CENTRAL VALLEY REGIONAL
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Benthic Macroinvertebrate Colonization of Artificial Substrates in Agriculture-dominated Waterways of the Lower Sacramento River Watershed

Surface Water Ambient Monitoring Program (SWAMP)
Lower Sacramento River Watershed

August 2005

CALIFORNIA ENVIRONMENTAL PROTECTION AGENCY

State of California
Arnold Schwarzenegger, Governor

California Environmental Protection Agency
Dr. Alan Lloyd, Ph.D., Secretary

REGIONAL WATER QUALITY CONTROL BOARD
CENTRAL VALLEY REGION

11020 Sun Center Drive #200
Rancho Cordova, CA 95670

Phone: (916) 464-3291
eMail: info5@waterboards.ca.gov
Web site: http://www.waterboards.ca.gov/centralvalley/

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REPORT PREPARED BY:

ROBERT W. HOLMES
Environmental Scientist
Sacramento River Watershed Unit

And

V. de Vlaming[1], D. Markiewicz[1], and K. Goding[1]

[1]U.C. Davis Aquatic Toxicology Laboratory
School of Veterinary Medicine: APC, 1321 Haring Hall
University of California, Davis 95616

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EXECUTIVE SUMMARY

Bioassessment data alone cannot identify cause(s) of water quality impact or impairment on aquatic ecosystem biota (beneficial uses). A primary objective of this study was to differentiate effects of physical habitat and water quality on invertebrate community condition in three agriculture-dominated waterways (ADW—Jack Slough, Main Drain, and Wadsworth Canal) in the lower Sacramento River watershed. The specific water quality focus of this investigation was whether stormwater runoff from agricultural lands adversely impact ADW invertebrate communities. A weight-of-evidence approach (i.e., multiple monitoring procedures) was applied. That is, chemical analysis of water column samples and aquatic species toxicity testing were used to assist in interpretation of benthic macroinvertebrate bioassessment (BMI) data. The primary biological assessment procedure was colonization of artificial substrate (rock-filled baskets) by BMI. The artificial substrate procedure is a tool that is widely applied to differentiate water quality from physical habitat (especially substrate) effects on BMI community structure and condition. Sampling in this investigation occurred December 2002 through April 2003, primarily following rainstorms. Data gathered in this study reveal that the artificial substrate performed effectively.

Colonization of artificial substrate at all three ADWs was by multivoltine (short life-span) and collector BMI taxa. A majority of taxa colonizing the artificial substrate were tolerant or of unknown tolerance. These results are not surprising given the poor physical habitat and water quality conditions in ADWs. Distinguishing water quality effects on ADW BMI communities is difficult because physical habitat is so degraded. Artificial substrate dominant taxa at the three ADW sites were naidid oligochaetes, chironomids, crangonyctid amphipods, and Hydra. However, the overall artificial substrate BMI community structure differed at the three ADW sites even on the same sampling dates. Change in BMI community structure through time also differed in the three ADWs. Further investigation is needed to define/explain these differences in BMI community structures in the three ADWs as well as the temporal changes. Almost certainly these

differences relate to dissimilarity in physical habitat and water quality conditions in the three ADWs at the sampling times and through time. Artificial substrate BMI community composition in all three ADWs varied temporally. In particular, December and early-January communities tended to diverge from those observed in late-February and March, especially in Jack Slough and the Main Drain. Whether these variations were a natural seasonal phenomenon or related to the high use of pesticides during the January and February dormant orchard application season is yet to be determined. More extensive (more sites) and long-term weight-of-evidence studies are essential to fully understand the relationship between BMI communities and deterministic environmental variables.

There was no consistent relationship between BMI metrics and specific environmental variables. These results clearly illustrate that correlations must not be interpreted as cause-and-effect relationships. These findings lead us to hypothesize that a number of interacting physical and water quality factors determine BMI community structure and condition.

Ceriodaphnia mortality was observed in Jack Slough samples collected on 1/22/03, 2/13/03, and 2/16/03. Diazinon, diuron, and simazine were detected in the toxic samples, but all at concentrations lower than LC_{50}s. Two Wadsworth Canal samples collected on 1/16/03 caused *Ceriodaphnia* mortality (40 and 100%). Diazinon, diuron, bromocil, norflurazon, and permethrin were identified in these samples at concentrations lower than individual pesticide LC_{50}s. While no one chemical appeared to be responsible for *Ceriodaphnia* mortality in these ADW samples, there is reason to suspect that a mixture of pesticides acted additively or synergistically (more than additive). Further, the limited pesticide analyte list, due to limited resources, may have excluded other contaminants that could have been present.

Artificial substrate BMI assemblages in Wadsworth Canal and the Main Drain shifted during periods when the highest pesticide concentrations were measured in these systems. BMI taxa list changes in Wadsworth Canal included a decline of coenagrionid damselflies and *Crangonyx* amphipods, and a dominance of chironomids. BMI taxa list shifts in the Main Drain included a decrease in EPT taxa and a dominance of *Hydra*.

Weekly qualitative field observation assessments (presence/absence and relative abundance) of BMI and zooplankton groups in aquatic edge habitat were used to identify taxa group trends throughout the study. Major BMI taxa group shifts were not observed. However, zooplankton abundance and chlorophyll decreased from late-February though mid-April. Whether the zooplankton/chlorophyll shifts were natural- or anthropogenic-caused events is unknown. The highest concentrations of pesticides were detected just prior to this period.

1. Introduction

Agriculture is the predominant land use in California's Central Valley (e.g., Domagalski et al., 1998; Groneberg et al., 1998). There are over 10,000 miles of waterways dominated by agricultural land use in the Central Valley. Land use affects benthic macroinvertebrate (BMI) community integrity/condition in stream ecosystems (Brown and May, 2000; Lenat and Crawford, 1994; Roy et al., 2003a, b; de Vlaming et al., 2004a, b). Impacts to freshwater ecosystems from agricultural land use practices can include sedimentation (e.g., Lenat 1984; Waters, 1995; Relyea et al., 2000), increased nutrient load (e.g., de Vlaming et al., 2004b), loss of riparian habitat (e.g., de Vlaming et al., 2004a), a variable flow regime (e.g., Nelson and Lieberman, 2002), occurrence of pesticides (e.g., Holmes and de Vlaming, 2003) and aquatic toxicity (e.g., de Vlaming et al., 2000; Anderson et al., 2003a; 2003b; Hunt et al., 1999). Data provided by Anderson et al., (In press) that pesticides have greater effects on BMI communities than suspended sediments in agricultural dominated waterways.

A weight-of-evidence/integrated monitoring approach has been recommended to establish cause-and-effect relationships with water quality contaminants and impairment to stream biological communities (e.g., Taylor and Kovats, 1995; Leslie et al., 1999; Culp et al., 2000; National Research Council, 2001; Collier, 2003; Hewitt et al., 2003; de Vlaming et al., 2004a). With the weight-of-evidence approach measures of biological condition (bioassessments) are conducted in concert with chemical and toxicological procedures to assess cause-and- effect to resident biota. Anderson et al. (2003a, b) used a weight-of-evidence approach on the Salinas River to establish impairment by the organophosphate pesticide, chlorpyrifos. Anderson et al. (2003a, b) were able to discount the effects of various stressors to BMI communities with increased BMI sampling replication, use of *in situ* toxicity tests, sediment toxicity tests, Toxicity Identification Evaluations (TIEs), chemical analyses, and controlled laboratory experiments.

Most often BMI communities are impacted by a combination of factors (e.g., habitat conditions and water quality—de Vlaming et al., 2004a). Differentiating effects of intercorrelated stressors on BMI community in multi-stressed systems, such as low gradient agricultural-dominated waterways (ADWs), poses a significant challenge.

Physical habitat is a major determinant of BMI community structure (e.g., Karr, 1991; Barbour et al., 1999). BMI colonization of artificial substrates can be a useful bioassessment approach in weight-of-evidence aquatic ecosystem investigations. Artificial substrates facilitate differentiation of water quality (contaminant) from habitat (e.g., substrate) related influences to BMI communities (review articles: Rosenberg and Resh, 1982; Taylor and Kovats, 1995). Artificial substrate provides standardization of substrate composition, which in turn reduces the biological variability associated with substrate. Substrate composition, particularly in low gradient agriculture-dominated waterways, can be highly variable (de Vlaming, et al., 2004a). Based on an extensive literature review, Taylor and Kovats (1995) recommend artificial substrate for BMI assessments in waterways with unstable bottoms of sand, mud, or organic ooze (typical of ADWs). We are not aware of any studies that used artificial substrate to examine BMI colonization in ADWs. No studies were found in the literature that examined, with a weight-of-evidence approach, the BMI community structure/condition and individual aquatic life stressors associated with stormwater runoff from agricultural land use.

A weight-of-evidence approach was applied to investigate BMI colonization of artificial substrates in ADWs of the lower Sacramento River watershed during a winter storm season. The primary objective was to use artificial substrate baskets to distinguish water quality from physical habitat effects on BMI communities. The variables monitored in this study included temperature, dissolved oxygen (DO), specific conductivity (SpC), turbidity, chlorophyll, and water column concentrations of organic pesticides (organophosphate insecticides, selected pyrethroid insecticides, triazine herbicides).

2. Methods

This investigation was conducted in ADWs of the lower Sacramento River watershed during the 2002/2003 stormwater season. Fixed-date and episodic sampling of artificial substrate (rock-filled baskets) were used to track BMI colonization trends. Laboratory water column toxicity testing using *Hyalella azteca* (amphipod) and *Ceriodaphnia dubia* (a cladoceran zooplankton species) was used to identify occurrences of toxicity in surface water samples. Weekly site visits were conducted throughout the study to document water quality chemistry (DO, temperature, pH, SpC, turbidity and chlorophyll) and physical habitat conditions (velocity and depth) at artificial substrate sites; BMI and zooplankton community composition and relative abundance in stream bank habitat samples were also assessed in these weekly visits. To leverage limited resources, the organic pesticide data was supplied by two concurrent investigations (Bacey et al., 2004 and Calanchini et al., 2003). The pesticide analyte list chosen by Bacey et al., (2004) and Calanchini et al., (2003) was limited and may not reflect all possible pesticide products or other contaminants that could be responsible for or contribute to any aquatic toxicity and/or any shifts in BMI community responses that may be observed. The University of California, Davis Aquatic Toxicology Laboratory (UCD ATL) and Bacey et al. (2003) collected toxicity testing data on samples collected at the artificial substrate sites.

2.1 Site selection rationale and locations

The ADWs in the lower Sacramento River Basin were Jack Slough, the Main Drainage Canal, and Wadsworth Canal (Figure 1; Table 1). De Vlaming et al. (2004a) documented low BMI index scores and poor habitat conditions in each of these waterways. These waterways have been monitored for occurrence of pesticides and aquatic life toxicity for over a decade (e.g., Domagalski, 1996; Holmes and de Vlaming, 2003). For the current investigation, one site was chosen near the lower portion of each waterway to reflect the cumulative effects of stormwater runoff from agricultural lands. Each sampling site is wadeable (< 1.5 m) under normal flow conditions in non-irrigation season.

Jack Slough is a small ADW north of Marysville in Yuba County and a Feather River tributary. During the winter months Jack Slough contains stormwater runoff from surrounding agricultural lands and drainage from local waterfowl wetland areas. The sampling location is in a semi-natural, less managed, riparian area less than 2.5 kilometers upstream of the confluence with the Feather River.

The Main Drainage Canal (Butte County) consists of a network of modified natural channels, historically natural channels, and partially constructed laterals that have been extensively aligned and modified to convey irrigation supply and return water for surrounding agricultural land practices. Irrigation water is supplied to the Main Drain from the Sutter Butte Canal, which consists of Feather River water from the Thermalito Afterbay. Peach and prune orchards are prominent in the upper watershed, while rice dominates the lower the watershed. The sampling location was at the intersection of the Main Drain and the Colusa Gridley Highway, adjacent to rice fields.

Wadsworth Canal is an ADW in Sutter County that receives water from the Feather River, irrigation runoff from extensive agricultural land east of the Sutter Buttes, and rainfall runoff. Wadsworth Canal connects with a number of small laterals that historically may have served as natural flow routes for rainfall runoff. Wadsworth Canal flows into the Sutter Bypass and Sacramento Slough above the confluences of these two waterways with the Sacramento River. The sampling location was near the intersection at Franklin Road. This site is at the lower end of Wadsworth Canal, where the channel is rather wide and bordered by levees. There is little riparian vegetation at the site, but instream vegetation is plentiful.

Sampling sites (Sites 4 and 5) were located on east and west side of the Sutter Bypass (Sutter County), respectively. Flow in the east side is predominately snow melt in Butte Creek as well as discharge from the Main Canal, Wadsworth Canal, and other low gradient ADWs. Water in the west side Sutter Bypass is from the upper Sacramento River and considered to be of high water quality. The east side of the bypass was chosen as a comparison site for the west side. No pesticide or toxicity data were available for these sites due to extreme flow fluctuations and flooding of the bypass. For the same reasons a complete trend monitoring or artificial substrate BMI data set could not be collected for either side of the Sutter Bypass. The limited number of BMI samples collected from the Sutter Bypass were archived for future analyses.

2.2 Toxicity testing

Toxicity tests were conducted by UCD ATL and Bacey et al. (2003). UCD ATL conducted 96-hour *Ceriodaphnia* and *Hyalella* water column toxicity tests. UCD ATL toxicity samples were collected near peak flow during storm events. Bacey et al. (2003) conducted *Ceriodaphnia* toxicity tests on samples collected during two storm events. Bacey et al. collected samples for toxicity testing at one- hour intervals for up to eight hours during two storm events. All toxicity tests were performed in undiluted, unfiltered samples using 96-hour, static renewal bioassays in accordance with current U.S. Environmental Protection Agency procedures (U.S. EPA, 1993). Bacey et al. (2003) collected samples for toxicity testing from only two sites (Jack Slough and Wadsworth Canal), and only during two storm events.

2.3 Pesticide chemistry

Bacey et al. (2003) analyzed water column samples for organic pesticides in Jack Slough and Wadsworth Canal during peak flow of two storm events (January 22 and February 15, 2003) at one-hour intervals for up to eight hours. Grab samples were collected from mid-channel using an extended pole and 1-liter amber glass bottles. Water samples were analyzed for two pyrethroid insecticides (esfenvalerate and permethrin), currently used organophosphate insecticides (OPs), and selected herbicides (Bacey et al., 2003).

Water samples were transported on ice and stored at 4^0C until extracted for chemical analyses or use in toxicity testing. Chemical analyses were performed by the California Department of Food and Agriculture's (CDFA) Center for Analytical Chemistry. OPs and pyrethroids were measured using gas chromatography/flame phometric detector (GC/FPD) and gas chromatography/electron capture detector (GC/ECD) confirmed with gas chromatography/mass spectrometry (GC/MS), respectively. Triazines were analyzed by liquid chromatography/atmospheric pressure chemical ionization mass spectrometry (LC/MS/MS). Full details on sampling, analyses, and quality assurance/quality control procedures can be found in Bacey et al. (2004). Bacey et al. observed the following pesticides in stormwater runoff samples from Jack Slough: chlorpyrifos, diazinon, simazine, diuron, and bromacil. These same chemicals also were observed in Wadsworth

Canal in stormwater runoff, in addition to norflurazon and permethrin. More specific details on pesticide analyses are provided in the results section.

Water column insecticide concentrations (diazinon and chlorpyrifos) were determined in samples collected by auto-samplers in the Main Canal and Wadsworth Canal (Calanchini et al., 2003). These composite samples were collected during select time periods of stormwater events. Samples were immediately placed on ice and delivered to the CDFA's Center for Analytical Chemistry in Sacramento within 48 hours of collection. Samples were then weighed and filtered with 0.45µ filter paper. After extraction, samples were stored in a –5°C freezer until analysis using Agilent Model 5973 GC-MSD with a HP-5MS or equivalent GC column. Analysis was performed in the selective ion monitoring mode. Complete details of the laboratory analyses can be found in Calanchini et al. (2003).

Both diazinon and chlorpyrifos were detected in stormwater runoff in Wadsworth Canal, but only diazinon was observed in the Main Canal.

2.4 Artificial substrate baskets

Cylindrical baskets, six inches in diameter and twelve inches in length, were constructed of plastic coated wire (0.5 inch diameter) mesh. The baskets were filled with a two-thirds to one-third mixture of large smooth gravel (2 inch diameter) and small crushed gravel (> 0.5 inch). Eight substrate baskets, secured to a (3 ft X 3 ft) wood pallet using plastic ties (Figure 2), were deployed at each site in a representative section of the stream. The pallet was secured in the stream channel using rebar. The pallet was oriented so stream flow would pass through the baskets in a longitudinal direction.

Adequate colonization time is essential for collecting representative BMI samples from artificial substrates (e.g., Rosenberg and Resh, 1982; Taylor and Kovats, 1995). A colonization time of at least four to six weeks was allowed prior to each sampling event. To characterize the baseline BMI community, two baskets (replicates) from each site were sampled after a four-week colonization period and prior to any storm events. The remaining sampling times were associated with storm events.

Each sampling event consisted of sampling two baskets (replicates) at a site. Baskets were gently removed from the pallet while wading in the stream channel and surrounding the basket with a 500 um mesh D-frame kick-net. Each basket was then placed into a tub of water, the mesh hand-scrubbed free of BMIs, and then emptied into the tub. The gravel was gently rinsed free of BMIs and removed from the tub. The contents of the tub were poured through the 500µm mesh d-frame kick-net, rinsed, and then placed into a sample container and preserved with 95% ethanol. The BMI collected from each basket were not composited, but rather processed separately as replicate samples.

2.5 Sub-sampling and taxonomy

Sub-sampling, the removal of 300 BMIs from each sample, was performed by hand using stereo-microscopes (7X minimum magnification). Each sample was first emptied into a 500µm sieve and gently rinsed to remove the majority of small particles. Large debris such as gravel, leaves, or twigs were removed after inspection for clinging BMI. The sample was homogenized as best as possible, and emptied into a white gridded (2 x 2 inch grids) tray. Grids, or grid partitions, were randomly processed until 300 BMI were removed from the sample. For abundance calculations, once a grid or grid partition was started, it was completely processed. All BMI removed after the 300 count were placed into a separate 'extra' vial. All processed material was placed in a 'remnant' container, and all remaining sample material was placed in an 'original' container and covered with alcohol. For quality assurance purposes, 'remnant' material was inspected for BMI in ten percent of the samples. For sub-sampling protocols, the UCD-ATL implements a ten percent rule, where no 'remnant' should contain more than ten percent of the total organisms removed from it. None of the samples violated this rule.

Taxonomic identification of the 300-sample BMI was completed under a stereo-microscope (17.5 to125X). If needed some BMI were slide-mounted and observed under a phase contrast compound microscope. Insects were identified to the genus level, except if monotypic designated as species. Most non-insect taxa were identified to the genus level, but several taxa could be distinguished only to family or higher level. Chironomidae midges were identified to tribe and oligochaetes to family. All taxa from a sample were sorted, counted, and placed into separate vials containing a site identification and taxon label. For quality assurance purposes, ten percent of the project samples were sent to an outside laboratory for taxonomic and enumeration verification. Tom King, of Bioassessment Services, was contracted for this task. All quality assurance samples were found to have correct identification and enumeration.

2.6 Trend monitoring

2.6.1 *Environmental parameters*

Weekly site visits (from 11/13/02 to 4/16/03) were conducted to document BMI and zooplankton abundance from stream bank samples, measure physical habitat variables (including depth, velocity), and measure conventional water chemistry. Temperature, DO, pH, turbidity, SpC, and chlorophyll were measured using an YSI 6600 multi-probe. The chlorophyll concentrations measured in this study were considered semi-quantitative because they were taken *in vivo* with a fluorescence probe (YSI 6025 chlorophyll sensor). The *in vivo* method does not involve disrupting the cells, as compared to the time-consuming and more costly quantitative laboratory extractive analyses. The limitations of the *in vivo* chlorophyll method include non-differentiation between the various forms of chlorophyll (a,b,c) and pheophytin a. However, the chlorophyll readings made *in vivo* will reflect changes (or trends) in chlorophyll from site to site, or over time at a site. Fouling (build up of biological and/or chemical debris) of the probes, which could lead to erroneous readings, was not observed because the probe was cleaned and calibrated prior to each spot sample event.

2.6.2 *BMI and zooplankton field observations*

A rapid visual survey of BMI and zooplankton observations (Rapid Bioassessment Level 1) was conducted by sampling the edge habitat from approximately a 50 meter reach adjacent and downstream of artificial substrate baskets with a 500µm d-frame kick net (Barbour et al., 1999). Stream bank samples were collected as a composite of 20 jabs of edge substrates and sweeping of overhanging and wetted stream vegetation. The composite sample was placed into a 12 inch by 12 inch white plastic tray with approximately ½ inch of field water. BMI and zooplankton families were visually enumerated in the field and recorded as either dominant (> 50 organisms), abundant (>100 organisms), common (3-9 organisms), rare (1 organism), and/or absent/not observed (0 organisms).

2.7 Rainfall data and pesticide use data

Rainfall data for Marysville, California were obtained from the California Department of Water Resources California Data Exchange Center (CDEC). Pesticide use data for 2002 and 2003 were obtained from the California Department of Pesticide Regulation (DPR) Pesticide Use Reporting (PUR) database.

2.8 Statistical analyses

Simple linear correlations were used to examine potential relationships of BMI colonization with the environmental parameters measured during the weekly site visits. Cluster analysis and nonmetric multidimensional scaling (NMS) were conducted to examine the relationships (associations) of benthic colonization, physical habitat conditions, water quality parameters, field survey data, and pesticide use data during the stormwater season at each site.

3. Results

Results are reported by waterway since the intent of this project was to examine each site independently through time rather than compare sites to one another. Artificial substrates in Jack Slough, Main Drain, and Wadsworth Canal were sampled four times post a four-week initial colonization time period, and coordinated with rainfall events (December 13, 2002, January 8, 2003, February 20, 2003, and March 12, 2003; Table 2).

Rainfall data are illustrated in Figure 3. The December 13 sampling event was taken prior to any rain events, after the initial colonization period, and to establish baseline conditions for each ADW. Jack Slough had an additional sampling on February 11, 2003 immediately following a large spike in turbidity. The largest rainfall events occurred in the beginning (12/13/02 to1/08/03) and toward the end (3/14/03t to 3/16/03) of this investigation.

Pesticide use data are reported (Bacey et al., 2004; Calanchini et al., 2003) only for those chemicals observed in surface water samples at the study sites. Pesticide use data are reported as county totals for each compound separated into three time periods 1/1/03 to 1/8/03, 1/9/03 to 2/20/03, and 2/21/03 to 3/12/03 (Table 3). These time periods coincide with rainfall events and sampling of artificial substrate baskets. The intermediate time period (1/9/03 to 2/20/03) had the highest total pesticide use in all three counties. Water samples taken during the early season heavy rain period (January 3, 2003) produced no significant mortality to C*eriodaphnia dubia* (Table 4).

3.1 Jack Slough

3.1.1 *Aquatic toxicity and pesticides*

Pesticide concentration data were collected during the January 22 and February 15 rain events at one-hour intervals for eight hours during peak flow. On January 22 only one of the eight hourly samples produced significant *Ceriodaphnia* mortality (35%--Figure 4). In the 1/22/ 03 samples diazinon concentrations (0.098 to 0.138 ug/L) were below the *Ceriodaphnia* LC_{50} level. Only trace amounts of chlorpyrifos and the herbicide diuron were observed in these samples. The pyrethroid insecticides, permethrin and esfenvalerate, were not detected during this storm event.

Three of the eight hourly samples collected on February 15 resulted in 40, 90, and 85% *Ceriodaphnia* mortality (Figure 4). Diazinon was present in each toxic sample at 0.195, 0.107, and 0.161 ug/L, respectively. These concentrations are below the *Ceriodaphnia* LC_{50} for diazinon. Some characteristics of these water samples may have potentiated diazinon and/or diazinon to act additively or synergistically with other contaminants in the samples. Other pesticide detections during this storm event included the herbicides diuron, bromacil and simazine. Pyrethroid insecticides were not detected.

Water samples collected by the UCD-ATL on February 13 caused 100% and 33% mortality to *Ceriodaphnia* (Table 5) and *Hyalella,* respectively. Only the *Ceriodaphnia* mortality was significantly different compared to laboratory control water. No pesticide analyses were performed on water samples collected on February 13. Samples collected on March 15 were not toxic to *Ceriodaphnia* (Table 6).

3.1.2 *Artificial substrate colonization trends*

Jack Slough was dominated by naidid (oligochaetes), crangonyctid (amphipods--genus *Crangonyx*), and hydrid (genus *Hydra*) taxa (Figure 5 and Appendix A). EPT taxa were extremely rare and consisted of only four organisms from two genera. Chironomid taxa also were collected in relatively low numbers. Over the course of the study, taxa in Jack Slough shifted from *Hydra*/Naididae dominated to *Crangonyx/*Naididae dominated (Figure 5). EPT taxa, Shannon diversity, and taxonomic richness metric scores were lower in Jack Slough than at all other sites (Tables 7, 8). Conversely, Jack Slough had the highest percent dominant taxon and highest taxa abundance metric scores. Although metric changes over time appear insignificant, BMI community dominance shifted throughout the study from oligochaetes to *Hydra* to amphipods.

3.1.3 *Environmental parameters*

Appendix D summarizes Jack Slough environmental variable data. Chlorophyll concentrations increased slightly from early December through early February (Figure 5). However, chlorophyll concentrations varied considerably (high: 11.6 µg/L; low 1.7 µg/L) from mid-February through the end of the study (April 16). Jack Slough was the most turbid waterway in this investigation. Turbidity averaged 26 NTU at the beginning and end of this study. Turbidity peaked on February 7 at 391 NTU. This was surprising as no rainfall occurred for two weeks prior to the peak turbidity. The high turbidity reading was potentially related to release from rice fields. Temperature and pH were relatively constant. However, temperature tended to decrease during storm events and was increasing during the last sample events in April. Conductivity was relatively constant during most of the study, but increased substantially in samples gathered in April (peak: 538 µS/cm). Velocity was relatively consistent throughout the study, but increased during storm events. The highest velocity reading (2.2 ft/s) was recorded during the large rain events during mid- to late-December. Most other velocity measurements were less than 1 ft/s. Depth also was greatest (170 cm) during the large storm events of mid- to late-December. The lowest depth measurement (29 cm) was noted on March 7.

3.1.4 *BMI and zooplankton field observations*

BMI abundance observed in weekly field surveys of stream bank samples was relatively consistent throughout the study. Zooplankton abundance was greatest in November through December (Figure 6). Although zooplankton abundance tended to vary depending on available stream edge habitat conditions, generally zooplankton was common until mid-March. From mid-March through the end of this study (April 16) zooplankton were absent from stream bank samples. This was surprising as habitat conditions (available riparian vegetation/pools) were favorable and consistent with conditions during November and December.

3.1.5 *Analyses*

Selected metric values are presented in Table 8. Correlations of environmental variables and metrics for Jack Slough are reported in Table 9. BMI communities in Jack Slough changed during the study period from being dominated by oligochaetes, *Hydra*, and chironomids (12/02) to consisting of predominately amphipods (3/03). Deeper water and abundance of planktonic cladocerans and copepods were correlated with December BMI communities, while higher SpC, pH and DO were associated with March amphipod dominated communities (Figure 7). During this transition, abundance of *Hydra* and clams spiked during the 2/11/03 sampling event. None of the measured environmental variables were strongly associated with this change in BMI community. The most significant correlation between environmental variables and taxa metrics was between chlorophyll concentrations and *Hydra* abundance ($r = 0.9094$; $p<0.05$; Table 9). However, *Hydra* abundance also was significantly correlated with SpC ($r = -0.7222$; $p<0.05$) and turbidity ($r = 0.8230$; $p<0.05$). Amphipod abundance was correlated with depth ($r = -0.7376$; $p<0.05$), DO ($r = 0.8366$; $p<0.05$), and SpC ($r = 0.7029$; $p<0.05$).

Observations of *Ceriodaphnia* mortality and occurrence of pesticides in Jack Slough samples were consistent with high pesticide use periods. *Ceriodaphnia*, but not *Hyalella*, mortality was observed in Jack Slough samples. The occurrence of toxicity and pesticides appeared to be of short duration, and associated with rain events.

3.2 Main Drainage Canal

3.2.1 *Aquatic toxicity and pesticides*

Continuous pesticide data were collected January 10 through January 15, 2003 and from February 13 through February 20, 2003. These data are reported in 8-hour composite samples as collected by streamside auto-samplers. Diazinon was detected only on January 10 at 0.021 µg/L during the January sampling event, and only in trace amounts thereafter (Figure 8). Simazine and chlorpyrifos also were detected only at trace amounts throughout this period. No other pesticides were detected.

During the February 13 through February 20 sampling event simazine and diazinon were detected most frequently, also with trace amounts of chlorpyrifos, methidathion, and carbaryl. However, all pesticide concentrations were below *Ceriodaphnia* and *Hyalella* toxicological threshold values. Samples collected on February 13 were found to be non-toxic to *Ceriodaphnia* (Table 5) and *Hyalella*. Samples collected on March 15, 2003 produced no significant mortality to either species (Table 6).

3.2.2 *Artificial substrate colonization trends*

The Main Drain was characterized by the highest tolerance values, but also characterized by some of the highest positive BMI metric scores in this study (Table 10). The baskets collected in December had the lowest average sample abundance (157) of all baskets collected. Average sample abundance was greater than 1000 on all other sample events. The samples collected in December and early January were dominated by oligochaete worms and chironomids, respectively (Figure 9 and Appendix B). EPT taxa were rare but present in all samples, with 23 organisms comprising five taxa. Chironomids were co-dominant in the January samples, but overall, were collected in moderately small numbers (18% total abundance for Main Drain).

3.2.3 *Environmental parameters*

Appendix D contains environmental variable data collected from the Main Canal. Chlorophyll concentrations ranged from 2.2 to 15.7 µg/L from November through early February in the Main Canal. Chlorophyll concentrations were lower from mid-February through mid-April, ranging from 0.8 to 4.1 µg/L. Turbidity was typically below 20 NTU, with a peak of 51 NTU during December rain events. Temperature was relatively consistent throughout the study with a peak (17.6˚ C) on April 10. Dissolved oxygen was relatively low in the beginning of the study, ranging from 2.9 to 3.7 mg/L prior to the first storm. Conductivity tended to increase over the course of the study ranging from 223 to 329 µS/cm in late November/early December to ranging from 559 to 578 µS/cm in late March/early and April. Similar to other agricultural waterways in this study, conductivity patterns did not appear to be related to rainfall patterns. Depth ranged from 72 to 200 cm in the Main Canal. Velocity was generally slow, ranging from 0.1 to 1.6 ft/sec. Peak velocity measurements were associated with storm events in December.

3.2.4 *BMI and zooplankton field observations*

BMI abundance observed in weekly field surveys from stream bank samples was relatively consistent throughout the study. Zooplankton abundance was greatest in November through December. Although zooplankton abundance tended to vary depending on available stream edge habitat conditions, generally zooplankton was common at all times until late February (Figure 10). From February 27 through March 12 zooplankton was absent from stream bank samples irrespective of habitat conditions (available riparian vegetation/pools) being favorable and consistent with conditions during November and December.

3.2.5 *Analyses*

Selected BMI metrics are presented in Table 8. Table 11 summarizes correlation values of environmental variables with Main Drain BMI metrics. The greatest change in Main Drain artificial substrate BMI communities over the sampling period was from naidid oligochaete domination to a higher prevalence of *Hydra* and snails. The greatest abundance of Hydra was observed in February. This change in community composition was associated with higher pH, higher SpC, as well as increased application of diazinon and simazine in Butte County (Figure 11). The two replicates collected from the Main Drain in December 2002 were the only replicate baskets to consist of notably different communities. One basket contained more mayflies, chironomids, and amphipods, while more flatworms characterized the other basket. Chlorophyll correlated significantly ($r = -0.8054$; $p < 0.05$) with EPT Index. Dissolved oxygen was significantly correlated with site abundance ($r = 0.7644$; $p < 0.05$). *Probezzia* (Ceratopogonidae—Diptera) abundance correlated significantly with DO ($r = 0.7648$; $p < 0.05$), pH ($r = 0.8148$; $p < 0.05$), and SpC ($r = 0.7365$; $p < 0.05$).

3.3 Wadsworth Canal

3.3.1 *Aquatic toxicity and pesticides*

Pesticide data were collected following the January 22 and February 15 rain events at eight- and nine-hour intervals, respectively. In January 22 samples diazinon concentrations ranged from 0.106 to 0.130 ug/L in the January 22. Only trace levels of chlorpyrifos were observed, and no pyrethroids were detected. The following herbicides also were observed in the January 22 samples: simazine, diuron, and norflurazon. No *Ceriodaphnia* toxicity was observed.

On February 15 diazinon concentrations ranged from 0.102 to 0.246 ug/L in surface water samples, trace levels of chlorpyrifos and esfenvalerate also were detected, as well as permethrin at 0.094 ug/L. One of nine hourly samples resulted in 100% mortality to *Ceriodaphnia*. The sample taken an hour later, despite also containing diazinon, diuron, bromacil, and norflurazon below toxicological significance values, resulted in 40% *Ceriodaphnia* mortality. Permethrin was detected only in the sample with the 100% mortality (Figure 12).

Continuous pesticide data were collected January 10 through January 14, 2003 and from February 13 through February 20 (Figure 13). During the January sampling event diazinon and chlorpyrifos peaked at 0.300 and 0.021 ug/L, respectively. During the

February sampling event diazinon and chlorpyrifos peaked at 0.960 and 0.030 ug/L, respectively. The peak diazinon and chlorpyrifos concentrations were observed on February 16. None of the samples collected on February 16 were toxic.
Samples taken by UCD-ATL on February 13 resulted in no mortality to *Ceriodaphnia* (Table 5). Diazinon (0.060 to 0.093 ug/L) and chlorpyrifos (0.012 to 0.013 ug/L) concentrations in these samples were below *Ceriodaphnia* and *Hyalella* toxicological threshold values.

3.3.2 *Artificial substrate colonization trends*

Wadsworth Canal manifested the highest taxonomic richness, highest number of EPT taxa, and highest Shannon diversity metric scores (Table 12). In general, EPT taxa were rare, but higher than the other ADW sites, with 55 individual organisms representing seven taxa. Crangonyctid (genus *Crangonyx*) amphipods, coenagrionid damselflies, and chironomids dominated December samples whereas January samples were dominated by oligochaetes, chironomids, and damselflies (Figure 14; Appendix C). February and March samples revealed a notable shift to chironomids with naidid oligochaetes relatively common; damselflies and amphipods became relatively uncommon. The community shifts in Wadsworth Canal may have been related to predator-prey interactions, as discussed in section 4.3. Most BMI metrics were consistent throughout the study, with the exception of percent Odonata and percent amphipods both of which decreased, while percent Chironomidae increased.

3.3.3 *Environmental parameters*

Appendix D contains environmental variable data collected from Wadsworth Canal. Chlorophyll concentrations ranged between 0.7 (November 29) and 6.3 µg/L (December 20). The lowest concentrations of chlorophyll were observed at the beginning and end of the study. Turbidity ranged between 1.4 and 36 NTU, with the highest turbidity occurring during the large rain events in December. Temperature ranged between 8.4 and 22.7˚ C, with lowest temperatures during the December rain events and highest temperatures during the last sample event in April. Dissolved oxygen ranged between 9.0 and 10.2 mg/L from November 13 through January 8. The lowest dissolved oxygen concentration was noted on January 24 (6.4 mg/L). Dissolved oxygen concentration peaked on March 7 and March 12 at 14.6 and 15.4 mg/L, respectively. The supersaturated oxygen measurements may reflect high photosynthesis rates. The pH in Wadsworth Canal ranged between 7.7 and 8.3. Conductivity ranged from 165 to 622 μS/cm. Conductivity increased over the study period, but sharply declined in the last week from 616 (April 10) to 165 μS/cm (April 16). Velocity and depth ranged between 0.1 to 1.3 ft/s and 34 (February 7) to 124.5 cm (January 8), respectively. The greatest depth measurements reflected the large rain events in December. Depths were lowest in early- to mid-February.

3.3.4 *BMI and zooplankton field observations*

BMI abundance recorded in weekly field surveys from stream bank samples at Wadsworth Canal was relatively consistent throughout the investigation. Zooplankton abundance was greatest in November through early December (Figure 15). Zooplankton was rare in one sample event in December (December 20). This was not surprising as the channel was flooded from heavy rains and there was no edge habitat from which to collect samples. Zooplankton abundance tended to vary between common and abundant during most of January. However, zooplankton was absent from stream bank samples mid- to late- February. This result was unexpected since habitat conditions (available riparian vegetation/pools) were favorable and consistent with conditions during November and early December.

3.3.5 *Analyses*

Table 8 summarizes selected BMI metrics. Correlation values of environmental variables with Wadsworth Canal BMI metrics are reported in Table 13. BMI communities in Wadsworth Canal were represented most faithfully by a 1-dimensional ordination (an ordination possessing only one axis) along one NMS axis (Figure 16). This axis juxtaposed communities dominated by chironomids, damselflies, and amphipods (12/02 samples) with communities dominated by mayflies, caddisflies, and chironomids (2/12/03 and 2/20/03 samples). This change in community composition appeared to be associated with increases in pH, SpC, and in use of the herbicide simazine in Sutter

County. All environmental variables measured were significantly correlated with at least one metric. Further, most environmental variables were significantly correlated with at least four or more metrics.

3.4 Sutter Bypass

3.4.1 *Aquatic toxicity and pesticides*

Toxicity and pesticides were not measured at either site in the Sutter Bypass due to funding limitations.

3.4.2 *Artificial substrate colonization trends*

Artificial substrate baskets were deployed at two sites (4 and 5) in the Sutter Bypass. However, due to flooding conditions baskets were retrievable from site 4 on a limited basis. Consequently deep-water baskets at site 5 were not sampled. Prior to the final sampling event, the baskets were not submerged as water levels suddenly decreased. BMIs collected from substrate baskets at site 4 were archived for analyses at a later date, funding permitting.

3.4.3 *Environmental parameters*

Environmental variable data collected from the Sutter Bypass sites appear in Appendix D. Chlorophyll concentrations at site 4 ranged from 1.8 to 13.4 µg/L. The peak chlorophyll concentration was observed on January 2 and the lowest concentration was noted on January 31. Turbidity ranged from 15 (November 13) to 559 NTU (January 2). Turbidity was usually less than 50 NTU, with spikes occurring after large rain events. Temperature ranged from 7.9˚ to 18˚ C, increasing in April. Dissolved oxygen, pH, and SpC ranged between 7.2 and 12.2 mg/L, 6.1 and 8.1, and 94 and 372 µS/cm, respectively.

3.4.4 *BMI and zooplankton field observations*

As with the other sites, BMI abundance noted in weekly field surveys from stream bank samples were relatively consistent throughout this study. Similar to the ADW sites, zooplankton abundance was greatest in November and December (Figure 17). The lowest zooplankton abundance was observed on January 8, when the slough flooded and sampling occurred along the non-vegetated levee since edge habitat was not available. On February 27 zooplankton was present (> 50 organisms). However, in March 12 through March 27 sampling events, zooplankton abundance decreased to between 3 to 9 organisms. The decreasing zooplankton abundance is similar to, although not as notable as, the trends observed in the upstream ADWs.

3.4.5 *Analyses*

Statistical analyses were not performed on samples collected from the Sutter Bypass due to inconsistent sample collections.

4. Discussion

4.1 Colonization of artificial substrates

Artificial substrate was used to evaluate BMI colonization trends in association with environmental parameters during the winter rain season in ADWs. Artificial substrate reduces BMI community variability associated with variable substrate composition. This approach assumes that the BMI community colonizing the artificial substrate reflects environmental parameters of concern, especially water quality. The ADW sites assessed in this investigation are not pristine and likely have been subject for many years to multiple stressors (poor physical habitat and water quality) associated with agricultural land use. Repeated, long-term exposure to such stressors most likely has influenced ADW BMI communities, complicating location of the true background (reference) state/sites for comparison.

Multivoltine and collector BMI taxa were prevalent at all three ADW sites. A majority of taxa colonizing the artificial substrate were tolerant (to degraded water quality and/or habitat) or of unknown tolerance. These results are not surprising given the poor physical habitat and water quality conditions in ADWs. Physical habitat is so degraded in most ADWs it may be difficult to distinguish water quality effects from due to degraded habitat on BMI communities. The difficulty of distinguishing water quality effects on BMI communities when physical habitat conditions are impaired has been hypothesized (e.g., Rogers et al., 2002; de Vlaming et al., 2004a).

Dominant taxa on artificial substrate at the three ADW sites were naidid oligochaetes, chironomids, crangonyctid amphipods, and Hydra. However, the overall artificial substrate BMI community structure differed at the three ADW sites even on the same sampling dates. Chironomid taxa were not dominant at Jack Slough, but were in the Main Drain and Wadsworth Canal. Hydra was never a dominant taxon in Wadsworth Canal, but was in Jack Slough and the Main Drain. Change in BMI community structure through time also differed in the three ADWs. Further investigation is needed to define/explain these differences in BMI community structures in the three ADWs as well as the temporal changes. Almost certainly the differences relate to dissimilarity in physical habitat, water quality conditions, and differing source water in the three ADWs at the sampling times and through time.

Artificial substrate BMI community composition in all three ADWs varied temporally. In particular, December and early-January communities tended to diverge from those observed in late-February and March, especially in Jack Slough and the Main Drain. Whether these variations were a natural seasonal phenomenon or related to the high use of pesticides in January and February can not be determined with this data set. More extensive (more sites) and long-term weight-of-evidence studies are essential to fully understand BMI communities and deterministic environmental variables.

4.2 Partitioning stressors

Statistically significant artificial substrate BMI metric correlations with environmental variables did not coincide among the three ADWs. That is, there was no consistent relationship between BMI metrics and specific environmental variables. These results clearly illustrate that correlations must not be interpreted as cause-and-effect relationships. Further, these findings lead us to predict that, as did de Vlaming et al. (2004a), a host of interacting environmental factors determine BMI community structure and condition. While we contend that artificial substrate sampling can be effective in distinguishing water quality variables that impact BMI communities in Central Valley ADWs, this data set was too small to achieve that goal. That is, a larger number of sites must be weight-of-evidence investigated through annual cycles and over years. Biological systems are extremely complex. To gain an adequate level of understanding of BMI community dynamics and deterministic variables will require a substantial spatial, temporal (several years), and economic commitment.

De Vlaming et al. (2004a, b) reported BMI community structure and integrity/condition in ADWs of the Central Valley as reflecting the ‘cumulative’ influences of surrounding land use, habitat conditions, and water quality. However, partitioning the individual influences of each stressor and linkage of cause-and-effect is not possible with bioassessment procedures alone. Using bioassessment data to assess causality is difficult for several reasons including (1) temporal and spatial variation, (2) sampling variability, (3) failure to measure important stressor(s), and (4) interactive effects of many stressors (Stevenson et al., 2004). Other studies also document that a limitation of standard bioassessment protocols is the inability for specific identification of cause(s) associated with BMI community degradation (e.g., Barbour et al., 1996; Clements and Kiffney, 1996; Holdway, 1996; McCarty and Munkittrick, 1996; Wolfe, 1996; Power, 1997; Bart and Hartman, 2000; Adams, 2003). As indicated in the Introduction, a weight-of-evidence approach is preferred in cause(s) of impact/impairment identification.

Water quality and habitat related stressors frequently co-vary in multi-stressed systems. De Vlaming et al. (2004a, b) hypothesized that BMI impacts from water quality related stressors were difficult to identify using a standard bioassessment approach because of the poor physical habitat condition of low gradient agriculture-dominated waterways (ADWs). The ability to distinguish physical habitat effects from water quality (contaminants) related impacts would be useful in understanding the regulatory utility of various BMI bioassessment approaches. The capacity to differentiate relative influences of various water quality related stressors on aquatic communities is critical for better regulatory and management decisions. We do contend that more extensive use of artificial substrate in agriculture-dominated waterways with poor physical habitat (especially unstable substrate) could be helpful in distinguishing water quality issues.

4.3 Toxicity and pesticides in ADWs

Ceriodaphnia mortality was observed in Jack Slough samples collected on 1/22/03 (one sample—33% mortality), 2/13/03 (one sample—100% mortality), and 2/16/03 (three samples—40, 85, and 90% mortality). Diazinon, diuron, and simazine were detected in the toxic samples, but all at concentrations lower than the LC_{50}s. Two Wadsworth Canal samples collected on 1/16/03 caused *Ceriodaphnia* mortality (40 and 100%). Diazinon, diuron, bromocil, norflurazon, and permethrin were identified in these samples at concentrations lower than individual pesticide LC_{50}s. These toxic samples were collected during the period of highest pesticide use in the study. While no one chemical appeared to be responsible for *Ceriodaphnia* mortality in these ADW samples, there is reason to suspect that the mixture of pesticides acted additively or synergistically (more than additive). For example, atrazine at non-toxic concentrations potentiate OP insecticide (chlorpyrifos and diazinon) toxicity to invertebrates (Pape-Lendstrom and Lydy, 1997; Belden and Lydy, 2000). Synergistic interactions of other pesticides and metals with OP insecticides have been documented by others (e.g., Macek, 1975; Fabacher et al., 1976; Bocquene et al., 1995; Forget et al., 1999). These data clearly reveal the need for studies on pesticide mixtures, preferably in matrices mimicking waters in agricultural drains. Previous experiments at UCD-ATL (de Vlaming et al., 2005) documented that the insecticide chlorpyrifos was more toxic (lower concentration) in agricultural drain water than in 'pristine' laboratory control water. Further, use of U.S. EPA TIE methods would help identify cause(s) of bioassay mortality including additive and synergistic effects.

Wadsworth Canal exhibited an artificial substrate BMI response following detection of the highest pesticide concentrations. On February 16 diazinon concentrations peaked around 0.96 ug/L. This level translates into approximately three *Ceriodaphnia* toxic units. Methidathion also was detected at 0.1 ug/L. After this period, the BMI taxa list and metrics manifested a large decline of coenagrionid damselflies and *Crangonyx* amphipods. Chironomids became very dominant, comprising 65% (up from 24%) of BMI abundance. This increased chironomid abundance was most likely due to a predator-prey interaction between the damselflies (known chironomid predators) and the chironomids (prey). Heckman (1981) as well as Lugthart and Wallace (1992) both reported that with removal of predator taxa by pesticide applications, prey species, most notably chironomids and oligochaetes, become numerically dominant and thrive. Whether or not the pesticides affected damselflies in Wadsworth Canal is uncertain without dose response data that suggest the damsel fly larvae are sensitive to the pesticides within the range of concentrations measured. The Heckman (1981) study clearly indicated that the Odonata (damselflies and dragonflies) was a group clearly unable to adapt to insecticides, disappearing completely from the system. That the Wadsworth Canal EPT taxa metric increased after February is puzzling. The EPT taxa identified were, however, fairly tolerant, were not numerically abundant, and were common in many ADWs sampled in a previous study (de Vlaming et al., 2004a).

Main Drain data also suggest a small biological response approximate to the most notable pesticide concentrations. Diazinon concentrations were not particularly high (around 0.14 ug/L), and about 0.5 *Ceriodaphnia* toxic unit. The herbicide simazine also was present at 1.14 ug/L (highest concentration observed). Another insecticide, methidathion, was detected at 0.31 ug/L, about one seventh of a *Ceriodaphnia* toxic unit. As indicated above, very little is known about interactive toxic effects, such as potentiation, of pesticides. However, during the period (February 13 to 20) the positive (indicating good biological condition) metrics taxonomic richness, Shannon diversity, and EPT taxa all decreased, while the negative metrics percent dominant taxon and percent *Hydra* both increased. The most noteworthy change during this sampling period was that *Hydra* abundance increased from 0.1 to 53% of the total BMI abundance. The higher *Hydra* abundance in the Main Drain is equivalent to that present in Jack Slough through most of the study. *Hydra* is a little studied taxon, and its ecological relevance is not fully understood. Although *Hydra* has been present in previously conducted ADW bioassessment studies, it was never a dominant taxon. After the February sampling period, BMI metrics at the Main Drain site indicated improved biotic conditions.

4.4 Pesticide toxicity data for BMIs are limited

Available toxicity data for diazinon and chlorpyrifos indicate a wide range of effect concentrations for BMI and cladocerans (Table 14). The diazinon cladoceran median

EC_{50} and BMI median LC_{50} are 1.22 (n=17) and 25 ug/L (n=19), respectively. Diazinon concentrations in Central Valley ADWs above the cladoceran median effect level, but not above the BMI median effect level, have been reported. Resident BMI toxicity data for most pesticides of interest are sparse in the published literature. Further, the tolerance values reported with bioassessment taxa were not developed as reflecting tolerance to pesticide concentrations.

Median LC_{50} values vary considerably within the same or similar taxa. Munn and Gilliom (2001) reported median LC_{50} values for diazinon ranging from 2 to 185 ug/L for amphipods within the genus *Gammarus* (Table 15). Toxicity testing data are available for two amphipods, *Gammarus lacustris* and *Hyalella azteca,* resident in Central Valley ADWs. Diazinon median LC_{50s} for *G. lacustris* and *H. azteca* are 185 and 6.5 ug/L, respectively (Munn and Gilliom, 2001). Central Valley ADW diazinon concentrations typically do not exceed these values. Given the relatively high diazinon tolerance thresholds for these two amphipods, using them as indicators of diazinon contamination seems inappropriate. Further, the presence of other amphipod species, for which there are no toxicity data, does not assist in discriminating insecticide contamination.

4.5 Utility of artificial substrates in assessing ADWs

Artificial substrate baskets are typically used to standardize habitat, particularly substrate, and limit variation due to habitat differences. In this regard the data gathered in this study reveal that the artificial substrate performed effectively. The taxa list for each set of baskets was uniform (low metrics replicate variability) as was the abundance of each taxon. Artificial substrate sample replicates are, for the most part, less variable than bottom grab or dredge sample replicates (e.g., Dickson et al., 1971; Beak et al., 1973; Mason et al., 1973; Hughes, 1975; Voshell and Simmons, 1977; Freedan and Spurr, 1978; Rabeni and Gibbs, 1978; Meier et al., 1979; Wells and Demas, 1979; Shaw and Minshall, 1980; Morin, 1985; De Pauw et al., 1986; Slack et al., 1986; Clements, 1991). Therefore, artificial substrate sampling generally requires fewer replicates per site than bottom sampling devices to achieve a given precision. This, in turn, establishes a greater ability to statistically distinguish BMI community structure and integrity among sites (e.g., increase in precision also improves statistical test sensitivity because smaller differences between sites can be statistically determined).

One drawback of artificial substrate is the possible creation of a taxa list that is not representative of a particular site. This can occur if the artificial substrate differs from the predominate substrate at the site(s) under investigation. ADWs in Central Valley contain substrates comprised primarily of mixtures of hardpan, clay, sand, gravel, cobble, silt, and mud. Nonetheless, if the objectives of a study are to characterize potential BMI colonizers and to assess water quality, this limitation may be irrelevant. Comparing taxa lists generated in this study with those from an earlier two-year study of ADWs in the Central Valley (de Vlaming et al., 2004a) only one taxon stood out. The damselfly *Argia*, usually a riffle dwelling insect, was not previously collected from Wadsworth Canal. This damselfly is most likely naturally present in Wadsworth Canal. The cluster and NMS analyses also support the performance of artificial substrate since data group together sequentially and spatially by site.

Central Valley ADWs do not have natural flow regimes and are heavily influenced by water augmentation projects. Flow augmentation for agricultural purposes typically results in relatively consistent flows during the summer irrigation season. However, due to upstream agricultural uses and other uses such as flooding fallow fields for waterfowl habitat and hunting results in unnatural retention and release of collected stormwater runoff during the winter season. Therefore, stream levels may fluctuate widely and sporadically during the winter months in ADWs, and may or may not coincide with stormwater events.

The use of artificial substrates in wadeable ADWs requires frequent monitoring of depth and careful placement in the streambed to ensure fluctuating water levels do not compromise data quality. The artificial substrates (gravel baskets) used in this study were monitored weekly through the study duration. Only on a few occasions in Jack Slough and Wadsworth Canal was the placement of the pallets holding the artificial substrates carefully moved while under water to avoid possible suspension out of the water column.

Further, at no time during this study did it appear that the baskets at a given sample time were or had previously been suspended out of the water. The low variability in sample replicates also supports this conclusion.

Shallowness was typically not an issue in this study. However, increased depth during and after storm events did result in high water levels and unsafe conditions for sampling crews to work in stream channels.

Assessment of water quality is the most typical use of artificial substrate sampling (e.g., Anderson and Mason, 1968; Arthur and Horning, 1969; Cairns and Dickson, 1971; Dickson et al., 1971; Beak et al., 1973; Benefield et al., 1974; Hughes, 1975; Hellawell, 1977; Cover and Harrrel, 1978; Rabeni and Gibbs, 1978; Janovic, 1979; Deutsch, 1980; Winner et al., 1980; Jones et al., 1981; DePauw and Vanhooren, 1983; De Pauw et al., 1986, 1994; Tolcamp, 1985; Van Hassel and Gaulke, 1986; Clements et al., 1988, 1989a, 1989b; Metcalfe, 1989; Clements, 1991; Battegazzore et al., 1994). According to US EPA (Weber, 1973) diversity of BMI on artificial substrate is an acceptable method for analysis of water quality. The effectiveness and efficiency of artificial substrate sampling BMI communities have been touted by many (e.g., Anderson and Mason, 1968; Dickson et al., 1971; Benfield et al., 1974; Crossman and Cairns, 1974; Voshell and Simmons, 1977; Cover and Harrel, 1978; Fredeen and Spurr, 1978; Rabeni and Gibbs, 1978; Deutsch, 1980; Shaw and Minshall, 1980; Wefring and Teed, 1980; De Pauw et al., 1986, 1994; Slack et al., 1986; Boothroyd and Dickie, 1989; Clements et al., 1989; Clements, 1991: Battegazzore et al., 1994).

Colonization time allowed in this study appears to have been adequate. Typically recommended colonization time is four to six weeks and the first samples were taken at five weeks. The December artificial substrate samples from Main Drain had the lowest recorded BMI abundance. The low DO (2.9 – 3.7 mg/L) in the Main Drain prior to this sampling period is the likely cause for this observation, hampering the initial colonization rate. Subsequently, DO increased to normal levels, and all baskets sampled were characterized by abundance equivalent to other ADWs, fluctuating little through time.

Most BMI community changes were gradual in all three ADWs and tended to be in a positive (improved biological condition) direction. These shifts likely reflect the progression of colonization of 'new' habitat in the substrate baskets. To test this hypothesis monitoring with artificial substrate should be for a longer period to assess whether stabilization could be achieved. Some evidence for colonization stabilization occurred in Sacramento Slough samples not included in this report. Initial colonizing taxa were mostly multivoltine (short-lived) with the later samples comprised primarily of longer-lived taxa. However, a thorough analysis of the Sacramento Slough samples was not performed.

4.6 Zooplankton trends

Weekly qualitative field observation assessments (presence/absence and relative abundance) of BMI and zooplankton groups in aquatic edge habitat were used to identify taxa group trends throughout the study. Major BMI taxa group shifts were not observed. Zooplankton abundance and chlorophyll decreased from late-February though mid-April. Whether this was a natural- or anthropogenic-caused event is unknown. The highest concentrations of pesticides were detected just prior to this period. The herbicide simazine was detected most often, as well as diuron and norflurazon. However, these chemicals were not present in all samples. The relationships among algae/chlorophyll, herbicides, and zooplankton diversity as well as abundance deserve further investigation. Although a critical component of freshwater ecosystem food webs little is known about zooplankton communities and abundance in waterways of California's central valley. In agriculture-dominated and -influenced waterways zooplankton diversity and abundance could well be impacted by insecticides and other pesticides. Downstream effects from agricultural drainage on Sacramento/San Joaquin Delta zooplankton are also possible.

4.7 Metrics and tolerance values

The issue of metrics used to interpret these data requires further scrutiny. Most of the metrics performed as expected and are not contentious. However, this and several other studies in California reveal the need for updating and developing BMI, as well as zooplankton, tolerance values. Most BMI tolerance values are based on organic pollution and poorly defined criteria. Although many BMI taxa labeled as sensitive respond to a wide range of stressors, responses to pesticides and several other variables are unknown for most invertebrate taxa. A considerable quantity of water chemistry data was available for this investigation. Nonetheless, the ecological relevance of these data, for the most part, remains unknown because of incomplete information on invertebrate tolerance levels. Refined or more specific tolerance values for a larger number of BMI and zooplankton would greatly enhance data interpretation and lend credence to collecting complete chemical analyses on water column samples, as well as enhancement of stressor identification.

5. Recommendations

• Need for updating, as well as developing, tolerance values for BMI and zooplankton communities.
• Need to study zooplankton abundance and diversity in agricultural areas of the Central Valley. It is thought that many of the backwater sloughs in the Central Valley serve as nursery grounds for zooplankton drifting to the downstream Sacramento-San Joaquin River Delta and San Francisco Estuary.
• BMI bioassessments using artificial substrates should be used as a component of monitoring projects intended to assess potential water quality impacts. This recommendation applies particularly to low gradient waterways with marginal to poor habitat conditions. We recommend a weight of evidence approach that may include toxicity testing (sediment and water column) with associated toxicity identification evaluations (TIE), chemical analyses, or other appropriate procedures. This approach is particularly essential if a study objective is to specifically identify the cause(s)/stressors responsible for community perturbations.
• Weight of Evidence studies using artificial substrates require extensive spatial and temporal analyses for use in ambient water quality assessment.

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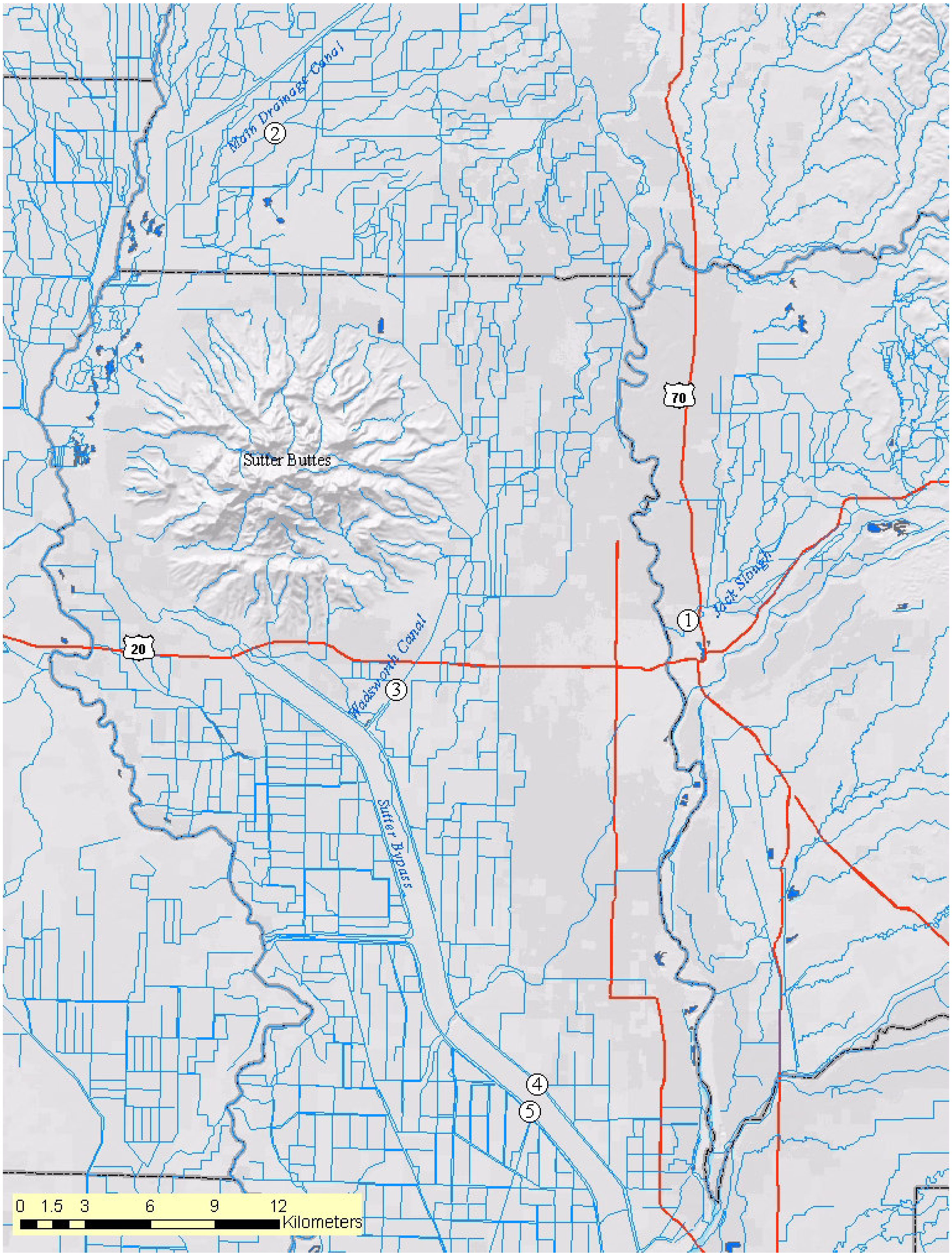

Figure 1. Map of sample sites for artificial substrate study. Numbered circles depict site locations. See Table 1 for site descriptions.

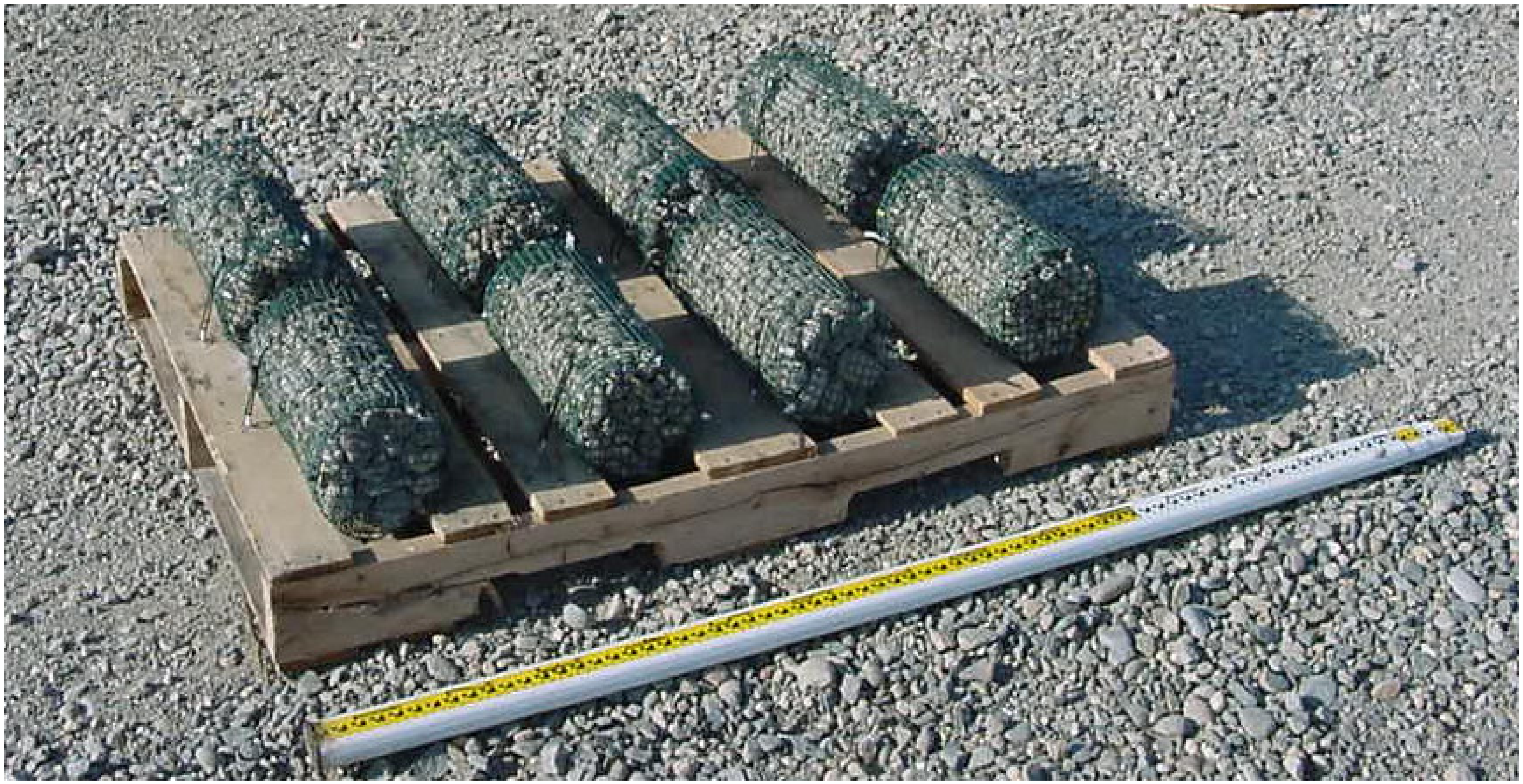

Figure 2. Artificial substrate baskets secured to pallet.

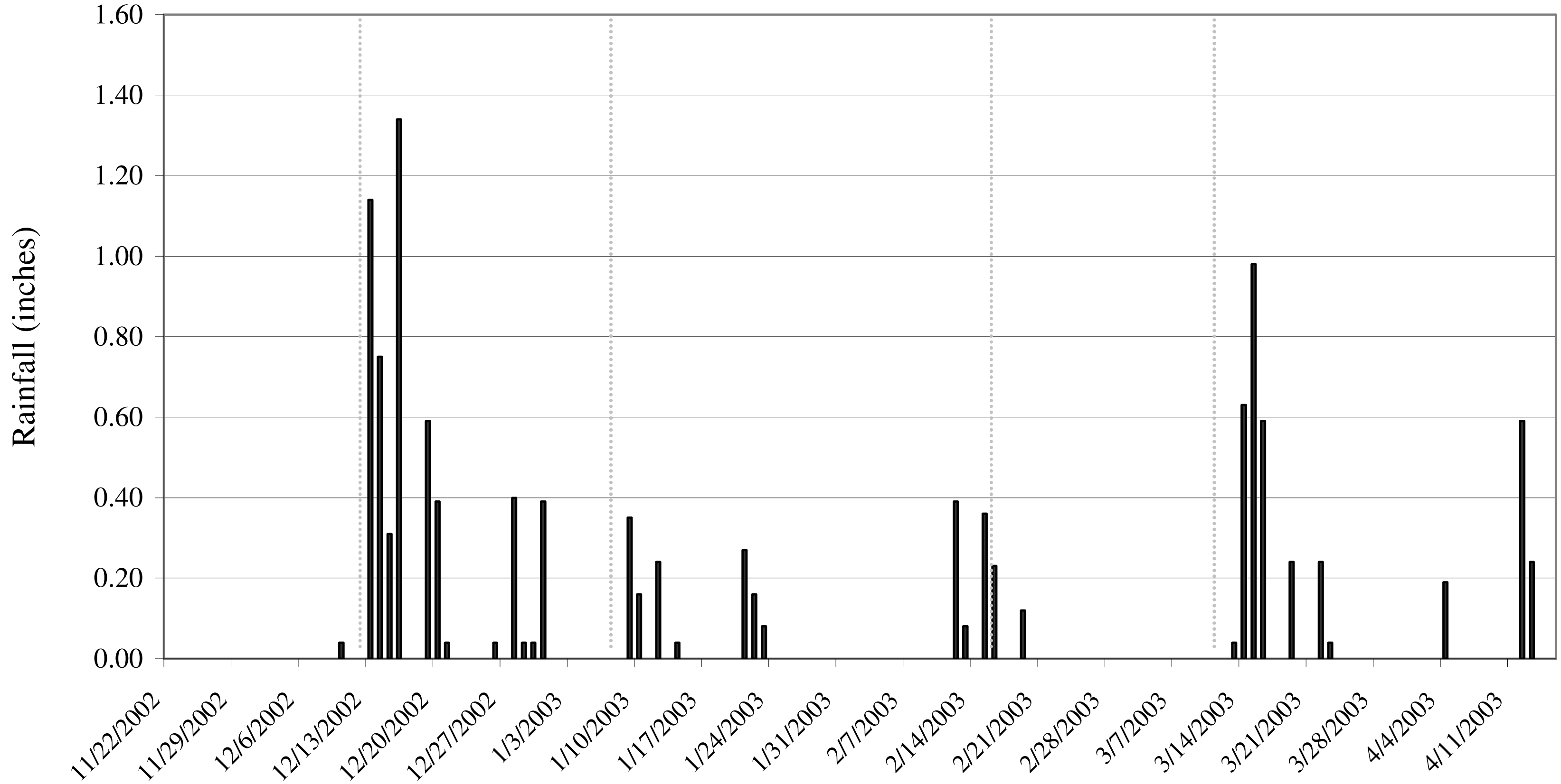

Figure 3. Rainfall data (inches) at Marysville, CA during the November 2002 to April 2003 artificial substrate study. Artificial substrates were collected on 12/13, 1/8, 2/11 (Jack Slough only), 2/20, and 3/12 from Jack Slough, Main Drainage Canal, and Wadsworth Canal.

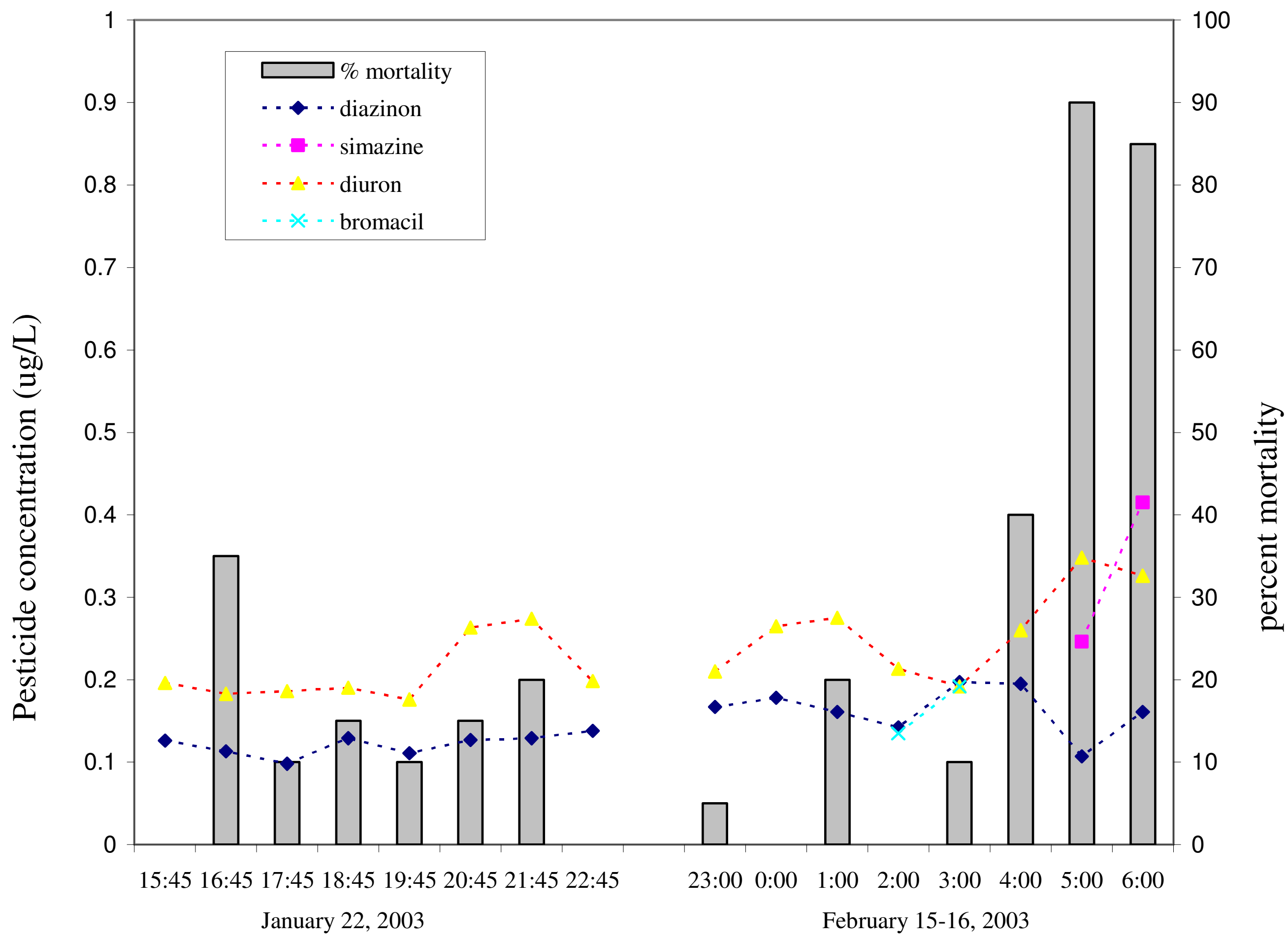

Figure 4. Pesticide detections (ug/L) and toxicity (*Ceriodaphnia dubia* percent mortality) detected in Jack Slough samples during two storm events in 2003. Source of data: Bacey *et al*., 2004.

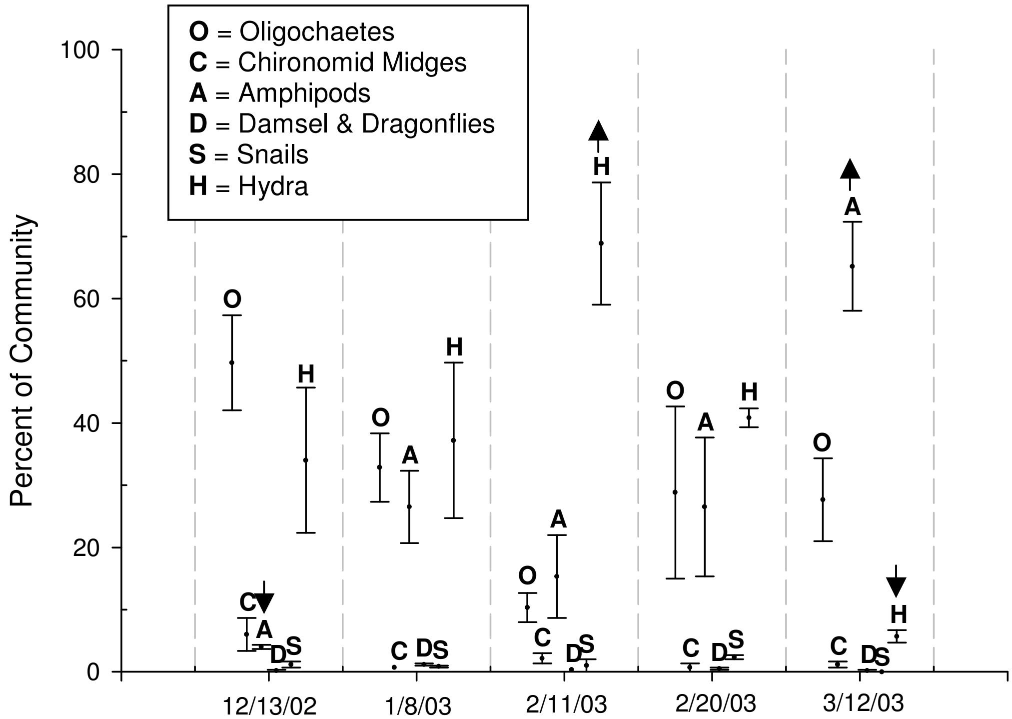

Figure 5. Profile of artificial substrate BMI community parameters that correlated with environmental variable changes in Jack Slough during winter 2002 to 2003. Plot of community composition indicates the mean of BMI taxa (bracketed by standard deviation) from two replicate artificial substrate baskets expressed as a percentage of total BMI community. Community components were significantly higher at timepoints marked with upward arrows than at timepoints marked with downward arrows.

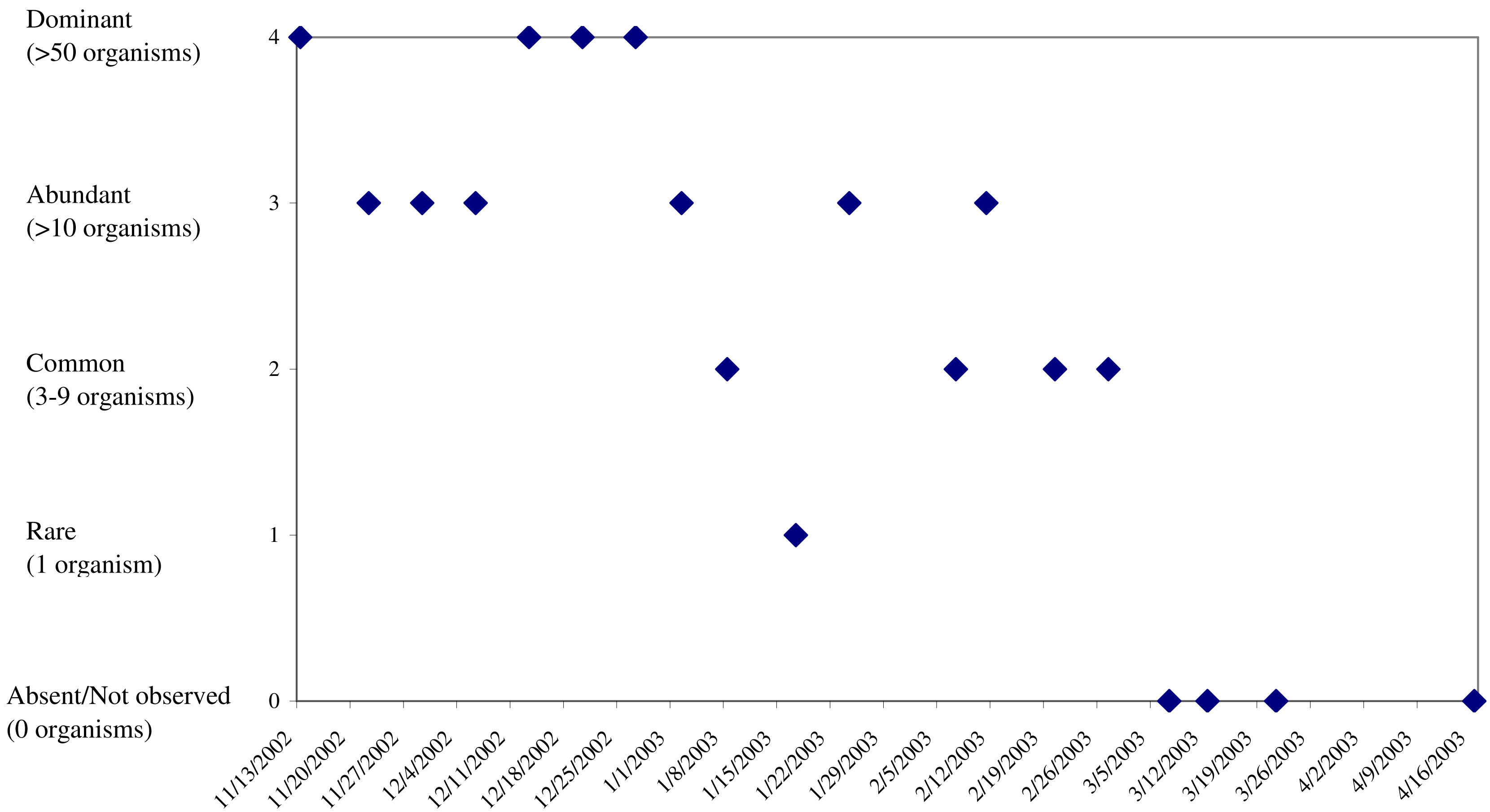

Figure 6. Estimated zooplankton abundance observed in Jack Slough November 2002 to April 2003 as determined by field observation using the method outlined by Barbour *et al*., (1999).

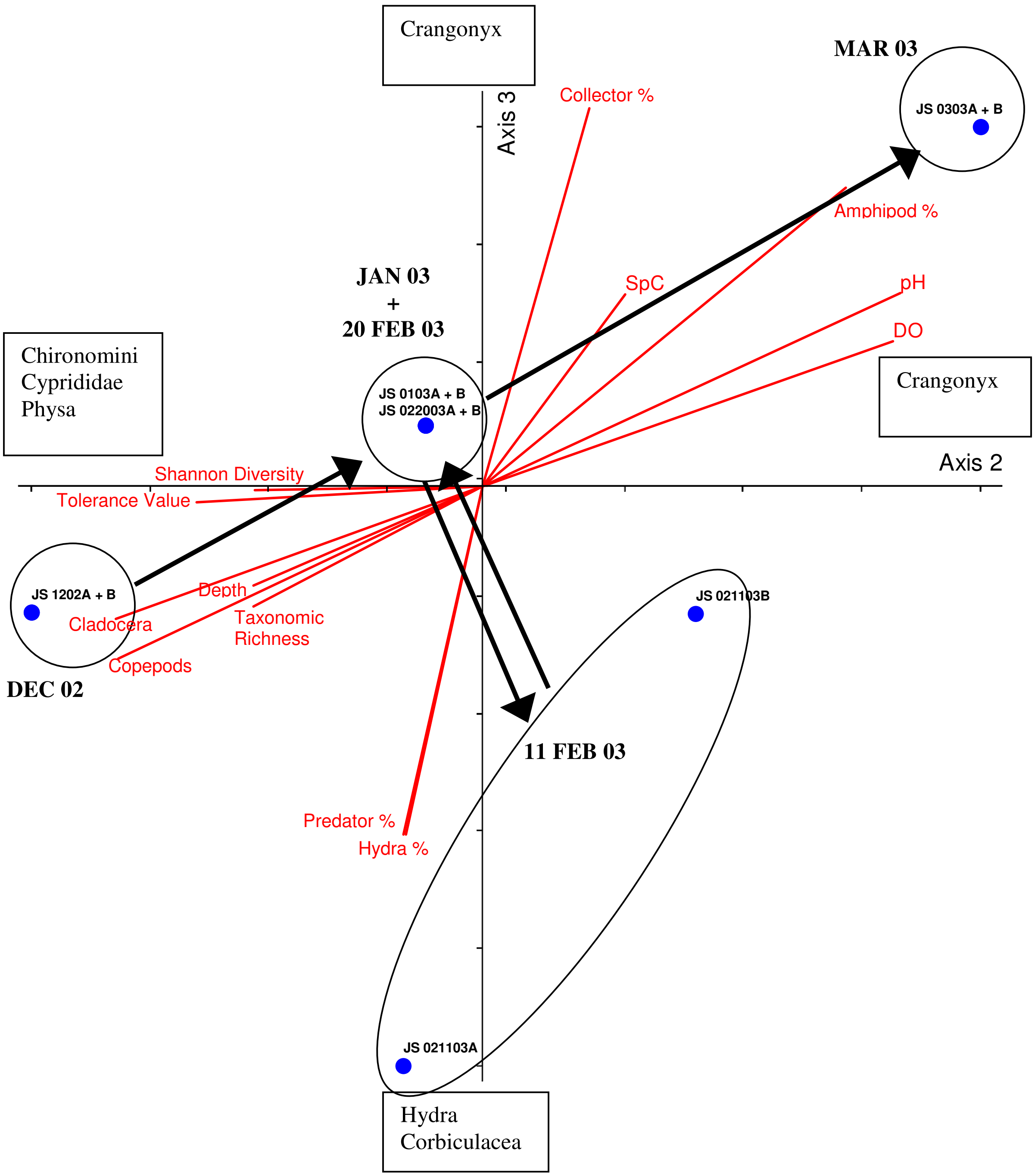

Figure 7. NMS ordination of artificial substrate BMI community composition data from Jack Slough during the winter of 2002 to 2003. “A” and “B” refer to artificial substrate basket replicates. Replicate samples collected during the same event are circled. Arrows indicate direction of change in community composition through time. BMI taxa, environmental parameters, and BMI community metrics associated with the NMS axis at $r^2 > 0.45$ are shown.

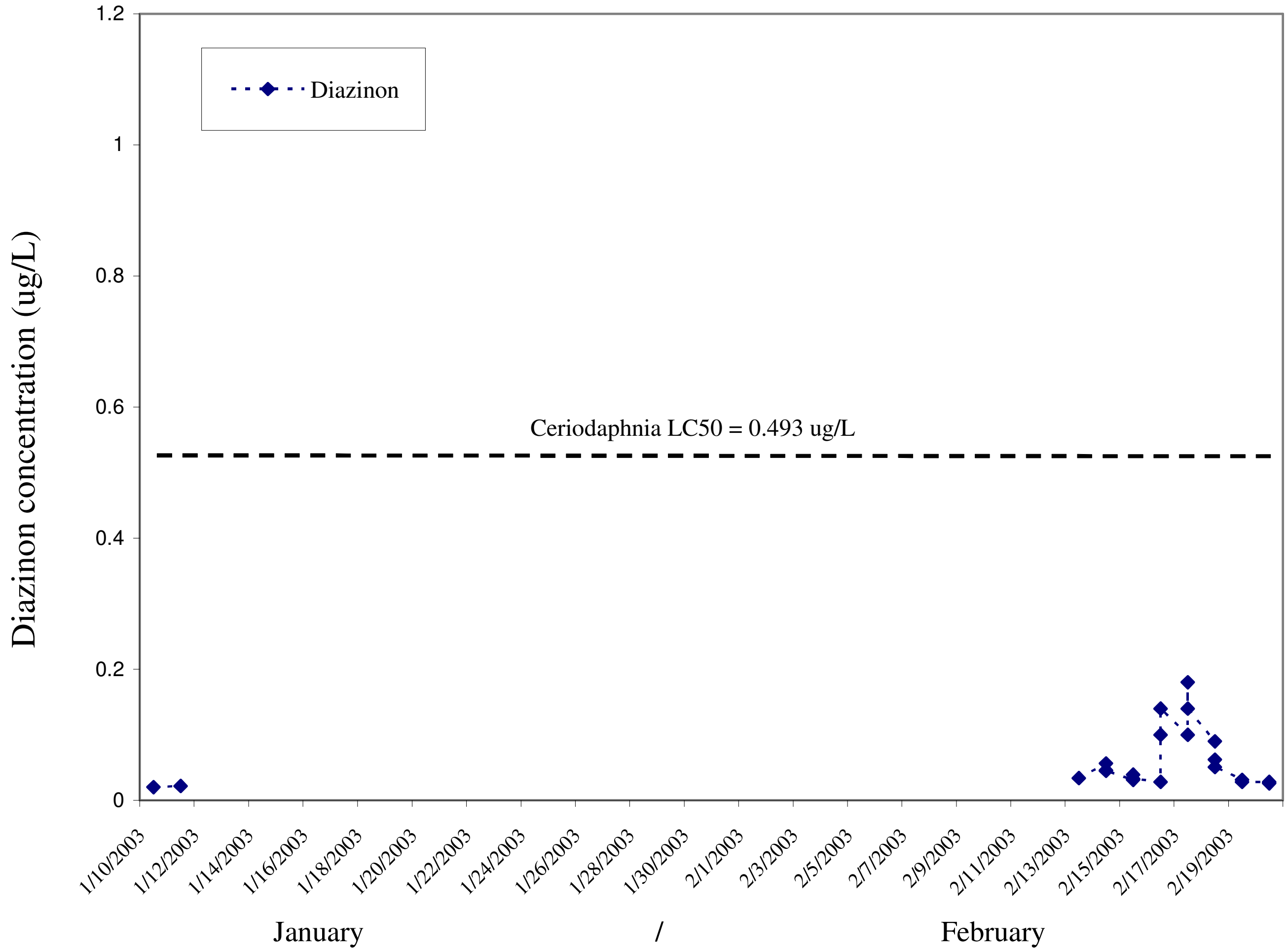

Figure 8. Diazinon concentrations detected in the Main Drainage Canal during January 10 to 16, and February 13 to 20, 2003. Source of data: Calanchini *et al*., 2003.

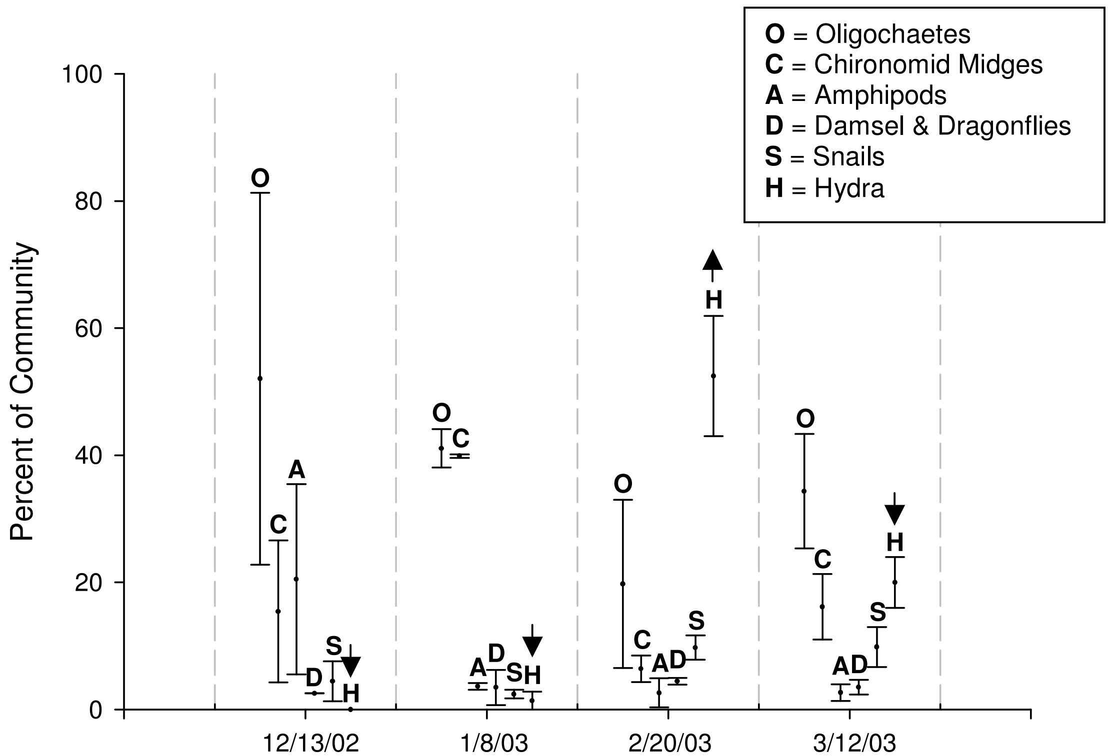

Figure 9. Profile of artificial substrate BMI community parameters correlated with environmental variable changes in Main Drainage Canal during winter 2002 to 2003. Plot of community composition indicates the mean of BMI taxa (bracketed by standard deviation) from two replicate artificial substrate baskets expressed as a percentage of total BMI community. Community components were significantly higher at timepoints marked with upward arrows than at timepoints marked with downward arrows.

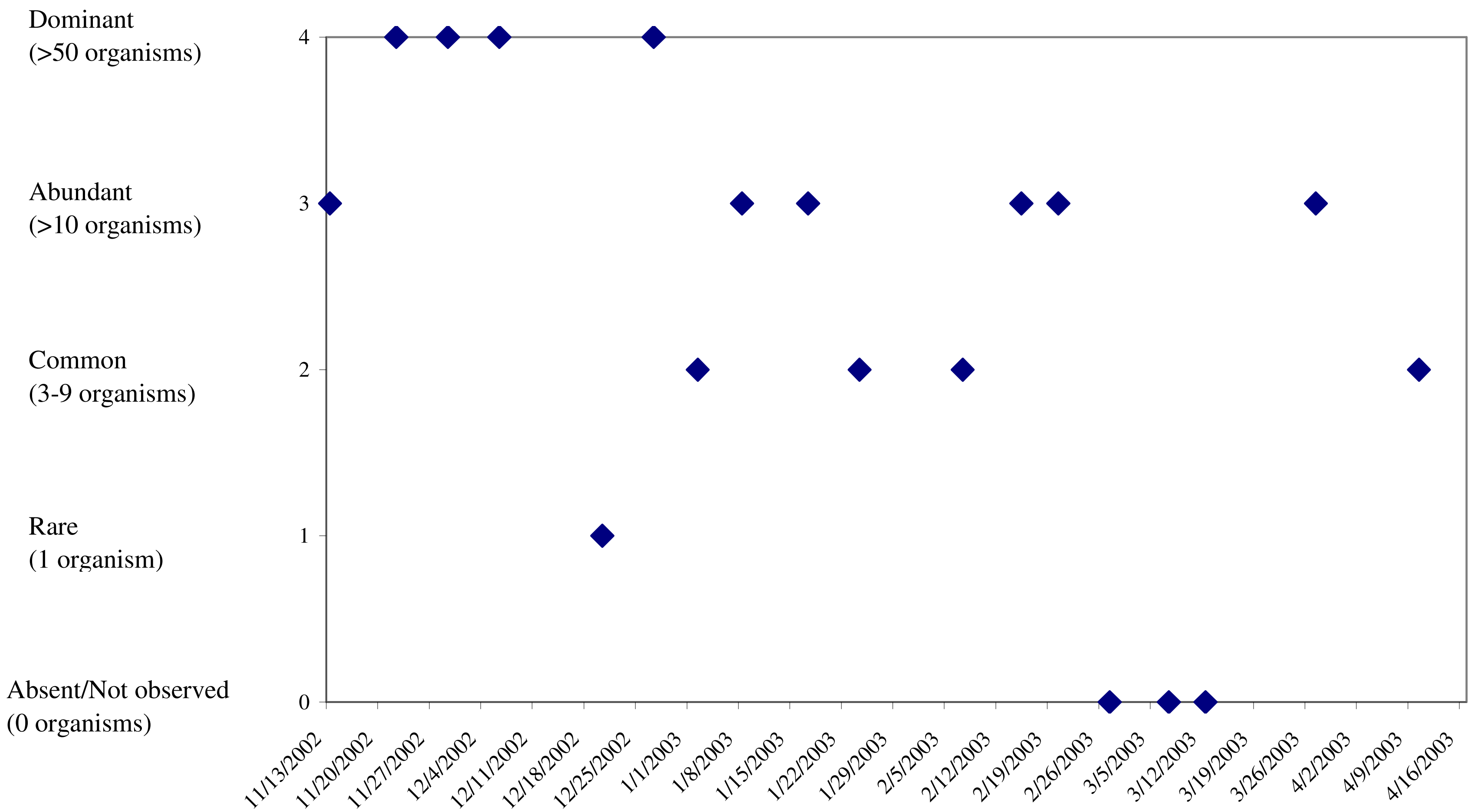

Figure 10. Estimated zooplankton abundance observed in the Main Canal November 2002 to April 2003 as determined by field observation using the method outlined by Barbour *et al.,* (1999).

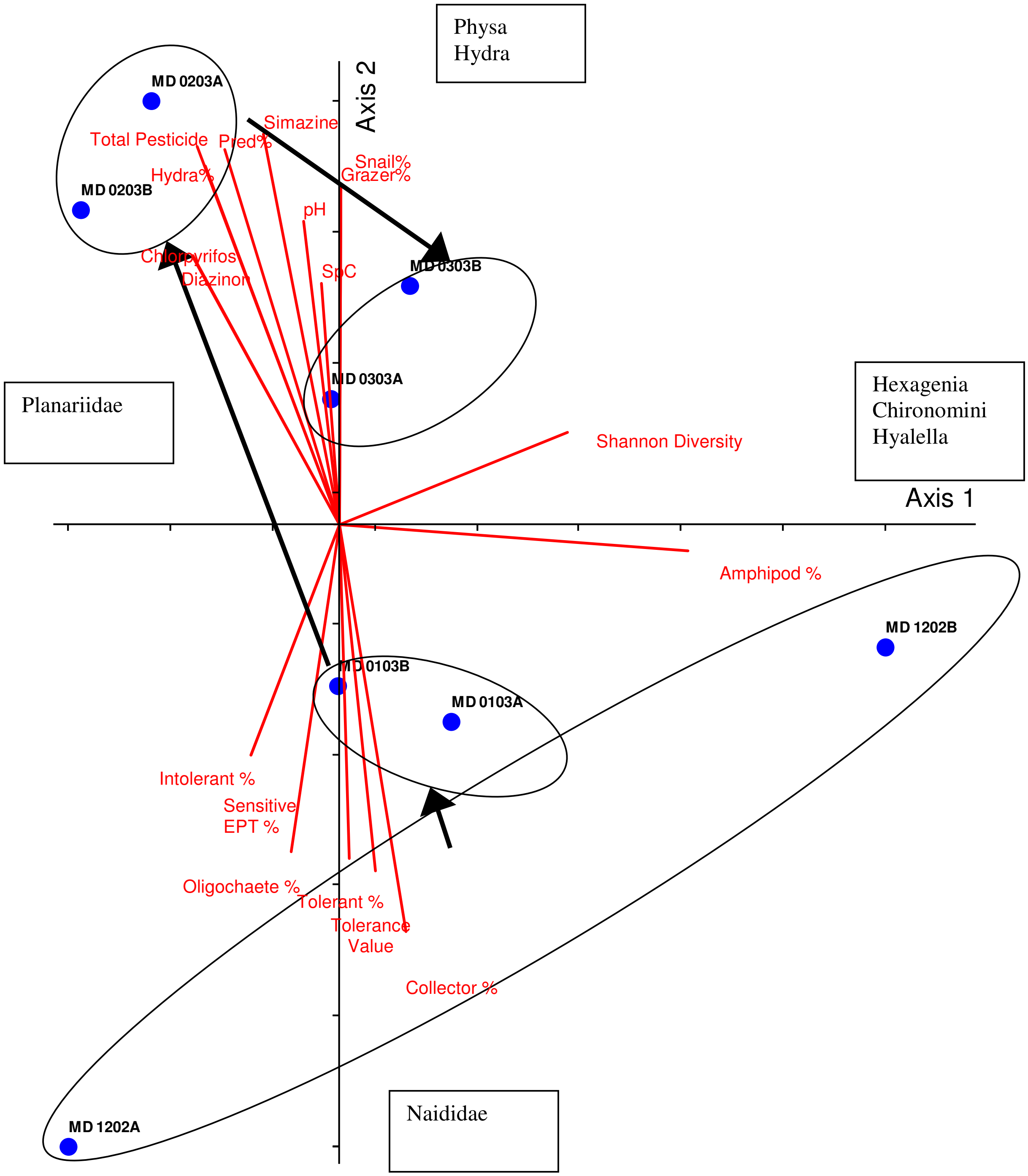

Figure 11. NMS ordination of artificial substrate BMI community composition data from Main Canal during the winter of 2002 to 2003. “A” and “B” refer to artificial substrate basket replicates. Replicate samples collected during the same event are circled. Arrows indicate direction of change in community composition through time. BMI taxa, environmental parameters, and BMI community metrics associated with the NMS axis at $r^2 > 0.45$ are shown.

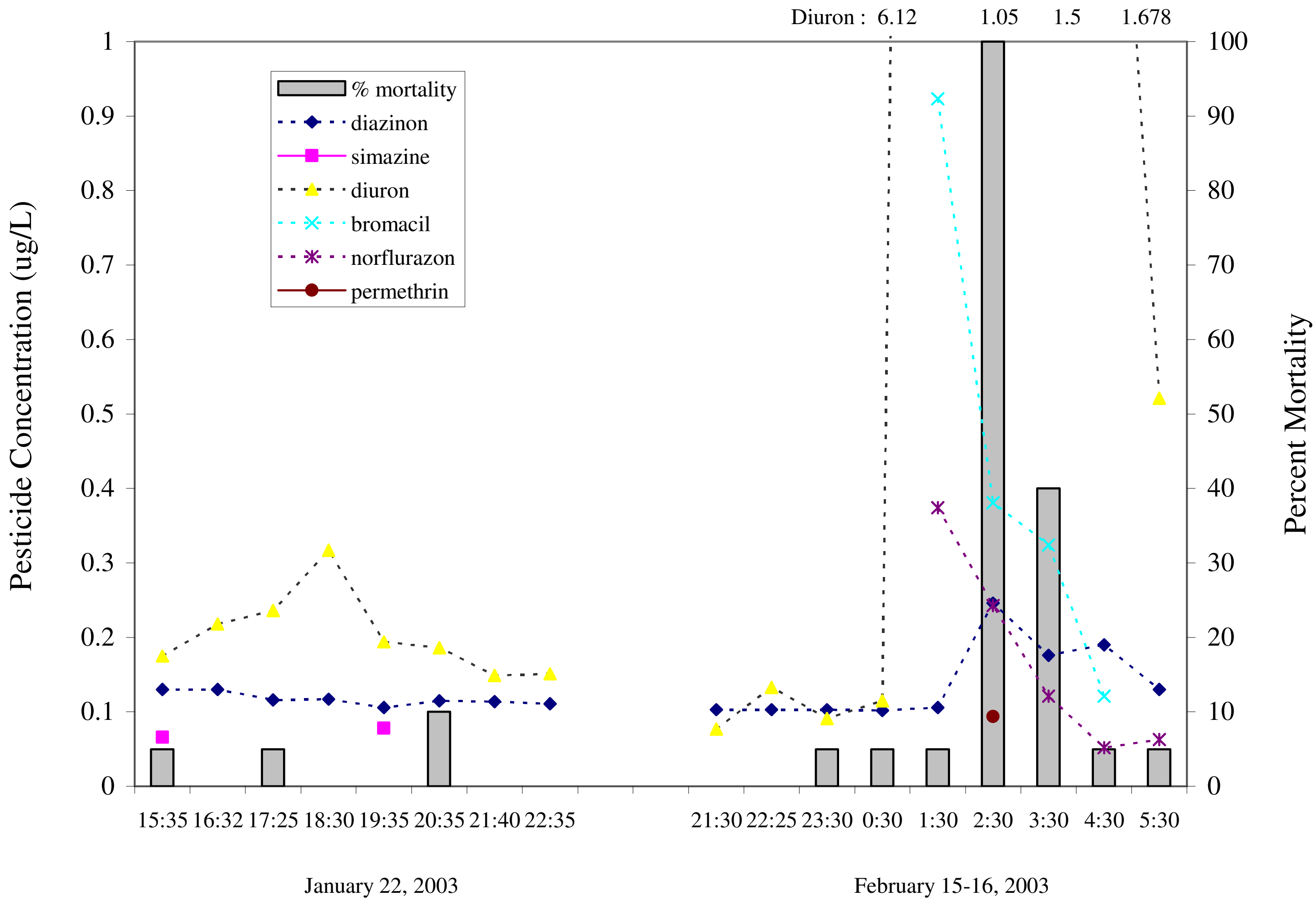

Figure 12. Pesticide detections (ug/L) and toxicity (*Ceriodaphnia dubia* percent mortality) detected in Wadsworth Canal during two storm events in 2003. Source of data: Bacey *et al.*, 2004.

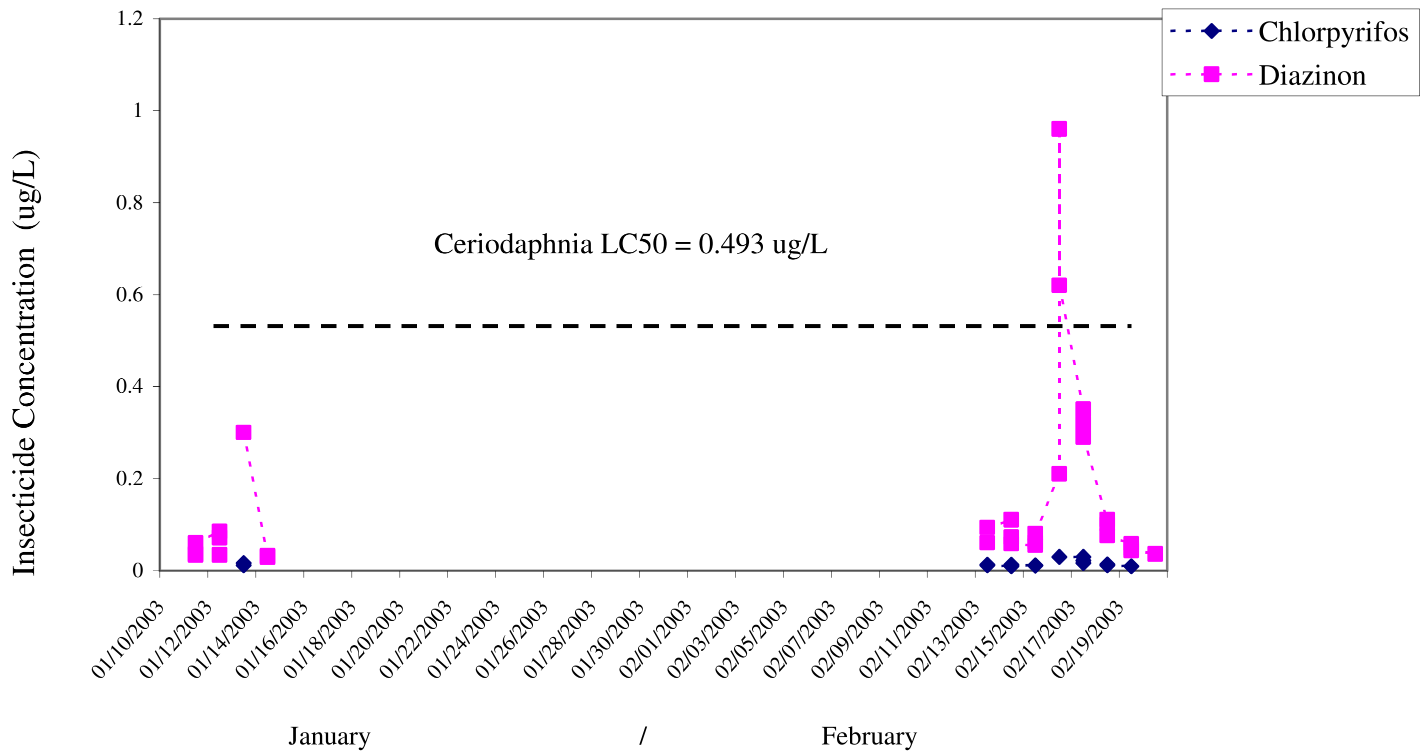

Figure 13. Chlorpyrifos and diazinon concentrations detected during January 10 to 16, and February 13 to 20, 2003 in Wadsworth Canal. Source of data: Calanchini *et al.,* 2003.

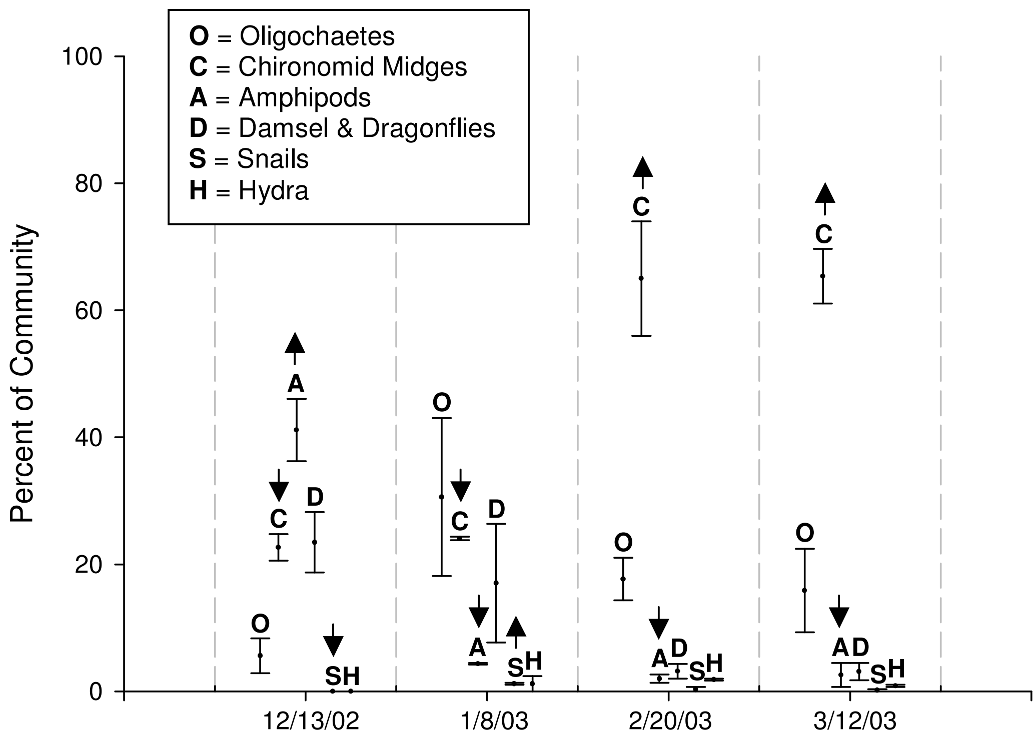

Figure 14. Profile of artificial substrate BMI community parameters correlated with environmental variable changes in Wadsworth Canal during winter 2002 to 2003. Plot of community composition indicates the mean of BMI taxa (bracketed by standard deviation) from two replicate artificial substrate baskets expressed as a percentage of total BMI community. Community components were significantly higher at timepoints marked with upward arrows than at timepoints marked with downward arrows.

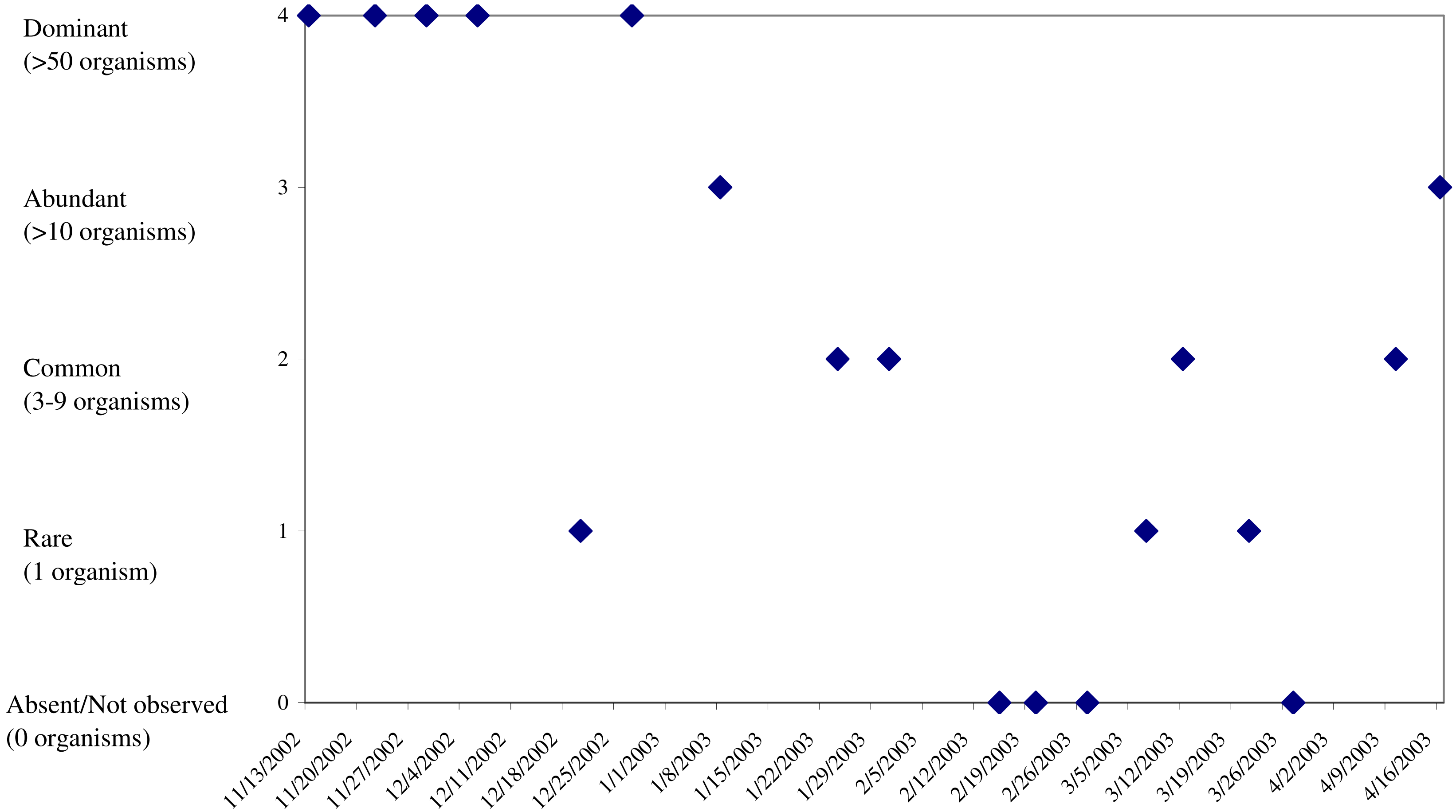

Figure 15. Estimated zooplankton abundance observed in Wadsworth Canal during November 2002 to April 2003 as determined by field observation using the method outlined by Barbour *et al.,* (1999).

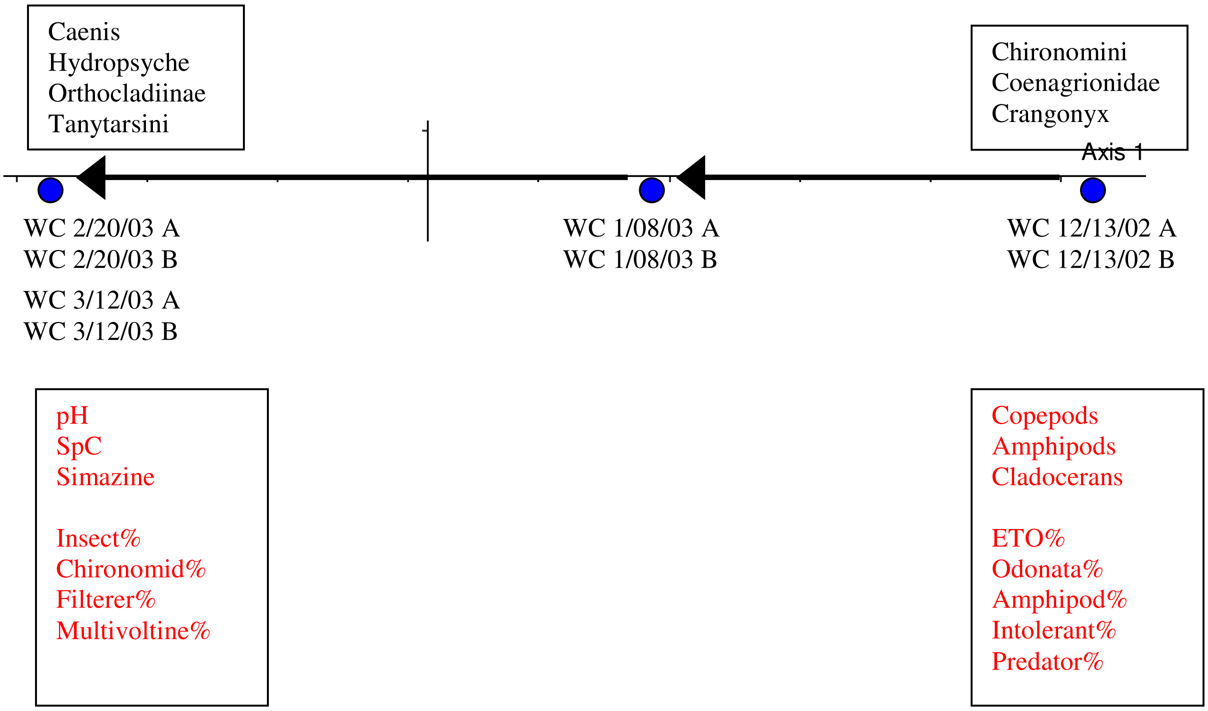

Figure 16. NMS ordination of artificial substrate BMI community composition data from Wadsworth Canal during the winter of 2002 to 2003. “A” and “B” refer to artificial substrate basket replicates. Arrows indicate direction of change in community composition through time. BMI taxa, environmental parameters, and BMI community metrics associated with the NMS axis at $r^2 > 0.45$ are shown.

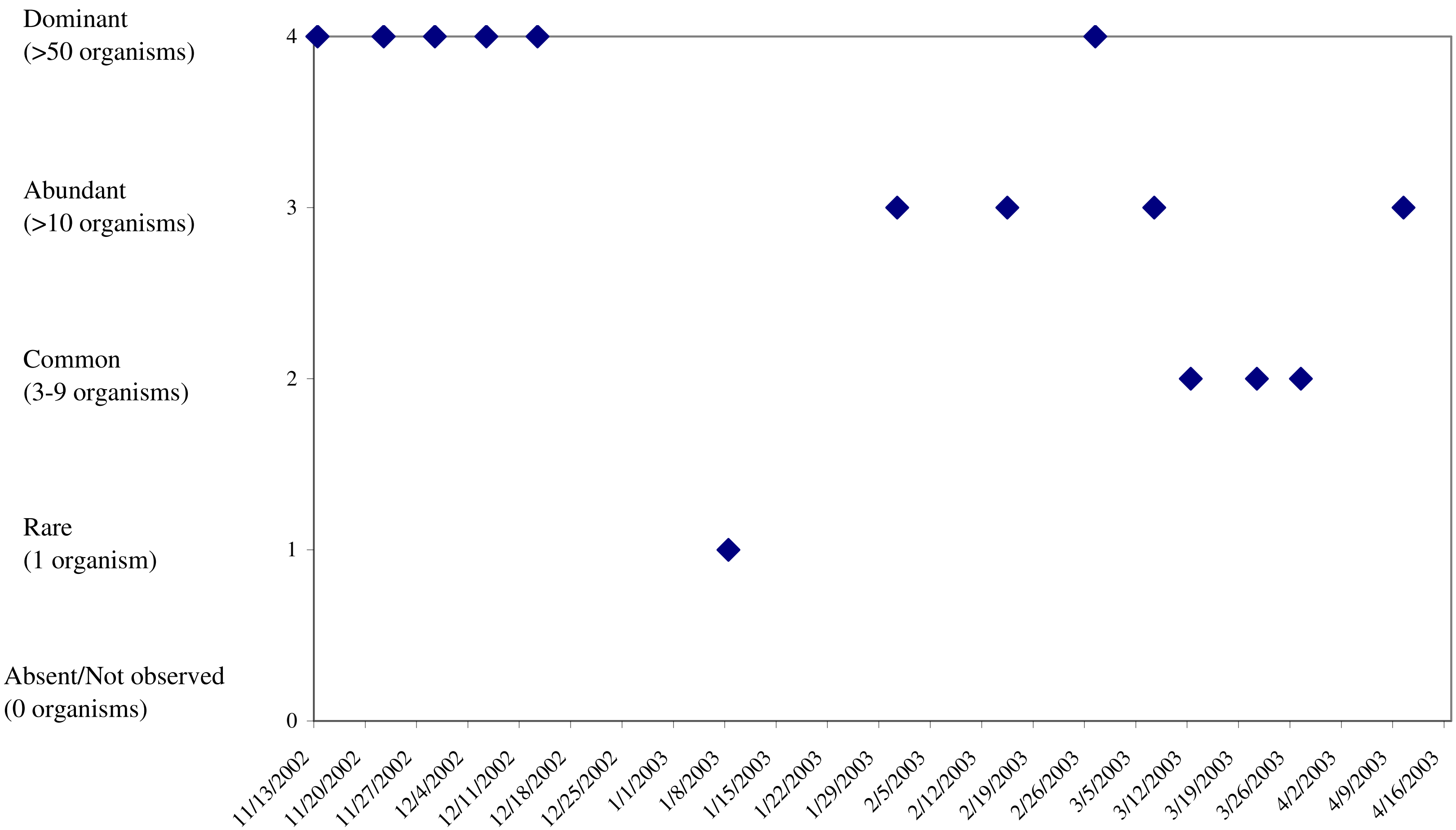

Figure 17. Estimated zooplankton abundance observed in the Sutter Bypass West during November 2002 to April 2003 as determined by field observation using the method outlined by Barbour *et al.,* (1999).

Table 1. Sampling site locations.

Site	Location	County	Latitude	Longitude
1	Jack Slough	Yuba	38.9549	-121.6769
2	Main Drainage Canal	Butte	38.9665	-121.6731
3	Wadsworth Canal	Sutter	39.1303	-121.7529
4	Sacramento Slough (East)	Sutter	39.1617	-121.5964
5	Sacramento Slough (West)	Sutter	39.3623	-121.8241

Table 2. Timeline of biological, toxicological, and chemical sample collections during the November 2002 to April 2003 artificial substrate study.

	11/22-12/13	12/14-1/8	1/9-2/20	2/21-3/12	3/13-4/11
Biology Artificial Substrates[1] Field Surveys[2]	12/13 ← 5 surveys →	1/8 ← 4 surveys →	(2/11) 2/20 ← 7 surveys →	3/12 ← 3 surveys →	← 4 surveys →
Toxicity Testing *Ceriodaphnia*[3] *Hyalella*[4]		1/3 1/3	1/22 2/13, 2/15-16 2/13		3/15
Chemistry CVRWQCB[5] DPR[6]			1/10-1/15, 2/13-2/20 1/22, 2/15-2/16		

[1] Artificial substrate baskets were deployed 11/22/02. Two replicate baskets were collected at each sample event. Jack Slough was also sampled on 2/11.
[2] Field surveys were made approximately once a weeks as conditions permitted.
[3] *Ceriodaphnia* 96-hour static renewal toxicity tests with percent mortality as endpoint.
[4] *Hyalella* 96-hour static renewal water column toxicity tests with percent mortality as endpoint.
[5] CVRWQCB OP TMDL (Calanchini et al, 2003) chemistry monitoring in Main Canal and Wadswoth Canal.
[6] Department of Pesticide Regulation (Bacey et al, 2003) chemistry monitoring in Jack Slough and Wadsworth Canal.

Table 3. Pesticide use (pounds) for selected compounds reported for Butte, Sutter, and Yuba counties during the 2003 storm water season.

County		1/1/03-1/8/03	1/9/03-2/20/03	2/21/03-3/12/03
Butte				
	Bromacil			34
	Chlorpyrifos		909	84
	Diazinon	15	6,510	599
	Diuron	28	1,870	2,012
	Esfenvalerate	1	188	31
	Norflurazon	51	2,353	601
	Permethrin		3	
	Simazine	188	2,366	2493
	Totals:	283	14,199	5,854
Sutter				
	Chlorpyrifos	980	4,343	73
	Diazinon	1,052	18,395	57
	Diuron	148	2,299	312
	Esfenvalerate	42	451	68
	Norflurazon	3		
	Permethrin	22	50	
	Simazine	150	478	484
	Totals:	2,397	26,016	994
Yuba				
	Chlorpyrifos		2,630	
	Diazinon	50	7,721	301
	Diuron		385	82
	Esfenvalerate	6	286	108
	Norflurazon		126	
	Simazine		828	93
	Totals:	56	11,976	584
Grand Totals:		2,736	52,191	7,432

Table 4. Summary of 96-hour *Ceriodaphnia* toxicity tests on samples collected in Sacramento Watershed on 3 January 2003.[1]

Treatment	Mortality (%)[1] ($\bar{x} \pm$ s.e.)
Laboratory Control	$0^{P} \pm 0.0$
Jack Sl @ Doc Adams	0.0 ± 0.0
Main Canal @ Gridley Hwy	0.0 ± 0.0
Wadsworth Canal @ Acacia	0.0 ± 0.0
Sutter Bypass west @ Hwy 113	0.0 ± 0.0
Sutter Bypass east @ Hwy 113	0.0 ± 0.0

[P] The laboratory control met the criteria for test acceptability.

[1] Test was set up on 4 January 2003.

Table 5. Summary of 96-hour *Ceriodaphnia* toxicity tests on samples collected in Sacramento Watershed on 13 February 2003.[1]

Treatment	Mortality (%)[1] (x ± s.e.)
Laboratory Control	0[P] ± 0.0
Jack Sl @ Doc Adams	100.0 ± 0.0
Main Canal @ Gridley Hwy	10.0 ± 5.8
Wadsworth Canal @ Acacia	0.0 ± 0.0
Sutter Bypass west @ Hwy 113	0.0 ± 0.0
Sutter Bypass east @ Hwy 113	5.0 ± 5.0

[P] The laboratory control met the criteria for test acceptability.

[1] Test was set up on 14 February 2003.

Table 6. Summary of 96-hour *Ceriodaphnia* toxicity tests on samples collected in Sacramento Watershed on 15 March 2003.[1]

Treatment	Mortality (%)[1] (x ± s.e.)
Laboratory Control	0.0[P] ± 0.0
Jack Slough @ Doc Adams	0.0 ± 0.0
Main Canal @ Gridley	0.0 ± 0.0
Wadsworth Canal @ Acacia	0.0 ± 0.0
Sutter Bypass east @ Hwy 113	5.0 ± 5.0
Sutter Bypass west @ Hwy 113	5.0 ± 5.0

[P] The laboratory control met the criteria for test acceptability.

[1] Test was set up on 19 March 2003.

Table 7. Jack Slough artificial substrate BMI metric data. Two replicate baskets were collected at each sampling event, and overall numbers of taxa or percentages of community composition were determined (shaded columns) from the replicates.

	12/13/02			1/8/03			2/11/03			2/20/03			3/12/03		
	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall
Taxonomic Richness	15	10	16	7	10	11	9	10	13	9	6	10	6	6	7
EPT Taxa	1	0	1	0	1	1	0	1	1	0	0	0	0	0	0
ETO Taxa	2	0	2	1	2	2	1	2	2	1	1	1	1	0	1
Ephemeroptera Taxa	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Plecoptera Taxa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Trichoptera Taxa	0	0	0	0	1	1	0	1	1	0	0	0	0	0	0
Odonata Taxa	1	0	1	1	1	1	1	1	1	1	1	1	1	0	1
Coleoptera Taxa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
EPT Index	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Sensitive EPT Index (<4)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETO Index	1	0	0	1	2	2	0	1	1	1	0	1	0	0	0
Shannon Diversity	1.41	1.25	1.37	1.24	1.32	1.33	0.82	1.19	1.05	1.24	1.13	1.26	0.92	0.80	0.88
Percent Odonata	0	0	0	1	1	1	0	0	0	1	0	1	0	0	0
Percent Amphipoda	4	4	4	21	32	27	9	22	15	38	15	27	58	72	65
Percent Hydra	22	46	34	50	25	37	79	59	69	42	39	41	7	5	6
Tolerance Value	6.9	6.4	6.7	5.7	5.9	5.8	5.3	5.2	5.2	5.2	6.2	5.7	5.5	4.9	5.2
Percent Intolerant Organisms	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percent Tolerant Organisms	61	46	54	29	41	35	11	14	13	18	45	32	35	21	28
Percent Hydropsychidae	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Percent Baetidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percent Dominant Taxon	56	46	49	50	38	37	79	59	69	42	43	41	58	72	65
Percent Insects	10	3	7	2	4	3	2	4	3	2	0	1	1	2	1
Percent Coleoptera	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percent Chironomidae	9	3	6	1	1	1	1	3	2	1	0	1	1	2	1
Percent Oligochaeta	57	42	50	27	38	33	8	13	10	15	43	29	34	21	28

(continued)

Table 7. (continued).

	Rep 1	Rep 2	12/13/02 Final	Rep 1	Rep 2	1/8/03 Final	Rep 1	Rep 2	2/11/03 Final	Rep 1	Rep 2	2/20/03 Final	Rep 1	Rep 2	3/12/03 Final
		12/13/02			**1/8/03**			**2/11/03**			**2/20/03**			**3/12/03**	
	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall
Estimated Site Abundance	4384	2572	3478	2704	3256	2980	3256	3544	3400	3648	4304	3976	3400	3704	3552
Percent Collectors	68	50	59	49	71	60	17	37	27	53	58	56	93	94	93
Percent Filterers	8	3	5	0	1	1	2	3	2	1	0	0	0	1	1
Percent Grazers	1	2	1	1	1	1	2	0	1	3	2	2	0	0	0
Percent Predators	23	46	35	51	27	39	79	60	70	43	40	42	7	5	6
Percent Shredders	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percent Multivoltine	99	100	99	99	98	985	99	99	99.5	99	99	99	99	100	99

Table 8. Selected metric scores of benthic macroinvertebrate colonization trends on artificial substrates during the 2002/2003 storm water season monitoring of three agriculture-dominated waterways in the lower Sacramento River Watershed.

	Dec 13	Jan 8	Feb 11	Feb 20	Mar 12
Taxa Richness					
Jack Slough	16	11	13	10	7
Main Canal	19	18	n/s	15	19
Wadsworth Canal	19	26	n/s	25	20
EPT Taxa					
Jack Slough	1	1	1	0	0
Main Canal	3	3	n/s	1	3
Wadsworth Canal	4	3	n/s	5	3
ETO Taxa					
Jack Slough	0	2	1	1	0
Main Canal	4	4	n/s	2	4
Wadsworth Canal	6	7	n/s	6	5
Percent Amphipoda					
Jack Slough	4	27	15	27	65
Main Canal	13	4	n/s	3	3
Wadsworth Canal	41	4	n/s	2	3
Percent Hydra					
Jack Slough	34	37	69	41	6
Main Canal	0	1	n/s	53	20
Wadsworth Canal	0	1	n/s	2	1
Estimated Site Abundance					
Jack Slough	3478	2980	3400	3976	3552
Main Canal	157	1300	n/s	1500	1080
Wadsworth Canal	870	610	n/s	1095	1040
Percent Oligochaeta					
Jack Slough	50	33	10	29	28
Main Canal	67	41	n/s	20	34
Wadsworth Canal	5	30	n/s	18	16

n/s = not sampled.

Table 9. Environmental variables significantly correlated with artificial substrate BMI metrics in the Jack Slough. Unshaded values are positive and shaded values negative correlations.

Environmental Variable	BMI Metric	Correlation [r]	Sample Size	*P*
Chlorophyll	% Collectors	-0.7789	10	0.0079
Chlorophyll	% Predators	0.7832	10	0.0074
Chlorophyll	Hydra Abundance	0.9094	10	0.0003
Depth	Amphipod Abundance	-0.7376	10	0.0149
Depth	ShanDiv	0.7051	10	0.0228
DO	TaxRich	-0.8009	10	0.0054
DO	TolValue	-0.7588	10	0.0109
DO	% Filterers	-0.7349	10	0.0155
DO	% Chironomidae	-0.7183	10	0.0193
DO	% Insects	-0.6904	10	0.0271
DO	ShanDiv	-0.6478	10	0.0428
DO	Amphipod Abundance	0.8366	10	0.0026
pH	TaxRich	-0.8146	10	0.0041
pH	TolValue	-0.7427	10	0.0139
pH	% Filterers	-0.7166	10	0.0197
pH	% Insects	-0.6964	10	0.0253
pH	% Chironomidae	-0.6888	10	0.0276
pH	ShanDiv	-0.6744	10	0.0324
pH	Amphipod Abundance	0.8728	10	0.0010
SpC	% Predators	-0.7510	10	0.0123
SpC	Hydra Abundance	-0.7222	10	0.0183
SpC	Amphipod Abundance	0.7029	10	0.0234
SpC	Abund No Hydra	0.7083	10	0.0219
SpC	% Collectors	0.7524	10	0.0120
Turbidity	% Collectors	-0.6789	10	0.0309
Turbidity	% Predators	0.7061	10	0.0225
Turbidity	Hydra Abundance	0.8230	10	0.0035

Table 10. Main Canal artificial substrate BMI metric data. Two replicate baskets were collected at each sampling event, and overall numbers of taxa or percentages of community composition were determined (shaded columns) from the replicates.

	12/13/02			1/8/03			2/11/03			2/20/03		
	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall
Taxonomic Richness	16	12	19	14	15	18	14	13	15	18	14	19
EPT Taxa	2	1	3	2	2	3	1	0	1	2	2	3
ETO Taxa	3	2	4	3	3	4	2	1	2	3	3	4
Ephemeroptera Taxa	1	1	2	1	2	2	1	0	1	1	2	2
Plecoptera Taxa	0	0	0	0	0	0	0	0	0	0	0	0
Trichoptera Taxa	1	0	1	1	0	1	0	0	0	1	0	1
Odonata Taxa	1	1	1	1	1	1	1	1	1	1	1	1
Coleoptera Taxa	0	0	0	0	0	0	0	0	0	0	0	0
EPT Index	0	1	1	0	2	1	1	0	0	2	2	2
Sensitive EPT Index (<4)	1	0	1	0	0	0	0	0	0	0	0	0
ETO Index	0	0	0	0	0	0	0	0	0	0	0	0
Shannon Diversity	1.00	1.95	1.45	1.75	1.66	1.76	1.50	1.47	1.57	2.15	1.99	2.12
Percent Odonata	2.55	2.53	2.55	6.23	0.69	3.47	3.91	5.00	4.45	2.33	4.67	3.50
Percent Amphipoda	6	35	13	4	3	4	5	0	3	4	1	3
Percent Hydra	0	0	0	0	3	1	62	43	53	24	16	20
Tolerance Value	7.9	7.4	7.8	7.1	7.0	7.1	5.9	6.6	6.3	6.7	7.4	7.0
Percent Intolerant Organisms	1	0	1	0	0	0	0	0	0	0	0	0
Percent Tolerant Organisms	94	70	88	55	53	54	24	51	38	42	65	53
Percent Hydropsychidae	0	0	0	0	0	0	0	0	0	0	0	0
Percent Baetidae	0	0	0	0	0	0	0	0	0	0	0	0
Percent Dominant Taxon	79	35	64	38	44	41	62	43	53	24	35	28
Percent Insects	8	33	14	50	44	47	17	10	13	36	23	30
Percent Coleoptera	0	0	0	0	0	0	0	0	0	0	0	0
Percent Chironomidae	4	27	10	40	40	40	8	4	6	21	11	16
Percent Oligochaeta	81	23	67	38	44	41	7	33	20	25	43	34

(continued)

Table 10. (continued).

	12/13/02			1/8/03			2/11/03			2/20/03		
	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall
Estimated Site Abundance	235	79	157	1700	870	1300	1200	1800	1500	860	1300	1080
Percent Collectors	92	81	89	53	58	55	15	36	25	37	51	44
Percent Filterers	0	3	1	32	31	31	5	2	3	19	10	14
Percent Grazers	1	8	3	2	3	3	8	12	10	7	13	10
Percent Predators	7	9	7	13	9	11	73	51	62	37	27	32
Percent Shredders	0	0	0	0	0	0	0	0	0	0	0	0
Percent Multivoltine	97	96	97	93	99	96	96	95	95	97	95	96

Table 11. Environmental variables significantly correlated with artificial substrate BMI metrics in the Main Canal. Unshaded values are positive and shaded values negative correlations.

Environmental Variable	BMI Metric	Correlation [r]	Sample Size	*P*
Chlorophyll	EPT Index	-0.8054	8	0.0158
Chlorophyll	EPT Taxa	-0.7538	8	0.0308
Depth	% Grazers	-0.7575	8	0.0295
Depth	% Chironomidae	0.7361	8	0.0373
DO	% Collectors	-0.7601	8	0.0286
DO	Site Abundance	0.7644	8	0.0272
DO	Probezzia Abundance	0.7648	8	0.0271
pH	% Collectors	-0.7555	8	0.0302
pH	Probezzia Abundance	0.8148	8	0.0137
SpC	% Grazers	0.7070	8	0.0499
SpC	Probezzia Abundance	0.7365	8	0.0372
Velocity	% Grazers	-0.7237	8	0.0424

Table 12. Wadsworth Canal artificial substrate BMI metric data. Two replicate baskets were collected at each sampling event, and overall numbers of taxa or percentages of community composition were determined (shaded columns) from the replicates.

	12/13/02			1/8/03			2/11/03			2/20/03		
	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall
Taxonomic Richness	15	17	19	18	24	26	23	18	25	17	19	20
EPT Taxa	3	3	4	0	3	3	4	5	5	2	3	3
ETO Taxa	5	5	6	3	5	7	5	6	6	4	5	5
Ephemeroptera Taxa	2	2	3	0	3	3	2	2	2	2	2	2
Plecoptera Taxa	0	0	0	0	0	0	0	0	0	0	0	0
Trichoptera Taxa	1	1	1	0	0	0	2	3	3	0	1	1
Odonata Taxa	2	2	2	3	2	4	1	1	1	2	2	2
Coleoptera Taxa	0	0	0	0	0	0	0	0	0	0	0	0
EPT Index	1	1	1	0	1	1	5	4	5	1	4	2
Sensitive EPT Index (<4)	1	0	0	0	0	0	0	0	0	0	0	0
ETO Index	20	29	24	8	27	18	10	6	8	5	5	5
Shannon Diversity	1.71	1.79	1.78	2.08	2.46	2.43	2.20	1.68	1.99	2.04	2.16	2.18
Percent Odonata	19	28	23	8	26	17	4	2	3	4	2	3
Percent Amphipoda	46	36	41	4	4	4	3	1	2	4	1	3
Percent Hydra	0	0	0	0	2	1	2	2	2	1	1	1
Tolerance Value	5.7	6.4	6.0	7.4	7.6	7.5	6.5	6.1	6.3	6.5	6.5	6.5
Percent Intolerant Organisms	1	0	0	0	0	0	0	0	0	0	0	0
Percent Tolerant Organisms	27	41	34	67	59	63	32	19	26	28	32	30
Percent Hydropsychidae	0	0	0	0	0	0	1	1	1	0	0	0
Percent Baetidae	0	0	0	0	0	0	0	0	0	0	0	0
Percent Dominant Taxon	46	36	41	39	25	22	30	40	35	33	26	29
Percent Insects	45	50	47	34	58	46	68	81	75	75	68	72
Percent Coleoptera	0	0	0	0	0	0	0	0	0	0	0	0
Percent Chironomidae	25	21	23	24	24	24	56	74	65	70	61	65

(continued)

Table 12. (continued).

	12/13/02			1/8/03			2/11/03			2/20/03		
	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall	Basket 1	Basket 2	Overall
Percent Oligochaeta	3	8	5	43	18	30	21	14	18	9	22	16
Percent Trichoptera	1	0	0	0	0	0	3	2	3	0	0	0
Estimated Site Abundance	750	990	870	760	460	610	890	1300	1095	1200	880	1040
Hydra Abundance	0	0	0	0	11	7	15	26	20	12	6	9
Est. Abundance Insects	336	493	410	258	265	280	605	1053	816	898	602	745
Percent Collectors	72	64	68	58	41	49	53	53	53	49	62	55
Percent Filterers	3	2	2	14	8	11	34	41	37	36	29	32
Percent Grazers	3	2	3	14	12	13	3	2	2	7	2	4
Percent Predators	22	31	26	14	40	27	11	4	8	9	7	8
Percent Shredders	0	0	0	0	0	0	0	0	0	0	0	0
Percent Multivoltine	81	71	76	92	73	82	93	96	94	95	97	96

Table 13. Environmental variables significantly correlated with artificial substrate BMI metrics in Wadsworth Canal. Unshaded values are positive and shaded values negative correlations.

Environmental Variable	BMI Metric	Correlation [r]	Sample Size	*P*
Chlorophyll	% Oligochaetes	0.7248	8	0.0419
Chlorophyll	% Tolerant Organisms	0.7602	8	0.0286
Chlorophyll	% Grazers	0.8205	8	0.0126
Chlorophyll	Tolerance Value	0.8399	8	0.0091
Depth	% Tolerant Organisms	0.8977	8	0.0025
Depth	Tolerance Value	0.9227	8	0.0011
Depth	% Grazers	0.9481	8	0.0003
pH	% Chironomidae	0.8057	8	0.0158
SpC	% Predators	-0.7647	8	0.0271
SpC	Insect Abundance	0.7484	8	0.0327
SpC	% Insects	0.8224	8	0.0122
SpC	% Filterers	0.8954	8	0.0026
SpC	% Chironomidae	0.9343	8	0.0007
Temp	% Tolerant	-0.8177	8	0.0132
Temp	% Grazers	-0.7139	8	0.0467
Temp	% Insects	0.7367	8	0.0371
Temp	Insect Abundance	0.751	8	0.0318
Temp	% Chironomidae	0.8111	8	0.0146
Turbidity	% Chironomidae	-0.8172	8	0.0133
Turbidity	Insect Abundance	-0.7508	8	0.0318
Turbidity	% Insects	-0.7403	8	0.0357
Turbidity	% Tolerant Organisms	0.8077	8	0.0153
Velocity	Hydra Abundance	0.7107	8	0.0482
Velocity	EPT Index	0.7335	8	0.0384
Velocity	Trichoptera Taxa	0.8447	8	0.0083
Velocity	% Hydropsychidae	0.9135	8	0.0015

(continued)

Table 13 (continued).

StudyDay	Amphipod Abundance	-0.7528	8	0.0311
StudyDay	% Predators	-0.7527	8	0.0312
StudyDay	Insect Abundance	0.7088	8	0.049
StudyDay	% Insects	0.8077	8	0.0153
StudyDay	% Chironomidae	0.9235	8	0.0011
StudyDay	% Filterers	0.9244	8	0.001

Table 14. Summary of median toxicity concentrations for diazinon and chlorpyrifos to cladocerans (EC_{50}) and benthic macroinvertebrates (LC_{50}), (Munn and Gillion, 2001).

		N	Min (ppb)	Median (ppb)	Max (ppb)
Diazinon	Cladocerans (EC_{50})	17	0.50	1.22	1.80
	BMI (LC_{50})	19	0.03	25	6,160
Chlorpyrifos	Cladocerans (EC_{50})	3	0.10	0.40	1.70
	BMI (LC_{50})	31	0.04	0.57	83

Table 15. Summary of LC_{50} median toxicity concentrations for diazinon and chlopyrifos to different species of amphipods (Munn and Gilliom, 2001).

		N	Min (ppb)	Median (ppb)	Max (ppb)
Diazinon	*Gammarus lacustris*	2	170.0	185.0	200.0
	Gammarus pseudlimneus	1		2.0	
	Hyalella azteca	1		6.5	
Chlorpyrifos	*Gammarus fasciatus*			0.3	
	Gammarus lacustris	3	0.1	0.1	0.1
	Gammarus pseudlimneus	1		0.2	
	Gammarus pulex	1		0.1	
	Hyalella azteca	2	0.0	0.1	0.1

Appendix A. Jack Slough artificial substrate BMI taxa list.

Phylum	Class	Order	Family	Final ID	TolVal	FFG	12/13A	12/13B	1/8A	1/8B	2/11A	2/11B	2/20A	2/20B	3/12A	3/12B
Annelida	Oligochaeta	Tubificida	Naididae	Naididae	8	c	169	122	72	114	24	38	45	128	103	63
Annelida	Oligochaeta	Tubificida	Tubificidae	Tubificidae	10	c	3	4	10	1						
Arthropoda	Insecta	Diptera	Ceratopogonidae	Bezzia/ Palpomyia	6	p	1									
Arthropoda	Insecta	Diptera	Ceratopogonidae	Probezzia	6	p				1						
Arthropoda	Insecta	Diptera	Chironomidae	Chironomini	6	c	1	1								
Arthropoda	Insecta	Diptera	Chironomidae	Orthocladiinae	5	c	2		2				2		1	1
Arthropoda	Insecta	Diptera	Chironomidae	Tanypodinae	6	p	1					2				
Arthropoda	Insecta	Diptera	Chironomidae	Tanytarsini	6	f	22	9		2	4	7	2		1	4
Arthropoda	Insecta	Diptera	Empididae	Chelifera	6	p				2			1			
Arthropoda	Insecta	Ephemeroptera	Caenidae	Caenis	7	c	1									
Arthropoda	Insecta	Odonata	Coenagrionidae	Coenagrionidae	9	p	1		3	4	1	1	2	1	1	
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Hydropsyche	4	f				2		1				
Arthropoda	Malacostraca	Amphipoda	Crangonyctidae	Crangonyx	4	c	13	11	62	97	26	66	113	46	174	217
Arthropoda	Malacostraca	Decapoda		Astacidea	8	c						2		1		
Arthropoda	Ostracoda	Ostracoda	Cyprididae	Cyprididae	8	c	8	8			2	1				
Coelenterata	Hydrozoa	Hydroida	Hydridae	Hydra	5	p	67	137	149	74	236	177	127	118	20	14
Mollusca	Bivalvia	Pelecypoda		Corbiculacea	9	f	1				1					
Mollusca	Gastropoda	Pulmonata	Ancylidae	Ferrissia	6	g		1			1		1			
Mollusca	Gastropoda	Pulmonata	Physidae	Physa/ Physella	8	g	2	4	2	3	5		7	6		
Nemertea	Enopla		Tertastemmatidae	Prostoma		c	8	3				5				
Platyhelminthes	Turbellaria	Tricladida	Planariidae	Planariidae	4	p										1

Appendix B. Main Canal artificial substrate BMI taxa list.

Phylum	Class	Order	Family	Final ID	TolVal	FFG	12/13A	12/13B	1/8A	1/8B	2/20A	2/20B	3/12A	3/12B
Annelida	Hirudinea	Rhyncobdellida	Glossiphoniidae	Helobdella	10	p	4			3	1			
Annelida	Hirudinea	Rhyncobdellida	Glossiphoniidae	Placobdella/Oligobdella	6	p			3					
Annelida	Oligochaeta	Tubificida	Naididae	Naididae	8	c	186	16	110	127	20	99	64	105
Annelida	Oligochaeta	Tubificida	Tubificidae	Tubificidae	10	c	5	2					12	25
Arthropoda	Insecta	Diptera	Ceratopogonidae	Probezzia	6	p		2	9	7	12	1	29	17
Arthropoda	Insecta	Diptera	Chironomidae	Chironomini	6	c	1	10	1	4	1	1	2	3
Arthropoda	Insecta	Diptera	Chironomidae	Orthocladiinae	5	c	4	6	15	18	4	2	3	1
Arthropoda	Insecta	Diptera	Chironomidae	Tanypodinae	6	p	5	3	8	5	6	4	3	
Arthropoda	Insecta	Diptera	Chironomidae	Tanytarsini	6	f		2	92	87	15	6	56	29
Arthropoda	Insecta	Ephemeroptera	Baetidae	Fallceon quilleri	4	c			1	3				
Arthropoda	Insecta	Ephemeroptera	Caenidae	Caenis	7	c	1			2	2		7	5
Arthropoda	Insecta	Ephemeroptera	Ephemeridae	Hexagenia limbata californica	4	c		1						1
Arthropoda	Insecta	Odonata	Coenagrionidae	Coenagrionidae	9	p	6	2	18	2	12	15	7	14
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Hydroptila	6	g			1					
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Oxyethira	3	c	2						1	
Arthropoda	Malacostraca	Amphipoda	Hyalellidae	Hyalella	8	c	13	28	12	9	15	1	12	4
Arthropoda	Malacostraca	Decapoda		Astacidea	8	c	1		9			1	2	
Arthropoda	Ostracoda	Ostracoda	Cyprididae	Cyprididae	8	c	2	1	5	3	3	3	8	8
Coelenterata	Hydrozoa	Hydroida	Hydridae	Hydra	5	p				8	190	129	72	48
Mollusca	Bivalvia	Pelecypoda		Corbiculacea	9	f				1			1	
Mollusca	Gastropoda	Pulmonata	Ancylidae	Ferrissia	6	g	1						1	
Mollusca	Gastropoda	Pulmonata	Physidae	Physa/ Physella	8	g	2	6	5	9	24	35	19	39
Nemertea	Enopla		Tertastemmatidae	Prostoma	8	c	1							
Platyhelminthes	Turbellaria	Tricladida	Planariidae	Planariidae	4	p	1				2	3	1	1

Appendix C. Wadsworth Canal artificial substrate BMI taxa list.

Phylum	Class	Order	Family	Final ID	TolVal	FFG	12/13A	12/13B	1/8A	1/8B	2/20A	2/20B	3/12A	3/12B
Annelida	Hirudinea	Pharyngobdellida	Erpobdellidae	Erpobdellidae	8	p				1				
Annelida	Hirudinea	Rhyncobdellida	Glossiphoniidae	Helobdella	10	p					2			
Annelida	Oligochaeta	Tubificida	Enchytraeidae	Enchytraeidae	8	c				2				
Annelida	Oligochaeta	Tubificida	Naididae	Naididae	8	c	9	24	111	16	53	35	6	50
Annelida	Oligochaeta	Tubificida	Tubificidae	Tubificidae	10	c			12	35	10	8	21	14
Arthropoda	Insecta	Diptera	Ceratopogonidae	Probezzia	6	p			7	17	5			6
Arthropoda	Insecta	Diptera	Chironomidae	Chironomini	6	c	51	45	23	36	11	5	28	40
Arthropoda	Insecta	Diptera	Chironomidae	Orthocladiinae	5	c	9	3	1	2	65	96	69	55
Arthropoda	Insecta	Diptera	Chironomidae	Tanypodinae	6	p	9	6	7	13	2	1	9	6
Arthropoda	Insecta	Diptera	Chironomidae	Tanytarsini	6	f	9	5	37	20	90	120	96	73
Arthropoda	Insecta	Diptera	Empididae	Chelifera	6	p					1			
Arthropoda	Insecta	Diptera	Simuliidae	Simulium	6	cf					1	4		
Arthropoda	Insecta	Ephemeroptera	Baetidae	Fallceon quilleri	4	c	1							
Arthropoda	Insecta	Ephemeroptera	Caenidae	Caenis	7	c				1	2	1	1	1
Arthropoda	Insecta	Ephemeroptera	Ephemeridae	Hexagenia limbata californica	4	c		1		1				
Arthropoda	Insecta	Ephemeroptera	Leptohyphidae	Tricorythodes	5	c	1	1		1	6	3	1	8
Arthropoda	Insecta	Odonata	Coenagrionidae	Argia	7	p	1	2		5			1	2
Arthropoda	Insecta	Odonata	Coenagrionidae	Coenagrionidae	9	p	58	79	19	72	13	6	12	3
Arthropoda	Insecta	Odonata	Gomphidae	Gomphidae	1	p			1					
Arthropoda	Insecta	Odonata	Libellulidae	Pachydiplax longipennis	9	p			2					
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Hydropsyche	4	f					3	3		1
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Hydroptila	6	g					5	3		
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Oxyethira	3	c	2	1				1		
Arthropoda	Malacostraca	Amphipoda	Crangonyctidae	Crangonyx	4	c	145	102	10	12	1			
Arthropoda	Malacostraca	Amphipoda	Hyalellidae	Hyalella	8	c		2	2	1	7	4	13	2
Arthropoda	Malacostraca	Decapoda		Astacidea	8	c	6	2	2	1		1		3

Appendix C. (continued).

Phylum	Class	Order	Family	Final ID	TolVal	FFG	12/13A	12/13B	1/8A	1/8B	2/20A	2/20B	3/12A	3/12B
Arthropoda	Ostracoda	Ostracoda	Cyprididae	Cyprididae	8	c	4	3	5	11	3	1	3	4
Coelenterata	Hydrozoa	Hydroida	Hydridae	Hydra	5	p				7	5	6	3	2
Mollusca	Bivalvia	Pelecypoda		Corbiculacea	9	f		1	3	2	8		7	8
Mollusca	Gastropoda	Pulmonata	Ancylidae	Ferrissia	6	g			3	2	2		1	
Mollusca	Gastropoda	Pulmonata	Lymnaeidae	Fossaria	6	g				2				
Mollusca	Gastropoda	Pulmonata	Physidae	Physa/ Physella	8	g	9	7	37	31	1	2	18	6
Nematoda				Nematoda	5	p		3						
Platyhelminthes	Turbellaria	Tricladida	Planariidae	Planariidae	4	p	1		4	1	4		1	1

Appendix D. Environmental variables measured during time periods (11/13/02-4/16/03) corresponding with collection of artificial substrate baskets.

Jack Slough	11/13-12/6			12/13-1/8			1/17-2/11			2/15-3/12			3/21-4/16		
	Range	Mean	N	Range	Mean	N	Range	Mean	N	Range	Mean	N	Range	Mean	N
Chlorphyll (ug/L)	4.7-5.8	5.1	4	5.0-7.9	6.3	5	9.4-14.5	11.3	3	1.9-11.5	6.2	5	(-1.7)-4.5	1.5	3
Turbidity (NTU)	19.4-34.5	26	4	49.7-166	96	5	52-391	198	4	15-154	75.6	5	13.1-37	26.7	3
Temperature C	10-15.8	12.3	4	6.9-10.6	8.8	5	7.6-12.2	10.5	4	10.2-16.7	12.8	5	14.2-23	17.1	4
DO (mg/L)	6.7-7.9	7.4	3	8.3-9.6	9.4	5	7.2-10.8	9.3	4	8.2-12.1	9.9	5	4.7-8.3	6.7	4
pH	7.1-7.3	7.2	4	7.4-7.69	7.5	5	7.2-7.8	7.5	4	7.5-8.2	7.7	5	7.7-8.4	8	4
SpC (us/cm)	175-199	188	4	104-149	124	5	110-153	137	4	162-324	226	5	211-538	325	4
Velocity (f/s)	0.3-0.4	0.35	4	0.1-2.2	0.88	5	0.1-1.1	0.4	5	0.3-1.4	0.9	5	0.3-1.3	0.8	4
Depth (cm)	80.5-153	105	4	80.5-170	114	5	48.5-112.5	82.3	5	29-57	48.7	5	41.5-61	52.75	4

Wadsworth Canal	11/13-12/6			12/13-1/8			1/17-2/11			2/15-3/12			3/21-4/16		
	Range	Mean	N	Range	Mean	N	Range	Mean	N	Range	Mean	N	Range	Mean	N
Chlorphyll (ug/L)	0.7-1.8	1.3	4	2.5-6.3	3.9	5	1.2-4.5	2.8	2	1.5-2.8	2.2	5	1.1-1.6	1.3	3
Turbidity (NTU)	7.7-24	12.5	4	15-36	25.7	5	12.6-14	13.1	3	1.4-11	6.1	5	7.0-15.0	10.4	3
Temperature C	11.5-14.5	12.9	4	8.4-11.9	10.7	5	12.3-14.3	13.6	3	14.1-18.2	15.2	5	15.2-22.7	18.6	4
DO (mg/L)	9.9-10.2	10.11	3	9-9.5	9.3	5	6.4-11.1	8.7	3	9.1-15.4	12.1	5	5.1-8.8	7.1	4
pH	7.7-7.8	7.7	4	7.6-7.7	7.7	5	7.6-8.1	7.8	3	7.9-8.3	8.1	5	7.8-8.3	8.1	4
SpC (us/cm)	198-274	243	4	241-356	283	5	324-549	459	3	539-613	584.5	5	165-622	505.6	4
Velocity (f/s)	0.1-0.4	0.2	4	0.1-0.5	0.2	5	0.1-1.2	0.5	3	0.1-1.3	0.6	5	0.1-0.1	0.1	3
Depth (cm)	58-80.5	69.4	4	57.5-124.5	112.1	4	34-124	86.5	3	48.5-613	66	5	52.5-74.5	64.8	4

Appendix D. (continued).

Main Canal	11/13-12/6			12/13-1/8			1/17-2/11			2/15-3/12			3/21-4/16		
	Range	Mean	N	Range	Mean	N	Range	Mean	N	Range	Mean	N	Range	Mean	N
Chlorphyll (ug/L)	2.8-3.2	3	4	2.5-6.9	4.4	5	2.2-15.7	8.9	2	0.8-4.1	2.5	5	1.1-2.1	1.6	3
Turbidity (NTU)	6.8-14.8	10.3	4	18.7-51.2	25.4	5	10.1-16	13.1	3	1.1-16.5	7.2	5	13-Aug	10	3
Temperature C	9.9-14.2	12.2	4	8-11.5	10.2	5	10-13.8	12.4	3	13.4-17.1	13.8	5	13.8-17.6	15.8	3
DO (mg/L)	2.9-3.7	3.3	3	8.2-10.2	8.9	5	6.7-11.2	8.6	3	7-11.1	9.2	5	6-7.4	6.9	3
pH	7.3-7.4	7.3	4	7.5-7.6	7.5	5	7.4-7.9	7.6	3	7.7-7.9	7.8	5	7.8-8.1	7.9	3
SpC (us/cm)	244-329	289	4	223-268	265.8	5	273-455	371	3	446-556	507.6	5	559-578	568.6	3
Velocity (f/s)	100-104.5	103.5	4	85-200	132	5	83-119	102.5	4	72-87	79.8	5	n/s	n/s	n/s
Depth (cm)	0.3-0.8	0.5	4	0.6-1.6	1.1	5	0.1-0.6	0.4	3	0.1-0.3	0.1	5	n/s	n/s	n/s

Sutter Bypass West	11/13-12/6			12/13-1/8			1/17-2/11			2/15-3/12			3/21-4/16		
	Range	Mean	N	Range	Mean	N	Range	Mean	N	Range	Mean	N	Range	Mean	N
Chlorphyll (ug/L)	3.2-4	3.5	4	2.9-13.4	6.5	5	1.8-3.5	2.7	2	2.7-7.3	5	4	4.6-5.9	5.1	3
Turbidity (NTU)	14.5-23	16.9	4	17.8-559	166.1	5	15.9-24.6	19.2	3	22-49	32.9	4	30.5-44	35.5	3
Temperature C	10.2-13.9	11.9	4	7.9-10.8	9.1	5	8.8-10.5	9.9	3	11.1	13.6	4	15.6-18	16.8	3
DO (mg/L)	7.8-8.6	8.2	3	8.2-10.5	9.3	5	8.7-10.9	10.1	3	9.8-12.2	10.8	4	7-8.2	7.7	3
pH	7.5-7.7	7.6	4	7.4-7.7	7.6	5	6.1-8.1	7.3	3	7.8-8	7.9	4	7.7-8.1	7.9	3
SpC (us/cm)	319-370	356	4	94-372	187.6	5	134-257	176.3	3	284-366	324	4	198-276	249	3
Velocity (f/s)	0.1-0.1	0.1	4	0.1	0.1	1	0.1-0.1	0.1	2	0.1	0.1	1	0.1	0.1	1
Depth (cm)	0-51.5	33.9	4	68.5	68.5	1	n/s	n/s	n/s	65	65	1	n/s	n/s	n/s

Appendix E. Common names for BMI taxa identified in artificial substrate study.

Order	Family	Final ID	Common Name
Pharyngobdellida	Erpobdellidae	Erpobdellidae	Leeches
Rhyncobdellida	Glossiphoniidae	Helobdella	Leeches
Rhyncobdellida	Glossiphoniidae	Placobdella/Oligobdella	Leeches
Tubificida	Enchytraeidae	Enchytraeidae	Segmented worms
Tubificida	Naididae	Naididae	Segmented worms
Tubificida	Tubificidae	Tubificidae	Segmented worms
Diptera	Ceratopogonidae	Bezzia/ Palpomyia	Midges
Diptera	Ceratopogonidae	Probezzia	Midges
Diptera	Chironomidae	Chironomini	Midges
Diptera	Chironomidae	Orthocladiinae	Midges
Diptera	Chironomidae	Tanypodinae	Midges
Diptera	Chironomidae	Tanytarsini	Midges
Diptera	Empididae	Chelifera	Midges
Diptera	Simuliidae	Simulium	Midges
Ephemeroptera	Baetidae	Fallceon quilleri	Mayflies
Ephemeroptera	Caenidae	Caenis	Mayflies
Ephemeroptera	Ephemeridae	Hexagenia limbata californica	Mayflies
Ephemeroptera	Leptohyphidae	Tricorythodes	Mayflies
Odonata	Coenagrionidae	Argia	Damselflies
Odonata	Coenagrionidae	Coenagrionidae	Damselflies
Odonata	Gomphidae	Gomphidae	Dragonflies
Odonata	Libellulidae	Pachydiplax longipennis	Dragonflies
Trichoptera	Hydropsychidae	Hydropsyche	Caddisflies
Trichoptera	Hydroptilidae	Hydroptila	Caddisflies
Trichoptera	Hydroptilidae	Oxyethira	Caddisflies
Amphipoda	Crangonyctidae	Crangonyx	Amphipods
Amphipoda	Hyalellidae	Hyalella	Amphipods
Decapoda		Astacidea	Crayfish
Ostracoda	Cyprididae	Cyprididae	Seed shrimp
Hydroida	Hydridae	Hydra	Proboscis worms
	Tertastemmatidae	Prostoma	Proboscis worms
Pelecypoda		Corbiculacea	Bivalves
Pulmonata	Ancylidae	Ferrissia	Gastropods
Pulmonata	Lymnaeidae	Fossaria	Gastropods
Pulmonata	Physidae	Physa/ Physella	Nematodes
		Nematoda	Nematodes
Tricladida	Planariidae	Planariidae	Flat worms