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Multi-Objective Water Management in Idaho's Henrys Fork Watershed: Leveraging Reservoir Operation and Groundwater Pathways to Benefit Aquatic Habitat

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MULTI-OBJECTIVE WATER MANAGEMENT IN IDAHO'S HENRYS FORK WATERSHED: LEVERAGING RESERVOIR OPERATION AND GROUNDWATER PATHWAYS TO BENEFIT AQUATIC HABITAT

by

Christina N. Morrisett

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in

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ABSTRACT

Multi-Objective Water Management in Idaho's Henrys Fork Watershed: Leveraging Reservoir Operation and Groundwater Pathways to Benefit Aquatic Habitat

by

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Utah State University, 2023

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Multi-user water management is a challenging arena further complicated by climate change. The Henrys Fork, Snake River, Idaho is an agricultural watershed that exemplifies those throughout the semi-arid American West and serves as the context and inspiration of this research. This dissertation uses an integrated water resource management approach that considers groundwater-surface water relationships, farm-scale decisions and basin-scale outcomes, upstream reservoir operation for downstream aquatic habitat, water rights, and collaborative stakeholder management to identify drought adaptation strategies accordingly.

Chapter 2 uses an interdisciplinary approach to quantify how cumulative farm-scale improvements in irrigation efficiency affect hydrology at the landscape-scale and alter groundwater-surface water relationships. 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.

Chapter 3 uses coupled streamflow-forecasting, reservoir-operation, and groundwater-surface water models to quantify 1) the potential to conduct agricultural managed aquifer recharge (Ag-MAR) in the lower watershed under a warming climate and 2) subsequent groundwater return flows and streamflow response. Water for Ag-MAR was largely available in April and October, reducing peak springtime streamflow at the watershed outlet by 10–14% after accounting for return flows. Streamflow contribution from recharge peaked in July and November, increasing July–August streamflow by 6–14% and November–March streamflow by 9–14%. I demonstrate Ag-MAR can recover groundwater return flows when applied as flood irrigation on agricultural land with senior water rights.

Chapter 4 developed streamflow-habitat models for three fish species in a reach where irrigation-season flows are managed by releases from an upstream reservoir. I used these models to 1) quantify aquatic habitat at different streamflows and 2) assess the differences in aquatic habitat across two management regimes. Using model output, I demonstrated that moving the location and magnitude of the management target to account for local irrigation diversions will contribute to more consistently suitable fish habitat in the reach while continuing to meet upstream management objectives.

(250 pages)

PUBLIC ABSTRACT

Multi-Objective Water Management in Idaho’s Henrys Fork Watershed: Leveraging Reservoir Operation and Groundwater Pathways to Benefit Aquatic Habitat

Christina N. Morrisett

Multi-user water management is a challenging arena further complicated by climate change. This research is based in the Henrys Fork, Snake River, Idaho—an agricultural watershed that exemplifies those throughout the semi-arid American West. This dissertation uses an integrated approach that considers groundwater-river relationships, farm-scale decisions and basin-scale outcomes, upstream reservoir operation for downstream aquatic habitat, water rights, and collaborative stakeholder management to identify drought adaptation strategies accordingly.

Chapter 2 uses an interdisciplinary approach to quantify how improvements to irrigation efficiency at the farm-scale (i.e., converting from flood to sprinkler irrigation) can add up to affect hydrology at the landscape-scale and alter groundwater-surface water relationships. 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 2,521 acre-ft (23%) and annual return flow to the river decreased by 2,431 acre-ft.

Chapter 3 uses streamflow predictions, local reservoir operation standards, and the relationship between groundwater and river flows to quantify 1) the potential to conduct

aquifer recharge in the lower watershed under a warming climate and 2) resulting streamflow response from groundwater. Water for recharge was largely available in April and October, reducing peak springtime streamflow at the watershed outlet by 10–14% after accounting for groundwater return. Streamflow contribution from recharge peaked in July and November, increasing July–August streamflow by 6–14% and November–March streamflow by 9–14%. I demonstrate recharge can recover groundwater return flows when applied as flood irrigation on agricultural land with senior water rights.

Chapter 4 developed relationships between streamflow and habitat for three fish species in a reach where irrigation-season flows are managed by releases from an upstream reservoir. I used these relationships to 1) quantify aquatic habitat at different streamflows and 2) assess the differences in aquatic habitat across two different streamflow management histories. Using these relationships, I demonstrated that moving the management target's location and flow amount will contribute to more consistently suitable fish habitat in the reach while continuing to meet upstream management objectives.

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CHAPTER 1

INTRODUCTION

Rivers in the arid and semi-arid American West supply water for irrigated agriculture, urban use, hydropower generation, recreational fisheries, and aquatic habitat. As a result, watershed stakeholders include farmers, municipalities, electric companies, recreationists, conservationists, and more. Such stakeholder multiplicity—where problems, and therefore solutions, are differently defined by a given party (Freeman, 2000; Lund, 2012)—makes for a challenging water management landscape. Climate warming exacerbates this challenge (Ficklin et al., 2022). Earlier snowmelt stresses storage infrastructure (Barnett et al., 2005; Rauscher et al., 2008; Stewart et al., 2005). Reduced snowpack decreases late-summer baseflow (Li et al., 2017; Tague et al., 2008), subsequently warming water (Null et al., 2013; Wenger et al., 2011), diminishing streamflow available for consumptive withdrawal and storage (Li et al., 2017; Null & Prudencio, 2016), and increasing reliance on groundwater extraction (AghaKouchak et al., 2015; Döll, 2009). Climate models predict substantial changes to precipitation and temperature in the American West (IPCC, 2014, 2018; USGCRP, 2017, 2017)—requiring water management strategies that are more resilient to periods of water scarcity (Barnett et al., 2008).

To best adapt water management for the 21st century, we must consider eco-hydrologic relationships as well as the social systems that demand and deliver water. Although general tools and global concepts are available for local adaptation and application (Petts, 2009), watershed-specific studies maximize local utility (Welsh et al., 2013) and buy-in (Pahl-Wostl, 2009; von Stackelberg & Neilson, 2014). Thus, my doctoral research is conducted in partnership with the Henry's Fork Foundation, a watershed conservation organization in eastern Idaho, USA. Here, I use an interdisciplinary, place-based approach to identify water management strategies best suited for water supply and ecosystem resiliency to drought in the Henrys Fork watershed, Snake River, Idaho.

In Chapter 2, I used an interdisciplinary approach to quantify how cumulative farm-scale improvements in irrigation efficiency affect hydrology at the landscape-scale and alter groundwater-surface water relationships. Irrigators are economically motivated in choosing their irrigation application mode and can have basin-scale hydrologic impacts over time. I discuss how agricultural managed aquifer recharge (Ag-MAR) can be implemented to recover return flows to rivers (Morrisett et al., In Review).

In Chapter 3, I used a series of models to quantify 1) the potential to conduct Ag-MAR in the lower Henrys Fork watershed under future climate and 2) subsequent groundwater return flows and streamflow response. Junior water rights for managed aquifer recharge limit the ability to conduct Ag-MAR at sanctioned sites, but senior water rights at flood irrigated sites allow for incidental aquifer recharge annually. Water rights and agricultural land conversion will continue to constrain the ability to conduct Ag-MAR and recover groundwater return flows diminished by improvements to irrigation efficiency.

In Chapter 4, I developed streamflow-habitat models for three fish species in a reach where irrigation-season flows are managed by releases from an upstream reservoir. I used these models to 1) quantify aquatic habitat at different streamflows and 2) assess the differences in aquatic habitat across two management regimes (Morrisett et al., 2023). Using model output, I demonstrated that moving the location and magnitude of the management target to account for local irrigation diversions will contribute to more consistently suitable fish habitat in the reach while continuing to meet upstream management objectives. I conducted this research to support efforts by the Henry's Fork Drought Management Planning Committee.

Overall, the research presented here explores mechanisms for achieving drought resiliency for multiple stakeholders while staying within the bounds of prior appropriation. Prior appropriation was designed to maximize the utility of scarce water resources (Craig et al., 2017). However, prior appropriation values historic priority and beneficial uses, leaving apparently little room for future societies to achieve water-use sustainability under the same framework. With diminishing water supply in the western United States given climate change and junior water rights holders subject to curtailment (Null & Prudencio, 2016), systems-thinking processes inherent to integrated water resources management (IWRM) are needed to identify adaptation strategies that will benefit diverse water uses. Rather than focusing on a narrow solution that may be uncertain, expensive, and contentious (Loucks, 2003), IWRM offers an opportunity to develop a portfolio of solutions that understands the system in its entirety, rather than considering single components (Brown et al., 2015). I use IWRM to consider the relationships between upstream reservoir operation and downstream flow, as well as on-farm irrigation application and subsequent groundwater-surface water responses to create a portfolio of solutions for multi-stakeholder water use in an agricultural watershed. This portfolio includes reservoir operation, instream flow considerations, irrigation application, and Ag-MAR for groundwater return flow recovery.

Additionally, with more stakeholders to consider, IWRM seeks to bring together physical and social science disciplines to engage stakeholders, understand and coordinate water use, and achieve adaptive management (Conallin et al., 2017). The physical sciences are foundational to water resource management, but effectively studying and incorporating social elements of water is not within the traditional toolkit of the discipline. In fact, civil engineers are often responsible for managing our water resources (Reisner, 1993; Worster, 1992). Thus, it is important to understand how social dimensions and stakeholder perspectives inform hydrologic dynamics and water resource management. Combining social and physical sciences in watershed studies also offers opportunity. Trust-building between scientists and stakeholders, and among scientists of different disciplines, creates opportunity for future collaboration beyond the water management issue at hand. It also facilitates interactions and knowledge building and can help identify and address tradeoffs among multiple and competing water needs (Cortese & Krannich, 2003; Loucks, 2003; Margerum, 2007). As such, in each chapter of this dissertation, I consider the role of collaboration and partnership in accomplishing suggested strategies to benefit aquatic habitat—a water use that does not have a water right within the watershed.

It has been a privilege to contribute science-based, stakeholder-driven research in the Henrys Fork watershed. This work has already informed collaborative management decisions and I look forward to integrating these findings on-the-ground to conserve local water resources.

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CHAPTER 2

THE IRRIGATION EFFICIENCY TRAP: RATIONAL FARM-SCALE DECISIONS CAN LEAD TO POOR HYDROLOGIC OUTCOMES AT THE BASIN SCALE

Abstract

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 (USA) 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.

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, Johnson and Pochop, 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, Lane and Neilson, 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, Hrast-Essenfelder and Perry, 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, Rosegrant and Ringler, 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, Kay and Weatherhead, 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 (USA)—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 2-1), a 28,000 km^2 unconfined aquifer that provides baseflow to the Snake River system (Hipke, Thomas and Stewart-Maddox, 2022). In addition to agriculture, the Henrys Fork hosts a recreational fishery worth $50 million USD (Van Kirk *et al.*, 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, Cosgrove and Johnson, 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, Thomas and Parham, 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 1978–2022 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?
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; Zalidis *et al.*, 1997; Burt *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

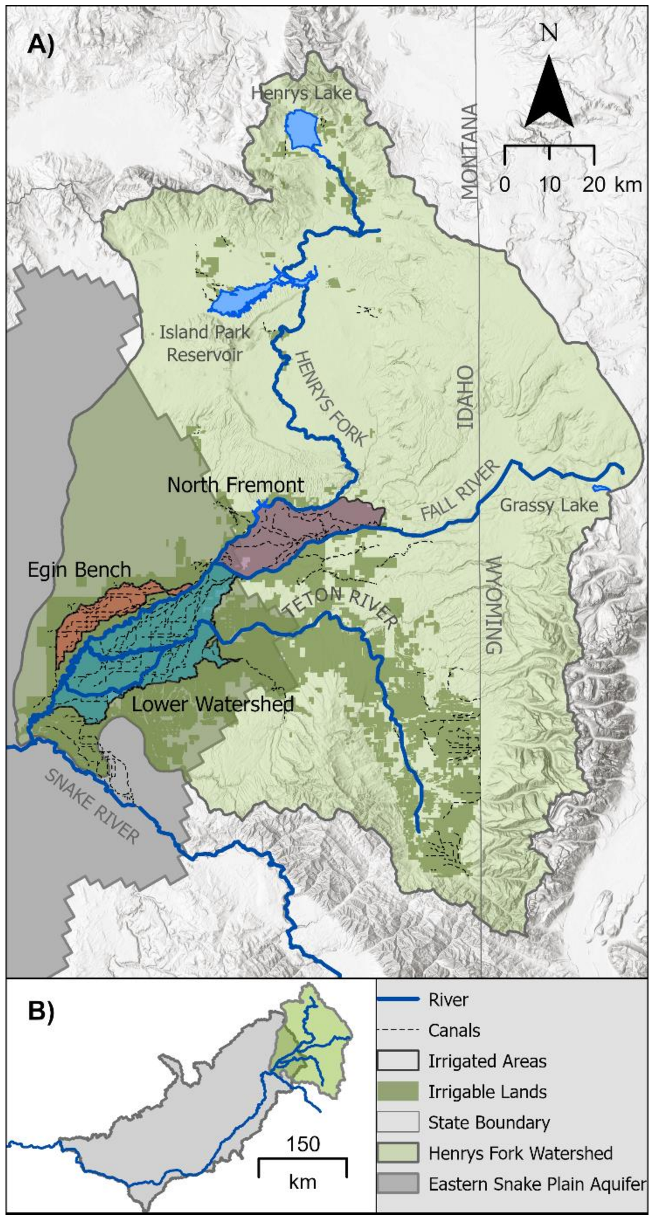

Figure 2-1. The Henrys Fork watershed (A) and the watershed relative to the Eastern Snake Plain Aquifer (B). Data sourced from Airbus, U.S. Geological Survey, NGA, NASA, CGIAR, NCEAS, NLS, OS, NMA, Geodatastyrelsen, GSA, GSI, and the GIS User Community.

The Henrys Fork watershed is 8,300 km^2 located in the headwaters of the Snake River Basin, Idaho, USA, ranging in elevation from 1,470 m to 3,800 m (Figure 2-1). 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, Thomas and Parham, 2018). The basin is fully adjudicated, and surface water rights include allowance for reasonable conveyance loss (Vonde *et al.*, 2016).

Irrigated land in the Henrys Fork watershed is separated into four regions: North Fremont, Egin Bench, Lower Watershed, and Teton Valley. 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 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.

Table 2-1. 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)

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 2-2). 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 1979–2008 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.

Table 2-2. 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 & 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 & 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% (Brown, 2008; King & Kincaid, 1997)

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, Hill and Patterson, 2009). Egin Bench and the Lower Watershed have predominantly flat terrain (≤0.5% slope), whereas the North Fremont region is steeper with greater heterogeneity (0–20% slope; Supplement). Regarding soil, Egin Bench is almost exclusively loamy fine sand, noted for its high infiltration and low runoff rates (Appendix A). 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 (Appendix A).

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, 2022b, 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, 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, Chapman and Francis, 2008). We used these analytical memos for inductive coding and thematic analysis (Attride-Stirling, 2001; Saldana, 2016). See Appendix A for interview instrument.

2.3 Geospatial analysis

We used aerial photography and Landsat satellite imagery from 1986–2020 to evaluate spatiotemporal trends in irrigation practices (Table A-2). 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 1988–2002 (every two years) and 2005–2020 (every five 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 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).

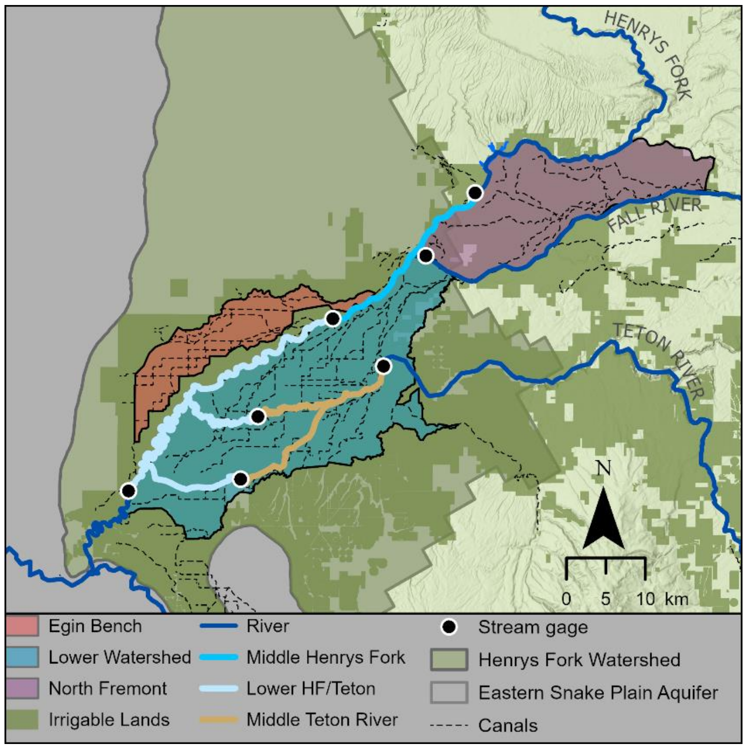

Figure 2-2. U.S. Geological Survey stream gages used in the water balance and reach gain calculations.

We estimated reach gain on reaches of the Henrys Fork and Teton River that interact with the ESPA (Figure 2-2). 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:

$$reach\ gain = reach\ outflow - reach\ inflow + diversions - exchange\ well\ injection \quad (1)$$

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:

$$flow_{unregulated} = flow_{regulated} + diversions + \Delta storage_{reservoir} + evaporation_{reservoir} - exchange\ well\ injection \quad (2)$$

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 (Supplement). 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. Equations (1) and (2) largely coincide with those used by Water District 1 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:

$$watershed\ inflow = watershed\ unregulated\ flow - \Delta storage_{reservoir} + exchange\ well\ injection - evaporation_{reservoir} \quad (3)$$

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 in downstream-most gage on the Henrys Fork near the bottom of the watershed at the confluence with the main Snake River (Figure 2-2). Equation (1) can be rearranged to yield:

$$diversion - reach\ gain = reach\ inflow - reach\ outflow + exchange\ well\ injection \qquad (4)$$

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 this 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

Of 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 (Supplement) 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 2-3B). This rate of pivot expansion slowed after 2000, with 87.2% of irrigated acres using center-pivot sprinklers by 2020 (Figure 2-3B). 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 2-3B). 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.

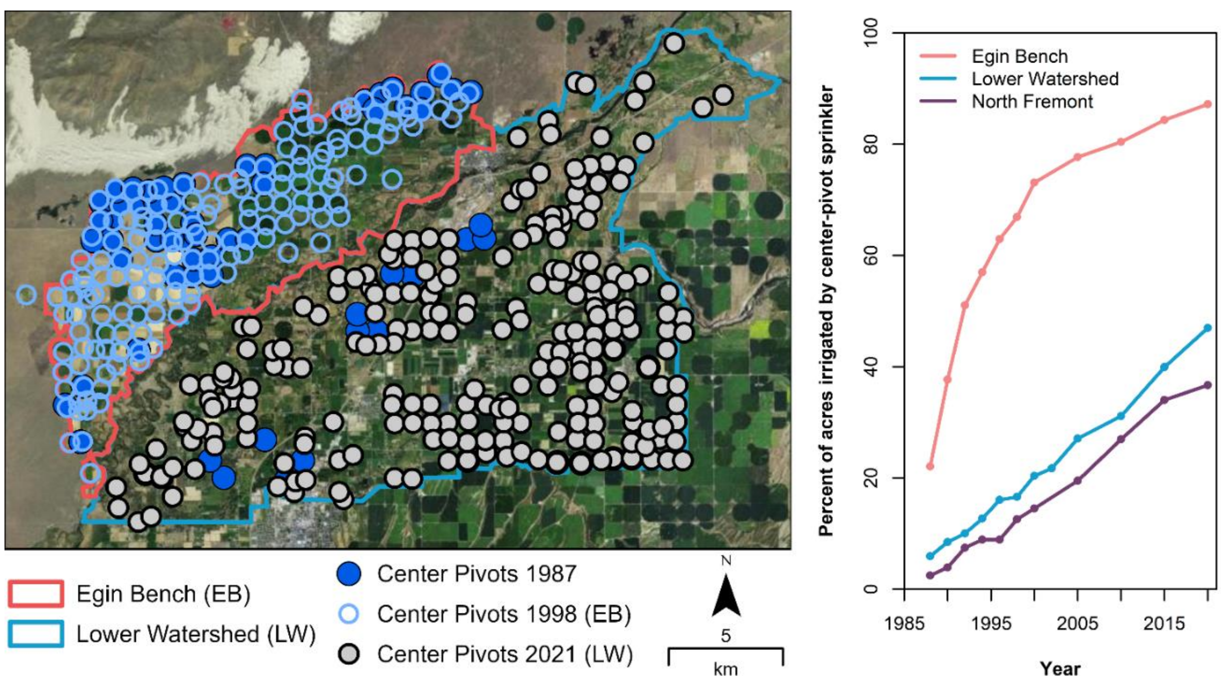

Figure 2-3. Panel A is change in pivot-irrigated acres for Egin Bench (1987–1998) and the Lower Watershed (1987–2021) (Imagery is from USDA FSA NAIP, July 2019). Panel B is percentage of acres irrigated with pivots for all three irrigation study areas for 1988–2020.

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 2001–2022 (Figure 2-3B). Six models accounted for 99.5% of the AICc weight, and all six included terms quantifying the continuous decline from 1978–2000. 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 2-1; Figure A-4). 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.

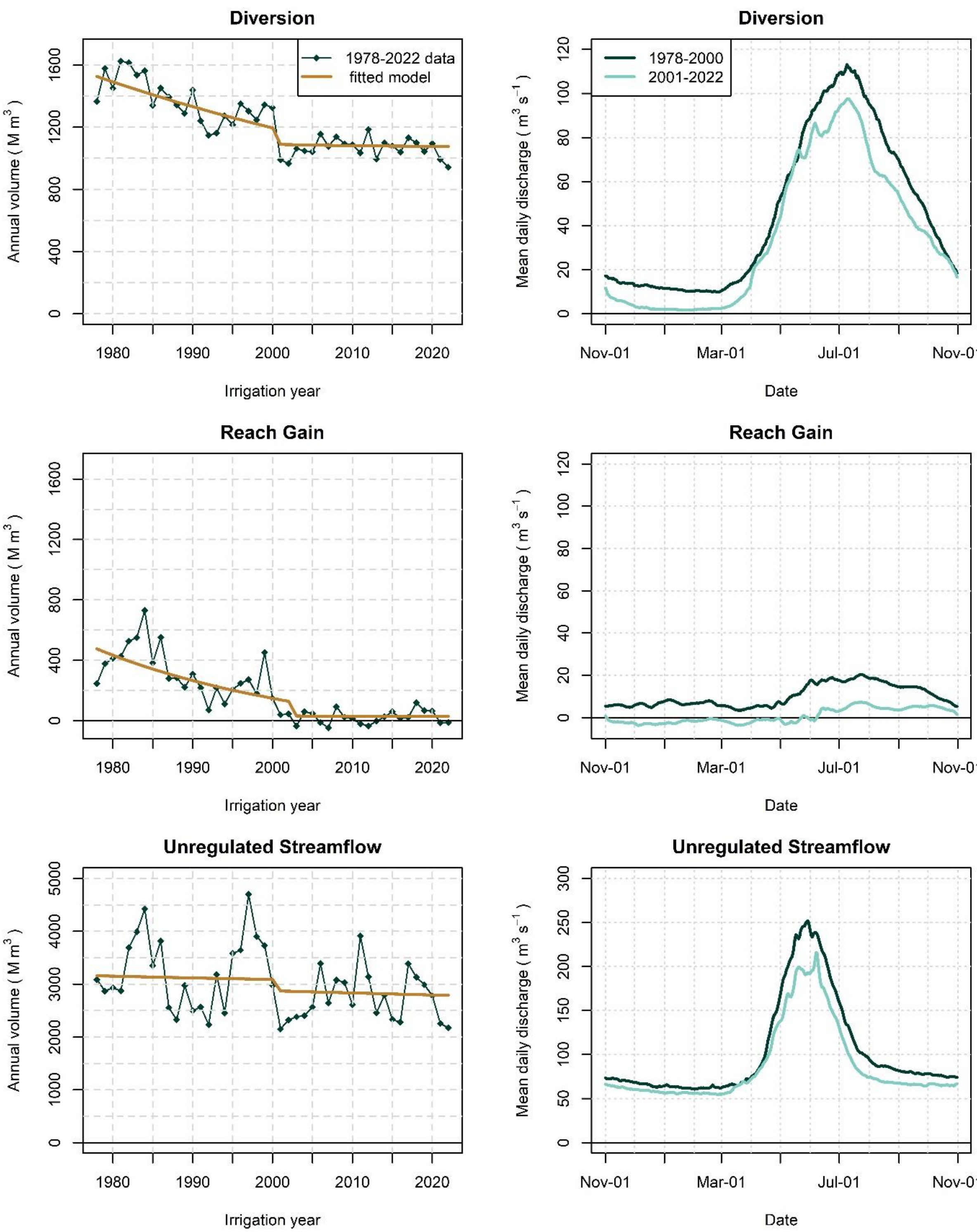

Figure 2-4. Trends in Henrys Fork watershed total diversion, reach gains, and unregulated streamflow for irrigation years 1978–2022.

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 2-4). 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 2-4). 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 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 eight 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 2-4). 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. 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 (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.

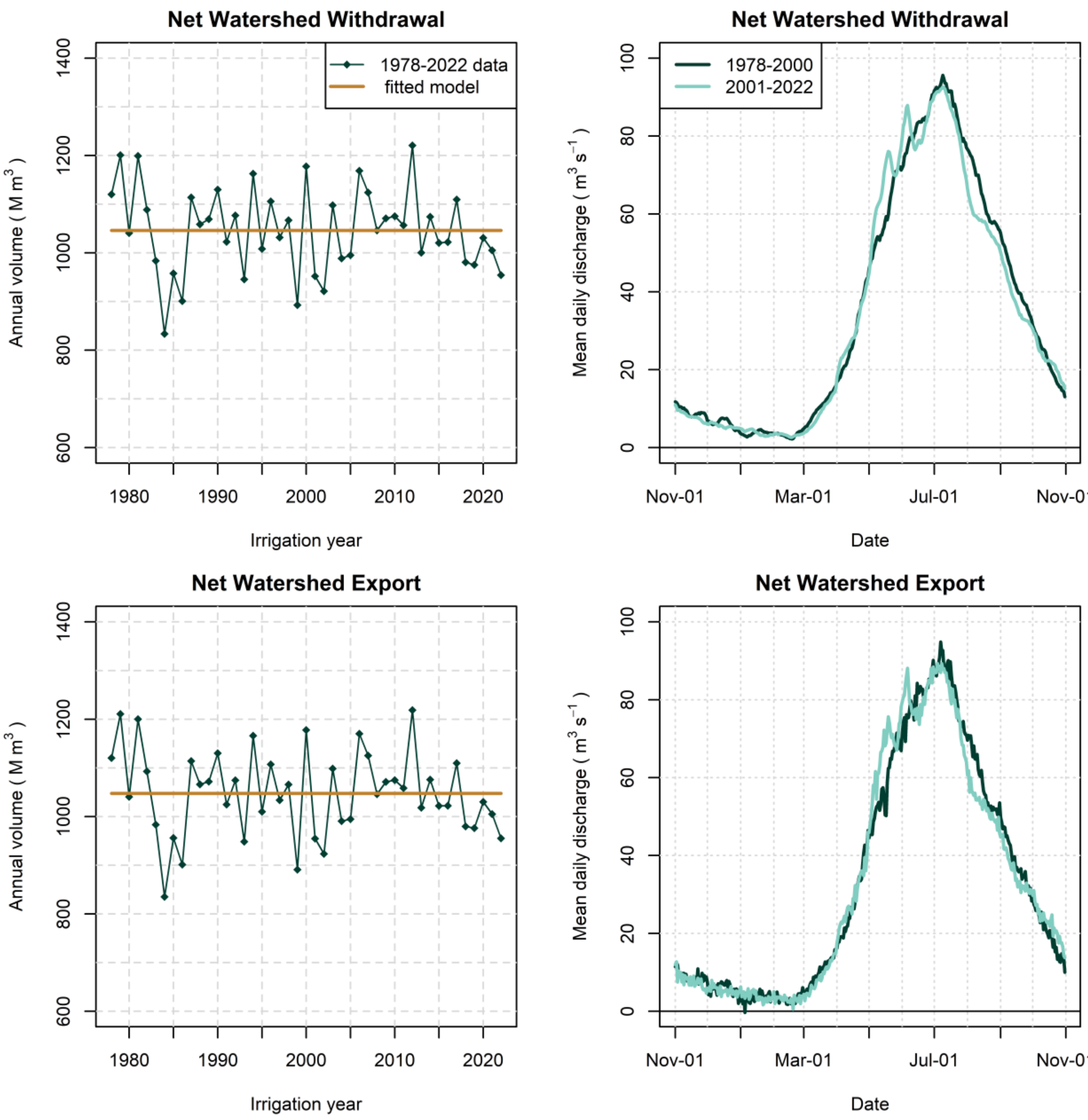

Figure 2-5. Net watershed withdrawal and export in the Henrys Fork watershed for irrigation years 1978–2022.

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 2-5). 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 2-4), net basin export showed little difference between the two time periods across the irrigation year (Figure 2-5).

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 2-3). 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 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.

Table 2-3. 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

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 2001–2022. 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, indicates 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 in the Egin Bench and Lower Watershed combined to center-pivot irrigation also aligns with irrigation conversion to more precise irrigation 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 hours 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 hours (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, Kay and Weatherhead, 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, Quílez and Aragüés, 2006), in the Chiredzi and Runde Rivers in Zimbabwe (Nhiwatiwa, Dalu and Brendonck, 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, Klier and Singh, 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, Donoso and Speelman, 2023). Critics of these subsidies argue that they facilitate increased consumptive use (Huffaker, 2008; Wheeler *et al.*, 2020), favor larger farms (Jordan, Donoso and Speelman, 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, Baker and Guldan, 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 2001–2022 (Figure 2-4). 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 1; Figure 2-3). The rapid rate of conversion on the Egin Bench from 1978–2000 coincides with the decrease in surface-water diversions in the watershed. Conversion on Egin Bench slowed after 2000 (Figure 2-3) 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 2-3). Reduced diversion as a result of irrigation efficiency improvements have also been observed in other studies (e.g. Sando, Borrelli and Brosz, 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, Rosegrant and Ringler, 2003; Toloei *et al.*, 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 1978–2000 before dropping abruptly in 2001, whereas reach gains continued to diminish more gradually through 2002 before stabilizing in 2003–2022. The two-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 2-3). 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 1978–2022 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, Thomas and Parham, 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 2-3) 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 1980–2002 due to irrigation efficiency improvements (Contor, Cosgrove and Johnson, 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, Van Kirk and Null, 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, Bastiaanssen and Immerzeel, 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, Van Kirk and Null (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, Neilson and Null, 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, Adams and Wallender, 2004; Peck *et al.*, 2004; Downard, Endter-Wada and Kettenring, 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).

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 can be achieved 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 three months (Contor, Taylor and Quinn, 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, Auken and Knight, 2019) and characterizing irrigator values, constraints, and enablers can identify potentially effective partnerships (Alonso *et al.*, 2019; Sketch, Dayer and Metcalf, 2020; Zuo, Wheeler and Xu, 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 *et al.*, 2018; Reznik *et al.*, 2022), and rebates on subsequent groundwater pumping fees (Miller, Fisher and Kiparsky, 2021). Lastly, social capital, civic engagement, and capacity building are important for developing cooperative partnerships with irrigators (Lubell, 2004; Alston and Whittenbury, 2011; Sketch, Dayer and Metcalf, 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 contributes to diminishing groundwater contributions to the river (Venn, Johnson and Pochop, 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 (USA) 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).

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

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 in the Egin Bench and Lower Watershed combined to center-pivot irrigation also aligns with irrigation conversion to more precise irrigation 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 hours 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 hours (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 by 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, Kay and Weatherhead, 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, Quílez and Aragüés, 2006), in the Chiredzi and Runde Rivers in Zimbabwe (Nhiwatiwa, Dalu and Brendonck, 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, Klier and Singh, 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, Donoso and Speelman, 2023). Critics of these subsidies argue that they facilitate increased consumptive use (Huffaker, 2008; Wheeler *et al.*, 2020), favor larger farms (Jordan, Donoso and Speelman, 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, Baker and Guldan, 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.

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.

6. 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.

7. Author Contributions

Conceptualization: C.N.M., R.W.V.K.; methodology: R.W.V.K., C.N.M., L.O.B., A.L.H., C.P.; formal analysis: R.W.V.K., C.N.M., L.O.B., A.L.H., C.P.; investigation: R.W.V.K., C.N.M., L.O.B., A.L.H., C.P.; resources: R.W.V.K and S.E.N.; data curation: R.W.V.K., C.N.M., L.O.B., A.L.H., C.P.; writing—original draft preparation: C.N.M.; writing—review and editing: R.W.V.K., S.E.N., L.O.B., A.L.H., C.P.; visualization: C.N.M, R.W.V.K., A.L.H., L.O.B.; supervision: R.W.V.K, C.N.M., S.E.N.; project administration: R.W.V.K.; funding acquisition: R.W.V.K. and C.N.M. As part of 10-week internships with the Henry's Fork Foundation: L.O.B. specifically contributed to the hydrologic time-series analysis; A.L.H. the geospatial analysis; C.P. the irrigator interviews. All authors have read and agreed to the published version of the manuscript.

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10. 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/.

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CHAPTER 3

CAN AQUIFER RECHARGE RECOVER RETURN FLOWS UNDER PRIOR APPROPRIATION IN A WARMING CLIMATE?

Abstract

Groundwater return flow to streams is important for maintaining aquatic habitat and providing water to downstream users, particularly in irrigated watersheds experiencing water scarcity. However, in some agricultural regions, increased irrigation efficiency has reduced return flows and their subsequent in-stream benefits. Agricultural managed aquifer recharge (Ag-MAR)—where recharge is conducted via irrigation canals and agricultural fields—may be a tool to recover these return flows, but implementation is challenged by water rights administration and water availability. Using climate-driven streamflow simulations, an integrated operations-hydrology model, and response functions from a regional groundwater model, we investigated the potential to use Ag-MAR to recover return flows in the Henrys Fork Snake River, Idaho (USA) under future temperatures for water years 2023–2052. We found sites where Ag-MAR is incidental to flood irrigation operations had more water available for recharge both in frequency and volume compared to sites requiring recharge rights, which are junior to agricultural rights. Mean annual recharge volume for the incidental recharge sites was 12% of annual natural streamflow, ranged from 269–335 Mm^3, and was largely available in April and October, reducing the springtime peak flow at the watershed outlet by 10–14% after accounting for return flows. Streamflow contribution from recharge peaked in July and November, increasing July–August streamflow by 6–14% and November–March streamflow by 9–14%. We

demonstrate Ag-MAR can effectively recover groundwater return flows when applied as flood irrigation on agricultural land with senior water rights.

1. Introduction

In agricultural watersheds, water delivered in earthen canals or applied via flood application can percolate into the aquifer, recharging groundwater and returning to the river in a lagged fashion as groundwater flow (Ferencz & Tidwell, 2022; Venn et al., 2004). Groundwater return flows supplement streamflow and, depending on when water returns to the river, can buffer periods of low flow, moderate stream temperature, and benefit water quality (Essaid & Caldwell, 2017; Fernald et al., 2010; Scherberg et al., 2018). Additionally, groundwater return flows may be an inherent part of contemporary basin hydrology in long-irrigated watersheds (Hu et al., 2017; Kendy & Bredehoeft, 2006) where water re-use is essential for aquatic ecosystems and junior water users (Owens et al., 2022). But groundwater return flows are at risk of decline as irrigation efficiency improves and efficient water savings are consumed by other uses (Pérez-Blanco et al., 2020; Scott et al., 2014). More precise irrigation conveyance and application, such as lined canals and sprinklers, reduce the ability for water to infiltrate soils, recharge the aquifer, and sustain return flows (Morrisett et al., In Review).

Agricultural managed aquifer recharge (Ag-MAR) may be a mechanism for recovering groundwater return flows. Ag-MAR uses the existing agricultural landscape—irrigation canals and agricultural fields—to capture excess streamflow, such as floods and snowmelt, for groundwater recharge (Levintal et al., 2023). Ag-MAR is often pursued to expand or recover water supply for groundwater users, particularly in watersheds where aquifers are in decline, but may also be conducted for conjunctive groundwater-surface water management in regions with unconfined aquifers (Miller, Milman, et al., 2021). Flood irrigation and earthen canal operations also often fit under the Ag-MAR umbrella. Although not usually accounted for and monitored in formal management, canal seepage and flood irrigation recharge shallow groundwater and maintain groundwater discharge to rivers (Ochoa et al., 2007). Flood irrigation is a fundamental component in groundwater-surface water management in many irrigation systems (Boelens & Vos, 2012; Oyonarte et al., 2022). Additionally, many modeling studies have demonstrated the effectiveness of Ag-MAR to supplement river base flows (e.g., Alam et al., 2020; Kourakos et al., 2019; Scherberg et al., 2018).

Many studies have considered Ag-MAR from the perspective of site suitability, as reviewed in Sallwey et al. (2019). To be suitable for both groundwater recharge and groundwater return flows to rivers, recharge sites should promote infiltration and not degrade recharged water quality (Dahlke et al., 2018; Ochoa et al., 2013), connect shallow, unconfined aquifers with river channels (Niswonger et al., 2017; van Roosmalen et al., 2009), have sufficient conveyance capacity, and be near enough to rivers for efficient conveyance and subsequent streamflow response. Canal capacity and diversion infrastructure are common Ag-MAR constraints (He et al., 2021; Niswonger et al., 2017). The impact of recharge on streamflow augmentation diminishes with increased distance from the river (Kendy & Bredehoeft, 2006; Kourakos et al., 2019), but recharge conducted too close to the stream channel risks groundwater mounding that waterlogs crops and reduces return flow lag (Fuentes & Vervoort, 2020; Kourakos et al., 2019), minimizing recharge benefits.

The intersection of water law and climate variability in Ag-MAR water availability is more challenging to navigate. The ability to conduct recharge is dependent both on availability of physical water and water rights. Physical water availability is climate-driven and climate variability is increasing (USGCRP, 2018). We do not have a good understanding of how changing flood frequency and timing, as well as other hydroclimate extremes like drought will affect Ag-MAR (Crosbie et al., 2010; He et al., 2021). When physical water is available, water rights may be needed for Ag-MAR operation (Fuentes & Vervoort, 2020; Scherberg et al., 2014; Van Kirk et al., 2020). Using integrated and dynamic water management models is one approach for assessing Ag-MAR feasibility within institutional limitations. For example, Zhao et al. (2021) routed water through an river-reservoir system to quantify water available for recharge while adhering to management rules. In contrast, Goharian et al. (2020) used optimization modeling to investigate how reservoir reoperation could benefit Ag-MAR feasibility. Dogrul et al. (2016) and Ghasemizade et al. (2019) integrated the hydrologic response from Ag-MAR into model simulations. These integrated water management models can investigate how Ag-MAR feasibility and subsequent streamflow response may change with climate. However, integrated operations-hydrology studies that examine water available for Ag-MAR rarely consider the intersection of water law and climate variability (He et al., 2021; Levintal et al., 2023). In this study, we expand the examination of Ag-MAR feasibility for groundwater return flow recovery by asking: Can aquifer recharge recover return flows under prior appropriation in a warming climate?

In the Henrys Fork watershed, Snake River, Idaho (USA)—a semi-arid agricultural region overlying the headwater portion of the Eastern Snake Plain Aquifer—increased irrigation efficiency has reduced groundwater return flow by 299 Mm3 since 1978 (Morrisett et al., In Review). Return flows in the watershed are important for aquatic habitat maintenance and for extending water supply for junior water rights holders. The watershed is located in a major groundwater management region where managed aquifer recharge is currently used for enhancing aquifer storage (Hipke et al., 2022) and recognized as a potential tool for addressing declining river reach gains and return flows (Burchenal et al., 2018; Van Kirk et al., 2020). In this study, we 1) use a watershed-scale irrigation-system operations model paired with climate-driven streamflow simulations to quantify the frequency, magnitude, and timing of possible aquifer recharge within water rights priority over the next 30 years (2023–2052), and 2) use groundwater-surface water response functions from a regional aquifer model to quantify streamflow response from recharge conducted at current and alternative Ag-MAR sites. We use our results to investigate barriers and pathways to implementing Ag-MAR for groundwater return flow recovery.

2. Methods

2.1 Study area

The Henrys Fork is an 8,300 km^2 watershed in the headwaters of the upper Snake River Basin that supports a $10 billion regional agricultural industry and a $50 million recreational fishery (Idaho Water Resource Board, 2009; Van Kirk et al., 2021). Watershed hydrology is snow-dominated and spring-fed (Bayrd, 2006; Benjamin, 2000), generating 3,140 Mm^3 mean annual natural flow (U.S. Bureau of Reclamation, 2012). Three headwater reservoirs (Henrys Lake, Island Park, and Grassy Lake) store a total of 297 Mm^3, which supplements natural streamflow to provide irrigation water for 1,012 km^2 of agricultural land in the lower watershed during irrigation season (April–October; Figure 3-1). On average, 1,400 Mm^3 of surface water and 200 Mm^3 of groundwater are used for agricultural irrigation (Morrisett et al., In Review; U.S. Bureau of Reclamation, 2012). Surface water is diverted from the Henrys Fork and its tributaries and is typically delivered via unlined, earthen canals. Groundwater is pumped from the headwater portion of the Eastern Snake Plain Aquifer, a 28,000 km^2 unconfined aquifer that also provides baseflow to the Henrys Fork and to the Snake River downstream (Hipke et al., 2022). The region largely transitioned from flood to center-pivot sprinkler application in the 1980s, but some flood irrigation continues (Morrisett et al., In Review).

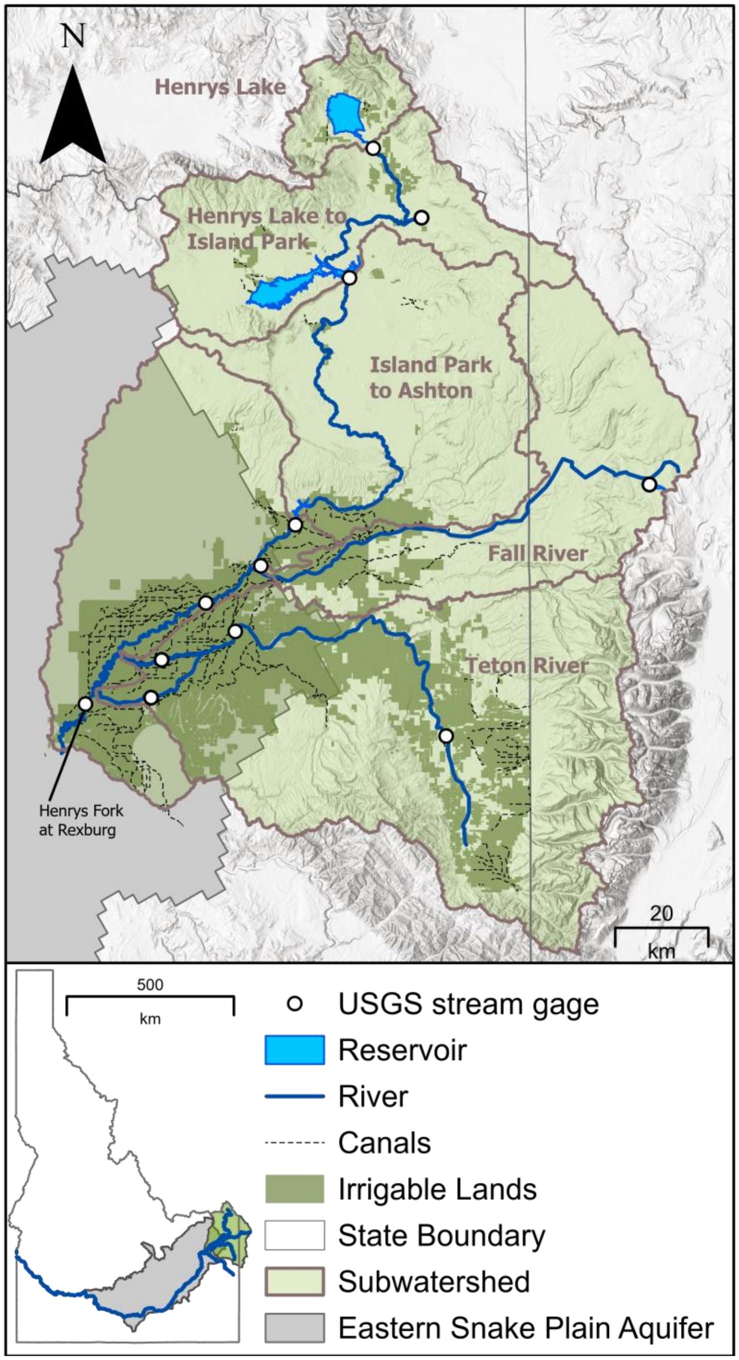

Figure 3-1. The Henrys Fork watershed by subwatershed and relative to the Eastern Snake Plain Aquifer. Data sourced from Idaho Department of Water Resources, Airbus, U.S. Geological Survey (USGS), NGA, NASA, CGIAR, NCEAS, NLS, OS, NMA, Geodatastyrelsen, GSA, GSI and the GIS User Community.

In Idaho, irrigation water is subject to the prior appropriation doctrine and managed via water rights priority (Van Kirk et al., 2019). The basin is fully adjudicated and water resources are conjunctively managed (Stewart-Maddox et al., 2018; Vonde et al., 2016). Aquifer recharge can be either managed or incidental. Water rights for managed aquifer recharge are junior to irrigation and reservoir storage rights and therefore can be used only when all other rights are met. Incidental aquifer recharge is conducted using existing natural streamflow rights for agricultural production that are senior in priority. Incidental recharge occurs via conveyance and application inefficiencies, i.e., canal seepage and flood irrigation during standard irrigation operations. Both managed and incidental aquifer recharge use existing irrigation conveyance, but differ in their site of application. Managed aquifer recharge uses designated basins specifically constructed for recharge, whereas incidental recharge uses existing agricultural fields. The literature captures these recharge modes under the umbrella of Ag-MAR. We use the term Ag-MAR for consistency with the literature, but specify legal differences of recharge types in our study as relevant and necessary. Aquifer recharge definitions may differ in other systems.

2.2 Model description

We used coupled streamflow simulation, reservoir operation, and groundwater-surface water response models to estimate future streamflow response from potential aquifer recharge in the Henrys Fork watershed. Here, we introduce each model and describe how they fit together (Figure 3-2). We describe each model in detail in the following subsections. First, we used statistical modeling to develop 30-year daily time series of climate-based streamflow projections and produced 1,000 independent stochastic replicates of these 30-year time series. We input these simulated time series into an integrated reservoir operations-hydrology model that uses local management rules and stakeholder-identified decision criteria to store and deliver water across the watershed, conforming to basin-wide water rights. For each of the 1,000 time series, the operations-hydrology model outputs the daily volume of water available for recharge at each of five current and potential recharge sites. Finally, we input daily recharge availability into site-specific streamflow response functions from the state of Idaho's regional groundwater model to produce 1,000 independent 30-year replicates of daily streamflow gain from aquifer recharge.

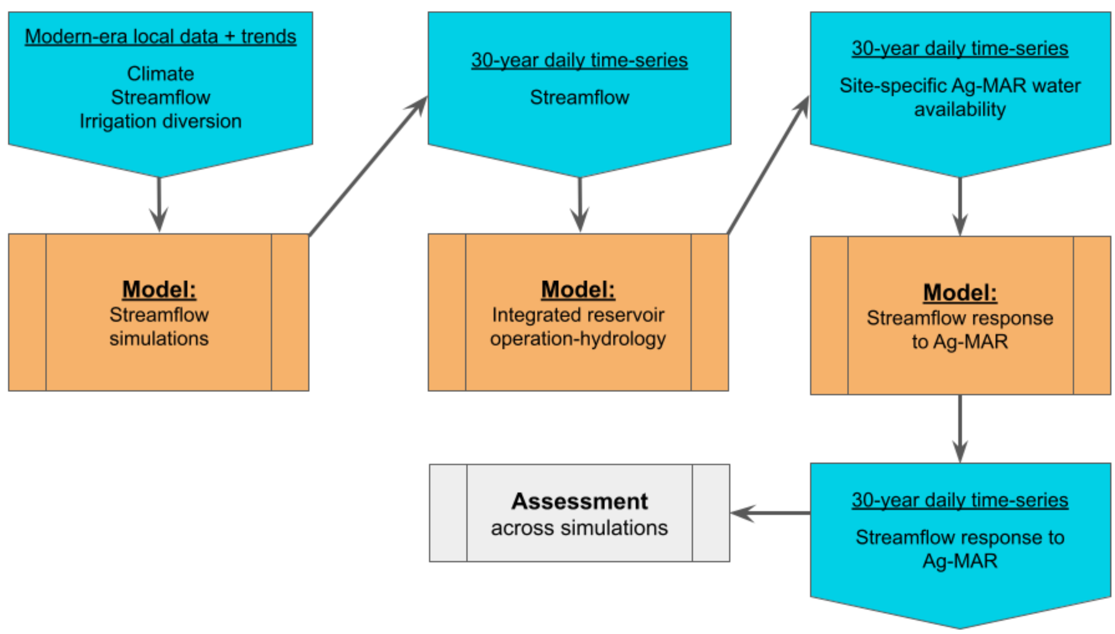

Figure 3-2. Conceptual diagram of model coupling, with input and output data flows.

2.2.1 Streamflow simulations

Ag-MAR feasibility depends on water available for recharge, which in our study system is natural streamflow in excess of that needed to meet irrigation demand and fill storage reservoirs. Before simulating water available for Ag-MAR, we generated daily time series of potential future natural streamflow for the Henrys Fork watershed with statistical modeling and model selection using Akaike's Information Criterion with small sample-size correction (AIC; Claeskens & Hjort, 2008). For the modeling process, we used publicly available hydrometeorologic data collected by state and federal agencies over water years 1989–2022 (Van Kirk, 2017a), where a water year is defined as October 1–September 30. We chose this set of water years because it was the longest period of record common to a set of 12 stations that represent the full spatial variability of subwatershed-scale climate across all areas of the watershed. This record was short enough that it excluded climatic conditions unlikely to be experienced in the future, but long enough to detect current temporal trends in climate parameters. Where such trends were detected, we projected those 30 years into the future beyond 2022 conditions to simulate the effects of ongoing climate change on streamflow. We used the full output of statistical models—including means, variances and temporal autocorrelation—to generate probability distributions of each streamflow parameter in each future year. We used these distributions to generate 1,000 independent, random 30-year streamflow time series that represent different realizations of streamflow possible over water years 2023–2052. We selected 30 years as a modeling time frame because it coincided with water-resources planning horizons, was long enough for effects of initial conditions to decay, and was greater than the 10–20 years required for aquifer response to reach steady state. The model contained roughly 50 stochastically generated inputs, so we selected 1,000 replications to ensure the full variability of possible future conditions would be captured, within reasonable computational burden. Capturing such variability also captures some of the error and uncertainty in the three coupled models.

To align simulated streamflow with irrigation-system operation, we generated natural streamflow regression models for each of five subwatersheds: 1) Henrys Lake, 2) Henrys Lake to Island Park, 3) Island Park to Ashton, Fall River, and Teton River (Figure 3-1). For each of these five subwatersheds, we generated three seasonal streamflow measures: October–March (base flow) volume, April–September (runoff) volume, and April–September hydrograph centroid (center of mass). We considered five potential climate predictor variables for these three streamflow measures: 1) precipitation, 2) temperature, 3) evapotranspiration, 4) snow water equivalent (SWE), and 5) the one-year average difference between precipitation and evapotranspiration, a surrogate for accumulated soil moisture surplus/deficit. We also considered streamflow over the previous semi-annual period as a potential predictor of streamflow over the subsequent period. We assessed all response and predictor variables for normality and natural-log transformed all right-skewed quantities prior to modeling. We used automated AICc model selection with the dredge function in R (Bartoń, 2022) to identify the best predictors. We assessed correlation among predictors and resulting variance inflation factor in the top model or models (Petrie, 2020; R Core Team, 2022; Wei & Simko, 2021), eliminating predictors and re-running model selection accordingly to produce final models that used a parsimonious set of relatively uncorrelated predictors and met all distributional assumptions. The latter was critical to use of these distributions to generate the stochastic replicates. We also assessed model inputs and outputs for temporal autocorrelation. This analysis confirmed model residuals were independent and autocorrelation inherent in observed streamflow time series was preserved in the simulated time series. In all cases where autocorrelation was observed, it appeared in simulations via dependence of semi-annual streamflow volume on that in the preceding semi-annual period. For example, October–March streamflow volume appeared as a predictor in the AICc-selected final model of subsequent April–September streamflow volume.

Once all streamflow regression models were selected, we analyzed temporal trends in each climatic variable that appeared in the final models. We also used AICc model selection for this analysis, but proposed a set of only four candidate models for each climate variable: a null model (constant mean through time), first-order autoregressive (constant mean but with one-year autocorrelation), trend (linear trend in time), and trend with first-order autocorrelation. These models were fit using R's arima function (R Core Team, 2022). We then used the best of these four models by AICc, including variance and autocorrelation, to simulate future time series of these variables that extended the observed 1989¬–2022 time series 30 years into the future. We used the arima.sim function in R to generate 1,000 independent, random time series of each climate variable. The regression model selection procedure described above ensured the climate variables were sufficiently uncorrelated so that we did not need to model cross correlation among them. Generally, we found increasing trends in variables related to temperature and evapotranspiration similar to a regional climate assessment (Hostetler et al., 2021), but no significant trend in SWE and other precipitation variables.

To simulate a 30-year series of each streamflow parameter, we stepped sequentially through each semi-annual period—starting with observed streamflow during the April–September period that ended water year 2022 and the first set of October–March climate variables simulated in a given climate time series. We used these predictors as inputs to the appropriate regression model for each respective subwatershed. The mean and variance defined by the regression model outputs, in addition to observed correlation among the subwatersheds, produced a mean vector and spatial covariance matrix defining the multivariate normal distribution of seasonal streamflow volume for the five subwatersheds. We then used the mvrnorm function in R to select a random set of streamflow volumes from the multivariate distribution. Those streamflow volumes and the second set of climate variables were then input to the appropriate regression models for the April-September period and so on until the 30-year series was complete. We repeated this procedure 1,000 times, each using a different one of the 1,000 climate time series as inputs. This resulted in 1,000 independent random 30-year time series of semi-annual subwatershed streamflow volumes and runoff-period centroids.

Next, we generated daily-scale hydrographs for natural streamflow. After preliminary analysis of observed October–March (base flow) hydrographs, we found no measure of hydrograph shape that could be predicted based on any other hydrometeorologic variable. Thus, we randomly selected years from 1989–2022, converted the observed base-flow hydrographs for each subwatershed from that year into unitless hydrographs, and multiplied by the semi-annual volume generated in the previous modeling step to obtain a dimensioned hydrograph. These same analog years and methods were also used to calculate time series of daily precipitation and evaporation, which were needed to calculate reservoir gain/loss from precipitation and evaporation in the operations model. April–September hydrographs were defined by the simulated centroids. We used an analog-year approach to selecting hydrographs by first constructing a multivariate normal distribution around the simulated watershed centroids for a given April–September period with the pmvnorm function in R (Genz et al., 2021). We then calculated the probability of obtaining the observed centroid vectors in each of water years 1978–2022 from that multivariate distribution. We used this longer period of record to increase the number of possible outcomes and combinations of runoff timing and streamflow volume. We then drew a random year from that set using those probabilities as weights. That produced an analog year that had a high probability of occurring, given the simulated centroids. We scaled the runoff-period hydrographs for each subwatershed with the simulated streamflow volumes to produce resulting April-September daily-scale hydrographs for each subwatershed. We found little discontinuity at the boundary of the October–March and April–September hydrographs, but we smoothed the boundaries between water years with a centered, 7-day moving average to prevent unrealistic or computationally problematic jumps in the operations model at the beginning of each new water year.

The last inputs needed for the operations model were daily time series of irrigation diversion and stream reach gains/losses, the latter reflecting interactions with aquifers in the lower watershed. We used the same model selection procedure describe above for these quantities but used irrigation years (November 1–October 31) instead of water years, modeled total diversion for the entire watershed before determining reach-specific diversion, and limited the analysis to years 2004–2022. We chose irrigation years for consistency with reporting by the state water resources agency. We used watershed-total diversion as the model response variable because diversions from specific stream reaches and tributaries are interconnected, with many locations in the watershed served by diversions from multiple sources. Finally, we used the shorter time period both because irrigation practices changed dramatically around the year 2000 (Morrisett et al., In Review) and because of shorter periods of gaging records required to calculate these quantities. We predicted total annual diversion by variables related to water supply, but we found no predictors for reach gains/losses. The output of the diversion regression model generated a normal distribution for watershed-total diversion in the particular simulation year. We then calculated the probability of selecting each of years 2004–2022 from that distribution with the pnorm function in R and used these probabilities as weights to randomly select an analog diversion year. Then, we used the actual observed reach-, tributary-, and canal-specific diversions and reach gains/losses from that year in the simulations.

2.2.2 Integrated reservoir operation-hydrology model

We developed a daily integrated reservoir operations-hydrology model for the Henrys Fork watershed that inputs the 30-year streamflow time series for each for the 5 reaches, dynamically routes water through storage reservoirs, rivers, and diversions, and then outputs the water available for Ag-MAR based on water supply, water right priorities, and local collaborative management criteria. This model was based on a seasonal model successfully used by stakeholders in the watershed since 2017 to plan operations for the upcoming irrigation season (Joint Committee, 2018; Van Kirk, 2017b, 2021). We ran the model for each of the 1,000 streamflow, diversion, and reach gain/loss replicates. This approach was similar to other studies (e.g., Ghasemizade et al., 2019; Goharian et al., 2020; Niswonger et al., 2017; Zhao et al., 2021).

The model reflects modern irrigation-system operation, as specified and constrained by basin-wide water rights administration, agency policies and jurisdiction, and local stakeholder agreements (Olenichak, 2020). The total water supply available to meet irrigation demand on any given day during the April–October irrigation season is the sum of natural streamflow, reservoir storage delivery, river reach gains (negative if the reach loses water to the aquifer) and groundwater injected into the surface-water system, less watershed stream outflow downstream of all diversions. The latter is constrained by a minimum target outflow set by local stakeholders on the mainstem Henrys Fork (Figure 3-3; Appendix B). When natural streamflow plus reach gain exceeds the sum of diversion and the outflow targets on the Teton River (Appendix B), reservoirs are either filled or held at full pool. Once natural streamflow plus reach gain is insufficient to meet diversion and streamflow targets, reservoir storage is released as needed to make up the difference. Outflow is held at the target flows until irrigation demand drops at the end of the season and reservoir delivery is no longer needed. At that point, reservoirs begin to store water again according to basin-wide fill strategy, stakeholder input, and infrastructure constraints during the period of reservoir ice cover. We first ran the full model without implementing Ag-MAR to obtain simulated streamflow at the bottom of the watershed (Henrys Fork at Rexburg streamflow gage; Figure 3-1) in absence of Ag-MAR, for the purposes of quantifying the net effect of Ag-MAR.

2.2.3 Site-specific Ag-MAR water availability

Ag-MAR water availability is dependent on water rights and conveyance infrastructure to recharge sites. We identified five sites for potential Ag-MAR (Figure 3-3) using local knowledge of the agricultural landscape. Two sites (Egin Lakes and St. Anthony Union Canal) were identified as sites designated for *managed* aquifer recharge and have the appropriate water rights. Three sites (Wilford, Sugar-Salem, and Burton) were identified as suitable for *incidental* aquifer recharge via agricultural water rights.

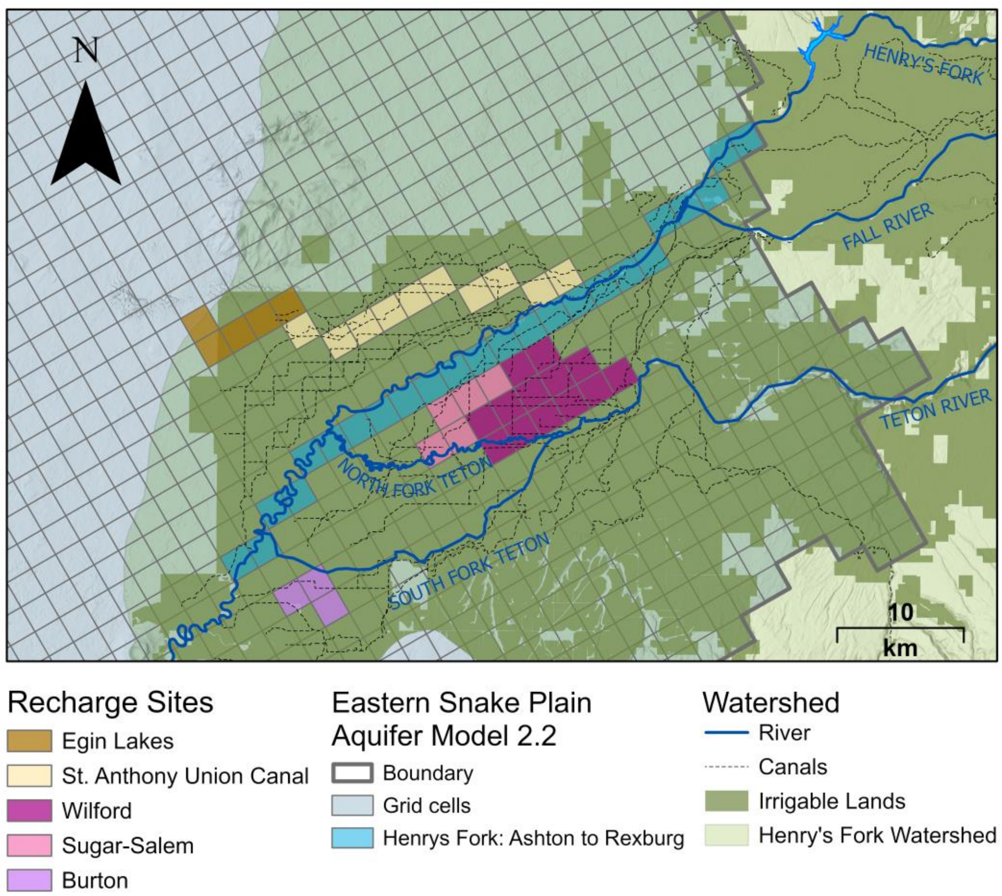

Figure 3-3. Five potential Ag-MAR sites relative to their Eastern Snake Plain Aquifer Model (ESPAM) Version 2.2 grid cells and modeled river reach of interest (target reach). ESPAM data sourced from the Idaho Department of Water Resources. Other data sourced from Airbus, U.S. Geological Survey, NGA, NASA, CGIAR, NCEAS, NLS, OS, NMA, Geodatastyrelsen, GSA, GSI, and the GIS User Community.

Egin Lakes is a state-designated managed aquifer recharge (MAR) basin, requiring water available only under junior MAR rights. These rights can be available at any time of year, if natural streamflow exceeds that needed for all senior irrigation, storage, and hydropower rights in the upper Snake River basin, as well as a minimum streamflow constraint on the mainstem Henrys Fork (Idaho Department of Water Resources, 1999; Appendix B). We also imposed a constraint preventing Ag-MAR during draft of the largest upstream reservoir (Appendix B). Previous work quantified excess streamflow available for MAR at Egin Lakes, finding that it was available only in relatively small subset of irrigation years in their 1980–2014 study period (CH2M & Henry's Fork Foundation, 2016). The year of lowest water supply in that study in which substantial MAR water was available at Egin Lakes was irrigation year 2000, where total natural streamflow was 3,010 Mm^3. Thus, we used watershed-total natural flow in that irrigation year as the threshold above which MAR was available and below which it was not in simulated years. We also allowed MAR at Egin Lakes during October and November in years when Island Park Reservoir storage at the end of irrigation season was higher than its winter operational capacity and the excess was evacuated between October 1 and November 15. Next, MAR availability at Egin Lakes was capped at the 2.8 m^3/s capacity of the recharge site. Lastly, we considered capacity of the St. Anthony Union Canal, which delivers water to the Egin Lakes site (Table 3-1). If Egin Lakes MAR water was available during the irrigation season, only canal capacity more than existing irrigation needs was allowed to convey MAR water to Egin. If Egin Lakes MAR water was available during the off-season, we allowed recharge to occur only during November and March because ice, snow, and cold weather prevent operation there during the middle of the winter. In that case, Idaho water law allows canal seepage to count toward MAR, and we allowed diversion beyond the site capacity but only up to the total amount of seepage the canal can accommodate, which is roughly 2.8 m^3/s based on data reported by Apple (2013). This allows the canal itself to serve as a recharge site during November and March.

Table 3-1. Recharge sites, site area, and canal capacities to each site for water year 2001–2022. Steady-state streamflow response in the target reach was attained in ~20 years at all five of our sites.

Recharge site type	Recharge site name	Site Area (km^2)	Canal	Canal capacity (m^3/s)	Steady-state streamflow response in target reach
Managed	St. Anthony Union Canal	25.9	St. Anthony Union	13.9	50.7%
	Egin Lakes	12.9	Recharge Canal	2.8	65.9%
Incidental	Sugar-Salem	12.9	Salem Union	8.9	71.8%
			Consolidated Farmers	9.1	
	Burton	7.8	Rexburg Irrigation	8.1	48.8%
	Wilford	33.7	Crosscut Canal (Southeast)	8.9	69.3%
			Fall River Canal	7.1	
			Farmers Friend	7.0	
			Wilford	5.7	
			Pioneer Ditch	0.8	
			Stewart Ditch	1.2	

The remaining three sites are not formal recharge basins but can conduct incidental recharge via flood irrigation using senior natural-streamflow irrigation rights, when these rights are in priority during the April-October irrigation season. These sites were identified as locations where flood irrigation was still in operation or could be restored based on local infrastructure and geography (Morrisett et al., In Review). Roughly, these natural-streamflow water rights are in priority when reservoir delivery is not needed to meet irrigation demand, so in the model we allowed Ag-MAR diversion at these sites between

April and October whenever reservoir storage was not needed. This was implemented in the model by identifying times when watershed outflow was at the minimum stakeholder-determined target. When water was available at these sites, we allowed diversion up to the maximum capacity of canals that can deliver water to these sites (Table 3-1), less existing irrigation water conveyed by the canals, as determined by actual diversion during the analog year used in the operations model. Diversion for Ag-MAR was further constrained by maintaining watershed outflow no lower than the stakeholder-determined minimum streamflow targets currently used to manage the irrigation system. Lastly, we considered seepage capacity at each of the sites. Seepage rates in the study area range from around 90 mm/day on typical soils to 150 mm/day at the Egin Lakes recharge basin to over 900 mm/day in canals (Apple, 2013; Contor et al., 2009; Peterson, 2011; Wytzes, 1980). Even the lower end of these estimates is high enough that the maximum amount of water that can be delivered to each recharge site (including Egin and St. Anthony Union Canal) can be recharged in the area available (Table 3-1). Thus, we did not constrain recharge by seepage rate. In the study area, these seepage rates are around one order of magnitude greater than evapotranspiration (U.S. Bureau of Reclamation, 2012), so we ignored evaporation and assumed all water delivered to the site according to the water rights, conveyance capacity, and streamflow target criteria was recharged.

The Ag-MAR subroutine in the operations-hydrology model was dynamic. If streamflow was allocated for Ag-MAR diversion at another site, it was removed from streamflow volume available to other recharge sites downstream. In the model, we prioritized Ag-MAR at the two designated managed aquifer recharge sites because recharge counts administratively toward basin-wide MAR objectives set by the state. Then, we prioritized the incidental recharge sites (in order: Sugar-Salem, Burton, Wilford), based on the physical arrangement of canals that can deliver water to those areas.

2.2.4 Modeling streamflow response to Ag-MAR

We used the Eastern Snake Plain Aquifer Model Version 2.2 (ESPAM2.2) to model streamflow response to Ag-MAR. ESPAM2.2 and is a regional, finite-difference groundwater accounting model programmed in MODFLOW that estimates the effect of aquifer stresses on rivers and springs hydraulically connected to the aquifer (Cosgrove et al., 2006). It has 1.6 km^2 grid cells. The Idaho Department of Water Resources conducted specific runs of ESPAM2.2 for this research to generate 30-year transient streamflow response functions for each of our five recharge sites. These response functions give the fraction of a given recharge event realized as streamflow in each subsequent two-week period following the recharge event. We interpolated these response functions to daily resolution and applied them to the recharge conducted on each day at each site in the model to obtain streamflow response in the Ashton-Rexburg target reach in all future days in the 30-year simulation (Sukow, 2021). We used an efficient convolution routine in the Matrix package in R (Bates et al., 2022) to conduct these calculations. The target reach is the finest-resolution reach in our study area built in to ESPAM2.2. Steady state streamflow response in this reach was attained by about 20 years at all five of our sites. At steady state, the percentage of recharged water realized as streamflow gains in the target reach was 50.7% and 65.9% for the two sites designated for managed recharge and 48.8%, 69.3, and 71.8% for the three incidental recharge sites (Table 3-1). The remaining water recharged in the study area produces streamflow increases in other reaches of the Snake River downstream of the Henrys Fork watershed.

Streamflow gains to the river from Ag-MAR were not dynamically included in the integrated operations-hydrology model. Instead, we accounted for them in the following post-hoc calculations:

1. First, all water diverted for Ag-MAR was subtracted from streamflow at the watershed outlet as calculated under the base scenario (no-Ag-MAR) to assess the potential effect of Ag-MAR diversion on peak flow characteristics important for ecological function in the lower part of the watershed.
2. When watershed stream outflow was at the stakeholder-determined minimum target in the base scenario (equivalent to times when reservoir storage is needed to meet irrigation demand), we subtracted the additional streamflow from Ag-MAR from the amount delivered from Island Park Reservoir, consistent with the goals of the stakeholder group (Joint Committee, 2018). This allowed us to assess the potential effectiveness of Ag-MAR in reducing the need for reservoir storage in the future.
3. When watershed stream outflow was greater than the minimum target in the base scenario, the additional streamflow from Ag-MAR was added to streamflow at the watershed outlet as calculated in step 1 above.

2.2.5 Ag-MAR assessment across streamflow simulations

Finally, we use summary statistics across simulations to evaluate the magnitude and timing of Ag-MAR and return flows, and how they vary across potential recharge sites. Lastly, we investigate specific cases that enable or hinder Ag-MAR. All analyses and visualizations were conducted in R Version 4.2.2 (R Core Team, 2022).

3. Results

3.1 Streamflow simulations

Total streamflow volume in April–September was always a predictor for total streamflow volume in the subsequent October–March for all reaches/nodes (Table 3-2). Fall precipitation was a common predictor for total October–March streamflow volume and was included in models for Henrys Lake inflow, Fall River, and the Teton River, whereas accumulated soil moisture was important for Henrys Lake to Island Park and Island Park to Ashton. Temperature in October–March was also a predictor for October–March streamflow volume for Henrys Lake to Island Park. For all reaches/nodes, total streamflow volume in October–March, snow water equivalent (SWE) on April 1, and precipitation in April–June were always predictors for total streamflow volume in the subsequent April–September. Evapotranspiration was also important for Fall River and the reach between Henrys Lake and Island Park. Predictors for the April–September total streamflow volume center-of-mass were more variable, but SWE on April 1 was a predictor for all reaches/nodes excluding the Teton River. April 1 SWE and total streamflow volume in the preceding October–March were predictors for total-watershed streamflow diversions in April–October. Predictors for each hydrologic parameter for all reaches or nodes are noted in Table 3-2.

Table 3-2. The climate and streamflow predictors used to model a given hydrologic parameter and the time-series component for each predictor used in the 30-year simulated projection. We also share the adjusted R^2 value for each hydrologic parameter model. Q refers to streamflow volume at the given node/reach, center-of-mass is a water year day, SWE, $Precip$, and $Temp$ refer to the snow water equivalent, precipitation, temperature for a given subwatershed, and $Moist$ refers to watershed-wide accumulated soil moisture, or the daily precipitation minus evapotranspiration.

Subwatershed	Node/Reach	Hydrologic Parameter	R^2	Predictor	Predictor time-series model
Upper Henrys Fork	Henrys Lake Inflow	$Q_{Oct-Mar_t}$	0.79	$\log(Q_{Apr-Sep_{t-1}})$	Null
				$Precip_{Oct-Nov_t}$	Null
		$Q_{Apr-Sep_t}$	0.90	$\log(Q_{Oct-Mar_{t-1}})$	1-year lag
				$\log(SWE_{Apr1})$	Null
				$Precip_{Apr-Jun_t}$	Null
		$CenterOfMass_{Apr-Sep_t}$	0.69	$\log(SWE_{Apr1_t})$	Null
				$Precip_{Apr-Jun_t}$	Null
				$\log(Precip_{Jul-Sep_t})$	Null
Upper Henrys Fork	Henrys Lake to Island Park	$Q_{Oct-Mar_t}$	0.91	$\log(Q_{Apr-Sep_{t-1}})$	1-year lag
				$Temp_{Oct-Mar_t}$	Trend
				$Moist_{Oct-Nov_t}$	5-year lag
		$Q_{Apr-Sep_t}$	0.89	$\log(Q_{Oct-Mar_{t-1}})$	1-year lag
				$\log(SWE_{Apr1_t})$	Null
				$Precip_{Apr-Jun_t}$	Null
				$\log(ET_{Jul-Sep_t})$	Trend
		$CenterOfMass_{Apr-Sep_t}$	0.42	$\log(SWE_{Apr1_t})$	Null
				$\log(ET_{Jul-Sep_t})$	Trend
Upper Henrys Fork	Island Park to Ashton	$Q_{Oct-Mar_t}$	0.75	$\log(Q_{Apr-Sep_{t-1}})$	1-year lag
				$Moist_{Oct-Nov_t}$	5-year lag
		$Q_{Apr-Sep_t}$	0.86	$\log(Q_{Oct-Mar_{t-1}})$	1-year lag
				$\log(SWE_{Apr1})$	Null
				$Precip_{Apr-Jun}$	Null
		$CenterOfMass_{Apr-Sep_t}$	0.65	$\log(SWE_{Apr1})$	Null
				$Temp_{Jul-Sep_t}$	Trend
Fall River	Fall River	$Q_{Oct-Mar_t}$	0.77	$\log(Q_{Apr-Sep_{t-1}})$	Null
				$Precip_{Oct-Nov_{t-1}}$	Null
		$Q_{Apr-Sep_t}$	0.94	$\log(Q_{Oct-Mar_{t-1}})$	1-year lag
				$\log(SWE_{Apr1_t})$	Null
				$Precip_{Apr-Jun_t}$	1-year lag
				$\log(ET_{Jul-Sep_t})$	Trend
			0.78	$\log(SWE_{Apr1})$	Null

		$CenterOfMass_{Apr-Sep_t}$		$Temp_{Apr-June_t}$	Null
Teton River	Teton River	$Q_{Oct-Mar_t}$	0.68	log $(Q_{Apr-Sep_{t-1}})$	Null
				$Precip_{Oct-Nov_{t-1}}$	Null
		$Q_{Apr-Sep_t}$	0.92	log $(Q_{Oct-Mar_{t-1}})$	1-year lag
				log (SWE_{Apr1})	Null
				log $(Precip_{Apr-Jun})$	Null
		$CenterOfMass_{Apr-Sep_t}$	0.72	$Temp_{Apr-June_t}$	Trend
				$Moist_{Jul-Oct_t}$	1-year lag
Henrys Fork Watershed	NA	$Diversion_{Apr-Oct_t}$	0.34	log $(Q_{Oct-Mar_{t-1}})$	1-year lag
				log (SWE_{Apr1})	Null

Most predictors did not have a time-series component (Table 3-2; Figure B-2–B-5). However, total streamflow volume in October–March exhibited a 1-year lag for all reaches/nodes and at the watershed scale. Total streamflow volume in April–September contained a 1-year lag for Henrys Lake to Island Park, Island Park to Ashton, and Fall River. Temperature in the Upper Henrys Fork and Teton subwatersheds demonstrated a positive trend across seasons (Table 3-2; Figure B-3–B-4). Watershed-scale accumulated moisture in October–November included a 5-year lag (Figure B-2) and a 1-year lag for July–October (Figure B-5).

We initialized our 30-year streamflow simulations on October 1, 2022, starting with the watershed conditions inherited from water year 2022—a significant drought year for the Henrys Fork watershed (Van Kirk, 2023). Overall, our streamflow simulations predicted the 2022 drought would take several years to recover from (Figure 3-4). When we removed this drought recovery period and considered the next 25 years in the simulation, two of the five subreaches demonstrated a decreasing trend in mean annual streamflow volume across all 1,000 simulations: 1) Henrys Lake to Island Park and Fall River. Additionally, for the 30-year study period, the April–September centroid date was predicted to shift to earlier in the spring in Henrys Lake to Island Park, Island Park to Ashton, and the Teton River (Figure 3-4). Alternatively, the April–September centroid date was predicted to advance to later in the spring for Henrys Lake inflow and Fall River.

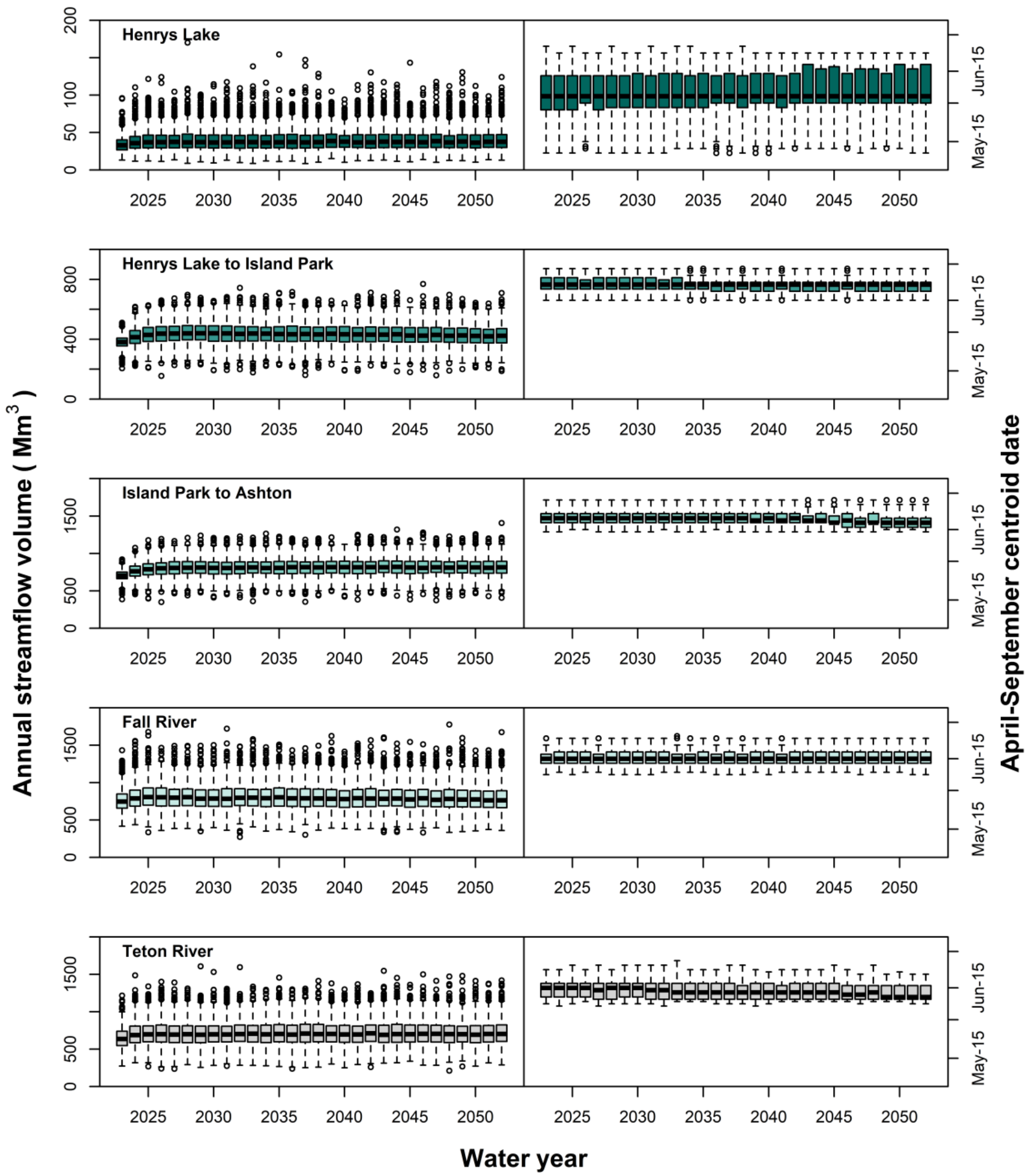

Figure 3-4. Boxplots of streamflow prediction and the April–September centroid date for each modeled stream reach for each year in the 30-year time series (2023–2052) across 1,000 simulations. Note the y-axis range for Henrys Lake and Henrys Lake to Island Park are different than the other three subreaches.

3.2 Simulated Ag-MAR water availability

Ag-MAR was available in all water years across all simulations, regardless of total annual streamflow, and there was little variability among years (Figure B-2). Summing across all recharge sites, total mean annual water available for Ag-MAR was 12% of annual natural streamflow (Figure 3-5). On average, total annual streamflow was 2,789 Mm^3 and 349 Mm^3 was available for Ag-MAR in any given year. The three *incidental* recharge sites were simulated to receive the most recharge volume via the integrated operations-hydrology model, per water rights priority (Figure 3-5). On average, 43% of water available for Ag-MAR could be recharged at Wilford, 37% at Sugar-Salem, and 15% at Burton. In our simulations, the two recharge sites with or associated with recharge water rights—Egin Lakes and St. Anthony Union Canal—received 4% and 1% of total water available for Ag-MAR on average, respectively.

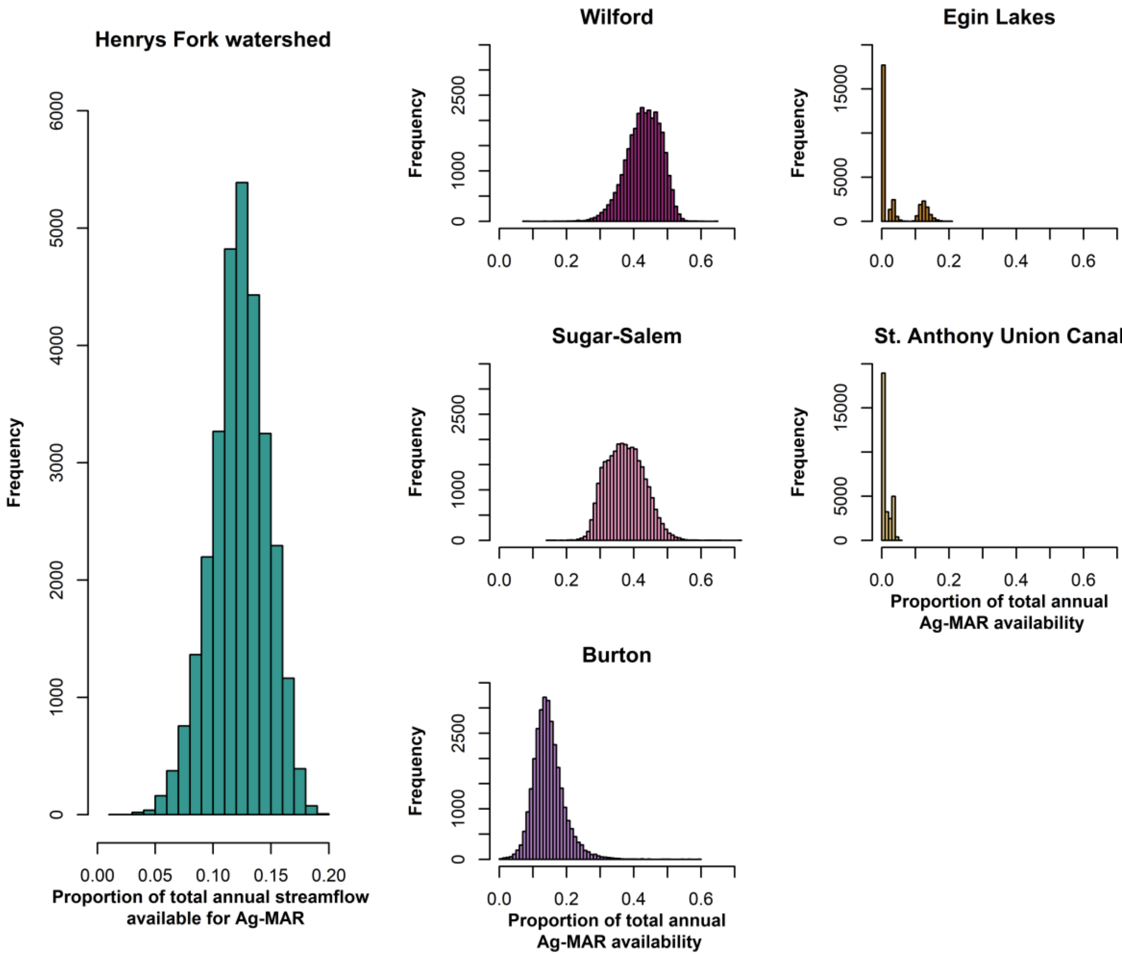

Figure 3-5. For all water years, the proportion of total annual streamflow available for Ag-MAR for the Henrys Fork watershed and the proportion of total annual Ag-MAR water availability conducted at each site.

A total annual streamflow threshold of 3,010 Mm^3 was required to conduct Ag-MAR at Egin Lakes or St. Anthony Union Canal. Across all simulations, 74% of years were below the threshold. Thus, the opportunity to recharge at Egin Lakes and St. Anthony Union Canal occurred in 26% of water years. Through a given 30-year time series, water was only available for Ag-MAR at Egin Lakes a median of 8 years (27%). Ag-MAR was never available every year within a 30-year time series, but 17.8% of simulations had water available for Ag-MAR in 10–18 years of the 30-year time series. For these sites, the recharge season is 275 days. In years when water was available for junior managed aquifer recharge rights, Ag-MAR could be conducted an average of 63–86% days at Egin Lakes and 21–22% days at St. Anthony Union Canal (Figure 3-6). In contrast, water was available for Ag-MAR at the three *incidental* recharge sites (Sugar-Salem, Wilford, and Burton) in 100% of water years for all 30-year time series simulations (Figure 3-6). Ag-MAR at these sites was limited to the 213-day irrigation season. On average, in any given irrigation season, Ag-MAR could be conducted during 59% of the season at Wilford, 78% at Sugar-Salem, and 62% at Burton.

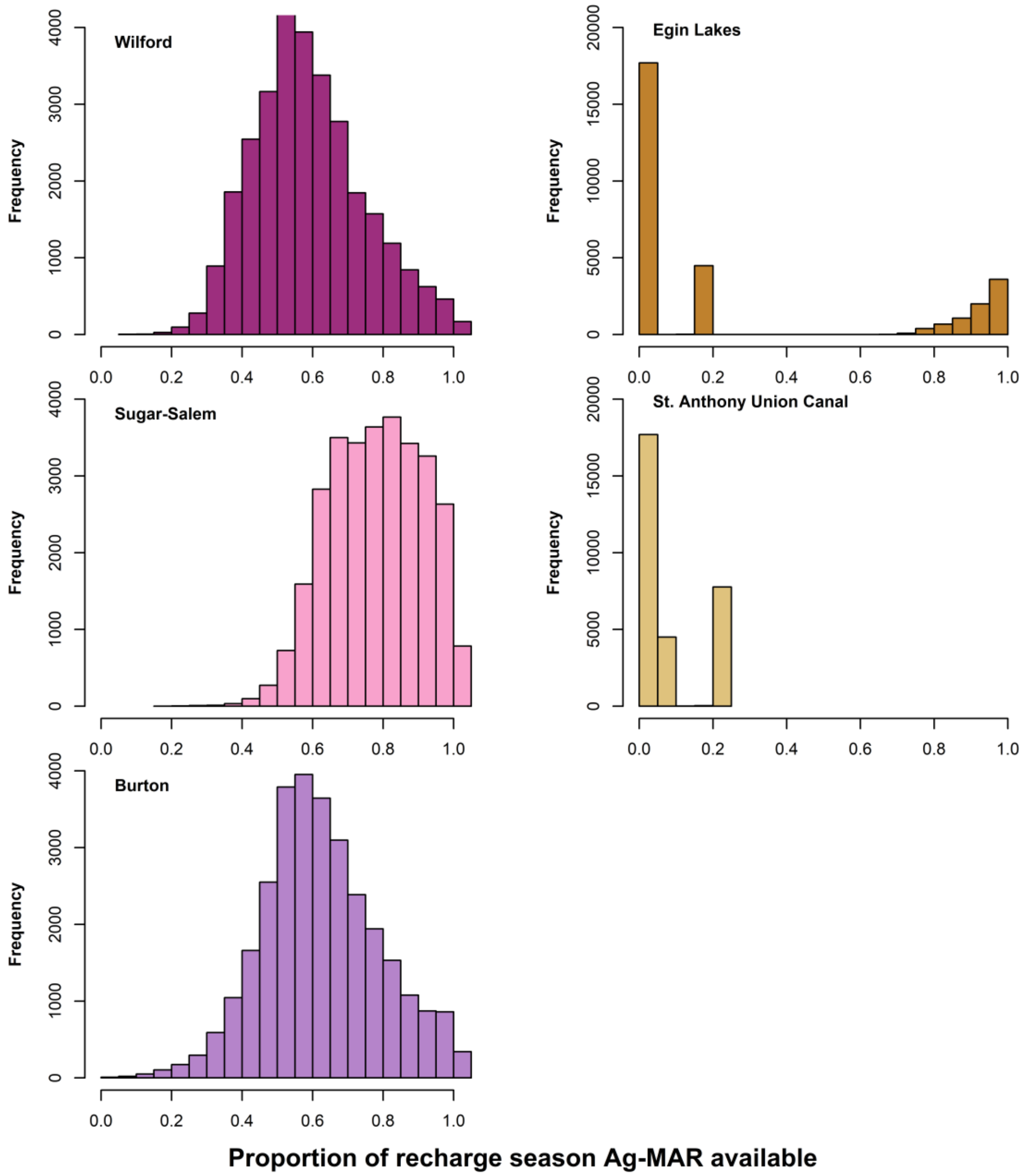

Figure 3-6. The proportion of recharge season when Ag-MAR is available for each site for all years in all 1,000 simulations. For reference, the recharge season is limited to 213 days for the *incidental* recharge sites and to 275 days for the *managed* aquifer recharge sites.

Canal capacity was not a limiting factor for conducting Ag-MAR at most sites. On average, canal capacity was never reached during Ag-MAR operations at three sites (*incidental:* Wilford and Sugar-Salem; *managed:* St. Anthony Union Canal) in all years across all simulations. However, canal capacity was reached an average of 13–14 days a year for recharge operations at one *incidental* recharge site (Burton) and 66–79 days a year at one *managed* recharge site (Egin Lakes).

3.3 Streamflow response to simulated Ag-MAR

Ag-MAR impacts streamflow because water is diverted for recharge and subsequently returned to the river. On average, our model output demonstrated Ag-MAR operations reduced the springtime streamflow peak at the watershed outlet by 10–14% and caused the runoff peak to occur up to 2 days earlier (Figure 3-7). Ag-MAR diversions reduced mean watershed outlet streamflow by 51–52% in April and October, respectively, when streamflow rises to its spring peak and when irrigation demand eases (Figure 3-7). However, on average, 62% of total recharged water returned to the target river reach annually. The remaining percentage enhances streamflow elsewhere in the Upper Snake River basin, further downstream.

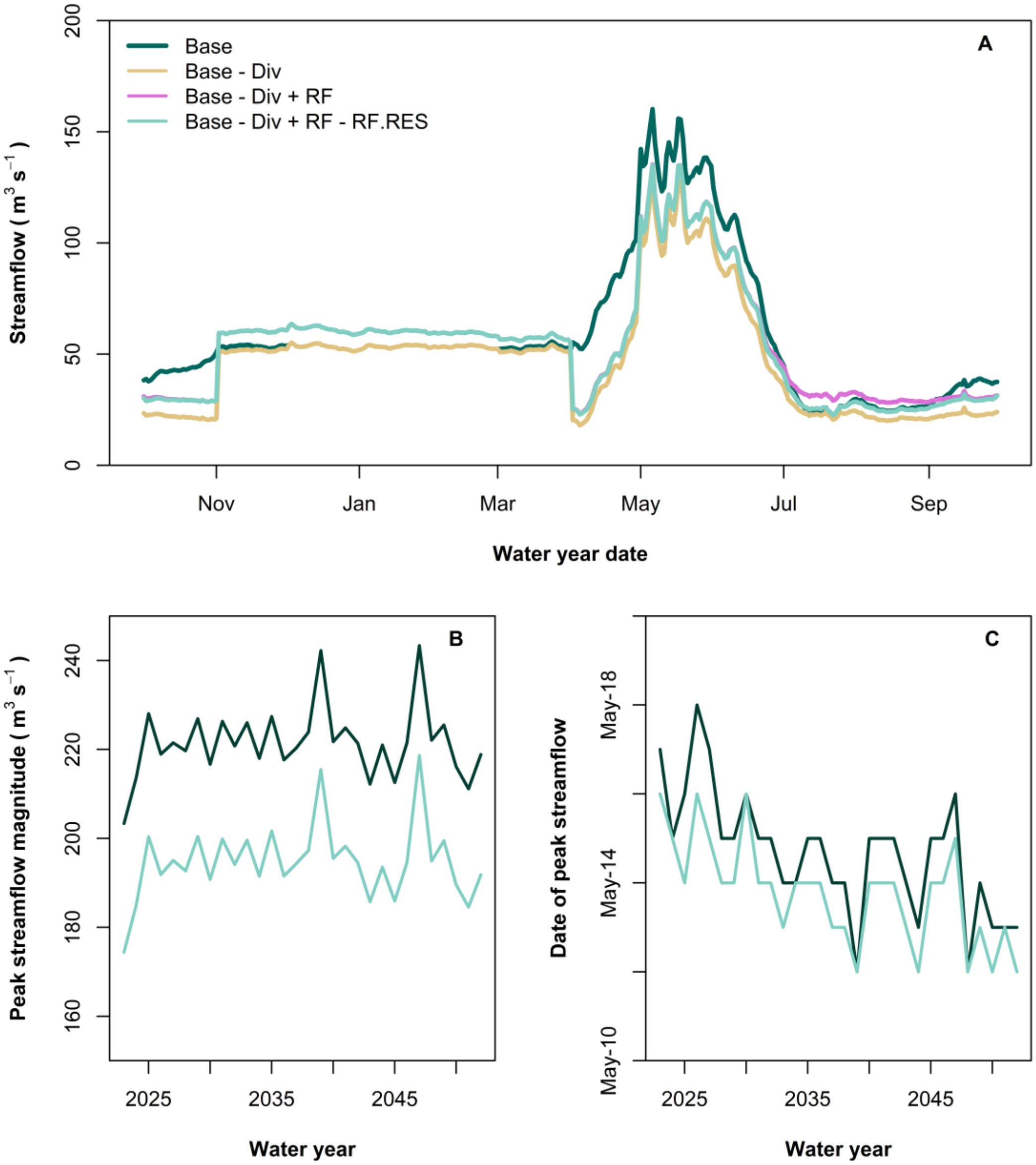

Figure 3-7. For streamflow at the Henrys Fork at Rexburg (the watershed outlet), across all simulations for conditions with and without Ag-MAR: Mean hydrograph for water year 2052 (steady state; A), mean annual peak streamflow magnitude (B), and date of peak (C). The base case refers to the no-Ag-MAR scenario, Div refers to the water diverted for Ag-MAR operations, RF refers to the return flows from Ag-MAR operations, and RF.RES refers to return flows added to upstream reservoir storage to reduce outflow.

Daily Ag-MAR water diversions had two general peaks for the *incidental* recharge sites—one in April and another in October (Figure 3-8). The timing of Ag-MAR water availability was different for the *managed* recharge sites, where the possible recharge season was March–November. At St. Anthony Union Canal, water availability was limited to March and November separately. At Egin Lakes, water availability peaked in March and November, but was available through the intervening months. However, water available for Ag-MAR was more evenly distributed at Egin Lakes May–September compared to the *incidental* recharge sites where Ag-MAR water availability declined to zero or near-zero between seasonal peaks. In contrast with all other sites, Ag-MAR water availability at Egin Lakes was slightly higher in the autumn than the spring. The Egin Lakes autumn peak ranged 0.7–1.3 m^3/s and the spring peak ranged 0.7–0.8 m^3/s. The peaks at St. Anthony Union Canal were near-equivalent.

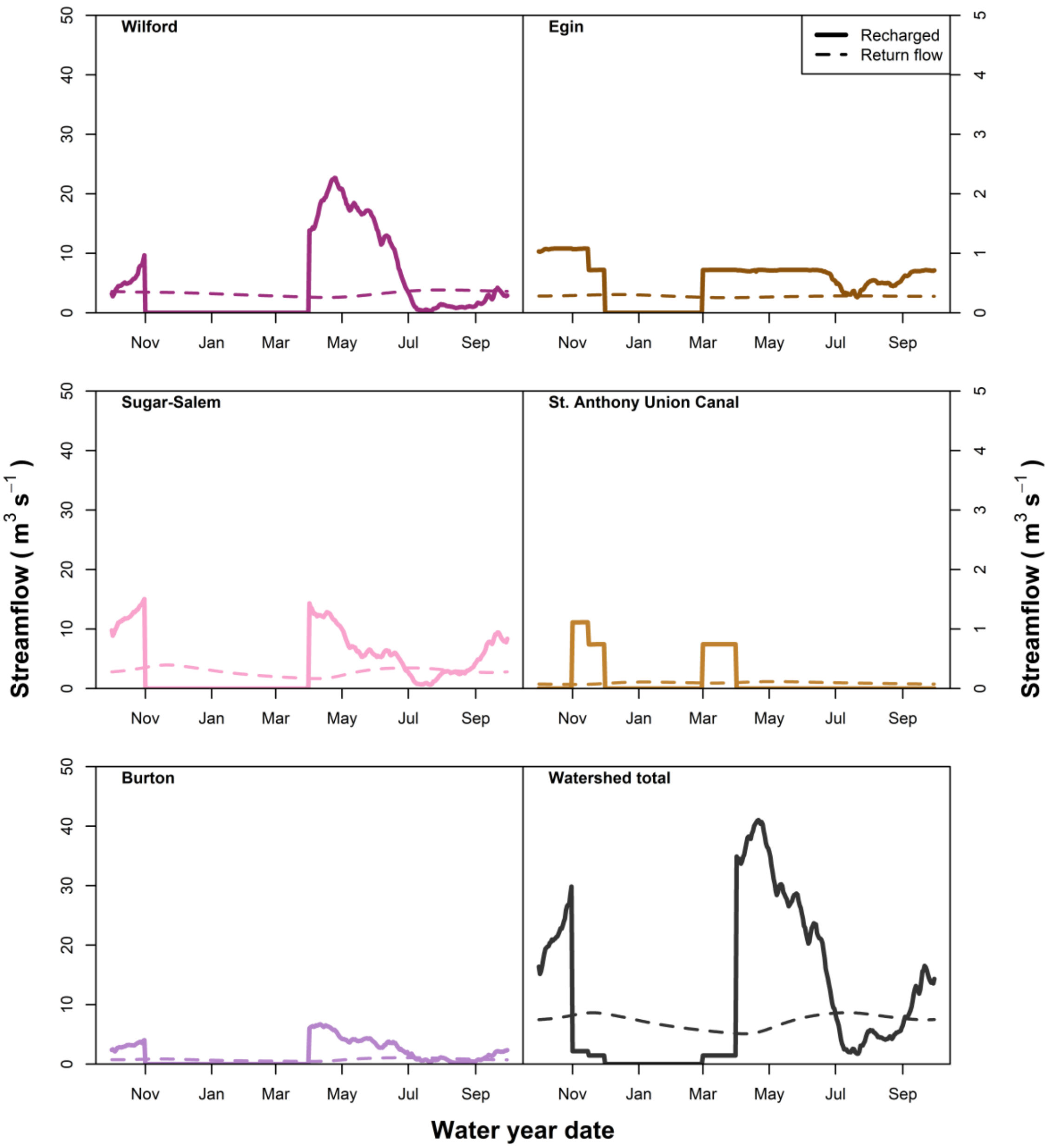

Figure 3-8. For the average water year 2052 (steady state) across all simulations, daily streamflow diverted for recharge and returned to the river for all Ag-MAR sites.

The model was initialized in conditions where no Ag-MAR was conducted. Thus, streamflow response to Ag-MAR had a 5-year ramp up before stabilizing. At steady state (water year 2052), mean daily return flow in the target reach ranged 5.1–8.7 m^3/s when

Ag-MAR was conducted consistently at all five sites (Figure B-3) and increased streamflow at the watershed outlet compared to the base scenario by 9–14% November through March (after accounting for diversion; Figure 3-7).

Additionally, our Ag-MAR simulations resulted in two return flow peaks: July 12–14 and November 19–20 (Figure 3-8). The return flow peak in mid-July coincided with the low-flow period for the reach (Figure 3-7). If all available return flows were applied in addition to base-scenario operations, return flows would increase streamflow at the watershed outlet by 6–14% July 2–September 3 (Figure 3-7). However, given Decision 2 in our post-hoc calculations (Section 2.2.4), return flows that increased streamflow above a local minimum target were subtracted from the watershed outlet and instead added to an upstream reservoir to increase storage and reduce outflow. In this case, return flow in July did not increase streamflow at the outlet but maintained streamflow in the base scenario (Figure 3-7).

On average, Ag-MAR conducted at two of the *incidental* recharge sites—Wilford and Sugar-Salem—contributed the most to return flow, accounting for 28% and 32% of total return flow, respectively (Figure 3-8). Ag-MAR conducted at Egin Lakes contributed 16% of total return flow, on average, while operations at Burton and St. Anthony Union Canal contributed 12% and 11%, respectively.

4. Discussion

In all simulations, water was available in less than a third of years for the designated MAR sites, where Ag-MAR access is limited by junior, MAR-specific water rights. Water was available for Ag-MAR in nearly all years, regardless of annual natural flow, at the *incidental* recharge sites. For water years 2025–2052, mean annual recharge volumes for the three *incidental* recharge sites totaled approximately 327 Mm^3 (12% annual natural flow) and largely occurred in April and October. Daily return flows to the target reach peaked in July and November, with the potential to increase July–August streamflow by 6–14% and November–March streamflow by 9–14%. Ag-MAR conducted at the *incidental* recharge sites contributed the most to return flow. However, Ag-MAR reduced the spring streamflow peak at the watershed outlet by 10–14% after accounting for both diversions and subsequent return flow.

4.1 Ag-MAR rules and tradeoffs

Diverting streamflow for Ag-MAR in the Henrys Fork was largely available when irrigation demand was low, namely during the spring streamflow pulse and autumn harvest periods. In Texas's Gulf Coast, a region with hurricane-caused flood events, Yang & Scanlon (2019) found water could be diverted for Ag-MAR from high magnitude flood events in each of the last 50 years. In the Sierra Nevada Range, where water for Ag-MAR is also limited to high magnitude flow events, water was available for Ag-MAR less often—30% of years in Carson Valley (Niswonger et al., 2017), 70% of years in the Sacramento Valley, and 47% of years in the San Joaquin Valley (Kocis & Dahlke, 2017).

Critics of using streamflow pulses for Ag-MAR note the geomorphic and phenological importance of flood and pulse flows. High magnitude flows are important for sediment transport and channel scouring, as well as seed dispersal and cueing fish migration (Arthington & Pusey, 2003; Stromberg et al., 2007). Indeed, we demonstrate diverting water for Ag-MAR measurably decreases streamflow pulses—with the watershed outlet streamflow peak diminishing by 10–14% after accounting for return flow.

However, Ag-MAR application volume and timing could be managed to maintain ecologically-important parts of the hydrograph, such as the springtime pulse. In California, where high magnitude flow events may occur multiple times in a season, Kocis & Dahlke (2017) suggest reserving the first high magnitude flow events for instream geomorphic and environmental needs and diverting water for Ag-MAR in the latter events. Other studies ensure Ag-MAR maintains 90–95% of daily streamflow in the river during high magnitude events (Levintal et al., 2023).

There is an inherent tradeoff: diverting streamflow for Ag-MAR in the springtime reduces peak spring streamflow, but can supplement streamflows mid-summer. Case studies in California's Central Valley and Washington's Walla Walla Basin, documented a 53% and 52–73% increase in summer flows due to Ag-MAR, respectively (Alam et al., 2020; Scherberg et al., 2014). In Cailfornia's Cosumnes River, modeling simulated 18% of recharged water would return to the river—providing continuous flow not otherwise experienced due to groundwater depressions from pumping (Gailey et al., 2019). In the Henrys Fork, groundwater return flows from springtime Ag-MAR provide cool water during a critical low-flow period when irrigation demand is high and stream temperatures exceed temperatures suitable for trout (Van Kirk et al., 2020). However, return flows have declined in volume as irrigation efficiency has increased in the watershed (Morrisett et al., In Review). Thus, Ag-MAR presents an opportunity to increase summer streamflow by approximately 6–14%.

4.2 Ag-MAR site suitability and water rights administration

Water rights at the *incidental* recharge sites were senior to those at the *managed* aquifer recharge sites. Thus, the *incidental* recharge sites had reliable access to water for

Ag-MAR and were able to conduct Ag-MAR more frequently than the *managed* sites specifically designated and appropriated for Ag-MAR. We identified five sites as suitable for Ag-MAR with steady-state streamflow response rates of 49–72% after 30 years. We chose the three *incidental* recharge locations either because flood irrigation was still in place or because they contained flood irrigation infrastructure with the potential for restoration, given proper incentives and relationship building. These sites could provide recharge *incidental* to flood irrigation practices under current water rights and their assigned beneficial use. In contrast, the two *managed* recharge sites were locations associated with specific water rights to conduct managed aquifer recharge. Our work highlights how water rights administration can act as a barrier or pathway to Ag-MAR success. In the Henrys Fork, sites specifically managed and maintained for Ag-MAR are least likely to receive water given junior water rights. Therefore, maintaining and enhancing flood irrigation in sites with more senior water rights may instead be a more impactful route.

4.3 Ag-MAR, reservoir operations, and a warming climate

Using senior rights to divert natural streamflow for flood irrigation (incidental recharge), we can conduct Ag-MAR in the Henrys Fork even in low water years—as demonstrated by Ag-MAR water availability in the drought recovery period following initial conditions (Figure B-6). Our finding contrasts with research in California's Central Valley, where reservoir re-operation modeling suggests Ag-MAR conducted in dry and below-average years requires draining surface reservoirs and is thus ill-advised (Goharian et al., 2020). In fact, our model did not allow recharge operations when Island Park

Reservoir was drafting (i.e., outflow is greater than inflow)—thus preventing Ag-MAR from drawing down reservoir storage.

However, our ability to conduct Ag-MAR at sites with junior recharge rights is likely to diminish through time. Inflow to Island Park Reservoir—streamflow in the reach between Henrys Lake and Island Park (Figure 3-1)—was projected to decrease through the 30-year simulation (Figure 3-4). With diminished inflow, Island Park Reservoir will draft for a longer period, further limiting the ability to conduct Ag-MAR. Furthermore, as hydrologic extremes intensify, the ability to store water in surface reservoirs is increasingly uncertain (Ficklin et al., 2022) and junior water rights are increasingly likely to face curtailment (Null & Prudencio, 2016). Thus, the ability to conduct Ag-MAR with senior water rights to divert natural streamflow is an important component of adapting water management portfolios in the American West.

4.4 Competition for flood irrigated land and other Ag-MAR limitations

Our study highlights how flood irrigated land with senior water rights has promise for Ag-MAR compared to sites specifically managed for recharge in the region. However, these incidental recharge sites are at greater risk of land conversion. Residential development is more likely to occur on flat terrain proximal to urban areas (Li et al., 2019). The potential incidental recharge sites meet both criteria as they have flat terrain necessary to conduct flood irrigation and are within 10 km of the largest city in the Henrys Fork watershed with housing development and city expansion pressure (Baker et al., 2014). As broadly demonstrated in other studies, urban encroachment on the canals that service these irrigated areas could also cause conflict or disrupt canal function (Cox & Ross, 2011; Hicks & Peña, 2003). Seepage from unlined canals and on flood irrigated land can damage residential property in mixed residential and agricultural neighborhoods (Deng & Bailey, 2020).

The interaction between Ag-MAR and crop cultivation also requires further examination. Ghasemizade et al. (2019) found their Ag-MAR simulation in California's Central Valley risked waterlogging agricultural land both in and near the Ag-MAR zone, potentially damaging crops. Compensating irrigators for allowing Ag-MAR on their land and risking such crop damage may incentivize Ag-MAR participation (Dahlke et al., 2018; Gailey et al., 2019). In our study, we modeled Ag-MAR implementation for all water available. However, this strategy is likely impractical given the specific soil conditions and labor requirements for flood irrigation to be effective. Flood irrigation cannot be conducted when soil is too wet (Bjorneberg & Sojka, 2005), and irrigators in the Henrys Fork have noted soil with high infiltration rates require frequent flood application (Morrisett et al., In Review). Flood irrigation is also a laborious task (Bjorneberg & Sojka, 2005). Thus, integrating Ag-MAR into current flood irrigation operations may require partners provide additional labor or compensate landowners accordingly. Lastly, we identified water available for Ag-MAR to flood irrigated sites into October. However, flood irrigating in October is likely impractical due to current crop harvesting schedules.

5. Conclusions and Next Steps

Ag-MAR is an inherently local process (Levintal et al., 2023). Conducting Ag-MAR on flood irrigated land in the Henrys Fork will require relationship building with landowners and irrigators, as well as an understanding of their values and limitations (Ricart & Clarimont, 2016; Tran & Kovacs, 2021). Water users generally support recharge projects on the Eastern Snake Plain Aquifer, but lack the financial willingness or ability to pursue recharge projects (Miller, Goulden, et al., 2021; Morrisett et al., In Review). Thus, economic incentives may successfully encourage Ag-MAR participation (Ricart & Clarimont, 2016). When developing Ag-MAR projects and engaging partners, it is also important to explicitly demonstrate the multi-dimensional feasibility of Ag-MAR (Harvey et al., 2023). Quantifying tradeoffs and assessing varying Ag-MAR operation rules with an optimization or other tradeoff analysis approach is a needed next step. Studies like Dogrul et al. (2016), Ebrahim et al. (2016), Tran et al. (2019), and Zhao et al. (2021) serve as examples. Nonetheless, our research provides the proof of concept required before pursuing partnerships and a foundation from which to adjust and explore in more depth. But our conclusion is clear: Leveraging prior appropriation to conduct aquifer recharge may be a valid option in a warming climate. Using senior water rights to flood irrigate should be considered within climate adaptation portfolios.

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CHAPTER 4

ASSESSING DOWNSTREAM AQUATIC HABITAT AVAILABILITY RELATIVE TO HEADWATER RESERVOIR MANAGEMENT IN THE HENRYS FORK SNAKE RIVER

Abstract

Reservoirs are sometimes managed to meet agricultural and other water demands, while also maintaining streamflow for aquatic species and ecosystems. In the Henrys Fork Snake River, Idaho (USA), irrigation-season management of a headwater reservoir is informed by a flow target in a management reach ~95 km downstream. The target is in place to meet irrigation demand and maintain aquatic habitat within the 11.4 km management reach and has undergone four flow target assignments from 1978 to 2021. Recent changes to irrigation-season management to maximize reservoir carryover warranted investigation into the flow target assignment. Thus, we created a streamflow-habitat model using hydraulic measurements, habitat unit mapping, and published habitat suitability criteria for Brown Trout (*Salmo trutta*), Rainbow Trout (*Oncorhynchus mykiss*), and Mountain Whitefish (*Prosopium williamsoni*). We used model output to compare habitat availability across two management regimes (1978–2017 and 2018–2021). We found that efforts to minimize reservoir releases in 2018–2021 did not reduce mean irrigation-season fish habitat relative to natural flow, but did reduce overall fish habitat variability during the irrigation season compared to streamflow management in 1978–2017. Field observations for this research led to an adjusted flow target in 2020 that moved the target location downstream of intervening irrigation diversions. Using our model output, we demonstrated

that moving the location of the target to account for local irrigation diversions will contribute to more consistently suitable fish habitat in the reach. Our study demonstrates the importance of site selection for establishing environmental flow targets.

1. Introduction

In highly regulated river systems, reservoir managers often attempt to balance reservoir storage and instream flow releases to meet human water demand and preserve aquatic species, habitats, and ecosystems (Owusu et al., 2021). Reservoir operations to meet downstream flow objectives are paramount to meeting multiple stakeholder needs (Kahil et al., 2016; Tickner et al., 2017), and are a challenging endeavor as drought conditions amplify water scarcity and management scrutiny (Castro et al., 2018; Null et al., 2022; Wineland et al., 2022).

In water-limited systems, there are many frameworks for managing flow to meet different environmental objectives. Major ideas within environmental flow management include minimum instream flows (Tharme, 2003), environmental flows (Acreman et al., 2014), functional flows (Yarnell et al., 2020), designer flows (Chen & Olden, 2017), or some combination thereof—each with a distinct definition and intent. Environmental water management is often motivated by a desire to maintain native biodiversity and ecosystem function by exceeding critical minimum flows (Acreman & Dunbar, 2004) or, more commonly, preserving components of natural flow regimes (Arthington et al., 2018). The functional flows approach identifies natural hydrograph components (e.g. magnitude and duration) and links them with relevant ecological responses (e.g. migration cues) to inform preservation of hydrograph functionality (Yarnell et al., 2020). Designer flows also seek to generate positive ecological response, but allow model simulations to find flow strategies outside the natural flow regime given the constraints of highly regulated and altered river systems (Chen & Olden, 2017).

Environmental flow assessments inform river management and establish or recommend flow strategies. Through time, these assessments have used hydrologic, hydraulic, habitat simulation, or holistic methods (Arthington et al., 2004). A common hydrologic approach assigns minimum flows proportional to average flow (Tharme, 2003). This is the simplest and least costly approach as it uses existing flow data and does not require fieldwork (Arthington et al., 2004). The hydraulic approach accounts for parameters like wetted perimeter and channel depth relative to streamflow and seeks to keep parameters above a given threshold assumed to be ecologically relevant, while habitat simulation relates streamflow to habitat suitability (Tharme, 2003). Ecohydraulic approaches build on habitat simulation to include data relevant to ecological dynamics (Rice et al., 2010). More holistic approaches blend hydrologic, hydraulic, and habitat simulation with socio-economic dependencies to craft environmental flows for ecosystems as a whole (King et al., 2003; Poff et al., 2010).

Processes to implement environmental flows are just as variable as their assessment methods. For example, South Africa’s 1998 National Water Act created an ecological “Reserve” that prioritizes water for aquatic ecosystems second to water needed for basic human needs; local water managers are responsible for ensuring water allocations account for the Reserve (Takacs, 2016). In Australia’s Murray-Darling Basin, the government re-allocated a proportion of water for environmental purposes (Grafton et al., 2014). In the United States, environmental flow regulations may be determined through consultation with relevant federal fish and wildlife agencies for species listed as endangered (e.g.

National Marine Fisheries Service, 2020). In many western states, environmental flows are subject to rigid prior appropriation rules and must have broad “beneficial use” (Wineland et al., 2022). Other approaches include agreements between local and regional stakeholders that require buy in; some are legally enforceable (Van Kirk et al., 2019).

In the Henrys Fork watershed, Snake River (USA), the Henry’s Fork Drought Management Planning Committee (henceforth, the Committee) was established by federal mandate in 2003 and collaborates across stakeholder groups to recommend seasonal reservoir storage and release strategies for multiple water users (Joint Committee, 2018). The Committee formally includes a federal agency, two irrigation entities, and three environmental conservation groups. The Committee aims to “maintain or enhance watershed health and ecology, even in years of below-average precipitation, in balance with agricultural needs through flexible and adaptive water management within the context of Idaho water law” (Joint Committee, 2018). The Committee does not have management authority as a whole, but individual members of the Committee do—namely the irrigation entities who operate headwater irrigation storage reservoirs. However, the Committee generally makes decisions based on consensus and usually does so based on expert opinion or stakeholder input pending further study.

In 2017, following a severe drought period, the Committee modified its reservoir management strategy to 1) minimize irrigation-season drawdown and 2) avoid extremely low flows and high variability observed within a management reach ~95 km downstream of a major reservoir during the previous summer. In modifying its strategy, the Committee requested a framework for assessing habitat quantity relative to streamflow management decisions. In this paper, we present work done to meet the Committee’s request. Our objectives were to 1) quantify aquatic habitat availability at different streamflows and 2) assess the differences in aquatic habitat availability across different management regimes. To address these objectives, we used a physical habitat approach and created streamflow-habitat models for three fish species in the downstream management reach where irrigation-season flows are managed by releases from an upstream reservoir. The intent of this study was to create a framework for assessing habitat quantity relative to management decisions in an applied setting, rather than to explicitly develop an environmental flow recommendation.

2. Study area

The Henrys Fork watershed is 8,300 km^2 in the upper Snake River Basin (Figure 4-1). The natural hydrologic regime is snow-dominated and spring-fed (Bayrd, 2006; Benjamin, 2000), with 3,140 Mm^3 mean annual natural flow (Bureau of Reclamation, 2012). Three headwater reservoirs (Henrys Lake, 111 Mm^3; Island Park, 167 Mm^3; Grassy Lake, 19 Mm^3) store water for 1,012 km^2 of irrigated agriculture in the lower watershed, with an average annual irrigation diversion of 1,400 Mm^3 (Bureau of Reclamation, 2012). In addition to supplying water to a $10 billion regional agricultural industry (Idaho Water Resource Board, 2009), the Henrys Fork also hosts recreational fisheries that contribute $50 million to local communities (Van Kirk et al., 2021).

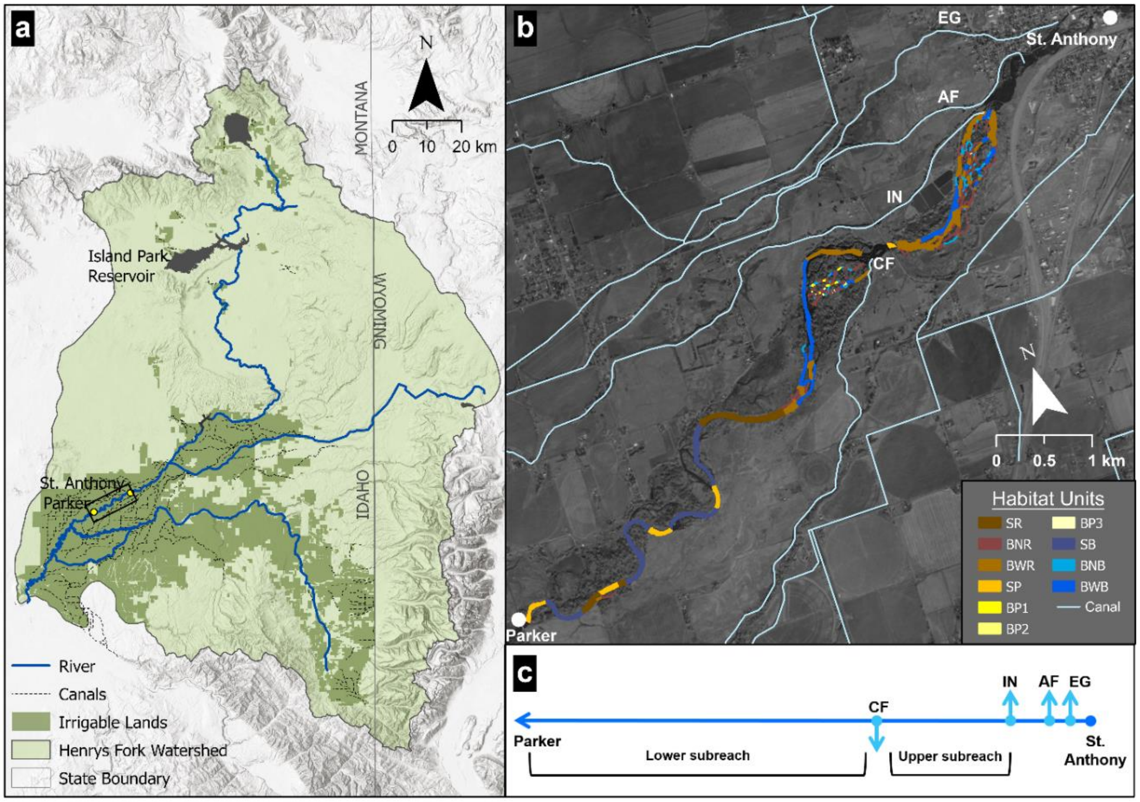

Figure 4-1. Henrys Fork Watershed, Idaho, USA (A), habitat units within study reach (B), and a schematic of study subreaches relative to canal diversions (C; not to scale). See Table 4-2 for habitat unit definitions. EG is Egin Canal, AF is St. Anthony Union Feeder Canal, IN is Independent Canal, and CF is Consolidated Farmers Canal. Credits: Airbus, USGS NGA, NASA, CGIAR, NCEAS, NLS, OS, NMA, Geodatastyrelsen, GSA, GSI and the GIS User Community (A) and Maxar for imagery (B).

2.1 History of irrigation-season flow management

During irrigation season (April–October), the Committee recommends strategies for operating the headwater reservoirs—particularly Island Park Reservoir—to meet irrigation demand and streamflow targets in the lower watershed. Releases are made from Island Park Reservoir to meet irrigation needs within a system of water-rights priority and administrative rules, with some streamflow left in the river downstream of all canal diversions. The amount left in the river constrains Island Park Reservoir releases, is colloquially termed the “irrigation-season flow target,” and effectively acts as a non-binding, de facto environmental flow that maintains aquatic habitat. The target is an irrigation-season management crux for the watershed. The target is triggered by and has direct impact on other watershed management goals such as reservoir carryover and streamflow in a tailwater fishery immediately below Island Park Dam. The term “target” is intentional: a streamflow to aim for within the Henrys Fork watershed, with the acknowledgement that the target may be missed or over-ridden by basin-wide operational needs for the upper Snake River, Idaho.

The Henrys Fork irrigation-season target is in an informal management reach from St. Anthony to Parker, Idaho (11.4 km; 0.27% gradient) where four irrigation canals cumulatively divert up to 22.7 m^3s^{-1} from the river (Figure 4-1C; Figure 4-2A). The management reach underwent multiple management approaches between 1978 and 2021 (Table 4-1). St. Anthony was the original location for the management target because of a U.S. Geological Survey streamflow gage located there. The creation of the Committee in 2003 formalized stakeholder input, and the Committee set the default target to 28.3 m^3s^{-1} at St. Anthony in the late-2010s to meet downstream irrigation demand and maintain aquatic habitat. Application of the 28.3 m^3s^{-1} at St. Anthony target was ad hoc prior to 2018 and carefully implemented after 2018 (Table 4-1).

Table 4-1. Irrigation-season (April–October) management history in the lower Henrys Fork, where water year is October–September.

Management regime	Water Year	Flow Target	Location	Basis
Ad hoc	1978–2002	None	St. Anthony	General “rules of thumb” for Upper Snake River system set by water managers
	2003–2017	22.6–34.0 m^3s^{-1}	St. Anthony	Formal stakeholder input applied ad hoc via the Committee
Precision	2018–2019	28.3 m^3s^{-1}	St. Anthony	Formal stakeholder input carefully implemented via the Committee
	2020–present	9.9 m^3s^{-1}	Parker	Recommendation from this research accepted by the Committee

Prior to the 2020 irrigation season, the Committee changed the target to 9.9 m^3s^{-1} target at Parker based on observations we brought to the Committee. In 2019, we observed that diversions to four canals downstream of St. Anthony resulted in significant flow variability and considerably lower streamflow than the 28.3 m^3s^{-1} St. Anthony target for most of the reach. For example, we measured streamflow at 6.2 m^3s^{-1} at Parker in mid-July—a day where we also observed increased streambed exposure and fish clustering in single-channel pools. We used these observations to inform a simple desktop analysis, where we demonstrate that, on average, a 9.9 m^3 s^{-1} target at Parker was equivalent to the 28.3 m^3s^{-1} target at St. Anthony for a given reservoir drawdown period due to intervening canal diversions (Figure 4-2). Thus, changing the flow target to 9.9 m^3s^{-1} target at Parker would not result in higher reservoir drawdown volume. Additionally, changing the flow target would prevent extremely low flows in the lower subreach. Diversion volume varies through the irrigation season and is highest in July (Figure 4-2A). Thus, a flow target at St. Anthony leads to less streamflow through the management reach in July and more in

September as diversions increase and decrease. Lastly, we documented the presence of cool groundwater springs in the lower subreach in summer 2019 (Van Kirk et al., 2020) and observed that flows ≤8.5 m^3s^{-1} in the lower subreach disconnected these springs from the river, eliminating potential thermal refugia. A 9.9 m^3s^{-1} target at Parker would create a m^3s^{-1} buffer for maintaining spring connectivity in the lower subreach (Figure 4-1C). Based on our observations, the Committee accepted our recommendation on a trial basis in 2020 pending the quantitative analysis in this study.

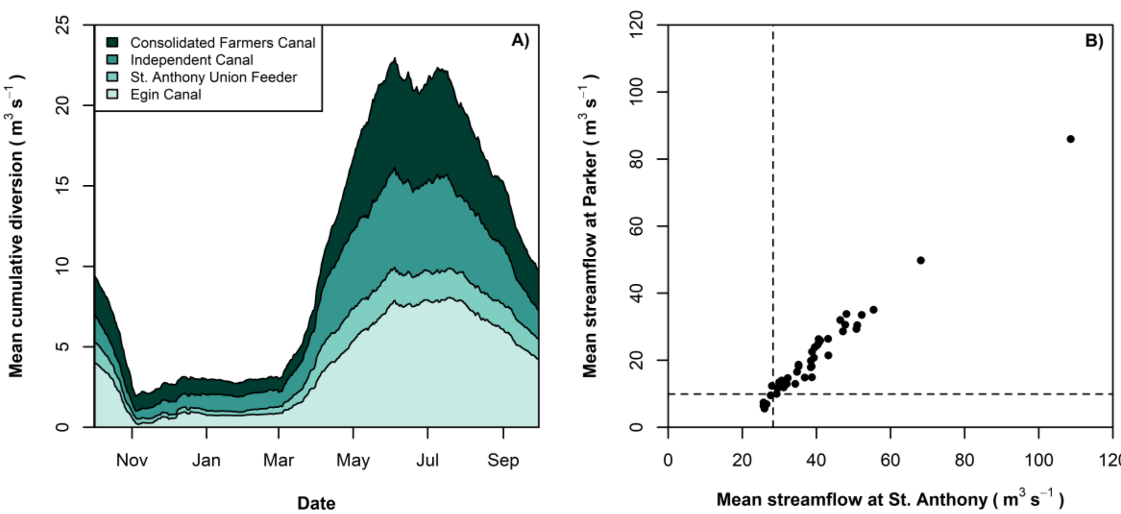

Figure 4-2. For water years 1978–2021, mean daily cumulative diversion for the four irrigation canals in the Henrys Fork irrigation-season management reach (A) and mean daily streamflow at St. Anthony and Parker during the reservoir drawdown period (B).

2.2 Management reach characteristics

The river reach between St. Anthony and Independent Canal is primarily fractured basalt, with a basalt ledge upstream of Independent Canal. Between Independent Canal and

Consolidated Farmers Canal (“upper subreach”; Figure 4-1C), the river is anastomosing and braided, with multiple channels, islands with a cottonwood riparian corridor, and canals that parallel much of the north bank of the reach. Downstream of Consolidated Farmers (“lower subreach”; Figure 4-1C), the river has some anastomosis with cottonwood islands, but is primarily single-channel meandering, with areas of slow, deep water and adjacent agricultural fields mostly used for cattle grazing.

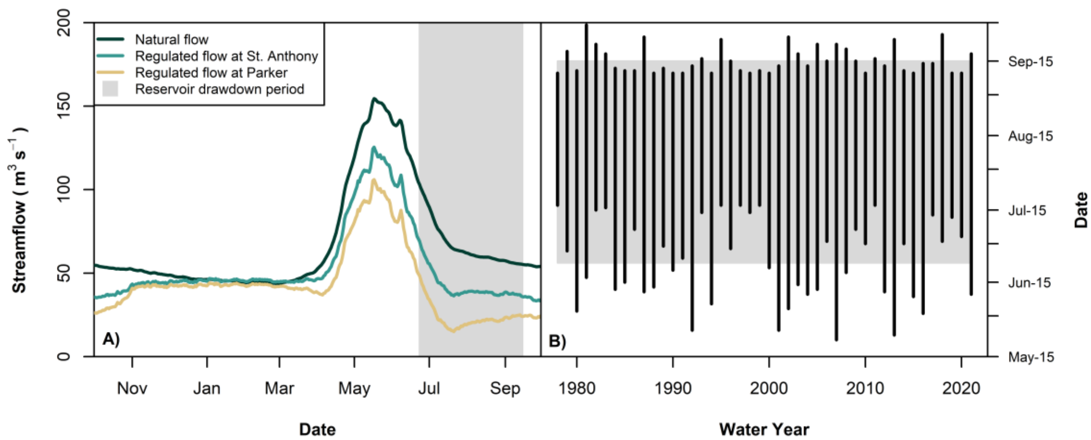

Figure 4-3. Hydrographs of mean daily natural flow and mean regulated flow at St. Anthony and Parker for water years (October–September) 1978–2021, using gaged streamflow and canal diversion data (A; Appendix C for total hydrograph range). The drawdown period at Island Park Reservoir for each water year 1978–2021 (B). The gray shading in each panel denotes the mean drawdown period across all water years (June 23 to September 15).

The management reach retains a snowmelt-driven runoff peak (Figure 4-3A), has a low-flow period that coincides with headwater reservoir drawdown to meet irrigation demand (Figure 4-3B), has considerable groundwater exchange year-round (Van Kirk et al., 2020), hosts a popular, recreational, wild, nonnative Brown Trout (Salmo trutta) fishery

(Vincent et al., 2023), and has high macroinvertebrate density (Marshall, 2018). Nonnative Rainbow Trout (Oncorhynchus mykiss) and native Mountain Whitefish (Prosopium williamsoni) are also present but less abundant (Vincent et al., 2023). For all study species, juveniles and adults are present in the reach during irrigation season. For Rainbow Trout, both life stages use pools and riffles (Raleigh et al., 1984; Sigler & Zaroban, 2018). For Mountain Whitefish and Brown Trout, adults prefer deep pool bottoms while juveniles prefer shallow riffles and backwaters. Mountain Whitefish and Brown Trout spawning seasons begin in September and October, respectively (Sigler & Zaroban, 2018), thus overlapping with the end of irrigation season.

3. Methods

We used a weighted usable area approach with hydraulic measurements, reach-scale habitat unit mapping, literature-based habitat suitability indices, and statistical modeling to quantify reach-scale habitat and understand how streamflow relates to aquatic habitat for the three study species.

3.1 Weighted Usable Area

We computed the streamflow-habitat relationship using a weighted usable area (WUA) approach (Bovee *et al.*, 1998), where WUA is the product of area and a suitability index between 0 and 1, using indices from habitat suitability curves for a given species (s) and life stage (l), summed across all habitat types (i) for a given reach flow (Q).

$$WUA_{Q,s,l} = \sum_{i=1}^{n} (Area_{i,Q})(suitability_{i,Q,s,l}) \tag{1}$$

Given diversion locations within the management reach, streamflow—and thus habitat—differ in the upper and lower subreaches of this study (Figure 4-1). Therefore, we calculated WUA by subreach according to their respective streamflows and summed for total WUA within the study reach:

$$WUA_{Total} = WUA_{LowerSubreach,Q_{Parker}} + WUA_{UpperSubreach,Q_{Parker}+Q_{CFdiversion}} \quad (2)$$

where the upper subreach is from the IN canal to the CF canal, and the lower subreach is from the CF Canal to Parker (Figure 4-1C).

3.2 Habitat types: definitions and mapping

We sampled 14 sites chosen to represent 10 habitat types within the 11.4 km management reach. Habitat types were assigned based on channel type, cross-section bathymetry, and width (Table 2). Rectangular cross-sections had near-uniform depth; bends had a deep thalweg along one bank and a point bar deposit along the opposing bank, matching descriptions of the management reach documented in Bayrd (2006). Pools varied from first-class (i.e. large and deep) to third-class (small and/or shallow), matching definitions in Raleigh et al. 1984. See Appendix C for visual depictions.

Table 4-2. Habitat types, locations, number of sites sampled per type, and the number of total samples per type. Habitat types are abbreviations of their characteristics: B/S for Braided/Single, W/N for Wide/Narrow, R/B for Rectangular/Bend, and P is for Pool with numbers to match definitions in Raleigh et al. 1984.

Habitat Type	Channel Type	Channel Width (m)	Geomorphic Unit	Subreach sampled	Sites	Samples
BWR	Braided	≥15.24	Rectangular	Upper	2	12
BNR	Braided	<15.24	Rectangular	Upper	2	12
BWB	Braided	≥15.24	Bends with point bar	Upper	2	13
BNB	Braided	<15.24	Bends with point bar	Upper	2	12
BP1	Braided	≥13.72	First-class pool	Lower	1	12
BP2	Braided	9.14–13.72	Second-class pool	Lower	1	11
BP3	Braided	<9.14	Third-class pool	Lower	1	9
SR	Single	NA	Rectangular	Lower	1	42
SB	Single	NA	Bends with point bar	Lower	1	31
SP	Single	NA	Pool	Upper	1	57

We used area mapping to extrapolate our sample sites to the reach scale (Figure 4-1). We mapped habitat types within the braided areas of our study reach via walking surveys in late July 2020, when streamflow averaged 20.7 m^3s^{-1} in the upper subreach and 12.7 m^3s^{-1} in the lower subreach. We mapped habitat units in the single-channel section of the lower subreach via floating surveys at flows 6.2–41.3 m^3s^{-1}. Two sections within our study area characterized by fractured basalt channels were excluded from the analysis: 1) between St. Anthony and the IN canal and 2) just upstream of the CF canal (Figure 4-1B).

We digitized habitat types onto aerial river imagery (Figure 4-1B) and calculated the geometric area of each habitat unit. We assumed unit length was constant across all flows, but wetted width was dependent on streamflow. Based on time of year and flows at which imagery was collected, we assumed aerial imagery surface area reflected bankfull width (Bayrd, 2006). We calculated the width of each habitat unit as a fraction of bankfull from field data collected at flows ≤bankfull. These calculations were based on statistical relationships between width and flow, in which width was constrained to be between 0 and bankfull width across the range of flows relevant to this study (Appendix C). We demonstrate channel morphology has not changed significantly during the 44-year time series based on comparison of aerial imagery from 1986 to 2022 (Figure 4-4). However, we acknowledge that historic aerial imagery only provides information of channel location and width, and cannot provide information on depth and velocity.

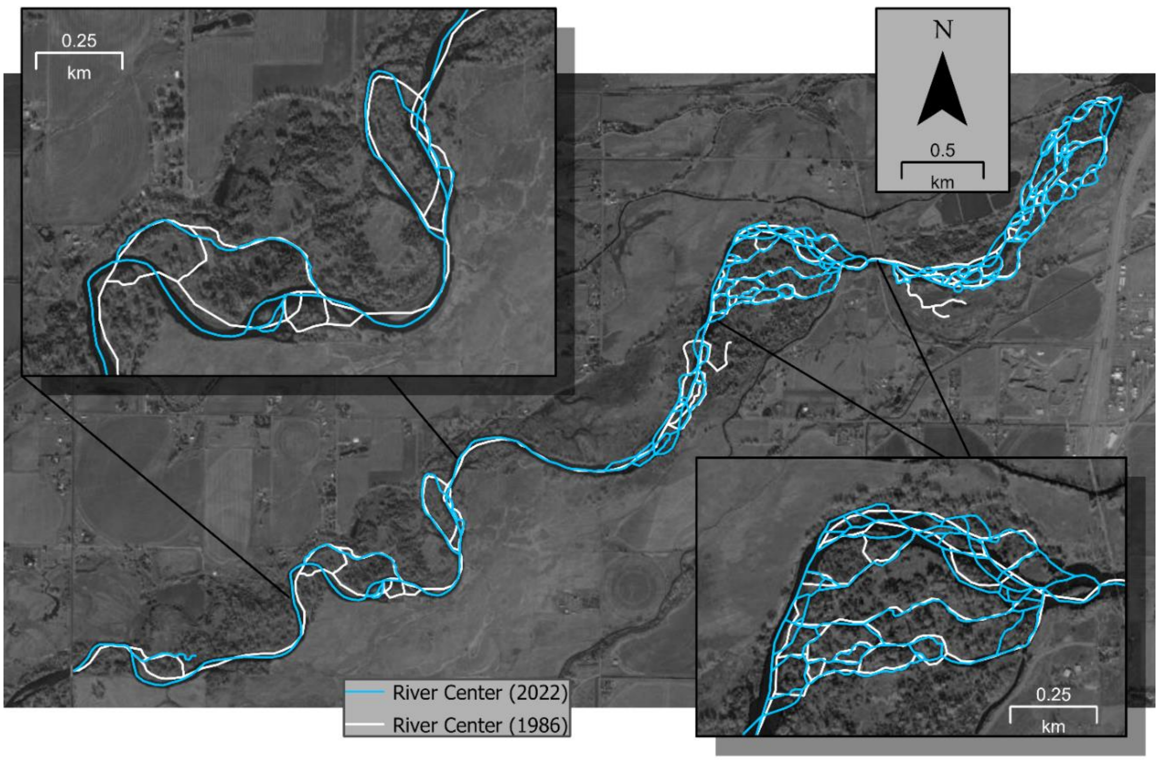

Figure 4-4. Center-line comparison for individual channels of the management reach from Independent Canal to Parker. Center lines were drawn using aerial imagery from 1986 (Idaho Department of Water Resources, 2022) and 2022 (background imagery; Maxar).

3.3 Habitat suitability

For each habitat type, we calculated the composite suitability of depth and velocity (Bovee et al., 1998). We chose to use composite suitabilities, rather than mean suitabilities, to reflect the assumption that suitable habitat conditions from one variable cannot compensate for unsuitable conditions from another variable (Muñoz-Mas et al., 2012). We did not consider cover, as woody debris is minimal, boulders are absent, and riparian cover is absent in the single-channel sections and minimal in braided sections. We assumed uniform substrate suitability, as previous work documented most of our study reach is gravel-bottomed (Bayrd, 2006). We did not consider stream temperature suitability as it is locally impacted by groundwater (Van Kirk et al., 2020) and otherwise largely driven by atmospheric conditions and travel time (Null et al., 2009). Particular to our system, stream temperature cannot be effectively managed with reservoir release volumes ~95 km upstream (McLaren et al., 2019).

No habitat suitability criteria are available for the lower Henrys Fork. Thus, we used suitability criteria found in the literature for each species and life stage of interest. For Brown Trout, we used the Category I habitat suitability curves from Raleigh et al. (1986) for juvenile, adult, and spawning adult trout. For Rainbow Trout, we used the Category I habitat suitability curves from Raleigh et al. (1984) for juvenile and adults. For Mountain Whitefish, we combined Category III habitat suitability curves from Hoffman et al. (2002) and Category II habitat suitability curves from Rempel et al. (2012) for juvenile and adult fish (Appendix C). We did not choose suitability curves related to abundance, carrying capacity, or other population metrics, as regional fisheries managers do not manage the reach by these criteria. We did not consider spawning habitat for Rainbow

Trout, as they are spring spawners and spawn prior to the irrigation-season low-flow period. Habitat suitability curves for spawning Mountain Whitefish were unavailable.

3.4 Habitat-streamflow data and relationships

At each site, we measured wetted width (m), depth (m), and velocity (m s^{-1}) along a channel transect using an Acoustic Doppler Current Profiler (Teledyne StreamPro on a Teledyne Riverboat). The StreamPro has a depth range of 0.1–7 m and measures velocity up to 5 m s^{-1}, with 1% accuracy for each metric (Teledyne RD Instruments, 2016). Each site measurement consisted of ≥3 passes across the channel and we strived for an in-field measurement error of ≤5%. We conducted measurements at full-channel nominal streamflows 10.3–68.0 m^3s^{-1} in the upper subreach in June–July 2019–2020 and 5.13–41.2 m^3s^{-1} in the lower subreach from June–September 2019–2021.

We used publicly available data to calculate expected subreach flow for the full time series (1978–2021) at the instantaneous and daily scales. We retrieved 15-minute and daily streamflow data for St. Anthony from the U.S. Geological Survey monitoring location 13050500 and for the four gaged canal diversions from the U.S. Bureau of Reclamation's Hydromet website for the Columbia-Pacific Northwest Region. To calculate instantaneous streamflow in the upper and lower subreaches at the time of our habitat measurements, we subtracted gaged canal diversions from gaged streamflow at St. Anthony, accounting for the 3.5-h travel time through the management reach (Appendix C). We used instantaneous streamflow values in statistical models to create relationships between 1) channel width and total-river streamflow, and 2) suitability and total-river streamflow for each habitat type, species, and life stage combinations (Appendix C).

We refer to streamflow as calculated above as "nominal flow." However, the study reach interacts dynamically with the regional aquifer (Van Kirk et al., 2020), and nominal daily flow calculations at Parker sometimes resulted in negative values—indicating the presence of groundwater exchange. To account for historical reach gains and losses, and also ensure streamflow remained nonnegative, we added a reach gain adjustment to all mean daily flows used in calculating WUA in the lower subreach (Appendix C). For water years 1978–2021, the reach gain adjustment is ≤30% of drawdown-period streamflow and, on average, made a 4.2–10.9% difference in the calculated mean drawdown-period WUA depending on species and life stage (Appendix C).

3.5 Management regime comparison

We analyzed WUA for the adult stage of each of the three species in the context of two management regimes (Table 4-2) for water years 1978–2021 (October 1–September 30). We input daily streamflow and diversions for water years 1978–2021 into Equation 2 to calculate the mean WUA and coefficient of variation (CV) to assess across-day within-year WUA variation during the reservoir drawdown period (Figure 4-3B). We specifically focus on the reservoir drawdown period, rather than the irrigation season overall, because streamflow and habitat are most tightly managed during the drawdown period. We assessed for normality in mean WUA calculations and identified that no transformations were needed. We also tested for autocorrelation in mean WUA and, finding no significant autocorrelation, we treated each data in each water year as an independent observation. We used two-sided t-tests assuming unequal variance to compare CV, mean WUA, and mean WUA adjusted for annual streamflow between the two management regimes (Table 4-2). We adjusted mean WUA for annual streamflow by dividing WUA by unregulated (natural) streamflow (Joint Committee, 2018). Within each of the three responses (CV, WUA, adjusted WUA), we used Bonferroni's correction to adjust the significance level to accommodate the three tests (three fish species) conducted. To obtain a family-wide Type I error rate of 0.05, individual tests were considered significant at 0.017. All calculations and analyses were conducted in R (R Core Team, 2022).

4. Results

Our model demonstrated a positive relationship between streamflow and WUA for each adult species, and that there was more suitable habitat in the ~6.4 km lower subreach than the ~2.4 km upper subreach (Figure 4-5).

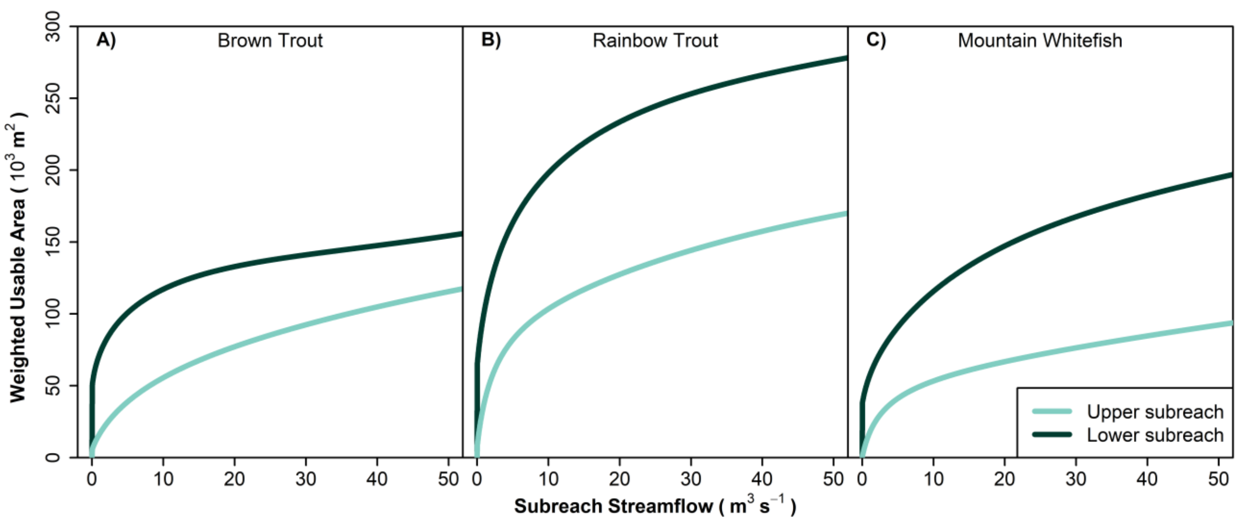

Figure 4-5. Weighted Usable Area (WUA) curves for each adult species in the upper and lower subreaches, separately. Note: a small initial amount of flow provides a sizeable increase in WUA.

Total WUA is the sum of WUA in the lower and upper subreaches at their respective streamflows, where streamflow in the lower subreach is water remaining in the river after the CF diversion. As a result, total WUA in the management reach increased with larger irrigation-season targets at Parker (Figure 4-6). By setting a flow target at Parker, streamflow and thus WUA in the lower subreach were more consistent and flow was added to the upper subreach to accommodate for the downstream CF diversion.

The range in total WUA varied given the combination between the flow target at Parker and the streamflow in the upper subreach, upstream of where water is diverted into the CF canal (Figure 4-6). For example, total WUA for adult Brown Trout could be ~172,000–190,000 m^2, a range of ~18,000 m^2, when the flow target is 9.9 m^3s^{-1}, depending on how much CF is diverting (Figure 4-6). In 2020 and 2021, when the 9.9 m^3s^{-1} flow target at Parker was implemented, nominal flow at Parker averaged 12.7 m^3s^{-1} during the reservoir drawdown period. At this flow, total WUA shifted upwards and its range shrunk by ~12% compared to a 9.9 m^3s^{-1} target (Figure 4-6). As the irrigation-season target at Parker decreased, total WUA decreased and the range increased. For example, when streamflow at Parker was 6.2 m^3s^{-1}—the lowest streamflow measured at Parker by the research team in 2019—total WUA could be between ~149–171,000 m^2, expanding the range to 22,000 m^2.

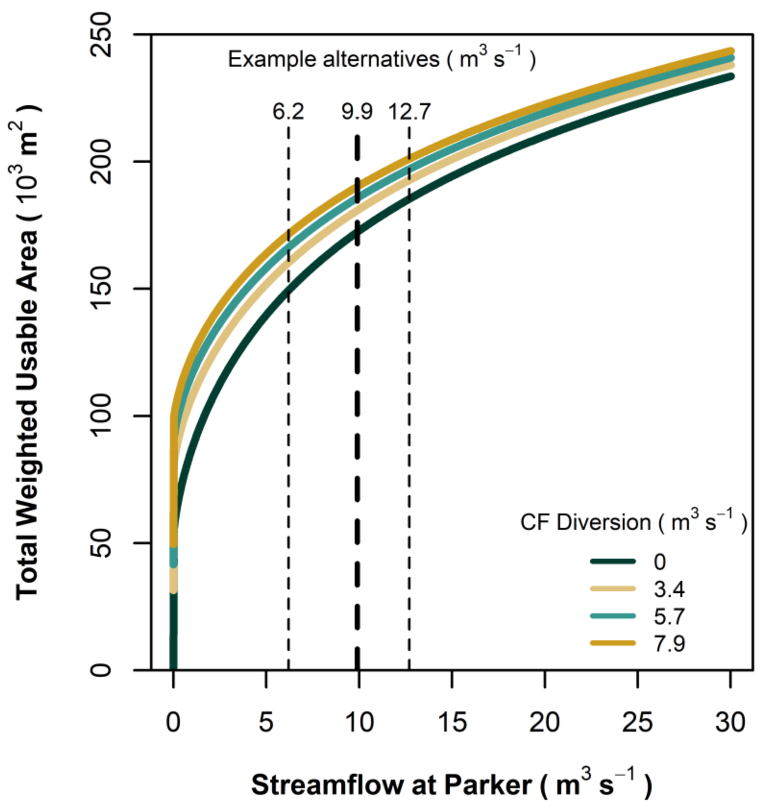

Figure 4-6. Total Weighted Usable Area (WUA) for adult Brown Trout as a function of streamflow at Parker, with higher streamflow in the upper subreach to support Consolidated Farmers (CF) diversions, where 7.9 m^3s^{-1} is the maximum water right for the canal. The thick dashed line is the 9.9 $m^3\ s^{-1}$ irrigation-season flow target implemented in water year 2020. The thin dashed lines are alternative streamflows observed in the lower subreach, with 6.2 m^3s^{-1} measured by researchers at Parker in 2019 and 12.7 m^3s^{-1} the mean streamflow at Parker during the reservoir drawdown period in 2020 and 2021. Together, the dashed lines demonstrate how total WUA changes depending on the irrigation-season flow target and the volume diverted by CF.

WUA over the water year largely reflected the annual hydrograph (Figure 4-7). Habitat for all species was lowest in July. Habitat available for juvenile Brown Trout and Mountain Whitefish during July was greater than that for adults of these species. Juvenile Rainbow Trout always had less suitable habitat than adult Rainbow Trout, but had similar suitable habitat ranges as adult and juvenile Brown Trout and Mountain Whitefish. The magnitude of WUA throughout the year was similar for adult and juvenile Brown Trout, whereas the WUA for adult Rainbow Trout almost doubled the adult WUA for the other two species. Brown Trout spawning habitat was nearly constant over their spawning season.

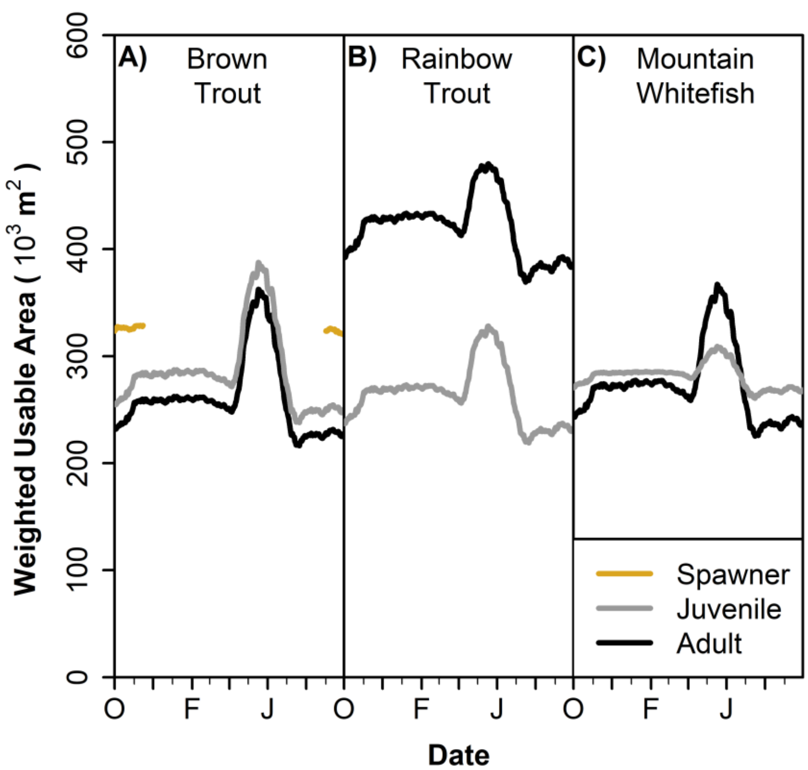

Figure 4-7. Mean Weighted Usable Area (WUA) by species for water years 1978–2021.

For water years 1978–2021, the mean annual drawdown-period WUA for adults of each species has varied, with significant difference between the two management regimes (1978–2017 and 2018–2021; Figure 4-8). Excluding water years 1983 and 1984—when Island Park Reservoir was drawn down for dam repair beyond what was needed for irrigation—mean drawdown-period WUA for adult Brown Trout in the latter regime significantly decreased from an average of ~223,000 m^2 to ~207,000 m^2 ($t_{10.0} = 3.3, p = 0.009$). However, there was no difference in mean drawdown-period WUA between the two regimes when accounting for total annual natural streamflow ($t_{4.7} = 0.5, p = 0.6$). Thus, WUA was lower in years with low natural flow and vice versa. However, there was no significant difference in either test for adult Mountain Whitefish ($t_{6.5} = 3.1, p = 0.02$; $t_{5.6} = 1.2, p = 0.3$) and adult Rainbow Trout ($t_{6.6} = 2.9, p = 0.03$; $t_{4.7} = 0.4, p = 0.7$). Results were similar when calculating WUA with nominal streamflow (Appendix C).

The coefficient of variation (CV) shows variation relative to the mean WUA within the drawdown period (Figure 4-8). Excluding water years 1983 and 1984, within-year CV was significantly lower for adult Brown Trout in water years 2018–2021 ($t_{15.5} = 4.1, p = 0.009$). Similar results occurred for adult Mountain Whitefish ($t_{3.0} = 10.4, p = 0.013$) and Rainbow Trout ($t_{3.0} = 10.7, p = 0.012$). For all adult species, within-year CV was 1.6–1.8 times larger before 2018. When using nominal streamflow to calculate WUA, within-year CV was significantly lower in water years 2018–2021 for all adult species (Appendix C).

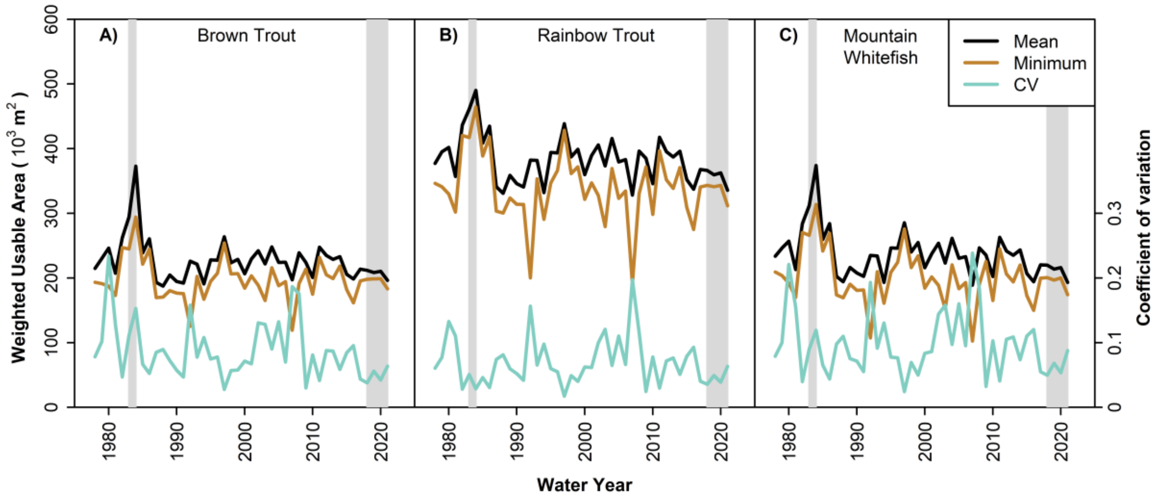

Figure 4-8. Mean Weighted Usable Area (WUA), minimum 7-day moving average WUA, and coefficient of variation (CV) in WUA for adult life stages across all species during the Island Park Reservoir drawdown period in water years 1978 through 2021. Two time periods are shaded within each panel: 1983–1984 when the reservoir was drawn down excessively for dam maintenance and 2018–2021 when the Committee changed irrigation-season management to focus on minimizing reservoir drawdown.

Minimum 7-day moving average WUA typically occurred in mid-July and coincided with peak demand at the CF canal (Appendix C). On average, adult Rainbow Trout had more suitable habitat compared to the other two species (Figure 4-8). However, during periods of low streamflow, adult Rainbow Trout and Mountain Whitefish had larger declines in minimum WUA compared to Brown Trout (Figure 4-8).

5. Discussion

Our streamflow-habitat models demonstrate that the longer, lower subreach contains the most suitable habitat for adult species within the management reach and that total irrigation-season WUA for fish is dynamic with streamflow, diversions, and management objectives in our study reach. Thus, a management approach that prioritizes the lower subreach and accounts for management reach dynamism aligns with Committee goals. Assuming channel morphology has not changed significantly during the assessment period (1978–2021), recent changes to watershed management have not decreased mean irrigation-season WUA relative to natural streamflow, but have reduced within-season habitat variation.

5.1 Flow variability

In 2018, the Committee adopted a precision-based management regime to minimize reservoir drawdown and avoid extremely low flows and high variability within the management reach. Our study identified that such precision management led to lower variability in drawdown-period WUA for water years 2018–2021 (Figure 4-8; Table 4-1). The reduction in variability was also due to state and federal water managers allowing the Henrys Fork watershed to be managed in isolation from the rest of the upper Snake River basin to meet water demands within the watershed. The new irrigation-season flow target of 9.9 m^3s^{-1} at Parker was implemented in 2020 and contributed to reduced WUA variability during the latter half of the 2018–2021 precision-based management regime. The management reach is subject to some large, but infrequent sub-daily flow fluctuations due to the dynamism of irrigation demand relative to the 24-h travel time of reservoir water delivery and temporary flow reductions to ~12 m^3s^{-1} by an intervening run-of-river hydroelectric facility when it trips offline. By moving the target to the downstream extent of the management reach, streamflow in the longer, lower subreach—where more than half of WUA for all adult species is located (Figure 4-5)—is more consistent and thus habitat more resilient to these sub-daily fluctuations. Overall, moving the target to the downstream reach extent contributed to, but was not wholly responsible for, reduced WUA variability compared to when the flow target was located at St. Anthony, upstream of four substantial diversions (Figure 4-8).

Low variability is important for fish during the low flow season. Ecological responses and habitat availability may be sensitive to small percentage changes in low flow magnitude (Rolls et al., 2012). Extreme low flows can disconnect habitats (Dzara et al., 2019), reduce productive riffle habitat (Bradford & Heinonen, 2008), and constrict fish to higher densities in refuge areas—increasing competition (Rolls et al., 2012) and predation risk (Jackson et al., 2001). Daily fish movements are typically small (Höjesjö et al., 2007; Wolf & Brewer, 2021) and high daily flow variability requires high energy expenditure as fish seek suitable habitat (Taylor et al., 2012). However, seasonably stable, low flows may facilitate non-native species invasion (Bunn & Arthington, 2002). In the Henrys Fork management reach, non-native Brown Trout abundance has increased relative to non-native Rainbow Trout since 2004 (Vincent et al., 2023). But the reach is also considered a watershed stronghold for native Mountain Whitefish (Heckel, 2021; Meyer et al., 2009). We acknowledge that differences in flow regimes may impact species composition, but do not have sufficient data to attribute environmental flow management to species composition changes in the Henrys Fork management reach. Ultimately, the Committee seeks to reduce streamflow variability in the management reach throughout the low flow season and their management approaches as of 2018 have met that goal.

5.2 Site specificity

Our study emphasizes the need for site-specific flow assessments to inform water management. The original lower Henrys Fork irrigation-season targets centered on real-time flow reported at a convenient USGS streamflow gage at St. Anthony. However, reliance on this gage alone failed to account for surface water withdrawals occurring immediately downstream, where flow and fish habitat depend inversely on diversion magnitude. As a result, we documented streamflow below the last diversion as low as 6.2 m^3s^{-1}.

Choosing environmental flow attributes like magnitude and location based on convenience and transferability are attractive starting points for environmental water management, given data and funding limitations. But specificity matters when setting instream flows (Arthington et al., 2018; Opperman et al., 2019). In our study, we moved beyond previous percentage-mean-annual-flow recommendations based on desktop methods (Idaho Department of Water Resources, 1999) and quantified streamflow-habitat relationships for species relevant to local recreational fisheries (Vincent et al., 2023). The Committee’s 2020 decision to move the flow target to the downstream extent of the management reach now accounts for canal diversions to provide more consistent habitat throughout the reach. Although expensive, field visits at a range of flows provided important local and regional context (Swales & Harris, 1995). Field visits can also ensure diverse river relationships are considered (Anderson et al., 2019) and may be mandatory to ensure flow assessments maintain Indigenous values (e.g., Māori Cultural Opportunity Assessments, Tipa & Nelson, 2012). In our study, frequent river floats allowed us to 1) experience first-hand the magnitude of flow variability from the upstream flow target and canal diversions and 2) document the presence and potential role of groundwater springs in the management reach (Van Kirk et al., 2020).

5.3 Tools for success: Collaboration, operational flexibility, and data products

Collaboration and relationship building are critical to environmental flow management (Conallin et al., 2017; Owusu et al., 2021; Richter et al., 2006). In the Henrys Fork watershed, water management authorities do not have a jurisdictional obligation to consider fisheries needs. Prior to Committee creation, state agencies made flow recommendations for aquatic habitat in the management reach (Cochnauer, 1978; Idaho Department of Water Resources, 1999) and, to the best of our knowledge, these recommendations were not formally adhered to or implemented. Given the Committee's federal mandate to regularly meet, it is the only standing, formal avenue for Henrys Fork fisheries stakeholders to work with water management authorities to seek mutually beneficial management solutions. Thus, the Committee is a long-established avenue for collaborative management.

We used the Committee platform to communicate our field observations, which resulted in adoption of a new 9.9 m^3s^{-1} flow target at Parker on a trial basis in 2020. Ultimately, changing the streamflow target to account for canal diversions did not result in additional reservoir storage in water year 2020. No change in reservoir drawdown volume was expected given the average equivalence of the old target at St. Anthony and the new target at Parker (Figure 4-2). However, the new target did result in 10 Mm^3 additional reservoir storage in water year 2021. Water year 2020 was near-average with natural flow 92% of average, whereas water year 2021 was drier with natural flow 75% of average(Van Kirk, 2020, 2022). As a result, the reservoir drawdown period in 2021 was four weeks longer than average—extending the period in which the flow target was applied. Beyond upstream reservoir storage, desired WUA consistency was retained in the management reach in both years (Figure 4-8). Environmental flow recommendations must be defensible for water management consideration (Arthington et al., 2018). Because the new irrigation-season flow target succeeded in meeting multiple objectives in the Henrys Fork watershed, it has since become the default operational strategy. Beyond the new target itself, our results emphasize the benefit of stakeholder collaboration for meeting the Committee's management goal to reduce streamflow variability in the management reach during the low-flow season.

Operational flexibility and adaptive management frameworks can be especially useful for revising environmental flow approaches as new information becomes available (Warner et al., 2014; Webb et al., 2018; Wineland et al., 2022). The non-legally binding nature of the Henrys Fork flow target allowed for swift adaptation by the Committee. Flow experimentation to test proposed dam operations have benefited environmental flow implementation globally (Owusu et al., 2021). However, legislation and regulation are also means to secure environmental flows (Harwood et al., 2018; Opperman et al., 2019; Owusu et al., 2021). Indeed, the flow target for the Henrys Fork was spurred by a federal mandate requiring stakeholders discuss headwater dam operations for human and aquatic ecosystem needs. Implementing environmental flows may be possible at local and hyper-regional scales without legislative requirements, provided strong stakeholder collaboration (Harwood et al., 2018; Van Kirk et al., 2019; Wineland et al., 2022).

Modeling and data tools for environmental flow management are also useful and increasingly necessary (Cantor et al., 2021; Richter et al., 2006, 2012). To support manager uptake of the new streamflow target and replicate the convenience of the St. Anthony streamflow gage, the Henry's Fork Foundation developed an R shiny web application that retrieves streamflow and diversion data and displays streamflow at Parker to inform real-time decision-making. This approach provides data products to support decision maker needs (Cantor et al. 2021). In addition, a federal grant funded installation of real-time gages and remote-controlled canal gates at critical locations throughout the watershed in 2020, allowing managers to more precisely adjust water delivery operations to meet the downstream flow target. Although these data tools and instrumentation have helped meet needs in the Henrys Fork, expense and maintenance can be barriers in other systems (Cantor et al., 2021; Richter et al., 2006, 2012).

5.4 Quantifying tradeoffs and adaptive design

Tradeoffs are inherent to managing water for many uses, and conflict is common (Harwood et al., 2018; Owusu et al., 2021; Wineland et al., 2022). In our applied case study, we present environmental flow management that has benefited both aquatic habitat and irrigation storage. Some water resource optimization studies have reported similar theoretical results. Porse et al. (2015) suggested that meeting environmental flow requirements for riparian species in the Big Bend reach of the Rio Grande would not impact total water supply allocations, with some tradeoffs on release timing. Using a multi-objective optimization approach, Chen & Olden (2017) designed dam releases predicted to increase native fish abundance and nonnative fish losses without threatening societal water needs—even in drought years. Owusu et al. (2021) moved beyond theory and identified six cases across multiple countries where dam re-operation successfully met environmental flow needs with no or minimal impact to human use.

Managing social-ecological tradeoffs will become increasingly challenging in water-limited systems as water scarcity and climate variability increase (Wineland et al., 2022). Integrating surface-groundwater interactions into environmental flow assessments for groundwater-influenced streams is one way managers can mediate low flows and high temperatures exacerbated by climate change (Lapides et al., 2022; Yarnell et al., 2022). More broadly, watershed-specific studies will maximize local utility, resiliency, and buy-in (Pahl-Wostl, 2009; Welsh et al., 2013).

6. Conclusion

Our research emphasizes the importance of site-selection and stakeholder collaboration for environmental flow target implementation and managing habitat variability. We provide an implemented, stakeholder-supported example where environmental flows were improved to meet goals for low variability within a regulated irrigation system. In addition to habitat assessments, designer flows (Chen & Olden, 2017; Tickner et al., 2017) and reservoir re-operation (Null et al., 2022; Owusu et al., 2021; Warner et al., 2014) can further facilitate efforts to balance human and environmental water needs.

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8. Conflict of Interest

The authors declare no conflict of interest. C.N.M and R.V.K are affiliated with or employed by the Henry's Fork Foundation, a member organization of the Henry's Fork Drought Management Planning Committee.

9. Data Availability Statement

The data and code that support the findings of this study are available on Hydroshare in the following repository: https://doi.org/10.4211/hs.f738261bc1ed42f0bc71a5b14e0f90ac

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CHAPTER 5

CONCLUSION

Water resource management for multiple stakeholders is an already challenging endeavor, further complicated by climate change. A systems-thinking approach that integrates the physical and social components of water resource management within multiple scales is imperative for climate adaptation, effective resource allocation, and watershed conservation. The integrated water resource management (IWRM) framework leverages interdisciplinarity to identify climate adaptation strategies for diverse water uses.

This dissertation is an example of integrated water resource management research in an applied, place-based context. In Chapter 2, I demonstrate the importance of considering water management from multiple scales, as individual farm-scale decisions can compound to have basin-scale impacts on hydrologic processes. I emphasize how improving irrigation efficiency should be pursued with caution and intent. In Chapter 3, I explore how aquifer recharge can effectively recover and maintain groundwater pathways to the river. I determine that flood irrigation paired with senior water rights is a mechanism for climate resiliency. Together, Chapters 2 and 3 hone in on groundwater-surface water relationships and how they can contribute to water scarcity or be leveraged to benefit streamflow and aquatic habitat. Chapter 4 is informed by groundwater-surface water relationships, but focuses on surface-water management of an upstream reservoir in maintaining summer aquatic habitat. In Chapter 4, I demonstrate the importance of familiarity with the resource and collaborative stakeholder management. Floating the river frequently and at multiple streamflows fostered my understanding of streamflow variability within the study reach and facilitated discovery of groundwater springs not previously documented. Collaborative stakeholder management allowed my findings to be integrated in real-time.

This work was stakeholder-driven and locally-informed. Chapters 2 and 3 document the history and potential future of the watershed should we leverage senior rights for broader adoption of aquifer recharge via flood irrigation. Chapter 4 puts current management into the context of past management, and has already had positive impact on watershed management to date—contributing to increased reservoir carryover that benefits fish and aquatic habitat throughout the watershed. Overall, this research uses a place-based context to demonstrate an integrated water resource management approach in-action.

APPENDICES

APPENDIX A: SUPPLEMENTAL INFORMATION FOR CHAPTER 2

1. Background

1.1 Estimation of groundwater withdrawal trends in the study area

From U.S. Geological Survey water use data, groundwater withdrawal for irrigation statewide increased at an average rate of 18.6 Mm^3/year from 1975-2015, while surface water withdrawal decreased at an average of 60.9 Mm^3/year over that time period (Figure A-1). From county-level data, groundwater withdrawal in the Henrys Fork watershed (Fremont, Madison, and Teton counties, Idaho) in 2015 was 358 Mm^3, which was 30% of total irrigation withdrawal. This matched the statewide figure of 32%. Trends in surface-water diversion in the watershed also match those reported statewide, so it is reasonable to assume that trends in groundwater withdrawal in the watershed match the statewide trend. In 2015, groundwater withdrawal for irrigation in the Henrys Fork watershed was 5.3% of the statewide total. Of the groundwater withdrawal in the Henrys Fork watershed in 2015, 54% was withdrawn specifically from the Eastern Snake Plain Aquifer, which is our study area. Thus, groundwater withdrawal in our study area in 2015 was around 2.9% of the statewide total. Applying this fraction to the 18.6 Mm^3/year increasing trend in groundwater use statewide yields an increase in groundwater withdrawal in our study area of 0.54 Mm^3/year. Over the 45-year study period, this equals an increase in groundwater withdrawal of 24 Mm^3.

Table A-1. Groundwater and surface-water withdrawal in the Henrys Fork watershed.

Year	Groundwater Withdrawal (Mm^3)	Surface water withdrawal (Mm^3)	Citation
1975	4810.6	16035.4	Murray and Reeves (1977)
1980	5550.7	16035.4	Solley, Chase and Mann IV(1983)
1985	4082.9	21339.4	Solley, Merk and Pierce (1988)
1990	9152.5	16652.1	Solley, Pierce and Perlman (1993)
1995	3478.4	14555.2	Solley, Pierce and Perlman (1998)
2000	5143.7	18502.4	Hutson *et al.* (2004)
2005	5353.3	17515.6	Kenny *et al.*(2009)
2010	5279.3	14185.1	Maupin *et al.*(2014)
2015	6771.9	14308.5	Dieter *et al.*(2018)

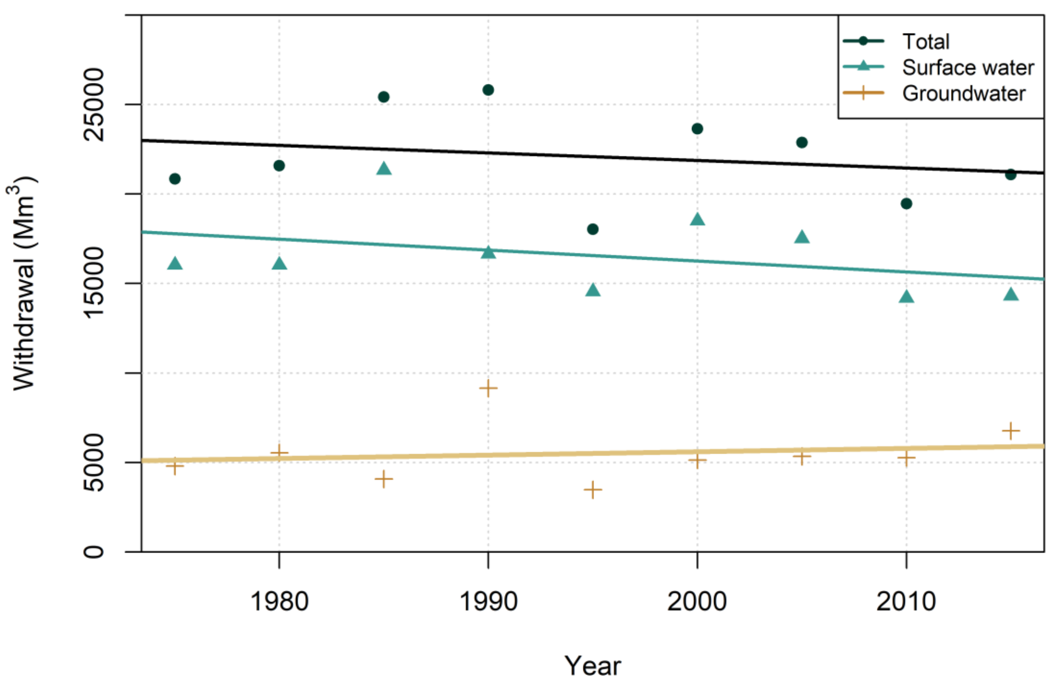

Figure A-1. Trends in groundwater and surface-water withdrawals in the Henrys Fork watershed.

1.2 Land slope and soil for irrigated areas

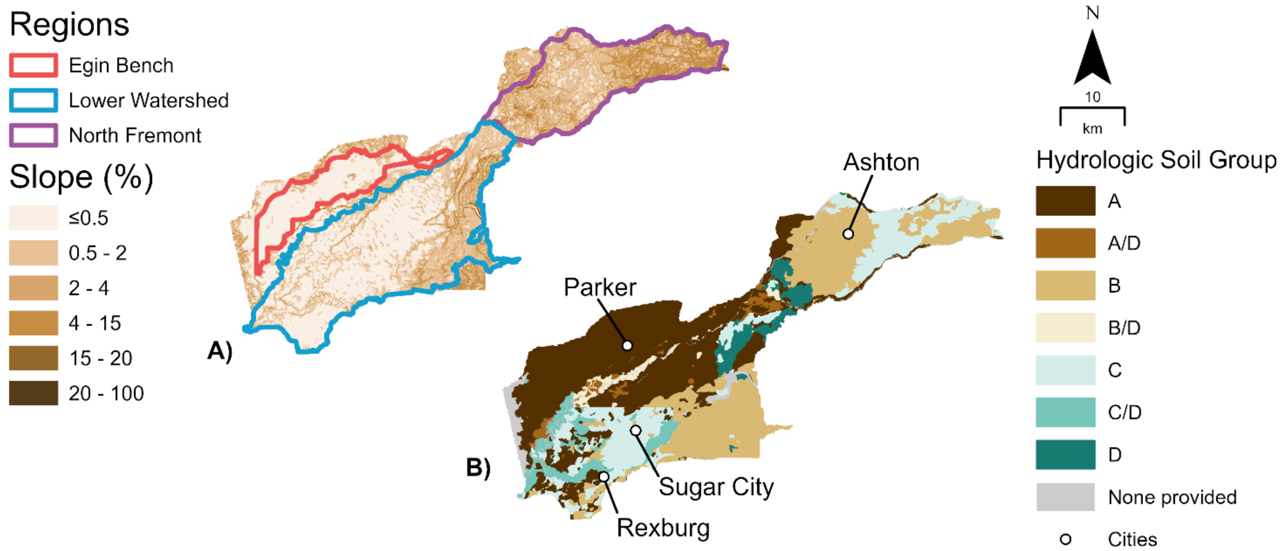

Figure A-2. Panel A shows land slope (% gradient) and Panel B depicts the hydrologic soil group relative to cities. Note that hydrologic soil groups are classified A–D and reference runoff potential as lowest, moderately low, moderately high, and highest (Natural Resources Conservation Service, 2007). Each soil group comprises multiple soil textures. The topographic slope layer was created using a 30-m Digital Elevation Model (U.S. Geological Survey, 2022). we also visualized soil layers for each irrigation area using hydrologic soil group data from the Soil Survey Geographic Database (USDA NRCS, 2022).

2. Methods

2.1 Interview instrument

- Which canal(s) (do/did) you (manage/get water from)?
- Have you/your farm/company converted from flood to sprinkler irrigation?
    1. If yes:
        - Can you tell me about how your canal company/farm converted from flood to sprinkler irrigation?
        - When did your company/farm convert?
        - Where did your company/farm convert?
        - Why did your company/farm convert?
            - Probing: Impact of commodity prices, neighboring farms, was conversion part of a cost-sharing conservation plan with NRCS or the FSA?
        - How did your company/farm convert?
            - Probing: How many acres, Phased conversion or all at once
    2. If no:
        - How long have you been flood irrigating?
        - Where do you flood irrigate?
            - How many acres?
        - What motivates you to continue to flood irrigate, rather than convert to sprinkler application?
        - What challenges do experience with flood irrigation?

- What else should I be asking that I didn't ask? What else should I know?
- Do you have any recommendations for other folks I should talk to? Would you mind sharing their contact information?

2.2 Geospatial analysis

Table A-2. Geospatial imagery characteristics. For Landsat imagery, we used color infrared to determine time within the summer growing season to ensure photos assessed had similar crop growth. We used natural color to determine pivot vs. not-pivot irrigation and used one cloud-free photo from a given 16-day photo set.

Imagery	Data Source	Region	Spatial Resolution	Temporal Resolution	Details	Year(s)
Aerial	Idaho State Water Adjudication	Snake River, Basin 22	3 m	2 years	Color infrared	1987–1988
Satellite	USGS Landsat 4	Path 38 Row 29 Path 38 Row 30	30 m	1 day	Color infrared (Bands 5, 4, 3) Natural color (Bands 4, 3, 2)	1982–1993
Satellite	USGS Landsat 5	Path 38 Row 29 Path 38 Row 30	30 m	1 day	Color infrared (Bands 5, 4, 3) Natural color (Bands 4, 3, 2)	1984–2013
Satellite	USGS Landsat 8	Path 39 Row 29 Path 38 Row 30	30 m	1 day	Color infrared (Bands 5, 4, 3) Natural color (Bands 4, 3, 2)	2013–2020

2.3 Hydrologic analysis: data compilation and computation

For Equation 1 in the main text, a diversion calculation adjustment was required to account for the Crosscut Canal injection to the Teton River. The Crosscut Canal diverts water from the Middle Henrys Fork and terminates in the Middle Teton River. The Crosscut Canal diverts two types of administrative water at the same point of diversion on the Middle Henrys Fork. One type of water is irrigation water, administered as storage and natural flow, subsequently distributed into canals within the Southeast Idaho Canal Company system on land between the Crosscut Canal diversion and terminus points. The second type of water is administrative storage delivered from Grassy Lake and Island Park Reservoir that is conveyed and injected into the Teton River. For the reach gain equation at the subreach scale (Equation 1 in main text), all water diverted into the Crosscut Canal count towards diversion from the Middle Henrys Fork. All physical water diverted from the Middle Teton River, including water injected from the Crosscut Canal, count towards diversion from the Middle Teton River.

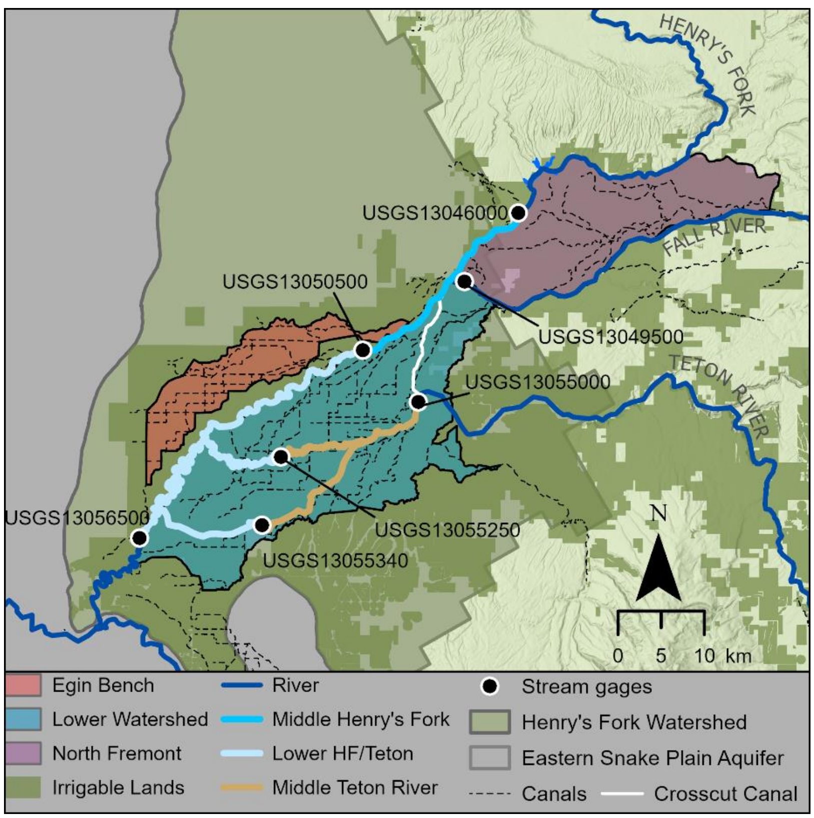

Figure A-3. U.S. Geological Survey stream gages (labeled by identification number) used in the water balance and reach gain calculations.

However, for the watershed-scale water balance, we cannot double count water diverted into the Crosscut Canal. Therefore, water diverted into the Crosscut Canal for injection to the Middle Teton River were subtracted from total Middle Henrys Fork diversions and added to total streamflow in the Middle Teton River. Thus, total flow in the Middle Teton River is the sum of natural flow, Crosscut Canal injection, and exchange well injection. Total inflow to the watershed is:

$$\begin{aligned} &watershed\ inflow \\ &= watershed\ unregulated\ flow - \Delta storage_{reservoir} \\ &+ exchange\ well\ injection - evaporation_{reservoir} \end{aligned} \tag{A1}$$

Where watershed unregulated flow is the sum of natural or unregulated flow in the Henrys Fork, Fall River, and Teton River. The Crosscut Canal does not enter into the watershed-scale inflow equation because it is subtracted from the Henrys Fork and subsequently added to the Teton River. The $\Delta storage_{reservoir}$ term is included to account for years when reservoir storage at the beginning and end of a year are not equivalent. If end-of-year storage is less than beginning-of-year storage, then the $\Delta storage_{reservoir}$ term is negative and indicates that additional water was added to total streamflow through reservoir storage.

For Equation 2 in the main text, regulated streamflow data used the following long-term USGS stream gaging stations: Henrys Fork at Ashton (USGS13046000), Fall River at Chester (USGS13049500), and Teton River at St. Anthony (USGS13055000; Figure A-3). Annual watershed outflow is regulated streamflow in the Henrys Fork at Rexburg (USGS13056500; Figure A-3), near the bottom of the watershed at the confluence with the main Snake River. Streamflow gages used in our calculations are noted in Table A-3.

Table A-3. Inflow and outflow U.S. Geological Survey stream gage(s) used to calculate reach gains at the watershed- or reach-scale.

Reach Name	Inflow Gage(s)	Outflow Gage(s)
Watershed Total	USGS13046000 USGS13049500 USGS13055000	USGS13056500
Middle Henrys Fork	USGS13046000 USGS13049500	USGS13050500
Middle Teton River	USGS13055000	USGS13055250 USGS13055340
Lower Henrys Fork/Teton	USGS13050500 USGS13055250 USGS13055340	USGS13056500

2.4 Hydrologic analysis: statistical modeling

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). The AIC is a relatively easily understood information criterion that has firm mathematical basis in theory of both statistical likelihoods and information. The basic AIC formula is:

$$AIC = -2\log(\mathcal{L}) + 2p \tag{A2}$$

where $\mathcal{L}$ is the statistical likelihood of a fitted model, log is the natural logarithm, and p is the number of parameters fitted in the model, including all structural parameters such as means, slopes, and intercepts, and any and all parameters describing the probability structure of the model such as variances, covariances, and autocorrelation coefficients. Lower AIC scores indicate better models in the sense that the data provide more evidence for that particular model among those in the candidate set. The $2p$ term penalizes models for the number of parameters included, so that AIC tends to favor more parsimonious models than might be selected based solely on statistical significance of parameter estimates. 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.

The best model out of the candidate set is the one with lowest AICc score, and then all other models are ranked in ascending order of ΔAICc with respect to this top model. These ΔAICc values can be converted into model weights w_i via the formula

$$w_i = \frac{\exp\left(-\frac{1}{2}\exp(\Delta AICc_i)\right)}{\sum_j \exp\left(-\frac{1}{2}\exp\left(\Delta AICc_j\right)\right)} \tag{A3}$$

where the sum in the denominator is taken over all models in the candidate set. This normalization produces a set of weights that sum to 1. Weighted averages of model parameters, fitted values, and covariances yield an evidence-based fitted model that reflects the relative evidence for all models in the candidate set, and parameter estimates that have optimal balance between bias and standard error. Model weights can also be used to identify particular model components that are more strongly or less strongly supported by the data.

Model averaging using AICc offers numerous advantages over traditional statistical hypothesis testing. First, it allows simultaneous evaluation of a number of scientific hypotheses, each represented by a particular model or subset of models in the candidate set. Second, the AICc can be used to compare models that are not nested within each other, a requirement for model selection using stepwise hypothesis testing. Third, the AICc can compare models with different variance and distributional structures, for example comparing models with lognormal variance against analogous models with normal variance. One drawback of the AICc method is that results depend on the particular models in the candidate set. The set of candidate models should be chosen based on reasonable and parsimonious descriptions of the data that are grounded in knowledge of the system being studied. Simulation modeling suggests that once a given set of parameters and model structures is determined, all possible combinations of the parameters and structures should be used in order to obtain the most appropriate set of model weights (Doherty, White and Burnham, 2012).

After ranking the models by ΔAICc, the additional step of removing redundant or "pretending" models must be taken to correctly calculate model weights (Anderson, 2008). Redundant models occur when a parameter with no predictive power occurs in a particular model that otherwise has reasonable model weight. In this case, the addition of the poor predictor is equivalent to adding a parameter whose value is 0. The addition of that parameter to a particular model does not increase the model likelihood but increases AIC by 2, the value of the $2p$ term when p is increased by one. Thus, the pretending model may still look very good in relation to other models, but it is actually redundant with the first one and should be removed so that the remaining models receive appropriate weights.

3. Results

3.1 Statistical hydrologic analysis

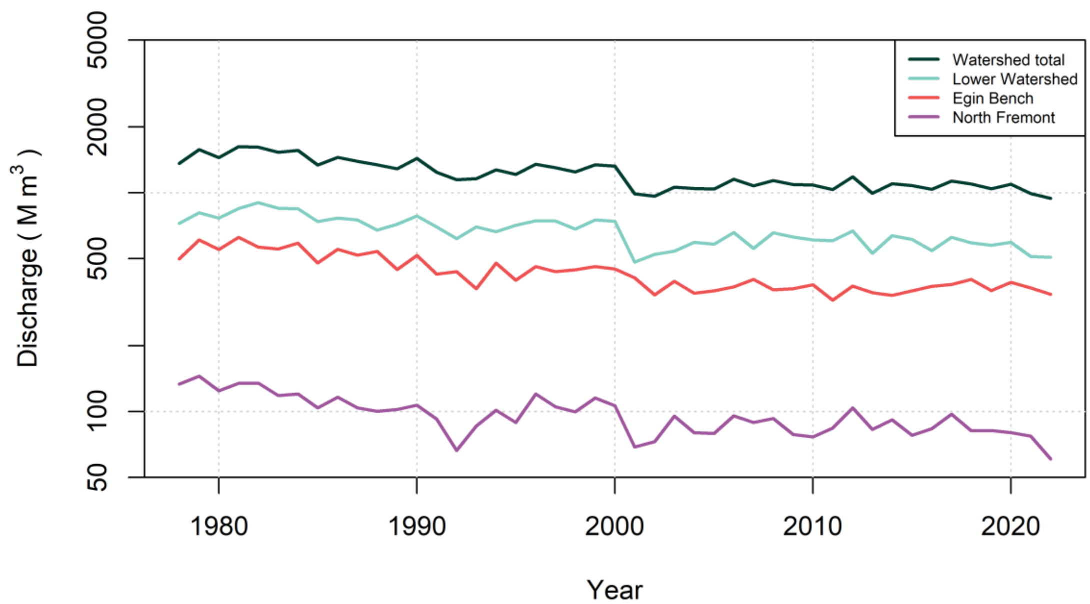

Figure A-4. Henrys Fork watershed diversions by watershed total and irrigation study regions.

The pattern and relative magnitude of decrease in diversion was uniform across all of the irrigated areas (Figure A-4).

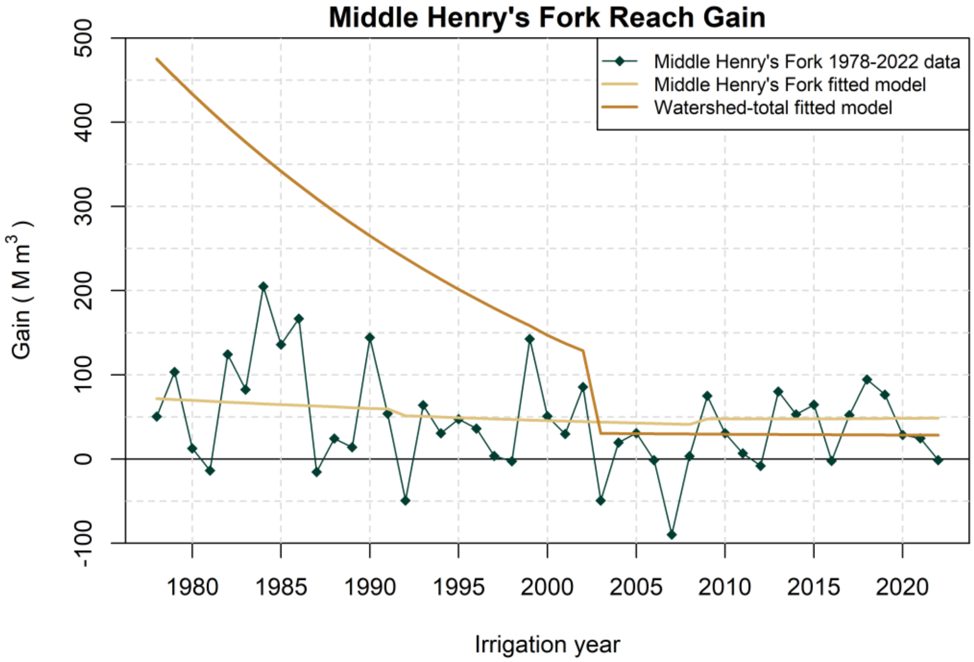

Figure A-5. Trends in reach gains for the Middle Henrys Fork subreach for irrigation years 1978–2022. The watershed-total trend is overlaid for reference.

Although sub-reach analysis was limited by data availability, there was little evidence for reach gain decline in the middle Henrys Fork reach (Figure A-5). Of ten models that accounted for ~100% of model weight, eight (77.2% of weight) included terms representing temporal change, but two of those (13.7% of weight) were quadratic models that indicated a minimum around the year 2000, followed by a mild increase since then. Four models, accounting for 49.3% of model weight, identified a step-wise change, but the steps all occurred at the endpoints of our pre-specified 1992–2009 range. None identified a step change in the early 2000s. Annual reach gain was 61.4 Mm^3 from 1978–2000 and 27.5 Mm^3 in 2001–2022. Middle Henrys Fork reach gain was negative in four years in the 1978–2000 period and six years in the 2001–2022 period. Subwatershed unregulated streamflow appeared as a covariate in the top five models (76.2% of weight).

Mean annual gain over the 2004–2022 period was 24.2 Mm^3 for the whole watershed and 28.4 Mm^3 in Middle Henrys Fork, -109.8 Mm^3 in Middle Teton, and 105.6 Mm^3 in Lower Henrys Fork/Teton. In the 1978–2000 time period, reach gain in the middle Henrys Fork reach was 19% of the watershed-total reach gain, whereas it was 120% of the watershed total in 2001–2022—indicating that other reaches in the watershed have transitioned from gaining to losing reaches. In absence of stream gage data prior to 2004 on the Teton River, we were unable to determine when losses in the Middle Teton exceeded gains in the Lower Henrys Fork/Teton.

Table A-4. AICc table for analysis of Henrys Fork watershed surface-water diversion, after removal of redundant models, where Q_{unreg} refers to the inclusion of unregulated streamflow as a covariate, AR refers to the inclusion of the first-order autocorrelation term in the model, log is the natural logarithm, $\mathcal{L}$ is the statistical likelihood of a fitted model, and p is the number of parameters fitted in the model. Models with weight less than 0.002 are not shown.

Structural model	Q_{unreg} included	AR term	Log transform	p	$\log(\mathcal{L})$	AICc	ΔAICc	Weight	Cum. weight
Piecewise trend	Yes	No	Yes	6	65.168	-116.125	0.000	0.644	0.644
Piecewise trend	Yes	Yes	Yes	7	65.535	-114.043	2.082	0.228	0.872
Quadratic trend	Yes	No	Yes	5	61.809	-112.080	4.045	0.085	0.957
Quadratic trend	Yes	Yes	Yes	6	62.106	-110.002	6.123	0.030	0.987
Piecewise trend	No	No	Yes	5	58.940	-106.341	9.783	0.005	0.992
Piecewise trend	No	Yes	Yes	6	59.724	-105.237	10.888	0.003	0.995

Table A-5. AICc table for analysis of Henrys Fork watershed reach gain, after removal of redundant models, where Q_{unreg} refers to the inclusion of unregulated streamflow as a covariate, AR refers to the inclusion of the first-order autocorrelation term in the model, log is the natural logarithm, $\mathcal{L}$ is the statistical likelihood of a fitted model, and p is the number of parameters fitted in the model. Models with weight less than 0.002 are not shown.

Structural model	Q_{unreg} included	AR term	Log transform	p	log ($\mathcal{L}$)	AICc	ΔAICc	Weight	Cum. weight
Piecewise trend	Yes	No	Yes	6	-5.135	24.480	0.000	0.625	0.625
Piecewise trend	Yes	Yes	Yes	7	-4.473	25.973	1.493	0.296	0.921
Piecewise trend	No	No	Yes	5	-9.466	30.470	5.990	0.031	0.952
Piecewise trend	No	Yes	Yes	6	-8.821	31.852	7.372	0.016	0.968
Linear trend	Yes	Yes	Yes	5	-10.388	32.314	7.833	0.012	0.980
Quadratic trend	Yes	Yes	Yes	6	-9.504	33.219	8.739	0.008	0.988
Piecewise constant	No	Yes	Yes	5	-11.334	34.205	9.725	0.005	0.993
Quadratic trend	Yes	No	Yes	5	-12.098	35.734	11.254	0.002	0.995

Table A-6. AICc table for analysis of Henrys Fork watershed unregulated streamflow, after removal of redundant models, where Q_{unreg} refers to the inclusion of unregulated streamflow as a covariate, AR refers to the inclusion of the first-order autocorrelation term in the model, log is the natural logarithm, $\mathcal{L}$ is the statistical likelihood of a fitted model, and p is the number of parameters fitted in the model. Models with weight less than 0.002 are not shown.

Structural model	Q_{unreg} included	AR term	Log transform	p	log ($\mathcal{L}$)	AICc	ΔAICc	Weight	Cum. weight
Constant	NA	Yes	Yes	3	14.613	-22.641	0.000	0.342	0.342
Piecewise constant	NA	Yes	Yes	5	16.881	-22.223	0.418	0.277	0.619
Linear trend	NA	Yes	Yes	4	15.602	-22.203	0.438	0.275	0.894
Piecewise trend	NA	Yes	Yes	6	17.025	-19.839	2.802	0.084	0.978
Piecewise constant	NA	No	Yes	4	12.321	-15.641	6.999	0.010	0.988
Linear trend	NA	No	Yes	3	10.561	-14.536	8.104	0.006	0.994

Table A-7. AICc table for analysis of net watershed diversion, after removal of redundant models, where Q_{unreg} refers to the inclusion of unregulated streamflow as a covariate, AR refers to the inclusion of the first-order autocorrelation term in the model, log is the natural logarithm, $\mathcal{L}$ is the statistical likelihood of a fitted model, and p is the number of parameters fitted in the model. Models with weight less than 0.002 are not shown.

Structural model	Q_{unreg} included	AR term	Log transform	p	log ($\mathcal{L}$)	AICc	ΔAICc	weight	Cum. weight
Constant	No	No	No	2	-565.551	1135.388	0.000	0.584	0.584
Constant	No	No	Yes	2	-565.892	1136.069	0.682	0.416	1.000

Table A-8. AICc table for analysis of net watershed export, after removal of redundant models, where Q_{unreg} refers to the inclusion of unregulated streamflow as a covariate, AR refers to the inclusion of the first-order autocorrelation term in the model, log is the natural logarithm, $\mathcal{L}$ is the statistical likelihood of a fitted model, and p is the number of parameters fitted in the model. Models with weight less than 0.002 are not shown.

Structural model	Q_{unreg} included	AR term	Log transform	p	log ($\mathcal{L}$)	AICc	ΔAICc	Weight	Cum. weight
Constant	No	No	No	2	-565.662	1135.609	0.000	0.585	0.585
Constant	No	No	Yes	2	-566.007	1136.299	0.690	0.415	1.000

Table A-9. AICc table for analysis of Middle Henrys Fork reach gains, after removal of redundant models, where Q_{unreg} refers to the inclusion of unregulated streamflow as a covariate, AR refers to the inclusion of the first-order autocorrelation term in the model, log is the natural logarithm, $\mathcal{L}$ is the statistical likelihood of a fitted model, and p is the number of parameters fitted in the model. Models with weight less than 0.002 are not shown.

Structural model	Q_{unreg} included	AR term	Log transform	p	log ($\mathcal{L}$)	AICc	ΔAICc	weight	Cum. weight
Piecewise constant	Yes	No	No	5	-542.156	1095.851	0.000	0.255	0.255
Constant	Yes	No	No	3	-544.877	1096.339	0.488	0.200	0.455
Piecewise trend	Yes	No	No	6	-541.693	1097.596	1.745	0.107	0.562
Trend	Yes	No	No	4	-544.322	1097.645	1.794	0.104	0.666
Quadratic trend	Yes	No	No	5	-543.141	1097.820	1.968	0.095	0.762
Piecewise constant	No	No	No	4	-544.451	1097.901	2.050	0.092	0.853
Linear trend	No	No	No	3	-546.328	1099.242	3.391	0.047	0.900
Piecewise trend	No	No	No	5	-544.027	1099.592	3.740	0.039	0.939
Quadratic trend	No	No	No	4	-545.481	1099.962	4.110	0.033	0.972
Constant	No	No	No	2	-547.992	1100.269	4.418	0.028	1.000

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APPENDIX B: SUPPLEMENTAL INFORMATION FOR CHAPTER 3

1. Methods

Our reservoir operation-hydrology model considered three target outflows (Figure B-1). The target on the mainstem Henrys Fork is at a location locally known as Parker and is 9.9 m^3/s. On the Teton River, target outflows are 0 m^3/s on the North Fork Teton and 2.8 m^3/s on the South Fork Teton.

The Egin Lakes managed aquifer recharge right can only be filled if streamflow on the mainstem Henrys Fork at St. Anthony (USGS 13050500) is at least 28.3 m^3/s. In our model, we also prevented MAR when Island Park Reservoir—the largest reservoir in our study system—was drafting (i.e., being drawn down to meet irrigation demand).

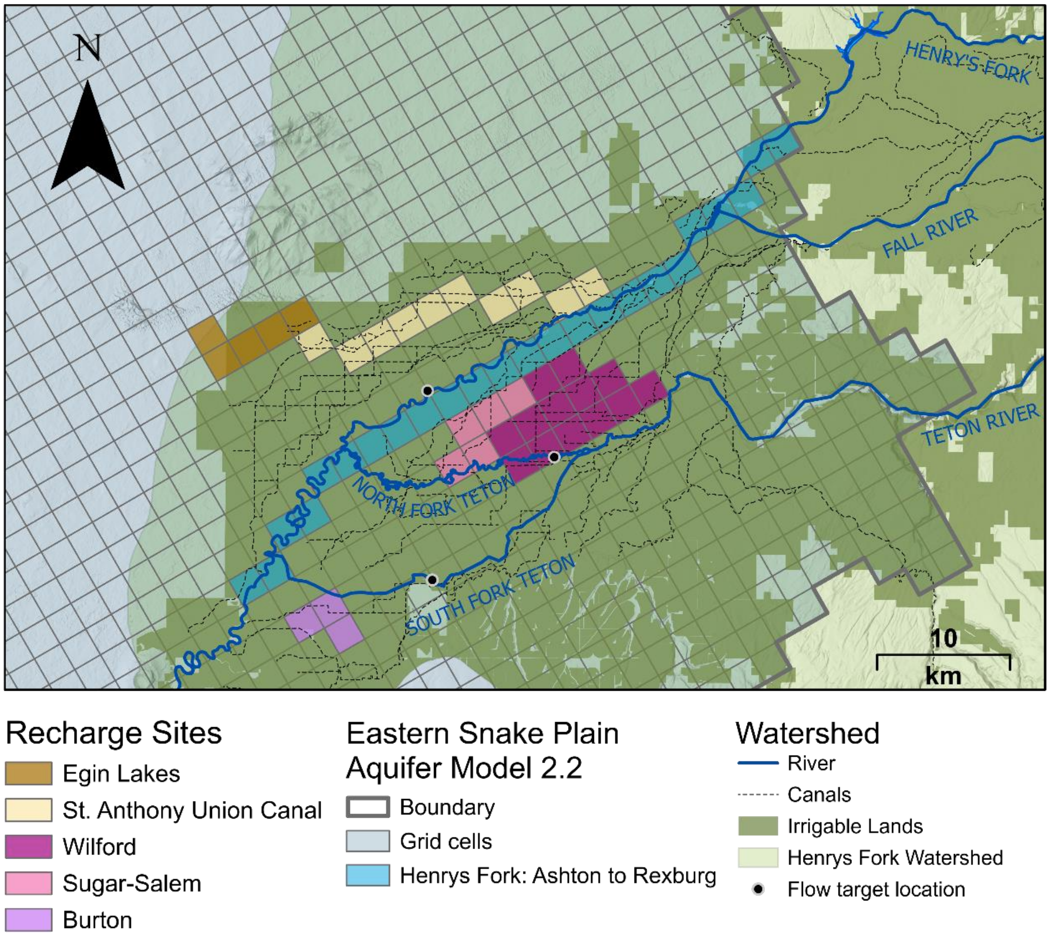

Figure B-1. The five potential Ag-MAR sites relative to the Eastern Snake Plain Aquifer (ESPAM) Version 2.2 grid cells, modeled river reach of interest (target reach), and location of flow targets on the forks of the Teton River and the mainstem Henrys Fork.

2. Results

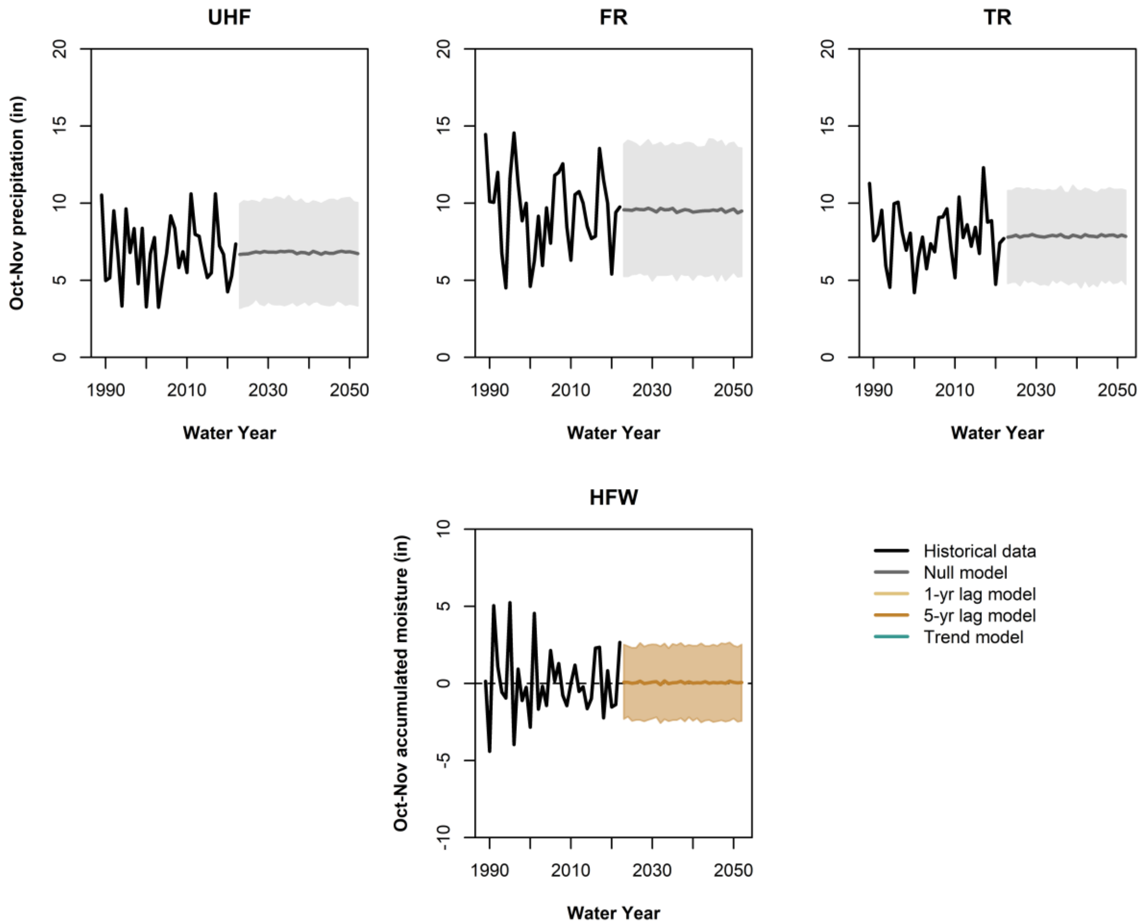

Figure B-2. For the autumn season (1 October–30 November), the hydroclimatic predictors projected by subwatershed where UHF is the Upper Henrys Fork, FR is Fall River, TR is Teton River, and HFW is the Henrys Fork Watershed. We depict the historical data from water years 1989–2022 and the 30-year simulated projections (2023–2052). For the projections, we depict the mean prediction within a given year. The associated shaded region is bounded by the 5th and 95th percentiles.

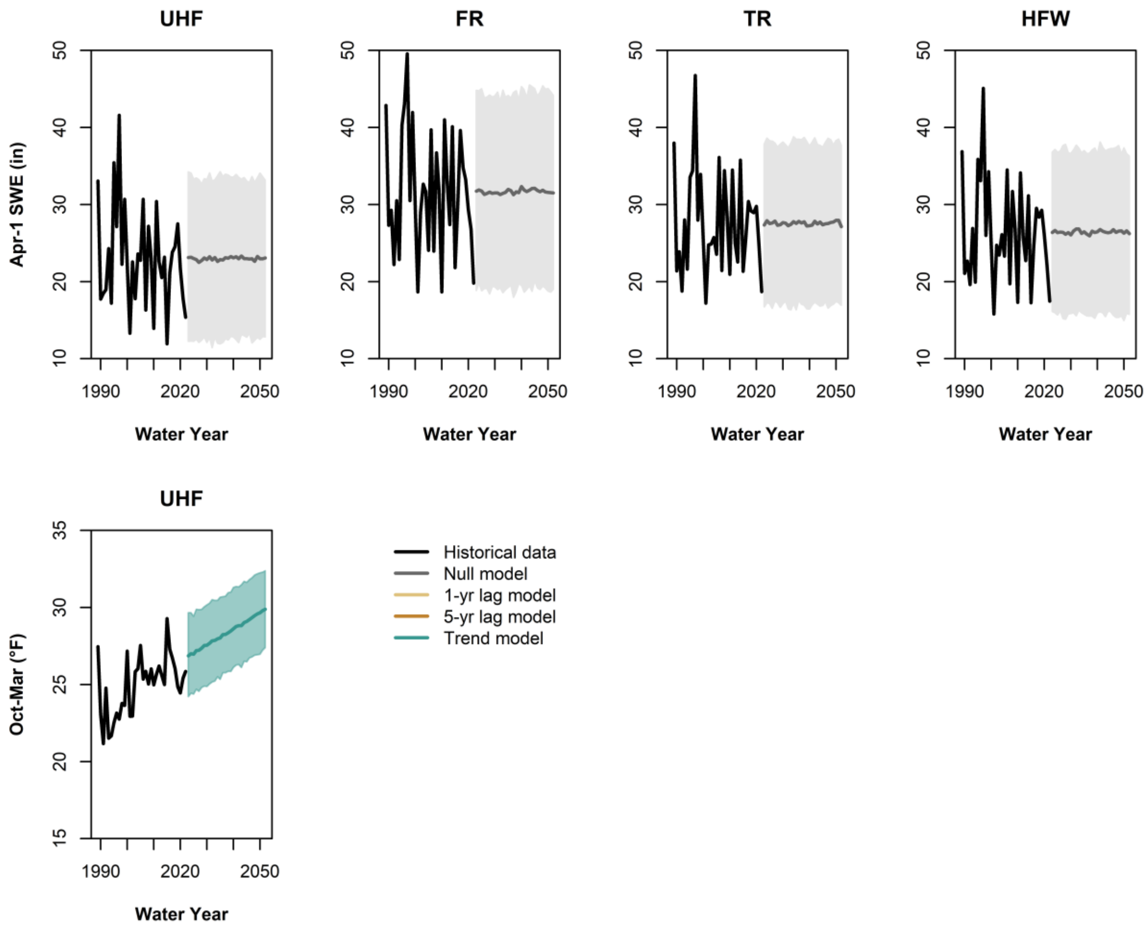

Figure B-3. For the winter season (1 October–31 March), the hydroclimatic predictors projected by subwatershed where UHF is the Upper Henrys Fork, FR is Fall River, TR is Teton River, and HFW is the Henrys Fork Watershed. We depict the historical data from water years 1989–2022 and the 30-year simulated projections (2023–2052). For the projections, we depict the mean prediction within a given year. The associated shaded region is bounded by the 5th and 95th percentiles.

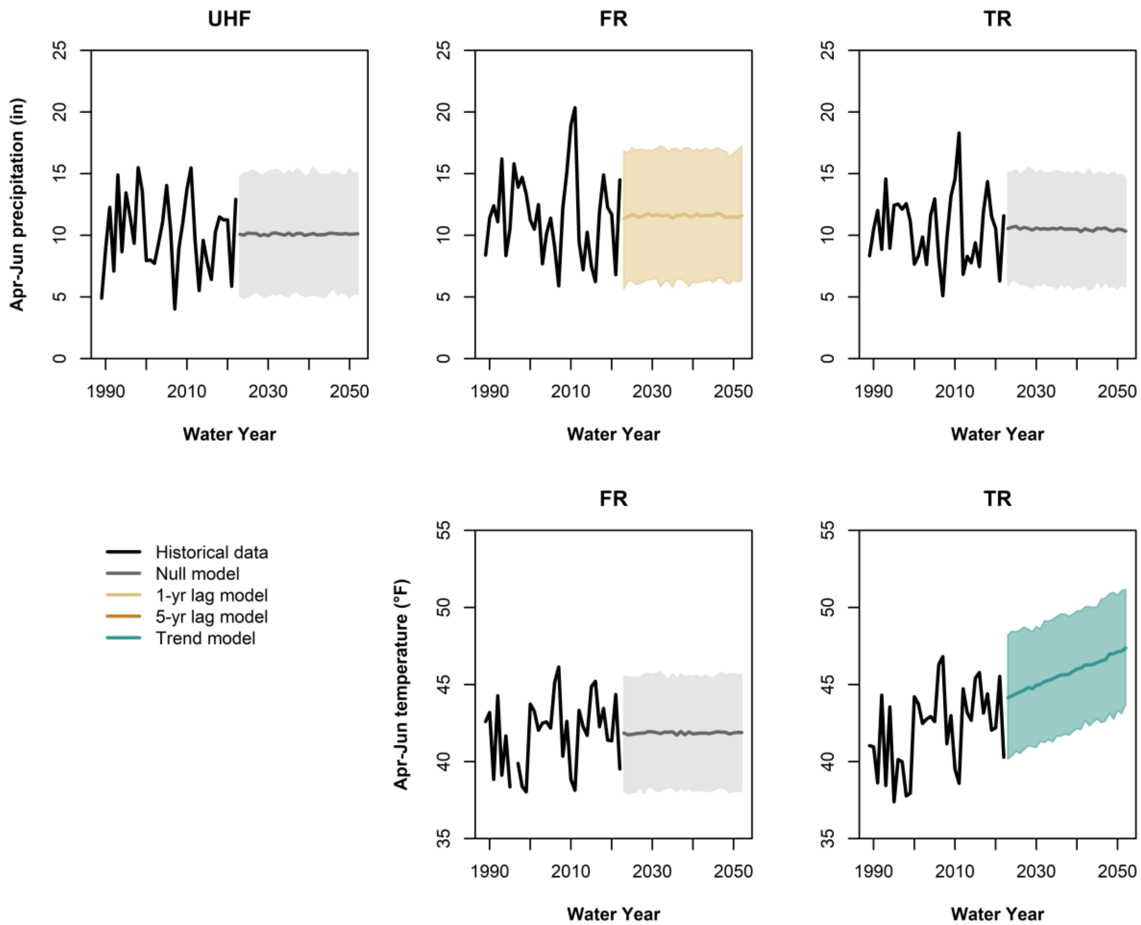

Figure B-4. For the spring season (1 April–30 June), the hydroclimatic predictors projected by subwatershed where UHF is the Upper Henrys Fork, FR is Fall River, and TR is Teton River. We depict the historical data from water years 1989–2022 and the 30-year simulated projections (2023–2052). For the projections, we depict the mean prediction within a given year. The associated shaded region is bounded by the 5th and 95th percentiles.

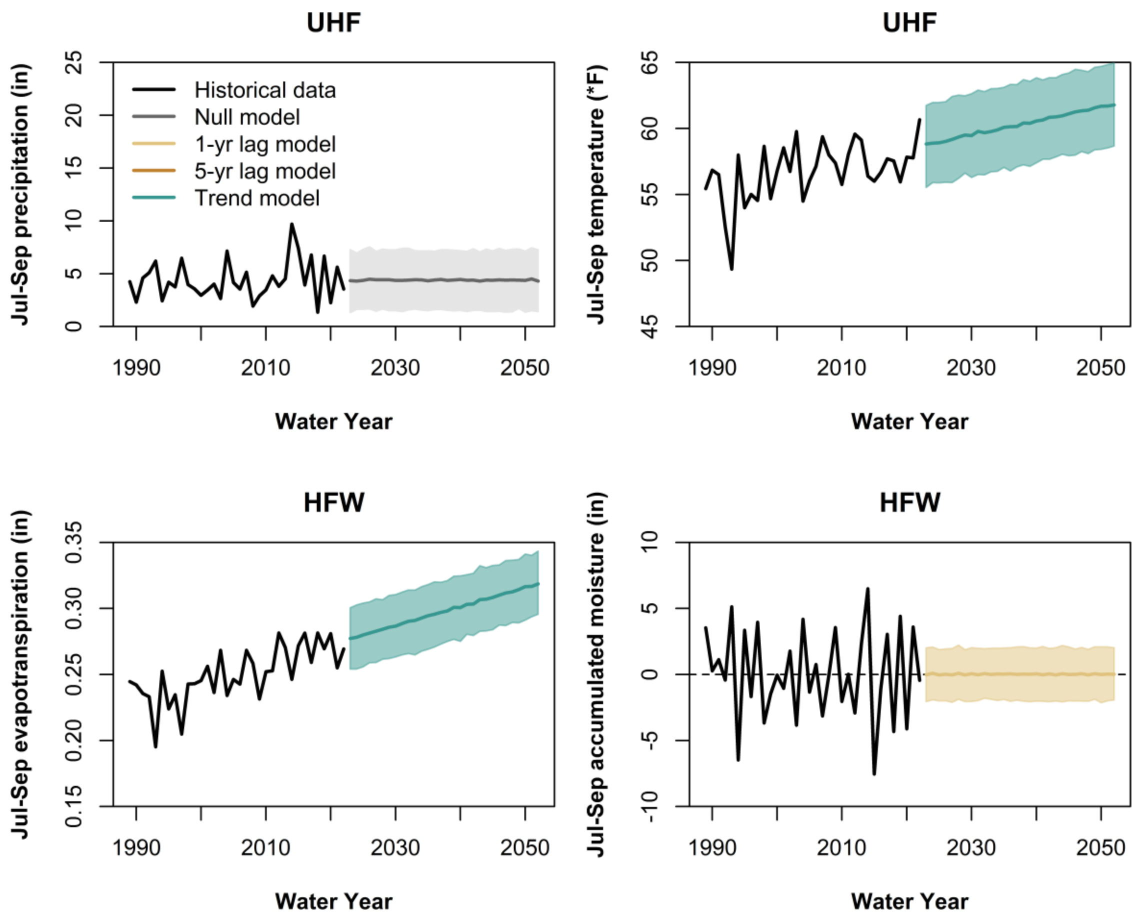

Figure B-5. For the summer season (1 July – 30 September), the hydroclimatic predictors projected by subwatershed where UHF is the Upper Henrys Fork and HFW is the Henrys Fork Watershed. We depict the historical data from water years 1989–2022 and the 30-year simulated projections (2023–2052). For the projections, we depict the mean prediction within a given year. The associated shaded region is bounded by the 5th and 95th percentiles.

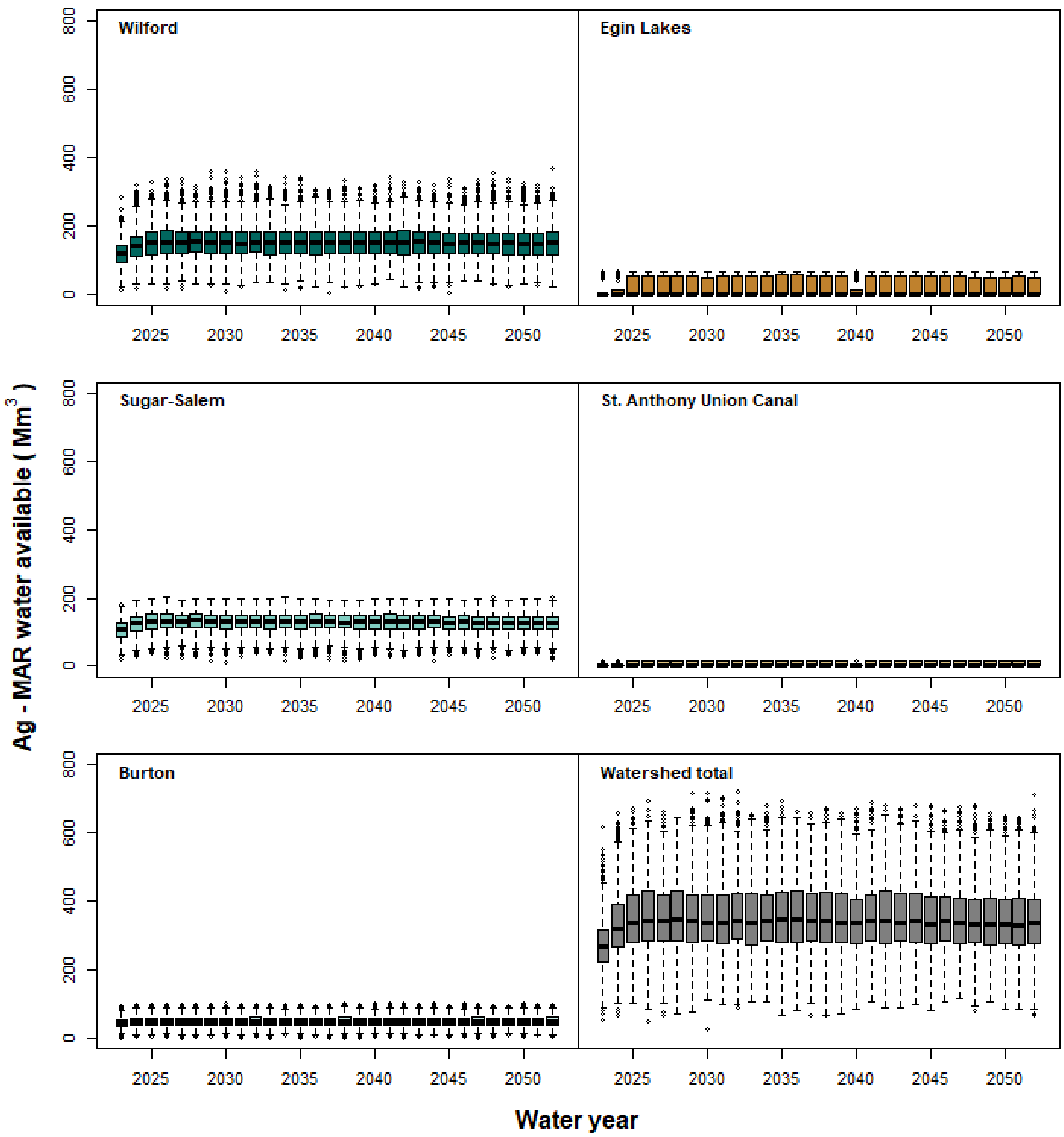

Figure B-6. Water available for Ag-MAR at each recharge site and the watershed total for the 30-year time series across all simulations.

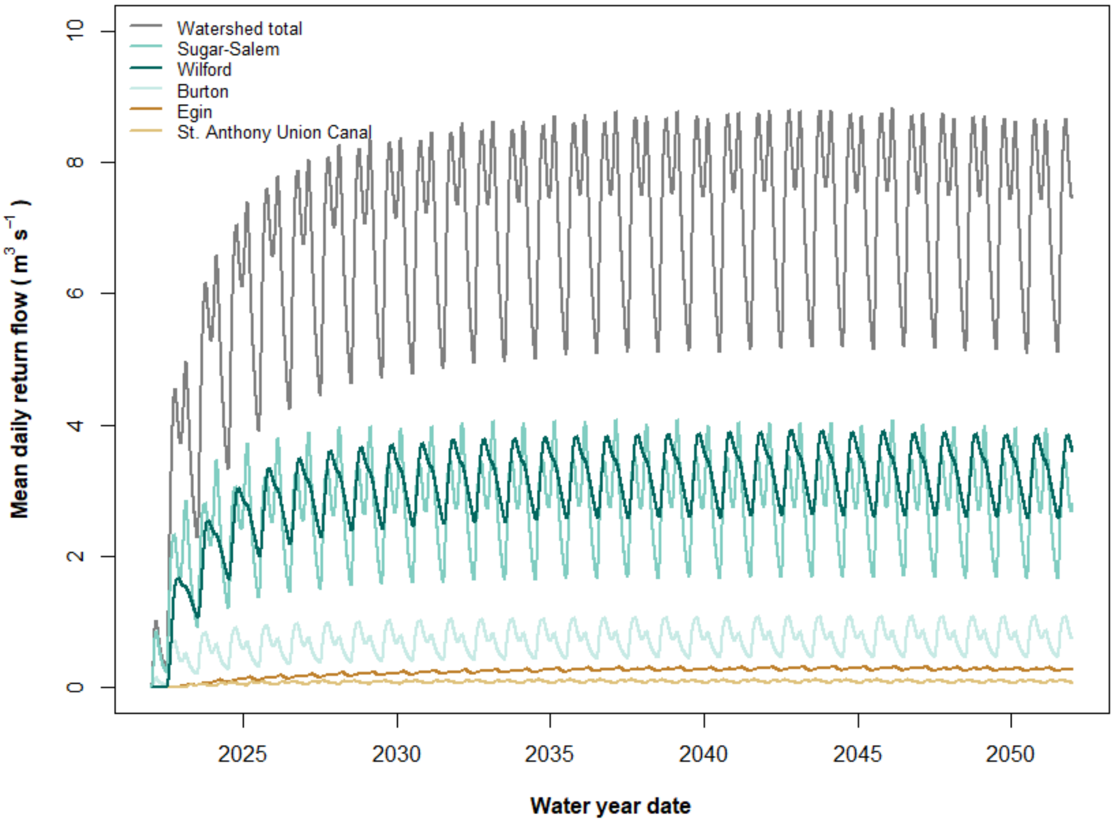

Figure B-7. For water years 2023–2052, mean daily return flow for all Ag-MAR sites individually and in sum across all simulations.

APPENDIX C: SUPPLEMENTAL INFORMATION FOR CHAPTER 4

1. Background

1.1 Study Area

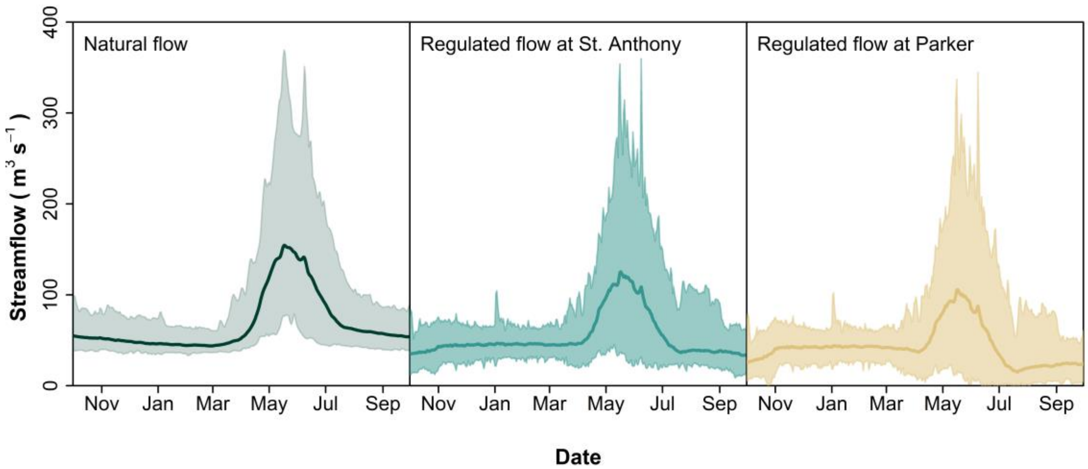

Figure C-1. The hydrographs for mean daily natural flow and mean regulated flow at St. Anthony and Parker for water years 1978–2021 (October–September), using gaged streamflow and canal diversion data with a nominal streamflow calculation. The surrounding polygons demonstrate the total range of flow within water years 1978–2021.

2. Methods

2.1 Habitat types: definitions and mapping

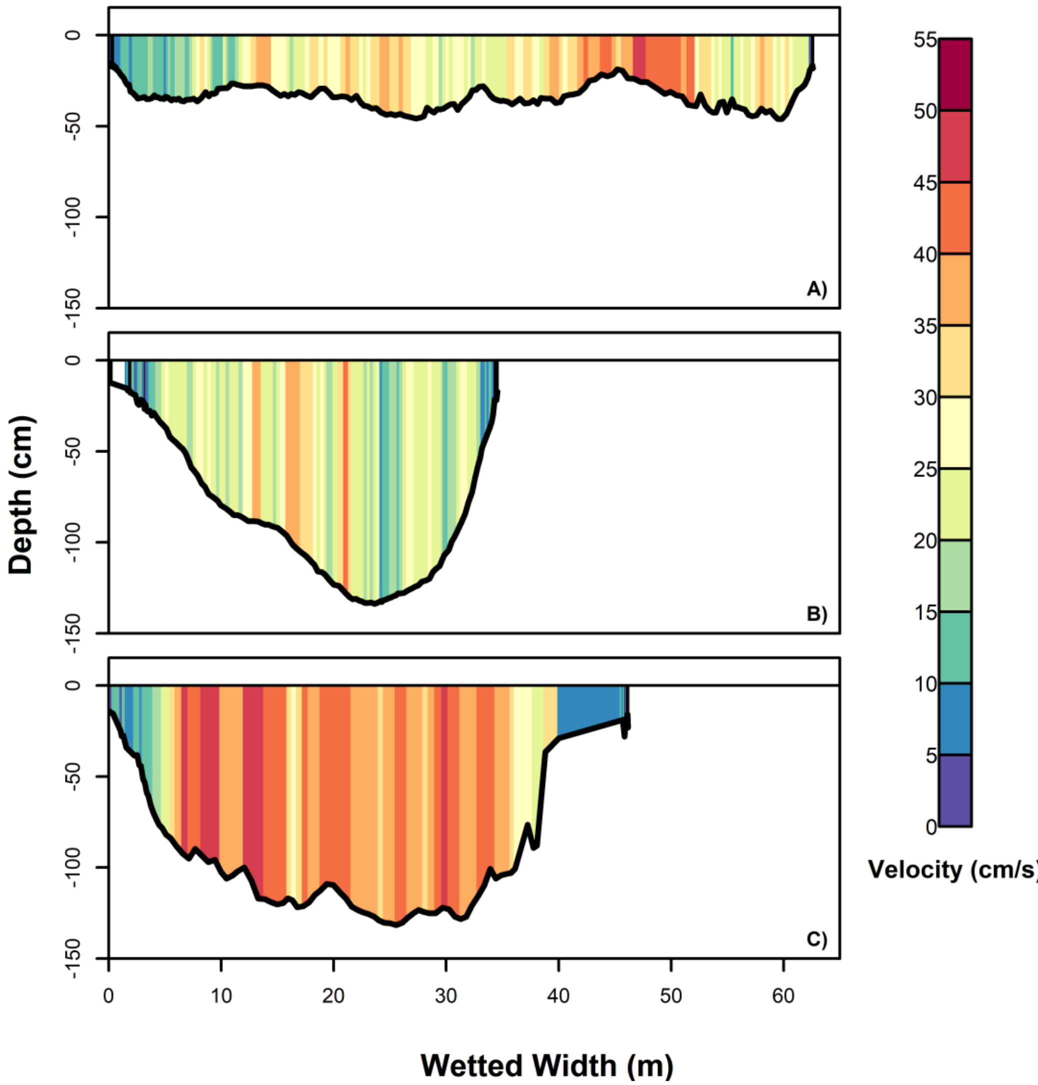

Figure C-2. Example cross-sections for a given habitat type with column-averaged velocity (cm/s). Panel A is a measurement taken at a single-channel site with a rectangular geomorphic unit ("SR"). This location demonstrates near-uniform depth with channel flow distributed uniformly across the section. Panel B is a measurement taken at a single-channel site with a point bar deposit on the left bank (not shown) and a deep thalweg along the right bank ("SB"). Panel C is a measurement taken at a single-channel pool ("SP"). Panels A and B are from sites in the lower subreach; Panel C is from a site in the upper subreach. Panel A represents a measurement taken on 2020-08-06 at a nominal streamflow of 12.1 m^3 s^{-1}. Panel B represents a

measurement taken on 2020-09-15 at a nominal streamflow of 14.2 m^3 s^{-1}. Panel C represents a measurement taken on 2021-08-18 at a nominal streamflow of 19.7 m^3 s^{-1}. There is an intervening diversion (CF) between Panel C and A-B that can take up to 7.9 m^3 s^{-1} from the river. Therefore, the flows shown in in this figure are representative of the irrigation season.

We scaled habitat unit area values by width, using statistical relationships between streamflow and width for each habitat type. More specifically, we normalized width as a proportion of bankfull width, used beta regression with logit link to ensure the fitted curve remained between 0 and 1 (Cribari-Neto & Zeileis, 2010). If the width proportionality increased with bankfull nominal streamflow, we tested two models: width as a function of flow and log(flow). We chose the best model according to AIC (R Core Team, 2022). If the width proportionality remained roughly constant or appeared to be decreasing (indicating a sampling error), we treated width as constant with flow.

2.2 Habitat suitability

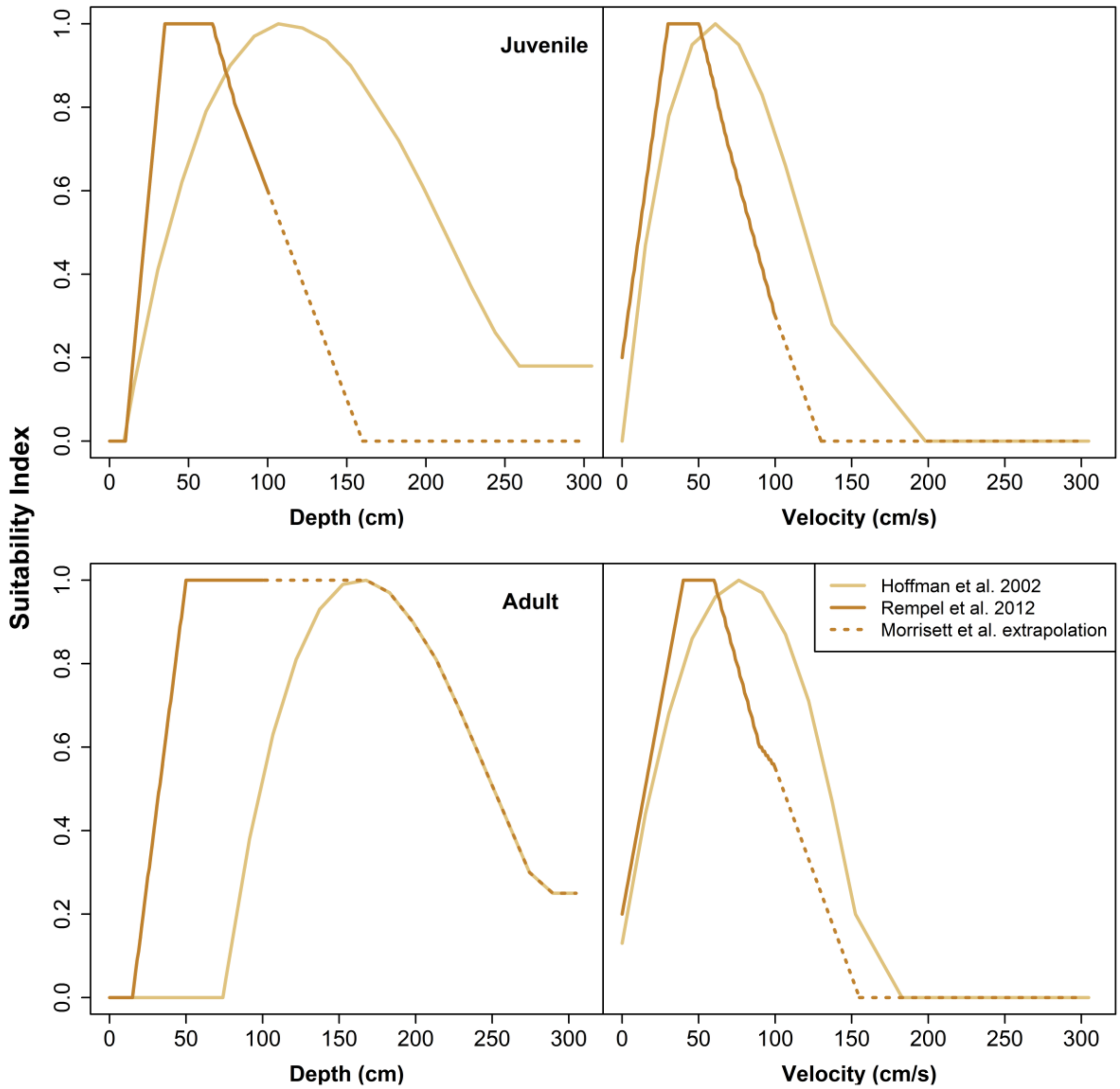

Figure C-3. The suitability curves for Mountain Whitefish used in this study.

We combined suitability criteria from two publications for Mountain Whitefish—Rempel et al. (2012) and Hoffman et al. (2002). The suitability curves for Rempel et al. 2012 extend to 100 cm depth and 100 cm s^{-1} velocity, whereas the suitability curves for Hoffman et al. 2002 exceed 300 cm and 300 cm s^{-1}. Although our study system is better represented by Rempel et al. 2012, some of our data exceed their curve. To compensate, we extrapolated the Rempel et al. 2012 curves beyond the given depth or velocity range until the curve either 1) intersected zero or 2) intersected the curve from Hoffman et al. 2002. If the extrapolation intersected the Hoffman et al. 2002 curve, we used the latter curve from that intersection onward.

2.3 Habitat-streamflow data and relationships

All hydraulic data collected with the Acoustic Doppler Current Profiler were processed in the WinRiverII software (Teledyne RD Instruments, 2018). We exported ASCII files for each pass within a site measurement with data for ensemble number (i.e. column of water), distance traveled (from bank), average beam depth, and velocity (BT Earth Magnitude) for analysis and visualization in R (R Core Team, 2022).

Because each site measurement at a given flow consists of ≥3 passes and each pass consists of dozens of ensembles (water column measurements across the channel), we used a series of averaging to compute a single composite suitability for a given habitat type at a given flow ($suitability_{i,Q,s,l}$):

1. Within a given pass for a given site measurement, computed composite suitability of depth and velocity for each water column ensemble within the cross-section.
2. Averaged all column composite suitabilities to compute a single suitability for the pass.
3. Averaged all passes to compute a single composite suitability for that site at a given full-channel nominal streamflow.

For each relevant habitat type, species, and life stage combination, we fit statistical relationships for habitat suitability and flow using the following systematic model selection:

1. If suitability increases with streamflow, assess two models: suitability as a function of flow and log(flow). Select the best model according to AIC.
2. If suitability decreases with increasing streamflow, assess a quadratic fit.
3. If suitability is roughly constant or decreasing, assess three models: suitability as a function of flow, log(flow), and a quadratic fit. Select the best model according to AIC. If the model produces suitability values between 0 and 1 (inclusive) within the irrigation-season flow range, accept the model. If the suitability values are outside 0 and 1 beyond the mean irrigation-season flow range ($\geq$85 $m^3 s^{-1}$), assign suitability to 0 for negative values and 1 for values >1.

In assessments 1 and 2, we used beta regression with logit link to ensure the fitted curve remained between 0 and 1. There was one habitat type, species, and life stage combination (Spawning Brown Trout at BNR) that required a mixed effects model given the significant difference between the two sites sampled. Here, we fit a mixed effects model for suitability as a function of log(flow) accounting for site differences using the glmr() function in R (Gelman et al., 2022).

To calculate travel time and river reach gains, we retrieved streamflow data for St. Anthony from the U.S. Geological Survey monitoring location 13050500. We retrieved 15-m and daily data for the gaged diversions from the U.S. Bureau of Reclamation’s Hydromet website for the Columbia-Pacific Northwest Region. Diversions included Egin

Canal (EGCI), St. Anthony Union Canal Feeder (AFCI), Independent Canal (INCI), and Consolidated Farmers Canal (CFCI); acronyms specific to database.

Nominal streamflow within the study reach for habitat measurements was calculated as:

$$Q_{location,t_1} = Q_{StAnthony,t_2} - Q_{AFCI,t_2} - Q_{EGCI,t_2} - Q_{INCI,t_2} - Q_{CFCI,t_3} * i$$

where $t_2 = t_1$ and $i = 0$ for location = Trestle and all rectangular and bend sites in the braided reach above Trestle; $t_2 = t_1 - 1.25h$ and $i = 1$ for location = pools in braided reach below Trestle; $t_2 = t_1 - 2.06h$ and $i = 1$ for location = SR; $t_2 = t_1 - 2.92h$ and $i = 1$ for location = ST; $t_2 = t_1 - 3.5h$ and $i = 1$ for location = Parker. Here, h is time in hours. The travel-time coefficients were determined by tracking large reductions in river stage during the reservoir drawdown period at the St. Anthony gage downstream through each of the four canal gates. This flow reduction occurred when a hydroelectric facility upstream of the management reach tripped and temporarily reduced outflow to ~12 m^3s^{-1} (before canal diversions). We used these travel time calculations when determining nominal streamflow at the time of individual habitat measurements taken during the reservoir drawdown period in years 2019–2021. Travel time through the reach likely changes during other times of the year when streamflow rebounds, but we use these travel-time estimates during periods applicable to the study when flows are relatively low and within a narrow range.

At the daily scale, nominal streamflow calculations are calculated as:

$$Q_{Parker} = Q_{StAnthony} - Q_{AFCI} - Q_{EGCI} - Q_{INCI} - Q_{CFCI}$$

using daily values reported by the relevant agency. Travel time through the management reach is 3.5-h—less than the 24-h day and thus irrelevant at this scale. As a result, we use daily nominal streamflow calculations when computing daily WUA for water years 1978–2021.

Historic nominal daily flow calculations at Parker—measured streamflow at the USGS St. Anthony gage minus the four downstream diversions—sometimes result in negative streamflow (Figure A-1). These negative values occurred on 28 days during our study period 1978–2021 and ranged as low as -3.9 $m^3 s^{-1}$. This is unrealistic, both because surface flow cannot be lower than zero and because the Consolidated Farmers (CF) canal diversion structure cannot physically withdraw the entire flow of the river. Due to placement of the structure in the channel, at least 0.57 $m^3 s^{-1}$ flows past the CF diversion, no matter how high diversion is relative to streamflow at the point of diversion. Thus, mainstem streamflow will always be ≥0.57 $m^3 s^{-1}$.

Additionally, previous work has identified our study reach as dynamic with groundwater exchange between the river and the Eastern Snake Plain Aquifer (Van Kirk et al., 2020). To account for these reach gains and losses within our study reach, we adjusted the watershed-wide reach gains modeled on the daily scale by Van Kirk (unpublished) using the flow measurements that accompany our habitat measurements. We measured flow above the CF diversion, at the bottom of the upper subreach, June 9–September 30 in water years 2019–2021 ($n = 58$). We computed reach gains for this site by subtracting nominal flow, accounting for travel time, from our measured flow. We created a summer-season reach gain curve for the site using a quadratic mixed effects model with observations grouped by water year, fit to maximize the log-likelihood, and adjusted for autocorrelation with the lme4 package in R (Bates et al., 2015). We used AIC to compare quadratic, linear, and null models (R Core Team, 2022). The equation for the modeled local reach gain for the site during the sampling period was:

$$Q_{StudyAreaReachGain} = -62.6 + 0.5x - 0.0008x^2$$

where x is water year day, with $x = 1$ for October 1 and $x = 365$ for September 30, limited to the time period June 9–September 30 (Figure C-4).

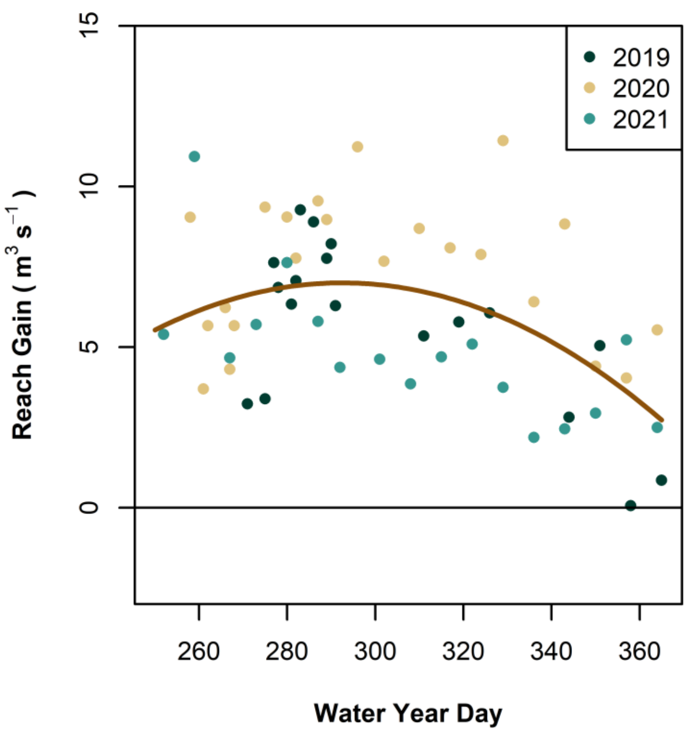

Figure C-4. Summer-season reach gain curve for the site above the CF diversion, at the bottom of the upper subreach. The points depict reach gains computed for the site by subtracting nominal flow, accounting for travel time, from our measured flow in 2019–2021. A water year is October 1–September 30 and we limited our model to the time period June 9–September 30 (water year days 252–365). Points below the zero line indicate reach losses.

We computed mean daily watershed-wide reach gains for water years 2019–2021, subtracted modeled local reach gains for our sample period, and averaged the output to compute a single sampling period adjustment:

$$Q_{StudyAreaAdjustment} = \overline{\overline{Q_{WatershedGain}} - Q_{StudyAreaReachGain}}$$

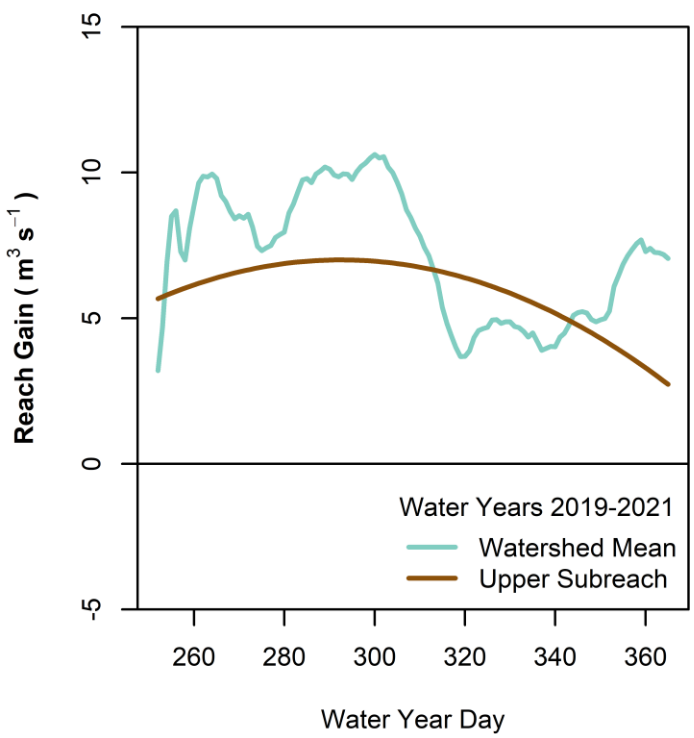

Figure C-5. The watershed-wide mean daily reach gains for water years 2019–2021 compared to the reach gain curve created for the study reach.

On average, our flow measurements above the CF diversion (at the bottom of the upper subreach) were 1.27 $m^3\ s^{-1}$ less than daily watershed-wide reach gains. We applied this adjustment when calculating daily streamflow at Parker for all years within our study (1978–2021):

$$Q_{Parker_{adjusted}} = Q_{Parker_{nominal}} + Q_{WatershedGain} + Q_{StudyAreaAdjustment}$$

Therefore, given the diversion constraints at CF, $Q_{Parker_{adjusted}}$ was computed as the maximum of 0.57 $m^3 s^{-1}$ and the above equation.

2.4 Sensitivity analysis

Sensitivity analyses identify how model output changes with variations in model input. Evaluating model sensitivity is important to understanding how uncertainties in model input impact model accuracy (Booker & Dunbar, 2004; Moriasi et al., 2007). Our WUA model results may be sensitive to the suitability criteria, hydraulic measurements, habitat type classification, and the flow used in WUA calculations. We acknowledge that site-specific curves are best for WUA modeling (Vismara et al., 2001), but collecting the data to develop such curves was beyond our resources. We are confident in our hydraulic measurements and habitat type classification.

Thus, we evaluated the sensitivity of our total-reach WUA calculation to our reach-gain flow adjustment. We compared the mean drawdown-period WUA, minimum 7-day moving average WUA, and the coefficient of variation to those calculated using nominal flow.

We computed the root mean squared error-observations standard deviation ratio (RSR—equation 2; Moriasi et al., 2007) for each case, between flows of 0 to 28.3 $m^3 s^{-1}$ (the pre-2020 low-flow target) to understand how sensitive the WUA curves are to changes in the area classified as a given habitat type.

$$RSR = \frac{RMSE}{STDEV_{obs}} = \sqrt{\frac{\sum_i^n \left(WUA_i^{orig} - WUA_i^{sim}\right)^2}{\sum_i^n \left(WUA_i^{orig} - WUA^{mean}\right)^2}}$$

where n = 1001, WUA_i^{orig} = the WUA calculation using reach-gain adjusted streamflow, and WUA_i^{sim} = the WUA calculation using nominal streamflow, and WUA^{mean} = the mean of the original WUA curve from 0 to 28.3 $m^3\ s^{-1}$. We used RSR performance ratings from Moriasi et al. (2007) to classify the sensitivity of each case.

3. Results

3.1 Comparison of results using nominal versus adjusted streamflow

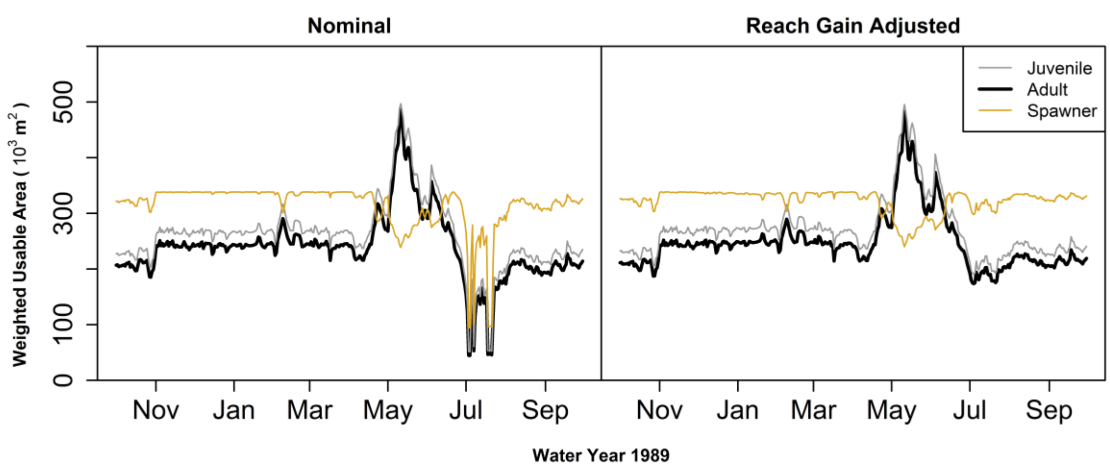

Figure C-6. For water year 1989, a comparison of daily Weighted Usable Area for Brown Trout using nominal streamflow versus a daily streamflow that has been adjusted for local reach gains.

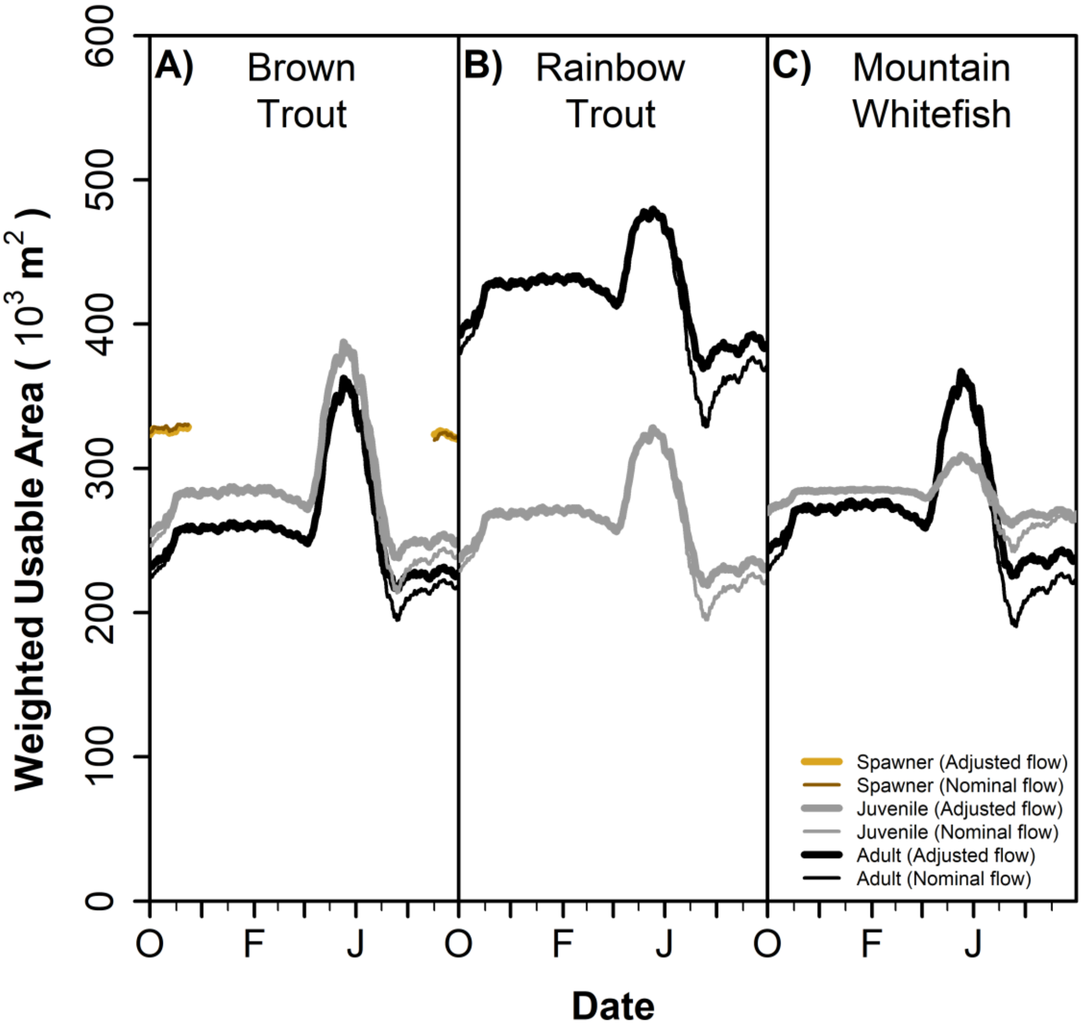

Figure C-7. Mean Weighted Usable Area (WUA) by species, using nominal or reach-gain adjusted streamflow, for water years 1978–2021.

Regardless of the flow-type input, WUA over the water year largely reflected the annual hydrograph (Figure C-7). Habitat for all species was lowest in July—and this was exaggerated using nominal streamflow to calculate WUA.

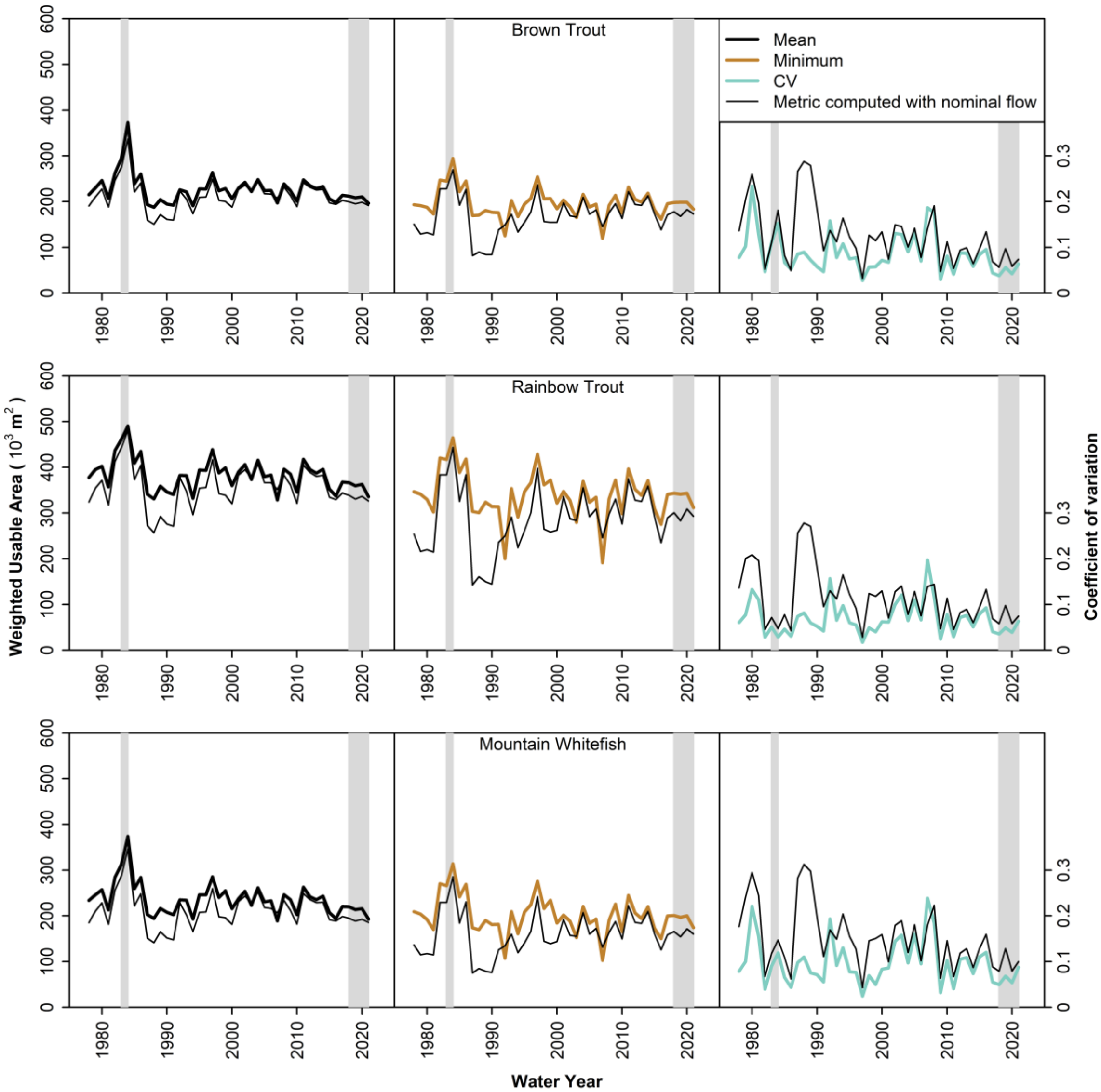

Figure C-8. Mean Weighted Usable Area (WUA), minimum 7-day moving average WUA, and coefficient of variation (CV) in WUA for adult life stages across all species during the Island Park Reservoir drawdown period in water years 1978 through 2021. The thin black line is the given metric in each panel where WUA is calculated using nominal streamflow, rather than reach-gain adjusted streamflow. Two time periods are shaded within each panel: 1983–1984 when the reservoir was drawn down excessively for dam maintenance and 2018–2021 when the Committee changed irrigation-season management to focus on minimizing reservoir drawdown.

Overall, WUA calculated using nominal and reach-gain adjusted streamflow reflect similar patterns. However, the reach-gain adjustment appears to have more impact in years prior to 2001 (Figure C-8). This reflects documented changes to reach gains in the watershed, where reach gains were greater and decreasing during 1978–2001 and have since leveled out at a lower value (Sukow, 2021).

Table C-1. Two-sided t-tests assuming unequal variance to compare mean WUA, mean WUA adjusted for annual natural streamflow, and coefficient of variation during the drawdown period for Island Park Reservoir between the two management regimes (1978–2017 and 2018–2021), for adult species where BRN = Brown Trout, MWF = Mountain Whitefish, and RBT = Rainbow Trout. The t-test outputs are shown for metrics where WUA was calculated using nominal vs. reach-gain adjusted streamflow. With Bonferroni's correction to account for three tests (each species) within each metric, significance at a family-wide error rate of 0.05 requires individual significance at 0.017.

Metric	Species	Flow used	*t*	df	*p*
Mean WUA	BRN	Nominal	2.6	36.9	**0.013**
		Reach-gain adjusted	3.3	10.0	**0.009**
	RBT	Nominal	2.5	37.3	0.018
		Reach-gain adjusted	2.9	6.6	0.03
	MWF	Nominal	3.1	38.5	**0.004**
		Reach-gain adjusted	3.1	6.5	0.02
Mean WUA adjusted for natural streamflow	BRN	Nominal	0.2	4.7	0.8
		Reach-gain adjusted	0.5	4.7	0.6
	RBT	Nominal	0.2	4.5	0.9
		Reach-gain adjusted	0.4	4.7	0.7
	MWF	Nominal	0.6	5.2	0.5
		Reach-gain adjusted	1.2	5.6	0.3
Coefficient of variation	BRN	Nominal	4.1	13.9	**0.001**
		Reach-gain adjusted	4.1	15.5	**0.008**
	RBT	Nominal	3.6	12.5	**0.003**
		Reach-gain adjusted	3.0	10.6	**0.012**
	MWF	Nominal	3.7	10.3	**0.004**
		Reach-gain adjusted	3.0	10.4	**0.013**

WUA calculated using nominal streamflow echoed results for all adult species for both mean annual drawdown-period WUA and mean annual drawdown-period WUA accounting for total annual natural streamflow using the reach-gain adjusted streamflow (Table C-1). However, the flow used to calculate WUA did create differences in conclusions regarding coefficient of variation. When using nominal streamflow, within-year coefficient of variation was significant at an alpha of 0.05 for all species. When using reach-gain adjusted streamflow, the metric was only significant for adult Brown Trout.

3.2 Sensitivity analysis

Our sensitivity analysis demonstrated that our WUA model output is highly sensitive to nominal versus reach-gain adjusted streamflow input. Only mean drawdown-period WUA for adult Brown Trout remained Satisfactory across flow inputs. Although the values may be sensitive in terms of magnitude, the overall conclusions across management regimes largely remain (Table C-2).

Table C-2. Sensitivity analysis results by drawdown-period WUA metric and adult species, where BRN = Brown Trout, MWF = Mountain Whitefish, and RBT = Rainbow Trout. Performance statistic "root mean squared error-observations standard deviation ratio" (RSR) shown with performance rating from (Moriasi et al., 2007).

Model output metric	Species	RSR	Performance Rating
Mean WUA	BRN	0.56	Good
	MWF	0.81	Unsatisfactory
	RBT	0.77	Unsatisfactory
Minimum 7-day moving average WUA	BRN	0.93	Unsatisfactory
	MWF	1.14	Unsatisfactory
	RBT	1.08	Unsatisfactory
Coefficient of variation	BRN	1.02	Unsatisfactory
	MWF	1.14	Unsatisfactory
	RBT	1.16	Unsatisfactory

3.3 Additional results with adjusted streamflow

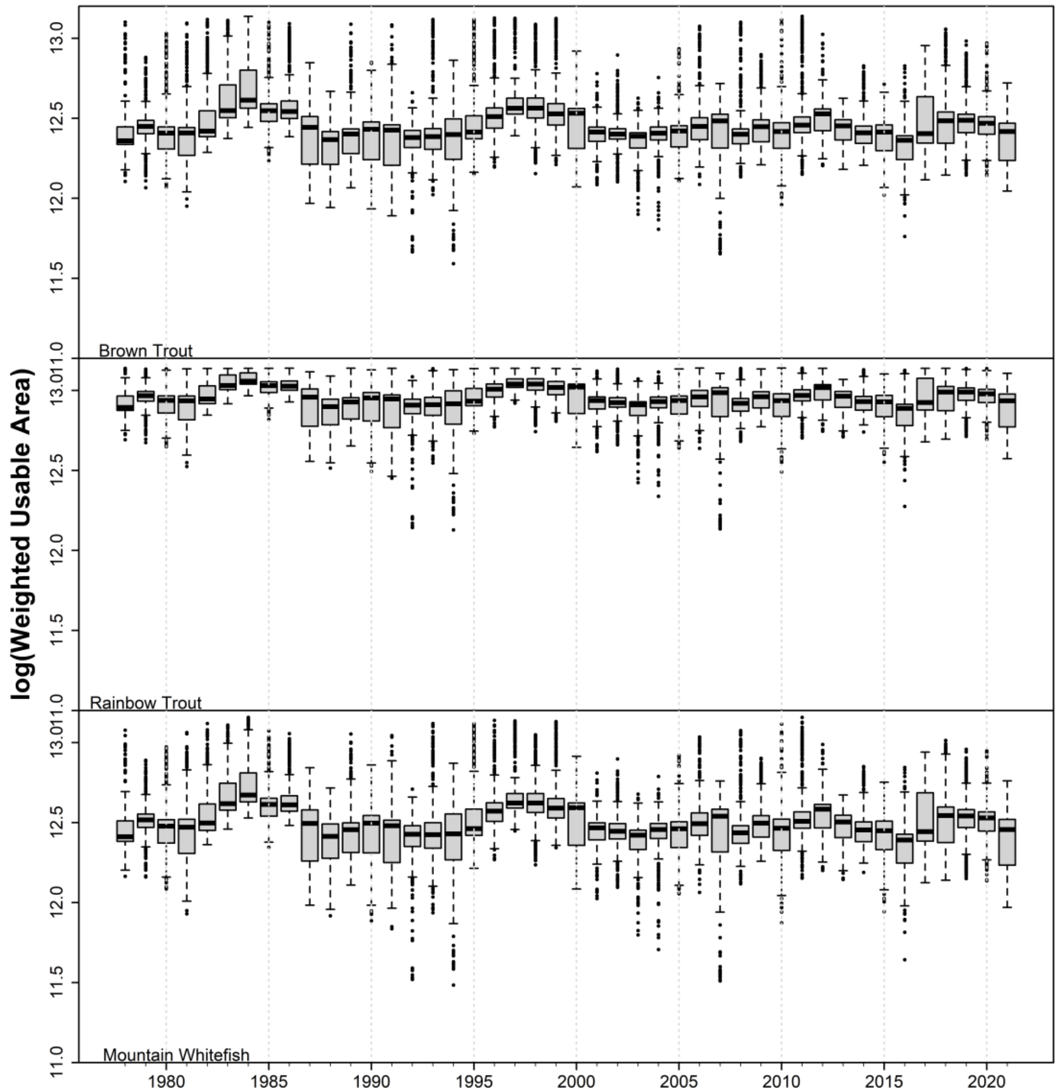

Figure C-9. Box plots of log(Weighted Usable Area) for the three study species for the reservoir drawdown periods in water years 1978–2021.

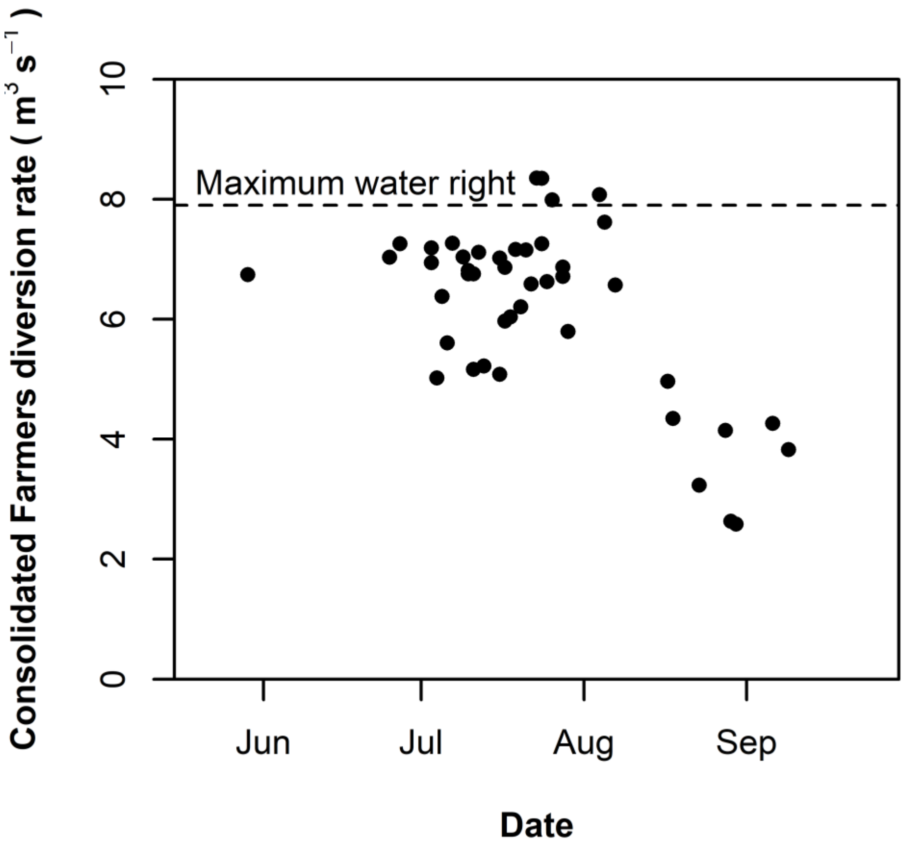

Figure C-10. The 7-day moving average diversion rate at the Consolidated Farmers (CF) canal relative to the date of the minimum 7-day moving average WUA for the study reach. The CF canal can divert more than its water right when diverting on behalf of other entities, in addition their own water right.

4. References

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APPENDIX D: CO-AUTHOR AUTHORIZATION FOR INCLUSION OF WORK

July 25, 2023

Dear Robert Van Kirk,

I am in the process of preparing my dissertation in the Department of Watershed Sciences at Utah State University.

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Thank you for your cooperation,
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christina@henrysfork.org

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(Morrisett et al., 2023); Morrisett, C. N., Van Kirk, R. W., & Null, S. E. (2023). Assessing downstream aquatic habitat availability relative to headwater reservoir management in the Henrys Fork Snake River. *River Research and Applications*.

(Morrisett et al., In Review); Morrisett, C. N., Van Kirk, R. W., Bernier, L. O., Holt, A. L., Perel, C. B., & Null, S. E. (In Review). The irrigation efficiency trap: Rational farm-scale decisions can lead to poor hydrologic outcomes at the basin scale.

(Morrisett et al., In Prep); Morrisett, C. N., Van Kirk, R. W., & Null, S. E. (In Prep). Can aquifer recharge recover return flows under prior appropriation in a warming climate?

Signed: Robert W. Va [illegible]

July 25, 2023

Dear Sarah Null,

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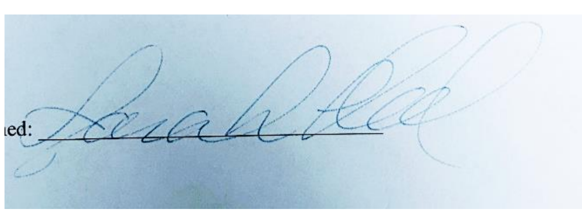
(Morrisett et al., In Review); Morrisett, C. N., Van Kirk, R. W., Bernier, L. O., Holt, A. L., Perel, C. B., & Null, S. E. (In Review). The irrigation efficiency trap: Rational farm-scale decisions can lead to poor hydrologic outcomes at the basin scale.

(Morrisett et al., In Prep); Morrisett, C. N., Van Kirk, R. W., & Null, S. E. (In Prep). Can aquifer recharge recover return flows under prior appropriation in a warming climate?

Signed: 

July 25, 2023

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I am in the process of preparing my dissertation in the Department of Watershed Sciences at Utah State University.

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Signed: London Bernier

July 25, 2023

Dear Andrea Holt,

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Signed: Andrea Holt

July 25, 2023

Dear Chloe Perel,

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Signed: ______________________________________

CURRICULUM VITAE

Christina N. Morrisett, Ph.D.

Interdisciplinary water resource management scientist (rivers)

EDUCATION:

2023 **Ph.D. in Watershed Sciences**, specialization in Climate Adaptation Science
Utah State University, Department of Watershed Sciences; Advisor: Dr. Sarah Null
Committee: Dr. Rob Van Kirk, Dr. Courtney Flint, Dr. Phaedra Budy, Dr. Patrick Belmont
Dissertation: *Multi-objective water management in Idaho's Henrys Fork watershed: leveraging reservoir operation and groundwater pathways to benefit aquatic habitat*

2018 **M.S. Aquatic and Fishery Sciences**
University of Washington, School of Aquatic and Fishery Sciences
Advisor: Dr. John Skalski
Committee: Dr. Julian Olden, Dr. Thomas Quinn
Thesis: *Assessing the utility of tributary passive integrated transponder (PIT) tag arrays in monitoring Snake River salmonid recovery*

2015 **B.S. Earth Systems (Oceans Track)**
Stanford University, School of Earth, Energy, & Environmental Sciences
Studied abroad with SEA Semester (2013), the Uni. of Queensland (2013), in the Pantanal (2014)

PROFESSIONAL EXPERIENCE:

07/2023–present **Water Resources Consultant**, Henry's Fork Foundation

- Develop stage-discharge curve for new stream gage installation.
- Build multi-stakeholder reservoir optimization model for winter-time fill.
- Model groundwater-surface water interactions for twelve sites of interest in Teton Valley, Idaho to inform aquifer recharge efforts for groundwater return flow restoration.
- Contribute to RShiny App development for several water management tools.

01/2023–present **Climate Ambassador,** American Fisheries Society

- Receive training in narrative structure and climate change communication.
- Develop outreach materials, presentations, and speaking engagements to inform target audiences on the impact of climate change on fish and fisheries.

08/2018–06/2023 **Graduate Research Assistant**, Utah State University (USU) /
Doctoral Research Associate, Henry's Fork Foundation (HFF)

- Facilitated stakeholder meetings to collaboratively develop a watershed-scale optimization model to manage water resources for multiple uses.
- Led three-person field crew and coordinated volunteers to measure streamflow parameters with an ADCP via river float surveys.
- Modeled streamflow-aquatic habitat relationships and recommended a flow target in a reach of interest. Flow target implemented by managers in 2020 has increased reservoir water savings while decreasing daily habitat variability in a downstream management reach.
- Simulated managed aquifer recharge scenarios under climate change to inform feasibility of groundwater return flow restoration.
- Deployed piezometers to monitor groundwater-surface water dynamics.
- Hired/trained four 10-week undergraduate summer interns for assistance with fieldwork and data analysis. Mentor undergraduate summer research project.
- Communicated results to irrigators, recreational anglers, and HFF members in presentations, meetings, field tours, and in written outreach materials.
- Wrote and contributed to 1) peer-reviewed scientific publications and 2) grant proposals for research funding.

01/2021–04/2021 **Lab Instructor**, WATS 4490/6490/6491: Small Watershed Hydrology, USU

- Led weekly labs to develop 15 graduate/undergrad students' comprehensive understanding of the hydrologic cycle and each of its primary components.
- Assisted students with lab assignments. Graded assignments.

09/2020–12/2020 **Grader**, GEOG 1000: Physical Geography, Utah State University

- Graded homework and final assignments for 170 undergraduate students.

04/2020–09/2020 **Climate Adaptation Science Intern**, Friends of the Teton River

- Coordinated water quality monitoring efforts for a pilot managed aquifer recharge project with irrigation, agency, university partners, and local landowners. Wrote groundwater quality monitoring plan.
- Collaborated with NGO partners to write a $200,000 Wildlife Conservation Society grant pre-proposal to implement managed aquifer recharge for conservation of ecosystem function in the Upper Snake River basin.

10/2019–10/2020 **Natural Resources Workforce Development Fellow**, Southwest Climate Adaptation Science Center (SWCASC)

- Conducted collaborative, interdisciplinary research with a cohort of seven SWCASC consortium students to study climate adaptation planning to support ecosystems and people in Arizona's Gila River watershed.

05/2019 **Climate Adaptation Science Intern**, Friends of the Teton River

- Collaborated with ag producers, NGOs, and academic researchers to write a $350,000 grant pre-proposal to study water-saving farming practices for the USDA Western Sustainable Agriculture Research and Education program.

08/2018–08/2020 **NSF Climate Adaptation Science Trainee**, Utah State University

- Conducted collaborative, interdisciplinary research studying water availability for cannabis cultivation in northern California.
- Received training in science communication, leadership, project management, risk assessment, and decision-making under uncertainty.

09/2016–07/2018 **Graduate Research Assistant**, Columbia Basin Research, Univ. of Washington

- Statistically assessed tag detection histories of hundreds of thousands of ESA-listed Snake River salmonids to understand how adult dam passage and smolt transportation affect upstream migration success.
- Interacted with federal entities to obtain data and dam operation orientation.
- Developed R scripts to compile and automate the cleaning, aggregation, and analysis of datasets with millions of rows.
- Communicated results and actionable management recommendations through written reports, visual presentations, and collaborative discussions.

09/2015–07/2016 **Research Assistant**, Henry's Fork Foundation

06/2015–08/2015 **Environmental Modeling Intern**, Henry's Fork Foundation

- Modeled natural and managed hydrology, water-rights administration, hydroelectric power operations, and irrigation system management of upper Snake River system.
- Conducted statistical analysis of fish abundance and species composition data.
- Conducted literature review and prepared figures for manuscript submission.
- Managed the Buffalo River fish ladder for the spring 2016 season.
- Authored a comprehensive report on the Buffalo River fish ladder.
- Electro-fished in remote Teton River tributaries for cutthroat trout study.
- Assisted with creel surveys, water quality monitoring, in-river macroinvertebrate sampling, and cattle fencing as needed.
- Participated in various outreach events with elementary and college students, irrigators, outfitters, recreational anglers, and HFF members.

01/2015–09/2015 **Student Consultant**, Comm. Eng. Learn. Prog., Stanford Uni.

- *For the Marine Science Institute:* Developed a summer camp lesson for 2nd-5th graders that introduces the water cycle, discusses human water use, and includes a simulation to explore virtual water in food.
- *For the California Governor's Office and the U.S. Dept. of Defense:* Collected news articles on biodiversity loss, climate disruption, pollution, invasives/diseases, and population change to populate an ArcGIS Story Map.
- *For the Peninsula Open Space Trust:* Created a baseline assessment to determine effectiveness of restoration efforts at Butano Creek. Adapted methods to suit a volunteer monitoring protocol.

04/2015–06/2015 **Research Assistant**, Stanford University

- Coded survey responses in NVivo for Stanford E-IPER Ph.D. candidate on using surfer wave knowledge to predict impacts of sea-level rise.

09/2014–06/2015 **Student Advisor**, Bing Overseas Study Program, Stanford Uni.

- Advertised the Stanford in Australia program; answered student questions.
- Led orientation for the incoming 40-student cohort.

06/2014–07/2014 **College Intern II**, Alaska Department of Fish and Game

- Enumerated salmon passage at Igushik River Counting Towers (remote camp).
- Deployed beach seine to collect specimens for biological sampling.

06/2013–08/2013 **Conservation Science Intern**, Turtle Island Restoration Network

- Participated in local and international advocacy efforts for endangered species via video editing, petition submission, and social media outreach.

05/2013 **Program Participant,** SEA Semester

- Crewed the SSV Robert C. Seamans on a five-week round-trip cruise from Oahu to Palmyra Atoll to Kiribati to Oahu. Aided in oceanographic CTD deployment, plankton counts, and ship navigation.

06/2011–09/2012 **Fish and Wildlife Technician II**, Alaska Dept. of Fish and Game

- Seasonal position. Extracted otoliths from three Pacific salmon species in processing plants and remote spawning locations. Delivered for hatchery mark evaluation.

AWARDS:

2022 **Legacy of Utah State Award**, Utah State University (awarded to 1 of 7 college nominees)

An annual award to recognize and highlight a student who represents the heart and soul of the university.

2022 **Legacy of Utah State Award**, Quinney College of Natural Resources, nominee to Utah State U.

An annual award to recognize and highlight a student who represents the heart and soul of the university.

2021 **Covey Leadership Award**, Stephen R. Covey Center, Huntsman School of Bus., Utah State U.

For students who show exceptional character and leadership, awarded to one student per college per year

2018 **Honorable Mention**, NSF Graduate Research Fellowship Program, Geosciences: Hydrology

Proposal Title: *Can managed aquifer recharge mitigate flow and temperature effects of a warming climate?*

FELLOWSHIPS AND SCHOLARSHIPS:

2019–2020	Natural Resources Workforce Development Fellowship, SWCASC, $5,000
2019–2020	NSF NRT Climate Adaptation Science Fellowship, Utah State U., $34,000
2019	Anchor QEA Scholarship, $3,000

GRANTS:

Funded

Morrisett, C. Open Access Funding Initiative, USU Merrill-Cazier Library. $1,075. *Competitive.*

Van Kirk, R. (PI), H. Blischke, **C. Morrisett**, M. Muradian, and J. Laatsch. Predictive hydrologic modeling and real-time data access to support water resources planning and management in the Henry's Fork Watershed. U.S. Bureau of Reclamation WaterSMART Applied Science Grant. $273,211 awarded to the Henry's Fork Foundation, 2020-2022. *Competitive.*

Van Kirk, R and **C. Morrisett**. Hydrologic and Wetland Assessment of the Lower Henry's Fork and Modeling to Support Multi-stakeholder River Management. Federal Highway Administration, Local Highway Technical Assistance Council, and Fremont County, Idaho. $60,000 awarded to the Henry's Fork Foundation, 2019. *Non-competitive.*

Not Accepted

Henry's Fork Foundation. Facilitating angler adaptation to drought in the Henry's Fork Watershed. Idaho Fish and Game Commission. $23,600 proposed; July 2022-June 2023. *Competitive.*

Van Kirk, R. (PI), **C. Morrisett**, J. McLaren, S.E. Null, and P. Budy. Development and communication of aquatic ecosystem responses to drought in a Yellowstone-region watershed that supports socio-economically important fisheries and recreational resources. NOAA Coping with Drought: Ecological Drought. $250,000 proposed; FY2022-2024. *Competitive.*

Van Kirk. R. (PI), J. Laatsch, **C. Morrisett**, M. Ludington, and A. Lindstedt. Implementing managed aquifer recharge to increase climate resilience of stream, wetland, and riparian ecosystems in the upper Snake River Basin. Pre-proposal submitted to the 2020 Climate Adaptation Fund, Wildlife Conservation Society. $215,520 requested. *Competitive.*

Morrisett, C. (PI), B. Contor, Z. Wolcott-MacCausland, J. Brandt, J. Pierce, M.A. de Graaff, W. Penfold, M. Reid, T. Hill, and S. Wright. A holistic assessment of an integrated crop-livestock cooperative. 2019 pre-proposal submitted to Western Sustainable Agriculture Research and Education (SARE). $350,000 requested. *Competitive.*

PUBLICATIONS:

Peer-reviewed

ORCID, Google Scholar Profile

[8] **Morrisett, C.N.**, R.W. Van Kirk, and S. E. Null. Can managed aquifer recharge recover return flow under prior appropriation in a warming climate? *In prep for submission to Water Resources Research.*

[7] **Morrisett, C.N.**, R.W. Van Kirk, L.O. Bernier, A.L. Holt, C.B. Perel, and S.E. Null. Changes to irrigation practices impact groundwater return flows in the Henry's Fork Snake River. *Accepted with Frontiers in Environmental Science, Special Issue: Women at the Frontier of Freshwater Science 2022-08-08.*

[6] **Morrisett, C.N.**, R.W. Van Kirk, and S.E. Null. 2023. Assessing downstream aquatic habitat availability relative to headwater reservoir management in the Henrys Fork Snake River. *River Research and Applications.*

[5] Null, S.E., A. Farshid, G. Goodrum, C.A. Gray, S. Lohani, **C.N. Morrisett**, L. Prudencio, and R Sor. 2021. A meta-analysis of environmental tradeoffs of hydropower dams in the Se Kong, Se San, and Sre Pok (3S) Rivers of the Lower Mekong Basin. *Water* 13(1):63.

[4] Morgan, B., K. Spangler, J. Stuivenvolt Allen, **C.N. Morrisett**, M. W. Brunson, S.-Y. Wang, N. Huntly. 2021. Water availability for cannabis in northern California: Intersections of climate, policy, and public discourse. *Water* 13(1):5.

[3] Van Kirk, R. W., B. A. Contor, **C.N. Morrisett**, S.E. Null, and A.S. Loibman. 2020. Potential for managed aquifer recharge to enhance fish habitat in a regulated river. *Water* 12(3):673.

[2] **Morrisett, C.**, J. R. Skalski, R B. Kiefer. 2019. Passage route and upstream migration success: A case study of Snake River salmonids ascending Lower Granite Dam. *North American Journal of Fisheries Management* 39(1): 58-68.

[1] Anderson, C., M. Krigbaum, M. Arostegui, M. Feddern, J. Koehn, P. Kuriyama, **C. Morrisett**, C. Akselrud, M. Davis, C. Fiamengo, A. Fuller, Q. Lee, K. McElroy, M. Pons, J. Sanders. 2019. How commercial fishing effort is managed. *Fish and Fisheries* 20(2): 268-285.

Editor-reviewed

[4] Van Kirk, R., B. Contor, **C. Morrisett**, and S. Null. 2019. Potential for managed aquifer recharge to mitigate climate-change effects on fish and wildlife in the Snake River Basin, USA. International Symposium on Managed Aquifer Recharge, Madrid, Spain.

[3] Oldemeyer, B., J. Flinders, **C. Morrisett**, and R. Van Kirk. 2017. Long-term effectiveness of flow management and fish passage on the Henry's Fork Rainbow Trout population. Proceedings of Wild Trout Symposium XII, West Yellowstone, MT.

[2] Laatsch, J., R. Van Kirk, **C. Morrisett**, K. Manishin, and J. DeRito. 2017. Angler perception of fishing experience in a highly technical catch-and-release fishery: How closely does perception align with biological reality? Proceedings of Wild Trout Symposium XII, West Yellowstone, MT.

[1] **Morrisett, C.** Supporting teachers to continue field-trip learning in the classroom. 2015. Environmental Education Research Bulletin. Issue 6.

Technical reports

[5] Eppehimer, D., E. Fard, L. Jennings, J. Kemper, **C. Morrisett**, J. Sturtevant, M. Sierks, A. Willis. 2021. Climate adaptation planning to support ecosystems and people in the Gila River watershed, Arizona. Southwest Climate Adaptation Science Center, University of Arizona.

[4] **Morrisett, C**, R. Van Kirk, A. Loibman. 2019. Lower Henry's Fork hydrology and habitat assessment: Progress report submitted to meet conditions of Ora Bridge mitigation agreement. Submitted by the Henry's Fork Foundation to the Federal Highway Administration, Fremont County Idaho, and the Local Highway Technical Assistance Council, Ashton, ID.

[3] Van Kirk, R. and **C. Morrisett**. 2017. Analysis of Caribou-Targhee National Forest Fisheries and Fish Habitat Database. Prepared for USDA Forest Service, Caribou-Targhee National Forest. 33 pp. plus appendices. Henry's Fork Foundation, Ashton, ID.

[2] **Morrisett, C**. 2016. Buffalo River Fish Ladder 2006-2016 Comprehensive Report. Prepared for Federal Energy Regulatory Commission for the Buffalo River Hydroelectric Project (P-1413-038). Henry's Fork Foundation, Ashton, ID.

[1] CH2M and **Henry's Fork Foundation**. 2016. Eastern Snake Plain Aquifer (ESPA): Review of Comprehensive Managed Aquifer Recharge Program. Prepared for Idaho Water Resource Board. 44 pp. plus appendices and addendum.

Theses

[2] **Morrisett, C**. 2023. Multi-objective water management in Idaho's Henrys Fork watershed: leveraging reservoir operation and groundwater pathways to benefit aquatic habitat. Ph.D. Dissertation, Utah State University, Logan, UT.

[1] **Morrisett, C**. 2018. Assessing the utility of tributary PIT-tag arrays in monitoring Snake River salmonid recovery. M.S. Thesis, University of Washington, Seattle, WA.

INVITED TALKS:

Morrisett, C. Lower Henry's Fork habitat modeling for flow management. Henry's Fork Watershed Council Annual Conference, St. Anthony, Idaho, December 2022.

Morrisett, C. Optimizing watershed management in Idaho's Henrys Fork: Using and assessing a collaborative modeling approach. Water in the West: Toward Convergent Solutions for Water Security, Workshop Facilitated by the Center for Science, Technology, Ethics and Society (STES) at Montana State University: Bozeman, May 2022.

Morrisett, C. Panel on Student Leadership and Mental Health. Utah State University, March 2022. [Virtual]

Morrisett, C. Optimizing multi-stakeholder watershed management in Idaho's Henrys Fork. University of Washington, School of Aquatic and Fishery Sciences, Quantitative Seminar, March 2021. [Virtual]

Morrisett, C., S.E. Null, R. Van Kirk, and L. Bernier. Summer Flows for Anglers and Agriculture: Identifying an Optimal Low Flow Target for Collaborative Watershed Management in Idaho's Henry's Fork. American Water Resources Association, Annual Conference, November 2020. [Virtual]

Morrisett, C. Student Engagement Break: Next Stop Grad School, American Water Resources Association, Annual Conference, November 2020. [Virtual Panel]

Morrisett, C. PhD Project on the Lower Henry's Fork: Progress and Preliminary Findings. Henry's Fork Watershed Council Meeting, Ashton, ID, October 2020. [Virtual]

Morrisett, C. My Career in Science: Out to Sea and Back Again. Boys & Girls Club, Eagle River, AK, July 2020. [Virtual]

Morrisett, C. Linking ecohydrology and social systems to support multi-stakeholder management of the lower Henrys Fork river. Henry's Fork Watershed Council Meeting, Ashton, ID, April 2019.

PRESENTATIONS:

* indicates presenter if not first author

2023 **Morrisett, C.** Considering irrigation efficiency, aquifer recharge, and return flows for environmental flow management in the Henrys Fork, Snake River (USA). Freshwater Down Under, Brisbane, Australia, June 2023.

Hoffner, B.*, R. Van Kirk*, and **C. Morrisett***. Henry's Fork Foundation: Overview, Impact, and Stanford Internship Program. Stanford University's Bill Lane Center for the American West Advisory Council, Island Park, ID, May 2023.

Morrisett, C. Multi-objective water management in Idaho's Henrys Fork watershed: leveraging reservoir operation and groundwater pathways to benefit aquatic habitat. PhD Defense, Utah State university, Department of Watershed Sciences, April 2023.

Van Kirk, R.*, M. Muradian, **C. Morrisett**, A. Roseberry, and H. Blischke. Henry's Fork Foundation WaterSMART Applied Science Grant Update. Henry's Fork Watershed Council, Rexburg, Idaho, March 2023.

2022 **Morrisett, C.**, R. Van Kirk, and S.E. Null. Low flow in the lower Henrys Fork, Snake River: Investigating streamflow-habitat relationships to inform water management. Idaho Chapter of the American Fisheries Society, Fort Hall, ID, March 2022.

Allison, K.*, **C. Morrisett**, R. Van Kirk, M. Muradian, and J. Laatsch. Incorporating DEI in the Henry's Fork Foundation Internship Program. Idaho Chapter of the American Fisheries Society, Fort Hall, ID, March 2022.

2021 **Morrisett, C.,** R. Van Kirk, S.E. Null, A. Loibman, and L. Bernier. The Irrigation-Groundwater Connection: Sustaining agriculture for water supply resilience to climate change. American Water Resources Association, 2021 Summer Conference. [Virtual]

Morrisett, C. Optimizing watershed management in Idaho's Henrys Fork: Using and assessing a collaborative modeling approach. Utah State University, Dept. of Watershed Sciences, Graduate Research Symposium, April 2021.

Morrisett, C. Partnering with farmers to benefit cutthroat trout in Teton Valley, Idaho. National Science Foundation Research Trainee Annual Meeting, January 2021. [Virtual]

2020 J. Stuivenvolt Allen*, K.A. Spangler, B. Morgan, **C.N. Morrisett**, M.W. Brunson, S.Y.S Wang, and N.J. Huntly. Water availability for cannabis in northern California: Intersections of climate, policy, and public discourse (poster). American Geophysical Union, Fall Meeting, December 2020.

Null, S.E.*, A. Farshid, G. Goodrum, C.A. Gray, S. Lohani, **C.N. Morrisett**, and L. Prudencio. Environmental tradeoffs of dams in the Lower Mekong Basin (poster). American Geophysical Union, Fall Meeting, December 2020.

Morrisett, C., B. Morgan, K.A. Spangler, J. Stuivenvolt Allen, M.W. Brunson, S-Y. Wang, and N.J. Huntly. Water availability for cannabis in northern California: Intersections of climate, policy, and public discourse (poster). American Water Resources Association, Annual Conference, November 2020.

Brunson, M.W.*, N.J. Huntly, S. Bogen, L. Capito, M. Christman, S. Koutzoukis, B. Morgan, **C. Morrisett**, W. Munger, and K.A. Spangler. Integrating ecological and social systems models and data: An application of the 4DEE approach for graduate education. Ecological Society of America, Annual Meeting, August 2020.

Pinto, D.*, **C. Morrisett**, S. Koutzoukis, and M. Christman. Graduate researchers collaborating on interdisciplinary climate adaptation science (poster). Ecological Society of America, Annual Meeting, August 2020.

Spangler, K. A.*, B. Morgan, **C. Morrisett**, J. Stuivenvolt Allen, N. J. Huntly, M. W. Brunson, and S. Wang. Climate realities and media conversations: Water availability for cannabis agriculture in California's North Coast (poster). Ecological Society of America, Annual Meeting, August 2020.

2019 **Morrisett, C.**, S.E. Null, R. Van Kirk. Fish, farms, and low flows: Quantifying management tradeoffs for multiple stakeholders in Idaho's Henrys Fork watershed (poster). American Geophysical Union, Fall Meeting, December 2019.

Morrisett, C., R. Van Kirk, S.E. Null, B. Contor, and A. Loibman. Quantifying and managing groundwater spring flows for climate change adaptation. Northwest Climate Conference, Portland, OR, October 2019.

Null, SE*, S. Lohani, L. Prudencio, G. Goodrum, **C. Morrisett**, CA Gray. Environmental effects of dam construction in the Se Kong, Se San, and Sre Pok (3S) Rivers of the Lower Mekong Basin: A literature review. American Fisheries Society and The Wildlife Society Joint Annual Conference, Reno, NV, October 2019.

Van Kirk, R.*, B. Contor, **C. Morrisett**, and S. Null. Potential for managed aquifer recharge to mitigate climate-change effects on fish and wildlife in the Snake River Basin, USA. International Symposium on Managed Aquifer Recharge, Madrid, Spain, May 2019.

Morrisett, C. Linking ecohydrology and social systems to support multi-stakeholder management of the lower Henrys Fork river. Utah State University, Dept. of Watershed Sciences, Graduate Research Symposium, April 2019.

2018 Van Kirk, R.*, J. Laatsch, J. McLaren, **C. Morrisett**, M. Muradian, B. Oldemeyer. Climate-change adaptation in the Henry's Fork Snake River to sustain agriculture, fish and wildlife, and recreation. Northwest Climate Conference, Boise, ID, October 2018.

Morrisett, C. Assessing the utility of tributary PIT-tag arrays in monitoring Snake River salmonid recovery. Master's Defense, University of Washington, School of Aquatic and Fishery Sciences, June 2018.

Van Kirk, R.* and **C. Morrisett**. Importance of Egin Lakes managed recharge site. Henrys' Fork Watershed Council Meeting. Rexburg, ID, March 2018.

Morrisett, C., J. R. Skalski, R. B. Kiefer. The effects of adult ladder passage at Lower Granite Dam on Snake River salmonid migration. Idaho Chapter of the American Fisheries Society, Idaho Falls, ID, March 2018.

2017 **Morrisett, C.** and J. R. Skalski. The effects of adult ladder passage at Lower Granite Dam on Snake River salmonid migration. School of Aquatic and Fishery Sciences Graduate Student Symposium, University of Washington, November 2017.

Oldemeyer, B.*, J. Flinders, **C. Morrisett**, and R. Van Kirk. Long-term effectiveness of flow management and fish passage on the Henry's Fork Rainbow Trout population. Wild Trout XII Symposium, West Yellowstone, MT, September 2017.

Laatsch, J.*, R. Van Kirk, **C. Morrisett**, K. Manishin, and J. DeRito. Angler perceptions in a highly technical catch-and-release fishery: How closely does perception align with biological reality? Wild Trout XII Symposium, West Yellowstone, MT, September 2017.

Laatsch, J.*, R. Van Kirk, **C. Morrisett**, K. Manishin, and J. DeRito. Angler perception of fishing experience in a highly technical catch-and-release fishery: How closely does perception align with biological reality? Joint meeting, Idaho chapters of American Fisheries Society and The Wildlife Society, Boise, ID, March 2017.

2016 Lien, M.*, **C. Morrisett**, and R. Van Kirk. Using trout population assessment data to identify threats to native trout and to prioritize native trout conservation projects. International Trout Congress, Bozeman, MT October 2016.

SCIENCE COMMUNICATION (MULTIMEDIA):

Morrisett, C. Photo blog. Available: https://www.instagram.com/lowerhenrysfork/

Morrisett, C. Blog posts. Available: https://www.henrysfork.org/profile/christina/profile

Morrisett, C. Henry's Fork Daily Water Report. (2023, March 16–24). Daily email sent to ~300 recipients that include water managers, irrigators, and interested water users.

Morrisett, C. Henry's Fork Foundation Annual Membership Meeting. (2022, June 17). *The Lower Henry's Fork Project.* Available: https://youtu.be/s_isCBVDTt4

Morrisett, C. Henry's Fork Watershed Council Meeting. (2021, August 19). Field tour host.

Morrisett, C. Henry's Fork Foundation Annual Membership Meeting. (2021, May 28). *The Lower Henry's Fork Project*. [Webinar]. Available: https://www.youtube.com/watch?v=NU3s15F7e-8&t=1176s

Morrisett, C. September 2020. *How do I fit in? Finding my place and leaning in to boundary spanning.* [Blog post]. Natural Resource Workforce Development Fellow Highlight, Southwest Climate Adaptation Science Center, University of Arizona. Available: https://www.swcasc.arizona.edu/sw-casc-blog/how-do-i-fit-finding-my-place-and-leaning-boundary-spanning-1

Morrisett, C. Henry's Fork Foundation Annual Membership Meeting. (2020, May 29). *The Lower Henry's Fork Project.* [Webinar]. Available: https://www.youtube.com/watch?v=5ytP0bzXl_U&feature=youtu.be

Morrisett, C. HFF Virtual Get Together. (2020, Apr 22). *The Lower Henry's Fork Project.* [Webinar] Available: https://vimeo.com/411160087/b59a056285?fbclid=IwAR3MvoVLkbTcdTiku-aFqg8_x-rbrqHmwtiKpmKebVJXG-ztUCwXRixKiAw

Morrisett, C. Winter 2019. *What Can Science Tell Us About How to Improve Lower River Habitat?* Henry's Fork Foundation Quarterly Newsletter. Available: https://henrysfork.org/files/Quarterly%20Newsletters/2019%20Winter%20Newsletter%20-%20web.pdf

Henry's Fork Foundation and Fremont County. (2016, Jan 6). *The Henry's Fork: Recreation and Conservation* [Story map]. Retrieved from http://arcg.is/1Qjtftz

Morrisett, C. Summer 2016. *Genetic study identified important spawning habitat*. Henry's Fork Foundation Quarterly Newsletter. Available: https://henrysfork.org/files/Quarterly%20Newsletters/2016%20Summer%20Newsletter.pdf

Van Kirk, R. and **C. Morrisett**. Fall 2015. *Spring 2015 third driest in 80 years*. Henry's Fork Foundation Quarterly Newsletter. Available: https://henrysfork.org/files/Quarterly%20Newsletters/FA2015%20newsletter.pdf

Morrisett, C. (2015, September 10). *Discharge and Diversions in a Dry Spring.* [Blog Post]. Retrieved from https://west.stanford.edu/news/blogs/out-west-blog/2015/discharge-and-diversions-dry-spring-morrisett

Morrisett, C. (2015, July 22). *Wild Trout, Turbulent Waters.* [Blog Post]. Retrieved from https://west.stanford.edu/news/blogs/out-west-blog/2015/wild-trout-turbulent-waters-morrisett

Morrisett, C (Contributor). (2015, June 5). *Geographic Impacts of Global Change: Mapping the Stories (US)* [Story map]. Retrieved from https://www.mappingglobalchange.org/.

Morrisett, C (Producer). (2015, July 15). *Mapping Global Change – Damasa Organics: Wyndmere, ND* [Audio podcast]. Retrieved from https://soundcloud.com/mapping-global-change/sets/food-producers.

Morrisett, C (Producer). (2014, May 5). *Friends Don't Let Friends Eat Farmed Fish* [Audio podcast]. Featured on Green Grid Radio and by the Center on Food Security and Environment at Stanford University.

Morrisett, C (Producer). (2013, Aug 28). *Costa Rican Ecotour: Saving Sea Turtles and Cleaning Beaches* [Video file]. Retrieved from https://www.youtube.com/watch?v=uEtW3xBzspU.

Morrisett, C (Producer). (2013, Jul 9). *Saving Sea Turtles One Sign at a Time* [Video file]. Retrieved from https://www.youtube.com/watch?v=alMIkVqGeFM.

Morrisett, C (Producer). (2013, Jun 24). *Creating Plastic-Free Sea Turtle Habitat in Costa Rica* [Video file]. Retrieved from https://www.youtube.com/watch?v=g7SepTtLu-E.

MEDIA FEATURES:

American Fisheries Society. (2022, Jan 31). Career Outreach Features hosted by the Equal Opportunities Section. [Videos]. Available here and here.

Water Water Everywhere. (2022, Jan 11). Episode 07: Water Resources Management with Christina Morrisett [Podcast Episode]. Available: https://www.waterwatereverywherepod.com/episodes/episode-07waterresourcesmanagement-me3kf-yam28

Conservation Paleobiology Network. (2021, Mar 30). Practitioner Perspective. [Newsletter Feature]. Available: https://conservationpaleorcn.org/wp-content/uploads/2021/03/CPN-Newsletter_30-March-2021.pdf

Yale Climate Connections. (2020, Jul 28). Deliberate flooding of agricultural land could recharge an Idaho river. [Podcast]. Available: https://yaleclimateconnections.org/2020/07/deliberate-flooding-of-agricultural-land-could-recharge-an-idaho-river/

Bill Lane Center for the American West. (2018, Oct 12). Alumna Profile: Christina Morrisett '15, A Country Scientist. [Article]. Available: https://west.stanford.edu/news/christina-morrisett-country-scientist

FIELD EXPERIENCE:

06/2020 – present	Henry's Fork Watershed (14 days; piezometer installation and monitoring)
06/2019 – present	Henry's Fork Watershed (45 days; ADCP river and canal deployments)
10/2020	Henry's Fork Watershed (1 day; boat electro-fishing with IDFG)
05/2020 – 06/2020	Teton River Watershed (3 days; infiltration test monitoring for managed recharge)
09/2015 – 06/2016	Buffalo River, Idaho (60 days; fish ladder monitoring)
06/2015 – 08/2016	Henry's Fork Watershed (15 days; water/invertebrate sampling, cattle fencing)
06/2015 – 08/2015	Teton River Watershed, Idaho (30 days; backpack electro-fishing)
02/2015	Butano Creek, California (2 days; salmonid habitat assessment)
08/2014	The Pantanal, Brazil (21 days; wetland ecology field seminar)
06/2014 – 07/2014	Igushik River, Dillingham, Alaska (35 days; salmon escapement)
09/2013 – 12/2013	Queensland, Australia (30 days; terrestrial and marine ecology field courses)
06/2013 – 08/2013	Marin County, California (7 days; salmon smolt surveys)

05/2013 – 06/2013	North Central Pacific (35 days/cruise; SEA Semester S-247)
06/2011 – 09/2012	Prince William Sound, Alaska (7 days; salmon straying surveys)

SERVICE (UNIVERSITY):

08/2020–04/2022 **Graduate Student Rep. for the Quinney College of Natural Resources (QCNR)**, Graduate Student Council, Utah State University

- AY2021-22: Co-led initiatives to hire a university ombudsperson and offer subsidized childcare options to graduate students.
- AY2020-21: Led initiative to preserve the dependent option for graduate student health insurance. Co-wrote a position letter for submission to the USU Graduate Council. After solicitation, received 280 signatures and 10 statements of personal impact to accompany submission. Dependent option successfully preserved, but perhaps not affordably.

08/2020–04/2022 **Chair**, Graduate Student Council, QCNR, Utah State University

- Led monthly council meetings and participated in monthly QCNR leadership meetings, with the QCNR Dean and department heads.
- AY2021-22: Facilitated spaces for sharing graduate student experiences with comprehensive/qualifying exams. Hosted a listening/brainstorming session re: QCNR quantitative resource offerings. Met with the QCNR Business Service Center to discuss the graduate student experience and brainstorm solutions.
- AY2020-21: Facilitated creation of graduate student committee on Justice, Equity, Diversity, and Inclusion. Co-organized panel on mental health and healthcare resources at USU for QCNR graduate student community. Facilitated Q&A space for graduate students to share "hacks" about work and student life.

10/2021–12/2021 **Graduate Student Rep.,** Climate Faculty Search Committee, Dept. of Watershed Sciences, Quinney College of Natural Resources, Utah State University

- Reviewed applications and provided feedback to search committee. Participated in first-round Zoom interviews and second-round in-person interviews. Solicited graduate student feedback and summarized for presentation to students, the search committee, and the department head.

12/2018–04/2021 **Undergraduate Mentor**, QCNR, Utah State University

- Met with students to answer questions about grad school application process.

02/2020–04/2021 **Member,** Ecology Center Seminar Committee, Utah State University

	• Assisted selection of top candidates for invitation among nominees. Invited guest speaker and co-hosted virtual visit.
04/2019–08/2020	**Graduate Student Rep. for Dept. of Watershed Sciences**, Graduate Student Council, QCNR, Utah State University • Participated in monthly faculty meetings and provided summaries to students. • Spring 2019: Wrote a position letter distributed to QCNR graduate students for signature regarding unannounced changes to the Graduate Enhancement Award evaluation criteria.
10/2019	**Graduate Student Rep.**, Dept. Chair Search Committee, Dept. of Watershed Sciences, QCNR, Utah State University • Solicited graduate student feedback and summarized for presentation to students and the search committee.
04/2017–07/2018	**Founding Member**, Students Exploring Aquatic Sciences (SEAS), School of Aquatic and Fishery Sciences, University of Washington
07/2017–12/2017	**Co-coordinator**, Graduate Student Symposium, School of Aquatic and Fishery Sciences, University of Washington

SERVICE (COMMUNITY):

03/2019–present	**Hutton Scholar App. Reviewer**, American Fisheries Society (AFS)
10/2022–05/2023	**Member Coach**, Legends Boxing Gym – Cache Valley
05/2022	**Room Lead,** National Ocean Sciences Bowl (Nat. Competition)
03/2022	**Session Moderator**, American Fisheries Society (ID Chapter), Annual Meeting
03/2022	**Presentation Judge**, American Fisheries Society (ID Chapter), Annual Meeting
03/2022	**Timekeeper**, Manatee Bowl (Regional Comp.), National Ocean Sciences Bowl
06/2018–06/2020	**Hutton Scholar Pen Pal**, American Fisheries Society
02/2017–02/2020	**Application Reviewer**, UW Doris Duke Conservation Scholars Program
12/2020	**Poster Judge**, American Geophysical Union, Fall Meeting
03/2018	**Presentation Judge**, American Fisheries Society (ID Chapter), Annual Meeting
02/2015–02/2018	**Scorekeeper**, Multiple regional competitions, National Ocean Sciences Bowl
10/2011–06/2015	**External Relations Director and Mentor**, Women and Youth Supporting Each Other (WYSE), Stanford Chapter
10/2011–06/2015	**Interim Chair, Non-Gameday Ops, and Member**, The Stanford Axe Committee

WORKSHOPS ATTENDED:

01–02/2023	And, But, Therefore Framework, Facilitated by Dr. Randy Olsen, Virtual. 10 hours.
05/2022	Water in the West: Toward Convergent Solutions to Water Security, Facilitated by the Center for Science, Technology, Ethics and Society at Montana State University, Bozeman. 3 days.
04/2022	Filmmaking for Science Communication, Facilitated by Provare Media, at Utah State University, Logan, UT. 12 hours.
11/2020	Water Conflict Management and Transformation, Facilitated by Aaron Wolf, Todd Jarvis, and Todd Votteler, American Water Resources Association. 8 hours.

SKILLS:

- Languages – Spanish (limited working proficiency)
- Computer Applications and Programming – R, WinRiver II, GAMS, ArcGIS Pro, NVivo, Atlas.ti
- Certifications – First Aid/CPR (exp. 6/24), CITI: Human Research, Hunter Safety (UT), USA Boxing Coach (exp. 12/23), SCUBA
- Fieldwork – ADCP deployment, piezometer monitoring, salmonid identification, electro-fishing, fish handling, otolith extraction, small boat operation, boat trailering, manual transmission vehicle operation

STUDENTS MENTORED:

06/2022–08/2022	Chloe Perel, Undergrad. Summer Intern, Brown Uni. (co-mentored) ***Project:*** *Investigating flood-to-sprinkler conversion history in the Henry's Fork watershed*
06/2021–08/2021	Erik Sauer, Undergrad. Summer Intern, St. Lawrence Uni. ***Project:*** *Mapping thermal infrared drone imagery of groundwater flow to the Henry's Fork*
11/2018–04/2021	QCNR Undergraduates (6), Utah State University ***Topics:*** *Applying for graduate school*
06/2020–08/2020	London Bernier, Undergrad. Summer Intern, St. Lawrence Uni. ***Projects:*** *Henry's Fork water glossary and time-series analysis of river reach gains*
06/2019–08/2019	Ashly Loibman, Undergraduate Summer Intern, Colgate University ***Project:*** *Characterizing groundwater seeps to the lower Henry's Fork*