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Baseline data for assessing beaver dam analogs as a restoration tool in fire-affected tributaries of the Methow and Okanogan watersheds

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Baseline data for Assessing Beaver Dam Analogs as a restoration tool in fire-affected tributaries of the Methow and Okanogan Watersheds

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ABSTRACT

Incised streams are disconnected from their floodplains and no longer store water effectively. This leads to diminished ecosystem function, loss of critical riparian and aquatic habitats, and reduced biodiversity. Beaver dams improve incised streams by raising surface and groundwater levels, leading to reconnected floodplains. When beaver establishment is not feasible, Beaver Dam Analogs (BDAs) may be used to mitigate damage from stream incision and facilitate beaver establishment. However, it is unclear how effective BDAs are at mimicking natural beaver dams, especially on streams affected by high-intensity wildfires. The objective of my research is to collect baseline data needed to assess BDA effectiveness in comparison to natural beaver dam complexes. I hypothesized that beaver dam sites would have lower channel incision, higher accumulation of fine sediment, higher abundance of wetland species, greater water storage, and higher soil moisture compared to non-beaver sites, and that BDA installation would make the BDA sites more similar to beaver sites. I used a Before-After-Control-Reference-Impact study design to compare five BDA restoration sites with paired control sites and three natural beaver dam complexes. In the summer of 2021, pre-restoration data was collected on 1) channel morphology using a laser level and stadia rod, 2) riparian vegetation accounting for riparian landform using the line-intercept method, 3) sediment composition using a Wolman pebble count, and 4) water storage using a salt drip to measure water travel time. In the summer of 2022, I assessed soil moisture above the stream channel (floodplain for beaver sites and terrace for non-dammed sites) one month after BDAs were installed on one restoration site. Overall, I found that beaver sites had width-to-depth ratios and floodplain widths over twice as large as non-beaver sites indicating they were less incised. They also had finer sediment, greater water travel times indicating greater water storage, and higher soil moisture that lasted through the summer months. Compared to beaver and control sites, pre-BDA sites had the lowest cover of wetland species. My study has shown that beaver dams effectively trap fine sediment, recharge soil moisture in floodplains, and increase the cover of wetland species. I have also provided critical baseline data needed to assess the impacts of BDAs over time after installation is complete to determine whether they effectively mimic beaver dams.

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INTRODUCTION

Riparian zones provide critical wildlife habitat and maintain biodiversity and productivity in water-limited environments. Unfortunately, anthropogenic disturbances such as extirpation of the North American beaver (*Castor canadensis*), timber harvest, mining, channel straightening, damming, grazing, wildfire, and climate change have reduced and degraded over 80% of riparian habitat (Knopf et al. 1988, Naiman et al. 1995, Patten 1998). Currently, riparian zones comprise less than 2% of the dryland regions of the western United States of America (USA) (Svejcar 1997, Capon et al. 2013, Isaak et al. 2018). In the Methow and Okanogan watersheds of Washington State, riparian zones have been reduced by intense grazing and high-intensity wildfires (Dennison et al. 2014, Whipple 2019).

Beaver populations were extirpated by the 1850s from many historic regions leading to greater riparian habitat loss (Jenkins and Busher 1979, Naiman et al. 1986). The combined loss of riparian zones and beaver ponds has resulted in alterations to natural flow regimes and sediment and pollutant runoff increases (Poff et al. 1997). Over time, these modifications have reduced ecosystem resilience to drought and wildfires, which negatively affect both terrestrial and aquatic organisms.

Anthropogenic disturbance and high-intensity wildfires have increasingly caused severe channel incision, which results in floodplains becoming disconnected from their streams, reducing ecosystem function (Shields Jr. et al. 2010). Channel incision is a growing issue in the Western USA where increasing drought and wildfire, caused in part by climate change, compound the problem (Westerling et al. 2006, Beechie et al. 2012, Bowman et al. 2020). Incision is caused by disturbances such as overgrazing, water management, road infrastructure, and high-intensity wildfires that reduce riparian vegetation cover, which in turn increases erosion (Pollock et al. 2014). Increased erosion leads to channel downcutting, causing streams to be incised and disconnected from their floodplains; thus water storage and riparian zones become diminished (Pollock et al. 2014). Natural flooding maintains nutrient cycling and complex exchanges between the riparian and stream ecosystems (Poff et al. 1997). When streams are disconnected, floods can no longer reach their floodplains, reducing nutrient exchange and dynamic channel-forming flows (Junk et al. 1989). Consequently, incised streams tend to have terrestrialized floodplains, with reduced biodiversity and wildlife habitat (Shields Jr. et al. 2010). Incised streams have a more homogeneous habitat, with reduced wetted area, woody debris, and deep pool habitat, resulting in lower fish species richness (Shields et al. 1994, Beechie et al. 2008).

The lower water tables and terrestrialized floodplains of incised streams have many negative consequences for ecosystem function. For example, the ability of fire to move and penetrate the upland is influenced by fire intensity and the width of the functioning floodplain (Pettit and Naiman 2007). More terrestrialized riparian zones allow for further accumulation of fuel loads, while decreased water surfaces allow fires to cross stream channels more freely (Naiman et al. 2010). Plant communities transition from moist riparian vegetation to dry upland vegetation, often including fire-prone invasive annual grasses. This can negatively affect wildlife that depends on native riparian vegetation, like the Columbian white-tailed deer (*Odocoileus virginianus leucurus*), which lives in the wide floodplains of the Columbia River (Suring and Vohs Jr. 1979).

Without a healthy riparian zone, stream quality is reduced for important fish communities. In incised streams, sedimentation is reduced and turbidity is dramatically higher compared to non-incised, urbanized streams, indicating that incision has a stronger influence than urbanization on turbidity (Shields Jr. et al. 2010). Additionally, phosphorus levels are higher in incised streams as there is not a wide riparian zone that can hold and process runoff (Shields Jr. et al. 2010, Whipple 2019). Increased phosphorus and nitrogen input from nearby agriculture can lead to nutrient uptake by algae and bacteria (Zaimes et al. 2008, Fox et al. 2016), resulting in reduced dissolved oxygen and increased turbidity, termed eutrophication (Rao 2007). Eutrophication can harm the salmonid populations of greatest ecological concern that rely on water with high dissolved oxygen by increasing physiological stress; in extreme scenarios this can lead to death (Cornelius et al. 1995).

One way to improve degraded incised streams is with the help of beaver dam complexes. Beaver dam complexes have been shown to restore incised streams by increasing water levels, nutrient exchange, riparian native plant diversity, and trapping sediment (Pollock et al. 2007). Beaver dam complexes increase water storage in incised streams by raising water levels in the stream and groundwater (Beechie et al. 2008): this increases interactions between stream water and sediments, allowing microbes to uptake nutrients (Law et al. 2016, Puttock et al. 2018). Increased water storage restores riparian water tables by extending hyporheic exchange over the floodplain (Westbrook et al. 2006, Janzen and Westbrook 2011).

Beaver ponds improve water quality by trapping and converting nutrients and increasing sediment deposition, thereby decreasing turbidity downstream (Gurnell 1998).

Turbidity is a measure of how clear the stream is, which can indicate the amount of suspended sediment and pollutants in a stream (Swanson and Baldwin 1965). Streams with riparian buffers have lower streambank erosion and contribute less soil and phosphorus into streams than those without a riparian buffer, thereby reducing turbidity (Zaimes et al. 2008). Unstable banks lead to erosion and high sediment input into the stream. When these are adjacent to agricultural fields, bank erosion can carry fertilizer into the stream, causing phosphorus and nitrogen inputs to increase. Beaver ponds retain phosphorus for longer periods, allowing it to be taken up by vegetation, thus reducing downstream runoff (Naiman and Melillo 1984, Law et al. 2016). This is especially important after elevated phosphorus runoff following wildfires (Whipple 2019).

The abiotic changes caused by beaver dams benefit vegetation and wildlife. Species diversity is increased in beaver ponds because they create heterogeneous zones in riparian and aquatic habitats by creating geomorphic areas with lower velocity, higher hydraulic residence times, and greater water depth and temperature variability (Wathen et al. 2019, Majerova et al. 2020). Additionally, beaver ponds increase heterogeneity by increasing channel aggradation, widening, and sinuosity, thereby lowering channel gradient (Bouwes et al. 2016). Beaver dam complexes provide more favorable conditions for the growth of plants such as willow and alder, providing bank stabilization, cover, and refuge during seasonal low flows or drought (Hammerson 1994, Penaluna et al. 2021).

Beavers coexist with and benefit many salmonids across North America by increasing water quality, habitat, and food sources, yet many organizations still trap beavers to improve salmon and steelhead habitats (Pollock et al. 2004, Bouwes et al. 2016, Wathen et al. 2019). However, prior to human settlement, beaver, salmon, and steelhead coexisted in high densities (Chapman 1986). After the extirpation of beavers from many watersheds along with contributing anthropogenic activities, salmonid habitat quantity and quality were reduced. For example, in the Stillaguamish River Basin in Washington, coho salmon (*Oncorhynchus kisutch*) summer habitat capacity was reduced by 61% compared to historic levels mostly due to the loss of beaver ponds (Pollock et al. 2004). Salmonid habitat quality has also been reduced with increased water temperatures from climate change (Ficklin et al. 2013). Beaver ponds increase groundwater infiltration and stream temperature heterogeneity, but whether they increase or decrease pond temperature is still understudied and variable across the existing literature (Błędzki et al. 2011, Majerova et al. 2015, Wathen et al. 2019). Flooding and side channels caused by beaver ponds create areas of high flow refugia and habitat for rearing juvenile salmonids (Bouwes et al. 2016). Additionally, beaver ponds increase food availability for fish. Lotic macroinvertebrate taxa are replaced by lentic taxa in beaver ponded areas and the total biomass of benthic macroinvertebrates is 2-5 times larger in the summer, providing a large food source (McDowell and Naiman 1986). In South America, non-native North American beavers create higher growth rates of non-native Brown Trout (*Salmo trutta fario*) as a result of high macroinvertebrate density (Arismendi et al. 2020).

Despite the many ecosystem functions beavers and their dams provide, they are absent from many watersheds where they historically occurred. North American beaver populations were estimated to be 60–400 million individuals prior to European colonization, but beavers became nearly extinct due to trapping between 1620 and 1900 (Seton 1929, Jenkins and Busher 1979, Naiman et al. 1986). In most areas, beavers were extirpated from their natural ecosystems (Jenkins and Busher 1979), and beaver habitat was converted into dry land for farming and ranching (Shaw and Fredine 1971). Now, with conservation and restoration efforts, their population numbers have risen, however, their level of recovery is unknown (Gibson and Olden 2014). Human conflict, trapping, and degraded ecosystems prevent them from reaching their historic population levels (Naiman et al. 1986). While beaver reintroduction may help restore streams, there are areas where reintroduction may not be feasible or may require prior restoration to ensure the habitat is viable for beavers.

Where beaver reintroduction is infeasible, beaver dam analogs (BDAs) may be a useful tool. BDAs are man-made structures mimicking natural beaver dams. Given that they are maintained less frequently, often use more porous materials, or construction is regulated by sediment disturbance restrictions, it is not known how effective BDAs are at mimicking natural beaver dam ecosystem functions. Land managers have begun looking to BDAs as a potential restoration strategy for incised streams when beavers are not present, or reintroduction is not feasible. BDAs are becoming increasingly popular despite limited research on their efficacy (**Table 12** summarizes all previous primary research). The few existing BDA studies (18 papers with data per Scopus, May 2023, **Table 12**) are based on one or few BDAs with short time scales (Bouwes et al. 2016, Munir and Westbrook 2021a, Pearce et al. 2021b), and have limited to no data collection prior to restoration (Scamardo and Wohl 2020, Davis et al. 2021). Individual BDAs, however, do not tell the whole story as multiple BDAs should be built along a reach of a stream to act as a complex rather than a single dam to more closely resemble natural beaver dam complexes, to increase complex resilience, and to reduce structural integrity stress on each BDA. This approach is more analogous to beaver dam complexes (Pollock et al. 2014).

In the short-term, the published studies show very promising results when utilizing BDAs to repair stream incision. Researchers have found that BDAs increase in-stream surface area directly upstream of BDAs or create active side channels, and they decrease in-stream surface area downstream of the structures (Vanderhoof and Burt 2018). Orr et al. (2020) found that, after one year, groundwater levels upstream of the BDA structures rose 18-30 cm, and the water spread out into the floodplain, causing the stream to reconnect with its floodplain. On the semiarid Red Canyon Creek in Wyoming, five BDAs were installed along a 250-meter stretch. Compared to the untreated site, the BDA reach showed less bank erosion, less overall erosion, and greater spatial heterogeneity in erosion and deposition patterns (Pearce et al. 2021b). However, this study was limited to one restoration reach, and after one year most of their structures failed. Davis et al. (2021) found that, along the same BDA stretch, sediment went from aggradation above the first BDA with the highest deposition along the inner edges of the meanders, and then it transitioned to erosion by the last BDA. The strongest influence on the local sediment supply was the BDA order and whether a structure breached. After construction of four BDA sites on French Creek in Northern California, aquatic invertebrate density, beta-diversity, and gamma diversity significantly increased in comparison to the control site (Corline et al. 2022). In the Great Salt Lake catchment, BDAs acted as habitat for tiger salamanders (*Ambystoma mavortium*) on the complex scale (Wolf and Hammill 2023).

The most studied BDA project thus far is on Bridge Creek in Oregon. Four BDA reaches on only one stream were compared to four control reaches with variable amounts of reference reaches as beavers moved around the landscape. The BDA structures moderated extreme summer temperatures and led to increased heterogeneity of stream temperatures on the channel scale (Weber et al. 2017). The survival and production of juvenile steelhead significantly increased after BDA installation without reducing migration ability (Bouwes et al. 2016). Additionally, the vegetation productivity remote sensing index normalized difference vegetation index (NDVI) was significantly higher in the BDA reaches compared to control sites, and pre-restoration (Silverman et al. 2019).

Not all studies of BDA restoration exhibit the desired outcomes, and some outcomes are not consistent across studies. Munir and Westbrook (2021b) saw that overall, as the number of BDAs installed in a sequence increased, the stream temperature increased. However, as the depth of the pond increased, the stream temperature decreased due to surface albedo lowering (2021b). Additionally, another study from 2018 investigated the BDA restoration sites on Fish Creek and Campbell Creek, Colorado. The BDAs were installed across large stretches. Researchers hoped to find that BDAs increased groundwater tables, and one BDA on Fish Creek did show an increase in groundwater levels upstream of the dam, however, they were not able to show a significant increase across the complexes (Scamardo and Wohl 2020).

In the Methow and Okanogan watersheds in north-central Washington, increasing stream incision has occurred for more than a century related to agricultural water abstraction, channel straightening, intensive livestock grazing, high-intensity wildfires, and beaver extirpation. BDA complexes have been built on Triple Creek and Meyers

Creek in the Okanogan Watershed by the Okanogan Highlands Alliance and the creeks visually appear to be recovering from incision (OHA 2023). A new largescale stream restoration project proposing to build BDA complexes in five wildfire impacted tributaries across these two watersheds began in 2020. I collected baseline data in these tributaries to assess the restoration impact of BDA installations as part of a Before-After-Control-Reference-Impact (BACRI) study design. This study compares five BDA restoration sites (impact) on severely incised streams to paired control sites with severe incision and no restoration and three natural beaver dam complexes (reference). In 2021, pre-restoration data was collected as a baseline on paired BDA and control sites and natural beaver dam complex sites. Two BDA complexes were installed in the summer of 2022 out of five streams with restoration plans throughout the watersheds to reconnect the streams to their historic floodplains. In 2022, continued pre-restoration baseline data was collected for the three untreated BDA sites, post-restoration for one site, and the second post-restoration site was excluded from analyses as it was installed during data collection. My study will be the first step of a long-term study assessing BDA success over time and will help guide land managers use of BDAs as a restoration tool.

This study is currently the only BDA study to look at BDA complexes on multiple streams across two watersheds using the BACRI design. Riparian and stream ecosystems are highly dynamic and my BACRI design will allow us to account for natural variability caused by site specific river and weather conditions when assessing the effects of BDAs.

Study Objectives

My study aims to test the hypothesis that natural beaver complexes increase stream and riparian habitat quality and that incised streams prior to restoration will have reduced ecosystem function compared to streams with beaver dam complexes. Specifically, I predicted that BDA sites will be more similar to incised, unrestored control sites prior to BDA installation, and they will be more similar to natural beaver dam complexes after restoration. Comparing the BDA complex and control sites to the natural beaver dam complexes in 2021, I predicted that within beaver dam complexes there would be lower channel incision, more diverse vegetation with a greater abundance of wetland and woody riparian species, and greater sediment retention. As a result of greater sediment retention in beaver complexes, I predicted that turbidity would not increase as much with discharge downstream from beaver dam complexes, and within complexes, streambed sediments would have finer particle sizes. I expected beaver sites to have greater water travel times relative to reaches without impoundments due to slower downstream movement of water. Lastly, during Year 2 (2022), I predicted that the soil moisture at newly built BDA sites would be more similar to beaver dam complexes than to control and pre-restoration sites.

METHODS

Study Area

The study area for this project included sites in the Methow and Okanogan watersheds located in Okanogan County, Washington, USA (**Figure 1**). The Methow River is a free-flowing tributary of the Columbia River, with its watershed on the eastern slope of the north Cascade Mountains located in north central Washington State. The watershed has a dry continental climate with precipitation primarily falling as snow November-March. The Okanogan River is in northeastern Washington, in the central portion of Okanogan County, and is a major tributary to the Columbia River.

The weather was unusually hot and dry in 2021, which may have affected my study. According to the Winthrop weather station (WINTHROP 1 WSW), Okanogan County has an average precipitation of 57.68 cm a year; the average summer daily temperature including the highs and lows from June to September in Okanogan County was 15.6℃ between 2000 and 2020 (NOAA 2023). The highest average temperature in those years was 16.7℃ in 2015 (NOAA 2023). In 2021, the average summer temperature was 17.2℃ (NOAA 2023). In the summer of 2021 temperatures rose to 47.2℃ on June 29th. According to the Winthrop weather station, the next highest temperature on record was in 1930 and 1939 at 41.1℃ (NEMAC 2023).

Along with record breaking temperatures, Okanogan County was experiencing low flows. Bonaparte Creek's USGS stream gauge in Okanogan County is an accurate representation of the tributaries in this study and it began recording discharge in March 2016. Throughout the majority of 2021 discharge was below the 6-year median daily flow (USGS 2023). Between 2016 and 2020, the average monthly discharge in May and April during the high flow season from snow melt was 24.97 ft^3/s (USGS 2023). In 2021, high flows averaged only 8.05 ft^3/s (USGS 2023). During the low flow season between 2016 and 2020, discharge averaged 1.24 ft^3/s in August and September, whereas in 2021, low flows averaged 0.62 ft^3/s (USGS 2023).

The streams in this study have all been impacted by recent large wildfires and subsequent debris flows. Okanogan County has seen an increase in forest fires due to drying conditions from climate change (Westerling et al. 2006, Bowman et al. 2020). In 2014, the Carlton Complex Fires burned Bear Creek, Texas Creek, Cow Creek, and Chiliwist Creek. The Carlton Complex Fires also burned much of the subbasin that Texas Creek's reference site is in, but not the actual site used in this study. In 2015, the Okanogan Complex Fires burned Tunk Creek. In 2021, the Walker Fire burned upstream of Tunk Creek's reference site.

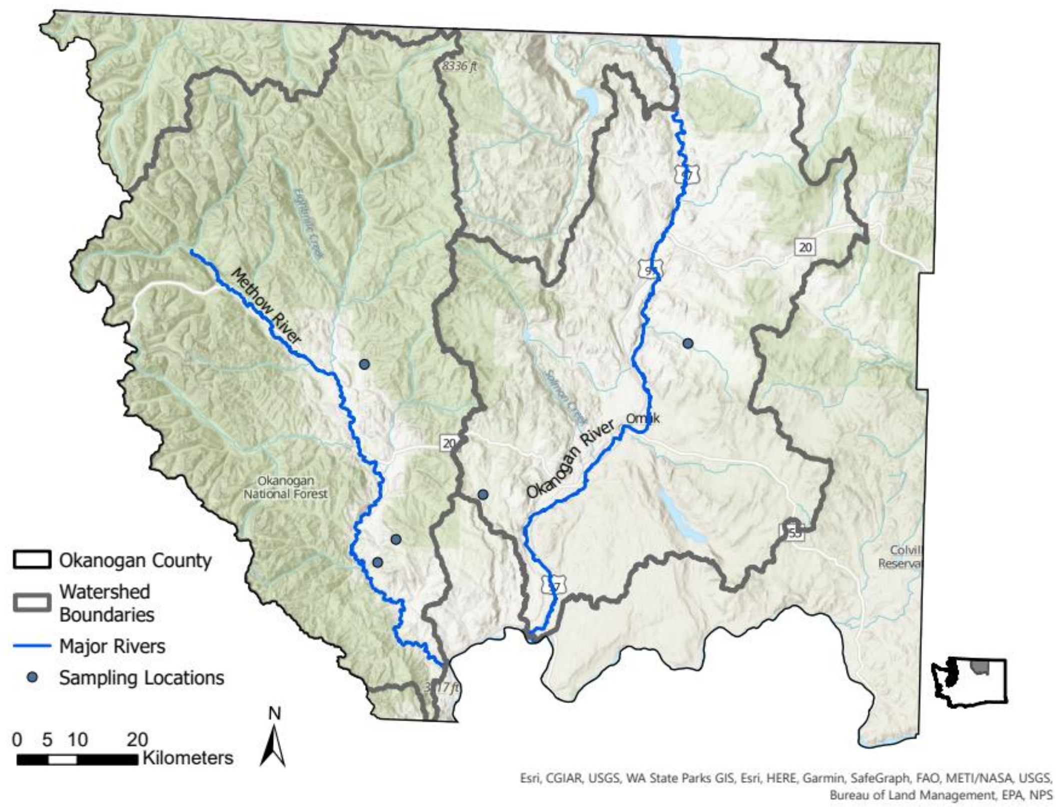

Figure 1. Site locations in the Methow and Okanogan watersheds, WA, USA. The stream names and locations of beaver sites were not included on the map to protect beaver communities.

The Methow watershed has relatively more forested habitat than the Okanogan watershed, which has more shrub/scrub habitat (**Figure 2**, **Table 1**). The Okanogan watershed has more agricultural (Hay/pasture and cultivated crops) use compared to the

Methow (4.63% compared to 0.58%). The Methow and Okanogan watershed canopies are dominated by ponderosa pine (*Pinus ponderosa)*, and Douglas fir (*Pseudotsuga menziesii*). The riparian zone shrubs/understory trees found in the headwater streams are serviceberry (*Amelanchier alnifolia*), Scouler's willow (*Salix scouleriana*), and grey alder (*Alnus incana*). Both watersheds are home to culturally important and federally listed salmonids including spring chinook (*Oncorhynchus tschawytscha*), steelhead trout (*Oncorhynchus mykiss*), and bull trout (*Salvelinus confluentus*).

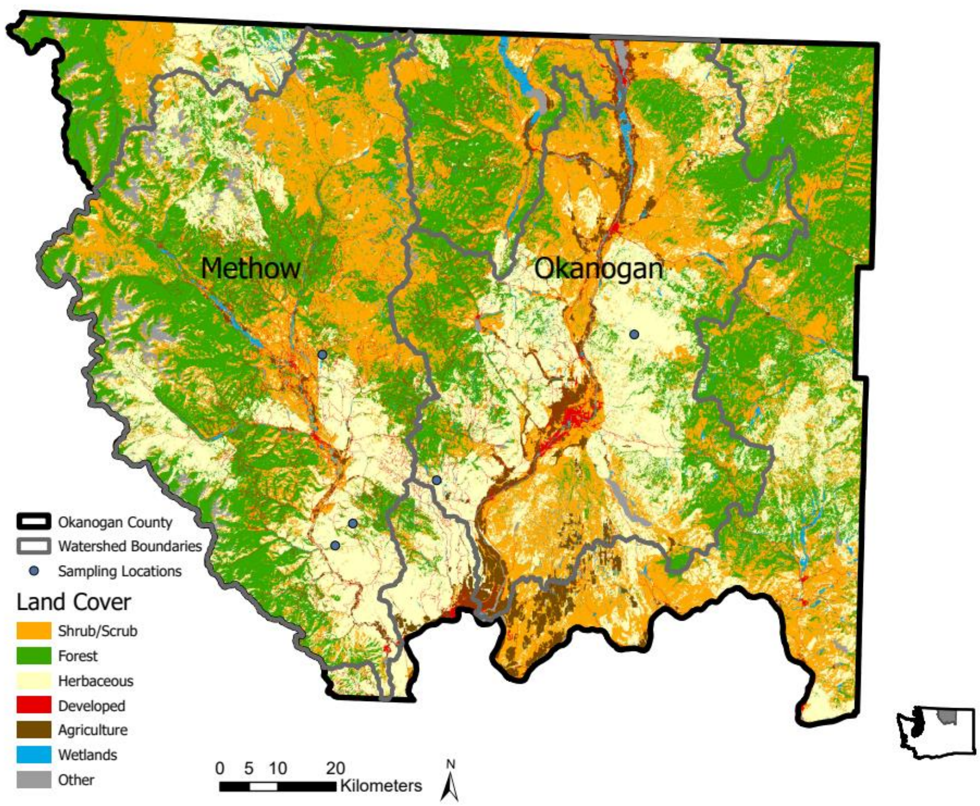

Figure 2. Land cover types in Okanogan County, WA, USA. The National Land Cover Data Set (NCLD 2019) was consolidated into seven land use categories using ArcGIS Pro. Open water, perennial snow/ice, barren land, and unclassified were reclassified as "Other". Developed open space, and developed low, medium, and high intensity were reclassified as "Developed". Deciduous, evergreen, and mixed forests were reclassified as "Forest". Hay/pasture and cultivated crops were reclassified as "Agriculture". Woody

and herbaceous wetlands were reclassified as "Wetlands". Shrub/Scrub and Herbaceous land use types were not reclassified.

Table 1. Land use types across the Methow and Okanogan Watersheds, USA listed from highest to lowest percent cover. Percentages are for seven consolidated landform types based on NCLD (2019) data.

Watershed	Land use type	Cover %
Okanogan	Shrub/Scrub	34.21
	Herbaceous	33.23
	Forest	22.25
	Agriculture	4.63
	Developed	3.06
	Other	1.36
	Wetlands	1.26
Methow	Forest	34.76
	Shrub/Scrub	30.81
	Herbaceous	25.53
	Other	4.11
	Developed	3.19
	Wetlands	1.03
	Agriculture	0.58

Study Design and Site Selection

BDA treatment reaches were selected by the Methow Beaver Project (MBP) based on the severity of wildfire impact, landowner interest in stream restoration and eventual beaver reestablishment, and feasible access to the site to install BDAs and conduct year-round monitoring. MBP used Relative Elevation Modeling (Powers et al. 2019) and the Beaver Restoration Assessment Tool (BRAT v. 3.1) to target and design BDA complex installations within the restoration reaches. The Relative Elevation Modeling helps identify and visualize historic stream conditions and assess risk related to current human infrastructure. The BRAT model identifies the potential for beaver dam building and beaver dam capacity based on drainage network characteristics, stream gradient, and vegetation cover (Macfarlane et al. 2017, Weirich 2021). Areas with high potential and capacity ratings are generally more suitable for beaver reintroduction and therefore more likely to respond well to the installation of BDAs. Originally twelve BACRI study sites were planned, including four where BDAs would be installed on incised streams, four matched incised control sites, and four sites with wild beaver complexes. Matched BDA, control, and beaver sites were chosen with similar elevation, vegetation, and water discharge to ensure comparability between sites. However, although we examined multiple beaver complexes, we were only able to identify three that were a reasonable match for the paired control and BDA sites in terms of stream discharge and slope, and for which we were able to obtain landowner permission to work. We added an additional pair of BDA and control sites to add greater statistical power for this comparison. In sum, we identified five BDA sites, five control sites with no planned restoration, and three beaver dam complex sites as references (**Table 2**). The locations of the beaver sites are left undisclosed for the protection of the beaver communities and therefore I used a naming scheme of Beaver 1, Beaver 2, and Beaver 3. In the Methow watershed, four streams were included: Bear Creek, Texas Creek, Cow Creek, and Beaver 2's stream. In the Okanogan watershed, three streams were included: Tunk Creek, Chiliwist Creek, and Beaver 3's stream. All control locations are upstream of the paired BDA complex sites by at least 50 meters. The streams were first through fifth order with slopes ranging from 3.4-9.4 % rise (**Table 3**).

Slope, stream order, and contributing watershed area were determined using a geographic information system (GIS). I calculated slope as the percent rise from 50 meters upstream of the highest sampling point to 50 meters below the lowest sampling point using a 10 m resolution National Elevation Dataset (NED 2023). The Hydrology Toolbox was used to determine stream order and contributing watershed area using the approximate locations of sampling sites along streams identified using the NED 10. All analysis and calculations were performed using ArcGIS Pro (ArcGIS 2023).

In the spring of 2022, one of the BDA sites was colonized by beavers before BDAs were installed. This site was removed from the study and excluded from turbidity and soil moisture analyses. Additionally, by the spring of 2022, beavers were no longer present at Beaver 2 for unknown reasons. The stream transitioned into hyporheic flow, with no obvious surface water present, other than the area with the beaver dam still which was slightly inundated. This site was still treated as a beaver site in the summer of 2022. Beaver 3 also lost its beavers due to agency trapping. The influence of the dams remained significant while the dams stayed intact. By July 2022, the dams were starting to degrade, and their influence was diminishing. However, they were re-colonized by beaver by the August sampling time, and so Beaver 3 remained a beaver site for summer 2022 sampling.

For the summer 2022 sampling period, Texas Creek's restoration site was classified as BDA as it was built in May, a month before summer sampling. Chiliwist Creek's BDA complex was built in July. As the stream is intermittent in the summer and was dry by August, Chiliwist's restoration site was excluded from the soil moisture models. Because BDAs were not completed before summer 2022 on all other restoration sites, they were classified as pre-BDA. BDA complexes in these other sites will be installed throughout 2023.

Table 2. Matched sites were used for this study. Creek names for the beaver sites are excluded to protect the beaver populations.

Site-matched Group Name	**Beaver Dam Analog**	**Control**	**Beaver Dam Complex**
Bear	Bear BDA	Bear Control	Beaver 1
Texas	Texas BDA	Texas Control	Beaver 2
Cow	Cow BDA	Cow Control	
Chiliwist	Chiliwist BDA	Chiliwist Control	
Tunk	Tunk BDA	Tunk Control	Beaver 3

Table 3. Stream slope, stream order, and contributing watershed of each study site. Slope is averaged across the entire site. Contributing watershed area is calculated to the most upstream location on the site.

Site	Average Slope (% Rise)	Stream Order	Contributing Watershed Area (km^2)
Bear BDA	3.37	4	27.99
Bear Control	7.44	3	17.76
Beaver 1	4.42	3	18.76
Texas BDA	9.40	3	8.43
Texas Control	6.86	3	6.49
Beaver 2	4.34	1	0.78
Cow BDA	6.90	3	12.60

Cow Control	8.69	3	11.32
Chiliwist BDA	3.43	4	33.34
Chiliwist Control	5.48	4	32.69
Tunk BDA	7.91	5	142.25
Tunk Control	5.92	5	142.06
Beaver 3	1.91	5	334.70

BDAs were constructed by MBP using untreated wood posts 3.25 inches in diameter and 6 to 8 feet long. The posts were installed into the channel substrate to a depth of 1 to 3 feet, depending on the substrate acceptance, using a handheld hydraulic post pounder powered by a small portable gas generator. The posts were placed in an upstream or downstream convex formation with two sets of posts arranged in offset post construction to temporarily withstand or deflect high stream power. Woody debris was frequently added upstream of BDAs to decrease stream power and increase the positive effects of the BDAs. For each BDA, first conifer bough mattresses were laid down in an upstream and downstream orientation and then locked in place with a conifer bough weave through the posts. This was repeated until the desired height of the BDA was achieved. Native species conifer boughs (typically Douglas fir, (*Pseudotsuga menziesii*) for weaving the BDAs were procured onsite when available or imported from the closest area of opportunity. Wood for the woody debris between BDA posts came from local wildfire burned and dead/down wood acquired on site except at Chiliwist Creek, where it was brought in from a forest thinning project. At the thalweg of treatment streams, the BDAs are approximately 3 to 4 feet high to provide a height above peak flow. In areas of tight access, vegetation was pruned and any species that could reproduce from these cuttings, such as willows, were planted opportunistically. The number of BDAs and other restorations are summed in **Table 4**. The BDAs are dissimilar from beaver dams because beavers add sediment to their dams to fill holes, whereas BDAs are expected to fill with some sediment over time from high flow events. It is difficult to make BDAs entirely non-porous and undercutting has resulted in previous projects. Additionally, BDAs are maintained less frequently and made with the goal of withstanding flooding events, however, they are not reliably subject to floods, and some fail during these events. BDAs must be built in consideration of the stream power year-round; thus some BDAs fail during high flows.

Table 4. The number of BDAs, log jams, and willow live stakes established by the summer of 2022 on the restoration streams in Okanogan County, WA, USA by the Methow Beaver Project. Streams with NA have restoration construction planned for 2023.

Action	Texas	Chiliwist	Cow	Bear	Tunk	Total
Year Treated	2022	2022	2022	2023	2023	
Month Treated	May	July	Nov.	NA	NA	
Structures						
BDAs	33	31	33	NA	NA	97
Log jams	21	16	18	NA	NA	70
Willow live stakes	25	25				50
Kilometers of stream treated	1.06	0.79	0.84	NA	NA	2.69

Floodplain and Channel Morphology and Riparian Vegetation

Floodplain and channel morphology, riparian vegetation, and sediment composition were determined by surveys conducted during the summer months (July-September) at all 13 sites. Within beaver dam complexes and BDA installation sites, transects were established at 25%, 50%, and 75% of the distance from the most downstream dam to the upstream extent of the complex (or expected upstream extent). If these distances fell on a dam or planned BDA location, the transect was moved five meters upstream. In undammed control sites, the three transects were located at 25%, 50%, and 75% of the total length of the reach. At sites that had steep, unstable banks with many fallen logs, some transects were moved a reasonable distance for the safety of the field technicians. The transects spanned between valley walls on both sides of the stream, and the ends were located one meter up the valley slope where it transitioned from the floodplain or terrace landform. Transect ends were permanently monumented using rebar to facilitate long-term repeat surveys.

To test whether there was lower channel incision within the beaver dam complexes, floodplain and channel morphology were measured using a stadia rod and level along the three cross-channel transects. Landform classifications were defined as stream (inundated stream channel), bar (within the stream channel area but not submerged), floodplain bank (hereafter "bank", high gradient transition between stream channel and floodplain), floodplain (low gradient riparian area adjacent to the stream channel that experiences inundation every 1-3 years), floodplain terrace (hereafter "terrace", former floodplain but currently isolated from seasonal high flow inundation, due to incision), and valley wall (high gradient transition away from the floodplain into dryland habitat) (Latterell et al. 2006, Whipple 2019). Bankfull width-to-depth ratios and floodplain widths were calculated (Beechie et al. 2008) to assess channel incision. Bankfull elevation was determined for all transects to use in width-to-depth ratios. Bankfull elevation is the height at which seasonal high flows overtop into the floodplains

(Harrelson 1994). In highly degraded and incised streams, bankfull height can be hard to determine due to high flows not connecting to the historic floodplain thus bankfull elevation is at a lower elevation than the height of the actual bank. Bankfull indicators are used to determine the elevation such as changes in sediment, bank vegetation, organic debris, bank undercutting, and crustose lichens with water stains (Harrelson 1994). The bankfull and surface water elevation, and base elevation for each transect were determined to calculate bankfull width and then the width-to-depth ratios and floodplain widths in streaMMetricsTM (Gemmill 2000). Floodplain width is the width at two times bankfull height. For the beaver sites, many of the transects did not go the full length into the flood prone area and thus the floodplain width became the length of the transect. This estimate was still sufficient to determine the differences in floodplain width between treatments, however, it was not sufficient to determine the differences in entrenchment ratios. Base elevation, as a reference for repeat sampling, was the actual elevation of the starting rebar.

To test whether there is more diverse vegetation with a greater abundance of wetland and woody riparian species within the beaver dam complex sites, I recorded plant species across the three transects using the line intercept method (Canfield 1941). Position in landform and strata (herb, shrub, understory, or canopy) were marked for each plant species. Landform type was determined using flood frequencies (Hupp and Osterkamp 1996). Wetland species were classified using the USDA Plants National Database wetland indicator status ratings (USDA 2021) to assess the abundance of wetland (hydrophyte) species (**Table 9**). *Epilobium obscurum* was unclassified and I classified it as facultative wetland hydrophyte because it was only found in the bar landform next to the stream, and I left every other unclassified species as unclassified.

Sediment Composition and Transport, and Water Quality

To test whether beaver dam complexes retain sediment better than control or pre-BDA sites, streambed particle size distributions and bed-level changes were used to assess sediment storage (Fischenich and Little 2007). On each of the three vegetation transects, particle size classifications were determined using Wolman Pebble Counts (Wolman 1954). At each site, the 100+ observations were split between the three transects. I recorded observations on each transect starting at the top of one incised bank and moving across the stream to the top of the other incised bank, thus within the historical channel. I crossed the stream along the transect until I reached approximately 33 samples on each transect and 100+ samples per site. If I was not at the top of the bank, I continued across the stream collecting samples until I was.

To test the ability of beaver sites to reduce suspended sediment concentrations compared to control and pre-BDA sites, I measured turbidity (NTU) at each study site monthly throughout the summer of 2021 and during changes in flow for the remainder of the year. During the summer of 2022, turbidity concentrations were measured monthly. Samples were collected at the upstream and downstream ends of planned BDA or beaver dam complexes (or comparable distances in control sites). I measured turbidity for each sampling location at each sampling date using a Turner Designs Aquafluor handheld fluorometer. I also measured water temperature, pH, conductivity, and dissolved oxygen concentration using a handheld YSI 556 Multimeter (**Table 11**, **Figures 12** to **15**). The

YSI was calibrated for pH and conductivity monthly and for dissolved oxygen daily. Some pH measurements had to be removed due to a pH meter malfunction. Lastly, discharge was estimated by the cross-sectional area method using a Hach FH950 flow meter (**Table 11**, Gordon et al. 2004).

Water retention

To test the hypothesis that streams with active beaver impoundments have greater water travel time relative to reaches without impoundments; water time travel was determined during the low flow of summer 2021 (July-September). Travel time is defined as the mean time for a particle of water to travel from the upstream end of the reach to the downstream end. For these measurements, an injection reach was selected within each of the 13 study sites to include multiple impoundments, except for Beaver 2 which includes only one beaver dam, and a comparable distance in the control sites. The injection reach length varied between sites and was later normalized to 200 meters.

I conducted conservative tracer injections to measure travel time using a saltwater mixture that was estimated by the stream discharge and was kept at safe concentration levels for wildlife. The saltwater mixture was continuously dripped for at least 2 hours (Gordon et al. 2004). A Fluid Metering International pump controlled the drip, which was located ~15 m (one pool and riffle sequence) above the upstream end of each injection reach. Conductivity was recorded to monitor the movement of the saltwater drip at the upstream and downstream ends of the study site. It was monitored and recorded once per minute at the upstream and downstream ends of the injection reach using a handheld YSI 556 Multimeter. The increase, plateau, and decrease of conductivity were recorded at each of these endpoints. Water travel time was estimated as the difference in the time conductivity reaches plateau concentrations at the upstream end of the reach and the time the conductivity reaches plateau concentrations at the downstream end of the reach (Stream Solute Workshop 1990).

Soil Moisture

Soil moisture was surveyed during the summer of 2022 along the three established transects at all 13 sites and was the only data collected on the post-restoration Texas BDA site. Along the floodplain in beaver sites or terrace in non-beaver sites, measurements were made every 1 m to a depth of 10 cm using a HOBO 10HS Soil Moisture Smart Sensor and USB Micro Station Data Logger. The probes were calibrated monthly according to manufacturer specifications. The top of the bank for beaver sites or incised bank for non-beaver sites was classified as zero meters and measurements were made on both sides of the stream. On beaver sites measurements were made on the floodplain, whereas the incised streams lacked a floodplain, so measurements were made on the terrace. Measurements were made up to 20 meters or to the rebar (one meter into the valley wall) if closer. When needed, the duff was carefully moved aside to expose mineral soil before data collection and then replaced to avoid altering soil moisture.

Data analysis

I processed the floodplain and channel morphology data to get width-to-depth ratios and floodplain widths using streaMMetricsTM (Gemmill 2000). A variable called “matched site group,” or “group” for short, was included as a random variable in models to account for matched sites (e.g., Bear control, Bear BDA, and Beaver 1 are all in the group "Bear"). To determine if beaver dam sites had lower channel incision than non-dammed sites, I tested the effect of treatment on width-to-depth ratios and floodplain widths using mixed-effects linear models with treatment as a fixed effect and matched site groups as a random effect. To fix issues with normality and to improve the data distribution, I used a log transformation on both models. These models were then visualized in RStudio using the ggplot2 package (Wickham et al. 2016, Rstudio 2023).

To compare wetland vegetation cover across sites, total wetland species cover was divided by transect length to create a variable called "proportion wetland". Plants were classified as wetland species if they had a facultative (FAC), facultative wetland (FACW), or obligate wetland (OBL) wetland indicator status (**Table 9**). To compare woody riparian cover across sites, total woody riparian cover was divided by transect length to create a variable called "proportion woody riparian". Plants were classified as woody riparian species if they were woody and had a wetland species status described above. *Salix melanopsis* was the only woody species with an OBL status. To compare upland vegetation cover across sites, total upland species cover was divided by transect length to create a variable called "proportion upland". Plants were classified as upland species if they had a facultative upland (FACU), or obligate upland (UPL) wetland indicator status (**Table 9**).

To test whether beaver sites had a greater proportion of wetland and woody riparian vegetation than other sites, I used mixed-effects linear models with the matched site groups as a random effect and landform type and treatment as fixed effects. To test whether non-dammed sites had a greater proportion of upland vegetation than beaver sites, I used a mixed-effects linear model with the matched site groups as a random effect and landform type and treatment as fixed effects. To fix issues with normality and to improve the data distribution, I used log transformations on the vegetation models. These models were then visualized in RStudio using the ggplot2 package (Wickham et al. 2016, Rstudio 2023).

To visualize patterns in species composition, I created a non-metric multidimensional scaling (NMDS) ordination, and I tested for statistically significant differences in community composition among treatments using a PERMANOVA (Rstudio 2023). The data points were split by both treatment and landform. Any species found in only one location were removed from the analysis (11 species out of the 162). The ordination used a Bray-Curtis dissimilarity index, and the data were transformed using a Wisconsin double standardization. Ordinations were created using the vegan package in R, with 999 iterations and six dimensions. The ADONIS function was used for the PERMANOVA, and the permanova_pairwise function from the ecole package was used for pairwise comparisons among sites (Rstudio 2023). Six dimensions were used based on a Sheppard's plot created using the pairwise.perm.manova function in the RVAideMemoire package (Figure 3)(Rstudio 2023). Lastly, I ran an indicator species analysis using treatments as groups with the multipatt function in the indicspecies package (Rstudio 2023).

To test whether beaver dam complexes had greater sediment retention than the pre-BDA and control treatments, I compared the distribution of the all of the Wolman pebble counts using a Kruskal-Wallis test (Kruskal and Wallis 1952). Additionally, to test if beaver dam complexes retained more sediment, I calculated a variable called transport, which was the average turbidity downstream minus the average turbidity upstream of the complex (or similar distance in non-beaver sites). If the number was negative, it meant sediment was retained within the complex. I tested the effect of treatment on retention using a mixed-effects linear model with the matched site groups as a random effect, and month and treatment as fixed effects. To fix issues with normality and to improve the data distribution, I used the output of the Box-Cox transformation (lambda=1.3) on retention (Box and Cox 1964, Rstudio 2023).

Water travel times were normalized across all sites to a 200 m stretch of stream. To determine if beaver sites had greater water travel times than pre-BDA or control sites, I used an ANOVA test with treatment as the predictor variable.

To determine if beaver sites had higher soil moisture above the stream channel (floodplain for beaver sites and terrace for non-dammed sites) between treatments, I first log-transformed soil moisture. To test if beaver sites would have greater soil moisture than non-dammed sites and that the BDA site would be similar to the beaver sites, I used a mixed-effects linear model with matched site groups as the random effect and distance from the edge of the incised bank, month, and treatment as fixed effects. To determine if pre-BDA and control sites had similar terrace soil moisture, I used a mixed-effects linear model with the same variables as the previous test. To determine if the recently built (a few weeks to three months during the sampling period) Texas BDA site had higher soil moisture than its matched control site, I used a mixed-effects linear model with the same variables as the previous two tests. These models were then visualized in RStudio using the ggplot2 package (Wickham et al. 2016, Rstudio 2023).

RESULTS

Floodplain and Channel Morphology

Beaver sites had width-to-depth ratios over three times as large as pre-BDA and control sites (conditional R^2=0.345, $p<0.001$ and $p= 0.003$ respectively, **Figure 3**), indicating lower channel incision in beaver occupied reaches. Pre-BDA and control sites had similar width-to-depth ratios of around 12.58 and 14.76 respectively. Beaver sites had floodplain ratios over twice as large as both pre-BDA sites and control sites (conditional R^2=0.285, $p<0.001$, and $p=0.004$ respectively, **Figure 4**). Pre-BDA and control sites had similar floodplain widths of around 6.33 and 8.35 meters respectively.

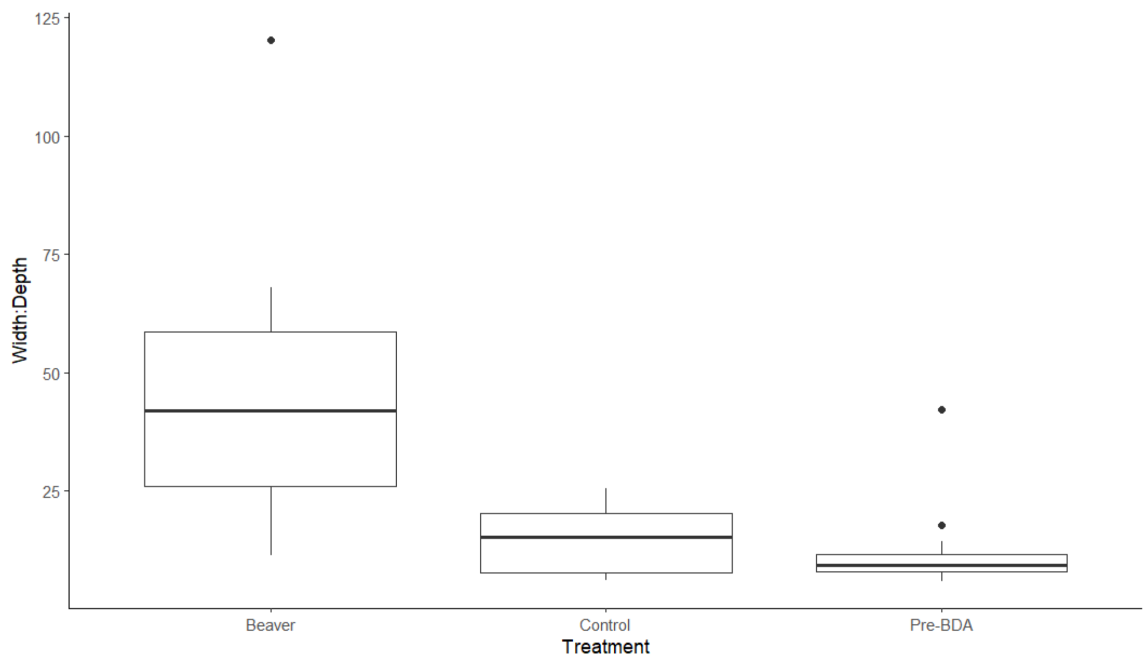

Figure 3. Effect of treatment (beaver, control, pre-BDA) on the width-to-depth ratio of headwater tributaries in the Methow and Okanogan watersheds, WA, USA.

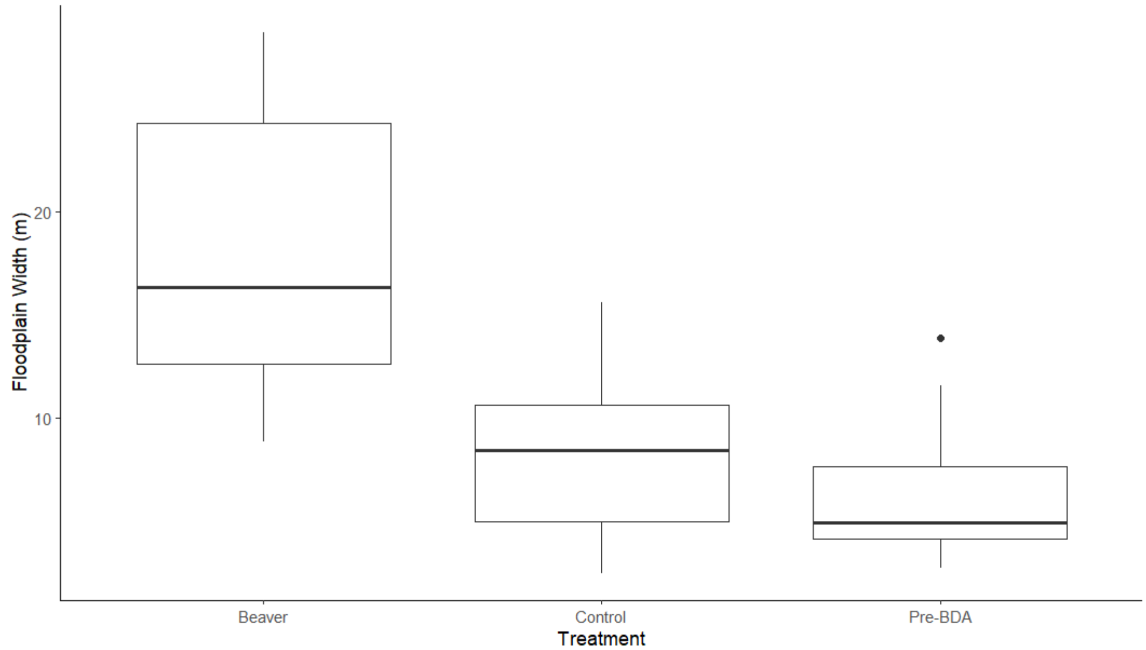

Figure 4. Effect of treatment (beaver, control, pre-BDA) on floodplain width of headwater tributaries in the Methow and Okanogan watersheds, WA, USA.

Vegetation

Beaver dam complexes did not have a higher proportional cover of woody riparian species than pre-BDA sites. Control sites had more woody riparian species than pre-BDA sites, but there were no differences between the beaver sites and any other treatments ($p<0.001$, conditional $R^2=0.173$, **Figure 5**, **Table 5**). The floodplain and terrace landforms both had a greater proportion of woody riparian species than the valley wall landform, and the terrace landform had a greater proportion of woody riparian species than in the stream (**Figure 5**, **Table 5**).

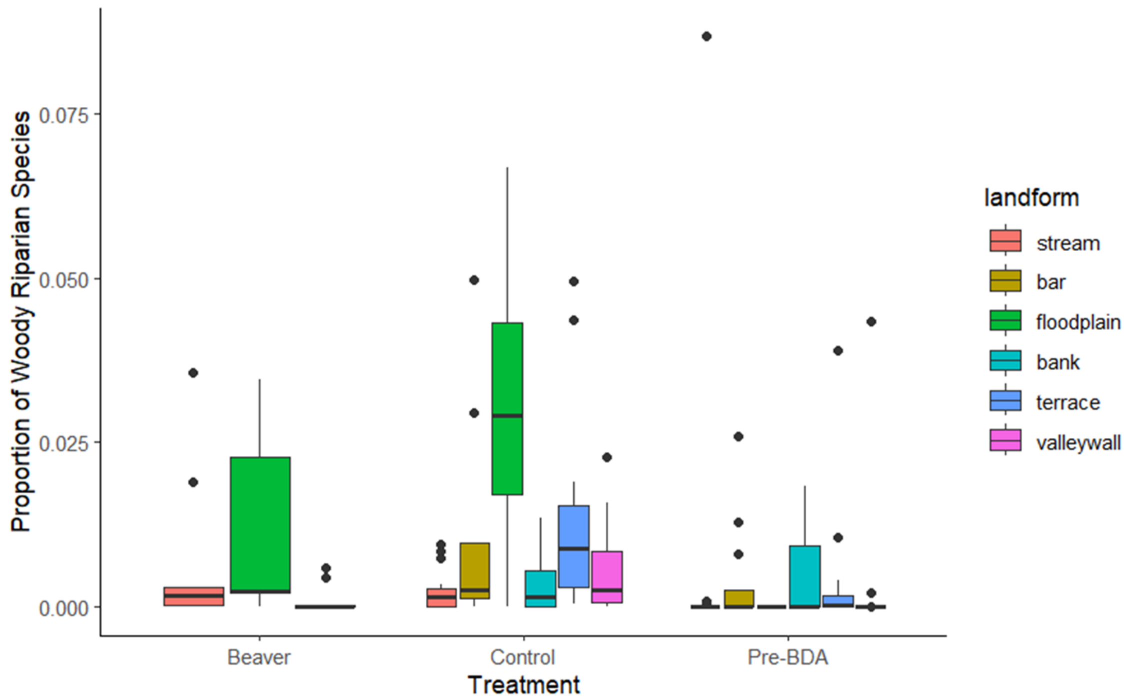

Figure 5. Effect of treatment (beaver, control, and pre-BDA) and landform type on the proportion of woody riparian species in headwater tributaries in the Methow and Okanogan watersheds, WA, USA. The proportion represents the total length of woody riparian species crossed by a transect divided by the transect length. Woody riparian species include woody species with a facultative, facultative wetland, or obligate wetland (FAC, FACW, and OBL) USDA wetland indicator status

Beaver dam complexes had higher proportions of wetland species than pre-BDA sites (conditional R^2= 0.405, p=0.002, **Table 5**). Control sites had higher proportions of wetland species than pre-BDA sites but were not significantly different from beaver sites (**Figure 6**, **Table 5**). Landform type was also a strong predictor (p<0.001, **Table 5**); floodplains had the highest proportion of wetland species followed by terraces; and both had higher proportions than stream and valley wall landforms (**Figure 6**, **Table 5**). Landform type was the only significant predictor of upland species proportion, with terraces having higher proportions than other landforms (conditional R^2=0.324, p<0.001, **Figure 7**, **Table 5**).

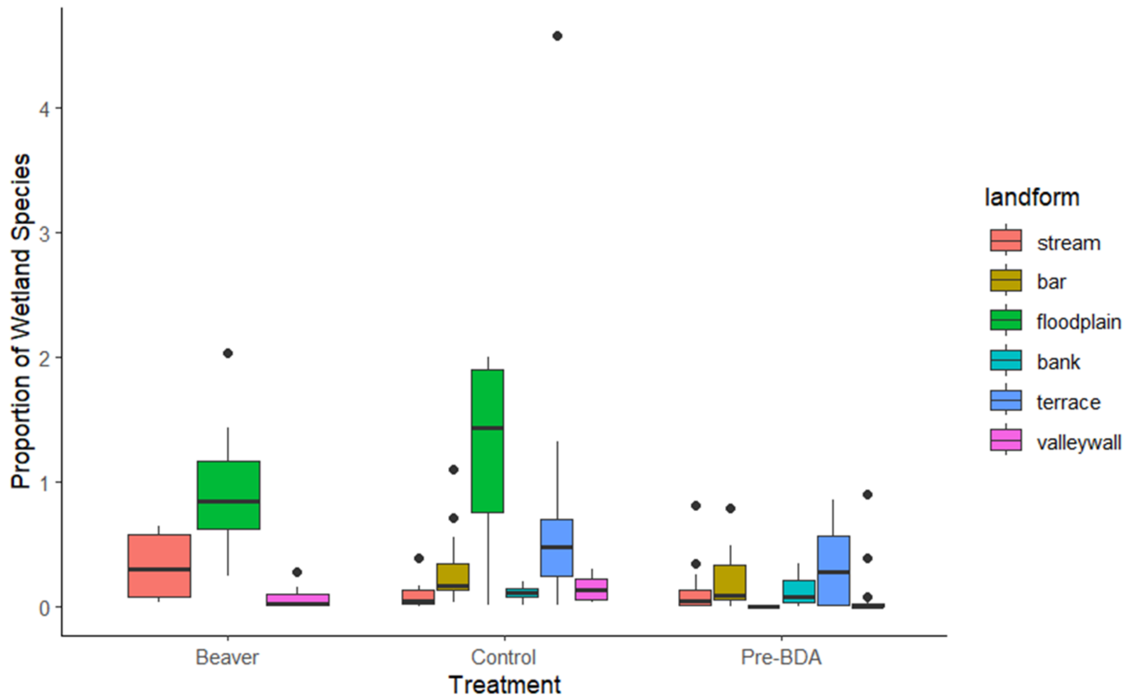

Figure 6. Effect of treatment (beaver, control, and pre-BDA) and landform type on the proportion of wetland species in headwater tributaries in the Methow and Okanogan watersheds, WA, USA. The proportion represents the total length of wetland species crossed by a transect divided by the transect length. Wetland species include species with a facultative, facultative wetland, or obligate wetland (FAC, FACW, and OBL) USDA wetland indicator status.

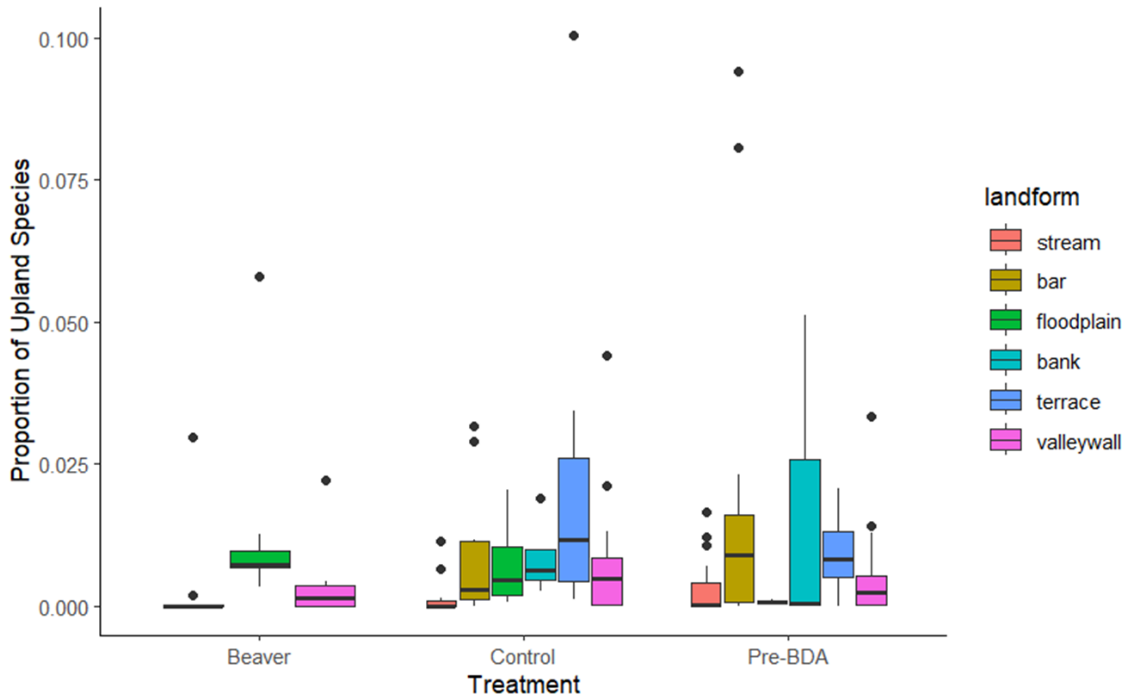

Figure 7. Effect of treatment (beaver, control, and pre-BDA) and landform type on the proportion of upland species in headwater tributaries in the Methow and Okanogan watersheds, WA, USA. The proportion represents the total length of upland species crossed by a transect divided by the transect length. Upland species include species with a facultative upland and obligate upland species (FACU and UPL) USDA wetland indicator status.

Plant species composition differed significantly across all treatments (p=0.032, **Figure 8**, **Table 7**). Of 162 total species, 22 were indicator species for beaver sites, 5 were indicator species for control sites, and 8 were indicator species for pre-BDA sites (**Table 8**). The obligate wetland species, *Typha latifolia*, *Carex simulata*, and *Lemna minor*, were strong indicators of the beaver dam sites, along with the facultative or facultative wetland species, *Ribes hudsonianum*, *Asclepias speciosa*, *Sonchus asper*, *Betula occidentalis*, *Leymus cinereus*, and *Symphyotrichum ericoides* var. *pansum*. Control sites were indicated by *Alnus viridis* ssp. *sinuate*, a facultative wetland species, and *Populus tremuloides* and *Acer glabrum* var. *douglasii*, both facultative upland species. Pre-BDA sites were indicated by the facultative upland species, *Solidago lepida*, *Pinus ponderosa*, and *Poa secunda* ssp. *Secunda*, and *Poa pratensis*, a facultative species.

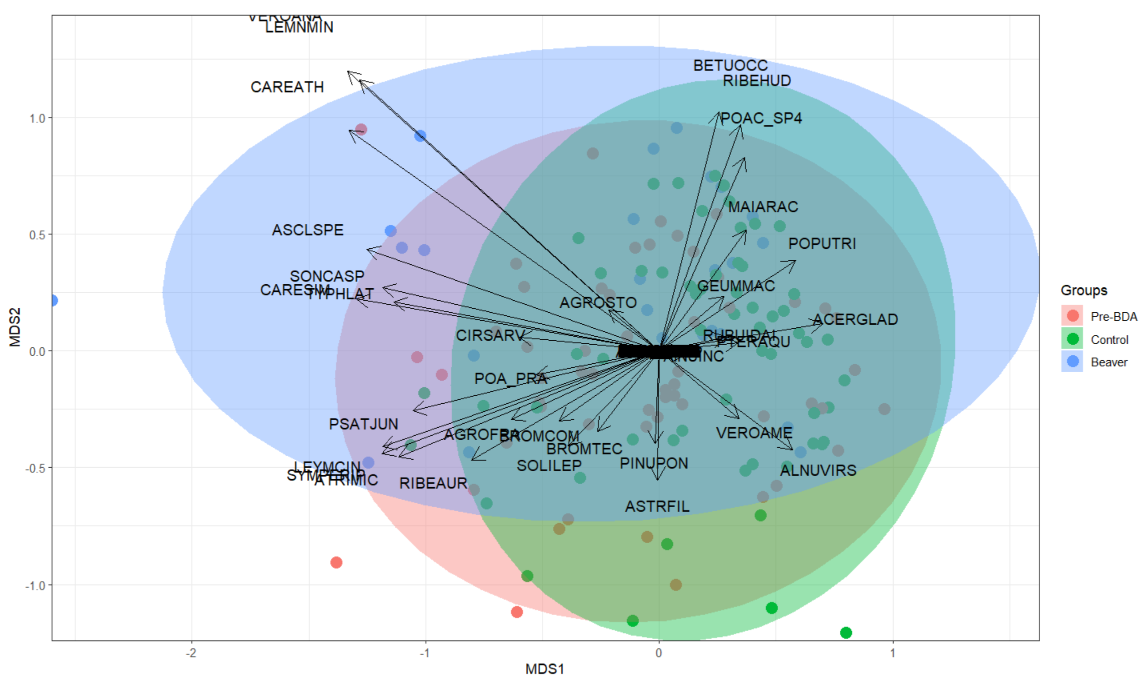

Figure 8. NMDS ordination assessing patterns in vegetation species composition associated with treatment type in the Methow and Okanogan watersheds, WA, USA. The vectors represent correlations between abundance and ordination axes of 35 indicator species (out of 162 total species) named using EWU species codes.

Sediment Composition and Transport,

Beaver dam complexes had greater sediment retention, and a higher proportion of fine sediment, whereas non-beaver sites had a wider distribution of sediment sizes ($p < 0.001$, **Figure 9**). There was little to no sediment retention throughout August 2021-August 2022 across all sites. Additionally, there was no difference among treatments in sediment transport showing that suspended sediment (turbidity) was not retained more efficiently by beaver sites, however month and the interaction between treatment and month were strong predictors of turbidity levels (**Table 5**, **Figure 10**). However, there was not a consistent change over time across the treatments. In beaver sites, there was a large increase in sediment transport in July during the collapse of the Beaver 3 complex.

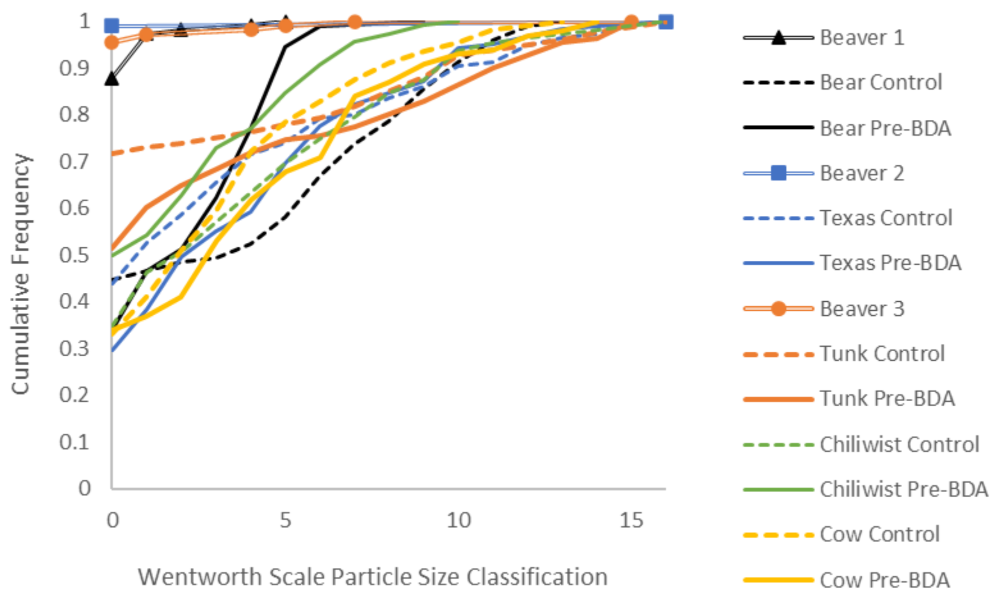

Figure 9. Cumulation frequency graph of sediment sizes (using the Wentworth Scale) from incised bank to incised bank pebble counts along transects in the headwater tributaries of the Methow and Okanogan watersheds, WA, USA. The line shapes indicate treatment (beaver, control, pre-BDA) and the color indicates matched sites.

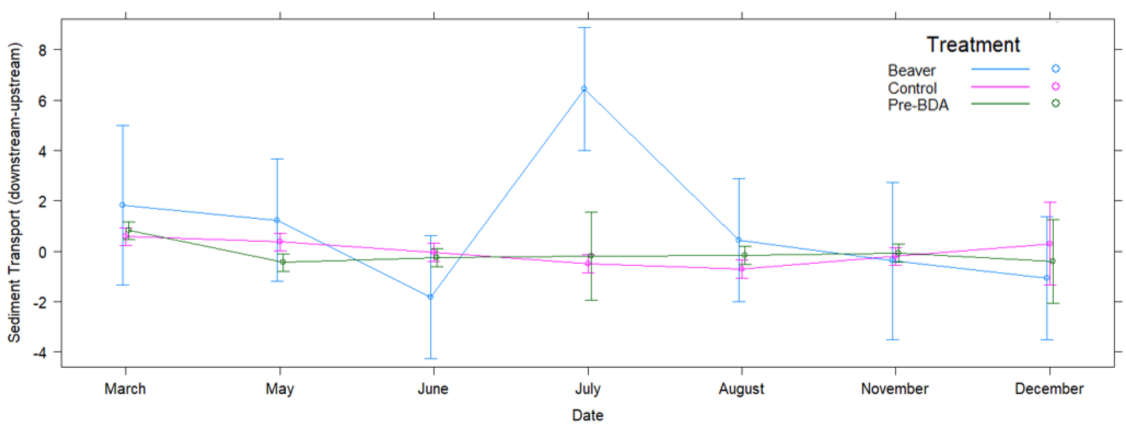

Figure 10 Effect of treatment (beaver, control, and pre-BDA) and month on sediment transport (average turbidity (NTU) at downstream location minus average turbidity at the upstream location) from August 2021- August 2022 in the Methow and Okanogan watersheds, WA, USA. The line color represents the treatment groups.

Water Quality, and Water Retention

Pre-BDA and control sites during the low flow of 2021 (July-September) had mean water travel times of 51 and 58 minutes respectively, whereas beaver sites were 7x to >400x slower (ANOVA, df = 2, p= 0.045, **Figure 11**). Discharge and the measurements taken from the YSI are displayed in **Table 11** and **Figures 12** to **15**. There were no consistent water quality patterns across treatments. Some outlier points on **Figures 12** to **15** represent the degradation of the Beaver 2 pond and the collapse of the Beaver 3 complex.

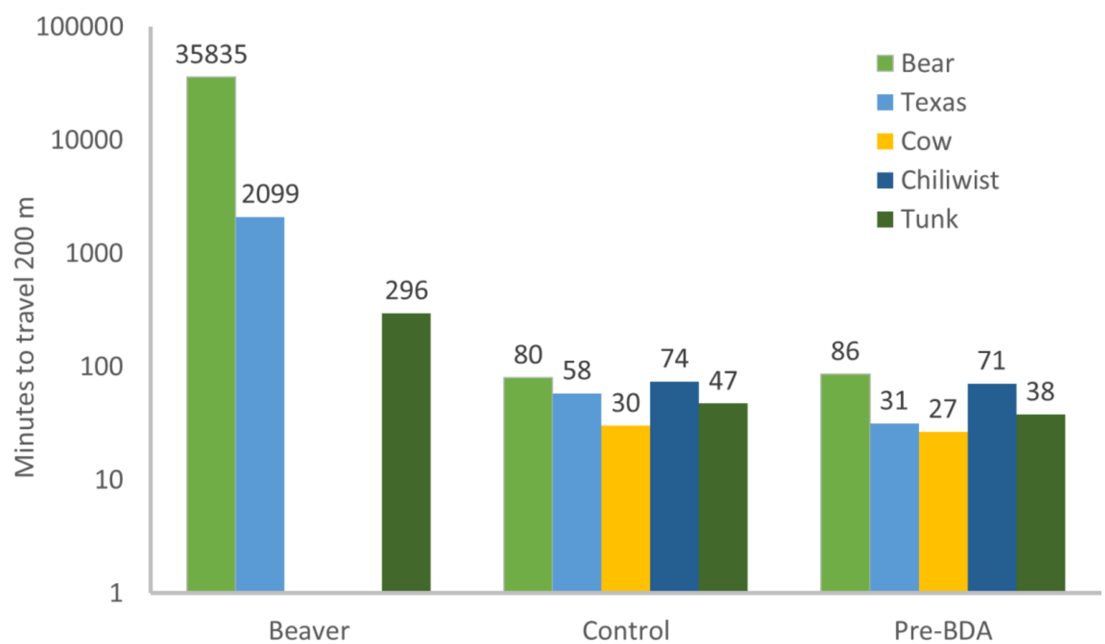

Figure 11. Effect of treatment (beaver, control, and pre-BDA) on the water travel times for 200 m of headwater tributaries in the Methow and Okanogan watersheds, WA, USA. Colors indicate matched sites.

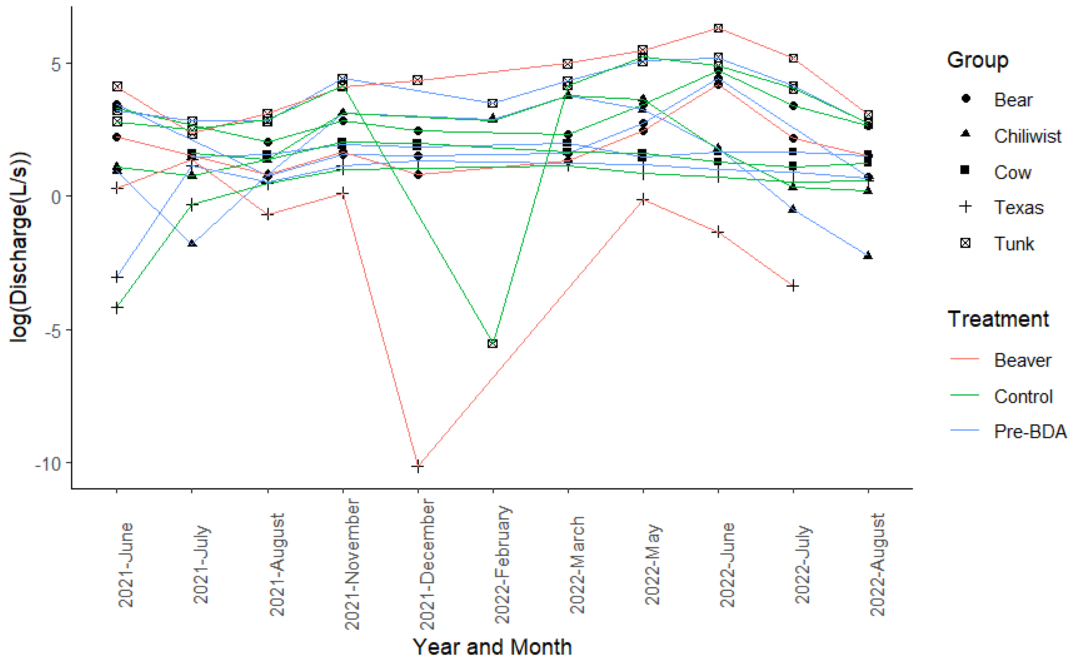

Figure 12. The discharge (L/s and log-transformed) of headwater tributaries of the Methow and Okanogan watersheds, WA, USA from June 2021 to August 2022. The line colors indicate treatment (beaver, control, and pre-BDA). Line shapes indicate matched sites.

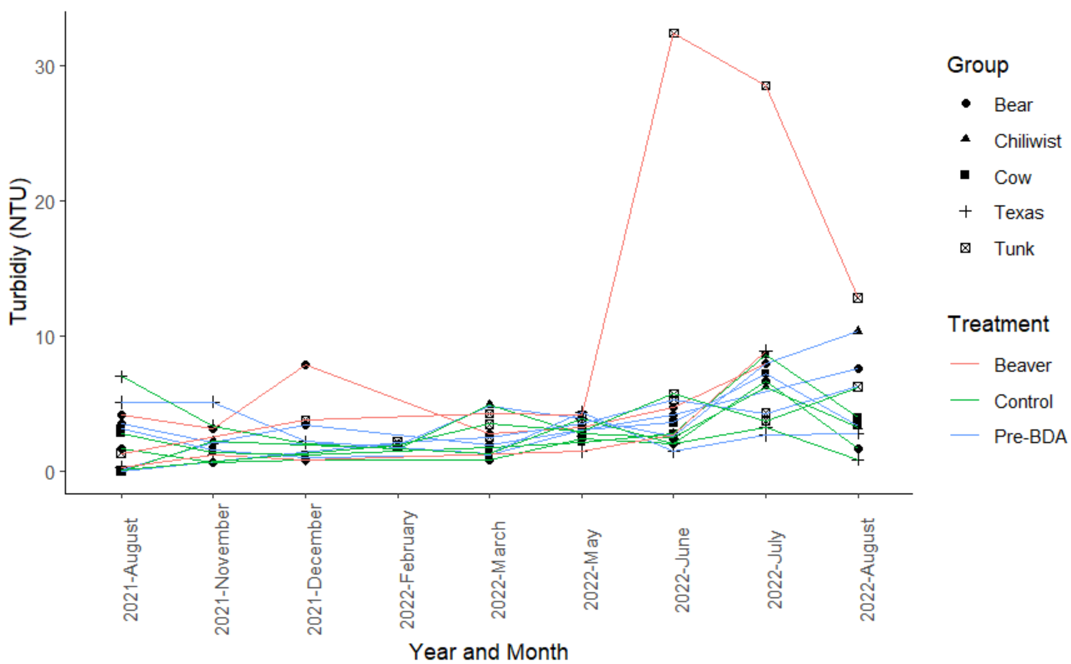

Figure 13. The turbidity (NTU) of headwater tributaries of the Methow and Okanogan watersheds, WA, USA from June 2021 to August 2022. Line color indicates treatment (beaver, control, and pre-BDA). Line shapes indicate matched sites.

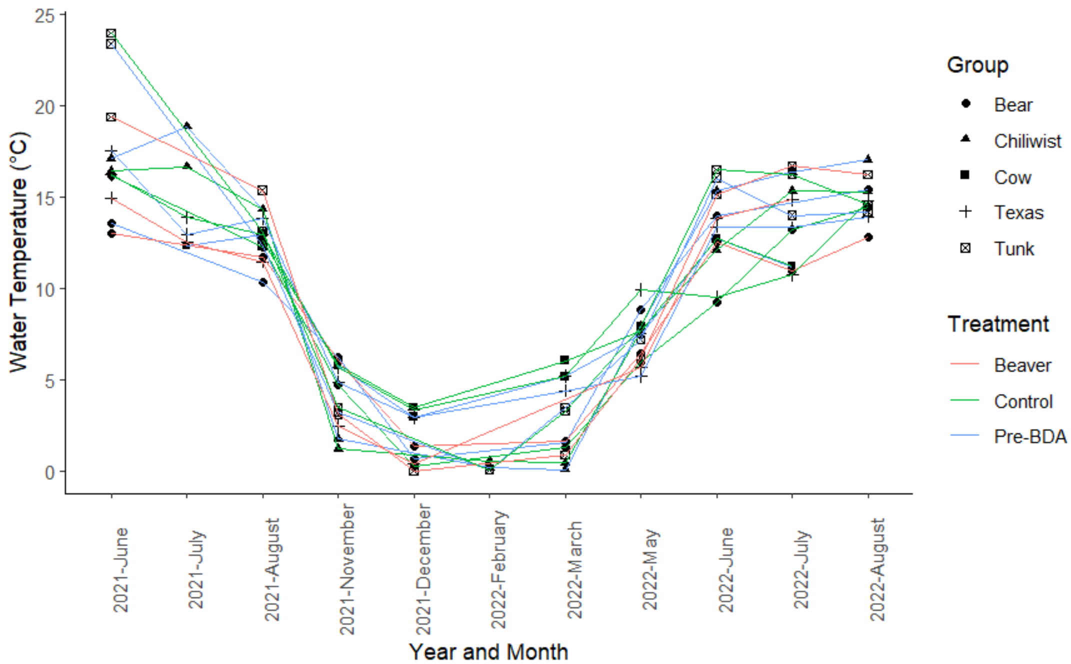

Figure 14. The water temperature (°C) of headwater tributaries of the Methow and Okanogan watersheds, WA, USA from June 2021 to August 2022. Line color indicates treatment (beaver, control, and pre-BDA). Line shapes indicate matched sites.

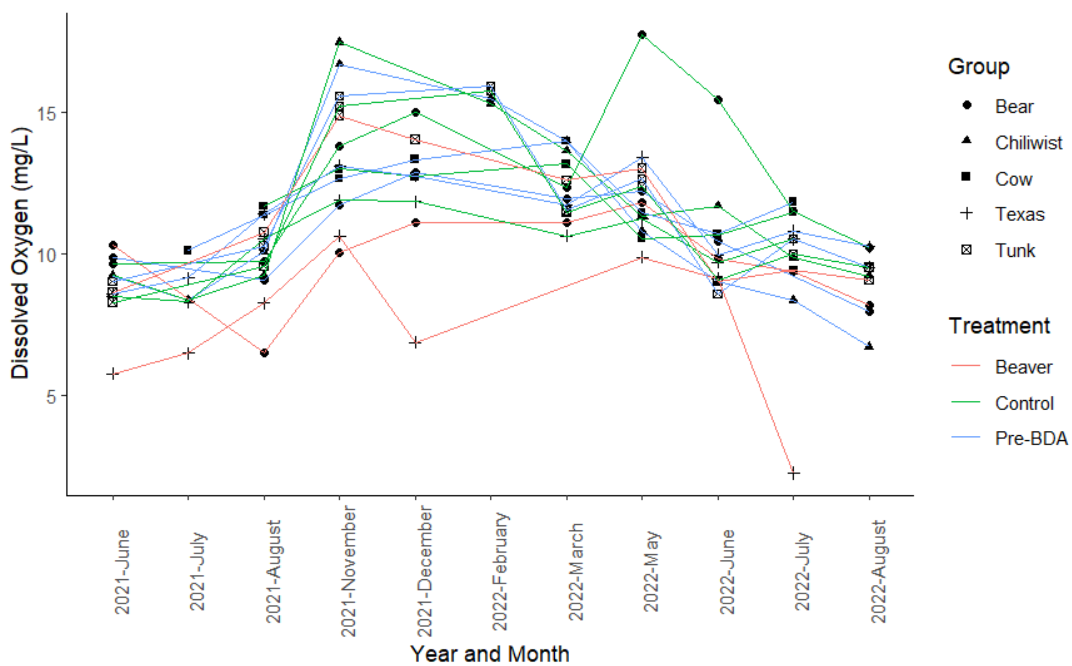

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Figure 15. The dissolved oxygen (mg/L) of headwater tributaries of the Methow and Okanogan watersheds, WA, USA from June 2021 to August 2022. Line color indicates treatment (beaver, control, and pre-BDA). Line shapes indicate matched sites.

Soil Moisture

Beaver dam complexes had higher soil moisture than the other treatments in every month (**Figure 16**, **Table 5**). Control, BDA, and pre-BDA sites had significantly decreasing soil moisture as the summer went on, whereas the beaver sites maintained high soil moisture throughout the summer (**Figure 16**, **Table 5**). There was no difference between the control and pre-BDA sites (conditional R^2= 0.358, p=0.074, **Figure 17**, **Table 5**). Texas Creek was the only stream to have its BDA complex built by the summer of 2022. Its soil moisture decreased as the summer went on, in a similar way to the Texas control site (conditional R^2= 0.774, p=0.102, **Figure 18**, **Table 5**).

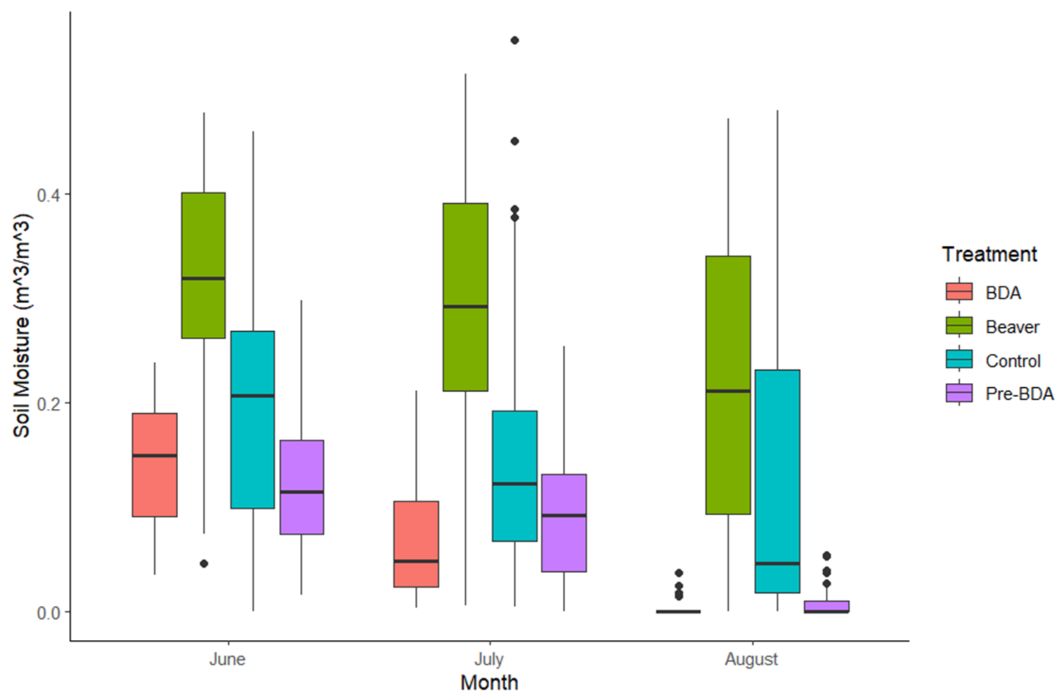

Figure 16. Volumetric soil moisture (m3/m3) content in the historic floodplains of the pre-BDA, control, and beaver sites including the recently built Texas Creek BDA site throughout the 2022 summer months in the headwater tributaries of the Methow and Okanogan watersheds, WA, USA.

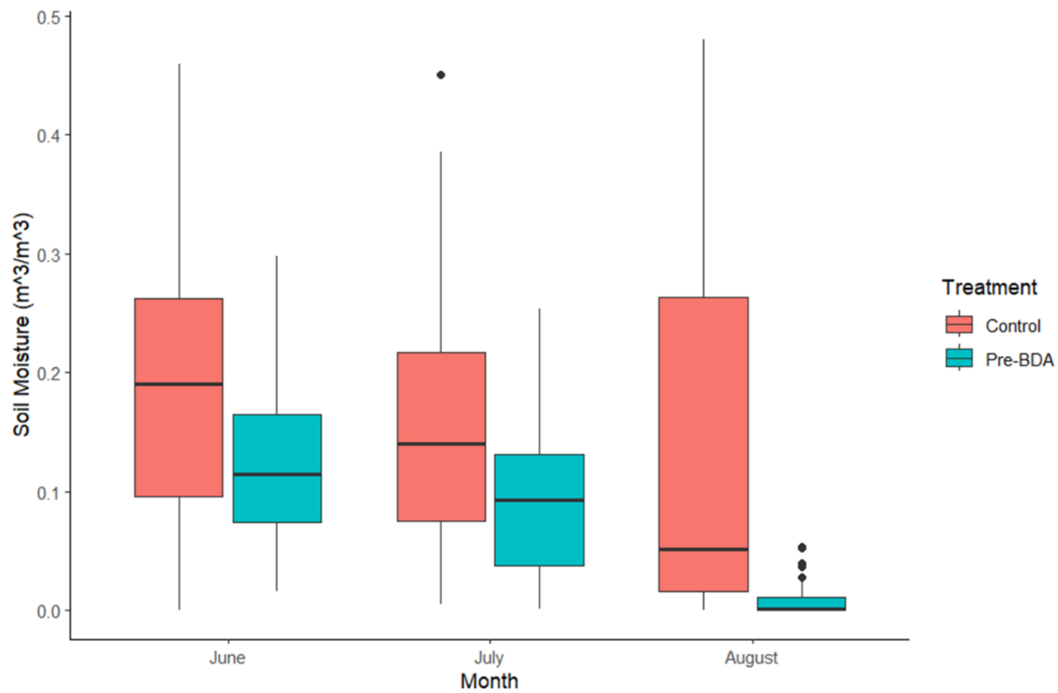

Figure 17. Volumetric soil moisture (m3/m3) content in the historic floodplains of the pre-BDA and control sites throughout the 2022 summer months in the headwater tributaries of the Methow and Okanogan watersheds, WA, USA.

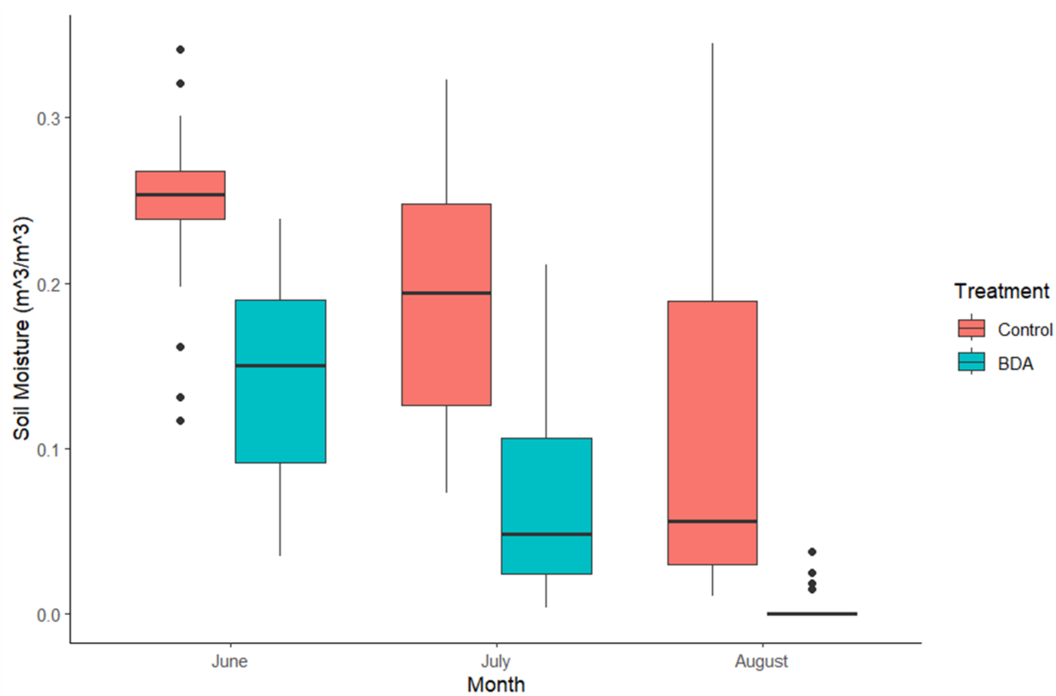

Figure 18. Volumetric soil moisture (m3/m3) content in the historic floodplains of Texas Creek's control and BDA sites throughout the 2022 summer months in the headwater tributary of the Methow watershed, WA, USA.

DISCUSSION

My study provides baseline data for a large-scale, multi-watershed effort assessing BDA effectiveness compared to three paired natural beaver dams and unrestored control reaches with a BACRI study design. Although it is too soon to tell the effects of BDAs, my results provide clear evidence that beaver dam complexes improve riparian and stream ecosystem function in the Methow and Okanogan watersheds. As predicted, beaver dam sites had higher width-to-depth ratios, floodplain widths, proportions of wetland species, fine sediment retention, and soil moisture through the summer than control and pre-BDA sites. The record hot and dry conditions throughout the summer of 2021 may have influenced my study. The benefits of increased water storage and soil moisture in beaver complexes will become increasingly important as droughts and high-intensity wildfires increase with climate change (Beechie et al. 2012, Whipple 2019, Bowman et al. 2020, Weirich 2021). However, my study also highlights the need to protect and change policies related to beavers, as two of my three beaver dam complex sites were lost by the end of the study due to human impacts: overgrazing or intentional beaver removal.

The increased width-to-depth-ratios and floodplain widths of beaver dam complexes provide strong evidence that beaver ponds widen incision trenches and reconnect streams to their historic floodplains, thus improving channel incision (Pollock et al. 2014), consistent with other studies (Pollock et al. 2007, Curran and Cannatelli 2014, Grudzinski et al. 2022). Side channels that formed around the Beaver 3 site widened the floodplain further and allowed for the establishment of riparian species (Pollock et al. 2014). My results are consistent with Whipple (2019) who found that beaver complexes in the Methow watershed had higher width-to-depth ratios than non-dammed sites. The beaver dams in that study were all built within four years of a major fire and helped restore the incised channels by widening and aggrading.

Unexpectedly, beaver dam sites were not found to influence the proportion of non-facultative upland woody riparian species, although they had a higher proportion of wetland species than the pre-BDA sites. Many of the deciduous riparian trees in this study were facultative upland species, however in arid ecosystems, riparian zones are one of the few places deciduous trees are found (USACE 2010), including those with facultative upland status, so the effect of beaver complexes on woody riparian species may be higher than shown. My findings are consistent with other studies that show that beaver dam complexes create important habitat for wetland species (McMaster and McMaster 2001), and beaver sites create unique community assemblages within complexes (Willby et al. 2018).

Similarly, non-beaver sites did not have a higher proportion of facultative upland species, as I had expected. This may be due to the large number of species in my dataset that appear to be upland species but were not classified by USDA (2021). Most indicator species in the beaver sites were classified as facultative wetland or obligate wetland species by the USDA (2021), whereas control and pre-BDA sites had more indicator species that were facultative upland or unclassified.

Incised streams can create large downstream inputs of sediment due to unstable banks (Pollock et al. 2014), while beaver dams accumulate fine sediment (Naiman et al. 1986, Pollock et al. 2014, Brazier et al. 2021), as I found in my study. However, unexpectedly, there were no significant differences among treatments with respect to their effects on downstream-upstream suspended sediments, suggesting that suspended sediment was transported similarly across all treatment groups (beaver, control, and pre-BDA). The amount of suspended sediment was not influenced by discharge, indicating that flooding was not a major contributor of sediment into the streams in the summer of 2021. The abundant fine sediment accumulated within complexes shown in the sediment composition data most likely built up in the past during previous fires or large floods when there were large sediment inputs. Large sediment inputs from flooding may have been absent in summer 2021 due to the below average discharge throughout the summer. Since the beaver sites were no longer retaining sediment more efficiently than non-dammed sites, this indicates that the beaver dam sites had enough time to recover from the large sediment inputs from the past. If the dams are removed or breached, like the collapsing of Beaver 3, this can lead to sediment lifting off the stream bed and transporting downstream (Błędzki et al. 2011). Maintaining functioning beaver dam complexes is important to prevent downstream sediment transport.

The water travel times of beaver sites were drastically higher than those of non-beaver sites, showing the critical role that beaver can play in increasing transient water storage within watersheds (Jin et al. 2009, Janzen and Westbrook 2011). My result is similar to previous studies that found beaver dam complexes increased travel times by accumulating sediment and creating areas of dead zones thus increasing transient water storage (Jin et al. 2009, Majerova et al. 2015). Transient storage describes temporary hydraulic storage through slow-moving parcels of water associated with hyporheic storage, vegetation, and hydraulic obstacles such as boulders or wood that create dead zones (Bencala and Walters 1983, D'Angelo et al. 1993). The longer water is stored, the higher potential it has to recharge groundwater (Westbrook et al. 2006). The beaver dam sites also had wetland vegetation like *Typha latifolia* in the ponds that may have reduced stream power and increased water travel times by obstructing flow (Ensign and Doyle 2005). In arid climates increased water storage will be important to increase stream resilience as reduced snowpack, droughts, and wildfires are increasing (Dierauer et al. 2019).

Beaver dam complexes increased resilience in the riparian zone by maintaining high levels of soil moisture in the floodplain throughout the summer. This was consistent with the findings of Weirich (2021) who showed increased levels of soil moisture in beaver dam complexes compared to non-dammed streams. Additionally, the soil moisture remained at higher levels at further distances from the stream in comparison to non-dammed sites. The wider wetted width of beaver ponds most likely contributed to the reduced burn severity found in beaver complexes compared to non-dammed sites. The differences in fire burn potential are exacerbated during times of drought. With climate change beaver complexes may become an increasingly important refugia habitat.

Only one BDA complex was constructed by the conclusion of my study on Texas Creek, and I assessed it one month after construction. This may not have been enough time to assess results, as no high flow, sediment transport events had occurred since dam construction. Nevertheless, immediately after construction, water pooled, and sediment collected directly above the BDAs.

The BDAs in this study are expected to initiate recovery in incised streams similar to previous studies by widening the incision trench, accumulating fine sediment, increasing the proportion of wetland vegetation, increasing water storage, and raising water tables. BDAs have already been shown to be effective at mimicking beaver dam complexes by reducing incision (Scamardo and Wohl 2020), and it is expected that the BDA sites in my study will begin to recover from incision rather than incise further. The BDA complexes in this study are expected to increase riparian vegetation productivity over time similar to the BDA complexes on Bridge Creek, OR (Silverman et al. 2019). However, BDAs are not expected to precisely imitate beaver dam complexes as beaver herbivory can cause large shifts in plant community structure and reduce the number of aquatic invasives (Parker et al. 2007). BDAs have been effective in past studies by aggregating sediment (Scamardo and Wohl 2020) and over time this study expects to see similar results. With more time, water is expected to spread into the floodplain, and it is predicted that the soil moisture on the historic floodplains in the BDA sites will begin to increase. The BDA sites in this study are expected to create flow reversals from stream to floodplain (Pearce et al. 2021a), to increase stream surface area above the structures, and to raise water levels similar to the restoration project on Alkali, and Robb creeks (Vanderhoof and Burt 2018), and increase groundwater levels similar to the South Fork (Orr et al. 2020). Results may be limited in Chiliwist Creek's restoration site as it is an intermittent stream, but there may be longer running periods as water storage is increased within the complex.

A limitation of this study was that the control sites were often in better condition than the pre-BDA sites. This may be because the BDA sites were established in areas where the biggest stream quality improvement was expected, many with severe wildfire degradation, and control sites were always upstream of the pre-BDA sites. As the most severely degraded locations were selected for BDAs, the control sites ended up with lower incision and slightly improved stream quality. At Bear Creek, the control site had higher woody vegetation cover, higher soil moisture, and lower incision than the planned restoration site downstream. Control sites may have had smaller width-to-depth ratios because many of them had larger slopes (**Table 4**) and incision depth tends to increase with decreasing slope (Beechie et al. 2008). Control sites had a higher proportion of non-facultative upland woody riparian and wetland species than the pre-BDA sites indicating that the control sites tended to be in a more connected condition to the stream and had different community composition than the pre-BDA sites.

It was difficult to identify accessible beaver complexes, especially in the Okanagan watershed, that met my site-matching criteria. Consequently, the sites I used were not ideal reference sites, or they were altered before the conclusion of the study. The Texas Creek reference site (Beaver 2) was on a first-order stream with a small pond and only one dam, whereas Texas Creek is a third-order stream at both site locations (control and pre-BDA), however other reference options were determined to have too high of discharge. Beaver 2's stream was primarily underground in many places in the watershed around the site. The site was densely covered in woody species, many of which were generalist and upland species. The pond was relatively new and if the beaver site had more time to develop it would be expected that more woody riparian species would grow as the conditions would become over-saturated for upland species. This site may have reduced the impact beaver sites had on the abundance of wetland species. Additionally, the Tunk Creek reference site (Beaver 3) may have contributed to the low influence beaver sites had on woody riparian species cover. Beaver 3 was established with many wetland species including cattails, sedges, and rushes. The most common cover included *Typha latifolia*, shrubs, and grasses. However, the site had minimal woody cover with woody, obligate wetland species. The site had some woody, facultative wetland species, *Alnus incana*, and *Salix amygdaloides*, and some woody, facultative species, *Ribes aureum*, *Salix scouleriana*, and *Kochia scoparia*. The site had a high cover of facultative wetland species and the highest cover of all sites of obligate wetland species. Each beaver site showed the positive influence of beavers, but together they could not tell a compelling story about the vegetation. This study would have benefited from more beaver reference sites, and in the future, more sites will need to be added with less strict site-matching criteria.

My study also illustrates the problem of continued human conflict with beavers. Often beavers are removed, and water quality and water storage are lost. Reintroduction is frequently limited or hindered by human conflict and thus we end up relying more heavily on manmade solutions. In the Beaver 3 site, the beavers were trapped in the fall of 2021 due to fear that the pond would impact a nearby bridge. By July 2022, the dams were degrading, and the pond levels were dropping dramatically. The side channels dried up and the stream started to disconnect from the floodplain. Turbidity rose from an average of 1.335 NTU in August 2021 to 28.52 NTU in July 2022. Stream power had risen causing the fine sediment that was settled at the bottom of the ponds to be transported downstream. This intentionally destroyed beaver complex was located in an arid, sagebrush steppe dominated landscape, where water storage is critical. Given the reduced predictability of water supply related to climate change, there may come a time when the lost water storage from trapping beaver exceeds that of using beaver friendly solutions to protect infrastructure.

The Beaver 2 site also lost its beaver community sometime between October 2021 and May 2022, during the 5-month, unlimited harvest season for beavers. Because I often encountered hunters, most of whom communicated negative views of beavers, I suspect that the beaver community at this site was trapped. After the beavers were gone, the pond quickly started to decline by June 2022. In July, cows moved in, heavily grazed the riparian vegetation, and trampled the pond (**Figure 20**). The few remaining pools had dissolved oxygen of 2.26% compared to 62.1% in July of the previous year. The pond and stream were fully dry by August (**Figure 21**). Once beavers were absent, the landscape dramatically changed.

Figure 19. Beaver site 2 with active beavers on July 9th, 2021, in Okanogan County, WA, USA.

Figure 20. Beaver site 2 on July 20th, 2022, after cows began to trample the pond in Okanogan County, WA, USA.

Figure 21. Beaver site 2 on August 19th, 2022, after the pond was fully trampled and dried up in Okanogan County, WA, USA.

MANAGEMENT RECOMMENDATIONS

In Okanogan County, where cattle significantly contribute to the degradation of streams and riparian zones (Wissmar 1994, Whipple 2019), limiting interactions between cattle and streams would help maintain water quality and stream ecosystem function. Overgrazing can lead to large riparian vegetation removal and trampling can increase turbidity and cause banks to collapse (Trimble and Mendel 1995). These effects were seen on the Texas Creek and Beaver 2 sites. More proactive cattle exclusion would benefit stream function and habitat quality, especially in areas of channel incision, to prevent further degradation. Complete or partial exclusion of cattle can limit the amount of contaminants, suspended sediment, and erosion that cattle cause in the riparian zone (Bremner et al. 2016, O'Callaghan et al. 2019). Allowing access to a singular part of the stream or “sacrifice” areas for drinking water or providing stock tanks can reduce the impact of cattle by reducing the interaction cattle have with the stream (Bremner et al. 2016). Bridges across the stream could encourage passage without direct contact with the water, thus decreasing disturbance and trampling-related erosion.

Land managers should prioritize providing protection for beaver communities, especially in wildlife or restoration areas. While BDAs may be an effective tool in restoring incised streams when we cannot feasibly reintroduce beavers due to site conditions or landowner priorities, they may not reach goals as efficiently or to the same degree as natural beaver complexes. BDAs may differ in porosity and hydraulic conductivity compared to natural beaver dams, which warrants further study. Beavers regularly push sediment into their dams to reduce pores, whereas BDAs are rarely built this way due to permitting restrictions. Further, beavers maintain their dams more frequently than humans can maintain BDAs. With time it is expected that BDA sites will become more similar to beaver sites if they fill with sediment during high-flow events or are adopted by reintroduced or dispersing beavers. More data is needed to clarify whether BDAs are an effective tool for reducing incision and restoring incised streams.

CONCLUSION

My project represents the first step in a large-scale, long-term study assessing BDA complexes compared to natural beaver dams across two watersheds using a BACRI study design. This study will be one of the most comprehensive assessments of BDA effectiveness to date and will help clarify their role. The baseline data I've collected shows that beaver dam complexes improve incised stream habitat by reducing incision, widening floodplains, and increasing stream sediment heterogeneity compared to both control and reference reaches. They also have higher floodplain soil moisture that is maintained throughout the summer allowing for a species composition with more facultative wetland and obligate wetland species. Yet despite the benefits that beaver provide, they are being actively removed in my study watersheds and many others throughout North America (Gibson and Olden 2014). My results are consistent with other beaver studies in the Methow watershed, including Whipple (2019), who showed that beaver increase stream resiliency to wildfire, and Weirich (2021), who showed that beaver dams increase soil moisture and resistance to severe burning in riparian zones. BDAs can help the situation but may not be as effective as natural beaver dam complexes. Until it is better known, BDAs should be looked at as temporary solutions to provide habitat for beaver reintroduction.

TABLES

Table 5. ANOVA results from all mixed model statistical analyses show the effects of the listed independent variables on dependent variables (* indicates p values < 0.05). The independent variables include treatment (beaver, pre-BDA, BDA, and control), landform type (stream, bar, bank, floodplain, terrace, and valley wall), distance from edge of bank, and month. Additionally, the results of pairwise comparisons among independent variables (Tukey post-hoc tests) for the mixed models in. Only significant (p<0.05) pairwise comparisons are shown.

ANOVA	**Factor**	**Chisq**	**df**	**p**	**Pairwise Comparison**	**Estimate**	**df**	**Adjusted p**
Topography								
Width:Depth	Treatment	32.528	2	**<0.001**				
					Beaver-Control	0.75	30	<0.001
					Beaver-Pre-BDA	1.18	30	<0.001
Floodplain Width	Treatment	24.010	2	**<0.001**				
					Beaver-Control	0.66	30	0.004
					Beaver-Pre-BDA	0.94	30	<0.001
Vegetation								
Proportion of Wetland Species	Treatment	12.587	2	**0.002**				
					Beaver-Pre-BDA	0.37	138	0.019
					Control-Pre-BDA	0.32	138	0.005
	Landform	39.531	5	**<0.001**				
					Floodplain-Stream	0.69	138	<0.001
					Floodplain-Valley wall	0.83	138	<0.001
					Stream-Terrace	-0.51	138	0.005
					Terrace-Valley wall	0.65	138	<0.001
Proportion of Upland Species	Treatment	0.825	2	0.662				
	Landform	51.955	5	**<0.001**				
					Bar-Terrace	-0.33	138	0.012

ANOVA	Factor	Chisq	df	p	Pairwise Comparison	Estimate	df	Adjusted p
					Floodplain-Stream	0.45	138	0.002
					Floodplain-Valley wall	0.35	138	0.031
					Stream-Terrace	-0.56	138	<0.001
					Terrace-Valley wall	0.47	138	<0.001
Proportion of Woody Riparian Species	Treatment	16.212	2	**<0.001**				
					Control-Pre-BDA	0.19	138	<0.001
	Landform	26.143	5	**<0.001**				
					Terrace-Valley wall	0.26	138	0.001
					Floodplain-Valley wall	0.24	138	0.050
					Stream-Terrace	-0.23	138	0.005
Soil Moisture								
Full model	Distance	2.799	1	0.094				
	Type	33.429	3	**<0.001**				
	Month	126.596	2	**<0.001**				
	Distance:Type:Month	24.664	6	**<0.001**				
	Distance:Type	10.705	3	**0.013**				
	Distance:Month	2.812	2	0.245				
	Type:Month	114.666	6	**<0.001**				
					June:			
					BDA-Beaver	-1.54	1481	<0.001
					BDA-Control	-1.18	1481	<0.001
					BDA-Pre-BDA	-1.15	1481	<0.001
					Beaver-Control	0.36	1481	<0.001
					Beaver-Pre-BDA	0.39	1481	0.001
					July:			

ANOVA	Factor	Chisq	df	p	Pairwise Comparison	Estimate	df	Adjusted p
					Beaver-Control	0.58	1481	<0.001
					Beaver-Pre-BDA	0.82	1481	<0.001
					Control- Pre-BDA	0.24	1481	0.043
					August:			
					BDA-Beaver	-3.96	1481	<0.001
					BDA-Control	-3.15	1481	0.013
					Beaver-Control	0.81	1481	<0.001
					Beaver-Pre-BDA	1.94	1481	<0.001
					Control- Pre-BDA	1.13	1481	<0.001
Control and Pre-BDA	Distance	0.254	1	0.614				
	Month	109.636	2	**<0.001**				
	Type	3.182	1	0.074				
	Distance:Month:Type	16.575	2	**<0.001**				
	Distance:Month	1.867	2	0.393				
	Distance:Type	7.762	1	**0.005**				
	Month:Type	35.799	2	**<0.001**				
					July:			
					Control- Pre-BDA	0.32	880	<0.001
					August:			
					Control- Pre-BDA	1.18	880	<0.001
Texas BDA and Control	Distance	2.200	1	0.138				
	Type	2.674	1	0.102				
	Month	99.522	2	**<0.001**				
	Distance:Type:Month	2.457	2	0.2927				
	Distance:Type	2.454	1	0.117				
	Distance:Month	2.211	2	0.331				
	Type:Month	29.599	2	**<0.001**				

ANOVA	Factor	Chisq	df	p	Pairwise Comparison	Estimate	df	Adjusted p
					June:			
					BDA-Control	-0.58	163	<0.001
					July:			
					BDA-Control	-0.99	163	<0.001
					August:			
					BDA-Control	-2.07	163	<0.001
Sediment								
Sediment Transport	Month	17.147	6	**<0.01**				
	Treatment	3.100	2	0.212				
	Month:Treatment	79.316	12	**0.009**				
Sediment Composition	Treatment	256.600	2	**<0.001**				
					Beaver-Pre-BDA	391.757	2	<0.05
					Beaver-Control	385.63	2	<0.05

Table 6. Vascular plant species richness across sites and transects in the Methow and Okanagan watersheds in Washington, USA. Shading indicates matched sites.

Watershed	Site	Transect 25%	Transect 50%	Transect 75%	Average
Methow	Bear BDA	43	31	29	34.3
	Bear Beaver	17	8	27	17.3
	Bear Control	9	18	17	14.7
	Benson Beaver	16	15	24	18.3
	Texas Control	11	8	16	11.7
	Texas BDA	13	12	14	13.0
	Cow BDA	12	15	16	14.3
	Cow Control	9	16	15	13.3
Okanogan	Chiliwist BDA	21	7	14	14.0
	Chiliwist Control	15	12	16	14.3
	Bonaparte Beaver	20	24	27	23.7
	Tunk BDA	27	22	31	26.7
	Tunk Control	23	24	28	25.0

Table 7. Results of PERMANOVA analysis comparing plant species composition across treatments.

pairs	Sum of Squares	F value	R^2	P value
Pre-BDA vs. Beaver	0.851	1.991	0.025	0.009
Pre-BDA vs. Control	1.107	2.624	0.023	0.003
Beaver vs. Control	0.872	2.078	0.025	0.003

Table 8. Results of indicator species analysis for plant species composition. The species are listed under what treatment group they indicate. The USDA wetland indicator status is given for each species.

Beaver

Species	stat	p.value	Wetland Status
Typha latifolia	0.529	0.005	OBL
Betula occidentalis	0.529	0.005	FACW
Agrostis stolonifera	0.499	0.030	FACW
Maianthemum racemosum	0.497	0.005	FAC
Ribes hudsonianum	0.490	0.005	FACW
Symphyotrichum ericoides var. *pansum*	0.447	0.005	FAC
Carex simulata	0.400	0.005	OBL
Leymus cinereus	0.394	0.005	FAC
Geum macrophyllum	0.393	0.005	FACW
Lemna minor	0.371	0.005	OBL
Veronica americana	0.346	0.010	OBL
Pteridium aquilinum	0.346	0.010	FACU
Rubus idaeus var. *idaeus*	0.345	0.015	FACU
Ribes aureum	0.312	0.020	FAC
Asclepias speciosa	0.283	0.050	FAC
Atriplex micrantha	0.283	0.040	NA
Carex athrostachya	0.283	0.035	FACW
Sonchus asper	0.283	0.050	FAC
Veronica anagallis-aquatica	0.283	0.035	OBL
Psathyrostachys juncea	0.283	0.045	UPL
Lepidium chalepensis	0.280	0.045	NA

Control

Species	stat	p.value	Wetland Status
Alnus viridis ssp. *sinuata*	0.432	0.005	FACW
Populus tremuloides	0.407	0.015	FACU
Acer glabrum var. *douglasii*	0.372	0.020	FACU
Poaceae sp.4	0.368	0.025	NA

Pre-BDA

Species	stat	p.value	Wetland Status
Bromus tectorum	0.473	0.015	NA
Bromus commutatus	0.410	0.005	NA

Solidago lepida	0.396	0.025	FACU
Pinus ponderosa	0.357	0.025	FACU
Astragalus filipes	0.302	0.045	NA
Poa pratensis	0.302	0.030	FAC
Poa secunda ssp. *secunda*	0.270	0.050	FACU

Beaver+Control

Species	**stat**	**p.value**	**Wetland Status**
Alnus incana	0.512	0.010	FACW

Beaver+Pre-BDA

Species	**stat**	**p.value**	**Wetland Status**
Agropyron fragile	0.461	0.010	NA

Control+Pre-BDA

Species	**stat**	**p.value**	**Wetland Status**
Clematis ligusticifolia	0.459	0.040	FAC

Table 9. Definitions for the wetland indicator statuses (USDA 2021).

Code	Status Designation	Description
OBL	Obligate Wetland Hydrophyte	99% of the time occurs in wetlands
FACW	Facultative Wetland Hydrophyte	67-99% of the time occur in wetlands
FAC	Facultative Hydrophyte	34-66% of the time occurs in wetlands
FACU	Facultative Upland Non-hydrophyte	1-33% of the time occurs in wetlands
UPL	Obligate Upland Non-hydrophyte	1% of the time occurs in wetlands

Table 10. Plant species list for all study sites in Okanagan County, WA, USA, including their native status and their wetland indicator status (USDA 2021). For plants with both native and non-native USDA status in Washington, the Burke Herbarium (2021) was used to distinguish the status in Okanogan County. Plants that are woody and had a FAC, FACW, or OBL wetland status were classified as 'woody riparian'.

Species name	**Common name**	**Native Status**	**Wetland Status**	**Woody Riparian**
Acer glabrum var. *douglasii*	Douglas' maple	native	FACU	
Achillea millefolium	common yarrow	both	FACU	
Achnatherum occidentale ssp. *pubescens*	pubescent western needlegrass	native		
Agastache urticifolia	nettle leaf giant hyssop	native	FACU	
Agropyron cristatum	crested wheatgrass	invasive		
Agropyron fragile	Siberian wheatgrass	invasive		
Agrostis stolonifera	creeping bentgrass	invasive	FACW	
Alnus incana	gray alder	native	FACW	yes
Alnus viridis ssp. *sinuata*	Sitka alder	native	FACW	yes
Amelanchier alnifolia	saskatoon serviceberry	native	FACU	
Anaphalis margaritacea	western pearly everlasting	native	FACU	
Apera interrupta	dense silkybent	invasive		
Apocynum cannabinum	Indian hemp	native	FAC	yes
Arctium minus	lesser burdock	invasive	FACU	
Artemisia ludoviciana ssp. *ludoviciana*	white sagebrush	native	FACU	
Asclepias speciosa	showy milkweed	native	FAC	
Asteraceae sp. 1	Asteraceae species	NA		
Astragalus filipes	basalt milkvetch	native		
Atriplex micrantha	twoscale saltbush	invasive		
Berberis aquifolium	Oregon-grape	native	UPL	
Betula occidentalis	water birch	native	FACW	yes
Bromus commutatus	meadow brome	invasive		
Bromus tectorum	cheatgrass	invasive		
Bromus ciliatus	fringed brome	native	FAC	
Capsella bursa-pastoris	shepherd's purse	invasive	FACU	
Cardamine pensylvanica	Pennsylvania bittercress	native	FACW	
Carex athrostachya	slenderbeak sedge	native	FACW	
Carex praegracilis	clustered field sedge	native	FACW	

Carex scoparia	broom sedge	native	FACW	
Carex simulata	analogue sedge	native	OBL	
Carex utriculata	northwest territory sedge	native	OBL	
Centaurea diffusa	diffuse knapweed	invasive		
Centaurea stoebe	spotted knapweed	invasive		
Chamerion angustifolium	fireweed	native	FACU	
Chrysothamnus viscidiflorus ssp. *lanceolatus*	yellow rabbitbrush	native		
Circaea alpina	small enchanter's nightshade	native	FAC	
Cirsium arvense	Canada thistle	invasive	FACU	
Cirsium vulgare	bull thistle	invasive	FACU	
Clematis ligusticifolia	western white clematis	native	FAC	yes
Collomia heterophylla	variable leaf collomia	native		
Cornus sericea	red osier dogwood	native	FACW	yes
Crataegus macracantha	large-thorn hawthorn	native		
Echinochloa crus-galli	barnyard grass	invasive	FACW	
Elodea canadensis	Canadian waterweed	native	OBL	
Elymus glaucus	blue wildrye	native	FACU	
Elymus glaucus ssp. *glaucus*	blue wildrye	native	FACU	
Elymus repens	quackgrass	invasive	FAC	
Epilobium ciliatum	fringed willowherb	native	FACW	
Epilobium glaberrimum	glaucus willowherb	native	FACW	
Epilobium obscurum	dwarf willowherb	invasive	FACW	
Equisetum arvense	field horsetail	native	FAC	
Equisetum hyemale	rough horsetail	native	FACW	
Ericameria nauseosa ssp. *nauseosa*	rubber rabbitbrush	native		
Erigeron divergens	spreading fleabane	native		
Erigeron philadelphicus	Philadelphia fleabane	native	FACU	
Erigeron subtrinervis	three nerve fleabane	native		
Eriogonum niveum	snow buckwheat	native		
Eurybia conspicua	western showy aster	native		
Festuca idahoensis	Idaho fescue	native	FACU	
Galium aparine	sticky willy	native	FACU	
Galium boreale	Northern bedstraw	native	FACU	
Galium triflorum	fragrant bedstraw	native	FACU	
Geum macrophyllum	largeleaf avens	native	FACW	

Glyceria elata	tall mannagrass	native	OBL	
Glyceria striata	fowl mannagrass	native	OBL	
Gnaphalium palustre	western marsh cudweed	native	FACW	
Gypsophila paniculata	baby's breath	invasive		
Hesperostipa comata ssp. *intermedia*	intermediate needle and thread	native		
Hieracium albiflorum	white-flowered hawkweed	native		
Iliamna rivularis	streambank wild hollyhock	native	FACW	yes
Ipomopsis aggregata	scarlet gilia	native		
Juncus balticus	baltic rush	native	FACW	
Juncus effusus ssp. *effusus*	common rush	native	FACW	
Juncus ensifolius	swordleaf rush	native	FACW	
Juncus parryi	Parry's rush	native	FAC	
Juncus saximontanus	rocky mountain rush	native	FACW	
Kochia scoparia	burning bush	invasive	FAC	yes
Lactuca biennis	tall blue lettuce	native	FAC	
Lactuca serriola	prickly lettuce	invasive	FACU	
Lemna minor	common duckweed	native	OBL	
Lepidium draba	whitetop, hoary cress	invasive		
Lepidium chalepensis	lenspod whitetop	invasive		
Leymus cinereus	basin wildrye	native	FAC	
Lithospermum ruderale	western stoneseed	native		
Lonicera involucrata	twinberry honeysuckle	native	FAC	yes
Lupinus sericeus ssp. *sericeus*	silky lupine	native		
Lycopus americanus	American water horehound	native	OBL	
Maianthemum racemosum	large false Solomon's seal	native	FAC	
Melilotus albus	white sweet clover	invasive	FACU	
Mentha arvensis	wild mint	native	FACW	
Microseris borealis	northern microseris	native	OBL	
Mimulus guttatus	yellow monkey-flower	native	OBL	
Myosotis laxa	bay forget-me-not	native	OBL	
Osmorhiza berteroi	sweet cicely	native	FACU	
Paxistima myrsinites	Oregon boxleaf	native	FACU	
Phacelia hastata var. *hastata*	silver-leaf scorpion-weed	native		

Phalaris arundinacea	Reed canarygrass	invasive	FACW	
Philadelphus lewisii	Lewis' mock orange	native		
Pinus ponderosa	ponderosa pine	native	FACU	
Poa bulbosa	bulbous bluegrass	invasive	FACU	
Poa pratensis	Kentucky bluegrass	both	FAC	
Poa secunda	Sandberg bluegrass	native	FACU	
Poa secunda ssp. *secunda*	Sandberg bluegrass	native	FACU	
Poaceae sp.1	Poaceae species 1	NA		
Poaceae sp.2	Poaceae species 2	NA		
Poaceae sp.3	Poaceae species 3	NA		
Poaceae sp.4	Poaceae species 4	NA		
Poaceae sp.5	Poaceae species 5	NA		
Populus tremuloides	quaking aspen	native	FACU	
Populus trichocarpa	black cottonwood	native		
Prunus virginiana	chokecherry	native	FAC	yes
Psathyrostachys juncea	Russian wildrye	invasive	UPL	
Pseudotsuga menziesii	Douglas fir	native	FACU	
Pteridium aquilinum	western brackenfern	native	FACU	
Purshia tridentata	antelope bitterbrush	native		
Rhaponticum repens	Russian knapweed	invasive		
Ribes aureum	golden currant	native	FAC	yes
Ribes bracteosum	stink currant	native	FAC	yes
Ribes divaricatum	spreading gooseberry	native	FAC	yes
Ribes lacustre	prickly currant	native	FACW	yes
Ribes viscosissimum	sticky currant	native	FAC	yes
Ribes aureum	golden currant	native	FAC	yes
Ribes cereum var. *colubrinum*	wax currant	native		
Ribes hudsonianum	northern black currant	native	FACW	yes
Rosa woodsii	Woods' Rose	native	FACU	
Rubus idaeus	American red raspberry	native	FACU	
Rubus parviflorus	thimbleberry	native	FAC	yes
Rubus idaeus var. idaeus	American red raspberry	invasive	FACU	
Rumex acetosella	common sheep sorrel	invasive	FACU	
Salix amygdaloides	peachleaf willow	native	FACW	yes
Salix melanopsis	dusky willow	native	OBL	yes
Salix scouleriana	Scouler's willow	native	FAC	yes

Salix exigua var. *exigua*	narrowleaf willow	native	FACW	yes
Schoenoplectus acutus	hardstem bulrush	native	OBL	
Scirpus atrocinctus	black girdle bulrush	native	OBL	
Scutellaria galericulata	marsh skullcap	native	OBL	
Secale cereale	cereal rye	invasive		
Silene latifolia	bladder campion	invasive		
Silene menziesii	Menzies' campion	native	FAC	
Sisymbrium altissimum	tall tumble mustard	invasive	FACU	
Solanum dulcamara	climbing nightshade	invasive	FAC	
Solidago lepida	Western Canada goldenrod	native	FACU	
Sonchus asper	spiny sowthistle	invasive	FAC	
Sonchus oleraceus	common sowthistle	native	UPL	
Symphoricarpos albus	common snowberry	native	FACU	
Symphyotrichum campestre	western meadow aster	native		
Symphyotrichum ericoides var. *pansum*	many-flowered aster	native	FAC	
Symphyotrichum lanceolatum ssp. *hesperium*	white panicle aster	native	OBL	
Taraxacum officinale	common dandelion	invasive	FACU	
Thalictrum occidentale	western meadow-rue	native	FACU	
Tragopogon dubius	yellow salsify	invasive		
Trifolium repens	white clover	invasive	FACU	
Typha latifolia	broadleaf cattail	native	OBL	
Unknown aquatic weed		NA	OBL	
Unknown species		NA		
Urtica dioica	stinging nettle	both	FAC	
Verbascum thapsus	common mullein	invasive	FACU	
Veronica americana	American speedwell	native	OBL	
Veronica anagallis-aquatica	water speedwell	native	OBL	
Vicia cracca	bird vetch	invasive		
Viola glabella	pioneer violet	native	FAC	
Viola palustris	marsh violet	native	FACW	

Table 11. Summaries statistics for the water quality measurements taken from June 2021 to August 2022.

		Discharge L/s		pH		Conductivity mS/cm		Conductivity mS/cm^c		Dissolved Oxygen mg/L		Dissolved Oxygen %		Water Temperature ◦C	
		min	max	min	max	min	max	min	max	min	max	min	max	min	max
Bear	BDA	0.54	85.52	7.53	9.91	0.06	0.13	0.08	0.17	6.95	12.91	69.60	105.50	0.55	15.70
	Control	6.87	110.63	7.77	10.16	0.04	0.82	0.06	0.16	9.31	17.80	85.80	142.90	0.14	16.23
	Beaver 1	0.11	97.14	7.43	10.24	0.05	0.12	0.07	0.16	4.45	12.13	40.20	103.60	0.97	13.43
Texas	BDA	0.02	4.28	7.22	11.14	0.23	0.36	0.37	0.42	8.23	13.50	80.30	114.90	2.42	17.74
	Control	0.01	3.78	7.87	11.06	0.23	0.35	0.37	0.42	8.15	12.06	80.10	106.50	3.32	16.24
	Beaver 2	0.00	6.87	7.16	10.14	0.07	0.31	0.09	0.39	2.26	12.36	23.30	95.80	0.36	15.76
Cow	BDA	2.86	8.37	7.38	9.40	0.29	0.40	0.48	0.53	9.72	14.30	91.00	111.90	2.82	13.08
	Control	2.46	8.49	8.09	9.92	0.30	0.40	0.44	0.53	10.22	13.68	86.10	116.60	3.26	12.82
Tunk	BDA	15.10	197.07	6.13	10.63	0.10	0.31	0.15	0.41	8.51	16.12	74.40	119.30	0.04	23.54
	Control	0.00	196.93	6.15	10.88	0.10	0.31	0.12	0.41	7.90	15.79	83.50	115.90	0.07	24.09
	Beaver 3	7.47	554.24	8.02	9.88	0.19	0.39	0.24	0.51	8.28	15.00	87.50	115.70	-0.08	19.44
Chiliwist	BDA	0.01	43.95	7.00	10.60	0.11	0.31	0.21	0.34	6.24	16.79	64.90	120.90	0.07	20.59
	Control	1.13	48.79	7.27	11.01	0.11	0.30	0.19	0.34	7.69	17.68	83.50	124.90	0.41	19.28

Table 12. Summaries of published primary research conducted on BDA structures.

Location	Study	BDAs	Years monitored	Question	Results
Bridge Creek and Murderers Creek, tributaries to the John Day River, OR	(Conner et al. 2016)	Four treatment reaches in Bridge Creek Murderers Creek was the control	Pre-manipulation (2007–2009) Post-manipulation (2010–2012)	BACI study with a Bayesian MCMC approach to evaluate the effectiveness of a river restoration project to increase juvenile steelhead (Oncorhynchus mykiss) survival and density	After restoration juvenile steelhead survival increased by 36% on average and density increased by 58% on average in BDA reaches in comparison to their controls.
	(Bouwes et al. 2016)	4 treatment reaches with 121 BDAs installed throughout them in Bridge Creek Two Bridge Creek tributaries had control reaches and Murderers Creek had 3 control reaches	Pre-manipulation (2007–2009) Post-manipulation (2010–2013)	Can BDAs aggrade a highly incised stream and improve habitat quantity and quality for steelhead?	Juvenile survival increased by 52% and juvenile production increased by 175% in Bridge Creek compared to Murderers Creek control site.

	(Weber et al. 2017)	Four treatment reaches, 2 controls, and many beaver sites in Bridge Creek	Pre-manipulation (2007–2009) Four years post-manipulation (2010–2014)	How do beaver dams and BDAs influence stream temperature throughout the year at multiple spatial scales?	Summer water temperatures were moderated at the reach scale, and there was increased temperature heterogeneity at the channel scale.
	(Silverman et al. 2019)	Same reaches as (Bouwes et al. 2016)	Pre-restoration 1997-1999, 2001, 2005, 2006, & 2008 Post-restoration 2010-2016	How has riparian vegetation via the normalized difference vegetation index changed since BDAs were installed?	Vegetative productivity increased by 20% and was higher longer into the growing season after BDAs were installed.
Long, Alkali, and Robb creeks on Missouri River Headwaters Basin, MT	(Vanderhoof and Burt 2018)	1 reach with 6 BDAs on Alkali Creek 1 reach with 12 BDAs on Robb Creek 2 reaches on Long Creek (9 and 7 BDAs)	2014 images were used as pre-restoration and 2017 images were used as post-restoration	How can multispectral high-resolution imagery be used to monitor stream condition and how have four restoration reaches changed since BDAs were installed?	In restoration areas, there were increases in stream surface area upstream of the BDAs or reactivated side channels, and there were decreases in stream surface area downstream of the BDAs.
California Creek A tributary to Hangman Creek, WA	Niezgoda	6 BDAs 4 built in the fall of 2016 And 1 in 2017 and 1 in 2018	April 2017 and April 2018	Are BDAs effective at trapping sediment and reducing the load that gets carried downstream?	On average behind each BDA sediment aggregated 3.4m^3.
	(Wade et al. 2020)	5 BDAs in a 250 m reach. BDAs 1 and 5 were excluded from this study due to	One-year post-restoration in 2019	What are the patterns of hyporheic exchange and biogeochemical redox zonation in a BDA reach? What causes variation in the hydrologic effects of	Biogeochemical cycling spatial patterns like natural beaver dams were produced by the BDAs. One of the BDAs that is similar in size to a natural beaver dam was able to

		structural failure		BDAs on the reach scale?	produce similar hyporheic fluxes.
Red Canyon Creek, 2nd order creek Lander, WY	(Pearce et al. 2021a, b)	5 BDAs in a 250 m reach In Pearce et al. b, they exclude BDAs 1 and 5 due to structural failure	Pre-restoration 2017 Post-restoration 2018 & 2019	How do BDAs affect surface water-groundwater interactions, and water levels and temperatures? What are the impacts of BDAs on streambank erosion and deposition patterns?	After restoration, the water flowed from the stream to the floodplain. Elevated stream and groundwater levels. No effect on stream temperature. BDA sites had less overall erosion and greater deposition, and greater spatial heterogeneity patterns. Breaches led to significant bank erosion observed downstream of the complex.
	(Davis et al. 2021)	5 BDAs in a 250 m reach BDA 1 and 5 were excluded from this study Control site upstream of the restoration area	August 2017, August 2018 (the week immediately following installation), and August 2019	How does the geomorphology of streams change after BDA installation using data from annual unoccupied aerial vehicle surveys?	Higher deposition on the inner edges of meanders in BDA reaches. Most aggradation was at the most upstream BDA then erosion dominates at the downstream BDAs. High erosion after the BDAs were breached
Fish Creek in the Colorado Front Range, CO	(Scamardo and Wohl 2020)	7 BDAs in a 700m reach 1 unrestored reference reach	May to October 2018	Are the BDAs increasing stream bed aggradation and raising water tables?	BDAs had no significant effect on groundwater tables. BDAs were less effective at trapping fine sediment than natural dams based on predicted values.
South Fork in the Deschutes River Basin, OR	(Orr et al. 2020)	5 BDAs were built in 2016 in a 2.25 km reach	Some measurements were taken pre-	How does a low gradient stream and its vegetation respond to BDAs	BDAs did not affect stream temperature. Groundwater levels rose by 18-30cm. This was spread

			restoration in 2016 Post-restoration 2017 and 2018	installed in the short term?	135 m upstream and 12 m into the floodplain. Willows planted in BDA reaches had 1.3 times more growth.
North arm of Pine Creek in the Canadian Rockies, AB	(Munir and Westbrook 2021a, Munir and Westbrook 2021b)	6 BDAs were installed in the 1075m reach A single, double, and triple configuration	Pre-restoration 2017 Post-restoration 2018 and 2019	How do BDAs affect the thermal regime of streams? How do BDAs affect the water tables of streams?	They found that there was stream warming downstream of BDAs and in general the more BDAs installed in the sequence, the more the stream and pond warmed. Thermal variation was decreased. There was an immediate rise in the water table into the riparian area after installation. Water now flowed from the stream to the riparian area whereas before it flowed the opposite. In the single BDA configuration flow peaks were reduced and low flows were raised.
French Creek, 3rd order tributary to the Scott River, CA	(Corline et al. 2022)	4 restoration reaches and 1 control	Pre-restoration 2017 Post-restoration 2018 & 2019	How do BDAs affect the invertebrate community?	Increase in particulate organic matter and pelagic phytoplankton. Beta and gamma diversity increased. High densities of invertebrates. The water temperature was buffered. Half of the invertebrates found were unique to the BDA habitat

Sugar Creek tributary to Klamath River, CA	(Pollock et al. 2022)	2 BDAs ~50 m and 200 m above its confluence with the Scott River	2016 and 2017 (one and two years after the BDAs were constructed)	Can juvenile coho salmon and steelhead trout pass over beaver dam analogs and go upstream?	21 days after salmonids were displaced from upstream of the BDAs to downstream of the dams, 91% of juvenile coho and 54% of juvenile steelhead trout were detected upstream of the dams showing that the BDAs did not affect their passage.
Blacktail Creek in Southwest MT	(Askam et al. 2022)	Treatment site with 10 BDAs, and 3 control sites	Post-restoration 2016-2021	Do BDAs increase the Enhanced Vegetation Index (EVI) (using Sentinel-2 satellites) of a mountain headwater stream?	BDAs did not increase the EVI, and it may be due to the site having high precipitation.
West Desert, Lower Bear, Weber, and Provo River HUC 6 catchments	(Wolf and Hammill 2023)	24 beaver dam complexes and 9 BDA complexes	sampled twice per year from 2019 to 2021	How do BDA and beaver complexes compare as amphibian habitat? Are BDAs suitable habitat for amphibian breeding?	BDA sites were younger, at a lower elevation, and contained a greater fish abundance. Although tiger salamanders (*Ambystoma mavortium*) did not co-occur with fish in the BDA sites, the BDAs acted as tiger salamander habitat on the complex scale.

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VITA

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Education

2021-20203	M.S. in Biology, Eastern Washington University, Cheney, WA
	Thesis: Baseline data to test Beaver Dam Analogs as a Stream Restoration Tool in fire-affected tributaries of the Methow and Okanogan Watersheds
2018-2020	B.S. in Conservation Biology and Ecology, Montana State University, Bozeman, MT
2016-2018	Running start program, Columbia Basin College, Pasco, WA
2014-2018	High school diploma, Pasco High School, Pasco, WA

Experience

2021-2023 Graduate Research Assistant, Laboratory of Rebecca Brown, and Camille McNeely, Eastern

Washington University, Cheney, WA, Principal Investigator: Rebecca Brown

- I performed vegetation surveys, topography surveys, pebble counts, and water travel time measurements.
- I keyed collected plants and performed total phosphorus and orthophosphate analyses on water samples.
- I lead a field crew of three people on proper data collection
- I was a teacher's assistant for the introductory biology series, field botany, and data analysis for biologists.

2022-2023 Researcher, Geosciences Department, Principal Investigator: Brian Buchanan

- Surveyed deciduous trees, determined their genus, measured their DBH, measured their heights in ArcGIS Pro using LIDAR, and then determined their carbon sequestration using Itree.

2021 Fisheries Technician II, Prince William Sound Aquaculture Corporation, Main Bay, AK

- At a sockeye salmon hatchery, I performed daily activities to maintain healthy growing salmon
- I helped take monthly weights to monitor growth
- I did other tasks to maintain the equipment and buildings

2020 Undergraduate Research Assistant, Laboratory of Ryan Thum, Montana State University,

Bozeman, MT, Principal Investigator: Ryan Thum

- I performed lab work on *Nymphoides peltata* such as DNA extractions, PCR, and gel electrolysis.
- I analyzed ten microsatellite markers to determine unique clones. The invasive species clones were used to determine where the source population came from.

Presentations

2022 Killoy, K., R. L. Brown, C. McNeely, and A. Whipple. Poster. Baseline data to test Beaver Dam Analogs as a Stream Restoration Tool in fire-affected tributaries of the Methow and Okanogan Watersheds, WA. Joint Aquatic Sciences Meeting.

2022 Killoy, K., R. L. Brown, C. McNeely, and A. Whipple. Poster. Baseline data to test Beaver Dam Analogs as a Stream Restoration Tool in fire-affected tributaries of the Methow and Okanogan Watersheds, WA. Eastern Washington University Creative Works Symposium.

2022 Killoy, K., R. L. Brown, C. McNeely, and A. Whipple. Poster. Baseline data to test Beaver Dam Analogs as a Stream Restoration Tool in fire-affected tributaries of the Methow and Okanogan Watersheds, WA. Washington American Water Resources Association National Conference.

Grants and Fellowships

2022 Washington American Water Resources Association Fellowship

2022 Northeast Washington Native Plant Society Student Research Grant

2020 Montana State University, Undergraduate Scholars Program Grant

Awards

2021-2022	Graduate Service Appointment, Eastern Washington University, Cheney, WA
2018-2020	Montana State's President's List
2018	Valedictorian for Pasco High School's 2018 graduating class of 465 students
2018	National Scholar/Athlete Award from the United States Army Reserve
2018	Superintendent' Eclipse Award for Academic Achievement from the Pasco School District #1
2018	Award for academic achievement in the top 10% of the Washington state high school graduating class of 2018
2018	United States Marine Corps Scholastic Excellence Award Washington State Honors