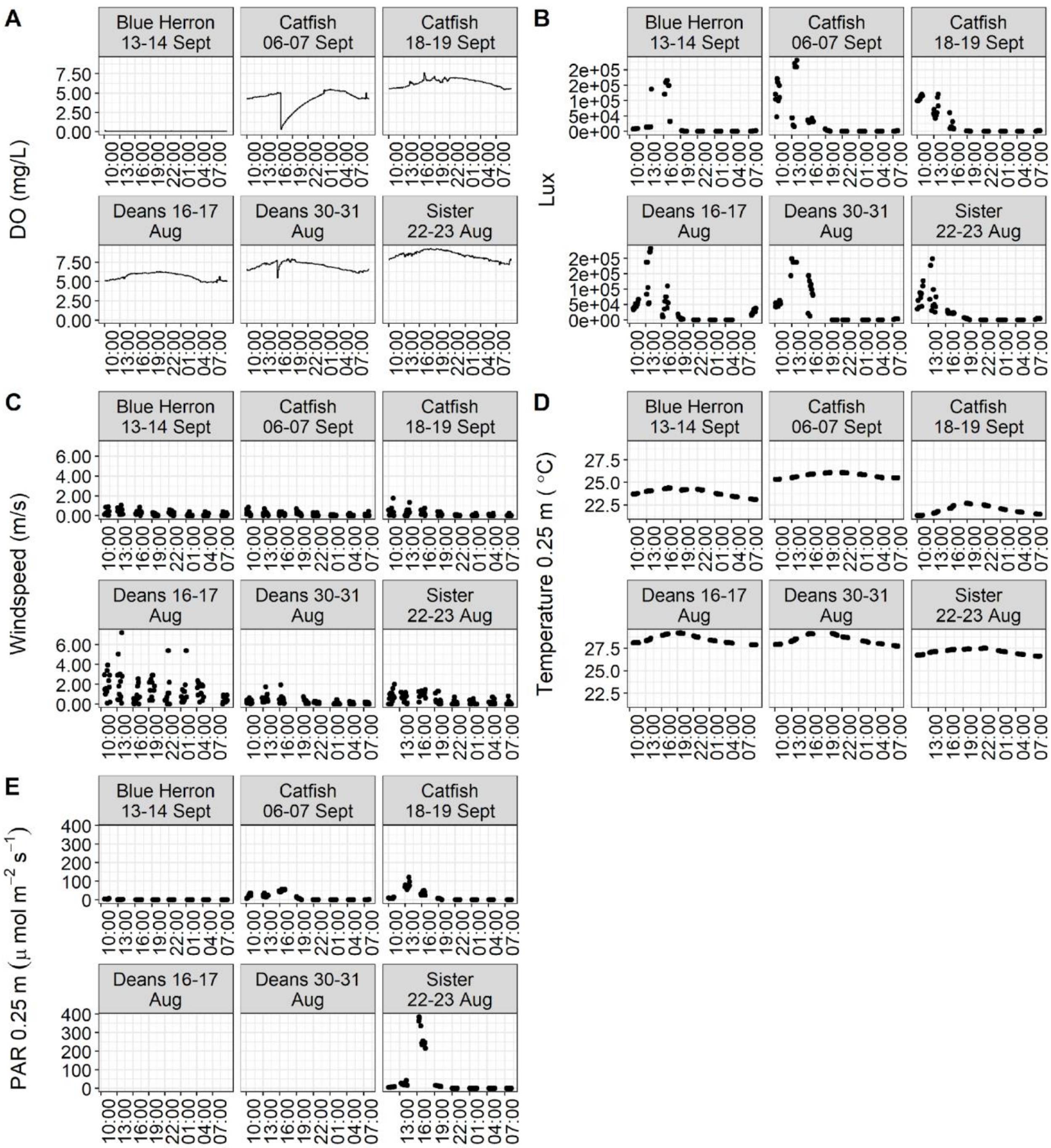

Figure A3. A) Dissolved oxygen concentration from 0.25 m depth, B) Lux, C) windspeed, D) temperature at 0.25 m depth, and E) PAR during each sampling event.

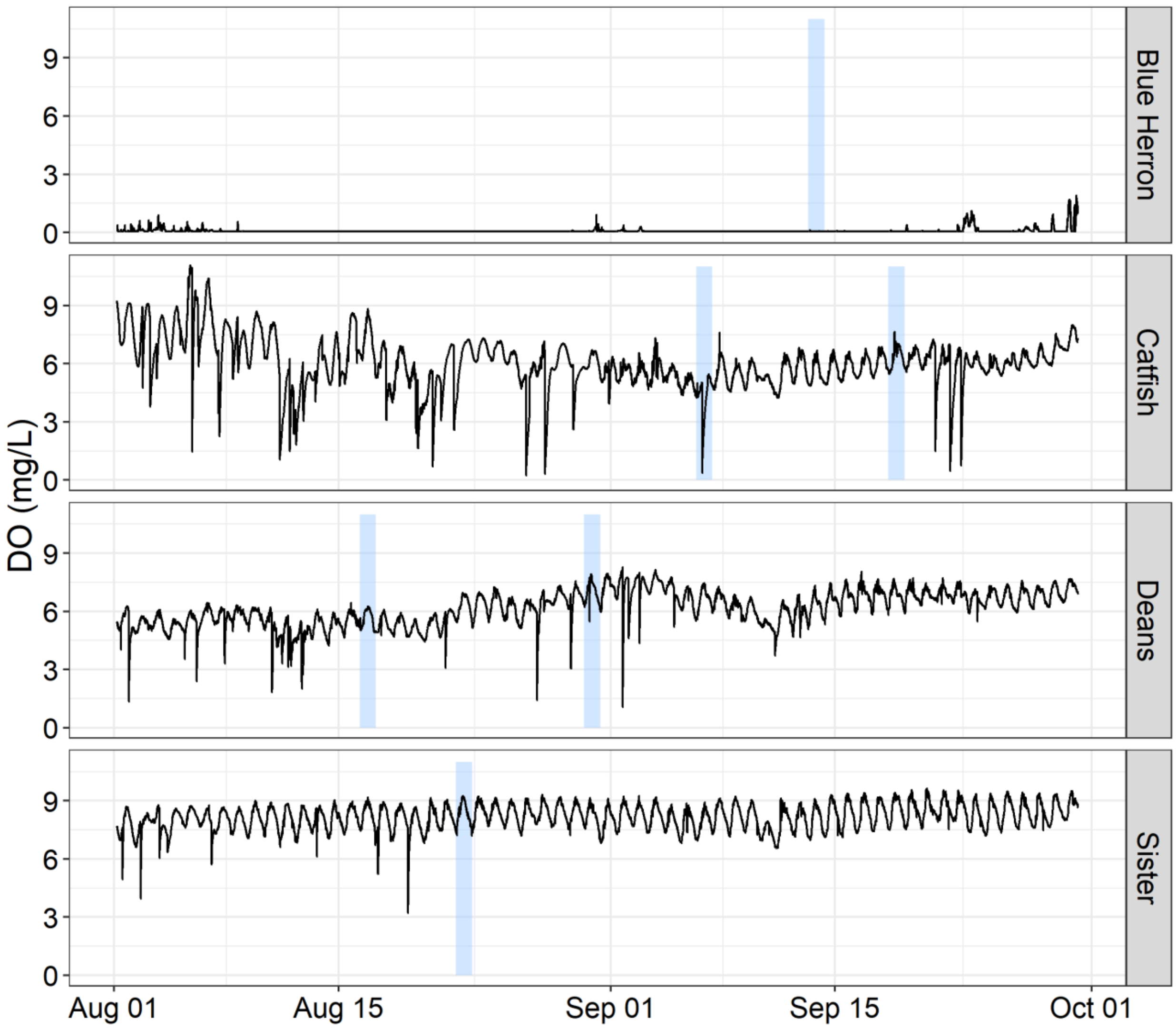

Figure A4. Time series of dissolved oxygen from 0.25 m depth in each sampled reservoir from August 1st to September 30th, 2022. Highlighted portions of the time series represent sampling windows.

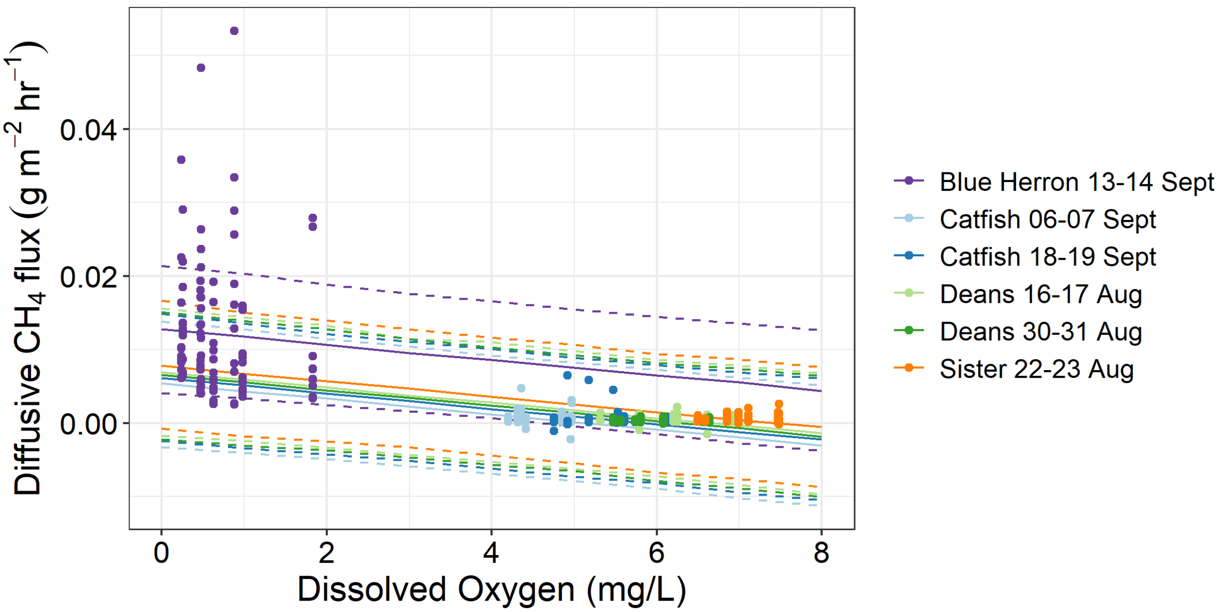

Figure A5. The relationship between dissolved oxygen recorded 0.25 m below the water surface and the diffusive flux of CH_4.

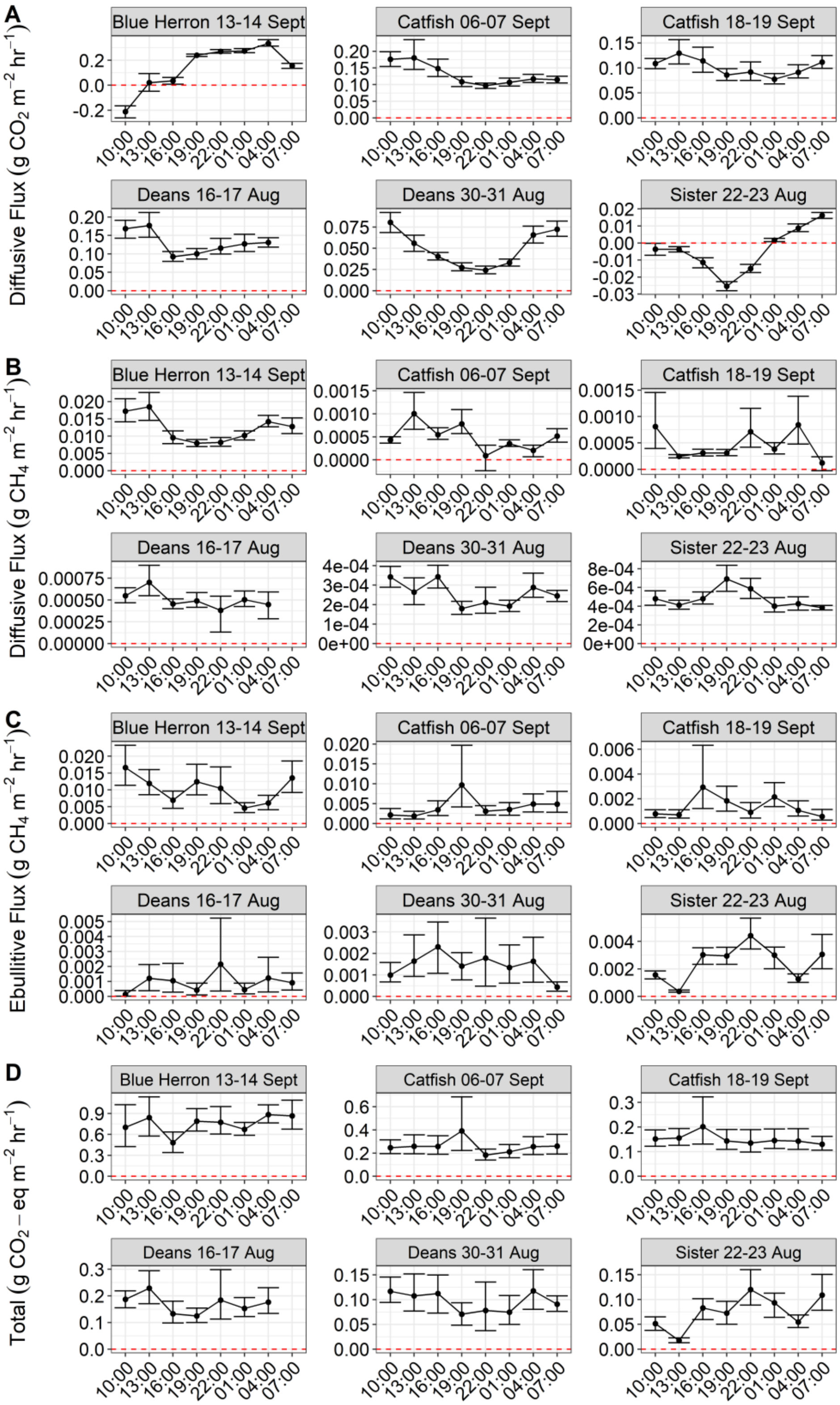

Figure A6. Diel patterns in interpolated A) CO_2 diffusion, B) CH_4 diffusion, C) CH_4 ebullition, and D) total CO_2-eq emissions. Note the different y axis ranges. Points represent average estimates and error bars represent the simulated 95% confidence interval. The times depicted for CH_4 ebullition are the end of the ebullition measurement period (i.e., when the gas volume was recorded). The 07:00 diffusive flux sampling for Deans on 16 Aug was delayed until 09:00 due to rain. These values were used to calculate total CO_2-eq flux over a 24-h period, but are not depicted here.

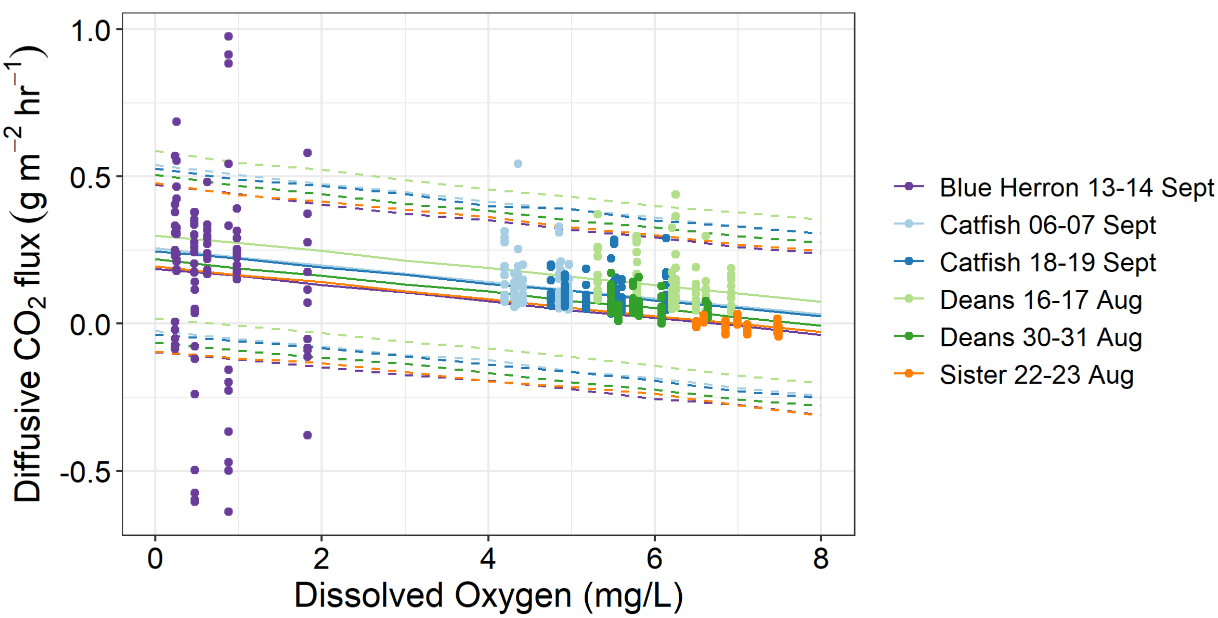

Figure A7. The relationship between dissolved oxygen recorded 0.25 m below the water surface and the diffusive flux of CO_2.

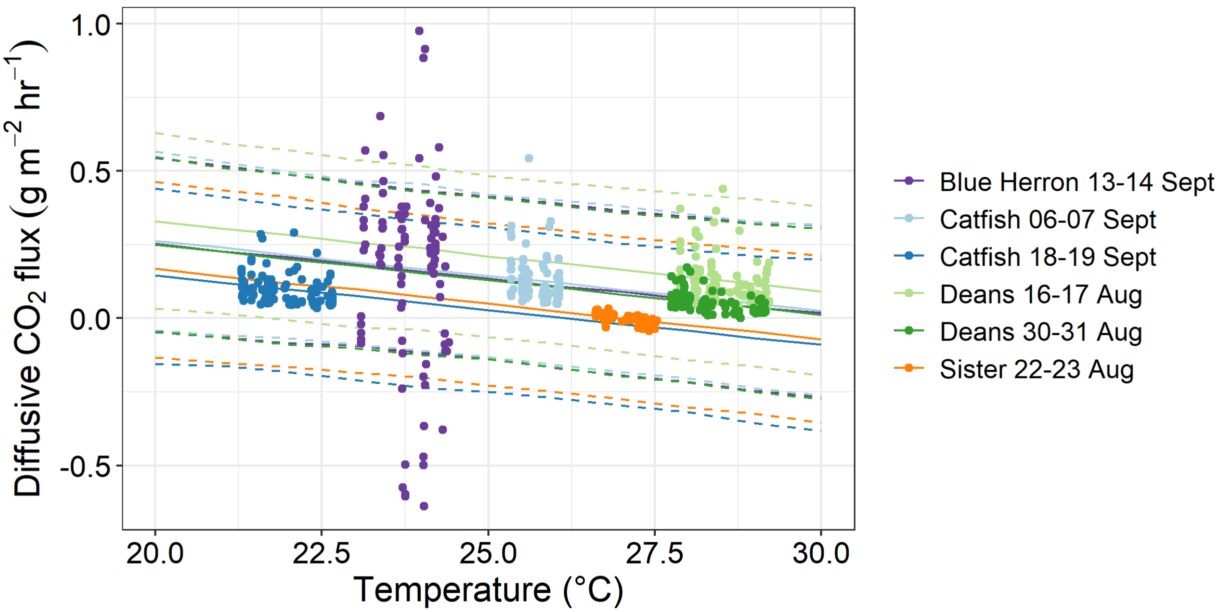

Figure A8. The relationship between temperature recorded 0.25 m below the water surface and the diffusive flux of CO_2.

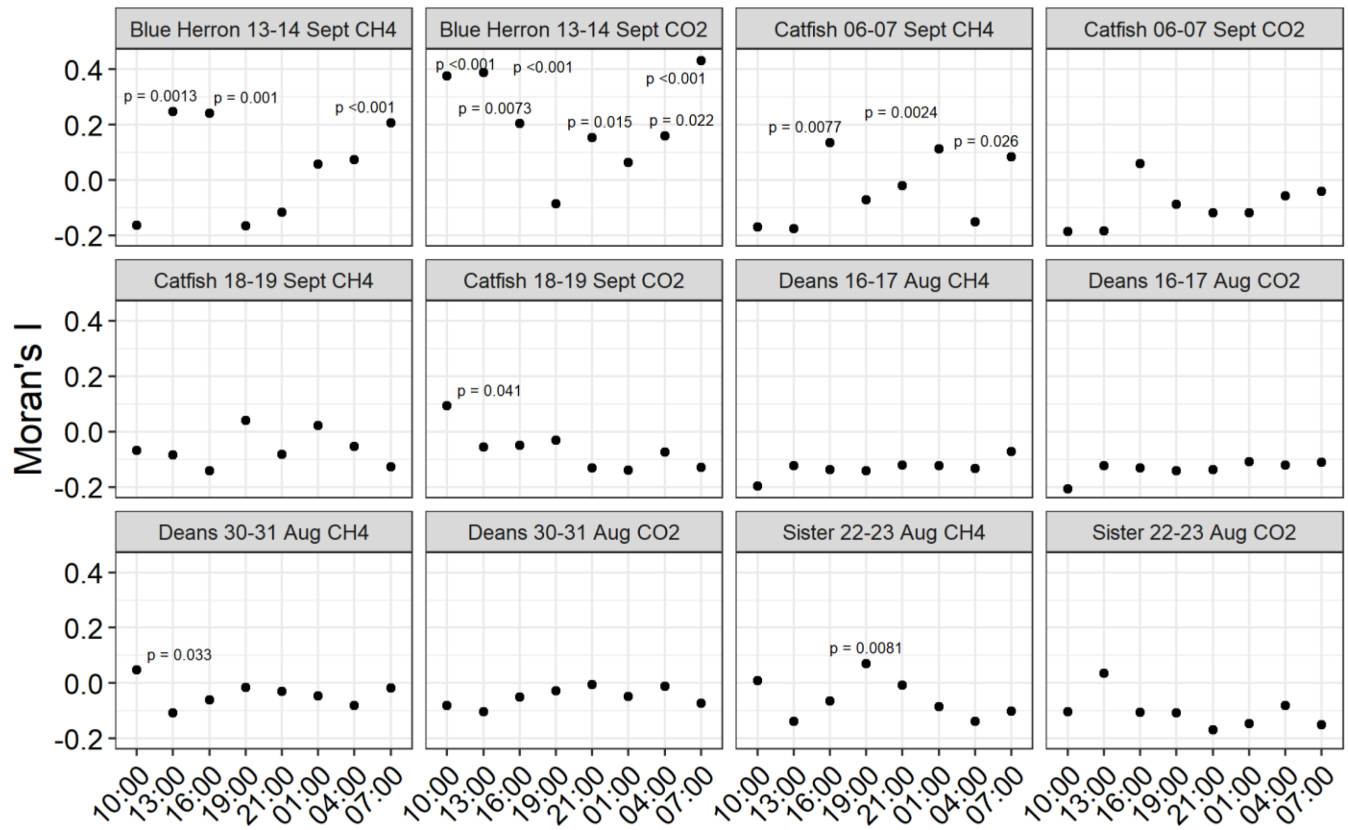

Figure A9. Moran's I calculated for every sampling period at every site for CO_2 and CH_4. *P* values < 0.05 reported above the corresponding points.

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APPENDIX B

SUPPLEMENTARY METHODS AND DATA: CHAPTER 3

Model caveats

General: Our models of emissions in the reservoir footprint after removal assume stationarity in environmental characteristics and responses over the next 100 years. We do not model, for example, how future changes in precipitation may alter NEP in the reservoir footprint.

Reservoir emissions: Because we were unable to recreate prediction intervals from the G-res model using the summary statistics reported in the tool's user interface, our estimates of reservoir emissions uncertainty may be relatively lower than our other flux estimates. We also did not estimate emissions from sediment exposed prior to dam removal due to e.g., drought, dam maintenance or operations.

Burial: The sedimentation model we used to estimate carbon burial in the Veazie reservoir does not account for dams upstream. Thus, our calculated burial flux for the Veazie Dam is likely an overestimate. Our approach also does not consider coarse woody debris storage in the reservoir before removal or after removal, which can constitute a large portion of the reservoir carbon storage (Stratton et al. 2019).

Exposed sediment emissions: In our model, sediment CO_2 and CH_4 emissions are only allowed to vary among reservoirs based on differences in exposed sediment area. Future work on sediment CO_2 and CH_4 emissions may support better constraining initial rates of sediment emissions using reservoir and environmental characteristics.

Terrestrial NEP: River floodplains tend to be productive relative to upland forests, so estimates of terrestrial NEP here may be biased low if most of the FLUXNET tower sites are not located in river floodplains. However, we also do not have sufficient information to determine the rate of forest establishment on former reservoir sediments versus in a clearcut or burned plot as in the FLUXNET plots from which we derived our estimates of terrestrial NEP.

Soil CH_4 flux: In contrast to the NEP model, the model of soil CH_4 flux did not account for changes in fluxes over time as the soils develop. We were also unable to recreate the summary statistics reported in Gatica et al. (2020) using data in the accompanying archive and our interpretation of their statistical methods. Because soil CH_4 flux was a minor component of CO_2-eq balance in the reservoir after removal, we considered this a minor issue.

River CH_4 emissions: We did not include river CH_4 ebullition in our estimates of CH_4 emissions from rivers because there were too few ebullition estimates in the size classes of our focal rivers.

Model results in g C

The Veazie (4.22 Mg C yr^{-1} (3.70, 4.74)) had lower emissions in g of C than the Glines (with degassing: 16.3 Mg C yr^{-1} (14.3, 18.3); without degassing: Mg C yr^{-1} 16.0 (14.1, 18.0)) before dam removal. The Glines buried much more C (-3180 Mg C yr^{-1} (-9210, -719)) than the Veazie prior to removal (-22.1 Mg C yr^{-1} (-118 -0.553)). The net C flux from the Veazie spanned values of source and sink, with the average estimate suggesting the Veazie was a small net C sink (-17.9 Mg C yr^{-1} (-115, 4.19)). The Glines, in contrast, was a large net C sink before removal (-3160 Mg C yr^{-1} (-9200, -700)).

The Veazie Dam removal “burped” 1.04 Mg C (0.536, 1.53) in the least emissions scenario, 64.0 Mg C (26.7, 101) in the moderate emissions scenario, and 397 (157, 633) in the most emission scenario. In every scenario for the Veazie, exposed sediment CO_2 emissions contributed the most C emissions. The Glines Canyon Dam removal "burped" 50900 Mg C (2000, 230000) in the least emissions scenario, 51500 Mg C (2260, 230000) in the moderate emissions scenario, and 54300 Mg C (3530, 235000) in the most emissions scenario. In every scenario for the Glines Canyon, eroded sediment emissions contributed the most C emissions.

After removal, the Veazie became a larger net source of C emissions (1350 Mg C yr^{-1} (-2.06, 6300)) and the Glines became a smaller net sink of C emissions (-123 Mg C yr^{-1} (-855, 882)). River CO_2 emissions was the dominant contributor to carbon balance in the Veazie reservoir footprint after dam removal (1380 Mg C yr^{-1} (90.9, 6230)) and terrestrial NEP contributed little given the small area of exposed sediment (-30.2 Mg C y^{-1} (-94.2, 33.9)). Terrestrial NEP in the Glines Canyon reservoir footprint, in contrast was the dominant contributor to landscape carbon balance (-283 Mg C yr^{-1} (-873, 308)), followed by river CO_2 emissions (161 Mg C yr^{-1} (20.8, 563)).

Results from specific flux parameterizations

Before

We were unable to determine the depth of the Glines Canyon dam intake from the literature, and were therefore unable to verify whether it was appropriate to model emissions with or without degassing. We therefore modeled Glines Canyon reservoir total emissions with and without degassing (Fig B1). The uncertainty deriving from this issue was ultimately much smaller than the uncertainty in reservoir carbon burial. For this reason, we chose to only report Glines Canyon reservoir emissions with degassing in the main text.

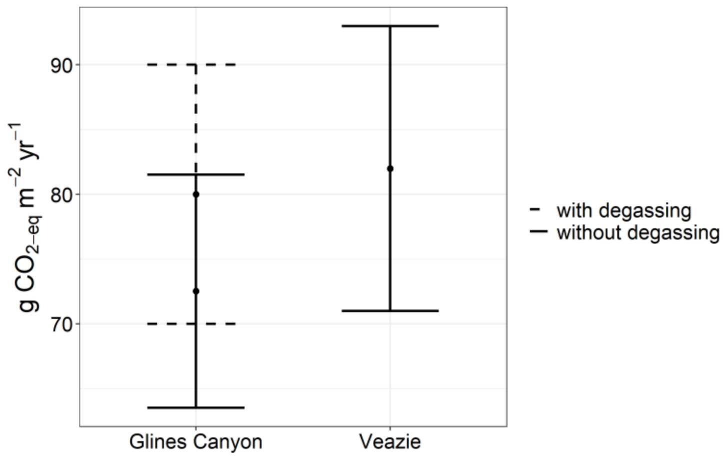

Figure B1. CO_2-eq emissions from the reservoir surface of the Glines Canyon and Veazie reservoirs, with and without degassing emissions from the Glines Canyon.

Burp

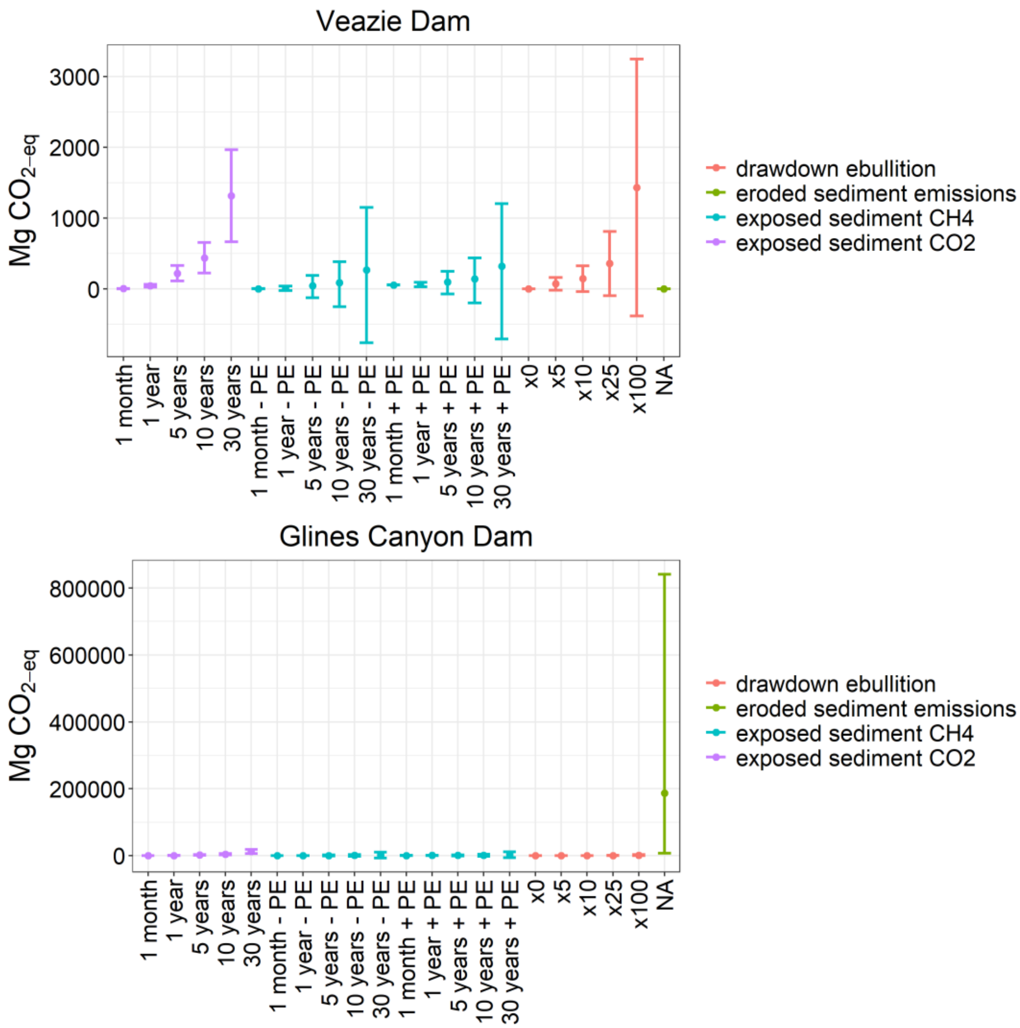

Figure B2. Veazie Dam and Glines Canyon Dam removal burp. Time periods (e.g., 1 month) indicate different scenarios for how long sediment emissions were allowed to decline exponentially until they reached near-zero emissions. +/-PE refers to whether pore CH_4 emissions were included or excluded in exposed sediment CH_4 emissions. Multipliers refer to the proportion of annual ebullition which was assumed to be emitted during reservoir drawdown. NA indicates that all parameters of flux could be reasonably constrained, and thus a single value and accompanying uncertainty is presented rather than values and uncertainties bracketed by parameter values.

After

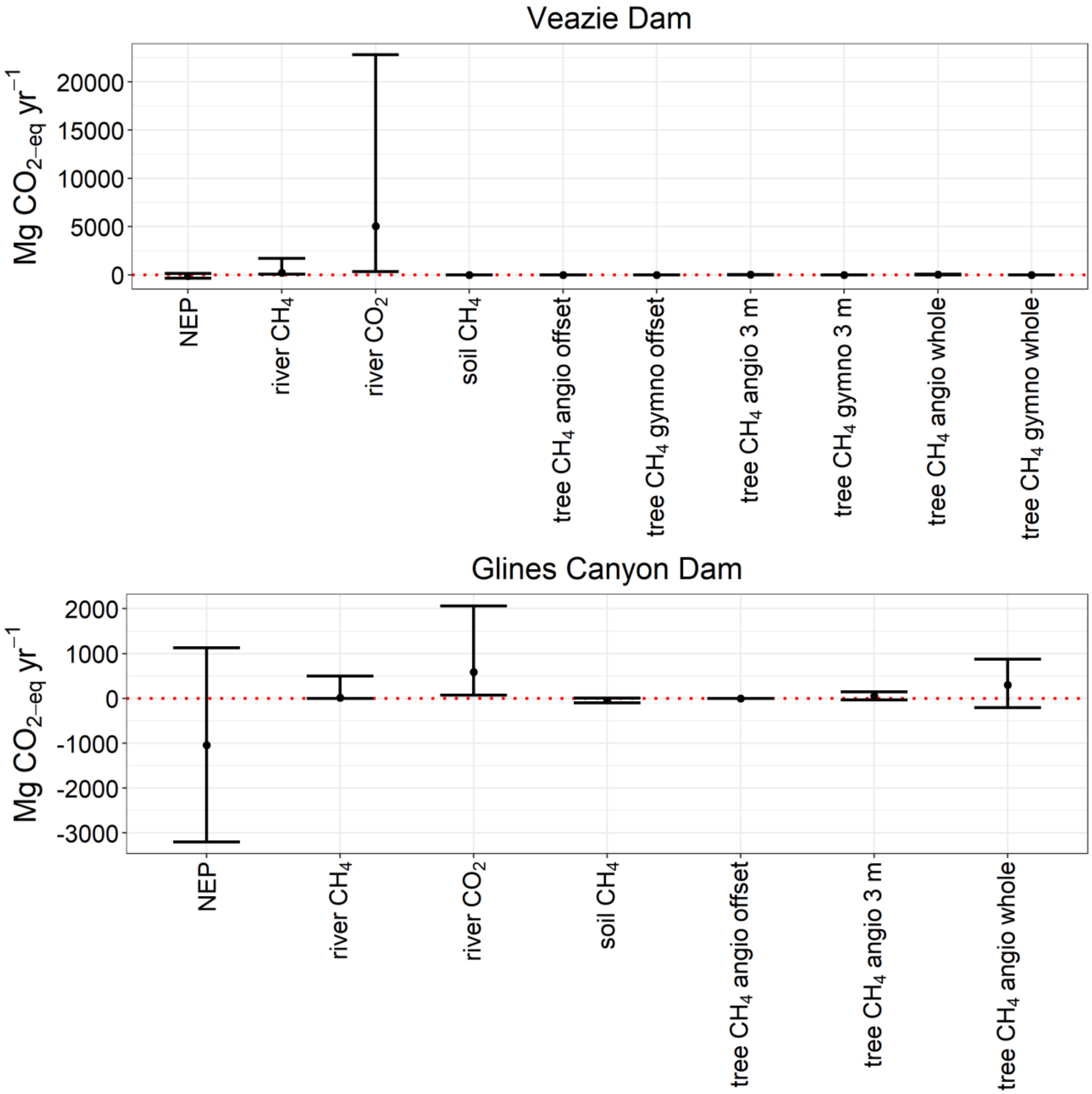

Figure B3. Veazie and Glines Canyon reservoir footprint mass flows after removal. Angio refers to scenarios modeled with angiosperms and gymno refers to scenarios modeled with gymnosperms. Offset is the scenario in which trunk CH_4 emissions are assumed to be entirely offset by oxidation of CH_4 in the canopy. 3 m is the scenario in which CH_4 emissions only occur within the first 3 m of trunk height, and whole refers to the scenario in which CH_4 emissions occur through the entire trunk.

Parameterizing drawdown ebullition

To our knowledge there have been no empirical measurements of ebullition during the complete drawdown of a reservoir. Further there have been few measurements of ebullition during smaller magnitude drawdowns which are accompanied by estimates of annual ebullition from the reservoir (but see Harrison et al. 2017; Beaulieu et al. 2018). We ground our parameterization of the proportion of annual ebullition emitted during complete drawdown in estimates from two reservoirs which experienced 1.5-2 m drawdowns (Harrison et al. 2017; Beaulieu et al. 2018). Partial drawdown of one reservoir emitted 3-10x annual ebullition and the other emitted 0.03x annual ebullition. We therefore included a parameterization with drawdown ebullition equivalent to 0, 5, and 10x annual ebullition in line with these measurements. Because these estimates were from partial drawdowns, we also modeled scenarios in which complete drawdown ebullition was 25 and 100x annual ebullition.

Parameterizing exposed sediment emissions

Rates of CO_2 production from sediments depend non-linearly on sediment moisture. At high values of sediment moisture, CO_2 production is low. CO_2 production peaks between 20-60% water content and declines precipitously below 20% water content (Isidorova et al. 2019). The relationship between sediment moisture content and CO_2 production could be used to estimate the amount of time it takes for exposed sediment CO_2 emissions to approach zero to parameterize our decay model; however, sediment drying after reservoir dewatering, is a complex process governed by both sediment and environmental characteristics (e.g., sediment grain size and the frequency and intensity of precipitation). For this reason, generating estimates of the amount of time it takes sediments to dry, and by extension the amount of time it takes sediment CO_2 emissions to reach nearly zero, is challenging. We therefore bracket our estimates of exposed sediment CO_2 emissions by the length of time it takes emissions to reach nearly zero in an exponential decay model. Full crust development from reservoir drying can take 20 to 39 months in fine grained reservoirs, and coarse grained sediments will dry more quickly (personal communication, Paul Schroeder). We ground our selections of 1 year and 5 years to nearly zero exposed sediment CO_2 emissions within this range of values. We select 1 month and 10 years as values substantially higher and lower than the values derived from sediment drying time, and we additionally include 30 years because this parameterization was used in a previously published estimate of exposed sediment CO_2 emissions (Marcé et al. 2019).

Parameterizing tree CH_4 emissions

Removal of the Glines Canyon Dam resulted in distinct landforms in the remaining sediment: valley walls and terraces. The valley walls were defined by steep slopes along the edge of the former reservoir while the terraces were flatter areas close to the river floodplain (Chenoweth et al. 2022). We modeled tree CH_4 emissions from the valley walls using upland tree CH_4 emissions and the terraces using wetland tree CH_4 emissions, as we expected greater inundation potential for the terrace region. However, if the terraces do not develop wetland characteristics because they are perched too high above the river floodplain, our calculated tree CH_4 flux may be an overestimation. Because tree CH_4 was a small component of landscape carbon balance after removal, we consider this a minor potential issue.

Table B1. Detailed description of the sources for statistical model input variables.

Flux	**Model/source**	**Input variables**	**Source**
Reservoir surface emissions	G-res (Prairie et al. 2017a)	Catchment area	StreamStats (U.S. Geological Survey 2019)
		Population in catchment	EnviroAtlas Dasymetric allocation of population 2010 CONUS (U.S. Environmental Protection Agency)
		Community wastewater treatment	State onsite wastewater treatment system utilization rate (National Environmental Services Center 2020)
		Landcover in catchment	2011 National Land Cover Database (Homer et al. 2015)
		River area	Literature reported values or, if unavailable, we approximated river length as the distance between reservoir inlet and outlet and used allometric equations built into G-res to calculate river area (Whipple et al. 2013).
		Age	Determined from year of dam closure reported in the literature.
		Reservoir surface soil C content	Harmonized World Soil Database (Fischer et al. 2008)

		Monthly temperature	World Clim (Fick and Hijmans 2017)
		% Littoral area	G-res estimate derived from mean and maximum reservoir depth (Prairie et al. 2017b)
		Cumulative global horizontal radiance	World Clim (Fick and Hijmans 2017)
		Water residence time	Calculated from literature reported discharge, mean depth, and reservoir area.
		Discharge	Literature reported values
		Reservoir area	Literature reported values or if not reported in the literature, from a delineation of the reservoir surface using NAIP imagery.
Reservoir carbon burial	Measured values (see Table B2) or (Clow et al. 2015)	Average catchment slope	WSIO Watershed Index Online (WSIO) (U.S. Environmental Protection Agency 2023)
		Crop cover in the catchment	2011 National Land Cover Database (Homer et al. 2015)
		Forest cover in the catchment	2011 National Land Cover Database (Homer et al. 2015)

		Reservoir area	Literature reported values or if not reported in the literature, from a delineation of the reservoir surface using NAIP imagery.
		Sediment organic carbon 0-5 cm	Soil Survey Geographic Database (SSURGO) (Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture)
		K factor	Soil Survey Geographic Database (SSURGO) (Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture)
		Wetland cover in the catchment	2011 National Land Cover Database (Homer et al. 2015)
		Barren cover in the catchment	2011 National Land Cover Database (Homer et al. 2015)
		Reservoir area	Literature reported values or if not reported in the literature, from a delineation of the reservoir surface using NAIP imagery.
River CO_2 emissions	(Hotchkiss et al. 2015)	Discharge	Literature reported values
River CH_4 emissions	(Stanley et al. 2023)	Discharge	Literature reported values

Terrestrial NEP	(Besnard et al. 2018)	Mean annual temperature	World Clim (Fick and Hijmans 2017)
		N deposition	EnviroAtlas (U.S. Environmental Protection Agency)
Tree CH_4 emissions	Mechanistic model based on tree surface area from FVS	Angiosperm tree CH_4 emissions	(Pitz et al. 2018)
		Gymnosperm tree CH_4 emissions	(Machacova et al. 2016)
		Tree density after 10 years	(Van Pelt et al. 2006)
		Tree species	Literature reported values
		FVS input: location	Nearest National Forest by Euclidean distance
		Aspect, elevation, slope	DEMs (Table B2)
Soil CH_4 emissions	(Gatica et al. 2020)	Mean annual precipitation	World Clim v2.1 (Fick and Hijmans 2017)
		Mean annual temperature	World Clim v 2.1 (Fick and Hijmans 2017)
		Soil bulk density	Global Soil Dataset for Earth System Modeling (Shangguan et al. 2014)

Soil organic content	Global Soil Dataset for Earth System Modeling (Shangguan et al. 2014)
Soil pH	Global Soil Dataset for Earth System Modeling (Shangguan et al. 2014)
Soil sand content	Global Soil Dataset for Earth System Modeling (Shangguan et al. 2014)
Biome	Literature reported values

Table B2. Sources of literature values used for model parameterization.

Dam	Source for literature reported values
Glines Canyon	(Stratton and Grant 2019): river area, discharge, reservoir area
	(Stratton et al. 2019): sediment volume, sediment carbon
	(US Bureau of Reclamation 2011): sediment volume uncertainty
	(Wing 2014): bulk density and sediment carbon uncertainty
	(Randle et al. 2015): eroded sediment mass
	(Van Pelt et al. 2006): tree species
	(Ritchie et al. 2018): DEM
Veazie	(Collins et al. 2020): eroded sediment volume, discharge
	Amazon Web Services Terrain Tiles: DEM
	(U.S. Forest Service 2018): tree species

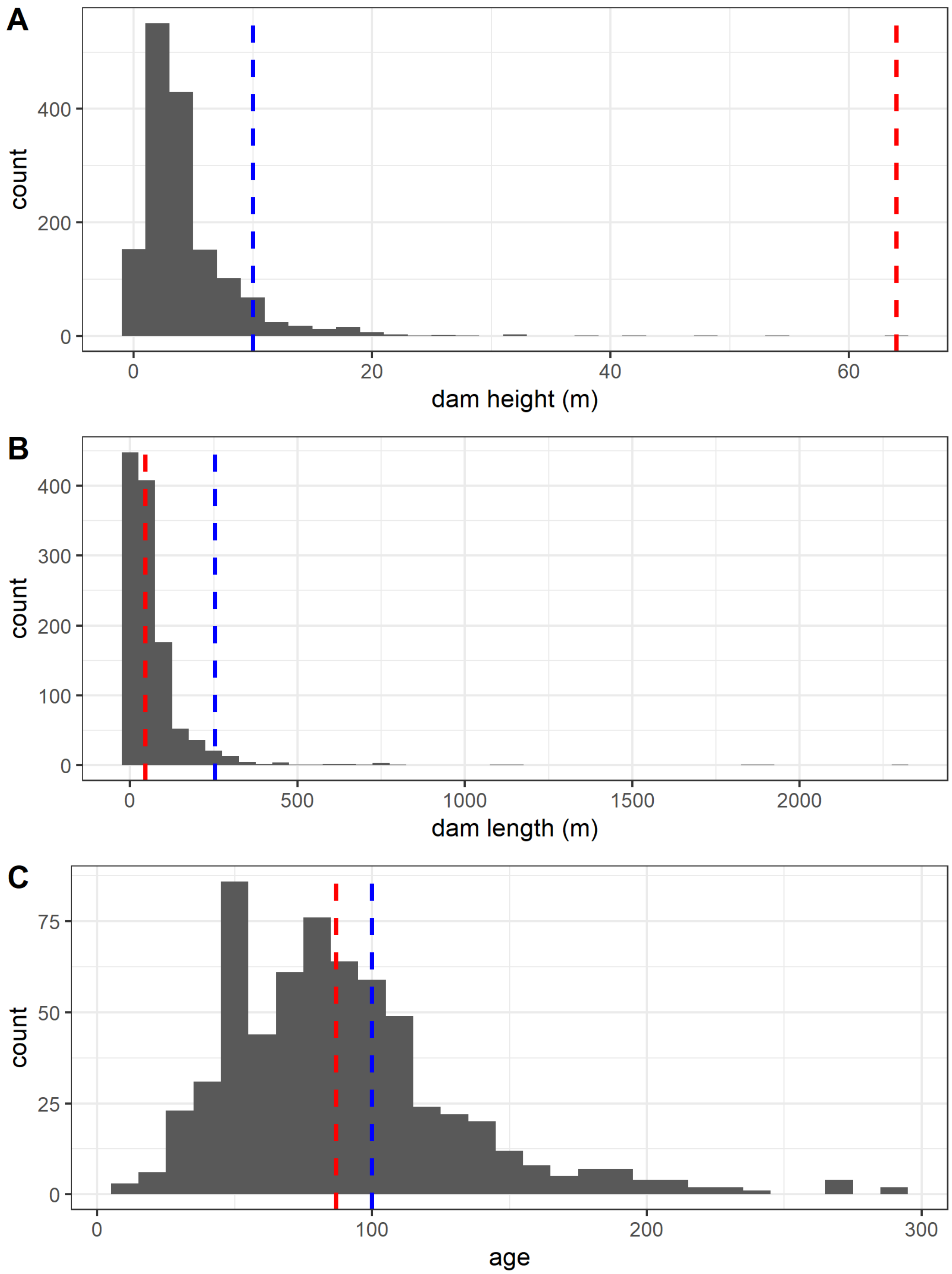

Figure B4. Comparison of characteristics of the Veazie and Glines Canyon Dam removals to other dam removals in the United States. The red line indicates values for the Glines Canyon Dam and blue line indicates values for the Veazie Dam removal.

APPENDIX C

SUPPLEMENTARY METHODS AND DATA: CHAPTER 3

Elaboration of author contributed metrics/methods

Objective 1a. Cost of repairing dam and appurtenant structures: Because repair/retrofit and removal likely incur different costs, we include repair costs as a separate metric. We recommend using expert evaluation or the Association of State Dam Safety Officials (ASDSO)'s cost estimation method which is based on previous project costs and dam characteristics.

Objective 1b. Loss of tax revenue: If dam removal decreases the value of surrounding properties, it may contribute to lower tax revenue. The impact of dam removal on property values is not well understood, and evidence of the effects of small dam removals on property values is mixed. We recommend expert evaluation to predict the loss of property value with dam removal and the subsequent loss of tax revenue. We define this loss as a maintenance cost because it is incurred on an on-going basis. We also note that in cases where dam removal could reduce property values, riparian residents will also likely oppose removal, which is a metric for objective 4d) minimize interruption to community sense of place.

Objective 2a. Change in water surface elevation relative to water intakes: A chief concern of municipalities regarding water supply is the elevation of their intake pipes, particularly during low-flow periods. For reservoirs where dam decommissioning is considered, it is imperative that the post-removal water surface elevation remains adequately high to maintain withdrawal through the current intake; otherwise, the cost of installing a new intake should be accounted for in the decision process. We recommend the use of an appropriate hydraulic model to estimate water surface elevation during low flow periods.

Objective 2a. Change in local average groundwater table depth: Twenty-five percent of water use in the US is from groundwater (Dieter et al., 2018), which is connected to reservoirs through surface water-groundwater exchange. In regions where this is a potential issue, dam decommissioning analyses should include groundwater modeling to evaluate how changes to stream and reservoir elevations propagate into the local aquifer to ensure that groundwater supply is not compromised (Berthelote, 2013).

Objective 2b. Users of dam power: The population served by dam-generated power provides an additional element for understanding its importance for service delivery.

Objective 2c. Users of in-stream dam navigation: The annualized number of vessels using a dam for navigation is an indicator of its importance for transportation.

Objective 2c. Users of terrestrial dam navigation: The annualized number of vehicle or axle crossings over a dam is an indicator of its importance for transportation.

Objective 2c. Proximity to nearest alternative stream-crossing infrastructure: An important consideration when evaluating decommissioning options is to determine the distance traffic must be re-routed to cross the stream if the road that a dam supports is removed. This is useful to couple with the metric for number of vehicles crossing the road over the dam to determine the complete effect of removing that road (e.g., if traffic must be re-routed, is the amount of traffic or re-routed distance sufficiently low?).

Objective 3a. Proportion of annual flows classified as dangerous submerged hydraulic jumps: Low head dams can be a significant hazard for dangerous submerged hydraulic jumps, but that hazard is dependent on flow conditions over the dam (Poff & Hotchkiss, 2023). Whereas the other provided method for this metric evaluated potential hazard based on simple dam characteristics, this method provides a more detailed assessment for how frequently these dangerous flows may occur. Combining flow frequency analysis with physical dam measurements to estimate this hazard helps decision makers quantify the prevalence of life-threatening flow conditions.

Objective 4a. Accessible stream length: Dams can impede swiftwater recreation by creating barriers to the movement of small craft like kayaks, canoes, and rafts. Connectivity methods can be used to estimate the changes to accessible areas for swiftwater recreation under different dam removal alternatives.

Objective 4b. Accessible flat-water area: Dams can create opportunities for flat-water recreation like motorized boating, swimming, and paddle boarding. Calculating accessible flat-water area can indicate opportunities for this type of recreation under different dam removal alternatives.

Objective 4c. Average community ranking of historical significance: Sites without formal historical recognition (e.g., National Register of Historic Places) may still be important to a community's historic memory (Fox et al., 2016). Asking community members to rank their evaluation of a dam's historical significance with other places in the community could indicate the relative importance of the dam to the community's historic memory.

Objective 5e. Net water quality improvement: Dam decommissioning can have variable effects on many aspects of water quality (e.g., temperature, dissolved oxygen, sediment, nutrients, etc.), so it is important to estimate the direction and magnitude of these changes. Generating a water quality model can inform a quantitative assessment of potential local and downstream benefits of dam decommissioning.

Objective 5f. Net change in channel width/depth ratio, slope: Dam decommissioning can release a volume of sediment that alters the local and downstream channel morphology, affecting flow characteristics and connection to the floodplain. Common methods to quantify the effects of bed and bank erosion are through channel slope and width-depth ratio. Substantial shifts in these parameters can trigger channel instability and impair stream and riparian function.

Objective 5f. Erosion potential: Erosion potential assesses the susceptibility of the stream to erode based on the amount of sediment available and the stream's ability to move that sediment. This is imperative to understanding potential geomorphic channel change that may result from dam decommissioning.

Elaboration of application features

We constructed the Dam Objectives & Metrics Selector web application using R Shiny version 1.7.4 (Chang et al., 2022). The Guidance landing page hosts this manuscript to provide context on the intended use of this application. On the Objectives tab, the user can select from among the listed objectives, and the associated metrics and methods will be displayed in a table on the Metrics tab. Clicking on the Objectives Categories labels displays dialogue boxes containing definitions and justifications for each of the objectives in the selected category. Similarly, clicking the information icon associated with each of the methods in the table in the Metrics tab displays a dialogue box containing information about the data requirements and sources for that method as well as citations. Methods not derived from existing published literature are cited as "Authors." The Tools tab displays citations for tools which include the user's selected objectives. The final Feedback tab provides a space for users to alert the authors to additional resources to include in the application. By this mechanism, we envision the application as a dynamic community resource.

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APPENDIX D

INCORPORATING BIOGEOCHEMICAL CYCLING INTO DECISION ABOUT DAM REMOVAL AND LONG-TERM MANAGEMENT[4]

Abstract

A comprehensive framework for evaluating the benefits and costs of dams is urgently needed to support decisions about the long-term management of this aging infrastructure. To date, such frameworks have largely focused on reservoir and dam condition, while excluding the ecosystem services and disservices caused by the conversion of flowing waters to reservoirs. In particular, dams and their reservoirs contribute many services and disservices through their impacts on elemental cycles. Carbon, growth-limiting nutrients, and other elements can be transformed by biological and chemical processes as they enter reservoirs from flowing waters. These biogeochemical changes can cause algal blooms and subsequent low oxygen conditions, resulting in undesirable effects in the reservoir and downstream aquatic ecosystems. The goal of this technical note is to review the published literature on the biogeochemical impacts of dams and argue for including elemental cycles more explicitly in dam management decisions.

Introduction

In the United States, over 2.5 million dams have supported river navigation, flood control, water supply, recreation, and hydropower generation (Conyngham et al., 2006). However, this critical infrastructure is aging. Approximately half of the 90,000 dams in the

[4] Naslund, L.C., Wenger, S., Rosemond, A., McKay, S.K. Submitted as *US Army Corps of Engineers Engineer Research and Development Center Technical Note*, *3/19/23.*

National Inventory of Dams are over 50 years old (Gonzales & Walls, 2020). With dams aging, infrastructure managers increasingly will be faced with decisions about whether to repair or remove dams. For some dams, removal may be an attractive alternative to repair and an opportunity for stream restoration (USACE 2018). Other dams may merit repair because they provide critical services (Doyle et al., 2003). Tools to support decisions about the management of aging dams are urgently needed because many dams in the United States are approaching or have passed their design life.

To develop these decision support tools we need to identify the criteria necessary to fully evaluate the costs and benefits of dam removal. Dam removals over the past two decades have been primarily motivated by securing public safety, reducing financial liability, and promoting fish passage (Conyngham et al., 2006; Walls, 2020); thus, research efforts have focused on decision criteria related to these outcomes (Bellmore et al., 2017; McKay et al., 2020). The impacts of dams and their removal on carbon, nutrients, and other elements have received less attention, although dams may be providing important services and disservices through these impacts, which are currently unaccounted for in cost-benefit analyses. For example, in systems with excess nitrogen (N) inputs, the oxygen-depleted reservoir sediments may decrease N loads and associated risks of harmful algal blooms by promoting microbially-mediated N removal (Baulch et al., 2019; Powers et al., 2015). On the other hand, reservoirs have been implicated as sources of excess methane, a carbon (C)-based greenhouse gas (GHG) which contributes to climate warming (Deemer et al., 2016). Expanding the criteria by which a dam is evaluated to include its biogeochemical impacts offers a fuller accounting of its benefits and costs (Maavara, Chen, et al., 2020). This technical note provides an overview of dam impacts on material fate and transport and refers to studies across a range of dam sizes and engineered purposes to argue for the inclusion of biogeochemical endpoints in long-term dam management decisions. The goal of this technical note is not to provide specific indicators for management decisions, although we affirm that this is a critical next step.

Reservoir effects on element transformation and fate

Materials transported in streams and rivers experience three major fates. They can be 1) buried in sediments, 2) transported downstream, or 3) exported to the atmosphere as a gas (Cole et al., 2007). While in transport in stream ecosystems, materials can experience several transformations, including incorporation into the tissues of living organisms (Stream Solute Workshop, 1990). While biological uptake could be construed as a fourth fate, it is a relatively temporary one, so we consider uptake a transformation rather than a fate. We can understand the biogeochemical impacts of dams by considering how they alter the fates and transformations of ecologically important elements transported in streams.

In general, dams reduce water velocity, increase water surface area and depth, and increase hydraulic residence time: the average retention time of water in a system (Harvey & Schmadel, 2021). Dams can promote material burial by reducing water velocity, causing particle-bound elements entrained in the reservoir inflow to settle (Friedl & Wüest, 2002). By prolonging hydraulic residence times, dams can also alter the extent to which materials are biologically transformed as they are transported downstream (Schlesinger & Bernhardt, 2013; Stream Solute Workshop, 1990). Prolonged residence time promotes biological transformation because it increases the exposure of materials to the organisms that transform them (Maavara, Chen, et al., 2020). Dissolved oxygen is one of the most important constituents of water quality altered by prolonged residence time (Friedl & Wüest, 2002). Lower turbulence with slower water velocity reduces gas exchange across the air-water interface, limiting reaeration and promoting biological oxygen depletion (Hall & Ulseth, 2020). In some reservoirs, dissolved oxygen can be further impacted by the spatial and temporal extent of summer stratification, which occurs when solar radiation heats the surface of the reservoir creating a top layer of warmer water separated from the bottom, cooler layer by differences in water density. Minimal mixing occurs across these layers which limits the replenishment of oxygen to the bottom layer of the reservoir and results in biological oxygen depletion (Dodds & Whiles, 2019). These changes in water temperature and dissolved oxygen content can propagate downstream as water is withdrawn from the reservoir and discharged downstream. By this mechanism, intake depth can be an important determinant of downstream water quality (Ignatius & Rasmussen, 2016).

In the following sections, we review the literature addressing how dams have altered the fate and transport of elements by modifying riverine characteristics like water residence time, surface area, material transport potential, and ecological productivity. We organize this review by ecologically important chemical element, acknowledging that elemental cycles are inextricably linked. Where relevant, we discuss these linked cycles below. We focus on carbon as an element essential to energy exchange and biological structure, nitrogen, phosphorus, and silicon as macronutrients in aquatic ecosystems, and mercury as a widespread, regulated pollutant.

Carbon

Organic C enters freshwater ecosystems through photosynthesis by aquatic plants and algae or by inputs of plant and animal material from land (e.g., leaves and wood from trees). The microbial breakdown of this organic matter in the presence of oxygen (O_2) produces carbon dioxide (CO_2), a greenhouse gas (GHG) which can be emitted to the atmosphere (Dodds & Whiles, 2019). Flooding land following dam closure can expose additional stores of organic matter in flooded soils to conditions prime for microbial decomposition, promoting CO_2 emissions to the atmosphere (Deemer et al., 2016).

Reservoirs also create conditions favorable for methane (CH_4) production and emissions (Deemer et al., 2016). When O_2 and other reactants (i.e., nitrate (NO_3^-), manganese (Mn^{4+}), iron (Fe^{3+}), and sulfate (SO_4^{2-})) used in the microbial breakdown of organic matter have been exhausted, microorganisms called methanogens split acetate (CH_3COOH) or shuttle electrons from hydrogen gas (H_2) to CO_2, producing CH_4 (Schlesinger & Bernhardt, 2013). The shift in the end product of microbial metabolism from CO_2 to CH_4 in O_2-depleted sediments is significant because CH_4 has 34 times the warming potential of CO_2 (Deemer et al., 2016). Although mean CH_4 fluxes from the surface of reservoirs estimated globally are lower than mean CO_2 fluxes (120.4 mg CH_4-C/m^2/day and 329.7 mg CO_2-C/m^2/day, respectively), CH_4 emissions are responsible for 80% of the warming effect caused by gas emissions from reservoirs over a 100 year time horizon (Deemer et al., 2016). By creating conditions for CH_4 production and emissions, dams can provide the disservice of promoting climate warming.

On the other hand, reservoirs may also serve as sinks for organic C by promoting burial in their sediments. Buried organic C is inaccessible to microbial conversion to CO_2 or CH_4. Prolonged water residence time in reservoirs can facilitate C burial by promoting the settling and subsequent burial of organic particles. Prolonged water residence time also promotes the proliferation of phytoplankton, which convert CO_2 to organic C that settles into reservoir sediment when algae die and sink (Clow et al., 2015). The assumption that organic C burial offsets some of the C lost to the atmosphere as CO_2 or CH_4 may be too simplistic; it is often unclear whether C burial in the reservoir represents “new” C burial, and thus a new C sink, or if the burial would have occurred elsewhere in the river network or in the coastal ocean in the absence of the reservoir (Prairie et al., 2018). It is also important to consider that the upland soils, vegetation, and floodplains which a dam inundated may have also been a larger carbon sink than the reservoir which replaced it, resulting in a net loss of C sink from dam construction (Prairie et al., 2018). The GHG Reservoir (g-res) Tool developed by the International Hydropower Association and UNESCO provides methods to approximate pre-reservoir landscape emissions and account for them in estimating reservoir-attributable emissions (https://g-res.hydropower.org/). While outside of the scope of this review, it is also important to note that reservoirs can alter the structure of food webs within and downstream of reservoirs by shifting the availability and reliance of consumers on algal versus detrital resources (Cross et al., 2013; Doi et al., 2008; Freedman et al., 2014; Ruhí et al., 2016).

Nitrogen

Streams are naturally dilute in bioavailable forms of nitrogen (N); however, agriculture, industry, and urban development have increased N loading to freshwater ecosystems, such that 43% of streams and rivers in the US are now impaired due to excess N (Manning et al., 2020; U.S. EPA, 2020). Excess N in aquatic ecosystems can promote harmful algal blooms (HABs), which endanger wild and domestic animals as well as human recreation and water supply. The estimated annual costs of HABs in the US was $2.2 billion over a decade ago (Dodds et al., 2009), and HABs are increasing in frequency and duration worldwide (Gobler, 2020; Ho et al., 2019). Adverse human health outcomes from high drinking water nitrate (NO_3^-) exposure in the US cost an estimated $250 million to $1.5 billion in medical expenses as well as $1.3 to $6.5 billion due to lost productivity annually (Temkin et al., 2019).

Reservoirs are typically considered sinks for N because they create conditions for N removal via microbial processing and particulate N burial (Akbarzadeh et al., 2019). The construction of small reservoirs has even been proposed as a management strategy for nutrient-impaired waters in the Great Plains region (Baulch et al., 2019). Globally, reservoirs are estimated to eliminate 3.7 Tg N annually by denitrification, a pathway of anaerobic metabolism used by microorganisms that transforms nitrate (NO_3^-) into nitrogen gas (N_2) (Maavara, Chen, et al., 2020). Denitrification permanently removes reactive N from the river, as most aquatic organisms cannot use N_2 as a source of N. Dams promote N burial by the same mechanism that they promote C burial: facilitation of particle-bound N settling and subsequent burial. Global reservoirs are estimated to bury 1.54 Tg N annually (Maavara, Chen, et al., 2020).

However, reservoirs also create conditions that can promote the fixation of atmospheric N, which adds available N in aquatic ecosystems. The increased water residence times, temperatures, and light conditions caused by reservoir creation can lead to proliferation of N-fixing cyanobacteria, resulting in the addition of 0.98 Tg N per year to rivers globally (Maavara, Chen, et al., 2020). While less than the total amount of N removed by reservoirs due to N burial and denitrification, N addition by N-fixing cyanobacteria must be considered in estimating the net impact of reservoirs on river N loads (Akbarzadeh et al., 2019). The addition of reactive N by N-fixing cyanobacteria may also promote the proliferation of toxin-producing cyanobacteria taxa, resulting in the disservice of water pollution by algal toxins (Beversdorf et al., 2013).

The role of a reservoir in reducing river N loads may be an important consideration for dam removal decisions (Hart et al., 2002). Denitrification rates in formerly impounded reaches are likely to decline as a stream returns to free-flowing conditions, resulting in a long-term loss of a N sink (Powers et al., 2013); however, restored riparian areas may be able to compensate for some losses in denitrification capacity upon dam removal (Lammers & Bledsoe, 2017). Dam removal may also mobilize N previously buried behind the dam. The capacity of the downstream aquatic ecosystems to assimilate the pulsed N addition and compensate for the loss of an upstream N sink—a capacity determined in large part by the current supply of nutrients to the downstream ecosystems—may be an important consideration for the timing and method of dam removal.

While reservoirs may provide the service of reducing river N loads, reservoirs can also provide the disservice of emitting the N-based GHG nitrous oxide (N_2O) (Lauerwald et al., 2019). Like CH_4, N_2O is a more potent GHG than CO_2, with 298 times its warming potential (Deemer et al., 2016). Global estimates of N_2O emissions from reservoirs range from 20 to 71.5 Gg N/yr (Maavara, Chen, et al., 2020). N_2O also depletes ozone in the stratosphere, increasing UV penetration to the Earth's surface (Ravishankara et al., 2009). N_2O is formed as a product of incomplete denitrification and from the decomposition of an intermediate product of nitrification, an aerobic process in which microbes oxidize ammonium (NH_4^+) to nitrate (NO_3^-). N_2O emissions from reservoirs are likely to vary widely across systems; thus, understanding the controls on N_2O emissions from impounded waters represents an important future research need.

Phosphorus

Like N, phosphorus (P) is a growth-limiting nutrient in aquatic ecosystems which promotes harmful algal blooms in waterbodies where it is present in excess (Dodds & Whiles, 2019). Natural P inputs to rivers are derived from rock weathering and, unlike N, there is no significant pool of P in the atmosphere (Schlesinger & Bernhardt, 2013). Humans have elevated P flux to surface waters by 800% by liberating P through mining of phosphate rock deposits and widely distributing it in agricultural fertilizer and consumer products. Consequently, 58% of rivers and streams in the US are impaired due to excess P (Mallin & Cahoon, 2020; U.S. EPA, 2020).

Reservoirs can efficiently sequester P in river inflows, with an individual reservoir retaining up to 94% of its P inputs (Maavara et al., 2015). Globally, reservoirs are estimated to retain 12% of their predicted annual P load. Because P readily binds to minerals in sediment, P movement in aquatic ecosystems is often strongly associated with sediment movement. Like with N and C, reservoirs can bury particulate P in their sediments (Maavara et al., 2015). Reservoirs can also retain dissolved inorganic P in the form of phosphate (PO_4^{3-}) because it readily sorbs to minerals and can be precipitated by ferric iron (Fe^{3+}) under oxic conditions (Maavara et al., 2015). Whether reductions in P loads result in services or disservices provided by dams depends on local context (Friedl & Wüest, 2002). Retention of P inputs by the Aswan High Dam was implicated in the collapse of local fisheries in the Nile delta region in 1964 following its construction (Nixon, 2003). Similarly, Sockeye salmon catches in Kootenay Lake, Canada declined after the construction of dams on its tributaries, leading managers to institute a fertilization program in the oligotrophic lake to compensate for nutrients sequestered by upstream dams (Ashley et al., 1997). In contrast, P retention by dams may be considered beneficial for rivers in the agricultural Midwest of the United States which are disproportionately impaired due to excess P (U.S. EPA, 2020).

Similar to N, mobilization of stored legacy P in reservoirs may be an important consideration for the timing and manner of dam removal. Unlike N, P does not have a gaseous form that is significant in nature. Thus, P cannot be removed from aquatic ecosystems through gas emissions to the atmosphere and mobilized legacy P may remain in downstream aquatic ecosystems for longer periods of time before being immobilized. Flushing of sediments from Guernsey Reservoir on the North Platte River, for example, mobilized stored P, contributing to a downstream bloom of filamentous algae (Gray & Ward, 1982). The capacity of a reservoir with sustained high P loading to retain P may also decline over time as storage capacity is exceeded and internal stores are mobilized (Powers et al., 2015). Potential declines in P retention over the lifetime of a reservoir may be an additional consideration in prioritizing dams for removal. Beneficial application of P-rich sediments extracted from former reservoirs to agricultural fields or farming former sediments in place may be an additional consideration for dam removal (Chuck Theiling, personal communication).

While reservoirs may efficiently sequester particulate P, they may also mobilize dissolved P when sediment O_2 levels are low. Sediments of thermally stratified reservoirs or reservoirs with high chemical or biological O_2 demand are frequently anoxic (Friedl & Wüest, 2002). In the absence of O_2, ferric phosphate ($FePO_4$) dissociates, releasing bioavailable phosphate (Dodds & Whiles, 2019). A cascade of reservoirs in the Upper Mekong River, for example, increased the proportion of bioavailable P from 22.0% of total sediment P above the dams to 83.7% below the dams, likely due to declines in sediment O_2 availability along the reservoir cascade (Chen et al., 2020). Thus, while reservoirs may reduce total P loads in rivers, they may also increase the bioavailable fraction of P, potentially promoting algal blooms.

Silicon

Silicon (Si) is an essential micronutrient for algae. Diatoms, a nutritionally rich and frequently abundant group of algae, can be limited in their growth by Si availability because they construct silica cell walls. Like N and P, Si can be sequestered behind dams by particulate burial in reservoir sediments. Dams can also sequester dissolved Si by promoting the growth of diatoms, which assimilate dissolved Si into their cell walls and transport it to reservoir sediments when they die and sink. The loss of dissolved Si inputs from rivers by these mechanisms has been associated with changes in nearshore marine algal communities (Ma et al., 2017). The closure of the Iron Gate dam on the Danube River, for example, caused an 80% decrease in the load of dissolved Si to the Black Sea over the course of 20 years, resulting in a decline in diatoms and an increase in blooms of toxic dinoflagellates (Humborg et al., 2000).

Unlike N and P, humans have not substantially increased Si loading to freshwater, which has altered expected ratios of N:P:Si (Maavara, Akbarzadeh, et al., 2020). Nutrient ratios can be a control on the composition of algal communities, as algal taxa are limited by N, P, and Si to different extents and changes in nutrient ratios can alter the outcome of competition (Klausmeier et al., 2008). Dams further alter N:P:Si ratios by retaining each nutrient with different efficiency. On average, P is retained most efficiently for dams with water residence time greater than 50 days and Si is most efficiently retained for dams with residence time less than 50 days (Maavara, Chen, et al., 2020). The resulting changes in N:P:Si ratios may complicate efforts to predict the consequences of changes to the biogeochemical cycle of any one element caused by dams or changes in nutrient inputs; however, modeling linked elemental cycles has been identified as a priority for biogeochemical research and future progress may be used to design dam and reservoir evaluation methods (Reinhold et al., 2019; Weathers et al., 2016)

Mercury

Mercury (Hg) is a trace element loaded to surface waters from the weathering of igneous rocks. Humans have dramatically increased Hg loading through mining and fossil fuel combustion and Hg contamination of freshwater ecosystems is widespread. In a survey of US lakes and reservoirs, fish mercury concentrations exceeded EPA criterion in nearly half of sampled sites (Stahl et al., 2009). Bacteria that consume sulfate in the anaerobic breakdown of organic matter can also transform dissolved Hg (Hg^{2+}) into the neurotoxic and bioaccumulative form of Hg, methylmercury (CH_3Hg^+) (Schlesinger & Bernhardt, 2013). The anoxic and high organic matter sediment conditions created by dams can promote CH_3Hg^+ production, resulting in the enhanced flux of CH_3Hg^+ into aquatic food webs. Fish within these food webs may accumulate high Hg burdens, rendering them unsafe to eat (Eagles-Smith et al., 2018). In a large survey of total Hg concentrations in fish in the western US and Canada, fish caught in reservoirs were found to have on average 1.4 times the total Hg concentration of fish caught in lakes (Willacker et al., 2016). Methylmercury concentrations in fish and other organisms are particularly elevated in young reservoirs, and can eventually decline as reservoirs age; however, reservoir conditions like fluctuating water levels can maintain high fish Hg concentrations (Eagles-Smith et al., 2018). As with other contaminants, the potential mobilization of Hg in reservoir sediments upon dam removal is an important consideration for the timing and method of removal (Stanley & Doyle, 2003). High Hg concentrations may warrant sediment dredging prior to dam removal, increasing costs.

Synthesis and future directions

Dams can simultaneously provide a range of services and disservices through their biogeochemical impacts but quantifying these contributions across the range of dams and reservoirs that exist in the US remains a challenge. Developing multi-element loading, processing, and transport budgets which quantify the impact of even one dam and its reservoir on material fate requires a level of effort that cannot be feasibly replicated across a portfolio of dams that may be of interest to managers. For this reason, the development of indicators and proxies of dam impacts on elemental cycles are needed to incorporate the accompanying ecosystem services and disservices into dam removal decisions. In this effort, small dams (operationally defined as those excluded from the National Inventory of Dams: >25ft tall, impounding >15 ac-ft and > 6ft tall, impounding >50 ac-ft of water) merit greater research effort, because they are nearly ubiquitous in many landscapes (occurring every <1-2 km of stream length), are typically the focus of dam removals, and may have large cumulative impacts on GHG emissions and nutrient sequestration (Gardner et al., 2019; Ollivier et al., 2019; Schmadel et al., 2019).

Fully leveraging knowledge about the biogeochemical impacts of dams in infrastructure management requires consideration of their cumulative impacts. Current regional scale biogeochemical models do not account for the different rates and pathways of biogeochemical transformation that may occur because of anoxic sediment conditions and thermal stratification in reservoirs (Wollheim, 2016). Empirical work on the effects of upstream reservoirs on downstream reservoirs, streams, and rivers is also needed to incorporate reservoirs into spatially explicit models of material transport that can be used to understand the ecosystem services and disservices of dams at the landscape scale.

Despite its challenges, expanding the ecosystem services and disservices considered in dam management decisions would further the goal of integrating and harmonizing infrastructure and environmental management, a goal explicitly expressed in multiple policy directives and executive orders (e.g., Executive Order on Tackling the Climate Crisis at Home and Abroad (86 F.R. 7619)).

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