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Technical Memorandum No. 86-68220-11-01

Stream Macroinvertebrate Assemblages Associated with the Las Vegas Wash Watershed 2000-2010

U.S. Department of the Interior
Bureau of Reclamation
Denver, Colorado

January 2011

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Abstract

Effects of stream erosion control structures on aquatic macroinvertebrates were studied (2000-2010) in a wastewater dominated drainage (Wash) in Las Vegas, Nevada. Mainstem sites with and without structures, wastewater treatment plant outfalls, reference sites above treatment plant inputs, and tributary sites unaffected by treatment plant water were sampled.

Ordination of samples collected on an annual basis suggested hydrology and channel characteristics (current velocity, stream depth, and width), and water quality (conductivity) were primary factors in organizing macroinvertebrate communities, with some variables altered at structures. Treatment plant inputs changed hydrology (increased flows), water chemistry (conductivity decreased below treatment plants), and temperature. Structures appeared important in retaining organic matter and, among mainstem Wash sites, coarse particulate organic matter was significantly higher at structures when compared to sites without structures. Invertebrate assemblages differed between site types, with midges and damselflies important at tributary sites and *Fallceon* mayflies, *Smicridea* caddisflies, and blackflies (Simuliidae) common at erosion control structures.

Examination of seasonally collected data showed that there was no significant difference in taxa richness between seasons. There were some differences in macroinvertebrate communities with the mayfly *Camelobaetidius* absent from samples collected in March.

Erosion control structures, coupled with warm effluent, high baseflows, and altered water quality resulted in development of a macroinvertebrate community that differed from both reference and tributary sites. Locally unique communities developed at structures, which also may have facilitated exotic species invasions. Analyses showed that taxa richness increased over time at these sites and differed significantly from richness at sites without structures.

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 & 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). It is often unclear which factors have the most impact to invertebrate communities.

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). Studies rarely consider specific mechanisms that cause urbanization effects (Paul & Meyer, 2001) or evaluate the effectiveness of stream restoration (Moerke & Lamberti, 2004, Miller et al., 2010). Macroinvertebrate studies in regard to stream restoration are especially rare (Miller et al., 2010). 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 & 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 & 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 analogous issues and Walsh et al. (2005) describe 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 Wash summer flow (Reclamation, 1982). In the 1930's and 1940's when groundwater was the basic water resource, wastewater treatment plants were built and began to discharge effluent into the Wash. By the early 1940’s water managers were expressing concerns with limited supplies (SNWA, 2006) and in 1942 water was imported from Lake Mead to process magnesium for industrial use, and then discharged into the Wash (Reclamation, 1982). These 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 Valley Water District, which included the city of Las Vegas and most of the populated areas of Clark County, became increasingly dependent upon Colorado River water from Lake Mead. Currently approximately 85-90% 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 cause increased flow volume as most of the water 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 Water Pollution Control Facility (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 (e.g., Figure 1).

Buckingham & Whitney (2007) found the hydrologic history of the Wash dominated by three periods. Small additions of wastewater prior to 1975 resulted in an extensive marsh development with limited erosion. Between 1975 and 1989 wastewater discharge 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 & Whitney, 2007). In response, the Las Vegas Wash Coordination Committee in 1999 completed the first of 22 grade control structures for channel stabilization 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 LW6.05, LW5.5, and LW3.85 (Table 1). Erosion control structures placed along the Wash are permanent, low height dams or weirs designed and engineered to endure and help disipate energy from large storm events. Building materials range from confined rock riprap to roller-compacted concrete secured to drilled concrete piles. Along with these constructed weirs, stabilization of the channel bed has utilized bank protection and revegetation. Revegetation with native plant species included structural dominants Fremont cottonwood (*Populus fremontii*) and willow (*Salix* spp.). In many cases revegetated sites were at erosion control sites where the terrain surface had been lowered. The effect was to create a hydologically functioning floodplain which provided the opportunity for occasional flooding of a portion of the terrestrial environment. 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.

Modifications of the Wash were similar to those identified by Miller et al. (2010) as typical of in-stream habitat restoration and include boulder and weir additions along with channel changes caused by cross-stream structures. Addition of large woody debris is also a common restoration technique (Miller et al., 2010). While this technique was not actively pursued in the Wash, woody material has been added incidently through inputs of woody debris generated from succesful riparian plantings.

The purpose of this study was to monitor changes in macroinvertebrate assemblages associated with the construction of Wash erosion control features 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. Therefore, the focus was on environmental and chemical variables at the local scale rather than at the landscape scale. Macroinvertebrate community composition was assessed in the Wash and its tributaries to (1) identify environmental factors that may control biotic structure in this urban-impacted area, (2) describe the relationship of biota with erosion control structures, and (3) examine seasonal components of invertebrate communities and biotic variables.

Methods

Study area

Las Vegas Wash, a natural wash east of the city of Las Vegas, Nevada, carries stormwater, 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 the Wash.

Annual sampling at sites took place from 2000 to 2010 in March or April of each year. In 2004 mainstem site sampling was expanded to a quarterly basis for examination of seasonal variability. Monitoring was started at the same time that construction of erosion control structures was initiated so comparisons of the effects of structures on macroinvertebrate communities could be determined over time. Samples were also collected from other sites in the Wash that lacked erosion control structures or the upstream sites that were not influenced by wastewater treatment plants. Furthest upstream sites were considered reference or benchmark sites that represent the "best of what's left" (Hawkins et al., 2010). 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 (e.g. Table 1). 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. As the study has continued several sites have been added or dropped as conditions change. Two 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). This tributary site was no longer available for sampling by 2007, due to the flow having been diverted in a buried concrete conduit. Upstream sites on the mainstem Wash (LW11.76 and LW11.1) were above the influence of wastewater treatment plants and also considered reference sites for the Wash. While LW11.1 was sampled from the beginning of the monitoring program, LW11.76 was added in 2010. LW11.1 may be compromised as a reference site in the near future because construction is progressing on a new wastewater treatment facility in North Las Vegas that plans to release effluent into the open, county-owned Sloan Channel, above LW11.1. LW7.0 was also added in 2010 at a site where an erosion control structure is planned for a later date. The site LWC10.6 was lost in 2010 when the effluent channel was replaced with a pipe to Las Vegas Wash from the City of Las Vegas treatment plant.

Site-types included: mainstem, mainstem-structure, reference, tributary, wastewater, and wetland environments (e.g., Table 1). Sampling of a variety of environmental variables was initiated in 2001.

Chemical, physical, and biological methods

Environmental variables measured for each site included water chemistry, physical parameters, 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. Stream wet width 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. Variables include estimates of plant density on the upper banks, the frequency of raw banks, and how much of the bottom is affected by scouring and deposition. 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.

Yearly sampling with the full set of variables took place in April 2001 to 2002, March 2003 to 2006, and April 2007 to 2009. 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. Insect taxa were generally identified to genus, although Chironomidae were identified to subfamily or tribe.

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.

Data analysis

Sites were not randomly assigned to treatments (mainstem, mainstem-structure, reference, tributary, wastewater, and wetland) nor were they randomly interspersed along the Wash. Thus differences in macroinvertebrate assemblages between and among sites may reflect something besides a difference in habitat, i.e., measurements may be biased in an unknown manner, limiting inferences. “Replicates” used in the site-type analysis are from different years and different sites. In some cases a site could shift category (e.g, a mainstem site one year could become a mainstem-structure site the next). Perhaps the best description of this study is the “quasi-experiment” of Hargrove and Pickering (1992) where some level of pseudoreplication is considered acceptable in exchange for realism.

ANOVA was used to compare environmental variables associated with site types in annually collected data. Ordination techniques were used to examine patterns in annually collected 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. Pearson correlation was used to examine whether taxa richness increased over time at Mainstem and Mainstem-structure sites in the Wash.

Factorial ANOVA followed by Tukey's test for comparisons were used to compare means of biotic variables including taxa richness, invertebrate abundance, CPOM, and periphyton biomass at different types of environments and months for seasonal data. Data were transformed, if needed to normalize distributions, using ln (X+1). Data analyses with ANOVA, in this case, are limited in interpretation by pseudoreplication (Hurlbert, 1984). Ordination (DCA) was used to determine whether the makeup of invertebrate communities differed seasonally.

Results

Environmental variables

Nitrate concentrations were highest at sites downstream of wastewater treatment facilities, while reference and tributary sites had significantly lower concentrations (Table 2). Other water quality parameters also indicated effects from wastewater on the mainstem Wash sites (alkalinity, conductivity, DO, hardness, pH, and temperature). Processed water from wastewater treatment plants was not exposed to sediment and wastewater sites had the lowest mean turbidity values (Table 2). Tributary sites were often similar to the reference site in water quality (Table 2).

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 2). Mainstem Wash and mainstem sites with structure had significantly higher values than tributary or wetland sites (Table 2). Significantly higher 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 2). Velocities at these two sites differed significantly from velocities at reference, tributary, and wetland sites. Stream width ranged from 1 m to 108 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; these differed significantly from other sites (Table 2). Pfankuch's Index varied from 38 to 139, with lower values associated with sites that were less prone to damage from floods. Wastewater sites that had stabilized flows and banks, and were lined with concrete, differed significantly in Pfankuch value from all other sites (Table 2).

Multivariate analysis of annually collected data

Results of CCA from the 2001 to 2010 studies (Figures 2 and 3) of the stream benthos had eigenvalues of 0.340 and 0.218 for the first two axes and explained 11.8% of the species data variation and 59.6% of the species-environment relation. Initial environmental variables used in the model included alkalinity, ammonia, conductivity, depth, DO, nitrate, percent sand, Pfankuch's Index, temperature, total phosphate, turbidity, and width. Hardness, pH, SI, and velocity were not used in the initial model because, respectively, they were highly correlated with conductivity (r=0.9226, $p<0.0000$), DO (r=0.6049, $p<0.0000$), percent sand (r=-0.8657, $p<0.0000$), and percent sand (r=-0.6062, $p<0.0000$). Variables found to be significant ($P<0.05$) in the model were alkalinity, conductivity, depth, DO, percent sand, Pfankuch's Index, total phosphate, temperature, turbidity, and width.

Alkalinity, conductivity, depth, percent sand, total phosphate, and width were correlated with the first axis, while DO, Pfankuch's Index, temperature and turbidity were correlated with Axis 2 (Table 3). No variables had their highest correlation with the third or fourth axis and these explained only a small portion of species-environment relationships.

Site samples tended to cluster in four areas (Figure 2) 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 mainstem reference site (LW11.1) took an intermediate position between tributary sites and mainstem sites with structure (Figure 2). Wetlands sites that were forming above erosion control structures were scattered along Axis 2.

Depth and width were relatively low at tributary sites with increased values at mainstem sites (Figure 2, Table 2). Percent sand was relatively high at tributary sites along Axis 1. Alkalinity and conductivity were higher at tributary sites (see Figure 2, Table 2) and this native water was diluted by high volumes of low conductivity wastewater downstream in the Wash (e.g., Table 2). Dissolved oxygen concentrations were lower at wastewater outfall sites receiving water from treatment plants (Figure 4, Table 2). It also appeared that wastewater treatment plant operations resulted in higher temperatures at wastewater outfalls. Relatively low Pfankuch index values were associated with hydrologically and physically stable wastewater outfall sites.

Relationship between biota and site types

Characteristic taxa were found at specific site types (Figure 3). Distributional data indicated midges (tanypodinae and tanytarsini), odonates (Coenagrionidae and Libellulidae), and a variety of Diptera such as Empididae, Ephydridae, Muscidae, and Tipulidae were found at tributaries (to the right on Axis 1, Figure 3). *Fallceon* mayflies and *Smicridea* caddisflies were found at mainstem and mainstem-structure sites (to the left on Axis 1, Figure 3). Blackflies (Simuliidae) were also characteristic of mainstem-structure sites (Figure 3 and Figure 4) although they were also common, but variable in abundance at reference sites. *Hyalella* was associated with increased depth and temperature along with lower DO and pH, characteristics found at wastewater outfall sites (Figure 3). The vast majority of macroinvertebrates collected from the Las Vegas Wash basin were tolerant of organic pollution and sediment (Nelson, *in press*).

Annual changes in taxa richness

Taxa richness increased significantly (r=0.5918, p=0.0005) over time at Wash sites where structures were built (Figure 5a), but was unchanged sites in the Wash that lacked these structures (r=0.0124, p=0.9349) (Figure 5b). Some taxa, despite multiple sampling years, were only found at structures. In 2003 *Corbicula* clams were first detected at structures, with thiarid snails appearing in 2007. In 2006, the tropical aquarium fish, shortfin molly (*Poecilia mexicana*) was first observed in the Wash, suggesting that dumping of aquarium contents into the Wash resulted in the introduction of both mollies and tropical thiarid snails. In 2008, relatively sensitive native taxa like the caddisfly *Culoptila* have appeared in the Wash. *Culoptila* have been found to be sensitive to sediment in systems at the species (Blinn & Ruiter, 2006), genus (U.S. Forest Service, 1989), and family level (Carlisle et al., 2007) and their presence in the Wash at this time may be a biological sign that sediment from erosion is declining. This decline in sediment is supported by turbidity data collected in the mainstem portion of the Wash (Figure 6). Most of these new taxa were exclusively found at erosion control structures in the Wash and not detected at tributary or reference sites. The continued addition of structures in the Wash and the colonization, over time, by cattail and common reed, and their contribution to CPOM on developing substrates may have also played a role in the ability of new taxa to colonize the area. It appears that taxa are continuing to come into the system, as the caddisfly *Nectopsyche* was detected during monitoring operations for the first time in 2010.

Seasonal characteristics

With the exception of periphyton biomass; biological metrics of taxa richness, invertebrate abundance, and CPOM did not differ between seasons at sites along the Wash. Periphyton biomass differed by season ($p=0.0029$), habitat type ($p<0.0000$), and the interaction term between season and habitat type ($p=0.0013$). Periphyton biomass was significantly higher in March compared to September and December, but June data overlapped with all other seasons. Periphyton mass was significantly lower at mainstem sites relative to reference and mainstem-structure sites. The latter two habitat types did not differ from each other. The interaction between seasons and habitat type was complicated and is shown in Figure 7.

Taxa richness and invertebrate abundance did not differ with season but did vary with habitat type ($p<0.0000$). Both were significantly lower at mainstem habitats relative to either reference or mainstem sites that contained structure (Figure 8a and 8b). CPOM values were also significantly lower at mainstem habitats (Figure 9).

While there were no significant differences in invertebrate richness or abundance seasonally, DCA was used to determine whether the makeup of communities differed with season (Figure 10). Results of DCA from the 2004 to 2010 studies of the stream benthos had eigenvalues of 0.358 and 0.241 for the first two axes and explained 20.7% of the species data variation. Figure 10 suggested that there was some difference seasonally with some separation of groups based upon the month in which data were collected. There was, however, also a fair amount of overlap, suggesting that most taxa were present throughout the year. There were specific genera, such as *Camelobaetidius*, which were only present at certain times of the year (Figure 11).

Discussion

Environmental Factors Associated with Macroinvertebrate Communities

Las Vegas Wash sites below effluent outfalls had significantly lower conductivity values compared to reference and tributary sites. Factors differentiating benthic invertebrate assemblages included hydrology/channel characteristics, catchment geology (salinity/conductivity), and water quality changes (temperature, pH, DO, phosphate) associated with inflows of wastewater treatment plant effluent. 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.

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. 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 & 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 & 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 & 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 from wastewater discharges likely are detrimental to certain invertebrate taxa, resulting in competitive exclusion by more tolerant species (e.g., Cairns, Jr., 1972). Taxon-specific themal tolerance may also be important and it is noted that some common taxa in the Wash such as the mayfly *Fallceon* and the amphipod *Hyalella* have very high tolerances to high temperatures (Carlisle et al., 2007). Wang & Kanehl (2003) found that increased water temperature caused by urbanization was one of the most influential factors, whatever the mechanism, 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 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 Litvan 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 & 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. As sites where structures were placed have developed over time, taxa richness has increased significantly.

Harrison et al. (2004) point out that macroinvertebrates have complex life cycles in which different life stages may use different parts of the aquatic or riparian environment. It may be that erosion control structures provide only some of these habitat requirements for Wash aquatic invertebrates. For example, the limited (but improving) 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 & 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). Erosion control structures were also important in development of habitat diversity, providing lotic habitat on the structures and lentic environments similar to that of tributaries at wetlands that formed above structures. This wetland environment is creating habitat for taxa that are typically associated with reference/tributary sites. Miller et al. (2010) indicate that this type of backwater habitat is especially critical to increasing biodiversity in river restoration.

Factors Important in Restoration from a Macroinvertebrate Perspective

Data from this study suggest that in-stream macroinvertebrate habitat at the Wash, in the absence of erosion control structures, is quite limited. 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 & 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. Also important to macroinvertebrates is the provision of important food resources in the form of increased CPOM and periphyton that are provided at structures. 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 a role in influencing macroinvertebrate assemblages. The paucity of sensitive taxa (Nelson, 2011) suggests 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 Wash, taxa richness significantly declined with increasing magnitude of recent discharges (Nelson, 2011). Others also consider stormwater runoff and floods to have major impacts on urban systems (Walsh et al., 2005; Trush et al., 2000). Increasing benthic biodiversity in the Wash may depend to some degree on decreasing the magnitude and frequency of flood events (e.g., Hollis, 1975). This may be difficult to achieve, although there is already an extensive effort to control flood discharge (e.g., through use of detention ponds) within the watershed (PBS&J, 2008). It is also likely that the restriction of habitat at unimproved portions of the Wash by the constant, high baseflow velocities will be exacerbated if the pattern of ever-increasing baseflow continues. Constant disturbance by floods and increasing baseflows may limit biodiversity to tolerant taxa. *Fallceon quilleri*, the most abundant mayfly in the Wash, is an example of a species adapted to frequent flooding. Floods adversely impact aquatic larval stages, but because of its rapid life cycle, aerial adults are typically present during floods (e.g., Gray, 1981) and quickly recolonize aquatic habitats post-flood. Channel stability in urban environments may not be achieved until decades after urban development ceases (Henshaw & Booth, 2000). Erosion control structures in the Wash, however, may mitigate for diminished stability and allow for a more rapid rehabilitation than otherwise achievable.

Presently, flows have little connectivity to the floodplain, in terms of area inundated at “normal” non-storm high flows. Harrison et al. (2004) suggest that connectivity is important and that rivers should be given lateral space for formation of side channel and stream margin habitats. Utilization of lateral space might result in a greater diversity of aquatic habitats and lead to increased invertebrate diversity. 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 protection and conservation of these sites is desirable. 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 as source habitats are lost, as well as contributing to more rapid water runoff and higher flood flows in the Wash. Political support to stop this process is unlikely because it has is generally undertaken to protect human life and property. Sociological aspects are typically not considered in restoration projects (e.g., Choi et al., 2008) but can have a large impact on project success. Degradation of the tributaries may increase the value of restoration activities in the Wash that enhance biodiversity.

Unintended consequences

Wash rehabilitation may have unintended consequences. It appears, for example, that the development of habitat around erosion control structures has made possible survival of exotic invasives 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 & Lake, 2003) and it is possible that negative interactions between exotics and native species will occur.

Padilla & 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. Aquarium dumpings may be responsible for the appearance of exotic mollusks in the Wash. Other introductions could have occurred with the transplanting of native emergents collected in other watersheds (e.g., plantings of bulrush (*Schoenoplectus*)) or with what appear to be deliberate introductions of game fish like Largemouth bass (*Micropterus salmoides*), first noted in 2007. Some organisms may move upstream into the Wash from Lake Mead, although transfer for aquatic stages would be inhibited, by erosion control structures and small waterfalls. In the case of sensitive native aquatic invertebrates, the Wash is largely isolated from other lotic drainages that might provide colonizers (but see the example of *Culoptila* and the recent detection of *Nectopsyche*). 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 discrepancy 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 at the more lotic sites 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 and disturbances. Muotka et al. (2002) make the point that stream restoration is a unique disturbance to which stream biota have not evolved a standard response and, similar to the present paper, found that restored stream communities differed from those found in natural streams. This is somewhat comparable to findings from a meta-analysis done by Miller et al. (2010) where richness levels at restored sites did not return to target levels derived from minimally impacted stream sites. Zedler & Callaway (1999) in a study of wetland mitigation draw similar conclusions and suggest that when ecosystems are restored, "development may proceed along complex paths that are difficult or impossible to predict". Hilderbrand et al. (2005) indicates that restoring an ecosystem of specific composition is difficult and that the dynamic nature of community assembly should be expected. In 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 communities.

Acknowledgements

I thank Amy Stephenson of the Bureau of Reclamation Lower Colorado Regional Offices' Technical Services Group for chemical analyses of water samples and for her assistance with field sampling over the years. Doug Andersen (USGS) helped with sampling and reviewed the document. Rich Durfee helped with identifying invertebrates in some of the samples. Thanks to Pam Adams and Becky Blasius, LC Regional Office, and Seth Shanahan, Southern Nevada Water Authority, for providing additional support. Funding support over the years has been provided by the LC Regional Office, Southern Nevada Water Authority, and the Reclamation S&T Program.

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Table 1. Sites used for study of Las Vegas Wash macroinvertebrates.

Site code	Description	Site type
MD_1	Monson Drain-East	Tributary
MD_2	Monson Drain-West	Tributary
TD_1	Tropicana Wash-East	Tributary
TD_2	Tropicana Wash-West	Tributary
DC_1	Duck Creek at Broadbent	Tributary
DC_2	Duck Creek at Boulder Highway	Tributary
LW11.76	LV Wash above Vegas Valley Drive-sampling initiated in 2010-furthest upstream site	Reference
LW11.1	LV Wash below Vegas Valley Drive	Reference
LWC10.6	Discharge channel from the City of Las Vegas Wastewater Treatment Plant (CLVWTP)-sampling halted in 2010 after effluent placed in pipe	Wastewater
LW10	LV Wash	Mainstem
LW9.1	LV Wash upstream of confluence with Clark County Advanced Wastewater Treatment Plant (CCAWTP)	Mainstem
LWC9.0	Discharge channel from CCAWTP	Wastewater

		Wastewater
LWC9.0_1	New discharge channel from CCAWTP	
LW8.85	LV Wash	Mainstem
LW7.0	LV Wash-sampling initiated in 2010	Mainstem
LW6.10	Backwater above structure-sampling initiated in 2005	Wetland
LW6.05	LV Wash at Pabco Road weir	Mainstem, Mainstem-structure
LWC6.3	Saline spring-consistently dry after 2007-sampling halted	Tributary
LWC6.1_1	City of Henderson discharge	Wastewater
LWC6.1_2	Pittman Bypass-discharge from TIMET	Wastewater
LW5.8	LV Wash	Mainstem
LW5.5	LV Wash at Bostick weir	Mainstem, Mainstem-structure
LWC5.5	Inflow at 5.5-flow diverted to conduit by 2007	Tributary
LW3.86	Wetland above structure-sampled from 2005 to 2008	Wetland
LW3.85	LV Wash at Demonstration weir	Mainstem-structure
LW0.55	LV Wash downstream from the Northshore Road Bridge. Weir present in 2003.	Mainstem, Mainstem-structure

Table 2. Invertebrate metrics and environmental variables associated with types of sites along the Las Vegas Wash from annual sampling. Sampling events occurred from 2001 to 2010. 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). The range of values for variables is enclosed in parentheses within the table.

Variable	Site type					
	Mainstem	Mainstem-structure	Reference	Tributary	Wastewater	Wetland
Alkalinity (mg/L)	119[b] (82-224)	122[b] (100-150)	191[a] (168-241)	168[a] (63-334)	113[b] (75-215)	126[b] (100-150)
Ammonia (mg/L)	0.066[a] (0.015-0.151)	0.062[a] (0.015-0.143)	0.079[a] (0.015-0.181)	0.095[a] (0.015-0.256)	0.169[a] (0.009-4.65)	0.052[a] (0.015-0.165)
Conductivity (µS/cm)	2283[c] (1760-2750)	2401[c] (2040-2670)	3725[b] (3430-3940)	5831[a] (1320-12520)	1781[d] (868-2260)	2112[c,d] (1372-2600)
Depth (m)	0.68[a] (0.20-1.10)	0.36[b,c] (0.15-1.20)	0.36[b,c] (0.05-0.50)	0.24[c] (0.02-0.70)	0.45[b] (0.10-0.90)	0.72[a] (0.10-1.20)
Dissolved oxygen (mg/L)	7.6[b] (6.09-9.25)	8.54[a,b] (6.90-10.38)	9.52[a] (8.19-11.04)	9.43[a] (3.15-15.66)	6.51[c] (4.10-8.33)	7.48[b,c] (5.41-9.40)
Hardness (mg/L)	624[b] (247-828)	680[b] (550-829)	1664[a,b] (1516-1942)	2716[a] (433-29834)	408[b] (214-551)	610[b] (435-766)
Nitrate (mg/L)	58.4[a] (15.4-91.4)	51.7[a,b] (13.9-69.8)	14.6[c] (3.2-21.9)	23.1[c] (0.2-87.0)	43.4[b] (0.6-104.0)	51.3[a,b] (23.0-72.5)
Percent sand	23[c] (0-100)	22[c,d] (0-65)	34[c] (0-95)	64[b] (0-100)	4[d] (0-30)	94[a] (80-100)
Pfankuch index	91[a] (63-120)	85[a] (57-123)	92[a] (47-127)	82[a] (39- 131)	54[b] (38-109)	94[a] (65-139)
pH (S.U.)	7.5[b,c] (6.8-8.8)	8.1[a] (7.1- 8.7)	8.1[a] (7.7- 8.5)	8.0[a] (7.0- 9.3)	7.2[c] (6.1-8.8)	7.9[a,b] (7.4-8.3)
Substrate index	5.4[b] (3.0-8.0)	5.4[b] (3.6-6.8)	5.0[b,c] (3.1-8.0)	4.5[c,d] (3.0- 8.0)	6.4[a] (4.5-8.0)	3.2[d] (3.0-3.8)
Temperature	22.0[a]	22.7[a]	13.7[c]	20.6[b]	22.9[a]	20.2[b]

(°C)	(19.1-25.1)	(20.0-25.6)	(8.9-17.9)	(12.1-28.8)	(18.3-26.2)	(15.7-24.6)
Total phosphate (mg/L)	0.30[a] (0.04-1.43)	0.29[a] (0.03-0.75)	0.09[b] (0.003-0.32)	0.16[b] (0.003-0.57)	0.21[b] (0.004-1.03)	0.20[a,b] (0.02-0.41)
Turbidity (NTU)	14.5[a] (0.5-112.0)	17.0[a] (2.0-78.8)	3.2[b,c] (0.5-15.8)	5.0[b] (0.7-44.5)	1.2[c] (0.3-3.4)	24.5[a] (4.6-95.7)
Velocity (m/S)	0.54[a,b] (0.00-1.66)	0.70[a] (0.30-1.47)	0.32[b,c] (0.13-0.62)	0.22[c] (0.00-1.18)	0.51[b] (0.15-1.18)	0.02[c] (0.00-0.09)
Width (m)	11[c] (3-38)	63[a] (17-108)	9[c,d] (2-32)	5[d] (1-25)	5[d] (2- 10)	42[b] (18-100)

Table 3. 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	**0.3219**	-0.2820	0.2147	0.2069
Conductivity	**0.6910**	-0.2810	0.1318	0.1223
Depth	**-0.4255**	0.3603	0.1716	0.0209
Dissolved oxygen	-0.0056	**-0.2876**	0.2604	0.3320
Pfankuch index	-0.0251	**-0.3257**	-0.1273	0.1275
Percent sand	**0.5148**	-0.1530	0.3093	-0.0237
Temperature	-0.1175	**0.2452**	-0.0679	-0.2132
Total phosphate	**-0.3794**	-0.0289	0.0894	0.0317
Turbidity	-0.2507	**-0.4021**	0.0046	-0.1636
Width	**-0.6035**	-0.3325	0.0561	-0.3267

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.

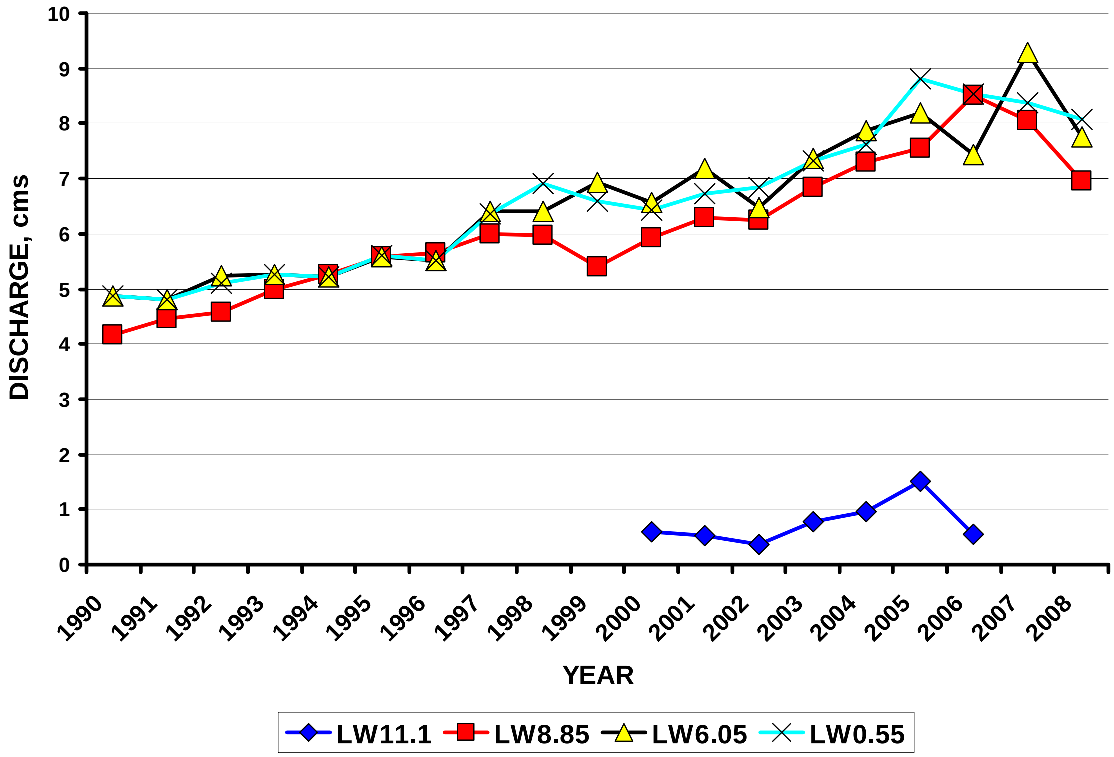


Figure 2. Biplot from data collected in March/April 2001-2010 based on a canonical correspondence analysis (CCA) of sites with respect to environmental variables. Environmental variables were related to community attributes as shown by arrows. Site samples are represented by geometric shapes as shown in the legend.

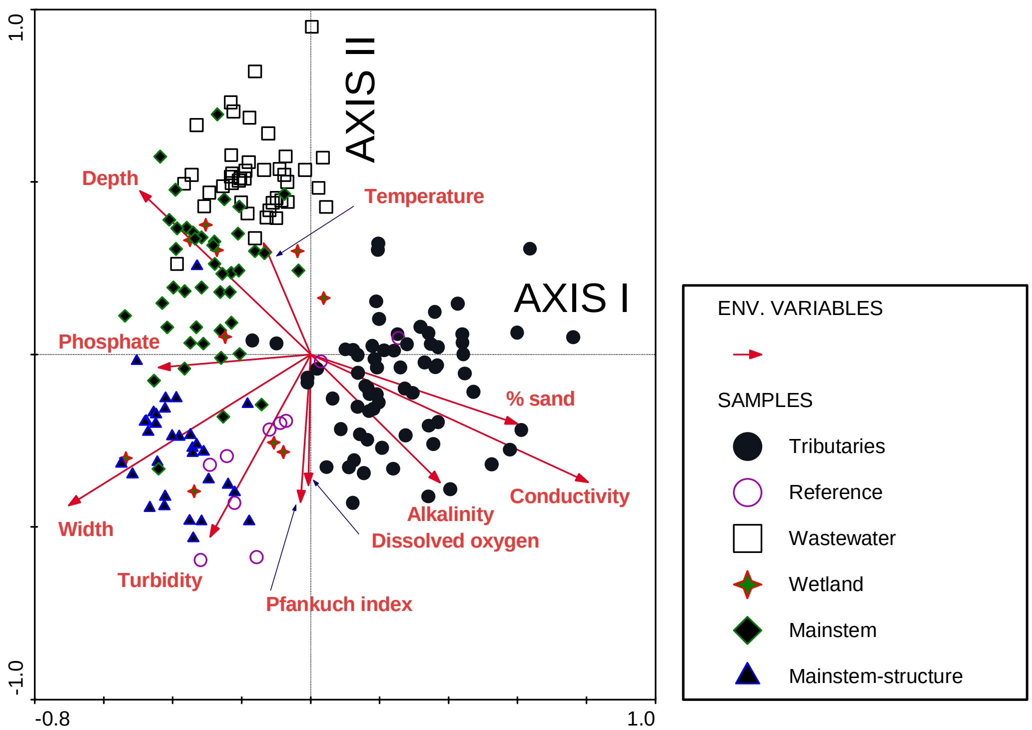


Figure 3. Biplot from data collected in March/April 2001-2010 based on a canonical correspondence analysis (CCA) of macroinvertebrate taxa in association with environmental variables. Only those species that had a fit and weight ≥ 1% are shown in the figure.

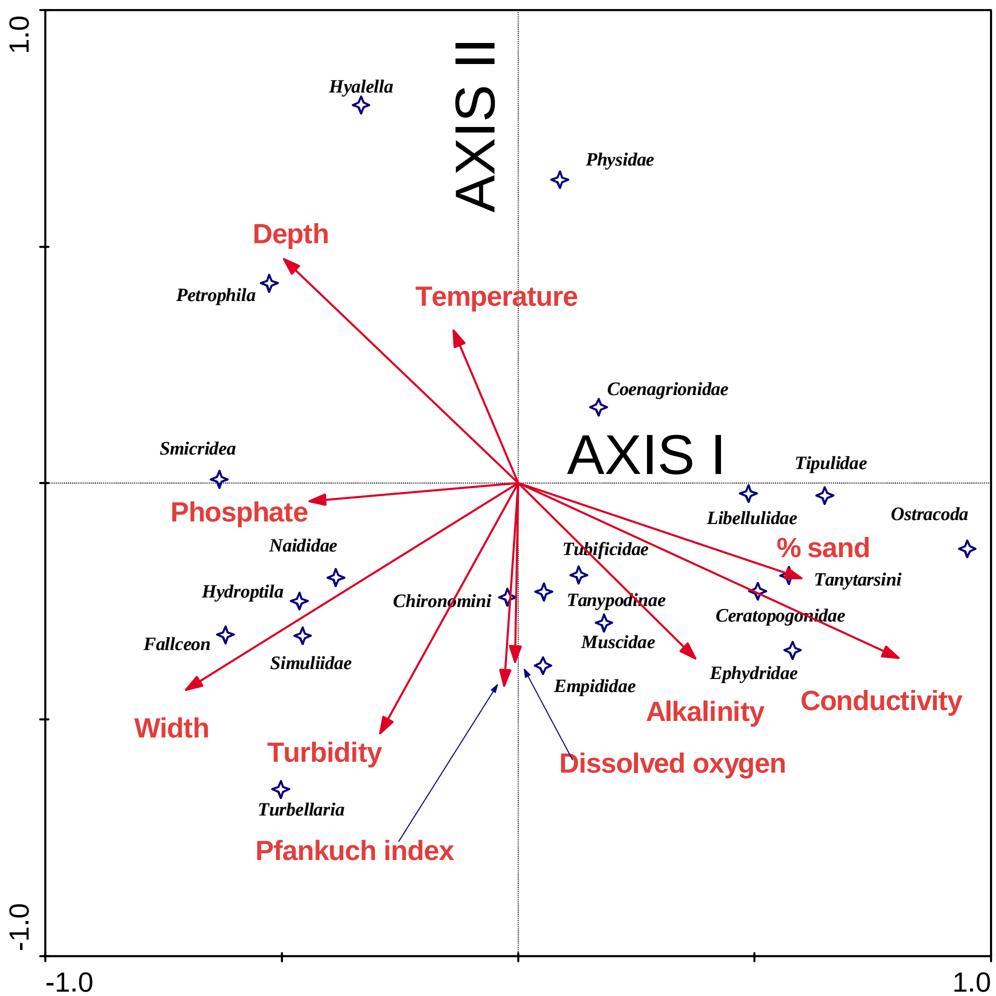


Figure 4. Simuliidae abundance at sites without (mainstem) and with erosion control structures (structure) in Las Vegas Wash. Data is also presented on the other habitat types in the area of the Wash. Error bars indicate standard error from mean values.

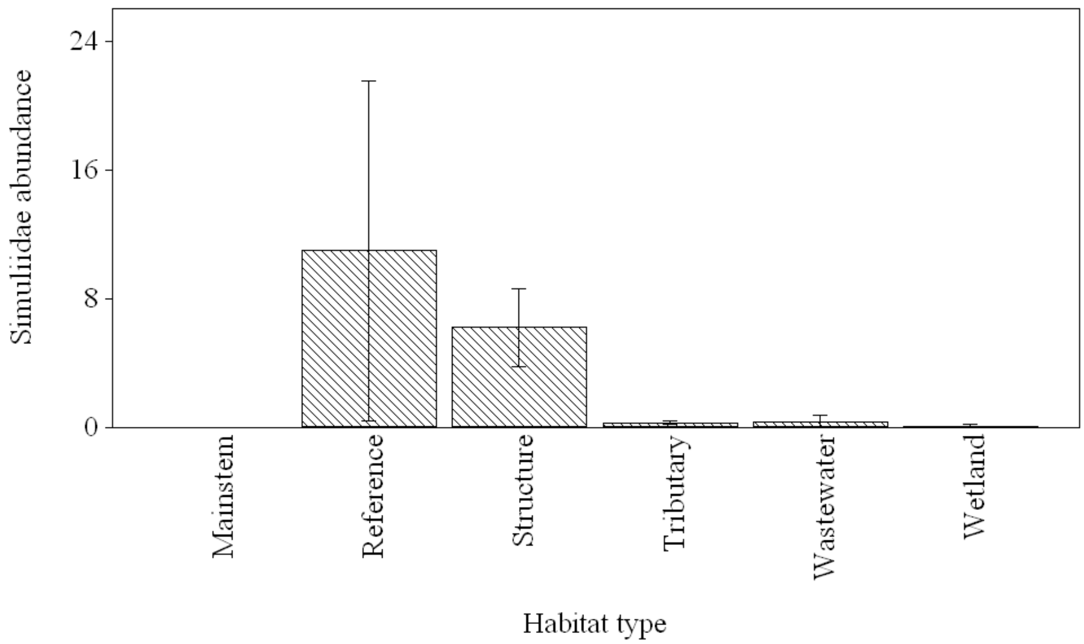


Figure 5. Taxa richness at Las Vegas Wash sites with structures (a) and at sites without structures (b) over time. Correlation analyses indicated that richness at structures was significantly correlated with yearly sampling (r=0.5918, p=0.0005) but there was no significant relationship at sites without structures (r=0.0124, p=0.0349).

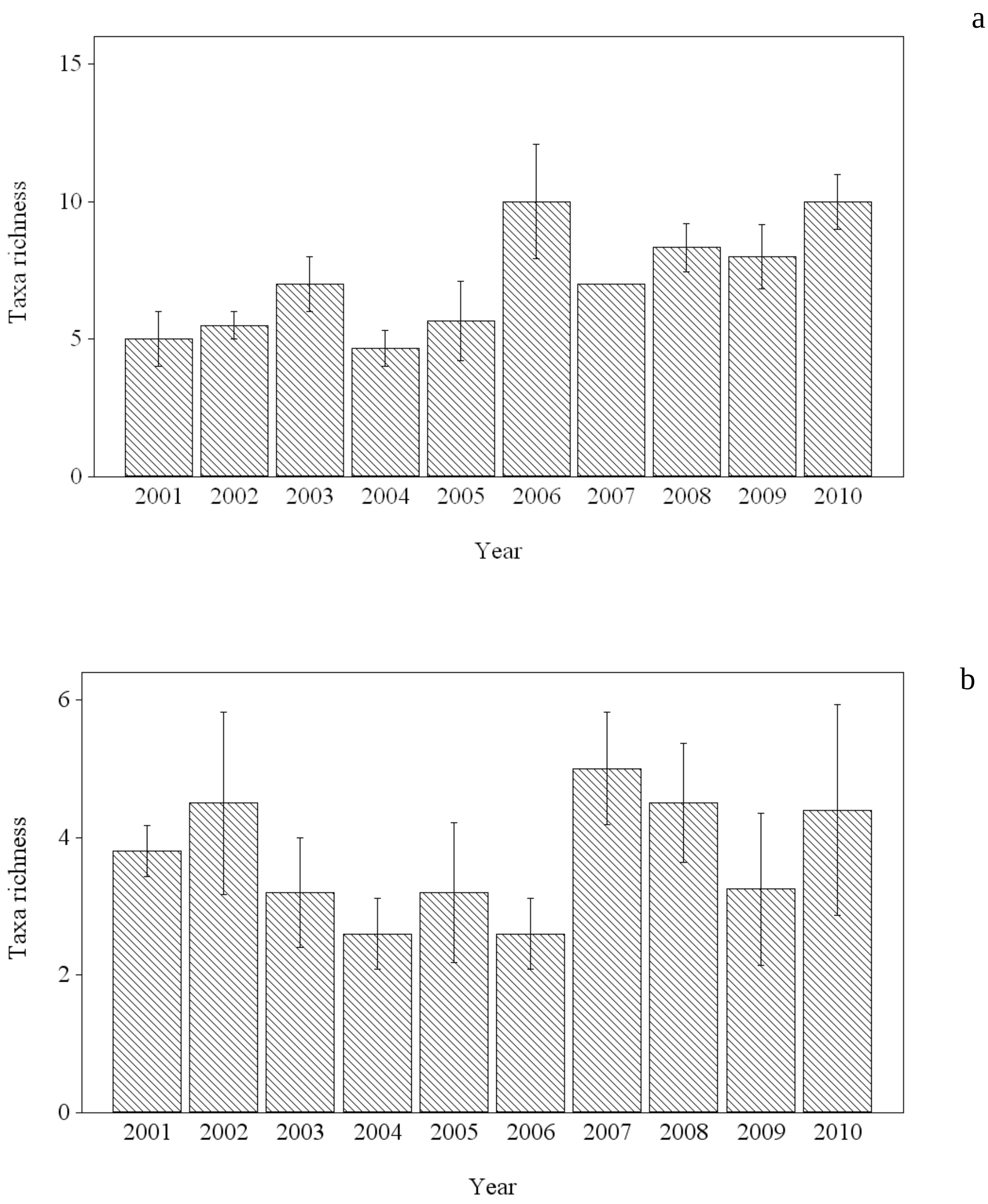


Figure 6. Turbidity measurements at Las Vegas Wash sites for a given sampling year.

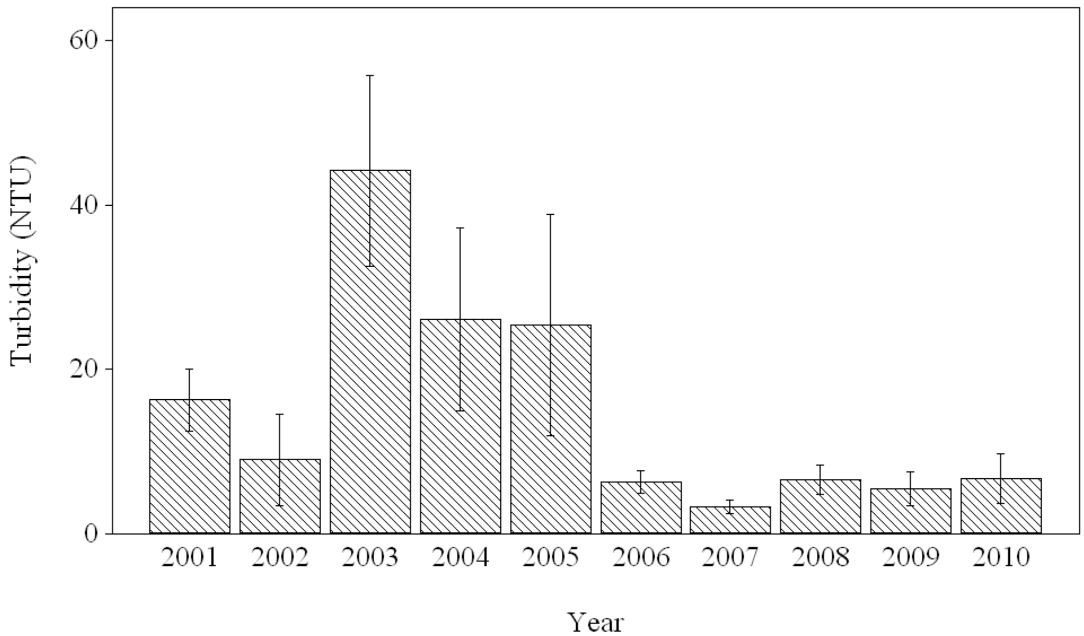


Figure 7. Periphyton biomass based on seasonal collections at Las Vegas Wash.

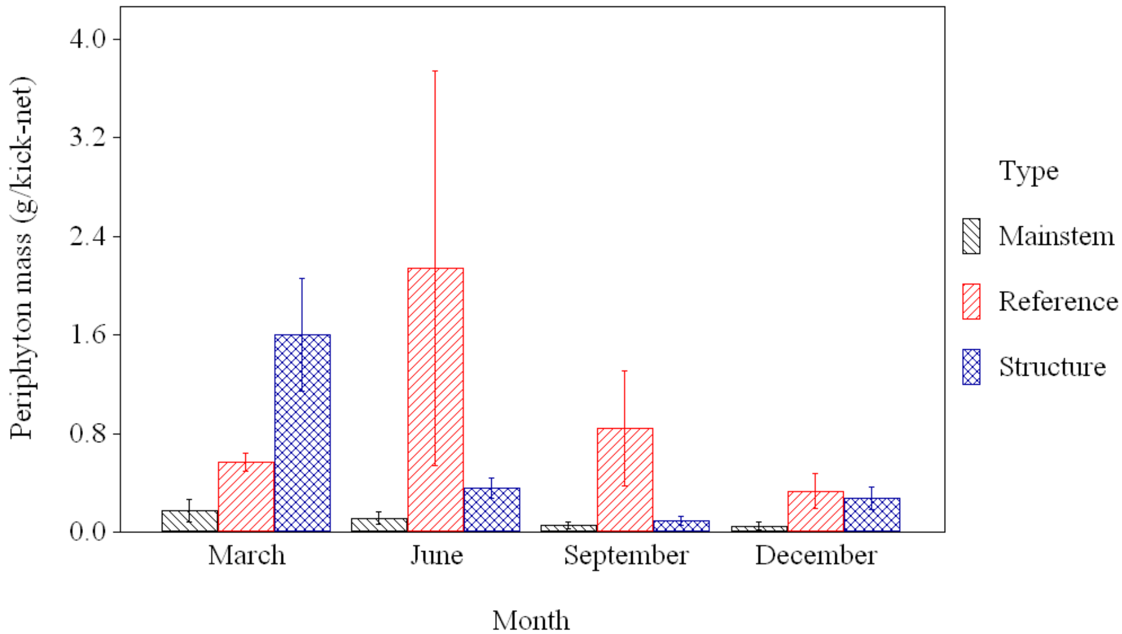


Figure 8. Taxa richness (a) and invertebrate abundance (b) from seasonally collected data in Las Vegas Wash. Taxa richness and invertebrate abundance did not differ with season but did vary with habitat type ($p<0.0000$). Both were significantly lower at mainstem habitats relative to either reference or mainstem sites that contained structure.

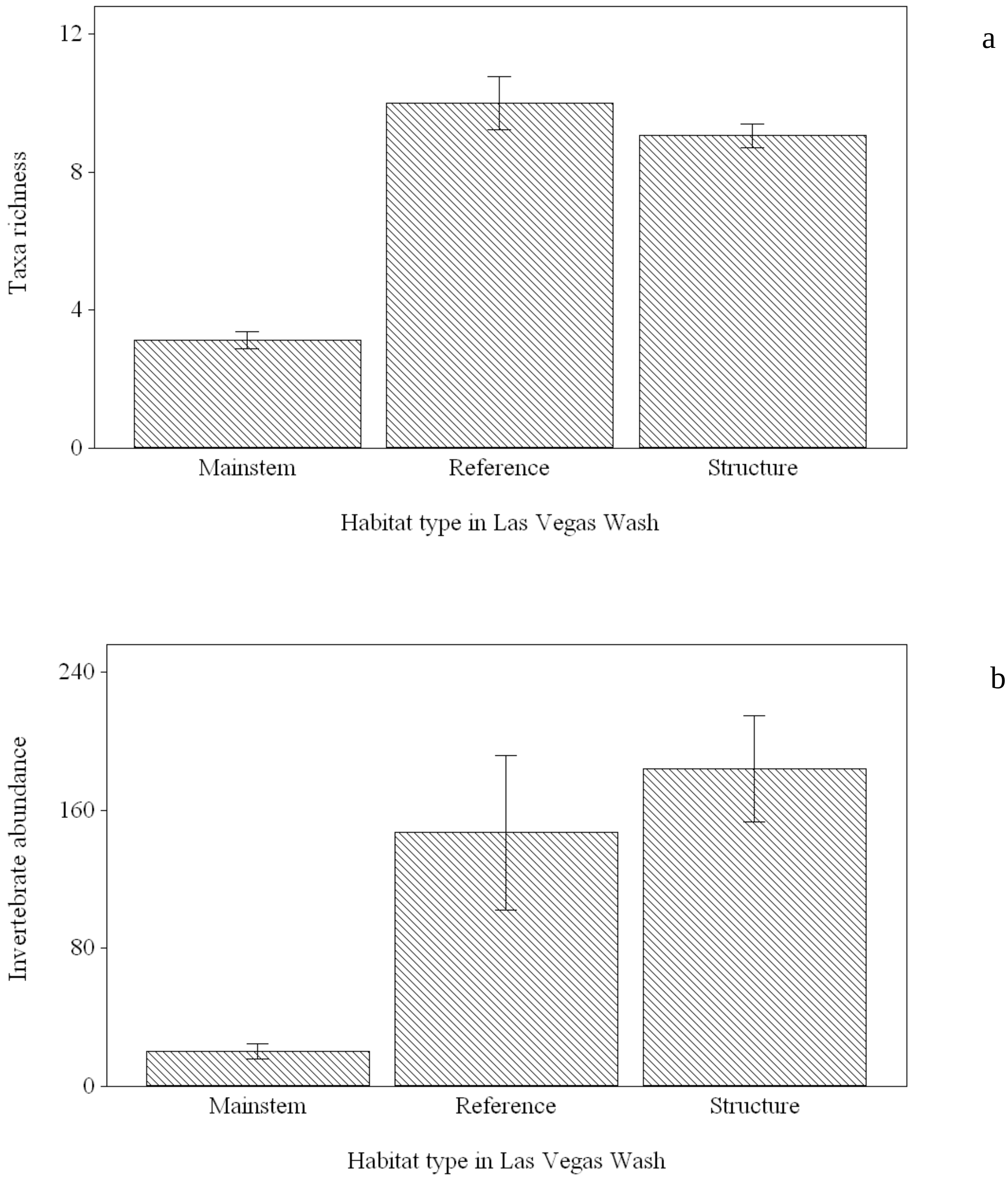


Figure 9. Coarse particulate organic matter (CPOM) from seasonally collected data in Las Vegas Wash. CPOM did not differ with season but did vary with habitat type ($p<0.0000$). CPOM was significantly lower at mainstem habitats relative to either reference or mainstem sites that contained structure.

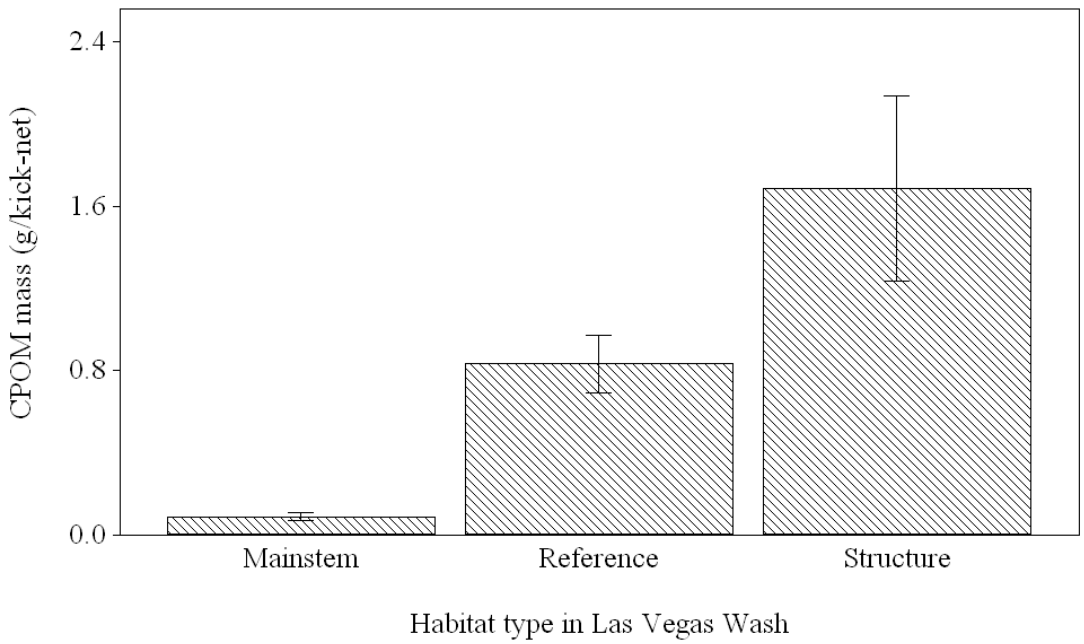


Figure 10. Biplot from seasonally collected data based on a detrended correspondence analysis (DCA) of benthic communites. Months of sample collection are represented by geometric shapes as shown in the legend.

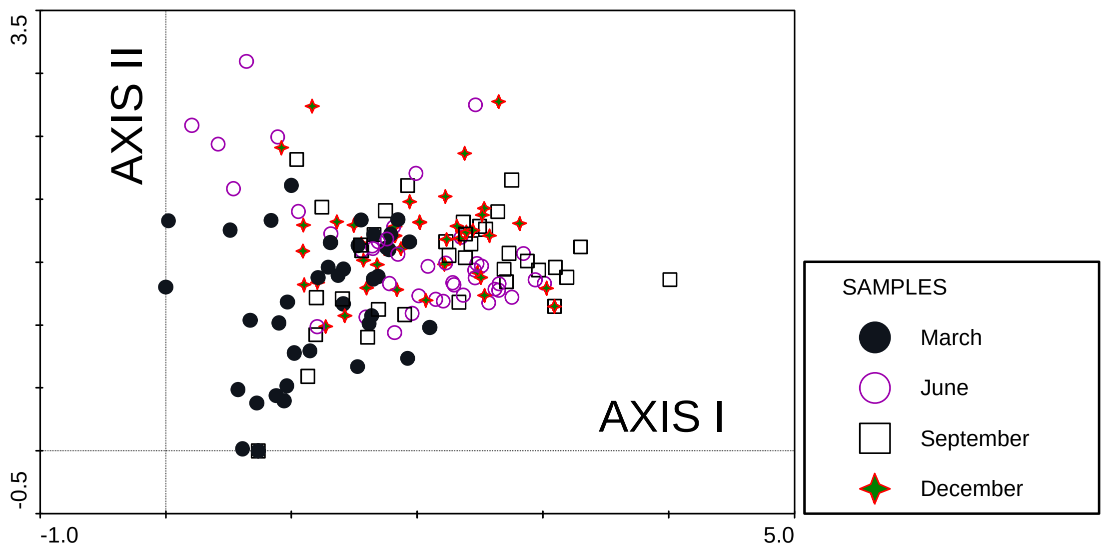
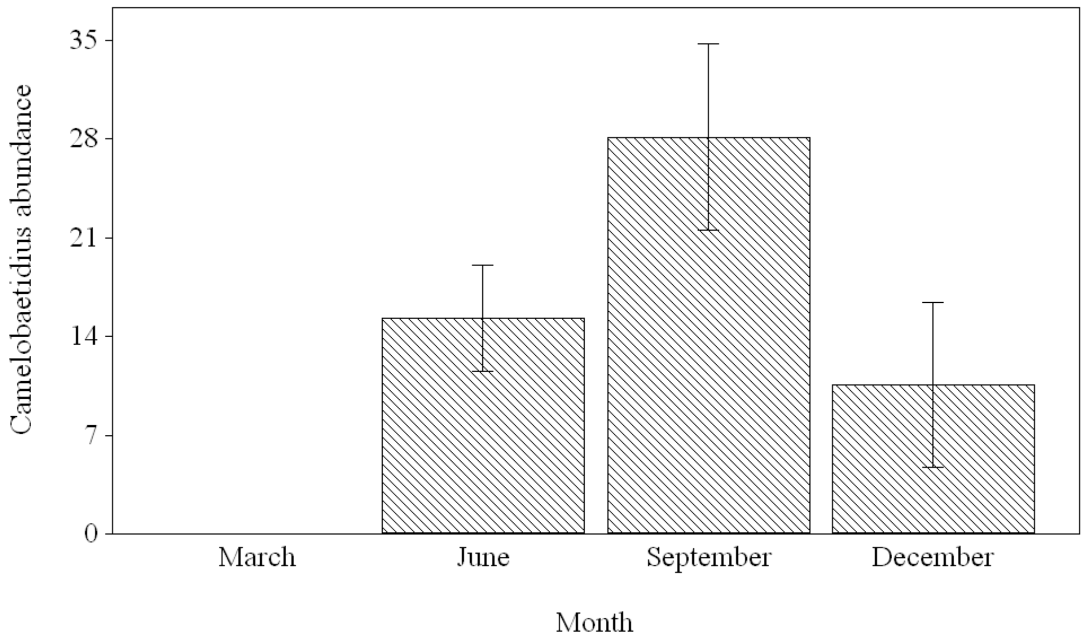


Figure 11. *Camelobaetidius* abundance by season in Las Vegas Wash.



PEER REVIEW DOCUMENTATION

PROJECT AND DOCUMENT INFORMATION

Project Name Aquatic Ecology WOID AF480

Document Stream Macroinvertebrate Assemblages Associated with the Las Vegas Wash Watershed 2000-2010

Document Date January 2011

Team Leader S.Mark Nelson

Document Author(s)/Preparer(s) S. Mark Nelson

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 31 Jan 11

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.

Team Member: S. Mark Nelson Date:
Signature 1-31-11