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August 12, 2003

3. Sources of Stormwater Pollutants, Including Pollutant Buildup and Washoff

Introduction

This section presents pollutant accumulation and washoff processes that have been observed during extensive field projects. These processes are fundamental components of many stormwater models. This section also describes pollutant characteristics of particulates that are removed during rains, and sheetflow quality from most source areas.

This material was mostly extracted from the final draft of:

Pitt, R. *Stormwater Quality Management*, CRC Press. New York, expected publication in 2000.

The accumulation and washoff information presented here was obtained from many research projects (as listed in the references) and initially described in Pitt's dissertation:

Pitt, R. *Small Storm Urban Flow and Particulate Washoff Contributions to Outfall Discharges*, Ph.D. Dissertation, Civil and Environmental Engineering Department, University of Wisconsin, Madison, WI, November 1987.

Descriptions of street dirt measurements and washoff tests are summarized from many studies and this discussion is from:

Burton, G.A. and R. Pitt. *Manual for Evaluating Stormwater Runoff Effects, A Tool Box of Procedures and Methods to Assist Watershed Managers.* CRC/Lewis Publishers, New York. Expected publication in 2000.

The Source Concept

Urban runoff is comprised of many separate source area flow components that are combined within the drainage area and at the outfall before entering the receiving water. Considering the combined outfall conditions alone may be adequate when evaluating the long term, area-wide effects of many separate outfall discharges to a receiving water. However, if better predictions of outfall characteristics (or the effects of source area controls) are needed, then the separate source area components must be characterized. The discharge at the outfall is made up of a mixture of contributions from different source areas. The "mix" depends on the characteristics of the drainage area and the specific rain event. The overall effectiveness of source area controls in reducing stormwater discharges is, therefore, highly site and storm specific, as site and rain characteristics control how important each source is in contributing pollutants to the overall flow.

Various urban source areas all contribute different quantities of runoff and pollutants depending on their characteristics. Impervious source areas may contribute most of the runoff during small rain events. Examples of these source areas include paved parking lots, streets, driveways, roofs, and sidewalks. Pervious source areas become important contributors for larger rain events. These pervious source areas include gardens, lawns, bare ground, unpaved parking areas and driveways, and undeveloped areas. The relative importance of the individual sources is a function of their areas, their pollutant washoff potentials, and the rain characteristics.

The washoff of debris and soil during a rain is dependent on the energy of the rain and the properties of the material. Pollutants are also removed from source areas by winds, litter pickup, or other cleanup activities. The runoff and pollutants from the source areas flow directly into the drainage system, onto impervious areas that are directly connected to the drainage system, or onto pervious areas that will attenuate some of the flows and pollutants, before they discharge to the drainage system.

Sources of pollutants on paved areas include on-site particulate storage that cannot be removed by usual processes such as rain, wind, and street cleaning. Atmospheric deposition, deposition from activities on these paved surfaces (e.g., auto traffic, material storage) and the erosion of material from upland areas that directly discharge flows onto these areas, are the major sources of pollutants to the paved areas. Pervious areas contribute pollutants mainly through erosion processes where the rain energy dislodges soil from between vegetation. The runoff from these source areas enters the storm drainage system where sedimentation in catchbasins or in the sewerage may affect their ultimate discharge to the outfall. In-stream physical, biological, and chemical processes affect the pollutants after they are discharged to the ultimate receiving water.

Knowing when the different source areas become "active" (when runoff initiates from the area, carrying pollutants to the drainage system) is critical. If pervious source areas are not contributing runoff or pollutants, then the prediction of urban runoff quality is greatly simplified. The mechanisms of washoff and delivery yields of runoff and pollutants from paved areas are much better known than from pervious urban areas (Novotny and Chesters 1981). In many cases, pervious areas are not active except during rain events greater than at least five or ten mm. For smaller rain depths, almost all of the runoff and pollutants originate from impervious surfaces (Pitt 1987). However, in many urban areas, pervious areas may contribute the majority of the runoff, and some pollutants, when rain depths are greater than about 20 mm. The actual importance of the different source areas is highly dependent on the specific land use and rainfall patterns. Obviously, in areas having relatively low-density development, especially where moderate and large sized rains occur frequently (such as in the Southeast portion of the US), pervious areas typically dominate outfall discharges. In contrast, in areas having significant paved areas, especially where most rains are relatively small (such as in the arid west of the US), the impervious areas dominate outfall discharges. The effectiveness of different source controls is, therefore, quite different for different land uses and climatic patterns.

If the number of events exceeding a water quality objective are important, then the small rain events are of most concern. Stormwater runoff typically exceeds some water quality standards for practically every rain event (especially for bacteria and some heavy metals). In the US's upper midwest, the median rain depth is about six mm, while in the US's southeast, the median rain depth is about twice this depth. For these small rain depths and for most urban land uses, directly connected paved areas usually contribute most of the runoff and pollutants. However, if annual mass discharges are more important (e.g. for long-term effects), then the moderate rains are more important. Rains from about 10 to 50 mm produce most of the annual runoff volume in many areas of the US. Runoff from both impervious and pervious areas can be very important for these rains. The largest rains (greater than 100 mm) are relatively rare and do not contribute significant amounts of runoff pollutants during normal years, but are very important for drainage design. The specific source areas that are most important (and controllable) for these different conditions vary widely. This section describes sources of urban runoff flows and pollutants based on many studies reported in the literature.

Sources and Characteristics of Urban Runoff Pollutants

Years of study reveal that the vast majority of stormwater toxicants and much of the conventional pollutants are associated with automobile use and maintenance activities and that these pollutants are strongly associated with the particulates suspended in the stormwater (the non-filterable components or suspended solids). Reducing or modifying automobile use to reduce the use of these compounds, has been difficult, with the notable exception of the phasing out of leaded gasoline. Current activities, concentrated in the San Francisco, CA area, focus on encouraging brake pad manufacturers to reduce the use of copper.

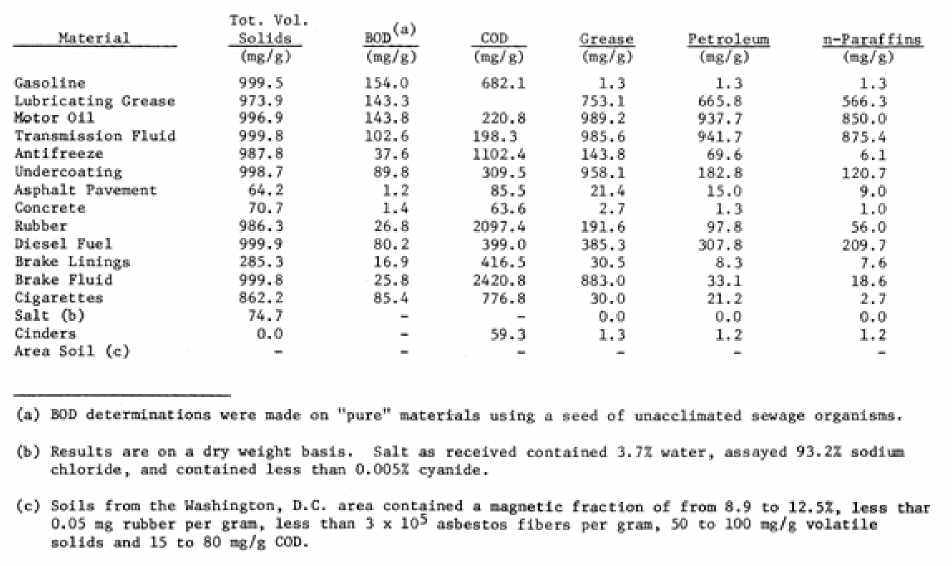
The effectiveness of most stormwater control practices is, therefore, dependent on their ability to remove these particles from the water, or possibly from intermediate accumulating locations (such as streets or other surfaces) and not through source reduction. The removal of these particles from stormwater is dependent on various characteristics of these particles, especially their size and settling rates. Some source area controls (most notably street cleaning) affect the particles before they are washed-off and transported by the runoff, while others remove the particles from the flowing water. This discussion, therefore summarizes the accumulation and washoff of these particulates and the particle size distribution of the suspended solids in stormwater runoff to better understand the effectiveness of source area control practices.

Table 3-1 shows that most of the organic compounds found in stormwater are associated with various human-related activities, especially automobile and pesticide use, or are associated with plastics (Verschueren 1983). Heavy metals found in stormwater also mostly originate from automobile use activities, including gasoline combustion, brake lining, fluids (e.g., brake fluid, transmission oil, anti-freeze, grease), undercoatings, and tire wear (Durum 1974, Koeppe 1977, Rubin 1976, Shaheen 1975, Solomon and Natusch 1977, and Wilbur and Hunter 1980). Auto repair, pavement wear, and deicing compound use also contribute heavy metals to stormwater (Field, *et al*. 1973 and Shaheen 1975). Shaheen (1975) found that eroding area soils are the major source of the particulates in stormwater. He also investigated many different materials that contribute to the street dust and dirt loading (Table 3-2). The eroding area soil particles, and the particles associated with road surface wear, become contaminated with exhaust emissions and runoff containing the polluting compounds. Shaheen found that gasoline and oils have heavy concentrations of many pollutants, while break linings and asphalt pavement wear have high concentrations of many heavy metals. Even litter materials (such as cigarette butts) can contribute metals and other pollutants. Most of these compounds become tightly bound to these particles and are then transported through the urban area and drainage system, or removed from the stormwater, with the particulates. Stormwater concentrations of zinc, fluoranthene, 1,3-dichlorobenzene, and pyrene are unique in that substantial fractions of these compounds remain in the water and are less associated with the particulates.

Table 3-1. Uses and Sources for Organic Compounds found in Stormwater (Verschueren 1983)

Compound	Example Use/Source
Phenol	gasoline, exhaust
N-Nitroso-di-n-propylamine	contaminant of herbicide Treflan
Hexachloroethane	plasticizer in cellulose esters, minor use in rubber and insecticide
Nitrobenzene	solvent, rubber, lubricants
2,4-Dimethylphenol	asphalt, fuel, plastics, pesticides
Hexachlorobutadiene	rubber and polymer solvent, transformer and hydraulic oil
4-Chloro-3-methylphenol	germicide; preservative for glues, gums, inks, textile, and leather
Pentachlorophenol	insecticide, algaecide, herbicide, and fungicide mfg., wood preservative
Fluoranthene	gasoline, motor and lubricating oil, wood preservative
Pyrene	gasoline, asphalt, wood preservative, motor oil
Di-n-octylphthalate	general use of plastics

Table 3-2. Concentrations of Materials Found on Urban Roadways (Shaheen 1975)

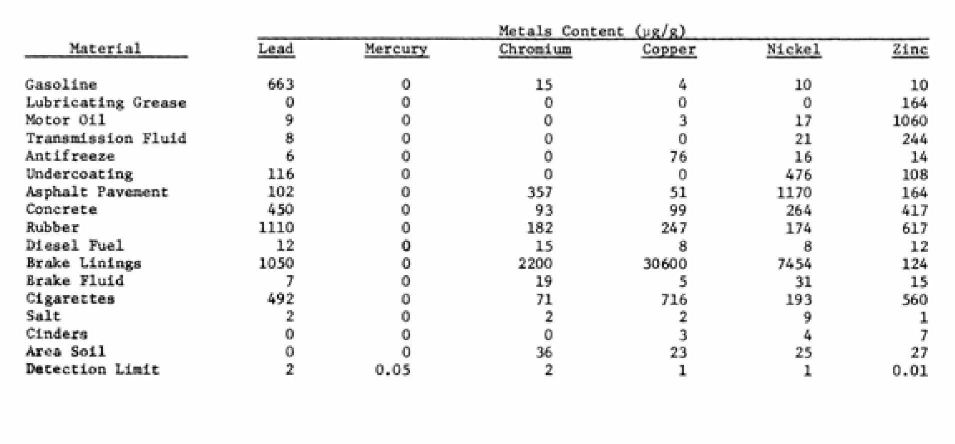


Material	Tot. Vol. Solids (mg/g)	BOD[(a)] (mg/g)	COD (mg/g)	Grease (mg/g)	Petroleum (mg/g)	n-Paraffins (mg/g)
Gasoline	999.5	154.0	682.1	1.3	1.3	1.3
Lubricating Grease	973.9	143.3		753.1	665.8	566.3
Motor Oil	996.9	143.8	220.8	989.2	937.7	850.0
Transmission Fluid	999.8	102.6	198.3	985.6	941.7	875.4
Antifreeze	987.8	37.6	1102.4	143.8	69.6	6.1
Undercoating	998.7	89.8	309.5	958.1	182.8	120.7
Asphalt Pavement	64.2	1.2	85.5	21.4	15.0	9.0
Concrete	70.7	1.4	63.6	2.7	1.3	1.0
Rubber	986.3	26.8	2097.4	191.6	97.8	56.0
Diesel Fuel	999.9	80.2	399.0	385.3	307.8	209.7
Brake Linings	285.3	16.9	416.5	30.5	8.3	7.6
Brake Fluid	999.8	25.8	2420.8	883.0	33.1	18.6
Cigarettes	862.2	85.4	776.8	30.0	21.2	2.7
Salt (b)	74.7	-	-	0.0	0.0	0.0
Cinders	0.0	-	59.3	1.3	1.2	1.2
Area Soil (c)	-	-	-	-	-	-

(a) BOD determinations were made on "pure" materials using a seed of unacclimated sewage organisms.

(b) Results are on a dry weight basis. Salt as received contained 3.7% water, assayed 93.2% sodium chloride, and contained less than 0.005% cyanide.

(c) Soils from the Washington, D.C. area contained a magnetic fraction of from 8.9 to 12.5%, less than 0.05 mg rubber per gram, less than 3 x 10^5 asbestos fibers per gram, 50 to 100 mg/g volatile solids and 15 to 80 mg/g COD.



Material	Metals Content (μg/g)					
	Lead	Mercury	Chromium	Copper	Nickel	Zinc
Gasoline	663	0	15	4	10	10
Lubricating Grease	0	0	0	0	0	164
Motor Oil	9	0	0	3	17	1060
Transmission Fluid	8	0	0	0	21	244
Antifreeze	6	0	0	76	16	14
Undercoating	116	0	0	0	476	108
Asphalt Pavement	102	0	357	51	1170	164
Concrete	450	0	93	99	264	417
Rubber	1110	0	182	247	174	617
Diesel Fuel	12	0	15	8	8	12
Brake Linings	1050	0	2200	30600	7454	124
Brake Fluid	7	0	19	5	31	15
Cigarettes	492	0	71	716	193	560
Salt	2	0	2	2	9	1
Cinders	0	0	0	3	4	7
Area Soil	0	0	36	23	25	27
Detection Limit	2	0.05	2	1	1	0.01

All areas are affected by atmospheric deposition, while other sources of pollutants are specific to the activities conducted on the areas. As examples, the ground surfaces of unpaved equipment or material storage areas can become contaminated by spills and debris, while undeveloped land remaining relatively unspoiled by activities can still contribute runoff solids, organics, and nutrients, if eroded. Atmospheric deposition, deposition from activities on paved surfaces, and the erosion of material from upland unconnected areas are the major sources of pollutants in urban areas.

Many studies have examined different sources of urban runoff pollutants. These significant pollutants have been shown to have a potential for creating various receiving water impact problems. Most of these potential problem pollutants typically have significant concentration increases in the urban feeder creeks and sediments, as compared to areas not affected by urban runoff.

The important sources of these pollutants are related to various uses and processes. Automobile related potential sources usually affect road dust and dirt quality more than other particulate components of the runoff system. The road dust and dirt quality is affected by vehicle fluid drips and spills (e.g., gasoline, oils) and vehicle exhaust, along with various vehicle wear, local soil erosion, and pavement wear products. Urban landscaping practices potentially affecting urban runoff include vegetation litter, fertilizer and pesticides. Miscellaneous sources of urban runoff pollutants include firework debris, wildlife and domestic pet wastes and possibly industrial and sanitary wastewaters. Wet and dry atmospheric contributions both affect runoff quality. Pesticide use in an urban area can contribute significant quantities of various toxic materials to urban runoff. Many manufacturing and industrial activities, including the combustion of fuels, also affect urban runoff quality.

Natural weathering and erosion products of rocks contribute the majority of the hardness and iron in urban runoff pollutants. Road dust and associated automobile use activities (gasoline exhaust products) historically contributed most of the lead in urban runoff. However, the decrease of lead in gasoline has resulted in current stormwater lead concentrations being about one tenth of the levels found in stormwater in the early 1970s (Bannerman, *et al*. 1993). In certain situations, paint chipping can also be a major source of lead in urban areas. Road dust, contaminated by tire wear products and zinc plated metal erosion material, contributes most of the zinc to urban runoff. Urban landscaping activities can be a major source of cadmium (Phillips and Russo 1978). Electroplating and ore processing activities can also contribute chromium and cadmium.

Many pollutant sources are specific to a particular area and on-going activities. For example, iron oxides are associated with welding operations and strontium, used in the production of flares and fireworks, would probably be found on the streets in greater quantities around holidays, or at the scenes of traffic accidents. The relative contribution of each of these potential urban runoff sources, is, therefore, highly variable, depending upon specific site conditions and seasons. Specific information is presented in the following subsections concerning the qualities of various rocks and soils, urban and rural dustfall, and precipitation.

Chemical Quality of Rocks and Soils

The abundance of common elements in the lithosphere (the earth's crust) is shown in Table 3-3 (Lindsay l979). Almost half of the lithosphere is oxygen and about 25% is silica. Approximately eight percent is aluminum and five percent is iron. Elements comprising between two percent and four percent of the lithosphere include calcium, sodium, potassium and magnesium. Because of the great abundance of these materials in the lithosphere, urban runoff transports only a relatively small portion of these elements to receiving waters, compared to natural processes. Iron and aluminum can both cause detrimental effects in receiving waters if in their dissolved forms. A reduction of the pH substantially increases the abundance of dissolved metals.

Table 3-3. Common Elements in the Lithosphere (Lindsay 1979)

Abundance Rank	Element	Concentration in Lithosphere (mg/kg)
1	O	465,000
2	Si	276,000
3	Al	81,000
4	Fe	51,000
5	Ca	36,000
6	Na	28,000
7	K	26,000
8	Mg	21,000
9	P	1,200
10	C	950
11	Mn	900
12	F	625
13	S	600
14	Cl	500
15	Ba	430
16	Rb	280
17	Zr	220
18	Cr	200
19	Sr	150
20	V	150
21	Ni	100

Table 3-4, also from Lindsay (1979), shows the rankings for common elements in soils. These rankings are quite similar to the values shown previously for the lithosphere. Natural soils can contribute pollutants to urban runoff through local erosion. Again, iron and aluminum are very high on this list and receiving water concentrations of these metals are not expected to be significantly affected by urban activities alone.

The values shown on these tables are expected to vary substantially, depending upon the specific mineral types. Arsenic is mainly concentrated in iron and manganese oxides, shales, clays, sedimentary rocks and phosphorites. Mercury is concentrated mostly in sulfide ores, shales and clays. Lead is fairly uniformly distributed, but can be concentrated in clayey sediments and sulfide deposits. Cadmium can also be concentrated in shales, clays and phosphorites (Durum 1974).

Street Dust and Dirt Pollutant Characteristics

Most of the street surface dust and dirt materials (by weight) are local soil erosion products, while some materials are contributed by motor vehicle emissions and wear (Shaheen 1975). Minor contributions are made by erosion of street surfaces in good condition. The specific makeup of street surface contaminants is a function of many conditions and varies widely (Pitt 1979).

Table 3-4. Common Elements in Soils (Lindsay 1979)

Abundance Rank	Element	Typical Minimum (mg/kg)	Typical Maximum (mg/kg)	Typical Average (mg/kg)
1	O	- -	- -	490,000
2	Si	230,000	350,000	320,000
3	Al	10,000	300,000	71,000
4	Fe	7,000	550,000	38,000
5	C	- -	- -	20,000
6	Ca	7,000	500,000	13,700
7	K	400	30,000	8,300
8	Na	750	7,500	6,300
9	Mg	600	6,000	5,000
10	Ti	1,000	10,000	4,000
11	N	200	4,000	1,400
12	S	30	10,000	700
13	Mn	20	3,000	600
14	P	200	5,000	600
15	Ba	100	3,000	430
16	Zr	60	2,000	300
17	F	10	4,000	200
18	Sr	50	1,000	200
19	Cl	20	900	100
20	Cr	1	1,000	100
21	V	20	500	100

Automobile tire wear is a major source of zinc in urban runoff and is mostly deposited on street surfaces and nearby adjacent areas. About half of the airborne particulates lost due to tire wear settle out on the street and the majority of the remaining particulates settle within about six meters of the roadway. Exhaust particulates, fluid losses, drips, spills and mechanical wear products can all contribute lead to street dirt. Many heavy metals are important pollutants associated with automobile activity. Most of these automobile pollutants affect parking lots and street surfaces. However, some of the automobile related materials also affect areas adjacent to the streets. This occurs through the wind transport mechanism after being resuspended from the road surface by traffic-induced turbulence.

Automobile exhaust particulates contribute many important heavy metals to street surface particulates and to urban runoff and receiving waters. The most notable of these heavy metals has been lead. However, since the late 1980s, the concentrations of lead in stormwater has decreased substantially (by about ten times) compared to early 1970 observations. This decrease, of course, is associated with significantly decreased consumption of leaded gasoline.

Solomon and Natusch (1977) studied automobile exhaust particulates in conjunction with a comprehensive study of lead in the Champaign-Urbana, IL area. They found that the exhaust particulates existed in two distinct morphological forms. The smallest particulates were almost perfectly spherical, having diameters in the range of 0.1 to 0.5 µm. These small particles consisted almost entirely of PbBrCl (lead, bromine, chlorine) at the time of emission. Because the particles are small, they are expected to remain airborne for considerable distances and can be captured in the lungs when inhaled. The researchers concluded that the small particles are formed by condensation of PbBrCl vapor onto small nucleating centers, which are probably introduced into the engine with the filtered engine air.

Solomon and Natusch (1977) found that the second major form of automobile exhaust particulates were rather large, being roughly 10 to 20 µm in diameter. These particles typically had irregular shapes and somewhat smooth surfaces. The elemental compositions of these irregular particles were found to be quite variable, being predominantly iron, calcium, lead, chlorine and bromine. They found that individual particles did contain aluminum, zinc, sulfur, phosphorus and some carbon, chromium, potassium, sodium, nickel and thallium. Many of these elements (bromine, carbon, chlorine, chromium, potassium, sodium, nickel, phosphorus, lead, sulfur, and thallium) are most likely condensed, or adsorbed, onto the surfaces of these larger particles during passage through the exhaust system. They believed that these large particles originate in the engine or exhaust system because of their very high iron content. They found that 50 to 70 percent of the emitted lead was associated with these large particles, which would be deposited within a few meters of the emission point onto the roadway, because of their aerodynamic properties.

Solomon and Natusch (1977) also examined urban particulates near roadways and homes in urban areas. They found that lead concentrations in soils were higher near roads and houses. This indicated the capability of road dust and peeling house paint to contaminate nearby soils. The lead content of the soils ranged from 130 to about 1,200 mg/kg. Koeppe (1977), during another element of the Champaign-Urbana lead study, found that lead was tightly bound to various soil components. However, the lead did not remain in one location, but it was transported both downward in the soil profile and to adjacent areas through both natural and man-assisted processes.

Atmospheric Sources of Urban Runoff Pollutants

Atmospheric processes affecting urban runoff pollutants include dry dustfall and precipitation quality. These have been monitored in many urban and rural areas. In many instances, however, the samples were combined as a bulk precipitation sample before processing. Automatic precipitation sampling equipment can distinguish between dry periods of fallout and precipitation. These devices cover and uncover appropriate collection jars exposed to the atmosphere. Much of this information has been collected as part of the Nationwide Urban Runoff Program (NURP) and the Atmospheric Deposition Program, both sponsored by the USEPA (EPA 1983a).

This information must be interpreted carefully, because of the ability of many polluted dust and dirt particles to be resuspended and then redeposited within the urban area. In many cases, the measured atmospheric deposition measurements include material that was previously residing and measured in other urban runoff pollutant source areas. Also, only small amounts of the atmospheric deposition material would directly contribute to runoff. Rain is subjected to infiltration and the dry fall particulates are likely mostly incorporated with surface soils and only small fractions are then eroded during rains. Therefore, mass balances and determinations of urban runoff deposition and accumulation from different source areas can be highly misleading, unless transfer of material between source areas and the effective yield of this material to the receiving water is considered. Depending on the land use, relatively little of the dustfall in urban areas likely contributes to stormwater discharges.

Dustfall and precipitation affect all of the major urban runoff source areas in an urban area. Dustfall, however, is typically not a major pollutant source but fugitive dust is mostly a mechanism for pollutant transport. Most of the dustfall monitored in an urban area is resuspended particulate matter from street surfaces or wind erosion products from vacant areas (Pitt 1979). Point source pollutant emissions can also significantly contribute to dustfall pollution, especially in industrial areas. Transported dust from regional agricultural activities can also significantly affect urban stormwater.

Wind transported materials are commonly called “dustfall.” Dustfall includes sedimentation, coagulation with subsequent sedimentation and impaction. Dustfall is normally measured by collecting dry samples, excluding rainfall and snowfall. If rainout and washout are included, one has a measure of total atmospheric fallout. This total atmospheric fallout is sometimes called “bulk precipitation.” Rainout removes contaminants from the atmosphere by condensation processes in clouds, while washout is the removal of contaminants by the falling rain. Therefore, precipitation can include natural contamination associated with condensation nuclei in addition to collecting atmospheric pollutants as the rain or snow falls. In some areas, the contaminant contribution by dry deposition is small, compared to the contribution by precipitation (Malmquist 1978). However, in heavily urbanized areas, dustfall can contribute more of an annual load than the wet precipitation, especially when dustfall includes resuspended materials.

Table 3-5 summarizes rain quality reported by several researchers. As expected, the non-urban area rain quality can be substantially better than urban rain quality. Many of the important heavy metals, however, have not been detected in rain in many areas of the country. The most important heavy metals found in rain have been lead and zinc, both being present in rain in concentrations from about 20 μg/L up to several hundred μg/L. It is expected that more recent lead rainfall concentrations would be substantially less, reflecting the decreased use of leaded gasoline since these measurements were taken. Iron is also present in relatively high concentrations in rain (about 30 to 40 μg/L).

Table 3-5. Summary of reported rain quality.

	Rural-Northwest (Quilayute, WA)[1]	**Rural-Northeast (Lake George, NY)[1]**	**Urban-Northwest (Lodi, NJ)[2]**	**Urban-Midwest (Cincinnati, OH)[3]**	**Other Urban[3]**	**Continental Avg. (32 locations)[1]**
Suspended solids, mg/L				13		
Volatile suspended solids, mg/L				3.8		
Inorganic nitrogen, mg/L as N				0.69		
Ammonia, mg/L as N					0.7	
Nitrates, mg/L as N					0.3	
Total phosphates, mg/L as P					<0.1	
Ortho phosphate, mg/L as P				0.24		
Scandium, μg/L	<0.002	nd				nd
Titanium, μg/L	nd	nd				nd
Vanadium, μg/L	nd	nd				nd
Chromium, μg/L	<2	nd	1			nd
Manganese, μg/L	2.6	3.4				12
Iron, μg/L	32	35				
Cobalt, μg/L	0.04	nd				nd
Nickel, μg/L	nd	nd	3			43
Copper, μg/L	3.1	8.2	6			21
Zinc, μg/L	20	30	44			107
Lead, μg/L			45			

1) Rubin 1976
2) Wilbur and Hunter 1980
3) Manning, *et al.* 1976

The concentrations of various urban runoff pollutants associated with dry dustfall are summarized in Table 3-6. Urban, rural and oceanic dry dustfall samples contained more than 5,000 mg iron/kg total solids. Zinc and lead were present in high concentrations. These constituents can have concentrations of up to several thousand mg of pollutant per kg of dry dustfall. Spring, *et al*. (1978) monitored dry dustfall near a major freeway in Los Angeles, CA. Based on a series of samples collected over several months, they found that lead concentrations on and near the freeway can be about 3,000 mg/kg, but as low as about 500 mg/kg 150 m (500 feet) away. In contrast, the chromium concentrations of the dustfall did not vary substantially between the two locations and approached oceanic dustfall chromium concentrations.

Much of the monitored atmospheric dustfall and precipitation would not reach the urban runoff receiving waters. The percentage of dry atmospheric deposition retained in a rural watershed was extensively monitored and modeled in Oakridge, TN (Barkdoll, *et al*. 1977). They found that about 98% of the lead in dry atmospheric deposits was retained in the watershed, along with about 95% of the cadmium, 85% of the copper, 60% of the chromium and magnesium and 75% of the zinc and mercury. Therefore, if the dry deposition rates were added directly to the yields from other urban runoff pollutant sources, the resultant urban runoff loads would be very much overestimated.

Tables 3-7 and 3-8 report bulk precipitation (dry dustfall plus rainfall) quality and deposition rates as reported by several researchers. For the Knoxville, KY, area (Betson 1978), chemical oxygen demand (COD) was found to be the largest component in the bulk precipitation monitored, followed by filterable residue and nonfilterable residue. Table 3-8 also presents the total watershed bulk precipitation, as the percentage of the total stream flow output, for the three Knoxville watersheds studies. This shows that almost all of the pollutants presented in the urban runoff streamflow outputs could easily be accounted for by bulk precipitation deposition alone. Betson concluded that bulk precipitation is an important component for some of the constituents in urban runoff, but the transport and resuspension of particulates from other areas in the watershed are overriding factors.

Table 3-6. Atmosphere dustfall quality.

Constituent, (mg constituent/kg total solids)	Urban[1]	Rural/ suburban[1]	Oceanic[1]	Near freeway (LA)[2]	500' from freeway (LA)[2]
pH				4.3	4.7
Phosphate-Phosphorous				1200	1600
Nitrate-Nitrogen, µg/L				5800	9000
Scandium, µg/L	5	3	4		
Titanium, µg/L	380	810	2700		
Vanadium, µg/L	480	140	18		
Chromium, µg/L	190	270	38	34	45
Manganese, µg/L	6700	1400	1800		
Iron, µg/L	24000	5400	21000		
Cobalt, µg/L	48	27	8		
Nickel, µg/L	950	1400			
Copper, µg/L	1900	2700	4500		
Zinc, µg/L	6700	1400	230		
Lead, µg/L				2800	550

1) Summarized by Rubin 1976
2) Spring 1978

Rubin (1976) stated that resuspended urban particulates are returned to the earth's surface and waters in four main ways: gravitational settling, impaction, precipitation and washout. Gravitational settling, as dry deposition, returns most of the particles. This not only involves the settling of relatively large fly ash and soil particles, but also the settling of smaller particles that collide and coagulate. Rubin stated that particles that are less than 0.1 µm in diameter move randomly in the air and collide often with other particles. These small particles can grow rapidly by this coagulation process. These small particles would soon be totally depleted in the air if they were not constantly replenished. Particles in the 0.1 to 1.0 µm range are also removed primarily by coagulation. These larger particles grow more slowly than the smaller particles because they move less rapidly in the air, are somewhat less numerous and, therefore, collide less often with other particles. Particles with diameters larger than 1 µm have appreciable settling velocities. Those particles about 10 µm in diameter can settle rapidly, although they can be kept airborne for extended periods of time and for long distances by atmospheric turbulence.

The second important particulate removal process from the atmosphere is impaction. Impaction of particles near the earth's surface can occur on vegetation, rocks and building surfaces. The third form of particulate removal from the atmosphere is precipitation, in the form of rain and snow. This is caused by the rainout process where the particulates are removed in the cloud-forming process. The fourth important removal process is washout of the particulates below the clouds during the precipitation event. Therefore, it is easy to see that re-entrained particles (especially from street surfaces, other paved surfaces, rooftops and from soil erosion) in urban areas can be readily redeposited through these various processes, either close to the points of origin or at some distance away.

Table 3-7. Bulk precipitation quality.

Constituent (all units mg/L except pH)	Urban (average of Knoxville St. Louis & Germany)[1]	Rural (Tennessee)[1]	Urban (Guteburg, Sweden)[2]
Calcium	3.4	0.4	
Magnesium	0.6	0.1	
Sodium	1.2	0.3	
Chlorine	2.5	0.2	

Sulfate	8.0	8.4	
pH	5.0	4.9	
Organic Nitrogen	2.5	1.2	
Ammonia Nitrogen	0.4	0.4	2
Nitrite plus Nitrate-N	0.5	0.4	1
Total phosphate	1.1	0.8	0.03
Potassium	1.8	0.6	
Total iron	0.8	0.7	
Manganese	0.03	0.05	
Lead	0.03	0.01	0.05
Mercury	0.01	0.0002	
Nonfilterable residue	16		
Chemical Oxygen Demand	65		10
Zinc			0.08
Copper			0.02

1) Betson 1978
2) Malmquist 1978

Pitt (1979) monitored airborne concentrations of particulates near typical urban roads. He found that on a number basis, the downwind roadside particulate concentrations were about 10% greater than upwind conditions. About 80% of the concentration increases, by number, were associated with particles in the 0.5 to 1.0 µm size range. However, about 90% of the particle concentration increases by weight were associated with particles greater than 10 µm. Pitt found that the rate of particulate resuspension from street surfaces increases when the streets are dirty (cleaned infrequently) and varied widely for different street and traffic conditions. The resuspension rates were calculated based upon observed long-term accumulation conditions on street surfaces for many different study area conditions, and varied from about 0.30 to 3.6 kg per curb-km (one to 12 lb per curb-mile) of street per day.

Murphy (1975) described a Chicago study where airborne particulate material within the city was microscopically examined, along with street surface particulates. The particulates from both of these areas were found to be similar (mostly limestone and quartz) indicating that the airborne particulates were most likely resuspended street surface particulates, or were from the same source.

Table 3-8. Urban bulk precipitation deposition rates (Betson 1978)[1].

Rank	Constituent	Average Bulk Deposition Rate (kg/ha/yr)	Average Bulk Prec. as a % of Total Streamflow Output
1	Chemical oxygen demand	530	490
2	Filterable residue	310	60
3	Nonfilterable residue	170	120
4	Alkalinity	150	120
5	Sulfate	96	470
6	Chloride	47	360
7	Calcium	38	170
8	Potassium	21	310
9	Organic nitrogen	17	490
10	Sodium	15	270
11	Silica	11	130
12	Magnesium	9	180
13	Total Phosphate	9	130
14	Nitrite and Nitrate-N	5.7	360

15	Soluble phosphate	5.3	170
16	Ammonia Nitrogen	3.2	1,100
17	Total Iron	1.9	47
18	Fluoride	1.8	300
19	Lead	1.1	650
20	Manganese	0.54	270
21	Arsenic	0.07	720
22	Mercury	0.008	250

1) Average for three Knoxville, KY, watersheds.

PEDCo (1977) found that the re-entrained portion of the traffic-related particulate emissions (by weight) is an order of magnitude greater than the direct emissions accounted for by vehicle exhaust and tire wear. They also found that particulate resuspensions from a street are directly proportional to the traffic volume and that the suspended particulate concentrations near the streets are associated with relatively large particle sizes. The medium particle size found, by weight, was about 15 µm, with about 22% of the particulates occurring at sizes greater than 30 µm. These relatively large particle sizes resulted in substantial particulate fallout near the road. They found that about 15% of the resuspended particulates fall out at 10 m, 25% at 20 m, and 35% at 30 m from the street (by weight).

In a similar study Cowherd, *et al.* (1977) reported a wind erosion threshold value of about 5.8 m/s (13 mph). At this wind speed, or greater, significant dust and dirt losses from the road surface could result, even in the absence of traffic-induced turbulence. Rolfe and Reinbold (1977) also found that most of the particulate lead from automobile emissions settled out within 100 m of roads. However, the automobile lead does widely disperse over a large area. They found, through multi-elemental analyses, that the settled outdoor dust collected at or near the curb was contaminated by automobile activity and originated from the streets.

Source Area Sheetflow and Particulate Quality

The following discussion summarizes the source area sheetflow and particulate quality data obtained from several studies conducted in California, Washington, Nevada, Wisconsin, Illinois, Ontario, Colorado, New Hampshire, and New York since 1979. Most of the data obtained were for street dirt chemical quality, but a relatively large amount of parking and roof runoff quality data have also been obtained. Only a few of these studies evaluated a broad range of source areas or land uses.

Source Area Particulate Quality

Particulate potency factors (usually expressed as mg pollutant/kg dry particulate residue) for many samples are summarized on Tables 3-9 and 3-10. These data can help recognize critical source areas, but care must be taken if they are used for predicting runoff quality because of likely differential effects due to washoff and erosion from the different source areas. These data show the variations in chemical quality between particles from different land uses and source areas. Typically, the potency factors increase as the use of an area becomes more intensive, but the variations are slight for different locations throughout the country. Increasing concentrations of heavy metals with decreasing particle sizes was also evident, for those studies that included particle size information. Only the quality of the smallest particle sizes are shown on these tables because they best represent the particles that are removed during rains.

Warm Weather Sheetflow Quality

Sheetflow data, collected during actual rain, are probably more representative of runoff conditions than the previously presented dry particulate quality data because they are not further modified by washoff mechanisms. These data, in conjunction with source area flow quantity information, can be used to predict outfall conditions and the magnitude of the relative sources of critical pollutants. Tables 3-11 through 3-14 summarize warm weather sheetflow observations, separated by source area type and land use, from many locations. The major source area categories are listed below:

1. Roofs
2. Paved parking areas
3. Paved storage areas
4. Unpaved parking and storage areas
5. Paved driveways
6. Unpaved driveways
7. Dirt walks
8. Paved sidewalks
9. Streets
10. Landscaped areas

11. Undeveloped areas
12. Freeway paved lanes and shoulders

Table 3-9. Summary of observed street dirt mean chemical quality (mg constituent/kg solids).

Constituent	Residential	Commercial	Industrial
P	620 (4)		670 (4)
	540 (6)	400 (6)	
	1100 (5)	1500 (5)	
	710 (1)	910 (1)	
	810 (3)		
TKN	1030 (4)		560 (4)
	3000 (6)	1100 (6)	
	290 (5)	340 (5)	
	2630 (3)	4300 (2)	
	3000 (2)		
COD	100,000 (4)		65,000 (4)
	150,000 (6)	110,000 (6)	
	180,000 (5)	250,000 (5)	
	280,000 (1)	340,000 (1)	
	180,000 (3)	210,000 (2)	
	170,000 (2)		
Cu	162 (4)		360 (4)
	110 (6)	130 (6)	
	420 (2)	220 (2)	
Pb	1010 (4)		900 (4)
	1800 (6)	3500 (6)	
	530 (5)	2600 (5)	
	1200 (1)	2400 (1)	
	1650 (3)	7500 (2)	
	3500 (2)		
Zn	460 (4)		500 (4)
	260 (5)	750 (5)	
	325 (3)	1200 (2)	
	680 (2)		
Cd	<3 (5)	5 (5)	
	4 (2)	5 (2)	
Cr	42 (4)		70 (4)
	31 (5)	65 (5)	
	170 (2)	180 (2)	

References; location; particle size described:
(1) Bannerman, *et al.* 1983 (Milwaukee, WI) <31µm
(2) Pitt 1979 (San Jose, CA) <45 µm
(3) Pitt 1985 (Bellevue, WA) <63 µm
(4) Pitt and McLean 1986 (Toronto, Ontario) <125 µm
(5) Pitt and Sutherland 1982 (Reno/Sparks, NV) <63 µm
(6) Terstriep, *et al.* 1982 (Champaign/Urbana, IL) >63 µm

Table 3-10. Summary of observed particulate quality for other source areas (means for <125 µm particles) (mg constituent/kg solids).

	P	TKN	COD	Cu	Pb	Zn	Cr
Residential/Commercial Land Uses							
Roofs	1500	5700	240,000	130	980	1900	77
Paved parking	600	790	78,000	145	630	420	47

Unpaved driveways	400	850	50,000	45	160	170	20
Paved driveways	550	2750	250,000	170	900	800	70
Dirt footpath	360	760	25,000	15	38	50	25
Paved sidewalk	1100	3620	146,000	44	1200	430	32
Garden soil	1300	1950	70,000	30	50	120	35
Road shoulder	870	720	35,000	35	230	120	25
Industrial Land Uses							
Paved parking	770	1060	130,000	1110	650	930	98
Unpaved parking/storage	620	700	110,000	1120	2050	1120	62
Paved footpath	890	1900	120,000	280	460	1300	63
Bare ground	700	1700	70,000	91	135	270	38

Source: Pitt and McLean 1986 (Toronto, Ontario)

Table 3-11. Sheetflow quality summary for other source areas (mean concentration and source of data).

Pollutant and Land Use	Roofs	Paved Parking	Paved Storage	Unpaved Parking/Storage	Paved Driveways	Unpaved Driveways	Dirt Walks	Paved Sidewalks	Streets
Total Solids (mg/L)									
Residential:	58 (5) 64 (1) 18 (4)	1790 (5)	73 (5)		510 (5)		1240 (5)	49 (5)	325 (5) 235 (4)
Commercial:	95 (1) 190 (4)	340 (2) 240 (1) 102 (7)							325 (4)
Industrial:	113 (5)	490 (5)	270 (5)	1250 (5)	506 (5)	5620 (5)		580 (5)	1800 (5)
Suspended Solids (mg/L)									
Residential:	22 (1) 13 (5)	1660 (5)	41 (5)		440 (5)		810 (5)	20 (5)	242 (5)
Commercial:		270 (2) 65 (1) 41 (7)							242 (5)
Industrial:	4 (5)	306 (5)	202 (5)	730 (5)	373 (5)	4670 (5)		434 (5)	1300 (5)
Dissolved Solids (mg/L)									
Residential:	42 (10 5 (5)	130 (5)	32 (5)		70 (5)		430 (5)	29 (5)	83 (5) 83 (4)
Commercial:		70 (2) 175 (1) 61 (7)							83 (5)
Industrial:	109 (5)	184 (5)	68 (5)	520 (5)	133 (5)	950 (5)		146 (5)	500 (5)

Table 3-11. Sheetflow quality summary for other source areas (mean concentration and source of data) (Continued).

Pollutant and Land Use	Roofs	Paved Parking	Paved Storage	Unpaved Parking/Storage	Paved Driveways	Unpaved Driveways	Dirt Walks	Paved Sidewalks	Streets
BOD_5 (mg/L)									
Residential:	3 (4)	22 (4)							13 (4)
Commercial:	7 (4)	11 (1) 4 (8)							
COD (mg/L)									
Residential:	46 (5) 27 (1) 20 (4)	173 (5)	22 (5)		178 (5)			62 (5)	174 (5) 170 (4)
Commercial:	130 (4)	190 (2) 180 (4) 53 (1) 57 (8)							174 (5)
Industrial:	55 (5)	180 (5)	82 (5)	247 (5)	138 (5)	418 (5)		98 (5)	322 (5)
Total Phosphorus (mg/L)									
Residential:	0.03 (5) 0.05 (1) 0.1 (4)				0.36 (5)		0.20 (5)	0.80 (5)	0.62 (5) 0.31 (4)
Commercial:	0.03 (4) 0.07 (4)	0.16 (1) 0.15 (7) 0.73 (5) 0.9 (2) 0.5 (4)							0.62 (5)
Industrial:	<0.06 (5)	2.3 (5)	0.7 (5)	1.0 (5)	0.9 (5)	3.0 (5)		0.82 (5)	1.6 (5)

Table 3-11. Sheetflow quality summary for other source areas (mean concentration and source of data) (Continued).

Pollutant and Land Use	Roofs	Paved Parking	Paved Storage	Unpaved Parking/Storage	Paved Driveways	Unpaved Driveways	Dirt Walks	Paved Sidewalks	Streets
Total Phosphate (mg/L)									
Residential:	<0.04 (5) 0.08 (4)				<0.2 (5)		0.66 (5)	0.64 (5)	0.07 (5) 0.12 (4)
Commercial:	0.02 (4)	0.03 (5) 0.3 (2) 0.5 (4) 0.04 (7) 0.22 (8)	<0.02 (5)						0.07 (5)
Industrial:	<0.02 (5)	0.6 (5)	0.06 (5)	0.13 (5)	<0.02 (5)	0.10 (5)		0.03 (5)	0.15 (5)

Pollutant and Land Use	Roofs	Paved Parking	Paved Storage	Unpaved Parking/Storage	Paved Driveways	Unpaved Driveways	Dirt Walks	Paved Sidewalks	Streets
TKN (mg/L)									
Residential:	1.1 (5) 0.71 (4)				3.1 (5)		1.3 (5)	1.1 (5)	2.4 (5) 2.4 (4)
Commercial:	4.4 (4)	3.8 (5) 4.1 (2) 1.5 (4) 1.0 (1) 0.8 (8)							2.4 (5)
Industrial:	1.7 (5)	2.9 (5)	3.5 (5)	2.7 (5)	5.7 (5)	7.5 (5)		4.7 (5)	5.7 (5)
Ammonia (mg/L)									
Residential:	0.1 (5) 0.9 (1) 0.5 (4)	0.1 (5)	0.3 (5)		<0.1 (5)		0.5 (5)	0.3 (5)	<0.1 (5) 0.42 (4)
Commercial:	1.1 (4)	1.4 (2) 0.35 (4) 0.38 (1)							<0.1 (5)
Industrial:	0.4 (5)	0.3 (5)	0.3 (5)	<0.1 (5)	<0.1 (5)	<0.1 (5)		<0.1 (5)	<0.1 (5)

Table 3-11. Sheetflow quality summary for other source areas (mean concentration and source of data) (Continued).

Pollutant and Land Use	Roofs	Paved Parking	Paved Storage	Unpaved Parking/Storage	Paved Driveways	Unpaved Driveways	Dirt Walks	Paved Sidewalks	Streets
Phenols (mg/L)									
Residential:	2.4 (5)	12.2 (5)	30.0 (5)		9.7 (5)		<0.4 (5)	8.6 (5)	6.2 (5)
Industrial:	1.2 (5)	9.4 (5)	2.6 (5)	8.7 (5)	7.0 (5)	7.4 (5)		8.7 (5)	24 (7)
Aluminum (μg/L)									
Residential:	0.4 (5)	3.2 (5)	0.38 (5)		5.3 (5)		<0.03 (5)	0.5 (5)	1.5 (5)
Industrial:	<0.2 (5)	3.5 (5)	3.1 (5)	9.2 (5)	3.4 (5)	41 (5)		1.2 (5)	14 (5)
Cadmium (μg/L)									
Residential:	<4 (5) 0.6 (1)	2 (5)	<5 (5)		5 (5)		<1 (5)	<4 (5)	<5 (5)
Commercial:		5.1 (7) 0.6 (8)							<5 (5)
Industrial:	<4 (5)	<4 (5)	<4 (5)	<4 (5)	<4 (5)	<4 (5)		<4 (5)	<4 (5)
Chromium (μg/L)									

Pollutant and Land Use	Roofs	Paved Parking	Paved Storage	Unpaved Parking/Storage	Paved Driveways	Unpaved Driveways	Dirt Walks	Paved Sidewalks	Streets
Residential:	<60 (5) <5 (4)	20 (5) 71 (4)	<10 (5)		<60 (5)		<10 (5)	<60 (5)	<60 (5) 49 (4)
Commercial:	<5 (4)	19 (7) 12 (8)							<60 (5)
Industrial:	<60 (5)	<60 (5)	<60 (5)	<60 (5)	<60 (5)	70 (5)		<60 (5)	<60 (5)

Table 3-11. Sheetflow quality summary for other source areas (mean concentration and source of data) (Continued).

Pollutant and Land Use	Roofs	Paved Parking	Paved Storage	Unpaved Parking/Storage	Paved Driveways	Unpaved Driveways	Dirt Walks	Paved Sidewalks	Streets
Copper (µg/L)									
Residential:	10 (5) <5 (4)	100 (5)	20 (5)		210 (5)		20 (5)	20 (5)	40 (5) 30 (4)
Commercial:	110 (4)	40 (2) 46 (4) 110 (7)							40 (5)
Industrial:	<20 (5)	480 (5)	260 (5)	120 (5)	40 (5)	140 (5)		30 (5)	220 (5)
Lead (µg/L)									
Residential:	<40 (5) 30 (3) 48 (1) 17 (4)	250 (5)	760 (5)		1400 (5)		30 (5)	80 (5)	180 (5) 670 (4)
Commercial:	19 (4) 30 (1)	200 (2) 350 (3) 1090 (4) 146 (1) 255 (7) 54 (8)							180 (5)
Industrial:	<40 (5)	230 (5)	280 (5)	210 (5)	260 (5)	340 (5)		<40 (5)	560 (5)

Table 3-11. Sheetflow quality summary for other source areas (mean concentration and source of data) (Continued).

Pollutant and Land Use	Roofs	Paved Parking	Paved Storage	Unpaved Parking/Storage	Paved Driveways	Unpaved Driveways	Dirt Walks	Paved Sidewalks	Streets
Zinc (µg/L)									
Residential:	320 (5) 670 (1)	520 (5)	390 (5)		1000 (5)		40 (5)	60 (5)	180 (5) 140 (4)

Pollutant and Land Use	Roofs	Paved Parking	Paved Storage	Unpaved Parking/Storage	Paved Driveways	Unpaved Driveways	Dirt Walks	Paved Sidewalks	Streets
	180 (4)								
Commercial:	310 (1) 80 (4)	300 (5) 230 (4) 133 (1) 490 (7)							180 (5)
Industrial:	70 (5)	640 (7)	310 (5)	410 (5)	310 (5)	690 (5)		60 (5)	910 (5)

References:
(1) Bannerman, *et al*. 1983 (Milwaukee, WI) (NURP)
(2) Denver Regional Council of Governments 1983 (NURP)
(3) Pitt 1983 (Ottawa)
(4) Pitt and Bozeman 1982 (San Jose)
(5) Pitt and McLean 1986 (Toronto)
(6) STORET Site #590866-2954309 (Shop-Save-Durham, NH) (NURP)
(7) STORET Site #596296-2954843 (Huntington-Long Island, NY) (NURP)

Table 3-12. Sheetflow quality summary for undeveloped landscaped and freeway pavement areas (mean observed concentrations and source of data).

Pollutants	Landscaped Areas	Undeveloped Areas	Freeway Paved Lane and Shoulder Areas
Total Solids, mg/L	388 (4)	588 (4)	340 (5)
Suspended Solids, mg/L	100 (4)	400 (1) 390 (4)	180 (5)
Dissolved Solids, mg/L	288 (4)	193 (4)	160 (5)
BOD_5, mg/L	3 (3)	- - - -	10 (5)
COD, mg/L	70 (3) 26 (4)	72 (1) 54 (4)	130 (5)
Total Phosphorus, mg/L	0.42 (3) 0.56 (4)	0.40 (1) 0.68 (4)	- - - -
Total Phosphate, mg/L	0.32 (3) 0.14 (4)	0.10 (1) 0.26 (4)	0.38 (5)
TKN, mg/L	1.32 (3) 3.6 (4)	2.9 (1) 1.8 (4)	2.5 (5)
Ammonia, mg/L	1.2 (3) 0.4 (4)	0.1 (1) <0.1 (4)	- - - -
Phenols, µg/L	0.8 (4)	- - - -	- - - -
Aluminum, µg/L	1.5 (4)	11 (4)	- - - -
Cadmium, µg/L	<3 (4)	<4 (4)	60 (5)
Chromium, µg/L	10 (3)	<60 (4)	70 (5)

Copper, µg/L	<20 (4)	40 (1) 31 (3) <20 (4)	120 (5)
Lead, µg/L	30 (2) 35 (3) <30 (4)	100 (1) 30 (2) <40 (4)	2000 (5)
Zinc, µg/L	10 (3)	100 (1) 100 (4)	460 (5)

References:
(1) Denver Regional Council of Governments 1983 (NURP)
(2) Pitt 1983 (Ottawa)
(3) Pitt and Bozeman 1982 (San Jose)
(4) Pitt and McLean 1986 (Toronto)
(5) Shelly and Gaboury 1986 (Milwaukee)

Table 3-13. Source area bacteria sheetflow quality summary (means).

Pollutant and Land Use	Roofs	Paved Parking	Paved Storage	Unpaved Parking/ Storage	Paved Driveways	Unpaved Driveways	Dirt Walks	Paved Sidewalks	Streets	Land-scaped	Un-developed	Freeway Paved Lane and Shoulders
Fecal Coliforms (#/100 ml) Residential: Commercial Industrial:	85 (2) <2 (3) 1400 (4) 9 (3) 1600 (4)	250,000 (4) 2900 (2) 350 (3) 210 (1) 480 (5) 23,000 (6) 8660 (6)	100 (4) 9200 (4)	18,000 (4)	600 (4) 66,000 (4)	300,000 (4)		11,000 (4) 55,000 (4)	920 (3) 6,900 (4) 100,000 (4)	3300 (4)	5400 (2) 49 (3)	1500 (7)
Fecal Strep (#/100 ml) Residential: Commercial: Industrial:	170 (2) 920 (3) 2200 (4) 17 (2) 690 (4)	190,000 (4) 11,900 (2) >2400 (3) 770 (1) 1120 (5) 62,000 (6) 7300 (4)	<100 (4) 2070 (4)	8100 (4)	1900 (4) 36,000 (4)	21,000 (4)	1800 (4)	3600 (4)	>2400 (3) 7300 (4) 45,000 (4)	43,000 (4)	16,500 (2) 920 (3)	2200 (7)
Pseudo, Aerug (#/100 ml) Residential: Industrial:	30,000 (4) 50 (4)	1900 (4) 5800 (4)	100 (4) 5850 (4)	14,000 (4)	600 (4) 14,300 (4)	100 (4)	600 (4)	3600 (4)	570 (4) 6200 (4)	2100 (4)		

References:
(1) Bannerman, *et al*. 1983 (Milwaukee, WI) (NURP)
(2) Pitt 1983 (Ottawa)
(3) Pitt and Bozeman 1982 (San Jose)
(4) Pitt and McLean 1986 (Toronto)
(5) STORET Site #590866-2954309 (Shop-Save-Durham, NH) (NURP)
(6) STORET Site #596296-2954843 (Huntington-Long Island, NY) (NURP)
(7) Kobriger, *et al.* 1981 and Gupta, *et al*. 1977

Table 3-14. Source area filterable pollutant concentration summary (means).

	Residential			Commercial			Industrial		
	Total	Filterable	Filterable (%)	Total	Filterable	Filterable (%)	Total	Filterable	Filt. (%)
Roof Runoff									
Solids (mg/L)	64 58	42 45	66 (1) 77 (3)				113	110	97 (3)
Phosphorus (mg/L)	0.054	0.013	24 (1)						
Lead (µg/L)	48	4	8 (1)						
Paved Parking									
Solids (mg/L)				240 102 1790	175 61 138	73 (1) 60 (4) 8 (3)	490	138	28 (3)
Phosphorus (mg/L)				0.16 0.9	0.03 0.3	19 (1) 33 (2)			
TKN (mg/L)				0.77	0.48	62 (5)			
Lead (µg/L)				146 54	5 8.8	3 (1) 16 (5)			
Arsenic (µg/L)				0.38	0.095	25 (5)			
Cadmium (µg/L)				0.62	0.11	18 (5)			
Chromium (µg/L)				11.8	2.8	24 (5)			
Paved Storage									
Solids (mg/L)				73	32	44 (3)	270	64	24 (3)

References:

(1) Bannerman, *et al*. 1983 (Milwaukee) (NURP)
(2) Denver Regional Council of Governments 1983 (NURP)
(3) Pitt and McLean 1986 (Toronto)
(4) STORET Site #590866-2954309 (Shop-Save-Durham, NH) (NURP)
(5) STORET Site #596296-2954843 (Huntington-Long Island, NY) (NURP)

Toronto warm weather sheetflow water quality data were plotted against the rain volume that had occurred before the samples were collected to identify any possible trends of concentrations with rain volume (Pitt and McLean 1986). The street runoff data obtained during the special washoff tests were also compared with the street sheetflow data obtained during the actual rain events (Pitt 1987). These data observations showed definite trends of solids concentrations verses rain volume for most of the source area categories, as shown later. Sheetflows from all pervious areas combined had the highest total solids concentrations from any source category, for all rain events. Other paved areas (besides streets) had total solids concentrations similar to runoff from smooth industrial streets. The concentrations of total solids in roof runoff were almost constant for all rain events, being slightly lower for small rains than for large rains. No other pollutant, besides SS, had observed trends of concentrations with rain depths for the samples collected in Toronto. Lead and zinc concentrations were highest in sheetflows from paved parking areas and streets, with some high zinc concentrations also found in roof drainage samples. High bacteria populations were found in sidewalk, road, and some bare ground sheetflow samples (collected from locations where dogs would most likely be "walked").

Some of the Toronto sheetflow contributions were not sufficient to explain the concentrations of some constituents observed in runoff at the outfall. High concentrations of dissolved chromium, dissolved copper, and dissolved zinc in a Toronto industrial outfall during both wet and dry weather could not be explained by wet weather sheetflow observations (Pitt and McLean 1986). As an example, very few detectable chromium observations were obtained in any of the more than 100 surface sheetflow samples analyzed. Similarly, most of the fecal coliform populations observed in sheetflows were significantly lower than those observed at the outfall, especially during snowmelt. It is expected that some industrial wastes, possibly originating from metal plating operations, were the cause of these high concentrations of dissolved metals at the outfall and that some sanitary sewage was entering the storm drainage system.

Table 3-14 summarizes the little filterable pollutant concentration data available for different source areas. Most of the available data are for residential roofs and commercial parking lots.

Sources of Stormwater Toxicants Case Study in Birmingham, AL

Pitt, *et al*. (1995) studied stormwater runoff samples from a variety of source areas under different rain conditions as summarized in Table 3-15. All of the samples were analyzed in filtered (0.45 µm filter) and non-filtered forms to enable partitioning of the toxicants into "particulate" (non-filterable) and "dissolved" (filterable) forms.

Table 3-15. Numbers of samples collected from each source area type.

Local Source Areas[1]	Residential	Commercial/ Institutional	Industrial	Mixed
Roofs	5	3	4	
Parking Areas	2	11	3	
Storage Areas	na	2	6	
Streets	1	1	4	
Loading Docks	na	na	3	
Vehicle Service Area	na	5	na	
Landscaped Areas	2	2	2	
Urban Creeks				19
Detention Ponds				12

1) All collected in Birmingham, AL.

The samples listed in Table 3-15 were all obtained from the Birmingham, AL, area. Samples were taken from shallow flows originating from homogeneous source areas by using several manual grab sampling procedures. For deep flows, samples were collected directly into the sample bottles. For shallow flows, a peristaltic hand operated vacuum pump created a small vacuum in the sample bottle, which then gently drew the sample directly into the container through a Teflon™ tube. About one liter of sample was needed, split into two containers: one 500 ml glass bottle with Teflon™ lined lid was used for the organic and toxicity analyses and another 500 ml polyethylene bottle was used for the metal and other analyses.

An important aspect of the research was to evaluate the effects of different land uses and source areas, plus the effects of rain characteristics, on sample toxicant concentrations. Therefore, careful records were obtained of the amount of rain and the rain intensity that occurred before the samples were obtained. Antecedent dry period data were also obtained to compare with the chemical data in a series of statistical tests.

All samples were handled, preserved, and analyzed according to accepted protocols (EPA 1982 and 1983b). The organic pollutants were analyzed using two gas chromatographs, one with a mass selective detector (GC/MSD) and another with an electron capture detector (GC/ECD). The pesticides were analyzed according to EPA method 505, while the base neutral compounds were analyzed according to EPA method 625 (but only using 100 ml samples). The pesticides were analyzed on a Perkin Elmer Sigma 300 GC/ECD using a J&W DB-1 capillary column (30m by 0.32 mm ID with a 1 µm film thickness). The base neutrals were analyzed on a Hewlett Packard 5890 GC with a 5970 MSD using a Supelco DB-5 capillary column (30m by 0.25 mm ID with a 0.2 µm film thickness). Table 3-16 lists the organic toxicants that were analyzed.

Table 3-16. Toxic pollutants analyzed in samples.

Pesticides Detention Limit = 0.3 µg/L	Phthalate Esters Detention Limit = 0.5 µg/L	Polycyclic Aromatic Hydrocarbons Detention Limit = 0.5 µg/L	Metals Detention Limit = 1 µg/L

BHC (Benzene hexachloride) Heptachlor Aldrin Endosulfan Heptachlor epoxide DDE (Dichlorodiphenyl dichloroethylene) DDD (Dichlorodiphenyl dichloroethane) DDT (Dichlorodiphenyl trichloroethane) Endrin Chlordane	Bis(2-ethylhexyl) Phthalate Butyl benzyl phthalate Di-n-butyl phthalate Diethyl phthalate Dimethyl phthalate Di-n-octyl phthalate	Acenaphthene Acenapthylene Anthracene Benzo (a) anthracene Benzo (a) pyrene Benzo (b) fluoranthene Benzo (ghi) perylene Benzo (k) fluoranthene Chrysene Dibenzo (a,h) anthracene	Fluoranthene Fluorene Indeno (1,2,3-cd) pyrene Naphthalene Phenanthrene Pyrene	Aluminum Cadmium Chromium Copper Lead Nickel Zinc

Metallic toxicants, also listed in Table 3-16, were analyzed using a graphite furnace equipped atomic absorption spectrophotometer (GFAA). EPA methods 202.2 (Al), 213.2 (Cd), 218.2 (Cr), 220.2 (Cu), 239.2 (Pb), 249.2 (Ni), and 289.2 (Zn) were followed in these analyses. A Perkin Elmer 3030B atomic absorption spectrophotometer was used after nitric acid digestion of the samples. Previous research (Pitt and McLean 1986; EPA 1983a) indicated that low detection limits were necessary in order to measure the filtered sample concentrations of the metals, which would not be achieved by use of a standard flame atomic absorption spectrophotometer. Low detection limits would enable partitioning of the metals between the solid and liquid phases to be investigated, an important factor in assessing the fates of the metals in receiving waters and in treatment processes.

The Microtox™ 100% sample toxicity screening test, from Azur Environmental (previously Microbics, Inc.), was selected for this research after comparisons with other laboratory bioassay tests. During the first research, 20 source area stormwater samples and combined sewer samples (obtained during a cooperative study being conducted in New York City) were split and sent to four laboratories for analyses using 14 different bioassay tests. Conventional bioassay tests were conducted using freshwater organisms at the EPA's Duluth, MN, laboratory and using marine organisms at the EPA's Narraganssett Bay, RI, laboratory. In addition, other bioassay tests, using bacteria, were also conducted at the Environmental Health Sciences Laboratory at Wright State University, Dayton, OH. The tests represented a range of organisms that included fish, invertebrates, plants, and microorganisms.

The conventional bioassay tests conducted simultaneously with the Microtox™ screening test for the 20 stormwater sheetflow and combined sewer overflow (CSO) samples were all short-term tests. However, some of the tests were indicative of chronic toxicity (e.g., life cycle tests and the marine organism sexual reproduction tests), whereas the others would be classically considered as indicative of acute toxicity (e.g., Microtox™ and the fathead minnow tests). The following list shows the major tests that were conducted by each participating laboratory:

1. University of Alabama at Birmingham, Environmental Engineering Laboratory
   Microtox™ bacterial luminescence tests (10-, 20-, and 35-minute exposures)
   using the marine *Photobacterium phosphoreum.*

2. Wright State University, Biological Sciences Department
   Macrofaunal toxicity tests:
      Daphnia magna (water flea) survival; *Lemma minor* (duckweed) growth;
         and *Selenastrum capricornutum* (green alga) growth.
   Microbial activity tests (bacterial respiration):
      Indigenous microbial electron transport activity;
      Indigenous microbial inhibition of β-galactosidase activity;
      Alkaline phosphatase for indigenous microbial activity;
      Inhibition of β-galactosidase for indigenous microbial activity; and
      Bacterial surrogate assay using *O*-nitrophenol-β-D-galactopyranside
         activity and *Escherichia coli.*

3. EPA Environmental Research Laboratory, Duluth, MN
   Ceriodaphnia dubia (water flea) 48-h survival; and
   Pimephales promelas (fathead minnow) 96-h survival.

4. EPA Environmental Research Laboratory, Narragansett Bay, RI

Champia parvula (marine red alga) sexual reproduction (formation of cystocarps after 5 to 7 d exposure); and

Arbacua punctulata (sea urchin) fertilization by sperm cells.

Table 3-17 summarizes the results of the toxicity tests. The *C. dubia. P. promelas,* and C. *Parvula* tests experienced problems with the control samples and, therefore, these results are therefore uncertain. The *A. pustulata* tests on the stormwater samples also had a potential problem with the control samples. The CSO test results (excluding the fathead minnow tests) indicated that from 50% to 100% of the samples were toxic, with most tests identifying the same few samples as the most toxic. The toxicity tests for the stormwater samples indicated that 0% to 40% of the samples were toxic. The Microtox™ screening procedure gave similar rankings for the samples as the other toxicity tests.

Table 3-17. Fraction of samples rated as toxic.

Sample series	Combined sewer overflows (%)	Stormwater (%)
Microtox™ marine bacteria	100	20
C. Dubia	60	0[1]
P. promelas	0[1]	0[1]
C. parvula	100	0[1]
A. punctulata	100	0[1]
D. magna	63	40
L. minor	50[1]	0

1) Results uncertain, see text

Laboratory toxicity tests can result in important information on the effects of stormwater in receiving waters, but actual in-stream taxonomic studies should also be conducted. A recently published proceedings of a conference on stormwater impacts on receiving streams (Herricks 1995) contains many examples of actual receiving water impacts and toxicity test protocols for stormwater.

All of the Birmingham samples represented separate stormwater. However, as part of the Microtox™ evaluation, several CSO samples from New York City were also tested to compare the different toxicity tests. These samples were collected from six CSO discharge locations having the following land uses:

1. 290 acres, 90% residential and 10% institutional.
2. 50 acres, 100% commercial.
3. 620 acres, 20% institutional, 6% commercial, 5% warehousing, 5% heavy industrial, and 64% residential.
4. 225 acres, 13% institutional, 4% commercial, 2% heavy industrial. and 81% residential.
5. 400 acres, 1% institutional and 99% residential.
6. 250 acres, 88% commercial. 6% warehousing, and 6% residential.

Therefore, there was a chance that some of the CSO samples may have had some industrial process waters. However, none of the Birmingham sheetflow samples could have contained any process waters because of how and where they were collected.

The Microtox™ screening procedure gave similar toxicity rankings for the 20 samples as the conventional bioassay tests. It is also a rapid procedure (requiring about one hour) and only requires small (<1 mL) sample volumes. The Microtox™ toxicity test uses marine bioluminescence bacteria and monitors the light output for different sample concentrations. About one million bacteria organisms are used per sample, resulting in highly repeatable results. The more toxic samples produce greater stress on the bacteria test organisms that results in a greater light attenuation compared to the control sample. Note that the Microtox™ procedure was not used during this research to determine the absolute toxicities of the samples or to predict the toxic effects of stormwater runoff on receiving waters. It was used to compare the relative toxicities of different samples that may indicate efficient source area treatment locations, and to examine changes in toxicity during different treatment procedures.

Results

Table 3-18 summarizes the source area sample data for the most frequently detected organic toxicants and for all of the metallic toxicants analyzed. The organic toxicants analyzed, but not reported, were generally detected in five, or less, of the non-filtered samples and in none of the filtered samples. Table 3-18 shows the mean, maximum, and minimum concentrations for the detected toxicants. Note that these values are based only on the observed concentrations. They do not consider the non-detectable conditions. Mean values based on total sample numbers for each source area category would therefore result in much lower concentrations. The frequency of detection is therefore an important consideration when evaluating organic toxicants. High detection frequencies for the organics may indicate greater potential problems than infrequent high concentrations.

Table 3-18. Stormwater toxicants detected in at least 10% of the source area sheetflow samples (µg/L, unless otherwise noted).

	Roof areas		Parking areas		Storage areas		Street runoff		Loading docks		Vehicle service areas		Landscaped areas		Urban creeks		Detention ponds	
	N.F.[1]	F.[2]	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.
otal samples	12	12	16	16	8	8	6	6	3	3	5	5	6	6	19	19	12	12
Base neutrals (detection limit = 0.5 µg/L)																		
,3-Dichlorobenzene detection frequency = 20% N.F. and 3% F.																		
No. detected[3]	3	2	3	2	1	1	1	1	0	0	3	2	3	2	2	0	1	1
Mean[4]	52	20	34	13	16	14	5.4	3.3			48	26	29	5.6	93		27	21
Max.	88	23	103	26							72	47	54	7.5	120			
Min.[5]	14	17	3.0	2.0							6.0	4.9	4.5	3.8	65			
Fluoranthene detection frequency = 20% N.F. and 12% F.																		
No. detected	3	2	3	2	1	0	1	1	0	0	3	2	3	2	1	0	2	1
Mean	23	9.3	37	2.7	4.5	0	0.6	0.5			39	3.6	13	1.0	130		10	6.6
Max.	45	14	110	5.4							53	6.8	38	1.3			14	
Min.	7.6	4.8	3.0	2.0							0.4	0.4	0.7	0.7			6.6	
Pyrene detection frequency = 17% N,F, and 7% F.																		
No. detected	1	0	3	2	1	0	1	1	0	0	3	2	2	0	1	0	2	1
Mean	28		40	9.8	8		1.0	0.7			44	4.1	5.3		100		31	5.8
Max.			120	20							51	7.4	8.2				57	
Min.			3.0	2.0							0.7	0.7	2.3				6.0	
Benzo(b)fluoranthene detection frequency = 15% N.F. and % F.																		
No. detected	4	0	3	0	0	0	1	0	0	0	2	0	1	0	2	0	0	0
Mean	76		53				14				98		30		36			
Max.	260		160								110				64			
Min.	6.4		3.0								90				8.0			
Benzo(k)fluoranthene detection frequency = 11% N.F. and % F.																		
No. detected	0	0	3	0	0	0	1	0	0	0	2	0	1	0	2	0	0	0
Mean			20				15				59		61		55			
Max.			1								103				78			
Min.			3.0								15				31			
Benzo(a)pyrene detection frequency = 15% N.F. and 0% F.																		
No. detected	4	0	3	0	0	0	1	0	0	0	2	0	1	0	2	0	0	0
Mean	99		40				19				90		54		73			
Max.	300		120								120				130			
Min.	34		3.0								60				19			

Table 3-18. Stormwater toxicants detected in at least 10% of the source area sheetflow samples (µg/L, unless otherwise noted).Continued.

	Roof areas		Parking areas		Storage areas		Street runoff		Loading docks		Vehicle service areas		Landscaped areas		Urban creeks		Detention ponds	
	N.F.[1]	F.[2]	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.
Total samples	12	12	16	16	8	8	6	6	3	3	5	5	6	6	19	19	12	12
Bis(2-chloroethyl) ether detection frequency = 12% N.F. and 2% F.																		
No. detected	3	1	2	0	0	0	1	0	0	0	1	1	1	0	1	0	1	0
Mean	42	17	20				15				45	23	56		200		15	
Max.	87	2	39															
Min.	20		2.0								6.0	4.9	4.5	3.8	65			
Bis(chloroisopropyl) ether detection frequency = 13% N.F. and 0% F.																		
No. detected	3	0	3	0	0	0	0	0	0	0	2	0	1	0	2	0	0	0
Mean	99		130								120		85		59			

Max.	150		400								160				78			
Min.	68		3.0								74				40			
Naphthalene detection frequency = 11% N.F. and 6% F.																		
No. detected	2	0	1	1	0	0	0	0	0	0	2	1	1	0	1	1	2	2
Mean	17		72	6.6							70	82	49		300	6.7	43	12
Max.	21										100						68	17
Min.	13										37						18	6.6
Benzo(a)anthracene detection frequency = 10% N.F. and 0% F.																		
No. detected	1	0	3	0	0	0	0	0	0	0	2	0	1	0	1	0	0	0
Mean	16		24								35		54		61			
Max.			73								39							
Min.			3.0								31							
Butylbenzyl phthalate detection frequency = 10% N.F. and 4% F.																		
No. detected	1	0	2	1	0	0	0	0	0	0	2	2	1	0	1	0	1	0
Mean	100		12	3.3							26	9.8	130		59		13	
Max.			21								48	16						
Min.			3.3								3.8	3						
Pesticides (detection limit = 0.3 µg/L)																		
Chlordane detection frequency = 11% N.F. and 0% F.																		
No. detected	2	0	2	0	3	0	1	0	0	0	1	0	0	0	0	0	0	0
Mean	1.6		1.0		1.7		0.8			0.8								
Max.	2.2		1.2		2.9													
Min.	0.9		0.8		1.0													

Table 3-18. Stormwater toxicants detected in at least 10% of the source area sheetflow samples (µg/L, unless otherwise noted).Continued.

	Roof areas		Parking areas		Storage areas		Street runoff		Loading docks		Vehicle service areas		Landscaped areas		Urban creeks		Detention ponds	
	N.F.[1]	F.[2]	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.
Total samples	12	12	16	16	8	8	6	6	3	3	5	5	6	6	19	19	12	12
Metals (detection limit = 1µg/L)																		
Lead detection frequency = 100% N.F. and 54% F.																		
No. detected	12	1	16	8	8	7	6	4	3	1	5	2	6	1	19	15	12	8
Mean	41	1.1	46	2.1	105	2.6	43	2.0	55	2.3	63	2.4	24	1.7	20	1.4	19	1.0
Max.	170		130	5.2	330	5.7	150	3.9	80		110	3.4	70		100	1.6	55	1.0
Min.	1.3		1.0	1.2	3.6	1.6	1.5	1.1	25		27	1.4	1.4		1.4	<1	1	<1
Zinc detection frequency = 99% N.F. and 98% F.																		
No. detected	12	12	16	16	8	7	6	6	2	2	5	5	6	6	19	19	12	12
Mean	250	220	110	86	1730	22	58	31	55	33	105	73	230	140	10	10	13	14
Max.	1580	1550	650	560	13100	100	130	76	79	62	230	230	1160	670	32	23	25	25
Min.	11	9	12	6	12	3.0	4.0	4.0	31	4.0	30	11	18	18	<1	<1	<1	<1
Copper detection frequency = 98% N.F. and 78% F.																		
No. detected	11	7	15	13	8	6	6	5	3	2	5	4	6	6	19	17	12	8
Mean	110	2.9	116	11	290	250	280	3.8	22	8.7	135	8.4	81	4.2	50	1.4	43	20
Max.	900	8.7	770	61	1830	1520	1250	11	30	15	580	24	300	8.8	440	1.7	210	35
Min.	1.5	1.1	10	1.1	10	1.0	10	1.0	15	2.6	1.5	1.1	1.9	0.9	<1	<1	0.2	<1
Aluminum detection frequency = 97% N.F. and 92% F.																		
No. detected	12	12	15	15	7	6	6	6	3	1	5	4	5	5	19	19	12	12
Mean	6850	230	3210	430	2320	180	3080	880	780	18	700	170	2310	1210	620	190	700	210
Max.	71300	1550	6480	2890	6990	740	10040	4380	930		1370	410	4610	1860	3250	500	1570	360
Min.	25	6.4	130	5.0	180	10	70	18	590		93	0.3	180	120	<5	<5	<5	<5
Cadmium detection frequency = 95% N.F. and 69% F.																		
No. detected	11	7	15	9	8	7	6	5	3	3	5	3	4	2	19	15	12	9
Mean	3.4	0.4	6.3	0.6	5.9	2.1	37	0.3	1.4	0.4	9.2	0.3	0.5	0.6	8.3	0.2	2	0.5
Max.	30	0.7	70	1.8	17	10	220	0.6	2.4	0.6	30	0.5	1	1	30	0.3	11	0.7

Min.	0.2	0.1	0.1	0.1	0.9	0.3	0.4	0.1	0.7	0.3	1.7	0.2	0.1	0.1	<0.1	<0.1	0.1	0.4
Chromium detection frequency = 91% N.F. and 55% F.																		
No. detected	7	2	15	8	8	5	5	4	3	0	5	1	6	5	19	15	11	8
Mean	85	1.8	56	2.3	75	11	9.9	1.8	17		74	2.5	79	2.0	62	1.6	37	2.0
Max.	510	2.3	310	5.0	340	32	30	2.7	40		320		250	4.1	710	4.3	230	3.0
Min.	5.0	1.4	2.4	1.1	3.7	1.1	2.8	1.3	2.4		2.4		2.2	1.4	<0.1	<0.1	<0.1	<0.1

Table 3-18. Stormwater toxicants detected in at least 10% of the source area sheetflow samples (μg/L, unless otherwise noted).Continued.

	Roof areas		Parking areas		Storage areas		Street runoff		Loading docks		Vehicle service areas		Landscaped areas		Urban creeks		Detention ponds	
	N.F.[1]	F.[2]	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.	N.F.	F.
Total samples	12	12	16	16	8	8	6	6	3	3	5	5	6	6	19	19	12	12
Nickel detection frequency = 90% N.F. and 37% F.																		
No. detected	10	0	14	4	8	1	5	0	3	1	5	1	4	1	18	16	11	8
Mean	16		45	5.1	55	87	17		6.7	1.3	42	31	53	2.1	29	2.3	24	3.0
Max.	70		130	13	170		70		8.1		70		130		74	3.6	70	6.0
Min.	2.6		4.2	1.6	1.9		1.2		4.2		7.9		21		<1	<1	1.5	<1
Other constituents (always detected, analyzed only for non-filtered samples)																		
pH																		
Mean	6.9		7.3		8.5		7.6		7.8		7.2		6.7		7.7		8.0	
Max.	8.4		8.7		12		8.4		8.3		8.1		7.2		8.6		9.0	
Min.	4.4		5.6		6.5		6.9		7.1		5.3		6.2		6.9		7.0	
Suspended solids																		
Mean	14		110		100		49		40		24		33		26		17	
Max.	92		750		450		110		47		38		81		140		60	
Min.	0.5		9.0		5.0		7.0		34		17		8.0		5.0		3.0	

1) N.F.: concentration associated with a non-filtered sample.
2) F.: concentration after the sample was filtered through a 0.45 μm membrane filter.
3) Number detected refers to the number of samples in which the toxicant was detected.
4) Mean values based only on the number of samples with a definite concentration of toxicant reported (not on the total number of samples analyzed).
5) The minimum values shown are the lowest concentration detected, they are not necessarily the detection limit.

Table 3-18 also summarizes the measured pH and SS concentrations. Most pH values were in the range of 7.0 to 8.5 with a low of 4.4 and a high of 11.6 for roof and concrete plant storage area runoff samples, respectively. This range of pH can have dramatic effects on the speciation of the metals analyzed. The SS concentrations were generally less than 100 mg/L, with impervious area runoff (e.g., roofs and parking areas) having much lower SS concentrations and turbidities compared to samples obtained from pervious areas (e.g., landscaped areas).

Out of more than 35 targeted organic compounds analyzed, 13 were detected in more than 10% of all samples, as shown in Table 3-18. The greatest detection frequencies were for 1,3-dichlorobenzene and fluoranthene, which were each detected in 23% of the samples. The organics most frequently found in these source area samples (i.e., polycyclic aromatic hydrocarbons (PAH), especially fluoranthene and pyrene) were similar to the organics most frequently detected at outfalls in prior studies (EPA 1983a).

Roof runoff, parking area and vehicle service area samples had the greatest detection frequencies for the organic toxicants. Vehicle service areas and urban creeks had several of the observed maximum organic compound concentrations. Most of the organics were associated with the non-filtered sample portions, indicating an association with the particulate sample fractions. The compound 1,3-dichlorobenzene was an exception, having a significant dissolved fraction.

In contrast to the organics, the heavy metals analyzed were detected in almost all samples, including the filtered sample portions. The non-filtered samples generally had much higher concentrations, with the exception of zinc, which was mostly associated with the dissolved sample portion (i.e., not associated with the SS). Roof runoff generally had the highest concentrations of zinc, probably from galvanized roof drainage components, as previously reported by Bannerman, *et al*. (1983). Parking and storage areas had the highest nickel concentrations, while vehicle service areas and street runoff had the highest concentrations of cadmium and lead. Urban creek samples had the highest copper concentrations, which were probably due to illicit industrial connections or other non-stormwater discharges.

Table 3-19 shows the relative toxicities of the collected stormwaters. A wide range of toxicities was found. About 9% of the non-filtered samples were considered highly toxic using the Microtox™ toxicity screening procedure. About 32% of the samples were moderately toxic and about 59% were considered non-toxic. The greatest percentage of samples considered the most toxic were from industrial storage and parking areas. Landscaped areas also had a high incidence of highly toxic samples (presumably due to landscaping chemicals) and roof runoff had some highly toxic samples (presumably due to high zinc concentrations). Treatability study activities indicated that filtering the samples through a range of fine sieves and finally a 0.45µm filter consistently reduced sample toxicities. The chemical analyses also generally found much higher toxicant concentrations in the non-filtered sample portions, compared to the filtered sample portions.

Replicate samples were collected from several source areas at three land uses during four different storm events to statistically examine toxicity and pollutant concentration differences due to storm and site conditions. These data indicated that variations in Microtox™ toxicities and organic toxicant concentrations may be partially explained by rain characteristics. As an example, high concentrations of many of the PAHs were associated with long antecedent dry periods and large rains (Barron 1990).

Pollution Prevention Associated with Selection of Building Materials

The selection of alternative building materials exposed to weather can have a significant effect on runoff quality. The above information showed obvious problems associated with roof runoff caused by the exposure of galvanized metal flashing to rain water. Treated wood has also been of concern as a likely source of heavy metal and organic toxicants.

Table 3-19. Relative toxicity of samples using Microtox™ (non-filtered).

Local Source Areas	Highly Toxic (%)	Moderately Toxic (%)	Not Toxic (%)	Number of Samples
Roofs	8	58	33	12
Parking Areas	19	31	50	16
Storage Areas	25	50	25	8
Streets	0	67	33	6
Loading Docks	0	67	33	3
Vehicle Service Areas	0	40	60	5
Landscaped Areas	17	17	66	6
Urban Creeks	0	11	89	19
Detention Ponds	8	8	84	12
All Areas	9	32	59	87

Microbics suggested toxicity definitions for 35 minute exposures:
Highly toxic - light decrease >60%
Moderately toxic - light decrease <60% & >20%
Not toxic - light decrease <20%

The detection of pentachlophenols in stormwater indicates important leaching from treated wood. Frequent detections of polycyclic aromatic hydrocarbons (PAHs) during the U.S. Environmental Protection Agency's Nationwide Urban Runoff Program (EPA 1983a) may possibly indicate leaching from creosote treated wood, in addition to fossil fuel combustion sources. High concentrations of copper, and some chromium and arsenic observations also indicate the potential of leaching from "CCA" (copper, chromium, and arsenic) treated wood. The significance of these leachate products in the receiving waters is currently unknown, but alternatives to these preservatives should be considered. Many cities use aluminum and concrete utility poles instead of treated wood poles. This is especially important considering that utility poles are usually located very close to the drainage system ensuring an efficient delivery of leachate products.

Many homes currently use wood stains containing pentachlorophenol and other wood preservatives. Similarly, the construction of retaining walls, wood decks and playground equipment with treated wood is common. Some preservatives (especially creosote) cause direct skin irritation, besides contributing to potential problems in receiving waters. Many of these wood products are at least located some distance from the storm drainage system, allowing some improvement to surface water quality by infiltration through pervious surfaces.

There is growing interest in the development and use of environmentally sensitive construction materials.
Studies conducted at the University of Alabama at Birmingham (Pitt, *et al.*, 1995) investigated toxic contributions to urban wet weather flow from sources such as roofs, parking areas, storage areas, streets, loading docks, vehicle service areas, and landscaped areas. Roof runoff, vehicle service area and parking lot samples were found to have the greatest organic toxicant detection frequencies and the highest levels of detected metals. However, relative pollutant contributions from various roofing, wooden and paving materials themselves are also a concern which has not been adequately addressed. Due to the common use of these surfaces in our urban environments, reduction of emissions at the source is desirable, and material substitution would seem a good place to start.

Roofing and Paving Materials

Other studies have verified the UAB research, confirming the important role played by roofs and paved surfaces to pollutant contributions to wet weather flow. Boller (1997) identified heavy metals such as cadmium, copper, lead and zinc as the critical metals in domestic wastewaters and, based on his flow studies, concluded that runoff from roofs and streets contribute 50-80% of these metals to the total mass flow in Swiss combined sewer systems. Roof runoff samples, from tile, polyester, and flat gravel roofs were analyzed and metal concentrations were found to vary tremendously with roof type. First flush analyses showed polyester roofs contributing highest concentrations of copper (6,817 μg/L), zinc (2,076 μg/L), cadmium (3.1 μg/L) and lead (510 μg/L). Concentrations in runoff from tile roofs were copper (1,905 μg/L), zinc (360 μg/L), cadmium (2.1 μg/L) and lead (172 μg/L). Runoff from flat gravel roofs also contributed copper (140 μg/L), zinc (36 μg/L), cadmium (0.2 μg/L) and lead (22 μg/L). Runoff from roofs was found to contain not only heavy metals, but polyaromatic hydrocarbons (PAHs) and organic halogens as well.
Mottier and Boller (1996), working in Zurich, measured metals concentrations in road runoff and found average values of 300 μg/L for lead, 4 μg/l for cadmium, 150 μg/L for copper and 500 μg/L for zinc. Information on pavement material type was not included. Averaged roof runoff concentrations (from tile and polyester roofs) were also measured at 16 μg/L for lead, 0.17 μg/L for cadmium, 225 μg/L for copper and 42 μg/l for zinc. Boller concluded that copper installations on buildings seem to represent the largest source for the discharges of this metal into the environment. He estimated annual zinc and copper corrosion from roof metal installations at 4-14 grams/sq. meter and 7.5-15 grams/sq. meter respectively. Stark, *et al* (1995) arrived at a similar conclusion, estimating that stormwater from roofs may be responsible for more the 60% of the copper in Austria's combined sewers.

Researchers in Marquette, Michigan, collecting wet weather flow concurrently at 33 sites during 12 storms detected discernable differences in runoff quality between a variety of impervious source areas (Steuer, *et al.* 1997). Commercial and residential rooftops were found to produce the lowest concentrations of suspended solids, but the highest concentrations of dissolved metals such as lead, zinc, cadmium, and copper. Parking lots produced the highest concentrations for all PAH compounds and high concentrations of zinc, total cadmium and total copper. Low traffic streets were also identified as a major producer of total cadmium.

Forster (1996) sampled and analyzed roof runoff for heavy metals (Cd, Cu, Zn, Pb) between April 1993 and May 1994. Measurement were made with an experimental roof system situated on the Campus of the University of Bayreuth and at various locations in the urban area of Bayreuth, Northern Bavaria. The experimental roof systems allowed the influence of different roof materials (concrete tiles, zinc sheet, pantiles, fibrous cement) on runoff quality to be compared. Large differences in runoff pollutant concentrations from various roofs were interpreted to indicate that the pollutants were not only being transported to the surface via the atmosphere, but also originating from the material itself. Extremely high values of zinc and copper were measured when the roof system or parts of it were made of metal panels, flashing, and gutters. For example, runoff concentrations from zinc sheet roofing started almost three orders of magnitude higher and remained more than twenty times above the values measured for the roofs affected only by atmospheric deposition. Forster noted the most critical effect of runoff pollution containing heavy metals is their high ecotoxicity in receiving waters. Mean runoff concentration values at his study sites exceeded by about two orders of magnitude local toxicity thresholds. Peak values exceeded thresholds by a factor of 1000 or more. Forster concluded by advocating abandoning the use of exposed metal surfaces on roofs and walls of buildings.

Good (1993) reported the results of one time sampling of runoff from a rusty galvanized metal roof, a weathered metal roof, a built-up roof of plywood covered with roofing paper and tar, a flat tar-covered roof which had been painted with a fibrous reflective aluminum paint, and a relatively new anodized aluminum material at a sawmill facility on the coast of Washington . The research was carried out following the discovery that stormwater samples from the site were acutely toxic and contained high concentrations of zinc. Differences in contributions of copper, lead, and zinc were noticed between each roof type. Built-up roofing contributed the highest concentrations of dissolved copper (128 μg/L) and total copper (166 μg/L), approximately 10 times higher than levels detected in runoff from the other roofs sampled. Runoff from the rusty galvanized metal roof contained the highest concentrations of dissolved lead (35 μg/L) and total lead (302 μg/L), dissolved zinc (11,900 μg/L) and total zinc (12,200 μg/L). High concentrations of zinc were noted in runoff from each type of roof sampled at the site. Dissolved metals concentrations and toxicity remained high in roof runoff samples collected three hours after the beginning of the storm event, indicating metals leaching continued throughout storm events. All roof runoff samples were found to be highly toxic to rainbow trout with the aluminum painted roof least toxic. Roof runoff sample concentrations exceeded the water quality criteria for copper, lead, and zinc in all samples, though the greatest exceedences were for zinc. Acid rain and the high ionic content of the coastal atmosphere were thought to have contributed to the rapid corrosion of the galvanized metal roofs and leaching of zinc. Interestingly, plastic rain gutters were also reported as a source of lead.

Thomas and Greene (1993) working in and near Armidale, Australia found differences in metal contaminate levels between urban and rural roofs associated with variations in atmospheric deposition and differences related to antecedent dry periods. He also found runoff water quality influenced by different roof types. Zinc concentrations were significantly higher in galvanized iron roof catchments, while pH, conductivity and turbidity levels were higher in concrete tile roof catchments.

Pitt, *et al*. (1995) found high concentrations of organic constituents in runoff from several types of paved source areas. Paved areas receive pollutant contributions from vehicle exhaust emissions, tire and brake wear, vehicle corrosion and leaks, carry-in and atmospheric deposition, which are then washed off to varying degrees in subsequent rains. However, differences noted between sampling sites indicate potential differences in contribution of organics from paving materials themselves. Polycyclic aromatic hydrocarbons (PAHs) in particular are of concern, because they are known to have potential for adverse effects to a large number of invertebrates, fishes, birds, and mammals (Kennish 1992). Chlorination of PAHs in water treatment plants have also been found to produce carcinogenic by-products (Kopfler, *et al*. 1977).

Exposed Wooden Material/Treated Wood

Typical treated woods include chromated-copper-arsenate (CCA), ammoniacal copper zinc arsenate (ACZA), pentachlorophenol (PCP), and creosote. The volume of treated wood produced in the United States in 1987 was as follows: CCA/ACZA – 11.9 million cubic meters, PCP – 1.4 million cubic meters, Creosote – 2.8 million cubic meters.

Both arsenic and chromium are heavy metals which have acute environmental health risks associated with them. Copper does not generally constitute a human health risk, however, low concentrations of copper, in certain ionic forms, are highly toxic to marine fauna and flora. The known toxicity of arsenic and chromium to humans has resulted in concern about the possible introduction into the environment of large amounts of these metals in treated wood products (Brooks 1993).

Pentachlorophenol is a highly chlorinated, synthetic preservative containing pentachlorophenol, 2,3,4,6-tetrachlorophenol, higher chlorophenols, dioxins and furans. Arsenault (1975) and Stranks (1976) reported the presence of pentachlorophenol around the base of, and in drainage ditches near treated utility poles. Stranks reported drainage ditch waters with 1.8 times the 96-h LC50 of chlorophenol for salmonids near PCP treated utility poles. In 1991, the EPA determined that the use of pentachlorophenol poses the risk of oncogenicity because of the presence of hexachlorodibenzo-p-dioxin and hexacholorobenzene, both of which have the potential to produce teratogenic/fetotoxic effects) (CALEPA 1996).

Creosote is a rather complex chemical that is comprised of more than 160 different distillates that occur in coal-tar, including aromatic hydrocarbons (such as naphthalene, anthracene, benzene, toluene, xylene, acenaphthene, phenanthrene, and fluorene), tar acids (such as phenols, cresols, xylenols, and naphthols), and tar bases (including pyridines, guinolines, and acridines) many of which are toxicants and carcinogens. The EPA determined that creosote has the potential for oncogenicity and mutagenicity (CALEPA 1996).

Preliminary Leaching Tests to Investigate Building Material Contributions to Stormwater Contamination

Pitt, *et al*. (1999) examined the leaching effects associated with different building materials that may affect runoff quality, as part of his studies on the construction of pilot-scale treatment units. This information is summarized in the following paragraphs as an indication of the potential benefits of using alternative building materials. An important consideration when constructing any treatability apparatus is potential contamination of the test solutions by materials used in the construction of the device. Therefore, before the pilot-scale Multi-Chambered Treatment Train (MCTT) was constructed, a series of tests were conducted to examine the leachability of different potential construction materials. Samples of the various materials were left to soak in de-ionized water for set periods of time, and then the water was analyzed for a broad list of constituents of interest.

Samples of each material were immersed for a period of 72 h in approximately 500 mL of laboratory grade 18 megohm water. A sample blank was also prepared. Analyses conducted on each of these samples, and the sample blank included toxicity screening, major ion, and toxicant analyses. Table 3-20 presents the contaminants that were found in the leaching water at the end of the test in high concentrations that may affect the test results. The most serious problems occur with plywood, including both treated and untreated wood. Attempting to seal the wood with Formica and caulking was partially successful, but toxicants were still leached. Covering of the Formica clad plywood with polyethylene plastic sheeting was finally used to eliminate any potential problem in the pilot-scale treatment constructed. Fiberglass screening material, especially before cleaning, also causes a potential problem with plasticizers and other organics. PVC and aluminum may be acceptable materials, if phthalate esters and aluminum contamination can be tolerated.

Table 3-20. Potential Sample Contamination from Materials that may be used in Treatability Test Apparatus

Material:	Contaminant observed:
untreated plywood	toxicity, chloride, sulfate, sodium, potassium, calcium, 2,4-dimethylphenol, benzylbutyl phthalate, bis(2-ethylhexyl) phthalate, phenol, N-nitro-so-di-n-propylamine, 4-chloro-3-methylphenol, 2,4-dinitrotoluene, 4-nitrophenol, alpha BHC, gamma BHC, 4,4'-DDE, endosulfan II, methoxychlor, and endrin ketone

treated plywood (CCA)	toxicity, chloride, sulfate, sodium, potassium, hexachloroethane, 2,4-dimethylphenol, bis(2-chloroethoxyl) methane, 2,4-dichlorophenol, benzylbutyl phthalate, bis(2-ethylhexyl) phthalate, phenol, 4-chloro-3-methylphenol, acenaphthene, 2,4-dinitrotoluene, 4-nitrophenol, alpha BHC, gamma BHC, beta BHC, 4,4'-DDE, 4,4'-DDD, endosulfan II, endosulfan sulfate, methoxychlor, endrin ketone, and copper (likely), chromium (likely), arsenic (likely)
treated plywood (CCA) and Formica	toxicity, chloride, sulfate, sodium, potassium, **bis(2-chloroethyl) ether***, diethylphthalate, phenanthrene, anthracene, benzylbutyl phthalate, bis(2-ethylhexyl) phthalate, **phenol***, N-nitro-so-di-n-propylamine, **4-chloro-3-methylphenol***, 4-nitrophenol, pentachlorophenol, alpha BHC, 4,4'-DDE, endosulfan II, methoxychlor, endrin ketone, and copper (likely), chromium (likely), arsenic (likely)
treated plywood (CCA), Formica and silica caulk	lowered pH, toxicity, **bis(2-chloroethyl) ether***, hexachlorocyclopentadiene, diethylphthalate, bis(2-ethylhexyl) phthalate, **phenol***, N-nitro-so-di-n-propylamine, **4-chloro-3-methylphenol***, alpha BHC, heptachlor epoxide, 4,4'-DDE, endosulfan II, and copper (likely), chromium (likely), arsenic (likely)
Formica and silica caulk	lowered pH, toxicity, 4-chloro-3-methylphenol, aldrin, and endosulfan 1
silica caulk	lowered pH, toxicity, and heptachlor epoxide
PVC pipe	N-nitrosodiphenylamine, and 2,4-dinitrotoluene
PVC pipe with cemented joint	**bis(2-ethylhexyl) phthalate***, acenaphthene, and endosulfan sulfate
plexiglass and plexiglass cement	naphthalene, benzylbutyl phthalate, and bis(2-ethylhexyl) phthalate, and endosulfan II
aluminum	toxicity, and aluminum (likely)
plastic aeration balls	2,6-dinitrotoluene
filter fabric material	acenaphthylene, diethylphthalate, benzylbutyl phthalate, bis(2-ethylhexyl) phthalate, and pentachlorophenol
sorbent pillows	diethylphthalate, and bis(2-ethylhexyl) phthalate
black plastic fittings	pentachlorophenol
reinforced PVC tubing	diethylphthalate, and benzylbutyl phthalate
fiberglass window screening	toxicity, dimethylphthalate, **diethylphthalate***, bis(2-ethylhexyl) phthalate, di-n-octyl phthalate, phenol, 4-nitrophenol, pentachlorophenol, and 4,4'-DDD
Delrin™	benzylbutyl phthalate
Teflon™	nothing
glass	zinc (likely)

* the observed concentrations in the leaching solution were very large compared to the other materials.

Metals exposed to rain water are of obvious concern, as indicated by the roof runoff data. Treated wood is of obvious concern and should be avoided in locations near directly connected paved areas. It is also likely that runoff from fresh asphalt pavement can produce toxic effects, while aged asphalt surfaces do not cause problems. In many cases, much reduced amounts of toxicants reach the drainage system if the sheetflow water from these materials is allowed to drain across landscaped areas, where most of the heavy metals and organic toxicants seemed to be tightly sorbed to soil particulates. Of course, these soil particulates can erode and contribute contaminated sediments to the stormwater, while others can adversely affect groundwater (Pitt, *et al*. 1996). Selection of alternative materials is preferred. Most plastics, or plastic-coated metals should be acceptable, along with many traditional building materials (glass, brick and concrete), but much additional work needs to be done in this area.

Street Dirt Accumulation

The washoff of street dirt and the effectiveness of street cleaning as a stormwater control practice are highly dependent on the available street dirt loading. Street dirt loadings are the result of deposition and removal rates, plus "permanent storage." The permanent storage component is a function of street texture and condition and is the quantity of street dust and dirt that cannot be removed naturally by rains or winds, or by street cleaning equipment. It is literally trapped in the texture, or cracks, of the street. The street dirt loading at any time is this initial permanent loading plus the accumulation amount corresponding to the exposure period, minus the re-suspended material removal by wind and traffic-induced turbulence. Removal of street dirt can occur naturally by winds and rain, or by human activity (e.g., by the turbulence of traffic or by street cleaning equipment). Very little removal occurs by any process when the street dirt loadings are small, but wind removal may be very large with larger loadings, especially for smooth streets (Pitt 1979).

It takes many and frequent samples to ascertain the accumulation characteristics of street dirt. The studies briefly described in this discussion typically involved collecting many hundreds of composite street dirt samples during the course of the one to three year projects from each study area. With each composite sample made up of about 10 to 35 subsamples, a great number of subsamples were used to obtain the data. Without high resolution (and effective) sampling, it is not possible to identify the variations in loadings and effects of rains and street cleaning. Figures 3-1 and 3-2 are examples of the measured street dirt loading as a function of time for both smooth and rough streets (Pitt (1979). These plots clearly show accumulation rates (and increases in particle size of the street dirt) as time between street cleaning lengthens.

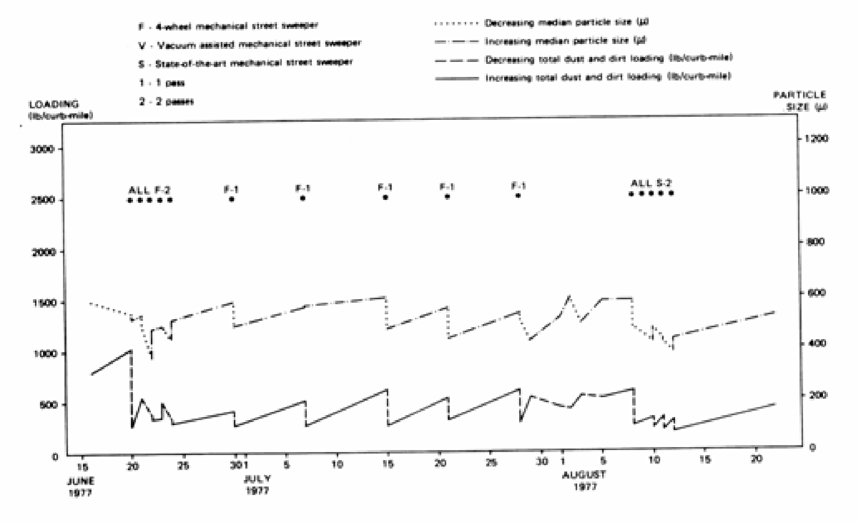


Figure 3-1. Street dirt accumulation and particle size changes on good asphalt streets in San Jose, CA (Pitt 1979).

Figure 3-2. Street dirt accumulation and particle size changes on rough asphalt streets in San Jose, CA (Pitt 1979).

Figure 3-3 shows very different street dirt loadings for two San Jose, CA residential study areas (Pitt 1979). The accumulation and deposition rates (and therefore the amounts lost to air) are quite similar, but the initial loading values (the permanent storage values) are very different. The loading differences were almost solely caused by the different street textures.

Figure 3-3. Deposition and accumulation of street dirt (Pitt 1979).

In early studies (APWA 1969; Sartor and Boyd 1972; and Shaheen 1975) it was assumed that the initial loading values were zero. However, the sampling procedures employed were very effective in removing all loose material from the streets, including the loadings that could not be removed by rains or street cleaning. Calculated accumulation rates for rough streets were therefore very large, as they were forced through the origin. The early, uncorrected, Sartor and Boyd accumulation rates that ignored the initial loading values were almost ten times the corrected values that had reasonable "initial loads."

A street dirt loading equation that can be used to represent street dirt loading (Pitt 1979) is:

$$Y = ax - bx^2 + c$$

where Y = street loading at time x,
a, b, and c are second order polynomial curve coefficients
ax represents the deposition loading
bx^2 represents the amount lost to the air
and c represents the initial storage loading

This curve should only be used over the range of observed accumulation periods. For long accumulation periods, this quadratic equation may predict decreasing loadings.

At very long accumulation periods relative to the rain frequency, the wind losses may approximate the deposition rate, resulting in very little loading increases. For Bellevue, Washington, with interevent rain periods averaging about 3 days, steady loadings were observed only after about 1 week (Pitt 1985). In Castro Valley, California, the rain interevent periods were much longer (ranging from about 20 to 100 days) and steady loadings were never observed (Pitt and Shawley 1982).

The accumulation period for each observed loading is needed before these accumulation curve coefficients can be calculated. It is the time since the streets were last cleaned, or the time since the last “significant” rain. A significant rain is usually considered to be about 10 mm, or larger, that occurs over a few hours. These rains normally remove at least 90% of the “available” street dirt washoff load, as will be described in the following discussion.

Street dirt loading data is difficult to fit to any curve because of many potential measurement and interpretation errors. The measurements are usually obtained with 25 percent allowable errors due to the large cost increases needed to collect enough sub-samples to significantly reduce these errors. As an example, it requires about five times as many street dirt subsamples for a 10 percent allowable error as compared to a 25 percent allowable error (Pitt 1979). Many areas also have frequent (every few days) rains. In most cases, frequent rains keep the street dirt loadings very close to the initial storage value, with little observed increase in dirt accumulation over time. If the loading value is not very well correlated with accumulation time, linear regression curve fitting may not be appropriate.

Other problems arise when attempting to use least squares regression techniques with data that contain different distributions of residuals (errors) over the range of predictor variables, or if the errors are not independent. This is especially true with street dirt accumulation data, as there are usually few street dirt loading observations associated with long accumulation periods. The shorter accumulation period observations usually have much smaller errors (due to smaller allowable data ranges) than the observations having longer accumulation periods (which are not as constrained). The short period loadings are relatively low, and the range of observed loadings at these low accumulation periods range from zero to values two or three times higher than the predicted loadings. The observed loadings at the longer accumulation periods are also constrained at zero for minimum values, but the range of possible values is much larger than for the lower loadings. The errors for these longer period observations can be greater because of the greater opportunity for other factors that are not included in the regression relationship to be prominent. These other factors include variable winds and moisture conditions. If the data is extensive, then it may be separated into seasonal groupings to reduce the variations of these other factors. Logarithmic transformations of the loading values can sometimes produce normally distributed residuals over the range of data that are necessary for least-squares regression analyses.

Early measurements of across-the-street dirt distributions made by Sartor and Boyd (1972) indicated that about 90 percent of the street dirt was within about 30 cm of the curb face (typically within the gutter area). These measurements, however, were made in areas of no parking (near fire hydrants because of the need for water for the sampling procedures that were used), and the traffic turbulence was capable of blowing most of the street dirt against the curb barrier (or over the curb onto adjacent sidewalks or landscaped areas) (Shaheen 1975). In later tests, Pitt (1979) and Pitt and Sutherland (1982) examined street dirt distributions across-the-street in many additional situations. They found distributions similar to Sartor and Boyd’s observations only on smooth streets, with moderate to heavy traffic, and with no on-street parking. In many cases, most of the street dirt was actually in the driving lanes, trapped by the texture of rough streets. If extensive on-street parking was common, much of the street dirt was found several feet out into the street, where much of the resuspended (in air) street dirt blew against the parked cars and settled to the pavement. Figure 3-4 shows across-the-street distributions of street dirt, both before and after street cleaning for a relatively busy roadway (having no parking) in Bellevue, WA (Pitt 1985). Only about 20% of the street dirt was near the curb before street cleaning, while 90% was within about 8 ft. After cleaning, the load was even more evenly distributed, as the street cleaner preferentially removed street dirt near the curb and blew some dirt out into the street.

Figure 3-4. Re-distribution of street dirt across the street during street cleaning (Pitt 1985).

Methodology for Street Dirt Accumulation Measurements

Pitt and McLean (1984) conducted street dirt accumulation studies as part of the Humber River study portion of TAWMS (Toronto Area Watershed Management Study). Detailed results were also presented by Pitt (1987). An industrial street with heavy traffic (Norseman) and a residential street with light traffic (Glen Roy) in Toronto were monitored about twice a week for three months. At the beginning of this period, intensive street cleaning (one pass per day for each of three consecutive days) was conducted to obtain reasonably clean streets. Street dirt loadings were then monitored every few days to measure the accumulation rates of street dirt. Street dirt sampling procedures developed by Pitt (1979) were used. Basically, industrial vacuums were used to clean many separate subsample strips across the roads which were then combined for analysis.

Street Surface Particulate Sampling Procedures

The street dirt sampling procedures described here were developed by Pitt (1979) and were extensively used during many of the EPA's Nationwide Urban Runoff Program (NURP) projects (EPA 1983) and other street cleaning performance studies and washoff studies (Pitt 1987). These procedures were developed to be much for flexible and more accurate indicators of street dirt loading conditions than previous sampling methods used during earlier studies (such as Sartor and Boyd 1972, for example).

Powerful dry vacuum sampling, as used in this sampling procedure, is capable of removing practically all of the particulates (>99%) from the street surface, compared to wet sampling. It can also remove most of the other major pollutants from the street surface (>80% for COD, phosphates and metals, for example). Wet sampling (used by Sartor and Boyd 1972), better removes some of these other constituents, but is restricted to single area sampling, requires long periods of time, requires water (and usually fire hydrants further restricting sample collection locations to areas having no parked cars), and basically is poorly representative of the variable conditions present. Dry sampling can be used in many locations throughout an area, is fast, and can also be used to isolate specific sampling areas (such as driving lanes, areas with intensive parking, and even airport runways and freeways, if special safety precautions are used). It is especially useful when coupled with appropriate experimental design tools to enable suitable numbers of subsamples to be collected representing subareas, and finally, the collected dry samples can be readily separated into different particle sizes for discrete analyses.

Equipment Description. A small half-ton trailer was used to carry the generator, two stainless steel industrial vacuum units, vacuum hose and wand, miscellaneous tools, and a fire extinguisher. This equipment can also be fitted in a pick-up truck, but much time is then lost with frequent loading and unloading equipment, especially considering the frequent sampling that is typically used for a study of this nature (sampling at least once a week, and sometimes twice a day before and after street cleaning or rains). A truck with a suitable hitch and signal light connections was used to pull the trailer. The truck also had warning lights, including a roof-top flasher unit. The truck operated with its headlights and warning lights on during the entire period of sample collection. The sampler and hose tender both wore orange, high-visibility vests. The trailer was equipped with a caution sign on its tailgate. In addition, both the truck and the street cleaner used to clean the test area were equipped with radios (CB radios were adequate), so that the sampling team could contact the street cleaner operator when necessary to verify location and schedule for specific test areas.

Experiments were conducted to determine the most appropriate vacuum and filter bag combination. Two-horsepower (hp) industrial vacuum cleaners with one secondary filter and a primary dacron filter bag were selected. The vacuum units were heavy duty and made of stainless steel to reduce contamination of the samples. Two separate 2-hp vacuums were used together by joining their intakes with a wye connector. This combination extended the useful length of the 1.5 in. vacuum hose to 35 ft. and increased the suction so that it was adequate to remove all particles of interest from the street surface. Unfortunately, two vacuums had to be cleaned to recover the samples after the sample collection. A wand and a "gobbler" attachment were also needed. The aluminum gobbler attached to the end of the wand and is triangular in shape and about 6 in. across. Since it was scrapped across the street during sample collection, it wore out periodically and needed replacement. The generator used to power the vacuum units was of sufficient power to handle the electrical current load drawn by the vacuum units, about 5000 watts for two 2-hp vacuums. Honda water-cooled generators are extremely quite and reliable for this purpose. Finally a secure, protected garage was used to store the trailer and equipment near the study areas when not in use.

Sampling Procedure. Because the street surfaces were more likely to be dry during daylight hours (necessary for good sample collection), collection did not begin before sunrise nor continue after sunset. During extremely dry periods, some sampling was conducted during dark hours, but that required additional personnel for traffic control. Two people were required for sampling at all times, one acting as the sampler, the other acting as the vacuum hose tender and traffic controller. This lessened individual responsibility and enabled both persons to be more aware of traffic conditions.

Before each day of sampling, the equipment was checked to make sure that the generator's oil and gasoline levels were adequate, and that vacuum hose, wand, and gobbler were in good condition. Dragging the vacuum hose across asphalt streets required periodic hose repairs (usually made using gray duct tape). A check was also made to ensure that the vacuum units were clean, the electrical cords were securely attached to the generator, and the trailer lights and warning lights were operable. The generator required about 3 to 5 minutes to warm up before the vacuum units were turned on one at a time (about 5 to 10 seconds apart to prevent excessive current loading on the generator). The amperage and voltage meters of the generator were also periodically checked. The generator and vacuums were left on during the complete subsamping period to lessen strain associated with multiple shutoffs and startups. Obviously, the sampling end of the vacuum hose was carefully secured between subsamples to prevent contamination.

Each subsample included all of the street surface material that would be removed during a severe rain (including loose materials and caked-on mud in the gutter and street areas). The location of the subsample strip was carefully selected to ensure that it had no unusual loading conditions (e.g., a subsample was not collected through the middle of a pile of leaves; rather, it was collected where the leaves were lying on the street in their normal distribution pattern). When possible, wet areas were avoided. If a sample was wet and the particles caked around the intake nozzle, the caked mud from the gobbler was carefully scraped into the vacuum hose while the vacuum units were running.

Subsamples were collected in a narrow strip about 6 in. wide (the width of the gobbler) from one side of the street to the other (curb to curb). In heavily traveled streets where traffic was a problem, some subsamples consisted of two separate one-half street strips (curb to crown). Traffic was not stopped for subsample collection; the operators waited for a suitable traffic break. On wide or busy roadways, a subsample was often collected from two strips several feet apart, halfway into the street. On busy roadways with no parking and good street surfaces, most particulates were found within a few feet of the curb, and a good subsample could be collected by vacuuming two adjacent strips from the curb as far into the traffic lanes as possible. Only a sufficient (and safe) break in traffic allowed a subsample to be collected halfway across the street.

Subsamples taken in areas of heavy parking were collected between vehicles along the curb, as necessary. The sampling line across the street did not have to be a continuous line if a parked car blocked the most obvious and easiest subsample strip. A subsample could be collected in shorter (but very close) strips, provided the combined length of the strip was representative of different distances from the curb. Again, in all instances, each subsample was representative of the overall curb-to-curb loading condition.

When sampling, the leading edge of the gobbler was slightly elevated above the street surface (0.125 in.) to permit an adequate air flow and to collect pebbles and large particles. The gobbler was lifted further to accept larger material as necessary. If necessary, leaves in the subsample strip were manually removed and placed in the sample storage container to prevent the hose from clogging. If a noticeable decrease in sampling efficiency was observed, the vacuum hoses were cleaned immediately by disconnecting the hose lengths, cleaning out the connectors (placing the debris into the sample storage container), and reversing the air flows in the hoses (blowing them out by connecting the hose to the vacuum exhaust and directing the dislodged debris into the vacuum inlet). If any mud was caked on the street surface in the subsample strip, the sampler loosened it by scraping a shoe along the subsample path (being certain that street construction material was not removed from the subsample path unless it was very loose). Scraping caked-on mud was done after an initial vacuum pass. After scraping was completed, the strip was revacuumed. A rough street surface was sampled most easily by pulling (not pushing) the wand and gobbler toward the curb. Smooth and busy streets were usually sampled with a pushing action, away from the curb.

An important aspect of the sample collection was the speed at which the gobbler was moved across the street. A very rapid movement significantly decreased the amount of material collected; too slow a movement required more time than was necessary. The correct movement rate depended on the roughness of the street and the amount of material on it. When sampling a street that had a heavy loading of particulates, or a rough surface, the wand was pulled at a velocity of less than 1 ft per second. In areas of lower loading and smoother streets, the wand was pushed at a velocity of 2 to 3 ft per second. The best indication of the correct collection speed was by examining how well the street was visually being cleaned in the sampling strip and by listening to the collected material rattle up the wand and through the vacuum hose. The objective was to remove everything that was lying on the street that could be removed by a significant rainstorm. It was quite common to leave a visually cleaner strip on the street where the subsample was collected, even on streets that appeared to be clean before sampling.

In all cases, the hose tender continuously watched traffic and alerted the sampler of potentially hazardous conditions. In addition, he played out the hose to the sampler as needed and kept the hose as straight as possible to prevent kinking. If a kink developed, sampling stopped until the hose tender straightened the hose. While working near the curb out of the traffic lane (typically an area of high loadings), the sampler visually monitored the performance of the vacuum sampler and periodically checked for vehicles. In the street, the sampler constantly watched traffic and monitored the collection process by listening to particles moving up the wand. A large break in traffic was required to collect dust and dirt from street cracks in the traffic lanes, because the sampler had to watch the gobbler to make sure that all of the loose material in the cracks was removed.

When moving from one subsample location to another, the hose, wand, and gobbler were securely placed in the trailer. All subsamples were composited in the vacuums for each study area. The hose was placed away from the generator’s hot muffler to prevent hose damage. The generator and vacuum units were left on and in the trailer during the entire subsample collection period. This helped dry damp samples and reduced the strain on the vacuum and generator motors.

The length of time it took to collect all of the subsamples in an area varied with the number of subsamples and the test area road texture and traffic conditions. The number of subsamples required in each area was determined using experimental design sample effort equations, with seasonal special sampling efforts to measure the variability of street dirt loadings in each area. The variabilities were measured using a single, small 1.5 HP industrial vacuum, with a short hose. The vacuum was emptied, the sample collected, and weighed (in the lab) after each individual sample so the variability in loadings could be directly measured. During the first phase of the San Jose study (Pitt 1979), the test areas required the following sampling effort in order to stay within a 25% allowable error goal:

Test Area	No. of Subsamples	Sampling Duration
Downtown - poor asphalt street surface	14	0.5 hr.
Downtown - good asphalt street surface	35	1 hr.
Keyes Street - oil and screens street surface	10	0.5-1 hr.
Keyes Street - good asphalt street surface	36	1 hr.
Tropicana - good asphalt street surface	16	0.5-1 hr.

The dirtiest streets required the least sampling effort because the coefficients of variation for loadings represented by the individual subsample strips was much smaller than for the cleaner streets. In the oil and screens test area, the sampling procedure was slightly different because of the relatively large amount of pea gravel (screens) that was removed from the street surface. The gobbler attachment was drawn across the street more slowly (at a rate of about 3 seconds per ft.). Each subsample was collected by a half pass (from the crown to the curb of the street) and therefore contained one-half of the normal sample. Two curb-to-curb passes were made for each Tropicana subsample because of the relatively low particulate loadings in this area, as several hundred grams of sample material were needed for the laboratory tests. In addition, an after street cleaning subsample was not collected from exactly the same location as the before street cleaning subsample (they were taken from the same general area, but at least a few feet apart).

A field-data record sheet kept for each sample contained:

- Subsample numbers
- Dates and time of the collection period
- Any unusual conditions or sampling techniques.

Subsample numbers were crossed off as each subsample was collected. After cleaning, subsample numbers were marked if the street cleaner operated next to the curb at that location. This differentiation enabled the effect of parked cars on street cleaning performance to be analyzed. In addition, photographs (and movies) were periodically made to document the methods and street loading conditions.

Sample Transfer. After all subsamples for a test area were collected, the hose and wye connections were cleaned by disconnecting the hose lengths, reversing them, and holding them in front of the vacuum intake. Leaves and rocks that may have become caught were carefully removed and placed in the vacuum can, the generator was then turned off. The vacuums were either emptied at the last station or at a more convenient location (especially in a sheltered location out of the wind and sun).

To empty the vacuums, the top motor units were removed and placed out of the way of traffic. The vacuum units were then disconnected from the trailer and lifted out. The secondary, coarse vacuum filters were removed from the vacuum can and were carefully brushed with a small stiff brush into a large funnel placed in the storage can. The primary dacron filter bags were kept in the vacuum can and shaken carefully to knock off most of the filtered material. The dust inside the can was allowed to settle for a few minutes, then the primary filter was removed and brushed carefully into the sample can with the brush. Any dirt from the top part of the bag where it was bent over the top of the vacuum was also carefully removed and placed into the sample can. Respirators and eye protection is necessary to minimize exposure to the fine dust.

After the filters were removed and cleaned, one person picked up the vacuum can and poured it into the large funnel on top of the sample can, while the other person carefully brushed the inside of the vacuum can with a soft 3- to 4-in. paint brush to remove the collected sample. In order to prevent excessive dust losses, the emptying and brushing was done in areas protected from the wind. To prevent inhaling the sample dust, both the sampler and the hose tender wore mouth and nose dust filters while removing the samples from the vacuums.

To reassemble the vacuum cans, the primary dacron filter bag was inserted into the top of the vacuum can with the filters's elastic edge bent over the top of the can. The secondary, coarse filter was placed into the can and assembled on the trailer. The motor heads were then carefully replaced on the vacuum cans, making sure that the filters were on correctly and the excessive electrical cord was wrapped around the handles of the vacuum units. The vacuum hoses and wand were attached so that the unit was ready for the next sample collection.

The sample storage cans were labeled with the date, the test area's name, and an indication of whether the sample was taken before or after the street cleaning test or if it was an accumulation (or other type) of sample. Finally, the lids of the sample cans were taped shut and transported to the laboratory for logging-in, storage, and analysis.

Summary of Observed Accumulation Rates

Table 3-21 summarizes many accumulation rate measurements obtained from throughout North America. In the earliest studies (APWA 1969; Sartor and Boyd 1972; and Shaheen 1975), the initial street dirt loading values after a major rain or street cleaning were assumed to be zero. Calculated accumulation rates for rough streets were therefore very large. Later tests measured the initial loading values close to the end of major rains and street cleaning and found that they could be relatively high, depending on the street texture. When these starting loadings were considered for the earlier measurements, the re-calculated accumulation rates were much lower. The early, uncorrected, Sartor and Boyd accumulation rates that ignored the initial loading values were almost ten times the corrected values shown on this table. Unfortunately, most urban stormwater models used these very high early accumulation rates as default values.

Table 3-21. Street dirt loadings and deposition rates.

	Initial Loading Value (grams/curb-meter)	Daily Deposition Rate (grams/curb-meter-day)	Maximum Observed Loading (grams/curb-meter)	Days to Observed Maximum Loading	Reference
Smooth and Intermediate Textured Streets					
Reno/Sparks, NV – good condition	80	1	85	5	Pitt and Sutherland 1982
Reno/Sparks, NV – good with smooth gutters (windy)	250	7	400	30	Pitt and Sutherland 1982
San Jose, CA – good condition	35	4	>140	>50	Pitt 1979
U.S. nationwide – residential streets, good condition	110	6	140	5	Sartor and Boyd 1972 (corrected)
U.S. nationwide – commercial street, good condition	85	4	140	5	Sartor and Boyd 1972 (corrected)
Reno/Sparks, NV – moderate to poor condition	200	2	200	5	Pitt and Sutherland 1982
Reno/Sparks, NV – new residential area (construction)	710	17	910	15	Pitt and Sutherland 1982
Reno/Sparks, NV – poor condition, with lipped gutters	370	15	630	35	Pitt and Sutherland 1982
San Jose, CA – fair to poor condition	80	4	230	70	Pitt 1979
Castro Valley, CA – moderate condition	85	10	290	70	Pitt and Shawley 1982
Ottawa, Ontario – moderate condition	40	20	Na	Na	Pitt 1983
Toronto, Ontario – moderate condition, residential	40	32	100	>10	Pit and McLean 1986
Toronto, Ontario – moderate condition, industrial	60	40	351	>10	Pit and McLean 1986
Believue, WA – dry period, moderate condition	140	6	>230	20	Pitt 1985
Believue, WA – heavy traffic	60	1	110	30	Pitt 1985
Believue, WA – other residential sites	70	3	140	30	Pitt 1985
Average:	150	9	>270	>25	
Range:	35 – 710	1 – 40	85 – 910	5 – 70	
Rough and Very Rough Textured Streets					
San Jose, CA – oil and screens overlay	510	6	>710	>50	Pitt 1979
Ottawa, Ontario – very rough	310	20	Na	Na	Pitt 1983
Reno/Sparks, NV	630	10	860	35	Pitt and Sutherland 1982
Reno/Sparks, NV – windy	540	34	>1,400	>40	Pitt and Sutherland 1982
San Jose, CA – poor condition	220	6	430	30	Pitt 1979
Ottawa, Ontario – rough	200	20	Na	Na	Pitt 1983
U.S. nationwide – industrial streets (poor condition)	190	10	370	10	Sartor and Boyd 1972 (corrected)
Average:	370	15	>750	>30	
Range:	190 - 630	6 - 34	370 - >1,400	10 - >50	

The most important factors affecting the initial loading and maximum loading values shown on Table 3-21 were found to be street texture and street condition. When data from many locations are studied, it is apparent that smooth streets have substantially less loadings at any accumulation period compared to rough streets for the same land use. Very long accumulation periods relative to the rain frequency result in high street dirt loadings. During these conditions, the wind losses of street dirt (as fugitive dust) may approximate the deposition rate, resulting in relatively constant street dirt loadings. At Bellevue, WA, typical interevent rain periods average about three days. Relatively constant street dirt loadings were observed in Bellevue because the frequent rains kept the loadings low and very close to the initial storage value, with little observed increase in dirt accumulation over time (Pitt 1985). In Castro Valley, CA, the rain interevent periods were much longer (ranging from about 20 to 100 days) and steady loadings were only observed after about 30 days when the loadings became very high and fugitive dust losses caused by the winds and traffic turbulence moderated the loadings (Pitt and Shawley 1982).

Pitt and McLean (1986) studied street dirt accumulation rates and the effects of street cleaning in Toronto. An industrial street with heavy traffic and a residential street with light traffic were monitored about twice a week for three months. At the beginning of this period, intensive street cleaning (one pass per day for each of three consecutive days) was conducted to obtain reasonably clean streets. Street dirt loadings were then monitored every few days to measure the accumulation rates of street dirt. The street dirt particulate loadings were quite high before the initial intensive street cleaning period and were reduced to their lowest observed levels immediately after the last street cleaning. After street cleaning, the loadings on the industrial street increased much faster than for the residential street. Right after intensive cleaning, the street dirt particle sizes were also similar for the two land uses. However, the loadings of larger particles on the industrial street increased at a much faster rate than on the residential street, indicating more erosion or tracking materials being deposited onto the industrial street. The residential street dirt measurements did not indicate that any material was lost to the atmosphere as fugitive dust, probably because of the low street dirt accumulation rate and the short periods of time between rains. The street dirt loadings never had the opportunity to reach the high loading values needed before they could be blown from the streets by winds or by traffic-induced turbulence. The industrial street, in contrast, had a much greater street dirt accumulation rate and reached the critical loading values needed for fugitive losses in the relatively short periods between the rains.

Washoff of Street Dirt

Background

The degradation of the road surface and traffic related discharges are responsible for most of the particulate discharges in urban runoff. The smallest particulates from urban areas are usually discharged during the early parts of storms, but small particulates from impervious surfaces may also be discharged during later parts of storms. Shaheen (1975) found that road surface particulates and polluted area soils (affected by traffic related pollutants) contribute most of the urban runoff particulate pollutants. Many urban runoff models assume that "all" of the pollutants and runoff flows in urban areas originate from directly connected impervious areas, ignoring contributions from pervious areas. The correct interpretation of particulate washoff from impervious surfaces is therefore critical to understanding urban runoff quality. This discussion summarizes some of the procedures that are commonly used to estimate particulate washoff from impervious surfaces, presents the results of washoff tests, and describes a revised street dirt washoff model.

Washoff of particulates from impervious surfaces is dependent on the available supply of particulates and the capacity of the runoff to transport the loosened material. The accumulation of the material is dependent on many site specific land use and geographic features, plus the intended or unintended losses of materials.

Brief descriptions follow of two methods (the Yalin equation and the Sartor and Boyd equation) currently used in most urban runoff studies for estimating particulate washoff from impervious surfaces. They can be used to obtain satisfactory estimates of particulate washoff, if their limitations are recognized and if rough estimates are all that are required. Unfortunately, they are often used in situations beyond their limits (such as for small rains, unusual street dirt loadings, or rough pavement textures). Certain washoff equation parameters have also been misunderstood (such as confusing total street dirt load with "available" street dirt load). The use of these washoff equations in large and well documented urban runoff computer models also implies more confidence in their accuracy than may be warranted.

A field study is briefly summarized that found significant washoff differences for various particle sizes. These observed washoff quantities are compared to the values obtained with these two washoff models, but the observed washoff quantities are shown to be much less than predicted with the washoff equations. These data observations and the existing washoff models' inabilities to accurately predict washoff lead to the series of washoff tests conducted by Pitt (1987) and the development of washoff models sensitive to important environmental conditions.

Yalin Equation

Novotny and Chesters (1981) presented the Yalin equation as the best candidate from the many models presented in the literature to describe sediment washoff and transport in urban areas. The Yalin equation relates the sediment carrying capacity to runoff flow rate (Yalin 1963). Yalin assumed that sediment motion begins when the lift force of flow exceeds a critical lift force. Once a particle is lifted from the bed, the drag force of the flow moves it downstream until the weight of the particle forces it back to the bed. The Yalin equation is used to predict particle transport, for specific particle sizes, on a weight per unit flow width basis. It is used for fully turbulent channel flow conditions, typical of shallow overland flow in urban areas. The receding limb (tail) of a hydrograph may have laminar flow conditions, and the suspended sediment carried in the previously turbulent flows would settle out. The predicted constant Yalin sediment load would therefore only occur during periods of rain; and the sediment load would decrease, due to sedimentation, after the rain stops. The equation is presented in the following form:

$$p = 0.635\, s\, [1 - (1 / a^*s) \ln (1 + a^*s)]$$

where p = particle transport, grams/meter-second

a and s are calculated, based on particle density, particle diameter, and shear velocity.

To use the equation, the particle shear velocity (v_*, m/sec) must be calculated:

$$v_* = (gHS)^{0.5}$$

where g = acceleration of gravity = 9.81 m/sec^2

H = flow depth, meters

S = energy gradient slope, m/m

The particle Reynolds number (X) must also be known:

$X = v_* D / u$

where D = particle diameter, meters

u = kinematic viscosity of fluid = 10^{-6} m^2/sec for water

The critical particle bedload tractive force (Y_{cr}), the tractive force at which the particle begins to move, can be obtained from a Shield's diagram (Figure 3-5). Shen (1981) warned that Shield's diagram cannot be used alone to predict "self-cleaning" velocities, it gives only a lower limit below which deposition will occur. It defines the boundary between bed movement and stationary bed conditions. The diagram does not consider the particulate supply rate in relationship to the particulate transport rate. Reduced particulate transport occurs if the sediment supply rate is less than the transport rate.

The actual tractive force is also calculated:

$Y = v_*^2 / (p_s - 1) g^* D$

where p_s = specific density of particle, g/cm3

Figure 3-5. Shield's diagram for particle tractive force (from Novotny and Chesters 1981).

The Yalin coefficients can be calculated knowing Y, Y_{cr}, and p_s:

$s = Y / Y_{cr}$

and $a = 2.45\ p_s - 0.4\ (Y_{cr})^{0.5}$

The Yalin equation by itself is therefore not sensitive to particulate supply; it only predicts the carrying capacity of flowing waters. Models must be used that account for total particulate discharge and "stop" transport when the particulate supply is exhausted.

Besides the particulate supply rate, the Yalin equation is also very sensitive to local flow parameters (specifically gutter flow depth); a hydraulic model that can accurately predict sheetflow across impervious surfaces and gutter flow is needed. Sutherland and McCuen (1978) statistically analyzed a modified form of the Yalin equation, in conjunction with a hydraulic model (the Basic Inlet Hydrograph Model - BIHM), for different gutter flow conditions. Except for the largest particle sizes, the effect of rain intensity on particle washoff was negligible. A set of equations, shown on Table 3-22, were developed relating the percentage washoff (TS_i) of each of six particle sizes to gutter slope, impervious area, initial solids loading, and the gutter length before the storm drain inlet. These washoff percentages assume a one-hour uniform rain of 13 mm. These washoff percentages can then be modified for other total rains, by the K_j factors given in Table 3-23:

$TS_j = K_j \, TS_i$

where TS_j = percent total solids removal (for a specific size range)

TS_i = percent total solids removal for the standard 13 mm rain (for a specific size range)

K_j = factor relating the standard rain to the actual rain

Table 3-22. Washoff Equations for Different Particle Sizes (Sutherland and McCuen 1978)

Table 3-23. Kj Values used in Yalin Sediment Transport Model (Sutherland and McCuen 1978)

The Yalin equation is based on classical sediment transport equations, and requires some assumptions concerning the micro-scale aspects of gutter flows and street dirt distributions. The Yalin equation, as typically used in urban runoff models, assumes that all particles lie within the gutter, and no significant washoff occurs by sheetflows traveling across the street towards the gutter. The early measurements of across-the-street dirt distributions made by Sartor and Boyd (1972) indicated that about 90 percent of the street dirt was within about 30 cm of the curb face (typically within the gutter area). These measurements, however, were made in areas of no parking (near fire hydrants because of the need for water for the sampling procedures that were used), and the traffic turbulence was capable of blowing most of the street dirt against the curb barrier (or over the curb onto adjacent sidewalks or landscaped areas). In later tests, Pitt (1979) examined street dirt distributions across-the-street in many situations. He found distributions similar to Sartor and Boyd's observations only on smooth streets, with moderate to heavy traffic, and with no on-street parking. In many cases, most of the street dirt was actually in the driving lanes, trapped by the texture of rough streets. If on-street parking was common, much of the street dirt was found on the outside edge of the parking lanes, where the resuspended (in air) street dirt blew against the parked cars and settled to the pavement. Some later modeling efforts (most notably later versions of the MUNP and PTM models, Sutherland personal communication) adjusted the total street loading to estimate the loading present only in the gutter. Washoff of in-street particulates was still not considered.

Another process that may result in washoff less than predicted by Yalin is bed armoring (Sutherland, *et al.* 1982?). As the smaller particulates are removed, the surface is covered by predominantly larger particulates which are not effectively washed off by the rain. Eventually, these larger particulates hinder the washoff of the trapped, under-lying, smaller particulates. Debris on the street, especially leaves, can also effectively armor the particulates, reducing the washoff of particulates to very low levels (Singer and Blackard 1978).

Sartor and Boyd Washoff Equation

Observations of particulate washoff during controlled tests may result in empirical washoff models that are not as limited as incomplete theoretical models. Washoff experiments using actual streets and natural street dirt and debris are affected by street dirt distributions and armoring. Their disadvantage is the assumption of transferability. If the washoff experiments are conducted for many situations then it may be possible to use the resultant model for other situations.

The earliest controlled street dirt washoff experiments were conducted by Sartor and Boyd (1972) during the summer of 1970 in Bakersfield, California. Their data are used in many urban runoff models (including SWMM, Huber and Heaney 1981; STORM, COE 1975; and HSPF, Donigian and Crawford 1976) to estimate the percentage of the available particulates on the streets that would wash off during rains of different magnitudes. They used a rain simulator having many nozzles and a drop height of 1-1/2 to 2 meters in street test areas of about 5 by 10 meters. Tests were conducted on concrete, new asphalt, and old asphalt, using simulated rain intensities of about 5 and 20 mm/hr. They collected and analyzed runoff samples every 15 minutes for about two hours for each test. Figure 3-6 shows two plots of their data, showing the asymptotic shape of the accumulative washoff curves for several particle sizes. Sartor and Boyd fitted their data to an exponential curve, assuming that the rate of particle removal of a given size is proportional to the street dirt loading and the constant rain intensity:

$$dN/dt = k\ r\ N$$

where dN/dt = the change in street dirt loading per unit time

k = proportionality constant

r = rain intensity (in/hr)

N = street dirt loading (lb/curb-mile)

This equation, upon integration, becomes:

$$N = N_o\ e^{-krt}$$

where N = residual street dirt load (after the rain)

N_o = initial street dirt load

t = rain duration

Figure 3-6. Street dirt washoff during high intensity rain tests (Sartor and Boyd 1973).

Street dirt washoff is therefore equal to No minus N. The variable combination rt, or rain intensity times rain duration, is equal to total rain volume (R). This equation further reduces to:

$$N = N_o\, e^{-kR}$$

Therefore, this equation is only sensitive to total rain, and not rain intensity.

These figures also did not show the total street dirt loading that was present during the tests and modelers have assumed that the asymptotic maximum shown was the total “before-rain” loading. However, the total street dirt loadings were several times greater than the maximum washoff amount observed.

Because of decreasing particulate supplies, the exponential washoff curve predicts decreasing concentrations of particulates with time since the start of a constant rain (Alley 1980 and 1981).

The proportionality constant, k, was found by Sartor and Boyd to be slightly dependent on street texture and condition, but was independent of rain intensity and particle size. The value of this constant is usually taken as 0.18/mm, assuming that 90 percent of the particulates will be washed from a paved surface in 1 hour during a 13 mm/hour rain. However, Alley (1981) fitted this model to watershed outfall runoff data and found that the constant varied for different storms and pollutants, for a single study area. Novotny (undated) examined “before” and “after” rain event street particulate loading data using the Milwaukee NURP data and found almost a three-fold difference between the constant value for fine (<45 microns) and medium sized particles (100 to 250 microns); 0.026/mm for the fine particles and 0.01/mm for the medium sized particles, both much less than the “accepted” value. Jewell, *et al*. (1980) also found large variations in outfall “fitted” constant values for different rains compared to the typical default value. Either the assumption of the high removal of particulates during the 13 mm/hr storm was incorrect or/and the equation cannot be fitted to outfall data (which assumes that all the particulates are originating from homogeneous paved surfaces during all storm conditions).

This washoff equation has been used in many urban runoff models (including SWMM, STORM, and HSPF), but the N_o factor has been frequently misinterpreted. It has been assumed to be the total initial street loading, when in fact it is only the portion of the total street load available for washoff (the maximum asymptotic washoff load observed during the washoff tests). STORM and SWMM use an availability factor (A) for particulate residue as a calibration procedure in order to reduce the washoff quantity for different rain intensities (Novotny and Chesters 1981):

$$A = 0.057 + 0.04\,(r^{1.1})$$

where r is the rain intensity (mm/hr), and A must be less than 1.0. This regression equation is used to adjust the relative importance of the particulate residue contributions from pervious and impervious source areas. This availability factor is equal to 1.0 for all rain intensities greater than about 18 mm/hr. For rains of 1 mm/hr, this availability factor reduces to about 0.10. HSPF does not use an availability factor in an attempt to be “more universally applicable” (Donigian and Crawford 1976). Instead, calibration of observed with predicted outfall yields are used to “adjust” the accumulation and washoff rates directly in HSPF. The availability factor in SWMM does not really have a significant effect on the variation of the predicted runoff load. However, it does affect the relationship between the runoff volume and the particulate washoff (and therefore concentration).

Jewell, *et al*. (1980) stressed the need to have local calibration data before using the exponential washoff equation, as the default values can be very misleading. The exponential washoff equation for impervious areas is justified, but washoff coefficients for each pollutant would improve its accuracy.

Street Dirt Washoff Observations and Comparisons with the Yalin, and Sartor and Boyd Washoff Equations

Particle dislodgement and transport characteristics at impervious areas can be directly measured using relatively easy washoff tests. These tests are used to supplement dry street dirt sampling at impervious source areas. Street dirt sampling, or other pavement dirt sampling, is misleading because little of the sampled dirt actually washes off during rains.

The Bellevue, Washington, urban runoff project (Pitt 1985) included about 50 pairs of street dirt loading observations close to the beginnings and ends of rains. These before and after loading values were compared to determine significant differences in loadings that may have been caused by the rains. The observations were affected by rains falling directly on the streets, along with flows and particulates originating from non-street areas. The net loading differences were therefore affected by street dirt washoff (by direct rains on the street surfaces and by gutter flows augmented by “upstream” area runoff) and by erosion products that originated from non-street areas that may have settled out in the gutters. When all the data were considered together, the net loading difference was about 10 to 13 grams/curb-m removed. This amounted to a street dirt load reduction of about 15 percent, which was much less than predicted using the previously described washoff models.

Very large reductions in street dirt loadings for the small particles were observed during rains in Bellevue, but the largest particles actually increased in loadings (due to settled erosion materials), as shown in Figure 3-7. The particles were not source limited, but armor shielding may have been important. Most of the weight of solid material in the runoff was in the fine particle sizes (<63 µm). Very few washoff particles greater than 1000 µm were found, in fact, loadings increased for the largest sizes. Urban runoff outfall particle size analyses in Bellevue (Pitt 1985) resulted in a median particle size of about 50 µm. Similar results were obtained in the Milwaukee NURP study (Bannerman, *et al*. 1983).

Particulate residue washoff predictions for Bellevue conditions were made using the Sutherland and McCuen modification of the Yalin equation, and the Sartor and Boyd equation. Three particle size groups (<63, 250-500, and 2000-6350 µm), and three rains, having depths of 5, 10, and 20 mm and 3-hour durations, were considered. The gutter lengths for the Bellevue test areas averaged about 80 m, with gutter slopes of about 4.5 percent. Typical total initial street dirt loadings for the three particle sizes were: 9 g/curb-meter for <63 µm, 18 g/curb-meter for 250-500 µm, and 9 g/curb-meter for 2000-6350 µm. The actual Bellevue net loading removals during the storms was about 45 percent for the smallest particle size group, 17 percent for the middle particle size group, and -6 percent (6 percent loading increase) for the largest particle size group. The predicted removals were 90 to 100 percent using the Sutherland and McCuen method, 61 to 98 percent using the Sartor and Boyd equation, and 8 to 37 percent using the availability factor with the Sartor and Boyd equation. The ranges given reflect the different rain volumes and intensities only. There were no large predicted differences in removal percentages as a function of particle size. The availability factor with the Sartor and Boyd equation resulted in the closest predicted values, but the great differences in washoff as a function of particle size was not predicted.

Figure 3-7. Observed washoff of street dirt during tests in Bellevue, WA (Pitt 1985).

The rain energy needed to remove larger particles is much greater than for small particles. Therefore, rains are much more effective in removing fine particles than large particles. In contrast, mechanical street cleaning equipment preferentially remove the larger particles compared to the small particles. Vacuum street cleaning equipment should be able to remove the finer particles better than the larger particles, but most vacuum street cleaners cannot remove the fine particles effectively under typically moist conditions and in the presence of larger particles that cover most of the finer street dirt. Therefore, particles of different sizes "behave" quite differently on streets. Typical street dirt total solids loadings show a "saw-tooth" pattern with time between street cleaning or rain washoff events. The patterns for the separate particle sizes are considerably different than the pattern for total residue. Typical mechanical street cleaners remove much (about 70 percent) of the coarse particles in the path of the street cleaner, but they remove very little of the finer particles (Sartor and Boyd 1972; Pitt 1979). Rains, however, remove very little of the large particles, but can remove large amounts (about 50 percent) of the fine particles (Bannerman, *et al*. 1983; Pitt 1985; Pitt 1987). The intermediate particle sizes show reduced removals by both street cleaners and rain.

The Bellevue street dirt washoff observations included effects of additional runoff volume and particulates originating from non-street areas. The additional flows should have produced more gutter particulate washoff, but upland erosion materials may also have settled in the gutters (as noted for the large particles). However, across-the-street dirt loading measurements indicated that much of the street dirt was in the street lanes, not in the gutters, before and after rains. This dirt distribution reduces the importance of these extra flows and particulates from upland areas. The increased loadings of the largest particles after rains were obviously caused by upland erosion, but the magnitude of the settled amounts was quite small compared to the total street dirt loadings.

Small-Scale Washoff Tests

Street dirt has a wide range of particle sizes and the chemical quality varied greatly for the different particle sizes. It is therefore important to mostly focus on the fraction that will be removed during rains. There is much confusion if the easily measured street dirt loadings are assumed to be totally available for washoff. Washoff tests can therefore be used to estimate the fraction of the total loading measured on the street that can be removed during rains.

In order to clarify street dirt washoff, Pitt (1987) conducted numerous controlled washoff tests on city streets in Toronto. These tests were arranged as an overlapping series of 2^3 factorial tests, and were analyzed using standard factorial test procedures described by Box, *et al.* (1978). The experimental factors examined included: rain intensity, street texture, and street dirt loading. The differences between available and total street dirt loads were also related to the experimental factors. The samples were analyzed for total solids (total residue), dissolved solids (filterable residue: <0.45 µm), and SS (particulate residue: >0.45 µm). Runoff samples were also filtered through 0.45 µm filters and the filters were microscopically analyzed (using low power polarized light microscopes to differentiate between inorganic and organic debris) to determine particulate size distributions from about 1 to 500 µm. The runoff flow quantities were also carefully monitored to determine the magnitude of initial and total rain water losses on impervious surfaces.

Table 3-24 presents the site data along with the basic rain and runoff observations obtained during these tests. All tests were conducted for about two hours, with total rain volumes ranging from about 5 to 25 mm. The test code explanations follow:

Test code	Rain intensity	Street dirt loading	Street texture
HCR	High	Clean	Rough
HDR	High	Dirty	Rough
LCR	Light	Clean	Rough
LDR	Light	Dirty	Rough
HDS	High	Clean	Smooth
HCS	High	Dirty	Smooth
LDS	Light	Clean	Smooth
LCS	Light	Dirty	Smooth

Table 3-24 shows the specific experimental levels that each variable was held to during each test. Unfortunately, the streets during the LDS test were not as dirty as anticipated and was actually a replicate with the LCS tests. The experimental analyses were modified to indicate these unanticipated duplicate observations.

Table 3-24. Experimental Levels for each Test Factor

	Rain intensity	**Street dirt loading**	**Street texture**
Expected to enhance percentage washoff:	High (11.0 to 12.2 mm/hr)	Dirty (10.5 to 12.6 g/m^2)	Smooth (0.3 to 0.4 mm detention storage)
Expected to retard percentage washoff:	Low (2.9 to 3.2 mm/hr)	Clean (1.7 to 2.6 g/m^2)	Rough (1.1 mm detention storage)

A simple artificial rain simulator was constructed using 12 lengths of “soaker” hose, suspended on a wooden framework about one meter above the road surface. “Rain” was applied by connecting the hoses to a manifold, having individual valves to adjust constant rain intensities for the different areas. The manifold was in turn connected to a fire hydrant. The flow rate needed for each test was calculated based on the desired rain intensity and the area covered. The flow rates were carefully monitored by using a series of ball flow gauges before the manifold. The distributions of the test rains over the study areas were also monitored by placing about 20 small graduated cylinders over the area during the rains. In order to keep the drop sizes representative of sizes found during natural rains, the surface tension of the water drops hanging on the plastic soaker hoses was reduced by applying a light coating of Teflon spray to the hoses.

It was difficult to obtain even distributions of rain during the light rain tests in Toronto using the manifold, so a single hose was used that was manually moved back and forth over the test area during the smaller rain tests (three people took 30-minute shifts). To keep evaporation reasonable for the rain conditions, the test sites were also shaded during sunny days. Blank water samples were also obtained from the manifold for background residue analyses. The filterable residue of the “rain” water (about 185 mg/L) could cause substantial errors when calculating total solids washoff.

The areas studied were about 3 by 7 meters each. The street side edges of the test areas were edged with plywood, about 30 cm in height and imbedded in thick caulking, to direct the runoff towards the curbs with minimal leakage. All runoff was pumped continuously from downstream sumps (made of caulking and plastic sand bags) to graduated 1000 L Nalgene containers. The washoff samples were obtained from the pumped water going to the containers every 5 to 10 minutes at the beginning of the tests, and every 30 minutes near the end of the test. Final complete rinses of the test areas were also conducted (and sampled) at the tests’ conclusions to determine total loadings of the monitored constituents.

The samples were analyzed for total residue, filtrate residue, and particulate residue. Runoff samples were also filtered through 0.4 micron filters and microscopically analyzed (using low power polarized light microscopes to differentiate between inorganic and organic debris) to determine particulate residue size distributions from about 1 to 500 microns. The runoff flow quantities were also carefully monitored to determine the magnitude of initial and total rain water losses on impervious surfaces.

These tests are different from the important early Sartor and Boyd (1972) washoff experiments in the following ways:

- They were organized in overlapping factorial experimental designs to identify the most important main factors and interactions.
- Particle sizes were measured down to about one micron (in addition to particulate residue and filterable residue measurements).
- The precipitation intensities were lower in order to better represent actual rain conditions of the upper midwest.
- Observations were made with more resolution at the beginning of the tests.
- Washoff flow rates were frequently measured.
- Emphasis was placed on total street loading, not just total available loading.
- Bacteria population measurements were also periodically obtained.

Figures 3-8 through 3-10 are plots of total solids, suspended solids, and filterable solids concentrations during these tests. The total solids concentrations varied from about 25 to 3000 mg/L, with an obvious decrease in concentrations with increasing rain depths during these constant rain intensity tests. No concentrations greater than 500 mg/L occurred after about two mm of rain. All concentrations after about 10 mm of rain were less than 100 mg/L. Total solids concentrations were independent of the test conditions. A wide range in runoff concentrations was also observed for SS, with concentrations ranging from about 1 to 3000 mg/L. Again, a decreasing trend of concentrations was seen with increasing rain depths, but the data scatter was larger because of the experimental factors. The dissolved solids (<0.45 μm) concentrations ranged from about 20 to 900 mg/L, comprising a surprisingly large percentage of the total solids loadings. For small rain depths, dissolved solids comprised up to 90 percent of the total solids. After 10 mm of rain depth, the filterable residue concentrations were all less than about 50 mg/L.

Figure 3-8. Total solids concentration decreases with rain depth increases during constant rain intensity washoff tests in Toronto (Pitt 1987).

Figure 3-9. Suspended solids concentration decreases with rain depth increases during constant rain intensity washoff tests in Toronto (Pitt 1987).

Figure 3-10. Dissolved solids concentration decreases during street dirt washoff tests in Toronto (Pitt 1987).

Manual particle size analyses were also conducted on the suspended solids washoff samples, using a microscope with a calibrated recticle. Figures 3-11 through 3-13 are examples of particle size distributions for three tests. These plots show the percentage of the particles that were less than various sizes, by measured particle volume (assumed to be similar to weight). The plots also indicate median particle sizes of about 10 to 50 µm, depending on when the sample was obtained during the washoff tests. All of the distributions showed surprisingly similar trends of particle sizes with elapsed rain depth. The median size for the sample obtained at about one mm of rain was much greater than for the samples taken after more rain. The median particle sizes of material remaining on the streets after the washoff tests were also much larger than for most of the runoff samples, but were quite close to the initial samples' median particle sizes. The washoff water at the very beginning of the test rains, therefore, contained many more larger particles than during later portions of the rains. Also, a substantial amount of larger particles remained on the streets after the test rains. Most street runoff waters during test rains in the 5 to 15 mm depth category had median suspended solids particle sizes of about 10 to 50 µm. However, dissolved solids (less than 0.45 µm) made up most of the total solids washoff for elapsed rain depths greater than about five mm.

These particle size distributions indicate that the smaller particles were much more important than indicated during previous tests. As an example, the Sartor and Boyd (1972) washoff tests (rain intensities of 50 mm/h for two hour durations) found median particle sizes of about 150 µm which were typically three to five times larger than were found during these lower-intensity tests. They also did not find any significant particle size distribution differences for different rain depths (or rain duration), in contrast to the Toronto tests, which were conducted at more likely rain intensities (3 to 12 mm/hr for two hours).

Figure 3-11. Particle size distributions during high rain intensity, clean and smooth street (HCS) tests (Pitt 1987).

Figure 3-12. Particle size distributions during high rain intensity, dirty and smooth street (HDS) tests (Pitt 1987).

Figure 3-13. Particle size distributions during light rain intensity, clean and rough street (LCR) tests (Pitt 1987).

Washoff Equations for Individual Tests

The particulate washoff values obtained during these Toronto tests were expressed in units of grams per square meter and grams per curb-meter, concentrations (mg/L), and the percent of the total initial loading washed off during the test. Plots of accumulative washoff are shown on Figures 3-14 through 3-21. These plots show the asymptotic washoff values observed in the tests, along with the measured total street dirt loadings. The maximum asymptotic values are the "available" street dirt loadings (N_o). The measured total loadings are seen to be several times larger than these "available" loading values. As an example, the asymptotic available total solids value for the HDS (high intensity rain, dirty street, smooth street) test (Figure 3-20) was about 3 g/m^2 while the total load on the street for this test was about 14 g/m^2, or about five times the available load. The differences between available and total loadings for the other tests were even greater, with the total loads typically about ten times greater than the available loads. The total loading and available loading values for dissolved solids were quite close, indicating almost complete washoff of the very small particles. However, the differences between the two loading values for SS were much greater. Shielding, therefore, may not have been very important during these tests, as almost all of the smallest particles were removed, even in the presence of heavy loadings of large particles.

The actual data are shown on these figures, along with the fitted Sartor and Boyd exponential washoff equations. In many cases, the fitted washoff equations greatly over-predicted suspended solids washoff during the very small rains (usually less than one to three mm in depth), possibly due to shielding. In all cases, the fitted washoff equations described suspended solids washoff very well for rains greater than about 10 mm in depth.

Tables 3-25 through 3-27 present the equation parameters for each of the eight washoff tests for total solids, suspended solids, and filterable solids. Pitt (1987) concluded that particulate washoff (defined by the suspended solids washoff) should be divided into two main categories, one for high intensity rains with dirty streets, possibly divided into categories by street texture, and the other for all other conditions. Factorial tests also found that the availability factor (the ratio of the available loading, N_o, to the total loading) varied depending on the rain intensity and the street roughness, as indicated below:

- Low rain intensity and rough streets: 0.045
- High rain intensity and rough streets, or low rain intensity and smooth streets: 0.075
- High rain intensity and smooth streets: 0.20

Figure 3-14. Washoff plots for HCR test (high rain intensity, clean, and rough street) (Pitt 1987).

Figure 3-15. Washoff plots for LCR test (light rain intensity, clean, and rough street) (Pitt 1987).

Figure 3-16. Washoff plots for HDR test (high rain intensity, dirty, and rough street) (Pitt 1987).

Figure 3-17. Washoff plots for LDR test (light rain intensity, dirty, and rough street) (Pitt 1987).

Figure 3-18. Washoff plots for HCS test (high rain intensity, clean, and smooth street) (Pitt 1987).

Figure 3-19. Washoff plots for LCS test (light rain intensity, clean, and smooth street) (Pitt 1987).

Figure 3-20. Washoff plots for HDS test (high rain intensity, dirty, and smooth street) (Pitt 1987).

Figure 3-21. Washoff plots for LCS replicate test (light rain intensity, clean, and smooth street) (Pitt 1987).

Table 3-25. Total solids washoff coefficients (Pitt 1987)[1]

Test condition code	Rain intensity category	Street dirt loading category	Street texture category	N_o (g/m^2) measured total initial total solids load	Calculated k (1/hr)	Standard error for k (1/hr)	N_o (g/m^2) available initial total solids load	Calculated k (1/hr)	Standard error for k (1/hr)
HCR	high	clean	rough	3.25	0.016	0.002	0.84	0.145	0.018
LCR	low	clean	rough	2.99	0.038	0.001	0.58	0.304	0.032
HDR	high	dirty	rough	12.82	0.004	<0.001	1.14	0.078	0.006
LDR	low	dirty	rough	11.22	0.013	0.001	0.74	0.383	0.024
HCS	high	clean	smooth	2.62	0.033	0.005	1.21	0.146	0.021
LCS	low	clean	smooth	2.32	0.026	0.001	0.35	0.301	0.024
HDS	high	dirty	smooth	13.82	0.012	0.001	2.74	0.138	0.008
LCS	low	clean	smooth	2.42	0.042	0.002	0.57	0.300	0.024

1) Note:
$N = N_o e^{-kR}$

Obviously, washoff was more efficient for the higher rain energy and smoother pavement tests. The worst case was for a low rain intensity and rough street, where only about 4.5% of the street dirt would be washed from the pavement. In contrast, the high rain intensities on the smooth streets were more than four times more efficient in removing the street dirt.

If a selected model requires available loading values instead of the total loading values, then a procedure must be used to adjust the total loading values (such as attempted by the availability term in STORM and SWMM). In all cases, the k term must be appropriate for the model form. However, the use of an available loading value for N_o requires the use of a substantially larger k term compared to using the total loading value.

Table 3-26. Suspended solids washoff coefficients (Pitt 1987)

Test condition code	Rain intensity category	Street dirt loading category	Street texture category	N_o (g/m^2) available suspended solids load	Calculated k (1/hr)	Standard error for k (1/hr)	Ratio of available load to total initial load
HCR	high	clean	rough	0.295	0.832	0.064	0.11
LCR	low	clean	rough	0.138	0.344	0.038	0.061
HDR	high	dirty	rough	0.375	0.077	0.008	0.032
LDR	low	dirty	rough	0.291	0.619	0.052	0.028
HCS	high	clean	smooth	0.462	1.007	0.321	0.26
LCS	low	clean	smooth	0.091	0.302	0.024	0.047
HDS	high	dirty	smooth	1.66	0.167	0.015	0.13
LCS	low	clean	smooth	0.209	0.335	0.031	0.11

Table 3-27. Filterable solids washoff coefficients (Pitt 1987)

Test condition code	Rain intensity category	Street dirt loading category	Street texture category	N_o (g/m^2) measured total initial filterable solids load	Calculated k (1/hr)	Standard error for k (1/hr)
HCR	high	clean	rough	0.651	0.061	0.004
LCR	low	clean	rough	0.745	0.139	0.006
HDR	high	dirty	rough	0.915	0.058	0.002
LDR	low	dirty	rough	0.680	0.163	0.006
HCS	high	clean	smooth	0.871	0.070	0.003
LCS	low	clean	smooth	0.395	0.154	0.007
HDS	high	dirty	smooth	1.223	0.085	0.002
LCS	low	clean	smooth	0.463	0.183	0.008

The total residue models were fitted using both total and available residue values to show the differences in the proportionality terms (k) for each loading type. In three cases (HCR, HCS, and HDS), the available residue form of the equations provided much better model residual analyses and were therefore preferred over the candidate equations using total loadings. The k values varied greatly (by about 5 to 30 times), depending on the use of total or available loadings.

Some of the attempts at fitting outfall data to the washoff model used total street dirt loading values, while the Sartor and Boyd values were based on available loadings. Obviously, this difference in loading definition easily could have been responsible for causing such different k values to be identified. The available loading forms of the equations for these washoff tests produced the largest k values (0.078 to 0.38), and are similar to the reported Sartor and Boyd value of 0.18 that is used as a "default" in many urban runoff models. The total loading model k terms are much smaller (0.004 to 0.042) and are close to those reported by Novotny (undated) (0.019 to 0.026) using Milwaukee NURP street dirt washoff observations and actual measured total street dirt loadings.

Selecting the appropriate k term for the correct form of N_o is critical. As an example, the rain volume needed to produce 90 percent washoff can be calculated using the standard washoff equation as follows:

$$N = N_o e^{-kR}$$

for 90 percent washoff, $N = 0.1 N_o$, and

$$0.1 N_o = N_o e^{-kR}, \text{ or}$$

$$0.1 = e^{-kR}, \text{ and}$$

$$(1/k) \log_e (0.1) = R, \text{ therefore}$$

$$R = 2.303/k \text{ for 90 percent washoff.}$$

For a k value of 0.3 (the LCS model for available total residue loadings), the rain needed for 90 percent washoff would be 8 mm. This rain would produce a washoff total of about 0.32 g/m^2 using the appropriate available N_o loading of 0.35 g/m^2. If the k value of 0.026 was used instead (appropriate for the total loading form of the LCS model), a rain of almost 90 mm would be needed for 90 percent washoff (more than ten times the rain depth predicted using the larger k value). In this case however, a total N_o value of 2.32 g/m^2 should be used, producing a washoff quantity of about 2.1 g/m^2 (more than 6.5 times the total residue washoff produced above). In all cases, the fitted models should obviously be used with caution beyond the test conditions. The 8 mm rain prediction is well within the test conditions, while the 90 mm rain prediction is almost four times the maximum rain used in these washoff tests. Other relationships between k values and rain quantities (mm) to produce specific percent washoffs are as follows:

Percent washoff	Rain needed (mm)
99.9	6.908/k
99	4.605/k
95	2.996/k
90	2.303/k
75	1.386/k
50	0.693/k
25	0.288/k
10	0.105/k

From these relationships, it is obvious that washoff occurs faster for larger k values (the washoff curves presented in Figures 3-14 through 3-21 would be steeper for larger k values if the figures were plotted without log scales).

The selected particulate residual washoff models were all based on the available loading model form because of superior model residual behavior. Therefore, an additional relationship is needed to predict available loading from total observed loading. The available particulate residue loadings ranged from about 3 to 25 percent (with an average of about 10 percent) of the total particulate residual loadings.

The filterable residue washoff models, however, were all based on total measured filterable residue loadings. These different preferred model forms for particulate and filterable residue were most likely caused by the differences in washoff efficiencies for different sized particles. Particulate residues were not nearly as efficiently removed during the washoff tests and were better related to much reduced “available” particulate residue loading values. Filterable residues in contrast, were much more efficiently removed and related well to total loadings (not much filterable residue was left on the streets after the washoff tests, making the available loadings very similar to the total loadings for filterable residue). Table 3-28 contains the availability relationship for suspended solids.

Table 3-28. Fraction of total street dirt suspended solids available for washoff (Pitt 1987).

Maximum Washoff Capacity

Another important consideration in calculating washoff of street dirt during rains is the carrying capacity of the flowing water. If the water velocity is high, it is much more capable of carrying particulates than for lower water velocities. This is the basic concept of the Yalin equation (using the Shield's diagram) and numerous other sediment transport equations: there is a physical limit to the ability of water to transport sediment. In contrast, the conventional washoff plots and equations presented earlier result in a "percentage" washoff of the total load, irrespective of the resultant concentration. However, when observing the plot of suspended solids concentration vs. rain depth for many washoff test plots (Figure 3-9), the pattern is quite distinct and appears to be generally independent on initial street loading (there is substantial scatter in this plot which likely reflects some site conditions). The washoff mostly is controlled by the carrying capacity of the water, and not source limitations, as there is substantial material on the street after the end of most rains. Therefore, this carrying capacity must be considered when predicting washoff quantities. If the calculated washoff is greater than the carrying capacity (such as would occur for relatively heavy street dirt loads and low to moderate rain intensities), then the carrying capacity is limiting. For high rain intensities, the carrying capacity is likely sufficient to transport most all of the washoff material.

In order to determine this carrying capacity for street runoff, data from washoff tests conducted by Pitt (1987) and Sartor and Boyd (1972), shown previously as Figures 3-6 and 3-14 through 3-21, were further examined. The maximum washoff amounts (g/m^2) for six different tests conducted on smooth streets were plotted against the rain intensity (mm/hr) used for the tests. This plot is shown in Figure 3-22, illustrating the exponential equation fitted to these data:

$$W = 0.0636\, e^{0.237P}$$

Where W = the maximum washoff, $grams/meter^2$
and P = average rain intensity, mm/hr

These are the maximum washoff values possible, representing the carrying capacity of the runoff. If the predicted washoff, using the previous "standard" washoff equations, is smaller than the values shown in this figure, then those values can be used directly. However, if the predicted washoff is greater than the values shown in this figure, then the values in the figure should be used.

Figure 3-22. Maximum washoff capacity for smooth streets (based on Pitt 1987 and Sartor and Boyd 1972 measurements).

The resulting sheetflow concentrations associated with these maximum washoff values depends on the rain durations at these average rain intensities. As an example, for typical 6 hour durations, the resulting concentrations are very similar to the fitted line on the suspended solids concentration vs. rain depth plot shown on Figure 3-9 (about 100 mg/L for 1 to 2 mm rains, decreasing to about 10 mg/L for rains of about 25 mm in depth). For very large rains, having sustained high rain intensities, the available street dirt loading would most likely be limiting.

Comparison of Particulate Residue Washoff Using Previous Washoff Models and Revised Washoff Model

This discussion briefly compares the washoff observations obtained during these washoff tests with predicted washoff values obtained using the Sartor and Boyd (1972) washoff model (with and without the “availability” factor). Table 3-29 shows the predicted washoff values along with the observed values for the conditions that occurred during the washoff tests. In all cases, serious over-predictions in street dirt washoff resulted by using these common washoff models. Even with the availability factor, the predicted Sartor and Boyd washoff quantities were almost two to more than five times greater than observed. Without the availability factor, the modeled washoff quantities were at least five times greater than the observed values. The residuals (all reflecting over-predictions) of these modeled estimates ranged from 0.2 to 7 g/m^2 when using the availability factor, compared to residuals mostly less than 0.05 g/m^2 when the model developed from these washoff tests was used. Lower residuals obtained by using the revised model could be expected because these data were not independent from the data used in developing the revised washoff model.

Table 3-29. Comparisons of Observed Washoff with Sartor and Boyd Equation Predictions (Pitt 1987).

As stated previously, over-predicted street dirt washoff quantities would result in under-predictions of particulate residue from other sources during model calibration. These over-predictions, especially combined with commonly over-predicted runoff flow volumes, dramatically affect the relative importance of different urban runoff pollutant source areas and estimated effectiveness of source area controls.

Summary of Street Particulate Washoff Tests

The above discussion summarized street particulate washoff observations obtained during special washoff tests, along with the associated street dirt accumulation measurements. The objectives of these tests were to identify the significant rain and street factors affecting particulate washoff and to develop appropriate washoff models. These tests and calculations were also used to clarify apparent confusion caused by misuse of washoff equations in urban runoff models.

The controlled washoff experiments identified important relationships between “available” and “total” particulate loadings and the significant effects of the test variables on the washoff model parameters. Past modeling efforts have typically ignored or misused this relationship to inaccurately predict the importance of street particulate washoff. The available loadings were almost completely washed off streets during rains of about 25 mm (as previously assumed). However, the fraction of the total loading that was available was at most only 20 percent of the total loading, and averaged only 10 percent, with resultant actual washoffs of only about 9 percent of the total loadings. Based on extrapolating the washoff models, only very large rains (possibly approaching 100 mm in depth) could ever be expected to wash off most of the total particulate street dirt load. These very large rains are well beyond the range of any washoff tests. However, observed street dirt washoff during actual rains near this size have not produced substantially greater washoff quantities than observed during the tests conducted during this research. The correctly used exponential washoff models only appear to be applicable for rains in the range of about 3 to 30 mm, which are the most important rains for water quality studies.

The fractions of the particulate residue loadings that were available for washoff was affected by both rain intensity and texture. In many model applications, total initial loading values (as usually measured during field studies) are used in conjunction with model parameters for available loadings, resulting in predicted washoff values that are many times over-predicted. This has the effect of incorrectly assuming greater pollutant contributions originating from streets and less from other areas during rains. This in turn results in inaccurate estimates of the effectiveness of different source area urban runoff controls.

Street dirt accumulation values have also been observed before and after rains. A tested industrial street experienced a much greater accumulation rate than the residential street, probably because of increased tracking of debris from unpaved driveways and parking areas and greater deposition of particulates from the heavy car and truck traffic. As shown in a summary of much accumulation data from throughout the US, smooth streets had much lower initial loadings immediately after street cleaning, but street texture did not affect particulate accumulations as much as land use.

These accumulation and washoff relationships were included in the Source Loading and Management Model (WinSLAMM) to describe street dirt washoff processes.

Observed Particle Size Distributions in Stormwater

A final note needs to be included in this section pertaining to the sizes of stormwater runoff particulates. The particle size distributions of stormwater greatly affect the ability of most controls to reduce pollutant discharges, and accumulation and washoff of particulates from source areas determines the particle sizes entering the storm drainage systems. Sedimentation and filtration controls are much more effective for large particles than for small particles, for example. Conventional street cleaning preferentially removes large particles from streets, but rains preferentially remove the smallest particle sizes. Inaccurate particle size assumptions of stormwater particulates than therefore dramatically affect performance predictions.

During several research projects, Pitt determined particle size analyses of 121 stormwater samples from three states that were not affected by stormwater controls (southern New Jersey as part of inlet tests; Birmingham, AL as part of MCTT pilot-scale tests; and in Milwaukee and Minocqua, WI, as part of the MCTT full-scale tests). These samples represented stormwater entering the stormwater controls being tested. Particle sizes were measured using a Coulter Multi-Sizer IIe and verified with microscopic, sieve, and settling column tests.

Figures 3-23 through 3-25 are grouped box and whisker plots showing the particle sizes (in μm) corresponding to the 10^{th}, 50^{th} (median) and 90^{th} percentiles of the cumulative distributions. If 90% control of SS is desired, for example, then the particles larger than the 90^{th} percentile would have to be removed by a sedimentation device. The median particle sizes ranged from 0.6 to 38 μm and averaged 14 μm. The 90^{th} percentile sizes ranged from 0.5 to 11 μm and averaged 3 μm. These particle sizes are all substantially smaller than have been typically assumed for stormwater. In all cases, the New Jersey samples had the smallest particle sizes, followed by Wisconsin, and then Birmingham, AL, which had the largest particles. The New Jersey samples were obtained from gutter flows in a residential semi-xeroscaped neighborhood, the Wisconsin samples were obtained from a public works yard in Milwaukee, and the Birmingham samples were collected from a long-term parking area.

Figure 3-23. Tenth percentile particle sizes for stormwater inlet flows.

Figure 3-24. Fiftieth percentile particle sizes for stormwater inlet flows.

Figure 3-25. Ninetieth percentile particle sizes for stormwater inlet flows.

“First-Flush” of Stormwater Pollutants from Pavement

“First flush” refers to the relatively high pollutant concentrations at the beginning of a wet weather event, with decreasing concentrations as the event progresses. Sutherland (personal communication) suggests examining it by preparing a double mass curve, with accumulative runoff volumes (x axis) vs. accumulative pollutant mass (y axis). If first flush occurs, the resulting curve will bow upward initially and generally stay above the diagonal straight line from 1 to 100% (unfortunately, I don’t have a good illustration). There is frequent mention of the phenomena of “first flush” as an opportunity for stormwater control, specifically as the reason why treatment of the first ½ inch of runoff is adequate. Concentrations at outfalls of most urban drainages do not routinely experience pronounced first flushes. However, they are well documented for combined systems, where CSO concentrations are very large at the beginning of events when accumulated sanitary solids in the sewerage can be easily scoured by a slight rise in the flow rate.

The controlled pavement washoff tests described in this section show large solids concentrations at the beginning of the tests, with significant decreases as the test progresses. These tests were conducted with constant “rain” intensities (and therefore constant kinetic energy). The initial abstractions and infiltration of water through the pavement also results in less runoff at the beginning of the test. However, there is an abundance of material on pavement surfaces that is not removed easily by low to moderate rain intensities. If the rain intensity increases later in the event, then pollutant concentrations would likely increase according as the available energy to dislodge and transport particulates increase. In addition, these tests were conducted with the simplest drainage conditions. In a real watershed, many source areas are contributing pollutants, but the travel times from the sources to the outfall are highly varied. This would moderate the high concentrations observed during the simple tests, as the first flushed material would arrive at different times at the outfall. In addition, as flows decrease during times of decreasing rain intensity, the transport ability (carrying capacity) of the water decreases, with deposition in the drainage system (onto pavement, in gutters, in grass channels, in the sewerage, etc.). These flow contribution irregularities, coupled with varying rain intensities during storms, generally masks significant first flush conditions at outfalls.

An example of first flush from a relatively simple watershed is shown in Figure 3-26 through 3-28 (Shaheen 1975). The test watershed was a portion of the Washington, D.C. beltway (I495), almost totally paved and guttered. This relatively small, but common rain (about 0.1 inch) produced peak flows of about 24 gal/min. The event had a relatively constant rain intensity and classical hydrograph shape with a rapid rise and drop. This event also had a pronounced first flush, with high concentrations of total solids, suspended solids, and lead at the beginning of the event, decreasing to about half. Constituents more associated with filterable fractions (soluble zinc and soluble lead) had little change over the period of the event. In contrast, another event at the same location is shown in Figures 3-29 and 3-30. The initial rainfall was about the same as for the other event, but significantly increased after about 2 hours. The hydrograph shows an initial rise and drop corresponding to the first part of the event, but the majority of the runoff occurred later in the event. The concentrations also showed an initial period of relatively high values, and then dropped, but later significantly increased when the rain intensity increased. The period of high concentrations (and high pollutant yields) occurred about two hours after rain started, conflicting with the first flush "theory." The concept of treating the first ½ inch of runoff from each event is usually successful, as almost all rains produce less than this amount, and about 80% of annual flows in many parts of North America, not because capturing the first flush allows treatment of a significantly more polluted and smaller portion of the runoff.

Figure 3-26. Rain and flow for storm event of Sept. 18, 1973, Washington, D.C. beltway freeway site (Shaheen 1975).

Figure 3-27. Solids concentrations for storm event of Sept. 18, 1973, Washington, D.C. beltway freeway site (Shaheen 1975).

Figure 3-28. Heavy metal concentrations for storm event of Sept. 18, 1973, Washington, D.C. beltway freeway site (Shaheen 1975).

Figure 3-29. Rain and flow for storm event of Aug. 21, 1973, Washington, D.C. beltway freeway site (Shaheen 1975).

Figure 3-30. Pollutant concentrations for storm event of Aug. 21, 1973, Washington, D.C. beltway freeway site (Shaheen 1975).

Comparisons of First-Flush vs. Composite Samples at Stormwater Outfalls

Maestre, *et al*. (2004) compared outfall sample concentrations from NPDES permits, using data obtained as part of a EPA 104b(3) project that compiled monitoring information from many permit holders. As part of their NPDES stormwater permit, some communities collected grab samples during the first 30 minutes of the event to evaluate a "first flush" in contrast to the flow-weighted composite data. More than 400 paired samples representing the first flush and composite samples from eight communities (mostly located in the southeast U.S.) from the National Stormwater Quality Database (NSQD) (http://civil.eng.ua.edu/~rpitt/Research/ms4/mainms4.shtml) were reviewed. Box and probability plots were prepared for 22 major constituents. Nonparametric statistical analyses were then used to measure the differences between the sample sets. This discussion presents the results of this preliminary analysis, including the effects of storm size and land use on the presence and importance of first flushes. Only concentration data were available for these analyses, so traditional accumulative mass curves could not be developed.

It is expected that peak concentrations generally occur during periods of peak flow (and highest rain energy). On relatively small paved areas, however, it is likely that there will always be a short period of relatively high concentrations associated with washing off of the most available material (Pitt 1987). This peak period of high concentrations may be overwhelmed by periods of high rain intensity that may occur later in the event. In addition, in more complex drainage areas, the routing of these short periods of peak concentrations may blend with larger flows and may not be noticeable. A first flush in a separate storm drainage system is therefore most likely to be seen if a rain occurs at relatively constant intensity over a paved area having a simple drainage system.

A total of 417 storm events with paired first flush and composite samples were available from the NSQD. The majority of the events were located in North Carolina (76.2%), but some events were also from Alabama (3.1%), Kentucky (13.9%) and Kansas (6.7%). All of the data were from end-of pipe samples in separate storm drainage systems.

The initial analyses were used to select the constituents and land uses that meet the requirements of the statistical comparison tests. Probability plots, box plots, concentration vs. precipitation, and standard descriptive statistics, were performed for 22 constituents for each land use, and for all land uses combined. Nonparametric statistical analyses were performed after the initial analyses. Mann Whitney and Fligner Policello tests were most commonly used. Minitab and Systat statistical programs, along with Word and Excel macros, were used during the analysis.

The Mann-Whitney and Fligner-Policello non-parametric tests were selected to determine if there were statistically significant differences between the first flush and composite data sets for each land use and constituent. These tests are very useful because they require only data symmetry, not normality, to evaluate the hypothesis. The null hypothesis during the analysis was that the median concentrations of the first flush and composite data sets were the same. The alternative hypothesis was that the medians were different, with a confidence of at least 95%.

Results

A complete description of these analyses is presented in Maestre, *et al.* (2004). Table 3-30 shows the results of the analysis. The ">" sign indicates that the median of the first flush data set is higher than for the composite data set. The "=" sign indicates that the there is not enough information to reject the null hypothesis. Events without enough data are represented with an "X". Also shown on this table are the ratios of the medians of the first flush and the composite data sets for each constituent and land use. The first flush samples were larger than for the composite samples if the ratio is great than one. Generally, a statistically significant first flush is associated with a median concentration ratio of about 1.4, or greater (the exceptions are where the number of samples in a specific category is small). The largest significant ratios are about 2.5, indicating that the first flush concentrations may be about 2.5 times greater than the composite concentrations. More of the larger ratios are found in the commercial and institutional land use categories, areas where larger paved areas are likely to be found. The smallest ratios are associated with the residential, industrial, and open space land uses, locations where there may be larger areas of unpaved surfaces.

Results indicate that for 55% of the evaluated cases, the median of the first flush data set was significantly larger than for the composite sample set. In the remaining 45% of the cases, both medians were expected to be the same, or the concentrations were possibly greater later in the events. About 70% of the constituents in the commercial land use category had first-flushes, while about 60% of the constituents in the residential, institutional and the mixed (mostly commercial and residential) land use categories had first flushes, and about 45% of the constituents in the industrial land use category had first-flushes. In contrast, no constituents were found to have first-flushes in the open space category.

COD, BOD_5, TDS, TKN, and Zn all had first flushes in all areas (except for the open space category). In contrast, turbidity, pH, fecal coliforms, fecal strep., total N, dissolved and ortho-P never showed a statistically significant first flush in any category. The conflict with TKN and total N implies that there may be some other factors involved in the identification of first flushes besides land use. If additional paired data becomes available during later project periods, it may be possible to extend these analyses to consider rain effects, drainage area, and geographical location.

Table 3-29. Presence of Significant First Flushes (ratio of first flush to composite median concentrations)

Parameter	Commercial	Industrial	Institutional	Open Space	Residential	All Combined
Turbidity	= (1.32)	X	X	X	= (1.24)	= (1.26)
pH	= (1.03)	= (1.00)	X	X	= (1.01)	= (1.01)
COD	> (2.29)	> (1.43)	> (2.73)	= (0.67)	> (1.63)	> (1.71)
TSS	> (1.85)	= (0.97)	> (2.12)	= (0.95)	> (1.84)	> (1.60)
BOD_5	> (1.77)	> (1.58)	> (1.67)	= (1.07)	> (1.67)	> (1.67)
TDS	> (1.82)	> (1.32)	> (2.66)	= (1.07)	> (1.52)	> (1.55)
O&G	> (1.54)	X	X	X	= (2.05)	> (1.60)
Fecal Coliform	= (0.87)	X	X	X	= (0.98)	= (1.21)
Fecal Strep.	= (1.05)	X	X	X	= (1.30)	= (1.11)
Ammonia	> (2.11)	= (1.08)	> (1.66)	X	> (1.36)	> (1.54)
NO_2 NO_3	> (1.73)	> (1.31)	> (1.70)	= (0.96)	> (1.66)	> (1.50)
Total N	= (1.35)	= (1.79)	X	= (1.53)	= (0.88)	= (1.22)
TKN	> (1.71)	> (1.35)	X	= (1.28)	> (1.65)	> (1.60)
Total P	> (1.44)	= (1.42)	= (1.24)	= (1.05)	> (1.46)	> (1.45)
P Dissolved	= (1.23)	= (1.04)	= (1.05)	= (0.69)	> (1.24)	= (1.07)
Phosphate Ortho	X	= (1.55)	X	X	= (0.95)	= (1.30)
Cd	> (2.15)	= (1.00)	X	= (1.30)	> (2.00)	> (1.62)
Cr	> (1.67)	= (1.36)	X	= (1.70)	= (1.24)	> (1.47)
Cu	> (1.62)	> (1.24)	= (0.94)	= (0.78)	> (1.33)	> (1.33)
Pb	> (1.65)	> (1.41)	> (2.28)	= (0.90)	> (1.48)	> (1.50)
Ni	> (2.40)	= (1.00)	X	X	= (1.20)	> (1.50)
Zn	> (1.92)	> (1.540	> (2.48)	= (1.25)	> (1.58)	> (1.59)

Summary

Pollutants can originate from many source areas in urban watersheds. During small storms, directly connected paved areas (such as streets) contribute the majority of pollutants and flows. However, as the rain events increase in size, other areas become important. Receiving water impacts are associated with a variety of storm types, and WinSLAMM can be used to identify the significant pollutant sources for each category of storm, and can also evaluate the benefits of alternative stormwater controls under a wide variety of conditions.

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