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PRIMARY RESEARCH PAPER

Response of stream macroinvertebrate assemblages to erosion control structures in a wastewater dominated urban stream in the southwestern U.S.

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Abstract Effects of stream erosion control structures on aquatic macroinvertebrates were studied (2000–2009) in a wastewater dominated drainage (Wash) in Las Vegas, Nevada. Mainstem sites with and without structures, wastewater treatment plant outfalls, a reference site above treatment plant inputs, and tributary sites were sampled. Ordination 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. Assemblages differed between site types, with midges and damselflies important at tributary sites and *Fallceon* mayflies and *Smicridea* caddisflies common at erosion control structures. 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. Structures appeared important in retaining organic matter and, among mainstem sites, coarse particulate organic matter was highest, but variable, at structures and at wetlands above the structures. Erosion control structures, coupled with warm effluent, high baseflows, and altered water quality resulted in development of a macroinvertebrate community that did not trend towards reference or tributary sites. In this case, ecological communities at structures used for river restoration were not on a continuum between disturbed and reference sites. Goal setting of community responses at these structures would have required insight beyond the simple use of reference site attributes.

Keywords Erosion control structures · Las Vegas Wash · Macroinvertebrates · Stream restoration · Thiaridae · Urban

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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 nineteenth 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 1930s and 1940s when groundwater was the basic water resource, wastewater treatment plants were built and began to discharge effluent into the Wash. By the early 1940s water managers were expressing concerns with limited supplies (SNWA, 2006) and 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 1950s 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 (Fig. 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

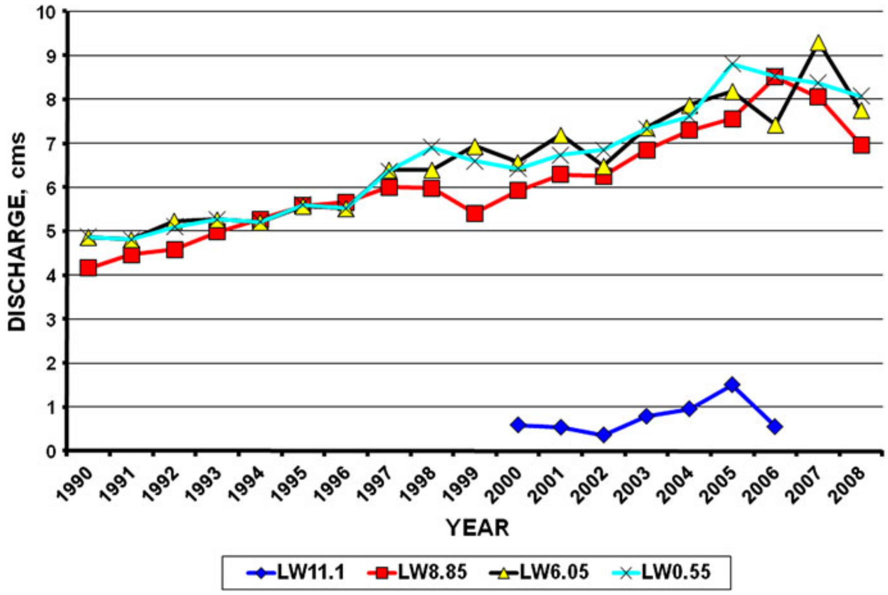

Fig. 1 Mean annual (USGS water year) discharge at several mainstem Las Vegas Wash sites with LW0.55 the site furthest downstream. LW11.1 is above the influence of wastewater treatment plants which discharge additional water into the Wash. Only a portion of the discharge record is available for LW11.1

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. 1975–1999; Buckingham & 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 LW6.05, LW5.5, and LW3.85 (Fig. 2). Erosion control structures placed along the Wash are permanent, low height dams or weirs designed and engineered to endure and help dissipate 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 hydrologically 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 instream habitat restoration and include boulder and weir additions along with channel changes caused by cross-stream structures. Addition of large woody debris is a common restoration technique (Miller et al., 2010) which was not actively pursued in the Wash; however, woody material was added incidently by shedding from successful 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 (e.g., Fig. 2). 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 between biota and erosion control

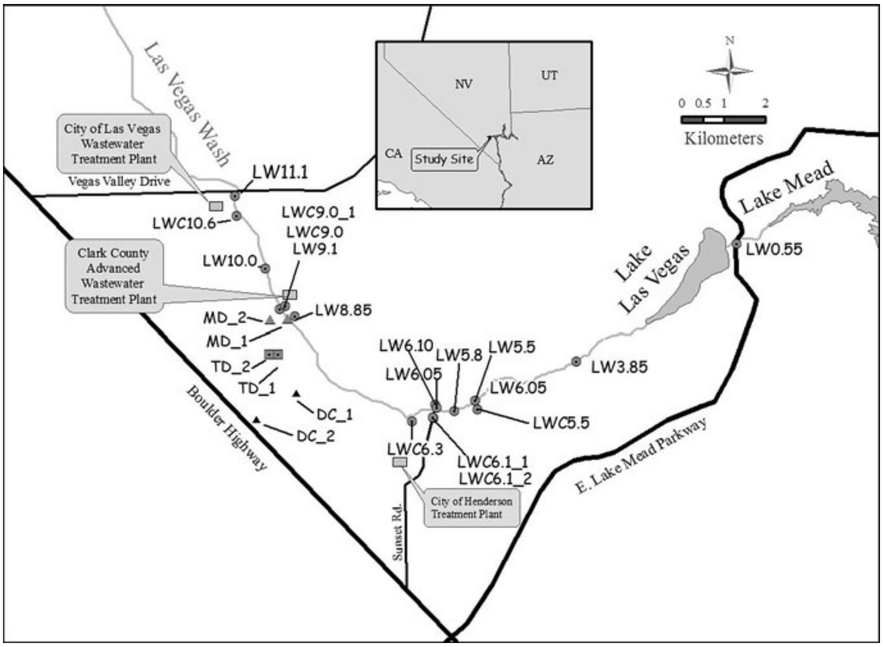

Fig. 2 Macroinvertebrate sampling sites associated with the Las Vegas Wash

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 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.

Sampling was initiated 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. This contrasts with other sites in the Wash that lacked erosion control structures or the furthest upstream site that was not influenced by wastewater treatment plants. This furthest upstream site was considered a reference or benchmark site that represents the "best of what's left" (Hawkins et al., 2010). Initially in 2000 only seven 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 nine Wash sites (with and without erosion control structures) and 11 tributary and wastewater discharge sites (Fig. 2, also see Table 1). 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). This tributary site was added in 2005, but was no longer available for sampling by 2007, due to 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 baseline communities that may not be present in the main channel of

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
LW11.1	LV Wash below Vegas Valley Drive-furthest upstream site	Reference
LWC10.6	Discharge channel from the City of Las Vegas Wastewater Treatment Plant (CLVWTP)	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
LWC9.0_1	New discharge channel from CCAWTP	Wastewater
LW8.85	LV Wash	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

the 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. Site types included: mainstem, mainstem-structure, reference, tributary, wastewater, and wetland environments (e.g., Table 1). Environmental variable sampling was initiated in 2001.

Chemical, physical, and biological methods

Yearly sampling with the full set of variables took place in April 2001–2002, March 2003–2006, and April 2007–2009. A 1-min kick method with a D-frame net (700–800 μ mesh) was used for sampling benthic invertebrates along a ca. 10-m reach at each sampling site. Samples were preserved in 70% propanol. In the laboratory, samples were washed in a 600-μ mesh sieve to remove alcohol, invertebrates were picked from the substrate with the aid of an illuminated $\times 10$ magnifier, and then the entire sample was enumerated and identified under a binocular dissecting scope. Insect taxa were mostly identified to genus while 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°C for 48 h 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, 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 & Richardson, 1990) using the formula S.I. = 0.08* %bedrock + 0.07* %boulde + 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.

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 characterize invertebrate assemblages. Tolerance values obtained from Barbour et al. (1999) were often mean values derived from regional tolerance values.

Repeated measures ANOVA followed by Tukey's test for comparisons were used to compare means of biotic and abiotic variables at different types of environments repeated over time. 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). Experimental sites were not assigned to treatments (mainstem, mainstem-structure, reference, tributary, wastewater, and wetland) and were not randomly interspersed throughout the region being considered. Because landscape treatments were not assigned, these comparisons may be detecting something besides a difference in habitat and may be biased in a way that limits inferences. "Replicates" used in the site-type analysis were from different years and different sites. In some cases a site could be in different categories in different years (e.g., a mainstem site 1 year and then a mainstem-structure site the next). Perhaps the best description of this study is the "quasi-experiment" of Hargrove & Pickering (1992) where some level of pseudoreplication is considered acceptable in exchange for realism. In this particular analysis temporal psuedoreplication also occurs. Repeated measures ANOVA is used to overcome this limitation (e.g., Taylor et al., 1996) and was the purpose of the analysis in the present study.

Trends in the data set were examined with 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 and deep with high stream velocities) and show no significant correlation with sampling occasion.

Stepwise multiple regression (SMR) (forward selection) was used to determine where 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 of discharge data in ordination 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 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 to 100 m and depths from 0.02 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 from all other sites (Table 2). Nitrate concentrations were highest at wastewater outfall sites and sites downstream of wastewater treatment facilities (Table 2). Other water quality parameters 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). Most physical attributes in the Wash did not significantly vary from year to year; however, some water quality parameters did (Table 2). In some cases, this may have been from changes in discharge, and ammonia and nitrate were significantly negatively correlated with discharge ($P < 0.01$, Wash sites only). 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.

Table 2 Invertebrate metrics and environmental variables associated with types of sites along the Las Vegas Wash

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^{b} (63–334)	114^{c} (75–215)	128^{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^{b,c} (0.15–1.20)	0.36^{b,c} (0.05–0.50)	0.25^{c} (0.02–0.70)	0.45^{b} (0.10–0.90)	0.69^{a} (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^{a,b,c} (5.41–9.40)
Hardness (mg/l)	630^{c} (247–828)	682^{c} (550–829)	1678^{b} (1516–1942)	2756^{a} (433–29834)	410^{d} (214–551)	602^{c} (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^{a,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^{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} (13.9–69.8)	14.7^{d} (3.2–21.9)	23.3^{c} (0.2–87.0)	43.7^{b} (0.6–104.0)	48.9^{a,b} (23.0–65.7)
Percent sand	23^{b} (0–100)	22^{b} (0–65)	34^{b} (0–95)	64^{a} (0–100)	4^{c} (0–30)	94^{d} (80–100)
Periphyton (g)	0.12^{b,c} (0.00–1.92) n = 25	1.22^{a,b} (0.00–6.71) n = 28	0.58^{a,b,c} (0.08–1.17) n = 7	2.05^{a,b} (0.00–21.75) n = 47	5.65^{a} (0.01–82.12) n = 34	0.08^{b,c} (0.00–0.68) n = 9
Pfankuch index	92^{a} (63–120)	85^{a,b} (57–123)	89^{a,b} (47–127)	83^{b} (39–131)	53^{c} (38–109)	98^{a} (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^{b} (3.0–8.0)	5.4^{b} (3.6–6.8)	5.0^{b,c} (3.1–8.0)	4.4^{c} (3.0–8.0)	6.5^{a} (4.5–8.0)	3.2^{d} (3.0–3.8)
Temperature* (°C)	22.2^{a,b} (19.9–25.1)	22.9^{a} (20.7–25.6)	14.2^{c} (10.1–17.9)	20.9^{b} (12.1–28.8)	22.9^{a} (18.3–26.2)	20.2^{a,b} (15.7–24.6)
Total phosphate* (mg/l)	0.30^{a} (0.04–1.43)	0.29^{a} (0.03–0.75)	0.08^{a} (0.003–0.32)	0.14^{a} (0.003–0.57)	0.21^{a} (0.004–1.03)	0.19^{a} (0.02–0.41)
Turbidity* (NTU)	15.2^{a,b} (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^{b} (0.00–1.31)	0.65^{a} (0.30–1.00)	0.30^{c} (0.13–0.62)	0.21^{c} (0.00–1.18)	0.51^{b} (0.15–1.18)	0.02^{d} (0.00–0.09)
Width* (m)	11^{c} (3–38)	63^{a} (17–100)	10^{c,d} (2–32)	5^{e} (1–25)	5^{d,e} (2–10)	40^{b} (18–100)

The number of samples collected for a given type of site is represented as n. Sampling events occurred 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). Variables with an *asterisk* indicate that these variables differed significantly from year to year. The range of values for variables is enclosed in *parentheses* within the table

Invertebrate food resources: CPOM/Periphyton

Tributary and wetland sites had significantly higher amounts of CPOM relative to mainstem and wastewater sites (Table 2). Mean CPOM values were high 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 that differed significantly from mainland and wetland sites (Table 2).

Taxa richness/invertebrate abundance

Sixty-two invertebrate taxa were identified from all samples (Table 3). Organism abundance suggested taxa sensitive ($\leq$ one half the maximum tolerance value) to both pollution and sediment were very uncommon, with only *Culoptila* ($n = 17$ individuals) qualifying in both categories (Table 3). Repeated measures ANOVA indicated a significant difference in macroinvertebrate taxa richness and abundance among site types (Table 2). Invertebrate abundance and taxa richness were higher at tributary, reference, and sites associated with erosion control structures (Table 2). Richness and abundance differed significantly between tributary/mainstem-structure sites and mainstem sites (Table 2). Abundance was also high at wastewater discharge sites, however, taxa richness was low (Table 2). 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 4 and Fig. 3. Sites below the influence of wastewater treatment plants that lacked structures did not have significant correlations with sampling year, while those collected at structures were significant or near significance (Table 4) suggesting an increase in taxa richness over time (Fig. 3). Taxa richness at the reference site LW11.1 was also significantly correlated with sampling occasion (Table 4). 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

Table 3 Taxa and numbers of individuals found in the Las Vegas Wash drainage with associated pollution and sediment tolerance values

Taxa	Total number of individuals	Pollution tolerance value[a]	Sediment tolerance value[b]
Collembola	11	10	108
Ephemeroptera			
Callibaetis	15	8.4	72
Fallceon quilleri	1,486	4	–
Siphlonurus	17	5.5	72
Odonata			
Aeshnidae	8	3	72
Calopterydidae	1	5	–
Coenagrionidae	127	8	108
Gomphidae	6	1	108
Libellulidae	38	9	72
Trichoptera			
Culoptila	17	2	32
Hydroptila	459	5.5	108
Smicridea	143	4	72
Lepidoptera			
Petrophila	110	5	72
Hemiptera			
Corixidae	2	8	108
Mesovellidae	4	–	72
Notonectidae	4	–	108
Veliidae	8	–	72
Coleoptera			
Agabetes	27	–	72
Agabinus	64	–	72
Agabus	2	6.5	72
Berosus	34	6.8	72
Enochrus	30	6.7	72
Neoclypeodytes	10	5	–
Optioservus	1	3.7	108
Tropisternus	13	8.3	72
Diptera			
Anopheles	41	7.6	108
Bezzia/Probezzia	45	5.8	108
Brachydeutera	2	–	–
Chironominii	778	6	108
Culex	88	9	108
Culicoides	31	8.8	108
Dasyhelea	5	–	108
Diamesinae	3	2	–
Dolichopodidae	17	5.9	108

Table 3 continued

Taxa	Total number of individuals	Pollution tolerance value[a]	Sediment tolerance value[b]
Empididae	28	5.9	–
Ephydridae	185	6	108
Limnophora	35	7	108
Orthocladiinae	18,375	5	108
Psychodidae	13	10	36
Sciomyzidae	13	6	–
Simuliidae	227	6	108
Stratyomyidae	21	–	108
Tabanidae	1	8	
Tanypodinae	42	7	108
Tanytarsini	1,063	6	108
Tipulidae	37	3	72
Turbellaria	60	4	108
Hirudinea	1	6.7	108
Oligochaeta			
Enchytraeidae	19	10	–
Lumbricidae	3	10	108
Lumbriculidae	12	7.6	–
Naididae	116	5	–
Tubificidae	114	10	108
Nemertea			
Prostoma	16	–	–
Ostracoda	3,927	8	108
Amphipoda			
Crangonyx	1	4	–
Hyalella	844	8	108
Decapoda			
Cambaridae	8	6	108
Gastropoda			
Physidae	8,888	8	108
Lymnaeidae	2	6.3	108
Thiaridae	17	–	–
Pelecypoda			
Corbicula	14	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

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, 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 below the concrete-line section (and just below 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.

Table 4 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/comments
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	Fall of 2002, but flows prior to sampling in 2005 removed most of this structure.
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

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 5). 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 5). Flow effects were

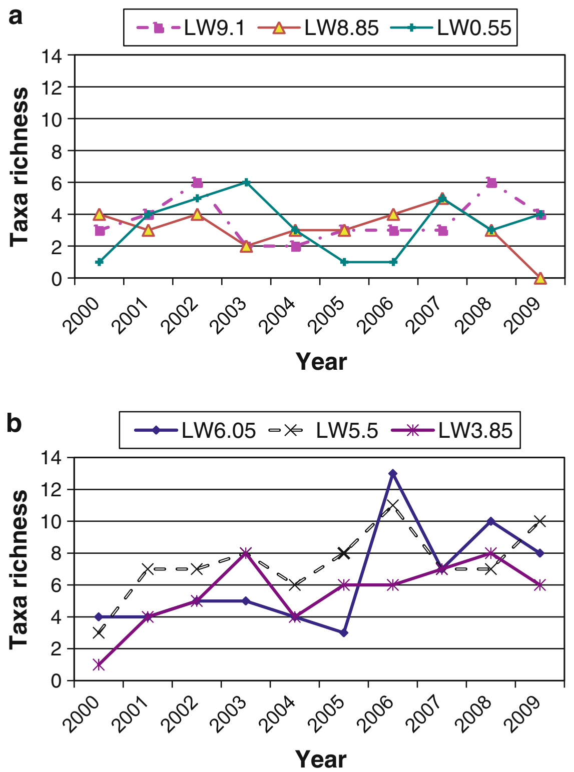

Fig. 3 Taxa richness over time at sites without (**a**) and with (**b**) erosion control structures

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).

Table 5 Results of stepwise multiple regressions for taxa richness. Variables that were not significant in the model included construction disturbance and site

Variable	Taxa richness			
	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			

Multivariate analysis

Results of CCA from the 2001 to 2009 studies (Figs. 4, 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 6). 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 (Fig. 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 (Fig. 4). This also seemed to be the case

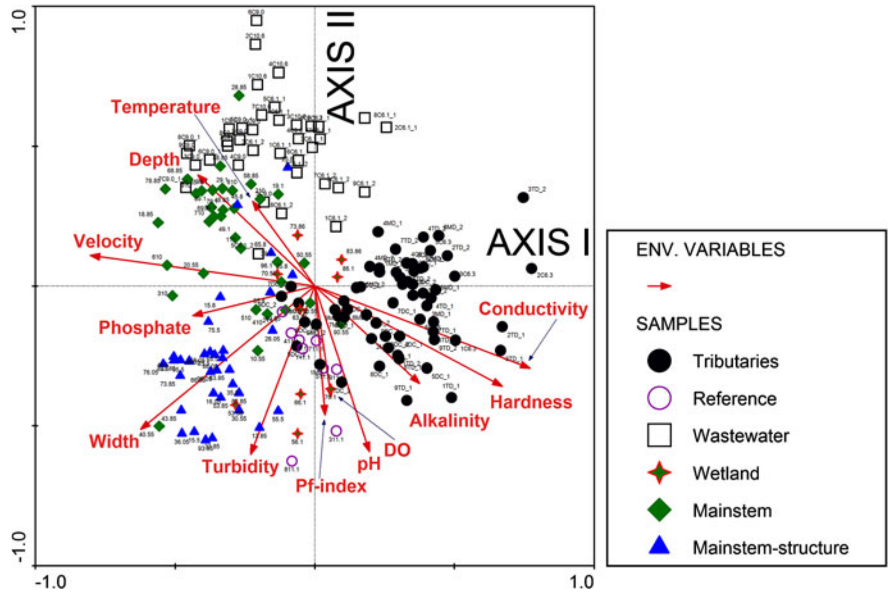

Fig. 4 Biplot from data collected in March/April 2001–2009 based on a canonical correspondence analysis (CCA) of sites with respect to environmental variables. Environmental variables were related to community attributes as shown by arrows. Site samples are represented by geometric shapes as shown in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

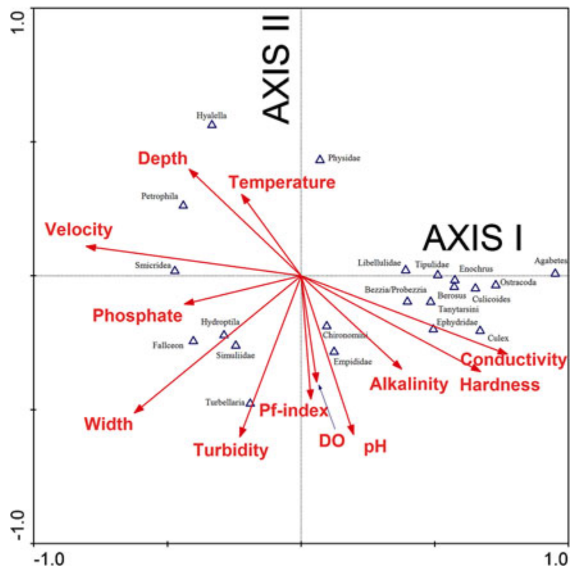

Fig. 5 Biplot from data collected in March/April 2001–2009 based on a canonical correspondence analysis (CCA) of macroinvertebrate taxa in association with environmental variables. Only those species that had a fit >5% are shown in the figure

Table 6 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	2	3	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
Hardness	**0.5868**	−0.2742	0.1323	0.1456
Pfankuch index	0.0318	**−0.3532**	0.0633	0.0214
pH	0.1710	**−0.4541**	−0.0647	0.1517
Phosphate	**−0.3823**	−0.0809	−0.0112	−0.2241
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

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 (Fig. 4; Table 2). Alkalinity and conductivity were higher at tributary sites and the upstream reference site (see Fig. 4; Table 2) and this native water was diluted by high volumes of low conductivity wastewater downstream in the Wash (e.g., Table 2). DO concentrations were lower at wastewater outfall sites receiving water from treatment plants (Fig. 4; Table 2). 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 (Fig. 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, Fig. 5), and *Fallceon* at mainstem-structure sites (to the left on Axis 1, Fig. 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 the abundance of larval *Smicridea* increased at structure sites but not at mainstem sites that lacked these structures (Fig. 6). *Hyalella* was associated with increased depth and temperature along with lower DO and pH, characteristics found at wastewater outfall sites (Fig. 5). Wetland sites contained low numbers of taxa (not shown in Fig. 5) 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 3).

Some taxa, despite multiple sampling years, were only found at structures in the Wash. In 2003 *Corbicula* 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 and it may be that dumping aquariums into the Wash resulted in the introduction of both mollies and tropical thiarid snails. In 2008, more sensitive native taxa like *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.

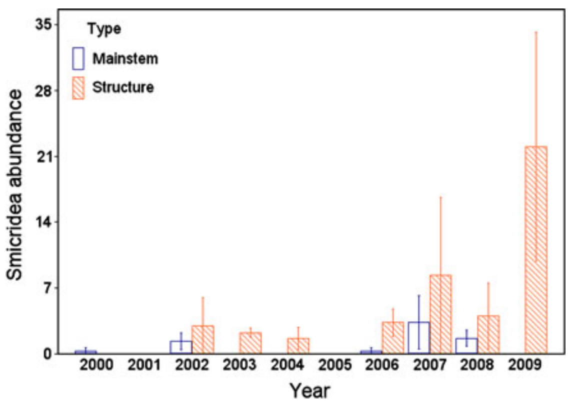

Fig. 6 *Smicridea* abundance at sites without (mainstem) and with erosion control structures in Las Vegas Wash. *Error bars* indicate standard error from mean values

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. For the most part these 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.

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 and beetle 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 (opposite of most urban studies; Brown et al., 2009) 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 & 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, 1972). Taxon-specific thermal 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 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 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 effect 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 these 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 & 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 & 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 terrestrial association (perhaps related to increased humidity needs by adult *Smicridea*?) 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. In a recent study it was found that *Hydropsyche* (a genus in the same family, Hydropsychidae, as *Smicridea*) abundance responded to changes in substrate mobility (Albertson et al., 2010). Increased substrate mobility resulted in decreased *Hydropsyche* abundance. It is likely that structures placed in the Wash would have a positive benefit for hydropyschids through increased substrate stability. Erosion control structures were also important in development of habitat diversity, providing lotic habitat on the structures and slow velocity environments similar to that of tributaries at wetlands forming above structures in the Wash. 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 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 & 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 a 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 (Trush et al., 2000; Walsh et al., 2005). Increasing benthic biodiversity in the Wash may depend to some degree on decreasing the magnitude and frequency of flood events that occur (e.g., Hollis, 1975) on a yearly basis. This may be difficult to achieve since there is already an extensive effort to control flooding (e.g., detention ponds) within the Wash watershed (PBS & J, 2008). It is also likely that the constant, high baseflow velocities in the Wash minimize habitat at unimproved portions of the Wash for stream invertebrates and increased negative effects could occur if the pattern of ever increasing flow continues over time. In the still-urbanizing watershed of the Wash, constant disturbance by floods and ever increasing baseline flows 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 which impact larval stages, but because of its rapid life cycle which allows for the presence of adults during periods of flood disturbance (e.g., Gray, 1981) adults can then recolonize areas 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 floodplain connectivity in terms of area inundated at 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 conservation of these sites is desirable and argues against further degradation. 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. It is unlikely that there would be any political support to stop this process since it has transpired for protection of 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 for maintaining some degree of 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 may 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 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 noted in 2007. Some organisms may move upstream into the Wash from Lake Mead. This sort of transfer for aquatic stages might be inhibited, however, 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*). 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 draws similar conclusions and suggests that when ecosystems are restored that "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.

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. While samples collected from wetlands forming above structures have similarities with reference/tributary sites, colonization trajectories at lotic portions of erosion control structures appear to be taking a more unique path. An adaptive management approach, utilizing baseline macroinvertebrate characteristics as targets/goals, with the understanding of how dynamic the process may be, should be used if changes in Wash operations occur.

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