Ecological Responses of Benthic Macroinvertebrate Communities to Streamflow Augmentation in Strawberry Creek, Berkeley, California

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ABSTRACT

Benthic macroinvertebrates in Mediterranean-climate streams undergo substantial stresses from seasonal and annual variation in streamflow, and these stresses can be magnified by urbanization. These organisms are especially susceptible to anthropogenic stresses during the low-flow period of the year. Streamflow augmentation is one potential solution to combat this water-stress problem; however, few studies have examined streamflow augmentation or collected ecological information on augmented flow. This study examines the responses of the benthic macroinvertebrate community to augmented streamflow at 3 sites in Strawberry Creek, an urban Mediterranean stream in Berkeley, California. I hypothesized that (1) benthic community diversity would increase, (2) water quality would improve, and (3) filterers and gatherers would benefit from the augmented flow. I installed control and augmented-flow treatments in the stream using plywood boards and used a variety of bioassessment metrics to evaluate the responses, including total abundance, taxa richness, EPT richness, and percent filterers and gatherers. I used a t-test and ANOVA to test for differences in streamflow and bioassessment metrics between control and augmented-flow treatments, and did non-metric multidimensional scaling (NMS) and linear regression analyses to examine the relationship between flow and bioassessment metrics. I found no significant difference in streamflow and metric values between control and augmented-flow treatments. However, I observed a trend that suggests an improvement in water quality and increase in filterer-collector population at augmented-flow treatments. As effective water management is critical to maintain aquatic ecosystem integrity, this study informs future studies to conduct flow augmentation on more urbanized and disturbed streams.

KEYWORDS

water stress, urbanization, habitat quality, Mediterranean climate, bioassessment

INTRODUCTION

Many factors affect the health of benthic macroinvertebrate communities in streams including, but not limited to, water quality and temperature, riparian vegetation, and streamflow (Dewson et al. 2007). For example, introductions of both chemical pollutants and invasive riparian vegetation often cause general declines in benthic-macroinvertebrate community diversity and species abundances (Pesek and Hergenrader 1976, Rios and Bailey 2006). Water-temperature changes, which affect the amount of dissolved oxygen available, can result in shifts in community composition towards warm-water or cold-water specialists (Lessard and Hayes 2003). Likewise, decreases in streamflow can have similarly detrimental effects. Flow volume directly affects substrate composition, the level of turbulence, and the delivery rate of dissolved ions and particulate matters downstream (Hart and Finelli 1999, James et al. 2009). As a result, streamflow plays a very important role in maintaining stream ecosystems, especially in streams in Mediterranean climate regions where flow volumes can be highly variable by season, leading to drastic changes in habitat and food availability (Gasith and Resh 1999).

In Mediterranean-climate streams, the lowest flows usually occur during the hot, dry summer months, whereas peak flows typically occur during the cold, rainy winter. These low and high flows impose great stresses on the invertebrate community (Suren and Jowett 2006). Low flows can cause a decline in habitat diversity, water quality, and food resources (Gasith and Resh 1999, Walters and Post 2011), which can result in reductions in macroinvertebrate species richness (McIntosh et al. 2002, Stubbington et al. 2009) and drift to areas of higher flow (Kohler 1985, James et al. 2009). Alternatively, large floods can produce high shear stress, which can reduce macroinvertebrate biomass, taxa richness and population density (Herbst and Cooper 2010, Siegfried and Knight 1977, Suren and Jowett 2006). A seasonally appropriate flow volume, even in highly variable Mediterranean-climate streams, is therefore essential to protect ecological integrity.

To improve the environmental conditions for aquatic invertebrates during unnaturally low flow periods that may result from dam operations, inter-basin water diversions, groundwater abstractions, or other forms of hydrological alteration in urban areas, managed streamflow augmentation has been implemented in some cases (Matlock et al. 2000). Because streamflow in these anthropogenically modified environments is not sufficient during these periods, the addition of water to the stream can help relieve the stress on benthic communities (Gasith and Resh 1999). For instance, flow augmentation can improve water quality by increasing dissolved oxygen concentration through increased turbulence (Matlock et al. 2000) and by reducing concentrations of pollutants through dilution (Gasith and Resh 1999). Flow augmentation is also associated with an increase in riparian vegetation due to more water availability and accrued benefits for specific functional feeding groups such as filterers and gatherers (Ponce and Lindquist 1990). Despite these findings, it is still not well understood how flow augmentation application benefits ecosystems because in most of the cases in which it has been done very little ecological information was collected before and after the application (McIntosh et al. 2002).

This study examines the ecological effects of streamflow augmentation using natural stream water on the community of benthic macroinvertebrates in Strawberry Creek, an urban Mediterranean stream located in Berkeley, California. The streamflow augmentation was performed during the most water-stressed period of the year when the ecological benefits are expected to be most pronounced. I hypothesized that: (1) community diversity would increase (e.g., increase in taxa richness), (2) water quality would improve (e.g., increase in percent Ephemeroptera, Plecoptera and Trichoptera (EPT) richness), and that (3) certain functional feeding groups would benefit from the flow augmentation more than others (e.g., increase in filterer and gatherer population). These results will have important implications for water resources management in urban areas where streamflow augmentations are often done, typically using treated wastewater, with little knowledge of the effects that these augmentations might have on the biota.

METHODS

Study site

Strawberry Creek is a small urban stream that flows through Strawberry Canyon in Berkeley, California which has a total watershed area of about 4.1 square kilometers. The stream is approximately 8 kilometers in length, and its width varies between 1 meter and 4 meters. Starting at its headwaters in the Berkeley Hills (37°52’ N; 122°15’W), Strawberry Creek flows through the University of California at Berkeley campus and the City of Berkeley draining into the San Francisco Bay estuary (Charbonneau 1987). The Strawberry Canyon watershed is 40% urbanized (i.e. covered with impervious surfaces), and urbanization has had a profound impact on the hydrologic regime of Strawberry Creek. Impervious surface, stream culverting and channel confinement have altered the natural flow regime of the stream, resulting in unnaturally flashy floods during winter rains, bank erosion and destruction of aquatic habitat (Charbonneau 1987). Non-natural contributions to the streamflow, besides from storm water and groundwater, include municipal discharge and landscape-irrigation runoff. The Strawberry Creek ecosystem now supports 5 native fish species, assemblages of benthic macroinvertebrates, macrophytes, periphyton and a wide range of riparian vegetation along different sections of the stream including non-native eucalyptus (*Eucalyptus globulus*), Monterey pine (*Pinus radiata*) and redwood trees (*Sequoia sempervirens*), and invasive shrubs and forbs such as English ivy (*Hedera helix*) (Hans and Maranzana 2006).

Experimental design

To establish the collection sites, we (Dr. Justin Lawrence and I) selected three locations along the less urbanized upper reach of Strawberry Creek along the fire trail in Strawberry Canyon, just below the University Botanical Gardens and above the retention dams (Figure 1). The section of stream under study had an average width and water depth of 2m and 6 cm, respectively, had thick riparian vegetation cover and was dominated by cobbles and boulders as substrate.

We installed two treatments at each location: one procedure-wise control (unaugmented flow) and one impact (augmented flow). The control treatment consisted of a pair of thick two-by-one-half-meter plywood boards installed upright using one-meter long rebar in the stream, with the long sides oriented in the direction of flow and separated by half a meter (Figure 2A, C). Water was able to flow normally through and around this treatment. The impact treatment had the same design as the control except the upstream end included 0.5-m plywood board extensions that were angled at 45^{o} to direct the entire flow into the area between the boards and the outer sides of the wood boards facing the banks were completely filled with sediment and rocks to create two “No Flow” zones outside of the new augmented-flow channel (Figure 2B, D).

We positioned all the control treatments approximately 10 meters upstream from the impact treatments.

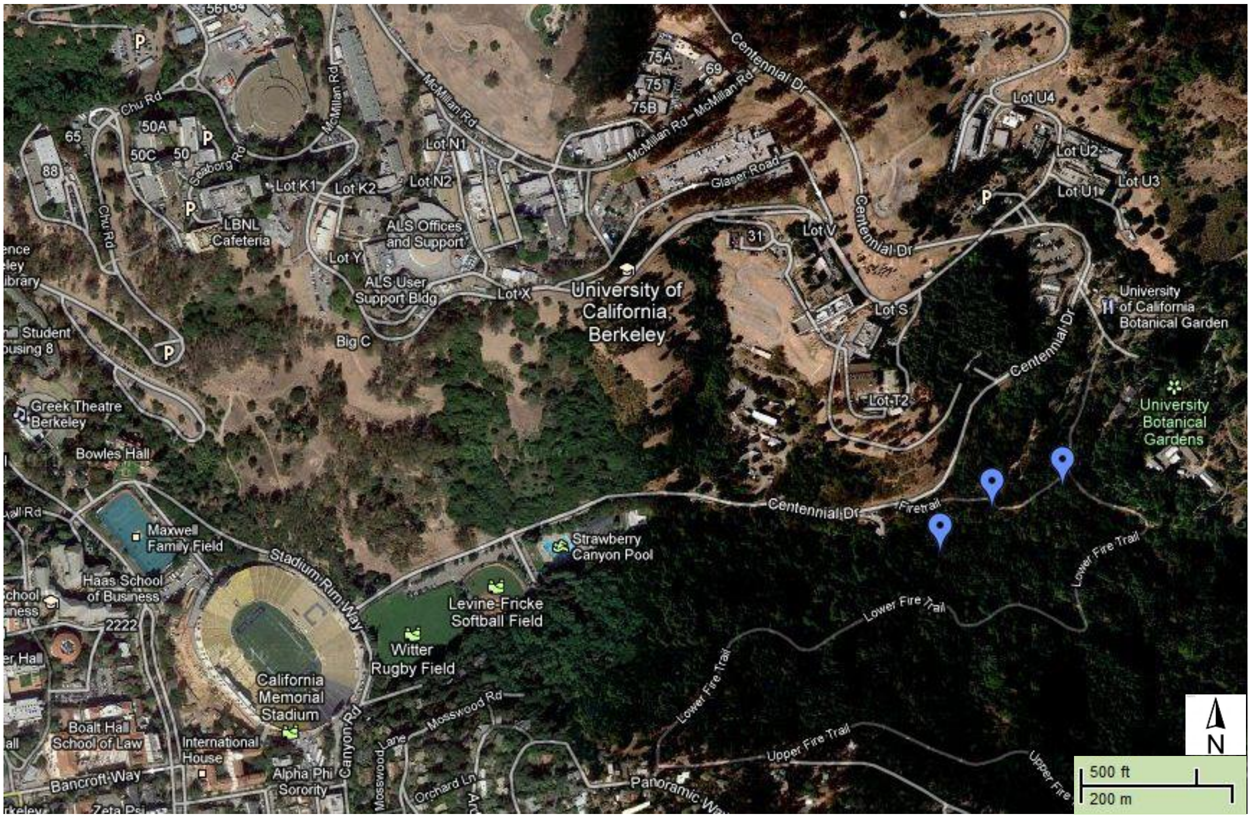

Figure 1. Sampling collection sites. The three sites (in blue) were located in Strawberry Creek along the Fire Trail in the Strawberry Canyon, below the University Botanical Gardens and above the retention dams.

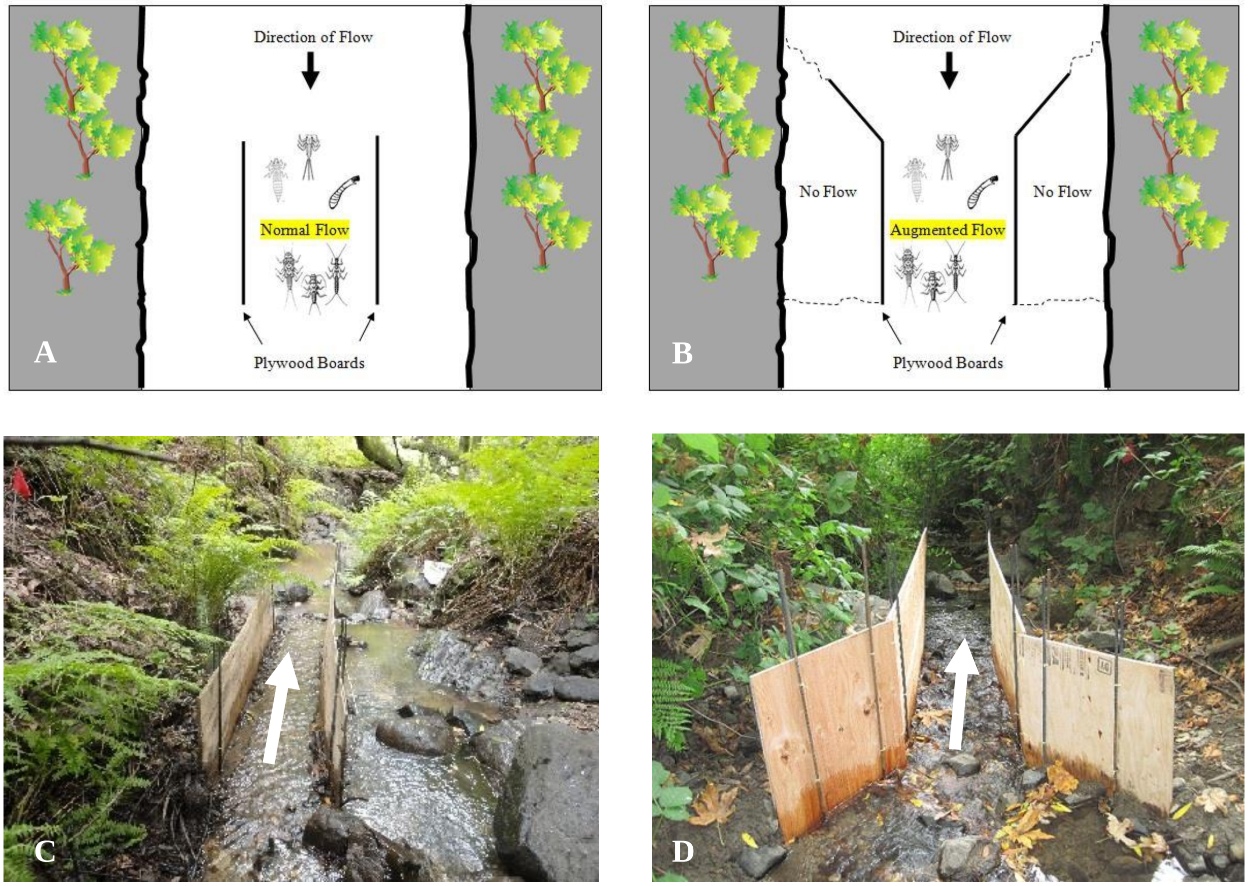

Figure 2. Schematic diagram (top view) of the (A) control treatment and (B) impact treatment; Photo image (looking downstream) of the (C) control treatment and (D) impact treatment. The dotted line in the impact treatment diagram refers to the boundary of the "No Flow" zones after the filling of sediments and rocks. The white arrows in the photo image of both control and impact treatments refer to the direction of flow.

Data collection

Data collection consisted of monthly biological sampling and stream physical measurements of the collection sites in the summer and fall of 2011. We collected benthic macroinvertebrate samples three times: one collection on August 20 before the treatment had been installed (baseline collection) and two collections afterward, one on September 15 (September collection) and the other on October 13 (October collection). At each sampling, we also measured stream depth (m) and flow velocity (m/s).

To adhere to standard bioassessment practices and ensure precision and accuracy of each collection, we followed the guidelines from the U.S. EPA's Rapid Bioassessment Protocol as closely as possible (Barbour et al. 1999); however, some modifications were necessary to accommodate the experimental objectives. We sampled benthic macroinvertebrates at three random locations in the region between the plywood boards at each treatment starting from downstream to upstream using a D-frame net with a 0.5mm mesh. At each location, we disturbed the substrate for exactly one minute by using hands or kicking the rocks to force organisms to drift into the net. Next, we put all three macroinvertebrate collections for each treatment into a single Ziploc bag to create one cumulative sample. We poured 95% ethanol into the Ziploc bag with the macroinvertebrate sample at the field site to preserve the organisms. At the end of each sampling period, we had a total of six benthic macroinvertebrate samples from all three sites with two treatments each.

To facilitate sorting of the organisms at the laboratory, I transferred the macroinvertebrate sample to a flat pan, drained the alcohol from the sample and added water back into it. This procedure allowed the organisms to float to the surface aiding in the sorting process. After sorting, I used the standardized taxonomic identification key by Harrington and Born (2003), McCafferty (1981) and Merritt et al. (2008) to identify the collected organisms to the family level and recorded their identifications and abundance into a database.

To quantify the difference in habitat quality, water quality and benthic macroinvertebrate community diversity and composition, I calculated fifteen bioassessment metrics from the EPA's Rapid Bioassessment Protocol including, but not limited to, total abundance, taxa richness, percent EPT, different percent functional feeding groups and percent Chironomidae individuals for each sample (Barbour et al. 1999). I excluded the non-insect organisms (i.e. aquatic earthworms, snails and scuds), which are less sensitive to disturbance or pollution, from the calculation of these metrics. Taxa richness is useful in determining the diversity and composition of a benthic macroinvertebrate community and is calculated by counting the number of genera present in a given sample (Barbour et al. 1999). Higher taxa richness in augmented-flow samples indicates that the habitat after treatment can support a more diverse assemblage of benthic macroinvertebrates, and thus flow augmentation can be considered as beneficial to the organisms. In addition, percent EPT can efficiently reflect water and habitat quality and is calculated by dividing the number of EPT organisms by the number of total macroinvertebrate organisms in a sample (Barbour et al. 1999). Benthic macroinvertebrates in the orders of Ephemeroptera, Plecoptera and Trichoptera (EPT) are known to be very sensitive to pollution and disturbance (Barbour et al. 1999), and thus the absence of EPT organisms in augmented-flow samples would indicate that the treatment caused measurable stress to the organisms.

Furthermore, percent functional feeding groups can be effectively used to determine how the food web changes after the flow augmentation and which feeding groups benefit the most from this application. Percent functional feeding group is calculated by dividing the number of benthic macroinvertebrates in each feeding group (e.g., predators, filterers, gatherers, scrapers and shredders) by the total number of macroinvertebrates organisms in a sample. As this study implements a Before After Control Impact (BACI) design, I calculated a difference in metric values between the control and impact treatments for each sample and assumed this difference to be as a result of the augmented flow.

Statistical analysis

I used a t-test to detect a significant difference in streamflow parameters (i.e. flow velocity and stream depth) between the control and impact treatments for each sample (i.e. whether the impact treatments significantly increased streamflow as expected). Additionally, I used a one-way ANOVA to determine whether the difference in metric values between the control and impact treatments is significantly higher in the September and October collections compared to the baseline (i.e. whether the metric values increased significantly in impact treatments compared to the values at baseline).

To examine the independence among treatments, collection times, and sites, I used non-metric multidimensional scaling analysis (NMS) on the $\log_{10}(n+1)$ taxa abundances of all taxa. I used the PC-ORD 5.10 software to obtain a 2-Dimensional solution based on Sørenson distance (McCune and Mefford 1999). I examined clustering among species in ordination space in relation to the categorical variables such as treatment, stream, order, and perenniality, and the continuous variables such as flow velocity and stream depth. I ran the NMS with 4 axes, 50 runs with real data using a stability criterion of 0.0001, 15 iterations to evaluate stability, and a maximum number of iterations of 250.

I did a linear regression analysis to determine if there was a relationship between physical flow parameters (i.e. flow velocity and stream depth) and the bioassessment metrics (i.e. taxa richness, percent EPT richness and percent filterers and gatherers). I plotted the physical parameters with respect to each of the three bioassessment metrics and determined the coefficient of determination to see if the relationship was significant.

RESULTS

Streamflow measurements

I found that all the sampling sites were similar in streamflow at baseline (Table 1). At control treatments flow velocity ranged between 0.190 and 0.248 m/s, and stream depth ranged between 0.043 and 0.060 m across the three collection times. At impact treatments flow velocity ranged between 0.250 and 0.407 m/s while stream depth ranged between 0.057 and 0.067 m. Additionally, I found that flow velocity and stream depth were not significantly different between the control and impact treatments. However, I observed a trend that suggests site-specific increases in flow velocity and stream depth observed across time.

Bioassessment metrics

I collected a total of 9,085 individual organisms representing 10 orders and 61 families of benthic macroinvertebrates. The most common orders observed were Diptera, Plecoptera, Ephemeroptera, and Trichoptera while the dominant families were Simulidae, Nemouridae, Chironomidae, Leptophlebiodae and Baetidae (in order of decreasing abundance). Rare taxa observed in less than 1% of the sample included Lepidoptera, Megaloptera and Collembola. The dominant functional feeding groups were filterers and gatherers.

I did not find any significant difference in bioassessment metrics between the control and impact treatments at all three sites across time evident by the large p-values (Table 2). However, I observed a directional non-significant trend in average percent EPT richness (i.e. averaged from individual metric values at all three sites) and average percent filterers and gatherers at impact treatments across time. Specifically, the average percent EPT richness was non-significantly elevated over baseline in September and October at impact treatments ($p = 0.327$). In contrast, the change in average percent EPT richness at impact treatments was minimal (Figure 3). Similarly, the average percent filterers and gatherers at impact treatments was non-significantly elevated over baseline in September and October ($p = 0.466$) while there was no trend for the control treatments (Figure 4).

Table 1. Physical measurements of sampling sites. I measured flow velocity (m/s) and water depth (m) at every sampling.

Collection	Site	Flow Velocity (m/s)		p-value	Stream Depth (m)		p-value
		Control	Impact		Control	Impact	
Baseline	1	0.232	0.237†		0.057	0.057	
	2	0.248	0.250		0.047	0.057	
	3	0.237	0.262		0.057	0.057	
	Mean*	**0.239 ± 0.012**	**0.249 ± 0.014**	**0.401**	**0.053 ± 0.009**	**0.057 ± 0.005**	**0.538**
September	1	0.238	0.247		0.053	0.067	
	2	0.207	0.345		0.043	0.047	
	3	0.190	0.407		0.053	0.057	
	Mean	**0.212 ± 0.042**	**0.333 ± 0.122**	**0.180**	**0.050 ± 0.009**	**0.057 ± 0.021**	**0.624**
October	1	0.218	0.353		0.060	0.057	
	2	0.197	0.317		0.053	0.057	
	3	0.242	0.252		0.053	0.063	
	Mean	**0.219 ± 0.040**	**0.307 ± 0.116**	**0.282**	**0.056 ± 0.007**	**0.059 ± 0.012**	**0.727**

*value ± SD
†value at impact treatment site at baseline before treatments were installed

Table 2. Average values of bioassessment metrics from all three sites for all sampling collections.

Bioassessment Metrics*	Treatment	Collection			p-value†
		Baseline	September	October	
Diversity & Composition					
Total Abundance	Control	384 ± 146	425 ± 130	580 ± 111	0.294
	Impact	544 ± 268	611 ± 217	485 ± 225	
Total Richness	Control	23 ± 11	19 ± 3	21 ± 3	0.241
	Impact	22 ± 8	22 ± 4	18 ± 5	
Water Quality					
EPT Abundance	Control	161 ± 87	196 ± 81	315 ± 191	0.329
	Impact	277 ± 120	352 ± 214	295 ± 216	
EPT Richness	Control	12 ± 8	9 ± 3	10 ± 1	0.407
	Impact	9 ± 3	11 ± 3	9 ± 5	
% EPT Abundance	Control	42 ± 11	45 ± 11	51 ± 27	0.617
	Impact	53 ± 8	57 ± 23	56 ± 25	
% EPT Richness	Control	52 ± 7	49 ± 9	50 ± 5	0.327
	Impact	41 ± 1	51 ± 6	51 ± 13	
Functional Feeding Groups					
Filterer-Gatherer Abundance	Control	268 ± 104	298 ± 67	352 ± 57	0.278
	Impact	316 ± 175	386 ± 118	276 ± 78	
% Filterer-Gatherers	Control	69 ± 5	71 ± 6	62 ± 17	0.466
	Impact	56 ± 7	65 ± 13	60 ± 12	
Scraper Abundance	Control	2 ± 2	18 ± 27	45 ± 68	0.410
	Impact	8 ± 5	27 ± 29	13 ± 16	
% Scrapers	Control	1 ± 0	3 ± 5	7 ± 11	0.397
	Impact	1 ± 0	4 ± 3	3 ± 4	
Shredder Abundance	Control	93 ± 39	65 ± 29	139 ± 70	0.515
	Impact	181 ± 80	153 ± 105	146 ± 132	
% Shredders	Control	25 ± 6	15 ± 4	23 ± 9	0.517
	Impact	34 ± 4	24 ± 9	26 ± 13	
Predator Abundance	Control	21 ± 10	44 ± 13	43 ± 29	0.511R‡
	Impact	39 ± 13	45 ± 18	50 ± 20	
% Predators	Control	5 ± 1	11 ± 1	7 ± 4	0.130
	Impact	8 ± 3	7 ± 2	11 ± 1	
% Chironomid Individuals	Control	13 ± 4	20 ± 12	29 ± 20	0.763
	Impact	12 ± 5	13 ± 2	25 ± 13	

*Mean ± SD (metric value averaged from all three sites)

†p-value represents whether the differences in metric values between the control and impact treatments increased significantly over time compared to baseline.

‡R indicates that a Wilcoxon signed-rank test was performed instead of an ANOVA because the parametric assumptions were not met.

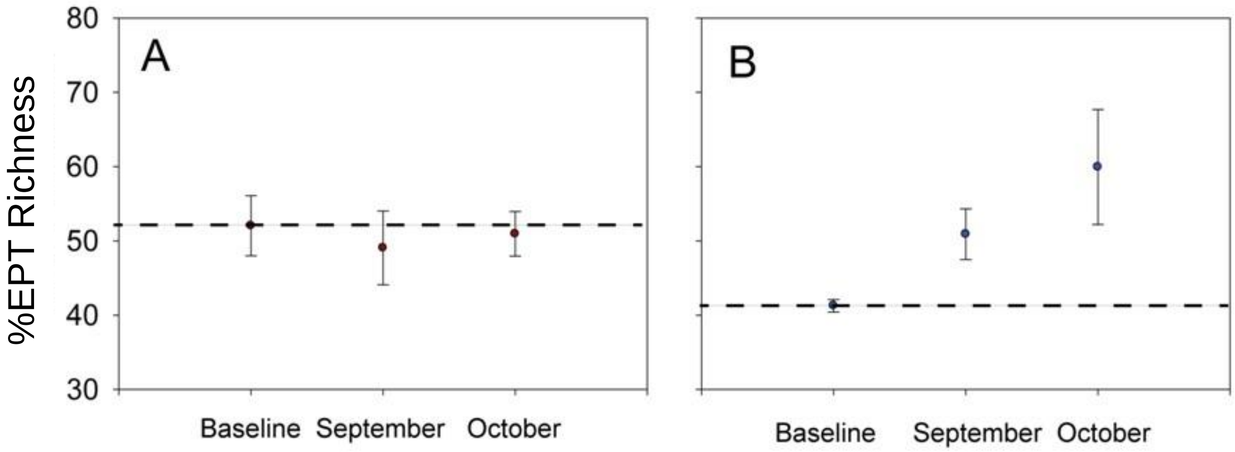

Figure 3. Observational trend in average %EPT richness at (A) Control and (B) Impact treatments across time. Error bars indicate standard error and dotted line indicates baseline value.

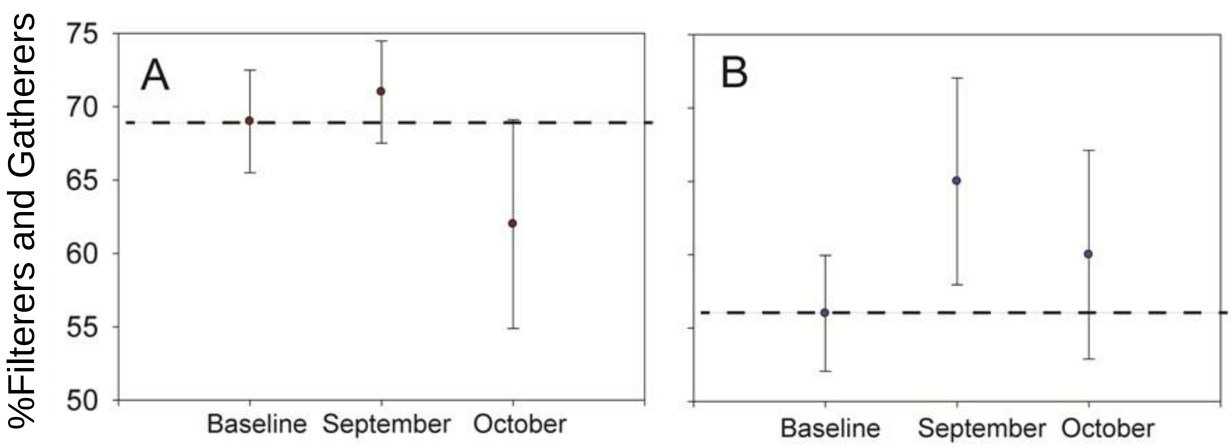

Figure 4. Observational trend in average %filterers and gatherers at (A) Control and (B) Impact treatments across time. Error bars indicate standard error and dotted line indicates baseline value.

Multivariate analysis

The first two NMS axes explained a cumulative total of 40.5% of the variability in the species abundances observed, with 25.9% explained by the first axis and 14.6% by the second axis (Figure 6). Stream depth and flow velocity aligned along the second axis suggesting that that they explained only a relatively small fraction of the total variability. The amount of variability that stream depth and flow velocity explained along the second axis represents less than 5% of the total variability and is proportional to the length of the arrows lines leading from the origin to their respective labels.

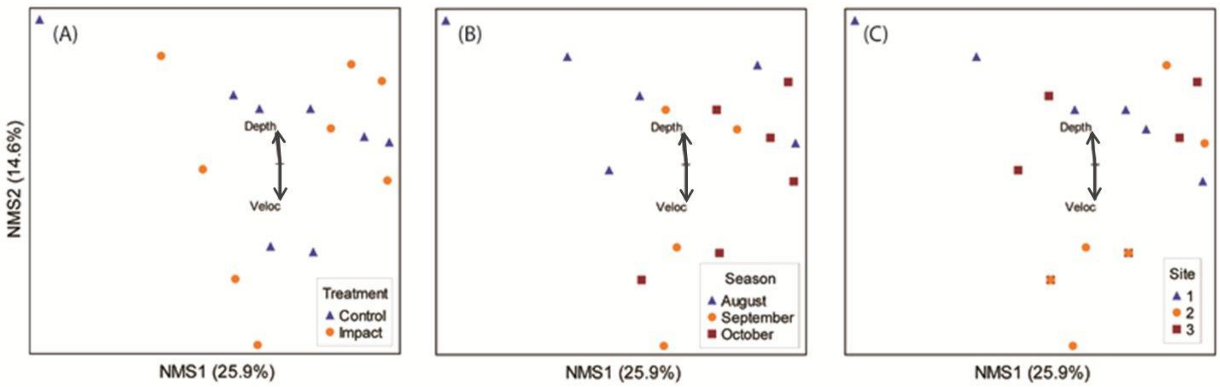

Figure 6. NMS of species distribution grouped by (A) treatment (B) season, and (C) site.

Relationship between physical stream features and bioassessment metrics

I found a non-significant relationship between the two physical stream features (flow velocity and stream depth) and the three bioassessment metrics (taxa richness, %EPT richness and percent filterer and gatherers). Specifically, linear regression tests between taxa richness, %filterers and gatherers and flow velocity gave R-squared values of 0.0007, 0.1047 and 0.0949, respectively. Similarly, R-squared values for the relationship between taxa richness, %EPT richness, %filterers and gathers and stream depth were also non-significant (Figure 7).

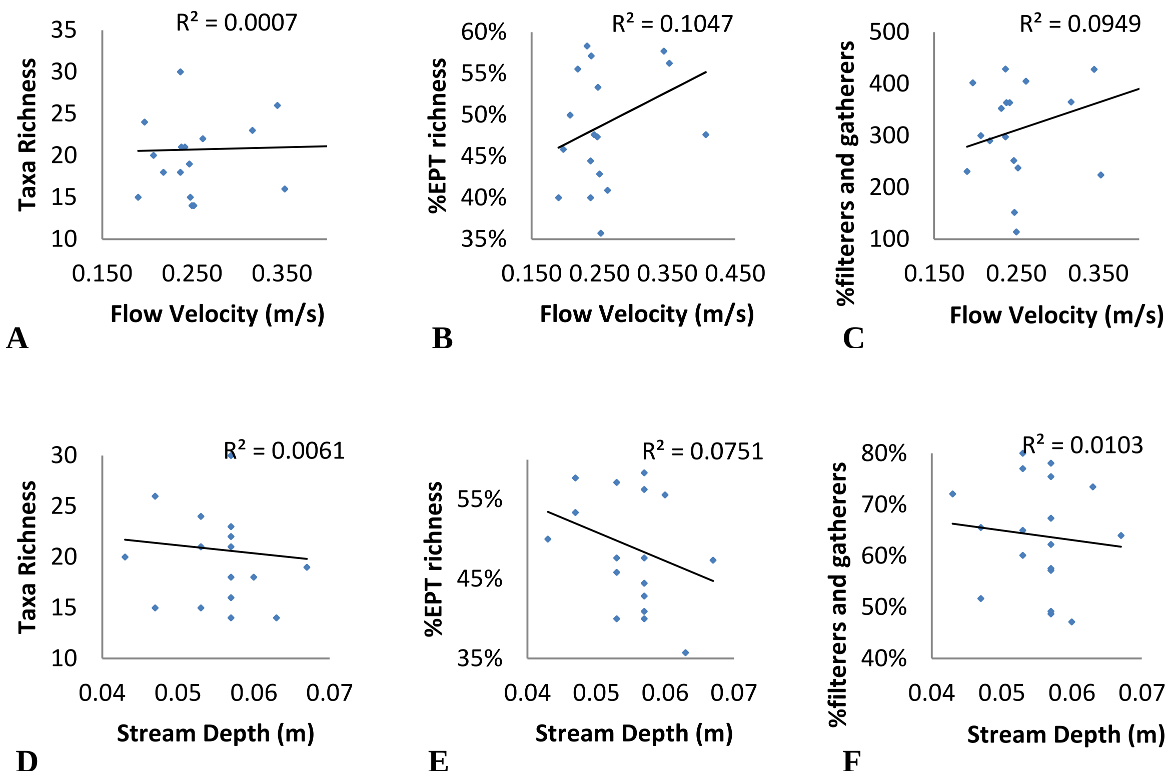

Figure 7. Linear regression with physical stream parameters (flow vecloty and stream depth) as predictor and bioassessment metrics as outcome (taxa richness, %EPT richness and % filterers and gatherers).

DISCUSSION

Streamflow augmentation of urban streams has been considered in some regions to help maintain ecological integrity during the dry season (Matlock et al. 2000). In this study, I assessed the relative influence of streamflow augmentation on the community of benthic macroinvertebrates in a small urban Mediterranean stream during the dry summer months (i.e. August through October). These benthic organisms are widely used as indicators of ecological integrity and a variety of metrics based on them are available for monitoring responses (Resh and Jackson 1993). In this study the among-site variability was so high that I found no significant difference in average streamflow or biological metric values between the control and impact treatments. Nevertheless, I observed directional trends in the data collected which suggests potential minor improvements in water quality and increases in filter-collector population at the impact treatments.

Responses of benthic macroinvertebrates to flow augmentation

The lack of significance observed in the changes in the metrics examined suggests that the augmented flow did not increase macroinvertebrate diversity, improve stream water and habitat quality and benefit the filterer and gatherer population as hypothesized, and the experimental design of this study may have been responsible for this. For example, by installing the plywood boards to increase the flow in my study, I created constricted artificial channels at the impact treatments, and channelization has been found to decrease species richness and abundance in urban streams (Michal et al. 2009, Rohasliney and Jackson 2008). It is thus possible that the augmented-flow procedure had a negative impact on the macroinvertebrate community. Similarly, the plywood-board treatments created a new habitat that was physically separated from the riparian vegetation, which has been shown to be positively correlated with invertebrate family richness (Arnaiz et. Al 2011). Additionally, increase in flow may have also caused considerable shear stress resulting in a downstream drift of benthic macroinvertebrates that are sensitive to hydraulic disturbance resulting in a community abundance loss (Borchardt 1993).

Natural life history of benthic macroinvertebrates in Mediterranean-climate streams may also serve as another explanation to why I did not see significant responses of these macroinvertebrates as expected. Specifically, certain benthic macroinvertebrates take refuge in the hyporheic zone after increased flow discharge (Marchant 1995, Holomuzki and Biggs 2000), and I did not sample this habitat in this study. Moreover, as streamflow in the Mediterranean climate is highly variable according to season, benthic communities are naturally adapted to changes in flow within the range examined (Gasith and Resh 1999). As a result, because streamflow was only non-significantly elevated at impact treatments, this increase in flow most likely was not ecologically significant for the resident organisms.

In contrast, increased streamflow has been associated with higher water quality, macroinvertebrate diversity and increase in filterer and gatherer individuals (Matlock et al. 2000, Gasith and Resh 1999, Peeters et al. 2004). Likewise, some benthic macroinvertebrates exhibit preferences for high flow velocity as a habitat variable (Jowett and Richardson 1990). The observed potential improvement in water quality evident by the (non-significant) increase in percent EPT richness at impact treatments suggests a weak signal that could be amplified with a different study design. An increase in discharge has been associated with higher concentration of dissolved oxygen through increased turbulence (Matlock et al. 2000), higher habit quality by providing sufficient water to support aquatic wildlife (Gratham et al. 2010), and higher water quality through dilution of pollutants (Gasith and Resh 1999). Moreover, because percent filterers and gatherers increased, although non-significantly, after flow augmentation it seems that that the filterer and gatherer population may have benefited from the augmented flow. Higher flow has also been associated with faster nutrient delivery rate, increased riparian vegetation growth as a result of increased water availability, and more organic matter suspension in the water column (Ponce and Lindquist 1990). This increase in food availability and preferred habitat could allow for an increase in the filterer and gatherer population, which we may have seen to a very small degree in this study (Peeters et al. 2004).

Flow augmentation in urban Mediterranean streams

Because low flow takes place in dry season and my study found a bigger increase in taxa richness and percent EPT after increased flow in the dry season than in the wet season, flow augmentation can be an effective solution to provide baseflow or supplement unnaturally low flow in disturbed streams during the water-stressed months. Flow augmentation is commonly suggested in restoration projects (Purcell et al. 2002). Specifically, the use of reclaimed water to augment streamflow has been considered as an effective water management strategy that provides benefits to stream organisms.

Benthic macroinvertebrates as biological indicators

The response in benthic macroinvertebrate metrics to changes in both physical and biological conditions of the study stream sites after flow augmentation suggest that benthic macroinvertebrates were effective biological indicators. Correspondingly, benthic macroinvertebrates have been recognized for their effectiveness in biological monitoring and have been the preferred method for rapid bioassessment (Resh et al. 1995). The bioassessment metrics (i.e. taxa richness, percent EPT richness, and percent filterer-gatherers) were useful in testing my study hypotheses.

Limitations

Although species level has been generally accepted to be the "best-available-taxonomy" level of identification, Lenat and Resh (2001) and Jones (2008) have argued that organism identification should be based on an optimization of trade-offs between taxonomic detail and identification resources. The family-level identification in this pilot study was most appropriate and less likely to introduce error in the results of biotic indices.

Nevertheless, a few limitations have to be considered with the results of this study. Given the scope of a pilot study, we only sampled benthic macroinvertebrates three times in total and two times after the augmented-flow treatments had been installed. This short timescale and small number of samples limited my ability to articulate that my study results would still be applicable at other times of the year or that the results would be relatively the same if I were to repeat it over many years.

The significance of these trends could have perhaps been better demonstrated if the study design had included higher statistical power (i.e. more replicate sites) or the magnitude of the flow manipulation was more dramatic (i.e. an 80-90% increase in streamflow between control and impact treatments). However, this would have required approaching zero flow through the control treatments, which would have required a different protocol for sampling the macroinvertebrates.

Future directions

As flow in urban Mediterranean-climate streams is highly variable, little is known on the flow level that is optimum for the survival for the benthic communities. Therefore, larger studies carrying out more frequent sample collections over a longer period of time, preferably multiple years, would be preferable because the effects of seasonal and annual variability could be better controlled. Future flow augmentation studies can also take place in more water-stressed and more urbanized or polluted streams than Strawberry Creek, which are ecologically degraded and thus have more room to improve. The responses in such a case may be easier to detect. Additionally, further research is needed to extensively study the effects of highly treated wastewater effluents, instead of natural water, on the benthic macroinvertebrate communities before flow augmentation applications can be applied on a large scale as this may be the only source of water available for this purpose in the future (Prat and Munné 2000).

Broader Implications

As more stream water is diverted or abstracted for human use from increasing water demand, stream water and habitat quality continue to deteriorate resulting in considerable stress on stream organisms (Fleckenstein et al. 2004, Wolff et al. 1989). In addition, global climate change is expected to cause a progressive decline in the average streamflow and changes in hydrologic regime in Mediterranean streams resulting in an unprecedented stress on benthic communities (Garcia-Ruiz 2011). The results from this study suggest that streamflow augmentation may act as a potential application to combat this human-environment interaction problem. Restoration of disturbed streams with unnaturally low flow in urban areas can use this flow augmentation application to restore flow to pre-disturbance levels (Fleckenstein et al. 2004). Notably, a number of flow augmentation projects have already been practiced using treated wastewater to provide base flow for disturbed streams such as Beargrass Creek in Kentuckey, Bell Creek in Washington, and the San Antonio River in Texas (Houmis et al. 2005). In conclusion, effective water and wastewater management strategies are thus suggested to prevent ecological damage and at the same time provide benefits to stream organisms.

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