The Effects of Anthropogenic Activity in Forested Campgrounds on Stream Ecosystems

Charlynn Teter

ABSTRACT

Urbanization along stream banks is a key cause of river degradation, impacting streams by increasing run-off, decreasing natural filtration of run-off, and adding chemical and organic pollutants to the water. Natural spaces, open to the public, may ameliorate some of these ill-effects, but may also pose similar risks, and the impacts of disturbances from campgrounds on river health is much less understood. This study investigates the physical and fauna health of Pescadero Creek in a County Campground along the gradient of sites and human activity in memorial park. Sites at the park were characterized by clean water taxa such as Diptera, Ephemeroptera, Plecoptera, and Trichoptera. The most downstream site showed consistently poorer measurements of biological health, scoring more poorly than the other sites on: Richness (10), EPT Richness (1), Percent EPT (3.09%), FBI (7.12), and CSCI (0.25). In contrast, the two most upstream sites had, on average: Richness 21.75, EPT Richness 8.5, Percent EPT 20.3%, FBI 5.19, and CSCI 0.61. The CSCI metric showed a statistically significant and consistent decrease between Site 4 (the most upstream site) and Site 1 (the most downstream site). Beyond human influence, physical environmental factors did not show extreme variation between sites; although factors such as substrate and flow varied slightly between sites, they did not consistently change moving from upstream to downstream. These results suggest that the campground and other recreational activity detrimentally impacted the stream's health along the campground reach.

KEYWORDS

Benthic macroinvertebrates; pollution; riparian ecosystems; campgrounds; biomonitoring

INTRODUCTION

All life requires clean water, and as humans reshape the environment, human activity and pollution have imperiled critical freshwater ecosystems (Stella and Bendix 2019). In addition to the direct consequences to the river itself, organisms in surrounding areas, including humans, also suffer increased mortality when the streams they rely on deteriorate (Paetzold et al. 2011). Traits of an imperiled stream include, but are not limited to: high concentrations of pollutants or other contaminants, lower species diversity and richness, and an unusually high level of pollution-tolerant species (Walsh et al. 2005). This finding is so ubiquitous that scientists coined a specific term for particularly degraded streams in cities: "urban stream syndrome" (Booth et al. 2015). Freshwater organisms are predicted to be five times more likely to become extinct than terrestrial organisms (Rasmussen and Ricciardi 1999), and in the United States, half of all freshwater bodies are too impaired for safe swimming or fishing (Kelderman et al. 2022).

Past studies frequently focused on these heavily urbanized streams, and have identified several key issues commonly leading to stream degradation, including: paved areas causing increased pollutant run-off, excess organic matter in the streams, and increased erosion due to deforestation (Schoonover et al. 2005, Walsh et al. 2005, Booth et al. 2015). These factors combine to make urban rivers some of the most degraded ecosystems in the world. Even across varied climatic conditions, these key problems remain consistent across urban streams (Booth et al. 2015). Other studies contrast these degraded rivers with rivers which maintain buffer zones of riparian vegetation along banks which have not suffered from human disturbance. These buffered rivers typically have proven more healthy over time (Hession et al. 2003).

However, urbanization impacts are not limited to streams flowing through city environments. Even in natural spaces like city parks, humans have affected the environment with many of the same issues found across archetypal urban streams, such as patches of paved ground for parking lots and campground sites. The biggest issue in this recreation management revolves around the volume and location of visitors. Analyses that distinguish between high and low traffic areas have clearly indicated that higher traffic areas suffer more degradation (Marion et al. 2016). Prior investigations into parks suggest a diminishing impact of each additional visitor, indicating that carefully allocating which spaces permit visitors is crucial, as is determining the appropriate volume of permitted guests (Marion and Sober 1987). Humans visiting parks can impact or imperil the natural resources in these parks, especially through vegetation soil trampling and wildlife disturbance (Gary 1982, Marion et al. 2016, Barnett et al. 2016).

Assessments of impacts on water features in parks have been more narrow in scope. A large proportion of the prior research in this field has focused on the narrow impacts of singular and specific use activities such as: the bacterial impact of cattle grazing or hiking (Derlet et al. 2004), vegetation and soil damage (Marion et al. 2016), disturbances caused by cattle or canoes (Marion et al. 2016), and bathing or swimming pollution (Butler et al. 2021). These findings provide insight into the impacts and consequences of individual types of human activities within parks, but they do not capture the overall water quality of rivers subjected to many different types of disturbances in parks.

Research on campgrounds' impacts is particularly minimal. Some earlier works investigated campgrounds' impact on environmental matters of deforestation or erosion that do ultimately affect rivers, but did not specifically address water quality (Marion and Sober 1987). More recent investigations similarly focused on impacts such as erosion and vegetation damage or destruction, factors which impact water quality, but did not specifically address water quality (Farrell and Marion 1998, Marion 2003, Eagleston and Marion 2017). Similarly, investigations into the campgrounds' campfires have generally focused on damage done to terrestrial soils and trees, not the nearby water (Marion et al. 2016). Where research specifically dedicated to the impact of campgrounds on water systems exists, the historical research is narrow and typically limited to the presence or absence of bacterial contaminants sourced from improper waste disposal (Gary 1982). More recent studies account for more factors when assessing campground water quality, but still typically address only a handful of chemical factors. Frequent analysis metrics include assessments of the levels of specific bacteria in conjunction with chemical metrics such as dissolved oxygen levels (Wasowski et al. 2013) or heavy metal measurements (Flack et al. 1988). They typically do not analyze benthic macroinvertebrates and decline to comprehensively investigate the impact of other features of campgrounds, such as increased impervious surfaces created through paving.

A crucial tool for assessing the overall health of streams in human-influenced parks is using benthic macroinvertebrates. Benthic macroinvertebrates are a well-established tool for assessing streams' water quality (Wang and Kanehl 2003). These organisms display a wide range of tolerances to ecological disturbances, including anthropogenic influences that cause changes to the water chemistry or sediment levels (Jackson and Füreder 2006, Oliveira and Callisto 2010, Pinto et al. 2014). When observing a representative sample of benthic macroinvertebrates, practitioners can assert that the stream is likely unhealthy, if highly sensitive macroinvertebrates are absent while tolerant organisms are present, or likely healthy, if a diverse range of pollutant-sensitive benthic macroinvertebrates exist in the sample (Tampo et al. 2021). Benthic macroinvertebrates provide a more complete picture of the stream ecosystem than simple point chemical assessments because they reflect conditions over a long time range (Tampo et al. 2021). Furthermore, when assessing stream health based on macroinvertebrates, stream monitors need not necessarily know what chemicals, pollutants, or disturbances to examine as narrowly as would be required to identify and test for specific bacteria or chemical concerns (Kebede et al. 2020). Benthic macroinvertebrates frequently reflect anthropogenic disturbances even when testing for individual bacteria does not yield meaningful results (Kebede et al. 2020). Paired with assessments of the physical habitat, the benthic macroinvertebrate analysis can offer a complete picture of stream community health.

This study assesses the impact of recreational campgrounds on an otherwise "preserved" river's health and stability. This assessment will involve evaluating: (1) variation in habitat condition along the river's gradient; (2) change in fauna along the gradient of human occupancy and activity; and (3) correlation in any factors between habitat and fauna. The campground features impervious surfaces in the form of roads, access to the stream by hiking trails, high occupancy during the summer months, and several parking lots. Given both the impervious surface and the likelihood of recreational swimming or wading in the stream itself, I anticipate that the habitat quality and fauna diversity will both be highest immediately upstream of the campground, and then will decrease as human activity increases. I will conduct this assessment by taking a series of habitat assessments and kicknet benthic macroinvertebrate samples.

METHODS

Study Area

The forests along Pescadero Creek are mostly redwoods and Douglas-fir trees, with some cypress, myrtle, laurel, madrone, maple, and oak trees ("Pescadero Creek Park Natural Features | County of San Mateo, CA" n.d.). Pescadero Creek flows through Memorial Park, which features a major campground. Memorial Park's campground was established in July 1924 (Staff 2021). There were approximately 141 campsites in Memorial Park as of 2019, a decline from the 300 campsites initially established in 1924, and on average approximately 57,000 campers visit the campground annually with the heaviest occupancy during the summer months (Staff 2021).

I selected four study sites along Pescadero Creek in Memorial County Park, California (Figure 1). Site 1, the most downstream site, was located near the Homestead Flat Youth Camp (Latitude and Longitude of the Youth Camp 37.273482N, -122.301996W). From this location, I walked directly to the closest point of access to the river. Site 2 was located at the Huckleberry Flat Picnic Area (37.374542N, -122.29644W), from which the river was accessible by the hiking trail which led to the closest point of entry to the river from the Picnic Area. I entered the river directly next to the out-of-use suspension bridge. Site 3 was located directly next to the amphitheater (37.273725N, -122.292513W). Site 4 was located next to the Legion Flat Picnic Area (37.374566N, -122.288460W). At the Legion Flat, I entered the river at a point of access signposted as the prior swimming hole, now undergoing restoration.

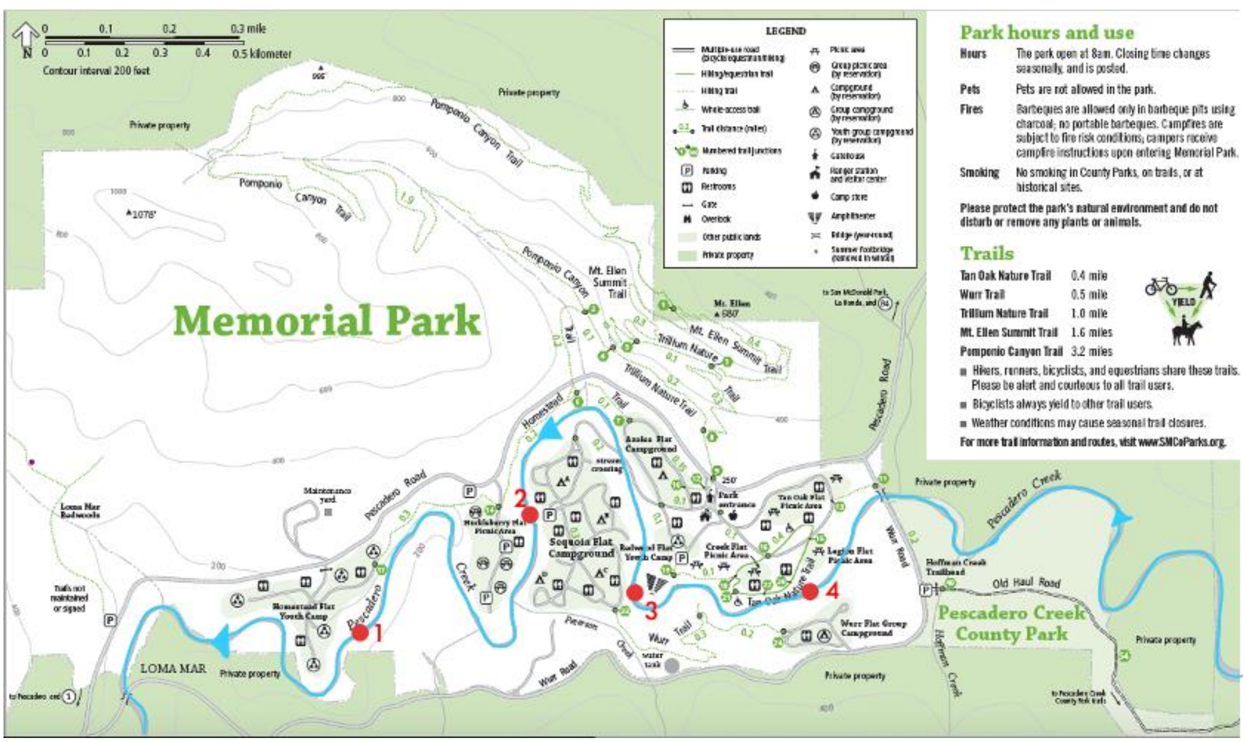

Figure 1. Site map of Memorial Park. Study site locations along the stream (in blue) are marked with red dots. The arrow indicates the direction of the stream flow.

At each site, Dr. Patina Mendez (University of California, Berkeley) and I constructed a single transect which spanned the entire river in width and up to 10 m upstream. We took all the samples and measurements within that transect space. We began our sampling at the most downstream site and moved upstream in a linear fashion so that organisms and sediment dislodged from sampling would not impact sites not yet sampled.

Physical Habitat Measurements

We took a series of physical habitat measurements at each site by following SWAMP protocols for measurements and using their datasheets ("SWAMP - Standard Operating Procedures | California State Water Resources Control Board" n.d.). Due to time constraints, we conducted a modified version of the standard procedure with only one transect per site, rather than 10 transects in 10 meter intervals.

To complete physical habitat assessments, we used the physical habitat datasheet of the SWAMP protocol and began by measuring the physical dimensions of the stream. We stretched a tape measure across the width of the stream. We measured the "bankfull width," defined as the width of the river that is submerged when the water is at the high water mark. Visually, this width is generally distinguished by changes in the gradient of the banks. Next, we measured the "wetted width," defined as the width of the river that was currently submerged in water. After taking this measurement, we left the tape measure stretched across the river. We used a meter stick to measure the "bankfull height," defined as the difference between the maximum water height where the bankfull width occurred and the current water height.

Next, we took flow measurements. We moved along the tape measure that we had already placed across the wetted width of the stream. Approximately every 1-2 meters (depending on the width of the stream and the depth, because the flow meter requires a minimum depth of water), we recorded the depth of the river using a measuring stick and the flow of the river using a Gurley pygmy meter.

We next took pebble counts to characterize the substrate. We moved along the tape measure wetted width of the stream and took substrate measurements at five points along the width of the stream (the left side, left center, center, right center, and right side of the river). At each of those points, we recorded the distance from the left bank and the distance between the water's surface and the beginning of the substrate material. Then, we picked up a substrate piece at random and passed it through a gravelometer and recorded the particle size code for this piece of substrate. We visually estimated the percentage of the cobble which was submerged in substrate. We recorded on a scale of 0 (none) to 5 (a thick coat) how much microalgae was present in the surrounding area. We noted whether the following plants were present or absent: attached macroalgae, unattached macroalgae, and macrophytes. We also recorded whether coarse particulate organic matter was present or absent in the area immediately surrounding the cobble.

To estimate shade coverage of the stream, we used a spherical crown densiometer. We held the densiometer flat and faced the center upstream, center downstream, the right bank, and the left bank of the river. While facing each direction, we recorded the number of dots on the mirror that shrubbery covered. At sites 1-3, we erroneously moved 1-2 meters in the direction we had pivoted to face each time. At site 4, after noting the error, we recorded both the values obtained using the erroneous method and the values determined using the correct method.

To measure the slope of the river we used a clinometer. We measured 20 meters along the river in length upstream and downstream of the transect. At the upstream end, one person held a 2 meter survey staff. On the downstream end, the other team member knelt at 80cm and used a clinometer as a level. The upstream team member slid their hand down the survey staff until it reached the line level and then read the measurement. After making this measurement, we checked the measurement mark on that upstream meter stick and recorded the difference between the two meter sticks.

Finally, we recorded coverage rankings for human-made and natural features along the bank of the river and within the river itself using categories and ratings defined in the SWAMP protocols. We recorded in-stream habitat complexity by ranking the extent of habitat features on a scale from 0 (completely absent) to 5 (covering more than 75% of the in-stream habitat). The complete scale is as follows: 0 (0% present), 1 (less than 10% covered), 2 (10-40% covered), 3 (40-75% covered), 4 (>75% covered). We ranked the following features: filamentous algae, aquatic macrophytes, boulders, small woody debris (<0.3m), large woody debris (>0.3m), undercut banks, overhang, live tree roots, and artificial structures. We rated the left and right banks as "eroded," "vulnerable," or "stable." Along the bank, we visually observed human disturbances (if any were present) and recorded on a scale from Y (present within wetted margins) to 0 (not present within 50m of the bank) whether specific types of human disturbances were present. We specifically ranked the presence of the following human influence features: walls/dams, buildings, pavement, roads/railroads, pipes, landfills, parks/lawns, row crops, pastures/ranges, logging operations, mining activity, vegetation management, bridges/abutments, and orchards/vineyards. We also recorded the coverage of different vegetation types along both sides of the bank. We ranked the vegetation coverage on a scale from 0 (completely absent) to 4 (more than 75% covered). We recorded the coverage of the following vegetation types: trees & saplings taller than 5 meters, vegetation between 0.5m and 5m, woody shrubs and saplings less than 0.5m, herbs and grasses, and barren/bare soil/duff.

Finally, we took photographs of the site.

Biological Sampling

To collect biological samples of benthic macroinvertebrates, we used a modified version of the SWAMP protocol ("SWAMP - Standard Operating Procedures | California State Water Resources Control Board" n.d.). At each site within the single transect, we took two single-habitat samples, one along the left-center of the river and one along the right-center of the river. For each sample, I used a 500 micrometer D-shaped kicknet.

For the single-habitat samples, we first placed the net on the bottom of the river, with the open side of the net facing upstream. Then, we designated an approximately 1x1 foot area of the river immediately in front of the net's opening. We kicked and dug beneath the mud and rocks in this 1x1 foot area for 30 seconds, allowing the dislodged material to wash into the open net. Then, we removed the net from the water.

I preserved the samples. I filled a shallow plastic pan with river water from the same area where I took my sample. Then, I turned the net inside-out and submerged the net in the shallow pan, manually "washing" the net's contents into the pan. I then visually inspected the net for any remaining organisms that had not been washed into the pan, and added them to the pan as well. Next, I took the pan with the river water and sample material and poured it through a #35 sieve. I then rinsed the contents of this sieve into a plastic sample bag with 95% ethanol. I filled the plastic sample bag with 95% ethanol until the sample was fully submerged in ethanol. Upon returning to the lab, I completed the preservation process by draining the 95% ethanol and replacing it with 75% ethanol.

After preserving the samples in ethanol, I sorted the samples and conducted a family-specific level of analysis of the organisms in each sample. I first observed the samples under a dissecting microscope and removed all of the organisms from the debris. Next, I identified individuals to a family level using a dichotomous key (Harrington and Born 1999, Merit et al. 2008).

Water Chemistry

At site 4 along the river, I took measurements of the chemical qualities of the river. I used a multiprobe to record the following measurements of the water: temperature, dissolved oxygen, specific conductivity, and pressure. I also filled a plastic container with water from the river at that same point. I used this water sample to test for turbidity, pH, and alkalinity. I tested for turbidity with a turbidity meter (Hanna Instruments HI93414-01 Turbidity and Chlorine Portable Meter). I tested pH with a Millipore pH strip. I tested alkalinity with a LaMotte 3467 alkalinity test kit by conducting a titration.

Statistical Analysis

Biological Samples

To assess the condition of the biological communities within each section of the stream, I calculated the California Stream Condition Index (CSCI) for each site. I only identified the organisms to the family level, so my CSCI calculation used a modified index which calculated the CSCI based on family-level rather than species-level identifications ("Family Level Index" n.d.). I entered family-level identifications including the life stages of the organisms identified as well as latitude and longitudinal data. The CSCI Family Level index accounts then automatically accounted for site-level characteristics as well as the family identifications.

After inputting the data, I ran the CSCI Family Level index code, which produced a CSCI score for each of the habitats. I also calculated the following indices: Family Richness, EPT Richness, Percent EPT, and the Family Biotic Index at each site. For the Family Biotic Index, I relied on a chart of FBI values to manually calculate it (Barbour et al. 1999).

Finally, I used Non-metric multidimensional scaling (NMDS) to visualize differences between sites in terms of relative abundance of organisms at the site to identify statistically significant differences in organisms per family at each site. I used the vegan package (Oksanen et al. 2022) in Rstudio (RStudio Team 2020) to generate a 2-axis solution, and then correlated taxa scores with the NMDS axes.

Physical and Chemical Samples

For my physical analysis of the site, I analyzed each of the meaningful physical metrics independently. I descriptively identified any noteworthy differences between the physical habitats at each site. I also correlated these physical variables for the site with the NMDS axes from the benthic macroinvertebrate analysis to identify which of these differences were statistically significant.

Joint Analysis

After separately analyzing the biological and physical data, I relied on prior literature to assess how much of the variation in fauna communities could be attributed to natural physical habitat variation.

Comparisons to Prior Data Collections

In addition to analyzing the data I personally collected, I compared the measurements I collected in the winter of 2022 with previous assessments completed by regulatory agencies in 2019 (*Integrated Monitoring Report Part B: Creek Status Monitoring* 2020).

RESULTS

Physical Habitat Measurements

Site Dimensions

The wetted width does trend longitudinally between sites (Table 1). The Bankfull Width and Bankfull Height does vary significantly with the location variable. The Bankfull Width is larger at sites 3 and 4 compared to the other sites. Bankfull Height is larger at Site 2 compared to the upstream or downstream sites.

Table 1. Site Dimensions. Site 1 is the most downstream site; site 4 is the most upstream site.

Site	Wetted Width (m)	Bankfull Width (m)	Sandbar	Bankfull Height (m)	Cross-sectional Area (m^2)
1	11m	11.3m	NONE	0.7m	2.8
2	4.4m	11.7m	NONE	1.2m	0.34
3	15.6m	16.2m	Present between; 13.7m-5.6m from left bank	0.5m	75.6
4	6.2m	16.3m	NONE	0.6m	10.16

Discharge Measurements

The flow in meters cubed per second does not appear to have a consistent trend with the upstream/downstream variable (Table 2). Site 1 is an outlier from the other points, with a flow of 0.

Table 2. Discharge Measurements. I calculated the value for average discharge per second from the measured values of velocity in rotations per second using the relevant Gurley Pygmy Meter formula: (rotions/seconds)*(0.3)*(cross-sectional area) and then averaged the calculated discharge across each of the measured points of flow at each site. Site 1 is the most downstream site; site 4 is the most upstream site.

Site	Cross-sectional Area (m^2)	Average Discharge (m^3/s)
1	2.8	0
2	0.34	0.21
3	75.6	0.086
4	10.16	0.07

The U.S. Geological Survey (USGS) average reported flow in Pescadero Creek, Pescadero, California in cubic feet per second in November of 2022 was 3.03 (Figure 2). In cubic meters per second, this value is approximately 0.085. In August of 2019, the reported USGS flow was 6.69 ft^3/s, equal to 0.19 m^3/s.

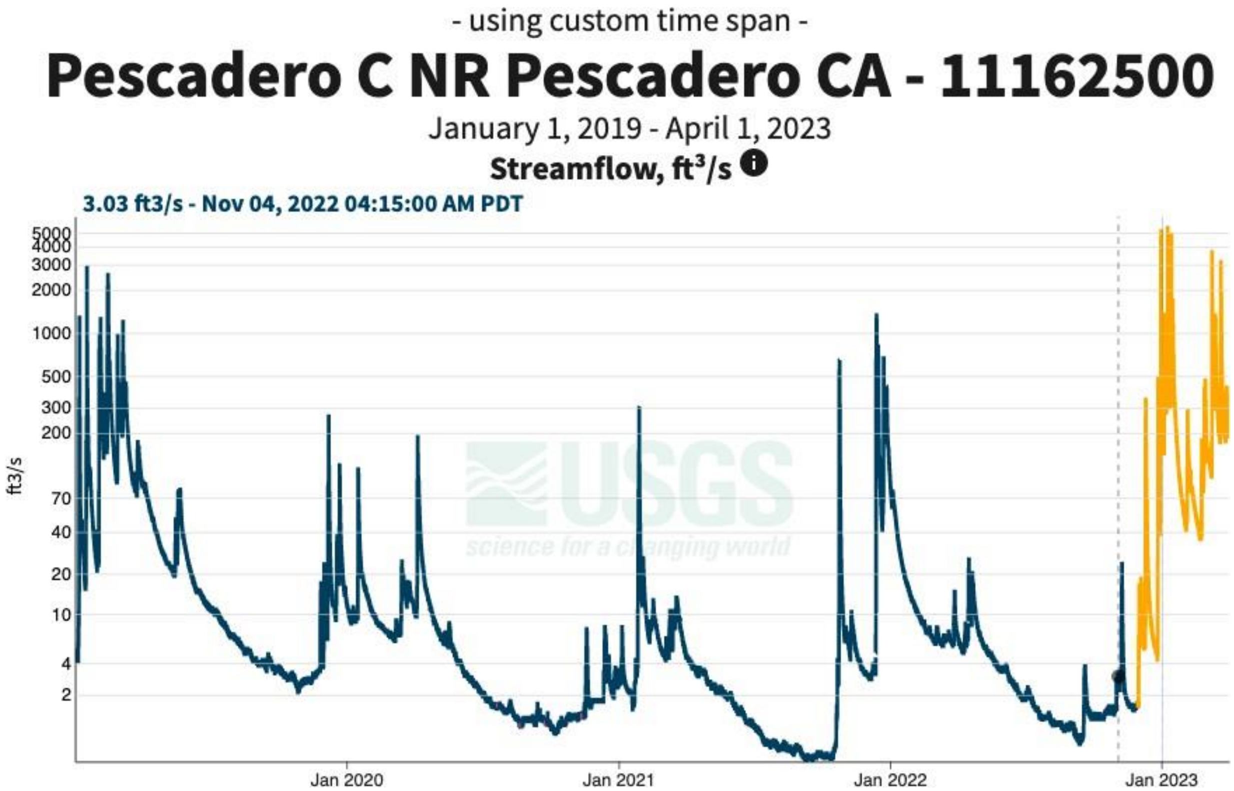

Figure 2. Flow measurements of Pescadero Creek. Data collected and figure generated by the USGS ("Pescadero C NR Pescadero CA" n.d.).

Substrate Measurements

The substrate size is significantly larger on average at Sites 3 and 4 (peaking in magnitude at Site 3) (Table 3). Average cobble embeddedness does not demonstrate any statistically significant trends across the upstream and downstream sites.

Table 3. Substrate Measurements. A larger numeric value for size class indicates a larger cobble. Site 1 is the most downstream site; site 4 is the most upstream site.

Site		**1**	**2**	**3**	**4**
Distance from Left Bank	Left Bank	2m	0.2m	1m	1m
	Left Center	4m	1.1m	2m	2m
	Center	6m	2.2m	3m	3m
	Right Center	8m	3.3m	4m	4m
	Right Bank	10m	4m	5m	5m
Depth (cm)	Left Bank	NOT MEASURED	3cm	13cm	12cm
	Left Center	NOT MEASURED	10cm	8cm	14cm
	Center	NOT MEASURED	9cm	16cm	18cm
	Right Center	NOT MEASURED	5.5cm	8cm	14cm
	Right Bank	NOT MEASURED	3.5cm	3cm	6cm
mm/size class	Left Bank	45	Bedrock	90	64
	Left Center	32	4	100	90
	Center	90	8	90	64
	Right Center	64	<4	128	64
	Right Bank	45	32	128	32
% Cobble Embedded	Left Bank	50%	0	70%	10%
	Left Center	50%	0	70%	20%
	Center	20%	0	40%	10%
	Right Center	10%	0	10%	10%
	Right Bank	10%	40%	10%	20%

Coarse particulate organic matter is present throughout the whole stream and is most prominently present at Site 3. Microalgae is only present at sites 3 and 4. At Site 3, it was less than 1 mm in thickness but visible; at Site 4 it was not visible but the texture of microalgae was clearly apparent. Attached macroalgae is present only at Sites 1 and 3, and not at Sites 2 and 4. Unattached macroalgae is only present at Site 2, in minimal quantities. Macrophytes are not present at any site.

Shade Coverage Measurements

The shade coverage does not appear to differ between upstream and downstream sites (Table 4). The average shade coverage ranges between 16 and 18.25 dots covered at every site both upstream and downstream.

Table 4. Shade coverage. The values in this chart describe the dots on the densiometer that shade does cover. There were 24 total dots on the densiometer, so the number of uncovered dots is equal to (24 - number in chart). Site 1 is the most downstream site; site 4 is the most upstream site.

Site	Center Left	Center Upstream	Center Right	Center Downstream
1	23	16	19	16
2	20	17	19	18
3	16	17	22	19
4	14	15	12	11

River Slope

The river slope appears to become steeper moving from upstream of the site to downstream of the site (Table 5). However, the change is small in magnitude.

Table 5. River Slope. Site 1 is the most downstream site; site 4 is the most upstream site. The slope indicated is reported as: (upstream elevation - downstream elevation) / difference between the upstream and downstream locations of elevation measurements.

Site	1	2	3	4
Slope	NOT MEASURED	30cm / 20m	20cm / 20m	5cm / 20m

In-Stream Habitat Complexity

The following in-stream habitat traits do not demonstrate any differences across the different sites: filamentous algae, macrophytes, emergent vegetation, undercut banks, overhang vegetation, live tree roots, and artificial structures (Table 6). The presence of woody debris is significantly higher in sites 3 and 4.

Table 6. In-Stream Habitat Complexity. Site 1 is the most downstream site; site 4 is the most upstream site. I ranked each trait on a scale from 0-4. These values correspond to the following meanings: "0" stands for 0% coverage, "1" stands for less than 10% coverage, "2" stands for 10-40% coverage, "3" stands for 40-75% coverage, and "4" stands for more than 75% coverage.

Site	Filamentous algae	Aquatic Macrophytes / Emergent Vegetation	Boulders	Woody Debris >0.3 m	Woody Debris <0.3m	Undercut Banks	Overhang Vegetation	Live Tree Roots	Artificial Structures
1	2	1	1	1	1	3	3	1	0
2	0	0	1	1	1	2	1	0	1
3	2	-	1	2	3	4	1	1	1
4	1	1	1	2	2	2	1	0	0

Riparian Vegetation

There are notable differences between the bank vegetation of different sites. Site 4 has fewer tall trees and saplings than Sites 1-3; Site 1 has more medium vegetation than sites 2-4; Site 1 has more small woody shrubs and saplings than sites 2-4; Site 4 has fewer herbs and grasses than sites 1-3 (Table 7).

Table 7. Riparian Vegetation. Site 1 is the most downstream site; site 4 is the most upstream site. I ranked each trait on a scale from 0-4. These values correspond to the following meanings: "0" stands for 0% coverage, "1" stands for less than 10% coverage, "2" stands for 10-40% coverage, "3" stands for 40-75% coverage, and "4" stands for more than 75% coverage.

Site	Trees & Saplings >5m high		All vegetation 0.5-5m		Woody shrubs & saplings <0.5 m		Herbs/grasses		Barren, bare soil/duff	
	Left	Right	Left	Right	Left	Right	Left	Right	Left	Right
1	4	4	4	4	4	4	2	2	0	0
2	4	4	3	3	2	2	2	2	1	1
3	4	4	3	3	2	2	2	2	0	0
4	3	3	3	3	2	2	1	1	2	2

Bank Stability

Both left and right banks are "stable" at Sites 1-3. At Site 4, the left bank is "eroded" and the right bank is "stable."

Human Influence

The following human influence features are not present at any site: walls/rip-raps/dams, pipes, landfill/trash, parks/lawns, row crops, pasture, logging, mining, or orchards.

Buildings are present within 10-50 meters of one bank at both Site 2 and Site 4, but not present at Site 1 and Site 3. Pavement and roads are present within 10-50 meters of the right bank of Site 4, but not present at any of the other sites.

Vegetation management is present within 10 meters of both banks at Sites 1, 2, and 3, but there is no vegetation management present at Site 4.

Notable Field Conditions

In winter, the notable field conditions remain roughly the same between both upstream and downstream areas of the river.

Table 8. Notable Field Conditions. Site 1 is the most downstream site; site 4 is the most upstream site.

Site	Evidence of Recent Rainfall	Evidence of fires in reach or immediately upstream (<500m)	Dominant landuse/ landcover in area surrounding reach	Site affected by recent scouring event?	Channel engineered?
1	>10% flow increase	NO	Forest	NO	NO
2	Minimal	NO	Forest	YES	NO
3	>10% flow increase	NO	Forest	YES	NO
4	Minimal	NO	Forest	YES	NO

Ambient Water Quality Measurements

The winter temperature was 11.0 degrees Celsius. There was 105.7 % dissolved oxygen in winter. Specific conductivity was 0.2 in winter.

Biological Measurements

Family Measurements

I identified 42 distinct families across the samples taken at each site and a total of 2,408 organisms (Table 9). The highest abundances were in Chironomidae (610 larvae and 27 pupae), Gastropoda (494 Snails and 30 Limpets), Elmidae (70 M1 and 248 M2), Chloroperlidae (180), Ostracoda (135), Oligochaeta (134), Copepoda (85), Acari (61), and Simulidae (56 larvae and 1 pupae). Of these highly abundant families, three were not present in any amount at site 1: Elmidae, Simulidae, and Chloroperlidae. Site 2 accounted for the vast majority of Elmidae; 63 of the M1 Elmidae and 190 of the M2 Elmidae originated from Site 2.

Chironomidae larvae were present in high numbers at all sites, with substantially larger amounts at sites 3 and 4. Acari, Copepoda, Ceratopogonidae, Chironomidae, Gastropoda, Oligochaeta, and Ostracoda all appeared in every single sample, and while Ephemerellidae did not appear in every sample, they were present at every site. Across the sites, I identified at least seven distinct Ephemeroptera families, five Trichoptera families, and five Plecoptera families. The most abundant Plecoptera family was the Chloroperlidae family.

Table 9. Macroinvertebrate Family Identifications. Each value in the columns indicate a count of the number of those organisms found at that particular site. The notation (A) indicates adult organisms, (P) represents organisms in the pupae life phase, and all rows without either notation represent larval organisms. Site 1 is the most downstream site; site 4 is the most upstream site. I will add additional rows for each identification. Elmidae included two morphospecies noted here as M1 and M2.

Order	Family	1A	1B	2A	2B	3A	3B	4A	4B
Acari (A)		3	2	17	8	23	7	1	2
Amphipoda (A)		0	0	1	0	0	0	0	0
Cladocera		2	0	2	3	0	1	0	0
Coleoptera	**Elmidae M1**	0	0	48	15	5	6	0	1
Coleoptera	**Elmidae M2**	0	0	180	10	33	21	1	3
Coleoptera	**Hydrophilidae**	0	0	0	0	0	0	0	1
Coleoptera	**Amphizoidae (A)**	0	0	5	0	0	0	0	0
Collembola		0	1	0	0	0	0	0	0
Copepoda		9	9	3	21	33	6	4	2
Diptera	**Ephydridae**	0	1	0	1	2	0	0	5
Diptera	**Ceratopogonidae**	8	1	1	4	2	2	4	2
Diptera	**Chironomidae**	4	21	31	13	313	97	47	84
Diptera	**Chironomidae (P)**	0	0	0	0	12	6	1	8
Diptera	**Simulidae**	0	0	1	0	54	0	0	1
Diptera	**Simulidae (P)**	0	0	0	0	1	0	0	0
Diptera	**Empididae**	0	0	0	0	1	0	0	0
Diptera	**Dolichopodidae (P)**	0	0	0	0	1	0	0	0
Ephemeroptera	**Heptageniidae**	0	0	1	0	8	0	0	3
Ephemeroptera	**Siphlonuridae**	0	0	0	0	2	0	0	0
Ephemeroptera	**Leoptophlebiidae**	0	0	0	0	0	1	1	10

Table 9, continued

Order	Family	1A	1B	2A	2B	3A	3B	4A	4B
Ephemeroptera		1	0	0	0	0	0	0	0
Ephemeroptera	**Ephemerellidae**	2	0	3	3	3	4	1	14
Ephemeroptera	**Baetidae**	0	0	0	0	0	0	0	1
Ephemeroptera	**Leptohyphidae**	0	1	0	0	0	0	1	0
Gastropoda (A)	**(Limpets)**	0	1	8	1	7	1	0	2
Gastropoda (A)	**(Snails)**	3	22	373	85	6	3	0	2
Megaloptera	**Sialidae**	0	0	0	0	4	0	4	22
Nematoda (A)		0	0	0	0	1	0	0	0
Odonata	**Coenagrionidae**	0	0	0	1	0	0	0	1
Odonata	**Gomphidae**	0	0	3	0	0	0	0	0
Oligochaeta (A)		8	21	17	14	5	5	9	55
Ostracoda		19	12	10	52	10	11	4	17
Plecoptera	**Chloroperlidae**	0	0	7	2	126	29	2	14
Plecoptera	**Perlodidae**	0	0	3	0	11	2	0	2
Plecoptera	**Nemouridae**	0	0	7	1	4	1	1	6
Plecoptera	**M1**	0	0	10	18	9	0	0	0
Plecoptera	**Pteronarcyidae**	0	0	0	0	1	0	0	0
Trichoptera	**Glossosomatidae**	0	0	0	1	0	1	0	2
Trichoptera	**Lepidostomatidae**	0	0	12	0	2	12	1	13
Trichoptera	**Hydroptilidae**	0	0	0	0	0	1	0	0
Trichoptera	**Hydropsychidae**	0	0	0	0	3	0	0	0
Trichoptera	**Polycentropodidae**	0	0	0	2	7	0	0	0

In the NMDS ordination, most samples were situated close to the other sample at the same site, with the most distance within a site occurring at site 1 and site 2. Correlations between species scores and NMDS axes revealed that site 1 features Ostracoda (tolerance value 8) at higher abundance than the other sites (Figure 3). Site 2 features more Gastropods (tolerance value 7) and

Elmidae (tolerance value 4). Site 3 and Site 4 are similar to each other in terms of family composition, and both feature higher abundance of Chironomidae (tolerance value 6).

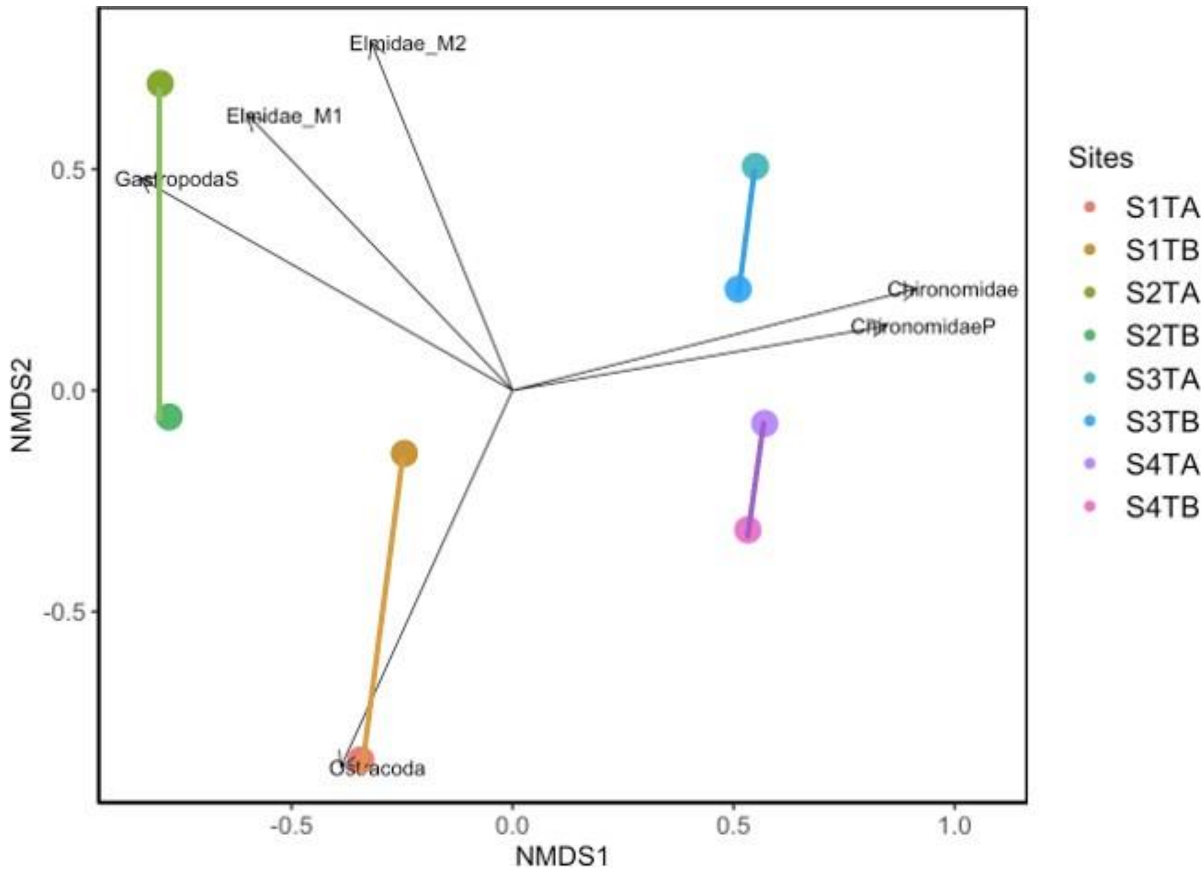

Figure 3. Family Trends NMDI Visualization. Arrows represent differences at a confidence interval above 95% statistical significance. Points representing samples from the same physical site are connected with a line. Site 1 is the most downstream site; site 4 is the most upstream site.

Biological Indices

I calculated six distinct biological indices (Table 10). For each metric, the Site 1 score is the poorest across all four sites. The CSCI, which measures disturbance on a scale from 0 (entirely altered stream habitat) to 1 (entirely intact stream habitat) is lowest at the downstream Site 1 (0.245) and highest at the upstream site Site 4 (0.63).

The family count (Richness) and EPT family counts (EPT Richness) are both highest at Site 3 (Richness of 23.5, EPT Richness of 9.5) and lowest at Site 1 (Richness of 10, EPT Richness of 1). All of the sites, including Site 1, do feature at least one family of the extremely pollution-sensitive EPT group. The two Site 4 samples resulted in markedly different results for percent EPT despite being drawn from the same physical area: one sample scored 8.33% and the other scored 23.81%. If the first sample was more accurate, Site 4's score most closely resembles Site 2 (average score of 7.52%) and, while higher than Site 1 (3.09%), is substantially lower than Site 3 (24.55%). However, if the second sample was more representative of Site 4, then Site 4 most closely resembles Site 3.

The Family Biotic Index (FBI) is relatively high for Site 1 (7.12), indicating that Site 1 has relatively pollution-tolerant taxa, while Site 3's score is much lower (4.80), indicating more sensitive taxa. Sites 2 and 4 have similar intermediary FBI scores, which correspond to a community sensitivity level in between Sites 1 and 3.

Table 10. Biological Indices. Site 1 is the most downstream site; site 4 is the most upstream site.

Site	1A	1B	2A	2B	3A	3B	4A	4B
CSCI Value	0.26	0.23	0.48	0.38	0.63	0.54	0.63	0.63
Richness	9	11	22	19	27	20	15	25
EPT Richness	1	1	7	6	11	8	6	9
% EPT	5.08%	1.09%	4.85%	10.19%	25.6%	23.50%	8.33%	23.81%
Family Biotic Index	7.09	7.14	5.64	6.31	4.82	4.77	5.90	5.27

The California Stream Condition Index Score (calculated using family-level identifications) increases along the length of the stream human activity gradient (Figure 4).

The CSCI consistently increases moving from the more human-influenced downstream (Site 1, mean CSCI of 0.25) to the more intact upstream (Site 4, mean CSCI of 0.63). The average CSCI across all sites is 0.4725.

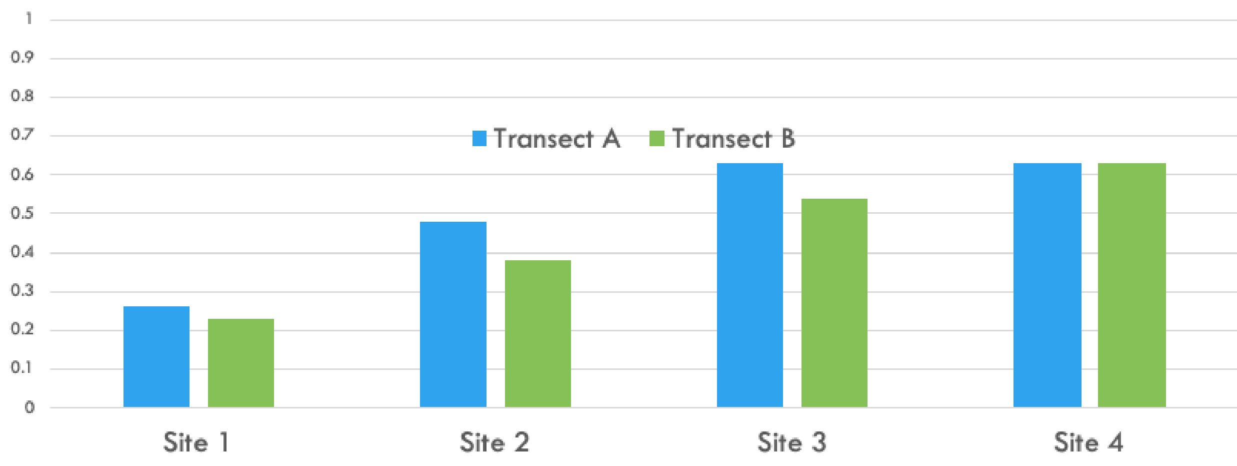

Figure 4. California Stream Condition Index. In the CSCI scoring system, a score of one indicates a fully intact stream, while a score of 0 indicates an extremely ecologically devastated stream. Site 1 is the most downstream site; site 4 is the most upstream site. The blue bars indicate values calculated from Transect A samples, and the green bars indicate values calculated from the Transect B samples.

The CSCI index consistently decreases moving downstream from more intact to more human-influenced areas. The smallest difference between sites is between sites 3 and 4, which differ by 0.045 units. The largest difference between sites is between sites 2 and 1, which differ by 0.19 units.

The Richness measures (both general family richness and EPT richness) both peak at Site 3 and reach the lowest values at Site 1 (Figure 5). Site 2 and Site 4 are similar in value and both fall in between the values observed at Site 1 and Site 4.

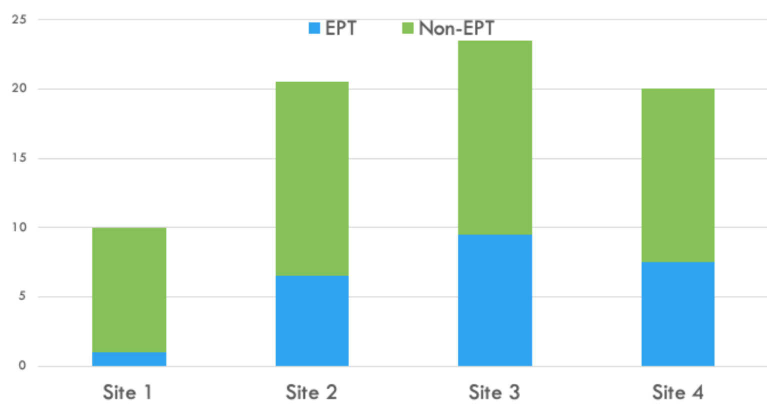

Figure 4. Family Richness. The blue section of the bars represent EPT families (Ephemeroptera, Plecoptera, and Trichoptera). The green section of the bars represent all non-EPT families. The entirety of the bar (both green and blue sections) represents the total richness of all family counts, both EPT and non-EPT alike. Site 1 is the most downstream site; site 4 is the most upstream site.

The Percent EPT, like the Richness metric, generally increases from downstream to upstream, with the exception of Site 4, which displays a decrease in Percent EPT compared to Site 3 (Figure 5).

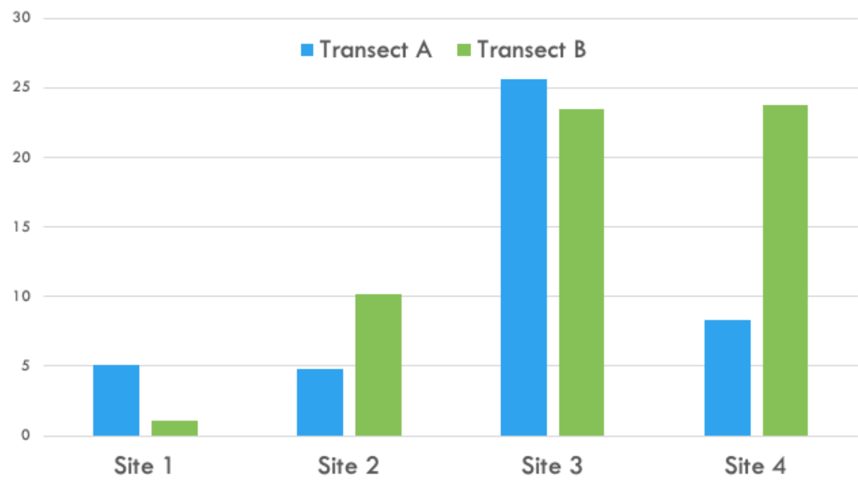

Figure 5. Percent EPT. Site 1 is the most downstream site; site 4 is the most upstream site. The blue bars indicate values calculated from Transect A samples, and the green bars indicate values calculated from the Transect B samples.

The Family Biotic Index (FBI) ranges from approximately 4.8 to 7.2 (Figure 6). It is highest at Site 1 and lowest at Site 3. Although Site 4 has a higher FBI than Site 3, it has a lower FBI than Site 2.

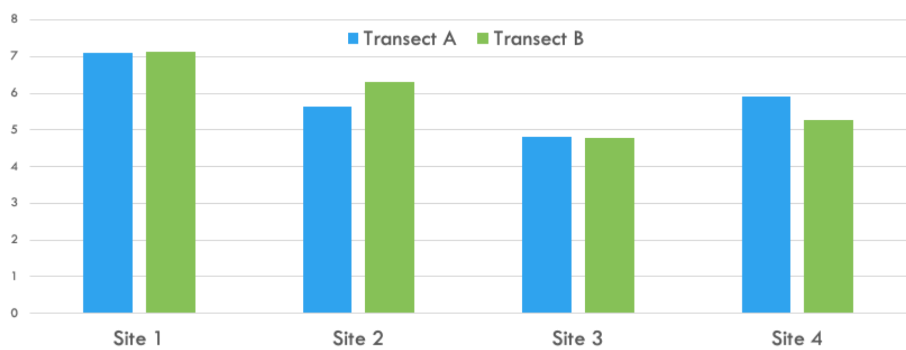

Figure 6. Family Biotic Index. Site 1 is the most downstream site; site 4 is the most upstream site. The blue bars indicate values calculated from Transect A samples, and the green bars indicate values calculated from the Transect B samples.

DISCUSSION

The index scores and physical habitat measurements are consistent with a negative impact from recreational campgrounds on riparian health. Physical habitat conditions do minimally vary across sites, even in regards to metrics that were likely not influenced by human recreational activity. Sites 3 and 4, the more upstream sites, have, to a minor degree, more variety in their in-stream physical habitat, which may be in part responsible for a healthier and more diverse fauna community at these sites. However, these natural variations in physical habitat likely do not fully account for the differences in fauna, richness, sensitivity, and overall community health. By all metrics, the most downstream site fauna was the least healthy while the more upstream sites were healthier. This discrepancy suggests that human recreational activity (including but not limited to camping, picnicking, and swimming) upstream of a given sampling site negatively affects the fauna community's health at all downstream locations.

Physical Habitat

On a few physical habitat metrics, Sites 3 and 4 appear more naturally prone to creating a diverse and healthy fauna community. Substrate size is significantly higher on average at Sites 3 and 4 than it is at Sites 1 and 2. Taxa richness and density of macroinvertebrates tends to increase in areas where sites have cobbles or stones, while it typically remains low where substrate is extremely small in size, like sandy areas (Duan et al. 2008). Macroinvertebrates tend to thrive in these areas with larger, more porous substrate, because this type of substrate offers more habitat in the form of nooks and crannies for them to live in (Duan et al. 2008). This difference suggests that Sites 3 and 4 might be naturally prone to be more diverse communities. However, the difference is small: Sites 1 and 2 also have non-sandy substrates and therefore also likely provide viable habitats for benthic macroinvertebrates, albeit probably fewer habitats in total.

Stream flow is another variable that noticeably differs across sites; as Site 1 has a stream flow of 0 and the site occurred upstream of a very large debris dam that likely backed up the water, slowing down the flow in the channel, while the other sites do measure a meaningful flow of water.

A low streamflow poses several possible problems pertaining to a river community's health: as the water slows down, it is able to carry less sediment, and so a sudden decrease in river speed can result in an unusually high buildup of sediment that decreases the amount of available habitat for benthic macroinvertebrates. Additionally, slower sections of river flow are less efficient at carrying away pollutants from the site, resulting in more pollutant loading ("Monitoring Our Rivers and Streams" n.d.). Therefore, the zero flow at Site 1 likely indicates that Site 1 will provide poor habitat quality and potentially higher levels of pollutant build-up than other sites along the river, both of which will likely result in less diverse and healthy macroinvertebrate communities.

It is not possible to definitively determine the impact on fauna health, but coarse particulate organic matter is most prominently present at Site 3, and woody debris is most prominently present at Sites 3 and 4. This trend might tend to create healthier sites upstream (Sites 3 and 4) because organic matter is a foodstuff for many different feeder types (notably collectors and shredders) that rely on a sufficient amount of organic matter within the stream, and large woody debris also offers habitat variety for organisms within the stream (Bundschuh and McKie 2016). However, excess CPOM can also undermine the health of stream ecosystems (Johnson et al. 2018). It is possible that the CPOM remains within healthy values across the entire stream, in which case Site 3 might benefit from larger amounts of CPOM being available as foodstuffs. It's also possible that the CPOM is high at Site 3 and would tend to create a less healthy environment for fauna at Site 3.

However, many other natural physical habitat metrics either do not differ between upstream and downstream sites or differ only slightly and not significantly, suggesting that without human intervention, the sites would be likely to cultivate similar fauna assemblages. Natural physical features including, but not limited to: cobble embeddedness, filamentous algae, macrophytes, emergent vegetation, undercut banks, overhang vegetation, live tree roots, and artificial structures vary slightly between sites, but are not significantly different between different sites. To the extent that these features contribute to fauna health, their impact on each of the sites would be similar, creating an inclination for the fauna communities to be similar at each site.

Benthic Macroinvertebrate Variation

The benthic macroinvertebrate results strongly support the hypothesis that recreational activity upstream of riparian communities negatively affects the benthic macroinvertebrate communities. Lower species diversity, richness, and pollution sensitivity are well-established to correlate with imperiled streams (Walsh et al. 2005). Family richness, EPT Richness, Percent EPT, FBI, and the CSCI measurements all reflect lower diversity levels and higher pollution tolerance at the most downstream site (Site 1) and higher diversity levels and less pollution tolerance at more upstream sites, suggesting that these sites are likely less disturbed and feature healthier or more intact fauna communities. Although there is not a precedent in the literature specifically addressing the question of recreational activity influence on streams, the result found here is similar to the result found related, but not identical studies of the impact of urban anthropogenic activities on stream banks. Stream community health is well-established to decrease when urban anthropogenic activity increases along the stream banks (Schoonover et al. 2005, Walsh et al. 2005, Booth et al. 2015).

Similarly, the results here suggest that higher levels of anthropogenic activity in non-fully urbanized but recreational areas also contributes to stream degradation and less healthy fauna communities. The upstream sites (Sites 3 and 4) have notably higher levels of "clean water" taxa compared to Sites 1 and 2, specifically Ephemeroptera, Plecoptera, and Trichoptera, and the presence of these sensitive species suggests that the pollution levels at these sites is lower. The EPT Richness is, on average, 3.75 at Sites 1 and 2 (and only 1 at Site 1) while on average Sites 3 and 4 have 7.75 EPT families, more than double the average amount at the downstream sites. All four sites have at least some moderately or very tolerant taxa: for example, Oligachaeta, with a tolerance value of 8, is present at every site. However, Site 1 has higher amounts of tolerant taxa, with an average weighted tolerance value of 7.12 across all families, while no other site has a weighted tolerance value above 6, also suggesting that Site 1 may feature higher levels of pollution and disturbance. However, although the river was partially impaired, across all sites at least one EPT family is present and very few, if any, extremely tolerant families (with a tolerance value of 9 or 10) appear. These results suggest that the stream is at least fairly healthy, especially at the most upstream sites. Similarly, the average CSCI across all sites, 0.47, equates to a "very likely altered" score, suggesting that the anthropogenic activity has had an impact, but prior CSCI calculations at different seasonal times resulted in a score of 0.93, indicating a "likely intact" environment (*Integrated Monitoring Report Part B: Creek Status Monitoring* 2020). These results suggest that the stream is altered, but still able to support moderately healthy benthic macroinvertebrate communities. This result seems reasonable with regards to the literature.

Physical Habitat & Fauna Interactions

The natural physical environment likely does contribute to the variation in fauna communities, even without considering the anthropogenic effects from recreational activities. In particular, Sites 3 and 4 show some characteristics that indicate their physical environment is naturally more favorable than Sites 1 and 2 along metrics that are unlikely to be strongly affected by typical recreational activities. For example, Sites 3 and 4 feature larger substrate sizes, which is likely to positively affect benthic macroinvertebrate communities (Duan et al. 2008) but which logically is not likely to vary significantly based on people's camping or swimming activities. However, along many physical metrics these sites are very similar aside from human activities, suggesting that the natural physical variation is not likely to account for all of the differences in fauna communities. In particular, Sites 1 and 2 scored very similarly in regards to physical habitat, even when Site 3 and Site 4 demonstrated more favorable physical characteristics. However, Site 2 was consistently more highly scored along fauna community indices despite very similar physical characteristics. This difference strongly suggests that the anthropogenic activities did indeed contribute to the differences in fauna scores across each site.

Limitations and Future Directions

This study was limited in scope to be able to accomplish it in one year: I identified only approximately 2,400 organisms, and only to a family level. I was only able to take two samples at each of four transects instead of the 10-transects taken by SWAMP protocols which results in a higher number of samples and organisms to better capture the variability in the stream reach. Additionally, I only identified organisms to the family level, while the ideal CSCI works based off a higher precision of identification at the genus or species-level identifications. Therefore, results may be more variable than if I had been able to collect a higher volume of samples or to identify organisms on a more detailed level. Furthermore, I only sampled in November of 2022 which was later than biomonitoring samples have historically been taken in Pescadero Creek. It is quite possible that the community fauna integrity varies throughout the year. In August of 2019, researchers measured the CSCI of Pescadero Creek to be 0.93, much higher than my reported value of approximately 0.47 (*Integrated Monitoring Report Part B: Creek Status Monitoring* 2020).

This discrepancy might represent an actual change in the stream health, indicate flaws in my analysis, or it might reflect differences in stream health throughout the year, particularly given that the streamflow was much lower in November of 2022 than it was in August of 2019 ("Pescadero C NR Pescadero CA" n.d.). Although I had planned to take samples in March 2023, California had record rainfalls and streamflows were much too high to safely sample.

This area of research offers numerous routes for future study. It would be advisable to conduct additional sampling to simply increase the volume of data available to analyze both in terms of sites, samples, and organisms collected. Additionally, assessing the stream at multiple various times of year would address concerns regarding seasonal variation in creek health. Conducting additional studies at different parks could help to address the external validity concern about whether or not this creek is indeed representative of all other campgrounds. It would also be advisable to intentionally select sampling sites with physical similarities. Finally, randomized experiments are the gold standard for addressing causality. If possible, a randomized controlled experiment regarding recreational activity could be conducted to more clearly establish causality between recreation and stream degradation.

Broader Implications

These results suggest that there are important policy considerations regarding recreational activity in parks because the results imply that recreational activity is detrimental for streams. Therefore, when managing parks, permitting recreational activity can undermine the conservation missions that frequently motivate these parks' creation. Recreational activities often do have positive effects as well, which cannot be discounted: they can provide important sources of revenue to maintain the parks and they can motivate people to participate in park preservation and maintenance by increasing their feeling of connection to the parks. However, the stream degradation resulting from recreational activity also has a strong negative trade-off, which should not be ignored when establishing limits and rules surrounding park activities. Streams are very vulnerable to negative impacts as well as serving as important components of the overall global ecosystem, and establishing a better understanding of the factors that drive rivers to be more or less healthy will allow us to protect these crucial ecosystems.

ACKNOWLEDGEMENTS

Thank you to the San Mateo County Park system for allowing us to access Pescadero Creek for sampling, and to Emma Kroencke for processing our application for a sampling permit (permit number SP-2022-15-MCP). Thank you to all the park rangers at Memorial Park for their assistance in locating our sampling locations, and to Park Ranger David Vasquez for coordinating the scheduling of our sampling days. Thank you to my thesis advisor, Dr. Patina Mendez, for helping me to plan this study, carry out sampling procedures, sample processing and identification, and data analysis. Thank you to Kevin Lunde, Kristina Yoshida, and Raphael Mazor for their advice on sampling and sample analysis. Thank you to my friends, family, and loved ones for supporting me throughout the process of writing this thesis and for attending my thesis presentation.

REFERENCES

Barnett, A., N. L. Payne, J. M. Semmens, and R. Fitzpatrick. 2016. Ecotourism increases the field metabolic rate of whitetip reef sharks. Biological Conservation 199:132–136.

Booth, D., A. Roy, B. Smith, and K. Capps. 2015. Global perspectives on the urban stream syndrome. Freshwater Science 35:412–420.

Bundschuh, M., and B. G. McKie. 2016. An ecological and ecotoxicological perspective on fine particulate organic matter in streams. Freshwater Biology 61:2063–2074.

Butler, B., R. G. Pearson, and R. A. Birtles. 2021. Water-quality and ecosystem impacts of recreation in streams: Monitoring and management. Environmental Challenges 5:100328.

Derlet, R. W., J. R. Carlson, and M. N. Noponen. 2004. Coliform and Pathologic Bacteria in Sierra Nevada National Forest Wilderness Area Lakes and Streams. Wilderness & Environmental Medicine 15:245–249.

Draft California 2018 Integrated Report (303(d) List/305(b) Report). 2018. . Regional Board 2 - San Francisco Bay Region.

Duan, X., Z. Wang, and S. Tian. 2008. Effect of streambed substrate on macroinvertebrate biodiversity. Frontiers of Environmental Science & Engineering in China 2:122–128.

Eagleston, H., and J. L. Marion. 2017. Sustainable campsite management in protected areas: A study of long-term ecological changes on campsites in the boundary waters canoe area wilderness, Minnesota, USA. Journal for Nature Conservation 37:73–82.

Family Level Index. (n.d.). . https://sccwrp.shinyapps.io/familyindex/.

Farrell, T., and J. Marion. 1998. An Evaluation of Camping Impacts and their Management at Isle Royale National Park.

Flack, J. E., A. J. Medine, and K. J. Hansen-Bristow. 1988. Stream Water Quality in a Mountain Recreation Area. Mountain Research and Development 8:11–22.

Gary, H. L. 1982. Stream water quality in a small commercial campground in Colorado. Journal of Environmental Health 45:10.

Harrington, J., and M. Born. 1999. Measuring the Health of California Streams and Rivers. Second edition. Sustainable Land Stewardship International Institute.

Hession, W., R. Horowitz, J. Pizzuto, and T. Johnson. 2003. Influence of Bank Vegetation on Channel Morphology in Rural and Urban Watersheds. Geology 31:147–150.

Integrated Monitoring Report Part B: Creek Status Monitoring. 2020. . Pages 1–208. San Mateo Countywide Water Pollution Prevention Program.

Jackson, J., and L. Füreder. 2006. Long-Term Studies of Freshwater Macroinvertebrates: A Review of the Frequency, Duration and Ecological Significance. Freshwater Biology 51:591–603.

Johnson, E. R., S. Inamdar, J. Kan, and R. Vargas. 2018. Particulate Organic Matter Composition in Stream Runoff Following Large Storms: Role of POM Sources, Particle Size, and Event Characteristics. Journal of Geophysical Research: Biogeosciences 123:660–675.

Kebede, G., D. Mushi, R. B. Linke, O. Dereje, A. Lakew, D. S. Hayes, A. H. Farnleitner, and W. Graf. 2020. Macroinvertebrate indices versus microbial fecal pollution characteristics for water quality monitoring reveals contrasting results for an Ethiopian river. Ecological Indicators 108:105733.

Kelderman, K., A. Phillips, T. Pelton, E. Schaeffer, P. MacGillis-Falcon, and C. Bernhardt. 2022. The Clean Water Act at 50: Promises Half Kept at the Half-Century Mark. Environmental Integrity Project.

Marion, J. 2003. Camping Impact Management on the Appalachian National Scenic Trail Camping Impact Management on the Appalachian National Scenic Trail.

Marion, J. L., Y.-F. Leung, H. Eagleston, and K. Burroughs. 2016. A Review and Synthesis of Recreation Ecology Research Findings on Visitor Impacts to Wilderness and Protected Natural Areas. Journal of Forestry 114:352–362.

Marion, J. L., and T. Sober. 1987. Environmental Impact Management in the Boundary Waters Canoe Area Wilderness. Northern Journal of Applied Forestry 4:7–10.

Merrit, R.W., Cummins, K.W. and Berg, M.B. (2008) An Introduction to Aquatic Insects of North America. 4th Edition, Kendall Hunt Publishers, Dubuque.

Monitoring Our Rivers and Streams. (n.d.). . https://pubs.usgs.gov/fs/fs-077-02/.

Oksanen, J., G. Simpson, G. Blanchet, R. Kindt, P. Legendre, P. Minchin, R. O'Hara, P. Solymos, M. H. Stevens, E. Szoecs, H. Wagner, M. Barbour, M. Bedward, B. Bolker, D. Borcard, G. Garvalho, M. Chirico, M. De Caceres, S. Durand, H. B. Antoniazi Evangelista, R. FitzJohn, M. Friendly, B. Furneaux, G. Hannigan, M. O. Hill, L. Lahti, D. McGlinn, M.-H. Ouellete, E. Ribeiro Cunha, T. Smith, A. Stier, C. Ter Braak, and J. Weedon. 2022. vegan: Community Ecology Package.

Oliveira, A., and M. Callisto. 2010. Benthic macroinvertebrates as bioindicators of water quality in an Atlantic forest fragment. Iheringia 100:291–300.

Paetzold, A., M. Smith, P. Warren, and L. Maltby. 2011. Environmental impact propagated by cross-system subsidy: chronic stream pollution controls riparian spider populations. Ecology 92:1711–6.

Pescadero C NR Pescadero CA. (n.d.). . https://waterdata.usgs.gov/monitoring-location/11162500/.

Pescadero Creek Park Natural Features | County of San Mateo, CA. (n.d.). https://www.smcgov.org/parks/pescadero-creek-park-natural-features.

Pinto, U., B. L. Maheshwari, and E. C. Morris. 2014. Understanding the Relationships among Phytoplankton, Benthic Macroinvertebrates, and Water Quality Variables in Peri-Urban River Systems. Water Environment Research 86:2279–2293.

Rasmussen, J., and A. Ricciardi. 1999. Extinction rates of North American freshwater fauna. Conservation Biology 13:1220–1222.

RStudio Team. 2020. RStudio: Integrated Development for R. RStudio, PBC, Boston, MA.

Schoonover, J., B. G. Lockaby, and S. Pan. 2005. Changes in Chemical and Physical Properties of Stream Water Across an Urban-Rural Gradient in Western Georgia. Urban Ecosystems 8:107–124.

Staff. 2021, March 3. Improved Memorial Park to welcome back campers in 2021.

Stella, J., and J. Bendix. 2019. Multiple Stressors in Riparian Ecosystems. Pages 81–110.

SWAMP - Standard Operating Procedures | California State Water Resources Control Board. (n.d.). . https://www.waterboards.ca.gov/water_issues/programs/swamp/bioassessment/sops.html.

Tampo, L., I. Kaboré, E. H. Alhassan, A. Ouéda, L. M. Bawa, and G. Djaneye-Boundjou. 2021. Benthic Macroinvertebrates as Ecological Indicators: Their Sensitivity to the Water Quality and Human Disturbances in a Tropical River. Frontiers in Water 3.

Walsh, C. J., A. Roy, J. Feminella, P. Cottingham, P. Groffman, and R. Morgan II. 2005. The Urban Stream Syndrome: Current Knowledge and the Search For A Cure. Am. Benthol. Soc 24:706–723.

Wang, L., and P. Kanehl. 2003. Influences of Watershed Urbanization and Instream Habitat on Macroinvertebrates in Cold Water Streams. Journal of the American Water Resources Association 39:1181–1196.

Wasowski, R., D. Alexander, S. Kolmes, N. Banet, A. Card, M. Haines, E. Kolmes, K. Northcutt, D. Roemer, and S. Webber. 2013. A Multi-Year Longitudinal Study of Water Quality Parameters in Four Salmon-Bearing and Recreational Streams on Mount Hood, Oregon. PLOS ONE 8:e70453.