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16. Abstract

This report is the first in a series which will address the water quantity and quality impacts of highway construction in the Austin, Texas area. This literature review evaluates the impact of highway construction and operation on surface water quality and on recharge of groundwater aquifers. The types of barriers for containment and retention of sediment and pollutants from runoff and the effectiveness of each device are discussed. The report also addresses the quantity and quality of highway runoff from different types of road surfaces, drainage and conveyance systems, and various types of highways. In addition, methods and strategies for the handling and control of highway runoff and effectiveness of pollution control devices are reviewed.

Highway construction may cause changes in turbidity, suspended solids concentration, and color of the receiving waters. The extent and persistence of the effects are very site specific and are usually transitory. Prevention of erosion during construction with the use of vegetative stabilization is the most effective way to minimize the adverse effects of runoff.

Previous research has identified surrounding land use, traffic volume, and rainfall characteristics as the most important factors for predicting the quality of highway stormwater runoff. Most studies have concluded that the type of paving material has a relatively small effect on runoff quality. The type and size of the receiving water, the potential for dispersion, the size of the catchment area, and the biological diversity of the ecosystem are some of the factors which determine the extent and importance of runoff effects.

Most of the pollutant load in highway runoff is either the suspended particulate matter, or material adsorbed to the suspended solids. The most effective control measures reduce the amount of particulates in runoff through settling or filtration. Most design references specify vegetated controls because of their wide adaptability, low costs, and minimal maintenance requirements. Wet ponds are recommended when site conditions are not conducive to vegetated controls. Infiltration practices, although offering excellent treatment potential, are the least desirable because of their high maintenance requirements.

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Water Quantity and Quality Impacts Assessment of Highway Construction in Austin, Texas, Area

conducted for the

Texas Department of Transportation

by the

CENTER FOR TRANSPORTATION RESEARCH
Bureau of Engineering Research
THE UNIVERSITY OF TEXAS AT AUSTIN

April 1993

IMPLEMENTATION STATEMENT

This report reviews previous studies pertaining to the quantity and control of pollution from highway runoff and construction. The authors report the amounts and types of pollutants identified by other researchers and evaluate the performance of runoff controls. Use of recommended runoff controls will help districts reduce the amount of nonpoint pollution attributed to highway stormwater runoff.

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Prepared in cooperation with the Texas Department of Transportation.

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EXECUTIVE SUMMARY

Sources of Pollutants

Vehicles directly and indirectly contribute much of the pollution found in highway runoff. Vehicles are a source of the metals, chemical oxygen demand, oil and grease, nitrates, sulfates, phosphorus, and other materials deposited on highways. Other major sources of contaminants include dustfall and even dissolved constituents in rain itself. Rainfall can contribute up to 78% of the major ionic contaminants leaving the road surface in runoff, and up to 48% of the suspended solids. Dustfall loadings can be a significant fraction of the loadings in runoff and an important source of highway pollution. This is especially true for highways near or in urban areas. Thus the surrounding land use has a major impact on the amount of pollution in dustfall deposited on a highway and on the ensuing quality of stormwater runoff.

Factors Affecting Highway Runoff

Traffic volume is an important factor in predicting runoff quality. This is especially true for the number of vehicles during a storm event. Less clear is the relationship between average daily traffic and the amount of pollutants in highway runoff. Removal processes such as air turbulence (natural or the result of vehicles) limit the accumulation of solids and other pollutants on road surfaces, thereby obscuring the relationship between the traffic volume and runoff loads.

The precipitation characteristics which may impact the water quality of highway runoff include the number of dry days preceding the event, the intensity of the storm, and storm duration. Only a weak correlation between antecedent dry days and runoff quality has been demonstrated. Storm intensity has a marked impact because many of the pollutants are associated with particles which are more easily mobilized in high-intensity storms. Constituents showing a strong correlation with suspended solids include metals, organic compounds, total organic carbon, and biochemical oxygen demand. For low-intensity storms, other mechanisms (i.e., vehicles) are partially responsible for the removal of pollutants from highway surfaces. Larger storms dilute highway runoff and lower concentrations of contaminants. However, the loading of pollutants (total mass transported) is generally higher in longer storms, as the transport of at least some constituents continues throughout the duration of the event.

Higher concentrations of pollutants are often observed in the first runoff from a storm, a phenomenon referred to as the "first flush." This is especially true for dissolved components including nutrients, organic lead, and ionic constituents. In general, concentrations of particle-associated pollutants show a more complex temporal variation related to rainfall intensity and the flushing of sediment through the drainage system.

The choice of highway paving material (asphalt versus concrete) seems to have only a small effect in determining the quality of highway runoff. Most studies have found that highway surface type was relatively unimportant compared to such factors as surrounding land use. It has also been reported that the type of collection and conveyance system for highway runoff (storm sewer, grassy swale, etc.) has a greater effect on runoff quality than pavement type.

In addition to the general factors discussed above, the range of pollutant concentrations and loads can also be attributed to site-specific conditions or seasonal variations. Excess solid loadings have been connected to environmental sources as well as to highway maintenance practices. Hydrological effects of highways and water quality effects of highway runoff on receiving waters are also highly site-specific.

Environmental Effects on Receiving Waters

The type and size of the receiving body, the potential for dispersion, the size of the catchment area, and the biological diversity of the receiving water ecosystem are just some of the factors which determine the extent and importance of highway runoff effects.

Highways increase the amount of impervious cover on a watershed and thus raise storm runoff volumes and peak discharges. Consequently, there is an increase in streambank erosion and greater loads of solids and other pollutants into receiving waters. Soluble pollutants adversely affect algae and zooplankton, while suspended solids interfere with the respiration of young fish. Bioassay tests of adult fish in receiving waters have yielded conflicting results.

Stream and lake sediments have been found to be a reservoir for heavy metals and the primary source for the bioconcentration of metals. Some of the toxic effects of metals in highway runoff can be greatly reduced by natural processes within the receiving water. For example, ionic species of incoming trace metals may be reduced by complexation.

Highways can have a significant impact on groundwater, including changes in water quality in the vadose and saturated zones. Metals concentrations in groundwater have been detected at elevated levels in the vicinity of highways and runoff control structures. Highway runoff may also increase the concentration of constituents other than metals such as Kjeldahl nitrogen and organic compounds.

Highway runoff effects on groundwater are often spatially limited due to local hydrological conditions as well as sorption processes within and above the aquifer. Furthermore, the effects of runoff on groundwater are minimized by processes in the soils such as precipitation and adsorption. These processes are highly dependent on the type and thickness of soil at a particular site. Lead is especially immobilized by soils, but organic and ionic constituents are relatively mobile and pose threats to groundwater quality.

Highway Construction

Highway construction may cause changes in turbidity, suspended solids concentration, and color of the receiving waters. The extent and persistence of the changes vary from site to site. It is often hard to distinguish the effects of highway construction activity from other influences such as concurrent commercial development or nearby industry. When construction impacts on stream quality are detected, they are usually transitory. Some authors have reported that highway construction has resulted in the loss of mangroves, seagrass, marshes, and swamps.

Prevention of erosion during highway construction is important to minimize the effects on receiving waters. Vegetative stabilization is the most effective method for reducing construction impacts. Sedimentation ponds, when designed with sufficient holding times, have also proved useful for reducing suspended sediment loads.

Controlling Pollution from Highway Runoff

The control of pollution from highway runoff can be accomplished by both source management and structural controls. Most of the pollutant load is either the suspended particulate matter, or material adsorbed to the suspended solids. The most effective control measures reduce the amount of particulates available for transport, or settle and/or filter the particulate material in runoff.

Source management includes transportation plans which can be designed to lower the total vehicle miles traveled and the implementation of land use plans which restrict developments which generate high traffic volumes in sensitive areas. Reduction of pollutant runoff can be accomplished by the elimination of curbs and other barriers, traffic flow regulation, and minimizing the use of fertilizers and pesticides.

Structural controls which are appropriate for highways include vegetative practices, ponds, infiltration methods, wetlands, and filters. Vegetative controls include the grassed swale and vegetated buffer strips. These controls are popular because of their low costs and minimum maintenance requirements. They have been shown to reduce concentrations of metals, oil and grease, and suspended solids. Removal of nutrients is often less effective. Factors which reduce the effectiveness of swales include steep slopes and fine-grained soils. Steep slopes contribute to high runoff velocities which mat the grass and reduce the time available for treatment and infiltration.

Different types of ponds used to treat highway runoff include detention, extended-detention, and "wet" ponds. Detention ponds are primarily flood control devices and are designed to be dry between storm events. Because of the short detention times associated with these structures, they are neither reliable nor effective in treating highway runoff. Extended-detention ponds are dry ponds designed to retain the runoff for 6 to 12 hours. This results in increased removal of particles and particle-associated pollutants. However, nutrient removal rates are low and sometimes even negative. The construction costs of dry ponds are generally the least of those for all pond options, but the maintenance burden is usually higher.

Wet ponds are considerably more effective at mitigating highway runoff pollution and are the best choice when vegetative controls are not feasible. These ponds are designed to maintain a permanent pool of water and to retain a certain amount of storm runoff. Pollutant removal is achieved primarily through sedimentation and biological processes. Many pollutants are retained in the pond sediments, but concentrations remain much lower than the EPA's criteria for hazardous waste designation. Pond depth, surface area, and shape are all important factors affecting pollutant removal. Costs for wet ponds are definitely higher than for other ponds, not including permitting costs which may equal or exceed design costs in some cases. In addition, land cost and surrounding land use may restrict their applicability.

Constructed wetlands have the ability to assimilate large quantities of dissolved and suspended solids and exhibit a high nutrient demand. Pollutant removal is achieved primarily through plant uptake, physical filtration, adsorption, gravitational settling, and microbial decomposition. The high cost of wetlands is usually associated with their increased land requirements, which may be two to three times the space required for other control methods. Wetlands are difficult to establish in areas with high soil permeabilities or high evapotranspiration rates.

Infiltration trenches and basins are designed to contain a certain volume of highway runoff and treat it through percolation into the underlying subsoil, or through a prepared porous media filter bed. Although not well documented, pollutant removal rates appear to be very high. These controls are highly dependent on specific site conditions, so they may not be applicable in many areas. Costs for infiltration structures are higher than for pond systems especially when based on volume of runoff treated, and maintenance appears to be a serious problem. Most infiltration basins have failed due to rapid clogging, usually within 5 years.

Sand filters treat stormwater runoff by percolating it through sand beds, after which it is collected in drainage pipes and discharged downstream. Removal rates are high for suspended solids and trace metals, and moderate for biochemical oxygen demand, nutrients, and fecal coliform. Sand filter performance can be increased by incorporating peat into the filter material. These filters are useful in areas with thin soils, soils with low infiltration rates, and areas of high evapotranspiration. Construction costs are very high and maintenance is required on a regular basis to prevent clogging of the sand bed with sediment.

Several structural additions have been used in conjunction with primary runoff controls to increase their performance. These additions include oil/grit chambers, sediment forebays, and granular-activated carbon filters. Oil and grit chambers used to remove heavy particulates and adsorbed hydrocarbons are relatively ineffective due to their high maintenance requirements. Sediment forebays have been shown to be useful in reducing the sediment load to infiltration structures and sand filters. Granular-activated carbon has been used to treat runoff before discharge to underground drainage wells, but it is very expensive.

Pollutant removal can also be increased by combining several of the structural control devices. Combinations may increase the ability to effectively filter suspended solids, or may be useful in reducing the site limitations of a single control measure. The redundancy of expected pollutant removal efficiencies increases the overall reliability and performance of the system.

For highway runoff, most design references specify vegetated controls as their first choice because of their wide adaptability, low costs, and minimal maintenance requirements. Wet ponds are recommended when site conditions are not conducive to vegetated controls. Infiltration practices, although offering excellent treatment potential, are the least desirable because of their high maintenance requirements.

Recommendations for Future Studies

Structural controls for treating stormwater runoff are becoming increasingly common; however, little quantitative work has been done to establish the most effective designs. The use of a combined treatment system which includes an extended detention pond and a vertical filter appears to offer many advantages to single-treatment technologies. Sediments and other particulate matter should not collect on a vertical filter as quickly as on a conventional horizontal sand filter, reducing maintenance requirements. Replacing part of the sand in the filter with other adsorbing materials may increase the removal of heavy metals and other pollutants. Design parameters which require a better understanding include optimum filter thickness, filter media composition, optimum detention time, and effect of antecedent dry periods on filter performance.

The testing of structural controls will require an accurate characterization of the composition of highway runoff. The ability to predict highway runoff quality has been limited by the many variables which combine to make each storm event unique. Differences in antecedent dry period, rainfall intensity, traffic volume, surrounding land use, highway surface type, and drainage method results in a wide range of concentrations for many of the pollutants observed in runoff. A system which could simulate rainfall at predetermined intervals might allow the individual effects to be determined with much greater accuracy, resulting in models which could be used to predict runoff quality. Such models could be used to predict the effects of highway construction in environmentally sensitive areas.

Studies of the constituents in highway runoff have been conducted in many parts of the United States; however, little research has been done in the Southwest. Many studies also do not consider important parameters such as rainfall intensity, or the temporal distribution of pollutants in runoff. Pollution control structures often are designed to collect the "first flush" of runoff (commonly the first 1/2 inch (13 mm)), but the highest concentrations of pollutants may occur only when rainfall intensity exceeds the level necessary to transport particles from the road surface. In addition, rain and dustfall have been shown to contribute significant amounts of pollutants to highway runoff. Consequently, a runoff sampling program in Texas could help establish whether regional differences are important, determine the types and amounts of pollutants contributed by the atmosphere in this area, and identify what portion of the runoff should be collected and treated.

Although a few studies have examined temporary runoff controls at construction sites, very little data are available on the relative effectiveness of silt fences and rock berms. These devices are commonly used for the containment and retention of sediment and pollutant load. A program to monitor these temporary controls at highway construction sites could provide valuable information on their performance, maintenance requirements, and life span.

Little is known about the effects of highway runoff on groundwater quality in a karst terrane (cavernous limestones with thin soils). Other studies which have shown minimal effects on groundwater quality have been located in areas with fairly thick soils, which immobilize many of the pollutants in runoff. In the Austin, Texas, area, groundwater recharge to the Edwards aquifer occurs primarily in stream beds during storm events. For this reason, it would be useful to establish a field sampling program of the quantity and quality of the surface water in the creeks and drainage ways affected by highway construction and operation.

1.0 INTRODUCTION

Regulatory agencies have recently focused attention on nonpoint sources of pollution such as urban runoff. The EPA's National Pollutant Discharge Elimination System (NPDES) regulations regarding stormwater runoff are evidence of this effort.

In Texas, the Barton Springs/Edwards Aquifer Conservation District (the District) and several environmentally oriented organizations became concerned about the potential for aquifer contamination as a result of proposed highway construction activities over the Edwards aquifer. The proposed construction of the extension of Loop 1 – MoPac South and a segment of SH 45, also referred to as "the Outer Loop," crosses and parallels three creeks and overlies a portion of the recharge zone of the Barton Springs segment of the Edwards aquifer. This concern resulted in litigation involving the Texas Department of Transportation (TxDOT) and the Federal Highway Administration (FHA) which temporarily halted construction activities on the project site.

Prior to this halt in construction, the District and TxDot negotiated a settlement between their two agencies which was approved by the U. S. District Court. The District removed itself from the litigation and TxDot began implementing certain actions and practices to answer the concerns of the District. By working cooperatively, the two agencies have been effective in preventing or reducing pollution from both point and nonpoint sources during roadway construction activities. Many improvements and innovations have been developed for structural and non-structural Best Management Practices (BMPs) which have gained both agencies local, state, and national recognition as leaders in the field of pollution prevention and mitigation.

The agreed-to Consent Decree also ordered a study of the water quality and quantity of highway runoff and the effects of highway construction and operation on the quality of receiving waters. TxDot and the District agreed to have the study conducted by the Center for Research in Water Resources (CRWR) at The University of Texas at Austin. A technical review committee consisting of three representatives of the District, two from TxDot, and two from the CRWR will meet regularly to review recent activities and progress reports. The committee will provide input and guidance to the CRWR on the overall study, its procedures, equipment, and upcoming work efforts.

The study requires a review of previous research into the quality of stormwater runoff, the environmental impacts of highway construction and operation, and feasible mitigation strategies to control the negative effects of highway runoff. This literature review has been prepared in partial fulfillment of that requirement for review of prior studies.

Although most of the published literature pertaining to the constituents within runoff from paved surfaces is focused on urban runoff, some literature dealing specifically with runoff from highways does exist. Many of the reports on this subject constitute "gray" literature, documents published by the Federal Highway Administration or state departments of transportation throughout the country. Little information appears in archival journals. Most of the reports were obtained through/from the Center for Transportation Research at The University of Texas at Austin, the Technology Transfer Library at TXDOT, DOT libraries in the states of Washington and Florida, and the National Technical Information Service (NTIS). The literature review has been aided by computer searches of data bases such as TRIS and COMPENDEX. The complete bibliography is contained in Appendix B. References shown in boldface type have been cited in the text.

This review is divided into five parts. The first, "Sources of Pollutants," discusses the amounts and types of pollutants derived from vehicles as well as other sources. "Factors Affecting Highway Runoff" reports on the pollutants found in highway runoff, several factors which influence the amount of runoff and pollution, and the processes involved in the transport and transformation of highway-related pollutants. Runoff concentrations and loads reported in several studies are discussed and the possible reasons for the wide range of values are evaluated. "Environmental Effects on Receiving Waters" analyzes the effect of highway runoff on streams, rivers, lakes, wetlands, soil water, and groundwater. The range of quantitative data is explained in terms of the relevant factors such as transport and transformation processes. "Highway Construction" discusses the important constituents (mainly solids, oil, and grease) in runoff from construction sites and analyzes the effects on receiving water quality. Also discussed are methods to minimize construction impacts. "Controlling

Pollution from Highway Runoff" summarizes the results from studies of source management as well as permanent pollution controls to protect receiving waters from the possible effects of highway runoff. Both structural devices such as filters, swales, and ponds, and non-structural measures such as planning and maintenance, are considered.

2.0 SOURCES OF POLLUTANTS

Major sources of pollutants on highways are vehicles, dustfall, and precipitation. Many factors affect the type and amounts of these pollutants, including traffic volume and type, local land use, and weather patterns.

Other possible, but infrequent, sources of pollutants include spills of recreational vehicle wastes, agricultural or chemical products, or oil and gas losses from accidents. These losses are related to traffic volume and may often go unnoticed, but could result in a large pollutant load locally (Asplund, 1980). Roadway maintenance practices such as sanding and deicing, or the use of herbicides on highway right-of-ways, may also act as sources of pollutants.

2.1 Vehicles

Vehicles are both a direct and an indirect source of pollutants on highways. As a direct source, vehicles contribute pollutants from the normal operation and frictional parts wear. Indirect or acquired pollutants are solids that are acquired by the vehicle for later deposition, often during storms (Asplund et al., 1980).

Vehicles directly contribute much of the metals, COD, oil and grease, nitrates, sulfate, and phosphorus deposited on highways through emissions and leakage. Tire wear contributes oxidizable rubber compounds and zinc oxides (Christensen and Guinn, 1979). Other studies have tried to quantify the amount of pollutants in highway runoff attributed directly to vehicles, but have been only partially successful. A more complete discussion of the effects of vehicles on highway runoff water quality is contained in Section 3.1.

Indirectly, vehicles contribute to highway pollution by carrying solids from parking lots, urban roadways, construction sites, farms, and dirt roads. Shaheen (1975) showed that more than 95% of solids on a given highway originate from sources other than the vehicles themselves.

2.2 Atmospheric Deposition

Atmospheric sources contribute a significant amount of the pollutant load in highway runoff. The deposition may occur in precipitation during storms or as dustfall during dry periods.

Annual loads of physical and chemical constituents in bulk precipitation to a rural highway bridge were estimated by Irwin and Losey

(1978) by extrapolating from five individual events. Bulk precipitation loads were a significant percentage of total loads in bridge runoff (Table 2.1). Precipitation loads were even higher than runoff loads for some dissolved constituents (e.g., chloride, sodium, and dissolved solids). Estimation error or removal mechanisms other than stormwater runoff (e.g., vehicular splashing) may account for these observations.

Many major ionic constituents originate from atmospheric pollution. Harrison and Wilson (1985a) found that rainfall can contribute up to 78% of the major ionic contaminants (Na^+, K^+, Mg^{2+}, Ca^{2+}, Cl^-, and SO_4^{2-}) leaving the road surface in runoff and up to 48% of the suspended solids.

Atmospheric dry fallout can also contribute large amounts of pollutants to highway surfaces. Irwin and Losey (1978) and Harrison and

Table 2.1
Estimated Loads of Selected Chemical and Physical Parameters in Bridge Surface Bulk Precipitation and in Bridge Surface Runoff
(Irwin and Losey, 1978)

Parameter	**Bridge Surface Runoff** (lb/yr)	**Bulk Precipitation** (lb/yr)
Dissolved solids	220	280
Sodium (Na)	2.9	14.3
Chloride (C1)	6.9	26
Suspended solids	1,210	138
Oil and grease	9.1	17.9
Nitrogen (N)	14.6	11.3
Phosphorus (P)	1.8	.58
Organic carbon	78.8	17.9
BOD	45.6	21.5
Chromium (Cr)	< .15	< .20
Copper (Cu)	.44	.08
Lead (Pb)	2.60	1.04
Mercury (Hg)	< .01	< .01
Nickel (Ni)	.11	.11
Zinc (Zn)	1.60	.32

SI units: To convert lb/yr to kg/yr multiply by 0.454

Table 2.2
Summary of Dustfall Loading Rate for Monitoring Sites (gm/m^2/day).
(Gupta et al., 1981c)

Monitoring sites	1976			1977			
	Nonwinter[a]		Winter	Nonwinter[a]		Winter [b]	
	Avg.	Range	Typical value	Avg.	Range	Avg.	Range
Milwaukee-Hwy. 794	0.30	0.12-0.52	0.87	0.56	0.11-2.45	0.15	0.10-0.21
Milwaukee-Hwy. 45	0.21	0.03-0.38	0.11	0.31	0.05-0.58	0.13	0.06-0.20
Harrisburg	0.13	0.07-0.16	0.07	0.06	0.04-0.09	0.07	0.05-0.09
Nashville	0.30	0.23-0.38	NS	0.90	0.37-2.07	1.43	0.53-2.17
Denver	0.37	0.30-0.49	NS	0.32	0.07-0.68	0.34	0.27-0.46

Customary units: To convert gm/m^2/day to lb/ac/day multiply by 8.9
Note: NS = no dustfall samples taken during this period
a Represents monitoring periods between April through October
b Represents monitoring periods between November through March

Table 2.3
Loadings of Total Solids in Highway Runoff Non-Winter Periods
of 1976 and 1977
(Gupta et al., 1981c)

Site	Average gm/m^2/event	Range gm/m^2/event
Milwaukee -HW 794	3.8	.2 - 9.2
Milwaukee -HW 45	3.2	.4 - 9.2
Harrisburg	1.9	.2 - 8.2
Nashville	3.7	.1 - 6.5
Denver	2.4	.2 - 7.3

Customary units: To convert gm/m^2/event to lb/ac/event multiply by 8.9

Wilson (1985a) discussed, but did not quantify, this phenomenon. Table 2.2 quantifies this source based on work by Gupta et al. (1981c). The significance of this dustfall loading can be seen by comparison with the highway runoff loading of total solids presented in Table 2.3. The average values in runoff are loadings per event, and for each site they are approximately ten times the dustfall values given in Table 2.2, which are loadings per day. It is interesting to note that the average dry period between events for these non-winter periods was approximately ten days. If all the dustfall remained on the highway, dustfall loading would approximately equal the loading in the runoff.

Surrounding land use has an important effect on the amount and types of pollution in dustfall. Highways in or near urban areas have been shown to have significantly higher levels of pollutant loading from dustfall than those in rural areas (Gupta et al., 1981c).

3.0 FACTORS AFFECTING HIGHWAY RUNOFF

There are many mechanisms for the removal of pollutants from highways. These include stormwater runoff, wind, vehicle turbulence, and the vehicles themselves.

The major pollution removal mechanisms in low-precipitation areas are natural surface winds and traffic-created turbulence (Aye, 1979). The mechanical scrubbing action of the tires together with wind (both natural and vehicle-created) scour the road and transport pollutants away from vehicle lanes and the highway (Asplund, 1980). Supporting this conclusion are studies, which have shown that the majority of pollutants are located within 3 feet (0.9m) of the curb (Laxen and Harrison, 1977, and Little and Wiffen, 1978).

During periods of wet weather the primary removal mechanism is stormwater runoff (Asplund, 1980). Removal may occur by other means as well. Chui et al. (1981) state that for low-intensity storms, a significant amount of pollutants accumulate on vehicles themselves.

The remainder of this report will concentrate on the pollutant loads and concentrations in stormwater runoff. The effect of the accumulation of pollutants on highway medians and shoulders will be considered only as it impacts runoff quality.

Stormwater runoff from highways may contain many constituents including solids, metals, nutrients, and hydrocarbons. Concentration and loading of highway runoff constituents have been reported in several studies, and the data from individual reports are included in Appendix A. A summary of these data, including the range of averages for each pollutant, is presented in Table 3.1. Because these values are averages, they do not reflect the maximum and minimum concentrations reported in the studies in Appendix A.

The averages may be for a particular site, or may represent an average of several sites examined in one study. Concentrations are reported in mass per volume of runoff, but loads are reported in several forms: mass/area/time, mass/area/event, mass/length of road/time, mass/length of road/number of vehicles, and mass/area/depth of runoff. The first two formats are the most prevalent and are the only two listed in Table 3.1.

To explain the wide range of values in Table 3.1, several factors must be considered, including the processes involved in the deposition, transport, and transformation of the pollutants.

Table 3.1
Constituents of Highway Runoff
Ranges of Average Values Reported in the Literature

Constituent	Concentration (mg/L unless indicated)	Load (kg/ha/year)	Load (kg/ha/event)
SOLIDS			
Total	437 - 1147		58.2
Dissolved	356	148	
Suspended	45 - 798	314 - 11,862	1.84 - 107.6
Volatile, dissolved	131		
Volatile, suspended	4.3 - 79	45 - 961	.89 - 28.4
Volatile, total	57 - 242	179 - 2518	10.5
METALS (totals)			
Zn	.056 - .929	.22 - 10.40	.004 - .025
Cd	ND - .04	.0072 - .037	.002
As	.058		
Ni	.053	.07	
Cu	.022 - 7.033	.030 - 4.67	.0063
Fe	2.429 - 10.3	4.37 - 28.81	.56
Pb	.073 - 1.78	.08 - 21.2	.008 - .22
Cr	ND - .04	.012 - 0.10	.0031
Mg	1.062		
Hg, X10-3	3.22	.007	.0007
NUTRIENTS			
Ammonia, total as N	.07 - .22	1.03 - 4.60	
Nitrite, total as N	.013 - .25		
Nitrate, total as N	.306 - 1.4		
Nitrite + nitrate	0.15 - 1.636	.8 - 8.00	.078
Organic, total as N	.965 - 2.3		
TKN	0.335 - 55.0	1.66 - 31.95	.17
Nitrogen, total as N	4.1	9.80	.02 - .32
Phosphorus, total as P	.113 - 0.998	.6 - 8.23	
MISCELLANEOUS			
Total coliforms organisms/100ml	570 - 6200		
Fecal coliforms organisms/100ml	50 - 590		
Sodium		1.95	
Chloride		4.63 - 1344	
pH	7.1 - 7.2		

Table 3.1 continued

Constituent	Concentration (mg/L unless indicated)	Load (kg/ha/year)	Load (kg/ha/event)
Total Org. Carbon	24 - 77	31.3 - 342.1	.88 - 2.35
Chemical Oxygen Demand	14.7 - 272	128 - 3868	2.90 - 66.9
Biological Oxygen Demand (five day)	12.7 - 37	30.60 - 164	0.98
Polyaromatic Hydrocarbon (PAH)		.005 - .018	
Oil and Grease	2.7 - 27	4.85 - 767	.09 - .16
Specific conductance (µmhos/cm at 25 C)	337-500		
Turbidity (JTU)	84 - 127		
Turbidity (NTU)	19		

Customary units: To convert kg/ha/yr to lb/ac/yr multiply by 0.87
To convert kg/ha/event to lb/ac/event multiply by 0.87

3.1 Traffic Volume

Vehicles are one of the major sources of pollutants in highway runoff; therefore, the amount of traffic on a given stretch of highway will influence the accumulation of pollutants on the highway surface. However, vehicle turbulence also can remove solids and other pollutants from highway lanes and shoulders (Kerri et al., 1985, and Asplund et al., 1980), obscuring the relationship between traffic volume and pollutant loads or concentrations in runoff. Furthermore, there are two measures of traffic volume which must be considered: average daily traffic (ADT) and vehicles during a storm (VDS).

The results of several reports indicate that the relationship between ADT and the quality of stormwater runoff is unclear. Dorman et al. (1988) found that ADT greatly influences runoff pollutant levels, but Horner et al. (1979) found a weak correlation between TSS concentrations and ADT, while Bourcier et al. (1980) found no correlation with metal loadings. Mixed results also were found by comparing runoff concentrations from highways of different traffic densities in studies by McKenzie and Irwin (1983), Irwin and Losey (1978), and Wanielista (1980). ADT was not related to concentrations of suspended solids, nitrogen, or phosphors; ADT was somewhat related to concentrations of lead and zinc; and ADT was strongly related to COD.

These results possibly are explained by the work of Driscoll et al. (1990b). Runoff concentrations are two to four times higher from high-traffic sites (ADT > 30,000) than from low-traffic (ADT < 30,000) sites (Table 3.2). Regression analyses of the data from the high-traffic sites produced weak correlations between ADT and concentrations of the pollutants.

Table 3.2
Pollutant Concentrations in Highway Runoff
Site Median Concentrations (mg/L)
(Driscoll et al., 1990b)

Pollutant	Urban Highways ADT > 30,000	Rural Highways ADT < 30,000
TSS	142	41
VSS	39	12
TOC	25	8
COD	114	49
nitrate plus nitrite	0.76	.46
TKN	1.83	.87
PO4	.4	.16
copper	.054	.022
lead	.4	.08
zinc	.329	.08

ADT seems to be significant on a broad scale only, and site-by-site variations are attributable to many factors, not just ADT. ADT might be most important as an indicator of land-use of the surrounding area, e.g., "urban" versus "non-urban." Stotz (1987) concludes that "the amount of pollutants discharged is not dependent on the traffic frequency, but much more on the characteristics of the area." Mar et al. (1982) reached a similar conclusion.

While surrounding land use may very well be more important than ADT, VDS seems to be a very important factor in the determination of pollutant loads. Kerri et al. (1985) found that VDS is better than either ADT or the antecedent dry period as an independent variable used to predict loads of lead, zinc, COD, TKN, and filterable residue.

A linear regression of cumulative TSS loads versus cumulative VDS for several sites in the state of Washington also disclosed a strong relationship (Chui et al., 1981). The slopes of the regression lines (pounds of TSS per curb-mile per 1000 VDS) for each site varied from 3.21 to 46.76 (0.91 to 13.3 kg per unit-km). This wide range can be attributed to differences in surrounding land use, differences in precipitation patterns throughout the state, ashfall from the eruption of Mount St. Helens volcano, and varied applications of deicing materials.

The importance of VDS has also been demonstrated in studies by Chui et al. (1982), Asplund et al. (1982), and Horner and Mar (1983). In the Asplund study, volume of traffic during dry periods had no effect on solids loadings in the runoff.

It might be expected that the type of traffic would also have a major effect on the type and volumes of pollutants from highways, but no studies have been identified which address this aspect directly.

3.2 Precipitation Characteristics

Three characteristics of a storm event which may be relevant to the ensuing water quality of runoff from a highway surface are: the number of dry days preceding the event, the intensity of the storm, and the duration of the storm. Several studies have attempted to determine the importance of these factors.

The number of antecedent dry days before an event effects the runoff quantity from highways (Kent et al., 1982, and Lord, 1987), but the evidence pertaining to runoff quality is mixed. Howell (1978a) found a relationship between solids build-up on highway surfaces and the duration of dry weather. Using correlation analysis, Hewitt and Rashed (1992) found an association between the antecedent dry period (ADP) and mean concentrations of dissolved lead, dissolved copper, and particulate-phase lead (significant at the 5% level). However, no correlation for the concentrations of dissolved Cd, particulate-phase Cu, particulate-phase Cd, or individual polyaromatic hydrocarbons (PAH) concentrations was evident. PAH compounds are believed to be lost from the highway surface by volatilization, photo-oxidation, or other oxidation processes.

Other reported studies have found that ADP is relatively unimportant. For example, Horner et al. (1979) found that the correlation was not strong enough to predict TSS loadings from ADP. Kerri (1985) determined ADP to be a poor independent variable for the predictions of Pb, Zn, COD, TKN, or filterable residue. Harrison and Wilson (1985a) did not find a correlation between ADP and peak lead concentration (Table 3.3), although the negative correlation with discharge in the previous 24 hours reflects the role of runoff in cleansing the road surface.

Table 3.3
Results of Correlation Analysis for Peak Pb Concentrations in Runoff Water (Harrison and Wilson, 1985a)

Relationship	Spearman rank correlation coefficient (r_S)	Significance level
Peak Pb concentration and length of preceding dry period	-0.075	
Peak Pb concentration and peak runoff rate	0.63	0.05
Peak Pb concentration and runoff discharge in previous 24 h	-0.58	0.05
Peak Pb concentration and rate of increase in runoff discharge over the time to peak discharge	0.66	0.025

From these results it can be infered that the duration between storms is an important factor only for short duration periods. Removal processes such as air turbulence (natural or the result of vehicles) and volatilization, photo-oxidation or other oxidation processes, limit the accumulation of solids and other pollutants on road surfaces, thereby decreasing the importance of dry periods between storms.

The intensity of the storm can have a marked impact on the type and quantity of pollutants in runoff. This is due in large part to the fact that many pollutants are associated with particles, which are more easily mobilized in high-intensity storms.

Metals are predominantly washed from highways after adsorption upon particulate materials such as bituminous road surface wear products, rubber from tires, and particles coated with oils. The degree of association with solids varies between different metals. Gupta et al. (1981c) found dissolved metal fractions in runoff were small for lead, zinc, and iron (Table 3.4). Lead values in particular were small and often below detectable limits of 0.05 mg/L. Metal loadings were tested for statistical correlation with solids loadings. Lead was significantly correlated with solids at a 99% confidence limit for six out of six sites. Zinc, iron, and cadmium were correlated at five of the six sites, copper and chromium at four sites, and mercury at only one (Gupta et al., 1981c).

Table 3.4
Concentration of Total and Dissolved Lead, Zinc, and Iron at Various Sites (Gupta et al., 1981c)

Site	Storm No.	Type of Sample	Lead (mg/L) Total	Lead (mg/L) Dissolved	Zinc (mg/L) Total	Zinc (mg/L) Dissolved	Iron (mg/L) Total	Iron (mg/L) Dissolved
I-794 Milwaukee	11	Composite	13.1	<0.05	3.4	0.21	43.0	0.03
		Discrete	160.0	<0.05	25.0	0.58	39.0	0.48
		Discrete	17.0	<0.05	3.3	0.20	52.0	0.09
		Discrete	2.5	<0.05	0.8	0.31	10.0	0.08
		Discrete	0.2	<0.05	0.1	0.09	0.4	0.07
Hwy. 45 Milwaukee	17	Composite	6.6	<0.05	1.9	0.36	35.0	0.11
		Discrete	8.6	<0.05	2.8	0.25	43.0	0.12
		Discrete	9.3	<0.05	3.0	0.29	51.0	0.14
		Discrete	2.3	<0.05	1.2	0.48	14.0	0.20
Hwy. 45 Milwaukee	18	Composite	2.2	<0.05	0.94	0.39	15.0	0.23
		Discrete	6.5	<0.05	2.35	0.33	39.0	0.13
		Discrete	6.4	<0.05	2.00	0.37	34.0	0.24
		Discrete	0.1	<0.05	0.35	0.34	1.1	0.16
Grassy Site Milwaukee	01	Composite	<0.05	<0.05	0.14	0.08	2.5	0.25
		Discrete	0.20	<0.10	0.16	0.08	3.6	0.19
		Discrete	0.40	<0.10	-	-	2.1	0.20
		Discrete	<0.10	<0.10	-	-	1.5	0.15
I-81 Harrisburg	15	Composite	<0.05	<0.05	0.15	0.02	6.6	0.13
		Discrete	<0.05	<0.05	0.09	0.08	1.9	0.05
I-40 Nashville	03	Discrete	2.0	<0.05	1.10	0.20	27.0	0.05
I-40 Nashville	04	Composite	0.5	<0.05	0.36	0.03	6.3	0.34
		Discrete	2.2	<0.10	1.30	0.16	32.0	0.43
		Discrete	0.8	<0.10	0.40	0.14	7.9	0.04
		Discrete	0.3	<0.10	0.19	0.01	3.7	0.05

Lead is also the metal most associated with particulates (> 0.45 μm (> 1.77 x 10^{-5} in)) in the work of Hewitt and Rashed (1992). The particulate fractions for lead, copper, and cadmium were respectively 90%, 75%, and 57%. Using geometric regression, Harrison and Wilson (1985a) found significant correlations between suspended solids and lead and between suspended solids and copper. Correlations between suspended solids and other metals, manganese, cadmium, and iron were not significant at the 0.1 level. A study of runoff concentrations of titanium and tungsten revealed no dissolved-colloidal fractions for either metal above the detectable limit of 2.5 mg/L (Bourcier et al., 1980). A linear relationship was observed between total solids and the individual metal concentrations, with titanium showing a higher correlation than tungsten.

The size of the particulates is very important to the transport of the associated pollutants. Finer grains have lower settling velocities and remain in runoff longer than larger grains. Harrison and Wilson (1985a) studied and discussed the topic of metals at length. Different metal concentrations are associated with different-sized particulate materials. Therefore, lead and iron do not follow the suspended sediment profiles exactly. Most metals are found in the finer street dust (<43 μm (1.7 x 10^{-5} in)): although fine street dusts constitute 6% of the total mass of solids, they contain more than 50% of the trace metals. Because storms and runoff can more easily mobilize the smaller grains, metal concentrations in suspended sediment are generally higher than in street dust.

Other pollutants found primarily associated with the particulate phase and/or showing a strong correlation with solids include PAH's, TOC, COD, and extractable organics. Although particulate and dissolved phase concentrations of PAH's were not separately determined by Hewitt and Rashed (1992), the similarity between concentration profiles of PAH's and suspended solids implies that all of the PAH compounds tend to be adsorbed to particles. Chui et al. (1981) found high degrees of correlation ($R > 0.83$ in all but one case) between solids and TOC and COD regardless of traffic or weather conditions.

Table 3.5 (Zawlocki et al., 1980) shows the phase concentrations in highway runoff for nine categories of extractable organics. For all three major classes (aromatic compounds, aliphatic hydrocarbons, and oxygenated) and all three storms, the majority of pollutants are found in the particulate phase. The lowest particulate/total ratio is 54% for aliphatic hydrocarbons during the first storm. Most of the ratios are over 90%.

Table 3.5
Extractable Organics in Runoff Classified into Nine Categories
(Zawlocki et al., 1980)

Class of Compounds	Storm 1-5-87 Particulate (µg/L)	Storm 1-5-87 Soluble (µg/L)	Storm 1-5-87 Total (µg/L)	Storm 1-5-131 Particulate (µg/L)	Storm 1-5-131 Soluble (µg/L)	Storm 1-5-131 Total (µg/L)	Storm 520-43 Particulate (µg/L)	Storm 520-43 Soluble (µg/L)	Storm 520-43 Total (µg/L)
Alcohols	T	160	160	478	155	633	327	126	453
Aliphatic Hydrocarbons	3710	3220	6930	1850	636	2490	913	20	933
Aromatic Compounds Including Heterocyclics	2050	148	2200	297	96	393	596	128	724
Halogenated Organics	T	T	T	175	9	82	114	T	114
Ketones and Aldehydes	1130	T	1130	87	39	126	385	128	513
Organosulphur Compounds	1200	62	1260	T	5.3	5.3	4.9	T	4.9
Oxygenates excluding alcohols, phenolics, ketones/aldehydes	3170	410	3580	3510	228	3740	2440	126	2570
Nitrogen containing Compounds	1420	62	1480	87	28	115	325	135	460
Phenolics	2830	80	2910	T	2.9	2.9	T	T	T
Total Chromatographed	10236	6803	17039	6308	1204	7512	4083	320	4403

Table 3.3 shows a positive correlation between Pb concentration and two factors: peak runoff rate and the increase in runoff discharge during the rising limb of the hydrograph. These factors are indicators of high-intensity storms. Such storms exhibit the vigorous flushing required to remove the mostly particle-associated lead. These results are supported by the work of Hoffman et al. (1985) who graphed rainfall intensity versus loads of hydrocarbons, lead, and suspended solids (Figure 3.1). Runoff concentrations for all of the parameters and loading rates generally followed the trend of rainfall intensity.

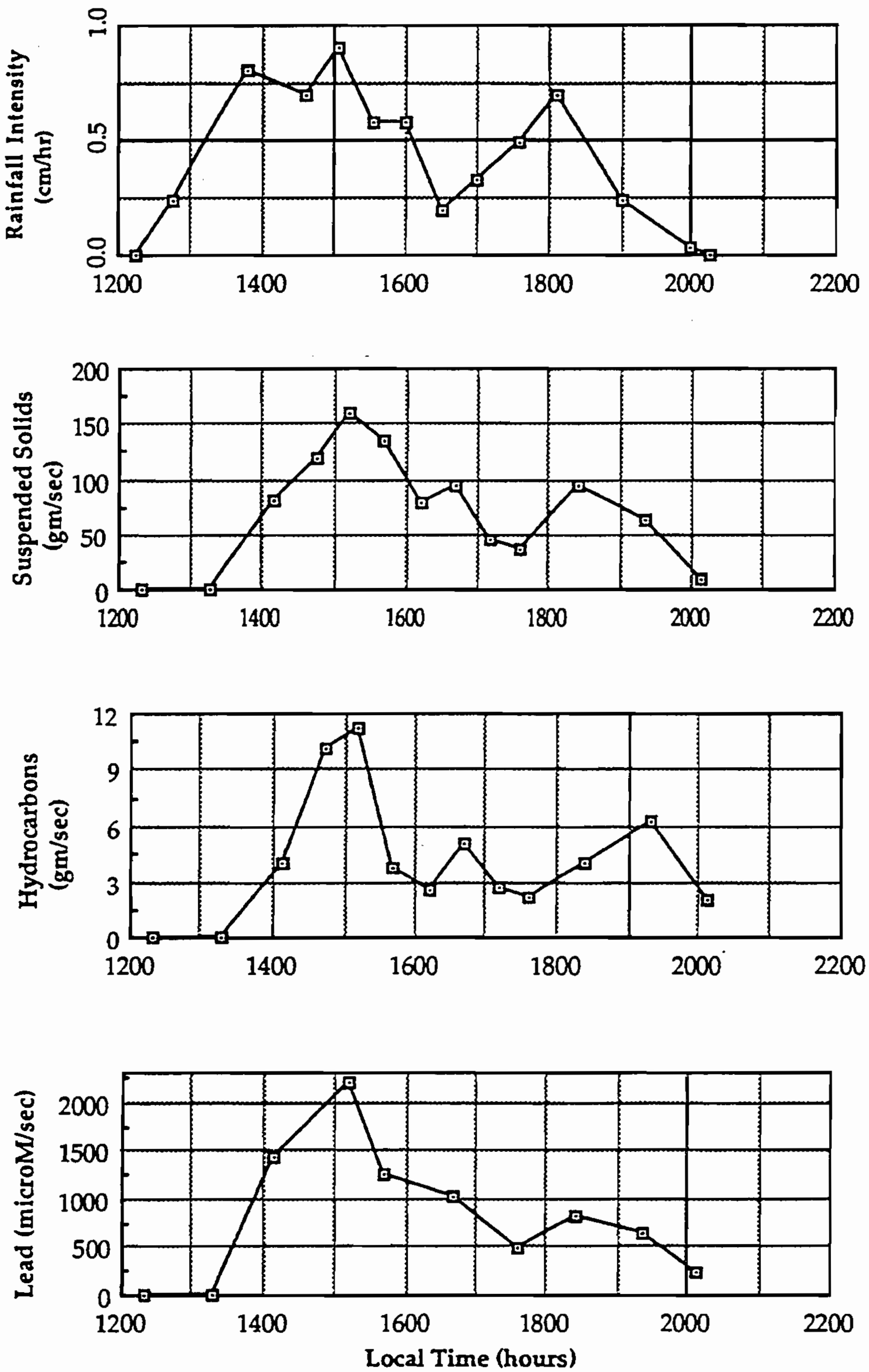
The positive correlation between loading rates and storm intensity was also shown in the work of Horner et al. (1990a). Seven storms were monitored during the winter of 1988-89, and intensities and loading rates were recorded. The ranges of loadings of the two lowest-intensity storms (0.026 and 0.020 inch/hour (0.66 and 0.51 mm/hr)) and the two highest-intensity storms (0.119 and 0.114 inch/hour (3.0 and 2.9 mm/hr)) are compared in Table 3.6. The upper range for all constituents of the higher-intensity storms was 2-3 orders of magnitude above the upper range of the less intense storms.

Table 3.6
Ranges of Loadings (1988-89 Storms)
(Horner et al., 1990)

Storm Type	TSS (mg/h)	VSS (mg/h)	TP (mg/h)	COD (mg/h)	Total Cu (μg/h)	Total Pb (μg/h)	Total Zn (μg/h)
low intensity	7-35,726	2-1631	0.4-31.1	0-920	3-2178	0-354	31-3516
high intensity	436-$14x10^6$	136-322,704	5.8-10,332.2	0-195,969	121-362,529	0-175,472	343-571,527

There was less correlation between storm intensity and loading rates for titanium and tungsten, two metals associated with studded tires (Bourcier et al., 1980). The storm considered by Bourcier was of relatively low intensity (< 0.08 inch/hour (2.0 mm/hr)) throughout its duration, so the apparent lack of association may be due to the small variation in rainfall intensity.

Nutrients are more likely than metals, PAH's, TOC, COD, or extractable organics to be found in the dissolved rather than in the particulate phase. Gupta et al. (1981c) found significant correlations between total solids and TKN at only



Customary units: To convert cm/hr to in/hr multiply by 0.39
To convert gm/sec to lb/sec multiply by 0.0022

Figure 3.1 Rainfall Intensity and Pollutant Volumes
(Hoffman et al., 1985)

two out of six sites. The parameter nitrate plus nitrite showed very weak correlation with TSS in highway runoff (Chui et al., 1981). Approximately 79% of organic phosphorus was found in the dissolved phase (Hvitved-Jacobsen et al., 1984).

Major ions also are found predominantly in dissolved form. Harrison and Wilson (1985a) found that particulate fractions (< 0.45 μm (< 1.77 x 10^{-5} in)) often constituted less than 1% of total concentrations of Na^{+}, Ca^{2+}, Cl^{-}, and SO_4^{2-}. Two ions (K^{+} and Mg^{2+}) were associated with particulate fractions, but their total concentrations were low (< 8 mg/L). Organic lead also tends to be primarily dissolved.

Higher concentrations of pollutants are often observed in the first runoff from a storm, and this is especially true for dissolved components. This phenomenon is commonly described as the "first flush," and has led many agencies to require retention and treatment of the first 1/2 inch (13 mm) of rainfall. Howell (1978a) reported higher concentrations of metals and nutrients during the initial 30 to 60 minutes of a runoff event. Horner et al. (1979) found concentrations to be higher in both magnitude and fluctuation during the first 30 to 60 minutes of a runoff event. Concentration profiles for TSS, VSS, and COD are presented in Figure 3.2. A "first flush" is evident for both measurements of solids.

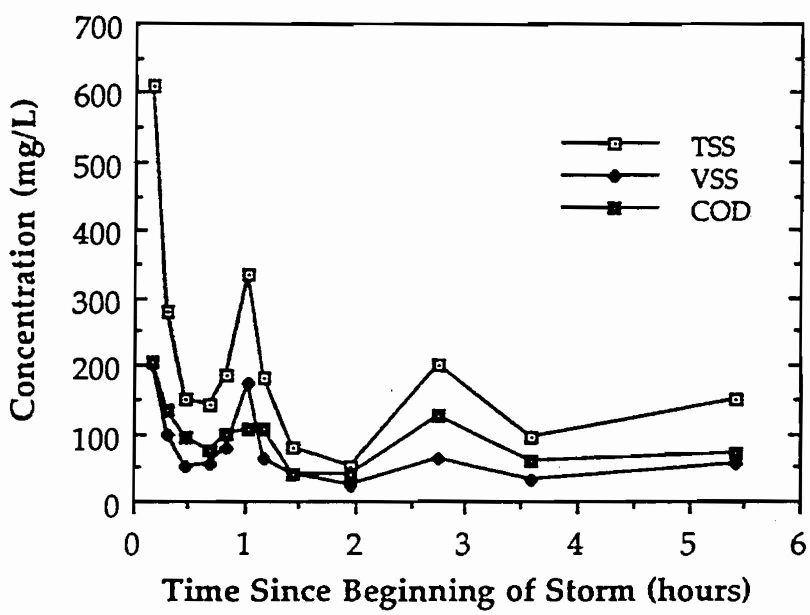


Figure 3.2 First Flush for Solids and COD (Horner et al., 1979)

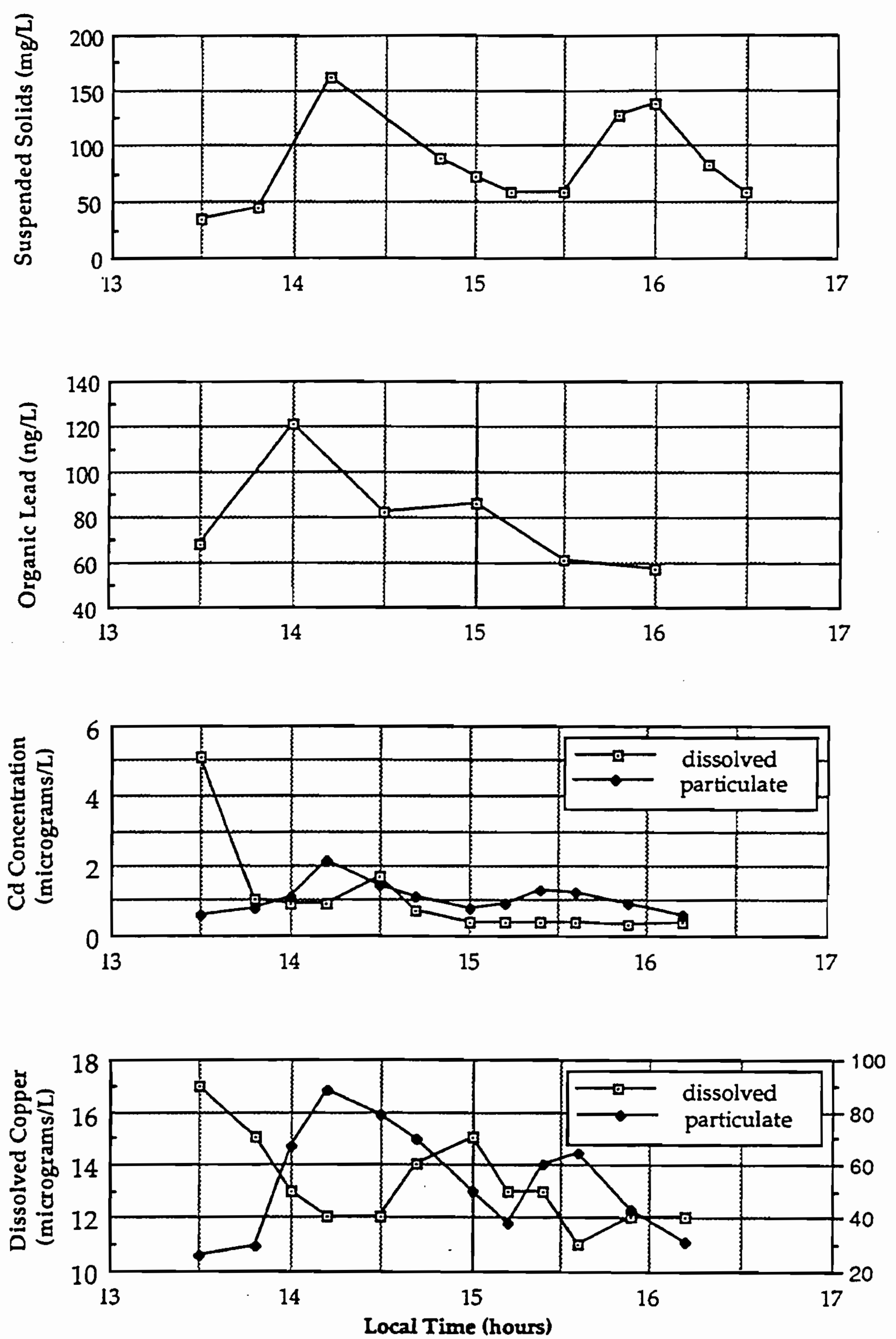


Figure 3.3 Concentration of Particles and Particulate-Associated Metals (Hewitt and Rashad, 1992)

Many studies have supported these findings only for dissolved constituents. Concentration profiles for particle-associated pollutants often display a discharge pattern more complex than a simple "first flush." In a study of a major rural highway in northwest England, Hewitt and Rashed (1992) observed a rapid, steady decline in concentrations of organic lead compounds, dissolved cadmium, and (to a lesser degree) dissolved copper (Figure 3.3); however, concentrations of suspended solids and particulate-associated metals fluctuated with the runoff hydrograph. For this report, "particulates" were defined as greater than 0.45 µm (1.77×10^{-5} in) in diameter and "dissolved" constituents as those which pass a 0.45 µm (1.77×10^{-5} in) in filter.

Harrison and Wilson (1985a) also reported a "first flush" for dissolved constituents, but found that the temporal variation of concentrations of particle-associated elements was more complex and related to rainfall intensity and the flushing of sediment through the drainage system. The work of McKenzie and Irwin (1983) produced similar results.

The third precipitation characteristic, the length of a storm and the ensuing runoff volume, also seems to have an unclear effects on runoff quality. Driscoll et al. (1990b) determined the correlation between runoff volume and eight pollutant concentrations using 184 paired data sets from 23 sites. They found that only 10% of the data sets were significantly correlated at the 95% confidence level, and only 15% were significantly correlated at the 90% confidence level. Furthermore, even for the few sets with significant correlation, the correlations are weak, i.e., on average they explain about 20% of the concentration variability.

Storm duration can be a significant factor if highway runoff is not completely isolated. In the work of Dorman et al. (1988), concentrations of runoff pollutants were greater during shorter, low-volume storms in which there was no runoff from unpaved areas. Larger storms dilute the highway runoff and lower the concentrations of most pollutants with runoff from unpaved areas.

Even though concentrations are lower, loadings of pollutants are generally greater from longer storms, as the transport of at least some constituents continues throughout the duration of the event. Many solids and other pollutants which accumulate on the pavement and in the gutter between storms are quickly washed off, but vehicles and atmospheric fallout continue to release pollutant constituents (Kerri et al., 1985).

3.3 Highway Surface Type and Drainage Mechanisms

The type of highway paving materials may also effect the amount of pollutants in highway runoff. Wiland and Malina (1976) measured several parameters in runoff from two highways near Austin, Texas. The data presented in Tables 3.7 and 3.8 show that concentrations and loadings of COD, TOC, lead, and zinc were greater from the asphalt surface (MoPac) than from the concrete surface (IH 35), even though traffic flow was 160% higher on IH 35. Oil and grease and TSS concentrations and loadings were higher from the concrete surface, which Wiland and Malina (1976) attribute to higher traffic flow, higher abrasiveness of concrete surfaces, and/or guard walls on IH 35 preventing removal of solids by wind.

Gupta et al. (1981c) in a study in Denver, Colorado, determined that oil and grease loadings were highest from an asphalt-paved surface, but concluded that land use was the most important factor in determining runoff quality. Driscoll et al. (1990b) also reported that highway surface type was unimportant compared to other factors.

Table 3.7
Average Concentrations
(Wiland and Malina, 1976)

Site	Date	Oil and Grease mg/L	COD mg/L	TOC mg/L	Lead µg/L	Zinc µg/L	TSS mg/L
IH 35 (concrete)	Feb. 11	6.8	117	18	229	75	–
	Mar. 15	3.2	169	9	98	60	–
	Apr. 22	2.1	76	17	137	95	–
	Jun. 16	3.3	128	28	366	165	79
	July 21	4.3	92	23	308	75	80
	Aug. 6	7.8	167	47	423	278	171
	Average	4.6	125	24	259	125	110
MoPac (Asphalt)	Feb. 11	3.4	266	66	581	406	–
	Mar. 15	2.3	205	22	113	62	–
	Apr. 22	1.4	116	30	101	83	–
	Jun. 16	2.2	280	70	527	362	44
	July 21	2.1	104	35	219	104	40
	Aug. 6	4.6	568	141	1098	970	50
	Average	2.7	257	61	440	331	45

Table 3.8
Mass Loadings
(Wiland and Malina, 1976)

Site	Date	Oil and Grease mg/sq ft	COD mg/sq ft	TOC mg/sq ft	Lead µg/sq ft	Zinc µg/sq ft	TSS mg/sq ft
IH 35	Feb. 11	5.4	92	14	181	59	–
	Mar. 15	7.3	387	20	224	137	–
	Apr. 22	4.8	175	39	303	218	–
	Jun. 16	7.6	295	64	839	378	181
	July 21	6.0	130	33	436	106	113
	Aug. 6	11.9	255	72	646	424	261
	Average	7.2	222	40	438	220	185
MoPac	Feb. 11	2.4	187	47	407	285	–
	Mar. 15	2.9	259	27	142	79	–
	Apr. 22	1.7	147	38	127	105	–
	Jun. 16	2.8	353	89	664	457	56
	July 21	4.5	225	76	475	226	86
	Aug. 6	10.0	1229	304	2374	2098	108
	Average	4.1	400	97	698	542	83

Customary units: To convert mg/sq ft to lb/sq•ft multiply by 2.2 x 10^{-6}
To convert mg/sq ft to lb/sq•ft multiply by 2.2 x 10^{-9}

SI units: To convert mg/sq•ft to mg/m^2 multiply by 10.76
To convert mg/sq•ft to mg/m^2 multiply by 10.76

Stotz (1987) investigated differences in paving materials in a study of three German highways, but found that drainage methods were more important than pavement type in determining the quality and quantity of highway runoff. Two of the highways had an impermeable system of curbs and stormwater sewers exclusively; on the third site (Ulm-West), three-quarters of the catchment area was drained through grass-covered ditches. Although traffic loads and precipitation values at the three sites were similar, runoff volumes were quite different. Runoff at the Ulm-West site was 178 cubic meters/hectare/month (2500 ft^3/ac/month) compared to values of 599 and 358 (8,400 and 5000 ft^3/ac/month) from the other highways.

Yearly pollutant loads are shown in (Table 3.9) and indicate that the loads were smaller for the Ulm-West highway for all of the constituents except

Table 3.9
Yearly Pollution Loads in Highway Surface Runoffs (in kg/hectare)
(Stotz, 1987)

	A81 Pleidelsheim	A6 Obereisesheim	A 8/B 10 Ulm-West
Paving Material	concrete	asphalt	asphalt
% of Drainage Area Paved	100	86	40
Filterable solids	873	848	479
COD	672	557	207
Mineral Oil	43.27	27.09	4.85
PAH	0.018	0.014	0.005
Cd	0.037	0.029	0.0072
Cr	0.062	0.100	0.012
Cu	0.621	0.544	0.130
Fe	23.37	28.81	4.37
Pb	1.332	1.155	0.360
Zn	2.329	2.892	0.715
C1	1011	777	1344
Ammonia	4.60	3.22	1.03
TP	1.62	1.45	0.63

Customary units: To convert kg/hectanc to lbal/ac multiply by 0.87

chlorides. The higher chloride concentrations were most likely due to salt in thawing ice. Loadings from the other two highways were very similar, even though one highway was concrete (Pleidelsheim) and one was asphalt (Obereisesheim).

The type of drainage from a highway bridge influences the runoff quality and subsequent effect on the receiving water. Yousef et al. (1982) compared concentrations of lead, zinc, chromium, nickel, copper, and iron in the sediments beneath highway bridges with and without scupper drains. On the bridge without scuppers, water drains toward the adjacent land on either side of the lake. The data show that sediment metal concentrations were two to three times greater beneath the bridge with scuppers.

3.4 Site-Specific and Seasonal Considerations

Ranges of pollutant concentrations and loads also can be attributed to site-specific conditions or seasonal variations. Excess solid loadings have been connected to a variety of sources: the eruption of Mount St. Helens (Asplund et al., 1980, and Chui et al., 1981), the use of studded tires (Chui, 1981), and the use of salt or sand for deicing (Gupta et al., 1981c, and Harrison and Wilson, 1985a). Deicing is an important source of constituents other than solids. Kobriger and Geinopolos (1984) found high correlations between metals, especially lead, zinc, iron, and saltating particulates. Increase in chloride concentration and conductivity due to the application of road salt on highways was also discussed and measured by Stotz (1987) and Besha et al. (1983).

Other site-specific or seasonal factors can outweigh the impact of sanding. Horner et al. (1979) found spring loads to be greater than winter loads (despite sanding in winter) due to the less frequent rains and the increased construction activity in the spring. The use of studded tires in winter results in higher loads of tungsten and titanium (Bourcier et al., 1980). High zinc concentration has been attributed to the proximity of a smelter (Driscoll et al., 1990b, and Chui et al., 1981).

4.0 ENVIRONMENTAL EFFECTS ON RECEIVING WATERS

The type and size of the receiving body, the potential for dispersion, the size of the catchment area, and the biological diversity of the receiving water ecosystem are just some of the factors which determine the extent and importance of highway runoff effects.

Hydrological effects of highways are highly site-specific. The extent of increased storm runoff volumes and peak discharges due to increased impervious cover depends on the relative sizes of highway right-of-way and total watershed area. Most highway projects are not large enough to create significant downstream flooding. A more likely problem is increased stream bank erosion due to the increased peak flows (Dupuis and Kobriger, 1985c). Methods of predicting the hydrologic effects of highways bridges and encroachments on surface waters were presented by Richardson et al. (1974).

Highways also may cause hydrogeologic effects (Parizek, 1971). These effects include the beheading of aquifers, the development of groundwater drains where cuts extend below the water table, changes in ground and surface water divides and basin areas, obstruction of groundwater flows by abutments, retaining walls, and sheet pilings, and changes in runoff and recharge characteristics.

Like hydrological effects, water quality effects of highway runoff are site-specific. Different types of water bodies react differently to the loading of pollutants. The processes controlling the transport and fate of pollutants in lakes and reservoirs differ from those in rivers, streams, and aquifers. The type of receiving water, the related dispersion characteristics, and the relative size determine the amount of dilution of highway runoff and related pollutants.

Seasonal variations in the water quality of both lentic and lotic systems can influence the impact of highway runoff (Dupuis and Kobriger, 1985c), and the size of a receiving water is also a consideration. Lange (1990) theorized that highway runoff is a problem for watercourses with catchment areas less than 5 km^2 (2 mi^2), but can be discounted for watercourses with catchment areas greater than 20 km^2 (7.8 mi^2).

The potential impacts of various pollutants have been discussed by Dupuis and Kobriger (1985c), Dorman et al. (1988), and McKenzie and Irwin (1983). Particulates and sediment in runoff also can cause problems by decreasing flow capacity in drainage ways, reducing storage volume in ponds and lakes, smothering benthic organisms, decreasing water clarity, and interfering with the respiration of small fish. Furthermore, toxic materials often are sorbed to and are transported by suspended solids. These toxins include metals, hydrocarbons, chlorinated pesticides, and PCB's, and present acute and chronic threats to receiving water organisms.

Researchers generally agree that nutrients (various forms of nitrogen and phosphorus) are a concern because of the long-term potential for eutrophication and the short-term problem of "shock-loading." Oxygen-demanding materials (measured by COD or BOD) can be relatively high in concentration, although the organics are usually particulate-associated and may settle rapidly before the demand can be exerted. Furthermore, DO depletion can be compensated by reaeration during stormflow periods.

Relatively high levels of pathogenic bacteria of non-human origin can be detected in runoff from highways, which routinely are used to haul livestock and/or are subjected to large amounts of bird droppings.

Effects of stormwater runoff from highways have been studied and quantified for three categories of receiving waters: streams, rivers, and lakes; wetlands; and groundwater and soil-water.

4.1 Streams, Rivers, and Lakes

Metal loadings to receiving waters are of particular concern due to the potential toxicity and relative abundance of metals in highway runoff. Yousef et al. (1982) found significant differences (often at greater than 95% confidence levels) between metal concentrations in Lake Lucien, a relatively undeveloped lake, and metal concentrations in a nearby highway runoff detention pond, near Orlando, Florida. The analysis included sampling of both the water column and bed sediments with pertinent data included in Tables 4.1 and 4.2. Some metal concentrations reached significantly higher levels in the detention pond. Lead concentrations in the water column were almost three times as high in the pond as lake concentrations, but total zinc concentrations in the pond were almost identical to those in Lake Lucien.

The effects of highway runoff were more dramatic in the sediments. Concentrations in pond sediments were 1.7 to 22 times higher than concentrations in lake sediments. Metal enrichment (2 to 4 times "normal" levels) also was observed in a small lake, catchment area of 2.4 km^2 (.94 mi^2), near Oslo, Norway, despite a traffic density below 20,000 ADT (Gjessing et al., 1984b).

Table 4.1
Significance of Differences in Heavy Metal Concentrations in Water Samples from Maitland Interchange
(Yousef et al., 1982)

Element	Total		Dissolved		Confidence Level (%)	
	Lake Lucien	West Pond	Lake Lucien	West Pond	Total	Dissolved
Zn	56	64	34	43	45.9	75.6
Pb	33	92	19	66	99.9	98.6
Cr	8.6	17	5.4	7	94.9	70.4
Ni	7.3	15	3.4	5	80.8	53.6
Ca	36	38	19	21	29.2	34.9
Fe	182	414	82	128	98.4	52.4

Average Concentrations in µg/L

Table 4.2
Significance of Differences in Heavy Metal Concentrations in Bottom Sediments from Maitland Interchange (t-test Analysis)
(Yousef et al., 1982)

Element	Lake Lucien	West Pond	Confidence Level (%)
Zn	21.1	35.2	80.27
Pb	3.4	76.0	97.51
Cr	2.5	33.9	98.87
Ni	1.2	10.7	97.64
Cu	5.0	15.2	93.17
Fe	421.4	3264.7	98.62
Cd	0.1	0.7	96.05

Mean Dry Weight (µg/g)

Dilution of metals concentrations was shown by Wanielista et al. (1980) by comparing concentrations in runoff from a highway bridge with scupper drains and concentrations in water samples taken from the lake itself (almost directly underneath the bridge). Table 4.3 shows a range of dilution ratios. Lead is diluted the most. The runoff concentration of lead is more than 20 times greater than the lake concentration. The fact that copper and chromium concentrations are greater in the lake indicates that other pollutant sources are present.

Table 4.3
Comparison Between Total Metal Concentrations in Bridge Runoff and Lake Ivanhoe Water Samples
(Wanielista et al., 1980)

	Zn	Pb	Ni	Fe	Cu	Cr	Cd	As
Average concentration in lake	104	75	15	192	74	14	4	57
Average concentration in runoff	498	1558	53	2427	52	11	5	58
Runoff/lake ratio	4.7	20.8	3.5	12.6	0.7	0.8	1.3	1.0

Concentrations in µg/L

Dupuis et al. (1985a) studied several highways with a wide range of traffic densities between 12,000 and 120,000 ADT. Lead was the only parameter in the water column even slightly affected by runoff with maximum concentrations at two of the three influenced stations in excess of 0.20 mg/L, while concentrations of lead at the control stations never exceeded 0.05 mg/L). Metal concentrations in the sediments showed little difference between the control and influenced stations.

Solids, pH, sulfate, turbidity, TOC, oil and grease, COD, nutrients, sodium, chloride, alkalinity, specific conductivity, calcium, indicator bacteria, and TKN concentrations also were measured in the sediments. TKN concentrations were higher for the highway-influenced stations at only one of the sites. During the August survey, concentrations at the control station were always below 1500 mg/kg dry weight, while concentrations at the influenced stations were almost always above 2000 mg/kg and ranged as high as 4000 mg/kg.

The toxic effects of metals in highway runoff can be greatly reduced by natural processes within the receiving water. Yousef et al. (1985a) described how ionic species of incoming trace metals are reduced by complexation. Lead often exists as $PbCO_3$ and much of the copper is associated with organic complexes. Most of the metal species in runoff eventually reside in the top few centimeters (top inch) of the sediment, and are unlikely to be released to the water column under aerobic conditions.

Bioassay tests of organisms in streams and lakes receiving highway runoff have yielded various results. Dupuis et al. (1985a) reported that the runoff from highways with various traffic densities, 12,000 to 120,000 ADT, had little effect on the biota of receiving waters. Flow-through in-situ bioassay studies at the lake site did not indicate an impact on six species of invertebrates. Bioassay testing included sampling for benthic macroinvertebrates and macrophytes. The data show that highway runoff had little or no influence on cattails. Similarly, toxicity tests on salmon, algae, bacteria, and fungi reported by Gjessing et al. (1984a) indicated that highway runoff had little effect on algae and no effect on fish or eggs; however, a stimulating effect on meterotrophes was detected.

The results reported in other studies indicate that highway-related pollutants, especially metals, were more threatening to nearby ecosystems. Portele et al. (1982) performed toxicity tests with highway runoff and dilution water. A dilution/runoff ratio of 100/1 is required to protect biota from heavy metals when traffic density exceeds 10,000 ADT. A 4/1 dilution/runoff ratio is required to avoid oxygen depletion. Soluble pollutants adversely affected algae and zooplankton, while suspended solids interfered with the respiration of rainbow trout fry. Adult fish showed no negative reaction to the runoff.

Stream sediments store heavy metals and are the primary source for the bioconcentration of metals (Van Hassel et al., 1980). Concentrations of lead, zinc, nickel, and cadmium in the water columns of streams near highways with low to moderate traffic volumes, around 15,000 ADT, were comparable to concentrations in uncontaminated waters. However, the dry weight concentrations of metals in benthic insects and fish were comparable to values reported in the literature for animals from contaminated waters. This suggests that the accumulation of metals in the bed sediments is important in the bioaccumulation of metals.

4.2 Wetlands

Schiffer (1988) studied the effects of highway runoff on two wetlands in central Florida. Highway runoff was compared to state water quality standards for alkalinity, trace metals, phosphate, total phosphorus, specific conductivity, pH, ammonia, ammonia plus organic nitrogen, nitrate plus nitrite, TOC, DO, and temperature. Lead concentrations exceeded state standards in 43% of the samples from one wetland inlet. In another wetland, lead exceeded state standards in only 19% of inlet samples, while zinc surpassed the standard in 81% of inlet samples.

Areal variations within the wetlands also were studied. Most constituent concentrations such as nutrients, aluminum, lead, zinc, specific conductivity, and pH decreased with distance from the inlets; however, color, TOC, and chromium concentrations sometimes increased with distance. Schiffer (1988) suggested possible explanations for the results obtained for chromium; this explanation was based on chromium remaining dissolved longer than other metals and/or atmospheric sources of chromium.

4.3 Groundwater and Soil-water

Researchers agree that highway runoff can have a significant impact on the hydrogeologic environment, including changes in water quality in the vadose and saturated zones. Many of theses changes are associated with stormwater runoff controls.

McKenzie and Irwin (1988) observed the effects of two types of runoff control devices on the water quality in an underlying surficial aquifer that is a major source of drinking water in southern Florida. The two types of controls evaluated were exfiltration trenches, and grassed swales, which received parking lot and roadway runoff. The exfiltration trench is essentially like an infiltration trench, except that the runoff collects first in an open trench, and then drains through a perforated pipe drain to an adjacent underground reservoir, filled with coarse aggregate, topped with pea gravel, filter fabric, and native soil on top. The reservoir is partially within the water table.

McKenzie and Irwin (1988) used the statistical non-parametric analysis of variance technique to evaluate the 11 water quality variables measured over a one-year period at the structures and at nearby groundwater monitoring wells. Results indicated that the exfiltration trenches did not adversely affect the groundwater. Lead and zinc apparently were trapped or removed by the exfiltration trenches. The swales showed signs of anaerobic conditions, most likely resulting from poor drainage conditions and high organic content soils. At the swale sites, groundwater measurements of ammonia nitrogen, iron, and dissolved solids were significantly greater, and groundwater measurements of sulfate were significantly less than those in the groundwater near the exfiltration trenches. The evidence of fairly active biological cycling associated with anaerobic conditions at the groundwater near the swales eliminated the possibility of evaluating the swales' influence on groundwater quality.

Yousef et al. (1986) discussed the potential for groundwater contamination near a highway runoff retention/detention pond. Heavy metal concentrations in the groundwater were compared to those in the pond. In general, mean concentrations of all heavy metals except copper were greater in groundwater beneath the pond than in the water within the pond. Reductions in the pH of the pond water, as a result of sediment accumulation, were believed to increase the release of metal ions into the groundwater. Yousef et al. (1986) recommend that pond sediments be removed every 10 to 20 years to reduce the leaching of metals.

Water quality in a well near the pond was compared to water quality in a well near a swale receiving highway runoff and also to water quality in a third control well. Iron, lead, and chromium concentrations beneath the swale were twice the concentration values beneath the pond and 3 to 10 times higher than concentrations in the control well (Yousef, 1986).

Zinc levels in groundwater can also be affected by highways. Schiffer (1988) monitored water quality in a surficial aquifer near a highway runoff detention pond and cypress wetland during dry (April) and wet (October) seasons. Samples from the wells closest to the highway exhibited higher concentrations of dissolved zinc than samples of concentrations from other wells and in the wetland. Zinc concentrations in wells near the highway were as high as 220 μg/L, whereas concentrations in the wells further from the highway were almost always below 50 μg/L and never above 100 μg/L. It is worth noting that lead concentrations were below the detection limit.

Highway runoff may contain constituents other than metals. Schiffer (1988) found that dissolved Kjeldahl nitrogen concentrations from wells closest to the highway were twice as high as concentrations from other wells and in the wetland. Organic compounds also may pose potential problems for groundwater quality (Schiffer, 1989). This particular study focused on inorganic constituents, and analyses of organic compounds were limited and qualitative.

Ku and Simmons (1986) measured concentrations of pollutants in groundwater below a recharge basin receiving stormwater runoff from a major highway. They found that in terms of the chemical and microbiological constituents of stormwater, there were no significant adverse effects on groundwater quality. Their data are presented in Table 4.4.

Schiffer (1989) measured the impacts of an infiltration basin, a detention pond/wetland system, two swales, and an exfiltration pipe on the surficial aquifer groundwater. The depth to the water table varied between sites: from 0.8 feet to 10 feet (0.24 to 3 meters) below the soil surface. Soils at all the locations were similar. Statistical differences between groundwater constituent background concentrations and groundwater constituent concentrations below each structure were assessed.

All of the methods were effective in removing metals and ions, but were less effective for nutrient removal. Nitrite plus nitrate and phosphorus concentrations were highest in groundwater near the swales and the exfiltration pipe, and Kjeldahl nitrogen was highest near the ponds. Turbidity and color were significantly higher in the groundwater near the ponds. Specific conductivity and pH were lowest beneath the swales. The pH was frequently below pH-6.5, a violation of drinking water standards, at both of the swales and at one of the ponds.

Lead, chromium, and copper were below detection levels (1 mg/L) near all of the structures. High iron concentrations near the swales (median value of 815 ppb) and near the detention pond (median value of 1700 ppb) were attributed to the surrounding soil type rather than to highway runoff. Potassium and sulfate concentrations were highest near the exfiltration pipe, but were still below drinking water standards. The nitrate standard (10 mg/L) was exceeded in only one sample near the exfiltration pipe. Schiffer (1989) reports that the organic compounds retained in pond sediments may present potential problems for groundwater quality.

The effects of highway runoff on groundwater are spatially dependent on local hydrological conditions, as well as on sorption processes within the aquifer. Transport of contaminants also is influenced by groundwater velocity. Yousef et al (1986) found limited effects at one site in Florida due to a groundwater velocity of only 10 meters per year. Furthermore, pollutants may be immobilized on the ground surface and in the vadose zone. Bell and Wanielista (1979) observed low concentrations of metals in groundwater near a highway, including lead concentrations at levels which did not exceed Florida water quality standards (<0.05 mg/kg). Their data indicate that heavy metals are retained by the soil, generally are immobilized, and hence do not leach downward.

Table 4.4
Median Values of Characteristics of Stormwater, Groundwater, and Precipitation, Recharge Basin, Plainview, New York
(Ku and Simmons, 1986)

Parameter	Surface Water	Groundwater	Precipitation
Turbidity (NTU)	20.0	0.4	0.5
Spec. Cond. (μmhos)	120	200	NA
Total Coliform (MPN)	24000	3	NA
Fecal Coliform (MPN)	640	3	NA
Fecal Strep. (MPN)	24000	3	NA
BOD, mg/L	10.0	2.5	NA
pH	6.9	6.6	7.1
Cadmium, diss., μg/L	1.0	1.0	0
Cadmium, susp., μg/L	0	0	0
Cadmium, tot., μg/L	1.0	1.0	1.0
Chromium, diss., μg/L	2.0	0.5	1.0
Chromium, susp., μg/L	15.0	7.0	6.0
Chromium, tot., μg/L	16.0	7.0	8.5
Lead, diss., μg/L	35.0	3.5	11.0
Lead, susp., μg/L	250	1.0	9.0
Lead, tot., μg/L	275	4.0	16.0
Potassium, diss., mg/L	2.3	1.6	0.2
Chloride, diss., mg/L	10.0	46.0	2.6
Sulfate, diss., mg/L	11.0	16.0	1.6
Fluoride, diss., mg/L	0.1	0.1	0.1
Arsenic, diss., μg/L	1.0	0	0
Arsenic, susp., μg/L	1.0	0.5	0
Arsenic, tot., μg/L	1.5	1.0	1.0
Phosphorous, diss., mg/L	0.05	0.01	0.02
Org. Carbon, diss., mg/L	6.9	2.4	1.8
Org. Carbon, susp., mg/L	7.3	0.9	1.6
Cyanide, tot., mg/L	ND	ND	NA
Calcium, diss., mg/L	7.85	9.0	0.4
Magnesium, diss., mg/L	1.2	3.6	0.2
Sodium, diss., mg/L	8.5	27.0	0.5
Nitrogen, $NH3$ + Org., mg/L	2.3	0.15	0.32
Nitrogen, $NO2$ +$NO3$, mg/L	0.49	0.82	0.28

These results indicate that where sufficient soil thickness is present, natural processes occurring in soils can attenuate pollutants in the runoff from highways prior to reaching the groundwater. On the other hand, Milligan and Betson (1985) studied an area with thin soils in a karst terrain, and suggest that urban stormwater runoff may have a significant impact on groundwater quality.

Because of the near surface immobilization of pollutants, highways may be a greater threat to soil-water than groundwater. Howie and Waller (1986) analyzed lithological material under highway swales, and found high concentrations of lead (1000-6600 μg/kg), iron (490-2400 μg/kg), and zinc (90-1800 μg/kg) in the top six inches (15 cm) of soil. Concentrations of this magnitude were not detected in lithological samples collected at an unaffected control site. Even with the concentrations of metals reported in the soil, no obvious impact of highway runoff on groundwater was found.

Bell and Wanielista (1979) sampled groundwater in the vicinity of several highways in east-central Florida in a study of overland flow and the deposition of highway-related heavy metals. The metals considered were lead, zinc, copper, chromium, nickel, and cadmium. Topsoils contained higher concentrations of metals than subsurface soils. This phenomenon was especially true for lead, which was less mobile in soil than other metals.

Kobriger and Geinopolos (1984) studied the sources and migration of highway runoff pollutants at four sites across the United States and found that metals are not the only pollutants immobilized during infiltration. Percolation to groundwater was one of the migration paths considered, and zero-tension lysimeters were used to measure the quantity and quality of water in the unsaturated zone. The data from this study are presented in Tables 4.5 and 4.6 for all sites. These data indicate an inverse relationship between distance from the highway pavement and sodium and chloride concentrations. Metals concentrations were also higher in near-highway samples, but to a lesser degree. Metals and sodium concentrations generally were higher in topsoil layers than in substrate layers, but the filtration process varied in effectiveness with different soil types.

Table 4.5
Lysimeter Water Quality Data (mg/L) - Milwaukee I-94 Site
(Kobriger and Geinopolos, 1984)

Parameter	Distance from the edge of the pavement					
	2 meters		12 meters		24.5 meters	
	Range	Mean median, m	Range	Mean median, m	Range	Mean median, m
pH	7.5-8.0	7.8	6.85-8.1	7.5	7.2-8.1	7.7
TS	733-3570	2150	218-440	326	308-418	379
TVS	--	NA	--	61	--	NA
SS	--	NA	10-98	33	3-11	7
Pb	0.2-3.5	1.5	ND-4.1	0.2	ND-0.1	ND
Zn	0.19-2.0	0.88	0.06-5.7	0.80	0.03-0.11	0.08
Fe	3.6-102	41.2	0.05-4.3	2.0	0.3-1.6	0.76
Cr	0.03-0.20	0.10	ND-0.06	0.03	0.01-0.02	0.02
Cu	0.49-2.2	1.15	0.06-0.43	0.16	0.01-0.04	0.03
Cd	ND-0.04	0.02	ND-0.08	ND	ND-0.02	ND
Ni	ND-0.2	ND	ND-0.20	ND	ND-0.1	ND
As	--	NA	--	ND	--	NA
Hg	--	NA	--	ND	--	NA
NO_2 +NO_3	--	NA	0.04-0.34	0.21	ND-0.04	0.02
TKN	--	NA	1.8-3.2	2.6	ND-2	1
PO_4	--	NA	0.21-1.93	0.60	ND-0.04	0.03
Sulfate	ND-110	ND		NA	18-26	24
Na	80-860	470	12-80	38	5-37	20
Cl	100-825	362	ND-272	20	42-125	87
Ca	14-98	56	2.8-38	14.5	21-78	59

Custom units: To convert meters to feet multiply by 0.30

ND = Not detectable.
NA = No analysis performed due to limited sample quantity.

Table 4.6
Lysimeter Water Quality Data (mg/L) - Harrisburg I-81 Site
(Kobriger and Geinopolos, 1984)

Parameter	Distance from the edge of the pavement							
	1.2 - 1.8 meters		2.1 meters		3.2 - 3.9 meters		13.7 meters	
	Range	Mean median, m	Range	Mean median, m	Range	Mean median, m	Range	Mean median, m
pH	5.6-8.4	6.6	6.95-9.1	7.0	5.90-8.0	7.5	5.6-7.8	6.3
TS	66-3550	587	918-1370	1140	101-3090	751	28-422	158
SS	6-1550	201	112-1040	574	40-2430	381	8-50	25
Pb	ND-0.6	0.1	ND-0.2	ND	ND-0.6	0.04	ND-0.50	ND
Zn	0.04-1.5	0.24	0.12-0.76	0.44	0.07-0.83	0.30	0.06-2.0	0.27
Fe	0.3-130	11.2	9.5-40.3	24.9	1.3-95.8	16.6	0.4-14	2.1
Cr	ND-0.07	0.01	--	0.08	ND-0.14	0.005	ND-0.07	ND
Cu	0.02-0.26	0.09	--	0.16	0.037-0.21	0.09	0.029-0.20	0.09
Cd	ND-0.05	ND	--	0.02	ND-0.02	0.004	ND-0.08	0.01
Ni	ND-0.3	ND	--	ND	ND-0.3	0.02	ND-0.2	ND
C1	6-2060	114	40-110	75	5-130	45	ND-41	10
NO_2+NO_3	0.46-27.5	7.0	--	NA	5.8-23.0	15.0	0.03-1.36	0.17
TKN	0.78-5.6	3.0	--	NA	2.62-3.60	3.01	ND-2.85	2.3
PO_4	0.77-3.20	1.59	--	NA	0.53-0.86	0.70	0.02-5	1.2
Na	3-1300	81	25.6-125	75.3	13.8-88	29.6	1.2-8.2	3.9
Ca	10-74	30	--	NA	--	NA	2-11	7
Hg	0.0005-0.0014	0.0010	--	NA	--	NA	--	0.0005
SO_4	ND-133	30	--	61	10-104	48	ND-22	7
O&G	2-3	3	--	NA	--	NA	--	NA

Customary units: To convert meters to feet multiply by 0.30

ND = Not detectable
NA = No analysis performed due to limited sample quantity

5.0 HIGHWAY CONSTRUCTION

5.1 Environmental Effects of Highway Construction

Highway construction effects on surface waters have been noted in the literature. Embler and Fletcher (1981) sampled streams for turbidity and suspended solids upstream and downstream of a construction site in Columbia, South Carolina. Stream measurements were taken before, during, and after construction. Rainfall was sampled as well. Peaks of turbidity and suspended solids concentrations were much greater after construction began. Turbidity never exceeded 25 NTUs during the preconstruction period, but after construction began turbidity peaks ranged from 50 to 80 NTUs. Suspended solids concentration remained below 30 mg/L prior to construction. The peak suspended solids concentration varied between 60 and 130 mg/L after construction began. Riley (1990) discussed some of the problems large volumes of sediment eroded from a highway construction site in Australia presented for discharge and sediment load sampling.

Yew and Makowski (1989) discussed an area along the Tennessee-North Carolina border where highway construction contributed to toxic conditions for fish in several area streams. Highway excavation exposed a pyritic shale material, which allowed leaching of the sulfides in the form of sulfuric acid. Analysis of the water quality data indicated that a combination of low pH (4.0 to 4.4) and alkalinity, along with increased toxic metal concentrations, contributed to the toxic conditions at these impacted sites. Temporary control measures included the addition of sodium hydroxide to the acidic streams. More permanent mitigation involved sealing the exposed pyritic material in the road embankments from surface water infiltration with lime and topsoil.

Streams near Richmondville, New York, were monitored two years prior to construction, during construction, and two years after completion of the construction (Besha et al., 1983). Little evidence of construction-related declines in water quality was observed. Rather, peak concentrations were the consequence of high rainfall rather than construction activity. Turbidity reached high levels in one of the creeks, but only infrequently and temporarily. Other variables experienced no detectable change due to construction activity. An increase in conductivity was attributed to the use of road salt.

The general destruction of wetlands is quantified by Hall and Naik (1989), and highway construction is listed as one of the principal activities contributing to this destruction. Highway construction has resulted in the loss of mangroves, seagrass, marshes, and swamps. The authors describe a number of highway construction sites and report on the effectiveness of the mitigation actions taken at each one.

Cramer and Hopkins (1982) focused on a Louisiana wetland in a study of construction impacts. Turbidity, color, salinity, DO, and pH were monitored before, during, and after construction for bridged highway construction techniques. The changes in salinity, DO, pH, and solids which occurred were attributed to factors other than construction. For example, changes in salinity at two sites could be traced to pollution from an automobile battery plant. However, the wetland did show an increase in turbidity and color resulting from construction activity. Turbidity began to decrease and color began to return to ambient conditions after conclusion of highway construction.

Crabtree et al. (1992) reviewed wetland mitigation efforts at 17 highway construction sites across the country. The mitigation efforts included restoration, creation, and enhancement of wetlands. They found that ineffective mitigation could usually be attributed to lack of attention to detail in the planning, design, and implementation processes. Effectiveness was improved through baseline monitoring, proper mitigation site selection, detailed mitigation plans, monitored construction activities, and post-construction remediation and monitoring.

Kobriger (1983) developed management guidelines which provide a general discussion on highway construction, design, and maintenance considerations for the preservation of wetlands. Heavy metals, deicing agents, hydrocarbons, pesticides, and fertilizers are the main constituents of highway runoff which can negatively affect wetland ecosystems.

Garton (1977) reported on an incident where highway construction had a very serious impact on groundwater quality. Construction of part of Interstate highway 79 in West Virginia uncovered pits and caverns overlying a karst aquifer which was the source of springs used as a water supply for the Bowden National Fish Hatchery. Large quantities of clay and silt were washed into the caverns, resulting in very turbid springflow during storms. During one event, more than 150,000 trout died due to silt build-up on their gills. Other fish kills were the result of poisoning by diesel fuel which was spilled on the construction site and washed into the caverns. In addition, damage to the hydrogeologic environment occurred because the construction beheaded the aquifer resulting in reduced flow and rechannelization of water away from the spring.

Finally, it should be noted that many factors may make it impossible to isolate the effects of highway construction (German, 1983). These include land-use changes, socioeconomic changes, and natural changes in a receiving water's plant community.

5.2 Methods to Prevent Construction Erosion

There is an abundance of literature on erosion control methods, but only a handful of reports focus on the control of erosion from highway runoff. Furthermore, only a fraction of these reports contain quantitative analysis of control methods.

Various mulches, blankets, chemical products, and silt fencing were tested for erosion control from highway construction sites in the state of Washington by Horner et al. (1990a). They reported on the effectiveness of these devices for erosion prevention and pollutant removal. Pollutant constituents measured included solids, metals, phosphorus, and organic content. Their results indicate that several methods were effective in removal of pollutant. A wood fiber mulch accompanied by grass seeding was the most cost-effective slope covering. Fabric fences, straw mulches, and a woven straw blanket were considerably more expensive than wood fiber mulch. Sedimentation ponds also were relatively expensive.

Schueler and Lugbill (1990) reviewed the performance of current designs of sediment basins and rip-rap outlet traps at several suburban developments in Maryland. They found that despite significant sediment removal, sediment levels in outflows remained elevated with a median TSS of 283 mg/L and a median turbidity of 200 NTU's. The overall performance of these sediment controls was estimated to be only 46% for storm events that produced measurable outflow runoff. The sediment removal capability was greatest for sediment controls in the earlier stages of construction and for storm events that produced less than 0.75 inches (19 mm) of rainfall. The authors reported that initial sediment settling was quite rapid with as much as 60% removal within 6 hours, but that additional removal occurred much more slowly.

Schueler and Lugbill (1990) found that erosion control measures, especially vegetative stabilization, were the most important line of defense, providing at least a six-fold reduction in downstream suspended sediment levels.

Also important was adequate sizing of the sediment basins, so that a minimum of 2 to 6 hours of detention was provided for larger storms.

Nawrocki and Pietrzak (1976) recommend the performance of a preconstruction site survey and analysis of the grain size distribution of the soils. This allows more accurate sizing of ponds for anticipated flow rates. They also recommend that ponds be provided with baffles, that a length-to-width ratio of 5 to 1 be maintained, and that inflow and outflow structures be as wide as feasible. They report that additional fine-grained sediment control can be achieved with the addition of chemical flocculants, and with the use of two ponds in series. Also emphasized in their study is the importance of source control. They include tables comparing the effectiveness different erosion control techniques on slopes of varying steepness.

6.0 CONTROLLING POLLUTION FROM HIGHWAY RUNOFF

The control of pollution from highway runoff can be accomplished by both source management and control measures. Examples of source management are transportation and land-use planning, and highway design and operation. Structural controls include vegetative practices, ponds, infiltration methods, wetlands, and filters. Certain structures enhance the performance of these structural control methods. Each type of control method will be characterized and its strengths and weaknesses will be identified. Additionally, where information is available, relative costs and expected maintenance activities for each method will be discussed.

In assessing the various methods available for the control of highway runoff, it is important to recognize the differences between urban storm drainage and highway runoff as related to structural control measures. Maintenance activities in urban watersheds can be performed on a more regular basis than can be expected alongside major highways; therefore, some urban runoff control structures prevalent in the literature are complex. Usually, these structures have outlet and/or inlet devices which are manually adjustable based on the prevailing hydrological conditions. These facilities also may involve pre-fabricated, commercially marketed, end-of-pipe treatment technologies. Such devices generally require more frequent monitoring, inspection, and maintenance than would be desirous for highway applications. End-of-pipe treatment approaches apparently may not be a suitable choice for urban runoff. Kuo and Lognathan (1988) recommended source control at the site of pollutant introduction to urban runoff as a more cost-effective measure than the upgrade of end-of-pipe treatment facilities.

Highway runoff pollution mitigation measures require a practical approach. Burch et al. (1985) stated that pollution associated with highway runoff usually is transported by stormwater runoff along the curbside, pavement, and shoulder areas. Most of the associated pollutant load is either the suspended particulate matter, or material adsorbed to the suspended solids. Therefore, the most effective control measures either will reduce the amount of particulates available for transport, or will settle and/or filter the particulate material in the runoff.

6.1 Source Management

Transportation plans can be designed to reduce water pollution by lowering total vehicle miles traveled. Providing alternate modes of transportation (mass transit) or encouraging carpooling can reduce traffic congestion and fuel consumption. Successful programs reduce the total load of pollutants deposited on highways and rights-of-ways from vehicles (Burch et al., 1985b).

Implementation of land-use plans provides indirect mitigation of highway runoff through effects on land use and associated traffic patterns. By controlling development location and density, traffic mix and density can also be controlled and runoff problems minimized. The goal of land-use planning for mitigation of highway runoff is to protect the environmental balance of an area in terms of runoff volume, rate, and water quality by restricting developments that generate high traffic volumes in sensitive areas (Burch et al., 1985b).

Design and operation of highways affect traffic characteristics and pollutant deposition. Reduction of pollutant runoff can be accomplished by elimination of curbs and other barriers; traffic flow regulation; animal control, fertilizer and pesticide/herbicide application; and control of debris from mowing (Burch et al., 1985b).

Eliminating barriers from highways prevents many sediments and other particulates from being trapped on the highway. Without curbs, wind and turbulence will remove much of the fine materials from the road. Materials that are not trapped on the roadway are generally immobilized on the roadside (Burch et al., 1985b).

Fencing of highway rights-of-way to minimize animal access to the highway, improves safety conditions, reduces accidents, and thereby lessens pollutants from spills, debris, and litter.

The use of fertilizers, particularly in areas with direct runoff to natural receiving waters, should be controlled to prevent unnecessary releases of nutrients. Pesticides and herbicides can cause significant acute and chronic toxic responses in terrestrial and aquatic ecosystems, but have not been found to be significant pollutants in highway runoff (Kobriger et al., 1984).

Grass mowing and other vegetative maintenance operations can leave cuttings and other debris along highway shoulders, which can form a thatch layer, reducing infiltration and preventing other vegetative growth. Mowing debris itself does not contribute significant amounts of pollutants to highway runoff. In areas with sparse vegetative cover, leaving cuttings on the ground can reduce erosion, preserve soil moisture, and add organic matter to the soil (Burch et al., 1985b).

6.2 Vegetative Controls

Vegetative controls include the grassed swale and vegetated filter (buffer) strips. These controls usually require a sizable amount of land, since they rely on gentle slopes to reduce water velocity to enable settling of suspended solids and some infiltration into the subsoil. Therefore, successful use of these vegetated control methods is highly site-specific. Maestri and Lord (1987) identify vegetative controls as the least costly management technique for controlling highway runoff.

Grassed swales are earthen runoff conveyance systems which make use of gentle slopes and wide shallow channels to remove pollutants through filtration by the grass, by some settling, and by some infiltration into the subsoil. Past research indicates that grass species is important to the overall filtering efficiency of swales. Grass density, blade size, blade shape, flexibility, and texture vary by species and influence the filtering efficiency (Umeda, 1988).

Schueler et al. (1991) proposed several swale design considerations including longitudinal slopes less than 6%, permeable subsoils, long swale contact times, swale lengths greater than 200 feet (60 m), and dense grass cover. Additionally, the maximum water velocity in the swale should not exceed 1.5 ft/s (0.45 m/s) and the peak discharge should not be in excess of 5 cfs. Swales operated above these conditions were relatively ineffective at reducing pollution. General costs of swale construction were found to be less than those associated with traditional curb and gutter conveyances, ranging from $5 to $15 per linear foot ($1.50 to $4.50 per linear meter), depending on swale dimensions. An added benefit of swales is the elimination of the curb and gutter system, which concentrates and transports pollutants in highway runoff.

Schueler et al. (1991) also report a range of pollutant removals for grassed swales. Of 10 swales monitored adjacent to highways and residential areas, 50% demonstrated high to moderate pollutant removal, while the remainder showed negligible or negative removal. The expected removal for well-designed, well-maintained grassed swales is reported as 70% for total suspended solids (TSS), 30% for total phosphorus (TP), 25% for total nitrogen (TN), and 50 to 90% for various trace metals.

Several investigators studied the effectiveness of swales, and the immobilization of metals is commonly reported. Yousef et al. (1985a) found swales to be effective in removing ionic species of metals from highway runoff. Processes involved included sorption, precipitation, co-precipitation, and biological uptake. The effects of flow through swales were quantified in terms of change in concentration between highway runoff and swale outflow for heavy metals. The data in Table 6.1 show mass retention values for both metals and nutrients and indicate the importance of channel length.

Table 6.1
Average Concentrations of Dissolved Pollutants
Flowing over Roadside Swales
(Yousef et al., 1985a)

	Maitland Site			EPCOT Site			
Pollutant	0.0 m	23 m	53 m	0.0 m	30 m	90 m	170 m
Zn	22	9	3	140	103	77	53
Pb	9	5	9	67	43	41	29
Cu	6	6	5	26	30	29	24
Fe	260	102	81	290	290	261	316
Cr	9	6	8	10	9	10	10
Cd	-	-	-	7	6	5	4
Ni	-	-	-	70	59	47	34
OP-P	368	290	279	580	546	514	530
TP-P	415	367	310	599	586	558	580
NH_4-N	1015	870	699	293	321	297	299
(NO_2+NO_3)-N	192	188	167	147	151	147	163
Organic N	842	1337	951	1833	1994	1683	1973
Total N	2049	2395	1817	2273	2456	2127	2435

Average Concentration in µg/l
Customary units: To convert meters to feet multiply by 0.30

Wang et al. (1980) used mass balance studies to show the effectiveness of well-vegetated surfaces in retaining metals from highway runoff at several sites in the Seattle area. They also compared the transport and deposition of heavy metals in a grass channel, a paved channel, and a mud channel, and found grass to significantly reduce metal travel distances (see Figure 6.1). Lead was immobilized most effectively; 95% of the lead entering a 24-meter- (79-feet) long channel was retained by the soil or grass. Bell and Wanielista (1979) also found that lead was immobilized more than other metals. Some of the concentrations actually increase through certain stretches of the channel (iron is the most obvious in Figure 6.1). Wang et al. (1980) attribute this to the channel being partially unvegetated and erosion carrying metals to more vegetated sections of the channel.

Yousef et al. (1987) pumped simulated highway runoff spiked with the appropriate levels of nutrients and metals over two swales in central Florida to study mass transport and removal efficiencies. Dissolved metals (particularly ionic species) were more effectively removed in the swales in this study than nutrients. In fact, nutrient concentration in highway runoff flowing over roadside swales may increase. Increases in nutrient concentrations especially are likely during fall and winter when vegetation growth is decreased and nutrients are released by dead plant material. Little et al. (1982) found the concentration of nutrients (total phosphorus, nitrate plus nitrite, and soluble reactive phosphorus) decreased by at least 20% in the summer in a 73-meter (240-feet) channel. Conversely, soluble reactive phosphorus concentrations increased along the length of the channel during the fall of the same year.

Other pollutants also can be removed from runoff using vegetated swales. Little et al. (1982) reported oil and grease removal efficiencies ranging from 67 to 93 percent in the 73-meter (240-feet) channel mentioned previously. The oil and grease tests also were performed during winter. TSS and VSS concentrations decreased at least 65%, and algal bioassays demonstrated reduced growth inhibition by toxicants in highway runoff when drained through vegetation. Wang et al. (1980) confirmed the effectiveness of swales in reducing concentrations of TSS, VSS, and COD.

Wanielista et al. (1978) conducted an evaluation of eleven field sites to measure the effectiveness of diverting highway runoff to shallow roadside ditches which remained aerobic. Of particular interest was the finding that indigenous, hydrocarbon-utilizing, bacteria populations are up to 90% higher during the rainy months than at other times of the year. The

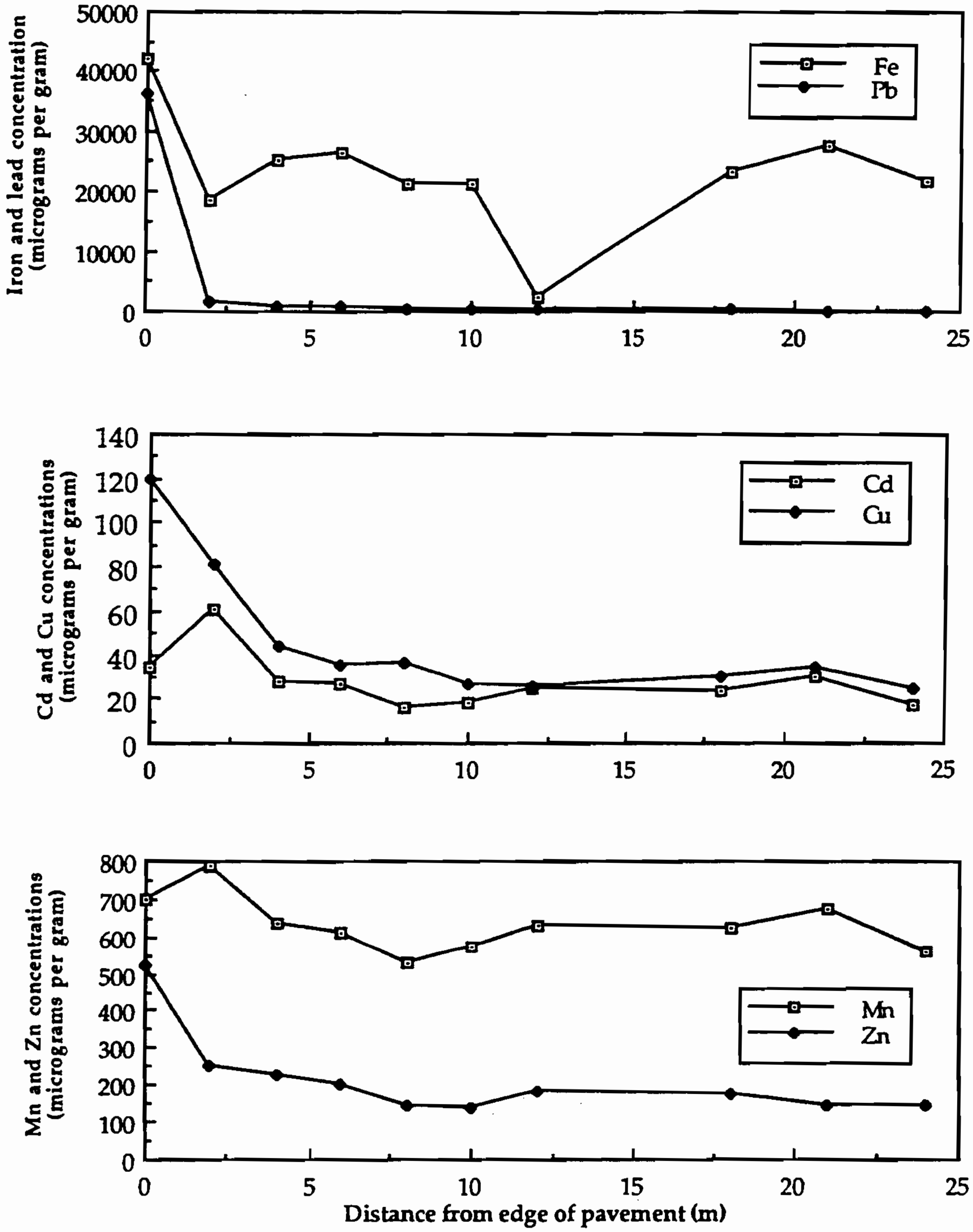


Figure 6.1 Metal Concentration in Surface Soil Near a Highway (Wang, 1980).

microbial concentrations are greatest near the pavement, decreasing until reaching the roadside ditches. The results of this study indicated a maximum hydrocarbon degradation of only 48% after 60 days along roads with no ditches. The roads with shallow aerobic ditches reached hydrocarbon degradation rates of 99%. The authors recommend leaving grass clippings on the ground to enhance nutrient availability for the hydrocarbon-degrading microorganisms. Additionally, the planting of a nitrogen-enriching legume cover crop, such as clover, can increase nitrogen availability.

Some of the reports offer design suggestions. Yousef et al. (1985a) indicate the importance of channel length (see Table 6.1), as well as the need to reduce velocity of flow. In this report, removal efficiencies at two different swale locations (Maitland and EPCOT) were compared under different flow-through conditions. A decrease in organic nitrogen removal over time was observed. A possible explanation was the increase of organic material deposited in the swale during periods of rapid grass growth. A thin grass cover (< 20%) was preferred to a thick one (≥ 80%), since a heavy grass cover would increase the amount of organic material, effectively lessening the available sorption sites for the nitrogen and phosphorus ionic species. The deposited organic material is also available for decay and resuspension in later storms.

Higher infiltration rates also might explain the greater efficiency observed at the Maitland site. Infiltration at the Maitland site ranged from 1.4 to 3.4 in/hr (36 to 86 mm/hr) and infiltration at the EPCOT site ranged from 0.5 to 1.3 in/hr (13 to 33 mm/hr). It was determined that swale removal efficiency on a mass basis for the heavy metals, nitrogen, and phosphorus is proportional to increased contact time and infiltration rates. Therefore, swale design should include longer channels with flatter slopes. It should be noted, however, that little nitrogen removal was achieved when the excess runoff exceeded 3 in/hr (76 mm/hr).

Yousef et al. (1985a) found that the efficiency of swales increased when soils are dry and infiltration rates are high, where lower water tables exist, and when contact time is increased. Thus flat-sloped swales with sandy soils which are high above the groundwater elevation are preferable.

Wanielista et al. (1988a) studied five swale sites (three of the sites were adjacent to highways) in central Florida to determine flow rates and when permeabilities. Flow rate was measured using 90-degree V-notch weirs, and a double-ring infiltrometer (adjusted for existing conditions) was used to determine permeability. Descriptions of soil and ground cover for each swale were reported. It was recommended that a design infiltration rate equal to 50% of the value obtained with the double-ring method be used. The flow data indicate that higher runoff volumes result in lower infiltration rates and higher discharge volumes. Discharge volume decreases when swale length is increased. Swale geometry and slope are also important design features, and the authors present design equations for the construction of swales.

Citing storm data from Florida, Wanielista et al. (1986) show that at least one storm of 1-hour duration and producing 3 inches (76 mm) or more of runoff will occur each year. Since swales in Florida are designed to convey the 3-inch (76-mm) runoff volume as a maximum, the authors propose the use of swale blocks to increase the retention capacity of the swale. Swale blocks are earthen check dams placed transversely across the longitudinal direction of the swale. The miniature detention pools created by the swale blocks increase the swale contact time, providing for increased settling of suspended particles and more infiltration. Design calculations and construction methods for swale blocks are presented in the report.

Several swale blocks were constructed in an existing swale according to the design parameters. After two years, and three storms producing runoff in excess of 3 inches (76 mm), the swale blocks were still operating. Flows during the large storms had yet to overflow the swale blocks. Swale blocks effectively retained suspended solids, but the authors offered no comparisons to solids retention efficiencies for swales without swale blocks. Schueler et al. (1991) state that additional investigations are necessary to evaluate such swale design enhancements and their ability to improve pollutant removal and inhibit downstream pollutant migration.

Very little maintenance is required for grass swales. Recommended practices included seasonal mowing, cleaning of trash and debris, and removal of heavy sediment deposits. Schueler et al. (1991) found that the biggest maintenance problem in Washington State was the accumulation of soil and grass alongside the highway edge; runoff was diverted away from the swales. Manual removal of sediments was recommended to preserve the infiltration capacity of the underlying soils (State of Maryland, 1985). Additionally, replanting of vegetation may be necessary. Burch et al. (1985) note that periodic thatch removal or mechanical aeration may be required to restore the original permeability of the soil.

Some concerns and unknown factors about the use of swales are noted in the literature. Umeda (1988) expressed concern about the potential for reduced removal efficiencies during times of summer drought, when vegetation dies or becomes dormant. Schueler et al. (1991) cite several other concerns. Swales in soils which are too sandy may experience side slope stability problems. Also, standing water in swales may create mosquito and odor nuisances. The State of Maryland (1984a) recommends that swale designs allow for a maximum ponding time of 24 hours. Another question to be answered is whether swale performance over longer periods will decline as adsorption sites are used. Finally, swales may adversely affect the groundwater in certain areas.

Vegetated buffer strips, also known as filter strips, are another vegetated control measure. These controls may appear as any vegetated form, grassland to forest, and are designed to intercept upstream flow, lower velocity of flow, and spread out the flow as sheet flow (Schueler et al., 1991). Unlike swales, which are concave conveyance systems, filter strips are fairly level. Filter strips, like swales, require flat slopes (generally less than 5%) and fairly permeable natural subsoils for effective performance. According to Schueler et al., filter strips are best suited for agricultural practices and low land usage areas. Strips are unable to treat high flow velocities, which typically are associated with impervious areas. Schueler et al. mention a study which evaluated filter strips in an urban environment. Removal for TSS was only 28%; efficiencies for nutrient removal were not reported. Although an inexpensive control measure, vegetated buffer strips are most useful in contributing watershed areas where peak runoff velocities are low.

6.3 Ponds

The use of ponds to treat highway runoff is well-documented. Basically, three types of ponds exist: detention ponds, extended-detention ponds, and "wet" ponds.

Detention ponds are primarily flood control devices, designed to reduce the peak flow associated with large storm events. As such, ponds are designed to be "dry" between storm events. They usually are designed as basins with a fixed orifice outlet pipe that controls the drainage release rate through the structure. These "dry" ponds are usually designed to detain runoff for 1 to 2 hours (Metropolitan Washington Council of Governments, 1983). Runoff treatment is limited and is primarily related to the detention time which affects the amount of particulate settling.

Extended detention ponds are similar to detention ponds, but are more suitable for improving runoff water quality. As the name implies, these structures are dry ponds with extended detention times. Detention times of up to 24 hours are common, with 6 to 12 hours being the preferred minimum (Schueler et al., 1991). Detention times are increased through the use of adjustable drainage orifices, or by using vertical, perforated riser pipe drains.

Pollutant removal is achieved through the increased detention time which allows the suspended particles time to settle out. The amount of treatment is moderate, but highly variable, depending on detention time and the fraction of the runoff effectively detained. Reported removals range from 30%-70% for TSS, 15%-40% for COD, and 10%-30% for total phosphorus (Schueler et al., 1991). Soluble nutrient removal rates are low, and sometimes even negative. Schueler et al. state that the treatment efficiencies for smaller storms (i.e., < 0.5 watershed inches (13 mm)) are usually higher than those associated with the larger ones. Lange (1990) also found chloride treatment to be relatively ineffective in dry ponds.

Dorman et al. (1988) attribute the low, and sometimes negative, efficiencies, to the fact that most pollutants associated with highway runoff are allied with the smaller particulate material which does not have adequate time to settle by gravity in a dry pond. Additionally, the heavier sediments and pollutants which settle during earlier storms may wash out during subsequent rainfall-runoff events. So, the dry basin becomes an effective pollutant source rather than a sink, and the long-term pollution abatement efficiency is low.

The construction costs of dry ponds are generally the least of those for all the pond options. However, the maintenance burden of these ponds is usually higher. Debris and sediment deposit quickly, requiring more frequent removal, and many of the dry ponds are difficult to mow. Although few of these ponds have been known to fail, the designed detention times are usually much higher than the actual detention times. The inability to predict the actual detention time creates difficulties when attempting to estimate the potential of a dry pond for removing pollutants prior to construction. Chronic clogging of inlets and outlets often poses problems. Downstream warming of natural waters may also adversely affect biota, if the pond is not shaded (Schueler et al., 1991). Since dry ponds are neither reliable nor particularly effective in treating highway runoff, the use of such structures has been deemed ineffective (Dorman et al., 1988).

Wet ponds are considerably more effective at mitigating highway runoff pollution. Maestri and Lord (1987) claim that wet ponds are the best choice for highway runoff treatment if vegetative controls are not feasible. These ponds are designed to maintain a permanent pool of water and to retain a certain amount of storm runoff. Pollutant removal is achieved primarily through the sedimentation of suspended particles and with some biological processes accounting for soluble nutrient reduction. Reported removals have ranged from poor to excellent and are a function of the basin size relative to the contributing watershed and area storm characteristics.

Schueler et al. (1991) report that monitoring studies indicate a range of wet pond removal efficiencies. Reported ranges for removal of TSS, total phosphorus, and soluble nutrients are 50%-90%, 30%-90%, and 40%-80%, respectively. Moderate to high removals of trace metals, organic matter, and coliforms often are reported as well. Yousef et al. (1985b) reported removal efficiencies for a wet pond at a highway interchange having a surface area of about 3 acres (1.2 ha) and a depth of 1.5 to 2 meters (5 to 6.5 feet). Removals for dissolved cadmium, zinc, copper, lead, nickel, chromium, and iron ranged from 27% to 63%. The average total removals for the same constituents varied from 47% to 97% of the incoming highway runoff concentrations.

Wanielista et al. (1988b) attempted to determine the removal efficiency of a detention pond for both individual events and for the annual average. Parameters included TSS, inorganic and organic carbon, dissolved metals, and total and fecal coliform. Removal efficiencies for all constituents were significant based on the concentration data. The average concentrations from the pond inlet were compared to average concentrations of a combined pond/outlet value. Removal efficiency was computed for individual runoff events and for yearly average concentration. Some single-event efficiencies are: 45% for organic carbon, 97% for total coliform, 50% for zinc, 49% for copper, 27% for iron, and 37% for lead. Reductions in yearly average concentration between the inlet and the outlet were 42% for organic carbon, 99% for total coliform, 69% for zinc, 60% for copper, 67% for iron, and 31% for lead. However, using mass data, the removal efficiencies are negative due to groundwater flow into the pond. Therefore, detention ponds should be designed to minimize groundwater inflow.

Of particular importance to wet ponds are quantitative and qualitative measurements of the bottom sediments. Sediment accumulation rates are important in determining recommended cleanout intervals. Qualitative investigations of the sediments are necessary prior to selecting disposal options, since accumulated metals in sediments may affect the underlying groundwater or be released back into the pond.

Previous results showed that a large portion of the dissolved and suspended heavy metals were deposited in the pond sediments. Yousef et al. (1985b) investigated the potential migration of the metals through the sediment. Their investigation discovered that the top 5 to 6.8 cm (2 to 2.5 inches) of the bottom sediments contained heavy metals concentrations greater than the background concentrations. These concentrations attenuated quickly with increasing depth through the sediment material. It was concluded that, upon reaching the pond bottom, the heavy metals formed stable associations, remained near the sediment surface, and decreased in concentration with increasing depth. Most of the heavy metals were bound to the iron-manganese (Fe-Mn) oxides and organic matter in the pond sediments. Although most of the lead and cadmium appeared in exchangeable form, the hydrous Fe-Mn oxides served as a sink for the trace metals since aerobic conditions were maintained. Therefore, under aerobic conditions, the potential for trace metal release to solution remained highly unlikely.

Hvitved-Jacobsen et al. (1984) investigated a wet pond's sediment bed over a seven-year period and found that the sediments served as a sink for both nitrogen and phosphorus. Throughout the monitored period, 99% of the phosphorus input was deposited in the sediments and remained there as a result of the aerobic conditions at the water-sediment interface. The aerobic conditions not only limited more phosphorus release, but increased the top layer of sediment's sorption potential for more phosphorus. In fact, after one storm, the orthophosphorus concentration in the pond was 130 μg/L, but after approximately three days, the concentration was reduced to 10 μg/L. Additionally, 85%-90% of the total nitrogen input was removed, most likely through the denitrification process in the lower anaerobic sediment layers where plenty of organic material was available. Ammonia is released by means of denitrification in the sediments and is subsequently nitrified, forming nitrate. The nitrate diffuses to the lower sediment layers and is denitrified. Additional ammonia is released through plant uptake. Since biomass growth in the pond was calculated to be nitrogen-limited, the phosphorus build-up should not cause future pond eutrophication.

Disposal methods for accumulated pond sediments are dependent on the concentrations of constituents. Additionally, sediment accumulation rates are useful in determining pond sediment removal intervals. In one study, Yousef et al. (1991) studied nine wet ponds in Florida which received mostly highway runoff. The parent soil in the pond bottom was chosen to represent background conditions. The top sediments showed higher moisture contents, greater organic contents, and lower densities than the soil beneath. Nutrient and heavy metals concentrations were also greater in the sediments than in the soils. Average pollutant concentrations for the sediments and underlying soils are presented in Table 6.2.

Table 6.2
Concentrations of Constituents In Wet Pond Sediments
(Derived from Yousef et al., 1991)

Constituent	Parent Soil	Sediment
Total N (mg/L)	0.41	3.62
Total P (mg/L)	0.32	0.58
Cd (μg/g)*	15	5
Cr (μg/g)*	61	18
Cu (μg/g)*	28	5
Fe (μg/g)*	3554	1969
Ni (μg/g)*	52	28
Pb (μg/g)*	374	67
Zn (μg/g)*	161	12

*Dry Wt Basis

The Environmental Protection Agency's (EPA) Toxicity Characteristic Leaching Procedure (TCLP) was performed on the bottom sediments to determine if the deposited sediments were hazardous waste. Concentrations resulting from the TCLP test were much lower than those allowed for hazardous wastes. Additionally, sediments containing increasing levels of silt and clay, as well as increased organic matter content, caused exponential reductions in the sediment TCLP concentrations. Sediment accumulation rates averaged between 1.1 and 4.2 cm/yr (0.4 to 1.6 in/yr).

Correlations between sediment accumulation rates and various potential influencing factors were also investigated. Relationships between sediment accumulation rate, impervious watershed area, average daily traffic volume, highway area, and other combinations were evaluated. The best correlation was that between the annual sediment accumulation rate and the pond surface area as a percentage of the contributing watershed area. Using a design standard of allowable pool volume reduction due to sediment accumulation of 10% to 15%, the authors recommend sediment removal every 25 years based upon the observed sediment accumulation rates.

Maestri and Lord (1987) mention a statistical analysis of wet pond runoff inflows over time. The analysis suggests that detention basin performance can be divided into two distinct periods. The first period is the dynamic period, which occurs during rainfall runoff events. The second is termed quiescent, and is considered the time period between storms. The authors point out that more knowledge of rainfall-runoff relationships, settling velocities of the particles in the runoff, and particle size distributions are necessary in order to facilitate the design of wet ponds to reach pollutant removal objectives.

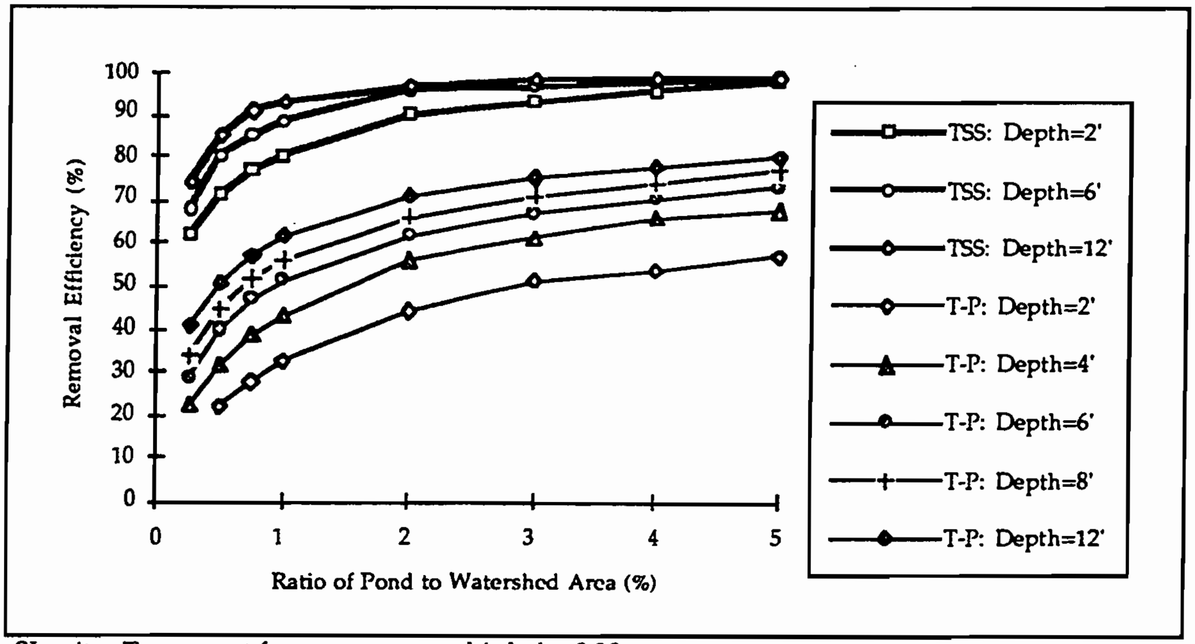
Haan and Ward (1978) conducted some research on sediment particle size as it relates to sedimentation basins. A predictive model was developed which provides an estimate of a basin's sediment trapping efficiency. The research concludes that the biggest influencing factor for sedimentation is the number of incoming sediment particles in the 5 to 20 micron range, and that particles less than 5 microns in size are unlikely to settle without the aid of a flocculant. The investigators affirm the need to more accurately estimate total sediment load flowing into the basin and to successfully determine sediment particle size distribution.

Wu et al. (1989) evaluated the removal efficiencies of three wet ponds as a function of the pond surface area to contributing watershed area. The evaluated ratios ranged from approximately 0.75% to 7.5%. At each pond, removal efficiencies for iron, zinc, TSS, total Kjeldahl nitrogen, and total phosphorus were calculated. For all of the measured constituents, removals increased when the pond surface area to contributing watershed area ratio was larger.

Maristany (1989) showed the relationships between wet pond surface area, capacity, and removal efficiencies for total suspended solids (TSS) and total phosphorus. Eleven wet ponds, with contributing watershed areas ranging from 128 to 23,393 acres (50 to 9240 ha), and with 9% to 54% impervious area, were evaluated. Treatment efficiencies for TSS and for total phosphorus were calculated using methods promulgated by the EPA and the Corps of Engineers, respectively. These methods required specific storm characteristics and pond dimension data. The phosphorus removal estimate also required data for inflow concentrations of orthophosphorus and total phosphorus. Detailed phosphorus data were not available for all of the ponds, so the same average values were assumed for all of the ponds evaluated. The calculated results show that for a given basin, there is a point of diminishing returns where increased removal occurs at a lesser rate than the rate of storage capacity increase. Since this point is always reached first for TSS removal, followed next by the total phosphorus, Maristany suggests that longer detention times are necessary for increased phosphorus removal.

Wet pond performance as a function of depth was also evaluated. The calculated data show that once the ratio of pond to watershed area exceeds 2%, total phosphorus removal is controlled by the pond depth rather than the surface area. As seen in Figure 6.2, at a 2% ratio, a pond 2 feet (0.6 m) in depth removes approximately 40% of total phosphorus, while a 4-foot (1.2-m) depth at the same ratio removes about 55%, and a 12-foot (3.6-m) depth results in the removal of 70%. Similar increases in TSS removals are also evident, only not so dramatic. Maristany (1989) concludes that these trends indicate that pond deepening may be the most cost-effective measure to enhance the performance of a wet pond if excavation costs are less than land acquisition costs.

Martin (1989) performed a tracer dye study under near steady-state discharge conditions to determine the extent of mixing which occurred in a wet pond. The pond has a surface area of about 0.2 acres (0.08 ha) and a contributing watershed of approximately 42 acres (17 ha). About 33% of this area is urban highway. The pond is rectangular in shape with a length to width ratio of close to 2:1. Dead storage depth is 8 feet (2.4 m), and the



SI units: To convert feet to meters multiply by 0.30

Figure 6.2. Effect of Pond Depth on Wet Pond Treatment Efficiencies for TSS and Total P (Modified from Maristany, 1989)

addition of the maximum live storage depth produces a maximum pond depth of 11 feet (3.3 m).

The collected data indicate that the amount of mixing occurring in the wet pond is related to the amount of available live storage space. The larger the available live storage space, the more closely the pond approaches completely mixed conditions, and short-circuiting is decreased. This relationship is even more pronounced for low discharge rates; measured residence times approach the theoretical completely-mixed values. However, under high discharge conditions, short-circuiting is more prevalent and residence times are reduced. One tracer dye experiment conducted under high discharge conditions revealed 40% of the particles short-circuiting in less than 6 minutes

The State of Maryland offers several reasons behind its wet pond design criteria (State of Maryland, 1984b). The design guidelines state that a wet pond's removal by settling in a wet pond is directly related to the pond geometry, detention time, volume, and particle size. Gravitational settling occurs when the average velocity in the ponds is less than the critical settling velocity of the particle. Care must be exercised when considering the pond geometry. Inlets and outlets should be positioned in such a way so as to minimize short-circuiting and to promote mixing. Baffles may be positioned in the pond to increase the flow length, if dead space is unavoidable. The Maryland design criteria recommend a 2:1 length-to-width ratio and a wedge-shaped pool, with the inlet at the narrow end. The area ratio is the drainage area divided by the pond surface area, and the volume ratio is the wet pond volume divided by the mean runoff volume. The author presents relationships between the two ratios as they pertain to wet pond performance. Evidence is presented which shows that a smaller area ratio and a larger volume ratio usually will increase pollutant removal performances of most wet ponds. Justification of the 9-day detention time design criteria for wet ponds also is given.

Costs for wet ponds are definitely higher than for other ponds. Wet ponds usually cost 25% to 40% more than other detention methods (Schueler et al., 1991). Permitting costs for wet ponds may equal or exceed design costs in some cases. Costs also are highly dependent on land acquisition costs. Costs per unit area treated generally decrease with increasing contributing watershed area (Burch et al., 1985). According to Schueler et al. (1991), annual maintenance costs range from 3% to 5% of construction costs. Maintenance typically consists of inspections, trash and debris removal, and mowing of embankments. Additionally, sediments must be removed as necessary, as failure to do so will decrease long-term performance. To date, very few wet ponds have failed. Well-designed ponds may last over 20 years.

Schueler et al. (1991) also list some additional considerations associated with wet ponds. Ponds are not useful in regions where the annual evapotranspiration rate exceeds the annual precipitation. Additionally, dry weather baseflow assists in maintaining the wet pool elevation and preventing stagnation. If not properly sited, wet ponds may cause downstream warming, although the downstream impacts of wet ponds are not fully known. Nutrient releases from pond sediment over time also have not been fully evaluated. Further studies investigating the trace metal uptake by wet pond biota (especially fish) are needed.

6.4 Wetlands

Wetlands which treat highway runoff are designed as shallow pools which create growing conditions conducive to marsh plant growth, maximizing pollutant removal through plant uptake (Schueler et al., 1991). Unlike constructed wetlands for wastewater treatment and NPDES requirements, storm runoff wetlands are not designed to replicate all of the natural wetland ecological functions. Pollutant removal is achieved primarily through wetland plant uptake, physical filtration, adsorption, gravitational settling, and microbial decomposition. Wetlands have the ability to assimilate large quantities of dissolved and suspended solids and exhibit a high nutrient demand (Dorman et al., 1988). Wetlands are particularly useful at removing BOD, TSS, and heavy metals. Nutrients also are removed, but rates are highly variable. Overall, wetlands' treatment efficiencies are similar to those associated with wet ponds (Schueler et al., 1991). Additionally, the degree of treatment is dependent on the surface area to volume ratio, the treatment volume, and the ratio of wetland surface area to contributing watershed area.

Dorman et al. (1988) provide additional wetland design guidance. A relatively long retention time (6 to 14 days) is the most important factor in removing heavy metals and other toxicants. Shallow water with a low basin gradient to slow the flow also is important. This configuration assists in maximizing the contact time between runoff and wetland vegetation and soils. Sufficient size to store the design storm runoff volume is also necessary. Finally, inlets to wetlands should be designed to eliminate or minimize the erosion potential. Wetlands for treating highway runoff are relatively new. Very little information is available with regard to performance.

Schiffer (1988) documented the reduction in the concentration of automobile-related chemicals between the inlets and outlets of wetlands. The greatest removals evaluated were those for lead and zinc, averaging 80% and 53%, respectively.

Increased costs for wetlands are usually associated with the increased land area necessary for their construction (Schueler et al., 1991). Often, wetlands require two to three times the space required for other control methods. Design costs are slightly higher than those for wet ponds, usually due to required environmental analyses. Construction costs should be slightly higher than those for wet ponds because of the use of special planting techniques. Wetlands generally require intensive maintenance to establish the marsh during the first three years. After this time, maintenance is similar to that associated with other ponds. Typically, annual maintenance costs should be about 3%-5% of construction costs. Well-designed wetlands should last for many years: the oldest ones in existence today are less than 10 years old. Linker (1989) postulates that by using basins adjacent to highways which have been created through "cut and fill" operations during highway construction, significant wetlands construction cost savings can be realized.

Schueler et al. (1991) highlight some concerns pertaining to wetlands. Wetlands are difficult to establish in sandy soils or in soils with high permeabilities. Additionally, wetlands may not work well in areas with high evapotranspiration rates. Wetland performance is greatest during the warmer months which are associated with the growing season. The extent to which removal rates are reduced during the colder months is unknown. Also, the annual dieback of wetland plants may generate a pulse of nutrients in the outflow. Annual harvesting of plants may increase removal rates, but this practice needs to be evaluated. Biota uptake of heavy metals may also be a concern.

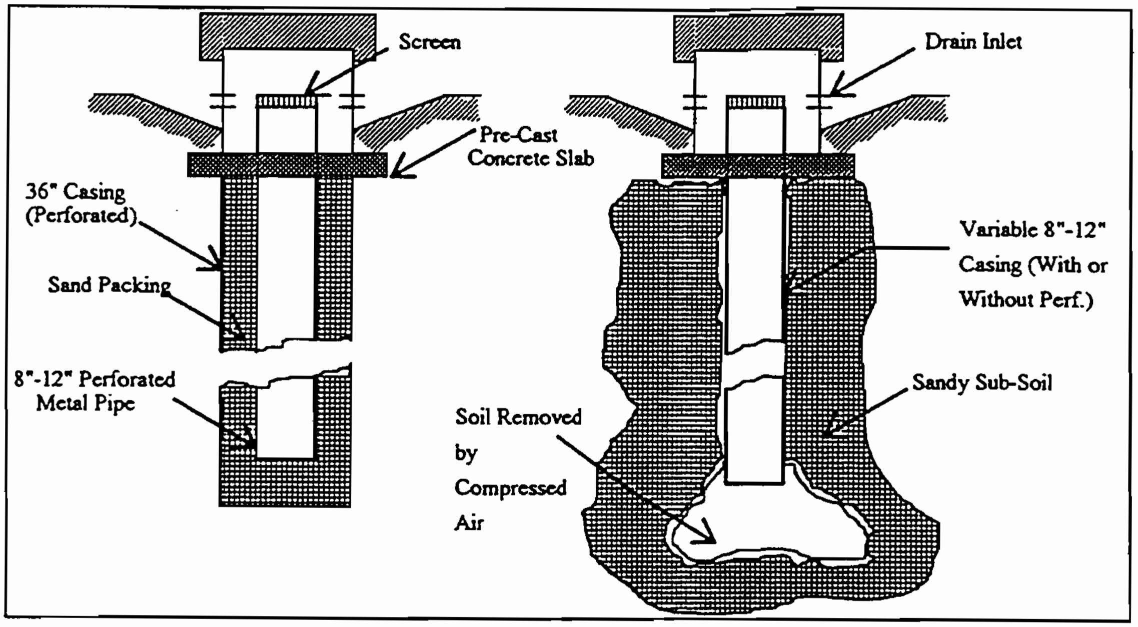
6.5 Infiltration Practices

Infiltration practices are designed to contain a certain volume of highway runoff. The runoff is treated as the water percolates into the underlying subsoil or through a prepared porous media filter bed. Infiltration structures include porous pavements, wells, trenches, and basins.

Porous pavements consist of a thin coat of open-graded asphalt placed over a base of crushed stone. Runoff water is held in the pore spaces until it percolates through the sub-base or drains laterally through underdrains. Chronic clogging of porous pavements is common. Since a dry sub-base is essential to good highway design, these structures are ineffective for treating highway runoff and are recommended for use only in parking lots and low-volume trafficways (Maestri and Lord, 1987).

Infiltration drainage wells are useful in intercepting runoff, treating it, and recharging the groundwater. Jackura (1980) describes the use of drainage wells to drain highway surface runoff in California beginning in the 1960's. A typical schematic of a drainage well appears in Figure 6.3.

The two major types of drainage wells are the open-end casing (with or without perforations) and the closed-end casing. Wells must extend through impervious strata and terminate a minimum of 10 feet (3 m) above the



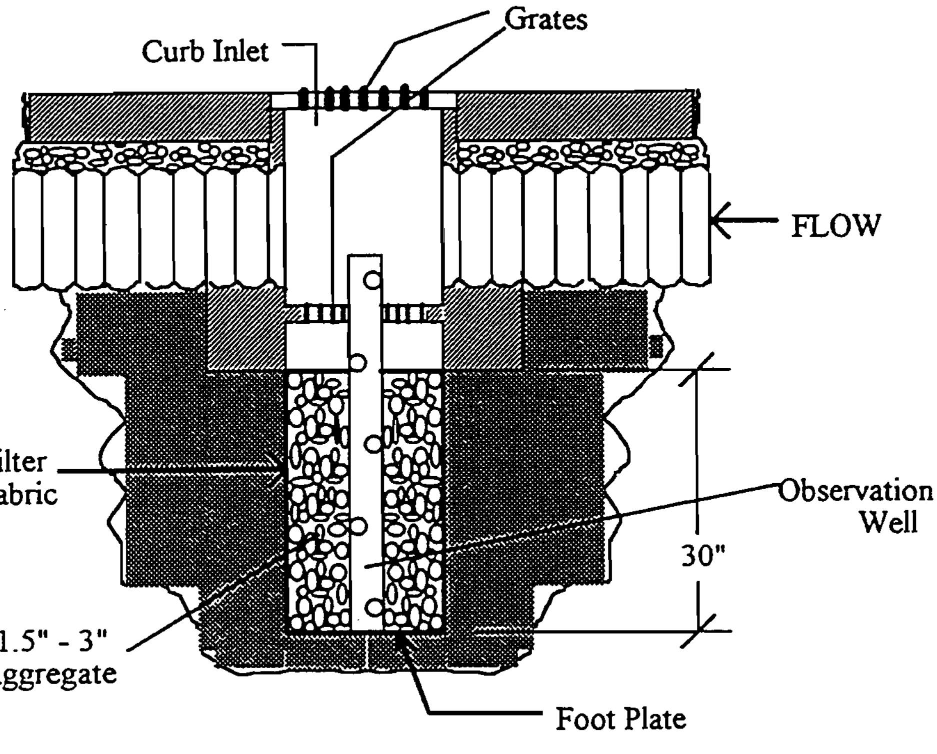
SI units: To convert inches to centimeters multiply by 2.54

Figure 6.3 Drainage Well Installations
(Modified from Jackura, 1980)

groundwater, unless the runoff has been treated. Usually, gravel and/or sand is mounded around the drainage inlet to filter the incoming water. It is much easier to replace the mounded filter media than it is to replace an entire well. Construction and operating costs of drainage wells are high, based on a unit volume of water drained, in comparison to cost of other infiltration practices. Air jets usually are used to rejuvenate the pore spaces adjacent to the well casings. The risk of groundwater contamination also is higher with drainage wells compared to that with other infiltration practices; therefore, use of wells for highway runoff is not highly recommended.

The State of Maryland (1984a) designed a drainage well for use with a highway curb and gutter inlet. An example of the structure is shown in Figure 6.4. The well is 3- to 12-feet (0.9- to 3.6-m) deep and requires a minimum soil percolation rate of 0.27 in/hr (6.9 mm/ha). The authors state that the longevity and performance of the structure is not well documented.

Infiltration trenches and basins are the classic infiltration systems associated with highway runoff. Often, these devices are termed "retention"



SI units: To convert inches to centimeters multiply by 2.54

Figure 6.4 Curb Inlet Dry Well
(Modified from State of Maryland, 1984a)

structures since the runoff water is essentially retained. The design storm runoff volume is captured by the device and slowly exfiltrates through the bottom and sides of the structure. The exfiltrated water passes through the sub-soil and eventually recharges the water table. Pollutant removal is primarily through sorption, straining, and microbial decomposition in the underlying soils (Schueler et al., 1991).

Very few monitoring studies have evaluated infiltration practices. Estimates of the effectiveness of infiltration systems are derived primarily from rapid infiltration tests on land applied wastewater treatment systems and through modeling. Schueler et al. (1991) report estimated sediment removals greater than 90%, and phosphorus and nitrogen removals of about 60%. Removals of approximately 90% of coliforms, trace metals, and organics can be anticipated. Lower removal efficiencies for chlorides, nitrate, and soluble trace metals are anticipated, particularly in sandy soils. Increased levels of organic material in the soil can increase the removal efficiencies.

The fact that these structures provide recharge to the groundwater, instead of discharge downstream, makes them a useful tool for restoring pre-development groundwater conditions and for reducing downstream flooding and erosion. As such, infiltration controls are the structural methods of choice in Florida and Maryland (Dorman et al., 1988).

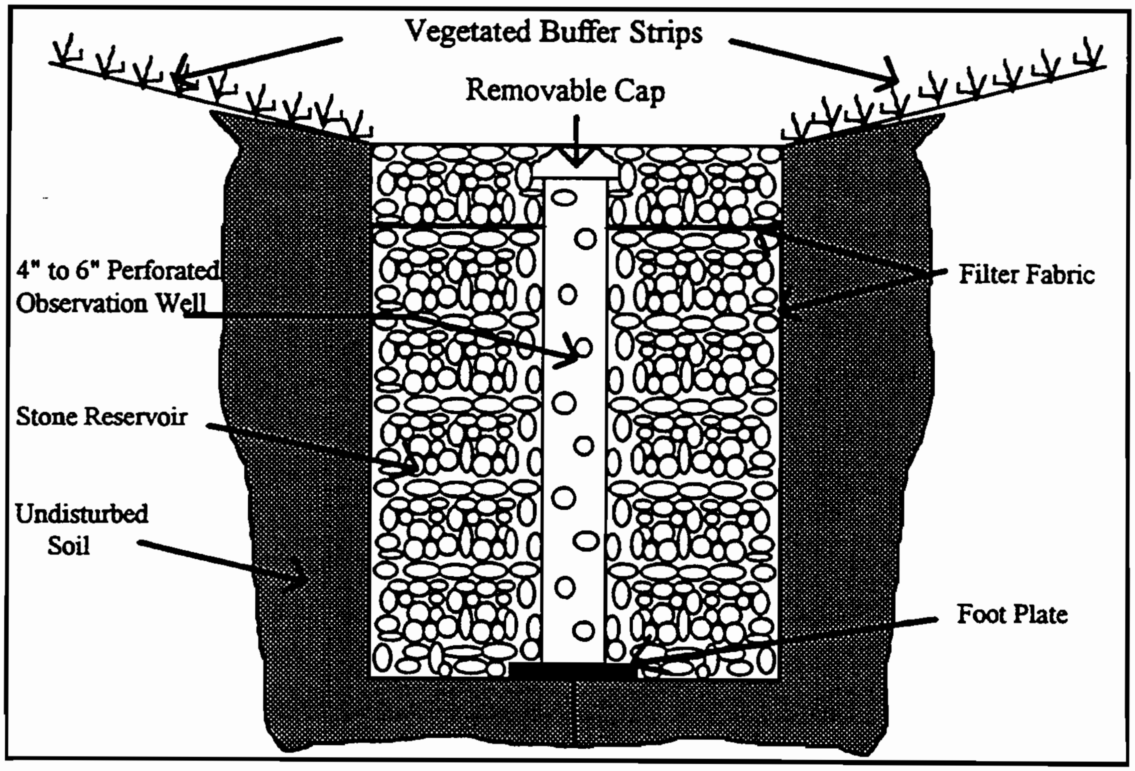
Infiltration devices (i.e., trenches and basins) are highly dependent upon site conditions. Schueler et al. (1991) affirm that most sites will require an on-site geotechnical investigation to ascertain infiltration feasibility. A wide range of minimum site criteria values appears in the literature. The important criteria include the saturated soil infiltration rate, maximum allowable dewatering time, minimum distance between facility bottom and underlying water, bedrock, or confining layer, and topographic features. The minimum saturated soil infiltration rate most commonly cited is 0.5 in/hr (13 mm/hr) (Schueler et al., 1991), but values as low as 0.3 in/hr (7.6 mm/hr) also are reported (Dorman et al., 1988). Maximum allowable dewatering rates are a function of the statistical evaluation of the average time between rainfall events for a given meteorological region (Dorman et al., 1988).

The infiltration devices should be designed to dewater completely between storms. Harrington (1989) states that this drying allows the soil pores to rejuvenate and prevents sealing of the soil pores as well. Burch et al. (1985a) assert that algal growth in porous media is best controlled by allowing the media to dry between storms, particularly during the warmer months. Maryland and Florida design for a 72-hour maximum dewatering period (Dorman et al., 1988).

Minimum depth to the underlying water or confining layer is important with respect to groundwater contamination. Enough contact time must be allowed for percolating runoff so that the potential for groundwater contamination is minimized. Typically, this value is dictated by the regional depth to the water table (Dorman et al., 1988). Values range from 3 feet to 10 feet (0.9 to 3 m), with eastern States generally opting for 2 to 4 feet (0.6 to 1.2 m) due to shallow water tables, and western States closer to the 10-foot (3-m) minimum standard. Steep slopes also may prohibit infiltration practices. The range of maximum allowable slopes reported is 7% (Dorman et al., 1988) to 5% (Schueler et al., 1991). Infiltration structures also should be located a minimum of 100 feet (30 m) from water supply wells (Schueler et al., 1991).

Although the treatment methods and efficiencies associated with infiltration trenches and basins are essentially the same, applications, designs, and costs are quite different. Infiltration trenches are best suited for smaller watersheds. Harrington (1989) recommends the use of trenches on watersheds less than 10 acres (4 ha), but a more commonly reported size is 5 acres (2 ha) or less (Schueler et al., 1991, and Dorman et al., 1988).

A typical cross-section of an infiltration trench is presented in Figure 6.5. An important aspect of this design is the inclusion of the observation well with a removable cap. Harrington (1989) says that this feature allows for inspection of the trench to determine if it is functioning properly and facilitates the evaluation of the drain time between storms. The use of filter fabric around the stone reservoir and approximately 1 foot (0.3 m) below the



SI units: To convert inches to centimeters multiply by 2.54

Figure 6.5 Infiltration Trench Cross-Section
(Modified after Harrington, 1989)

trench surface also prevents the migration of fines into the larger pore spaces and prevents clogging failures. Buffer strips at least 20 feet (6 m) wide on each side of the trench assist in coarser sediment removal.

Clogging failures are common for infiltration trenches. A survey in Maryland found that the oldest working infiltration trench was 5 years old (Schueler et al., 1991). The same survey discovered that 20% of the trenches did not operate as designed immediately after construction. Less than 50% of the surveyed structures were operating properly after 5 years. Approximately one-third showed signs of chronic clogging. Construction practices and maintenance activities were most responsible for failure or reduced performance.

The State of Maryland (1985) provides specific construction and maintenance guidelines which are known to improve an infiltration structure's longevity and performance. Areas where infiltration facilities are to be built should be surveyed and roped off prior to any construction in the area to prevent vehicular traffic from compacting the underlying soils. Trench excavation should be done with a backhoe, wheel or ladder type trencher. Bulldozers and scrapers may seal the soil pores with their weight and/or equipment blades. Runoff should be diverted away from the facility until all adjacent construction is complete and all surrounding slopes are stabilized with vegetation to prevent erosion. The trench should be inspected for any roots or other objects which could puncture the filter fabric. Finally, each trench should be inspected at least once annually after a major storm to see if it is draining at its designed rate.

Yim and Sternberg (1987) conducted a comprehensive study of infiltration trenches in an attempt to refine infiltration trench design criteria. Of particular interest was the use of a granular filter medium on top of the trench base soil. The granular filter was believed to assist in maintaining the original permeability of the soil by trapping the finer sediments. An optimum grain size for the granular filter medium exists for a given infiltration rate. The general relationship is that, the higher the infiltration rate, the lower the required grain size to effectively filter out the fine sediments. Various empirical relationships for sizing the granular filter media are provided. For infiltration rates ranging from 0.016 to 0.645 cm/s (0.006 to 0.25 m/s), optimum granular filter media sizes were determined and found to maintain about 50% of the underlying soil's original permeability after about 7.5 years of service. Trenches with no granular filters and subjected to the same loading rates experienced a reduction of approximately one-order-of-magnitude reduction in base soil permeability after the same time period. The use of the granular filter effectively increased the life of the facility by a factor of two. Finally, the addition of the granular filter increased the sediment removal efficiency of the trench by about 40% to 45% under uniform sediment loads, compared to the trench with no granular filter.

Stahre and Urbonas (1989) describe the success of infiltration facilities (both trenches and basins) in Sweden. A more conservative design approach than is currently practiced in the U.S. provides more effective infiltration facilities in Sweden. The authors point out that when intensity-duration-frequency (I-D-F) curves are used to estimate design storm runoff (as is often the case in the U.S.), only the most intense portions of the storms are considered. The "tails" of rainfall before and after the intense period are omitted. However, an effective infiltration facility must be designed to hold all of the runoff associated with a design storm. A previous Swedish study determined that an adjustment of 25% to the Rational Formula improved runoff predictions for the sizing of infiltration facilities.

Another conservative approach used is to divide the field-determined infiltration rates by a safety factor of 2 or 3, in recognition of the fact that the soil pores will clog over time. When calculating infiltration areas of pits and basins, the floors are considered impervious. Only infiltration through the walls is considered. All of these design measures have greatly increased the longevity and performance of Swedish infiltration structures. Stahre and Urbonas (1989) present several design examples as well.

Kuo et al. (1989) developed a finite element model to simulate transient flow of water in a variably saturated porous medium. Pressure heads and soil moisture content distributions in the soil surrounding an infiltration trench were calculated as were fluxes across the boundaries. The predicted values using the model compared well to the experimentally measured values. Geometric design of trenches was found to be important. The infiltration rates for deep narrow trenches are higher than those for trenches which are shallow and wide, due to higher pressure heads in the deeper trenches. However, the larger horizontal surface area of the wide shallow trench allows a greater volume of water to infiltrate.

Costs for infiltration trenches are generally greater than those for pond systems, especially when based on unit of runoff per volume treated basis (Schueler et al., 1991). However, trenches are suitable for smaller watersheds where ponds cannot be used. A significant portion of design costs are incurred during the site investigation. Future EPA regulations concerning groundwater injection may require a permitting procedure. Limited maintenance costs are available. Based on the available data, if adequate maintenance procedures are ignored, trench rejuvenation or replacement may be required every 10 years. Such a cost could approach the original construction cost.

Schueler et al. (1991) also mention several areas pertaining to infiltration trenches which require additional investigation. Pre-treatment systems should be evaluated to determine the most effective method of increasing trench longevity. Effective maintenance routines and schedules are necessary, to increase trench performance. The removal efficiency of trenches located in sandy soils with shallow water tables needs evaluation. Finally, infiltration trench performance under freezing and snowmelt runoff conditions is unknown.

Infiltration basins operate in a manner similar to trenches, but with a few exceptions. One difference is that the runoff storage volume in infiltration basins is primarily ponded water in the basin. Most basins capture the runoff from a specified design storm. Usually, this amount is the first 1/2 inch (13 mm) associated with the first flush. Any additional runoff is either diverted to other discharge devices or overflows the top of the basin via an overflow spillway. These two differences are commonly referred to as "off-line" and "on-line" basins, respectively (Schueler et al., 1991). The increased surface area for exfiltration from the basins provides more potential for groundwater recharge than other infiltration techniques. Basins are also applicable to larger contributing watershed areas. Schueler et al. (1991) cite a range of 2 to 15 acres, while Dorman et al. (1988) state an upper limit of 50 acres (20 ha). In aquifer recharge zones, infiltration basins have successfully treated moderately polluted runoff (Whipple et al., 1987). However, in areas where chronic oil spills are possible, or in areas of sole-source aquifers, the use of infiltration basins should be carefully reviewed (Schueler et al., 1991).

To date, most infiltration basins have failed due to rapid clogging (Schueler et al., 1991). Of those investigated, 60% to 100% could no longer exfiltrate water after five years. Once basins are clogged, it is nearly impossible to restore their exfiltration capacity. Many are converted to wet ponds or wetlands. Large sediment inputs, large contributing watershed areas, and long dewatering times adversely affect the pollutant removal performance and decrease the basin's life. Deep pools of standing water should be avoided, since this tends to cause soil compaction. Control measures that reduce sediment input to the basin and/or off-line systems that bypass large storms with high sediment yields are also beneficial. Lining the basin floor with sand and installing back-up underdrains is another useful technique.

Aside from using the construction and maintenance techniques previously mentioned for infiltration trenches, several additional methods can be applied to infiltration basins. Burch et al. (1985) state that sediment should be removed from basins only when they are dry and that light tractors should be utilized to minimize soil compaction. Additionally, the authors recommend tilling the basin bed once per year with a disc harrow or rotary tiller, after sediment removal, to maintain the basin's exfiltration capacity.

Costs for infiltration basins generally range 10% to 20% higher than those of dry ponds (Schueler et al., 1991). Costs may increase significantly depending on improvements made to increase the basin life. Design costs involve substantial site investigation expenses. As with infiltration trenches, future EPA regulations may require a groundwater injection permit. Annual maintenance costs are projected to be about 5% of construction costs. However, reported maintenance costs are higher due to conversions to wetlands or ponds.

Additional studies are necessary to establish the effects of temporary deep ponding on the clogging of infiltration basins (Schueler et al., 1991). Pretreatment controls which can lengthen basin life also should be assessed. The risk of groundwater contamination and the performance of infiltration basins under freeze/thaw conditions also require study. Finally, more evaluations of actual infiltration basin pollutant removal efficiencies are necessary, particularly for soluble nutrients.

Lange (1990) evaluated several forms of highway runoff treatment practices used in the Federal Republic of Germany (FRG) for their impact on groundwater contamination. Groundwater below infiltration basins showed increases in certain inorganic constituents, but at levels still below the standards set for drinking water. Organics deposited in the basins were sorbed by the natural soils, or degraded by indigenous microorganisms. Chloride concentrations were the only constituents which sometimes exceeded the drinking water criteria. As a result, the FRG authorized infiltration practices for treating highway runoff, even in areas in close proximity to water resources.

Armstrong and Llena (1992) used fate and transport models for chemicals in soils and aquifers to assess the potential of contaminant leaching from soils which are beneath infiltration structures. Their work provided a general evaluation based on existing knowledge of pollutant properties and transport characteristics. Groups of organic (32) and inorganic (7) pollutants were selected for study based on their presence in runoff. By coupling the estimated distribution coefficient (K_d) and retardation factor (R_f) of a pollutant value with water loading and infiltration rates, predicted leaching rates were estimated. Results were presented as pollutant mobility index classifications. The indices were based on the calculated pollutant leaching rate relative to the water infiltration rate. The results also included estimated residence times of pollutants in specific types and depths of soils. Chemicals with K_d values less than 10mL/g could move through the soil at a rate greater than 1% the rate of a non-retained chemical. Twenty-four chemicals were mobile when the soil organic content (f_{oc}) was less than 0.01%, but when the f_{oc} value was increased to 1%, only 9 chemicals remained in the same classification. Most of the remaining 9 had octanol-water partition coefficients (K_{ow}) less than 10^3.

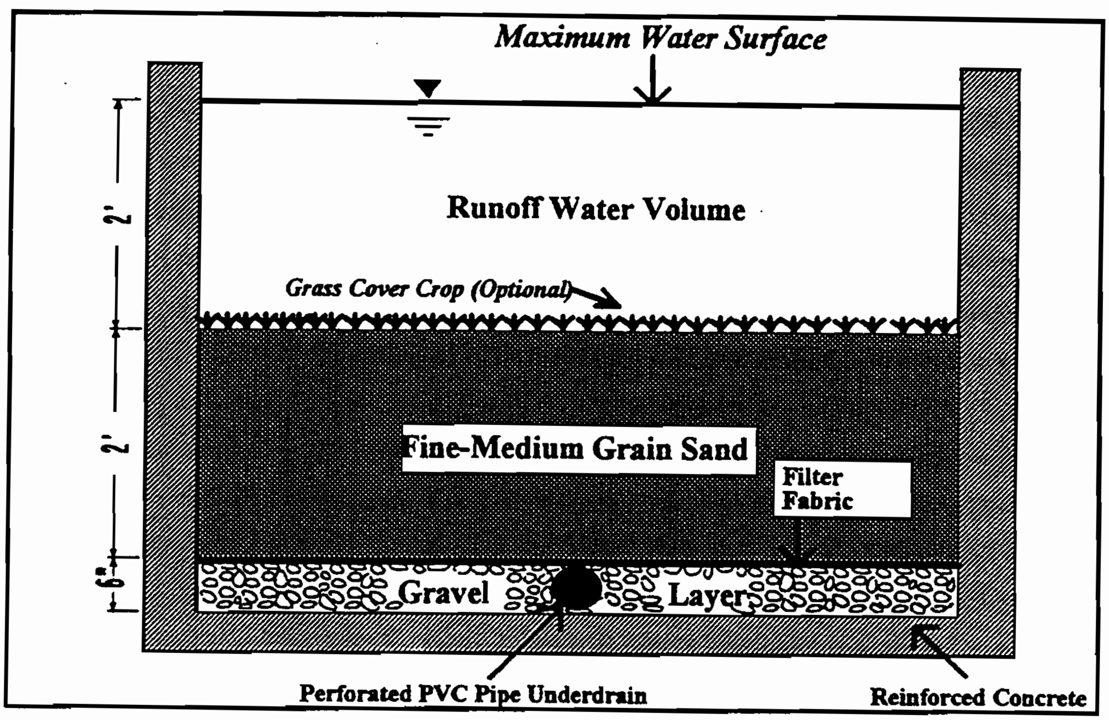
The model showed that inorganic pollutants are significantly less mobile than the organic compounds. The experimental published data for two "typical" soils was the basis for K_d values used in the evaluation of inorganic mobility. At low pollutant loading rates, no organic constituent received a mobile or intermediately mobile classification. At higher loadings, most inorganics displayed a low mobility ranking; expected migration was 0.1% to 1% that of a conservative chemical. Accuracy of the inorganic mobility is dependent on how well the soils beneath the infiltration facility represent the "typical" soils used in the mobility evaluations. Even when the f_{oc} value was 1%, a fairly high value, several polar pesticide compounds were predicted to be fairly mobile.

Infiltration site selection should minimize the potential for pollutant leaching by considering such factors as soil type, depth to groundwater, and water loading rate. For high loading rates, a few meters of soil beneath the facility will probably provide insufficient protection. Soils with high organic content can reduce leaching potential.

6.6 Filters

Sand filters are a relatively new treatment device for highway runoff (Schueler et al., 1991). Actually, the concept is similar to that of an infiltration basin, except that the treated water is discharged downstream rather than into the groundwater. The first flush of storm runoff is detained in the surface impoundment, which has a sand bottom covering drainage collection pipes. The bottom of the impoundment is impervious (i.e., concrete) or relatively impervious. Runoff is filtered through the sand, collected by the drains, and discharged downstream. A typical sand filter is illustrated in Figure 6.6.

Both on-line and off-line systems exist, but off-line systems work best since the larger storms with heavier sediment loads are bypassed. Pollutant removal is achieved primarily by straining through the porous media, and through gravitational settling on top of the sand bed. Removal rates are high



SI Units: To convert feet to meters multiply by 0.30
To convert inches to centimeters multiply by 2.54

Figure 6.6 Typical Sand Filter Cross-Section for TSS and trace metals, and moderate for BOD, nutrients, and fecal coliforms. Nutrient removal can be increased through plant uptake if a cover crop is planted on top of the sand layer.

Most of the sand filters in place today are in the Austin, Texas, area. Performance results have been good. Three sand filters in Austin have shown 85%, 35%, 40%, 40%, and 50% to 70% removal efficiencies for sediment, total nitrogen, dissolved phosphorus, fecal coliforms, and trace metals, respectively (Schueler et al., 1991).

Sand filters are useful in areas with thin soils, soils with low infiltration rates, and areas of high evapotranspiration (Schueler et al., 1991). Sand filters also pose little threat of groundwater contamination and are useful in areas of limited space. Most sand filters have contributing watershed areas of 0.5 to 10 acres (0.2 to 4 ha), with a maximum of about 50 acres (20 ha). Minimum sand bed thicknesses of 18 inches (46 cm) and drawdown times of 24 to 40 hours are necessary for effective pollutant removal. Head required to effectively operate the sand filters has usually been 2 to 4 feet (0.6 to 1.2 m). Of the approximately 1000 sand filters in place in the Austin area, public works officials state that most are performing as designed and few have failed. The oldest filter is about 10 years old.

The construction costs of sand filters range from \$3 to \$10 per cubic foot (\$100 to \$350 per m^3) of runoff treated (Schueler et al., 1991). This cost is about 2 to 3 times the cost for similarly sized infiltration trenches. One reason for increased costs is the use of structural concrete for the filter. Frequent (i.e., quarterly) manual maintenance is required, consisting primarily of raking, leaf removal, trash and debris removal, and surface sediment removal and disposal. Surface sediments from Austin's sand filters have been analyzed and can be safely landfilled. Since most maintenance is performed manually, the sand filter should be designed for easy access. Maintenance costs are estimated to be 5% of construction costs per year.

Some drawbacks and unknown qualities pertaining to sand filters do exist. Schueler et al. (1991) assert that sand filters provide very little flood control benefits. Sand filters also may appear unsightly, if no cover crop is planted. Odor problems have been associated with some sand filters. The impact of sand filters on downstream warming, and their performance in colder climate, should be evaluated. The frequency of surface sediment removal also requires further investigation. Finally, additional media combinations which could increase nutrient removal should be studied.

Welborn and Veenhuis (1987) evaluated a sand filter in Austin, Texas. The structure is an on-line system which treats runoff from an 80-acre site (32-ha), of which about half is impervious parking lots and roads. The sand bed consists of an 18-inch (46-cm) fine sand top layer, followed by a 12-inch (30-cm) coarse sand intermediate layer, followed by a 6-inch (15-cm) gravel layer with 6-inch (15-cm) perforated pipe underdrains. The pond bottom is lined with a 24-inch (10-cm) clay liner. The maximum pond depth is 14 feet (4.2 m), and the storage capacity is 3.5 acre-feet (5000 m^3). A total of 22 storm events were monitored over a two-year period, with total rainfall ranging from 0.14 to 2.88 inches (3.5 to 7.3 mm). All inflow to the device was filtered through the sand beds, except for three large storms which crested over the emergency spillway. Peak outflow from the filter was measured at 3.1 cfs (0.1 m^3/s). Average discharge rates tended to decrease during the duration of the study, as the sand bed became clogged. The filter was cleaned twice during the study, which caused peak and average discharge rates to improve, but not to the levels achieved when the filter was new. Peak and average discharges also lessened noticeably after larger storms, most likely due to the larger sediment loads associated with the storms.

The sand filter system was efficient in removing bacteria, suspended solids, BOD, total phosphorus, TOC, COD, and dissolved zinc. Average removals ranged between 60% and 80%. The average total dissolved solids (TDS) load was approximately 13% greater in the outflow than in the inflow. Possible explanations for the increase were the dissolution of previous deposits left on the filter, leaching from the pond bed and sand filter, and mineralization of the organic material deposited on the pond bed. Organic nitrogen and ammonia nitrogen concentrations in the inflow were substantially larger than that in the outflow. Total nitrate plus nitrite levels in the outflow were about 110% larger than the inflow concentrations. These measurements indicate that nitrification occurs in the pond.

Galli (1990) describes an enhanced sand filter design which incorporates peat into the filter material. Peat is a highly organic, complex material which is primarily composed of cellulose and of humic and fulvic acids. Its structure ranges from open and porous, to granular and colloidal-size. The porous peats tend to have the highest water holding capacities. Measured hydraulic conductivities of peat range from 0.025 cm/hr (0.01 in/hr) to 140 cm/hr (55 in/hr). Low bulk densities are also common. Peats also exhibit a high buffering capacity. The adsorptive surface area reportedly is 2 to 4 times that of montmorillonite clay. Peat also maintains a high cation exchange capacity which is particularly good for copper, zinc, lead, and mercury. The carbon:nitrogen:phosphorus composition ratio of peat is around 100:10:1, providing substrate for microbial growth. Pure peat materials typically contain large populations of nitrifying and denitrifying organisms. Although phosphorus assimilation in peat has been reported, phosphorus retention in peat appears to be more closely linked to the calcium, aluminum, iron, and ash content in the peat. All of these qualities make peat a useful additive for sand filters.

Galli points to earlier studies which evaluated the effectiveness of peat for sewage treatment. Removals were high (i.e., greater than 80%) for nutrient, BOD, and pathogenic bacteria. Peat has also been effectively used to treat electroplating wastewaters and to clean up oil spills. The peat-sand filter was first tested in the early 1970's, consisting of a 10- to 30-cm (4- to 12-in) peat layer on top of a 75- to 90-cm (3- to 35-in) layer of fine sand. Grass was planted on top of the peat. Removals achieved were greater than 90% for phosphorus, 98% for BOD, and 99% for fecal coliforms. Improvements since then include a multi-layered design. The top layer is 12- to 18- inches (30- to 46-cm) of peat with calcitic limestone mixed in to enhance phosphorus removal. The middle layer is a 4-inch (10 cm) thickness of 50% peat-50% sand mixture. This layer assists in providing a uniform flow through the bed and helps to increase the peat-water contact time. The bottom layer is a 6-inch (15-cm) gravel layer with a perforated PVC pipe underdrain.

A peat-sand filter has been constructed in Maryland where an existing off-line infiltration basin failed. The contributing watershed area is 140 acres (55 ha). Although removals have not been evaluated, estimated removal efficiencies for TSS, total P, total N, BOD, trace metals, and bacteria are 90%, 70%, 50%, 90%, 80%, and greater than 90%, respectively. Since the peat-sand filter performs best during the warmer months, a wet pond precedes the filter in order to provide limited treatment during the winter when the peat-sand filter is bypassed. The pond also provides some sediment removal.

Design requirements for sizing peat-sand filters for treating runoff are not rigid. Generally, an increase in the areal pollutant and hydraulic loadings corresponds to an increased requirement for the peat surface area. One general rule of thumb provided is that 0.5 hectares (1.2 ac) of peat surface area is required for each 100 hectares (250 ac) of contributing watershed area treated. Galli stresses the importance of analyzing peat for hydraulic conductivity, CEC, iron, aluminum, calcium carbonate, ash, and nutrient content prior to bulk purchase. He notes that negative nutrient removal may also be experienced during filter start-up as some nutrients may wash out from the peat.

Known maintenance requirements for peat-sand filters include trash and debris removal, sediment removal, and grass cover mowing and clippings disposal.

More research is required pertaining to the amount of sediment which can be deposited on peat before filter efficiency is diminished. The effects of different peat hydraulic conductivities on overall removal efficiencies are also unknown. The sizing relationships for designing peat-sand filters must be studied as well. In addition, the effects of optimum peat mixtures and thicker peat depths on performance and longevity require investigation.

6.7 Performance Enhancements

Several performance enhancements for highway runoff controls are mentioned in the literature. These enhancements are primarily structural additions to pre-treat the influent or to polish the effluent, but effective maintenance management measures can also increase the pollutant removal capability and longevity of a structure (Schueler et al., 1991). Specifically, oil/grit treatment and separators, sediment forebays, and granular-activated carbon (GAC) filters will be discussed.

6.7.1 Oil/Grit Treatment

Oil and grit chambers are often used in conjunction with highway runoff controls to remove heavy particulates and adsorbed hydrocarbon particulates (Schueler et al., 1991). These structures cost around $8000 per unit, which is 3 to 4 times the cost per unit volume of treated runoff associated with infiltration trenches. Berg (1991) mentions that these structures are intended for small contributing watersheds, usually 1 acre (0.4 ha) or smaller. Frequent maintenance and cleanout are recommended on a quarterly basis, and the associated operating costs are high. Of 500 constructed in Maryland, only a few have been properly maintained, causing retained oil and sediments to be released downstream during intense storms. If properly maintained, these systems can function as pre-filtering devices for other runoff controls.

Silverman et al. (1989) acknowledge the relative ineffectiveness of oil and grit chambers. They cite studies which showed that 40% to 60% of oil and grease associated with urban runoff is in a dissolved or colloidal state. Thus, the classic oil and grit separators which are designed to separate free-floating oil and grease products would exhibit low removal efficiencies for urban and highway runoff. According to Schueler (1991), actual removal efficiencies for water quality inlets used as runoff controls have yet to be studied. Silverman et al. (1989) show that the most effective treatment for oil and grease in highway runoff is one of source reduction. The use of porous pavements, greenbelts, storm drain inlet adsorbents, and wet pavement scrubbing are all shown to provide beneficial oil and grease reduction.

Kile et al. (1989) designed a pre-cast concrete sedimentation/filtration system for curb and gutter inlets. A schematic diagram illustrating the design features is presented in Figure 6.7. The two box chambers at the inlet serve as sediment traps prior to the filtration chamber. The 6 ft x 6 ft x 6 ft (1.8 m x 1.8 m x 1.8 m) concrete filtration chamber has a PVC underdrain beneath 18 inches (46 cm) of sand, followed by filter fabric, with 6 inches (15 cm) of sand on top. The authors offer no expected pollutant removal efficiencies or recommended maintenance practices for this device.

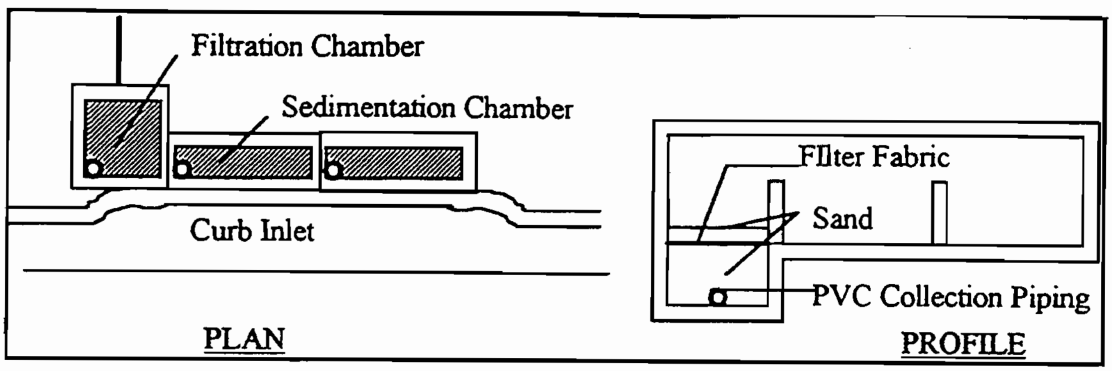


Figure 6.7 Moduled Sedimentation/Filtration Chambers
(Modified from Kile et al., 1989)

6.7.2 Sediment Forebays

Forebays are useful devices for enhancing the performance of most runoff control devices. They provide an extra storage space, or pool, prior to the inlet of the runoff control device and remove a fair amount of the initial coarse sediment load (Schueler et al., 1991). Sediment forebays are particularly useful for infiltration structures and sand filters, as they inhibit the more rapid clogging failures. Driscoll (1989) showed the effectiveness of using even a small forebay in sediment removal prior to entering a secondary control device. Using a probabilistic model, National Urban Runoff Program particle size/settling velocity mass percentile groupings, and rainfall statistics typical of the eastern U.S., the performance of a forebay 1 foot (0.3 m) deep with a surface area of 43.5 square feet (4 m^2) was evaluated. The contributing watershed area was 1 acre (0.4 ha) of completely impervious material. The predicted results are presented in Table 6.3. Notice that the forebay's performance is best during dynamic conditions. This is the desired result for a forebay. Also, the forebay removes an estimated 25% to 30% of all incoming TSS, and approximately 80% of the largest incoming fraction. The performance estimate also assumes that a regular maintenance program removes accumulated sediment from the forebay.

6.7.3 GAC Filters

The adsorptive properties of granular-activated carbon (GAC) are often used to capture organic compounds in industrial air and wastewater streams. In Florida, concern over the trihalomethane-forming potential (THMFP) of the organic constituents associated with highway runoff was increasing in areas where runoff was discharged directly to underground drainage wells (Wanielista et al., 1991). From 1905 until 1970, about 400 drainage wells were constructed in Florida in an attempt to reduce some runoff flooding problems. The practice was halted in 1970 amid increasing suspicion of groundwater contamination.

A detention pond (3.2-acre surface area) which receives runoff from an Interstate highway and a commercial area (130-acre contributing watershed area) and discharges into a 200-foot-deep drainage well in an aquifer that is an underground source of drinking water (USDW) was retrofitted with a GAC filter bed prior to the drainage well discharge. Measurements of the THMFP

Table 6.3
Estimated Performance of a Sediment Forebay
(from Driscoll, 1989)

Rainfall-Runoff Event Statistics

	Mean	COV
Volume (inches)	0.40	1.50
Intensity (in/hr)	0.07	1.30
Duration (hours)	6.00	1.10
Interval (hours)	90.00	1.00

Forebay Performance Estimates

Size Fraction	Settling Velocity (ft/hr), Vs	Effective Vol. Ratio, VE/VR	Dynamic Removal %	Quiescent Removal %	Combined Removal %
1	0.03	0.01	1.4	0.4	1.0
2	0.30	0.02	5.0	1.2	6.1
3	1.50	0.03	12.5	2.2	14.5
4	7.00	0.03	32.5	2.4	34.2
5	65.00	0.03	80.6	2.4	81.0

SI units: To convert inches to centimeters multiply by 2.54
To convert ft/ha to m/hr multiply by 0.3

of the water before and after carbon treatment noted a significant amount of treatment. Approximately 6.3 milligrams of TOC was adsorbed per gram (6.3 lb per 1000 lb) of activated carbon.

However, the GAC treatment was expensive. The annual cost to treat the water for THMFP precursors was calculated as $316,000, or $4.39/1000 gallons ($1.15/1000 L) treated after detention and before well injection. Because of the rapid breakthrough experienced in the GAC beds, replacement of the carbon would be required after every 1-inch (25 mm) storm event. Regeneration of the used carbon was estimated as more expensive than the purchase of new carbon. The recommended use for the spent carbon was as a replacement for sand in concrete.

A separate problem encountered involved the rapid growth of bacteria in the GAC bed. The bacteria were associated with iron and manganese which were present in high concentrations in the groundwater below the pond's underdrain system leading to the carbon bed. The bacteria formed layers in the pipe and in the bed which sloughed off, partially clogging the bed inlet pipe and the underdrains. The problem was alleviated by surface-draining the pond, which eliminated the groundwater intrusion.

Although the GAC filter performed well in reducing the THMFP of the runoff, it was considered a cost-prohibitive procedure for this application. The study recommends applying the pond water as irrigation to the highway median and shoulder vegetation as a more cost-effective measure for reducing THMFP in the USDW.

6.8 Combined Systems

Combined systems link the use of the previously mentioned control devices for enhanced and more uniform overall pollutant removal performance. In fact, a combination of runoff control measures is recommended whenever possible (Burch et al., 1985a). Combinations may increase the ability to effectively filter suspended solids, or may be useful in reducing the site limitations for a single control measure. Vegetative controls are the only control structures which treat the runoff as it is conveyed; therefore, these systems are recommended wherever possible as collection and conveyance links between treatment systems (Burch et al., 1985a).

When treatment systems are combined two restrictions exist. First, infiltration devices should be the last structures in the treatment train, since these systems are adversely affected by high sediment loads. Preceding structures should remove as much of the suspended sediment as possible to increase the effectiveness and longevity of the infiltration devices. Second, wetlands should not be used in conjunction with infiltration practices. Wetlands have the potential to discharge large sediment loads and decaying matter which can clog infiltration devices. Wetlands are best positioned in the middle of treatment trains. They should discharge to ponds or vegetated control structures.

One design combination involves ponds. Schueler et al. (1991) describe multiple pond systems (MPSs) as combinations of wet ponds, extended detention ponds, and wetlands in series. No one MPS design exists; the flexibility of the design allows the designer to minimize negative impacts associated with one system and maximize site-specific conditions. The redundancy of expected pollutant removal efficiencies increases the overall reliability and performance of the system. Many MPSs have demonstrated increased pollutant removals over single treatment systems. Different designs utilize longer flow paths, longer retention times, and more quiescent settling conditions to achieve the noted improved performance and reliability. Longevity is expected to be at least as long as for single ponds or wetlands. However, MPS designs may have longer life spans, since one part of the design may protect the long-term performance of another. Because of their complexity, costs associated with MPS designs are usually higher than those for regular ponds and wetlands. Maintenance is comparable to the single systems. In some cases, the MPS maintenance burden may actually be less due to the design.

Gain and Miller (1989) evaluated an MPS system which is a wet pond-wetlands design. The evaluation focused primarily on the removal efficiency of dissolved pollutants, particularly phosphorus, zinc, and lead. The wet pond has an approximate 0.2-inch (5 mm) runoff "live" storage space. The wetlands receive water from a spillway at the end of the pond and drain into a drainage channel. During a 22-month period, 22 storm events were monitored. Total runoff ranged between 0.10 inches and 0.63 inches (2.5 mm to 16 mm), and intensity ranged from 0.037 in/hr to 0.972 in/hr (0.94 mm/hr to 24.6 mm/hr). The durations of most storms were about 5 hours, three were greater than 5 hours, and the maximum duration was around 11 hours.

A tracer dye study in the pond revealed that once the pond neared its maximum live storage capacity, the water began to short-circuit through the center of the pond in about 5 to 15 minutes, failing to displace or mix with any of the water in dead storage. If the entering runoff was significantly warmer than the water already in the pond, thermal stratification might cause the runoff to float across the pond's surface.

The entire system is relatively ineffective at treating the dissolved chlorides. No net removal of these pollutants was observed. Removals of total dissolved solids (TDS) was less in the pond than in the wetlands. The belief is that sediments in the pond bottom dissolve into the dead storage space and are discharged to the wetlands. Also, since the pond has limited biological activity, fewer TDS materials are used through biological uptake. The TDS removal in wetlands is slightly better. The greater biological activity and the larger amount of organic matter on the pond bottom available for adsorption probably account for the increase. In general, however, overall TDS treatment by the system is poor.

Dissolved phosphorus concentrations are much greater in the wetlands than in the pond. The biological activity in the wetland causes the wetland to cycle phosphorus at a faster rate than the pond. The pond shows higher retention rates of total phosphorus. One reason provided is that the lessened pond biological activity corresponds to limited biological phosphorus assimilation. Also, studies have indicated that iron and aluminum presence may increase the retention of phosphorus. Under anaerobic conditions, dissolved iron and phosphorus in the pond sediments may diffuse upward into the aerobic zone of the pond, where the two metals combine with the phosphorus and form insoluble solids that settle back to the pond bottom. Dissolved oxygen levels of nearly zero at the bottom of the pond seem to support this explanation of increased phosphorus retention.

Virtually all of the lead is removed in the pond. Some of the zinc stays in the pond, but the majority is discharged into the wetlands. Since most of the lead and zinc entering the pond is in the suspended form, both metals initially settle to the pond bottom. However, zinc is fairly soluble under anaerobic conditions, while lead solubility remains low. Zinc leaches from the pond bottom into the water and moves as dissolved zinc to the wetlands. Biological activity in the wetlands takes up some of the zinc. The remaining zinc, now under aerobic conditions, precipitates as zinc oxide.

Wulliman et al. (1989) described the use of a wet pond, followed by a series of infiltration/wetland areas. The design was developed to achieve an overall total phosphorus removal of 50%. Based on predictive modeling for the design, the most conservative estimated phosphorus removal is 52%; the most optimistic removal is 87%.

In California, a system draining a 39.5-acre (15.6 ha) auto auction site uses an oil and grit chamber to treat runoff from the site. Oil-adsorbing pillows floating on top of the chamber collect the oil and grease in the runoff. Water from the chamber is drained through an 1100-foot-long (330-meter-long) combination grassed swale/infiltration ditch. No estimation of treatment efficiencies or maintenance activities is provided.

Holler (1990) evaluated a combined extended-detention/filtration system for total phosphorus and total orthophosphorus removal efficiency. The general design of the facility is shown in Figure 6.8. The pond is 200 ft x

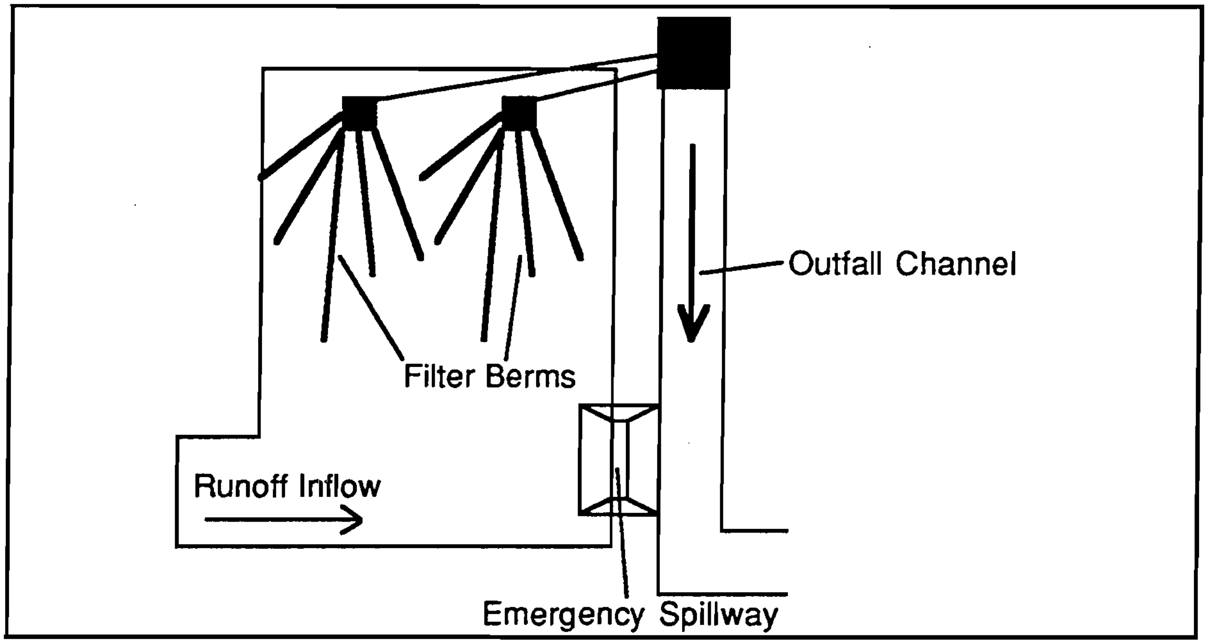
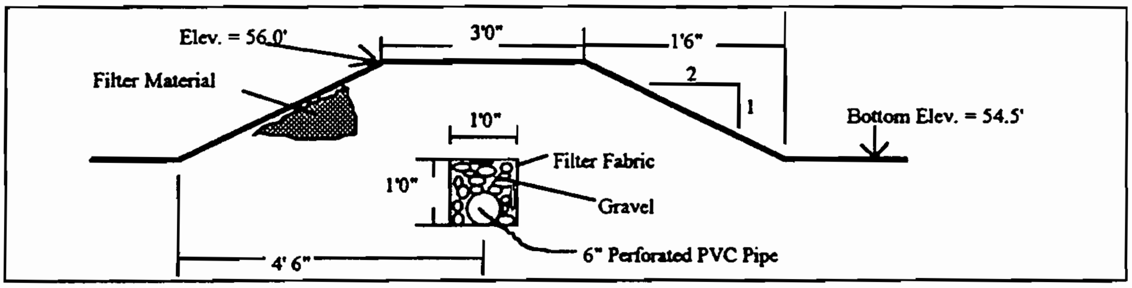


Figure 6.8 Site Layout of the Combination Detention/Filtration Pond (Modified from Holler, 1990)

400 ft (60 m x 120 m) , and designed for a maximum 3-foot (0.9-meter) storage depth. The storage capacity is for the first 0.5 inches (13 mm) of runoff (1.46 acre-feet (2,100 m^3)) from the 75-acre (30 ha) urban/commercial contributing watershed. Excess runoff overflows the emergency spillway into the drainage channel. Filtration berms, each 100-feet (30 meters) long, are located at one end of the pond and slope slightly to one of two concrete drop boxes. Five filter berms extend radially from each drop box. A filter berm cross-section is shown in Figure 6.9. The filtration medium is a combination of limestone, sand, and native fill. The 6-inch (15 cm) PVC pipe underdrains connect to the drop box. The drop box tops are screened at 0.75 feet (23 cm) below the emergency spillway elevation to allow for berm overflow discharge. The intended extended-detention time is 48 hours.

Six storm events were monitored during a 1-year period. The basin water level receded much more slowly than the design values. Head losses of about 0.11 ft/day (3.4 cm/day) were typical. Apparently, the design lacks sufficient head to effectively operate the system. Severe reductions in the designed filter berm surface loadings also occurred. This probably happened because of the rapid growth of vegetation on top of the berms, and because the media became clogged with fines.



SI units: To convert feet to meters multiply by 0.70.
To convert inches to centimeters multiply by 2.54.

Figure 6.9 Cross-Section of Filter Berm
(Modified from Holler, 1990)

Statistical analysis of the data showed significant treatment for both total phosphorus and total orthophosphorus in the extended-detention pond. The average removal for both was 77%. Statistical analysis of the measured removals at the filter berms showed no statistical significance between pre- and post-filter berm concentrations. Therefore, the filter berms apparently offer no additional treatment for total phosphorus or total orthophosphorus. The treatment measured for each component and for the total system is summarized in Table 6.4. Holler hypothesizes that the vast improvement shown for filtration berm treatment of orthophosphorus during storms 4 through 6 may indicate a filter ripening process.

Table 6.4
Detention Pond/Filtration Berm Phosphorus Treatment Potential
(from Holler, 1990)

Event No.	Treatment					
	Wet Detention		Filtration		Overall	
	$o\text{-}PO_4\text{-}P$	$T\text{-}PO_4\text{-}P$	$o\text{-}PO_4\text{-}P$	$T\text{-}PO_4\text{-}P$	$o\text{-}PO_4\text{-}P$	$T\text{-}PO_4\text{-}P$
1	83	91	-321	-86	29	83
2	77	80	-250	8	19	82
3	76	52	-92	60	54	81
4	78	84	26	37	84	90
5	81	82	50	38	91	89
6	69	71	42	38	82	82
$\bar{x} \pm s$	77 ± 4	77 ± 13	-91 ± 147	16 ± 48	60 ± 28	85 ± 4

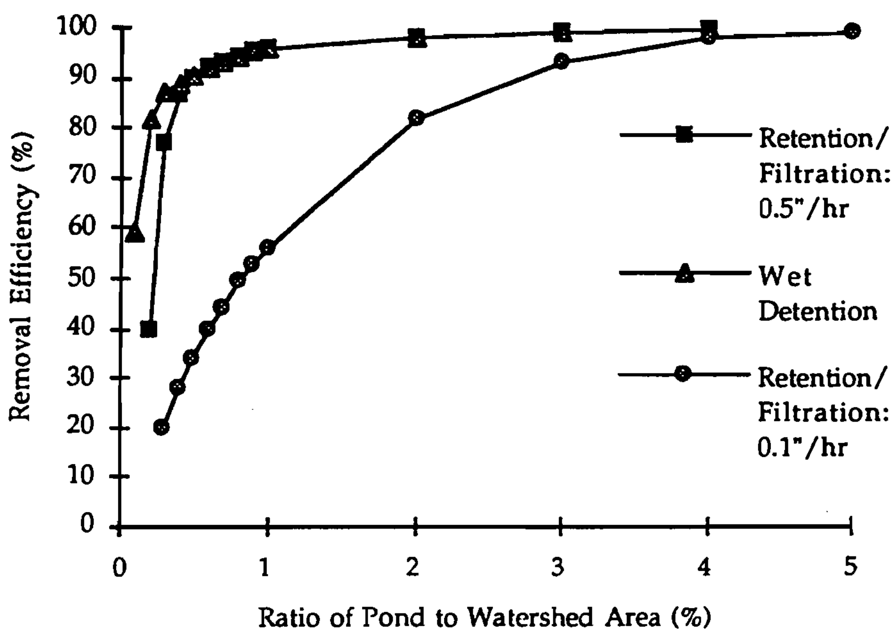
* ALL VALUES IN PERCENT

6.9 Comparative Studies

Several comparative analyses of the relative effectiveness of highway runoff control structures are in the literature. Studies have documented the differences between the performances of various control structures. The comparisons are either estimated or measured. The influence of different types of structures on groundwater has also been evaluated. Studies and surveys of relative maintenance requirements and practices are also available.

6.9.1 Predicted Performance

Maristany (1989) used a predictive model from the EPA, specific storm data, and structural dimension data to compare TSS removal efficiencies of wet ponds and infiltration basins. The ponds evaluated are located in Florida and have contributing watershed areas ranging from 128 to 23,393 acres (50 to 9,240 ha). The percent imperviousness of the watershed ranges from 9% to 54%. A graphical comparison of some of the results is presented in Figure 6.10. According to the estimates, at percolation rates between 0.1 in/hr (2.54



SI units: To convert in/hr to cm/hr multiply by 2.54

Figure 6.10 Comparison of Treatment Efficiencies for TSS of Wet Ponds and Retention/Filtration Basins (Modified from Maristany, 1989) mm/hr) and 0.5 in/hr (13 mm/hr), TSS removal efficiencies are better for wet ponds. The decrease in the infiltration basin's performance shown for percolation rates of 0.1 in/hr (2.54 mm/hr) indicates the infiltration basin's sensitivity to clogging and decreased soil permeabilities.

Driscoll et al. (1986) used U.S. Weather Service data along with watershed area information and a probabilistic analysis procedure to develop a computer code that will estimate long-term pollutant removal for various runoff control structures subjected to different operating conditions. The predicted results match actual measured performance data quite nicely in most cases. The authors point out that the program is a useful tool for optimizing a structure's design parameters.

6.9.2 Field Evaluations

Personnel of the City of Austin, Texas (1990), performed evaluations of six runoff water quality control structures including one wet pond, two detention/filtration basins, and three sand filters. The two detention/filtration facilities were off-line systems, designed to treat the first 0.5 inch (13 mm) of runoff. The other four structures were on-line systems. Average drawdown and detention times were calculated for the structures. Drawdown time is the time required for the total outflow to pass through the structure (the time range of the outflow hydrograph). The detention time is the volume of inflow divided by the outflow discharge. Since the discharge rate is not constant, the detention time was estimated as the time distance between inflow and outflow hydrograph centroids for a structure. For the 0.5-inch (13-mm) runoff events, average drawdown times ranged from 20 to 26 hours, and average detention times were from 4 to 6 hours. The Drainage Criteria Manual (1991) prepared by the City of Austin recommends drawdown times of 40 hours or less.

The measured ranges of removal efficiencies for the structures are presented in Table 6.5. The study notes that detention times greater than 4 hours may be necessary to increase the removal efficiencies of wet ponds. An increased dry pond detention time also increases the removal efficiency. Even so, with a relatively high detention time of about 6 hours, the removals observed for dry ponds were less than those associated with the filtration basins and the wet pond. Effective detention time and drawdown time for the filtration basins were determined to be 6 and 25 hours, respectively.

Table 6.5
Removal Efficiencies (%) Measured at City of Austin Facilities
(Modified from City of Austin, 1990)

System	TSS	BOD	COD	TOC	NO_2+NO_3	TN	TPO_4	Metals
Filtration	70-87%	15-51%	34-67%	44-61%	-82 to -26%	18-32%	3- 61%	19-86%
Wet Pond	46%	30%	31%	-9%	36%	29%	37%	41-72%
Dry Ponds	16%	23%	8%	18%	43%	22%	3%	-64-19%
Retention/ Filtration	86%	59%	82%	87%	-38%	47%	65%	71-84%

The filtration basins used a 12-inch to 18-inch (30- to 46-cm) layer of fine sand (0.02-inch- to 0.04-inch- (0.05- to 0.10-cm) diameter), and a layer of 0.5-inch to 2-inch (1.3- to 51-cm) gravel surrounding the underdrains. Some filtration basins had geotextile fabric placed over the sand layer to facilitate easier sand rejuvenation; the filter fabric was removed and replaced instead of the sand. Regular removal of the deposited sediment on top of the filters was required to maintain optimum performance and to prevent clogging. When this procedure was not performed, the drawdown time reached several days. This reduced the overall efficiency of the infiltration basin, since the design runoff of the following storm could not be totally captured.

The off-line systems performed better than the on-line ones during concurrent rainfall-runoff events. The on-line systems were unable to capture all of the pollutants. Overflow also resuspended some of the previously deposited sediments and carried the solids out of the structure. In all cases, sediment traps and forebays upstream of the filtration basins assisted in maintaining a working filter.

Yu and Benelmouffok (1988) compared the removals of a wet pond and a level spreader/vegetated buffer strip system (LS/VBS). The pond has a surface area of 1.67 acres (0.66 ha), an average depth of 7.6 feet (2.3 m), and a contributing watershed of 137 acres (54 ha) adjacent to a highway and commercial area. The level spreader drains an 18-acre (7.1-ha) area near a highway and shopping center complex. Water flows into an infiltration trench/detention, area then overflows downslope through a 150-foot-long (45-m) vegetated buffer strip. Five rainfall-runoff events were monitored for the pond and three were monitored for the LS/VBS. The data in Table 6.6

Table 6.6
Overall Pollutant Removal Efficiency (%), Effect of Filter Strip Length
(from Yu and Benelmouffok, 1988)

Pollutant	Length of Filter Strip, (ft) 20	40	70	150
Suspended Solids (TSS)	28	40	70	71
NO_3+NO_2	2	6	11	10
Total Phosphorus (TP)	23	14	28	38
Pb	2	18	20	25
Zn	18	24	51	51

Si units: To convert feet to meters multiply by 0.30

show that a 70-foot-long (21-m) strip is virtually as effective as a 150-foot-long (45-m) strip for removal of most of the measured pollutants. The measured average pollutant removals for both systems are presented in Table 6.7. Removal efficiencies for TSS and zinc are similar for both facilities, but the wet pond is better at removing nitrite-nitrate nitrogen, total phosphorus, and lead.

Table 6.7
Overall Removal Efficiency of Two Systems
(Modified from Yu and Benelmouffok, 1988)

Pollutant	Wet Pond	LS/VBS (Full Length)
Suspended Solids (TSS)	77	71
Total Phosphorus (TP)	70	38
Nitrite-Nitrate	75	10
Lead	57	25
Zinc	50	51

The Metropolitan Washington Council of Governments (MWCOG) evaluated several existing control structures (MWCOG, 1983). Removals were statistically calculated in three ways: by averaging the average removal for each storm, by determining the median load from all storms for both inlet and outlet and calculating the median removal efficiency, and by summing the entire total inflow loads and total outflow loads over time and calculating the long term removal efficiency. The differences in total removal efficiencies for the three types of ponds evaluated are summarized in Figure 6.11. The dry detention ponds (1- to 2- hour detention time) are relatively

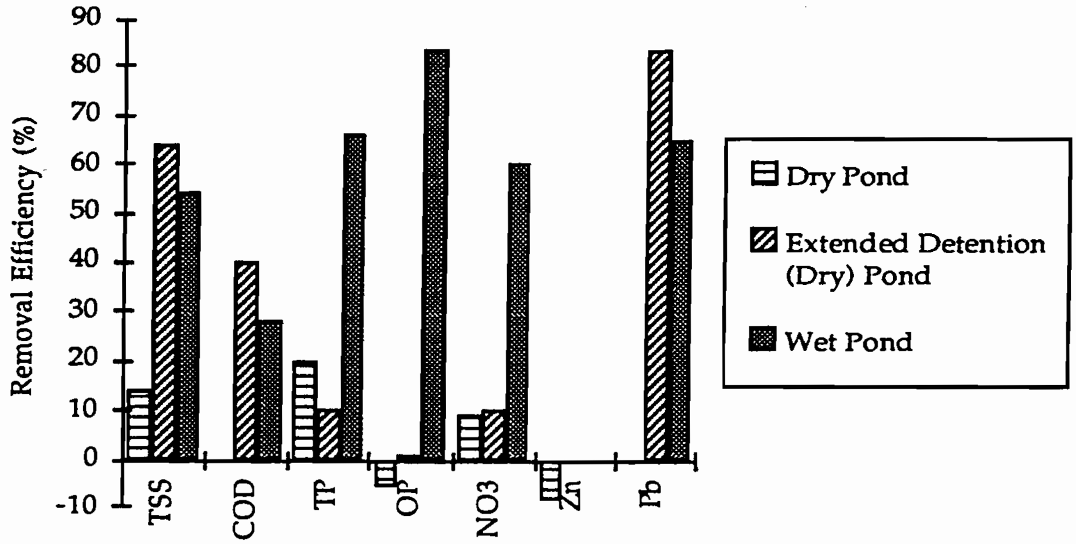


Figure 6.11 Pollutant Removal Efficiencies Associated with Different Pond Structures (Modified from MWCOG, 1983)

ineffective compared to the extended-detention (about 8 hours) and the wet ponds. The increased particulate removal associated with increased detention times is evident when comparing the dry pond with the extended detention pond. Nutrient removals for these two types of ponds are lower than those of the wet pond, since inadequate time is available for biological assimilation and flocculation/sedimentation of the finer particulates associated with the nutrients to occur.

The data presented in Figure 6.12 clearly show the influence of larger storms on particulate removal as opposed to soluble nutrient removal. Since the larger storms are associated with increased particulate loads, the total removal efficiency calculation shows the increase in particulates associated with the larger storm events. Since an increase in nutrient load generally is not associated with an increased storm size, the three methods of removal efficiency calculation show similar values for the nutrient removal efficiencies.

Swales evaluated did not display appreciable pollutant removal results. A careful analysis of the swale sites used indicated four possible reasons. First, all of the swale sites monitored had silt soils beneath them. Second,

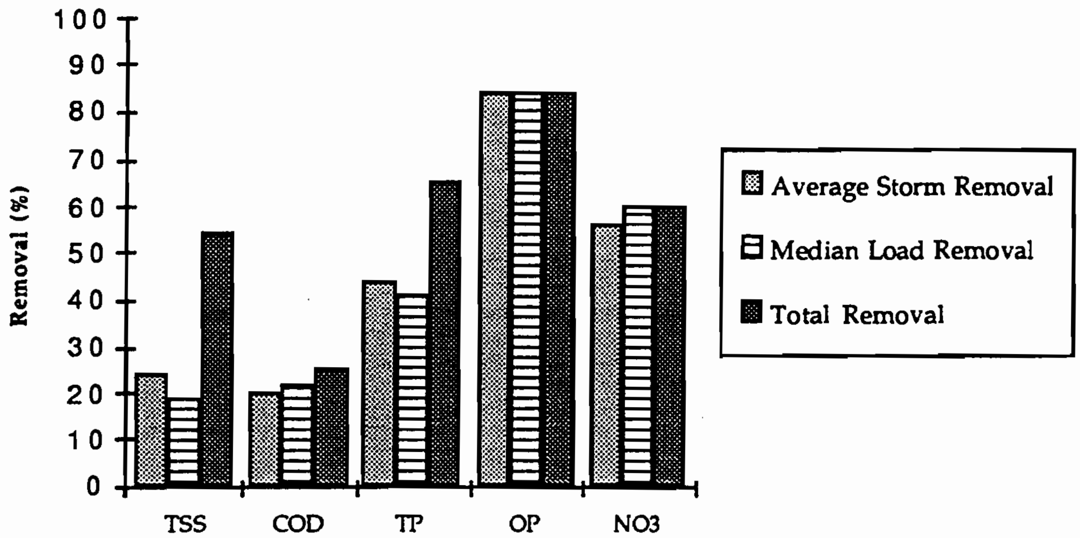


Figure 6.12 Comparison of Independent Methods Used to Calculate Wet Pond Removal Efficiencies (Modified from MWCOG, 1983)

measured contact times for runoff in the swale were from 5 to 10 minutes, while the underlying silt loams required much longer times for percolation. Third, the swale slopes were from 4% to 6%. Often, the steep slopes caused an increase in the downstream water velocity, which in turn matted the grass, and conditions in the swale approached those of open flow. Finally, grass in the swales may have been mowed too frequently to effectively lower the swale water velocity.

The MWCOG cite settling studies which show that a detention time of 6 to 12 hours in ponds removes approximately 67% of the theoretical upper limit for TSS. The differences associated with 6-hour and with 48-hour settling times measured in the settling column tests are presented in Figure 6.13. Since settling times greater than 12 hours achieve only incremental increases in removal rates, and the cost of additional storage space in a pond is a relative constant, the authors point out that designing for long detention times should be closely scrutinized. However, the authors recommend design retention times of 24 to 48 hours for infiltration structures, since these devices normally have smaller storage space and throughput.

6.9.3 Maintenance Considerations

As previously mentioned, maintenance practices affect the performance and longevity of control devices designed to mitigate highway runoff. The amount and type of maintenance have an important bearing on which control systems are appropriate for a given application. Many authors and agencies have published specific maintenance guidelines for runoff controls. These recommendations often are specific to a certain area of the

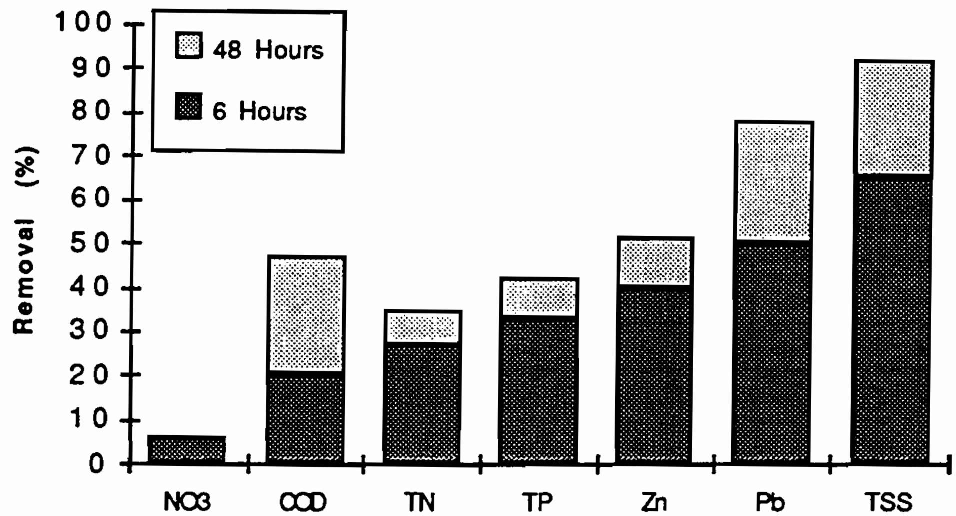


Figure 6.13 Observed Removal Rates in Experimental Settling Column Studies After 6 and 48 Hours (Modified from MWCOG, 1983)

United States. The State of Maryland, Washington Department of Transportation, City of Austin, Texas, and the MWCOG have published extensive manuals which provide general maintenance recommendations, as well as specific criteria particular to the respective geographic region. However, some authors have recommended general maintenance efforts not previously identified in this review that are not region-specific.

Pazwash (1991) recommends trash racks for the outlets of all ponds. These trash retainers should be cleaned out at least once in the spring and once in the fall. Detention ponds should have a concrete or rip-rap-lined low-flow channel constructed through the basin bottom. The channel prevents water from ponding and prevents a soggy basin bottom from forming between storms. This allows for ease of mowing the basin, and prevents wetland plant establishment. Pazwash suggests that swales be irrigated during times of low moisture content to ensure the survival of vegetation.

Roenigk et al. (1992) conducted an extensive maintenance survey of over 800 stormwater control structures in four North Carolina cities. The structures surveyed include conveyance structures (e.g., culverts, inlet devices, channels, etc.), detention ponds, wet ponds, and infiltration trenches and basins. Although the survey focused on the maintenance efforts required to meet a structure's flood control design, the survey did offer some insight into the level of maintenance needed to provide water quality improvement.

The conditions of the facilities were rated as good, adequate, inadequate, or dangerous with respect to various maintenance activities required at a given type of structure. The conveyance structures typically had fewer than 5% rated as inadequate or dangerous due to maintenance activities, while ponds and infiltration trenches and basins had the worst ratings. Since the survey evaluated only the flood control capability of a structure and did not evaluate water quality, a higher maintenance failure rate may actually exist for these structures with respect to water quality. The report suggests that as the concern over runoff water quality increases, the level of required maintenance will increase and overall maintenance ratings may decline.

In a phone survey of city stormwater officials in all North Carolina cities with populations in excess of 5,000 individuals, respondents were asked to cite the runoff control structure maintenance activities which were performed on a regular basis. Mowing was the most regularly performed task (58%). Inlet cleaning was the next most frequently conducted activity (56%) followed by facility inspections (35%). Regular sediment removal at detention facilities was the least frequently cited activity (8%).

Of the cities surveyed which had retention/detention facilities, less than 50% had conducted sediment removal. Facilities which had undergone sediment removal averaged 10 years of use prior to sediment removal. Most removals were performed due to the observed reduced effectiveness of the facility rather than as scheduled maintenance. Although most experts recommend a scheduled preventive maintenance approach, most communities surveyed used a reactive maintenance approach. The survey found no apparent cost differences between the two.

Four localities had their structures field-surveyed. This survey included a total of 46 retention/detention facilities. The retention/detention facilities had the most problems attributed to maintenance practices when compared to the culverts, lined channels, and vegetated channels. The maintenance requirements of the retention/detention facilities were greater, but were overlooked most often. The ponds and basins were cited as dumping grounds in three of the four cities. The ponds also were the leaders in the category of amount of debris and trash collected in the structure. In 33% of the facilities surveyed, embankments were too steep for mowers. Clogging failures and sediment accumulations were most apparent at these facilities as well. Algal blooms, odors, and mosquito problems were rare at these structures.

Finally, a survey of 25 experts in stormwater management was conducted. All of the panelists agreed that inspections, mowing, and sediment removal were important to maintaining a facility, but they failed to agree on the recommended frequency of such activities. Few of them believed that annual sediment removal was required. All panelists agreed that the annual maintenance intensities (person-weeks per year and annual equipment/material expenses) were greatest for the ponds, with the infiltration basins' costs slightly less.

6.10 Design Aids

Several design manuals for runoff control structures are currently available. Most of these provide screening criteria for applicability of the structures to differing site conditions. Additionally, some provide methods for sizing the structures based on rainfall/runoff relationships.

Stahre and Urbonas (1990) provide numerous design examples for conventional basins and ponds. Although their text is aimed primarily at designing for flood control, they provide valuable insight into using rainfall/runoff relationships in a conservative manner so as to increase the performance and longevity of runoff control structures. This approach is equally applicable to the sizing of runoff control structures for pollution abatement.

The City of Austin, Texas, provides extensive design criteria for ponds, sand filters, and combined systems in its Drainage Criteria Manual (1991) and its Environmental Criteria Manual (1991). These two references provide designs which are meant to facilitate compliance with the city's strict watershed ordinances and provide protection over the Edwards Aquifer recharge zone.

For highway runoff, most design references specify vegetated controls as the preferred choice, followed by wet ponds or extended detention basins as the next best choice (Horner, 1985a, and Burch et al., 1985a). Vegetated controls are recommended first, primarily because of their wide adaptability, low cost, and minimal maintenance requirements, not solely for their pollutant removal capability. Wet ponds are recommended when space limitation and/or site conditions are not conducive to vegetated controls. Infiltration practices, although offering excellent treatment potential, are recommended as the third choice based on their site-specific requirements and because of their increased likelihood of failure if not properly maintained (Burch et al., 1985a).

However, the State of Maryland recommends infiltration as the method of choice for runoff treatment. The state's extensive manuals for the design and maintenance of such facilities presumably increases the structure's success rate in Maryland.

The most comprehensive design reference was written by Schueler (1987). This manual provides a comprehensive guide for screening, designing, cost estimating, and estimating pollutant removal efficiencies for a wide variety of control methods. These controls include extended detention ponds, wet ponds, infiltration trenches and basins, porous pavements, oil/grit chambers, grassed swales, and vegetated buffer strips. Schueler (1987) states that a structure's attainable pollutant removal is governed by (1) the removal mechanisms used, (2) the fraction of the annual runoff volume actually treated, and (3) the nature of the pollutant itself. Of these three, the designer has control over items 1 and 2 but no control over item 3. A graphical comparison of various designs based on field studies, laboratory investigations, modeling predictions and other theoretical considerations is presented in Table 6.8. From the figure, one can see the relationships between pollutants and different designs' capabilities for removing them.

Table 6.8
Comparative Pollutant Removal of Control Structure Designs
(From Schueler, 1987, and Galli, 1990)

CONTROL METHOD:	TSS	Total P	Total N	O_2 Demand	Metals	Bacteria	Overall Capability
EXTENDED DETENTION POND:							
DESIGN 1	◕	◔	◔	◔	◑	⊛	MODERATE
DESIGN 2	●	◑	◔	◑	◕	⊛	MODERATE
DESIGN 3	●	◕	◑	◑	◕	⊛	HIGH
WET POND:							
DESIGN 4	◕	◑	◔	◔	◔	⊛	MODERATE
DESIGN 5	◕	◑	◔	◔	◕	⊛	MODERATE
DESIGN 6	●	◕	◑	◑	◕	⊛	HIGH
INFILTRATION TRENCH:							
DESIGN 7	◕	◑	◑	◕	◕	◕	MODERATE
DESIGN 8	●	◑	◑	◕		◕	HIGH
DESIGN 9	●	◕	◕	●	●	●	HIGH
INFILTRATION BASIN:							
DESIGN 7	◕	◑	◑	◕	◕	◕	MODERATE
DESIGN 8	●	◑	◑	◕	●	◕	HIGH
DESIGN 9	●	◕	◕	●	●	●	HIGH
VEGETATED FILTER STRIP:							
DESIGN 10	◔	○	○	○	◔	⊛	LOW
DESIGN 11	●	◑	◑	◕	●	⊛	MODERATE
GRASSED SWALE:							
DESIGN 12	○	○	○	○	○	⊛	LOW
DESIGN 13	◔	◔	◔	◔	○	⊛	LOW
SAND FILTER:							
DESIGN 14	◕	◔	◔	◕	◑	◕	MODERATE
PEAT-SAND FILTER:							
DESIGN 15	●	◕	◑	●	◕	●	HIGH

Design 1: First-flush runoff volume (i.e., 0.5 inches runoff per impervious acre) detained for 6 - 12 hours
Design 2: Runoff volume produced by 1 inch rainfall detained 24 hours
Design 3: Same as Design 2, but with shallow marsh in bottom stage
Design 4: Permanent pool equal to 0.5 inch storage per impervious acre
Design 5: Permanent pool equal to 2.5 x V_r (V_r = mean storm runoff)
Design 6: Permanent pool equal to 4.0 x V_r; approximately 2 weeks retention
Design 7: Facility exfiltrates first-flush: 0.5 inches runoff per impervious acre

Table 6.8 continued

Design 8: Facility exfiltrates 1.0 inches runoff per impervious acre
Design 9: Facility exfiltrates all runoff up to the 2 year design storm
Design 10: 20 foot wide turf strip
Design 11: 100 foot wide forested strip with level spreader
Design 12: High slope swales with no check dams
Design 13: Low gradient swales with check dams
Design 14: 0.5 inches of runoff storage per watershed acre with pretreatment detention
Design 15: 0.5 inches of storage and exfiltration per impervious acre

KEY: ○=0 to 20 % removal, ◔=0 to 20 % removal, ◑=20 to 40 % removal, ◕=40 to 60 % removal, ●=60 to 80 % removal, ⊗=Insufficient Knowledge

SI units: To convert inches to centimeters multiply by 2.54
To convert acre to hectare multiply by 0.395

APPENDIX A:

Constituents in Highway Runoff

TABLE A-1
Water Quality Characteristics for Runoff Through Bridge Scuppers on I-4 and Lake Ivanhoe (Wanielista et al., 1980)

Element	Form	Number of Samples	Range (mg/L)	Mean (mg/L)	Standard Deviation	Average % In Solution
Zn	Total	11	0.228-1.120	0.498	0.263	67
	Dissolved		0.028-1.120	0.336	0.300	
Cd	Total	11	0.003-0.009	0.005	0.002	20
	Dissolved		0.000-0.005	0.001	0.002	
As	Total	11	0.000-0.145	0.058	0.049	86
	Dissolved		0.000-0.145	0.050	0.055	
Ni	Total	11	0.013-0.261	0.053	0.071	92
	Dissolved		0.006-0.261	0.049	0.072	
Cu	Total	11	0.032-0.101	0.052	0.023	60
	Dissolved		0.010-0.055	0.027	0.014	
Fe	Total	11	0.510-6.850	2.429	2.290	12
	Dissolved		0.034-2.170	0.287	0.626	
Pb	Total	11	0.690-3.250	1.558	0.939	12
	Dissolved		0.063-0.504	0.187	0.136	
Cr	Total	11	0.003-0.027	0.011	0.008	20
	Dissolved		0.000-0.009	0.002	0.003	
Ca	Total	11	25.10-53.80	38.073	9.837	97
	Dissolved		21.80-53.80	36.800	11.110	
Mg	Total	11	0.493-2.790	1.062	0.673	78
	Dissolved		0.294-2.790	0.831	0.738	
P	Total	11	0.160-1.220	0.426	0.382	16
	Dissolved		0.000-0.137	0.067	0.059	

TABLE A-2
Summary of Water-Quality Analyses of Stormwater Runoff from a 1.43-Acre (0.57 ha) Bridge Section of Interstate 95 Collected During Five Storms, November 3 and 20, 1979; March 23, 1981; and May 1 and 20, 1981. Concentrations in mg/L, except as indicated
(McKinzie and Irwin, 1983)

Property	Number of Samples	Mean	Median	Standard Deviation	Range
Spec. Cond.(µmhos/cm)	31	337	182	297	36-845
Turbidity (NTU)	11	19	20	2	14-22
Solids, total	43	437	524	312	31-1,190
Solids, dissolved	43	356	388	273	20-784
Volatile dissolved solids	43	131	136	92	11-327
Volatile suspended solids	32	64	48	60	5-266
Suspended solids, total	43	81	42	85	7-433
COD	40	223	285	139	26-530
Ammonia, total as N	34	.17	.14	.09	.04-.50
Nitrite, total as N	34	.25	.04	.32	.01-1.0
Nitrate, total as N	34	1.4	1.0	1.1	.06-3.2
Organic, total as N	34	2.3	2.0	1.9	.06-5.8
Nitrogen, total as N	34	4.1	3.2	3.0	.49-8.2
Phosphorus, total as P	34	.17	.16	.12	.02-.66
Organic carbon, total	44	77	86	55	3.2-205
Cadmium, total (µg/L)	34	ND	1	--	<1-8
Chromium, total (µg/L)	32	ND	<10	--	<10-30
Copper, total (µg/L)	35	54	40	50	7-250
Lead, total (µg/L)	35	680	590	590	41-2,400
Zinc, total (µg/L)	36	370	330	300	50-1,300
Oil and grease, total	8	7	4	6	2-16
PCB's	2				

TABLE A-3
I-90 Stormwater Runoff Quality North Bend Area
(Farris, 1973)

Parameter		Max.	Min.	Mean
Temperature, ^{o}C		9.0	4.0	5.6
Conductivity, µmhos/cm		5700	41	-
Turbidity, JTU'S		155	58	119
Settleable Solids, mg/L		2.8	0.1	0.66
Total Suspended Solids, mg/L		900	60	285
Volatile suspended Solids, mg/L		324	0.0	4.3
COD, mg/L		243	55	-
Oil, mg/L		51	7.7	24
BOD, mg/L	(S)	50	4.6	12.7
Total coliforms, organisms/100 ml		48,000	700	6200(1)
Fecal coliforms organisms/100 ml		2200	20	590(1)
Ammonia, mg N/L		0.16	0.04	0.07
Nitrate + Nitrite, mg N/L (S)		0.30	0.06	0.15
Kjeldahl Nitrogen, mg N/L (I)		1.56	0.19	0.58
Phosphorus, mg P/L (S)		0.14	0.04	0.07
Phosphorus, mg P/L (I)		0.36	0.06	0.16
Lead, mg/L (S)		0.13	0.04	0.09
Lead, mg/L (I)		1.64	0.1	0.59
Zinc, mg/L (S)		0.06	0.008	0.05
Zinc, mg/L (I)		0.17	0.04	0.09
Copper, mg/L (S)		0.33	0.01	0.02
Copper, mg/L (I)		0.27	0.01	0.09
Cadmium, mg/L, Total		0.03	0.005	0.012
Chromium, mg/L, Total		0.01	0.01	0.01

(1) Median Concentration (S) Soluble (I) Insoluble

TABLE A-4
Stormwater Runoff Quality, South Bellevue Interchange
(Farris, 1973)

Parameter	Max.	Min.	Mean
Temperature, ^{o}C	8.2	4.0	6.0
pH	7.8	6.4	7.1
Conductivity, µmhos/cm	4500	48	500
Turbidity, JTU'S	175	24	84
Settleable Solids, mg/L	2.7	0.1	0.4
Suspended Solids, mg/L	1496	42	246
Volatile Solids mg/L	396	12	57
Total Coliforms, organisms/100 ml	14,000	20	650(1)
Fecal Coliform, organisms/100 ml	2200	20	50(1)
COD, mg/L	433	30	123
Oil, mg/L	96	0.0	14.9
BOD, mg/L	42	5	13
Ammonia, mg N/L	0.44	0.05	0.22
Nitrate-Nitrite, mg N/L (S)	1.85	0.30	0.75
Kjeldahl Nitrogen, mg N/L (I)	4.20	0.09	0.50
Phosphorus, mg P/L (S)	0.12	0.02	0.05
Phosphorus, mg P/L (I)	1.10	0.03	0.26
Lead, mg/L (S)	0.52	0.10	0.17
Lead, mg/L (I)	4.50	0.10	0.99
Zinc, mg/L (S)	0.64	0.005	0.059
Zinc, mg/L (I)	0.749	0.010	0.111
Copper, mg/L (S)	0.20	0.01	0.034
Copper, mg/L (I)	0.16	0.01	0.118
Cadmium, mg/L, Total	0.016	0.004	0.004
Chromium, mg/L, Total	0.01	0.01	0.01

(1) Median Concentration (S) Soluble (I) Insoluble

TABLE A-5
Stormwater Runoff Quality, Lacey V. Murrow Bridge
(Farris, 1973)

Parameter	Max.	Min.	Mean
Temperature, ^{o}C	9.0	5.0	7.0
pH	8.3	6.3	7.2
Conductivity, Micromhos/cm	1620	109	385
Turbidity, JTU'S	230	37	127
Settleable Solids, mg/L	4.0	0.1	1.0
Suspended Solids, mg/L	1274	66	374
Volatile Solids mg/L	238	15	71
Total Coliforms, organisms/100 ml	77,000	20	570$^{(1)}$
Fecal Coliform, organisms/100 ml	9,000	20	120$^{(1)}$
COD, mg/L	548	136	272
Oil, mg/L	78	12	27
BOD, mg/L	111	13	37
Ammonia, mg N/L	0.58	0.08	0.22
Nitrate-Nitrite, mg N/L (S)	4.40	0.35	1.19
Kjeldahl Nitrogen, mg N/L (I)	2.20	0.08	0.76
Phosphorus, mg P/L (S)	0.13	0.03	0.08
Phosphorus, mg P/L (I)	1.70	0.04	0.42
Lead, mg/L (S)	0.44	<0.10	0.15
Lead, mg/L (I)	4.52	0.47	1.63
Zinc, mg/L (S)	0.27	<0.005	0.08
Zinc, mg/L (I)	1.26	0.01	0.32
Copper, mg/L (S)	0.49	<0.01	0.07
Copper, mg/L (I)	0.16	<0.01	0.03
Cadmium, mg/L, Total	0.16	<0.004	<0.01
Chromium, mg/L, Total	0.08	<0.01	<0.02

(1) Median Concentration (S) Soluble (I) Insoluble

Table A-6
Site Median Concentrations for Monitored Storm Events.
(Driscoll, 1990a)

Site No.	SS (mg/L)	VSS (mg/L)	TOC (mg/L)	COD (mg/L)	NO2+3 (mg/L)	TKN (mg/L)	PO4-P (mg/L)	Cu (mg/L)	Pb (mg/L)	Zn (mg/L)
1	112	20		94	0.71			0.019	0.108	0.167
2	172			196		3.35	0.453		0.987	0.666
3	90	20	22	51	0.21	1.67	0.099	0.068	0.278	0.269
4	218			125		2.01	0.408		0.900	0.341
5	408	77	88	291		3.51	0.821	0.104	0.705	0.644
6	9		12	41	0.23	0.46	0.036	0.005	0.236	0.071
8	67	70	46	169	1.02	1.25	0.140	0.043	0.623	0.303
11	51		15			1.04	0.227	0.020	0.116	
12	85		20			1.56	0.429	0.030	0.407	
13	20	6	24	67	0.19	1.68	0.124	0.038	0.011	0.050
14	25	8	11	31	0.61	1.14	0.267	0.029	0.091	0.051
15	184	18	16	34	3.32	2.16	1.075	0.087	0.026	0.167
17	190	49	29	113		1.86	1.687	0.056	0.411	0.259
18	126	21	3	46	0.73	0.64	0.168	0.036	0.175	0.100
19	101	25	10	114	0.81	3.32	0.476	0.025	0.101	0.325
21	104	21	17	60	0.57	0.75	0.428	0.026	0.130	0.099
23	93	26	13	106	0.83	0.90	0.217	0.037	0.451	0.382
25	244	59	33	145	0.79	1.09	0.415	0.172	1.065	0.280
26	43	9	2	41	0.53	0.38	0.123	0.025	0.065	0.071
27	119	29	10	158	1.11	1.69	0.865	0.041	0.173	2.892
28	34	9	7	32	0.45	0.60	0.098	0.017	0.046	0.040
29	334	72	32	111	0.77	2.77	0.417	0.075	0.738	0.371
30	140	47	27	88	1.27	1.86	0.287	0.088	1.457	0.336
31	143	47	30	122	0.79	3.09	0.315	0.155	0.817	0.465
Mean	143	36	24	103	0.84	1.79	0.435	0.052	0.525	0.368
Median	93	26	16	84	0.66	1.48	0.293	0.039	0.234	0.217
COV	1.16	0.97	1.06	0.71	0.77	0.67	1.10	0.87	2.01	1.37
N	24	19	21	22	18	23	23	22	24	22

TABLE A-7

Statistical Analysis of Urban Highway Runoff at Maitland Interchange near Orlando, Florida. Concentrations in µg/L.
(Yousef et al., 1991)

Parameter	Number of Observations	Mean	COV	Median	Range
Organic N	6	965	0.64	741	292-1891
NH_3-N	13	152	0.63	78	9-972
NO_2-N	13	13	0.35	8	1-37
NO_3-N	12	306	0.49	246	46-665
OP	13	76	0.81	65	26-178
TP	13	170	0.66	147	62-346
Total Pb	16	163	0.70	119	30-379
Diss. Pb	16	34	0.81	25	13-128
Total Zn	16	71	0.68	53	13-173
Diss. Zn	16	40	0.74	33	13-134
Total Cu	16	37	0.63	35	10-101
Diss. Cu	16	26	0.59	22	10-64

Table A-8

Summary of Pollutant Concentrations (mg/L) in Washington State Highway Runoff, 1980-1981 (Including Road Sand and Volcanic Ash). Mean Values with Ranges in Parentheses (Chui et al., 1981)

SITE	TSS	VSS	COD	Pb	Zn	Cu	TKN	nitrate plus nitrite	TP
I-5	106 (14-552)	37 (4-428)	150 (34-1291)	0.466 (.08-1.166	.638 (.112-3.093)	.043 (.013-.283)	1.175 (.409-2.968)	1.281 (.118-3.032)	.226 (0.0-.671)
Vancouver	106 (21-833)	13 (0-35)	45 (4-122)	.073 (.002-.165)	.056 (.003-.186)	.025 (.01-.093)	55.0 (.387-339)	1.636 (.552-5.811)	.113 (.007-.323)
Snoqualmie Pass	63 (13-270)	12 (1-39)	44 (14-136)	.086 (.01-.408)	.101 (.002-.559)	.022 (.006-.053)	.335 (0-1.054)	.693 (.129-2.430)	.150 (.047-.479)
Montesano	798 (22-15010)	45 (4-304)	109 (7-685)	.556 (.032-4.184)	.380 (.004-4.008)	.089 (0-.648)	1.586 (0-5.832)	1.067 (.094-7.321)	.441 (.033-2.328)
Pasco	570 (31-5050)	79 (9-585)	265 (40-1106)	.196 (.055-.691)	.352 (.219-7.190)	1.039 (.011-.099)	8.628 (2.296-19.43)	.793 (.287-2.070)	.666 (.019-2.010)
Spokane	370 (48-1037)	65 (15-174)	219 (106-437)	.167 (.04-.326)	.929 (4.96-10.774)	7.033 (.004-.068)	7.156 (.800-6.312)	1.208 (.332-1.713)	.998 (.34-1.454)
Pullman-9	622 (59-4545)	37 (7-117)	110 (27-523)	.227 (.028-1.303)	.199 (.029-.527)	.044 (.015-.116)	1.146 (.313-1.713)	.791 (.258-2.489)	.810 (.155-3.732)

Table A-9
Pollutant Mass Loadings and Concentrations
(Zawlocki, 1980)

Parameter	Storm 520-16		Storm 520-13		Storm 1-5-131		Storm 1-5-87		Storm 1-5-79		Storm 1-5-79		Storm 1-5-93		Storm 520-11	
	mg/L	lb/ acre	mg/L	lb/ acre	mg/L	lb/ acre	mg/L	lb/ acre	mg/L	lb/ acre	mg/L	lb/ acre	mg/L	lb/ acre	mg/L	lb/ acre
Particulate	147	16.3	362	45.8	123	19.2	63	12.	95	4.4	23	1.5	128	28.1	404	89.0
TSS	144	15.9	408	51.6	57	8.9	72	14.	97	4.5	32	2.1	118	25.9	435	95.8
VSS	53	5.9	74	9.4	19	2.97	10	2.0	26	1.2	12	0.79	26	5.7	115	25.3
COD	130	14.4	280	35.4	166	2.59	59	12	61	2.8	61	4.0	52	11.4	271	59.7
Pb	0.5	0.05	0.9	0.1	0.7	0.11	0.3	0.06	0.8	0.04	0.1	0.007	0.8	0.2	0.8	0.2
Particulate Extract	19.1	2.1	17.6	2.2	13.4	2.1	13.5	2.66	15.6	0.72	7.3	0.48	0.5	2.1	30.8	6.79
Aqueous Extract	2.1	0.4	4.6	0.6	9.3	1.4	1.1	0.22	7.8	0.36	5.1	0.33	3.0	0.66	0.62	0.14
Alcohols			0.453	0.057	0.633	0.099	0.16	0.032								
Aliphatic			0.933	0.118	2.49	0.389	6.93	1.37								
Aromatic			0.724	0.092	0.393	0.061	2.20	0.43								
Halogens			0.114	0.014	0.082	0.013	T	T								
Ketones/ Aldehydes			0.513	0.065	0.126	0.020	1.13	0.22								
Organo Sulfur			0.005	0.0006	0.005	0.001	1.26	0.25								
Oxygenates			2.57	0.325	3.74	0.586	3.58	0.71								
Phenols			T	T	0.003	.0005	2.91	0.57								
Nitrogen			0.46	0.058	0.115	0.018	1.15	0.29								

SI units: To convert 16/acre to kg/ha multiply by 1.15

Table A-10
Annual Loading Rate in kg/ha/yr (Including Applications and Volcanic Ashfall)
(Chui et al., 1981)

SITE	TSS	VSS	COD	TOC(1)	Pb	Zn	Cu	TKN	NO_3 $+NO_2$-N	TP
I-5	980	231	787	40.4	4.42	2.32	.223	4.96	5.70	2.40
I-5w/Grit(2)	1561	--	--	--	--	--	--	--	--	--
I-5*	923	254	740	--	4.62	2.84	.355	1.66	1.24	1.96
SR-520	3766	851	2101	342.1	15.92	4.21	.732	15.65	6.75	6.75
Vancouver	314	55	195	46.3	.13	.22	.034	31.95	2.17	.60
Snoqualmie Pass	2770	361	1026	218.3	1.67	2.01	.361	6.99	6.56	6.23
Montesano	6175	961	3868	63.6	5.48	2.56	.565	13.72	8.00	8.23
Pasco	320	45	181	31.3	.08	.47	.030	2.22	.80	.73
Spokane	1498	241	1187	44.9	.69	10.40	.118	17.27	3.52	3.06
Pullman-9	1355	68	206	--	.32	.39	.065	2.78	2.42	1.00

Notes: (1) TOC loadings reported exclude 1980-81
(2) Includes large particles which settled in flume upstream of flow splitter and were collected for analysis
Customary units: To convert kg/ha/yr to lb/ac/yr multiply by 0.87

TABLE A-11
Summary of Highway Runoff Quality Data for All Six Monitoring Sites - 1976-77.
(Gupta et al., 1981c)

	Pollutant Concentration (mg/L)		Pollutant Load lb/ac/event		Pollutant Load lb/ac/in-runoff	
Parameter	**Avg.**	**Range**	**Avg.**	**Range**	**Avg.**	**Range**
pH		6.5-8.1				
TS	1147	145-21640	51.8	0.04-535.0	260	33-4910
SS	261	4-1656	14.0	0.008-96.0	59	0.9-375
VSS	77	1-837	3.7	0.004-28.2	17	0.2-190
BOD_5	24	2-133	0.88	0.000-4.1	5.4	0.5-30
TOC	41	5-290	2.1	0.002-11.5	9.3	1.1-66
COD	14.7	5-1058	6.9	0.004-34.3	33	1.1-240
TKN	2.99	0.1-14	0.15	0.000-1.04	0.68	0.02-3.17
NO_2+NO_3	1.14	0.01-8.4	0.069	0.000-0.42	0.26	0.002-1.90
TPO_4	0.79	0.05-3.55	0.047	0.000-0.36	0.18	0.011-0.81
Cl	386	5-13300	13.0	0.008-329.0	88	1.1-3015
Pb	0.96	0.2-13.1	0.058	0.000-4.8	0.22	0.005-2.97
Zn	0.41	0.01-3.4	0.022	0.000-0.48	0.053	0.002-0.771
Fe	10.3	0.1-45.0	0.50	0.000-3.5	2.34	0.023-10.2
Cu	0.103	0.01-0.88	0.0056	0.000-0.029	0.023	0.002-0.199
Cd	0.040	0.01-0.40	0.0017	0.000-0.014	0.009	0.002-0.091
Cr	0.040	0.01-0.14	0.0028	0.000-0.029	0.009	0.002-0.032
Hg, (µg/l)	3.22	0.13-67.0	0.00059	0.000-0.002	0.730	0.029-15.2
Ni	9.92	0.1-49	0.27	0.007-1.33	2.25	0.023-11.1
TVS	242	26-1522	9.34	0.01-44.	55	5.89-345

Metric units: To convert lbs/ac to kg/ha multiply by 1.15

TABLE A-12
I-90 Stormwater Runoff Loads, Lacey V. Murrow Bridge
(Farris, 1973)

Parameters	Daily Loading ($mg/m^2/day$)
Suspended Solids	239
Volatile Solids	49
COD	171
Oil	15.6
BOD	23.7
Ammonia as N (S)	0.12
Nitrate-Nitrite as N (S)	0.59
Total Nitrogen as N (I)	1.15
Phosphorus as P (S)	0.05
Phosphorus as P (I)	0.28
Lead (S)	0.09
Lead (I)	0.96
Zinc (S)	0.03
Zinc (I)	0.21
Copper (S)	0.03
Copper (I)	0.02
Mean Daily Traffic Volume	50,600

(S) Soluble (I) Insoluble

Customary units: To convert $mg/m^2/day$ to lb/ac/day multiply by 0.0089

TABLE A-13
Stormwater Runoff Loadings North Bend Area.
(Farris, 1973)

Parameter	Daily Loading $mg/m^2/day$	Daily Loading mg/m/vehicle
Suspended Solids	3250	36×10^{-1}
Volatile Solids	690	7.6×10^{-1}
COD	35	3.6×10^{-2}
BOD (S)	45	5×10^{-2}
Oil	210	2.3×10^{-1}
Ammonia as N (S)	0.53	5.8×10^{-4}
Nitrate + Nitrite as N (S)	1.53	1.7×10^{-3}
Kjeldahl Nitrogen as N (I)	0.62	6.9×10^{-4}
Phosphorus as P (S)	0.53	5.8×10^{-4}
Phosphorus as P (I)	1.29	1.4×10^{-3}
Lead (S)	0.12	1.4×10^{-4}
Lead (I)	5.7	6.4×10^{-3}
Zinc (S)	0.52	5.8×10^{-4}
Zinc (I)	0.77	8.4×10^{-4}
Copper (S)	0.017	1.8×10^{-4}
Copper (I)	1.26	1.4×10^{-3}

(S) Soluble (I) Insoluble
Customary units: To convert $mg/m^2/day$ to lb/ac/day multiply by 0.0089
Customary units: To convert mg/m/vehicle to lb/mile/vehicle multiply by 0.0035

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Note: General references are listed; text references are highlighted in bold