## Title

Balancing trade-offs for sustainable water resources management: reconciling climate change, hydropower, and environmental flows in the Central Sierra Nevada, California



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# Balancing trade-offs for sustainable water resources management: reconciling climate change, hydropower, and environmental flows in the Central Sierra Nevada, California

Dissertation submitted in partial satisfaction of the requirements for the degree of

Doctor of Philosophy

in

Environmental Systems
(Hydrology and Water Resources Engineering)

by

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*"Nessun effetto è in natura sanza ragione,*
*intendi la ragione e non ti bisogna sperienza."*

Leonardo da Vinci

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**Acknowledgments**

In 2019, I read the first sentence of my acceptance letter into this PhD program at least four times to make sure I did understand that I got admitted into a fully funded doctorate in the US. I was so delighted that this, a faraway thought from my high school self, was becoming true. When I was 8, collecting scrap metal with my parents during a financial hardship, I would never imagine I'd make it to my bachelor's degree and master's degree in public universities in Brazil, let alone a PhD in a renowned American institution.

I would like to thank my parents for giving me the opportunity of getting higher education, which they were not given growing up in Sao Paulo's countryside.

I thank my home institutions, Sao Paulo State University and Federal University of Mato Grosso do Sul, for the great and free education that allowed me to get here.

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## Curriculum Vitae


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### Education

**PhD in Environmental Systems (Hydrology and Water Resources Engineering)** – University of California, Merced (UCM - 2024)
**MSc in Environmental Systems** – University of California, Merced (UCM - 2022)
**MSc in Environmental Technology (Environmental Sanitation and Water Resources)** – Federal University of Mato Grosso do Sul, Brazil (UFMS - 2018)
**BSc in Agronomy/Agricultural Engineering -** São Paulo State University, Brazil (UNESP - 2014)

### Skills

- Portuguese (native), English (fluent), Spanish (advanced/fluent), French/Italian (basic/intermediary).
- Comfortable with Microsoft Windows, Office programs (Word, Excel, PowerPoint) and QGIS.
- Comfortable working with R and Python programming languages.
- Experience with ENVI, Stella, CALVIN, Google Earth Engine and Ecoinvent.
- Good communication skills, multitasking and problem-solving abilities.
- Good ability to work in team and to adapt to multicultural environment.
- Public speaking, teaching and presentation experience.

### Awards

- UCM - FARMERS Fellowship 2022-2023 – stipend, tuition and fees + Conference travel award
- UCM - Spring 2022 Clean Energy Research Center for Water-Energy Technologies Fellowship
- UCM - Summer 2021 ES Professional Fellowships
- UCM - Summer 2021 ES Bobcat Fellowship
- UCM - Fellowship Award 2020-2021 Miguel Velez Scholarship – prestigious award to Latin American students with good academic standing and excellence in character and ability
- UCM - NSF-Funded National Research Training (UCM) - Computational Education - Spring 2020 + Conference travel award
- UCM - Non-Resident Supplemental Tuition (NRST) Fellowship – 2019-Present
- Government of Canada - Scholarship from the Emerging Leaders in the Americas Program - Exchange student for one semester at the University of British Columbia, Okanagan campus
- UFMS - Federal fellowship awarded to the best graduate students for their period of study
- UNESP - Scholarship to be an exchange student for one semester in South Korea (GPA: 4.25/4.5)
- UNESP - Scholarship to be an intern for one semester in Argentina
- UNESP - Fellowship to be an intern for ten months in a local NGO

### Work Experience

**Lecturer – Federal University of Mato Grosso do Sul (Brazil)** Apr-Jun 2017
Taught, and prepared content, homework and exams for the General Geology undergraduate course for Civil Engineering and Environmental Engineering majors.

**Corporate Ag Engineering Intern – São Francisco Farm, Agro-Pecuária CFM (Brazil)** Sep-Oct 2014
Participated in all field activities with the farm staff (manager, office workers, stockmen and farmhands) in the production of sugar cane, cattle, eucalyptus and reforestation activities; produced a report of all the farm activities, considered the best report ever done by the farm manager.

**English Teacher – Open Beyond Kids School (South Korea)** Apr-Jun 2014
Taught English to Korean elementary school students; prepared the content for classes using books with linguistic content and storybooks; made discussions, games and prepared exams.

**Ag Engineering Intern – Association of Parents and Friends of Exceptional People – NGO (Brazil)**
Mar-Dec 2012
Participated in the production of ornamental, aromatic and medicinal plants, fruits, vegetables, and seedlings, with children with disabilities and the kitchen garden staff.

## Research Experience

**Graduate Student Researcher – VICE Lab, University of California, Merced** Jul 2019-Present
Research on the implications of changes in climate, infrastructure and policy on water allocation to agriculture, water supply, hydropower and environmental flows, and flood control, in California as part of the projects "CERC-WET: Sustainable Hydropower Operations" and "Optimizing Hydropower Operations While Sustaining Ecosystem Functions in a Changing Climate", funded by the Department of Energy and California Energy Commission. Participating in activities such as managing data and code repositories of water system simulation models, coding water and power infrastructure of four basins and their operational schedules (e.g., hydropower generation, flood control releases, environmental flow and agricultural/urban deliveries) in Python (*Pywr* library), model calibration, gray and scientific literature review (e.g., hydropower licenses), writing reports and papers, in addition to data collection, visualization and analysis of observed and simulated climate/hydrological/spatial data. Other activities include modeling the agricultural suitability of winegrowing regions on the Pacific Coast using machine learning models, remote sensing of reservoirs in R/Google Earth Engine, statistical modeling, and eventually helping others with data collection/analysis, presentations/classes, manuscript/grant writing or fieldwork.

**Graduate Intern – LabGIS, Federal University of Mato Grosso do Sul (Brazil)** Mar 2017-May 2018
Participated in research activities related to GIS/remote sensing mostly associated with data collection and analysis and literature review on mapping of land use/land cover change, classification and mapping of fires, changes in vegetation and water content on the land surface in the Cerrado (Brazilian savanna) using QGIS and Landsat imagery, at UFMS.

**Visiting Research Student – University of British Columbia (Canada)** Sep 2016-Feb 2017
Audited hydrogeology classes and did research about groundwater protection zones and forest hydrology, and literature review on land use/land cover change.

**Ag Engineering Intern – National Agricultural Technology Institute (Argentina)** Sep-Nov 2014
Participated in extension visits to farmers, lectures, field day, with field, lab and greenhouse practices and research in different essays about in vitro plant breeding, floriculture, horticulture and phytopathology of stone fruit.

**Agronomic Engineering Intern – National University of La Plata (Argentina)** Aug-Dec 2014
Participated in an essay about phytopathology in wheat, with lab and field activities, and prepared a report about the essay and its foliar diseases.

**Agronomic Engineering Intern – São Paulo State University (Brazil)** Apr-Jul 2013
Used ArcGIS, Landsat imagery, and Google Earth to map and assess forest fragmentation in protected areas and forest remnants in Jaboticabal, São Paulo, to propose a restoration plan with wildlife corridors.

## Peer-Reviewed Papers

Rheinheimer, D.E., Medellín-Azuara, J., Ramirez, A.I., Brown, C.M., Park, D., Torres, E., **Facincani Dourado, G.**, Knox, S., Garza-Diaz, L.E., Abdallah, A.M. (in preparation). OpenAgua: A web-based framework for collaborative water system modeling. **Environmental Modelling & Software**.

Parker, L.E., et al. (in preparation). Vineyard of the Future. **American Journal of Enology and Viticulture.**

**Facincani Dourado, G.**, Rheinheimer, D.E., Abatzoglou, J.T., Viers, J.H. (in review). Stress testing California's hydroclimatic whiplash: Potential challenges, trade-offs and adaptations in water management and hydropower generation. **Water Resources Research**.

Rheinheimer, D.E., Sood, A., **Facincani Dourado, G.**, Rallings, A.M., Viers, J.H. (in review). Water system simulation modeling with hydropower optimization and environmental flows: An example with Pywr. **Water Resources Research**.

**Facincani Dourado, G.**, Rallings, A. M., & Viers, J. H. (2023). Overcoming persistent challenges in putting environmental flow policy into practice: A systematic review and bibliometric analysis. **Environmental Research Letters,**18(4), 043002**.** https://doi.org/10.1088/1748-9326/acc196

**Facincani Dourado, G.**, Hestir, E.L., Viers, J.H. (2022). Establishing reservoir surface area-storage capacity relationship using Landsat imagery. In 2022 IEEE International Geoscience and Remote Sensing Symposium. **IGARSS**. https://doi.org/10.1109/IGARSS46834.2022.9884132

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**Book Chapters**

Gabas, S.G., **Dourado, G.F**., Uechi, D.A., Cavazzana, G.H., Lastoria, G. (2022). The role of groundwater in economic and social development of Mato Grosso do Sul state, Midwest Brazil. In: Groundwater, Resilient Livelihoods and Equitable Growth. **International Association of Hydrogeologists**. https://doi.org/10.1201/9781003024101

Facincani, E. M., Gregório, E. C., Amorim, G. M., **Dourado, G. F.** (2018). Depositional geoforms of the current distributary lobe of the fluvial megafan of the Aquidauana River, southeast border of the Pantanal in Mato Grosso do Sul. In: Geografia e suas linguagens: a construção de novas leituras sobre o espaço regional sul-mato-grossense (Vol. 3), **Life Editora**, Campo Grande, Mato Grosso do Sul, Brazil (in Portuguese).

San Pedro, M. M. I., et al. (2013). Factors to consider for the control of diseases that affect flowers and ornamentals. In: Sanidad en cultivos intensivos, 38-48, **Instituto Nacional de Tecnología Agropecuaria**. Buenos Aires (in Spanish).

**Reports**

**Facincani Dourado, G.**, Rallings, A.M., Viers, J.H. 2021. Unlocking life from blocked waters: Balancing trade-offs for reconciling human and environmental water needs. University of California, Merced, Merced, California. 20 ppd.

Rheinheimer, D.E., Rallings, A.M., Willis, A., **Facincani Dourado, G.**, Maskey, M., Sood, A., Cai A., Viers, J.H. 2022. Final Project Report: Optimizing Hydropower Operations While Sustaining Ecosystem Functions in a Changing Climate. California Energy Commission. 135 pp. https://www.energy.ca.gov/sites/default/files/2022-09/CEC-500-2022-008.pdf

**Scientific Presentations**

**Posters**

**Facincani Dourado, G.**, Rheinheimer, D.E., Sood, A., Cai, A., Rallings, A.M., Viers, J.H. 2023. Climate, Soils and Landscape: Applying Machine Learning and Principles of Vinecology to Sustain Current and Future Winegrowing along the Pacific Coast of the Americas. **American Geophysical Union**. San Francisco, California. December. 2023.

- Winner of the Outstanding Student Presentation Award.

**Facincani Dourado, G.**, Sarwar, A., Abatzoglou, J. T., Medellín-Azuara, J., Viers, J.H. 2023. Applying Machine Learning and Principles of Vinecology to Sustain Current and Future Winegrowing in California. **AgAID Annual Meeting**. Wenatchee, Washington. September, 2023.

**Facincani Dourado, G.**, Rheinheimer, Abatzoglou, J.T., Viers, J.H. 2023. California's climate whiplash: Uncertainty in hydropower production, agricultural and environmental water supplies. **AgAID Annual Meeting**. Wenatchee, Washington. September, 2023.

**Facincani Dourado, G.**, Rheinheimer, D.E., Sood, A., Cai, A., Rallings, A.M., Viers, J.H. 2020. The consequences of the Sierra losing its Nevada: Inconsistent water year classification and a nonstationary hydroclimate alter water allocation for humans and the environment. **American Geophysical Union**. Online. December, 2020.

**Talks**

**Facincani Dourado, G.**, Rheinheimer, Abatzoglou, J.T., Viers, J.H. 2023. CA's Climate Whiplash: Fluctuations and Uncertainty in Hydropower Production, and Agricultural and Environmental Water Supplies. **2023 Yosemite Hydroclimate Meeting**. Yosemite National Park, California. October, 2023.

**Facincani Dourado, G.** Using water-energy systems models to better inform decision-making. Technical Presentation. **Turlock Irrigation District**. Turlock, California. September, 2023.

**Facincani Dourado, G.**, Rheinheimer, Abatzoglou, J.T., Viers, J.H. 2023. Climate whiplash in California: too much to bear, too little to handle?. **European Geosciences Union**. Vienna, Austria. May, 2023. https://doi.org/10.5194/egusphere-egu23-10605

**Facincani Dourado, G.**, Willis, A., Rheinheimer, D.E., Yarnell, S.M., Rallings, A.M., Maskey, M., Viers, J.H. 2021. Water Year Typing, Nonstationary Hydroclimates, and Consequences for Management of Environmental Flows. **American Water Resources Association Summer Conference**. Online. July, 2021.

**Data and Model Repositories**

Rheinheimer, D. E., **Facincani Dourado, G.**, Sood, A., Tran, D., & Scaturchio, L. (2024). CenSierraPywr (Version v2024.02.20). [Software]. Zenodo. https://doi.org/10.5281/zenodo.10689850

**Facincani Dourado, G.**, Sood, A., Rallings, A.M., Willis, A., Maskey, M. L., Cai, A., Viers, J.H. (2022). Bias-corrected VIC historical runoff data (1950-2013) for the Central Sierra Nevada [Dataset]. Dryad. https://doi.org/10.6071/M3609B

**Other Training, Education and Contributions**

- CALVIN (Python version) – Winter 2024 course – UC Davis (7h)
- Wilderness First Aid and CPR training (valid for 2023-2024)
- Forecast Informed Reservoir Operation Colloquium – 3 week-long training at Scripps Institution of Oceanography, UC San Diego
- Groundwater, Watersheds, and Groundwater Sustainability Plans – UC Davis (15h)
- Watershed Management – São Paulo State University (120h)
- Soil and Water Degradation and Management Policies - São Paulo State University (20h)
- Introduction to Forest Fire – Kangwon National University (45h)
- Compiled the modules "Hydrological Setting", "Hydropower and environmental flows" and "Climate Impacts and Resilience" for the graduate course ES 242 - Geospatial Analysis for Watershed Science and Management, at UC Merced

Balancing trade-offs for sustainable water resources management: reconciling climate change, hydropower, and environmental flows in the Central Sierra Nevada, California

by Gustavo Facincani Dourado

Doctor of Philosophy in Environmental Systems
(Hydrology and Water Resources Engineering)

University of California, Merced, 2024

Professor Joshua H. Viers, Advisor

**Abstract**

Decisions on water allocation to humans and the environment depend on physical engineering structures, various operations and allocation policies, supplies, and demands of numerous end-users. Different assumptions of current and future scenarios can anticipate decisions that best meet human and environmental objectives, under different stressors (e.g., climate change, increased demands). Environmental water allocation especially presents intricate challenges, given the interplay of various regulations and the complexities of managing water resources across different regions. Therefore, the goal of this collection of studies is to provide new insights on reservoir operations, hydropower generation and water management in the Central Sierra Nevada, California, aiming to balance human demands, while achieving greater environmental benefits.

This work involves the use of a novel method for water-power modeling with a specific application to the Central Sierra Nevada, California introduced in Chapter 1, and used in Chapters 2 and 4. The modeling framework includes more detailed and facility-specific information to provide a more comprehensive and finer temporal resolution (daily time-step) of water allocation decisions than those found in most modeling efforts. This is a potentially crucial method for modeling water management, due to the reconciliation of water and power systems through the integration of hydroeconomic needs (e.g., hydropower operations) and rule-based simulation (e.g., instream flow requirements), which is one of the biggest challenges in modeling water systems. Better representation of real-world systems is essential to address the difficulties in water management and to analyze solutions. These models are made available for use in a broad range of scenario analyses, including different hydrological inputs (historical and future climates), electricity prices, and a variety of management objectives.

Chapter 2 delves into the nuanced landscape of environmental flow (e-flows) requirements, primarily anchored on water year types (WYTs), to understand the efficacy and adaptability of current strategies. Through an extensive examination of pertinent hydropower licensing documents, the research identifies a lack of standardized adoption of

WYTs in many river reaches, manifesting as minimal variation across different year types and limited seasonal fluctuations. Incorporating climate change projections from multiple Global Circulations Models, the study reveals significant variability in WYT distributions under existing management strategies. This variability has led to inconsistencies in e-flow management, exacerbating potential conflicts among stakeholders. To address these challenges, an adaptive strategy is proposed, employing a method to recalibrate WYT thresholds, aiming to bolster the reliability and resilience of e-flows. As a result, Chapter 3 critically analyzes the systemic barriers hindering the effective implementation of e-flows. A comprehensive systematic review and bibliometric analysis were conducted, yielding insights into the major impediments such as competing priorities of human water uses, data deficiencies, and resource and capacity limitations. To enhance the successful implementation of e-flows, the dissertation recommends a system analysis approach, utilizing modeling tools to navigate competing demands and foster holistic flow allocations based on hydroecological principles. In turn, Chapter 4 evaluates the resilience of water systems and hydropower against climate whiplash. Through 200 synthetic hydrologic sequences of different lengths of dry-wet-dry combinations, the research underscores the vulnerability of water storage and the implications for water resource management, offering policy suggestions to enhance system flexibility and resilience against climatic shocks. Finally, Chapter 5 concludes by providing policy insights and recommendations based on these studies to help inform stakeholders and decision-makers in the search for sustainable solutions to water management problems.

# Chapter 1. Introduction

Freshwater ecosystems are the most severely impacted by human actions, with disproportional loss of species in comparison to other ecosystems (Brewer et al., 2016). Most of their fragmentation has been caused by the heavy reliance on water infrastructure for irrigation, energy generation, flood control and water supply (The Brisbane Declaration, 2018). Dams and levees, for instance, disrupt the connectivity and alter the movement of water, sediment and organisms in river systems (Opperman et al., 2019). The change in sediment and bedload transport also affects the geomorphology and habitat formation in downstream reaches (Schramm et al., 2016). In particular, large dams tend to reduce the natural variability of flows, homogenizing streamflow patterns by diminishing flood flows and elevating baseflows (Brewer et al., 2016). River regulation reduces flow variability, impedes longitudinal and lateral connectivity and alters seasonal flow patterns (Thompson et al., 2018), leading to global freshwater biodiversity declines, fish species extinctions and floodplain degradation (Grantham et al., 2014a). Consequently, the altered flow affects river-dependent ecosystems that support immense biological diversity and productivity (Opperman et al., 2019).

Decisions on the allocations to meet societal needs for water supply, agricultural production, energy generation, and flood management require careful evaluation and integration of competing uses (Kendy et al., 2012). Particularly, understanding the trade-offs between allocating water for the environment and for hydropower in regulated rivers can inform decision-making about hydropower system planning, policy, and operations, specially under a changing climate (Yarnell et al., 2013). Currently, many governments have recognized the environmental water needs through policies or legal provisions to protect ecosystems and dependent communities (The Brisbane Declaration, 2018). However, the implementation of environmental flows (e-flows) has been limited in many places due to insufficient political will, lack of stakeholder support, lack of capacity and resources, institutional roadblocks and conflicts of interest (The Brisbane Declaration, 2018).

In the United States, minimum instream flows were already designed in the late 1940s (Arthington et al., 2006a), although the environmental water needs were only effectively recognized in the late 1970s, as minimum stream flow requirements, established to maintain fisheries below dams (Whipple & Viers, 2019a). Even though California is leading the implementation of e-flows in the US (Grantham et al., 2022; Schramm et al., 2016; Taniguchi-Quan et al., 2022; Whipple, 2018), the allocation of environmental water is poorly accounted for and poorly understood in the state. This has been largely due to the lack of current, transparent, and adequately detailed information to guide management, such as the nonexistence of official estimates of environmental water, lack of details for estimating applied versus net environmental water use (Gartrell et al., 2017) and over-allocation of water to agriculture (Grantham & Viers, 2014a).

In California, hydroclimatic and socioeconomic factors make the state particularly vulnerable to climate change impacts. Non-stationarity imposes a shift and increasing variability in hydrology, affecting how and when water is naturally distributed (Milly et al., 2008b), exposing aquatic and riparian species to more frequent and intense extreme hydroclimatic events (Poff, 2018). In California's Sierra Nevada, many of the larger reservoirs and water projects are operated for multiple uses, such as water supply, hydropower, flood control, environmental mitigation, and recreation (Null et al., 2010). Water allocation is determined by different "water year type" (WYT) classifications in the region, based upon historical and/or forecasted hydrologic data (Null & Viers, 2013a). WYTs are defined, forecasted, and applied by different agencies and utilities for specific regions or facilities. Consequently, changes in the distribution of WYTs under climate change are expected (He et al., 2021), which in turn will likely affect facilities that are operated under different WYT classifications unevenly due to the inconsistent categorization methods.

In addition, e-flows decrease the ability of a hydropower operator to operate exclusively based on energy prices, thereby potentially reducing revenue (Null & Viers, 2013a). As a result, maintaining e-flows can be a particular challenge, especially in central and southern Sierra Nevada, where significant agricultural and urban demands for limited water resources exist. For instance, Stewart et al. (2020), using observed data the authors found that water management under drought in the Tuolumne watershed emphasizes on safeguarding urban drinking water supplies, meanwhile e-flows and agricultural deliveries are disproportionately affected. Therefore, the long-term water planning and management under the assumption of a stationary hydrology based historical records is inadequate (Milly et al., 2008a).

Consequently, considering the potential impacts of climate change is a key factor to further understand the increasing challenges of meeting human and environmental demands. Previous studies have considered the response of high-elevation hydropower to climate change in the Sierra Nevada (Madani & Lund, 2009, 2010a; Rheinheimer et al., 2014; Vicuna et al., 2007). Null et al. (2010) used the Water Evaluation and Planning System (WEAP) to consider the effects of increased air temperature (2, 4 and 6ºC) on the hydrologic response of the Sierra Nevada and its consequent impacts on hydropower generation. Mehta et al. (2011) and Rheinheimer et al. (2016) also considered a similar approach to study potential hydrologic impacts on hydropower generation in northern Sierra Nevada and Upper Yuba River watershed, respectively. Similarly, Kiparsky et al. (2014) also used this approach to assess water supply reliability in the Merced and Tuolumne rivers, while others (e.g., Jager & Martinez (2012) and Jager et al. (2015)) have used operational and environmental flow releases to better understand impacts to species of concern.

As noted by Loucks & van Beek (2017), planning, designing, and managing water resource systems inevitably involve impact prediction, which can be assessed through mathematical simulation and optimization models. Models assimilate different time-varying boundary conditions (e.g., streamflow), operational rules and a representation of the water system to produce a prediction of state in a system overtime (e.g., daily, weekly, or monthly time-steps), although with several limitations (Tomlinson et al., 2020). For instance, Rheinheimer et al. (2023) calls for the need of better representing hydropower operations in water and energy systems, by reconciling their competing priorities due to the modeling discrepancies. As discussed by the authors, hydropower is typically represented as a single-priority output in energy models, meanwhile in water models the competing demands are increasingly resolved for non-energy components (e.g., hydrological input, agricultural, and urban demands). According to the authors, oversimplified assumptions on operational objectives, constraints, and priorities (e.g., unconsidered policies, instream requirements, flood control rules, and physical constraints) as well as coarse spatial and temporal resolutions, among other limitations, are part of the reasons that modelling efforts can produce misleading results. Considering that, Rheinheimer et al. (2021) developed the *CenSierraPywr*, a multi-objective water system simulation model for the Central Sierra Nevada in California, composed of the basins that contribute the most to the San Joaquin River (SJR) flow. This coupled water-energy modeling framework has innovative features and is capable of running various climate and management scenarios.

To address the aforementioned research gaps, this study consists of three chapters based on the major themes surrounding hydropower development, especially in the Central Sierra Nevada, California. The chapters will cover the subsequent major topics: 1) the perspectives on overcoming persistent challenges of putting e-flow policy into practice, 2) the impacts of climate whiplash (i.e., hydrological extremes) on hydropower generation and water management, 3) the inconsistency among WYT classifications and management implications of adopting rules based on a stationary climate.

Chapter 1 is composed of a systematic literature review and bibliometric analysis, meanwhile Chapters 2 and 3 involve the use of the *CenSierraPywr*. These chapters, their respective approaches and goals are described below. These studies aim to provide a better understanding of system behaviors, climate change impacts, management options and trade-offs among water uses for e-flows and human objectives at a scale relevant to facility operations and in a more realistic way. Presumably this collection of studies is going to provide useful outcomes for better decision-making for development plans, management policies and reservoir operations in the SJR Basin. These can also bring insights for water management in California in general, as well as in regions with similar characteristics and challenges to achieve more sustainable solutions for water management problems.

## 1.1. Study area

The study area is comprised of the four major basins in the Central Sierra Nevada, California, that contribute the most to the SJR, one of the two main rivers that flow to the Sacramento-San Joaquin River Delta. These basins include the Stanislaus, Tuolumne, Merced and Upper San Joaquin rivers. The basins are mostly formed by highly regulated rivers with high-altitude reservoirs and hydropower facilities and low altitude, multipurpose "rim" dams that store water for water supply and flood control, regulating the flow entering the SJR.

This region has a Mediterranean-montane climate, with a notably variable hydrology (Null & Viers, 2013a), where hydropower has historically accounted for an about 25% of in-state California's hydroelectricity. The facilities are operated by several distinct utility companies, and therefore, regulated differently by each owner/operator in many cases. The basins vary from highly managed systems driven mostly by hydropower generation (Upper San Joaquin and Stanislaus), to less regulated basins mostly managed for agricultural and/or urban deliveries (Merced and Tuolumne). In the Tuolumne basin, urban water deliveries also occur mainly to the San Francisco Public Utilities Commission (SFPUC), but also to the Groveland Community Services District.

### 1.2. *CenSierraPywr* model

The study area is represented in the *CenSierraPywr* modeling framework (**Figure 1-1**), a daily water system simulation model implemented with *Pywr* (Tomlinson et al., 2020) in Python. *CenSierraPywr* is composed of four independent models created for each of the major basins in the SJR system (Rheinheimer et al., 2022), for which the code, and associated workflow and changes are hosted on GitHub (Rheinheimer et al., 2024). A linear programming basis is used to allocate water within a water network given the system's physical and legal constraints (e.g., minimum and maximum instream flow requirements, reservoir storage capacity) and water value, based on pre-defined rules or numerical input.

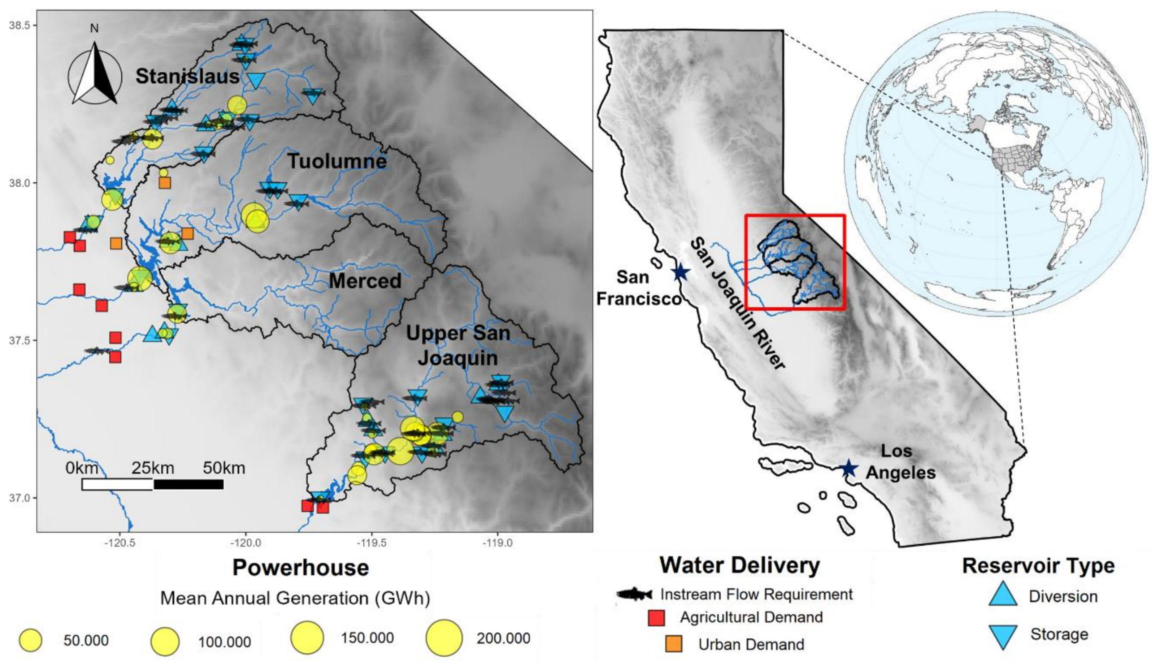


**Figure 1-1.** Study area and main nodes represented in the *CenSierraPywr* framework

This modeling framework was derived from a water system schematic originally used in WEAP in previous efforts that encompassed all of the west slope of the Sierra Nevada (Rheinheimer et al., 2012, 2014). The data available for the Central Sierra Nevada were updated, corrected, and extended to include more detailed information, that allow the examination of basin-scale options and trade-offs in a more realistic way based on real-world system constraints. That includes instream flow requirements prescribed in Federal Energy Regulatory Commission (FERC) licenses and other regulatory agreements, and the inclusion of the rim reservoirs, their powerhouses, flood control rules and downstream dependent urban and agricultural water users (SFPUC, irrigation districts and the Central Valley Project). Water allocation is determined by the relative water value given for each node/link, affecting the model decision based on the “cost” of moving/storing water. More details on modeling framework are further described in **Appendix S2: *CenSierraPywr*’s details**.

*CenSierraPywr* can also be coupled with other models (e.g., hydrologic and energy models), and optionally allow the inclusion of energy price-based optimization for hydropower allocations, a key methodological advancement over typical water system models. The hydropower optimization component is composed of a 12-month, monthly time step, planning-scale optimization model with imperfect foresight of hydrology and perfect foresight of energy prices. However, the hydropower optimization is currently employed only in the hydropower generation-driven basins (Upper San Joaquin and Stanislaus), to optimize discretionary hydropower releases. These hydroeconomic decisions are assumed to occur at the hourly time step across days and months and incorporated into the model by piecewise linear price curves.

Outputs from hydrologic and energy models can be used as inputs into this modeling framework to simulate historical and future climate scenarios and current or alternative regulatory/infrastructure constraints on basin operations, that can pose new challenges and opportunities to managers. More details on hydrological data inputs and their bias-correction for use into *CenSierraPywr* are provided in **Appendix S2: *CenSierraPywr*’s details**. The outputs from *CenSierraPywr* include reservoir storage, hydropower flow and generation at all facilities, instream flows in river segments where instream flow requirements exist, and urban and agricultural water deliveries. Considering that, this modeling tool can be used to examine these uncertainties and to develop alternative scenarios to identify trade-offs that provide insights for management-relevant decision making.

### 1.3. Research team

This work was essentially and primarily one of the outcomes of the projects “CERC-WET: Sustainable Hydropower Operations” and “Optimizing Hydropower Operations While Sustaining Ecosystem Functions in a Changing Climate”, funded by the Department of Energy and California Energy Commission (CEC), respectively. The overarching goal of these projects was to develop *CenSierraPywr*, a water systems simulation, with a hydropower optimization modeling framework, to consider institutional and physical constraints placed on hydropower operations and water allocation. The team that planned and executed this project included principal investigator Joshua H. Viers; senior contributors Dr. Daniel Nover, Dr. David E. Rheinheimer, and Ms. Anna M. Rallings; and researcher affiliates Dr. Ann Willis, Dr. Mahesh L. Maskey, and Dr. Aditya Sood, and Ms. Jenny Ta; as well as graduate students Mr. Alan Cai, Dr. Vicky Espinoza, Dr. Britne Elizabeth Clifton, Dr. Zhuo Hao and me. Beyond those names, this research received essential contributions from the technical advisory committee and collaborators from public and private institutions who provided operational insights, feedback on model outputs, and guidance on project development. All of this body of work resulted in the CEC report available at: https://www.energy.ca.gov/sites/default/files/2022-09/CEC-500-2022-008.pdf, among several other studies and publications, including the work that resulted in my master's along the way (**Appendix S1: Partial satisfaction of the requirements for the degree of Master of Science**), this doctoral dissertation, and CERC-WET reports.

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## Chapter 2. Reliability and resilience of environmental flows under uncertainty: reconsidering water year types and inconsistent flow requirements in California

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**Keywords:** instream flows, climate change, adaptive management, San Joaquin River

### Abstract

Environmental water allocation in California is a complex and highly regulated process that involves a combination of federal, state, and local laws, as well as the management of water resources by various government agencies and stakeholders. Environmental flow (e-flow) requirements measured by volume, timing, and duration are often based on a codified typology of annual runoff at the supplying facilities, commonly referred to as water year types (WYTs). In this study, we examined hydropower licenses and related documents of the major water and power projects in the Central Sierra Nevada to catalog instream flow requirements for ecosystem benefit by WYT where codified. We used a case study of relevant basins within the greater San Joaquin River, consisting of the Stanislaus, Tuolumne, Merced and Upper San Joaquin) to identify how environmental flow mitigation and allocation strategies vary across and within different basins as a function of management priority and authority. In addition, we assessed the impacts of climate change on hydrology, on the frequency of WYTs identified, and on the reliability and resilience of e-flows using future projections (2031-2060) of 10 Global Circulation Models. We then propose a potential adaptation strategy using a 30-year moving percentiles approach to recalculate WYT thresholds. We identified 8 WYT classification systems in 9 projects that classify WYTs using numerical thresholds supported by historic and/or forecasted hydrologic data and can be established independently per facility or hydropower project, using a variety of methods (e.g., indices, models, numerical thresholds). However, in most river reaches WYTs are not adopted, as e-flows in many cases include little to no variation across different year types, and also limited seasonal fluctuations. In the context of climate change, the hydrological impact across future projections is generally not statistically significant in most scenarios across basins. However, variability in WYT distributions under current management strategies is evident and statistically significant in all projects and scenarios. Disparities in impacts are observed among and within hydropower projects, with some river reaches showing negative impacts on reliability and resilience. The

adaptively recalculated WYTs can generally boost reliability and improve resilience, but simply updating existing WYT thresholds without flexible regulatory frameworks reconsidering WYTs, e-flows thresholds, may not yield substantial improvements. Challenges in managing e-flows in California within regulatory and hydroclimatic contexts are intricate due to the lack of standardized approaches, leading to inconsistencies and potential conflicts among stakeholders, that will likely be exacerbated by climate change. Thus, we emphasize that targeted, site-specific, and adaptive management strategies are crucial, as well as the need for a harmonized and consistent approach to defining and applying WYT categories and methods and/or e-flow assessment approaches.

## 2.1. Introduction

California is a leader in the implementation of environmental flows (e-flows) (Schramm et al., 2016); however, the allocation and delivery of water for environmental objectives is often poorly accounted for in the state. The implementation of e-flows generally aims to reduce the environmental impacts of river regulation and water diversions, which is most often through the adoption of minimum instream flow requirements (MIFs) (Facincani Dourado et al., 2023). MIFs are often a subjectively determined flow (i.e., discharge) or water level (i.e., stage) maintained to provide in-channel (e.g. environmental allocation) and off-channel uses (e.g. agricultural allocation) (Whipple & Viers, 2019b). In California, the management decisions on when and how much water is allocated to MIFs are many times based on the typology of annual runoff at the supplying facilities, commonly referred to as “water year type” (WYT) (Null & Viers, 2013b).

For instance, the San Joaquin Valley Index (SJVI) is a water year index used to categorize WYTs based on the unimpaired runoff from the main tributaries to the San Joaquin River (SJR). This index is used to classify water years as either critically dry, dry, below normal, above normal, or wet, for environmental and specific agricultural water delivery allocation schemes (Null and Viers, 2013). The majority of e-flow thresholds and schedules are defined in hydropower licenses, which can be based on WYTs, and are applied to specific points within natural river channels below storage or diversion dams (Rheinheimer et al., 2022).

Non-federal hydropower projects, which account for more than half of the total hydropower capacity in the US, are regulated by the Federal Energy Regulatory Commission (FERC) (Office of Energy Projects, 2017). To date, FERC has yet to consider climate change in the licensing process for e-flow assessments and WYT calculations (Federal Energy Regulatory Commission, 2009; Viers, 2011a; Viers & Nover, 2017). The competing water use priorities, differing e-flow methodologies and authorities involved already make e-flow implementation especially challenging, and climate change increases that complexity due to nonstationarity (Chen & Wu, 2019b; Facincani Dourado, 2023; Milly et al., 2008b).

To explore the impacts of this information gap, previous studies have assessed how climate change affects WYT distribution in the San Joaquin Valley Index (SJVI) and

Sacramento Valley Index, using hydrological data from 6 Global Circulation Models (GCMs) from CMIP3 (Null & Viers, 2013) and 4 GCMs from CMIP5 climate projections (He et al., 2021). The authors found that the increasing incidence of hydrological extremes will introduce inaccuracies and uncertainties into the long-term regulatory framework, potentially compromising its stability and undermining efforts to sustain especially environmental water deliveries. Anticipating climate change impacts on water allocations is vital to adapt management frameworks to a climate-driven shift in hydrology. Envisioning that, Rheinheimer et al. (2016) also used an assemblage of 4 GCMs from CMIP5 to consider climate change impacts in the WYT distribution of the Yuba River Index in the Upper Yuba River watershed, using a hydrologic model developed for the western Sierra Nevada. The authors' findings point to the need for climate-adaptive options for water typing to help maintain instream flow requirements (IFRs).

If current water year typologies remain static while the distribution of hydroclimatology shifts in timing and volume, resulting reservoir and hydropower operations could be highly affected throughout the cascade of water conveyance systems (Maskey et al., 2022). Furthermore, infrastructural and operational constraints (e.g. reservoir sizes, flow release schedules, flood control capacity) could potentially limit any future adaptation strategy (Willis et al., 2022). Given that water years are classified through numerical thresholds based pm historical and/or forecasted hydrologic data, as calculated independently per facility or hydropower project using a variety of methods (e.g., indices, models, statistical cutoffs), any shift in distribution or non-stationary behavior is likely to jeopardize operational objectives.

In this study, we illustrate this issue by identifying the WYT classifications found in the SJR basin, in the Central Sierra Nevada, California. Through an in-depth analysis, we aim to investigate (1) the flow mitigation requirements adopted to allocate water and reduce environmental impacts, (2) the characteristics and discrepancies in the independent WYT classifications used, (3) the efficacy of these WYT classifications given hydroclimatic alteration, and (4) the potential ramifications of projected changes on environmental water allocations within and across study basins. The objective of this study is to evaluate the impacts of climate change and management alternatives on the resilience and reliability of e-flows, exploring potential trade-offs, and proposing adaptive strategies that can mitigate adverse consequences. Results from this study can provide can inform more robust and adaptive water resource management practices given the hydroclimatic alteration now underway (Ficklin et al., 2012; He et al., 2021; Maskey et al., 2022; Mehta et al., 2011).

## 2.2. Methods

### 2.2.1. Study area

The Sierra Nevada has been an area of interest to the scientific community given the interplay of climate change and hydropower energy in the region (Mehta et al., 2011; Rheinheimer, Bales, et al., 2016; Vicuna et al., 2007; Vicuña et al., 2011; Young et al., 2009). The Central Sierra Nevada has been identified as the area more prone to hydroclimatic impacts in California, where it is also a key supplier of surface water for agricultural and urban demands (Null, Viers, et al., 2010; Viers & Nover, 2018). This study region is comprised of the four major basins that contribute the most to the SJR, which is one of the two main rivers that flow to the Sacramento-San Joaquin River Delta. These basins include the Stanislaus, Tuolumne, Merced and Upper San Joaquin rivers. Three of the four basins are highly regulated, consisting of small, high-altitude reservoirs and numerous hydropower facilities. All four basins include large, terminal low-altitude, multi-purpose storage reservoirs that regulate the flow entering the SJR. This complex water management system is operated by several distinct entities, and therefore, regulated differently by each owner/operator in many cases.

### 2.2.2. *CenSierraPywr* model

The study area is captured within the *CenSierraPywr* modeling framework, a tool designed for simulating these regional water systems on a daily resolution (Rheinheimer et al., 2022). This framework developed on *Pywr* (Tomlinson et al., 2020), a Python-based platform, consists of four models tailored to each basin within the SJR system (**Figure 2-1**). We employed *CenSierraPywr* to simulate the instream flows in the region, as it uses linear programming to manage water distribution across various links (such as rivers and canals) and nodes (such as reservoirs, powerhouses, and water demand for agriculture and instream flow requirements) while considering real-world physical and operational constraints. Water allocation is prioritized based on the relative water value given to storing/moving water in the network, in which the objective function aims to minimize the costs in each time step; environmental water allocation has the highest priority (i.e., lowest cost).

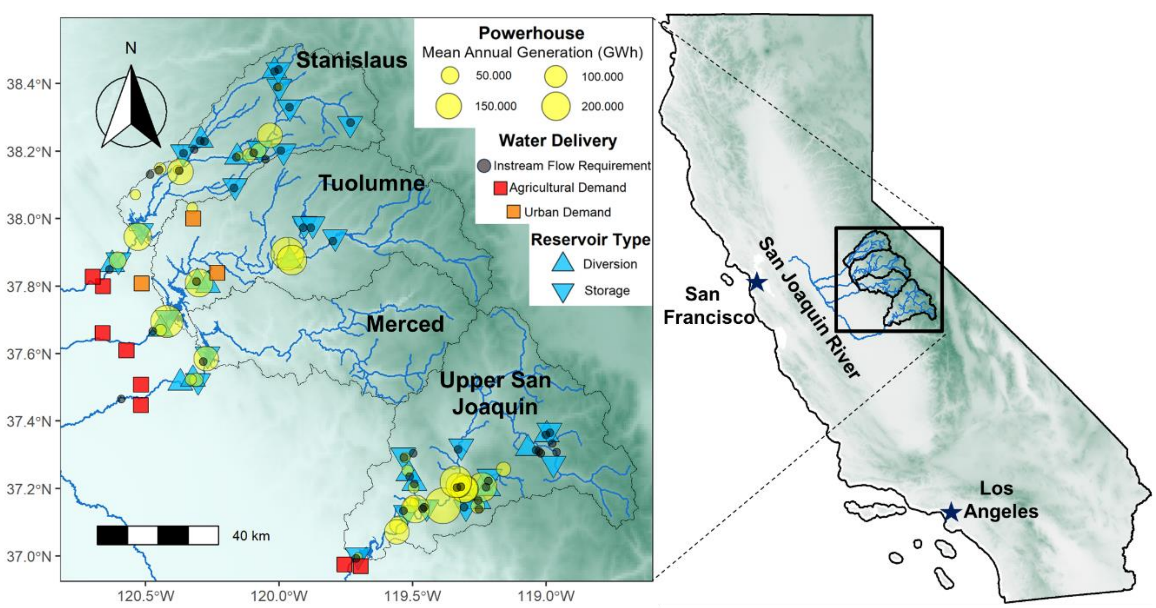


**Figure 2-1.** Nodes added to the *CenSierraPywr* framework in the study area. Canals and other conveyance infrastructure are not represented here.

The hydrological model inputs are streamflow data derived from runoff and baseflow outputs of the Variable Infiltration Capacity (VIC; Liang et al., 1996) model. The gridded streamflow data at a 1/16° (~6 km) resolution for water years 1951-2010 were obtained from Livneh et al. (2015), downscaled to subbasin level using the *raster* R package (Hijmans, 2020) using a normalized area-weighted approach. Bias correction was then applied at the basin level using historical unimpaired flow estimates from the California Data Exchange Center (CDEC) (CDWR, 2016), through the *hyfo* R package (Xu, 2020), and further bias-corrected at the sub-basin level with US Geological Survey (USGS) data for specific gauges that had at least 15 years of continuous and complete data (**Figure S2.1-1**). Model calibration was then conducted using recent observed reservoir operations (1981-2010) and historical reservoir storage gauges, flow gauges, and powerhouse electricity generation data from USGS and the Energy Administration Information.

*CenSierraPywr* offers optional hydropower optimization based on energy prices for the basins in which energy generation is their primary objective (Stanislaus and Upper San Joaquin), particularly beneficial for hydropeaking facilities. This optimization module operates at a planning scale with an 8-month, monthly time step, imperfect hydrology foresight and perfect energy price foresight. It schedules discretionary releases using piecewise linear price curves derived from wholesale energy prices from the HiGRID model, developed by Tarroja et al. (2016, 2019). The price curves are transformed into relative costs within *Pywr*, allowing for hourly hydroeconomic decisions. Currently, 2009 prices are used due to their stability and representativeness of modern energy demand (Rheinheimer et al., 2022). For the non-optimized basins, the model is driven by hydrology alone following existing operational objectives, with hydropower generation considered a secondary benefit.

*CenSierraPywr* simulates 47 river reaches of the study area, including their minimum instream flow schedules (**Figure S2.1-2**) according to water year typing, maximum flow requirements, and sub-daily constraints on flow variability (i.e., ramping rates, especially present in the Stanislaus River). To manage this latest complexity, the model employs piecewise linear functions to break down flow schedule regimes into three parts, corresponding to different costs of water allocation, in which minimum flows thresholds are prioritized, flows greater than maximum flow thresholds are penalized, and flows within both are neutral. Further details regarding model schematics, inputs, parameters, and assumptions can be found in Rheinheimer et al. (2022).

### 2.2.3. Analyses

#### 2.2.3.1. Flow mitigation requirements

We conducted a manual review of FERC hydropower licenses obtained from the FERC e-library (http://www.ferc.gov/docs-filing/elibrary.asp), and related documents publicly available (e.g., State Water Resources Control Board Order, Water Quality Control Plan, Environmental Impact Assessments, SJR Restoration Program reports) of the 16 major water and power projects found in the region. These include 28 storage reservoirs and 35 powerhouses, besides many diversion dams, canals, and aqueducts, operated by 12 utility companies, irrigation districts or government agencies. The review was followed by keyword searches to identify environmental flow mitigation requirements, including but not limited to, flow schedules and their respective water year typing.

#### 2.2.3.2. Water year type classifications

*CenSierraPywr* can be coupled with streamflow data from other models, such as VIC outputs from projections of Global Circulation Models (GCMs). For this study we adopted mid-21st century conditions (2031-2060) from 10 GCMs that have been identified by the California Department of Water Resources (Lynn et al., 2015) and California's 4th Climate Change Assessment (Herman et al., 2018) as best representing regional hydrology. Data were originally developed by Pierce et al. (2016) and subsequently downscaled to a 1/16° (~6 km) resolution using the Localized Constructed Analogues (LOCA) statistical approach (Pierce et al., 2018). Like the Livneh dataset, streamflow data from GCMs were downscaled and bias corrected to the subbasin level as a model inputs.

Following (Saplıoğlu & Güçlü, 2022), we used the non-parametric Mann-Whitney U test to test whether the distribution of more recent historical (1981-2010) annual streamflow data – a period in which all hydropower projects were already implemented – significantly differs from earlier historical data (1951-1980) and future projections (2031-2060). . The WYT classification frequencies were compared and assessed for distributional consistency under climate change under historical and future projections. We used the two-tailed Fisher's exact test as a robust nonparametric method to account for distribution of categorical variables given small sample sizes (Carlisle et al., 2011), and to assess whether WYTs significantly differed between the GCM predictions and the historical period. For the latter analysis, we tested the null hypothesis that there was no statistical difference between the distribution of historical (1981-2010) and future (2031-206) WYTs ($p < 0.05$). All statistical analyses were carried out in R version 4.0 (R Core Team, 2022).

#### 2.2.3.3. Environmental flows

E-flows deliveries were assessed through the volumetric reliability ($R_v$), represented as the actual delivered portion of the total demand, following Jain & Bhunya (2008) and Nagy et al. (2013), as:

$$R_v = 1 - \frac{\int_{Q<D}(D-Q)dt}{\int_0^T DdT} = 1 - \frac{\sum \Delta V}{TD} \tag{1}$$

in which, $\Delta V$ represents the quantity of shortfall within a period $T$ during which the supply $Q$ falls below the constant draw-off rate $D$ at time $t$. Furthermore, we calculate the resilience ($R_s$) index as described by Hashimoto et al. (1982), to evaluate how probable a recovery from failure is, once failure in achieve the demand has occurred, defined as:

$$R_s = \frac{\lim_{n\to\infty}\left(\frac{1}{n}\right)^n \sum_{t=1}^{n} Wt}{1 - \lim_{n\to\infty}\left(\frac{1}{n}\right)^n \sum_{t=1}^{n} Zt} = \frac{\rho}{1-\alpha} \tag{2}$$

where, $Zt$ denotes a satisfactory system state, and $Wt$ the transition from a satisfactory to an unsatisfactory state within an $n$-period. Hence, $\alpha$ represents the probability of the system being in a satisfactory state, and $\rho$ signifies the likelihood of the system transitioning from satisfactory to failure during a given period $t$.

Volumetric reliability was calculated monthly and resilience annually for each river reach and summarized by WYT classification; classification systems used across multiple river reaches were summarized using weighted averages, so that each reach was assigned weights correspondent to their respective daily flow contribution. Consequently, the resilience index was determined for each climate scenario based on the total instances of rebounding from demand-related failures.

#### 2.2.3.4. Adaptation to climate change

As previously mentioned, WYTs are generally set through fixed thresholds based on short-term forecasts and/or historical observations of inflows into reservoirs (He et al., 2021; Null & Viers, 2013b), which are established during the licensing process, and therefore are inflexible for the validity of the license, generally 30-50 years (Viers, 2011; Viers & Nover, 2018). In this study, we propose the process of recalculating percentiles annually from annual streamflows based on a moving time window instead, which can be represented as:

$$P_{Year(t),p} = Percentile(S_{t-i,1}\ S_{t-i,2}, S_{t-i,m} \ldots , S_{t,m-1}, S_{t,m}) \tag{1}$$

where $P_{(t),p}$ is the $p^{th}$ percentile of streamflows calculated at year $t$, $S_{t,m}$ is the streamflow value at year $t$ and position $m$ within a moving time window, *Year(t)* represents the year $t$ index. And, $i$ = 0, 1, 2 … $n$-1, where $n$ is the total number of years of the moving time window. Here, we adopt n = 30, as 30-year time windows are the shortest license's validity period and are generally adopted in hydrological and climatic studies for capturing meteorological patterns and long-term variability, smoothing out short-term variations and providing statistical stability for 'normal' or 'mean' conditions (Bakker et al., 2011; Poórová et al., 2023; WMO, 2015). We conducted targeted sampling of specific WYT classifications to evaluate the efficacy of this climate change adaptation approach. Subsequently, for the chosen river segments, we quantified shifts in both reliability and resilience resulting from climate change, relative to historical benchmarks serving as the baseline. To gauge the effectiveness of this adaptation strategy, we further analyzed alterations in reliability and resilience in comparison to the established baseline.

## 2.3. Results

### 2.3.1. Flow mitigation requirements

We identified 48 stream reaches in which minimum instream flow requirements (MIFs) are prescribed in the licenses, which are many times not dependent on WYTs (**Table 2-1**). In addition, many reaches also include other IFRs, such as ramping rates (RR), flushing and/or supplemental flows (e.g., outmigration pulse flow) and maximum flows (MAF). Minimum flows aim to protect and restore particularly rivers, streams, wetlands, and habitats for fish and wildlife, meanwhile maximum flows are applied in certain locations to avoid erosion, protect water quality and riparian habitat, or set a limit based on the channel conveyance capacity (FERC, 2003a). Flushing and/or supplemental flows can include attraction pulse and out-migration pulse flows (i.e., flows used to attract upstream-migrating adult fall-run Chinook salmon, and Chinook salmon smolts, respectively) (FERC, 2019b). In addition, ramping rates seek to avoid sudden fluctuations, changing the flow of water in a controlled and gradual manner (SJRRP, 2017).

**Table 2-1.** The operator, hydropower project, the dependencies of their environmental flow schedules and their related documents in the San Joaquin River basin. Flow prescriptions are MIF = Minimum Instream Flow; MAF = Maximum Flow Requirements; RR = Ramp Rate; F/S = Flushing or Supplemental flows.

| *Basin* | *Operator* | *Hydropower Projects (n)* | *Flow prescription (n)* | *Dependency (n)* | *Source* |
|---|---|---|---|---|---|
| **Stanislaus** | Northern California Power Agency | Upper Utica (2) | MIF (1)<br>MIF and RR (1) | None (2) | (FERC, 2003d) |
| | Utica Power Authority | Utica (1) | MIF and RR (1) | WYT (1) | (FERC, 2004) |
| | South San Joaquin and Oakdale Irrigation Districts | Beardsley-Donnells (1) | MIF, RR and F/S (1) | WYT (1) | (FERC, 2006) |
| | Pacific Gas & Electric Company | Spring Gap-Stanislaus (4) | MIF and RR (4)<br>MAF (2)<br>F/S (1) | WYT (4) | (FERC, 2009) |
| | Utica Power Authority | Angels (2) | MIF, RR and F/S (2) | None (2) | (FERC, 2003e) |
| | Oakdale and South San Joaquin Irrigation Districts | Sand Bar Water Power (1) | MIF and RR (1) | None (1) | (FERC, 1983) |
| | Calaveras County Water District | North Fork Stanislaus (5) | MIF and RR (5) | None (5) | (FERC, 1982b, 1997) |
| | Pacific Gas & Electric Company | Phoenix (1) | MIF and RR (1) | None (1) | (FERC, 1994) |
| | Tri-Dam and Stockton East Water District | Goodwin (1) | MIF and RR (1) | WYT (1) | (NOAA, 2009) |
| **Tuolumne** | San Francisco Public Utilities Commission | Hetch Hetchy (3) | MIF, RR and F/S (3) | None (2)<br>WYT (1) | (SFPD, 2008) |
| | Turlock and Modesto Irrigation Districts | Don Pedro (1) | MIF, RR and F/S (1) | WYT (1) | (FERC, 2019a) |
| **Merced** | Merced Irrigation District | Merced River (2) | MIF (2)<br>RR and F/S (1) | None (1)<br>WYT (1) | (FERC, 1964) |
| **Upper San Joaquin** | Southern California Edison Company | Big Creek (18) | MIF and RR (18) | None (10)<br>WYT (8) | (FERC, 1959, 1978, 2003c; SCE, 2000) |
| | Pacific Gas & Electric Company | Crane Valley (4) | MIF and RR (4)<br>MAF (1) | None (4) | (FERC, 2003b) |
| | Pacific Gas & Electric Company | Kerckhoff (1) | MIF (1)<br>RR (2) | None (1) | (FERC, 1979) |
| | US Bureau of Reclamation | Friant Division (1) | MIF and RR (1) | WYT (1) | (SJRRP, 2017) |

**Figure 2-2** shows that 28 of the 49 river reaches do not have flows varying by WYT (0 WYTs); and, in practice, 29 have no changes in-between years (i.e., 'Used interannual flow variations', from the 'Prescribed interannual flow variations'). Meanwhile, certain projects and/or facilities adopt 1-6 variations of WYTs, which can be defined using 8 different classification systems found in their licenses. For instance, e-flows in the Spring Gap-Stanislaus Project just use 3 WYT categories, even though 5 exist in the license based

on inflows into Stanislaus River's rim dam (FERC, 2006b). In this case, different WYTs adopt the same flow schedule in the licensing process (e.g., 'normal' and 'wet' years, or 'critically dry' and 'dry' years, are assigned the same flow thresholds). In addition, no within-year (seasonal) change in flows is present in flow schedules of 23 reaches. The natural flow dynamics is also completely removed from at least one WYT (when one or more different categories are selected) in 28 reaches, in which seasonal variations are not prescribed, and 44 reaches have four or fewer different magnitudes of flows being prescribed at different times throughout the year (excluding S/F flows prescribed apart from MIF). The lower Stanislaus River is the reach with the most prescribed seasonal variations, with changes in flows occurring between 19 and 48 times a year below the Goodwin Project, depending on the WYT (NOAA, 2009).

In addition, seven river reaches are required to keep temperature targets in the region particularly for salmonid spawning and egg incubation. Among them is the lower Tuolumne River, currently considered impaired due to elevated temperatures. However, temperature needs required for salmonids in the licensing process are based on populations from the Pacific Northwest (EPA, 2003), even though their distribution may be locally adjusted to warmer temperatures relative to northern populations (FERC, 2019b). Another distinction between policies is an exception for not meeting MIFs, a 'release inflows' policy adopted in more than half of the river reaches. This gives operators a certain flexibility in which when natural inflows into a reservoir are lower than the prescribed MIFs, they are allowed to release the natural inflows instead.

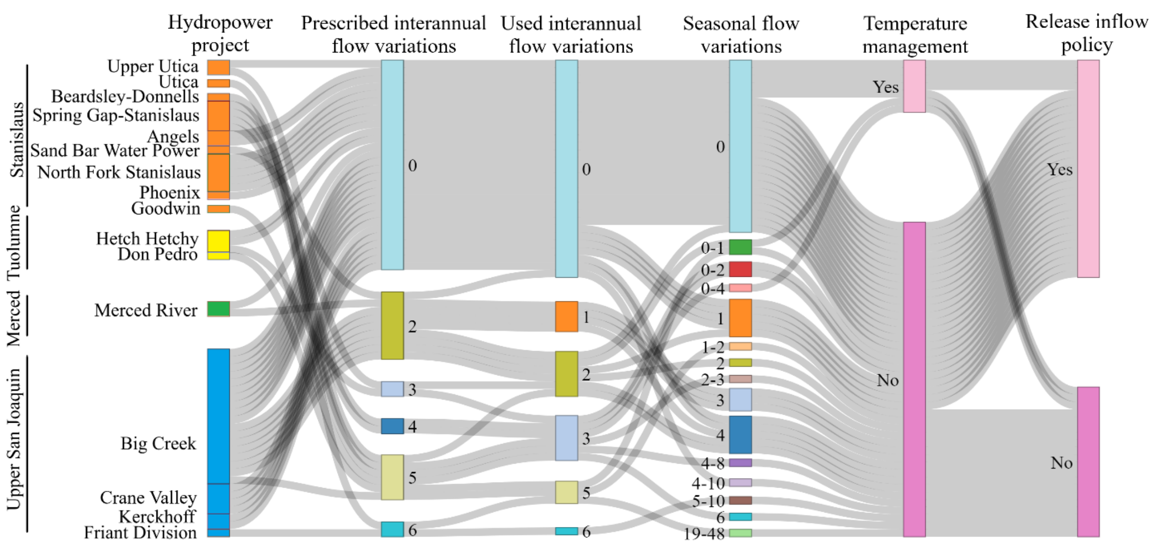


**Figure 2-2.** Sankey diagram of river reaches with minimum instream flow requirements (MIFs) in the region per hydropower project, the number of flow variations based on water year types prescribed and implemented, the number of seasonal variations adopted, in addition to the occurrence of temperature management and a 'release inflows' policy. Basins are organized from north to south, and hydropower projects from upper to lower watersheds.

### 2.3.2. Water year type classifications

There are five dimensions generally considered to assign and apply WYTs; the classification depends upon (1) the location where inflows are forecasted, (2) the methods and (3) time period used to calculate it, (4) the date(s) when it is performed, and (5) the duration of validity of water year classification. WYTs are generally defined, forecasted, and applied by different agencies and utilities for specific regions, hydropower projects or facilities within them (**Table S2.1-1**) and that leads to inconsistent classifications among and within watersheds (**Figure 2-3**). As shown in **Table S2.1-1**, the estimated natural inflows into Millerton Lake (terminal reservoir) in the Upper San Joaquin basin's outlet are used to set instream flow schedules of river reaches upstream, in the Big Creek Hydroelectric Project. Each WYT classification has its own operator, applicable area, and forecast period and method. Forecasts are often provided by the California Department of Water Resources (CDWR) and/or US Bureau of Reclamation (USBR).

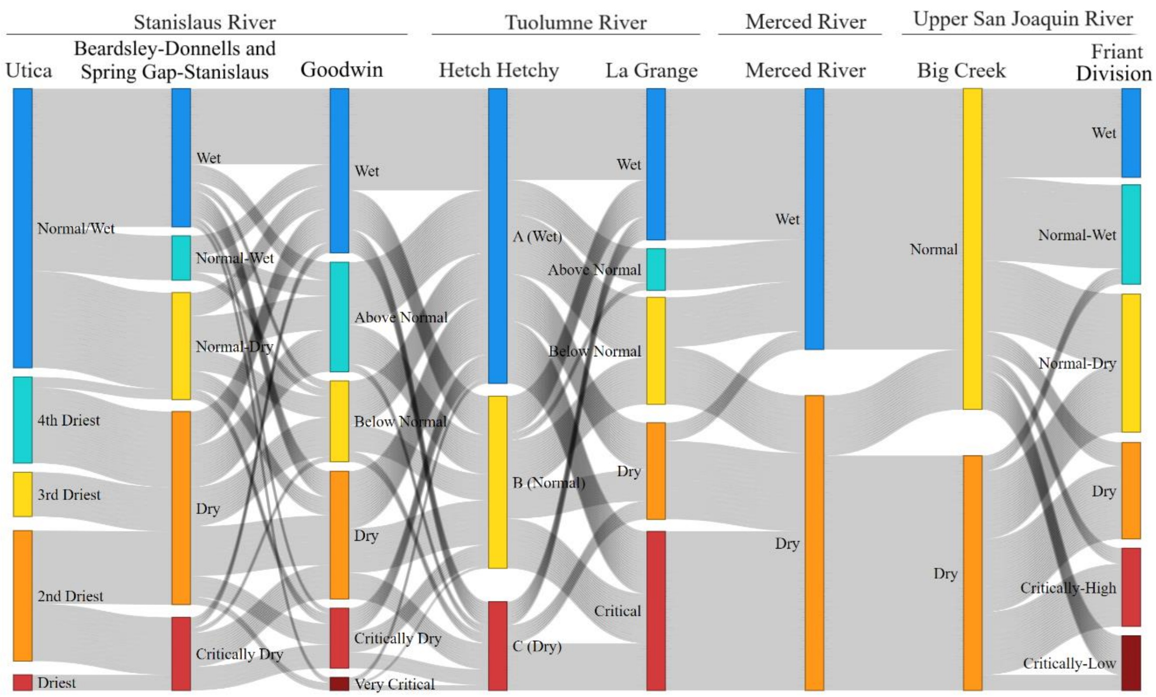


**Figure 2-3.** Sankey diagram mapping historical (1951-2010) occurrences of water year types of different classification systems found in the San Joaquin River basin. Basins are organized from north to south, and hydropower projects from upper to lower watersheds.

Considering climate change, results indicate a change in the distribution of WYTs and that facilities operated under different water year type categorizations will likely be affected unevenly due to the inconsistent categorization methods (**Figure 2-4**). Consequently, these different water year typologies will likely affect asynchronously how water is distributed in the system, making it even more difficult to manage resources in order to meet multiple demands. For instance, 9 and 10 of the 10 GCMs indicate an increase in the frequency of Dry water years for the water year typing defined for the WYT classifications used for the Big Creek and Hetch Hetchy, respectively.

Wilcoxon Rank Sum test results (**Table 2-2**) show that there is a statistically significant difference in annual unimpaired streamflows only between the future predictions of CNRM-CM5 (wettest GCM) for all basins, and of the driest GCMs for especially for the USJ. Meanwhile, no significant change occurs when compared to the more recent historical data and almost all other future scenarios. The statistical significance implies that it is unlikely this hydrological shift occurred by random chance alone, especially as these are knowingly the wettest and driest scenarios for the mid-21$^{st}$ century in the region (Maskey et al., 2022; Rheinheimer et al., 2022). However, according to the Fisher's exact test results, these two scenarios do not specifically cause more statistically significant differences in WYT frequency. The Fisher's exact test pointed to significant differences in all cases, in which the classification systems individually can be impacted differently due to their own classification methodology. Besides, p-values of 0.03 and 0.01 for the ACCESS2-0 and CMCC-CMS scenarios in the Big Creek, and a p-value of 0.02 for the CNRM-CM5 scenario in the Merced River, all other scenarios showed highly statistically significant changes ($p<0.01$) for all WYTs. As seen in **Table S2.1-1**, WYTs are generally defined by fixed thresholds, therefore affect WYT frequencies across all future scenarios.

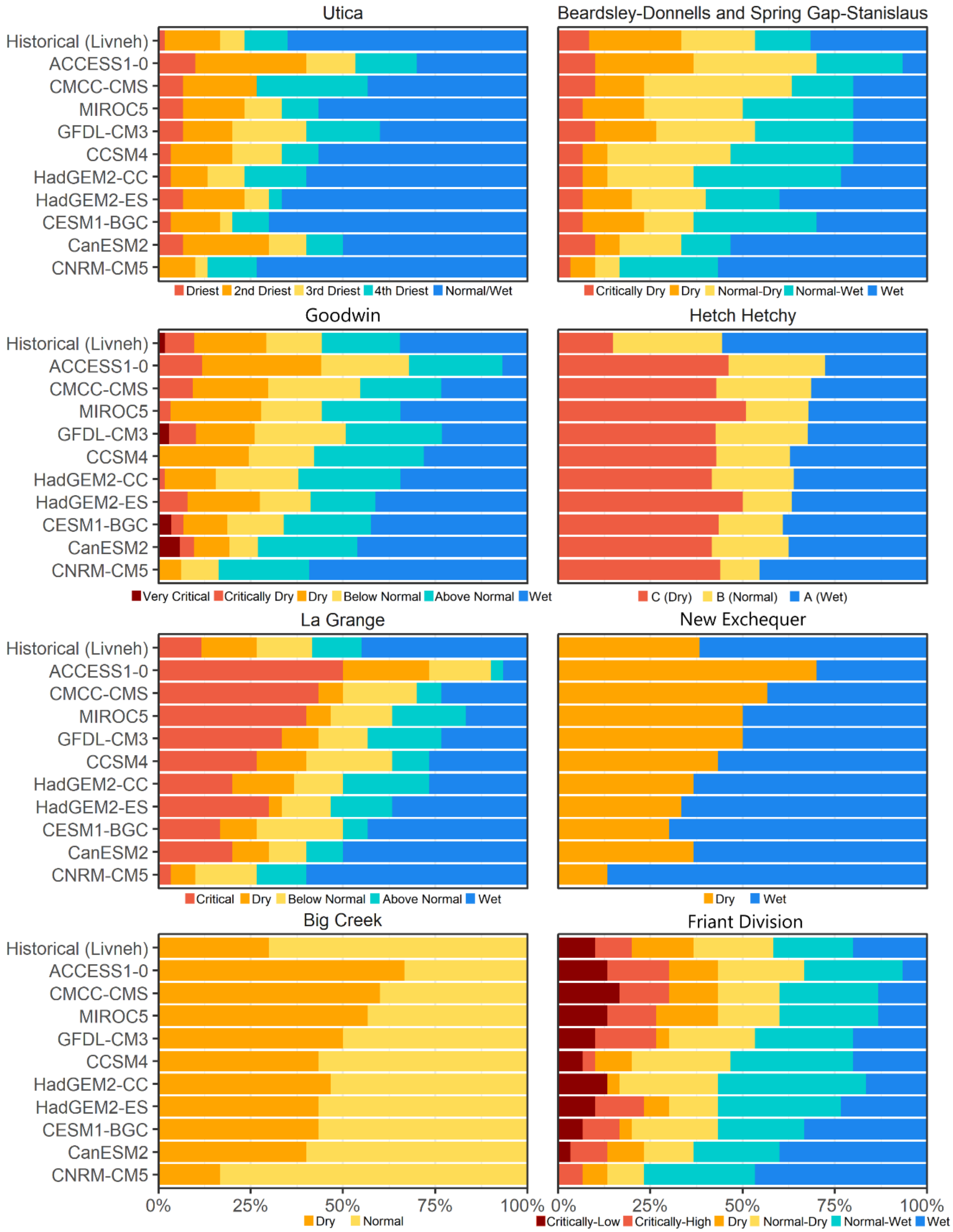


**Figure 2-4.** Frequency of water year types as defined for the different hydropower projects in the Central Sierra Nevada. GCMs are organized from driest to wettest.

**Table 2-2.** Pairwise comparison of simulated annual unimpaired streamflows used as model inputs to *CenSierraPywr* using the Wilcoxon Rank Sum Test assessing whether differences in mean runoff between scenarios were statistically significantly different. We compare the more recent historical data (1981-2010), to earlier historical data (1951-1980) and all future GCM projections (2031-2060); p-values in bold are less than or equal to the significance level ($p \leq \alpha$).

| **Scenario** | **Wilcoxon Sum Rank test (p-value)** | | | |
|---|---|---|---|---|
| | **Stanislaus** | **Tuolumne** | **Merced** | **Upper San Joaquin** |
| **Historical (Livneh) (1951-1980)** | 0.83 | 0.69 | 0.63 | 0.64 |
| **ACCESS2-0** | 0.30 | **0.03** | 0.15 | **<0.01** |
| **CMCC-CMS** | 0.59 | 0.25 | 0.29 | **0.02** |
| **MIROC5** | 0.95 | 0.21 | 0.31 | **0.04** |
| **GFDL-CM3** | 0.70 | 0.49 | 0.53 | 0.21 |
| **CCSM4** | 0.55 | 0.84 | 0.54 | 0.63 |
| **HadGEM2-CC** | 0.82 | 0.89 | 0.74 | 0.54 |
| **HadGEM2-ES** | 0.40 | 0.81 | 0.87 | 0.49 |
| **CESM1-BGC** | 0.45 | 0.49 | 0.44 | 0.88 |
| **CanESM2** | 0.19 | 0.36 | 0.21 | 0.55 |
| **CNRM-CM5** | **<0.01** | **0.01** | **0.01** | **0.04** |

### 2.3.3. Environmental flows

According to **Figure 2-5**, e-flow volumetric reliability in the Tuolumne and Merced basins (Hetch Hetchy and La Grange, and Merced River, respectively) is not greatly affected by the climate change scenarios. Meanwhile, river reaches that do not have flow prescriptions varying according to hydrological conditions (No WYT) still face lower reliability especially across late summer to early winter (July-December). The hydropower projects in the Stanislaus basin (top row) generally maintain a high reliability, which however falls below 75-80% many times especially in the Beardsley-Donnells and Spring Gap-Stanislaus water year typology around the summer and winter (June-July and November-January). The simulated e-flows for Friant Dam are based on the 'restoration flows' schedule currently being implemented by the USBR, as historical flow releases did not really follow a flow schedule as mentioned before, relying mostly on flood control releases. Still, the lower San Joaquin River faces higher reliability risks around end of water year transition (August-November) in this new flow schedule. Meanwhile, the Big Creek hydroelectric system in the Upper San Joaquin also shows more declines in reliability around this time of the water year, but also impacted between the summer and early spring (June-March).

**Figure 2-6** shows the overall annual reliability of all river reaches per WYT classification. E-flows with No WYT prescriptions tend to generally have increasingly greater resilience in future scenarios depicting increasingly greater water volumes, as e-flows are mostly static and depend solely on water availability, not on specific thresholds of inflows. Resilience scores predominantly around or below 50% in the Stanislaus, Tuolumne, and Merced rivers, despite high reliability, suggest that after a failure to meet e-flow requirements occurs, there is a considerable chance for the system to not promptly return to satisfactory performance conditions. This observation, paired with high reliability, indicates that e-flows occasionally fail to meet demands, but are still somewhat likely to recover quickly. Similarly, even though Friant Dam struggles to maintain e-flows around the end and beginning of the water years, it also maintains resilience levels near or below 50% even in the wettest future scenarios. Meanwhile, the Big Creek reaches show the lowest resilience across all WYT classifications. This suggests that while failures to meet demands are not uncommon, as depicted in **Figure 2-5**, they persist for longer durations within this system. Based on these results, we decided to sample the Beardsley-Donnells and Spring Gap-Stanislaus, and Big Creek projects, as highly regulated systems in the Stanislaus and Upper San Joaquin River, respectively, to investigate the impacts of our proposed climate change adaptation strategy. These projects also have smaller catchment contributing areas when compared to most other projects in which WYTs are adopted, such as at or downstream of the terminal dams (Goodwin, La Grange, Merced River and Friant Division), and therefore are more likely affected by hydroclimatic variability.

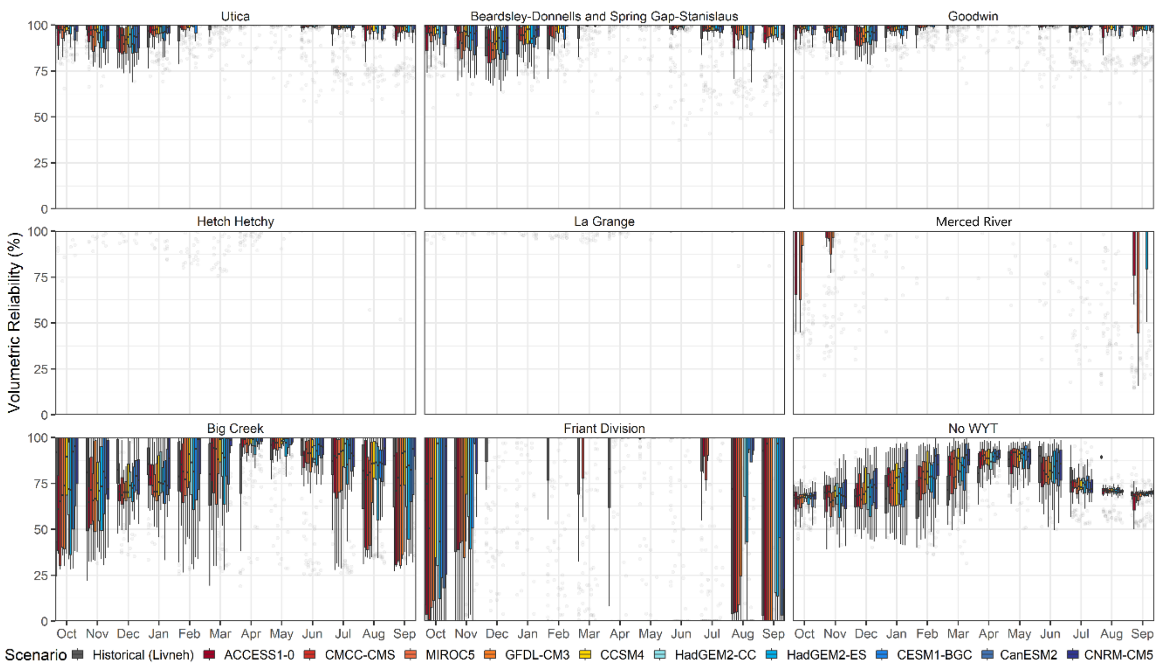


**Figure 2-5.** Volumetric reliability of environmental flows according to their water year typology. GCMs are ordered from driest to wettest.

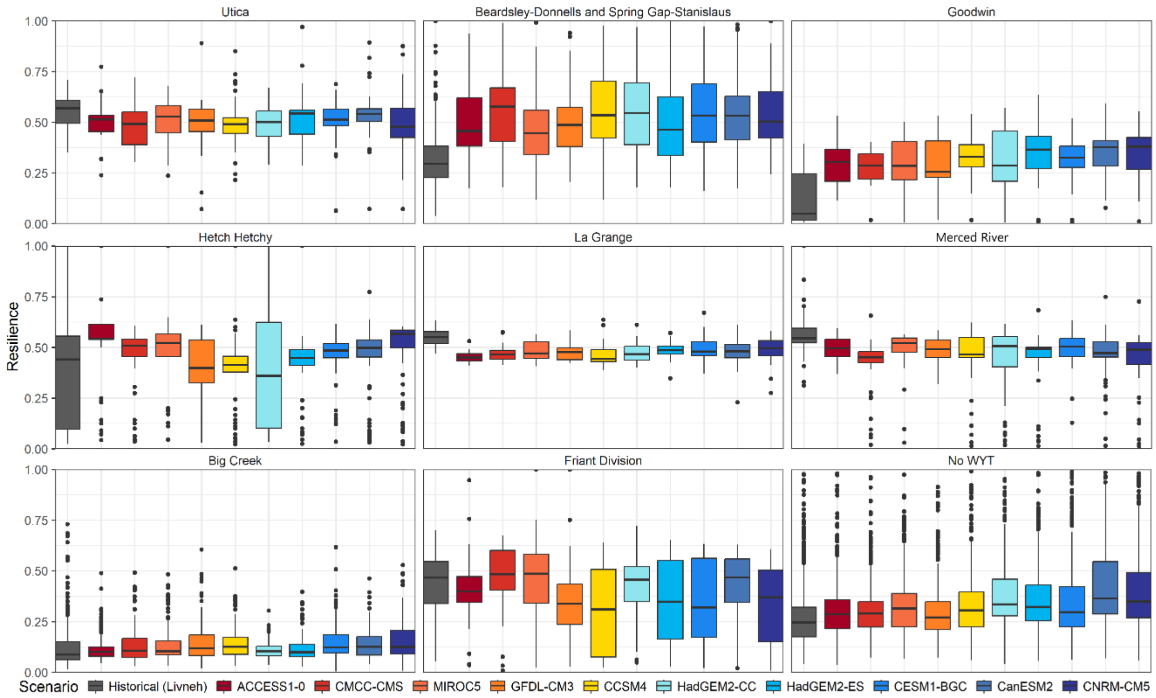


**Figure 2-6.** Resilience of environmental flows according to their water year typology. GCMs are ordered from driest to wettest.

### 2.3.4. Adaptation to climate change

E-flows in the Beardsley-Donnells and Spring Gap-Stanislaus projects are disproportionately affected by climate change. For instance, the IFR below Philadelphia Diversion (**Figure 2-7**, top right), is downstream of the IFR below Pinecrest Lake (**Figure 2-7**, bottom left), the first reservoir in line upstream, among the smallest in storage capacity in the basin. Even though both river reaches have similar flow requirements, the extra natural inflows from sub-watersheds downstream of Pinecrest Lake make the IFR below Philadelphia Diversion more reliable, being mostly not affected by climate change, when considering changes in volumetric reliability from historical averages. Meanwhile, the IFR below Relief Reservoir (first reservoir in line in the upper watersheds) is the most impacted river reach, with more significant drops in reliability especially in the winter, when flow prescriptions are higher, however with potential increases in reliability around the summer (May-July) as projected by all GCMs. Likewise, even though the IFR below Donnell Lake receives water from Relief Reservoir and other free flowing sub-watersheds, it shows drops in reliability around the winter (November-January). Also, the IFR below Sand Bar Diversion receives water from two powerhouses above the diversion point, showing increased reliability March-July, but with diminished reliability in the winter extending through February, as projected by all GCMs. **Figure 2-8** illustrates the change in resilience based on historical averages for each river reach. While there are different levels of positive and negative variances across individual IFRs, climate change generally slightly improves resilience, pointing to mostly shorter recovery times for meeting e-flow demands following a failure. However, recovering more quickly does not imply a lower occurrence of such failures, as pointed out by the reliability index.

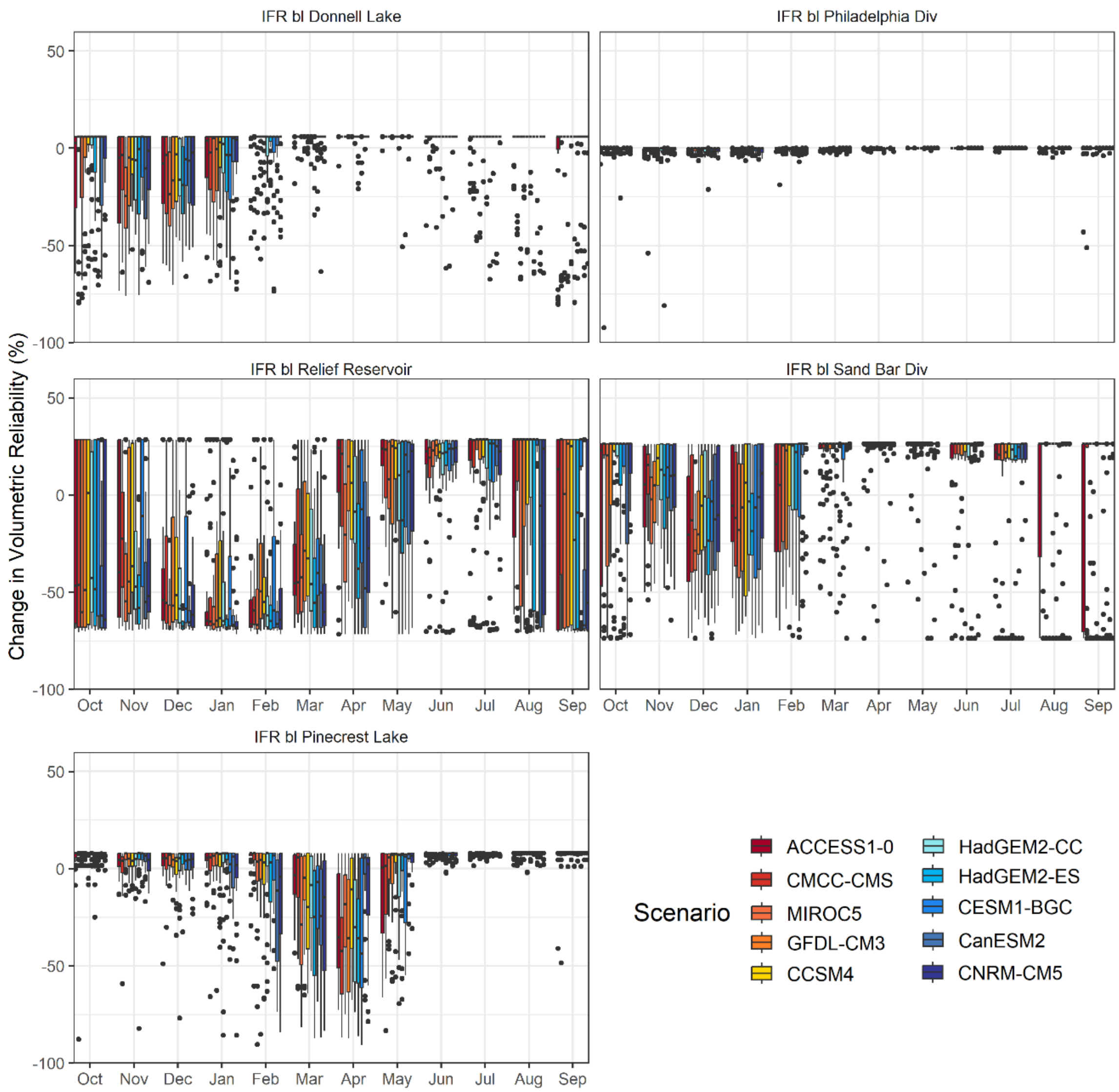


**Figure 2-7.** Change in volumetric reliability of environmental flows of each river reach of the Beardsley-Donnells and Spring Gap-Stanislaus hydropower projects in future climate change scenarios, compared to historical averages. GCMs are ordered from driest to wettest.

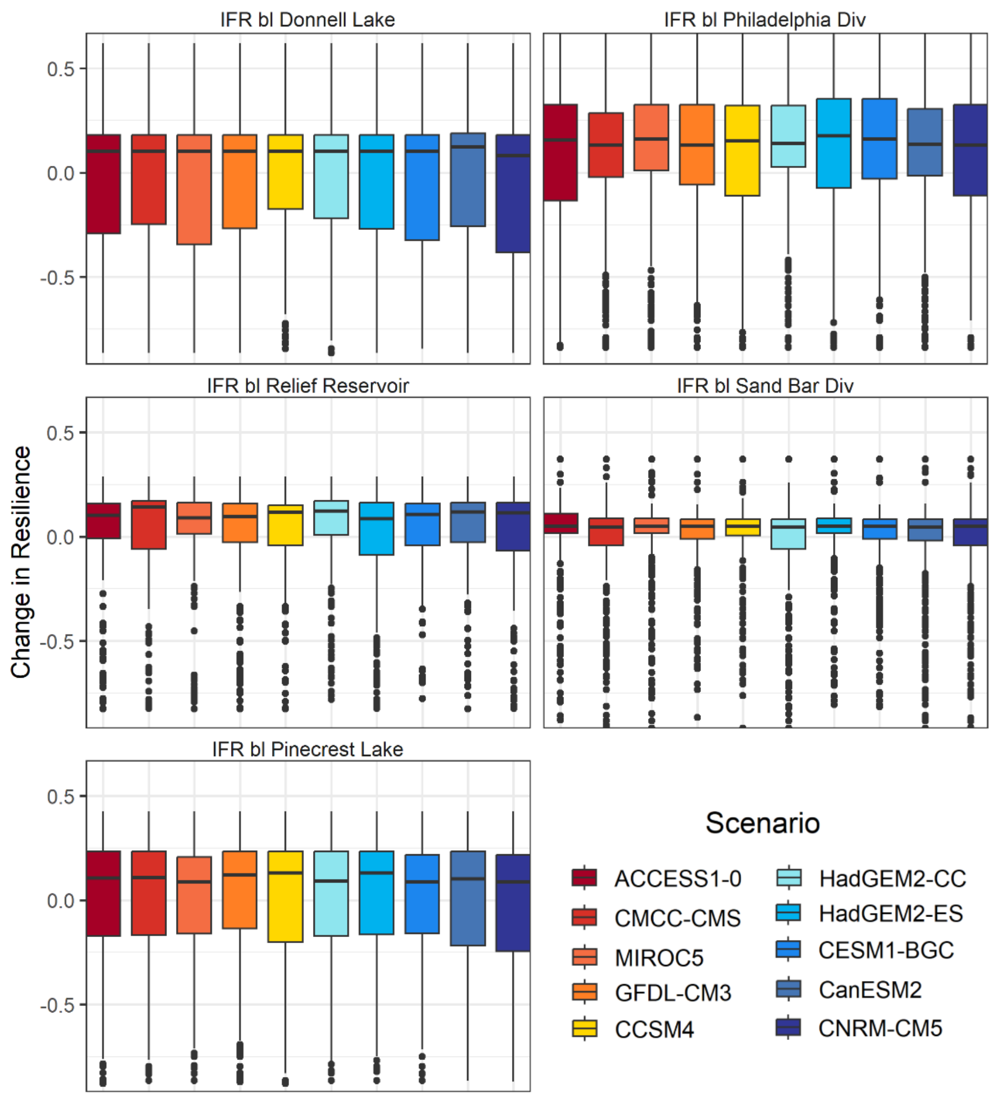


**Figure 2-8.** Change in resilience of environmental flows in each river reach of the Beardsley-Donnells and Spring Gap-Stanislaus hydropower projects in future climate change scenarios, compared to historical averages. GCMs are ordered from driest to wettest.

**Figure 2-9** shows the changes in volumetric reliability of e-flows in the Big Creek project caused by climate change, compared to historical averages. IFR above Shakeflat Creek is below Mammoth Pool Reservoir, which receives natural and regulated inflows from a large contributing catchment area, however, besides increases in reliability projected especially by the wetter GCMs around the spring and summer (March-July), there are also projected decreases in reliability throughout the rest of the year. Meanwhile, the IFRs below Big Creek 5 Diversion and below Redinger Lake suffer no significant changes in any future projection. These are likely because both of these river reaches receive water from most reservoirs, free-flowing rivers and/or powerhouses upstream. All other river reaches tend to face mostly losses in reliability, with more significant drops in reliability within late summer and winter (September-March), and occasional losses in reliability around the late spring and early summer seasons (April-July). The IFR North Fork Stevenson Creek above Shaver Lake also receives natural inflows from free-flowing creeks and outflows from Balsam Meadows Forebay, a re-regulating reservoir that receives water from facilities upstream as well. Meanwhile, the IFRs below Mono Creek Diversion and below Hooper Creek are below the first reservoirs in line, being their only possible sources of water. Similar to e-flows in river reaches in the hydropower projects in the Stanislaus River, these in the Upper San Joaquin River, generally show slightly increased resilience under climate change, especially under wetter future scenarios, with different degrees of variance (**Figure 2-10**).

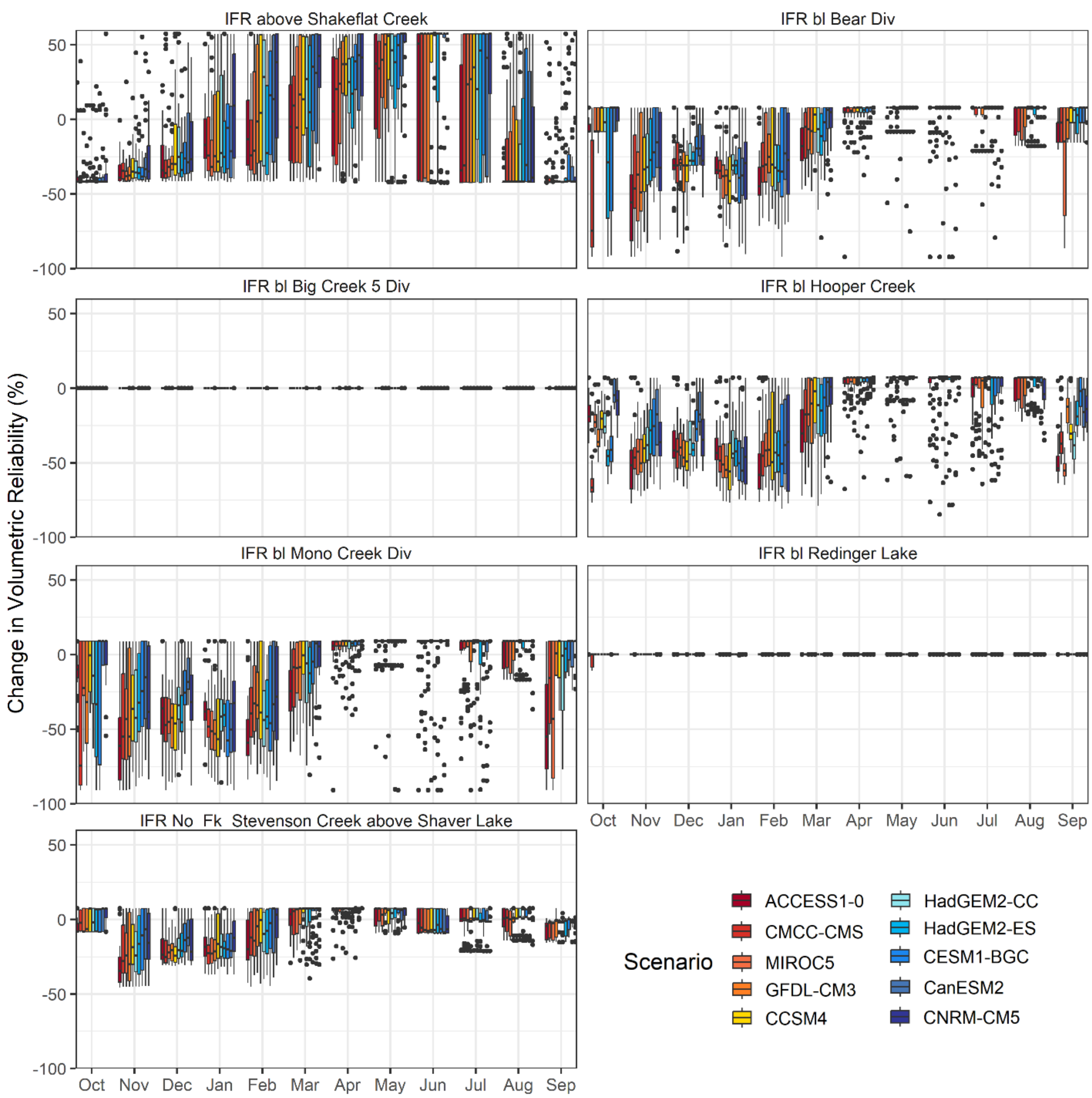


**Figure 2-9.** Change in volumetric reliability of environmental flows in each river reach of the Big Creek hydropower project in future climate change scenarios, compared to historical averages. GCMs are ordered from driest to wettest.

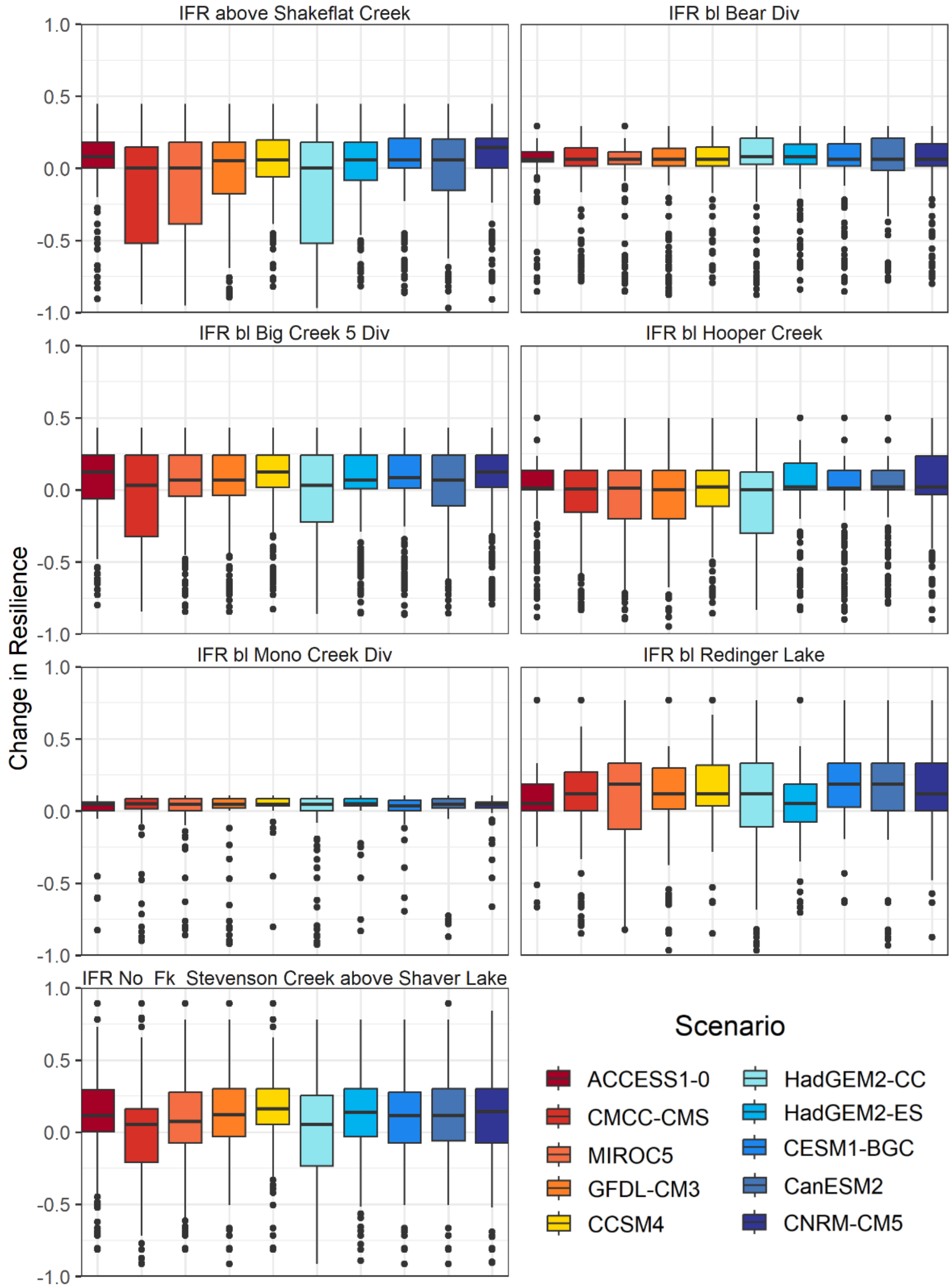


**Figure 2-10.** Change in resilience of environmental flows in each river reach of the Big Creek hydropower project in future climate change scenarios, compared to historical averages. GCMs are ordered from driest to wettest.

Considering the adaptation strategy of constantly updated WYTs through 30-year moving percentiles (**Figure 2-11**), the IFRs below Donnell Lake, below Relief Reservoir and below Pinecrest Lake show eventual increases in reliability when compared to the *status quo* management. Meanwhile, the IFR below Philadelphia Diversion, shows no significant difference, as it is not generally affected by climate change. The IFR below Sand Bar Diversion shows a similar behavior, however showing eventual decreases in reliability, especially when climate change increases reliability in **Figure 2-10**, such as in June-July. The resilience index (**Figure 2-12**) on the other hand, shows mostly improvements in e-flows, according to the generally positive similar median changes across all scenarios.

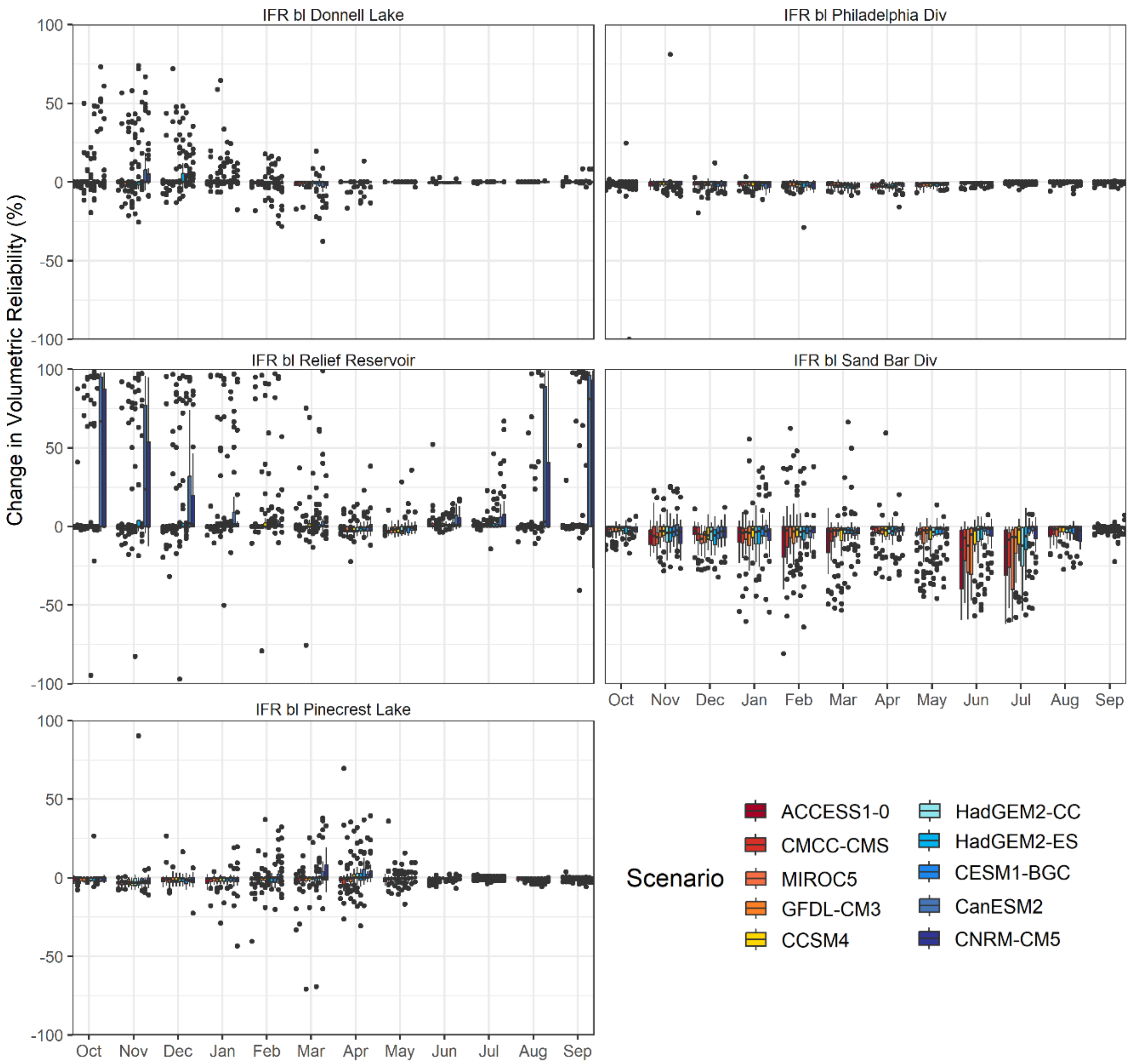


**Figure 2-11.** Change in volumetric reliability of environmental flows in each river reach of the Beardsley-Donnells and Spring Gap-Stanislaus hydropower projects using the adaptive management approach, in future climate change scenarios, compared to status quo management. GCMs are ordered from driest to wettest.

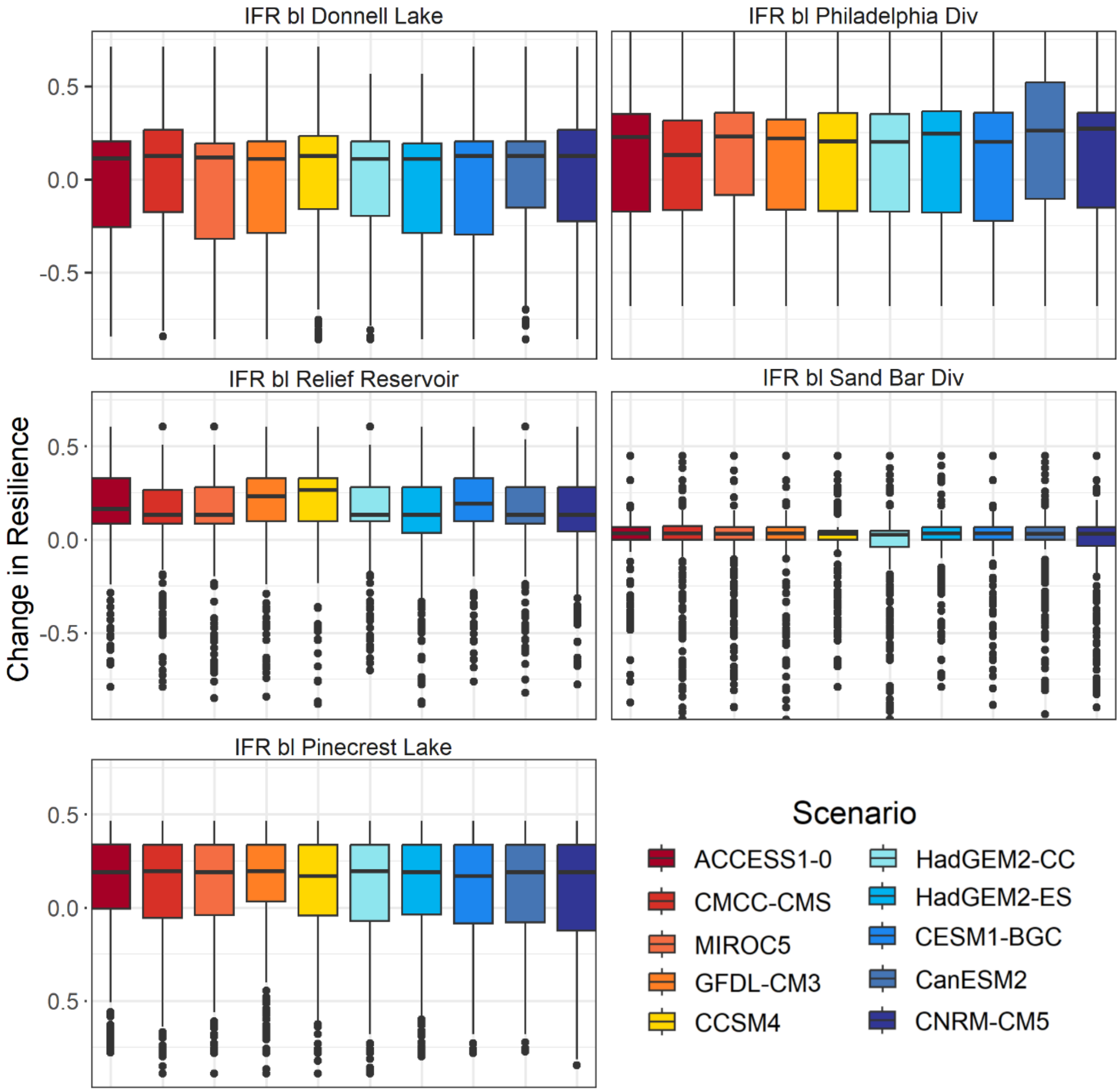


**Figure 2-12.** Change in resilience of environmental flows in each river reach of the Beardsley-Donnells and Spring Gap-Stanislaus hydropower projects using the adaptive management approach, in future climate change scenarios, compared to status quo management. GCMs are ordered from driest to wettest.

In the Big Creek system, the adaptation resulted in higher volumetric reliability mostly when e-flows are most affected by climate change under *status quo* management in the IFRs below Mono Creek Diversion, below Hooper Creek and North Fork Stevenson Creek above Shaver Lake (**Figure 2-13**). However, improvements generally remained within a 0-20% range. As seen before, reliability is not affected by climate change in the IFRs below Big Creek 5 Diversion and below Redinger Lake, under the adaptation strategy they continue to not be negatively affected. Meanwhile, the IFRs above Shakeflat Creek and below Bear Diversion, show mostly sporadic improvements in volumetric reliability.

Considering resilience, in almost all river reaches all future scenarios have predominantly positive median responses to the adaptive water year typing, implying that when failures in meeting IFRs occur, in most cases they tend to recover more quickly.

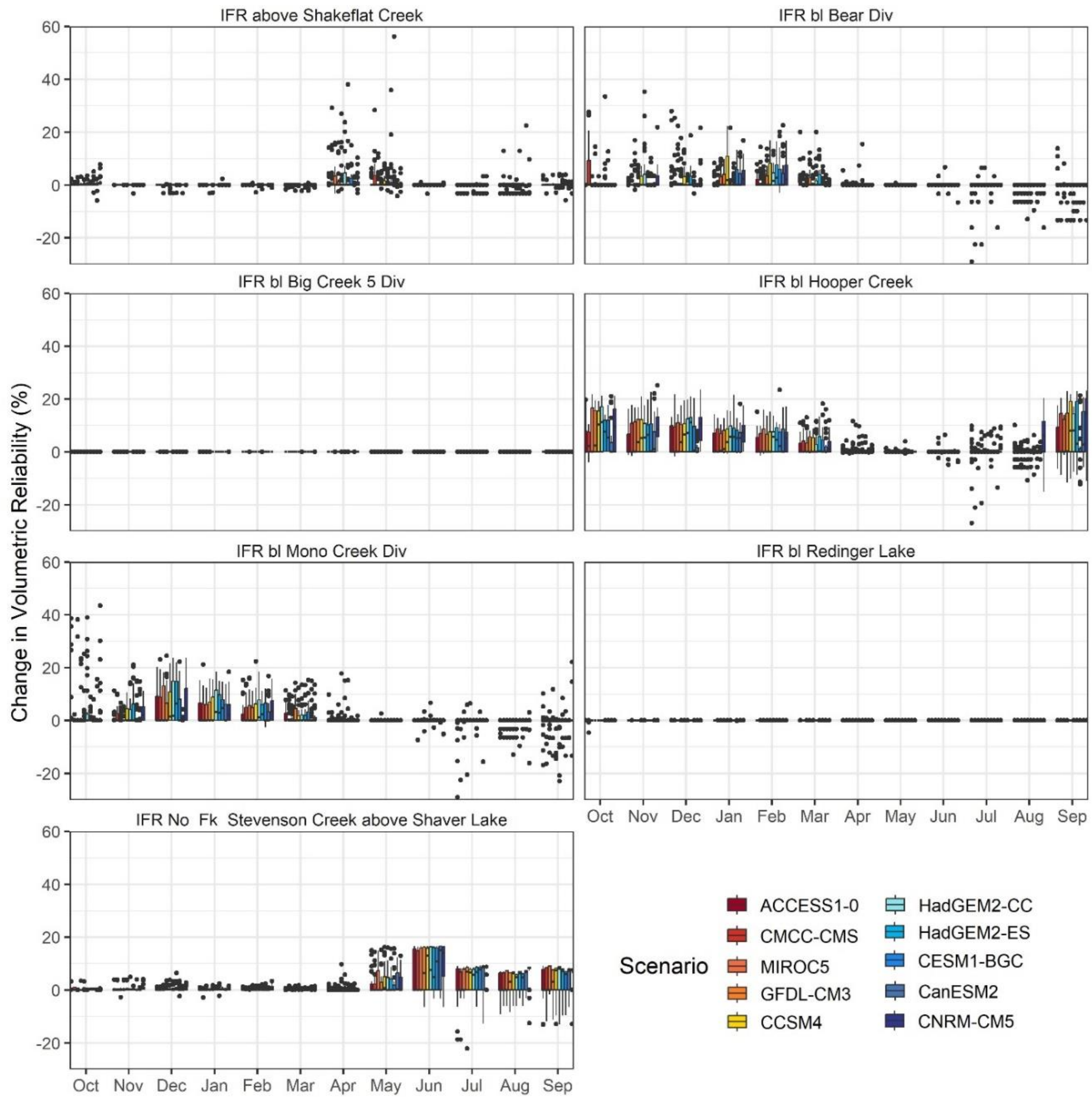


**Figure 2-13.** Change in volumetric reliability of environmental flows in each river reach of the Big Creek hydropower project using the adaptive management approach, in future climate change scenarios, compared to status quo management. GCMs are ordered from driest to wettest.

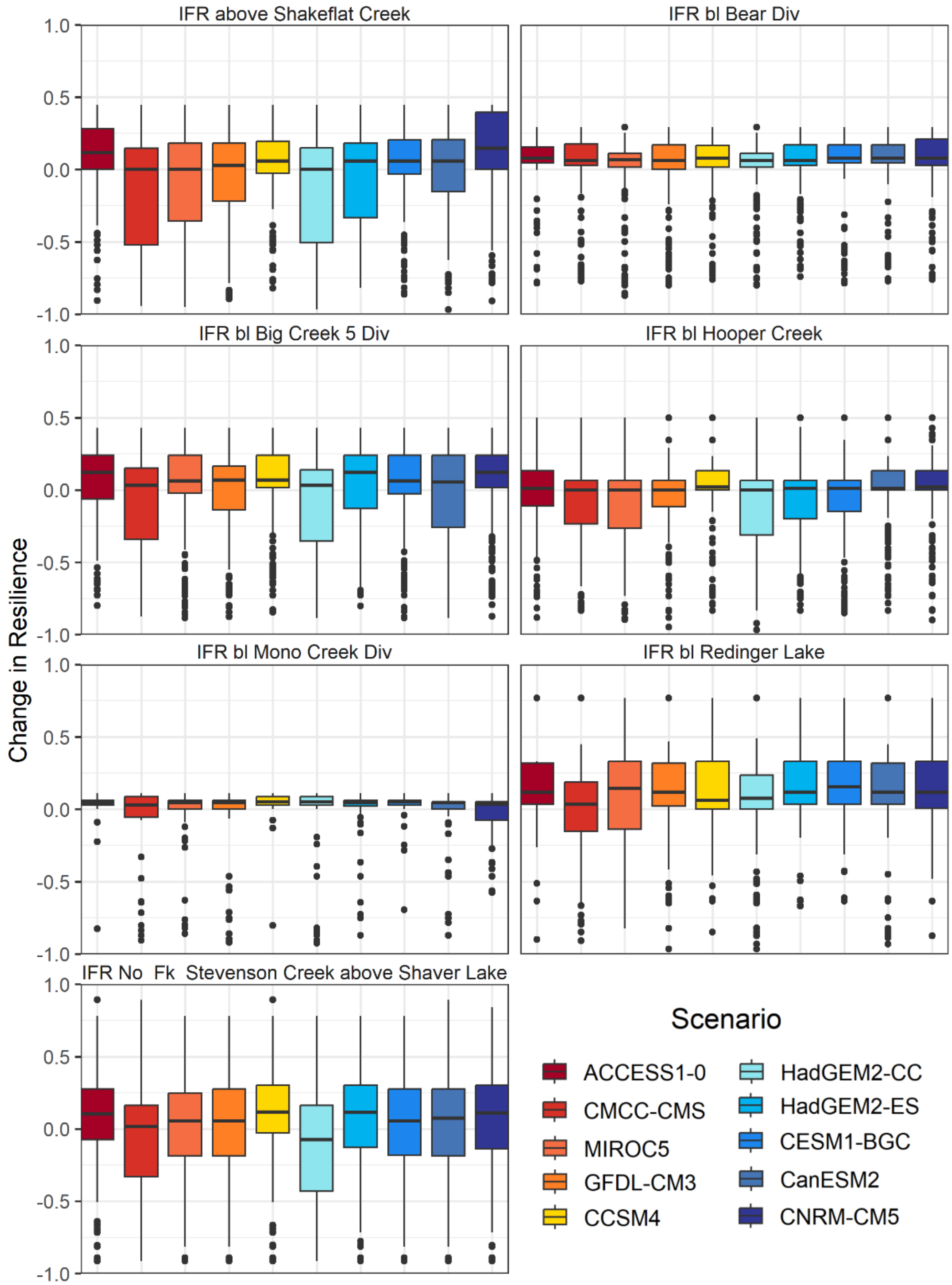


**Figure 2-14.** Change in resilience of environmental flows in each river reach of the Big Creek hydropower project using the adaptive management approach, in future climate change scenarios, compared to status quo management. GCMs are ordered from driest to wettest.

## 2.4. Discussion

Even though climate change does not severely impact hydrology according to the Wilcoxon Rank Sum Test in most future projections, WYT distributions under the current management strategies can become highly variable as per the two-tailed Fisher's exact test results. Meanwhile, on a river reach scale, impacts can be disparate among and within hydropower projects. The negative impacts on reliability and variable levels of resilience in the Beardsley-Donnells and Spring Gap-Stanislaus, and Big Creek projects are evident, with certain river reaches experiencing disproportionate effects while others are not affected. For instance, as seen in the Stanislaus basin, the IFR below Philadelphia Diversion benefits from the additional natural inflows, making it more reliable compared to the IFR below Pinecrest Lake upstream. The IFR below Relief Reservoir and Donnell Lake show decreased volumetric reliability, especially during winter, but potential increases during summer as projected by all GCMs. The IFR below Sand Bar Diversion exhibits increased volumetric reliability from March-July but diminishes during winter.

Generally, climate change appears to slightly improve resilience, suggesting shorter recovery times to meet e-flow demands post failure in most times. The adaptation using the 30-year moving percentiles approach to recalculate WYTs based on new information shows varying impacts across the IFRs. However, the adaptation strategy generally produces increases in reliability especially when IFRs are mostly affected by climate change, as noted in some IFRs in the Big Creek. Conversely, the IFR Sand Bar Diversions displays no significant difference or even decreased reliability under with the adoption of the adaptation strategy.

The moving percentiles approach offers a potential first-step to enhance e-flow reliability and resilience on a case-by-case basis. Nonetheless, merely updating the existing WYT thresholds while preserving the current instream flow schedules, along with maintaining other current regulatory frameworks (e.g., inflexible reservoir operations established during the hydropower licensing process), may not yield substantial improvements. For instance, besides instream flows not varying across water years, seasonal variations are mostly absent in many river reaches as well. River dynamics include flood flows, which often are impeded by MAF requirements. Extremely low to no-flow events, are many times also not adopted, as in many instances the 'release inflows' policy is still not implemented. Even though both cases are naturally-occurring, they are generally erased from e-flow hydrographs.

While counterintuitive to freshwater conservation objectives, managed no-flow events could be a preferred solution to manage invasive species, as recently considered for redeye bass management in the Phoenix Project within the Stanislaus River basin (USFS, 2021). Furthermore, the lack of flexibility is also observed regarding temperature targets. Naturally warmer inflows, diversions and agricultural return flows in the lower Tuolumne basin are uncontrollable disturbances that impede meeting temperature targets, which are delaying the Don Pedro Reservoir relicensing process, forcing local irrigation districts to operate under a provisional license since 2012 (FERC, 2019b; Rheinheimer et al., 2015).

Once again, regulatory frameworks need to be flexible enough for the adoption of alternatives also in these cases.

Therefore, additional strategies are essential, especially for e-flows that do not show significant enhancements. Our study sheds light on the intricate challenges of managing e-flows in California, especially within the regulatory and hydroclimatic contexts. The lack of a standardized approach in defining, forecasting, and applying WYTs has led to inconsistencies and discrepancies in the management of e-flows. This inconsistency can result in asynchronous and inequitable water allocation and could potentially exacerbate conflicts among various stakeholders. Our findings highlight the need for more targeted, site-specific, and adaptive management strategies that can account for the variable impacts of climate change on water availability. Policy harmonization, i.e., making e-flow regulatory requirements identical or at least more similar (e.g., homogenizing year type categories, year type methods and/or e-flow assessment approaches), could help solve or simplify this problem. These issues require further discussion and consideration to guide more robust and resilient water resource management strategies.

## 2.5. Conclusion

This study highlights several challenges and vulnerabilities in the current regulatory framework governing e-flows in California, exacerbated by the complexities introduced by climate change. There is a lack of coherence in water policy that has the potential to limit sociohydrological adaptation. There is a lack of dependence of IFR requirements on WYTs, despite their usage for operations. Further, the inconsistent application of WYT classification within and across basins propagates this incoherence. And, moreover, the increased variability in inter- and intra-annual hydrology due to climate change exacerbates the incoherence in policy prescriptions. Modeling results presented here underscore how poor e-flow reliability and resilience necessitate the need for a more adaptive and resilient regulatory approach. These results also suggest that site-specific and data-driven approaches to e-flow implementation are needed. The difficulty in maintaining higher reliability and more stable resilience responses in specific WYT classifications suggests that the existing systems may not be capable of effectively stabilizing against climate change effects to maintain environmental needs under current WYT based operating rules. A more harmonized and consistent approach to defining, forecasting, and applying WYTs is crucial to establish IFRs, to ensure equitable and sustainable management of water resources across different regions and watersheds. That could be achieved by the adoption of feedback policies, updated based on new data to achieve especially greater reliability in e-flow deliveries (e.g., revise WYT classification periodically and update WYT and flow thresholds based on more recent inflows, for instance). Moreover, we emphasize the need for greater collaboration and coordination among different agencies and utilities to achieve that and responsibly managing water resources.

Our findings suggest that current management strategies may not be sufficiently flexible or robust for e-flows to cope with the uncertainties and variability associated with climate change in certain hydropower projects. WYT calculations constantly been updated can improve resilience, and reliability in certain cases, but may not be relied upon solely, which raises concerns about the environmental water resources and the integrity of aquatic ecosystems. This calls for a comprehensive review and revision of the existing regulatory framework to ensure more adaptive and equitable management of water resources, balancing the needs of various stakeholders, especially environmental needs. Some options are currently being considered by SWRCB, such as 'water month types', for instance. Future research should focus on developing more integrated and dynamic management approaches to water year typing and/or e-flow prescriptions that can effectively respond to the challenges posed by climate change and ensure the long-term sustainability of California's water resources.

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## Appendix 2.1: Supporting information for Chapter 2

**Table S2.1-1.** Different water year type (WYT) classification systems in the San Joaquin River Basin

| *Project* | *Forecasting Agency* | *Applicable Area* | *Forecast* | *Release Year* | *Source* |
|---|---|---|---|---|---|
| **Utica Power Authority** | CDWR | Specific river reaches below dams | Apr-Jul unimpaired runoff to New Melones Dam | May-Apr | (FERC, 2004) |
| **Beardsley-Donnells and Spring Gap-Stanislaus** | CDWR | Specific river reaches below dams | May 1 unimpaired runoff to New Melones, updated monthly between Feb-Apr | May-Apr | (FERC, 2006) |
| **Goodwin** | CDWR | Below the terminal dam | Apr-Jul unimpaired runoff to Don Pedro Reservoir + 20% of the current year's Oct-Mar unimpaired runoff + 20% of the previous year's total unimpaired runoff | Apr 14-Apr 15 | (NOAA, 2009) |
| **Hetch Hetchy** | SFPUC | Below the dam | Cumulative Oct-Jun precipitation and Jul-Aug inflows at Hetch Hetchy Reservoir, updated monthly | Jan-Dec | (SFPD, 2008) |
| **La Grange** | CDWR | Below the diversion dam downstream of the terminal dam | April-July unimpaired runoff to Don Pedro Reservoir + 20% of the current year's Oct-Mar unimpaired runoff + 20% of the previous year's total unimpaired runoff | May-Apr | (FERC, 2019a) |
| **Merced River** | CDWR | Below a re-regulating reservoir downstream of the terminal dam | April-July unimpaired runoff to New Exchequer Dam | May-Apr | (FERC, 1964) |
| **Big Creek** | CDWR/ USBR | Specific river reaches below dams | April-July unimpaired runoff to Millerton Lake | May-Apr | (FERC, 1959, 1978, 2003c; SCE, 2000) |
| **Friant** | USBR | Below the terminal dam | Jan 20-Jul 10 unimpaired runoff to Millerton Lake, updated at least monthly | Mar-Feb | (SJRRP, 2017) |

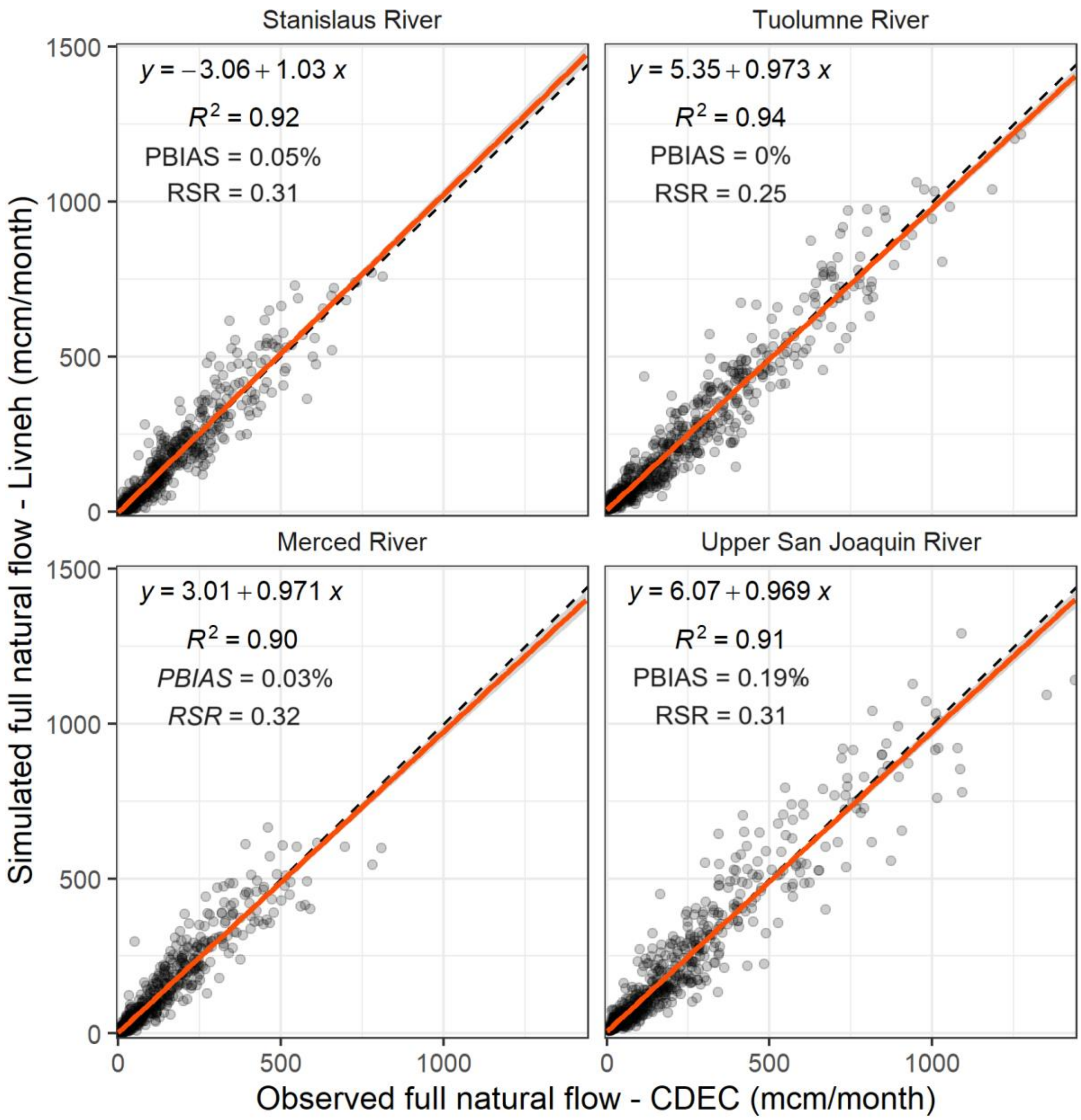


**Figure S2.1-1.** Bias-corrected simulated monthly streamflow data (Livneh 1950-2013) compared with historical full natural flow estimates from the California Data Exchange Center data for each basin.

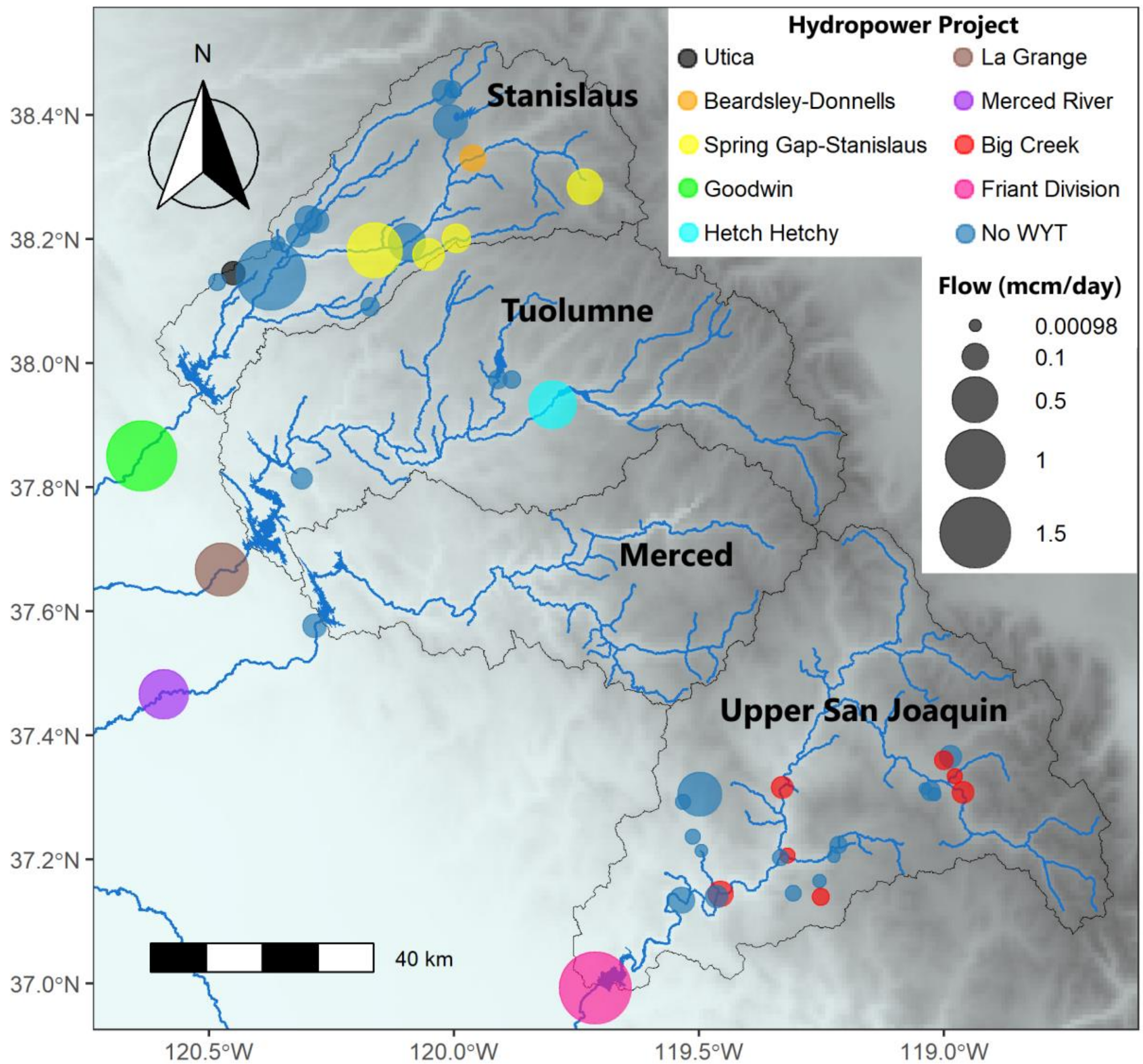


**Figure S2.1-2.** River reaches with instream flow requirements (IFRs) represented in the *CenSierraPywr* framework. Hydropower projects are colored after water year type (WYT) classification systems and size by average daily minimum instream flow requirements (MIFs).

## Chapter 3. Overcoming persistent challenges in putting environmental flow policy into practice: A systematic review and bibliometric analysis[a]

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**Keywords**: ecological flows, instream flows, minimum flows, flow alteration, flow regulation, dams, freshwater conservation, river management

### Abstract

The implementation of environmental flows (e-flows) aims to reduce the negative impacts of hydrological alteration on freshwater ecosystems. Despite the growing attention to the importance of e-flows since the 1970s, actual implementation has lagged. Therefore, we explore the limitations in e-flows implementation, their systemic reasons, and solutions. We conducted a systematic review and a bibliometric analysis to identify peer-reviewed articles published on the topic of e-flows implementation research in the last two decades, resulting in 68 research and review papers. Co-occurrence of terms, and geographic and temporal trends were analyzed to identify the gaps in environmental water management and propose recommendations to address limitations on e-flows implementation. We identify the underlying causes and potential solutions to such challenges in environmental water management. The limitations to e-flow implementation identified were categorized into 21 classes. The most recognized limitation was the competing priorities of human uses of water (n = 29). Many secondary limitations, generally co-occurring in co-causation, were identified as limiting factors, especially for implementing more nuanced and sophisticated e-flows. The lack of adequate hydrological data (n = 24) and ecological data (n = 28) were among the most mentioned, and ultimately lead to difficulties in starting or continuing monitoring/adaptive management (n = 28) efforts. The lack of resource/capacity (n = 21), experimentation (n = 19), regulatory enforcement (n = 17), and differing authorities involved (n = 18) were also recurrent problems, driven by the deficiencies in the relative importance given to e-flows when facing other human priorities. In order to provide a clearer path for successful e-flow implementation, system mapping can be used as a starting point and general-purpose resource for understanding the sociohydrological problems, interactions, and inherited complexity of river systems. Secondly, we recommend a system analysis approach to address competing demands, especially with the use of coupled water-energy modeling tools to support decision-making

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when hydropower generation is involved. Such approaches can better assess the complex interactions among the hydrologic, ecological, socioeconomic, and engineering dimensions of water resource systems and their effective management. Lastly, given the complexities in environmental water allocation, implementation requires both scientific rigor and proven utility. Consequently, and where possible, we recommend a move from simplistic flow allocations to a more holistic approach informed by hydroecological principles. To ease conflicts between competing water demands, water managers can realize more 'pop per drop' by supporting key components of a flow regime that include functional attributes and processes that enhance biogeochemical cycling, structural habitat formation, and ecosystem maintenance.

### 3.1. Introduction

The concept of environmental flows (e-flows) emerged from the need to recognize the needs of specific species, such as economically important salmonid fisheries (Tharme, 2003), with infrequent consideration of the water needs of entire river ecosystems and the people who directly depend on them (Matthews et al., 2014). E-flows then evolved to whole-community and ecosystem perspectives to mitigate the undesirable hydrological impacts of dams and water diversions (Poff & Matthews, 2013), and protect or restore the benefits of naturally flowing rivers, in regulated systems (Owusu et al., 2021). In response to well documented global degradation of freshwater ecosystems, several efforts have emerged over the previous four decades to document and support e-flow development and implementation (Arthington, Kennen, et al., 2018). At the policy level, several instruments have been adopted, such as the Brisbane Declaration (2007) which formalized the e-flow paradigm as “the quantity, timing, and quality of freshwater flows and levels necessary to sustain aquatic ecosystems which, in turn, support human cultures, economies, sustainable livelihoods, and well-being”. This approach was reiterated in 2018 with additional guidelines for practitioners of different regions and disciplines, intending to set a common vision and direction for e-flows globally (Arthington, Bhaduri, et al., 2018).

The actual implementation of e-flows has remained limited to date despite the abundance of theories and concepts for e-flows (Owusu et al., 2022). Several factors have contributed to this disparity in implementation, including a lack of research on e-flow implementation and trade-off analysis with other uses (Pahl-Wostl et al., 2013), uncertainty over method choice (Opperman et al., 2018), and physical and policy constraints that limit flow releases from dams (Aldous et al., 2011; Pittock & Hartmann, 2011). Delays in implementation can also occur due to differing technical term definitions or other incongruences among implementing and regulatory authorities, thus leading to misunderstanding among stakeholders and managers (Pahl-Wostl et al., 2013).

E-flow implementation efforts can also miss the systematic and integrated conceptualization of a river system as one complex socio-hydrological system (Madani & Shafiee-Jood, 2020). This can lead to fragmentation of effort, failure to take advantage of local adaptive management learnings, and poor public understanding of why decisions are being made and where responsibility lies (Thompson et al., 2018). Indeed, there is a pressing need for a more committed effort to protect and restore freshwater ecosystems as resilient human-water systems through the implementation and adaptation of e-flows

(Arthington, Bhaduri, et al., 2018). Poff et al. (1997) introduced a paradigm shift in environmental water management when presenting the natural flow concept, in which the natural streamflow dynamics supports native habitats and species assemblages. Yet, the role of the natural flow regime in creating the spatiotemporal variation in biogeographic patterns and processes has been neglected (Meitzen et al., 2013). A disregard for the natural system complexity of river ecosystems persists despite substantial progress in understanding how natural flow variation maintains river health (Sofi et al., 2020). Given the complexities in water allocation for e-flows, implementation requires both scientific rigor and proven utility.

Therefore, a crucial priority for freshwater conservation is to accelerate the implementation of effective e-flows and their potential benefits such as improvements in water quality, critical habitat maintenance, and hydrologic connectivity (Tickner et al., 2020). Consequently, our review addresses the gaps in e-flow implementation to help guide water management decisions and better meet ecosystem needs while satisfying human demands (Viers, 2017). Within this context, the goals of this paper are to (1) explore the limitations in e-flow implementation by emphasizing current and future challenges of environmental water management and their implications; (2) identify systemic reasons for the lack of implementation and solutions for overcoming them; and (3) present a conceptual framework as the basis for decision-making to help managers and stakeholders select the most appropriate methods based on their resource availability, physical and legal constraints and objectives. We conclude by identifying existing data and conceptual gaps and discussing important recommendations for the effective implementation of e-flows.

### 3.2. Background

The objective of setting e-flows is (or should be) to modify water abstraction from water bodies or flow releases from water infrastructure to restore natural or normative flow regimes that benefit river and riparian ecosystems downstream (Poff & Matthews, 2013). The "natural flow regime" determines the geomorphic processes that shape river channels, floodplains, and other riverine habitats, consequently governing the ecological processes and the composition of flora and fauna (Poff et al., 1997; Taniguchi-Quan et al., 2022). The maintenance of natural flow patterns (i.e., magnitude, frequency, duration, timing, predictability, and rate of change) allows lateral and longitudinal habitat connectivity, a major determinant of biotic diversity, and controls invasive species while triggering life-history strategies of native species that are adapted to the natural flow regime (Bunn & Arthington, 2002; Poff et al., 2007; Poff & Matthews, 2013; Koster & Crook, 2017). Conversely, disruptions to natural flow regimes can have long-ranging impacts on adapted species. For instance, the Balbina Dam in the Brazilian Amazon removed the periods of extended low flow that allowed floodplain forests to establish, causing the death of tree species in waterlogged areas for over 100 km downstream (Assahira et al., 2017).

The application of e-flows emerged in the mid-twentieth century in developed countries within Europe and in the US in response to the biodiversity impacts of flow regulation and diversion of surface waters (Matthews et al., 2014). E-flow assessments began in the late 1940s in the western US to establish minimum flows required for the protection of valuable cold-water fisheries in snow-dominated environments (Poff et al., 2017). Since then, international policy landmarks that favored the implementation of e-flows have been created, such as the UK Water Resources Act of 1963, which required minimum acceptable flows to maintain natural beauty and fisheries (Neachell & Petts, 2017; Overton et al., 2014b). After decades, environmental water science and assessment have advanced with the development of many approaches and tools in response to changing societal objectives and values, paradigms, and increasing knowledge base and modeling capabilities (Poff et al., 2017). Still, one of the methods widely used to 'preserve' river flows is to set a minimum flow below which any water abstraction must be reduced or ceased (Acreman, 2005).

Water allocation for the environment also involves trade-offs with other competing needs, such as hydropower and urban/agricultural water supply, and can be limited by different objectives or strategies occurring under different jurisdictional boundaries and institutional settings (Rheinheimer et al., 2022). Simply reallocating water from human uses to the environment often faces uncertainties due to overallocation (Loch et al., 2011; Stein et al., 2021) primarily hampered by competition between human and environmental needs, and lack of political will (Overton et al., 2014b). Trade-offs among contrasting goals need to be identified so that the appropriate strategies can be prioritized, especially in hydropower and multiple-purpose water projects which are prone to conflicting interests at different scales (**Figure 3-1**). Trade-off analyses are especially important when considering the need for the implementation of more nuanced e-flows, which necessitate a better understanding of ecological needs and seasonal variability (Willis et al., 2022). In that sense, system analysis in water operations is increasingly needed due to the more intense competition for limited supplies, necessitating efficient allocation among conflicting objectives (Brown et al., 2015). Based on that, technical processes can be developed to better guide practitioners in the development of e-flow standards for rivers and streams focusing on the habitat needs of native species, to deliver broad benefits for people and nature (Grantham et al., 2020; Null et al., 2021).

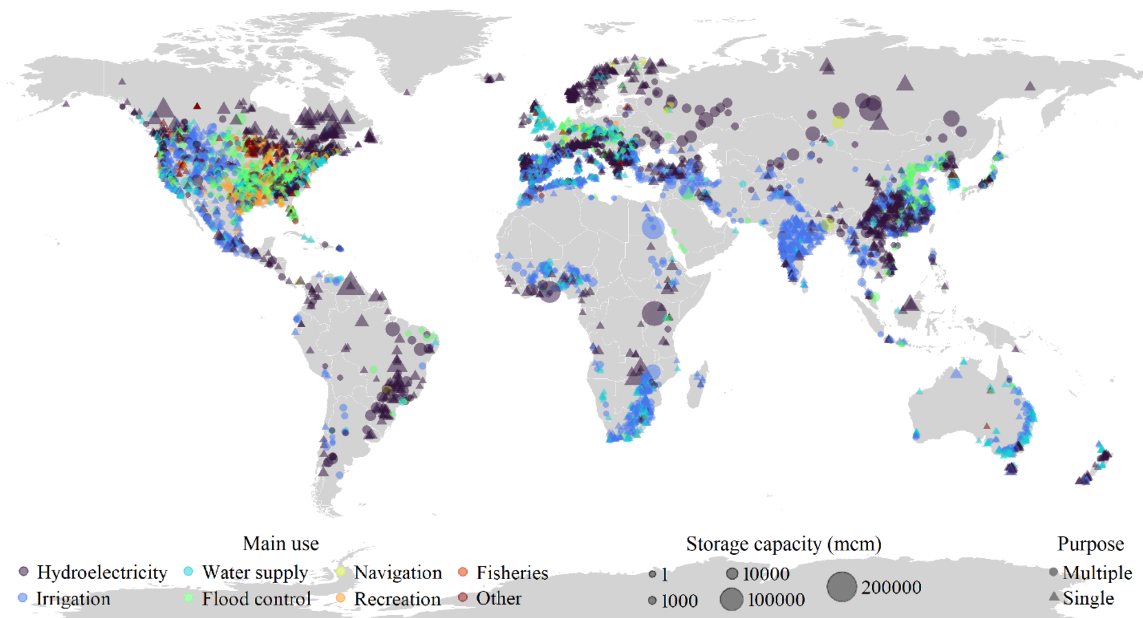


**Figure 3-1.** Global distribution of dams, according to their main use, storage capacity in million $m^3$ (mcm), and purpose. Most single and multi-purpose water projects are designed for hydropower generation. Data source: Georeferenced global Dam And Reservoir (GeoDAR) dataset v1.0 (Wang et al., 2022)

In the last 20 years, many reviews on e-flows have been specific to certain policies or regions. For instance, Adams (2014) assessed the environmental water requirements of estuaries, Hayes et al. (2018) focused on the advances in functional e-flows for temperate floodplain rivers, and O'Brien et al. (2021) assessed good e-flows practice for the small hydropower sector in Uganda. In addition, other reviews considered the influence of e-flows on the abundance of native riparian vegetation on lowland rivers (Miller et al., 2012), the methodologies and application in the Qianhe River in China (Hao et al., 2016), the socioeconomic values of restoring e-flows (Jorda-Capdevila & Rodríguez-Labajos, 2017), e-flows within the process of Water Framework Directive (WFD) implementation in Europe (Ramos et al., 2018), gaps between the science and implementation of e-flows in China (A. Chen et al., 2019), practical experiences of dam reoperation (i.e., change in the operational schedule for storing and releasing water to different us and volumes) (Owusu et al., 2021), and challenges on e-flow implementation in water-limited systems (Wineland et al., 2022). More generalized studies involve the review of global trends in the development and application of e-flow methodologies by Tharme (2003), the review and categorization of e-flow methods and requirements by Acreman & Dunbar (2004), and predicting ecological responses to e-flows (Webb et al., 2015).

Consequently, in the last two decades, many countries have recognized the importance of e-flows in water management and have incorporated e-flow provisions in updated water policy (Harwood et al., 2018). However, despite efforts, aquatic ecosystems continue to degrade at alarming rates, mainly due to habitat loss, direct overexploitation of resources (i.e., species, ecosystems, and water), and hydrological alteration (Salinas-Rodríguez et al., 2021). Policy does not always translate into practice. A good example of this shortfall is the adoption of the Water Resources Act, enacted by the State of Washington in 1971, which firmly established the need for instream flows to preserve fish, wildlife, and other environmental values (Hurst, 2015). Still, e-flows are frequently unmet during at least part of the year in the state's watersheds, even when instream flow rules are followed, as they do not prevent senior water rights holders (those with earlier priority dates) from using the water downstream (Hurst, 2015). Shortcomings happen elsewhere too, such as the lack of enforcement of California's Fish and Game Code statutes intended to maintain fish populations in "good condition" below dams (Grantham et al., 2014). These cases illustrate the underlying difficulty in shifting water away from human uses to streamflow because of economic and political resistance.

As suggested by Arthington, Kennen, et al. (2018), increased flow alteration and less water dedicated to the environment are expected in coming decades as human demands (e.g., flood and drought protection, electricity generation, urban/agricultural water supply, recreation) increase; and the consequences of that growing demand initiate a cascade of biophysical changes to ecosystems (Viers et al., 2017). The management of competing demands with imperfect knowledge and constraints of existing governance structure requires a systems approach, challenging the more linear thinking often applied to policy development and implementation (Thompson et al., 2018). Systems thinking can be a useful tool to better understand the various processes and interrelationships of complex systems, to provide effective decision-making strategies towards a more sustainable water resources management (Ram & Irfan, 2021; Zhang et al., 2021). In this study, we apply systems thinking concepts described hereafter to explore complex system interdependences in the implementation of e-flows and their implications for environmental water management.

### 3.3. Systems Thinking

Systems thinking provides a structured approach to understanding complex problems by viewing the overall systems, their components, interdependencies, and purpose (Mijic, 2021). System feedbacks can exacerbate existing problems and result in ineffective policy interventions when they are neglected (Refulio-Coronado et al., 2021). Consequently, systems thinking aspects have been applied to understand and address a wide range of issues in different settings, including environmental policy (Castro, 2022). Tasca et al. (2020) identified a need to bring systems thinking more generally into water resources planning and management because of the increasing complexity, scope, and urgency of environmental issues. The authors illustrate how a river system can be represented as a sociohydrological system with hierarchically organized sub-systems at successively lower levels (e.g., stream segments, reaches, pool riffle sequences, and microhabitat subsystems, as well as governance, consisting of institutions, networks, bureaucracies, and policies). Riverine ecosystems form a complex system of human and natural biotic and abiotic feedbacks, thus identifying clear ecological responses – either positive or negative – to flow alteration can become a significant challenge due to the inherited complexity of river systems (Arthington et al., 2006; Wu & Chen, 2018). As stated by Pahl-Wostl et al. (2013), implementation of e-flows requires a more systematic and integrated approach in order to capture the nuanced interaction between sociopolitical and environmental systems. According to the authors, the combined effect of poor governance and unrecognized complex feedbacks can lead to ineffective management, overexploitation of resources, and the ultimate long-term degradation of ecological integrity. Any remedy will require better and more explicit ways of acknowledging the enabling conditions and underlying drivers of conflict, explicit recognition and incorporation of systems interactions, and transparent accountability in water allocation decision-making and resulting trade-offs (Hjorth & Madani, 2023). Understanding the complexity of each sub-system and connectedness to overall system behavior allows scientists or managers to identify appropriate points of intervention to meet management objectives (**Figure 3-2**). In that way, systems thinking can help evolve policy-making from narrow, sectoral, and little coordinated, or even overlapping and conflicting, towards more integrated decision-making (Voulvoulis et al., 2022).

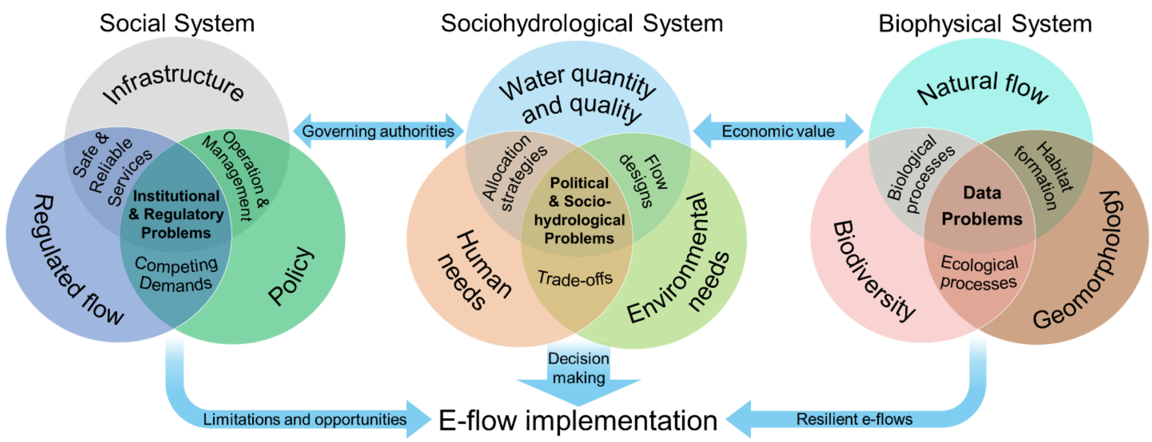


**Figure 3-2.** Conceptual model showing elements to be considered in each system and sub-system, within the e-flow implementation framework, to identify actions and interactions that can produce more effective results in balancing multiple human objectives and environmental needs.

## 3.4. Methods

### 3.4.1. Systematic Review and Bibliometric Analysis

We gathered key studies on the implementation of e-flows, including theory, concepts, and applications associated. For that purpose, thematic searches of published, peer-reviewed literature using topic-relevant keywords were conducted on Web of Science (WoS) (*https://www.webofknowledge.com*, August 8, 2022) search engine. Keywords used for this systematic review include "environmental flow", and "environmental flows", and "functional flows", or "implementation of environmental flow", or "environmental flow implementation", or "dam reoperation", or "implementation of environmental flows", or "environmental flows implementation". The search for articles published since 2001 resulted in 68 research and review articles, retrieved as Bibtex for further bibliometric analysis. The articles were analyzed to identify general limitations to the implementation of e-flows, when mentioned, and their co-occurrence.

A bibliometric analysis was conducted to identify the state of the intellectual structure and emerging trends in e-flows research. The WoS Bibtex file dataset was analyzed in RStudio (R version 4.0) (R Core Team, 2022) using the *Bibliometrix* R package (version 3.2.1) and its web application counterpart called *Biblioshiny* (Aria & Cuccurullo, 2017). *Bibliometrix* calculates frequency statistics and performs data visualization of leading authors, conceptual and intellectual maps, collaboration and co-citation networks, and overall trends of e-flows science (*sensu* Hao et al., 2021). Herein, a limitation is that e-flow implementation is not necessarily a scientific process that is being captured and reported in the peer-reviewed literature. Therefore, this analysis reflects findings on the research around implementation when reported and may not reflect all findings in e-flow practice. Additionally, bibliographies of selected papers were reviewed to find related and relevant publications for broadening the discussion below.

### 3.5. Results and Discussion

Our review identified 21 obstacles in the implementation of e-flows (**Table 3-1**), and their co-occurrence in 59 out of the 68 studies analyzed (**Figure 3-3**). The limitations found in the literature are either specific local barriers or broadly recognized obstacles mentioned by the authors. In general, a combination of these factors reinforces these impediments, such as insufficient political will, institutional roadblocks, limited scientific methods, conflicting interests, and lack of stakeholder support, capacity and resources (The Brisbane Declaration, 2018).

We identified the difficulty in shifting water from competing human uses (n=29) as the primary factor and leading challenge to overcome. Traditional approaches to water management have mostly focused on basin productivity, as indicated by Overton et al. (2014b), and thus these measures of economic development skew analysis toward valuing human benefits over environmental needs (Pahl-Wostl et al., 2013). Shinozaki & Shirakawa (2021) illustrate this problem in Japan, where even though e-flows can be reassessed during relicensing of hydropower projects every 10 years, conventional water withdrawals for consumptive use by rice paddies tend to be prioritized.

Richter (2009) states that the degree of 'sustainability' achieved in a water system is directly proportional to the degree to which stakeholders are satisfied with water allocation and management. Human use objectives can impose direct or indirect system constraints that cannot be countered without a prior change in system configuration. For instance, flood flows releases from a dam might be restricted due to downstream urban development, while at the same time water quality objectives might require elevated flows to dilute pollution (Aldous et al., 2011). In this case, human objectives prevent the implementation of high and low (natural) flow levels.

The remaining limitations identified form a host of other problems associated with implementing more sophisticated e-flows. Two of the most recognized and often co-occurring limitations identified were the lack of adequate hydrological data (n=24) and ecological data (n=28). These, for instance, may ultimately lead to difficulties in starting or continuing monitoring/adaptive management (n=28) efforts. The lack of resource/capacity (n=21), experimentation (n=19), regulatory enforcement (n=17), and differing authorities involved (n=18) were also recurrent problems, generally driven by the absence of funding and deficiencies in the relative importance given to e-flows when facing competing human priorities (n=29).

Major conflicts in water allocation are expected between hydropower generation and irrigation, drinking water and irrigation, and/or between conventional energy and agricultural purposes (Sharma & Kumar, 2020). Provisioning services arguably are often perceived to provide the most direct socio-economic benefits, and therefore, guide governance and management (Pahl-Wostl et al., 2013). Consequently, when faced with other competing demands e-flows are generally given low priority in allocation systems and are limited to low or 'minimum' flows (Richter, 2009). The prioritization of human uses is generally a result of politics and power differentials among competing interests, where economic and political power have precedence and resist changes in allocation (Sharma & Kumar, 2020). Misaligned purposes and inappropriate resource allocations are common governance problems, in addition to institutional fragmentation, unclear roles and responsibilities, poorly drafted legislation, and lack of long-term strategic planning (Hjorth & Madani, 2023).

Governance that fails to recognize the systems nature of decision-making (Thompson et al., 2018) produces fragmentation and duplication of authority, policy inconsistencies and high transaction costs (Folke et al., 2005). Action, then, is often compartmentalized and fragmented, where the bigger, integrated picture is lost (Tasca et al., 2020). Several common patterns have emerged that encompass the body of persistent challenges now facing the practice in e-flows. The challenges identified in this review are classified into data, institutional and regulatory, sociohydrological, and political problems, and are further discussed below. Therefore, we explore examples to show their occurrence and co-causality within the core system's problems illustrated in **Figure 3-2**, and points in a system's structure where interventions and action can produce more effective results.

**Table 3-1.** List of challenges in e-flow implementation mentioned in the literature.

| Limitations (n) | Country (Publications) |
|---|---|
| Competing priorities* (29) | Australia (Aldous et al., 2011; Conallin, Campbell, et al., 2018; Conallin, Wilson, et al., 2018; Loch et al., 2011; Stuart & Sharpe, 2022; Tennant & Sheed, 2001; Warner, 2014), Brazil (Brambilla et al., 2017), Chile (MacPherson & Salazar, 2020), China (Chen et al., 2019; Vonk et al., 2014; Wang et al., 2009), France (Warner, 2014), Japan (Shinozaki & |

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| | Shirakawa, 2021), Mexico (Sandoval-Solis et al., 2022), Poland (Dubel & Godyń, 2018), South Africa (Ramulifho et al., 2019; Russell, 2011), UK (Neachell & Petts, 2017; Warwick, 2012), US (Aldous et al., 2011; Sandoval-Solis et al., 2022; Stein et al., 2021), Multiple (Acreman & Ferguson, 2010; Arthington, Bhaduri, et al., 2018; O'Keeffe, 2018; Opperman et al., 2019; Overton et al., 2014b; Pahl-Wostl et al., 2013; Tickner et al., 2020; Watts et al., 2011; Wineland et al., 2022) |
| Lack of ecological data (28) | Australia (Koster & Crook, 2017; Stuart & Sharpe, 2022; Warner, 2014; Whiterod et al., 2017a), China (Chen & Wu, 2019; Cheng et al., 2018; Xingong Wang et al., 2012; Xiqin Wang et al., 2009; Wu et al., 2020b), Chile (MacPherson & Salazar, 2020), France (Warner, 2014), US (Grantham et al., 2014; Julian et al., 2016; Yarnell et al., 2020), South Africa (Quinn, 2012), South Korea (Kim et al., 2022), South Africa (Dube et al., 2015; Ramulifho et al., 2019), Spain (Mezger et al., 2019), Sweden (Bejarano et al., 2017), Tanzania (Olden et al., 2021), UK (Neachell & Petts, 2017), Multiple (Opperman et al., 2018; Overton et al., 2014b; A. Owusu et al., 2022; Pahl-Wostl et al., 2013; Poff et al., 2010; Tickner et al., 2020; Watts et al., 2011; Wineland et al., 2022) |
| Lack of monitoring/ adaptive management (28) | Australia (Aldous et al., 2011; Conallin et al., 2018; Conallin et al., 2018; Stuart & Sharpe, 2022; Whiterod et al., 2017), China (Chen et al., 2019; Cheng et al., 2018), South Africa (Quinn, 2012), Spain (Mezger et al., 2019), US (Aldous et al., 2011; DeWeber & Peterson, 2020; Grantham et al., 2014; Stein et al., 2021), Multiple (Acreman & Ferguson, 2010; Arthington, Bhaduri, et al., 2018; Arthington, Kennen, et al., 2018; Bruno & Siviglia, 2012; Harwood et al., 2018; O'Keeffe, 2018; Overton et al., 2014b; Pahl-Wostl et al., 2013; Pittock & Hartmann, 2011; Poff et al., 2010b; V. Ramos et al., 2018; Rolls et al., 2018; Watts et al., 2011; Wineland et al., 2022; Yang et al., 2016) |
| Lack of hydrological data (24) | Australia (Whiterod et al., 2017a), China (Chen & Wu, 2019a; Cheng et al., 2018; Xingong Wang et al., 2012; Xiqin Wang et al., 2009; Wu et al., 2020b), South Africa (Dube et al., 2015; Quinn, 2012), South Korea (Kim et al., 2022), Spain (Mezger et al., 2019), Sweden (Bejarano et al., 2017), UK (Neachell & Petts, 2017), US (Grantham et al., 2014; Julian et al., 2016; Yarnell et al., 2020), Tanzania (Olden et al., 2021), Multiple (Opperman et al., 2018; Overton et al., 2014b; A. Owusu et al., 2022; Pahl-Wostl et al., 2013; Poff et al., 2010; Tickner et al., 2020; Watts et al., 2011; Wineland et al., 2022) |
| Lack of resource/ capacity (21) | Australia (Conallin, Campbell, et al., 2018; Conallin, Wilson, et al., 2018; Warner, 2014), China (A. Chen et al., 2020; A. Chen & Wu, 2019a), France (Warner, 2014), Japan (Shinozaki & Shirakawa, 2021), South Africa (Quinn, 2012; Ramulifho et al., 2019), US (Grantham et al., 2014; Stein et al., 2021), Multiple (Acreman & Ferguson, 2010; Arthington, Bhaduri, et al., 2018; Harwood et al., 2018; O'Keeffe, 2018; Opperman et al., 2018; Overton et al., 2014b; Pahl-Wostl et al., 2013; Rolls et al., 2018; Watts et al., 2011; Wineland et al., 2022; Yang et al., 2016) |
| Differing authorities involved (19) | Australia (Loch et al., 2011), Chile (MacPherson & Salazar, 2020), China (Chen et al., 2020; Chen & Wu, 2019a; Cheng et al., 2018; Wu et al., 2020b), Japan (Shinozaki & Shirakawa, 2021), Mexico (Sandoval-Solis et al., 2022), South Africa (Ramulifho et al., 2019), UK (Warwick, 2012), |

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| | US (Acreman & Ferguson, 2010; Sandoval-Solis et al., 2022; Stein et al., 2021), Multiple (Acreman & Ferguson, 2010; Arthington, Bhaduri, et al., 2018; Overton et al., 2014b; Pahl-Wostl et al., 2013; Tickner et al., 2020; Watts et al., 2011; Wineland et al., 2022) |
| Lack of experimentation (18) | Australia (Whiterod et al., 2017a), China (Cheng et al., 2018), Japan (Mori et al., 2018; Shinozaki & Shirakawa, 2021), Mexico (Salinas-Rodríguez et al., 2018), US (DeWeber & Peterson, 2020), Multiple (Acreman & Ferguson, 2010; Arthington, Kennen, et al., 2018; Arthington, Bhaduri, et al., 2018; Bruno & Siviglia, 2012; A. Owusu et al., 2022; Pahl-Wostl et al., 2013; Poff et al., 2010; V. Ramos et al., 2018; Tickner et al., 2020; Watts et al., 2011; Wineland et al., 2022; Yang et al., 2016) |
| Lack of regulatory enforcement (17) | Chile (MacPherson & Salazar, 2020), China (Chen & Wu, 2019a; Cheng et al., 2018), South Africa (Quinn, 2012; Ramulifho et al., 2019) , UK (Neachell & Petts, 2017), US (Grantham et al., 2014a; Stein et al., 2021), Multiple (Arthington, Bhaduri, et al., 2018; Harwood et al., 2018; O'Keeffe, 2018; Opperman et al., 2019; A. Owusu et al., 2022; Pahl-Wostl et al., 2013; Pittock & Hartmann, 2011; Watts et al., 2011; Wineland et al., 2022) |
| Lack of stakeholder engagement (15) | Australia (Conallin, Campbell, et al., 2018; Conallin, Wilson, et al., 2018), China (Cheng et al., 2018), Japan (Shinozaki & Shirakawa, 2021), South Africa (Russell, 2011), US (DeWeber & Peterson, 2020; Stein et al., 2021), Multiple (Acreman & Ferguson, 2010; Arthington, Bhaduri, et al., 2018; O'Keeffe, 2018; Overton et al., 2014b; Owusu et al., 2022; Pahl-Wostl et al., 2013; Watts et al., 2011; Wineland et al., 2022) |
| Lack of standard definitions (14) | China (A. Chen & Wu, 2019a; Vonk et al., 2014; Wu et al., 2020), Poland (Dubel & Godyń, 2018), South Korea (Kim et al., 2022), Chile (MacPherson & Salazar, 2020), UK (Neachell & Petts, 2017; Warwick, 2012), US (Stein et al., 2021), Multiple (Acreman & Ferguson, 2010; O'Keeffe, 2018; Pahl-Wostl et al., 2013; V. Ramos et al., 2018; Wineland et al., 2022) |
| Differing e-flow methods (13) | China (Chen & Wu, 2019a; Vonk et al., 2014; Wang et al., 2009; Wu et al., 2020b), Chile (MacPherson & Salazar, 2020), Poland (Dubel & Godyń, 2018), US (Stein et al., 2021; Yarnell et al., 2020), Multiple (Acreman & Ferguson, 2010; Overton et al., 2014b; Pahl-Wostl et al., 2013; V. Ramos et al., 2018; Wineland et al., 2022) |
| Lack of political willingness (12) | Australia (Conallin, Campbell, et al., 2018; Conallin, Wilson, et al., 2018), China (Cheng et al., 2018), South Korea (Kim et al., 2022), Multiple (Arthington, Bhaduri, et al., 2018; Harwood et al., 2018; O'Keeffe, 2018; V. Ramos et al., 2018; Watts et al., 2011; Wineland et al., 2022) |
| Lack of initiative (12) | Australia (Warner, 2014), Chile (MacPherson & Salazar, 2020), France (Warner, 2014), Japan (Shinozaki & Shirakawa, 2021), South Africa (Quinn, 2012), UK (Neachell & Petts, 2017), US (Grantham et al., 2014), Multiple (Arthington, Bhaduri, et al., 2018; O'Keeffe, 2018; Overton et al., 2014b; A. Owusu et al., 2022; Pahl-Wostl et al., 2013; Wineland et al., 2022) |
| Implementation requires change to infrastructure (11) | Australia (Aldous et al., 2011; Warner, 2014), China (A. Chen et al., 2020; Vonk et al., 2014; X. Wang et al., 2009) France (Warner, 2014), US (Aldous et al., 2011; Grantham et al., 2014a), Multiple (Opperman et al., |

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| | 2019; Pittock & Hartmann, 2011; V. Ramos et al., 2018; Tickner et al., 2020; Watts et al., 2011) |
| Implementation requires system reoperation (10) | Australia (Loch et al., 2011; Warner, 2014), China (Vonk et al., 2014; Wang et al., 2009), France (Warner, 2014), US (Grantham et al., 2014a), Multiple (Arthington, Bhaduri, et al., 2018; Opperman et al., 2019; Pittock & Hartmann, 2011; Tickner et al., 2020; Watts et al., 2011) |
| Lack of awareness of human impacts (8) | Australia (Aldous et al., 2011; Warner, 2014), China (Vonk et al., 2014), France (Warner, 2014), US (Aldous et al., 2011; Stein et al., 2021), Multiple (Arthington, Bhaduri, et al., 2018; Tickner et al., 2020; Watts et al., 2011; Wineland et al., 2022) |
| Climate change (8) | Japan (Shinozaki & Shirakawa, 2021), South Africa (Ramulifho et al., 2019), Spain (Mezger et al., 2019), Tanzania (Olden et al., 2021), UK (Neachell & Petts, 2017), Multiple (O'Keeffe, 2018; Watts et al., 2011; Wineland et al., 2022) |
| Lack of criteria to select target sites (4) | China (Wang et al., 2009), US (Yarnell et al., 2015), Multiple (Meitzen et al., 2013; Overton et al., 2014b) |
| Lack of consideration to geomorphology (4) | South Korea (Kim et al., 2022), US (Grantham et al., 2014a), Multiple (Watts et al., 2011; Wineland et al., 2022) |
| Lack of information on riparian condition (1) | China (Wang et al., 2009) |
| Change in mindset (1) | Multiple (O'Keeffe, 2018) |

*Region-specific competing demands for water prioritized over environmental needs

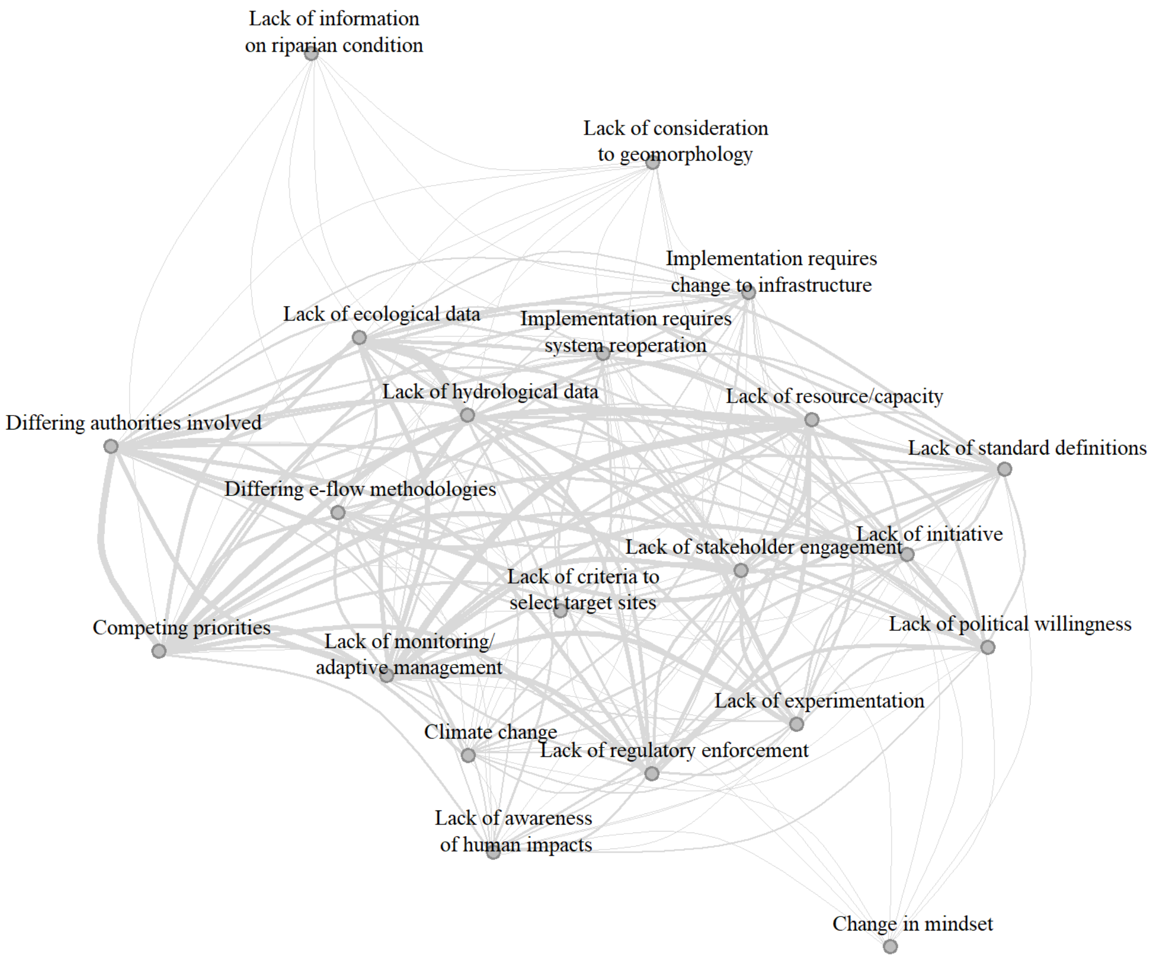


**Figure 3-3.** Co-occurrence network analysis of challenges in the implementation of e-flows identified in the systematic review (n = 59).

### 3.5.1. Data Problems

Two of the most recognizably mentioned obstacles to e-flow implementation were the **lack of hydrological data** (n=24) and **lack of ecological data** (n=28), generally co-occurring (**Figure 3-3**). The significant data needs, and consequently, financial and technical resources usually required to apply the Ecological Limits of Hydrologic Alteration (ELOHA) framework (Poff et al., 2010) for developing regional e-flow standards was a specific example, as discussed by Richter et al. (2012). References to the **lack of consideration to geomorphology** (n=4) and **lack of information on riparian condition** (n=1) are generally considered as data needs about the riverine environment to be maintained or restored, as these are mediating factors that can alter flow-ecology relationships (Taniguchi-Quan et al., 2022). Although not common limitations, these characteristics of the river channel and its surroundings can help policymakers implement a broader set of management options, such as land-use restrictions to slow development

(Giacomoni et al., 2013), pest control, grazing management, and riparian restoration (Thompson et al., 2018).

Consequently, these data problems lead to practical gaps, i.e., **lack of experimentation** (n=18), as actions at a dam cannot be predicted to produce specific results downstream with certainty, and many times it is unclear which dam releases will provide the desired results (Owusu et al., 2021). As stated by Meadows (2008), decision-makers cannot respond to information they do not have, cannot respond accurately to inaccurate information nor in a timely way to late information. For instance, a program aimed to create a 'sustainable' balance between human and environmental water uses in the UK, yet resulting in the maintenance of the *status quo*, due to the vague information flows that obscured inequities in water rights and constraints (Warwick, 2012). The ambiguous goal of 'sustainability' was unsuccessful. This also demonstrates the relevance of **adaptive management/monitoring** (n=28) ecological benefits produced by e-flows and their adequacy to achieve the environmental goals defined (Ramos et al., 2018).

For instance, monitoring of fish populations in China showed they have been impacted, in part by water infrastructure, with a decline of approximately 90% in the total number of fish fry for the four economically-important Chinese carp species (Cheng et al., 2018). To counter this problem, shifts in policy priorities have been promoted with a greater focus on river restoration and e-flow implementation (Cheng et al., 2018). Likewise, Mexico adopted measures to implement e-flows nationwide. E-flows are determined based on the Mexican Environmental Flows Norm, established at a river basin scale through a presidential decree for 50 years; and prioritization of basins is based on information on water availability and demand, biological richness, and conservation values (Salinas-Rodríguez et al., 2018).

Our bibliometric analysis revealed a potential lack of collaboration on e-flows research that corroborates to information asymmetries, with little representation from developing and emerging countries, where it is unknown to the degree that e-flows are being implemented and, even if present, unlikely to be well represented in the scientific literature (**Figure 3-4**). Brazil is a remarkable example, which despite recent widespread dam-building (Aledo Tur et al., 2018; Arias et al., 2020), was not detected in this analysis. Unfortunately, international collaboration has been mostly limited among the countries with the most scientific production, namely the USA (particularly California), Australia and the UK, followed by the Netherlands, Canada, Germany and South Africa. A lack of cross-collaboration in scientific research can also be a limiting factor to e-flow implementation, as most river research still operates within local paradigms (Tasca et al., 2020), especially when considering the bias in the literature with a prevalence of regions with a Mediterranean-montane climate (notably California, Australia, Chile, and South Africa).

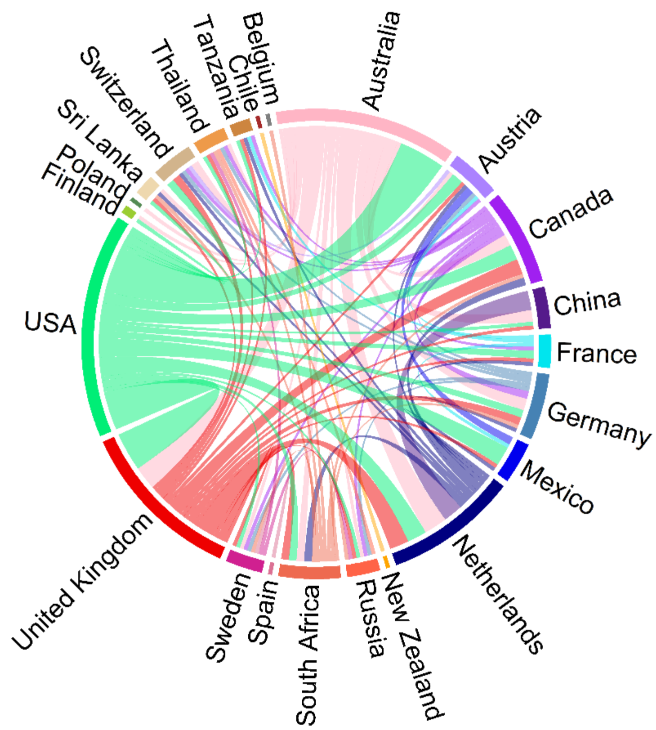


**Figure 3-4.** Country collaboration network based on the bibliometric analysis of the literature (n=68), showing collaboration among countries based on each country's publication output

### 3.5.2. Institutional and Regulatory Problems

Other limitations identified were the institutional and regulatory problems. Notably, **differing authorities involved** (n=19) constitute barriers, at times with overlapping roles. This unclear jurisdiction can also lead to the use of conflicting or ambiguous definitions in policy goals resulting in a **lack of standard definitions** (n=14). Thus, policies with unclear goals, such as to achieve a "sustainable" balance between water users (Warwick, 2012), have "beneficial uses" of water (Hurst, 2015), maintain a "dry weather flow" (Neachell & Petts, 2017), or allow water abstractions within "reasonable limits" (MacPherson & Salazar, 2020) are found in the literature. Numerous competing definitions for e-flows, such as "ecological flow", "ecological minimum flow", and "minimum acceptable flow" (Ramos et al., 2018), can also be found in addition to **differing e-flow methods** (n=13). For instance, according to Wu et al. (2020b), e-flow implementation in hydropower projects is regulated by the National Energy Administration of China, meanwhile water projects with other main purposes are regulated by the Ministry of Water Resources of the People's Republic of China. Both entities together use seven different terms to refer to e-flows, with inconsistent definitions, and both recommend various differing standard methodologies to assess e-flow requirements, with no specific explanation on the methods selection principle. The lack of integrated calculation methods in China is also highlighted by Wang et al. (2009), as flow prescriptions are not easily transferable across a country with a such diverse geography. Therefore, the **lack of regulatory enforcement** (n=17) was also identified as a recurrent problem in e-flow implementation. On the other hand, flow releases that comply with the law can also be difficult to implement due to their specific standards and objectives, and onerous enforcement (Owusu et al., 2021). Although such regulations pave the way for the implementation of e-flows in Europe, legislations at the national and regional levels and obligations under the WFD, Habitats Directive, other European Directives, and international commitments need to be considered (European Commission, 2016).

Similarly, another limitation we identified was **climate change** (n=8), as traditional approaches assume stationarity (Overton et al., 2014b). For example, in the state of Washington, instream flow rules do not require scientifically-grounded standards, by definition are not meeting desired “maximum net benefits” and cannot be modified to changing conditions except through additional notice-and-comment rulemaking (Hurst, 2015). Acknowledging potential climate change impacts into planning can be an opportunity to re-examine policies and management procedures for rivers and infrastructure (Thompson et al., 2018; Watts et al., 2011), as the increasing uncertainty and conflicts will require constant updates to system rules to adapt to nonstationary conditions (Brown et al., 2013).

Pittock & Hartmann (2011) suggest that opportunistic policy windows for reoperation (i.e., change in the operational schedule for storing and releasing water from reservoirs) of dams are safety reviews, utility management, systems operations, and relicensing. However, physical, financial, and legal constraints can limit the implementation of e-flows when a **change to infrastructure** (n=11) and **system reoperation** (n=10) are needed. Changes in infrastructure may necessitate reconsideration to operational design and/or siting of infrastructure (Opperman et al., 2019), often leading to retrofitting, or perhaps decommissioning (Arthington, Kennen, et al., 2018) in cases of infrastructure without specific e-flow release devices (Ramos et al., 2018).

Another complication that emerges from institutional and regulatory problems is the rigid structures that lack adaptability. Due to the complex and relatively uncertain feedback responses among water demands, land uses, hydrological variability, biodiversity, and aquatic ecosystem services, the governance systems that manage e-flows must be adaptive, flexible, and capable of learning from experience (Pahl-Wostl et al., 2013). A response to feedback systems is the creation of feedback policies. Static policies cannot respond to system dynamics and are more likely to produce temporary solutions and a greater number of escalating problems (Grigg, 2016). In this way, adaptive management is a promising approach necessary to implement long-term strategies to maintain riverine ecosystems; however, this approach is difficult to implement under rigid regulatory and institutional governance (Folke et al., 2005; Bruno & Siviglia, 2012). Thus, management objectives and physical constraints (e.g., hydropower generation, type and size of dam outlets), can directly negate consideration of dam reoperation (Wang et al., 2009).

For instance, the US Federal Energy Regulatory Commission (FERC) is responsible for the licensing process of non-federal hydropower projects in the country. FERC has been disregarding the potential impacts of climate change on hydropower operations, stating that “although there is consensus that climate change is occurring, we are not aware of any climate change models that are known to have the accuracy that would be needed to predict the degree of specific resource impacts and serve as the basis for informing license conditions” (Federal Energy Regulatory Commission, 2009). Viers (2011) argues that the issuance of FERC licenses will ensure a series of fixed operating rules based on stationary hydrology for the life of the license, typically 30-50 years in length. Viers & Nover (2018) suggest the inclusion of formal environmental impact studies, more academic sensitivity analyses, or the development of climate-informed “worst case” scenario planning into their licensing process. According to the authors, FERC should make licenses adaptive (adaptively alter operations based on new information) to offset that oversight. However, the agency continues to dismiss the need for adaptation and flexibility in operations to support environmental water needs under a changing climate. According to FERC, predicting future flow scenarios in climate change studies is “too speculative given the state of the science at this time” (Ulibarri & Scott, 2019). With the rapid changes in climate science as well as the evolving body of e-flows work, these cases illustrate the existing challenges of e-flow implementation.

### 3.5.3. Sociohydrological Problems

Securing e-flows is also limited by challenges in sociohydrology (Pande & Sivapalan, 2017), such as the **lack of awareness of the multiple human impacts** (n=8) on the environment caused by river regulation. This is linked to the ignorance of the multiple ecological and social benefits provided by river systems, their related ecosystem (e.g., wetlands and floodplains) and water needs, as well as the exclusion of specific stakeholders from central decision-making and priority-setting in river basins (The Brisbane Declaration, 2018). Stakeholder engagement in these cases works as an information-dissemination exercise for government departments or implementing agencies, with an opportunity to comment (Acreman & Ferguson, 2010). Ignorance on the importance and functioning of natural systems leads to disregard for human impacts on the environment and to how people benefit from the different components of a river system. Another limitation we identified was the **lack of stakeholder engagement** (n=15), reflected as indifference toward or ignorance of the problem. Conflicts regarding what stakeholders want from water allocation and flow patterns persist when decision-making processes are opaque and/or unbalanced in representation (Carr, 2015; Mehrparvar et al., 2020).

Comparing successful and unsuccessful cases of e-flow implementation, Owusu et al. (2022) found that the key difference between them was stakeholders’ involvement, especially the support of scientists, who increase the odds of successful dam reoperation. Yet, the authors emphasize that scientists should play a supportive role rather than drive the process. O’Keeffe (2018) indicates that the **change in mindset** (n=1) of all levels of stakeholders, including water policymakers, managers, and scientists, is the most intractable limiting factor. The mindset is determined by their paradigm (i.e., the pattern of values, beliefs, and assumptions) that sees rivers as resources to be used to maximum benefit, which ultimately drives planning, policies, and decisions.

The minimum flow paradigm became a widely accepted form of e-flows also as a reflection of sociohydrological problems. Despite advances in e-flow science and policy, minimum flows remains the most implemented approach (Owusu et al., 2021). For example, in China the minimum flow demand is widely accepted and usually adopted as 10% of the annual average flow as an empirical rule (Wu & Chen, 2018). Minimum flows are also the most widely legal provisioning form of e-flows adopted in the US. Schramm et al. (2016) analyzed 300 licenses of hydropower projects in the US and found that most plants throughout the country are required to release a static minimum flow or the natural inflow, whichever is less, in either the facility tailrace or bypass. The authors mention that the California hydrologic region is the only area where the majority of minimum flow releases change by season or annual water conditions.

Although minimum flows mimic some hydrological flow characteristics, they are not designed to capture more nuanced and critical aspects of the flow regime throughout the year (Grantham et al., 2020). And even when implemented, e-flow allocations with lesser seniority or priority are among the first to be sacrificed when water is in short supply; the complete drying of rivers by water extractions, particularly in arid and semi-arid regions, is not uncommon (Richter, 2009). Therefore, the water governance systems need to be designed to send feedback about the consequences of decision-making directly, quickly, and compellingly to the decision-makers.

A classic example is California's 'first in time, first in right' system of water rights, combined with the overallocation of many river systems (Grantham & Viers, 2014). The state's water system is an enigma of interconnections of geographic, sociopolitical, infrastructure, and environmental factors. As a result, human and natural systems form a complex web of competing demands for freshwater, which has been the focus of continuous political, legislative, and legal battles (Stewart et al., 2020). The unrealistic allocation of water creates "paper water", as the water proposed for transfer does not translate into the natural system's capacity of producing water for human supply (Chong & Sunding, 2006).

Similarly, unfeasible water allocations have happened in Australia, where such rights are termed "sleeper rights", which can be later activated for larger use of water than in previous years (Chong & Sunding, 2006). Overallocation in the Hawkesbury-Nepean River in Australia has reduced e-flows from the recommended 80% to around 3%; a scenario similar to the Durance River in France, in which 97.5% of river flows are diverted for hydropower production (Warner, 2014). One approach to this type of problem is the regulation of the commons enforced by policing and penalties, to create the feedback link from the condition of the resource through regulators to users. When there is a commonly shared resource, every user benefits directly from its use but also shares the costs of its abuse with everyone else.

After California's 2012-2015 drought, in which low flows and high temperatures reduced water quality and impaired habitat for native fish species and supported expansions of invasive species, the discussion of an environmental water right began (Lund, Medellin-Azuara, et al., 2018). Agricultural demands for irrigation supply under drought-induced water scarcity have resulted in widespread groundwater overdraft, resulting in decreased base flow and localized subsidence (Pinter et al., 2019), leading government agencies to expand e-flow regulations (Lund, Medellin-Azuara, et al., 2018). Likewise, in the UK, water rights were perpetual, as established by the Water Resources Act in 1963, therefore, new licenses could only be issued if they did not impact existing rights (Warwick, 2012). To change that, time-limited licenses started being issued in the 1990s, together with curtailments to protect environmental features on a case-by-case basis, in which the Environment Agency can, for instance, ban on spray irrigation or non-essential water use during drought (Warwick, 2012).

### 3.5.4. Political Problems

We identified that the abovementioned limitations are generally and ultimately driven by the absence of funding and deficiencies in the relative importance given to e-flows when facing **competing priorities** (n=29), such as pollution (Neachell & Petts, 2017a; Wang et al., 2009), agricultural, industrial, or municipal use (Shinozaki & Shirakawa, 2021; Tickner et al., 2020; Wineland et al., 2022), flood control and/or hydropower (Brambilla et al., 2017; Vonk et al., 2016; Warner, 2014; Watts et al., 2011). The controversies among these limitations are generally caused by political problems, i.e., the **lack of political willingness** (n=12) to recover water for the environment, especially in over-allocated systems with conflicts between economic development and conservation. This results in a **lack of resource/capacity (n=21)**, for instance, due to changes in funding cycles or priorities within government agencies, and short-term commitments to environmental water management and monitoring (Conallin, Wilson, et al., 2018). That, in turn, leads to a **lack of initiative** (n=12) in reallocating water resources for protecting riverine environments, as human uses are prioritized (Shinozaki & Shirakawa, 2021). Ultimately, political decisions determine the level of acceptable compromise of human uses in face of environmental water requirements (Warner, 2014).

As stated by Meadows (2008), “If a government proclaims its interest in protecting the environment but allocates little money or effort toward that goal, environmental protection is not, in fact, the government’s purpose. Purposes are deduced from behavior, not from rhetoric or stated goals.”. For instance, in 2007, e-flows were required to maintain the normal function and state of streams in South Korea through the River Act (Kim et al., 2022). In addition, e-flows to conserve the health of aquatic ecosystems have been endorsed in the Water Environment Conservation Act of 2017; however, implementation of e-flows is still in its early stages due to the **lack of established criteria for the selection of target sites** (n=4) (Kim et al., 2022). Similarly, in Chile, e-flows have been applied in a discretionary and ad hoc manner, as safeguarding e-flows may be costly and politically unpalatable, potentially requiring the redirection of water away from consumptive, economic purposes (MacPherson & Salazar, 2020). Therefore, policy goals can also be eroded due to political and economic pressures.

The systems nature of decision-making and the need for information sharing can lead to the fragmentation of effort, and a failure to take advantage of local adaptive management learnings (Thompson et al., 2018). This surfaces the need for policy coherence, in order to reduce conflicts and strengthen interactions, coordination, and effects of governmental actions to achieve a desirable goal. Assessing policy coherence and the potential implications of one sectoral policy across the system is key to minimizing trade-offs and establishing compromises (Pereira Ramos et al., 2021). Policy coherence might include policy harmonization, i.e., making the regulatory requirements, laws, or governmental policies of different jurisdictions identical or at least more similar, or by assigning decisions to a common political authority, as defined by Majone (2014). Polycentric governance may be of significance in responding to ecosystem dynamics at different scales (Folke et al., 2005) and provide better results when involving all stakeholders and therefore addressing all the economic activities within a resource system (Refulio-Coronado et al., 2021). For instance, in 2000 the WFD was the first legislation ruling in the European Union to use ecological conditions as the benchmark for the management of "ecological flows" (Wu & Chen, 2018). The WFD sets a common definition and understanding of how ecological flows should be calculated to facilitate their integration into river basin management plans in Europe (European Commission, 2016).

Representative of these political challenges are efforts for restoring fish population. Salmonid populations in the Yuba River, California illustrate a case in which e-flows alone are unable to achieve the purpose of maintaining native species. As stated by Viers (2012), salmon "cannot go beyond the dam because there is no water, and there is no water because they cannot go beyond the dam". Restoration in this case would require not only e-flow releases in proper timing, quantity, and quality from an upstream hydropower project but also fish passage downstream for the reintroduction of salmonids in the system. Fish hatcheries are another example of efforts that try to balance the effects of dysconnectivity, and hydrologic alteration caused by river regulation and inappropriate e-flows. According to Sturrock et al. (2019), fish hatcheries in California often eliminate the entire migratory corridor by trucking fish directly to the estuary to prevent in-river mortality, particularly during droughts. However, the authors state that this practice has inadvertently caused excessive straying rates due to genetic homogenization and increasingly synchronized population dynamics. Similarly, Brown et al. (2013) found that restoration projects of Atlantic salmon have not yielded self-sustaining populations in any eastern US river, despite hundreds of millions of dollars spent in hatcheries, although a complete extinction was avoided in a few rivers, albeit at the expense of genetic integrity. In addition, the authors mention the poor performance of fish ladders by portraying the mean passage efficiency of <3% from the first dam up to the spawning grounds for American shad. The authors indicate that the systemic cause of fish declines (i.e., main stem dams and overfishing) were not properly addressed, exerting a greater pressure on natural resources management agencies to restore fisheries. Consequently, more funding is applied to the problem, more agency personnel are hired, and it becomes difficult to dismantle ineffective programs.

Likewise, flow restoration projects that neglect how sediment availability influences abiotic and biotic responses to flow by reshaping channel morphology and creating habitat (Wohl & Brian, 2015) can be unsuccessful or even deleterious. For instance, channel incision in sediment-deprived reaches can be aggravated, and inevitably floodplain connectivity is further jeopardized (Consoli et al., 2022). Gravel augmentation projects have been implemented in California to improve anadromous salmonid spawning habitat, however beneficial results tend to be temporary, as placed gravels were usually scoured and transported downstream by subsequent high flows (Harvey et al., 2005). Moreover, ecosystem restoration with poor consideration of the influence of hydrological alteration on freshwater biodiversity across spatial scales creates a paradoxical situation where even e-flows may inadvertently contribute to further biodiversity declines (Rolls et al., 2018). Restoration considering hydrology (i.e., e-flows) in isolation addresses symptoms, meanwhile process-based restoration also accounting for geomorphology, connectivity and biology addresses the causes of degradation (Beechie et al., 2010).

### 3.6. Recommendations

The limitations identified in this study, including lack of financial resources, organizational capacity, and regulatory enforcement, are a reflection of the primary factor identified in the review, which is the prioritization of human uses over environmental needs. Below, we provide a set of recommendations based on the reviewed literature and the systemic problems identified to address this overarching factor. Although these recommendations do not consider resource limitations and other practical on-the-ground considerations (e.g., data-poor river basins), we discuss several examples from the literature to provide a clearer path for successful e-flow implementation.

#### 3.6.1. System Mapping: Understand the system to be managed and the management system

A systemic map or conceptual model can express insights about the purpose, processes, and structures governing the system and producing its behavior (Mingers & White, 2010; Tasca et al., 2020). A systems perspective allows the integration of the subsystem goals, as it can bring information not only about existing problems (system unintended outcomes) but also on elements (resources), their arrangement (hierarchy), rules, and consequences. System mapping can be produced in terms of causalities (causal loop diagrams) and flows (flow charts), that ultimately reveal effective intervention points (Haraldsson & Sverdrup, 2021), by surfacing the system problems (Mijic, 2021), or areas where reliable quantitative information is not available (Mingers & White, 2010). It is also suggested that system mapping be an interactive participatory process of managers, governments, infrastructure operators, farmers, and other stakeholders to integrate multiple general perspectives (Mijic, 2021; Ram & Irfan, 2021).

Understanding the system to be managed may involve mapping physical, natural, and/or human components, meanwhile, the management system may involve mapping people, resources, and procedures (Grigg, 2016). If the system to be managed is a water supply reservoir, the management system involves data, operational rules based on legal requirements, and a decision support system (Grigg, 2016). In this way, we can identify the level of flexibility of the systems involved, i.e., the case-specific context, limits, and barriers that can or cannot be adapted or overcome (System accommodation to constraint *vs.* Constraint accommodation to the system). For instance, Mijic (2021) used system mapping to identify that most infrastructure and technological solutions to improve water quality at Lake Windermere in the UK, would fail unless implemented across the system as a whole.

As described in section 5.2., information on system deficiencies might not be enough for triggering action due to institutional constraints, requiring therefore an accommodation to the system constraint. For instance, FERC can reject mitigation measures due to ‘high’ costs, if their cost represents more than ten percent of a project’s annual power benefits (Black et al., 1998). To counter that the economic analysis could consider the recreational fishing benefits, which could outweigh the costs to implement it (Black et al., 1998). However, the construction of a fish ladder to minimize the impact of damming on migratory species is not always carried out due to the costs involved. This constraint can be accommodated to the system needs in specific windows of opportunity that are open during the relicensing process. In the relicensing of existing hydropower projects, other US Federal agencies have a “conditioning authority” through which they can issue conditions that FERC must incorporate into a license, including requirements to change a project’s design or its operation, e.g., a retrofit to include a fish passage structure or change its e-flow release schedule (Opperman et al., 2019).

The main operating purpose of a dam influences dam reoperation strategies and reoperation might require integration across sectors or involve multiple dams to simultaneously achieve human and environmental objectives (Vonk et al., 2014). Systems thinking can assist different bodies to work together with a shared view to develop more coherent management options and policies, to provide multiple outcomes while considering and preventing unintended consequences (Mijic, 2021). Considering that, we recommend that the first step for e-flow implementation be system mapping to surface the problems in effect and the limitations in place, as discussed in the examples above. This intends to fix information flows and avoid delays in response by identifying mechanisms that impose conditions and constraints that can limit success. A system map allows the first considerations of alternative practices and policy options, as well as inflexible frameworks to be reconsidered, countered, and overcome. The identification and extraction of relationships among and within social and ecological systems can also later be used as input variables into empirical models (Bouchet et al., 2022), which can be used in the next step, the system analysis.

### 3.6.2. System Analysis: Support decision-making using suitable modeling tools

The conceptualization in the system mapping works as an actual model development (Haraldsson & Sverdrup, 2021), and building a model can change paradigms as the builder is forced to see the system as a whole (Meadows, 2008). The delineation and quantification of system interconnections and influences allow for building and testing a computer model (Mingers & White, 2010). That allows the learning process during planning and implementation, by identifying the problems to be solved and the questions to be answered based on the information available and the understanding provided by it.

For instance, water supply systems require careful simulation as their outcomes need to provide very high reliability, with sensitivity restricted to critical periods (e.g., droughts) (Marchau et al., 2019). Consequently, a detailed representation of temporal and spatial variability (i.e., demands, inflows, outflows, competing needs) is required to accurately assess these systems (Marchau et al., 2019). The model outcomes can then be effective communication tools to engage stakeholders in technical decision-making. In hydropower systems, energy generation tends to follow electricity price signals and can also reflect constraints imposed on a facility (e.g., multi-objective reservoir) that can limit the timing, period, and intensity of power generation (Stoll et al., 2017). Therefore, adding the economic-driven factor of hydropower in water systems can produce more realistic information for better decisions. In that way, adverse effects of hydropower can be better assessed to provide alternatives that minimize them by restoring vital features of the natural flow regime and/or avoiding hydropower-induced habitat bottlenecks, such as through the adoption of restricted ramping rates (Freeman et al., 2001).

Hydropower system design and planning that fully integrates environmental and social resources remain relatively rare, although this integration can provide relevant information to energy planners and operators, and provide better opportunities to achieve climate and energy goals while also supporting the ecological integrity of rivers (Opperman et al., 2023; Rheinheimer et al., 2023). Sector integration, aided by system analysis is needed to address the main challenge of competing priorities by promoting the transparent assessment of needs, allocations and inherent trade-offs (Rheinheimer et al., 2022). Better modeling, including forecasted energy prices and/or hydrological conditions, can also help explore alternative flood control rules for flexibility in the operation and management of reservoirs, resulting in "extra" water that could be used for other purposes, including e-flows (Lee et al., 2006; Zarei et al., 2021). In addition, stress testing can assess the system's performance to meet the desired objectives under critical periods (namely, the driest or wettest season, the highest recorded flood) (Nagy et al., 2013). Modeling of e-flow requirements has been employed for planning flow standards in Texas, USA, to advance statewide implementation efforts (Wurbs & Hoffpauir, 2017). Similarly, modeling studies have allowed the analysis of trade-offs among water uses to allow preliminary discussions on the implementation of functional flows by water agencies in Brazil (Dalcin et al., 2022), and the initial implementation stages of the California Environmental Flows Framework on a small subset of watersheds in California (CEFWG, 2021).

Hydropower planning and operations, due to the significant infrastructure investment and high value as a source of renewable energy, is perhaps the most viable area for continued improvement of systems analysis tools in water systems operations (Brown et al., 2015). They also state that especially in hydropower-dominated systems economic incentive justifies optimizing the use of operational flexibility for maximizing hydropower revenue subject to nonpower constraints and objectives. Water and energy systems can be represented separately in modeling frameworks; their interdependency is indirectly considered through the input variables employed in each model (Voisin et al., 2016). For instance, Jager & Martinez (2012) used an energy model to estimate relative electricity value and one to estimate relative salmon production to optimize e-flows for hydropower generation and the environment in the Tuolumne River, California. However, as stated by Stevanato et al. (2021), model integration can consider a joint optimization according to one objective function that allocates the resources in the two systems. In that way, coupled water-energy models allow the assessment of their feedbacks without coarse approximations in their dynamic behavior, and therefore, a more scientifically solid energy and hydrological planning (Stevanato et al., 2021). In that sense, coupled water-energy models can help integrate the management of the complex interactions among the hydrologic, environmental, engineering, and socioeconomic dimensions of water resource systems when hydropower systems are present.

Simulation models can be used to assess the performance of alternative water management system configurations, plans, or policies, including economic and environmental performance indicators and trade-off analysis (Brown et al., 2015; Loucks & van Beek, 2017). For instance, Rheinheimer et al. (2016) considered climate-adaptive instream flow requirements in hydropower systems using wholesale electricity prices to better inform release decisions. Similarly, Willis et al. (2022) and Maskey et al. (2022) used a multi-objective water system simulation model built in *Pywr* written in Python (van Rossum, 1995), which simulates customizable water allocation and operation rules throughout complex managed water systems (Tomlinson et al., 2020). The modeling framework simulates a daily time step basin-scale water resources system with optimization of discretionary hydropower based on day-ahead market, a key methodological advancement over typical water system models (Rheinheimer et al., 2022). In this coupled water-energy modeling effort, forecasted energy prices and hydrological conditions drive generation, as water is allocated based on a monthly planning optimization model with partial foresight and a long (multi-month) planning horizon, and a daily scheduling model. In that way, competing demands for human uses are accounted for, meanwhile the influence of current or alternative scenarios can be assessed, including climate change impacts (Maskey et al., 2022) and the implementation of alternative e-flow schedules (Willis et al., 2022).

Therefore, we recommend the employment of modeling as a useful tool for identifying more realistic, reliable, and robust infrastructure designs and operating policy rule sets for a given hydrological input, aiming at historical or future scenarios. Models have had an increasingly important role in providing a common way for planners and managers to predict the behavior of any proposed water resources system design or management policy before it is implemented (Loucks & van Beek, 2017). Thus, a system analysis with coupled water-energy models can help better identify and assess different scenarios by allowing the modeler to test different rules and goals, producing new information flows for decision-making. Based on the problems assessed, trade-offs among conflicting goals are identified and the appropriate strategies can be prioritized.

### 3.6.3. System Resilience: Move from simplistic flow designs to ecosystem process-based approaches

Resilience is a system's ability to survive and persist within a variable environment. As defined by Meadows (2008), resiliency is not achieved by static or constant states over time, as systems can be very dynamic, with short-term oscillation, periodic outbreaks, long cycles of succession, climax, and collapse within their nature. Comparably, the resilience of a river system is in its dynamic behavior. Therefore, building resilience into e-flows involves setting flow targets necessary to achieve ecosystem goals by mimicking key components of a natural river's flow.

Discrepancies between the desired and actual state of a system compel managers to intervene, however a lack of understanding of system structure and functions results in ineffective interventions (Endreny, 2020). Flow designs based on institutional and technological panaceas without long-term monitoring of their performance and effectiveness produce no revision and critical reflection on practice (Pahl-Wostl et al., 2013). As noted by Viers & Nover (2018), despite the extraordinary effort and money generally directed toward hydropower relicensing, the requirements for post-licensure performance monitoring are comparatively negligible. The authors state that, even though requirements for monitoring in hydropower licenses are commonplace, they are not used in an adaptive manner as licenses typically do not specify any consequences based on unfavorable outcomes of monitoring. For instance, large efforts such as the WFD in Europe propose the implementation of e-flows to counter the negative ecological impacts caused by river regulation in many countries. Yet, a minimum flow paradigm that disregards the different flow regimes for normal or dry years is still largely adopted in the participating countries (Acreman & Ferguson, 2010; Ramos et al., 2018). Ultimately, the unachievable goals can lead to more problems such as when only "paper water" is available to the environment due to overexploitation (Chong & Sunding, 2006), or when restoration addresses the only symptoms of hydrologic alteration (Sturrock et al., 2019).

Protecting the dynamism of river systems involves setting dynamic boundaries in water withdrawals from/releases to rivers, reflecting changing societal needs and values over time as well as new scientific knowledge (Richter, 2009). The Building Block Methodology was created to reproduce the dynamic flow regime components consisting of different 'blocks' of flow in South Africa (King et al., 2000). The method establishes monthly volumes of low flows, and the duration, timing, and magnitude of floods, for both maintenance and drought years, based on data, global literature, expert opinion, and local knowledge (King, 2016). More recently, progress in e-flows has focused on specific functional flows that support natural disturbances that promote the physical dynamics and drive ecosystem functions (Taniguchi-Quan et al., 2022; Yarnell et al., 2015, 2022). The functional flows approach emphasizes process-based hydrograph components that are key to specific environmental outcomes and can guide management of regulated river systems (Viers et al., 2017). Developing inherent relationships between ecological responses to flow alteration is necessary to enhance the scientific credibility of flow designs (Poff & Matthews, 2013).

E-flows that fail to support ecological functions can result in inefficiencies in water allocation that foster conflicts between competing water demands (Stein et al., 2021). As noted by O'Keeffe (2018), misunderstanding and resistance to implementing e-flows are not uncommon. Moyle et al. (2018) address a vivid example revolving the controversies around the Delta smelt, an "economically insignificant" fish protected by state and federal conservation mandates. Complicated by the intersection of declining water quality, aging infrastructure, climate change, and multi-institutional governance, the deficiencies of single species management have been laid bare (Luoma et al., 2015). A species recovery plan remains in limbo as the state negotiates with local water authorities on how to interpret the science and how best to implement water allocations for the environment (Hanemann & Dyckman, 2009). Although rigor and ecologically comprehensive processes in setting e-flows targets can increase the challenge of implementing e-flows (e.g., the need for better data, capacity, and resources) and exacerbate competing demands, they are indeed required if e-flows are to be effectively adopted.

In California, functional flow components such as peak flows, dry-season low baseflows, wet season initiation flows, spring recession flows, and interannual variability can be identified in all rivers although their dimensions (timing, magnitude, frequency, and duration) vary regionally (Grantham et al., 2020). These flow regime components are key for focal native species such as salmonids to thrive in the State, for instance, providing cues for migration and spawning (Yarnell et al., 2015). Functional flows rest on the assumption that reservoirs releases that reproduce these key flow components will produce the necessary hydrologic signals that trigger biophysical processes upon which native biological communities depend (Yarnell et al., 2020).

Viers et al. (2017) applied this framework to connect natural flow regime components to specific biophysical riverine functions in various hydroclimatic systems, to help inform water resource management strategies and functional flow requirements globally. The authors consider the functional flow approach a plausible management option to maintain ecosystem services and biodiversity while also continuing conventional reservoir operations. For instance, in a system analysis, Willis et al. (2022) modeled the adoption of functional flows in the central Sierra Nevada, California. The authors found that generally functional flows can provide enhanced ecological functions and reduce uncontrolled spills, although at the expense of lower agricultural deliveries, with the extent of trade-offs depending on the river basin and water year.

However, institutional and regulatory structures need to be flexible enough so that water releases can be timed to accommodate the needs of target species or ecosystems. Thus, rather than focusing on specific species or life-stage requirements through static minimum instream flow requirements that do not enhance ecological integrity, broadly, the implementation of e-flows should focus on the functional attributes and processes that enhance biogeochemical cycling, structural habitat formation, and ecosystem maintenance (Grantham et al., 2020; Yarnell et al., 2020). In addition, knowledge-driven decisions to solve problems based on research and analysis are key to adapting water management and reservoir operations. Therefore, we recommend that essential hydrograph components be identified for guiding the implementation of ecosystem process-based e-flows. The focus on elements of the natural flow regime should replace generalized e-flows based on simplistic thresholds that work against ecosystem health.

### 3.7. Conclusion

E-flow implementation is primarily constrained by the unbalanced competing human priorities having precedence over environmental needs, as the result of poor environmental water governance. When governance fails to address the entrenched interests and legitimate expectations of water allocation from stakeholders, other limitations represent only incidental hurdles for implementing e-flows. For instance, the need for education, stakeholder engagement, or better data and science has been recognized in the literature (Conallin, Campbell, et al., 2018a; Harwood et al., 2018; Mezger et al., 2019; Owusu et al., 2022; Tasca et al., 2020). Technological fixes alone, however, are unlikely to overcome historical and structural impediments to cooperation. Implementation in the absence of cooperation, therefore, is likely to be limited to those portions of an e-flow regime that do not conflict with other purposes, and thus reduced to minimum flow targets. Consequently, the smallest amount of water that can maintain a wetted channel is allocated for the environment, limiting functional attributes and processes that enhance biogeochemical cycling, structural habitat formation, and ecosystem maintenance.

Many other secondary challenges, generally co-occurring in causality, limit the implementation of more nuanced and sophisticated e-flows. The implementation of e-flows might require regulatory and even institutional changes that allow for their pre-existence, stakeholder engagement at all levels, setting achievable flow designs based on the available data, resources, and human-driven limiting factors (e.g., financial and infrastructural constraints, other demands) to guide decision-making. Implementation also requires timely, reliable, and available data, not only on the historical and current environmental states of river systems but also on potential future scenarios. These observations are even more pressing as hydroclimatic non-stationarity requires the acknowledgment that climate change impacts are imposing limitations on the natural system’s ability to provide water for allocation systems (Milly et al., 2008a).

Ecologically functioning systems require the presence of functional flow components. However, as the most common controlling factor of e-flow implementation is regulation (Owusu et al., 2021), re-operating reservoirs to accommodate more sustainable e-flow strategies requires flexibility and policy changes. Although water resources management is not reduced to computer-based models, water-energy modeling is an important part of decision-making in these processes when hydropower projects are present. Simulation models have been widely adopted for evaluating the impacts of changes in supply and demand under different scenarios when planning and managing water resources systems. To achieve the goal of sustainable water management, the investigation of water resources system design and management policies establishes the foundation for assessing system performance. Modeling tools can represent important interactions among the various control structures (i.e., different designs and policies) and users of a water resource system, and therefore, can help inform planners and managers on the trade-offs when allocating water resources (Loucks & van Beek, 2017). Assessment of management actions and frameworks is needed to find sustainable compromises between the different values, or to at least allow trade-offs to be explicit (Conallin, Wilson, et al., 2018). Although different levels of impacts on human uses are expected for different locations, they may not be greatly affected by the environmental water allocation (Owusu et al., 2021; Willis et al., 2022). Therefore, modeling studies constitute a powerful tool to balance these often-conflicting interests towards sustainable solutions to environmental water management problems.

The growing demands of human water uses associated with the uncertain water supplies caused by global changes, most notably climate change, will likely affect current management and operations and further stress already impaired and threatened riverine ecosystems and processes (Overton et al., 2014b). Consequently, planning and management decision-making processes increasingly need science-based approaches to guide e-flow implementation. Further studies involving facility-specific to system re-operation can better inform decision-makers and managers to maintain the health of freshwater ecosystems, building long-term resilience rather than short-term survival. Future research on e-flows should assess water system performance of conflicting objectives at different scales, to identify efficient trade-offs by using combinations of different scenarios of system designs, operating policies, and stressors. Some challenges for the future include the encouragement of stakeholder participation and awareness about the human impacts on the environment as well as the importance and benefits of e-flows. In addition, a greater focus on habitat formation by considering improvements on floodplain reconnection, sediment transport, geomorphology, temperature gradients, and restoration of riparian ecosystems, for example, can lead to greater environmental benefits than improvements in hydrologic conditions alone.

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## Chapter 4. Stress testing California's hydroclimatic whiplash: Potential challenges, trade-offs and adaptations in water management and hydropower generation[b]

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### Abstract

Inter-annual precipitation in California is highly variable, and future projections indicate an increase in the intensity and frequency of hydroclimatic 'whiplash'. Understanding the implications of these shocks on California's water system and its degree of resiliency is critical from a planning perspective. Therefore, we quantify the resilience of reservoir services provided by water and hydropower systems in four basins in the western Sierra Nevada. Using downscaled runoff from 10 climate model outputs, we generated 200 synthetic hydrologic whiplash sequences of alternating dry and wet years to represent a wide range of extremes and transitional conditions used as inputs to a water system simulation model. Sequences were derived from upper (wet) and lower (dry) quintiles of future streamflow projections (2030-2060). Results show that carryover storage was negatively affected in all basins, particularly in those with lower storage capacity. All basins experienced negative impacts on hydropower generation, with losses ranging from 5% to nearly 90%. Reservoir sizes and inflexible operating rules are a particular challenge for flood control, as in extremely wet years spillage averaged nearly the annual basins' total discharge. The reliability of environmental flows and agricultural deliveries varied depending on the basin, intensity, and duration of whiplash sequences. Overall, wet years temporarily rebound negative drought effects, and greater storage

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capacity results in higher reliability and resiliency, and lesser volatility in services. We highlight potential policy changes to improve flexibility, increase resilience, and better equip managers to face challenges posed by whiplash while meeting human and environmental needs.

## 4.1. Introduction

In California, a mix of high-elevation hydropower reservoirs in the Sierra Nevada and large multi-purpose water storage reservoirs throughout the state provide multiple benefits (Tarroja et al., 2016b), such as 10-20% of the state's electricity supply (California Energy Commission, 2022). The design of these systems, from infrastructure to operations, includes assumptions about reasonable circumstances in the lifespan of the system. For instance, flood control design criteria for dams include operational rules that detail the volume of storage necessary to absorb a large, predictable flood event from probabilities in the observed historical record (Şen, 1980).

Rapidly changing hydroclimatic conditions are challenging, especially when managing reservoirs for meeting competing demands (Wyrwoll & Grafton, 2022). Therefore, the historical record alone is unlikely to provide sufficient information on reliability (i.e., probability that a system will perform its intended function), especially if a reservoir operates with carryover storage (i.e., if it provides storage to meet required withdrawals over a period of several dry years) (Nagy et al., 2013). Beyond this limitation, historical data are insufficient for the design, planning and operation of water resources systems, due to incomplete records, short duration and/or sparse spatial distribution (Fagherazzi et al., 2007).

### 4.1.1. California's Whiplash

Recent studies have pointed to hydroclimatic 'whiplash' (Swain et al., 2018) or increasing volatility in interannual precipitation, especially for the Mediterranean climate regime found in California (He & Gautam, 2016). Swain et al. (2018) and Persad et al. (2020) characterized whiplash events as rainy seasons with precipitation totals below the 20th percentile (droughts) followed by events exceeding the 80$^{th}$ percentile (floods), or vice versa (D. Chen et al., 2022). These extreme transitions can happen, for example, when prolonged droughts punctuated by pronounced wet periods (Dettinger, 2013), and are also expected to alternate more rapidly with climate change (Swain et al., 2018). Null & Viers (2013) and He et al. (2021) also identified the skewed bimodal distribution towards wet and critically dry water years in climate projections for California.

According to Swain et al. (2018), future multi-year droughts in California tend to be interrupted by very wet interludes. Consequently, managing seasonal and multi-year whiplash events (e.g. a multi-year drought followed by an extreme wet year) will increase the complexity of water resources management (D. Chen et al., 2022). Extreme events can potentially have deleterious impacts, such as disruption to urban and agricultural water supply, and compromised flood control and groundwater recharge (AghaKouchak et al., 2014; Casson et al., 2019; Liu et al., 2022). Extremely wet and dry years, as well as their sequencing in time, can further stress water management and infrastructure (Persad et al., 2020) and exacerbate the challenge of sustainable water management under a changing climate (Scanlon et al., 2023). Moreover, different arrangements of water years result in different end-of-year storage (Johnson et al. 1995) highlighting the need to assess supply-demand dynamics for a variety of sequences under a changing climate.

While the impacts of extreme and prolonged droughts are far-reaching, the cascading impacts on the energy system are just now being understood (Jääskeläinen et al., 2018; Naumann et al., 2015; Van Vliet et al., 2016; Y. Wang et al., 2020). Meeting competing demands for limited water resources can be a particular challenge, especially in California, where significant agricultural, environmental, and urban demands exist (Stewart et al., 2020; Willis et al., 2022). As McNally et al. (2009) point out, managing for system stability  is not the same as designing or managing water infrastructure for system resilience. Managing these whiplash effects requires flexible and adaptive reservoir operations to optimize storage levels based on changing hydrological conditions, to capture this excess water for future use during dry periods. System resilience is the inherent ability for a system to return to a normal operating level after perturbation (Boltz et al., 2019; Meadows, 2008). Thus, considering the resilience of water systems (i.e., probability of rapidly and effectively recovering after disruptions) is important as they are usually designed with an assumption of a stationary hydroclimate, which is problematic given anthropogenic climate warming and non-stationary behavior (Milly et al., 2008a), such as consecutive multi-year droughts (Lund, Asce, et al., 2018).

To date, the impacts of hydroclimatic whiplash on water allocation and hydropower generation are poorly known. Understanding these implications on California's water system and its degree of resiliency to these shocks is critical from a planning perspective. In this paper, we explore the capacity of the existing water management infrastructure system to rebound to an acceptable system state after against perturbations. Given the potential increase in whiplash sequences, we examine the relative impact of whiplash on hydropower, flood control and water allocation systems in four basins in the western Sierra Nevada in California to identify how reliably services can still be provided after consecutive whiplash events. We also seek to resolve at what point these systems become less reliable and resilient to droughts and floods. In addressing these questions, we both examine historical data and impose a stress test modeling approach (Viers & Nover, 2018), whereby we perturb regional water system models with synthetic hydrologic sequences that are presumed to represent a range of plausible future multi-year whiplash conditions. We build on this growing body of research into disruptive and abrupt shifts in hydrological regimes and their consequences on water and hydropower systems.

## 4.2. Methods

### 4.2.1. Study Area

In this study, we consider the four major basins in the Central Sierra Nevada, California, which collectively contribute the most to the flow into the San Joaquin River (SJR), one of the two main rivers in the state that form the Sacramento-San Joaquin River Delta. The Stanislaus (STN), Tuolumne (TUO), Merced (MER) and Upper San Joaquin (USJ) basins form a complex system of highly regulated river systems, with high-elevation reservoirs designed for hydropower generation, and low-elevation major multi-purpose storage reservoirs. This region has a Mediterranean-montane climate, with a notably variable hydrology (**Figure 4-1**), that has accounted for approximately 25% of California's hydroelectricity. The facilities are operated by several distinct utility companies and government agencies (**Table 4-1**) and therefore, are in many cases regulated differently by each owner/operator. The projects' purposes range from mostly hydropower generation-driven management (in USJ and STN) to water supply deliveries (in MER and TUO), with varying degrees of storage capacity (Maskey et al., 2022).

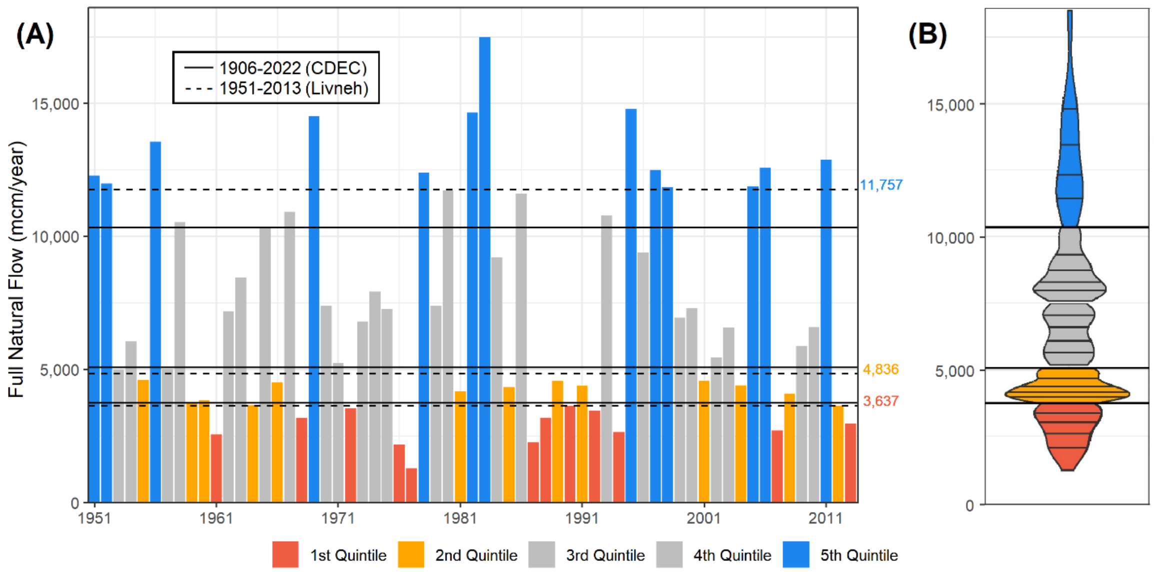


**Figure 4-1.** A) Time series of total full natural flow (historical simulated streamflow) from the four basins, where sequences of critically dry, dry and wet years can be observed (below the 20$^{th}$ and 40$^{th}$, and above the 80$^{th}$ percentile, in red, orange and blue, respectively) derived from the Livneh dataset (daily data available for 1951-2013). B) Violin plot containing the full natural flow estimates from the California Data Exchange Center (CDEC) (monthly data available for 1906-2022 water years). The dashed lines in A, and full lines in A and B show the upper and lower quintiles from the Livneh and CDEC datasets, respectively.

**Table 4-1.** Main characteristics of each basin. Adapted from: Rheinheimer et al. (2022).

| **Basin** | **Stanislaus** | **Tuolumne** | **Merced** | **Upper San Joaquin** |
|---|---|---|---|---|
| **Primary objectives** | Hydropower and agricultural deliveries | Urban/agricultural deliveries | Agricultural deliveries | Hydropower and agricultural deliveries |
| **Area (km$^2$)** | 3,100 | 4,851 | 3,288 | 4,245 |
| **Maximum elevation (m)** | 3,373 | 3,749 | 3,759 | 3,925 |
| **Terminal reservoir (Dam)** | New Melones Lake (New Melones Dam) | Lake Don Pedro (New Don Pedro Dam) | Lake McClure (New Exchequer Dam) | Millerton Lake (Friant Dam) |
| **Average historical water yield (mcm) (*ac-ft*)** | 1,439.2 (*1,166,778*) | 2370 (*1,921,390*) | 1225.6 (*993,610*) | 2215.4 (*1,796,054*) |
| **Reservoirs (n)** | 10 | 4 | 2 | 10 |
| **Total reservoir storage capacity (mcm) (*ac-ft*)** | 3,469.1 (*2,812,445*) | 3,317.8 (*2,689,784*) | 1,306.5 (*1,059,197*) | 1,409.4 (*1,142,619*) |
| **Average historical diversions (%)** | 45% | 81% | 59% | 77% |
| **Major agricultural and/or urban deliveries** | Oakdale and South San Joaquin Irrigation Districts | Turlock, and Modesto Irrigation Districts and SFPUC | Merced Irrigation District | Central Valley Project (CVP)* |
| **Utilities (Number of powerhouses)** | Utica Power Authority (2), Tri-Dam Project & Tri-Dam Power Authority (4), Northern California Power Agency (2), Pacific Gas & Electric Company (3) and US Bureau of Reclamation (1) | SFPUC (3) and Turlock Irrigation District (1) | Merced Irrigation District (3) | Southern California Edison (9), Pacific Gas & Electric (6), US Bureau of Reclamation and Friant Power Authority (1) |
| **Generation capacity (MW)** | 803 | 586 | 107 | 1,222 |

*Federal power and water management project that serves many districts, cities and thousands of family farms.

### 4.2.2. Water system model

The study area is represented in the *CenSierraPywr* modeling framework (**Figure 4-2.**), a daily time step water system simulation model implemented with *Pywr* (Tomlinson et al., 2020) in Python. *CenSierraPywr* is composed of four independent models created for each of the major basins in the SJR system (Rheinheimer et al., 2022). *Pywr* uses linear programming to allocate flows within a network of links (e.g., river, canal) and nodes (e.g., reservoirs, instream flow requirements points), given a system's physical and operational constraints and relative water value, written in Python or described in other formats, described in **Table 4-2**. Water allocation depends on pre-defined rules or numerical input, such as flow schedules or channel capacity. Allocation is then determined by the relative water value given for each node/link, affecting the model decision based on the cost of moving/storing water, with a goal of minimizing costs in each time step (Tomlinson et al., 2020).

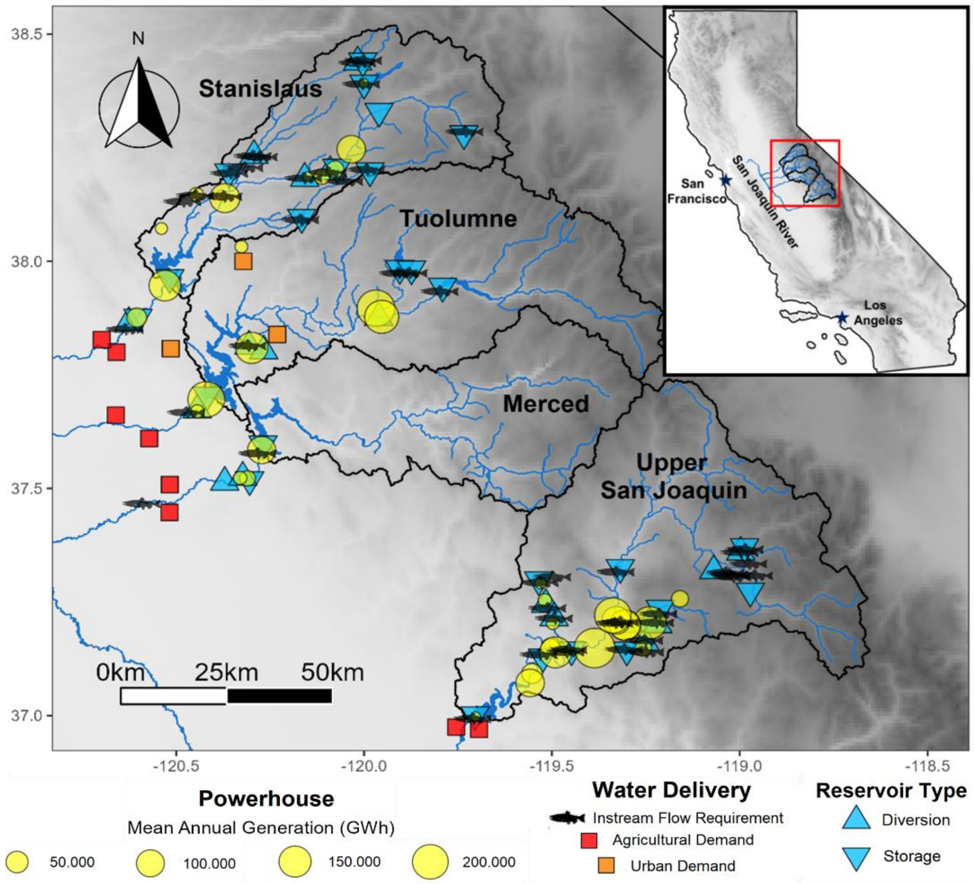


**Figure 4-2.** Map showing the network of main nodes incorporated into the *CenSierraPywr* modeling framework.

**Table 4-2.** Model parameters and parameter sources incorporated into CenSierraPywr.

| Data | Model Parameters | Parameter Sources |
|---|---|---|
| **Infrastructure** | Storage/turbine capacity, storage elevation curve, powerhouse flow and head, aqueduct/canal conveyance capacity, dead pool storage, among others | FERC Licenses, Environmental Impact Assessments, State Water Resources Control Board (SWRCB) Manual, & other manuals (e.g., SJR Restoration Flows Guidelines) |
| **Reservoir operations** | Flood control pool and releases, recreational flows, water supply (downstream demands) | FERC Licenses, US Army Corps of Engineers documents |
| **Instream Flow Requirements** | Minimum flow schedules, maximum flows and ramping rates from storage and diversion dams | FERC Licenses, SWRCB order, Water Quality Control Plan |
| **Urban and agricultural water demands** | Daily demands based on historical urban and agricultural deliveries | San Francisco Public Utilities Commission, Irrigation Districts, CVP contracts |
| **Relative water value** | Positive costs for penalties (e.g., spills) and negative costs for priority allocation (e.g., instream flows) | Established by trial-and-error to have model decisions on water allocation closer to real historical operations |

*CenSierraPywr* can also be coupled with other models (e.g., hydrologic and energy models), and optionally allow the inclusion of energy price-based optimization for hydropower allocations; this option was adopted for the hydropower-driven basins to better represent hydropeaking facilities. The hydropower optimization component is composed of an 8-month, monthly time step, planning-scale optimization model with imperfect foresight of hydrology and perfect foresight of energy prices that schedules discretionary releases based on the forecasts. These hydroeconomic decisions are assumed to occur at the hourly time step across days and months and are incorporated into the model by piecewise linear price curves, transformed to relative costs within the *Pywr* model. Energy pricing was derived from wholesale energy prices from the independent energy model Holistic Grid Resources Integration and Deployment (HiGRID), developed by Tarroja et al. (2016, 2019). Price duration-curves from HiGRID were linearized by iteratively dividing prices into equally divided successive blocks, such that the sum of each level of division resulted in minimal differences of the actual prices and piecewise-averaged prices between each of the two respective blocks. Five blocks were used in the piecewise linearization; the piecewise linear price curves are then used by the *CenSierraPywr* routine. Currently, 2009 prices are adopted as they generally reflect approximate modern and stable energy demand (see Rheinheimer et al., 2022). Meanwhile, non-optimized system components are simulated following existing operational objectives; in all cases hydropower generation is a secondary benefit.

The main hydrological inputs to the model are streamflow data, constructed by routing runoff and baseflow outputs from the Variable Infiltration Capacity (VIC; Liang et al., 1996) model. Gridded runoff data at a 1/16° (~6 km) resolution were used for the simulated historical water years 1951-2010, developed by Livneh et al. (2015). We further downscaled the dataset from the VIC cells to the subbasin level using the ‘extract’ function from the *raster* R package (Hijmans, 2020) using a normalized area-weighted approach. Then, we bias-corrected the streamflow data at the basin level using historical monthly unimpaired flow estimates from the California Data Exchange Center (CDEC) developed by the Department of Water Resources (DWR) (CDWR, 2016), through the *hyfo* R package (Xu, 2020). The historical data was further bias-corrected using US Geological Survey (USGS) data for specific gauges that had at least 15 years of data, mostly in the upper subbasins, where most of the precipitation occurs. The Livneh dataset was then used to calibrate model inputs from **Table 4-2**, and outputs, when needed (e.g., when powerhouse or canal flow capacity stated in licensing documents differs from observed flow gauges). The models were calibrated to reflect more recent, real-world observed reservoir operations (Rheinheimer et al., 2022). Therefore, hydrological and power generation historical data (1980-2011) gathered from the USGS and the Energy Administration Information were also used to calibrate water allocation and hydropower generation. For more details on model inputs, parameters and assumptions, see Rheinheimer et al. (2022).

Among the limitations in this study, we recognize the factors that affect hydropower planning particularly important in California, but are not included, such as net energy demand, the value of ancillary services, infrastructure damage (e.g., the Creek fire in 2020 destroyed hydropower stations in the USJ), land use/cover change (e.g., impacted runoff in subsequent years in the Big Creek region) and the penetration of other renewable sources in the grid and associated energy prices. Other limitations include the use of simulated hydrology as model inputs (model uncertainty), and the adoption of historical agricultural deliveries and reservoir operations (parameter uncertainty) and 2009 electricity prices (data uncertainty) for the historical baseline and whiplash scenarios.

#### 4.2.3. Hydrologic sequences

While droughts are slower to develop than extreme rain events caused by atmospheric rivers, this climate variability makes balancing of flood protection and water supply storage in local, State and Federal projects challenging (US Bureau of Reclamation, 2021). As stated by De Luca et al. (2020), floods and droughts are expected to become more frequent and severe problems, underscoring the importance of research on concurrent wet and dry hydrological extremes. Therefore, system analyses considering the stochasticity of streamflow processes should preferably become part of the hydrologic studies for all reservoirs that utilize natural inflows (Nagy et al., 2013). Johnson et al. (1995) indicated the need to use a range of historical and synthetic hydrologic sequences to note the sensitivity of systems, such as for water supply-demand comparisons. Generating hydrologic sequences, each of a specified desired length, allows the creation of a much broader base for hydrologic design; for that, it is usually considered that 10-20 sequences would be adequate and that their length should generally correspond to the period of expected project amortization (USACE, 1993).

##### 4.2.3.1. Data Preparation

Francis et al. (2022) notes that temporal hydrological changes in the future under representative concentration pathway (RCP) 8.5 forcing (a “business-as-usual” greenhouse gas emissions scenario) are more robust than recent historical data. Thus, here we use wet and dry years derived from future climate projections forced by RCP 8.5. In order to do that, we conducted an analysis of dry and wet year frequency within the projections of 10 Global Circulation Models (GCMs) for the mid-21$^{st}$ century conditions (2030-2060) (**Figure S4.1-1**). These GCMs have been identified by the California Department of Water Resources (Lynn et al., 2015) and California’s 4$^{th}$ Climate Change Assessment (Herman et al., 2018) as the best representative of the regional hydrology.

These datasets were produced by Pierce et al. (2016) and downscaled to a resolution of 1/16° (~6 km) using the Localized Constructed Analogues (LOCA) statistical method (Pierce et al., 2018). Similar to the Livneh dataset, GCMs’ streamflow data were originally constructed by routing runoff and baseflow outputs from LOCA VIC runs and were also bias-corrected and downscaled to the subbasin level.

To address the inherent uncertainty of hydrological processes, we constructed a wide range of dry-wet-dry sequences stochastically. For that, we used the splice method, generally used to alter the sequencing, duration, or frequency of events (Albano et al., 2021), to combine water years sampled from the GCMs projections and create multi-year whiplash events.

##### 4.2.3.2. Creation of Whiplash Sequences

Similar to Swain et al. (2018) and Persad et al. (2020), we synthesized the different sequences by sampling water years with total aggregated regional simulated hydrologic runoff below the 20$^{th}$ and 40$^{th}$, and above the 80$^{th}$ percentiles. These quintiles were selected to approximate critically dry, dry and wet water year conditions in the Central Sierra Nevada, respectively. Water years were derived by sampling from these upper and lower quintiles, with replacement, across 3 decades of future streamflow projections from the 10 GCMs as a uniformly random sample pool (**Figure S4.1-1**).

To determine the desired lengths of sequences, we considered the historical occurrence (1906-2023) of wet-to-dry and dry-to-wet transitions, as seen on **Figure S4.1-2**. The probability of occurrence of a wet (> $80^{th}$ percentile) and dry/critically dry (<$40^{th}$ percentile) years are 20.3% and 39.8%, respectively, in any given year. The CDEC data shows no serial correlation or memory (**Table S4.1-1**), therefore, the probability of occurrence of 2, 3, 4 and 5 successive dry years, and 2 successive wet years are 15.9%, 6.3%, 2.5% and 1%, and 4.1%, respectively. Consequently, these were adopted as plausible whiplash conditions in this study.

Each multi-year whiplash consisted of 2 dry spells of the same length interspersed with either 1 or 2 wet years, to represent dry-to-wet and wet-to-dry transitions. More specifically twenty-five (25) random realizations were generated for 8 different multi-year whiplash lengths, wherein sequences consisted of 2 to 5 dry (D2 to D5) years, followed by 1 or 2 wet (W1 to W2) years, followed again by 2 to 5 dry years, for a total of 200 multi-year whiplash sequences. GCM predictions show increased variance, lower base flows and weaker snowmelt signal due to more precipitation in form of rain occurring earlier in the year, as seen in the flow duration curves, daily and monthly hydrographs (**Figure S4.1-3**, **Figure S4.1-4** and **Figure S4.1-5**) for the historical and sampled future years. The hydrologic sequences used in this study are shown in **Figure 4-3.** . Initial reservoir levels at the beginning of the simulation (October 1) are assumed to be mostly at 50% of storage capacity (**Table S4.1-2**), as during that period of the year the halfway mark is near or within one standard deviation from the state most reservoirs tend to hover around historically. Model results from this stress test on these systems are discussed below.

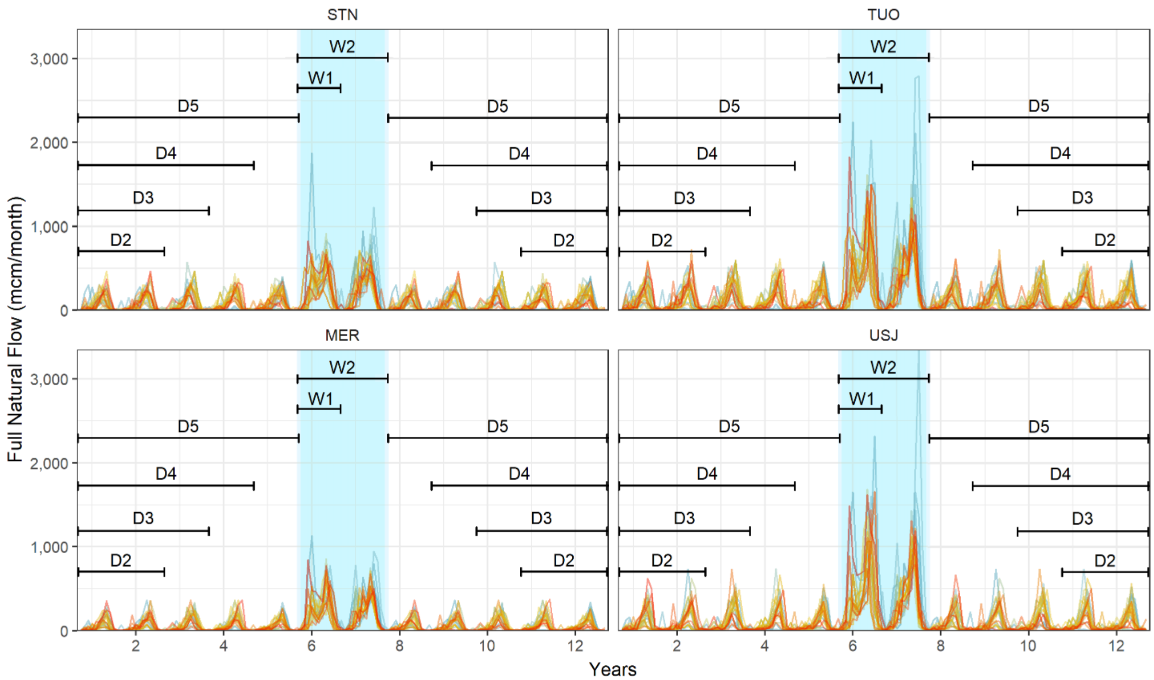


**Figure 4-3.** Timeseries showing the different combinations of synthetic hydrological sequences used in this study, formed by 25 combinations of two periods of 2-5 dry years (D2 to D5) interspaced by either 1 or 2 wet years (W1 or W2) that result in 5-to-12 year-long sequences for the Stanislaus (STN), Tuolumne (TUO), Merced (MER) and Upper San Joaquin (USJ) basins.

### 4.2.4. Analyses

We explore the trade-offs among the different services provided by reservoirs to help reduce potential impacts of hydroclimatic whiplash on these regional water and energy systems. Trade-offs between hydropower generation and storage were quantified as monthly and annual outcomes for each sequence. Flood control releases were quantified as the amount of water released downstream into the SJR either as controlled spill to maintain the flood control pool of the terminal reservoirs or eventual uncontrolled spill. Results were summarized by using the daily modeled output to calculate the total monthly and annual storage, flood control releases and hydropower generation. Hydropower and storage were then compared to historical averages to assess changes and the ability of these systems to rebound back to a baseline system state. In addition, as there are two major agricultural deliveries and numerous river reaches with environmental flow requirements in each basin, these were summarized as the total basin-wide water delivered to irrigation districts and the environment. Environmental and agricultural water deliveries were assessed through the subsequent indices as they generally have target demands that mostly vary according to water year types.

When the occurrence of a failure and subsequent success are probabilistically independent events, reliability is considered a measure of resiliency (Hashimoto et al., 1982). Reliability indices generally followed in water resources planning and management are the time and volumetric reliabilities, as defined by Jain & Bhunya (2008) and Nagy et al. (2013) as:

*Time reliability* ($R_t$), also known as duration-based reliability, is the probability that the system state lies in the set of satisfactory states in which demands (D) are fully satisfied by water deliveries (Q):

$$R_t = \mathrm{P}[x(t) \in S] \tag{2}$$

where P(·) is the probability, $x(t)$ is the system's state in the given time period $t$, and $S$ is a set of satisfactory states. It is given as the sum of all non-failure periods $\Delta t$, as a fraction of the total operation period $T$ – and can be estimated as:

$$R_t = \frac{1}{T}\sum_{Q=D} \Delta t \tag{3}$$

*Volumetric reliability* ($R_v$) is the actual delivered portion of the total volume of demand during a period $T$. The volume or quantity-based reliability is expressed as:

$$R_v = 1 - \frac{\int_{Q<D}(D-Q)dt}{\int_0^T DdT} = 1 - \frac{\sum \Delta V}{TD} \tag{4}$$

where, $\Delta V$ is the quantity of short-fall within a period $T$ for which the supply $Q$ is curtailed below the constant draw-off rate $D$. In the case of constant (and continuous in time) target, the following relationship holds true: $R_t \leq R_v$. $R_t$ and $R_v$ were calculated on an annual basis. In addition, we determine the *recovery rate* (*Rr*) and *resiliency index* (*Rs*) as described by Hashimoto et al. (1982), to assess how quickly and how probable a recovery from failure is, once failure has occurred. These are defined as:

$$R_r = \frac{1 - \lim\limits_{n\to\infty}\left(\frac{1}{n}\right)^n \sum_{t=1}^{n} Zt}{\lim\limits_{n\to\infty}\left(\frac{1}{n}\right)^n \sum_{t=1}^{n} Wt} = \frac{1-\alpha}{\rho} \tag{5}$$

$$R_s = \frac{\rho}{1-\alpha} \tag{6}$$

where, $F$ is a set of unsatisfactory or failure states, $Zt$ indicates a satisfactory system state, and $Wt$ a transition from a satisfactory to an unsatisfactory state, in an $n$-period. Therefore, $\alpha$ is the probability that a system is in a satisfactory state and $\rho$ is the probability of the system being in the set $S$ in some period $t$ and going to the set $F$. Thus, recovery rates were obtained for each time a recovery from failure in meeting demands happened, and the resilience index was calculated per sequence, as failures propagated throughout water years.

### 4.3. Results

### 4.3.1. Impacts on Reservoir Storage and Flood Control

The total storage in all basins is negatively affected in most sequences (average results for the longest sequences are shown in **Figure 4-4**, average and full range of results for all sequences are shown in **Figure S4.1-6** and **Figure S4.1-7**, respectively). All supplemental figures and tables are available in **Appendix 4.1: Supporting information for Chapter 4**. Fluctuations are particularly notable in the MER which has no upper watershed reservoirs, where storage goes from approximately +40% in wet years to below -60% in the last year of prolonged droughts, compared to historical averages. Very wet years could compensate for the dry years overall, making these systems snap back closer to or above an average system state. Refilled reservoirs provided enough water to keep storage above average for an extra year in the TUO and MER basins. Even though the STN and TUO have greater storage capacity, their average maximum increase averaged approximately +20% after two subsequent wet years, likely due to spillage as show in the total flood control releases from the terminal reservoirs into the SJR in **Figure 4-5.** and **Figure S4.1-8.**, and **Table S4.1-3**. In addition, the TUO shows a more stable response to drought, with losses in storage averaging around -30% in most dry years. And the STN generally needed two wet years to reach above average storage values in the second wet year, holding enough water to still be above the historical mean in the following dry water year. On average, even though storage in the USJ benefited from the wet years, the gains are already lost in the subsequent dry water year. The USJ also shows the smallest average relative losses in storage during droughts (approximately 15-30% for all dry years).

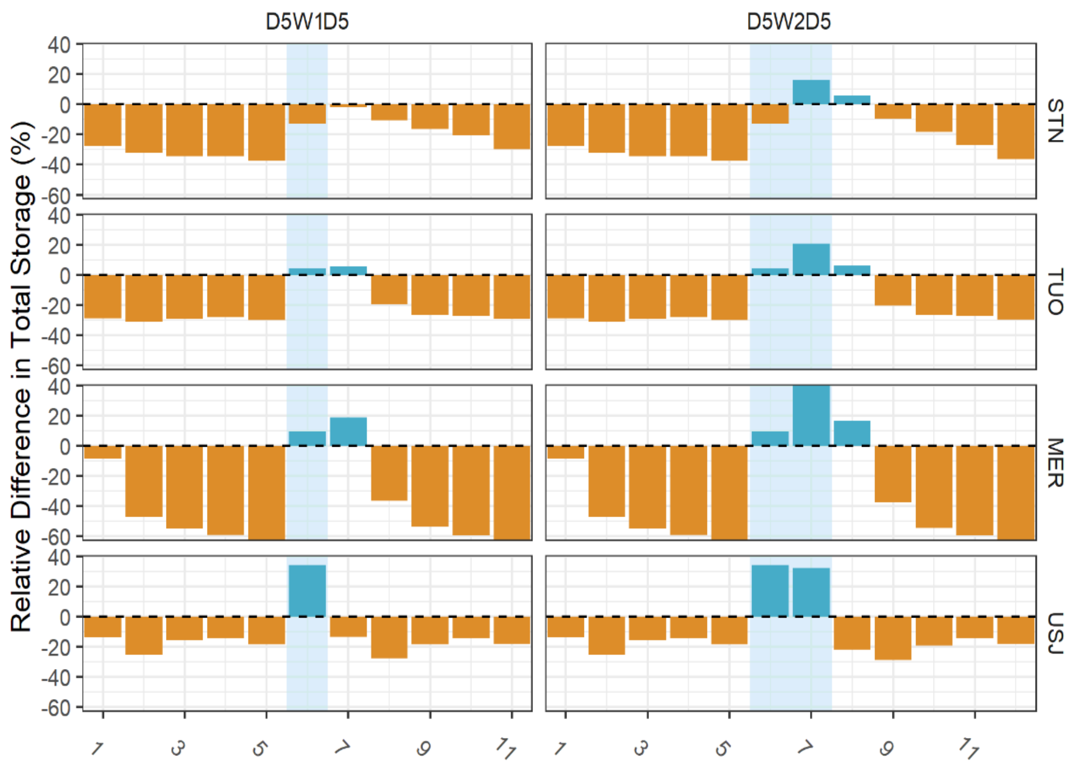


**Figure 4-4.** Average annual relative change in storage compared to the historical mean (1951-2010 water years) during the longest sequences of 5 dry-year long spells (D5)

interspaced by one or two wet years (W1 or W2) in the Stanislaus (STN), Tuolumne (TUO), Merced (MER) and Upper San Joaquin (USJ) basins. Wet years are shaded in light blue.

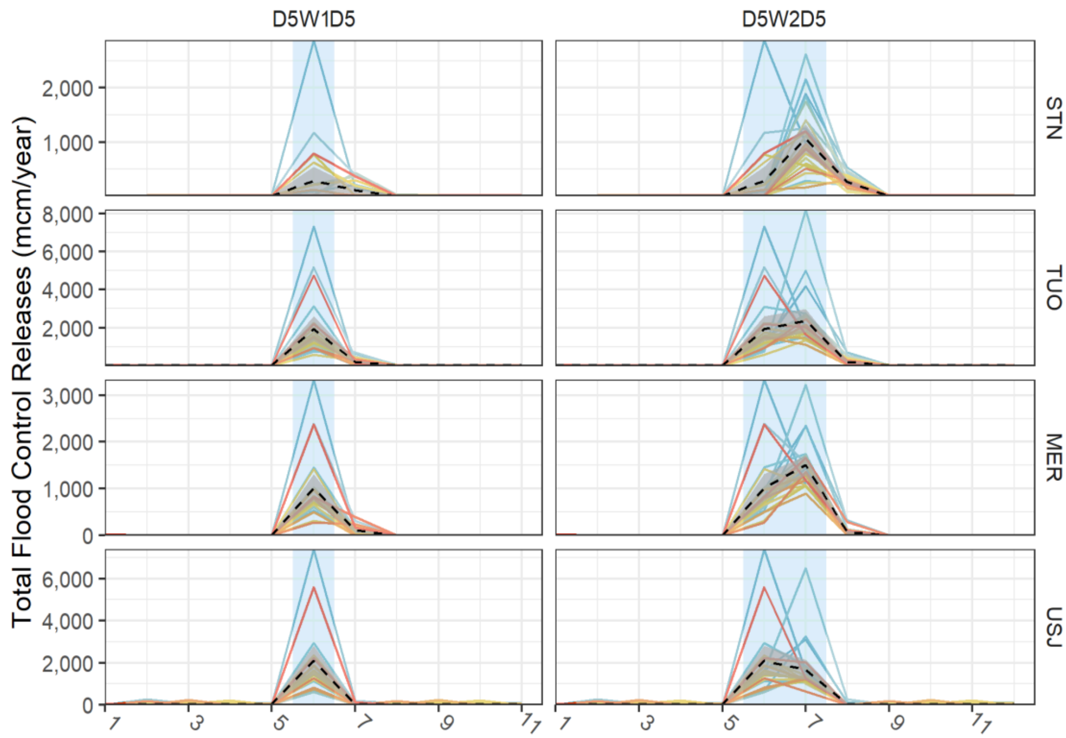


**Figure 4-5.** Annual flood control releases from the terminal dams during the longest sequences (wet years shaded in light blue). The black dotted lines with the gray background show the mean values with a 95% confidence interval. Note the different y axis.

**Figure S4.1-9** shows a comparison of monthly basin-wide storage values in the historical period to the ones in the first and second dry spells, and first and second wet years produced by the sequences. Both dry periods reduced storage in all basins, especially later in the water year. The impacts of droughts following the wet years are generally weakened in the STN and TUO due to their greater storage capacity, but also in the MER. The USJ tends to have a lower performance during droughts after wet years likely due to how water allocation happens in the basin, as discussed in the next sections. All basins also show greater storage at the beginning of the second wet water year as the reservoirs are already fuller, showing lower differences later in the year when compared to the previous (first) wet year.

Maximum flows are adopted specially below the terminal to avoid erosion, protect water quality and riparian habitat, and/or are limited to channel conveyance capacity to avoid flooding (FERC, 2003a). However, as seen in **Figure S4.1-10** and **Figure S4.1-11**, there are occurrences of flows above the maximum flow threshold of each basin in certain sequences, showing the importance of proper infrastructure design and maintenance, along with well-managed peak flow releases across multiple days. The basins with highest water yields (TUO and USJ) are more prone to flooding, despite their difference in storage capacity. According to **Table S4.1-4**, many times the flooding happens for no more than

1-5 days, though in certain occasions the maximum flow thresholds are surpassed for over a month. Considering the percentage of exceedance of such flows, the basins with greater storage capacity (STN and TUO) tend to have small exceedances (<1-6%), meanwhile in the MER and USJ, had median flows that exceed the established limits in 14-77%. In addition, in most cases the standard deviation of such flows is close to or over 100%, and all basins had high magnitude flows that were over 6 times greater than the maximum flows allowed, as a result of uncontrolled spillage.

### 4.3.2. Impacts on Hydropower Generation

The mean relative changes on hydropower generation compared to the historical mean for the longest sequences are shown in **Figure 4-6**. Average and full range of results for all sequences are found in

**Figure S4.1-12** and **Figure S4.1-13**, respectively. The northern basins (STN and TUO) basins have some buffer in the first dry year after wet years, with approximately half the generation losses compared to preceding and subsequent dry years, which generally go around 30% and 40%, respectively. The MER basin exhibits the widest variation in generation, with generation in wet years averaging around +35-70%, and losses in dry years around -40%. Similarly, the USJ basin shows the poorest performance during droughts, typically experiencing losses of around 40-50% or more on average for all dry years. Within year changes in generation can be seen in **Figure S4.1-14**. During the droughts, the TUO presents major losses in generation between the winter and summer, especially in Feb-Jul. Hydropower losses in the MER concentrate in the summer, when water is needed downstream to meet agricultural demands. In the STN and USJ, hydropower is generally lower year-round, with greater losses in late spring (May-Jun) in the STN. Meanwhile, in the USJ, losses during droughts as well as gains during the wet years are both concentrated in the winter and summer. Gains in generation during the wet rebound are mostly apparent in the winter and spring (Jan-May) in the STN, and in the winter of the second wet year in the TUO (Jan-Mar). In the MER, gains are most apparent later, between late winter and spring (Feb-Jun).

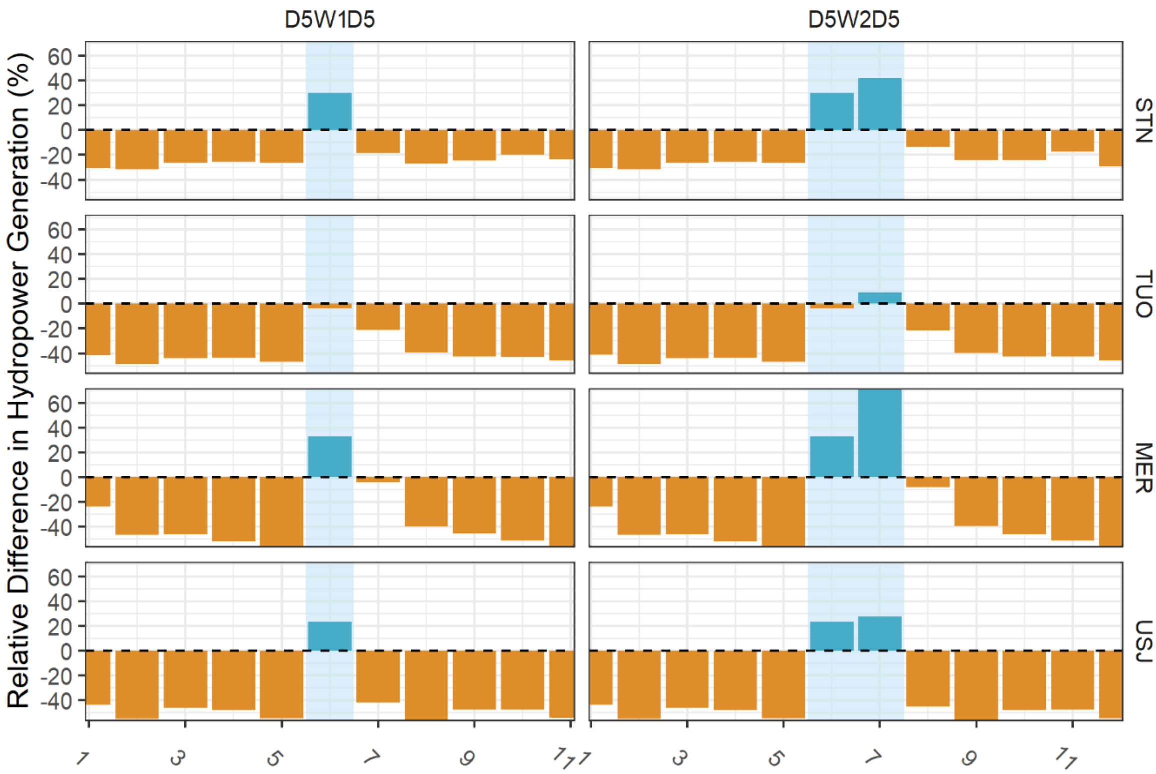


**Figure 4-6.** Average annual relative change in generation compared to the historical mean (1951-2010 water years) during the longest sequences (wet years shaded in light blue).

### 4.3.3. Impacts on Agricultural Water Deliveries

**Figure 4-8** shows the volumetric and time reliability of agricultural deliveries, respectively, in which we can see the substantial role of storage capacity among the basins for the longest sequences. The average and full range of responses for all sequences can be seen in **Figure S4.1-15** and **Figure S4.1-16**. Both indices show similar behaviors, however the time reliability tends to show a lower performance, especially in the USJ basin. This indicates that demands tend to be unmet more constantly throughout the years, but with not as significant cutoffs in volume so that part of the water supposed to be allocated to users is still delivered.

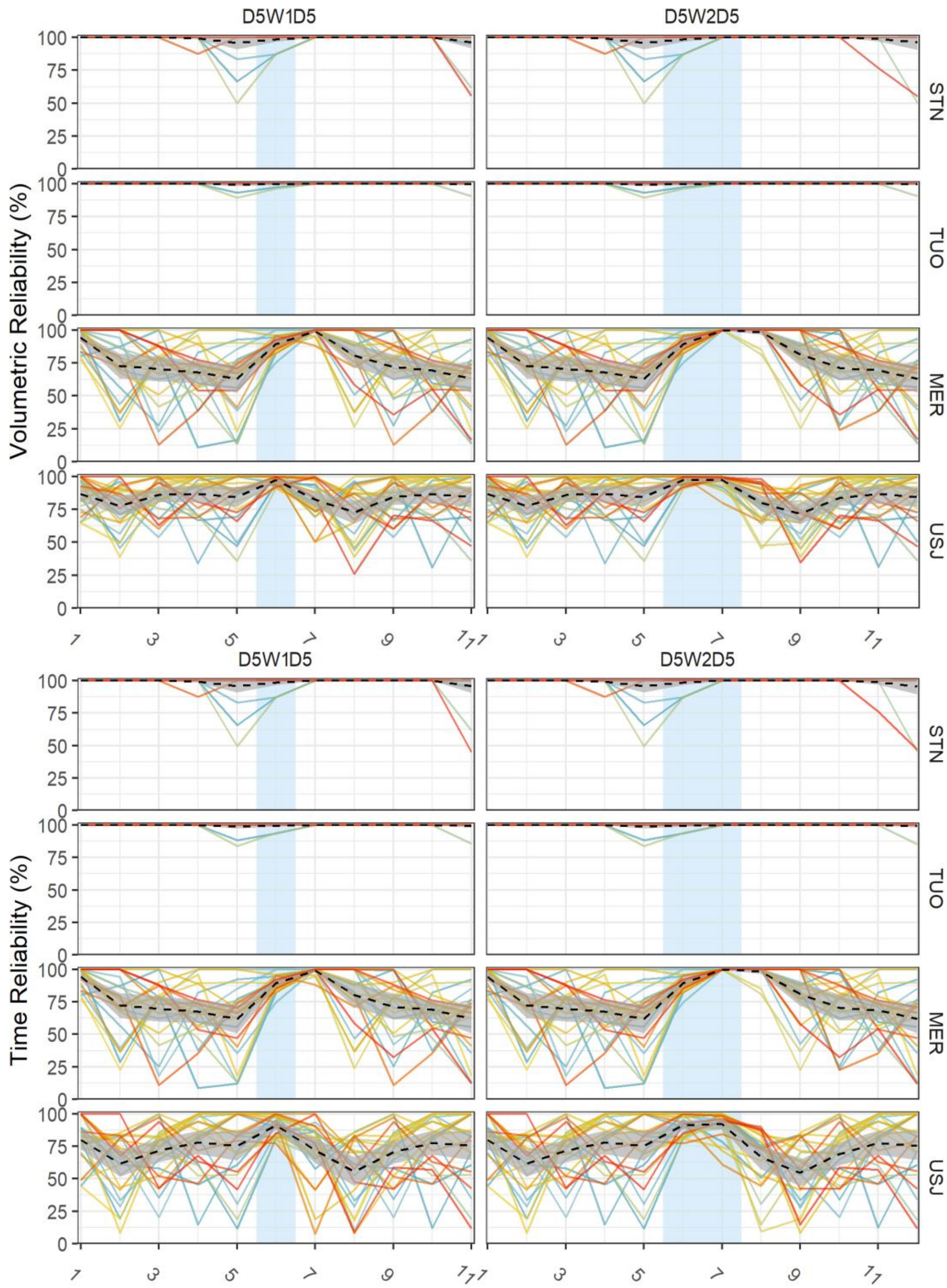


**Figure 4-7.** Annual volumetric and time reliability of agricultural deliveries during the longest sequences. The black dotted lines with the gray background show the mean values with a 95% confidence interval (wet years shaded in light blue).

The STN and TUO basins can withstand the dry periods, with few drier sequences impacting the reliability of this service in longer droughts. However, some extremely dry sequences can reduce the volumetric reliability down to about 50% during 5-year droughts. The USJ and MER basins show higher variability, in which sequences with extremely wet years can meet 100% of the target water allocation, while others reach only 25% or even less under longer droughts. Certain drought sequences can make deliveries unachievable for as much as 75-90% of the time in both basins. The MER tends to be capable of meeting demands most of the time during the introduction of 1-2 wet years, while keeping a better performance for one more year after the drought interruptions by wet years, in which deliveries still tend to be mostly met. However, as the dry spells last longer, the lower the overall reliability of agricultural deliveries gets; at the end of 5-year droughts (D5) only about 60% of demands were met on average, in both dry periods before and after the wet years. Meanwhile, the USJ basin is able to keep a better performance than the MER in longer droughts, meeting at least about 73% of demands on average, due to the significant reductions in the CVP deliveries in dry and critically dry water years. In the USJ, the sequences with 2 wet water years (W2) show an alleviation of the impact of droughts on the quantity of deliveries, especially in the driest sequences, as shown in the minimum volumetric reliabilities achieved after severe drought sequences. The sudden fall in performance in the USJ reliability indices right after the wet years can be explained by the SJVI calculation, as water year types are partially defined based on the previous' years index. Therefore, based on the considerably high previous year's calculated index, more water might be required to be delivered in the first dry year than the actual natural system's yield capacity to produce, or the infrastructural system's capacity to store.

In addition, agricultural deliveries hardly went from a satisfactory to an unsatisfactory state (*Wt* in equation 5) in the STN and TUO basins, while the USJ and MER had increasing occurrences of failures with longer droughts, in which some sequences were more impactful (**Figure S4.1-17**). **Figure S4.1-18** shows that agricultural deliveries in the STN and TUO have high resilience, with few exceptions in the 5-year long dry spells, when deliveries can be unmet for several consecutive days. The MER and USJ tend to have a very little resilience (generally below 7%), with recovery rates mostly under 50 days, however with few occurrences of interrupted deliveries happening for more than 200-300 days. Despite that, certain sequences in all scenarios can still allow these basins to achieve maximum volumetric and time reliability in multiple years.

### 4.3.4. Impacts on Environmental Flows

The reliability indices also show similar behaviors for e-flow deliveries, in which the time reliability also tends to show a lower performance, with the greater differences from the volumetric index occurring in the USJ. Additionally, time reliability (**Figure S4.1-8.** and **Figure S4.1-20**) shows a much greater range of responses among sequences, indicating a higher degree of discrepancies in the timing of deliveries while the total water quantity delivered is more consistent with flow schedules (**Figure S4.1-8.** and **Figure S4.1-19**).

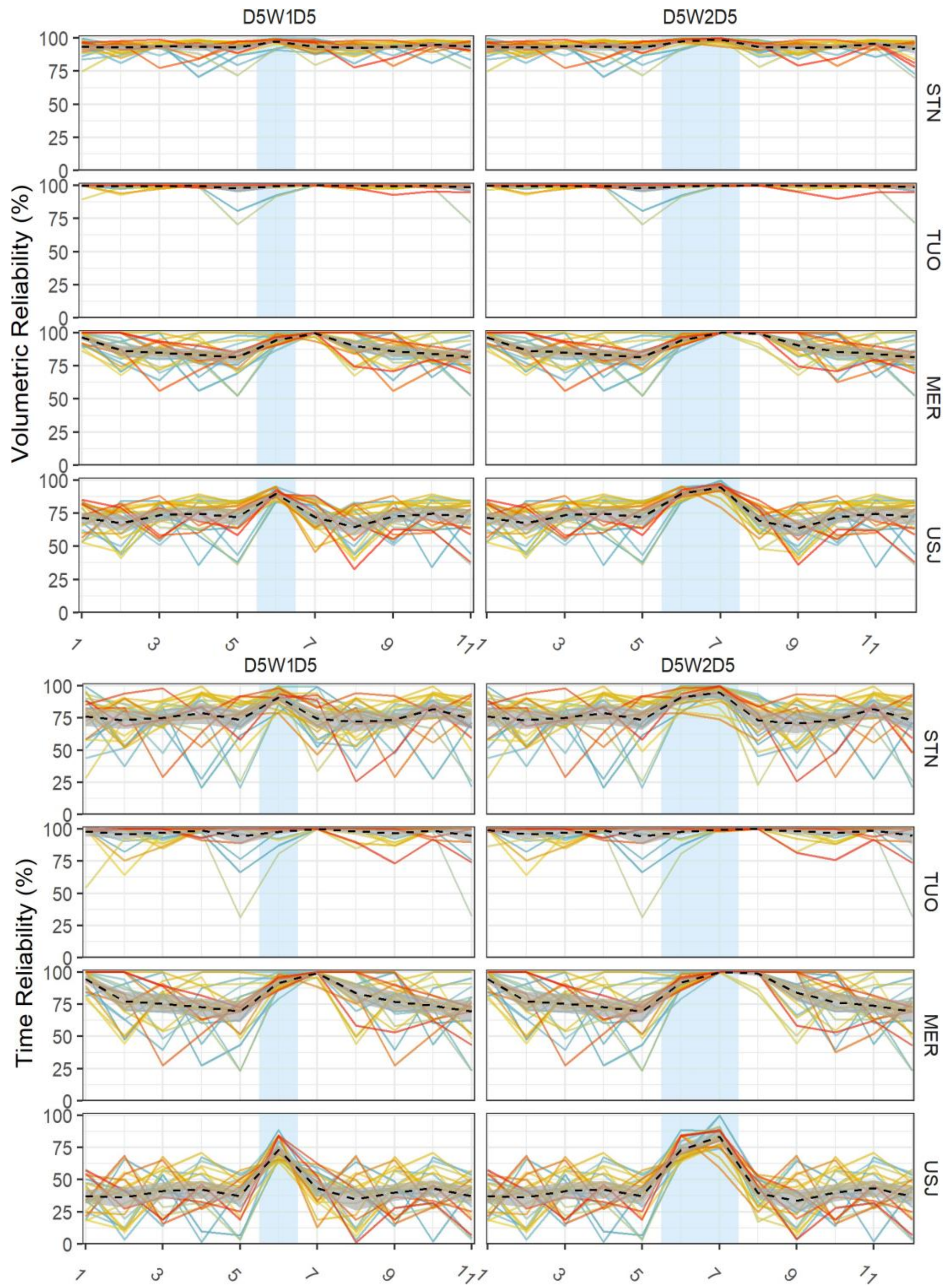


**Figure 4-8.** Annual volumetric and time reliability of environmental flows during the longest sequences. The black dotted lines with the gray background show the mean values with a 95% confidence interval (wet years shaded in light blue).

Similar to the MER, the STN and USJ have highly variable responses. The STN presents volumetric reliabilities generally around 90-100%, with some sequences reaching down to 73-76% especially during longer droughts. However, the time reliability averages around 75%, with responses varying between 22-100% in dry years, and mostly around 90-100% in all wet years. On the other hand, USJ rarely gets to maximum reliability in both indices even in wet years, with volumetric and time reliabilities ranging around 35-86% and 0-70% in dry years, respectively. Both basins also show a decline in both indices right after the wet years, most notably in time reliability, with a recovery in reliability after the second dry year. Like the CVP agricultural deliveries, this is likely due to the e-flow schedules based on the SJVI being partially calculated on the previous year's high inflows.

**Figure S4.1-21** shows the number of times e-flow deliveries went from a satisfactory to an unsatisfactory state (*Wt* in equation 5) in each sequence. In the STN and USJ the count of failures increases with longer droughts, showing that the length of droughts progressively impacts agricultural deliveries in these basins. Meanwhile, the TUO shows inconsistent responses among different lengths, being more affected by the severity of certain sequences instead, i.e., the intensity of the droughts is more important to the system performance; in turn, the MER presents an intermediate response. Once these failures in meeting e-flow requirements occur, e-flow deliveries tend to recover within a week in the TUO, meanwhile the other basins take mostly up to about three weeks (**Figure S4.1-22**, left). However, longer failures that get to about 150 days happen in the STN, MER and USJ in all scenarios. The 5-year long dry spells can cause failures greater than 100 days in all basins, especially in the USJ, where droughts longer than 4 years can cause consecutive failures for almost a year. Therefore, the resilience index (**Figure S4.1-22**, right) stays mostly below 10-15% for almost all sequences in the STN, MER and USJ. Due to the quicker recovery rates, the TUO has a better resiliency performance, however, rarely reaching a probability of recovery above 50%.

## 4.4. Discussion

### 4.4.1. Reservoir Storage and Flood Control

Significant storage fluctuations during prolonged drought periods have been observed in the historical period for the major storage reservoirs in the region (Facincani Dourado, 2023). In addition, considering that about 33-40% of all droughts in California end by landfalling atmospheric river storms (Dettinger, 2013), extreme wet years can relieve some immediate effects of extended droughts, while also producing widespread flooding and threatening dam safety throughout the State (Wahl et al., 2020). As noted by Kocis & Dahlke (2017) high-magnitude flows are often not captured to maintain the flood control pool of reservoirs to hold eventual additional flood waters. To handle these problems, some infrastructure, water and watershed management options can be considered and are further discussed below.

The STN and USJ are both highly regulated, however the relative storage capacity in the USJ is considerably smaller. Even though the USJ shows the smallest average relative losses in storage during droughts (approximately 15-30% for all dry years), these impacts are still very significant as the basin has the largest hydropower potential and agricultural demands. On the other hand, the TUO and MER have fewer reservoirs, though the TUO has greater storage capacity besides a greater water yield than all other basins. Still, the TUO had the lowest gains in storage in wet years, and the MER showed the highest volatility in storage levels among all basins.

Extreme wet years can cause the terminal reservoirs to release on average a water volume that can be equivalent to a total year of water yield in each basin (as per **Table 4-1**), generally with greater spillage in scenarios with two wet years (W2). In these cases, additional water storage and conveyance infrastructure, such as off-channel storage and bypass systems, are often considered to alleviate some system vulnerabilities, although this option can be generally cost prohibitive (Hamilton et al., 2022; Hanak et al., 2019) if not ineffectual (Nover et al., 2019).

### 4.4.2. Hydropower Generation

As noted by Tarroja et al. (2019), climate projections anticipate greater losses in hydropower generation due to an increase in spillage, further complicating the balance of water and energy needs, besides economic and environmental implications. In general, losses in hydropower generation are not as severe in the first dry year after the wet year(s) as in the remaining dry years, except for the USJ, where differences are not as noticeable, similar to the basin’s storage behavior. This shows that one very wet year can already alleviate the stress caused by droughts, by generally bringing the system response closer to or above average. Likewise, previous studies have shown a general decrease in hydropower generation under climate change driven by combinations of shifting hydrology (e.g., earlier center of mass), large flood driven spill events, and drought (Madani & Lund, 2009, 2010b; Null, Viers, et al., 2010). In addition, studies of high elevation facilities found earlier snowmelts and streamflows can affect basins that lack sufficient storage capacity (Madani et al., 2014; Vicuna et al., 2007).

The STN and TUO can sustain a little buffer for the first dry year after all wet years on average, in which we can see that generation losses tend to be about half of the ones occurring in the precursory and following dry years. However, on average the TUO does not get above mean historical generation in the first wet year, with little compensation in generation in the second wet year, demonstrating a slower recovery rate as a potential result of spillage. The MER basin shows the widest range of variation in generation, as the powerhouses in the region all depend on the water stored in the terminal reservoir in the lower watershed, and agricultural water demand are mostly also met downstream of the powerhouses. Similarly, the USJ presents the worse performance during droughts, as a result of the relatively low storage capacity of reservoirs within the basin.

Planners need to account for these whiplash effects and develop flexible strategies to manage power production during both wet and dry periods. This may include investing in advanced forecasting and monitoring systems, adopting adaptable operational strategies, and exploring alternative sources of power generation to balance energy supply during extreme conditions. The significant decrease in hydropower generation, especially during the summer season indicates that the limited stored water is likely being utilized to attempt preserving environmental flow requirements. These findings indicate that in the near future, hydropower generation could be enhanced by implementing optimization analyses that prioritize both sustainable ecological function and power generation. This can be more effectively achieved by the use of coupled water-energy system models to better represent these conflicting water-centric and energy-centric operational priorities (Rheinheimer et al., 2023). In addition, physical modifications of reservoirs and their associated powerhouses could mitigate potential power losses tied to increased spill events (Forrest et al., 2018).

### 4.4.3. Agricultural and Environmental Water Deliveries

Management decisions on water allocation for human and environmental uses are typically based on the typology of annual runoff at the supplying facilities, commonly referred to as “water year type”, as described by Null & Viers (2013). These constraints have been imposed so that the water resources systems must be operated to meet water quantity and quality objectives. However, policies implemented in practice tend to address only short-term changes in objectives (Loucks & van Beek, 2017). A common example for dealing with low flow periods on the short-term is releasing e-flows below the prescribed flow schedule, i.e., in case inflows into a reservoir are lower than prescribed e-flow releases downstream, inflows are released instead. Therefore, the changing inter-annual hydroclimate’s volatility could present greater challenges to water management and require long-term changes in operational decision-making and planning (Cheng & Liu, 2022). This is particularly important for the basins with smaller storage capacity, as extreme low flows could propagate below reservoirs, which could instead alleviate these effects through the incorporation of changes in e-flow schemes.

In the study area, water allocation to irrigation deliveries is limited in drier years despite the higher demands, according to observed data used in the model calibration process. Historically, water deliveries in the STN, TUO, MER basins are generally reduced by 12%, 16%, 20%, in critically dry years, mostly due to increased water rate schedules in dryer years and/or limited surface water status (Davids Engineering, 2021; MID, 2013; TID, 2012). In the USJ, water allocation to the CVP is governed by the San Joaquin Valley Index (SJVI), used to establish whether the current water year is classified as critically dry, dry, below normal, above normal or wet. The SJVI is calculated based on the total inflow into the four terminal reservoirs in the region (0.6×April-July runoff + 0.2×October-March runoff + 0.2×previous year’s index) (Null & Viers, 2013b). Historically, CVP deliveries are reduced by 63% on average in a critically dry, when compared to a wet year.

On the other hand, e-flows in the region are set as minimum flow requirements established by year type classifications, and may also depend on existing flows, reservoir storage, and the specific fishery needs for any given year (Cain et al., 2003). The majority of e-flow thresholds and schedules are also defined in hydropower licenses based on the SJVI, applied to specific points within natural river channels below storage or diversion dams (Rheinheimer et al., 2022). One of the few exceptions is the flow schedule for the MER river, in which only dry or wet years are classified based on the forecasted April-July unimpaired runoff (FERC, 1964). The MER can generally keep volumetric reliability above 80% and time reliability above 65%, although with total ranges from 52-100% and 24-100%, respectively, depending on the length and severeness of drought. Wet years bring the reliability indices up to or closer to 100%, and the improved reliability lasts for at least the first dry year in the MER basin. However, similar to the irrigation deliveries, the reliability of e-flows also continuously drops with longer droughts, likely due to the differing water year type classification. The TUO basin has greatest reliability for e-flows, with few sequences affecting flow deliveries especially when the basin is hit by 5 dry years (D5).

Recovery times for e-flow deliveries ranged from mostly around a week in the TUO to mostly up to 20-30 days in the other basins, with much longer failures occurring in certain sequences. The STN, but especially the MER, and USJ are particularly vulnerable to droughts longer than four years. Overall, the basins show low resilience indices, except for the TUO. The occurrence of failures, recovery rates and resiliency index highlight that the STN and TUO basins have higher resilience in agricultural deliveries, being able to recover a satisfactory state and rebound quickly after a failure; the MER and USJ basins have limited resilience with longer droughts leading to more frequent and prolonged failures. These metrics were not greatly affected by the occurrence of 1 or 2 wet years.

### 4.4.4. Policy Implications

As stated by Hanak et al. (2019), there is potential to capture more water, especially during very wet years in California, by expanding surface reservoir capacity or groundwater recharge. These high-magnitude event flows could also be managed more efficiently through forecast improved reservoir operations (FIRO). Data-driven FIRO decisions based on inflow forecasts can enhance water storage by adopting flexible flood control operational criteria that can lead to greater reliability for water supply and ecosystems, power generation, and managed aquifer recharge (MAR), while maintaining existing flood control capabilities (Cobb et al., 2023; Ralph et al., 2014; Woodside et al., 2022; Zarei et al., 2021). Moreover, the use of flood waters for MAR (Flood-MAR) (Kocis & Dahlke, 2017) has been adopted to address the widespread groundwater overdraft in the state (Leahy, 2016), especially when using the existing irrigation infrastructure for on-farm recharge (Ag-MAR) (Levintal et al., 2023).

FIRO operations can also help maintain hydroelectricity production and water deliveries, especially during the summer, when water and power are needed the most (Kocis & Dahlke, 2017; Naz et al., 2018). Furthermore, MAR operations can repurpose flood flows to balance overdrafted groundwater basins, as required by the Sustainable Groundwater Management Act (SGMA) (Hanak et al., 2019). Our results indicate greatest MAR potential in the TUO and USJ basins (**Table S4.1-3**). These alternatives not only enhance energy and water security but also align with California's policy objectives of achieving groundwater sustainability, and also reducing greenhouse gas emissions as hydropower generation compensates for carbon-emitting power plants (Tarroja et al., 2019).

FIRO's adaptive flood space, responsive to forecast ability, allows operators to empty reservoirs below the pre-determined flood curve when anticipating the arrival of an atmospheric river in the region. Conversely, if no significant storm is forecasted, the reservoir can retain water above the original flood control curve. Monitoring meteorological and watershed conditions could enable safe controlled reservoir releases ahead of time, as a risk-mitigation strategy to avoid flooding, such as those seen in **Figure S4.1-10** and **Figure S4.1-11**. In addition, greater groundwater storage would facilitate conjunctive water use during periods of scarcity. For instance, California's extreme dry year in 2015 resulted in ~50% reduction in hydropower production due to insufficient supply of stored water, which in turn was replaced with natural gas and offset by increasing penetration of solar and wind sources (Gleick, 2017).

In addition, as recommended by Rheinheimer et al. (2016), climate-adaptive options for water year typing should be considered to better allocate resources. Adaptations to nonstationarity require a response to the climate-driven change in runoff timing and magnitude; to reconcile competing demands, water year type definitions (i.e., their calculations and respective timing and volume of deliveries) should be regularly updated and regionally concordant to improve reliability. Such adaptations are important especially for maintaining e-flows, as human uses are commonly prioritized worldwide in detriment of e-flow targets (Facincani Dourado, 2023), especially in areas such as the Central Sierra Nevada, where significant competing demands for limited water resources exist.

Furthermore, adopting adaptation strategies will require policy changes, as water storage and release schemes of non-federal facilities are determined in the long-term licensing process conducted by the Federal Energy Regulatory Commission (FERC) (Viers, 2012). Existing frameworks often rely on fixed rules and structures that lead to institutional inertia and resistance to change. For instance, most hydropower facilities in the study area, as well as in the US in general, are regulated by FERC. In the licensing process, FERC sets the reservoir operating rules (e.g., flood control operations, water year type classifications and environmental flow schedules) based on stationary hydrology for the life of the license, typically 30–50 years in length (Viers, 2011). Yet, as suggested by Viers & Nover (2018), alternative reservoir operations could be considered in the relicensing process through the incorporation of formal environmental impact studies, sensitivity analyses, climate-informed 'worst case' scenario and adaptive management.

In addition to the strategies mentioned above, others can be implemented, such as the physical modification of reservoirs and their associated powerhouses to mitigate potential power losses tied to increased spill events (Forrest et al., 2018). Furthermore, water utilization patterns could be more balanced with the stochastic hydrologic processes comprised of random sequences of high flow and low flow periods (Nagy et al., 2013). For instance, taking advantage of hydro and solar power generation complementarity for load and peak demand-balancing, therefore, producing more hydropower when water is naturally more available (Marshall & Chen, 2022).

If failures are expected to be reoccurring, prolonged, and system recovery is slow, especially for environmental and agricultural water deliveries in the STN, MER and USJ, these systems' designs, allocation strategies and management policies need to be reconsidered to allow for greater reliability and resiliency. Implementing forward-thinking policies that account for unpredictable shifts in climate patterns is crucial for sustainable water management, to ensure the availability of water for both ecosystems and human needs in the face of a changing hydroclimate.

## 4.5. Conclusion

Droughts longer than one year can already disrupt services to a certain level, especially in basins with lower storage capacity. Extremely wet years temporarily increased carryover storage; however, subsequent dry years typically negate these gains. Besides, extreme wet years can also pose flooding risks and threaten dam safety caused by extreme spillage. Meanwhile, 5 year-long droughts can cause significant drops in energy generation and water deliveries in the TUO, MER and USJ basins. One or two very wet years generally mitigate the effects of droughts and bring the hydropower systems closer to or above average generation levels. The STN and TUO basins demonstrate a partial buffer in the first dry year after wet years, with generation losses about half of those in precursory and following dry years; however, the TUO basin shows a slower recovery rate. The MER basin exhibits the widest variation in generation, impacting also agricultural and environmental water deliveries downstream of the powerhouses. The USJ basin performs the worst during droughts, experiencing significant generation losses due to its low storage capacity.

Regarding agricultural deliveries, the STN and TUO basins exhibit relatively better performance, with drier sequences occasionally impacting reliability during longer droughts. The USJ and MER basins show greater variability, with extremely wet years meeting target allocations while longer droughts result in significant reductions. The MER basin performs better during short drought interruptions, with declining performance as the drought proceeds. Meanwhile, the USJ basin maintains a lower but more stable performance during longer droughts due to substantial reductions in CVP deliveries.

The reliability indices for e-flow deliveries exhibit similar patterns, with lower performances, observed in the MER, and especially USJ, which also showed the lowest resilience. Time reliability shows a wider range of responses, notably in these two basins, indicating discrepancies in delivery timing, while the volumetric index aligns moderately more consistently with flow schedules. The TUO basin shows greater e-flow reliability and resilience, with more noticeable impacts in few sequences of longer droughts. The STN and USJ basins display variable responses, with the STN maintaining higher volumetric reliability and resilience but lower time reliability.

As a significant increase in the frequency of whiplash events is expected (Mount & Hanak, 2018), the need for greater climate resilience is imperative, and requires water resource infrastructure and management to incorporate flexibility (Ficklin et al., 2022). Future research efforts could assess the occurrence, changes and driving factors of whiplash events in future climate scenarios, the impacts of water year types in the agricultural and environmental water deliveries, and the efficiency of implementing Flood/Ag-MAR and FIRO. Further research could also test hydrologic sequences or specific extreme years, to assess the intra-annual impacts of specific timing, length and magnitude of flood and drought events, to help bring awareness that such conditions are possible to help decision makers consider risks and surprises in water management.

### 4.6. Open Research

The source code for the case study, which includes all code and *Pywr* extensions noted in this work, is available in a publicly available Zenodo repository at https://doi.org/10.5281/zenodo.10689850 (Rheinheimer et al., 2024). Intermediary data needed for model input is generated via Python scripts, as described in the main GitHub repository’s README file. Electricity price data from HiGRID is within the same model code repository and from Tarroja et al. (2021). Runoff data is available from Facincani Dourado et al. (2021).

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## Appendix 4.1: Supporting information for Chapter 4

### Content

Figures S4-1 to S4-22 and Tables S4-1 to S4-4.

### Introduction

This supporting information provides extra figures that help complement explanations of the methodology used in this paper (Figures S4-1 to S4-5, and Tables S4-1 and S4-2), and results (Figures S4-6 to S4-22, and Tables S4-3 and S4-4).

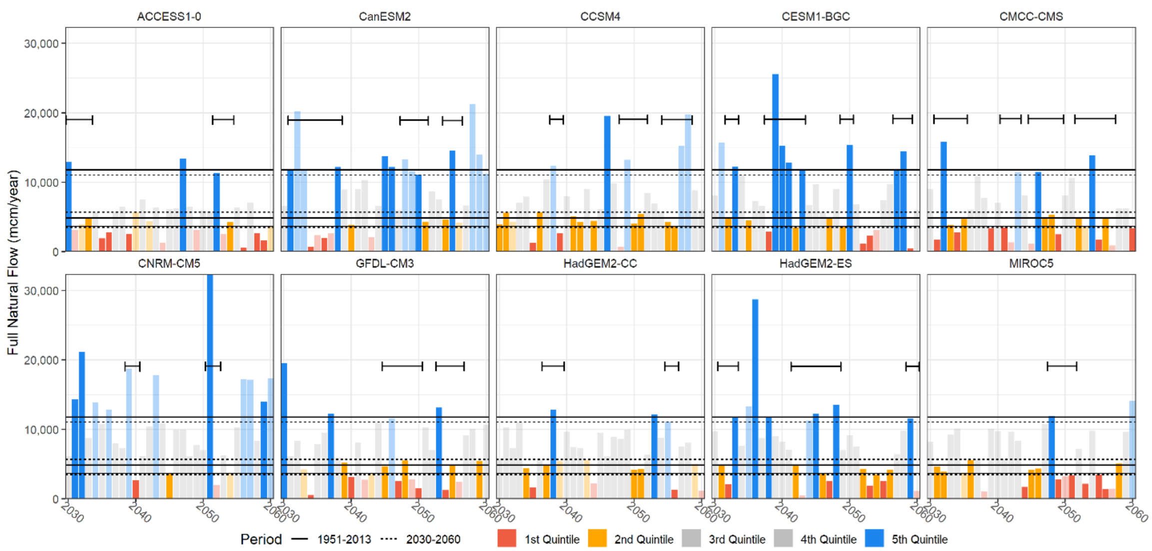


**Figure S4.1-1.** Future full natural flow projections from the 10 GCMs, highlighting the occurrence of multi-year whiplash. Water years not sampled for creating the synthetic sequences are transparent. Solid and dashed lines represent the 20$^{th}$, 40$^{th}$ and 80$^{th}$ percentiles for the historical and future data, respectively.

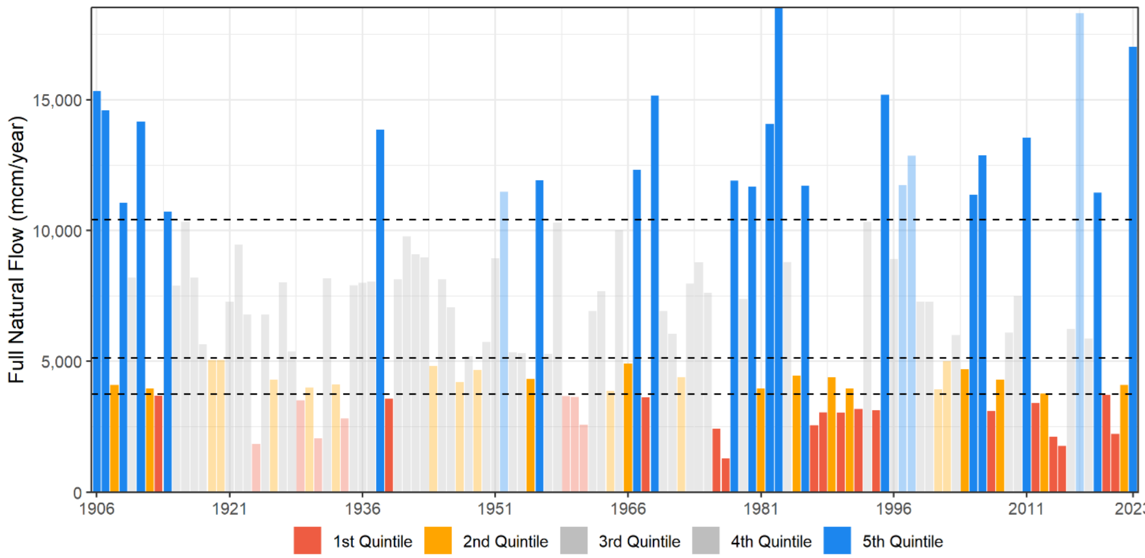


**Figure S4.1-2.** Historical occurrence of oscillations between wet and dry water years from full natural flow estimates from the California Data Exchange Center (CDEC). Non-whiplash years are transparent.

**Table S4.1-1.** Autocorrelation analysis of annual full natural flow estimates from CDEC. Pearson's correlation coefficient (r) and Spearman's rank correlation coefficient (ρ) test the linear relationship and strength of association within between time series, and lagged versions of itself. Here we included lags of up to twelve years (k=1 to k = 12), to represent the length of the synthetic hydrological sequences created in this study.

| Lag autocorrelation (k) | Pearson's correlation (r) | | Spearman's rank correlation coefficient (ρ) | |
|---|---|---|---|---|
| | r | p-value | ρ | p-value |
| 1 | 0.05 | 0.62 | 0.08 | 0.40 |
| 2 | 0.06 | 0.55 | 0.06 | 0.54 |
| 3 | -0.12 | 0.22 | -0.14 | 0.13 |
| 4 | -0.01 | 0.88 | -0.02 | 0.83 |
| 5 | 0.00 | 0.98 | 0.05 | 0.61 |
| 6 | 0.01 | 0.88 | -0.02 | 0.82 |
| 7 | -0.13 | 0.18 | -0.08 | 0.40 |
| 8 | -0.16 | 0.09 | -0.15 | 0.13 |
| 9 | -0.13 | 0.17 | -0.13 | 0.16 |
| 10 | -0.16 | 0.11 | -0.15 | 0.12 |
| 11 | 0.15 | 0.13 | 0.14 | 0.16 |
| 12 | -0.02 | 0.82 | -0.11 | 0.28 |

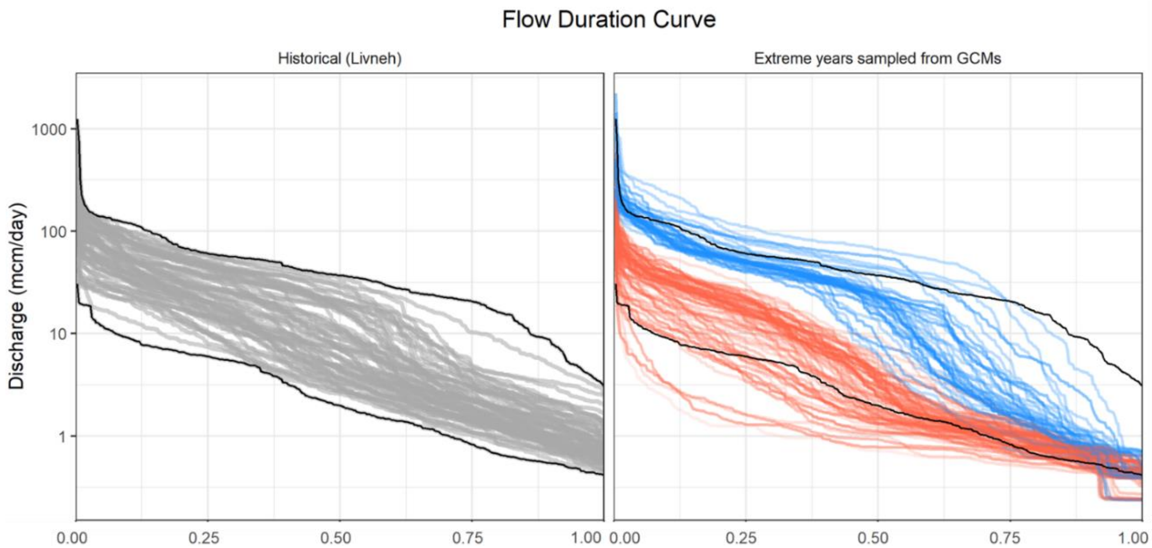


**Figure S4.1-3.** Flow duration curves of the historical (left) and extreme years sampled from GCMs (right) streamflow data used in this study. The solid black lines show the range of flows in the historical period. Dry and wet years sampled from future climate change scenarios to construct the synthetic whiplash sequences are in orange and blue, respectively.

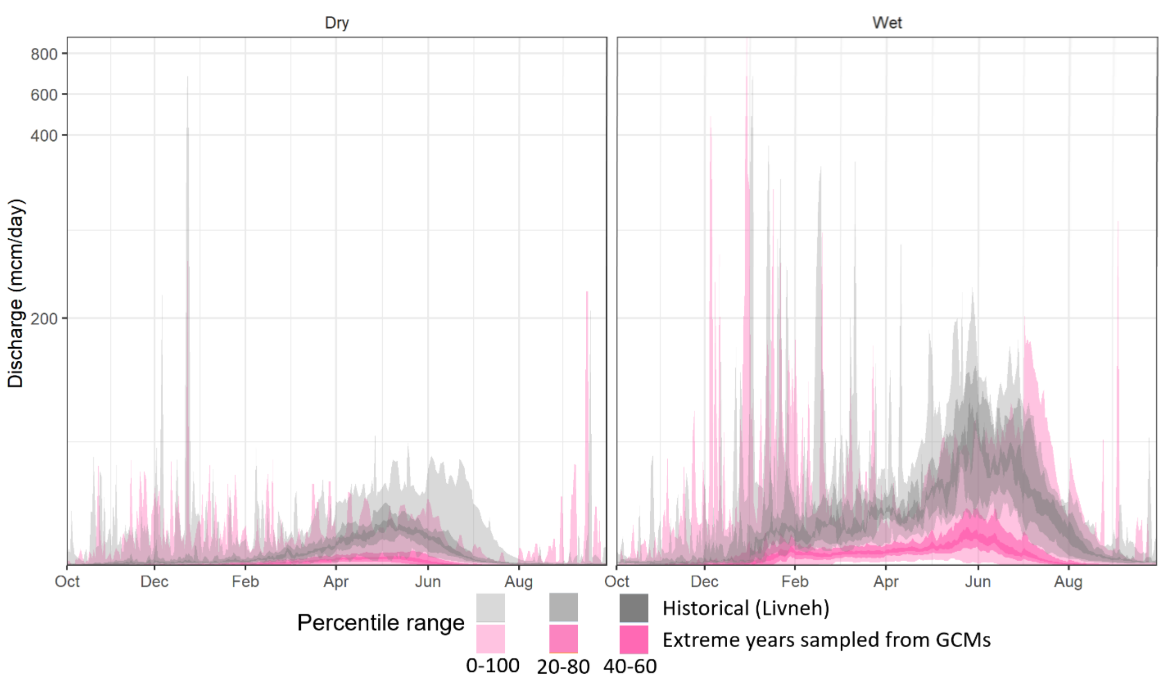


**Figure S4.1-4.** Percentile ranges of daily flows for dry (left; <40th percentile) and wet (right; >80th percentile) from the Livneh dataset (gray) and the extreme years sampled from the GCMs (pink).

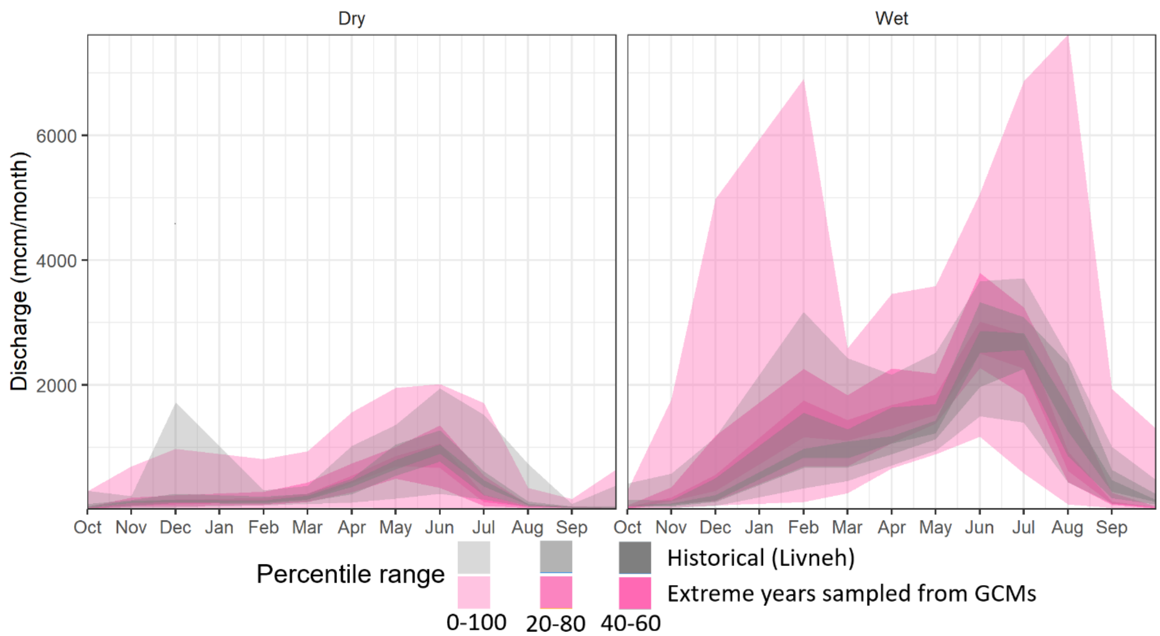


**Figure S4.1-5.** Percentile ranges of monthly flows for dry (left; <40$^{th}$ percentile) and wet (right; >80$^{th}$ percentile) from the Livneh dataset (gray) and the extreme years sampled from the GCMs (pink).

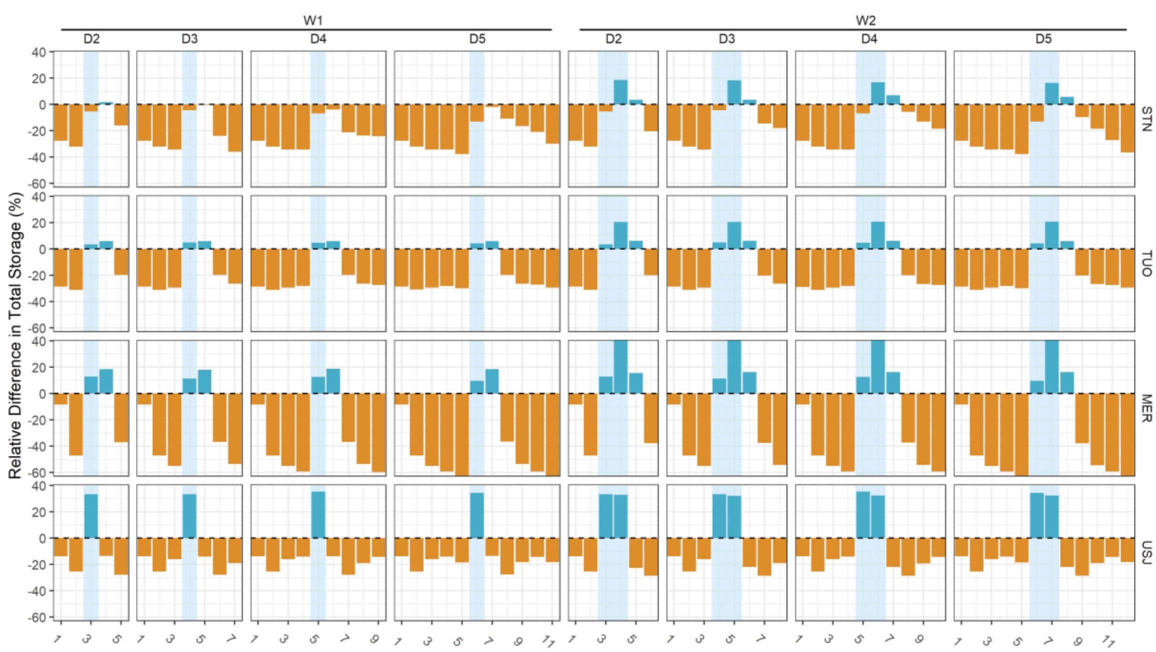


**Figure S4.1-6.** Average annual relative change in storage compared to the historical mean (1951-2010 water years) in the Stanislaus (STN), Tuolumne (TUO), Merced (MER) and Upper San Joaquin (USJ) basins. Wet years are shaded in light blue.

**Table S4.1-2.** Initial storage levels adopted at the start of model runs (Modeled) on October 1, and mean and standard deviation of observed storage levels by USGS gauges, when available, for the months of September and October (1980-2011). Reservoirs are ranked by storage capacity.

| Basin | Reservoir | Storage Capacity (mcm) | Initial Storage Level (%) | | |
|---|---|---|---|---|---|
| | | | Modeled | Observed | |
| | | | | Mean | ± Standard Deviation |
| STN | New Melones Lake | 2985 | 50 | 53.5 | 27.4 - 79.6 |
| TUO | Don Pedro | 2504 | 50 | 69 | 53.9 - 84.1 |
| MER | Lake McClure | 1238.6 | 50 | 49.6 | 25.9 - 73.3 |
| USJ | Millerton Lake | 642 | 50 | 43.9 | 28.9 - 58.9 |
| TUO | Hetch Hetchy | 444.6 | 50 | 76.2 | 62.9 - 89.5 |
| TUO | Cherry Lake | 338.2 | 50 | 71.6 | 48.9 - 94.3 |
| STN | New Spicer Meadow | 227.2 | 50 | 58.6 | 43.6 - 73.6 |
| USJ | Shaver Lake* | 167.2 | 55.8 | 67.3 | 41.8 - 92.8 |
| USJ | Lake Thomas A Edison | 154.2 | 50 | 57.8 | 26.6 - 89 |
| USJ | Mammoth Pool | 147.9 | 50 | 34.6 | 13.5 - 55.7 |
| STN | Beardsley | 118.7 | 50 | 64.1 | 40.6 - 87.6 |
| USJ | Huntington Lake | 104.1 | 50 | 96.3 | 86.1 - 100 |
| STN | Lake Tulloch* | 82.6 | 82 | 90.1 | 80.7 - 99.5 |
| STN | Donnells | 79.8 | 50 | 50 | 27.8 - 72.2 |
| USJ | Florence Lake | 76.8 | 50 | 28.1 | 1.6 - 54.6 |
| USJ | Bass Lake | 55.6 | 50 | 67 | 57 - 77 |
| TUO | Lake Eleanor | 33.9 | 50 | 56.1 | 44.9 - 70.1 |
| USJ | Redinger Lake | 30.4 | 50 | 57.5 | 25.1 - 87.1 |
| STN | Pinecrest | 22.6 | 50 | 70.2 | 56.6 - 83.8 |
| STN | Relief | 18.4 | 50 | 44.4 | 19.7 - 69.1 |
| STN | Lyons | 7.5 | 50 | 37.4 | 21.5 - 53.3 |
| STN | Union Utica | 6.8 | 50 | 73.8 | 64.8 - 82.8 |
| USJ | Kerckhoff Lake* | 4.8 | 87.5 | 95.8 | 86.3 - 100 |

*Reservoirs with rules establishing minimum storage levels above the 50% initial storage level adopted

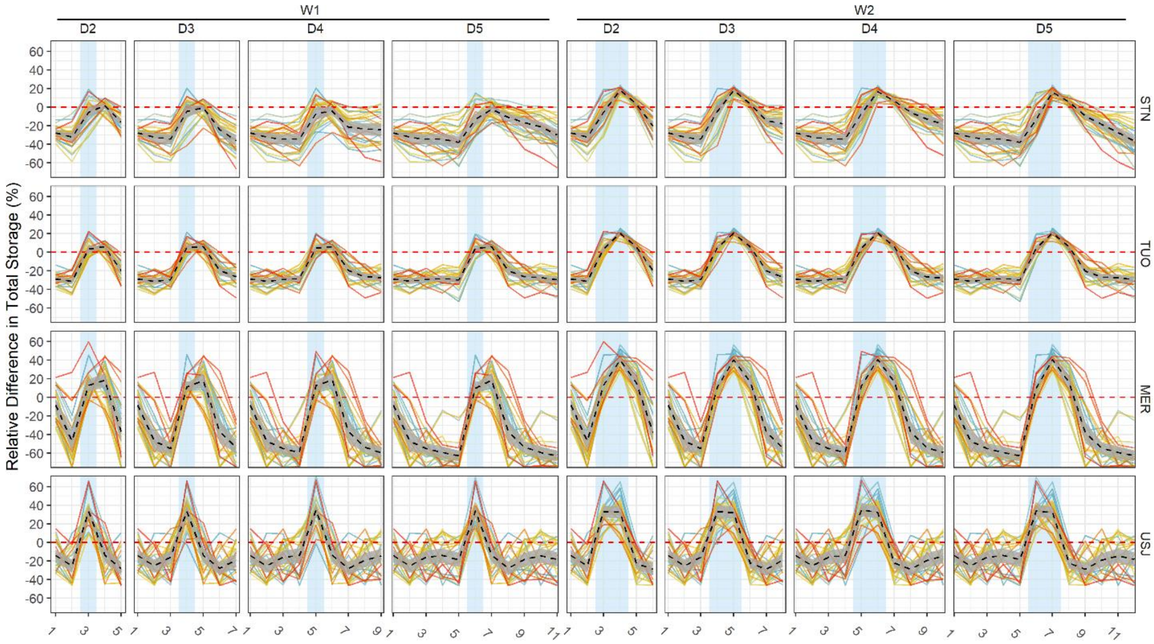


**Figure S4.1-7.** Annual relative change in reservoir storage compared to the historical mean (water years 1951-2010) for each whiplash sequence (wet years shaded in light blue). The black dotted lines with the gray background show the mean values with a 95% confidence interval.

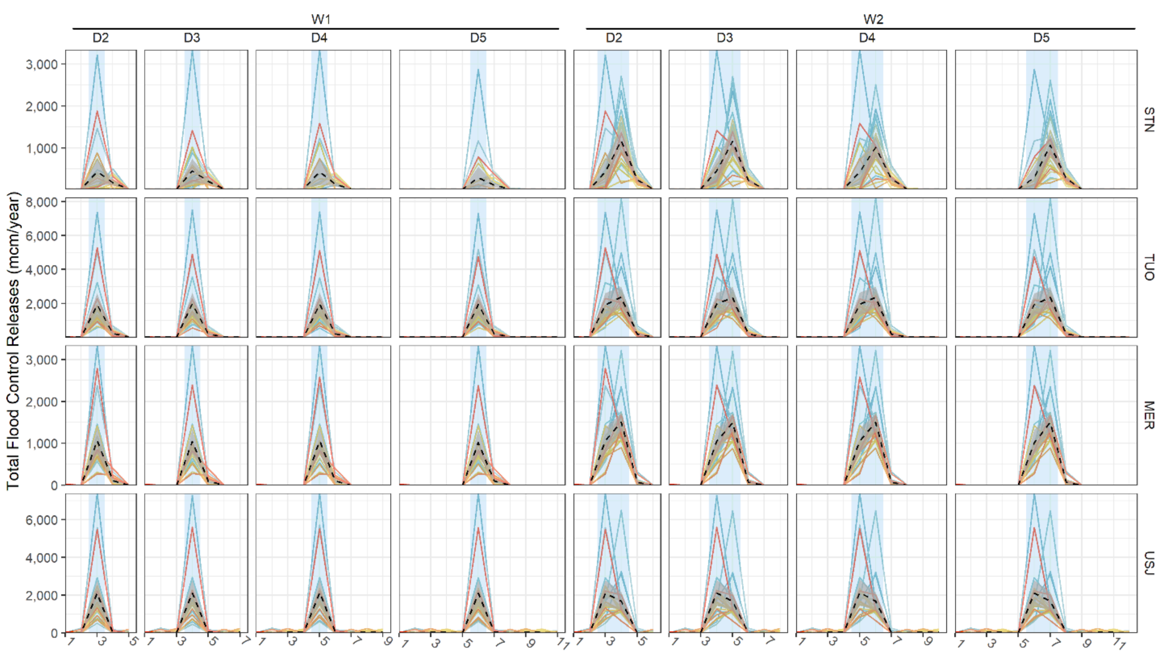


**Figure S4.1-8.** Annual flood control releases from the terminal dams (wet years shaded in light blue). The black dotted lines with the gray background show the mean values with a 95% confidence interval. Note the different y axis.

**Table S4.1-3.** Median, range and standard deviation of flood control releases from the terminal dams, that could be repurposed, such as for managed aquifer recharge.

| Basin | Scenario | Flood control releases (mcm/year) | | |
|---|---|---|---|---|
| | | Median | Range | Standard deviation |
| Stanislaus | W1 | 82.4 | 10.4 - 3345.3 | 705.0 |
| | W2 | 615.5 | 10.4 - 3345.3 | 755.7 |
| Tuolumne | W1 | 1394.4 | 554 - 7527.4 | 1598.7 |
| | W2 | 1646.8 | 554 - 8241.7 | 1549.4 |
| Merced | W1 | 827.6 | 272.1 - 3337.3 | 704.5 |
| | W2 | 1078.9 | 272.1 - 3337.3 | 650.3 |
| Upper San Joaquin | W1 | 1656.6 | 472.5 - 7397.2 | 1637.9 |
| | W2 | 1454.5 | 472.5 - 7397.2 | 1425.3 |

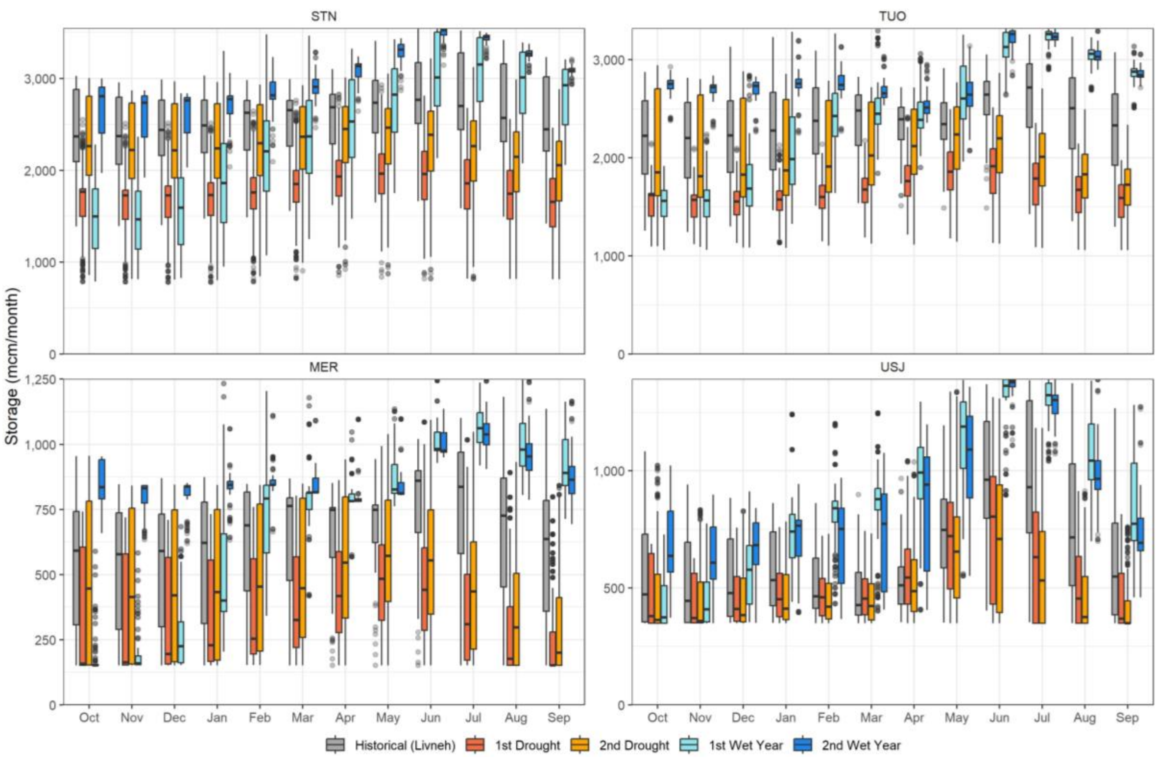


**Figure S4.1-9.** Monthly basin-wide reservoir storage in the historical period (water years 1951-2010), first and second dry sequences of dry years, and first and second wet years. Note the different y axes.

**Table S4.1-4.** Occurrence (count of successive days) and magnitude (percentage of exceedance) of flows higher than the maximum flow requirement threshold below the terminal dams.

| Basin | Maximum flow requirement (mcm/day) | Scenario | Occurrence (days) | | | Exceedance (%) | | |
|---|---|---|---|---|---|---|---|---|
| | | | Median | Range | Standard deviation | Median | Range | Standard deviation |
| **Stanislaus** | 19.6 | W1 | 5 | 1-54 | 14 | 1 | 0-632 | 93 |
| | | W2 | 3 | 1-54 | 10 | 1 | 0-632 | 60 |
| **Tuolumne** | 22.0 | W1 | 1 | 1-46 | 6 | 6 | 0-1301 | 103 |
| | | W2 | 1 | 1-46 | 5 | 1 | 0-1301 | 96 |
| **Merced** | 15.9 | W1 | 2 | 1-6 | 2 | 14 | 0-807 | 183 |
| | | W2 | 3 | 1-28 | 8 | 57 | 0-807 | 126 |
| **Upper San Joaquin** | 19.6 | W1 | 5 | 1-39 | 9 | 77 | 0-2497 | 179 |
| | | W2 | 6 | 1-62 | 10 | 73 | 0-2497 | 155 |

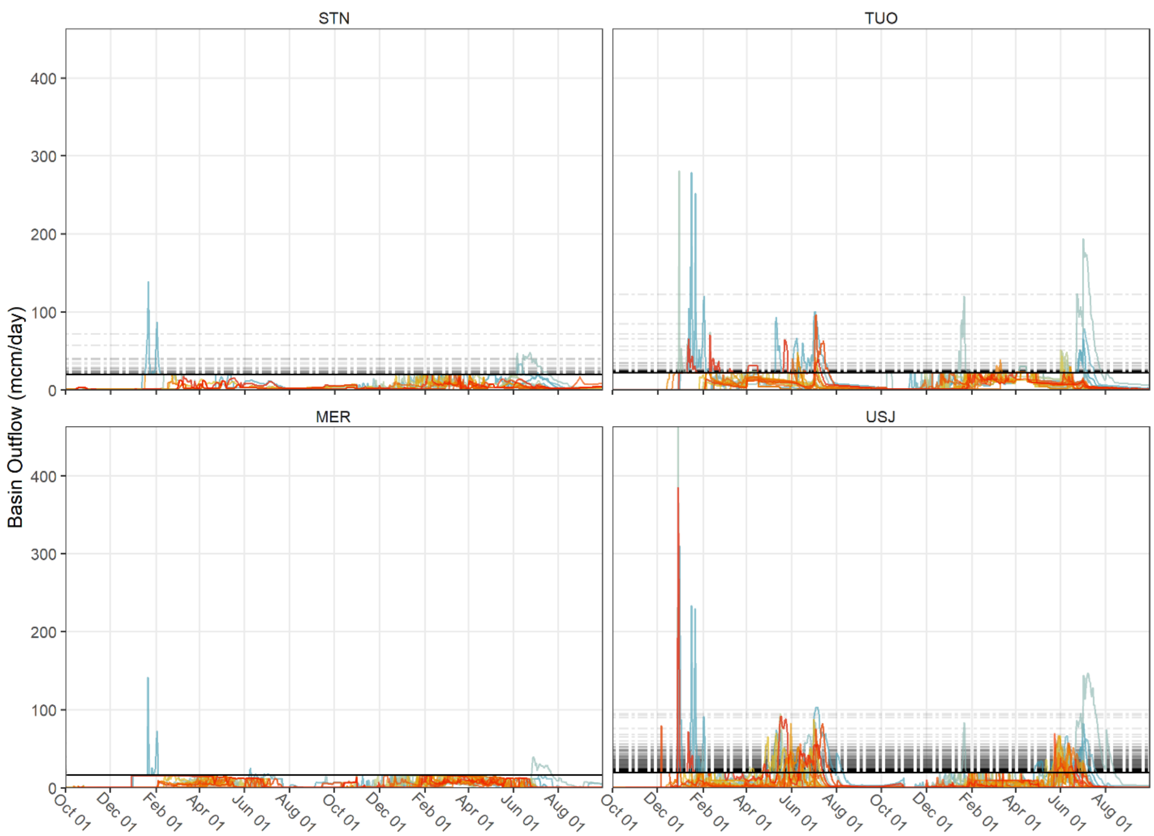


**Figure S4.1-10.** Daily basin outflows from the two consecutive wet water years from the D5W2D5 sequences. Solid black horizontal lines represent the maximum flow according to channel capacity, to prevent flooding. Dashed transparent black lines represent the occurrence of weekly flows above the maximum flow thresholds, as per observed USGS gauges.

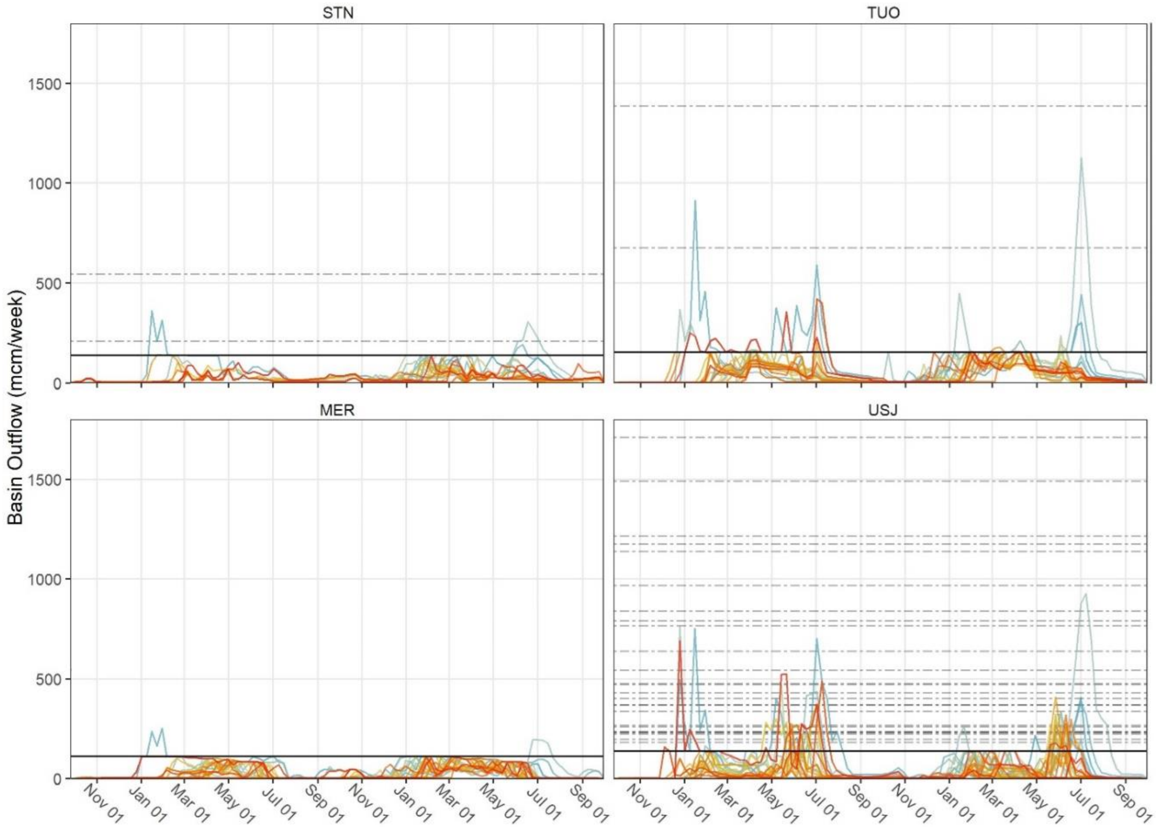


**Figure S4.1-11.** Weekly basin outflows from the two consecutive wet water years from the D5W2D5 sequences. Solid black horizontal lines represent the maximum flow according to channel capacity, to prevent flooding. Dashed transparent black lines represent the occurrence of weekly flows above the maximum flow thresholds, as per observed USGS gauges.

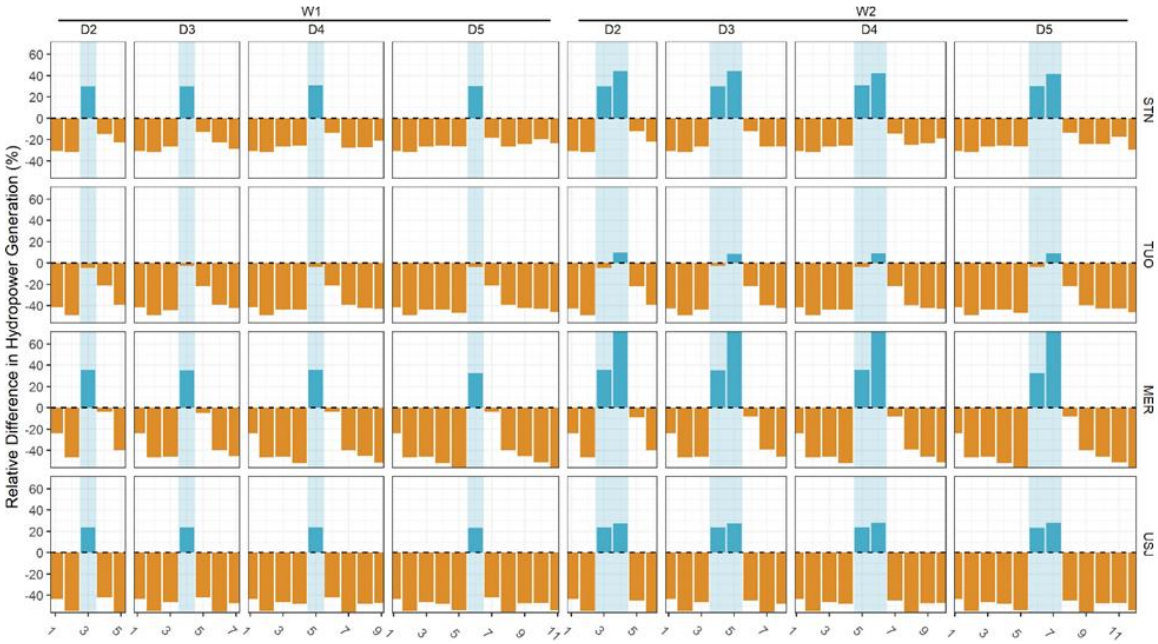


**Figure S4.1-12.** Average annual relative change in hydropower generation compared to the historical mean (1951-2010 water years) in the Stanislaus (STN), Tuolumne (TUO), Merced (MER) and Upper San Joaquin (USJ) basins. Wet years are shaded in light blue.

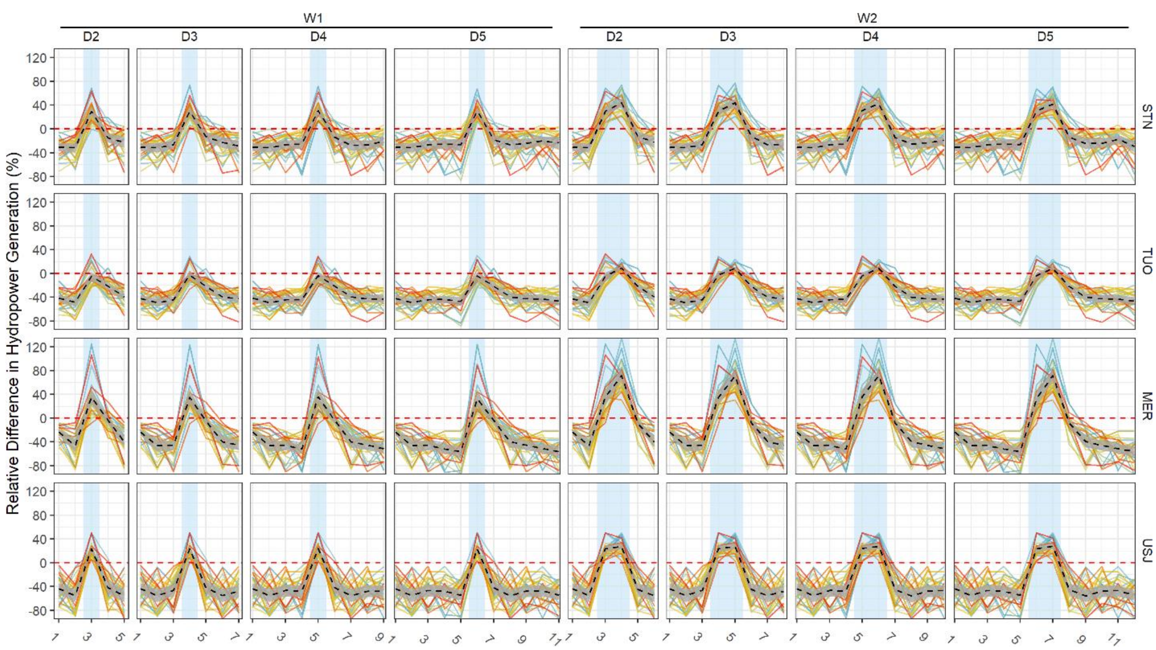


**Figure S4.1-13.** Annual relative change in hydropower generation compared to the historical mean (water years 1951-2010) for each whiplash sequence (wet years shaded in light blue). The black dotted lines with the gray background show the mean values with a 95% confidence interval.

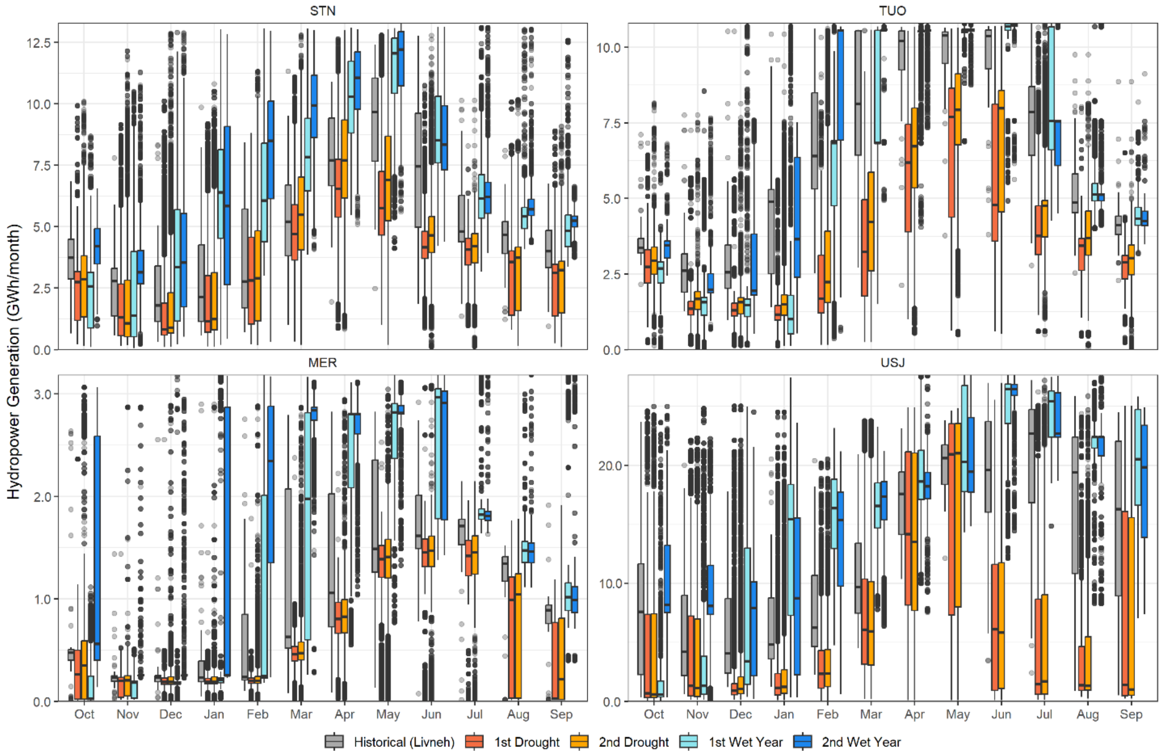


**Figure S4.1-14.** Monthly basin-wide hydropower generation in the historical period (water years 1951-2010), first and second dry sequences of dry years, and first and second wet years. Note the different y axes.

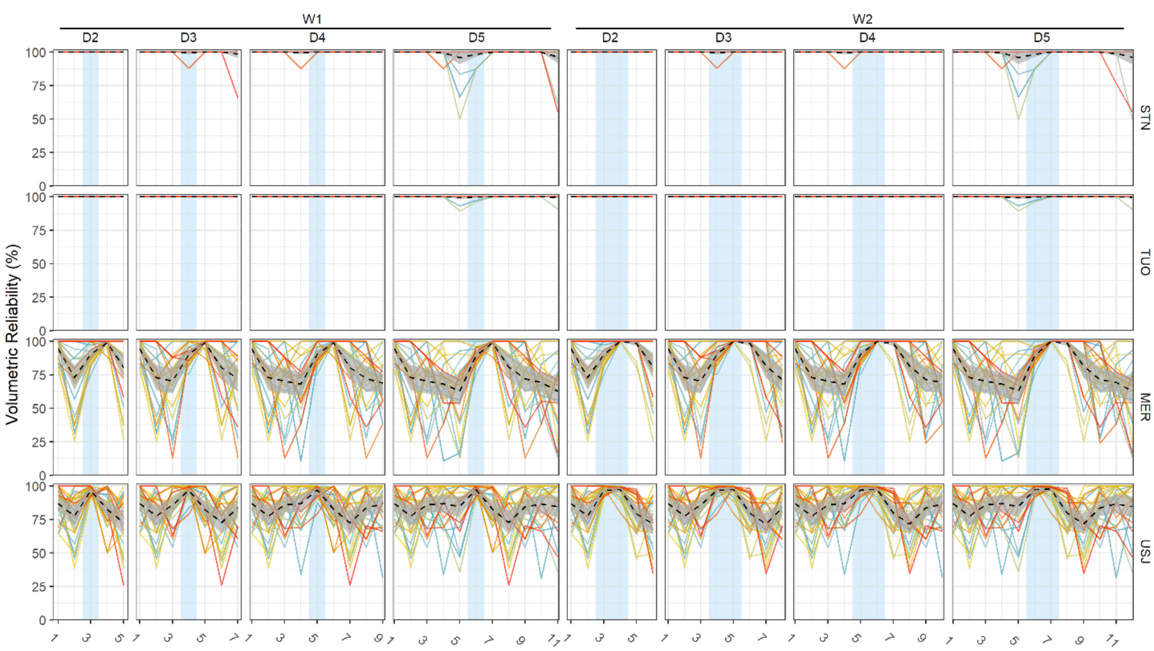


**Figure S4.1-15.** Annual volumetric reliability of agricultural deliveries. The black dotted lines with the gray background show the mean values with a 95% confidence interval. Wet years are shaded in light blue.

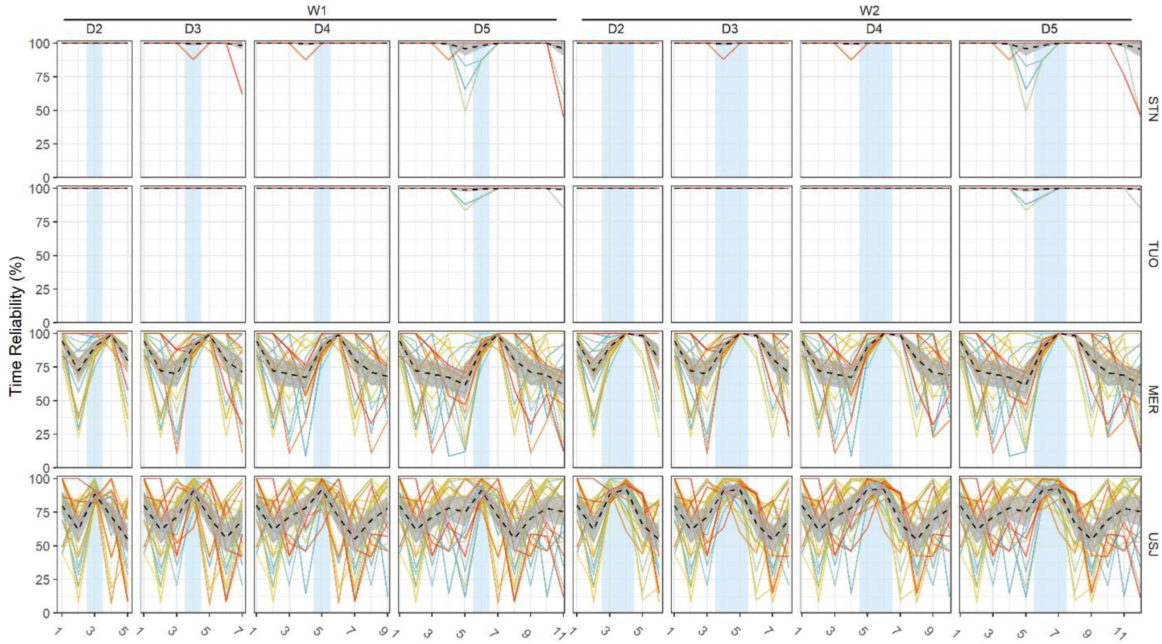


**Figure S4.1-16.** Annual time reliability of agricultural deliveries. The black dotted lines with the gray background show the mean values with a 95% confidence interval. Wet years are shaded in light blue.

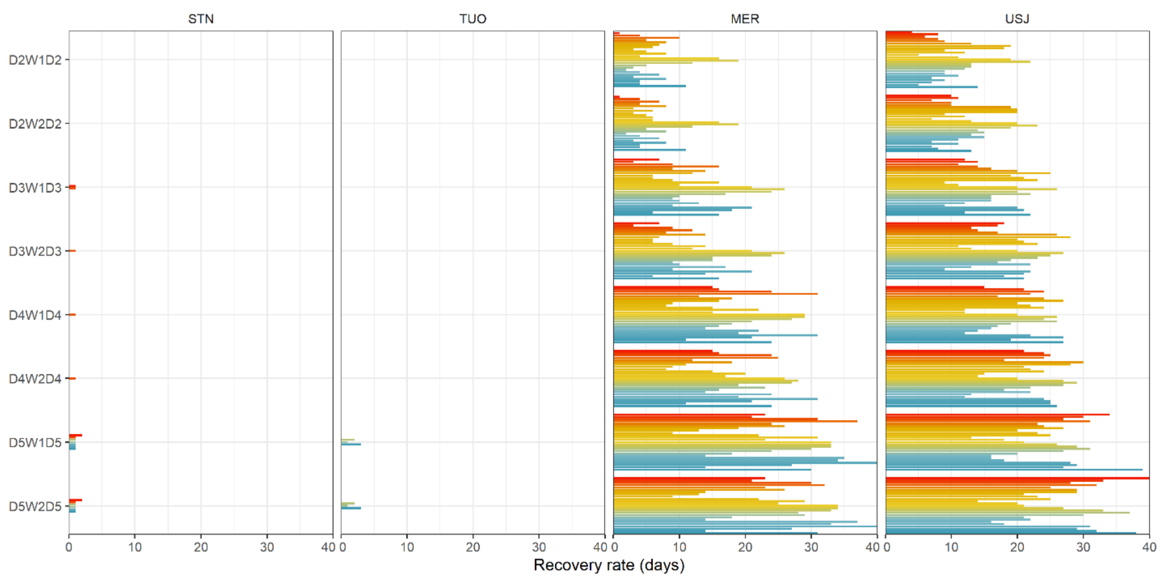


**Figure S4.1-17.** Total count of days in which agricultural deliveries went from a satisfactory to an unsatisfactory state of unmet demands for each sequence.

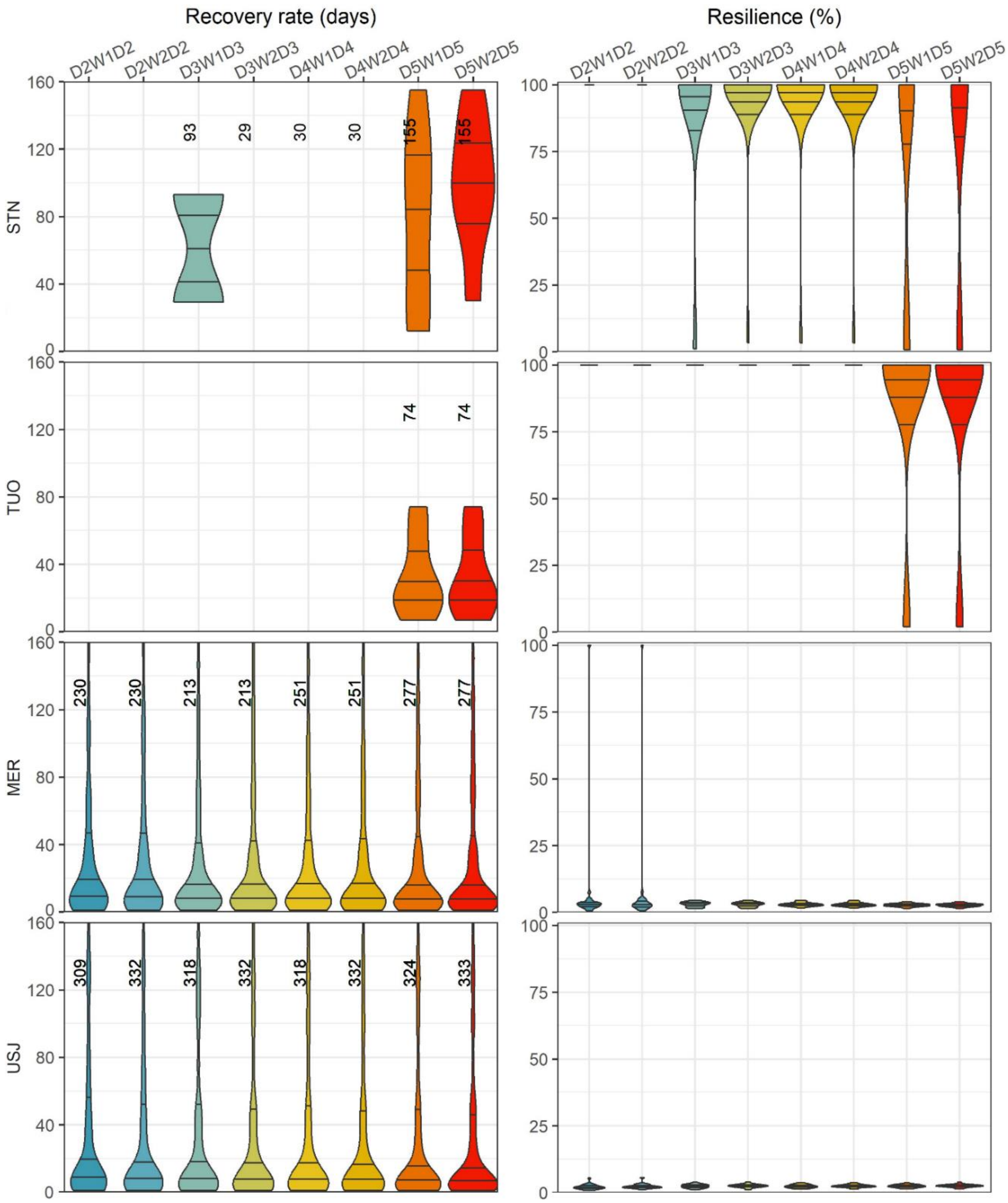


**Figure S4.1-18.** Violin plots showing the distribution of recovery rates and the longest recovery rate per group of sequences (left), and the resiliency (right) of agricultural deliveries.

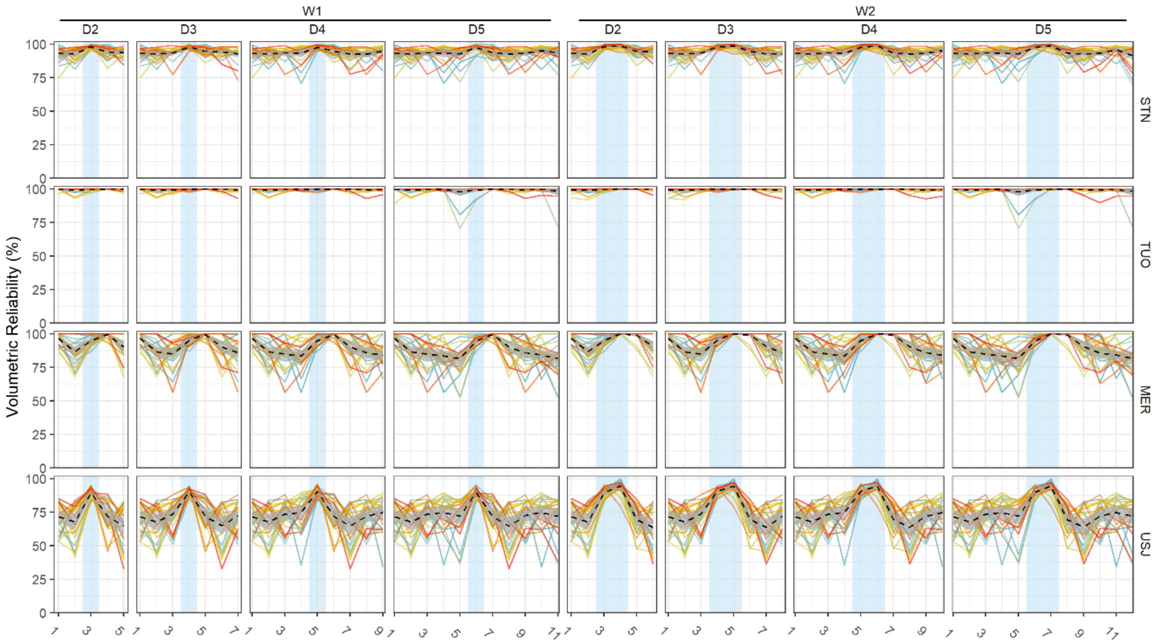


**Figure S4.1-19.** Annual volumetric reliability of environmental flows. The black dotted lines with the gray background show the mean values with a 95% confidence interval. Wet years are shaded in light blue.

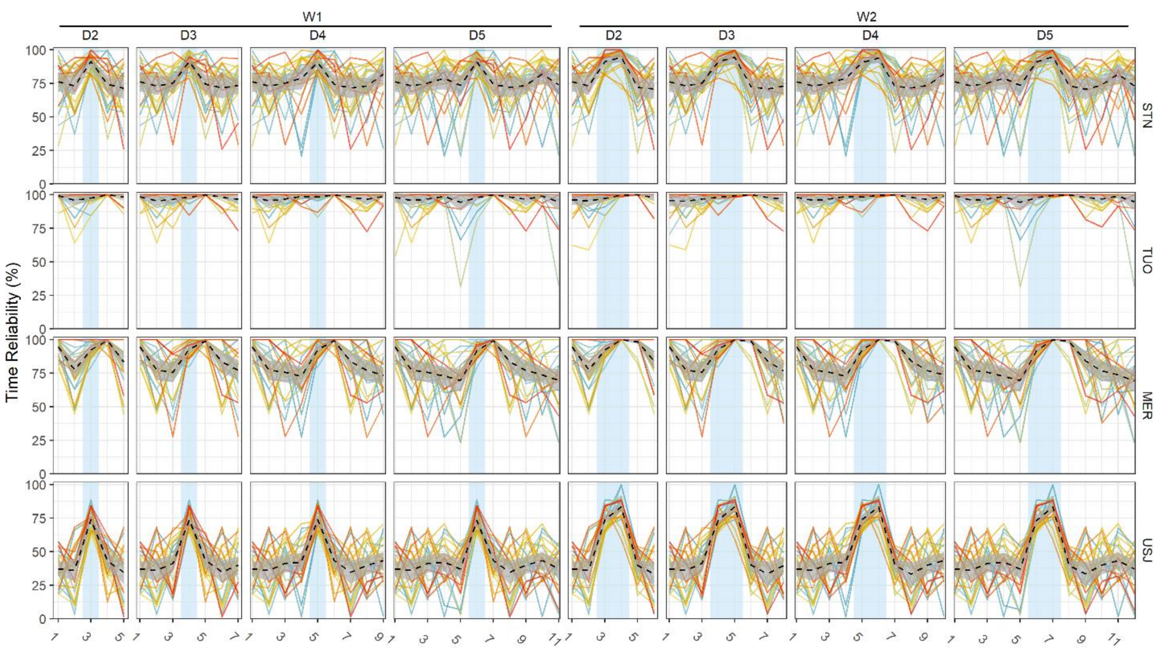


**Figure S4.1-20.** Annual time reliability of environmental flows. The black dotted lines with the gray background show the mean values with a 95% confidence interval. Wet years are shaded in light blue.

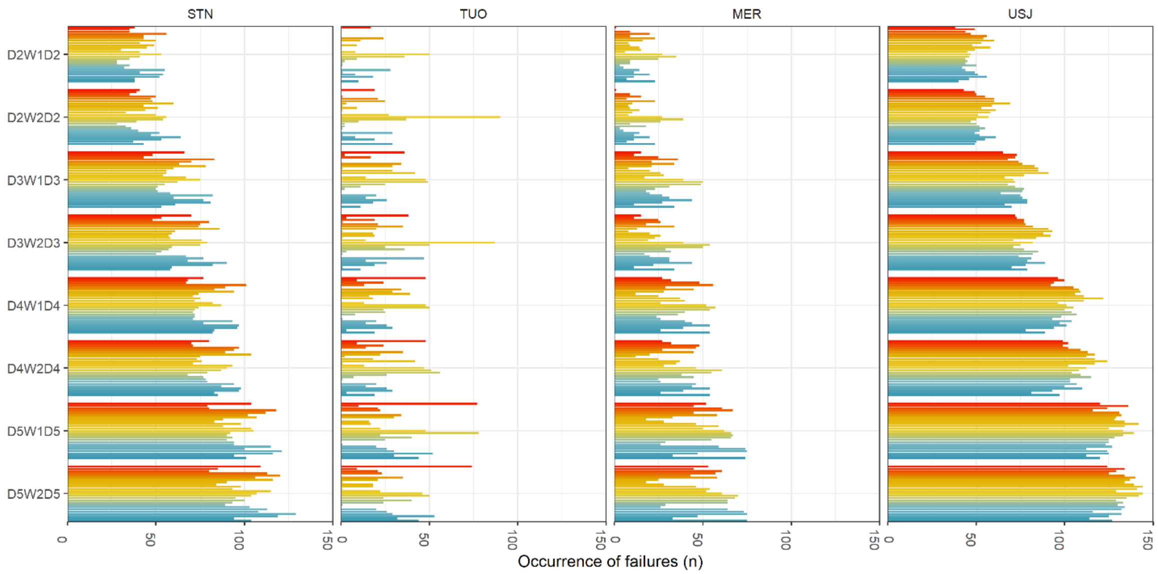


**Figure S4.1-21.** Total count of days in which environmental flow deliveries went from a satisfactory to an unsatisfactory state of unmet demands for each sequence.

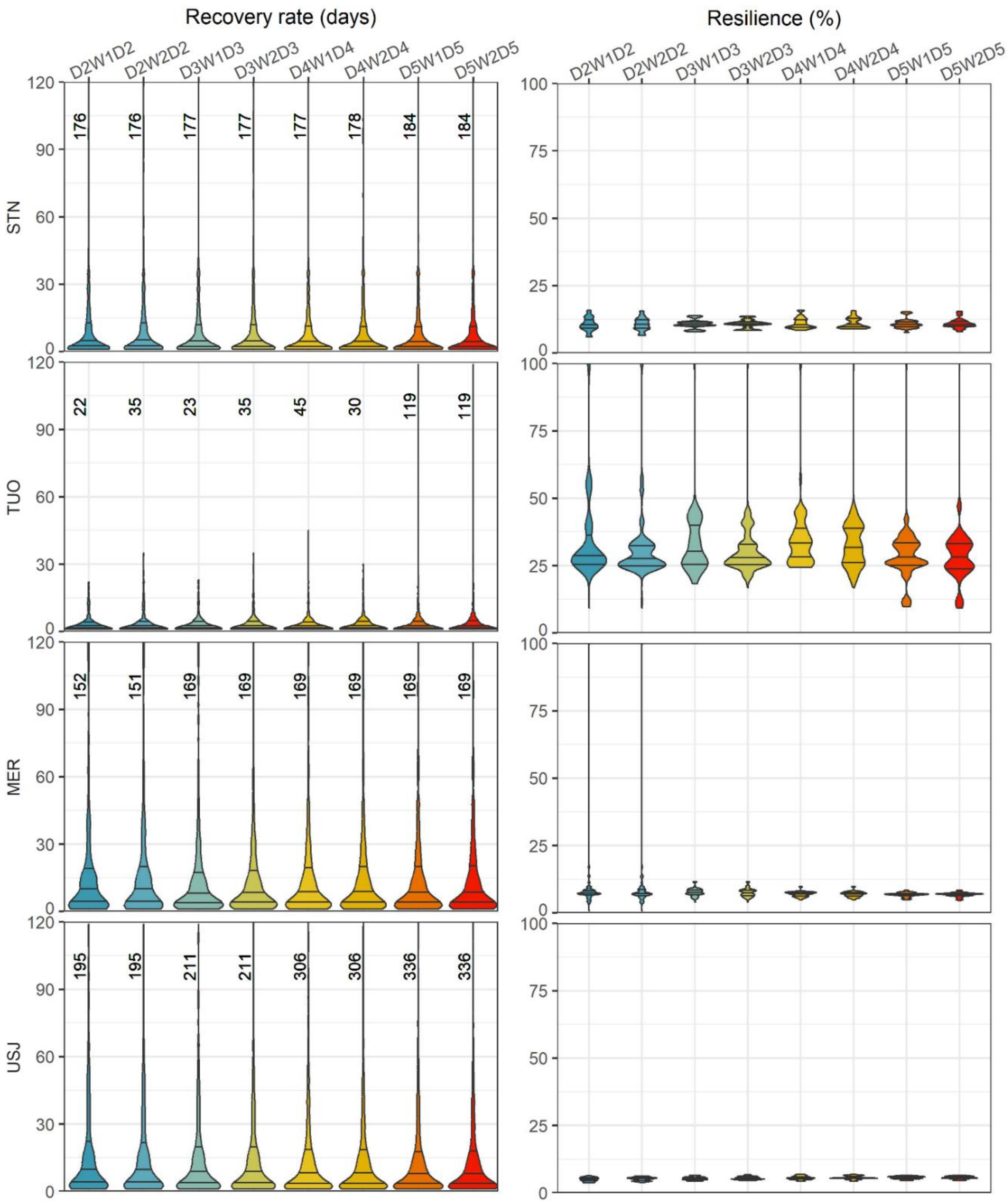


**Figure S4.1-22.** Violin plots showing the distribution of recovery rates (left) and the longest recovery rate per group of sequences, and the resiliency (right) of environmental flow deliveries.

## Chapter 5. Conclusion

The allocation of water to both human and environmental needs is influenced by a combination of physical infrastructure, operational strategies, regulatory frameworks, and the competing objectives of various stakeholders. Anticipating the best decisions for both human and environmental interests in the multifaceted nature of water resource management requires the consideration of the intricacies of each system. This dissertation demonstrates that through fresh perspectives on reservoir operations, hydropower generation, and water management, with a focus on the Central Sierra Nevada, California. Overall, the findings advocate for a holistic approach to integrated water resource management, emphasizing the importance of adaptive regulatory frameworks, stakeholder collaboration, and climate-resilient strategies to ensure the long-term sustainability and equitable allocation of California's water resources.

Implementing these comprehensive policies can guide more resilient and adaptive water resource management strategies amidst a changing hydroclimate. Continued discussion and consideration are essential to effectively address these complex challenges. To help guide this process, our findings suggest the need for:

**Integrated Approaches:** The governance setting, natural environment, spatial scale, stakeholders involved, local objectives, demands and uncertainties need to be accounted for in management and (re)planning of decisions. Establishing platforms for knowledge sharing, capacity building, and global collaboration among researchers, policymakers, and practitioners could lower sociohydrological barriers, facilitate innovation, and the adoption of best practices. For instance, the promotion of conjunctive water use by encouraging groundwater storage and recharge can optimize water resources human uses and enhance overall water security.

**Policy Standardization and Flexibility:** Policy coherence and harmonization could minimize inconsistencies, such as for standardizing and streamlining water year type and environmental flow definitions, categories, and methods. For instance, introducing climate-adaptive water year type approaches could be a first step for better resource allocation. A feedback policy using a 30-year moving percentiles approach could be a first step to enhance e-flow reliability and resilience.

**Adaptive Management and Monitoring for Risk Mitigation:** The adoption of adaptive management strategies requires continuing monitoring to assess and optimize e-flow ecological benefits. This facilitates the dissemination of best practices and guidelines for e-flow implementation based on successful case studies. Investments in advanced forecasting and monitoring systems can help adapt to climate change impacts. Data-driven decisions can enhance water storage and guide appropriate flow releases, such as through the implementation of risk-mitigation strategies. One example is the Forecast Informed Reservoir Operation's adaptive flood space, which can prevent flooding and facilitate decision-making on Flood-Managed Aquifer Recharge for win-win solutions.

**Regulatory and Institutional Reforms:** The need for revising reservoir operations is clear, especially if more dynamic flow managements across different WYTs and seasons are to be adopted. Advocating for policy changes in FERC's long-term licensing process to integrate climate-informed scenarios is crucial. Fostering stakeholder collaboration to overcome resistance and implement sustainable water management policies can be more easily achieved by regularly updated regulatory frameworks to incorporate the latest scientific insights.

**Infrastructure and Hydropower Optimization:** Investing in additional water storage and conveyance infrastructure to address vulnerabilities caused by extreme events can be an option, however, exploring other strategies that optimize water allocation based on science using the current system settings should be prioritized. This includes the adoption of adaptive operational strategies, that can be driven or further explored by coupled water-energy system models for sustainable hydropower generation, such as *CenSierraPywr*.

**Reconsideration of Agricultural and Environmental Water Deliveries:** Human uses are generally prioritized to the detriment of the environment, therefore, trade-offs need to be identified to reach to an appropriate compromise. Nonstationary imposes the need for updating operational planning to adapt to changing hydroclimate volatility. Hence, it is imperative to review and adjust water rights and allocation policies to ensure equitable distribution, especially during dry periods, to enhance e-flow schemes and mitigate the impacts of extreme low flows, particularly in basins with limited storage capacity.

## Appendix S1: Partial satisfaction of the requirements for the degree of Master of Science

# Establishing reservoir surface area-storage capacity relationship using Landsat imagery[c]

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## Abstract

Remote sensing is a powerful tool for tracking surface waterbodies at different scales and spatial and temporal resolutions. The efficient management of water resources, at basin or regional scales, requires the monitoring of water storage in reservoirs. In reservoirs where storage observations are available or a surface water-storage curve exists, storage can be estimated using optical remote sensing. Therefore, this study uses image segmentation to estimate the surface area of three large reservoirs to predict surface water storage volumes. We established a surface area-storage relationship using the Modified Normalized Difference Water Index (MNDWI), produced from Landsat imagery using the Google Earth engine code editor interface, to estimate surface water storage for the New Melones, Don Pedro and McClure reservoirs located in central California. Observed storage showed a very high correlation ($R \geq 0.990$) with remotely sensed surface area estimates. The results show that the modeled storage values derived from fitted equations of the remote sensing methodology were highly correlated ($R \geq 0.993$, p-value $< 0.001$) with observed storage data. Monitoring surface water storage using satellite data is thus demonstrated, as is seasonal and inter-annual variations of storage levels, which are estimated with small errors and high predictive power ($R^2 \geq 0.987$, PBIAS $\leq +0.6\%$, NSE $\geq 0.989$).

**Keywords:** MNDWI, water index, surface water, reservoir volume

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[c] This work is published as a peer-reviewed conference paper in the International Geoscience and Remote Sensing Symposium (IGARSS) as partial satisfaction of the requirements for the degree of Master of Science in Environmental Systems: Facincani Dourado, G., Hestir, E. L., & Viers, J. H. (2022). Establishing Reservoir Surface Area-Storage Capacity Relationship Using Landsat Imagery. In *IGARSS 2022-2022 IEEE International Geoscience and Remote Sensing Symposium* (pp. 863-866). IEEE. https://dx.doi.org/10.1109/IGARSS46834.2022.9884132

## S1.1. Introduction

Water storage facilities require the availability of regularly updated information on reservoir level and capacity for effective basin-wide water resources management [1]. However, storage levels are either not measured or data are not readily available for most reservoirs around the world, due to financial, political, or legal considerations [1]. Considering that, remote sensing can be a useful technology for estimating reservoir storage, as it allows the observation of spatial and temporal surface water dynamics at multiple scales and at regular and frequent time intervals [2]. These observational data can be used to map the extent of water bodies at local, regional, or even global scales [2]. For instance, the volume of water bodies can be estimated using several methods depending on the availability of morphometric and areal data [3]. Water storage in reservoirs, in particular, is a vital parameter for flood control and/or hydropower generation dispatch, in which the reservoir storage curve is a key issue for strategic risk management [4]. Besides that, quantifying surface water dynamics is fundamental for hydrological, biogeochemical and ecological studies in order to provide useful information for reservoir management decision-making [4].

Regarding reservoir storage, the reservoir area-storage curve is an approach extensively accepted to estimate storage fluctuation [5]. The use of mathematical equations relating area and volume are the most often adopted method to determine changes in water volume over time [6]. Satellite have been used to estimate reservoir based on their surface areas. For instance, Moderate Resolution Imaging Spectroradiometer (MODIS) 16-day 250 m vegetation product and satellite altimeter-based estimates of reservoir water elevations have been used to estimate the surface water areas of large reservoirs [7, 8]. Similarly, Landsat imagery has been employed to establish surface area-storage relationships to retrieve information on storage variations in small reservoirs [1,9], which can achieve a high coefficient of determination ($R^2 = 0.95$) [10]. Since optical remote sensing data are widely applied for tracking surface water [4], this study used optical remote sensing to evaluate the feasibility of remote detection and quantification of storage volumes. The objective of this study was to develop a reservoir surface area-storage capacity curve from remote sensing data and observations of stored surface water volumes at varying times.

## S1.2. Methodology

### S1.2.1. Study area

The Stanislaus, Tuolumne and Merced rivers are the main tributaries that flow into the San Joaquin River, one of the two major rivers that form the Sacramento-San Joaquin River Delta in California. The lower watershed of each river has a major low-elevation “rim” dam, each of which are managed for multiple purposes, such as water supply for agriculture and/or urban water supply, hydropower generation, flood control, recreation and environmental mitigation [11]. Water stored at the Don Pedro Reservoir also serves as a water bank for the San Francisco Public Utilities Commission, and all rim dam water releases feed environmental flows into the San Joaquin River, as well as deliveries to local irrigation districts, who operate the facilities [11]. Therefore, these important facilities were selected for this study case. The study period selected comprises the year 1984-2021, in which lake levels varied significantly, more notably during the California droughts (1987–1992, 2007–2009 and 2012-2016). These storage facilities were selected as historical daily storage and elevation measurements were available for the reservoirs, but without local reservoir area observations or surface area-storage curves.

### S1.2.2. Remote sensing index

Remote sensing indices based on the visible and infrared spectrum can provide a direct means to observe variations of reservoir area. The Normalized Difference Water Index (NDWI) have been suggested as an effective approach as it is insensitive to subpixel vegetation component [5]. However, MNDWI is more stable and reliable than NDWI, as the shortwave infrared band used in MNDWI is less sensitive to concentrations of sediments and other optical active constituents within the water column as compared to the near infrared band used in NDWI [2].

MNDWI has been widely used to extract waterbodies easily due to its distinct response in comparison to other land cover features [12]. MNDWI is appropriate for either natural or urban environments, as it can readily differentiate water between vegetation, soil, and built-up areas compared to other indices. For instance, MNDWI has been shown to delineate water from land with better performance than Normalized Difference Water Index, Automated Water Extraction Index, Water Index [2], Normalized Difference Vegetation Index, Normalized Difference Pond Index, Normalized Difference Moisture Index and Normalized Difference Turbidity Index [12]. It is calculated as follows (Eq. 1):

$$MNDWI = (\rho G - \rho SWIR)/(\rho G + \rho SWIR) \qquad \text{(Eq. 1)}$$

where: $\rho G$ = reflectance in the green (Band 3 in the Landsat 8 OLI TIRS, Band 2 in the Landsat 5 TM); $\rho R$ = reflectance in the short-wave infrared (Band 6 in the Landsat 8 OLI TIRS, Band 5 in the Landsat 5 TM). MNDWI varies from -1 to 1, in which water features have positive values, while non-water features have negative values. MNDWI relies on the greater reflectance that water has in the green band, compared to the SWIR, which is opposite of terrestrial features [2]. Therefore, a threshold of zero is often set to segment MNDWI results into water versus non-water pixels. Although snow has a higher reflectance in the visible and infrared wavelengths, MNDWI cannot readily discriminate between snow and water because the index response is similar to the value of water [2]; however, this is not a problem for this study area as snowfall occurs only in the high-elevation upper watersheds [11].

### S1.2.3. Data collection and analysis

The reservoir boundaries at its conservation capacity were extracted from GIS data provided by the California Natural Resources Agency [13]. To avoid that any pixels were missed due to the changes of surface elevation, a buffer of 150 meters was added to reservoir boundaries. To identify the reservoir surface area, Landsat imagery from 1984-2021 were used. Landsat 8 Operational Land Imager (OLI sensor) and Landsat 5 Thematic Mapper (TM sensor) images were selected from the Google Earth Engine code editor interface using the JavaScript API. Selected images used in the analysis have cloud coverage less than 1% over the reservoirs of interest. Daily storage data for New Melones, Don Pedro and McClure reservoirs were retrieved from the USGS database for gauges 11299000 (1927-2009 recorded at 12am), 11287500 (1930-2021 recorded at 5pm) and 11269500 (1930-2020 recorded at 12am), respectively.

Landsat imagery is one the most popular for calculating water indices, due in part to its suitable spectral bands and medium spatial resolution [2] and its widespread availability. Furthermore, Landsat images are already orthorectified, georeferenced and atmospherically corrected to surface reflectance, with low absolute radiometric calibration uncertainties. Training (80%) and validation (20%) data were selected randomly to represent time periods in which the reservoir was at a variety of storage conditions and thus different water surface elevations, with an attempt to characterize lowest to highest historical storage capacity. Then, first- and higher-order polynomial models were fitted to the training data and used for predicting storage values based on the remotely sensed reservoir surface areas using the validation dataset.

Image selection was constrained by cloud cover condition and revisit time of the instrument. Satellite-derived storage estimates were then compared to local observations using the coefficient of determination ($R^2$), Nash–Sutcliffe model efficiency coefficient (NSE), Percent Bias (PBIAS), Root Mean Square Error (RMSE) and Pearson correlation coefficient (R) as goodness-of-fit metrics for model evaluation [14]. Correlation was tested at a significance level of 5% ($\alpha = 0.05$) and relative differences were calculated for each reservoir. Processing algorithms and data analyses were developed and conducted in the R environment [15].

Reservoir water surface areas were determined by extracting pixels with MNDWI values greater than zero (**Figure S1-1**) and summing the total number of classified water pixels by their respective area (~ 900 $m^2$). We assumed that the mismatch of few hours between image acquisition and time of storage recording is insignificant given the large volume and surface area of such reservoirs (27 to over 50 $km^2$).

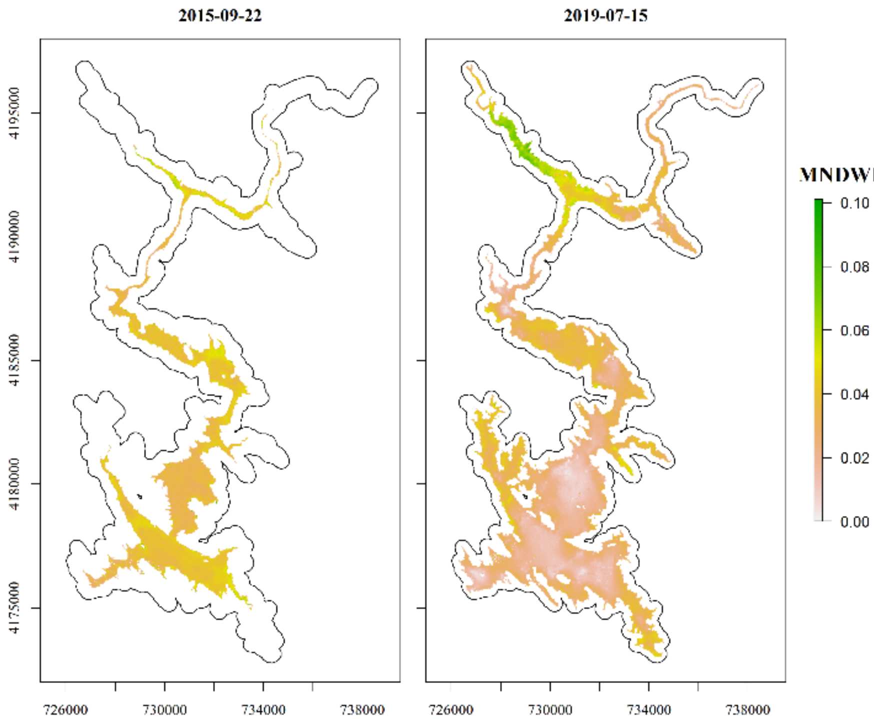


**Figure S1-1.** Don Pedro Reservoir at its lowest (2015) and highest level (2019) in the Landsat 8 imagery, with a 150-m buffer around the reservoir boundaries

### S1.3. Results and discussion

During the study period, water reservoir storage estimates from classified satellite imagery showed a high correlation ($R \geq 0.990$) with water surface area using second-order polynomial regression analysis due to the nonlinear relationship between dependent and independent variables (**Figure S1-2**, left). Higher-order polynomial models did not improve model performance and were not adopted to avoid overfitting. By using the second-order polynomials, almost 100% of the change in storage can be explained by the change in surface area ($R^2 \geq 0.977$). By using this fitted regression model to predict storage values of the validation data, a very consistent linear relationship with the observed storage values, as seen in **Figure S1-2** (right). Predicted water volume estimates showed a very high correlation ($R \geq 0.993$) with the observed storage values, showing a statistically significant correlation even at the significance level of 1% ($p$-value $< 0.01$). These modeled results show that at least 98.7% of the variation in the observed storage can be explained by satellite-estimated reservoir surface area. **Table S1-1** shows model outcomes for each lake, including NSE, which measures model efficiency and predictive power (i.e., perfect prediction is equal to 1). PBIAS values show the general over or underestimation of modeled storage, in comparison to observed data. Lastly, RMSE refers to how far modeled estimates are from the fitted regression line (i.e., normalized residual distance).

Overall, this approach shows measurably good agreement between observed and estimated storage values from mid-resolution remote sensing data (e.g., Landsat derived MNDWI) to map reservoir surface area. As shown in **Table S1-1**, mean bias of estimations is close to zero, though underestimations of -9.8% and overestimations of up to +22.3% were observed. Notably these mismatches occur when reservoirs were at very low storage capacity. Outside these periods of reduced reservoir storage, most observations had differences within ±5%. Although this uncertainty measured as tens of millions of cubic meters of water is non-trivial, these differences in this study are relatively small considering these are large storage facilities. The RMSE estimates show that in general, errors represent approximately 2.4%, 1.2% and 1.1% of the total reservoir water storage for the New Melones, Don Pedro and McClure reservoirs, respectively. Therefore, a generally small bias in this approach should not significantly impact its use for planning for hydropower generation, flood control, water supply deliveries or other management purposes at these specific facilities. The segmented images, when combined with observational data, were used to develop surface area to storage relationships for each of the reservoirs and in turn predicted the seasonal and inter-annual variations in storage capacity. Due to the large area covered by the reservoirs, uncertainties are likely due to complex canyon topography and/or shorelines such that mixed pixels between the land/water interfaces occur. Thus, these estimations could be improved with higher spatial and temporal remote sensing data, such as Sentinel-2 imagery.

**Table S1-1.** Model assessment, and absolute and relative difference of the results for each lake.

| Lake | Capacity (mcm) | NSE | PBIAS (%) | RMSE (mcm) | Median relative difference (range) (%) |
|---|---|---|---|---|---|
| **New Melones** (n = 218) | 2985 | 0.989 | -0.3 | 72.463 | -0.2 (-9.8 – +22.3) |
| **Don Pedro** (n = 317) | 2504 | 0.995 | +0.6 | 29.895 | -0.7 (-3.3 – +2.7) |
| **McClure** (n = 220) | 1238.6 | 0.998 | -0.3 | 13.279 | +0.2 (-4.9 – +5.4) |

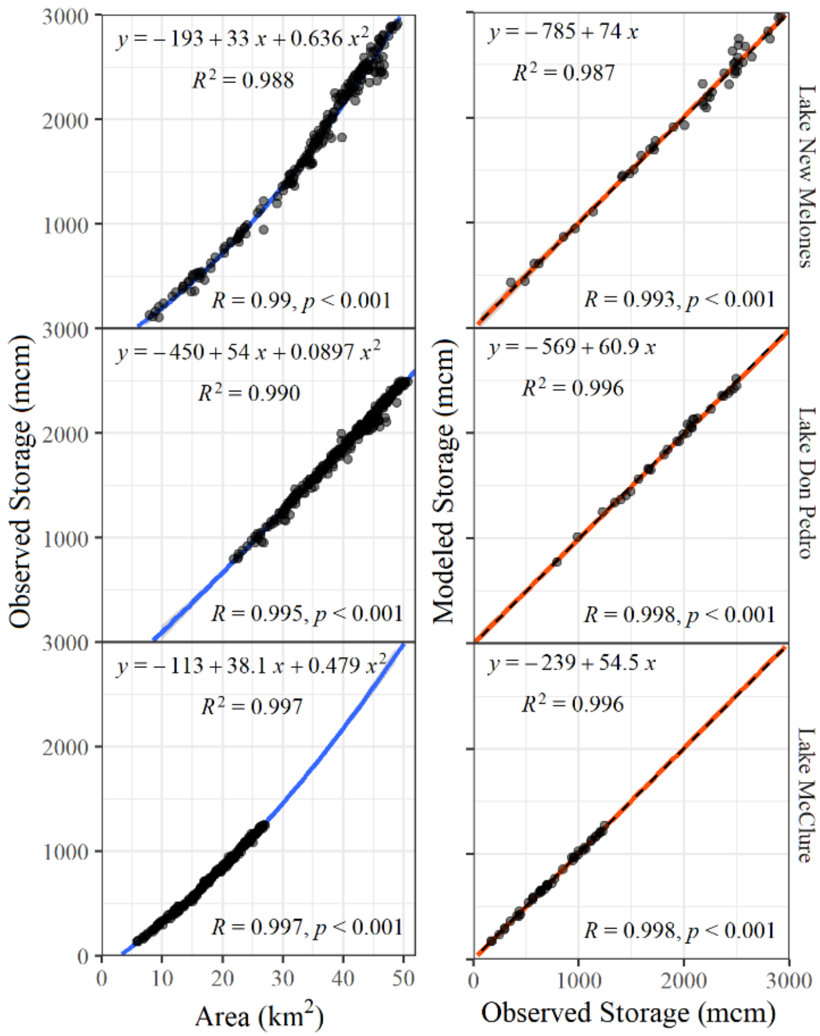


**Figure S1-2.** Regression analyses showing the relationship between remotely sensed surface area (km$^2$) and volume (mcm) for each reservoir for training data (left), as well as fit statistics and parameters of each validation model (right).

### S1.4. Conclusion

In this study, remotely sensed surface water reservoir storage volume estimates were generated using satellite and observational data. Observed reservoir storage values and Landsat images were used to establish surface area-storage relationships for three major reservoirs, by taking advantage of the highly variable inter- and intra-annual water surface elevations found in California [16]. The second-order polynomial regression models produced from this analysis demonstrated an effective means for estimating reservoir storage capacity over time, with high accuracy and relatively low errors. This approach requires storage observations for calibration; therefore, data-sparse regions cannot use optical remote sensing alone. However, other approaches, such as fusing optical imaging of water properties with bathymetric data derived from terrain models or active sensor surveys could be used [6]. Landsat imagery may not be adequate for reservoirs with small surface areas and great hourly variations in elevation, due to the subpixel land-cover components and possible mismatches between observations and image acquisition time. Imagery with higher spatial resolution and acquired at shorter satellite revisit intervals could improve model precision, but at greater computational cost. Going forward, these limitations are likely to be overcome by a combination of cloud computing and new Earth observation satellites [6].

### S1.5. References

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## Appendix S2: *CenSierraPywr*'s details

### S2.1. Data inputs

The main hydrological inputs to the model are runoff outputs from the Variable Infiltration Capacity (VIC) hydrologic model, which will be examined under historical and future climate change scenarios using 10 representative Global Circulation Models (GCMs). Daily gridded runoff data will be used for the simulated historic and future streamflow for the water years 1951-2013 and 2031-2090, produced by Livneh et al. (2015) and Pierce et al. (2016), respectively. These datasets are publicly available at the Scripps Institution of Oceanography server as *NetCDF* files. Similarly, precipitation data for Hetch Hetchy reservoir's region was extracted, due to specific water management rules in the Tuolumne River dependent on precipitation. These datasets were downscaled to a resolution of 1/16° (~6 km) using the Localized Constructed Analogues (LOCA) statistical method by Pierce et al. (2018), since global models have coarser spatial resolutions (100 km or more).

The extract function in the *'raster'* R package (Hijmans, 2020) was applied to extract the data from the *NetCDF* files using a normalized area-weighted approach to downscale the dataset from the VIC cells to the subbasin level. Therefore, the basin's shapefiles were used to extract the runoff data based on the fraction of each cell that is covered by each subbasin (**Figure S2-1**, A). Moreover, these data were preprocessed for bias-correction to account for systematic over- and under-estimations. Bias-correction was performed using the *'hyfo'* R package (Hydrology and Climate Forecasting) developed by Xu (2020). The hydrology from the projected future climate change scenarios was bias-corrected using monthly bias-corrected data from the Scripps Institution of Oceanography. The historical hydrology data was bias-corrected at the basin level using monthly unimpaired flow estimates from the California Data Exchange Center (CDEC) developed by the Department of Water Resources (DWR) (CDWR, 2016); the data was further bias-corrected using US Geological Survey (USGS) data for specific gauges that had at least 10-15 years of data, mostly in the upper subbasins, where most of the precipitation occurs (**Figure S2-1**, B). That caused an over-estimation bias in the total basin hydrology; therefore, the total contribution of the lower watersheds was evenly reduced trough linear scaling due to the lack of gauges in the area, in order to achieve a basin-wide bias close to zero.

For the basin-wide bias correction, the linear scaling method was adopted, in which multiplicative correction factors were applied to the modeled data to reduce the monthly bias. Meanwhile, for the subbasins, bias-correction was performed using the empirical quantile mapping approach, using the gap-filling technique to overcome inconsistent observations. A set of the 10 GCMs most relevant to California were selected due to their representativeness of different global, regional and local spatiotemporal climate patterns, as suggested in the 4th California Climate Assessment (Pierce et al., 2016). The GCMs are forced by the Representative Concentration Pathways (RCPs) 4.5 and 8.5, respectively corresponding to an intermediate scenario where emissions peak around 2040 and then decline, and a "business-as-usual" scenario in which greenhouse gas emissions continue rising throughout the 21st century (Pierce et al., 2018).

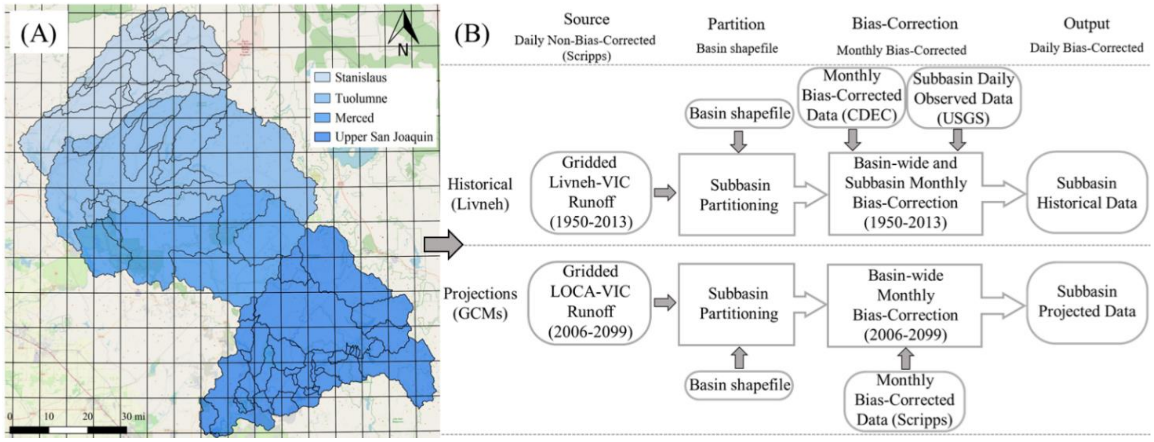


**Figure S2-1.** Diagram showing (A) the overlap of VIC cells over the study area, and (B) the bias-correction process adopted for the historical (1950-2013) and future (2030-2090) runoff data adopted in this study.

Additional data for the optimization module include energy pricing derived from the wholesale energy prices, output produced by the independent energy model Holistic Grid Resources Integration and Deployment (HiGRID), developed by Tarroja et al. (2016, 2019). Currently, 2009 prices are adopted as they generally reflect approximate modern and stable energy demand. The hourly electricity prices were pre-processed to convert them into daily prices to be used in the daily allocation model. The hourly prices require a piecewise linearization to be used by the planning-scale optimization model, therefore, the data is separated into 5 linear blocks (fraction of the day) and equivalent prices for each block to better reflect revenue-driven hydropower generation decisions. For non-optimized basins, operations are simulated following existing operational objectives and regulations while producing hydropower energy as a secondary benefit.

Monthly performance metrics were calculated for model calibration and evaluation. Historical data from USGS gauges and the Energy Information Administration for the water years 1980-2011 were used to assess model performance of instream flows, hydropower flows, water deliveries and reservoir storage, and energy generation, respectively. The main metrics used were the Nash–Sutcliffe efficiency (NSE) and percent bias (PBIAS), besides root mean square error (RMSE), RMSE-observations standard deviation ratio (RSR) (Moriasi et al., 2007) and Kling-Gupta efficiency (KGE) (Gupta et al., 2009).

Average daily values per month were used for comparing the observed data measured by USGS gauges in each location for hydrological data to avoid problems with possible data gaps, meanwhile the electricity generation was based on monthly totals, as that is the temporal resolution of data provided by the Energy Information Administration (EIA) for each power plant. As the basin models are based upon the information provided by the licenses and their updates, certain patterns of operations might differ overtime (due to previous updates) or are not captured due to different operations determined by the operator's decisions or limitations in the hydrology (Livneh data). In the observed data, it is noticeable some facilities presented different patterns of operations at certain times in the past. For instance, some hydropower flows showed a different pattern of operation prior to 1994/1995, certain IFRs showed changes in flow regimes overtime or had flows higher than the minimum requirements delivered downstream. These differences affect the model performance metrics.

## S2.2. Bias-correction overview

The *hyfo* (Hydrology and Climate Forecasting) R package developed by Xu (2020) has been used for bias correcting simulated data in previous studies (Bouabdelli et al., 2020; Cooper, 2019; Mendez et al., 2020; Shen et al., 2020). The bias correction will use the *getBiasFactor*() function to get the bias factors for correcting the simulated data. It can be done in different scales; in this case we are getting monthly bias factors. The inputs are observed and simulated data frames, with the same length, a first column with dates, and a second column with streamflow. Then, the bias factors are applied to get the whole simulated data using the *applyBiasFactor*() function, using as arguments, the bias factors and the simulated data only. Using these two functions can return random errors about the format of the data, asking the columns to be read as date and numeric/double, even when they're already in this format, inputting them using *as.data.frame*(), solves the problem. The *hyfo* package offers different methods for bias correction, including:

• *delta*: This method adds to the observations the mean change signal. It should be avoided to bounded variables as it can produce values out of the variable range (e.g., negative streamflow values).

• *scaling*: The data is corrected by scaling the simulation with the difference (additive) or quotient (multiplicative) between the observed and simulated means in the train period. The *scaleType* argument can be "*multi*" or "*add*", so that the bias factors can be derived for multiplying the simulated data or added to the simulated data. The multiplicative method can be chosen for correcting river flows, as it is indicated for variables with a lower bound and it also preserves the frequency.

• *eqm* (empirical quantile mapping): this method is applicable to any variable, as it's used to calibrate the simulated Cumulative Distribution Function (CDF) by adding to the observed quantiles, the mean delta change and the individual delta changes, in the corresponding quantiles. The extrapolate argument can be set to "*no*", so that the simulated data does not surpass the limits found in the observed data, bounding it to the range of observed, not producing biased extremes. It requires an extra argument ("*obs*") when applying the bias factor. The "*preci*" argument needs to be set to "*FALSE*" when using this method to variables other than precipitation.

• *gqm* (gama quantile mapping): used only for precipitation.

Bias correction is an active area of research; a variety of techniques have been examined, ranging from simple scaling to more complex distribution mapping methods (Cooper, 2019). Bias corrected results can vary by bias correction technique, model, climate output (Miralha et al., 2021), season (Ratri et al., 2019) or even study area (Cooper, 2019). Therefore, it is recommended that bias correction methods be fully documented and results from pre- and post- correction presented (Cooper, 2019). In this case, one problem identified with the multiplicative scaling is that when flows are low in the simulated data, the bias factor can be 5-7 (increasing the flows in 5-7 times), and that causes higher flows in that period to be overestimated. The option "*add*" doesn't cause this problem. However, for correcting streamflow data at the sub catchment level, the *eqm* method provided the best results. According to Mendez et al. (2020), the quantile mapping approach corrects the distribution of the simulated data, so that the variability of corrected data is more consistent with the observed. The authors used this approach to bias correct precipitation data, stating that it non-linearly corrects the mean, standard deviation (variance), quantiles, wet frequencies and intensities preserving the extremes, outperforming methods such as linear scaling, power transformation of precipitation, gamma quantile mapping and gamma-pareto quantile mapping. This method adjusts 99 percentiles and linearly interpolates inside this range every two consecutive percentiles (Miralha et al., 2021). This is a major advantage as the entire distribution matches that of the observations for the training period, while maintaining the rank correlation between models and observations (Mishra et al., 2020). Ratri et al. (2019) also used this method to bias correct daily precipitation data. Mishra et al. (2020) used the *eqm* method to bias correct historical and future simulations of precipitation, minimum and maximum temperatures at the daily time scale.

### S2.2.1. Application to hydrological data

The runoff data were preprocessed for bias-correction to account for systematic over- and under-estimations, using the *hyfo* R package. The hydrology from the projected future climate change scenarios was bias-corrected using monthly bias-corrected data from the Scripps Institution of Oceanography (http://albers.cnr.berkeley.edu/data/scripps/streamflow/). The historical hydrology data was bias-corrected at the basin level using monthly unimpaired flow estimates from the California Data Exchange Center (CDEC) developed by the Department of Water Resources (DWR) (CDWR, 2016); the data was further bias-corrected at the subbasin level using US Geological Survey (USGS) data for specific gauges at the outlet of subbasins. Only gauges that had at least 10-15 years of data with no major gaps, or several sequences of small gaps were used, mostly in the upper subbasins, where most of the precipitation occurs. That caused an over-estimation bias in the total basin hydrology as the Livneh data underestimated precipitation in the upper watersheds; therefore, the total contribution of the lower watersheds was evenly reduced trough linear scaling due to the lack of gauges in the area, in order to achieve a basin-wide bias close to zero.

For the basin-wide bias correction, the linear scaling method was adopted, in which multiplicative correction factors were applied to the modeled data to reduce the monthly bias. Meanwhile, for the subbasins, bias-correction was performed using the empirical quantile mapping approach, using the gap-filling technique to overcome inconsistent observations.

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# Appendix S3: Knowledge gaps, water caps and regulatory traps: Policies, practices, and lessons from environmental flow requirements in the San Joaquin River, California

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## S3.1. Background

The implementation of environmental flows (e-flows) aims to reduce the impacts of river regulation and water diversions, however, many times simplistic, static, minimum flow requirements (MIFs) are adopted (Facincani Dourado et al., 2023). MIFs are often a subjectively determined water level or flow maintained to provide water for in-channel (e.g. environmental allocation) and off-channel uses (e.g. agricultural allocation) (Whipple & Viers, 2019b). The main objectives of maintaining e-flows is for the "protection of water quality" and conservation of "beneficial uses" of water (The San Francisco Public Utilities Commission, 2014).

In California, the allocation of environmental water is primarily aimed at protecting and restoring ecosystems, particularly its rivers, streams, wetlands, and habitats for fish and wildlife. However, in the US, dams built earlier than 1960s were generally not required to account for potential environmental impacts, except for rivers with important runs of anadromous salmon (Kondolf & Yi, 2022). Still, as pointed by Grantham & Viers (2014), several river systems are overallocated, in which the San Joaquin River (SJR) is at the top of the list, with 861% of its natural water yield capacity aimed for human purposes.

The SJR is one of the two main rivers in the state that flow to the Sacramento-San Joaquin River Delta. Since late 1940s, most of SJR's water has been diverted for agricultural uses by the construction of the Friant Dam, and approximately 60 miles of the river ran dry in most years for decades despite the instream flow needs, causing the extinction of a distinct run of Chinook salmon (Kondolf & Yi, 2022). The controversy surrounding Friant Dam prompted numerous studies on the impacts of dams on fish and surrounding environmental resources. This led to a comprehensive engagement with local communities and various stakeholders, including regulatory bodies entrusted with the preservation of fish and wildlife (Kondolf & Yi, 2022). Consequently, after court's ruling leading to mounting legal fees and the possibility of dramatic cuts to water diversions, the SJR Restoration Program was started through a settlement agreement to restore fisheries (Stern & Sheikh, 2021). In addition, other restoration efforts try to reintroduce steelhead and salmon in the SJR and its tributaries through the operation of four fish hatcheries (FERC, 2019b).

**S3.2. Environmental flows in the San Joaquin River**

There are four major basins in the Central Sierra Nevada, California that contribute the most to the lower SJR (Stanislaus, Tuolumne, Merced and Upper San Joaquin rivers) (Facincani Dourado et al., 2022). The basins are mostly highly regulated river systems with high-altitude smaller reservoirs and hydropower facilities, and low-altitude, multi-purpose large storage reservoirs (rim dams) that regulate the flow entering the SJR (Willis et al., 2022). This complex system is operated to capture and control the entire average annual yield of the basins, by several distinct utility companies and agencies, and therefore, regulated differently by each owner/operator in many cases (Maskey et al., 2022).

**S3.2.1. Policies**

In this region, 16 major water and power projects include 28 storage reservoirs and 35 powerhouses, besides many diversion dams, canals, and aqueducts, operated by 12 utility companies, irrigation districts or government agencies. Each project's license prescribes a minimum instream flow requirement (MIF) for 48 stream reaches, in addition to ramping rates (RR), flushing and/or supplemental flows (F/S) and maximum flows (MAF) (Rheinheimer et al., 2022).

Regarding water quantity, MIFs are designed to protect and restore rivers, streams, wetlands, and habitats for fish and wildlife. Conversely, MAFs are implemented in specific areas to prevent erosion, safeguard water quality and riparian habitat, or establish a limit based on the channel's conveyance capacity (FERC, 2003a). F/S may consist of attraction and out-migration pulse flows, which are intended to lure upstream-migrating adult fall-run Chinook salmon and Chinook salmon smolt, respectively (FERC, 2019b). Additionally, ramping rates are used to prevent abrupt fluctuations, ensuring the water flow changes in a controlled and gradual manner (SJRRP, 2017).

Water allocation management decisions are typically determined by the 'water year type' (WYT), which classifies annual runoff at water supply facilities using numerical thresholds based on historic or forecasted hydrologic data (Null & Viers, 2013b). The California Department of Water Resources (CDWR) and/or US Bureau of Reclamation (USBR) generally provide these forecasts. For example, the San Joaquin Valley Index (SJVI) is a WYT index established by State Water Resources Control Board (SWRCB), categorizing WYTs based on historical unimpaired runoff from four basins to allocate water for agricultural and environmental purposes. Various agencies and utilities define and apply different WYTs using five key dimensions, including forecast location, calculation method, time period, calculation date, and validity duration. In relation to water quality parameters, no clear prescriptions are given besides temperature management in a few locations, mostly in major river reaches that feed into the lower SJR.

### S3.2.2. Practices

Water quantity of e-flows is defined in 9 projects, according to 1-6 different WYTs categories are established considering 8 WYT classification systems (FERC, 1959, 1964, 1978, 2003c, 2004, 2006, 2009, 2019a; NOAA, 2009; SCE, 2000; SFPD, 2008; SJRRP, 2017). For instance, the Spring Gap-Stanislaus Project uses only 3 WYT categories; however, 5 categories are described in its license. Some WYTs share similar flow schedules, and 23 out of 48 reaches do not include seasonal flow variations. Most e-flow schedules do not utilize WYT classifications, and those that do, have minimal inter-annual flow variations. Many policies allow reservoir operators to release inflows when natural inflows are below the MIFs, giving them operational flexibility.

With respect to water quality, the SWRCB is responsible for issuing a water quality certification beforehand for the construction and operation of projects, which can still be modified or revoked if monitoring results indicate the violation of water quality objectives or impairment of beneficial uses (FERC, 2006c). Prior to the certification, environmental impact assessment studies conducted may consider hydrologic and hydraulic properties of streams and reservoirs, sediment and nutrient transport, and temperature regimes. For instance, a reservoir with different intake elevations might need a thermal prediction analysis to estimate temperature of water releases at each intake point and determine the type of outlet structure needed for suitable downstream fishery and recreational uses (FERC, 1982a). Parameters both modeled and monitored need to be consistent with all water quality standards and implementation plans adopted or approved pursuant to the Porter-Cologne Water Quality Control Act or section 303 of the Clean Water Act (FERC, 2006a).

However, such policies may bring further complications. For instance, specific temperature standards are mandated for 7 river segments in the region to support the spawning and egg incubation of salmonid fish. One of these is the lower Tuolumne River, which is currently classified as impaired due to higher than acceptable temperatures. Interestingly, the temperature requirements set for salmonids during the licensing process are based on data from populations in the Pacific Northwest (EPA, 2003). This approach may not accurately reflect the local conditions, where the fish populations might be adapted to warmer temperatures compared to those in the northern regions (FERC, 2019b).

### S3.2.3. Lessons

Generally, political and sociohydrological problems regarding human competing uses of water remain the main barrier to implement e-flows (Facincani Dourado et al., 2023). In federal hydroelectric projects, such as Friant Division and New Melones projects in the region, human uses prevailed being authorized by the Congress, meanwhile the non-federal projects, that account for more than half of the total hydropower capacity in the US, are regulated by the Federal Energy Regulatory Commission (FERC) (Office of Energy Projects, 2017). Other example is the Raker Act, passed by the U.S. Congress in 1913 to authorize the construction of the Hetch Hetchy Reservoir in the Yosemite National Park, California, for water and power supply for the City of San Francisco. The means, criteria and regulations for licensing for non-federal and federal projects may question the fairness of this difference, as federal projects may be biased and subject to privileges, such as less rigorous licensing process and provision of subsidies (Perkins, 1997), what might stir up inequity (government failure), and consequently, imperfect competition (new market failure).

FERC does not consider climate change in the licensing process; this information failure (market failure) causes policies to not to have the impact they should to be completely effective, requiring new policies to be created. Therefore, the different water use priorities, e-flow methodologies and authorities involved already make e-flow implementation especially challenging. Added to that, climate change will likely affect facilities operating under different WYT classifications unevenly. Previous studies in the Sierra Nevada assessed the hydrological impacts of climate change on WYT distribution of three classification systems (He et al., 2021; Null & Viers, 2013b; Rheinheimer, Null, et al., 2016). Climate change-induced shifts in runoff volume and timing are expected; therefore, provided that these benchmarks remain unchanged, adaptive water year typing options need to replace fixed thresholds set under the assumption of stationary hydrology. Besides that, most river reaches have no WYT defined, especially in the upper watersheds and the impact of within-basin asynchrony of the inconsistent WYT classifications has not been assessed. Therefore, maintaining e-flows to benefit the environment can be a particular challenge, where competing demands for limited water resources exist.

To find mutually beneficial solutions to complex adjudicatory proceeding related to water rights, voluntary agreements have been negotiated throughout the state among various stakeholders, such as water agencies, environmental organizations, agricultural groups, and government entities. Discussions for the SJR have been halted due to insufficient progress, highlighting the need for increased management flexibility. This could be achieved through feedback policies that are updated based on new data to enhance reliability in e-flow deliveries, such as periodic revisions of water year type classifications and thresholds. Additionally, policy harmonization, by aligning e-flow regulatory requirements, could simplify the problem. Seasonal variations are largely absent in current regulations, hindering river dynamics like flood flows and no-flow events. Flexible regulatory frameworks are essential for adopting alternative solutions, e.g., managing invasive species through no-flow events.

Regarding temperature targets, naturally warmer inflows, diversions and agricultural return flows, for instance, can further impede reaching such targets. When reservoir releases do not meet the objective targets, water managers need to investigate and consult with agencies such as National Park Service, US Forest Service and US Fish and Wildlife Service, regarding potential management modifications (The San Francisco Public Utilities Commission, 2014). In these cases, longer and more complex land and water use planning and riparian conservation can help meet targets, if reasonably considered; e.g. a mature riparian habitat can provide shade to reduce water temperatures sufficiently to maintain cold-water fisheries (e.g., brown and rainbow trout) throughout the river corridors (Kessler and Associates, 2004). Consequently, local irrigation districts have been forced to operate under a provisional license since 2012 (FERC, 2019b).

Other problems the region faces include data gaps on agricultural diversions downstream, forcing certain operators to release extra water in an attempt to avoid unmet e-flows. Furthermore, non-native species, reduction of historical native population of salmonids, extensive flow modification, return flows, altered habitat conditions, water quality and dissolved oxygen, further complicate these challenges (FERC, 2019b). An assessment of alternative policies (e.g., policy harmonization and/or feedback policies) is needed to identify impacts on facility operations and trade-offs caused by the different management and climate change scenarios, for better, proactive and adaptive decision-making in the SJR Basin.

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