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Natural Resource Stewardship and Science

Stream Community Response to the 2011 Las Conchas Fire in Bandelier National Monument

Aquatic Macroinvertebrate Monitoring 2007–2019

Natural Resource Report NPS/SCPN/NRR—2022/2486

ON THIS PAGE
RIT02 monitoring site on Rito de los Frijoles during 2013 in Bandelier National Monument
NPS/SCPN

ON THE COVER
CAP01 monitoring site on Capulin Creek during 2014 in Bandelier National Monument
NPS/SCPN

Stream Community Response to the 2011 Las Conchas Fire in Bandelier National Monument

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Contents

Page

Figures v

Tables vi

Appendices viii

Abstract ix

Acknowledgments x

Introduction 1

Importance of Aquatic Macroinvertebrates to Stream Ecosystems 1

Wildfire and Stream Ecosystems 2

Las Conchas Fire 3

Methods 6

Study Area 6

Capulin Creek 6

Rito de los Frijoles 8

Field Methods 10

Data Analyses 11

Richness and Diversity 11

Community Analyses 12

Abundance and Post-fire Trend Analyses 13

New Mexico Stream Condition Index 13

Results 15

Diversity and richness 15

Community Analyses 20

Community Composition 20

Indicator Species Analysis 23

Abundance and Post-Fire Trend Analyses 26

Overall Site Abundance 26

Contents (continued)

Page

Abundance of Dominant Taxa....27

Common Family Post-Fire Abundance....30

Post-fire Family Group Trend....32

Functional Feeding Group Abundance....33

Post-fire Functional Group Trend....34

Mountain-Stream Condition Index....37

Discussion....39

Effects of Fire and Historic Flows on Community Structure....39

Post-fire trend in community structure recovery....41

Ecosystem Function....42

Management Implications....43

Conclusions....45

Literature Cited....46

Figures

Page

Figure 1. Southern Colorado Plateau Network (SCPN) long-term aquatic monitoring sites and fire history affecting streams in Frijoles and Capulin Canyons in Bandelier National Monument from 1977–2019 3

Figure 2. Annual peak stream flow (cubic feet per second) at USGS stream gage 08313350 on the Rito de los Frijoles RIT01 5

Figure 3. Southern Colorado Plateau Network (SCPN) long-term aquatic monitoring sites at Bandelier National Monument and burn severity during the Las Conchas Fire of 2011 7

Figure 4. Photo of Capulin Creek taken at the CAP01 monitoring site in 2013 after first large post-fire flood event. 8

Figure 5. Photo of the Rito de los Frijoles at RIT02 in 2013 following the Las Conchas Fire and monsoon flood. 9

Figure 6. Family level annual diversity (top) and richness (bottom) estimates at CAP01 monitoring site on Capulin Creek (Chao and Jost 2011). 16

Figure 7. Family level annual diversity (top) and richness (bottom) estimates at CAP02 monitoring site on Capulin Creek (Chao and Jost 2011). 17

Figure 8. Family level annual diversity (top) and richness (bottom) estimates at RIT01 monitoring site on Capulin Creek (Chao and Jost 2011). 18

Figure 9. Family level annual diversity (top) and richness (bottom) estimates at RIT02 monitoring site on Capulin Creek (Chao and Jost 2011). 19

Figure 10. Non-Metric Multidimensional Scaling (NMS) ordination of samples collected at four (CAP01, CAP02, RIT01, RIT02) monitoring sites in Bandelier National Monument between 2007–2019. 21

Figure 11. Mean annual aquatic macroinvertebrate sample abundance for four monitoring sites at Bandelier National Monument (Capulin Creek N = 2, Rito de los Frijoles N = 2). 26

Figure 12. Dominant family abundance at Capulin Creek 2007–2019. 28

Figure 13. Dominant family abundance at the Rito de los Frijoles 2009–2019. 29

Figure 14. Post-fire trend in shredder abundance at RIT02. 35

Figure 15. Post-fire trend in scraper abundance at RIT02. 36

Figure 16. Post-fire trend in predator abundance at RIT02. 37

Tables

Page

Table 1. Quantitative aquatic macroinvertebrate sampling on Capulin Creek and Rito de las Frijoles in Bandelier National Monument. 10

Table 2. State of New Mexico's Mountain Stream Condition Index (M-SCI) metrics. 14

Table 3. Condition categories for M-SCI scores. 14

Table 4. Permutation test results for difference in means between pre-fire and post-fire diversity and richness estimates at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). 15

Table 5. Family taxa correlation to NMS ordination axes at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). 22

Table 6. Joint plot environmental variables used to describe variation in aquatic insect community at monitoring sites in Bandelier National Monument. 22

Table 7. Indicator Species Analysis results from four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). 23

Table 8. Families absent from the post-fire community at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). 24

Table 9. Families novel to the post-fire community at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). 25

Table 10. Permutation test results on the difference in means of pre-fire and post-fire site abundance. 27

Table 11. Permutation test summary of results for differences in pre-fire and post-fire abundance in aquatic macroinvertebrate family abundance at two sites on Capulin Creek and two on the Rito de los Frijoles in Bandelier National Monument 2007–2019 (N=48). 31

Table 12. Permutation test results on the difference in means of pre-fire and post-fire functional feeding group abundance at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). 33

Table 13. Post-fire trend analysis of functional feeding groups at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). 34

Tables (continued)

Page

Table 14. Monitoring site Mountain Stream Condition Index scores at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). 38

Table A1. Total abundance and relative abundance of the top five dominant taxa at the CAP01 monitoring site for each year. 53

Table A2. Total abundance and relative abundance of the top five dominant taxa at the CAP02 monitoring site for each year. 55

Table A3. Total abundance and relative abundance of the top five dominant taxa at the RIT01 monitoring site for each year. 56

Table A4. Total abundance and relative abundance of the top five dominant taxa at the RIT02 monitoring site for each year. 58

Table B1. Difference in pre-fire and post-fire means in abundance for each family taxa at each monitoring site. 61

Table C1. Theil-Sen estimator results of post-fire trend for family taxa at each monitoring site. 72

Appendices

Page

Appendix A. Dominant Taxa 53

Appendix B. Results from Family Abundance Permutation Tests 61

Appendix C. Post-fire Family Trend Analysis Results 72

Abstract

Bandelier National Monument is a landscape with an extensive fire history. During the last 50 years, Bandelier experienced three major wildfires that affected streams within the park (1977, 1996, 2011). These fires led to repeated hydrologic disturbance and substantial impacts on the aquatic communities within park streams. This report investigates changes in aquatic macroinvertebrate community structure and function in two streams (Capulin Creek and the Rito de los Frijoles) following the 2011 Las Conchas Fire and the series of historic floods that followed.

The Southern Colorado Plateau Network (SCPN) has been collecting aquatic macroinvertebrate data from sites on Capulin Creek and the Rito de los Frijoles since 2007 and 2009, respectively. We use a combination of multivariate analyses and permutation procedures to describe change in the aquatic macroinvertebrate community. Trend analyses are used to investigate recovery among family and functional groups in post-fire years. Finally, we take the data collected from our monitoring program and apply them to the state of New Mexico's Mountain Stream Index to describe ecological impairment occurring at our sites over time.

Our results indicate that the effects of this fire and subsequent flooding on the aquatic community are ongoing. These disturbances resulted in the decline of aquatic macroinvertebrates across all taxonomic orders. Few taxa have been able to recolonize streams following the fire. Organisms with the ability to recolonize are disturbance-tolerant taxa with functional traits that promote resilience. The loss of several sensitive taxa has likely disrupted ecosystem function at all monitoring sites. Functional feeding groups responsible for processing primary production (shredder, scrapers) experienced the largest post-fire declines, and trend analyses indicate recovery of these functional groups, as well as many other taxa, has not occurred. Recovery of community structure and function, in addition to being slow, is primarily limited to the upstream monitoring site on the Rito de los Frijoles. Data also suggest the aquatic macroinvertebrate community at the lower Rito de los Frijoles monitoring site is likely undergoing additional perturbation not experienced at other monitoring sites.

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Introduction

Aquatic macroinvertebrates were identified as a high-priority vital sign for aquatic ecosystems in Southern Colorado Plateau Inventory and Monitoring Network (SCPN) parks (Thomas et al. 2006). SCPN has been monitoring the aquatic macroinvertebrate communities in Bandelier National Monument (Bandelier) in northern New Mexico since 2007. The purpose of this monitoring program is to determine the annual status and condition of the aquatic macroinvertebrate communities with the goal of providing park managers a scientific foundation for management decisions regarding stream ecosystems in the park. This report investigates aquatic macroinvertebrate community data collected in the first 12 years of the program (2007–2019) and documents changes occurring as a result of the 2011 Las Conchas Fire.

Importance of Aquatic Macroinvertebrates to Stream Ecosystems

Aquatic macroinvertebrates are a globally ubiquitous and diverse group of organisms that are fundamental to stream ecosystems. Their importance lies in their ability to control and govern the movement of energy through aquatic ecosystems (Cummins et al. 1973, Covich et al. 1999). As a group, aquatic macroinvertebrates influence detrital decomposition, nutrient cycling, primary production, and the transport of materials through the ecosystem (Covich 1988, Wallace and Webster 1996). Nutrients enter the streams via autochthonous energetic inputs, i.e., those created within the stream as primary production (e.g., algae, cyanobacteria, fungi, macrophytes), and as allochthonous inputs, i.e., those created from outside the aquatic ecosystem (e.g., riparian vegetation, woody debris, leaf litter, etc.). Aquatic macroinvertebrates are responsible for processing the majority of these energetic inputs through direct predation. Energy is transferred up through the aquatic food web resulting in an increase in community biomass. Such secondary production can account for up to 86% of biomass in aquatic systems (Lindegaard 1997). Covich et al. (1999) suggest that aquatic macroinvertebrates are responsible for processing an estimated 20–73% of the riparian leaf litter that enters low order (headwater) streams. In streams of the Southwestern U.S., secondary productivity is 2–13 times higher compared to streams found in the Eastern United States, with some taxa processing 4.2 times their body weight a day (Fisher and Gray 1983). Aquatic macroinvertebrates also play an important role in nutrient cycling within the aquatic ecosystem. Many aquatic macroinvertebrate taxa are benthic, dwelling in or on bottom substrates, where they release dissolved nutrients into the system that are stored in sediments via burrowing, foraging, and migration (Palmer et al. 1997).

Aquatic macroinvertebrates have a long history of use as a surrogate for ecosystem health. Their sensitivity to environmental change makes them an ideal model for use in biotic indices designed to assess the integrity of aquatic ecosystems (Barbour et al. 1999, Jacobi et al. 2006, Herman and Nejadhashemi 2015). Studies investigating aquatic macroinvertebrate community structure have been used to identify causes of ecosystem degradation that range from chemical pollutants (Kearns and Karr 1994) to urbanization (Roy et al. 2003). Their sensitivity to changes in the aquatic environment makes them important indicators of water quality and aquatic ecosystem condition (Allan 1995, Karr and Chu 1999). For example, many taxa have evolved gill structures that make them sensitive to decreases in dissolved oxygen within the aquatic environment. Because the aquatic

macroinvertebrate community directly reflects conditions in the aquatic environment, the presence of gilled taxa has been considered an indicator of high quality aquatic habitat (Gaufin and Tarzwell 1956, Goodnight 1973).

Wildfire and Stream Ecosystems

Streams in the western United States are disturbance-driven ecosystems that have an inseparable link to the surrounding riparian forests (Cummins 1974, Vannote et al. 1980). In addition to crucial energetic inputs provided to the aquatic system, riparian vegetation interacts with natural geomorphic processes to control the movement of water and sediments through the system, slowing the residence times of both (Fisher 1995). The riparian canopy adjacent to streams controls surface water exposure to solar radiation, thus regulating stream temperature (Barton et al. 1985). Autumnal losses of canopy cover, and subsequent leaf deposition in streams, can result in seasonal pulses of nutrients into stream water that are then available for microbial and aquatic macroinvertebrate uptake (Gregory et al. 1991).

Wildfires decouple the dynamic links between riparian forests and stream ecosystems, altering system processes across multiple levels and spatial scales. Stand replacing wildfires can lead to large post-fire hydrologic events that change sediment loads, flow rates, and can lead to the extirpation of taxa from the stream ecosystem (Minshall et al. 1989, Gresswell 1999, Veenhuis 2002, Vieria et al. 2014). Fire removes streamside vegetation and results in the loss of vital habitat and food resources (woody debris, leaf packs, detritus) for aquatic fauna. The loss of the riparian canopy in headwater streams can affect autotrophic production, limiting both nitrogen and phosphorous within the system (Newbold et al. 1983, Grimm and Fisher 1986). Furthermore, the loss of riparian canopy can result in changes in seasonal light patterns that can affect the timing and rates of metabolic, growth, and reproductive processes of stream biota (Sweeny 1993).

The forest and stream ecosystems within Bandelier National Monument have experienced major wildfire three times in recent decades (Figure 1). The La Mesa Fire burned 15,270 acres in and around Frijoles Canyon during 1977. Nineteen years later, in 1996, the Dome Fire burned 16,516 acres in Capulin Canyon. Then the Las Conchas Fire burned 156,591 acres of private and federal lands in north-central New Mexico in 2011 (Walker 2018). Pippin and Pippin (1980) inventoried the aquatic macroinvertebrate community in the Rito de los Frijoles following the 1977 La Mesa Fire. Vieira et al. (2004) tracked change and recovery in the aquatic macroinvertebrate community in Capulin Creek following the Dome Fire. Both studies documented a decline in aquatic macroinvertebrate abundances to near zero in the years immediately following fire. Both studies also attribute change in the aquatic community to post-fire flood events.

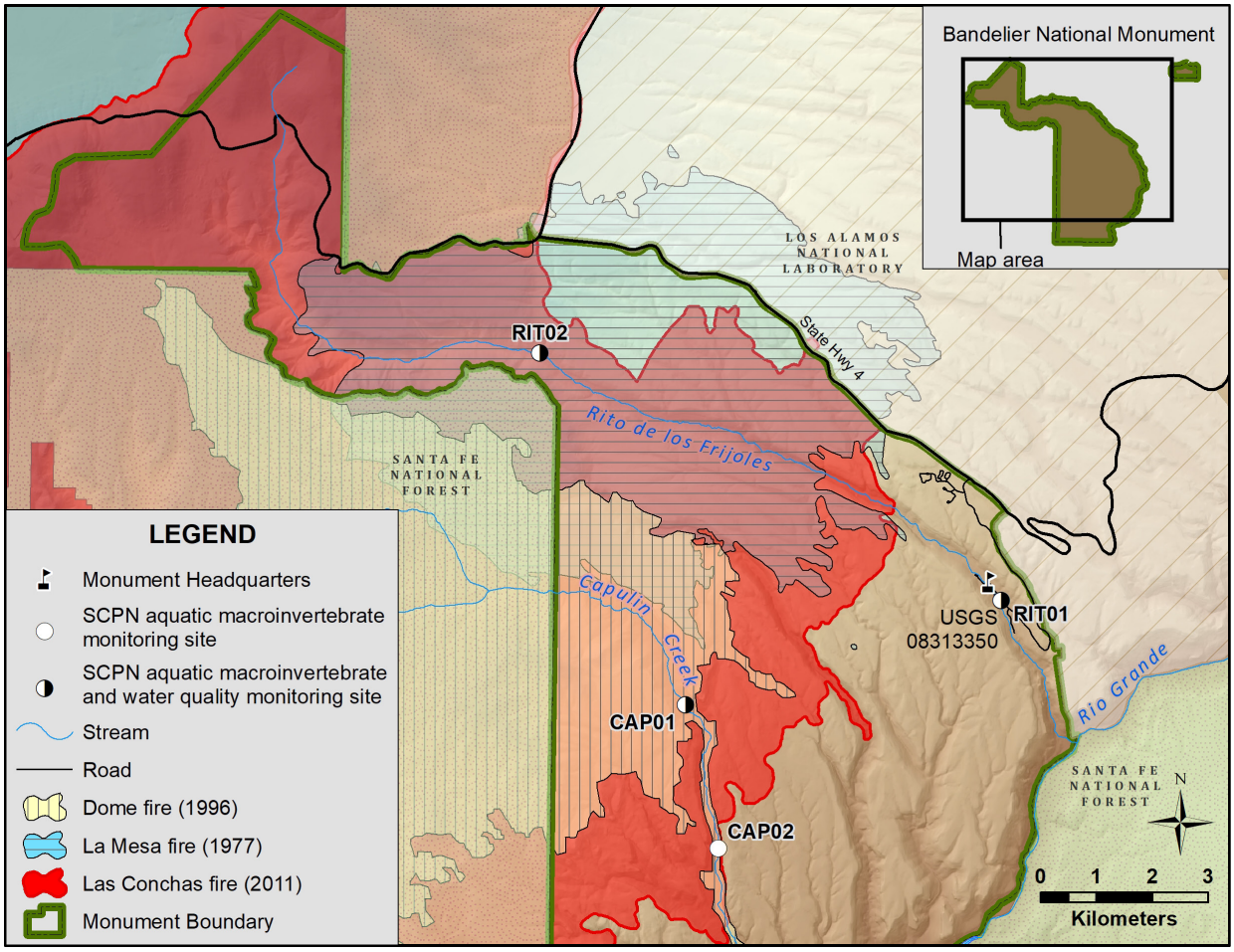

Figure 1. Southern Colorado Plateau Network (SCPN) long-term aquatic monitoring sites and fire history affecting streams in Frijoles and Capulin Canyons in Bandelier National Monument from 1977–2019.

The average annual peak flows in Capulin Creek and the Rito de los Frijoles typically occur as a result of snow melt in early spring or from summer monsoonal precipitation. Post-fire peak flows increased dramatically after the La Mesa and Dome fires (Veenhuis 2002). Median annual peak flow in the Rito de los Frijoles prior to the La Mesa Fire was 5.7 ft^3/sec (cfs) with a maximum peak flow recorded at 19 cfs (Veenhuis 2002). At Capulin Creek, prior to the Dome Fire, median annual peak flow was 15.2 cfs with an estimated maximum peak flow of 25 cfs. In the years following these fires, peak flows increased by approximately 160 times pre-fire maximum peak flows. Maximum post-fire peak flow following the La Mesa Fire was 3,030 cfs in the Rito de los Frijoles. Following the Dome Fire peak flow was estimated to be 3,630 cfs in Capulin Creek (Veenhuis 2002).

Las Conchas Fire

On June 26, 2011, a fallen power line ignited the Las Conchas Fire. The fire burned 156,591 acres (244 mi^2) of private and federal lands throughout the Jemez Mountains in north-central New Mexico until it was fully contained on August 3rd (Walker 2018). Approximately 62% of the land within the boundary of Bandelier National Monument burned during the Las Conchas Fire (Figure 1). Unlike the previous fires which were contained to a single canyon, the Las Conchas Fire burned through

large portions of both Capulin and Frijoles Canyons. Prior to the Las Conchas Fire, SCPN established four long-term aquatic macroinvertebrate monitoring sites. Two sites were established in 2007 on Capulin Creek and an additional two sites were established in 2009 on the Rito de los Frijoles. Three of four long-term monitoring sites experienced moderate to high severity fire leading to the complete removal of the riparian canopy and understory at those sites (SCPN unpublished data). The loss of vegetation in the canyon bottoms led to several unprecedented peak flow events that affected all monitoring sites in the years following the fire.

Annual peak streamflow in the years immediately following the Las Conchas Fire were the largest ever recorded in the Rito de los Frijoles (Figure 2). Monsoonal precipitation resulted in two large flood events, one on August 21, 2011, which reached 7,000 cfs and a second, larger event that occurred on September 13, 2013, that increased flow to 9,500 cfs. Following the 2011 floods, park staff requested assistance from the USGS to set up an early warning stream gauge in upper Frijoles Canyon. The purpose of this temporary gaging station was to track large flow events in real time to provide monument staff with ample warning for the evacuation of visitors from downstream locations prior to the arrival of dangerous flows. In the years following the Las Conchas Fire, this early warning stream gauge has collected data showing that 30 large monsoon driven flow events occurred between 2011–2014 in Frijoles Canyon (Anne Tillery, USGS, unpublished data). Flooding was not confined to Frijoles Canyon. Tillery and Hass (2016) modeled debris-flow probabilities in subbasins around the Jemez Mountains following the Las Conchas Fire and found that 671 subbasins returned high probabilities (81%) for large post-fire floods. While Capulin Creek is not instrumented with a stream gauge, the appearance of large debris flows and downcutting suggest it experienced similar sized flood events.

This report describes how the aquatic macroinvertebrate communities in Capulin Creek and the Rito de los Frijoles changed as result of the Las Conchas Fire and subsequent hydrologic disturbances.

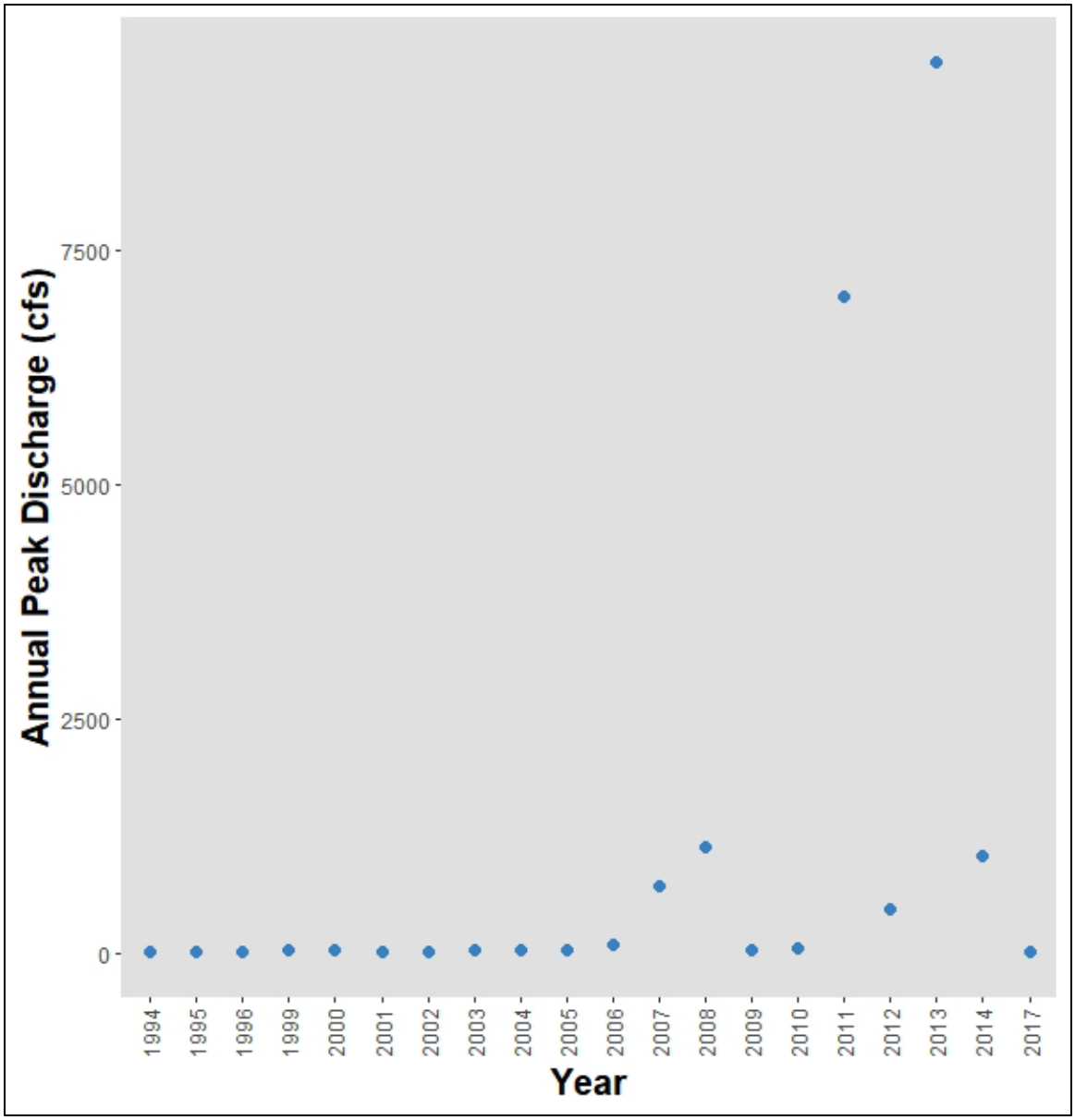

Figure 2. Annual peak stream flow (cubic feet per second) at USGS stream gage 08313350 on the Rito de los Frijoles RIT01 (https://nwis.waterdata.usgs.gov/nwis/peak?site_no=08313350&agency_cd=USGS&format=html).

Methods

Study Area

Bandelier National Monument (Bandelier) is 13,629 hectares (136.2 km^2) and is located on the Pajarito Plateau in the Jemez Mountains of north-central New Mexico. The monument was established in 1916 to protect ancestral Puebloan cultural resources such as kivas, pictographs, and petroglyphs. The topography within the monument is a series of wide mesas separated by southeast-trending, deeply-incised, steep-walled canyons (Muldavin et al. 2011). The climate at Bandelier is characterized by warm, wet summers and cool dry winters.

SCPN and Bandelier staff selected two perennial streams within the park boundary to monitor aquatic macroinvertebrate community structure and diversity (Figure 1). In 2007, two monitoring sites were established on Capulin Creek, and two additional monitoring sites were established on the Rito de los Frijoles in 2009.

Capulin Creek

Capulin Creek is a perennial montane stream that originates on the eastern ridge of the Jemez Mountains within the Santa Fe National Forest and flows downslope to its confluence with the Rio Grande. Pippin and Pippin (1981) reported that Capulin Creek flows for approximately 11.5 km through Capulin Canyon, noting that the final 5.1 km experience intermittent flows during the summer months. The watershed is designated wilderness and managed for recreational use within park boundaries. The upper reaches of the watershed start outside of park boundaries and are managed for recreation and timber harvest by the U.S. Forest Service. Capulin Creek experienced wildfire during the 1996 Dome Fire and the 2011 Las Conchas Fire. Following the Dome Fire, native Rio Grande cutthroat trout (*Oncorhynchus clarkii virginalis*) populations were extirpated in Capulin Creek (Allen, 2004). Although the New Mexico Department of Game and Fish (NMDGF), Bandelier, and the Santa Fe National Forest collaborated to reintroduce cutthroat trout in 2006, no trout have been detected in the creek following the Las Conchas Fire and subsequent flooding events.

Capulin Creek at Base Camp Gaging Station (BANDCAP01), identified in this report as CAP01, is located less than 0.3 km upstream from the site of the historic backcountry ranger cabin and is co-located with an SCPN water quality monitoring site (Figure 1). The center-point of the site is located at a discontinued U.S. Geological Survey (USGS) streamflow gaging station (Capulin Canyon at Ranger Cabin, # 083133655). The channel substrate at this site is primarily gravels and fine sediments.

Capulin Creek above Painted Cave (BANDCAP02), identified in this report as CAP02, is located 2.6 km downstream from the base camp gaging station at CAP01, and 1.8 km up canyon from a popular backcountry site named Painted Cave. This site was selected using the Generalized Random Tessellation Stratified (GRTS) design, and channel substrate is primarily fine/sand substrate.

The Las Conchas Fire burned at high severity through portions of the upper reaches of Capulin Canyon, upstream of the study sites (Figure 3). Most of the canyon, including CAP01 and CAP02 study sites, burned at moderate severity (Figure 3). The Las Conchas Fire and subsequent hydrologic

disturbances removed riparian and forest vegetation from both monitoring sites on Capulin Creek (Figure 4).

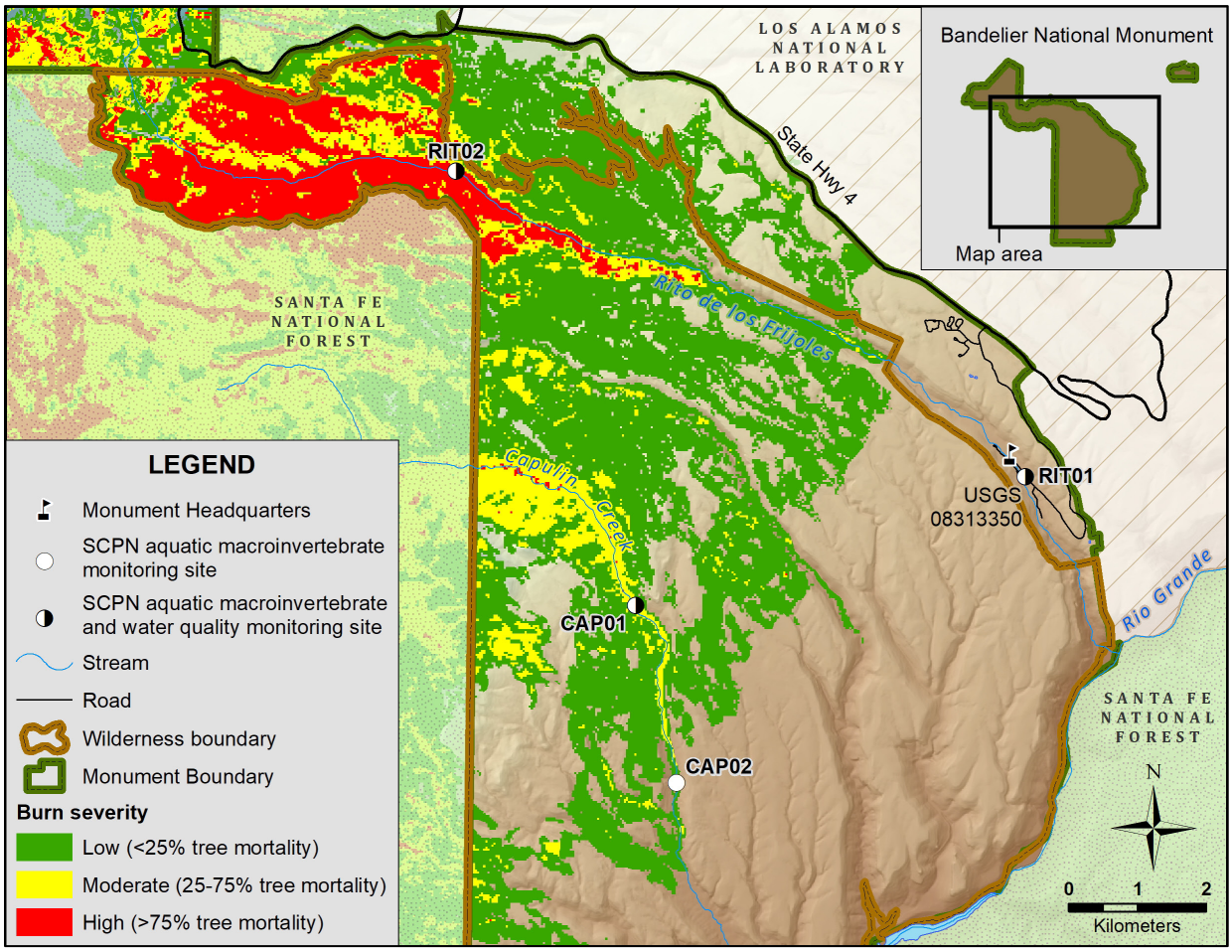

Figure 3. Southern Colorado Plateau Network (SCPN) long-term aquatic monitoring sites at Bandelier National Monument and burn severity during the Las Conchas Fire of 2011 (https://burnseverity.cr.usgs.gov/baer/baer-imagery-support-data-download).

Figure 4. Photo of Capulin Creek taken at the CAP01 monitoring site in 2013 after first large post-fire flood event. The Las Conchas Fire and subsequent floods removed most of the forest and riparian vegetation.

Rito de los Frijoles

The Rito de los Frijoles is a perennial stream that flows entirely within Bandelier National Monument. It originates high in the Jemez Mountains on the Pajarito Plateau from a series of springs in the upper watershed and flows eastward through the park where it eventually flows into the Rio Grande River. The upper reaches of the watershed are designated wilderness and managed for recreational use. Because the Bandelier visitor center and numerous archeological sites are located near the stream in the middle portion of the watershed, it experiences a high level of visitor use. Similar to the Capulin watershed, the Frijoles watershed has burned multiple times in recent fire history (Figure 1). Approximately 6,070 ha of the watershed burned during the La Mesa Fire in 1977. The upper reaches of the watershed burned again during the Cerro Grande Fire of 2000. In 2011, the Las Conchas Fire burned through the upper half of Frijoles canyon.

Rito de los Frijoles near Visitor Center (BANDRIT01), identified in this report as RIT01, is co-located with an SCPN water quality monitoring site and a USGS streamflow gaging station (Rito de los Frijoles in Bandelier Nat Mon, NM 08313350) (Figure 3). Riparian vegetation is dominated by narrowleaf cottonwood (*Populus angustifolia*), box elder (*Acer negundo*), and New Mexico olive

(*Forestiera neomexicana*) (Muldavin et al. 2011). Channel substrates are dominated by fine particles, gravel, and cobble.

Rito de los Frijoles at Upper Crossing (BANDRIT02), identified in this report as RIT02, is located at the intersection with Upper Crossing Trail. This site is co-located with a discontinued (1960–1963) USGS streamflow gaging station (Rito de los Frijoles near Los Alamos, NM 08313300). Riparian and forest vegetation was burned and scoured from the upper reaches of the Rito de los Frijoles resulting in the complete removal of vegetation from this site. Cobble, gravel, and fine sediments dominate stream bottom substrates.

The Las Conchas Fire burned at high severity through the upper reaches of Frijoles canyon, which includes the RIT02 monitoring site (Figure 3 and 5). The lower RIT01 site did not burn during the Las Conchas Fire, as wildland firefighters successfully stopped fire spread down canyon. Several large hydrologic disturbances occurred on the Rito de los Frijoles in the years following the fire, and post-fire flooding in the upper reaches impacted both monitoring sites (Figure 2 and 5).

Figure 5. Photo of the Rito de los Frijoles at RIT02 in 2013 following the Las Conchas Fire and monsoon flood.

Field Methods

The methods in this paper followed those described in Brasher et al. (2011), the Aquatic Macroinvertebrate Monitoring Protocol for the Southern Colorado Plateau. This report includes analysis of the aquatic macroinvertebrate samples collected from 2007–2019 (Table 1).

Table 1. Quantitative aquatic macroinvertebrate sampling on Capulin Creek and Rito de las Frijoles in Bandelier National Monument (sampling years denoted with "x"). The asterisks for 2013 represents years where data collection did not occur due to a government-wide shut down. Dashes represent years where data could not be collected due to a lack of appropriate habitat, and n/a represents years prior to site establishment.

Site	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
CAP01	x	x	x	x	x	–	*	–	x	x	x	x	x
CAP02	x	x	x	x	–	–	*	–	x	x	x	x	x
RIT01	n/a	n/a	x	x	x	x	*	x	x	x	x	x	x
RIT02	n/a	n/a	x	x	x	x	*	x	x	x	x	x	x

The State of New Mexico defines its aquatic macroinvertebrate sampling window for high-elevation streams as August to mid-November (Jacobi et al. 2006). All monitoring sites were sampled annually during this sampling window, typically during October (Table 1). During each site visit, two sets of invertebrate samples were collected. These consisted of a qualitative multihabitat sample used as an annual site inventory and a quantitative targeted habitat sample that provided estimates of organism abundances. Qualitative samples were collected by walking the length of the 150m sampling reach and sampling all habitat types present at the time of the visit. Frequency of habitat types sampled was dependent upon the frequency of their occurrence along the sampling reach. For example, if a reach was dominated by cobbles, algal mats, and leaf packs, those habitats were sampled with greater frequency along the 150m monitoring reach than less dominant habitat types. For the quantitative sample, five replicate targeted riffle samples were collected by placing a 0.25m^2 sampling frame on the streambed of a chosen riffle. A slack sampler with a 500µm mesh net was placed directly downstream of the sampling frame. All substrates within the frame were disturbed and scrubbed for 2 minutes. All organisms collected were stored in 95% ethyl alcohol for preservation.

Quantitative samples were exclusively collected in riffle habitat. This allowed for consistency of habitat type across years and reduced sample variation. In the years immediately following the fire, when flooding and channel instability were greatest, monitoring sites often lacked a complete set (five) of riffles within the 150m sampling reach. During years where riffle habitat was scarce, field crews shifted the monitoring reach up or downstream to incorporate five riffle samples. When this occurred, the site was shifted 75m in either direction. By limiting movement of the site to 75m, we ensured half of sample overlapped with the reach sampled in previous and subsequent years (Brasher et al. 2011). During 2011–2014, floods removed all riffle habitat from CAP01 and CAP02. Lack of riffle habitat prevented us from collecting quantitative samples at CAP01 for two years (2012, 2014)

and at CAP02 for three years (2011, 2012, 2014) (Table 1). No data were collected in 2013 due to a government shutdown during the sampling window (Table 1).

All samples were sent to the National Aquatic Monitoring Center at Utah State University for identification. Detailed information regarding identification techniques and quality control and assurance procedures are described in Brasher et al (2011).

Data Analyses

Richness and Diversity

Taxonomic richness and diversity were assessed using a fixed-coverage rarefaction method and reported at the family level. Family level identifications were selected for measuring diversity and richness because 97.9% of individuals collected could be identified to family, while only 68.5% of individuals could be identified to genus. We chose a mixed-coverage rarefaction methodology over raw counts to account for differences in annual sampling area (number of riffles), number of individuals collected, and laboratory subsampling across years. (Chao and Jost 2012). This method used rarefaction or extrapolation to compare species richness based on samples of equal completeness. Traditional rarefaction methods extrapolate to equal-sized samples, which can bias results towards larger samples that may not characterize the full richness of a community (Chao and Jost, 2012). Analysis methods were based on equal completeness to help ensure the estimated coverage (total taxa) of the community captured was equal across samples. Richness was measured as a count of taxa estimated to be present at that sample coverage level (99.5%), which gives rare species the same importance as common species. Rarefaction or extrapolation curves were derived using R statistical software package iNext (Hsieh et al. 2019, Chao et al. 2014b). Samples where extrapolations were twice the original sample size have been identified in tables and figures. Samples in years where the number of individuals was too small to calculate a diversity estimation were removed from annual mean calculations.

Richness and diversity were calculated at the family level from the composited quantitative sample (summed values of individual riffle samples) for each monitoring year. Diversity and richness metrics were only calculated for samples having two or more taxa and at least 30 individuals. Individuals identified to a coarser taxonomic level (e.g., order), with no other family members present in the sample, were included as a single taxon at the higher resolution.

Diversity was calculated using the inverse of the Simpson concentration. The Simpson concentration places greater emphasis on the value of evenness among species. Using the inverse value of the Simpson index (corresponding to a Hill number of order 2) provides greater consistency in comparing relative diversity values among samples (Hill 1973, Chao et al. 2014a).

Diversity and richness measures used to calculate the New Mexico Stream Condition Index were calculated separately and followed the methods developed for that index (Jacobi et al. 2006). A detailed description of those methods is described in the relevant section below.

Community Analyses

Quantitative samples were composited by site and year (sampling unit) at the family level for community analyses unless otherwise noted. The response matrix consisted of 59 taxa in 39 sampling units. Taxa that occurred in only a single sampling unit were removed from the matrix. Data were log(x+1) transformed to normalize variances among all sample units. All community analyses were run in PC-ORD version 6.22 (McCune and Mefford 2010).

To visualize dissimilarity in community structure between sampling units, a non-metric multidimensional scaling ordination (NMS) was performed (Kruskal 1964, McCune and Mefford 2010) using Sørensen (Bray-Curtis) distances on random starting configuration with 999 runs on the real data. Sørensen (Bray-Curtis) distances were calculated as the shared abundance between sample units divided by the total abundance of all samples (Bray and Curtis 1957, Faith et al. 1987, McCune and Grace 2002). Ties in the distance matrix were not penalized (Kruskal's strategy 1, McCune and Grace 2002). Sample units were grouped by flow status: pre-fire: 2007–2010, unstable: 2011–2014, and stabilizing: 2015–2019 based on timing of post-fire hydrologic disturbances. To test for differences in community structure (similarity of SUs) across sample unit groupings, an Analysis of Similarity (ANOSIM) was run using the anosim function in the R package Vegan. The same number of permutations and distance measures were used for all ANOSIM tests and for NMS ordination. The placement of sample units in ordination space depict a sample's relationship to the gradients described by the resulting ordination axes. Dissimilarities in community structure of a sample with respect to other samples in the analysis were represented by the distance between sample units.

We used the joint plot function in PC-Ord to investigate relationships between environmental factors and ordination scores (Peck 2010). Environmental variables were Canopy Cover, Stream Depth, Particle Size, Particle Embeddedness, pH, Conductance, Dissolved Oxygen, Discharge, and Turbidity. A description of how these data were collected can be found in Brasher et al. (2011) and Monroe et al. (2016). The joint plot function created directional vectors originating from the centroid of all samples that described the relationship between the environmental variables and the two ordination axes (Peck 2010). The length and direction of the vector indicated the strength of the association between the variable and the axes. If a vector pointed to a particular sample unit or group of sample units, then those sample units were positively correlated to the corresponding stream condition variables displayed in the ordination. Stream condition variables with $r^2>0.20$ were displayed as joint plot vectors in the ordination.

An Indicator Species Analysis (ISA) was performed to determine which taxa or groups of taxa best represented sample unit flow status groupings (Dufrêne and Legendre 1997). An ISA assesses the value of a specific taxa as an indicator for environmental conditions. An Indicator Value (IV=0–100) was assigned to each taxon describing its value with respect to each stream flow status grouping. Indicator Values were calculated based on a taxon's exclusivity (relative abundance) to the group, and its fidelity (relative frequency) to samples within the group. Exclusivity was determined by calculating the average abundance of each taxon in each group. That average was then divided by the sum of average abundances across all groups. Fidelity was calculated by dividing the number of sample units in each group where a particular taxon was present by the total number of sample units

in the that group. Exclusivity and fidelity were then multiplied by 100 to calculate an Indicator Value (Peck 2010). All indicator values were tested for significance using a Monte Carlo randomization procedure. Taxa were representative of a group (i.e., an indicator) if the IV value was ≥50.0, $p<0.05$. Additionally, the ISA highlighted any taxa present prior to the fire that failed to appear in post-fire samples (i.e., unstable and stabilizing periods) and any novel taxa that appeared post-fire.

Abundance and Post-fire Trend Analyses

Two-sample permutation tests were run on group means for each family at each site to determine the differences in pre-fire and post-fire abundance of aquatic macroinvertebrate families and functional groups at the site level. Permutation tests on group means are a non-parametric equivalent of the t-tests (Hesel 2012). Permutation tests are ideal for skewed datasets and require no distributional assumptions about the data. The two-sample permutation test randomly shuffled the group assignment (e.g., pre-fire/post-fire) for all distinct combinations of the data and computed a test statistic, the difference in means (Δ in means), for each shuffle. A p-value was computed from the percent of results that were equal to or more extreme than results in the observed data.

Post-fire trend of family and functional feeding groups (organisms with similar feeding structures that share similar ecological roles) were examined using the Thiel-Sen estimator (Thiel 1950, Sen 1968). The Thiel-Sen estimator is a non-parametric regression analysis that computes a slope and intercept for the line connecting each pair of samples in a dataset. A median slope and intercept are then computed to provide the Thiel-Sen line. This test uses the Kendall's Rank Correlation (τ) to produce a value similar to a r^2 value in traditional regression analyses. Unlike a traditional r^2 value, tau has a range of −1 to 1. Tau values approaching 1 represent a positive trend over time and values approaching −1 indicate a negative trend over time. Tau values near zero are plotted as a flat Thiel-Sen line and indicate no trend over time. This test was chosen because of its lack of distribution requirements, resistance to outliers, and power for detecting trends in skewed data (Hesel 2012).

New Mexico Stream Condition Index

The multimeric Mountain Stream Condition Index (M-SCI) developed by the State of New Mexico was used to investigate stream ecological integrity and impairment at our monitoring sites. (Barbour et al. 1996, Jacobi et al. 2006). This index is composed 12 aquatic macroinvertebrate metrics designed to address the following ecosystem attributes: diversity, composition, disturbance tolerance, habit, and functional feeding group (Table 2). All metric values are calculated and standardized to obtain a metric score ranging from 0–100. Metric scores are summed and averaged to obtain an overall M-SCI index score for the site. Index scores are given a condition category that serves as a method of comparison to reference sites where conditions represent a healthy, sustainable stream ecosystems in the state of New Mexico. Sites with M-SCI rankings of "Very Good" 100–78.35 and "Good" 78.35–56.70 are designated as meeting the water quality standard for aquatic life use (Table 3). This standard is defined as "High quality cold-water perennial surface water in a minimally disturbed condition with considerable aesthetic value and superior cold-water aquatic life habitat with, stream bed characteristics and other attributes of habitat sufficient to protect and maintain a propagating cold-water aquatic life population." (Jacobi et al. 2006). To make the data comparable to the state's methodology, a finer taxonomic resolution was applied using operational

taxonomic units, as defined by the National Aquatic Monitoring Center (Cuffney et al. 2007, Miller and Judson 2011). This provided us a resolution identical to the index's intended use. Individual quantitative samples were pooled for a site and year. Metrics were calculated on raw data and standardized using methods described in Jacobi et al. (2006).

Table 2. State of New Mexico's Mountain Stream Condition Index (M-SCI) metrics.

Ecosystem Attribute	Index Metric
Taxonomic Composition	Shannon Diversity ($\log_2$)
	Evenness
	Plecoptera %
Taxonomic Richness	Ephemeroptera Taxa
	Plecoptera Taxa
Tolerance	Sensitive EPT%
	Intolerant %
Habit	Clinger Taxa
	Sprawler Taxa
	Swimmer Taxa
Functional Feeding Group	Scraper %
	Scraper Taxa

Table 3. Condition categories for M-SCI scores. Only sites with scores in categories "Very Good" and "Good" are considered to meet the state's designated aquatic life use.

Very Good	Good	Fair	Poor	Very Poor
100–78.35	78.35–56.70	56.70–37.20	37.20–18.90	<18.90

Results

Diversity and richness

Three of four sites exhibited significant differences in species diversity between pre- and post-fire periods (Table 4, Figures 6–9). Rito de los Frijoles sites contained less diverse communities post-fire (Table 4, Figures 8–9, RIT01 Δ means = 2.4, p = 0.04; RIT02 Δ means = 2.9, p = 0.03), as did the upstream site (CAP01) on Capulin Creek which burned at moderate severity (Table 4, Figure 6). There was no difference between pre- and post-fire diversity at the downstream site (CAP02) on Capulin Creek (Table 4, Figure 7).

Table 4. Permutation test results for difference in means between pre-fire and post-fire diversity and richness estimates at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). Significant differences detected when p ≤ 0.05.

Metric	Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL
Diversity	CAP01	6.8	3.9	2.9	0.008	6.8	3.9
	CAP02	3.5	3.2	0.3	0.905	−1.6	1.9
	RIT01	5.1	2.8	2.4	0.040	1.6	3.1
	RIT02	6.1	3.2	2.9	0.030	1.7	4.1
Richness	CAP01	22.7	13.8	8.8	0.030	4.1	13.3
	CAP02	15.9	14.8	1.1	0.802	−8.1	7.7
	RIT01	20.6	13.4	7.2	0.179	3.6	10.0
	RIT02	20.6	12.5	8.1	0.083	20.6	12.5

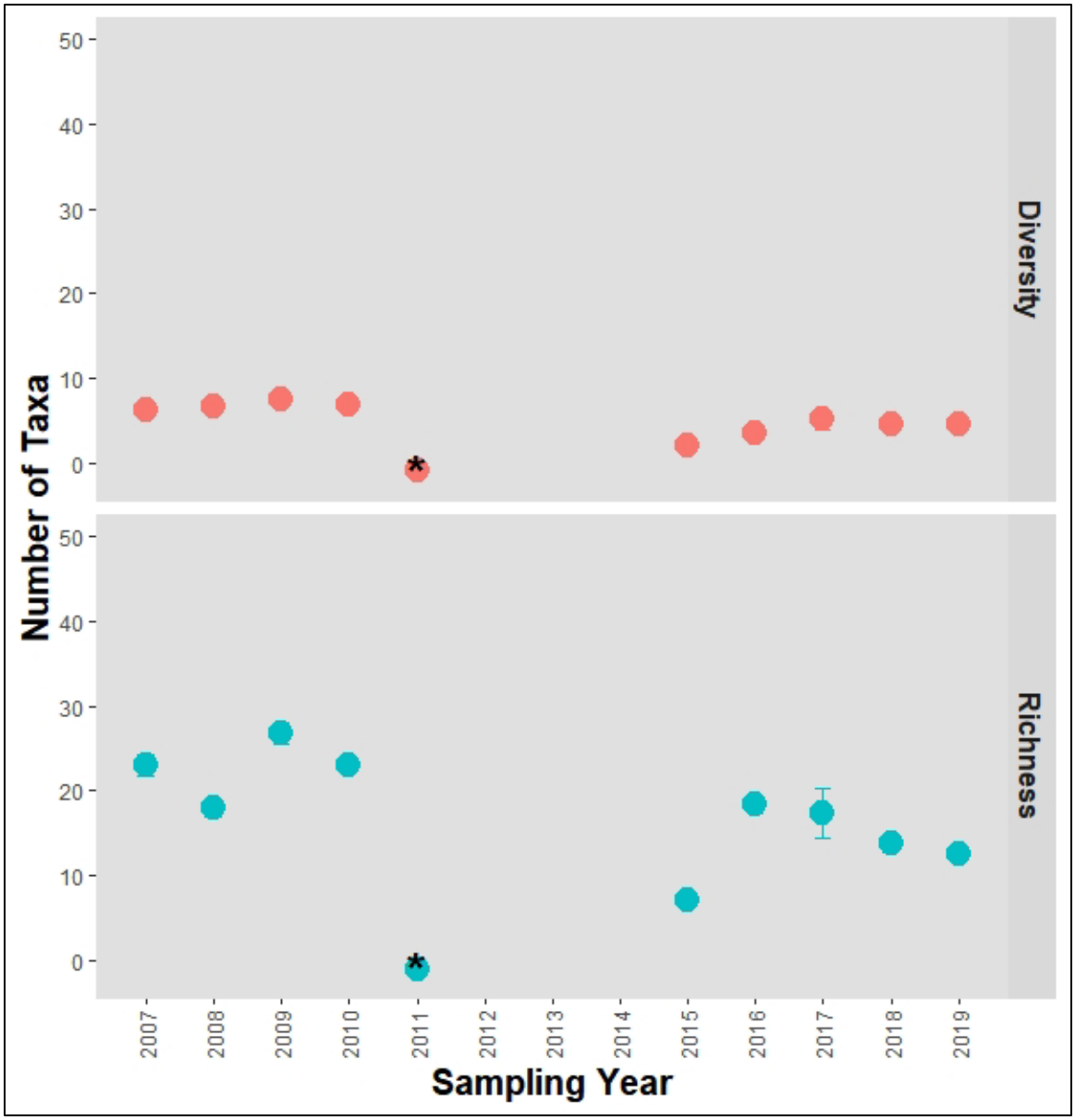

Figure 6. Family level annual diversity (top) and richness (bottom) estimates at CAP01 monitoring site on Capulin Creek (Chao and Jost 2011). Samples annotated by an asterisk (*) were too small to run estimates. Missing values represent years where sites lacked sufficient habitat to sample.

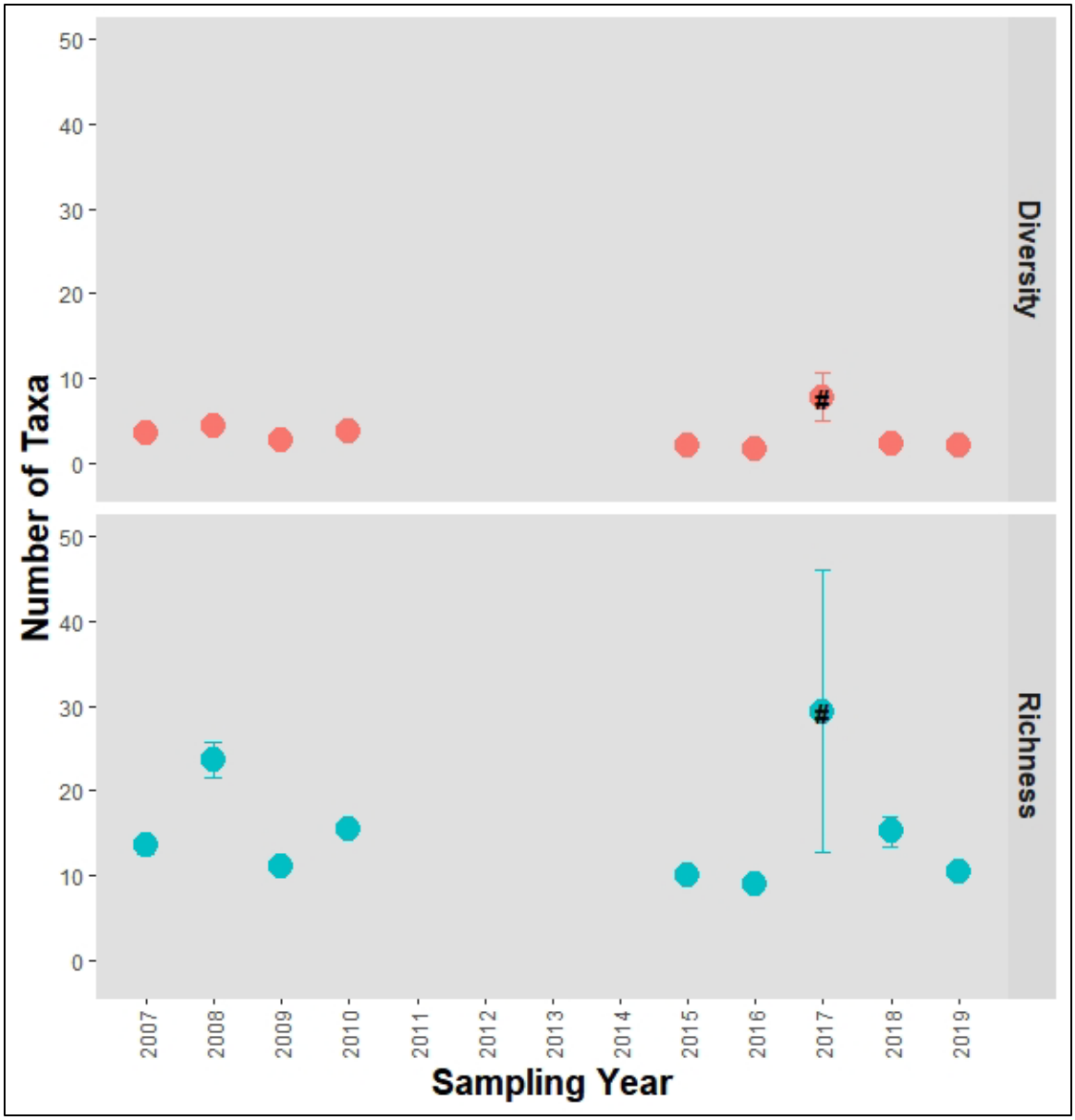

Figure 7. Family level annual diversity (top) and richness (bottom) estimates at CAP02 monitoring site on Capulin Creek (Chao and Jost 2011). Samples annotated with a # are estimates that are subjected to high bias due to over extrapolation. Missing values represent years where sites lacked sufficient habitat to sample.

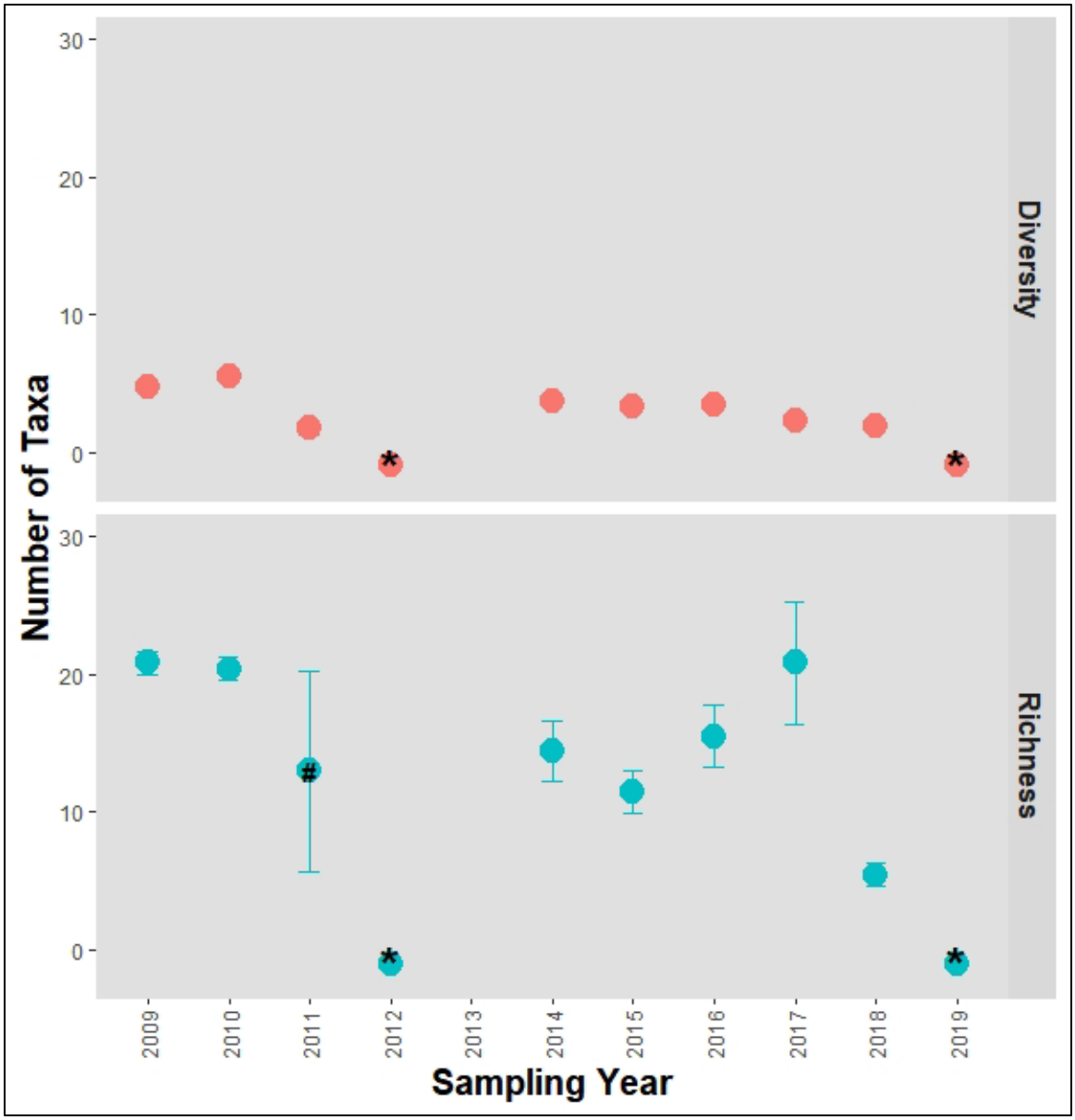

Figure 8. Family level annual diversity (top) and richness (bottom) estimates at RIT01 monitoring site on Rito de los Frijoles (Chao and Jost 2011). Samples annotated by an asterisk (*) were too small for estimates. Samples annotated with a # are estimates that are subjected to high bias due to over extrapolation. Missing values represent years where sites lacked sufficient habitat to sample.

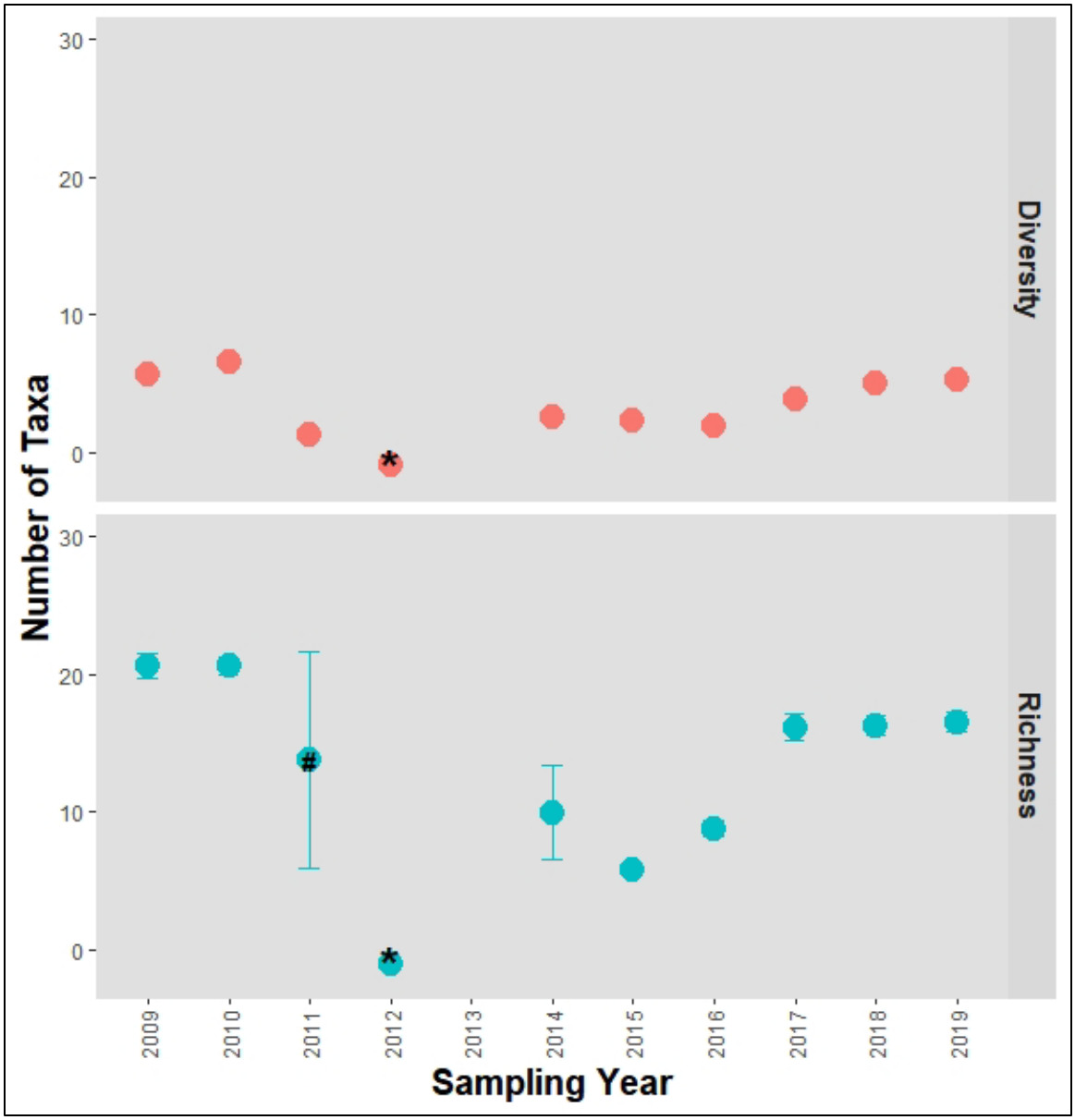

Figure 9. Family level annual diversity (top) and richness (bottom) estimates at RIT02 monitoring site on Rito de los Frijoles (Chao and Jost 2011). Samples annotated by an asterisk (*) were too small for estimates. Samples annotated with a # are estimates that are subjected to high bias due to over extrapolation. Missing values represent years where sites lacked sufficient habitat to sample.

Post-fire richness estimates were lower at one (CAP01) of four sites when compared to pre-fire estimates (Table 4, Figures 6–9), but pre-fire and post-fire richness values were not statistically different at the other three sites (Table 4). Mean annual richness at CAP01 declined by 39.0% from a pre-fire value of 22.7 taxa to a post-fire value of 13.8 taxa (Table 4).

Community Analyses

Community Composition

The community composition matrix used was composed of 39 sample units (site/year) and 59 families. The final NMS ordination solution was a two-dimensional ordination with a stress value of 8.97 ($p = 0.004$). The ordination represented 92% of the variation in our matrix with axis 2 explaining 66% of the variation in the dataset (Figure 10). Axis 2 was related to the dominance of Dipteran families (Table 5). Baetidae (Ephemeroptera), Perlodidae (Plecoptera), Hydropsychidae (Trichoptera), and Sperchontidae (Trombidifomes) also exhibited a strong correlation with axis 2 (Table 5). Axis 1, the weaker of the two axes in explaining variation among all sample units, was dominated by the presence of Chironomidae, Tipulidae, Elmidae (Coleoptera), Leptophlebiidae (Ephemeroptera), and plecopteran families (Table 5).

Results from the ANOSIM analysis to test for differences between pre-defined stream flow status groups (i.e., pre-fire, unstable, and stabilizing) demonstrate distinct partitioning of samples, i.e., significant differences in community structure among the three groups ($r = 0.581$, $p = 0.0001$). Pre-fire samples (2007–2010, blue diamonds in Figure 10) from all monitoring sites are clustered in the upper right quadrant of the ordination. Pre-fire samples from CAP02 are separated from other pre-fire samples, suggesting some pre-fire structuring among sites. All pre-fire sample units (blue diamonds) show clear separation from the unstable (2011–2014, red triangles) and stabilizing (2015–2019, orange diamonds) sample groups (Figure 10). Further, there is little overlap between unstable and stabilizing group samples, indicating distinct differences in community structure across all three groups. Samples from the stabilizing group (orange triangles) fall between pre-fire (blue diamonds) and unstable (red triangles) groups, with the most recent samples (2017–2019) clustering at the top of axis 2 and closer to pre-fire samples (Figure 10).

Five stream condition variables describe the relationship between ordination scores (sample units) and axes (Table 6, Figure 10). Axis 1 describes a gradient where increased canopy closure ($r^2 = 0.503$) and water temperature ($r^2 = 0.237$) are working in opposition to describe recovery status (Figure 10). The canopy closure vector extends from the centroid towards the pre-fire sample cloud and represents a period of increased canopy cover (Figure 10). The temperature vector moves in the opposite direction (left) along axis 1 and represents a time of increased stream temperatures (when canopy cover was low). Similarly, turbidity ($r^2 = 0.341$) is working in opposition to stream depth ($r^2 = 0.269$) and particle embeddedness ($r^2 = 0.200$) along axis 2. The vector for turbidity points directly towards the unstable sample grouping, when stream flows were most dynamic (bottom of Figure 10). An increase in stream depth and particle embeddedness are represented by samples grouped towards the top of the ordination (when turbidity was relatively lower).

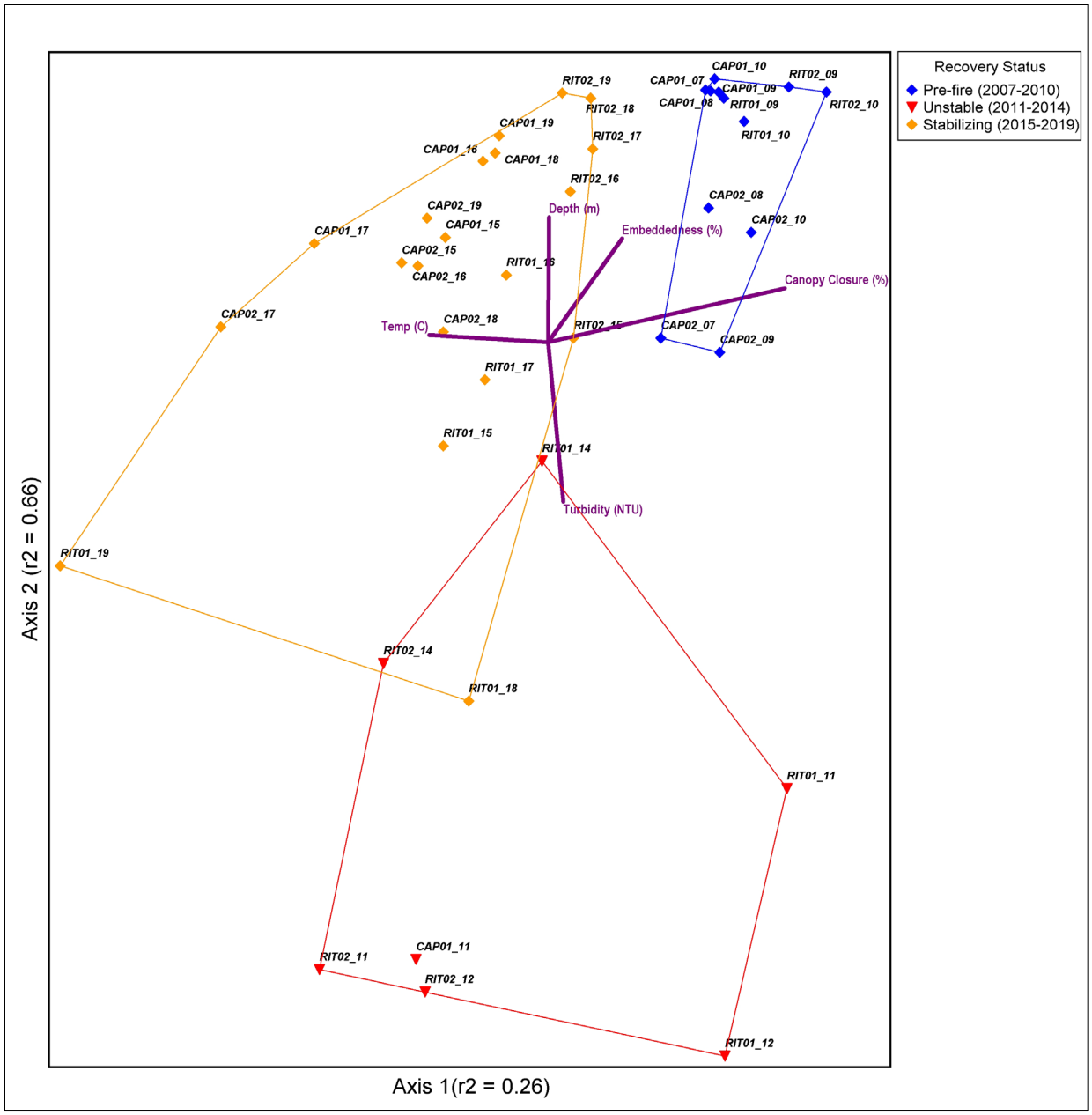

Figure 10. Non-Metric Multidimensional Scaling (NMS) ordination of samples collected at four (CAP01, CAP02, RIT01, RIT02) monitoring sites in Bandelier National Monument between 2007–2019. Individual sample units are labelled by site code and year (e.g., RIT01_12 = Rito de los Frijoles site number 1 during 2012). Purple vectors represent environmental variables that contribute to groupings and point to samples where those relationships are the strongest.

Table 5. Family taxa correlation to NMS ordination axes at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). Pearson's R Coefficients (r) describe the correlation of the family with respect to the individual axes.

Order	Family	Axis 1 r	Axis 2 r
Coleoptera	Elmidae	0.780	0.561
Diptera	Ceratopogonidae	0.445	0.715
	Chironomidae	0.727	0.707
	Empididae	0.264	0.728
	Psychodidae	0.440	0.696
	Simullidae	0.216	0.687
	Tipulidae	0.709	0.731
Ephemeroptera	Baetidae	0.086	0.764
	Leptophlebiidae	0.619	0.564
Plecoptera	Capniidae	0.737	0.307
	Chloroperlidae	0.612	0.404
	Nemouridae	0.698	0.497
	Perlodidae	0.590	0.711
Trichoptera	Hydropsychidae	0.439	0.794
Trombidiformes	Sperchontidae	0.269	0.800
Veneroida	Pisidiidae	0.695	0.421

Table 6. Joint plot environmental variables used to describe variation in aquatic insect community at monitoring sites in Bandelier National Monument. Axis r^2 values describe the correlation between environmental variables and corresponding axes.

Environmental Variable	Axis 1 r^2	Axis 2 r^2
Temperature (°C)	0.237	0.015
pH	0.115	0.000
Conductance (µs/cm)	0.079	0.165
Dissolved O2 (mg/L)	0.003	0.018
Turbidity (NTU)	0.005	0.341
Discharge (cfs)	0.001	0.039
Canopy Closure (%)	0.503	0.101

Table 6 (continued). Joint plot environmental variables used to describe variation in aquatic insect community at monitoring sites in Bandelier National Monument. Axis r^2 values describe the correlation between environmental variables and corresponding axes.

Environmental Variable	Axis 1 r^2	Axis 2 r^2
Median Riffle Depth (m)	0.016	0.267
Flow Velocity (m/s)	0.000	0.024
Particle Size (mm)	0.158	0.006
Particle Embeddedness (%)	0.173	0.200

Indicator Species Analysis

Sixteen of the 59 family taxa in the response matrix were determined to be indicators of pre-fire aquatic macroinvertebrate communities (Table 7). Two families, Ephemerellidae (Ephemeroptera) and Pisidiidae were perfect indicators of pre-fire communities. Both were present in all pre-fire samples and were present only in samples that belong to the pre-fire grouping (Table 7). Most families (62%) found to be indicators of pre-fire community structure belong to the sensitive EPT (Ephemeroptera, Plecoptera, Trichoptera) taxa group. In addition to Ephemerellidae, the ephemeropteran families Heptageniidae and Leptophlebiidae were strong indicators of pre-fire community structure (Table 7). Plecopteran families Nemouridae, Chloroperlidae, Perlodidae, and Capniidae were among the stronger indictors of pre-fire community structure (Table 7). Among trichopteran families, Lepidostomatidae also served as a strong indicator of pre-fire communities with Glossosomatidae and Brachycentridae being moderate indicators of pre-fire communities. Only one family, the odonate Libellulidae, was found to be a strong indicator of our stabilizing group. There were no indicators found for our unstable group.

Table 7. Indicator Species Analysis results from four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). Indicator Value (IV) for taxa was calculated from relative abundance (RA%) of the taxa for a particular group and its relative frequency (RF%), how often the taxa occurred within samples of that group.

Order	Family	Indicator Group	RA %	RF%	IV	p-value
Ephemeroptera	Ephemerellidae	Prefire	100	100	100.0	0.0002
Veneroida	Pisidiidae	Prefire	100	100	100.0	0.0002
Plecoptera	Nemouridae	Prefire	96	92	87.9	0.0002
Coleoptera	Elmidae	Prefire	86	100	85.6	0.0002
Plecoptera	Chloroperlidae	Prefire	100	83	83.3	0.0002
Trichoptera	Lepidostomatidae	Prefire	97	75	73.1	0.0002
Plecoptera	Perlodidae	Prefire	63	100	68.3	0.0002

Table 7 (continued). Indicator Species Analysis results from four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). Indicator Value (IV) for taxa was calculated from relative abundance (RA%) of the taxa for a particular group and its relative frequency (RF%), how often the taxa occurred within samples of that group.

Order	Family	Indicator Group	RA %	RF%	IV	p-value
Ephemeroptera	Heptageniidae	Prefire	100	67	66.7	0.0002
Plecoptera	Capniidae	Prefire	70	92	64.1	0.0002
Ephemeroptera	Leptophlebiidae	Prefire	84	75	63.1	0.0004
Diptera	Tipulidae	Prefire	63	100	62.9	0.0002
Odonata	Aeshnidae	Prefire	65	83	53.8	0.003
Diptera	Ceratopoginidae	Prefire	54	100	53.6	0.0008
Trichoptera	Glossosomatidae	Prefire	79	67	52.7	0.0002
Trichoptera	Brachycentridae	Prefire	100	50	50.0	0.002
Trombidiformes	Torrenticolidae	Prefire	100	50	50.0	0.0006
Odonata	Libellulidae	Stabilizing	100	75	70.0	0.0002

There were 13 families of aquatic macroinvertebrates that were not found in post-fire samples at any of the monitoring sites (Table 8). Of the 13 families missing in post-fire samples, 10 (77%) belonged to the sensitive EPT taxa. Five of the EPT taxa not seen in post-fire samples occurred in ≤ 33% of the pre-fire samples indicating that they were not dominant taxa prior to the fire. For example, Calamoceratidae and Rhyacophilidae, both trichopterans, were found in only two (17%) of the pre-fire samples.

Table 8. Families absent from the post-fire community at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). Pre-fire relative abundance (RA%) and relative frequency values (RF%) indicating how often these taxa were found in grouped samples.

		Exclusivity (RA%)			Frequency (RF %)		
Order	Family	Pre-fire	Unstable	Stabilizing	Pre-fire	Unstable	Stabilizing
Ephemeroptera	Ameletidae	100	0	0	33	0	0
Ephemeroptera	Ephemerellidae	100	0	0	100	0	0
Ephemeroptera	Heptageniidae	100	0	0	67	0	0
Odonata	Cordulegastridae	100	0	0	42	0	0
Plecoptera	Chloroperlidae	100	0	0	83	0	0
Plecoptera	Pteronarcyidae	100	0	0	25	0	0

Table 8 (continued). Families absent from the post-fire community at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). Pre-fire relative abundance (RA%) and relative frequency values (RF%) indicating how often these taxa were found in grouped samples.

		Exclusivity (RA%)			Frequency (RF %)		
Order	**Family**	**Pre-fire**	**Unstable**	**Stabilizing**	**Pre-fire**	**Unstable**	**Stabilizing**
Trichoptera	Brachycentridae	100	0	0	50	0	0
Trichoptera	Calamoceratidae	100	0	0	17	0	0
Trichoptera	Leptoceridae	100	0	0	33	0	0
Trombidiformes	Torrenticolidae	100	0	0	50	0	0
Veneroida	Pisidiidae	100	0	0	100	0	0

Seven novel taxa were found in post-fire samples (Table 9). The dipteran Ephydridae and the hemipteran Corixidae appeared in unstable (2011–2014) and stabilizing (2015–2019) group samples (Table 9). Five taxa appeared for the first time at our monitoring sites in samples from the stabilizing period. Physidae (Basommatophora), Curculionidae (Coleoptera), Tabanidae (Diptera) and Mesoveliidae (Hemiptera) were sampled relatively infrequently, occurring in 30% or less of samples (Table 9). The odonate Libellulidae occurred in 70% of the samples during system stabilization (Table 9).

Table 9. Families novel to the post-fire community at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). Post-fire relative abundance (RA%) and relative frequency values (RF%) indicate how often these taxa were found in grouped samples.

		Exclusivity (RA%)			Frequency (RF %)		
Order	**Family**	**Pre-fire**	**Unstable**	**Stabilizing**	**Pre-fire**	**Unstable**	**Stabilizing**
Basommatophora	Physidae	0	0	100	0	0	25
Coleoptera	Curculionidae	0	0	100	0	0	15
Diptera	Ephydridae	0	60	40	0	14	15
Diptera	Tabanidae	0	0	100	0	0	30
Hemiptera	Corixidae	0	62	38	0	14	15
Hemiptera	Mesoveliidae	0	0	100	0	0	10
Odonata	Libellulidae	0	0	100	0	0	70

Abundance and Post-Fire Trend Analyses

Overall Site Abundance

Mean annual sample abundance dropped to near zero at all monitoring sites immediately following the fire (Figure 11). Post-fire annual abundance was 93% lower at RIT01 compared to pre-fire samples, but we did not detect differences between pre- and post-fire annual sample abundance at the other three sites (Table 10, Figure 11).

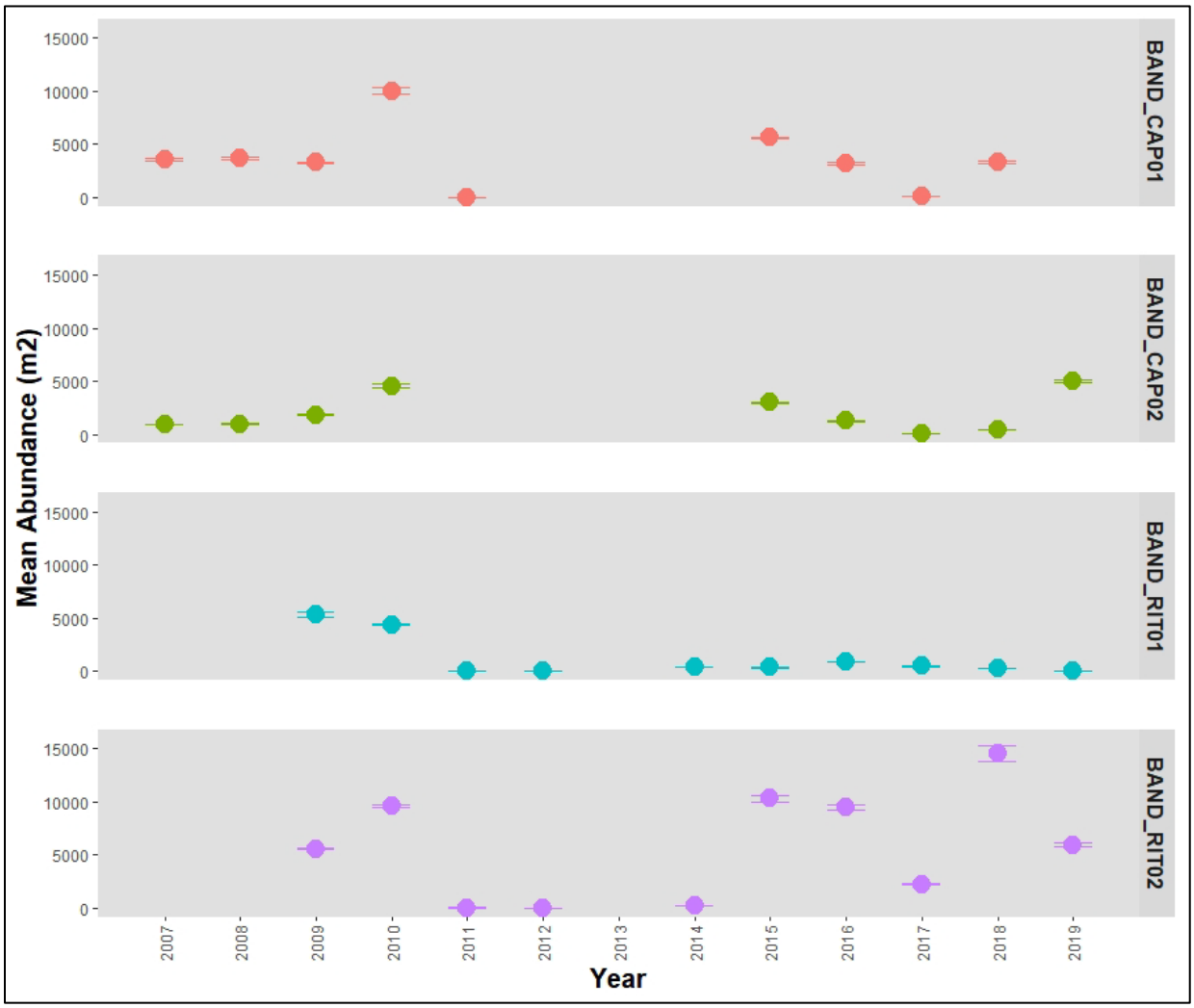

Figure 11. Mean annual aquatic macroinvertebrate sample abundance for four monitoring sites at Bandelier National Monument (Capulin Creek N = 2, Rito de los Frijoles N = 2). Error bars represent +/- 1 standard deviation from the sample mean. Monitoring at RIT01 and RIT02 began in 2009. We were unable to collect samples from CAP01 during 2012 and 2014, and from CAP02 during 2011, 2012, and 2014 due to a lack of appropriate habitat at the monitoring site. No data were collected in 2013 because of a government shutdown.

Table 10. Permutation test results on the difference in means of pre-fire and post-fire site abundance. Significant differences in means are represented where p ≤ 0.05.

Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL
CAP01	5153.3	5377.0	223.7	0.961	−3565.2	4854.7
CAP02	2089.9	1965.5	124.4	0.871	−1158.1	1608.4
RIT01	4875.5	360.0	4515.5	0.0001	2907.6	6700.5
RIT02	7617.1	6107.8	1509.3	0.613	−4706.2	2126.3

Abundance of Dominant Taxa

Prior to the Los Conchas Fire, the dominant families sampled at our monitoring sites were a mix of taxa with varying degrees of tolerance to disturbance. Chironomidae, a highly tolerant midge, was the most common dominant taxon across all monitoring sites in all years (Figures 12–13). Although total abundances of dominant taxa declined immediately following the fire at RIT01, relative abundances of dominant taxa increased. For example, Chironomidae was a dominant taxon in pre-fire samples (2009–2010) at both Rito de los Frijoles sites (RIT01, 12.9%; RIT02, 7.7%) and exhibited post-fire relative abundances of 72% at RIT01 and 91% at RIT02 (Appendix A).

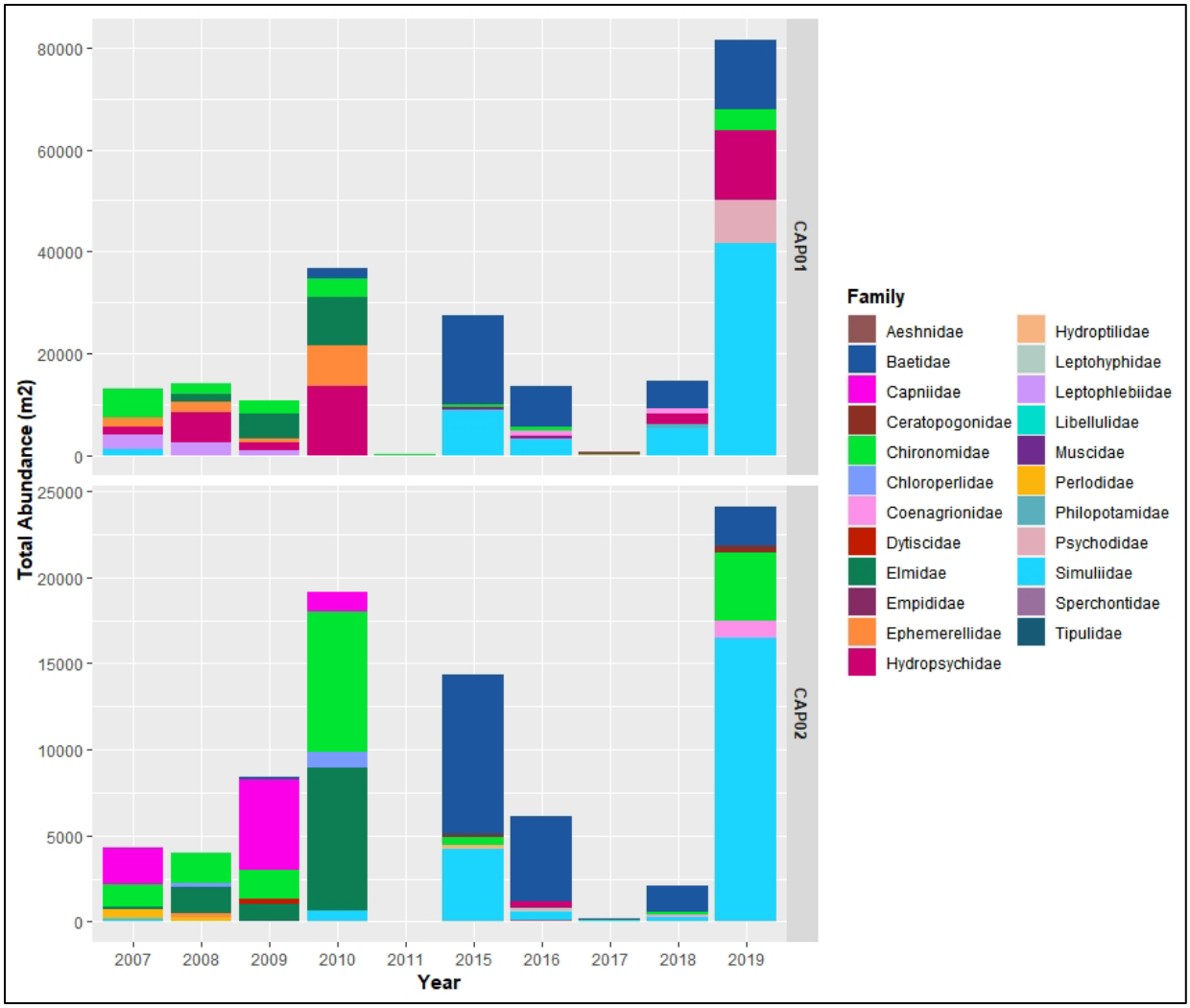

Figure 12. Dominant family abundance at Capulin Creek 2007–2019. Abundance for the top five dominant taxa during each year are presented as stacked bars. Values of individual taxa can be found in Appendix A. We were unable to collect samples from CAP01 during 2012 and 2014, and from CAP02 during 2011, 2012, and 2014 due to a lack of appropriate habitat at the monitoring site. No data were collected in 2013 because of a government shutdown.

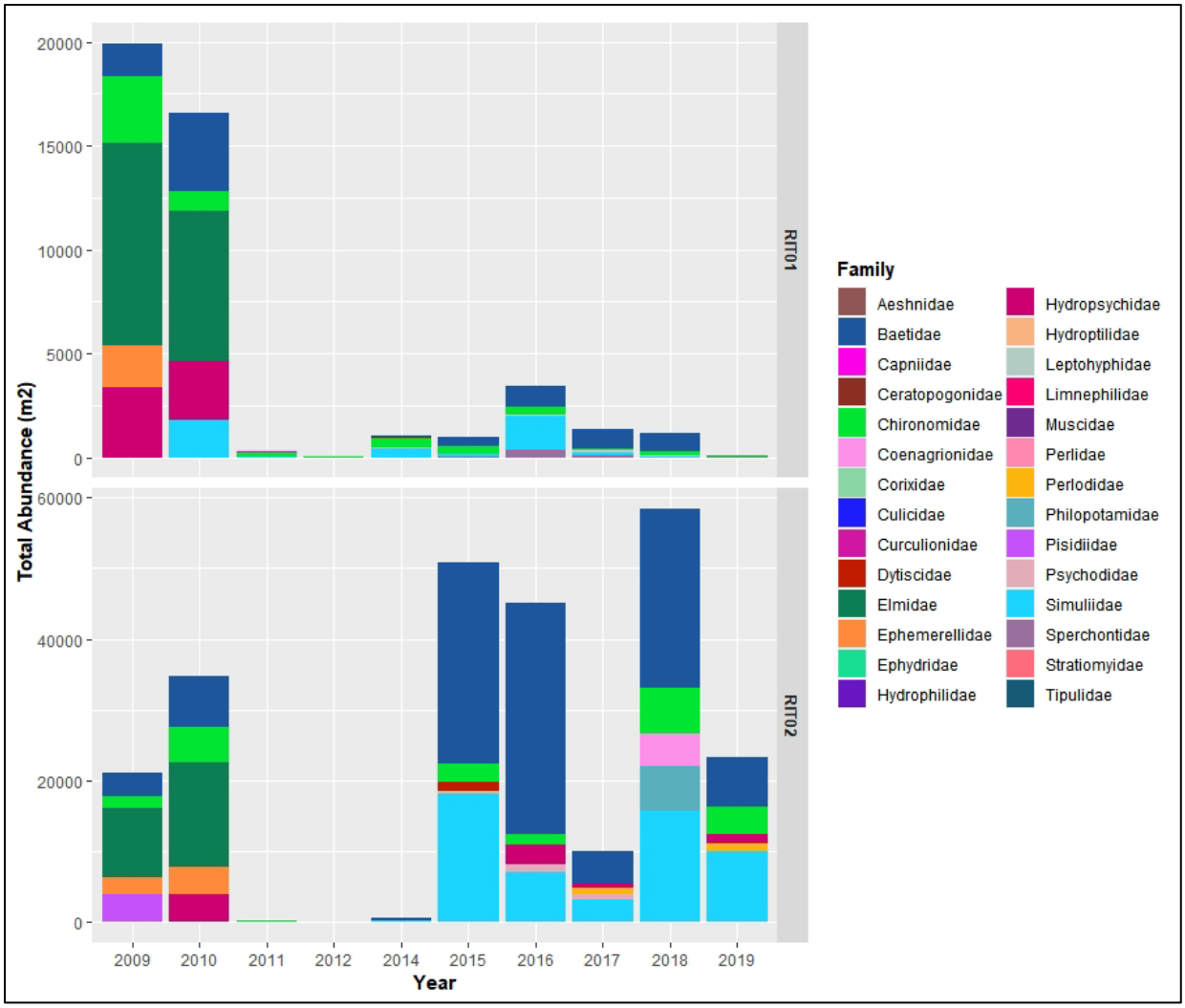

Figure 13. Dominant family abundance at the Rito de los Frijoles 2009–2019. Abundance for the top five dominant taxa during each year are presented as stacked bars. Values of individual taxa can be found in Appendix A. No data were collected in 2013 because of a government shutdown.

Chironomidae was a dominant taxon at both sites on Capulin Creek before the fire (Figure 12). Dominant pre-fire taxa at CAP01 also included disturbance-intolerant mayflies in Leptophlebiidae, the second most abundant family group in 2007 and 2008 and fifth most abundant in 2009 (Figure 12). Ephemerellidae were dominant across all pre-fire years and at both sites, and Hydropsychidae, a caddisfly that is intolerant to disturbance, was among the most dominant taxa at CAP01 (Figure 12). Riffle beetles (Elmidae), which are sensitive to large flow events, were among the most dominant taxa at CAP01 from 2008–2010 (pre-fire). The immediate post-fire community at CAP01 was dominated by Baetidae, Simullidae, and Chironomidae. Baetids were the dominant taxa in five of six post-fire years. Other dominant post-fire taxa at CAP01 included flies and midges (Chironomidae, Muscidae, Psychodidae, Simullidae) and odonates (Coenagrionidae, Libellulidae, Aeshnidae, Figure12).

Similar to CAP01, pre-fire taxa dominance at CAP02 was a mix of tolerant taxa (Chironomidae) and disturbance-intolerant taxa (Figure 12). The stonefly (Plecoptera) families Capniidae, Perlodidae, and Chloroperlidae were among the most dominant taxa in pre-fire samples. The mayflies Ephemerellidae and Baetidae were dominant in 2008 and 2009, respectively, but not consistent across pre-fire years at CAP02. Baetidae is the most dominant taxon in samples at CAP02 in most post-fire years (Figure 12). The most diverse group of dominant taxa in the post-fire community at CAP02 were the flies and midges (Ceratopogonida, Chironomidae, Simuliidae, Tipulidae). The caddisfly families (Hydropsychidae, Hydroptilidae) and odonates (Coenagrionidae and Libellulidae) were dominant during certain years at CAP02 but lack consistency across post-fire years.

The pre-fire community at the Rito de los Frijoles was dominated by Elmidae at both monitoring sites (Figure 13). Baetidae, Chironomidae, and Hydropsychidae also were in the top five dominant taxa across both sites for all pre-fire years. The freshwater pea clam Pisidiidae was the second most abundant taxon at RIT02 in 2009 but was not found to be a dominant taxon in any other year.

The post-fire community in the Rito de los Frijoles was most consistently dominated by Baetidae, Simuliidae, and Chironomidae (Figure 13). From 2015–2019, Baetidae and Simullidae accounted for the majority of taxa abundance at RIT02, driving total abundance of dominant taxa above pre-fire values (Figure 13). The post-fire community at RIT01 also was dominated by these three disturbance-tolerant taxa (Baetidae, Simuliidae, and Chironomidae), but post-fire abundances were considerably lower than pre-fire values (Figure 13).

Common Family Post-Fire Abundance

Among aquatic macroinvertebrate families detected before and after the Las Conchas Fire (N=48), 60% exhibited lower post-fire abundance compared to pre-fire estimates at one or more monitoring sites (Table 11, Appendix B). The other 19 families examined exhibited no difference in abundance between pre- and post-fire periods at any site, and none of 48 families exhibited increased post-fire abundance compared to pre-fire estimates (Appendix B). The proportion of families exhibiting reduced post-fire abundance varied across orders, as 71–100% of families within the EPT group (Ephemeroptera, Plecoptera, Tricoptera) exhibited lower post-fire abundance at one or more sites, but 33–50% of fly (Diptera), damselfly/dragonfly (Odonata), and beetle (Coleoptera) families exhibited reduced abundance post-fire, compared to pre-fire estimates (Table 11).

Table 11. Permutation test summary of results for differences in pre-fire and post-fire abundance in aquatic macroinvertebrate family abundance at two sites on Capulin Creek and two on the Rito de los Frijoles in Bandelier National Monument 2007–2019 (N=48). Appendix B provides site specific results for each family.

Order	Number of Families	Declines in post-fire abundance	No difference in post-fire abundance
Ephemeroptera	7	5 (71%)	2 (29%)
Plecoptera	6	6 (100%)	0 (0%)
Trichoptera	13	10 (77%)	3 (23%)
Coleoptera	6	2 (33%)	4 (67%)
Odonata	4	2 (50%)	2 (50%)
Diptera	12	4 (33%)	8 (67%)

Abundance estimates (i.e., monitoring site and occurrence) of Ephemeroptera families (N=7) indicate a significant post-fire decline in five families (71%) at one or more monitoring sites (Table 11, Appendix B). Forty-three percent (N=12) of ephemeropteran post-fire abundance estimates were lower than pre-fire estimates (Appendix B). Four of five ephemeropteran families that occurred at RIT01 exhibited post-fire declines, and Ephemerellidae exhibited significantly lower abundance across all monitoring sites and was not found in any post-fire samples (Appendix B). Heptageniidae and Lepthoplebiidae occurred at three sites (CAP01, RIT01, and RIT02) prior to the fire, and abundance of both families exhibited a post-fire decline at all three monitoring sites.

Compared to pre-fire estimates, all Plecoptera families exhibited significantly lower post-fire abundance at one or more monitoring sites (Appendix B). Seventy-six percent (N=16) of post-fire plecopteran abundance estimates were lower than pre-fire estimates (Appendix B). Pre-fire and post-fire abundance estimates for Capniidae, Chloropedlidae, and Perlodidae were similar at the RIT02 site, and post-fire estimates were similar to pre-fire estimates for Perlidae and Pteronarcyid at CAP01 (Appendix B). Two families of plecopterans (Chloroperlidae and Pteronarcyidae) were present prior to the fire but were not detected in any post-fire samples. RIT01 was the only monitoring site where post-fire abundance estimate declined for all plecopteran families (Appendix B).

Seventy-seven percent of Trichoptera families (10 of 13) sampled during this study exhibited lower post-fire abundance compared to pre-fire estimates at one or more sites (Table 11, Appendix B). Forty-six percent (N=16) of abundance estimates for all trichopterans (N=35, 2007–2019) were significantly lower post-fire compared to pre-fire (Appendix B). Brachycentridae and Leptoceridae were found in pre-fire samples but were not detected in post-fire samples. Glossosomatide exhibited significant post-fire declines across all three monitoring sites where it occurred pre-fire (CAP01, RIT01, RIT02), and Lepidostomatidae exhibited post-fire declines at three of the four sites (CAP01, RIT01, RIT02, Appendix B). Limnephilidae declined in post-fire abundance at both sites on Capulin Creek, but pre- and post-fire abundance were similar on the Rito de los Frijoles (Appendix B).

Abundance estimates (N=19, 2007–2019) for three of six coleopteran (beetles) families were significantly lower post-fire at one or more sites (Table 11, Appendix B). Elmidae was the only coleopteran family to exhibit significantly lower post-fire abundances across all monitoring sites, and Dytiscidae (CAP02) and Hydrophilidae (RIT01) exhibited lower post-fire abundance at one of four sites (Appendix B).

Two of four Odonata families exhibited lower post-fire abundance compared to pre-fire estimates (Table 11, Appendix B), with 36% of abundance estimates (N=1) lower post-fire (Appendix B). Aeshindae exhibited significantly lower post-fire abundance estimates at three of four monitoring sites (CAP01, CAP02, and RIT01) and was not found in post-fire samples at CAP02 (Appendix B). Cordulegastridae post-fire abundance estimates declined at CAP01 and RIT02 and was not sampled at either site post-fire (Appendix B).

Thirty-three percent of Diptera families (N=12) exhibited lower post-fire abundance compared to pre-fire estimates at one or more sites (Table 11, Appendix B). Seventeen percent of dipteran abundance estimates (8 of 45) were lower post-fire (Appendix B). Tipulidae was the only dipteran family exhibiting lower post-fire abundance across all monitoring sites. Chironomidae declined at CAP01 and RIT01. Psychodidae declined at RIT01 and Ceratopogonidae at RIT02 (Appendix B).

Post-fire Family Group Trend

There was little indication of post-fire recovery among EPT taxa (Appendix C). Of the 26 EPT taxa collected in pre-fire samples, only four (15%) exhibited an increase in post-fire trend in abundance at one or more sites (Appendix C). Ten EPT taxa (38%) were not found in post-fire samples. There were no significant trends in post-fire abundance among ephemeropteran families found in post-fire samples (N=3, Appendix C). Four (67%, N=6) plecopteran families found in pre-fire samples were found in post-fire samples. One of four plecopteran families (Perlodidae) exhibited a positive trend in post-fire abundance at CAP01 (increasing by an average of 8.3 individuals per year), but another (Capniidae) exhibited a negative post-fire trend at RIT01 (declining in abundance by 1.1 individuals per year, Appendix C). Thirty-three percent of trichopteran families found in post-fire samples (three of nine) showed a positive trend in post-fire abundance at RIT02 (Appendix C). Glossosomatidae increased by 2.4 individuals per year (Appendix C), Hydropsychidae increased by 36.8 individual per year (Appendix C), and Philopotamidae increased by 7.3 individuals per year.

Aside from EPT taxa, post-fire abundance trends were investigated in 21 families from three orders (Coleoptera, N=5; Odonata, N=3; Diptera, N=13; Appendix C). Elmidae (Coleoptera) exhibited a positive trend in post-fire abundance at RIT02, increasing by 2.5 individuals per year (Appendix C), and two Odonate families exhibited positive trends in post-fire abundance (Appendix C). Aeshnidae increased by 4.0 individuals per year at RIT02 (Appendix C), and Coenagrionidae exhibited positive post-fire trends at three of four monitoring sites (CAP01 increased by 70.9 individuals per year; CAP02 increased by 3.6 individuals per year; RIT02 increased by 10.5 individuals per year, Appendix C). Seven of 13 dipteran families showed a positive trend in post-fire abundance at RIT02 (Psychodidae, Ceratopogonidae, Empididae, Tipulidae, Dixidae, Stratiomyidae, Tabanidae, Appendix C). Psychodidae and Tipulidae also exhibited positive post-fire trends at CAP01 and CAP02, respectively (Appendix C).

Functional Feeding Group Abundance

Post-fire abundance was lower for the shredder functional feeding group across all monitoring sites compared to pre-fire estimates (Table 12). Annual shredder post-fire abundance was between 326 (CAP01) and 495 (CAP02) individuals lower across sites compared to pre-fire abundance (Table 12). Scrapers exhibited reduced post-fire abundance at three of four sites, all except CAP01 (Table 12). Collector-gatherers and collector-filterers did not exhibit significant differences in pre- and post-fire abundance at three (CAP01, CAP02, RIT02) of four sites, but abundance was lower for both collector-gatherers and collector-filterers during the post-fire period at RIT01 (Table 12).

Table 12. Permutation test results on the difference in means of pre-fire and post-fire functional feeding group abundance at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). Significant differences in means are represented where p ≤ 0.05.

Site	Functional Group	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL
CAP01	Collector-Gatherers	2140.2	1758.7	381.5	0.667	−1013.7	1840.6
	Collector-Filterers	1231.5	2557.7	−1326.2	0.829	−2204.3	1157.2
	Predators	513.4	341.4	172.0	0.529	−249.3	550.2
	Scrapers	809.1	345.6	463.6	0.352	−1298.7	362.3
	Shredders	337.1	10.6	326.6	0.005	208.3	530.1
CAP02	Collector-Gatherers	794.2	939.7	−145.5	0.778	−762.6	914.4
	Collector-Filterers	64.4	882.1	−817.6	0.270	−1974.3	30.7
	Predators	225.1	122.1	103.0	0.333	−32.3	231.3
	Scrapers	490.1	13.0	477.1	0.016	−1492.00	−90.7
	Shredders	498.0	2.9	495.1	0.008	246.0	795.9
RIT01	Collector-Gatherers	1835.8	197.0	1638.8	0.022	968.2	1835.8
	Collector-Filterers	931.4	77.0	854.5	0.022	630.7	1090.2
	Predators	294.4	39.4	255.0	0.022	81.8	444.0
	Scrapers	1392.0	3.8	1388.0	0.001	1254.7	1520.9
	Shredders	385.7	4.4	381.3	0.022	342.0	422.0
RIT02	Collector-Gatherers	3408.8	2987.8	421.0	0.911	−2615.9	1981.7
	Collector-Filterers	1308.7	1703.0	−394.3	0.822	−1706.4	473.7
	Predators	381.4	443.5	−62.1	0.844	−467.7	248.2
	Scrapers	2071.9	196.3	1875.6	0.022	1046.3	2756.76
	Shredders	365.2	15.5	349.7	0.022	215.71	484.27

We found significantly lower post-fire abundances across all functional groups at the RIT01 monitoring site (Table 12). The largest declines occurred in collector-gatherer abundance, which decreased by an average of 1638.8 individuals per year (p = 0.02). Collector-filterer annual abundance declined by 854.5 (p = 0.02) individuals per year, and post-fire scraper and predator annual abundance decreased by 1388 (p = 0.001) and 255 individuals (p = 0.02), respectively (Table 12).

Post-fire Functional Group Trend

Post-fire trends in abundance were not observed in collectors (gatherers and filterers), and RIT02 was the only site where we detected any trend among functional groups (Table 13). Shredders, scrapers, and predators at RIT02 exhibited positive post-fire trends (Table 10, Figures 14–16). Although not statistically significant, collectors (gatherers, p = 0.06; filterers, p = 0.06) exhibited some evidence of a positive post-fire trend in abundance (Table 13).

Table 13. Post-fire trend analysis of functional feeding groups at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). The Thiel-Sen Estimator was used to determine if functional groups are showing signs of recovery indicated by ↑ in the Trend column. Asterisks represent sample sizes that were too small to determine 95% confidence limits.

Site	Functional Group	y-intercept	Slope	Kendall's τ	Lower CL	Upper CL	p-value	Trend
CAP01	Collector-Gatherers	−349719.10	174.15	0.200	−1778.11	1810.35	0.719	↔
	Collector-Filterers	−847734.00	420.83	0.467	−915.97	5601.39	0.272	↔
	Shredders	−3990.67	1.98	0.276	−14.40	14.75	0.566	↔
	Scrapers	−104232.20	51.70	0.733	−3.06	894.95	0.056	↔
	Predators	−136420.70	67.73	0.600	−26.20	594.08	0.136	↔
CAP02	Collector-Gatherers	512222.70	−253.33	−0.200	*	*	0.817	↔
	Collector-Filterers	9198.64	−4.47	0.000	*	*	1.000	↔
	Shredders	−2307.02	1.15	0.527	−1.85	4.75	0.312	↔
	Scrapers	23.72	−0.01	0.000	*	*	1.000	↔
	Predators	−4724.61	2.36	0.000	*	*	1.000	↔
RIT01	Collector-Gatherers	−64594.37	30.66	0.286	−60.75	77.60	0.398	↔
	Collector-Filterers	1688.42	−0.83	−0.214	−32.75	43.72	0.548	↔
	Shredders	2177.90	−1.08	−0.500	−2.64	0.34	0.109	↔
	Scrapers	−541.44	0.27	0.322	−1.77	2.81	0.352	↔
	Predators	−2846.40	1.43	0.143	−15.80	21.21	0.720	↔

Table 13 (continued). Post-fire trend analysis of functional feeding groups at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). The Thiel-Sen Estimator was used to determine if functional groups are showing signs of recovery indicated by ↑ in the Trend column. Asterisks represent sample sizes that were too small to determine 95% confidence limits.

Site	Functional Group	y-intercept	Slope	Kendall's τ	Lower CL	Upper CL	p-value	Trend
RIT02	Collector-Gatherers	−691922.70	344.02	0.571	−53.11	1720.98	0.061	↔
	Collector-Filterers	−712362.70	353.99	0.571	−0.26	953.40	0.060	↔
	Shredders	−8103.85	4.03	0.909	2.62	7.47	0.003	↑
	Scrapers	−86734.20	43.07	0.837	9.93	139.31	0.006	↑
	Predators	−206895.00	102.79	0.714	2.31	276.96	0.010	↑

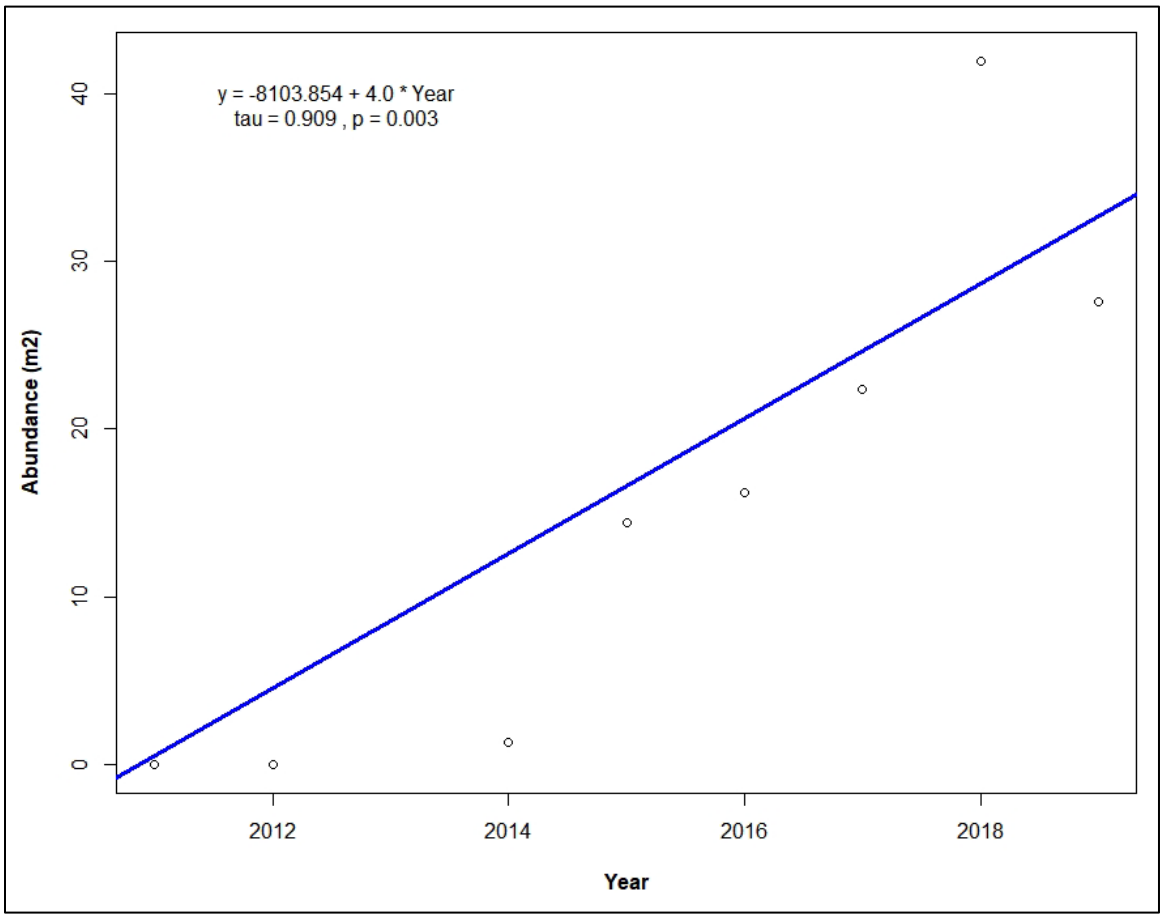

Figure 14. Post-fire trend in shredder abundance at RIT02. The Thiel-Sen estimator (in blue) represents the median slope and intercept of all paired values in the figure. The tau value represents the strength of the correlation between abundance and time. Trend is significant at a $p \leq 0.05$.

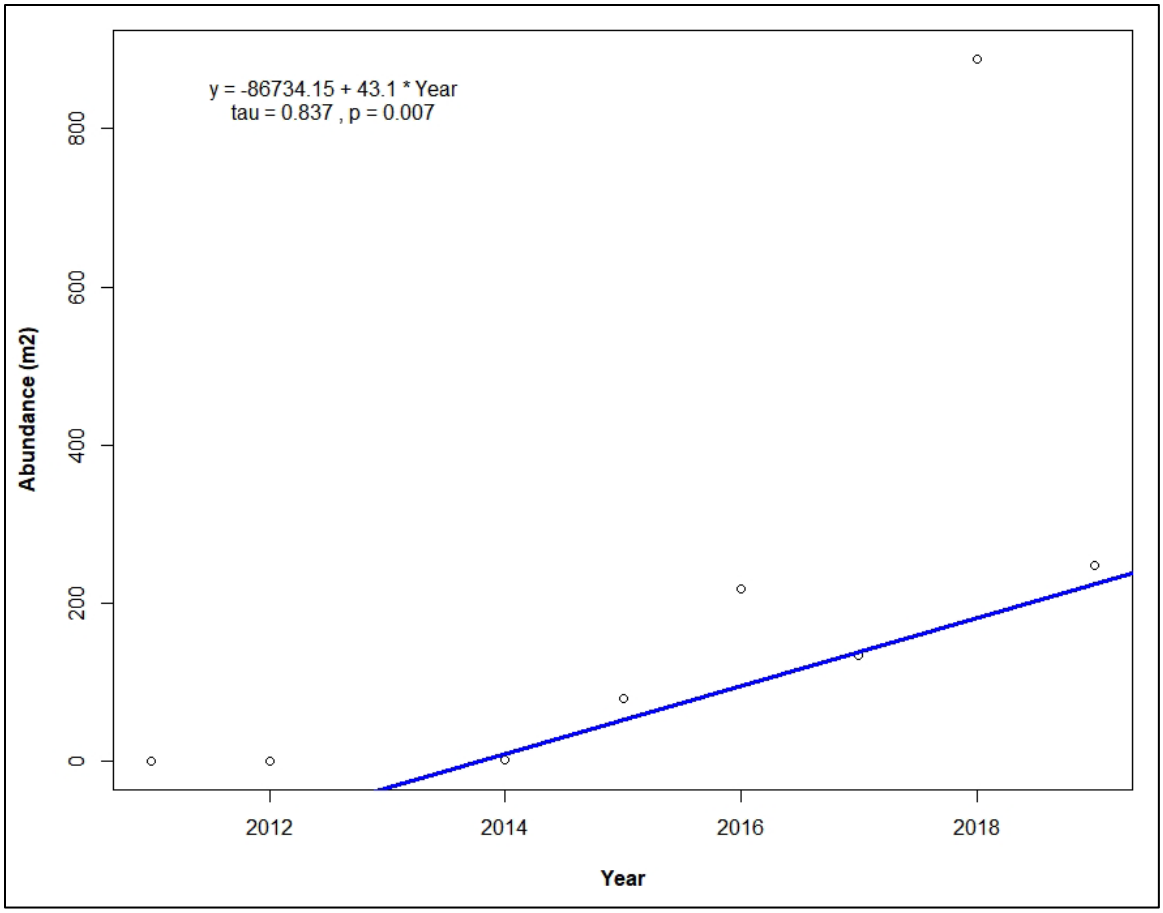

Figure 15. Post-fire trend in scraper abundance at RIT02. The Thiel-Sen estimator (in blue) represents the median slope and intercept of all paired values in the figure. The tau value represents the strength of the correlation between abundance and time. Trend is significant at a $p \leq 0.05$.

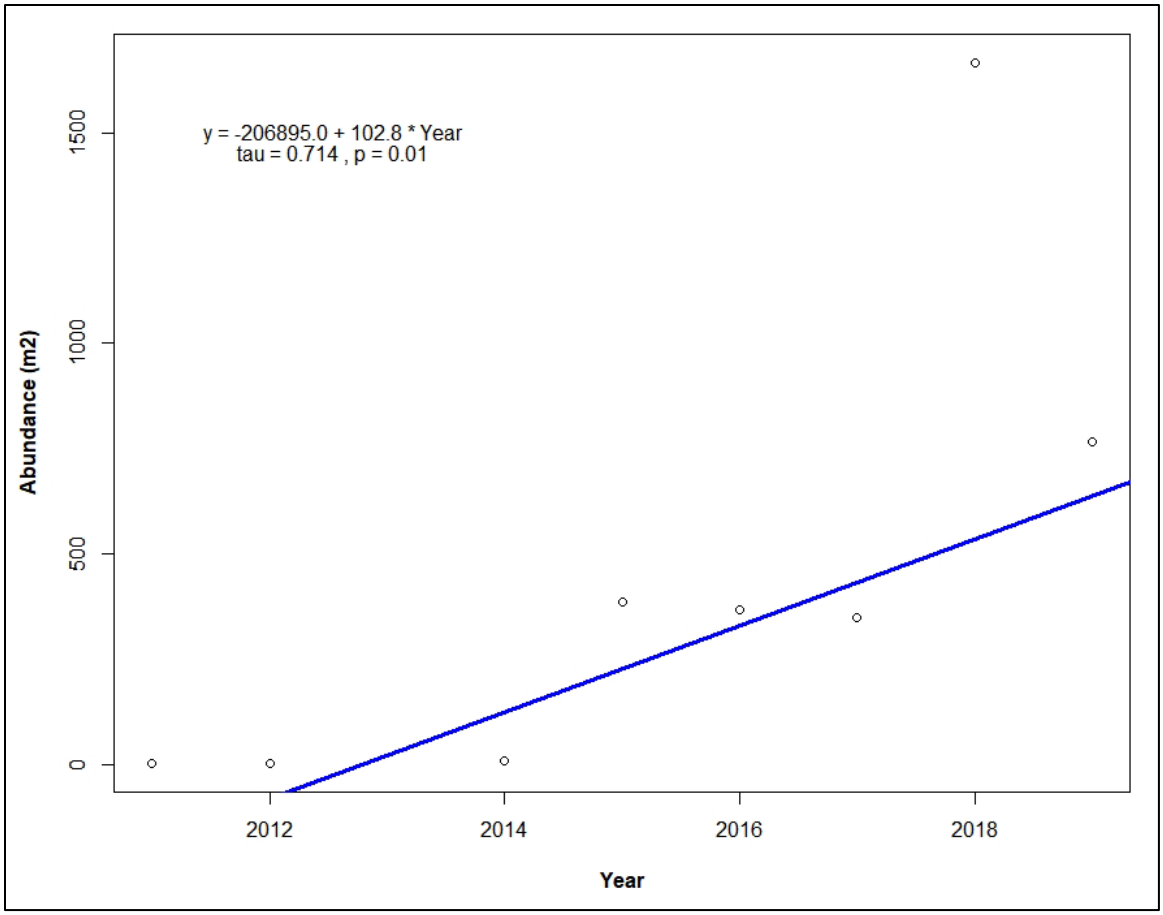

Figure 16. Post-fire trend in predator abundance at RIT02. The Thiel-Sen estimator (in blue) represents the median slope and intercept of all paired values in the figure. The tau value represents the strength of the correlation between abundance and time. Trend is significant at a $p \leq 0.05$.

Mountain-Stream Condition Index

New Mexico's Mountain Stream Condition Index (M-SCI) indicated all monitoring sites met the water quality standard for aquatic life use prior to the Las Conchas Fire (2007–2010, Table 14). Pre-fire M-SCI scores at CAP01 ranged from 77.4 to 83.8 and averaged 81.9. At CAP02, pre-fire scores were relatively lower with a range of 68.4–75.9 and an average of 72.6. Average pre-fire M-SCI scores at the Rito de los Frijoles sites RIT01 and RIT02 were 82.7 and 78.9, respectively.

We found the lowest M-SCI scores (16.4–30.4) immediately post-fire (2011–2014), coinciding with years when hydrologic disturbances were the greatest (Figure 2, Table 14). During the years of stabilizing streamflow (2015–2019), M-SCI scores were generally higher compared to the years immediately post-fire (Table 14). Site scores were fair across sites, with poor conditions detected at one site during 2017 (CAP02) and at another site (RIT01) during 2018–2019 (Table 14). Two sites, CAP01 (2018) and RIT02 (2017–2018, Table 14), had post-fire M-SCI scores that suggested they were not impaired and met the state's water quality standard for aquatic life use.

Table 14. Monitoring site Mountain Stream Condition Index scores at four sampling sites in Bandelier National Monument (2011–2019, Capulin Creek N = 2, Rito de los Frijoles N = 2). Values in green[a] (good) and blue[b] (very good) indicate index scores that meet the state water quality standard for aquatic life use. All other colors[c] indicate impairment and failure to meet the water quality standard. Gray boxes (dashes) indicate years where sample data do not exist.

Site	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
CAP01	77.4[a]	83.2[b]	83.8[b]	83.6[b]	18.3[c]	–	–	–	43.2[c]	46.5[c]	40.0[c]	57.5[a]	48.7[c]
CAP02	68.4[a]	75.9[a]	71.4[a]	74.9[a]	–	–	–	–	44.4[c]	38.7[c]	30.5[c]	40.4[c]	37.9[c]
RIT01	–	–	80.8[b]	84.6[b]	22.3[c]	24.4[c]	–	30.4[c]	41.2[c]	46.6[c]	44.9[c]	32.4[c]	22.1[c]
RIT02	–	–	76.6[a]	81.0[b]	16.4[c]	28.9[c]	–	22.8[c]	46.3[c]	53.2[c]	58.4[a]	60.4[a]	42.9[c]

Discussion

Recovery of aquatic macroinvertebrate densities in stream ecosystems following a fire can vary from a few years up to a decade (Minshall 2003). While this study presents nine years of post-fire data, our results suggest that recovery of aquatic macroinvertebrate communities to pre-fire structure and densities has not occurred. These systems experienced unprecedented post-fire stream flows resulting in the prolonged loss of many taxa. High intensity fires in lower order streams can be particularly harmful to aquatic macroinvertebrate communities (Minshall et al. 1997, Gresswell 1999). These fires typically result in complete removal of adjacent riparian vegetation and large hydrologic disturbances that remove vital food and habitat resources. For many aquatic macroinvertebrate taxa, drift (hydraulic transport downstream) is unidirectional with flow (Rader 1997), and as such, the burning of headwater streams eliminates an important mechanism of post-disturbance recolonization of downstream sites. Recolonization is then dependent on taxa with specific functional traits (Godoy et al. 2021) or aerial adults from undisturbed streams and riparian habitat. Post-fire recovery times of sensitive taxa may be prolonged in Capulin Creek and the Rito de los Frijoles because of the size and intensity of fire. Large stand replacing wildfires that burn through multiple neighboring watersheds have the potential to reduce the opportunities for recolonization by many taxa and increase recovery times. The fire and floods at Bandelier worked successively to increase the distance to potential recolonization populations. This was facilitated first, by increasing the distance of populations from unburned riparian and forested communities, then by reducing or removing populations via flooding, which likely resulted in the extirpation of taxa as well as habitat and refugia.

Effects of Fire and Historic Flows on Community Structure

Natural disturbance is the primary force structuring aquatic macroinvertebrate communities in stream ecosystems (Resh et al. 1998, Fisher and Grimm 1991, Poff 1992). Following high severity fire, hydrologic disturbances are the primary factor controlling post-fire aquatic macroinvertebrate survival and persistence (Gresswell 1999). Much like previous studies investigating the effects of fire on the aquatic community in Bandelier (Pippin and Pippin 1980, Vieira et al. 2004), results from this study indicate fire and post-fire floods reduced aquatic macroinvertebrate densities and diversity to near zero, suggesting the aquatic community showed little resistance to those disturbances. Following the La Mesa Fire, peak stream flows resulted in the transport of suspended-sediment loads that were 220 times the annual sediment load in Frijoles Canyon (Veenhuis 2002). Stream flows on the Rito de los Frijoles between 2011–2014 produced the largest flow events ever recorded in the system, including multiple peak flow events that were 2–3 times larger than those that followed the La Mesa Fire (Figure 2). It is reasonable to assume that these flow events contributed to the observed declines found in aquatic macroinvertebrate abundances. The long-term impacts of these disturbances to the aquatic community are determined by the community's resistance (extent of change caused by the disturbance) to the disturbance as well as its resilience (rate of recovery). Resistance and resilience differ among aquatic taxa and are the result of an organism's morphology and life history strategies (Poff et al. 2018). Thus, aquatic macroinvertebrate communities not only reflect the long-term physical and chemical conditions of the stream, but its natural disturbance regime as well (Resh et al. 1988).

Thirteen families of aquatic macroinvertebrates sampled pre-fire were not found in post-fire samples at any of the monitoring sites; 77% of families absent from post-fire samples are sensitive EPT taxa (Table 8). Post-fire invertebrate abundances declined to near zero at most sites (Figure 11). In 2015, after the large historic flows began to subside, we observed an increase in abundance at most monitoring sites. Our data indicate post-fire mean total abundances have increased above pre-fire values at CAP01 and that CAP02 and RIT02 have recovered >80% of pre-fire mean abundance values (Figure 11). This result is similar to other studies showing burned aquatic systems can result in increased aquatic macroinvertebrate abundances and densities. Martens et al. (2019) showed that burned catchments along the eastern slopes of the northern Rocky Mountains exhibited macroinvertebrate abundances that were three times higher than unburned catchments. Similarly, Mellon et al. (2008) found that densities of drifting and benthic aquatic macroinvertebrates were three times higher in burned sites compared to unburned sites in forested headwater streams in northeastern Washington. Musetta-Lambert et al. (2019) also reported headwater streams within burned areas contained significantly higher aquatic macroinvertebrate abundance compared to reference (unburned) streams in eastern forests.

The resilience in macroinvertebrate abundance observed at Bandelier was not evenly distributed across the community. Post-fire richness and diversity values remained lower than pre-fire at all sites (Figures 6–9). In the years immediately following the fire, we observed a shift in taxa dominance at Capulin Creek and the Rito de los Frijoles from sensitive taxa such as Lepthophlebiidae, Ephemerllidae (mayflies), Hydropsychidae (caddisfly), Capniidae and Perlodidae (stoneflies) and Elmidae (riffle beetle) to an overwhelming dominance by Baetidae (mayflies), Simullidae (blackflies), and Chironomidae (midges) (Figures 12–13). Results are in accord with previously described post-fire abundance in these streams (Pippin and Pippin 1980, Vieira et al, 2004), and several other studies have shown results similar in distinct systems. Repeated experimental flood events on the River Spöl and adjacent tributaries resulted in a reduction of taxa richness, and the majority of post disturbance aquatic macroinvertebrate density was attributed to recolonization by Chironomidae, Simuliidae, and Baetidae (Robinson et al. 2004). DeColibus et al. (2014) reported a large hydrologic disturbance decreased diversity, primarily through the loss of riffle dominant EPT taxa (mayflies, stoneflies, and caddisflies) that were replaced by chironomids in a third-order stream in Ohio. Mellon et al. (2008) reported dominance by chironomids and baetids in the two years post-fire, and Martens et al. (2019) found chironomids and baetids can make up the bulk of relative abundances for several years following fire.

Several factors contribute to an organism's ability to recolonize a stream following disturbance (i.e., resilience). These include the magnitude and timing of the disturbance, an organism's life history traits, presence of refugia, and distance to intact populations (Resh et al. 1988, Niemi et al. 1990). Poff et al. (2018) described life history traits that promote resistance and resilience following a 100-year flood. Resistance traits can include morphological adaptations that allow organisms to remain in place during large flow events, such as structures that allow taxa to attach to substrates and body shapes that are resistant to shear forces experienced during high flow events (Mazuuco et al 2015, Godoy et al. 2021). Resistance to crushing from substrates can promote persistence at times of high flows when debris in the system is expected to be at its greatest. Conversely, resistance traits may

also confer greater mobility (swimmers and drifters). Resistance traits can allow organisms greater opportunity to find local refugia or to escape downstream to higher order reaches where the effects of high flows may be muted. Resilience traits include the presence of terrestrial adults at the time of disturbance as well as the ability for larvae to drift long distances (i.e., disperse). The presence of ovipositing terrestrial adults or multivoltine (multiple annual reproductive cycles) aerial adults (resilience traits) increase the likelihood that an organism can recolonize an area after a disturbance. Resistance and resilience traits combined with a generalist feeding strategy allow an organism to take advantage of post-fire environmental conditions (Vieira et al. 2004).

The dominance of post-fire abundances by chironomids, simullids, and baetids can be explained by traits that promote resistance and resilience to disturbance. All exhibit morphological, life history, and functional traits that promote resistance and resilience. They are highly mobile taxa that can drift or swim long distances (Rader 1997). These taxa are generalist feeders that prey on allochthonous and autochthonous organic inputs found in the stream ecosystem (Merritt et al. 2008). For example, baetids have feeding appendages that are specialized for feeding on periphyton (Martens et al. 2019, Merritt et al. 2008), and algal biomass has been shown to increase 5–20 times following fire and flooding in burned streams (Klose et al. 2015). The post-fire dominance of baetid mayflies could be attributed, in part, to their ability to take advantage of post-fire algal blooms. Additionally, taxa exhibiting post-fire dominance (chironomids, baetids, simuliids) have terrestrial adult life stages that are multivoltine and have extended periods of oviposition throughout the year (Merritt et al. 2008). Because these taxa have multiple broods over a longer period of the year, they have a greater opportunity to recolonize a stream once flows stabilize (Mellon et al. 2008, Minshall 2003). Aquatic taxa with contracted periods of oviposition, or those that have only one brood annually (univoltine), are less resilient to disturbance timing. Many organisms have evolved reproductive strategies synched with typical seasonal flows, e.g., spring runoff or summer precipitation events (Huryn et al. 2008). Disturbances that occur outside of the typical period of high flows can suppress resilience for these taxa (Niemi et al. 1990, Vieira et al. 2004).

Indicator species analysis results suggest voltinism was an important factor in describing pre-fire communities. Univoltine or semivoltine (one reproductive cycle every two years) life histories occurred in 93% of families indicative of pre-fire community structure (Table 7). The prevalence of these taxa in pre-fire samples suggests annual monsoonal disturbance was not detrimental to persistence. All eight of the EPT families extirpated from the post-fire community at our monitoring sites (Table 8) were univoltine (Merrit et al. 2008). While this reproductive strategy is not expected to be the sole cause for decline in these taxa, it is likely that reproductive strategies work concordantly with other life history traits, trophic relationships, and habits (mode of existence) to restrict resilience (Niemi et al. 1990, Vieira et al. 2004, Poff et al. 2018).

Post-fire trend in community structure recovery

Trend analyses of post-fire (2011–2019) aquatic community structure indicated few families present prior to the fire are showing signs of recovery. We found 38% of families (14 of 37) exhibited a positive post-fire trend in abundance at one or more sites, and one plecopteran family (Capniidae) exhibited a negative trend in abundance (Appendix C). Recovery appears to be strongest at the

upstream site on the Rito de los Frijoles (RIT02) as 93% of families exhibiting positive post-fire abundance trends were increasing in abundance at this site (Appendix C). Half of the families exhibiting a positive post-fire abundance trend (seven of 14) were dipterans (flies and midges). In addition to being tolerant to disturbance, many dipteran families are multivoltine with very short generation times. These life history characteristics allow them to quickly recolonize disturbed stream ecosystems (Thorpe and Rogers 2011).

Four families of sensitive EPT taxa (Capniidae, Hydropsychidae, Glossosomatidae, Philopotamidae) exhibited a post-fire trend in abundance (Appendix C). Capniidae (Plecoptera) at RIT01 was the only family exhibiting a negative post-fire trend in abundance (Appendix C), but a second plecopteran (Perlodidae) exhibited a positive trend in abundance at CAP01 (Appendix C). Capniidae are univoltine and are extremely weak swimmers (Hilsenhoff 2001). Large summer flows can remove these taxa from the system. Because they are poor swimmers, nymphs are not likely to recolonize upstream reaches once flooding subsides. Adult capniids typically emerge from streams in winter; this timing would prevent local populations from recolonizing sites disturbed by summer flooding.

Three trichopteran (EPT) families exhibited an increasing post-fire trend in abundance at RIT02 (Appendix C). Hydropsychidae showed the strongest positive trend in abundance of any EPT taxa, increasing at a rate of 36.8 individuals per year. Philopotamidae was rare prior to the fire (15.2 individuals/m^2) but increased at a rate of 7.3 individuals/year post-fire. The saddle casemaker caddisfly, Glossosomatidae, showed signs of slow recovery, having recovered only 10% of its pre-fire abundance at RIT02 and increasing by 2.4 individuals per year. These trichopteran families are univoltine or semivoltine, attach themselves to cobble sized rocks in riffles along the stream bed, and lack mobility (Rader 1997, Merritt et al. 2008), making them particularly susceptible to extreme floods. Previous fires delivered sediment loads that were orders of magnitude larger than the average annual loads (Veenhuis 2002) seen in these systems. The tuft and pumice that make up the surrounding canyon walls and bottom lands (Reneau et al. 2007) are extremely erodible, and fine sediments were quite abundant in the system prior to Las Conchas. It is likely that habitat loss through the aggradation of fine sediments has slowed trichopteran recovery. However, increasing trend in abundance of these trichopteran families at RIT02 suggests this site may be the only site where we are starting to see recovery in cobble and riffle habitat. Two of these families, Philopotamidae and Glossosomatidae, are case-building trichopterans that rely on smaller size gravels to build cases and cobbles to attach those cases to. Likewise, Hydropsychidae build retreats into gravels and woody debris which provide shelter when foraging. These taxa are all considered to be sensitive to disturbance and have tracheal gills which make them particularly vulnerable when dissolved oxygen declines. Furthermore, Hydropsychidae and Glossosomatidae are both scraper taxa that feed on periphyton, suggesting an important energetic pathway is being restored at this site. The presence and increasing abundance of these trichopterans at the RIT02 site suggests that recovery is beginning along this middle reach of the Rito de los Frijoles.

Ecosystem Function

The aquatic macroinvertebrate community reflects energy flows (function) through stream ecosystems. Ecosystem function can be described by looking at the feeding mechanisms of taxa

present (Cummins and Klug 1979). Taxa are clustered into functional feeding groups (FFG) based on their morphological similarities in feeding appendages. These groups describe how the stream community processes autochthonous and allochthonous inputs of energy (primary production) and converts that energy into biomass. In undisturbed stream ecosystems, the aquatic community would be expected to be structured to efficiently process energetic inputs with minimal loss to downstream reaches (Vannote et al. 1980). First order, or headwater, streams like those in Bandelier are streams where allochthonous inputs in the form of riparian vegetation and forest detritus dominate. Vannote et al. (1980) theorize that in low order stream communities we would expect to see a codominance in shredder and gatherer taxa. Shredder and collector-gatherer taxa have evolved feeding appendages designed to break down coarse particulate matter such as leaves, woody debris, and aquatic vegetation. Collector-filterer taxa feed on the finer particulates that leak down from the shredders and collector-gatherer processing of coarser materials. Scrapers, who primarily break down autochthonous inputs of energy (periphyton), and predators are present but less abundant in headwater streams. Our pre-fire data suggest that streams in Bandelier differ somewhat from this theorized model. One important distinction is that scrapers appear to be an important functional group in pre-fire Bandelier communities. Scraper abundance was high across all sites prior to the fire. This suggests periphyton (algae, bacteria, fungi) were an important autochthonous source of energy in streams prior to the fire. Post-fire loss of riparian canopy (foliar cover) increases the amount of sunlight hitting burned reaches. As a result, we would expect a large increase in periphyton at our sites (Vannote et al. 1980).

Results from FFG analyses suggest the Las Conchas Fire led to alterations of ecosystem function in both streams. Our data show significant declines occurred in shredder and scarper taxa. Post-fire shredder abundance decreased by >95% compared to pre-fire values at all monitoring sites. Significant declines in scraper abundance occurred at both sites on the Rito de los Frijoles and at CAP02 (Table 12). While not statistically significant, scraper abundance decreased by over half of pre-fire abundance at CAP01. We expect to see an increase in woody debris over time as standing and downed dead trees decay and wood enters the post-fire system (Gresswell, 1999). Loss in abundance of shredders and scrapers suggests study reaches may be ill-equipped to process expected increases in energetic inputs, and these systems are likely functioning less efficiently comparted to pre-fire conditions. Recovery of post-fire functional group abundance is limited to the RIT02 monitoring site. Shredders, scrapers, and predators at RIT02 show significant increasing trends in post-fire abundance; however, recovery is very slow. Shredder abundance at RIT02 decreased by >95% of its pre-fire value (365.2 individuals/m^2, Figure 14). Our results show that while there is a positive trend in post-fire shredder abundance, the increase is occurring at a rate of 4.0 individuals per year. Similarly, scrapers lost 90% of their pre-fire abundance (2071.9 individuals/m^2) and appear to be increasing by 43.1 individuals/year (Figure 15). Recovery of shredder and scraper taxa at RIT02 suggests the community may (slowly) be regaining the capacity to break down primary production at this site.

Management Implications

Bioassessments like the SCI have long been favored by ecologists as an inexpensive tool for investigating ecological integrity of streams (Plafkin et al. 1989, Barbour et al. 1999). In 2006 the

State of New Mexico developed a multimeric Mountain Stream Condition Index (M-SCI) to determine stream impairment through the assessment of the aquatic macroinvertebrate community (Jacobi et al. 2006). The design of the M-SCI was based on the conditions found at reference stream sites throughout New Mexico. Both Capulin Creek and the Rito de los Frijoles were sampled as reference stream sites to create this index.

Index scores calculated from pre-fire aquatic macroinvertebrate samples indicate that all sites were unimpaired and met the cold-water aquatic life use water quality standard. The upstream sites on both streams had higher pre-fire condition scores than downstream counterparts. During the post-fire years of unstable streamflow (2011–2014), condition scores indicated extreme impairment, with samples designated as "poor" or "very poor" (Table 14). Condition scores fluctuated during the post-fire years when stream flows begin to stabilize. Index scores reached the "good" designation at RIT02 in 2017 and 2018, and at CAP01 in 2018, indicating a lack of impairment for those years. However, scores from both sites regressed to an impaired state the following year (Table 14). Index scores from the two downstream monitoring sites on our streams suggest that impairment has gone unabated since the fire and floods.

Index scores at RIT01 support results from family abundance and functional feeding group analyses that suggest the aquatic macroinvertebrate community continues to experience disturbance at RIT01. In 2018 and 2019, condition scores from RIT01 declined to the "poor" designation and were the lowest at that site since stream flows stabilized in 2015. These results are peculiar as RIT01 was the only site in our study that did not burn. While flooding resulted in the decline of some streamside riparian vegetation (unpublished data), the canopy at RIT01 remained mostly intact. RIT01 was the only monitoring site where all functional feeding groups showed significant declines in post-fire abundance (Table 12). There is a growing body of literature showing repeated disturbance to aquatic macroinvertebrate communities can negatively affect community resilience and function (Graeber et al. 2017, Haghkerdar et al. 2019). In their investigation of aquatic macroinvertebrate community response to fire across multiple biogeographical regions, Verkaik et al. (2015) suggested that drought/drying following large fire induced floods can suppress community recovery and resistance to future disturbance. Anecdotal evidence from crew observations and communications with monument staff suggest that this lower reach has experienced increased flow intermittency since the fire. There appears to be a drying period during the warmest months of the year, with flows returning to the site briefly following monsoon precipitation and then more consistently as temperatures cool in October. Index scores at RIT01 are likely due to a combination of extended flow intermittency at the site and the timing of our sampling.

The data provided by the M-SCI can provide monument managers with valuable information regarding food base for stream fishes. The Las Conchas Fire and subsequent floods are believed to have extirpated all fish from both Capulin Creek and the Rito de los Frijoles. This provided an opportunity for monument staff at Bandelier to collaborate with NMDGF and local NGOs to develop a reintroduction plan for the native Rio Grande cutthroat trout. In 2018, 306 trout were introduced into the Rito de los Frijoles upstream from our RIT02 monitoring site (Brauer and Bakevich 2019). Our results suggest that the aquatic life use water quality standard was being met in 2017 and 2018 at

our upper Rito de los Frijoles site, but conditions appear to fluctuate from year to year. The site fell back into an impaired state in 2019. Data from the lower site on the Rito de los Frijoles suggest that our lower reach is not suitable to sustain cold-water fish populations and that fish migrating downstream may find conditions uninhabitable. The benefit of the M-SCI index is that it uses aquatic macroinvertebrate based metrics that describe ecosystem attributes (community structure, diversity, function, and disturbance tolerance) and contextualizes that information to describe a stream's ability to sustain cold-water fish populations. Because this study suggests that resilience in abundance of most taxa in the Rito de los Frijoles is slow or non-existent, fish may have difficulty establishing populations in lower reaches of this stream.

Conclusions

The Las Conchas Fire and subsequent hydrologic disturbances resulted in severe and lasting negative impacts on the structure and function of the aquatic macroinvertebrate community. Although previous studies have attempted to describe the effects of fire on the aquatic community in Bandelier National Monument (Pippin and Pippin 1980, Vieira et al. 2004), this is the first long-term study to document change across multiple stream ecosystems in the monument. The data presented in this report suggest that the impacts of this fire on the aquatic community are ongoing. There is very little consistency in post-fire recovery across family groups or monitoring sites. The loss in abundance of two key functional groups, shredders and scrapers, suggest a disruption in ecosystem function is occurring in both streams. Annual increases in post-fire family and functional group abundance are small and observed at few sites, suggesting recovery is slow (Appendix C, Table 13). Evidence of recovery is highest at the RIT02 monitoring site, which experienced the highest burn severity among sites. Our results suggest recovery is localized and that stream communities in Bandelier are depauperate nearly a decade after the fire.

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Appendix A. Dominant Taxa

The top five dominant taxa for sites CAP01, CAP02, RIT01, and RIT02 are presented in tables A1 through A4.

Table A1. Total abundance and relative abundance of the top five dominant taxa at the CAP01 monitoring site for each year.

Year	Family	Total Abundance/ m^2	Mean Relative Abundance %
2007	Chironomidae	5595.4	27.0
	Leptophlebiidae	2977.7	19.5
	Ephemerellidae	1854.3	8.2
	Hydropsychidae	1470.3	9.5
	Simuliidae	1054.3	4.8
2008	Hydropsychidae	5890.3	26.8
	Leptophlebiidae	2527.1	13.9
	Chironomidae	2148.0	13.4
	Ephemerellidae	2000.2	10.9
	Elmidae	1563.8	8.8
2009	Elmidae	4873.7	27.4
	Chironomidae	2648.0	15.9
	Hydropsychidae	1651.4	9.7
	Ephemerellidae	835.4	4.5
	Leptophlebiidae	825.1	6.0
2010	Hydropsychidae	13556.0	23.6
	Elmidae	9484.0	20.4
	Ephemerellidae	7904.0	15.2
	Chironomidae	3800.0	7.4
	Baetidae	1904.0	5.2
2011	Chironomidae	24.0	28.6
	Simuliidae	8.0	22.2
	Muscidae	4.0	4.8
	Baetidae	4.0	11.1

Table A1 (continued). Total abundance and relative abundance of the top five dominant taxa at the CAP01 monitoring site for each year.

Year	Family	Total Abundance/ m^2	Mean Relative Abundance %
2015	Baetidae	17635.2	62.6
	Simuliidae	8959.5	32.1
	Chironomidae	462.4	1.7
	Hydropsychidae	277.3	1.0
	Empididae	210.7	0.7
2016	Baetidae	8053.1	49.0
	Simuliidae	3273.1	18.4
	Coenagrionidae	888.0	4.3
	Chironomidae	705.9	4.1
	Hydropsychidae	561.9	2.6
2017	Baetidae	268.0	44.4
	Coenagrionidae	96.0	11.6
	Libellulidae	56.0	7.3
	Dytiscidae	48.0	7.9
	Aeshnidae	32.0	4.3
2018	Simuliidae	5406.7	30.1
	Baetidae	5252.0	31.3
	Hydropsychidae	1938.7	13.2
	Coenagrionidae	1093.3	6.2
	Philopotamidae	786.7	5.2
2019	Baetidae	13681.1	18.1
	Psychodidae	8431.7	10.3
	Chironomidae	4062.9	5.8
	Coenagrionidae	2834.4	4.1
	Sperchontidae	1918.9	1.9

Table A2. Total abundance and relative abundance of the top five dominant taxa at the CAP02 monitoring site for each year.

Year	Family	Total Abundance/ m^2	Mean Relative Abundance %
2007	Capniidae	2144.0	43.1
	Chironomidae	1264.0	36.0
	Perlodidae	480.0	5.4
	Simuliidae	224.0	4.5
	Elmidae	192.0	2.4
2008	Chironomidae	1770.3	41.9
	Elmidae	1494.9	22.2
	Chloroperlidae	272.6	4.0
	Ephemerellidae	248.6	4.5
	Perlodidae	237.7	2.1
2009	Capniidae	5236.0	55.4
	Chironomidae	1692.0	18.1
	Elmidae	1032.0	10.5
	Dytiscidae	272.0	2.9
	Baetidae	192.0	2.1
2010	Elmidae	8308.0	28.4
	Chironomidae	8188.0	39.0
	Capniidae	1104.0	8.0
	Chloroperlidae	868.0	3.3
	Simuliidae	672.0	3.8
2015	Baetidae	9250.9	62.2
	Simuliidae	4223.7	28.6
	Chironomidae	415.0	2.3
	Hydroptilidae	234.1	1.2
	Ceratopogonidae	211.0	1.1
2016	Baetidae	4932.0	76.9
	Simuliidae	472.0	7.8
	Hydropsychidae	380.0	6.2
	Leptohyphidae	248.0	3.6
	Sperchontidae	104.0	1.5

Table A2 (continued). Total abundance and relative abundance of the top five dominant taxa at the CAP02 monitoring site for each year.

Year	Family	Total Abundance/ m^2	Mean Relative Abundance %
2017	Baetidae	72.0	28.7
	Libellulidae	36.0	15.6
	Coenagrionidae	32.0	12.3
	Simuliidae	24.0	11.1
	Tipulidae	16.0	3.2
2018	Baetidae	1460.0	65.6
	Simuliidae	256.0	9.9
	Chironomidae	176.0	7.0
	Leptohyphidae	116.0	5.9
	Coenagrionidae	56.0	1.8
2019	Simuliidae	16462.2	66.9
	Chironomidae	3931.6	14.9
	Baetidae	2333.9	9.0
	Coenagrionidae	1008.7	3.7
	Ceratopogonidae	385.1	1.5

Table A3. Total abundance and relative abundance of the top five dominant taxa at the RIT01 monitoring site for each year.

Year	Family	Total Abundance/ m^2	Mean Relative Abundance %
2009	Elmidae	9763.1	38.7
	Hydropsychidae	3370.2	15.0
	Chironomidae	3196.5	10.8
	Ephemerellidae	2022.1	3.2
	Baetidae	1582.1	8.9
2010	Elmidae	7244.3	33.6
	Baetidae	3764.5	16.8
	Hydropsychidae	2776.5	13.4
	Simuliidae	1823.5	7.5
	Chironomidae	984.3	4.1

Table A3 (continued). Total abundance and relative abundance of the top five dominant taxa at the RIT01 monitoring site for each year.

Year	Family	Total Abundance/ m^2	Mean Relative Abundance %
2011	Chironomidae	224.0	71.9
	Capniidae	24.0	7.5
	Simuliidae	8.0	3.5
	Psychodidae	4.0	1.2
	Limnephilidae	4.0	1.2
	Aeshnidae	4.0	1.2
	Tipulidae	4.0	1.2
2012	Capniidae	20.0	33.3
	Chironomidae	8.0	3.5
	Simuliidae	4.0	1.8
2014	Chironomidae	452.0	33.5
	Simuliidae	400.0	30.4
	Baetidae	92.0	7.1
	Corixidae	56.0	6.1
	Ceratopogonidae	52.0	3.2
2015	Baetidae	416.0	38.5
	Chironomidae	372.0	35.2
	Simuliidae	112.0	12.0
	Muscidae	28.0	2.5
	Sperchontidae	28.0	2.3
2016	Simuliidae	1612.0	34.7
	Baetidae	1000.0	36.2
	Chironomidae	376.0	7.4
	Sperchontidae	348.0	7.7
	Leptohyphidae	96.0	1.8
2017	Baetidae	944.0	69.2
	Sperchontidae	116.0	7.7
	Leptohyphidae	116.0	4.5
	Simuliidae	100.0	3.4
	Chironomidae	84.0	5.0

Table A3 (continued). Total abundance and relative abundance of the top five dominant taxa at the RIT01 monitoring site for each year.

Year	Family	Total Abundance/ m^2	Mean Relative Abundance %
2018	Baetidae	904.0	68.5
	Chironomidae	176.0	16.2
	Leptohyphidae	76.0	4.4
	Elmidae	8.0	0.6
	Simuliidae	8.0	0.3
2019	Sperchontidae	36.0	37.9
	Baetidae	24.0	27.1
	Hydrophilidae	8.0	7.1
	Chironomidae	8.0	7.1
	Hydroptilidae	4.0	5.0

Table A4. Total abundance and relative abundance of the top five dominant taxa at the RIT02 monitoring site for each year.

Year	Family	Total Abundance/ m^2	Mean Relative Abundance %
2009	Elmidae	9766.9	34.9
	Pisidiidae	3991.3	13.7
	Baetidae	3272.2	11.9
	Ephemerellidae	2334.5	8.7
	Chironomidae	1742.0	6.2
2010	Elmidae	14846.7	29.9
	Baetidae	7125.3	15.5
	Chironomidae	4912.0	9.2
	Hydropsychidae	3954.7	8.8
	Ephemerellidae	3856.0	8.1
2011	Chironomidae	280.0	90.6
	Ephydridae	16.0	4.3
	Simuliidae	4.0	1.0
	Culicidae	4.0	1.1

Table A4 (continued). Total abundance and relative abundance of the top five dominant taxa at the RIT02 monitoring site for each year.

Year	Family	Total Abundance/ m^2	Mean Relative Abundance %
2011 (continued)	Muscidae	4.0	1.0
	Hydrophilidae	4.0	1.2
2012	Chironomidae	12.0	42.9
	Muscidae	4.0	14.3
	Simuliidae	4.0	14.3
	Perlidae	4.0	14.3
	Baetidae	4.0	14.3
2014	Baetidae	356.0	56.1
	Simuliidae	212.0	29.8
	Chironomidae	60.0	8.6
	Dytiscidae	12.0	1.5
	Muscidae	12.0	1.7
2015	Baetidae	28397.6	55.0
	Simuliidae	18166.2	33.3
	Chironomidae	2558.1	6.2
	Dytiscidae	1288.7	2.6
	Hydroptilidae	344.1	0.9
2016	Baetidae	32689.6	69.4
	Simuliidae	7129.1	15.3
	Hydropsychidae	2721.9	6.4
	Chironomidae	1492.0	2.8
	Psychodidae	1053.3	1.2
2017	Baetidae	4496.0	41.8
	Simuliidae	3304.0	26.2
	Perlodidae	1008.0	9.6
	Psychodidae	604.0	4.5
	Hydropsychidae	552.0	5.0
2018	Baetidae	25300.0	33.7
	Simuliidae	15845.3	24.4
	Chironomidae	6495.1	9.1

Table A4 (continued). Total abundance and relative abundance of the top five dominant taxa at the RIT02 monitoring site for each year.

Year	Family	Total Abundance/ m^2	Mean Relative Abundance %
2018 (continued)	Philopotamidae	6167.6	7.4
	Coenagrionidae	4631.6	6.9
2019	Simuliidae	10108.5	31.4
	Baetidae	7040.5	24.5
	Chironomidae	3913.3	13.0
	Hydropsychidae	1289.9	4.4
	Perlodidae	1039.7	3.7

Appendix B. Results from Family Abundance Permutation Tests

Table B1. Difference in pre-fire and post-fire means in abundance for each family taxa at each monitoring site. Significant differences in abundance are shown where p ≤ 0.05. Direction of difference is shown in the Change column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL	Change
Ephmeroptera	Ameletidae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Ameletidae	CAP02	0.20	0.00	0.20	0.44	0.00	0.60	↔
	Ameletidae	RIT01	1.42	0.00	1.42	0.02	1.28	1.55	↓
	Ameletidae	RIT02	1.07	0.00	1.07	0.20	0.00	2.13	↔
	Baetidae	CAP01	148.71	1496.53	−1347.82	0.12	−2687.43	−420.66	↔
	Baetidae	CAP02	15.29	721.95	−706.67	0.07	−1324.23	−203.05	↔
	Baetidae	RIT01	534.67	115.85	418.82	0.02	155.26	666.42	↓
	Baetidae	RIT02	1039.75	2463.09	−1423.34	0.62	−2849.22	498.08	↔
	Ephemerellidae	CAP01	629.69	0.00	629.69	0.00	235.14	1227.37	↓
	Ephemerellidae	CAP02	23.63	0.00	23.63	0.01	9.20	39.13	↓
	Ephemerellidae	RIT01	242.37	0.00	242.37	0.02	80.32	404.43	↓
	Ephemerellidae	RIT02	619.05	0.00	619.05	0.02	466.91	771.20	↓
	Heptageniidae	CAP01	7.30	0.00	7.30	0.01	3.87	11.31	↓
	Heptageniidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Heptageniidae	RIT01	1.44	0.00	1.44	0.02	0.89	2.08	↓
	Heptageniidae	RIT02	117.66	0.00	117.66	0.02	46.78	188.53	↓
	Leptohyphidae	CAP01	6.09	11.20	−5.11	0.42	−15.25	6.85	↔

Table B1 (continued). Difference in pre-fire and post-fire means in abundance for each family taxa at each monitoring site. Significant differences in abundance are shown where p ≤ 0.05. Direction of difference is shown in the Change column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL	Change
Ephmeroptera (continued)	Leptohyphidae	CAP02	1.42	18.89	−17.47	0.10	−30.39	−6.33	↔
	Leptohyphidae	RIT01	1.97	10.07	−8.09	0.67	−17.61	0.50	↔
	Leptohyphidae	RIT02	0.00	2.34	−2.34	0.67	−4.68	−0.10	↔
	Leptophlebiidae	CAP01	359.10	0.00	359.10	0.01	167.71	545.42	↓
	Leptophlebiidae	CAP02	1.46	0.00	1.46	0.44	0.00	4.37	↔
	Leptophlebiidae	RIT01	141.20	0.00	141.20	0.02	60.93	221.44	↓
	Leptophlebiidae	RIT02	329.28	37.17	292.12	0.02	129.92	479.95	↓
	Oligoneuriidae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Oligoneuriidae	CAP02	3.00	0.00	3.00	0.44	0.00	9.00	↔
	Oligoneuriidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Oligoneuriidae	RIT02	n/a	n/a	n/a	n/a	n/a	n/a	–
Coleoptera	Curculionidae	CAP01	0.00	0.27	−0.27	1.00	−0.80	0.00	↔
	Curculionidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Curculionidae	RIT01	0.00	0.25	−0.25	1.00	−0.75	0.00	↔
	Curculionidae	RIT02	0.00	0.16	−0.16	1.00	−0.64	0.00	↔
	Dryopidae	CAP01	0.40	0.67	−0.27	0.50	−1.26	0.56	↔
	Dryopidae	CAP02	0.43	0.16	0.27	0.40	−0.32	0.82	↔
	Dryopidae	RIT01	1.20	0.13	1.08	0.20	−0.25	2.40	↔
	Dryopidae	RIT02	0.00	0.20	−0.20	1.00	−0.80	0.00	↔

Table B1 (continued). Difference in pre-fire and post-fire means in abundance for each family taxa at each monitoring site. Significant differences in abundance are shown where p ≤ 0.05. Direction of difference is shown in the Change column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL	Change
Coleoptera (continued)	Dytiscidae	CAP01	4.60	8.41	−3.81	0.60	−13.05	3.92	↔
	Dytiscidae	CAP02	32.14	5.50	26.65	0.03	9.39	40.03	↓
	Dytiscidae	RIT01	1.20	3.63	−2.43	0.51	−5.76	−0.80	↔
	Dytiscidae	RIT02	57.80	36.77	21.03	0.80	−31.49	76.15	↔
	Elmidae	CAP01	829.85	0.96	828.89	0.01	178.54	1627.56	↓
	Elmidae	CAP02	551.34	0.00	551.34	0.01	122.40	1292.50	↓
	Elmidae	RIT01	170.73	0.49	1700.24	0.00	1448.09	1952.43	↓
	Elmidae	RIT02	2461.35	24.12	2437.23	0.02	1914.36	2966.82	↓
	Hydraenidae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Hydraenidae	CAP02	2.00	0.00	0.20	0.44	0.00	0.58	↔
	Hydraenidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Hydraenidae	RIT02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Hydrophilidae	CAP01	0.40	3.56	−3.16	0.28	−7.07	0.20	↔
	Hydrophilidae	CAP02	0.80	4.27	−3.47	0.36	−8.58	−0.13	–
	Hydrophilidae	RIT01	3.02	0.50	2.52	0.04	0.06	4.48	↓
	Hydrophilidae	RIT02	0.00	0.56	−0.56	0.82	−1.11	−0.01	↔
Plecoptera	Capniidae	CAP01	28.89	0.36	28.53	0.01	13.66	45.47	↓
	Capniidae	CAP02	429.60	0.32	429.28	0.01	71.08	891.92	↓
	Capniidae	RIT01	152.53	2.46	150.07	0.02	119.81	181.26	↓

Table B1 (continued). Difference in pre-fire and post-fire means in abundance for each family taxa at each monitoring site. Significant differences in abundance are shown where p ≤ 0.05. Direction of difference is shown in the Change column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL	Change
Plecoptera (continued)	Capniidae	RIT02	9.60	4.81	4.79	0.38	−8.51	15.69	↔
	Chloroperlidae	CAP01	66.84	0.00	66.84	0.00	28.23	127.81	↓
	Chloroperlidae	CAP02	57.23	0.00	57.23	0.05	2.23	126.50	↓
	Chloroperlidae	RIT01	64.04	0.00	64.04	0.02	24.53	103.54	↓
	Chloroperlidae	RIT02	2.68	0.00	2.38	0.20	0.00	5.36	↔
	Nemouridae	CAP01	177.96	0.00	177.96	0.00	92.08	293.31	↓
	Nemouridae	CAP02	27.06	0.00	27.06	0.05	0.81	77.75	↓
	Nemouridae	RIT01	93.31	0.00	93.31	0.02	37.12	149.51	↓
	Nemouridae	RIT02	117.42	0.43	117.00	0.02	45.46	188.51	↓
	Perlidae	CAP01	0.60	0.31	0.29	0.73	−0.53	1.44	↔
	Perlidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	Perlidae	RIT01	22.27	0.13	22.14	0.02	19.71	24.59	↓
	Perlidae	RIT02	184.57	0.17	184.40	0.00	151.74	217.07	↓
	Perlodidae	CAP01	175.36	16.59	158.78	0.00	106.44	223.03	↓
	Perlodidae	CAP02	48.69	0.48	18.21	0.00	24.98	72.62	↓
	Perlodidae	RIT01	15.75	0.62	15.12	0.02	13.68	16.56	↓
	Perlodidae	RIT02	56.89	84.12	−27.23	0.84	−82.68	55.99	↔
	Pteronarcyidae	CAP01	0.40	0.00	0.40	0.40	0.00	0.12	↔
	Pteronarcyidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–

Table B1 (continued). Difference in pre-fire and post-fire means in abundance for each family taxa at each monitoring site. Significant differences in abundance are shown where p ≤ 0.05. Direction of difference is shown in the Change column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL	Change
Plecoptera (continued)	Pteronarcyidae	RIT01	50.06	0.00	50.06	0.02	40.75	59.37	↓
	Pteronarcyidae	RIT02	n/a	n/a	n/a	n/a	n/a	n/a	–
Odonata	Aeshnidae	CAP01	110.88	5.08	105.07	0.00	66.29	156.19	↓
	Aeshnidae	CAP02	6.09	0.00	6.09	0.00	0.40	16.66	↓
	Aeshnidae	RIT01	17.94	17.00	17.77	0.02	3.13	32.41	↓
	Aeshnidae	RIT02	0.40	76.16	−75.76	0.64	−175.26	−3.50	↔
	Coenagrionidae	CAP01	50.83	164.35	−113.52	0.41	−264.54	21.15	↔
	Coenagrionidae	CAP02	0.60	46.42	−45.82	0.42	−123.73	−5.80	↔
	Coenagrionidae	RIT01	0.80	0.13	0.68	0.20	−0.25	1.60	↔
	Coenagrionidae	RIT02	0.00	135.42	−135.41	0.42	−341.37	−3.84	↔
	Cordulegastridae	CAP01	2.86	0.00	2.86	0.03	0.40	5.31	↓
	Cordulegastridae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Cordulegastridae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Cordulegastridae	RIT02	0.80	0.00	0.80	0.02	0.80	0.80	↓
	Libellulidae	CAP01	0.00	4.13	−4.13	0.13	−7.55	−0.62	↔
	Libellulidae	CAP02	0.00	6.51	−6.51	0.06	−10.31	−2.75	↔
	Libellulidae	RIT01	0.00	0.23	−0.23	0.38	−0.48	0.00	↔
	Libellulidae	RIT02	0.00	0.84	−0.84	0.69	−4.54	−0.02	↔

Table B1 (continued). Difference in pre-fire and post-fire means in abundance for each family taxa at each monitoring site. Significant differences in abundance are shown where p ≤ 0.05. Direction of difference is shown in the Change column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL	Change
Trichoptera	Brachycentridae	CAP01	1.66	0.00	1.66	0.13	0.00	3.60	↔
	Brachycentridae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Brachycentridae	RIT01	100.06	0.00	100.06	0.02	19.05	181.07	↓
	Brachycentridae	RIT02	169.88	0.00	169.88	0.02	164.27	175.49	↓
	Calamoceratidae	CAP01	0.55	0.00	0.55	0.09	0.00	1.37	↔
	Calamoceratidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Calamoceratidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Calamoceratidae	RIT02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Glossosomatidae	CAP01	112.56	0.00	112.56	0.00	6.59	279.92	↓
	Glossosomatidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Glossosomatidae	RIT01	34.68	0.50	34.18	0.02	31.91	36.90	↓
	Glossosomatidae	RIT02	67.63	6.84	60.80	0.02	12.12	104.49	↓
	Helicopsychidae	CAP01	21.42	1.33	20.09	0.04	0.59	50.42	↓
	Helicopsychidae	CAP02	0.20	0.00	0.20	0.44	0.00	0.60	↔
	Helicopsychidae	RIT01	5.49	0.00	5.49	0.20	0.00	10.97	↔
	Helicopsychidae	RIT02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Hydropsychidae	CAP01	1128.40	553.22	575.18	0.43	−494.65	1955.88	↔
	Hydropsychidae	CAP02	2.63	24.53	−21.90	0.33	−44.22	0.87	↔
	Hydropsychidae	RIT01	614.67	0.42	614.26	0.00	554.82	674.01	↓

Table B1 (continued). Difference in pre-fire and post-fire means in abundance for each family taxa at each monitoring site. Significant differences in abundance are shown where p ≤ 0.05. Direction of difference is shown in the Change column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL	Change
Trichoptera (continued)	Hydropsychidae	RIT02	534.29	154.43	379.86	0.07	73.91	709.47	↔
	Hydroptilidae	CAP01	0.20	19.08	−18.88	0.21	−39.24	−2.52	↔
	Hydroptilidae	CAP02	10.09	10.22	−0.13	1.00	−18.63	17.77	↔
	Hydroptilidae	RIT01	1.44	0.88	0.57	0.64	−1.32	2.36	↔
	Hydroptilidae	RIT02	0.00	10.16	−10.16	0.67	−27.09	−1.39	↔
	Lepidostomatidae	CAP01	34.32	0.00	34.32	0.01	2.53	96.20	↓
	Lepidostomatidae	CAP02	0.40	0.16	0.24	0.72	−0.32	1.18	↔
	Lepidostomatidae	RIT01	26.30	0.00	26.30	0.02	4.70	47.89	↓
	Lepidostomatidae	RIT02	67.19	0.00	67.19	0.02	15.99	118.40	↓
	Leptoceridae	CAP01	3.60	0.00	3.60	0.00	1.05	7.00	↓
	Leptoceridae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Leptoceridae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Leptoceridae	RIT02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Limnephilidae	CAP01	16.40	0.13	16.26	0.01	2.90	41.71	↓
	Limnephilidae	CAP02	6.83	0.00	6.83	0.00	0.26	18.20	↓
	Limnephilidae	RIT01	3.36	0.50	2.86	0.20	−0.82	6.54	↔
	Limnephilidae	RIT02	0.00	4.29	−4.29	0.44	−7.32	−1.29	↔
	Philopotamidae	CAP01	0.00	27.07	−27.07	0.40	−79.15	−0.31	↔
	Philopotamidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–

Table B1 (continued). Difference in pre-fire and post-fire means in abundance for each family taxa at each monitoring site. Significant differences in abundance are shown where p ≤ 0.05. Direction of difference is shown in the Change column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL	Change
Trichoptera (continued)	Philopotamidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Philopotamidae	RIT02	15.23	175.52	−160.30	0.76	−468.63	12.45	↔
	Polycentropodidae	CAP01	7.80	0.27	7.53	0.03	0.02	21.32	↓
	Polycentropodidae	CAP02	0.00	0.16	−0.16	1.00	−0.48	0.00	↔
	Polycentropodidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Polycentropodidae	RIT02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Rhyacophilidae	CAP01	0.00	0.27	−0.27	1.00	−0.80	0.00	↔
	Rhyacophilidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Rhyacophilidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Rhyacophilidae	RIT02	26.97	0.00	26.97	0.02	25.15	28.80	↓
	Uenoidae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Uenoidae	CAP02	0.00	0.16	−0.16	0.00	−0.48	0.00	↔
	Uenoidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Uenoidae	RIT02	21.04	0.00	21.04	0.02	14.93	27.16	↓
Diptera	Ceratopogonidae	CAP01	18.67	22.52	−3.86	0.92	−33.42	21.02	↔
	Ceratopogonidae	CAP02	9.60	26.08	−16.48	0.43	−46.28	6.92	↔
	Ceratopogonidae	RIT01	21.46	3.83	17.63	0.09	−2.10	36.89	↔
	Ceratopogonidae	RIT02	23.86	5.77	18.09	0.04	10.35	27.09	↓
	Chironomidae	CAP01	709.57	202.60	506.78	0.05	163.54	801.24	↓

Table B1 (continued). Difference in pre-fire and post-fire means in abundance for each family taxa at each monitoring site. Significant differences in abundance are shown where p ≤ 0.05. Direction of difference is shown in the Change column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL	Change
Diptera (continued)	Chironomidae	CAP02	645.71	186.63	462.09	0.23	−77.41	1257.72	↔
	Chironomidae	RIT01	418.08	64.15	353.93	0.02	126.21	606.50	↓
	Chironomidae	RIT02	665340.00	379.76	285.64	0.53	−219.79	722.12	↔
	Dixidae	CAP01	7.12	3.58	3.54	0.17	0.17	6.62	↔
	Dixidae	CAP02	0.40	2.37	−1.97	0.41	−4.96	0.24	↔
	Dixidae	RIT01	0.40	1.38	−0.98	0.49	−2.04	0.13	↔
	Dixidae	RIT02	1.61	10.89	−9.28	0.62	−19.64	2.33	↔
	Dolichopodidae	CAP01	0.00	1.87	−1.87	0.40	−5.48	0.00	↔
	Dolichopodidae	CAP02	0.20	0.16	0.04	1.00	−0.32	0.40	↔
	Dolichopodidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Dolichopodidae	RIT02	0.00	1.51	−1.51	0.67	−2.84	0.00	↔
	Empididae	CAP01	3.98	18.83	−14.85	0.26	−30.00	−1.10	↔
	Empididae	CAP02	4.54	6.64	−2.09	0.69	−9.39	5.54	↔
	Empididae	RIT01	1.98	2.13	−0.15	1.00	−2.23	1.78	↔
	Empididae	RIT02	5.19	21.42	−16.23	0.49	−32.38	−4.27	↔
	Ephydridae	CAP01	0.00	0.40	−0.40	0.47	−0.93	0.00	↔
	Ephydridae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Ephydridae	RIT01	0.00	0.17	−0.17	1.00	−0.48	0.00	↔
	Ephydridae	RIT02	0.00	0.40	−0.40	1.00	−1.20	0.00	↔

Table B1 (continued). Difference in pre-fire and post-fire means in abundance for each family taxa at each monitoring site. Significant differences in abundance are shown where p ≤ 0.05. Direction of difference is shown in the Change column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL	Change
Diptera (continued)	Muscidae	CAP01	0.00	3.38	−3.38	0.06	−6.10	−1.12	↔
	Muscidae	CAP02	1.00	11.46	−10.46	0.41	−23.42	0.47	↔
	Muscidae	RIT01	0.00	3.50	−3.50	0.69	−7.12	0.00	↔
	Muscidae	RIT02	0.00	8.35	−8.35	0.42	−16.05	−0.70	↔
	Psychodidae	CAP01	58.48	310.81	−252.33	0.87	−812.63	67.24	↔
	Psychodidae	CAP02	2.40	0.80	1.60	0.64	−1.26	5.84	↔
	Psychodidae	RIT01	102.07	2.33	99.73	0.00	43.04	154.91	↓
	Psychodidae	RIT02	21.54	153.88	−132.37	0.51	−269.88	8.52	↔
	Simullidae	CAP01	83.70	1977.40	−1893.70	0.30	−4524.90	−289.14	↔
	Simullidae	CAP02	49.20	857.52	−898.31	0.31	−1990.97	17.34	↔
	Simullidae	RIT01	210.25	76.58	133.67	0.13	−31.81	309.22	↔
	Simullidae	RIT02	83.55	1372.93	−1289.38	0.33	−2047.23	−450.68	↔
	Stratiomyidae	CAP01	2.97	21.58	−18.61	0.49	−42.92	2.04	↔
	Stratiomyidae	CAP02	2.26	6.96	−4.70	0.93	−17.22	3.11	↔
	Stratiomyidae	RIT01	1.33	1.38	−0.04	0.89	−1.73	1.90	↔
	Stratiomyidae	RIT02	0.00	3.89	−3.89	0.58	−7.83	−0.86	↔
	Tabanidae	CAP01	0.00	0.53	−0.53	0.47	−1.30	0.00	↔
	Tabanidae	CAP02	0.00	0.21	−0.21	1.00	−0.64	0.00	↔
	Tabanidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–

Table B1 (continued). Difference in pre-fire and post-fire means in abundance for each family taxa at each monitoring site. Significant differences in abundance are shown where p ≤ 0.05. Direction of difference is shown in the Change column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	Pre-fire Mean	Post-fire Mean	Difference in Means	p-value	Lower CL	Upper CL	Change
Diptera (continued)	Tabanidae	RIT02	0.00	11.35	−11.35	0.84	−18.37	−0.03	↔
	Tipulidae	CAP01	105.08	13.14	91.95	0.01	51.01	136.68	↓
	Tipulidae	CAP02	46.37	0.28	43.57	0.00	6.05	92.05	↓
	Tipulidae	RIT01	88.35	1.02	87.33	0.02	59.97	114.76	↓
	Tipulidae	RIT02	82.43	12.14	70.28	0.00	44.25	99.64	↓

Appendix C. Post-fire Family Trend Analysis Results

Table C1. Theil-Sen estimator results of post-fire trend for family taxa at each monitoring site. Strength of trend is determined by Kendall's tau value where values approaching −1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trend is shown where p ≤ 0.05. Asterisks (*) are used to denote tests where there were insufficient data to develop confidence intervals. Direction of trend is shown in the Trend column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	y-intercept	Slope	Kendall's tau	Lower CL	Upper CL	p-value	Trend
Ephmeroptera	Baetidae	CAP01	300875.60	149.87	0.20	−1736.72	1341.31	0.72	↔
	Baetidae	CAP02	699410.40	−346.53	−0.40	*	*	0.48	↔
	Baetidae	RIT01	−56310.94	27.98	0.47	−31.67	64.67	0.14	↔
	Baetidae	RIT02	−457796.80	227.71	0.50	−738.96	1419.88	0.11	↔
	Leptohyphidae	CAP01	−5289.87	2.62	0.55	−31.07	14.83	0.18	↔
	Leptohyphidae	CAP02	10017.74	−4.96	−0.20	*	*	0.82	↔
	Leptohyphidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Leptohyphidae	RIT02	−100.78	0.05	0.45	0.00	1.56	0.20	↔
	Leptophlebiidae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Leptophlebiidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Leptophlebiidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Leptophlebiidae	RIT02	−5641.80	2.80	0.81	0.00	42.22	0.81	↔
Coleoptera	Curculionidae	CAP01	0.00	0.00	−0.35	−0.40	0.00	0.56	↔
	Curculionidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Curculionidae	RIT01	0.00	0.00	0.50	*	*	0.19	↔
	Curculionidae	RIT02	0.00	0.00	0.50	*	*	0.19	↔

Table C1 (continued). Theil-Sen estimator results of post-fire trend for family taxa at each monitoring site. Strength of trend is determined by Kendall's tau value where values approaching −1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trend is shown where p ≤ 0.05. Asterisks (*) are used to denote tests where there were insufficient data to develop confidence intervals. Direction of trend is shown in the Trend column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	y-intercept	Slope	Kendall's tau	Lower CL	Upper CL	p-value	Trend
Coleoptera (continued)	Dryopidae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Dryopidae	CAP02	0.00	0.00	0.00	*	*	1.00	↔
	Dryopidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Dryopidae	RIT02	0.00	0.00	0.21	0.00	0.00	0.21	↔
	Dytiscidae	CAP01	8286.83	−4.10	−0.41	−8.53	2.46	0.34	↔
	Dytiscidae	CAP02	2826.20	−1.40	−0.20	*	*	0.82	↔
	Dytiscidae	RIT01	2.33	0.00	0.08	−2.00	2.20	0.90	↔
	Dytiscidae	RIT02	0.40	0.00	−0.80	−15.81	6.33	0.89	↔
	Elmidae	CAP01	0.00	0.00	0.58	0.00	1.44	0.58	↔
	Elmidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	↔
	Elmidae	RIT01	−425.49	0.21	0.12	0.00	0.33	0.54	↔
	Elmidae	RIT02	−4954.93	2.46	0.70	0.00	22.55	0.03	↑
	Hydrophilidae	CAP01	−267.53	0.13	0.14	−7.20	4.27	0.85	↔
	Hydrophilidae	CAP02	−2015.40	1.00	0.20	*	*	0.82	↔
	Hydrophilidae	RIT01	0.00	0.00	0.50	0.00	0.00	0.19	↔
	Hydrophilidae	RIT02	0.00	0.00	−0.37	−0.13	0.00	0.32	↔
Plecoptera	Capniidae	CAP01	0.00	0.00	−0.35	−0.53	0.00	0.56	↔
	Capniidae	CAP02	0.00	0.00	0.32	0.00	0.80	0.72	↔

Table C1 (continued). Theil-Sen estimator results of post-fire trend for family taxa at each monitoring site. Strength of trend is determined by Kendall's tau value where values approaching −1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trend is shown where p ≤ 0.05. Asterisks (*) are used to denote tests where there were insufficient data to develop confidence intervals. Direction of trend is shown in the Trend column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	y-intercept	Slope	Kendall's tau	Lower CL	Upper CL	p-value	Trend
Plecoptera (continued)	Capniidae	RIT01	2127.97	−1.10	−0.81	−1.40	0.00	0.01	↓
	Capniidae	RIT02	−1476.03	0.73	0.32	−1.96	3.64	0.32	↔
	Nemouridae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Nemouridae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Nemouridae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Nemouridae	RIT02	0.00	0.00	0.37	0.00	0.26	0.32	↔
	Perlidae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Perlidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Perlidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Perlidae	RIT02	0.00	0.00	−0.36	0.00	0.00	0.38	↔
	Perlodidae	CAP01	−16660.53	8.27	0.83	0.00	23.09	0.04	↑
	Perlodidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Perlodidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Perlodidae	RIT02	−56084.47	27.85	0.73	0.00	51.99	0.73	↔
Odonata	Aeshnidae	CAP01	−2494.19	1.24	0.41	−3.12	3.79	0.34	↔
	Aeshnidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Aeshnidae	RIT01	0.00	0.00	−0.50	0.00	0.00	0.19	↔
	Aeshnidae	RIT02	−8061.60	4.00	0.81	0.00	66.95	0.01	↑

Table C1 (continued). Theil-Sen estimator results of post-fire trend for family taxa at each monitoring site. Strength of trend is determined by Kendall's tau value where values approaching −1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trend is shown where p ≤ 0.05. Asterisks (*) are used to denote tests where there were insufficient data to develop confidence intervals. Direction of trend is shown in the Trend column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	y-intercept	Slope	Kendall's tau	Lower CL	Upper CL	p-value	Trend
Odonata (continued)	Coenagrionidae	CAP01	−142790.80	70.86	0.87	0.93	273.84	0.02	↑
	Coenagrionidae	CAP02	−7254.80	3.60	0.95	0.00	190.54	0.04	↑
	Coenagrionidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Coenagrionidae	RIT02	−21094.10	10.47	0.87	0.27	132.33	0.01	↑
	Libellulidae	CAP01	2.67	0.00	0.00	*	*	1.00	↔
	Libellulidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Libellulidae	RIT01	0.00	0.00	0.26	0.00	0.13	0.51	↔
	Libellulidae	RIT02	−597.19	0.30	0.62	0.00	0.64	0.62	↔
Trichoptera	Glossosomatidae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Glossosomatidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Glossosomatidae	RIT01	0.00	0.00	0.21	0.00	0.00	0.66	↔
	Glossosomatidae	RIT02	−4849.84	2.41	0.81	0.00	6.31	0.01	↑
	Helicopsychidae	CAP01	0.00	0.00	0.35	0.00	1.14	0.56	↔
	Helicopsychidae	CAP02	0.00	0.00	0.35	0.00	1.14	0.56	↔
	Helicopsychidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Helicopsychidae	RIT02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Hydropsychidae	CAP01	−223254.70	110.76	0.73	−25.33	1377.07	0.06	↔
	Hydropsychidae	CAP02	−4089.76	2.03	0.20	*	*	0.82	↔

Table C1 (continued). Theil-Sen estimator results of post-fire trend for family taxa at each monitoring site. Strength of trend is determined by Kendall's tau value where values approaching −1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trend is shown where p ≤ 0.05. Asterisks (*) are used to denote tests where there were insufficient data to develop confidence intervals. Direction of trend is shown in the Trend column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	y-intercept	Slope	Kendall's tau	Lower CL	Upper CL	p-value	Trend
Trichoptera (continued)	Hydropsychidae	RIT01	0.00	0.00	0.16	0.00	0.22	0.16	↔
	Hydropsychidae	RIT02	−74167.52	36.83	0.64	0.00	105.66	0.04	↑
	Hydroptilidae	CAP01	−4563.27	2.27	0.28	−33.60	20.59	0.57	↔
	Hydroptilidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Hydroptilidae	RIT01	−503.38	0.25	0.48	0.00	0.53	0.14	↔
	Hydroptilidae	RIT02	0.00	0.00	−0.09	−3.70	0.67	0.89	↔
	Limnephilidae	CAP01	0.00	0.00	0.35	0.00	0.11	0.56	↔
	Limnephilidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Limnephilidae	RIT01	0.00	0.00	−0.34	−0.33	0.00	0.37	↔
	Limnephilidae	RIT02	−2823.72	1.40	0.48	0.00	2.76	0.14	↔
	Philopotamidae	CAP01	895.29	0.44	0.60	−1.87	77.73	0.16	↔
	Philopotamidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Philopotamidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Philopotamidae	RIT02	−14780.33	7.33	0.71	0.00	67.73	0.03	↑
	Polycentropodidae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	↔
	Polycentropodidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	↔
	Polycentropodidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Polycentropodidae	RIT02	n/a	n/a	n/a	n/a	n/a	n/a	–

Table C1 (continued). Theil-Sen estimator results of post-fire trend for family taxa at each monitoring site. Strength of trend is determined by Kendall's tau value where values approaching −1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trend is shown where p ≤ 0.05. Asterisks (*) are used to denote tests where there were insufficient data to develop confidence intervals. Direction of trend is shown in the Trend column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	y-intercept	Slope	Kendall's tau	Lower CL	Upper CL	p-value	Trend
Trichoptera (continued)	Rhyacophilidae	CAP01	0.00	0.00	0.58	0.00	0.40	0.24	↔
	Rhyacophilidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Rhyacophilidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Rhyacophilidae	RIT02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Uenoidae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Uenoidae	CAP02	0.00	0.00	−0.32	−0.40	0.00	0.72	↔
	Uenoidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Uenoidae	RIT02	n/a	n/a	n/a	n/a	n/a	n/a	–
Diptera	Ceratopogonidae	CAP01	−9675.73	4.80	0.55	−15.52	46.19	0.18	↔
	Ceratopogonidae	CAP02	−8867.60	4.40	0.20	*	*	0.82	↔
	Ceratopogonidae	RIT01	0.00	0.00	−0.18	−3.47	0.89	0.67	↔
	Ceratopogonidae	RIT02	−4252.94	2.11	0.81	0.00	5.35	0.02	↑
	Chironomidae	CAP01	−53591.98	26.63	0.60	−43.04	403.09	0.14	↔
	Chironomidae	CAP02	−45952.40	22.80	0.20	*	*	0.82	↔
	Chironomidae	RIT01	39021.27	−19.33	−0.29	−30.00	7.20	0.40	↔
	Chironomidae	RIT02	−203455.10	101.04	0.57	−120.00	262.47	0.06	↔
	Culicidae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Culicidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–

Table C1 (continued). Theil-Sen estimator results of post-fire trend for family taxa at each monitoring site. Strength of trend is determined by Kendall's tau value where values approaching −1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trend is shown where p ≤ 0.05. Asterisks (*) are used to denote tests where there were insufficient data to develop confidence intervals. Direction of trend is shown in the Trend column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	y-intercept	Slope	Kendall's tau	Lower CL	Upper CL	p-value	Trend
Diptera (continued)	Culicidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Culicidae	RIT02	0.00	0.00	−0.50	0.00	0.00	0.19	↔
	Dixidae	CAP01	−918.95	0.46	0.41	−3.33	3.33	0.34	↔
	Dixidae	CAP02	2994.28	−1.48	−0.74	−6.27	0.80	0.13	↔
	Dixidae	RIT01	0.67	0.00	−0.08	−1.00	0.75	0.89	↔
	Dixidae	RIT02	−3350.85	1.66	0.72	0.00	9.48	0.02	↑
	Dolichopodidae	CAP01	0.00	0.00	−0.26	−3.20	0.40	0.65	↔
	Dolichopodidae	CAP02	0.00	0.00	0.63	0.00	0.40	0.29	↔
	Dolichopodidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Dolichopodidae	RIT02	0.00	0.00	0.36	0.00	1.28	0.32	↔
	Empididae	CAP01	−3430.96	1.71	0.33	−20.67	24.08	0.47	↔
	Empididae	CAP02	2426.30	−1.20	−0.20	*	*	0.82	↔
	Empididae	RIT01	0.00	0.00	0.00	*	*	1.00	↔
	Empididae	RIT02	−10130.21	5.03	0.64	0.00	13.97	0.04	↑
	Ephydridae	CAP01	−403.30	0.20	0.78	0.00	0.80	0.07	↔
	Ephydridae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Ephydridae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Ephydridae	RIT02	0.00	0.00	−0.50	0.00	0.00	0.19	↔

Table C1 (continued). Theil-Sen estimator results of post-fire trend for family taxa at each monitoring site. Strength of trend is determined by Kendall's tau value where values approaching −1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trend is shown where p ≤ 0.05. Asterisks (*) are used to denote tests where there were insufficient data to develop confidence intervals. Direction of trend is shown in the Trend column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	y-intercept	Slope	Kendall's tau	Lower CL	Upper CL	p-value	Trend
Diptera (continued)	Muscidae	CAP01	79.49	−0.04	−0.07	−4.00	2.93	1.00	↔
	Muscidae	CAP02	n/a	n/a	n/a	n/a	n/a	n/a	–
	Muscidae	RIT01	0.00	0.00	−0.18	−3.20	0.44	0.67	↔
	Muscidae	RIT02	116.24	−0.06	−0.11	−1.33	1.07	0.80	↔
	Psychodidae	CAP01	−94622.21	46.93	0.83	0.00	840.37	0.04	↑
	Psychodidae	CAP02	−1210.20	0.60	0.84	0.00	2.40	0.10	↔
	Psychodidae	RIT01	1.33	0.00	−0.16	−0.89	0.89	0.69	↔
	Psychodidae	RIT02	−82266.52	40.85	0.72	0.00	114.67	0.02	↑
	Simullidae	CAP01	−429372.50	213.36	0.47	−893.15	4161.33	0.27	↔
	Simullidae	CAP02	−24916.40	12.40	0.00	*	*	1.00	↔
	Simullidae	RIT01	1697.58	−0.83	−0.21	−32.93	43.57	0.55	↔
	Simullidae	RIT02	−540739.20	268.81	0.57	−154.72	774.60	0.06	↔
	Stratiomyidae	CAP01	5.60	0.00	0.00	*	*	1.00	↔
	Stratiomyidae	CAP02	2421.68	−1.20	−0.53	−30.29	1.28	0.31	↔
	Stratiomyidae	RIT01	−452.99	0.23	0.32	−0.50	1.00	0.35	↔
	Stratiomyidae	RIT02	−3048.52	1.51	0.73	0.00	0.32	0.02	↑
	Tabanidae	CAP01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Tabanidae	CAP02	0.00	0.00	0.63	0.00	0.53	0.29	↔

Table C1 (continued). Theil-Sen estimator results of post-fire trend for family taxa at each monitoring site. Strength of trend is determined by Kendall's tau value where values approaching −1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trend is shown where p ≤ 0.05. Asterisks (*) are used to denote tests where there were insufficient data to develop confidence intervals. Direction of trend is shown in the Trend column where ↑ = increasing and ↓ = decreasing trend.

Order	Family	Site	y-intercept	Slope	Kendall's tau	Lower CL	Upper CL	p-value	Trend
Diptera (continued)	Tabanidae	RIT01	n/a	n/a	n/a	n/a	n/a	n/a	–
	Tabanidae	RIT02	−3627.90	1.80	0.80	0.00	10.00	0.02	↑
	Tipulidae	CAP01	−1610.93	0.80	0.28	−28.00	16.69	0.57	↔
	Tipulidae	CAP02	−3086.55	1.53	1.00	*	*	0.02	↑
	Tipulidae	RIT01	77.18	−0.04	−0.24	−0.80	0.27	0.51	↔
	Tipulidae	RIT02	−8287.74	4.11	0.84	0.61	7.27	0.01	↑

The Department of the Interior protects and manages the nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

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