BIOLOGICAL ASSESSMENT OF URBAN AND AGRICULTURAL STREAMS IN THE CALIFORNIA CENTRAL VALLEY (FALL 2002 THROUGH SPRING 2004)

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"Chemical measurements are like taking snapshots of the ecosystem, whereas biological measurements are like making a videotape."
Prof. David M. Rosenberg Ph.D., Univ. of Manitoba

ABSTRACT

This project was designed to establish baseline aquatic biological community structure and physical habitat conditions in select wadeable streams within the California Central Valley. A secondary objective was to evaluate possible water quality differences between site types and seasons. Two agricultural and two urban streams were monitored in spring and fall for two consecutive years beginning in the fall of 2002. Bioassessment sampling was conducted according to modified U.S. EPA methods. The study included physical habitat assessment, water and sediment chemical analysis and characterization of the benthic macroinvertebrate community at each site. Water samples were analyzed for selected organophosphate insecticides, pyrethroid insecticides and herbicides, while sediment samples were analyzed for pyrethroids only. All sites had substantial physical habitat and water quality impairments, and the absence of pollution intolerant macroinvertebrates and dominance of pollution tolerant macroinvertebrates were indications of biological impairment. Due to the limited amount of water quality and pesticide data collected, it was not possible to definitively demonstrate any cause and effect relationships between BMI community structure and water quality or pesticide concentrations. There were no significant differences in physical habitat between urban and agricultural sites ($p=0.354$). Dominant taxon found at all sites were Chironomids, Amphipods, and Oligochaetes. Benthic macroinvertebrate metrics were significantly different between both types of sites ($p=0.002$) and seasons ($p=0.036$). Chironomidae taxon and those of the functional feeding group scrappers were greater at urban sites, while those of the functional feeding group filterers were greater at agricultural sites. In addition, all three of these metric groups were found in greater numbers in the spring than the fall.

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1. Introduction

California has over 200,000 miles of rivers and streams. Determining the integrity or current condition of these waters and the aquatic environment within is challenging. Biological assessment, or bioassessment, is a common method used to determine current conditions of a water body. Bioassessment is a quantitative survey of the physical habitat and biological community of a water body. By examining the biological community rather than one species, a more comprehensive survey of the health of a water system can be determined. They provide a time-integrated measure of overall water quality conditions at an aquatic site.

Aquatic benthic macroinvertebrates (BMI) are most often used to represent the biological community. Not only are BMIs ubiquitous, but BMI surveys are not labor intensive in comparison to fish and algae surveys, and can be very informative. BMIs are affected by changes in a stream's chemical and/or physical structure because they complete the majority of their life cycle in water and are relatively stationary. The variety of species and population sizes present in the stream are reflective of the overall health of that biological community and can be used as water quality indicators (SWRCB, 2001).

Bioassessment has been conducted at over 3000 sites throughout the state by various agencies, universities and other entities (Tetra Tech, 2003). The California Department of Water Resources has collected bioassessment data since 1975, while the United States Geologic Survey began its long-term program in 1992 as part of the National Water Quality Assessment Program. The California Department of Fish and Game also began conducting projects in 1992, and has developed standard protocols for bioassessment based on the U.S. EPA Rapid Bioassessment Protocols. The State Water Resources Control Board (SWRCB) and its nine regional boards are responsible for implementing water quality standards for the state of California. Within the past five years, they also have begun to use bioassessment practices in their monitoring programs.

DPR's Surface Water Protection Program developed this pilot project to familiarize staff with current bioassessment monitoring methods and to establish baseline aquatic biological community structure and physical habitat conditions in select wadeable streams within the Central Valley. An additional objective of this study was to evaluate water quality parameter and BMI differences between site types and seasons. Monitoring was conducted in four streams that are tributaries to the Sacramento or San Joaquin rivers. Two are dominated by agriculture (AG) and two are dominated by urban development (UB). Monitoring occurred in the fall and spring for two consecutive years beginning in the fall of 2002.

In developing this project DPR collaborated with the Central Valley Regional Water Quality Board (CVRWQCB) by assisting them with their bioassessment monitoring and data collection needs. CVRWQCB staff are currently evaluating bioassessment as a water quality monitoring tool, with the hope that it's future application will be in a regulatory capacity (R. Holmes, personal communication, 2004). The CVRWQCB's Total Maximum Daily Load (TMDL) process includes provisions for use of bioassessment (CVRWQCB, 2002a), as well as their Surface Water Ambient Monitoring Program (CVRWQCB, 2002b).

2. Materials and Methods

Site Description

This project targeted sites with a potential for anthropogenic impacts. Sites were selected using the following criteria:

- Receives drainage from AG or UB runoff
- Has a history of previous pesticide detections
- Had no current condition assessment

Four creeks were selected: two UB-dominated creeks (Elder and Elk Grove) in the vicinity of Elk Grove, California and two AG-dominated creeks (Little Johns and Lone Tree) in the vicinity of Stockton, California (Figures 1-3). Each creek selected had two sampling sites. Each sampling site consisted of a 100-meter long section of the creek (reach). Year-round water flow has been observed at all sites.

Sampling Plan

Monitoring was conducted in the fall and spring in order to collect information on seasonal variation. Habitat modifications and pesticides are potential stressors that can impact BMI populations. Therefore, a physical habitat assessment was completed for each reach, along with the collection and analysis of water, sediment and BMI samples. Water samples were analyzed for selected organophosphates (OPs), pyrethroids (PY) and herbicides, and sediment samples were analyzed for pyrethroids (Table 1). Some of the listed pesticides had been previously detected in these streams.

Sampling Method

Each site or reach was selected based on available access using a non-point source design. This design is used when there is no obvious point of discharge into the stream. Typically, several sampling reaches are selected to better assess the entire stream. Due to logistics and funding, two sites were selected on each stream for this study.

BMI's were collected using a D-framed kick net with a mesh of 0.5mm. Sampling was conducted in accordance with the U.S. EPA modified multi-habitat method SOP FSWA010.00 (Bacey, 2003) during year one and the U.S. EPA modified EMAP method SOP FSWA015.00 (Bacey and Moncada, 2004) during year two.

A physical habitat assessment was conducted within each reach during each sampling event. Physical habitat scores were determined using U.S. EPA national standardized methods (U.S.EPA, 2001). Other parameters measured at each site included gradient, canopy cover, depth, substrate particle size and substrate embeddedness (Bacey and Moncada, 2004). Water quality parameters included temperature, specific conductance (EC), dissolved oxygen (DO), pH, and turbidity.

Water samples were individually collected in 1-liter amber glass bottles from as close to center channel as possible, and were sealed with Teflon-lined lids.

One sediment sample was collected from each site, during each sampling event. These were collected using a 24-inch long, 2-inch diameter, polycarbonate cylinder tube. One end of the tube was thrust into the sediment and then removed.

The top 2.0 cm of the sediment collected in the tube was placed into a clear 1-pint glass jar. This was repeated several times in the same general area and composited.

Water and sediment samples were transported on wet ice. Water samples were stored refrigerated at 4^{o}C until extraction for chemical analysis and sediment samples were stored frozen at –14^{o}C until extraction for chemical analysis.

Benthic Macroinvertebrate Identification
The Bidwell Environmental Institute of California State University, Chico, conducted identification of BMIs. Quality control was conducted in accordance with previously established California Department of Fish and Game procedures. For analysis of each sample, a random sub-sample of 500 macroinvertebrates were identified as to genera and, when possible, species.

Pesticide Analysis
The California Department of Food and Agriculture's Center for Analytical Chemistry performed chemical analyses. Quality control was conducted in accordance with standard Department of Pesticide Regulation QC procedures (Segawa, 1995), and included approximately 10 percent of samples as blind spikes. Samples with no residue above the MDL are reported as non-detections. Samples with a residue concentration that falls between the RL and the MDL are reported as trace detections. The analytical chemist uses his/her best professional judgment to make this determination. Samples with residues above the RL are detections and analytical concentrations are reported.

Pyrethroid whole water samples, including any suspended sediment, were extracted *in toto* with methylene chloride. Sample bottles were rinsed with extraction solvent and added to the sample extracts for analysis. The extract was passed through sodium sulfate to remove residual water. The anhydrous extract was evaporated on a rotary evaporator and then a solvent exchange performed with hexane. Extracts were concentrated using a Brinkmann R110 rotary evaporator (Brinkmann, Westbury, NY), and analyzed using a gas chromatograph (GC) equipped with a HP-1 column (Hewlett Packard, Avondale, PA) and an electron capture detector (ECD). Pyrethroid analysis results are reported on a whole water basis (water plus suspended sediment). Reporting limits were 0.005 to 0.08 µg/L.

Pyrethroid sediment samples were homogenized and extracted with acetonitrile. The filtered extracts were salted out with sodium chloride. An aliquot of acetonitrile extract was evaporated to dryness in a water bath under a stream of nitrogen for solvent exchange to hexane. Extracts were analyzed using GC with ECD. Pyrethroids in sediment were confirmed using GC equipped with a mass selective detector. Reporting limits were 0.01 µg/g.

Organophosphate samples were extracted with methylene chloride and the extract was passed through sodium sulfate to remove residual water. The anhydrous extract was evaporated to near dryness on a rotary evaporator and diluted to a final volume of 1.0 mL with acetone.

The extract was then analyzed by a Hewlett-Packard model 5890 GC (Hewlett Packard, Avondale, PA) equipped with an Rtx OP Pesticides column (Restek, State College, PA) and a flame photometric detector. Reporting limits were 0.03 to 0.05 µg/L.

For herbicide analysis, the water samples were passed through two Oasis MCX cartridges (Waters, Millford, MA) connected in tandem. The cartridges were then eluted under vacuum with 5 percent ammonium hydroxide in methanol. The eluant was filtered through a nylon Acrodisc 0.2-micron filter (Gelman Sciences, Ann Arbor, MI), concentrated, reconstituted in 75/25 water/methanol and analyzed by a ThermoQuest/ThermoSeparation HPLC with a Finnigan LCQ Deca mass spectrometer (Finnigan/ThermoQuest, San Jose, CA). Reporting limits were 0.05 µg/L.

Data Analysis

BMI taxa were summarized into biological metrics (Table I). Physical habitat and water quality parameters, as well as BMI metrics were transformed using a $\log_{10}(x + 1)$ or arcsin (sq. root x) transformation as required to meet assumptions of normality and/or homogeneity of variance (Townsend, 2002). Differences between AG and UB sites and seasonality were examined using two-sample t-tests, paired t-tests and multivariate analyses. Differences were considered significant at $p < 0.05$.

Table 3. Benthic macroinvertebrate metrics and definitions

Taxonomic Richness	Total number of individual taxa
Percent Dominant Taxon	Percent of organisms in sample that is the single most abundant taxon
EPT Taxa	Number of families in the Ephemeroptera (mayfly), Plecoptera (stonefly) and Trichoptera (caddisfly) insect orders
EPT Index	Percent of organisms in sample that consists of Ephemeroptera, Plecoptera, and Trichoptera (EPT)
Sensitive EPT Index	Percent of EPT in sample with tolerance values of 0 through 3
Tolerance Value	Value between 0 and 10 weighted for abundance of individuals designated as pollution tolerant (higher values) and intolerant (lower values)
Intolerant Taxa	Organisms in sample that are highly intolerant to impairment as indicated by a tolerance value of 0 through 2
Tolerant Taxa	Taxon-specific organisms in sample that are highly tolerant to impairment as indicated by a tolerance value of 8 through 10
Chironomidae	Of the order Diptera (true flies) mainly consisting of midges
Collectors	A total of BMIs of the groups Gatherers and Filterers
Gatherers	BMIs that collect or gather fine particulate matter
Filterers	BMIs that filter fine particulate matter
Scrapers	BMIs that graze upon periphyton
Predators	BMIs that feed on other organisms
Shredders	BMIs that shred coarse particulate matter
Modified from Harrington and Born, 1999	

The **Tolerance Value** reflects a community level tolerance. This metric was originally designed to serve as a measure of community tolerance to organic pollution (decaying plants and animals, manure, sewage). The regionally specific tolerance values for BMI communities in the Pacific Northwest are used here (CAMLnet, 2003). In addition, the EPA has established a list of tolerance values applicable to BMI communities in the northwestern U.S. based on their bioassessment program in Idaho.
If a taxon found in California is not assigned a value in the Pacific Northwest, then this EPA value is used. A moderately disturbed stream typically has a tolerance value in the mid-range values (Harrington and Born, 1999).

The number of **Chironomid** species found in most water systems usually accounts for 50% of the total BMI species richness (Merritt and Cummins, 1996). Chironomids occur in most aquatic ecosystems, tolerating a wide range of conditions (i.e. temperature, pH, salinity, oxygen concentration). They are also tolerant to water pollution, and in general their dominance at a site may indicate increased nutrients (Harrington and Born, 1999).

The **Functional Feeding Groups** (collectors, filterers, etc.) represent the processes or feeding habits of different macroinvertebrates in the stream. They also represent ecology production and food source availability within the stream. An imbalance of the feeding groups may reflect an unstable food process and indicate a stressed condition (Harrington and Born, 1999).

3. Results and Discussion

Water Quality

Water quality data were collected twice annually at each sampling site, once in the spring (April) and once in the fall (October) during the two-year study. Therefore, there were four measurments of each water quality parameter at each site with the exception of turbidity; turbidity was measured during the second year only. No data were collected during winter or summer seasons. Consequently, because sampling was limited, no definitive "cause and effect" relationships between water quality and BMI metrics are identified. These data are presented here to show water quality conditions during the BMI sampling periods, indicate which water quality parameters may be impacting BMIs, and to show possible differences in water quality parameters between sites and or seasons.

Flow measurements showed that only one of the four creeks surveyed had flow greater than 0.03 ft/sec, the minimum detectable flow velocity of the current meter. Flow at that site (Lone Tree creek) never measured greater than 1.0 ft/sec. Water temperature and turbidity measured at the time of BMI sampling were within acceptable ranges for BMIs (Table 4). Water quality measurements by site and season are presented in Figure 4. Extreme water temperatures can have a negative effect on such things as BMI hatching success, larval growth, and emergence.

Dissolved oxygen concentrations were below the U.S. EPA national warm water quality criteria of 5 mg/L (1986) in 34 percent of the samples. Nine of the eleven low DO samples were from UB sites, while two were from AG sites. Very low DO measurements were most likely due to the stagnant, warm conditions of these creeks. DO levels below 3 mg/L can be stressful to most aquatic organisms, while levels below 2 or 1 mg/L will not support fish. DO levels at 4mg/L or less are toxic to many species of invertebrates (U.S. EPA, 1986).

The SWRCB freshwater aquatic life criterion for pH is 6.5 to 9.0 (2003). Courtney and Clements (2000) found that in some aquatic environments, the abundance of mayflies could be significantly lower in an acidic environment (pH 4.5). In this study, pH measurements were never measured below 5.5, but were below 6.5 in 10 percent of the samples.

Table 4. Water quality measurements of California central valley monitoring sites, 2002-2004, urban vs. agricultural and fall vs. spring.

Parameter	Urban Sites	Mean	Median		Agric. Sites	Mean	Median
Normal range	Range				Range		
Temperature (°C) <35°C[2]	8.7 – 26.5	16.27	16.9		13.4 – 19.7	16.47	15.9
Dissolved Oxygen (mg/L) 5 mg/L[3] (min. range)	1.2 – 10.9	5.97	6.34		3.05 – 9.95	6.97	7.25
pH 6.5 – 9.0[4]	5.5 – 8.9	7.47	7.57		6.13 – 9.48	7.29	7.23
Specific Conductance (µS/cm) 150 – 500µS/cm[5]	208.1 – 615	358.6	301.8		60.1 - 839	210.4	125.9
Turbidity (NTU) 0 – 100[4]	25 – 90	48.63	38.5		1 - 99	43.25	38.5

Parameter	Fall Sites	Mean	Median		Spring Sites	Mean	Median
Normal range	Range				Range		
Temperature (°C) <35°C[2]	14.9 – 19.7	17.35	17.5		13.4 – 18.6	15.57	15.55
Dissolved Oxygen (mg/L) 5 mg/L[2]	3.05 – 7.87	5.84	6.28		5.73 – 9.95	8.10	8.32
pH 6.5 – 9.0[4]	6.56 – 9.48	7.34	7.12		6.13 – 7.95	7.24	7.3
Specific Conductance (µS/cm) 150 – 500µS/cm[5]	60.1 – 243.7	138.9	109.2		84.9 - 839	281.9	189.4
Turbidity (NTU) 0 – 100[4]	38 - 99	61	53.5		1 - 39	25.5	31

1. N=16 for each site type (AG and UB), except for turbidity where N=8 each site type
2. Aquatic macroinvertebrae requirements (various sources)
3. 7-day mean minimum, freshwater mixed fisheries & BMIs. (U.S.EPA, 1986)
4. Freshwater aquatic life criteria (SWRCB, 2003)
5. Supports freshwater mixed fisheries & BMIs (U.S. EPA, 2005)

U.S. EPA guidelines for specific conductance (EC) in streams supporting good mixed fisheries are 150 to 500µS/cm. Conductivity outside this range may not be suitable for certain species of fish or macroinvertebrates (U.S. EPA, 2005). In this study EC was measured below 150µS/cm in 30 percent of the samples. Nine of the these ten samples were at AG sites while one was at a UB site. Another ten percent of these measurements were measured above 500µS/cm. In a recent study, Brown and May (2004) suggested that EC may be an important parameter affecting the composition of macroinvertebrate communities. Chambers and Messinger (2001) found that the number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa present was negatively correlated with rising EC levels.

Physical Habitat

Baseline physical habitat (PHab) conditions were determined using varied physical habitat parameters such as: flow, canopy coverage and substrate size. All PHab parameters are listed in Attachments A-D. Only one of the four creeks surveyed had flow greater than 0.3 ft/sec, the minimum detectable flow velocity of the current meter. In no case were measured flow velocities greater than 1 ft/sec. Of the four UB sites, only one site had a density of overhead canopy coverage of 2%, the other three sites had 0% canopy coverage.

Of the AG sites, two sites had 30 and 68% canopy cover (different creeks) while the remaining two had 6% and 0%. Other PHab data that were similar at all sites with little variability were elevation (50-110 ft) and gradient (0-2 percent slope).

Substrate of a stream can consist of inorganic matter (large boulders, cobble of various sizes, gravel, sand, fine silt or mud, clay) and particulate organic matter (detritus). A description of substrate types and sizes is given in Appendix B. Substrate particle size was determined by visually inspecting substrate at 55 points within the sampling reach, following a modified U.S. EPA method (Bacey and Moncada, 2004). Substrate particle size is important because many pollution intolerant taxa require open interstitial spaces in the substrate. Chambers and Messinger (2001) found the numbers of EPT taxa in a sample were positively correlated with median particle size. Though both types of sites in this study had high amounts of fine silt or mud, AG sites had a wider variety of substrate than UB sites (Figure 5).

Sediment, a mixture of fine organic and inorganic substrates, is the dominant non-point source pollutant of U.S. rivers (Simon, 2002). When there is significant sediment in a stream, substrate embeddedness can occur. Substrate embeddedness is the degree to which substrate such as cobble or boulders are firmly surrounded by finer organic or inorganic materials such as sand or mud. Embeddedness was determined by visually inspecting substrate at 55 points within the sampling reach, following a modified U.S. EPA method (Bacey and Moncada, 2004; Appendix B). All of the UB sites surveyed were 98% or greater embedded, with the main cause being fine silt or mud. Three of the four AG sites fell within the 80th percentile range of embeddedness; the fourth had a mean embeddedness of 99%. Embeddedness at the AG sites ranged from 53 to 100 percent. Heavy sedimentation within a stream will decrease insect diversity and growth (Resh and Rosenberg 377; Table 5).

In addition to the physical habitat data, a habitat assessment for low gradient streams was conducted (U.S. EPA, 2001, Appendix D). The physical habitat assessment score ranges from 0 to 200, with 0 being the most impacted by anthropogenic activities and 200 being the least impacted. The score can be subjective due to the experience of the individual making the assessment. During this study, several individuals assessed the physical habitat of each site a total of four times. This allowed us to mitigate the subjective nature of the assessment, and enabled us to determine a mean score for each site. In determining this score ten habitat parameters were used. The first four were: variability of pools within the reach, sediment deposition, channel sinuosity, and the width of the riparian vegetative zone beyond the bank of the stream. All of these parameters received the lowest score (poor = 0-5) in the majority of sites, which greatly dominated on the overall score. The other six parameters were: favorable habitat available for epifaunal colonization, characterization of substrate in pools, water level within channel, channel alteration, bank stability, and streambank vegetation coverage. All UB and AG sites had a low mean PHab score of <100 (Table 5). Statistical analysis was performed on the transformed means of the PHab scores (to achieve normality). There was no significant difference in Physical habitat between AG and UB sites based on a two-sample t-test of the transformed scores ($p = 0.354$).

Table 5. Physical habitat measurements of California central valley monitoring sites, 2002-2004, urban vs. agricultural.

Parameter	Urban Sites	Mean	Median	Agric. Sites	Mean	Median
	Range			Range		
Substrate Embeddedness	98 – 100%	99.5	100	53 – 100%	84.63	88
Physical Habitat Score	56 – 100	71	66	34 – 124	81.5	75.5

Pesticide Detections

Water and sediment pesticide samples were collected four times over a two-year period at each site. Consequently, the sampling data are inadequate for characterizing the temporal and seasonal variation of pesticide concentrations in the creeks. Generally, BMI communities are affected by changes in a stream's chemical and/or physical structure. Since they inhabit a water system for the majority of their life cycle, they often reflect the effects of their environment over time. However, the limited pesticide monitoring conducted for this study is not sufficient to show any potential long-term cumulative exposure of the BMI communities, therefore no definitive "cause and effect" relationships between pesticides and BMI metrics are identified. The only detection of a pesticide in a sediment sample was a trace detection of permethrin in a UB site at Elk Grove creek. Trace organophosphate (OP) detections in water occurred at two AG sites (Little John Creek), while there were no OP detections at the AG sites at Lone Tree Creek. OP detections in water occurred at every UB site (trace to 0.212 μg/L), suggesting the potential for impacts on the BMI communities

Several studies have reported the effects of pesticides, especially insecticides, on BMI communities. Anderson et al. (2003) collected water samples downstream of an agricultural drain and found that toxicity to *Ceriodaphnia dubia* was highly correlated with combined toxic units of the organophosphates (OP) chlorpyrifos and diazinon. He also found that BMI metrics were negatively correlated with combined toxic units of chlorpyrifos and diazinon. Schulz and Liess (1999) found that pyrethroid insecticide runoff during summer irrigation season resulted in a negative effect of the BMI community. Eight of eleven common species disappeared for a period of 3 to 6 months while the 3 other species were significantly reduced after an initial runoff event (fenvalerate concentration of 0.10μg/L). The aquatic stage of the 11 species would normally have continued during that period. The disappearance of the species was attributed to a drift response to fenvalerate by BMIs.

In this study, pyrethroids and OPs were detected in the water column. Either acute or long-term chronic exposures to these two classes of pesticides can cause negative effects to BMI communities (Coates, et al, 1989, McCutchan, 2000). OP detections occurred at every sampling period at every UB site, suggesting that potential for impacts on the BMI communities. However, analytical results from samples collected biannually are insufficient to quantify the pesticide effects on BMI communities, or to distinguish between effects due to other factors such as low DO or high EC. All detections are presented in Table 7.

Benthic Macroinvertebrates

BMI results have been summarized and select biological metrics are presented in Table 8. Metrics include classification of BMIs by select taxon, abundance and by feeding habits (functional feeding groups (FFG)).

In general, a healthy stream, that which is cool, clean and highly oxygenated (i.e. headwater stream) may have only a few dominant species due to the cold temperature and low nutrient levels (Peckarsky et al. 1990). As temperatures and nutrients increase further downstream taxa richness will increase as well. The following indicator BMIs are often dominate in clean to moderately polluted waters: Ephemeroptera, Plecoptera, and Tricoptera taxa (EPT), as well as Amphipoda and Odonata taxa (Peckarsky et al. 1990, Pennak, 1989). Some families of Plecoptera and Ephemeroptera can be highly sensitive to pesticides, with many species within the EPT taxa orders being highly intolerant to pollutants.

Those indicator BMIs that dominate in fairly to severely polluted waters include: Oligochaetes, Diptera (chironomidae), and Gastropoda (physa spp) (Peckarsky et al. 1990, Pennak, 1989). Oligochaeta are aquatic worms that tolerate low levels of oxygen and are generally found in large numbers in organically polluted habitats (i.e. sewage, manure, decaying vegetation). Chironomidae occur in large populations in all habitat types. The number of Chironomid species found in most water systems usually accounts for 50% of the total BMI species (Merritt and Cummins, 1996). They tolerate a wide range of conditions (i.e. temperature, pH, salinity, oxygen concentration), and are also tolerant to water pollution. Their dominance at a site may indicate increased inorganic pollutants (i.e. nutrients) (Harrington and Born, 1999). The snail species Gastropoda, *Physa spp.*, is a low-oxygen indicator species. It has lungs and can tolerate water conditions with little or no oxygen (Pennak, 1989). It is normally found in such conditions.

In this study, EPT taxa were found at only one of the four UB sites and the number of families never exceeded two during the four sampling events (ephemeroptera and trichoptera). At the AG sites, Ephemeroptera were found at two of the four sites (both on the same creek), but not consistently over the length of the study (). On each occasion taxa consisted of 1 to 4 families. No plecoptera taxa were found at any of the sites. The dominant taxa found at UB sites were the pollution tolerant Chironomidae, Oligochaetes, and Amphipoda. Chironomidae accounted for 49 percent of the total BMIs and Oligochaetes accounted for 13 percent. Amphipoda accounted for 30 percent of the total at one site only (Elder Creek). At the AG sites, Oligochaetes accounted for 25 percent of the total BMIs and Chironomidae accounted for 9 percent. While at one AG creek, Amphipoda accounted for 40 percent of the total BMIs (Lone Tree Creek). BMI indicator taxa results are presented in Figures 6-7.

It's important to note that the absence of a particular family or species of aquatic insect may be due to the presence of a restricting pollutant such as high nitrate or pesticide, or due to other factors such as habitat limitations. Habitat limitations may be due to either anthropogenic impacts or naturally occurring conditions. One example is fingernet caddisflies of the family philopotamidae (order trichoptera). These organisms are restricted to riffle areas of streams. None of the streams surveyed in this study contained riffles. Historical data is not available for these sites, so whether riffles were once present and have vanished due to anthropogenic activities is unknown.

Metrics of FFGs are related to stream resources. An imbalance in FFGs will result when food resources are not stable, reflecting stressed conditions (U.S. EPA, 1999).

In this study it was evident that the FFGs, collectors and gatherers, (i.e. some chironomidae, amphipoda, and oligochaeta) were dominant at the majority of sites (up to 95%, Figures 8-9). Filterers (i.e. some Chironomidae and Mollusca) were also relatively abundant at a few sites (e.g., Little John creek-AG, Elder Creek-UB). These groups feed on fine particulate matter. They are considered generalists, accepting a broader range of food materials than specialists and are more tolerant to pollution (U.S. EPA, 1999). The physical habitat assessments revealed that fine sediments and organic matter were the dominant substrate at all sites. These conditions explain the high numbers of gatherers and filterers. Other dominant taxa included predators (Elk Grove creek-UB, up to 75%), which may increase in response to gatherer numbers if habitat conditions are suitable.

Scrapers (i.e. gastropoda taxa) and shredders (i.e. some diptera spp) are specialized feeders and are more sensitive organisms than other FFGs. In healthy streams they will be well represented (U.S. EPA, 1999). Only during one sampling event did scrappers numbers reach 36% (Elder creek-UB). The remainder of the events and sites had no greater than 5%. The primary food source of scrappers is attached periphyton (i.e. algae) on submerged substrates. Shredder populations did not exceed 4 percent of the total community, or approximately less than 8 percent of the total invertebrate biomass, at any of the sites. Shredder populations in these low gradient streams (4-5 order streams) should represent approximately twenty percent of the total invertebrate biomass (Graca et al., 2001). Low shredder populations may be indicative of low coarse leaf litter. Coarse particulate organic matter (CPOM) is the primary food source of shredders. CPOM consists of organic matter such as twigs, leaves and flowers that are greater than 1mm in diameters. Low amounts of CPOM are reflected in the low physical habitat scores (low bank and riparian vegetation). The means of the BMI metrics are presented in Table 9.

A multiple analysis of variance of the transformed BMI metrics was used to determine potential differences between type of site (UB or AG), and season (spring or fall). Significant differences were found in each case (site type, $p=0.002$; season, $p=0.036$). Significant differences in filterers, scrappers, and chironomidae taxa were found between AG and UB sites in subsequent t-tests comparing BMI metrics ($p< 0.017$, 0.011, 0.001, respectively). Both scrappers and chironomidae taxa were greater in UB sites than AG sites, while filterers were greater in AG sites than UB sites (Figure 10). Likely explanations include increased algae growth, increased nutrients, or reduced oxygen levels in UB sites in comparison to AG sites, therefore providing a suitable habitat for scrappers and chironomidae. These conditions tend to occur if waters became stagnant or reduced due to reduced flows. In contrast, filterers, although very tolerant, require constant flowing waters in order to filter fine organic matter. Consequently, they would tend to have lower populations in the non- or very low-flowing UB sites relative to AG sites. Flow was never greater than 0.09 m/sec at UB sites while one of the two AG creeks had flow up to 0.3 m/sec.

Significant differences were seen in these same three metrics (filterers, scrappers, chironomidae) when comparing seasons (p-value = 0.00, 0.016, 0.002, respectively). All three were greater in the spring than the fall. Generally, water flows are greater in the spring than in the fall, offering optimal hydrologic conditions for BMI growth and development. No significant differences were seen in taxa richness between the various sites and seasons.

Table 9. Benthic macroinvertebrate metric means and standard deviation of California central valley monitoring sites, 2002-2004, urban vs. agricultural and fall vs. spring.

	Agric			Urban			Fall			Spring	
BMI Metric	**Mean**	**StDev**		**Mean**	**StDev**		**Mean**	**StDev**		**Mean**	**StDev**
Taxonomic Richness	22.06	8.15		19.69	11.93		19.94	11.24		21.81	9.14
EPT Taxa	1.188	1.642		0.313	0.602		0.938	1.063		0.563	1.504
Chironomidae	0.2731	0.2261		0.5763	0.2207		0.3419	0.2462		0.5075	0.2712
Collectors	0.8619	0.1652		0.71	0.1991		0.7513	0.1883		0.8206	0.203
Gatherers	0.7375	0.213		0.6681	0.2047		0.7106	0.2066		0.695	0.2166
Filterers	0.1244	0.1375		0.0419	0.0853		0.0406	0.0676		0.1256	0.1463
Scrapers	0.0094	0.0124		0.0631	0.0944		0.02	0.03502		0.0525	0.0938
Predators	0.1125	0.1625		0.1888	0.2121		0.2044	0.1875		0.0969	0.1821
Abundance	4174	9745		2880	6661		3669	9897		3386	6500

Conclusion

Due to the limited amount of water quality and pesticide data collected, it was not possible to definitively demonstrate any cause and effect relationships between BMI community structure and water quality or pesticide concentrations. However, the BMI metrics do indicate that the water bodies are impacted, and the water quality and Physical habitat data do suggest some potential reasons.

Water quality measurements were within normal aquatic life criteria ranges, with the exception of DO and EC. DO levels were below the accepted minimum (5 mg/L) numerous times in both UB and AG sites (44 percent and 15 percent, respectively). Such low levels are toxic to many species of invertebrates (U.S.EPA, 1986). EC exceeded the normal range (150-500 µs/cm; U.S.EPA, 2005a) at both AG and UB sites. Studies have shown that EC can be an important parameter affecting macroinvertebrate communities, negatively impacting pollution sensitive EPT taxa (Chambers and Messinger, 2001).

Physical habitat assessments of both AG and UB sites revealed highly impaired conditions. Both AG and UB sites contained high amounts of fine sediment (up to 100 percent). Substrate embeddedness ranged from 80 to 99 percent at both types of sites. U.S. EPA Physical habitat assessment scores revealed mean scores below 100 for all UB and AG sites, with the most significant parameters being, variability of pools within the reach, sediment deposition, channel sinuosity, and the width of the riparian vegetative zone beyond the bank of the stream.

Pesticides were detected at every site, while OP insecticides were detected at three of the four creeks monitored. Concentrations of the OP insecticide chlorpyrifos exceeded the EC50s of some sensitive aquatic species such as daphnia magna in several samples (0.1 – 1.7 ppb; U.S.EPA, 2005b).

The dominant taxa found at all sites were pollution tolerant Chironomidae, Oligochaeta and Amphipoda, while clean water taxa (EPT) represented less than 1 percent of the total BMIs. Of

the functional feeding groups, gatherers were dominant at all sites. There were significantly greater scrappers and chironomidae taxa in the UB sites than AG sites (p=0.011 and 0.001 respectively), while filterers were greater in AG sites than UB sites (p=0.017). Likely explanations include increased algae growth, increased nutrients, or reduced oxygen levels in UB sites in comparison to AG sites, therefore providing a suitable habitat for scrappers and chironomidae. These conditions tend to occur when waters are stagnant or have reduced flows. In contrast, filterers, although very tolerant, require constant flowing waters in order to filter fine organic matter. Consequently, they would not show the same increase in the non-flowing UB sites relative to AG sites as scrappers and Chironomidae.

In addition, a significant difference was observed between seasons for these same three metrics (filterers, scrappers and Chironomidae), with all three being greater in the spring than the fall (p=0.00, 0.016, 0.002, respectively). Generally, water flows are greater in the spring than in the fall, offering optimal hydrologic conditions for BMI growth and development.

The combined water quality, Physical habitat, pesticide, and BMI data suggest that a variety of factors may be impairing the biological community at all sites surveyed. There were significant differences in BMI taxa metrics for types of sites and seasons, but determining which parameters, or combination of parameters, may be causing these differences is difficult to determine. The results of this study have led to further research to identify impacts on the BMI community due to surface runoff of pesticides.

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Table 1. Method titles, method detection and reporting limits of OPs and herbicides

Organophosphate Pesticides in Water Method: GC/FPD			**Triazines/Herbicides in Water Method: LC/MS/MS**		
Compound	**Method Detection Limit (µg/L)**	**Reporting Limit (µg/L)**	**Compound**	**Method Detection Limit (µg/L)**	**Reporting Limit (µg/L)**
Azinphos methyl	0.0099	0.05	Atrazine	0.02	0.05
Chlorpyrifos	0.0109	0.04	Bromacil	0.031	0.05
Diazinon	0.011	0.04	Diuron	0.022	0.05
DDVP (dichlorvos)	0.0098	0.05	Hexazinone	0.04	0.05
Dimethoate	0.0079	0.04	Metribuzin	0.025	0.05
Disulfoton	0.0093	0.04	Norflurazon	0.019	0.05
Ethoprop	0.0098	0.05	Prometon	0.016	0.05
Fenamiphos	0.0125	0.05	Prometryn	0.016	0.05
Fonofos	0.008	0.04	Simazine	0.013	0.05
Malathion	0.0117	0.04	DEA	0.010	0.05
Methidathion	0.0111	0.05	ACET	0.030	0.05
Methyl Parathion	0.008	0.03	DACT	0.016	0.05
Thimet (Phorate)	0.0083	0.05			
Profenofos	0.0114	0.05			
Tribufos	0.0142	0.05			

Pyrethroid Pesticides in Surface Water Method: GC/ECD, confirmed with GC/MSD		
Compound	**Method Detection Limit (µg/L)**	**Reporting Limit (µg/L)**
Fenvalerate/Esfenvalerate	0.0225	0.050
Permethrin	0.0169	0.050
Bifenthrin	0.00216	0.005
Lambda Cyhalothrin	0.00776	0.020
Cyfluthrin	0.0555	0.080
Cypermethrin	0.0566	0.080
Pyrethroid Pesticides in Sediment Method: GC/ECD, confirmed with GC/MSD (MG/G)		
Fenvalerate/Esfenvalerate	0.008	0.01
Permethrin	0.006	0.01
Bifenthrin	0.007	0.01
Lambda Cyhalothrin	0.009	0.01
Cyfluthrin	0.008	0.01
Cypermethrin	0.008	0.01

Table 2. Physical habitat and water quality parameters

Physical Habitat parameters	Water Quality Parameters
Avg. reach gradient	Temperature
Avg. gradient of reach	EC
Avg. canopy cover of reach	Dissolved oxygen
Avg. depth of reach	pH
Adjacent land use	Turbidity
Evident NPS pollution	Alkalinity
Evident watershed erosion	Ammonia Nitrate
Water odors	Nitrate
Water surface oils	Phosphate.
Avg. substrate size	
Avg. substrate embeddedness	

Table 6. Toxicity of detected insecticides

Pesticide	***Ceriodaphnia dubia*** (µg/L)	***Daphnia magna*** (48hr EC50, µg/L)	***Oncorhynchus mykiss*** (96hr LC50, µg/L)	**Plecoptera - varied species** (96hr LC50, µg/L)	Freshwater Criterion Maximum Concentration [g]
Bifenthrin	**0.07** [b]	**1.6** [a]	0.15 [a]	ND	ND
Malathion	1.14 [d]	1.0-2.2 [d]	2.0-7.0 [d]	0.2-4.3 [d]	0.43 [g]
Diazinon	0.436 [c]	0.96 [d]	230-2900 [d]	20.0-30.0 [d]	0.08 [g]
Chlorpyrifos	0.038 [e]	0.1-1.7 [d]	7.1-27 [d]	0.4-13.0 [d]	0.02 [g]
Dimethoate	ND	**1700** [f]	4100 – 10900 [d]	0.036-0.05 [d]	ND

NOTES:

- ND - No data available
- Numbers in **Bold** are for 48-hour LC50 toxicity tests.

Number ranges are for all studies listed in the indicated source and may represent 2-6 individual studies.

SOURCES:

a. DPR Ecotox Database, 2003
b. Mokry and Hoagland, 1990
c. CA Dept. of Fish & Game, 1998
d. U.S. EPA Ecotox Database, 2003
e. CA Dept. of Fish & Game, 1999
f. Siepmann S. and T. Yargeau, 1996
g. Siepmann and Finlayson, 2002

Table 7. Pesticide detections at California central valley monitoring sites, fall and spring, 2002-2004

	Elder Creek		Elk Grove Creek		Little John Creek		Lone Tree Creek	
Pesticide	**At BR**	**At EGFR**	**At EV**	**At EGF**	**At AR**	**At SR**	**At EBR**	**At LTR**
Fall 2002								
Organophosphates								
Diazinon	nd	nd	trace	0.0599	nd	nd	nd	nd
Chlorpyrifos	0.0684	nd	nd	nd	nd	nd	nd	nd
Dimethoate	nd	nd	nd	nd	trace	nd	nd	nd
Herbicides								
Diuron	0.174	nd	nd	nd	nd	nd	nd	0.063
Prometon	nd	nd	nd	nd	nd	nd	nd	nd
DACT	nd	nd	nd	nd	nd	nd	nd	nd
Pyrethroids								
in water	nd	nd	nd	nd	nd	nd	nd	nd
in sediment	nd	nd	nd	nd	nd	nd	nd	nd
Spring 2003								
Organophosphates								
Diazinon	nd	nd	0.14	0.212	nd	nd	nd	nd
Chlorpyrifos	0.108	trace	nd	nd	nd	nd	nd	nd
Dimethoate	nd	nd	nd	nd	nd	trace	nd	nd
Herbicides								
Diuron	0.15	0.379	3.65	5.84	3.79	0.154	14.24	6.3
Prometon	nd	nd	0.133	0.131	nd	nd	nd	nd
DACT	nd	nd	nd	nd	nd	nd	0.135	nd
Pyrethroids								
in water	nd	nd	nd	nd	nd	nd	nd	nd
in sediment	nd	nd	nd	nd	nd	nd	nd	nd

Fall 2003								
Organophosphates								
Diazinon	nd	nd	0.203	tr	nd	nd	nd	nd
Chlorpyrifos	0.163	nd	nd	nd	nd	nd	nd	nd
Dimethoate	nd	nd	nd	nd	nd	nd	nd	nd
Herbicides								
Not collected this season								
Pyrethroids (water) ppt								
Bifenthrin	27.5ppt	nd	nd	nd	nd	nd	nd	nd
No other pyrethroid detections								
Pyrethroids (sediment) ppm								
	nd	nd	nd	nd	nd	nd	nd	nd

Table 7. Continued. Pesticide detections

Spring 2004								
Organophosphates								
Diazinon	nd	nd	0.0604	trace	nd	trace	nd	nd
Chlorpyrifos	0.156	nd	nd	nd	trace	nd	nd	nd
Dimethoate	nd	nd	nd	nd	nd	nd	nd	nd
Malathion	nd	nd	nd	trace	nd	nd	nd	nd
Herbicides								
Atrazine								
Simazine	trace	trace	nd	trace	nd	0.12	0.154	0.326
Diuron	0.095	0.098	0.365	0.25	0.959	0.233	1.94	4.45
Prometon	nd	nd	0.06	trace	nd	nd	nd	nd
Bromacil	nd	nd	0.061	0.146	nd	nd	nd	nd
DACT	nd	nd	nd	nd	nd	nd	nd	nd
Pyrethroids (water)								
Bifenthrin	trace	nd	trace	nd	nd	nd	nd	nd
* no other detections								
Pyrethroids (sediment)								
Permethrin	nd	nd	trace	nd	nd	nd	nd	nd
* no other detections								
*** nd = no detection, **tr= trace, *** All detections are in ppb.**								

Table 8. Benthic macroinvertebrate results at California central valley monitoring sites, fall and spring, 2002-2004

Project Name:	**Elk Grove Creek**							
Site Name:	**AT EGF**				**AT EV**			
Collection Date:	12/6/2002	4/18/2003	10/21/2003	4/8/2004	12/6/2002	4/18/2003	10/23/2003	4/9/2004
Taxonomic Richness	10	5	3	22	12	6	12	20
Percent Dominant Taxon	45	75	75	41	43	74	33	48
Ephemeroptera Taxa	0	0	0	0	0	0	0	0
Plecoptera Taxa	0	0	0	0	0	0	0	0
Trichoptera Taxa	0	0	0	0	0	0	0	0
EPT Taxa	0	0	0	0	0	0	0	0
EPT Index (%)	0	0	0	0	0	0	0	0
Sensitive EPT Index (%)	0	0	0	0	0	0	0	0
Shannon Diversity	1.2	9	0.7	1.8	1.7	1	2	1.7
Tolerance Value	7.5	9.8	6	7.1	8.2	6.4	6.5	6.2
Percent Intolerant Taxa (0-2)	0	0	0	0	0	0	0	0
Percent Tolerant Taxa (8-10)	50	95	25	39	64	11	29	22
Percent Baetidae	0	0	0	0	0	0	0	0
Percent Chironomidae	44	100	25	82	62	98	43	41
Percent Hydropsychidae	0	0	0	0	0	0	0	0
Percent Collector-Gatherers	48	21	88	81	44	89	73	82
Percent Collector-Filterers	0	0	0	2	0	0	2	1
Percent Scrapers	1	0	0	8	1	2	14	9
Percent Predators	49	79	13	6	55	9	11	6
Percent Shredders	0	0	0	0	0	0	0	0
Abundance (#/ sample)	173	291	8	1728	77	47	63	246

Table 8. Continued. Benthic macroinvertebrate results

Project Name:	**Elder Creek**							
Site Name:	**AT EGFR**				**AT BR**			
Collection Date:	12/6/2002	4/18/2003	10/21/2003	4/8/2003	12/6/2002	4/23/2003	10/23/2003	4/9/2004
Taxonomic Richness	30	36	43	36	10	18	22	25
Percent Dominant Taxon	48	33	18	34	30	40	36	30
Ephemeroptera Taxa			1	0			0	0
Plecoptera Taxa		0	0	0		0	0	0
Trichoptera Taxa		0	1	1		0	0	0
EPT Taxa	1	1	2	1	0	0	0	0
EPT Index (%)	5	0	4	1	0	0	0	0
Sensitive EPT Index (%)	0	0	0	1	0	0	0	0
Shannon Diversity	2	2.5	2.7	2.5	1.9	2	2	2.1
Tolerance Value	5.8	6.8	5.8	7.7	8.4	7.6	6.3	7.4
Percent Intolerant Taxa (0-2)	0	0	0	0	0	0	0	0
Percent Tolerant Taxa (8-10)	22	31	24	62	70	70	17	54
Percent Baetidae		0	4	0		0	0	0
Percent Chironomidae	33	58	41	40	65	64	84	55
Percent Hydropsychidae		0	0	0		0	0	0
Percent Collector-Gatherers	84	39	82	66	74	91	86	58
Percent Collector-Filterers	9	35	2	10	0	2	1	2
Percent Scrapers	2	13	2	2	0	2	4	36
Percent Predators	5	12	8	20	26	3	1	4
Percent Shredders	0	0	3	0	0	1	0	0
Abundance (#/ sample)	2405	9679	2445	26216	93	721	976	905

Table 8. Continued. Benthic macroinvertebrate results

Project Name:	**Little John Creek**							
Site Name:	**AT AR**				**AT SR**			
Collection Date:	10/15/2002	4/16/2003	10/20/2003	4/7/2004	10/15/2002	4/16/2003	10/20/2003	4/7/2004
Taxonomic Richness	33	12	19	15	34	20	22	35
Percent Dominant Taxon	15	49	41	71	24	49	62	33
Ephemeroptera Taxa			2	0			2	6
Plecoptera Taxa		0	0	0		0	0	0
Trichoptera Taxa		0	0	0		0	1	0
EPT Taxa	1	0	2	0	2	0	3	6
EPT Index (%)	1	0	1	0	9	0	3	6
Sensitive EPT Index (%)	0	0	0	0	0	0	0	2
Shannon Diversity	2.9	1.9	1.7	1.2	2.8	1.6	1.6	2.5
Tolerance Value	7.7	7.7	7.6	5.6	6.7	5.8	6.1	5.7
Percent Intolerant Taxa (0-2)	0	0	0	0	3	0	0	7
Percent Tolerant Taxa (8-10)	57	51	54	8	32	8	23	17
Percent Baetidae		0	1	0		0	0	4
Percent Chironomidae	57	83	18	28	36	46	23	32
Percent Hydropsychidae		0	0	0		0	0	0
Percent Collector-Gatherers	55	73	42	90	34	59	79	40
Percent Collector-Filterers	12	26	1	9	27	35	2	45
Percent Scrapers	0	1	1	0	1	2	0	1
Percent Predators	32	0	56	1	37	4	17	8
Percent Shredders	0	0	0	0	0	0	0	0
Abundance (#/ sample)	1078	2741	579	928	579	2470	698	3083

Table 8. Continued. Benthic macroinvertebrate results

Project Name:	**Lone Tree Creek**							
Site Name:	**AT EBR**				**AT LTR**			
Collection Date:	10/16/2002	4/15/2003	10/22/2003	4/10/2004	10/16/2002	4/14/2003	10/22/2003	4/10/2004
Taxonomic Richness	12	19	11	30	23	21	19	21
Percent Dominant Taxon	95	66	53	30	75	86	38	40
Ephemeroptera Taxa			0	1			0	0
Plecoptera Taxa		0	0	0		0	0	0
Trichoptera Taxa		0	0	0		0	2	0
EPT Taxa	0	0	0	1	2	0	2	0
EPT Index (%)	0	0	0	0	1	0	1	0
Sensitive EPT Index (%)	0	0	0	0	0	0	0	0
Shannon Diversity	0.3	1.4	1.2	2	1.2	0.8	1.6	1.7
Tolerance Value	4.1	5	3.7	6.4	4.7	5.3	3.9	6.1
Percent Intolerant Taxa (0-2)	0	0	0	0	0	0	0	0
Percent Tolerant Taxa (8-10)	1	14	40	31	9	6	21	7
Percent Baetidae		0	0	0		0	0	0
Percent Chironomidae	1	16	4	26	8	8	3	48
Percent Hydropsychidae		0	0	0		0	0	0
Percent Collector-Gatherers	95	93	93	78	80	96	88	89
Percent Collector-Filterers	3	6	0	19	4	1	3	7
Percent Scrapers	1	0	1	0	5	1	0	1
Percent Predators	1	1	5	1	8	1	8	1
Percent Shredders	0	0	0	0	0	0	0	1
Abundance (#/ sample)	40196	1772	2031	1707	7036	1210	259	424

Figure 1. Bioassessment monitoring Sites in the California central valley, fall and spring, 2002-2004

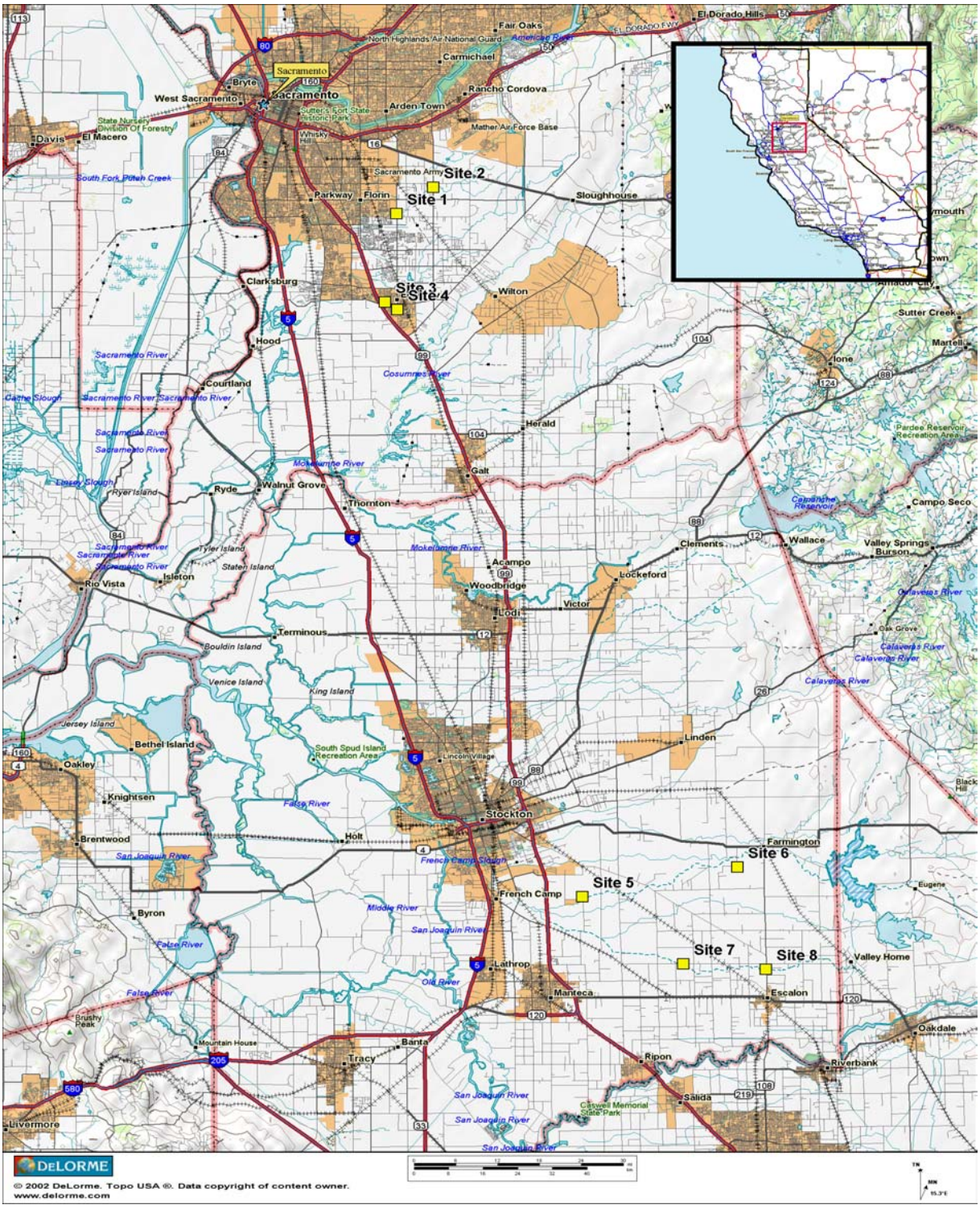


Figure 2. Urban monitoring sites, California central valley

Site 1. Elder creek at EGFR

Site 2. Elder creek at BR

Site 3. Elk Grove creek at EV

Site 4. Elk Grove creek at EGF

Figure 3. Agricultural monitoring sites, California central valley

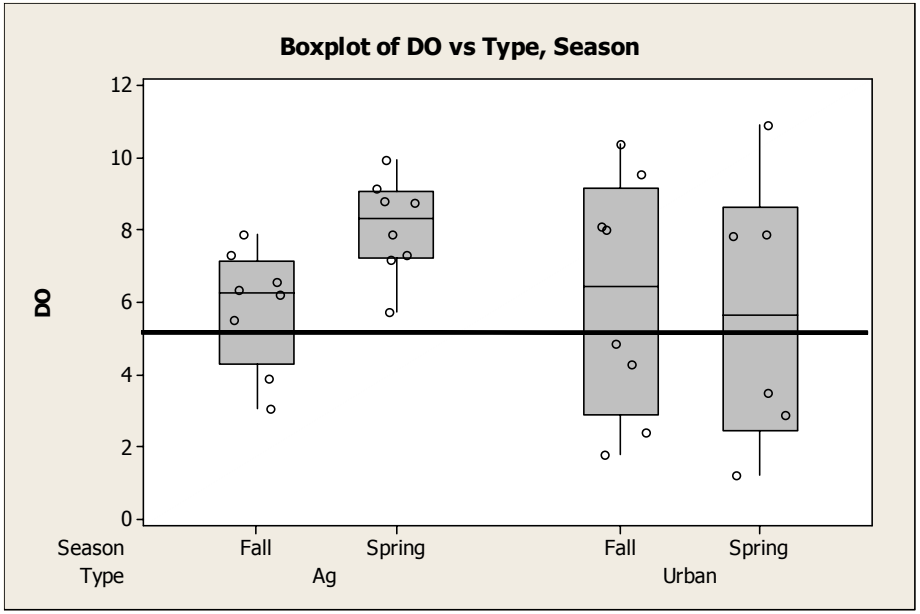
Site 5. Little John creek at AR

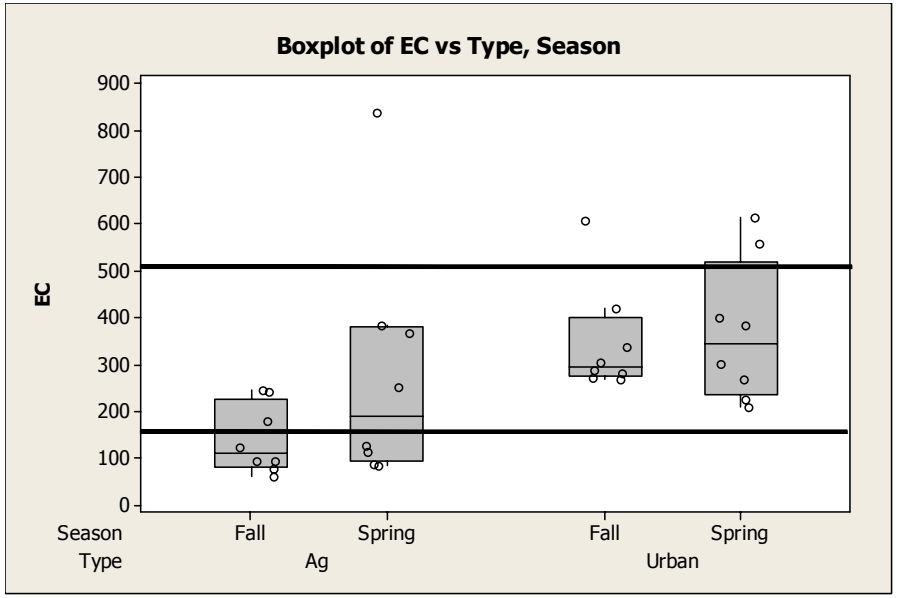
Site 6. Little John creek at SR

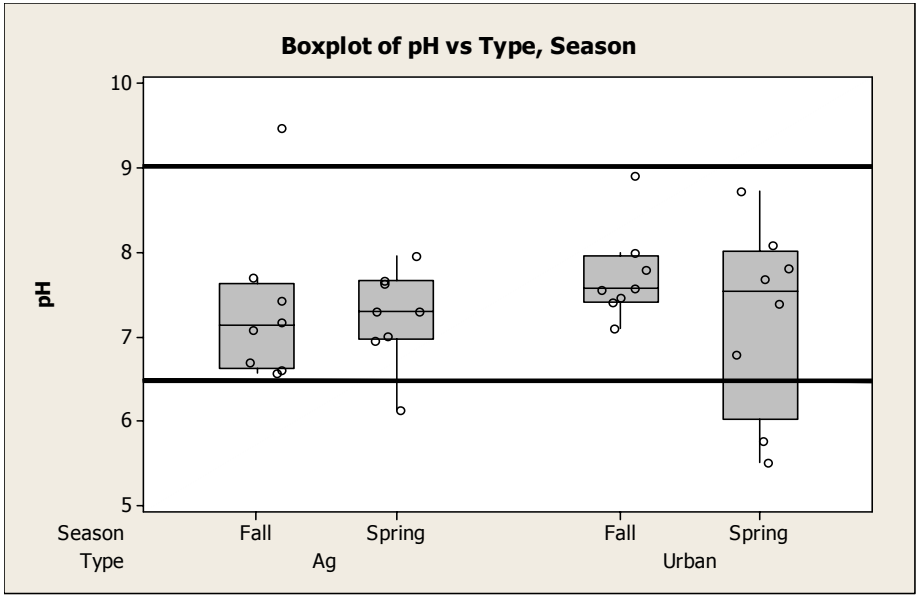
Site 7. Lone Tree creek at LTR

Site 8. Lone Tree creek at EBR

Figure 4. Water quality measurements by site and season

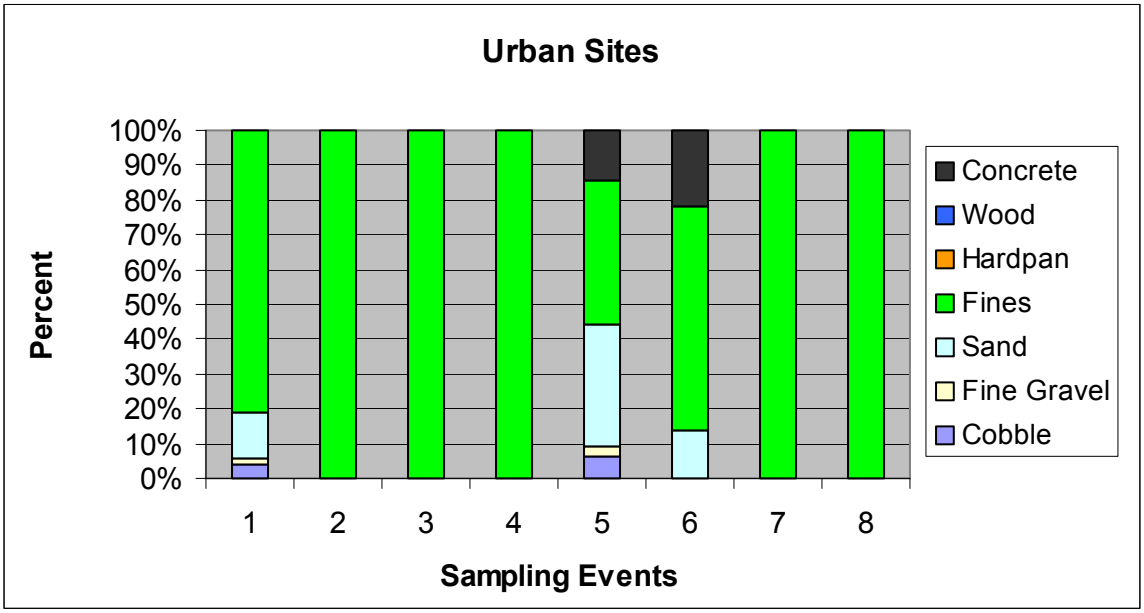






Note: Bold lines (▬▬) represent water quality requirements and criteria.

Figure 5. Substrate results at California central valley monitoring sites, fall and spring, 2002-2004



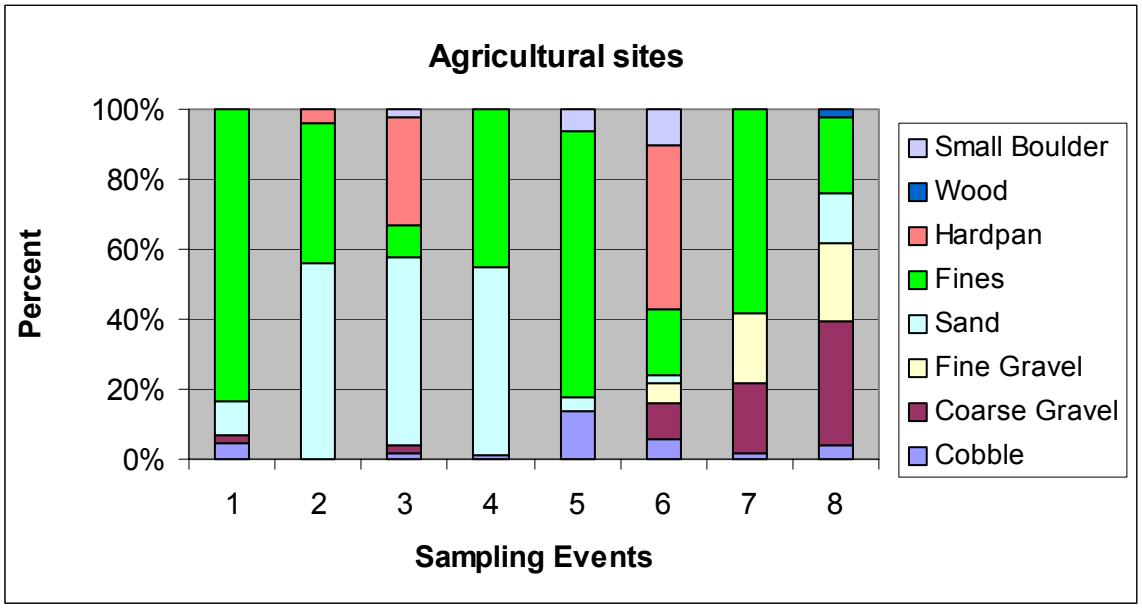
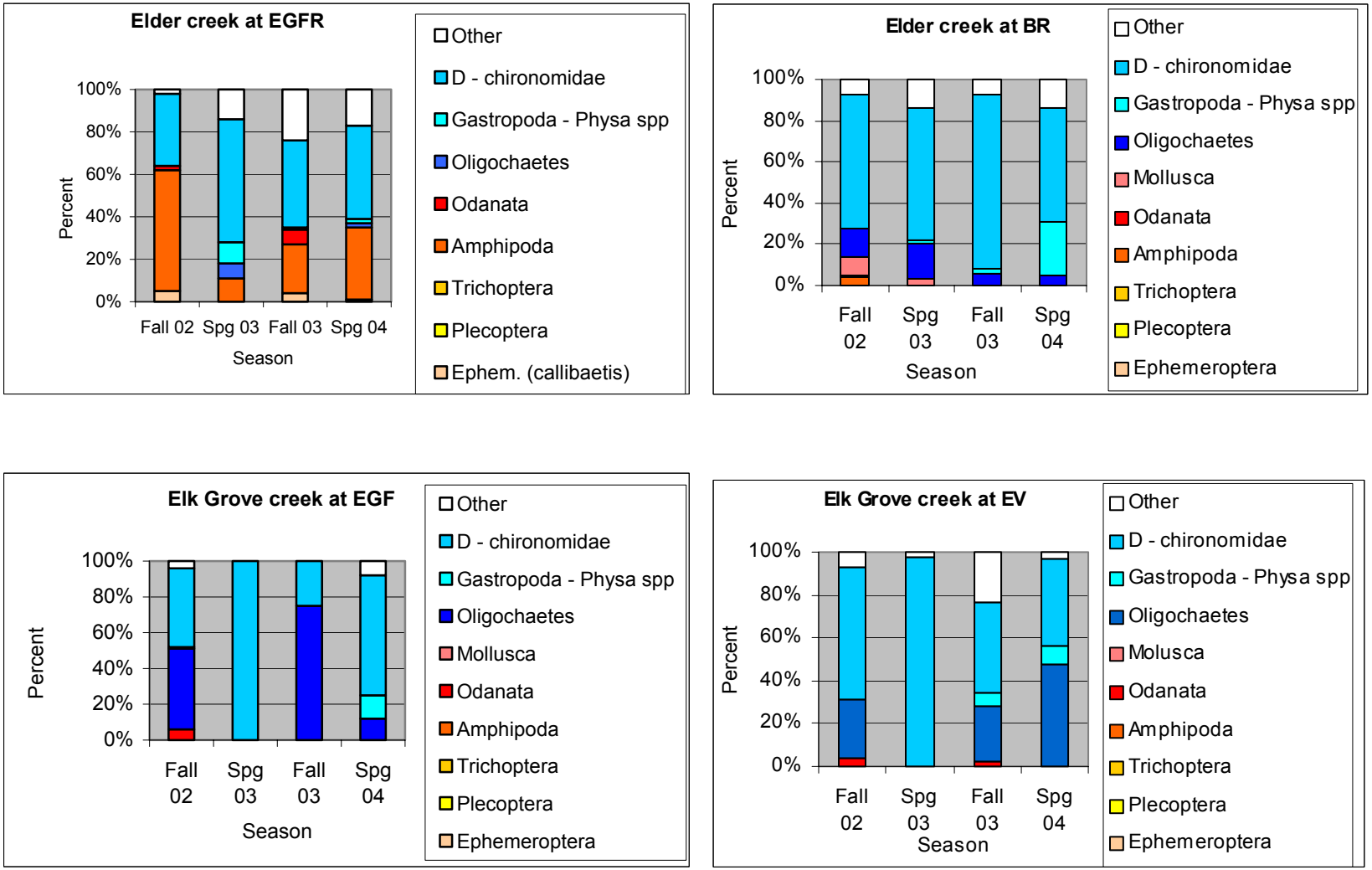
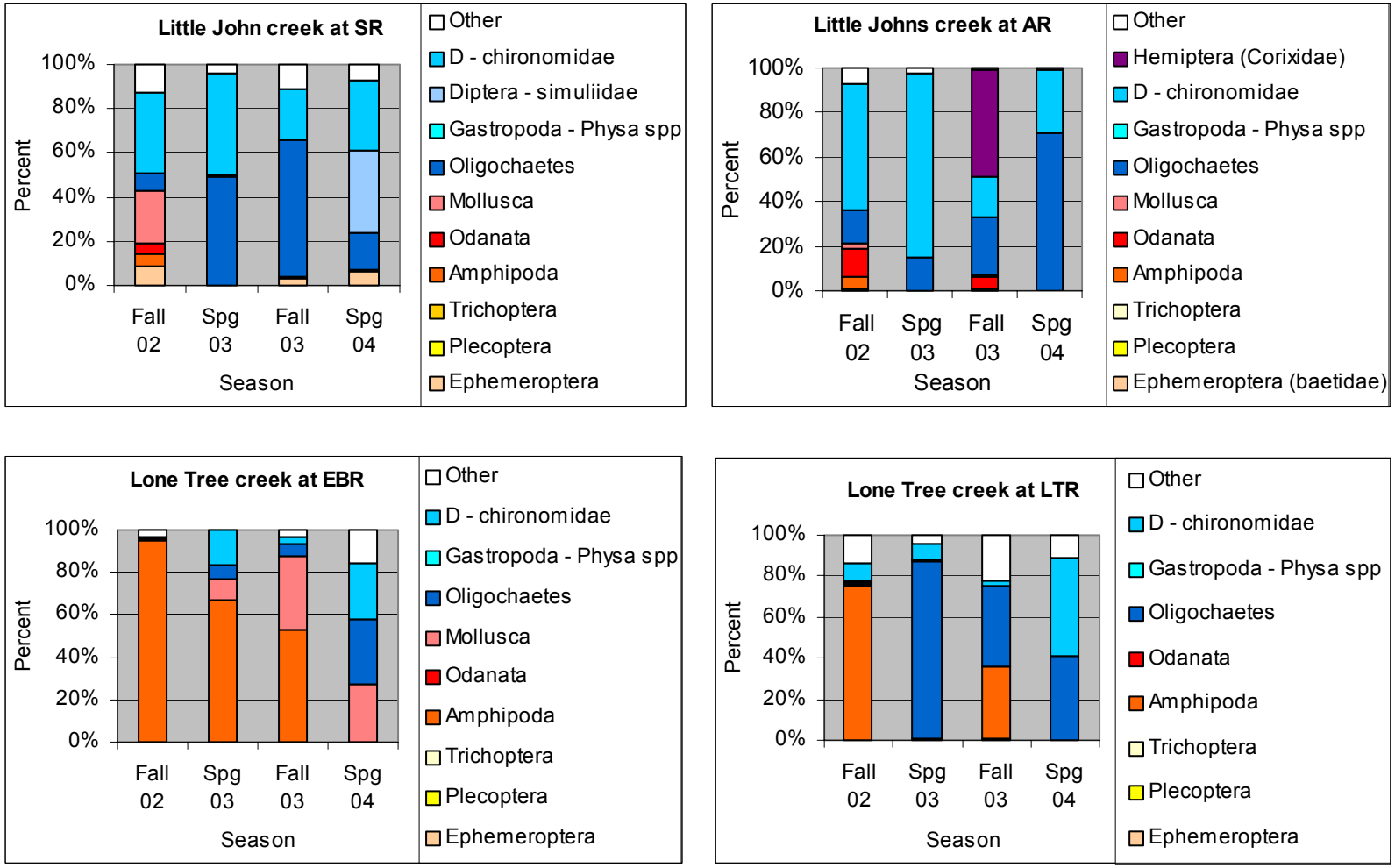


Figure 6. Benthic macroinvertebrate indicator taxa of urban sites, at California central valley monitoring sites, fall and spring, 2002-2004



*** Blue colors represent those taxa found in fairly to severely polluted waters.**

Figure 7. Benthic macroinvertebrate indicator taxa of agricultural sites, at California central valley monitoring sites, fall and spring, 2002-2004



*** Blue colors represent those taxa found in fairly to severely polluted waters.**

Figure 8. Benthic macroinvertebrate functional feeding groups found at urban sites, California central valley, fall and spring, 2002-2004

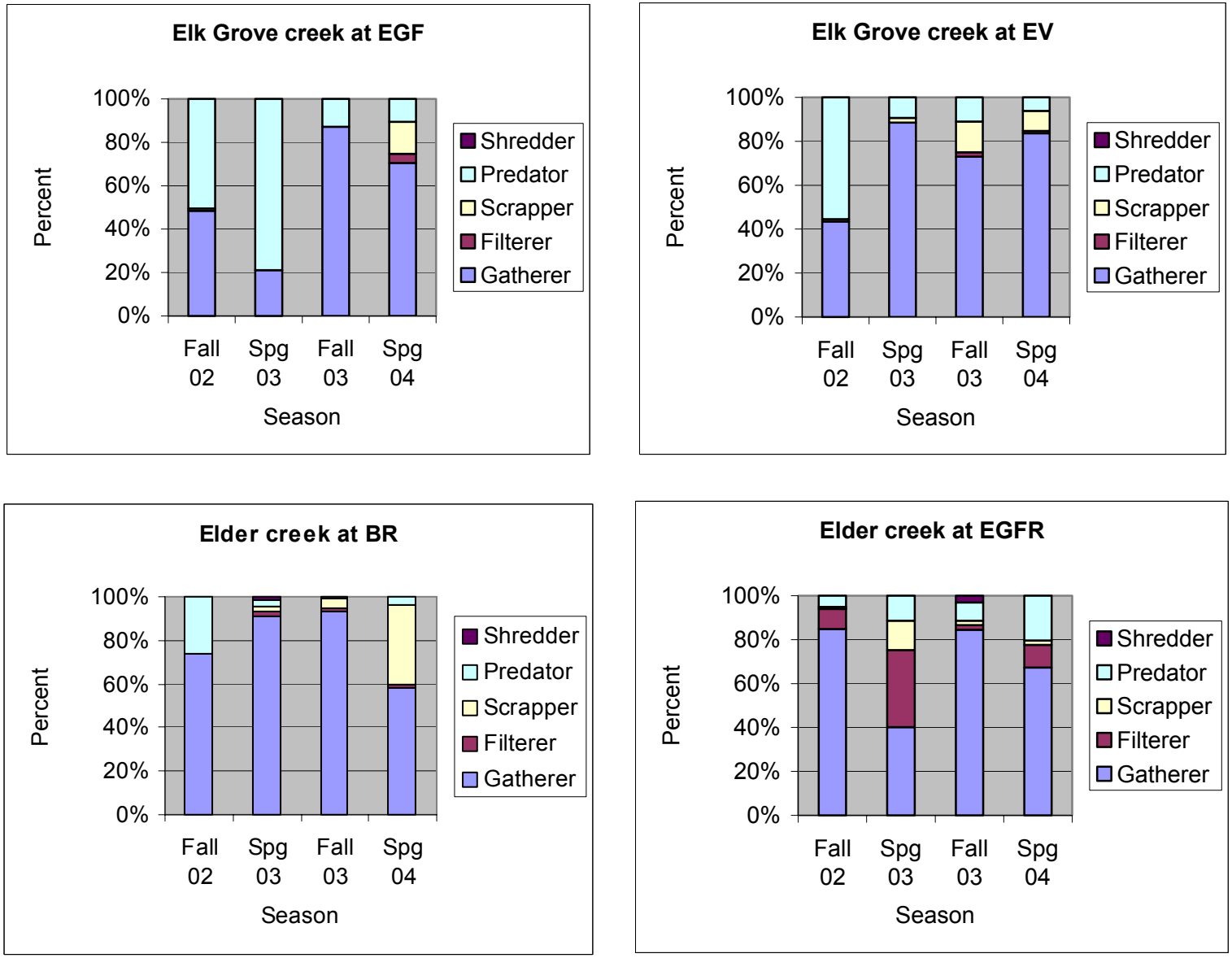


Figure 9. Benthic macroinvertebrate functional feeding groups found at agricultural sites, California central valley, fall and spring, 2002-2004

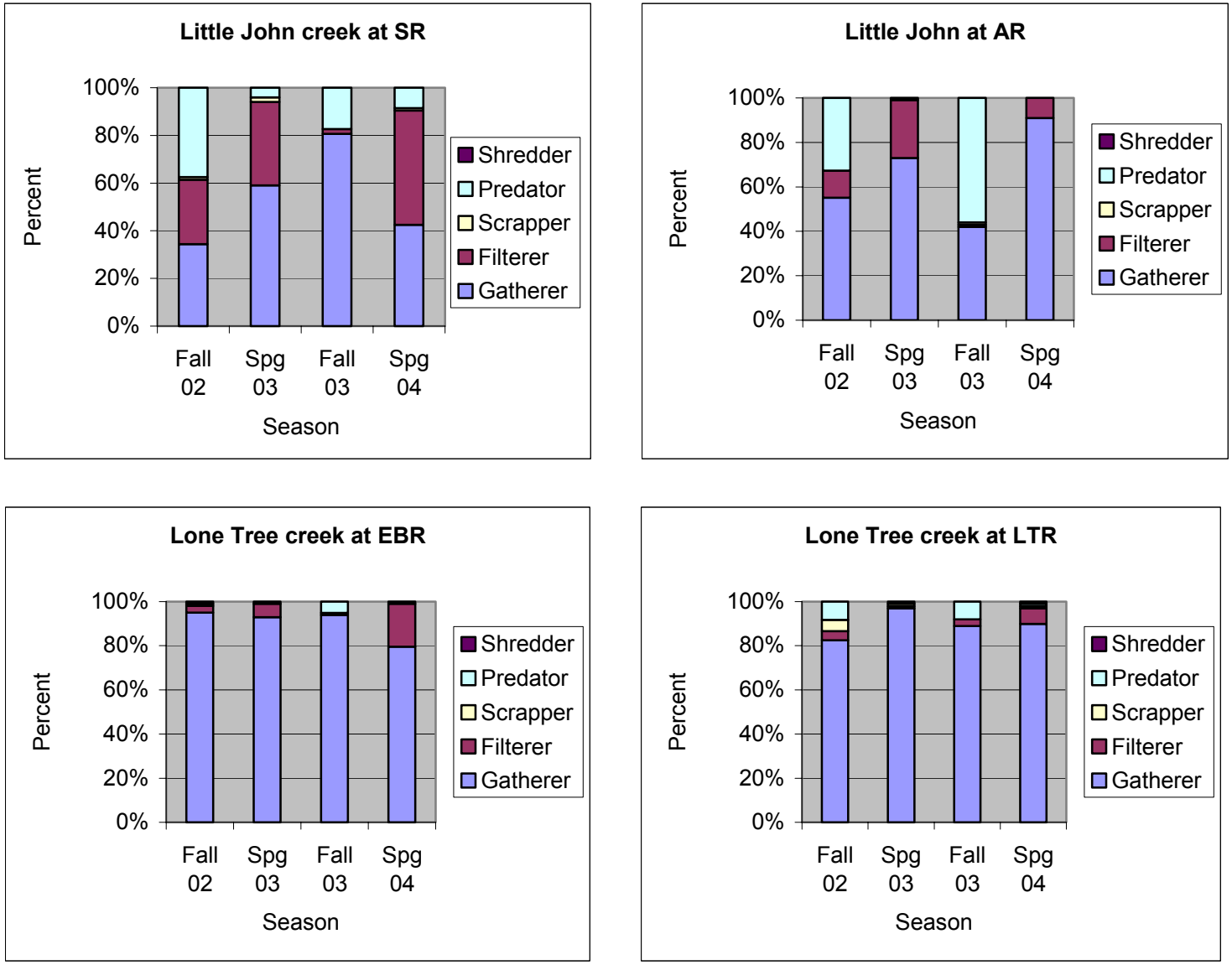
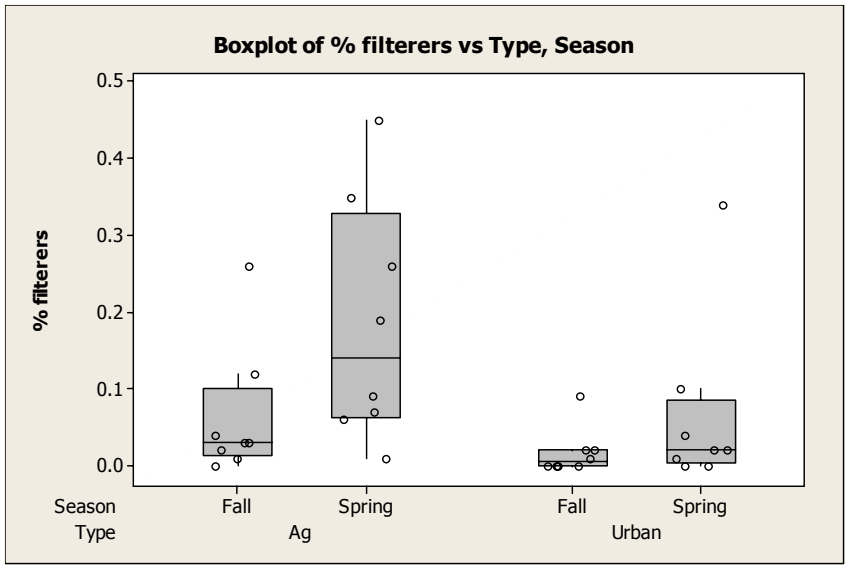
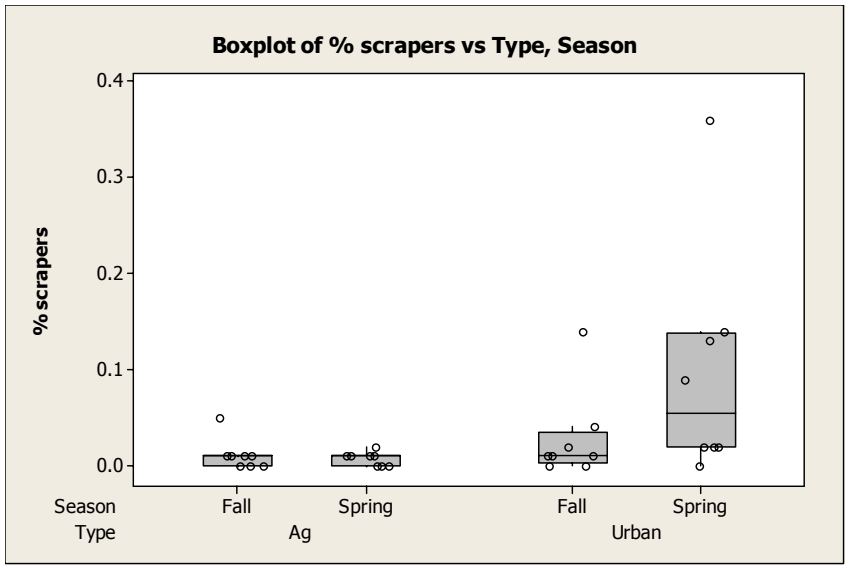


Figure 10. Select benthic macroinvertebrate metrics by site and season





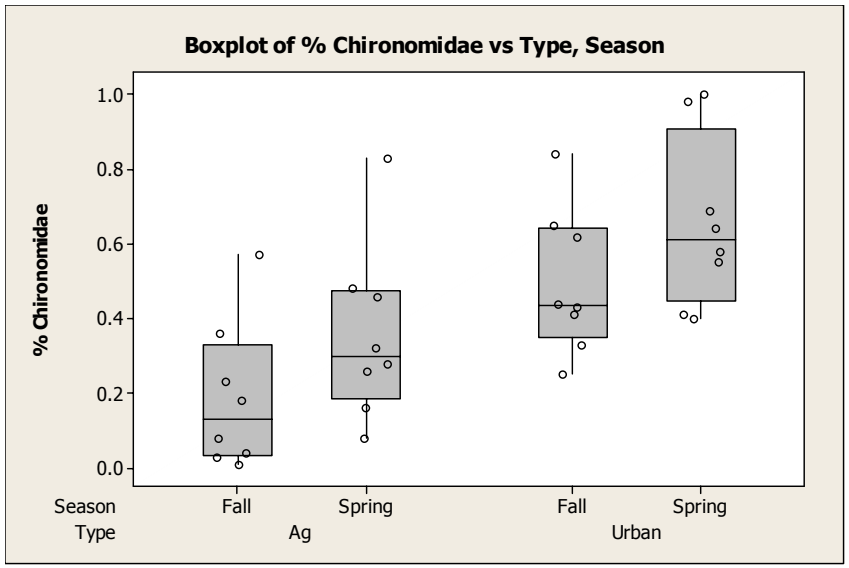


Figure 10. Continued. Select benthic macroinvertebrate metrics by site and season