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Assessing Water Source and Channel Type as Factors Affecting Benthic Macroinvertebrate and Periphyton Assemblages in the Highly Urbanized Santa Ana River Basin, California

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Abstract.—The Santa Ana River basin is the largest stream system in Southern California and includes a densely populated coastal area. Extensive urbanization has altered the geomorphology and hydrology of the streams, adversely affecting aquatic communities. We studied macroinvertebrate and periphyton assemblages in relation to two categorical features of the highly engineered hydrologic system—water source and channel type. Four water sources were identified—natural, urban-impacted groundwater, urban runoff, and treated wastewater. Three channel types were identified—natural, channelized with natural bottom, and concrete-lined. Nineteen sites, covering the range of these two categorical features, were sampled in summer 2000. To minimize the effects of different substrate types among sites, artificial substrates were used for assessing macroinvertebrate and periphyton assemblages. Physical and chemical variables and metrics calculated from macroinvertebrate and periphyton assemblage data were compared among water sources and channel types using analysis of variance and multiple comparison tests. Macroinvertebrate metrics exhibiting significant ($P < 0.05$) differences between water sources included taxa and Ephemeroptera-Plecoptera-Trichoptera richness, relative richness and abundance of nonchironomid dipterans, orthoclads, oligochaetes, and some functional-feeding groups such as parasites and shredders. Periphyton metrics showing significant differences between water sources included blue-green algae biovolume and relative abundance of nitrogen heterotrophic, eutrophic, motile, and pollution-sensitive diatoms. The relative abundance of trichopterans, tanytarsini chironomids, noninsects, and filter feeders, as well as the relative richness and abundance of diatoms, were significantly different between channel types. Most physical variables were related to channel type, whereas chemical variables and some physical variables (e.g., discharge, velocity, and channel width) were related to water source. These associations were reflected in correlations between metrics, chemical variables, and physical variables. Significant improvements in the aquatic ecosystem of the Santa Ana River basin are possible with management actions such as conversion of concrete-lined channels to channelized streams with natural bottoms that can still maintain flood control to protect life and property.

Introduction

Human influence on the environment has been extensive for thousands of years. The introduction of agriculture changed human–environmental relations in virtually all parts of the world (Grimm et al. 2000). However, some of the more severe human-induced environmental impacts are those associated with urbanization. Even in ancient cities, dense human populations caused extreme regional degradation, resulting

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in abandoned cities and reductions in agricultural productivity leading to the collapse of entire civilizations (Grimm et al. 2000).

Currently, urbanization is second only to agriculture as the main cause of stream impairment (Paul and Meyer 2001). Urbanization alters the geomorphic and hydrologic characteristics of streams that drain urban catchments. Drainage basin boundaries are made meaningless by imports and exports of water and construction of drainage and flood control structures. Flow regimes are altered by diversions of water to and from natural channels, water storage and release activities, and changes in surrounding land use that alter runoff patterns (Leopold 1968; Klein 1979; Booth 1991). In addition, flow may be augmented by discharges from wastewater treatment plants. These changes in stream geomorphology, hydrology, and landscape affect water quality. Urban catchments typically have elevated concentrations of phosphorus and other nutrients (U.S. Geological Survey 1999; Winter and Duthie 2000); inorganic ions such as chloride, sodium, and potassium (Paul and Meyer 2001); metals (Paul and Meyer 2001); pesticides (U.S. Geological Survey 1999); and other organic contaminants (Moring and Rose 1997; Frick et al. 1998; Burton 2002).

Changes in landscape and water quality also alter aquatic biotic communities. Taxa richness (number of taxa) declines and communities shift to more tolerant species in association with measures of urbanization including increased population density, increased impervious surface area, and changes in streambed substrate and water quality (Garie and McIntosh 1986; Winter and Duthie 1998; Ourso 2001; Kennen and Ayers 2002). Tolerant species increase in abundance and often dominate the biological community (Barbour et al. 1999; Paul and Meyer 2001).

The Santa Ana River basin is the largest stream system in Southern California, encompassing about 6,900 km^2 of the densely populated coastal area (Figure 1). The river begins in the San Bernardino Mountains (which reach altitudes exceeding 3,000 m) and flows more than 160 km to the Pacific Ocean. The watershed is home to almost 5 million people, and the population is expected to reach almost 7 million by the year 2025 (Santa Ana Watershed Project Authority 2003).

The hydrologic system is highly engineered. Flow from most headwater tributaries is diverted to groundwater-recharge facilities (Figure 1), public supply, and, on the Santa Ana River, to hydroelectric plants. Because of diversions and hot, dry summers, the Santa Ana River and almost all its tributaries lose surface flow once they reach the alluvium-filled basin. Groundwater is withdrawn and supplemented with imported water for public use. Some of the water used by the public returns to the ground through infiltration from lawns, some is lost through evapotranspiration from ornamental plants, some becomes urban runoff, and most is sent to wastewater treatment plants. Farther downstream, flow is reestablished in some channels due to inputs from wastewater treatment plants (Figure 1). In a few small streams, urban runoff or groundwater forced upward by faulting or by bedrock outcrops restores surface flow. Imported water is occasionally discharged to a stream and then diverted for groundwater recharge farther downstream. Most water in streams is from sources impacted by human activities. Treated wastewater and urban runoff maintain perennial flow in some streams that historically were ephemeral. Flow in these streams is 70–100% treated wastewater (Mendez and Belitz 2002). Moreover, the volume of treated wastewater is increasing every year (Burton et al. 1998). In addition to changes in source water, the tributary network on the basin floor has been greatly altered. Natural stream courses are typically channelized and, in many cases, concrete-lined.

Ecological conditions in the Santa Ana River basin have also been significantly altered. Watersheds have been converted from natural habitats to urban land uses and riparian vegetation has been removed. Many streams have been channelized and concrete-lined for the protection of life and property. Human activities have introduced many toxic compounds to surface- and groundwater, rerouted the distribution of streamflow, and changed the timing of these flows. Changes to the landscape and streams imposed by urbanization cannot be fully corrected (Booth and Reinelt 1993); however, that does not mean ecosystem conditions cannot be improved. The problem is identifying differences in biological conditions related to anthropogenic factors that may be mitigated.

Many urban studies focus on an urbanization gradient either within one catchment or among several catchments. Other studies compare reference conditions to urban condition. Because of the engineered hydrologic system, the arid climate, and the high percentage of urban land use in the basin, gradient studies are not feasible in the Santa Ana River basin. However, some aspects of the hydrologic system in the Santa Ana River basin are well constrained. It is possible to identify the source of water as well as the channel modification or type for many streams. This provides an opportunity for assessing the effects of these two anthropogenic factors. Objectives of this

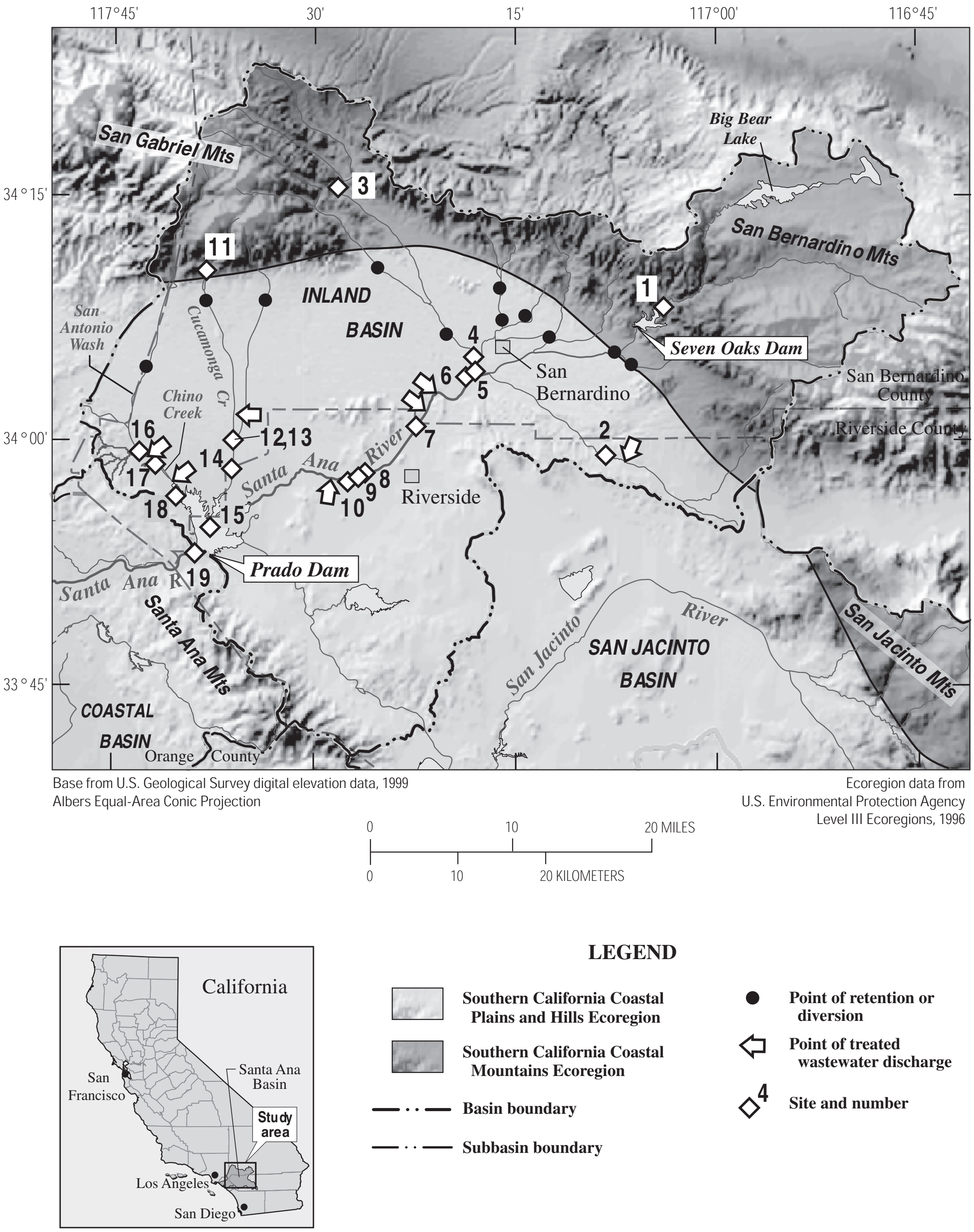

FIGURE 1. Location of stream sites, retention basins, and points of treated-wastewater discharge in the Santa Ana River basin, California.

paper are to relate condition of the benthic macroinvertebrate and periphyton (attached algae) assemblages to water source and channel type. We also address some challenges of studying and understanding highly urbanized streams. In a companion paper (Brown et al. 2005, this volume), we use multivariate ordination techniques to better understand the ecological processes affecting aquatic communities in the Santa Ana River basin and generating the patterns observed in this study.

Methods

Study Area

Urban and agricultural land uses occur primarily in the alluvium-filled valleys and coastal plain, which are located in the Southern California Plains and Hills ecoregion (Omernik 1987) (Figure 1). The uplands, which include the San Gabriel, San Bernardino, and San Jacinto Mountains, lie within the Southern California Coastal Mountain ecoregion and are generally steep and undeveloped. Land use in the entire basin is about 35% urban; 10% agricultural; and 55% open space, primarily steep, mountain slopes. The alluvium-filled valleys and coastal plain are more than 70% urban. Population density in the entire basin is about 580 people/km^2; population density in the alluvium-filled valleys and coastal plains is about 1,160 people/km^2. The climate is Mediterranean and characterized by hot, dry summers and cool, wet winters. Average annual precipitation ranges from 25 to 60 cm in the coastal plains and inland valleys, and from 60 to 122 cm in the San Gabriel and San Bernardino Mountains (U.S. Army Corps of Engineers 1994).

Site Selection

Preliminary assessments identified four categories of water source (natural, human-impacted groundwater, urban runoff, and treated wastewater) and three categories of channel type (natural, channelized with natural bottom, and concrete-lined). Conceptually, this represents a 4 × 3 categorical design, but perennial stream reaches that represented all combinations were not available (Table 1). At least one study reach was selected to represent all combinations available in the basin for a total of 19 sites (Table 2).

TABLE 1. Number of sites in each combination of water source and channel type, Santa Ana River basin, California.

	Channel type		
Water source	Natural	Channelized	Concrete
Natural	3	0	0
Groundwater	0	1	3
Urban runoff	0	2	1
Treated wastewater	4	2	3

Natural landscape variability was minimized by locating reaches within one ecoregion, the Southern California Plains and Hills (Omernik 1987), with one exception. Sites with little to no urban land use could not be found within the ecoregion. Therefore, three sites (1, 3, and 11) located near the interface with the Southern California Mountain ecoregion were selected to represent least-impacted conditions (Table 2; Figure 1). These mountain-runoff sites are supplied with natural water, have a natural channel type, and represent conditions as streams enter the valley floor.

Four sites are primarily supplied with human-impacted groundwater. Of these, one is channelized (9) and the remaining three are concrete-lined (4, 6, and 8). Three sites are primarily supplied with urban runoff. Two of these sites are channelized (5 and 16), and the third is concrete lined (12). Nine sites are dominated by discharge from wastewater treatment plants. Four of these sites are located in natural channels (2, 10, 15, and 19), two in channelized streams (7 and 18), and three in concrete-lined channels (13, 14, and 17).

Basin area and percentage of urban land use were calculated for all sites for the natural, topographical drainage basin. The actual contributing drainage area also was calculated based on information about retention basins, storm drains, and treated wastewater discharge locations (Table 2). Drainage areas behind retention basins were subtracted from the topographical drainage basin, and basin boundaries were adjusted to account for storm drains and treated wastewater discharge locations. The contributing area was usually a subset of the topographical drainage area that is separated from mountain runoff by retention basins. In the case of the three least-impacted sites, the topographical drainage basin area and contributing basin area are the same. Percentage of urban land use also was calculated using the contributing basin area. In the case of a stream receiving 100% treated wastewater, the contributing basin area has no meaning because the flow emerges from a pipe and all of the water is from urban uses (100%).

Data Collection

Water samples for major ions, nutrients, pesticides, and field parameters (specific conductance, pH, water temperature, dissolved oxygen, and discharge) were collected using standard U.S. Geological Survey (USGS) protocols (Shelton 1994). Water samples were collected in Teflon bottles using either the equal-width-incremental method or the multiple-vertical method,

TABLE 2. Site number, site name, drainage area, land use, population density, major source of water, and channel type for study sites, Santa Ana River basin, California.

Site no.	Station name	Drainage area (topographical, km^2)	Drainage area (contributing, km^2)	Urban land use (topographical, %)	Urban land use (contributing %)	Population density (contributing, people per km^2)	Major contributing water source[a]	Channel type[b]
1	Santa Ana River (SAR) at upper powerhouse	398	398	5	5	20	N	N
2	San Timoteo Creek near Eastside Ranch	141	NA[c]	20	100[d]	NA[c]	W	N
3	Cajon Creek below Lone Pine	145	145	4	4	10	N	N
4	Warm Creek above Orangeshow	4	4	90	90	1,990	G	C
5	Warm Creek above E Street	30	30	93	93	1,890	U	Ch
6	Warm Creek near San Bernardino	32	32	94	94	1,890	G	C
7	SAR above Riverside Road	1,918	743	21	46	690	W	Ch
8	Sunnyslope channel near Rubidoux Nature Center	19	19	52	52	870	G	C
9	Sunnyslope channel in SAR Regional Park	19.4	19.4	53	53	870	G	Ch
10	SAR at MWD Crossing	2,136	960	25	49	730	W	N
11	Cucamonga Creek near Upland	26	26	0	0	2	N	N
12	Cucamonga Creek at Chino Ave, main channel	180	132	58	80	1,460	U	C
13	Cucamonga Creek at Chino Ave, wastewater channel	180	NA[c]	65	100[d]	NA[c]	W	C
14	Cucamonga Creek near Mira Loma	208	160	56	72	1,280	W	C
15	Mill Creek near Slatter S Duck ponds	234	186	52	63	1,120	W	N
16	Little Chino Creek above Pipeline	16	16	46	46	560	U	Ch
17	Chino Creek above Central Ave	234	155	55	78	1,680	W	C
18	Chino Creek below Pine Road	259	191	51	69	1,520	W	Ch
19	SAR below Prado Dam	3,726	2,394	32	45	690	W	N

[a] N, natural; W, treated wastewater; G, urban ground water; U, urban runoff.
[b] C, concrete-lined channel, Ch, channelized; N, natural channel.
[c] Contributing area is not applicable (NA) because all flow in the channel comes from a wastewater treatment plant.
[d] Urban land use is 100% because all flow in the channel comes from wastewater treatment plant.

depending on site conditions. Water samples for determination of dissolved nutrients and major ions were filtered through a 0.45-μm Gelman capsule filter. Water samples for determination of cations were preserved with nitric acid. Water samples collected for pesticides were filtered through a baked glass-fiber filter. Water samples for major ions, nutrients, and pesticides were chilled to 4°C and shipped to the USGS National Water-Quality Laboratory (NWQL), Denver, Colorado, for analysis. Samples for major ions, nutrients, and pesticides were collected once, when the algae samples were collected. Field parameters were collected during each site visit.

Habitat variables were measured at each of 11 transects within each sampling reach (Fitzpatrick et al. 1998). Reaches ranged from 150 m for small streams and concrete-lined channels to 900 m for larger streams. Basin area, urban land use, and population densities were determined using geographic information system databases. Land-use information was obtained from the Southern California Association of Governments (1997), and population density information is from the 1990 U.S. Census (Hitt 1994).

Periphyton, benthic macroinvertebrate, and water quality samples were collected and the habitat was characterized at the 19 sites from July to September 2000. Artificial substrates were used for periphyton and benthic macroinvertebrates to decrease the effect of radically different substrate types among sites (cobbles, sand, concrete), facilitating comparisons among sites (Aloi 1990; Lowe and Pan 1996).

Unglazed clay tiles (approximately 7.5 × 7.5 cm) attached to concrete paving blocks were used to collect periphyton. Four paving blocks with two tiles each were placed at each site. Water depth and velocity were measured at each paving block when the substrates were deployed and when they were collected. After a 2-week colonization period, the clay tiles were removed from the paving blocks. Periphyton was collected and processed using the top-rock scrape method (Moulton et al. 2002). Samples were preserved in 4% formalin. Periphyton taxa were identified and enumerated at the Philadelphia National Academy of Science following the methods of Charles et al. (2002).

Artificial substrates for benthic macroinvertebrates consisted of a section of bristled plastic doormat (approximately 15 × 15 cm) and an 18-cm length of 3.2-cm polyvinyl chloride (PVC) pipe wrapped three times with plastic fencing (1.9-cm mesh) attached to a concrete paving block (Figure 2). Four substrates were placed at each site. Water depth and velocity were measured at each paving block when the substrates

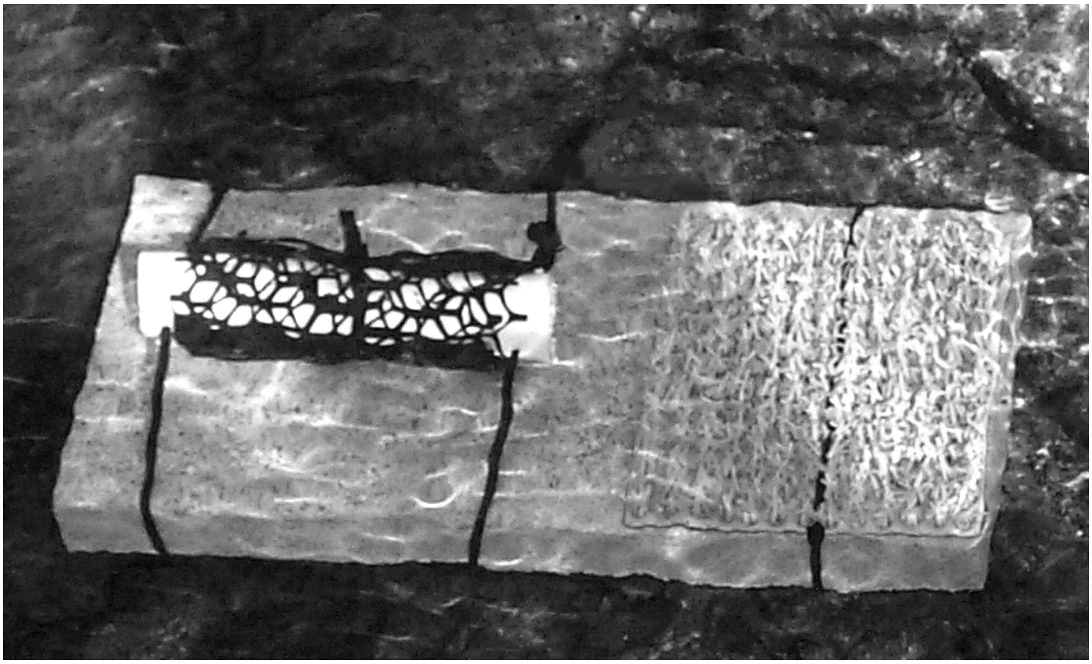

FIGURE 2. Artificial substrate used to collect macroinvertebrate samples from the Santa Ana River basin, California.

were deployed and when they were collected. Up to three substrates were removed after a 6-week colonization period. The doormat and PVC pipe were removed from the paving block, and placed in a bucket. A 500-μm-mesh net was placed downstream from the substrate to collect any macroinvertebrates that were dislodged in the removal process. The doormat, PVC pipe, and fencing were scrubbed and inspected to remove macroinvertebrates. The sample was sieved (500 μm) and preserved in 10% formalin (Moulton et al. 2002). Macroinvertebrates were identified and enumerated at the NWQL following protocols for a 100-organism fixed-count (Moulton et al. 2000).

Data Analysis

More than 80 chemical and 30 physical variables were measured, including concentrations of dissolved nutrients, concentrations of dissolved major ions, concentrations of dissolved pesticides, and habitat characteristics. Analysis of variance (ANOVA) was used on ranked chemical and physical variables to identify those that were significantly different ($P < 0.05$) between water sources or channel types. These selected chemical and physical variables were used in all further analyses. Principal components analysis (PCA) was used to explore overall patterns among the final set of environmental variables.

Benthic macroinvertebrates were generally identified to genus. Ambiguous individuals identified at a higher taxonomic level (usually family) were distributed among the lower taxa (usually genera) in accordance with the relative abundance of each genus when most of the individuals were identified at the lower level. Otherwise, data were aggregated at the higher level of taxonomy. Species lists and counts are not presented in this paper but are available from the corresponding author (Carmen Burton). The taxa data were summarized as biological metrics using the Invertebrate Data Analysis System program (Cuffney 2003).

Periphyton was identified to species in most cases. As with the macroinvertebrates, species lists and counts are not presented in this paper. Metrics were calculated using autecological and tolerance information described by Van Dam et al. (1994), Bahls (1993), and Lange-Bertalot (1979).

More than 120 macroinvertebrate metrics and 40 periphyton metrics were calculated. Macroinvertebrate metrics included taxa richness, relative taxa richness, taxa abundance (density), and relative taxa abundance, functional-feeding group (taxa having similar adaptations for feeding) richness and abundance, and relative richness and abundance. Periphyton metrics include taxa richness, relative richness, abundance, relative abundance, biovolume, and percentage of biovolume of periphyton classes, as well as relative abundance of taxa according to autecological characteristics such as motility and pollution tolerance. Only those metrics that were statistically different among water sources or channel types are discussed further.

Biological metric values were compared among water sources or channel types using ANOVA. The data were transformed by ranking before using the general linear model procedure in SAS. Tukey's multiple comparisons tests (Helsel and Hirsch 1992) were conducted for biological metrics showing significant differences among water sources or channel types. The test was modified using the harmonic mean to account for unequal sample sizes (Helsel and Hirsch 1992). Significant metrics ($P < 0.05$) were correlated (Spearman's correlation) with the selected chemical and physical variables. The significant macroinvertebrate metrics and the significant periphyton metrics also were analyzed by PCA to determine patterns of correlation among metrics within each taxa group and facilitate interpretation of general patterns. The number of important axes was determined by Kaiser's rule, which states that the minimum eigenvalue for an axis should be greater than the average of the eigenvalues for all axes.

Results

Chemical and Physical Variables

Nineteen chemical and physical variables were significantly different between sources of water or channel type (ANOVA, $P < 0.05$). Fifteen chemical and physical variables showed significant differences between sources of water, and nine physical variables showed significant differences between channel types (Tables 3 and 4). Concentrations of dissolved chemicals were highest in streams with treated wastewater and lowest in streams with natural water (Table 3). Inorganic ions such as chloride (Cl), potassium (K), and sodium (Na) were significantly different among water sources, but specific conductance was not. Of the physical variables, discharge, water velocity, and wetted channel width were highest in streams supplied with treated wastewater and usually lowest in natural streams (Table 3). Streams supplied with water from natural sources had the highest values for the remaining physical variables. Streams supplied with urban runoff and groundwater had the highest percentages of urban land use.

TABLE 3. Median values for chemical and physical variables that were significantly different among water-source categories (ANOVA, $P < 0.05$). For each variable, water sources with different letters were significantly different (Tukey tests, $P < 0.05$).

Variable	Water source: Natural	Groundwater	Urban runoff	Treated wastewater
Cl (mg/L)	6 A	7 A	61 AB	94 B
K (mg/L)	2 A	3 A	4 A	11 B
Na (mg/L)	19 A	71 AB	64 AB	82 B
Ammonia (mg/L as N)	0.01 A	0.02 AB	0.04 AB	0.09 B
PO_4 (mg/L as P)	0.005 A	0.005 A	0.087 B	1.05 C
Discharge (m^3/s)	0.03 A	0.07 A	0.06 A	0.88 B
cv[a] bank-full width	33 A	5 B	14 AB	14 B
Channel width, wetted (m)	3.8 AB	4.4 A	5.7 AB	14.1 B
cv depth	61 A	52 B	49 AB	36 B
% riffle	53 A	0 B	0 B	0 B
Streambed substrate size[b]	7.0 A	1.7 B	2.5 B	3.1 AB
Velocity (m/s)	0.18 A	0.12 A	0.21 AB	0.47 B
cv velocity	101 AB	98 AB	102 A	44 B
% urban land use (contributing)	4 A	71 B	80 B	69 AB
Water temperature (°C)	20.0 A	29.8 B	30.5 B	28.0 B

[a] cv, coefficient of variation.
[b] The streambed substrate size was characterized as 1, concrete; 2, silt, mud, or detritus; 3, sand (>0.063–2 mm); 4, fine/medium gravel (>2–16 mm); 5, coarse gravel (>16–32 mm); 6, very coarse gravel (>32–64 mm); 7, small cobble (>64–128 mm); 8, large cobble (>128–256 mm); 9, small boulder (>256–512 mm); 10, large boulder, irregular bedrock, irregular hardpan, or irregular artificial surface (Fitzpatrick et al. 1998).

Water temperature, open canopy, and percentage of urban land use were highest in concrete-lined channels and lowest in natural channels (Table 4). Bank shading, coefficient of variation (cv) of bank-full width, cv of open canopy, presence of riffles, and streambed substrate size were lowest in concrete-lined channels and highest in natural channels. Values for channelized streams fell between the values for concrete-lined and

TABLE 4. Median values for chemical and physical variables that were significantly different among channel-type categories (ANOVA, $P < 0.05$). For each variable, channel types with different letters were significantly different (Tukey tests, $P < 0.05$).

	Channel type: Natural	Channelized	Concrete
% bank shading	87 A	69 A	9 B
cv[a] of bank-full width	24 A	21 A	0 B
Depth (m)	0.19 A	0.23 A	0.10 B
Open canopy (degrees)	61 A	122 AB	153 B
cv open canopy	47 A	24 AB	9 B
% riffle	17 A	7 AB	0 B
Streambed substrate size[b]	4.9 A	4.1 A	1.0 B
% urban land use (contributing)	45 A	53 B	79 B
Water temperature (°C)	23.5 A	26.0 B	30.3 B

[a] cv, coefficient of variation.
[b] The streambed substrate size was characterized as 1, concrete; 2, silt, mud, or detritus; 3, sand (>0.063–2 mm); 4, fine/medium gravel (>2–16 mm); 5, coarse gravel (>16–32 mm); 6, very coarse gravel (>32–64 mm); 7, small cobble (>64–128 mm); 8, large cobble (>128–256 mm); 9, small boulder (>256–512 mm); 10, large boulder, irregular bedrock, irregular hardpan, or irregular artificial surface (Fitzpatrick et al. 1998).

natural streams; however, they were usually more similar to values for natural streams.

Principal components analysis of chemical and physical variables resulted in four PCA axes accounting for more than 80% of the variance in the data (Table 5). The first two PCA axes accounted for most of the variance (65%). Principal components analysis axis 1 explains about 39% of the variation among sites and is composed of variables associated with water source (Figure 3). Loadings for the chemical variables of chloride, potassium, sodium, phosphate, and ammonia increase in the positive direction toward the sites supplied

TABLE 5. Loadings of ranked chemical and physical variables and biological metrics derived from principal component analysis.

	Loadings					
Variable	Axis 1	Axis 2	Axis 3	Axis 4	Axis 5	Axis 6
Chemical and physical variables:						
Cl	0.28	—[a]	—	0.42	NA[b]	NA
K	0.31	—	—	—	NA	NA
Na	0.28	—	—	0.30	NA	NA
Ammonia	0.26	—	—	0.32	NA	NA
PO_4	0.31	—	—	—	NA	NA
Discharge	0.26	—	—	—	NA	NA
cv[c] bank-full width	—	—	0.25	—	NA	NA
Bank shading	—	0.31	–0.28	—	NA	NA
Channel width, wetted	—	—	0.43	—	NA	NA
Depth	—	0.36	—	0.29	NA	NA
cv depth	—	—	—	—	NA	NA
Open canopy	—	–0.26	0.39	—	NA	NA
cv open canopy	—	0.32	–0.33	—	NA	NA
% riffle	–0.24	—	—	—	NA	NA
Streambed substrate size	—	0.35	—	—	NA	NA
% urban land use (contributing)	—	—	–0.46	—	NA	NA
Velocity	0.26	—	—	–0.35	NA	NA
cv velocity	–0.25	—	—	0.47	NA	NA
Water temperature	—	–0.31	—	—	NA	NA
Cumulative variance explained:	39	65	75	83	NA	NA
Macroinvertebrate taxa metrics:						
Richness	—	0.22	—	0.22	0.24	—
EPT richness	0.21	—	0.24	—	—	—
Trichoptera richness	—	—	0.23	—	—	—
% nonchironomid dipteran richness	—	0.28	—	–0.24	—	—
% nonchironomid abundance	—	0.28	—	—	—	—
% Orthoclad richness	—	–0.23	–0.21	–0.30	—	—
% Orthoclad abundance	—	—	—	–0.24	0.21	0.31
% Oligochaete richness	—	–0.22	—	—	—	—
% Oligochaete abundance	—	—	—	—	—	—
EPA tolerance, based on richness	–0.22	—	–0.20	—	—	0.21
% parasite richness	—	—	–0.23	—	—	—
% parasite abundance	—	—	–0.23	—	—	—
Scraper richness	—	—	0.38	—	—	—
% shredder richness	—	–0.25	—	—	—	0.46
% shredder abundance	—	—	0.22	—	—	0.28
% predator richness	—	—	—	0.32	—	0.37
% gatherer richness	—	—	–0.25	—	—	0.31
% Trichoptera abundance	0.25	—	—	—	—	—

TABLE 5. Continued.

Variable	Loadings Axis 1	Axis 2	Axis 3	Axis 4	Axis 5	Axis 6%
%Tanytarsini abundance	0.20	—	—	0.22	0.23	—
% Tanytarsini abundance/Chironomid abundance	—	—	–0.20	—	0.25	—
% noninsect abundance	–0.23	—	—	—	—	—
Nonmidge diptera plus noninsect abundance	–0.23	—	—	—	—	—
% filterer richness	—	—	—	–0.26	—	—
% filterer abundance	0.24	—	—	—	—	—
Cumulative variance explained:	38	65	75	83	88	92
Periphyton taxa metrics:						
% diatom richness	—	0.56	—	NA	NA	NA
% diatom abundance	—	0.52	0.27	NA	NA	NA
Green algae biovolume	–0.29	—	–0.55	NA	NA	NA
Blue-green algae biovolume	–0.44	—	0.33	NA	NA	NA
% blue-green algae biovolume	–0.36	—	0.53	NA	NA	NA
% Nitrogen-heterotrophic diatom abundance	–0.38	0.33	–0.34	NA	NA	NA
% eutrophic diatom abundance	–0.35	0.37	–0.28	NA	NA	NA
% motile diatom abundance	–0.33	0.31	—	NA	NA	NA
% Pollution-sensitive diatom abundance	0.39	—	—	NA	NA	NA
Cumulative variance explained:	43	69	84	NA	NA	NA

[a] —, loading < 0.20 for macroinvertebrate metrics; loading < 0.24 for periphyton metrics and chemical and physical variables.
[b] NA, not applicable.
[c] cv, coefficient of variation.

with treated wastewater. Physical variables of discharge and velocity also increase in the positive direction. Physical variables of percentage of riffles and cv of velocity increase in the negative direction toward the sites supplied with natural water (Table 5; Figure 3).

Principal components analysis axis 2 explains 26% of the variation among sites and includes primarily variables associated with channel type (Table 5; Figure 3). Loadings for the physical variables of bank shading, depth, streambed substrate size, and cv of open canopy increase in the positive direction toward sites that are channelized or have natural channels. Open canopy and water temperature increase in the negative direction toward sites that are concrete-lined.

Biological Metrics and Water Source

Twenty-four benthic macroinvertebrate and six periphyton metrics were significantly different among water sources (ANOVA, $P < 0.05$). Eight benthic macroinvertebrate metrics are measures of taxa richness, 6 are measures of taxa abundance, and 10 are measures of functional-feeding groups. Several macroinvertebrate metrics based on slightly different measures of the same taxa (e.g., oligochaetes) gave similar results, suggesting that they were redundant. In these cases, only results for relative richness and abundance are given (Tables 6 and 7). Macroinvertebrate metrics varying with water source included taxa richness, Ephemeroptera-Plecoptera-Trichoptera (EPT) richness, nonchironomid dipterans, orthoclad chironomids, oligochaetes, richness of tolerant taxa, parasites, shredders, predators, scrapers, and gatherers.

Streams supplied by natural water and streams supplied by treated wastewater were significantly different from other sources for more biological metrics than were streams supplied by either human-impacted groundwater or urban runoff (Table 6). Streams supplied by natural water sources generally had higher values for total richness, EPT richness, trichopteran

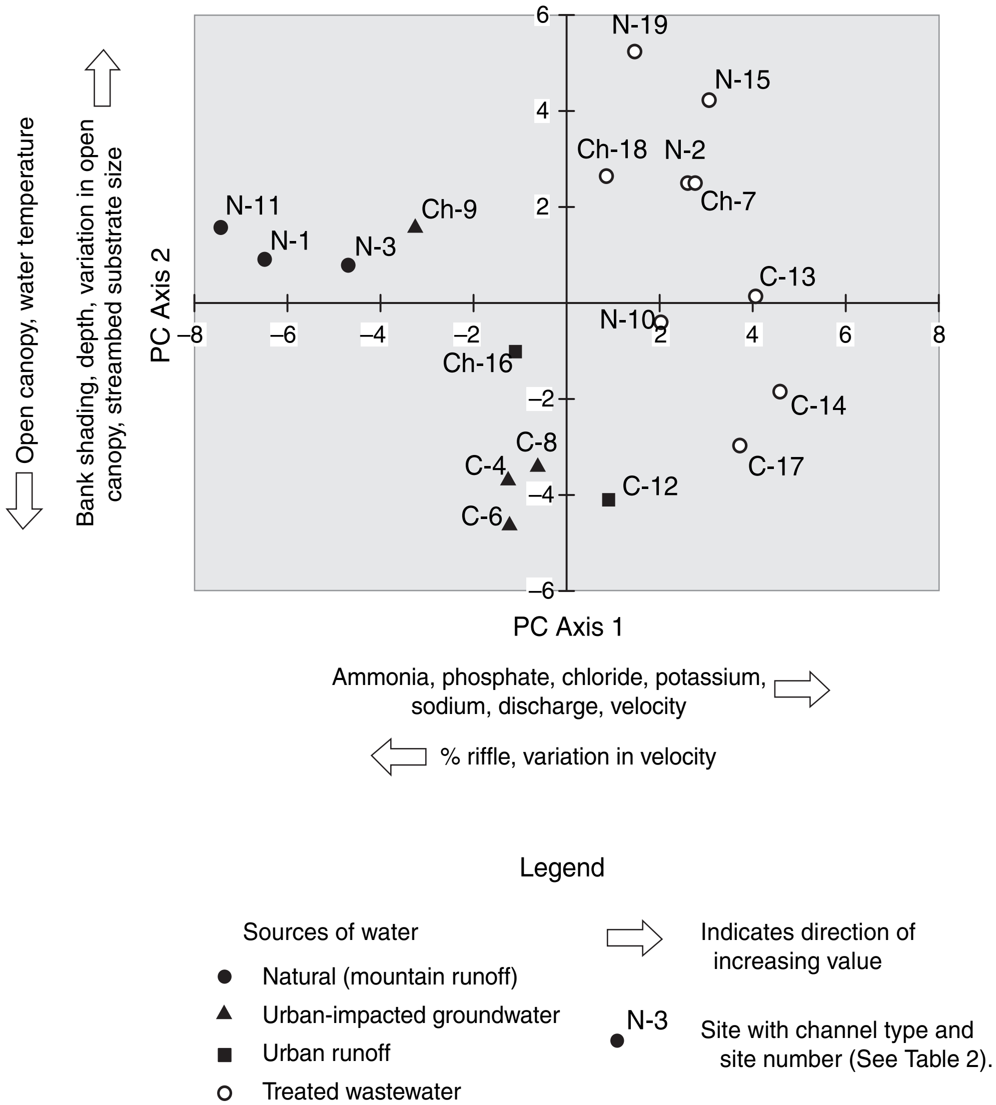

FIGURE 3. Principal component analysis (PCA) site scores based on chemical and physical data from the Santa Ana River basin, California. Principal component analysis axis 1 is controlled by water-source variables. Principal component analysis axis 2 is controlled by channel-type variables.

richness, and measures of nonchironomid dipteran abundance (predominantly Simuliidae and *Caloparyphus* sp.) than did streams dominated by treated wastewater (Table 6). Urban runoff and treated wastewater generally had higher values than those of natural water or groundwater sources for measures of orthoclad chironomids (dominated by *Cricotopus* sp.) and oligochaetes.

The importance of functional-feeding groups varied among water sources (Table 6). Measures of parasites (Nematoda) and predators (Turbellaria) were greatest at urban groundwater sites. Measures of shredders (predominantly *Cricotopus* sp.) were greatest at urban runoff and treated wastewater sites. Scraper richness (dominated by *Physella* sp., *Petrophila* sp., and *Helicopsyche* sp.) was significantly greater in natural waters and urban runoff compared with treated wastewater and urban groundwater. Percentage gatherer richness (predominantly Naididae, *Fallceon* sp., and *Baetis* sp.) was lowest in natural waters compared with urban runoff or groundwater.

Thirteen of the 19 selected chemical and physical

TABLE 6. Median values for biological metrics that were significantly different among water-source categories (ANOVA, $p < 0.05$). For each variable, medians with different letters were significantly different (Tukey tests, $P < 0.05$).

	Water source			
Metric	Natural	Groundwater	Urban runoff	Treated wastewater
Macroinvertebrate taxa metrics:				
Taxa richness	19 A[b]	17 AB	14 AB	9 B
EPT[a] richness	6 A	5 B	1.5 B	2 B
Trichoptera richness	4 A	3 AB	1.5 AB	1 B
% nonchironomid dipteran richness	11 AB	23 A	11 AB	0 B
% nonchironomid dipteran abundance	9 A	11 A	1 B	0 B
% Orthoclad richness	0 A	10 AB	9 AB	14 B
% Orthoclad abundance	0 A	<1 A	35 B	4 B
% Oligochaete richness	0 A	0 A	14 B	14 B
% Oligochaete abundance	0 A	0 A	28 B	17 B
EPA[b] tolerance, based on richness	4.9 A	6 B	7.2 B	5.5 AB
Macroinvertebrate functional-feeding group metrics:				
% parasite richness	0 A	7 B	0 A	0 A
% parasite abundance	0 A	<1 B	0 A	0 A
% shredder richness	8 A	7 A	19 B	17 AB
% shredder abundance	1 AB	<1 A	36 B	5 B
% predator richness	23 AB	42 A	33 AB	20 B
Scraper richness	4 A	2 B	3 A	1 B
% gatherer richness	30 A	42 B	48 B	33 AB
Periphyton taxa metrics:				
Blue-green algae biovolume	0 A	0.004 B	0 AB	0.1 B
% blue-green algae biovolume	0 AB	<1 AB	<1 A	5 B
% Nitrogen heterotrophic diatom abundance	11 A	16 A	86 B	94 B
% eutrophic diatom abundance	57 AB	22 A	87 BC	97 C
% motile diatom abundance	33 A	53 A	81 AB	76 B
% pollution-sensitive diatom abundance	46 A	2 AB	3 A	<1 B

[a] EPT, Ephemeroptera-Plecoptera-Trichoptera.
[b] EPA, U.S. Environmental Protection Agency.

variables were correlated with more than 2 of the 24 benthic macroinvertebrate metrics associated with water source. Measures of taxa richness, EPT richness, and nonchironomid dipterans were commonly negatively correlated with concentrations of dissolved chemicals (Table 7). These metrics tended to be positively correlated with cv of bank-full width, presence of riffles, and streambed substrate size. That is, these metrics were higher at sites that had lower concentrations of Cl, K, Na, orthophosphate (PO_4), ammonia, and lower percentages of urban land use and more heterogeneous habitat. Measures of orthoclad chironomids, oligochaetes, and U.S. Environmental Protection Agency's (EPA) tolerance tended to show the opposite patterns. Correlations of functional-feeding groups with chemical and physical variables were more limited, and there were no strong general patterns (Table 7).

Two of six periphyton metrics associated with water source are measures of blue-green algal biovolume, and four metrics are autecological measures (Table 6). Streams with natural waters were lower in blue-green algal biovolume and relative abundance of nitrogen heterotrophic, eutrophic, and motile diatoms (predominantly *Nitzschia amphibia, N. palea,* and *Diadesmis confervacea*) than were streams with treated wastewater (Table 6). The relative abundance of pollution-sensitive diatom species (predominantly *Cymbella affinis*) was greater in natural waters than in treated wastewater (Table 6). The relationships of sites with urban runoff and groundwater were mixed.

All fifteen chemical and physical variables differing among sources of water were correlated with at least one of six periphyton metrics. The periphyton metrics associated with water source, except pollution-

TABLE 7. Results of Spearmans' correlations for chemical and physical variables with invertebrate metrics that were associated with water-source categories.

Metric	Cl	K	Na	Ammonia	PO_4	Discharge	cv[a] bankfull width	cv depth	% riffle	Streambed substrate size	Velocity	% urban land use (contributing)	Water temperature
Macroinvertebrate taxa metrics:													
Taxa richness	—[b]	—	—	—	—	–	+	NS	NS	NS	–	–	NS
EPT[c] richness	–	NS	—	–	–	NS	++	NS	+	+	NS	—	NS
Trichoptera richness	—	NS	—	–	—	NS	++	NS	+	+	NS	—	NS
% nonchironomid dipteran richness	–	–	NS	NS	–	NS	NS	NS	NS	NS	NS	NS	++
% nonchironomid dipteran abundance	—	—	—	—	—	–	NS	NS	NS	NS	NS	NS	NS
% Orthoclad richness	++	+	++	+	++	+	NS	NS	NS	NS	+	NS	NS
% Orthoclad abundance	+	NS	+	++	+	NS	NS	NS	NS	NS	NS	+	NS
% Oligochaete richness	NS	++	++	++	++	NS	NS	NS	NS	NS	–	NS	–
% Oligochaete abundance	NS	NS	NS	++	+	NS	NS	NS	NS	NS	NS	NS	NS
EPA tolerance, based on richness	NS	NS	NS	NS	NS	NS	–	–	—	—	NS	+	NS
Macroinvertebrate functional-feeding group metrics:													
% parasite richness	NS	–	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	+
% parasite abundance	NS	–	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	+
Scraper richness	—	NS	—	NS	–	–	NS	NS	+	NS	NS	–	NS
% shredder richness	NS	++	NS	+	+	+	NS	NS	NS	NS	+	NS	NS
% shredder abundance	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
% predator richness	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
% gatherer richness	NS	NS	NS	NS	NS	NS	-	-	—	—	NS	+	NS

[a] cv, coefficient of variation.

[b] +, positive correlation at $P < 0.05$; –, negative correlation at $P < 0.05$; ++, positive correlation at $P < 0.01$; —, negative correlation at $P < 0.01$.

[c] EPT, Ephemeroptera-Plecoptera-Trichoptera.

TABLE 8. Results of Spearmans' correlations for chemical and physical variables with periphyton metrics that were associated with water-source categories.

Metric	Cl	K	Na	Ammonia	PO_4	Discharge	cv[a] bankfull width	Channel width, wetted	cv depth	% riffle	Stream bed substrate size	Velocity	cv velocity	% urban land use (contributing)	Water temperature
Blue-green algae biovolume	++[b]	NS	++	NS	NS	+	NS	++	NS	NS	NS	NS	NS	NS	NS
% blue-green algae biovolume	NS	NS	+	NS	NS	+	NS	++	NS	NS	NS	NS	NS	NS	NS
% Nitrogen-heterotrophic diatom abundance	+	++	+	+	++	+	NS	++	NS	–	NS	++	—	NS	–
% eutrophic diatom abundance	+	++	+	+	++	++	NS	++	NS	NS	NS	++	—	NS	–
% motile diatom abundance	+	+	++	+	+	NS	NS	+	NS	NS	NS	NS	–	NS	–
% pollution-sensitive diatom abundance	–	–	—	–	–	NS	++	NS	+	++	++	NS	NS	—	NS

[a] cv, coefficient of variation.

[b] +, positive correlation at $P < 0.05$; –, negative correlation at $P < 0.05$; ++, positive correlation at $P < 0.01$, —, negative correlation at $P < 0.01$.

TABLE 9. Median values for biological metrics that were significantly different among channel types (ANOVA, $P < 0.05$). For each variable, medians with different letters were significantly different (Tukey tests, $P < 0.05$).

Metric	Channel type		
	Natural	Channelized	Concrete
Macroinvertebrate taxa metrics:			
% Trichoptera abundance	29 A	57 A	0.3 B
% Tanytarsini abundance	1 A	2.9 AB	0 B
% Tanytarsini/Chironomid abundance	0.2 A	0.4 AB	0 B
% noninsect abundance	9.2 AB	6.7 A	49 B
Nonmidge diptera plus noninsect abundance	5,346 A	9,863 A	44,878 B
Macroinvertebrate functional-feeding group metrics:			
% filterer richness	25 A	22 AB	0 B
% filterer abundance	50 A	57 A	0 B
Periphyton taxa metrics:			
% diatom richness	20 A	21 A	13 B
% diatom abundance	86 A	90 A	45 B
Green algae biovolume	0.003 A	0 A	1.5 B

sensitive diatom abundance, tended to be positively correlated with chemical variables and with most physical variables but had negative correlations with cv of stream velocity, water temperature, and presence of riffles (Table 8). Therefore, these metrics were higher at sites with higher Cl, K, Na, PO_4, and ammonia concentrations but cooler water temperatures. Pollution-sensitive diatom abundance tended to be negatively correlated with these chemical variables and urban land use and positively correlated with most physical variables (Table 8).

Biological Metrics and Channel Type

Thirteen benthic macroinvertebrate and three periphyton metrics were significantly different among channel types (ANOVA, $P < 0.05$). Nine macroinvertebrate metrics are measures of taxa abundance. Four macroinvertebrate metrics are measures of functional-feeding groups. Similar to water source results, several macroinvertebrate metrics based on slightly different measures of the same taxa gave similar results suggesting redundancy. Therefore, only results for relative abundance are given (Tables 9 and 10). Macroinvertebrate metrics differing with channel type included measures of abundance of trichopterans, tanytarsini chironomids, noninsects, and filterers.

Concrete-lined channels had significantly lower abundances of trichoptera, tanytarsini chironomids, and filterers (predominantly *Hydropsyche* sp. and *Rheotanytarsus* sp.) in comparison with natural and channelized channels (Table 9). Concrete-lined channels were highest in noninsect abundance (predominantly Naididae, Turbellaria, *Hyalella* sp., and *Physella* sp.). No significant differences in macroinvertebrate metrics were observed between natural and channelized sites (Table 9).

Nine physical variables showing significant differences between channel types were correlated with at least 1 of the 13 benthic macroinvertebrate metrics associated with channel type (Table 10). Several strong patterns were evident in macroinvertebrate correlations. Metrics for trichopterans and tanytarsini chironomids were positively correlated with percentage of bank shading, depth, and streambed substrate size and negatively correlated with open canopy, water temperature, and urban land use (Table 10). This was generally reflected in results for filterers because the two taxonomic groups were dominated by filtering genera, *Hydropsyche* sp. and *Rheotanytarsus* sp., respectively. Hence, trichopterans and tanytarsini chironomids were found in higher abundance at sites with more shading, larger streambed substrate, cooler water temperatures, and less urban land use. Noninsects showed reverse patterns with negative correlations to percentage of bank shading, depth, streambed substrate size, and urban land use. Noninsect metrics also showed negative correlations with cv of bank-full width and presence of riffles indicating higher abundance of noninsects in concrete-lined channels.

Relative diatom richness and abundance were lowest in concrete-lined channels (Table 9). Biovolume of green algae was greater in concrete-lined channels than in channelized and natural streams (Table

TABLE 10. Results of Spearman's correlations for physical and chemical variables with invertebrate metrics that were associated with channel type.

Metric	% bank shading	cv[a] of bank-full width	Depth	Open canopy	cv open canopy	% riffle	Streambed substrate size	% urban land use (contributing)	Water temperature
Macroinvertebrate taxa metrics:									
% Trichoptera abundance	+[b]	+	+	NS	NS	+	++	–	NS
% Tanytarsini abundance	++	NS	+	–	+	NS	+	–	NS
% Tanytarsini/Chironomid abundance	++	+	+	–	+	NS	+	–	–
% noninsect abundance	–	NS	NS	NS	NS	NS	–	NS	NS
Nonmidge diptera plus noninsect abundance	NS	–	–	NS	NS	–	—	++	NS
Macroinvertebrate functional-feeding group metrics:									
% filterer richness	++	NS	++	–	++	NS	+	NS	NS
% filterer abundance	++	NS	++	–	+	NS	++	NS	NS

[a] cv, coefficient of variation.

[b] **+**, positive correlation at $P < 0.05$; –, negative correlation at $P < 0.05$; **++**, positive correlation at $P < 0.01$; —, negative correlation at $P < 0.01$.

TABLE 11. Results of Spearman's correlations of physical variables with periphyton metrics that were associated with channel type.

Metric	cv[a] bank-full width	Streambed substrate size	Water temperature
% diatom richness	++[b]	+	–
% diatom abundance	++	+	–
Green algae biovolume	–	–	NS

[a] cv, coefficient of variation

[b] +, positive correlation at $P < 0.05$; –, negative correlation at $P < 0.05$; ++, positive correlation at $P < 0.01$.

9).Three variables were correlated with at least one of three periphyton metrics (Table 11). Relative richness and the abundance of diatoms were positively correlated with cv of bank-full width and streambed substrate size but negatively correlated with water temperature (Table 11). Therefore, more diatoms were found in higher abundances in natural and channelized streams than in concrete channels. Biovolume of green algae showed a reverse pattern and was negatively correlated to both cv of bank-full width and streambed substrate size (Table 11).

Principal Components Analysis

Principal components analysis effectively summarized the relations between macroinvertebrate and periphyton metrics. Principle components analysis of the macroinvertebrate metrics resulted in six PCA axes that accounted for more than 90% of the variance in the macroinvertebrate metrics among sites (Table 5). The first axis explained 38% of the variance among sites. Most of the metrics dominating this axis were associated with channel type; however, three metrics associated with water source—EPT richness, EPA tolerance, and shredder abundance—were also important (Table 5; Figure 4). The second axis explained 27% of the variation among sites and was dominated by macroinvertebrate metrics associated with water source. Loadings for orthoclads, oligochaetes and shredders increase in the negative direction toward the sites supplied with treated wastewater. Taxa richness and nonchironomid dipteran metrics increase in the positive direction toward sites supplied by natural water or groundwater.

Principal component analysis of periphyton metrics resulted in three PCA axes that accounted for 84% of the variance among sites (Table 5). The first axis accounted for 42% of the variance among sites and is dominated by metrics associated with water source (Figure 5). The second axis accounts for 26% of the variance and is also dominated by metrics associated with water source; however, metrics associated with channel type also contribute to the second axis. In general, sites with treated wastewater are in the upper left portion of the graph, whereas sites with natural sources of water are more toward the right of the graph; sites with urban runoff and groundwater are between the two (Figure 5).

Discussion

The categorical study design successfully characterized associations of biological assemblages with water source and channel type. Many macroinvertebrate and periphyton metrics showed significant differences among the four sources of water or the three types of channels (Tables 6 and 9). In addition, many chemical and physical variables that correlated to these biological metrics also showed significant differences between sources of water and channel types (Tables 3 and 4). Although water source and channel type were treated largely as independent factors for analysis, organisms are responding to physical and chemical conditions generated jointly by these two factors and others not explicitly considered in the study (such as annual patterns in hydrology). For example, PCA axis 1 from the analysis of chemical and physical variables (Figure 3) clearly separates sites with natural and wastewater sources, but the gradient includes chemical and physical variables related to differences among streams with natural channels and streams in concrete-lined channels. Brown et al. (2005) address these issues more fully.

Artificial substrates were used to mitigate some effects caused by different channel types; however, it is likely that the macroinvertebrate and periphyton assemblages sampled from the artificial substrates were somewhat different from the natural assemblages. Lamberti and Resh (1985) found that clay tiles were similar to natural substrates for algae and macroinvertebrates after colonizing for 28 d. Other studies showed artificial substrates had different compositions and abundances of invertebrates in comparison with natural substrates (Garie and McIntosh 1986; Casey and Kendall 1996). Colonization rates and replacement of colonizing periphyton by more persistent species can be affected by environmental conditions, including current velocity, nutrient concentrations, water temperature, and light (Oemke and Burton 1986). Although assemblages on artificial substrates are likely different from assemblages in the stream,

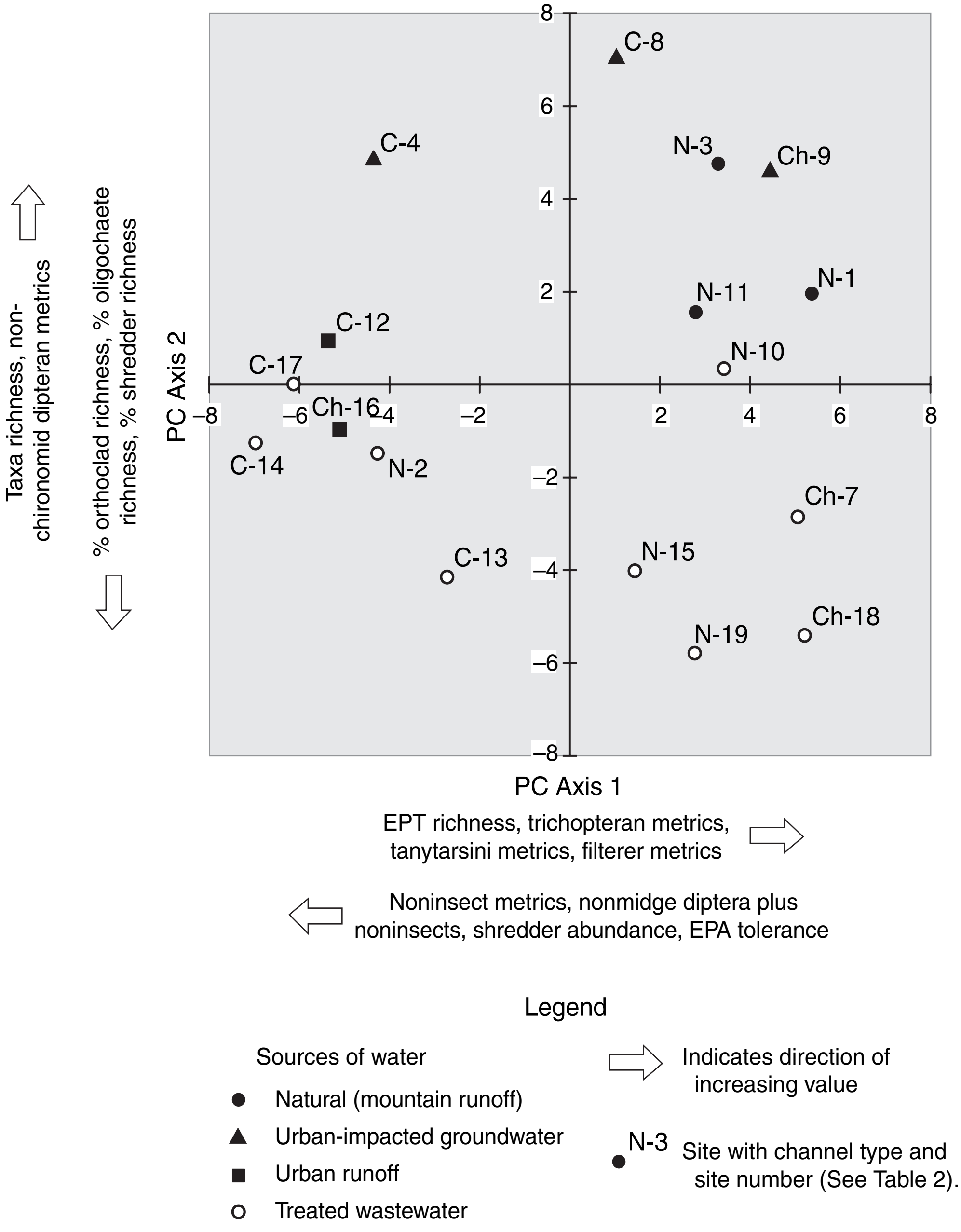

FIGURE 4. Principal component analysis (PCA) site scores based on macroinvertebrate metrics from the Santa Ana River basin, California. Principal component analysis axis 1 is controlled principally by metrics associated with channel type. Principal component analysis axis 2 is controlled principally by metrics associated with water source.

artificial substrates should represent a similar sub-assemblage, including the taxa best at colonizing and exploiting new habitats. Therefore, assemblages on artificial substrates likely reflect differences between sites.

Macroinvertebrate Response

Macroinvertebrate metrics appeared most responsive to habitat alteration. The first PCA axis (Figure 4) emphasizes tanytarsini, trichoptera, noninsect, and filterer metrics that differed among channel types.

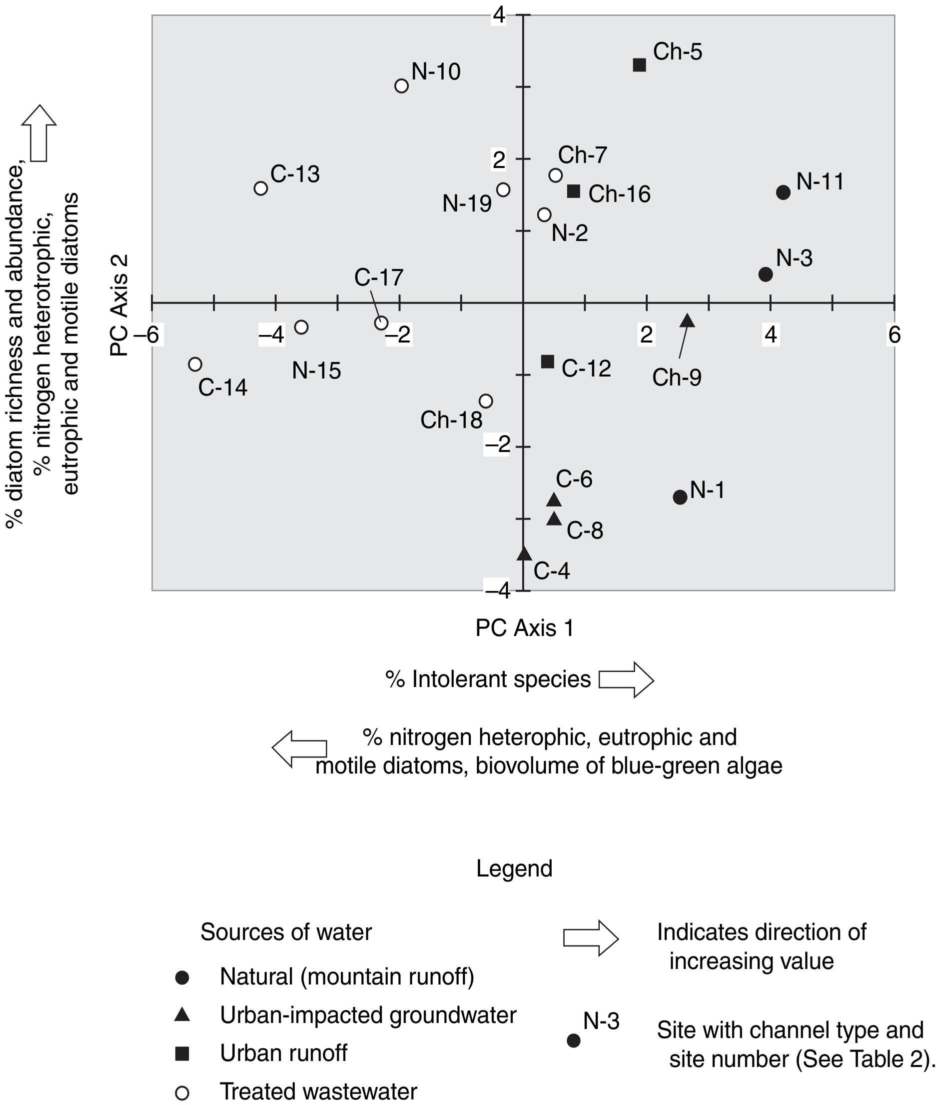

FIGURE 5. Principal component analysis (PCA) site scores based on periphyton metrics from the Santa Ana River basin, California. Principal component analysis axis 1 is controlled principally by metrics associated with water source. Principal component analysis axis 2 is controlled by metrics associated with both water source and channel type.

Concrete-lined channels always differed from one or both of the other channel types (Table 9). Although channelized sites appeared to be intermediate between concrete-lined and natural sites, differences between channelized sites and natural sites were not statistically significant. This suggests that channelized sites are able to support the same assemblages as natural channels. Most habitat measures were similar between these two channel types (Table 4). In particular, some riparian vegetation was present at most channelized sites, ranging from grasses and cattails to trees. There was less riparian vegetation in channelized streams compared with natural channels, but channelized streams were not as bare as concrete channels.

Several metrics related to water source—EPT richness, trichoptera richness, EPA tolerance, and shredder abundance—mainly reflect differences among the natural, least-impacted sites and the other sites with a variety of water sources (Table 6). Natural-water source sites were chemically and physically different from other sites (Figure 3; Table 3). These sites are at the interface of the mountains and valley and have little urban land use in their drainage areas (Figure 1; Table 3).

Specific conductance and other surrogates for salinity are commonly related to macroinvertebrate assemblages (Leland and Fend 1998; Brown and May 2000), but they were not correlated in this study. The lack of such correlation, despite correlations of metrics with ionic species, could be due to the relatively narrow range of specific conductance (286–1,250 µS/cm) and one of the least-impacted sites having a fairly high value (804 µS/cm at site 3). In addition, the least-impacted site was high in calcium, magnesium, and sulfate instead of Na, Cl, or K. There were differences in ionic composition among water sources (Table 3); however, it is unknown if differences of this relatively small magnitude can affect macroinvertebrate taxa.

The macroinvertebrate metric responses observed in this study are generally consistent with results of other studies. The least-impacted streams had the highest macroinvertebrate taxa richness, EPT richness, and trichopteran richness in comparison with urban streams. Barbour et al. (1999) observed that increases in tolerance and relative abundance of oligochaetes were expected responses to stream degradation. The proportion of oligochaetes is expected to be higher in urban areas (Winter and Duthie 1998; Ourso 2001; Paul and Meyer 2001; Kennen and Ayers 2002) and to increase with stream degradation (Karr and Kerans 1992; Fore et al. 1996).

A number of other macroinvertebrate metrics suggest differences among the urban water sources (Table 6). For example, macroinvertebrate metrics at urban-impacted groundwater sites appear to be different from those for other water sources as indicated by the dominance of nematodes (parasite metrics), turbellarians (predator metric), and nonchironomid dipterans and relatively low values for oligochaete metrics (Table 6). These metrics were responsive to environmental stresses including urbanization in other studies (Garie and McIntosh 1986; Karr and Kerans 1992; Barbour et al. 1996; Fore et al. 1996). However, those studies were conducted in less altered streams across a relatively wide range of urbanization. It is unclear what such differences mean ecologically in the context of highly altered streams all within a highly urbanized area.

Several functional-feeding group metrics were responsive to water source or channel type and warrant discussion because of implications for ecological functions within highly urbanized areas. With respect to water source, the most interesting relationships involved shredders and scrapers. Shredder metrics tended to be higher at sites affected by urban runoff and treated wastewater (Table 6), mainly because of the abundance of *Crictopus* sp., a pollution tolerant genus of chironomids that has been associated with highly urbanized sites (Jones and Clark 1987; Kennen and Ayers 2002). Cummins et al. (1989) and Kerans and Karr (1994) suggested that the presence of shredders may be more indicative of local riparian habitat rather than more general indicators of urbanization. Although a number of these sites have riparian vegetation present, the highest abundance of this genus occurred at an urban-runoff site (site 12) with no riparian vegetation and high urbanization. Shredders could be feeding on macroalgae present at the site.

Scraper richness was higher at sites supplied with natural water or urban runoff compared to sites supplied with groundwater or treated wastewater. Decreasing scraper richness has been considered an indicator of environmental degradation, although Fore et al. (1996) suggested that the response of scrapers can be variable. Scraper taxa at urban-runoff sites and at groundwater sites consisted of both highly pollution-tolerant genera (i.e., *Physella* sp.) and pollution-sensitive genera (i.e., *Petrophila* sp. and *Helicopsyche* sp.). However, the pollution-tolerant *Physella* sp. was significantly more abundant at sites with urban runoff than at sites with groundwater.

Filterer metrics were strongly affected by channel type (Table 9), with concrete-lined channels having the lowest abundances. Primary filtering taxa in this study were the trichopteran *Hydropsyche* sp. and the chironomid *Rheotanytarsini* sp. These taxa were also most responsible for responses of trichoptera and tanytarsini metrics to channel type. Previously reported responses of filterers to environmental degradation are variable (Karr and Kerans 1992; Barbour et al. 1996; Fore et al. 1996; Winter and Duthie 1998). *Hydropsyche* sp. and *Rheotanytarsini* sp. are generally considered pollution sensitive (DeShon 1995; Barbour et al. 1999); however, in this study, water chemistry of concrete channels was variable (Figure 4). A simple physical explanation might be applicable. Both of these filterer taxa are net builders (*Rheotanytarsini* sp. can also build tubes) (Merritt and Cummins 1984). The relatively two-dimensional nature of the concrete substrate may limit the number of locations appropriate for net and tube construction, therefore limiting the population.

Periphyton Response

Periphyton metrics were most responsive to water source. Principal component analysis axis 1 included only metrics responsive to water source. Principal component analysis axis 2 included metrics responsive to water source and those responsive to channel type (Figure 3; Table 5). Sites with treated wastewater generally had the highest values for the periphyton metrics, except for percentage pollution-sensitive diatoms, which had the lowest value (Table 6). Other water sources were more variable, with each being at the other extreme from treated wastewater for at least one metric. Clearly, treated wastewater has a strong effect on periphyton assemblages.

Three periphyton metrics highest in treated wastewater—nitrogen-heterotrophic diatom abundance, eutrophic diatom abundance, and motile diatom abundance—were dominated by *Nitzchia palea*, a species most abundant in the most disturbed streams in a New Jersey urban study (Kennen and Ayers 2002). All three metrics show a positive relationship with inorganic salts, ammonia, phosphate, and wetted channel width and a negative relationship with cv of velocity. Treated wastewater has the highest concentrations of these chemical constituents and sites with treated wastewater had the widest channels and the most uniform velocities. Similar relationships between periphyton metrics and physical and chemical variables were found in the mid-Appalachian region (Hill et al. 2000).

Two periphyton metrics—relative diatom richness and abundance—were lowest in concrete-lined channels, whereas green algae biovolume was highest in concrete-lined channels (Table 9). Response of the diatom metrics is indicative of higher environmental stress in concrete channels (Barbour et al. 1999). Diatom metrics have a positive correlation to streambed substrate and cv of bank-full width, indicating that diatoms prefer more complex aquatic habitat than is available in concrete-lined channels. Also, diatom metrics had a negative correlation with water temperature. Previous studies showed that diatom richness decreases when water temperature is greater than 30°C and diatoms are often replaced with green or blue-green algae (DeNicola 1996). In our study, fewer diatoms and a higher biovolume of green algae were present in concrete channels, which were warmer than the other channel types. Median water temperature at sites with concrete channels was 30°C compared to 23°C at sites with natural channels and 26°C in channelized streams.

Challenges of Studying Highly Urbanized Areas

A general approach to urban aquatic studies is to examine hydrologic, geomorphic, chemical, habitat, and biological characteristics of streams over a gradient of urban land use intensities in specific environmental settings (e.g., Tate et al. 2005, this volume). The basic requirements for these studies include (1) consistent definition of the land-use gradient; (2) consistent environmental setting to minimize natural gradients (e.g., temperature, stream size, hydrologic variability, soil type, and precipitation); (3) collection of sufficient physical, chemical, and biological data to characterize stream response to land use; and (4) sufficient distribution and number of sites to accurately represent a response over the range of the land-use gradient. Conditions in the Santa Ana River basin did not support this approach.

The traditional view of precipitation falling on the landscape and flowing toward stream channels through various land uses does not apply in the Santa Ana River basin and in other basins in the arid and semiarid Southwest. Some streams are ephemeral in nature. Other streams are diverted for public supply or into retention basins for groundwater recharge. As a result, many streams lose surface flow shortly after reaching the valley floor. Flow is reestablished in some stream channels from anthropogenic sources (e.g., treated wastewater, urban runoff). Therefore, the traditional view that water quality in a stream channel reflects the proportions of land uses in a basin also does not apply. Alternatively stated, land use is not a surrogate for the source of water in the Santa Ana River basin, and land use and basin area are not surrogates for the volume of water in the streams. These discontinuities in the hydrologic cycle owing to the semiarid climate and human engineering in combination with extreme urbanization made it impossible to find least-impacted sites in the valley. The rapid change from undeveloped mountains to the highly urbanized valley made it impossible to define an urbanization gradient because there were few streams with low to intermediate levels of urbanization (Table 2). This also precludes another common approach to urban studies, comparing reference or least-impacted conditions with conditions in urbanized areas.

An alternative model is to explicitly identify aspects of highly urbanized streams, such as impervious area, and determine the associations of the chosen factor with some of the effects this has on habitat quality, water quality, or aquatic communities. However, it is difficult to study just one or two aspects of urbanization because the observed effect may also be caused by other factors not included in the study. In the Santa Ana River basin, some aspects of the hydrologic system are well constrained despite the complexities of human involvement. Therefore, it was possible to identify the source of water and channel type for many of the streams. This provided an opportunity for assessing the effects of these two anthropogenic factors; however, even in this highly constrained system, the effects of the two factors could not be separated completely.

Conclusions

Based on analysis of macroinvertebrate and periphyton metrics and their correlations with chemical and physical variables, water source and channel type had significant associations with ecosystem condition. The source of water affected not only chemical variables such as Cl and K, but also some physical variables such as velocity and water temperature, whereas channel type affected only physical variables such as streambed substrate size. Macroinvertebrate assemblages appeared to be most sensitive to physical variables, particularly those associated with differences between concrete-lined channels and streams with natural channels or channelized streams with natural bottoms. In contrast, periphyton metrics mainly reflected associations with aspects of water quality related to water source.

Our results suggest that significant improvements in aquatic ecological condition are possible in streams of the highly urbanized Santa Ana River basin even though conditions will not be returned to full ecological function. These results are likely to be similar for other urban streams in arid or semiarid areas. Given appropriate design criteria, converting concrete-lined channels to channelized streams with stabilized sides and natural bottoms could support aquatic ecosystems more similar to those of less degraded sites and still maintain flood control to protect life and property. Reducing chloride, sodium, potassium, ammonia, and phosphate concentrations in treated wastewater could also contribute to such changes.

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References

Aloi, J. E. 1990. A critical review of recent freshwater periphyton field methods. Canadian Journal of Fisheries and Aquatic Sciences 47:656–670.

Bahls, L.L. 1993. Periphyton bioassessment methods for Montana streams (Revised January 1993). Department of Health and Environmental Sciences, Water Quality Bureau, Helena, Montana.

Barbour, M. T., J. Gerritsen, G. E. Griffith, R. Frydenborg, E. McCarron, J. S. White, and M. L. Bastian. 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. Journal of the North American Benthological Society 15:185–211.

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, 2nd edition. U.S. Environmental Protection Agency Report 841-B-99–02, Washington, D.C.

Booth, D. B. 1991. Urbanization and the natural drainage system—impacts, solutions, and prognoses. Northwest Environmental Journal 7:93–118.

Booth, D. B., and L. E. Reinelt. 1993. Consequences of urbanization on aquatic systems—measured effects, degradation thresholds, and corrective strategies. Pages 545–550 *in* Proceedings of Watershed '93, a national conference on watershed management, March 12–24, 1993, Alexandria, Virginia.

Brown, L. R., and J. T. May. 2000. Macroinvertebrate assemblages on woody debris and their relations with environmental variables in the lower Sacramento and San Joaquin River drainages, California. Environmental Monitoring and Assessment 64:311–329.

Brown, L. R., C. A. Burton, and K. Belitz. 2005. Aquatic assemblages of the highly urbanized Santa Ana River basin, California. Pages 263–287 *in* L. R. Brown, R. H. Gray, R. M. Hughes, and M. R. Meador, editors. Effects of urbanization on stream ecosystems. American Fisheries Society, Symposium 47, Bethesda, Maryland.

Burton, C. A. 2002. Effects of urbanization and long-term rainfall on the occurrence of organic compounds and trace elements in reservoir sediment cores, streambed sediment, and fish tissue from the Santa Ana River basin, California, 1998. U.S. Geological Survey Water-Resources Investigations Report 02–4175, Sacramento, California.

Burton, C. A., J. A. Izbicki, and K. S. Paybins. 1998. Water-quality trends in the Santa Ana River at MWD Crossing and below Prado Dam, Riverside County, California. U.S. Geological Survey Water-Resources Investigations Report 97–4173, Sacramento, California.

Casey, R. J., and S. A. Kendall. 1996. Comparisons among colonization of artificial substratum types and natural substratum by benthic macroinvertebrates. Hydrobiologia 341:57–64.

Charles, D. F., C. Knowles, and R. S. Davis, editors.

2002. Protocols for the analysis of algal samples collected as part of the U.S. Geological Survey National Water-Quality Assessment Program. Patrick Center for Environmental Research Report No. 02–06, Academy of Natural Sciences, Philadelphia.

Cuffney, T. F. 2003. User's manual for the National Water-Quality Assessment program invertebrate data analysis system (IDAS) software: Version 3. U.S. Geological Survey Open-File Report 03–172, Raleigh, North Carolina.

Cummins, K. W., D. M. Wilzbach, D. M. Gates, J. B. Perry, and W. B. Taliaferro. 1989. Shredders and riparian vegetation. Bioscience 39:24–30.

DeNicola, D. M. 1996. Periphyton responses to temperature at different ecological levels. Pages 149–181 *in* R. J. Stevenson, M. L. Bothwell, and R. L. Lowe, editors. Algal ecology: freshwater benthic ecosystems. Academic Press, San Diego, California.

DeShon, J. E. 1995. Development and application of the invertebrate community index. Pages 217–243 *in* W. S. Davis and T. P. Simon, editors. Biological assessment and criteria: tools for water resource planning and decision making. Lewis Publishers, Boca Raton, Florida.

Fitzpatrick, F. A., I. A. Waite, P. J. D'Arconte, M. R. Meador, M. A. Maupin, and M. E. Gurtz. 1998. Revised methods for characterizing stream habitat in the National Water-Quality Assessment Program. U.S. Geological Survey Water-Resources Investigations Report 98–4052, Raleigh, North Carolina.

Fore, L. S., J. R. Karr, and R. W. Wisseman. 1996. Assessing invertebrate responses to human activities: Evaluating alternative approaches. Journal of the North American Benthological Society 15:212–231.

Frick, E. A., D. J. Hippe, G. R. Buell, C. A. Crouch, and E. E. Hopkins. 1998. Water quality in the Apalachicola-Chattahoochee-Flint River basin, Georgia, Alabama, and Florida, 1992–1995. U.S. Geological Survey Circular 1164, Reston, Virginia.

Garie, H. L., and A. McIntosh. 1986. Distribution of benthic macroinvertebrates in a stream exposed to urban runoff. Water Resources Bulletin 22:447–455.

Grimm, N. B., J. M. Grove, S. T. A. Pickett, and C. L. Redman. 2000. Integrated approaches to long-term studies of urban ecological systems. BioScience 50:571–584.

Helsel, D. R., and R. M. Hirsch. 1992. Statistical methods in water resources. Elsevier, New York.

Hill, B. H., A. T. Herlihy, P. R. Kaufmann, R. J. Stevenson, F. H. McCormick, and C. B. Johnson. 2000. Use of periphyton data as an index of biotic integrity. Journal of North American Benthological Society 19:50–67.

Hitt, K. J. 1994. Refining 1970's land-use data with 1990 population data to indicate new residential development. U.S. Geological Survey Water-Resources Investigations Report 94–4250, Reston, Virginia.

Jones, R. C., and C. C. Clark. 1987. Impact of watershed urbanization on stream insect communities. Water Resources Bulletin 23:1047–1055.

Karr, J. R., and B. L. Kerans. 1992. Components of biological integrity: Their definition and use in developments of an invertebrate IBI. Pages 1–16 *in* W. S. Davis and T. P. Simon, editors. Proceedings of the 1991 Midwest Pollution Control Biologists Meeting, March 19–22, 1991, Lincolnwood, Illinois. U.S. Environmental Protection Agency 905/R-92/002, Chicago.

Kennen, J. G., and M. A. Ayers. 2002. Relation of environmental characteristics to the composition of aquatic assemblages along a gradient of urban land use in New Jersey, 1996–98. U.S. Geological Survey Water-Resources Investigations Report 02–4069, West Trenton, New Jersey.

Kerans, B. L., and J. R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. Ecological Applications 4:768–785.

Klein, R. D. 1979. Urbanization and stream quality impairment. Water Resources Bulletin 15:948–963.

Lamberti, G. A., and V. H. Resh. 1985. Comparability of introduced tiles and natural substrates for sampling lotic bacteria, algae and macroinvertebrates. Freshwater Biology 15:21–30.

Lange-Bertalot, H. 1979. Pollution tolerance of diatoms as a criterion for water quality estimation. Nova Hedwigia 64:285–304.

Leland, H. V., and S. V. Fend. 1998. Benthic invertebrate distributions in the San Joaquin River, California, in relation to physical and chemical factors. Canadian Journal of Fisheries and Aquatic Sciences 55:1051–1067.

Leopold, L. B. 1968. Hydrology for urban land planning—a guidebook on the hydrologic effects of urban land uses. U.S. Geological Survey Circular 554, Washington, D.C.

Lowe, R. L., and Y. Pan. 1996. Benthic algal communities as biological monitors. Pages 705–739 *in* R. J. Stevenson, M. L. Bothwell, and R. L. Lowe, editors. Algal ecology: freshwater benthic ecosystems. Academic Press, San Diego, California.

Mendez, G. O., and K. Belitz. 2002. Identifying sources of baseflow in the Santa Ana River, California. Pages 567–572 *in* J. F. Kenny, editor. Ground water/surface water interactions. Proceedings American Water Resources Association 2002 Summer Specialty Conference, TPS-02–2, Keystone, Colorado.

Merritt, R. W., and K. W. Cummins. 1984. An introduction to the aquatic insects, 2nd edition. Kendall/Hunt Publishing Company, Dubuque, Iowa.

Moring, J. B., and D. R. Rose. 1997. Occurrence and concentrations of polycyclic aromatic hydrocarbons in semipermeable membrane devices and clams in three urban streams of the Dallas-Fort Worth Metropolitan Area, Texas. Chemosphere 34:551–566.

Moulton, S. R. II, J. G. Kennen, R. M. Goldstein, and J. A. Hambrook. 2002. Revised protocols for sampling algal, invertebrate, and fish communities as part of the National Water-Quality Assessment Program. U.S. Geological Survey Open-File Report 02–150, Reston, Virginia.

Moulton, S. R., II, J. L. Carter, S. A. Grotheer, T. F. Cuffney, and T. M. Short. 2000. Methods of analysis by the U.S. Geological Survey National Water-Quality Laboratory—processing, taxonomy, and quality control of benthic macroinvertebrate samples. U.S. Geological Survey Open-File Report 00–212, Denver.

Oemke, M. P., and T. M. Burton. 1986. Diatom colonization dynamics in a lotic system. Hydrobiologia 139:153–166.

Omernik, J. M. 1987. Ecoregions of the conterminous United State. Annals of the Association of American Geographers 77:118–125.

Ourso, R. T. 2001. Effects of urbanization on benthic macroinvertebrate communities in streams, Anchorage, Alaska. U.S. Geological Survey Water-Resources Investigation Report 01–4278, Anchorage, Alaska.

Paul, M. J., and J. L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32:333–365.

Santa Ana Watershed Project Authority. 2003. About the watershed; population. Available: http://www.sawpa.org/about/watershed.htm#Population. (June 2003)

Shelton, L.R. 1994. Field guide for collecting and processing stream-water samples for the National Water-Quality Assessment Program. U.S. Geological Survey Open-File Report 94–455, Sacramento, California.

Southern California Association of Governments. 1997. Draft 98 regional transportation plan. Southern California Association of Governments, Los Angeles.

Tate, C. M., T. F. Cuffney, G. McMahon, E. M. P. Giddings, J. F. Coles, and H. Zappia. 2005. Use of an urban intensity index to assess urban effects on streams in three contrasting environmental settings. Pages 291–315 *in* L. R. Brown, R. H. Gray, R. M. Hughes, and M. R. Meador, editors. Effects of urbanization on stream ecosystems. American Fisheries Society, Symposium 47, Bethesda, Maryland.

U.S. Army Corps of Engineers. 1994. Water control manual, Prado Dam and Reservoir, Santa Ana River, California. U.S. Army Corps of Engineers, Los Angeles.

U.S. Geological Survey. 1999. The quality of our nation's waters—nutrients and pesticides. U.S. Geological Survey Circular 1225, Reston, Virginia.

Van Dam, H., A. Mertens, and J. Sinkeldam. 1994. A coded checklist and ecological indicator values of freshwater diatoms from the Netherlands. Netherlands Journal of Aquatic Ecology 28:117–133.

Winter, J. G., and H. C. Duthie. 1998. Effects of urbanization on water quality, periphyton and invertebrate communities in a southern Ontario stream. Canadian Water Resources Journal 23:245–257.

Winter, J. G., and H. C. Duthie. 2000. Export coefficient modeling to assess phosphorus loading in an urban watershed. Journal of American Water Resources 36:1053–1061.