



Long-term Trend in the Aquatic Macroinvertebrate Community of the Mancos River at Mesa Verde National Park

Aquatic Macroinvertebrate Monitoring 2008–2019



Mancos River, Mesa Verde National Park (February 2022)

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Long-term trend in the aquatic macroinvertebrate community of the Mancos River at Mesa Verde National Park: Aquatic macroinvertebrate monitoring 2008–2019

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Contents

	Page
Figures.....	v
Tables.....	vi
Abstract.....	vii
Introduction.....	1
Methods.....	4
Study Area.....	4
Field Methods.....	8
Biological Data Collection.....	9
Physical Habitat Data Collection.....	9
Data Analysis.....	10
Richness and Diversity.....	10
Community Analyses.....	10
Trend Analyses.....	11
Colorado Multi-metric Index (MMI).....	11
Results.....	14
Water Quality.....	14
Stream Temperature.....	14
NPS Core Parameters.....	14
Physical Habitat.....	17
Aquatic Macroinvertebrate Community.....	21
Community Structure.....	21
Richness, and Diversity.....	22
Community Metrics.....	24
Colorado Multi-metric Index for Biotypel Streams.....	27
Discussion.....	29
Stream Temperature and Discharge.....	29
Trichopteran Response to Multiple Stressors.....	32
Ecological Integrity.....	34

Contents (continued)

	Page
Management Implications/Concerns	35
Literature Cited	37
Appendix A. Aquatic Macroinvertebrate Metrics.....	44
Appendix B. Trend Analysis Results	47

Figures

	Page
Figure 1. Map of the Mancos River Watershed, Colorado, showing the locations of MAN01 and MAN02 long-term monitoring sites (2008–2019).....	5
Figure 2. Map of the Mancos River through Mesa Verde National Park, Colorado, showing the locations of MAN01 and MAN02 long-term monitoring sites (2008–2019).....	6
Figure 3. Photo of the Mancos River at the MAN01 monitoring site in 2016 (NPS).....	7
Figure 4. Drying at riffle along the Mancos River at the MAN02 monitoring site in 2018 (NPS).....	8
Figure 5. Daily stream temperature values, derived from continuous temperature monitoring, for the MAN01 monitoring site (2008–2019).....	14
Figure 6. Stream flow velocity trend over time at physical habitat transects along MAN01.	19
Figure 7. Canopy cover trend at MAN01 and MAN02 monitoring sites.....	20
Figure 8. Nonmetric multidimensional scaling (NMDS) ordination of sample units (a) and taxa (b).	22
Figure 9. Boxplots showing the interquartile range of estimated sample richness for each year at MAN01 (a, or top) and MAN02 (b, or bottom).	23
Figure 10. Boxplots showing the interquartile range of estimated sample diversity for each year at MAN01 (a, or top) and MAN02 (b, or bottom).	24
Figure 11. Trend in relative abundance of moderately tolerant (a, or top) and intolerant taxa (b, or bottom).....	25
Figure 12. Trend in relative abundance (a, or top) and percent richness (b, or bottom) of Trichopterans at MAN01.	26
Figure 13. State of Colorado multi-metric index results for samples collected at MAN01 (a, or top) & MAN02 (b, or bottom) (2008–2019).	27
Figure 14. Stream discharge for 2010–2015 from decommissioned USGS Anita’s Flat stream monitoring gage located at MAN01 monitoring site.....	30

Tables

	Page
Table 1. Aquatic macroinvertebrate sampling on the Mancos River in Mesa Verde National Park.	8
Table 2. List of metrics used for the state of Colorado’s multi-metric index for Biotype 1.	12
Table 3. Results of trend analyses for NPS core parameter water quality data for the MAN01 and MAN02 monitoring sites.	16
Table 4. Average median values and ranges for NPS core water quality parameters for MAN01 (n=10) and MAN02 (n=7) for 2008–2019.	17
Table 5. Physical habitat trend analysis results.	18
Table 6. Aquatic macroinvertebrate metrics used in trend analyses.	44
Table 7. Trend analyses for chosen aquatic macroinvertebrate metrics for the MAN01 monitoring site.	47
Table 8. Trend analyses for chosen aquatic macroinvertebrate metrics for the MAN02 monitoring site.	49

Abstract

The Southern Colorado Plateau Inventory and Monitoring Network monitors status and long-term trends in the aquatic macroinvertebrate community and associated habitat in order to track changes in the ecological integrity of stream ecosystems. This report investigates changes in key aquatic macroinvertebrate community indices and physical habitat metrics collected from two long-term monitoring sites on the Mancos River in Mesa Verde National Park, Colorado. Results suggest that both monitoring sites have experienced periods of thermal and hydrologic stress. These stresses occur concurrently and are reflected in key metrics describing the aquatic community of the Mancos River. Daily stream temperature maxima were in exceedance of the State of Colorado's aquatic life use water quality standard during the summer and fall months (June–September) across several years (2010–2014). These thermal exceedances match timing of low stream flows on the river. Sensitive taxa such as Trichoptera and taxa assigned as intolerant to perturbation exhibited significant negative trends in abundance, as did Trichoptera richness. These declines are likely attributed to thermal and hydrologic stress. Data analyzed using the state of Colorado's multi-metric index indicate that both monitoring sites showed signs of impairment over the course of this study (2008–2019). Multiple index metrics contributed to impairment and support trend analyses demonstrating that declines in sensitive intolerant taxa, Ephemeroptera, Plecoptera and Trichoptera taxa (EPT) and clingers (trichopteran habit) contributed to the overall impairment of monitoring sites. As the effects of climate change increase over the coming decades, future management action on the river should include monitoring of thermal and hydrologic stress to better understand community responses to those stresses as well as their effects on the outcome of future management actions.

Introduction

The Southern Colorado Plateau Inventory and Monitoring Network (SCPN) identified aquatic macroinvertebrates as a high-priority vital sign for monitoring the ecological integrity of streams in parks (Thomas et al. 2006). The aquatic macroinvertebrate community was chosen as a vital sign due to their sensitivity to change in the aquatic environment, and their use to detect physical and biological perturbations in rivers and streams (Brasher et al. 2011). In 2006 SCPN established two monitoring sites on the Mancos River in Mesa Verde National Park (MEVE). These sites were chosen as part of a pilot project to test field methods for the network's monitoring protocol. Concerns by park staff regarding the effects of past wildfires as well as water extraction practices upstream made the aquatic community of the Mancos River a suitable candidate for monitoring ecological integrity. The Mancos River watershed is an important resource in southwestern Colorado. Stacey (2007) reports that Native American tribes have been in the watershed for "many thousands of years" farming, hunting, and fishing much of the watershed. European settlement of the area started in 1876 and led to intense modification of flows for irrigation and water storage (Stacey 2007). Today there are nearly 50 diversion ditches within the watershed and two large storage reservoirs. These extraction practices have greatly modified the hydrologic regime of the river within Mesa Verde National Park. These modifications have undoubtedly affected and likely altered the aquatic community over time.

The aquatic macroinvertebrate community serves as an important indicator of the integrity and health of freshwater ecosystems. As a group, their relatively short life spans, and confinement to specific habitat types allow them to reflect both current and recent conditions in the system (Graber et al. 2017, Brasher et al. 2011). As such, tracking the status and long-term trends using metrics that describe the aquatic macroinvertebrate community can alert managers to stresses occurring within freshwater ecosystems. Aquatic community responses to natural and anthropogenic disturbances such as metal pollutants (Cadmus et al. 2018, Malaj et al. 2012, Clements 1994), sedimentation (Wagenhoff et al. 2012, Kaller & Heartman 2004, Angradi 1999), streamflow reductions (Herbst et al. 2019, Growns & Growns 2001, Rader & Belish 1999) and thermal fluctuations (Cheney et al. 2019, Haidekker & Herring 2008) are well documented and often result in series of negative effects for taxa. Sedimentation, for example, has been shown to lead to decreased densities, biomass, and richness in aquatic macroinvertebrate communities (Agrandi 1999) as well as leading to decreases in the richness and density in sensitive EPT taxa, *Ephemeroptera* (mayflies), *Plecoptera* (stoneflies), and *Trichoptera* (caddisflies) (Wagenhoff et al. 2012). Similarly, increased concentrations of nutrients and heavy metals are often reflected in aquatic macroinvertebrate community structure (Cadmus et al. 2018). These changes can occur across taxonomic and functional groups as well as affecting individual taxa. Clements (1994) demonstrated a pattern of decreased abundance and richness in sensitive ephemeropteran taxa due to increasing concentrations of zinc in the Arkansas River and found that additional metal inputs from mining operations were responsible for structuring the distribution of plecopteran and coleopteran (beetle) communities. Cadmus et al. (2018) showed that ferric (oxidized) iron (Fe), a mining effluent, resulted in negative direct and indirect effects on EPT taxa. When subjected to an increasing gradient of ferric Fe, EPT taxa abundance and richness

significantly decreased after 10 days of exposure. Furthermore, ephemeropteran grazers, taxa feeding on periphyton and bacteria, suffered more significant decreases in abundance than non-grazing *Ephemeroptera* because ferric oxides decreased food resources by inhibiting algal growth (Cadmus et al. 2017). In addition to these known anthropogenic stresses, freshwater ecosystems are now facing additional threats from a warming climate.

Climate change induced drought and climate warming are expected to be primary factors affecting aquatic communities in rivers and streams (lotic systems) across the United States for the next 20-50 years (Dai 2010). Rivers and streams of the southwestern United States are currently experiencing the effects of climate change. The first two decades of this century (2000-2020) have been the warmest in 120 years of recorded temperatures, with temperatures in the southwest approximately 1 °C higher than historic averages (World Meteorological Organization 2016, Garfin et al. 2014). The aquatic macroinvertebrate community will reflect changes in the physical system due to a warming climate (Bonacina et al. 2022, Cheney et al. 2019, Herbst et al. 2018). Climate change induced drought is projected to decrease discharge in lotic systems leading to the reduction of aquatic macroinvertebrate taxa by an estimated 30–40% and abundances by 42% (Kakouei et al. 2018, Pyne & Poff 2017). Streams and rivers in the intermountain west are reliant on precipitation events to maintain natural flow regimes (Poff et al. 2002). The potential loss of late spring runoff and summer monsoonal flows associated with climate change (Garfin et al. 2014) will likely result in structural and functional changes in the aquatic community. Herbst et al. (2019) demonstrated a 40% reduction in EPT densities due to prolonged drought in headwater streams of the Sierra Nevada range in California. Increasing periods of drought can lead to changes in community structure that promote the persistence of short-lived, strong dispersing taxa, taxa less sensitive to decreases in dissolved oxygen, and taxa with more generalist feeding requirements (Kakouei et al. 2018, Bogan et al. 2015, Verdonschot & van den Hoorn 2010) thus decreasing diversity and functional redundancy in the system. Furthermore, water abstraction, a common disturbance to streams in the rural west, can exacerbate these effects leading to greater declines in abundances and in some cases extirpation of fauna from lotic communities (Salmaso et al. 2021, Pyne & Poff 2017, Rader & Belish 1999). In addition to the adverse effects of decreasing discharge, climate change will increase water temperature in lotic ecosystems. Mohseni et al. (1999) suggest that climate change could increase average stream temperatures in the US by 2–5°C and that average weekly temperature maxima and minima in streams in the Rocky Mountains could increase by 1–2 °C and 1–3°C, respectively. The responses of the aquatic community to chronic thermal stress are well documented by Bonacina et al. (2022) and the references within. These deleterious responses include genetically expressed decreases in metabolic processes, decreases in body size & growth rates as well as shifts in phenology that affect hatching and emergence cycles of aquatic insects. Additionally, thermal stress can lead to mortality and prolonged decreases in sensitive taxa. Voelz et al. (1994) found that prolonged summer maxima increase of 4–5 °C resulted in the near elimination of trichopteran taxa from the Colorado River. In addition to the immediate reductions in trichopteran abundances, some taxa exhibited a prolonged response which they attributed to reduced hatching success. Monitoring physical attributes such as temperature and investigating trends in metrics associated with aquatic

macroinvertebrate community structure and function allow us to interpret the effects of climate change on the aquatic community of the Mancos River.

To determine long-term trend in the aquatic community SCPN utilizes a combination of metrics and indices designed to detect change in community structure, sensitivity to disturbance, ecosystem function, and habit, or mode of existence (Brasher et al 2011, CDPHE 2003). The purpose of this report is to 1) understand the long-term trend in metrics describing those important ecological attributes, 2) contextualize monitoring data within the framework of the state of Colorado's multi-metric index to determine if the aquatic community is displaying signals of impairment, and 3) determine if the community is experiencing signs of stress that can help inform management of the Mancos River within Mesa Verde National Park. This report analyzes data collected from two monitoring sites on the Mancos River between 2008–2019.

Methods

Study Area

Mesa Verde National Park (MEVE) occupies 52,485 acres of shrub-steppe, pinyon-juniper woodland, Douglas fir and Ponderosa pine woodland located in southwestern Colorado. Within the park, several aquatic habitats such as seeps, springs, and riverine habitats can be found. The largest aquatic resource within MEVE is the Mancos River. The Mancos River watershed is approximately 2,072 sq km (800 sq mi.) in size. The river flows from the western side of the La Plata mountains and runs for approximately 187 km before its confluence with the San Juan River in New Mexico (San Juan National Forest). Only 6 km of the river flows within MEVE along the eastern park boundary through a deep canyon bracketed by Weber Mountain to the east and Mesa Verde National Park to the west. There are several diversion points which remove water for human use within the Mancos Watershed including two reservoirs (Jackson Gulch & Weber Reservoir) and over 47 diversion ditches (Figure 1). These diversions result in reduced flows once the river enters the park. Site characteristics of the entire watershed including the reach within MEVE are well documented in Stacey (2007). Three vegetation communities exist along the river within MEVE: Coyote Willow/Mesic Forb shrubland, located along the banks of the river; Narrowleaf Cottonwood-Rocky Mountain Juniper Woodland, occurring adjacent to the river on the floodplains; and the Lanceleaf Cottonwood Riparian Woodland, located adjacent to the river in lowly areas (Thomas et al. 2006).

A pilot project designed to test SCPN annual aquatic macroinvertebrate monitoring protocols occurred from 2005–2007 and led to the establishment of two long-term monitoring sites on the river (Figure 2) (Brasher et al. 2011). *Mancos River at Anita's Flat* (MEVEMAN01), identified in this report as MAN01, is located upstream near the northern park boundary (Figure 2). MAN01 is co-located with a now decommissioned USGS streamflow gauging station (USGS 09370600). It is characterized by a wide channel with bed substrates consisting primarily of cobbles, gravel, and fine sediments.

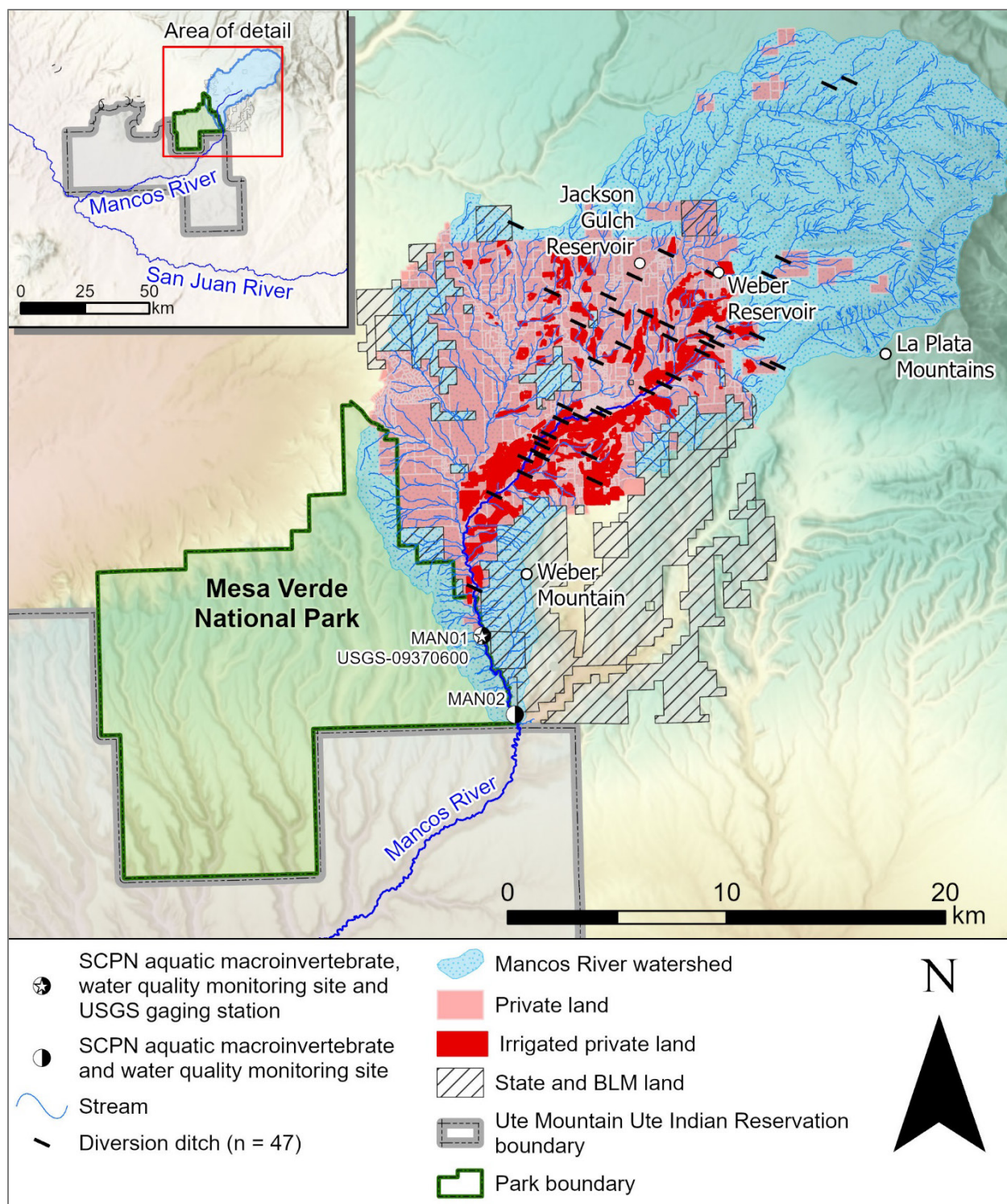


Figure 1. Map of the Mancos River Watershed, Colorado, showing the locations of MAN01 and MAN02 long-term monitoring sites (2008–2019).

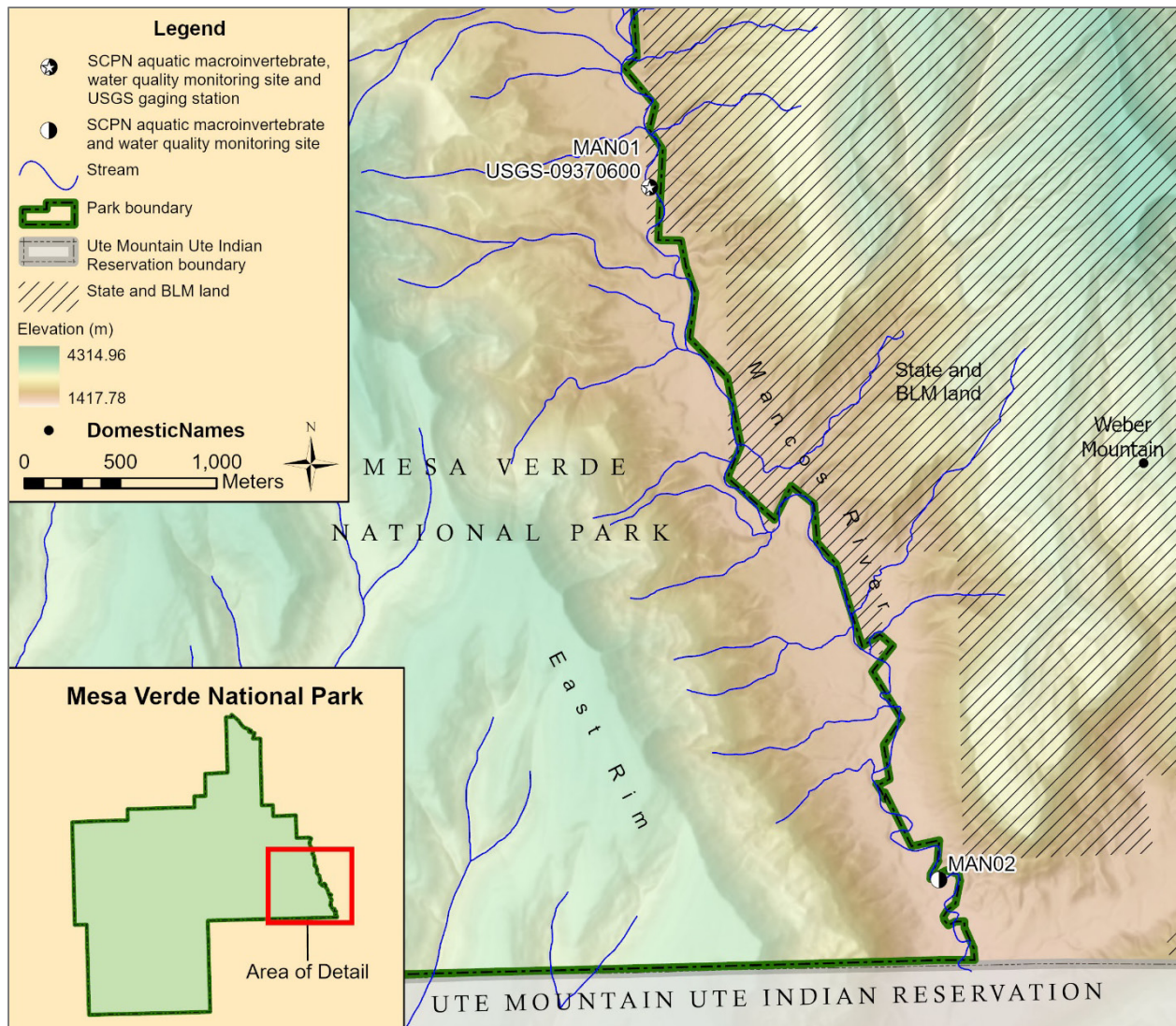


Figure 2. Map of the Mancos River through Mesa Verde National Park, Colorado, showing the locations of MAN01 and MAN02 long-term monitoring sites (2008–2019).

Coyote willow (*Salix exigua*) line the channel along the entirety of the MAN01 reach (Figure 3). Our second monitoring reach, *Mancos River at Southern Boundary* (MEVEMAN02), identified in this report as MAN02, was established in 2007 near the downstream boundary of the park. This site was chosen to allow spatial variability along the river while meeting habitat criteria (e.g., no inflows within 150 m, appropriate substrate, and consistent streamflow) and accessibility for field crews. Bed substrates along this reach are a mix of cobbles and fine sediments (shale) (Figure 4). Unlike MAN01, the banks of the river along MAN02 are a mix of coyote willow and woodland species.



Figure 3. Photo of the Mancos River at the MAN01 monitoring site in 2016 (NPS).



Figure 4. Drying at riffle along the Mancos River at the MAN02 monitoring site in 2018 (NPS).

Field Methods

The field methods in this paper follow those described in Brasher et al. (2011), the *Aquatic Macroinvertebrate Monitoring Protocol for the Southern Colorado Plateau Network*. Aquatic macroinvertebrate samples and physical habitat data are collected within a 150 m monitoring reach that includes 11 equally spaced physical habit cross-sectional transects. This report includes the analyses of biological and physical habitat data collected from 2008–2019 (Table 1).

Table 1. Aquatic macroinvertebrate sampling on the Mancos River in Mesa Verde National Park. Years where sampling occurred at each site are denoted with “x”. The “N/A” for 2013 represent the year where data were not collected due to a government-wide shut down. Asterisks represent years where conditions on the ground (e.g., 2009-pollution spill, 2016 & 2018-drying) prevented data collection.

Site	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
MAN01	x	x	x	x	x	N/A	x	x	x	x	x	x
MAN02	x	*	x	x	x	N/A	x	x	*	x	*	x

Biological Data Collection

Data were collected to match the timing of state regulatory agency sampling windows. As such, SCPN follows Colorado's recommendation for collecting annual aquatic macroinvertebrate samples during baseflow conditions in the fall for mountain/transitional streams (Paul et al. 2005). All monitoring sites were sampled annually during this sampling window. Annual site visits include the collection of 5 quantitative targeted habitat (riffle) samples and 1 qualitative multihabitat sample. Quantitative samples are collected exclusively at riffles to allow for consistency of habitat type across years thereby reducing sample variation. Quantitative samples were collected by placing a 0.25 m² sampling frame directly on the streambed of the chosen riffles. A 500 µm mesh kick net was placed downstream of the sampling frame. Substrates within the frame were disturbed and cleaned for two minutes.

Qualitative multihabitat samples were collected by walking the length of the sampling reach and sampling all aquatic macroinvertebrate habitat types present. The frequency of sampling a particular habitat type was dependent on the frequency of occurrence of that habitat along the sampling reach. For example, if the reach was dominated by cobbles, and undercut banks, those habitat types were sampled with greater frequency than less dominant habitat types. Qualitative samples were collected to provide an annual species list of organisms present at the site.

All organisms and organic matter collected during sampling were elutriated and stored in 95% ethyl alcohol for preservation. Samples were sent to the National Aquatic Monitoring Center at Utah State University for identification. Detailed information regarding identification techniques as well as laboratory quality control and assurance procedures are described in detail in Brasher et al. (2011).

Physical Habitat Data Collection

In combination with aquatic macroinvertebrate sampling physical habitat data were collected at three spatial scales: riffle, transect and reach levels. A collection of five substrate particles representing the distribution of bed substrate sizes within each riffle were measured for size and estimates of percent embeddedness (Brasher et al. 2011). Stream depth, flow velocity and discharge were collected using a standard wading rod and current meter. At each of the 11 physical habitat transects, measurements of stream depth and flow were taken at 5 equidistant locations within the wetted channel. Canopy cover was collected using a concave densiometer at the stream's left bank, channel center and stream right bank at each physical habitat transect (Brasher et al. 2011).

In addition to physical habitat data, SCPN collected the following NPS core water quality parameters.

- Discrete measurements of temperature, pH, dissolved oxygen, and conductance were collected using a multi-parameter sonde. The sonde is deployed at a location with consistent flow at each site and where little fouling can occur. The sonde is set up to record each parameter every 15 minutes during the length of the aquatic macroinvertebrate sampling effort (approximately 8–12 hrs.). Median values for each parameter are reported herein.

- To determine stream discharge during the sampling effort, a location with appropriate stream depth, flow, and low sinuosity was chosen. A minimum of 20 flow depth and velocity measurements were taken along each cross-section to provide an accurate assessment of channel structure and flow (Monroe et al. 2016).
- Turbidity samples were collected from each site at a location where good streamflow mixing occurred. Samples were placed on ice and stored until they could be analyzed by the field crew at the end of the day. For turbidity analyses, samples were placed in clear glass vials and the presence of suspended solids was measured using a portable turbidimeter. Median turbidity values for each monitoring effort are reported in nephelometric turbidity units (NTU).
- Continuous stream temperature data were collected using Hobo Tidbit temperature loggers at 15-minute intervals per day at the MAN01 monitoring site. SCPN monitors stream temperature to track cyclical patterns in temperature and to determine if stream temperature meets the state's warm water aquatic life use water quality standards. The state of Colorado has defined two different water quality standards describing temperature maxima exceedances. The first occurs during December–February (25.2°C), and for the months of March–November (28.6°C).

Data Analysis

Richness and Diversity

Taxonomic richness and diversity were assessed using a fixed-coverage rarefaction method and reported at the Genus level. 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 & Jost 2012). This method uses 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 & Jost 2012). Analysis methods used in this study 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%). 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 may be erroneous and have been identified in tables and figures. Diversity was calculated using the inverse of the Simpson diversity index. Simpson's diversity 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). Richness and diversity were calculated at the Genus level for individual quantitative samples for each monitoring year.

Community Analyses

All community analyses were run using the Vegan package 2.6-4 in R (Oksanen, 2020). Quantitative samples from targeted riffle habitats were used as sampling units. Taxa were resolved to the Genus

level for community analyses where possible, otherwise individuals were resolved to the Family level. The response matrix consisted of 55 taxa in 91 sampling units. Taxa that occurred in only a single sampling unit were removed from the matrix. Underlying patterns in community structure across sampling units were investigated using an NMDS (non-metric multidimensional scaling) ordination (Kruskal 1964, McCune & Mefford 2010). Data were transformed using the Hellinger standardization transformation. This standardization minimizes the effects of large differences (variation) in sample abundances by taking the square root of the sample total for all samples (Legendre & Gallagher 2001, Oksanen, 2020). Ecological distances (dissimilarity) of sampling units were determined using Sørensen (Bray-Curtis) distance measure. Sørensen (Bray-Curtis) distances were calculated as the shared abundance between sample units divided by the total abundance of all samples (Bray & Curtis 1957, Faith et al. 1987).

An analysis of similarity (ANOSIM) was used to determine if site or year contributed to differences in sample community structure. ANOSIM is a non-parametric test that determines if the natural groupings that exist in the data matrix (i.e., site, year) are statistically significant based on the average rank dissimilarity within groupings and between groupings (Clarke & Green 1988, Clarke, 1993). The magnitude of the effect of a particular grouping is reported as an R value where values close to 1 are interpreted as maximum distance between groups and values near 0 are interpreted as having an even distribution of dissimilarity within and between groups.

Trend Analyses

Physical habitat and aquatic macroinvertebrate metrics were evaluated for monotonic (directional) trends using the Mann-Kendall test (Mann 1945, Kendall 1975). The Mann-Kendall test is a rank correlation test that produces a value, tau (τ), that is used to determine if the metric of interest reliably increases or decreases through time. Tau values, scaled from -1 to 1, describe the strength of trend where values approaching 1 indicate a positive trend, and values approaching -1 indicate a negative trend. Values near 0 indicate no trend in the data. The Theil-Sen estimator was used to compute intercepts and slopes to describe trend (Theil 1950, Sen 1968). The Theil-Sen estimator is a non-parametric regression analysis that computes intercepts and slopes for each pair of samples in the dataset. The fitted Theil-Sen line represents the median intercept and slope for all pairs of data. The Theil-Sen estimator was chosen because it lacks distributional requirements, is resistant and insensitive to outliers and has power for detecting trends in skewed data (Helsel 2012).

Colorado Multi-metric Index (MMI)

To evaluate the ecological integrity of the aquatic macroinvertebrate community in the Mancos River, the state of Colorado's Multi-Metric Index (MMI) Bioassessment Tool (CDPHE 2003) was applied to sample data. The MMI is a tool designed to detect ecological stress in the system that could potentially alter the biological community. It is not designed to identify the specific stress, however metrics used in the MMI are designed to respond to a wide range of ecological perturbations that can affect community attributes. These attributes include community structure, function, habit, and sensitivity to disturbance. During development of the MMI the state classified streams and rivers as belonging to one of three Biotypes (1-transitional, 2-mountains, and 3-plains) based on aquatic macroinvertebrate community composition similarities resulting from ecological gradient similarities

found across the state (CDPHE 2003). The stretch of the Mancos River that flows within MEVE is classified as Biotype 1- transitional. Unlike the state’s water quality standards, which are designed to describe a waterbody’s ability to maintain species with particular thermal ranges (i.e., coldwater, warmwater) MMI metrics are designed to reflect the expected aquatic macroinvertebrate community based on the physical attributes of the waterbody within the Biotype. Metrics used to determine the index value describe community richness, composition, function, habit, and tolerance to disturbance (Table 2, Appendix A). Further information regarding metric calibration and analytical methods for the MMI can be found in CDPHE (2003).

Table 2. List of metrics used for the state of Colorado’s multi-metric index for Biotype 1.

Metric Code	Metric	Metric Description
Taxa Richness	EPTTax	Number of EPT taxa in the sample
	NonInPct	Non-insect taxa as a percentage of all taxa (worms, mites, snails)
	TotalTax	Total number of taxa in the sample
Taxa Composition	EPTnoBPct	Percent EPT excluding Baetidae of all individuals
	ColeoPct	Percent Coleopteran (beetles) individuals of all individuals
	NonInsPT	Percent Non-Insect individuals of all individuals
Habit	CIngrTax	Number of Clinger Taxa
	SprwlTax	Number of Sprawler Taxa
Pollution Tolerance	pt_Intol	Intolerant taxa as a percentage of all taxa
	pi_IncrMidElev	Percent increaser indicator individuals in Biotype 1
	IntolTax	Number of intolerant taxa
	pi_DecrMtnTrn	Percentage decreaser indicator taxa in Biotypes 1 and 2.
	pi_IncrPlains	Percent increaser indicator individuals in Biotype 3
Functional Feeding Group	PredShrTaxFAC	Number of Facultative Predator and Shredder Taxa
	PredTaxFAC	Number of Facultative Predator Taxa
	ScrapPctFAC	Percent Facultative Scraper individuals of all individuals

Metrics used for Biotype 1 streams are divided into five different community attributes: taxa richness, taxa composition, habit, pollution tolerance, and functional feeding group (Table 2). Richness metrics describe the percentage of sensitive taxa (EPTTax), percentage of tolerant non-insect taxa (NonInPct), and total number of taxa in the sample (TotalTax). Compositional metrics

describe percentage of sensitive EPT taxa excluding tolerant Ephemeropteran belonging to the Family Baetidae (EPTnoBPct); percentage of coleopterans, or beetle taxa (ColeoPct) which represents one of the most diverse and widespread groups; and the percentage of non-insect individuals in the sample (NonInsPT). Two metrics are used to describe habit, where organisms exist within the aquatic environment. The number of Clinger taxa (ClingerTax) can be viewed as a surrogate of habitat availability and the interstitial space between bed substrates. Number of sprawlers (SprwlTax) are taxa that exist on the top of fine sediments and migrate via organic material in the system. Pollution tolerance metrics emphasize the percentage and number of taxa intolerant to pollution (pt_Intol & IntolTax). Additionally, there are a number of metrics for “increaser/decreaser” taxa. The decreaser metric (pi_DecrMtnTrn) describes the percentage of taxa decrease with increasing stress in Biotype 1 and 2 streams. Increaser metrics describe taxa that are considered tolerant to stresses and increase as they occur. There are two increaser metrics for our stretch of the Mancos, one that tracks the percent of increaser individuals indicated as increasers for Biotype 1 streams (pi_IncrMidElev) as well as the percent of individuals that increase with stress for identified in Biotype 3 streams (pi_IncrPlains). The final group of metrics describe ecosystem function through use of functional feeding groups. Two metrics describe the number of facultative predators, PredTaxFAC & PredShrTaxFAC, with the latter including taxa that are facultative shredders as well as predators. The percent of individual that are facultative scrapers tracks the presence of individuals that feed on algae and periphyton in the system.

Overall scores for the MMI are derived by combining scores for the individual metrics listed in Table 2. Samples with MMI scores below 34 are designated as impaired (i.e., fail to meet the warm water use standard). Samples with MMI scores of 45 or higher are in attainment of the standard. Samples that fall between impairment and attainment are in the what the state defines as the “gray zone”. Gray zone samples have two additional metrics applied to them before an impairment or attainment designation can be made. For gray zone samples to meet attainment they must score <5.8 on the Hilsenhoff Biotic Index and >2.1 on the Shannon Diversity Index (CDPHE, 2003). A failure to meet either of those criteria results in an impairment designation.

Because the state of Colorado does not currently have a public facing portal for the Ecological Data Application System (EDAS), the system used for calculating individual metric scores and MMI index values, SCPN aquatic macroinvertebrate monitoring data were sent to the Colorado Department of Public Health and Environment-Water Quality Control Division. A senior data analyst harmonized identifications and entered the data into EDAS. Metric scores and MMI values were calculated for individual riffles at both monitoring sites.

Results

Water Quality

Stream Temperature

Daily maximum stream temperature exceedances for the March–November water quality standard were found for a total of 61 days (N =2632) (Figure 5). The summer months of 2012 experienced the greatest number of daily max exceedances at 34 days beginning in late June and ending in early September. The second highest year for daily maximum exceedances was 2014 when 14 days in July and 1 day in August saw daily max temperatures above 28.6 degrees C. There were 7 days in August and September of 2013 where max temperature exceedances occurred. The summer of 2010 had 5 consecutive days from June 16–20 where daily max temperatures were exceeded. There were no daily maximum exceedances found for the water quality standard for the months of December–February between 2009–2019. Data are missing for the summer months during 2011 and 2017–2019 due to equipment failure or loss.

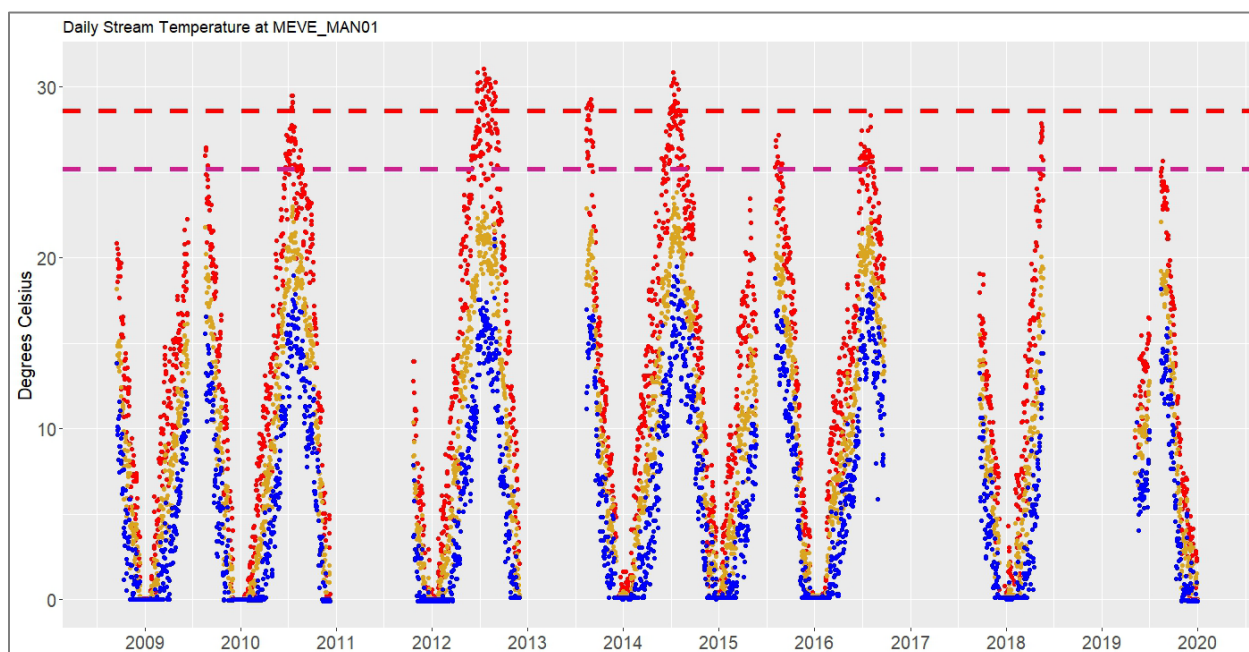


Figure 5. Daily stream temperature values, derived from continuous temperature monitoring, for the MAN01 monitoring site (2008–2019). Daily mean stream temperatures are represented in orange. Daily maximum and minimums are represented in red and blue, respectively. Red dashed line at 28.6 degrees represents the daily maximum water quality standard for March–November. The daily maximum standard for December–February is denoted by the pink dashed line at 25.2 degrees.

NPS Core Parameters

There were no significant increasing or decreasing trends found in NPS core water quality parameters at either monitoring site (MAN01, MAN02) from 2008–2019 (Table 3). Mean discharge at MAN01 was 9.00 cfs (cubic feet per second) (N=10) and 9.71 cfs at MAN02 (N=7) (Table 4). Mean pH was 8.24 at the upstream site (MAN01) compared to 8.38 at MAN02. Mean dissolved

oxygen and conductance at MAN02 were 101.04% and 1484.67 $\mu\text{S}/\text{cm}$, respectively (Table 4). At MAN01 dissolved oxygen averaged 99.94% and conductance averaged 1341.90 $\mu\text{S}/\text{cm}$ (Table 4). Mean turbidity at MAN01 was 28.26 NTU (nephelometric turbidity units) compared to 17.59 NTU at MAN02 (Table 4).

Table 3. Results of trend analyses for NPS core parameter water quality data for the MAN01 and MAN02 monitoring sites. Strength of trend was determined by Kendall's tau where values approaching -1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trends occur when $p \leq 0.05$. Slopes and intercepts were calculated using the Theil-Sen estimator. Direction of trend is shown in the Trend column where $\uparrow$ = increasing and $\downarrow$ = decreasing trend and $\leftrightarrow$ indicates no trend was found.

Site	Parameter	Intercept	Slope	Kendall's tau (τ)	Lower Confidence Limit	Upper Confidence Limit	p-value	Trend
MAN01	Discharge (ft ³ /sec)	10.89	-0.41	-0.20	-2.05	0.95	0.44	$\leftrightarrow$
	Median pH	8.56	-0.03	-0.47	-0.10	0.08	0.07	$\leftrightarrow$
	Median Temp (°C)	20.13	-0.24	-0.24	-1.66	0.48	0.38	$\leftrightarrow$
	Median Dissolved Oxygen (%)	100.56	-0.26	-0.11	-1.05	1.90	0.73	$\leftrightarrow$
	Median Conductance (μ S/cm)	1287.08	-2.17	-0.02	-117.00	128.44	1.00	$\leftrightarrow$
	Turbidity (NTU)	8.03	0.84	0.13	-1.87	4.13	0.65	$\leftrightarrow$
MAN02	Discharge (ft ³ /sec)	10.36	0.14	0.00	-3.78	2.79	1.00	$\leftrightarrow$
	Median pH	8.42	0.00	-0.08	-0.04	0.03	0.83	$\leftrightarrow$
	Median Temp (°C)	14.97	-0.23	-0.20	-1.17	0.86	0.53	$\leftrightarrow$
	Median Dissolved Oxygen (%)	106.60	-0.75	-0.20	1.80	1.06	0.53	$\leftrightarrow$
	Median Conductance (μ S/cm)	1112.25	35.63	0.31	-50.00	189.29	0.30	$\leftrightarrow$
	Turbidity (NTU)	9.03	1.30	0.28	-2.45	3.45	0.36	$\leftrightarrow$

Table 4. Average median values and ranges for NPS core water quality parameters for MAN01 (n=10) and MAN02 (n=7) for 2008–2019.

Site	Parameter	Mean	Range
MAN01	Discharge (ft ³ /sec)	9.00	(0.07–19.00)
	Median pH	8.24	(7.48–8.48)
	Median Temp (°C)	15.10	(6.86–20.17)
	Median Dissolved Oxygen (%)	99.94	(91.70–116.80)
	Median Conductance (μS/cm)	1341.90	(346.00–2674.00)
	Turbidity (NTU)	28.26	(1.66–169.00)
MAN02	Discharge (ft ³ /sec)	9.71	(0.78–18.90)
	Median pH	8.38	(8.10–8.53)
	Median Temp (°C)	14.12	(8.66–18.71)
	Median Dissolved Oxygen (%)	101.04	(91.30–106.10)
	Median Conductance (μS/cm)	1484.67	(923.00–2579.00)
	Turbidity (NTU)	17.59	(3.08–39.20)

Physical Habitat

Significant trends in physical habitat metrics were found only at the transect level (Table 5). A negative (decreasing) trend in transect level flow velocity was found at the MAN01 monitoring site (Table 5). Flow velocity at transects along MAN01 decreased by an estimated 0.02 m/sec between 2008–2019 ($\tau = -0.19$, $p = <0.01$) (Figure 6). Canopy cover was the only physical habitat metric where a positive (increasing) trend was found at both monitoring sites (Figure 7). Estimated average annual increase in canopy cover was 1.44% at the downstream site (MAN02, $\tau = 0.19$, <0.01) and 1.39% at the upstream site (MAN01, $\tau = 0.24$, <0.01).

Table 5. Physical habitat trend analysis results. Strength of trend was determined by Kendall's tau where values approaching -1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trends occur when $p \leq 0.05$. Slopes and intercepts were calculated using the Theil-Sen estimator. Direction of trend is shown in the Trend column where $\uparrow$ = increasing and $\downarrow$ = decreasing trend and $\leftrightarrow$ indicates no trend was found.

Site	Habitat Level	Metric	Intercept	Slope	Kendall's τ	Lower Confidence Limit	Upper Confidence Limit	p-value	Trend
MAN01	Riffle	Particle Size (mm)	102.00	0.00	-0.01	-2.00	1.80	0.96	$\leftrightarrow$
	Riffle	Particle Embeddedness (%)	20.00	0.00	-0.10	1.67	0.46	0.35	$\leftrightarrow$
	Riffle	Stream Flow Depth (m)	0.11	0.00	0.06	0.00	0.01	0.55	$\leftrightarrow$
	Riffle	Stream Flow Velocity (m/sec)	0.55	0.00	0.01	-0.02	0.03	0.96	$\leftrightarrow$
	Transect Level	Median Stream Depth (m)	0.17	0.00	0.06	0.00	0.01	0.40	$\leftrightarrow$
	Transect Level	Median Flow Velocity (m/sec)	0.46	-0.02	-0.19	-0.32	-0.01	<0.01	$\downarrow$
	Transect Level	Canopy Closure	8.90	1.39	0.19	0.33	1.40	<0.01	$\uparrow$
MAN02	Riffle	Particle Size (mm)	84.20	0.20	0.03	-1.22	2.00	0.78	$\leftrightarrow$
	Riffle	Particle Embeddedness (%)	35.00	-2.50	-0.32	-3.75	0.00	0.01	$\leftrightarrow$
	Riffle	Stream Flow Depth (m)	0.07	0.01	0.40	0.00	0.01	<0.01	$\leftrightarrow$
	Riffle	Stream Flow Velocity (m/sec)	0.40	0.02	0.11	-0.02	0.04	0.37	$\leftrightarrow$
	Transect Level	Median Stream Depth (m)	0.21	0.00	0.05	0.00	0.01	0.49	$\leftrightarrow$
	Transect Level	Median Flow Velocity (m/sec)	0.34	0.00	0.03	-0.01	0.01	0.69	$\leftrightarrow$
	Transect Level	Canopy Closure	15.73	1.44	0.24	0.05	2.50	<0.01	$\uparrow$

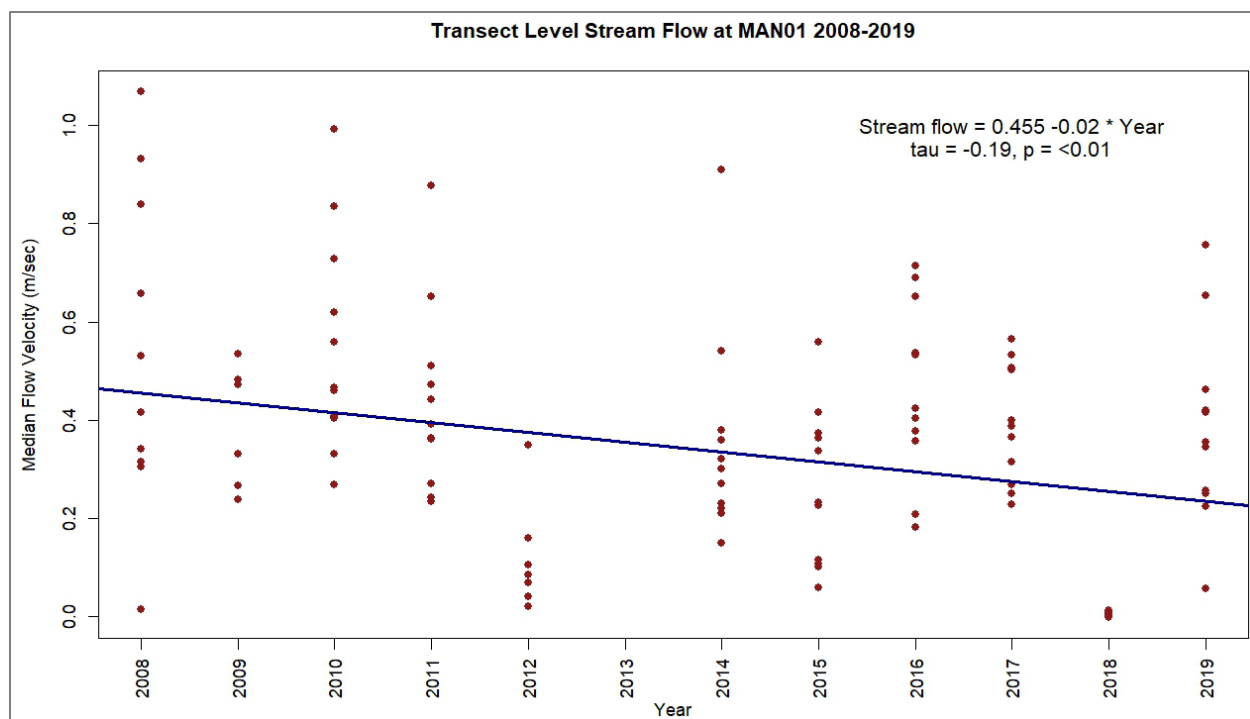


Figure 6. Stream flow velocity trend over time at physical habitat transects along MAN01. Red points represent median stream flow for each physical habitat transect collected during each monitoring year. Blue line represents the Theil-Sen estimated slope for negative trend in flow velocity at MAN01.

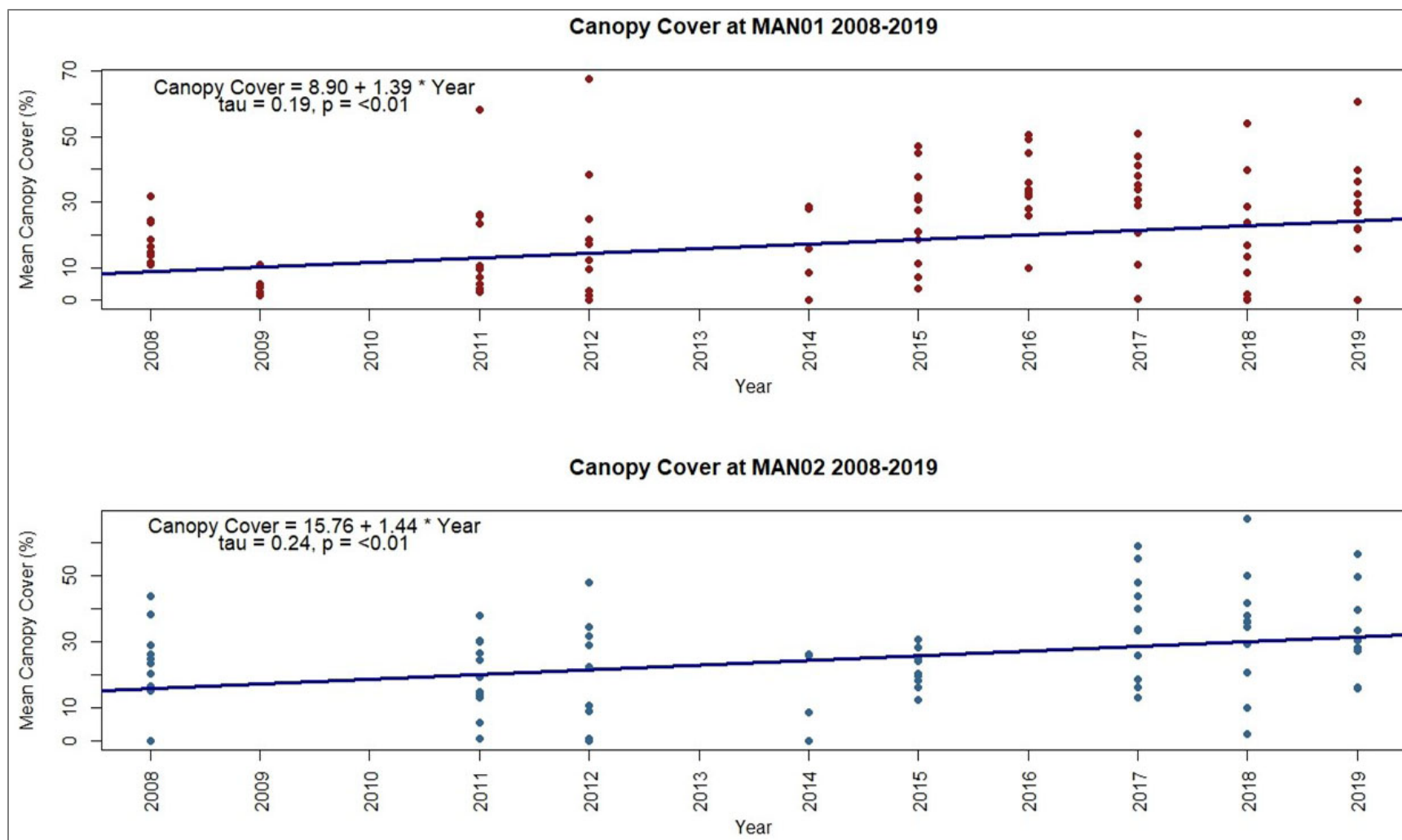


Figure 7. Canopy cover trend at MAN01 and MAN02 monitoring sites. Colored dots represent mean canopy cover values for MAN01 (a) and MAN02 (b). Blue lines in both figures represent the Theil-Sen estimated slope for a positive trend in canopy cover over time at both sites.

Aquatic Macroinvertebrate Community

Community Structure

To investigate patterns in community structure among sampling units a nonmetric multidimensional scaling (NMDS) ordination was used. This technique is used to describe similarity/dissimilarity among samples. The ordination space describes the dissimilarity between samples, represented by the distance between each sample in ordination space. The full ordination represents dissimilarity in samples collected from the Mancos River between sites (shape) across years (color). NMDS identifies how similar samples are based on the taxa represented and their abundance in each sample.

The ordination was run on a taxa response matrix of 92 samples (individual riffle samples) and 55 taxa (Figure 8 a & b). The goal of using the ordination was to: 1) determine community composition patterns among sites and 2) assess changes in communities during this period both overall, and inter-annual changes in pattern if they existed.

The resulting ordination is displayed in two dimensions with a stress of 0.22 (Figure 8 a & b). Both figures represent identical ordination space. The top figure (8a) represents the dissimilarity among individual riffle samples from each site across years. The bottom figure (8b) represents the taxa responsible for the dissimilarity in Figure 8a. If a sample or group of samples in Figure 8a coincides with the direction of a vector in Figure 8b, then those samples have a higher abundance of that taxa than other samples.

There is no clear visual partitioning (separation) among samples by site (shape) or year (color), in the ordination (Figure 8a). Figure 8a shows that the greatest dissimilarity is found along the x-axis, NMDS 1. That dissimilarity appears to be driven by differences in sensitivity across samples (Figure 8b). Samples clustering to the left of the origin and moving in a negative direction along axis 1 are dominated by mayflies (Baetis & Heptageniidae) and riffle beetles (Optioservus, Helichus, Postelichus). Samples placed in the opposing positive direction are dominated by trichopterans (Hydroptila, Hydroptilidae), and stoneflies (Taeniopterygidae, Capniidae). Samples moving along the y-axis, NMDS 2 indicate dissimilarity in taxa that are resistant to disturbance and stress and include midges (Chironomidae, Hemerodromia, Orthoclaadiinae, Empididae, Simuliidae, Neoplasta) and worms (Clitellata) in a negative direction along axis 2 and Tanypodinae (midge) and Collembolla (springtails) in a positive direction.

Ordinations serve as a visual representation of dissimilarity among samples. These figures are intended to display dissimilarity in samples across sites and years (Figure 8a) as well as displaying the taxa that are responsible for those differences (Figure 8b). While there is no intuitive partitioning of samples by site or year in Figure 8a, we tested for differences in community structure between the groupings of site and year using an Analysis of Similarity (ANOSIM). ANOSIM tests for dissimilarities in the natural groupings of these data (i.e., Site, Year) based on presence and abundance of taxa. Results indicate a significant partitioning in samples by Year ($R = 0.49$, $p < 0.01$). When grouping by Site, dissimilarity was evenly distributed within and across groups ($R = 0.09$, $p < 0.01$). These results suggest that inter annual variation influences the composition and structure of

samples and that differences between monitoring sites have little to no effect on community composition.

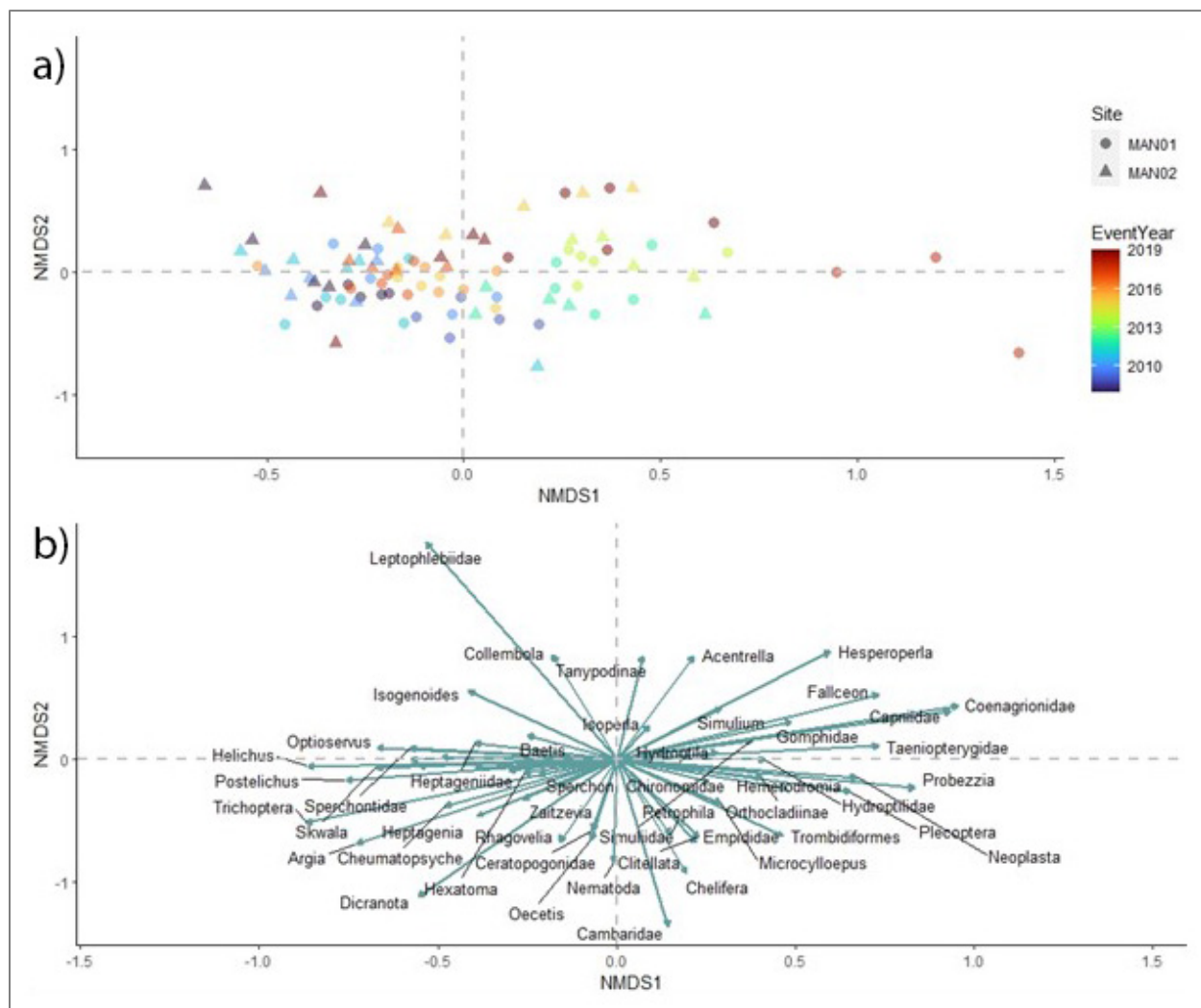


Figure 8. Nonmetric multidimensional scaling (NMDS) ordination of sample units (a) and taxa (b). Figure 8a shows individual sample units. Year is represented by color, site represented by shape. Distance between sample units represent the dissimilarity in taxa present and their abundance in the sample. Figure 8b displays identical ordination space with taxa represented as vectors. Taxa vectors point to areas where samples have the highest abundance of that taxa. Movement in the opposing direction of any vector indicates samples with little or no representation of that taxa.

Richness, and Diversity

There were no statistical trends found in richness or diversity metrics for either site for the monitoring years between 2008–2019 (Appendix B). Mean richness was 17.42 (SD = 9.51) at MAN01 and 17.52 (SD = 10.16) at MAN02. Calculations of sample richness indicate that at least one sample in the years 2010, 2014, and 2019 at MAN01 had estimated richness values that were more than double the coverage estimate for the sample (Figure 9). This occurs when the sample is found to

have a high number of taxa with one or two individuals (singlets, doublets) that inflate the estimation of richness. As such these values should be viewed with skepticism as they are likely inflated estimates of the sample. Similar inflations occurred at MAN02 for the years 2010, 2011, 2012, 2017, and 2019. Diversity was estimated using the inverse of the Simpson's Diversity index (Figure 10). Mean diversity was identical at both monitoring sites with an average of 2.98 (MAN01 SD = 1.32, MAN02 SD = 1.08)

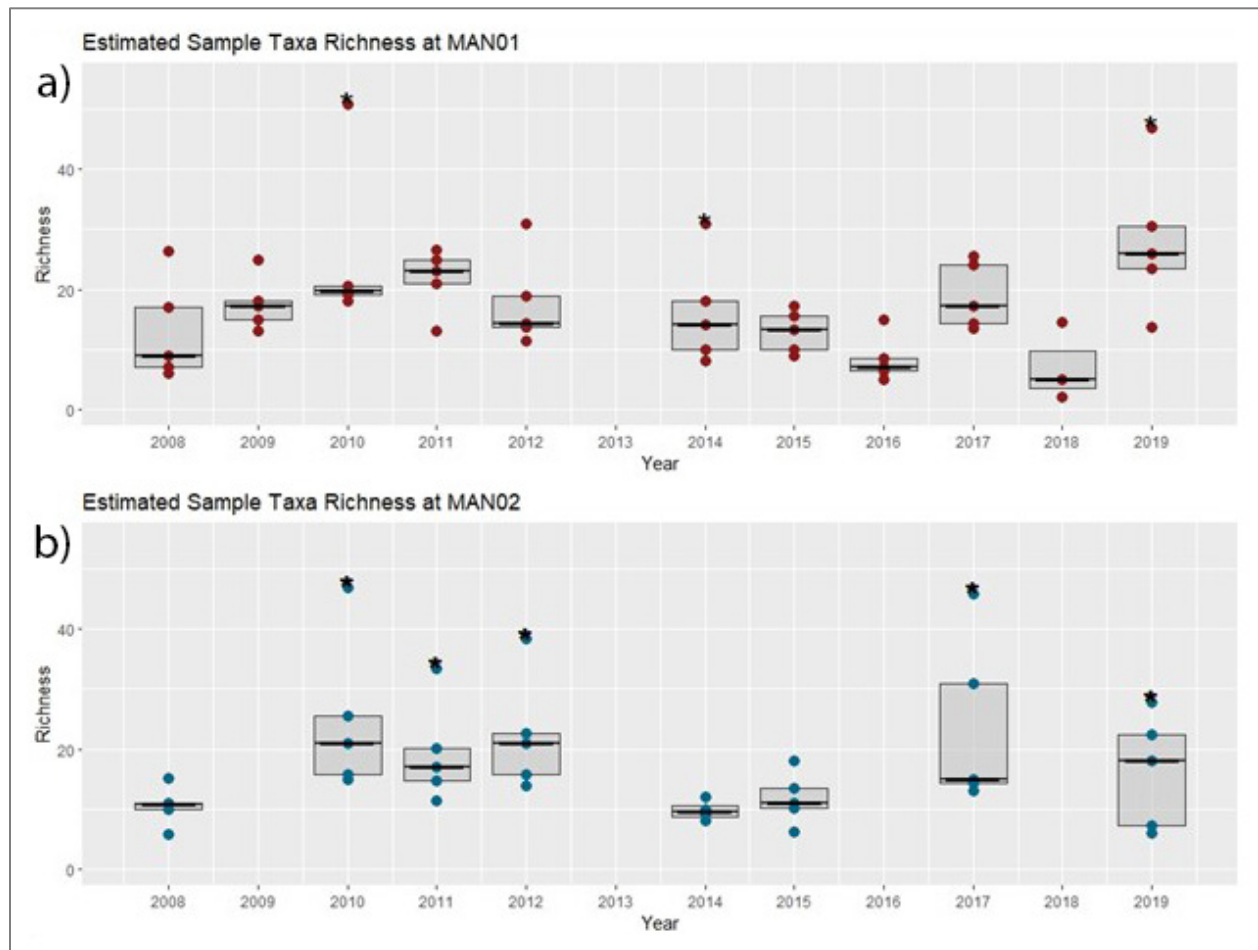


Figure 9. Boxplots showing the interquartile range of estimated sample richness for each year at MAN01 (a, or top) and MAN02 (b, or bottom). Colored dots represent estimated richness of individual samples, dark black line represents the median of all richness values for that year. Samples annotated with an * are estimates that are subjected to high bias due to over extrapolation.

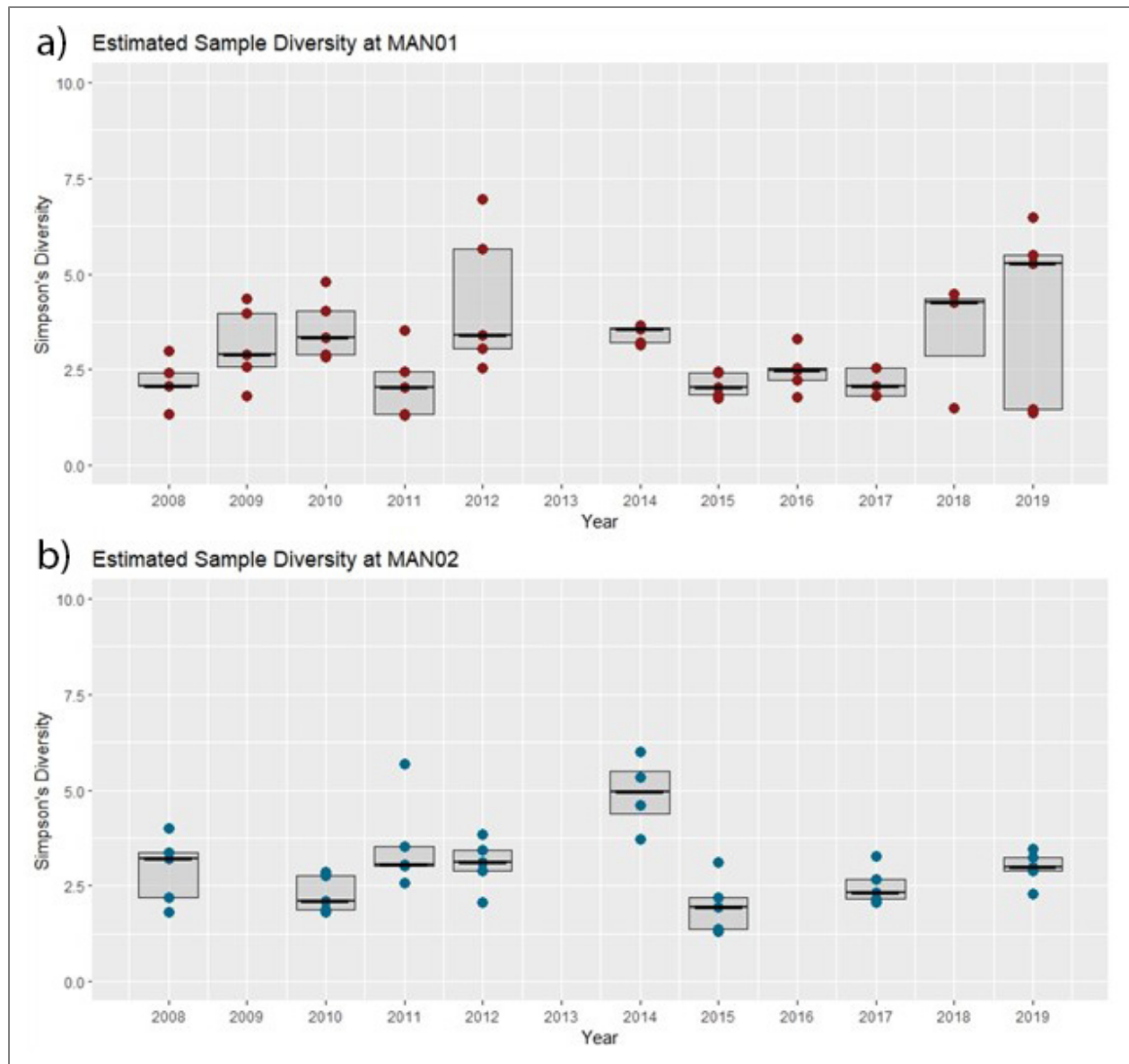


Figure 10. Boxplots showing the interquartile range of estimated sample diversity for each year at MAN01 (a, or top) and MAN02 (b, or bottom). Colored dots represent estimated diversity of individual samples, dark black line represents the median of all diversity values for that year. Diversity was calculated using the inverse of the Simpson's Diversity Index.

Community Metrics

Long-term trends were found in 6 of the 38 community metrics used to describe the ecological attributes of structure, function, sensitivity, and habit. All metrics that exhibited a long-term trend were found at the upstream MAN01 monitoring site, while no trend was found in any metric at the downstream MAN02 monitoring site (Appendix B). Four sensitivity metrics exhibited a trend at MAN01. Relative abundance of moderately tolerant taxa increased by an average of 2.60% a year ($\tau = 0.27$, $p < 0.01$) (Figure 11a) while the relative abundance of intolerant taxa decreased by an average -2.72% a year ($\tau = -0.26$, $p < 0.01$) (Figure 11b). Trichoptera, an order of aquatic

macroinvertebrates that are sensitive to stress in the aquatic environment, exhibited a negative trend in abundance and richness at MAN01 (Figure 12 a & b). Relative abundance of trichopteran taxa at MAN01 decreased by an average of -2.66% a year ($\tau = -0.26$, $p = 0.01$) (Figure 11a). Percent richness of Tricoptera saw a smaller decline averaging an annual decrease of -0.64% a year ($\tau = -0.24$, $p = 0.02$) (Figure 11b). Predator richness was the only functional group metric that exhibited a trend over time (Appendix B). Predator taxa increased in richness by 0.98% a year ($\tau = -0.24$, $p = 0.02$). Sprawlers, a taxonomic group that inhabit substrate surfaces and the surface of floating leave and hydrophytes, exhibited a negative trend in relative abundance at MAN01. Sprawler individuals decreased by an average of -0.20% ($\tau = -0.22$, $p = 0.04$) per year.

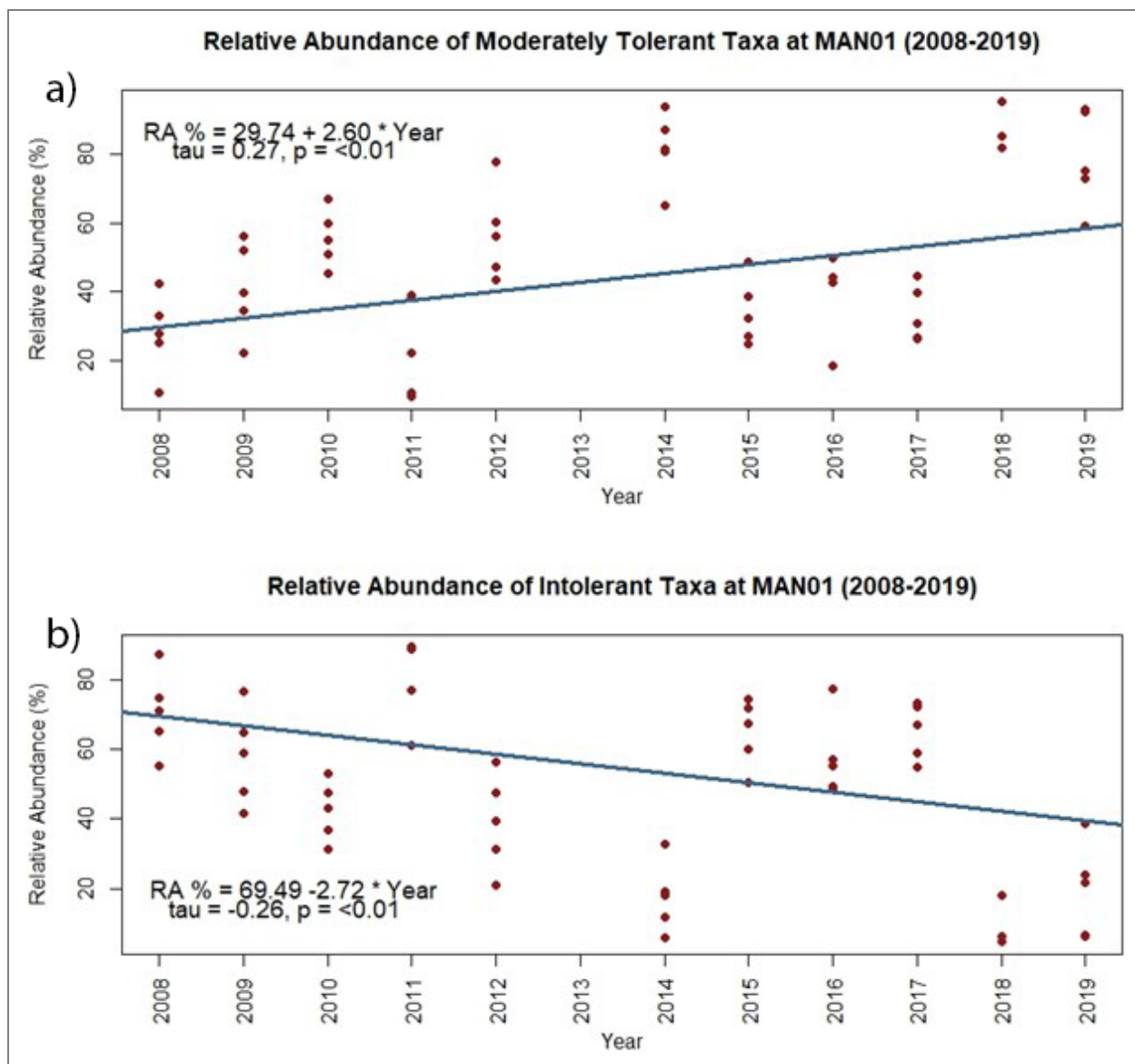


Figure 11. Trend in relative abundance of moderately tolerant (a, or top) and intolerant taxa (b, or bottom). Blue line represents the Theil-Sen estimated slope for trend in relative abundance.

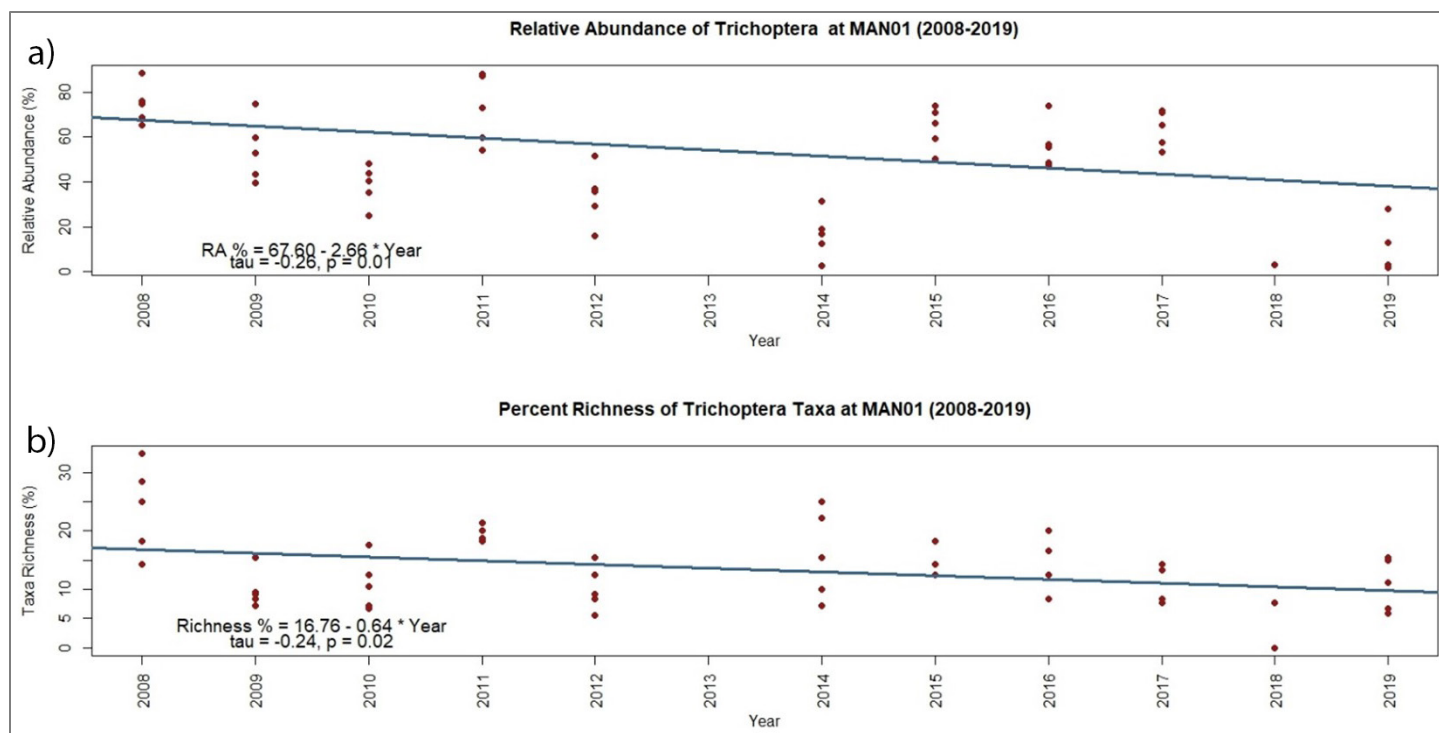


Figure 12. Trend in relative abundance (a, or top) and percent richness (b, or bottom) of Trichopterans at MAN01. Blue line represents the Theil-Sen estimated slope for trend in relative abundance.

Colorado Multi-metric Index for Biotype1 Streams

Ecological integrity of the aquatic macroinvertebrate communities at MAN01 and MAN02 was assessed using the state of Colorado's Multi-Metric Index (MMI). Results from the MMI indicate that 28.3% of samples from MAN01 ($n = 53$) and 20.5% ($n = 39$) did not meet the threshold for attainment of the state's water quality standard for warm water life use. Impairment was detected in multiple samples for the 2014, 2015, 2016, and 2018 monitoring years at MAN01 (Figure 13). For the years between 2014–2016, MMI scores were higher than the impairment threshold but not the attainment threshold (Figure 13). These samples occur in what the state has designated as “the gray zone”. For such samples, the HBI and Shannon Diversity indices are applied to the sample. If either metric is outside the defined range for attainment the sample is deemed to be impaired. For those years, all samples in the “gray zone” failed to meet the criteria for the Shannon Diversity Index (> 2.1). During 2018 at MAN01, no sample MMI score was above the threshold for impairment. At MAN02 the largest concentration of impaired samples occurred in 2015 when all samples were designated as impaired. It should be noted that samples were not collected at MAN02 during 2016 and 2018 due to drying conditions and a lack of flowing water at the monitoring site. For those years, the site should be considered in a state of impairment.

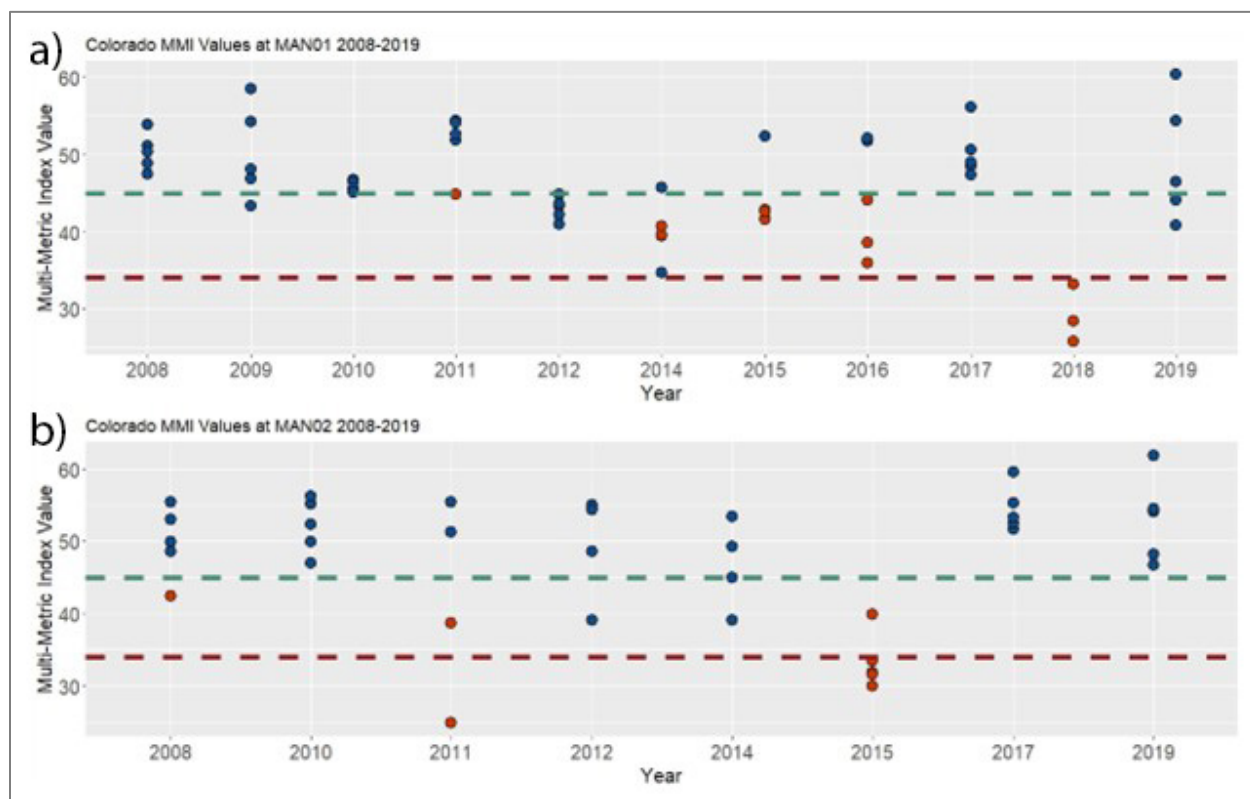


Figure 13. State of Colorado multi-metric index results for samples collected at MAN01 (a, or top) & MAN02 (b, or bottom) (2008–2019). Figure shows individual index scores for each sample by year. Blue circles represent years when site met aquatic use standard. Orange circles indicates years where site did not meet standard and is considered in an impaired state. Blue-green dashed line indicates index attainment threshold. All samples above line represent reference conditions. Red dashed line indicates impairment threshold. All samples below line represent impaired conditions.

When comparing samples that met attainment of the standard (n = 69) to those that were designated as impaired (n = 23) there were 5 metrics that appear to have the greatest contribution to impairment in MMI scores. The percentage of all intolerant taxa (pt_Intol) showed the greatest decline, decreasing by -68.13%. The number of EPT taxa (EPTTax) decreased by -35.47% and decreased further when baetid mayflies were removed (EPTnoBPct) to -42.95%. In impaired samples, facultative predators and shredders (PredShrTaxFAC) decreased by -52.40% and clinger taxa (ClngrTax) decreased by -38.91%.

Discussion

The Southern Colorado Plateau Inventory & Monitoring Network monitors the aquatic macroinvertebrate community to track the ecological integrity of stream ecosystems in parks. Long-term trends of metrics describing important ecological attributes are used to investigate possible points of stress in the aquatic community. Trend results from individual metrics and the State of Colorado's multi-metric index show declines in sensitive taxa, particularly Trichoptera and taxa classified as intolerant to disturbance. Additionally, these results demonstrate that across several years, the aquatic community at both sites were in a state of impairment and failed to meet the water quality standard for warm water streams. Physical habitat and biological data indicate that these monitoring sites are experiencing variable amounts of hydrologic and thermal stress which may be associated with a warming climate. Data from 2008–2019 provide multiple lines of evidence that suggest SCPN long-term monitoring sites on the Mancos River are experiencing stress affecting these communities. These stresses include both abiotic and biotic factors affecting this ecosystem. The most apparent source of stress was drying at the lower reach (MAN02) during 2016 and 2018. A lack of streamflow at the lower site prevented sample collection during those years. Additionally, streamflow data upstream from MAN01 indicate an overall negative trend in flow velocity at habitat transects at MAN01 from 2008–2019 (Figure 6). Further signs of stress were found in stream temperatures at MAN01. While the record is incomplete, there were 61 days where summer and fall daily maximum stream temperatures were in exceedance of the water quality standard. These exceedances extended across multiple years (Figure 5).

Evidence suggests that warming is well underway across the western United States (Udall & Bates 2007). Mote et al. (2005) reported significant warming trends across the west using air temperature data collected from 1950–1997, suggesting that the current rate of warming is 2–3 °C per century. Leppi et al. (2012) investigated August stream discharge across the Central Rocky Mountains and found that decreases in average discharge were caused or influenced by changes in air temperature over the last half century (1950–2008).

Stream Temperature and Discharge

Streamflow and temperature are the primary influences structuring aquatic macroinvertebrate communities in lotic ecosystems (Bonacina et al. 2023, Kalaninoviá et al. 2014, Sweeny 1993, Vannonte & Sweeny 1980). As the climate warms aquatic ecosystems will suffer the effects of multiple stressors acting concurrently on the system (Graeber et al. 2017, Piggott et al. 2012, Wagenhoff et al. 2012). Globally, near-term future warming is projected to increase stream temperatures by 1–2 °C and summer maxima & minima of streams in the Rocky Mountains are expected to increase by 1–2°C and 2–3 °C, respectively (Piotrowski et al. 2021, Mohseni et al. 1999). Moreover, stream discharge for some areas of the Rocky Mountains have already shown a mean reduction of 0.22% per year (Rood et al. 2005). Discharge data from the now decommissioned USGS stream gage (Mancos River at Anitas Flat Below Mancos, CO-093706000) at MAN01 show that between 2010–2015 patterns in increased stream temperature closely followed decreasing summer discharge values at MAN01 (Figure 14). Summer and fall discharge values for 2010, 2012, 2013 and

2014 indicate that flows dissipated to 1.76 cfs (cubic feet per second) in 2010 and fell below 1.00 cfs in subsequent years reaching a low of 0.02 cfs on July 14, 2014.

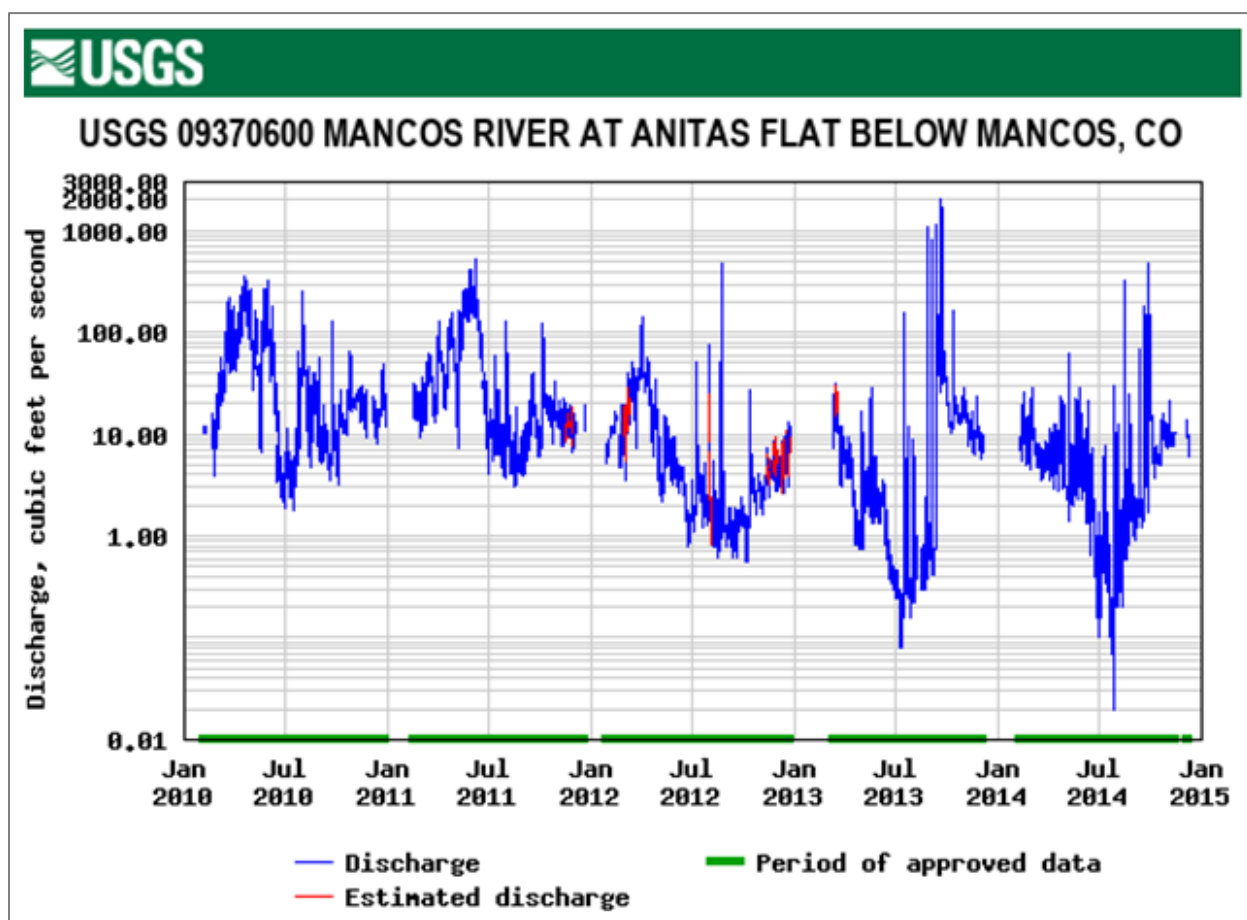


Figure 14. Stream discharge for 2010–2015 from decommissioned USGS Anita's Flat stream monitoring gage located at MAN01 monitoring site.

While stream temperature increases are expected to accompany decreased discharge patterns for lotic systems (Booker & Whitehead 2021), the mechanisms by which they affect the aquatic community will differ. Streams experiencing lowered discharge will have less power to scour streambeds and banks, and to transport gravels, fine sediments, and nutrients to downstream communities. As a result, the increased deposition of fine sediments settling out from low flows can reduce habitat availability for aquatic macroinvertebrates. Fine sediments can embed streambed substrates filling in the interstitial spaces between cobbles reducing available attachment and foraging habitat. Angradi (1999) was one of the first studies to demonstrate negative responses in aquatic macroinvertebrate densities, biomass and EPT richness due to increased sedimentation. Taxa that inhabit riffles may be particularly susceptible to sedimentation via decreased discharge (Walters & Post 2011, Kaller & Hartman 2004). In one study, Kaller & Hartman (2004) ascribed the 50% reduction of available riffle habitat to increased fine sediments during low summer flows. The significant decrease in EPT richness was attributed to this reduction in habitat. Like streambed particle transport, resource

distribution of allochthonous (outside) sources of energy (e.g., woody debris, leaf packs, organic materials) are negatively affected by reduced stream flows. Decreasing flow velocities can trap organic materials in place and prevent the distribution of this important resource downstream. Taxa that process those materials (i.e., shredders and grazers) are expected to be particularly vulnerable to the reduction of organic material availability (Pyne & Poff 2017, Verdonshot & van der Hoorn 2010). The effects from the potential loss of shredders and grazers would cascade through the aquatic ecosystem changing patterns of energy flow through the system. Since the processing and breakdown of organic material by shredder and grazer taxa provides finer organic material for other taxa further down the food chain, their loss has the potential of creating resource scarcity for those taxa (collector gatherers and collector filterers) that rely on finer organic food particles (Vannote et al. 1980).

In addition to lower summer flows, climate change will bring an increase in frequency of extreme events and expansion of the interannual variation of ecosystem processes. For streams in the southwest this will result in extended periods of drying and more extreme hydrologic events (IPCC 2023, Dettinger et al. 2015). Extremely large precipitation events that were once uncommon, occurring on multi-decadal scales, are projected to increase in frequency to every 12 years in the Southwest (Dettinger et al. 2015). Large flow events like those in the summer of 2013 (Figure 13) are likely to increase in frequency offset by longer periods of drying. This shift in the hydrologic regime will likely lead to alternating periods where nutrients and sediments are immobile and unavailable to downstream communities followed by sudden scouring events that remove and widen stream channels, flushing nutrients, sediments, and habitat from large portions of the river thereby altering community structure along different reaches of the Mancos. Poff et al. (2002) warn that this increased year to year variability could challenge the resiliency of aquatic ecosystems by inflicting novel or rare environmental conditions on aquatic biota. Shifts in community structure are likely to favor biota that are more generalist and resistant to extreme events as the projected rapidity of climate change is likely prohibitive to allowing for communities to adapt to change via “natural genetic selection” (Poff et al. 2002).

During years where discharge was at its lowest values, stream temperature maxima was high (Figures 5 and 14). Daily stream temperature maxima during low discharge years (2010, 2012, 2013, 2014) exceeded the daily temperature maxima water quality standard for aquatic life use set by the state of Colorado. This standard was designed to describe the desirable temperature limit for aquatic life in the Mancos River. There was a total of 61 days between 2010 and 2015 where that standard was exceeded. This result shows that during the warmer months the aquatic community at the MAN01 monitoring site is exposed to thermal stress. Patterns and timing of decreasing discharge suggest that reduced flows are the mechanism responsible for temperature exceedances. Under historic conditions the natural discharge regimes of lotic systems buffer stream temperatures from extreme fluctuations of air temperature. The simultaneous stresses of decreasing discharge and increasing air temperatures have a larger effect on stream temperatures than increasing air temperatures alone. A study by van Vliet et al. (2011) showed that an increase in air temperature of 4 °C combined with a 40% decrease in stream discharge resulted in higher stream temperatures than a 6 °C increase in air temperature alone. The combined effects of these stressors could pose a serious threat to the current structure and function of the aquatic community in the Mancos River.

Stream temperature is a principal organizing component of freshwater ecosystems affecting aquatic macroinvertebrate physiological responses, phenology, behavior, and ecology (Bonacina et al. 2023, Sweeney 1993, Vannote & Sweeney 1980). As streams increase in temperature, respiration of aquatic organisms increases (Bonacina et al. 2023, Sinsabaugh 1997). This poses a particularly sensitive stress for the aquatic community because the increased demand for oxygen occurs as oxygen solubility decreases in aquatic environments (Bonacina et al. 2023, Forester et al. 2012). Additionally, thermal stress can lead to reduced body size and growth rates among aquatic taxa. The increased energetic demands for metabolic processes that occurs as temperature increases, has been shown to result in a divestment in growth among some taxa (Bonacina et al. 2023). Aquatic macroinvertebrate taxa have demonstrated several phenological responses to increased stream temperatures including decreases in developmental period, hatching time, and emergence (Bonacina et al. 2023, Woods et al. 2021, Sweeney et al. 1986). These shifts have been attributed to smaller overall body sizes and decreased fitness for different taxa within the aquatic community. Behavioral responses to thermal stress include increased drift and shifts in feeding patterns. As temperatures increase beyond an optimal range, organisms, particularly EPT taxa, are more likely to leave protective bottom substrates to seek out more ideal conditions downstream (Bonacina et al. 2023).

Because increased temperature exposure is accompanied by increasing metabolic rates, organisms that are passive feeders under optimal conditions may become more active foragers as they seek higher-quality resources (Bonacina et al. 2023, Vannote & Sweeney 1980). Differential sensitivity to thermal stress among taxa within the aquatic community implies that both chronic and acute increases in summer temperature maxima will affect community structure and function. Changes in summer temperatures have been shown to be a key factor in altering aquatic community composition. Haidekker & Herring (2008) attribute a 3 °C change in mean summer temperature as the primary attribute responsible for shifts in community structure in streams. Thermal stress can lead to instantaneous reductions via mortality in aquatic macroinvertebrate abundance and richness as well as inflicting prolonged effects, such as hatching success, which can be reduced in some taxa (Voelz et al. 1994). Pyne & Poff (2017) have suggested that thermal warming could lead to the loss of 30–40% of taxa in lotic systems. Furthermore, they argue that the differential sensitivity to thermal stress could result in the loss of taxa responsible for processing allochthonous and autochthonous (inside) (e.g., aquatic vegetation, algae) sources of primary production. Such loss could drastically alter how energy moves through aquatic food webs.

Trichopteran Response to Multiple Stressors

Trichoptera are a large and diverse group of aquatic organisms that exhibit a high degree of sensitivity to changes in the aquatic environment (Morris 2009, Wiggins & Currie 2008). Trichopterans have proven to be useful bioindicators for tracking sedimentation (Blinn & Ruiter 2006), water quality (Hamid & Rawi 2017, Stuijzand et al. 1999) and metals (Vuori & Kukkonen 1996). Because of this sensitivity, trichopteran metrics are useful in monitoring the integrity of

aquatic ecosystems through time (Brasher et al. 2011, CDPHE 2003). Trend results for the trichopteran community at our MAN01 monitoring site indicates that abundance and richness show significant declines (Figures 12 a & b). This is an important result in that the steady decline of trichopteran abundance and taxa indicates the site is experiencing periods of stress. Patterns in trichopteran relative abundance decline at MAN01 follow similar patterns of temperature maxima exceedances and low summer discharge. The highest number of daily stream temperature maxima exceedances occurred in 2010, 2012 and 2014, the same years that trichopteran abundance fell below the estimated median abundance levels, i.e., the Theil-Sen slope line (Figures 5 & 12a). Similarly, patterns in abundance decline match up with years where stream discharge was measured to be its lowest (2010–2014) (Figure 14).

Temperature has been shown to be one of the most important abiotic factors in structuring trichopteran communities (Kalaninová et al. 2014). Many taxa native to North America are adapted to cooler lotic systems (Wiggins & Currie 2008). Taxa found in the Mancos River are no exception. Many trichoptera taxa are enclosure building taxa that attach themselves to the underside of substrates (clingers) in riffles and other heavily oxygenated stream reaches. As such, they are extremely limited in their ability to disperse when conditions fluctuate outside of optimal ranges rendering chronic stress to be a serious concern. The thermal and hydrologic stress demonstrated in stream temperature and discharge data are the likely cause for declining trichopteran abundance and richness. Climate change induced shifts in air temperature and discharge have been linked to alterations in patterns of trichopteran community structure in high- and mid-elevation streams (Sáinz-Barián et al. 2016). Increasing stream temperatures can lead to trichopteran community collapse and alter behavior that can increase susceptibility to mortality. Rogowski & Stewart (2016) demonstrated that signs of increased trichopteran mortality begin to occur at 22.3 °C with nearly complete mortality occurring at 25 °C. All documented daily maxima exceedances on the Mancos occurred above 28.6 °C. Voelz et al. (1994) documented the removal of Trichoptera from reaches in the Colorado River after prolonged exposure to stream temperatures that were found to be 4–5 °C above normal summer maxima. Furthermore, they noted a significant lag in trichopteran recovery that was attributed to a lack of hatching success from thermal stress suggesting that the effects of temperature can have protracted ecological consequences.

Trichopterans shift their behavior as they reach thermal limits. Dallas & Rivers-Moore (2012) found that as thermal stress approaches the “Critical Thermal Endpoint” (the temperature that results in mortality), Trichoptera leave their protective enclosures and lose their ability to cling to the underside of substrates. This is an important finding because it demonstrates that non-lethal temperatures can subject organisms to behaviors that can increase vulnerability to environmental conditions and predation. All trichopteran taxa found in the Mancos River are enclosure builders (Wiggins & Currie 2008). Once enclosures are built and attached to substrates most taxa will use their salivary glands to deploy nets that cause fine particulate organic matter to get trapped as it flows through the enclosure. These nets serve as the mechanism to obtain food resources that are free floating in the water column. Leaving enclosures and losing the ability to attach to substrates reduces the individual’s ability to forage. Case making taxa (SubOrder Integripalpia) found in the Mancos River which include the families Ryacophilidae, Hydroptilidae, and Glossosomatidae, are particularly vulnerable

when leaving their cases because these organisms rely on their cases for respiration. These cases are osmotically semipermeable and allow the organisms to obtain oxygen via diffusion as water passes by the structure (Wichard et al. 1993).

It's unclear why other taxonomic groups, particularly other sensitive taxa such as ephemeropterans and plecopterans, have not demonstrated declines like those found in the trichopteran community. Among these sensitive taxonomic groups, trichopterans may be the first community members in the Mancos River to reflect effects of chronic stress due to their sensitivities to thermal and hydrologic stress. If patterns of increasing stream temperature and decreasing streamflow continue, further shifts in aquatic community structure and function should be expected.

Ecological Integrity

Results from the Colorado Multi-Metric Index (MMI) indicate that both monitoring sites on the Mancos River are experiencing stress and have failed to meet the water quality standard for attainment of the warm water aquatic life use at different points through time (Figures 12a & 12b). Data indicate that 28.3% (n=53) of samples from MAN01 and 20.5% (n=39) from MAN02 were impaired. While data for 2016 and 2018 do not exist for MAN02, drying and the lack of stream flow at that site suggest that the site should be considered impaired for those years as well. The utility of the MMI is in its ability to determine if the aquatic community is meeting its aquatic life use or if it exists in a state of impairment. The aquatic life use for this stretch of the Mancos River states that attainment is met by: “...waters that currently are capable of sustaining a wide variety of warm water biota including sensitive species or could sustain such biota but for the correctable water quality conditions. Waters shall be considered capable of sustaining such biota where physical habitat, water flows or levels, and water quality conditions result in no substantial impairment of the abundance and diversity of species” (CDPHE 2003). Metrics were chosen for the MMI based on qualities that are ecologically meaningful, can represent multiple ecological attributes, and can differentiate between reference (attainment) and stressed (impaired) conditions (Table 2, CDPHE 2003). While the MMI is not designed to specifically identify causes of impairment, investigation into the metrics can help identify potential impairment points.

Five metrics describing sensitive taxa and habitat appear to be contributing to impairment in samples from the sites on the Mancos River. Decreases in the number of EPT taxa (EPTTax), percent intolerant taxa in sample (pt_intol), number of facultative predators and shredders (PredShrTaxFAC), number of clinger taxa (ClingerTax), and percent EPT excluding Baetidae (EPTnoBPct) ranged from -35.5% to -68.1%. The decrease in the number of facultative predators and shredders is due to the decrease in shredder taxa in samples. This result corresponds with the findings of Pyne & Poff (2017) who suggested that thermal and hydrologic stress will result in few scrapers and shredders as decreased flows restrict the movement of basal resources through the system. EPT metrics can be explained primarily through the decline in Trichoptera abundance and richness (Figures 11 a & b). While variability in ephemeropteran and plecopteran abundances occurred from year to year there was significant decline in trichopteran abundance at MAN01 over the course of this study. Furthermore, while not statistically significant, trichopteran richness (slope = -0.44, p = 0.12) and relative abundance (slope = -1.84, p = 0.08) both showed patterns of decline at MAN02, and those

declines likely contributed to impairment at the site (Appendix B). The loss of clinger taxa in impaired samples is an expected response, given the declines in EPT taxa metrics among impaired samples. Most clinger taxa found in the Mancos River belong to an EPT Order. Furthermore, decreases in clinger densities have been directly linked to decreased stream flows during drought in the Sierra Nevada of California (Herbst 2019). As discussed earlier, when stream discharge decreases dissolved oxygen decreases as a direct result of increased temperature (Bonacina et al. 2022, Forster et al. 2012, van Vliet 2011). Clinger taxa found in the Mancos River are adapted to well oxygenated flows found at higher velocities (Cummins et al. 2008) and as discharge decreases taxa dependent on those conditions should decrease accordingly (Verdonschot & van den Hoorn 2010). Stream flow velocity data from MAN01 indicate a significant decrease in velocity over time (Figure 6) which could be the potential mechanism for the decline in clinger taxa at that site.

Management Implications/Concerns

Continuous monitoring of stream discharge and temperature will be key to understanding how a warming climate will affect the aquatic community in the Mancos River. Thermal and hydrologic stress have been documented in the upper section of the river over the course of this study. Further, flow at MAN02 was not sufficient to permit sampling during three of the monitoring years included in this study. The lack of streamflow likely had negative impacts on the aquatic community during those years. Understanding annual and seasonal variations in thermal and hydrologic regimes will help managers appropriately plan for community responses.

The reach of the Mancos River within Mesa Verde National Park is situated downstream from ranchlands where water extraction occurs through the summer and fall months. A warming climate will likely result in greater extraction rates upstream and reduced flow is the critical concern for the future of the Mancos River in the park. Water abstraction can exacerbate the effects of thermal and hydrologic stress and lead to greater declines in abundance and in some cases extirpation of fauna from lotic communities (Salmaso et al. 2021, Pyne & Poff 2017, Rader & Belish 1999). The likely increase in water extraction upstream of the park that will accompany increased warming will likely have significant consequences on future management actions within the park. Monitoring the changing hydrologic regime of the river in Mesa Verde will be key to understanding the effects of extraction on the aquatic community.

During 2023, MEVE natural resource staff implemented a beaver dam analog (BDA) project at the MAN01 monitoring site. The goal of this project was to increase habitat heterogeneity as well as increasing residency time of streamflow through the reach thereby increasing beaver immigration into these reaches of the river. Beaver dams and their analog structures can serve as a sufficient buffer against summer stream temperature maxima (Weber et al. 2017). However, the benefit of buffering will only extend to portions of the reach where the BDAs exist. Pooling that occurs behind the structures can provide refugia for aquatic taxa when thermal stress increases. However, one tradeoff with such efforts will be that the loss of riffle habitat will likely decrease dissolved oxygen content in those reaches as water is slowed and less turbulent. An increased frequency in deeper pools and few riffles will likely slow recovery of the trichopteran community. Other restoration techniques have proven useful to buffering stream temperature increases. Pierce et al. (2014) has

shown that a combination of channel narrowing, and restoration of riparian vegetation was successful at lowering summer stream maxima temperatures by 3 °C. Canopy cover at both monitoring sites is increasing over time (Figures 6 a & b), but this represents an increase in streamside willow cover and not Cottonwood overstory. Restoration of overstory canopy along the length of the Mancos River in MEVE could assist in lowering stream temperatures. However, any overstory restoration could be hindered by the expansion of beaver which has been occurring within the park in recent years. Areas of low beaver activity in the Mancos should be prioritized for overstory restoration to avoid wasting valuable park resources.

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Appendix A. Aquatic Macroinvertebrate Metrics

Table 6 shows the aquatic macroinvertebrate metrics used in the trend analysis presented in this report.

Table 6. Aquatic macroinvertebrate metrics used in trend analyses. Metrics describe important ecological attributes including function, tolerance to disturbance, and habitat availability.

Metric Type	Metric	Metric Description
Abundance/Richness/Diversity	Total Abundance	Total number of individuals
	Taxa Richness	Total number of taxa, estimated from rarified sample coverages (Chao et al. 2014, Hsieh et al. 2016)
	Diversity (Simpson's Diversity Index)	Estimated diversity (Simpson's diversity index) based on rarified sample coverages (Chao et al. 2014, Hsieh et al. 2016)
Tolerance to perturbation	Tolerance Relative Abundance (%)	Percent of individuals considered not to be sensitive to perturbation.
	Moderately Tolerant Relative Abundance (%)	Percent of individuals considered to be moderately sensitive to perturbation.
	Intolerant Relative Abundance (%)	Percent of individuals considered to be sensitive to perturbation.
	Tolerance Richness (%)	Percent of taxa considered not to be sensitive to perturbation.
	Moderately Tolerant Richness (%)	Percent of taxa considered to be moderately sensitive to perturbation.
	Intolerant Richness (%)	Percent of taxa considered to be sensitive to perturbation.
Functional Feeding Groups	Collector-Gatherer Relative Abundance (%)	Percent of individuals that collect fine particulate matter from the streambed.
	Collector-Filterer Relative Abundance (%)	Percent of individuals that filter fine particulate matter from the water column.
	Scraper Relative Abundance (%)	Percent of individuals that consume algae and bacteria.
	Shredder Relative Abundance (%)	Percent of individuals that consume periphyton and coarse particulate organic matter.
	Predator Relative Abundance (%)	Percent of individuals that prey on other invertebrates in the stream.
	Collector-Gatherer Richness (%)	Percent of taxa that collect fine particulate matter from the streambed.
	Collector-Filterer Richness (%)	Percent of taxa that filter fine particulate matter from the water column.

Table 6 (continued). Aquatic macroinvertebrate metrics used in trend analyses. Metrics describe important ecological attributes including function, tolerance to disturbance, and habitat availability.

Metric Type	Metric	Metric Description
Functional Feeding Groups (continued)	Scraper Richness (%)	Percent of taxa that consume algae and bacteria.
	Shredder Richness (%)	Percent of taxa that feed on periphyton and coarse particulate organic matter.
	Predator Richness (%)	Percent of taxa that prey on other invertebrates in the stream.
Habit Groups	Burrower Relative Abundance (%)	Percent of individuals that live in fine sediments.
	Climber Relative Abundance (%)	Percent of individuals that live on hydrophytes and detritus.
	Clinger Relative Abundance (%)	Percent of individuals that attach to substrate in riffles and swift flowing waters.
	Sprawler Relative Abundance (%)	Percent of individuals that live on the surface of floating detritus or hydrophytes.
	Swimmers Relative Abundance (%)	Percent of individuals that swim through water column and live on submerged substates.
	Burrower Richness (%)	Percent of taxa that live in fine sediments.
	Climber Richness (%)	Percent of taxa that live on hydrophytes and detritus.
	Clinger Richness (%)	Percent of taxa that attach to substrate in riffles and swift flowing waters.
	Sprawler Richness (%)	Percent of taxa that live on the surface of floating detritus or hydrophytes.
	Swimmer Richness (%)	Percent of taxa that swim through water column and live on submerged substates.
Sensitive EPT group	Ephemeroptera Relative Abundance (%)	Percent of individuals that are mayflies.
	Plecoptera Relative Abundance (%)	Percent of individuals that are stoneflies.
	Trichoptera Relative Abundance (%)	Percent of individuals that are caddisflies.
	Ephemeroptera Richness (%)	Percent of taxa that are mayflies.
	Plecoptera Richness (%)	Percent of taxa that are stoneflies.
	Trichoptera Richness (%)	Percent of taxa that caddisflies.
	Ratio of Non-sensitive to Sensitive Trichoptera	Ratio of Trichopterans in Hydroptilidae + Hydropsychidae: total caddisfly abundance (sensitive Trichoptera)

Table 6 (continued). Aquatic macroinvertebrate metrics used in trend analyses. Metrics describe important ecological attributes including function, tolerance to disturbance, and habitat availability.

Metric Type	Metric	Metric Description
Non-Insect	Non-Insect Relative Abundance (%)	Percent of individuals that are not insects.
	Chironomidae Relative Abundance (%)	Percent of individuals that are midges.

Appendix B. Trend Analysis Results

Tables 7 and 8 show the trend analysis results for aquatic macroinvertebrates evaluated during this study.

Table 7. Trend analyses for chosen aquatic macroinvertebrate metrics for the MAN01 monitoring site. Strength of trend was determined by Kendall's tau where values approaching -1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trends occur when $p \leq 0.05$. Slopes and intercepts were calculated using the Theil-Sen estimator. Direction of trend is shown in the Trend column where $\uparrow$ = increasing and $\downarrow$ = decreasing trend and $\leftrightarrow$ indicates no trend was found.

Category	MAN01	Intercept	Slope	Kendall's τ	Lower Confidence Limit	Upper Confidence Limit	p-value	Trend
Metrics	Total Abundance	163.00	11.00	0.13	-3.00	28.00	0.19	$\leftrightarrow$
	Richness	16.51	-0.17	-0.05	-0.93	0.54	0.57	$\leftrightarrow$
	Diversity (Simpson's Diversity Index)	2.58	-0.01	-0.02	-0.11	0.09	0.85	$\leftrightarrow$
Tolerance	Tolerance Relative Abundance (%)	1.42	-0.03	-0.06	-0.10	0.06	0.60	$\leftrightarrow$
	Moderately Tolerant Relative Abundance (%)	29.74	2.60	0.27	0.73	4.60	0.01	$\uparrow$
	Intolerant Relative Abundance (%)	69.48	-2.72	-0.26	-4.65	-0.75	0.01	$\downarrow$
	Tolerance Richness (%)	7.69	0.00	0.02	-0.24	0.33	0.86	$\leftrightarrow$
	Moderately Tolerant Richness (%)	60.32	-0.53	-0.12	-1.52	0.30	0.23	$\leftrightarrow$
	Intolerant Richness (%)	30.84	0.81	0.17	0.00	1.59	0.09	$\leftrightarrow$
Functional Group	Collector-Gatherer Sample Relative Abundance (%)	26.20	0.89	0.12	-0.59	2.30	0.24	$\leftrightarrow$
	Collector-Filterer Sample Relative Abundance (%)	66.89	-0.74	-0.10	-2.39	0.89	0.33	$\leftrightarrow$
	Scraper Relative Abundance (%)	0.83	-0.03	-0.12	-0.12	0.06	0.45	$\leftrightarrow$
	Shredder Relative Abundance (%)	0.88	-0.04	-0.17	-0.10	0.09	0.38	$\leftrightarrow$
	Predator Relative Abundance (%)	4.40	-0.04	-0.03	-0.29	0.29	0.79	$\leftrightarrow$
	Collector-Gatherer Richness (%)	37.66	0.32	0.09	-1.04	0.40	0.35	$\leftrightarrow$

Table 7 (continued). Trend analyses for chosen aquatic macroinvertebrate metrics for the MAN01 monitoring site. Strength of trend was determined by Kendall's tau where values approaching -1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trends occur when $p \leq 0.05$. Slopes and intercepts were calculated using the Theil-Sen estimator. Direction of trend is shown in the Trend column where $\uparrow$ = increasing and $\downarrow$ = decreasing trend and $\leftrightarrow$ indicates no trend was found.

Category	MAN01	Intercept	Slope	Kendall's τ	Lower Confidence Limit	Upper Confidence Limit	p-value	Trend
Functional Group (continued)	Collector-Filterer Richness (%)	21.78	-0.60	-0.17	1.41	0.00	0.10	$\leftrightarrow$
	Scraper Richness (%)	0.00	0.00	0.02	0.00	0.00	0.83	$\leftrightarrow$
	Shredder Richness (%)	0.00	0.00	-0.09	0.00	0.00	0.42	$\leftrightarrow$
	Predator Richness (%)	29.41	0.98	0.24	0.21	1.79	0.02	$\uparrow$
Habit	Burrower Relative Abundance (%)	10.97	0.28	0.05	-0.57	1.52	0.63	$\leftrightarrow$
	Climber Relative Abundance (%)	0.50	-0.02	-0.06	-0.09	0.09	0.87	$\leftrightarrow$
	Clinger Relative Abundance (%)	68.01	-0.66	-0.09	-2.17	1.01	0.37	$\leftrightarrow$
	Sprawler Relative Abundance (%)	0.31	-0.20	-0.22	-0.44	-0.01	0.04	$\downarrow$
	Swimmers Relative Abundance (%)	8.60	0.54	0.16	-0.13	1.25	0.11	$\leftrightarrow$
	Burrower Richness (%)	30.95	0.40	0.14	0.00	1.19	0.16	$\leftrightarrow$
	Climber Richness (%)	0.00	0.00	-0.01	0.00	0.00	0.91	$\leftrightarrow$
	Clinger Richness (%)	41.79	-0.30	-0.01	-1.30	0.38	0.37	$\leftrightarrow$
	Sprawler Richness (%)	14.29	0.00	-0.04	-1.01	0.57	0.72	$\leftrightarrow$
	Swimmer Richness (%)	1.00	0.00	0.17	0.00	0.00	0.15	$\leftrightarrow$
EPT	Ephemeroptera Relative Abundance (%)	9.82	0.40	0.12	-0.27	1.17	0.22	$\leftrightarrow$
	Plecoptera Relative Abundance (%)	1.45	-0.03	-0.04	-0.16	0.16	0.71	$\leftrightarrow$
	Trichoptera Relative Abundance (%)	67.60	-2.67	-0.26	-4.89	-0.55	0.01	$\downarrow$
	Ephemeroptera Richness (%)	13.81	0.08	0.05	-0.46	0.65	0.63	$\leftrightarrow$
	Plecoptera Richness (%)	6.73	0.21	0.14	0.00	0.96	0.19	$\leftrightarrow$
	Trichoptera Richness (%)	16.76	-0.64	0.24	-1.19	-0.08	0.02	$\downarrow$

Table 7 (continued). Trend analyses for chosen aquatic macroinvertebrate metrics for the MAN01 monitoring site. Strength of trend was determined by Kendall's tau where values approaching -1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trends occur when $p \leq 0.05$. Slopes and intercepts were calculated using the Theil-Sen estimator. Direction of trend is shown in the Trend column where $\uparrow$ = increasing and $\downarrow$ = decreasing trend and $\leftrightarrow$ indicates no trend was found.

Category	MAN01	Intercept	Slope	Kendall's τ	Lower Confidence Limit	Upper Confidence Limit	p-value	Trend
EPT (continued)	Non-Sensitive Trichoptera Relative Abundance (%)	100.00	0.00	0.01	0.00	0.00	0.98	$\leftrightarrow$
Non-Insects	Non-Insect Relative Abundance (%)	3.06	-0.08	-0.08	-0.59	0.20	0.60	$\leftrightarrow$
Chironomids	Chironomidae Relative Abundance (%)	8.40	0.28	0.07	-0.52	1.41	0.52	$\leftrightarrow$

Table 8. Trend analyses for chosen aquatic macroinvertebrate metrics for the MAN02 monitoring site. Strength of trend was determined by Kendall's tau where values approaching -1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trend is shown where $p \leq 0.05$. Slopes and intercepts were calculated using the Theil-Sen estimator. Direction of trend is shown in the Trend column where $\uparrow$ = increasing and $\downarrow$ = decreasing trend and $\leftrightarrow$ indicates no trend was found.

Category	MAN02	Intercept	Slope	Kendall's τ	Lower CL	Upper CL	p-value	Trend
Metric	Total Abundance	204.67	2.83	0.06	-14.50	20.50	0.63	$\leftrightarrow$
	Richness	14.98	0.00	-0.01	-0.83	0.82	0.96	$\leftrightarrow$
	Diversity (Simpson's Diversity Index)	2.87	0.00	0.02	-0.09	0.08	0.90	$\leftrightarrow$
Tolerance	Tolerance Relative Abundance (%)	2.05	-0.13	-0.29	-0.38	0.01	0.05	$\leftrightarrow$
	Moderately Tolerant Relative Abundance (%)	35.78	0.96	0.10	-1.32	2.91	0.38	$\leftrightarrow$
	Intolerant Relative Abundance (%)	59.40	-0.62	-0.08	-2.73	1.21	0.49	$\leftrightarrow$
	Tolerance Richness (%)	6.67	0.00	0.12	0.00	0.79	0.35	$\leftrightarrow$
	Moderately Tolerant Richness (%)	60.47	-0.83	-0.16	-2.08	0.38	0.18	$\leftrightarrow$
	Intolerant Richness (%)	35.85	0.65	0.15	-0.60	1.91	0.20	$\leftrightarrow$

Table 8 (continued). Trend analyses for chosen aquatic macroinvertebrate metrics for the MAN02 monitoring site. Strength of trend was determined by Kendall's tau where values approaching -1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trend is shown where $p \leq 0.05$. Slopes and intercepts were calculated using the Theil-Sen estimator. Direction of trend is shown in the Trend column where $\uparrow$ = increasing and $\downarrow$ = decreasing trend and $\leftrightarrow$ indicates no trend was found.

Category	MAN02	Intercept	Slope	Kendall's τ	Lower CL	Upper CL	p-value	Trend
Functional Group	Collector-Gatherer Sample Relative Abundance (%)	28.83	-0.96	-12.00	-2.24	0.92	0.33	$\leftrightarrow$
	Collector-Filterer Sample Relative Abundance (%)	59.00	0.87	0.10	-1.08	2.80	0.39	$\leftrightarrow$
	Scraper Relative Abundance (%)	0.32	0.11	0.06	-0.18	0.57	0.82	$\leftrightarrow$
	Shredder Relative Abundance (%)	1.06	0.04	0.06	-0.76	2.08	0.87	$\leftrightarrow$
	Predator Relative Abundance (%)	8.29	-0.06	-0.03	-0.60	0.61	0.81	$\leftrightarrow$
	Collector-Gatherer Richness (%)	34.26	0.26	0.07	-0.79	1.32	0.62	$\leftrightarrow$
	Collector-Filterer Richness (%)	23.08	0.00	0.02	-0.81	0.87	0.83	$\leftrightarrow$
	Scraper Richness (%)	0.00	0.00	-0.13	0.00	0.00	0.32	$\leftrightarrow$
	Shredder Richness (%)	0.00	0.00	-0.30	0.00	0.00	0.83	$\leftrightarrow$
	Predator Richness (%)	33.70	0.32	0.06	-1.19	1.54	0.59	$\leftrightarrow$
Habit	Burrower Relative Abundance (%)	4.45	-0.08	-0.07	-0.46	0.32	0.59	$\leftrightarrow$
	Climber Relative Abundance (%)	0.90	0.03	0.15	-0.49	0.90	0.84	$\leftrightarrow$
	Clinger Relative Abundance (%)	65.35	1.40	0.14	-0.87	2.84	0.22	$\leftrightarrow$
	Sprawler Relative Abundance (%)	1.92	-0.02	-0.02	-0.27	0.51	0.87	$\leftrightarrow$
	Swimmers Relative Abundance (%)	20.56	-0.61	-0.08	-2.11	1.22	0.49	$\leftrightarrow$
	Burrower Richness (%)	30.16	-0.40	-0.11	-1.32	0.56	0.38	$\leftrightarrow$
	Climber Richness (%)	0.00	0.00	0.05	0.00	0.00	0.74	$\leftrightarrow$
	Clinger Richness (%)	48.86	0.28	0.06	-1.07	1.79	0.60	$\leftrightarrow$
	Sprawler Richness (%)	1.00	0.00	-0.14	-0.14	0.00	0.28	$\leftrightarrow$
	Swimmer Richness (%)	10.50	0.50	0.19	0.00	1.11	0.11	$\leftrightarrow$

Table 8 (continued). Trend analyses for chosen aquatic macroinvertebrate metrics for the MAN02 monitoring site. Strength of trend was determined by Kendall's tau where values approaching -1 describe a strong decreasing trend and values approaching 1 describe a strong increasing trend. Significant trends occur when $p \leq 0.05$. Slopes and intercepts were calculated using the Theil-Sen estimator. Direction of trend is shown in the Trend column where $\uparrow$ = increasing and $\downarrow$ = decreasing trend and $\leftrightarrow$ indicates no trend was found.

Category	MAN02	Intercept	Slope	Kendall's τ	Lower CL	Upper CL	p-value	Trend
EPT	Ephemeroptera Relative Abundance (%)	21.25	-0.61	-0.09	-2.13	1.05	0.48	$\leftrightarrow$
	Plecoptera Relative Abundance (%)	2.71	0.30	0.18	-0.12	0.76	0.17	$\leftrightarrow$
	Trichoptera Relative Abundance (%)	58.07	-1.84	-0.21	-4.98	0.17	0.08	$\leftrightarrow$
	Ephemeroptera Richness (%)	17.05	-0.33	-0.10	-1.08	0.42	0.42	$\leftrightarrow$
	Plecoptera Richness (%)	8.26	0.57	0.17	-0.28	1.45	0.16	$\leftrightarrow$
	Trichoptera Richness (%)	16.04	-0.44	-0.19	-0.86	0.00	0.12	$\leftrightarrow$
	Non-Sensitive Trichoptera Relative Abundance (%)	100.00	0.00	0.24	0.00	0.00	0.08	$\leftrightarrow$
Non-Insects	Non-Insect Relative Abundance (%)	2.11	-0.26	-0.35	-1.84	0.09	0.13	$\leftrightarrow$
Chironomids	Chironomidae Relative Abundance (%)	2.69	0.02	0.02	-0.24	0.29	0.85	$\leftrightarrow$

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