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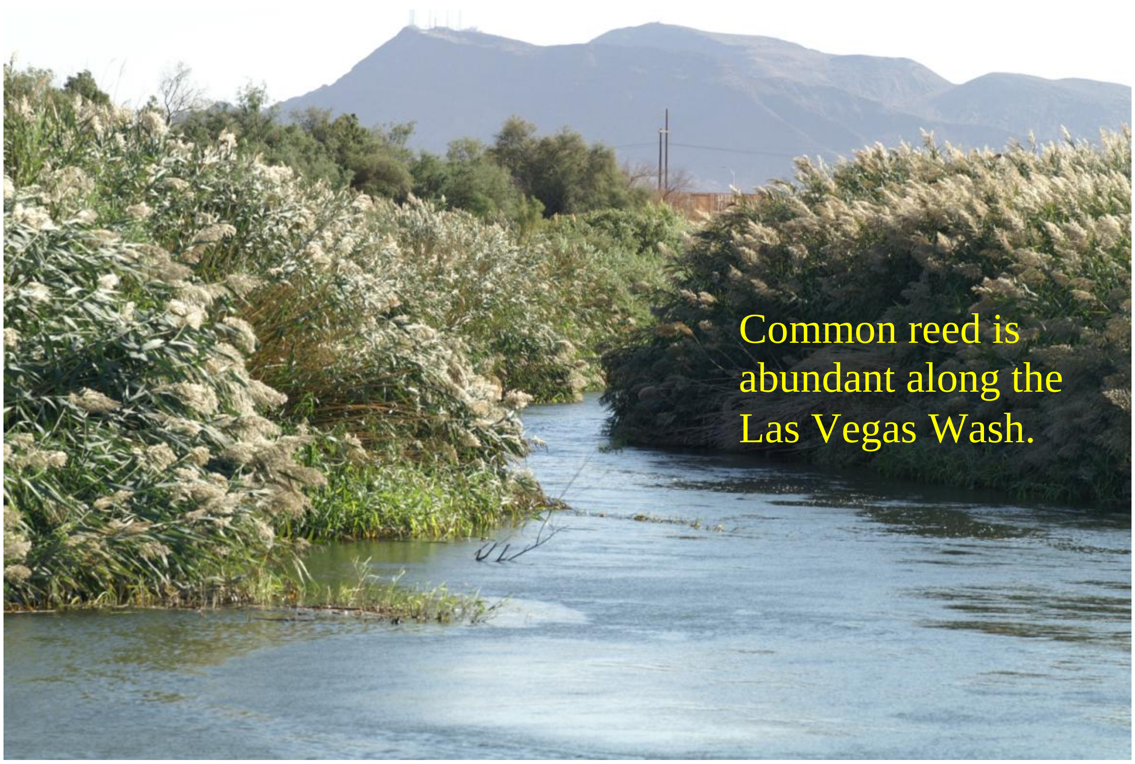
Stream Macroinvertebrate Assemblages Associated with the Las Vegas Wash Watershed 2000-2009



U.S. Department of the Interior
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Stream Macroinvertebrate Assemblages Associated with the Las Vegas Wash Watershed 2000-2009

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Abstract

Effects of stream erosion control structures on aquatic macroinvertebrates were studied in the urban Las Vegas Wash (Wash) drainage in Nevada from 2000-2009. Natural flow in this drainage is augmented by wastewater treatment plant inputs. Sampled areas included mainstem sites with and without erosion control structures, sites at wastewater treatment plant outfalls, an upstream reference site above the influence of treatment plant inputs, and tributary sites.

Direct ordination suggested hydrology and channel characteristics (current velocity, stream depth, and width), and water quality (conductivity) were primary factors in organizing macroinvertebrate communities and that some of these variables were altered by structures. Wastewater treatment plant inputs change hydrology (increased flows and erosion), water chemistry and alter water temperature. Data suggest that an increase in water temperatures may be facilitating invasion of an exotic tropical snail. Because the snail serves as an intermediate host for parasitic trematodes, its presence may impact fishes, amphibians, or other vertebrate species. Conductivity is decreased below wastewater treatment plants because municipal drinking water (and the resulting processed wastewater) is obtained from a lower conductivity source, Lake Mead.

Highest invertebrate taxa richness was found in Wash tributaries and was similar to samples collected at Wash sites with structures, and significantly higher than at mainstem sites below wastewater inputs that lacked erosion control structures. Taxa richness in the Wash was also impacted by flood flows as shown by the negative association with flow magnitude in the month preceding sample collection. Invertebrate assemblages differed between types of sites, with midges and damselflies important at tributary sites while *Fallceon* mayflies and the caddisfly *Smicridea* were common at erosion control structures. Ordination indicated that distinctive communities with unique taxa developed at sites with erosion control structures and correlation analysis showed that taxa richness increased over time at these sites. Structures placed in the Wash appeared important in retaining organic matter and among mainstem Wash sites, coarse particulate organic matter (CPOM) was highest, but variable, at structures and at wetlands formed upstream of structures.

It appears that the presence of erosion control structures, coupled with warm effluent, high baseflows, and altered water quality has resulted in development of a macroinvertebrate community that is locally unique to the Las Vegas Wash drainage and one that is not trending towards those found at reference or tributary sites. These data suggest that ecological communities associated with erosion control structures used for river restoration do not, in this case, lie on a continuum between disturbed and reference sites. Data also suggest that prediction of community responses (i.e., goal setting) to these structures would have required insight beyond the simple use of reference site material.

Important conclusions of this study are: wastewater discharges affect baseflows, water temperatures, and conductivity of the Wash; aquatic invertebrate taxa richness is

decreased at unimproved mainstem Wash sites below the influence of wastewater inputs; and erosion control structures modify some impacts to the Wash and are developing unique macroinvertebrate communities, which include exotic invasive species. It may be desirable to manage water temperatures in the Wash; in any case, aquatic invertebrate monitoring should be used to track successional patterns and evaluate responses to operational changes as part of an adaptive management scheme to determine appropriate ecological alterations in the Wash.

Introduction

Urbanization impacts to stream invertebrate communities result from multiple factors. Aquatic invertebrate assemblages in urban settings are often modified because of changes in sediment regimes, higher nutrient loads, alterations in trophic relationships, and presence of toxic compounds (e.g., Jones and Clark, 1987). Increased imperviousness of urban watersheds, caused by replacement of runoff-absorbing natural areas with rooftops and road surfaces (Klein, 1979), results in increased stream discharge, which can lead to changes in stream channel morphology. In arid environments, desert soil surfaces surrounding waterways may be naturally hydrophobic to some degree and other hydrologic metrics may be more important than impervious area in the linkage between biology and urbanization (Booth et al., 2004).

The numerous possible contributors to the stream urbanization response illuminates the need for study of these non-pristine systems. Efforts to conserve and restore stream biota in urbanized watersheds require quantitative models that describe and identify the relationship between environmental variables and stream communities. In urban areas this entails understanding stressors that connect human actions to changes in biota (e.g., Grimm et al., 2000). Few studies have considered specific mechanisms that cause urbanization effects (Paul and Meyer, 2001) and stream restoration effectiveness is rarely considered (Moerke and Lamberti, 2004). Where restoration has been evaluated, it has been noted that efforts to rehabilitate or restore urban streams fail because of narrowly prescribed solutions (Booth et al., 2004) that lack understanding of the breadth of stressor/biota interactions. Biotic response to restoration has often been less than expected. Urban stream restoration in Christchurch, New Zealand resulted in no improvement to stream ecosystems after riparian plantings and in-stream habitat modifications (Blakely and Harding, 2005). Larson et al. (2001) likewise found that large woody debris habitat features proved ineffective at improving biological conditions over a time scale of 2-10 years. Bond and Lake (2003) list a variety of factors that cause the expected link between habitat creation and biotic restoration to break down. Urban stream restorations tend to deal with many similar factors and Walsh et al. (2005) describe many of the characteristics general to urban streams as “urban stream syndrome”. Las Vegas Wash (Wash) in Nevada has many of the symptoms characterizing this syndrome.

In the 19th century, the Wash was ephemeral for most of its length, except for a small wetland area and several springs, which at that time were common in the Las Vegas Valley (Stave, 2001). Before 1928, approximately 0.03 m^3/s of discharge was the normal summer flow (Reclamation, 1982). In the 1930's and 1940's, wastewater treatment plants were built and began to discharge effluent into the Wash. During these early years groundwater was the basic water resource in the Las Vegas area. By the early 1940’s, however, water managers were expressing concerns about the limited supplies (SNWA, 2006). In 1942, water was imported from Lake Mead to process magnesium, and then discharged into the Wash (Reclamation, 1982). Increased inflows produced a wetland area that extended nearly the entire length of the Wash and provided important habitat for waterfowl and other wildlife.

Following the end of World War II, the Las Vegas metropolitan area continued to grow, with the Las Vegas Valley in Clark County containing the highest concentration of people in the state. In the 1950’s the Las Vegas Water District, which included the city of Las Vegas and most of Clark County, became increasingly dependent upon Colorado River water from Lake Mead. Currently approximately 85% of Clark County’s drinking water is delivered from Lake Mead at Saddle Island via water intakes, pumping plants, and pipelines. Because of the mechanisms of water use and flow in the Las Vegas Valley, increases in the human population result in increases in Wash flow volume as most of the flow in the Wash is treated wastewater (Sartoris et al., 2005). Thus, except for occasional flash floods during storm events, the lower 17 km of Las Vegas Wash, from the outfall of the City of Las Vegas Wastewater Treatment Plant (LWC10.6) to Las Vegas Bay on Lake Mead, can be characterized as an effluent-dominated stream. Average annual discharge in the Wash has generally increased over time and now approximates 8.0 m^3/s where it flows into Las Vegas Bay (Figure 1).

Buckingham and Whitney (2007) indicate that the hydrologic history of the Wash is dominated by three periods. Small additions of wastewater prior to 1975 resulted in an extensive marsh development with just a small amount of erosion. Between 1975 and 1989 hydrology changed as wastewater and storm runoff increased with the expansion of the city of Las Vegas. Down cutting, and channelization of the Wash lowered the water table adjacent to the Wash and drained much of what was once floodplain (Reclamation, 1982) resulting in decreased wetlands. Intensified erosion occurred between 1989 and 1999 as wastewater discharges continued to increase. By 1999, the Wash essentially flowed in a confined channel to Lake Mead. Urban development resulted in impervious surface area increases from 8,900 ha in 1960 to 75,600 ha by 1999, intensifying flash flood effects (Stave, 2001). The volume of sediment lost from the Wash is believed to be the largest ever documented for an urban expansion (6,588,000 m^3 of material eroded, ca. 1975 to 1999; Buckingham and Whitney, 2007). In response, the Las Vegas Wash Coordination Committee in 1999 completed the first of 22 grade control structures proposed to stabilize the channel at headcut locations in the Wash. By January 2008, 12 structures were in place, with construction started on several others. Three of these erosion control structures are located at sampling stations LW 6.05, LW5.5, LW3.85 (Figure 2). Erosion control structures placed along the Wash are permanent, low height dams designed to endure and help disipate energy from large storm events. These structures are engineered and capable of withstanding large flows. Building materials range from confined rock riprap to roller-compacted concrete secured to drilled concrete piles. It appears that sediment from erosion has been successfully reduced as evidenced by delisting in 2004 of the lower portion of the Wash from the state list for impairment to aquatic life caused by total suspended solids (USEPA, 2006). Flood events, however, still impact portions of the Wash, resulting in some erosion over the course of a year.

The purpose of this study was to monitor changes in macroinvertebrate assemblages associated with the construction of the erosion control features throughout the Wash in conjunction with resulting channel changes and development of some wetland and riparian areas. The area of study was limited in this case, extending only 17 km along the Wash (e.g., Figure 2). Therefore, the focus of this study was on environmental and chemical variables at the local scale rather than at the landscape scale. Macroinvertebrate community composition was assessed in the Las Vegas Wash and its tributaries to (1) identify environmental factors that may control biotic structure in this urban-impacted area, (2) describe the relationship between biota and erosion control structures and determine whether communities are similar to reference communities (i.e., communities unaffected by additions of water to the system), and (3) describe key ways in which restoration efforts can be aided by identifying variables important to aquatic invertebrates in the system.

Methods

Study area—Las Vegas Wash, a natural wash east of the city of Las Vegas, Nevada, carries storm water, groundwater drainage, and treated effluent from three wastewater treatment plants to Lake Mead. The Wash provides nearly the only surface water outlet for the entire 5,680 km^2 of Las Vegas Valley. A drainage area of 4,108 km^2 contributes directly to the Wash through surface flow which is channeled to Las Vegas Bay of Lake Mead, while drainage of the remaining 1,572 km^2 is presumably subsurface and may drain toward Las Vegas Wash.

Sampling was initiated at the same time that construction of erosion control structures was initiated so comparisons of the effects of structures on macroinvertebrate commmunities were limited to changes that occurred over time, contrasts with other sites in the Wash that lacked erosion control structures, and similarities to reference sites that were not influenced by wastewater treatment plants. Initially in 2000 only 7 mainstem sites (with and without control structures) along the Wash were sampled; however, starting in 2001, 20 sites were selected within the Wash watershed, including 9 Wash sites and 11 tributary and wastewater discharge sites (Figure 2). One new wastewater discharge site (LWC9.0_1) was added in 2003. The numbering system used in this study corresponds to that utilized by the Southern Nevada Water Authority with the site number related to the distance (in miles) upstream from Lake Mead and the letter “C” indicating an inflow at the confluence with the Wash. Two additional Wash sites were added in 2005 to sample wetlands forming above erosion control structures (LW6.10 and LW3.86), along with one additional tributary site (LWC5.5). The additional tributary site added in 2005 was gone by 2007, the flow having been diverted in a buried concrete conduit. Tributary sites were considered important in this study because they can potentially provide a source of reference communities that may not be present in the degraded main channel of Las Vegas Wash. The furthest upstream site on the mainstem

Wash (LW11.1) is above the influence of wastewater treatment plants and was also considered a reference site for the Wash. Environmental variable sampling began in 2001.

Chemical, physical, and biological methods—Yearly sampling with the full set of variables took place in April 2001 to 2002, March 2003 to 2005, March 2006, and April 2007 to 2009. Quarterly sampling at the original seven mainstem sites was initiated in December of 2004 to document biota that might be available seasonally. A 1-minute kick method with a D-frame net (700-800 micron mesh) was used for sampling benthic invertebrates along a ca. 10-meter reach at each sampling site. Samples were preserved in 70% propanol. In the laboratory, samples were washed in a 600-micron mesh sieve to remove alcohol, invertebrates were picked from the substrate with the aid of an illuminated 10X magnifier, and then the entire sample was enumerated and identified under a binocular dissecting scope.

Starting in 2003 biomass of coarse particulate organic matter (CPOM) and plant matter related to autotrophic production (periphyton) were obtained from the macroinvertebrate sample. These samples were dried at 60^{o}C for 48 hrs and weighed to the nearest hundredth of a gram.

Environmental variables measured for each site included physico-chemical parameters, water chemistry analyses, and measurements of habitat qualities. Dissolved oxygen (DO), conductivity, pH, temperature, and turbidity were measured with portable meters. Water samples for alkalinity were analyzed using titration methods, while hardness was determined by calculation from Ca and Mg concentrations or from titration. Water samples for analyses of major ions and nutrients (nitrogen and phosphorus compounds) were collected in high-density-polyethylene bottles and transported to the laboratory in an iced, insulated cooler. Water samples were analyzed by Reclamation's Lower Colorado Regional laboratory using standard methods (APHA, 1975; APHA 1998; USGS, 1979).

Size composition of the substrate was visually estimated at each site in the area where macroinvertebrates were collected. Categories were expressed as percent bedrock, boulders, cobble, coarse gravel, fine gravel, and sand/fines. Percentage categories were converted to a single substrate index (S.I.) value (e.g., Jowett and Richardson, 1990) using the formula S.I.=0.08* %bedrock + 0.07* %boulder + 0.06* %cobble +0.05* %gravel +0.04* %fine gravel + 0.03* %sand and fines. Wet width of the stream was measured with a measuring tape or a range finder. Depth was measured with a calibrated rod.

Water velocity at 10 cm above the substrate was measured at three discrete points in the channel cross-section within the invertebrate collection area. The average of these three measurements was used in analysis.

Habitat disturbance was estimated with Pfankuch's Index (Pfankuch, 1975). This subjective, composite index involves scoring 15 stream channel variables along the upper bank, lower bank, and stream bottom. High scores represent unstable channels at the reach scale. This index has been use to measure stream disturbance in other studies (Townsend et al., 1997). Information was also noted on impairment within the stream related to construction activities in the Wash. Imperviousness of the watershed was not measured because the relatively small geographic area in which the study took place would likely be uniformly impacted.

Data analysis—Multivariate analysis (CANOCO 4.5), invertebrate abundance and richness, and tolerance measures for pollution [Barbour et al., 1999; Aquatic Bioassessment Laboratory, 2003 (www.dfg.ca.gov/cabw/cabwhome.html)] and sediment (U.S. Forest Service, 1989) were used to relate invertebrate assemblages to pollutants. Tolerance values obtained from Barbour et al. (1999) were often mean values derived from regional tolerance values.

ANOVA followed by Tukey's test for comparisons were used to compare means of taxa richness, invertebrate abundance, and organic matter resources at different types of environments. Data were transformed, if needed to normalize distributions, using ln (X+1). Trends in the data set were examined by looking at correlations (Pearson) between annual sampling occasion and taxa richness. The assumption was that richness values would increase over time at sites modified by erosion control structures and therefore be positively correlated with sampling occasion. Theoretically, sites without these structures would remain unchanged (cutbanks, narrow, deep, with high stream velocities) and show no significant correlation with sampling occasion.

Stepwise multiple regression (SMR) (forward selection) was used to determine which candidate variables were important ($P \leq 0.05$) in determining taxa richness and invertebrate abundance at Wash sites. Data from March/April 2000 to 2009 were used in this analysis. Initial independent variables used for the model included: highest daily mean (Q_{max}) discharge (m^3/S) for the 1-month period prior to sampling (obtained from USGS water resources data), whether or not an erosion control structure was present at the sample location, the year the samples were collected, whether construction was noted in the immediate area, and sample location (site). Dummy variables were created for structure presence or absence and whether construction activity was observed at the site. Macroinvertebrate variables and Q_{max} were transformed using ln (X+1) to normalize distributions. Data were not necessarily available for all sites during all years because of changes in USGS monitoring that was sometimes interrupted or lost due to damage to flow gages. The irregularity of these data and absence of flow data collection from most sites precluded use in multivariate analysis.

Ordination techniques were used to examine patterns in the macroinvertebrate data, and to identify physical and chemical variables most closely associated with invertebrate distributions. Initial analyses of the macroinvertebrate data sets used detrended correspondence analysis (DCA), and revealed a data gradient length > 3, suggesting that unimodal models were appropriate for analysis. Therefore, canonical correspondence analysis (CCA) was used for direct gradient analyses. Faunal data were transformed (square root transformation) before analysis. Forward selection of environmental variables and Monte Carlo permutations were used to determine which and to what extent environmental variables exerted a significant ($P \leq 0.05$) effect on invertebrate distributions. If environmental variables were strongly correlated (Pearson correlation, $r \geq 0.6$), only a single variable was selected for use in CCA to avoid problems with multicollinearity. Environmental variables were normalized [(ln (X+1)) or arcsin squareroot transformation for percentage data] if the Shapiro-Wilks Test indicated non-normality. In the ordination diagram, taxa and sites are represented by points and the environmental variables by arrows. Arrows roughly orient in the direction of maximum variation of the given variable. Environmental variables were not measured in 2000, therefore only data from 2001 to 2009 were used in direct ordinations. CPOM and periphyton biomass variables were not collected until 2003 and were not a part of ordination analysis.

Results

Environmental variables—The substrate type was highly diverse, ranging from mud to cobble and bedrock (concrete lined) and S.I. values ranged from 3 to 8 (Table 1). Mainstem Wash and mainstem sites with structure had significantly higher values than tributary or wetland sites (Table 1). Highest mean values were at wastewater outfall sites because of the tendency for these sites to be concrete-lined. Velocity ranged from 0.00 to 1.31 m/S with highest mean velocities at mainstem and mainstem sites with structure (Table 1). Velocities at these two sites differed significantly from velocities at tributary and wetland sites. Stream width ranged from 1 m to 100 m and depths from 0.02 m to 1.2 m. Sites that were widest were those where erosion control structures had been placed. Sites that were deepest were those at mainstem sites without structures and wetland sites and these differed significantly from mainstem sites with structure, reference, and tributary sites (Table 1). Pfankuch′s Index varied from 38 to 139, with lower values associated with sites that were less prone to damage from floods. Nitrate and phosphate concentrations were highest at wastewater outfall sites and sites downstream of wastewater treatment facilities (Table 1). Other water quality parameters indicated differences associated with wastewater affects on the mainstem Wash sites (alkalinity, conductivity, DO, hardness, pH, and temperature) or observed increases in turbidity downstream (Table 1). Tributary sites were often similar to the reference site in water quality (Table 1). The Shapiro-Wilks Test indicated that transformation was necessary for most of the variables used in CCA analysis. Nitrate concentrations, Pfankuch's Index, pH, S.I., temperature, and velocity were not transformed for analyses.

Differences in temperature found in annual data (Table 1) were also reflected in data from quarterly seasonal collections along the Wash. Mean temperatue and coefficient of variation (CV) of temperature were calculated from these data for the upstream reference site and the other mainstem sites that were downstream of wastewater treatment plants. The CV provided a measure of heterogeneity or variability in temperature. Mean temperature at the reference site was $16.7^{o}C$ with a CV of 44.6 (n=20) while at the other downstream sites the mean temperature was 24.2 with a CV of 14.1 (n=118). These data support the annual data that shows differences in temperature regimes between mainstem Wash sites and the reference site.

Invertebrate food resources-CPOM/Periphyton—Tributary and wetland sites had significantly higher amounts of CPOM relative to mainstem and wastewater sites (Table 1). Mean CPOM values were higher at sites with erosion control structures, but high variance precluded finding significant differences between mainstem-structure sites and other mainstem sites. Cattail (*Typha* sp.) and common reed (exotic invasive haplotype of *Phragmites australis*) tended to colonize structures over time. During high flows, much of this material was scoured from structures. Wastewater sites had high mean periphyton biomass and this differed significantly from reference and wetland sites (Table 1).

Taxa richness/invertebrate abundance—Seventy invertebrate taxa were identified from all samples (Table 2). ANOVA indicated a significant difference in macroinvertebrate taxa richness and abundance among site types (Table 1). Invertebrate abundance and taxa richness were higher at tributary, reference, and sites associated with erosion control structures (Table 1). Abundance was also high at wastewater discharge sites, however, taxa richness was low (Table 1). Correlation analyses of taxa richness with sampling year for Wash sites with and without erosion control structures that were sampled over the 10 year period 2000 through 2009 are presented in Table 3 and Figure 3. Sites that lacked structures at sites below the influence of wastewater treatment plants did not have significant correlations with sampling year, while those collected at structures were significant or near significance (Table 3) suggesting an increase in taxa richness over time (Figure 3). Taxa richness at the reference site LW11.1 was also significantly correlated with sampling occasion (Table 3). The response at LW11.1 appeared to be related to large changes in the substrate that occurred after 2003. The channel at LW11.1 was made up of fine sediment and lined with dense cattails in 2000. During sampling in 2003, heavy equipment was operating in the channel removing sediment and emergent vegetation and exposing a buried concrete liner. On that occasion, sampling occurred just upstream of this operation. Beginning in 2004, sampling took place downstream of the original site just below the concrete-line section (and boulder material downstream of the concrete) that had been exposed in 2003. In 2005, a large flood occurred, resulting in plant material and trash being deposited up to 2 m above baseflow stage. It is believed that the correlation between taxa richness and sampling year at the reference site was a result of the changes that coincided with sampling from 2000 to 2005 and channel modifications that occurred post-2005. Early taxa richness values ranged from 1 to 6, while values post-2005 ranged from 6 to 17.

Invertebrate abundance was not predicted by any of the variables in the SMR model. However, the presence/absence of structures was an important predictor of taxa richness in the SMR model from Wash sites (Table 4). The flow variable Q_{max} was also important in predicting taxa richness as was the year. Other variables such as construction impacts and site were not significant in the model. Large flows negatively impacted taxa richness, while sampling year and presence of structures positively affected taxa richness (Table 4). Flow effects were often visually noticeable in the Wash. Damage and movement of boulder material above LW11.1 was noted in December 2004 and some of the concrete structure at LW0.55 was uncovered. In March of 2005, deposited debris lines up to 3 m above the baseflow water surface were present at some sites, and emergent vegetation had been scoured out at LW6.05, LW5.5, and LW3.85. Evidence at LWC6.1_1 indicated water had flowed out of the Wash and into this wastewater outfall. Downcutting in March 2005 occurred at LW8.85 and LW0.55 with a deepening of the channel of approximately 3 m at LW0.55, changing the habitat from lotic to lentic. Mean annual values (from all sites) for Q_{max} at mainstem sites varied between the years, with highest values in 2000 (33.7 m^3/S) and 2005 (36.0 m^3/S). Discharge was lower in 2001 (7.3 m^3/S), 2002 (6.1 m^3/S), 2006 (7.6 m^3/S), 2007 (7.1 m^3/S), and 2008 (8.6 m^3/S). Values were more moderate in 2003 (15.4 m^3/S) and 2004 (19.4 m^3/S).

Multivariate analysis—Results of CCA from the 2001 to 2009 studies (Figures 4 and 5) of the stream benthos had eigenvalues of 0.310 and 0.219 for the first two axes and explained 16.5% of the species data variation and 67.1% of the species-environment relation. Initial environmental variables used in the model included alkalinity, conductivity, depth, DO, hardness, NH_3, NO_3, Pfankuch's Index, PO_4, pH, S.I. (correlated with % sand), temperature, turbidity, velocity, and width. Variables found to be significant ($p<0.05$) in the model were alkalinity, conductivity, depth, DO, hardness, Pfankuch's Index, PO_4, pH, temperature, turbidity, velocity, and width.

Alkalinity, conductivity, depth, velocity, and width were correlated with the first axis, while DO, Pfankuch's Index, pH, temperature and turbidity were correlated with Axis 2 (Table 5). No variables had their highest correlation with the third or fourth axis and these explained only a small portion of species-environment relationships. A permutation test used to examine the relationship between species and environmental variables was significant for all axes ($P=0.0010$).

Site samples tended to cluster in four areas (Figure 4) of the ordination diagram. Wash mainstem sites without structures were mostly to the left on Axis 1; mainstem sites with erosion control structures were in the lower left portion of the diagram; effluent dominated wastewater outfalls towards the upper end of Axis 2; and most tributaries were in the right portion of the diagram. It appeared that the furthest upstream mainstem reference site (LW11.1) took an intermediate position between tributary sites and mainstem sites with structure (Figure 4). This also seemed to be the case with the wetlands sites that were forming above erosion control structures.

Depth, velocities, and width were relatively low at tributary sites and greater at mainstem sites (Figure 4, Table 1). Alkalinity and conductivity were higher at tributary sites and the upstream reference site (see Figure 4, Table 1) and this native water was diluted by high volumes of low conductivity wastewater downstream in the Wash (e.g., Table 1). Dissolved oxygen concentrations were lower at wastewater outfall sites receiving water from treatment plants (Figure 4, Table 1). It also appeared that wastewater treatment plant operations resulted in lower pH at wastewater outfalls.

Relationship between biota and site types—Characteristic taxa were found at specific site types (Figure 5). Distributional data indicated midges (chironominii and tanytarsini), Odonates (Libellulidae) and Coleoptera (*Enochrus*, *Berosus*, and *Agabetes*) were found at tributaries (to the right on Axis 1, Figure 5) and *Fallceon* at mainstem-structure sites (to the left on Axis 1, Figure 5). *Smicridea* was associated with mainstem-structure sites but was also present (but in lower numbers) at mainstem sites without structures. As erosion control structures developed over time it appeared that the abundance of larval *Smicridea* increased at structure sites but not at mainstem sites that lacked these structures (Figure 6). *Hyalella* was associated with increased depth and temperature along with lower DO and pH, characteristics found at wastewater outfall sites (Figure 5). Wetland sites contained low numbers of taxa such as *Callibaetis*, Corixidae, Ephydridae, Psychodidae, and Sciomyzidae that typified reference and tributary sites. The vast majority of macroinvertebrates collected from the Las Vegas Wash basin were tolerant of organic pollution and sediment (Table 2).

Exotic mollusks appear to be invading the system. The clam *Corbicula* was first noted in the Wash in 2003 and snails in the family Thiaridae (one specimen was identified as *Melanoides*) detected in 2007. In 2006, the tropical aquarium fish, shortfin molly (*Poecilia mexicana*) was first observed in the Wash and it may be that dumping aquariums into the Wash resulted in the introduction of both mollies and tropical thiarid snails. While evidence of *Corbicula's* impacts on native fauna is weak (e.g., Strayer, 1999), declines in some native mollusks have been associated with introduction of *Melanoid*es (U.S. Fish and Wildlife Service, 1998). Thiaridae have also been noted to carry a fish parasite, a heterophyid trematode that is causing problems with listed fish in Texas (Fuller and Brandt, 1997). Rader et al. (2003) reported that the invasive Thiaridae snail (native to tropical regions such as parts of Africa) *Melanoides tuberculata* is restricted to waters between 18 and 31^{o}C and the temperatures found in the Wash below wastewater inputs appear to be maintained within this range. The recent discovery of quagga mussels (*Dreissena rostriformis bugensis*) in Lake Mead may indicate that yet another mollusk may soon reside in the Wash. It is unknown what impact this may have on the Wash fauna, however, large negative impacts to native macroinvertebrates have been attributed to other *Dreissena* species (Lozano et al., 2001).

Other taxa seem to be appearing over time at erosion control structures, and in some cases, despite multiple sampling years, have only been found at structures in the Wash. In December of 2005, after 9 sampling occasions, the damselfly *Hetaerina* (family Calopterygidae) appeared for the first time. Even more recently a new caddisfly (*Culoptila,* family Glossosomatidae) was detected in March of 2008 after 18 previous sampling occasions. A pulmonate snail in the family Ancylidae (*Ferrissia*) was found in December of 2007 after 17 previous samplings. In all cases these were not single incidents but appeared to represent an initial absence and then discovery followed by additional detections. For the most part these taxa are exclusively found at erosion control structures in the Wash and not detected at tributary or reference sites. The detection of *Culoptila* may be related to increased water quality in the Wash. Some species of *Culoptila* in the western United States have been found to be highly sensitive to sediment (Blinn and Ruiter, 2006; also see Table 2) and their presence in the system at this time may be a biological sign that sediment from erosion is declining. Taxa such as *Culoptila* and the recently detected *Ferrissia* may also be favored by thermal regimes that are modified towards decreased variability and higher temperatures (e.g., Houghton and Stewart, 1998; Hadderingh et al., 1987); conditions similar to those found in the Wash below wastewater treatment plants.

The continued addition of structures in the Wash and the colonization, over time, of cattail and common reed, and their contribution to CPOM on developing substrates may have played a role in the ability of new taxa to colonize the area. The increase in suitable environmental conditions at the erosion control structures, where dense emergents and corresponding CPOM accumulates, may be the type of environment that favors some taxa such as *Hetaerina* (e.g., Westfall and May, 1996). Blinn and Ruiter (2006) have also reported increased caddisfly species richness at sites associated with native willow and cottonwood vs. those that contained saltcedar vegetation.

Discussion

Environmental Factors Associated with Macroinvertebrate Communities—Factors differentiating benthic invertebrate assemblages included hydrology/channel characteristics, catchment geology (salinity/conductivity), and water quality changes (temperature, pH, DO, phosphate) associated with wastewater treatment plants. Many of these environmental gradients were expressed in the CCA. Habitat simplification to a narrow, deep, high- velocity channel was especially evident in the upper portion of the Wash mainstem below treatment facilities. Within the Wash, taxa richness was higher at the reference site above the influence of wastewater impacts and in areas associated with erosion control structures where the channel was wider and shallower. Invertebrate abundance was significantly lower in the unimproved incised sections of the Wash compared with other types of habitats sampled. SMR indicated that the presence of erosion control structures increased macroinvertebrate richness. It appeared that the channel in the vicinity of some of these structures is becoming quite complex, with, for example, a braided or multiple channel appearance at LW5.5

Macroinvertebrate assemblages differed between tributaries, wastewater outfalls, the mainstem reference site, structure-associated communities, and mainstem sites without structures. Tributary communities were taxa-rich compared with other groups and tended to contain odonates and a variety of dipteran taxa. Multivariate analysis identified high conductivity as one of the variables that was associated with tributary communities. Dissolved salts and minerals in the water, as measured by conductivity, are likely influenced by catchment geology and urbanization. Prior to the introduction of increased volumes of Lake Mead water via wastewater discharge, these constituents may have been higher in downstream portions of the Wash. Decreased conductivity is one type of urban impact in this study and is related to increased flows of water transported from wastewater treatment plants. Conductivity is often related to chloride concentrations and chloride values ranged from 271 to 345 mg/L in the Wash and 445 to 1030 mg/L in tributaries (March 2004 data). These values exceed the final chronic value for chloride of 226.5 mg/L promulgated by the Environmental Protection Agency for water quality (USEPA, 1988). The relatively high conductivity associated with this catchment may place an upper limit on invertebrate biodiversity, while lower conductivities in the Wash below treatment plants may increase survivability for some taxa in the watershed.

The altered thermal regime in the Wash could also affect macroinvertebrate assemblages. Vannote and Sweeney (1980) have noted large changes in invertebrate communities exposed to thermal impacts. Increased temperatures often lead to changes in invertebrate densities and reduced size at maturity, results that may decrease the ability of particular species to persist in the environment (Hogg and Williams, 1996). Higher temperatures in the Wash may also permit invasion by exotic species restricted to warmer water temperatures. The Thiaridae snails which have been recently found in the Wash may be an example. Mitchell and Brandt (2005) showed that in colder regions these snails can only survive where springs or power plants moderate temperatures. Wastewater effluent appears to provide similar opportunities. As an intermediate host for parasitic trematodes this snail may impact species of concern including fishes and amphibians (Rader et al., 2003). Schueler (1987) found that water temperature increases in urban area streams are not only a function of warm water entering streams from wastewater treatment facilities, but could also occur from water being heated by impervious surfaces (e.g., van Buren et al., 2000), by solar radiation in unshaded conveyance channels, and from impoundments such as stormwater detention ponds. Increased water temperatures associated with wastewater discharges likely are detrimental to certain invertebrate taxa, resulting in competitive exclusion by more tolerant species (e.g., Cairns, Jr. 1972). Wang and Kanehl (2003) found that increased water temperature caused by urbanization was one of the most influential factors in structuring macroinvertebrate assemblages. Management efforts that restore a natural thermal regime may result in communities with greater similarity to reference/tributary sites.

Relationship between Biota and Erosion Control Structures—Below wastewater inputs, greatest taxa richness and abundance in the mainstem Wash was found at in-channel erosion control structures that resulted in a shallow and wide stream with relatively high velocities. Several of these structures appear to have high values of relative roughness, which may indicate a greater diversity of hydraulic conditions. In-stream structures that promote such variability will increase benthic diversity to some degree. These structures appeared to trap particulate organic matter that then serves as both food and additional habitat for invertebrates. In many cases, these stable structures also provided substrate for periphyton growth. Finally, it appeared that sand accumulations occurred within these structures, providing habitat for burrowing organisms (*Corbicula*) within a matrix of stable substrate. Stewart et al. (2003) and Livan et al. (2007) found a positive response for benthos from stone habitat structure placed in streams and suggested that increased organic matter and habitat diversity were responsible. Negishi and Richardson (2003) found that placement of boulders in a stream increased organic matter storage that was accompanied by a 280% increase in macroinvertebrate abundance but had little affect on taxa richness. Other studies in urban areas (Larson et al., 2001; Harrison et al., 2004) have found no change in biological condition after habitat addition, and suggested that watershed-scale factors controlled overall biotic diversity.

Harrison et al. (2004) point out that macroinvertebrates have complex life cycles where different life stages may use different parts of the aquatic and riparian environment. It may be that erosion control structures provide only a part of life history requirements for

Wash aquatic invertebrates. For example, the limited (but increasing) riparian environment along much of the Wash may not yet provide the resources needed by aerial adults of species with aquatic larvae. Populations often exhibit thresholds in response to overall habitat area. Below this level they may not exist, regardless of habitat quality (e.g., Miller and Hobbs, 2007). Altered riparian vegetation has been associated with reduced stream invertebrate diversity in other studies (Urban et al., 2006). Low amounts of riparian vegetation may also limit aquatic- terrestrial linkages important for transfer of instream biomass to terrestrial consumers (e.g., Paetzold et al., 2005). Recent data from sticky-traps placed in terrestrial environments along the Wash indicated that adult caddisflies (*Smicridea*) were captured at sites revegetated with native vegetation but not at sites dominated by invasive exotic vegetation (unpublished data). Remsburg and Turner (2009) found that riparian vegetation influenced larval odonate assemblages within the adjacent aquatic environment and suggest that larvae need the rigid vertical structure of tall riparian plants when emerging from the water as adult dragonflies. This may explain, in part, the pattern of increased larval *Smicridea* abundance that was observed only at sites with erosion control structures and corresponding riparian plantings. Erosion control structures were also important in development of habitat diversity, providing both lotic habitat and slow velocity environments similar to that of tributaries at wetlands forming above structures in the Wash. It appears that this environment is creating habitat for taxa otherwise not detected in the Wash.

Factors Important in Restoration from a Macroinvertebrate Perspective—Data from this study suggest that in-stream habitat at the Wash, in the absence of erosion control structures, is habitat limited for macroinvertebrates. In other studies of urban streams, physical habitat differences were not important in structuring the macroinvertebrate community because of poor water quality (Beavan et al., 2001), and streams receiving wastewater effluent often contain highly modified invertebrate assemblages (e.g., Kondratieff and Simmons, 1982). Although nutrients were elevated at some sites in this study, the strong relationship of river width, depth, and velocity with benthic communities suggests that hydrological and channel characteristics are among the main driving forces in structuring communities in the Wash. The different benthic community found at the upstream reference site also provides some evidence that increased temperatures, baseflows, and water quality (i.e., lower conductivity) provided by wastewater treatment plants at downstream sites plays some role in influencing macroinvertebrate assemblages. The low numbers of sensitive taxa suggest that large-scale processes are resulting in decreased diversity. It is unclear if organic compounds such as pesticides are impacting invertebrates. Bevans et al. (1998), however, detected a wide range of organic compounds in the Wash.

Wang et al. (2001) suggest that large-scale landscape features have a major impact on urban streams and can overwhelm local structures designed to improve habitat. Walsh et al. (2001) suggest that the most effective means of restoring degraded urban streams may be retrofitting stormwater drainage systems to decrease flood flows along with minimizing catchment imperviousness. In the present study, taxa richness significantly declined with increasing magnitude of recent discharges. Others also consider stormwater runoff and floods to have major impacts on urban systems (Walsh et al., 2005; Trush et al., 2000). It is also likely that the constant, high baseflow velocities in the Wash minimize habitat at unimproved portions of the Wash for stream invertebrates. Water velocities were identified as an important factor in distribution of invertebrates in the Wash and surrounding habitats. In the still-urbanizing watershed of the Wash, constant disturbance by floods and ever increasing baseline flows may limit biodiversity to tolerant taxa. Channel stability in urban environments may not be achieved until decades after urban development ceases (Henshaw and Booth, 2000). Erosion control structures in the Wash, however, may mitigate for lack of stability and allow for a more rapid rehabilitation than could otherwise be achieved.

Increasing benthic biodiversity in the Wash will likely depend to some degree on decreasing the magnitude and frequency of flood events that occur (e.g., Hollis, 1975) on a yearly basis. Perhaps this can be accomplished by an extensive effort of flood control (e.g., detention ponds) within the Las Vegas Wash watershed. One of the problems with this approach, however, might be increased water temperature, although this could vary with the season in which flooding occurred. Increased baseflow in the Wash could have negative consequences if this pattern of ever increasing flow continues over time. Even presently, flows are such that there is very little connectivity with the floodplain in terms of area inundated at high flows. Harrison et al. (2004) suggest that lateral connectivity is important and that rivers should be given lateral space and allowed to form floodplain habitats including side channels and stream margin habitats. Utilization of lateral space might result in a great diversity of aquatic habitats and likely increase diversity of invertebrates. Some of the aquatic taxa associated with backwater habitats, however, are those found above structures in the Wash and this spatial displacement from backwater lentic environments to above structure lentic environments appears to be effective in providing habitat that might only be found at lateral environments in a natural stream. Construction of side channels, however, could increase the area of this sort of habitat and might also encourage survival of terrestrial vegetation through increased soil moisture.

The relatively high invertebrate taxa richness found in tributaries suggests that conservation of these sites is desirable and their further degradation should be avoided. These sites may serve as reservoirs of biodiversity important for providing source material for the Wash. Unfortunately, it appears that some of these tributaries are being simplified (lined) to transport higher stormwater flows to the Wash. This may result in decreased biodiversity along with more rapid water runoff into the Wash causing higher flood flows.

Unintended consequences —Rehabilitation of the Wash may have unintended consequences. It appears, for example, that the development of habitat around erosion control structures has made it possible for exotic invasives to survive in what was originally a very harsh environment. Introductions from aquarium dumpings may have occurred sporadically since urbanization of the area, but it was only when the environment was modified that populations could persist and become self-sustaining. Invasive species have been recognized as a concern in other aquatic restoration projects (e.g., Bond and Lake, 2003) and it is possible that negative interactions between exotics and native species may occur.

Padilla and Williams (2004) provide evidence that aquarium and ornamental species are a group that may be especially invasive because of the large size and generally robust nature of the organisms released. This vector may be responsible for the appearance of exotic mollusks in the Wash. Other introductions could have occurred with plantings of native emergents that were recovered from other watersheds (e.g., plantings of bulrush (*Schoenoplectus*) or with what appear to be deliberate introductions of game fish like Largemouth bass (*Micropterus salmoides*), that were first noticed in 2007. Some organisms may be able to move upstream into the Wash from Lake Mead. This sort of transfer for aquatic stages might be inhibited, however, by erosion control structures, small water falls, and passage through the pipeline which contains the Wash flow below Lake Las Vegas. In the case of sensitive native aquatic invertebrates, the Wash is largely isolated from other lotic drainages that might provide colonizers. Langford et al. (2009) suggest that the absence of proximal sources of sensitive taxa may result in considerable time lags (decades) between stream improvements and the appearance of sensitive macroinvertebrate taxa. A variety of transfer methods will likely be responsible for the eventual make-up of macroinvertebrate communities in the Wash and the differences in hydrology and water quality from surrounding drainages may increase differences between the Wash and other proximal communities in the watershed.

Development of physical habitat has emerged as a key activity for managers charged with river restoration. It is often assumed that the biotic response to such development will proceed in a characteristic manner from degraded to reference site communities and that assemblages will be found somewhere on a continuum between these two extremes. Ordination and unique taxa associated with mainstem erosion control structures suggest that this is not the case in the Wash and indicates that the "reference" approach may not necessarily characterize expectations of habitat restoration activities, especially when the reference site is exposed to disturbances which do not occur at other monitored sites. This, in hindsight, might be expected, since "restoration" activities often involve creation of unique habitats. Zedler (1999) in a study of wetland mitigation draws similar conclusions. In the case of the Wash, physical restoration activities are overlayed on a template of altered water quality and hydrology that results in what may be a greater divergence from expected conditions.

Conclusions

There are several primary mechanisms that limit biodiversity in the Wash. High conductivities and possible low regional biodiversity may be considered large-scale limits to biodiversity, reflected to some degree by assemblages presently found in tributaries. Other limits include disturbance from flood flows, altered temperatures, high baseflows, low connectivity with the floodplain, and the small extent of riparian vegetation. Revegetation of riparian areas should continue and it would be especially desirable to increase connectivity of aquatic and riparian areas with the addition of side channels. Management efforts that allow for decreases in water temperature may also be helpful in increasing similarities between reference and restored macroinvertebrate assemblages richness and may also limit invasions of some exotic taxa. Colonization trajectories at erosion control structures appear to be taking a unique path relative to reference and tributary sites. An adaptive management approach, utilizing macroinvertebrate characteristics as targets/goals, should be used if changes in Wash operations occur.

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Table 1. Invertebrate metrics and environmental variables associated with types of sites along the Las Vegas Wash. Sampling events occurred annually (March/April) from 2001 to 2009 except for coarse particulate organic matter and periphyton biomass. Las Vegas Wash sites include those designated as Mainstem (without erosion control structures), Mainstem-structure (those at structures), Reference (LW11.1), and Wetland sites upstream of erosion control structures. Letters associated with a given variable that are different indicate significant (*P* < 0.05) inter-site type differences (Tukey's post-hoc test).

Variable	Site type					
	Mainstem (n=41)	Mainstem-structure (n=28)	Reference (n=9)	Tributary (n=62)	Wastewater (n=42)	Wetland (n=9)
Alkalinity (mg/L)	119[c] (83-180)	124[c] (105-150)	194[a] (180-241)	167[a,b] (63-334)	114[c] (75-215)	128[b,c] (100-150)
Ammonia (mg/L)	0.067[a] (0.015-0.151)	0.064[a] (0.015-0.143)	0.081[a] (0.015-0.181)	0.092[a] (0.015-0.256)	0.178[a] (0.009-4.65)	0.054[a] (0.015-0.165)
Coarse particulate organic matter (g)	0.27[b,c] (0.00-0.96) n=25	3.30[a,b] (0.00-26.64) n=28	0.41[a,b,c] (0.00-1.05) n=7	2.59[a] (0.00-14.65) n=47	0.06[c] (0.00-0.75) n=34	3.32[a] (0.09-6.50) n=9
Conductivity (µS/cm)	2287[c] (1760-2750)	2397[c] (2040-2670)	3725[b] (3430-3940)	5828[a] (1320-12520)	1766[d] (868-2260)	2078[c,d] (1372-2600)
Depth (m)	0.67[a] (0.20-1.10)	0.36[c,d] (0.15-1.20)	0.36[c,d] (0.05-0.50)	0.25[d] (0.02-0.70)	0.45[b,c] (0.10-0.90)	0.69[a,b] (0.10-1.20)
Dissolved oxygen (mg/L)	7.62[b] (6.09-9.25)	8.42[b,c] (6.90-10.38)	9.40[b,c] (8.19-11.04)	9.29[c] (3.15-15.66)	6.46[a] (4.10-8.33)	7.52[b,c] (5.41-9.40)
Hardness (mg/L)	630[b] (247-828)	682[b] (550-829)	1678[a] (1516-1942)	2756[a] (433-29834)	410[c] (214-551)	602[b] (435-766)
Invertebrate abundance	41.5[c] (0.0-223.0)	187.1[a,b] (9.0-1245.0)	124.8[a,b,c] (2.0-357.0)	341.0[a] (1.0-3078.0)	229.2[b,c] (0.0-4178.0)	37.3[b,c] (6.0-107.0)
Invertebrate richness	3.6[b,c] (0.0- 7.0)	6.7[a] (3.0-13.0)	7.1[a,b] (1.0- 17.0)	8.5[a] (1.0-16.0)	3.3[b,c] (0.0-7.0)	6.1[a,b] (3.0-14.0)
Nitrate (mg/L)	56.9[a] (15.4-91.4)	50.3[a,b] (13.9-69.8)	14.7[c] (3.2-21.9)	23.3[c] (0.2-87.0)	43.7[b,c] (0.6-104.0)	48.9[a,b] (23.0-65.7)

Percent sand	23[c] (0-100)	22[c] (0-65)	34[c] (0-95)	64[b] (0-100)	4[d] (0-30)	94[a] (80-100)
Periphyton (g)	0.12[a] (0.00-1.92) n=25	1.22[a,b] (0.00-6.71) n=28	0.58[a,b] (0.08-1.17) n=7	2.05[b,c] (0.00-21.75) n=47	5.65[b] (0.01-82.12) n=34	0.08[a,c] (0.00-0.68) n=9
Pfankuch index	92[a] (63-20)	85[a] (57-123)	89[a] (47-127)	83[a] (39- 131)	53[a] (38-109)	98[b] (74-139)
pH (S.U.)	7.5[b,c] (6.8-8.8)	8.0[a] (7.1- 8.7)	8.0[a] (7.7- 8.5)	7.9[a] (7.0- 9.3)	7.2[c] (6.1-8.8)	7.8[a,b] (7.4-8.3)
Substrate index	5.4[a] (3.0-8.0)	5.4[a] (3.6-6.8)	5.0[a,b,c] (3.1-8.0)	4.4[c,d] (3.0- 8.0)	6.5[b] (4.5-8.0)	3.2[c,d] (3.0-3.8)
Temperature (^{o}C)	22.2[b,c] (19.9-25.1)	22.9[c] (20.7-25.6)	14.2[a] (10.1-17.9)	20.9[b] (12.1-28.8)	22.9[c] (18.3-26.2)	20.2[b,c] (15.7-24.6)
Total phosphate (mg/L)	0.30[a] (0.04-1.43)	0.29[a] (0.03-0.75)	0.08[b] (0.003-0.32)	0.14[b] (0.003-0.57)	0.21[a,b] (0.004- 1.03)	0.19[a,b] (0.02-0.41)
Turbidity (NTU)	15.2[a] (0.5-112.0)	17.9[a] (2.6-78.8)	3.4[b,c] (0.5-15.8)	5.3[b] (0.7-44.5)	1.2[c] (0.3-3.4)	26.7[a] (4.6-95.7)
Velocity (m/S)	0.53[a,b] (0.00-1.31)	0.65[a] (0.30-1.00)	0.30[b,c] (0.13-0.62)	0.21[c,d] (0.00-1.18)	0.51[a,b] (0.15-1.18)	0.02[d] (0.00-0.09)
Width (m)	11[b] (3-38)	63[a] (17-100)	10[b,c] (2-32)	5[d] (1-25)	5[c,d] (2- 10)	40[a] (18-100)

Table 2. Taxa found in the Las Vegas Wash drainage with their associated pollution and sediment tolerance values.

TAXA		Pollution Tolerance Value[a]	Sediment Tolerance Value[b]
COLLEMBOLA		10	108
EPHEMEROPTERA			
	Baetis	4.7	72
	Caenis	6.3	72
	Callibaetis	8.4	72
	Camelobaetidius	4	--
	Fallceon	4	--
	Siphlonurus	5.5	72
ODONATA			
	Aeshnidae	3	72
	Calopterydidae	5	--
	Coenagrionidae	8	108
	Corduliidae	3.5	--
	Gomphidae	1	108
	Libellulidae	9	72
TRICHOPTERA			
	Culoptila	2	32
	Hydroptila	5.5	108
	Smicridea	4	72
LEPIDOPTERA			
	Petrophila	5	72
HEMIPTERA			
	Corixidae	8	108
	Mesovellidae	--	72
	Notonectidae	--	108
	Saldidae	10	--
	Veliidae	--	72
COLEOPTERA			
	Agabetes	--	72
	Agabinus	--	72
	Agabus	6.5	72
	Berosus	6.8	72
	Carabidae	4	--
	Enochrus	6.7	72
	Neoclypeodytes	5	--
	Optioservus	3.7	108
	Tropisternus	8.3	72
DIPTERA			
	Anopheles	7.6	108
	Bezzia/Probezzia	5.8	108
	Brachydeutera	--	--
	Chironominii	6	108
	Culex	9	108
	Culicoides	8.8	108

	Dasyhelea	--	108
	Diamesinae	2	--
	Dolichopodidae	5.9	108
	Empididae	5.9	--
	Ephydridae	6	108
	Limnophora	7	108
	Orthocladiinae	5	108
	Psychodidae	10	36
	Sciomyzidae	6	--
	Simuliidae	6	108
	Stratyomyidae	--	108
	Tabanidae	8	
	Tanypodinae	7	108
	Tanytarsini	6	108
	Tipulidae	3	72
BRYOZOA		--	--
TURBELLARIA		4	108
HIRUDINEA		6.7	108
OLIGOCHAETA			
	Enchytraeidae	10	--
	Lumbricidae	10	108
	Lumbriculidae	7.6	--
	Naididae	5	--
	Tubificidae	10	108
NEMERTEA			
	Prostoma	--	--
OSTRACODA		8	108
AMPHIPODA			
	Crangonyx	4	--
	Hyalella	8	108
DECAPODA			
	Cambaridae	6	108
GASTROPODA			
	Ancylidae	6	--
	Physidae	8	108
	Lymnaeidae	6.3	108
	Thiaridae	--	--
PELECYPODA			
	Corbicula	4.7	--

[a]Pollution tolerance values range from 0 to 10 with 0 being most sensitive and 10 most tolerant.
[b]Sediment tolerance values range from 2 to 108 with 2 being most sensitive and 108 most tolerant.

Table 3. Correlation analysis of taxa richness and time (sampling occasion) at sites in Las Vegas Wash.

Site type/Locations	*r* value	P value	Date of structure
Reference			
LW11.1	0.6345	0.0488	Site alterations may have been responsible for significant correlation.
Mainstem			
LW9.1	0.1540	0.6710	--
LW8.85	-0.3348	0.3444	--
LW0.55	0.0301	0.9342	--[a]
Mainstem-structure			
LW6.05	0.6367	0.0478	November 2000
LW5.5	0.6256	0.0530	December 2000
LW3.85	0.6833	0.0294	October 1999

[a]Fall of 2002, but flows prior to sampling in 2005 removed most of this structure.

Table 4. Results of stepwise multiple regressions for taxa richness. Variables that were not significant in the model included construction disturbance and site.

	Taxa richness			
Variable	Coefficient	Standard error	T	P
Constant	-88.4808	37.1674	-2.38	0.0206
Year	0.04506	0.01854	2.43	0.0182
Structure presence	0.47174	0.09618	4.90	0.0000
Discharge (m^3/S)	-0.16591	0.05780	-2.87	0.0057
R squared	0.4072			

Table 5. Weighted correlation matrix showing relationship between species axes and significant environmental variables. Highest correlations associated with a given variable are shown in **bold**.

Variable	Axis 1	Axis 2	Axis 3	Axis 4
Alkalinity	**0.3279**	-0.2666	0.1679	0.0263
Conductivity	**0.6740**	-0.2250	0.0510	0.0351
Depth	**-0.3662**	0.3044	-0.1131	0.0625
Dissolved oxygen	0.0510	**-0.3045**	-0.0719	0.2739
Pfankuch index	0.0318	**-0.3532**	0.0633	0.0214
pH	0.1710	**-0.4541**	-0.0647	0.1517
Temperature	-0.1944	**0.2325**	-0.0169	-0.2123
Turbidity	-0.2010	**-0.4614**	-0.1407	-0.1177
Velocity	**-0.7033**	0.0838	0.1156	-0.0338
Width	**-0.5452**	-0.3929	-0.1314	-0.1539

Figure 1. Mean annual (USGS water year) discharge at several mainstem Las Vegas Wash sites with LW0.55 the site furthest downstream. LW11.1 is above the influence of wastewater treatment plants which discharge additional water into the Wash. Only a portion of the discharge record is available for LW11.1.

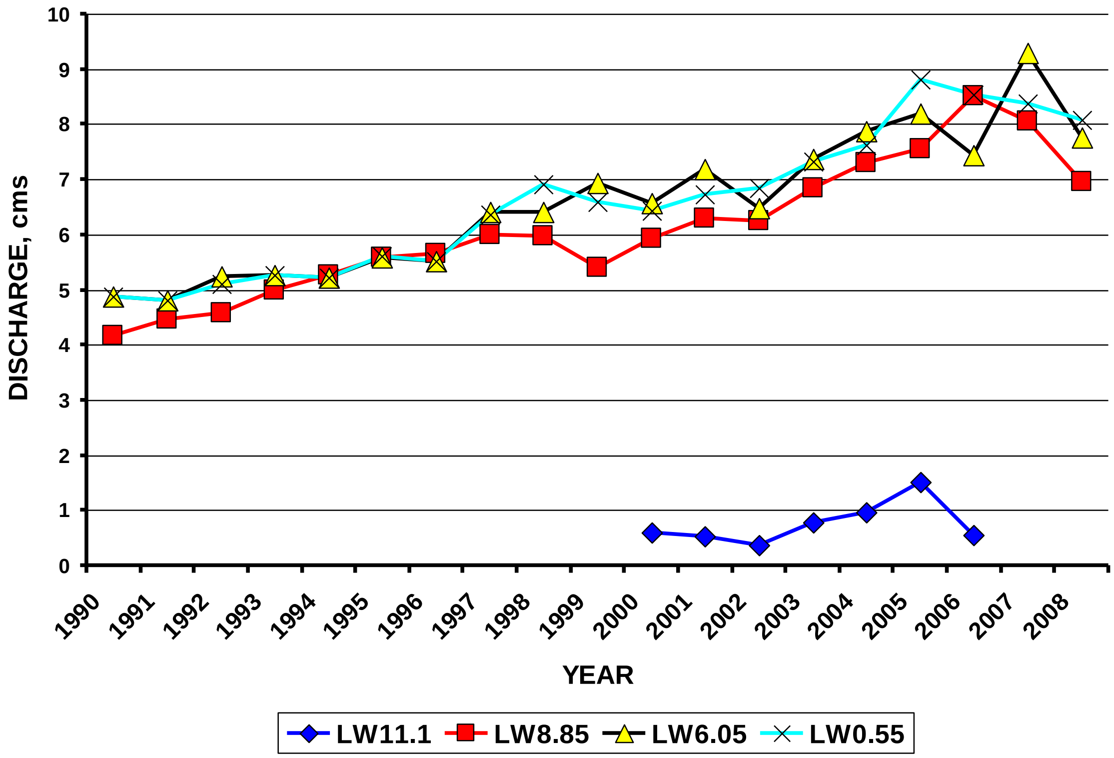


Figure 2. Macroinvertebrate sampling sites associated with the Las Vegas Wash.

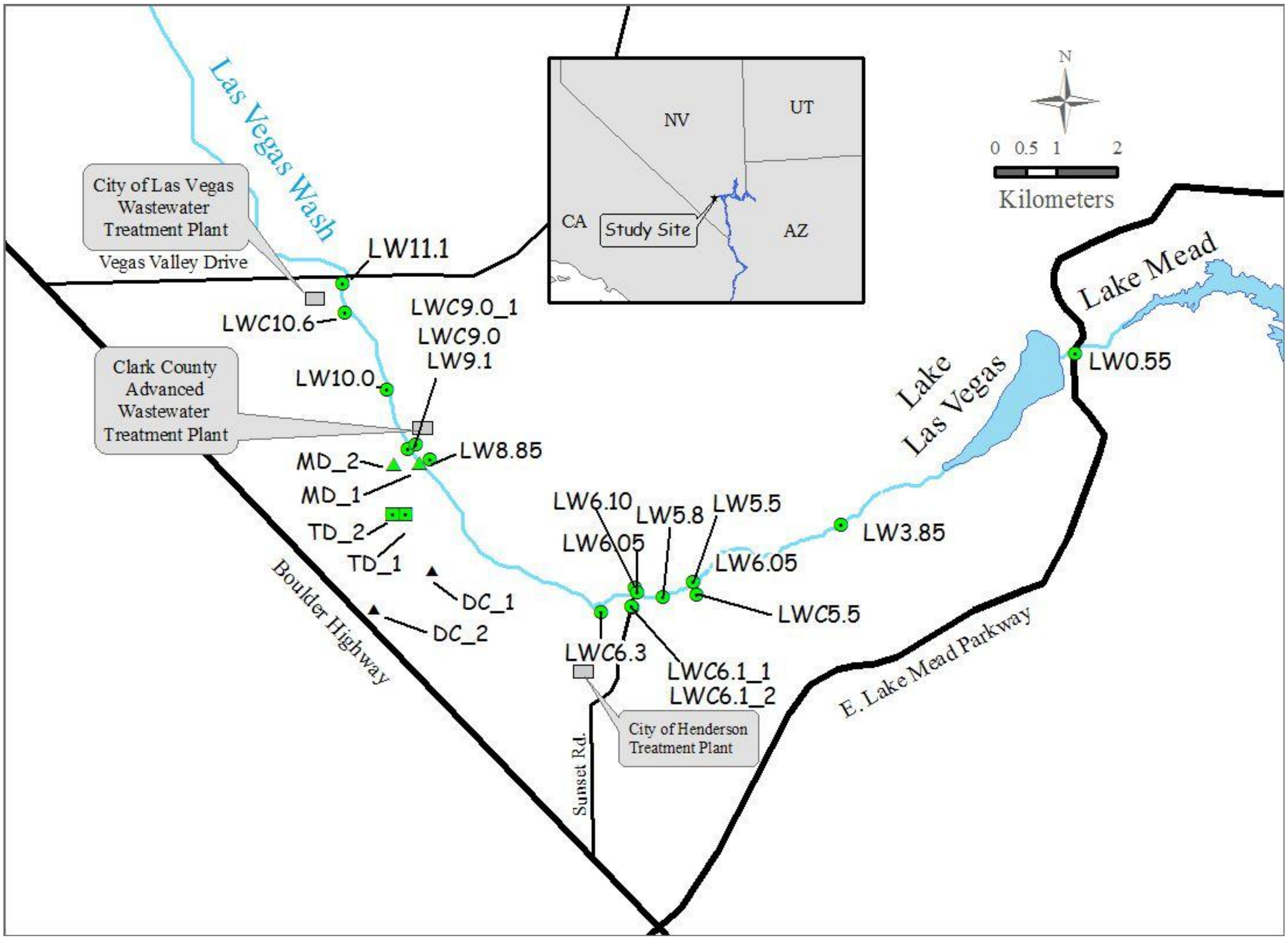
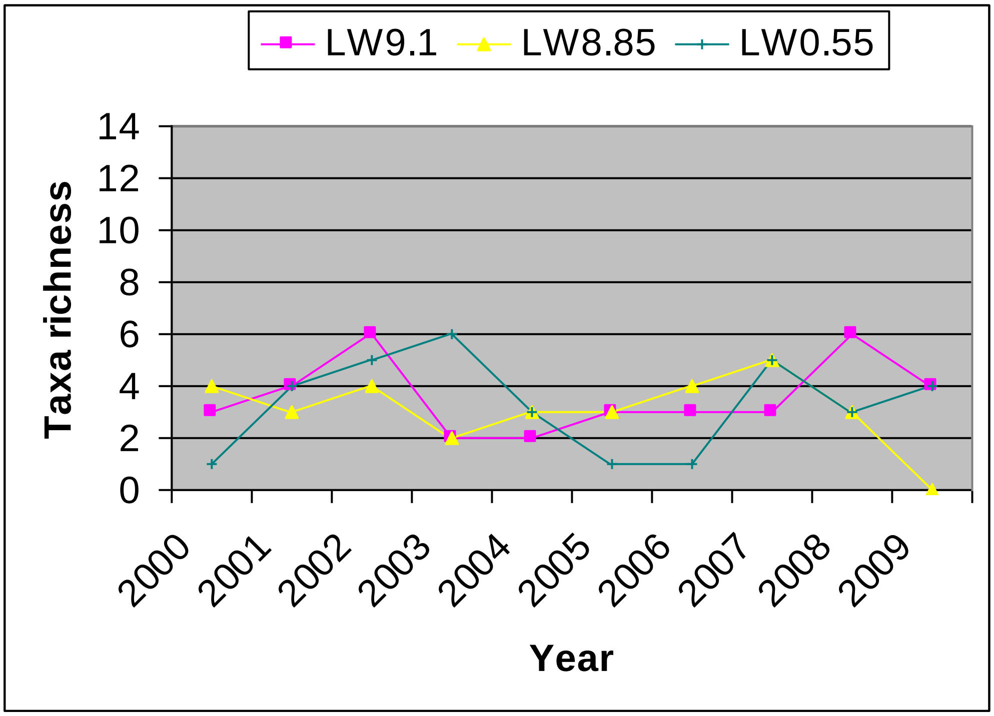
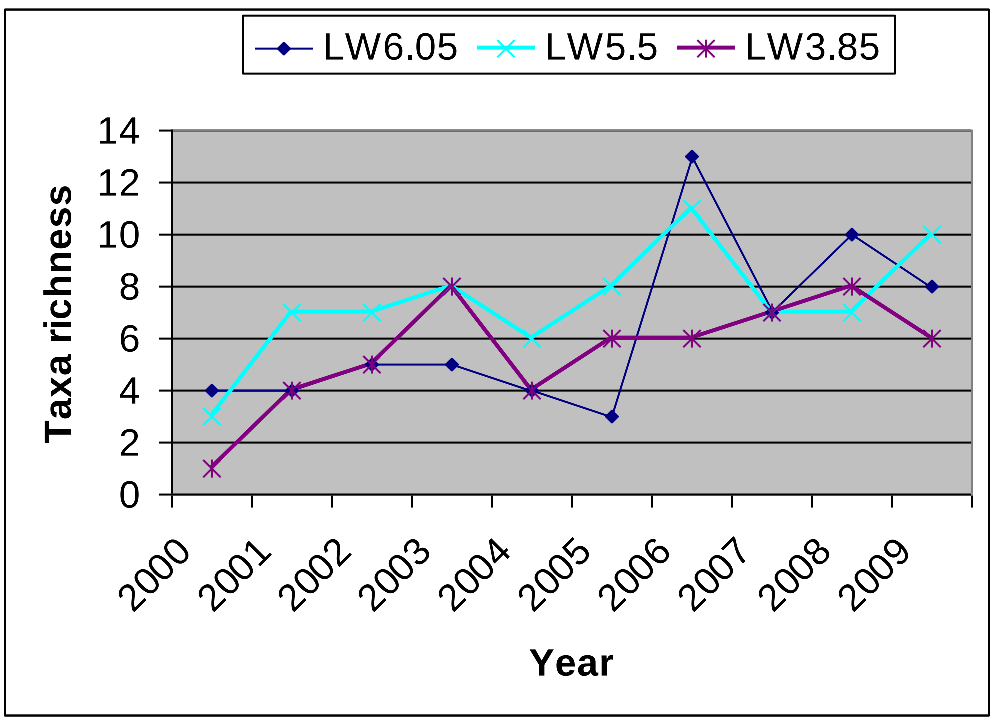


Figure 3. Taxa richness over time at sites without (a) and with (b) erosion control structures.



a



b

Figure 4. Biplot from data collected in March/April 2001-2009 based on a canonical correspondence analysis (CCA) of sites with respect to environmental variables. Environmental variables were related to community attributes as shown by arrows. Site samples are represented by geometric shapes as shown in the legend. Year of collection (1 to 9) precedes abbreviated site code.

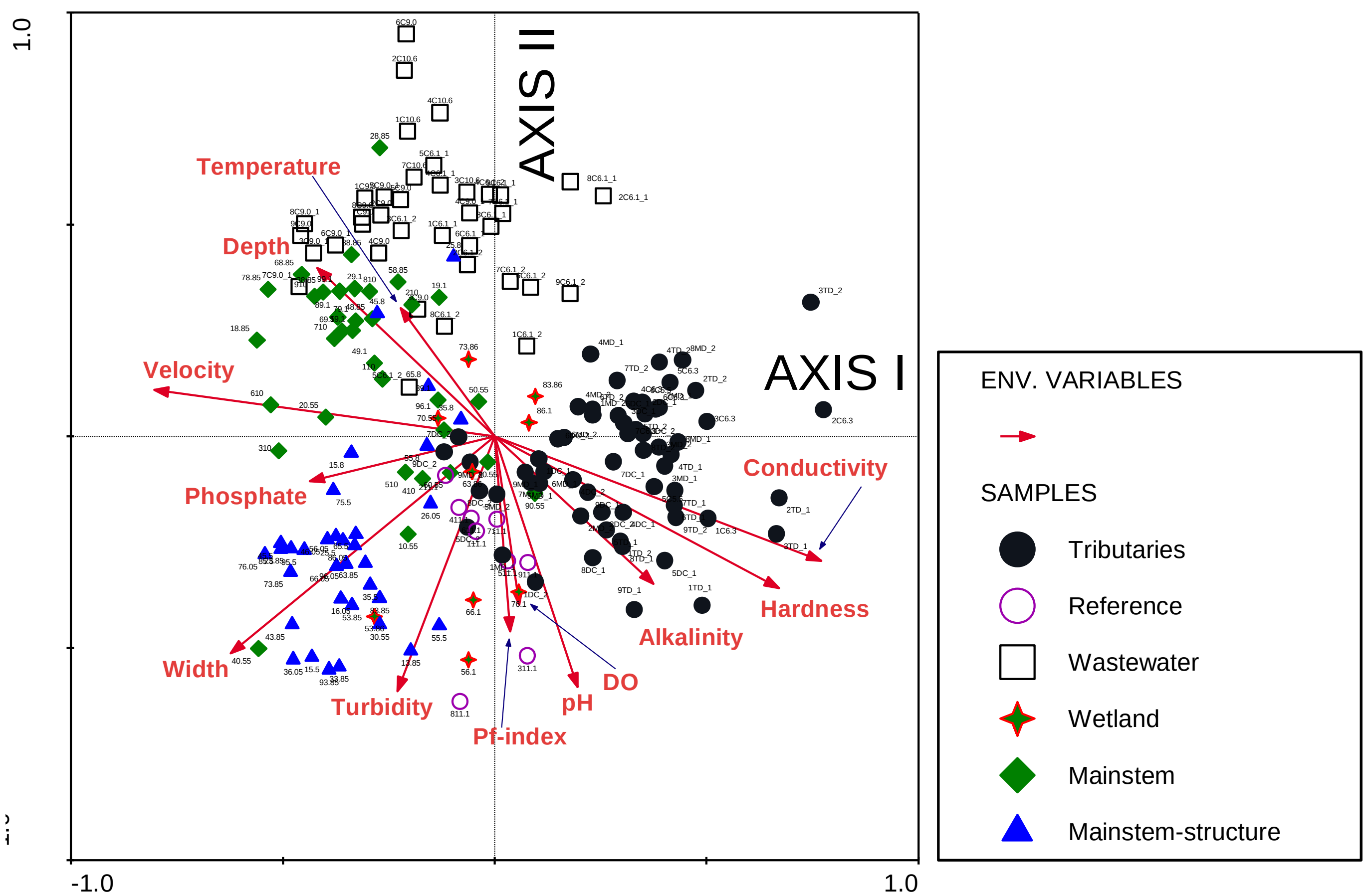


Figure 5. Biplot from data collected in March/April 2001-2009 based on a canonical correspondence analysis (CCA) of macroinvertebrate taxa in association with environmental variables. Only those species that had a fit $\geq$ 5% are shown in the figure.

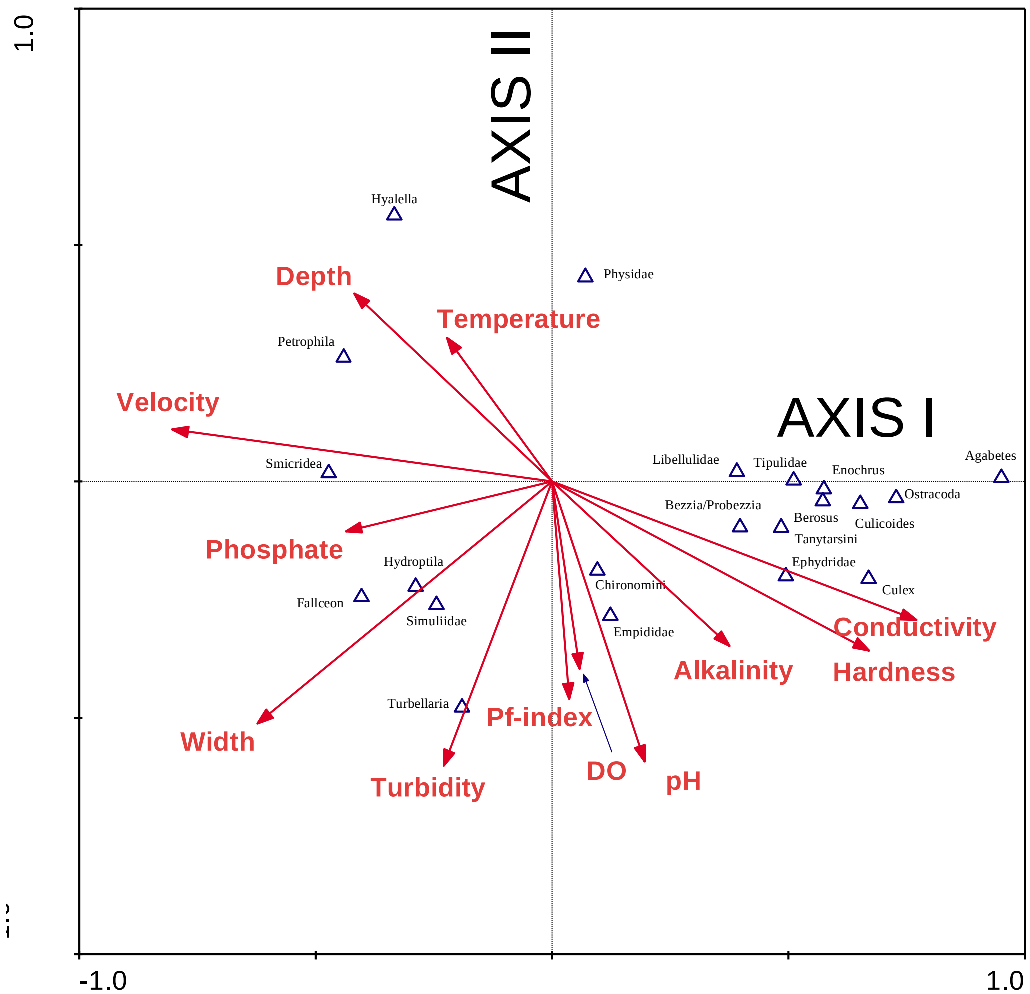
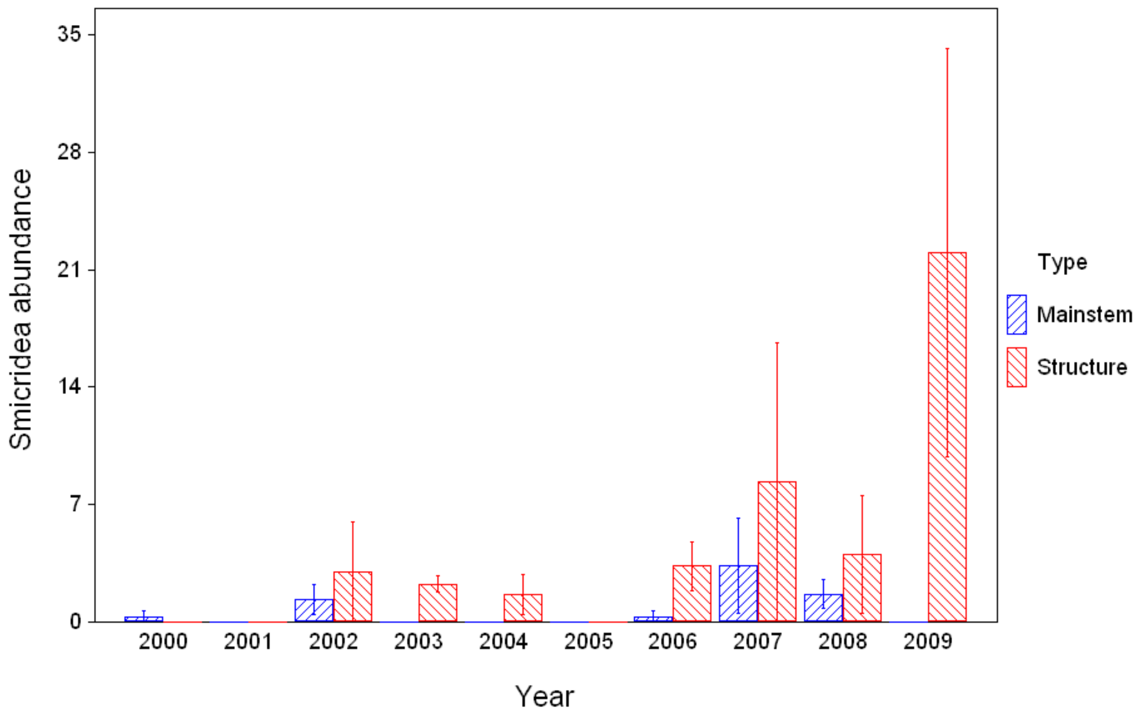


Figure 6. ***Smicridea*** **abundance at sites without (mainstem) and with erosion control structures in Las Vegas Wash. Error bars indicate standard error from mean values.**



PEER REVIEW DOCUMENTATION

PROJECT AND DOCUMENT INFORMATION

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REVIEW CERTIFICATION

Peer Reviewer - I have reviewed the assigned Items/Section(s) noted for the above document and believe them to be in accordance with the project requirements, standards of the profession, and Reclamation policy.

Reviewer: Doug Andersen Review Date:
Signature 11 Feb 2010

Preparer - I have discussed the above document and review requirements with the Peer Reviewer and believe that this review is completed, and that the document will meet the requirements of the project.

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