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Ecological benefits and risks to native salmonids from beaver dam analogs

Stacey Meyer[1*], Joshua L. McCormick[2], Timothy Copeland[3] and Amber Young[1]

[1]Idaho Department of Fish and Game, Salmon Regional Office, Salmon, ID, United States, [2]Idaho Department of Fish and Game, Fisheries Research Office, Nampa, ID, United States, [3]Idaho Department of Fish and Game, Headquarters Office, Boise, ID, United States

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EDITED BY
Greer Maier,
State of Washington, United States

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Jukka Syrjänen,
University of Jyväskylä, Finland
Tyson Hallbert,
University of California, Davis, United States

*CORRESPONDENCE
Stacey Meyer
stacey.meyer@idfg.idaho.gov

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In degraded river systems, beaver dam analogs (BDAs) are an increasingly popular low-tech treatment used to reduce water velocity, increase floodplain connectivity, activate secondary side channels, and thus increase juvenile salmonid rearing habitat. However, BDAs may benefit non-native species as well, posing a potential conservation risk. In the Lemhi River basin in Idaho, an Intensively Monitored Watershed program quantifies responses of salmonid populations to restoration actions intended to remediate the effects of agricultural development. In 2017, BDAs were installed in Hawley Creek, to improve habitat conditions for native Rainbow Trout (*Oncorhynchus mykiss*). Non-native Brook Trout (*Salvelinus fontinalis*) also reside in Hawley Creek. We evaluated native and non-native salmonid responses to BDAs to understand their implications for achieving restoration goals. A BACI analysis was used to evaluate the effects of BDAs on the intrinsic rate of population growth of Rainbow Trout and Brook Trout. Demographic analysis was used to estimate the effects of treatment (i.e., BDA or control) on abundance and demographic rates of Rainbow Trout and Brook Trout. Our results suggested that Brook Trout did not displace Rainbow Trout in sites with BDAs, indicating that BDAs may not greatly change conservation risk to native salmonids. Rainbow Trout abundance and apparent survival in Hawley Creek post-treatment were typically higher than for Brook Trout. Our study suggests that BDAs in degraded western streams did not favor Brook Trout over Rainbow Trout.

KEYWORDS
beaver dam analogs, brook trout, rainbow trout, intensively monitored watershed, habitat restoration, ecological risk, salmonid conservation

1 Introduction

Beaver dam analogs (BDAs) are a low-tech restoration tool which are built to mimic natural beaver dams (Bouwes et al., 2016; Pollock et al., 2012; Wheaton et al., 2019). Beaver dam analogs are used to reduce water velocity, increase floodplain connectivity, activate secondary side channels, reduce water temperatures through groundwater exchange, and increase fish rearing habitat (Bouwes et al., 2016). A BDA provides enhanced habitat to a variety of species by creating complex habitats to meet biological needs such as increased nutrient availability and reduced water velocity (Hallbert and Keeley, 2023; Wall et al., 2017). These structures are engineered to serve as an alternative to beavers in locations where beavers are no longer present on the landscape (Wheaton et al., 2019; Lahr, 2023). The use of BDAs as a restoration tool has become very popular (Scamardo et al., 2025). They are relatively low-cost, are easily implemented without heavy machinery, and are a natural-looking manipulation. However, the implementation of BDAs has outpaced research on efficacy and best practices (Pilliod et al., 2018).

In the Western United States, BDAs are implemented to provide water year-round in streams, to reinvigorate the riparian zone, and to enhance the complexity of poor-quality habitat for a variety of fish species (Munir and Westbrook, 2021; Bobst et al., 2022). Similar to other watersheds in the West, the Lemhi River basin in Idaho has experienced major alterations to its habitat through irrigation, livestock grazing, and land-use development (e.g., land agriculture and developed recreation; Watershed Advisory Group, 1998). These alterations, among others, have led to protection of Steelhead Trout (the anadromous form of *Oncorhynchus mykiss*), under the US Endangered Species Act (ESA). In addition to Steelhead, Rainbow Trout (the non-anadromous form of *Oncorhynchus mykiss*) and non-native Brook Trout (*Salvelinus fontinalis*) are present. A restoration program began in 2007 to benefit native salmonids. As part of this program, BDAs were deployed in selected Lemhi River tributaries to improve habitat in newly re-connected stream reaches. The presence of Brook Trout, which competes with native salmonids, was observed in pools formed behind the BDAs, posing a potential conservation risk (Dunham et al., 2002). How native salmonids and non-native Brook Trout interact and benefit from the implementation of BDAs is unclear.

The initiation of the Lemhi River restoration program was accompanied by an intensive monitoring program (an intensively monitored watershed, IMW). The goal of an IMW project is to understand how habitat actions and fish response are related at the watershed scale (Bennett et al., 2016). The initial emphasis of the Lemhi IMW was to restore flows in key tributaries such that they were re-connected to the Lemhi River. The hierarchical structure of the Lemhi IMW study design links results at the population scale to the finer scales most relevant to restoration practitioners. Thus, the IMW design allowed for assessment of BDAs and their consequences. Our goal in this study was to evaluate the ecological benefits and risks to native salmonids from implementing BDAs. Our objectives were to compare the abundance and selected demographic rates of native salmonids versus Brook Trout in relation to BDAs. Given the invasiveness of Brook Trout in the West (Dunham et al., 2002), we hypothesized that abundance, survival, and recruitment of Brook Trout would increase after installation of BDAs relative to the abundance, survival, and recruitment of native trout. We chose Hawley Creek, a headwater tributary of the Lemhi River, on which to focus the study. However, because Brook Trout are widely distributed, and BDAs are widely used across the West, our results are broadly applicable.

2 Study area

Hawley Creek is a 5th-order system located in east-central Idaho with a drainage basin encompassing approximately 16,835 ha (Figure 1). The creek originates at 2,256 m. above sea level in the Beaverhead Mountains on the Idaho-Montana border. Hawley Creek flows in a westerly direction for 16 km before joining with Eighteenmile Creek and forming the headwaters of the Lemhi River near Leadore, Idaho. The majority of Hawley Creek is on public land (76%), with the lowest reach and mouth on private property. Hawley Creek originates in the mountains in alpine forests and then enters the valley across an alluvial fan where the creek naturally loses some discharge to the aquifer. The riparian zone of Hawley Creek was once lush in riparian vegetation in the valley bottom. However, in the early 1900s, Hawley Creek was diverted from its native channel approximately 5.5 km upstream from the mouth where it enters the valley into an open ditch for irrigation purposes. During the 100 years that Hawley Creek was diverted into the irrigation ditch, the alluvial fan dried up and the valley bottom transitioned from riparian to upland vegetation.

Historically, Hawley Creek supported anadromous and fluvial salmonids but these species were displaced due to alterations to their habitat and dewatering of the creek (Watershed Advisory Group, 1998). In 2014, the creek was returned to its original channel and conservation efforts were made to enhance instream flow, facilitate fish migration, and improve fish rearing habitat. During this study, the creek flowed in the original channel and riparian habitat started to rebound. However, during drought conditions, the lower portion of Hawley Creek typically dries in the summer months due to a combination of losses to the aquifer and irrigation withdrawals. During 2017-2018, 37 BDAs were installed (Figure 2) to restore riparian habitat, reinvigorate the floodplain, provide water in the creek for longer periods of time, and create complex habitats for salmonids (pools, side channels, etc.) in approximately 2.5 km of stream immediately downstream from the old irrigation diversion (Figure 3). Unfortunately, mid-study in 2022, an irrigator altered 13 BDAs in treatments A and D which included partial removal of BDAs or notched BDAs (center cut out) to allow additional water to spill downstream in Hawley Creek.

Our control streams included Canyon Creek and Big Timber Creek, which are neighboring 5th-order watersheds (Figure 1). Canyon Creek, to the northeast of Hawley Creek, originates in the Beaverhead Mountains and the watershed encompasses 37,322 ha and is a combination of private and public lands. Big Timber Creek originates to the northwest of Hawley Creek in the Lemhi Mountains and the watershed encompasses 55,275 ha and is a combination of private and public lands. Both creeks join the Lemhi River in Leadore, Idaho. Historically, Canyon Creek and Big Timber Creek were disconnected from the Lemhi River. Through flow augmentation projects (e.g. barrier removal and water savings, similar to Hawley Creek), Canyon Creek and Big Timber Creek

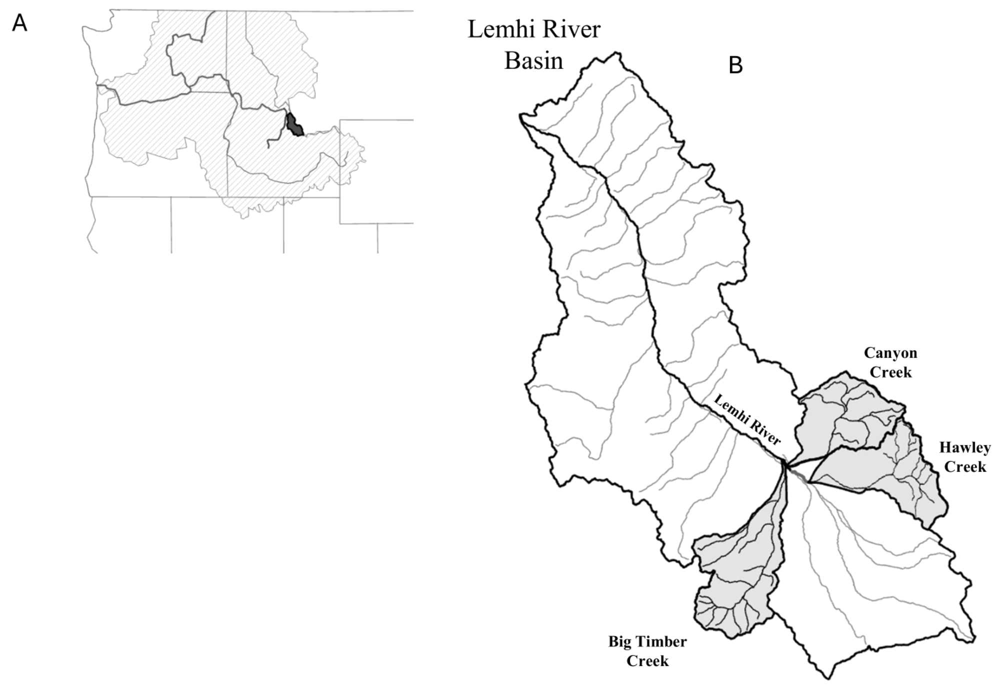

FIGURE 1
Location of the Lemhi River watershed in the Pacific Northwest **(A)**. Shaded portions show Hawley Creek, Big Timber Creek, and Canyon Creek **(B)**.

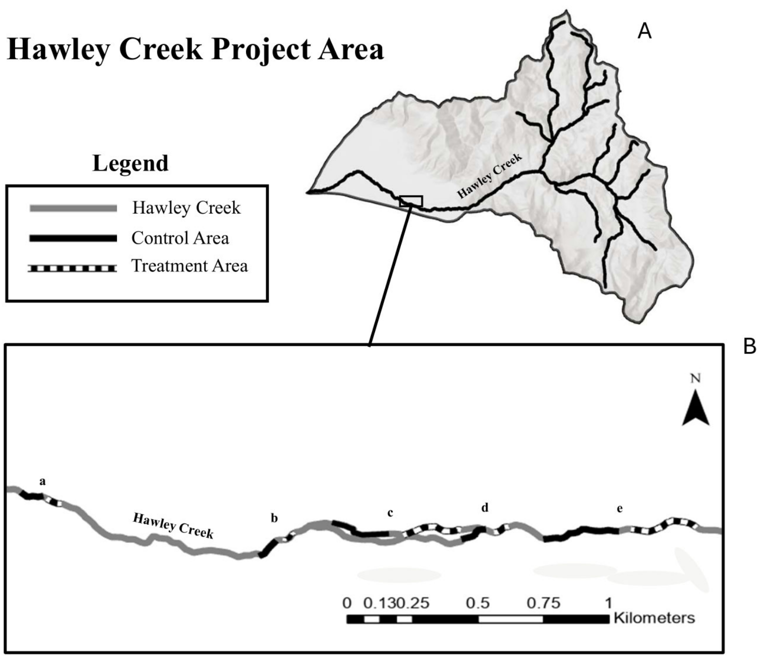

FIGURE 2
Location of survey site in Hawley Creek watershed **(A)**. Paired treatment and control reaches in Hawley Creek **(a-e)** were surveyed 2021-2024 **(B)**.

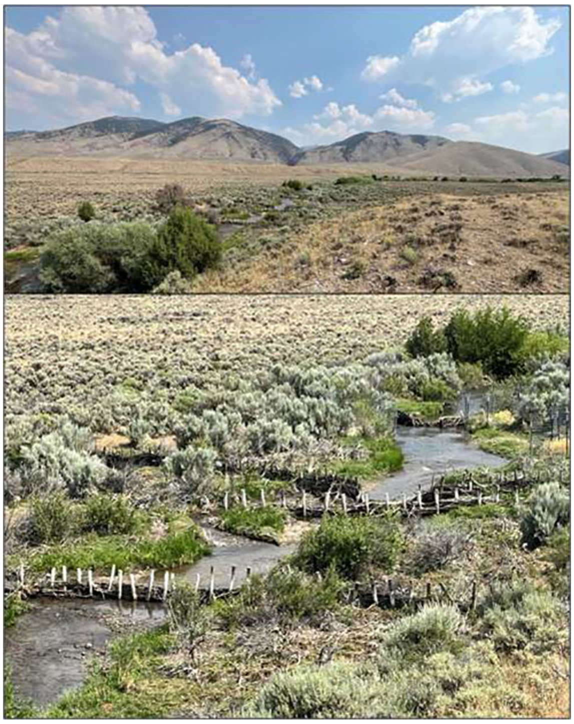

FIGURE 3
Hawley Creek in the treatment reach, July 2024, showing the beginning of riparian re-vegetation (willows along the stream). The bottom photograph is zoomed into the middle of the top photograph to show three BDAs.

were connected to the Lemhi River in 2011. While numerous flow augmentation projects were implemented in both watersheds, BDAs were never installed in Canyon and Big Timber creeks. Following reconnection, Rainbow Trout and Brook Trout were observed in the two control streams.

3 Methods

3.1 Data collection

The survey design assessed salmonids within paired treatment and control sites in Hawley Creek. The study included five reaches, each consisting of a treatment site (BDA present) and a control site (BDA absent). Control sites were located directly upstream of a paired treatment site (Figure 2). Paired treatment and control sites were the same length but lengths of site pairs varied such that the sum of total lengths sampled by site type was the same (total treatment length sampled 756.8 m, total control length sampled 756.8 m). The number of BDAs varied among treatment sites (4 to11 BDAs per treatment site). A total of 36 BDAs were sampled in this study. Sampling occurred twice a year in late May and early October for several days, from 2021 to 2024.

Pretreatment data were collected as part of the IMW surveys to investigate fish abundance and distribution in reconnected tributaries (Meyer et al., 2024). The two closest reconnected

TABLE 1 Total stream length sampled via mark-recapture electrofishing efforts in Big Timber Creek, Canyon Creek, and Hawley Creek from years 2013-2024.

Total stream length sampled (meters)			
Year	Big timber creek	Canyon creek	Hawley creek
2013	16000	12000	5900
2014	7600	9000	6300
2015	8950	7460	8200
2016	8818	9510	11254
2017	6324	7765	9193
2018	8003	4907	10394
2019	10593	980	8764
2020	2780	1020	NA
2021	4626	1619	1514
2022	7190	7689	1514
2023	774	1098	1514
2024	5052	1098	1514

tributaries were Big Timber Creek and Canyon Creek, which we use as control streams. All streams were surveyed consistently during spring through summer with single pass mark-recapture electrofishing methods beginning in 2013, although stream length surveyed varied (774-16,000 m) due to available resources and the target number of tags needing to be deployed to track distribution and survival (Table 1). The lengths surveyed were sufficient to provide estimates of fish densities in the valley reaches of the two streams and reaches surveyed were similar among years for comparable fish densities. These surveys were continued in Big Timber and Canyon creeks, while the effort in Hawley Creek was changed to evaluate BDAs.

Sampling consisted of mark-recapture electrofishing surveys. Two backpack electrofishers (Smith-Root, Inc., Vancouver, Washington; model LR-24) with pulsed DC were used to sample fish in the upstream direction. Crews consisted of 6–7 technicians. Voltage, pulse, and frequency were adjusted to maximize capture probability without causing fish injury (settings range: voltage 150-210, frequency = 30 Hz, pulse width = 24–30 ms). During the sampling event, all fish captured within the same habitat unit were held in a live well containing ambient stream water with an aerator to provide additional oxygen. To ensure fish health, sampling ceased when stream temperatures reached 17 °C (Copeland et al., 2021). Prior to handling fish, all fish were anesthetized in an anesthetic bath (Aqui-S) following the Investigational New Animal Drug protocol (U.S. Fish and Wildlife Service and Fish and Aquatic Conservation, 2023). All fish captured were scanned for previously implanted passive integrated transponder (PIT) tags using a HPR Lite scanner (Biomark Inc.) and any tag codes detected were recorded. All untagged *O. mykiss* (≥ 80 mm fork length, FL) and Brook Trout (≥ 60 mm total length, TL) were inserted with a pre-loaded 12 mm PIT tag (Biomark Inc.) in the body cavity using a PIT-tag injector. Lengths were measured from all salmonids (46–320 mm) and young-of-year-fish (i.e., individuals smaller than the minimum tag lengths) were excluded from study. All fish captured on the mark survey were fin clipped in the upper caudal fin and fin clipped fish were noted the following day on the recapture survey. Anesthetized fish were held in a live well of ambient water to recuperate before release into the habitat unit of captured.

3.2 Data analysis

Two analyses were conducted at two spatial and temporal scales in this study. The first analysis, which was based only on abundance estimates, was a before-after-control-impact (BACI) analysis where Hawley Creek as a whole was treated as the treatment population, and Big Timber Creek and Canyon Creek were treated as control populations. The "before" treatment period was from 2013 to 2018, and the "after" treatment period was from 2019 to 2024. To be consistent with the timing of other surveys, only data from the spring surveys at Hawley Creek were used in the BACI analysis. The second analysis was a post-treatment demographic analysis where recruitment, survival, emigration, and immigration rates were evaluated among treatment and control sites within Hawley Creek. Abundance estimates as well as PIT-tag data from 2021 to 2024, which were collected twice annually for a total of eight sampling periods, were integrated in the demographic analysis. Analysis focused on Rainbow Trout and Brook Trout because few individuals of other salmonid species were captured.

3.2.1 BACI analysis

A state-space model was used to evaluate the effects of BDAs on the intrinsic rate of population growth (r) of Rainbow Trout and Brook Trout using a BACI analysis. The analysis was based on the general framework outlined in Chevalier et al. (2019) and Prochazka et al. (2023). The state-space model included a state-process model that described the true but unknown abundance of fish in the population that was conditional on the year prior, and an observation model that was conditional on the process model (Kéry and Schaub 2011). The process model was defined as

$$log(Ni, j, t) = log(Ni, j, t-1) + ri, j, t,$$

where Ni,j,t is the abundance of fish of species i, in stream j, in year t, and r is the intrinsic rate of population growth, which was assumed to be normally distributed with mean $\mu i,j,t$ and variance $\sigma 2$. $\mu i,j,t$ was further modeled as

$$\mu i, j, t = \alpha i, TP + \gamma i, t,$$

where αi, $T\,P$ represents the mean intrinsic rate of population growth for each treatment (T; control or impact) and time period (P; before or after) for each species and γ is a random year effect for each species which was assumed to be normally distributed with mean 0 and variance $\sigma 2$. The observation model was based on the mark-recapture electrofishing data and was defined as

$$Ci, j, t \sim binomial(Ni, j, tlj, t, pi, j, t),$$

$$Qi, j, t \sim binomial(Mi, j, t, , pi, j, t),$$

where $Ci,j,\ t$ is the total number of fish captured on the recapture pass, l is the length of each site, pi,j,t is the capture probability, Qi,j,t is the number of fish that were recaptured that were marked on the marking pass ($Mi,j,t,$). The inclusion of site length in the likelihood functionally changes abundance to linear density in the model interpretation. Abundance was treated this way because site lengths varied among years.

Three metrics were calculated to evaluate the effect of BDAs on population growth rate of Rainbow Trout and Brook Trout. The first contrast was the BACI contrast (Chevalier et al., 2019).

$$BACI = (\alpha i, IA - \alpha i, IB) - (\alpha i, CA - \alpha i, CB),$$

where the IA subscript refers to the impact location during the after-treatment time period, IB refers to the impact stream during the before-treatment period, CA refers to the control streams during the after-treatment period, and CB refers to the control streams during the before- treatment period. A positive value for the BACI contrast suggests that the intrinsic rate of population growth increased at impact streams between time periods relative to the control sites and a negative value indicates that the rate decreased. Two control-impact (CI) measures were also calculated: the CI-contribution and the CI- divergence measures. The CI-contribution measure was calculated as

$$CI - contribution = |\alpha i, IA - \alpha i, IB| - |\alpha i, CA - \alpha i, CB|\,.$$

The CI-contribution measure describes the absolute magnitude of change in intrinsic rate of population growth at the impact location compared to the control location. A positive value for CI-contribution suggests that changes are mainly due to changes at impact sites, whereas a negative value suggests that changes in intrinsic rate of population growth were greater at control sites. The CI-divergence measure was calculated as

$$CI - divergence = |\alpha i, IA - \alpha i, CA| - |\alpha i, IB - \alpha i, CB|\,.$$

The CI-divergence measure describes how much more dissimilar intrinsic rate of population growth was at the impact location compared to the control locations. A positive value for CI-divergence suggests that the impact and control sites are more dissimilar, whereas a negative value suggests that that they are more similar. See Chevalier et al. (2019) for a more detailed description of the CI-contribution and CI-divergence measures.

3.2.2 Demographic analysis

An integrated population model (IPM) was used in the demographic analysis to estimate the effects of treatment (i.e., BDA or control) on demographic rates and population growth rate of Rainbow Trout and Brook Trout at specific sites within Hawley Creek. Two datasets were used in the analysis including mark-recapture electrofishing data when the population was closed (i.e., no births, deaths, immigration, or emigration) and mark-recapture PIT-tag data when the population was open. Abundance of a fish population of species i, at location j (treatment or control), at time t, Ni,t can be expressed as

$$Ni, j, t = Si, j, t-1 + Ri, j, t-1 + Ii, j, t-1 - Ei, j, t-1,$$

where S is the number of fish that survived throughout the time period, R is the number of local recruits, I is the number of immigrants, and E is the number of emigrants. In short, fish can be added to the population through local recruitment or immigration, and fish can be lost through deaths or emigration. It should be noted that, in this model, t represents the time between sampling events, which took place in late May and early October, and j represents treatment or control sites within Hawley Creek. One objective of BDAs is generally to increase local abundance, which is influenced by the demographic rates described above. As such, to gain a mechanistic understanding of the effects of BDAs on abundance, it is important to evaluate their effects on demographic rates.

The IPM used in this study is a type of state-space model that integrates multiple sources of data in a joint likelihood (Schaub and Kéry, 2021). As such, similar to the BACI model described above, the IPM included a state-process model that described the true but unknown state of the population and an observation model that was conditional on the process model. The process model was defined as

$$Ri, j, t \sim Poisson(Ni, j, t-1 \times \rho i, j, t-1),$$

$$Si, j, t \sim binomial(\phi i, j, t-1, Ni, j, t-1),$$

$$Ni, j, t = Si, j, t + Ri, j, t,$$

where ρ is the product of the local recruitment rate and immigration (apparent recruitment), and ϕ is the product of the survival and one minus the emigration rate (apparent survival). It was not possible to separately estimate survival and emigration, as well as recruitment and immigration in this study; however, the product of the two parameters are included in ϕ and ρ respectively.

Apparent survival ϕ (which is the product of survival and emigration) was estimated using a state-space parameterization of the Cormack-Jolly-Seber (CJS) model. Survival and detection were modeled using group effects for treatment type in the CJS model with random time effects. See Kéry and Schaub (2011, page 204) for a more detailed description of the CJS model. The observation model for the electrofishing data was the same as was used for the BACI analysis but with t representing time between sampling events rather than years as was used in the BACI analysis, and j representing treatment or control sites within Hawley Creek only. In addition, site length was not included in the observation model as all site lengths were consistent among years. Unlike the apparent survival estimates (ϕ), no data were used to directly inform the ρ parameter in the IPM. Thus, this parameter was not considered to be a "standard" parameter as defined by Riecke et al. (2019) but could be estimated as an "additional" parameter using the other data sources within the IPM. The difference in apparent survival and apparent recruitment estimates between treatment and control sites was evaluated by subtracting control values from the treatment values. Thus, a value greater than one would suggest that there was a positive effect of BDAs on demographic rates. All models were fit using Bayesian methods. Markov Chain Monte Carlo (MCMC) algorithms were used to estimate the posterior distributions for all model parameters. Analyses were performed using the JAGS program (Plummer, 2003) implemented in R using the R2jags package (Team RC, 2018; Su and Yajima, 2012). Posterior distributions were generated using three chains of 500,000 iterations with a burn in of 200,000 that were thinned by three. Parameters were checked for convergence based on the Gelman-Rubin statistic (i.e., $\hat{R}$ less than 1.05; Brooks and Gelman, 1998). The JAGS code used to fit the model, including all prior distributions, can be found in Supplementary 1. Parameter estimates were expressed as the mean of the posterior distribution along with 95% credible intervals which were based on the quantiles of the posterior distribution in the results.

4 Results

4.1 BACI analysis

The model-estimated density of Rainbow Trout was variable among streams and years over the duration of the study (Figure 4). Density in Hawley Creek over the entire study period was more stable than in either control stream. The mean rate of intrinsic population growth (i.e.,the $\alpha i,TP$ parameter) for Rainbow Trout was 0.042 (95% credible interval: -0.576—0.672) in the impact stream after the treatment, -0.129 (-0.806—0.543) in the impact stream before the treatment, -0.064 (-0.825—0.694) in the control streams after the treatment, and 0.060 (-0.757—0.882) in the control streams before the treatment. The BACI contrast estimate for Rainbow Trout was 0.295 (-0.765—1.354; Figure 4). Density in Hawley Creek increased slightly after treatment, whereas it was declining slightly before; the opposite pattern occurred in the control streams. Approximately 73% of the posterior distribution of BACI contrast was greater than zero, suggesting that there was a 73% probability that BDAs had a positive impact on the intrinsic rate of population growth of Rainbow Trout. The estimated CI-contribution measure for Rainbow Trout was -0.101 (-1.010—0.756), signifying changes were stronger in control streams, and the estimated CI-divergence measure was -0.048 (-0.766—0.626), signifying a slight convergence of impact and control streams after treatment.

Similar to Rainbow Trout, the model-estimated density of Brook Trout was variable among streams and years (Figure 4). Big Timber Creek had higher densities than the other streams except in 2019. The mean rate of intrinsic population growth (i.e., the $\alpha i,TP$ parameter) for Brook Trout was 0.318 (-0.086—0.764) in the impact stream after the treatment, 0.416 (0.004—0.834) in the impact stream before the treatment, 0.245 (-0.034—0.588) in the control streams after the treatment, and 0.320 (0.004—0.674) in the control streams before the treatment. The BACI contrast estimate for Brook Trout was -0.023 (-0.758—0.721; Figure 4). Brook Trout

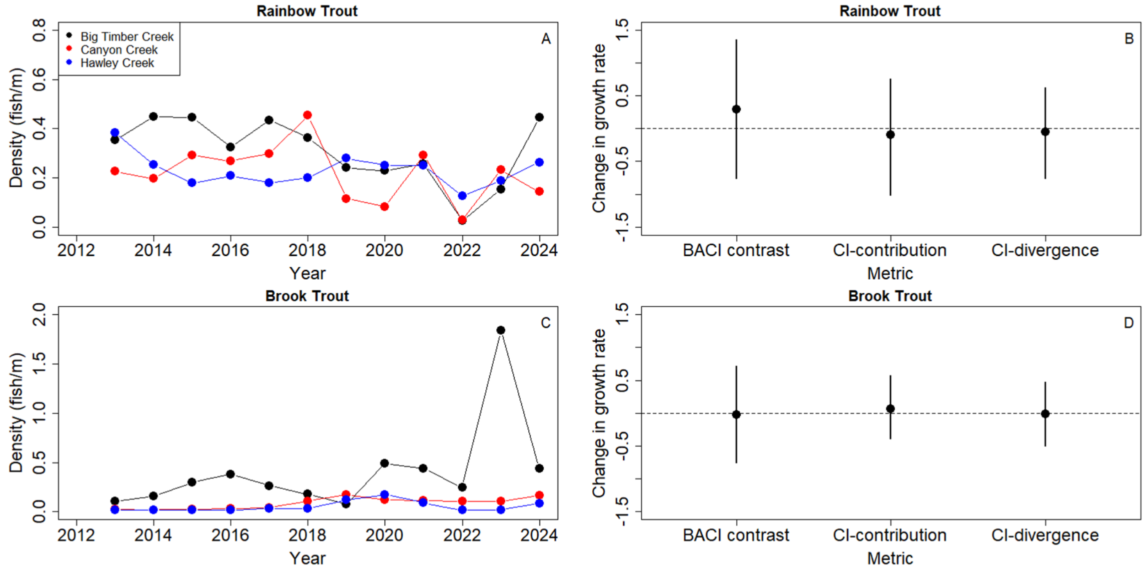

FIGURE 4
Spring density estimates of Rainbow Trout **(A)** and Brook Trout **(C)** in two control streams (Big Timber Creek and Canyon Creek) and in the BDA treatment stream (Hawley Creek). The right panel shows the results of the BACI analysis and contrasts for Rainbow Trout **(B)** and Brook Trout **(D)**. The before treatment period is from 2013 to 2018 and the after period is from 2019 to 2024. Vertical bars represent 95% credible intervals.

densities in Hawley Creek increased more slowly after treatment. Approximately 53% of the posterior distribution of the BACI contrast was less than zero, suggesting that there was a 53% probability that BDAs had a negative impact on the intrinsic rate of population growth of Brook Trout. The estimated CI-contribution measure for Brook Trout was 0.068 (-0.393—0.570), signifying a slightly stronger change in Hawley Creek, and the estimated CI-divergence measure was -0.011 (-0.496—0.475).

4.2 Demographic analysis

Rainbow Trout abundance, apparent survival rate and apparent recruitment rate changed in relatively similar ways between treatment and control sites within Hawley Creek over the eight post-treatment sampling events (Figures 5, 6). Estimated mean apparent survival of Rainbow Trout was 0.539 (0.424 – 0.639) at the treatment sites and 0.576 (0.482 – 0.658) at the control sites. The estimated difference between survival at treatment and control sites was -0.037 (-0.128 – 0.039). Approximately 18% of the posterior distribution of mean difference in apparent survival was greater than zero suggesting there was an 18% probability that BDAs had positive effect on post-treatment Rainbow Trout apparent survival. Estimated mean apparent recruitment rate of Rainbow Trout was 0.704 (0.390 – 1.106) at the treatment sites and 0.638 (0.313 – 1.069) at the control sites. The estimated difference between apparent recruitment rate at treatment and control sites was 0.066 (-0.455 – 0.578). Approximately 60% of the posterior distribution for mean apparent recruitment rate was greater than zero suggesting there was a 60% probability that BDAs had a positive effect on post-treatment Rainbow Trout apparent recruitment.

Brook Trout abundance, apparent survival rate and apparent recruitment rate had different patterns than in Rainbow Trout but were also relatively similar between treatment and control reaches within Hawley Creek over the eight post-treatment sampling events (Figures 5, 6). Estimated mean apparent survival of Brook Trout was 0.453 (0.313 – 0.620) at the treatment sites and 0.414 (0.254 – 0.612) at the control sites (Figure 5). The estimated difference between survival at treatment and control sites was 0.039 (-0.096 – 0.177). Approximately 73% of the posterior distribution of mean difference in apparent survival was greater than zero suggesting there was a 73% probability that BDAs had positive effect on post-treatment Brook Trout apparent survival. Estimated mean apparent recruitment of Brook Trout 0.977 (0.569 – 1.440) at the treatment sites and 1.034 (0.590 – 1.513) at the control sites. The estimated difference between recruitment rate at treatment and control sites was -0.057 (-0.687 – 0.579). Approximately 57% of the posterior distribution for mean apparent recruitment rate was less than zero suggesting there was a 57% probability that BDAs had a negative effect on post-treatment Brook Trout apparent recruitment.

5 Discussion

Beaver dam analogs are intended to bolster riparian vegetation growth and create complex habitats for aquatic species. Beaver dam analogs provide deeper pools, sediment deposition, and habitat for fish (Wolf, 2023). Additionally, BDAs restore degraded riverscapes and positively influence adult and juvenile salmonids by creating high-quality habitat (Cook, 1940; White and Rahel, 2008). Reinvigorated riparian vegetation can mitigate the effects of increasing temperatures, resulting in a more resilient system

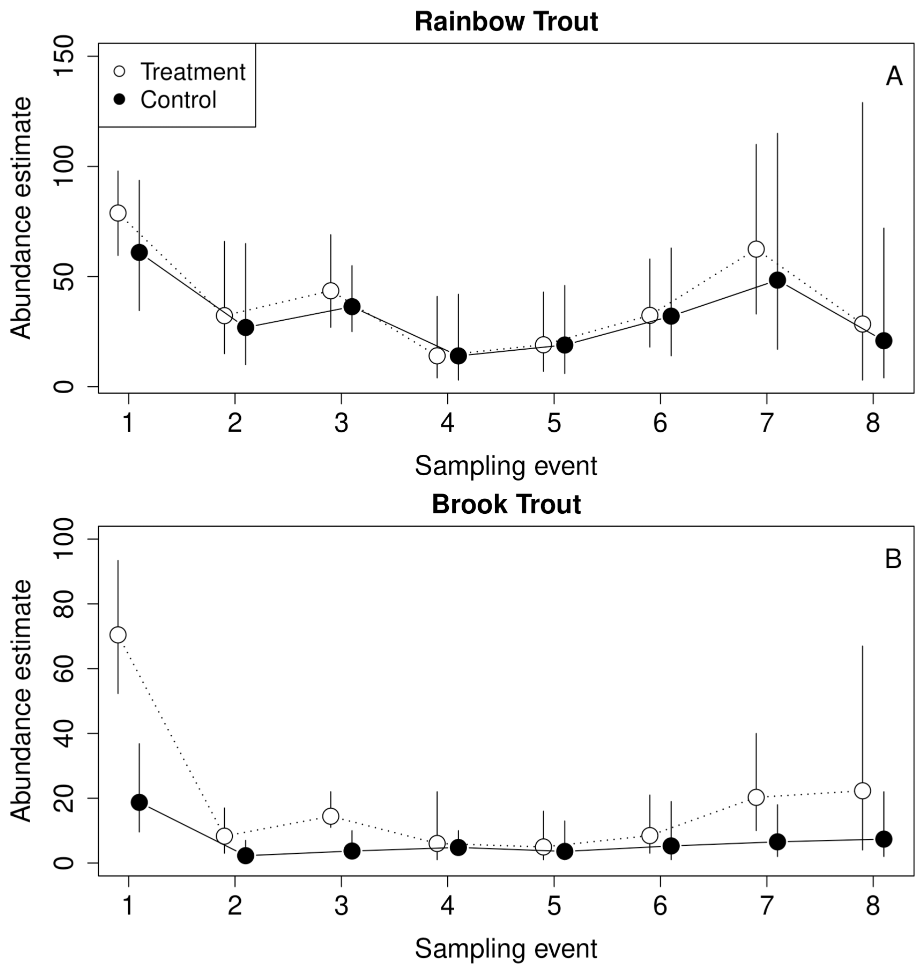

FIGURE 5
Abundance estimates of Rainbow Trout **(A)** and Brook Trout **(B)** at treatment and control sites within Hawley Creek from 2021 to 2024. Odd-numbered events are spring samples and even-numbered events are fall samples. Vertical lines represent 95% credible intervals.

(Justice et al., 2017). Beaver dam analogs can create refuges for native salmonids and are becoming a popular restoration tool throughout the western United States (Scamardo et al., 2025). Many restoration practitioners are finding that BDAs are useful in storing water and sediment, increasing riparian vegetation, and decreasing water temperatures (Scamardo et al., 2025), especially in the western United States where land use practices have negatively altered aquatic habitat (Grafton et al., 2013).

A common conservation concern in the West is displacement of native salmonids by Brook Trout (Levin et al., 2002; Benjamin and Baxter, 2012; Warnock and Rasmussen, 2013), which could be facilitated by the implementation of BDAs. Our study suggests this is not always the case. We did not find evidence that BDAs mediated the displacement of Rainbow Trout by Brook Trout. When comparing Hawley Creek to neighboring tributaries, we found BDAs likely had a positive influence on Rainbow Trout and a negative influence on Brook Trout population growth, although there was much variability about point estimates. Rainbow Trout density in Hawley Creek stayed relatively constant before and after BDA installation, while declining in the control streams. Concurrently, Brook Trout density didn't change much in Hawley Creek, while increasing in the control streams, especially Big Timber Creek. Our statistical results from the BACI analysis were ambiguous because of changes in survey sites among years, but clearly BDAs did not favor Brook Trout over native Rainbow Trout at the stream scale.

The post-treatment demographic analysis confirmed the BACI results. Rainbow Trout densities at sites in Hawley Creek were typically higher than Brook Trout densities. Rainbow Trout abundances in treatment and control sites tracked each other quite closely, whereas Brook Trout abundance was slightly higher in treatment versus control sites such that 95% credible intervals overlapped in all but two sampling events. Rainbow Trout were commonly found in riffle habitats but Rainbow Trout apparent survival estimates (survival and residency combined) did not suggest evidence of displacement from treatment reaches. Brook Trout may favor pool habitat upstream of BDAs thus influencing the location of where Brook Trout are commonly found in our study (Davis and Wagner, 2016). In Hawley Creek, the results suggested greater apparent recruitment of Brook Trout into the survey site than Rainbow Trout but not much difference between treatment and control reaches. However, results were rather

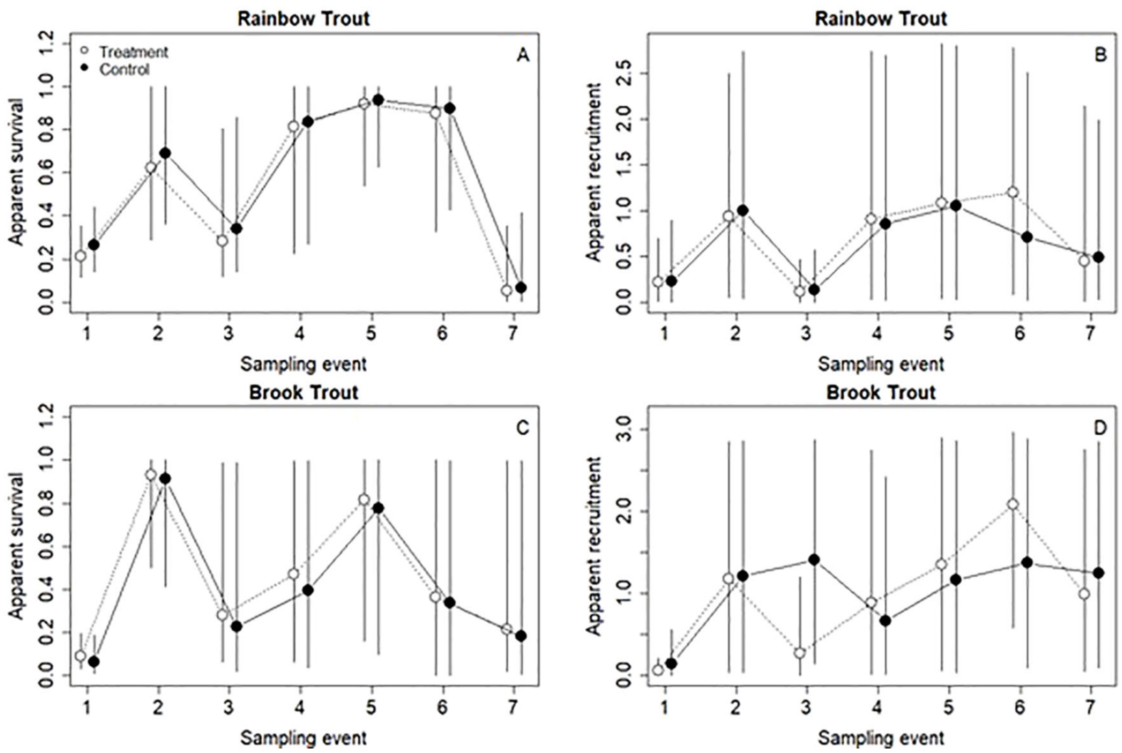

FIGURE 6
Apparent survival estimates of Rainbow Trout **(A)** and Brook Trout **(C)** and recruitment/emigration rates of Rainbow Trout **(B)** and Brook Trout **(D)** at treatment and control sites within Hawley Creek from 2021 to 2024. Odd-numbered events are spring samples and even-numbered events are fall samples. Vertical bars represent 95% credible intervals.

ambiguous due to the lack of data to inform the ρ parameter and low recapture rates, likely driven by movement of fish between sampled and unsampled reaches. Additionally, treatment sites were located upstream of control sites and likely influenced flow in the control sites; while controls lacked habitat complexity, the presence of fish within control sites was likely mediated by the effects of BDAs and upstream pool habitats.

When we initiated this study, there were few published studies evaluating the effect of BDAs on fish populations pre- and post-treatment. The study that initiated much of the BDA activity in the restoration community, (Bouwes et al., 2016) showed increased steelhead productivity in the BDA treatment stream compared to control streams. Since then, a few other studies have been completed that assessed fish response to BDAs. In western Montana, Lahr (2023) found that, although study reaches often went dry due to drought, BDAs did not negatively affect native Cutthroat Trout (*Oncorhynchus clarkii*) through non-native Brook Trout interactions (Lahr, 2023). In a Utah headwater drainage, Wolf (2023) found no difference in adult Brown Trout (*Salmo trutta*) abundance in BDA treatment sites compared to control sites. In the Klamath River basin, Tonty (2023) suggested that Coho Salmon (*Oncorhynchus kisutch*) growth and survival was similar or higher in BDA sites when compared to reference sites, however growth and survival varied depending on season. In southeast Idaho, Cutthroat Trout growth and young-of-the-year abundance was greater in reaches with instream structures and pool habitat that mimicked beaver dams than in sites without these structures (Hallbert and Keeley, 2023). Differences among the foregoing studies are likely attributed to factors that influence fish populations that cannot be controlled in the study (e.g., drought, water temperature, irrigation practices). However, there are still relatively few published studies about the effects of BDAs on native salmonids.

We implicitly assumed a step effect by BDAs but some factors likely reduce effectiveness. Typically, BDAs have a lifespan before they start to deteriorate without proper maintenance (Scamardo et al., 2025). While the majority of BDAs in our study were creating habitat complexity throughout the study, some were starting to deteriorate and spill additional water from the upstream pool. Social factors also make studying the outcome of BDA implementation challenging. Mid study, an irrigator decided to remove and alter all the BDA structures in two of our treatment reaches. Prior to removal, Brook Trout and Rainbow Trout were found in these reaches; after removal, none were found in one of the reaches. Rangeland streams such as Hawley Creek are socio-ecological systems in which concerns over water rights may limit BDA projects (Dunham et al., 2018; Charnley et al., 2020; Pilliod et al., 2018).

The Lemhi IMW program provided the framework and survey effort for monitoring of the stream reconnections required to address conservation goals for native salmonids. Although the focus of restoration in the watershed is to benefit anadromous

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salmonids, trout species serve as a good surrogate for anadromous species in headwater tributaries. The first step was to reconnect headwater reaches with the Lemhi River and the next was to improve habitat quality in the re-established stream reaches. The IMW design directed surveys of streams as they were reconnected, which were easily redirected to evaluate the BDAs, and provided the control data from streams where BDAs were not deployed. The hierarchical nature of the Lemhi IMW design connects specific management actions (like BDA deployment) to a population response and subsequently adaptation of the restoration program to what is learned.

In summary, our findings provide insight into how native and non-native salmonids respond to BDAs that can be applied to other low-tech restoration practices. Our study suggested that BDAs in highly degraded western streams provided habitat for native Rainbow Trout and non-native Brook Trout; however, Brook Trout did not displace Rainbow Trout in reaches with BDAs, indicating the BDAs did not greatly change conservation risk for native salmonids in the watershed. The leveraging of resources and context via the Lemhi IMW was an efficient way to use an adaptive management approach to gain reliable knowledge on fish response to BDAs. Moreover, BDAs are effective at providing complex habitat for a variety of cold-water species in the Western United States.

Data availability statement

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

Ethics statement

Ethical approval was not required for the study involving animals in accordance with the local legislation and institutional requirements because not applicable for this study with IDFG.

Author contributions

SM: Project administration, Writing – review & editing, Data curation, Writing – original draft, Investigation, Supervision, Conceptualization. JM: Writing – review & editing, Writing – original draft, Formal Analysis. TC: Writing – original draft, Writing – review & editing. AY: Writing – review & editing, Supervision.

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Conflict of interest

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

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