Dry weather water quality loadings in arid, urban watersheds of the Los Angeles Basin, California, USA

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Abstract

Dry-weather runoff in arid, urban watersheds may consist entirely of treated wastewater effluent and/or urban non-point 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 was used to calculate mean concentrations and loads for various sources. Dry weather loads were compared to 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.

Introduction

Increased urbanization often results in increased runoff and pollutant loading to receiving waters (USEPA 1995, Schueler and Holland 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 non-point 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 ten 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 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 to 51 $\mu g\ L^{-1}$ and 10 to 60 $\mu g\ L^{-1}$, in California and Nevada, respectively. Mizell and French (1995) also reported total ammonia levels in the Flamingo Wash in Las Vegas, Nevada, ranging from less than 1 to 9.9 mg L^{-1}. 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 $year^{-1}$ 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 km^2 lower San Gabriel watershed to the 2,160 km^2 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 km^2 San Gabriel watershed is completely isolated from the lower watershed during non-storm periods by a series of dams and diversions; consequently, this area is not addressed by this study.

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

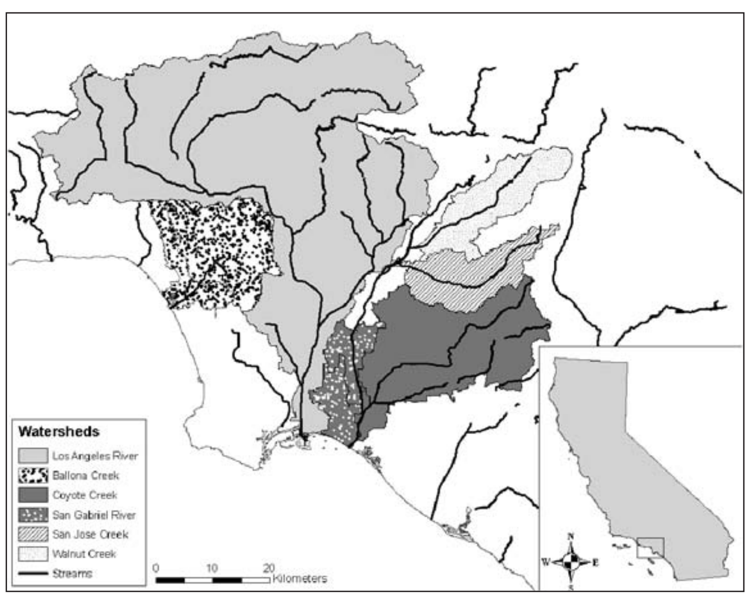

Figure 1. Location of the six monitored watersheds.

Table 1. Size and land use distribution of sampled watersheds.

		PERCENT LAND USE IN WATERSHED					
Watershed	Area (km2)	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.

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 year period) to help asses 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 was 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).

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-minute 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-hour 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 the EPA 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, 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 (USEPA 1983). Metals were analyzed using inductively coupled plasma (ICP) mass spectroscopy and bacteria were analyzed using the Idexx QuantiTray® chromogenic substrate method. Nitrate and nitrite were analyzed using the cadmium reduction method,

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	Sep-00	52	3	19
	Jul-01	95	3	24
Coyote	Sep-02	19	1	3
	Sep-03	20	1	5
San Gabriel	Sep-02	19	1	4
	Sep-03	9	1	4
San Jose	Sep-02	33	1	5
	Sep-03	34	1	8
Walnut	Sep-02	10	0	2
	Sep-03	14	0	3
Ballona Creek	May-03	35	0	12
	Jul-03	37	0	12
	Sep-03	47	0	12

Table 3. Sampled water quality constituents and their detection limits.

Constituent	Detection Limit	Units
Nutrients		
Total Ammonia-N	0.02	mg L^{-1}
Nitrate-N	0.03	mg L^{-1}
Nitrite-N	0.01	mg L^{-1}
Nitrite + Nitrate	0.05	mg L^{-1}
Total Kjeldhal Nitrogen	0.10	mg L^{-1}
Dissolved Phosphorous	0.01	mg L^{-1}
Total Phosphorous	0.01	mg L^{-1}
Metals (total and dissolved)		
Arsenic	0.40	µg L^{-1}
Cadmium	0.08	µg L^{-1}
Chromium	0.70	µg L^{-1}
Copper	1.50	µg L^{-1}
Iron	24.00	µg L^{-1}
Lead	3.00	µg L^{-1}
Nickel	0.24	µg L^{-1}
Zinc	2.00	µg L^{-1}
General		
Hardness	2.00	mg L^{-1}
Total Suspended Solids	1.00	mg L^{-1}
Bacteria		
Total Coliforms	20.00	MPN 100 ml^{-1}
E. coli	10.00	MPN 100 ml^{-1}
Enterococcus	10.00	MPN 100 ml^{-1}
Fecal Coliform*	10.00	MPN 100 ml^{-1}

* San Gabriel River watersheds 2002 only.

total 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, non-detects 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 two 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:

$$Load = \sum F_i C_i$$

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, non-detectable 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 done 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 m^3 km^{-2} day^{-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,

$$Load = A * i * c * Conc * K$$

where: A is drainage area (km^2); i is rainfall (mm); c is the unitless runoff coefficient; Conc is the water quality concentration (mg L^{-1}); 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 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^{-1}, 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^{-1}, respectively. The proportion of total volume in each watershed attributed to WRP discharge varied from 34% in the Los Angles 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^{-1}. However, mean storm drain discharge varied from 4.6 - 5.0 x 10^{-3} m^3 s^{-1} in San Gabriel River to 39.3 - 40.2 x 10^{-3} m^3 s^{-1} 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.

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^{-1} and 3,337.2 - 8.061.6 kg day^{-1} of total ammonia were discharged to the San Gabriel and Los Angeles rivers, respectively. In contrast, Walnut Creek, which

Table 4. Measured flows (m^3 s^{-1}) in the sampled watersheds and their relative contribution to the total volumetric output from the watershed. WRP = Water reclamation plant. NA = not analyzed because of lack of flow from the specific source.

		Flow m^3s^{-1}		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%	

Table 5. Average storm drain flow (10-3 m3 s-1) in the six monitored watersheds. 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.

		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
	Sept 2003	21.4	8.8	30

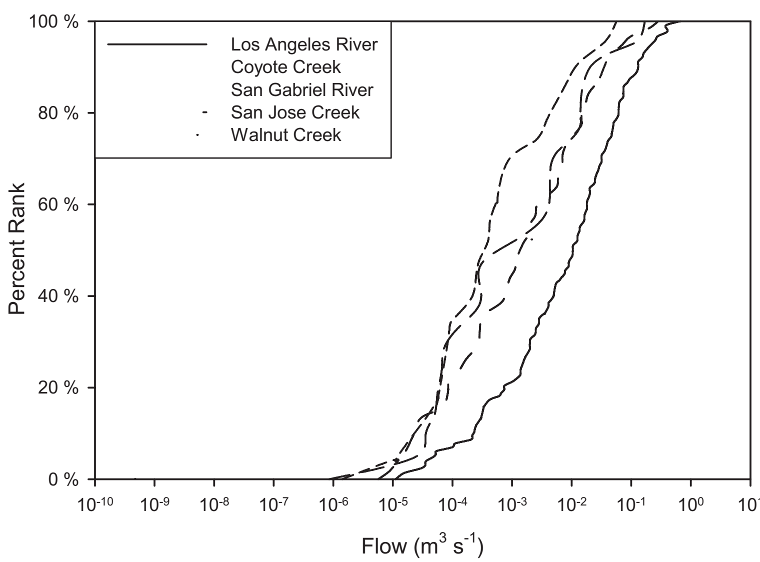

Figure 2. Cumulative distribution of measured storm drain flows in the sampled watersheds.

does not receive any WRP discharge had a total ammonia load of 0.1 - 0.2 kg day^{-1}. 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 dis)erage 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 x 10^5 m^3 day^{-1} vs. 8.2 x 10^5 m^3 day^{-1}).

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^{-1} and 137.2 - 323.6 g day^{-1}, whereas daily zinc loads were 1,701.9 - 1,947.6 g day^{-1}. 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, 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^{-1}.

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^{-1} in Coyote and Walnut Creeks to 1 mg L^{-1} in the Los Angeles River. Total phosphate concentrations ranged from 0.3 mg L^{-1} in Walnut Creek to 0.8 mg L^{-1} in the San Gabriel River. Storm

Table 6. Pollutant loading by source and watershed. 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.

					Percent Contribution							Percent Contribution		
			Mass Emissions	Units	Storm Drains	WRPs	Boundary			Mass Emissions	Units	Storm Drains	WRPs	Boundary
E. coli	Los Angeles	2000	12,022.40	10^9/d	100%	0%	-	Copper	2000	3,706.00	g/d	15%	85%	-
		2001	20,534.60	10^9/d	96%	4%	-		2001	12,296.40	g/d	51%	49%	-
	Coyote	2002	14,156.70	10^9/d	100%	0%	0%		2002	167.6	g/d	88%	0%	12%
		2003	3,244.60	10^9/d	88%	0%	12%		2003	195.9	g/d	100%	0%	0%
	San Gabriel	2002	4,646.40	10^9/d	100%	0%	0%		2002	84	g/d	100%	0%	0%
		2003	9.7	10^9/d	100%	0%	0%		2003	7.3	g/d	97%	3%	0%
	San Jose	2002	3,255.00	10^9/d	72%	0%	28%		2002	291.1	g/d	55%	0%	45%
		2003	2,318.10	10^9/d	91%	0%	9%		2003	384.5	g/d	25%	0%	75%
	Walnut	2002	1,638.30	10^9/d	100%	0%	0%		2002	312.8	g/d	100%	0%	0%
		2003	2,680.30	10^9/d	100%	0%	0%		2003	69.2	g/d	100%	0%	0%
	Ballona	2003a	NS	10^9/d					2003a	NS	g/d			
		2003b	299.1	10^9/d	100%				2003b	571.3	g/d	100%		
		2003c	5,436.10	10^9/d	100%				2003c	421.4	g/d	100%		
Total Ammonia	Los Angeles	2000	3,357.20	kg/d	0%	100%	-	Lead	2000	533.2	g/d	100%	0%	-
		2001	8,061.60	kg/d	32%	68%	-		2001	0	g/d	-	-	-
	Coyote	2002	0.8	kg/d	32%	68%	0%		2002	131.3	g/d	84%	0%	16%
		2003	64.1	kg/d	0%	100%	0%		2003	52.2	g/d	51%	0%	49%
	San Gabriel	2002	534.7	kg/d	0%	100%	0%		2002	27.4	g/d	100%	0%	0%
		2003	247.2	kg/d	0%	100%	0%		2003	1.1	g/d	97%	3%	0%
	San Jose	2002	946.8	kg/d	2%	98%	0%		2002	82	g/d	29%	0%	71%
		2003	174.9	kg/d	2%	98%	0%		2003	55.7	g/d	100%	0%	0%
	Walnut	2002	0.1	kg/d	100%	0%	0%		2002	47.1	g/d	100%	0%	0%
		2003	0.2	kg/d	100%	0%	0%		2003	71	g/d	100%	0%	0%
	Ballona	2003a	NS	kg/d	NS				2003a	NS	g/d			
		2003b	NS	kg/d	NS				2003b	323.6	g/d	100%		
		2003c	NS	kg/d	NS				2003c	137.2	g/d	100%		
Nitrate + Nitrite	Los Angeles	2000	363	kg/d	63%	37%	-	Zinc	2000	11,217.00	g/d	9%	91%	-
	(Nitrate-N)	2001	2,529.50	kg/d	31%	69%	-		2001	45,977.70	g/d	41%	59%	-
	Coyote	2002	60.4	kg/d	60%	0%	40%		2002	1,733.60	g/d	89%	0%	11%
		2003	240.6	kg/d	38%	34%	28%		2003	7,937.70	g/d	43%	47%	10%
	San Gabriel	2002	805.2	kg/d	1%	99%	0%		2002	5,363.10	g/d	20%	80%	0%
		2003	479	kg/d	0%	100%	0%		2003	7,965.40	g/d	4%	96%	0%
	San Jose	2002	663.4	kg/d	7%	69%	24%		2002	7,678.40	g/d	15%	76%	9%
		2003	674	kg/d	11%	75%	14%		2003	16,626.80	g/d	19%	77%	4%
	Walnut	2002	4	kg/d	100%	0%	0%		2002	495.3	g/d	100%	0%	0%
		2003	8.8	kg/d	100%	0%	0%		2003	1,070.70	g/d	100%	0%	0%
	Ballona	2003a	NS	kg/d	NS				2003a	NS	g/d			
		2003b	NS	kg/d	NS				2003b	1,941.60	g/d	100%		
		2003c	NS	kg/d	NS				2003c	1,701.90	g/d	100%		

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.

	Units	Los Angeles		Coyote		San Gabriel		San Jose		Walnut		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
Total Suspended Solids	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

drain concentration of lead was in the 1 - 3 µg L^{-1} range, copper was in 5 - 25 µg L^{-1} range, and zinc was in the 50 - 200 mg L^{-1} range. For the three bacteria indicators sampled, *E. coli* concentrations were in the 10^2 - 10^3 MPN 100 ml^{-1} range, *Enterococcus* were in the 10^4 MPN 100 ml^{-1} range, and total coliform were in the 10^4 - 10^5 MPN 100 ml^{-1} 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 (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^{-1} for *E. coli*. Therefore, unlike metals and nutrients, sources of bacteria appear to be relatively evenly distributed across the study watersheds.

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.

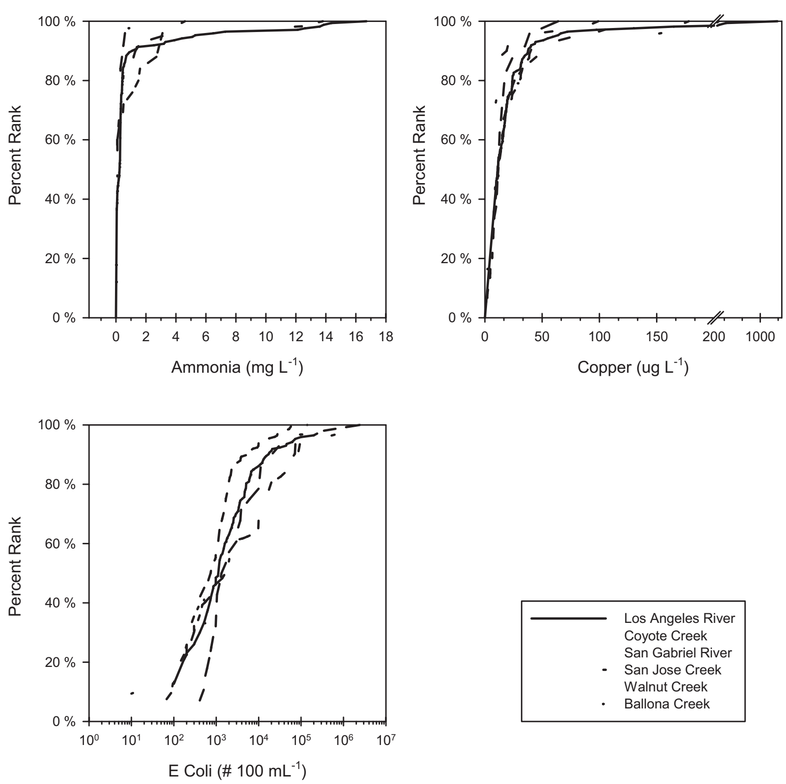

Figure 3. Cumulative distribution of storm drain water quality concentrations for total ammonia, copper and *E. coli*.

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 and volumetric loading data collected in this study (Table 8). During dry years (i.e., years in the 10^{th} 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 rainfall totals are 45% of the annual average, dry weather load-

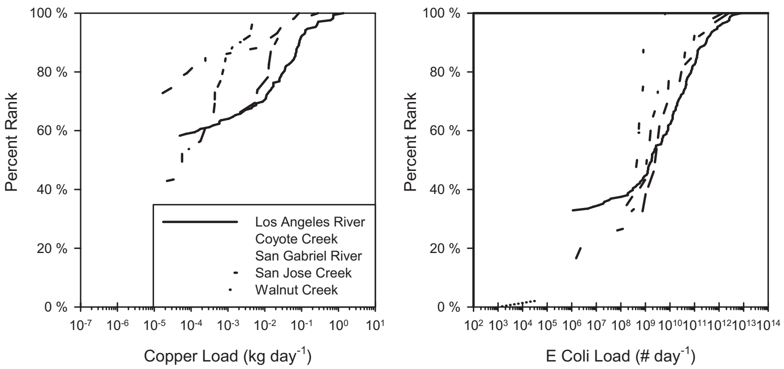

Figure 4. Cumulative distribution of storm drain mass loading for *E. coli* and copper.

ing accounts for 46, 31, and 36 percent 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 non-storm 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 due to 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 five-fold: Mean copper concentrations were 29 µg L^{-1} 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 µg L^{-1} 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 indica-

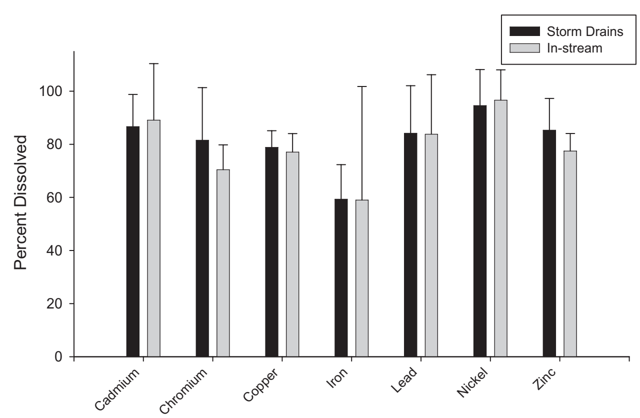

Figure 5. Comparison of percent dissolved metals. Percent of total metals as dissolved fraction in samples collected from storm drains and in-stream sites in the Ballona Creek watershed.

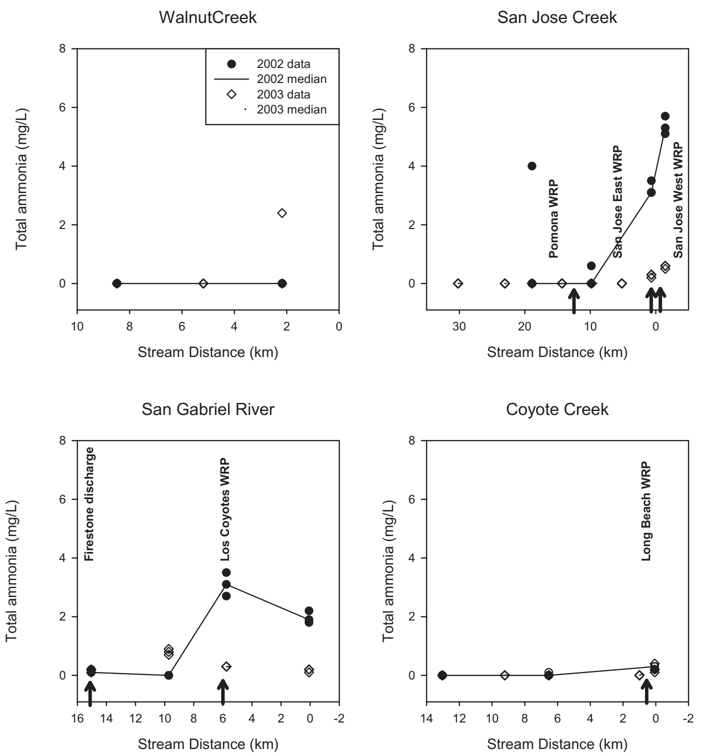

Figure 6. In-stream total ammonia concentrations by stream and year. Vertical arrows show locations of wastewater reclamation plants (WRPs).

tor). 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 non-storm flows in Ballona Creek and the Los Angeles River can vary by up to five-fold over the course of a year. For example dry weather flow in Ballona Creek may range from 0.2 $m^3 s^{-1}$ to 1.4 $m^3 s^{-1}$. In a system such as the San Gabriel River (Figure 8) 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 $m^3 s^{-1}$). 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 non-detectable 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. Non-detectable 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 non-detectable values. For the metals focused on in this study, only storm drain lead samples had a substantial fraction of non-detectable values (60%). If we had assumed that non-detectable 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 non-detectable 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 non-detectable 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 metals analyzed

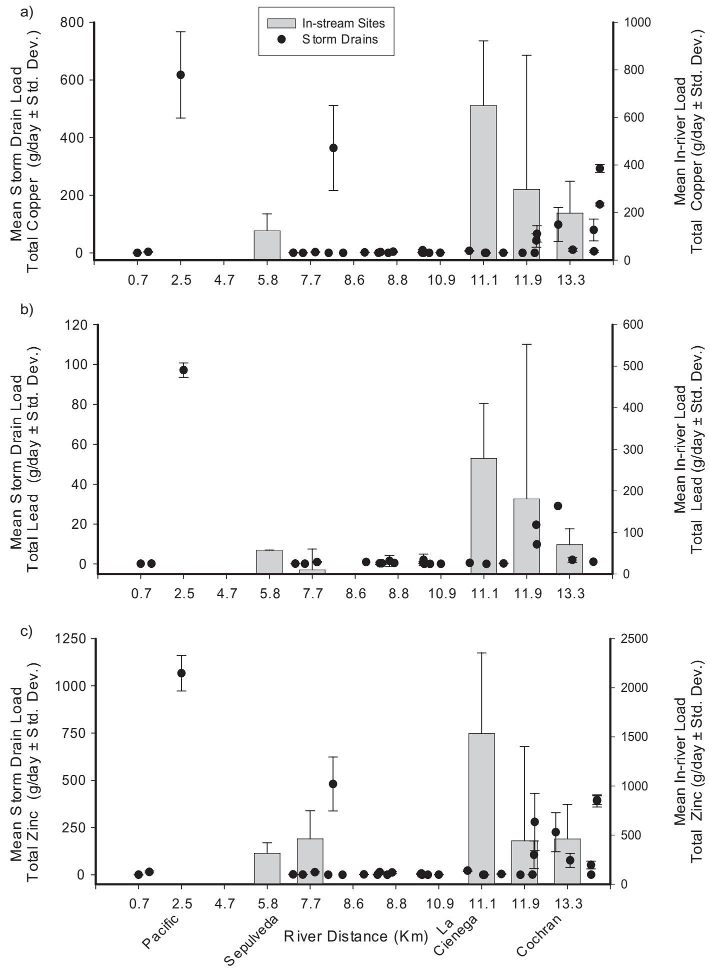

Figure 7. Change in mean in-stream metals loads. Graph shows the change in mean in-stream load (± standard deviation) between successive sampling locations in Ballona Creek for total copper (a), total lead (b), total zinc (c; right y-axis). Left y-axis shows mean storm drain load (± standard deviation)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

(Figure 9). As previously stated, we chose a conservative strategy by assigning a value of zero to all non-detectable 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

Table 8. Fraction of total annual load of various constituents accounted for by dry-weather loading, for varying annual precipitation conditions in Ballona Creek.

Percent of average annual rain	45	64	100	132	176
Rain percentile rank	**10**	**25**	**50**	**75**	**90**
	Dry Load as a Percent of Total Annual Load				
Volume	57	48	37	31	25
Fecal Coliform	3	2	2	1	1
Total Coliform	25	19	13	10	8
Total Suspended Solid	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

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 (Southern California Coastal Water Research Project, 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^{-1}. This level is several orders of magnitude lower than the 10^4 -10^6 MPN 100 ml^{-1} 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^{-1}.

This study has demonstrated that in semi-arid, urban watersheds, dry weather loadings can com-

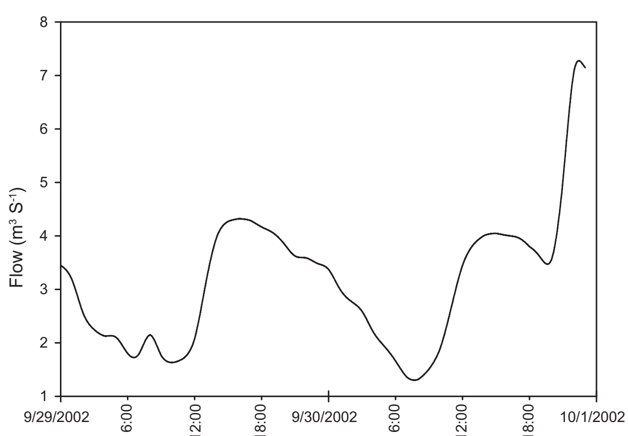

Figure 8. San Gabriel River flow variability during the 2002 sampling event.

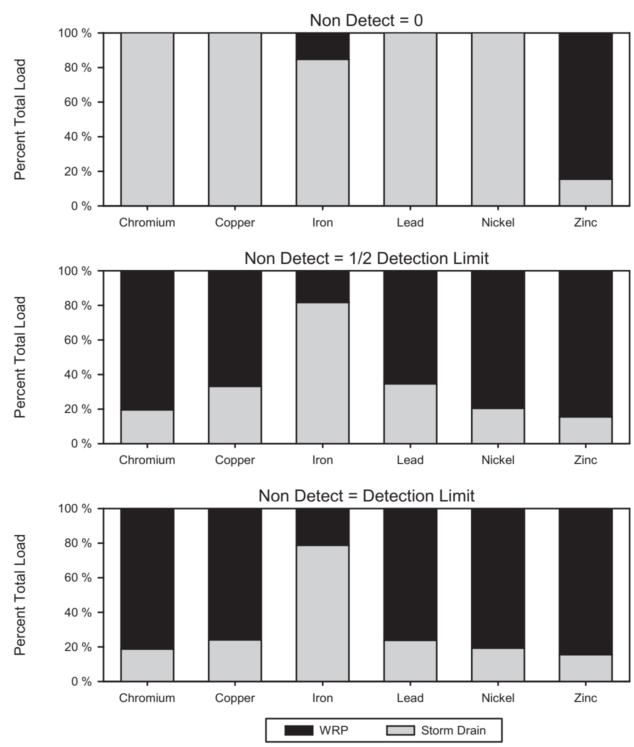

Figure 9. Effects of non-detected constituents on mass loadings of metals in the greater San Gabriel River Watershed. Graphs show how the estimated relative distribution of load varies based on the values assigned to "non-detect" sample results. WRP = water reclamation plant.

prise 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 used 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.

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