DRY WEATHER WATER QUALITY LOADINGS IN ARID, URBAN WATERSHEDS OF THE LOS ANGELES BASIN, CALIFORNIA, USA[1]

Eric D. Stein[2] and Drew Ackerman

ABSTRACT: Dry weather runoff in arid, urban watersheds may consist entirely of treated wastewater effluent and/or urban nonpoint source runoff, which can be a source of bacteria, nutrients, and metals to receiving waters. Most studies of urban runoff focus on stormwater, and few have evaluated the relative contribution and sources of dry weather pollutant loading for a range of constituents across multiple watersheds. This study assessed dry weather loading of nutrients, metals, and bacteria in six urban watersheds in the Los Angeles region of southern California to estimate relative sources of each constituent class and the proportion of total annual load that can be attributed to dry weather discharge. In each watershed, flow and water quality were sampled from storm drain and treated wastewater inputs, as well as from in-stream locations during at least two time periods. Data were used to calculate mean concentrations and loads for various sources. Dry weather loads were compared with modeled wet weather loads under a range of annual rainfall volumes to estimate the relative contribution of dry weather load. Mean storm drain flows were comparable between all watersheds, and in all cases, approximately 20% of the flowing storm drains accounted for 80% of the daily volume. Wastewater reclamation plants (WRP) were the main source of nutrients, storm drains accounted for almost all the bacteria, and metals sources varied by constituent. In-stream concentrations reflected major sources, for example nutrient concentrations were highest downstream of WRP discharges, while in-stream metals concentrations were highest downstream of the storm drains with high metals loads. Comparison of wet vs. dry weather loading indicates that dry weather loading can be a significant source of metals, ranging from less than 20% during wet years to greater than 50% during dry years.

(KEY TERMS: dry weather water quality; urban runoff; nonpoint source pollution; pollutant loads; urban watersheds.)

INTRODUCTION

Increased urbanization often results in increased runoff and pollutant loading to receiving waters (U.S. EPA, 1983, 1995; Schueler, 2000; Davis *et al.*, 2001; Paul and Meyer, 2001). Runoff from highly impervious urban landscapes occurs at amplified magnitude and frequency during both wet and dry weather conditions (Roesner and Bledsoe, 2003). Increased urban

runoff contributes to higher loadings of a broad range of constituents, including nutrients and metals, primarily from discharge of treated wastewater effluent and nonpoint source (i.e., storm drain) runoff (Paul and Meyer, 2001). Many of those pollutants, such as heavy and trace metals, can accumulate and result in downstream bioaccumulation and toxicity (Schueler, 2000). Similarly, bacterial loading to streams in urban areas has been well documented as one of the most common pollutants affecting aquatic systems (Porcella and Sorenson, 1980; Simpson *et al.*, 2002).

Over the past 10 years, management of urban runoff has focused primarily on evaluation and control of stormwater. However, dry weather pollutant discharge may also constitute a significant impact to water quality both in terms of concentration and load (Piechota and Bowland, 2001; McPherson *et al.*, 2002; Ackerman *et al.*, 2003; Stein and Tiefenthaler, 2005). This is especially true for urban watersheds in arid environments where stream flow may be comprised entirely of urban runoff and treated effluent for the majority of the year. Furthermore, during dry weather, streams have lower flow and a lower assimilative capacity than during wet weather, resulting in water column concentrations that may exceed levels that pose a toxicity risk to aquatic organisms (Duke *et al.*, 1999; Bay *et al.*, 2003).

Previous studies have shown that concentrations of many water quality constituents in dry weather flow are generally lower than in wet weather; nevertheless, concentrations may be high enough to be of concern with regard to aquatic life use (Mizell and French, 1995; Duke *et al.*, 1999). Duke *et al.* (1999) and Mizell and French (1995) reported dry weather copper and zinc concentrations of 5–51 $\mu g/l$ and 10–60 $\mu g/l$, in California and Nevada, respectively. Mizell and French (1995) also reported total ammonia levels in the Flamingo Wash in Las Vegas, NV ranging from less than 1 to 9.9 mg/l. Few studies have investigated the contribution of dry weather loading to overall annual load, and those that have found that the proportion of total annual load discharged during the dry weather can vary dramatically based on flow conditions and rainfall patterns. For example, McPherson *et al.* (2002) characterized long-term wet and dry weather flow and loading from the Ballona Creek watershed and determined that between 8% and 42% of the total annual trace metals load occurs during dry weather. This translates to between 100 and 500 kg/yr of dry season loading for most metals.

Previous investigations of dry weather water quality have focused on relative comparisons of constituent concentrations during wet *vs.* dry conditions. Substantially less attention has been devoted to the assessment of dry weather constituent load in several streams of a similar setting. More importantly, no studies have investigated relative sources of dry season loading in arid, urban watersheds and related them to responses in in-stream concentrations. Such information is necessary to allow decision makers to draw general conclusions about expected concentrations and loads during dry conditions and potential sources where management measures may be considered.

The goal of this study was to demonstrate the potential importance of dry weather constituent loading in arid, urban watersheds. This was accomplished by quantifying the relative contribution of dry weather nutrient, metals, and bacteria loading in six urban watersheds in southern California that receive runoff from wastewater effluent, storm drain discharge, or a combination of the two. The predominant sources of the various constituents were also investigated in order to assess the relevance of this research to other arid, urban watersheds, and to provide insight for decisions regarding management of dry weather pollutant loading.

METHODS

Study Areas

The six study watersheds drain the highly urbanized greater Los Angeles area in southern California and represent a range of typical conditions for arid, urban streams (Figure 1, Table 1). The watersheds range in size from the 73 km² lower San Gabriel watershed to the 2,160 km² Los Angeles River watershed. The proportion of developed land use ranges from 49% to 94% of total watershed area. The Ballona Creek watershed drains much of Los Angeles and flows through Marina del Rey to the Pacific Ocean. The Los Angeles River (LAR) extends 90 km, starting from its headwaters in the San Fernando Valley, flowing past downtown Los Angeles, and eventually draining to San Pedro Bay near Long Beach. The remaining four watersheds, lower San Gabriel River, Coyote Creek, San Jose Creek, and Walnut Creek, are catchments in the greater San Gabriel River watershed. During dry weather, flow control structures isolate each of these four watersheds. In addition, the upper (undeveloped) portion of the greater 1,866 km² San Gabriel watershed is completely isolated from the lower watershed during nonstorm periods by a series of dams and diversions; consequently, this area is not addressed by this study.
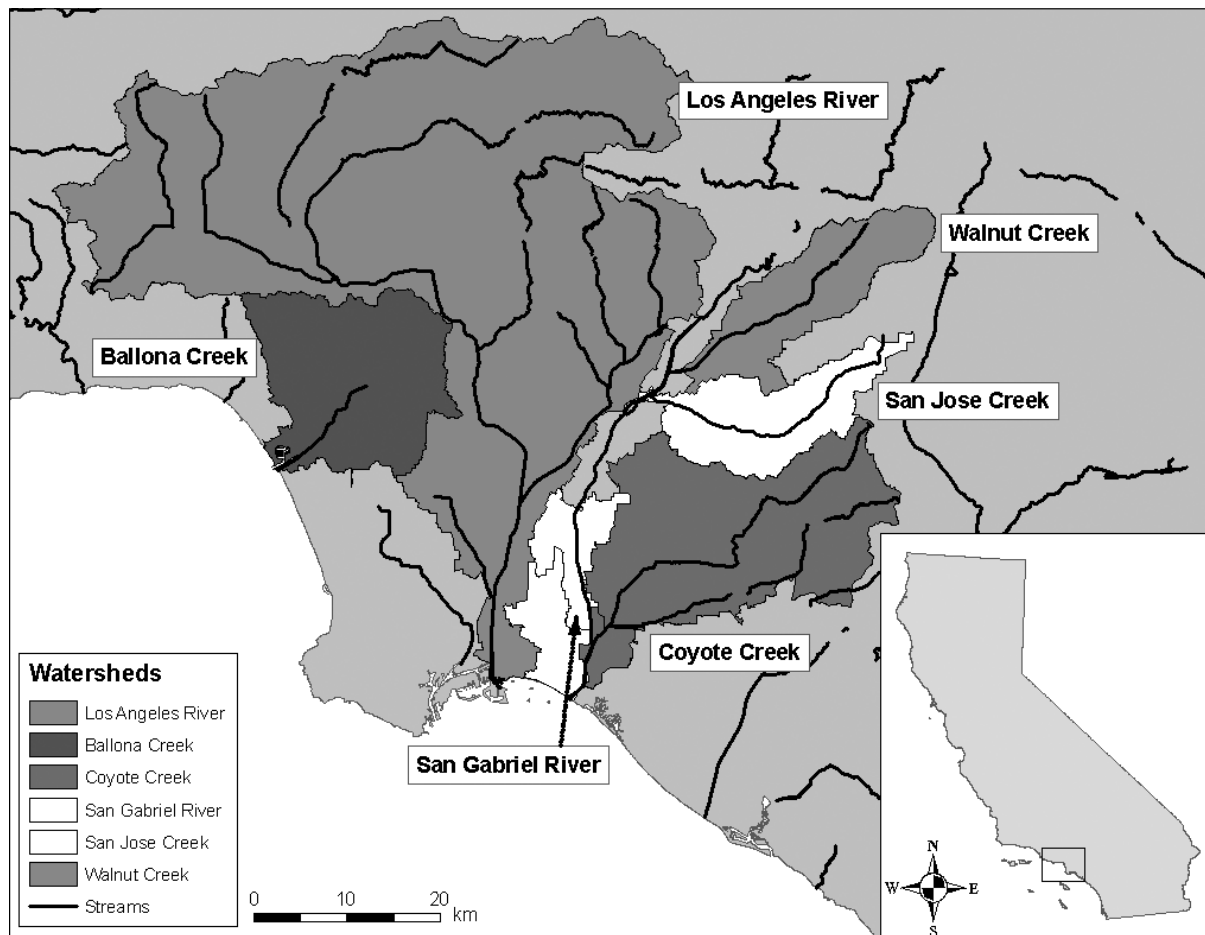
FIGURE 1. Location of the Six Monitored Watersheds.

TABLE 1. Size and Land Use Distribution of Sampled Watersheds.

Watershed	Area (km²)	Percent Land Use in Watershed					
		Commercial (%)	High Density Residential (%)	Industrial (%)	Low Density Residential (%)	Open Space (%)	Other (%)
Los Angeles	2,160	8	7	10	30	43	2
Coyote	487	13	7	14	41	24	1
San Gabriel	73	19	7	13	52	6	3
San Jose	194	11	4	15	41	25	4
Walnut	205	8	4	5	31	51	1
Ballona	338	16	22	7	36	18	1

Source: Southern California Association of Governments 2000 land use data. http://www.scag.ca.gov/landuse/

Sampling

Flow and water quality data were collected from inputs and in-stream locations in each watershed between 2000 and 2004 to characterize sources and effects of dry weather loading. Potential sources that were sampled include point-source discharges from water reclamation plants (WRPs) and untreated non-

point source discharges from storm drains. Industrial discharges, when present, typically occur either directly into the storm drain system or only during the wet season; therefore, they were not considered in this study. Data were collected synoptically in each watershed to provide a "snapshot" of conditions at the time of each sampling event. Each watershed was sampled two or three times (typically over a multiple

TABLE 2. Number of Storm Drains, Water Reclamation Plants (WRPs) and Instream Locations Sampled During the Dry Weather Surveys.

Watershed	Year	Number of Storm Drains	Number of WRPs	Number of Instream Sites
Los Angeles River	September 2000	52	3	19
	July 2001	95	3	24
Coyote	September 2002	19	1	3
	September 2003	20	1	5
San Gabriel	September 2002	19	1	4
	September 2003	9	1	4
San Jose	September 2002	33	1	5
	September 2003	34	1	8
Walnut	September 2002	10	0	2
	September 2003	14	0	3
Ballona Creek	May 2003	35	0	12
	July 2003	37	0	12
	September 2003	47	0	12

year period) to help assess temporal variability in the data.

Storm drains were selected for sampling based on the presence of consistent dry season flow (Table 2). Storm drains along the mainstem creek in each of the six study watersheds were visually surveyed 2-3 times during the month prior to each sampling event. Drains that were flowing during all pre-surveys were included, and drains that were not flowing during at least one of the surveys were excluded. At each storm drain sampled, flow was measured using a timed-volumetric or depth-velocity method (whichever was more appropriate for the conditions at a given location). Storm drain flow was estimated based on the mean of three replicate measurements at each drain. WRP effluent flow data were obtained from the Los Angeles County Sanitation Districts (LACSD) and in-stream flow information was acquired from existing flow gages maintained by the Los Angeles County Department of Public Works (LADPW) (2004).

Storm drains samples were collected by directly filling a single bottle by holding it under the discharge from each drain until the bottle was full. At the in-stream locations, three composite samples were collected at 20-min intervals. Each composite consisted of three grab samples collected at approximately equal intervals across the channel cross-section. A fill bottle was dipped into the stream just below the surface and the collected water was then transferred to a pre-cleaned sample bottle. Upon collection, water quality samples were immediately placed on ice for subsequent analysis. The WRP effluent was collected by LACSD as a 4-h composite sample and analyzed for the parameters listed in Table 3.

Water samples were analyzed for constituents for which the specific water body was listed as impaired by EPAs under Section 303(d) of the Clean Water Act (Table 3). In all cases, except Ballona Creek, this included metals, bacteria, and some form of nutrient

impairment (e.g., nutrients, algae, and total ammonia toxicity). Ballona Creek is listed as impaired for only metals and bacteria; consequently, no nutrient analysis was conducted. Analyses were conducted following protocols provided by Standard Methods for the Examination of Water and Wastewater (Greenberg et al., 2000) and EPA Chemical Methods for the Examination of Water and Wastes (U.S. EPA, 1983). Metals were analyzed using inductively coupled plasma (ICP) mass spectroscopy and bacteria were analyzed using the Idexx QuantiTray® (IDEXX

TABLE 3. Sampled Water Quality Constituents and Their Detection Limits.

Constituent	Detection Limit	Units
Nutrients		
Total ammonia-N	0.02	mg/L
Nitrate-N	0.03	mg/L
Nitrite-N	0.01	mg/L
Nitrite – nitrate	0.05	mg/L
Total Kjeldhal nitrogen	0.1	mg/L
Dissolved phosphorous	0.01	mg/L
Total phosphorous	0.01	mg/L
Metals (total and dissolved)		
Arsenic	0.4	μg/l
Cadmium	0.08	μg/l
Chromium	0.7	μg/l
Copper	1.5	μg/l
Iron	24	μg/l
Lead	3.0	μg/l
Nickel	0.24	μg/l
Zinc	2.0	μg/l
General		
Hardness	2	mg/l
TSS	1	mg/l
Bacteria		
Total coliforms	20	MPN/100 ml
E. coli	10	MPN/100 ml
Enterococcus	10	MPN/100 ml
Fecal coliform*	10	MPN/100 ml

*San Gabriel River watersheds 2002.

Laboratories Inc., Westbrook, ME) chromogenic substrate method. Nitrate and nitrite were analyzed using the cadmium reduction method, total ammonia was analyzed using distillation followed by the automated phenate colorimetric method, and total Kjeldahl nitrogen (TKN) was analyzed using the semi-micro-Kjeldahl digestion/distillation method. Standard quality assurance (QA) measures, including laboratory blanks, matrix spikes, and duplicate samples were analyzed along with every batch of samples. If the data quality objectives were not met for a given batch of samples, the data were either qualified or rejected, depending on the source of the error. Detection limits for all constituents analyzed are shown in Table 3. In all cases, nondetects were assigned a value of zero.

All sampling occurred between June and September during the period when surface flow in the streams originates exclusively from urban runoff. No measurable rain fell within 2 weeks of any sampling period and all sampling was conducted in the morning to minimize potential effects of diurnal variability.

Data Analysis

Means and ranges of flow and water quality concentrations and loads were calculated and analyzed for spatial and temporal patterns. Constituent loads for storm drain and in-stream sites were calculated by multiplying flow and concentration for each sample:

$$\text{Load} = \sum F_i C_i \tag{1}$$

where F_i was the flow at sampling location i averaged over the period when each water sample was collected and C_i was the constituent concentration at location i resulting from the composite grab sampling described above. Where replicate samples were collected, results of the replicates were averaged. In all cases, nondetectable results were assigned a value of zero. For bacteria, results that were greater than the maximum quantifiable levels were assigned the maximum value for that test.

For one selected watershed, additional calculations were performed to estimate annual loadings for both dry and wet weather conditions. Ballona Creek was selected for this analysis because it is representative of highly urbanized watersheds in southern California. Furthermore, there are no WRP discharges into Ballona Creek making it easier to directly compare dry and wet weather urban storm drain runoff (as opposed to treated effluent). Dry weather loads were calculated using the mean downstream concentrations measured during the 2003 sampling events.

Average dry weather flows were derived by multiplying the watershed area by a scaling factor from an analysis that showed average dry weather runoff in the watershed was 180 $\text{m}^3 \, \text{km}^{-2} \, \text{d}^{-1}$. Volume and concentration were then multiplied to get an annual load.

A GIS-based stormwater runoff model was used to estimate wet weather pollutant load based on land use, rainfall, and local water quality information. Ackerman and Schiff (2003) developed a model that established a relationship between rainfall and total storm runoff volume for six land use categories with an associated water quality concentration:

$$\text{Load} = A * i * c * \text{Conc} * k \tag{2}$$

where A is the drainage area (km^2), i the rainfall (mm), c the runoff coefficient (unitless), conc. the water quality concentration (mg/L), and K is the constant (unit conversion factor).

Fifty-two years of rainfall data from the Los Angeles International Airport was used to determine the 10th, 25th, median, 75th, and 90th rainfall volumes. These volumes were scaled using the 30-year orographic average rainfall information (Daly and Taylor, 1998) for each modeled watershed. Annual land use runoff volumes were multiplied by average stormwater concentration from each land use (Ackerman and Schiff, 2003) to estimate annual wet weather loads.

Statistics

Mean storm drain concentrations for nutrients and metals are reported as arithmetic means ± 1 standard error of the mean (SEM). Bacteria levels are reported as geometric means ± 1 SEM. Storm drain concentration data between surveys and across watersheds were log-transformed and compared using a one-way analysis of variance (ANOVA), with a significance of $p < 0.05$. In cases where the ANOVA revealed significant differences in the data set as a whole, a Tukey's means-separation technique was used to identify specific differences between pairwise comparisons (Sokal and Rohlf, 1969). Results were back-transformed for presentation in summary tables to allow easier comparison with other studies.

RESULTS

Flow

Dry weather stream flows were much higher in streams that receive treated WRP effluent than

TABLE 4. Measured Flows (m^3 s^{-1}) in the Sampled Watersheds and Their
Relative Contribution to the Total Volumetric Output From the Watershed.

	Flow (m^3/s)		Percent of Total Volume		
Year	WRP	Storm Drain	WRP (%)	Storm Drain (%)	Flow From Upstream of Study Area (%)
Los Angeles					
2000	3.3	2.0	62	38	
2001	2.0	3.8	34	66	
Coyote					
2002	0.0	0.5	0	87	13
2003	0.5	0.6	41	42	17
San Gabriel					
2002	2.8	0.1	97	3	
2003	3.0	0.1	98	2	
San Jose					
2002	1.7	0.4	73	19	8
2003	2.5	0.6	78	18	4
Walnut					
2002	NA	0.2	-	100	
2003	NA	0.2	-	100	
Ballona					
2003	NA	0.3	-	100	

"Flow from upstream of the study area" is primarily from storm drains that discharge upstream of the area we sampled, but may also include some ground-water discharge from both natural and anthropogenic sources. There were no WRP discharges upstream of the study area in any of the six watersheds. WRP = Water reclamation plant. NA = not analyzed because of lack of flow from the specific source.

those that receive only storm drain discharge (Table 4). For example, average daily flow during our surveys in the San Gabriel and Los Angeles Rivers was 2.9 and 5.5 m^3/s, respectively, much of which was from WRP effluent. In contrast, Ballona Creek and Walnut Creek, which lack WRP discharge and receive flow mainly from storm drain inputs, had average daily flows of 0.3 and 0.2 m^3/s, respectively. The proportion of total volume in each watershed attributed to WRP discharge varied from 34% in the Los Angeles River to 98% in the San Gabriel River. The variability in relative contribution from WRPs was primarily a function of differences in storm drain discharge. In general, the WRP discharge rate was consistently between 1.7 and 3.3 m^3/s. However, mean storm drain discharge varied from 4.6–5.0 × 10^{-3} m^3/s in San Gabriel River to 39.3–40.2 × 10^{-3} m^3/s in the Los Angeles River (Table 5).

The distribution of storm drain flows was comparable among the study watersheds (Figure 2). For every watershed sampled, a few large storm drains dominate the overall daily storm drain volume, with 20% of the flowing storm drains typically accounting for approximately 80% of total storm drain flow. This pattern was the same, despite differences in the size and shape of the six watersheds, and the magnitude of storm drain flows. This suggests that regardless of watershed size

or shape, management of dry season storm drain discharge could be focused on relatively few drains.

TABLE 5. Average Storm Drain Flow (10^{-3} m^3/s^{-1}) in the Six Monitored Watersheds.

Year	Average Flow (10^{-3} m^3/s^{-1})	SEM (10^{-3} m^3/s^{-1})	Number of Drains Sampled
Los Angeles River			
2000	39.3	5.5	52
2001	40.2	4.1	95
Coyote Creek			
2002	28.2	6.5	19
2003	27.3	6.1	20
San Gabriel River			
2002	4.6	1.1	19
2003	5.0	1.7	9
San Jose Creek			
2002	13.1	2.3	33
2003	16.8	2.9	34
Walnut Creek			
2002	20.4	6.5	10
2003	11.4	3.1	14
Ballona Creek			
May 2003	24.2	10.1	25
July 2003	10.4	6.2	28
September 2003	21.4	8.8	30

Mean ± SEM for flowing storm drains. Flow was measured using a timed-volumetric or depth-velocity method (whichever was more appropriate for the conditions at a given location). Storm drain flow was estimated based on the mean of three replicate measurements at each drain.
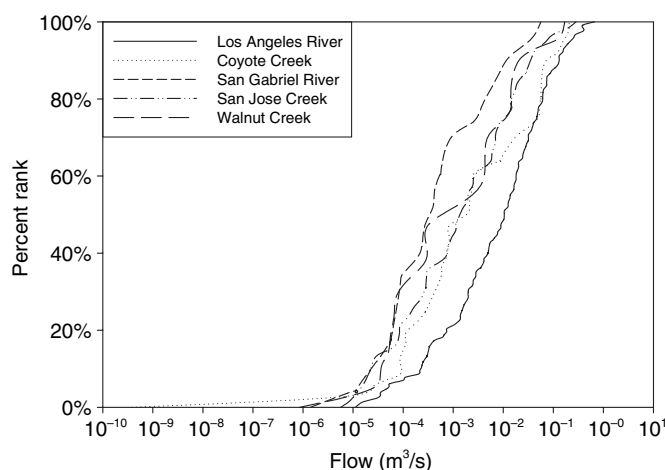
FIGURE 2. Cumulative Distribution of Measured
Storm Drain Flows in the Sampled Watersheds.

Water Quality

Estimated daily dry weather loads for representative bacteria, nutrients, and metals, by source, are summarized in Table 6. Constituent loading exhibited some consistent patterns between watersheds; however, measured concentrations and estimated loads varied considerably within a watershed, between sampling events. For example, estimated metals loading varied by 16% to 357% between successive sampling years, with the mean annual difference being 48% ± 37%. In Ballona Creek, storm drain *E. coli* concentrations varied by 18% to 270% between successive sampling events, with the mean difference being 121% ± 76%.

The majority of nutrients discharged during dry weather were associated with treated WRP effluent. Total ammonia and nitrate+nitrite loads were substantially higher in streams that receive WRP discharge. For example, 247.2-534.7 kg/day and 3,337.2-8.061.6 kg/day of total ammonia were discharged to the San Gabriel and Los Angeles rivers, respectively. In contrast, Walnut Creek, which does not receive any WRP discharge had a total ammonia load of 0.1-0.2 kg/day. The higher total ammonia loads in Los Angeles River are due to a combination of higher storm drain contributions and higher concentrations and volumes from the WRPs. Ammonia concentrations from WRPs discharging to the Los Angeles River are, on average 66% higher than those that discharge to the San Gabriel River. In addition, average daily WRP discharge volumes are 61-116% higher in the Los Angeles River (e.g., 3.8×10^5 m³/day *vs.* 8.2×10^5 m³/day).

Metals loadings were generally higher from storm drains than from WRPs, but there were some differences based on individual metals. With the exception of copper in the Los Angeles River, storm drains accounted for the majority of daily copper and lead load, and the daily loads were comparable between watersheds. In contrast, WRPs contributed 47-91% of the daily zinc load, which was 1-2 orders of magnitude higher than that of copper or lead. If the contribution of WRPs is removed, daily storm drain loads of zinc are still one order of magnitude greater than those of copper and lead. For example, in Ballona Creek daily copper and lead loads were 421.4-571.3 g/day and 137.2-323.6 g/day, whereas daily zinc loads were 1,701.9-1,947.6 g/day. It is interesting to note that storm drains discharged appreciable lead loads, despite the fact that many of the major sources of lead have been restricted by regulations over the last several decades, suggesting that legacy sources of lead persist in these developed watersheds.

Storm drains were the primary source of bacteria in every study watershed. Daily loads of bacteria were comparable between watersheds, with the exception of the Los Angeles River, which had significantly higher loads for all constituents sampled, most probably associated with higher discharge volumes from both WRPs and storm drains (we assumed that the fecal coliform analyzed in 2002 was equivalent to *E. coli*). For all study watersheds *E. coli* loads were in the range of 10^{12} organisms/day.

Mean storm drain concentrations were generally comparable within and between watersheds for each of the constituents sampled (Table 7). Total ammonia concentrations ranged from 0.1 mg/l in Coyote and Walnut Creeks to 1 mg/l in the Los Angeles River. Total phosphate concentrations ranged from 0.3 mg/l in Walnut Creek to 0.8 mg/l in the San Gabriel River. Storm drain concentration of lead was in the 1-3 μg/l range, copper was in 5-25 μg/l range, and zinc was in the 50-200 μg/l range. For the three bacteria indicators sampled, *E. coli* concentrations were in the 10^2-10^3 MPN/100 ml range, Enterococcus were in the 10^4 MPN/100 ml range, and total coliform were in the 10^4-10^5 MPN/100 ml range. Storm drain constituent concentrations in the Los Angeles River were generally comparable (or slightly lower) than those in the other five watersheds. This is in contrast to the higher estimates of loading in the Los Angeles River, once again suggesting that higher loads are due mainly to higher discharge volumes.

The distribution of storm drains with various concentration ranges varied by constituent (Figure 3). For metals and nutrients, less than 20% of the storm drains had appreciable constituent concentrations. From a cumulative mass loading perspective, approximately 90% of the daily storm drain load for most metals was accounted for by 10% of the storm drains

TABLE 6. Pollutant Loading by Source and Watershed.

Constituent/Watershed	Year	Mass Emissions	Units	Percent Contribution		
				Storm Drains (%)	WRPs (%)	Boundary (%)
E. coli						
Los Angeles River	2000	12,022.4	10^9/day	100	0	-
	2001	20,534.6	10^9/day	96	4	-
Coyote Creek	2002	14,156.7	10^9/day	100	0	0
	2003	3,244.6	10^9/day	88	0	12
San Gabriel River	2002	4,646.4	10^9/day	100	0	0
	2003	9.7	10^9/day	100	0	0
San Jose Creek	2002	3,255.0	10^9/day	72	0	28
	2003	2,318.1	10^9/day	91	0	9
Walnut Creek	2002	1,638.3	10^9/day	100	0	0
	2003	2,680.3	10^9/day	100	0	0
Ballona Creek	2003a	NS	10^9/day			
	2003b	299.1	10^9/day	100		
	2003c	5,436.1	10^9/day	100		
Total ammonia						
Los Angeles River	2000	3,357.2	kg/day	0	100	-
	2001	8,061.6	kg/day	32	68	-
Coyote Creek	2002	0.8	kg/day	32	68	0
	2003	64.1	kg/day	0	100	0
San Gabriel River	2002	534.7	kg/day	0	100	0
	2003	247.2	kg/day	0	100	0
San Jose Creek	2002	946.8	kg/day	2	98	0
	2003	174.9	kg/day	2	98	0
Walnut Creek	2002	0.1	kg/day	100	0	0
	2003	0.2	kg/day	100	0	0
Ballona Creek	2003a	NS	kg/day	NS		
	2003b	NS	kg/day	NS		
	2003c	NS	kg/day	NS		
Nitrate + nitrite						
Los Angeles River (Nitrate-N)	2000	363.0	kg/day	63	37	-
	2001	2,529.5	kg/day	31	69	-
Coyote Creek	2002	60.4	kg/day	60	0	40
	2003	240.6	kg/day	38	34	28
San Gabriel River	2002	805.2	kg/day	1	99	0
	2003	479.0	kg/day	0	100	0
San Jose Creek	2002	663.4	kg/day	7	69	24
	2003	674.0	kg/day	11	75	14
Walnut Creek	2002	4.0	kg/day	100	0	0
	2003	8.8	kg/day	100	0	0
Ballona Creek	2003a	NS	kg/day	NS		
	2003b	NS	kg/day	NS		
	2003c	NS	kg/day	NS		
Copper						
Los Angeles River	2000	3,706.0	g/day	15	85	-
	2001	12,296.4	g/day	51	49	-
Coyote Creek	2002	167.6	g/day	88	0	12
	2003	195.9	g/day	100	0	0
San Gabriel River	2002	84.0	g/day	100	0	0
	2003	7.3	g/day	97	3	0
San Jose Creek	2002	291.1	g/day	55	0	45
	2003	384.5	g/day	25	0	75
Walnut Creek	2002	312.8	g/day	100	0	0
	2003	69.2	g/day	100	0	0
Ballona Creek	2003a	NS	g/day			
	2003b	571.3	g/day	100		
	2003c	421.4	g/day	100		

TABLE 6. (Continued)

| Constituent/Watershed | Year | Mass Emissions | Units | Percent Contribution | | |
				Storm Drains (%)	WRPs (%)	Boundary (%)
Lead						
Los Angeles River	2000	533.2	g/day	100	0	-
	2001	0.0	g/day	-	-	-
Coyote Creek	2002	131.3	g/day	84	0	16
	2003	52.2	g/day	51	0	49
San Gabriel River	2002	27.4	g/day	100	0	0
	2003	1.1	g/day	97	3	0
San Jose Creek	2002	82.0	g/day	29	0	71
	2003	55.7	g/day	100	0	0
Walnut Creek	2002	47.1	g/day	100	0	0
	2003	71.0	g/day	100	0	0
Ballona Creek	2003a	NS	g/day			
	2003b	323.6	g/day	100		
	2003c	137.2	g/day	100		
Zinc						
Los Angeles River (Nitrate-N)	2000	11,217.0	g/day	9	91	-
	2001	45,977.7	g/day	41	59	-
Coyote Creek	2002	1,733.6	g/day	89	0	11
	2003	7,937.7	g/day	43	47	10
San Gabriel River	2002	5,363.1	g/day	20	80	0
	2003	7,965.4	g/day	4	96	0
San Jose Creek	2002	7,678.4	g/day	15	76	9
	2003	16,626.8	g/day	19	77	4
Walnut Creek	2002	495.3	g/day	100	0	0
	2003	1,070.7	g/day	100	0	0
Ballona Creek	2003a	NS	g/day			
	2003b	1,941.6	g/day	100		
	2003c	1,701.9	g/day	100		

Ballona loads were calculated using the most downstream flow and water quality concentrations. Boundary is the load entering the stream from upstream boundary of the study area. WRP = Water reclamation plant. NS = not sampled.

TABLE 7. Mean Storm Drain Water Quality Concentration by Watershed. Bacteria data are geometric means ± SEM, All other constituents are arithmetic means ± SEM. NS = not sampled.

| | | LA River | | Coyote Creek | | San Gabriel River | | San Jose Creek | | Walnut Creek | | Ballona | |
		Average	SEM	Average	SEM	Average	SEM	Average	SEM	Average	SEM	Average	SEM
Enterococcus	MPN/100 ml	2,177	897	21,321	7,882	22,225	6,563	12,130	3,337	13,373	3,735	775	204
E. Coli	MPN/100 ml	644	141	1,152	374	1,041	210	754	134	1,767	459	359	77
Total coliform	MPN/100 ml	48,148	17,522	140,637	56,498	149,700	26,688	56,464	10,077	65,209	13,527	25,518	5,698
TSS	mg/l	208	187	13.	6	13	5	34	20	17	4	22	6
Hardness	mg/l	NS		323	21	319	19	415	25	289	31	457	55
Chromium	μg/l	3	2	0.3	0.3	0.4	0.4	0.7	0.7	0	0	2	0.2
Copper	μg/l	25	8	5.8	1	26	9	8	2	13	3	19	3
Iron	μg/l	288	75	469	259	571	301	1,911	1,373	558	105	515	105
Lead	μg/l	2	0.8	2	0.3	3	1	2	0.7	3	0.9	4	1
Nickel	μg/l	3	2	1.	0.9	9	4	5	3	1	1	5	0.4
Zinc	μg/l	122	63	57	7	213	85	117	31	73	7	79	22
Total ammonia-N	mg/l	1	0.2	0.1	0.1	0.6	0.2	0.7	0.3	0.1	0.04	NS	NS
Nitrate-Nitrite	mg/l	1*	0.1*	4	1	3	2	3	0.4	1	0.2	NS	NS
TKN	mg/l	6	3	2	0.3	4	1	2	0.5	2	0.3	NS	NS
Total phosphate-P	mg/l	0.6	0.6	0.4	0.4	0.8	0.5	0.4	0.5	0.3	0.2	NS	NS

*LA River data is nitrate-N

(Figure 4). In contrast, almost all storm drains sampled had high bacteria concentrations, i.e., greater than the state standard of 10^2 MPN/100 ml for *E. coli*. Therefore, unlike metals and nutrients, sources of bacteria appear to be relatively evenly distributed across the study watersheds.
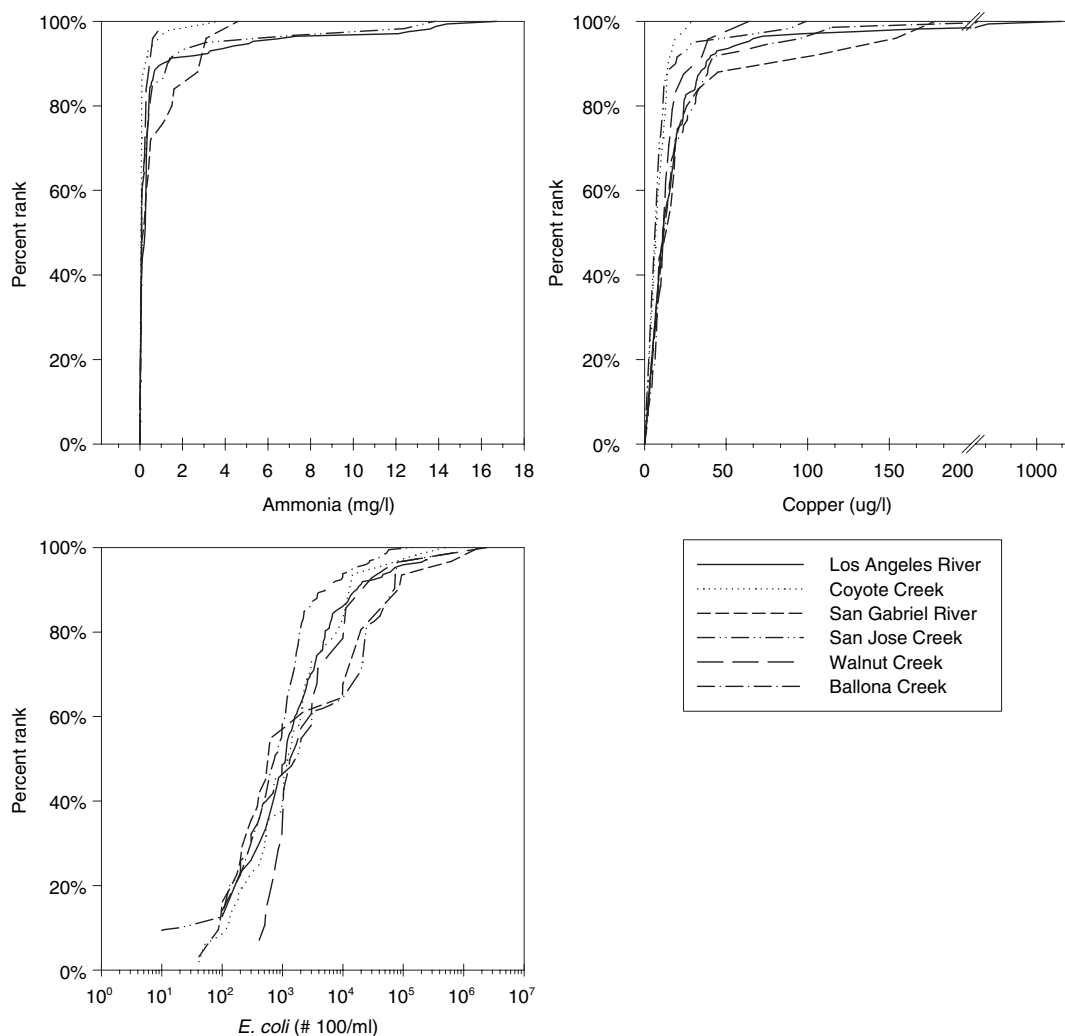
FIGURE 3. Cumulative Distribution of Storm Drain Water Quality Concentrations for Total Ammonia, Copper and *E. coli*.
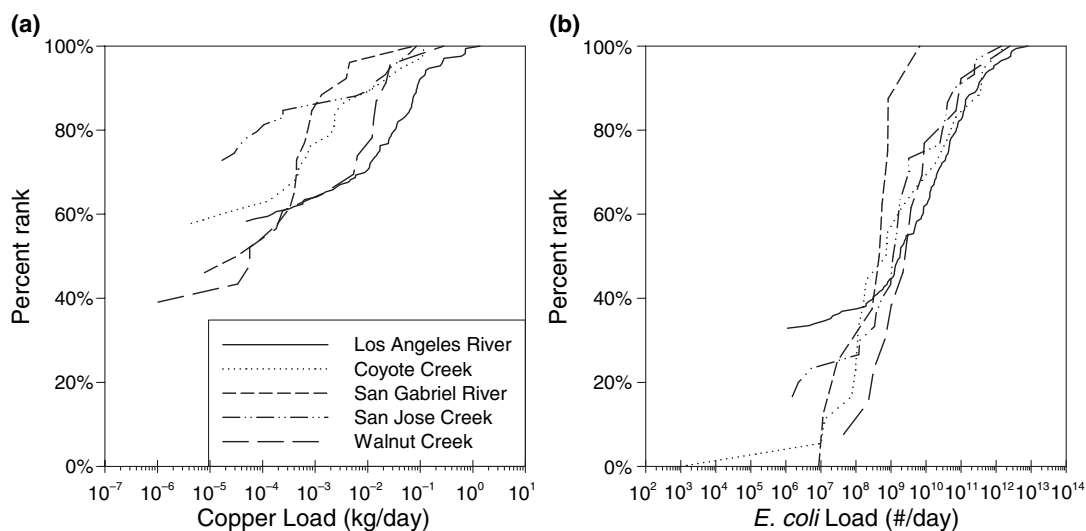

FIGURE 4. Cumulative Distribution of Storm Drain Mass Loading for *E. coli* and Copper.
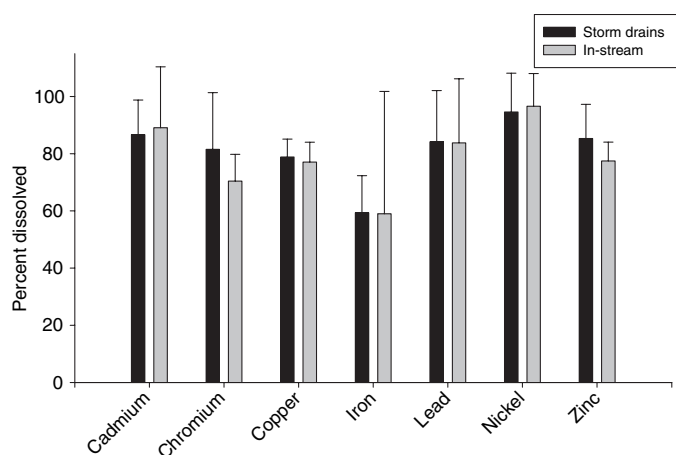
FIGURE 5. Comparison of Percent Dissolved Metals. Percent of total metals as dissolved fraction in samples collected from storm drains and in-stream sites in the Ballona Creek watershed.

Comparison of total and dissolved metals concentrations in the Ballona Creek watershed for both storm drains and in-stream sites showed that, unlike stormwater, dry-season metals occur predominantly in the dissolved phase, ranging from 60% dissolved for iron to 95% dissolved for nickel, with all metal except iron being at least 75% dissolved phase (Figure 5).

The spatial distribution of pollutants in the receiving waters generally reflected the influence of major sources. For example, in-stream total ammonia levels in San Jose Creek and the San Gabriel River were markedly higher downstream of the WRPs (Figure 6). Where storm drains were the only inputs; i.e., upper Coyote Creek and Walnut Creek, nutrient concentrations were consistently low. In Ballona Creek, the highest mean in-stream concentrations and loads of copper, lead, and zinc were observed immediately downstream of two large storm drains (Figure 7). Bacteria concentrations were generally high throughout all stream reaches in the study watersheds, with no apparent spatial pattern, corresponding to the uniformly high bacteria concentrations observed in most storm drains sampled.
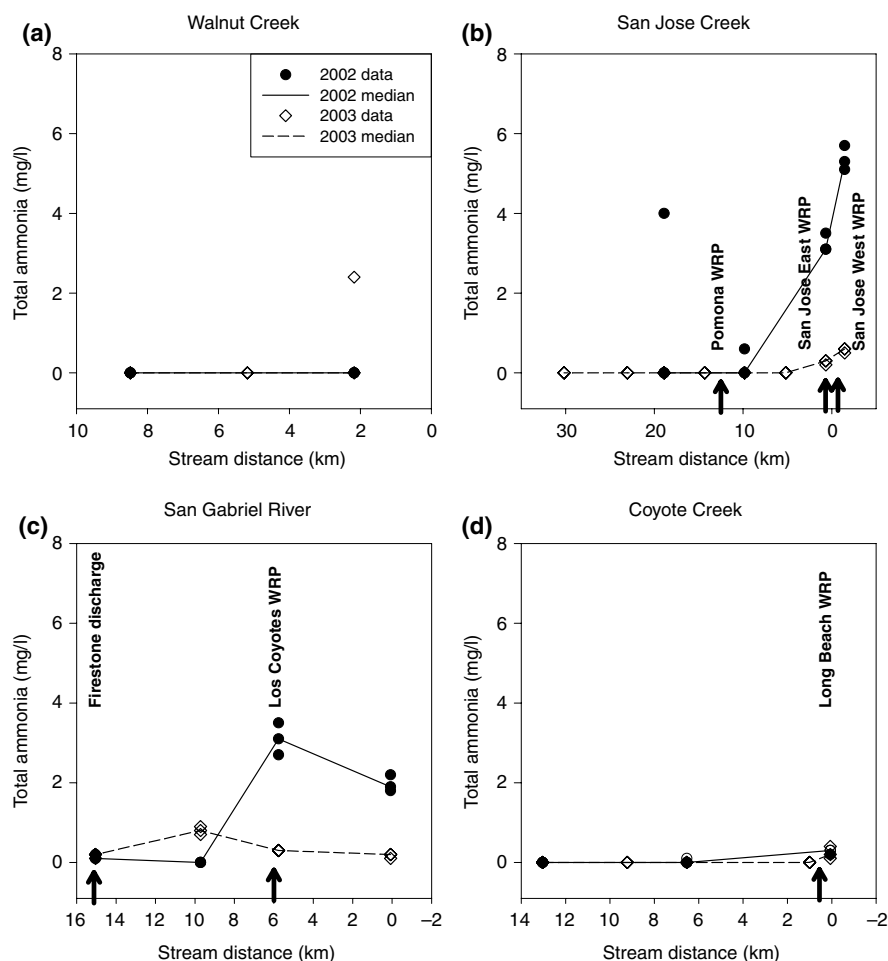

FIGURE 6. In-Stream Total Ammonia Concentrations by Stream and Year.
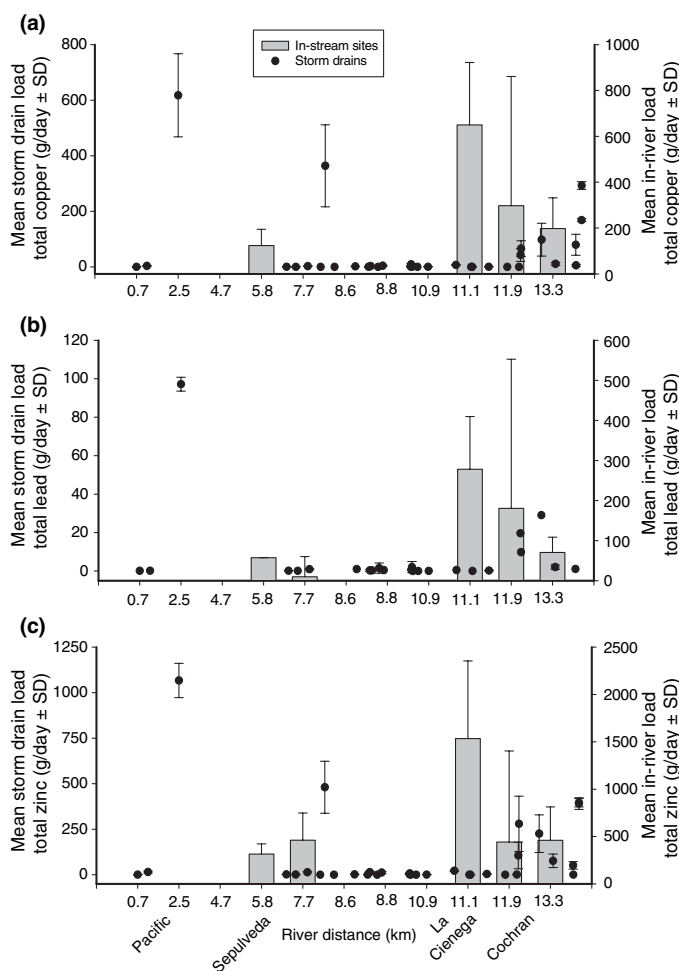Vertical arrows show locations of wastewater reclamation plants (WRPs).

FIGURE 7. Change in Mean In-stream Metals Loads. Graph shows the change in mean in-stream load (± standard deviation) between successive sampling locations in Ballona Creek for (a) total copper, (b) total lead, (c) total zinc (right y-axis). Left y-axis shows mean storm drain load (± standard deviation) by position along Ballona Creek.

Wet vs. Dry Weather Volume and Loadings

For most metals, a substantial portion of the total annual load during years with low rainfall can be attributed to dry weather loading. The relative contribution of dry weather loading to overall annual load for Ballona Creek was estimated by modeling annual stormwater loading under a range of total annual rainfall amounts. Modeled output for stormwater loading was compared to extrapolated dry weather loads based on the empirical instream water quality data collected in this study (Table 8). During dry years (i.e., years in the 10th percentile of rainfall), dry weather discharge may account for up to 57% of the total annual volume. Similarly dry weather loading may account for between 30% and 50% of the total annual load of many metals. For example, when

TABLE 8. Fraction of Total Annual Load of Various Constituents Accounted for by Dry Weather Loading, for Varying Annual Precipitation Conditions in Ballona Creek. Dry weather loads were calculated using the mean downstream concentrations measured during the 2003 sampling events multiplied by the annual nonstorm discharge volume.

	Dry Load as a Percent of Total Annual Load				
Percent of average annual rain	45	64	100	132	176
Rain percentile rank	10	25	50	75	90
Volume	57	48	37	31	25
Fecal Coliform	3	2	2	1	1
Total Coliform	25	19	13	10	8
TSS	19	14	9	7	6
Cadmium	29	22	15	12	9
Chromium	29	22	15	12	9
Copper	46	38	28	23	18
Lead	31	24	17	13	10
Nickel	47	39	29	23	19
Zinc	36	29	21	16	13

Wet weather loads were estimated using a validated watershed model based on land use, rainfall, and local water quality information. Fifty-two years of rainfall data from the Los Angeles International Airport was used to determine the 10th, 25th, median, 75th, and 90th rainfall volumes. These volumes were scaled using the 30-year orographic average rainfall information.

rainfall totals are 45% of the annual average, dry weather loading accounts for 46%, 31%, and 36% of the total annual load of copper, lead, and zinc, respectively. In median years 37% of total annual volume and between 15% and 30% of total annual metals load may occur during the dry season. During wet years (i.e., years in the 90th percentile of rainfall, or with 176% of the average annual rain), stormwater becomes the dominant pollutant source. Nevertheless, dry weather discharge may still comprise 25% of the total annual volume and dry weather metals loading may still comprise between 9% and 19% of the total annual load. The majority of E. coli loading occurs during wet weather in all years. However, concentrations are high during both wet and dry conditions.

DISCUSSION

The results of this study yielded three important conclusions regarding dry weather water quality in watersheds where the nonstorm flow is dominated by urban runoff and other effluent. Estimates of dry weather loading are inherently variable; therefore, repeated measurement will be necessary in order to bound the uncertainty associated with these estimates. Despite this uncertainty, this study indicates

that constituent loading during the dry weather period can comprise a substantial portion of the total annual load in semi-arid urban watersheds, such as those investigated in this study. Finally, comprehensive measurement of dry weather discharges can provide insight into specific sources of constituents.

Variability in Loading Estimates

Dry weather constituent concentrations were highly variable, resulting in uncertainty in the loading estimates. This study presents data collected at several points in time that provide a "snapshot" of conditions. The main sources of uncertainty are the inherent (and unpredictable) variability in both flow and concentration that may occur at time scales from hours to months (Hatje *et al.*, 2001). These fluctuations are resulting from a variety of sources, such as illicit discharges, permitted periodic discharges of industrial or construction-related effluent, diurnal patterns, and random variability in storm drain concentration and flow. For example, copper concentrations in Ballona Creek varied by 137% between the May and July sampling. On an inter-annual basis, copper concentrations varied by up to fivefold: Mean copper concentrations were 29 $\mu g/l$ in 1999 (McPherson *et al.*, 2002), "not-detected" in 2002 (City of Los Angeles, unpublished monitoring data – http://www.lacity.org/SAN/wpd/index1.htm), and 6 $\mu g/l$ in 2003 (this study). Others have reported similar variability in metals concentrations. For example, Nimick *et al.* (2003) reported that metals concentrations in streams vary by 100-500% over the course of a day, and Hatje *et al.* (2001) reported fluctuations in metals concentrations of 100-200% on a month-to-month basis, with variability increasing with increasing anthropogenic input.

Bacterial counts typically vary by up to five orders of magnitude on daily, seasonal, and inter-annual scales. Furthermore, between 5% and 22% of storm drain samples exceed the maximum detectable bacterial counts (depending on the specific indicator). Therefore, mean concentrations reported from storm drains likely underestimate the actual bacteria levels being discharged to urban streams. The greater variability observed in bacteria *vs.* metals data (i.e., several orders of magnitude *vs.* several fold) is expected. As living organisms, many processes that do not influence metals, such as growth, die-off, and random fluctuations in population size, may affect bacterial counts. In addition, the analytic method used to quantify bacteria is based on colorimetric estimation of bacterial density, which is inherently less precise than the approach used to quantify metals (mass spectroscopy). It is important to note that variability in estimations of bacteria density is not solely a function of the method

chosen. All three methods typically used, membrane filtration (MF), multiple tube fermentation (MTF), and chromogenic substrate (CS, the method used by this study) are based upon measuring products of bacterial growth. This approach can result in errors associated with growth of bacteria in the lab and in interpretation of the growth results by laboratory technicians. Noble *et al.* (2003) compared results from 22 laboratories using MF, MTF, and CS using laboratory fabricated samples and found the three methods to be comparable. More recently, Griffith *et al.* (2006) compared results from 26 laboratories using ambient water samples in a variety of matrices and also found the three methods to be comparable.

In addition to variability in concentration, flow may also vary at multiple time scales. Review of daily flow data from the Los Angeles County Department of Public Works shows that nonstorm flows in Ballona Creek and the Los Angeles River can vary by up to 5-fold over the course of a year. For example dry weather flow in Ballona Creek may range from 0.2 m^3/s to 1.4 m^3/s. In a system such as the San Gabriel River (Figure 9) or Coyote Creek, that is affected by WRPs that discharge intermittently, dry weather flow may vary by up to 40-fold over the course of a year (i.e., Coyote Creek flows ranged from 0.2–8.3 m^3/s). In addition, dry weather flow in arid, urban watersheds varies in a predictable manner by up to 40% over the course of a single day.

The manner in which samples with nondetectable levels of a particular constituent are treated may also affect overall estimates of load and introduce additional uncertainty into conclusions regarding the relative magnitude of sources of dry weather loading. Nondetectable values could be assigned a value ranging from zero to the detection limit. The degree to which this choice influences general conclusions about loading depends on the frequency of nondetectable values. For the metals focused on in this study, only storm drain lead samples had a substantial fraction of nondetectable values (60%). If we had assumed that nondetectable values were equal to the detection limit, our estimate of storm drain load would have increased by 100%. Similarly, the manner in which samples with nondetectable levels of a particular metal were treated may affect conclusions regarding distribution of load among sources. Due to the large volumetric input by the WRPs, small differences in these estimates can have a dramatic effect on the overall distribution of trace metal sources. For example, assuming that nondetectable samples for nickel were equal to a concentration of zero led to estimation that storm drains account for 100% of the nickel loading. If this assumption were changed to a concentration equal to one-half the detection limit, the WRPs would become the dominant source for nickel as well as five of the six other
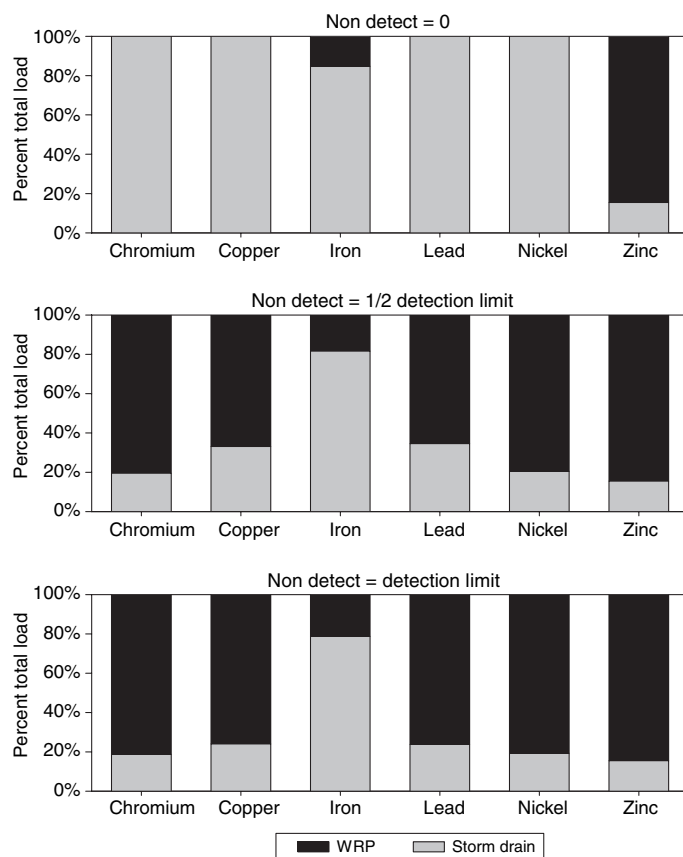
FIGURE 8. Effects of NonDetected Constituents on Mass Loadings of Metals in the Greater San Gabriel River Watershed. Graphs show how the estimated relative distribution of load varies based on the values assigned to "nondetect" sample results. WRP = water reclamation plant.

FIGURE 9. San Gabriel River Flow
Variability During the 2002 Sampling Event.

metals analyzed. (Figure 8). As previously stated, we chose a conservative strategy by assigning a value of zero to all nondetectable results so as to not artificially assume a load associated with a particular source.

Uncertainty in loading estimates can be reduced by compiling data representing numerous observations over time. This is typically accomplished by either repeated field measures or by long-term simulation models. For dry weather loading estimates, we believe simulation models are not as useful as they are for bounding the uncertainty associated with stormwater loading. Variability in stormwater loading (within a given watershed) is largely a function of differences in rainfall-runoff patterns, which conform to fairly well established physical principles. In contrast, variability in dry weather loading is largely a function of unpredictable and somewhat random events (e.g., discharges from industrial sites, construction sites, or illicit sources). Consequently, estimates of dry weather loading should be based on repeated measures of concentration and flow over time that can be used to bound the expected range of variability.
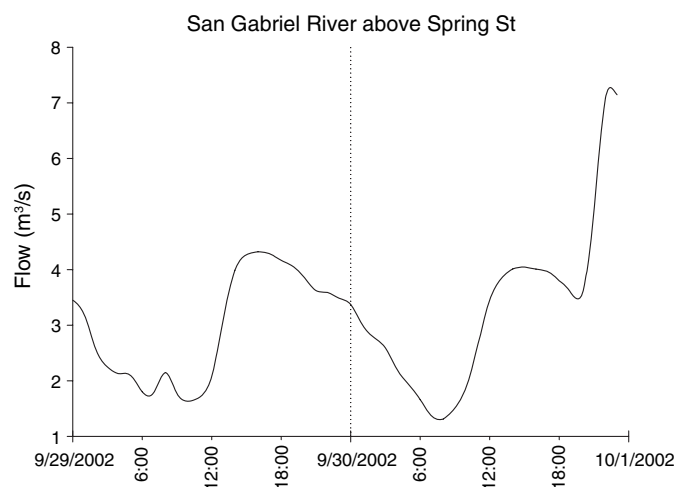
Contribution of Dry Weather Loading

Data from the six watersheds sampled in this study showed similar patterns, allowing us to conclude that dry weather loading can be an appreciable source of total annual constituent load for the watersheds in the greater Los Angeles area, in recognition of the inherent variability associated with dry weather water quality.

In arid systems, water quality loading is of particular interest because the majority of dry weather stream flow is derived from urban runoff and effluent discharge for approximately 85% of the year. Although concentrations may be appreciably lower during dry weather conditions, as noted by Mizell and French (1995) and Duke et al. (1999), this study showed that in arid, urban watersheds, dry weather loading could contribute between 20% and 50% of the total annual metals load. This analysis was based on modeling for Ballona Creek, which lacks WRP discharge, thereby allowing a direct comparison of loading from dry and wet weather urban runoff. In watersheds where WRPs are an appreciable source, dry weather loading would comprise a larger proportion of the total annual load than estimated for Ballona Creek for certain constituents, such as zinc and ammonia.

The emphasis on load vs. concentration is important from a management perspective for several reasons. First, many urban streams are subject to Total Maximum Daily Load (TMDL) requirements, that limit mass loadings to ensure that water quality standards are met and require managers to ensure that both dry and wet weather loads meet regulatory requirements. Second, many urban streams drain to lentic or semi-lentic waterbodies, such as lakes or

estuaries, where accumulation of toxic compounds in sediment and subsequent bioaccumulation may be of concern. In this case, dry weather loading is important not only in terms of its contribution to the total annual load, but because the lower velocities associated with dry weather flow (*vs*. storm flow) may facilitate deposition of metals and other toxic compounds in downstream water bodies. Furthermore, dry weather loads occur predominantly in the dissolved form (Figure 5), which is more bioavailable to organisms than the particle-bound constituents that are predominant in stormwater (SCCWRP, unpublished data). The exception to this is iron, which has a relatively higher particulate phase than other metals. This is likely due to a combination of the fact that iron occurs in much higher concentrations than other metals and transforms to its reduced (soluble) form at a much lower redox potential. Consequently, reductive dissolution is less effective at transforming particulate iron. Third, management strategies for dry and wet weather loading are typically different. Stormwater management typically focuses on retention or detention of flows, whereas dry weather runoff control focus on treatment, diversion, infiltration, and source control.

For bacteria, concentration (or counts) is a more appropriate management endpoint because unlike metals and other toxic compounds, bacteria do not typically accumulate in receiving water sediments. Dry weather counts of *E. coli* are typically around 10^3 MPN/100 ml. This level is several orders of magnitude lower than the 10^4–10^6 MPN/100 ml typically observed in urban stormwater (City of Los Angeles, unpublished monitoring data – http://www.lacity.org/SAN/wpd/index1.htm); however it is still consistently above the state freshwater standard of 400 MPN/100 ml.

CONCLUSIONS

This study has demonstrated that in semi-arid, urban watersheds, dry weather loadings can comprise a substantial proportion of the total annual load for a range of constituents, especially during years with low rainfall. Consequently, water quality management strategies should focus on dry weather runoff in addition to stormwater loadings, which are typically the focus of most management efforts.

Source identification can be particularly problematic in urban watersheds where flows and constituent concentrations vary in somewhat unpredictable ways. Investigation of all potential sources of dry weather loading over multiple time periods can help reduce the uncertainty and allow managers to begin identifying patterns that can be use to focus management efforts. Using this approach, several consistent patterns were observed in the six watersheds analyzed in this study: The WRPs consistently contributed the majority of nutrients to the receiving waters, while storm drains contributed the majority of bacteria. In the case of trace metals, the dominant source varied by specific metal. Analysis of storm drain loading from the six study watersheds revealed that relatively few storm drains (i.e., < 20%) had high concentrations and contributed relatively large metal loads, whereas most drains (i.e., > 90%) had high concentrations of bacteria. The consistently high bacteria concentrations throughout the system make establishing linkages between sources and receiving water concentrations difficult. In addition, potential in-stream sources of bacteria (e.g., birds or regrowth) were not evaluated in this study. Therefore, from a management perspective, nutrients and metals could be managed by targeting specific sources; whereas, management of bacteria loading would require a more coordinated systematic approach.

REFERENCES

Ackerman, D. and K. Schiff, 2003. Modeling Stormwater Mass Emissions To The Southern California Bight. *Journal of Environmental Engineering* 129:308-317.

Ackerman, D., K.C. Schiff, H. Trimm, and M. Mullin, 2003. Characterization of Water Quality in the Los Angeles River. *Bulletin of the Southern California Academy of Sciences* 102:17-25.

Bay, S., B.H. Jones, K. Schiff and L. Washburn, 2003. Water Quality Impacts of Stormwater Discharges to Santa Monica Bay. *Marine Environmental Research* 56:205-223.

Daly, C. and G. Taylor, 1998. *California Average Monthly or Annual Precipitation, 1961–90.* http://www.ftw.nrcs.usda.gov/prism/prism.html.

Davis, A.P., M. Shokouhian, and S. Ni, 2001. Loading Estimates of Lead, Copper, Cadmium, and Zinc in Urban Runoff From Specific Sources. *Chemosphere* 44:997-1009.

Duke, L.D., T.S. Lo, and M.W. Turner, 1999. Chemical Constituents in Storm Flow *vs*. Dry Weather Discharges in California Storm Water Conveyances. *Journal of American Water Resource Association* 35:821-836.

Greenberg, A.E., L.S. Clesceri, and A.D. Eaton, 2000. *Standard Methods for the Examination of Water and Wastewater*, 20th Edition. American Public Health Association, Washington, District of Columbia.

Griffith, J.G., L.A. Aumand, I.M. Lee, C.D. McGee, L.L. Othman, K.J. Ritter, K.O. Walker, and S.B. Weisberg, 2006. Comparison and Verification of Bacterial Water Quality Indicator Measurement Methods Using Ambient Coastal Water Samples. *Environmental Monitoring and Assessment* 116:335-344.

Hatje, V., K. Rae, and G.F. Birch, 2001. Trace Metal and Suspended Solid Concentrations in Freshwater: The Importance of Small-Scale Temporal Variation. *Journal of Environmental Monitoring* 3:251-256.

Los Angeles County Department of Public Works (LADPW), 2004. *Imported and Reclaimed Water Data 2002-2003 Final*. http://www.ladpw.org/wrd/spreadingground/watercon, April 29, 2004.

McPherson, T.N., S.J. Burian, H.J. Turin, M.K. Stenstrom, and I.H. Suffet, 2002. Comparison of the Pollutant Loads in Dry and Wet Weather Runoff in a Southern California Watershed. *Water Science and Technology* 45:255-261.

Mizell, S.A. and R.H. French, 1995. Beneficial Use Potential of Dry Weather Flow in The Las Vegas Valley, Nevada. *Water Resources Bulletin* 31:447-461.

Nimick, D., C.H. Gammons, T.E. Cleasby, J.P. Madison, D. Skaar, and C.M. Brick, 2003. Diel Cycles in Dissolved Metal Concentrations in Streams: Occurrence and Possible Causes. *Water Resources Research* 39:1247.

Noble, R.T., S.B. Weisberg, M.K. Leecaster, C.D. McGee, K. Ritter, K.O. Walker, and P.M. Vainik, 2003. Comparison of Beach Bacterial Water Quality Indicator Measurement Methods. *Environmental Monitoring and Assessment* 81:301-312.

Paul, M.J. and J.L. Meyer, 2001. Streams in the Urban Landscape. *Annual Review of Ecological Systems* 32:333-365.

Piechota, T.C. and L. Bowland, , 2001. Characterization of Wet and Dry Weather Flows From Urban Runoff in An Arid Region. In: *Proceedings of the Specialty Symposium Held in Conjunction with The World Water and Environmental Resources Congress, May 20-24, 2001, Orlando, Florida*, B.K. Panigrahi, * (ed) (Editor). American Society of Civil Engineers, Reston, Virginia: 195-203.

Porcella, D. B. and D. L. Sorenson, , 1980, *Characteristics of Nonpoint Source Urban Runoff and its Effects on Stream Ecosystems, EPA-600/3-80-032*. United States Environmental Protection Agency, Corvallis, Oregon, pp. 99.

Roesner, L.A. and B.P. Bledsoe, , 2003. *Physical Effects of Wet Weather Flows on Aquatic Habitats: Present Knowledge and Research Needs*. Water Environment Research Foundation Technical Report # 00-WSM-4, Alexandria, Virginia, pp. 197.

Schueler, T.R, 2000, The importance of imperviousness. *In*: *The Practice of Watershed Protection, Watershed Protection Techniques*, T.R. Schueler and H.K. Holland (Editors). Center for Watershed Protection, Ellicott City, Maryland pp. 100-111.

Simpson, J.M., J.W.S. Domingo, and D.J. Reasoner, 2002. Microbial Source Tracking: State of the Science. *Environmental Science and Technology* 36:5279-5288.

Sokal, R.R. and F.J. Rohlf, 1969. *Biometry: The Principles and Practice Of Statistics In Biological Research*, W.H. Freeman and Company, San Francisco, California, 776 pp.

Stein, E.D. and L.L. Tiefenthaler, 2005. Dry-Weather Metals and Bacteria Loading in an Arid, Urban Watershed: Ballona Creek, California. *Water, Air, and Soil Pollution* 164:367-382.

U.S. EPA, 1983. *Chemical Methods for the Examination of Water and Wastes*. EPA-600/4-79-020, Cincinnati, Ohio.

U.S. EPA, 1995. *National Water Quality Inventory; 1994 Report to Congress*. EPA/841/R-95/005, Washington, District of Columbia.