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Accuracy and precision of the volume–concentration method for urban stormwater modeling

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ABSTRACT

Land use based models have been widely used to estimate stormwater pollutant loads for establishing standards and selecting best management practices. Land use information is required to assign imperviousness or runoff coefficients and event mean concentrations for selected pollutants to the areas of the watersheds. This approach is useful to estimate the total mass discharges, but is dependent on various assumptions for parameters, which have rarely been validated with full-scale field data. This paper compares the assumptions, methodologies and predictions of six independent modeling efforts using literature and local data, and shows the variability of the different approaches and methodologies for the Upper Ballona Creek Watershed in Los Angeles CA. Differences in land use definitions among the six studies produced up to 14 % differences in average load predictions for TSS. Differences in runoff coefficient and EMC assumptions produced −70 to 124% differences in average TSS load. The combined effects of all assumptions produced differences in the estimated TSS mass loads by −68 to 118%. The most inaccurate estimates were based on the Nationwide Urban Runoff Program data, which were far too high. These results underscore the need for additional data collection and model validation, and provide error bounds for the use of this modeling approach for regulation and best management practice selection.

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1. Introduction

Stormwater modeling is critical to minimize the cost associated with stormwater management. The quantity and quality of urban stormwater runoff are often affected by site-specific conditions of geophysical and climatic factors such as imperviousness, slope, soil type, rainfall intensity and duration, and antecedent dry days. However, stormwater models generally require land use information because hydrological and water quality parameters of a watershed are highly related to its land uses and associated human activity (Marsalek, 1978; Novotny and Olem, 1994; Brezonik and Stadelmann, 2002). For instance, commercial and industrial land uses generate more heavy metals and organic pollutants than other land uses (Bannerman et al., 1993; Mikkelsen et al., 1994; Asaf et al., 2004). Construction sites contribute large

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amounts of sediment (Sonzogni et al., 1980; Brezonik and Stadelmann, 2002). Transportation land use is identified as a source of many pollutants (Barrett et al., 1998; Brezonik and Stadelmann, 2002; Han et al., 2006) and highways, streets and parking lots are the major source of hydrocarbons, sulfur and heavy metals because of vehicular emission (Bannerman et al., 1993; Hermann et al., 1994; Yuan et al., 2001). Roofing and paint materials generate copper and zinc pollution from corrosion (Förster, 1996; Gromaire et al., 2001; Davis et al., 2001; He et al., 2001; Asaf et al., 2004).

Land use information, obtained either from public records or aerial and satellite images for watershed analysis (Hill et al., 2003; Park and Stenstrom, 2007), has been used to estimate stormwater pollution and its impact on surface water quality (Bannerman et al., 1993; Charbeneau and Barrett, 1998). USEPA models such as the Stormwater Management Model (SWMM) and Hydrological Simulation Program–Fortran (HSPF) packaged in BASINS are examples that employ land use information in watershed modeling. Both models require a significant investment of time to adapt and calibrate for complex watersheds.

Other examples are empirical models or volume–concentration approaches that require land use information to assign the imperviousness and pollution concentrations to land parcels in order to estimate total pollutant mass. The logic behind the link of land use and pollutant load is the human activity that occurs on each land use type. This approach has been widely used for estimating stormwater pollution in Los Angeles region (Wong et al., 1997; LADPW, 2000; Ackerman et al., 2005; Psomas, 2005; Susilo et al., 2006) since Stenstrom and Strecker (1993) inventoried land use and estimated pollutant loads. It has also been widely used in other areas (Charbeneau and Barrett, 1998; Wu et al., 1998; Mitchell, 2005; Ellis and Mitchell, 2006).

Land use information for these models can be obtained from public agencies, but is often problematic because the agencies may use different land use/land cover classifications that are not optimal for environmental purposes. Existing land use/land cover classification systems such as USGS land cover/land use classification system (Anderson et al., 1976) have different levels with more than 100 categories. The challenge for modelers is to reclassify these land uses into a manageable number of environmentally meaningful categories. In the Los Angeles region, different modelers have employed different ways of reclassifying land uses. Different reclassifying systems can result in differences in model predictions that can have large consequences on estimated pollutant loads and receiving water impacts.

The goal of this paper is to compare the predictions of the different groups using the volume–concentration approach for a common watershed in order to assess the accuracy and variability of the methodology. The Santa Monica Bay watershed provides a good example because it has been independently modeled in approximately the same timeframe by several independent modelers. Each modeler has used sound principles and similar data sources but the results have not been compared. The comparison performed in this study serves to evaluate the approaches and can function as a qualitative indictor of the state of art for volume–concentration stormwater modeling. A well-developed and understood scientific approach should produce similar results even when performed by different groups working independently. The assumptions for the key parameters – land use classification, runoff coefficients, and pollutant concentrations – are compared and their impact on annual pollutant loads are quantified. The results will be useful for planning and regulatory agencies to understand how they can use and rely upon the predictions of volume–concentration models.

2. Materials and methods

2.1. *Volume–concentration models*

For modeling stormwater pollution, runoff volume and pollution concentrations are highly related to land use and impervious surfaces, and stream degradation begins at 10% watershed imperviousness and becomes severe at 30% imperviousness (Arnold and Gibbons, 1996; Schuler and Claytor, 1997) Often imperviousness and high pollutant loads occur in the same land uses, such as highway land use.

Stormwater runoff volume is correlated with storm characteristics, e.g. rainfall amount, duration, intensity and antecedent dry days (Brezonik and Stadelmann, 2002) but is usually estimated from imperviousness and runoff coefficients. The runoff coefficient (RC) is defined as the ratio of runoff to rainfall over a given time period and depends on the percent impervious surfaces, slope, and soil condition (Chow et al., 1988). The following equation shows an example of estimating RCs related to the impervious surfaces of a given area:

$$\mathrm{RC} = \kappa \times \mathrm{Imp} + 0.1 \tag{1}$$

where RC is runoff coefficient, Imp is impervious fraction and κ is the coefficient, e.g. 0.7 used by Driscoll et al. (1990b). Morgan et al. (1993) suggested an empirical relationship for the north Texas area as follows:

$$\mathrm{RC_p} = 0.3 \times \mathrm{Imp} + 0.2 \tag{2}$$

$$\mathrm{RC_{imp}} = \mathrm{Imp} - 0.15 \tag{3}$$

where $\mathrm{RC_p}$ is runoff coefficient for the pervious areas (Imp < 50%) and $\mathrm{RC_{imp}}$ is runoff coefficient for the impervious areas (Imp > 50%). Storm runoff volume can then be calculated as follows:

$$\mathrm{RV} = \mathrm{RC} \times A \times \mathrm{CF} \times \mathrm{RF} \tag{4}$$

where RV is stormwater runoff (m^3/yr), *A* is drainage area (m^2), CF is conversion factor, and RF is rainfall depth (mm).

Pollutant concentration of a storm event is represented by event mean concentrations (EMCs) because the concentration of the pollutant varies during the event. Generally, EMCs are characterized mainly by the land uses in the watershed and are dependent on sites and storm events (Smullen et al., 1999). An EMC is the average pollutant concentration during the storm event and defined as the total pollutant mass divided by total runoff volume as follows (Huber and Dickinson, 1988):

$$\mathrm{EMC} = \frac{M}{V} = \frac{\int_0^T c(t)q(t)\mathrm{d}t}{\int_0^T q(t)\mathrm{d}t} = \frac{\sum_0^T c(t)q(t)}{\sum_0^T q(t)} \tag{5}$$

where M is total mass of pollutant during the storm event (kg), V is total stormwater runoff volume (m^3), $c(t)$ is pollutant concentration varied over time (mg/L), $q(t)$ is flow over time (L/min), and T is total duration of runoff (min).

Given information of runoff volume and EMCs for each pollutant, pollutant loads can be estimated using the following equation (Wong et al., 1997):

$$PL_i = RV \times EMC_i \tag{6}$$

where PL_i and EMC_i are pollutant loads and the EMCs for pollutant i, respectively. Table 1 shows EMCs measured or calculated for a variety of pollutants and land uses, and many of the values were used by the groups modeling the Ballona Creek Watershed. Careful review of the estimates for the same land use and pollutants shows variability which suggest the need for site-specific calibration and comparison, as performed in this study.

2.2. *Watershed characteristics*

The Upper Ballona Creek Watershed (232 km^2) watershed includes the majority of the Ballona Creek Watershed, which is located in the western portion of the Los Angeles Basin and is the largest in the City of Los Angeles. The watershed is highly urbanized (~65% impervious), and the runoff from the watershed is the primary contributor of impairments to Santa Monica Bay. Typical stormwater pollutants include trash, TSS, bacteria, heavy metals, petroleum hydrocarbons, and nutrients. A toxicity study (Bay et al., 2003) showed that stormwater runoff, especially the seasonal first flush (Lee et al., 2004), was toxic. Data from the Los Angeles County Department of Public Works (LADPW) mass emission monitoring station and a USGS flow gauge located at the exit of Upper Ballona Creek Watershed (Fig. 1) were used for model validation.

2.3. *Pollutant mass estimates*

We compared six different modeling approaches for estimating stormwater pollution from the watersheds (Stenstrom and Strecker, 1993; LADPW, 2000; Psomas, 2005; Susilo et al., 2006; Stein et al., 2007; and Ha and Stenstrom, 2008). All of these models are based on land use information using Eqs. (4) and (6) with different assumptions. The runoff volumes (Eq. (4)) were compared using each group's land use definition as well as a common land use definition. It should be noted that we do not consider any of the assumptions as right or wrong or better or worse, only different.

First, different land use reclassification systems by different modeling groups were compared to see the effect of the land use definitions on the runoff estimation. The Southern California Association of Government (SCAG) provides public land use information for this area, and uses USGS land cover and land use classification system with some minor differences. According to SCAG's land use information (SCAG, 2005), there are more than 100 different land uses in Ballona Creek Watershed. Each modeling group reclassified these land uses into a manageable number of environmentally significant definitions (Table 2).

Second, we compared the RCs assigned for each land use, which were used by each modeling group (Table 3). Stenstrom and Strecker (1993) used the imperviousness based on LADPW, NPDES Permit No. CA0061654, Attachment 1, Santa Monica Bay Drainage Basin, drainage area characterization. Stein et al. used the pervious surface area for each land use, as established by LADPW (DePoto et al., 1991). For these cases, Eq. (1) was used to calculate RCs for each land use with κ of 0.7 for Stenstrom and Strecker's and Stein et al.'s models and with κ of 0.8 for LADPW and Psomas models. Susilo et al. used RCs for each land use in the Southern California Bight (Ackerman and Schiff, 2003). Ha and Stenstrom (2008) used a GIS to calculate RCs from Browne's (1990) relation as a function of hydrologic soil group, slope, and land use type to estimate the RCs for each land use in the Ballona Creek Watershed. Using the reclassified land uses and the assigned RCs to each land use, runoff volume from the watershed was estimated and compared assuming annual rainfall of 381 mm, which is approximately the 70-year average annual rainfall for the watershed.

Third, we compared the EMCs assigned for each land use, which were developed by each modeling group for selected water quality parameters, i.e. chemical oxygen demand (COD), total suspended solid (TSS), total Keldahl nitrogen (TKN), total phosphorus (TP), total copper (Cu), total lead (Pb), total zinc (Zn), and oil and grease (Table 3). Stenstrom and Strecker (1993) developed EMC data sets for seven land uses based upon the analysis of all previously collected data for the Santa Monica Bay Watershed. Much of the data were not useful due to differences in protocol, but there were sufficient data to estimate EMCs for TSS, COD, biochemical oxygen demand (BOD), Cu, and Zn. These were compared to data reported in the Nationwide Urban Runoff Program (NURP) and found to be higher than national means, corresponding to the 90th percentile in most cases. The remaining constituents were assumed to also correspond to the 90th percentile of NURP data except oil and grease EMCs, which were assumed to be the same as those previously collected by Fam et al. (1987) for six different land uses near San Francisco, CA. LADPW collected data using both grab samples during the initial portion of the storm and flow composite samples using an automated sampler from eight land use monitoring sites throughout the County. Their cumulative EMC data during 1994–2000 were used in this study. Psomas' data were previously calculated EMCs based on LADPW's 1994–2000 mean EMCs for each land use. Stein et al.'s EMC data were collected from seven land use monitoring sites in Santa Monica Bay watersheds. The EMC values shown here are mean EMCs during 2000–2001 except transportation and open land use. EMCs for transportation land use were measured in a rail park in 2001 and EMCs for open land use were measured in 2003. Susilo et al.'s EMC data were log mean EMCs of LADPW's 1994–2000 flow-weighted composite-sampled runoff monitoring data except agriculture, which was developed from Ventura County 1994–2004 land use EMC data. Ha and Stenstrom developed calibrations for TSS, Zn and TKN EMCs with recent monitoring results by LADPW using optimized regression by use of antecedent dry days and rainfall duration and intensity.

The parameter sets for each group are shown in Table 3 and were used with a GIS (ArcView, ESRI, Redlands, CA) to

Table 1 – Runoff coefficients and stormwater EMCs for urban land uses.

(a) RC	Location	SFR	MFR	C	P	I	T	O	Other
Runoff coefficients									
Ackerman and Schiff, 2003	S. CA	0.39	0.39	0.61		0.64		0.06	0.41
Line et al., 2002	NC	0.57	0.57					0.47	0.52
									0.70
Charbeneau and Barrett, 1998	TX	0.04 (LD)	0.62	0.83			0.56/0.85	0.007	
		0.17(MD)		0.75					
Imperviousness									
Pitt et al., 2004	USA	0.37		0.85	0.45	0.75	0.8	0.02	
Goonetilleke et al., 2005	Australia	0.7						0.02	
		0.5							

(b) EMC	Location	COD (mg/L)	TSS (mg/L)	TKN (mg/L)	NH_3 (mg/L)	$NO_{2\&3}$ (mg/L)	TP (mg/L)	Cu (μg/L)	Pb (μg/L)	Zn (μg/L)	O&G (mg/L)
SF Residential											
Charbeneau and Barrett, 1998	TX										
Low density			102								
Medium density			110								
City of Austin, 1995	TX	46	171			1.14	0.29	10	16	46	
Choe et al., 2002	Korea	226	414.1	6.81				2.85	99		
MF Residential											
Charbeneau and Barrett, 1998	TX		55.7								
Butcher, 2003	NC			1.24		0.69	0.19				
Choe et al., 2002	Korea	211.2	145.8	4.46				1.21	77		
Goonetilleke et al., 2005	Australia		156.4				0.4				
			69.5				0.7				
Residential											
Ackerman and Schiff, 2003	S. CA		102		0.53	0.12	0.60	25.2	12.9	141	
Butcher, 2003	GA										
1–2 acres				0.2		2.85	0.67				
1/2–1 acres				1.37		0.69	0.47				
1/4–1/2 acres				2.36		0.96	0.47				
Davis et al., 2001	MD							7	2	110	
Line et al., 2002	NC		42	1.48	0.34	0.49	0.40				
US EPA, 1983	USA	73	101	1.9		0.74	0.38	33	144	135	
Pitt et al., 2004	USA	54.5	49	1.5	0.31	0.60	0.31	12	12	73	4.0
Ellis and Mitchell, 2006	UK	80	85.1		0.56				141		
Goonetilleke et al., 2005	Australia		146.7				0.2				
Commercial											
Ackerman and Schiff, 2003	S. CA		118		0.7	0.11	0.55	32.64	12.22	233	
Butcher, 2003	GA			3.20		1.18	0.66				
Charbeneau and Barrett, 1998	TX										
Medium impervious			39.9								
High impervious			64.4								
Davis et al., 2001								29	12	760	
US EPA, 1983	USA	57	69	1.18		0.57	0.20	29	104	226	
Pitt et al., 2004	USA	58	43	1.5	0.50	0.6	0.22	17	18	150	4.6
Ellis and Mitchell, 2006	UK	146.2	50.4						133		
Choe et al., 2002	Korea	501.4	276.1	14.08				1.88	79		
Public											
Davis et al., 2001								2100	64	460	
Pitt et al., 2004	USA	50	17	1.35	0.31	0.6	0.18		5.75	305	
Industrial											
Ackerman and Schiff, 2003	S. CA		174		0.38	0.07	0.41	46.2	17.4	326	
Butcher, 2003	NC										
Light/office				1.6		0.65	0.17				
Heavy				1.28		0.63	0.24				
Line et al., 2002	NC		170	0.99	0.14	0.31	0.23				
City of Austin, 1995	TX	93	221			1.82	0.45	22	34	149	
Industrial/commercial											
Pitt et al., 2004	USA	58.6	81	1.4	0.42	0.69	0.25	20.8	24.9	199	4.8
Ellis and Mitchell, 2006	UK	146.2	50.4						133		
Choe et al., 2002	Korea										
Metal		118.4	88.3	4.4				2.6	44		
Food		71.7	90.7	3.6				1.3	45		
Textile		50	139.8	7.2				1.9	20		

Table 1 (*continued*)

(b) EMC	Location	COD (mg/L)	TSS (mg/L)	TKN (mg/L)	NH_3 (mg/L)	$NO_{2\&3}$ (mg/L)	TP (mg/L)	Cu (μg/L)	Pb (μg/L)	Zn (μg/L)	O&G (mg/L)
Transportation											
Barrett et al., 1998	TX										
		130	129			1.07	0.33	37	53	222	4.2
		39	91			0.71	0.11	7	15	44	1.4
		37	19			0.37	0.10	12	3	24	0.4
Driscoll et al., 1990a											
>30,000 vehicles/day		114	142			0.76		54	400	329	
<30,000 vehicles/day		49	41			0.46		22	80	80	
Charbeneau and Barrett, 1998	TX										
Medium impervious			143								
High impervious			129								
Sansalone and Buchberger, 1997	OH							135	64	4274	
Wu et al., 1998	NC										
Site I		48	215	1.00	0.66	0.38	0.20	15	15		3.3
Site II		32	88	0.95	0.62	0.19	0.37	12	13		2.1
Site III		24	14	1.02	0.42	0.08	0.26	2.5	6		1.1
Khan et al., 2006	CA	85.2									6.57
Pitt et al., 2004	USA	100	99	2.0	1.07	0.28	0.25	34.7	25	200	8.0
Ellis and Mitchell, 2006	UK										
Motorways			194.5						33		
Main roads		136.5	156.9						201		
Pagotto et al., 2000	France										
Conventional		80	46		1		6.7		30	40	228
Porous		80	8.7		0.27		2.1		20	8.7	77
Gnecco et al., 2005	Italy	129	140					19	13	81	
Shinya et al., 2000	Japan		54.26					0.16	68.25	30.5	718.38
Open											
Ackerman and Schiff, 2003	S. CA		371		0.09	0.02	0.6	22.9	4.89	45	
Charbeneau and Barrett, 1998	TX		59.6								
Line et al., 2002	NC		150	5.12	0.35	0.79	0.82				
US EPA, 1983	USA	40	70	0.97		0.54	0.12		30	195	
Pitt et al., 2004	USA	42.1	48.5	0.74	0.18	0.59	0.31	10	10.0	40	1.3
Ellis and Mitchell, 2006	UK	36	126.3	0.1					61		
Urban											
Smullen et al.,	USA										
(Pooled) Mean		52.8		1.73		0.66	0.32	13.5	67.5	162	
Median		44.7		1.47		0.53	0.26	11.1	50.7	129	
(NURP) Mean		66.1		1.67		0.84	0.34	66.6	175	176	
Median		55.0		1.41		0.66	0.27	54.8	131	140	

Note that SFR is single-family residential, MFR is multiple-family residential, C is commercial, P is public, I is industrial, T is transportation, and O is open land use and Pitt et al.'s values are median imperviousness.
Ackerman and Schiff, 2003 values are arithmetic mean, COA's, Sansalone and Buchberger, 1997 and Kim et al.'s values are mean EMCs, and Line et al.'s, Barret et al.'s, Wu et al.'s, Khan et al.'s, US EPA and Pitt et al.'s values are median EMCs. Davis et al.'s values are roof runoff EMCs.

estimate the pollutant load for each parameter for each land use parcel using Eqs. (4) and (6), and then summed for the entire watershed. The summations of each group were compared to each other and to the measured values at the mouth of the watershed (Fig. 1). Details of the GIS techniques have been published previously (Park and Stenstrom, 2007; Ha and Stenstrom, 2008)

3. Results and discussion

3.1. *Differences in land use categorization*

The composition of the reclassified land uses defined by each group's categorization method (Fig. 2) shows the differences in residential, transportation (or other urban) and open land uses due to different categorization methods. High-density residential land use in Stein et al.'s model contains high-density single-family residential and is the predominant land use (57%) of the watershed. In other groups' models, single-family residential is the predominant land use (39%). These classifications differ with classifications from other researchers in other areas, who divided land use into single-family and multiple-family residential (Charbeneau and Barrett, 1998; Choe et al., 2002; Goonetilleke et al., 2005) or lot size (Butcher, 2003). Others (USEPA, 1983; Davis et al., 2001; Line et al., 2002; Ackerman and Schiff, 2003; Pitt et al., 2004, Ellis and Mitchell, 2006) simply lumped residential land uses into a single category, implying no differences in parameter values between low- and high-density residential land use. This

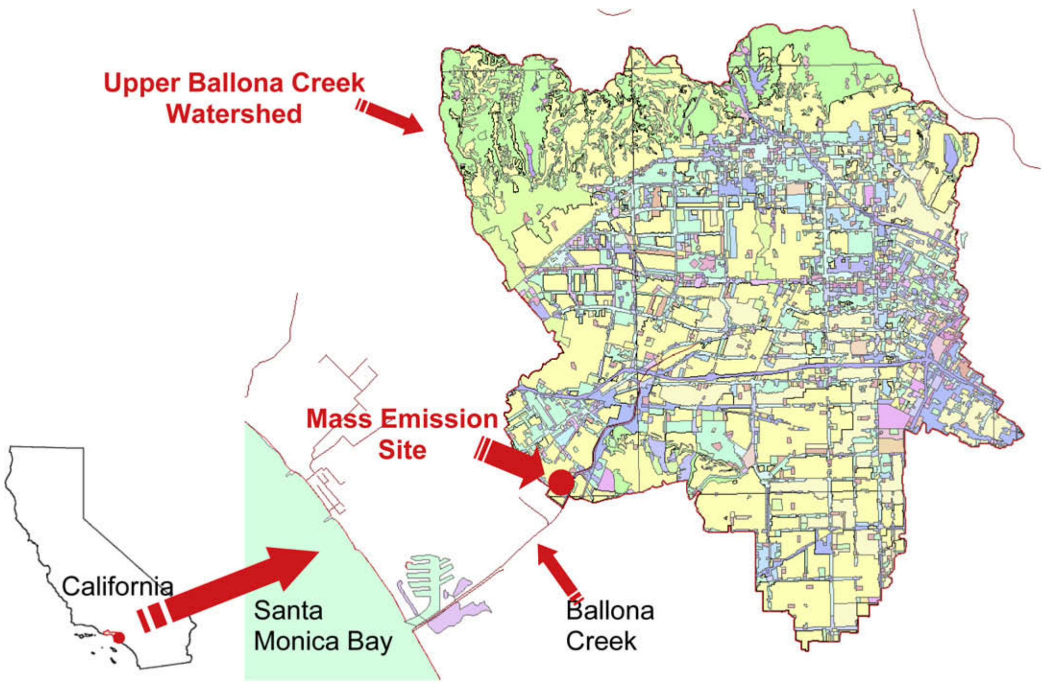

Fig. 1 – Study area: Upper Ballona Creek Watershed, California, USA.

indicates that there is no commonly used categorization of residential land uses.

Large differences were found in the other land use category. Susilo et al.'s other urban land use category contains not only transportation but also land parcels that were labeled as vacant or open by other groups. As a result, their sum of other urban in their model is four times greater than transportation land use in other groups' models. Transportation land uses were often sub-categorized by other researchers depending on traffic amount (Driscoll et al., 1990a), road types (motorways or main roads, Ellis and Mitchell, 2006), or pavement material (Pagotto et al., 2000). In addition, some researchers used sub-categories of commercial and industrial land uses depending on their type (Choe et al., 2002; Butcher, 2003) and imperviousness (Charbeneau and Barrett, 1998).

3.2. *Differences in runoff coefficients*

The next step was to compare the assumptions for RCs and their impact on runoff volume and load estimates. RCs associated with each land use are only slightly different among different models despite the differences in methodologies (Table 3). Generally, the values of RCs used in LADPW and Psomas models are greater. The greatest range among models for RCs was for multiple-family residential and transportation land uses, which differed by 0.31 and the smallest range in RCs was for single-family residential and public land uses, which were 0.15 and 0.14, respectively.

The difference in RCs creates different runoff volumes (Fig. 3) and is directly proportional to differences in load estimates. The runoff volume estimation using each group's RCs

Table 2 – Land use definition used for Upper Ballona Creek Watershed models.

Stenstrom et al.	LADPW/Psomas/Ha et al.	Stein et al.	Susilo et al.
Single-family residential	High-density single Family Residential	Low density residential	Single-family residential
Multiple-family Residential	Multiple-family Residential	Medium-high density Residential	Multiple-family residential
	Mixed Residential		
Commercial	Retail/commercial	Commercial	Commercial
Public	Educational	Public facilities & institution	
Industrial	Light industrial	Industrial	Industrial
Transportation	Transportation	Transportation & utilities	Other urban
Open	Vacant	Open space & recreation	Open
		Vacant	Agriculture
		Agriculture	

Table 3 – RCs and EMCs related to each land use for Upper Ballona Creek Watershed.

Land use	Stenstrom et al.[a]	LADPW[b]	Psomas[c]	Stein et al.[d]	Susilo et al.[e]	Ha et al.[f]
RC						
SF Residential	0.39	0.44	0.35	0.38	0.39	0.29
MF Residential	0.58	0.64	0.69	0.52	0.39	0.33
Commercial	0.74	0.84	0.89	0.7	0.61	0.72
Public	0.66	0.74	0.69		0.61	0.75
Industrial	0.74	0.83	0.79	0.63	0.64	0.69
Transportation	0.66	0.74	0.81	0.45	0.64	0.75
Open	0.1	0.1	0.25	0.12	0.06	0.23
TSS						
SF Residential	290	105	95	22	65	140
MF Residential	210	46	46	25	33	34
Commercial	180	67	66	15	58	123
Public	180	103	95		58	128
Industrial	180	229	240	130	81	194
Transportation	210	75	78	13	81	77
Open	490	165	186	135	28	235
TKN						
SF Residential	4.3	2.8	2.9			1.82
MF Residential	2.4	1.86	2.0			1.80
Commercial	2.0	3.37	3.4			8.23
Public	2.0	1.62	1.6			1.94
Industrial	2.0	3.07	3.0			1.59
Transportation	2.4	1.81	1.9			0.57
Open	2.8	0.81	0.8			0.70
$NO_{2\&3}$						
SF Residential	1.85	1.13	0.96		0.3	
MF Residential	1	1.82	1.2		0.57	
Commercial	1.2	0.71	0.64		0.46	
Public	1.2	0.71	0.6		0.46	
Industrial	1.2	0.95	0.96		0.49	
Transportation	1	0.84	0.79		0.49	
Open	1.45	1.16	1.1		1	
TP						
SF Residential	0.85	0.39	0.4	0.3		
MF Residential	0.62	0.19	0.2	0.3		
Commercial	0.43	0.41	0.4	0.2		
Public	0.43	0.31	0.3			
Industrial	0.43	0.44	0.4	0.9		
Transportation	0.62	0.44	0.4	0.1		
Open	0.52	0.11	0.2	0		
Cu						
SF Residential	0.095	0.015	0.015	0.017	0.015	
MF Residential	0.100	0.012	0.012	0.015	0.012	
Commercial	0.072	0.035	0.039	0.012	0.019	
Public	0.072	0.021	0.024	0.019		
Industrial	0.072	0.031	0.032	0.031	0.032	
Transportation	0.1	0.052	0.056	0.008	0.032	
Open	0.055	0.009	0.015	0.008	0.004	
Pb						
SF Residential	0.350	0.010	0.010	0.004	0.005	
MF Residential	0.440	0.005	0.006	0.007	0.003	
Commercial	0.225	0.012	0.018	0.004	0.002	
Public	0.225	0.005	0.005	0.002		
Industrial	0.225	0.015	0.017	0.018	0.004	
Transportation	0.44	0.009	0.010	0.002	0.004	
Open	0.14			0.001		
Zn						
SF Residential	0.350	0.080	0.079	0.059	0.053	0.04
MF Residential	0.380	0.135	0.146	0.196	0.116	0.06
Commercial	0.694	0.239	0.241	0.133	0.128	0.20
Public	0.694	0.124	0.138		0.128	0.07
Industrial	0.694	0.566	0.639	0.583	0.290	0.40

(continued on next page)

Table 3 (*continued*)

Land use	Stenstrom et al.[a]	LADPW[b]	Psomas[c]	Stein et al.[d]	Susilo et al.[e]	Ha et al.[f]
Transportation	0.380	0.279	0.291	0.064	0.290	0.22
Open	0.440	0.039	0.046	0.023	0.002	0.03
O&G						
SF Residential	3	1.36	1.3			
MF Residential	22					
Commercial	22	3.65	3.3			
Public	22					
Industrial	22	1.87	1.7			
Transportation	22	3.19	3.1			
Open	0					

a Adapted from Stenstrom and Strecker (1993). The EMC data were collected from Santa Monica Bay Watershed and compared to the EPA's Nationwide Urban Runoff Program (NURP) database. The SMB concentrations are generally 90th percentile NURP concentrations (Stenstrom and Strecker, 1993). EMC values for commercial sites were used for public and light industrial land use. Oil and grease concentrations were estimated from previous findings by Stenstrom et al. (1984).
b LACDPW 1994–2000 cumulative monitoring data.
c Based on Los Angeles County 1994–2000 mean concentration.
d Collected from monitoring sites in Santa Monica Bay watersheds. The EMC values shown here are mean EMC (FWMC) data during 2000–2001 except transportation and open land use. EMC for Transportation land use was measured in rail park in 2001. EMC for open land use were measured in 2003.
e Based on Los Angeles County 1994–2000 flow-weighted composite-sampled land use runoff monitoring data except agriculture, which was developed from Ventura County 1994–2004 land use EMC data.
f Calibrated data based on 13–33 storms from LADPW 1998–2003 monitoring data.

with uniform land use reclassification (i.e. Stenstrom et al.'s) shows that the higher RCs generally observed by LADPW and Psomas models produced greater runoff volume and the greatest difference in runoff volume among the groups was 15 million m^3/year, or approximately 25%. The runoff volume estimation using each group's own RCs and land use reclassifications shows that the runoff volume estimation by Stein et al. and Susilo et al. increased by 6 and 9 million m^3/year, respectively, and the runoff volume estimation by Psomas model decreased by 2 million m^3/year. These results demonstrate that the difference in land use classification creates large differences in runoff estimation.

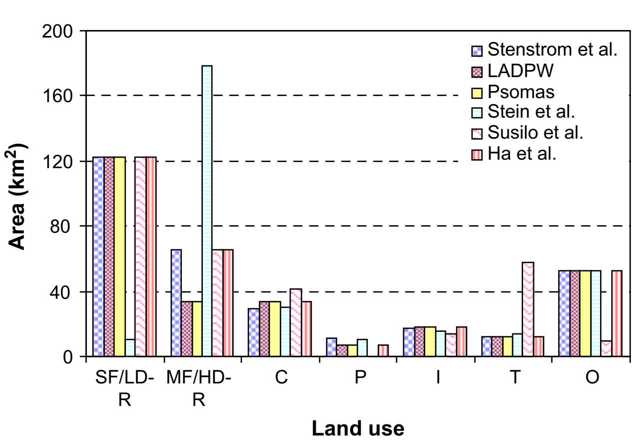

Fig. 2 – Land use composition in Upper Ballona Creek Watershed, CA. SF/LD-R: single-family/low density residential; MF/HD-R: multiple-family/high-density residential; C: commercial; P: public or education; I: industrial; T: transportation or other urban; O: open land use.

3.3. *Differences in event mean concentrations*

EMC values from each land use among the groups were quite different (Table 3) and some are outside the ranges shown in Table 1. For example, the TSS EMC for open land use was the highest in Stenstrom and Strecker, Stein et al. and Ha and Stenstrom models, whereas the EMC for industrial land use was the highest in the LADPW, Psomas and Susilo et al. models. The difference of TSS EMCs among the groups was 3–8 times different for each land use. For Zn, industrial land use was the highest concentration generating land uses by all groups. Commercial and other urban land uses were also the highest concentration generating land use by Stenstrom and Strecker, and Susilo et al., respectively. However, Stein et al.'s EMCs for transportation land use were much lower compared to those used by other groups because their monitoring station received runoff from a rail park, as opposed to roads or highways. In addition, the calibrated Zn EMCs by Ha and Stenstrom were the lowest or the second lowest for each land use. The difference of Zn EMCs among the groups varied by a factor 2–210 for each land use with the open land use making the greatest difference. Generally, EMCs of Stenstrom et al.'s model were greater than EMCs of other models, which indicate that the stormwater pollution concentrations in the watershed have declined since late 1980s.

EMCs used by the six models are shown in Fig. 4, along with the mean EMCs in NSQD (Pitt et al., 2004) and NURP databases (Smullen et al., 1999). The NURP data sets were collected almost 20 years before NSQD data sets, and the EMC values in NURP data sets are greater than those values in the NSQD data sets. This suggests that stormwater pollution concentrations have declined since the 1980s. The EMC estimates used by the models, except for those based on NURP data (Stenstrom and Strecker, 1993) are clearly lower than the

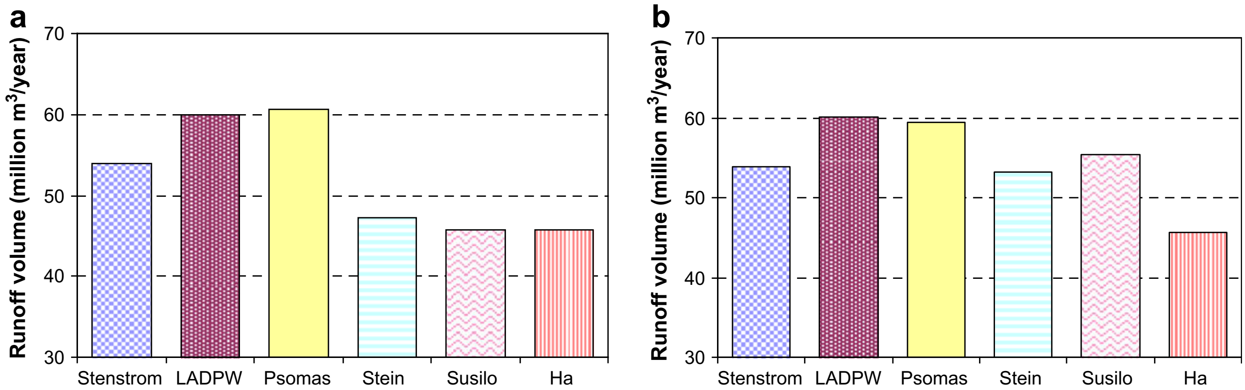

Fig. 3 – Estimated annual Runoff Volume from the Upper Ballona Creek Watershed. (a) using uniform land use reclassification and each group's runoff coefficients. (b) using each group's land use reclassification and runoff coefficients.

90th percentile NURP EMCs, and are even lower than mean NURP EMCs, with the exception of Zn. The Zn EMCs, especially for commercial, transportation and industrial land use are greater than NURP EMCs. The Pb EMCs are only 3–54% of the NSQD EMCs on average for each land use, which probably results from genuine pollutant source reductions, since California was among the first states to convert from leaded gasoline (banned in CA in 1992 vs. nationwide in 1996). Extensive data sets of EMCs from other areas of the world are generally not available, but Ellis and Mitchell (2006) have provided EMCs (Table 1) for locations in the UK. Except for TSS, UK EMCs are up to 30 times higher than EMCs for the Los Angeles region.

3.4. *Differences in pollutant loading estimation*

Using the RC and EMC data from each modeling group, we calculated annual mass loads from the watershed (Fig. 5).

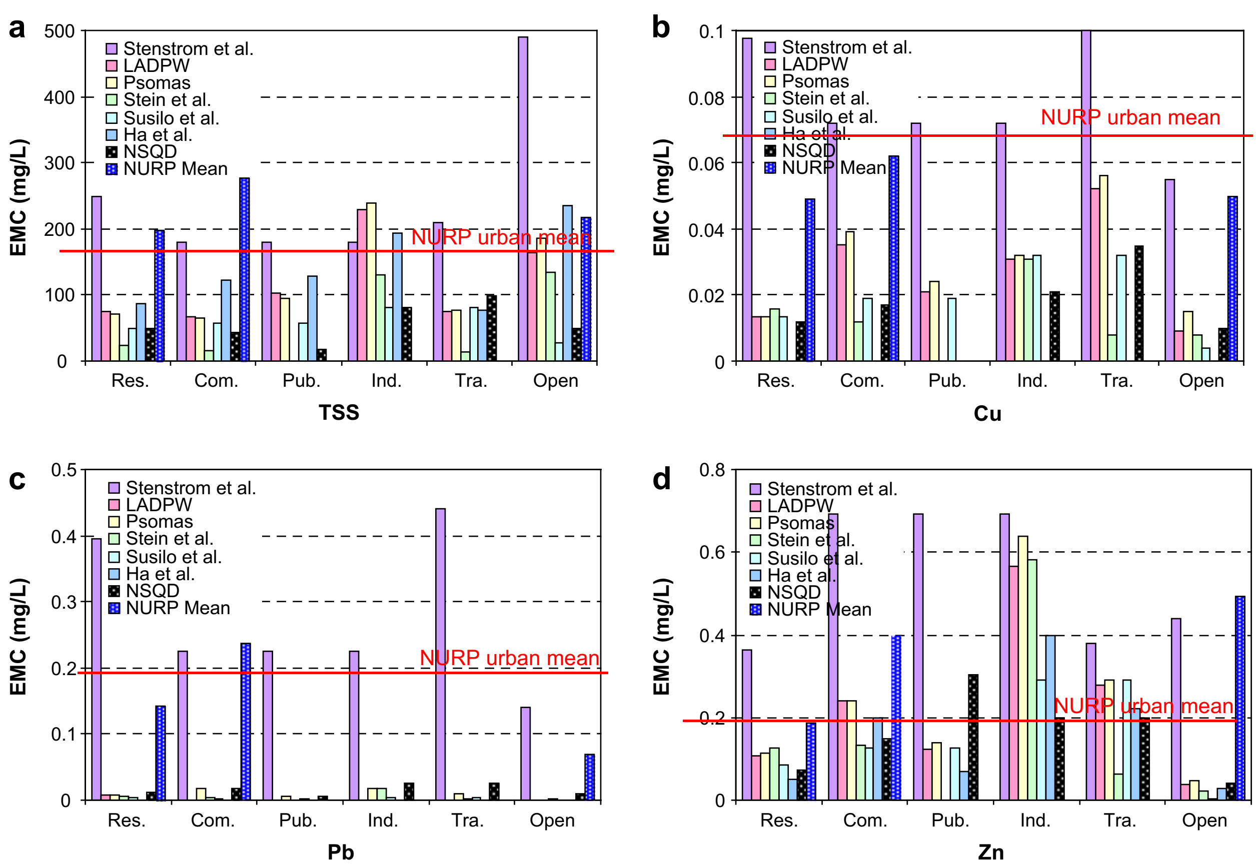

Fig. 4 – Comparison of EMCs with existing data. Res: residential; C, Com: commercial; Pub: public; Ind: industrial; Tra: transportation; O, Open: open land use.

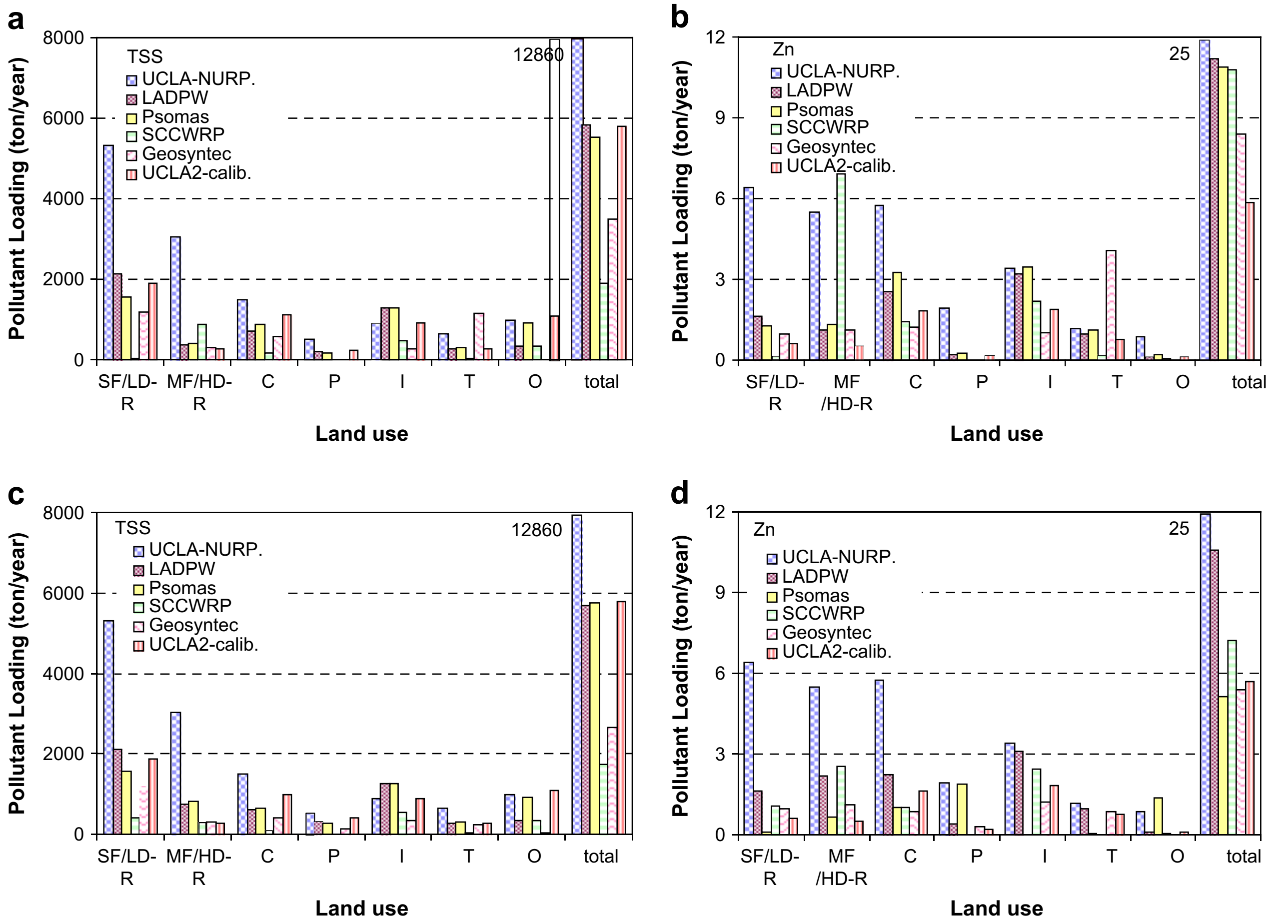

Fig. 5 – Estimated annual TSS and Zn loads. Note that different RCs and EMCs were used for all cases and (c) and (d) were based on Stenstrom et al.'s land use reclassification. (a) TSS load estimation using each modeler's land use reclassification. (b) Zn load estimation using each modeler's land use reclassification. (c) TSS load estimation using uniform land use reclassification. (d) Zn load estimation, using uniform land use reclassification. SF/LD-R: single-family/low density residential; MF/HD-R: multiple-family/high-density residential; C: commercial; P: public or education; I: industrial; T: transportation or other urban; O: open land use.

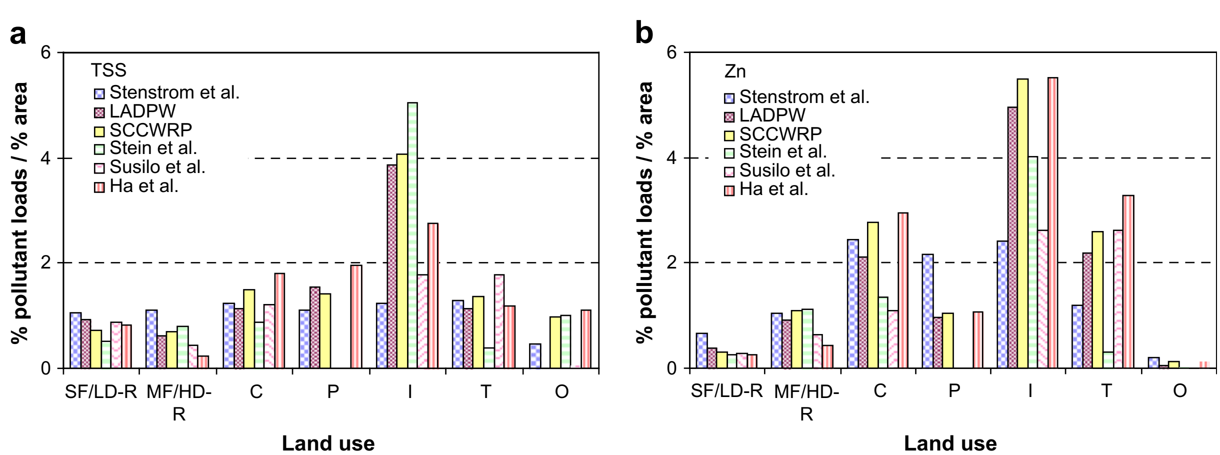

Fig. 6 – Leverage for selected pollutants from Upper Ballona Creek Watershed. (a) TSS, (b) Zn. SF/LD-R: single-family/low density residential; MF/HD-R: multiple-family/high-density residential; C: commercial; P: public or education; I: industrial; T: transportation or other urban; O: open land use.

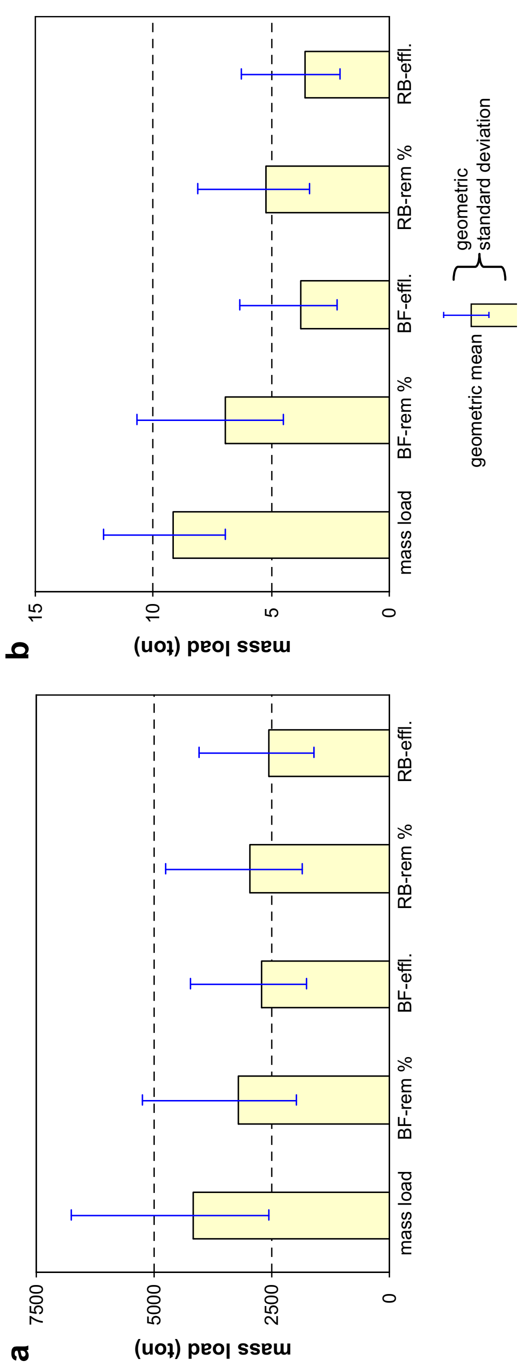

Fig. 7 – Variability of BMP performance. (a) TSS, (b) Zn. BF: biofilter; RB: retention basins; rem %: BMP performance with removal efficiencies; effl.: BMP performance with median average effluent concentrations.

Annual mass loads of selected water quality parameters were estimated using each group's RCs and EMCs with their own land use reclassification, and with uniform land use reclassification (i.e. Stenstrom and Strecker). The results show that the standard deviations for TSS and Zn load estimates among the different groups are 3751 ton and 6.70 ton, respectively. After removing the 90th percentile NURP estimates, the standard deviations among groups declines to 1751 ton and 2.29 ton for TSS and Zn, respectively. If the 90th percentile NURP estimates are ignored, loads estimated by Ha and Stenstrom are generally highest for TSS and lowest for Zn. These are the only calibrated results among the models, which used data collected at LADPW's monitoring station, and therefore should be the most representative. Estimates by other groups for these two constituents generally do not show consistent trends.

The results show that the difference among land uses, RCs and EMCs among groups are not strictly additive, and partially compensate. The coefficient of variations (CV) for TSS and Zn loads using uniform land use definition are 0.46 and 0.33, respectively, and are greater than those using each group's own land use definition. (0.39 and 0.24 for TSS and Zn, respectively). These values are lower than the CVs generally observed in large monitoring programs (Lee et al., 2007), which ranged from 1.8 to 3.3. The uncertainty ranges of the annual measured loads were greater than the estimated loads using models because of high variation of measurements and incomplete monitoring.

In addition, the leverage, the percentage of pollutant loading divided by the percentage of the area of each land use (Stenstrom et al., 1984), for each modeling group was compared (Fig. 6). The result shows general agreement among the modelers and that industrial land use generally has the highest leverage ranged from 1.2 to 5.1 for TSS and 2.4–5.5 for Zn. Transportation and commercial land uses also show high leverage values for TSS and Zn. The leverage values for land uses were different among groups, but generally the ranking order was similar. This suggests that any of the models can be used to rank land uses for BMP prioritization.

3.5. *Differences in BMP implementation*

The previously described modeling effort and the variability of the various approaches and modeling groups can be used to better understand the precision of stormwater quality estimates. Such estimates are often used to define achievable concentrations for TMDLs or to evaluate the potential impact of BMP construction. Alternate BMP applications can be compared using the models to understand the precision of expected water quality after the application of the BMPs.

To show this result, two BMPs, biofilters and retention basins, were applied to the mean modeling results (Fig. 7). The geometric means of the variability of the model results and BMP performance were added to estimate the overall the variability of the expected water quality after BMP implementation. BMP implementation was assumed for only the high leverage areas. The performance of BMPs was predicted using both removal efficiencies based on existing literature data and effluent concentration distributions. The removal efficiencies are reported to be 20–80% for TSS and 2–65% for metals by biofilters (USEPA, 1993; Young et al., 1996; Yu and Kaighn, 1992) and 50–80% for both TSS and metals by retention basins (USEPA, 1993). The median average effluent concentrations are 17.9 and 9.7 mg/L for TSS and 28.0 and 21.5 μg/L by biofilter and retention basins, respectively (Geosyntec, 2007).

The variability of stormwater quality and BMP performance is so large that in most cases the untreated and treated stormwater qualities are not statistically different. The error bars for the predicted concentrations of Zn, using effluent distributions, are on the verge of being different. Small differences in the assumptions might create or diminish the difference. The overall result of this analysis is disturbing, in that the application of expensive BMPs might not produce observable differences in water quality. Clearly there should be differences, since removal of pollutants by BMPs obviously occurs and is verified by the need to remove pollutants that accumulate in the BMPs. Two important conclusions can be drawn from this result. The first is that the application of BMPs may not produce statistically different improvements in stormwater quality, not because the BMPs are ineffective, but because of the poor water quality characterization, before the application of the BMPs. This result should not be considered a failure of the BMPs. The other useful conclusion is the need for additional monitoring and performance data to reduce the uncertainty of water quality and BMP performance estimates. Planners will need to weigh the potential benefits of an improved BMP application strategy that might result from improved monitoring results, with the environmental degradation that results with delayed BMP application.

The final comment we offer is on the use of such simple models. Models such as SWMM and HSPF can provide more realistic results as well as non-steady-state results (Ackerman et al., 2005). They are far more complicated, requiring many more parameters, more complex calibration and more effort. Is it better to use the more complex models with the additional data burden, or are the simple models sufficiently accurate for decision making? Volume–concentration models with user friendly graphical user interfaces (GUI) have already been developed to assist permittees in developing their TMDL compliance programs (Susilo et al., 2006). The comparisons of the independent efforts provided in this paper suggest that widely varying modeling results can occur, but can largely be reduced through judicious calibration (Ha and Stenstrom, 2008).

4. Conclusions

This study shows the effect of different model assumptions on the estimation of stormwater pollutant loads by evaluating and comparing the quantitative results of six different stormwater modeling groups. The groups worked independently except for the use of some common data sources. The following specific conclusions are made:

- The different modeling groups had slightly different interpretation of land use definitions, which impacted runoff volume and pollutant load estimates.
- Runoff coefficients for each land use were slightly different except for multiple-family residential and transportation land use. The difference of RCs in conjunction with the different land use categorization resulted in difference in runoff volume estimation as great as 25%.
- The EMCs values for each land use among the groups were quite different. EMCs based on long term historical data (NURP) are obviously too high to characterize current runoff water quality. Particularly, EMCs based on 90th percentile of NURP data, which were judged to be a best fit in 1993 for Los Angeles, are now far too high. Even mean NURP EMCs are now too high for Los Angeles area runoff, with the possible exception of Zn. If the estimates based on NURP are not considered, differences in load estimates among the groups varied by a factor of 2–3.
- Very large reductions in lead emissions, as great 97%, were predicted by the models using recent lead EMC estimates (<10 years old), when compared to lead emissions based upon NURP EMCs (>20 years old).
- Differences in assumptions for RCs and EMCs assumed by different modelers were larger than the differences in load estimates, showing that the high assumptions for runoff volume were associated with low assumptions for EMCs and vice versa.
- The EMCs that fit Los Angeles data well can be quite different than EMCs reported in other areas, which suggest that climate, geography and cultural differences will require site-specific calibration.

The overall results of the model comparisons suggest that volume–concentration modeling is a useful tool at the planning level for stormwater management. The distributing conclusion from this research is that the variability between modeling results and the reported variability in BMP performance is so large that statistically significant improvements in stormwater quality after resulting from BMP construction may not be detectable. The possibility of this result should be anticipated and should also provide motivation for better pre-BMP monitoring.

Increasing the accuracy and precision of water quality improvements with large scale application of BMPs such as Proposition O (Park et al., 2008) will depend upon continuing improvement in the definition of land use, RCs and EMCs. The effect of the land use reclassification on stormwater runoff quantity and quality is significant and the differences of EMCs for each land use can translate into large differences in mass load estimates.

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