The Effects of Urbanization on Water Quality: A Biological Assessment of Three Bay Area Watersheds using Benthic Macroinvertebrates as Biological Indicators

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Abstract. Many of the watersheds in the United States face water quality issues caused by anthropogenic sources such as pollution and urbanization. This study seeks to evaluate biological conditions of three urban creeks in the east San Francisco Bay Area (Sausal Creek, Strawberry Creek, and Codornices Creek in Berkeley and Oakland, California). This evaluation of creek health was used to determine the effects of varying degrees of urban land use in the upstream watershed by examining the composition of macroinvertebrate communities. For comparison, three sites on each stream were selected with increasing levels of urbanization. Spatial analysis was used to determine the degree of urban land use in subwatersheds. Water quality was evaluated by calculating biological metrics such as Family Biotic Index scores based on benthic macroinvertebrate samples, and habitat quality was assessed using the EPA's Rapid Bioassessment Protocol for habitats. Upstream sites were found to have significantly better water quality than downstream sites based on taxa richness and percent of sensitive organisms. Codornices Creek, the watershed most impacted by urbanization, had poorer water quality scores than Strawberry Creek and Sausal Creek. Land managers and policy makers can use the results of this study to predict and remediate the impacts of urbanization on these impacted urban watersheds.

Introduction

Water is one of the most important natural resources in the United States and around the world (EPA 2001, Voelz *et al.*2005). It is a necessity for life and provides a variety of uses from drinking water in cities to the irrigation of crops in agricultural areas. Water also provides recreational uses as well as habitat for wildlife. According to the Environmental Protection Agency (2001) "rivers, lakes, estuaries, and wetlands are among the Nation's most precious resources." Billions of dollars of the U.S. economy rely on the health of watersheds; clean water is also necessary for various recreational, industrial, and agricultural uses (EPA 2001). Even though many acknowledge the importance of clean water and healthy watersheds, about forty percent of watersheds in the United States face water quality issues from urbanization to agricultural pollution or a combination of many "complicated" factors (EPA 2001). Although watersheds provide many benefits to society, they can easily be impacted by human influences (ACCWP 2004).

Urbanization is one of the most detrimental forces affecting stream health and one of the biggest challenges facing watershed managers. However, methods to determine how large-scale changes in watersheds affect local habitats are still producing varied results (Kearns *et al*. 2005). Urbanization affects "patterns of ecologic structure and function" (Walsh 2006) by altering the physical landscape, increasing imperviousness, and changing channel morphology (Paul and Meyer 2001, Sponseller *et al*. 2001, Walsh et al. 2001, Kearns *et al*. 2005, Chadwick *et al*. 2006). Modification of the physical landscape by human development can exacerbate erosion, sedimentation, and bank undercutting thus reducing habitat for organisms such as fish and benthic macroinvertebrates (ACCWP 2004). Urban storm water enters creeks and rivers more readily from impervious surfaces and can increase the flashiness of the flow regime. Urban runoff can also affect water chemistry by changing levels of heavy metals and nutrients like phosphorus and nitrogen (Porcella and Sorenson 1980, Morse *et al.* 2002). These impacts from urbanization can cause changes in the biological communities of the stream ecosystems (Morse *et al*. 2002, Chadwick *et al*. 2006, Voelz *et al*. 2005, Walsh 2006). In addition, urban impacts are especially concerning because they can be seen throughout watersheds, and not just on a local level (Kearns *et al*. 2005). As stream ecosystems are changing, it is apparent that there is a need to develop consistent methods to track these changes, and monitor the environment within streams (Kearns *et al*. 2005). Impacts can be seen on multiple scales, and it is important to look at how watershed impacts of increasing urban development and land use affect habitats on a local scale.

One of the most effective ways to assess stream health is through the use of benthic macroinvertebrates as biological indicators (or bioindicators) of stream health. Benthic macroinvertebrates are commonly used in water quality assessments because they have a close link to the chemical and physical states of their habitats (Karr 1981, Resh *et al*. 1996, Simon and Stewart 1999, Sawyer *et al*. 2004). Assessments using benthic macroinvertebrates as biological indicators can be quite successful and reliable at determining stream health, and allow for a simple method to identify water quality issues (Resh *et al*. 1996, Hutchinson and Iyengar 2003). Benthic macroinvertebrates are widely used because of the large number of diverse species that have different tolerances to water quality, long life cycles, and a well-known taxonomy (Resh *et al.* 1996). Species with long lifecycles allow for long-term changes to be tracked, and a well-known taxonomy allows for easy identification of organisms in the field and the lab (Resh *et al*. 1996). For this reason, Hutchinson and Iyengar (2003) explain how macroinvertebrates can be seen as an "ecological memory" in aquatic habitats. By looking at the composition of benthic macroinvertebrates in relation to their pollution tolerance scores, the health of streams within a watershed can be evaluated, and the affect of habitat conditions on water quality can be determined (Resh *et al*. 1996, Sawyer *et. al*. 2004). Regular monitoring of watershed health can be useful in order to assess damage, protect wildlife as well as habitats, and to provide stakeholders with planning information (ACCWP 2004). By comparing bioindicators among sites, the effects of urban development and anthropogenic disturbances in the surrounding watershed can be determined. This comparison can also be used to examine how land use on a watershed scale affects water quality on a local, reach scale in streams.

This study addresses the following objectives: 1) determine the effects of varying percentages of urban land cover in watersheds on water quality within streams, 2) understand how large scale changes of land cover in watersheds affect water quality on a local level, and 3) assess the current health of streams in the Sausal Creek, Strawberry Creek, and Codornices Creek watersheds. It is hypothesized that sites with less upstream urban land cover will have a better biological condition as determined by biological metrics using benthic macroinvertebrates.

Methods

Study Sites Three creeks in the San Francisco Bay Area (Sausal Creek in Oakland, Strawberry Creek and Codornices Creek in Berkeley) were selected for use in this study (Fig. 1). Each watershed had an urban gradient from low to high levels of urban land use. In addition, according to the Alameda County Clean Water Program (2004), each of these creeks has many beneficial uses including fish spawning, recreation, wildlife habitat, and cold/warm water habitats. In order for these streams to continue to support wildlife and remain safe for humans, it is important to monitor their water quality. Three study sites (one downstream, one midstream, and one upstream) were selected along each creek for sampling and included the following (see Fig. 1 for specific locations):

Figure 1: Map of Oakland and Berkeley, showing locations of study sites. Sites are indicated by labels and black dots on streams. Map is from the Oakland Museum's "Oakland and Berkeley Watershed Finder."

- Codornices Creek - Site 1 (downstream) was located at 5th and Harrison Street, site 2 (mid-stream) was located in Live Oak Park, and site 3 (upstream) was located in Codornices Park.
- Strawberry Creek - Site 1 (downstream) was located in Strawberry Creek Park, site 2 (mid-stream) was located on the UC Berkeley campus adjacent to Valley Life Science Building, and site 3 (upstream) was in Strawberry Canyon off of Centennial Drive.
- Sausal Creek - Site 1 (downstream) was on Bona Street off of Fruitvale Avenue, site 2 (mid-stream) was located in Dimond Park, and site 3 (upstream) was located off of Monterey Blvd.

Physical Habitat Assessment At each site, physical habitat was evaluated and scored based on the parameters in the EPA's Rapid Bioassessment Protocol (Barbour *et al*. 1999). Ten parameters were scored on a scale of 0-20 (with 20 as a high score) including: Epifaunal Substrate/ Available Cover, Embeddedness, Velocity/Depth Regime, Sediment Deposition, Channel Flow Status, Channel Alteration, Frequency of Riffles (or bends), Bank Stability, Vegetative Protection, Riparian vegetative Zone Width (Barbour *et al*. 1999). Overall scores were out of a total 200 points and were used to determine habitat ratings of optimal, suboptimal, marginal, or poor.

Collection of Benthic Macroinvertebrates In order to collect benthic macroinvertebrates, the California Stream Bioassessment Procedure from the California Department of Fish and Game (2003) was used. At each site, three riffles within a 100-meter reach were randomly selected for collection. Benthic macroinvertebrate sampling was conducted by placing a 500 μm D-frame net in the stream and disturbing the substrate in a 0.9 m^2 area directly upstream from the collection net for a one-minute interval. Samples were preserved in 95% ethanol in the field and then transferred to 70% ethanol in the laboratory. Benthic samples were collected August 11th, 12th, and 13th 2007.

Calculation of Biological Metrics The following biological metrics were calculated for each stream site: taxa richness, Family Biotic Index (FBI), Shannon Diversity Index, proportion of individuals from the pollution-sensitive EPT orders (Ephemeroptera, Plecoptera, Trichoptera), and proportion of individuals from the pollution-tolerant groups Oligocheata, Chironimidae, and Hirudinea. The FBI was calculated by multiplying the number of organisms in each family by the tolerance score taken from the List of California Macroinvertebrate Taxa and Standard Taxonomic Effort (2003). The average was then taken across all the families collected for a site.

Spatial Analysis Subwatershed boundaries for each study site were hand digitized in ArcGIS 9.2 (ESRI, Inc. Redlands, CA). These boundaries were determined using a Geographic Information Systems (GIS) database for the 2000 edition of the Oakland Museum creek and watershed map of Oakland and Berkeley that was created in ArcView 3.2. Subwatershed boundaries were used to create polygons representing the watershed area upstream of study sites. Polygons were clipped to a land cover data set with a 30-m resolution (NOAA, 2000) to calculate the percentage of urban land cover within each subwatershed. The two land cover categories that were summed to produce an overall calculation of urban land cover in polygons were: high density developed (defined as areas containing greater than 75 percent impervious surfaces), and low density developed (defined as areas with greater than 25 percent impervious surfaces but less than 75 percent impervious surfaces).

Statistical Analysis One-way ANOVAs and student's t-tests were used in JMP 5.1 (SAS Institute Inc. 2004) to compare biological metrics among watersheds and among sites (i.e. upstream (most urban), midstream, and downstream (least urban)). Linear regression was also used to look at the effects of varying degrees of urban land cover on biological metrics. A Bonferroni correction was used to compare 3 groups, so the significance threshold was $p<0.02$.

Results

Physical Habitat Assessment Scores Total habitat scores at all sites ranged from 77 (marginal habitat) to 140 (suboptimal habitat). Strawberry Creek had the highest overall habitat scores at all three sites (140 upstream, 109 midstream and downstream). In Codornices and Sausal Creeks the total habitat score at the midstream site was lower than the downstream site. However, the downstream and midstream sites in Strawberry Creek scored the same overall. Of ten habitat parameters, the frequency of riffles (range 12-18) and embeddedness (range 12-17) had the highest scores. Some of the parameters that received low scores overall were channel flow (range 1-9), vegetative protection (range 1-9), and riparian vegetative zone width (range 1-9). For a full summary of physical habitat assessment scores see Appendix I.

Benthic Macroinvertebrates and Biological Metrics Overall, 36 taxa were identified from benthic samples. The rarest families were Thumaleidae (only at Codornices Creek downstream), Glossomatidae (only at Strawberry Creek downstream), Cordulegastridae (only at Strawberry Creek upstream), and Astacidae (only at Codornices Creek downstream). The most common organisms, found at each site, were: Acariforms, Turbellaria, and Chironomidae. The family Beatidae was found at every site except for the downstream site on Codornices Creek. The Codornices Creek downstream site also did not have any organisms within the insect orders Plecoptera (stoneflies) and Trichoptera (caddisflies). Individuals within the order Coleoptera (beetles) were absent from Codornices Creek and the downstream site on Strawberry Creek. The upstream Strawberry Creek site had no gastropods (snails) present. For a complete taxa list see Appendix II.

Mean taxa richness in the Strawberry Creek downstream site was 9.5 ± 2.12 taxa (mean ± SD) compared to 22 ± 1.41 taxa richness in the Codornices Creek downstream site. The Strawberry Creek downstream site also had the highest values for percent Oligocheata (6.25 ± 2.95) and percent Chironomidae (35.63 ± 5.60). The lowest percentage of Oligocheata (0%) was at the upstream Strawberry Creek site. No EPT individuals were found in the Codornices Creek downstream site. The Sausal Creek and Strawberry Creek upstream sites had the highest values of percent EPT and EPT richness (Appendix III). FBI scores were fairly consistent among watersheds and sites (Fig. 2). The lowest mean FBI score (3.53 ± 0.64) was in the upstream Sausal Creek site and the highest mean FBI score (7.01 ± 0.51) was in the upstream Codornices Creek site. For a complete list of values for each metric see Appendix III.

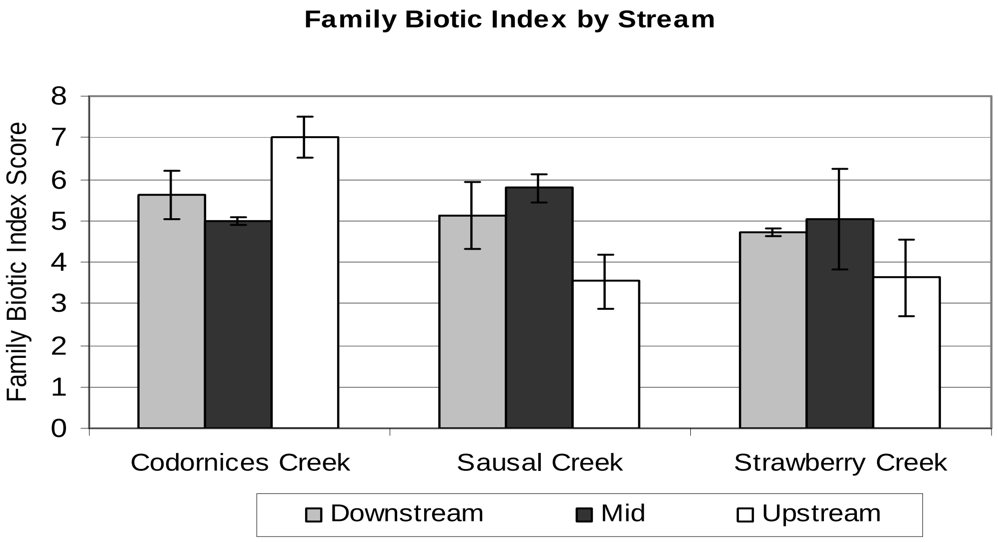

Figure 2. Average Family Biotic Index scores (± SD) for each creek divided by study site. Lower scores indicate better overall water quality.

Spatial Analysis Polygons labeled "upper" were used to calculate percentage urban land cover in the upstream watershed areas. Midstream urban land cover percentages were calculated by combining polygons labeled "mid" and "upper." Downstream urban land cover percentages were calculated using the polygons labeled "low," "mid," and "upper." Refer to Figures 3-5 for subwatershed boundaries that are specific to each study site.

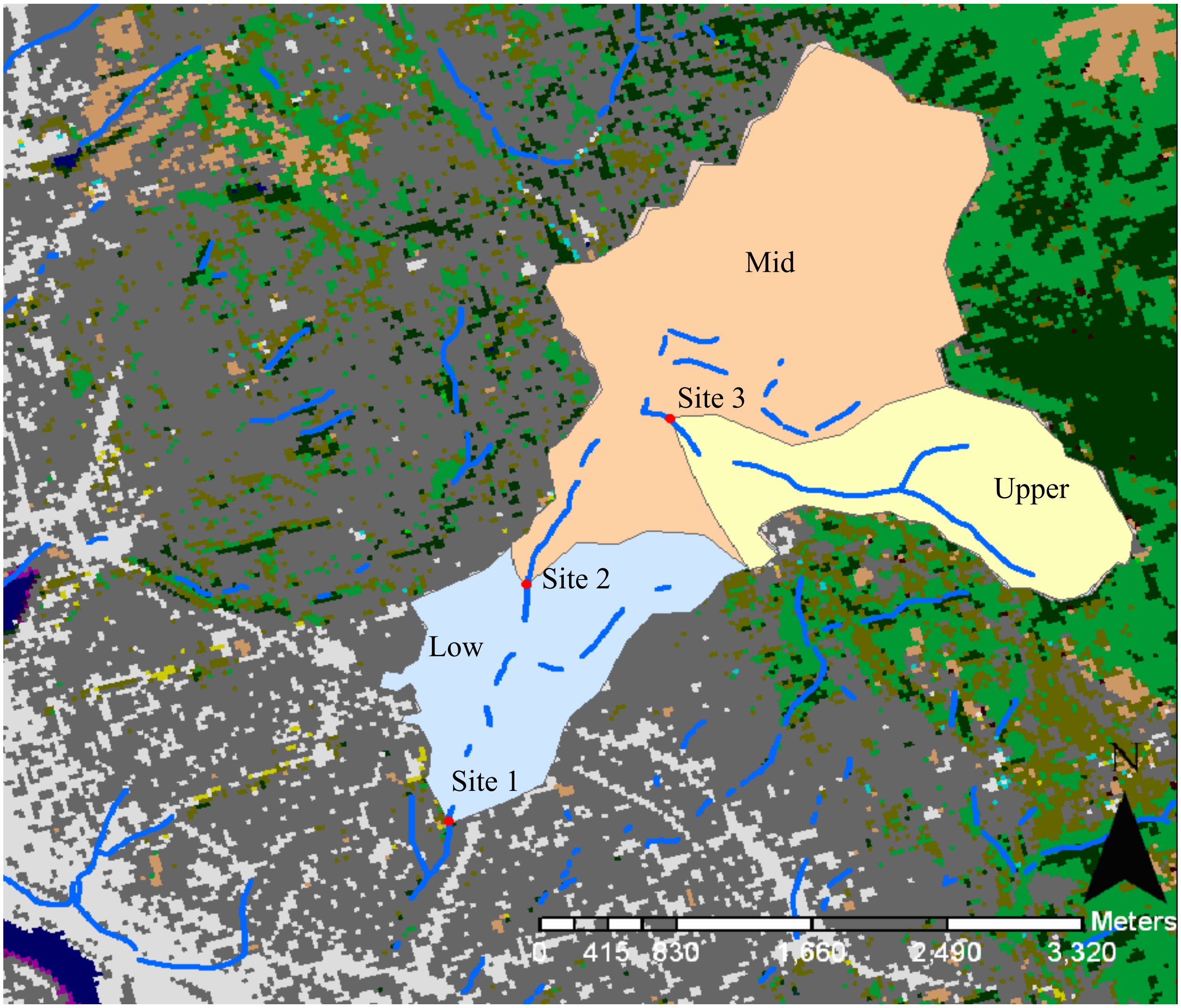

Figure 3. Sausal Creek watershed boundaries. The yellow polygon represents the upstream subwatershed, the orange polygon is the midstream subwatershed, and the blue polygon is the downstream subwatershed. Study sites are indicated by red dots and blue lines represent stream features.

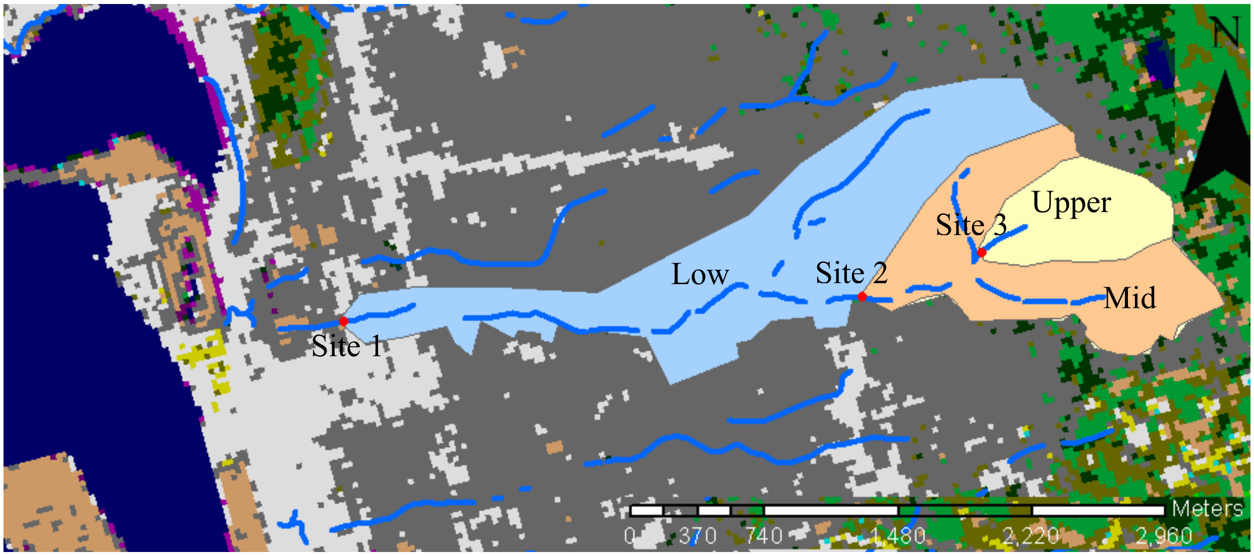

Figure 4. Codornices Creek subwatershed boundaries. The yellow polygon represents the upstream subwatershed, the orange polygon is the midstream subwatershed, and the blue polygon is the downstream subwatershed. Study sites are indicated by red dots and blue lines represent stream features.

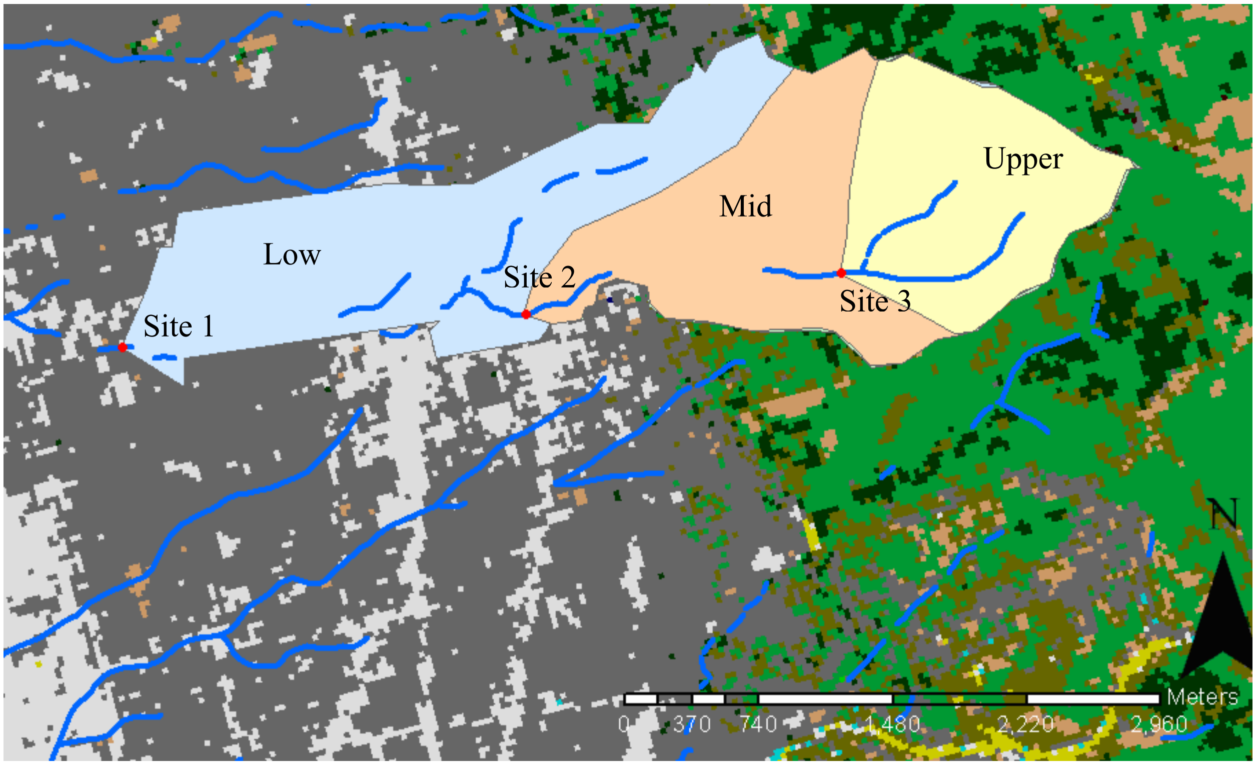

Figure 5. Strawberry Creek subwatershed boundaries. The yellow polygon represents the upstream subwatershed, the orange polygon is the midstream watershed, and the blue polygon is the downstream subwatershed. Study sites are indicated by red dots and blue lines represent stream features.

The Codornices Creek watersheds had significantly higher percentages of urbanization than both Strawberry Creek ($p<0.02$) and Sausal Creek ($p<0.02$). Sausal Creek had a significantly higher percentage of urbanization than Strawberry Creek ($p = 0.03$, $\alpha<0.5$). Across all three watersheds, upstream sites were the most urban, midstream sites at mid levels of urbanization, and downstream sites were the least urban (Fig. 6).

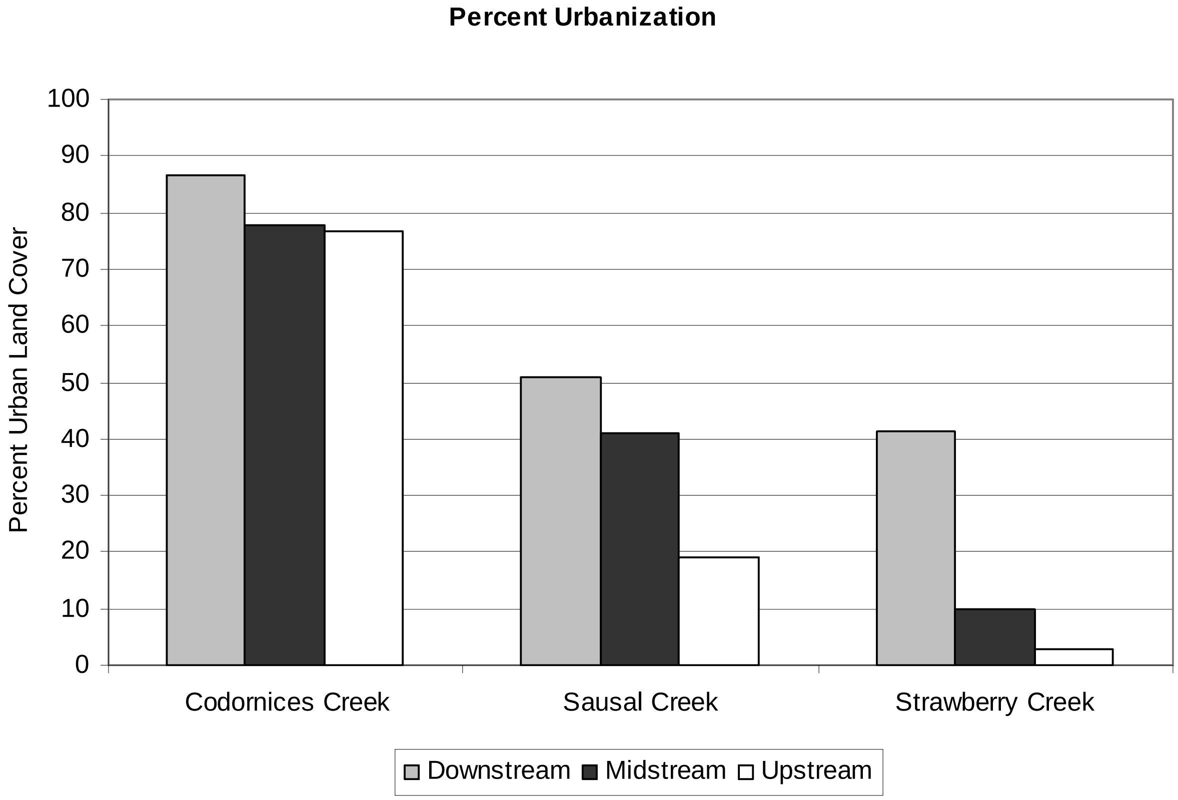

Figure 6. Percentage of urban land cover in the watershed area above each study site. Strawberry Creek had the lowest amount of urban land cover at the downstream (41.43%), midstream (9.83%), and upstream (2.76%) sites. Codornices Creek had the highest percentage of urban land cover at each site (86.72%, 77.78%, and 76.77% respectively).

Statistical Analysis Few significant results ($p<0.02$) were found using the ANOVA analysis. Taxa richness was significantly higher ($p = 0.006$) at the midstream sites compared to the downstream sites. Also, the percentage of EPT not including the families Baetidae and Hydropsychidae was significantly higher ($p = 0.012$) in upstream sites than in downstream sites (Fig. 7). Using a higher significance value as a threshold ($p<0.05$) upstream sites were also shown to have a significantly higher ($p = 0.048$) percentage of EPT, not including the families Beatidae and Hydropsychidae, than the downstream sites. Furthermore, Sausal Creek had significantly higher ($p = 0.016$) percentages of EPT than Codornices Creek. FBI was marginally significantly higher in Codornices Creek than Strawberry Creek ($p = 0.032$).

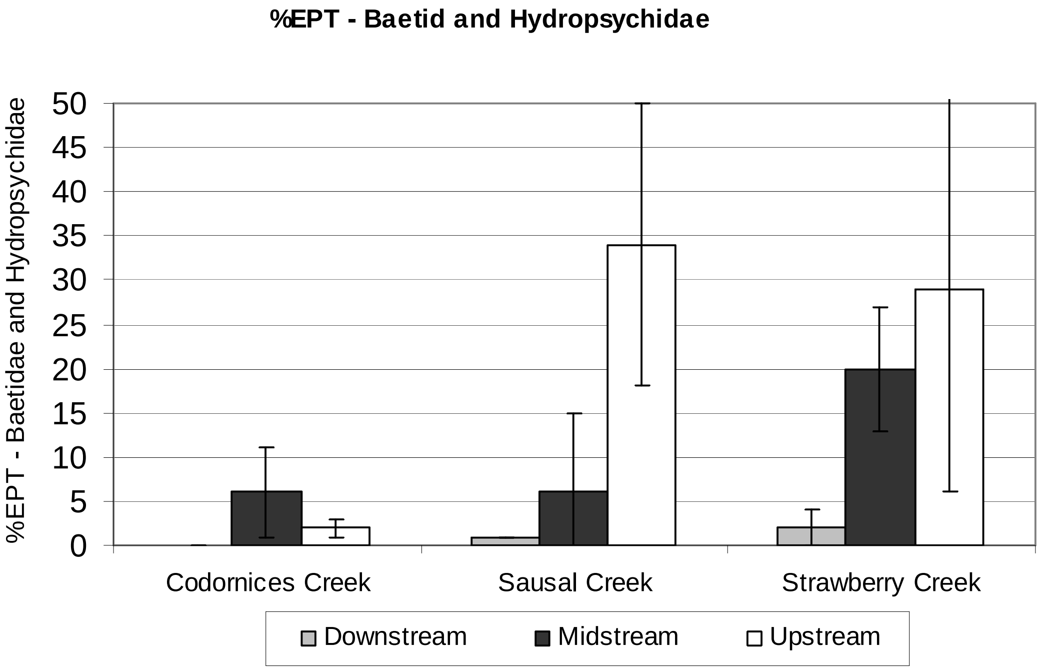

Figure 7. Average (± SD) percentage EPT not including the tolerant families Baetidae and Hydropsychidae.

Regression analysis revealed significant correlations between the biological metrics and percentage of urbanization in the upstream watersheds. As urbanization increased, percent EPT showed a significant decrease (r^2=0.48, F=14.81, $p<0.001$) (Fig. 8). EPT Richness and percent EPT, not including the families Baetidae and Hydropsychidae, also showed significant decreases (r^2=0.25, F=5.20, $p<0.036$ and r^2=0.48, F=15.01, $p<0.001$, respectively) as urbanization increased. The fourth comparison showed a positive correlation between FBI and percentage of urbanization (r^2=0.39, F=10.29, $p<0.006$) (Fig. 9).

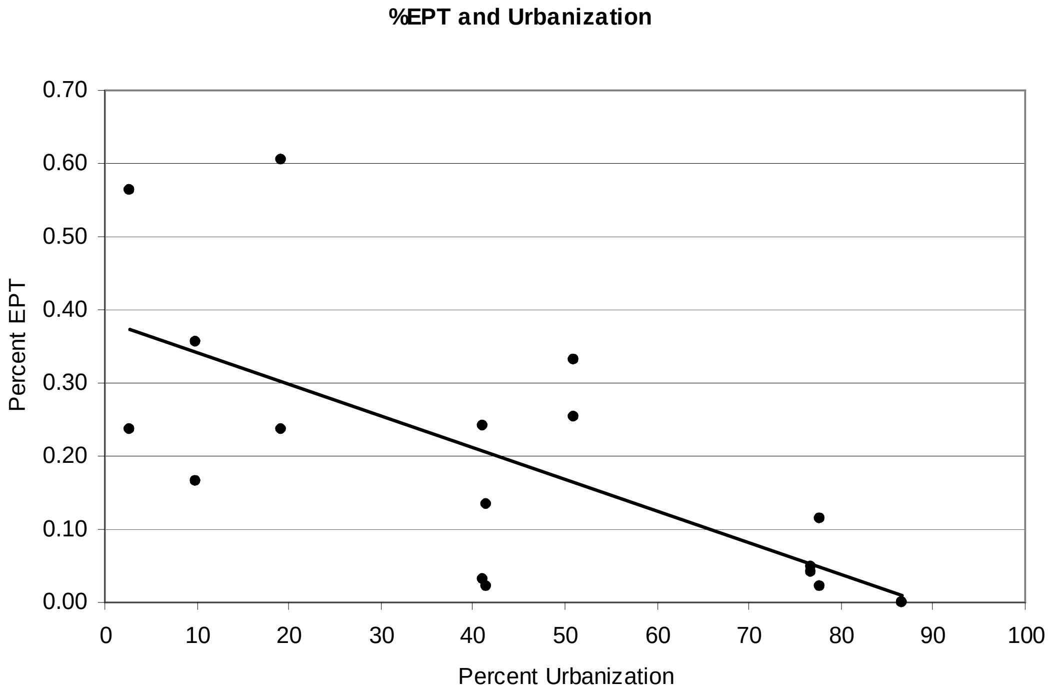

Figure 8. Linear regression of percent EPT and percent urbanization. As urbanization increases, there are less sensitive taxa present (r^2=0.48, F=14.81, p<0.001).

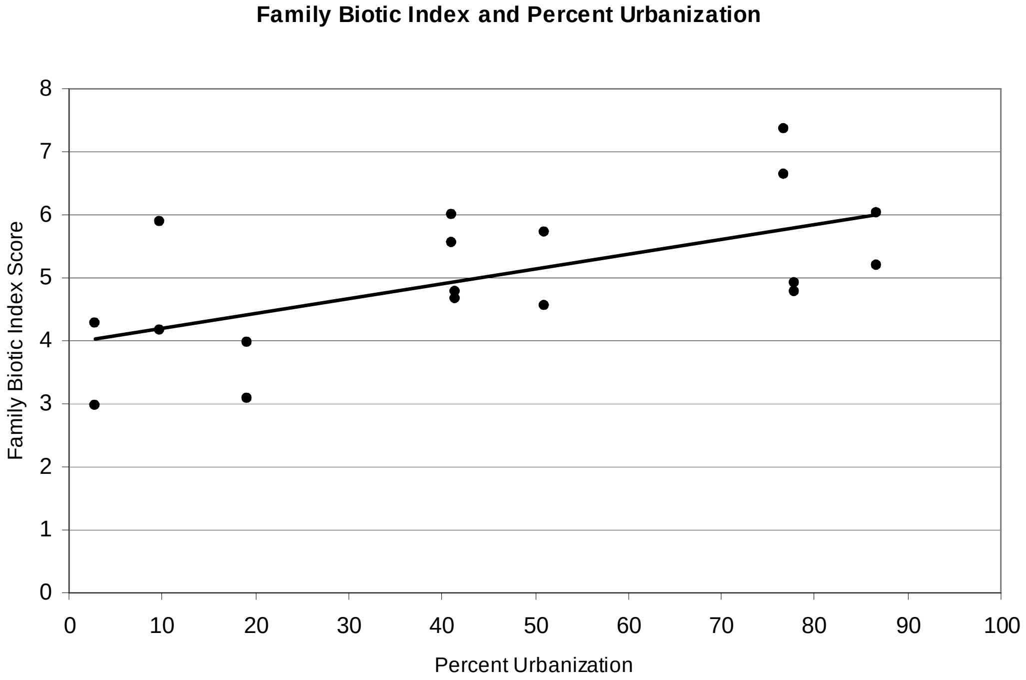

Figure 9. Linear regression of FBI and percent urbanization. As urbanization decreases, FBI scores also decrease indicating that water quality is improving (r^2=0.39, F=10.29, p<0.006).

Discussion

The results of this study found that watersheds with lower levels of urban land use have better water quality as reflected by the presence of a greater proportion of sensitive taxa and higher taxa richness. Upstream sites (with lower percentages of urban land use) had more diverse organisms than the downstream sites (with higher percentages of urban land use). Linear regression analysis indicated that as urban land cover increases within watersheds, water quality declines as determined by FBI scores (Fig 6). As urban land cover increases, the presence of sensitive taxa also decreases (Fig. 8). According to other studies, it is expected that there should be a decline in the percent of taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera which are sensitive to perturbations in water quality (Pratt *et al.* 1981, Hachmoller *et al.* 1991, Paul and Meyer 2001, Walsh *et al.* 2005).

Differences among watersheds also found that urban land cover had a negative impact on water quality. Codornices Creek was the most urban watershed (Fig.3) and this was reflected in its significantly lower water quality scores when compared to Sausal Creek and Strawberry Creek. The higher percentage of EPT in Sausal Creek may indicate better water quality than in Codornices Creek. Higher FBI scores in Codornices Creek may also show that its water quality is worse than in Strawberry Creek. Improved water quality conditions at Strawberry Creek may be attributed to the long term restoration projects that have taken place in the Strawberry Creek watershed. Restoration efforts on Strawberry Creek have shown a "detectable increase in invertebrate diversity and abundance" (Paul and Meyers 2001, Charbonneau and Resh 1992).

While these results are promising and may indicate changes in water quality along an urban gradient, only two (taxa richness and percent EPT minus Beatidae and Hydropsychidae) out of ten metrics showed a significant difference by site. In addition, linear regression results found only four metrics with significant trends along the urban gradient. There are a few considerations to take into account that could have affected these results. One consideration is the land cover dataset used. The NOAA dataset is from the year 2000 while samples were collected in August of 2007. Changes in land use over seven years could make a difference for the purposes of analyzing results. Another consideration is the potential of a critical urban threshold above which all sites are more likely to be degraded (King *et al.* 2005, Walsh *et al.* 2005). Thresholds have been indicated in watersheds with extremely low percentages of urbanization, some ranging from only one to ten percent (Walsh *et al.* 2005). If sites are above this threshold, it may be difficult to distinguish differences in water quality metrics. This could be the case in this study considering the percentages of urbanization in subwatersheds were extremely high (Fig. 6).

Other, more intensive, studies have shown stronger differences in water quality along urban gradients (Pratt *et al.* 1981, Hachmoller *et al.* 1991, Paul and Meyer 2001, Walsh *et al*. 2005). Decreases in invertebrate diversity have been shown to be correlated with spatial metrics like housing density, human population densities, and increased impervious surface cover (Klein 1979, Jones and Clark 1987, Paul and Meyer 2001). According to Paul and Meyer 2001, regardless of catchement size, a decrease in invertebrate diversity is expected to occur with an increase of urban land use. Areas that are highly impacted by urbanization should also be dominated by many Oligochaetes and Chironomids, which are considered tolerant taxa compared to EPT (Walsh *et al*. 2005). Increases in fine sediment, often caused by urban impacts, favor Chironomids and Oligochaetes because these taxa are adapted to living in depositional habitats (Collier 1995, Paul and Meyer 2001). In addition, toxins in sediments caused by runoff could have an affect on assemblages of macroinvertebrates (Paul and Meyer 2001). With the depletion of riparian zones and deforestation, toxins flow more easily into streams; this reduction of riparian zones can also cause changes in food availability for organisms, and changes in stream temperature (Paul and Meyer 2001, Walsh *et al*. 2005). In the three study watersheds, habitat assessment scores indicate that the vegetative protection and riparian vegetative zones have been degraded (Appendix I). This may be indicative of the overall high FBI scores (Fig. 2) and high percentages of Oligochaeta and Chironomidae in some study sites (Appendix III).

While changes can be seen along these urban gradients, it can be hard to determine the exact causes in these trends because of the “multivariate nature of urban disturbance” (Paul and Meyer 2001). This in turn presents many challenges for watershed managers and land owners when they are deciding on optimal management practices and strategies. Walsh *et al*. (2005) found that watershed management strategies in the past have been structured around the goals of protecting against flood and disease in urban areas. These are important goals for watershed managers to keep in mind, but the need to preserve ecosystem health is becoming more important with the degradation of aquatic ecosystems (Walsh *et al*. 2005).

There is a still a great need for future research that explores the question of how large scale changes within watersheds are affecting the local habitats within streams (Kearns *et al*. 2005).

Urbanization has a great impact on the "patterns of ecologic structure and function" (Walsh 2006) and should be taken into consideration by watershed managers. Consistent methods are also needed in order to monitor the changing environment within streams (Kearns *et al.* 2005) and preserve habitat for organisms such as fish and benthic macroinvertebrates (ACCWP 2004).

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Appendix I

Habitat scores for ten parameters described in Barbour *et al*. 1999. Each parameter is scored on a scale from zero to 20 with 20 being a high score. When parameters are divided among left bank and right bank, each bank is scored on a scale of zero to ten. A habitat rating is determined for each site based on overall score.

Stream Name	Codornices Creek			Sausal Creek			Strawberry Creek		
Location	Downstream	Midstream	Upstream	Downstream	Midstream	Upstream	Downstream	Midstream	Upstream
Habitat Parameter									
Epifaunal Substrate and Available Cover	11	7	15	8	9	11	12	16	15
Embeddedness	14	15	14	13	12	17	17	15	15
Velocity and Depth Regime	11	10	10	9	9	9	14	14	10
Sediment Deposition	9	8	8	11	4	11	6	9	11
Channel Flow Status	9	7	5	1	3	4	8	4	4
Channel Alteration	13	4	14	11	8	14	5	10	19
Frequency of Riffles	12	14	16	14	16	18	17	17	18
Bank Stability (Left)	5	4	8	6	6	6	6	4	7
Bank Stability (Right)	4	4	8	7	7	7	8	3	7
Vegetative Protection (Left)	7	1	5	9	7	8	4	4	8
Vegetative Protection (Right)	7	1	5	9	6	8	6	2	8
Riparian Vegetative Zone Width (Left)	8	1	6	1	3	9	3	6	9
Riparian Vegetative Zone Width (Right)	7	1	6	1	1	9	3	5	9
Total Score	117	77	120	100	91	131	109	109	140
Habitat Rating	Suboptimal	Marginal	Suboptimal	Marginal	Marginal	Suboptimal	Suboptimal/ Marginal	Suboptimal/ Marginal	Suboptimal

Appendix II

List of the average number of organisms at each site ± the standard deviation. Dashes indicate taxa not present at a particular site. The names of sensitive taxa (in the orders Ephemeroptera, Plecoptera, and Trichoptera) are highlighted in gray.

Stream Name	Codornices Creek			Sausal Creek			Strawberry Creek		
Location	Downstream	Midstream	Upstream	Downstream	Midstream	Upstream	Downstream	Midstream	Upstream
Taxa									
Elmidae	--	--	--	0.5 ± 0.71	18.5 ± 26.16	8 ± 8.49	--	7.5 ± 0.71	1.5 ± 0.71
Hydrophilidae	--	--	--	--	--	0.5 ± 0.71	--	--	0.5 ± 0.71
Psephenid	--	--	--	--	--	0.5 ± 0.71	--	1 ± 1.41	0.5 ± 0.71
Dytiscidae	--	--	--	0.5 ± 0.71	--	--	--	--	--
Astacidae	1 ± 1.41	--	--	--	--	--	--	--	--
Ceratopogonidae	--	0.5 ± 0.71	--	--	--	1 ± 0	--	0.5 ± 0.71	--
Empididae	--	7.5 ± 3.54	0.5 ± 0.71	2.5 ± 3.54	0.5 ± 0.71	4 ± 5.66	0.5 ± 0.71	2.5 ± 3.54	2.5 ± 0.71
Ephydridae	--	3.5 ± 3.54	--	--	1 ± 1.41	--	--	--	--
Psychodidae	--	3.5 ± 0.71	0.5 ± 0.71	--	1 ± 1.41	1 ± 0	0.5 ± 0.71	10.5 ± 14.85	2.5 ± 0.71
Dixidae	0.5 ± 0.71	--	0.5 ± 0.71	--	0.5 ± 0.71	1.5 ± 0.71	--	--	--
Chironomidae	39 ± 5.66	178.5 ± 55.86	23.5 ± 20.51	93 ± 32.53	23.5 ± 7.78	64.5 ± 55.86	19 ± 0	98.5 ± 64.35	61 ± 35.36
Simuliidae	0.5 ± 0.71	4 ±0	29.5 ± 21.92	--	2 ± 0	15.5 ± 12.02	--	5.5 ± 7.78	15.5 ± 10.61
Tipulidae	--	3 ± 2.83	1 ± 1.41	--	--	1.5 ± 0.71	--	1 ± 1.41	1.5 ± 2.12
Baetidae	--	15 ± 11.31	11 ± 9.90	126 ± 12.73	39.5 ± 51.61	22.5 ± 27.58	3.5 ± 3.54	34.5 ± 43.13	24.5 ± 0.71
Physidae	174 ± 59.40	19 ± 19.80	4.5 ± 0.71	60 ± 56.57	91.5 ± 106.77	1 ± 1.41	--	--	--
Planorbidae	10.5 ± 0.71	23 ± 28.28	26.5 ± 37.48	1 ± 1.41	9.5 ± 2.12	3.5 ± 3.54	1 ± 1.41	1 ± 1.41	--
Hydrobiidae	54 ± 28.28	230.5 ± 301.93	274 ± 50.91	12.5 17.68	79.5 ± 106.77	1 ± 1.41	1.5 ± 2.12	2.5 ± 2.12	--
Lymnaeidae	7 ± 9.90	1 ± 1.41	3.5 ± 4.95	--	3 ± 1.41	0.5 ± 0.71	0.5 ± 0.71	1 ± 1.41	--
Cordulegastridae	--	--	--	--	--	--	--	--	0.5 ± 0.71
Coenagrionidae	7.5 ± 6.36	14.5 ± 13.44	25.5 ± 27.58	2 ± 2.83	--	1 ± 0	--	19.5 ± 26.16	--
Aeshnidae	--	--	4 ± 5.66	2 ± 2.83	5 ± 7.07	0.5 ± 0.71	--	0.5 ± 0.71	--
Nemouridae	--	31 ± 22.63	5 ± 4.24	4.5 ± 0.71	29.5 ± 41.72	16.5 ± 0.71	--	94.5 ± 85.56	9.5 ± 10.61
Glossomatidae	--	--	--	--	--	--	0.5 ± 0.71	--	--
Brachycentridae	--	1.5 ± 0.71	3.5 ± 4.95	0.5 ± 0.71	10 ± 14.14	70.5 ± 47.38	--	8 ± 0	57 ± 62.23
Hydroptilidae	--	0.5 ± 0.71	--	1 ± 0	--	--	0.5 ± 0.71	--	--
Hydropsycidae	--	--	1.5 ± 2.12	--	--	--	--	1 ± 0	0.5 ± 0.71
Rhyacophilidae	--	--	--	--	--	--	--	--	0.5 ± 0.71
Oligochaeta	5 ± 2.82	2.5 ± 2.12	2.5 ± 3.54	18 ± 18.38	5.5 ± 7.78	8.5 ± 6.36	3.5 ± 2.12	9.5 ± 2.12	--
Turbellaria	84.5 ± 16.26	13 ± 5.67	9 ± 12.73	22.5 ± 4.95	8.5 ± 0.71	2 ± 1.41	0.5 ± 0.71	12 ± 11.31	27.5 ± 7.78
Amphipoda	133 ± 103.24	0.5 ± 0.71	--	28 ± 26.87	7.5 ± 10.61	--	18 ± 1.41	55.5 ± 38.89	0.5 ± 0.71
Bivalvia	29.5 ± 14.85	11.5 ± 14.85	--	--	--	--	--	3 ± 2.82	4 ± 5.66
Ostracoda	120.5 ± 70.00	18 ± 16.97	--	35.5 ± 9.19	11 ± 0	--	--	61.5 ± 84.15	1 ± 1.41
Acari	5 ± 1.41	8.5 ± 9.19	10.5 ± 0.71	13.5 ± 0.71	11 ± 8.49	6 ± 4.24	1 ± 1.41	0.5 ± 0.71	0.5 ± 0.71
Terrestrial	2.5 ± 0.71	2.0 ± 1.41	1.5 ± 2.12	0.5 ± 0.71	3 ± 1.41	6.5 ± 6.36	3.5 ± 0.71	2 ± 1.41	2.5 ± 0.71
Worm Other	42 ± 21.21	55 ± 15.56	15.5 ± 14.85	30.5 ± 33.23	5.5 ± 7.78	16 ± 5.66	--	28.5 ± 40.31	18 ± 7.07
Thumaleidae	--	--	0.5 ± 0.71	--	--	--	--	--	--

Appendix III

Average score (± SD) for ten biological metrics calculated by site.

Stream Name	Codornices Creek			Sausal Creek			Strawberry Creek		
Location	Downstream	Midstream	Upstream	Downstream	Midstream	Upstream	Downstream	Midstream	Upstream
Metrics									
Total Number of Individuals	716 ± 236.27	647.5 ± 234.05	454 ± 103.24	454.5 ± 41.72	366.5 ± 385.37	254 ± 19.80	54 ± 8.49	462 ± 253.14	232 ± 7.07
Taxa Richness	15 ± 1.41	22 ± 1.41	15.5 ± 2.12	15.5 ± 0.71	16.5 ± 4.95	20.5 ± 0.71	9.5 ± 2.12	19.5 ± 3.54	16 ± 0
% Oligocheata	0.67% ± .17%	0.35% ± 0.20%	0.47% ± 0.67%	3.79% ± 3.70%	0.86% ± 1.22%	3.26% ± 2.25%	6.25% ± 2.95%	2.27% ± 0.79%	0.00%
% Chironomidae	5.89% ± 2.73%	31.16% ± 19.89%	4.79% ± 3.43%	20.88% ± 9.07%	11.84% ± 10.33%	26.33% ± 24.05%	35.63% ± 5.60%	20.60% ± 2.64%	26.07% ± 14.44%
% EPT	0.0%	8.23% ± 4.50%	4.57% ± 0.52%	29.29% ± 5.49%	13.72% ± 14.90%	42.10% ± 25.95%	7.71% ± 7.95%	26.17% ± 13.52%	40.01% ± 23.26%
% EPT – Baetidae and Hydropsychidae	0.0%	6.09% ± 5.48	1.94% ± 0.60%	1.33% ± 0.12%	6.18% ± 8.74%	33.64% ± 15.75%	1.67% ± 2.36%	20.14% ± 7.49	29.23% ± 22.84%
EPT Richness	0.0	3.5 ± 0.71	3 ± 1.41	3.5 ± 0.71	2 ± 1.41	3 ± 0	2 ± 1.41	4 ± 0	4 ± 0
Shannon Diversity	2.10 ± 0.17	1.82 ± 0.15	1.44 ± 0.74	2.03 ± 0.16	2.24 ± 0.04	2.04 ± 0.05	1.64 ± 0.35	2.02 ± 0.25	1.93 ± 0.13
Evenness	0.79 ± 0.058	0.59 ± 0.06	0.52 ± 0.24	0.74 ± 0.04	0.81 ± 0.10	0.68 ± 0.01	0.73 ± 0.08	0.68 ± 0.04	0.70 ± 0.05
FBI	5.61 ± 0.58	4.98 ± 0.08	7.01 ± 0.51	5.14 ± 0.81	5.78 ± 0.33	3.53 ± 0.64	4.73 ± 0.08	5.03 ± 1.21	3.63 ± 0.93