frontiers | Research Topics

Women at the frontier of freshwater science

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Women at the frontier of freshwater science

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Editorial: Women at the frontier of freshwater science

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women in science, freshwater science, gender barriers, gender equity, affirmative action, human-ecological systems

Editorial on the Research Topic
Women at the frontier of freshwater science florianne

1 Introduction

The Research Topic "Women at the Frontier of Freshwater Science" presents ten examples of recent contributions by women to freshwater science. In this editorial, we provide an overview of the papers and reflect on our experiences as women freshwater scientists in different continents (e.g., Europe, South America and Australia).

2 Overview of the Research Topic

The coupling of ecological and human systems is an overarching and integrative theme in the papers for this Research Topic. Angela Arthington's challenge paper calls for a global freshwater conservation strategy, with four main priorities: 1) assessment and research; 2) restoration; 3) protected areas; and 4) socioecological science and governance. She expands on Tickner et al.'s (2020) Emergency Recovery Plan for freshwater biodiversity to guide policy responses that "bend the curve" of freshwater biodiversity loss. Her main message is that without shared knowledge, trust, understanding and respectful partnerships in these human–ecological systems it is not possible to live in harmony with nature.

Meghan Halabisky et al. validated the application of the Australian Water Observations from Space (WOfS) algorithm to the Landsat archive for Africa. This enables near real-time spatial data on surface water dynamics, supporting better understanding of Africa's water resource changes and long-term water security.

Five papers focus on water quality and pollution.

- Eugenia López-López et al. investigated water quality changes in Basin of Mexico lakes by comparing historical data from Alexander von Humboldt (a European naturalist who visited the Americas in 1799–1804) with modern data and noted significant declines due to urbanization and land use change.

- Eva Bacmeister et al.'s microcosm study showed that in USA streams, suspended sediment concentration has a positive nonlinear effect on nitrogen uptake, which varies by sediment source and size.
- Jordyn Wolfand et al. modelled contaminants of emerging concern in the Los Angeles River (USA) and reported that increased wastewater reuse reduces contaminant concentrations downstream.
- Katharine Owens et al. combined scientific data and community input in Uganda, Indonesia, and the USA, discovering stakeholder perceptions of pollution closely matched debris measurements.
- Camila Campos et al. studied Brazilian Savanna streams, identifying conductivity as the key factor influencing ecological metrics and highlighting that nonlinear responses need to be considered when setting monitoring guidelines.

Three papers address environmental flows and water use efficiency

- Xiaoying Liu et al. found that environmental water from an irrigation canal helped sustain refuge habitats in ephemeral Thule Creek and boosted productivity in the downstream Wakool River, Australia.
- Christina Morrisett et al. reported that improved irrigation efficiency in Idaho increased crop yields but also raised water use and reduced river return flows, leading them to recommend a holistic management approach.
- Meegan Judd et al. surveyed Australian water managers to determine how uncertainty affects decision making, highlighting that more work is required to establish robust decision-making frameworks for environmental water management.

3 Reflections

The Research Topic provides an opportunity to reflect on diverse contributions women are making to freshwater science. It also invites reflection on broader gender dynamics within the field, for which we draw on our own experiences as women in freshwater science in Europe, Latin America and Australia, and relevant literature.

Women have significantly shaped freshwater science research since the 19th century, contributing important insights into ecology and conservation (Downes and Lancaster, 2020; Togood et al., 2020; Catalán et al., 2023). Today many women are active in freshwater science, including in academia and research; application of freshwater science through policy, planning and management; and leadership of professional associations. The papers in the Research Topic showcase the breadth of their contributions: thought leadership (Arthington), technical advances (Halabisky et al.), foundational research (Bacmeister et al.), applied science (Campos et al., Lopez-Lopez et al., Wolfand et al., Liu et al., and Morrisett et al.), management (Judd et al.) and community engagement (Owens et al.).

Despite underrepresentation, women have recently played key roles in freshwater policy and research in Europe and Australia, with rising productivity and a narrowing publication gender gap. However, persistent gender barriers continue to limit women's full participation and advancement (Downes and Lancaster, 2020; Lester and Rosten, 2020). Fieldwork can present logistical challenges for women, including safety and harassment concerns. Although female enrolment in environmental science programs has increased, women are underrepresented in senior academic and leadership positions, perhaps constrained by the so-called "glass ceiling" effect (Sánchez-Montoya et al., 2016; Lester and Rosten, 2020; Slobodian et al., 2021). In Latin America, these issues are especially challenging and compounded by patriarchal cultural norms, limited institutional support, and political instability (Rico, 1998; Márquez-García et al., 2024).

Affirmative action and positive discrimination are helping to address these imbalances. Sector-specific initiatives in the last decade have included Australia's Peter Cullen Water and Environment Trust "Women in Water Leadership" program (Australia)[1], Brazil's "Ictiomulheres" and "Mulheres na Zoologia" collectives, and the recent establishment of the "Red Latinoamericana de Ictiólogas" as part of the Global Network of Women in Ichthyology.

Collections of papers like this Research Topic support greater recognition of women's scientific contributions. All papers in this Research Topic have a woman as first author, with women comprising 52% of all co-authors across all five continents, a significantly higher proportion than in other *Frontiers* freshwater science Research Topics not specifically targeting women. This Research Topic showcases the breadth of subjects women are tackling, often with integrative and interdisciplinary perspectives (Figure 1). The majority of women contributors to this Research Topic are from the Global North[2], with only five women contributors from the Global South. This reflects the underrepresentation of women from the Global South in the international literature on freshwater science.

1 Peter Cullen Water and Environment Trust 'Women in Water Leadership Program', https://www.petercullentrust.org.au/women-in-water/, viewed 19/07/2025.

2 The United Nations uses the terms 'Global North'; and 'Global South' to refer to the socioeconomic and political differences between developed countries (North) and developing and emerging countries (South) (United Nations Department of Economic and Social Affairs 'What is 'South-South cooperation' and why does it matter?', https://www.un.org/pl/desa/what-%E2%80%98south-south-cooperation%E2%80%99-and-why-does-it-matter, viewed 19/07/2025).

FIGURE 1
Keyword cloud[3] of the papers in the Research Topic "Women at the Frontier of Freshwater Science"; locations were excluded to avoid bias due to few keywords with locations.

4 Conclusion

The collection of papers in this Research Topic provides some examples of the spectrum of contributions made by women to freshwater science. Collaborative work among researchers and scholars with their students coming from different countries and areas of expertise is quite well represented.

Although the gender gap has been narrowing, partly due to affirmative action, barriers persist, particularly in Latin America and in Africa. It is paramount to identify and celebrate the stories and contributions of women in science in general, and in freshwater science in particular, to raise the visibility of our work and affirm our role in shaping a more sustainable world.

Author contributions

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3 Prepared using WordClouds (https://www.wordclouds.com/).

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Grand Challenges to Support the Freshwater Biodiversity Emergency Recovery Plan

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Keywords: biodiversity, ecosystem services, multiple stressors, restoration, protected areas, socio-ecological governance, stakeholders

INTRODUCTION

The year 2021 offers a critical opportunity for concerted action to influence the future of freshwater biodiversity, ecosystem services and human well-being. The United Nations Decade on Biodiversity 2011–2020 has ended, and governments around the world are reviewing major international agreements relevant to biodiversity conservation, including the Convention on Biological Diversity (CBD)[1], the Sustainable Development Goals (SDGs)[2], and the UN Framework Convention on Climate Change (UNFCCC)[3]. A Post-2020 Global Biodiversity Framework[4] is under development, with the grand mission to "Halt the loss of species, ecosystems and genetic diversity by 2030; restore and recover biodiversity to ensure a world of people "living in harmony with nature' by 2050".

Freshwater ecologists have acted quickly to draw attention to the global dimensions of the freshwater biodiversity crisis and address the lack of a comprehensive framework to guide policy responses (Bunn, 2016; Darwall et al., 2018). An Emergency Recovery Plan for freshwater biodiversity, published by 25 authors from 14 organizations (Tickner et al., 2020), sets out six major priorities for global action and policy development to "bend the curve of freshwater biodiversity loss." It has been submitted to the working committees of the Post-2020 Global Biodiversity Framework, and further promoted as a dramatic OUPblog "Bring living waters back to our planet[5]" Comprehensive reviews have since enumerated many research questions, actions and policy refinements needed to "bend the curve" and protect the world's freshwater ecosystems (van Rees et al., 2020; Buxton et al., 2021; Harper et al., 2021; Maasri et al., 2021). Each review cuts across important scientific, societal, management and policy issues.

The purpose of this brief challenge paper is, likewise, to strengthen and support the Emergency Recovery Plan, but in a different way, by advocating a broader package of strategic activities that too often operate in silos, with patchy coverage of the world's freshwater ecosystem types and biogeographic diversity. This package presents traditional areas of scientific and societal activity that require more strategic, integrated and collaborative global effort to deliver evidence-based freshwater conservation outcomes, conjoined with terrestrial and estuarine/marine conservation, depending on context: (i) inventory, evaluation and research; (ii) restoration and rehabilitation; (iii) protected area design and management; and (iv) socio-ecological science and governance. The paper is intended to motivate greater interest, commitment and collaboration of all stakeholders in the most urgent and ambitious conservation enterprise of the next decade—to protect and sustain freshwater biodiversity in the socio-ecological systems of the Anthropocene.

[1]https://www.cbd.int/convention/guide/?id=web4
[2]https://sustainabledevelopment.un.org/sdgs
[3]https://www.iucn.org/theme/global-policy/our-work/united-nations-framework-convention-climate-change-unfccc
[4]http://www.fao.org/forestry/48209-0cb7240cc9f200dcf507a40e71c39a591.pdfs
[5]https://blog.oup.com/2020/09/bring-living-waters-back-to-our-planet/

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INVENTORY, EVALUATION AND RESEARCH

Evidence-based ecosystem restoration and biodiversity protection depend upon a credible foundation of scientific and sociological data, process understanding and a capacity to model, predict and evaluate ecological/societal outcomes from natural processes, pressures and management actions. Notwithstanding a huge body of erudite freshwater research, there remains an ongoing need to increase understanding of the biodiversity, biophysical processes and ecosystem services of the world's freshwater and connected terrestrial and estuarine/marine ecosystems. The IUCN Commission on Ecosystem Management has developed a globally consistent, spatially explicit Ecosystem Typology for conservation purposes (Keith et al., 2021). It is designed to help identify the ecosystems most critical to conservation of biodiversity and supply of ecosystem services, as well as structuring global risk assessments for the Red List of Ecosystems and reporting against CBD and SDG targets and other framings. The typology distinguishes 28 natural freshwater ecosystem types within subterranean systems, palustrine wetlands, streams, rivers, freshwater and saline lakes, artesian springs, oases, and transitional waters (fjords, estuaries, intermittently closed and open lakes and lagoons–ICOLLS).

Depending on ecosystem type, geography and knowledge gaps, freshwater inventory and research is traditionally integrated around taxonomy, genetics and organismal biology, population and community ecology, and ecosystem functions, the latter including the processes that link landscapes, connected boundary systems (riparian areas, floodplains, wetlands/lakes, and groundwater systems) and freshwater ecosystems (Geist, 2011; Reis et al., 2017; Flitcroft et al., 2019). Likewise, the pathways and processes that connect rivers and estuaries via surface flows and submarine groundwater discharges are vital dimensions of interconnected freshwater and coastal ecosystems. The IUCN Ecosystem Typology provides a geographic framing and scientific resource to help guide priorities for basic inventory and ecological research on understudied ecosystem types and biogeographic regions. For example, groundwater-dependent ecosystems such as artesian springs and oases are relatively poorly studied but coming to attention globally (Cantonati et al., 2020). Intermittent rivers and ephemeral streams (IRES) and episodic arid-zone floodplains are of growing interest because even when dry they perform multiple ecosystem services that complement those of nearby perennial rivers (Datry et al., 2018). Given the exceptional biodiversity of the Amazon Basin and poor knowledge of many aquatic taxa (e.g., migratory fishes), there is an outstanding need for inventory, knowledge synthesis and risk assessment to guide recovery and conservation (Duponchelle et al., 2021).

Innovative biodiversity assessment techniques (remote sensing, GIS, environmental DNA, camera traps, sound recordings, radiotelemetry) can be integrated with established field methods to document biodiversity patterns and hotspots, and track flagship, umbrella and endangered species of high conservation value (Harper et al., 2021). Systematic reviews, meta-analysis, natural and laboratory experiments and modeling offer scope to relate biodiversity patterns and processes with dominant environmental drivers (climate, hydrological regime and water quality, etc). Broad stakeholder engagement is essential across the spectrum of biodiversity inventories, identification of knowledge gaps and research priorities, evaluation of ecosystem services and formulation of targets for restoration and protection of species, ecosystem processes and valued services.

RESTORATION AND REHABILITATION

The major threats to freshwater ecosystems have been comprehensively synthesized in six main categories: hydrological alterations, habitat degradation and loss, pollution, overexploitation, invasive species, and climate change (Dudgeon et al., 2006). These have been mapped at global scale (Vörösmarty et al., 2010; Reis et al., 2017; Grill et al., 2019), elaborated as new pollutants and configurations of stress emerge (Reid et al., 2019) and widely publicized (Bunn, 2016; Flitcroft et al., 2019). Yet despite prodigious management efforts, biodiversity loss and ecosystem degradation continue, creating huge deprivation for millions of people whose diets and livelihoods depend directly on freshwater biota (Lynch et al., 2016). Biodiversity decline has significant implications for ecosystem resilience, recovery potential and adaptation to climate change.

The Emergency Recovery Plan offers a blueprint focused on reducing biodiversity decline and recovering from these major threats, a well as a new threat category on connectivity to highlight the implications of habitat fragmentation for freshwater biota and ecosystems (Grill et al., 2019). Numerous methods and sound protocols already enable mitigation of these major threats, as demonstrated in successful ecological restoration projects around the world (Palmer et al., 2005). For example, the restoration of connectivity patterns and processes has contributed to recovery of biodiversity and ecosystem processes in many regulated rivers (Horne et al., 2017; Opperman et al., 2019). The bolder objective of the Emergency Recovery Plan is to transition from local freshwater restoration successes to a strategic approach that achieves biodiversity and ecological recovery at larger spatial scales. The European Water Framework Directive[6] offers one well-established jurisdictional framing for freshwater ecosystem recovery to good ecological status. Building on European case studies, challenges and successes under this and other directives, van Rees et al. (2020) extend the ideas of the freshwater Emergency Recovery Plan into 15 special recommendations with potential to protect freshwater life globally.

Beyond the main categories of threat to freshwater biodiversity and ecosystems lie new kinds of stress and new configurations of familiar stressors (Reid et al., 2019). Many, if not most, freshwater ecosystems are affected by several types of stress that interact, often with effects greater than (synergism), less than (antagonism) or equal to the sum of their individual effects (Sabater et al., 2018). The daunting scientific challenge is to identify the most significant causes of stress

[6]https://ec.europa.eu/environment/pubs/pdf/factsheets/wfd/en.pdf

and define the most beneficial blend, geographic placement and timing of management actions (Omerod et al., 2010; Craig et al., 2017). This approach has worked reasonably well for the urban stream "syndrome" (Sheldon et al., 2012; Booth et al., 2016). Other multiple-stressor syndromes that threaten freshwater ecosystems include irrigated agriculture, forestry, mining, energy production, transport systems and the recreation and tourism sectors. Climate change, itself a complex mix of stressors, already compounds multiple stressor syndromes (Sabater et al., 2018), by altering river flow and flooding regimes, while rising temperatures are driving higher evaporation rates, water scarcity, and aquatic habitat loss. Shifting climatic regimes intensify the urgency of multiple stressor research and adaptive management solutions.

In multiple-stressor contexts, Tickner et al. (2020) recommend the assembly of "strategic portfolios of measures" rather than relying on interventions that address individual stressors, although these will always be necessary in particular contexts. Methods for mapping individual and cumulative stressors are well-developed (e.g., Vörösmarty et al., 2010), and analytical tools for prioritizing ecological restoration among sites in multi-stressor landscapes are emerging (Hermoso et al., 2015; Neeson et al., 2016). Strategic portfolios of restoration measures require development of cause-and-effect relationships to understand and predict the responses of species and communities to individual and multiple-stressor configurations. Maasri et al. (2021) recommend assessment of restoration outcomes using large-scale replication of before-after-control-impact (BACI) designs, and long-term post-monitoring phases. Relatively few restoration projects meet these stringent design and monitoring requirements (Palmer et al., 2005; Geist and Hawkins, 2016). Meta-analyses of results from post-monitoring can help to identify restoration failures (often under-reported, Geist, 2011) as well as successes, extract learnings and guide adaptation toward more effective strategies.

In many situations with a long history of anthropogenic stress it is important to be realistic about the potential for restoration of near-natural ecological systems (Geist and Hawkins, 2016). Rehabilitation or remediation to recover and sustain selected ecosystem values and species may be the only feasible approach, especially where novel ecosystems with well-established alien species have replaced natural system structures, biodiversity and processes, as in many impounded rivers and degraded floodplain wetlands (Acreman et al., 2014; Poff et al., 2017). These novel circumstances require careful development of explicit and realistic targets for the recovery of the system at project onset (Geist, 2011, 2015; Geist and Hawkins, 2016). A framing termed Strategic Adaptive Management (SAM) offers a structured step-wise process from development of a shared vision and hierarchy of objectives linked to management actions, monitoring, evaluation and publication of outcomes (Kingsford et al., 2021). It amply meets the criteria for measuring restoration and management success from an ecological perspective (Palmer et al., 2005) and provides a powerful model of effective stakeholder collaboration. Broad stakeholder engagement throughout project design, implementation and monitoring strengthens comprehension of the multiple challenges of ecosystem restoration, and encourages appreciation of what can be achieved and is worthy of investment.

Freshwater ecosystem restoration, rewilding, rehabilitation and remediation are technically feasible with existing and emerging technologies, collaborative human commitment and adequate resourcing. The IUCN ecosystem typology provides a template for identification of risks and restoration priorities at global scale. As an example, severe threats to freshwater biodiversity in the Amazon Basin (overexploitation, deforestation, extensive hydroelectric dam development and climate change) demand a portfolio of recovery actions (Duponchelle et al., 2021) and spatially explicit prioritization of future hydropower developments to minimize loss of aquatic connectivity and biodiversity (Winemiller et al., 2016).

PROTECED AREA DESIGN AND MANAGEMENT

Ecosystem restoration is challenging, expensive and may require decades of sustained effort to maintain the desired outcomes. Prevention of biodiversity loss is a far better option than struggling for cures. Perfectly located, designed and managed freshwater protected areas (PAs) represent the pinnacle of global conservation policy. Many categories of area-based protected ecosystems (IUCN I–VI PAs, Ramsar list of Wetlands of International Importance, private protected areas, landholder covenants, indigenous stewardship) play significant roles in freshwater biodiversity conservation. In 2010, the Convention on Biological Diversity (CBD) included an area target of 17% protection for inland waters. However, 70% of river reaches (by length) have no protected areas in their upstream catchments, and only 11.1% (by length) achieve full integrated protection (Abell et al., 2017). Seasonal inland wetlands represent ∼6% of the world's land surface, yet around 89% are unprotected by IUCN PAs and Ramsar sites (Reis et al., 2017).

Urgent calls for increased protection of freshwater ecosystems and biodiversity include free-flowing rivers (Perry et al., 2021), river-wetland mosaics (Reis et al., 2017), springs (Cantonati et al., 2020) and other groundwater-dependent ecosystems, as well as integrated terrestrial-freshwater-estuary/marine protection coordinated across spatial scales, jurisdictions and sectors (Abell et al., 2017; Leal et al., 2020; Buxton et al., 2021). Systematic conservation planning offers data-driven methods for prioritizing restoration and protected area strategies (Abell et al., 2017; Linke et al., 2019). Applications of these approaches have addressed vital issues for freshwater conservation planning (source catchment condition, dimensions of river connectivity, integrated river, wetland and aquifer protection, threatening processes, species distribution shifts under climate change, and trade-offs between freshwater biodiversity conservation and human water requirements). Other tools that can aid similar spatial analysis, provide insights into trade-offs, and inform strategic multi-objective decision-making include pareto-optimal assessments (Hurford and Harou, 2014), Strategic Environmental Assessment (Lazarus et al., 2018) and system-scale infrastructure planning

(Winemiller et al., 2016; Opperman et al., 2019). Significant improvements in the placement, spatial configuration and connectivity of protected areas are feasible using these techniques.

Recent studies have sought to evaluate the benefits of freshwater protected areas for conservation of freshwater biodiversity. A systematic review found that only 51% of 75 case studies demonstrated beneficial outcomes relative to comparable unprotected areas (Acreman et al., 2020). Activities within and external to protected areas were held responsible, including landscape modifications, riparian loss, alterations to hydrological regimes, loss of floodplain connectivity, habitat alterations, chemical contamination, fishing, harvesting (e.g., turtle eggs) and the presence of non-native species. Over-exploited and degraded protected areas add to the burden of ecosystem restoration and recovery facing many societies. Ecological principles and guidelines for improved use, management and monitoring of freshwater protected areas and their surrounding landscapes warrant far wider appreciation and application (Finlayson, 2018; Acreman et al., 2020).

Strengthening the conservation benefits of freshwater protected areas requires engagement and collaboration among scientists, management agencies and the people who visit, know and use these areas. Increased public engagement, citizen science and participatory monitoring of trends in condition or species abundance by committed stakeholders can raise the profile of freshwater biodiversity and help to change behaviors that might otherwise lead to ecosystem damage. Positive socio-economic outcomes as well as biodiversity conservation are important, and more likely to occur when PAs adopt co-management regimes (e.g., fisheries), empower local people, reduce economic inequalities, and maintain cultural and livelihood benefits (Oldekop et al., 2016).

SOCIO-ECOLOGICAL SCIENCE AND GOVERNANCE

Freshwater ecosystems and their catchments are increasingly viewed as coupled human and natural systems, wherein setting objectives and devising management solutions, require engagement and collaboration among engineers and hydrologists, ecologists, social scientists and citizens (Bunn, 2016). This has been advocated and implemented in the field of environmental water management for decades (Poff et al., 2003, 2017) and is a strong element of The Brisbane Declaration and Global Action Agenda on Environmental Flows (Arthington et al., 2018; Anderson et al., 2019). Ecosystem-based Management (EBM), also referred to as the ‘Ecosystem Approach’, jointly considers societal and ecological goals and scenarios in an impressive modeling framework (Langhans et al., 2019). The EBM and similar framings (e.g., SAM) recognize the need for coupling of social and ecological systems, and engagement of *all* stakeholders. The concept of “stakeholders” has often meant token representation of indigenous, marginalized or poorly recognized societal groups. Yet increasingly, solving complex conflicts about water use and management, especially in times of scarcity and uncertainty, requires collaboration and enduring partnerships among all stakeholders with indigenous, societal and scientific knowledge, technical expertise, and credentials at all levels of governance.

Recent reviews consistently call for improved practices to enhance communication, understanding and respect for different “ways of knowing,” and methods for blending of stakeholder knowledge (especially indigenous knowledge) with conventional science (Anderson et al., 2019; Buxton et al., 2021; Maasri et al., 2021; Perry et al., 2021). Others call for evidence-based and targeted guidance to facilitate working with the complex dynamic interactions of ecological and societal systems (Harper et al., 2021). The framing termed Coupled Human and Natural Systems (CHANS) is especially relevant. It proposes strategic integration of patterns and processes that connect human and natural systems, as well as within-scale and cross-scale interactions and feedbacks between human and natural components of such systems (Liu et al., 2021). Interesting applications to freshwater systems include evaluation of water availability, use, quality, management and governance in Canadian agricultural watersheds (Liu et al., 2019) and fisheries management (Lynch and Liu, 2014).

CHANS, SAM and EBM embrace important principles of socio-ecological collaboration and governance, including building trust, maintaining respectful interactions, upholding rights, embracing mutual understanding, and development of enduring partnerships. These integrated socio-ecological frameworks and partnership models offer fundamental tools to guide understanding and management of increasingly degraded Anthropocene ecosystems, in which societal and ecological processes are deeply entwined and interact. Socio-ecological systems in turn require participatory management and governance regimes that can foster biodiversity conservation alongside societal benefits and social justice. For example, a “Just Aquatic Governance” framework has been proposed for the Amazon Basin, based on three pillars of social justice: recognitional, procedural and distributional (Lopes et al., 2021). The need for inclusive socio-ecological freshwater science and governance is particularly acute in the biodiverse, multicultural Amazon Basin (Castello, 2021).

CONCLUDING COMMENTS

The Post-2020 Global Biodiversity Framework is visionary and compelling, and especially relevant to the recovery of freshwater biodiversity—the most overlooked and urgent conservation challenge of the next decade. The IUCN has distinguished 28 global freshwater ecosystem types, a powerful framing for activities to promote the recovery and conservation of freshwater biodiversity. This challenge paper supports the freshwater Emergency Recovery Plan by promoting a broader package of strategic activities that too often operate in silos, with patchy coverage of the world’s freshwater ecosystem types and biogeographic diversity and cultural heritage.. This portfolio urges integration of biodiversity inventory and basic ecosystem science, stressor assessment and mapping with systematic restoration and protected area management in a strategic global freshwater conservation strategy, with links to terrestrial and estuarine/marine realms as required. An overarching and integrative theme is the coupling of ecological and human systems, and the importance of collaboration among all stakeholders with indigenous, societal and scientific knowledge, technical expertise, and experience with governance models and policy development. There is an urgent need to build shared knowledge, trust, mutual understanding and enduring respectful partnerships in coupled human-ecological systems if we want a world of people "living in harmony with nature."

AUTHOR CONTRIBUTIONS

The author confirms being the sole contributor of this work and has approved it for publication.

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Setting thresholds of ecosystem structure and function to protect streams of the Brazilian savanna

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Freshwater environments are among the most threatened by human activities, consequently, their ecosystem structures and functions are targets of significant transformations. It makes monitoring an essential tool in the management of these environments. Ecological metrics have been proven to be effective in monitoring programs aimed at assessing freshwater ecosystem integrity. Structural and functional aspects of the ecosystem may allow for a comprehensive view of the multiple human impacts that occur at different scales. However, a gap in the effective use of such ecological tools lies in the identification of the relative importance of different mechanisms that cause impacts and the interactions between them. Using Boosted Regression Tree (BRT) models, we evaluated the relative importance of natural and human impact factors, from local to catchment scales, on metrics related to diatom and macroinvertebrate assemblages and ecosystem processes. The study was carried out in 52 stream reaches of the Brazilian savanna in central Brazil. Conductivity was the most relevant factor to explain the variation of ecological metrics. In general, macroinvertebrate metrics and algal biomass production responded to both water quality and land use factors, while metrics of diatoms and microbial biomass responded more strongly to water quality variables. The nonlinear responses allowed the detection of gradual or abrupt-changes curves, indicating potential thresholds of important drivers, like conductivity (100–200 µS cm^{-1}), phosphate (0.5 mg L^{-1}) and catchment-scale urbanization (10–20%). Considering the best performance models and the ability to respond rather to stress than to natural factors, the potential bioindicators identified in the study area were the macroinvertebrates abundance, the percentage of group Ephemeroptera/Plecoptera/Trichoptera abundance, the percentage of group Oligochaeta/Hirudinea abundance, the percentage of genus *Eunotia* abundance, the Trophic Diatom Index and the algal biomass production. The results reinforced the importance of consider in the national monitoring guidelines validated ecological thresholds. Thus, maintaining the balance of aquatic ecosystems may finally be on the way to being achieved.

KEYWORDS

ecosystem integrity, boosted regression tree, ecological metrics, freshwater management, monitoring programs

1 Introduction

Freshwater ecosystems are among the most threatened by human activities (Gatti 2016). The knowledge of the various components of these ecosystems is of paramount importance to the elaboration of public policies on conservation or recovery (Bunn et al., 2010). Biomonitoring data has been increasingly used in determining the ecological conditions of aquatic environments, in addition to the traditional physical and chemical indicators of water quality (Leese et al., 2018; Pardo et al., 2018; Gieswein et al., 2019). A comprehensive ecosystem integrity assessment should consider both structural and functional characteristics (Bunn & Davies 2000). While the structure of an ecosystem comprises physical and chemical attributes related to water quality, composition of biological assemblages and habitat conditions, its functioning is related to the processes regulating energy and matter fluxes (Tilman et al., 2014).

The most commonly used metrics to assess freshwater ecosystem integrity are those related to biological assemblages, such as species richness and diversity, abundance, the proportion of tolerant and sensitive taxa, organismal traits (e.g., feeding habits, body size, mobility), and indices of sensitivity to pollution (Hering et al., 2006). Macroinvertebrates, diatoms, macrophytes and fish are often used for that purpose (Son et al., 2018; Waite et al., 2019) as they are robust to the identification of several human disturbances and present particular features that facilitate such application (e.g., life cycle, habitat, size; Merritt & Cummins 1996; Kelly et al., 2008). Much less explored in the context of biomonitoring are aquatic fungi and bacteria, which are key decomposers of organic matter in streams. The responses of some ecosystem processes to stressors are fundamental to understanding the effects of human disturbances on ecosystem services that produce direct benefits to people. But despite this, there is still a lot of reluctance among managers and little use of functional indicators (e.g., litter decomposition) in monitoring programs (Schiller et al., 2017).

Although many studies have pointed out to the applicability of several ecological metrics for assessing freshwater ecosystem integrity, the main gap lies in the relative importance of different mechanisms that cause impacts and the interactions between them (Wenger et al., 2009). According to Sutherland et al. (2013), one solution is the use of modelling as a tool for measuring and monitoring systems. In the context of environmental management, most models used in monitoring programs consider biological assemblages, especially benthic invertebrates (AUSRIVAS, Smith et al., 1999; RIVPACS, Wright et al., 1984; USEPA, 2016), as indicators. Some studies suggest the use of ecosystem processes for this purpose (Gessner & Chauvet 2002; Feio et al., 2010; Woodward et al., 2012), and rare are those that present a multi-metric approach using structural and functional aspects (but see Castela et al., 2008; Clapcott et al., 2014).

In Brazil, as in many other tropical countries, monitoring programs focus on physical and chemical variables of water, with a substantial gap in knowledge about the structure and functioning of aquatic ecosystems. The Brazilian savanna (Cerrado) is a global biodiversity hotspot (Myers et al., 2000), and its headwaters are responsible for 70% of all water supply to other Brazilian regions (Lima & Silva 2007). However, the devastation of Cerrado has been taking place at levels proportional to its ecological and social relevance (Strassburg et al., 2017). It is urgent to know the behaviour of these threatened ecosystems through two valuable management tools: 1) the identification of variables that respond strongly to anthropogenic impacts than to natural variations, and 2) the identification of ecological thresholds that are adopted as standards in monitoring programs. We define the term 'ecological threshold' as a point along a stressor gradient where the relationship between the stressor and an ecological indicator shows an abrupt change in the response curve that can be ecologically explained and significantly relevant for management (Wagenhoff et al., 2017).

In this context, Boosted Regression Tree (BRT) models have been used as a robust tool to identify the influence of environmental variables, natural or those related to human activities, on ecological metrics, making it possible to evaluate the shape of the responses and to make forecasts by using new data (Clapcott et al., 2012; Waite et al., 2019). This approach allows the identification of gradients, from which is possible to detect non-linear responses, interactions among predictors and potential threshold zones. However, identification of potential thresholds alone is not helpful for management if it does not accompany by an analysis of their ecological consequences relevant to management decisions. Furthermore, it is important to consider a group of non-redundant variables so that the impacts of different stressors on the structure and function of ecosystems are detected. This approach would lead to more robust in-stream objectives and provide options for adopting goals that protect the aspects of ecosystems that people value most (Wagenhoff et al., 2017). Several studies have used BRT models to identify thresholds in a diversity of ecological areas, highlighting the potential of this tool to environmental management (Davis et al., 2019; Giri et al., 2019; Wherry et al., 2021).

This study evaluated how ecological metrics respond to natural environmental gradients and human-related stressors (local and catchment) at different spatial scales. We also identified the most suitable metrics to be used as indicators of stream integrity and assessed the response and potential thresholds of ecological metrics along environmental gradients to inform the ecological management of Cerrado freshwaters.

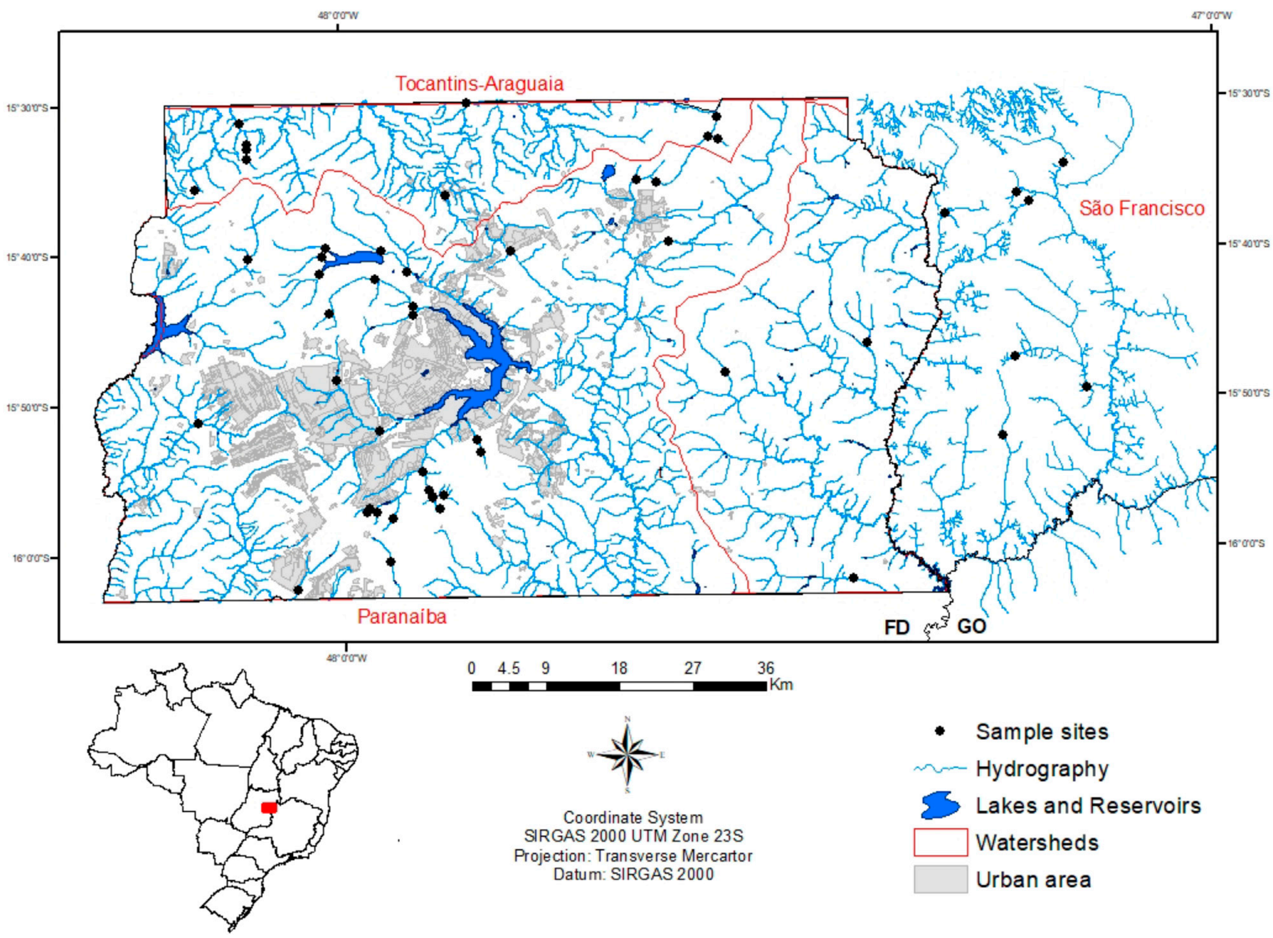

FIGURE 1
Spatial distribution of the 52 stream reaches in the regional river network. The red limits represent the portion of the national watersheds (Tocantins-Araguaia, São Francisco and Paranaíba) located in the study area (Federal District and Goiás) in central Brazil.

2 Materials and methods

2.1 Study area

The study was conducted in the central Brazilian plateau (*ca.* 1,000 m a.s.l.) in an area of approximately 6,700 km^2 dominated by Cerrado (Brazilian savanna) vegetation. Fifty-two stream reaches were selected to represent a broad range of natural environmental conditions (Figure 1). Briefly, sample sites were chosen to represent regions with different land uses and watersheds with different natural characteristics; accessibility for sampling was also taken into account for site selection. When more than one reach was sampled in the same stream, they were at least 500 m apart from each other (to reduce their spatial dependence) and comprised different natural characteristics. All streams are wadeable and perennial of up to 5th order (Strahler 1957).

2.2 Sampling, analysis and metrics

Two sampling campaigns were conducted in 2018, one at the end of the wet season (April/May)—which is from November to March—and a second at the end of dry season (August/September)—which is from May to September.

2.2.1 Predictor variables

A large number of variables related to natural conditions and human stressors were previously measured at each stream reach (Campos et al., 2021). From this dataset, we retained only uncorrelated variables (absolute Pearson's $r < 0.6$) which include natural characteristics (drainage area, elevation, riparian shading, and percentage of organic matter and coarse sediments in the riverbed), water quality variables commonly used in monitoring programs and considered as indirect indicators of human disturbances (dissolved oxygen, conductivity, turbidity, nitrate,

TABLE 1 Description, average and range (minimum and maximum) of natural and human disturbances variables. (*) Data collected four times, but for analysis, we consider the average between April/May and August/September. (**) For categorical variables, we indicated the number of samples in each category.

Variables	Description	Average (min-max)	Category—number of samples**
drai_area	Drainage area upstream of the sample site (Km^2)	40.52 (2.21–215.42)	
elevation	Altitude of the sample site relative to the sea level (m)	1,015 (744–1,220)	
shading	% of riparian shading (0 = 0%; 1 = < 30%; 2 = between 30 and 60%; 2 = > 60%)		0–3; 1–9; 2–7; 3–33
OM	% of organic matter in the riverbed sediment	6.15 (0.61–26.66)	
coa_sed	% of coarse sediments (>2000–710 mm) in the riverbed sediment	60.49 (4.19–97.28)	
DO*	Dissolved Oxygen (mg L^{-1})	7.11 (1.88–8.85)	
cond*	Electrical conductivity (µS cm^{-1})	56 (1–584)	
turb*	Turbidity (NTU)	8 (0.04–197)	
NO_3^{-}*	Nitrate (mg L^{-1})	0.35 (0–10.29)	
PO_4^{-3}*	Phosphate (mg L^{-1})	0.23 (0–6.26)	
RIP_urb	% of urban area in the riparian corridor	1 (0–33)	
RIP_agr	% of agricultural and livestock areas in the riparian corridor	8 (0–56)	
CAT_urb	% of urban area in upstream catchment	7 (0–70)	
CAT_agr	% of agricultural and livestock areas in upstream catchment	21 (0–86)	
CAT_mod	% of modified area in upstream catchment (allotment, exposed soil, eucalyptus)	3 (0–39)	
SR**	Presence (1)/absence (0) of point-source treated sewage release upstream		0–49; 1–3
Dam**	Presence (1)/absence (0) of dams upstream		0–39; 1–13

and phosphate), and primary sources of human disturbances (urbanization and agriculture in the catchment area and in the riparian corridor, other uses in the catchment, presence of upstream point-source sewage release and dam) (Table 1). All of them will be considered hereinafter as predictors. The season (wet and dry) was also considered as a predictor since it may affect some of our biological response metrics.

2.2.2 Response metrics

A large number of ecological metrics were considered in this study (Table 2). The structural metrics are related to the diatom and macroinvertebrate assemblages' composition. The functional metrics include relevant ecosystem processes such as leaf litter decomposition (microbial and total), sediment respiration and algal and microbial biomass production.

2.2.3 Biological assemblages sampling

Macroinvertebrates were sampled using a *surber* (0.09 m^2 area and 0.25 mm mesh size) to collect five sub-samples per site covering the proportional diversity of habitats. The sub-samples were then integrated and preserved in 96% alcohol to be sorted and identified under a stereomicroscope to the lower taxonomic level possible (until family). Diatoms were sampled from five 10 × 10 cm pieces of artificial substrates (slate stones) that were incubated in the riverbed for approximately 30 days. Nearly 250 cm^2 were scraped and the shaved material was preserved in vials containing 0.33% Lugol solution. The identification and quantification of the organisms were carried out under an inverted microscope (Utermöhl 1931). Identification of macroinvertebrates and diatoms was carried out mostly to family and species level, respectively, with the assistance of taxonomic specialists (see Acknowledgments).

2.2.4 Biological assemblage metrics

We considered in this study metrics related to the structure and sensitivity to pollution of diatom and macroinvertebrate assemblages. The structure was composed of richness, abundance, diversity (Shannon-Wiener, Simpson), and evenness (Pielou) indices. The percentage abundance of pollution-sensitive taxa was calculated for the diatom genus *Eunotia*, for diatoms, and for the macroinvertebrate orders Ephemeroptera/Plecoptera/Trichoptera (EPT) and the Plecoptera order alone. The percentage abundance of pollution-tolerant taxa was calculated for the diatom species *Nitzschia palea*, and for the macroinvertebrate classes Oligochaeta and Hirudinea.

Some pollution sensitivity indices were adapted for diatoms and macroinvertebrates. The TDI (Trophic Diatom Index) was adapted from Kelly (1998). Although this index has been developed in Europe, it has the most complete species list. Only 8 of the 74 species identified were not described in the TDI list, hence we attributed the lowest value 1) to them, not to have too much influence on the result. The Biological Monitoring Working Party (BMWP) was adapted from four BMWP indices developed in different regions. The main reference was Monteiro et al. (2008), followed by Junqueira & Campos (1998), Uherek & Gouveia (2014), and Alba-Tercedor & Sánchez-Ortega (1988).

TABLE 2 Description, average and range (minimum and maximum) of ecological response metrics.

Ecological group	Response metrics	Description	Average
Diatoms	Diat_Rich	Diatom species richness	7.52 (1–17)
	Diat_Abund	Diatom species abundance	2,857.61 (6.12–9 × 10^4)
	Diat_Shannon	Shannon-Wiener index	1.40 (0–2.52)
	Diat_Simpson	Simpson index	0.64 (0–0.9)
	Diat_Pielou	Pielou index	0.74 (0–1)
	%Eunotia	% abundance of *Eunotia*	56.81 (0–100)
	%Nitz_palea	% abundance of *Nitzschia palea*	2.28 (0–76.76)
	TDI	Trophic Diatom Index (Kelly (1998), adapted)	15.44 (0–92.01)
Macroinvertebrates	Inv_Rich	Macroinvertebrate taxa richness	14.66 (3–28)
	Inv_Abund	Macroinvertebrate taxa abundance	513.94 (6–6.4 × 10^3)
	Inv_Shannon	Shannon-Wiener index	1.49 (0.43–2.23)
	Inv_Simpson	Simpson index	0.62 (0.19–0.86)
	Inv_Pielou	Pielou index	0.58 (0.19–0.96)
	%EPT	% abundance of Ephemeroptera, Plecoptera and Trichoptera	17.72 (0–76.47)
	%Plecoptera	% abundance of Plecoptera	2.53 (0–31.82)
	%OLI_HIR	% abundance of Oligochaeta and Hirudinea	3.74 (0–70.29)
	BMWP	Biological Monitoring Work Party (Monteiro et al. (2008); Junqueira & Campos (1998); Uherek and Gouveia (2014); & Alba-Tercedor & Sanches-Ortega (1988), adapted)	87.00 (15–170)
	ASPT	Average Score per Taxon Armitage et al., (1983)	5.90 (4.83–6.93)
Ecosystem Processes	Mic_dec	% of decomposed leaf litter in fine mesh litter bags	65.2 (32.79–119.70)
	Tot_dec	% of decomposed leaf litter in coarse mesh litter bags (microbial + invertebrates)	61.69 (15.28–118.05)
	Resp	Sediment respiration rate (mg O_2 h^{-1})	0.13 (0–1.31)
	Chl	Algal biomass (Chlorophyll *a* concentration ug m^{-2})	0.69 (0–11.63)
	Erg	Fungal biomass (Ergosterol concentration mg Erg/g AFDM)	0.05 (0–0.27)
	ATP	Microbial biomass (ATP concentration nmol ATP/g AFDM)	0.01 (0–0.07)

Taxa without published sensitivity grades were attributed with the lowest score (1). The Average Score per Taxon (ASPT) index Armitage et al. (1983) was calculated by dividing the score of each taxon by the total number of scoring taxa.

2.2.5 Ecosystem processes

The respiration rates on river sediments were measured following Feio et al. (2010), with some adaptations, as an indication of river metabolism. Three PVC chambers (30 cm long, ø 4.4 cm) were half-filled with riverbed sediment (<1 cm diameter; collected up to 15 cm depth) and then filled in with stream water and sealed with rubber stoppers. To control, one PVC chamber was filled in only with river water. Respiration rates were measured as the depletion of dissolved oxygen in the chambers after approximately 30 min. The volume of water in each chamber was measured using a beaker.

The respiration rate for each site was given by the expression (1):

$$Rr = \sum \mathbf{s}\left[\mathbf{Vx}\left(\mathbf{Of} - \mathbf{Oi}\right)\mathbf{xt}\right] - \mathbf{c}\left[\mathbf{Vx}\left(\mathbf{Of} - \mathbf{Oi}\right)\mathbf{x}\,\mathbf{t}\right] \quad (1)$$

where Rr (mg O_2 L^{-1} h^{-1}) is the respiration rate, "s" is each chamber, V is the volume (L) of water in each chamber, Of is the final O_2 concentration (mg L^{-1}), measured with a YSI probe), Oi is the initial O_2 concentration (mg L^{-1}), "t" is the incubation period (hours) and "c" is the control chamber. Respiration was measured only in September (dry season).

The microbial (fine mesh bag-FMB) and total (coarse mesh bag-CMB) leaf litter decomposition rates were calculated by the decrease in leaves weight after 30 days of incubation on riverbeds. Portions with approximately 3 ± 0.5 g of dry air leaves (*Hyeronimia alchorneoides*) were placed in fine- (0.25 mm mesh; 13 cm × 20 cm size) and coarse-mesh litter bags (10 mm mesh; 18 cm × 23 cm size). The use of FMB (only microbial effects) and CMB (microbial and invertebrates assemblages' effects) allows distinguishing the contribution of microorganisms and macroinvertebrates to the loss of leaf litter mass. Moreover, CMB may also add the physical water abrasion effect (Tonin et al., 2018).

In the laboratory, six leaf discs (10 mm diameter) were cut from each sample. A set of a three-leaves disc was used to determine ergosterol content (as an indirect measure of fungal biomass on decomposing leaves; Gessner 2005) and another similar set was used to determine the total ATP content (as an indirect measure of the total microbial biomass; Abelho 2005). The results were expressed in % of decomposed biomass standardized for 30 days.

A similar piece of artificial substrate area scraped for diatoms (approx. 250 cm^2) was scraped off for Chlorophyll *a* determination, an indirect measure of periphytic algal biomass. The material was filtered (glass fibre 0.45 mm filters) and frozen until analysis. Chlorophyll *a* concentration ($\mu g\ m^{-2}$) was determined spectrophotometrically after acetone extraction (Wetzel & Likens 1991).

2.3 Data analysis

To quantify the relationships between selected predictors and response metrics we used Boosted Regression Tree (BRT) analysis. BRTs provide a means to fit nonlinear relationships between predictors to response metrics, including interaction effects, by using a boosting strategy to combine results from a large number (often thousands) of simple regression tree models (Friedman 2001). Three elements are fundamental in the execution of the BRT models: 1) tree complexity (*tc*), which controls whether the interactions are fitted; 2) the learning rate (*lr*), which determines the contribution of each tree to the growing model; and 3) the number of trees (*nt*) necessary for the optimization of the model, which is determined based on the two previous parameters (Elith et al., 2008). We adopted the tree complexity (*tc*) equal to 5, and the learning rate varying between 0.01 and 0.0001, guaranteeing that at least 1,000 trees were generated for each metric (see all settings in Supplementary Material). The bag fraction (*bf*) represents the proportion of training data to be selected, without replacement, at each interaction, thus controlling the stochasticity of randomization. We applied *bf* equal to 0.75. Within the BRT, the cross-validation (CV) technique provides a means for testing the model using part of the training data, while still using all data at some stage to fit the model. It is useful especially in cases of relatively low sample sizes (Elith et al., 2008), as is the case of this study.

BRT outputs included the performance of training data (% variation explained) and test data (CV correlation), the relative influence (contribution) of each predictor to explain the training data (sum adds up to 100%). Lastly, partial dependence plots indicated the shapes of relationships between predictors and the response variable (e.g., linear, curvilinear, and sigmoidal) taking into account the average effect of all other predictors (Elith et al., 2008). We also used the shapes for visual identification of thresholds (Wagenhoff et al., 2017).

In a second step, the models were reduced with the exclusion of predictor variables that contributed less than 2% to explain each response variable, since the reduction of variables is desirable considering that BRT models tend to overfit models (Elith et al., 2008; Brown et al., 2012). The results presented refer to the reduced final models. Sewage release (SR) was excluded from the reduced models in all response metrics (less than 2% of relative contribution). All statistical analyses were performed using the gbm package (Greenwel et al., 2018) from R v.4.0.3 (R Core Team 2020) and specific code for BRT provided by Elith et al. (2008).

3 Results

3.1 Performance of boosted regression tree models

For macroinvertebrate metrics, the highest percentages of variance explained were observed for % Oligochaeta/Hirudinea (91%), Macroinvertebrate abundance (84%), % Plecoptera (82%) and % EPT (70%). For diatom metrics, BRT models explained the highest percentage of variation for: %Eunotia (87%), Trophic Diatom Index (TDI, 84%) and Diatom richness (77%). Metrics of ecosystem processes were best predicted for algal biomass production (Chl, 96%), microbial decomposition (Mic_dec, 82%) and total decomposition (Tot_dec, 73%) (Figure 2). The medians of the structural and functional metrics were very similar, around 60% (Figure 2).

3.2 Relative contributions of predictor variables

Predictors related to the river size (drainage area and elevation) were important to explain some metrics, but especially %Plecoptera, for which the two predictors combined explained 29% of its variation. Habitat variables explained large portions of variation in a few metrics, most noteworthy among them was the percentage of organic matter in river sediment for macroinvertebrate metrics, ergosterol, and ATP (Figure 3).

Water quality variables were relevant in explaining almost all metrics. Conductivity highly contributed for most metrics (macroinvertebrates, diatoms and ecosystem processes). Turbidity, dissolved oxygen, nitrate, and phosphate were also relevant for some response metrics (Figure 3).

Among the land use predictors, agricultural and urban cover in the upstream catchment (CAT_agr and CAT_urb) explained the

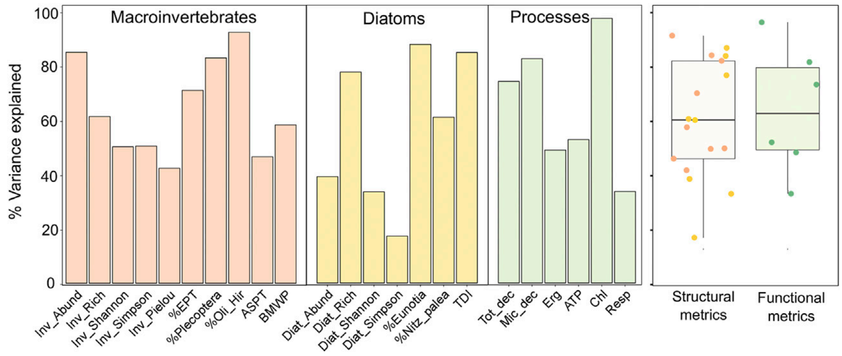

FIGURE 2
Percentage of variance explained for Macroinvertebrates, Diatoms, and Ecosystem Processes metrics models (see the metrics description in Table 2). Diat_Pielou is not shown because it was not possible to run the model. Boxplot of structural (Diatoms and Macroinvertebrates) and functional (ecosystem processes) metrics results. See all model settings and statistics in the Supplementary Material.

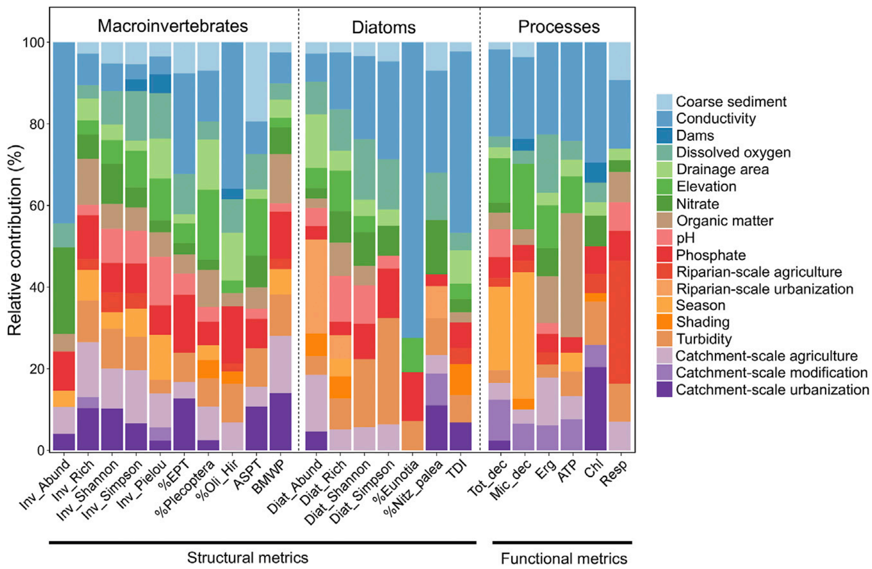

FIGURE 3
Relative contribution (0–100%) of predictor variables on the variance explained of each response ecological metrics (structural and functional; see the metrics description in Table 2). Sewage release (SR) was excluded because its contribution was 0% in all models.

largest fraction of variation of the response metrics (Figure 3). Macroinvertebrates metrics were the most influenced by them, but also the abundance of diatoms and sediment respiration. For macroinvertebrates, some metrics were rather explained by urban cover in the upstream catchment (e.g., %EPT, 13%), others by agricultural (e.g., Inv_Simpson, 13%) and others by both, like the macroinvertebrates richness (CAT_agr 14%, CAT_urb 10%) and the BMWP (CAT_agr 14%, CAT_urb 13%). Generally, catchment-scale metrics explained more variation in ecological variables than riparian-scale metrics, except for the abundance of diatoms and respiration rate, which were mostly influenced by urbanization (RIP_urb) and agricultural activities in the riparian corridor (RIP_agr), respectively. The influence of the presence of dams was minimal in all models.

3.3 Ecological response relationships with environmental gradients

The relationships between predictors and response metrics presented some features in common: 1- most response shapes were non-linear; 2- some of the response metrics presented an early increase or decrease followed by the continuity of the curve in the opposite direction; 3- for some of them, it is possible to identify common values from which the curves abruptly changed, which points out to the existence of potential thresholds. For example, change points of most conductivity curves were around 100 µS cm^{-1}. For phosphorus, change points were around 0.5 mg L^{-1}, and CAT_urb between 10 and 20% (Figures 4–6).

Conductivity, phosphate, nitrate and land use in the catchment had a positive influence on Macroinvertebrates abundance, %Oligochaeta/Hirudinea, TDI, Diatom richness and algal biomass; and positive on %EPT and %Eunotia. The increase in the drainage area and the reduction in elevation were negatively related to the %Plecoptera, %Eunotia and Diatom richness, and positively related to the increase in % Oligochaeta/Hirudinea, TDI, total and microbial decomposition (Figures 4–6).

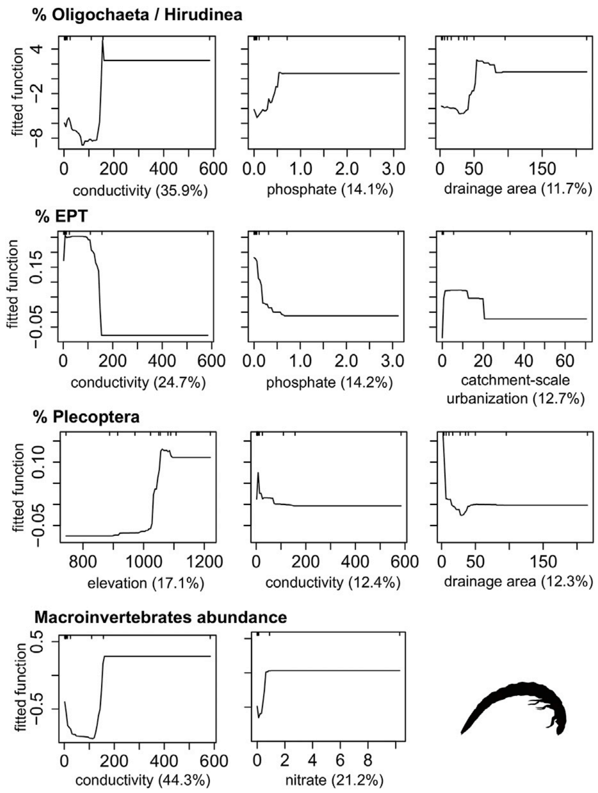

FIGURE 4
Boosted regression tree (BRT) fitted functions for the best performance models of Macroinvertebrate metrics. Plots are only shown for those predictors that explained more than 10% deviance in the metric. Rug plots show the distribution of data, in deciles, of the variable on the X-axis.

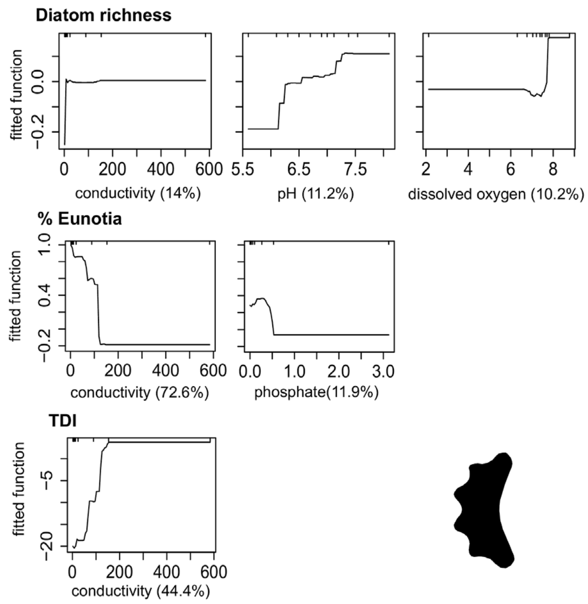

FIGURE 5
Boosted regression tree (BRT) fitted functions for the best performance models of Diatom metrics. Plots are only shown for those predictors that explained more than 10% deviance in the metric. Rug plots show the distribution of data, in deciles, of the variable on the X-axis. (TDI) Trophic Diatom Index.

4 Discussion

The study made it possible to identify the main predictors driving each ecological metric and how metrics responded to natural and human-related predictors, allowing the detection of potential indicators of stream integrity. While the percentage of EPT group abundance and the algal biomass would be good indicators of urbanization in the upstream catchment, the percentage of *Eunotia* abundance would indicate changes in water quality. In contrast, other metrics were poorly explained by the predictors or mainly influenced by natural predictors, making them inappropriate indicators of environmental disturbances for management purposes (Norris & Hawkins 2000). Like the Simpson Index for Diatoms showed a low variance explained (17%), the percentage of Plecoptera, which was influenced mainly by natural characteristics (elevation and drainage area), and decomposition primarily influenced by seasonality. Most of our models presented a unidirectional response for direct (land use) and indirect (water quality) human disturbances. Overall, increasing human disturbance (e.g., conductivity and changes in land use) led to a decrease in pollution-sensitive taxa (e.g., percentage of EPT group and *Eunotia*) and an increase in pollution-tolerant taxa (e.g., percentage of Oligochaeta and Hirudinea, the Trophic Diatom Index and the algal biomass production).

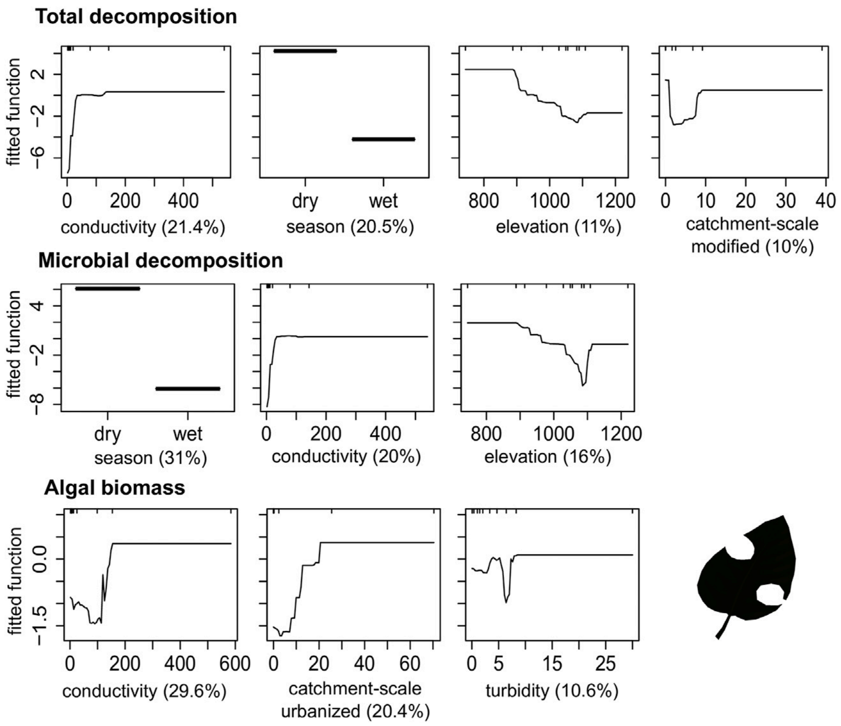

FIGURE 6
Boosted regression tree (BRT) fitted functions for the best performance models of functional metrics. Plots are only shown for those predictors that explained more than 10% deviance in the metric. Rug plots show the distribution of data, in deciles, of the variable on the X-axis.

The nonlinear responses promoted insights into the subsidy-stress theory (too much of a good thing syndrome; Odum 1983), which predicts that the increase of limited resources (e.g., nutrients, light) in an environment may have an initial positive effect on biological communities and ecosystem functions. However, this effect rises to a certain threshold; after then, it can lead to adverse effects. In this context, considering that Brazilian savanna streams are poor in nutrients (Markewitz et al., 2006), nutrient inputs possibly promote the maintenance of more species/individuals. But at the other extreme of the gradient, intense disturbances are expected to reduce the number of species that can colonize or tolerate high impact levels (Odum 1983). The shape of the EPT curve (initial low value followed by a sharp rise, lately a decrease) indicates their sensitivity to disturbed environments face to the increase in conductivity, and catchment urbanization was an example of this (Ligeiro et al., 2013; Siegloch et al., 2017). The evaluation of the response curves from BRT models was also a good starting point for discussing thresholds for the considered predictors. Notable change points could be observed, such as conductivity, phosphate, and the percentage of urbanization in the upstream catchment.

Our study showed the most important predictors to explain the ecological metrics were physical and chemical variables often used to indicate human disturbances (Heathwaite 2010; Uriarte et al., 2011; Alvarez-Cabria et al., 2016), such as phosphorus and nitrate concentrations, but especially conductivity. For instance, we reported significant changes in ecological metrics when conductivity stood between 100 and 200 μS cm^{-1}, suggesting a potential threshold. Values above this threshold indicate loss of water quality, except when high conductivity is due to the natural background (Fravet & Cruz 2007; Fundação Nacional de Saúde, 2014; CETESB 2020). Conductivity was the main predictor for the studied metrics in terms of relative importance. Comparing water bodies of preserved and anthropogenic (especially those without vegetation protection) areas, the diffuse sources of pollution resulted in higher electrical conductivity (Gardiner et al., 2009; Rezende et al., 2014). Like in anthropogenic areas with inadequately treated effluents flowing to water bodies, increasing the nutrient concentrations of the water (Myrka et al., 2008).

Phosphorus increase is responsible for triggering the eutrophication of freshwaters (Figueredo et al., 2016; Zhang et al., 2017) coming from agricultural fields and urban effluents (Ockenden et al., 2016). Our results showed potential thresholds for phosphate around 0.5 mg L-1, and its contribution was especially relevant for metrics sensitive to pollution, such as %EPT, %Oli_Hir and %Eunotia, as shown elsewhere (Kelly.

1998; Salomoni et al., 2006; Ferreira et al., 2014; Pardo et al., 2020). Both conductivity and phosphorus were positively related to effluent discharge and deforestation. Anthropogenic areas (remarkably urbanized areas) strongly influence biological assemblages, and their effects are disproportionate to the size of the area used (Rezende et al., 2014; Campos et al., 2021).

Urban and agricultural cover in the upstream catchment was the most important land-use factor to explain the response metrics. The adverse effects of replacing native vegetation with urban or agricultural areas in the upstream catchment have been reported for the stream via complex pathways (Allan 2004) like changes in temperature, habitat diversity, hydromorphology, sunlight, and nutrient availability (Einheuser et al., 2013). These changes have translated into alterations in the structure and functioning of the stream ecosystem (Clapcott et al., 2012). We observed that values between 10 and 20% of urban cover in the upstream catchments led to a decrease in the abundance of the EPT group and an increase in algal biomass. Brito et al. (2020) reported abrupt changes in the composition of macroinvertebrates with the removal of 57–79% of native vegetation in the Amazon Forest, while Dala-Corte et al. (2020) reported threshold values between 3 and 40% of native vegetation removal across biomes in Brazil. Therefore, our results in the study region indicated more restrictive values suggesting that parts of the Brazilian savanna are more susceptible to the conversion of native areas. Additionally, the increase of algal biomass related to the urbanization process confirms a recent study that shows a 32% greater effect on stream functioning than in its structure in the tropics (Wiederkehr et al., 2020).

Changes in biological assemblages and ecosystem processes are commonly associated with alterations in the riparian plants (Encalada et al., 2010; Fierro et al., 2017), especially in headwaters that are light-limited systems and rely on plant litter inputs from surrounding vegetation (Bunn & Davies, 2000; Perona et al., 2009). However, we did not observe a robust relationship with macroinvertebrates. On the other hand, we found a consistent negative relationship among diatoms, urbanization and agriculture in the riparian zone, indicating a higher local than catchment-scale effect. This finding suggests reliable benefits of forested riparian buffers for stream biological diversity in urban environments, supported by previous studies (e.g., Mutinova et al., 2020).

The different responses to the set of predictors, including structural and functional ecosystem metrics, can lead to a comprehensive interpretation of river conditions (Feio et al., 2010). The prediction of ecological conditions is relevant from the management's point of view since these are more complex data to be acquired but of extreme relevance for understanding the health of water bodies (Karr 2006). Knowledge about the importance of each predictor for the response metrics allows, for example, to predict some ecological conditions in places with limited availability of biological data.

Finally, the potential thresholds identified in the present study are important signs of significant changes in ecological responses. They should be employed in eventual review processes of guidelines to public policies for river health preservation and recovery (Huggett 2005). Brazilian national environmental guidelines do not consider, for example, conductivity (CONAMA n° 357, Brasil 2005), notwithstanding the importance of this variable as an ecosystem driver, as demonstrated in our study. In addition, further attention should be paid to the context of land use, especially to the urbanization processes in the upstream catchment. Currently, Brazil has increased awareness of riparian vegetation (Federal Law n° 12.651, Brasil 2012). However, for purposes of biodiversity conservation and maintenance of ecosystem processes, we also have shown it necessary to consider the entire context of the catchment in which the stream is located.

5 Conclusion

Our results demonstrated the importance of considering a set of ecological response metrics (structural and functional) and environmental factors (natural and disturbances), allowing a complete view of the freshwater ecosystem condition. The relative importance of predictors on ecological metrics pointed to metrics most affected by factors on a local scale (e.g., percentage of *Eunotia* abundance) and catchment scale (e.g., algal biomass). Also, the nonlinear responses permitted the detection of gradual or abrupt change curves, pointing out the existence of potential thresholds of important drivers, like the conductivity (100–200 µS cm-1), phosphate (0.5 mg L-1), and catchment-scale urbanization (10–20%). The potential bioindicators (considering the best performance models and the ability to respond more strongly to the human disturbances) were macroinvertebrates abundance, EPT abundance percentage, Oligochaeta and Hirudinea abundance percentage, percentage of *Eunotia* abundance, Trophic Diatom Index, and algal biomass. Although we have worked with many biotic and abiotic variables and the BRT model considered the interaction between them, models are simplified representations of a complex system, therefore presenting limitations. Nevertheless, the consistency and reasonableness of influential metrics within a given set of ecological metrics provide a weight of evidence in support of the models' results. The BRT models approach proved to be powerful tools that can be effectively employed to enhance and give better direction to freshwater management, not only to the streams of the Brazilian savanna but also to water bodies in other regions.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

Campos, CC: Term, Conceptualization, Methodology, Formal analysis, Investigation, Writing–Original Draft, Writing–Review and Editing, Visualization. Tonin, AT: Writing–Review and Editing, Visualization. Kennard, MK: Conceptualization, Writing–Review and Editing, Supervision. JG: Conceptualization, Resources, Funding acquisition, Writing–Review and Editing, Supervision.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2022.867905/full#supplementary-material

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Stream nitrogen uptake associated with suspended sediments: A microcosm study

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Despite significant advances in our understanding of nitrogen (N) removal pathways along river networks, the role of water column processes remains largely understudied. This knowledge gap not only limits our capacity to determine N transport and retention in mid-to-large rivers but also hampers our understanding of N removal processes in smaller streams during stormflow conditions, in which significant increases in suspended sediment concentrations (SSC) typically occur. High SSC in the water column can provide abundant substrate for microbial growth and water column N uptake. However, storms of different size mobilize different quantities of sediment of varying properties and sizes, which can ultimately modulate water column N uptake rates in the stream during stormflows. To assess water column N uptake associated with suspended sediment particles of different sources and sizes, we quantified assimilatory and dissimilatory N uptake rates in a set of microcosms representing a gradient of sediment properties (organic matter, N content, and microbial activity) and surface area (fine vs. coarse size) availability. Water column assimilatory uptake (U_{sed}) ranged from 12.7 to 187.8 μg N [g sediment]$^{-1}$ d^{-1} across all sediment sources and size fractions, and was higher on average than denitrification rates (DN_{sed}) in agricultural and stream bank sediments but not in streambed sediments (mean DN_{sed} = 240.9 ± 99 μg·N [g sediment]$^{-1}$·d^{-1}). Sediment-bound C in suspended sediment varied among sediment sources and was directly related to U_{sed} rates, but not to DN_{sed} rates, which were less predictable and more variable. Overall, our results showed a positive nonlinear relationship between water column N uptake and SSC, while indicating that water column N uptake may scale differently to SSC depending on sediment source, and to a lesser degree, particle size. Because low, moderate, and large storms can mobilize different quantities of sediment in the watershed of different sources and sizes, it is likely that storm size will ultimately modulate the contribution of water column uptake during storm events to whole-reach N retention.

KEYWORDS

stream, nitrogen, suspended sediment, uptake, denitrification

Introduction

Global nitrogen (N) export from watersheds has exceeded more than twice its preindustrial value due to modern human activities (Schlesinger, 2009), with streams of many regions experiencing up to a 5-fold increase in total N concentrations (Dodds and Smith, 2016). In North America, only a quarter of the monitored streams and rivers are currently showing long-term decreasing trends in N concentrations (Shoda et al., 2019), which indicates that nutrient enrichment remains a major threat to the ecological integrity of running waters. The improved management of excessive N loads requires both a reduction of anthropogenic N inputs and a more comprehensive understanding of N removal pathways along river networks and among different streamflow conditions (Wollheim et al., 2017, Wollheim et al., 2018). Previous research has focused largely on benthic N removal in headwater streams during baseflow, while much less is known about the role of water column N removal in larger streams or during stormflow conditions.

Most of the benthic N uptake in headwater streams is associated with biological assimilation and storage, *aka* assimilatory N uptake (Peterson et al., 2001; Arango et al., 2008), a temporary N retention that can last from hours to days (Peipoch et al., 2014; Tank et al., 2018). The remainder of benthic N uptake is associated with dissimilatory uptake, including denitrification, a major pathway by which N is permanently removed from aquatic ecosystems through anaerobic microbial respiration (Craig et al., 2008; Mulholland et al., 2009). Denitrification can occasionally account for more than 40% of benthic nitrate uptake in headwater streams (Mulholland et al., 2009). However, N uptake also occurs in the water column of streams of varying size (Reisinger et al., 2015, Reisinger et al., 2016), but the paucity of water column uptake measurements limits our understanding of the relative contribution of assimilatory and dissimilatory pathways to water column N uptake and the major controlling factors of these pathways. Previous studies have shown that water column N uptake is strongly related to suspended sediment concentration, particle size, and nutrient availability (Jia et al., 2016; Reisinger et al., 2021). In particular, large surface area generated by high concentrations of fine particles in the water column and associated organic matter can provide abundant substrate for microbial growth and N uptake (Liu et al., 2013; Jia et al., 2016), including anaerobic processes such as denitrification since suspended particles can retain anoxic microsites (Jia et al., 2016; Xia et al., 2017, Xia et al., 2018). Results to date indicate a strong dependence of water column N uptake rates on suspended sediment concentration. While some rivers can have high concentrations of suspended sediments during baseflow conditions (e.g., 20 $g \cdot L^{-1}$ in Xia et al., 2017), most streams and mid-size rivers only carry high suspended sediment loads during and immediately after storm events, highlighting the importance of better understanding N uptake processes during stormflow conditions.

Stormflows cause significant increases in suspended sediment concentrations of small-to mid-order streams (< 4th order; Cashman et al., 2018; Noe et al., 2020). During and after significant storm events, suspended sediment particles in relatively small streams may provide a comparable amount of water-sediment interface to that in the streambed, akin to the increase in bioavailable surface area associated with suspended sediment when streams become rivers (Gardner and Doyle, 2018). In fact, much like the longitudinal transition from small streams to large rivers, storm events cause short-term increases in water depth, suspended sediment concentrations, and sediment-bound nutrients (Wood and Armitage, 1997) that can promote water column uptake at event scales even in headwater networks. Other factors may also contribute to the potential nutrient uptake capacity in the water column during storm events. For instance, anthropogenic land use can play a role in the quantity and character of particles in suspension (Gellis and Mukundan, 2013; Gellis and Gorman-Sanisaca, 2018). Low intensity summer thunderstorms have been associated with resuspension of sediment particles originating from the stream channel (e.g., stream bed and banks), while larger storm events bring a greater contribution of particles from the surrounding landscape (Karwan et al., 2018; Jiang et al., 2020). Moderate storm events following freeze-thaw cycles in the winter have been implicated in stream bank erosion, yielding significant fluxes of fine sediment particles from stream banks (Inamdar et al., 2018). Overall, suspended sediment particles from different sources can vary in grain size and nutrient content (Jiang et al., 2020; Lutgen et al., 2020), which may ultimately affect the potential rates of water column N uptake during and after storm events.

Here, we used a microcosm approach to simulate suspended sediment concentrations in streams during storm events of different magnitude. Our goal was to assess how the source, size, and concentration of suspended sediments influence water column N uptake. Sediment particles of different sources and size generated a gradient of sediment properties (organic matter, N content, and microbial activity) and surface area availability. In particular, we evaluated fine (<63 μm) and coarse (63–2000 μm) fractions of three of the most common sediment sources that are mobilized by storm events of varying intensity in the Mid-Atlantic region of the US: Streambed, stream bank, and agricultural soils (Gellis and Noe, 2013; Cashman et al., 2018; Jiang et al., 2020; Noe et al., 2020). We hypothesized that fine fractions of stream bank and upland agricultural sources will result in higher water column N uptake due to the greater surface area and sediment-bound nutrient content (e.g., Jiang et al., 2020) associated with their dominance of fine sediment particles. In the microcosms, we experimentally manipulated suspended particle concentrations (up to 5,000 $mg \cdot L^{-1}$) for each size fraction and sediment source, and measured both assimilatory N uptake and denitrification rates in each microcosm using ^{15}N tracer methods.

Materials and methods

Sediment sampling and characterization

In the summer of 2021, we collected sediments at three different locations (one for each sediment source) within the White Clay Creek watershed, Pennsylvania, United States. The White Clay Creek (WCC) is a temperate third order stream of characteristic conditions in the Piedmont physiographic province of southeastern Pennsylvania. Mean annual precipitation and temperature in WCC are 1,190 mm and 11.7°C, respectively (period 2009–2020). At the location of sediments collection, the stream drains a 7-km two watershed dominated by pasture/hay (48%), deciduous forest (19%), cultivated crops (17%) and woody wetlands (4%) with a mostly closed-canopy stream network (~65% of forested riparian areas). We collected streambed and stream bank sediments from the East Branch of WCC and surface soils (top 15 cm; 39°51′32.5″N 75°47′00.6″W) from agricultural fields located upland of this stream branch (39°51′38.5″N 75°46′50.3″W). The location of sediment collection was partially guided by our previous sampling described by Lutgen et al. (2020). At each location, sediment was collected from three different sites within the location area and composited in a single sample. We collected, homogenized, and transported the necessary amount of sediment (500 g) for microcosm experiments to the laboratory, where it was processed immediately and kept in the dark overnight at 4°C. In all cases, sediment collection and the starting of each microcosm experiment occurred in less than 24 h. Prior to each sediment collection for the microcosm study, we took sterile samples of each sediment source for the characterization of denitrifying *nosZ* genes (clades I and II) abundances; these samples were collected in WhirlPaks and immediately stored at 0°C. We also collected additional samples to characterize the particle size distribution of each sediment source by volume and surface area (assuming spherical geometry of all particle sizes) using a Beckman Coulter LS 13 320 particle size analyzer.

In the laboratory, we first separated sediments from each source into clay/silt (<63 μm) and sand (63–2000 μm) size fractions by sequentially wet sieving the collected sediments through 2000 and 63 μm sieves using stream water from WCC. Then, we vacuum filtered all the sieved water in the previous step using 0.70-μm pore size glass fiber filters (Whatman filter Sigma-Aldrich, Missouri, United States) to compile <63 μm particles while keeping the filtered water for the preparation of the microcosm experiments. This was done to ensure that microbes dislodged during the wet sieving process were added to the microcosms (Jia et al., 2016). The sieving process was repeated until sufficient sediment and stream water were collected for the execution of each microcosm experiment. Prior to the microcosm incubations, we subsampled the 0.70-μm filtered water used in the sieving process of each sediment source for the analysis of dissolved organic carbon (DOC) and nitrate ($N\text{-}NO_3^-$) concentrations. DOC concentrations were determined using an Aurora 1030 W TOC Analyzer (Oceanographic Int., College Station, Texas, United States) and chemical oxidation (Menzel and Vaccaro, 1964). The concentrations of $N\text{-}NO_3^-$ were determined by discrete colorimetric analysis using an AQ300 discrete analyzer (SEAL Analytical, Wisconsin, United States) following standard procedures (APHA 2017).

For the characterization of denitrifying *nosZ* genes in each sediment source, the genomic DNA was extracted using DNeasy PowerSoil Pro Kit (Qiagen, Germantown, MD) and the *nos*Z genes were quantified with qPCR on a QuantStudio TM three system with Analysis Software v1.5.1 (Thermo Fisher Scientific, Waltham, WA). For *nos*Z clade I the qPCR was performed using the primers *nos*Z 1840F(CGCRACGGCAASAAGGTSMSSGT) and *nos*Z2090 R (CAKRTGCAKSGCRTGGCAGAA) (Henry et al., 2006). 20 μl reactions contained 1X Power Up SYBR Green Master Mix (Applied Biosystems, Waltham, MA), 0.5 μM each primer, and 0.6 mg/ml BSA (Invitrogen, Waltham, MA). The protocol is as follows: An initial 50°C for 2 min and 95°C for 10 min, followed by six cycles of 95°C for 15 s, 65°C for 30 s, 72 for 30 s, and 80°C for 15 s, then 45 cycles of 95°C for 15 s, 60°C for 30 s, 72 for 30 s, and 80°C for 15 s, ending with a melt curve step. The quantification of nosZ clade II was performed using the primers *nos*ZIIF (CTIGGICCIYTKCAYAC) and *nos*ZIIR (GCIGARCARAAITCBGTRC) (Jones et al., 2013). 20 μl reactions contained 1X Power Up SYBR Green Master Mix, two uM each primer, and 0.5 mg/ml BSA. The protocol was as follows: an initial 50°C for 2 min and 95°C for 5 min, followed by 45 cycles of 95°C for 20 s, 52 for 35 s, and 72 for 1 min 10 s, followed by a melt curve step. Each qPCR run contained a standard curve of 10X serial dilutions of gBlocks Gene Fragments from Integrated DNA Technologies (Coralville, IA), and the gene quantification was standardized to gene copy numbers per Gram of soil.

Suspended sediment microcosm incubations

We quantified nitrate uptake rates (assimilatory and denitrification) associated with different suspended sediment sources, size, and concentrations using a three-level factorial design of microcosm incubations. Microcosms consisted of a modified Kimble 250 ml wide-mouth media bottles with screw caps with rubber septa installed. To generate a gradient of suspended sediment concentrations (SSC), we individually weighed aliquots of sediment from each source and size class to generate four different treatment levels of SSC: 500, 1,000, 2,500, and 5,000 mg·L^{-1}. The selected SSC range (0.5–5 g·L^{-1}) is similar to that observed in WCC for storm events of varying size (Jiang et al., 2020). Unfortunately, targeted concentrations were difficult to pinpoint precisely due to the unknown water content contribution to wet sediment weight, and despite our best efforts, we ended up with a continuous gradient of SSC across the 12 replicates per sediment source and size. Consequently, SSC was included as a continuous variable in our data analysis with comparable ranges across sediment source and size treatments: 130–5,408 mg·L^{-1} for streambed, 177–6,216 mg·L^{-1} for stream bank, and 224–5,281 mg·L^{-1} for agricultural sediments. We replicated each treatment by triplicate (sediment source × particle size × concentration) making a total of 75 microcosms including sediment-free blanks bottles. After sediment aliquots were added, we poured 250 ml into each microcosm using the stream water previously used to separate sediment size fractions. We placed a magnetic stir bar in each microcosm and then added 1 ml of a 796 mg·L^{-1} solution of $Na^{15}NO_3^-$ specifically prepared to generate less than a 5% increase of the N-NO_3^- concentration in each microcosm. We capped the microcosms and evacuated (3 min) and flushed with He gas (1 min) by inserting tubing with syringe needles attached into the septa of each microcosm chamber (Dodds et al., 2017), repeating the evacuating and flushing cycle three times prior to the beginning of each incubation.

Then, microcosms were placed on magnetic stir plates within an incubator set to 25°C. We set the stir plates to 360 rpm to ensure particles remained in suspension during the incubation period (Jia et al., 2016), and then closed the incubator to keep chambers in the dark. We sampled dissolved gas at 4 and 24 h after the start of incubation, using a gas-tight syringe (Hamilton 25 ml Model 1025TLL) to sample 12 ml of gas from each chamber into 12 ml pre-evacuated Exetainers (Labco Ltd., High Wycombe, United Kingdom). The timing of gas sample collection was determined after conducting preliminary microcosm incubations that showed consistent linear increases of N_2 and N_2O mass between 4 and 24 h. Gas samples were analyzed for $\delta^{15}N$ of N_2, and N_2O *via* IRMS (ThermoScientific Delta V Plus) at the University of California-Davis Stable Isotope Facility.

At the end of incubations, we carried out a suite of additional analyses of the sediment and water in each microcosm. These included SSC, organic matter (OM) content, C and N content, and $\delta^{15}N$ signatures of suspended sediments. Sediment samples for SSC analysis were collected on pre-weighted FVF glass fiber filters, oven-dried at 60°C for 72 h, and weighed on a Sartorius (Goettingen, Germany) MC1 analytical balance. OM samples were oven-dried at 60°C for 72 h, weighed on a Sartorius (Goettingen, Germany) MC1 analytical balance, combusted at 500°C for 5–6 h, and reweighed for calculation of dry mass and ash-free dry mass (e.g., OM). Sediment samples for elemental and isotopic analyses of C and N were collected on pre-weighted glass fiber filters, encapsulated in tins, and sent to the UC Davis Stable Isotope Facility (California, United States). The C and N content (as a percentage of total dry mass) and ^{15}N isotope signatures were determined using a PDZ Europa ANCA-GSL elemental analyzer interfaced to a PDZ Europa 20–20 isotope ratio mass spectrometer (Sercon, Cheshire, United Kingdom). We also analyzed filtered water from each microcosm for DOC and N-NO_3^- concentrations using the same methodology described above.

Uptake calculations and data analysis

We calculated rates of assimilatory uptake by total suspended sediment in each microcosm following Mulholland et al. (2000), using the increase in tracer ^{15}N mass associated with suspended OM ($^{15}N_{susp\text{-}OM}$) at the end of the incubation and the tracer ^{15}N:^{14}N ratio in the microcosm NO_3^- (^{15}N-NO_3^-) as follows:

$$U_{micro} = \frac{^{15}N_{susp-OM}}{^{15}N - NO_3 * \Delta t},$$

where U_{micro} is the microcosm-specific rate of assimilatory uptake (μg N d^{-1}); $^{15}N_{susp\text{-}OM}$ is the background-corrected ^{15}N mass associated with suspended OM; and Δt is the incubation time. We calculated $^{15}N_{susp\text{-}OM}$ in μg ^{15}N microcosm^{-1} as the product of the ^{15}N molar fraction, N %, and OM mass in each microcosm and subtracting from it the background ^{15}N mass. We estimated ^{15}N-NO_3^- using background N-NO_3^- concentrations in each microcosm and the amount of tracer ^{15}N-NO_3^- added assuming a background ^{15}N:^{14}N ratio equal to 0.0036765 (that is $\delta^{15}N$-NO_3^- = 0‰).

Denitrification rates by total suspended sediment in each microcosm were calculated from the production rate of $^{29}N_2$ (r_{29}) and $^{30}N_2$ (r_{30}) following calculations for total denitrification rate associated with sediments described by Nielsen (1992) as follows:

$$DN_{micro} = (r_{29} + 2 * r_{30}) * \frac{r_{29}}{2 * r_{30}},$$

where DN_{micro} is the microcosm-specific denitrification rate (μg·N·d^{-1}). Prior to this calculation, r_{29} and r_{30} were calculated from the difference in $^{29}N_2$ and $^{30}N_2$ (in moles) between 4 and 24 h after the start of incubation. We determined the molar amount of $^{29}N_2$ and $^{30}N_2$ in each microcosm and time by multiplying total N_2 molar amount for the molar fraction of $^{29}N_2$ and $^{30}N_2$, respectively. Total molar N_2 amount was determined as the sum of N_2 moles in the water and gas phase using the specific Bunsen coefficients for N_2 according to the incubation temperature (Weiss, 1970; Dodds et al., 2017). Finally, we converted U_{micro} and DN_{micro} rates to sediment-specific rates (U_{sed} and DN_{sed} as μg·N [g sediment]$^{-1}$·d^{-1}) by dividing them for the dry weight of suspended sediment in each microcosm.

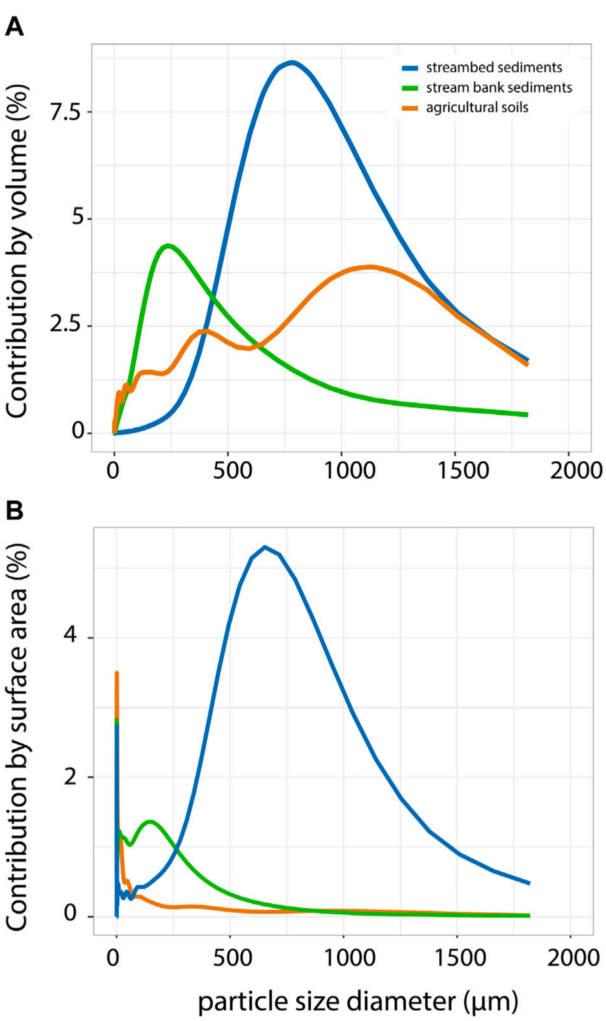

FIGURE 1
Sediment size distribution by volume **(A)** and surface area **(B)** of each sediment source. The top of the curve corresponds to the median particle diameter or surface area, for which the percentage of particles with diameters or area smaller and larger than this are 50%.

Statistical analysis of the differences in sediment properties and N uptake rates among sediment sources and sizes were addressed using non-parametric approaches with $\alpha = 0.05$. Specifically, we performed Kruskal–Wallis followed by unpaired Wilcoxon tests for comparisons among multiple sources and particle size and to address comparisons between two groups, respectively. Effects of sediment source, size and concentration on both assimilatory uptake and denitrification rates were assessed using simple linear regressions and Analysis of Covariance (ANCOVA) on $\log_{10}$-transformed variables with $\alpha = 0.05$. When necessary, appropriate constants were added to ensure positive values before transformation to meet linear model assumptions. ANCOVA models on assimilatory uptake and denitrification were performed separately, with sediment source and size as independent factors and SSC as a covariate. All statistical analyses were performed in the R environment (R Core Team, 2013).

Results

Biogeochemical properties and size across sediment sources

Beckman-Coulter particle size analysis indicated that for both surface area and volume, contributions of sand particles (> 63 µm) to stream bed sediment were significantly greater than for stream bank or agricultural sediments (Figure 1). By volume, agricultural sediments showed a more variable distribution than streambed and stream bank sediments (Figure 1A). Particle size distribution of streambed sediments was almost normally distributed and centered on 600 µm (Figure 1A), while the most common size in stream bank sediments (d50) was 235.6 µm with a strongly right-skewed distribution indicating a low presence of very large particles (Figure 1A). By surface area, all sediment sources showed a significant peak in the clay/silt fraction (<63 µm; Figure 1B), indicating the important role of fine sediments in providing available substrate area for microbial biota.

We found similar organic matter (OM) content on average among the sediment sources but substantial variation within each of them, with higher mean OM in the sand fraction of each sediment source (Table 1; Figure 2A). Carbon and nitrogen content (%C and % N) in suspended sediment were highest in agricultural sediments, intermediate in stream bank, and lowest in stream sediment (Table 1). For agricultural sediments, mean %C and %N in the sand fraction nearly doubled those of the clay/silt fraction (Table 1; Figure 2B); whereas both stream bank and streambed sediments showed the opposite pattern (Figure 2C), and a particularly low %N in the sand fraction of streambed sediment that resulted in mean C:N ratio for this source and size being more than tenfold higher than any other value (Table 1). Mean concentrations of both nitrate and DOC were much higher in microcosms with agricultural sediments than in those with stream bank or streambed sediments (Table 1). In contrast, mean concentrations of $N\text{-}NO_3^-$ and DOC varied little between size fractions of each sediment source (Table 1). Only clade I *nosZ* genes were detected in our samples, and the results showed that agricultural sediments contained a larger content of denitrifying *nosZ* genes than stream bank and streambed sediments (Table 1). Overall, differences in mean values of the measured sediment properties (OM, %C, %N, and *nosZ*) were greater among sources than between size fractions examined within each source (Table 1; Figure 2).

Variation in water column nitrogen uptake

Water column assimilatory uptake (U_{sed}) varied over an order of magnitude for each sediment source, with mean U_{sed}

TABLE 1 Biogeochemical properties of sediment sources and size fractions, including organic matter content (OM), C content in OM (C), N content in OM (N), molar C-to-N ratio (molar C:N), concentration of nitrogen-nitrate (NO_3-N), concentration of dissolved organic C (DOC), and number of denitrifying nosZ genes per sediment mass (nosZ genes). Values in bold show the means +/− SEM based on all replicates for each sediment type (N = 12), except for nosZ genes. Means +/− SEM for associated size fractions (N = 6) within each sediment type are listed below bolded value. Mean values within a column with unique superscripts are significantly different (p < 0.05) following Kruskal–Wallis and unpaired Wilcoxon tests.

Sediment source and size	OM (%)	C (%)	N (%)	molar C:N	NO_3-N ($\mu g{\cdot}L^{-1}$)	DOC ($mg{\cdot}L^{-1}$)	*nosZ* genes (10^3 copies $[g{\cdot}sed]^{-1}$)
agricultural	**14.3 ± 0.6**	**4.2AB ± 0.4**	**0.4A ± 0.03**	**12.3A ± 0.8**	**4.46A ± 0.15**	**5.7A ± 0.04**	**20,617 ± 2,143**
Sand	13 ± 0.5	5.2 ± 0.6	0.4 ± 0.06	14.5 ± 1.3	4.66 ± 0.11	5.6 ± 0.06	
clay/silt	15.5 ± 1.1	3.1 ± 0.1	0.4 ± 0.01	10.1 ± 0.3	4.27 ± 0.27	5.8 ± 0.05	
streambank	**11.5 ± 1.2**	**4.4A ± 0.3**	**0.4A ± 0.04**	**23.8B ± 6.9**	**2.83B ± 0.12**	**1.9B ± 0.04**	**661.4 ± 65.6**
sand	8.6 ± 1.8	4.3 ± 0.6	0.4 ± 0.08	34.5 ± 13.4	2.79 ± 0.12	1.9 ± 0.07	
clay/silt	14.3 ± 1.2	4.5 ± 0.2	0.4 ± 0.03	13 ± 0.3	2.87 ± 0.2	1.9 ± 0.05	
streambed	**15.4 ± 2.2**	**3.2B ± 0.2**	**0.2B ± 0.03**	**71.5C ± 17.3**	**3.74C ± 0.04**	**2B ± 0.05**	**16.6 ± 3.1**
sand	13.9 ± 1.7	3.4 ± 0.4	0.03 ± 0.01	130.5 ± 24.7	3.6 ± 0.03	1.8 ± 0.03	
clay/silt	17 ± 4.1	3.2 ± 0.2	0.3 ± 0.01	12.5 ± 0.3	3.9 ± 0.05	2.2 ± 0.05	

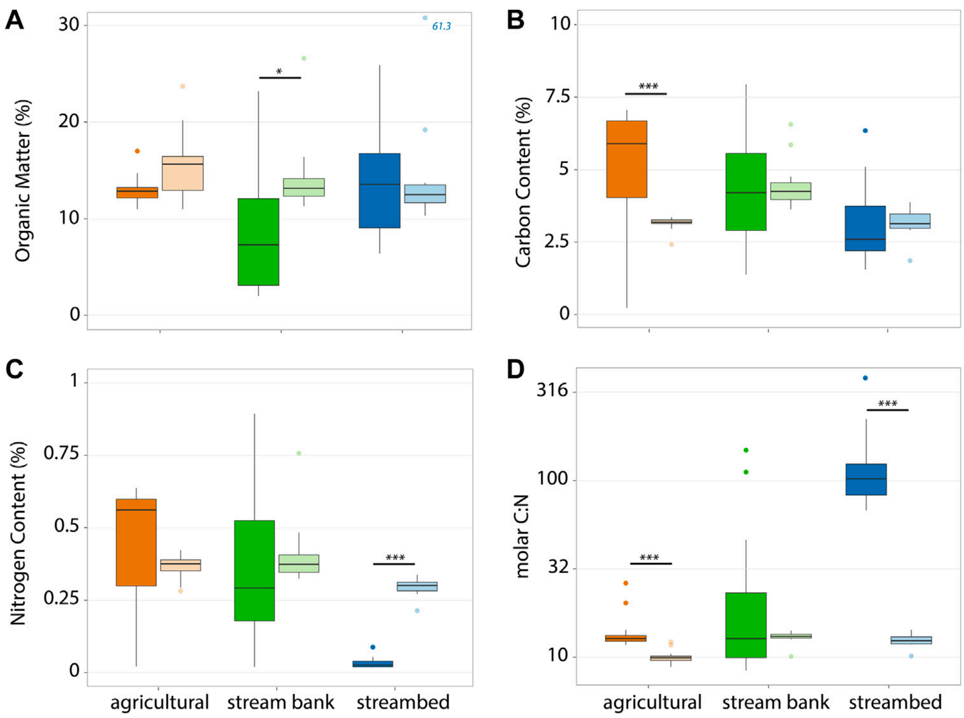

FIGURE 2
Box-plots of organic matter **(A)**, C content **(B)**, N content **(C)**, and M C:N ratio **(D)** for agricultural (orange), stream bank (green), and streambed (blue) sediments. Data for each size fraction are represented in dark and light colors for sand and silt/clay, respectively. Within each sediment source, significantly different values for each size fraction (unpaired Wilcoxon tests following significant Kruskal–Wallis) are indicated as: [blank] $p > 0.05$; *$p < 0.05$; **, $p < 0.01$; ***$p < 0.001$.

from each size fraction ranging from 12.7 to 56.6 $\mu g{\cdot}N$ [g sediment]$^{-1}{\cdot}d^{-1}$ for streambed sediments, from 30.1 to 187.8 $\mu g{\cdot}N$ [g sediment]$^{-1}$ d^{-1} for stream bank sediments, and from 114.2 to 118 $\mu g{\cdot}N$ [g sediment]$^{-1}{\cdot}d^{-1}$ for agricultural sediments (Figure 3A). U_{sed} was higher in clay/silt sediments from the streambed than in sand, and the opposite for stream

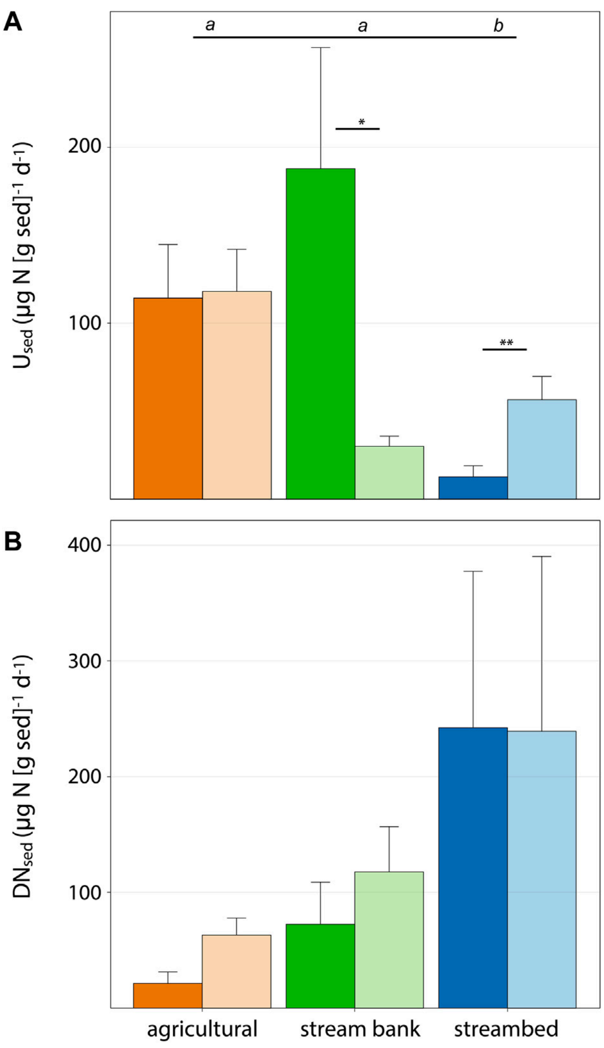

FIGURE 3
Sediment-specific rates of assimilatory uptake **(A)** and denitrification **(B)** for each sediment source and size. Bar height and error bars correspond to mean and SEM values, respectively (N = 12). Color legend is consistent with Figure 2. Size fractions are represented in dark and light colors for sand and silt/clay, respectively. Among sediment sources, bars with unique letters on the top of each panel are significantly different (unpaired Wilcoxon tests following significant Kruskal–Wallis). Within each sediment source, significantly different values between size fractions are indicated as: [blank] $p > 0.05$; *$p < 0.05$; **, $p < 0.01$; ***$p < 0.001$.

banks sediments (Figure 3A). On average, U_{sed} was significantly lower in streambed sediments than in agricultural and stream bank sediments. In contrast, the mean denitrification rate (DN_{sed}) in streambed sediments was higher compared to agricultural and bank sources for both clay/silt and sand fractions (Figure 3B). Water column DN_{sed} also varied over two orders of magnitude for each sediment source, with DN_{sed} rates in streambed sediments being more variable than DN_{sed} in the other two sediment sources (Figure 3B). Specifically, streambed sediments showed the highest DN_{sed} rates (1.7 mg·N [g sed]$^{-1}$·d^{-1}) and a similar number of non-detectable DN_{sed} rates compared to the other sediment sources (Figure 3B). Overall, we found similar or greater N uptake rates (U_{sed} and DN_{sed}) in the microcosms containing clay/silt than in those filled with sand, but this pattern was not consistent across all sediment sources and showed limited statistical significance (Figure 3B).

Controls on water column nitrogen uptake

Sediment %C was positively correlated to U_{sed} ($r = 0.37$, $p < 0.01$), but not to DN_{sed}, when considering all sediment sources and size fractions. More specifically, we found that the relationship between sediment %C and U_{sed} was positive and of identical effect size across the three sediment sources (Figure 4A), whereas only DN_{sed} rates in streambed sediments showed a positive relationship with sediment %C (Figure 4B). Similarly, at the microcosm scale, the effects of increasing SSC were much more apparent on assimilatory N uptake (U_{micro}) than on denitrification rates (DN_{micro}; Figure 5). U_{micro} showed positive and significant log-log relationships with increasing SSC of clay/silt and sand particles for both agricultural and stream bank sediments (Figures 5A,C). The exponents of the U_{micro}-SSC relationships were very similar between these two sediment sources and slightly higher for the sand fraction (Figures 5A,C). Unlike for agricultural and stream bank sediments, U_{micro} in streambed sediments were not related to increasing SSC of either clay/silt or sand particles (Figure 5E). DN_{micro} rates were only significantly related to increasing SSC of stream bank silt (Figures 5B,D,F). Overall, individual relationships between SSC and water column N uptake changed abruptly when comparing streambed sediments to the other sediment sources, indicating differences in how water column N uptake scales with SSC depending on sediment source. In concordance, ANCOVA models showed significant, positive effects of both sediment source and SSC on assimilatory uptake rates, but not for denitrification (Table 2). For assimilatory uptake, ANCOVA ($R^2 = 0.72$) results showed that the slopes of the uptake *versus* SSC relationships were significantly different among the sediment sources, and particularly between streambed and the other two sources (Table 2).

Discussion

Our microcosm approach attempted to recreate the turbulent and turbid conditions in streams during stormflows to estimate water column N uptake associated with sediment of different sources, and which are mobilized by storm events of different

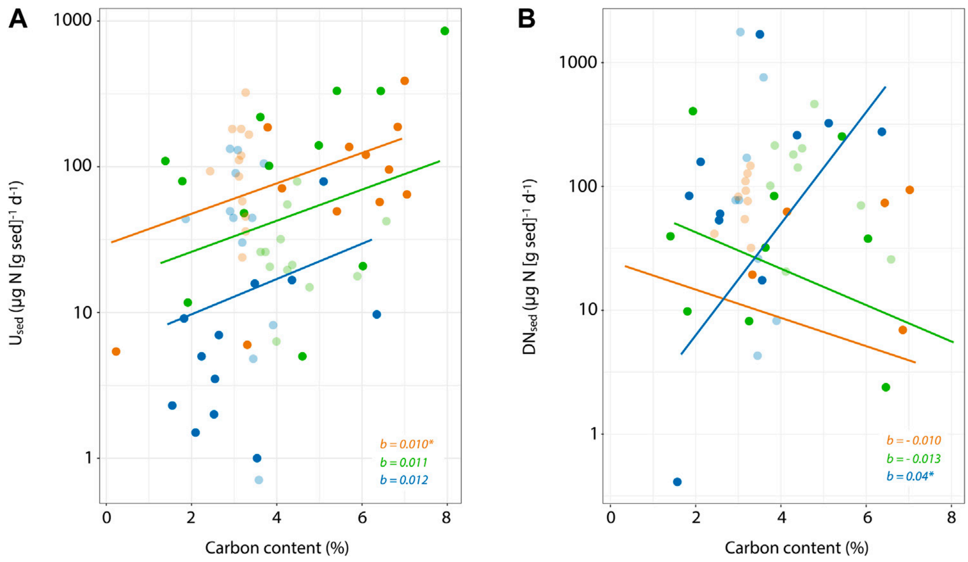

FIGURE 4
Relationships of sediment C content with sediment-specific assimilatory uptake rates **(A)** and denitrification rates **(B)** among sediment sources. Note vertical axis is $\log_{10}$ transformed and that color legend is consistent with previous figures. Slope lines were computed including data from both size fractions for each sediment source, and their values and significance ([blank] *p* > 0.05; **p* < 0.05; **, *p* < 0.01; ****p* < 0.001) are indicated in each panel. DN_{sed} values below our detection limit are omitted in the plot but were considered for the slope calculation.

intensity. We successfully measured assimilatory and dissimilatory N uptake across a range of suspended sediment concentrations that are characteristic of low (<0.3 $mg{\cdot}L^{-1}$), to moderate (0.3–2 $mg{\cdot}L^{-1}$), to very large (>2 $mg{\cdot}L^{-1}$) storms in the Mid-Atlantic region. Sediment-bound C in suspended sediment varied among sediment sources and was directly related to assimilatory N uptake rates, but not to denitrification rates, which were less predictable and more variable. Like others before (Liu et al., 2013; Xia et al., 2017; Reisinger et al., 2021), we generally found a positive and significant relationship between the concentration of suspended sediments and water column uptake; however, our results also showed that water column N uptake scaled differently to suspended sediment concentrations depending on sediment source, and to a lesser degree, particle size. These results are complementary to previous work quantifying whole-reach N retention during stormflow conditions. By comparing predicted and observed NO_3^- fluxes in a watershed's outlet, Wollheim et al. (2017) estimated 65% net retention of nitrate at the network scale during small storm events and no net retention during large storms. They explained the decline in network-scale N retention with storm size due to NO_3^- fluxes increasing at a faster rate (log-log slope >1) with storm runoff at the mouth of the watershed than in its headwaters (log-log slope ≤1). Others have estimated similar values of whole-reach retention during storms (~40%) and attributed it to significant in-stream N demand during stormflows despite shorter water residence time (Bernal et al., 2019). Since low, moderate, and large storms can mobilize different quantities of sediment in the watershed of varying sources and sizes, it is likely that storm size will ultimately modulate the contribution of water column uptake during storm events to whole-reach N retention.

Particle size and chemical analysis in our experiments revealed important differences among sediment sources that can affect water column N uptake. Based on our limited particle size analysis, streambed sediments contained a greater proportion of sand-sized particles than the other two sources, as well as a higher contribution of coarse particles to sediment surface area. Nonetheless, it is important to note that source-specific proportions of fine and coarse materials in suspended sediments will vary as a function of stream discharge (Slattery and Burt, 1998), and their proper characterization was beyond the scope of our study. In our study, we purposely assessed similar SSC gradients of fine and coarse fractions of each sediment source to independently test the effects of increasing surface area on water column N uptake—*i.e.*, given equal SSC, fine particles will provide more surface area than coarse ones. Our results showed no statistical evidence of the expected positive effects of sediment surface area (clay/silt vs. sand) on water column N uptake; although we generally observed similar or greater sediment-specific N uptake rates in clay/silt microcosms than in those with sand-sized particles, with the one exception for assimilatory uptake in stream banks sediments. However, due to methodological difficulties when adding sediment to the microcosms, clay/silt microcosms for stream

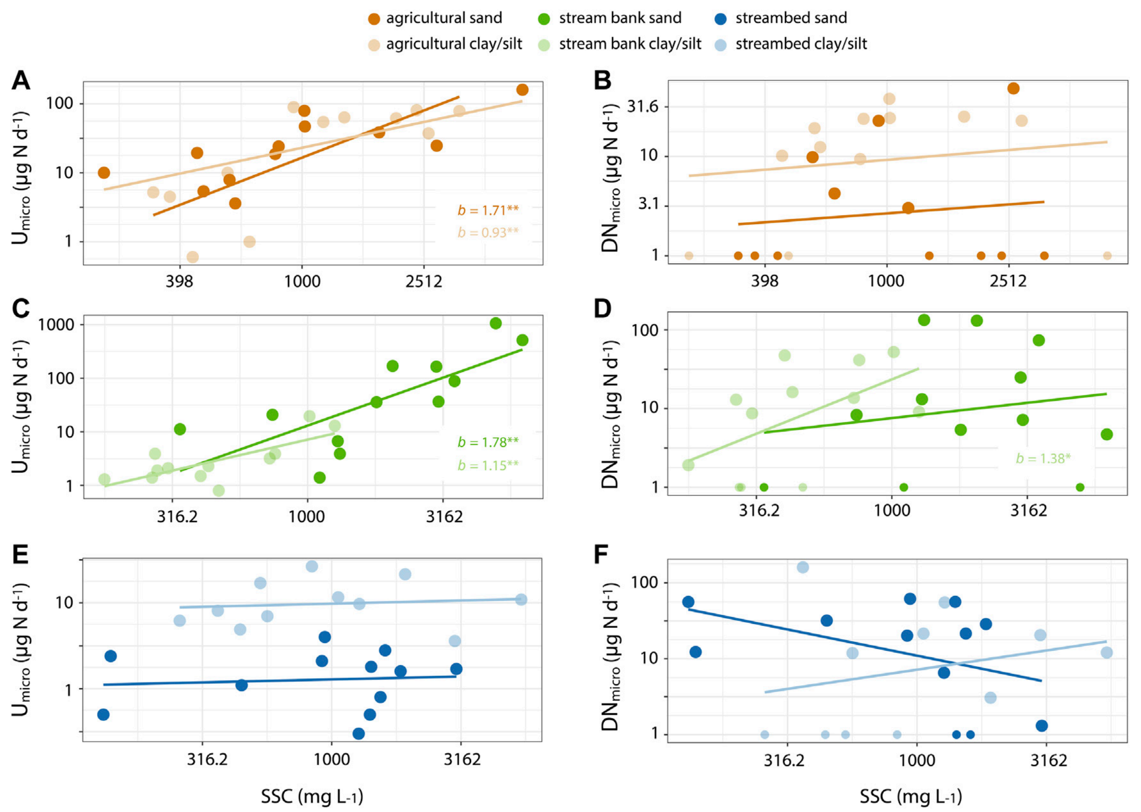

FIGURE 5
Log-log relationships between suspended sediment concentration (SSC) and microcosm-specific rates of assimilatory uptake **(A,C,E)** and denitrification **(B,D,F)** for each sediment source and size. Color legend is consistent with previous figures. Slope values and significance [(blank) $p > 0.05$; $^*p < 0.05$; **, $p < 0.01$; $^{***}p < 0.001$] are indicated in each panel.

bank sediments covered a much narrower range than those with stream bank sand, 177–1,260 and 337–6,216 mg·L^{-1}, respectively. This was unintentional and it was not the case for the other two sediment sources. But it is plausible that the higher N uptake rates in stream bank sand was due to a lower SSC range being tested for the clay/silt fraction.

Beyond differences in particle size, C and N content varied significantly among sediment sources, and they seem to be relevant factors influencing microbial colonization and uptake in suspended sediments. We found higher C:N ratios in streambed sediments than in stream bank or agricultural sediments, along with generally higher C:N ratios of coarser sediments within each source. These results are similar to the negative relationship between particle size and C:N ratios reported by Sinsabaugh and Linkins (1990) in a forested, New England stream and by Zhang et al. (2021), who also suggested that high C:N can constrain bacterial colonization and denitrifying functional genes. Indeed, agricultural soils and stream bank sediments in our study with lower C:N ratios contained higher denitrifying bacterial gene (*nosZ*) abundances than streambed sediments. This is also concordant with the higher *nosZ* abundance in suspended sediments of WCC during one of the highest stormflow on record that very likely mobilized large amounts of hillslope sediment (Kan 2018). However, differences in *nosZ* gene abundance across sediment sources were completely opposite to those of measured denitrification rates, which were highest for streambed suspended sediments. We speculate that this mismatch can be partly explained by the irregular presence and high variation of water column denitrification in streambed sediments, which showed both the highest DN_{sed} values and the largest amount of non-detectable rates compared to the other two sediment sources. High variation in water column denitrification rates have been previously observed in suspended sediments of streambed origin across rivers of contrasting size (Reisinger et al., 2016). Recent work has emphasized the role of heterogeneous anoxic/hypoxic microsites on the activation of anaerobic microbial activity in suspended sediments (Zhu et al., 2018; Schulz et al., 2022). We contend that large variation in denitrification

TABLE 2 Results of two separate ANCOVA models testing the effects of sediment source, size and suspended sediment concentration (SSC) on assimilatory uptake (U_{micro}) and denitrification (DN_{micro}) rates, respectively. All data were $\log_{10}$-transformed prior to the analysis. Bold values indicate significant ANCOVA effects as stated in the table caption.

Factors and covariate	Df	Mean sum sq	F-value	*p*-value
Assimilatory uptake (U_{micro})				
SSC	**1**	**12.8**	**61**	**<0.001**
Sediment Source	**2**	**3.3**	**15.6**	**<0.001**
Sediment Size	1	0.4	1.9	0.165
SSC:source	**2**	**3.6**	**17.3**	**<0.001**
SSC:size	1	0.4	1.9	0.164
Residuals	61	0.2		
Denitrification (DN_{micro})				
SSC	1	0.2	0.4	0.507
Sediment Source	2	0.9	0.9	0.379
Sediment Size	1	0.2	0.4	0.505
SSC:source	2	0.5	1.0	0.362
SSC:size	1	1.7	3.5	0.066
Residuals	61	0.5		

could be attributed to most denitrifying bacteria being facultative anaerobes that can respond rapidly to small-scale and/or short-term heterogeneity in oxygen availability.

Water column NO_3^- uptake varies considerably across rivers of different size, SSC, and N availability [from 0.001 to 363 mg·N·m^{-3} h^{-1} in Reisinger et al., (2015)]. In our microcosm study, water column NO_3^- uptake (U_{micro} + DN_{micro}) showed a narrower range (from 0.1 to 177 mg·N·m^{-3}·h^{-1}), with a high contribution of denitrification, when present, to water column NO_3^- uptake (mean ± SD: 41.7 ± 4.4%). Assuming a stream depth of 1 m, we estimated that microcosm-specific denitrification represented a mean areal rate of 3.3 ± 0.6 mg·N·m^{-2}·h^{-1}, which is very similar to median areal denitrification rates (1.7 mg·N·m^{-2}·h^{-1}) measured in multiple rivers during the warmest months of the year (Piña-Ochoa and Álvarez-Cobelas, 2006). Areal denitrification rates in our microcosm study ranged from 0 to 26.5 mg·N·m^{-2}·h^{-1}, which is comparably higher than the range found by Reisinger et al. (2016) of 0–4.9 mg·N·m^{-2}·h^{-1} in rivers with lower SSC and N availability. In contrast, we found similar sediment-specific denitrification rates to those reported by Reisinger et al. (2016). Therefore, these comparisons most likely highlight the positive effects of high SSC on water column N uptake, as SSCs in our microcosm study were much higher than in any of these previous studies. On the other hand, other studies using microcosms with even higher SSC (up to 20 g·L^{-1}) and 25-day incubations observed tenfold lower rates of water column denitrification than in our study (Liu et al., 2013; Jia et al., 2016). This could be due to the also ten times lower mean number of denitrifying genes they found compared to *nos*Z abundances in our study, or due to the effects of much different incubation times on net N processes. Overall, our results suggest that even during periods of high SSC associated with stormflow conditions, water column denitrification seems to be highly irregular and variable, akin to the patterns previously observed for benthic denitrification. Comparatively, assimilatory uptake in our study responded more strongly to *a priori* predictors of biological activity in suspended sediment such as %OM or increasing SSC than denitrification.

Consideration of other uptake processes besides denitrification is critical within the context of N removal in the water column. Assimilatory uptake can remove a comparatively larger amount of NO_3^- from the water column, which slows downstream N export and can eventually be permanently removed *via* remineralization and coupled nitrification/denitrification (Mulholland et al., 2004; Arango et al., 2008; Tank et al., 2018). Both microcosm- and sediment-specific assimilatory uptake rates were higher for stream bank and agricultural soils than for streambed sediments. When comparing assimilatory uptake rates in clay/silt and sand fractions for each sediment source, only streambed sediments showed significant effects of the greater surface area associated with fine sediments, even though we expected a similar result across all sediment sources. One explanation is that the larger median particle size in streambed sediments (Figure 1) may have resulted in larger differences in surface area between the clay/silt and sand fractions. In other words, the coarse fraction of agricultural and stream bank sediments in our microcosms most likely contained on average smaller sediment particles than the coarse fraction of streambed sediments. The effects of smaller sediment particles (i.e., greater surface area) in agricultural and stream bank sediments could also explain the higher intercepts in the %C-U_{sed} relationships for agricultural and stream bank sources compared to that of streambed sediments (Figure 4).

Similarly, the effects of increasing SSC and surface area on assimilatory N uptake were also much more notable for agricultural and stream bank sources. However, power exponents (i.e., scaling coefficients) in Figure 5 were similar or greater than 1, much higher than the expected ⅔ power exponents for sediment surface area to volume scaling, which indicates that assimilatory N uptake in the water column increased out of proportion with SSC and was likely also depending on additional factors beyond sediment surface area. Further assessment of scaling relationships between increasing SSC and water column uptake is necessary to improve the ability of existing watershed models to characterize N removal during storm events along stream watersheds.

In summary, results from our study suggest that the role of water column uptake on whole-reach N removal may be greater in watersheds with a high presence of agricultural and stream bank sediments that can be mobilized by storm events. Our microcosm study indicates that assimilatory N uptake is positively and nonlinearly related to increasing SSC with varying scaling coefficients depending on sediment sources and size. In our watershed, agricultural soils and stream bank sediments with higher C and N content than streambed sediments, and greater surface area per sediment load, were more reactive to increasing SSC. Accordingly, the contribution of water column N uptake to N retention at the watershed scale may be positively related to the contribution of agricultural and/or stream banks sources to stormflow sediment loads. Our microcosm study provides valuable data on how water column N uptake may scale with increasing storm size; however, more research on how these scaling relationships change across streams of contrasting land use, size, and channel forms is necessary to improve our understanding of water column processes at the watershed scale.

Data availability statement

The datasets analyzed for this study are deposited in the open-access repository, https://github.com/evabacmeister/microcosm2022.

Author contributions

EB, JK, SI, EP, and MP designed the microcosm experiments. EB, SB, and EP collected sediment samples and performed the microcosms experiments. EB and MP analyzed the data and wrote the manuscript with contributions from EP, JK, SB, and SI.

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River engage: Insights on plastic debris polluting the Aturukuku River in Uganda, the Ayung River in Indonesia, and the Connecticut River in the United States

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Introduction: Plastic waste in freshwater ecosystems is increasingly recognized as an economic, ecological, and environmental problem with potential health consequences. This article shares the results of a project to train local stakeholders to collect debris in their communities using scientific methods, then share the results with policymakers.

Methods: Workshops were held in Uganda, Indonesia, and the United States in the spring of 2022. This article presents baseline data from collections on the Aturukuku River in Uganda, the Ayung River in Indonesia, and the Connecticut River in the United States as well as survey results measuring participant attitudes, behaviors, and their perceptions around plastic waste and policy. Surveying participants sheds light on the nuances of perception of the problem and policies to combat pollution at each locale.

Results: We found deposited debris at each riverbank location: Aturukuku River, 0.45 pieces/m2 of which 89.4% was plastic; Ayung River, 7.62 pieces/m2 of which 91.1% was plastic, and the Connecticut River 0.29 pieces/m2 of which 63% was plastic. Environmental attitudes and behaviors were comparable among countries. Participants in all three countries expect plastic will be the most frequently found material.

Discussion: In all cases, perceptions about the kind of debris in their communities corresponds well with collection results. Perceptions around policy solutions included a wide range of solutions, though countries differed in whether solutions addressed the source or the symptoms of the problem; solutions focused more on waste management in Uganda and Indonesia.

KEYWORDS

plastic, freshwater, Uganda, Indonesia, United States, perception, policy, content analysis

Introduction

Researchers recognize plastic waste as a threat to water resources (Teuten et al., 2009; Lechner et al., 2014; Jambeck et al., 2015; Baldwin et al., 2016; Alshawafi et al., 2017; Blettler et al., 2017; Cable et al., 2017; Blettler et al., 2018; van Emmerik et al., 2019a; Battulga et al., 2019; Blettler and Wantzen 2019; Castro-Jiménez et al., 2019; Azevedo-Santos et al., 2021). Decades of research on the topic of what was initially called 'marine debris' indicates that it has a significant negative impact on ecosystems (Barnes et al., 2009; Teuten et al., 2009; Nkwachukwu et al., 2013; Rochman et al., 2016), wildlife (Ryan 1989; Laist 1997; Kuhn et al., 2015; Reynolds and Ryan 2018), and economies (McIlgorm et al., 2011; Newman et al., 2015). Plastics, often the most frequently found item in cleanups, have been found in human blood (Leslie et al., 2022), meconium and placenta (Braun et al., 2021), and human lung tissue (Amato-Lourenço et al., 2021). While the potential health impacts for humans are poorly understood, it is clear that plastic pollution significantly impacts life on Earth. This research builds knowledge of freshwater macro plastic pollution through first analyses of the Aturukuku River in Uganda, the Ayung River in Indonesia, and the Connecticut River in the United States. These rivers are valuable culturally and socially. In Indonesia, the Ayung River is the longest river on Bali island, which is called "The Island of Gods" because of its high value in religious and cultural matters. This river which flows across the island, holds specific cultural, agricultural, and tourism importance for the locals. The river Aturukuku in Tororo, is one of the few existing small riverine ecosystems in Eastern Uganda, and is currently an important resource supporting local fisheries, harvesting of craft materials, crop irrigation, and livestock rearing by the riparian communities, especially the rural poor and those who are ecosystem dependent. In the United States, the Connecticut river is the New England region's longest river and notable as an American Heritage River, a designation that recognizes its importance for nature, the economy, agriculture, history, culture, and recreation. Taking an interdisciplinary approach, we seek to better understand debris density at these sites as well as the perceptions of workshop participants around debris and policy.

Why freshwater and macro plastics?

Experts note the relative dearth of studies on freshwater plastic pollution when compared to research on marine environments (Sigler 2014; Blettler et al., 2017; Sruthy and Ramasamy 2017; Vincent and Hoellein 2017; Blettler et al., 2018; Reynolds and Ryan 2018; van Emmerik et al., 2019a; Blettler and Wantzen 2019; van Emmerik et al., 2019b; van Calcar and van Emmerik 2019; van Emmerik and Schwarz 2020; Meijer et al., 2021). Researchers have understood for some time that rivers may serve as a pathway connecting litter from land-based communities to marine environments (Jambeck et al., 2015; Lebreton et al., 2017; Schmidt et al., 2017). But rivers are not simply a mode of transferring debris, they are also deposition sites that hold plastic over years or decades as it degrades (McCormick and Hoellein 2016; Blettler and Wantzen 2019).

Several studies estimate plastics accumulation by combining variables including (high) population density, (lack of) waste infrastructure, and hydrological modeling (e.g., Jambeck et al., 2015; Lebreton et al., 2017; Schmidt et al., 2017). Lebreton et al. (2017) predict that the twenty most-polluting rivers exist in Asia (fifteen rivers: 75%), Africa (three rivers: 15%), and South America (two rivers: 10%). Further investigations of river systems may shed light on the so-called 'missing' plastic problem, or the difference between known plastic inputs compared to outputs in the environment (Cózar et al., 2014; Schmidt et al., 2017; Willis et al., 2017). Researchers point to the importance of assessing river debris (in water, sediment, and shorelines) to fully understand debris flows, better justifying these estimates through field observations (Gasperi et al., 2014; Lechner et al., 2014; Blettler and Wantzen 2019; Castro-Jiménez et al., 2019; Mihai et al., 2022). Several scientists recommend research emphasizing coastal regions with high population density (Vince and Hardesty 2017; Jambeck et al., 2018). Blettler et al. (2018) promote research on the world's most polluted rivers, namely in places that feature both quickly developing economic systems and a lack of waste management. Blettler and Wantzen (2019) suggest a focus on macro debris originating as "mismanaged[1] household solid waste;" they note that the scholarly emphasis on micro-plastics is an export from the global North to the global South: scientific imperialism that does not focus on the core local problem (p. 1). Not only is macro debris important to study for its emphasis on local context, Willis et al. (2017) note that land-based interventions at the local level will be more successful at preventing debris from entering the ocean at all. As such, more

1 While the term 'mismanaged waste' is a regular feature in the literature, we take issue with the idea that the plastic pollution found in global waterways is merely "mismanaged." This term implies that under different waste management conditions, the problem would be solved. Instead, under the best waste management, plastic is burned or buried: both are detrimental to the environment. Even eell-managed waste systems cannot address the core problem: that single use plastics were designed to be used once and thrown 'away.'

data on macro debris in freshwater systems is critical to understanding how plastic pollution harms freshwater and marine systems.

River debris studies in Uganda

Sadan and De Kock (2021) write that without intervention, both the production and the consumption of plastic will increase not only in Africa but across the globe in the coming decade. The African rivers included in the top twenty of Lebreton et al.'s (2017) list are the Cross River in Nigeria and Cameroon, the Imo in Nigeria, and the Kwa Ibo, also in Nigeria. Nigeria is the most populous country in Africa (with 225 million inhabitants) and the sixth most populous country worldwide (CIA 2022). The population density of Nigeria was 232 persons per square kilometer in 2021 (World Bank 2022). In comparison, Uganda has far fewer inhabitants (46.2 million) and a comparably high population density (235 individuals per square kilometer) (CIA 2022; World Bank 2022). No rivers in Uganda are included in Lebreton et al.'s (2017) list.

In a systematic literature review, Akindele and Alimba (2021) analyze fifty-nine plastic pollution studies from across the continent of Africa ranging in time from 1987 to 2020. They find that east African countries have the fewest studies while southern Africa has the most (Akindele and Alimba 2021). The majority of the studies (71%) focus on marine or estuarine systems while fewer (29%) center on freshwater habitats (Akindele and Alimba 2021). There are no studies of Ugandan plastic debris in freshwater systems. They call for more research, particularly in East Africa, North Africa, and West Africa (Akindele and Alimba 2021). For an in-depth review on microplastic research on the African continent, see Alimi et al. (2021); for research on both macro and micro plastic in Africa, see Akindele and Alimba (2021).

There are a few studies of microplastic in East Africa, including a study of plastic from the stomachs of Nile perch and Nile tilapia in Lake Victoria (Biginagwa et al., 2016), analysis of microplastic from the surface of Lake Victoria (Egessa et al., 2020b), documentation of ingestion of microplastic by zooplankton on Kenyan coasts (Kosore et al., 2018), and an analysis of microplastic abundance and composition from the surface waters of Kenya's Lake Naivasha (Migwi et al., 2020). Analyzing micro and macro particles in fish and sediment from Ethiopia's Lake Ziway, Merga et al. (2020) find plastic ingested by 35% of fish sampled and a median of 30,000 plastic particles/m^3 of sediment.

Due to a lack of macro freshwater research in Uganda, this review focuses on macro plastic studies across the African continent. Trawling the seafloor near Morocco, Loulad et al. (2017) found 603 kg of macro debris. Plastic was found in 54% of their trawling stations and made up 34.4% of the total weight of materials found and 83.6% of total number of items counted (Loulad et al., 2017). The most frequently found plastic material were displaced traps for *Octopus vulgaris* (Loulad et al., 2017). Working in a coastal wetland in Morocco, Alshawafi et al. (2017) found 57% of the macro debris recorded in their study was plastic. The authors attribute the debris to tourism, land-based usage, and commercial fishing (Alshawafi et al., 2017). Madzena and Lasiak (1997) investigate an undeveloped beach in South Africa, finding that plastic make up 83% of the debris by count and 47% by weight. It should be noted that at the time, researchers did not necessarily distinguish between macro and microplastic. Ebere et al. (2019) working on (microplastic and) macro debris research in Nigeria, find a total of 3,487 macro debris items (across five locations of 1,000 m^2 each) of which 59% were plastic. Their work indicates a significant relationship between macro-debris and microplastic abundance, suggesting microplastics are formed as macro plastics break down rather than distributed in another way (Ebere et al., 2019). They recommend improved management of waste, increased recycling, and consequences for those engaged in illegal dumping (Ebere et al., 2019). Egessa et al. (2020a) investigate micro-, meso-, and macro plastic in sediments and on shorelines of the northern side of Lake Victoria. They note that rates of accumulation are higher at fish landing beaches compared to recreational beaches and recommend focusing management at these sights (Egessa et al., 2020a). Ngupula et al. (2014) survey 68 stations by trawling in Lake Victoria in 2013, finding 44% (by weight) is multifilament gillnets, 42% monofilament gillnets, 7% longlines and hooks, 4% plastic bags, 2% floats, and 1% clothing; the authors attribute the waste to fishing, human activities, and transportation.

Research on plastic accumulation in African countries at times centers on ruminants ingesting debris—often plastic bags or polythene-causing impaction (Ramaswamy and Sharma 2011; Akinrinmade and Akinrinde 2013). This is sadly a phenomenon across the developing world (Priyanka and Dey 2018). Akinrinmade and Akinrinde (2013) note this impaction may be exacerbated by high numbers of livestock, lack of fodder, and the prevalence of dumping. Through abattoir surveys in northern Nigeria, Remi-Adewumi et al. (2011) find 80.9% of sheep and 19.1% of goats in their sample have foreign material within the rumen in their stomachs, most commonly plastic. Analyzing animals after slaughter at an abattoir in Ethiopia, Abebe and Nuru (2011) report that of 768 sheep and goats (384 each) 6.1% are positive cases of rumen or reticulum impaction, with plastic as the foreign substance in 59.6% of the positive cases.

Wandeka et al. (2022) note the complexities of plastic packaging in Uganda, where it is essential to maintaining quality, particularly for foods. The authors report that from 1994 to 2017 Uganda imported 1.9 million tons of raw and finished plastic (totals for the whole of the continent during this time is 117 million tons) (Wandeka et al., 2022). Akindele and Alimba (2021) concur that the African continent is the destination of many plastics manufactured globally, that they lack waste infrastructure, and that African countries are least studied. A 2018 article by Jambeck et al. names the usual suspects of lack of infrastructure and high population growth rates contributing to this important challenge. Focusing on Kenya, Rayne (2008) describes some of the challenges across Africa, namely lack of waste or sewage infrastructure as well as lobbying from plastics manufacturers to maintain access to plastic bags. In conclusion, the global problem of plastic pollution is significant across the African continent. There are many countries, coastlines, and freshwater systems with little or no data. Overall, studies tend to focus on a few areas and emphasize coastal and microplastic pollution. This research centers on macroplastic in the freshwater environment and shares data from an as yet unstudied river in Uganda: the Aturukuku.

River debris studies in Indonesia

Lebreton et al. (2017) include four river systems in Indonesia in their predictive list of top-twenty most polluting rivers: the Brantas, the Solo, the Serayu and the Progo. All can be found on Java, the most populous island on Earth. Indonesia is the second most populous country in East and Southeast Asia behind China and the fourth most populous country in the world with 277 million inhabitants (CIA 2022). The population density of Indonesia is 147 people per square kilometer (World Bank 2022). The Ayung River, the focus of this study is found on Bali and is not included in Lebreton et al.'s (2017) top twenty list.

Regarded as a 'hot spot' for plastic pollution, there are more studies in Indonesia as compared to Uganda, but like the bulk of studies from the African continent, Indonesian studies emphasize microplastics and coastal or marine settings (see Purba et al., 2019 and Vriend et al., 2021 for excellent overviews of studies across Indonesia) instead of macro debris and freshwater systems. As with our Ugandan review, we focus on macroplastic and freshwater analyses.

Scenario-modeling waste distribution patterns throughout Indonesia to guide policy, Sakti et al. (2021) estimate freshwater inputs for various regencies as 0.65–11.9 tons of plastic waste per day (low to high scenarios). The authors recommend using these data to name priority zones for management (Sakti et al., 2021). Surveying a number of points on the Pesanggrahan and Grogol Rivers (Java) using floating booms, Sari et al. (2022) find 74% of the litter in their sample is plastic. The authors calculate roughly 9.9 g of plastic per person discharges per day in the rainy season for these two rivers (Sari et al., 2022). They recommend more studies, effective cleaning, and strategies to prevent litter (Sari et al., 2022). Pamungkas et al. (2021), studying plastic flows in the Citarum River (Java) find an influx of 24,813.7 items per cubic meter, with the most commonly seen material being thin plastic wrap and foamed plastic. Studying visible plastic debris on the shore of the Madura Strait of the Wonorejo River (Java) estuary, Kurniawan and Imron (2019) track debris seasonally, finding accumulation is significantly greater in the rainy season. The researchers note that low-density polyethylene is the most abundant plastic (73.1%) in their dry season samples while polyethylene terephthalate is the most abundant plastic (59.8%) in their rainy season samples (Kurniawan and Imron 2019). They recommend cleanups in the rainy season to maximize efficiency (Kurniawan and Imron 2019). Studying riverbank debris of the Lower Citarum River (Java) Hidayat et al. (2022) sample quadrats of 30 × 30 × 10 cm^3 to analyze plastic composition. The researchers find plastic in all samples ranging from 0.7 to 301 g per 9,000 cm^3 quadrat and recommend improving management of waste, fewer single use plastics, and improved recycling to combat the problem.

Honingh et al. (2020) study plastic waste accumulation at trash racks, which may increase risk of flooding. Using flume experimentation, the authors analyze how waste from plastic and other debris may impact trash rack blockage, finding plastic has higher blockage density when compared to organic waste (Honingh et al., 2020). In addition to modeling experiments, the researchers performed fieldwork at the Cikapundung River (Java) a headwater of the Citarum, finding approximately 100 kg of which plastic make up from 11% to 78% of the samples by weight. Plastic bags (57%), food packaging (21%), and plastic cups (16%) dominate the sample (Honingh et al., 2020). Cordova et al. (2022) also research the Citarum River (Java) outputs to the ocean, approximating 6,043 ± 567 pieces (1.01 ± 0.19 tons by weight) of macro debris release each day. Monitoring outputs from nine rivers into Jakarta Bay (Java) Cordova and Nurhati (2019) determine plastic is the most frequently polluted material, comprising 59% of the sample by piece and 37% of its weight. Their analysis finds an average daily "release of 97,098 ± 28,932 items or 23 ± 7.10 tons into Jakarta Bay" of which they determine "8.32 ± 2.44 tons" per day are plastic (Cordova and Nurhati 2019, p. 1). While an incredible amount of debris, the authors note this is "8–16 times less than global-scale model estimates" (Cordova and Nurhati 2019, p. 1). Also working in Jakarta Bay (Java) Dwiyitno et al. (2020) find a concentration of 10,300 plastic items/km^2 in the wet season and 7,400 plastic items/km^2 in the dry season, with packaging and consumer products made of plastic representing the most abundance in the sample. In two articles modeling macro debris accumulation in Java's Jakarta Bay, Jasmin et al. (2019, 2020) find accumulation in the rainy season in the eastern part of the Bay and accumulation in the dry season on the western and eastern areas of the Bay.

Researching Pantai Indah Kapuk mangrove (Java) Hastuti et al. (2014) note plastic is the most frequent item they sample (77.7%) followed by foamed plastic (18.1%) which is, of course, also plastic; the authors recommend restoration and widening of the mangrove ecosystem to improve conditions. Tracking outputs of local rivers into Banten Bay (Java) Rahmania et al.

2 This is wet weight.

(2021) find plastics originating in the Cibanten River drift west to the Sunda Strait in the east monsoon and east to Jarkarta Bay in the west monsoon. Their analysis leads to a recommendation of local governmental cooperation to control both unmanaged and managed solid waste sites (Rahmania et al., 2021). Bridge sampling emissions of macro-plastics on the Ciliwung, Pesanggrahan, Sunter, and Cakung rivers (Java) in Indonesia, van Emmerik et al. (2019a) estimate that 2.1×10^3 tons of plastic waste travels *via* rivers and canals in Jakarta to oceans annually. Analysis of Ciliwung, Pesanggrahan, and Banjir Kanal Timur rivers (Java) in Indonesia by van Calcar and van Emmerik (2019) indicate an average of over 10^4 plastic items per hour for the Ciliwung, just under 10^4 plastic items per hour for the Pesanggrahan, and well over 10^3 plastic items per hour for the Banjir Kanal Timur, with seasonal variation. They find high rates of soft polyolefin in these Indonesian rivers (van Calcar and van Emmerik 2019).

An analysis of the Musi River (Sumatra) by Maherlsa et al. (2019) use a net/manta system to survey floating debris at ten stations along the river. The researchers discover 87 macro debris items, with a range of 1–33 items found per station; plastics comprise 86.2% of the sample (Maherlsa et al., 2019). Working with the point intercept transit method in the Musi River of South Sumatera (Sumatra), Almiza and Patria (2021) find an abundance range of 5–32 items/m^2 and 27.8–126.9 g/m^2. Their most frequently sampled items are "plastics fragments, food wrappers, other jugs/containers, bags/films, . . . cups, . . . [and] bags/films" (p. 1). Owens and Kamil's (2020) study is the only example of freshwater analysis for macro debris on Bali published to date. This research on two sites along the riverbanks of the Tukad Badung finds at the floodplain site 598 pieces of debris weighing 14.8 kg (or 1.19 pieces/m^2 and 0.029 kg/m^2), 92.8% of which was composed of plastic and at the transition zone site 147 pieces weighing 3.58 kg (or 0.58 pieces/m^2 and 0.014 kg/m^2) of which 88.4% was plastic (Owens and Kamil 2020). When analyzing research of freshwater macro plastic pollution in Indonesia, it is clear most analyses are centered on Java, with limited research in Sumatra or Bali and no published research tracking freshwater macro debris inputs on smaller islands.

River debris studies in the United States

The population of the United States is 337 million people, and the population of the state of Connecticut is 3.6 million people (CIA 2022; U.S Census 2022). Connecticut is the fourth most densely populated state in America, with 286 inhabitants per square kilometer (Statista, 2022). Lebreton et al. (2017) do not include any rivers in North American in their predictive list of top-twenty most polluting rivers. As there are no studies of the Connecticut river (as is the case with the Aturukuku and the Ayung) this review focuses on broader freshwater studies of macro plastic in the United States.

There are studies of US freshwater systems focusing on microplastics (e.g., Baldwin et al., 2016 in the Great Lakes Tributaries, Barrows et al., 2018 in the Gallatin River watershed, and Cable et al., 2017 in the Great Lakes) and more comprehensive reviews of freshwater research on microplastics with US examples (see Peller et al., 2020). Many reviews on microplastics in freshwater include little data on the United States though Bellasi et al. (2020) incorporate some data on tire wear particles and wastewater treatment plants. Baldwin et al. (2016) assess floating micro and macro debris across 29 tributaries of the Great Lakes in six US states, finding plastics in all of their 107 samples, of which 98% are microplastic. The authors attribute many of the plastics including fragments, pellets, foams, and films to urban runoff events though this is not true of fibers (Baldwin et al., 2016).

Carpenter et al. (1972) note the presence of polystyrene spherules in samples from Niantic Bay, Block Island, Long Island Sound, and the Great Salt Pond on Block Island. Collecting at the outlet pipes of factories in Massachusetts (the Chicopee River, the Connecticut River) and at the mouth of the Connecticut River in Saybrook, Hays and Cormans (1974) find similar polystyrene spherules at both locations. They find cylindrical polyethylene particles at additional factory sites in Ludlow Massachusetts, Stonington Connecticut, and Gilman (Thames River) Connecticut as well as sites in New York and New Jersey (Hays and Cormans 1974). Poletti and Landberg (2021) find 14,520 pieces of debris, 56.4% of which is polystyrene in their assessment of the Mill Creek of the Blue Heron Nature Preserve in Atlanta, Georgia. Analyzing the Los Angeles and San Gabriel Rivers (Moore et al., 2011) estimate an average 2.3 billion pieces of plastic (30 metric tons) flow through these rivers every 72 h with foams being most abundant (71%). Working on the Long Island Sound at Meig's Point, eight miles west of the Connecticut river's outlet and at Bluff Point State Park on the Poquonock River in Connecticut, Owens (2018) finds 1,623 individual pieces of debris weighing 19.4 kg of which 61.5% by piece and 16.2% by weight is plastic. In a review of studies on the Laurentian Great Lakes, Driedger et al. (2015) found litter along shorelines is predominantly plastic (>80%) while density in water is comparable to that of the ocean gyres. Vincent and Hoellein (2017) studying Lake Michigan's Pratt Beach collect 79,915 items and primarily attribute the material to litter and its accumulation. One comprehensive study of 15 sites in five rivers in Illinois and Indiana (Salt Creek, Turkey Creek, North Branch of the Chicago River, Hickory Creek, Plum Creek) finds that while in the riparian zone debris density compares to global beach averages, in benthic zones density is higher for riverine than marine environments (McCormick and Hoellein 2016). As such, this review indicates that there have been few analyses of the Connecticut River since the 1970's and there is a dearth of freshwater macro pollution studies in the United States in general.

Experts have weighed in for decades on solutions for the plastic pollution problem, whether touting education, legislation, or some combination of the two (Derraik 2002; Sheavly and Register 2007). Cleanups are valuable tools to improve awareness, influence behavior, and provide context to this complex global issue (5IMDC 2011; Owens 2018). Our analysis includes not only baseline data on three rivers (the Aturukuku, the Ayung, and the Connecticut) but also an analysis of surveys of workshop participant environmental attitudes and behaviors as well as asking how they conceptualize the problem of and the solution for plastic pollution in their home communities. According to Wandeka et al. (2022) many of the solutions presented in the Ugandan system stem from "the private sector and plastic recycling businesses" (p.19) Wandeka et al. (2022) recommend more support from government, increased regulations on imports, and local solutions for packaging and use. Nkwachukwu et al. (2013) recommend redesigning plastics to alleviate the problem, but note the importance of coupling bioplastic design with proper labelling, outreach, and education. The authors astutely point out "that the concept of degradable plastics has been oversold as a solution to the waste disposal problem" (Nkwachukwu et al., 2013 p.12). They also recommend creating infrastructure for disposal and recycling of waste (Nkwachukwu et al., 2013). Sadan and De Kock (2021) note that policy fragmentation exists at all levels and recommend that African nations address the problem globally (contributing to the development of a global treaty), regionally (aligning global policy for the regional context), and nationally (considering their own priorities and challenges). Jambeck et al. (2018) find that African communities show greater propensity toward recycling, remaking, and reusing materials in creative ways, finding that community-based solutions to the problem may be particularly apt in African countries. Mihai et al. (2022) find that whether discussing macroplastic in coastal or freshwater systems and in developed or developing countries, the linear economy model will yield waste mismanagement problems; they recommend the circular economy model "as a key mitigation strategy in the prevention of plastics materials, improvement of the production sector, and providing better waste management practices to reduce this global environmental threat" (p. 242). These authors rightly note that waste management in itself is not the solution, as developed countries with ample management of waste produce plastic pollution of both freshwater and marine systems (Mihai et al., 2022).

While the number of studies of freshwater systems is increasing, there are as yet no studies including data from the Aturukuku River in Uganda, the Ayung River in Indonesia, or the Connecticut River in the United States. In addition, we have found no studies that compare perceptions in these three countries to better understand the nuances around plastic pollution globally. This research provides additional information about three river systems, the Aturukuku River in Uganda, the Ayung River in Indonesia, and the Connecticut River in the United States and the people who live and work nearby.

Materials and methods

This interdisciplinary research approach seeks to create a baseline report for sites along each of the three rivers, and to highlight how communities understand the problem of plastic pollution, its sources, and its solutions. Maps of the Aturukuku, the Ayung, and the Connecticut River can be seen in Figure 1. At each study site (Uganda, Indonesia, United States) members of the research team led 2-day workshops for local residents. The goal of the workshop was to inform participants about the issue of freshwater debris, train participants on the cleanup methods, and share strategies for communicating results to policymakers. In addition to debris collections at each site, participants were surveyed before the workshop, at the end of the workshop, and 2 months after the workshop.

Debris collection

Researchers at each site complete an analysis using a river collection method (Owens and Kamil 2020) modeled on the National Oceanic and Atmospheric Administration (NOAA) Marine Debris Shoreline methodology (Opfer, Arthur, and Lippiatt 2012). Researchers use survey flags to demarcate an area along the shore of the respective river, marking off a 100 m long by 5-m wide (landward from the river) portion of shoreline for a total collection area of 500 m^2. In the Indonesian case, it was not possible to find an accessible shoreline area with 100 consecutive meters. Therefore, the Indonesian team opted for a shorter collection area of 16 m in length. After flagging the study area, researchers walk a precise pattern dictated by NOAA, covering the complete area by walking back and forth, scanning from side to side in order to collect all macro debris visible within the given area that is attributable to humans. As described in Owens and Kamil (2020) and Owens et al. (2022), this method combines the NOAA methodology for a standing stock survey (which typically covers 100 m of shoreline but does not include removing debris) and the accumulation method (which includes the entire shoreline and does include removing debris). This practical combined approach allows removing debris from a limited area to take a snapshot of accumulation of plastic and other debris. Researchers collect the debris and log it onto data collection sheets by type of material. The debris from a limited portion of shoreline is removed using this replicable, inexpensive, scientific method to serve a broader goal of informing policymakers and community members about the problem in a local context (Owens et al., 2022).

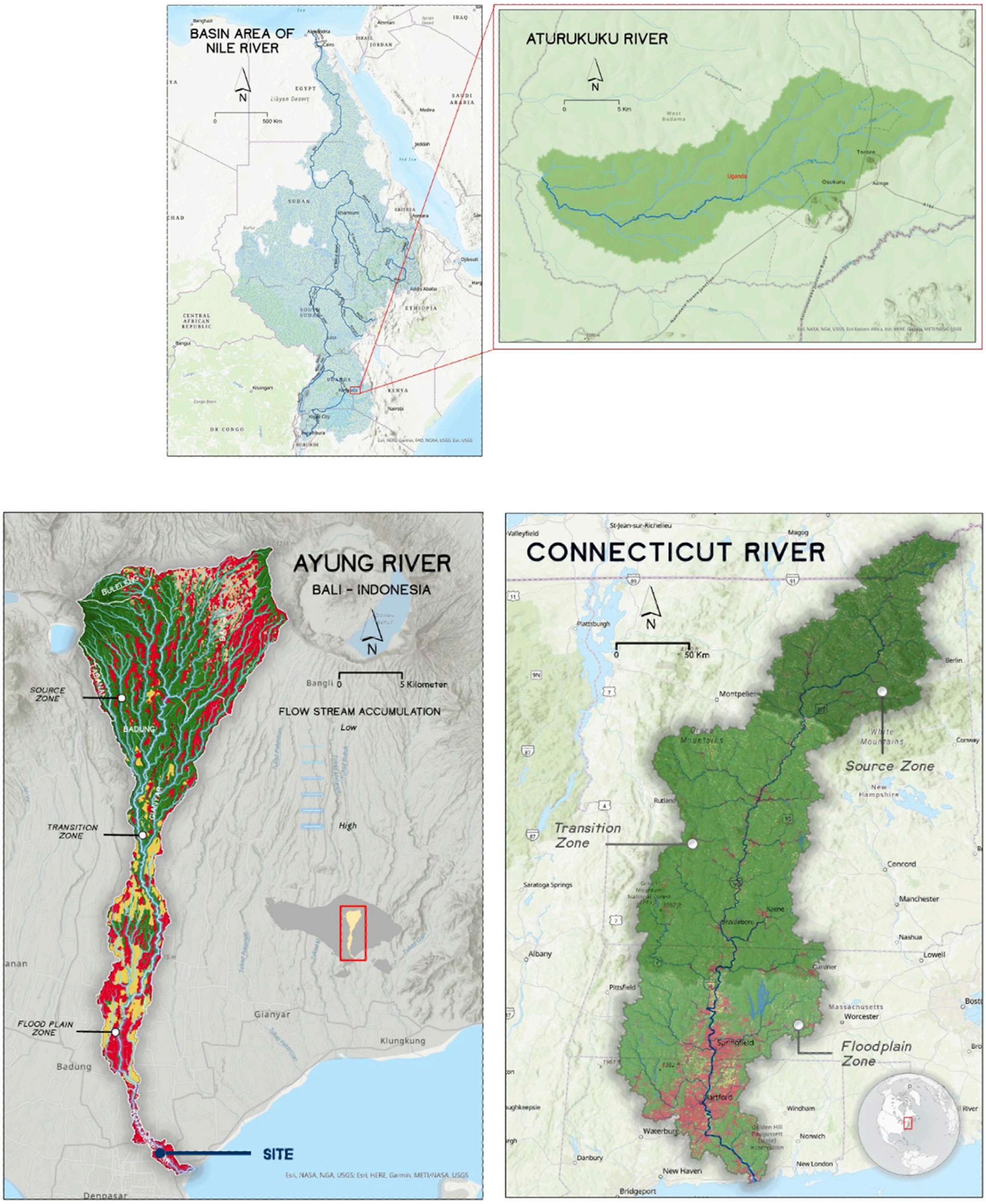

FIGURE 1
Maps showing the locations of the Aturukuku River in Uganda, the Ayung River in Indonesia, and the Connecticut River in the United States (Map credit: Muhammad Azmi).

Surveys

We measure participant environmental attitudes using the New Ecological Paradigm (NEP) Scale (Dunlap et al., 2000) which is used globally to shed light on "the impact of educational programs on environmental world views" (Anderson, 2012), p. 260. Self-reported environmental behaviors are measured *via* the Environmentally Responsible

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TABLE 1 Data about collection sites, with number of items collected, pieces/m², total weight, and kg/m².

Location	River	Coordinates decimal degrees	Area meter²	Date	Participants	Items collected total items (pieces/m²)	Total weight² kilograms (kg/m²)
Uganda	Aturukuku	.8194444, 34.2277778	500	19 March 2022	11	226 (.45)	7.15 (.014)
Indonesia	Ayung	−8.638729, 115.241841	80	2 April 2022	15	610 (7.62)	78.68 (.98)
United States	Connecticut	41.520483, −72.558784	500	11 April 2022	9	146 (.29)	3.80 kg (.007)

TABLE 2 Collected material by type from three sites in Uganda, Indonesia, and the United States; in parentheses find the percentage of the total amount of accumulated debris at each site.

River with location	Plastic	Metal	Glass	Rubber	Processed trees	Cloth, clothes, fabric	Natural materials	Other
Aturukuku Uganda	202 (89.4%)	0	0	1 (.4%)	0	18 (8.0%)	5 (2.2%)	0
Ayung Indonesia	556 (91.1%)	2 (0.3%)	0	1 (.2%)	0	31 (5.1%)	0	20 (3.3%)
Connecticut United States	92 (63.0%)	14 (9.6%)	31 (21.2%)	1 (.7%)	4 (2.7%)	1 (0.7%)	0	3 (2.0%)

Behavior Index (ERBI) described by Thapa (1999) and developed by Smith-Sebasto and D'Costa (1995). This instrument has a high internal consistency reliability (.94), and high validity (82%) (Smith-Sebasto and D'Costa 1995).

To shed light on perceptions around the problem of pollution in freshwater, we ask respondents about the items they expect to find in their cleanup and policies that can address the problem. We analyze responses using content analysis to look for patterns of meaning. When classifying responses into more discrete categories, we err on the side of counting an item as 'indeterminate' rather than ascribing meaning to it. For example, if a respondent writes they expect to see "straw" in the river, we do not assume this is a single use plastic straw. If they write "shoe" we cannot determine if they mean flip-flop (aka thong) made of plastic or a leather or cloth shoe, therefore we count it as indeterminate. In this way, we would rather under-report than over-report perceptions ascribed to respondents.

Why use qualitative methods to understand the problem of plastic pollution? As we hail from different societies with different cultures, as well as social and economic systems, we did not want to make presumptions about the conceptualization of the problem, the policy, or the solutions by respondents. Instead, we ask them to share the problem, the policy, and the solutions in their own words. Using an inductive, social-constructivist method we use content analysis to systematically observe, review, and pull meaning from the responses. This allows us to remove some of our own biases and conceptions from the analysis. The categories to which we subdivide responses vary by study area, as each derives from the responses themselves. In other words, we do not impose one set of categories on all three study sites, but instead build categories based on responses. By asking local participants about the problem, the policy, and the solutions, we highlight their perceptions and understanding of the issue, creating a unique profile for each community. We then compare these to draw meaning about the global problem of freshwater pollution. Please note: While participants share their results with local policymakers, this study is not configured to measure the impact of their advocacy on policy over time. Policymaking is a slow, incremental process. Work training local stakeholders to understand their local resources and advocate for them may improve policy in the future, but it is outside the scope of this project to measure it.

Results

Debris collection

Basic information about each site and collection can be seen in Table 1. Debris was present at all three riverbank sites, with the Ayung River's debris density (7.62 pieces/m²) an order of magnitude higher than the Aturukuku (0.45 pieces/m²) and the Connecticut (0.29 pieces/m²). Plastic was the most frequently found item at each site, ranging from 63.0% to 91.1% of the samples. Material by category can be seen in

FIGURE 2
Proportional distribution of material found in the Aturukuku River, Uganda, the Ayung River, Indonesia, and the Connecticut River, United States with material type (count, percentage).

TABLE 3 Comparative results of the New Ecological Paradigm.

	Pre-test results	Two months post-test results
Uganda	(n = 13) M = 3.65, SD = 0.92	(n = 9) M = 4.06, SD = 1.00
Indonesia	(n = 15) M = 3.85, SD = 0.86	(n = 14) M = 3.24, SD = 3.47
United States	(n = 8) M = 4.05, SD = 0.68	(n = 4) M = 4.40, SD = .64

TABLE 4 Comparative results of the Environmentally Responsible Behavior Index.

	Pre-test results	Two months post-test results
Uganda	(n = 13) M = 3.36, SD = 1.05	(n = 9) M = 3.86, SD = .65
Indonesia	(n = 15) M = 3.48, SD = 0.81	(n = 14) M = 3.13, SD = 3.47
United States	(n = 8) M = 3.84, SD = .64	(n = 4) M = 4.18, SD = .53

Table 2 and proportional distribution of each sample by site is shown in Figure 2.

Survey data: Environmental attitudes and behaviors

The New Ecological Paradigm (NEP) Scale (Dunlap et al., 2000) measures environmental attitudes while the Environmentally Responsible Behavior Index (ERBI) measures self-reported environmental behaviors. Participants respond to questions about their attitudes or behaviors on a scale ranging from rarely to usually. Their answers are given a score ranging from 0 = Not applicable, 1 = Rarely, 2 = Occasionally, 3 = Sometimes, 4 = Frequently, and 5 = Usually. Here we share the mean scores of the sample and the standard deviation. Due to the low sample sizes, we have opted not to administer statistical analysis on the responses. Results can be found in Tables 3 and Table 4.

Survey data: Perception of content of debris in Uganda

Ugandan respondents were asked: *Please list the top five most frequently found items in the Aturukuku River*. Supplementary Materials Table 1 compiles and categorizes responses for the Aturukuku river in Uganda. Figure 3 shows a comparison of participant responses by type of material for each survey period. Please note the changing sample size between the pre-test (n = 61), the post-test (n = 55), and the 2 months post-test (n = 45).

Survey data: perception of content of debris in Indonesia

Indonesian respondents were asked: *Please list the top five most frequently found items in the Ayung River.*

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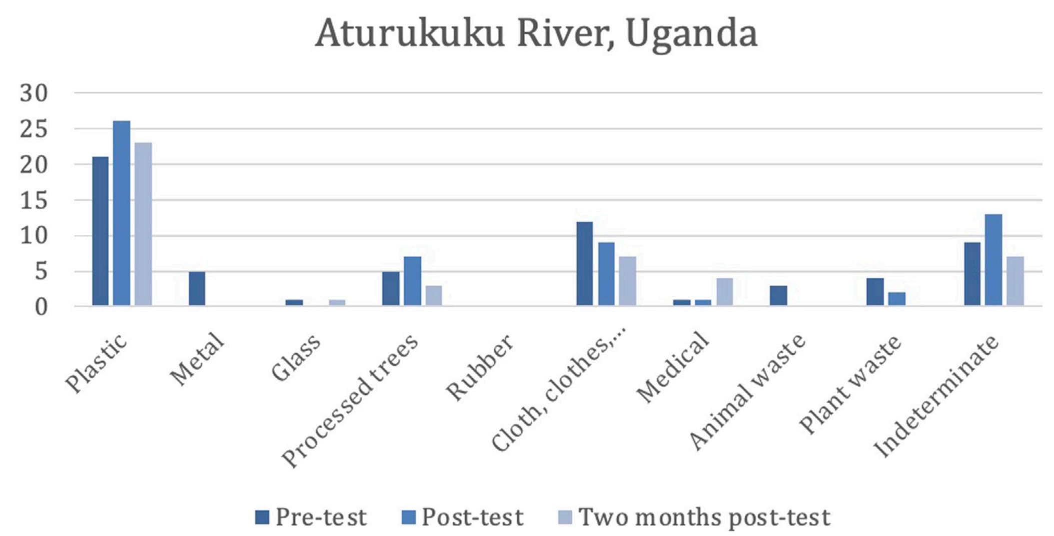

FIGURE 3
Participant perceptions of the most frequently found items in the Aturukuku River by type.

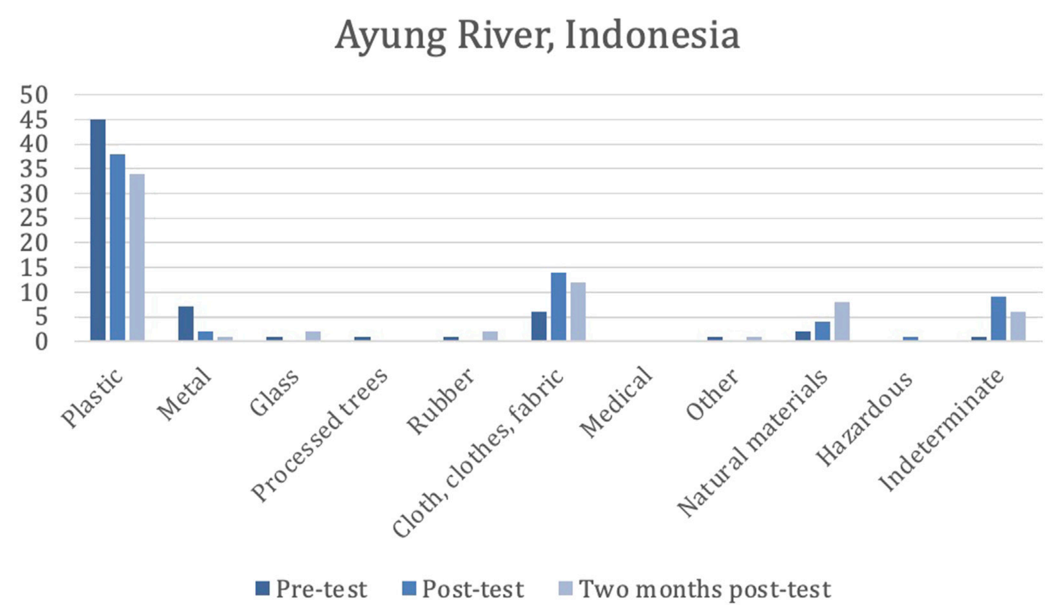

FIGURE 4
Participant perceptions of the most frequently found items in the Ayung River by type.

Supplementary Materials Table 2 compiles and categorizes responses for the Ayung River in Indonesia. Figure 4 shows a comparison of participant responses by type of material for each survey period. Please note the changing sample size between the pre-test (n = 65), the post-test (n = 68), and the 2 months post-test (n = 66).

Survey data: perception of content of debris in America

American respondents were asked: *Please list the top five most frequently found items in the Connecticut River.* Supplementary Materials Table 3 compiles and categorizes responses for the Connecticut River in the United States. Figure 5 shows a

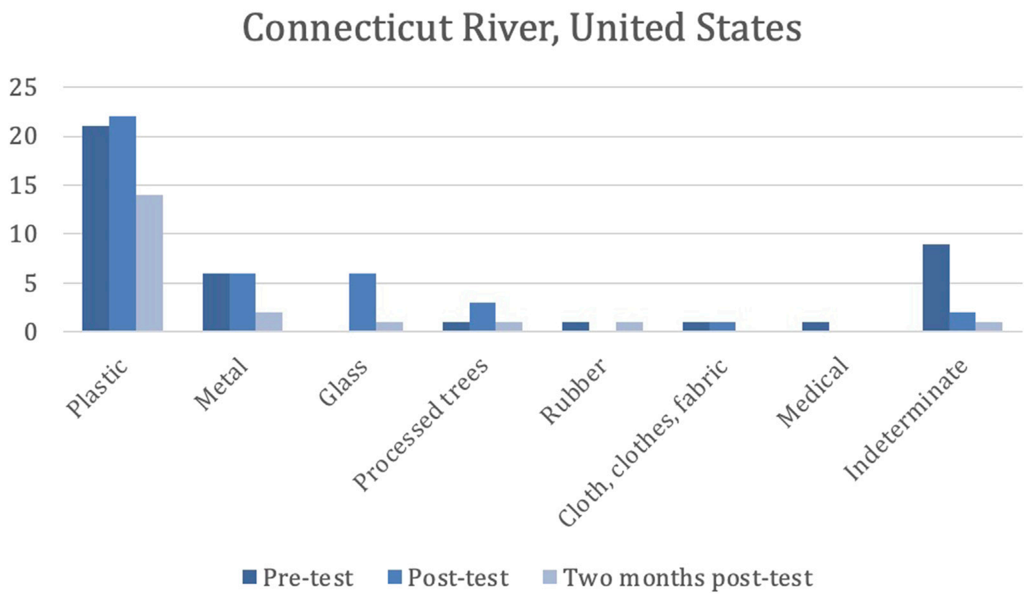

FIGURE 5
Participant perceptions of the most frequently found items in the Connecticut River by type.

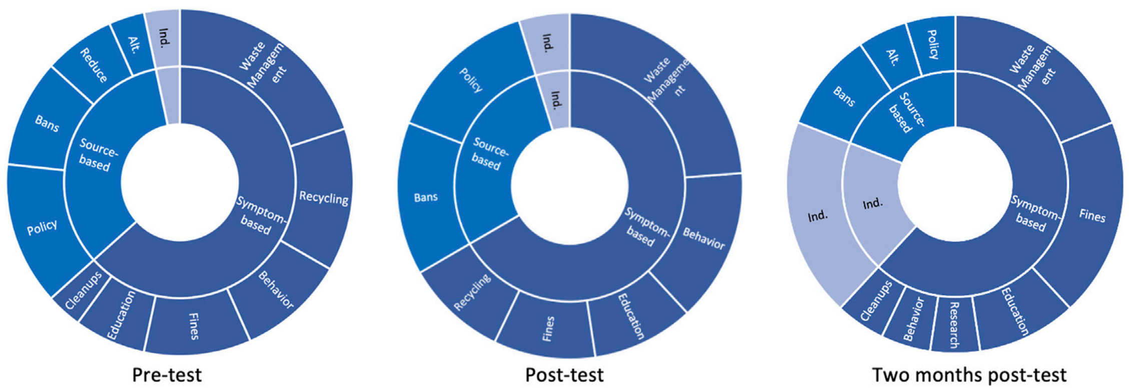

FIGURE 6
Comparison of stakeholder perceptions of policies in Uganda, pre-test (n = 30), post-test (n = 21), and 2 months post-test (n = 20). Alt. = Alternatives to plastic. Ind. = Indeterminate or unknown.

comparison of participant responses by type of material for each survey period. Please note the changing sample size between the pre-test (n = 40), the post-test (n = 40), and the 2 months post-test (n = 20).

Survey data: stakeholder perception of policies in Uganda

All workshop participants were asked: *Please list the top three policies that are helpful for combating the river litter problem in your community.* We categorize the responses based on whether the listed policy focuses on the symptoms or the sources of plastic pollution. For example, policies dealing with the symptoms might include recycling, fines on littering, or beach cleanups. Policies dealing with the source include those that ban plastic, encourage reducing, re-using, or alternatives to single use plastics. This is to better understand if policy conceptualization highlights the underlying core problem (unsustainable production of single use plastics).

Results from Uganda can be seen in Supplementary Materials Table 4, the data are summarized in Figure 6 below. In the pre-test (n = 30) 63% of responses focus on

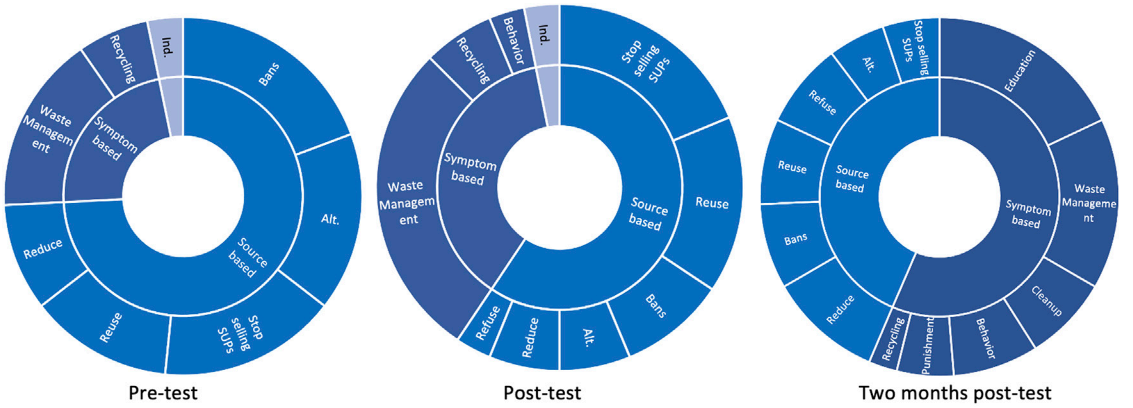

FIGURE 7
Stakeholder perceptions of policies in Indonesia, pre-test (n = 31), post-test (n = 32), and 2 months post-test (n = 39). Alt. = Alternatives to plastic. Ind. = Indeterminate or unknown.

symptoms-based approaches, including recycling, fines, cleanups, education, waste management, and behavior. Thirty-three percent focus on the source, including improving policy, reducing plastics, banning certain plastic, and promoting alternatives to plastic. Three percent of responses are indeterminate or unknown (e.g., "To circle the plastic"). For the immediate post-test (n = 21) in Uganda, 68% include policies associated with the symptoms of plastic pollution, including recycling, fines, education, waste management and behavior. Twenty-eight percent address the source of pollution, including improving policy and banning certain plastics. Five percent of responses are indeterminate or unknown (e.g., "Burning off plastic factories"). In the final Ugandan survey (n = 20) 65% of responses focus on symptoms-based approaches, including fines, cleanups, education, waste management, behavior and research. Twenty percent address the sources of pollution, including improving policies, banning certain plastics, and alternatives to plastic. Fifteen percent are indeterminate or unknown (e.g., "Throwing bottles is dangerous"). For the Ugandan participants, responses more frequently focus on the 'end of the pipe' rather than the source of debris and this does not change drastically over the course of the survey period.

Survey data: stakeholder perception of policies in Indonesia

Results from Indonesia can be seen in Supplementary Materials Table 5 and are summarized in Figure 7 below. In the pre-test (n = 31) 22% of responses focus on symptoms-based approaches, including recycling and waste management. Seventy-four percent focus on the source, including reusing materials, businesses no longer using single-use plastics, reducing plastics, banning certain plastic, and promoting alternatives to plastic. Three percent of responses are indeterminate or unknown (e.g., "reward and punishment for people"). For the immediate post-test (n = 32) in Indonesia, 37% include policies associated with the symptoms of plastic pollution, including recycling, waste management, and behavior. Fifty-nine percent of responses focus on the sources, including refusing, reusing, businesses no longer using single-use plastics, reducing, bans, and alternatives to plastic. Three percent of responses are indeterminate or unknown (e.g., "reward and punishment for people"). In the final Indonesian survey (n = 39), 57% percent of responses focus on the symptoms including recycling, waste management, behavior, education, cleanups, and punishment for people who litter. Forty-four percent address the sources of debris including refusing, reusing, businesses no longer using single-use plastics, reducing, bans, and alternatives to plastic. Initially, Indonesian responses focus more on the sources of debris, but then shift to being more balanced between policies focusing on the sources and symptoms of pollution.

Survey data: stakeholder perception of policies in America

The American results can be seen in Supplementary Materials Table 6 and are summarized in Figure 8 below. In the pre-test (n = 14), 50% of responses focus on the symptoms, including recycling, fines, cleanups, and education. Thirty-five percent of responses

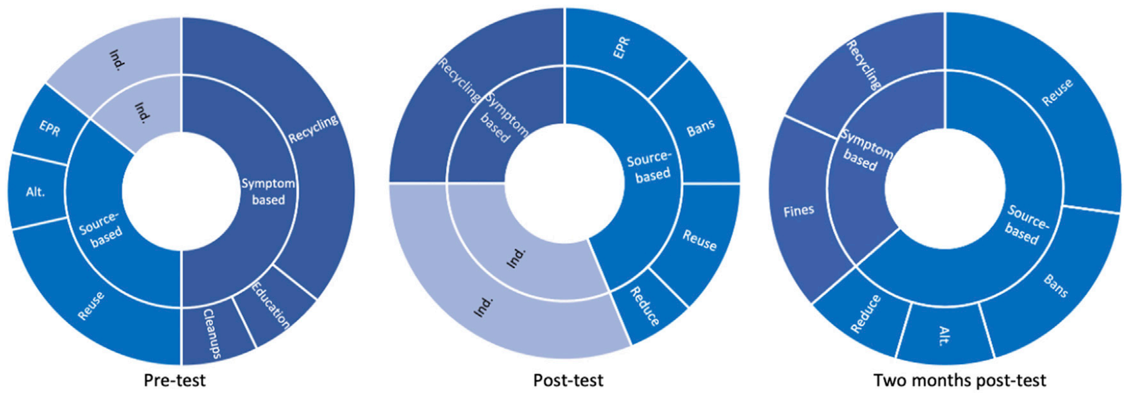

FIGURE 8
Stakeholder perceptions of policies in the United States, pre-test (n = 14), post-test (n = 16), 2 months post-test (n = 14). Alt. = Alternatives to plastic. Ind. = Indeterminate or unknown.

focus on the sources, including re-using, extended producer responsibility, and alternatives. Fourteen percent of responses are indeterminate or unknown (e.g., "commercial fishing," "none that I am aware of"). In the immediate post-test (n = 16), 25% of responses are symptom-based, emphasizing recycling. Forty-five percent emphasize the sources of debris, including re-using, bans on certain plastics, extended producer responsibility, and reducing. Thirty-one percent of responses are indeterminate or unknown (e.g., "plastic bag," "straw," "to go containers," "unaware of any"). For the final Connecticut survey (n = 11), 36% of responses are symptom based, including recycling and fines. Sixty-three percent are source-based and focus on re-using, bans, reducing, and promoting alternatives to plastic. Connecticut responses emphasize the symptoms of pollution initially, but shift to the sources of pollution between the pre-test and the survey occurring 2 months after the workshop.

Discussion

Debris collection

To better understand the context of the density results we use the classification of the Clean Coast Index (Alkalay et al., 2007). According to this classification system, the Ayung riverside rates as "extremely dirty" and the Aturukuku and the Connecticut riversides rate as "moderate."

Survey data: Environmental attitudes and behaviors

Due to both the low enrollment and the discrepancy in number of participants for each workshop, we were unable to run comparative statistical analysis on these data. That said, what we can see is that in this limited sample, the mean responses are comparable and there are not obvious differences between the respondents in Uganda, Indonesia, and the United States. In each country, the respondents are generally responding in the sometimes/frequently range for both pro-environmental attitudes and behaviors. In other words, workshop participants in each country display quite positive environmental attitudes and pro-environmental behaviors.

Survey data: Perception of content of debris in Uganda

How do the responses change over time to the question *Please list the top five most frequently found items in the Aturukuku River*? There is some growth of proportion for plastic, with responses for plastic rising from the pre-test (34%), through the immediate post-test (47%), and having the highest percentage for the 2-month post-test (51%). That said, plastic always ranks as the most frequently selected material, with cloth, clothes, fabric maintaining a steady proportion through the pre-test (20%), through the immediate post-test (16%), and the 2-month post-test (16%). Metal and animal waste are responses for the pre-test sample, but not beyond that. Glass disappears in the immediate post-test but resurfaces in the 2-month post-test. Plant waste is mentioned in the pre-test and immediate post-test, but not the 2-month post-test, whereas medical waste and processed trees are named in all three survey samples. As such, while some responses shift in minor ways throughout the sampling series, there are not drastic changes in respondent understanding of local waterway pollutants.

How do the responses compare to the collected sample? The respondents accurately presume the types of material they will find in the river, acknowledging that it will be primarily composed of plastic with cloth, clothes, fabric being the next-highest category. While these numbers shift between the pre, immediate post and 2-month post surveys, they are the two highest-ranking determined categories. This indicates the Ugandan participants have a strong grasp of the issues that plague their local waterway though they underestimate the proportion of plastic. In the pre-test, respondents presumed metal waste and animal waste would play a larger role, but this was not borne out through subsequent surveys, likely because the riverside collection did not include either of these substances. In other words, as their presumptions were ground-truthed, the Ugandan participants shifted their knowledge and understanding of the problem accordingly.

Survey data: Perception of content of debris in Indonesia

How do the responses change over time to the question *Please list the top five most frequently found items in the Ayung River*? There is a loss of proportion for plastic over the sampling series, with responses for plastic decreasing from the pre-test (69%), through the immediate post-test (56%), and through the 2-month post-test (52%). And yet, plastic always ranks as the most frequently selected material and proportionally makes up more than half of the responses. In the pre-test, metal ranks second-highest (11%) but this shifts to cloth, clothes, fabric for the immediate post-test (21%), and the 2-month post-test (18%). Metal is consistently named, but the proportion diminishes over time while conversely, Natural materials (including items like rope, organic waste, jute bags) is listed consistently and increases as a proportion of the sample over time. Glass and rubber are each a relatively small proportion found in the pre-test and the 2-month post-test, but not the immediate post-test. Hazardous materials appear only in the immediate post-test, and the other category includes varied materials in the pre-test (tetrapak) and the 2-month post-test (religious waste) but never comprises more than 2% of the sample. As with the Ugandan responses, the sample shifts in small ways through the sampling series, but there is a consistent understanding of local waterway pollutants.

How do the responses compare to the collected sample? The respondents accurately presume the types of material they will find in the river, acknowledging that it will be primarily composed of plastic. This indicates the Indonesian participants have a strong grasp of the issues that plague their local waterway, though they underestimate the predominance of plastic. While cloth, clothes, fabric rank third behind metal in the pre-test, these materials are the second most frequently found material in the collection. Responses shift accordingly, with cloth, clothes, fabric's proportion of the sample increasing from the pre-test (9%) to the post-test (21%) and the 2-month post-test (18%) while metal drops precipitously in the ranking from pre-test (11%) to the post-test (3%) and the 2-month post-test (2%). As such, as the Indonesian respondents learned more by engaging in a collection, their reported knowledge and understanding of the problem shifted accordingly.

Survey data: perception of content of debris in America

How do the responses change over time to the question *Please list the top five most frequently found items in the Connecticut River*? There is an increase in proportion for plastic over the sampling series, with responses for plastic increasing from the pre-test (53%), through the immediate post-test (55%), and the 2-month post-test (70%). Plastic always ranks as the most frequently selected material. In the pre-test, the indeterminate category ranks second-highest (23%) but that category shifts to 5% in both subsequent surveys. Metal is consistently named, and the proportion diminishes slightly over the series from pre-test (15%) to post-test (15%) to 2-month post-test (10%). Glass is not named in the pre-test, but is in the post-test (15%) tapering down for the 2-month post-test (5%). Rubber in the form of tires is noted in the pre-test (3%), not in the post-test, and again in the 2-month post-test (5%). Cloth, clothes, fabric rank low in the pre-test (3%), maintain that level for the post-test (3%) and are not mentioned in the 2-month post-test. Medical waste is mentioned in the pre-test (3%) but not in subsequent tests. As with the Ugandan and Indonesian responses, the sample shifts in small ways through the sampling series, but there is a consistent understanding of local waterway pollutants.

How do the responses compare to the collected sample? The respondents accurately presume the types of material they will find in the river, acknowledging that it will be primarily composed of plastic, though proportionately it is first slightly too low and then too high in comparison to the actual results. This indicates the American participants have a strong grasp of the issues that plague their local waterway, though they underestimate then overestimate the predominance of plastic. While glass is not listed in the pre-test, it is the second most frequently found material in the collection. Responses shift accordingly, with glass' proportion of the sample increasing for the post-test (15%) then decreasing for the 2-month post-test (5%). Metal is accurately presumed to be a part of the waste found in the sample, though its proportion decreases by the 2-month post-test (to 5%). As with the Ugandan and Indonesian respondents, the post-test responses are not a perfect match but do reflect the ground-truthing from the field experience. As such, the knowledge and understanding of the problem shifts accordingly.

How do the responses compare across countries?

The Ugandan responses are unique in the ranking of organic waste (animal remains, plant waste, logs, and invasive species) along with other pollutants (plastic, metal, glass, cloth, and rubber). The responses are consistent over time, never shifting widely. Participant responses indicate the Ugandans have a strong grasp of local river pollution and make subtle changes based on their real-world experience. Indonesian responses are also broadly consistent over time and a good match for the results found on-the-ground. When looking at the perceptions of the problem across categories, it is clear that respondents expect plastic will be a frequent find, which is borne out in real life. Both Ugandan and Indonesian responses note the importance of plastic and cloth, which are in fact the most frequently found items in their cleanup events. In comparison, Indonesian respondents do not expect as much organic waste (animal waste and plant waste), but do expect products created from natural materials such as jute bags and rope. The American respondents rightly presume they will find more non-organic materials (plastic, glass, and metal) and do not presume to find or find as much cloth as in the Ugandan and Indonesian samples.

In addition to the shifts within countries across the three surveys, it is important to note that both Ugandan and Indonesian participants describe a need for waste management, which is not mentioned by the US participants. This indicates the importance of waste management infrastructure for alleviating the symptoms of plastic pollution.

Survey data: Comparison of stakeholder perception of policies across countries

In general, the participants are well-informed about policy solutions, naming a wide range of policy options and strategies, particularly considering they are not prompted or given choices from which to select. The participants are self-selected into the workshop cohorts, which may mean they are more likely to be informed about the issues and aware of policies.

The Indonesian group begins with a strong focus on the sources of debris and provide more types of solutions when considering sources of pollution. When considering symptom-based approaches to alleviating debris, participants in all three countries note recycling, cleanups, education, and some type of negative consequences for littering, though respondents in Indonesia characterize this as punishment while those in Uganda and the US classify this as fines. As previously described, both Uganda and Indonesia make a note of waste management policies; participants from both countries also mention individual littering behavior, which the American respondents do not. In considering source-based approaches, respondents in all three countries mention policies that encourage reducing plastics, banning plastics, and promoting alternatives to plastics. Ugandan participants also mention forming new policies or bylaws. Both the American and Indonesian participants talk about programs to re-use materials. In addition, American respondents mention Extended Producer Responsibility, while Indonesian participants also mention programs that promote refusing plastics and business no longer carrying single-use plastics. Over time the Uganda sample remains relatively consistent, with the range of proposed policy responses focused on the sources of plastic pollution ranging from 33 to 20%. In contrast, the Indonesian population shifts over time, at first having a higher proportion devoted to source-based solutions (74%) though that falls over time to 44%. The group from Connecticut, United States, also shifts, but in the opposite direction, with the percentage of source-based policy solutions changing from 35% to 63% over time.

In conclusion, pollution, particularly plastic pollution, is an ongoing threat to freshwater as well as marine systems. This pervasive material has many uses, is convenient, and perhaps most importantly—cheap to produce in large quantities. That said, the long-term consequences as plastics break down in the environment harm wildlife, ecosystems, the economy, and human communities. The River Engage workshops sought to make connections between science and policy for local communities—collecting data on local systems to report the results to local lawmakers, though the project was hampered by the COVID-19 global pandemic. This research sheds light on the nuances among the way local workshop participants understand and grapple with the issue. These results indicate that one size does not 'fit all' and that the approaches to the problem should encapsulate local issues, nuances, and context.

The results, particularly from participant responses in Uganda and Indonesia, indicate that basic waste management must be a part of the solution. This must be shared with a major caveat, however, as in the United States and Europe, waste management systems are in place—yet plastic pollution remains a problem. In many ways, the global north has not solved the problem of plastic pollution—but are better at hiding it. Waste management systems should be in place in rural areas—for public health reasons as well as environmental and ecological health. It cannot be emphasized enough that waste management systems allow for some *tidying up of* but not *the true elimination of* the majority of plastic waste. When waste management systems are in place for plastic pollution, they result in incineration or landfilling—neither of which is an environmentally or ecologically neutral option. So, while these results indicate that waste management will improve the health of rivers in Uganda and Indonesia, this should be considered a stop-gap measure, never intended to solve the problem of plastic pollution. How we define problems has bearing on how we define solutions. The real solution for solving the problem of plastic pollution can only come from greatly reducing the number of single use plastics being manufactured, sold, used, and disposed of globally.

Data availability statement

The datasets presented in this study can be found in online repositories. The names of the repository/repositories and accession number(s) can be found below: https://bit.ly/3Db4i04 Research Gate and https://bit.ly/3f7hdYU Research Gate.

Ethics statement

The studies involving human participants were reviewed and approved by University of Hartford Institutional Review Board. The University of Hartford has an Assurance of Compliance on file with the Office of Human Research Protections (Federalwide Assurance—FWA #00003578). The patients/participants provided their written informed consent to participate in this study.

Author contributions

KO: conceptualization, methodology, formal analysis, investigation, writing—original draft, visualization, supervision, project administration, funding acquisition PK: conceptualization, methodology, formal analysis, investigation, writing—review and editing, supervision, project administration, funding acquisition HO: conceptualization, methodology, investigation, writing—review and editing, supervision, project administration, funding acquisition.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2022.1081208/full#supplementary-material

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Perhaps, perhaps, perhaps: Navigating uncertainty in environmental flow management

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Uncertainty can be an impediment to decision making and result in decision paralysis. In environmental flow management, system complexity and natural variability increase uncertainty. Climate change provides further uncertainty and can hinder decision making altogether. Environmental flow managers express reluctance to include climate change adaptation in planning due to large knowledge gaps in hydro-ecological relationships. We applied a hybrid method of hypothetical scenarios and closed ended questions within a survey to investigate ecological trade off decision making behaviours and cognitive processes of environmental flow managers. The scenarios provided were both similar to participants' past experiences, and others were entirely unprecedented and hence unfamiliar. We found managers were more confident making decisions in situations they are familiar with, and most managers show low levels of confidence in making trade off decisions under uncertain circumstances. When given a choice, the most common response to uncertainty was to gather additional information, however information is often unavailable or inaccessible–either it does not exist, or uncertainties are so great that decisions are deferred. Given future rainfall is likely to be different from the past, environmental flow managers must work to adopt robust decision making frameworks that will increase confidence in decision making by acknowledging uncertainties. This can be done through tools developed to address decision making under deep uncertainty. Adapting these tools and methods to environmental flow management will ensure managers can begin to consider likely, necessary future trade-offs in a more informed, transparent and robust manner and increase confidence in decision making under uncertainty.

KEYWORDS

environmental flow, uncertainty, decision making, climate change, adaptation

1 Introduction

"If you can't make your mind up, we'll never get started". Doris Day.

Without acknowledging uncertainties of the future, managers of environmental flows are effectively aiming to maintain a museum of the past. Environmental flows are an important tool in river health management, primarily used in regulated river systems (Arthington et al., 2018). Determination of environmental flows are typically guided by scientifically based flow assessments that develop recommendations for water delivery linked to specific ecological objectives (Tharme, 2003; Poff et al., 2017). Flow recommendations may include the magnitude, frequency or timing of water releases required to achieve a specific objective. Environmental water (used to deliver flow recommendations) is defined as all legally available water that can be used in a river system to provide environmental benefit such as protection of specific species, habitat maintenance or ecosystem function (Horne et al., 2017).

There is clear evidence that anthropogenic climate change is occurring, yet agreement on how to adapt environmental flow management is far from absolute. Environmental flow assessments (and recommendations) typically aim to restore ecosystems to an historic condition or protect them from further change and use historic flow regimes to make recommendations (Capon et al., 2018; Horne et al., 2022). However the past is not a good representation of the future and climate adaptation is required when determining flow objectives and recommendations for future environmental water (Judd et al., 2022). Under climate change, temperature and rainfall patterns are predicted to change along with rainfall run off relationships and streamflows (Saft et al., 2016). As these hydro-climatic changes occur, some current ecological relationships are unlikely to remain and consequently objectives are also unlikely to be achievable (Judd et al., 2022). Future water use and flow recommendations need to incorporate climate change adaptation (John et al., 2020). Yet, there are many known barriers to climate change adaptation, including insufficient staff and funding, lack of political leadership, backward looking legislative requirements, and uncertainty (Abunnasr et al., 2015; Kiem et al., 2016; Oberlack and Eisenack, 2018; Judd et al., 2023). Uncertainties include future greenhouse gas emissions; the direction, magnitude and intensity of change in response to emissions with large variances in predictions at particular geographic locations (Hallegatte et al., 2012; Shepherd et al., 2018). Managers of water for environmental flows (environmental water managers) cite uncertainty of hydrological and ecological systems' response to climate change as a major barrier to adaptation (Judd et al., 2023). Adaptation to climate change also suffers from considerable uncertainties regarding how social, economic and political systems will react (Kundzewicz et al., 2018).

Part of the role of environmental water managers is to prioritise flow recommendations based on flow assessments and antecedent conditions (Doolan et al., 2017). This 'active' management of water allows managers to determine how much and when to use water to achieve the desired ecological objectives. Management can include releasing water from dams to restore part of a flow regime or adding water releases to a natural flow to hit flow recommendations. Decisions to manage water are made in real time based on current ecological conditions, water availability and other constraints such as channel capacity or social/recreational requirements. At times flow recommendations for one component of the ecosystem (e.g., vegetation) may conflict with recommendations for another component (e.g. fish) and water managers will be required to make a trade off decision between two ecosystem components within the one system. In times of drought management tends to focus on maintaining drought refuges, avoiding species loss, and providing opportunities for ecosystem recovery once drought breaks (Doolan et al., 2017). In times of water scarcity, decision making is substantially more difficult as uncertainties in water availability and ecosystem response become more unknown and constrained.

Uncertainty in environmental flow management can be separated into nature and level. The nature of uncertainty relevant to the types of decisions environmental water managers must make are epistemic uncertainty and aleatoric uncertainty. Epistemic uncertainty arises due to a lack of knowledge or information about a phenomenon or process. This type of uncertainty can be reduced by gaining new knowledge or doing more research. Reducing uncertainty with new information can help increase confidence in decisions (Singh et al., 2020). Aleatoric uncertainty is defined as uncertainty that cannot be reduced by increasing knowledge due to the inherent variability and unpredictability of the phenomenon (Dewulf and Biesbroek, 2018; Singh et al., 2020). Both these sources of uncertainty will increase with climate change.

Levels of uncertainty have been defined by Kwakkel et al. (2010) as the "assignment of likelihood to things or events" with the likelihood able to be expressed either qualitatively or quantitatively. Kwakkel et al. (2010) identified four levels of uncertainty ranging from shallow uncertainty (level one) to recognised ignorance (level four). Following their definition environmental water management under climate change would fall into level three uncertainty. Level three is also referred to as 'deep uncertainty'. This is where alternative options can be specified but probability functions for the likelihood of alternatives cannot be determined, and an order ranking of alternatives is unknown or cannot be agreed on by experts or decision makers. Lempert et al. (2003) defines deep uncertainty as unknown or unagreeable boundaries surrounding the external context of a system, how the system works and where its boundaries lie, and the outcomes of interest from the system or their relative importance.

Both the nature and level of uncertainty impose impediments to making well informed decisions about future management of environmental water. The uncertainty surrounding climate change projections coupled with the natural variability and complexity of ecosystems means there will always be uncertainty in making environmental management decisions. Aleatoric uncertainty will always exist, and epistemic and deep uncertainty provide barriers to timely decision making if the uncertainties cannot be resolved. Herein lies the challenge for environmental water managers: there are uncertainties that will always exist, and others where a decision will need to be made prior to information becoming available. Understanding how water managers respond to such uncertainty will enable development of appropriate tools for planning and policy decisions to be made.

People respond to uncertainty in different ways as detailed by Lipshitz and Strauss (1997) and Pasquini et al. (2019). The three principal response categories to uncertainty are:

- Suppress - through complete denial, ignoring or distorting undesirable information, relying on intuition and taking a gamble. Additionally, fear of making the wrong choice can also result in avoidance of making a decision (Retief et al., 2013). Using traditional rational decision making approaches under climate change could be considered a form of suppression.
- Reduce - by collecting more information, deferring decision making, extrapolating from existing data, and shortening the time frames for decisions. However, deciding to collect more information should be an option employed with caution to

ensure it will add value, be available in time and change the decision outcome (Dietz, 2013).

- Acknowledge–often adopted when reducing uncertainty is not possible. This response involves preparing to confront potential risks under a chosen course of action, and may include incorporating reversible or no regret actions, or making an informed decision of the pros and cons. The use of multiple plausible futures or confidence intervals in data analysis also helps acknowledge uncertainty (Brugnach et al., 2008).

Lipshitz and Strauss (1997) undertook an empirical experiment and determined the following response strategies were used most frequently (in order); reduction, forestalling, assumption based reasoning, weighing pros and cons where there was difficulty choosing between alternatives, and suppression. They also note that different strategies were used in different situations e.g., assumption based reasoning was used where there was incomplete information and reduction was adopted where there was a lack of understanding. When people feel uncertain or a decision is difficult, the response can be to delay, avoid and/or be paralysed to make decisions (Weber et al., 2001; Höllermann and Evers, 2017). In a study by Doerner (1990) people's decision responses to maintaining or improving a complex and dynamic system were observed. Results showed common faults of people's decisions included failing to establish clear goals, treating a complex system as separate variables rather than an integrated system, and making decisions without checking the effects of these to other parts of the system. Doerner (1990) found even though participants had enough information they were not very adaptative in their thinking and devoted most of their time to problems they currently faced rather than looking to potential future problems or how their actions today may lead to future issues.

Decision making in water planning and policy has traditionally relied on rational and probabilistic methods to reduce uncertainty and optimise one preferred option (Haasnoot et al., 2013; Horne et al., 2016; Siders and Pierce, 2021). This approach follows a 'reduce uncertainty' principle where a set of possible actions are determined and compared through probability distribution functions, with the best performing option chosen to optimise a desired outcome (Citroen, 2011; Pasquini et al., 2019). This requires information such as averaged hydrological parameters, the likelihood of alternate states, how actions will combine to form outcomes and the benefit of one outcome over another (Polasky et al., 2011). During assessment of options this approach acknowledges uncertainties, but ultimately aims to maximise one particular outcome within the knowledge of some spread in performance (and uncertainty) amongst all options. The approach often employs data intensive mathematical models that follow generalised principles. They do not provide reasons for why or how a decision may deviate from the norm (Pasquini et al., 2019).

We know that river system dynamics–particularly under climate change–include significant aleatoric uncertainty, where these traditional approaches of 'reduction' can no longer be applied. Managers of environmental water and other natural resources express reluctance to include climate adaptation due to large knowledge gaps surrounding ecological and hydrological relationships (Stein et al., 2013, Judd et al., 2023). Although knowledge is improving, researchers suggest there is a clear need to increase understanding of flow ecology relationships (Thompson et al., 2019). Many models developed to predict biological responses to climate change ignore fundamental biological functions such as species interactions, demography (births, deaths, phenology etc) and evolutionary potential (Urban et al., 2016). Even when attempts are made to include this information in ecological models there is limited data due to funding constraints for long term monitoring programs. Further, the translation of global climate scenarios into meaningful and useful localised hydrological and ecological information for water supply remains a barrier (Kiem et al., 2016). Consequently, probability based decision methods that optimise for one solution will no longer be appropriate (Brugnach et al., 2008; Hallegatte, 2009; Polasky et al., 2011; Maier et al., 2016; Fletcher et al., 2017). Future planning needs to be either robust; include objectives or actions that can be achievable over a range of plausible futures, or dynamic; objectives and policies that are flexible and can change over time as new information becomes available. This causes further challenges for a technocratic industry where data and 'uncertainty reduction' for optimisation has always dominated thinking (Pahl-Wostl et al., 2013; McLoughlin et al., 2020).

This paper examines the readiness and response of environmental water managers to make ongoing decisions under uncertainty. A survey approach is adopted to challenge environmental water managers to make decisions in situations they are familiar with, and possible future scenarios. The data analysis links decision making in three hypotheses with the human responses of Pasquini et al. (2019) and Lipshitz and Strauss (1997). This study investigates the following three hypotheses:

H1: Environmental water staff have a high confidence in making ecological trade off decisions when situations are similar to their past lived experience (reduce)

H2: Environmental water staff are unwilling to make ecological trade off decisions when there is insufficient information (supress)

H3: Environmental water staff will be confident in making trade off decisions when provided with climate change information they consider to be important (acknowledge)

This study provides the first step in understanding how environmental water managers are making decisions, what type of information they consider important and how confident they feel making these decisions. By framing the managers response to uncertainty around 'acknowledge, reduce, suppress' we will be in a better position to support decision making under uncertainty and provide recommendations for future research, and development of methods or tools that enable increased confidence and ability of managers to make decisions in such situations. Bearing Doris Day's point in mind, we can help environmental water managers to make their mind up and get started despite not knowing with certainty.

2 Method

This research used a self administered online survey to measure how environmental water managers make decisions when faced with incomplete information (thereby introducing uncertainty). The questions were predominately a mix of behavioural and self assessment style questions including a combination of closed end

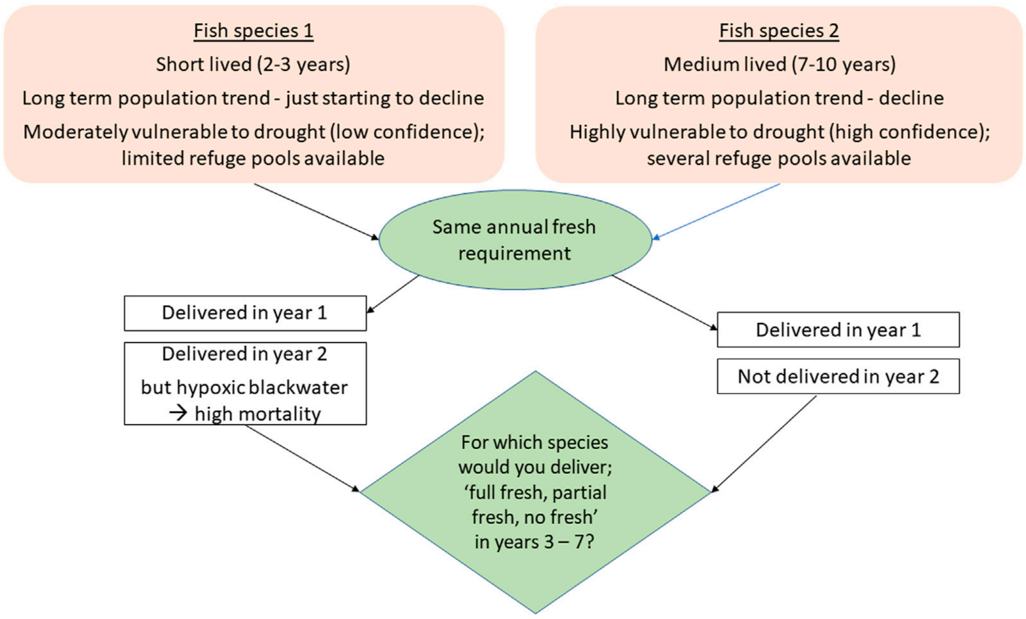

FIGURE 1
Hypothetical scenario (one) provided in the online survey for environmental water managers to measure trade off decision making response during a drought and in an uncertain situation.

questions and hypothetical scenarios (see Supplementary Material). Scenarios were presented as descriptive 'stories' which can be useful when decision making is required in uncertain conditions (Peterson et al., 2003). Scenarios can be based on past events or situations similar to existing conditions, they can be imagined and combined with climate change information, but importantly they should be plausible (Bryman, 2016; Shepherd et al., 2018).

The inclusion of imagined, hypothetical scenarios in this research allowed participants to improve their awareness of their decisions, think deeply about the scenarios provided, and demonstrate their current thinking and ideas. The survey scenarios were developed to determine how confident water managers were in situations they were familiar with, and likely future scenarios under a water scarce future to link back to hypotheses one to three.

The first scenario (Figure 1) asked participants to decide how they would use environmental water to support two different species located in different rivers of the same catchment during a drought with limited water. Historically water availability has been such that both species can be supported, but under this scenario there is restricted water availability and hence an ecological trade off decision is required between the two species. A drought scenario was chosen as there is a high likelihood that most participants have managed water during a drought, and it is a likely future scenario under climate change. Participants were asked to make a trade off decision between providing a full fresh, partial fresh or no fresh for one species *versus* the other. A fresh is defined as a short duration flow event greater than median flow and provides functions such as biological triggers for migration and/or spawning and physio chemical changes (DEPI, 2013). A full fresh was the recommended action for both species and benefits of a partial fresh were uncertain along with water trade and operational rules, broader population status including vulnerability and most recent antecedent conditions. This deliberate omission of certain information relates to hypothesis two; "staff are unwilling to make ecological trade off decisions when there is insufficient information". Questions about this scenario also linked to hypothesis one; "staff have high confidence in making ecological trade off decisions when situations are similar to their past lived experience" by testing whether participants have previously experienced a similar situation and their level of confidence in their water use decision.

In scenario two (Figure 2), participants were given information deemed important for decision making under climate change as identified by Judd et al., 2023. Information provided included species vulnerability assessments, temperature and water availability changes, drought refuge availability, extreme event frequency and habitat connectivity, species life span, social value and ecological function. The confidence levels of these projections were also provided. Considering the information provided participants were asked if they wanted to continue delivering water and pursuing the objective for the nominated species, despite climate data showing this species had a moderate to high level of vulnerability and survival looked unlikely. This scenario tests hypothesis three; "staff will be confident in making trade off decisions when provided with climate change information they consider to be important" and adds uncertainty in the moderate confidence levels provided for future conditions. This scenario enabled respondents to think about the types of decisions they will have to make more commonly in the future.

The survey was developed through an iterative process of review and pre-testing with a small number of environmental water managers. The online survey was developed in Question Pro (Inc) survey software and distributed by email to approximately 80–100 environmental water staff or researchers identified through purposive sampling (Gideon, 2012). The survey was distributed to staff working in government water management

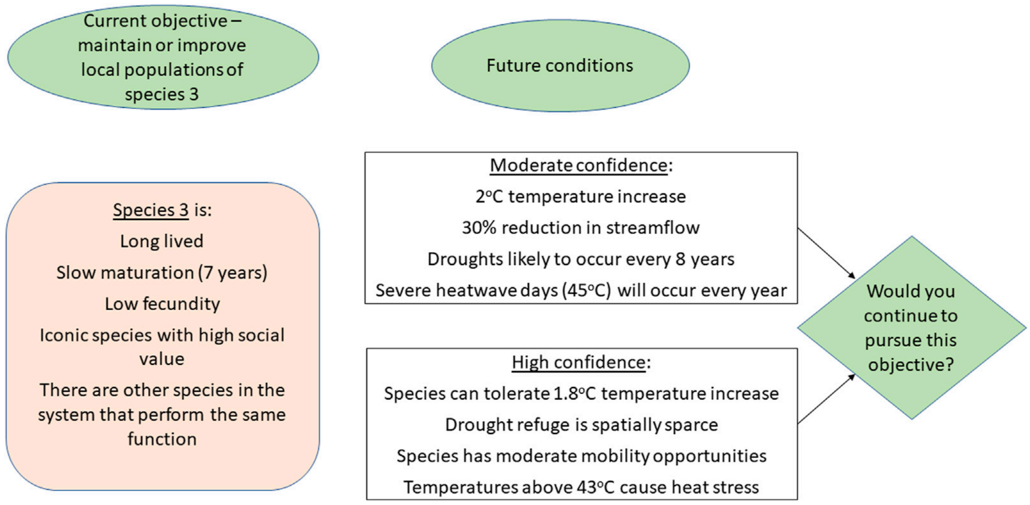

FIGURE 2
Hypothetical scenario (two) provided in the online survey for environmental water managers to measure their willingness to change objectives when provided with relevant climate change information.

agencies who are responsible for making decisions on the long term use of riverine environmental water throughout Australia, with a strong focus on the south east. Although researchers are not responsible for making decisions about water use and/or delivery, they are often approached by environmental water staff to help inform decision making. The survey was distributed in late January 2022 and open for approximately 6 weeks. This timing was chosen as it was thought staff would be returning to work after summer holidays and likely to have time available to complete the survey. Two reminder emails were sent to everyone on the distribution list at 4 weeks and just before the closing date. There were 25 completed responses and average completion time for the survey was 45 min.

The use of scenarios followed by specific survey questions meant data analysis could link to the study hypotheses. Questions in the survey that linked directly to the study hypotheses include "how confident are you with your decision" and "how closely does this reflect your own experience". Other questions used in the data analysis came from questions specifically asking the participants their perception of the amount/type of data provided for decision making, if there were specific pieces on information that assisted with their decision making, and if they found certain decisions difficult to make. Some of these questions were repeated with each scenario and additionally asked in a general manner. Questions asking the same question in different situations were collected and compared to provide results described here.

3 Results

3.1 Demographics and general response to uncertainty

There were 25 complete responses to the online survey encompassing a diversity of experience and training, and roles and responsibilities (Table 1). There were no partially completed survey responses.

Participants were asked about their general approach to decision making; how confident they are in making a decision with a lack of information and how they would go about making such a decision. Eight of the 25 participants had a very high or high level of confidence, while 14 had a medium level of confidence. Three had low confidence. Responses to how participants go about making a decision with uncertainty are shown in Figure 3. Linking the answers to Lipshitz and Strauss (1997) the overwhelmingly most common response was to reduce uncertainty by gaining additional information from experts or colleagues or drawing on a previous similar experience. An additional six of the 25 would delay their decision while reducing uncertainty by sourcing more information. Although fewer respondents aimed to acknowledge uncertainty by making reversible decisions a substantial number (14/25) still chose this option while some participants (7/25) chose to suppress uncertainty by relying on intuition.

3.2 Hypothesis 1—Environmental water managers have high confidence in making ecological trade off decisions when situations are similar to past lived experiences

Environmental water managers were found to be most confident in decision making when they feel the situation is similar to what they have previously experienced (Figure 4), however there was some spread in participant levels of confidence. This result is a common human response with much literature showing that personal experience, especially recent experiences, can be the most frequently drawn on source of information in decision making (Giehl et al., 2017; Ausden and Walsh, 2020; Page and Dilling, 2020; Kong et al., 2021). Two participants showed relatively very high or somewhat high levels of confidence without having been in a similar situation. Both these participants had more than 20 years' experience working in environmental water management.

Using drought as a previous experience aligns with the expectation that past droughts may provide a reasonable analogue for more permanent

TABLE 1 Participant demographics.

	n = 25	%		n = 25	%
Age			Trained as:		
22–32	2	8	Ecologist/biologist	14	56
33–44	12	48	Physical scientist	3	12
45–54	9	36	Engineer	3	12
55–65	2	8	Geographer	1	4
			Other (NRM, land/water manager)	4	16
Years of experience			Currently working as:		
0–4	6	24	Operational manager/officer	4	16
5–9	2	8	Strategic manager/officer	6	24
10–15	10	40	Both strategic and operational	10	40
16–20	4	16	Researcher	3	12
Longer than 20	3	12	Other (community engagement, project officer)	2	8

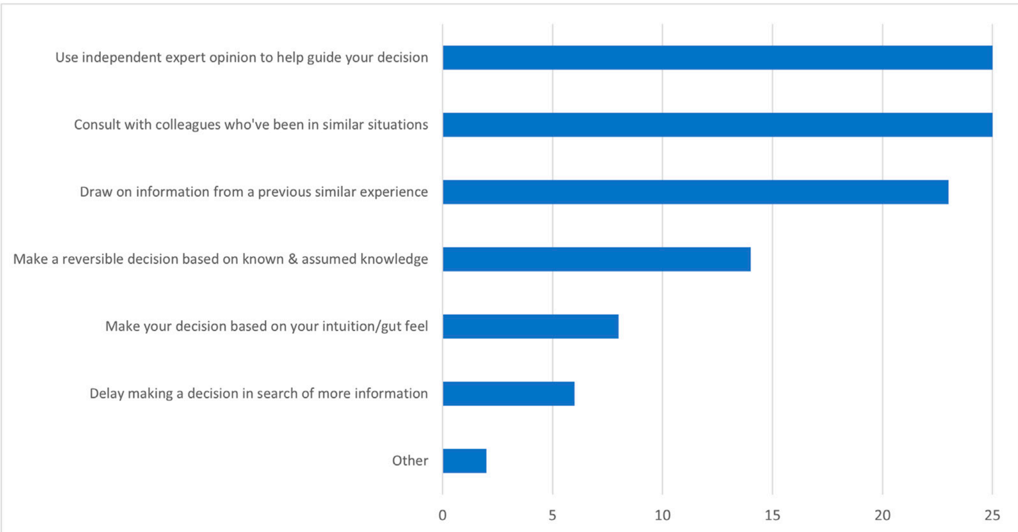

FIGURE 3
Online survey responses to a question asking how participants generally respond to decision making with uncertainty. Participants were Australian environmental flow managers (n = 25) responsible for making decisions on water use.

climate shifts in the region under investigation. These results show participants who had previously managed water during a drought were more confident in decision making than participants who had not managed during a drought (Figure 4). In fact four participants who had managed water during a drought indicated that scenario one was almost identical to an experience of their own management situations. This may also link to their level of confidence in similar situations as per the drought scenario one of the online survey, but the separation of managing water in a drought *versus* the total previous experience of these managers was not asked and hence cannot be concluded.

3.3 Hypothesis 2—Environmental water managers are unwilling to make ecological trade off decisions when there is insufficient information

Environmental water managers were found to be willing to make trade off decisions despite incomplete information (contrary to our hypothesis). Sixteen respondents (64%) chose to deliver a partial fresh with no certainty of the benefits this would provide to either species, while eight (32%) made the choice to only deliver water

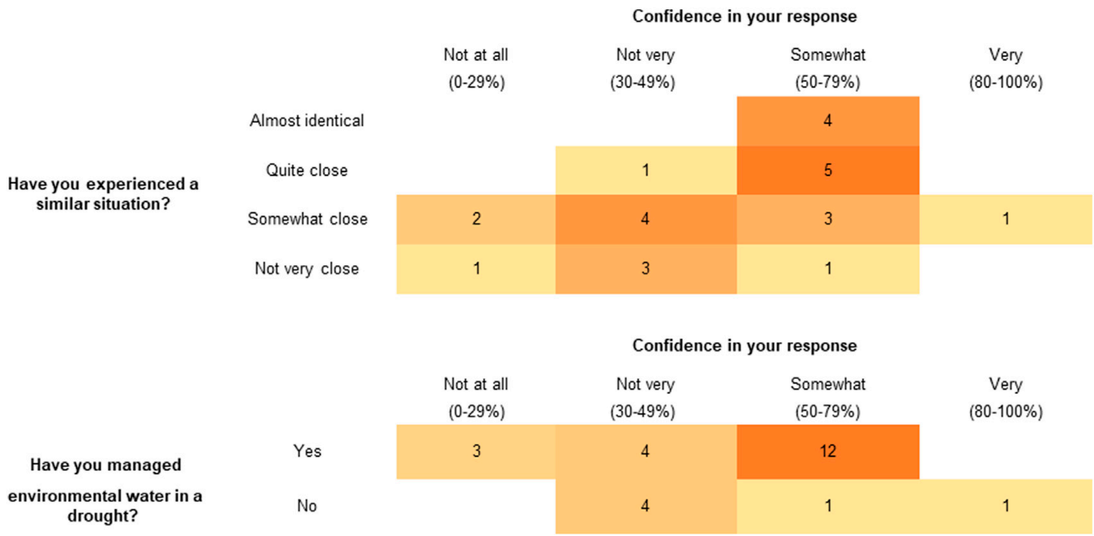

FIGURE 4
Participants' level of confidence in decision making in situations similar to their own experience and level of confidence for those who have managed water during drought conditions. Darker colours represent higher numbers of participant responses for this category.

when availability was such that a full fresh (as per recommendations) could be delivered.

Further, the same proportion of respondents decided to favour a higher value species despite the lower certainty of information. When asked how they would use water to support either one of two species or to try and balance water for both species, the shorter lived species with high social and ecological value and moderate, yet uncertain, vulnerability to drought was favoured over the longer lived species that was highly vulnerable and moderately valued. Seven respondents (~30%) chose to use water in a balanced way (i.e., alternating target species in alternate years) to aim for survival of both species showing a lack of willingness to 'give up' on a species and citing the lack of information as reason for their decision. The most critical pieces of information for all participants were species life span and water availability. When asked how long they would continue pursuing both these objectives given the information provided, the majority of participants said they would aim to reduce the uncertainty by gathering more information before 'giving up on one species' (Table 2).

These results show that approximately two thirds of participants are willing to make ecological trade off decisions when there is a lack of information and a level of uncertainty. However, 21 of the 25 participants agreed they found it difficult to make a decision for the following reasons: uncertainty of water availability now and in future years (including baseflows), uncertain benefits of a partial fresh for these specific species and the overall ecosystem, lack of detail on location and connection of other populations and refuges (e.g., how long will they retain water), and a lack of experience in making similar decisions. Results from the open ended question also show some participants were looking for information from contemporary monitoring data indicating the importance of up to date monitoring to some environmental water managers' decision making processes.

3.4 Hypothesis 3—Environmental water managers will be confident in making trade off decisions when provided climate change information deemed important

In general terms, the participants acknowledged and supported the need to prioritise sites, make trade offs and allow some sites to transition to a different structure, along with incorporating adaptation actions into environmental watering programs (Figure 5). There is also a large proportion of participants who are wanting to do this and not willing to wait for policy guidance, and as shown above believe the procurement of information will assist in this decision making.

However, the results also show that despite being willing to make adaptation decisions, participants are still challenged with how to go about making these decisions without complete information. Despite being provided the information previously deemed important for decision making (see Judd et al., 2023), a large portion of participants (10 or 40%) thought the information provided in the scenarios was not detailed enough. Another eleven agreed the information was useful with only three participants suggesting there was an overwhelming amount of information. Of the information provided, the most important or trigger pieces of information were species life span and sensitivity to temperature, species social value, the potential benefit this watering event may have on other components of the ecosystem (i.e., flow on effects to other species), frequency of drought and availability of refuges. Contrary to our hypothesis, provision of this information was not enough for managers to confidently change the management objective provided in this scenario. Nineteen of the 25 participants decided to continue to pursue the existing objective.

Clifford et al. (2020) found managers were opposed to ecosystem transformation due to a lack of confidence in climate projections,

TABLE 2 Participant responses to uncertainty for future water use.

Action	Count	Response to uncertainty
I would not stop delivering freshes for either species and lobby the government for increased environmental water entitlements	4	acknowledge
I would not stop delivering freshes for either species, however I would revise the relevant flows study	2	reduce
I would continue for 1–2 years while I seek expert opinion on issues such as climate change vulnerability and ecological function and then make a decision	17	reduce
I would continue for 1–2 years while I conduct a community survey to determine the community value of both species and then make a decision	1	acknowledge
Other (e.g., investigate complementary measures, prioritise and deliver for fish only)	2	suppress and acknowledge

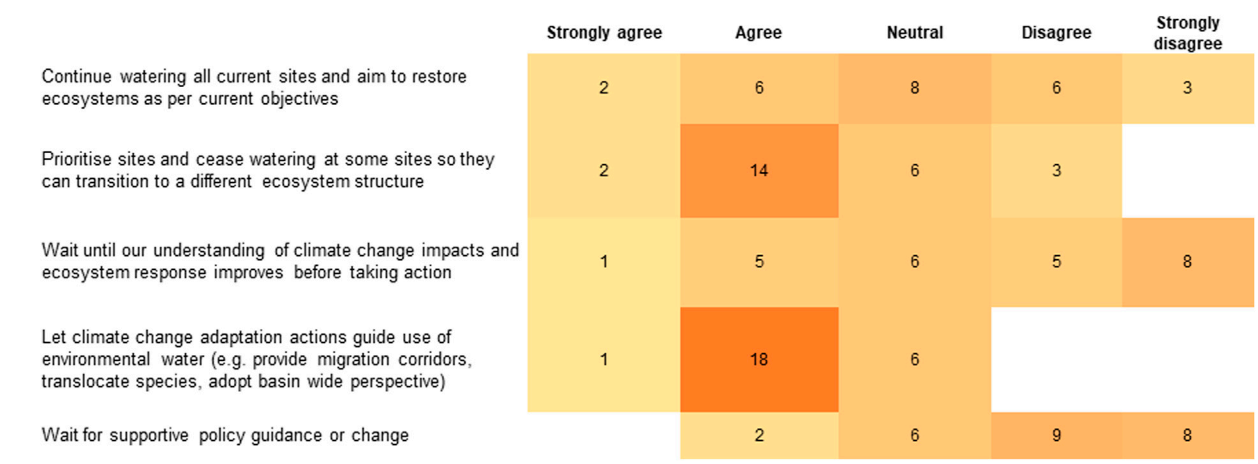

FIGURE 5
Participant agreement with suggested climate change management actions and adaptations. Participants used a Likert scale to select their level of agreement with each action. Darker colours represent higher numbers of responses for this action. All 25 participants provided their level of agreement to each management action.

while Azhoni et al. (2018) concluded a lack of confidence in climate information was as a barrier to adaptation. This also occurred in our study were some participants suggested they did not trust the provided climate change and vulnerability assessment information strongly enough to allow this to influence their decisions. As two participants said:

> *"Some thresholds and tolerance results from lab work are exceeded in natural systems and whilst they do cause impacts - they are not necessarily curtains"*

> *"Whilst the evidence shows the species cannot survive above 1.8°C annual average increase, I would view this with scepticism as similar research fails to take into account the potential for rare, but important genetic traits to influence selection."*

Nineteen (19) of the 25 respondents said they would continue pursuing a watering objective even when climate data showed significant vulnerability and unlikely survival. They sought more information in deciding whether to cease pursuing this objective as captured by this response:

> "Due to the uncertainty of the future, doing whatever is possible today to ensure survival of (the) species 'buys' time and provides motivation to look for ways to protect these species in the long term using alternative options"

4 Discussion

The results of the survey support one of the three hypotheses tested. Results show environmental water staff are not very confident in decision making when there is a lack of information with only one participant having greater than 80% confidence. These results show past experiences increase managers confidence in making ecological trade off decisions (hypothesis one). In decision making, past experiences are usually combined with a person's scientific knowledge but can also be subject to strong biases (Cook et al., 2010; Höllermann and Evers, 2017; Ausden and Walsh, 2020; Clifford et al., 2020). Further, the non stationarity of climate and weather experienced under climate change makes the past a poor reference point for the future, and one where past actions may not have the same outcome. As Helmrich and Chester (2020) emphasise, although a system has been shown to be resilient in the past, does not ensure its resilience in the future.

Using the past to inform future decisions can provide benefit if past decisions were the 'best' possible decisions. However, without clear monitoring each time a decision and action are taken, this will not be known. Adaptive management is one method to acknowledge this uncertainty and improve knowledge through testing hypotheses, implementing and monitoring actions, and adjusting future decisions and actions based on the results. Adaptive management is well acknowledged and included in environmental water management (Tonkin et al., 2020; Watts et al., 2020; Horne et al., 2022), but there can be lagged ecological feedbacks, and thresholds or tipping points may pass before managers are even aware. Other reasons also prohibit widespread adoption of adaptive management such as institutional risk aversion of looking like a 'failure' and insufficient resourcing, amongst others (Allan et al., 2008; Stults and Larsen, 2020).

Results from this research do not support hypothesis two which suggests managers would be unwilling to make a decision with insufficient information. In fact the results show environmental water managers are willing to adapt and make trade off decisions with a lack of information, but they find it difficult and would like an extensive list of additional information. The need to reduce uncertainty is a common response by people and organisations and can lead to the believe that improvement in information will be a solution to their decision problem. However, the improvement in information or science may not always be accessible, in a format reachable to practitioners or available in a timely manner. According to Ryder et al. (2010) providing knowledge from research for decision making is difficult due to misalignment of academic and manager's requirements, including different questions from practitioners and researchers, research timeframes are too slow for practitioners and different personal or organisational goals (Ryder et al., 2010). There can also be difficulty for practitioners to access research information along with different views on legitimacy of information, while recommendations provided by researchers can be perceived as irrelevant or impractical in practice (Cook et al., 2013; Dilling et al., 2015).

These results show even when provided with climate change information, water managers were reluctant to change objectives, rebutting hypothesis three. Under climate change, improvement in currently available science is simply not sufficient. Knowledge of future greenhouse gas emissions, and the direction, timing and severity of changes is unavailable. How this will change soil moisture capacity and rainfall runoff relationships is largely unknown. Human behavioural change and the complexity of ecosystem responses to these changes is also unknown (Stults and Larsen, 2020). Consequently, consideration of time and money invested in searching for more information needs to be weighed against the benefits extra information will provide. It is pointless to delay a decision if the new knowledge is not available and/or will not improve a decision outcome. This also makes it vitally important to ensure any additional research and/or monitoring is addressing well thought out endpoints to capture information important for long term decision making and the uncertainty of hydro-climatic change and consequent ecological responses.

To assist decision making the use of adaptation decision frameworks such as the expanded "resistance-resilience-transformation" or "resist-accept-direct" as per St-Laurent et al. (2021) and Thompson et al. (2021) respectively can be useful for environmental water managers. These frameworks aim to assist decision making under climate change by offering options for management actions; namely resisting change, accepting and adapting to change or transforming systems to a new state. These decision frameworks are useful to inform which adaptation path to follow but fail to acknowledge uncertainty. Several methods that do acknowledge uncertainty are already available and should be adopted by environmental water managers. Firstly, scenario planning provides managers with multiple possible future scenarios to consider. For example, the scenarios prepared for this research could be modified based on the results of this study and presented at a workshop of environmental water managers and other stakeholders. The scenarios can highlight parameters and inclusion of information deemed of high importance to managers in their decision making process (Kong et al., 2021). Each scenario can be as complex or simple as required and incorporate non flow and social or economic related constraints. Scenarios should be supported by real data and include the main concerns and uncertainties, significant driving factors and the plausible changes in those factors (Wodak and Neale, 2015; Shepherd et al., 2018; Gray et al., 2020). Numerous scenarios can be presented in a workshop yet there is no assumption on the likelihood of any particular scenario occurring. The workshops do not require specialist technical skills (other than a facilitator) thereby keeping costs low, with scenarios 'tested' prior to a workshop to ensure their feasibility and realism. Scenario planning workshops can include a large number of stakeholders and allow managers to think about events outside their own experience, and consider what type of policy/strategic decisions are required under a range of possible futures (Wodak and Neale, 2015; Shepherd et al., 2018). Fact sheets can be sent out prior to the workshop to ensure all participants have the same base level of information (Serrao-Neumann et al., 2019). Scenario planning workshops can challenge managers assumptions and improve knowledge of complex and dynamic issues.

Secondly, another option is to adopt robust and adaptive methods that have been developed specifically to deal with deep uncertainty. These are innovative methods for environmental water managers and can assist by providing information on long term policy and strategic direction to inform issues such as setting achievable ecological objectives. All such methods principally test system vulnerabilities across multiple scenarios to determine where the objective, or policy, fails. By testing objectives through a wide range of future conditions and 'stress testing' a system until the point of failure, the method delivers robust decisions rather than optimising for one ideal solution (i.e., robust being where performance is insensitive to which future may occur (Maier et al., 2016)). These methods aim to achieve a 'satisfactory' outcome under multiple scenarios rather than optimise one preferred option. Other methods aim to ensure performance is flexible enough depending on what future outcome may occur allowing for changes of approach if things 'fail' (Maier et al., 2016; Lawrence et al., 2018). Examples of these methods include; Robust Decision Making and Multi Objective Robust Decision

Making (Lempert et al., 2003; Lempert and Groves, 2010; Herman et al., 2014), Info-Gap analysis (Ben-Haim, 2006), Adaptation Pathways and Dynamic Adaptation Policy Pathways (Haasnoot et al., 2013). There are pros and cons in all these methods and reviews have been provided by Matrosov et al. (2013), Kwakkel et al. (2016), Bosomworth et al. (2017) and Bartholomew and Kwakkel (2020). One consideration when using, or adapting, these methods to environmental flow management is the requirement of potentially new data (e.g., vulnerability assessments), but the results of this study have shown managers are willing to make decisions in the absence of information/data. Therefore, perhaps the need for additional information is not as important as providing managers with tools and experiences to increase their confidence and ability to make decisions under uncertainty.

We will briefly review Robust Decision Making (RDM) as we consider this a suitable initial method to test vulnerabilities of existing environmental water ecological objectives and policies and can support subsequent implementation of other methods. The results from this study support the trialling of such methods as environmental water managers have shown they are able to make decisions under uncertainty and willing to test new methods of decision making. This method could be used to support long term decision making and objective setting as proposed in scenario two of this study. RDM allows analysts to propose an objective and stress test, or evaluate its vulnerabilities, across a range of plausible futures (Radke et al., 2017). For environmental water plausible futures may include those such as climate change scenarios (e.g., RCPs), response of species or communities including distribution models and vulnerability assessments, changes in water availability, trade and water quality, occurrence of disturbance events (e.g., drought, hypoxic blackwater) and change in land use. RDM allows managers to determine under what conditions the existing objectives or strategy performs well or fails, and what conditions affect performance. This would be ideal for testing vulnerabilities of existing environmental water goals under a range of possible futures. Alternative combinations of problems and uncertainties allow iterative assessments of scenarios to achieve satisfactory performance over a range of futures. Scenario two provided in this research could be tested in a RDM model to determine if and/or when to cease delivering environmental water to support the species identified by the objective, or in scenario one by running the options through numerous scenarios available in RDM which would indicate when different volumes of water can support both species, or when one will not survive under different water availability scenarios. Using RDM results of the scenarios will provide trade off curves that compare alternate strategies for achieving the goals identified by participants in this research and assist in making informed trade off decisions so environmental water managers can achieve the best bang for their environmental water buck. RDM outputs can also show where system ‘tipping points’ or vulnerabilities are, which can then be used in other methods such as Adaptation Pathways. While there are a number of academic examples of applying the RDM method to water resource management (Matrosov et al., 2013; Singh et al., 2015), RDM is currently not widely used or accepted in testing/setting objectives or policy (Jensen and Wu, 2016). This may be due to the following downsides of RDM; the model is data intensive, requires large computing capacity, and often needs specialist skills to run, analyse and interpret the results all of which make it expensive to execute (Jensen and Wu, 2016; Shi et al., 2019). Despite these challenges testing this method with environmental water use under climate change uncertainty would allow managers to consider future plausible hydrological and ecological changes and assist them in becoming more confident in their future decision making.

This research specifically focused on participants’ decisions regarding ecological trade offs, and deliberately omitted other factors in environmental water decision making (e.g., recreational use, socio-economic or political influences). The study acknowledges managers’ decisions may be different for reasons other than those investigated here and people make decisions based on a combination of their personal values and judgement, experience, organisational values, risk perception, political influence, availability of resourcing, and chances of success (Dietz, 2013; Mukherjee et al., 2018; Moallemi et al., 2020). Hence incorporation of climate change adaptations and the ultimate decision to implement adaptations will be affected by all these factors, along with geographic, legacy, economic and political differences, resulting in potentially different choices from managers in similar situations (O’Brien, 2009; Maani, 2013; Hagerman and Satterfield, 2014; Clifford et al., 2022).

It is also acknowledged that the methodology and data collected in this research have limitations. The data was gained from a small, purposive sample and cannot be generalised to the entire population of environmental water managers (Walter, 2019). Participation was voluntary so it is likely that participants are environmental water staff interested in climate change, and therefore likely to skew results to higher climate change interest than the whole population representing a level of sampling bias (Bryman, 2016). An additional limitation is the lack of opportunity for participants to request clarifications or explanations, especially when using scenarios, in online surveys (Walliman, 2015). Despite these limitations, this exploratory study acknowledges existing limitations of addressing uncertainty and provides research into new ways of embracing uncertainty in aquatic ecosystem management.

5 Conclusion

This research has shown that environmental water managers display all three responses to uncertainty (suppression, reduction and acknowledgement) with a large focus on reduction. The results highlight managers’ hesitancy in making decisions without full information. As climate change becomes embedded into legislative and strategic requirements of businesses and governments, the ability to incorporate adaptations despite these uncertainties becomes fundamental. However, environmental water managers will ultimately need to have conversations about if, or when, they cease managing waterways for certain species or communities, with or without what is deemed sufficient information. To ignore the need for such radical shifts will mean, in some cases, management becomes focused on unachievable ecological goals (Campbell et al., 2021; Judd et al., 2022). As well as failing to achieve those goals, such actions would be a poor use of a shared resource, would likely lead to loss of community support for environmental water, and may be maladaptative by reducing the likelihood of achieving other goals (environmental or otherwise). With the range of methods available to support decision making under deep uncertainty, environmental water managers have options to support decisions that can incorporate uncertainty and assist in water planning options and management decisions.

The results of this paper demonstrate that significant effort is still required to adopt decision making frameworks in environmental water management that are robust and well suited to handling the high levels of uncertainty associated with the future. Trialling methods for decision making under deep uncertainty will empower managers to acknowledge uncertainty, increase confidence, inform decision making and support conversations on future ecological objectives. Prior to widespread adoption (if deemed appropriate), less intensive options should be adopted immediately; such as scenario planning, climate change vulnerability assessments, use of adaptation decision frameworks and inclusion of reversible or low regret decisions. We acknowledge that all decision making is contextual, so we encourage managers to determine which method, or ideally range of methods, is best suited for their decision situation and employ the appropriate steps to get started.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Ethics statement

The studies involving human participants were reviewed and approved by La Trobe University Human Ethics Committee. The patients/participants provided their written informed consent to participate in this study.

Author contributions

MJ developed methods and performed data collection and analysis and prepared the manuscript. AH and NB reviewed methods and contributed to manuscript revision.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2023.1074896/full#supplementary-material

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A comparing vision of the lakes of the basin of Mexico: from the first physicochemical evaluation of Alexander von Humboldt to the current condition

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The Basin of Mexico is an endorheic lacustrine basin with an outstanding ecological and social history. There is evidence that it hosted human settlers since the late Pleistocene. This basin was home to great antique civilizations and many endemic species of flora and fauna. The main lake in the Basin was the Great Lake of Mexico, which was divided into five lakes and provided goods and services to the native communities. After the Spanish conquest, a rule was established to drain the lakes to prevent flooding in the city. The naturalist Alexander von Humboldt visited Mexico City in the early 1800s, and carried out the first formal scientific water quality analysis of the lakes of the basin. The Basin of Mexico gone through serious modifications due to urbanization and changes of land use reducing the lacustrine area to the virtual extinction of the lakes. The lakes are currently reduced to wetlands accounting for only 2.83% of the former lake and receiving mainly treated wastewater discharges. We carried out a comparative study between Humboldt's results and the current characteristics of water from these lake remnants analyzed with the same methods that he used. In addition, we assessed several morphometric parameters and performed water quality assessments using modern methods. Changes in water quality characteristics and ionic composition were detected, with Xochimilco being the lake with the highest water quality score and Texcoco and Chalco showing major alterations. The drastic reduction in the area of the remaining water bodies and the modifications in their water quality are discussed.

KEYWORDS

former great lake of Mexico, contrasting conditions over time, freshwater and saline lakes, water quality index, lake extinction

1 Introduction

The Basin of Mexico (also known as the Anahuac Basin), located south of the Mexican Plateau and at the center of the Trans-Mexican Volcanic Belt, is bordered by mountain systems that circumscribe it as an endorheic basin. This basin dates from the late Tertiary (Álvarez and Navarro, 1957) and previously harbored a large lake named "The Great Lake of Mexico". However, during the dry season, this large lake was separated into five lakes which, during pre-Hispanic times, were named Lake Zumpango, Lake Xaltocán, Lake Texcoco, Lake Xochimilco, and Lake Chalco (Legorreta, 2006). Thus, lakes were connected during periods of heavy rainfall and formed a single large water body, while in drought periods, the lakes became separate water bodies.

A sedimentary succession analysis with stable isotopes, diatoms, organic geochemistry, and tephrochronology have been allowed to identify conspicuous changes in the former Lake Texcoco shoreline between the late Pleistocene and the late Holocene. Additional switches included the exchange between aquatic and terrestrial plants (C3 and C4 plants), shifts from saline to alkaline and freshwater conditions, the influence of volcanic activity, marginal reworking of lake sediments, and the inflow of water drained from the basin (Lamb et al., 2009). Evidence has been found that humans settled in this basin since the late Pleistocene. Findings of the first humans in the Basin of Mexico are recorded near Chalco and El Peñón (Texcoco Lake springs), and Tepexpam. The presence of these skeletal remains close to the former lake suggests that this water body offered appealing resources for the development and survival of the first human settlers of the basin (González et al., 2003). Given the presence of humans since those times, is possible that they may have exerted environmental pressure by using the resources available in the basin and the water bodies. Other studies have shown that changes have occurred from saline to alkaline and freshwater conditions (Lamb et al., 2009), and other strong impacts due to volcanic activity and climatic changes, such as the last Glacial period.

The basin was home to several Mesoamerican civilizations. Among them, the most relevant was the Aztec empire, also named Mexicas, who settled on an island at the center of Lake Texcoco. The Aztec empire was at its peak when the Spaniards arrived. Other important civilizations settled in the Basin of Mexico were the Teotihuacans, Colhuas, Xochimilcas (in the littoral of Lake Xochimilco), Chalcas (in the littoral of Lake Chalco), and Xaltocanmecas (in the littoral of Lake Xaltocan), among others (Candiani, 2014; Torres-Alves and Morales-Nápoles, 2020). The lakes provided several ecosystem services to the local inhabitants, including water supply, food, and a means of transportation for the people and goods (Berres, 2000; Biar, 2020). The Aztecs modified the lacustrine system by building infrastructure such as dikes. In addition, they invented the farming system so-called *chinampas*—an interesting crop system composed by small artificial islands built in strips with sediments from the lake bottom, branches, and decaying vegetation, creating a network of channels serving as the irrigation system, with an average depth of 1.5 m.

The lakes functioned as means of communication: Lake Chalco was fed by freshwater draining from the mountains at the south and from springs. Lake Xochimilco was fed by the large number of springs in the area and by Lake Chalco; Lake Zumpango received water from the Cuautitlán River and fed Lake Xaltocan. Finally, Lake Texcoco, being at the bottom of the basin, received water from all the lakes and rivers (Torres-Alves and Morales-Nápoles, 2020).

Despite the dikes built by pre-Hispanic peoples, the basin maintained its lacustrine areas with changes in level associated with the rainy and dry seasons, suggesting that the original populations sustainably exploited the local resources (León Portilla, 1988). During the Spanish colonial period, the population increased and the urban area located within the lacustrine influence area expanded; consequently, the city was subjected to continuous floods that jeopardized its growth. In the 17th century, a work known as the *Tajo de Nochistongo* (Gurrfa, 1978) was built to divert the water of the Cuautitlán River, which fed Lake Texcoco. This work, together with the Tequixquiac tunnel, the Grand Canal and the so-called "Deep Drainage", was undertaken to drain off the lake system that had remained until then. These artificial drainages and urban growth, have caused drastic modifications in the basin and its lake systems (Alcántara and Escalante, 2005). As a result, neither the former lacustrine system nor the Nezahualcoyotl dike remains nowadays (Torres-Alves and Morales Nápoles, 2020; Montero-Rosado et al., 2022).

In the early 1800s, many European naturalists conducted scientific journeys to land still unknown at the time. The purpose of those journeys was not only the discovery of unexplored territories, but also investigation, including the collection of samples, the use of advanced scientific instruments, and the proposal of new taxonomic classification systems (Heck, 2020). The primary aim of Alexander von Humboldt (Berlin, Germany, 1769 – 1859) in his trip to the Americas (1799–1804) was data collection for the development of a science that had not yet been outlined and that was then called "the physics of the world". This was later called "theory of the Earth" and then Physical Geography (Heck, 2020). In Volume Eight of "*The American Travel Journal*", under the heading "*Chemical Analysis of the Lakes in the Valley of Mexico*", Humboldt detailed the analyses he carried out in early 1804 (in the dry season), on water samples from Lake Tescuco, Lake Zumpango, Lake Xaltocan, Lake San Cristóbal (the last one, was a water body separated from Lake Xaltocan, possibly by a dike or by the process of drying up of the former lake Texcoco), Lake Chalco, and Lake Xochimilco (Humboldt, 1802–1804), thus providing the first formal scientific analysis of lakes in the Basin of Mexico.

When Alexander von Humboldt visited Mexico City, the local population was only 160000 inhabitants (Humboldt, 2003). He noticed trends of increasing aridity and decreasing soil fertility from south to north (Humboldt, 2003). To date, the basin is home to one of the most densely populated cities in the world, considered a "Megacity" (UNESCO, 2018) with a population size above 20 million inhabitants (INEGI, 2023). According to Tortajada (2008), this huge urban area makes it extremely hard to provide services to the entire population, and generates large-scale environmental issues in the atmosphere and water bodies. This megacity lacks proper management supporting sustainable resource use, which has led to the depletion of some of the water bodies in the Basin of Mexico. Over several centuries, different environmental and social factors have led to the almost total disappearance of the lake system of the Basin of Mexico.

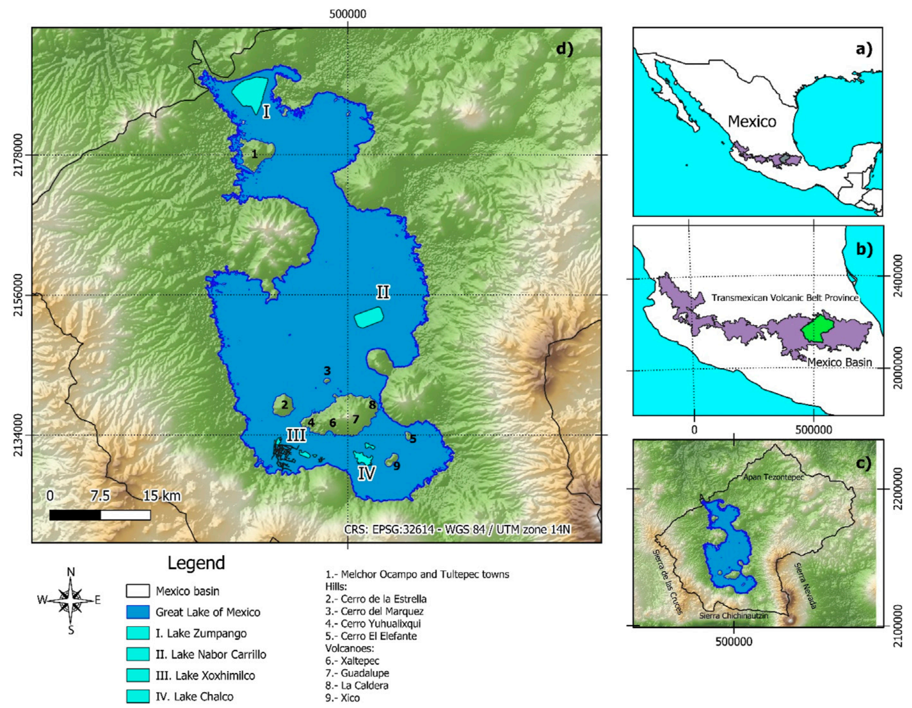

FIGURE 1
Study Area. **(A)** Reference map of the study area in Mexico. **(B)** Location of the BM into the Transmexican Volcanic Belt Province. **(C)** Basin of Mexico. **(D)** Surface area and shape of the former Great Lake of Mexico based on remote sensing and the location of the four remnant lakes studied.

This contribution aims to provide a comparative perspective not only in relation to Humboldt's analysis of the early 19th century, but also considering a more detailed recent study on water quality. The following particular objectives are derived from this work, a) Compare the results of the characteristics of the lakes of the basin of Mexico with the tests carried out by Humboldt with the results of recent water samples analyzed with the same methods used by Humboldt, b) analyze the significant changes in surface area of the still existing waterbodies concerning the surface area of the whole basin and based on Humboldt´s map, c) characterize water quality and assess a water quality index of the lakes using modern test methods in the dry and rainy seasons to obtain a current diagnosis. The implications regarding the state of conservation and the perspectives this work entails for management purposes are discussed. The implications regarding the state of conservation and the perspectives that this work entails for management purposes are discussed.

2 Methods

2.1 Study area

The endorheic Basin of Mexico (BM), also called the Valley of Mexico Basin (19° 29′52″N, 99° 7′37″W), is located in the central part of the Trans-Mexican Volcanic Belt Province belonging to the Mexican Transition Zone (Morrone et al., 2017; Morrone et al., 2022), with a mean altitude of 2,276 m above sea level. According to Arce et al. (2019) the area of the basin is of 9,620 km^2. The basin is delimited by the volcanic ranges of Sierra de las Cruces to the west, Sierra Nevada to the east, Sierra Chichinautzin to the south, and the Apan-Tezontepec volcanic range to the north (Arce et al., 2019; Martínez-Abarca et al., 2021) (Figure 1).

The predominant soil types in the area of the former Great Lake of Mexico are Phaeozem, Vertisol, and Solonchac; the latter is characterized by a high content of soluble salts (Sedeño-Díaz et al., 2019; CONABIO, 2023) (Supplementary Material; Figure 1A).

Today, the predominant climate in the Basin of Mexico (BM hereafter) is temperate [C (w0), C (w1), and C (w2)] according to Köppen's classification (García, 2004). Temperature ranges from 13°C to 25°C, with a mean annual temperature of 16°C; the mean annual precipitation varies between 700 mm and 900 mm, with the rainy season in the summer and up to 5% of rainfall in the winter (Alcocer and Williams, 1996; López-López et al., 2016) (Supplementary Material, Figure 1B).

During the Upper Cretaceous and Lower Tertiary, the BM was an open basin, with a surface drainage to the south through two main rivers, which produced large deposits of alluvial material at the bottom of the basin. Subsequently, during the Pleistocene, the basin closed off (Palma et al., 2022). In this sense, the lithological layers in the BM are dominated by alluvial deposits (with sedimentary rocks) and andesite-basalt (Supplementary Material, Figure 2).

The former Great Lake of Mexico was located within the BM (Figure 1) and was split into five lakes during the dry season, namely,: Texcoco, Zumpango, Xaltocan, Chalco, and Xochimilco (Berres, 2000; Montero-Rosado et al., 2022). Nowadays, only four lakes remain: Chalco, Texcoco (Nabor Carrillo reservoir, built in

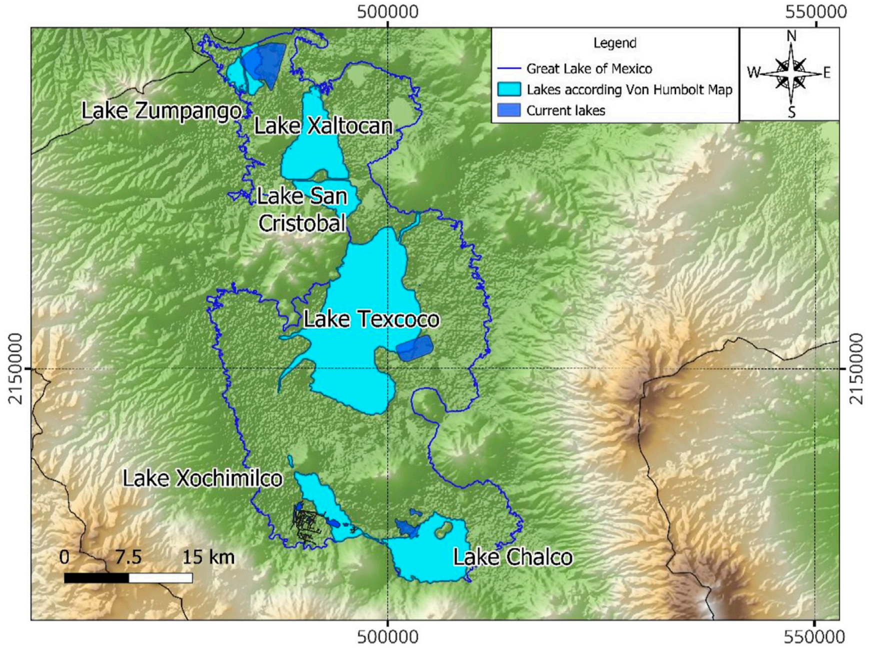

FIGURE 2
Estimated aspect of the BM lakes in 1807, based on Humboldt's map.

1983, covering a small area located where the great Texcoco Lake once was), Xochimilco, and Zumpango, all with major modifications (Figure 1). Texcoco and Xochimilco are currently protected natural areas considered wetlands and designated as Ramsar sites (Numbers 2469 and 1,363, respectively); however, both lakes receive the effluents of different wastewater treatment facilities (López-López et al., 2016; Morales-García et al., 2020). For its part, Lake Zumpango, with a storage capacity of 100 million m^3, functions like a regulating vessel. During approximately 2 months in the rainy season, it receives part of the water from the Cuautitlán River and the *Emisor Poniente* through the Santo Tomás canal. The water thus stored is used for agriculture, and the rest is diverted to the "*Gran Canal del Desagüe*", which drains into the Pánuco basin. Lake Chalco is mainly filled with rainwater; however, municipal and industrial wastewater treated and untreated is also discharged into it (Ortiz-Zamora and Ortega-Guerrero, 2007). The original connectivity between lakes was lost due to urban settlements in the former lacustrine area.

When Alexander von Humboldt visited Mexico, in addition to analyzing the water of the BM lakes, he drafted a map of the BM, including the lakes, which had already lost a considerable area by that time (the early 1800s). In the present work, we studied the same lakes Humboldt analyzed, except for Lakes Xaltocan and San Cristobal since both became extinct (Figure 2).

2.2 Spatial analysis

We conducted a morphometric analysis of the BM to determine its perimeter, maximum and mean areas, and the compactness coefficient (defined as the ratio between the basin perimeter and the circumference of a circle with an area equal to the basin area, as proposed by Gravelius, 1914). The model designed in this work from the surface of the Great Lake of Mexico was developed using the Digital Elevation Model (DEM) by Hydrosheds.org (2023), with a 3s resolution, and public shapefiles from Mexico's Institute of Statistics, Geography and Computer Sciences (*Instituto Nacional de Estadística, Geografía e Informática*; INEGI, 2023). The map entitled *Carte de la Vallée de Mexico*, drafted by Humboldt and published in 1811, was obtained from the website at David Rumsey Map Collection (2023). This map was georeferenced by the nearest neighbor method, using the landmarks marked in Humboldt's map to elaborate a new map superimposed on the DEM of Mexico Basin mentioned above. All spatial analyses were performed using the QGIS geographic information system (open-source system).

2.3 Fieldwork

The water bodies studied were monitored in two contrasting seasons: dry (March 2022) and rainy (October 2022). In each sampling period, the following environmental variables were recorded with the use of a Quanta® multiparameter probe (Hydrolab DS5): water and air temperature (°C), dissolved oxygen (DO mg L^{-1}), pH, salinity (UPS), turbidity (NTU), and conductivity (μS cm^{-1}). Additionally, 500 mL water samples were collected in polyethylene bottles, in duplicate. Samples were transferred in the dark and refrigerated (4 °C) for subsequent testing in the laboratory. The following water quality parameters were determined with HACHDR 3900, Hach® spectrophotometer techniques: nitrites (NO_2, mg L^{-1}), nitrates (NO_3, mg L^{-1}), ammonium (NH_4, mg L^{-1}), total nitrogen (TN, mg L^{-1}),

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TABLE 1 Results of the main morphometric parameters of the Basin of Mexico obtained with GIS and the digital model elevation.

Parameter	Calculation method	Value	Units
Length (L)	Directly in GIS	142.5	km
Perimeter (P)	Directly in GIS	529.8	km
Maximum Width (MW)	Directly in GIS	112.7	km
Area (A)	Directly in GIS	9,219.3	km^2
Mean Width (W)	$W = A/L$	64.70	km
Compactness coefficient	$IK = 0.282P/\sqrt{A}$	1.55	Dimensionless

orthophosphates (PO_4, mg L^{-1}), total phosphorus (mg L^{-1}), alkalinity (mg L^{-1}), biochemical oxygen demand (BOD_5, mg L^{-1}), color (U-PtCo), and total suspended solids (TSS mg L^{-1}). Total hardness was quantified by titration with EDTA ($CaCO_3$, mg L^{-1}). Alkalinity ($CaCO_3$, mg L^{-1}) was quantified by titration according to APHA (2005) techniques. In addition, at each study site, 100 mL of water was collected in Whirl-Pak bags for microbiological analyses. These water samples were tested for total and fecal coliforms according to the APHA MPN technique (2005).

Separately, chlorophyll *a* (Chl *a*) was quantified as per the standardized APHA technique (APHA, 2005). To this end, 500 mL of water collected from each study site was filtered through Whatman filters (0.45 µm), followed by extraction in acetone (90%) for 24 h in the dark under refrigeration (4 °C) prior to the Chl *a* analysis.

Another 50 mL water sample was acidified and digested to determine Cl^-, SO_4^{2-}, Na^+, K^+, Ca^{2+}, and Mg^{2+} using Inductively Coupled Plasma Optical Emission Spectrometry (ICP OES) techniques (Perkin Elmer Optima 8,000) at the CICATA Legaria laboratory of the *Instituto Politécnico Nacional*.

2.4 Water quality index

To obtain a single parameter indicating the water quality characteristics of the current lakes, we calculated the water quality index (WQI) proposed by Dinius (1987) for each study lake and period. This index includes 13 environmental variables: dissolved oxygen saturation (%), atmospheric and water temperature, pH, biochemical oxygen demand, alkalinity, nitrates, conductivity, chlorides, hardness, true color, and fecal and total coliforms. The output of this WQI ranges from 0 to 100, which is suitable for a better understanding of water quality and decision-making on water use, and as support of management programs. Mean WQI values were assessed using two different data sets: individual lakes (WQI values for all study periods), and study periods (WQI values for all study lakes for each study period).

2.5 Statistical analysis

All data were assessed for normality and homoscedasticity. Then, ANOVA followed by Tukey's multiple comparison test or Kruskal–Wallis followed by Duncan's multiple range test were performed for parametric or non-parametric data, respectively, to test the data for significant differences between lakes and between study periods. Box-plot graphs were constructed to depict the physicochemical and biological variables evaluated, by study lake and study period, using the XLSTAT software (Addinsoft, 2023). Heatmaps were constructed with the mean values and the standard error of the main physicochemical characteristics evaluated. A principal component analysis (PCA) was performed, after a factorial analysis of all the physicochemical factors evaluated, to identify the variables with the greatest contribution to the trends between study

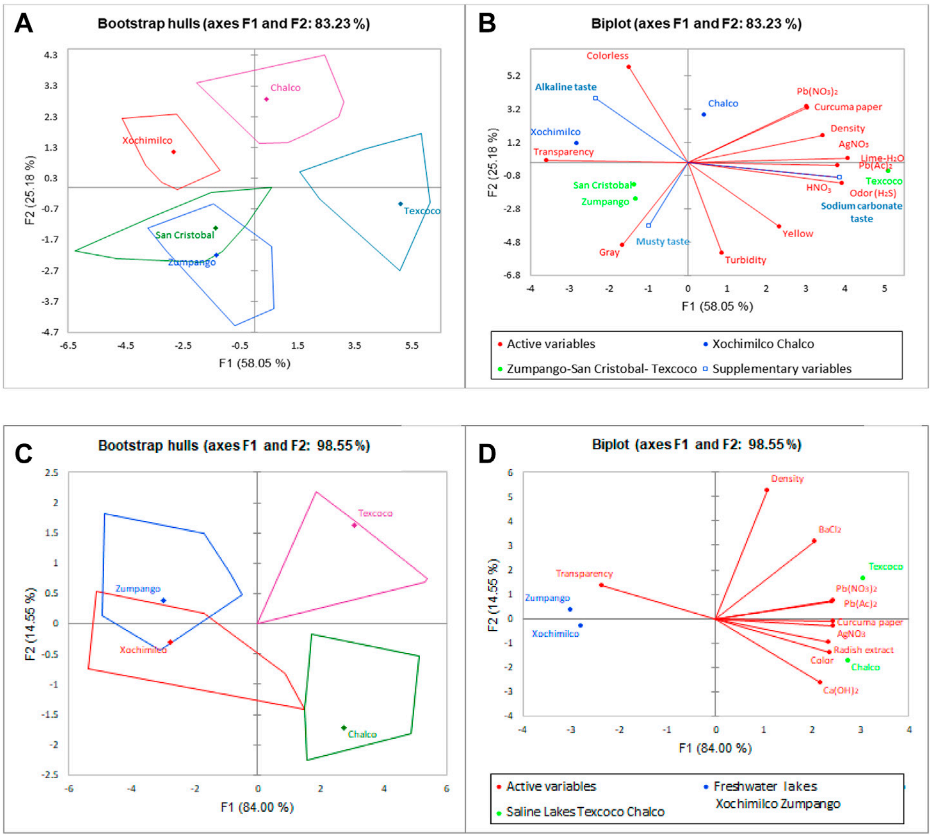

FIGURE 3
Biplot of the lakes of the BM according to the variables studied by Humboldt. **(A)** Convex hulls of each lake studied in the early 1800s; **(B)** arrangement of the lakes along environmental gradients, and vectors of the tests carried out by Humboldt; **(C)** convex hulls of each lake studied in 2022; **(D)** arrangement of the lakes along environmental gradients, and vectors of the tests carried out in the present study using the same methods as Humboldt.

sites. The data included the four lakes studied, the factors recorded in the field, and the results of the physicochemical testing carried out in the laboratory; all the variables were transformed to log(x+1). The PCA was performed with Pearson's correlation using the XLSTAT software (Addinsoft, 2023).

In the case of Na^+, Ca^{2+}, K^+, Mg^{2+}, Cl^-, and HCO_3^-, a Piper diagram was constructed using the free software Easy Quim V.5 (Vazquez-Suñe and Serrano-Juan, 2012) to classify the lakes according to their ionic composition.

As for the information generated by Alexander von Humboldt, we reviewed his diary (Humboldt, 1802–1804). From it, we obtained his records about the test results of the water samples collected from the center of the lakes (Zumpango, San Cristobal, Texcoco, Xochimilco, and Chalco). These tests and observations included color, odor, taste, density, and the reactions from testing water samples with turmeric Curcuma paper (alkalinity) and the reactants lead acetate (hydrogen sulfide), silver nitrate (chloride), lead nitrate (sulfate), lime water (carbonates), and nitric acid (bicarbonates) (Richter and Engshuber, 2014).

With Humboldt's results, a table was elaborated to summarize all the test results. Given that the results obtained by Humboldt are semi-quantitative, they were coded to provide quantitative data for a PCA. The coding involved using a 0-to-5 scale, where 0 means no reaction to the test and 5 indicates the maximum reaction reported by Humboldt. Variables such as colour and taste, were considered as categorical. The PCA aims to identify the similarities and differences between lakes, as well as any environmental gradients in the water bodies based on the lake parameters recorded by Alexander von Humboldt. The PCA was performed with Pearson coefficient and using the XLSTAT software (Addinsoft, 2023). The same procedure used for the PCA of our data was used for Humboldt´s data.

3 Results and discussion

3.1 The basin of Mexico

The analysis of basin morphometry indicates that the BM comprises an area of 9,219.3 km^2, with an approximate total length of 142.5 km in its main axis, oriented in a southwestern–northeastern direction, a maximum width of 112.7 km, a perimeter of 529.8 km, and a compactness coefficient score of 1.55. According to Faye and Ndiaye

TABLE 2 Estimation of the lake area lost from pre-Hispanic times to the present day.

Water body	Lake area(km^2)			Percentage of remaining lake area
	Pre-Hispanic period Estimated in this study	Humboldt 1803	2023	
Chalco	1280.00	83.05	4.20	5.06
Xochimilco		35.90	4.07	11.32
Texcoco		210.12	9.29	4.42
San Cristobal		31.31	---	---
Xaltocan		50.53	---	---
Zumpango		17.35	18.68	107.66
Total area	1280.00	428.26	36.24	
Percentage of remaining lake area	100.00	33.46	2.83	

TABLE 3 Summary of the variables assessed by Humboldt in the lakes of the Basin of México during his visit to the Basin of Mexico (1804) and the recent samples in 2022.

Lake	Xochimilco		Zumpango		San Cristobal		Chalco		Texcoco	
	Humboldt study	Current	Humboldt study	Current	Humboldt study	Current	Humboldt study	Current	Humboldt study	Current
Color	-	Light brownish	Yellow-Gray	Green colorless	Gray	NA	-	Green	Yellow	Green
Transparency	+++	+++	++	++++	+	NA	+	-	+	+
Turbidity	-	++	++	-	+++	NA	-	++++	++	++++
Odor	-	-	H_2S +	-	-	NA	-	H_2S +	H_2S ++	-
Taste	Alkaline	Alkaline	Musty	Musty	Alkaline	NA	Alkaline	Alkaline	Sodium carbonate	Sodium carbonate
Density	1.0009	1.002	1.0111	1.003	1.0129	NA	1.0171	1.002	1.0215	1.005
Curcuma paper	-	-	-	+	+	NA	+++	++++	++	++++
$PbNO_3$ Pp White	-	+	+	-	++	NA	++++	+++	+++	++++
*$Pb(C_2H_3O_2)_2$ Pp White	++	++	+	+	++	NA	++	+++	Black Pp	++++
$AgNO_3$ for Cl^-	-	+	++	-	+	NA	+++	++	++++	++
Lime water	Precipitate	-	Precipitate	-	Precipitate	NA	Precipitate	+	$NaCO_3$	++
HNO_3	Almost nule bubble	-	Almost nule bubble	-	Almost nule bubble	NA	Almost nule bubble	-	Bubble ++	-
Radish extract	No effect	Violet	Blue green	Blue green	-	NA	Dark green	Dark green	Green	Green

Note: The curcuma filter paper was used to detect alkalinity the filter paper used reacts red in alkaline solution.
$Pb(NO_3)_2$ and $Pb(C_2H_3O_2)_2$ indicate H_2S. Lime water $Ca(OH)_2$ and HNO_3 indicate carbonate (in the case of HNO_3 the presence of carbonate is evidenced by bubbling). Radish extract. Green colour is presented under alkaline conditions.
NA: Extinct Lake, data no available

(2021), the compactness coefficient (IK) can be used to classify lakes into four different shapes: circular (1), squat (1–1.15), intermediate (1.15–1.5), and elongated (1.5 and above); therefore, the BM has an elongated shape, with its main axis oriented as mentioned above (Table 1). Some of these values differ from those obtained by Arce et al. (2019), namely, 9,620 km^2 in area, 100 km in length, and 80 km in width; separately, Alcocer and Williams (1996), based on the work of Alvarez and Navarro (1957), reported a total length of 125 km, 90 km in width, and a mean area of 9,600 km^2, which are similar to the figures reported by Arce et al. (2019).

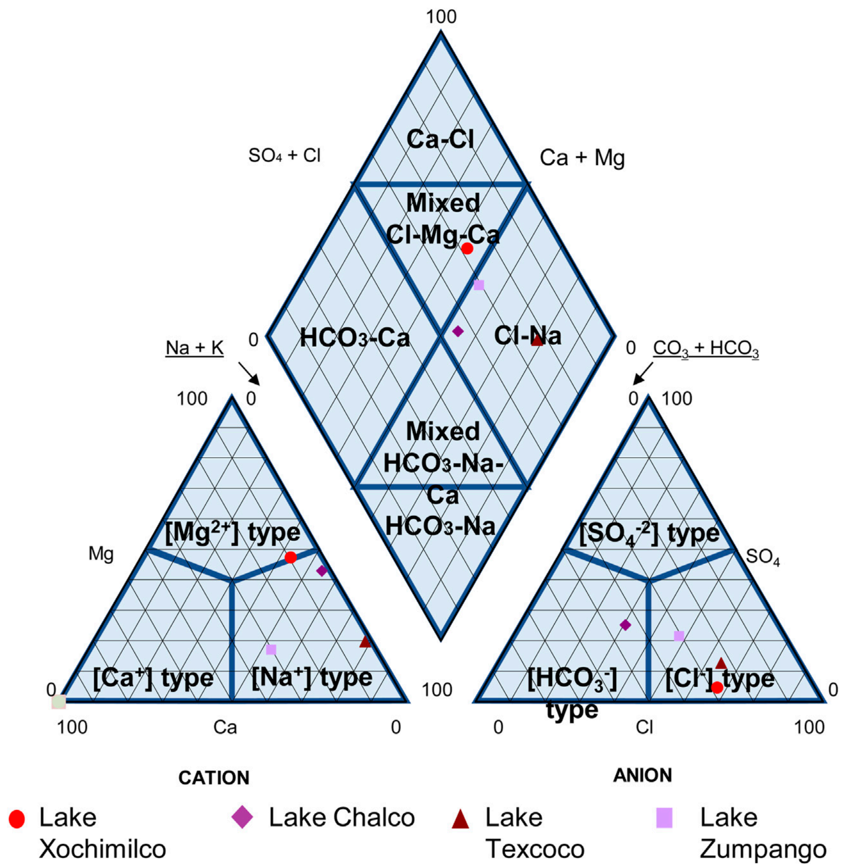

FIGURE 4
Piper diagram of the lakes of the Basin of México.

3.2 Reduction in Lake surface area

In this study, using GIS tools, the Digital Elevation Model (DEM) from Hydrosheds.org (2023), and considering 2,250 masl as the high watermark of the lakes as proposed by Torres-Alves and Morales-Nápoles (2020), we derived a new shape for the Great Lake of Mexico, estimating an area of 1,280 km^2 and a perimeter of 644.2 km (Figure 1). Based on this new shape, it is possible to observe that the hills named La Estrella, Yuhualixqui, El Marquez, and El Elefante, and the volcanoes Xico, Guadalupe, La Caldera, and Xaltepec were previously islands located within the Great Lake of Mexico, as well as the area that currently harbors the Tultepec and Melchor Ocampo towns, located in the northern area of the former lake, since this area exceeds 2,250 m asl. This result is consistent with the alluvial deposits in the BM (Supplementary Material, Figure 2). Several authors have proposed models that estimate the surface area of the Great Lake of Mexico. Cruickshank García (1998) estimated that the lake region measured almost 2000 km^2; Torres-Alves and Morales-Nápoles (2020) indicated an area of 1,000 km^2; and Alcocer and Williams (1996) calculated an area of 7,868 km^2.

Therefore, considering the original lacustrine area of 1,280 km^2 as the original (baseline) area (100%), the remaining lacustrine area in Humboldt's time (1803) was 428.26 km^2, accounting for 33.46% of the original area. This implies a loss of two-thirds of the original lake area by the early nineteenth century. Today, the remaining lacustrine area is a mere 2.83% of the original area of the Great Lake of Mexico (Table 2). Given its current role as a regulating vessel, Lake Zumpango has increased in area by 7.66% compared to the area shown in Humboldt's map (Figure 2). The lake with the greatest loss of area from Humboldt's time to date is Lake Texcoco (95.57%). For its part, Lake Xochimilco represents 11.32% of the area of all lakes in Humboldt's time; this scenario may be due to the increasing number of *chinampas* being built in this area. Tussupova et al. (2020) state that many saline lakes around the world are drying up rapidly due to anthropogenic activities, causing local adverse effects on health (lung diseases) air quality (excess dust), and ecological impacts (e.g., biodiversity), among others. In the case of the BM lakes, the decision to drain the lakes and urban growth since the Spanish colonial period has brought about major consequences.

These results highlight the extreme reduction of the lake area in the BM, which has brought with it a shift in the cultural perception of the BM landscape from a lakescape in the pre-Hispanic era to an urban landscape today (Biar, 2020).

3.3 Revisiting Humboldt's results

The values of the water variables tested in the five lakes in the early 1800s and obtained from Humboldt´s diary (Humboldt, 1802–1804) are summarized in Table 3.

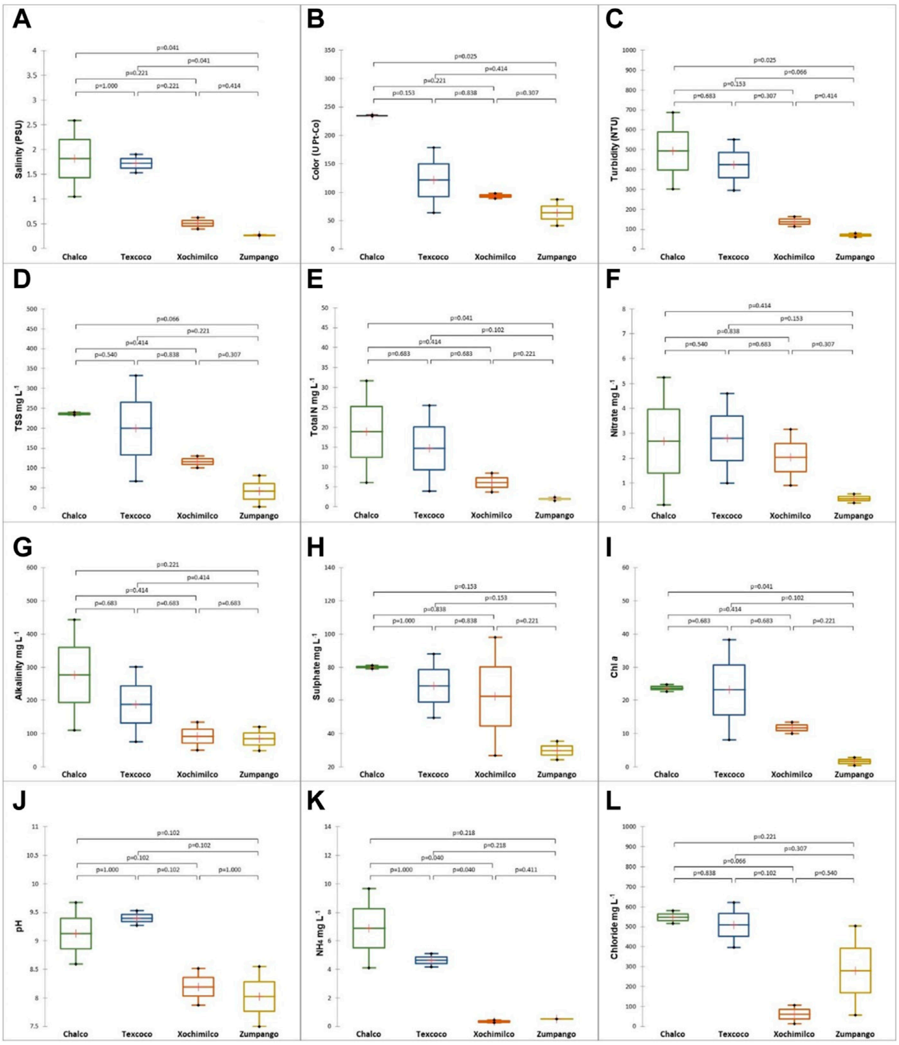

FIGURE 5
Box plots of environmental variables that show marked differences between lakes. **(A)** Salinity, **(B)** Color, **(C)** Turbidity, **(D)** Total Suspended Solids (TSS), **(E)** Total Nitrogen, **(F)** Nitrates, **(G)** Alkalinity, **(H)** Sulfates, **(I)** Chl a, **(J)** pH, **(K)** Ammonia, **(L)** Chloride.

Based on Table 3, Lakes Texcoco and Chalco were denser and more saline than the rest of the BM lakes; however, Lake Chalco had colorless clear water (i.e., no turbidity). On the other hand, Lake Xochimilco water was odorless and showed the lowest density. Currently extinct, Lake San Cristobal had turbid water with a density intermediate between Lakes Texcoco and Xochimilco. Contrary to some reports (Berres, 2000; Torres-Alves and Morales-Nápoles, 2020), Lake Chalco was relatively saline, not entirely freshwater. Humboldt recorded this water characteristic in the early 1800s, and Caballero and Ortega-Guerrero (1998) mentioned that Lake Chalco had been saline in some periods of the recent past.

The principal component analysis of the lakes based on the variables evaluated by Humboldt shows an explained variance of 83.23% in its first two components (Figures 3A, B). In the convex hulls diagram, Lakes Texcoco, Chalco, and Xochimilco are separate units, while Lakes Zumpango and San Cristobal are grouped in the same cluster (Figure 3A). The dispersion of the lakes along the environmental gradients formed by the variables tested by Humboldt is notorious. On the far right (Figure 3B), we can see the vectors of the variables that are associated, including density, and several tests such as the turmeric Curcuma paper, $Pb(NO_3)_2$, $AgNO_3$, lime water, $Pb(Ac)_2$, HNO_3, sodium carbonate taste, hydrogen sulfide odor, and yellowish color, all of which attain the highest values at the far right of the diagram; Lake Texcoco Lake is located at this end of the gradient. In contrast, on the far left of the diagram, the vectors corresponding to transparency, alkaline taste, colorless, greyish color, and musty taste are observed. Lake Xochimilco is located on the far left of the diagram, characterized by its non-turbid, colorless water with an alkaline taste, characteristics that contrast with those of Lake Texcoco. The environmental conditions in Lakes Xochimilco and Texcoco also form a gradient ranging from less dense and less saline water to denser and more saline conditions. Along this gradient, Lakes Zumpango

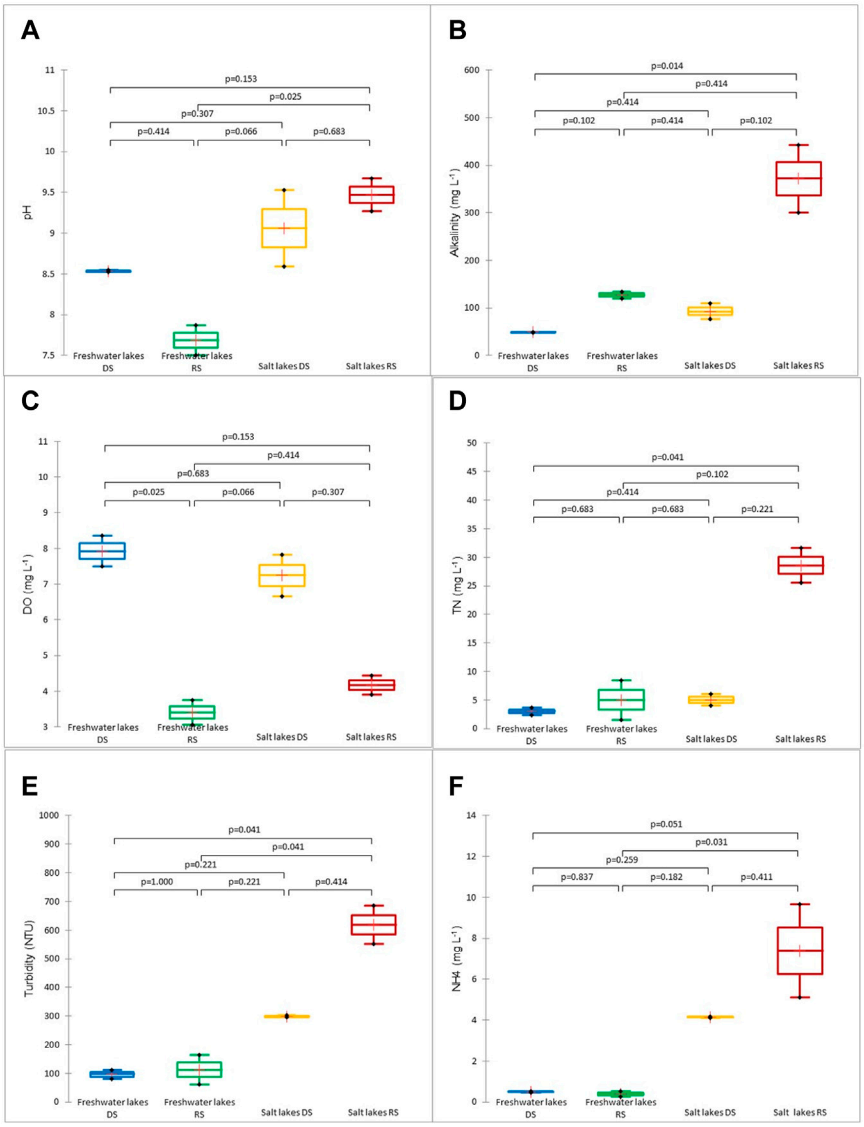

FIGURE 6
Box plots of environmental variables that showed significant differences between study periods for the types of lakes. **(A)** pH, **(B)** alkalinity, **(C)** dissolved oxygen (DO), **(D)** total nitrogen (TN), **(E)** turbidity, **(F)** ammonium.

and San Cristobal are positioned on the lower left quadrant, characterized by grayish water with greater turbidity and a musty taste; regarding the tests with reactants, these lakes were less reactive than Texcoco but more than Xochimilco. Finally, Lake Chalco, positioned in the upper right quadrant, was characterized by reacting to turmeric curcuma paper and silver nitrate, and by a higher density than that of Lakes Xochimilco, Zumpango, and San Cristobal, but lower than that of Lake Texcoco. Lake Chalco was positioned opposite to Lakes Zumpango and San Cristobal since its water was more transparent, colorless and with an alkaline taste, and reacted to silver nitrate and turmeric paper (Figure 3B).

According to Humboldt's results, the high Cl^-, HCO_3^-, and CO_3^{2-} contents are worth noting; together with alkalinity, these results reveal the alkaline soda-lake nature of Lake Texcoco. According to Jones et al. (1998), soda lakes are characterized by the predominance of NaCl, $NaHCO_3$, and Na_2CO_3, which are consistent with the findings of Humboldt from his qualitative chemical tests. In contrast, Lake Xochimilco turned out to be a freshwater lake with conditions opposite to those of Lake Texcoco, Lake Chalco stood out for its high alkalinity, with carbonate-rich waters, and ranked second in terms of its high density. Lakes Zumpango and San Cristobal showed intermediate characteristics between Lake Texococo (soda lake) and Lake Xochimilco (freshwater lake). Based on the above, the BM lakes showed an environmental gradient ranging from freshwater to saline soda lakes, with a strongly alkaline lake that does not reach a soda-lake state (Lake Chalco). Likewise, the spatial view of the BM is consistent with Humboldt's perception in the early 1800s that everything was interconnected (Holl, 2018). Humboldt claimed that the lack of water might turn the valley sterile and unhealthy, increasing the salinity and aridity. He observed that the aquatic plants covering the lakes released

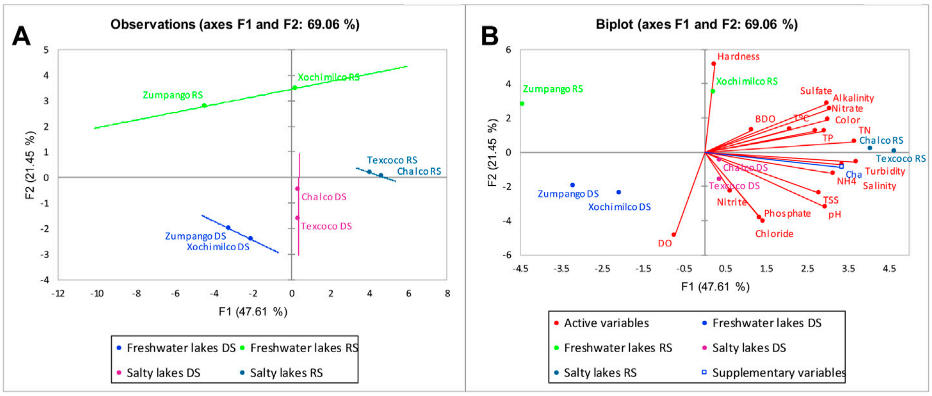

FIGURE 7
Biplot of the PCA with samples of the study lakes in the Basin of Mexico. **(A)** Groups of lakes by season, **(B)** arrangement of the lakes along environmental gradients, and vectors of the environmental factors assessed.

hydrogen sulfide, which could be perceived when the wind blew across Lake Texcoco (Humboldt, 2003, 256).

3.4 Physicochemical characteristics of the lacustrine remnants of Lake Texcoco

3.4.1 Testing with Humboldt's methods

The results of our tests of water samples collected from the study lakes using Humboldt's methods are shown in Table 3. The PCA of these data shows that the first two principal components accounted for 98.55% of the explained variance. The lakes in the bootstrap hulls diagram showed the distribution of the lakes along the gradient of the environmental conditions assessed (Figure 3C). Lakes Zumpango and Xochimilco were grouped in the same cluster positioned on the left side of the biplot, while Lakes Chalco and Texcoco were on the right side of the biplot, although as separate lakes, each with its own particular conditions. The vectors of the tests showed that Lakes Texcoco and Chalco have colorful water of the highest density that reacted with $Pb(NO_3)_2$, $Pb(Ac)_2$, $AgNO_3$, curcuma paper, and radish extract. Both lakes reacted with $Ca(OH)_2$, but the reaction was more intense for water from Lake Chalco; furthermore, water from both lakes also reacted with $BaCl_2$, but the reaction was stronger in water from Lake Texcoco (Figure 3D). These results suggest remarkable changes in the chemical composition of the lakes. In contrast to Humboldt's findings, Lake Chalco is currently more saline, sharing some environmental characteristics with Lake Texcoco, such as high alkalinity, color (green color due to algal blooms), chloride, HS_2, sulfates, and, particularly, carbonates.

Another contrast versus Humboldt's data is that Lakes Xochimilco and Zumpango are currently very similar in chemical composition, both being less saline, more alkaline, and colorless; in general, the reactions tested in water from both lakes were null or less intense than those in water from Lakes Chalco and Texcoco.

3.4.2 Ionic composition

The Piper diagram showed that almost all lakes, except for Lake Xochimilco, were plotted in the [Na+] type field of the lower-left triangle (Figure 4), while Xochimilco lies in the boundary with the [Mg+] and [Na+] type fields, suggesting that the lakes of the Basin of México are dominated by the cation Na+. These conditions are common in areas with arid climates, as stated by Shengbin, et al. (2022) for the water bodies of the Tibetan Plateau. In the case of the major anions, Lakes Xochimilco, Texcoco, and Zumpango are plotted in the [Cl−] type field of the lower-right triangle (Figure 4), and Lake Chalco was plotted in the [HCO3−] type field. This diagram suggests that the Basin of México lakes evolved from the fresh hydrochemical facie of the [HCO3−] type to the saline Cl− type. Furthermore, the central diamond shape of the Piper diagram (Figure 4) also shows that the hydrochemical facies of the Basin of Mexico lakes evolved from the fresh HCO3–Ca type to the saline Cl–Na type. The Basin of México lakes are currently dominated by the saline Cl–Na type, with Lake Chalco being very close to the HCO3–Ca type, and Lake Xochimilco being of a mixed Cl–Mg–Ca type.

3.4.3 Physicochemical characterization

According to the physicochemical results of the water bodies that still persist in the Basin of Mexico, the lakes showed remarkable differences in salinity. Lakes Chalco and Texcoco reached the highest values (1.82 ± 0.77 UPS and 1.72 ± 0.19 UPS, respectively), while Lakes Xochimilco and Zumpango had the lowest (0.51 ± 0.11 UPS and 0.27 ± 0.01 UPS, respectively) (Figure 5A). Accordingly, Lakes Texcoco and Chalco are referred to as saline lakes hereafter. At the same time, Xochimilco and Zumpango are considered freshwater lakes. The color, turbidity, and total suspended solids, total nitrogen, nitrates, alkalinity, sulfates, and Chl *a* showed the same behavior, where the highest mean values were observed in the following ranking order: Chalco > Texcoco > Xochimilco > Zumpango (Figures 5B–I; Table 4). The pH showed its highest values in Lake Texcoco, followed by Chalco (9.3 ± 0.03 and 9.1 ± 1.08, respectively), and with the lowest values in

TABLE 4 Heatmap of the Mean values and ± standard error of the main variables showing differences between lakes.

Lake	Salinity (PSU)	Color (Pt-Co)	Turbidity (NTU)	TSS (mg L^{-1})	TN (mg L^{-1})	Nitrate (mg L^{-1})	Alkalinity (mg L^{-1})	Sulfate (mg L^{-1})	Chl a (mg L^{-1})	NH4 (mg L^{-1})	Chloride (mg L^{-1})	pH
Chalco	1.82±0.77	234.75±2.5	494±384	236.5±7	18.85±25.6	2.68±5.14	276.4±332.8	80±2	23.69±2.08	6.88±5.53	547.05±65.02	9.13±1.08
Texcoco	1.72±0.19	121.25±57.25	423±128	199.75±132.75	14.25±11.25	2.65±1.95	173.2±112.88	68.72±19.27	23.18±15.09	4.65±0.47	508.49±112.69	9.3±0.03
Xochimilco	0.515±0.11	93.5±4.5	137.77±25.22	116±15	6.075±2.42	2.025±1.12	92.1±42.1	62.42±35.57	11.74±2.08	0.35±0.11	60.78±46.64	8.19±0.32
Zumpango	0.27±0.01	64.25±23.25	70.75±9.35	41.75±39.25	2.0	0.37±0.17	84±36	29.85±5.65	1.63±1.26	0.53±0.07	279.88±223.34	8.02±0.52

Lakes Xochimilco and Zumpango (8.19 ± 0.32 and 8.02 ± 0.52, respectively) (Figure 5J; Table 4). Ammonium reached higher values in the saline lakes (6.88 ± 5.53 mg L^{-1} and 4.65 ± 0.47 mg L^{-1}, corresponding to Chalco and Texcoco, respectively), and values lower than 1 mg L^{-1} were recorded in Lakes Xochimilco and Zumpango (Figure 5K; Table 4). Finally, chlorides were higher in the saline lakes, with average values of 508.49 ± 112.69 and 547.05 ± 65.02 in Lakes Texcoco and Chalco, respectively, while Lakes Xochimilco and Zumpango showed chloride values of 60.78 and 279 mg L^{-1}, respectively (Figure 5L). To note, all the lakes tested positive for fecal coliforms, with peak values of up to 800 ± 1600 MPN in Lake Chalco.

Regarding the differences between study periods, when comparing the mean salinity of saline and freshwater lakes, no significant differences were detected between the dry and rainy seasons (p > 0.05) (Table 5). Seasonal differences (although not significant) in pH were only observed for Lakes Xochimilco and Zumpango, but not for the saline lakes (Figure 6A). As for alkalinity and dissolved oxygen, seasonal differences (although not significant) between the dry and rainy seasons were observed for saline and freshwater lakes (Figures 6B, C). The highest alkalinity values were recorded in the rainy season for both types of lakes; in the case of dissolved oxygen, the highest values were observed in the dry season in both types of lakes (Table 5). On the other hand, for total N, ammonium, and turbidity, higher values were observed in saline lakes during the rainy season (Figures 6D–F). On the other hand, temperature, salinity, total P, orthophosphates, sulfates, color, chlorides, hardness, total suspended solids, BOD_5, fecal coliforms, nitrites, and Chl *a* did not show seasonal differences (p > 0.05) between the dry and rainy seasons.

3.4.4 Integration of environmental variables and study periods

The principal component analysis of the study sites and the environmental variables showed an explained variance of 69.06% in its first two components. The diagram shows the dispersion of the study sites along environmental gradients in which clusters can be identified, with the saline lakes (Chalco and Texcoco) positioned on the right quadrants of the diagram (Figure 7A). In general, these lakes are characterized by the highest salinity, turbidity, and total suspended solids, and were also rich in nutrients (N and P) and attained high color scores (Figure 7B). Additionally, during the rainy season, these lakes showed higher ammonium and nitrate levels, as well as the highest Chl *a* concentration. On the other hand, in the dry season, these lakes had higher values of nitrites, orthophosphates, and chlorides, as well as the highest pH values (Figure 7B). For their part, the freshwater lakes, Xochimilco and Zumpango, were positioned on the left quadrants of the diagram (Xochimilco in the rainy season, on the margin of the upper right quadrant) (Figure 7A). During the dry season, the freshwater lakes, in addition to their lower salinity, attained the lowest alkalinity, nitrites, sulfates, nutrients (TN and TP), and BOD_5. For its part, Lake Xochimilco during the rainy season showed high hardness and lower orthophosphates, nitrites, chlorides, and dissolved oxygen, while in the dry season, it showed lower salinity and

TABLE 5 Heat map of the mean values and ± standard error of the variables that show differences between seasons (RS: Rainy season, DS: Dry season).

Lake group	pH	Alkalinity (mgL^{-1})	DO (mgL^{-1})	TN (mgL^{-1})	Nitrate (mgL^{-1})	Turbidity (NTU)
Texcoco and Chalco RS	9.47±0.2	371.6±71.2	4.16±0.26	28.57±3.07	4.925±0.32	618.5±67.5
Texcoco and Chalco DS	9.06±0.37	93±17	7.24±0.59	5.02±1.52	0.555±0.35	298.5±3.5
Xochimilco and Zumpango RS	7.68±0.18	127.1±7.1	3.405±0.345	5.02±3.45	1.85±1.3	112.2±50.8
Xochimilco and Zumpango DS	8.535±0.43	49±1	7.9225±0.43	3.02±0.62	0.55±0.35	96.32±16.22

concentration of nutrients (TN, NH_4, nitrates, nitrites, TP, and orthophosphates), as well as the lowest turbidity, total suspended solids, pH, alkalinity, sulfates, color, and BOD_5 (Figure 7B).

The physicochemical factors assessed suggest an environmental gradient: Lakes Chalco and Texcoco are clustered for their high salinity, while Lakes Xochimilco and Zumpango showed the lowest salinity. Saline lakes are also characterized by high alkalinity, nutrient enrichment, and, particularly, high chloride levels.

Despite the geographic proximity of Lake Chalco to Lake Xochimilco (both in the south of the basin), our results show that they currently have contrasting conditions in terms of salinity, chloride content, and alkalinity. All the lakes studied have faced a declining volume of water due to water drainage and extraction from their aquifers (Soto-Coloballes, 2019). The water bodies that were virtually driven to extinction are Lake Chalco (Ortega-Guerrero, et al., 1993) and Lake Texcoco (Soto-Coloballes, 2019). The intense exploitation of the aquifers of the Basin of Mexico has led to the progressive subsidence of this area, which in the case of Lake Chalco amounts to 40 cm/year, giving rise to an extensive plain. This subsidence has led to the formation of a "new Lake Chalco" in this topographic depression, with the water surface 12 m below the original ground level (Ortega-Guerrero, et al., 1993; Ortíz-Zamora and Ortega-Guerrero, 2007). This new Lake Chalco is fed by runoff water and the inflow of streams from mountainous areas, but also by untreated wastewater from adjacent towns and industrial areas that reaches the new lake through canals (Ortíz-Zamora and Ortega-Guerrero, 2007). According to paleolimnological studies, between 39,000 years BP and approximately 6,000 years BP, Lake Chalco has faced several episodes of high salinity and alkalinity alternating with acidity and freshwater conditions (Bradbury, 1989; Caballero and Ortega, 1998). Berres (2000), in his study on the ichthyofauna of the basin of Mexico, pointed out that the southern lakes of the Basin of Mexico, i.e., Lakes Chalco and Xochimilco, are freshwater lakes; however, our results show that Lake Chalco currently has alkaline and saline conditions, indicating a drastic environmental change. These conditions may result from the flooding of the so-called "New Lake Chalco", thereby incorporating solutes previously precipitated during the draining of the former Lake Chalco. Those sediments were exposed to air and dried up through evaporation, resulting in saline soils; when the New Lake Chalco floods, these solutes are incorporated into the lake water.

Lake Xochimilco was also subjected to water extraction to supply drinking water to the downtown area. The declining inflow from springs led to a reduction in the surface area of the lake; this water body currently receives treated wastewater in addition to untreated domestic wastewater from households adjacent to the canals. According to our results, its waters show freshwater conditions, although with high contents of chloride and fecal coliforms.

Today, Lake Zumpango is an artificial reservoir located in the depression of the former Lake Zumpango and receives an inflow of runoff water from rainfall and treated wastewater. The former Lake Zumpango received the water from its tributary, the Cuautitlán River. Recently, however, this river has been diverted and now discharges its water in the Tajo de Nochistongo to avoid inputs to Lake Zumpango and, from it, to Lake Texcoco. The main water source entering Lake Zumpango is treated wastewater; consequently, the lake is now covered by the aquatic weed *Eichornia crassipes*. *E. crassipeps* (Water hyacinth) is an invasive species included among the 100 world´s worst invasive alien species (Lowe et al., 2000). This species reaches a fast growth leading to covering the total surface of lakes which produces high evapotranspiration, prevents the passage of light into the water column (limiting the photosynthetic activity), the gaseous exchange of the atmosphere with the water surface (causing depletion of dissolved oxygen in the water column), and promotes high evapotranspiration, thus represents a risk to remaining native biodiversity and the maintenance of water volume of lakes.

All the lakes studied, remnants of the former Great Lake of Mexico, have been desiccated and currently receive mainly treated wastewater from various treatment plants (PAOT, 2014; Soto-Coloballes, 2019), which brings about major changes in water quality, including nutrient enrichment. Lakes Chalco and Texcoco are indeed facing salinization and eutrophication processes.

Lake Texcoco has evolved from its previous soda-saline condition to a new one characterized by high levels of Na^+ and $Cl^-$from evaporation and the concentration of salts from the precipitation of $CaCO_3$. For its part, Chalco, despite being a lake with a lower density than Texcoco that previously showed high alkalinity, today shows signs of a salinization process with a trend towards increasing Na^+ and Cl^- levels. Our findings are consistent with Martínez-Abarca et al. (2021), who pointed out that Lake Chalco has been reduced to a shallow and subsaline wetland. Both lakes were completely drained off, and the water bodies currently monitored are new. The area previously covered by Lake Texcoco today has systems built for recovering the lake (Soto-Coloballes, 2019), so that its evolution towards conditions with a predominance of Na^+ and Cl^- is the result of its previous draining and precipitation of calcium and sodium salts. Besides being drained, Lake Chalco, has also subsided due to the intensive exploitation of its groundwater. This groundwater extraction has resulted in the consolidation of its aquitard (Ortiz-Zamora and Ortega Guerrero, 2007) and, consequently, the formation of a depression or basin that has given rise to the "new" Lake Chalco, into which treated and untreated wastewater

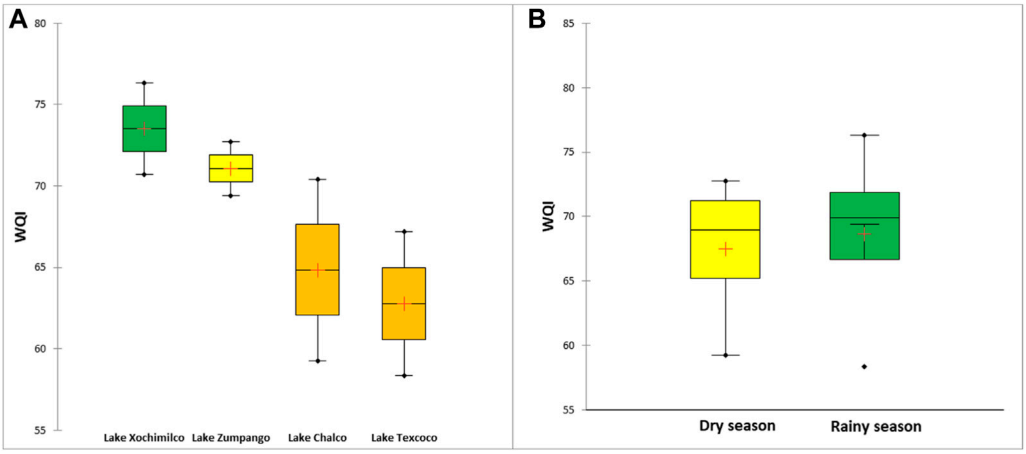

FIGURE 8
Box and whisker plot of **(A)** WQI score by lake in both study periods, and **(B)** WQI score by study period, considering all WQI scores for each lake.

(Ortíz-Zamora and Ortega-Guerrero, 2007) is also discharged, in addition to runoff from the valley itself.

3.4.5 Water quality index

The Water Quality Index scores calculated in this study agree with the physicochemical analyses. Lake Xochimilco had the highest WQI score, with a mean value of 73.5, while Lakes Chalco and Texcoco attained the lowest WQI scores, with mean values of 64.8 and 62.7, respectively (Figure 8A). WQI scores were not significantly different between lakes.

Although the differences in WQI between study periods were not statistically significant ($p > 0.05$), our results show that the mean WQI scores during the dry season are slightly lower than during the rainy season (67.47 vs. 68.61, respectively) (Figure 8B).

Variables such as chlorides, conductivity, and total hardness can bring down the WQI scores, as pointed out by López-López et al. (2019), who recorded low WQI scores in a river with high values of these parameters. Maansi et al. (2022), estimating various water quality indices in Lake Sukhna, in India, found that higher hardness and alkalinity influenced the water quality. In this case, Lakes Chalco and Texcoco showed high values of multiple water parameters such as salinity and concentrations of chlorides, nitrates, sulfates, total nitrogen, and orthophosphates.

The WQI scores make evident that water from the lakes does not have conditions for its use in human supply and its use is limited to other uses such as agriculture and harbor wildlife. In the past, the lakes of the basin of México represented a resource not only for the antique human civilizations, including the first humans in America, but also to populations during the Spanish colonial period, offering several ecosystem services such as water supply, food, water for agriculture, transportation, climate regulation, and maintained high biodiversity.

In addition, given the endorheic nature of the basin, biological richness in the BM was characterized by species considered microendemic to the lakes of the BM, such as the fish *Evarra bustamantei* (Xochimilco carp), *E. tlahuacensis* (Tláhuac carp), *E. eigenmanni* (green carp), and *G. viviparus* (mexclapique). Unfortunately, the three species of the genus Evarra are extinct (IUCN, 2023), and *G. viviparus* (a viviparous fish) is currently listed as threatened (Sedeño-Díaz and López-López, 2009; IUCN, 2023). A species that stands out among the endemic amphibians is *Ambystoma mexicanum*, listed as critically endangered (IUCN, 2023). In all cases, the current conservation status has been associated with human intervention and habitat loss.

Unfortunately, currently, these lakes have received multiple stressors from wastewater pollution and even have faced desiccation resulting from human activities in the basin of one of the largest cities in the world, these conditions evidence improper management limiting the potential ecosystem services of lakes. The current diagnosis shows that the water quality of the lakes in the BM undergoes a eutrophication process more pronounced in saline lakes (Texcoco and Chalco lakes). Furthermore, during the rainy season, a depletion in dissolved oxygen and an increase in nutrient concentration were detected, showing a higher deterioration during this season. In this sense, lakes need urgent attention to diminish the input of pollutants and to carry out appropriate rehabilitation measures for each remnant lake. It is mandatory to conserve the endemic species that still prevail in the basin and promote the rehabilitation of the ecosystem services that these water bodies provide in the past. Furthermore, macroinvertebrates and microalgae species of ancestral human consumption such as *Notonecta unifasciata*, *Krizousacorixa azteca*, *Corisella texcocana*, *Cambarellus montezumae*, *Phormidium tenue*, *Nostoc commune*, and *Chroococcus turgidis*, among others (Ortega, 1972), as well as the cyanobacterium *Spirulina* (Arthorspora) (Grant, 1992) should be recovery.

Future directions are necessary to take into account topics on best practices of wastewater management, rainwater harvesting to conserve aquifers, and aquatic weed management control methods. Likewise, it is important to draw up management programs containing restoration and conservation measures in the declared natural areas and Ramsar sites (Texcoco and Xochimilco), and if necessary, to establish declarations for the conservation of the Zumpango and Chalco lakes. This study establishes the baseline for comparison on the dramatic loss of lake area, which should not be allowed to continue, therefore suggesting increasing surface lake area, and slowing urban growth by setting buffer areas around the remaining lakes.

4 Conclusion

Different environmental and anthropogenic factors have led to the almost total disappearance of the lake system of the Basin of Mexico. The first formal assessment of the lakes was carried out by Alexander von Humboldt and allowed us to infer the state of these lakes in the early nineteenth century and assess the changes that have occurred since then. The former lakes no longer exist; the only water bodies that remain today are shallow wetlands representing only 2.83% of the original lake surface area. These water bodies have received treated wastewater discharges affecting their original water quality and ionic composition, with Lake Xochimilco being the lake with the best water quality. The most critical change is the case of Lake Chalco, which previously was less saline according to Humbodlt´s test results; today, Chalco is almost as saline as Texcoco, although the former is dominated by carbonates while Texcoco is dominated by Na^+ and Cl^-.

The assessment by Humboldt provides valuable information on the state of the BM lakes in the early 1800s and allows for comparing it versus current data to visualize the contrasting conditions of these lakes today. Population growth, urbanism, and the loss of natural land cover have been the leading factors that led to the current deplorable conditions observed in the present study. These water bodies should be the subject of conservation and recovery programs, as they are the habitat for multiple species that are microendemic to the Basin of Mexico, as well as for migratory birds; furthermore, some of these lakes have been declared as UNESCO World Heritage sites. American Public Health Association, 2023.

Data availability statement

The datasets presented in this article are not readily available because The data will be made available strictly for academic purposes upon justification. Requests to access the datasets should be directed to Dra. EL-L, eulopez@ipn.mx.

Author contributions

Conceptualization, analyzed the data, and wrote the article EL-L methodology, EL-L and JS-D; monitoring EL-L, VH, JS-D, MG, and AR-R; data curation EL-L and JS-D, GIS tools, JS-D review and editing EL-L, VH, JS-D, MG, and AR-R, original draft preparation EL-L and JS-D, validation EL-L, VH, JS-D, and MG, funding acquisition EL-L and JS-D. All authors contributed to the article and approved the submitted version.

Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2023.1217343/full#supplementary-material

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The irrigation efficiency trap: rational farm-scale decisions can lead to poor hydrologic outcomes at the basin scale

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Agricultural irrigation practices have changed through time as technology has enabled more efficient conveyance and application. In some agricultural regions, irrigation can contribute to incidental aquifer recharge important for groundwater return flows to streams. The Henrys Fork Snake River, Idaho (United States) overlies a portion of the Eastern Snake Plain Aquifer, where irrigated agriculture has occurred for over a century. Using irrigator interviews, aerial and satellite imagery, and statistical streamflow analysis, we document the impact of farm-scale decisions on basin-scale hydrology. Motivated to improve economic efficiency, irrigators began converting from surface to center-pivot sprinkler irrigation in the 1950s, with rapid adoption of center-pivot sprinklers through 2000. Between 1978–2000 and 2001–2022, annual surface-water diversion decreased by 311 Mm^3 (23%) and annual return flow to the river decreased by 299 Mm^3 over the same period. Some reaches that gained water during 1978–2000 lost water to the aquifer during the later period. We use an interdisciplinary approach to demonstrate how individual farm-scale improvements in irrigation efficiency can cumulatively affect hydrology at the landscape scale and alter groundwater-surface water relationships. Return flows are an important part of basin hydrology in irrigated landscapes and we discuss how managed and incidental aquifer recharge can be implemented to recover return flows to rivers.

KEYWORDS

groundwater-surface water, aquifer recharge, Idaho, Eastern Snake Plain Aquifer, irrigation efficiency, return flow, reach gain

1 Introduction

Improving irrigation efficiency is typically framed as a way to minimize water not put to its intended beneficial use (Burt et al., 1997), water often colloquially characterized as "lost" or "wasted" during conveyance and application (Jensen, 2007; Lankford, 2012). Lining or piping canals and converting to more precise application—in contrast to more traditional techniques, like earthen canals and flood irrigation—are methods touted to increase irrigation efficiency (Richter et al., 2017). Increasing irrigation efficiency is often prescribed in water-limited systems as means of basin-scale water conservation (Contor and Taylor, 2013) and can be attractive to those seeking to reduce stream withdrawals to provide water for environmental objectives or junior water rights-holders (Richter et al., 2017; Owens et al., 2022). Indeed, state, federal, and international programs and policies incentivize increasing irrigation efficiency to conserve water for reallocation to other users (Huffaker, 2008; Levidow et al., 2014; Pérez-Blanco et al., 2021).

But irrigation water lost at the farm-scale to inefficient irrigation practices is retained within basin-scale hydrology. Water delivered in earthen canals or applied in excess of crop uptake infiltrates soils and can recharge aquifers or follow surface and subsurface pathways to return to the river (Venn et al., 2004; Ferencz and Tidwell, 2022). Streamflow diverted for irrigation and recovered in rivers is often referred to as "return flow" and allow water to be used more than once (Jensen, 2007). In fact, in long-irrigated agricultural watersheds, return flows may be a fundamental component of the modern hydrologic cycle (e.g., Kendy and Bredehoeft, 2006; Hu et al., 2017; Oyonarte et al., 2022) and important to junior water users and aquatic ecosystems. Return flows can contribute streamflow during critical low-flow periods (Fernald and Guldan, 2006; Walker et al., 2021; Ferencz and Tidwell, 2022) and provide cool streamflow input (Essaid and Caldwell, 2017; Alger et al., 2021), although return timing is dependent on irrigation application, soil conditions, and local geology (Ochoa et al., 2007; Linstead, 2018). Thus, return flows can bolster the ability to meet environmental flow and temperature objectives in water-limited systems (Lonsdale et al., 2020; Van Kirk et al., 2020) while also supplying water to other users (Owens et al., 2022). In short, return flows are an important part of basin hydrology, but are at risk of decline as policy- and climate-induced water scarcity nudges agricultural regions towards increasing irrigation efficiency (Scott et al., 2014; Pérez-Blanco et al., 2020; Walker et al., 2021).

This sets the stage for an irrigation efficiency trap—where market forces incentivize farmers toward irrigation efficiency improvements that often do not result in the intended basin-scale water conservation—and in fact, may increase water consumption (Grafton et al., 2018; Wheeler et al., 2020). Increased resource consumption due to increased efficiency is described by the Jevons paradox (York and McGee, 2016) and has been well documented in theoretical and modeling studies related to irrigation. Such a change in water consumption is partially due to a difference in scale, where improving irrigation efficiency is perceived differently at the farm scale than the basin scale (Qureshi et al., 2011; Lankford et al., 2020). Irrigators consider increasing irrigation efficiency as a component of improving their individual economic efficiency, i.e., maximizing the difference between production benefits and input costs (Cai et al., 2003; Qureshi et al., 2011). Thus, incentive is strong for irrigators to use their full water allocation by putting more land into production or harvesting an additional or more water-intensive crop (English, 1990; Ward and Pulido-Velazquez, 2008; Xu and Song, 2022)—particularly within water management structures that lack mechanisms for reducing water allocations to a given user to reallocate for other purposes (e.g., doctrine of prior appropriation). Social scientists have documented that some farmers perceive increased irrigation efficiency as a means to maximize revenue, rather than to reduce total on-farm water consumption (Knox et al., 2012; Wheeler et al., 2020; Hamidov et al., 2022). Physical scientists have clearly documented that high irrigation efficiency risks an increase in consumptive water use for a given water allocation (Ward and Pulido-Velazquez, 2008; Scott et al., 2014; Grafton et al., 2018), thus diminishing river return flow (Hu et al., 2017; Linstead, 2018). Yet, the idea to use farm-scale irrigation efficiency for basin-scale water conservation persists (Pérez-Blanco et al., 2021).

Combatting the irrigation efficiency trap requires understanding how humans interact with irrigated landscapes and water resources at multiple scales. Combining irrigator surveys with physical measurements of landscape characteristics, irrigation conversion, streamflow diversion, water availability, and return flows allow for cross-scale examination and integrate the socio-hydrological nature of the problem. Few studies document the irrigation efficiency trap from farm-scale decisions to basin-scale hydrologic outcomes with measured social and physical data (e.g., Wheeler et al., 2020; Anderson, 2022). But irrigation systems are complex social-ecological systems (Lam, 2004) and integrating the hydrologic and social components of irrigation efficiency are important for system understanding and resilience (Fernald et al., 2015; Dunham et al., 2018). To adapt and prepare accordingly, we must examine place-based farm-scale irrigation decisions and how these decisions collectively impact basin-scale hydrology. We can then identify strategies that maintain agricultural and environmental water uses, are robust to climate variability, and are actionable for decision makers (Welsh et al., 2013; Lankford et al., 2020).

We use the Henrys Fork watershed, Snake River, Idaho (United States)—an agricultural watershed that exemplifies those throughout the American West—for place-based research on the relationship between farm-scale decisions and watershed-scale hydrology. Irrigated agriculture has been in place since 1879 (Van Kirk and Griffin, 1997) and contributes to a $10 billion USD regional economy (Idaho Water Resources Board, 2009). The Henrys Fork overlies the headwater portion of the Eastern Snake Plain Aquifer (ESPA; Figure 1), a 28,000 km^2 unconfined aquifer that provides baseflow to the Snake River system (Hipke et al., 2022). In addition to agriculture, the Henrys Fork hosts a recreational fishery worth $50 million USD (Van Kirk, 2021) and is an important component of local watershed management (Joint Committee, 2018). However, studies have modeled a decline in irrigation return flow and groundwater discharge to the river since 1980 (Contor et al., 2004; Sukow, 2021). The reduction of return flow in the Henrys Fork is part of a larger regional hydrologic change, where groundwater pumping, increased irrigation efficiency, and decreased surface-water diversion across southern Idaho has diminished ESPA storage (Stewart-Maddox et al., 2018) and contributions to Snake River streamflow (Olenichak, 1998). Thus, the irrigation efficiency trap is on display in the Henrys Fork and surrounding region.

Therefore, we use a unique interdisciplinary dataset that includes 1) irrigator interviews to understand motivations for irrigation conversion through time, 2) landscape imagery analysis to quantify spatiotemporal irrigation conversion, and 3) hydrologic measurements with statistical analysis from 1978 to 2022 to quantify changes in surface-water diversion, reach gains, and return flows to the river and examine hydrologic change from the farm-to basin-scale. Our research questions are:

1) What motivated farmers to convert to more efficient irrigation application?

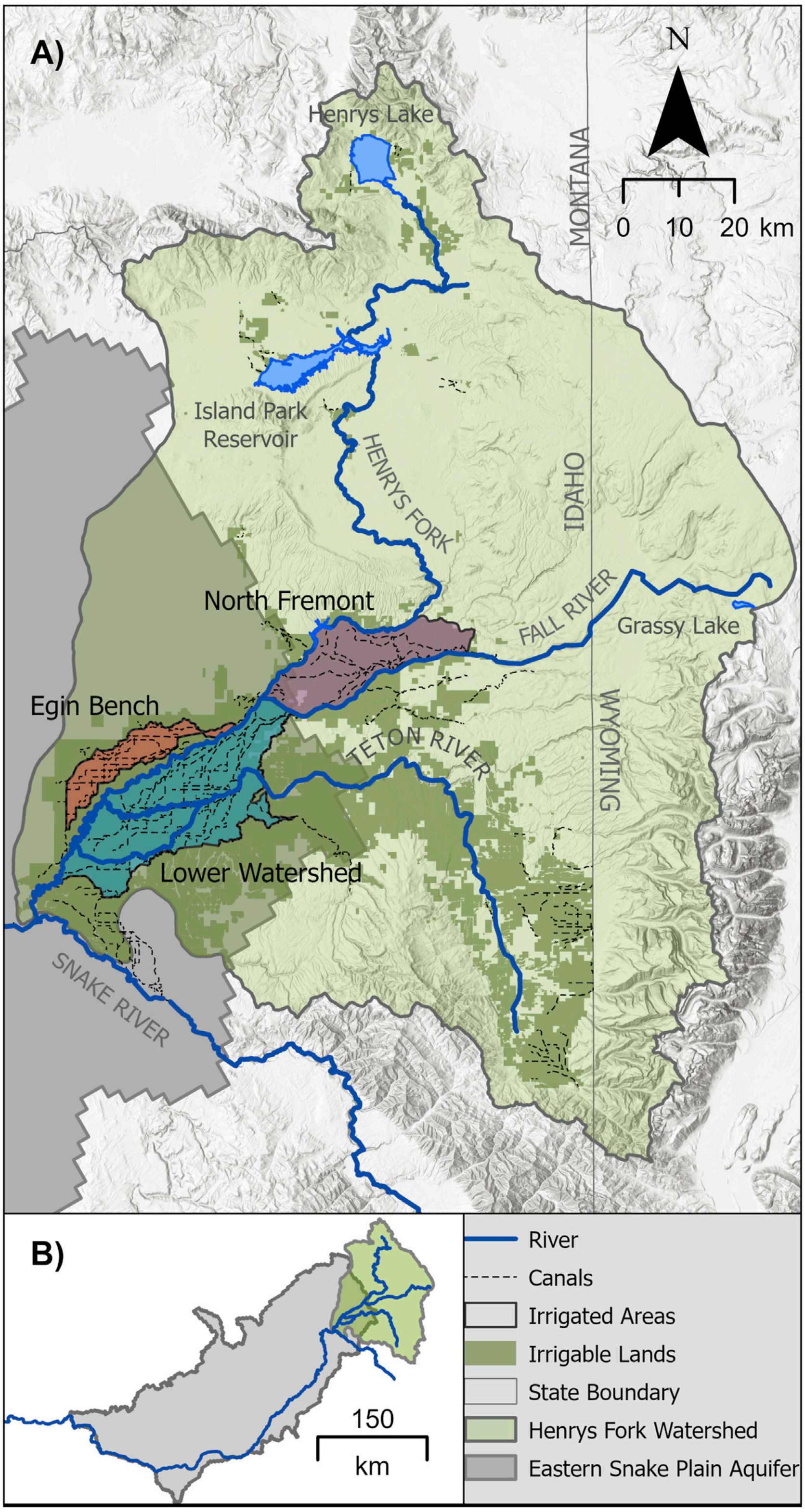

FIGURE 1
The Henrys Fork watershed **(A)** and the watershed relative to the Eastern Snake Plain Aquifer **(B)**. Data sourced from Airbus, U.S. Geological Survey, NGA, NASA, CGIAR, NCEAS, NLS, OS, NMA, Geodatastyrelsen, GSA, GSI, and the GIS User Community.

2) When and at what rate did farmers improve their irrigation efficiency?
3) How did these changes affect basin-scale hydrology?

Our first two questions consider on-farm irrigation efficiency, defined as evapotranspiration divided by the water applied to a field. Our third research question considers project-level irrigation efficiency, defined as water consumptively used by crops (i.e., evapotranspiration) divided by total water withdrawn (Thompson, 1988; Burt et al., 1997; Zalidis et al., 1997). Project-level efficiency accounts for two sources of inefficiency: 1) loss of water in the conveyance system between the point of diversion and the point of field application, and 2) water applied at the field scale that is not consumed by crops. Losses in both components of the irrigation system can be due to evaporation and to seepage into soils and aquifers below the crop root zone.

We use our results to outline the potential for aquifer recharge to maintain and recover return flows.

2 Materials and methods

2.1 Study area

The Henrys Fork watershed is 8,300 km^2 located in the headwaters of the Snake River Basin, Idaho, United States, ranging in elevation from 1,470 m to 3,800 m (Figure 1). Snowmelt and headwater springs provide an average annual unregulated streamflow of 3,140 Mm^3. The surface-water system is managed to provide irrigation to 1,012 km^2 of agricultural land in the low-elevation areas of the watershed, where producers primarily grow potato, alfalfa, and grain crops (U.S. Bureau of Reclamation, 2012b). Surface water is stored in three reservoirs in the watershed (Henrys Lake, 111 Mm^3; Island Park Reservoir, 167 Mm^3; Grassy Lake, 18.8 Mm^3). Teton Dam, on the Teton River, was completed in 1975 to store 247 Mm^3, but the dam failed in 1976 as the reservoir was filling for the first time and was not rebuilt (Reisner, 1993; U.S. Bureau of Reclamation, 2012a).

On average, 1,400 Mm^3 of surface water (45% of average annual unregulated flow) is diverted for agricultural irrigation (U.S. Bureau of Reclamation, 2012b) and is largely delivered by unlined, earthen canals that divert water directly from the Henrys Fork and its tributaries. Irrigators also use groundwater, which accounts for ~25% of the total water withdrawn for irrigation in the watershed. Proportional use of groundwater for irrigation is similar across the ESPA and the state of Idaho as a whole. In 2015, total annual groundwater pumped from the ESPA in the Henrys Fork watershed was ~200 Mm^3 (Lovelace et al., 2020). Although long-term watershed-specific data on groundwater withdrawal are not available, groundwater withdrawal for irrigation in Idaho has been increasing at a rate of ~19 Mm^3 per year, while withdrawal of surface water for irrigation has been decreasing at ~61 Mm^3 per year (see Supplementary Material).

Access to irrigation water is subject to water-rights priority based on the prior appropriation doctrine (Van Kirk et al., 2019) and largely organized under one irrigation district and ~30 canal companies (Van Kirk and Griffin, 1997). Under the prior appropriation doctrine in the western United States, state governments allocate surface water based on the date water was first diverted and put to "beneficial use" as defined by the state (Van Kirk et al., 2019). Irrigation districts and canal companies are local entities responsible for managing conveyance systems for water delivery to individual irrigators who are shareholders within the organization (Armstrong and Jackson-Smith, 2017). In the Henrys Fork, surface water users have rights senior to those of groundwater users and water resources are conjunctively managed (Stewart-Maddox et al., 2018). The basin is fully adjudicated, and surface water rights include allowance for reasonable conveyance loss (Vonde, 2016).

Irrigated land in the Henrys Fork watershed is separated into four regions: North Fremont, Egin Bench, Lower Watershed, and Teton Valley (Table 1). These four primary irrigated regions account for >95% of surface-water diversion in the watershed and >95% of the current and historic canal conveyance system (Joint Committee, 2018); all other irrigated acreage is primarily groundwater-irrigated. Regarding water rights, North Fremont has predominantly junior water rights and experiences significant water shortages annually (U.S. Bureau of Reclamation and Idaho Water Resource Board, 2015). Egin Bench has predominantly senior water rights, surplus water in average water years, and meets its demand even in successive drought years. The Lower Watershed meets most of its irrigation demand in average water years, but experiences a deficit in drought years that follow a drought year (U.S. Bureau of Reclamation and Idaho Water Resource Board, 2015). Essentially all conveyance in the Lower Watershed and Egin Bench is delivered through the 19th-century earthen canal system. Most conveyance in North Fremont has been converted to pipelines, beginning with small canals in the 1970s. We exclude Teton Valley from our analysis because the irrigated region does not interact with the ESPA, but rather a smaller, hydraulicly distinct aquifer (Bayrd, 2006). For all irrigation regions studied, we can assume a constant value for total irrigable area as no new irrigation rights have been granted in decades, particularly since the groundwater moratorium in the 1990s (Van Kirk et al., 2019). Thus, no new land has been put into agricultural production.

Our study considers two irrigation efficiency scales: on-farm and project. At the farm scale, efficiency is related to mode of irrigation application. Four modes of irrigation application are currently used in the watershed: flood irrigation and sprinkler irrigation via hand-line, wheel-line, and center-pivot (Table 2). In the Henrys Fork watershed, the estimated 1980–2010 average for on-farm irrigation efficiency (evapotranspiration divided by water applied) was 60% for North Fremont and 55% for each of the Egin Bench and Lower Watershed (U.S. Bureau of Reclamation, 2012b). Project-scale efficiency for the entire Henrys Fork watershed from 1979 to 2008 was 26% (U.S. Bureau of Reclamation, 2012b). Project-scale irrigation efficiency is water consumptively used by crops (i.e., evapotranspiration) divided by total water withdrawn and includes loss within canal conveyance.

Each irrigated region differs in terms of its gradient and soil type, important factors for irrigation application. Flood irrigation requires flatter terrain (0.5%–4% gradient), whereas wheel-line and center-pivot sprinklers are appropriate for steeper slopes ≤15% and hand-line sprinklers can handle slopes ≤20% (Brown, 2008; Barnhill et al., 2009). Egin Bench and the Lower Watershed have predominantly flat terrain (≤0.5% slope), whereas the North Fremont region is

TABLE 1 Characteristics of irrigated study regions within the Henrys Fork watershed by irrigation year (November–October). The standard deviation for mean annual precipitation and ET are reported parenthetically. We report data for two periods of time, 1978–2000 and 2001–2022. The year division for these time periods was determined through analysis in this paper. Diversion data are from Idaho Water District 01. Average annual precipitation and evapotranspiration were calculated from gridMET for alfalfa reference within each irrigated study region (Abatzoglou, 2013). The gridMET period of record begins in 1980 and has 4 km resolution. We assume a constant value for total irrigable land.

Study region	Irrigated land (km^2)	Irrigation year	Diversion (Mm^3)	Irrigation year	Precipitation (mm)	Alfalfa reference ET (mm)
North Fremont	131.5	1978–2000	109.6	1981–2000	475 (117)	1,335 (116)
		2001–2022	83.4	2001–2022	437 (84)	1,352 (66)
Egin Bench	123.4	1978–2000	495.7	1981–2000	349 (90)	1,396 (124)
		2001–2022	367.9	2001–2022	318 (69)	1,415 (70)
Lower Watershed	295.4	1978–2000	749.7	1981–2000	349 (88)	1,427 (130)
		2001–2022	583.7	2001–2022	321 (69)	1,443 (74)

steeper with greater heterogeneity (0%–20% slope; Supplementary Figure S2). Regarding soil, Egin Bench is almost exclusively loamy fine sand, noted for its high infiltration and low runoff rates (Supplementary Figure S2). North Fremont has soils that range from moderate infiltration and runoff to soils that are near-impervious with high runoff potential. Hydrologic soil groups in the Lower Watershed are heterogeneous (Supplementary Figure S2).

2.2 Irrigator interviews

We conducted 20 semi-structured phone interviews in July 2022 to 1) identify sociological, economic, and geographic factors that prompt farmers to convert to more efficient irrigation in the Henrys Fork watershed and 2) extend temporal flood-to-sprinkler conversion data beyond the period aerial and satellite imagery were available. Staff at the Henry's Fork Foundation, a local watershed conservation organization and sponsor of this research, developed a key informants list for initial contact; additional participants were identified using the snowball method (Hay, 2005). We interviewed current and former agricultural irrigators with a variety of farm acreage, irrigation district and canal company representatives, and second- or third-generation irrigators with knowledge of historic family operations related to surface-water irrigation. Our study area is rural, with a population of ~28,500 (United States Census Bureau, 2022a; United States Census Bureau, 2022b; United States Census Bureau, 2022c). Most farms in our study area are family-owned and operated. Eighty percent of farm operations in the study area are <500 acres, 10% are 500–999 acres, and the remaining 10% are ≥1,000 acres (USDA National Agricultural Statistics Service, 2017a; USDA National Agricultural Statistics Service, 2017b). It is likely our sample was biased towards individuals who are highly active in and knowledgeable about local and regional water management. Participation rate may have been negatively impacted by conducting interviews during the irrigation season when irrigators have limited capacity, drought limiting water rights allocation and contributing to high tension around water conversations, and perceptions of the Henry's Fork Foundation and its intent in conducting this research.

Interview data were collected in field notes and summarized in analytical memos (Hay, 2005)—a reflexive activity where researchers explore topics in a narrative structure (Birks et al., 2008). We used these analytical memos for inductive coding and thematic analysis (Attride-Stirling, 2001; Saldana, 2016). See the Supplementary Material for interview instrument.

2.3 Geospatial analysis

We used aerial photography and Landsat satellite imagery from 1986 to 2020 to evaluate spatiotemporal trends in irrigation practices (Supplementary Table S2). From satellite imagery, it was difficult to differentiate fields that were flood irrigated versus those that were irrigated via hand- or wheel-line sprinkler. Thus, we visually assigned irrigation type as pivot vs. not-pivot in June or July for each field using imagery from 1988 to 2002 (every 2 years) and 2005–2020 (every 5 years). We assigned pivots to circular fields and quantified pivot acres, assigning full pivot circles 0.63 km^2, three-quarter circles 0.47 km^2, and half pivot circles 0.32 km^2.

To verify the presence and extent of flood irrigated land currently in production, we identified eighteen fields in the Lower Watershed and two fields on the Egin Bench that appeared to be flood irrigated in Google Earth imagery from September 2015 and June 2017. We traveled to these sites in July 2021 to verify irrigation type.

2.4 Hydrologic analysis

We used statistical model selection and multi-model inference with Akaike's Information Criterion (AIC) to analyze annual time series data for five key measures of water supply and use: 1) surface-water irrigation diversion, 2) river reach gain, 3) unregulated streamflow, 4) total diversion minus reach gain (net watershed withdrawal), and 5) total watershed inflow minus watershed outflow (net watershed export). We conducted our analysis at

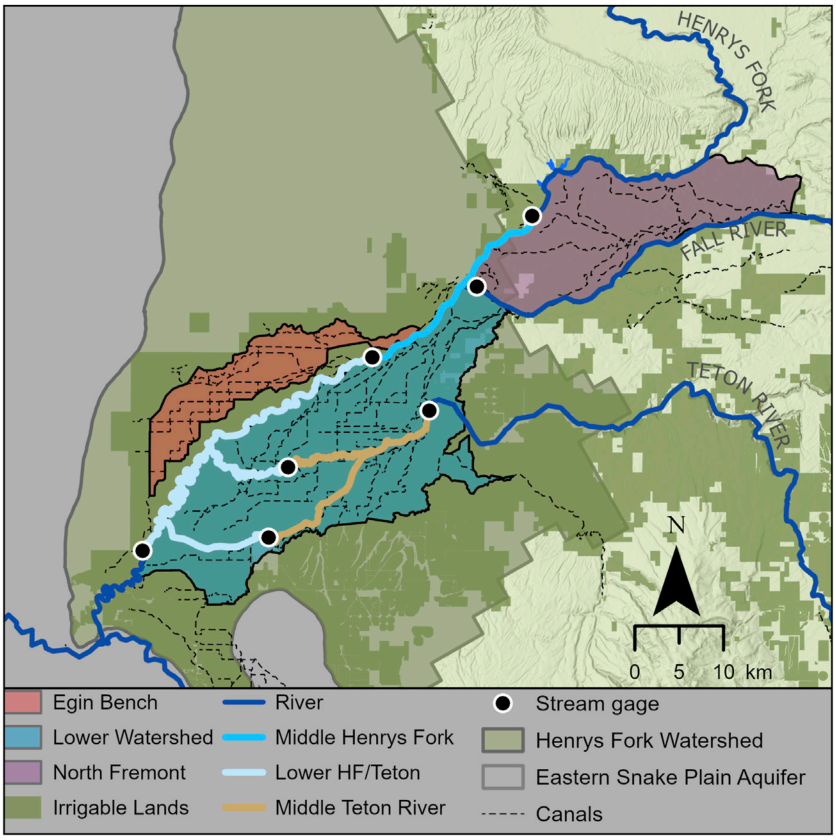

FIGURE 2
U.S. Geological Survey stream gages used in the water balance and reach gain calculations.

two spatial scales—watershed and subreach. We conducted the watershed-scale analysis for irrigation years 1978–2022, where the irrigation year is defined as November 1 through October 31. The 1978–2022 period is the longest over which complete daily data are available. Some sub-reach analysis was done for irrigation years 2004–2022, the longest period over which streamflow data were available for the sub-reaches.

2.4.1 Data compilation and computation

The primary hydrologic data used in the analysis were daily streamflow from U.S. Geological Survey (USGS) monitoring stations, surface-water diversion and exchange well injection reported by Idaho Water District 01 (the basin-wide water administration agency), reservoir volume from the U.S. Bureau of Reclamation, and precipitation and evapotranspiration data from U.S. Bureau of Reclamation and Natural Resources Conservation Service. Exchange wells inject groundwater directly into the Teton River (Olenichak, 2020). The exchange wells are operated only during very dry years, as are other exchange wells in the watershed, which inject water into the Henrys Fork (U.S. Bureau of Reclamation and Idaho Water Resource Board, 2015). Of the five key measures assessed, all but surface-water diversion required computation (detailed below).

We estimated reach gain on reaches of the Henrys Fork and Teton River that interact with the ESPA (Figure 2). These reaches do not gain appreciable water from tributary streams and do not contain storage reservoirs. Hence the net gain from a combination of surface-irrigation return flow and groundwater input into these reaches can be calculated as:

$$\begin{aligned} reach\ gain = reach\ outflow - reach\ inflow + diversions \\ - exchange\ well\ injection \end{aligned} \tag{1}$$

Negative reach gains indicate a reach loss.

Unregulated streamflow for the three sub-watersheds was calculated for upper Henrys Fork, Fall River, and Teton River as:

$$\begin{aligned} flow_{unregulated} = flow_{regulated} + diversions + \Delta storage_{reservoir} \\ + evaporation_{reservoir} - exchange\ well\ injection \end{aligned} \tag{2}$$

Regulated streamflow data for Equation 2 used three long-term USGS stream gaging stations downstream of all source tributaries and immediately upstream of interactions with the ESPA (Supplementary Figure S3 and Supplementary Table S3). The reservoir evaporation term in Equation 2 is the net difference between evaporation and precipitation on reservoir surfaces. If positive, this represents a loss via evaporation, and if negative represents a gain via direct precipitation in reservoirs. Eqs 1, 2 largely coincide with those used by Water District 01 to administer water rights in the watershed (Olenichak, 2020). Total watershed unregulated flow is the sum of unregulated flow in the three sub-watersheds.

For the watershed-scale water balance (total inflow minus outflow; net basin export), we included all sources of inflow available for surface-water diversion, which is given by:

$$\begin{aligned} watershed\,inflow = {} & watershed\,unregulated\,flow \\ & - \Delta storage_{reservoir} + exchange\,well\,injection \\ & - evaporation_{reservoir} \end{aligned} \tag{3}$$

Note: We define net basin export as the sum of consumptive use and water that exits the basin as groundwater flow to the ESPA.

Annual watershed outflow is regulated streamflow at the downstream-most gage on the Henrys Fork near the bottom of the watershed at the confluence with the main Snake River (Figure 2). Eq. 1 can be rearranged to yield:

$$\begin{aligned} diversion - reach\,gain = {} & reach\,inflow - reach\,outflow \\ & + exchange\,well\,injection \end{aligned} \tag{4}$$

At the watershed scale, Equations 1–3 can be used to obtain an alternate derivation of Equation 4 showing that net withdrawal of water from the watershed can be calculated either as the difference between diversion and unregulated flow or as the difference between total watershed inflow and watershed outflow. We analyze both to demonstrate this equivalence and better interpret the role of reach gains in the watershed-scale water balance.

2.4.2 Statistical modeling

We used an AIC-based approach to statistically model each of our five key hydrologic measures through the 1978–2022 study period and quantify changes through time. The basic AIC method is to propose a set of candidate models, rank them according to AIC, and then use a measure of relative evidence for the models in the candidate set to calculate a final model that is a weighted average of all models in the set (Burnham and Anderson, 2002; Anderson, 2008; Claeskens and Hjort, 2008). We used a modification of AIC known as AICc (AIC with small-sample correction), which includes an additional term that increases the overfitting penalty when the number of fitted parameters becomes large relative to the sample size.

All of the data analyzed here occur in a time series of 45 annual values, and all models were fit in the framework of autoregressive time series models using the `arima` function in the R programming environment (R Core Team, 2022). We proposed five types of structural models describing potential temporal trends in the data:

1. Null model: data described by a single mean (one structural parameter).
2. Piecewise constant: data described by two means, one for each of two distinct time periods (two structural parameters describing the means plus a third defining the time period breakpoint).
3. Linear trend (two structural parameters).
4. Piecewise trend: data described by linear trend over the first time period and constant mean over the second (three structural parameters plus a fourth defining the time period breakpoint).
5. Quadratic (three structural parameters).

The breakpoints in models 2 and 4 were not specified *a priori* but were determined through the maximum-likelihood model-fitting process. However, to avoid the possibility of a few extreme water years at the beginning or end of the time series artificially introducing a breakpoint near the endpoints of the study period, we restricted the range of breakpoints to 1991–2009. This ensured that each of the two time periods was at least 13 years long.

For each of the above, we proposed two sub-models, one in which unregulated flow was used as a covariate (one additional parameter) and another without the covariate. We included unregulated flow as a covariate because diversion in prior appropriation systems is generally greater in years of greater water supply. Incorporation of water supply as a covariate removes the confounding effect of short-term variability in water supply on actual long-term trends. For each of the models described so far, we proposed one each with and without first-order serial autocorrelation (one additional parameter). Finally, we fit one set of models to normally distributed residuals and another with lognormally distributed residuals, the latter achieved by log-transforming the response variable. Because reach gains could be negative and were on the order of 125 Mm^3, we used the transformation log $(y + 125)$ for reach gain data. Given five structural models and two choices for each of the other components, this gave a maximum of 40 possible models. However, for most of the response variables we tested, lognormal models accounted for most of the model weight, so we ended up eliminating the normal models. After removing redundant models, all final AICc results were based on 10 or fewer models. Where the AIC analysis indicated strong evidence for two distinct time periods, we compared observed means between the two periods.

Lastly, we calculated Pearson correlations (r) among diversion, reach gain, and unregulated streamflow at watershed and sub-reach scales. For each sub-reach, diversion was defined as that over all irrigated regions upstream of the reach, and unregulated streamflow was defined as that available to meet natural-streamflow water rights in that reach. We assigned $0 \le |r| < 0.5$ as weak, $0.5 \le |r| < 0.7$ as moderate, and $|r| \ge 0.7$ as strong (Chan, 2003).

3 Results

3.1 Irrigator interviews by irrigation region

From the twenty irrigator interviews, some had experience across irrigation study regions and could describe practices across the watershed. Thus, we received a total of 24 responses: 9 from North Fremont, 6 from Egin, and 9 from the Lower Watershed. Nineteen irrigators reported experience with either flood-to-sprinkler conversion or increasing sprinkler mechanization (i.e., converting from hand- or wheel-line to center pivot irrigation). Five irrigators continue to flood irrigate to a degree and mostly in the Lower Watershed. We recognize small sample size can carry bias, particularly with our non-random interviewee selection. However, we prioritized representation within each irrigated area given limited resources and previous work identifying each area as different in their irrigation practices, due to differences in physical geography and water rights priority (U.S. Bureau of Reclamation and Idaho Water Resource Board, 2015).

Across the study regions, economic efficiency and physical geography were primary motivators for converting irrigation practices. Responses about economic efficiency centered on water and labor, separately. Irrigators with flood irrigation experience noted how pivot irrigation reduced water lost to seepage and evaporation. Other irrigators noted that hand- and wheel-line sprinklers are subject to water loss through wind, sometimes double-watering crops while leaving others dry. With the water savings earned through increased irrigation efficiency, irrigators noted their ability to harvest an additional crop during the growing season—producing higher crop yields and crops of better quality. Conversion to pivot irrigation also significantly reduced the labor required to successfully irrigate via flood, hand-line, or wheel-line, improving economic efficiency.

Responses about physical geography noted how irrigation conversion better accommodated for land slope and soil profiles. Some regions are not conducive to flood irrigation. For North Fremont irrigators, steeper terrain prevented flood irrigation success and motivated increased sprinkler mechanization in the 1950s and 1960s as technology became available. In the Lower Watershed, irrigators with land impacted by the 1976 Teton Dam Failure noted that sediment deposition altered land slope and reduced flood irrigation efficiency, thus motivating their conversion to sprinkler irrigation. Irrigators on the Egin Bench coalesced around one story: the region has sandier soils (Supplementary Figure S2) and historically used subirrigation—subsurface application that raises the water table to crop roots (Bjorneberg and Sojka, 2005)—until a single irrigator converted to sprinkler application in the late 1970s/early 1980s, thus lowering the local water table and making subirrigation untenable. This initiated a conversion to sprinkler irrigation on the Egin Bench, where initial adopters converted to sprinkler application due to the physical limitations of subirrigation and secondary adopters converted to sprinklers to participate in the increased yield experienced by their neighbors. We do not know why one irrigator in Egin Bench first converted from subirrigation to sprinkler.

Topics related to environmental stewardship were evoked as justification for both converting and not converting to more efficient irrigation. Irrigators who converted to sprinkler application noted its benefit for minimizing soil erosion and improving soil health, oftentimes pairing these benefits with mention of higher yield and crop quality. Irrigators who continue to flood irrigate drew attention to its benefits for wildlife, aquifer recharge, and maintenance of groundwater springs.

Respondents noted cost, water right seniority, and land composition as factors limiting their ability to convert to more mechanized application and/or center-pivot sprinklers. Irrigators identified the high upfront cost of center-pivot sprinklers as the primary barrier to conversion, with the applications for federal cost-sharing programs to purchase equipment described as "a pain in the ass" by one interviewee. Irrigators also highlighted that those with senior water rights lack incentive to convert to more efficient sprinkler application, as they are less likely to face curtailment. Irrigators with rocky and vegetated land noted center-pivot installation is infeasible.

In terms of conversion through time, interviewees in the North Fremont region converted from flood to sprinkler irrigation prior to the 1970s. Irrigators from the Egin Bench and Lower Watershed lagged in their flood-to-sprinkler conversion by at least a decade, with conversion beginning largely in the 1970s. Conversion to sprinkler on the Egin Bench was completed by 2000, whereas respondents in the Lower Watershed reported converting their flood operations through to 2010. Increased sprinkler mechanization continued through the 2000s in all regions. However, Egin Bench mechanized prior to the 1990s while North Fremont and the Lower Watershed mostly increased their sprinkler mechanization prior to the 2000s.

3.2 Geospatial analysis by irrigation region

Overall, center-pivot sprinkler irrigation increased between 1988 and 2020. On the Egin Bench, total acres irrigated by pivots increased rapidly between 1988 and 2000—from 22.1% to 73.1% (Figure 3B). This rate of pivot expansion slowed after 2000, with 87.2% of irrigated acres using center-pivot sprinklers by 2020 (Figure 3B). The rate of conversion on the Egin Bench, where water users have senior water rights of the three study regions, did not align with commentary in irrigator interviews about senior water rights holders lacking incentive to convert to more efficient irrigation application. However, slowed expansion after 2000 aligns with irrigator interviews, where none of our interviewees on the Egin Bench reported conversion after 2000. In contrast, the rate of conversion from non-pivot irrigation to center-pivot sprinklers has been consistent through time in the Lower Watershed. Between 1988 and 2020, the percentage of irrigated acres with center-pivot sprinklers increased from 5.9% to 47.0%—an average annual rate of 1.3% (Figure 3). This result also aligns with irrigator interviews, particularly given some irrigators in the Lower Watershed continue to flood irrigate. Flood irrigation has been negligible in North Fremont since sprinkler irrigation became available because of the steeper terrain. The rate of center-pivot installation in North Fremont paralleled that of the Lower Watershed and, as of 2020, 36.7% of North Fremont was irrigated with center-pivot sprinklers. However, much of the land with irrigation rights cannot be irrigated due to its gradient, rocky substrate, and wetlands. Therefore, we estimate center-pivot sprinklers are used on ~80% of the total land area that is regularly irrigated from year to year.

Lastly, ground-truthing 2015 and 2017 satellite imagery confirmed the presence of flood irrigation as of July 2021. Of the twenty fields observed, fifteen were flood irrigated and five were irrigated by wheel-line sprinklers. Of the fifteen flood irrigated parcels, thirteen were growing barley, hay or alfalfa and two were pasture fields. This exercise confirmed that aerial imagery could not be used to distinguish wheel-line sprinkler irrigation from flood irrigation, as both have rectangular irrigation patterns.

3.3 Watershed-scale statistical analysis

The AICc analysis provided strong evidence for a steady decline in diversion from the late 1970s until 2000, followed by a sharp drop to a much lower, but constant level of diversion from 2001 to 2022 (Figure 3). Six models accounted for 99.5% of the AICc weight, and

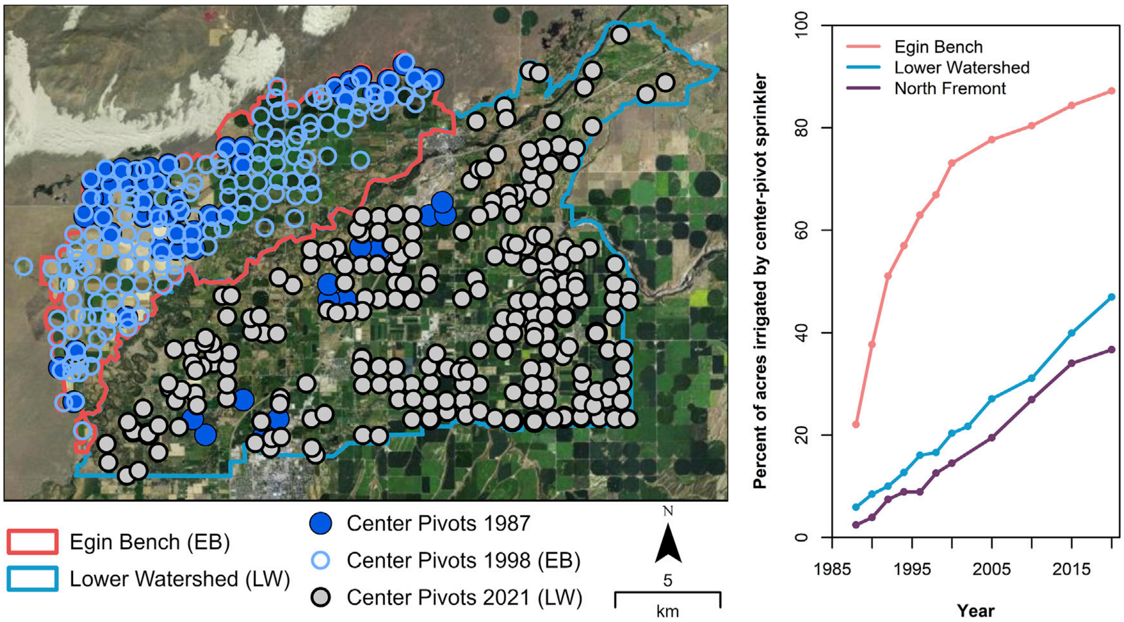

FIGURE 3
Panel A is change in pivot-irrigated acres for Egin Bench (1987–1998) and the Lower Watershed (1987–2021) (Imagery is from USDA FSA NAIP, July 2019). Panel B is percentage of acres irrigated with pivots for all three irrigation study areas for 1988–2020.

TABLE 2 Irrigation type definitions adapted from Bjorneberg and Sojka (2005) and Lonsdale et al. (2020) and irrigation type application efficiencies with appropriate citations. Application efficiency is defined as the fraction of average irrigation water applied that meets a target irrigation depth for an irrigation event (Burt et al., 1997).

Irrigation type	Definition	Application efficiency
Flood	Water spread across a field via furrows and ditches	30%–60% (Neibling, 1997)
Hand-line sprinkler	Segments of aluminum pipe laid on the ground and connected to create an irrigation line up to 400 m in length. Each segment has 1–2 mounted sprinklers and the irrigation line must be manually moved across a field	70%–80% (Trimmer and Hansen, 1994)
Wheel-line sprinkler	Elevates irrigation line above the ground with a 1.5–3 m diameter wheel and rolls along a field via engine power	70%–80% (Trimmer and Hansen, 1994)
Center-pivot sprinkler	Approx. 400 m of sprinkler pipe rotates around a pivot. The pipe is elevated 2–4 m above the ground with wheeled towers and tubes with low-pressure nozzles hang on the pipe 1–3 m above the soil	85%–95% (King and Kincaid, 1997; Brown, 2008)

all six included terms quantifying the continuous decline from 1978 to 2000 (Supplementary Table S4). Four of those, accounting for 87.9% of the AICc weight, identified the step-wise drop between 2000 and 2001. Watershed-total unregulated streamflow appeared as a covariate in the top four models, accounting for 98.7% of the model weight. Annual watershed-total diversion dropped from a mean of 1,374 Mm^3 in the 1978–2000 period to 1,063 Mm^3 in 2001–2022, a decrease of 311 Mm^3 (23%). The pattern and relative magnitude of decrease in diversion was uniform across all irrigated areas (Table 2; Supplementary Figure S4). Within the irrigation year, diversion was similar between the two time periods early and late in the irrigation season—April/May and October—but greater in the 1978–2000 period during June–September and during the winter. Winter diversion is allowed under water rights for stock water and other non-irrigation uses.

Evidence was equally strong that watershed-total reach gain has declined. Eight models accounted for 99.5% of the model weight, and all eight included terms modeling a decrease from 1978 until the early 2000s (Figure 4; Supplementary Table S5). Watershed-total unregulated streamflow appeared as a covariate in four of these models, accounting for 94.3% of model weight. Models containing a step-wise drop in the early 2000s accounted for 98.3% of model weight, but the location of the step differed across models. The top two models (93.1% of model weight) identified the step-wise drop as occurring between irrigation years 2002 and 2003; three other models (5.2% of weight) fit the step-wise drop between 1999 and 2000 or 2000 and 2001. The averaged model thus shows that the decline in reach gains lags that of diversion and is slightly more gradual (Figure 4). Using the 1978–2000 vs. 2001–2022 time division identified by the diversion trends, reach gain dropped from an annual mean of 322 Mm^3 in the 1978–2000 period to 23.1 Mm^3 in

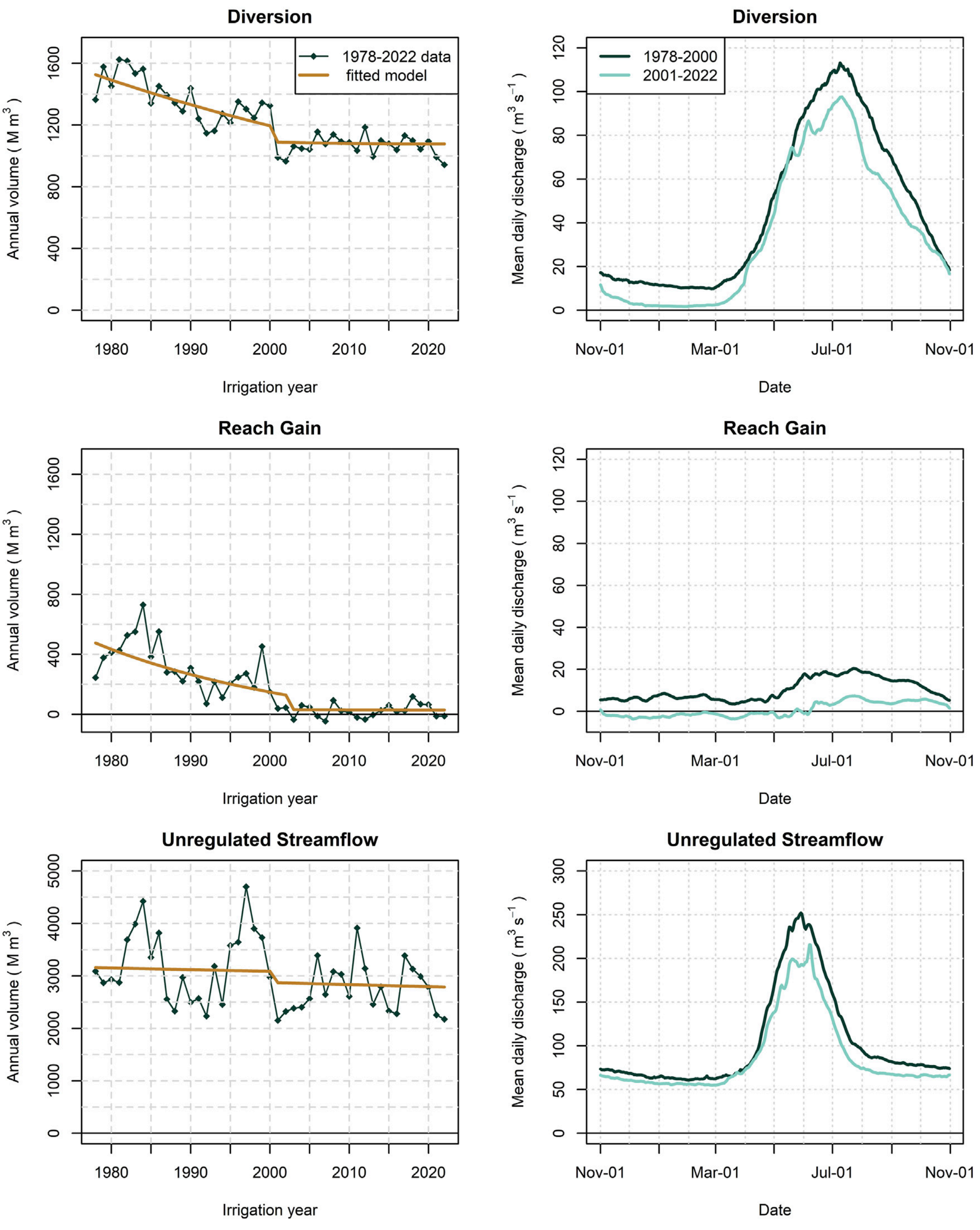

FIGURE 4
Trends in Henrys Fork watershed total diversion, reach gains, and unregulated streamflow for irrigation years 1978–2022.

2001–2022, a decrease of 299 Mm^3. We cannot calculate percent decrease in reach gains because reach gains can sometimes be zero or negative. Watershed-total reach gain was negative in 8 years in the recent period, whereas gain was positive in each year prior to 2001. Mid-summer reduction in reach gain between the two time periods averaged ~11 m^3/s.

Even though unregulated streamflow was a strong and positive covariate in all models of diversion and reach gain through time, on its own, it showed only a very modest decrease since 1978 (Figure 4). Six models accounted for 99.4% of the model weight, and the top model (34.2% of model weight) included only a constant term and first-order autocorrelation (Supplementary Table S6). Three of the models (37.2% of weight) identified a step-wise decline, and in all three, the step occurred between 2000 and 2001. Annual unregulated streamflow averaged 3,234 Mm^3 in the 1978–2000 period and 2,738 Mm^3 in the later time period, a decline of 496 Mm^3

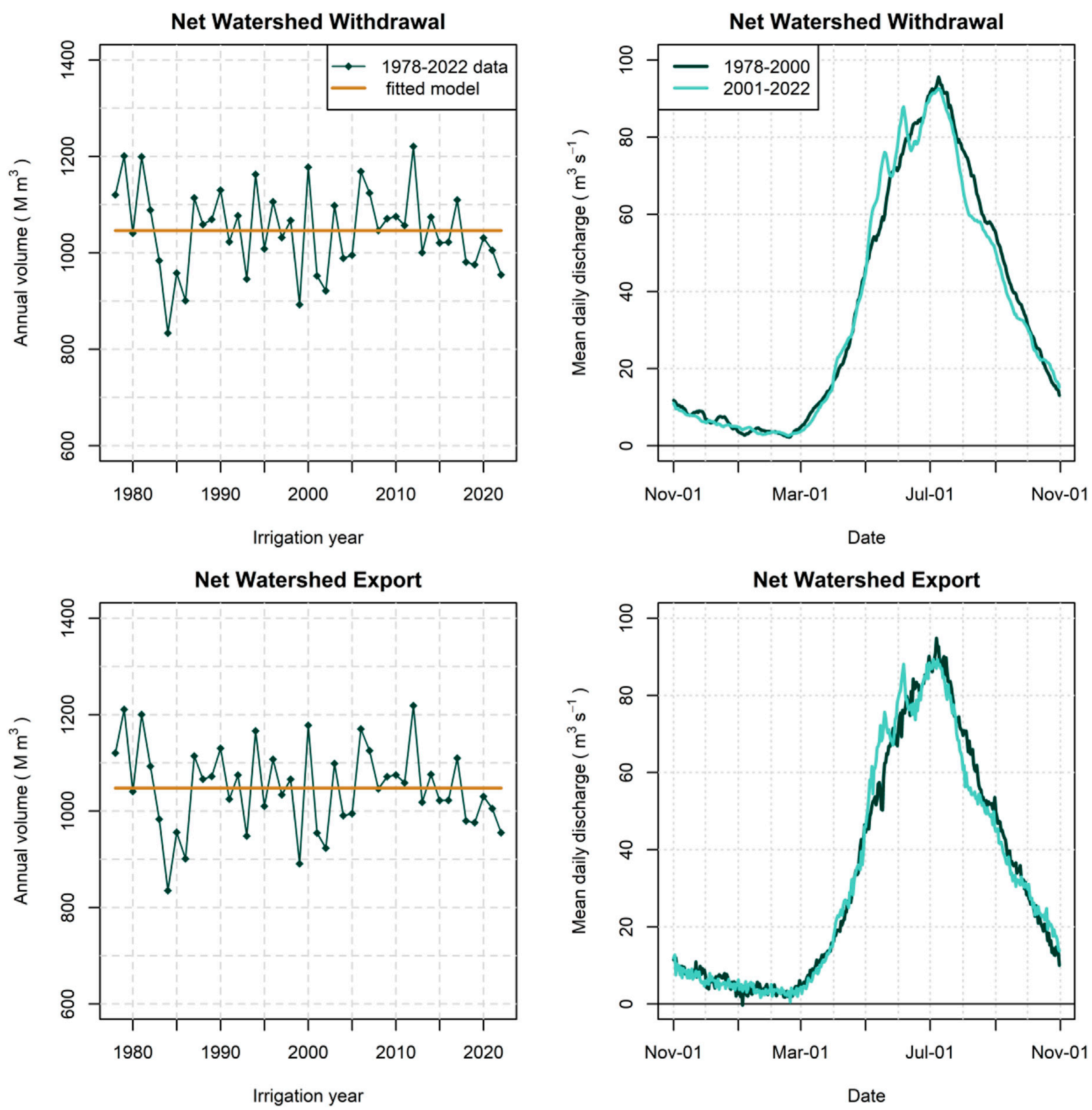

FIGURE 5
Net watershed withdrawal and export in the Henrys Fork watershed for irrigation years 1978–2022.

(15.3%). Unregulated flow was nearly constant during the early period but has decreased at a rate of 3.9 Mm^3 per year since 2001, for a total reduction of 82.1 Mm^3 (2.9%) in the last 20 years.

Net watershed withdrawal—the difference between watershed-total diversion and reach gain—showed no evidence of change since 1978. The top two models accounted for ~100% of model weight, and both were models of a constant over the entire study period (Figure 5; Supplementary Table S7). As expected from the mathematical definitions, net watershed export—the difference between total watershed inflow and outflow—was equivalent to net withdrawal, excluding differences from reservoir evaporation/precipitation, which is highly variable at the daily scale. Net watershed withdrawal averaged 1,052 Mm^3 in 1978–2000 and 1,041 Mm^3 in 2001–2022, a 1% decline. Over the entire study period, the net annual withdrawal of water from the watershed, measured either as diversion minus gain or inflow minus outflow, averaged 1,046 Mm^3 with an interannual coefficient of variation of 8.3%. Despite much higher winter and mid-summer diversion in the 1978–2000 period (Figure 4), net basin export showed little difference between the two time periods across the irrigation year (Figure 5).

Pearson correlations among the three primary response variables were strong only between reach gain and diversion and then only at the watershed scale and only over the entire study period (Table 3). Correlations between diversion and reach gain were weak otherwise. Correlations between diversion and unregulated flow were positive and moderate for all reaches and time periods except the watershed total over 1978–2022. Reach gain

TABLE 3 Correlation coefficients between diversion, unregulated flow, and reach gains within a given subreach or spatial extent (ex. Comparing diversion upstream of the middle Henrys Fork to unregulated flow into that node). Cell shading uses light to dark to signify weak to strong correlations. Correlations were computed based on data availability; subreach data for the Teton River were limited to 2004–2022.

Subreach	Irrigation years	Diversion vs. Unregulated flow	Reach gain vs. Unregulated flow	Reach gain vs. Diversion
Watershed Total	1978–2022	0.49	0.55	0.90
Watershed Total	2004–2022	0.57	−0.01	0.14
Middle Henrys Fork	1978–2022	0.54	0.36	0.33
Middle Henrys Fork	2004–2022	0.63	−0.03	−0.20
Teton River	2004–2022	0.64	0.15	−0.08
Lower Henrys Fork/ Teton	2004–2022	0.57	−0.05	0.22

and unregulated flow showed little correlation, other than a correlation of 0.55 for the watershed total over 1978–2022. Thus, reach gains were largely independent of unregulated streamflow whereas diversions were generally higher in wet years.

4 Discussion

On-farm irrigation efficiency in the Henrys Fork watershed has increased over the last 70 years. Local irrigators began converting flood irrigation to more mechanized sprinkler application in the 1950s in North Fremont and in the 1970s in the Egin Bench and Lower Watershed to improve their economic efficiency and accommodate for land composition. As of 2020, 87% of the Egin Bench, 47% of the Lower Watershed, and ~80% of North Fremont used center-pivot sprinkler application. Those changes to irrigation efficiency have altered Henrys Fork hydrology. Between 1978 and 2000, surface-water diversion and reach gains both decreased substantially and by about the same volume—311 Mm^3 and 299 Mm^3—then stayed relatively constant from 2001 to 2022. Hydrologic changes have been largest in the lower Henrys Fork/ Teton River—most likely in response to rapid changes in irrigation practices on the Egin Bench through 2000. Although reach gains declined through the period of record, stream gage data show that net watershed export—the sum of consumptive use and water that exits the basin as groundwater flow to the ESPA—has not changed, despite a 3% decrease in unregulated streamflow during 2001–2022 from extended drought in the West (Williams et al., 2020). This result, in combination with interpretation of additional regional studies, indicate consumptive use has increased with irrigation efficiency in the Henrys Fork watershed. Furthermore, our data show that prior to 2001, reach gains in our system were equivalent to irrigation return flows, i.e., water diverted from the river in excess of what could be consumed by crops or recharged to the regional aquifer.

4.1 Irrigation conversion: Comparing the Henrys Fork watershed with other regions

Farm-scale decisions in irrigation application have changed the irrigated landscape within the Henrys Fork watershed. The timing and rate of sprinkler adoption on the Egin Bench aligns with previous work in the watershed documenting conversion to mostly center-pivot sprinkler irrigation by the mid-1990s (Contor, 2004). The conversion of 61% of total irrigable land to center-pivot irrigation in the Egin Bench and Lower Watershed combined also aligns with irrigation conversion to more precise application elsewhere in the United States (Maupin et al., 2014). Irrigator motivations and inhibitors toward adopting more efficient irrigation application in the Henrys Fork are similar to those of irrigators elsewhere in the United States and globally. The irrigators we interviewed noted a desire to reduce water loss, a common perspective when water intended for a specific beneficial use is apparently “lost” or “wasted” to seepage or evaporation (Lankford, 2012; Cantor, 2017).

Reduced labor costs were also a factor in the adoption of more irrigation-efficient application technologies in the Henrys Fork. Flood irrigation can take 12–24 h to execute, depending on crop, soil, field size, and slope, and requires monitoring to move tarp dams (Bjorneberg and Sojka, 2005). Hand-line sprinklers need to be connected, disconnected, and moved to their new application location every 8–24 h (Bjorneberg and Sojka, 2005). Center-pivot sprinklers, on the other hand, uniformly water large areas with little labor (Bjorneberg and Sojka, 2005; Brown, 2008), and can be operated remotely (Avello Fernández et al., 2018)—reducing labor costs up to 90% (Brown, 2008). Irrigators elsewhere in the world have also switched from surface to sprinkler irrigation due to labor costs. In Spain, Lecina et al. (2010) documented that irrigation modernization partially occurred due to the high labor requirement of surface application and a diminishing workforce. Irrigators surveyed in Alberta, Canada also reported reduced labor cost as a factor in adopting more efficient irrigation technologies (Wang et al., 2015).

In addition to labor, Henrys Fork irrigators noted the benefit of increased irrigation efficiency to crop yield and quality, which directly affect income. Globally, irrigators report adopting more efficient irrigation technology to improve crop yield and quality too. For example, onion and potato farmers in Morocco’s Saïss plain largely adopted drip irrigation to increase their yield (Benouniche et al., 2014). Irrigators of low-value crops like wheat and barley in Alberta, Canada also reported yield as a motivator for improving their irrigation efficiency (Wang et al., 2015). English vegetable farmers for high-value grocery markets receive higher financial benefit from crop quality than crop yield and make irrigation decisions accordingly (Knox et al., 2012).

In our study, soils informed decisions regarding flood versus sprinkler application and, in combination with local geology, soils contributed to the lagged response of reach gains to surface-water diversion. In regions where soil salinity and nutrient loading are concerns, increasing irrigation efficiency may be a worthwhile pursuit to address water quality degradation created by return flows to streams, as has been documented in Spain's Ebro Basin (Causapé et al., 2006), in the Chiredzi and Runde Rivers in Zimbabwe (Nhiwatiwa et al., 2017), and in the Murray-Darling Basin in Australia (Walker et al., 2021).

Irrigators in the Henrys Fork who have yet to increase their irrigation efficiency noted the high cost of sprinklers. The financial barriers to increasing irrigation efficiency are documented in farming communities worldwide (Koech et al., 2021; Babin et al., 2022). Advocates for increased irrigation efficiency acknowledge these financial barriers and sponsor subsidies to promote access to more efficient irrigation application technologies (Huffaker, 2008; Molle and Tanouti, 2017; Jordan et al., 2023). Critics of these subsidies argue that they facilitate increased consumptive use (Huffaker, 2008; Wheeler et al., 2020), favor larger farms (Jordan et al., 2023), and may put irrigators at greater financial risk as these subsidies enable operation expansion (Scott et al., 2014; Schirmer, 2017). We were unable to determine the role of subsidies in local irrigation conversion. However, we did receive separate comments on the nuisance of cost-share applications, general wariness of government influence, and a concern that larger farms were more adaptable than smaller operations. Although we do not necessarily advocate for subsidies to increase irrigation efficiency, when creating watershed-scale water conservation or irrigation intervention programs, we recommend assessing local attitudes towards the program and program sponsors, as well as their accessibility to diverse farm operations (e.g., Ricart and Clarimont, 2016; Sanchis-Ibor et al., 2021).

Overall, most irrigators in the Henrys Fork watershed who we interviewed revealed that they made decisions regarding irrigation efficiency based on economic efficiency. These results adhere to the common framing of irrigators as economically rational actors who seek to maximize their individual benefit (Qureshi et al., 2011; Contor and Taylor, 2013; Graveline, 2016). Boelens and Vos (2012) note that adopting irrigation efficiency for economic gain is a settler-colonial standard and ignores the values of social efficiency that inform Indigenous irrigation practices, with examples from the Andes. Similar characterizations have been made regarding irrigation modernization in Spain (Oyonarte et al., 2022) and the southwestern United States (Hicks and Peña, 2003; Fernald et al., 2007). Ultimately, the framing that irrigators pursue irrigation efficiency as part of their journey toward economic efficiency holds in highly productive agricultural regions like the Henrys Fork.

4.2 Watershed-scale hydrologic response and implications

In the Henrys Fork watershed, farm-scale decisions to increase irrigation efficiency caused surface-water diversion to decrease by 23% between 1978 and 2000 then remain stable at reduced levels from 2001 to 2022 (Figure 4). We were unable to definitively identify the cause for the abrupt decline in 2001 with our methods. However, two factors may have contributed: drought and irrigation conversion on the Egin Bench. The year 2001 was a severe drought year in the Henrys Fork. State water managers have observed increases in on-farm irrigation efficiency in Idaho in drought years (Mathew Weaver 2023; personal communication, 18 May) and studies elsewhere document drought as a catalyst for increasing irrigation efficiency in the early 2000s (Schuck et al., 2005; Scott et al., 2014). Nonetheless, senior water users like those on the Egin Bench were almost always in priority for water allocation (U.S. Bureau of Reclamation and Idaho Water Resource Board, 2015) and still reduced their surface-water diversion as they converted to more efficient irrigation application (Table 2; Figure 3). The rapid rate of conversion on the Egin Bench from 1978 to 2000 coincides with the decrease in surface-water diversions in the watershed. Conversion on Egin Bench slowed after 2000 (Figure 3) for reasons unknown, coinciding with the stable surface-water diversions 2001–2022. Therefore, the dynamics of irrigation conversion on the Egin Bench may have also been a factor in the dynamics of surface-water diversion through time. Our statistical analysis confirmed a reduction in watershed-total diversion and provided strong evidence for temporal change in diversion even after accounting for the confounding effect of reduced unregulated flow identified within our correlation analysis (Table 3). Reduced diversion as a result of irrigation efficiency improvements have also been observed in other studies (e.g., Sando et al., 1988; Bigdeli Nalbandan et al., 2023).

As irrigation efficiency improved and diversion decreased in the Henrys Fork watershed, reach gains decreased by 299 Mm^3. Elsewhere in the upper Snake River basin, reach gain decline was largely attributed to decreased surface return, but the potential for changes in groundwater use to affect reach gains was acknowledged (Olenichak, 1998). Although we did not specifically investigate groundwater use, groundwater pumping was ~25% of total irrigation withdrawal in 2015, and the 299 Mm^3 decrease we observed in reach gains was larger than the 200 Mm^3 of total groundwater withdrawal from our study area in 2015 (Lovelace et al., 2020). Based on statewide data, we estimate that groundwater use for irrigation in our study area increased by ~24 Mm^3 between 1978 and 2022 (see Supplementary Material). Thus, we conclude that the decline in reach gains in 1978–2000 were from flood-to-sprinkler irrigation conversion. Effectively, then, reach gains prior to 2000 were irrigation return flows to the river. Our result aligns with other studies that have modeled 23%–77% declines in return flows following conversion to sprinkler or drip irrigation (Cai et al., 2003; Toloei, 2015; Hu et al., 2017; Malek et al., 2021).

Return flows are the combination of surface and groundwater returns to the river, where seepage from field application and canal conveyance contribute to groundwater returns specifically. Olenichak (1998) documented return flows were typically supplemented by surface return in river reaches downstream of the Henrys Fork watershed. However, based on field work done in the late 2000s, very little return flow occurs via surface return in the Henrys Fork (U.S. Bureau of Reclamation, 2012b). Our results suggest that return flows at least partially travel through shallow groundwater. The AICc analysis identified diversion decreasing from 1978 to 2000 before dropping abruptly in 2001, whereas reach gains continued to diminish more gradually through 2002 before stabilizing in 2003–2022. The 2-year lag between diversion and reach gain decline likely reflects attenuation in the groundwater system, further emphasizing the relationship between surface-water diversion and reach gains that is also demonstrated in our correlations (Table 3). A lag in streamflow response to groundwater recharge has been documented elsewhere in the Snake River basin (Miller et al., 2003) as well as in other systems (e.g., Kendy and Bredehoeft, 2006; Stoelzle et al., 2014). Given the increase in irrigation efficiency at the field scale, seepage from earthen canals is likely a major contributor in maintaining return flows at present. Thus, when considering a basin-scale shift in irrigation efficiency, it is important to assess the roles of soil, local geology, and conveyance seepage in both farm-scale decisions and the resulting basin-scale hydrology.

Critics of the effort to increase irrigation efficiency as a means for basin-scale water conservation specifically cite how these economically rational decisions at the farm-scale lead to higher consumptive water use and negate water conservation efforts (Ward and Pulido-Velazquez, 2008; Grafton et al., 2018). Overall, our analysis of streamflow data from 1978 to 2022 demonstrated no change in net basin export—the sum of consumptive use and water that exits the basin as groundwater flow to the ESPA. Our study did not include detailed groundwater data. Thus, we cannot quantify how consumptive use and groundwater stored in the ESPA individually contribute to net basin export. However, regional studies have documented a decline in ESPA storage and discharge from 1950 to present (Stewart-Maddox et al., 2018; Sukow, 2021)—suggesting a likely decrease in groundwater export from the watershed. If groundwater export in the Henrys Fork has declined, consumptive use would need to increase to maintain the average annual 1,046 Mm^3 net basin export. Our documented wide-spread conversion to center-pivot sprinklers (Figure 3) demonstrate a mechanism for increased consumptive use within the watershed. Furthermore, the observed reduction of 11 m^3/s in mid-summer reach gain is equivalent to previous scenario modeling predicting a 11.1 m^3/s reach gain decline from 1980 to 2002 due to irrigation efficiency improvements (Contor et al., 2004). Consumptive use of irrigation water by crops in the study area was estimated at 350 Mm^3 in 1980–2010 (U.S. Bureau of Reclamation, 2012b), around one-third of the total water exported from the watershed.

Thus, increases in irrigation efficiency in the Henrys Fork watershed may have increased consumptive use of surface water diversion and decreased return flows available to downstream users. The observed reduction of 11 m^3/s in mid-summer reach gain is the same order of magnitude as a 2020 irrigation-season flow target of ~10 m^3/s in the lower Henrys Fork (Morrisett et al., 2023) and is approximately one-third of the 31 m^3/s average mid-summer streamflow in the Henrys Fork at Rexburg for 2001–2022. Return flows can provide streamflow to downstream users (Simons et al., 2015; Owens et al., 2022), and irrigation systems may be managed with inherent assumptions of return flow reuse downstream (e.g., Boelens and Vos, 2012; Simons et al., 2020). Similar assumptions were made throughout the western United States until a 2007 Supreme Court case determined that the doctrine of recapture within prior appropriation does not require an irrigator to return unused water to its original source. Thus, irrigators are allowed to improve their irrigation efficiency and consumptive use as part of their original water right (MacDonnell, 2011). The loss of return flows has particular implications for downstream users, as they may have junior water rights and be especially sensitive to climate-induced water scarcity (Null and Prudencio, 2016). In the Henrys Fork watershed, the lower Teton River would be a losing reach without irrigation return flows (Apple, 2013). In mid-summer, when upstream users are diverting administrative storage water, the downstream-most water users on the lower Teton River have rights only to reach gains, and the river is managed so that the only physical water available to them are reach gains (Olenichak, 2020). Historically, irrigation return flows were likely a major source of water for lower Teton River irrigators, and return flow reduction has since diminished water availability for these downstream users—an issue that has been discussed numerous times by the local watershed council.

It is not apparent if the loss of irrigation return flows to the lower Henrys Fork watershed has impacted local aquatic ecosystems. Morrisett et al., 2023 did not identify a reduction in trout habitat for 1978–2021 that aligned with the declining reach gains observed in this study; the uniform flow-dependent habitat is consistent with our results that net diversion and streamflow have not changed despite decreased reach gains. However, another study has documented a shift in fish demographics that may be partially explained by thermal stress (Moore et al., 2016), due to a loss of cool groundwater inflow.

Irrigation return flow may be a beneficial climate adaptation tool in many types of systems. In the semi-arid western United States, reduced streamflow and warmer stream temperatures are expected with climate change (Ficklin et al., 2018). In irrigated watersheds, return flows can add resilience by mediating low streamflow and providing cool water refugia (Fernald and Guldan, 2006; Dzara et al., 2019; Van Kirk et al., 2020). Although increasing irrigation efficiency for aquatic ecosystem conservation was not a motivating factor for irrigation conversion in the Henrys Fork, our work provides an example for how increasing irrigation efficiency alone is not a successful tool for increasing streamflow for aquatic habitat. To best benefit aquatic ecosystems, managers and policymakers need to formally allocate water for environmental purposes (Batchelor et al., 2014; Pérez-Blanco et al., 2021; Anderegg et al., 2022). Otherwise, conserved water will continue to be allocated for human demands (Scott et al., 2014; Linstead, 2018). These ideas and methods are broadly applicable to other systems. For example, return flow reduction as a result of increased irrigation efficiency has made wetlands more vulnerable to change (Burke et al., 2004; Peck et al., 2004; Downard et al., 2014), diminished inland lake volume and habitat (Scott et al., 2014; Micklin, 2016; Parsinejad et al., 2022), and degraded delta ecosystems (Frisvold et al., 2018).

4.3 Opportunities for the future: Aquifer recharge as a potential adaptation for watershed management

Options for recovering return flows in the lower Henrys Fork watershed include 1) conducting managed aquifer recharge and 2) maintaining and expanding flood irrigation for incidental recharge. In Idaho, managed aquifer recharge is appropriated through water rights administration and incidental recharge occurs incidental to standard irrigation operations (i.e., seepage via canal conveyance and flood irrigation). Within the scientific literature, agricultural managed aquifer recharge (Ag-MAR) generally references the practice of using irrigation infrastructure or fields for recharge (Levintal et al., 2023) and captures both incidental and managed aquifer recharge as defined by Idaho's state water law.

Managed aquifer recharge is already being conducted in the watershed. In an effort to increase aquifer levels and spring discharge in the ESPA, the Idaho Water Resources Board recently invested over $1M USD to expand managed aquifer recharge infrastructure in the lower Henrys Fork (Patton, 2018). Managed aquifer recharge may only occur when its water rights are in priority and is thus conducted from November to March using existing irrigation infrastructure (i.e., canals) to route streamflow to the Egin Lakes recharge site—8 km from the river near the Egin Bench irrigation study area—for aquifer infiltration and percolation (Idaho Department of Water Resources, 1999). Groundwater models have shown that water recharged at Egin Lakes returns as base flow to the lower Henrys Fork in 3 months (Contor et al., 2009), and if effectively timed, recharge can supplement summer low-flow periods when irrigation diversion peaks (Idaho Department of Water Resources, 1999; Van Kirk et al., 2020).

Achieving recharge incidental to standard irrigation operations will be challenging. Given the economic inertia of irrigation development in the Henrys Fork watershed, it is unlikely irrigators will revert from center-pivot sprinkler application to flood irrigation. Flood irrigation continues to be conducted on some parcels within the Lower Watershed, as evidenced by our 2021 ground-truthing, and has potential to continue given relationship building and proper incentives. Implementing incidental recharge in the Henrys Fork at a scale meaningful for irrigation return flows will require irrigator buy-in.

To incentivize and collaborate with irrigators appropriately, managers and water conservation interests must understand and consider irrigator values and limitations, as well as the impact of climate change and market forces on agricultural production (Ricart and Clarimont, 2016). Our interviews suggested that irrigators who continue to flood irrigate may do so due to financial and land limitations, but also because of their values towards maintaining wildlife habitat and groundwater springs. Ag-MAR needs and constraints are inherently local (Levintal et al., 2023). Honing in on land parcels suitable for Ag-MAR using GIS-based multi-criteria decision analysis (Kazakis, 2018; Sallwey et al., 2019) or computer modeling (Behroozmand et al., 2019) and characterizing irrigator values, constraints, and enablers can identify potentially effective partnerships (Alonso et al., 2019; Sketch et al., 2020; Zuo et al., 2022). Given the economic incentives for increasing on-farm irrigation efficiency highlighted in our interviews, as well as the subsidies in place locally and globally to facilitate adoption of more efficient irrigation, economic incentives will likely be a key factor for implementing incidental recharge. Once the legal and regulatory framework are in place to allow Ag-MAR, economic incentives to conduct Ag-MAR include compensating irrigators for taking on risk through their participation (Dahlke et al., 2018; Gailey et al., 2019), access to the groundwater recharged via property rights or credit (Niswonger et al., 2017; Hanak, 2018; Reznik et al., 2022), and rebates on subsequent groundwater pumping fees (Miller et al., 2021). Lastly, social capital, civic engagement, and capacity building are important for developing cooperative partnerships with irrigators (Lubell, 2004; Alston and Whittenbury, 2011; Sketch et al., 2020) and should be a valued part of Ag-MAR pursuits.

However, the ability to conduct Ag-MAR may be limited by agricultural land availability as irrigators decide to sell their land for residential, urban, and commercial development. Conversion of agricultural land is increasing in the Henrys Fork watershed and is shifting water use to groundwater resources (Baker et al., 2014). Generally, increased groundwater withdrawal combined with decreased groundwater recharge further contribute to diminishing groundwater contributions to the river (Venn et al., 2004; Essaid and Caldwell, 2017). Furthermore, urban encroachment on surface water canals can disrupt their function and hinder local irrigation operations (Hicks and Peña, 2003; Cox and Ross, 2011). Mixed residential and agricultural neighborhoods may also limit the ability of an irrigator to flood irrigate due to the proximity of residential basements (Deng and Bailey, 2020). Thus, residential development within an irrigated landscape can indirectly limit groundwater recharge activities.

Hence, managers and water conservation interests must also be aware of how agricultural land development and conservation play a role in the hydrologic cycle. Li, Endter-Wada and Li (2019) analyzed agricultural land conversion in Utah (United States) and noted that irrigable lands are more likely to be developed due to their proximity to urban areas and flatter terrain, compared to non-irrigated agricultural land that is more rural and on hill slopes. In a nearby Idaho watershed, Huang et al. (2019) found that conservation of agricultural land with riparian buffers may indeed reduce water scarcity, nutrient loading, and sediment export under climate change.

Ag-MAR is not a panacea, however. Water rights priority, irrigator interests, and continued development of irrigable agricultural land may limit its implementation and effectiveness. Therefore, it is imperative water managers and policymakers consider how farm-scale decisions can compound to have watershed-scale hydrologic impacts. Ricart and Clarimont (2016) offer an approach for mapping stakeholder priorities in changing irrigation systems. Lankford et al. (2020) propose the 'irrigation efficiency matrix' framework in which multiple spatial scales and social dimensions are classified for consideration to prevent unintended consequences of changing irrigation landscapes. Numerous scholars urge accounting for basin-scale hydrology in water conservation policy, rather than focusing on maximizing on-farm irrigation efficiency alone (Huffaker, 2008; Ward and Pulido-Velazquez, 2008; Lankford et al., 2020).

5 Conclusion

Increasing irrigation efficiency is an economically attractive option to irrigators in the semi-arid Henrys Fork region to reduce water lost to seepage and improve their agricultural production under water scarcity. However, watershed-wide adoption of more efficient irrigation application has increased consumptive use and reduced return flows. Loss of cool groundwater return flow may exacerbate the effects of climate change on summer streamflow and stream temperature—and Ag-MAR may be a tool to mitigate such loss. Here, we demonstrate an interdisciplinary approach that combines interviews, geospatial analysis, and statistical streamflow analysis to identify the historical motivations and progression of irrigation conversion through time and investigate the watershed-scale response to these farm-scale decisions. Moving forward, when considering water conservation strategies within an irrigated watershed, we recommend managers and policymakers assess current and possible interactions between irrigation efficiency and irrigator behavior, as well as irrigation efficiency and basin-scale hydrology to identify and anticipate potential hydrologic outcomes. A holistic approach that seeks to understand how irrigator priorities contribute to landscape-scale changes in hydrologic regimes will allow watershed management to adapt to water scarcity accordingly.

Data availability statement

The datasets generated and analyzed for this study can be found on Hydroshare in the following repository: https://www.hydroshare.org/resource/5bf4e21aa33d4e7b8a65f0791396d30c/.

Ethics statement

The study involving humans was approved by Utah State University Institutional Review Board under Protocol #12846. The study was conducted in accordance with the local legislation and institutional requirements. The ethics committee/institutional review board waived the requirement of written informed consent for participation because interviews were not recorded and interview notes were collected with unique participant IDs (rather than names). Instead, interviewers read a Letter of Information to the participant that included the minimum elements for exempt applications and asked participants if they agreed and wished to continue.

Author contributions

Conceptualization: CM, RVK; methodology: RVK, CM, LB, AH, and CP; formal analysis: RVK, CM, LB, AH, and CP; investigation: RVK, CM, LB, AH, and CP; resources: RVK and SN; data curation: RVK, CM, LB, AH, and CP; writing—original draft preparation: CM; writing—review and editing: RVK, SN, LB, AH, and CP; visualization: CM, RVK, AH, and LB; supervision: RVK, CM, SN; project administration: RVK; funding acquisition: RVK and CM. As part of 10-week internships with the Henry's Fork Foundation: LB specifically contributed to the hydrologic time-series analysis; AH the geospatial analysis; CP the irrigator interviews. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2023.1188139/full#supplementary-material

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Impact of wastewater reuse on contaminants of emerging concern in an effluent-dominated river

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Contaminants of emerging concern such as pharmaceuticals, personal care products, per- and polyfluoroalkyl substances, and plasticizers, are ubiquitous in effluent-dominated rivers and have potential adverse effects on humans and aquatic life. Demands on water supply have prompted conservation and water reuse measures, impacting the discharge in these rivers, yet the effects of these management decisions on water quality are largely intuited and not quantified. This research examines how changes in water reuse practices will impact concentrations of contaminants of emerging concern, specifically carbamazepine, diclofenac, galaxolide, gemfibrozil, 4-nonylphenol, and perfluorooctane sulfonic acid (PFOS), in the effluent-dominated Los Angeles River (Los Angeles County, California). A water quality module was added to a calibrated hydrologic model of the system and parametrized with observed water quality monitoring data in EPA SWMM. Results indicate that water reuse (i.e., reduced effluent flow) will consistently improve in-stream water quality for all compounds studied except PFOS. However, the improvements are often not substantial enough to mitigate high concentrations directly downstream of treated effluent discharge points. Concentrations of these pharmaceuticals are substantially reduced through attenuation as dilution and degradation occur downstream, though the rate of this attenuation is variable and based on the contaminant. In contrast, concentrations of PFOS increase under some wastewater reuse scenarios and decrease under others but remain below the recommended environmental screening levels. Our work also highlights that management decisions regarding water quantity should integrate water quality modeling to help identify priority monitoring locations and constituents.

KEYWORDS

wastewater reuse, water quality, contaminants of emerging concern, los angeles, urban water management, PFOS, EPA SWMM

1 Introduction

Rivers are used as both sources of drinking water (and other beneficial uses) as well as places to discharge treated wastewater. Many rivers near highly populated areas are effluent-dominated, where most river discharge is due to treated wastewater effluent, particularly during dry weather (Brooks et al., 2006). The water quality of this effluent can then dictate the water quality in the river (Wolfand et al., 2022a). Treated wastewater effluent can be of greater water quality than surface waters, as it is typically low in solids and metals (Wolfand et al., 2022a). However, it can also be a source of other pollutants such as nitrogen and contaminants of emerging concern (CECs), such as pharmaceuticals, personal care products, and perfluorinated alkylated substances (Brooks et al., 2006).

CECs, also called trace organic contaminants or micropollutants, are often not regulated yet pose a major risk to ecosystems, aquatic organisms, and human health (Brooks et al., 2006; Patel et al., 2019). For example, at the parts per trillion level, endocrine-disrupting pharmaceuticals cause adverse effects to fish (Corcoran et al., 2010), and perfluorinated alkylated substances, such as perfluorooctane sulfonic acid (PFOS), are toxic to humans (U.S. Environmental Protection Agency, 2022). Compared to conventional pollutants, CECs pose a unique risk to aquatic life and human health because they are broad in chemical characteristics, difficult and expensive to monitor, and may pose increased toxicity in mixture (Schwarzenbach et al., 2006; Geissen et al., 2015; Mutzner et al., 2022). Both monitoring and modeling have been used, often in conjunction, to develop risk assessments related to CEC exposure or predict the impacts of management decisions (Johnson et al., 2008). Due to the complexity, there is no standard for modeling CECs in river systems (Keller, 2015). Various process-based and empirical water quality models such as EPA SWMM (Park et al., 2007; Jackson et al., 2011; Dittmer et al., 2020), SWAT (Wang et al., 2019), QUAL2K (Zhi et al., 2022), GREAT-ER, (Koormann et al., 2006; Sumpter et al., 2006; Lämmchen et al., 2021a; Lämmchen et al., 2021b), and WASP (Agustin et al., 2023), have been modified or developed to simulate the fate and transport of CECs (Sharma and Kansal, 2013). Others, more recently, have applied various machine learning approaches to simulate concentrations and loads of CECs in rivers (Yun et al., 2021).

Treated wastewater is a pathway for CECs to enter surface waters (Roberts and Thomas, 2006) as many of these pollutants are not fully removed during the conventional wastewater treatment process (Kim et al., 2003). Monitoring studies have shown that these contaminants are present in effluent-dominated rivers (Mandaric et al., 2018), though they are often transformed and attenuated downstream (Gross et al., 2004; Wiegel et al., 2004; Moldovan, 2006). However, the impact of water reuse practices on CEC concentrations in-river is generally unknown.

There is increasing motivation (from the public and regulators) for enhanced water efficiency at the local level. One approach to improving water efficiency is encouraging water reuse and water conservation, especially for outdoor uses. Water reuse practices are increasing globally, especially in the United States, but also in Australia, Belgium, and Singapore (WateReuse Association, 2023), where wastewater is treated centrally for reuse for either irrigation or public drinking water supply (Council, 2012; Ghernaout et al., 2019; Luthy et al., 2020; Olivieri et al., 2020; Radcliffe et al., 2020; Jeffrey et al., 2022). Domestic, commercial, and industrial water conservation practices reduce the amount of raw wastewater flows to wastewater treatment plants (Chappelle et al., 2019). Both conservation measures and water reuse will likely decrease the contribution of treated effluent to flows in urban rivers.

As water conservation and reuse become increasingly efficient, chemical characteristics and quantity of treated wastewater discharges may be drastically altered, impacting both the flow and the water quality in receiving water bodies. The purpose of this study is to quantify the effects of wastewater reuse on in-river concentrations of CECs, using the Los Angeles River as a case study. We use a calibrated hydrologic model created in EPA SWMM to investigate how proposed reductions in treated wastewater flow may impact concentrations of pollutants within the river.

2 Methods

2.1 Study site

The Los Angeles River is an effluent-dominated river located in Los Angeles County, California (Figure 1). The contributing watershed is highly developed, with approximately 32% impervious cover. The primary land use is residential, though there are pockets of industrial and commercial use surrounding the downstream reaches, as well as undeveloped open space surrounding upstream tributaries (Wolfand et al., 2022b). There are eight major flood control dams located along tributaries to the Los Angeles River mainstem. Most of the Los Angeles River mainstem and many of its tributaries are armored with concrete for flood control purposes (Los Angeles County and Los Angeles County Public Works, 2020), but the river also provides recreation and habitat, particularly in the soft-bottom portions of the river such as Sepulveda Basin (Figure 1; approx. from LA20 to LA20_2) and Glendale Narrows (Figure 1; approx. from Glendale WRP to F57C). (Los Angeles County and Los Angeles County Public Works, 2020)

Three water reclamation plants (WRPs) are located within the Los Angeles River watershed: Donald C. Tillman (Tillman) and Glendale, which are located on the mainstem, and Burbank, which discharges to the tributary Burbank Channel. The climate is semi-arid Mediterranean with spatially variable rainfall of 200–460 mm (8–18 in.) annually (Wolfand et al., 2022b). The drinking water supply is primarily from imported water and local groundwater supplies (Los Angeles Department of Water and Power, 2020). The availability and reliability of these sources are expected to be strained by climate change and continuing population growth in the region, so there are regional plans to diversify water supply portfolios, especially by increasing recycled water (Los Angeles Department of Water and Power, 2020). On average, Tillman, Glendale, and Burbank WRPs typically discharge approximately 46 MGD (2.0 m^3/s) of treated wastewater to the river (for water years 2011–2017), but they have plans to collectively reuse approximately 20% (California State Water Resources Control Board, 2017b; California State Water Resources Control Board, 2017a; City of Los Angeles Bureau of Sanitation, 2021). The flow, ecology, water quality, and temperature implications of these changes have been predicted by various other studies (Abdi et al., 2021; Stein et al., 2021a; Stein et al., 2021b; Abdi et al., 2022; Wolfand et al., 2022a; Wolfand et al., 2022b). Notably, previous work has found that indicator species may experience habitat degradation due to just a 4% decrease in wastewater discharge during the dry season (Wolfand et al., 2022b).

CECs have been previously monitored in treated effluent, freshwater, and sediments in the Los Angeles River basin (City of Los Angeles Bureau of Sanitation, 2014; Southern California Coastal Water Research Project, 2017). An expert panel in the state

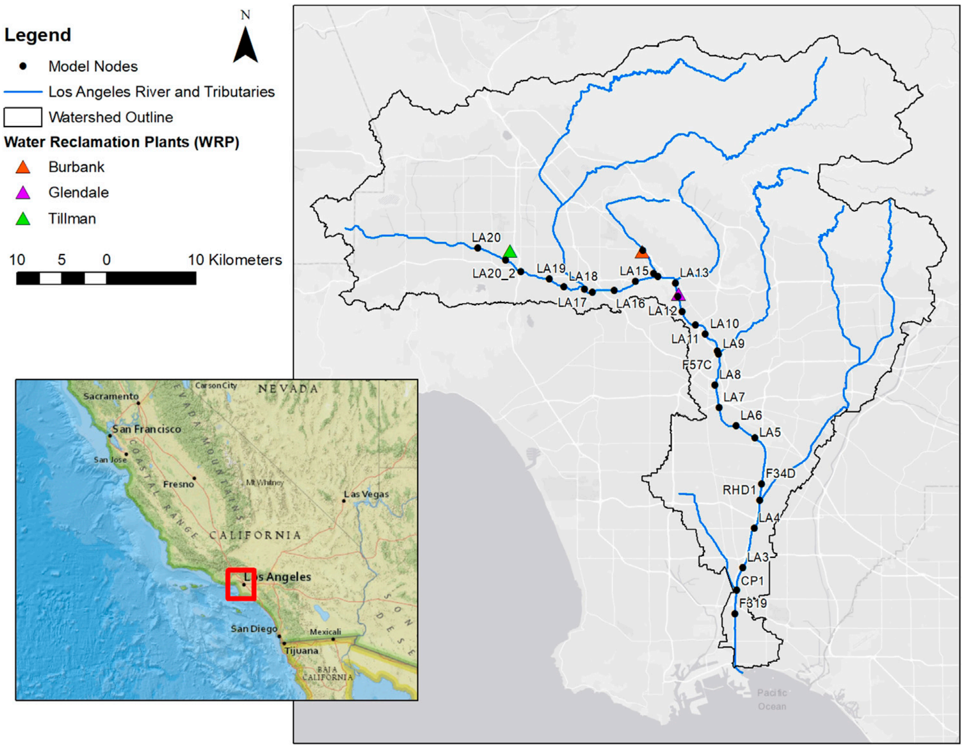

FIGURE 1
Map of the Los Angeles River watershed and model nodes. The model was calibrated at node LA12, immediately downstream of all three water reclamation plants.

TABLE 1 Compounds of interest, average concentrations in wastewater effluent, and calibration parameters.

Compound	Description	Water reclamation plant concentrations* (ng/L)			Calibration parameters		
		D.C. Tillman	Glendale	Burbank	First order k (d^{-1})	Half-life (h)	Observed/Modeled conc. (percent difference**)
Carbamazepine	pharmaceutical; anti-convulsant drug used to treat seizures, nerve pain, and bipolar disorder	140	140	140	2.25	7.4	56.25/56.04 (−0.38)
Diclofenac	pharmaceutical; nonsteroidal anti-inflammatory drug, pain reliever	76	85	85	1.36	12.2	41/40.69 (−0.75)
Galaxolide	fragrance; used as an odor enhancer in personal care products	2600	2100	2600	17.5	0.95	180/180.9 (0.50)
Gemfibrozil	pharmaceutical; cholesterol medication	270	220	270	3.17	5.2	83.5/83.53 (0.037)
4-nonylphenol	endocrine disrupting compound; used in industrial processes and products such as detergents and plastics	50	50	50	0	n/a	385/39.18 (−90)
PFOS	fluorosurfactant; used in household products as fabric protector	3.5	5.6	5.6	1.35	12.3	2.525/2.516 (−0.38)

*Concentrations in effluent are historical average concentrations as reported by the City of Los Angeles. (City of Los Angeles Bureau of Sanitation, 2014).
**Percent difference is the difference between simulated and observed concentrations.

established monitoring trigger levels (MTLs) for CECs that pose the greatest risk to aquatic systems (Anderson et al., 2012). Six emerging CECs were chosen based on historical monitoring data: carbamazepine, diclofenac, galaxolide, gemfibrozil, 4-nonylphenol, and perfluorooctane sulfonic acid (PFOS) (Table 1). Diclofenac, galaxolide, and 4-nonylphenol are consistently present at higher concentrations at stations downstream of WRPs in the Los Angeles River Basin. Carbamazepine, gemfibrozil, and PFOS are also observed in the treated effluent (City of Los Angeles Bureau of Sanitation, 2014). The observation of these six pollutants in the Los Angeles River is consistent with other studies that show that many of these compounds are not fully removed in current wastewater removal processes (Kim et al., 2018).

2.2 Water quality model

2.2.1 Model design

An hourly hydrologic model was created in EPA's Storm Water Management Model (SWMM), for the Los Angeles River. Detailed documentation on the hydrologic model development, calibration, and validation is provided in Wolfand et al., 2022b. Briefly, the basin was discretized into 77 catchments, and precipitation data was retrieved from 72 gages in Los Angeles County and spatially interpolated for each catchment using kriging methods. The model was run for water years 2011–2017. This time range was chosen due to 1) the availability of high-resolution hydrologic and climate data, and 2) relatively constant wastewater discharge during this period before major water reuse practices were implemented. Observed time series data, such as WRP discharges, dam discharges, and dry weather baseflows were included in the model. The model was calibrated and validated at 7 observed gage stations (4 on the mainstem, 3 on tributaries), with Nash Sutcliffe Efficiency (NSE) between 0.67 and 0.94 and percent bias between −20% and 17.3%.

For this study, channel geometry within the SWMM model was updated based on a previously developed HEC-RAS hydraulic model of the system (Stein et al., 2021b). The model was then parameterized to include CECs of interest: carbamazepine, diclofenac, galaxolide, gemfibrozil, 4-nonylphenol, and PFOS. There are three primary inputs of CECs in the river: stormwater runoff, dry-weather runoff from stormdrains (from activities such as car-washing and irrigation return flows), and WRP effluent. The analysis focused on flows during May and June, so that contribution of stormwater runoff to river discharge was negligible; during these months, there is typically no precipitation due to seasonal rainfall patterns.

Dry-weather runoff from stormdrains, which contributes approximately 20%–50% of the river discharge (Wolfand et al., 2022b), was not explicitly modeled due to lack of available data; instead, the contribution of this source to CEC concentrations was captured by parameterizing river water concentrations (baseflows). River water concentrations of the selected pollutants were included in the model upstream of the three WRPs at LA20 based on historical monitoring data. However, only PFOS was detected at an initial concentration of 6.85 ng/L.

WRP effluent concentrations for each CEC were included in the model based on monitoring grab samples taken in June 2014 (Table 1) (City of Los Angeles Bureau of Sanitation, 2014). 4-nonylphenol was not detected in WRP effluent, due to challenges in low-level quantification, so it was assumed concentrations were equal to the method detection limit (MDL), 50 ng/L. No effluent monitoring data was provided for Burbank WRP, so it was assumed that concentrations were the highest average value of those calculated for Tillman or Glendale WRPs.

2.2.2 Calibration

First-order decay was used to approximate the natural decay of these compounds due to photolysis, biodegradation, sorption, and other environmental conditions (Lin et al., 2006). The first-order decay constant for each compound was manually calibrated to minimize the percent difference between simulated and observed concentrations downstream of the last wastewater discharge point, node LA12 (Figure 1; Table 1). Simulated concentrations were the monthly median concentrations from the model for May and June, while observed concentrations were grab samples taken within the river below Glendale WRP in May and June 2018 (Maruya et al., 2022).

2.2.3 Scenarios

Seven WRP reuse scenarios were assessed: 0%, 10%, 25%, 50%, 75%, 90%, and 100% reuse, where 100% reuse means all effluent is treated and directed to other uses and is not directly discharged to the river. Flows and concentrations were evaluated at each node along the river mainstem. Outputs are reported as the median value for May and June. To distinguish between the effects of dilution and degradation, an additional scenario of 0% reuse (baseline conditions) that did not include chemical degradation was simulated. Note baseline conditions are defined by the simulation period (WY 2011–2017), during which time wastewater reuse was minimal. Simulated concentrations were compared to monitoring trigger levels for diclofenac (100 ng/L) and galaxolide (700 ng/L) determined by a Science Advisory Panel convened by the California State Water Resources Control Board (Anderson et al., 2012). Regulatory standards for PFOS are developing globally. The U.S. EPA released draft aquatic life ambient water quality criteria for PFOS in April 2022 of 3.0 mg/L for acute exposure (1-h average) and 0.0084 mg/L for chronic exposure (4-day average) in freshwater (EPA, 2022). As of 2020, California has an Interim Final Environmental Screening Level of 0. 56 μg/L (560 ng/L) for groundwater for direct exposure to freshwater aquatic species (California Water Boards, 2023). EPA's non-enforceable drinking water advisory level for PFOS is 70 ng/L. (U.S. Environmental Protection Agency, 2022)

3 Results

3.1 Calibration

The percent bias of simulated river concentrations was minimized to <1% for all compounds except for 4-nonylphenol (Table 1). Concentrations for 4-nonylphenol are much higher in the river than can be explained by discharge from WRP effluent alone, and thus calibration was unsuccessful (percent bias: 90%; Table 1). We have several hypotheses for why calibration may have been unsuccessful: 1) there is limited observational data, which may not have fully captured river concentrations; 2) there are permitted industrial dischargers to the river, which were not captured by the model; and 3) the presence of this pollutant in bed sediment could be an additional flux to freshwater; 4-nonylphenol has been observed in estuarine sediments collected in other parts of California (Anderson et al., 2012). Model results for 4-nonylphenol were therefore not reported. Calibrated half-lives for other compounds are within the

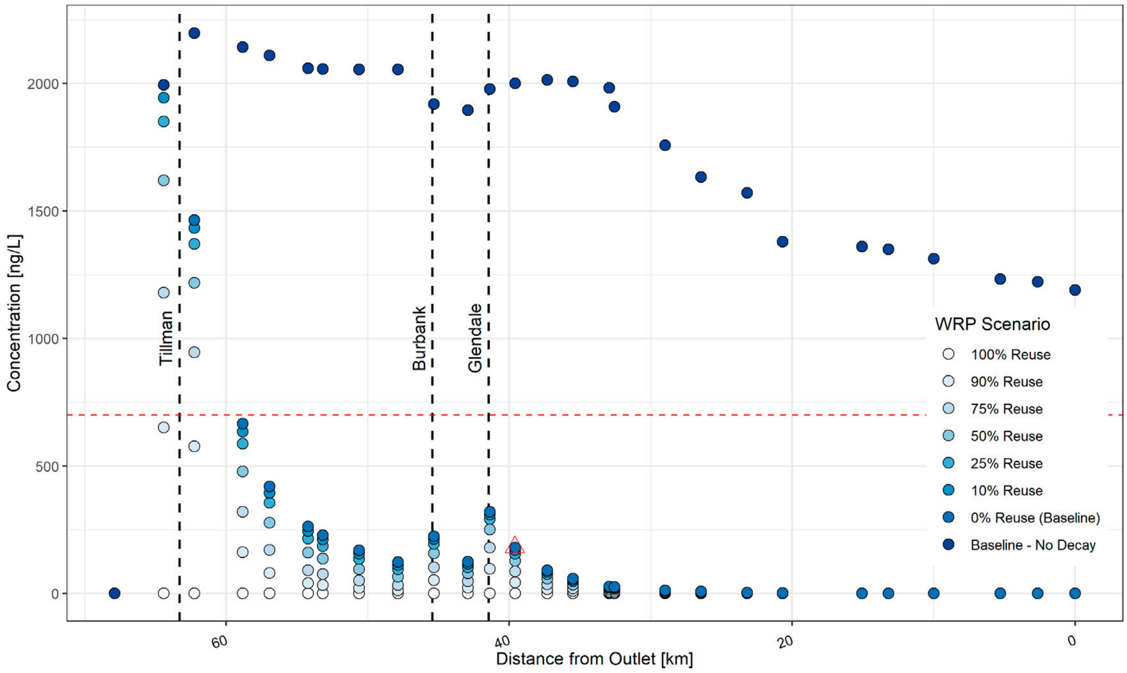

FIGURE 2
Simulated galaxolide concentrations (median across May and June) in the Los Angeles River under various water reclamation plant (WRP) scenarios. The locations of the three WRPs along the river are marked with vertical dashed lines. The horizontal dashed line indicates the monitoring trigger level for galaxolide of 700 ng/L. The red triangle indicates observed monitoring data.

observed ranges of experimental half-lives. For example, the calibrated half-life of carbamazepine was 7.4 h, which is similar to the half-life range of 8–12 h found in solar simulator experiments in river water (Matamoros et al., 2009). The calibrated half-life of diclofenac was 12.2 h, which falls within the reported range of 10–21 h found in field observations from lakes in Sweden (Bu et al., 2016).

3.2 Scenario analysis

3.2.1 Existing conditions

Existing flows are substantially augmented by WRP discharge, particularly in May and June when precipitation and stormwater runoff are limited (Supplementary Figure S1). (Wolfand et al., 2022b) Under baseline conditions (0% wastewater reuse), concentrations of CECs typically increase immediately downstream of the wastewater discharge point. For example, galaxolide concentrations increase from approximately 0 ng/L to 1,994 ng/L in the river after the last discharge point from Tillman (Figure 2). Note that Tillman WRP has multiple discharge points between node LA20 and Sepulveda Dam (USGS gage # 11092450; Supplementary Figure S9). The same pattern holds for pharmaceuticals carbamazepine, diclofenac, and gemfibrozil (Supplementary Figures S2–S4). However, concentrations of PFOS in the river decrease downstream of WRPs, due to the relatively low concentration of PFOS in WRP effluent compared to background concentrations. Because of this, WRP discharge serves to dilute PFOS concentrations immediately downstream of all three WRPs.

Simulated concentrations of CECs are attenuated along the river as flows increase. The rate of decrease is due to both dilution and degradation of the compounds, described by first-order decay in the model. Under existing conditions, at the outlet of the watershed (node F319), simulated concentrations of carbamazepine are reduced to 6.1 ng/L, diclofenac is reduced to 8.4 ng/L, and gemfibrozil is reduced to 4.8 ng/L (Supplementary Figures S2–S4). The simulated concentration of PFOS is 0.5 ng/L and galaxolide is 0.02 ng/L at the watershed outlet (Figures 2, 3). A monitoring study in the nearby Santa Ana River also found that CECs were substantially attenuated downstream of wastewater discharge points (Gross et al., 2004).

3.2.2 Water reuse scenarios

Simulated concentrations of CECs in the Los Angeles River decrease with increased water reuse for all compounds except PFOS. WRP discharge contains most of these CECs, so increased reuse results in lower concentrations in the river, due to dilution by dry-weather baseflow (primarily from dry-weather runoff). For example, concentrations of galaxolide downstream of Tillman WRP at node LA20_2 decreases from 1,994 ng/L to 1,620 ng/L with 50% reuse (Figure 2). The same is true for pharmaceuticals carbamazepine, diclofenac, and gemfibrozil (Supplementary Figures S2–S4).

In the case of PFOS, the concentration in treated effluent from Tillman WRP (3.5 ng/L) is less than the background levels of PFOS in the river, so the treated effluent helps dilute the concentration in the river. Simulated WRP discharge from Burbank and Glendale have higher concentrations of PFOS (5.6 ng/L) than the Los Angeles River water, so when chemical degradation is not considered, these WRPs increase concentrations of PFOS in the river (Figure 3;

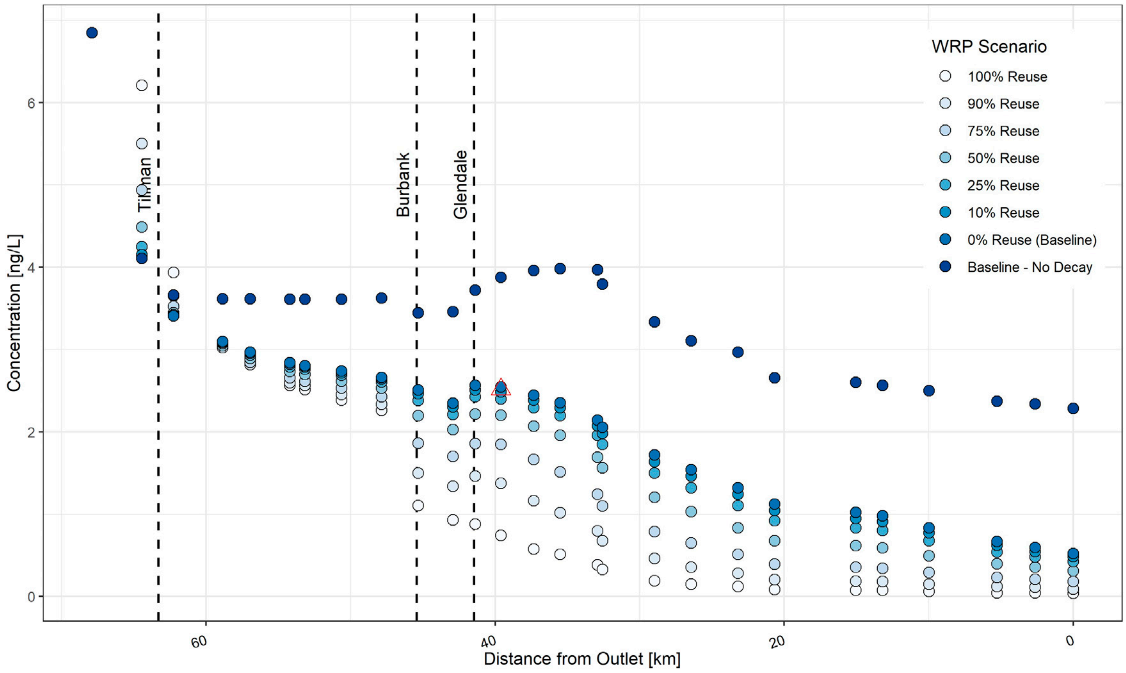

FIGURE 3
Simulated PFOS concentrations (median across May and June) in the Los Angeles River under various water reclamation plant (WRP) scenarios. The locations of the three WRPs along the river are marked with vertical dashed lines. The red triangle indicates observed monitoring data. The U.S. EPA drinking water advisory level for PFOS is 70 ng/L (beyond the y-axis scale).

"Baseline–No Decay"). When both dilution and chemical degradation are considered, PFOS concentrations either increase or decrease depending on the WRP reuse scenario and the location in the river (Figure 3). All simulated concentrations of PFOS fall well below the environmental screening level of 560 ng/L and the EPA drinking water advisory level of 70 ng/L, with a maximum simulated in-river concentration of less than 7 ng/L across all scenarios (Figure 3). However, simulated concentrations likely underestimate PFOS because only background in-river concentrations and WRP effluent were parametrized as sources of PFOS in the model. PFOS has been observed in wet weather runoff (stormwater) (Houtz and Sedlak, 2012; Xiao et al., 2012), and dry weather runoff likely contains PFOS, though there was no monitoring data for this potential source. Dry weather runoff should be targeted for future monitoring efforts.

The rate of attenuation is important when considering the potential impacts of these compounds on aquatic species. Under existing conditions, the recommended monitoring trigger level of 700 ng/L for galaxolide is exceeded at some locations directly downstream of the Tillman WRP and Burbank WRP (Figure 2; Supplementary Figure S9). Simulated median concentrations fall below the monitoring trigger level at approximately 3.5 km downstream of Tillman (Figure 2) and at the confluence of Burbank Channel with the mainstem of the Los Angeles River (Supplementary Figure S9). Wastewater reuse will decrease in-stream concentrations of galaxolide, below the monitoring trigger levels, especially when reuse is upwards of 90%. However, as mentioned previously, the proposed WRP reuse is likely only about 20%, at which point concentrations are diminished but only minimally (Figure 2). This suggests that there may be localized impacts of the WRP, regarding galaxolide, associated with currently planned wastewater reuse.

According to simulation results, the monitoring trigger level for diclofenac is not exceeded at any point along the river under baseline conditions or wastewater reuse scenarios (Supplementary Figures S3, S7). Concentrations of PFOS are also below EPA's draft chronic aquatic exposure criteria (0.0084 mg/L) and drinking water advisory level (70 ng/L; Figure 3). Aquatic life criteria have not been established for the three other compounds (carbamazepine, gemfibrozil, and 4-nonylphenol).

4 Discussion

4.1 Significance and limitations

Water quality modeling of CECs can be especially powerful because it is often difficult to draw river-scale or watershed-scale conclusions with the limited availability of CEC monitoring data. This work provides a framework for future studies to estimate the impacts of management scenarios on downstream conditions without intensive monitoring. The developed approach uses industry-standard software (EPA SWMM) to predict concentrations of CECs across management scenarios. Few studies have applied EPA SWMM in this way (Park et al., 2007; Jackson et al., 2011; Dittmer et al., 2020). The model can be readily applied to multiple compounds and can be used as a screening tool to determine which compounds and/or locations should be prioritized for monitoring and/or intervention.

However, the limited availability of monitoring data results in models that are inherently limited in their scope and applicability. In this study, no data was available on concentrations of the pollutants of interest in sources other than wastewater such as industrial discharges, or dry weather runoff. Because the study was limited to the dry season (summer months), impacts from stormwater runoff are negligible. However, dry weather runoff, from activities such as irrigation, car washing, and industrial discharges, may contain the pollutants of interest, particularly PFOS, which is a household fluorosurfactant. The pharmaceuticals diclofenac, carbamazepine, and gemfibrozil are likely not present in dry weather runoff, except in the case of illicit discharges, which were not captured by the model. A study of the South Platte River (Denver, CO) reported monitoring data that showed that wastewater treatment plant discharge was not the primary factor controlling water quality, instead, industrial discharges and stormwater runoff contributed (Schliemann et al., 2021), both of which were not included in our model due to limited data. Even just one or two more monitoring grab samples, either spatially or temporally, could improve model calibration, and therefore the accuracy, of predictions. In addition, a more complete dataset would allow for the validation of the model.

An additional limitation of this study includes assuming a first-order decay of these compounds as a proxy for combined degradation processes in the environment. Photodegradation is likely a primary degradation pathway for these compounds in the shallow and wide Los Angeles River, but other processes are occurring including hydrolysis, oxidation, sorption, and biodegradation. The degradation rates were assumed to be constant, but degradation is variable based on factors such as sunlight, temperature, and pH.

4.2 Management implications

Los Angeles plans to recycle approximately 20% of existing wastewater, which will result in modest reductions in the CEC concentrations simulated in this study, except for PFOS, which may increase in concentration immediately downstream of Tillman WRP but remain below environmental screening levels. Assuming 20% wastewater reuse, galaxolide concentrations will still be above recommended monitoring thresholds, for kilometers downstream of Tillman WRP. Therefore, there are still potential adverse effects from CECs, particularly near WRP discharge points. This may be of particular concern because areas downstream of the WRP discharges are often the locations where wetland and riparian habitat, recreational boating, fishing, and wading occur due to the persistence of flows. Therefore, while the adverse effects may be concentrated in specific areas, the risk to humans and wildlife could be intensified, emphasizing the need for prioritized monitoring of these locations in the future.

In this work, we found that as water reuse increases, CEC concentrations typically decrease (except for PFOS). This contrasts with our previous work that shows concentrations of conventional pollutants total suspended solids, total dissolved solids, copper, and lead increase with increased water reuse (Wolfand et al., 2022a). Treated WRP effluent is an input of CECs to the river, whereas it serves as a mechanism for dilution for conventional pollutants such as solids and metals. Therefore, the impact on water quality in the Los Angeles basin as a result of water reuse is expected to vary depending on the specific pollutant, leading to both improvements and potential deterioration. In addition, water reuse will reduce flows in the river, also potentially having a negative impact on wetland and riparian habitats (Wolfand et al., 2022b). The implications of water reuse for river flow and water quality should be weighed against the benefits of increased local water supply.

While we only evaluated a subset of CECs, we know that there are an indeterminate number of unmonitored compounds and degradation products. Some degradation products are equally if not more toxic to aquatic life in the environment (Li et al., 2016). Our analysis shows that CEC concentrations generally improve with increased reuse, but the true ecological impacts of this are much more complex as toxicity can be compounded by CECs in mixture. As water reuse increases, particularly for potable reuse, the urban water cycle approaches a closed-loop system, particularly within one region or city. Advanced water purification processes can help ensure that CECs are fully removed from the urban water cycle.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary Material, further inquiries can be directed to the corresponding author.

Author contributions

JW, ES, and TH designed the study. JW completed the water quality modeling and analysis. KT-Q and AS provided input on the model development and data analysis. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2023.1091229/full#supplementary-material

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An environmental flow to an ephemeral creek increases the input of carbon and nutrients to a downstream receiving river

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Although intermittent and ephemeral rivers lack surface flow for part of the year, they provide vital refuges for biota in otherwise dry semi-arid and arid landscapes. The hydrology of many such rivers has been altered due to river regulation and climate change. Environmental flows can be delivered to address the negative impacts of regulated flows, however there is limited knowledge of how dry ephemeral ecosystems respond following environmental flows. This study examined changes in water quality of the ephemeral Thule Creek in the southern Murray-Darling Basin, Australia, following delivery of environmental water from an irrigation canal. We also examined how the environmental flow influenced water quality of Wakool River that receives inflows from Thule Creek. Six sites in Thule Creek, three in Wakool River, and one in Yarraman irrigation channel (source water) were monitored for dissolved organic carbon (DOC), nutrients and dissolved oxygen (DO) once per week over 15 weeks from October 2019 to January 2020. The environmental flow resulted in high DOC concentrations (4.4–76 mg/L). Although low DO levels at sites in Thule Creek were recorded on some dates below the threshold for fish stress (< 4 mg/L) there were no fish kills observed during the environmental flow. The carbon-rich and nutrient-rich water (DOC >10 mg/L, total phosphorus up to 94 µg/L, total nitrogen up to 1,125 µg/L) was detected in the Wakool River downstream of Thule Creek confluence compared to the Wakool River upstream of Thule Creek confluence (DOC 6.6 mg/L, total phosphorus up to 64 µg/L, total nitrogen up to 805 µg/L) during the period when the environmental flow in Thule Creek was connected with the Wakool River. This research provides an example of how irrigation canal networks can be used to deliver environmental water to an ephemeral river to maintain refuges and contribute to the productivity of a receiving river further downstream. Careful management of the timing, volume and duration of environmental flows in arid or semi-arid landscapes is needed to avoid the development of poor water quality during, or following, the delivery of environmental water.

KEYWORDS

environmental flows, intermittent and ephemeral rivers, irrigation infrastructure, instream productivity, dissolved organic carbon, dissolved oxygen, refuge, Murray-Darling Basin

1 Introduction

Intermittent and ephemeral rivers constitute over half the world's river network (Datry et al., 2017; Messager et al., 2021). Although intermittent and ephemeral rivers lack permanent surface flow, they provide a vital source of water and refuge for biota in otherwise dry landscapes. It has been recognised that they support a high level of biodiversity, provide valuable habitat for endemic and rare species and contribute to biogeochemical processes of the ecosystems (Capon, 2003; Deil, 2005; Vorste et al., 2020). However, intermittent and ephemeral rivers are threatened by climate change and intensive anthropogenic pressures resulting in their rapid loss and degradation both globally (Dudgeon et al., 2006; Palmer et al., 2008; Pekel et al., 2016) and within Australia (Finlayson et al., 2011). Despite their prevalence throughout the world, intermittent and ephemeral rivers are less studied than perennial rivers.

The hydrological regimes of intermittent rivers are harder to characterise than those of permanent rivers (Graf and Lecce, 1988) because they are more variable as the rivers stop flowing or dry out at some point in time and space. These systems fluctuate in time and space in response to inter-annual hydrological variability, and seasonal hydrological variability in some circumstances (Zanor et al., 2012). Intermittent rivers receive inconsistent surface water flow and may cease to flow during dry seasons. Runoff from precipitation or upstream sources might be a supplement water source for these river systems (Zollitsch and Christie, 2014). Ephemeral rivers experience periodic wetting and drying cycles that substantially vary in terms of the timing, frequency and duration of their inundation events (Williams, 1996; Brock et al., 2003). The hydrology of many lowland river systems has been altered as a result of river regulation (e.g., Ward and Stanford, 1995; Nilsson and Berggren, 2000; Nilsson et al., 2005), with small to medium flood events being captured by upstream impoundments and water diverted out of river systems for irrigation and other uses. The flow regime of intermittent and ephemeral rivers has also been altered due to river regulation. Some intermittent and ephemeral rivers receive inflows from upstream permanent rivers. However, some inland intermittent and ephemeral rivers are situated high on floodplains and are only occasionally inundated by floodwater following large flow events and floods. Under unregulated conditions (modelled natural) these river systems would have received more frequent flows.

The hydrological connection between river channels and their floodplains promotes the transportation of carbon and nutrients, influencing in the functioning of the entire river system (Harris et al., 2017). The productivity and biodiversity of floodplain river ecosystems is closely linked to flows that can mobilise resources (e.g., carbon and nutrients) from low lying geomorphic features during in-channel flows and from floodplains during large flow events (Junk et al., 1989). These resources are subsequently utilised by a range of organisms, including bacteria, invertebrates, waterbirds and fish over a range of temporal and spatial scales. River regulation has disrupted river-floodplain connectivity through alterations in the volume, timing, and duration of high flow events (Ward and Stanford, 1995; Bunn and Arthington, 2002). Due to the loss of river-floodplain connectivity, the condition of terrestrial-aquatic habitats of intermittent and ephemeral rivers have declined and their biogeochemical processes have been altered.

Environmental water is water allocated and managed to improve and restore degraded river, wetland, and floodplain ecosystems (including the plants and animals that depend on these ecosystems), mitigate detrimental environmental outcomes of alteration of flow cycles (Konrad et al., 2011) and return a more natural cycle of flows to river and wetland ecosystems. Research on environmental flows has largely focussed on water delivered to perennial-permanent river systems and associated wetlands (e.g., Lind et al., 2007; Shafroth et al., 2010). The majority of environmental flows to rivers are delivered by the release of water from dams and weirs (Konrad et al., 2011; Opperman et al., 2019) to meet the requirements of water-dependant ecosystems further downstream. Due to the way in which river regulation infrastructure has been developed in many river systems, it may be difficult to deliver environmental water to intermittent and ephemeral rivers that are less connected to the main sources of water. Without environmental water, key refuges in these intermittent and ephemeral systems would dry up. In some circumstances and locations there may be the opportunity to use irrigation canal network infrastructures to deliver environmental water to rivers (e.g., Watts et al., 2018). However, there is limited knowledge of how dry intermittent and ephemeral ecosystems respond following environmental watering, and to what extent this can impact the water quality of river systems further downstream.

In this research we examined outcomes of the delivery of environmental water from an irrigation canal network on the water quality and instream productivity of the ephemeral Thule Creek in the southern Murray-Darling Basin (MDB), Australia. We hypothesised that the delivery of environmental flows to this ephemeral creek via the irrigation canal would result in the release of carbon and nutrients from leaves, organic matter and soil in the previously dry creek bed into the water, and this would influence the water quality of the receiving river system downstream. This research addressed two questions; 1) Does the delivery of environmental water via an irrigation canal to Thule Creek create detectable pulses of carbon and nutrients in this ephemeral creek?, and 2) Is the input of carbon and nutrients from the ephemeral creek detectable in the receiving Wakool River system?

2 Methods

2.1 Study area

The Edward/Kolety-Wakool River system is a large anabranch of the Murray River in the southern MDB, Australia. It is a complex network of permanent rivers, inter-connecting streams, creeks, wetlands and ephemeral and intermittent rivers. The Edward/Kolety-Wakool region has an extensive network of irrigation canals and channels which in some places can be used to release water into rivers and creeks via irrigation canal escape infrastructure, hereafter referred to as "irrigation escapes". The irrigation escapes are traditionally used to drain excess water from the irrigation system at the end of the irrigation season or in times of heavy rainfall or floods. In more recent times they have also been to deliver water from environmental accounts to rivers and creeks which have suffered reduced frequency of flows due to

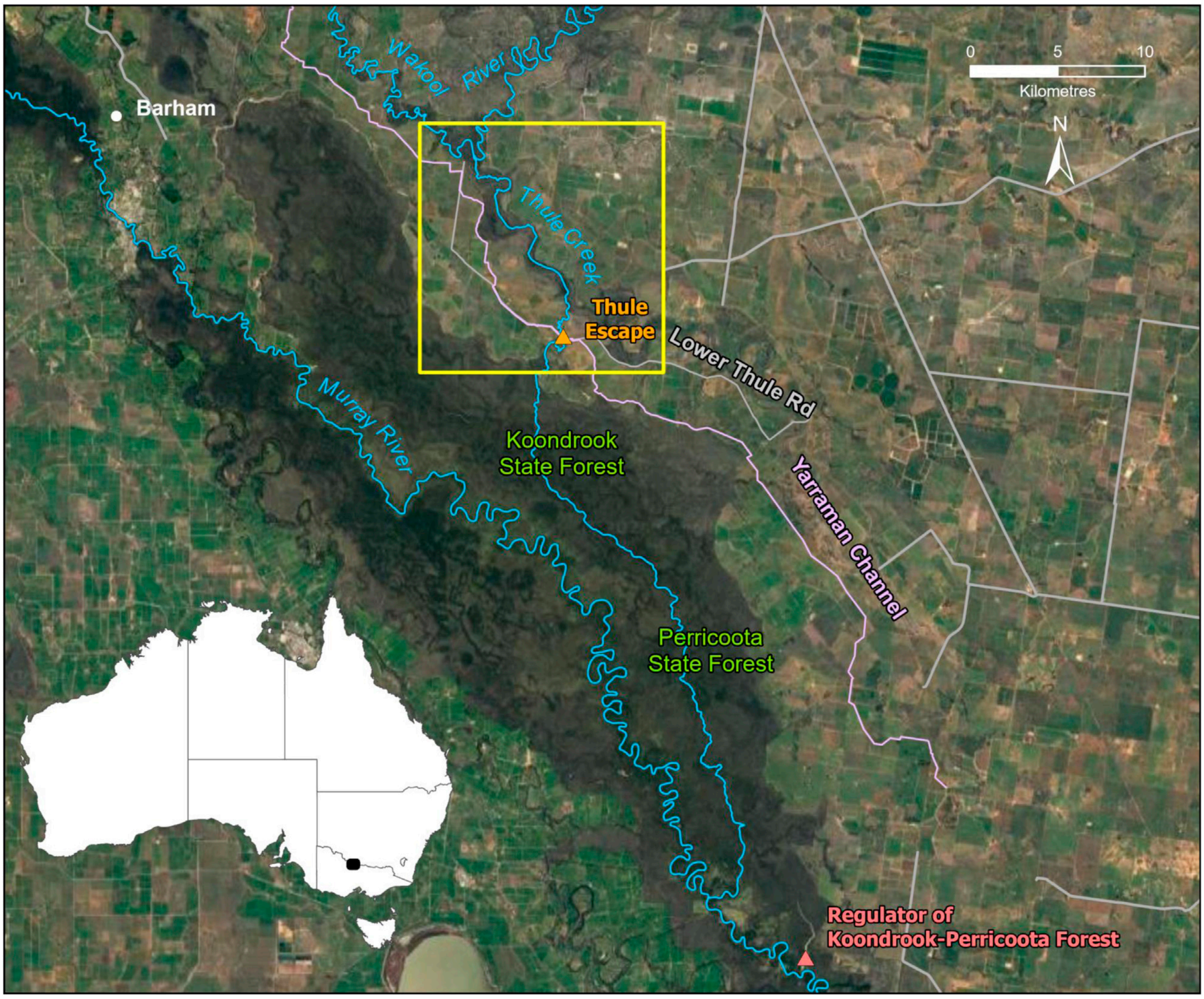

FIGURE 1
Yellow square indicates the location of the Thule Creek area within the mid-Murray River in the Murray–Darling Basin, Australia. (Source: ESRI, 2024).

reduced flood frequency as a result of the heavily modified flow regime that now persists in the Edward/Kolety-Wakool River system (Watts et al., 2020). The floodplain landscape in the region is a mosaic of agricultural land (irrigation, cropping and pasture for grazing) and native vegetation, including large river red gum floodplain forests and other riparian vegetation along river corridors. Like many other rivers of the MDB, the flow regimes of the Edward/Kolety-Wakool system have been considerably changed by river regulation (Green, 2001; Hale and SKM, 2011; Murray-Darling Basin Authority, 2018). Under regulated flow conditions the number of small to medium overbank flows have decreased in winter and early spring, and there have been fewer opportunities for carbon and nutrients to be exported from the floodplain to receiving creeks and rivers.

Flows in the permanent rivers and tributaries of the Edward/Kolety-Wakool system remain within the channel under regulated conditions, whereas during high unregulated flow events there is connectivity between the river channels, floodplains and several large forests including the Barmah-Millewa Forest, Koondrook-Perricoota Forest and Werai Forest that together form the NSW Central Murray Forests Ramsar site (Department of Climate Change, Energy, the Environment and Water, 2013). As some of the rivers in the Edward/Kolety-Wakool system are highly regulated and have low base flows during summer there is a risk of poor water quality developing in this system, particularly during warm weather.

Thule Creek is one of the many ephemeral rivers within the Edward/Kolety-Wakool River system (Figure 1). Flows in Thule Creek are reliant on Koondrook-Perricoota Forest being inundated from the Murray River. Water from the flooded forest then drains into Thule Creek and then flows downstream to the Wakool River (Murray-Darling Basin Authority, 2012). As with many ephemeral creeks in the MDB, river regulation has severely reduced the frequency of flows in Thule Creek, which threatens the ecosystem of the creek. The frequency of overbank flows that connect the Koondrook-Perricoota Forest to Thule Creek has been reduced by at least 50% due to reduced overbank flows in the Murray River as a result of river

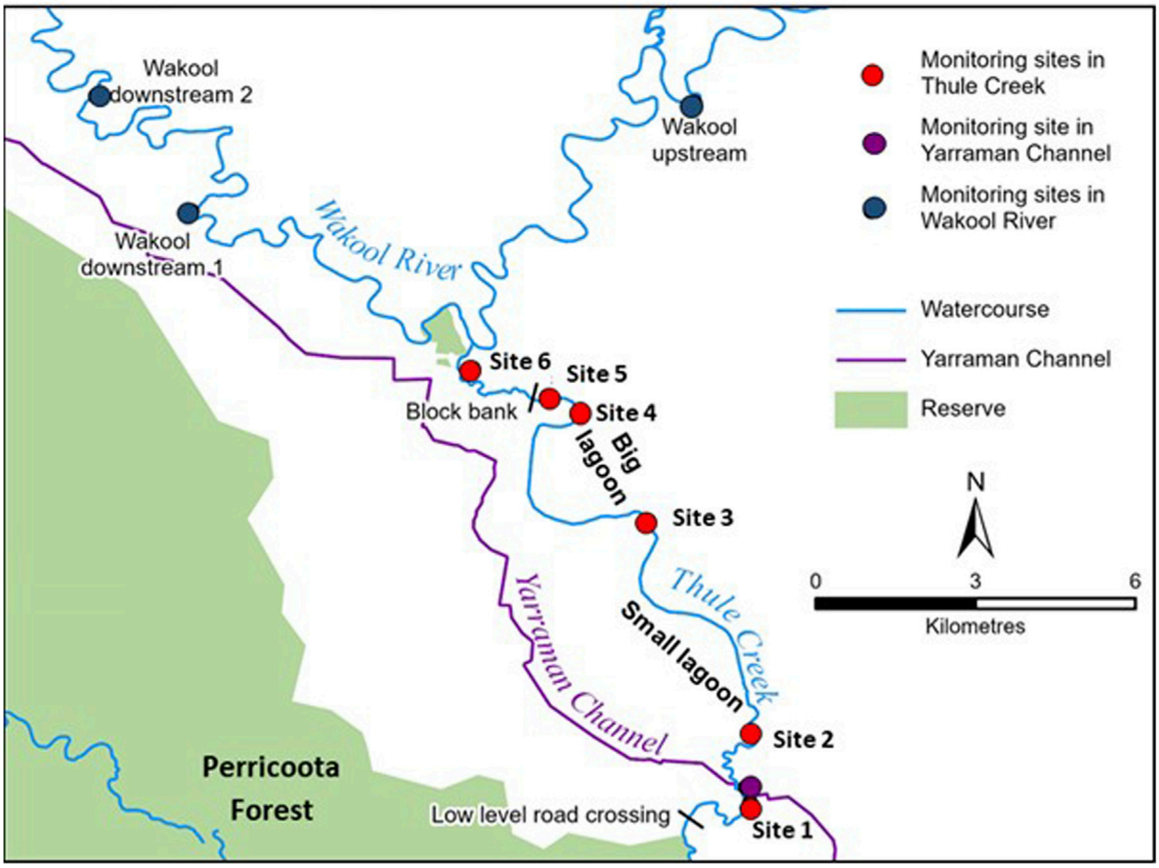

FIGURE 2
Location of monitoring sites along Thule Creek within the Edward/Kolety-Wakool River system. Numbered sites (red dots) show the six sites in Thule Creek from upstream to downstream. Dark blue dots are the three sites in Wakool River. The purple dot is the site within the Yarraman Channel (source water). The locations of a low-level road crossing, small and big lagoons and a block bank that influenced the progression of flow down Thule Creek are shown.

regulation (Tuteja, 2008; State of New South Wales and Department of Planning, Industry and Environment, 2020). The method for determining the declined frequency of overbank flows in Koondrook-Perricoota Forest was based on a comparison between frequency of events (as defined by environmental water requirement in long term water plan) under modelled natural (without development) and modelled current condition flow regimes. Environmental water requirement achievement over the long term was evaluated through statistical analysis of modelled or observed flow records (State of New South Wales and Department of Planning, Industry and Environment, 2020). Koondrook-Perricoota Forest would have received more frequent flows under modelled natural regime, and thus Thule Creek would have received more frequent outflows from Koondrook-Perricoota Forest under natural flow conditions. Whereas Thule Creek only experienced flows once in the 5 years prior to this study (J. Dyer, personal communication, 4 August 2023). Through the use of an irrigation channel, environmental water has been delivered directly to Thule Creek to restore a component of the natural flow regime of this ephemeral system by reducing the duration of dry periods. The environmental flows delivered to Thule Creek are not continuous throughout the year, so thus there continues to be a wetting/drying regime in this ephemeral system.

Thule Creek and its associated fringing redgum wetlands areas provide a nursery and refuge habitat for a range of native fauna, particularly native fish and waterbirds that have significant cultural value in the area (Forestry Corporation of NSW, 2019). Thule Creek has been home to a relatively high diversity of small bodied native fish compared to other waterbodies in the area, including flathead gudgeon, carp gudgeon, Australian smelt, Murray rainbow fish and unspecked hardyhead (Gannon et al., 2019).

Ten sites were monitored for this study, including six in Thule Creek, three in the Wakool River, and one in Yarraman irrigation channel (source of environmental water) (Figure 2). Site 1 in Thule Creek was downstream of Perricoota Forest and upstream of the inflows of environmental water from Yarraman Channel. Sites 2 to 6 in Thule Creek were downstream of the environmental water delivery from Yarraman Channel. There is a small in-channel lagoon in Thule Creek between sites 2 and 3 and a larger in-channel lagoon in the creek between sites 3 and 4. Both of these lagoons were dry at the commencement of the environmental watering action. There was a sediment bank across Thule Creek between sites 5 and 6, that blocked the downstream flow of water until it was manually breached. In the Wakool River there was one monitoring site upstream of the Thule Creek confluence (Wakool upstream) and two sites in the Wakool River downstream of the Thule Creek confluence (Wakool downstream 1 and 2) (Figure 2).

2.2 Environmental watering action

In many river systems, environmental water is released from dams to meet the requirements of water-dependant ecosystems further downstream. In the case of Thule Creek, environmental flows were delivered to the creek via an escape from a nearby irrigation channel, Yarraman Channel. In September 2019 Forestry Corporation of NSW and the NSW Department of Planning, Industry and the Environment (DPIE) planned an environmental watering action to deliver water via Thule Creek to Wakool River from Yarraman Channel via the Murray Irrigation Limited (MIL) Thule Escape (Commonwealth Environmental Water Office, 2019; Forestry Corporation of NSW, 2019).

The environmental watering action commenced to Thule Creek on 3rd October 2019 and ceased on 9th January 2020 (98 days) to achieve connectivity to the Wakool River and maximise wetted area in the wetland sections of the creek. Environmental flows ranged between 30 and 38 ML/day which was the maximum capacity of the Thule Escape, and total environmental water delivery was 3528.99 ML. Prior to the commencement of the environmental watering action to Thule Creek in 2019-20 there was a small refuge pool of water remaining in Thule Creek near site 1 from an environmental watering action delivered from the Thule Escape in 2018-2019. This refuge pool (hereafter referred to as the Thule Escape refuge pool) extended from upstream of the Lower Thule Road Bridge to approximately 3 km downstream of the Thule Escape, but it did not extend as far as the smaller lagoon on Thule Creek that was dry. There was also a pool of water in the lower section of Thule Creek backing up from Wakool River to just upstream of the Wakool-Barham Road Bridge. The bed of Thule Creek between these two pools was dry at the commencement of the environmental watering action.

2.3 Hydrological and weather data

Daily discharge (ML/day) and water level (m) for automated hydrometric gauges were obtained from the New South Wales Office of Water website (https://realtimedata.waternsw.com.au/water.stm). The daily discharge data for Thule Creek at Lower Thule Road was from gauge 409109. For this gauge there was no data for discharge available from 1/10/2019 and it is not representative of the Thule Escape flows as it is upstream of the escape with no flow. This gauge does however measure the effect of the escape deliveries on creek heights, so water level data have been used to describe hydrology. Daily discharge data for the release of water from the Thule Escape from Yarraman Channel Canal to Thule Creek was obtained from the Murray Irrigation Limited.

The temperature at weather station 80023 at Kerang, Victoria for the study period was obtained from the Australian Government Bureau of Meteorology website (http://www.bom.gov.au/climate/averages/tables/cw_080023.shtml).

2.4 Analysis of travel time of environmental flow down Thule Creek

The timing of the environmental water flowing down Thule Creek was documented through the analysis of Sentinel-2 satellite imagery from 3rd October 2019 to 11th March 2020. Initially the watercourse was digitised using satellite imagery and Light Detection and Ranging (LiDAR) (Digital Elevation Model, DEM derived) datasets as reference. Due to the very narrow width of channel in some parts of Thule Creek it was not possible to use an automated classification procedure to analyse the presence of water in the imagery. A basic imagery classification was to identify water presence and the imagery was manually assessed to determine the presence of water on each image. The pixels sizes of downloaded images ranged between 14 and 15 m. The analysis of imagery was undertaken by the Spatial Data Analysis Network (SPAN) at Charles Sturt University.

2.5 Sample collection and laboratory analysis

Collection of water samples and spot measures of water quality was undertaken once per week over a period of 15 weeks between mid-October 2019 and the end of January 2020. On all sampling dates water quality parameters (temperature (°C), electrical conductivity (mS/cm), dissolved oxygen (mg/L), pH, and turbidity (NTU)) were measured as spot recordings using a hand-held Horiba U-50 multi-parameter water quality meter.

Two replicate water samples were collected at each site on each sampling occasion. Water samples could not be collected from some sites on all sampling occasions, because some of the sites were dry. Water quality parameters used to monitor the water quality responses to the environmental watering actions included dissolved organic carbon (DOC), nutrients (phosphorus and nitrogen were measured in their total forms (total phosphorus (TP) and total nitrogen (TN)) and also in the soluble/bioavailable forms (filterable reactive phosphorus (FRP), ammonia (NH_3) and nitrates + nitrites (NOx)) and Chlorophyll-*a* (Chl *a*).

Water samples for TP and TN were collected using pre-rinsed syringes into sterile jars. Water samples for DOC, FRP, NOx and NH_3 were filtered through 0.2-μm pore-size membrane syringe filters into sterile jars at the time of sampling. The water samples for chlorophyll analysis were vacuum filtered through GF/C filter papers (Whatman®) and these were immediately wrapped in aluminium foil. Two replicate water samples for each indicator were collected at each site per week. All collected samples were stored on ice until returned to the laboratory and were frozen for analysis later. All water samples then were sent to the National Association of Testing Authorities (NATA) accredited laboratory at CSIRO, Albury, for analysis, thereby ensuring the integrity of data and analysis procedures.

Water samples for TP and TN determination were analysed by simultaneous digestion ultraviolet spectrometry using an oxidising regent solution of $NaOH\text{-}K_2S_2O_8$ (Hosomi and Sudo, 1986). The analyses of NH_3 and NOx were undertaken by using flow injection analysis and DOC analysis was undertaken by using high-temperature conversion to CO_2 followed by infrared detection. Water samples for chlorophyll analysis were collected and filtered

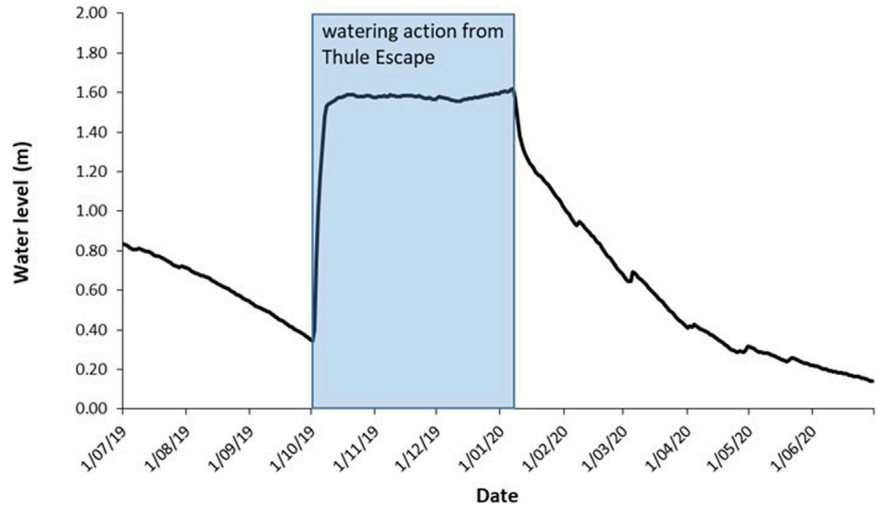

FIGURE 3
Water level (m) in Thule Creek at Lower Thule Road (gauge 409109) from 1 July 2019 to 30 June 2020. The shaded bar indicates the timing of the environmental watering action from Yarraman Channel via the Murray Irrigation Limited Thule Escape.

using the method proposed by APHA (American Public Health Association, 2005) and analysed using spectrophotometry.

The monitoring results were assessed against the trigger levels for aquatic ecosystems from the ANZECC (2000) water quality guidelines. If the concentration of a particular water quality parameter exceeded the trigger level or falls outside of the acceptable range, the guidelines are written with the intention that further investigation of the ecosystem is "triggered" to establish whether the concentrations are causing ecological harm. Systems may vary in their sensitivity to various parameters and therefore exceeding a trigger level is not an absolute indicator of ecological harm. It is quite common for water quality parameters to briefly fall outside of guideline values during periods of very high flow, this is not necessarily a sign of poor ecosystem health. The ANZECC water quality guidelines do not provide trigger levels for total organic carbon (TOC) and dissolved organic carbon (DOC), and this reflects the expectation that there will be large variation in the "normal" concentrations of organic carbon between ecosystems and also in the chemical and biological reactivity of the mixture of organic compounds making up the DOC and TOC at a particular site. Given the variable make-up of organic carbon, and the possible range of ecological responses to this mixture, a trigger level for this parameter would not be appropriate. However, trigger levels are provided for a number of nutrients and these are discussed below.

2.6 Data analysis

The influence of outflows from Thule Creek on the dissolved organic carbon and nutrients flowing into Wakool River was assessed by two qualitative comparison approaches, including the comparison of time series within each site and the comparison of sites on each sampling occasion in the Wakool River upstream and downstream of the Thule confluence.

3 Results

3.1 Hydrology

The water level in Thule Creek at site 1 at Lower Thule Road (gauge 409109), shows that there was water retained in the Thule Escape refuge pool prior to the commencement of the environmental watering action. The pool was slowly drying down as the air temperature increased during August and September 2019 (Figure 3). Environmental water released from Yarraman Channel from the Thule Escape to Thule Creek commenced on 3rd October 2019 and ceased on 9th January 2020. Water delivered from Yarraman Channel to Thule Creek was approximately 37 ML/day and was consistent at that rate for 98 days. On 3rd October 2019 when the watering action commenced, the water level at the gauge increased from 0.40 m to approximately 1.58 m and remained at this level until the watering action ceased and the pool started to slowly dry down (Figure 3). Between 1st September 2019 and 29th February 2020, the discharge in Wakool River at Wakool-Barham Road (gauge 409045) was average 332 ML/day (minimum 212 ML/day and maximum 482 ML/day).

3.2 Air temperature

The daily maximum air temperature at Kerang, Victoria (weather station 80023) during the study period ranged from 16.3°C on 1st December to 46.6°C on 20th December 2019 (Figure 4).

3.3 Travel time of environmental flows down Thule Creek

The analysis of Sentinel-2 imagery showed that the environmental water took more than 80 days to travel approximately 11 km down the Thule Creek channel and reach the junction with Wakool River (Figure 5).

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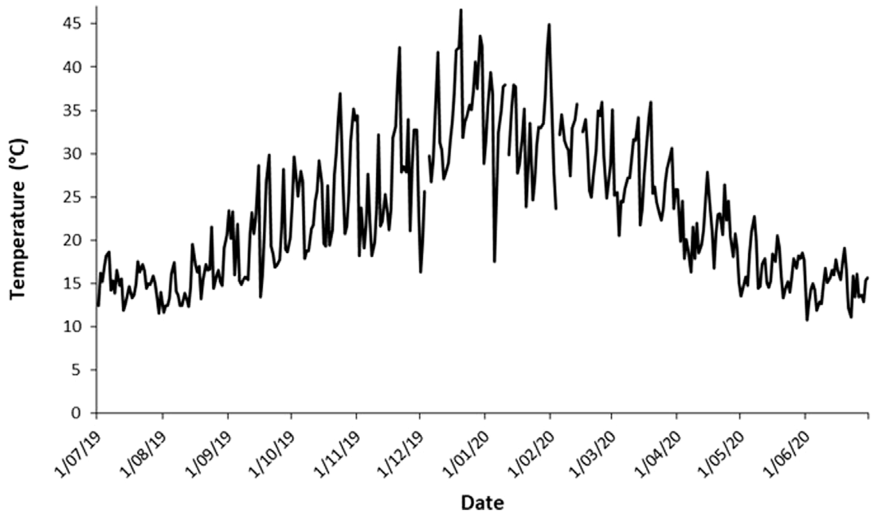

FIGURE 4
Daily maximum air temperature at Kerang, Victoria (weather station 80023) from 1 July 2019 to 30 June 2020 (Bureau of Meteorology, 2020).

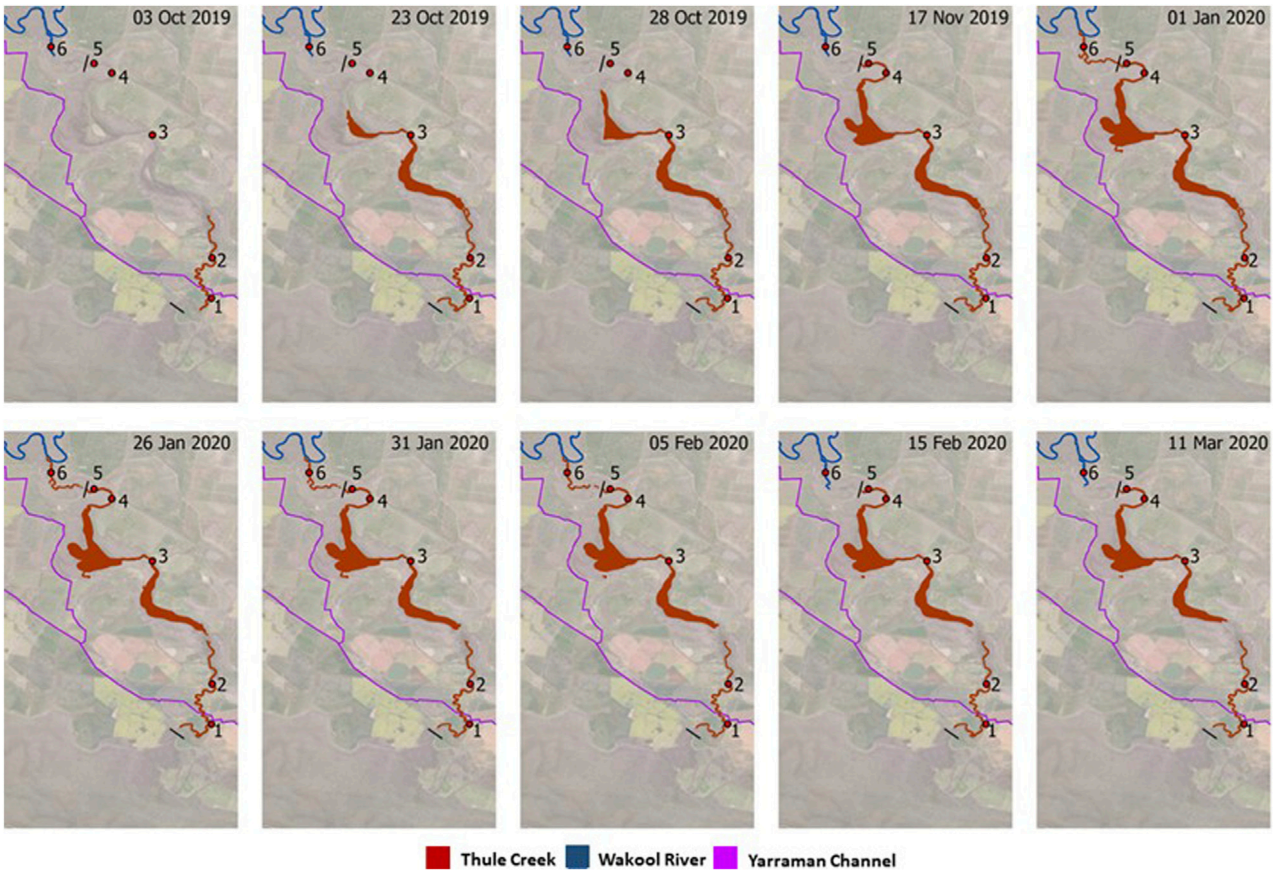

FIGURE 5
Annotated Sentinel-2 images showing changes in the presence of water in Thule Creek on selected dates from 3 October 2019 to 11 March 2020. Numbers indicate the location of six monitoring sites along Thule Creek. The blocked bank at site 5 influenced the flow of water down Thule Creek.

FIGURE 6
Left: Water was blocked by a bank in Thule Creek. Right: Water flowing over the blocked bank on 14th December 2019 after it was manually breached on 13th December 2019. (Photos Xiaoying Liu).

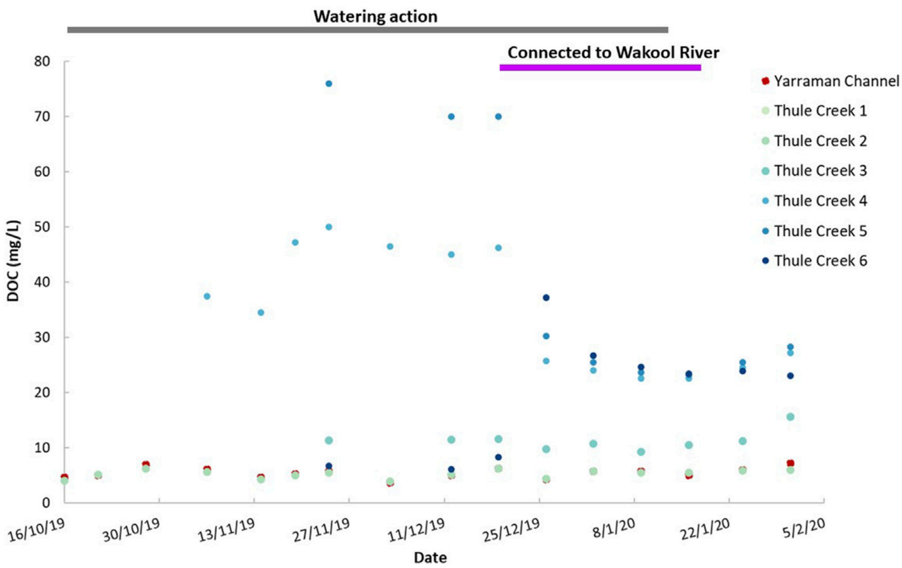

FIGURE 7
Dissolved organic carbon (DOC, mg/L) concentrations at all study sites in Thule Creek and Yarraman Channel from mid-October 2019 to the end of January 2020. The bold grey line indicates when environmental watering commenced and ceased, and the purple line indicates the period when Thule Creek was connected with the Wakool River.

At the start of the watering action there was a remnant pool in Thule Creek near the escape (site 1), but the two in-channel lagoons downstream of the escape were dry. It took approximately 2 weeks for the water to reach and fill the smaller lagoon between sites 2 and 3 and approximately four more weeks to fill the larger second lagoon between sites 3 and 4. The progress of the water downstream to the Wakool River was obstructed by a block bank at site 5 for approximately a month until the bank was manually breached on 13th December 2019 to allow the flow to continue to flow downstream and connect with Wakool River (Figure 6). Approximately 1 week later, the environmental water connected with the Wakool River, between 19th and 26th December 2019. The real-time connection was not included in the Sentinel-2 images (Figure 5) due to extensive cloud cover making water detection impossible in images at that time. The water in Thule Creek started to dry back after the watering action ceased on 9th January 2020. Sentinel-2 imagery viewed on 1st August 2020 showed that water was retained in the two lagoons and in the refuge pool near the Thule Escape.

3.4 Dissolved organic carbon and nutrients in Thule Creek during and after the environmental watering action

The water in Yarraman Channel had DOC concentration between 4 and 5 mg/L throughout the study, which is in the

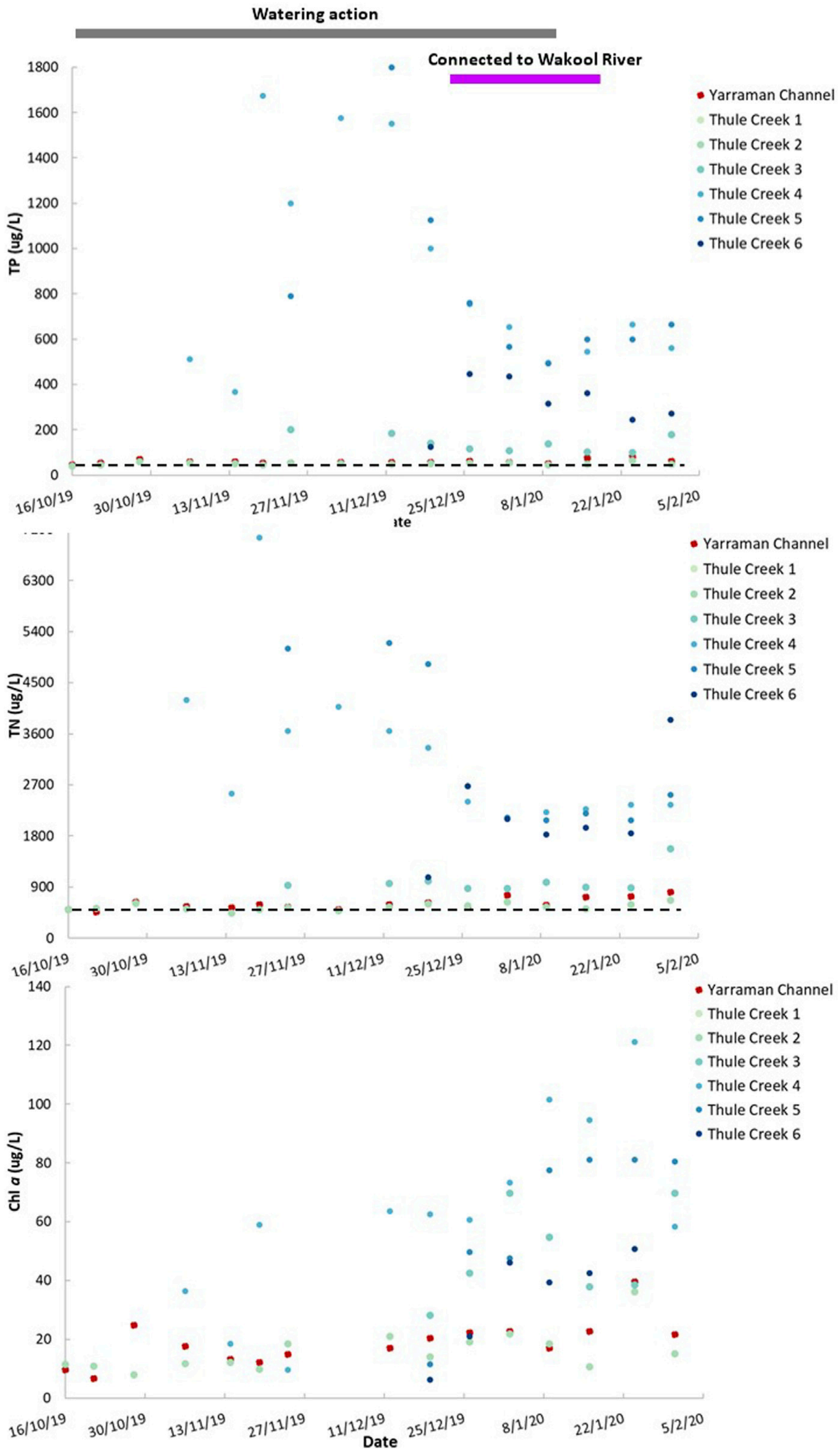

FIGURE 8
Total phosphorus (TP), total nitrogen (TN) and chlorophyll *a* (Chl *a*) concentrations (μg/L) at sites in Thule Creek and Yarraman Channel from mid-October 2019 to the end of January 2020. The bold grey line indicates when environmental watering commenced and ceased, the purple line indicates the period when Thule Creek was connected with the Wakool River, and the black dashed lines indicate ANZECC trigger values for TP and TN.

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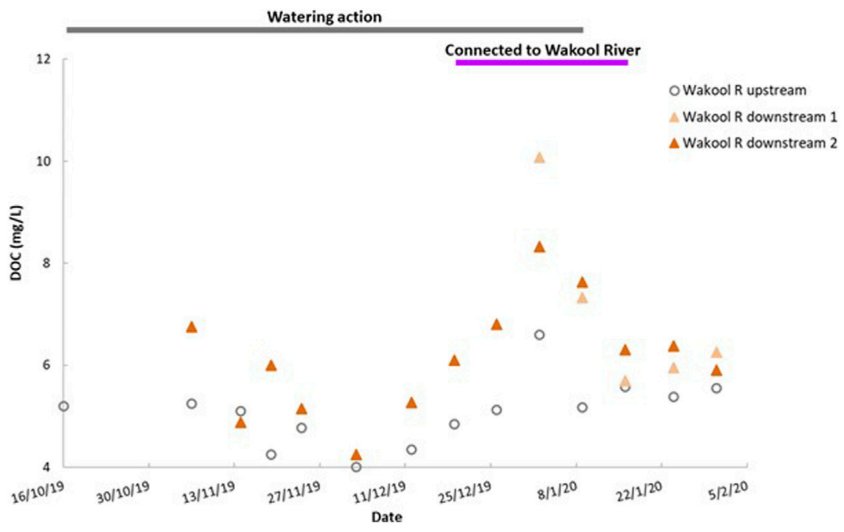

FIGURE 9
Dissolved organic carbon (DOC, mg/L) concentrations at sites in Wakool River from mid-October 2019 to the end of January 2020. The bold grey line indicates when environmental watering commenced and ceased, and the purple line indicates the period when Thule Creek was connected with the Wakool River.

normal range observed in this system (Watts et al., 2019). The water released from the Thule Escape increased in DOC concentration as it travelled downstream in Thule Creek (Figure 7). The concentration of DOC in water samples collected from sites 1 and 2 was similar to that from the Yarraman Channel. At sites 3, 4, and 5 the water inundated shallow areas of small red gum trees, grass, leaf litter and bare soil (Supplementary Figure S1). At site 3, downstream of the smaller in-channel lagoon, the concentration of DOC increased to 10–12 mg/L. The concentration of DOC increased to about 50 mg/L at site 4 after the water travelled through the larger in-channel lagoon. At site 5 it increased to concentration between 70 and 80 mg/L after the water had pooled in the creek for several weeks behind the blocked bank (Figure 7).

The block bank between Thule Creek sites 5 and 6 was breached on 13th December 2019 enabling the water in Thule Creek to flow downstream through the dry creek bed that had a thick cover of leaf litter and sticks which had accumulated for 3 years since the last major flow event (Supplementary Figure S2). The soil was very dry and the leading edge of the flow progressed slowly downstream. The colour of the water at the leading edge of the flow was extremely dark (Supplementary Figure S3, top), suggesting it contained a very high concentration of DOC.

After the water commenced to flow downstream the concentration of DOC at sites 4 and 5 reduced to between 30 and 40 mg/L and the colour of the water became lighter (Supplementary Figure S3, bottom). At the end of December 2019 when the Thule Creek flow connected with the Wakool River, the concentration of DOC at Thule Creek site 6 increased and was similar to Thule Creek sites 4 and 5 (Figure 7). On 26th January 2020 when the water ceased to flow downstream and the air temperature was over 35°C, the concentration of DOC at sites 3, 4, and 5 started to increase (Figure 7).

Similar to the patterns of DOC concentration, the water released from the MIL Thule Escape increased in concentration of nutrients as it travelled downstream. There were particularly high concentrations of nutrients at sites 4 and 5 (Figure 8) after the water had travelled through the larger lagoon and resided in the system for several weeks behind the block bank.

Total phosphorus (TP, 45–86 μg/L) was at low concentrations in the source water from Yarraman Channel. TP concentrations were elevated at downstream sites of Thule Creek during the watering action, suggesting a considerable amount of phosphorous was leached from soil, leaf litter and organic matter as the water flowed down the Thule Creek channel. Most of the TP was released between site 4 (368–1,675 μg/L) and site 5 (493–1,800 μg/L) and highest TP concentration was measured at site 5 on 19th November 2019 with a concentration of 1800 μg/L (Figure 8). The pattern observed for TN was similar to TP. The total nitrogen concentration (TN, 465–815 μg/L) was low in the source water from Yarraman Channel. The majority of TN was leached between site 4 (2,125–7,050 μg/L) and site 5 (2,075–5,200 μg/L) (Figure 8). During and after the watering action, TP and TN concentrations at most of the study sites were above the ANZECC trigger value of 50 μg/L and 500 μg/L respectively.

Chlorophyll *a* (Chl *a*) concentrations were closely associated with nutrient concentrations. At sites 4 and 5 concentrations of Chl *a* were considerably higher than all other sites in the study area and increased over time as water temperature was increased over December 2019 and January 2020 (Figure 8).

3.5 Dissolved organic carbon and nutrients in the Wakool River during and after connection with outflow of Thule Creek

The outflow of Thule Creek connected with the Wakool River water between 19th and 26th December 2019, resulting in a carbon-rich pulse detected in the Wakool River system (Figure 9). The

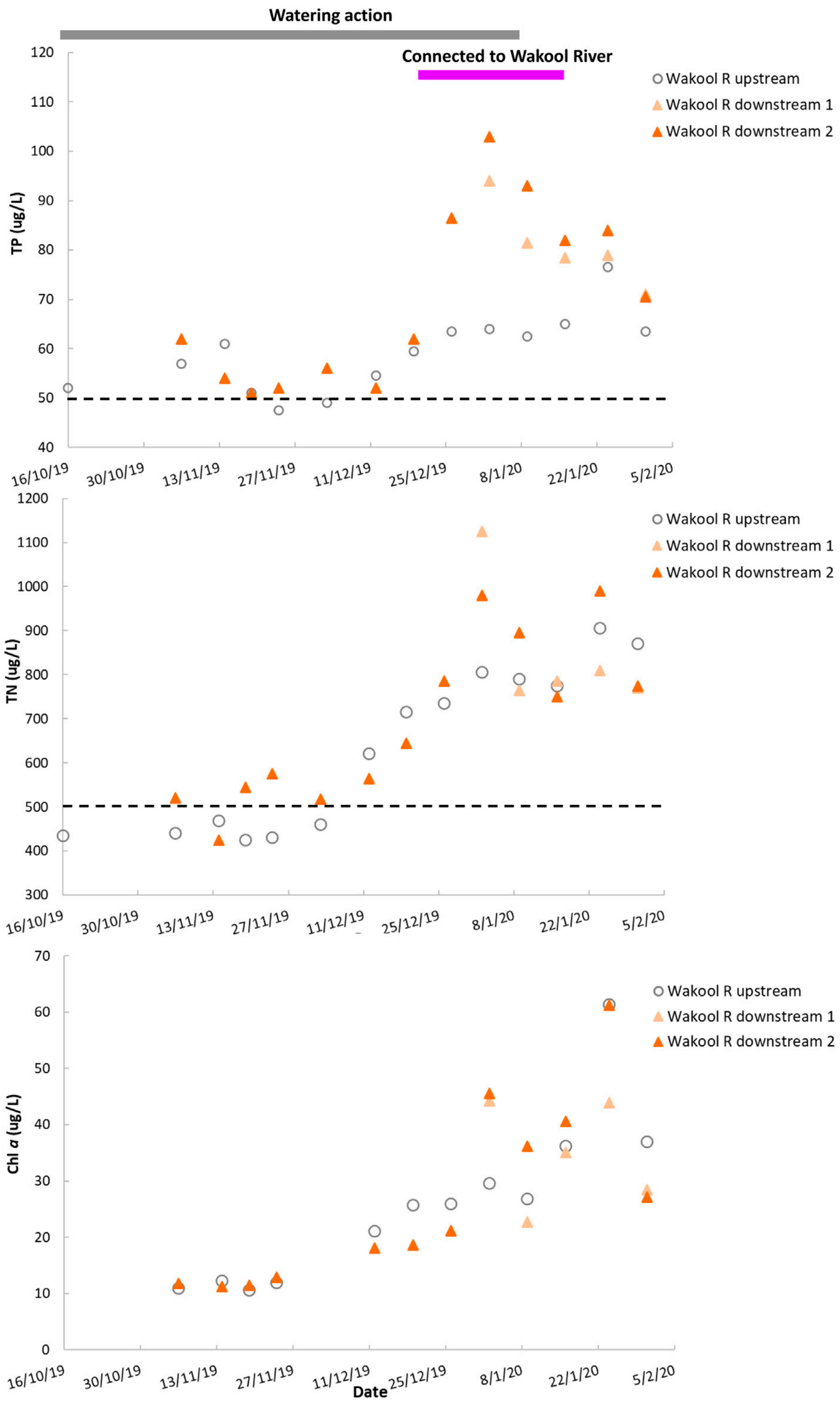

FIGURE 10
Total phosphorus (TP), total nitrogen (TN) and chlorophyll *a* (Chl *a*) concentrations (µg/L) at sites in Wakool River from mid-October 2019 to the end of January 2020. The bold grey line indicates when environmental watering commenced and ceased, the purple line indicates the period when Thule Creek was connected with the Wakool River, and the black dashed lines indicate ANZECC trigger values for TP and TN.

TABLE 1 Range and mean values of water physico-chemical parameters measured using handheld water quality meter for all study sites in Yarraman Channel, Thule Creek and Wakool River from mid-October 2019 to the end of January 2020. Mean values are shown in brackets.

	Yarraman channel	Thule site 1	Thule site 2	Thule site 3	Thule site 4	Thule site 5	Thule site 6	Wakool upstream	Wakool downstream 1	Wakool downstream 2
Temperature (°C)	15.8–32.5 (22.7)	17.3–32.2 (19.4)	15.5–31.0 (21.4)	17.2–30.8 (23.0)	18.9–30.8 (22.4)	16.2–32.4 (22.0)	19.2–31.0 (21.8)	17.7–26.7 (20.7)	19.1–26.4 (21.9)	16.8–27.5 (20.6)
DO (mg/L)	7.3–11.3 (9.4)	5.6–10.3 (7.4)	6.5–11.2 (8.4)	1.4–4.1 (2.7)	1.2–6.7 (3.1)	0.4–4.3 (1.8)	3.0–9.8 (7.2)	6.0–9.2 (8.0)	4.6–7.7 (6.1)	4.6–10.6 (8.3)
pH **6.5–8**	6.6–8.3 (7.6)	6.7–7.9 (7.3)	6.7–7.8 (7.3)	6.3–7.6 (7.0)	6.4–7.8 (7.3)	6.4–7.8 (7.2)	6.9–8.0 (7.4)	6.5–8.2 (7.4)	6.4–7.6 (6.7)	6.4–7.7 (7.0)
Turbidity (NTU) **50**	71–233 (134)	12–42 (36)	13–116 (82)	6–50 (10)	7–34 (26)	11–26 (18)	40–127 (78)	75–125 (91)	96–141 (123)	66–161 (95)
EC (mS/cm) **0.125**	0.042–0.080 (0.055)	0.051–0.079 (0.067)	0.045–0.067 (0.058)	0.065–0.112 (0.083)	0.150–0.250 (0.177)	0.158–0.312 (0.185)	0.060–0.221 (0.165)	0.039–0.068 (0.056)	0.061–0.075 (0.065)	0.039–0.074 (0.055)

ANZECC (2000) trigger levels for available water parameters are in bold font. Values exceeding ANZECC trigger levels at some sites are highlighted in blue.

concentration of DOC in the Wakool River downstream of Thule Creek increased in early January 2020 when the DOC-rich water from Thule Creek was flowing into Wakool River. DOC in the Wakool River downstream of Thule Creek, Wakool River downstream site 1 was 10.1 mg/L and Wakool River downstream site 2 was 8.3 mg/L compared to the site in the Wakool River upstream of Thule Creek confluence with 6.6 mg/L on 2nd January 2020. On 9th January the DOC concentration at Wakool River downstream site 1 (7.3 mg/L) and Wakool River downstream site 2 (7.6 mg/L) continued to be higher than that observed at Wakool River upstream site (5.2 mg/L).

There was a noticeable difference in the colour of the carbon-rich Thule Creek water and the turbid Wakool River water at the junction of these two systems (Supplementary Figure S4). On 16th January there was no difference in DOC concentration between the upstream and downstream Wakool River sites, as flows from Thule Creek to Wakool River had stopped flowing.

The flow in Thule Creek connected with Wakool River resulting in a nutrient-rich pulse detected in the Wakool River downstream of Thule Creek compared to the Wakool River upstream of Thule Creek (Figure 10). For example, on 2nd January 2020, TP and TN at the Wakool River downstream site 1 of Thule Creek confluence were 94 µg/L and 1,125 µg/L at Wakool River downstream site 2 were 103 µg/L and 980 µg/L compared to the Wakool River site upstream of Thule Creek with 64 µg/L and 805 µg/L. Higher Chl *a* levels were measured at the Wakool River downstream site 1 of Thule Creek with 44.2 µg/L and Wakool River downstream site 2 with 45.6 µg/L, compared to the Wakool River site upstream of Thule Creek with 29.6 µg/L when flow from Thule Creek connected with Wakool River.

There was minimal difference in nutrient concentrations between the upstream and downstream Wakool River sites on 16th January 2020, as flows from Thule Creek to Wakool River had stopped flowing.

3.6 Dissolved oxygen

Spot dissolved oxygen (DO) measures were taken at different times of the day. The time of day will affect DO readings: early morning spot measures of DO are often lower than measurements taken at the same site later in the day. While these measurements do not provide the same high resolution indication of changes with time, the data presented here provide an indication of the extent of hypoxia at each site at the time of sampling.

The DO concentration was less than 4 mg/L (threshold for fish stress, Gehrke, 1988) between sites 3 and site 6 in Thule Creek during the environmental watering action. At times the concentration of DO at sites 4 and 5 dropped into the range of lethal to fish populations (below 2 mg/L) (Table 1). On 8th January 2020 before the environmental watering action ceased, at most sites the DO concentrations were above 4 mg/L and DO levels at sites 3 and 4 were above 2 mg/L which were a little higher (better) than observed in previous weeks. However, after the Thule Creek ceased to flow on 9th January 2020 and air temperatures were very high, the DO levels at sites 3, 4, and 5 were below 2 mg/L again.

In general, a 3–4°C increase in water temperature was observed at all sites on 12th December 2019 coinciding with the time when rapid onset of hypoxia was observed. Temperatures at all sites were similar although wider daily fluctuations were recorded at sites in Thule Creek than at the sites in the Wakool River. This may have been due to the shallower water depth at the sites in Thule Creek.

4 Discussion

4.1 Outcomes of delivery of environmental water from an irrigation canal to Thule Creek

The inundation of a range of floodplain land uses (including redgum forests, crops, pastures and bare soil) during floods can be a major source of DOC and nutrients and a major contributor to dissolved oxygen depletion in rivers (Liu et al., 2019). Drying of sediments followed by re-wetting events in inland ephemeral systems can lead to a flush of nutrients (such as phosphorus and nitrogen) released to the water column (Twinch, 1987; De Groot and Van Wijck, 1993; Qiu and McComb, 1994; Mitchell and Baldwin, 1998; Liu, 2017). The results of detected flushes of DOC and nutrients in the Thule Creek channel support our hypothesis that a considerable amount of carbon and nutrients would be leached into the water column as during the environmental flow as the water flowed down Thule Creek.

Previously studies have shown that large amounts of organic matter accumulated in the bed of ephemeral rivers in the period preceding inundation can be mobilised by the high flow events (Sabo et al., 1999; Junk and Wantzen, 2004; Kobayashi et al., 2009). In our study we observed that a substantial amount of organic matter had accumulated on the bed of Thule Creek over the preceding years the creek had been dry. As sites 3 to 6 in Thule Creek (the lower part of the creek) had not been flooded or inundated since 2016, substantial inputs of DOC and nutrients derived from small red gum trees, grass, leaf litter and bare soils were released to the water column during the delivery of environmental water. For example, although the DOC concentration in the water released from Yarraman Channel was in the normal range observed in this system (Watts et al., 2019), the DOC concentration increased considerably after the water had travelled downstream and inundated shallow areas within in the Thule Creek channel.

A study conducted to examine the contribution of water delivery through Koondrook-Perricoota Forest to the productivity of Wakool River in spring of 2016 (Watts et al., 2017b) recorded DOC values ranging from 10 to 32.5 mg/L in Thule Creek. The carbon and nutrients concentrations observed in current study were even more elevated compared to the results of the 2016 study, but were similar to results of a glasshouse experiment where DOC was examined following inundation of soil and vegetation from a lowland river floodplain (Liu et al., 2019), simulating the leaching of carbon in standing water on a floodplain. Both the results of the glasshouse experiment and the current study suggest that high concentrations of DOC can develop when water flow is standing on the floodplain or, in the case of the current study, held up behind a block bank that obstructed the flow in the creek. In contrast, when the water is flowing through a system the DOC can increase but not to the same extent as in standing water.

Blackwater occurs when large quantities of organic material (carbon-based substances), such as sticks, leaves, grass or crops are washed off the floodplain and into rivers and creeks resulting in a dark tea color in the water. Blackwater becomes hypoxic (low dissolved oxygen concentration) when large amounts of organic material in rivers are broken down by bacteria, consuming dissolved oxygen in the water (Department of Climate Change, Energy, the Environment and Water, 2023). The likelihood of the occurrence of hypoxic blackwater events is increased if a large volume of organic matter has built-up on the floodplain or water temperatures are high. The high water temperatures at the time of this study may have accelerated respiration beyond the capacity of the system to maintain DO in the water column, combined with lower solubility at higher temperatures (Howitt et al., 2007; Kerr et al., 2013; Whitworth et al., 2014; Pasco et al., 2016). DO concentrations below the 2 mg/L threshold associated with fish deaths (Gehrke, 1988) occurred only at sites where water was held up behind the block bank with high DOC concentrations. Thus, the poor water quality was only in a relatively small section of Thule Creek for a brief period during very hot weather conditions. A study of fish monitoring was done in the refuge pool near site 1 prior to this study and it has documented fish community in Thule Creek (Gannon et al., 2019) and the landholder who owned the property at sites 3 and 4 had noted fish moving around in the water in Thule Creek because the water was clear and fish could be seen through the water. Despite the high DOC concentration and hot weather, the researchers and landholders did not observe any dead fish in Thule Creek at any time during the study.

After the blocked bank was breached and water started to flow downstream to Wakool River, the DO concentrations at most sites were a little higher than in the previous weeks, possibly because the blackwater was diluted by the canal source water. Furthermore, there may have been less carbon leaching out of the soil and grasses as the creek had already been wet for several weeks. Thus, the risk to fish was localised in the vicinity of Thule site 5 and the increased connectivity from the flow action provided fish with the opportunity to move upstream to a section of the creek that had higher DO concentration near the Thule Escape, or downstream to higher DO water in Wakool River.

4.2 Contribution of delivery of environmental water via an ephemeral system to Wakool River

In the current study, the carbon-rich and nutrient-rich water was detected in the Wakool River downstream of Thule Creek after the environmental flow in Thule Creek connected with Wakool River. The discharge in Wakool River was sufficient to dilute the inputs from Thule Creek in hot weather, so there was no evidence of low DO in Wakool River that would increase the risk of the development of hypoxia or fish deaths. Excessive inputs of nutrients and organic carbon can result in poor water quality with the development of algal blooms or hypoxic blackwater events resulting in very low DO concentrations (Howitt et al., 2007; Hladyz et al., 2011). Particularly during hot weather, the temperature affects the rates of microbial processes and organic matter leaching (Howitt et al., 2007; Whitworth et al., 2014). In 2016 there was a large flood in the southern Murray-Darling Basin that resulted in hypoxic blackwater events that extended throughout the Murray River system, including the Edward/Kolety-Wakool River system. Fish kills were reported in many areas with very low DO levels because large amounts of floodplain-derived carbon and nutrients were present in the water throughout the system (Watts et al., 2017a; 2018). The main issue with that large hypoxic event was that it was extremely widespread throughout the Murray River system and there were very few refuges having higher DO concentration that fish could migrate to.

In contrast, during the environmental watering action in 2019, only a short length of Thule Creek experienced high DOC and low DO concentrations, thus fish and other aquatic organisms had the opportunity to move upstream or downstream to areas where there was water with higher concentration of DO. This suggests that when delivering environmental water to create pulses of carbon to improve productivity it is essential to maintain connectivity with other parts of the system to allow fish and other aquatic organisms to move to reaches having better water quality.

Inputs of nutrients and organic carbon can have a positive influence on river systems through the stimulation of productivity and increased food availability for downstream (Robertson et al., 1999). Studies have shown that the connection between a river and its floodplain can create essential carbon stores to sustain the system through drier periods (Baldwin et al., 2013). Our study indicates that the delivery of environmental flows via the Thule Creek resulted in the addition of nutrients and organic carbon that could support microbial productivity and create food for aquatic organisms such as fish in the Wakool River.

The delivery of environmental flows has been shown to provide ecological benefits to permanent river systems and associated wetlands. Environmental flows to permanent river systems have been delivered to help prevent loss of critical taxa from droughts by creating refuges (Rayner et al., 2009), to promote hydrological connection and primary productivity (Chester and Norris, 2006), support spawning activity and recruitment of fish (King et al., 2010), sustain breeding of colonial waterbirds (Kingsford and Auld, 2005) and flush excessive growth of nuisance biofilms (Watts et al., 2010) by creating pulsed flows or flood events. Our study shows that a small discharge of environmental water delivered to a previously dry ephemeral river can result in the release of carbon and nutrients from leaves, grass and bare soils from the ephemeral waterway and influence the productivity of a receiving permanent river system. This possibly achieved a greater response than could have been achieved by delivering the same small amount of environmental water to a permanently river.

Our study suggests that careful planning of timing, magnitude and duration of delivery of environmental water through Thule Creek could accelerate ecosystem processes and increase the input of floodplain-derived carbon and nutrients to the Wakool River when it is under regulated flows, but at the same time minimise the risk of hypoxia, particularly in hot weather.

4.3 Other ecological benefits of the environmental watering action in Thule Creek

Environmental flows to Thule Creek also provided other visually assessed ecological benefits. Apart from facilitating nutrient and carbon transport between Thule Creek and the Wakool River there were several other ecosystem responses observed during the watering action. Along Thule Creek channel, the water was very clear with low turbidity, and there was a large amount of productivity. There were a high abundance of aquatic plants emerging on the riverbed that could be seen through the clear water. In amongst the plants there a high abundance of aquatic invertebrates observed in the shallow edges of the creek, suggesting it was creating a pulse of “fish food” as the flow moved downstream. The response of the riparian vegetation along the creek was also very notable, with trees adding a thick cover of new leaves. In addition, a lot of waterbirds were observed including pelicans, darters and cormorants in Thule Creek, and there were over 50 colonial waterbird nests observed by researchers, including darters nesting at the small lagoon and little black cormorants nesting in thick red gum regrowth between the two lagoon sites. Thus, the delivery of the environmental water to Thule Creek was visually observed to improve the condition of riparian vegetation, stimulate emergence and growth of aquatic plants, increase aquatic invertebrate activity, and support waterbird feeding and nesting in the Thule Creek channel. There were no fish kills observed during the environmental flow, and fish were observed moving around at sites in the creek.

4.4 Using irrigation infrastructure to deliver environmental flows to ephemeral rivers

In the Edward/Kolety-Wakool River system, the presence of an extensive irrigation canal network infrastructure provides an opportunity to deliver environmental water to ephemeral creeks. In this study the environmental flow from Yarraman Channel via the Murray Irrigation Limited (MIL) Thule Escape continued for 98 days and it took more than 80 days for the water to travel down Thule Creek and reach the junction with Wakool River. The reason the flow took a long time to reach the Wakool River was due to the very flat landscape, small-scale environmental watering action, some of the water soaking into the dry creek bed along the way, and hot weather resulting in high rate of water evaporation. Despite the long travel time, the environmental flow in Thule Creek did reach the Wakool River and created a small pulse of carbon and nutrients, contributing to the productivity of that system.

In May/June 2019 an upgrade to the MIL Thule Escape was completed, increasing the discharge that can be released from 30 ML/day to 120 ML/day. This would considerably reduce travel times between the Thule Escape and the Wakool River, reducing time for hypoxia to develop. The upgrade on MIL infrastructure to increase the water delivery capacity provides an opportunity to increase the discharge in Thule Creek of future environmental watering actions, thus increasing the opportunity for water managers to manage water deliveries to increase carbon inputs to stimulate productivity and at the same time avoid the risk of hypoxia. Further flow trials could be undertaken under different conditions to improve our understanding of how to balance the flows from Koondrook-Perricoota Forest regulator and the MIL Thule Escape with flows in Wakool River to achieve the best outcomes for the river ecosystem.

One challenge in many arid or semi-arid landscapes is that there will be limited options to deliver environmental water to ephemeral systems (Hughes, 2005). It may not be possible to create an environmental flow of sufficient discharge in permanent river systems that will connect to ephemeral or intermittent tributaries. The use of irrigation canal network infrastructure is an option that can be explored and trialled elsewhere worldwide.

This study focussed on hydrology and water chemistry outcomes of the environmental flow, combined with a visual assessment of ecological benefits. Future similar studies would be benefit from a more comprehensive integrated evaluation, including foodweb and ecological responses to environmental flows, to extend our knowledge of ephemeral river systems.

5 Conclusion

Due to the high evaporation rates of ephemeral systems in semi-arid and/or arid areas, careful management of the timing, volume and duration of environmental flows is required to avoid the development of poor water quality during, or following, the delivery of environmental water. This research provides an example of how an environmental flow delivered via an irrigation canal network infrastructure provided ecosystem benefits to an ephemeral creek and also contributed to productivity of a permanent river further downstream. The flow in the Wakool River at the time was sufficient to dilute the inputs from Thule Creek so at no time was there any risk of hypoxia or fish deaths. In general, the outcome of this small blackwater pulse was positive rather than negative. The small pulse increased hydrological connectivity and also improved vegetation condition including fringing vegetation and emergent/submerged aquatic plants in Thule Creek. Despite there being some sections of the creek that had low DO levels, no fish deaths were observed in Thule Creek. The pulse also provided opportunities for movement, reproduction and recruitment of invertebrates, frogs and native fish and other propagules.

Historically, before river regulation and agricultural modifications to the landscape, ephemeral systems would have flowed during large unregulated flow events. However, many of these ephemeral systems are now disconnected for extended periods of time from the rivers and permanent creeks that would have once supplied them with water and, without environmental water, would be dry in most years. In many landscapes there will be limited options to deliver environmental water to ephemeral systems. This research provides an example of the important contribution of delivering environmental water from irrigation canal network infrastructures toward promoting instream productivity in ephemeral creeks and permanent rivers and preventing loss of critical taxa. Water delivery options should be explored to find ways to increase river-floodplain connectivity and increase river productivity through the input of small pulses of dissolved inorganic and organic matter as seen during this study.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary Material, further inquiries can be directed to the corresponding author.

Author contributions

XL, RW, and JD contributed to design of the study. XL wrote the first draft of the manuscript. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2024.1213001/full#supplementary-material

SUPPLEMENTARY FIGURE S1
Inundated vegetation and organic matter in Thule Creek during delivery of environmental water. (Photos Xiaoying Liu).

SUPPLEMENTARY FIGURE S2
Leading edge of the water flowing downstream of site 5 block bank on 14th December 2019 (left) and 19th December 2019 (right). (Photos Xiaoying Liu).

SUPPLEMENTARY FIGURE S3
Water samples collected on 19th December 2019 (top) and 8th January 2020 (bottom), showing color differences of sampling sites on two dates during the environmental watering action. (Photos Xiaoying Liu).

SUPPLEMENTARY FIGURE S4
Photo taken at the junction of Thule Creek and Wakool River on 2nd January 2020. The darker coloured carbon rich water from Thule Creek on the left of the photo is flowing into the turbid water in the Wakool River on the right. (Photo Xiaoying Liu).

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A dynamic surface water extent service for Africa developed through continental-scale collaboration

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Spatially explicit, near real time information on surface water dynamics is critical for understanding changes in water resources, and for long-term water security planning. The distribution of surface water across the African continent since 1984 and updated as every new Landsat scene becomes available is presented here, and validated for the continent for the first time. We applied the Water Observations from Space (WOfS) algorithm, developed and well-tested in Australia, to every Landsat scene acquired over Africa since the mid 1980s to provide spatial information on surface water dynamics over the past 30+ years. We assessed the accuracy of WOfS using aerial and satellite imagery. Four regional geospatial organisations, coordinated through the Digital Earth Africa Product Development Task Team, conducted the validation campaign and provided both the regional expertise and experience required for a continental-scale validation effort. We assessed whether the point was wet, dry, or cloud covered, for each of the 12 months in 2018, resulting in 34,800 labelled observations. As waterbodies larger than 100 km^2 are easy to identify with Landsat resolution data and can thus boost accuracy, these were masked out. The resulting overall accuracy of the water classification was 82%. WOfS in Africa is expected to be used by ministries and departments of agriculture and water across the continent, by international organisations, academia, and the private sector. A large-scale collaborative effort, which included regional and technical skills spanning two continents was required to create a service that is regionally accurate and is both hosted on, and implemented operationally from, the African continent.

KEYWORDS

surface water, waterbodies, water security, co-production, Landsat, wetlands

1 Introduction

In the coming century, Africa is projected to see an increase in precipitation variability with both wetter and drier extremes (Gan et al., 2016; Nicholson, 2017; Barry et al., 2018), resulting in an increase in the length and severity of agricultural and ecological droughts as well as the frequency and intensity of floods (Milly et al., 2005; de Wit and Stankiewicz, 2006; Gan et al., 2016). High water stress is currently estimated to affect around 250 million people in Africa (World Metereological Organization, 2021); further changes to the hydro-climate will exacerbate water security issues (Boko et al., 2007; Leal Filho et al., 2021).

The 2030 Agenda for Sustainable Development adopted in 2015 called on governments and stakeholders to ensure the availability and sustainable management of water and sanitation for all (United Nations, 2015). Within the 17 Sustainable Development Goals (SDGs), Goal six includes, but importantly also goes beyond, drinking water, sanitation, and hygiene, to address the quality and sustainable use of water resources, as well as the protection and restoration of water-related ecosystems. The African Ministers' Council on Water (AMCOW) whose mission is to provide political leadership, policy direction and advocacy in the provision, use and management of water resources for sustainable social and economic development and maintenance of African ecosystems, works towards a vision of an Africa where there is equitable and sustainable use and management of water resources for poverty alleviation, socio-economic development, regional co-operation and the environment (AMCOW Initiative, 2023). But a recent assessment found that four out of five African countries are unlikely to have sustainably managed water resources by 2030 (World Metereological Organization, 2021). Strategies to support water resources management decisions require up-to-date information as well as long-term archival data on surface water dynamics.

Digital Earth Africa (DE Africa) addresses these data needs by processing openly accessible and freely available satellite data to produce demand driven, decision-ready products and services. Guided by a Governing Board co-chaired and represented by African ministers, advised by a Technical Advisory Committee (TAC) with the majority of its members based in Africa, and working closely with the AfriGEO community enables the program to respond to the information needs, challenges, and priorities, of partners across the African continent. In 2022, DE Africa transitioned out of an establishment phase into a distributed network of implementing partners across Africa with a program management office in South Africa (SANSA).

Development of operational data and services prioritised by the DE Africa governing bodies in response to end user demand, are created using cloud-native data processing tools from the archive of satellite data captured over Africa since 1984. Powered by Open Data Cube (ODC) technology, (Dhu et al., 2017; Lewis et al., 2017; Killough, 2018)], the platform, which runs on Amazon Web Services (AWS), currently archives over 3.5 petabytes of data including satellite imagery from the Landsat Collection 2 Level-2 in Cape Town, South Africa. The architecture implements AWS Elastic Kubernetes Service to enable the use of thousands of parallel processes on hundreds of servers enabling transferability and scalability of workflows exploiting decades of satellite images across the continent.

The DE Africa Water Observations from Space (WOfS) service, presented here, exploits this infrastructure. Based on an algorithm developed to provide a nationally consistent service for understanding surface water dynamics in Australia (Mueller et al., 2016), WOfS uses the entire Landsat satellite archive over Africa to obtain multiple surface water observations each month across decades of data. WOfS was developed on the Digital Earth Australia ODC and is used in Australia to support a wide variety of applications such as water allocation management by governments (Krause et al., 2021), identification of waterbodies for wildfire management, flood inundation modelling (Huang et al., 2019), floodplain monitoring (Hou et al., 2019), wetland monitoring (Dunn et al., 2023), wildlife management (Perry et al., 2021), and groundwater exploration (Hoare et al., 2016).

Development of complex algorithms for dynamic mapping over large areas requires the use of training data that captures the variability of features across space and time (Halabisky et al., 2018). Transferring and extending algorithms trained in one location to a new location with a different range of temporal and spatial variability across target features can result in poor results (Orynbaikyzy et al., 2022). However, given the similarity in the spectral response of water in different landscapes, and the use of consistent analysis-ready Landsat satellite data, it is possible to extend the WOfS algorithm to the African context without needing to re-train it from the beginning. Applying WOfS to the continent of Africa would allow for an initial deployment of a surface water map service without the need for a lengthy research and development phase. However, while the two continents have some similar climate and geographies, there are many differences and a rigorous accuracy assessment requires local and regional knowledge and expertise. As DE Africa is guided by the principle of fostering national and regional co-production to develop ownership of both the DE Africa program and all products and services, the WOfS validation process was undertaken by the DE Africa Product Development Task Team (PDTT). By leveraging the expertise of the PDTT we are able to test the transferability of the WOfS algorithm to map the distribution of surface water across the African continent to develop a new continental service starting from 1984 and updated as every new Landsat scene becomes available.

2 Materials and methods

2.1 Product development task team

The Product Development Task Team (PDTT) is a working group of Digital Earth Africa composed of program partner members representing several African regional and national geospatial organisations. The PDTT works together to identify shared needs and data gaps for African countries and selects, designs, plans, develops, and validates DE Africa's continental-scale services and products. The PDTT also provides support in the use and application of DE Africa data products by stakeholders and end users across their broader institutional networks. At the time this project was carried out, the PDTT consisted of the following organisations: L'Observatoire du Sahara et du Sahel

TABLE 1 List of Agro-Ecological Zones (AEZ) used for this project and the countries covered for each zone.

AEZ	Countries
Eastern	Tanzania, Kenya, Uganda, Ethiopia, Rwanda, and Burundi
Western	Nigeria, Benin, Togo, Ghana, Côte d'Ivoire, Liberia, Sierra Leone, Guinea, and Guinea-Bissau
Northern	Morocco, Algeria, Tunisia, Libya, and Egypt
Sahel	Mauritania, Senegal, Gambia, Mali, Burkina Faso, Niger, Chad, Sudan, South Sudan, Somalia, and Djibouti
Southern	South Africa, Namibia, Botswana, Lesotho, and Eswanti
Central	Angola, Democratic Republic of the Congo, Congo, Gabon, Cameroon, Equatorial Guinea, and Central African Republic
Indian Ocean	Madagascar, Mauritius, Reunion, and Comoros

(OSS, Tunisia), Regional Centre for Mapping of Resources for Development (RCMRD, Kenya), African Regional Institute for Geospatial Science and Technology (AFRIGIST, Nigeria), and AGRHYMET (Niger), which collectively represented the interests of 43 African countries.

2.2 Study area

The geographical coverage of the data is the African continent, including surrounding islands. Africa exhibits a remarkable range of climate zones, each characterized by distinct weather patterns and ecological conditions (FAO, 2021). The Food and Agriculture Organization of the United Nations (FAO) and the International Institute for Applied Systems Analysis (IIASA) developed a geospatial framework known as agro-ecological zones (AEZ) that enhance understanding and management of the diverse agricultural potential and limitations within a given region (FAO, 2021). These zones are specific geographic areas delineated based on a combination of climate, soil conditions, and other environmental factors that influence agricultural productivity and suitability. Similar to Xiong et al. (2017) in the production of the Global Food Security Support Analysis Data (GFSAD) crop mask we used simplified AEZ to assess regional accuracy by combining smaller AEZs and snapping boundaries to country borders resulting in seven different zones (Table 1).

2.3 Water observations from space (WOfS)

The Water Observations from Space (WOfS) service for Africa was created using an algorithm that has been developed and well-tested in Australia (Mueller et al., 2016). The WOfS algorithm maps surface water for every pixel in an image using a decision tree algorithm that considers surface reflectance measurements in selected spectral bands and a number of normalised difference indices, such as the Modified Normalised Difference Water Index (MNDWI) (Xu 2006). This algorithm is applied to the Landsat Collection two surface reflectance product hosted by DE Africa (DE Africa, 2022). Cloud and cloud shadow classifications are inherited from the Landsat Collection two quality assessment band. Ancillary information derived from observation metadata and the SRTM digital elevation model is used to flag areas where the classification is less reliable. Such areas include steep slopes (greater than 12°), observations with a low solar incidence angle (less than 10°), and regions shadowed by terrain. The WOfS product suite includes daily water observations and statistical summaries. Daily water observations, referred to as Water Observation Feature Layers (WOFLs), identify water in each satellite scene and are generated for new Landsat observations as soon as they become available on the DE Africa platform following satellite acquisition (a typical latency of 2°days). The frequency of inundation of every pixel is calculated in two ways: as a single all-time summary, and as annual summaries, which allow for greater understanding of the dynamic nature of waterbodies and flooding (Figure 1). Through the DE Africa cloud based platform on AWS, around one million Landsat scenes across the entire time series were processed in under 10 hours.

2.4 Validation methods

2.4.1 Validation sampling design

Given the extensive time period of analysis, as well as the continental extent, a validation approach needs to provide insights on both the spatial and temporal accuracy of WOfS. A stratified random sampling scheme was therefore selected, with points assessed as "truth" through interpretation of imagery. Because of the large effort required to create a continental-scale, multi-temporal reference dataset, we selected a sampling design that is independent of the WOfS classification so that it can be used to compare future versions of WOfS (which may use new algorithms or other types of satellite images, i.e., Sentinel-2) as well as other existing maps of surface water.

A key aspect of validation is the creation of a reference dataset, which in this case would be sensitive to differences between water classifiers and source imagery (Landsat v. Sentinel-2). Water classifiers typically produce accurate results when applied to large open waterbodies, which was not the focus of this map service. Therefore, we masked out large water features with an area of more than 100km^2 (FAO, 2020) from the sample frame. This, along with the stratified sampling mechanism, focused validation on areas that are more challenging to map, such as small waterbodies with different colours, depths and surrounding environments, and edges of waterbodies that often contain mixed pixels. The sample scheme provides a more sensitive comparison of WOfS to other datasets than a purely random sample. However, it should be noted that this scheme may diminish accuracy statistics when compared to validation results obtained using sampling schemes without large waterbody masking or stratified sampling.

We generated sample points covering the continent including the main islands using a stratified random sample to select locations with different water occurrences and waterbody types. First, samples were stratified by the simplified AEZs. We generated 2,900 sample points with a minimum of 300 sample points per AEZ. We increased the number of points to 500 for 4 AEZs that had a higher number of waterbodies. The number of points per AEZ is as follows; Central =

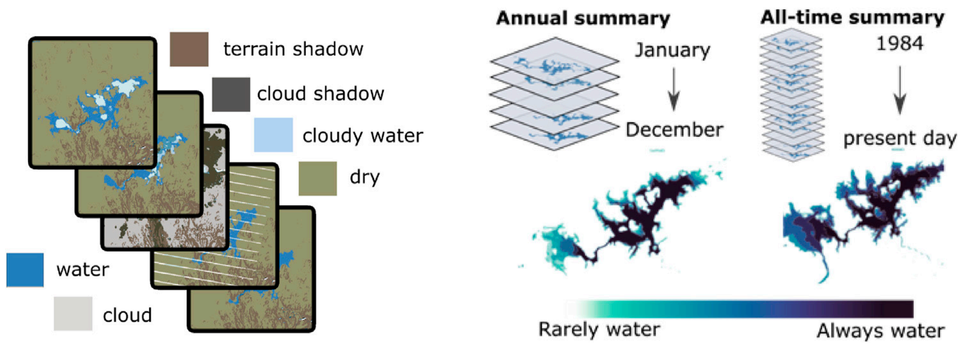

FIGURE 1
Example WOfS products for Lake Barrage El Mansour Eddahbi near Ouarzazate, Morocco. For every observation, pixels are classified as water or dry, as well as if they are covered by cloud, cloud shadow, or terrain shadow (left). An all-time summary and annual summaries are created from the time series of WOfS calculating the percentage of time a pixel is classified as water, identifying ephemeral and permanent surface water and how it changes over time.

500, Eastern = 500, Western = 500, Southern = 500, Sahel = 300, Indian Ocean = 300, Northern = 300.

Second, within each AEZ, we stratified points using the median Normalised Difference Water Index (NDWI) (McFeeters, 1996) calculated using provisional Collection 2 Landsat-8 data from 2013 to 2019 with clouds and cloud shadows masked out. This is based on the assumption that the median NDWI correlates with water occurrence frequency. We selected NDWI as opposed to MNDWI because NDWI was not included in the WOfS algorithm and we felt it provided a more dependent measure of water. Even though NDWI is not always reliable, we expect water to generally have a high NDWI value and dry land to have a low NDWI value; therefore permanent water will have a high median NDWI calculated over time. Sometimes flooded area will have a median NDWI value between median NDWI measured for permanently wet and permanently dry areas. Cumulative distributions for median NDWI values and WOfS all-time summary detection frequency were compared for each AEZ. In order to ensure a distribution of points across the gradient from permanently flooded to ephemerally flooded, the sample points were stratified into three classes based on the NDWI median: Permanently flooded (median NDWI greater than 0.03, corresponding to approximately >90% detection frequency), Sometimes flooded: (median NDWI between −0.03 and 0.03, or between 60%–90% detection frequency), Rarely flooded (median NDWI less than −0.03, or < 60% detection frequency). We further stratified the Rarely flooded sample points into three NDWI value ranges to ensure that more sample points fell in areas that are more likely to be confused by any water classifier.

For example, for the Central AEZ the sample point breakdown of 500 points was as follows.

- Permanently flooded: 150/500 (30% of points, median NDWI greater than 0.03).
- Sometimes flooded: 150/500 (30% of points, median NDWI between −0.03 and 0.03).
- Rarely flooded: 200/500 (40% of points further stratified into three classes):
  - 50/500 points (10% of sample points, median NDWI less than −0.06).
  - 50/500 points (10% of sample points, median NDWI between −0.06 and −0.04).
  - 100/500 points (20% of sample points, median NDWI between −0.04 and −0.03).

We imposed 30 km as a minimum distance between points to avoid clustering in the same waterbody. A further 100 samples stratified in the same way were added to the dataset that all analysts were asked to classify for cross-validation purposes. Points were assessed monthly for 2018. This provided us with 36,000 potential observations (3,000 * 12˚months), with the expectation that roughly 50% or less would be cloud-free and that the number of cloud-free observations would vary in the wet and dry seasons.

2.4.2 Validation assessment (response)

We used Collect Earth Online (CEO) an open source, free online tool initially developed by NASA SERVIR, to label our sample points (CEO, 2023). For every sample point, analysts used Sentinel-2 images taken within the first 5˚days of each month to assess whether a point was inundated or not. We selected this approach, rather than using monthly composites, to avoid potentially ambiguous interpretation of mean monthly values, and to enable identification of the exact date of the labelled inundation observation. Analysts were able to view the Sentinel-2 imagery as true color or false color composites as well as several different spectral indices including NDVI and NDWI. Analysts marked down all the months where water was detected. They also marked months where there was no water, the image quality was low (i.e., cloudy) or the image could not be assessed because it was hard to determine (e.g., mixed pixel, muddy, sediment). Mapbox high resolution base maps were available to assist in image interpretation, but did not have a date associated and so could not be used for assessment of inundation patterns. In addition, analysts noted the basic land cover type using the classes modified from the NASA Globe Observer land cover classification (NASA Globe Observer, 2023).

To assist image interpretation and to ensure consistency in labelling across analysts, analysts underwent a group training, assessed a common training dataset, and followed up with a one-on-one meeting with the DE Africa Science Team. The original CEO tool was tested out and improved based on input from the PDTT. Each analyst assessed a common set of 80 sample points in order to develop a cross-validation dataset, allowing for an estimation of the consistency of labelling between analysts. Leveraging the rich temporal reference dataset, separate accuracy assessments were performed for all cloud-free observations, and for both the wet and dry seasons.

For all of these accuracy assessments, we removed any points where there were not clear observations due to cloud cover. Because Sentinel-2 images from the first 5°days of a month are used to increase temporal alignment, matching WOfS data are chosen from a window that's extended by 5°days on each side. If water is detected in any images within this window, the WOfS classification is "water." We created confusion matrices for the data to allow us to assess the accuracy and calculated overall accuracy, user and producer's accuracy and F1 (Maxwell et al., 2021). We then examined this information for each AEZ for the wet and dry seasons. We assigned wet and dry seasons for each AEZ based on monthly rainfall information and selected clear observations within 1 month close to the middle of each season, wet and dry. This ensures the data points chosen represent typical conditions for the relevant season.

2.4.3 Qualitative assessment

In addition to the quantitative assessment, the PDTT also conducted a qualitative assessment of the WOfS algorithm performance across the continent by engaging with their network of stakeholders in a myriad of different ways. PDTT members engaged with stakeholders through workshops, phone calls, emails and WhatsApp messages, explaining the purpose of and expectations from the exercise. Stakeholders were asked to examine the WOfS map service on the DE Africa platform and to provide feedback on areas that they were familiar, assessing whether WOfS mapped these areas accurately. The PDTT team also asked potential end users for feedback on the usefulness and potential uses of the WOfS as a continental, near-real time service. The qualitative assessment served two purposes, the first was to assess the visual accuracy and useability of the product, and second was to identify demand for map service derivatives (e.g., summary products), training, and analytical tools that could support uptake and use. In total 44 stakeholders, regional experts, and potential end users were surveyed.

2.5 Co-production of use cases at national and local scales

As part of our co-development process we built open-source workflows (i.e., Jupyter notebooks) for applications at the local and continental scale and applied them to two use cases. The purpose of these use cases was to highlight how WOfS can be summarised at national and continental scales for large-scale assessment and analysis, as well as finer local scales, including hydrologic basins or even individual waterbodies.

TABLE 2 Accuracy assessment for all labelled observations across the continent. Overall accuracy (OA) was 82.1% and F1 score for water classification is 0.87.

	No water	Water	Total	Producer's (%)
No Water	2,721	369	3,090	88.1
Water	1,669	6,604	8,273	79.8
Total	4,390	6,973	11,363	
User's (%)	62.0	94.7		

Using WOfS, we can calculate water detection frequencies over any defined time period, then estimate the typical water extent above a detection threshold. Summarising WOfS at an annual interval and within each country supports reporting on SDG 6.6.1 (UN water, 2023) change in the extent of water-related ecosystems over time. The annual water extent timeseries further supports detection of anomalies. For example, by comparing the water extent measured in 2020 to the all-time mean for each country, we can identify countries that had an increase or decrease in permanent water availability.

3 Results

For each of the 12°months, 3,000 points were assessed. Not all observations could be labelled as water or not water due to cloud cover, lack of image acquisitions, or uncertainty due to spatial resolution and mixed pixels. After removing data points that could not be labelled, a total number of 11,363 observations in 2,377 locations remained, and were used for the accuracy assessment. 72.8% of these observations were labelled as water (8,293) and 27.2% as not water (3,090). It was expected that cloud cover would limit the number of observations, and that this would vary between wet and dry seasons as well as between AEZ regions. The number of valid points generally correlates with monthly rainfall, a proxy for cloud cover. The PDTT had 93.0% agreement between analysts for the common training dataset.

3.1 All cloud-free observations

The overall accuracy of WOfS at the continental-scale was 82.1%, with a producer's accuracy of 79.8% and user's accuracy 94.7% for the water class (Table 2). To reiterate, this accuracy assessment is based on the constrained sample design where large waterbodies were masked out and sampling was focused in more challenging areas (e.g., edges of waterbodies and small waterbodies).

WOfS accuracy varies across each AEZ with the highest overall accuracy in the Eastern AEZ and lowest in the Indian Ocean AEZ (Figure 2). WOfS performs well in the Eastern and Northern AEZs with an overall accuracy of more than 85%. For all AEZs, except the Western and Indian Ocean AEZs, WOfS achieved overall accuracies of more than 80%. Relatively low producer's accuracies, or high omission errors, were measured in the Western, Southern and Indian Ocean AEZs. F1 scores for the water classification range from 0.83 to 0.91 in the AEZs.

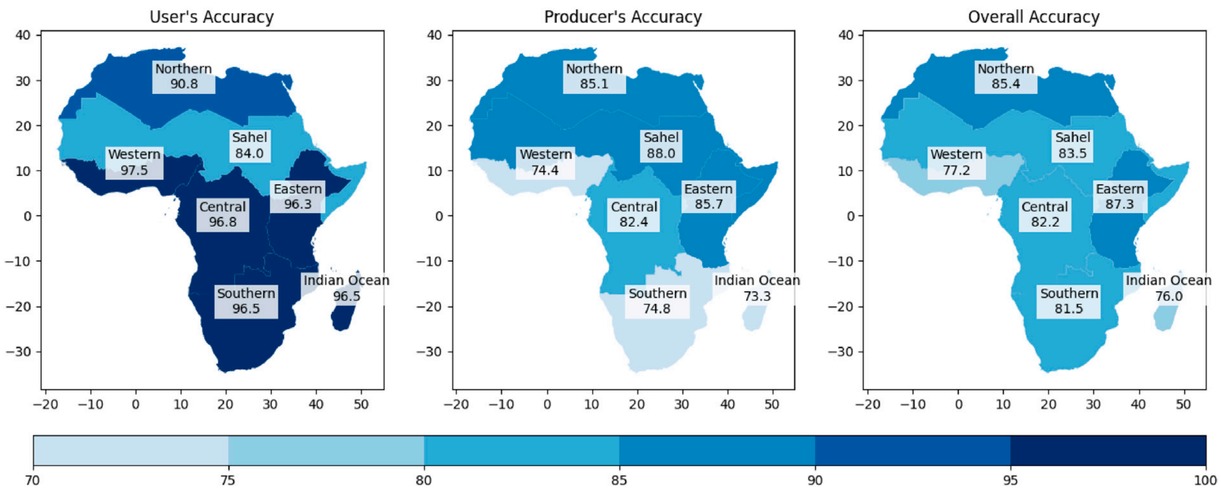

FIGURE 2
Accuracy assessment results for each AEZ showing user's accuracy (left), producer's accuracy (middle), overall accuracy (right).

TABLE 3 Accuracy assessment for the wet and dry seasons of each AEZ (DE Africa, 2022).

AEZ	Wet month	# Of points in wet month	Overall accuracy in wet month (%)	Dry month	# Of points in dry month	Overall accuracy in dry month (%)
Northern	December	95	84.2	July	89	76.4
Sahel	August	66	77.3	February	78	87.2
Western	June	45	64.4	December	228	86.0
Eastern	April	168	83.3	October	264	86.4
Central	October	75	74.7	June	169	86.4
Southern	January	156	82.7	July	223	82.1
Indian Ocean	January	98	72.5	August	193	78.2

3.2 Seasonal accuracy

The accuracy of the WOfS varied across wet and dry months for each AEZ. For most AEZs the accuracy improved in the dry season (Table 3). A different trend was measured for the Northern AEZ, where the rainfall variation is small across the year. The lowest accuracy was measured in the wet month in the Western AEZ, coinciding with the lowest number of clear observations. It is in general challenging to map water accurately during high rainfall months in the Western, Central and Indian Ocean AEZs.

Feedback from the qualitative assessment was very positive across all surveyed participants. One stakeholder felt that the service would be a very useful resource not only for researchers in water resources management, but also for practitioners and decision makers in the formulation of water resources management policies. The most common criticism is that the spatial resolution of WOfS was not adequate for small waterbodies and for applications and areas where change was below the spatial resolution on Landsat (30m^2).

3.3 Applying WOfS at continental scale

WOfS measures surface water extent and water detection frequencies that can be used to classify permanent and seasonal water bodies. Our continental scale application demonstrated how WOfS can be used to monitor water availability and understand drought and flooding relative to historical data (Figure 3). We found that the anomaly pattern measured by WOfS roughly correlates with the rainfall anomaly estimated for the same period using the monthly CHIRPS rainfall (Funk et al., 2014). Such analysis can be applied at different scales to help users understand changes in surface water extent at a finer detail across the continent.

3.4 Applying WOfS at local scale

The results from the use case with AGRYHMET, demonstrated how WOfS can be used to map and monitor changes in Lake Chad to help decision-makers understand year-to-year and long-term changes in surface water area (Figure 4). Floods are recurrent

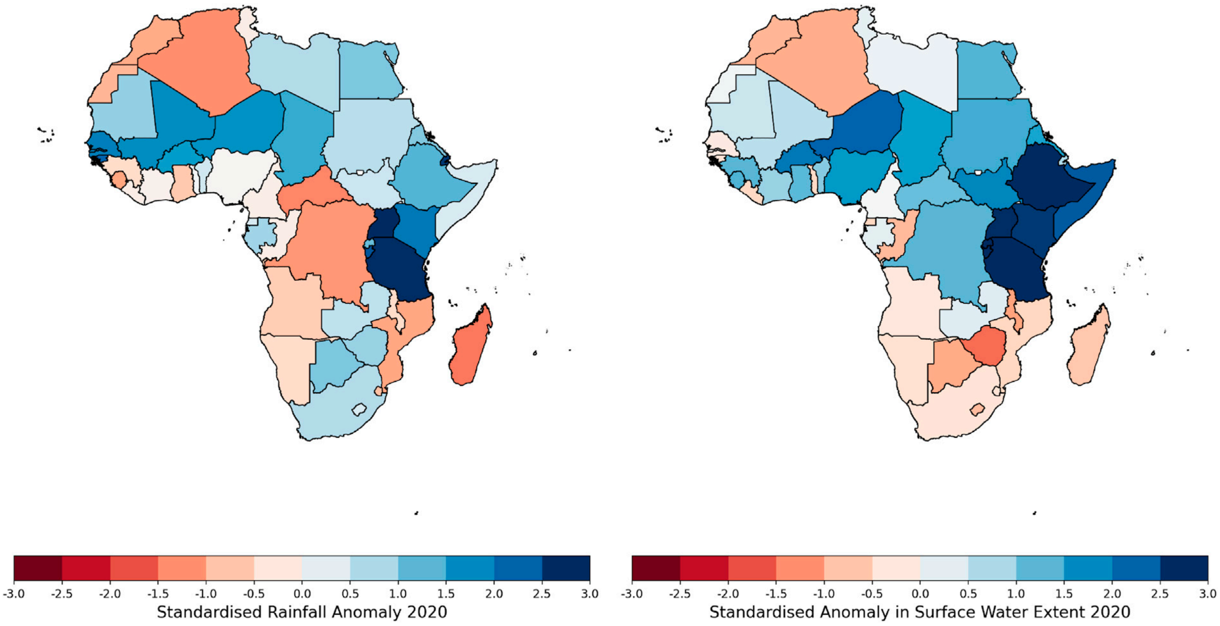

FIGURE 3
Rainfall (left) and surface water extent (right) anomaly measured for 2020. Surface water extent for a given year includes areas where water is detected in more than 30% of the observations. This threshold was selected to reduce noise. A reference period of 2000–2020 (inclusive) has been used to estimate the mean and standard deviation of total rainfall and total surface water extent within each country. Standardised Anomaly is calculated as the difference between the estimate for 2020 and the mean for the reference period, divided by the standard deviation.

around the Lake and along the rivers that flow into it in the countries of the Lake Chad Basin Commission (LCBC).

AGRYHMET found the surface water area was three times greater in 2021 compared to 1986. Not only was there a significant gain in surface water area, there was a shift in the spatial distribution of water. AGRYHMET was able to identify and map areas that have remained wet (1905 km^2), wet areas that have changed to dry (1,415 km^2), dry areas that have converted to wet areas for (4,389 km^2) and finally areas that have remained dry (20,930 km^2). The change in surface water area is likely due to solid transport in the area which from year to year carries a significant amount of sand to be laid down at the bottom of Lake Chad and altered its bathymetry. This is the result of cultivation practices and climatic changes with an increase in violent winds over time. Some parts of the lake have become shallower than they were in the past. The same amount of water that could be contained in a limited area now covers a larger shallow area increasing the risk of flooding in the Lake Chad area. AGRYHMET presented the summarised outputs to the community surrounding Lake Chad, including the Chad Lake Basin Commission, the governors, the mayors, the traditional chiefs, and producers' associations (agriculture, fishing, livestock) to help understand historic changes for better land use planning and to identify areas at high risk of flooding. This co-production process not only helped develop capacity within the PDTT, but also provided use cases that allowed the PDTT to engage with their regional stakeholders.

4 Discussion

The WOfS algorithm was able to accurately measure surface water across the African continent and through time, with only small variations in accuracy between rainy and dry months and different AEZs. The results from both the quantitative and qualitative accuracy assessment efforts provided the necessary confidence for the DE Africa team to publish the WOfS service as an operational product, updated on a monthly basis, for continent-wide use.

4.1 Considerations and limitations

Our validation effort is limited to what can be detected visually in the Sentinel two imagery. It should be noted that water features obscured by vegetation canopy, terrain shadows, or are too small to detect in Sentinel two imagery may have been missed in the validation labelling process. In addition, the identification of the presence of water is tied to the timing of the satellite overpass. Sentinel two imagery was used here for validation, where it matched a Landsat image acquisition within ± 5°days. There may be instances where waterbodies have dried or shifted within that time period, although we expect the impact on accuracy assessment to be very small.

The WOfS algorithm has several limitations that should be considered when using the WOfS dataset for a specific application. The errors associated with WOfS across the African continent were found to be similar to those noted in Australia, and are largely driven by issues of spatial resolution (i.e., a 30m^2 pixel size) and the difficulty in detecting areas where a pixel covers both water and dry surfaces (i.e., mixed pixels) (Mueller et al., 2016). These areas tend to be on the edges of lakes and wetlands, or over small waterbodies where there is a mix of water and vegetation within a single 30m^2 pixel. Waterbodies with high levels of sediment or

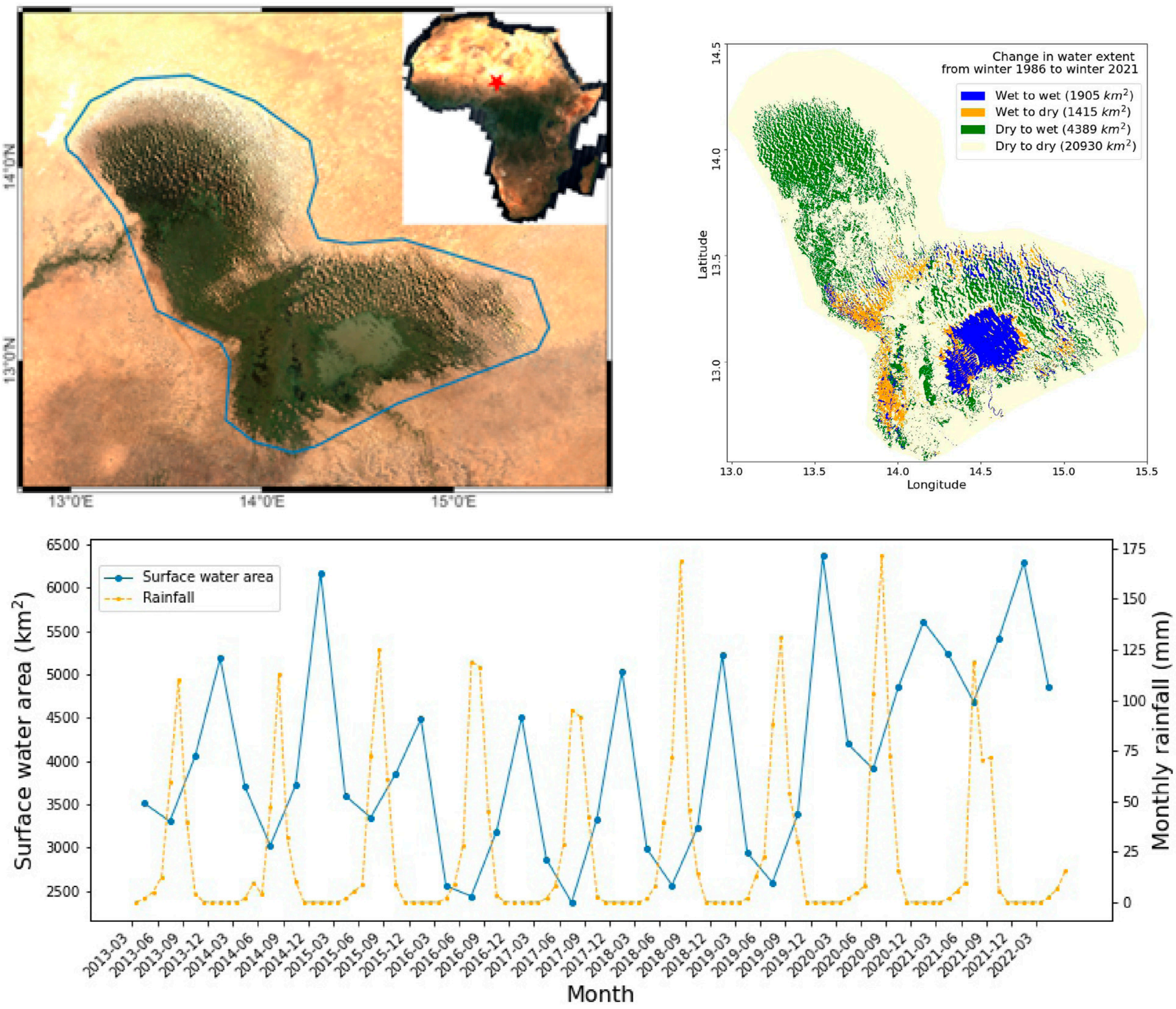

FIGURE 4
Location (left inset) and spatial extent of the Lake Chad area of interest (upper left); Change in water extent from winter 1986 to winter 2021 (upper right). For each winter season, water observations from December to February next year are summarised. Pixels with more than 20% of water detection frequency within a season are labelled as wet, otherwise as dry. Seasonal change of surface water extent from Spring (March to May) 2013 to Spring 2022, compared to monthly CHIRPS rainfall (bottom). Within each season, surface water extent is calculated including pixels with more than 20% water detection frequency. While this threshold is somewhat arbitrary we selected it to include more dynamic extent (from fewer observations). Lake Chad has increased in surface water area with a shift in the spatial distribution of surface water area. Overall, 24% of the areas has shifted from wet to dry while 76% are from dry to wet.

floating vegetation may also be missed as they are spectrally similar to terrestrial pixels. Through our qualitative assessment we noticed that WOfS had errors of omission related to narrow stretches of rivers, especially those with high sediment levels in Western and Central Africa, which likely contributed to the lower producer's accuracy in these regions. While small or narrow features may get missed in a single observation, they are more likely to get captured at the annual summaries and the all-time summary. However, because small or narrow features may get missed in individual observations, it important to note that frequency of flooding may be underestimated in any summary products or time series analysis (Figure 5).

Users who want to map and monitor changes to waterbodies smaller than a Landsat pixel should consider applying sub-pixel methods like NDWI (McFeeters, 1996), tasselled cap indices (Fickas et al., 2016), or spectral mixture analysis (Halabisky et al., 2016) and/or use satellite images with a finer spatial resolution such as Sentinel-2, or other very high resolution satellite imagery (Mishra et al., 2020).

The WOfS product is also limited by the 16-day acquisition frequency of Landsat and poor historical coverage by Landsat five over the African continent prior to the year 2000. In some cloudy regions there may be insufficient cloud-free pixels to adequately track changes in surface water dynamics. Specific applications such as flood monitoring may also be limited by the temporal resolution in the rainy season when there is a higher frequency of cloud cover.

The WOfS algorithm was developed for application to analysis ready surface reflectance data with physical reflectance values between 0 and 1. The algorithm is sensitive to inaccuracy in

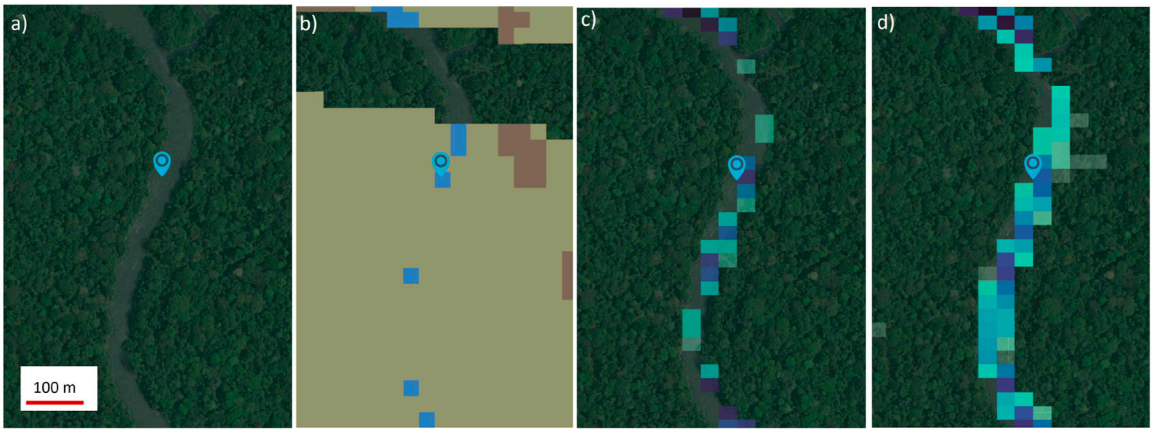

FIGURE 5
Narrow or small waterbodies **(A)** may be omitted from individual WOfS observations (WOfLs) **(B)** if they are close to scales near or below a 30 m pixel resolution. These waterbodies are more likely to be detected in the annual **(C)** or all-time summary **(D)**, but may have inaccurate estimates of flood frequency. Water is colored light blue in Figure 5B. For a detailed description of the WOfLs classification please reference Figure 1.

FIGURE 6
WOfS can be viewed and accessed through the Digital Earth Africa maps platform https://maps.digitalearth.africa.

input data and breaks down when negative reflectance values are used. A significant increase in false negative classifications, i.e., where water was detected as dry was observed in the results in situations where aerosol values are estimated to be high in the Landsat Collection two data. This is likely caused by over-correction for atmospheric effects, which has a larger impact on low reflectance values and typically occurs over oceans and large lakes. To avoid masking out potentially useable data over areas with poor coverage, we decided to keep these classifications and to flag pixels with negative reflectance values. Despite this limitation, the validation results demonstrate that WOfS offers a balanced performance over all types of water bodies present within the African landscape.

There are other water classifiers available or are being developed. In principle, we can adopt or develop new open source methods to improve WOfS in the future and the validation dataset created will help to benchmark these new methods. We welcome the use of this dataset by other teams to assess their method. Although we emphasize that matrices created against this validation dataset should be used in combination with other validation methods relevant for specific applications. WOfS is continually being updated as each Landsat scene becomes available. Annual summaries for the previous year and the all-time summary are updated at the start of each year.

4.2 Accessing the methods and dataset

Technical documentation for the WOfS algorithm, product specifications, and guidance for accessing the data through different interfaces is provided in the Digital Earth Africa Data Catalogue (DE Africa, 2022).

WOfS can be explored interactively through the Digital Earth Africa Maps user interface (Figure 6). For further analysis the data are freely accessible through the DE Africa Sandbox; this includes the full WOfS archive with annual and all-time summaries from 1984 to 2022, as well as additional DE Africa analytical workflows to query, analyse and view the data, which can be customized for different user locations of interest.

5 Conclusion

The Water Observations from Space (WOfS) is a Digital Earth Africa operational service that translates decades of satellite imagery into easy-to-consume information on the presence, location and recurrence of surface water across Africa.

The WOfS algorithm, originally developed for Australia, has been successfully applied to the full Landsat satellite archive (since the mid 1980s) for the entire African continent, and has been validated for different time periods and types of waterbodies. WOfS enables users to understand the location and movement of inland (and coastal) water over time. It shows where water is usually present; where it is seldom observed; and where inundation of the surface has been observed by satellite. This allows users across Africa to map, assess, visualise, and manage surface water resources and understand trends over time.

Easily accessible and frequently updated information which is produced consistently across space and time, such as WOfS, is critical for understanding the past as well as present distribution of surface water, and in understanding surface water dynamics and changes to these which are occurring as a result of climate change. This information is important for many critical water security issues such as identifying accessible water sources during dry seasons, planning for and preventing the impacts from flooding and drought, and managing the sustainable use of water resources across scales (site, national, regional).

Data availability statement

The original contributions presented in the study are included in the article/Supplementary Materials, further inquiries can be directed to the corresponding author.

Author contributions

Conceptualization, MH, FY, AL, and LH; methodology, MH, FY, and CB; validation, GA, EB, BM, FM, DO, and NM; formal analysis, MH, FY, CB, EB, and AL; data curation, CB, FY, EB, and NM; writing—original draft preparation, MH and FY; writing—review and editing, CB, L-MR, MH, and FY; visualization, E-FC; project administration, MH; funding acquisition, LH. All authors listed have made a substantial, direct, and intellectual contribution to the work and approved it for publication.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

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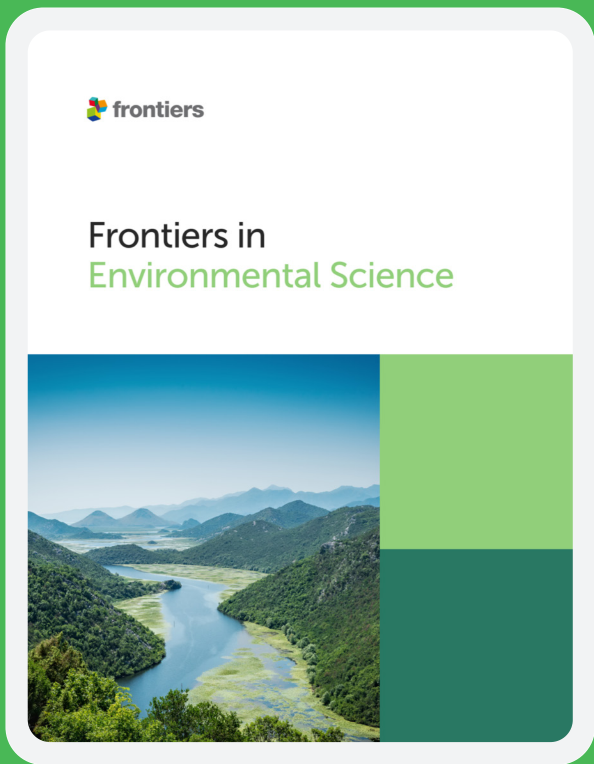

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