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Assessment of water and streambed sediment quality near a historic mine site in Lincoln National Forest, New Mexico: Influence of beaver dam analogs and downstream distance

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Research Article

Keywords: BDAs, Mining, Heavy-Metals, Contamination, Watershed, Restoration

Posted Date: November 24th, 2026

DOI: https://doi.org/10.21203/rs.3.rs-11627652/v1

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Additional Declarations: No competing interests reported.

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Assessment of water and streambed sediment quality near a historic mine site in Lincoln National Forest, New Mexico: Influence of beaver dam analogs and downstream distance

Abstract

Forests provide crucial ecosystem services, including climate regulation, biodiversity, and water conservation; however, forests are vulnerable to threats, including drought, climate change, wildfires, and resource extraction, such as mining. In response, forest managers utilize restoration strategies, including the installation of beaver dam analogs (BDAs) to improve water retention and reduce erosion. This study investigates whether a historic mine in Lincoln National Forest, New Mexico, is a point source of heavy metal pollution and whether recently installed BDAs improve downstream water quality.

Water and sediment samples were collected at the mine, at a nearby reference stream, immediately upstream and downstream of the first BDA, and immediately downstream of each of the remaining 10 BDAs along Bear Creek. Water and sediment samples were analyzed using ICP-AES and pXRF, respectively, to quantify element concentrations. Spearman's rank correlation and variation partitioning analyses were utilized to evaluate the relative importance of distance and number of BDAs in predicting metal concentrations.

Water samples from the mine site exhibited the highest turbidity and specific conductance, and the lowest dissolved oxygen. Elevated concentrations of Cd, Cr, Co, Fe, Pb, Mn, Mo, P, S, Ti, and Zn in sediment samples were detected at the mine. Metal concentrations generally decreased with downstream distance. In both water and sediment samples, distance had a stronger influence on metal concentrations than the number of BDAs, although Cu in sampled water and Co in sediment concentrations showed evidence of partial attenuation by BDA filtering. The historic mine site remains a source of metal pollution.

Keywords: BDAs, Mining, Heavy-Metals, Contamination, Watershed, Restoration

1.0 Introduction

Ecological and forest restoration are crucial strategies for reducing degradation, recovering biodiversity, and improving ecosystem resilience following anthropogenic or natural disturbances (Edwards & Cerullo, 2024; Stephens et al., 2020; Wortley et al., 2013). Restoration activities can improve forest resilience by promoting ecological functions that can help ecosystems resist and recover from future stressors, including stabilizing hydrologic processes, reducing soil erosion, and improving habitat quality for aquatic and terrestrial biota (Scamardo et al., 2025; Silliman et al., 2024). Restoring ecological functions promotes interventions that can enhance ecosystem services and mitigate biodiversity loss (Bullock et al., 2011). However, the effectiveness of restoration efforts depends on the extent to which previous disturbances have altered conditions within a particular ecosystem.

Ecological disturbances are primary drivers of ecosystem transformation because they alter physical and biological properties of landscape (Gough et al., 2024). Both natural and anthropogenic disturbances can degrade soils, vegetation, and water resources. For example, high-severity wildfires, caused by lightning or human activities, can trigger post-fire erosion, sediment deposition, and debris-flow flooding that alter vegetation, soils, and watershed processes sustaining municipal and other downstream water supplies (Bladon et al., 2014; Robichaud & Padowski, 2024; Shephard et al., 2025). Similarly, anthropogenic disturbances, including deforestation, overgrazing, landfill leakage, and mining operations, contribute to environmental degradation and reduce soil and sediment quality (Bisht et al., 2020). In particular, abandoned and active mine sites can degrade ecosystem structure and functions through erosion,

contaminated runoff, and acid mine drainage, which can lower pH, mobilize dissolved heavy metals, and degrade both surface and groundwater quality (Huang et al., 2013; Issaka & Ashraf, 2017; Ngole-Jeme & Fantke, 2017; Ziqiu et al., 2025). These changes adversely impact aquatic organisms through bioaccumulation of heavy metals (Grajal-Puche et al., 2024).

Mine contaminants impact forest ecosystems and water resources, including municipal use of water many miles downstream from a mine site (Church et al., 2007; Niyogi et al., 1994; Todd et al., 2024). These contaminants often contain suspended particulate matter, which can increase turbidity and eventually reduce dissolved oxygen by limiting light penetration and increasing microbial oxygen demand (Cheng et al., 2024; Diallo et al., 2019; Shi et al., 2026). Dilution may limit the extent of metal contamination, as increased streamflow, precipitation, and runoff can reduce concentrations of mine-derived contaminants in stream water downstream from a point source (Caruso, 2001; Huang et al., 2020; Lasagna et al., 2013).

Severe wildfires and mining operations each increase rates of soil erosion, augmenting turbidity, which is a major contributor to the decline of aquatic species populations (Henley et al., 2000; Ramli et al., 2020). High-severity fires cause extensive tree mortality, reduce infiltration, and augment surface runoff, causing major floods (Ebel et al., 2012; Neary et al., 2005). One place where mining contaminants, wildfire, erosion, and forest restoration all intersect is in the Lincoln National Forest in south-central New Mexico. The Smoky Bear District of the Lincoln National Forest is adjacent to the village of Ruidoso, a community with a high degree of wildland-urban interface (WUI) and high vulnerability to fire, erosion, and flooding. Recent floods that occurred in Ruidoso in 2025 (Lee & Montoya, 2025) underscore the urgent need for managers to implement forest restoration strategies that mitigate against post-fire erosion and flooding (Razo et al., 2004).

Beaver dam analogs (BDAs) are an ecological restoration technique designed to mimic the functions of natural beavers, thereby contributing to erosion control, improving water quality, and the restoration and maintenance of stream ecosystems (Bailey et al., 2019; Loredo et al., 2025; Neace, 2023; Pearce et al., 2021; Pollock et al., 2014). BDAs can play a pivotal role in improving water quality by reducing pollution in freshwater and trapping heavy metals from mine waste discharge (Dittbrenner et al., 2022; Yang et al., 2023). Recently, BDAs have been installed in the wake of mixed-severity fires, postfire erosion, and flooding in Big Bear Canyon, which lies at the headwaters of the Rio Bonito Watershed, the primary source of water for Ruidoso and other surrounding communities (U.S. Army Corps of Engineers, 2013). The Big Bear Canyon was also the location of historic mining activities.

The Mescalero Apache were the first to discover gold in the area now known as Lincoln National Forest (Opler & Opler, 1950). By the 1880s and into the early 1930s, major mining industries in this region included the American Mine, the Helen Rae Mine, and the Parson's Mine. The Parson's Mine was noted for placer gold from 1860 and was abandoned in the 1920s (Griswold, 1959; Thompson, 1966). Miners utilized early technologies such as placer mines and amalgamation plants to run surface water through elongated troughs (sluice boxes or "rockers") to wash alluvial gravel. Mercury was used in gold extraction through an amalgamation process that caused gold particles to coalesce (Donkor et al., 2024). Other minerals and metals recorded in mining production in Lincoln County include: Pb, Cu, Ag, Zn, W, coal, iron ore, CaF_2, and $Ce(CO_3)F$ (Cherotich & McLemore, 2023). The Great Western Mine was among the several mining companies located in the area around Big Bear Canyon. The Great Western Mine has an unknown ore tenor and shows no evidence of operation since the 1950s. A historic mine site in Big Bear Canyon is just one of more than 500,000 abandoned and inactive mines across the

126 United States, with a significant proportion located in sensitive mountain watersheds (Church et
127 al., 2007; Peplow, 1999; US EPA, 2025).

128 Much of the Rio Bonito watershed burned in the 17,664-acre Little Bear Fire in June
129 2012 (Tillery & Matherne, 2013). Erosion after high-intensity fire caused sediment buildup in the
130 Rio Bonito drainage basin, including headwaters in Big Bear Canyon (Tillery & Matherne,
131 2013). The historic mine site was exposed due to erosion in Big Bear Canyon after the Little
132 Bear Fire. The severe erosion has motivated managers of the Smokey Bear Ranger District of the
133 Lincoln National Forest to install BDAs along the riparian corridor.

134 The objectives of this research are (1) to investigate whether the exposed mine site in Big
135 Bear Canyon functions as a source of heavy metal pollution, and (2) to evaluate whether the
136 recently installed BDAs are filtering heavy metals from water, thereby causing metals to be
137 accumulated near the first BDAs and improving water quality downstream (Brazier et al., 2021;
138 Müller & Mcfadzean, 2019; Yang et al., 2023). Alternatively, metal concentrations may be
139 unaffected by the presence of BDAs, but could be diluted farther downstream (Aelion et al.,
140 2009; Besser et al., 2007; Lin et al., 2022a; Tian et al., 2018). If there is no pollution source in
141 Big Bear Canyon, heavy metal concentrations will remain low in water and sediment samples at
142 all sampling locations (Fig. 1A). It is also possible that there are numerous sources of heavy
143 metal pollution in Big Bear Canyon (Fig. 1D), or that heavy metal pollution emanating from the
144 mine site is not reduced by the presence of BDAs or by short downstream distances less than 2
145 km. In the latter case, elevated metal concentrations may persist along the entire 1.5 km stretch

of the riparian corridor that was evaluated (Fig. 1E).

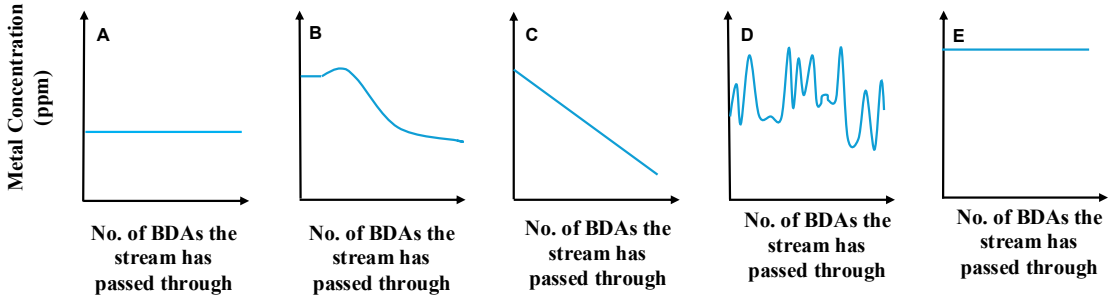


Figure 1. Graphical representation of the five alternate hypotheses evaluated for streambed sediments in this study (Question 2). The blue lines represent heavy metal concentrations over the downstream distance from the historic mine site. Metal concentrations may decrease abruptly in water samples but remain elevated in sediments (Salomons & Stigliani, 2012) at BDA locations (Fig. 1B); however, concentrations in both samples remain high across all locations due to multiple sources of pollution (Briffa et al., 2020) (Fig. 1D).

2.0 Materials and methods

2.1 Study area

The Big Bear Canyon study area is located within the Smokey Bear Ranger District of the Lincoln National Forest (LNF) in south-central New Mexico (Fig. 2). The Lincoln National Forest receives an average annual precipitation of 600 mm/yr, with a mean maximum temperature of 26 °C in July and a mean minimum temperature of - 5 °C in January (Ryniker et al., 2006). Sixty to seventy percent of precipitation falls during summer periods (July-September), and an appreciable amount occurs in winter as snowfall (Brown et al., 2001). Historically, fires have been mixed in severity, creating both stand replacement and surface fire patterns (Agee, 1998; McClure et al., 2024). Historic fires have shaped this southwestern mixed-conifer forest with minimum fire return intervals between 3 and 13 years and a maximum fire return interval of 18 to 32 years (Holden et al., 2010). In 2021, a total of 11 BDAs were installed

in Big Bear Canyon, beginning 185 m downstream from the mine site (Fig. 2**)**. The BDAs spanned a total distance of 829 m.

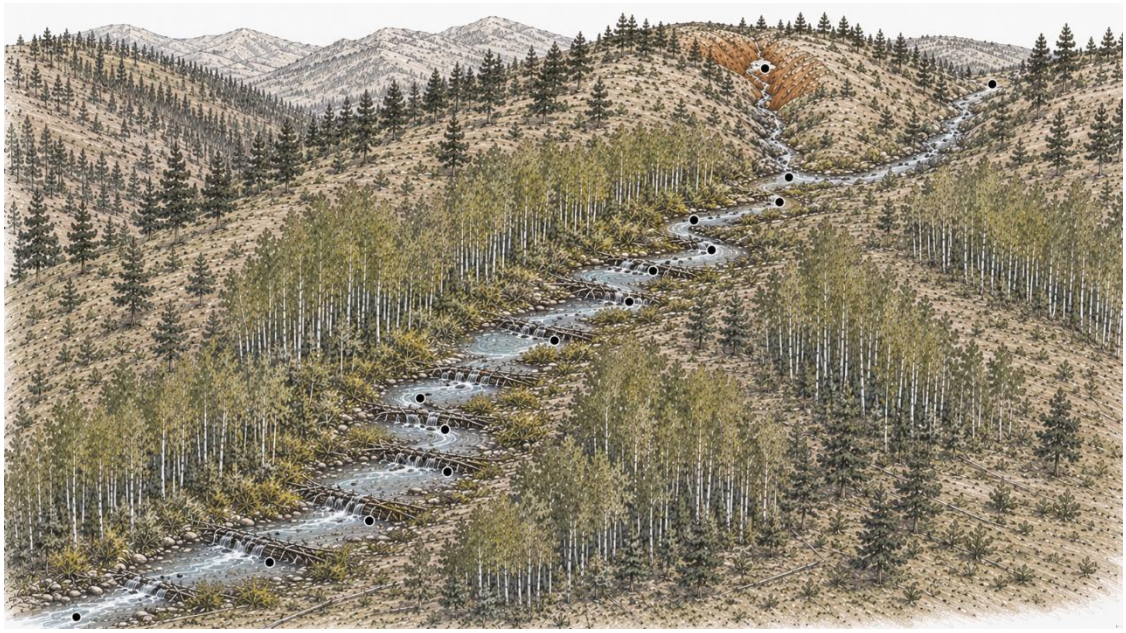


Figure 2. A conceptual diagram of our sampling design along the 1.5 km stretch of Bear Creek. Black dots indicate sediment and water sampling locations, and horizontal brown lines represent BDAs locations. The reference and the historic mine site are parallel, both located 51 m upstream of the confluence, while BDA 1 is located 184 m downstream of the confluence point. The farthest downstream sampling point was located 1473 m from the mine site.

2.2 Water quality and metal concentrations in water and sediment samples

Water quality parameters along the Bear Creek were measured during September 2023. A nephelometer was utilized to measure turbidity (NTU) (Davies-Colley & Smith, 2001), and a thermometer to measure water temperature (°C). A YSI Pro2030 device was used to measure dissolved oxygen (DO) (µg/L) and specific conductance (mS/cm at 25 °C) (VeeAnn & Bratton, 2018). Sixteen sampling locations, including the mine site, reference site on an adjacent stream, a site downstream of the confluence between the Bear Creek and the adjacent stream (upstream of the first BDA), immediately above and below the first BDA, downstream of the remaining 10 BDAs, and at a site located approximately 1,500 m downstream from the mine site (Fig. 2). The

latter sampling location was included to evaluate whether increasing distance farther downstream and streamflow contribute to dilution of metal contaminants originating from the historic mine site. Samples were taken from downstream locations first to minimize any risk of cross-contamination.

Water samples were collected at a depth of 0.15 m below the creek's surface by submerging a 250 mL acid-rinsed polyethylene bottle while facing upstream into the current. In situations where the water depth was shallow, adjustments were made to accommodate the shallow water by sampling water at a reduced depth or at an alternative location nearby where the water depth met the requirements. Two replicate water samples were collected at each of the 16 designated sampling locations ($n = 32$). Collected samples were labeled (location, date, time) according to the site description and kept in an ice chest at a cooler temperature (5 °C) before transporting them to the Geosciences Department, Texas Tech University. Samples were preserved and prepared for total recoverable metal analysis following U.S. EPA Method 200.2 and subsequently analyzed using inductively coupled plasma-atomic emission spectroscopy (ICP-AES) following U.S. EPA Method 200.7 (U.S. EPA, 1994b, 1994a). Water samples were preserved using an acidified nitric acid (HNO_3) to a pH < 2 and stored them in a refrigerator for 104 days until analysis. Water samples were composited and digested using concentrated nitric acid (HNO_3) to lower the pH below 2.0 to prevent metal precipitation (Miranda et al., 2022; Sayo et al., 2020). To ensure analytical accuracy, we utilized the instrument calibration standard 2 as the reference standard material and diluted according to the $C_1V_1 = C_2V_2$ equation to prepare a series of calibration standards at 0.01, 0.05, 0.1, 1, and 5 µg/L (US EPA, 2019). These standards were used to build calibration curves for accurate detection and quantification of metal and other elements concentrations across the analytical range (U.S. EPA, 2019).

Calibration/reference standards were run after every five samples, and analytical accuracy was assessed by comparing measured concentrations with the calibration/ referenced values. Analytical errors were calculated for each metal or element and summarized them in Table S1. Heavy metal composition in sampled water was measured by comparing the emission intensities generated from water samples with those of the calibration standards.

Sediments were sampled on the same day and with the same general protocol as water samples to ensure consistency. The pXRF instrument was used to quantify elemental concentrations in sediment samples. Sediments were collected from the same sampling locations immediately after water sampling to avoid cross-contamination. At the mine site, a star pattern sampling design approach was used to capture spatial heterogeneity of sampled sediments (Jordan, 2022; Fig. 3). Sediment samples were collected at the center of the mine and eight additional locations ~ 0.4 m intervals around the center of the mine. This yielded a total of 48 sediment samples, 2 replicates each from the reference stream (*n = 2*), historic mine site (*n = 18*), and the remaining 14 sampling locations (*n = 28*).

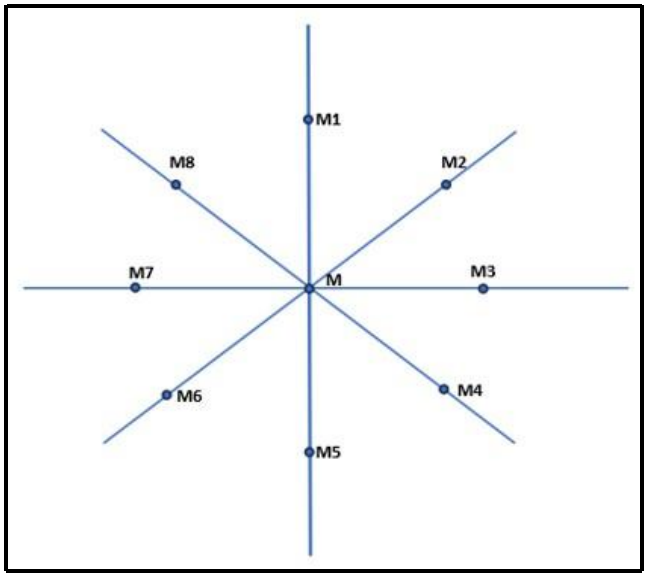


Figure 3. Illustration of the star-pattern sampling design used for sediment collection at the historic mine site. The middle point (M) represents the center of the mine location, and the eight

surrounding points (M1-M8) represent eight additional sampling locations arranged radially at approximately 0.4 m intervals from the center of the mine to capture spatial heterogeneity in sediment metal concentrations.

Streambed sediments were collected at 0.15-0.30 m below the sediment surface using a 0.19 m footstep soil probe marked at 0.15 m intervals, and samples were stored in sealed plastic bags (Cao et al., 2022). Similarly, streambed sediment samples were labeled (location, date, time) according to the site description and kept in an ice chest at a cooler temperature (5 °C) before transported to the Plant and Soil Science Department, Texas Tech University. Sediments were air-dried at 55 °C for four weeks to remove soil moisture. Sediments were gently crushed, homogenized, and sieved through a clean 0.002 m mesh (Liu et al., 2022). Sediments were placed into a 23 mm diameter × 25 mm height sample cup and inspected at the bottom to ensure the film was free from debris before undergoing ex-situ portable X-ray fluorescence (pXRF) analysis (Weindorf & Chakraborty, 2020; Zhou et al., 2020). The pXRF instrument was utilized to quantify elemental and metal concentrations in sediment samples (Weindorf & Chakraborty, 2020). Detection limits for many elements in sediments were generally in the range of 5-10 mg/kg (Chakraborty, 2016; Schmidt, et al., 2019). The cup was centered over the pXRF aperture and covered with a safety lid to prevent X-ray exposure (Zhou et al., 2018). Sediment samples were scanned for 240 s to improve analytical precision. The pXRF instrument was checked for stability using an Olympus quartz puck and verified for analytical accuracy using certified reference materials from the National Institute of Standards and Technology (NIST; 2711a and 2710a) (Chakraborty, 2016; Zimmerman et al., 2021). Standard reference samples certified by the NIST were scanned following the analysis of sediments and compared against their certified values to validate instrument calibration and for quality control measures (Weindorf & Chakraborty, 2020). For each measured metal, recovery was calculated by comparing the

measured concentration with the reference/certified value of the reference material, and analytical error was also determined accordingly (Table S2).

2.3 Data analysis

Spearman's rank correlation analyses were performed in R version 4.4.1 (R Core Team, 2016) to evaluate monotonic relationships between metal or element concentrations in water and sediment samples and distance from the mine site, as well as the number of BDAs. Cd was not included in the Spearman correlation analysis because cadmium's concentrations were below the detection limits of both the ICP-AES and pXRF devices, resulting in non-detect values (0 µg/L, and 0 mg kg^{-1}, respectively) across all sampling locations except at the mine site that exhibited elevated concentrations.

Principal component analysis (PCA) was conducted on water and sediment samples using the vegan package in R (Dixon, 2003) to reduce the data dimensionality and generate derived variables as linear combinations of the original multivariate data. Further, variation partitioning analysis (VPA) was used immediately after PCA to evaluate the relative importance explained by the two variables in predicting elements or metal concentrations: (1) distance from the mine site and (2) the number of BDAs that stream water/sediment passed through before being sampled (Mood, 1969; Peres-Neto et al., 2006).

3.0 Results

3.1 Water quality and metal concentrations at the historic mine site

Water samples at the historic mine site were different than other sampling locations. The mine site exhibited the highest turbidity (21.3 NTU), highest specific conductance (1.698 mS/cm), and lowest dissolved oxygen (1.10 µg/L) relative to the adjacent stream site (3.36 NTU, 0.337 mS/cm, and 1.89 µg/L, respectively) and to all other samples (Table 1).

Through ICP-AES analysis, a total of 10 metals and other elements were identified in water samples (Table S3), including Cd (only at mine site), Co, Fe, Mn, Cr, Zn, Mo, Pb, As, and Cu (Fig. 4). Cd, Co, Fe, Mn, Cr, Zn, Mo, and Pb concentrations in sampled water were elevated at the historic mine site compared to other sampling points (Fig. 4A-H). At the mine site, Cd, Co, Fe, and Mn concentrations in sample water exceeded both acute and chronic thresholds established by the United States Environmental Protection Agency (U.S. EPA; Table 2) and National Recommended Water Quality Criteria for the protection of aquatic life in freshwater (U.S. EPA, 1984, 1987, 2015). A total of 15 metals and other elements were detected through pXRF analysis of sediment samples, including As, Al, Cr, Co, Ni, Cu, Cd, Fe, S, Zn, P, Mn, Pb, Ti, and Mo (Table 3; Fig. 5). At the mine site, mean concentrations of Cd, Fe, S, Zn, and Mo in sediments exceeded the United States Geological Survey's (USGS) soil contaminant threshold (Fig. 5 G-I, K, O). Elevated concentrations of P, Mn, and Ti in sediments were observed at the mine site, although their mean concentrations fell below the USGS soil contaminant threshold (Fig. 5 J, L, N).

Table 1. Results of in situ water quality parameters, including turbidity (NTU), temperature (°C), dissolved oxygen (DO, % and µg/L), and specific conductance (mS/cm) collected at sampling locations along the Bear Creek in Lincoln National Forest.

Sampling Location	Number of BDAs	Turbidity (NTU)	Temp (°C)	DO (%)	DO (µg/L)	Specific Conductance (mS/cm)
Reference	0	3.36	14.8	18.60	1.89	0.337
Mine Site	0	21.30	15.0	10.90	1.10	1.698
Between BDA1 and Confluence	0	6.34	15.7	17.80	1.77	0.337
Upstream of BDA 1	0	3.62	17.3	19.30	1.85	0.482
Downstream of BDA 1	1	5.06	17.0	18.85	1.82	0.481
Downstream of BDA 2	2	5.60	18.6	18.70	1.75	0.479
Downstream of BDA 3	3	3.58	19.9	18.95	1.72	0.485
Downstream of BDA 4	4	3.22	19.8	19.60	1.79	0.490
Downstream of BDA 5	5	6.07	20.4	19.45	1.76	0.486
Downstream of BDA 6	6	8.75	18.5	20.10	1.88	0.505
Downstream of BDA 7	7	5.40	18.5	22.50	2.10	0.509
Downstream of BDA 8	8	7.99	18.4	26.30	2.47	0.506
Downstream of BDA 9	9	5.20	17.7	28.60	2.73	0.453
Downstream of BDA 10	10	5.99	17.8	35.90	3.41	0.506
Downstream of BDA 11	11	6.48	17.3	28.40	2.73	0.504
1.5 Km Downstream	11	11.99	13.1	32.80	3.44	0.563

3.2 Metal concentrations in water and sediment passing through BDAs

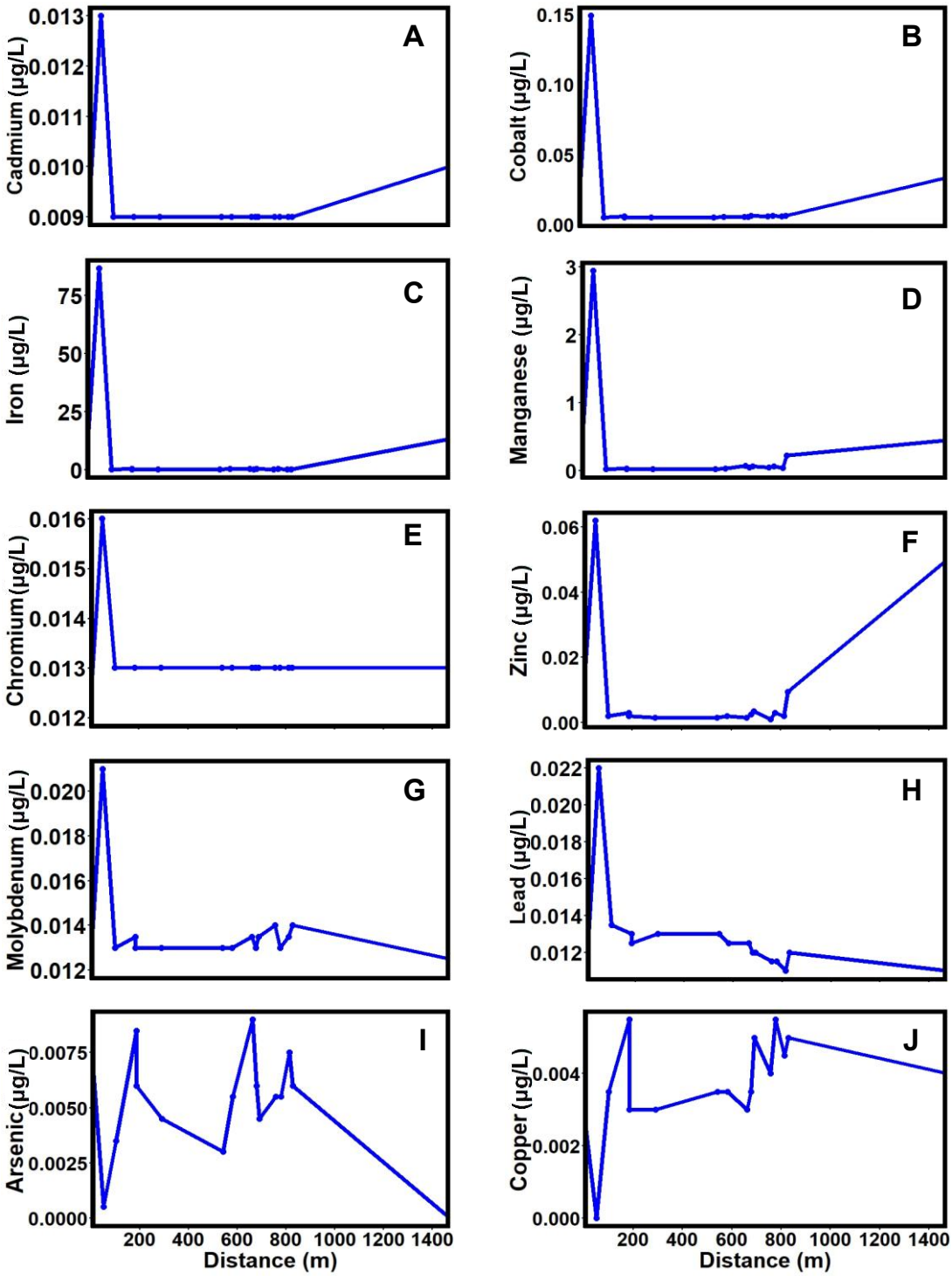
All metal or element concentrations in sampled water amidst the BDAs were elevated at the mine site relative to the adjacent reference stream except for As and Cu; however, their observed concentrations remained below the thresholds established by the United States Environmental Protection Agency (Table 2 and Fig. 4).

Table 2. National recommended water quality criteria for aquatic life. Acute Criteria Maximum Concentration is the highest concentration that aquatic organisms can be exposed for a short time without causing harmful effects. Chronic Criteria Continuous Concentration defines the highest concentration that aquatic species can be exposed over extended time without causing adverse effects.

Metal/ Element	Acute Criteria Maximum Concentration (µg/L)	Chronic Criteria Continuous Concentration (µg/L)	Reference
Cadmium	0.0018	0.0007	(USEPA, 1985a)

Cobalt	0.1200	0.0190	(USEPA, 1985b)
Iron	N/A	1.0000	(USEPA, 1985c)
Manganese	0.8090	0.0900	(USEPA, 1997)
Chromium	0.5700	0.0740	(USEPA, 1985b)
Zinc	0.1200	0.1200	(USEPA, 1980)
Molybdenum	0.8000	7.2000	(USEPA, 1998)
Lead	0.0650	0.0025	(USEPA, 1985c)
Arsenic	0.3400	0.1500	(USEPA, 1985c)
Copper	N/A	N/A	(USEPA, 1985c)

Also, in sediment samples, concentrations of 6 of 12 elements analyzed were below the established soil threshold levels, including As, Al, Cr, Co, Ni, and Cu (Fig. 5A-F). Co, Cu, Mo, and Pb concentrations in water showed a significant correlation with the number of BDAs (Table 4). The same four elements, including Co, Cu, Mo, and Pb in water samples, also showed a significant correlation with distance from the mine site. However, arsenic, chromium, and iron concentrations in water samples showed no significant correlations with either predictor variable (Table 4). Principal component 1 accounted for most of the variation in element concentrations, explaining 84.2% and 60.2% of variability in water and sediments samples, respectively (Table S4). Cu and As concentrations in water loaded negatively on the first principal component (PC1 = -0.214 and -0.274, respectively; Table S5).



308

309 **Figure 4**. Metal and other element concentrations (µg/L) in water samples over the downstream
310 distance from the historic mine site. A total of 11 Beaver Dam Analogs (BDAs) are located
311 downstream from 185-829 m. The blue dots indicate metal and element concentrations measured
312 at each sampling location.

313 The negative values indicate that As and Cu concentrations decline in water as distance from the
314 mine site and the number of BDAs increase farther downstream. The variation partitioning
315 analysis (VPA) of water sample data showed that only copper concentrations were primarily
316 driven by the number of BDAs, whereas the remaining elements were predominantly explained
317 by distance from the mine site (Table S6).
318 Cd, Fe, S, and P concentrations in sediments were elevated above the USGS soil
319 contaminant threshold at the mine site and decreased over distance farther downstream (Fig. 5G-
320 J). The sediment concentrations of all four of these elements became sharply reduced before
321 reaching the first BDA (185 m from the historic mine site), followed by a more gradual decrease
322 as the sediment passed through several BDAs farther downstream. There was no statistically
323 significant correlation between Fe, S, and P concentrations in sediments and the number of
324 BDAs (Table S7). Because no Cd data were detected at any location away from the mine site, the
325 correlation between Cd concentrations and the driver variables (distance from mine site and
326 number of BDAs) could not be evaluated. Fe, S, and P concentrations in sediments exhibited
327 negative loadings on the first principal component (Table 5). Similar to the pattern in water
328 sample data, sediment data showed that distance from the mine site was a stronger determinant
329 of all element concentrations than the number of BDAs, except for Co concentrations
330 (contribution: BDAs = 10.32%, distance = 8.51%; Table 6). For both water and sediment
331 samples, collinearity was observed between the two predictors (Fig. S1 and Fig. S2,
332 respectively). While many elements showed declining concentrations as samples progressed
333 through more BDAs, the distance from the mine site consistently explained a higher amount of
334 variance in metals' and other elements' concentrations in water and sediment samples, compared

to the variances explained by the number of BDAs

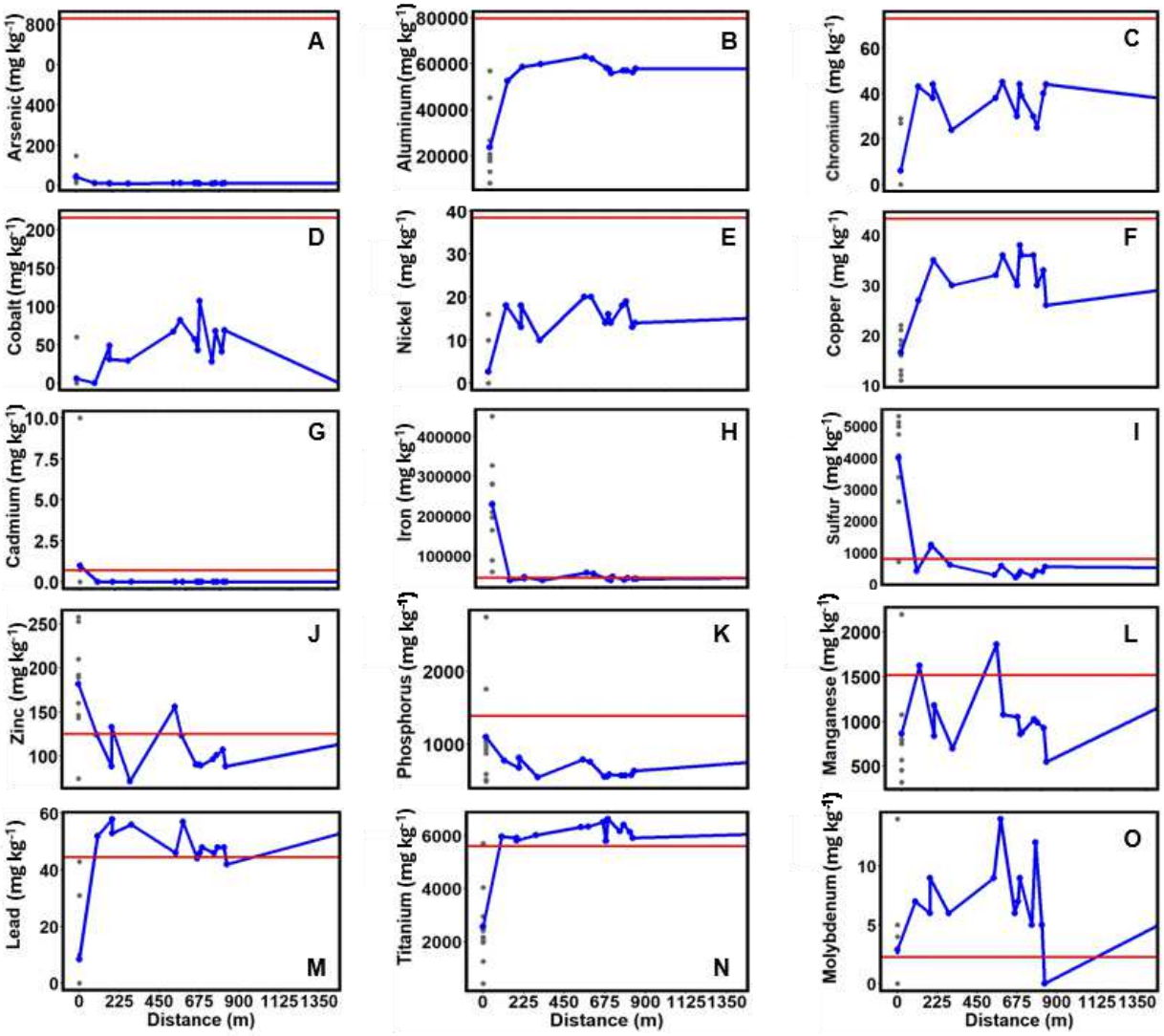


Figure 5. Metal and other element concentrations in streambed sediments over the downstream distance from the historic mine site. A total of 11 BDAs are located 185-829 m downstream from the mine site. The red line indicates the USGS soil contaminant threshold, while the grey dots represent individual samples of metal concentrations at the mine site, taken in a star pattern. The blue dot at zero m represents the average at the mine site, while the other blue dots indicate the metal concentration measured in individual samples at each other location.

Table 3. Concentrations of heavy metals and other elements in streambed sediments over distance from the historic mine site (0 m). Downstream from the mine, water and sediment pass through a series of eleven Beaver Dam Analogs (BDAs). Units for all metals and other elements are mg kg^{-1}.

Distance (m)	No. of BDAs	Al	As	Cd	Co	Cr	Cu	Fe	Mn	Mo	Ni	P	Pb	S	Ti	Zn
0	0	45257	15	10	0	0	16	196792	876	3	0	1031	0	4740	2629	210
0	0	17820	43	0	0	0	18	450341	323	0	0	2745	0	2613	404	189
0	0	18144	42	0	0	0	11	164720	800	0	0	1070	10	5124	2951	253
0	0	12971	52	0	60	29	22	59639	458	5	10	586	43	4047	5712	74
0	0	20724	27	0	0	0	13	210020	1074	0	0	974	0	5315	2402	160
0	0	56886	14	0	0	0	21	327115	568	0	0	871	0	4074	1240	192
0	0	26614	29	0	0	0	19	281524	791	3	0	514	0	4992	2063	143
0	0	8097	148	0	0	0	16	279780	784	4	0	486	0	5008	1956	146
0	0	23988	35	0	0	27	17	89180	2194	14	16	1757	31	701	4043	258
0	0	19372	38	0	0	0	10	244024	508	0	0	773	8	7094	1509	247
102.0	0	52616	11	0	0	43	27	39345	1621	7	18	771	52	414	5956	124
185.0	0	58633	11	0	49	38	35	43673	840	6	13	671	58	1195	5903	88
185.5	1	58598	10	0	31	44	35	46560	1176	9	18	812	53	1247	5811	133
291.0	2	59761	10	0	29	24	30	38380	697	6	10	544	56	613	6011	71
542.0	3	63209	12	0	67	38	32	58219	1857	9	20	786	46	298	6309	156
582.0	4	62257	11	0	82	45	36	55894	1077	14	20	756	57	588	6338	123
663.0	5	58244	11	0	57	30	30	41985	1047	6	14	547	44	204	6503	90
679.0	6	57548	11	0	43	44	38	38792	856	7	16	554	46	299	5794	90
691.0	7	55843	10	0	107	39	36	47322	878	9	14	578	48	397	6622	89
758.0	8	57125	10	0	28	30	36	39549	1021	5	18	568	46	267	6168	96
778.0	9	57147	11	0	68	25	30	44154	986	12	19	566	48	421	6403	101
813.0	10	56206	10	0	41	40	33	42309	923	5	13	575	48	404	6131	107
829.0	11	57897	11	0	69	44	26	41771	550	0	14	628	42	557	5906	88
1473.0	11	57897	11	0	0	38	29	44525	1156	5	15	748	53	529	6052	113

Zn and Mn concentrations in sediment samples exhibited a pattern of many irregular variable gradients over distance, crossing the sediment contaminant threshold twice (Fig. 5K, L). Zn and Mn concentrations in sediment samples showed no statistically significant correlation with distance from the mine site and number of BDAs (Table 4).

Table 4. Results of Spearman's rank correlation tests between heavy metal concentrations in water samples and distance from the mine site, and between heavy metal concentrations and the number of BDAs along Bear Creek in Lincoln National Forest. Correlation coefficient (rho; ρ) and associated p-values are reported for each element. (*) denotes statistical significance at the 0.05 level for each element.

Heavy Metal/ Element	Distance from mine site		Number of BDAs	
	p-value	rho (ρ)	p-value	rho (ρ)
Arsenic	0.9520	0.0178	0.8829	0.0434
Cobalt	*0.0223	0.6035	*0.0198	0.6128
Chromium	0.1089	0.4472	0.1414	0.4137
Copper	*0.0438	0.5452	*0.0492	0.5340
Iron	0.2145	-0.3538	0.2354	-0.3392
Manganese	0.1098	0.4462	0.1051	0.4515
Molybdenum	*0.0151	0.6331	*0.0186	0.6178
Lead	*0.0461	-0.5402	*0.0299	-0.5795
Zinc	0.7963	0.0760	0.8017	0.0739

Zn and Mn concentrations in sediment samples loaded negatively and positively on the first principal component (PC1 = -0.237 and 0.128, respectively; Table 5). The negative value indicates Zn concentrations decline as both predictor variables increase farther downstream. Additionally, the concentrations of both Zn and Mn in water samples spiked 1.5 km downstream from the mine (Fig. 4F, D).

Pb, Ti, and Mo concentrations in sediment samples were each consistently high across almost all sampling locations (Fig. 5M-O). Mo concentrations in sediment were above the soil threshold across all sampling locations, other than a singular dip 830 m from the mine site (after passing through 11 BDAs), before reaching elevated levels again 1,500 m downstream.

Table 5. Loading scores of element/metal concentrations for principal components one, two, and three (PC1, PC2, and PC3, respectively) on streambed sediments. Positive and negative loading scores indicate the direction and strength of each element's contribution to the primary axis. NA represents no data.

Element	PC 1	PC2	PC3
Al	0.269	N/A	0.405
As	-0.192	N/A	-0.571
Cd	N/A	N/A	0.651
Co	0.245	0.218	N/A
Cr	0.315	N/A	N/A
Cu	0.305	0.111	N/A
Fe	-0.300	N/A	N/A
Mn	0.128	-0.617	-0.100
Mo	0.230	-0.363	-0.190

Ni	0.315	-0.181	N/A
P	-0.150	-0.412	N/A
Pb	0.319	N/A	N/A
S	-0.299	0.142	N/A
Ti	0.323	N/A	N/A
Zn	-0.237	-0.433	N/A

Pb concentrations in sampled water were higher at the initial five BDAs sampling locations relative to the downstream BDA locations (Fig. 4H), but concentrations were high throughout. The concentrations of Mo and Pb in water samples never crossed contaminant thresholds (Table 2), but each did exhibit a peak at the mine site (Fig. 4G, H), while sediment concentrations at the mine site were all variable for three elements: Mo, Pb, and Ti.

Table 6. Variation partitioning analysis results on streambed sediments showing the contribution of BDAs and distance from the mine site on metal concentrations.

Element	Predictor	Contribution	R^2	Adjusted R^2
Al	BDAs	0.1485	0.3369	0.3068
	Distance	0.2501	0.4341	0.4084
As	BDAs	0.0397	0.1549	0.1165
	Distance	0.0759	0.1895	0.1527
Cd	BDAs	-0.0316	0.0349	-0.0159
	Distance	-0.0246	0.0283	-0.0089
Co	BDAs	0.1032	0.2579	0.2241
	Distance	0.0851	0.2405	0.2059
Cr	BDAs	0.1406	0.3284	0.2978
	Distance	0.2298	0.4137	0.3871
Cu	BDAs	0.1480	0.3334	0.3031
	Distance	0.2533	0.4341	0.4084
Fe	BDAs	0.1191	0.2977	0.2647
	Distance	0.1861	0.3607	0.3316
Mn	BDAs	0.0149	0.0000	-0.0455
	Distance	0.0291	0.0137	-0.0312
Mo	BDAs	0.0073	0.0377	-0.0060
	Distance	0.0561	0.0844	0.0428
Ni	BDAs	0.1339	0.3092	0.2778
	Distance	0.2320	0.4029	0.3758
P	BDAs	0.0199	0.1271	0.0818
	Distance	0.0255	0.1217	0.0874
Pb	BDAs	0.1335	0.2918	0.2596
	Distance	0.2500	0.4032	0.3762
S	BDAs	0.1739	0.3958	0.3683

	Distance	0.2580	0.4763	0.4525
Ti	BDAs	0.1693	0.3838	0.3558
	Distance	0.2616	0.4721	0.4481
Zn	BDAs	0.1127	0.2907	0.2585
	Distance	0.1362	0.3132	0.2820

Note: Variation partitioning analysis does not provide *p*-values. Negative values indicate neither of the predictors meaningfully explains the variation in metal or element concentrations.

4.0 Discussion

4.1 Assessing water quality conditions in Bear Creek based on physicochemical parameters

The finding of low water quality at the mine site is consistent with previous studies that identified elevated turbidity, conductance, and specific conductance (McAuley & Kozar, 2006; Sadar, 2017), low dissolved oxygen levels, and the presence of toxic heavy metals such as Cd and Pb at a mine (Saviour, 2012). Relative to the historic mine site, the downstream and reference sites generally exhibited lower turbidity and specific conductance and higher dissolved oxygen, indicating better water-quality conditions overall. Nevertheless, the historic mine site showed the most degraded water-quality conditions compared to the concentration levels of the National Recommended Water Quality Criteria for aquatic life. Elevated turbidity potentially decreases dissolved oxygen levels, which increases biological oxygen demand and causes hypoxia, resulting in a high risk of mortality in aquatic organisms (Nan et al., 2025; Ryan, 1991). Previous studies identified abandoned mine sites as sources of acid mine drainage enriched in suspended solids and heavy metals, which degrade water quality (Chileshe et al., 2020; Neuberger et al., 1990; Sheoran & Sheoran, 2006). Our study corroborates these findings.

4.2 The historic mine site functions as a point source of metal pollution

The results support the hypothesis that the historic mine site exhibits contaminant concentrations above the U.S. Geological Survey soil contaminants and U.S. Environmental Protection Agency threshold values and acts as a point source of Cd, Fe, S, Zn, and Mo, and to some degree, Pb and Ti relative to the adjacent stream. Although a lot of methodologies exist for

assessing environmental health risks associated with heavy metal contamination in soils, most do not account for the specific elements or chemical forms. The United States Environmental Protection Agency (U.S. EPA) has established dose-based guideline values that evaluate potential environmental risks associated with elements present in soils. The approach used to determine the threshold values for contaminated (elevated) sediments was based on the national percentiles (Ander et al., 2013; Smith et al., 2014); however, an enrichment factor analysis could provide additional insight into the degree of anthropogenic enrichment and the specific influence of historic mining activity.

Cd is naturally derived from weathering and mining of lead-zinc ores (Y. Zhao et al., 2020); thus, Cd co-exists with lead and zinc, especially in contaminated environments due to their geochemical relation (Tuhý et al., 2021). The result is consistent with previous studies that identified higher Cd concentrations in mine sites than in downstream areas (Huang et al., 2020; Sun et al., 2010; Zhu et al., 2019). Generally, Cd has a strong affinity for organic matter, and this immobilizes Cd (Autier & White, 2004). However, at the historic mine site, lower pH conditions increase Cd mobility, resulting in elevated dissolved concentrations in sampled water (Nordstrom & Alpers, 1999; Wang et al., 2026). Aquatic organisms found in the Bear Creek and downstream watershed may experience high exposure to Cd concentrations, which have detrimental effects. Elevated Cd concentrations at the historic mine site indicate potential ecological concern; however, biological exposure and organism-level effects were not directly assessed in this study. Although As concentrations remained below the freshwater threshold, even lower exposure to As concentrations can cause adverse biochemical and physiological effects (Han et al., 2019), including poisoning, reduced reproduction and growth, and cell death in aquatic species (Ahmed et al., 2013; D'Amico et al., 2014).

Fe is an essential nutrient for algae and other organisms; however, high Fe content may also have deleterious effects on aquatic invertebrates, such as decreasing species abundance and diversity (Cardwell et al., 2023; N. Zhao et al., 2025). Higher Fe concentrations in sediments may influence contaminant adsorption and mobility (Swanson et al., 2023). Although iron oxide can increase P sorption capacity (Dayton & Basta, 2005), iron oxide concentrations were not measured directly in our study. This may explain the higher P content in sediment samples detected at the mine site. This is consistent with a previous study that identified high iron pollution in mine sediments, which was correlated with increased amounts of P pollution (Kommana et al., 2023).

Sulfur supports the structure of proteins and forms part of living organisms (Luo, 2018). Elevated S content at the mine site could be due to acid mine drainage, which occurs when minerals containing a reduced form of S, usually pyrite (FeS_2), reacts with water and oxygen to produce sulfuric acid, mobilizing acid-soluble metals (Soucek et al., 2003). Findings from this current study are consistent with a previous study that identified higher content of sulfur at a mine site (Sołek-Podwika et al., 2016). Sulfuric acid may enter soil and water through the dissolution of pyrite, which causes acidification of water bodies (Shamshuddin et al., 2004) and impacts aquatic organisms in freshwater by making it uninhabitable (Elphick et al., 2011; Talukder et al., 2024).

Zn functions as a micronutrient needed by plants; however, high Zn concentrations above normal physiological values are considered toxic (Fashola et al., 2016). Zn is commonly present in high concentrations, especially in acid-mine drainage, which contaminates surface waters, groundwaters, stream sediments, tributaries, and adversely affects aquatic organisms (Schmitt et

al., 2005). Exposure to higher levels of Zn contamination in aquatic organisms reduces biodiversity, population size, reproductive size, and development of fish (Johnson et al., 2016).

Pb is naturally non-degradable, and even low exposure to Pb is toxic (Fashola et al., 2016). High Pb content observed at the mine site may be explained by lead's strong affinity for organic matter, clays, and iron oxides, which promotes precipitation and sorption into insoluble oxalate, which becomes immobile in soils (Davidson, Hursthouse, Tognarelli, Ure, & Urquhart, 2004). Elevated Pb concentrations from mine waste can cause adverse effects on aquatic biota, such as reducing fish populations (Brabant et al., 2025; Kovacik & Helczman, 2025).

4.3 The influence of distance from mine site and number of BDAs on metal concentration along Bear Creek

The results suggest that all of the elements that exhibited elevated concentrations at the historic mine site (including As, Cd, Fe, S, P, Mn, and Zn in sediment samples as well as Cr, Mo, and Pb in sampled water) were detected at lower concentrations in downstream locations. Metal concentrations generally declined with increasing distance downstream from the mine site, possibly due to natural attenuation processes such as dilution, adsorption, and precipitation that reduce dissolved metal transport (Nash & Stillings, 2003). However, local geology may also influence these patterns, as natural rock weathering and geochemical background can contribute to measured metal concentrations (Herndon et al., 2025; Shahrestani et al., 2024). In addition, as pH increases and acid mine drainage is progressively neutralized, heavy metals precipitate as hydroxides or scavenge onto reactive mineral surfaces, which attenuates and dilutes metal concentration downstream (Berger et al., 2000). A previous study identified metal concentrations in mine sediments as a function of distance from the point source of pollution (Lin et al., 2022b). That study, combined with our findings support the hypothesis that dilution over distance is a

stronger determinant of metal concentration than BDAs filtering for most of the metals that were tested.

Evidence indicated that downstream distance from the historic mine site was a strong determinant of many metal and other pollutant concentrations, which corroborates previous studies (Peplow, 1999; Razo et al., 2004). Metals and elements that support this hypothesis in our study include Al, As, Cd, Cr, Fe, Pb, Mn, Mo, Ni, P, S, Ti, and Zn. This is likely due to dilution, dispersion, adsorption, and/or neutralization (McAuley & Kozar, 2006). Previous studies found that heavy metal concentrations decreased with increasing distance from the mine area, possibly due to sedimentation and migration (Ding et al., 2017; Velea et al., 2009). This finding is consistent with previous studies and supports the hypothesis that dilution over distance is a stronger driver of many elements and heavy metal concentrations than BDAs filtering (Lin et al., 2022b; S. Nie et al., 2024).

However, the greater explanatory power of BDAs over distance for Cu and Co contaminants underscores BDAs' effectiveness as biogeochemical filters. The observed decline in Cu concentrations may, in part, be because Cu has a strong binding affinity for a large number of functional groups, including clays and organic matter (Park et al., 2016). This enhances precipitation of insoluble copper sulfide (CuS) and eventual immobilization within streambed sediments and consequently decreases their concentrations in sampled water downstream. Similarly, the decline in cobalt concentrations is likely due to dissolution of cobalt under low pH to form an aqueous solution, which becomes immobilized by co-precipitation or sorption onto iron and manganese oxides (Ziwa et al., 2020). Our results suggest that BDAs may contribute to attenuation of Cu and Co; however, distance from the mine site remained the stronger overall predictor of declining metal concentrations. This interpretation is consistent with previous

studies showing that beaver-related restoration structures can improve water quality and retain some sediment-associated contaminants (Brazier et al., 2021; Neace, 2023). More broadly, these findings are comparable to findings reported from other mining-affected watersheds where metal concentrations decline downstream from point sources but may still reflect additional local or geologic influences (Manning et al., 2024; Shahrestani et al., 2024). Although enrichment analysis was not conducted in this study, such an approach could help further clarify the degree of mining-related pollution and the extent of ecosystem change.

4.4 Evidence of multiple sources of metal contamination in the riparian corridor

P, Zn, and Mn concentrations all supported the hypothesis that multiple sources of pollution of these elements may exist in Big Bear Canyon. Pb, Mo, and Ti concentrations in sediment samples crossed the USGS threshold at many distinct points downstream, indicating that more than one source exists, although the mine site showed concentrations just below the threshold. Pb, Mo, and Mo results all supported the hypothesis that concentrations of these metals remained high throughout the stream, regardless of any downstream dilution or BDA filtering. Elevated Ti concentrations may reflect its natural abundance in the Earth's crust (Rudnick & Fountain, 1995; Rudnick & Gao, 2014), where it occurs widely in nearly all rocks and sediments (Schulz et al., 2017). Previous studies indicated that higher Ti concentrations potentially cause severe gill tissue damage in mussels (Sibiya et al., 2025) and changes in the diversity of fauna compositions (Olsgard & Hasle, 1993). Mo concentrations in our study were elevated, with the mine site and BDAs 4 and 9 (14, 14, and 12 mg kg^{-1}, respectively) and exceeded 10 mg kg^{-1} possibly because Mo has a high affinity (adsorption) for oxide minerals, particularly Al, Fe, and Mn (Kurmysheva et al., 2023; Li et al., 2024). Besides the mine, other

potential sources of pollution could be due to natural rock weathering (Niu et al., 2021) or other unknown historic mine activities within the region.

5.0 Study limitations

One limitation of this current study is that the distance from the mine site and number of BDAs were highly collinear. While all lines of evidence suggest that distance was a stronger predictor overall, the apparent influence of BDAs as filters of heavy metals may have been dampened because of this collinearity (Molnar, 2020). The study was also based on a one-time sampling event and therefore does not capture seasonal variability. In addition, vegetation-related factors, including plant type, species composition, and life cycle, were not evaluated in this study, although they may influence filtering process and flow reduction in the riparian corridor.

The BDAs used in this study were originally installed primarily as a postfire erosion control measure; therefore, any downstream metal filtering observed in the Bear Creek should be considered as an additional benefit rather than a primary management objective. Our findings support the idea that it may be possible to strategically place BDAs to intercept locations of high pollution, perhaps especially for Cu and Co, but for many other metals, the influence may be weaker than the influence of natural dilution downstream. Managers in Lincoln National Forest and elsewhere can apply BDAs as an ecological restoration tool to improve water quality and maintain healthy habitats for aquatic organisms, especially when those BDAs are intended to serve another primary purpose, such as erosion control.

6.0 Conclusion

In summary, elevated concentration levels of Cd, Cr, Co, Fe, Pb, Mn, Mo, P, S, Ti, and Zn were observed in water and sediment samples near the historic mine site in Bear Creek, New Mexico. The historic mine appears to be an important source of contamination, although some contaminants may also reflect additional downstream or background sources. Distance from the

mine site was the primary determinant of declining metal concentrations for most elements, more so than BDA filtering. Among the heavy metals and other elements analyzed, only Cu in water and Co in sediments showed notable evidence consistent with BDA-related attenuation. Overall, these findings suggest that downstream dilution and attenuation played a stronger role than BDAs in explaining most observed patterns, although BDAs may still provide secondary water-quality benefits within riparian restoration. These conclusions are based on both water and sediment results. Although the results suggested distance was the major influence on metal concentrations, BDAs provide a critical management tool within riparian ecosystem restoration, including a possible contribution to the reduction of heavy metal contamination.

Acknowledgments

The authors would like to extend our sincere gratitude to Dr. Kevin Wertz and Emeritus Dr. Melanie Barnes, Research Scientists from the Department of Geosciences at Texas Tech University. We express our profound gratitude to Austin Ibarra, Davis Mathews, Macy Tran, and McKenzie Campos for their help with data collection. We are grateful to Mr. Larry Cordova for his collaboration in providing information about the Beaver Dam Analogs in Big Bear Canyon.

Author contributions

George K. Gyan: Data curation, Conceptualization, Formal analyses, Investigation, Methodology, Project administration, Validation, and Visualization. **Nathan S. Gill:** Data curation, Conceptualization, Formal analyses, Investigation, Methodology, Project administration, Validation, and Visualization. **Reynaldo Patino:** Conceptualization, Methodology, Writing – review and editing. Also provided input in the relevant sampled water data needed to conduct statistical analysis. **Matthew G. Siebecker:** Data curation, Conceptualization, Methodology, and pXRF sediment lab analysis. The first draft of the

manuscript was written by **George K. Gyan**, and all authors provided extensive inputs on different versions of the manuscript. All authors read and approved the final manuscript.

Funding

This research was supported by United States Department of Agricultural-National Institute of Food and Agriculture (USDA-NIFA grant number # 2020-38422-32244). This research was supported by a Texas Tech strategic initiative award with funding from the State of Texas TUF (Texas University Fund).

Data availability

Datasets are available upon request from the corresponding author.

Declarations

Ethical responsibilities of authors

All authors have read, understood, and complied as applicable with the statement on "Ethical Responsibilities of Authors" as found in the Instructions for Authors.

Competing interest

The authors declare that there are no known financial or personal relationships that could be perceived as potential conflicts of interest.

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