River Research and Applications

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RESEARCH ARTICLE OPEN ACCESS

Off-Highway Vehicle Use Degrades a Mountain Stream: A Case Study Evaluating Recreation Impacts, Restoration Efficacy, and Assessing Vulnerability of Headwater Streams in the Central Rocky Mountains

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Received: 4 October 2024 | **Revised:** 7 February 2025 | **Accepted:** 14 February 2025

Funding: This work was supported by Colorado Parks and Wildlife; U.S. Forest Service.

Keywords: BACI | degradation | headwater stream | high-clearance road | open-water crossing | restoration

ABSTRACT

Headwater streams are abundant in mountain landscapes and where municipal watersheds occur on public lands; they co-occur with road networks open to off-highway vehicle (OHV) use. Despite well-known, deleterious interactions of roads and streams, few studies have empirically examined the biophysical impacts of OHV use where backcountry roads enter streams at open-water crossings. We present a case study using a controlled design where we quantified OHV road impacts in a Rocky Mountain stream and evaluated the restoration efficacy by examining aquatic biota and habitat before and after restoration treatments. Prior to restoration, stream habitat structure was simplified and the abundance, biomass, and diversity of aquatic biota were severely reduced in OHV-affected compared to reference reaches. Restoration included the removal of riparian OHV roads, rebuilding streambanks and bedforms, and additions of large wood. Following restoration and 2 years of spring snow-melt runoff, some, but not all, aquatic habitat had recovered, and aquatic biota were more similar to conditions in reference reaches. We included a vulnerability assessment for streams draining ~5 M Ha of public lands in the Rocky Mountains, indicating the greatest threat of similar OHV impacts to 2nd- and 3rd-order streams in watersheds deemed important for municipal supplies. Our study provides insights that may aid in effective prevention approaches and restoration of similar sites.

1 | Introduction

Headwater streams amass the greatest stream lengths in watersheds (Wohl 2017), support numerous ecological functions (Hill et al. 2014; Ferreira et al. 2023), and contribute biological productivity to larger streams in well-connected, functioning watersheds (Wohl 2017; Colvin et al. 2019). Often seen as source areas for water yield in a watershed (Alexander et al. 2007; Hill et al. 2014; Wohl 2017), the integrity of such small streams is intricately linked to adjacent terrestrial processes and land use conditions (Gomi et al. 2002), making headwaters vulnerable to a variety of land management activities (Richardson 2019). In montane landscapes across much of the United States, the headwaters of many watersheds occur on public lands where backcountry roads provide access for management activities (e.g., timber harvest, grazing) and critical infrastructure for energy transmission and water delivery (Kampf et al. 2021). Beyond management

https://doi.org/10.1002/rra.4433

needs for public services, abundant unpaved and unimproved roads on public lands also provide access for recreational use (Havlick 2002; Federal Highway Administration 2015) including motorized (e.g., camping, off-highway vehicle [OHV] driving, snowmobiling, boating) and non-motorized recreation (e.g., backcountry camping, hiking, paddling). Despite the dominance of small streams relative to larger streams, the numerous calls highlighting the role and importance of headwater streams (e.g., Lowe and Likens 2005; Alexander et al. 2007; Kampf et al. 2021), and the amount of water supply derived from them (Brown et al. 2008; Liu et al. 2022), studies on the ecological integrity of low-order streams in the context of land management are not as common as studies on larger-order streams (Freeman et al. 2007; Richardson and Danehy 2007; Richardson 2019).

Environmental impacts of roads have been well studied and are the subject of extensive reviews (e.g., Trombulak and Frissell 2000; Gucinski 2001). Roads in forests and wildlands have been linked to physical resource degradation in the form of surface erosion (Bilby et al. 1989; Montgomery 1994; Luce and Black 1999), hillslope erosion (Montgomery 1994; Luce 2002), poor water quality (Clinton and Vose 2003; Sheridan et al. 2006), and stream sedimentation (Furniss et al. 1991; Waters 1995). Sediments derived from eroding road networks may cause changes in fish and macroinvertebrate communities by degrading spawning habitat for fish (e.g., Eaglin and Hubert 1993) and filling interstitial living space for macroinvertebrates (e.g., Sarriquet et al. 2007). Traffic on backcountry roads may displace organisms through noise, cause direct mortality following collisions, and contribute to a loss of biodiversity (Ouren et al. 2007, Robinson et al. 2010). Degradation in aquatic habitats is most pronounced when roads are hydrologically connected with the stream network due to the conveyance of runoff and sediment at road–stream crossings (Eaglin and Hubert 1993; Al-Chokhachy et al. 2016). Stream crossings not only extend the hydrologic network in roaded watersheds, leading to increased peak flows and stream sedimentation (Jones et al. 2000, Wemple and Jones 2003), but also include physical infrastructure, such as culverted stream crossings, which often fragment aquatic habitat networks (Diebel et al. 2015) and reduce the extent of habitat available (Gibson et al. 2005).

Growth in OHV recreation over the past decades has highlighted conflicts for land managers (e.g., Havlick 2002; Ouren et al. 2007; Albritton and Stein 2011) and challenged attempts to balance public access and natural resource protections (e.g., Yankoviak 2005; Shilling et al. 2012; Thomas and Reed 2019). Recent assessments estimated nearly 50-M recreational OHV riders across the U.S. (Cordell et al. 2005; Cordell 2012) and found that the popularity of OHV recreation was greatest in rural areas where use is allowable on public lands, with public participation rates reaching a fifth to over one-quarter of adults in western, southern, and mid-western regions (Cordell 2012). Herein, we use OHV, similarly to Marion et al. (2014), inclusive of any motorized high-clearance vehicle (e.g., all-terrain, four-wheel drive, utility) capable of travel on forest roads and trails. OHV recreation is expected to grow by 29%–56% over the coming decades (Cordell 2012) as technological advances in the OHV manufacturing shift user access from niche user markets to the general population (see Yankoviak 2005).

With the growth of human populations in locales neighboring scenic public lands, the reach of recreational influence into relatively pristine environs has expanded (Havlick 2002), particularly for motorized vehicle–based outdoor recreation (Burgin and Hardiman 2012). Further, OHV use on public lands occurs on large portions of the 234,000 miles of unpaved, high-clearance public roads managed by the federal land management agencies (e.g., Forest Service and Bureau of Land Management), most of which are aging infrastructure and not designed to any standard (Federal Highways Administration 2015). Given the amount of unmaintained low-volume forest roads and the lack of comprehensive OHV management across federal lands, OHV activity has been and remains a significant threat to riparian ecosystems across the western U.S. (Theobald et al. 2010). Further, the juxtaposition of areas receiving high demand for OHV use and high-clearance road networks within municipal watersheds may pose long-term water resource management issues for rural communities reliant upon public lands for such ecosystem services.

We examined the impacts of a high-clearance, low-volume road with frequent OHV use on aquatic habitat and stream communities of a Rocky Mountain stream and subsequently identified other regional headwater streams likely vulnerable to impacts from similar OHV use. Objectives of the field examinations were to (1) determine if OHV use in a stream affects trout, macroinvertebrates, and stream habitat, (2) determine if trout and macroinvertebrates respond to road removal and habitat restoration in OHV-affected reaches, and (3) if trout abundance changed in OHV-affected or OHV-restored reaches, then determine which habitat factors were linked to positive or negative responses. To address these objectives, we used a before–after control–impact (BACI) study design to compare stream habitat characteristics, relative abundance, biomass, and body condition of resident trout, as well as the abundance, diversity, richness, and composition of macroinvertebrates in reference reaches and reaches affected by OHV use before and after restoring habitat in OHV-affected reaches. We predicted the following for responses before and after the removal of OHVs and restoration:

- Before restoration
    a. Trout and macroinvertebrates would be present but in reduced abundance.
    b. Stream habitat would be less complex.
    c. Streambed composition would contain more fine sediment.
- After restoration
    a. Trout and macroinvertebrates would increase in abundance.
    b. Stream habitat would be more complex.
    c. Streambeds would have less fine sediment.

For our last objective, we used geospatial techniques to assess other headwater streams that could be vulnerable to OHV impacts on National Forest lands within the Front Range of the Central Rocky Mountains, Colorado, USA. Aspects of this investigation are discussed in the context of road infrastructure management, environmental management of drinking water supply watersheds, and aquatic resource conservation in headwater streams on public lands.

2 | Materials and Methods

2.1 | Study Area

This study occurred along Jenny Creek, a headwater stream within the South Boulder Creek watershed located at approximately 2925 m asl in Gilpin County, Colorado. The stream heads from two alpine lakes and descends into a mixed spruce-fir coniferous forest type overlying the glacial deposits derived from the weathered felsic and biotite gneiss geology (Gable 1969; Young 1991). The U-shaped valley geometry and coarse glacial deposits support mesic patches of mixed conifer (*Abies lasiocarpa*, *Picea engelmanni*, and *Pinus contorta*) subalpine forest and glacial wetlands fed by shallow groundwater seeps. The stream channel is moderately confined by valley geometry, producing a step-pool to riffle-pool geomorphology controlled by large clasts of boulders and inputs of large wood and dominated by a 3- to 5-m wide gravel-cobble streambed. Understory riparian vegetation varies in extent and patch size but primarily consists of *Salix* spp., subalpine forbs (e.g., mountain bluebells *Mertensia ciliata*, arrowleaf groundsel *Senecio triangularis*, twisted stalk *Streptopus amplexifolius*), *Equisetum* spp., moss, and ferns. Aquatic fauna are predominately eastern brook trout (*Salvelinus fontinalis*) and typical macroinvertebrate taxa (Ward and Kondratieff 2002). A historic wagon trail (circa 1890) traversed 1.21 km of the valley bottom and had become a popular OHV route with an expanded network of trails within, crossing, and adjacent to Jenny Creek.

2.2 | Study Design

We used a BACI experimental study design to evaluate the biophysical characteristics of Jenny Creek before (study years 2011, 2013, and 2014) and after (study year 2016) stream restoration treatments. For our study, we use the restoration treatment as the "impact" aspect of the experiment and focused comparisons of OHV and restoration effects on trout, macroinvertebrates, and habitat where "control" (i.e., reference) and degraded (i.e., OHV-affected) reaches were sampled concurrently before and after the "impact" of restoration where OHVs were removed and habitat was restored. Reference reaches were located primarily in upstream extents where the trail was > 50 m from the stream. OHV-affected, later OHV-restored, reaches were primarily located in the downstream extent where the trail crossed and was adjacent to Jenny Creek. Fish community, macroinvertebrate community, and stream habitat characteristics were quantified from stream reaches stratified into two groups: (1) areas adjacent to the trail or showing visible evidence (e.g., tread marks along streambanks, logs, and boulders) of OHV use in the stream (OHV-affected) and (2) areas buffered by at least 50 m of intact riparian area without signs of OHV use or other recent anthropogenic influence (Reference) based on the efficacy of riparian buffers in other mountain streams (Sweeney and Newbold 2014). In September 2014, traffic was removed from degraded stream reaches, the OHV trail was rerouted to an upland area > 50 m from Jenny Creek, and stream habitat and riparian areas were restored by obliterating 1200 m of riparian OHV trail route, reconstructing bedform features (i.e., steps, riffles, pools), reconstructing streambanks, and placing large wood throughout degraded portions of Jenny Creek. Restoration treatments occurred over the course of a 4-week period during base streamflow (~ 0.14 m^3 per s; USGS 2019). Biophysical evaluations were repeated in August 2016 to capture the "after" (i.e., post-restoration) characteristics for stream fauna and habitat, following two cycles of spring runoff, organismal colonization, and biological production. Following restoration, areas previously evaluated as "OHV-affected" reaches were evaluated as "OHV-restored" in comparison to the "reference" reaches.

Resident fish communities and stream habitat factors were surveyed within variable-length reaches (minimum of 40 m) classified as either OHV-affected, OHV-restored, or reference. Reaches varied in total length depending on breaks in macrohabitat features (i.e., pools, cascades, riffles) and in approximate location by year. Most, but not all, reaches were repeated in the same approximate configuration; however, interannual shifts in habitat features (i.e., movement of riffle crests) resulted in adjustments to reach start and end. Reference reaches were spaced a minimum of 15 m apart. Benthic macroinvertebrates (BMIs) were sampled from reaches stratified as OHV-affected or reference, some within the bounds of fish-habitat reaches, but others outside of the fish-habitat reaches.

2.3 | Trout Evaluation

Trout were collected in each reach using the two-pass removal method, relying on natural breaks in habitat units, particularly at upstream endpoints of reaches, such as the base of steep riffles or log drop features, where fish were unlikely to emigrate from the reach. Trout > 65 mm were measured to the nearest millimeter (total length) and weighed to the nearest 1 g (2014 and 2016). Estimates of population abundance (i.e., number of individuals per kilometer) were derived for trout > 100 mm with Seber (1982) and Seber and LeCren (1967) regressions of two-pass depletion data using JOM (JakeOMatic) version 2.4 (Rogers 2006; Colorado Parks and Wildlife 2023). The same program was used to determine biomass (kg per Ha) of all trout in 2014 and 2016 when weights were recorded. Relative weight index (W_r) for brook trout (Hyatt and Hubert 2001) and Fulton's K condition factor were calculated for each fish > 120 mm to compare the fish condition in OHV-affected, reference, and OHV-restored reaches. Outliers of individual trout weights were detected using exponential regression and inner-quartile analyses, then removed from analyses of trout biomass and condition indices. We did not find outliers in other trout data.

2.4 | Stream Habitat Evaluation

Habitat conditions (i.e., pool frequency, mean proportions of stable streambanks, mean depth, mean maximum depth, and mean width: depth ratio) were evaluated along systematically placed transects in each reach, where wetted width, depths at four equidistant intervals, and maximum depth were measured to the nearest 0.01 m. Transects were placed at 10-m increments in 2011 and 5-m increments in 2014 and 2016 surveys. At each transect, habitat type was categorized as either predominantly pool or riffle. Bank stability was visually estimated as percent stable for the length of the stream between transects. Whole-reach channel substrate composition was determined by the zig-zag Wolman pebble count (Wolman 1954), wherein we measured the medial axis of a minimum of 100 inorganic streambed particles between the margins of the bankfull channel following Overton et al. (1997). All large wood within the reach at least half as long as bankfull width and greater than 0.1 m in diameter was measured to determine wood volume. The total volume of large wood was summed and reported as cubic volume (m^3) of wood per kilometer of stream.

2.5 | Macroinvertebrate Evaluation

Density, diversity, richness, and composition of BMIs were assessed in reaches stratified as OHV-affected/OHV-restored or reference by sampling five fast-water (e.g., riffle) habitats using a 500-μm surber sampler with a 30 by 30-cm grid (0.09 m^2) following the fixed area procedure outlined by Hawkins and others (2003). In reaches lacking distinct pool features, additional benthic samples were collected within the same riffle using the randomization method of Hawkins et al. (2003). Contents of the five surber-net collections were combined into a bucket, filtered, poured through a 500-μm screen, and transferred to 500-mL bottles to create a single composite sample to represent the BMI community for the reach. Composite samples were preserved in 70% EtOH and processed using standard methods of Barbour and others (1999), with a minimum sort counting and identification of 300 organisms to the lowest practical taxonomic unit. Samples with high organism abundance, > 300, were subsampled following Barbour et al. (1999), and density calculations were extrapolated (Vinson and Hawkins 1996). All BMI sample processing, sorting, and taxonomic identification were performed by Timberline Aquatics LLC in Fort Collins, Colorado. BMI response metrics were calculated for (1) taxonomic richness and diversity, (2) total, Ephemeroptera, Plecoptera, Trichoptera (EPT), and Diptera density, and (3) percent functional feeding groups (filterer, scraper, predator, shredder). Based on the resolution of taxonomic identification, organisms were resolved to operational taxonomic units (OTUs) for comparisons of community assemblage structure. Taxa were assigned to functional feeding groups based on sources built into the EDAS Program (Jessup 2009), including Merritt et al. (2019).

2.6 | Data Analyses

We compared trout responses and stream habitat conditions among OHV-affected, reference, and OHV-restored reaches using generalized linear mixed effects models (GLMMs; Bolker et al. 2009) to test for fixed effects of OHV use (degraded vs. reference), time (before or after restoration), and their interaction, which quantifies the restoration effect (see the Supporting Information for details). Random intercepts for site identity were included to account for multiple samples within a site. All GLMM comparisons followed this framework, though tests for some responses were parameterized differently based on data distributions and time dependencies of comparison (see the Supporting Information for details). We hypothesized that restoration activities would have a significant effect but only in the year after they were implemented, resulting in a significant OHV-by-time interaction. Note that for some measured variables (e.g., trout abundance, large wood volumes), small values were added to avoid log(0) occurrences using the data-driven method described in Stahel (2002). GLMMs were performed using the glmmTMB (Brooks et al. 2017) package in R (version 4.2.2, www.r-project.org).

Streambed composition in OHV-affected, OHV-restored, and reference reaches was compared using three approaches. First, cumulative proportions of fine sediments < 8 mm pooled by reach type were compared using an analytical tool (Potyondy and Bunte 2002) based on the confidence interval method described by Bevenger and King (1995). Eight millimeters was chosen based on the well-known deleterious effect fine sediment has on salmonids and BMI (see Waters 1995). Second, distributions of the streambed particle size between reach types were compared using two-sample conditional distribution tests following Hu and Lei (2024) to determine the difference between estimated size distributions, conditioned by slope (see Hyndman et al. 1996). This analytical approach tests for the general effect of OHV use and subsequent restoration on entire distributions of streambed particles by controlling for stream slope, a factor well known to influence the size of surface particles in a streambed (Knighton 2014). Finally, due to interest in any shifts in particle distributions, confidence intervals of 16th and 84th sample quantiles of particle size distributions were compared between reach types using methods and the GSDtools package developed by Eaton et al. (2019; see the Supporting Information for details). All analyses of streambed particles were restricted to similarly sloped reaches (i.e., 3%–6.2%), encompassing the range of slopes representative of each reach type. Apart from the fine sediment particle analysis, all analyses were performed in R 4.2.2.

To determine habitat factors that strongly influenced trout abundance responses among reach types and across years, a generalized linear model (GLM; McCullagh and Nelder 1989) Lasso regression was performed using the glmmLasso package (Groll 2017) in R, where the reach-scale predictors (i.e., slope, mean depth, maximum depth, width-to-depth ratio, pool frequency, large wood volume, and streambank stability) constituted the set of plausible explanatory variables. Random Forest regression was employed to confirm variable importance from the GLM Lasso using the randomForest (Liaw and Wiener 2002) package in R. Although included, random effects were ignored due to focal interest in which predictor variables influence trout abundance.

For BMI responses (e.g., density, EPT), fixed (i.e., OHV, time) and interaction (i.e., restoration) effects were analyzed using a similar model configuration described for trout and habitat analyses (see the Supporting Information for details). Total BMI and EPT densities were analyzed using GLMs with gamma distributions and log-link functions. Taxonomic richness and Shannon–Weiner diversity index were compared using GLMs with gaussian distributions and log-link functions. Random effects were not included in GLMs because BMIs were collected in different riffles between years. Permutational multivariate analysis of variance (PERMANOVA, Anderson 2017) was conducted for comparisons of functional feeding group percentages using the *adonis2* function in the R package vegan (Oksanen et al. 2022). To determine if the BMI community structure was altered by OHV (2013) or restoration (2016) relative to reference conditions, ordination analyses were performed for 2013 and 2016 separately by applying nonmetric multidimensional scaling (NMDS; Kruskal 1964) to taxon (i.e., OTU) count data using the *metaMDS* function in the R package vegan (Oksanen et al. 2022). A distance matrix was constructed with 20 random starts, and the solution with the lowest stress was selected. Bray–Curtis dissimilarity indices with 999 permutations were used for statistical comparisons of taxa vectors among OHV-affected, OHV-restored, and reference reaches by year.

2.7 | Stream Vulnerability Assessment

We assessed the vulnerability of streams to potential OHV impacts by identifying road–stream crossings with open-water fords on the existing motorized public road networks of National Forests within the Front Range Mountains of Colorado, USA (see the Supporting Information for details). A dataset of presumptive open-water stream crossings was produced from linear roads and motorized trails open to public use (USDA Forest Service 2019), the U.S. Forest Service culvert inventory data (USDA Forest Service n.d.), and the National Hydrography Dataset perennial stream data (USGS 2018) using intersect and query procedures in ArcMap (ESRI 2021) (see the Supporting Information). To corroborate geospatial outputs, the dataset was cross-validated with the locations of field-identified open-water fords by informally interviewing natural resource managers familiar with road networks on the assessed land base. Natural resource managers provided GPS coordinates for open-water crossings on roads they were familiar with or had visited personally. For geospatial analysis and field observations, descriptive statistics and spatial characterization of the extent and distribution of open-water fords on perennial streams are summarized in the context of Strahler stream order and watersheds with "moderate" to "high" values from the Index of Importance to Surface Drinking Water (IMP) reported by Mack et al. (2022).

3 | Results

3.1 | Trout and Habitat Responses

Adult trout (> 100 mm) abundance and biomass were > 80% lower in OHV-affected reaches compared to those in reference reaches prior to restoration, whereas estimated increases in abundance and biomass in OHV-restored reaches were 3-fold higher from pre-restoration conditions (Figure 1; Table 1). OHV-affected reaches were occupied by smaller individuals ($x_{OHV} = 125.1 \pm 4.2$ mm) compared to individuals in reference reaches ($x_{reference} = 141.6 \pm 3.3$ mm; Table 1, Figure S1). Following restoration, trout found in OHV-restored reaches were similar in length to those found in reference reaches (Table 1, Figure S1). There was an estimated overall decrease in the body condition of brook trout from 2014 to 2016 using either index (year effect [K: −22.8%, $p < 0.001$; W_r: −23.5%, $p < 0.001$]). Trout condition appeared to decline slightly as a result of the OHV "effect" (Figure S1); however, results were not significant (Table 1). The estimated percentage increase in Fulton's K and W_r due to the

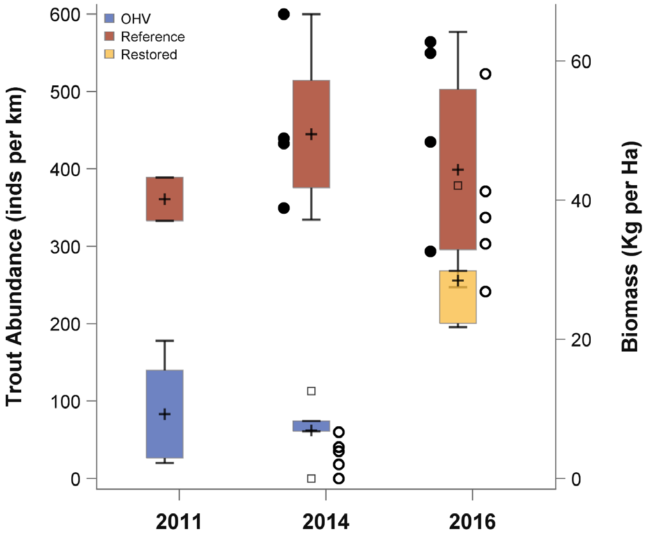


FIGURE 1 | Box plots of adult brook trout (> 100-mm length) abundance and scatterplots of trout biomass compared between reach categories (OHV [off-highway vehicle] impact, restored, reference), grouped along the *X*-axis as time series. Boxes represent the range in minimum and maximum densities observed in the 25th and 75th percentiles (the interquartile range, IQR); means shown as (+); whiskers represent the spread of data that fall within 1.5 times the IQR; and outliers (▫) are datapoints beyond 1.5 times the IQR. Closed circles (●) represent reference, and open circles (○) represent OHV-affected/restored reaches for biomass.

restoration was significant (Table 1), yet confounded by the overall decrease from 2014 to 2016.

We observed strong differences in habitat factors due to the OHV "effect" and estimated improvement for some factors due to the restoration "effect." Pool frequency was 1-fold lower in OHV-affected reaches in comparison to those in reference reaches, whereas post-restoration comparisons revealed pool frequencies in OHV-restored reaches being more similar to reference conditions following two cycles of spring snow-melt runoff (Figure 2, Table 1). Specifically, the estimated increase in pool frequencies in OHV-affected reaches following restoration was > 15-fold, essentially recouping the original loss of pools in OHV-affected reaches as compared to those in reference reaches (Figure 2, Table 1). We detected more than an 80% decrease in the volume of large wood within OHV-affected reaches in comparison to those in reference reach conditions (Table 1); yet following restoration and despite the intentional placement of large wood in OHV-restored reaches, the volume of large wood had not increased to the levels observed in reference reaches (Figure 2, Table 1). Width-to-depth ratios increased by ~51% due to the OHV effect and decreased by an estimated ~37.2% following restoration treatments (Table 1). Channel widths were often widest where the adjacent OHV road had eroded away the streambank and vehicles traversed the eroding bank or widened channel segment. For mean depth, we did not detect a strong OHV "effect," nor was there evidence that restoration had an effect relative to reference conditions (Table 1). Yet, maximum depths were reduced by nearly one-fifth in OHV-affected compared to those in reference reaches, whereas following restoration, they had increased an estimated 30% (Table 1), corresponding with the availability of pool habitat. Bank stabilities in OHV-affected

TABLE 1 | Statistical results (Wald test statistic z; significance p; effect size β) from generalized linear mixed models for measures of the trout community and stream habitat compared between reaches affected by off-highway vehicles (OHVs), then later restored in a before–after control–impact field study in Jenny Creek, Gilpin County, Colorado. Measures shown in bold font were statistically significant.

		$OHV_{affected}$ to reference (before effect)			$OHV_{restored}$ to $OHV_{affected}$ controlled for time (after effect)		
Response	**Distribution**	**z-value**	**p**	**β**	**z-value**	**p**	**β**
Trout abundance	Gamma	**−6.15**	**< 0.001**	**−84.7%**	**3.58**	**< 0.001**	**319.1%**
Trout biomass	Gamma	**−13.59**	**< 0.001**	**−86.3%**	**10.26**	**< 0.001**	**460.1%**
Trout length	Gamma	**−3.76**	**< 0.001**	**−19.4%**	**2.53**	**0.012**	**22.1%**
Relative weight (W_r)	Gamma	−1.43	0.153	−8.1%	2.74	0.006	17.0%
Condition (K')	Gamma	−1.71	0.088	−9.5%	2.96	0.0031	18.6%
Pool frequency	Poisson	**−3.69**	**< 0.001**	**−93.4%**	**2.63**	**0.008**	**889.5%**
Wood volume	Gamma	**−3.30**	**< 0.001**	**−84.6%**	0.36	0.722	n.e.
Width:Depth ratio	Gamma	**2.81**	**0.005**	**51.1%**	**−1.96**	**0.05**	**−37.2%**
Mean depth	Gamma	−1.64	0.1	n.e.	0.16	0.877	n.e.
Maximum pool depth	Gamma	**−2.85**	**0.004**	**−18.5%**	**2.22**	**0.027**	**29.3%**
Bank stability	Beta	−1.81	0.07	n.e.	1.61	0.107	n.e.

reaches compared to those in reference reaches, although lower, were not significantly different (Table 1). Likewise, although restoration in OHV-affected reaches appeared to increase bank stabilities, the change was not significant (Figure 2, Table 1).

For comparisons of streambed material, the composition and particle size distribution did not indicate that fine sediments had accumulated in OHV-affected reaches. Rather, fine sediments of < 8 mm comprised smaller fractions of the particle size distribution in OHV-affected compared to those in reference reaches, 5.3% and 10.3%, respectively, indicating a significant difference ($p = 0.002$). For OHV-restored reaches, the percentage of fine sediments remained at 5.3% after restoration. Comparisons of conditional particle size distributions were significantly different between OHV-affected and reference reaches ($p < 0.001$) and between reference and OHV-restored reaches ($p < 0.001$; Figure 3). However, we did not detect a significant difference between the OHV-affected and OHV-restored reaches ($p = 0.187$), meaning the entire size distribution of streamed particles was not measurably different in OHV reaches following restoration. Moreover, comparisons of the 16th and 84th quantiles of particle size distributions appeared coarsened and lacked finer-grained particles in OHV-affected relative to those in reference reaches and remained so following restoration (Table 2). Furthermore, confidence intervals for quantile comparisons between OHV-affected and OHV-restored reaches overlapped (Table 2), reflecting that we did not detect an appreciable shift in these quantiles following restoration.

Pool frequency, volume of large wood, and bank stability were identified as important reach-scale predictors of trout abundance, explaining > 70% of variance based on GLMM Lasso regression. Random Forest regression produced results similar to the Lasso analysis.

3.2 | Macroinvertebrate Responses

BMIs followed a similar trend to brook trout in response to the OHV and subsequent restoration effect. The OHV "effect" on the total BMI density was a 50.9% decrease (GLM, $t = -3.97$, $p < 0.001$) in comparison to unaffected reference reaches, whereas the estimated restoration "effect" was a 112.3% increase (GLM, $t = 3.04$, $p = 0.009$) from the degraded condition where OHVs drove through the stream. When examined together, densities of EPT taxa declined an estimated 40.6% as a result of the OHV "effect" (GLM, $t = -2.58$, $p = 0.023$) and increased an estimated 78.6% due to a plausible restoration "effect" (GLM, $t = 2.08$, $p = 0.058$). When we examined densities of EPT orders separately, Plecoptera and Trichoptera taxa showed strong responses to the OHV "effect," with estimated density reductions of 66.1% (GLM, $t = -3.15$, $p = 0.008$) and 83.8% (GLM, $t = -4.76$, $p < 0.001$), respectively (Figure 4A), whereas the time-adjusted restoration "effect" showed an estimated 220.8% and 359.1% increase in Plecoptera (GLM, $t = 2.47$, $p = 0.028$) and Trichoptera (GLM, $t = 2.89$, $p = 0.013$), respectively (Figure 4A). Ephemeroptera densities did not exhibit responses to either OHV (GLM, $t = -1.23$, $p = 0.239$) or restoration (GLM, $t = 1.216$, $p = 0.246$) "effects." Diptera showed a significant decrease from the OHV "effect" (GLM, −72.3%, $t = -4.23$, $p < 0.001$) and a significant restoration "effect" (GLM, 265.7%, $t = 3.09$ $p = 0.009$). For taxonomic richness, there was a loss of 6.6 taxa from the OHV "effect" (GLM, $t = -2.58$, $p = 0.024$) and a return of 5.4 taxa from a likely restoration "effect" (GLM, $t = 2.11$, $p = 0.056$; Figure 4B). Shannon–Weiner indices were reduced by −1.03 as a result of the OHV "effect" (GLM, $t = -5.53$, $p < 0.001$) and increased by an estimated 0.55 following restoration (GLM, $t = 2.98$, $p = 0.011$; Figure 4B). Proportions of functional feeding groups differed between years (PERMANOVA, $r^2 = 0.238$, $p = 0.019$)

and marginally differed between OHV-affected and reference reaches (PERMANOVA, $r^2 = 0.13$, $p = 0.059$), with the greatest difference in the proportion of predators (Figure 4C); however, proportions did not differ between OHV and OHV-restored reaches (PERMANOVA, $r^2 = 0.062$, $p = 0.232$). Using multivariate analysis, taxon-specific frequencies in the BMI community had different dissimilarity matrices with $k = 2$ dimensions (stress $= 0.095$, $r^2 = 0.99$). The NMDS revealed

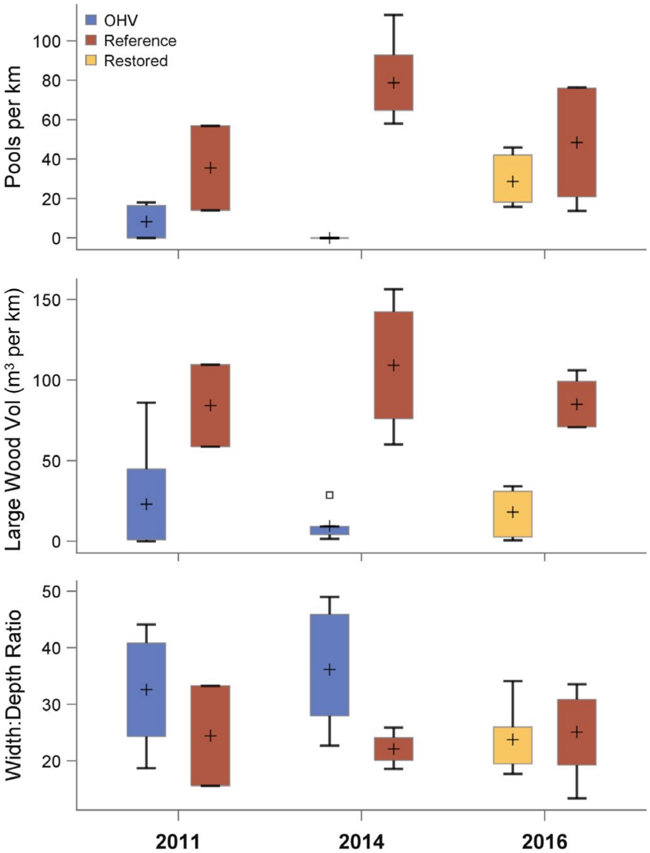


FIGURE 2 | Box plots of pool frequencies, large wood volume, and mean bank stability across OHV (off-highway vehicle)-affected, reference, and -restored reaches by year. Boxes represent the range in minimum and maximum densities observed in the 25th and 75th percentiles (the interquartile range, IQR); means shown as (+); whiskers represent the spread of data that fall within 1.5 times the IQR; and outliers (▫) are datapoints beyond 1.5 times the IQR.

strong differences in taxonomic assemblages found in reference compared to those in OHV reaches, whereas assemblages in OHV-restored reaches were more similar, yet still distinct from reference reaches after restoration ($r^2 = 0.45$, $p = 0.002$; Figure 5). Reference reaches contained greater richness of taxa, several of which are sensitive to water quality degradation (e.g., *Drunella coloradensis*, *Rhycophila* sp., *Zapada cinctipes*), whereas the dipteran *Simulium* appeared more dominant in OHV reaches and heptageniid mayflies *Cinygmula* and *Epeorus*, along with the midge *Synorthocladius*, appeared more frequently in OHV-restored reaches.

3.3 | Stream Vulnerability

An estimated 929 open-water ford crossings were determined using geospatial analyses, whereas only 228 fords were identified by resource managers across the 4.7-M Ha of public land we examined. Using either approach, the locations of open-water fords were distributed disproportionately across the landscape, occurring primarily on 2nd- and 3rd-order streams and the majority occurring in watersheds identified as *high* or *very high* drinking water supply (Figure 6). Cross-validation of open-water fords was only associated with 85 (~9.1%) fords identified from the geospatial analysis. Conversely, there were 143 open-water fords identified through cross-validation but not represented as geospatially identified road–stream intersections, most likely due to the classification of road type attributed in the transportation infrastructure database. Extracting georeferenced locations for crossing infrastructure proved difficult at best for information contained within the INFRA database. Unstandardized infrastructure inventories (USDA Forest Service n.d.) and field-based knowledge were the most reliable data sources for our deductive assessment.

4 | Discussion

We observed a strong, negative influence of OHV use on stream habitat factors as well as the structure and abundance of aquatic fauna in a Rocky Mountain headwater stream. Stream habitat characteristics were simplified into homogeneous riffles, lacking large wood, without pool habitats, and with wide, shallow bedform configurations of uniform channel width and heavily armored streambeds. Pronounced reductions in aquatic fauna included the near absence of trout and half of the number of

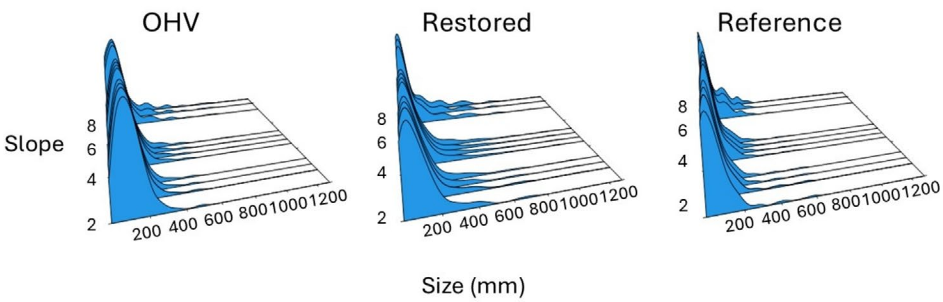


FIGURE 3 | Slope conditioned, grain-size distributions of streambed sediment across a range of reach slopes for OHV (off-highway vehicle)-affected, reference, and -restored reaches of Jenny Creek, Colorado.

TABLE 2 | Estimated particle sizes and results of two-sample confidence interval (CI) comparisons for 16th and 84th particle size quantiles among reference, off-highway vehicle (OHV)-affected, and OHV-restored reach types based on the methods of Eaton and others (2019).

			95% CI overlap		
Type	**Quantile**	**Particle size (mm)**	**Reference**	**$OHV_{affected}$**	**$OHV_{restored}$**
Reference	D_{16}	14	—	**No**	**No**
	D_{84}	142	—	**No**	**No**
$OHV_{affected}$	D_{16}	30	**No**	—	Yes
	D_{84}	176	**No**	—	Yes
$OHV_{restored}$	D_{16}	26	**No**	Yes	—
	D_{84}	189	**No**	Yes	—

Note: CIs that do not overlap are statistically different and highlighted in bold font.

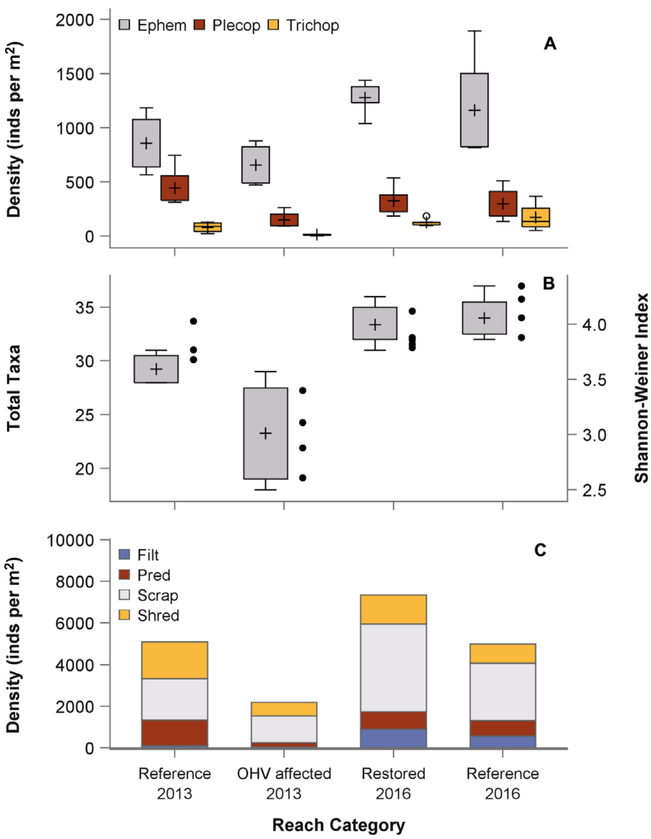


FIGURE 4 | Comparisons of benthic macroinvertebrate densities by order (A), taxonomic richness and Shannon–Weiner diversity (●) (B), and densities of functional feeding groups (C) before (2013) and after (2016) restoration across OHV (off-highway vehicle)-affected, reference, and OHV-restored reaches of Jenny Creek, Colorado. Abbreviations: Ephemeroptera (Ephem); Plecoptera (Plecop); Trichoptera (Trichop); Filterer (Filt); Predator (Pred); Scraper (Scrap); Shredder (Shred).

BMIs in reaches affected by OHV use. Despite the proliferation of motorized recreation on public lands in recent decades (Havlick 2002; Cordell et al. 2005; Pinyon Environmental Report 2016), to our knowledge, this study is the first to report on the apparent impacts of OHV use on habitat quality and biota in streams. Other examinations of OHV effects have focused on vegetation loss, soil erosion, wildlife interactions, and stream channel impacts associated with road–stream intersections (e.g., Buckley 2004; Ouren et al. 2007; Arp and Simmons 2012), yet impacts in streams have been limited to sediment deposition (Brown 1994; Marion et al. 2014; Miniat et al. 2019), channel instability (Arp and Simmons 2012; Marion et al. 2014), and quality of pool habitat (Chin et al. 2004). We documented the reduced abundance of BMIs with shifts toward sediment-tolerant taxa in response to OHV use in streams. A few other studies on macroinvertebrate responses to stream crossings have found varied results; Bylak and Kukuła (2018) claimed reduced diversity and compositional shifts associated with an open-water, concrete ford in a Polish stream; Kidd et al. (2014) attributed reduced EPT taxa and biotic metrics downstream from non-motorized recreational trails on a stream in Appalachia; and Heth et al. (2016) were unable to detect changes in any macroinvertebrate metric downstream from trail crossings, albeit lacking vehicular use, on Ozark streams.

Our results not only describe habitat degradation and biological losses associated with an OHV trail but also suggest mechanistic causes for degradation related to the loss of quantity and quality of pool habitat, removal of large wood, and erosion of streambanks associated with reduced biomass and abundance of trout. Our results are not unique as pool loss and systematic removal of large wood from decades of detrimental land use have been linked to reduced trout densities and less macroinvertebrate production (e.g., Herdrich et al. 2018). Yet, this finding is striking in the degree of effect and is the first linking dramatic reductions in trout occurrence to OHV use in streams. We observed far fewer and shallower pools in OHV compared to those in reference reaches; however, we did not observe pools filled with fine sediment. We suspect that consistent wheel action led to elongated riffles and diminishing depth of pool bedforms. Contrary to our prediction and observations by others (MacDonald and Coe 2008; Sosa-Pérez and MacDonald 2017; Al-Chokhachy et al. 2016), we did not observe an increase in fine sediment in OHV-affected reaches. Rather, our results suggest a coarsening of streambed substrate in reaches exposed to OHV use. This observation may be related to fines being winnowed away from the frequently disturbed streambed surface, leaving less, rather than more fine sediment, which may also explain, in part, why we found some highly mobile taxa associated with high-quality water (e.g., *Rithrogena*, *Epeorus*, *Cinygmula*) in surprisingly high numbers within OHV-affected reaches. Underlying hard

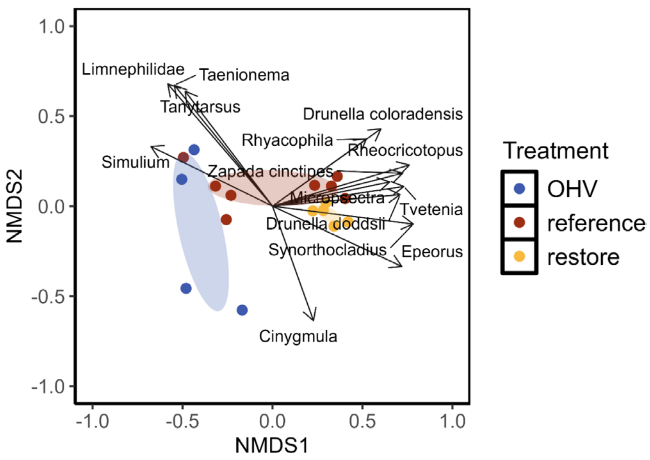


FIGURE 5 | Nonmetric multidimensional scaling (NMDS) plots based on Bray–Curtis distances for the operational taxonomic units (OTUs) that were significantly distinct between OHV (off-highway vehicle)-affected (2013), -restored (2016), and reference (2013, 2016) reaches from riffle collections in Jenny Creek, Colorado.

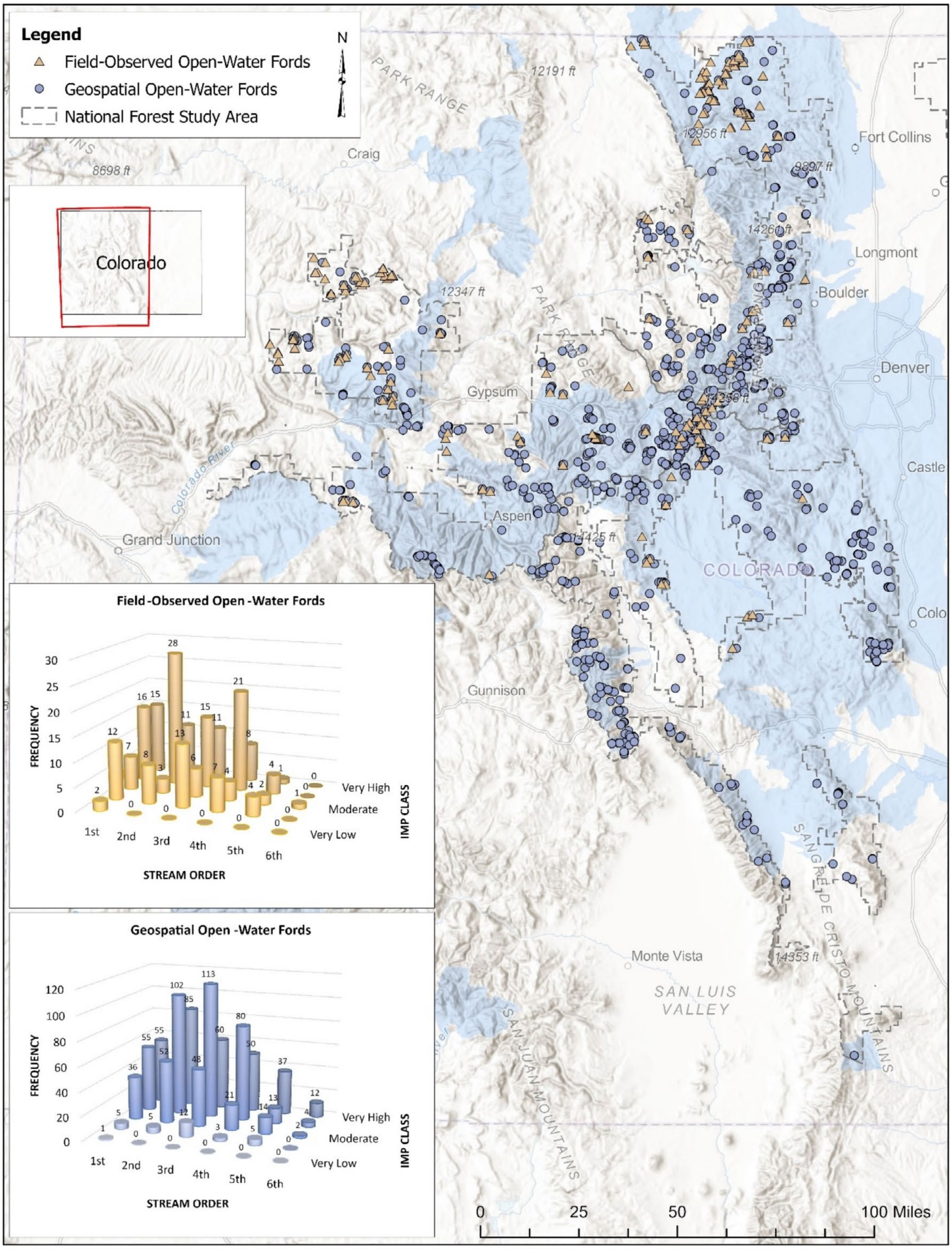
lithology (i.e., felsic gneiss) of our study stream likely yields less sediment supply relative to other mountain headwaters (see Mueller and Pitlick 2013), contributing to our observations of streambed coarsening rather than sediment deposition, which may not be observed in other geologic types.

Stream habitat restoration has been a widespread practice for natural resource managers throughout North America and elsewhere, yet studies evaluating the efficacy of those efforts conclude mixed results owing to misguided technical approaches, misunderstanding of physical processes causing degradation, and lack of appropriate project monitoring (Kondolf and Micheli 1995; Roni et al. 2008). In such cases, restoration focused on instream habitat manipulations that lack management action addressing systemic watershed issues does little to improve the aquatic environment (e.g., Kauffman et al. 1997; Rosi-Marshall et al. 2006; Palmer et al. 2010). In the present study, the greatest restoration act was the removal of the road from the valley bottom, a process far more related to effective engagement with OHV user groups than the technical approach related to stream restoration. Nonetheless, we found that the reconstruction of pool habitat and streambanks, additions of large wood, and decompaction of armored streambeds collectively appeared to result in restored reach-scale habitat characteristics and favorable biological responses. Within the context of the process-oriented restoration, our results align with other studies that have shown improvements in the number and biomass of cold-water salmonids in streams restored following the systematic removal of large wood (Martin et al. 2010; Kratzer 2018) and homogenization of stream channels (Pierce et al. 2013). Like Herdrich et al. (2018), we found a positive influence of large wood on the numbers and size of trout; however, restoration did not resupply large wood to the volumes matching upstream reference conditions, despite restoration intent and measurable gains observed following restoration.

We observed negative shifts in BMI abundance, diversity, composition, and structure where OHV use occurred, as well as evidence of positive responses to stream restoration. Specifically, Plecoptera and Trichoptera taxa showed more striking reductions in OHV-affected reaches, and while we did not detect significant shifts in functional feeding groups, there were visible reductions in the prevalence of predators and shredders in OHV-affected reaches. Although early work has suggested that large wood additions to streams increase habitat for macroinvertebrates (Benke and Wallace 2003) and ample stream restoration projects have occurred in a variety of environments (Roni et al. 2015; Palmer et al. 2010), most projects are not monitored (Bernhardt et al. 2005). When monitored, only a few studies have detected gains in macroinvertebrate density and diversity from aquatic habitat restoration, with large wood features eliciting stronger signals than boulder features (Miller et al. 2010), due to either the ineffectiveness of restoration to address biotic needs (e.g., Lepori et al. 2005) or latency in biotic recovery following restoration (Pillotto et al. 2018). In our case, recovery in the abundance, but not composition, of stream macroinvertebrates in affected reaches occurred within 2 years, indicating that relocating roads from valley bottoms of headwater streams and channel restoration realized a positive biotic response.

Open-water fords are but one type of stream crossing design for forest roads. Some suggest open-water fords provide benefits of low maintenance, flood resilience, and limited influence on aquatic passage (e.g., Warren and Pardew 1998; Clarkin et al. 2006; Keller and Clarkin 2007). Alternate designs, particularly culverts installed in prior eras, have been associated with stream habitat fragmentation and habitat loss for motile stream animals (e.g., Trombulak and Frissell 2000; Pépino et al. 2012). Other designs, such as bridges and modern culverts, constructed on public lands today account for aquatic organism passage and fluvial processes that do not restrict the movement of aquatic animals (Cenderelli et al. 2011). While transportation managers may often view fords as the only cost-effective, viable option for high-clearance vehicle roads, such crossing designs are not without risks to streams or aquatic ecosystem health. Our study clearly demonstrated degraded aquatic habitat and loss of fauna where a headwater stream had multiple road crossings and long sections of the stream were driven through due to avulsions into the road and bank erosion where the road paralleled the stream. We introduce the suggested term *stroads* for headwater streams that co-occur with and are severely altered by high-clearance roads. Lastly, restoring our study stream was costly, exceeding US $300,000, owing to the remoteness of the site and expense of rerouting 1.5 km of the road. Although restoring streams in degraded condition works to increase habitat complexity and abundance of aquatic life (present study, others), proactive prevention of such natural resource degradation is likely to be far more cost-effective. Taken together with evidence from others (Luce et al. 2001; Al-Chokhachy et al. 2016), managers seeking to reduce the impacts of unmaintained roads on stream networks would be wise to focus on stream crossings.

Stream networks are prominent features across public lands (Kampf et al. 2021), providing ecological services, such as high-quality clean water (Liu et al. 2022), biological production, resource delivery to downstream waters and adjacent terrestrial environments, and natural regulation of flood waters (Furniss et al. 2010). Transportation networks are essential for managing public lands and many of the extractive ecological services (e.g., timber, grazing, water storage and delivery, recreation) we derive from them. Where both intersect, road networks can have a



Esri, TomTom, Garmin, FAO, NOAA, USGS, Bureau of Land Management, EPA, NPS, USFWS, Esri, USGS

FIGURE 6 | Map of the central Rocky Mountains in Colorado displaying "high" and "very high" value watersheds for public water supply in blue shading (Forest to Faucets, version 2; Mack et al. 2022) and approximate distributions of open-water ford crossings on high-clearance vehicle roads within the White River, Arapaho-Roosevelt, and Pike-San Isabel National Forests study area. Blue circles represent geospatially identified fords, whereas orange triangles represent fords verified by natural resource managers. The inset histograms show the frequency of open-water crossings by Strahler stream order and importance (IMP) class for public water supplies from Mack et al. (2022).

multitude of effects on the supporting, regulating, provisioning, and cultural types of ecological services that streams and headwaters provide (see Furniss et al. 2010; Ferreira et al. 2023). Our field study detected deleterious effects on aquatic habitat quality and aquatic fauna where an OHV road traversed and intersected a headwater stream in the central Rocky Mountains. Further, our assessment suggests that many other headwater streams in municipal watersheds may also be at risk where high-clearance roads allow public OHV use. We caution readers that reported numbers of open-water fords are estimates, not absolute numbers. Further, the geospatial approach likely overestimates the actual number of open-water fords, whereas the field-based approach underestimates the number of fords. Furthermore, factors such as the alignment of roads with stream courses in close proximity to open-water fords likely affect the degree to which a ford poses risks to a stream. Despite inaccuracies, our assessment highlights that many other headwater streams are at risk from OHV use in this mountain landscape and underscores the lack of consistently inventoried infrastructure of road networks on public lands. Although roads are critical infrastructure needed for aspects of delivering ecosystem services, road networks may be best managed at a size and extent that is sustainable. Strategies for sustainable outdoor recreation growth have been posed (Thomas and Reed 2019) but have not yet been implemented in a manner that comprehensively addresses road networks, in the case of motorized vehicle recreation.

Increased OHV use on public lands has occurred on an aging transportation network that has, until recently, been maintained with minimal resources for land management agencies. OHV trail riding on public lands has steadily increased in recent years, which has triggered, albeit unaddressed, systemic concerns regarding the impacts and sustainability of such uses on public resources. Concomitantly, natural resource policies, laws, and proactive management of this issue have lagged behind the increased OHV use, leaving public lands prone to degradation from OHV trail networks. Managers have had long-standing awareness of the vast, unmaintained road networks occurring on federal public lands (Havlick 2002), yet roads remain a significant threat to aquatic biodiversity on public lands (Flitcroft et al. 2023). Our assessment suggests that many streams are vulnerable to OHV impacts, either presently or in the future. We do not suggest that all open-water fords present uniform risk to streams, as some may be well designed to minimize localized effects based on alignment, approach, and surfacing (Keller and Clarkin 2007; Keller and Sherar 2003). However, there may be many other poorly configured crossings and portions of unmaintained roads affecting headwater streams in the central Rocky Mountains and elsewhere. We recommend that future research examine environmental impacts in other headwater streams where high-clearance vehicle roads allow for OHV use, particularly across a spectrum of lithologies, hydrology, and eco-regions.

Acknowledgments

Restoration was funded by the Colorado Department of Natural Resources Off-Highway Vehicle State Trail grant 14-USFSTO-65380 and Boulder Flycasters. Bev Baker, Kevin Zimlinghaus, Deb Entwistle, Judy Kittson, and Colin Hutten contributed to restoration activities. Numerous seasonal technicians over many years helped collect field data. Natural resource managers from the Pike-San Isabel, White River, and Arapaho-Roosevelt National Forests provide valuable knowledge on locations of open-water crossings. Dan Cenderelli provided advice on restoration design, and colleagues Robert Zuellig, Travis Schmidt, Amy Odom, and the late Boris Kondratieff provided support or counsel for various components of the analytical approach. We thank volunteers from Boulder Flycasters and Rising Sun OHV Club providing support with field work during restoration. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section.