CHAPTER I
INTRODUCTION

A serious problem of urbanizing areas today is contamination of water supplies by nonpoint sources of pollution. Nonpoint sources are nondiscrete discharges such as construction site drainage, agricultural land drainage, and urban runoff. Urban runoff may contribute pollutants such as heavy metals, nutrients, suspended solids, and biochemical oxygen demand, which may be detrimental to area surface and groundwaters. Heavy metals are trace inorganics such as mercury, lead, silver, and zinc. Sources of pollutants may include pavement components, motor vehicles, atmospheric fallout, vegetation, litter, spills, and anti-skid chemicals. In order to maintain a receiving water of high quality, the problem of urban runoff must be addressed and considered as thoroughly as point source discharges because of the large volumes of runoff in urban areas due to the highly impervious nature of the catchments.

This is a problem which has received attention since the 1978 Water Quality Management Planning Study (NVPDC/Virginia Tech, 1978). One of the purposes for enacting the 1972 amendments was to initiate a coordinated approach towards solving the problems of water pollution from nonpoint sources as well as point sources.
in the Northern Virginia suburbs of Washington D.C., the Occoquan Reservoir serves as a major component of a drinking water source for over 106,000 persons. Impounded in 1957, by the mid-1960's it was found that the quality of the reservoir waters was rapidly declining. Following a study by the State Water Control Board and Upper Occoquan Sewage Authority (UOSA), which greatly reduced the pollution from point sources entering the reservoir, it was found that pollutants had continued to remain at a level

high enough to cause decreasing water quality. This was attributed to nonpoint sources such as stormwater runoff from agricultural and urbanizing areas (Randall, Grizzard and Hoehn, 1978). The major pollutants of interest were nutrients (phosphorus and nitrogen species) which cause eutrophication of surface waters, subsequent oxygen depletion, and ultimate water quality degradation.

In order to reduce the water quality impact of these pollutants, Best Management Practices (BMP's) are being tested and implemented by many jurisdictions in the watershed. There are three general classes of BMP's; source control, volume control, and infiltration measures. Source controls are those designed to keep potential pollutants at the source and include zoning, erosion control practices, street sweeping, and encouragement of site grading patterns that increase distances over pervious and impervious areas (Whipple, 1979).

Volume control BMP's are those BMP's which reduce the delivery of pollutants to receiving waters through collection in detention and retention facilities such as sediment traps, dry ponds, extended detention dry ponds, and wet ponds. Detention facilities detain the flow of runoff for a designed period of time so that pollutants may precipitate out or change to inert forms. Retention ponds hold the volume of stormwater indefinitely to accomplish the same goals. They rely upon a combination of physical, chemical, and biological processes to remove pollutants from the stormwater flow:

Process	Example
Physical	Sedimentation
	Filtration
	Flocculation
Chemical	Ion Exchange
	Adsorption
	Precipitation
	Coagulation
	Oxidation/Reduction
Biological	Uptake by Flora and Fauna
	Pollutant Conversion
	Oxidation/Reduction

Infiltration practices include such measures as grassed swales, French drains, infiltration trenches, recharge basins and porous pavement designed to reduce pollutants as well as retard the flow of runoff. To the extent possible, such practices take advantage of many of the same practices that are found at work in volume controls, but in addition, have the capacity to remove pollutants by utilizing the infiltration capacity of the underlying soils.

The Virginia Division of Soil and Water Conservation Commission and the Occoquan Watershed Monitoring Laboratory (OWML), of the Virginia Tech Department of Civil Engineering, have completed a 36 month study to examine the effectiveness of BMP's in reducing stormwater runoff and removing pollutants from urban parking lot runoff which would otherwise enter a watercourse. Two BMP's were selected for demonstration in the Davis Ford Park Best Management Practices project in Manassas, Virginia, which is located in the Occoquan Watershed.

Porous pavement is a type of asphalt pavement which is pervious to stormwater runoff. The pavement is characterized as having 16% voids as compared with conventional pavement which has 3-5 percent voids. It is this greater void ration that allows runoff to filter through the pavement and subbase and into the underlying soils and ultimately to the groundwater.

This characteristic allows water to filter through to the soil depositing pollutants on a covered subgrade and also allows infiltration of stormwater to the sub-grade, thus theoretically reducing the pollutant load which would ordinarily enter a watercourse with the runoff. Dissolved pollutants are also reduced by the mechanism of adsorption onto soil particles in the sub-base. Although research has been done on the hydraulic aspects of porous pavement projects (Campbell, 1978), little detailed research has been undertaken to examine pollutant export reductions resulting from the use of this BMP.

The second BMP, an infiltration trench, was constructed to act as a stormwater management practice for conventional impervious pavement so that the soil may receive the majority of pollutant load.   An Infiltration trench is a trench installed at the base of a modest slope which is designed to accept overland flowing urban runoff. The trench is sized in accordance with the maximum volume of stormwater that it is projected to receive.

One weakness of earlier porous pavement and infiltration trench studies was the absence of an experimental control to provide pollutant export data from conventionally paved surfaces control. As a control in the Davis-Ford project, runoff from an adjacent area of conventional pavement without an infiltration trench was also analyzed (V.P.I. and S.U. Department of Civil and Environmental Engineering, 1983).

The major objectives of this study were:

1. to characterize the pollutant export from conventionally-paved surfaces and to compare these values to others in the literature.
2. to observe the seasonal variation of wetfall and dryfall deposition, and to determine the effects if any, on subsequent storm loads.
3. to measure the pollution reduction by the two BMP's implemented for 15 synoptic storm events.
4. to observe the effect of antecedent dry period on runoff pollutant loading.
5. to observe the transformation of pollutants in the soil profile.

CHAPTER II

REVIEW OF LITERATURE

Stream standards have historically been used to provide a baseline against which the relative impact of a waste on a receiving watercourse may be measured. An oxygen sag model is often used to predict minimum dissolved oxygen concentrations for given hydrological conditions, typically the 10 year 7 day-low flow. During such extreme low flow conditions, one may generally assume no contribution by nonpoint sources, as extreme low flows most often occur in the absence of direct runoff. However it is conceivable that critical conditions may not be typified by the period immediately following the 10 year 7 day-low flow. One possibility is rainfall removes accumulated urban pollutants from the surface and transports them overland to a receiving watercourse, greatly increasing the pollutant load without significantly increasing the watercourse volume. This decrease in water quality may be attributed to the lack of diluting power due to the 10 year 7 day-low flow situation (Colston, 1974).

Such concentrated runoff events may occur several times during the year, and permanent changes in the downstream biota may result, even though the quality of the water may revert to normal after the cessation of runoff (Shaheen, 1975). Before this return to normal water quality, dissolved oxygen levels may be depressed below antecedent conditions at critical impact locations (Rimer, 1978).

In a Chicago study, the American Public Works Association (APWA) found that runoff constitutes approximately 1 percent of sewage load from the analysis of 18 sampling points (acreage unspecified) in that city. The basis of comparison for this study was the differing land use types of industrial, commercial, multiple family, and single family areas.

They found that the pollution potential from runoff was 5 percent of BOD discharged from the area's secondary waste treatment facilities. Assuming that a 14 day accumulation of street litter and all the soluble BOD in the dust and dirt fraction would be discharged into street inlets during a 2 hour storm, APWA estimated that the shock pollution load on receiving waters would be 160 percent of the raw sewage BOD and 800 percent of the secondary treatment effluent during the 2 hour period (Environmental Science and Technology, 1969).

The gravity of the situation is evident when considered on a shock load basis. For example, if a 3 day accumulation of traffic-related roadway materials were flushed into receiving waters during the course of a 2 hour runoff event, the rates of traffic-related runoff would be uniformly increased by a factor of 36 (three days * 24 hours/day / two hour runoff) (Environmental Science and Technology, 1969). Traffic-related deposits alone would then constitute a significant source of pollution on a shock load basis for each parameter.

Characterization of Urban Runoff

Atmospheric Sources

A logical point to begin the discussion of the sources of contaminants would be with atmospheric fallout. A large fraction of the particulate matter contributing to the water pollution effects of street surface contaminants is of a size sufficiently fine that they could have been transported by air currents prior to being deposited on street surfaces. The extent to which this occurs is difficult to measure, and therefore, may generally only be estimated (Sartor, 1972). In urbanizing areas, major sources of such materials are industrial stacks and vents, construction and excavation projects, agricultural operations, and exposed vacant land areas. Automotive traffic and heavy commercial air traffic are also sources of fine airborne particles (Sartor, 1972). Many of these forms of fallout are virtually inert and would add only turbidity and solids loadings to receiving waters. Others, however, are reactive and would impose loadings of oxygen demand, algal nutrients, toxic metals, and pesticides (Sartor, 1972).

All airborne particles are ultimately removed from the air by one of four mechanisms:

1. dry sedimentation
2. impact against an object on or near the ground
3. rainout (snowout) in which the particles are incorporated in cloud droplets by diffusion and are subsequently carried to the ground when the droplets grow to falling drops (or fall as snowflakes).

4. washout in which falling rain (or snow) scavenges the particles below the cloud.

While these processes are extremely variable and depend on meteorological conditions, their relative effectiveness is in the order listed, the most important being washout and the least important being sedimentation (Rasool, 1973).

Wind transported material is called dustfall. Dustfall is normally associated with samples collected during dry periods, and excluding rainfall and snowfall. In a Chicago study, Heany and Sullivan estimated that approximately 70 percent of the material found on street surfaces could be attributed to dustfall. They also found the seasonal distribution of dustfall loadings on the unit area exhibited the commonly observed heavier loadings in the winter months. During 1966, the average annual dustfall loading in Chicago was estimated to be 36.9 tons/month/sq mile. This level was significantly lower than a decade earlier when the average loading during the period from 1954 to 1956 was 54.9 tons/month/sq mile (Heaney and Sullivan, 1971).

Precipitation, in the forms of rainfall and frozen precipitation, dissolve and remove materials from the air such as particulates, carbon monoxide, nitrogen oxides, and sulfur oxides. Gambell and Fisher in 1966 observed the composition of rainfall over a 34,000 square mile area in southeastern Virginia and North Carolina. They examined data on sodium (Na^{+}), magnesium ($Mg2^{+}$), nitrate ($NO3^{-}$), and chlorine (Cl^{μ}) and observed that, excluding bicarbonate, almost one half of the dissolved solids carried by streams were contributed by rainfall. They also found that rainfall added more nitrates and sulfates to the streams than were originally carried by the streams prior to the rainfall (Gambell and Fisher, 1966).

In summary, the literature shows the chemical constituents of urban rainfall and air masses represent a significant quantity of pollutants which may mix with larger parts of organic and inorganic matter constituting urban litter (Sartor, 1972).

Direct Sources

Direct sources of urban runoff pollutants are varied and include:

1. Vegetation and animal activity which lead to the deposition of grass cuttings, leaves, and fecal matter.
2. Roof and road surfaces, which are sources of inorganic solids, cement, sand, eroded road material, and salt.
3. Motor traffic, which is a source of oil, exhaust gases, gasoline, and particles from car bodies and tires.
4. Human activity, which leads to accumulation of litter, detergents, and garden fertilizers (Sartor, 1972).

Street surface contaminants are deposited on roadways via mechanisms which may or may not be related to traffic. Loadings of related depositions will be proportional to total traffic and may arise directly (tire rubber, motor oil) or indirectly (abraded material from the roadway surfaces) from the motor vehicle. Less than 5 percent by weight of the traffic related deposits originate directly from the motor vehicles, however these are most important by virtue of their potential toxicity (Shaheen, 1975).

Motor vehicles contribute a broad range of materials to stormwater runoff. Lubricants, hydraulic fluids, and coolants, in addition to being pollutants themselves, may degrade asphaltic surfaces, thereby increasing the amount of inorganic solids loadings on streams (Shaheen, 1975). Zinc and other trace metal particles are worn off of tires and clutch/brake linings. Prior to the reformulation of gasoline in the early 1980's particulate exhaust emissions contributed high lead loadings. Dirt, rust, and decomposing coatings drop off of fender linings and undercarriages. Particulates may originate from vehicle components broken by vibration or impact (glass, plastic, metals etc.). Lead, zinc, and chromium originate from petrochemicals (Shaheen, 1975). Zinc compounds are also used as a filler in tires and it is deposited on asphalt surfaces as tires are abraded.

Many of the automobile-unrelated street surface contaminants are representative of products abraded from the roadway surfaces, and are largely inorganic. Phosphorus and chloride are most likely derived from roadway surface abrasion. Other pollutants unrelated to traffic are those deposited by atmospheric fallout or from sources of surrounding overland flow.

Soil Erosion

The same processes that result in the incorporation of deposited materials into urban stormwater from highly impervious catchments may also create an additional source of sediment and other pollutants reaching receiving waters. Goldman et al, (19__) defined five specific types of erosion, listed in order of occurrence from the point of raindrop impact to the flowing stormwater in an urban drainage channel:

1. Splash Erosion
2. Sheet Erosion
3. Rill Erosion
4. Gully Erosion
5. Channel Erosion

Splash erosion generally may be observed to take place from the direct impact of rainfall upon bare soil, and its degree of occurrence appears to be strongly related to raindrop size. A process similar to this may take place on impervious urban surfaces, when rainfall kinetic energy results in the detachment and incorporation into the flow of materials deposited on the surface, or of materials abraded from the surface. Significant splash erosion losses may also be envisioned during the construction phase of an urban catchment, when large areas of soil are stripped of vegetation. However, once a highly impervious catchment is stabilized, the vegetative cover that exists is generally well-maintained, and splash erosion at the site level is not often found to be a significant source of sediment loss.

Sheet erosion is generally viewed as the process of transport of solid particles that have been previously detached by soil erosion. Likewise, sediment transport from the site in a stable urban catchment is not thought to be a major mode of soil loss.

Rill and gully erosion are similar to channel erosion, but at the site level. Rill erosion is the removal of soil by water from small well-defined channels, while gully erosion is erosion from channels larger than rills, making the issue largely one of scale (Schwab). As with the previous cases, most highly impervious urban catchment generally do not exhibit excessive rill and gully erosion at the site level in the post-development condition.

The erosion and transport of sediments in urban waterways is largely a function of runoff waters acquiring sufficient energy to detach soils from pervious surfaces, and to transport them downstream. In highly impervious urban systems, such as the Davis Ford Park Catchment, there is little exposed pervious surface across which storm runoff passes. In the absence of detention, the conveyance routes the increased storm flow to a natural stream channel where increased erosion by flow over the streambank, or undercutting of the channel is likely to be one of the consequences. Schwab, et al subdivide the transport of such material into:

1. Suspension: Material carried in the stream flow with sufficient energy to prevent contact with the streambed

2. Saltation: Materials swept along in the flow, and may be observed to skip or bounce along the streambed
3. Bed load: Materials that move in constant contact with the streambed, being rolled and pushed along the bottom.

Other factors affecting urban runoff.

Colston (1974), in an early study of urban runoff in Durham, N.C., proposed the following factors as the principal determinants of the water quality effects of urban runoff:

1. hydrology of the region
2. percent imperviousness
3. land use
4. season
5. antecedent dry period storms

(Colston, 1974).

The rate of rainfall varies in time over a catchment. In general, precipitation is more intense the shorter the duration of the storm (Stephenson, 1981). Storm intensities and volumes tend to be more variable in the summer while the coefficient of variation of duration and the time between storms remains relatively constant throughout the year and very nearly equal to 1.0 (U.S. EPA).

Barry has stated that the intensity (amount/duration) of rainfall during an individual storm, or a still shorter period, is of vital interest to water engineers concerned with flood forecasting as well as conservationists dealing with soil erosion. He further states that the quantity of runoff from a given storm is determined by:

1. the moisture deficiency of the basin at the beginning of rainfall, and
2. the storm characteristics, such as rainfall amount, intensity, and duration.

(Barry and Chorley, 1968)

Storm runoff effectively represents the increment of rainfall which is discharged after the moisture deficiency of the basin has been satisfied. In other words storm runoff is the difference between precipitation and basin recharge, and is commonly called the precipitation excess (Linsley, 1949).

High intensity rain is generally associated with increased drop size rather than an increased number of drops (Barry and Chorley, 1968). Another useful characteristic is the average time period within which a rainfall of specified amount or intensity may be expected to occur. This is known as recurrence interval or return period (Barry and Chorley, 1968). It is important because it is a standard factor for the design of Best Management Practices.

In a runoff study by Angelotti it was found that three general parameters had an affect on contaminant removal from an asphaltic surface. In order of decreasing importance, he found them to be rainfall duration, rainfall intensity, and raindrop energy (Angelotti, 1985).

Mulvaney, in 1851 proposed the rational formula which described a relationship between catchment physiology and rainfall intensity.

$$Q = CiA$$

where: $C =$ coefficient governed by the physiology of the catchment.

$i =$ rainfall intensity

$A =$ surface area of the catchment

(Stephenson, 1981)

The rate at which runoff occurs depends not only on the rate of precipitation, but also on the surface configuration and the depth-discharge relationship. As a land surface is developed for urban use, the region is transformed from the natural state to a manmade state. In urbanization, as building densities increase so does the amount of impervious area. A greater proportion of the incident rainfall appears as runoff than when the catchment was in it's rural state. So water quality aspects of the hydrological cycle are affected by both the rise in population and the increase in the extent of the impervious area (Hall, 1984). Saturation and consequent surface runoff occur relatively rapidly in the urban watershed since storage and infiltration have been reduced to practically zero.

In contrast to the increase in the volume of runoff and peak rate of flow, urbanization tends to decrease the watershed's lag time (tc) (Hall, 1984). This decrease in lag time adds to the shock load effect of runoff entering receiving waters in the sense that the effect is more immediate than if the watershed had been in it's original state.

Kluesener and Lee state that percent runoff is approximately equal to the area of the basin covered by streets (Kluesener and Lee, 1974). In the study of the Occoquan Watershed mass balances show that the largest pollutant loads originated from urbanized areas of the watershed as compared to agricultural areas, even though the agricultural area was 1.85 times larger than the area containing urbanized sections (Randall et al, 1978).

A simplified equation for estimating annual runoff is:

$$AR = (0.15 + 0.75\ I/100)P$$

where: AR = annual runoff

I = percent imperviousness

P = annual precipitation (inches/year)

(U.S. EPA Nonpoint Sources Branch)

For STORM (Storage Treatment Overflow Runoff Model) Water Resources engineers computed a runoff coefficient weighted between pervious and impervious areas defined by:

$$CR = .15\ (1 - I/100) + .90(I/100)$$

where:

I = percent imperviousness

coefficients .15 and .90 are default values used in STORM for runoff coefficients from pervious and impervious areas respectively.

(U.S. EPA, Nonpoint Sources Branch)

Wide ranges in urban runoff and it's constituents are attributed to seasonal variations. Changnon, found in a Champaign, Illinois study that on a seasonal basis, the largest excesses of urban over rural precipitation occurred in winter (16 percent), with autumn (14 percent), spring (13 percent), and summer (7 percent) next in order of magnitude. (Changnon).

Volatile solids, BOD, and COD depositions are generally higher in summer and fall. Lead and zinc deposition rates were found to be considerably higher during warm seasons while other heavy metals were deposited at relatively uniform rates during the year (Shaheen, 1975). In general, Kluesener found in his Madison, Wisconsin study that phosphorus concentrations of overland runoff were greatest in spring and fall while nitrogen concentrations were greatest in spring (Kluesener and Lee, 1974).

The following chart relates traffic/roadway related pollutants with season of the year:

Table 1
Average Seasonal Loading Rates
gram/axle/km

Pollutant	Fall	Winter	Spring	Summer
Dust and dirt	0.028	0.027	0.019	0.029
Volatile solids	0.003	0.001	0.001	0.002
BOD	0.192	0.077	0.073	0.068
COD	4.40	2.240	1.760	2.830
Grease	0.290	0.240	0.330	0.410
Lead	0.044	0.041	0.110	0.085
Zinc	0.010	0.009	0.029	0.021
Rubber	0.029	0.012	0.062	0.041

Antecedent dry period effects.

Although deposition is theoretically uniform, it has been found that materials do not accumulate on roadways at a linear rate. Data acquired by the Environmental Protection Agency indicate that accumulated loads begin to level off after several days (Shaheen, 1975).

Public works practices play a major role in contaminant loadings of runoff. The quantity of contaminant material existing at a given test site was found to depend on the length of time that had elapsed since the site was last cleaned, either intentionally by sweeping or flushing, or by rainfall. In street cleaning the efficiency of removal of dust and dirt is generally low (Sartor, 1972).

CONSTITUENTS OF URBAN RUNOFF

PHOSPHORUS AND NITROGEN

Phosphorus

Phosphorus may exist in several chemical forms in nature, although it is only found in a single oxidation state in both inorganic and organic forms. Inorganic forms are mainly ortho-phosphate (PO_4^{3-}) and polyphosphates.

Particulate phosphorus, which is attached to solid particles, can be divided into nonapatite (largely Fe-and Al- associated), apatite and organic forms. The inorganic forms tend to control the phosphate concentration in solution through adsorption-desorption and precipitation-dissolution reactions. Particulate organic phosphorus can release dissolved organic phosphorus to solution, but dissolved organic P must be converted to inorganic P before it becomes available (Metcalf and Eddy, 1979).

Organic forms are usually associated with complex cellular substances, nucleic acids and phospholipids, and most phosphorus in natural waters is in the organic form (Welch, 1978). Total phosphate is a valuable measure of nutrient impact because polyphosphates are generally hydrolyzed to orthophosphates in aqueous environments within several hours (Sartor, 1972).

The most available form is ortho-phosphate and is readily assimilated by algae during photosynthesis while polyphosphate is of only minor consideration (Sartor, 1972). Yousef has found that on the order of 10 to 30 percent of the particulate phosphorus present in urban stormwater drainage would likely become available for algal metabolism in a receiving stream (Yousef, 1979).

Eutrophication is the process by which a water body progresses from it's origin to it's extinction according to the level of nutrient and organic matter accumulation. It is the result of over-enrichment of surface waters with nutrients (primarily nitrogen and phosphorus), potentially resulting in algal blooms at certain times of the year. It may occur naturally with nutrients derived from decaying organic matter or phosphate deposits, or artificially, as the product of excessive nutrient loading from agricultural, industrial, or urban activities (Welch, 1978).

Degradation of water quality may occur as a result of the growth of the algal cell mass produced from a phosphorus excess. Algal blooms destroy recreational and aesthetic values of lakes, decrease the value of lake-shore property and increase the costs of water treatment (Welch, 1978). Cultural eutrophication is the effect of man's activities which accelerate the natural process of eutrophication to the extent of causing a nuisance (Harms and Southerland, 1975).

Oxygen depletion is a major detrimental result of algal blooms. As a case study, oxygen depletion has presumably been the responsible factor for the devastation of benthos in Lake Erie (Welch, 1978).

Phosphorus, applied to the land surface in fertilizers, combines with clays and remains locked in the soil until either it is utilized by plant life or is removed in eroded soil, in which case the phosphorus is removed along with suspended sediment (Whipple, 1977). In many cases, phosphorus concentration levels have appeared to be very closely associated with the suspended solids concentration (Rimer, 1978).

Urban runoff typically contains appreciable quantities of both soluble orthophosphate and particulate forms of phosphate. The concentration of total phosphorus may generally be seen to increase markedly after heavy rainfalls, and consequently the storm loadings are most often larger than base flow. Most of the P loadings for urban areas fall in the range of 5 to 9 lbs/sq mi/day (Whipple, 1977). In a study by Sawyer in Madison, Wisconsin, it was found that urban runoff contained phosphorus in the range of 235-262 lb/mi2/yr (0.37-0.41 lb/acre/yr). (Sawyer, 1968). Wanielista estimates a range of total phosphorus loading in urban areas to be 1.0 to 5.0 kg/ha/year which converts to 570 to 2850 lb/mi2/yr (0.89-4.45 lb/acre/yr) (Wanielista, 1978).

Seasonal variations in a study by Whipple in Morristown, New Jersey resulted in phosphate contributions of 69 lb/day in August and 585 lb/day in March for an area of

5.2 square miles with average concentrations of 1.5 and 2.1 mg/L respectively (Whipple et al, 1974).

Phosphorus is usually the nutrient which is the limiting factor for the eutrophication of freshwater lakes. This follows from Liebig's Law of the Minimum which says "any nutrient will become limiting if it is present in the smallest quantity relative to the amount needed for growth" (Harms and Southerland, 1975). Because of the low solubility of phosphorus compounds, it is present in small amounts relative to need.

It is the subject of some controversy as to how much of a given nutrient is too much, but the E.P.A.'s Committee on Water Quality Criteria recommend a maximum level of 0.015 mg/L inorganic phosphorus to retard eutrophication (Sartor, 1972).

Of the limiting nutrients, phosphorus is generally the easiest to control. This is because nitrogen and carbon can enter a lake from the atmosphere or other natural sources. Phosphorus, however enters a lake primarily from man-made sources; point and nonpoint sources (Welch, 1972).

Because anthropogenic sources of phosphorus are generally so much larger than natural sources, it is most often easier to control relative to other nutrients. Secondly, phosphorus is more easily controlled than nitrogen from a nonpoint source point of view because of four soil removal mechanisms:

1. rapid removal or sorption
2. slow mineralization or sorption
3. plant uptake
4. biological immobilization

(Tofflemire and Chen, 1977).

Nitrogen

Nitrogen, along with phosphorus, is also the most common nutrient element in receiving waters. Aside from the radical effects of eutrophication on aquatic communities, human populations using lake resources may also be affected by excessive nitrogen loading. Consumption of water having nitrate levels exceeding 10 mg/L by infants may lead to methemoglobinemia, a serious blood disease (Sartor, 1972). USHPS recommends a limit of 10 mg/L of nitrate nitrogen in surface and drinking waters.

Ammonium toxicity to fish and other aquatic life is also a direct effect of excess nitrogen loading if conditions of pH and temperature are within certain ranges. In addition, during the process of nitrification ammonia may combine with oxygen in a receiving stream, thus depleting the oxygen supply.

In a study of the Occoquan Watershed, it was found that stormwater runoff generated and transported 85.2 percent of the nitrogen species entering the reservoir (Randall et al, 1978). For Whipple's Morristown, New Jersey study nitrate contributions amounted to 131 lb/day in August and 339 lb/day in March for a 5.2 mi2 area with average concentrations of 2.9 mg/L and 1.2 mg/L respectively. Ammonia contributions were around 110 lb/day in March and negative 146 lb/day in August (Whipple et al, 1974).

Like phosphorus, nitrogen may also exist in several different forms but the important forms in terms of availability are nitrates, and ammonium. Nitrogen species are more readily leached from the soil than are phosphorus species (Whipple et al, 1974). The total nitrogen test is indicative of nitrogen nutrient availability (Sartor, 1972).

Inorganic nitrogen forms such as ammonium, nitrite, and nitrate are readily available for algal growth in surface waters. The availability of organic nitrogen is a function of the rate at which organic nitrogen can be mineralized by bacteria to form inorganic nitrogen in the reactions shown schematically as follows:

Organic N -----------> Ammonium N (hydrolysis)

Ammonium N ----------->Nitrite N (oxidation)

Nitrite N -----------> Nitrate N (oxidation)

(Cowen et al, 1976).

Trace Metals.

Trace metal constituents of urban runoff consist of lead, zinc, chromium, copper, iron, manganese, nickel, mercury, arsenic, and cadmium (Randall et al, 1981). In a study by Randall et al it was found that urban runoff contained metals which predominated as Zn > Pb > Cr > Cu. Zinc and lead correlated well with volume on a per unit runoff basis (Randall et al, 1981).

Of particular significance in the consideration of trace metals, is the fact that they are non-degradable and hence persist in the environment for extended periods of time.

Trace metals tend to precipitate from solution at relatively neutral pH values and may be adsorbed on clay particles. As a result, these materials may become concentrated in the solid phases of aquatic systems. Consequently, even though the water itself may contain only small amounts of these materials, the particulate matter in the water, and especially the benthic or bottom deposits, may contain considerable quantities (Randall et al, 1981).

Toxicity to aquatic environments by metals depend upon their chemical/physical states (valence, whether they are present in complex inorganic or organic compounds etc.). On the basis of studies conducted over several storm hydrographs in Lodi, New Jersey, it appeared that the major contributors of heavy metals in urban runoff were zinc, lead, and copper. Furthermore, these appeared in greatest concentration shortly after the initiation of runoff, usually within the first 30 minutes, thus exhibiting the so-called first flush effect. Peak concentrations were found over a short period of time usually in the range of 10 to 20 minutes after the initiation of runoff (Wilber and Hunter).

Lead and zinc, like nutrients and solids, have higher average storm concentrations in areas of greater impervious cover.

Zinc and lead also have the heaviest loadings from the standpoint of concentration alone (Sartor, 1972). It is not known presently whether they are the worst polluters. Sartor found that lead loading intensities on street surfaces were highest in residential areas, comparatively high in industrial areas, and lowest in commercial areas. He also found that zinc was highest in industrial areas, moderately high in residential areas, and relatively lower in commercial areas (Sartor, 1972).

Table 2
Heavy Metals Export in Urban Runoff
kg/ha/yr

Land Use	Zn	Pb
Single family residential	0.12-0.23	0.10-0.20
Multifamily	0.68-0.84	0.31-0.39
High-rise residential	1.72-1.91	1.03-1.15
General Residential	0.06	0.02
Industrial	2.20-7.00	3.50-12.00
General Urban	0.14-0.50	0.90-1.60

Zinc

Most common zinc compounds are not toxic in low to moderate concentrations. In drinking water supplies, 5 mg/L is the USHPS maximum level. Aquatic organisms are more sensitive than humans to zinc. Concentrations as low as 0.1 to 1.0 mg/L have been found lethal to fish and aquatic organisms (Sartor, 1972).

In a study by Shaheen, it was shown that zinc may be added to roadways by the abrasion of tire particles and the leaking of motor oils (Shaheen, 1975). Analysis of street surface contaminant samples show that zinc is higher in loading intensities than any other metals with a mean value of 0.75 lb/curb mile (Sartor, 1972).

Relatively large quantities of zinc have been observed in wet and dry atmospheric fallout. The majority of these materials have been associated with combustion of fossil fuels and processing of metals.

Lead

Lead is toxic to fish, may be transferred through the food chain, and because of it's accumulative effect may at elevated levels in water supplies be a hazard to public health (Oliver et al, 1974). In vertebrate animals, lead is a cumulative poison which typically concentrates in bone (Sartor, 1972). USHPS drinking water standards limit lead to 0.05 mg/L. At higher concentrations lead has been found moderately toxic to fish and other aquatic organisms.

Initially, highly soluble inorganic lead compounds are converted to less soluble compounds in the atmosphere or in the soil. Coarser lead particles are rapidly deposited and are effectively immobilized within the top few centimeters of soil. This fraction causes an insignificant contribution to water pollution (Laxen and Harrison, 1977).

Lead is a major constituent of leaded gasoline automobile emissions. Newton, Colemane and Shepherd state that since there is a large concentration of automobiles in urban areas, lead accumulates in considerable quantities. These authors computed that in Oklahoma City, 143,000 g/day (315 lb/day) of lead is emitted by cars and they speculate that as much as 50 percent is contributed to streams in runoff (Newton et al). Again it is important to realize that this value was recorded before the widespread use of unleaded gasoline.

Lead concentrations in urban runoff have been found to be 1000-10,000 times general background concentrations in surface waters (Laxen and Harrison, 1977). This data was found prior to the widespread use of unleaded motor fuels. Heavy metal monitoring in Durham, North Carolina indicated that lead is related to land use and the increased traffic density that corresponds to commercial development. The flow weighted mean lead concentration for a particular monitoring position was 0.48 mg/L (Bryan, 1972). It is important to note that this figure also was recorded prior to the widespread use of unleaded motor fuels (1979).

Of the trace metals, lead loading intensities from street runoff are second only to zinc. The intensities ranged from 0.03 lb/curb mile in Tulsa, Oklahoma to 0.68 lb/curb mile in San Jose, California (Sartor, 1972).

Solids

Mineral matter, common sand, and silt are major constituents of street surface runoff. Consequences of overloading of solids in urban runoff are widely varied.

Sediment-laden water may increase the cross-sectional areas of streams. Other physical changes include deposition of channel bars, erosion of channel banks as a result of deposition within the channel, and obstruction of flow and increased flooding (Whipple, 1977).

Biological effects of high sediment loadings may include burial of bottom flora and fauna, thereby reducing water transparency inhibiting the transmission of light required for photosynthesis. Other effects are damage to biological structures and the clogging of respiratory, feeding, and digestive organs (Sartor, 1972).

High suspended solids concentrations may increase treatment costs when degraded sources are used for water supply. High sediment loads are of further importance because other types of pollutants may be associated with sediment. For example, sediment transports and stores adsorbed phosphorus and nitrogen (Whipple, 1977). Cations, anions, and organic compounds may be surface adsorbed onto particles and strongly held. Pesticides and heavy metals also exhibit this property and so their biological effects are modified (Sartor, 1972).

Erosion is a natural process and may contribute vast loads of sediment, largely from construction, agricultural activities, and other culturally-induced land management practices. However, within urban areas, increases in storm runoff and high peaks of energy augment natural erosive forces and greatly accelerate erosion.

Very high concentrations of suspended solids have been recorded in urban stormwater. Individual studies have recorded total suspended solids concentrations in urban runoff which range from 147 mg/L in Oklahoma City, Oklahoma to 1223 mg/L in Durham, North Carolina (Colston, 1974). Whipple estimated that the mean concentration of suspended sediments is in the range of hundreds of milligrams per liter, usually rising dramatically with peak flows (Whipple, 1977). In Tulsa, Oklahoma, a study was done to relate TSS concentration of stormwater runoff to land use type. Authors investigated characteristics of urban runoff that derived from land with different land uses. They found, as would be expected, that as urban activity (i.e. percent imperviousness) increased, the concentration of suspended solids in stormwater increased (Sartor, 1972).

With regard to land use types, Whipple found TSS concentrations which ranged from 2052 mg/L at a light industrial site to 84 mg/L at a residential site. Median concentrations were 445 mg/L at a golf course and 300 mg/L at a commercial site (Whipple, 1977).

Along with concentrations, loadings also vary with land use. Urban construction sites produced the largest annual yields of suspended sediment with a range of from 7 to 10 tons/acre (Hall, 1984). In the Durham, North Carolina study, sediment yield on the basin was estimated to be 4,500 tons/sq mi/yr (Bryan, 1972). This is over an order of magnitude larger than the 200-300 tons/sq mi/yr suggested by Guy and Ferguson as being normal (Bryan, 1972).

Randall et al (Randall et al, 1977) calculated the yearly loading of suspended solids in urban runoff to be about 48 times the yearly loading contributed by effluent from secondary treatment plants serving the area.

Infiltration BMPs

The term BMP is defined as a practice or a combination of practices determined to be the most effective and most affordable methods for preventing or reducing pollution from nonpoint sources to a level compatible with water quality goals. They may be structural devices or management practices (Novotny and Chester, 1981).

Structural BMP's are porous pavement, wet and dry detention ponds, infiltration trenches, grass swales, and rooftop storage. Management practices are those such as street sweeping, zoning, and anti-litter legislation.

Volume control BMP's are those which substantially reduce the volume of runoff, thereby reducing the amount of pollutant generated.

The application and selection of BMP's should be based on :

1. type of land-use activity
2. physical conditions in the watershed
3. pollutants to be controlled
4. site-specific conditions

Porous Pavement Design

The major reason for the investigation of porous pavement is that it allows the dual use of land surfaces. In areas that employ the use of combined sewers, porous pavement is beneficial because it alleviates combined sewer overflow pollution.

Porous pavement, as with all volume control BMP's, allows stormwater to percolate into the soil rather than overflow combined sewer systems and could alleviate much of this pollution. The use of porous pavement could help to alleviate water shortage problems for areas of the country subject to water supply deficiencies. By allowing precipitation to percolate back into the soil, this Best Management Practice serves to preserve natural drainage patterns, prevent flash flooding, and provide skid resistance by preventing hydroplaning.

Thelen presented design specifications for porous paving applications. Design specifications planned for many applications and locations are:

1. it shall imbibe all or most of the rainfall on it and water from melted snow and ice.
2. it shall carry the loads without damage
3. it shall survive freeze-thaw and weathering: long life is required.
4. it shall be hard to damage but easy to repair: maintenance costs must be moderate.

5. it shall not plug up: removal of trash by occasional weeping or vacuuming may be permissible.

(Thelen, 1972).

Thelen found that there are at least two important advantages to the treatment of urban runoff by porous pavement:

1. the water is filtered as it passes through the pavement, sub-base, and subgrade. Aerobic bacteria are able to survive under the pavement and may destroy organic contamination that could clog the soil or pollute the water.

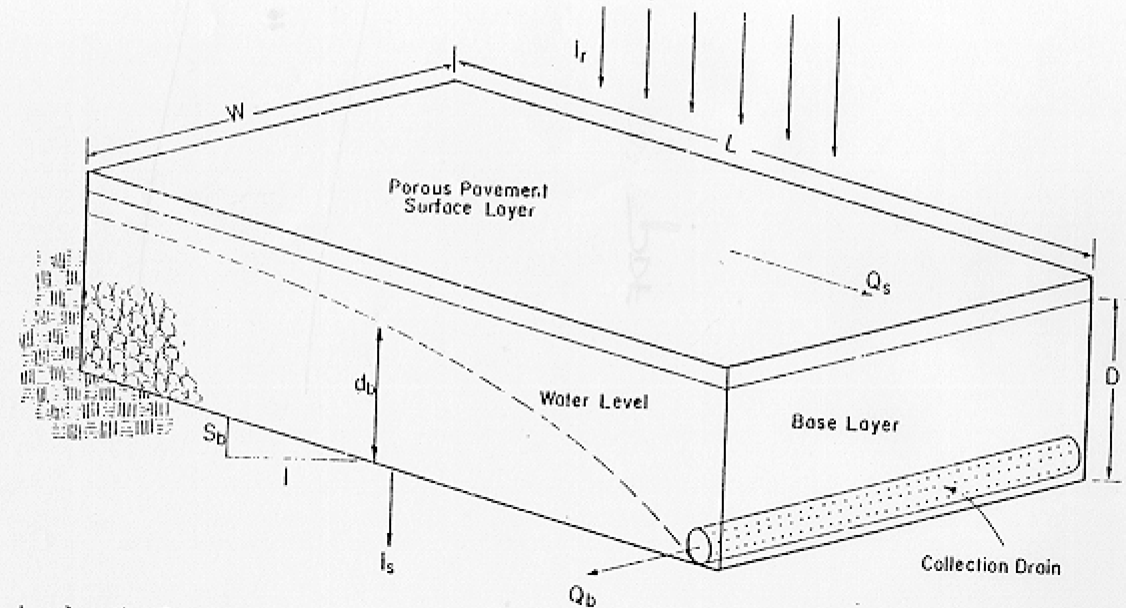
2. Depending on the pore size of the system in question, much of the dust and dirt fraction would be washed through to the underlying soil. The soluble dust and dirt fraction and any biodegradable wastes would be metabolized by soil bacteria. Pesticides, herbicides, air pollutants, nitrate and phosphate fertilizers would also be metabolized.

Thelen also found such open graded asphalt concrete exhibited superior physical characteristics, was low in cost, and could be laid by conventional paving methods. Infiltration rates varied from five in/hr to 25 in/hr. Thelen defines porous asphaltic concrete as "a graded aggregate cemented together by asphalt cement into a coherent mass, with sufficient interconnected voids to show a high permeability to liquid water." Because porous asphalt is more subject to scuffing than is conventional asphalt, it is suggested that 50 to 60 open-graded asphalt be used in the south, 65 to 80 in the mid-Atlantic states, and 85 to 100 penetration grade in the northern-most states. Also it is recommended for low traffic volume areas because it has a lower tensile strength than conventional pavement (Northern Virginia Planning District Commission, 1987).

Porous pavement is characterized as having 16 percent voids as opposed to conventional asphalt pavement which has three to five percent voids (NVPDC, 1987). In manufacture, the graded aggregate is derived by heating to 275-300^{o} F and mixing with asphalt cement. Typical thickness of the pavement is two and one half to four inches (NVPDC, 1987). Aggregate size is typically one to two inches. The hot mix is applied in smooth layers and compacted to design density by heavy rollers. Long term tests are still required to evaluate clogging resistance and quality of water that filters through. A four inch pavement and six inch base, with a void ratio of 16 percent could store two and four tenths inches of runoff volume (Thelen, 1972). Cross sections of typical porous asphalt pavement can be seen in Figures 1 and 2.

Infiltration trench design.

An infiltration trench is an overland flow structural modification used to reduce runoff and pollutant loadings. Typically, they are installed at the base of a modest slope to allow runoff to filter down into subsoil to prevent overland runoff from leaving the site (Maryland Department of Natural Resources, 1984).



L length of pavement
W width of pavement
D depth of base layer
S_b slope of base layer
i_r rainfall intensity
i_s infiltration rate
d_b depth of water in base layer
Q_s surface discharge
Q_b collection drain discharge

Figure 1. Cross Section of Typical Porous Pavement Facility [39]

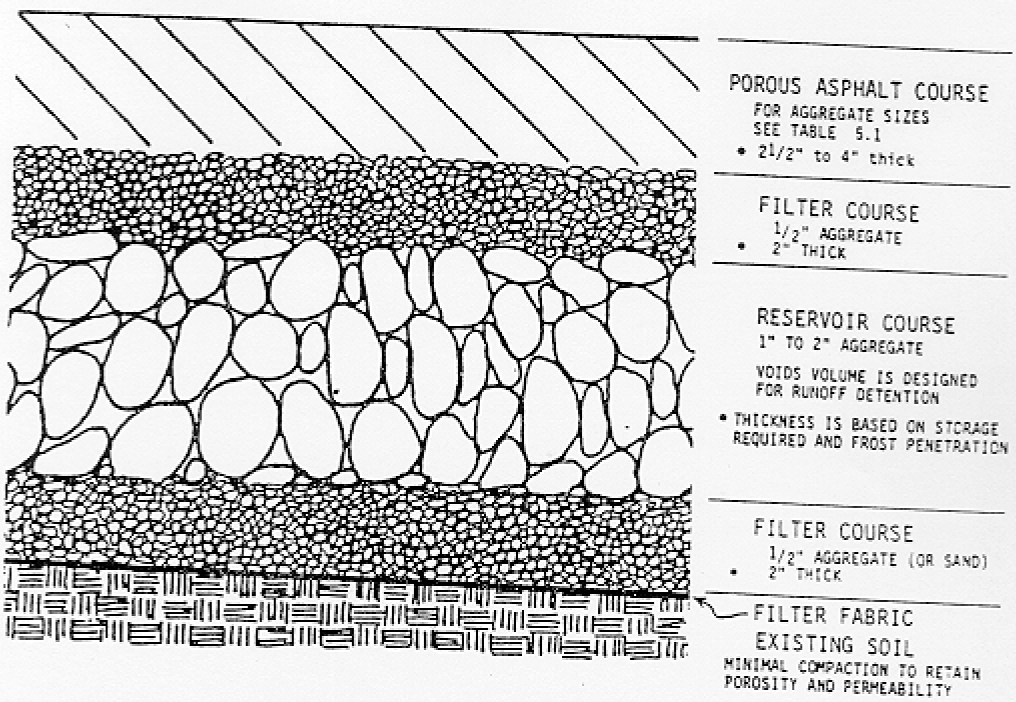


Figure 2. Porous Asphalt Paving Typical Section [38]

Field et al have published recommended design criteria for infiltration trenches:

1. one foot of grassed soil over the aggregate reservoir
2. seasonal groundwater table approximately 2-4 feet below the bottom of the trench
3. a minimum three foot depth of aggregate
4. trenches should be located at least 100 feet horizontally from any water supply well
5. aggregate material for the infiltration trenches shall consist of a clean aggregate with a maximum diameter of 3 inches and a minimum diameter of 1.5 inches
6. void space for aggregates should be between the range of 30-40 percent
7. recommended storage time is between 48 and 72 hours.

(Field et al, 1977).

The cross-section of a typical infiltration trench is shown in Figure 3.

Schematic of an Infiltration Trench

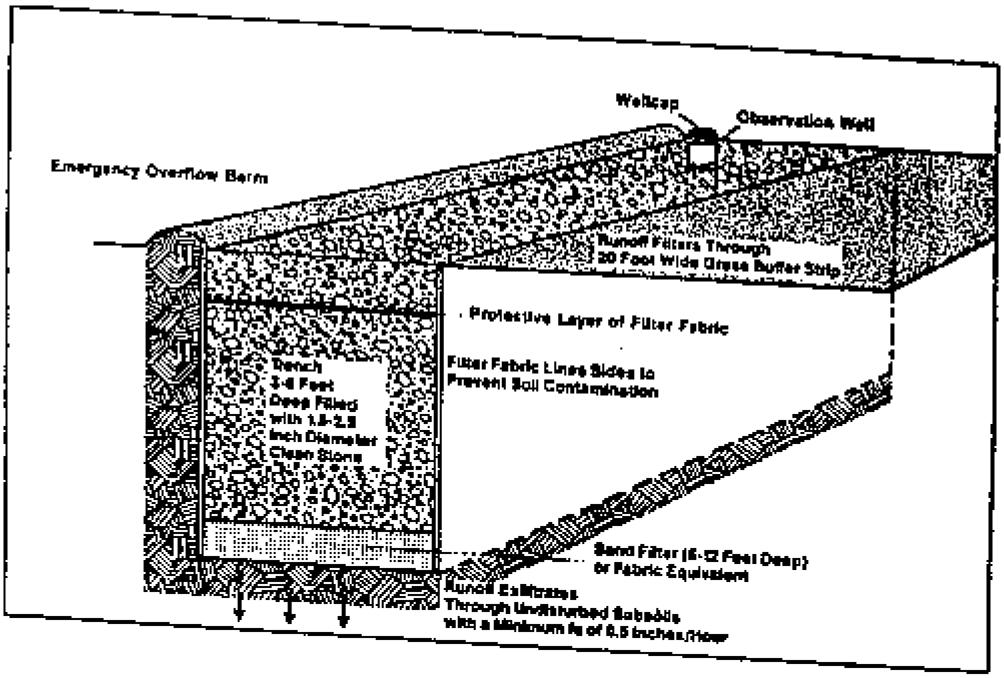


Figure 3. Schematic of an Infiltration Trench [38]

INFILTRATION PRACTICES &

POLLUTANT REMOVAL MECHANISMS

Fate of Nitrogen Species in Soil

The application of runoff to natural soils is a sound, inexpensive biological treatment. Nitrification is predominantly effected by autotrophic nitrifying bacteria if conditions are favorable. Nitrobacter are autotrophic bacteria with fairly precise requirements: pH around neutrality, sufficient CO_2, moisture a little below field capacity, and moderate aeration. The buildup of NH_4^+_N is often the indication of inefficient nitrification. Likewise, presence of nitrate-N is a sign of favorable conditions for nitrification (higher pH, fairly dry, good supply of ammonium-N) (Hebert, 1979).

Nitrosomonas, a common soil bacterium converts ammonium, NH_4^+, under aerobic conditions to nitrites, NO_2^-, and derives energy from the oxidation. Nitrification occurs as:

$$2NH_4 + 3O_2 \text{ ------> } 2NO_2^- + 2H_2 + 2H_2O$$

(Metcalf and Eddy, 1979).

Nitrobacter, another common soil bacterium, also obtains energy from the subsequent oxidation of nitrites to form nitrates:

$$2NO_2^- + O_2 \text{ -------------> } 2NO_3^-$$

(Metcalf and Eddy, 1979).

Nitrates, NO_3^-, produced in excess of the needs of plant life are carried away in the water percolating through the soil because the soil groundwater generally does not have sufficient anion exchange capacity to immobilize them (Metcalf and Eddy, 1979). Nitrates are easily leached from the soil whereas ammonia is better adsorbed and retained on the surfaces of clay particles (Walker, 1975).

Autotrophic nitrifying bacteria are ubiquitous where ammonia and nitrite are available. Numbers of nitrifiers in soil are never very high, usually some thousands per gram in a fertile arable soil. This is probably because of their slow growth rate and requirements for considerable amounts of ammonia or nitrite as energy sources (Walker, 1975).

Nitrogen removed in infiltration practices may be stored in the soil or may diffuse into the atmosphere (Lance, 1972). Nitrogen not removed from the water will eventually reach the water table as nitrate or ammonium ion, NH_4^+ depending on the amount of oxygen available (Lance, 1972). Acid soils often contain fairly constant amounts of ammonia during the year, but nitrate is much more variable. The most desirable nitrogen removal mechanism is reduction of nitrate to nitrogen gas by biological denitrification. In this process, nitrogen is transformed into a relatively stable form that will not contaminate groundwater or contribute to eutrophication.

Denitrification, shown as the terminal electron acceptor in the oxidation of glucose occurs as:

$$C_6H_{12}O_6 + 4NO_3 \text{---------}> 2N_2 + 6H_2O + 6CO_2$$

(Lance, 1972).

Under anaerobic conditions, nitrates and nitrites are reduced by denitrification. Nitrates are reduced to nitrites, which are further reduced to ammonia by a few bacteria, but most carry the reduction to nitrogen gas, which escapes to the atmosphere. Denitrification is desirable because when nitrates are reduced to nitrogen gas they are lost from the aquatic system and are not available to spur the growth of algae and other aquatic plants in receiving waters. Soils contain an abundance of denitrifying bacteria that can use oxygen as a terminal electron acceptor under aerobic conditions or nitrate in the absence of oxygen. According to Lance there are three conditions which control the establishment of a system for denitrification:

1. oxidation of ammonium into nitrate.
2. passage through an anaerobic zone following oxidation to nitrate.
3. provision of an adequate energy source (organic carbon) in the anaerobic zone for the denitrifying bacteria.

(Lance, 1972).

Experimentally, it is found that denitrification is affected by pH, temperature, soil moisture content, and extent of aeration (Bremmer and Shaw, 1958). Denitrification rates may be determined by measuring the losses of nitrogen gas during incubation of soil samples. A considerably slower rate was found in soils of pH 5.8 than in soils with higher pH values. Rates were very slow in soils with pH 4.1 and almost undetectable in soils of pH 3.8 or less.

Rates increased rapidly with a temperature from 2^{o} C to 25^{o} C but was not significantly affected by an increase in temperature beyond 25^{o} C (Bremmer and Shaw, 1958). Little denitrification takes place if the moisture content is less than 60 percent of the water holding capacity of the soil. Denitrification is much more rapid in soil saturated with water than in soil at lower moisture levels. This is because the supply of oxygen in water saturated soils is not adequate to meet the requirements of the microorganisms so the denitrifying microorganisms utilize nitrate instead of oxygen as a hydrogen acceptor and so causes denitrification (Bremmer and Shaw, 1958).

One of the observed effects of bubbling oxygen through soil suspensions was decreased denitrification (Lance, 1972). Rates were also observed to decrease with decreasing amounts of organic matter as substrate. For every mg of nitrate nitrogen reduced to nitrogen gas, 1.3 mg of carbon is required. The actual amount necessary may vary with carbon source and amount of carbon consumed by microorganisms other than denitrifiers (Lance, 1972).

Myers states that denitrifying activity in unamended surface soils and subsoils is directly related to the total content of native organic matter which contains the energy substrate utilized by the microorganisms. Because organic matter decreases with depth, denitrifying activities of subsoils are consequently low when compared with surface soils (Myers and McGarity, 1972).

The accumulation of nitrite during denitrification under adverse conditions has been observed (Bollag and Barabasz, 1979). Certain unfavorable growth conditions such as the presence of various pesticides, nitrification inhibitors, and heavy metals can cause the accumulation of nitrite, which is not usually found during the denitrification process (Bollag and Barabasz, 1979).

In a laboratory study by Gilliam it was demonstrated that there was a relationship between oxidation reduction potential, (Eh), and the loss of nitrogen by denitrification. When the Eh of soil dropped to 300 millivolts (mv), or below, there was a large loss of nitrogen by denitrification. There was very little change in Eh or loss of NO_3-N when the soil water varied from 34-41 percent but a large change when the soil moisture was increased above 41 percent (Gilliam et al, 1979).

Nitrogen adsorption

Ammonium adsorption to soil particles has been observed to follow the Freundlich isotherm:

$$A = KC^{1/n}$$

where: A = adsorption in micrograms of NH_4^+ per gram of dry soil

C = concentration of NH_4^+ in the solution when equilibrium is reached.

k,n are empirical constants.

(Preul and Schroepfer, 1968).

Adsorption and biological action are the main factors which control the movement of nitrogen through soils. One of these factors may dominate the other depending on the soil environment. Adsorption on soil may be an important mechanism in the inhibition of nitrogen travel when the nitrogen is in the form of ammonium. However, the presence of other ions may have a limiting effect on NH_4^+ adsorption.

Where nitrogen is in the form of nitrates, and at the usual pH of wastewaters (7.5-8.5), practically no inhibition to movement is offered. In NH_4 flow through a soil bed under limited oxygen conditions, as with solution saturation of the soil interstices, adsorption occurred with only minor biological interference. Under well aerated conditions, a major portion of the total denitrification resulted within one to two feet of the influent surface (Preul and Schroepfer, 1968).

Ammonium ions in runoff may be adsorbed by negatively charged clay and organic colloids of soil. The cation exchange capacity of different soils varies tremendously and depends on the amount of organic matter and amount and type of clay minerals in soil. Immobilization can be induced by adding organic matter, by microbial activity in the rhizosphere, or by fluctuations of available soil carbon during climatic variations. The lower the C/N ratio, the shorter is the period of immobilization (Hebert, 1979).

Large quantities of ammonium ions can be adsorbed by clays, but such immobilization is not permanent because it can be replaced by other cations or removed by nitrifying bacteria. Microorganisms immobilize much more ammonium nitrogen than nitrate (Lance, 1972).

Ammonia reacts with the organic portion of the soil, forming complexes resistant to leaching and decomposition (Lance, 1972). In many cases the organic fraction plays a greater role than the mineral fraction in the retention of ammonia in soil (Lance, 1972). Nitrogen may be transformed to the atmosphere by non-biological volatilization of ammonia. Volatilization of significant quantities of ammonia requires considerable air-water contact, and this would not be provided in a ground water recharge system (Lance, 1972).

Fate of Phosphorus species in soil

Interest in the movement of phosphorus by water transport in soils tends to center on the inorganic forms almost to the exclusion of it's organic forms. Dissolved inorganic phosphorus is of major importance because it is readily available to aquatic plants and is the fraction which influences the water and sediment phosphorus components in lakes and other surface waters (Reddy et al, 1978). Organic phosphorus represents 20 percent - 60 percent of the total phosphorus in many soils and seems to be the form most mobile under leaching of soils and upon hydrolysis or microbial mineralization also becomes a source of phosphorus to aquatic plants (Conesa et al, 1979).

As with nitrogen, organic phosphorus undergoes cycles of mineralization and immobilization (Tofflemire and Chen, 1977). Phosphate can be removed from runoff by several mechanisms;

1. rapid removal or sorption
2. slow mineralization and insolubilization
3. plant uptake
4. biological immobilization

For rapid infiltration systems, because of the rapid kinetics, the first two mechanisms are most important. The Langmuir adsorption isotherm implies an adsorption maximum and has been used by a number of authors (Tofflemire and Chen, 1977; Preul and Schroepfer, 1968) to describe the rapid removal mechanism. Many soil components adsorb phosphate ions: iron, aluminum hydroxide, mineral and organic colloids, and calcium carbonate. After two to five days, the additional phosphorus removal by soils slows and is termed slow mineralization. Factors such as soil moisture tension, pH, temperature, and redox potential affect phosphorus removal (Tofflemire and Chen, 1977).

Because of the strong retention, phosphorus migrates little in the soil profile and quantities leached are low, except in very sandy soils (Tofflemire and Chen, 1977). Although most rapid infiltration systems do well in phosphate removal it is possible to saturate the soil and observe subsequent phosphate breakthrough. The capacity of individual soils to remove phosphorus varies widely and can be measured by simple adsorption isotherm tests.

The volume of soil in rapid infiltration systems also affects the total phosphorus removal capacity. For many soils, the "B" horizon has a much higher phosphate capacity per unit weight than the C horizons. However, in most rapid infiltration systems, the B horizon is not utilized, although it could be (Tofflemire and Chen, 1977).

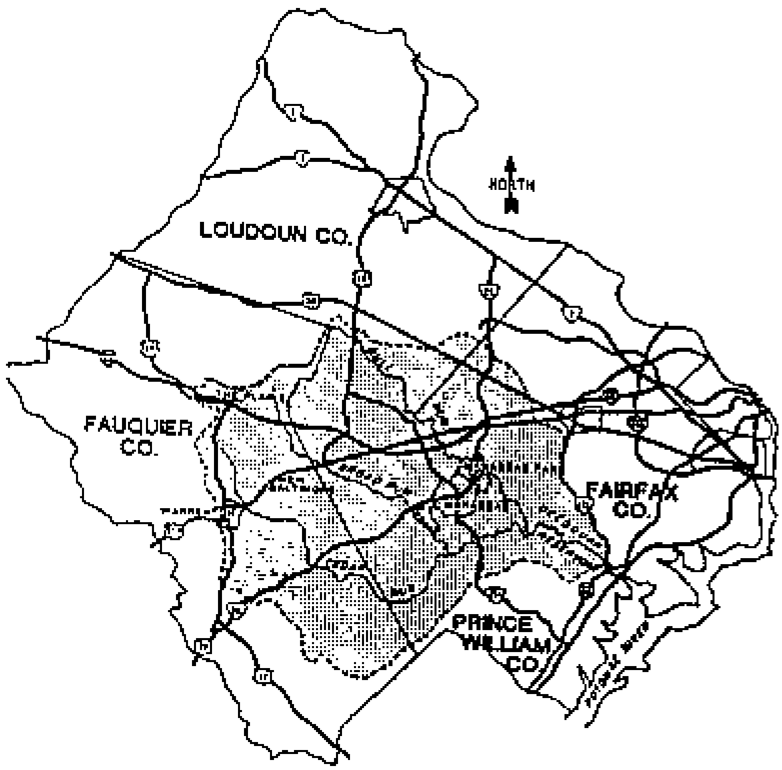
Chapter III
Material and Methods

Site description

The city of Manassas lies in the foothills of the Blue Ridge Mountains about 60 miles west of the Chesapeake Bay and 25 miles southwest of Washington D.C. Location in the middle latitudes of the Northern Hemisphere makes general atmospheric flow from west to east. Summers are warm and humid with a mean ambient air temperature of 87 F (30.5 C) and winters are mild with an average temperature of 23 F (-5 C). Annual rainfall ranges from 30 - 60 inches (76 -152 cm) and average snowfall averages 24 inches (61 cm) a year. The prevailing winds are out of the south, except during winter months when they come from the northwest (National Oceanic and Atmospheric Administration, 1982).

The site for the current study was located 6 miles southeast of Manassas, Virginia on Davis Ford Road (Rt. 663). The location is shown in Figures 4 and 5. A street map is shown in Figure 6. The parking area was the public parking facility for the Davis Ford Park, which consists of a baseball stadium and two public diamonds.

Traffic on Rt. 663 has been observed to be moderately heavy during rush hours and is composed of private automobiles, construction vehicles, and occasionally tractor trailers. Traffic on the parking lot itself may be described as very light with 2-5 park employee-owned private vehicles occupying spaces six days a week. The lot may be filled to capacity three nights a week during the summer baseball season.



OCCOQUAN WATERSHED LOCATION MAP

Scale: 1 in. = 8 mi.

Figure 4. Occoquan Watershed Location Map [20]

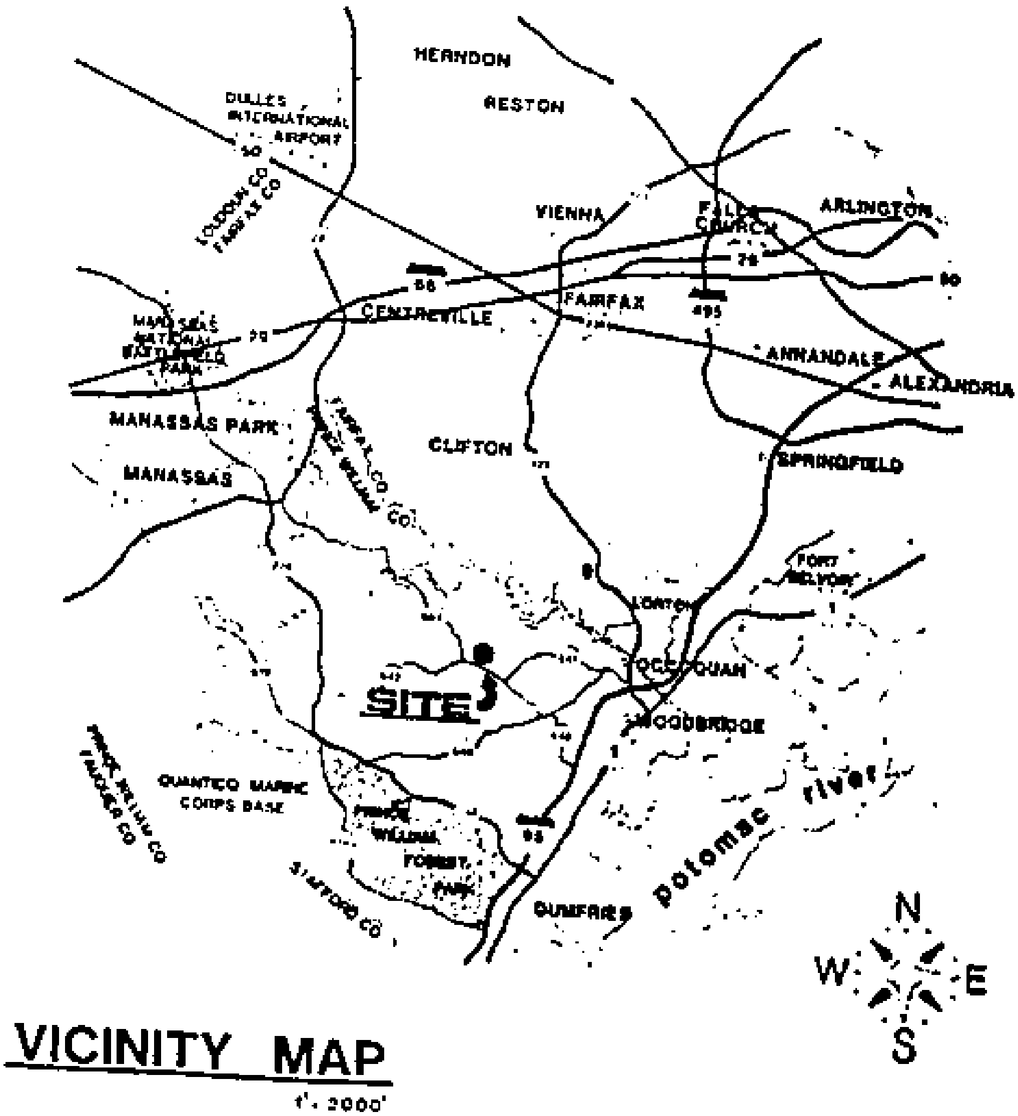


Figure 5. Vicinity map [54]

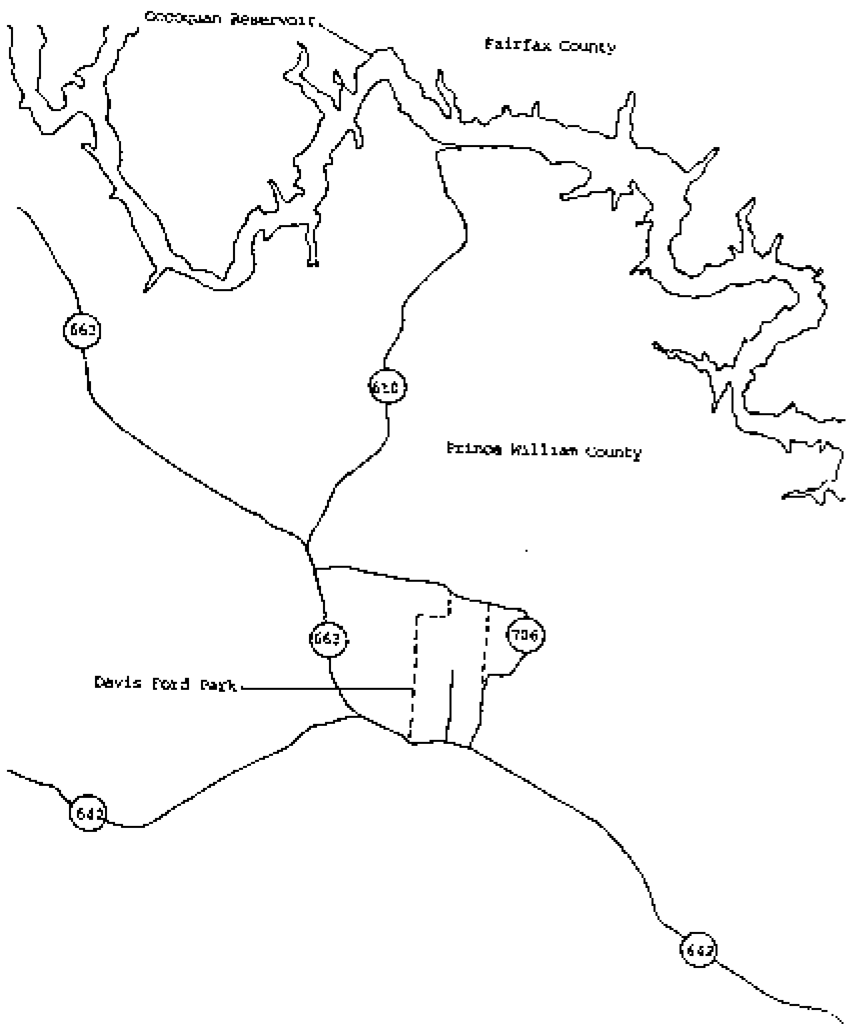


Figure 6. Area Street Map [53]

The study area may be classified as a rural, light industrial area 500 yards north of Rt. 663. The study area included 0.537 acres of conventional pavement contiguous to a 0.553 acre lot paved with porous pavement. Adjacent to these two areas is a 0.269 acre lot of conventional pavement draining to an infiltration trench.

A general site plan is shown in Figure 7. The catchment identification system was as follows:

PP01: 0.537 acre control

PP02: 0.553 acre porous pavement

PP03: 0.269 acre conventional pavement bordering an infiltration trench

Total parking spaces for the three spaces were as follows:

PP01: 81 spaces

PP02: 79 spaces

PP03: 101 spaces

Examination of the applicable site soils map showed the site soils to be described as sandy loams, loams, and silt loams from the Hoadly, Neabsco, and Meadowville soil classifications respectively (USGS, 1994). These soils described being from hydrologic

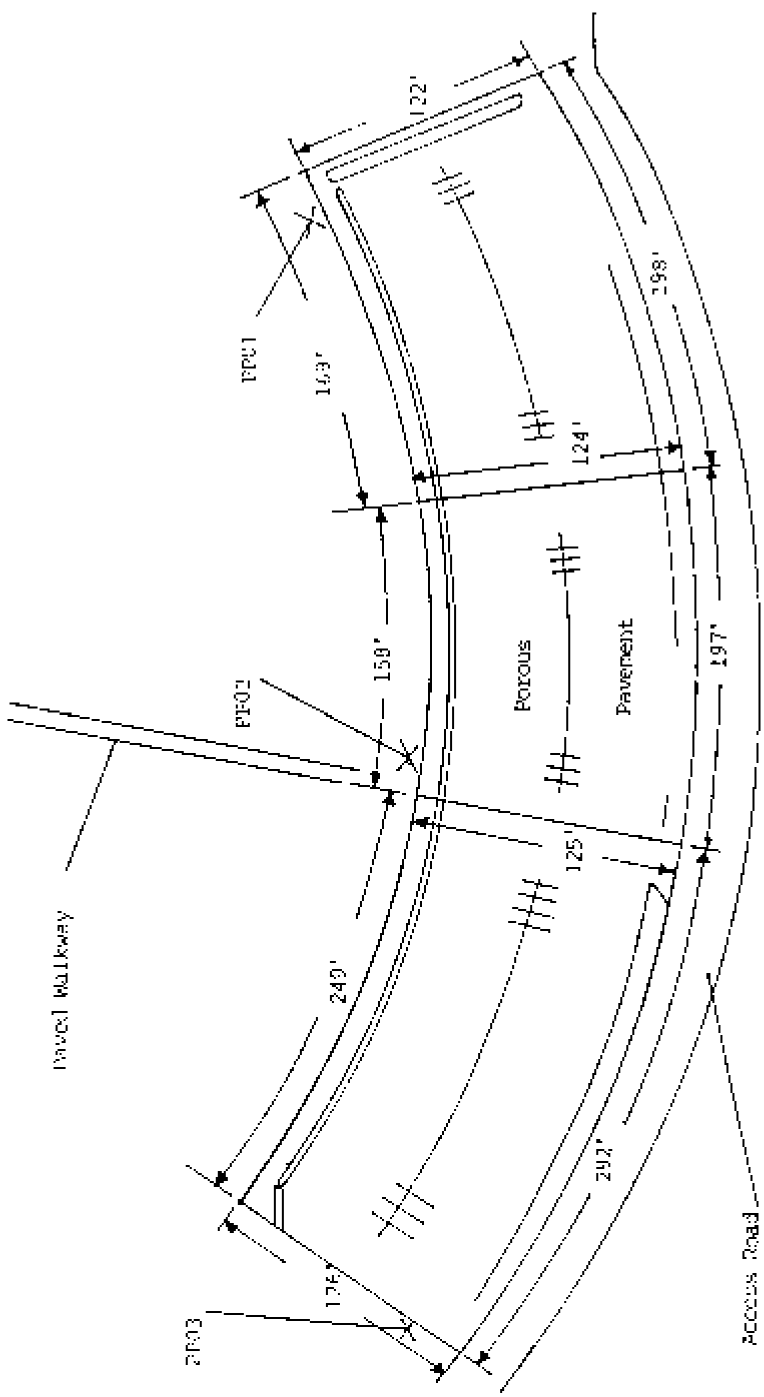


Figure 7. General Site Plan [54]

groups A and B, and as being moderately well drained to well drained with infiltration rates ranging from 0.2 to 6.0 inches/hour.

The physical construction of the porous pavement BMP consisted of 2.5" of porous bituminous pavement applied over 2" of Virginia Department of Highways and Transportation (VDHT) #7 (0.75" diameter or smaller) stone base. A VDHT primer coat was applied between these two top layers. Beneath this base was laid 12" of VDHT #2 (3.0" diameter or smaller) stone. A filter fabric was placed below the subbase and above the compacted subgrade.

The infiltration trench BMP surface was made up of a 9' wide gabion mat which held in place a 2' deep covering of 4" diameter rip-rap. Below this covering was the infiltration area made up of 1" to 2" aggregate. The dimensions of the aggregate area were 63' * 6' * 4.5'. Filter fabric of the same type used in the porous pavement construction was used to line the perimeter of the trench. A 6" perforated PVC pipe was placed vertically in the aggregate at the upward sloped end of the trench to be used as a site well.

The collection system for both BMP's consisted of underdrain modified 8" perforated pipe. The outlet works for the three areas utilized 18" reinforced concrete pipe. The monitoring stations were built above standard median drop inlets. Bituminous curbing 6" in height and 9" wide were constructed to direct the runoff flow toward the drop inlet at the control catchment and toward the infiltration trench at PP03.

The Davis-Ford porous pavement facility was designed for a two year storm. In the Prince William County area a two year storm's volume is 3.5 inches of precipitation within a 24-hour period. The storage volume was calculated from the following equation:

$$Vol = L * W * D * Vr$$

where Vol = volume of voids in facility (cf)

L = length of facility (ft)

W = width of facility (ft)

D = depth of reservoir course (ft)

Vr = voids ratio of stone aggregate

The total storage volume for the porous paving facility was calculated to be 9,636 cubic feet (cf). However the site had an approximate 2.4 percent slope in the direction of the principal flow, and an upslope width at that point of 125 feet. Given this site geometry, it may be calculated that the hydraulic grade line would have intersected the downstream paving surface at a point when the storage pool was 83 feet from the upslope site boundary. This would indicate a somewhat reduced net storage of approximately 1926 cf. Even so, the site had a potential storage volume equal to 0.50 inches of rainfall on the paving surface.

Using the same equation, the storage volume may be calculated for the infiltration trench. It was calculated to be 454 cf.

The original study design included stormwater monitoring from the three pavement areas for a period of 26 months from May 1, 1985 through June 30, 1987. Following the discovery of a clogging problem in the upper layers of the infiltration trench, the site was modified and the study of all catchments was extended through June 30, 1988.

FIELD MONITORING TECHNIQUES

General

Each of the 3 pavement areas were supplied with a flow monitoring/sample collection system. Housing consisted of 3.5' x 3.5' x 4.5' fiberglass utility sheds which housed all of the monitoring equipment.

Each station was equipped with 2 wet-cell 12 volt automotive batteries as a power source. Batteries were changed once a week to insure ample power to all equipment. A schematic diagram of the major equipment appears in Figure 8 (OWML, 1987).

OWML staff performed site maintenance visits on a minimum frequency of once per week. The same site visitation schedule was also adopted for wetfall/dryfall sampling stations. Routine maintenance activities included the following:

1. battery changes.
2. equipment performance checks.
3. equipment changes as required.
4. minor instrumentation repairs as required.
5. major and minor site maintenance and repair as required.

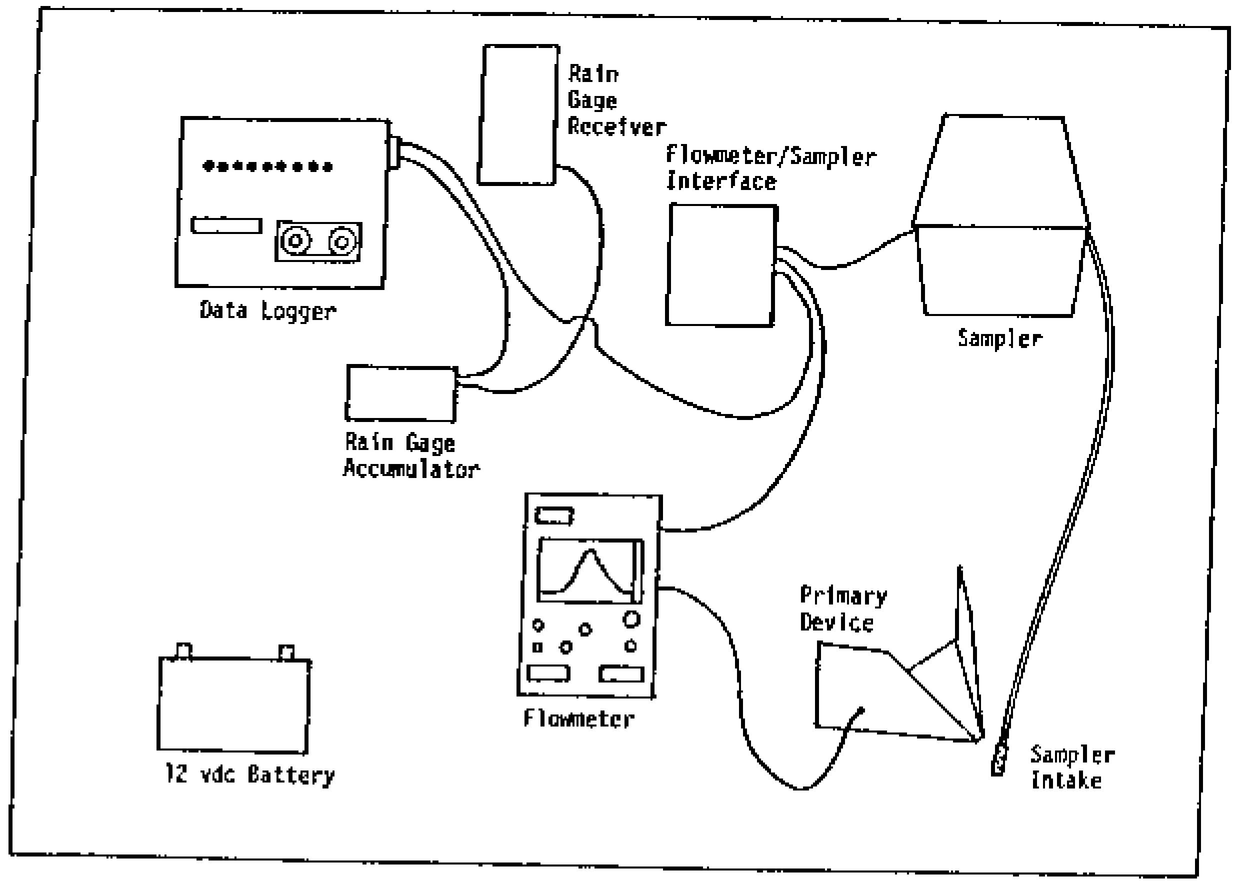


Figure 8. Schematic diagram of flowmetering and sampling equipment [55]

Precipitation Measurement

A rain gage was installed approximately 70 meters (230 feet) northeast of the parking lot location. A Weathertronics tipping-bucket rain gage (Weathertronics Inc.,1987) with a measurement sensitivity of 0.25 mm (0.01 inch) was used for precipitation measurements. The gage had excellent response characteristics at rainfall intensities up to 4 inches per hour. At each 0.01 inch increment of rainfall, the tip of the rainfall bucket caused a momentary contact closure of an attached mercury switch. The closure was sensed at monitoring station PP02 via a buried wire. An electrical counting circuit was utilized to total the number of contact closures and to record the accumulated total to a digital channel on the station data logger.

Wetfall/Dryfall Sampling

Atmospheric deposition rates were monitored using a Model 30 Aerochem Metrics wetfall/dryfall monitor (Aerochem Metrics). Automatic collection and segregation of wetfall and dryfall samples was achieved using this device. The dryfall collector was exposed to the atmosphere during intervals of dry weather while a "roof" covered the wetfall collector. At the beginning of a rainfall or snowfall event a moisture sensor would initiate the movement of the roof over the dryfall collector to expose the wetfall collector to the atmosphere. Both wetfall and pre-storm dryfall samples were collected after the completion of a storm event.

The wetfall\dryfall collection device was located at a fenced site 70 meters northeast of PP03. The collector was located about 1.8 meters above the ground surface.

The wetfall/dryfall collection system was visited prior to and following each storm event. Wetfall/dryfall sample buckets were collected and replaced with clean, acid washed buckets for the next storm.

Sample buckets were transported to and from the lab with lids attached. The date and time each sampling container was placed into service and taken out of service was noted on each container. Buckets were not exchanged prior to the completion of a precipitation event. Dryfall buckets were exchanged every two weeks if a precipitation event had not occurred during that allotted time. Otherwise, the normal procedure was to exchange the dryfall bucket when the wetfall bucket was changed.

Wet and dryfall accumulation rates were computed using known relationships between the horizontal surface area of the collection bucket exposed to the atmosphere, duration of the collection interval, and concentration (wetfall) or mass (dryfall) of the captured constituent. Upon return to the laboratory, dryfall samples were retrieved from the collectors by washing and policing the interior surfaces with sequential aliquots of reagent grade water. Samples were then transferred to a volumetric flask and diluted to a known volume, so that subsequent concentration measurements could be related back to the original mass of any contaminant of interest.

The adopted procedure required that the concentration of dryfall be calculated by recovering the bucket contents with 250 ml of reagent grade water. In the case of wetfall, the contents of the collection vessel were carefully mixed, and transferred to a graduated cylinder in order to measure the volume.

Flow Measurement

During construction of the site, a separate storm sewer was installed for each pavement area to collect surface runoff or infiltration. Each was fitted with a primary flow control device which allowed the measurement of flow. The primary device used at each site was an 18 inch Palmer-Bowlus flume (Palmer and Bowlus, 1936). Outlet storm sewer slopes were maintained at values less than three percent in order to avoid affecting the rating curve of the flumes.

The primary flow measurement devices provided a relationship between static head on the device and discharge, thus allowing flow determination by making stage measurements with a secondary stage recording device. In order to accomplish the required accuracy in measurements of static head, pressure transducer flowmeters of the bubbler type were employed. The devices operated by sensing the static head above a bubbler orifice anchored in the channel in such a way as to avoid being subject to any velocity head influences.

In addition to sensing static head, the flowmeters employed had the capability of supervising sample collection by activating an associated sampler. The provision of a rating curve stored in Erasable-Programmable-Read-Only Memory (EPROM), and totalizing circuitry in the equipment made possible the initiation of sample collection at equal increments of flow volume through the primary device (Instrumentation Specialties Company, 1980).

Sampling Methods

Stormwater samples were collected during runoff events at each monitoring site by automatic samplers. The sample retrieval device was selected on a set of criteria including:

1. intake velocities equal to or greater than 3.0 fps to insure adequate suspended solids recovery.
2. composite sample collection capability
3. collection of constant sample volume with changing suction head to allow equal volume compositing
4. pre- and post-sample purges of intake lines to clear lines and avoid sample carry-over
5. automatic retry upon collection of insufficient sample volume to allow equal volume compositing
6. activation by either an internal timer or an external signal
7. operation on 12 vdc electrical supply

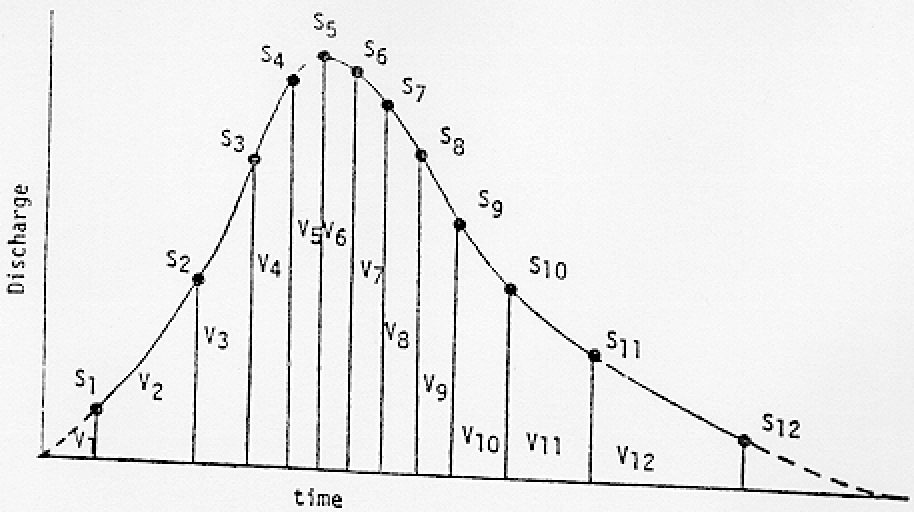
A Manning S 4040 automatic sampler (Manning Environmental Corporation, 1974) was installed at each station to collect samples of runoff at minimum designated intervals of 100 cubic feet for PP01 and PP02 and 10 cubic feet for PP03. These samplers employ a vacuum intake chamber design and sample control electronics that are well matched to the criteria listed above. Sample intakes were situated in the drainage channels proximate to the flowmetering equipment locations, and at points where the flow was determined to be well-mixed.

Stations were configured to automatically produce a flow weighted composite sample of the runoff event in the field. This was done using a flow totalizing circuit. This approach allowed a real time integration of the total volume passing through the primary device. Analysis of the samples retrieved from the field using this procedure resulted in the determination of event mean concentrations, or EMC's, for the constituents of interest. EMC may be taken as the concentration that would have been determined had the entire stormwater flow been captured, mixed and a single sample abstracted for analysis. Samples were collected in clean, acid washed, 15 liter Nalgene sample carboys (Nalgene, 1989). The process involved collecting a fixed volume sample at equal increments of total flow and placing it into a composite vessel in the insulated sampler base. Because each of the components of the total composite represented the same volume of total flow, constant, and therefore equal, volumes could be automatically collected by the sampling equipment at each station. The method is shown schematically in Figure 9 (Instrumentation Specialties Company, 1980). Stormwater samples from the BMP stations and control were collected immediately following the cessation of runoff. At times, portions of storm runoff composite samples were collected and returned to the lab prior to event completion in order to prevent samples from standing in the field for extended time periods. Samples were logged in and refrigerated to await analysis.

Data Recording

Sample times and flow information were stored on an AD Data Systems model ML 10-A data logger (A.D. Data Systems Inc., 1980). This device made possible the recording of all precipitation, flow, and sample collection data on a single, machine- readable data cassette, a factor which substantially reduced the data transcription errors invariably encountered with manual methods. Tapes from the instrument were retrieved at the same time as stormwater samples. The data from the tapes were reduced using a cassette reader, and software developed in-house at OWML.

Equipment modifications during the course of the study included installation of data loggers at each of the three stations. For the period September 27, 1985 to June 27, 1987 data from each of the three monitoring stations was collected by a single data logger housed at PP02. It was hypothesized that the distance between the three stations was the cause of extensive interference in signal recording. Following the



NOTES:

1. With the exception of V_1, all V_i are equal.
2. If V_i is very small, the error induced by unequal volume represented by sample 1 and unsampled volume past sample 12, will also be small.
3. If V_i is small, a better representation of flow near peak is also obtained.

Figure 9. Schematic of automated flow weighted compositing procedure [55]

installation of a separate data logger for each station, the result was a reduction in random erroneous signals.

Groundwater Monitoring

As a study enhancement installation of four soil pore water lysimeters on the perimeter of the infiltration trench allowed the collection of soil pore water as it filtered through the soil after running off of the pavement surface. They were installed at a depth of about 1.45 meters.

By analyzing runoff water samples from this depth, it was possible to observe how constituents passing through the soils were affected by:

1. microbial action within the soil.
2. filtering action by soil particles.
3. sorption
4. precipitation
5. coagulation/flocculation
6. ion exchange

Following six months of monitoring, it was found that the uppermost layers of the infiltration trench had become clogged. Modifications to the infiltration trench included removal and replacement of the stone bed.

A typical lysimeter may be seen in Figure 10. Lysimeter locations are shown in Figure 11. These lab-built lysimeters consisted of a 2 inch diameter PVC pipe with a porous ceramic cup glued to one end and a cap at the other end. Two lengths of tygon tubing were installed through the capped end; one long enough to reach to the bottom of the pipe to the cup end. The other tube reached to the middle of the pipe length. With these devices, a vacuum could be established inside the lysimeter by pumping out the air and clamping off the two tygon tubes. In this manner, the lysimeters were prepared (charged) for a storm event and infiltration of runoff. Following a storm event, the tubes were unclamped and the soil pore water was ejected from the lysimeter into a clean sample bottle, labeled, and transported back to the lab for analysis.

Laboratory Methods

Upon arrival in the Laboratory, all samples were assigned a unique laboratory identification number (LABID) and held at 4^{o} C in dedicated refrigerator compartments. A designated storm number was assigned to each event. The number consisted of the last two digits of that year and the Julian Day on which the storm event began. For example, the first storm occurred on September 26, 1985, the 269th day of the year and therefore was assigned the number 85269. No preservation other than refrigeration was used, with the exception of samples prepared for metals analysis. These samples were acidified to pH 4.0 using nitric acid. Laboratory analyses performed on the runoff samples included: ammonium, oxidized nitrogen, total and soluble Kjeldahl nitrogen,

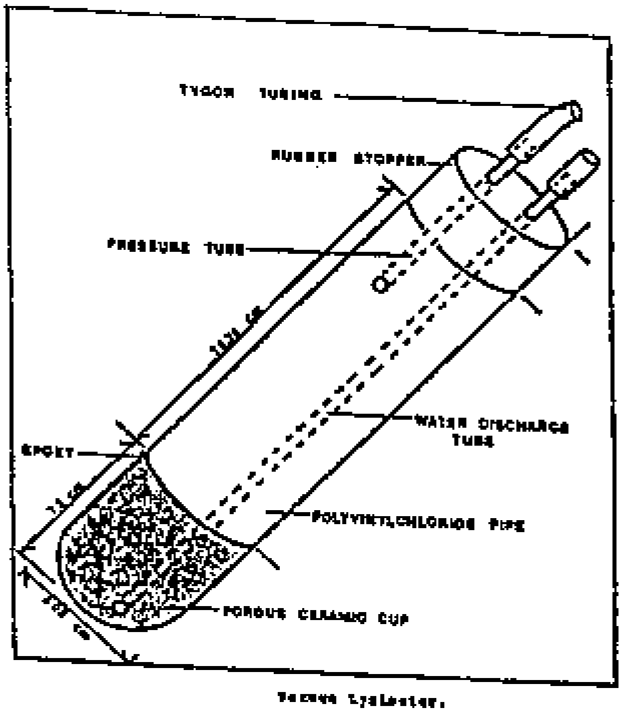


Figure 10. Diagram of typical lysimeter [63]

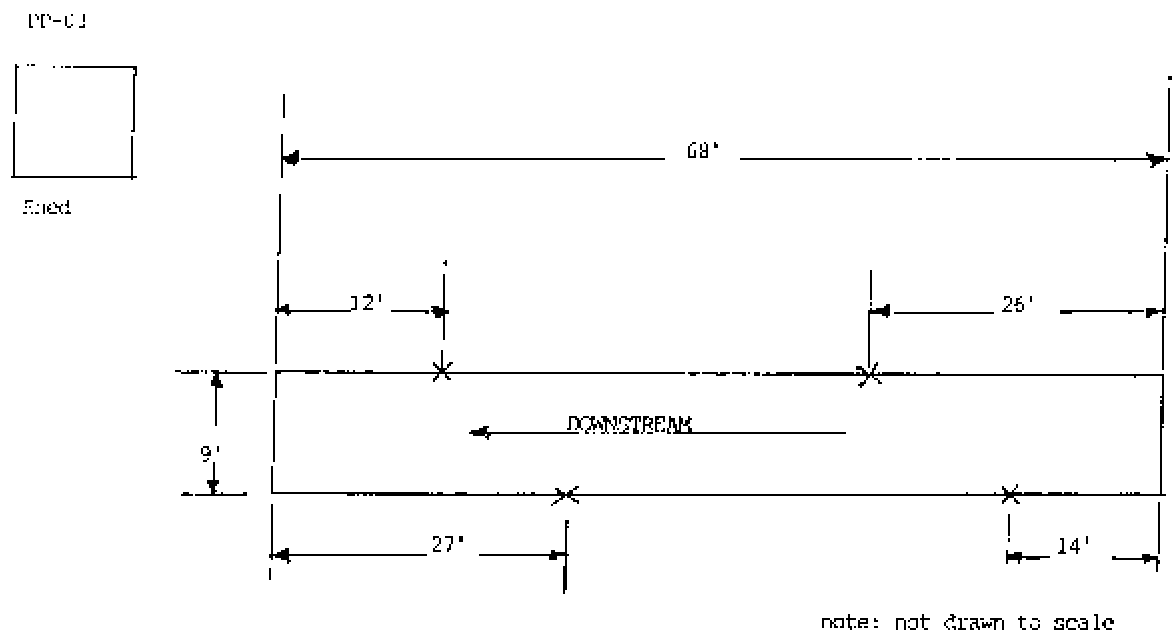


Figure 11. Lysimeter locations at PPO-3

soluble reactive phosphorus, total phosphorus, total suspended solids, and the trace metals lead and zinc.

The volume of each wetfall sample was measured and recorded. The pH of each wetfall sample was measured and recorded. In instances where limited sample was available, the following analytical priority list was employed:

1. total kjeldahl nitrogen (TKN), total phosphorus (TP)
2. Digested metals lead and zinc (Pb, Zn)
3. oxidized nitrogen; nitrates and nitrites (OX-N), orthophosphorus (OP), ammonium nitrogen (NH_4^+N)
4. soluble kjeldahl nitrogen (SKN), total soluble phosphorus (TSP)
5. total suspended solids (TSS)

Dryfall buckets were examined and cleaned of any gross contamination, such as leaves or bird dung. The contents of the buckets were washed out with 250 mL of reagent grade water and transferred to a 250 mL volumetric flask and diluted to capacity.

Because wetfall and dryfall samples were analyzed for metals, the buckets were cleaned with 1 + 1 HCl.

Lysimeter samples were analyzed for:

1. OP, TSP

2. NH_4^+N, SKN, OX-N

Analytical methods utilized are listed and referenced in Table 3.

Table 3

Chemical analyses used in the study.

- o total suspended solids 61,62 (TSS)
- o soluble oxidized Nitrogen N (NO_2_N + NO_3_N) 63 (OX-N)
- o ammonium 63 (NH_4_N)
- o soluble and total Kjeldahl Nitrogen 63 (SKN + TKN)
- o Ortho-phosphate Phosphorus 62 (OP)
- o Total Soluble Phosphorus 63 (TSP)
- o Total Phosphorus 62 (TP)
- o Lead 61 (TPB)
- o Zinc 61 (TZN)

Total suspended solids were determined according to Standard Methods using glass fiber filters.

Nutrient analyses were performed by OWML staff using a Technicon Autoanalyzer II. Orthophosphate, ammonia, and oxidized nitrogen analyses were performed on undigested sample filtrates. Ammonium was quantified using the Berthelot Reaction.

Trace metals zinc and lead were measured by flame atomic absorption spectroscopy, using a Perkin Elmer Atomic Absorption Spectrophotometer.

Quality Assurance.

Acceptability of the precision and accuracy of laboratory methods was governed by a comprehensive quality assurance program. Control charts constructed on the basis of statistical calculations were employed in quality assurance evaluations. These charts indicate whether observed errors were random or systematic in nature and were also used to depict the relative magnitude of errors.

Precision.

Replicate samples were used to predict the precision of a particular analytical procedure. OWML used 60 sets of duplicate samples to determine precision criterion; well above the minimum 15 sets of duplicates recommended by the U.S. EPA. (U.S. EPA, 1978).

After a minimum number of replicate samples has been analyzed, the range of each set of results was calculated. For a replicate set of two samples

$$R = [x_1 - x_2]$$

where: R = absolute difference in results between the replicate sets. "

The mean range (R) can then be calculated by summing the individual ranges and dividing by the number of replicate sets:

$$R = R/n$$

where: n = no. of replicate sets

The upper control limit on the range (UCLR) was then calculated according to the formula:

$$UCLR = D_4R$$

The factor D_4 is dependent upon the size of the replicate group.

By plotting the individual ranges from a replicate measurement on a chart on which the UCLR has been drawn, it was determined whether or not a given procedure is in or out of control. A range above the UCL was considered an indication of an out-of-control procedure. In such an instance the observed error was greater than what would be expected from random sources alone, and the analysis was stopped until the problem was resolved. Questionable data was discarded, and the analyses rerun if possible.

Performance.

A minimum of 15 spiked analyte samples were analyzed during normal lab operations to establish the mean recovery of analytical procedures. Percent recovery for any analytical method was calculated using the formula:

$$P = 100(O - B)/T$$

where: P = percent recovery expressed on mean basis

O = observed value in spiked recovery

B = background value (from unspiked sample)

T = added value (spike)

The mean percent recovery , P', and the standard deviation, SP, were then calculated by:

$$P' = Pi/60$$

$$SP = \{(1/n - 1)[Pi2 - (Pi)2/n]\}1/2$$

where: n = number of results.

The accuracy control limits are then calculated using the following formulas:

$$UCL = P' + 3Sp$$

$$UWL = P' + 2Sp$$

$$LCL = P' - 3Sp$$

$$LWL = P' - 2Sp$$

These values are then used to construct an accuracy control chart. A typical precision control chart is shown in Figure 12.

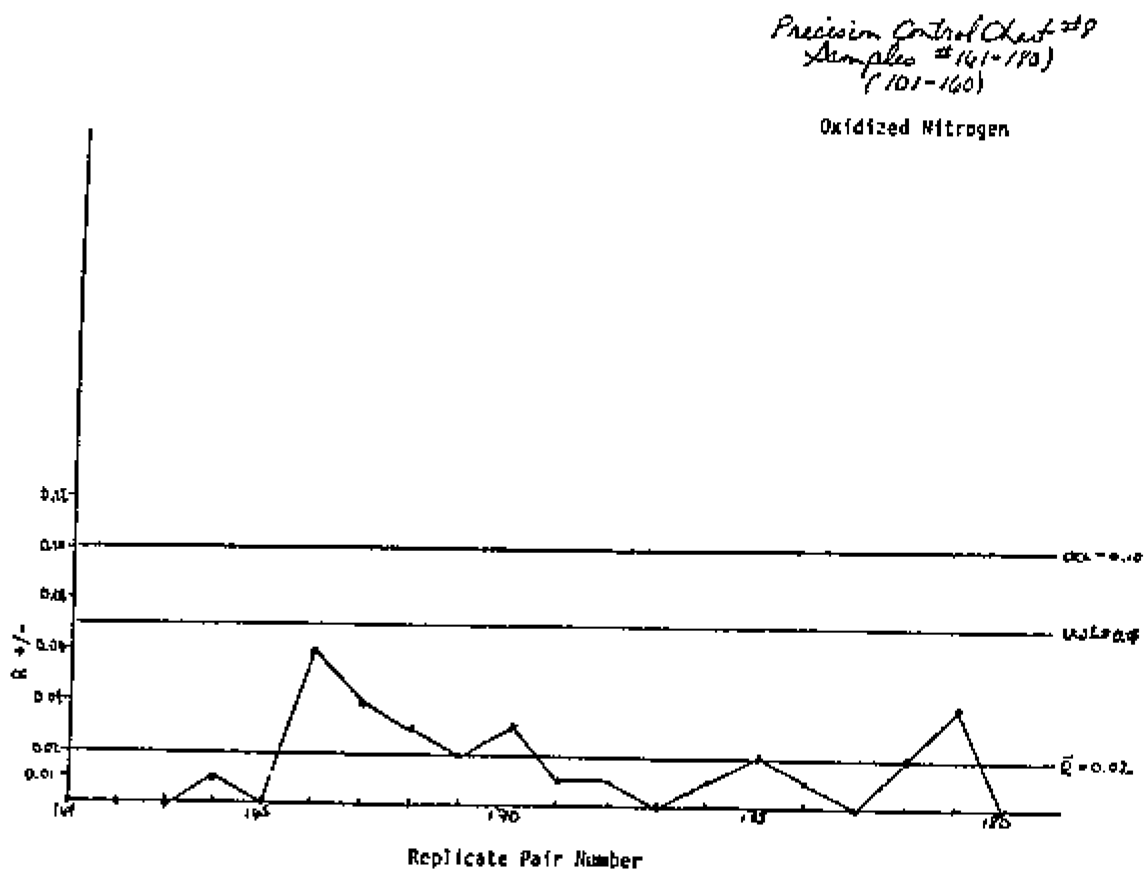


Figure 12. Typical Precision Control Chart

CHAPTER IV

STUDY RESULTS AND DISCUSSION

Storm Event Sampling

A total of 63 storm events occurred over the study period from August 1, 1985 to April 30, 1988. Storm events were defined as precipitation events which produced a minimum runoff volume of 100 cubic feet at PP01 (the control catchment) and could be detected by the flow meter at the site. A listing of the storm events is found on Page 70.

For synoptic storm events, minimum flow meter settings were taken to be 100 cubic feet, 10 cf, and 10 cf for the control (PP01), porous pavement (PP02), and infiltration trench (PP03), respectively. The temporal distribution of storm events was 7 storms in 1985, 29 storms in 1986, 19 in 1987, and 8 in 1988. Monitoring was temporarily halted during the period of June 30, 1987 to September 8, 1987 in order to reconstruct the infiltration trench which had become filled with extraneous solid matter during the previous 2 years of monitoring.

The infiltration trench was 100 percent efficient in reducing runoff for 17 storm events. A runoff reduction efficiency of 100 percent occurred at the porous pavement site for 4 storm events. In general, an average of 1.17" of rainfall was required to initiate sampling at all three stations for an individual storm event. This is a further indication of the role of infiltration in BMP efficiency.

Several equipment malfunctions occurred prior to the trench repairs which were performed June 30 to September 8, 1987. Following these modifications there were few malfunctions. The modifications included the installation of separate data loggers for each station in an effort to reduce signal noise and logging of erroneous signals which were being recorded. It was thought that the problem was rooted in the separation distances between the 3 stations. Over the entire monitoring period, flow-meters accounted for malfunctions during 3 storm events, samplers malfunctioned during 1 event, and data loggers malfunctioned during 2 events.

For this study, the entire database was analyzed in two ways in order to analyze the pollutant removal effectiveness of the BMP's:

1. Synoptic Storms: This included 57 events in which the BMP pair of PP01 and PP02 were synoptically sampled and 39 events in which the BMP pair of PP01 and PP03 were synoptically sampled. Synoptic events were selected where hydrologic and chemical analyses were complete. The number of events thus sampled exceeded the requirements of the work plan by over 50 percent.

2. Entire Data Base: The entire population (synoptic and nonsynoptic) of runoff events (i.e. including events in which one or both of the BMP's were 100 percent effective in reducing runoff) was used in order to incorporate the effect of the volume control aspect of the BMP's and to make a comparison with the synoptic

set. Runoff events which involved an equipment malfunction were not included in the analysis.

Successfully sampled and analyzed synoptic storms at PP01 and PP02 were:

85269	85275	85294	85307	85326	85331
86025	86037	86049	86054	86073	86095
86105	86111	86133	86140	86157	86175
86179	86190	86197	86214	86229	86286
86305	86309	86315	86324	86343	86352
87018	87031	87074	87089	87093	87123
87139	87177	87251	87254	87255	87262
87273	87276	87279	87300	87314	87331
87344	88017	88019	88033	88034	88046
88050	88086	88097			

Successfully sampled and analyzed synoptic storms at PP01 and PP03 were:

85269	85275	85294	85307	85331	86025
86034	86037	86049	86054	86073	86095
86105	86111	86133	86157	86175	86179
86183	86190	86214	86286	86305	86309
86315	86324	86343	87031	87139	87262
87344	88017	88019	88033	88034	88046
88050	88064	88097			

Wetfall and Dryfall Contribution

Wet and dryfall accumulation rates were computed using known relationships between the surface area of the collection bucket, duration of the collection interval, and concentration (wetfall) or mass (dryfall) of the captured constituent. The adopted procedure required that concentration of dryfall be calculated by recovering bucket contents with 250 ml of reagent grade water (U.S. EPA, 1979).

Wetfall accumulation rates were expressed as mg/m2/inch rainfall and dryfall accumulation rates were expressed as mg/m2/day. Dryfall duration was calculated as the number of days between precipitation events.

Accumulation rates were computed as follows:

$$Ra = M/A * t$$

where,

R_a = accumulation rate, $M/L^2 - t$

M = mass in collector, M

A = collector area, L^2

t = exposure time, t

Calculated median values of accumulation rates for the entire data set are shown in Table 4 and in Figures 13 and 14. In wetfall, ortho-phosphate phosphorus and total phosphorus rates were calculated to be 0.27 mg/m^2/in while total soluble phosphorus exhibited a slightly lower accumulation rate of 0.26 mg/m^2/inch precipitation. Rates of nitrogen accumulation in wetfall ranged from 4.71 mg/m^2/in. ppt. for NH_4_N to 10.23 mg/m^2/in. ppt. for SKN.

Table 4
Wetfall and Dryfall Median Deposition Rates

	OP	TSP	TP	NH_4^+	SKN	TKN	OX-N	TSS	E-PB	Ezn
Wetfall Mg/m^2/cm rain	0.107	0.104	0.109	1.85	4.03	3.38	2.95	11.07	0.101	0.101
Wetfall Mg/m^2/day	0.031	0.030	0.032	0.542	1.18	0.99	0.863	3.232	0.030	0.030
Dryfall Mg/m^2/day	0.008	0.013	0.023	0.079	0.199	0.243	0.121	NA	0.009	0.023

In dryfall, Total Kjeldahl Nitrogen (TKN) exhibited the highest accumulation rate of the nitrogen forms at 0.243 mg/m^2/day. Soluble Kjeldahl nitrogen and oxidized nitrogen exhibited similar accumulation rates of 0.199 mg/m^2/day and 0.121 mg/m^2/day respectively. NH4_N accumulated at a lesser rate of 0.079 mg/m^2/day.

In Table 4, wetfall was also expressed as mg/m^2/day, assuming an average yearly rainfall of 42 inches (Huber et al, 1979). This was done in order to express deposition rates in similar units.

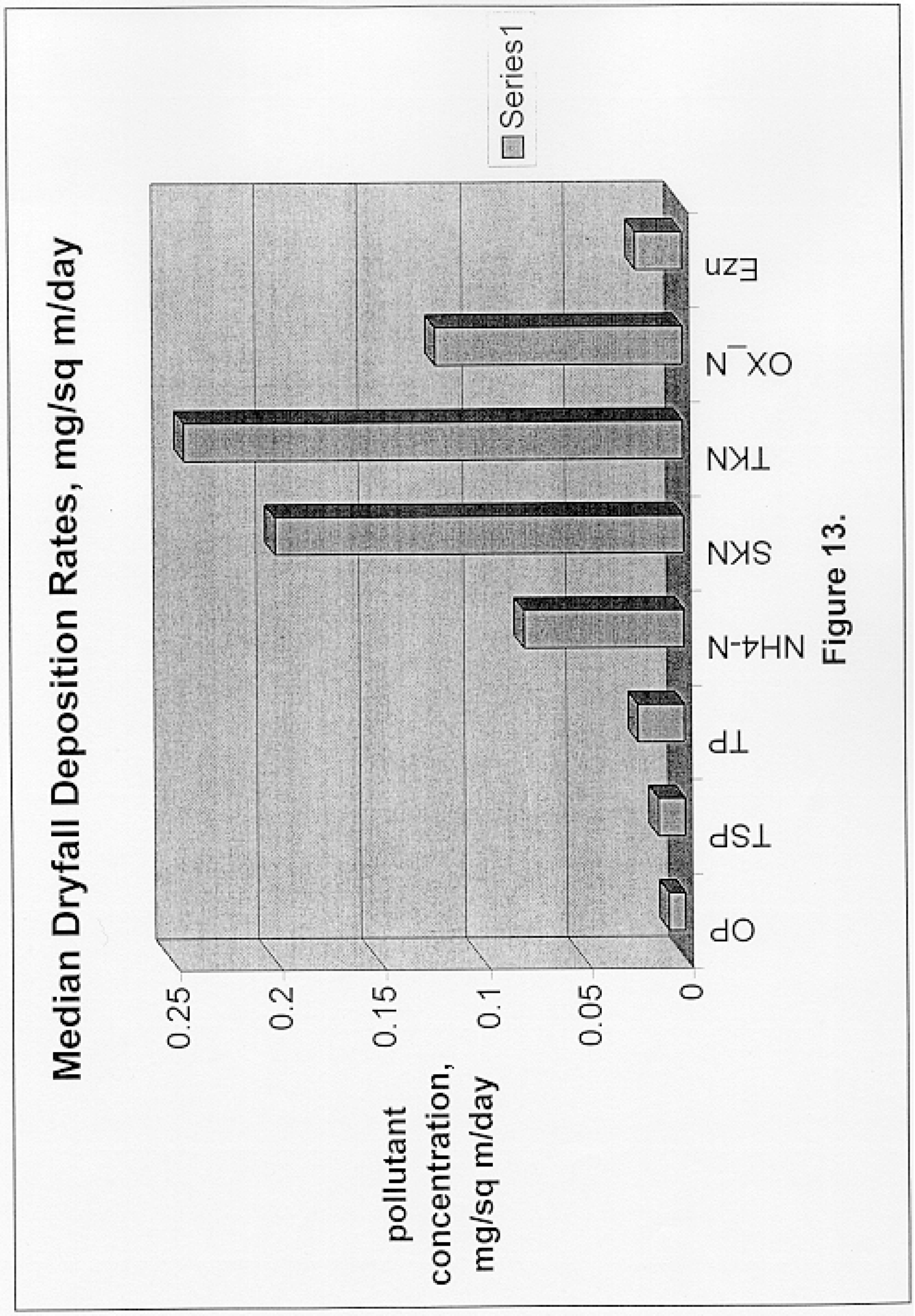

Median Dryfall Deposition Rates, mg/sq m/day
pollutant concentration, mg/sq m/day
0.25
0.2
0.15
0.1
0.05
0
OP
TSP
TP
NH4-N
SKN
TKN
OX_N
Ezn
Series1

Figure 13.

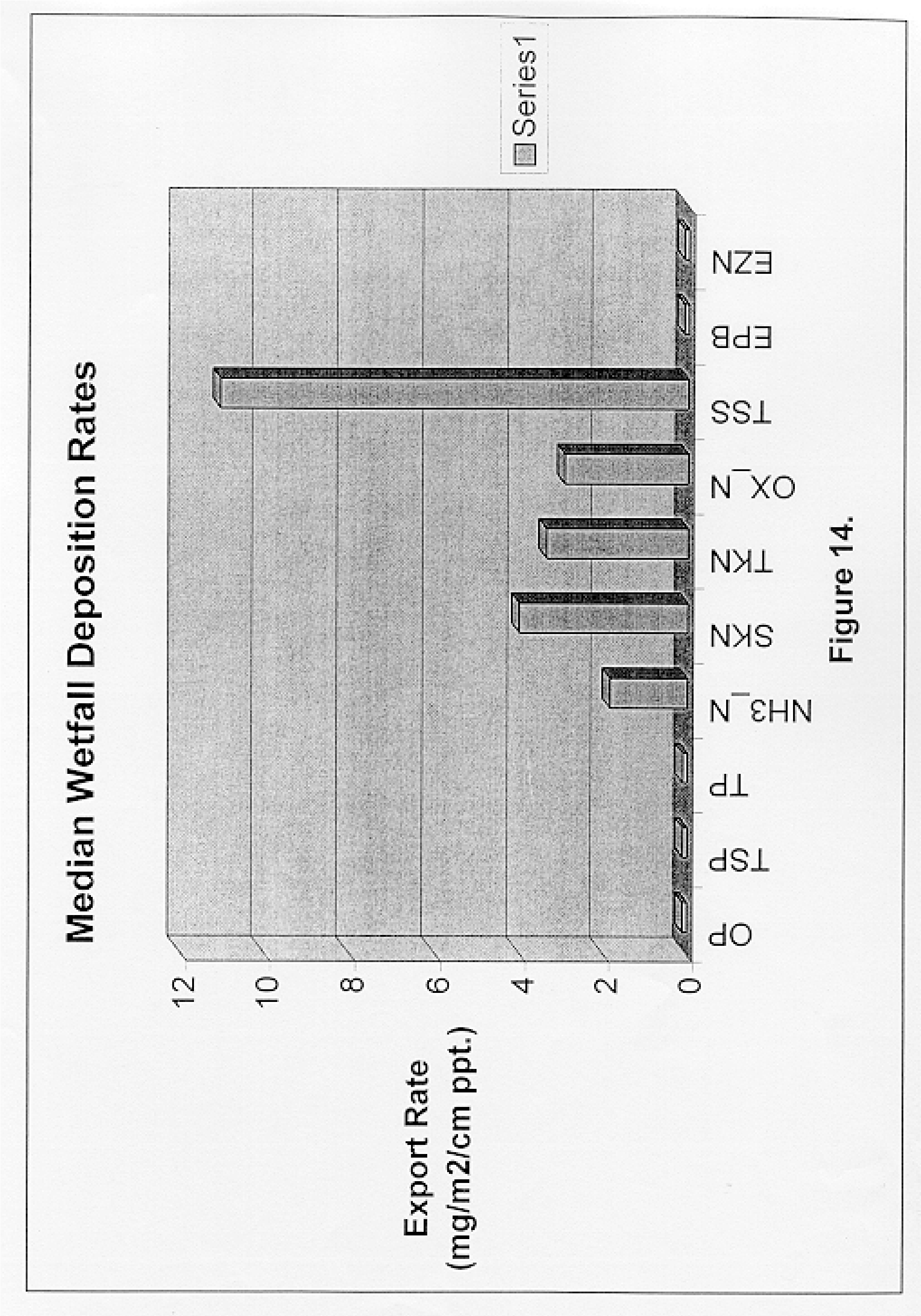

Median Wetfall Deposition Rates
Export Rate (mg/m2/cm ppt.)
12
10
8
6
4
2
0
OP
TSP
TP
NH3_N
SKN
TKN
OX_N
TSS
EPB
EZN
Series1

Figure 14.

With regard to dryfall, median accumulation rates of all nutrients are a great deal less than those calculated for the Metropolitan Washington Council of Governments National Urban Runoff Program study (MWCOG, 1983). Rates of accumulation in dryfall of NH_4_N, OP, and TP are an order of magnitude less than those rates calculated for the MWCOG NURP study. This may be due to the lower density of development in the area surrounding the Davis-Ford site.

Annual dryfall projections were computed by converting the entire dryfall database to lbs/acre/day and finding the median value to be multiplied by 365. These data, expressed as lbs/acre/year, are shown in Table 5.

Table 5
Wetfall and Dryfall Median Yearly Projections (lbs/acre/year)

	OP	TSP	TP	NH_4^+	SKN	TKN	OX_N	TSS	E-Pb	E-Zn
Wetfall	0.101	0.098	0.104	1.77	3.84	3.22	2.81	10.52	0.098	0.098
Dryfall	0.026	0.042	0.075	0.257	0.648	0.791	0.394	NA	0.029	0.075
Total	0.127	0.143	0.179	2.027	4.49	4.011	3.204	10.52	0.127	0.173
COG-total	0.35	NA	0.84	1.02	NA	NA	NA	243.3	NA	NA

In these calculations wetfall periods were assumed to be small enough to be discounted. Because of the inability to measure the period of time the dryfall collector was closed (e.g. during rain events), the accumulation rates have been calculated assuming exposure during the entire time period. Therefore it may be seen that most of a given period of time consists of dry conditions with relatively short periods of intervening wetfall. Yearly unit area projections of atmospheric contributions may be estimated assuming an average yearly precipitation of 42 inches. These yearly projections are shown in Table 5.

These yearly projections of atmospheric fallout are less than the yearly average rural projections calculated by the Water Resources Planning Board and MWCOG (MWCOG, 1983).

Based on median loadings of pollutants in runoff at the control (PP01), the percentage contribution by wetfall and dryfall sources was calculated. Median concentrations were converted to pounds per acre for both precipitation and antecedent dry periods. Table 6 illustrates these percentages. The highest loadings of airborne pollutants and atmospheric fallout were of oxidized nitrogen and total Kjeldahl nitrogen for both wetfall and dryfall phases. The value for total suspended solids was higher but was only analyzed in the wetfall phase, because of the low availability of solid material in the dryfall collectors. Wetfall loadings were consistently higher than dryfall loadings. This is in agreement with results of past studies (MWCOG, 1983).position it is
expected to be very near 100 percent. The entire data set of the storm events at PP01 were analyzed for this aspect of the study. These were chosen on the basis of having complete dryfall data prior to the storm event and wetfall data during the storm event. Wetfall and dryfall loadings were calculated (lbs/acre) for the entire data base and

expressed as a percentage of the runoff loadings (lbs/acre) at PP01. Table 6 displays these percentages.

Table 6
Wetfall/Dryfall percentage of all PP01 Runoff Events

	OP	TSP	TP	NH_4^+	SKN	TKN	OX-N	TSS	Epb	Ezn
Wetfall	86	72	50	131	111	105	98	43	NA	67
Dryfall	33	34	39	25	29	36	17	Na	NA	39
Total	119	106	89	156	140	141	115	Na	NA	106

Dryfall plus wetfall loadings of TSP, TP, and EZn were very nearly equal to 100 percent of the control loading and yet atmospheric OP, NH_4^+, SKN, and, TKN were much more than 100 percent of the storm loading at PP01. The total percentages are also shown in Table 6. The high total percentages can be attributed to the lack of complete washout of pollutants from the control catchment.

Acid rain, which is defined as rain with a pH less than 5.6, is a result of sulfuric and nitrate emissions from urban, industrial, and electric utility burning operations that use sulfur- and nitrogen- containing fuels.

In comparison to the pH of distilled water, which is around 5.6, the median lab pH value of precipitation for the entire data set was 3.72. The weighted mean pH for the entire data set was computed to be 3.57. Past studies have shown that only the total suspended solids concentration in runoff is significantly affected by the reduced pH of rainwater (Angelotti, 1985). Among the other implications of acid precipitation, undesireable oligotrophication (severe loss of productivity by the low pH conditions) and fish kills are the most dangerous and visible consequences of acidification.

Seasonal Aspects of Runoff Pollution.

All wetfall and dryfall data for the calendar year of 1986 were separated seasonally in an attempt to recognize a pattern between season of the year and amount of pollutants from atmospheric sources. For ease of separation and calculation, seasons were separated as follows:

June-August; summer

September-November; fall

December-February; winter

March-May; spring

Results of median dryfall loadings are shown in Table 7 and Figure 15. Dryfall loading rates are expressed as mg/m2/day. As may be seen in the table, dryfall loadings of all pollutants of interest except zinc (Zn), orthophosphorus (OP), and total soluble phosphorus (TSP) were heaviest during the spring season. Zn. OP, and TSP were most heavily deposited in summer months. These results of spring being the month

Seasonal Dryfall Contribution Expressed as mg/sq m/day

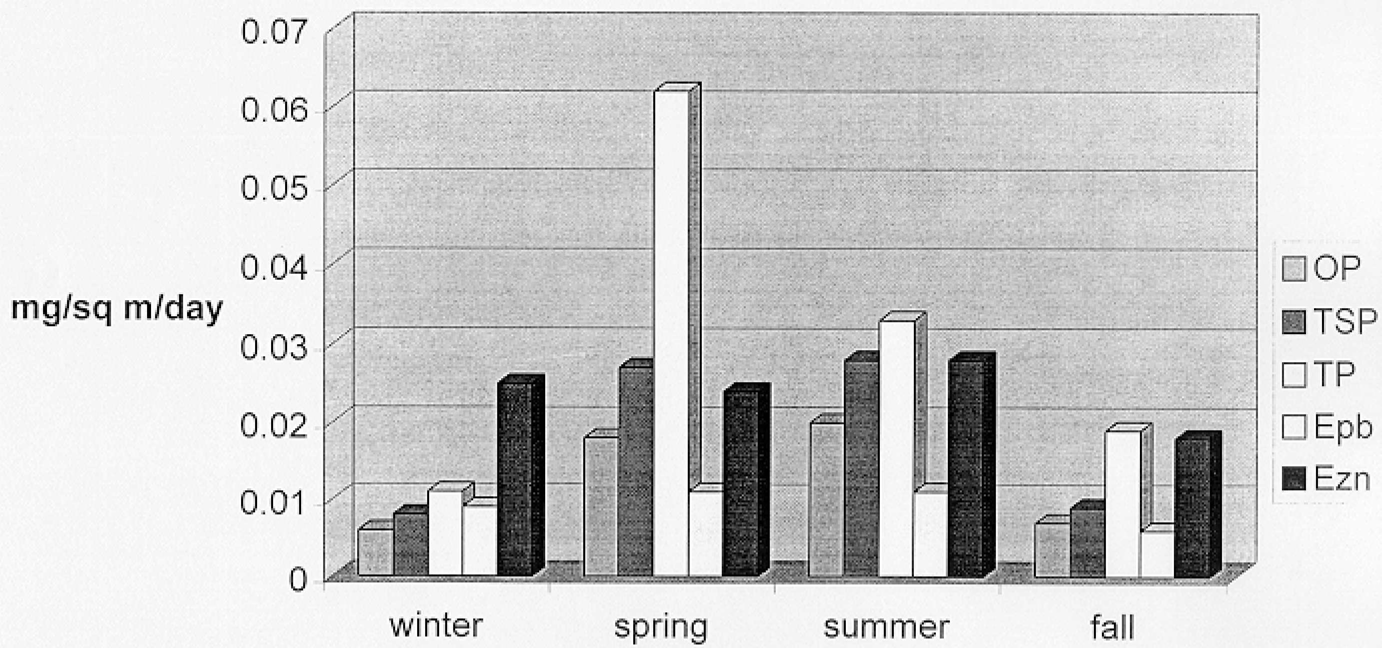


Figure 15

with the highest loadings is in agreement with past studies on seasonal variation of dryfall loadings.

Table 7
Seasonal Dryfall Contribution
(mg/m^2/day)

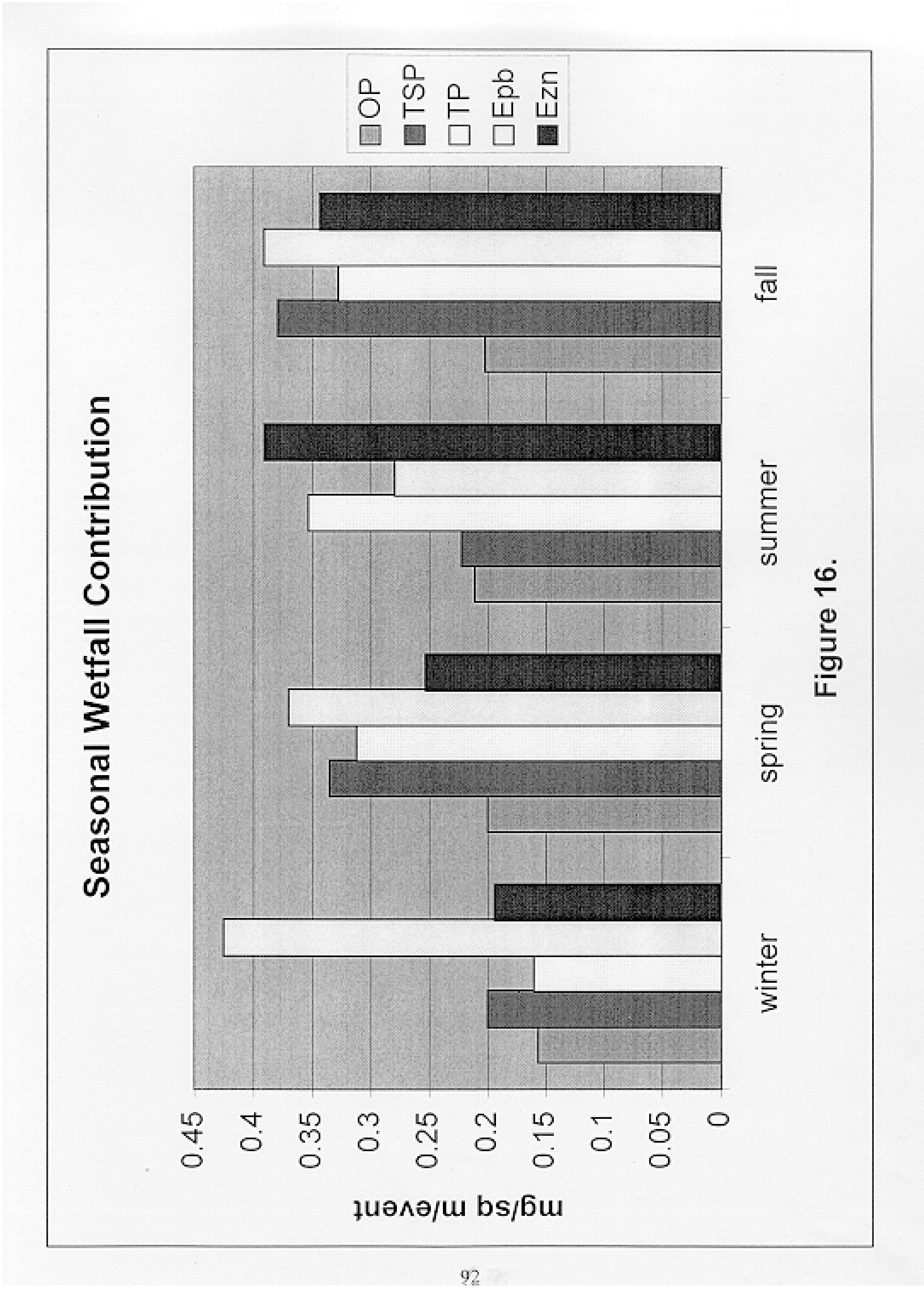
	OP	TSP	TP	NH_4^+	SKN	TKN	OX-N	TN	EZn	Ezn
Winter	0.006	0.008	0.011	0.073	0.147	0.164	0.136	0.373	0.009	0.025
Spring	0.018	0.027	0.062	0.079	0.323	0.568	0.171	0.818	0.011	0.024
Summer	0.020	0.028	0.033	0.047	0.179	0.201	0.046	0.294	0.011	0.028
Fall	0.007	0.009	0.019	0.120	0.183	0.253	0.104	0.477	0.006	0.018

Seasonal wetfall loadings were less predictable. For many of the precipitation events a rainfall total was not recorded, therefore it was necessary to express wetfall loading as mg/m^2/event as opposed to mg/m^2/inch rain. Wetfall loadings of TSP, NH_4^+, SKN, TKN , OX-N, and EPb were heaviest during the spring months. Wetfall samples produced the heaviest loadings of OP and EZn during the summer months. The heaviest TSP loadings occurred during the fall of the year. These values are displayed in Table 8 and Figure 16.

TABLE 8

Seasonal Wetfall Contribution (mg/m^2/event)

Season	OP	TSP	TP	NH_4^+	SKN	TKN	OX-N	TN	EPb	EZn
Winter	0.156	0.200	0.160	3.041	3.759	4.234	6.407	13.68	0.425	0.194
Spring	0.200	0.335	0.312	5.752	10.09	9.355	8.233	23.34	0.370	0.253
Summer	0.211	0.222	0.353	5.335	6.024	7.997	5.909	19.24	0.279	.391
Fall	0.202	0.379	0.327	3.596	8.161	6.871	4.313	14.78	0.363	0.343



Seasonal Wetfall Contribution
mg/sq m/event
0.45
0.4
0.35
0.3
0.25
0.2
0.15
0.1
0.05
0
winter
spring
summer
fall
OP
TSP
TP
Epb
Ezn

Figure 16.

Table 9 displays seasonal wetfall contribution based on seasonal distribution of rainfall. All wetfall data in which rainfall totals were complete were used in this analysis. Average figures for precipitation, distributed seasonally as found by Dulles International Airport weather data for the years of 1986 and 1987 follow:

Winter: 7.95" Spring: 12.19"

Summer: 10.39" Fall : 10.08"

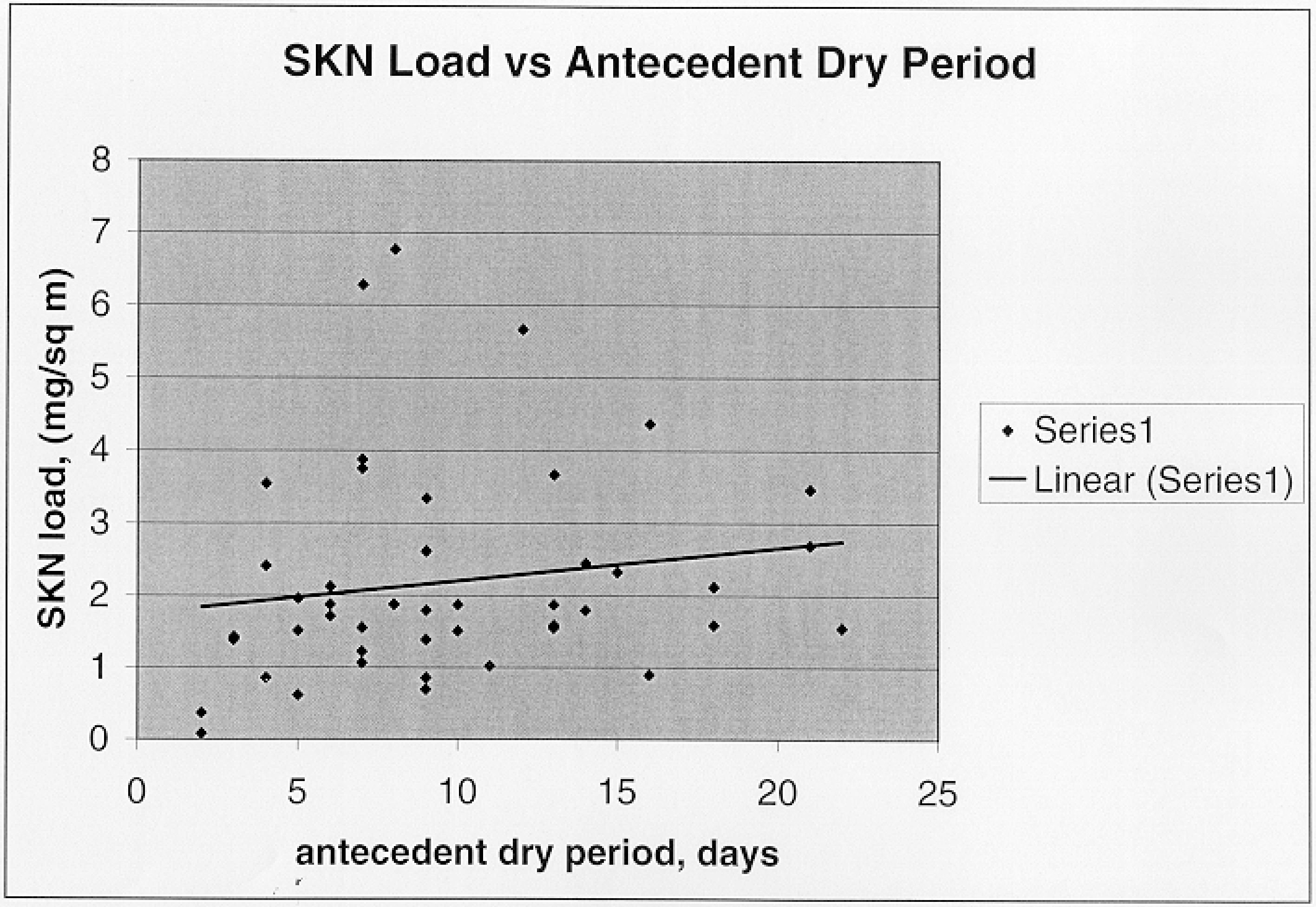
Table 9.
Seasonal Wetfall Contribution
(mg/m^2/season)

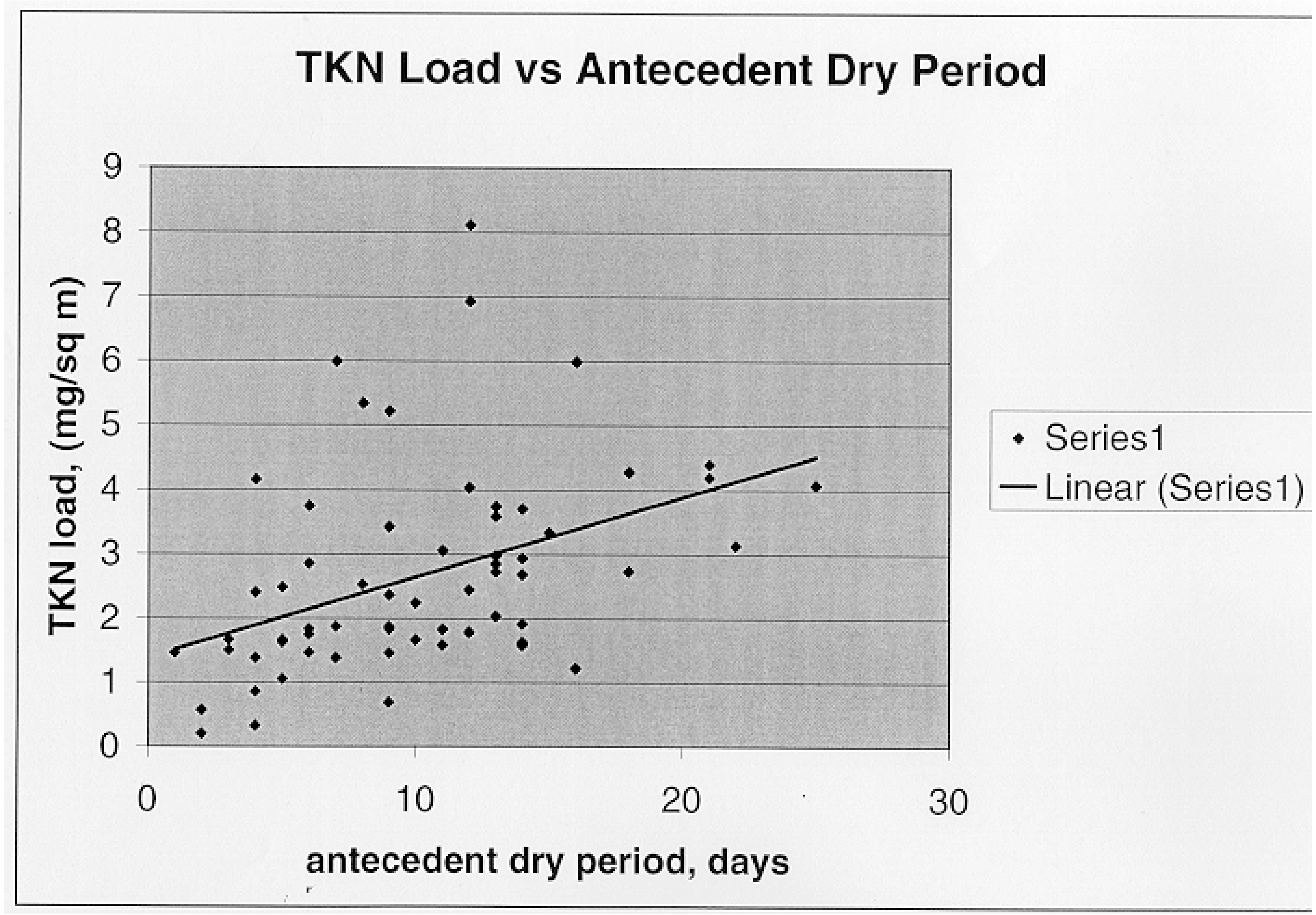
	OP	TSP	TP	NH_4^+	SKN	TKN	OX-N	TN	EPb	EZn
Winter	2.15	2.09	2.21	37.44	81.32	68.24	59.63	165.3	2.04	2.04
Spring	3.41	3.21	4.39	54.49	94.11	119.5	111.7	285.7	6.10	6.83
Summer	2.91	2.73	2.89	48.94	106.3	89.15	71.93	210.0	2.67	2.67
Fall	2.73	2.65	2.80	47.48	103.1	86.49	75.60	209.6	5.04	5.64

Antecedent dry period.

Past studies have shown that pollutants will increase in accumulation with increasing antecedent dry period, but only to a certain point. At this point they no longer accumulate and they reach a maximum level.

The entire dryfall data set was used to discern a correlation between antecedent dry period and pollutant loading. The parameters used for this part of the study were


SKN Load vs Antecedent Dry Period
SKN load, (mg/sq m)
0
1
2
3
4
5
6
7
8
0
5
10
15
20
25
antecedent dry period, days
Series1
Linear (Series1)



TKN Load vs Antecedent Dry Period
TKN load, (mg/sq m)
0
1
2
3
4
5
6
7
8
9
0
10
20
30
antecedent dry period, days
Series1
Linear (Series1)

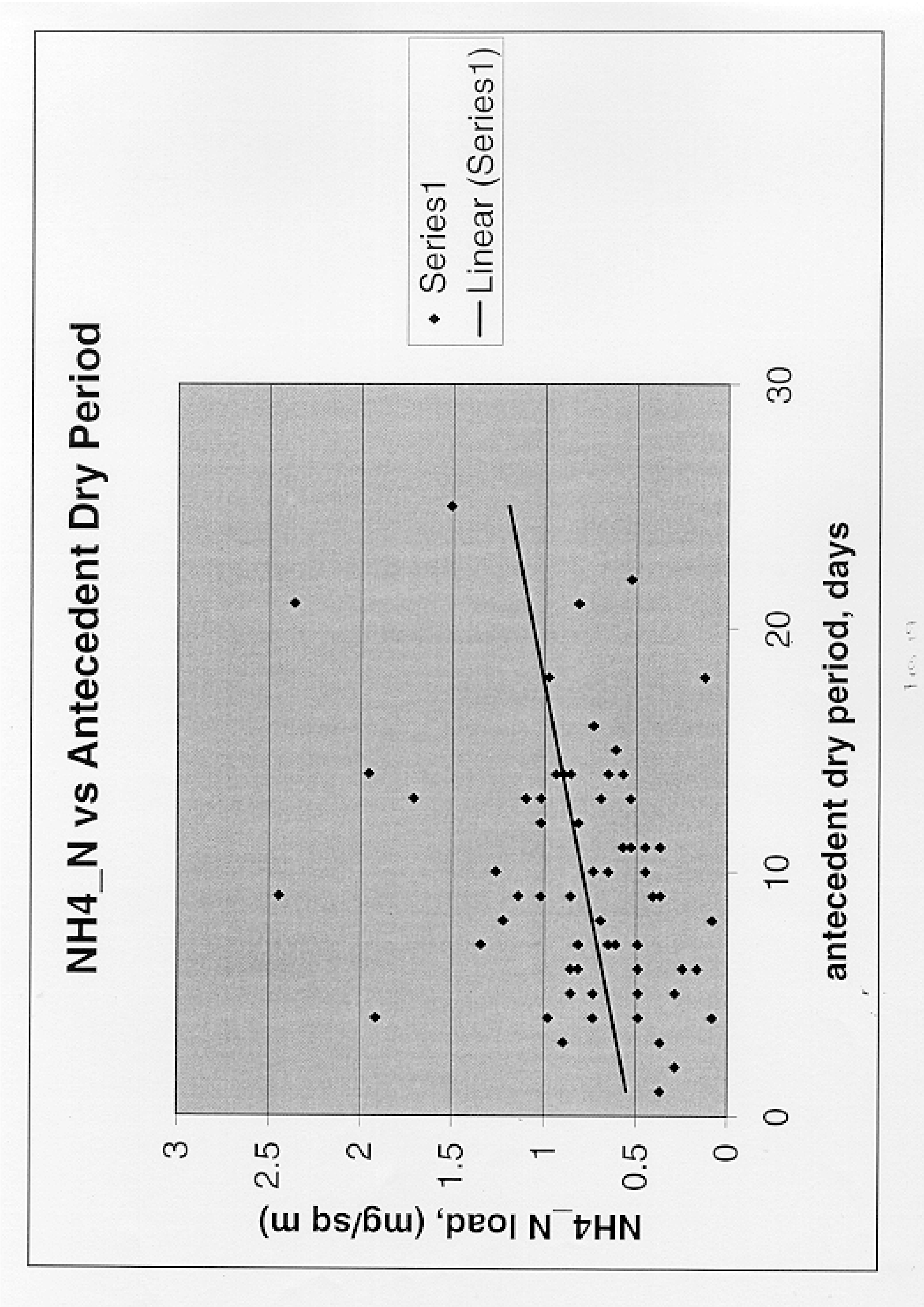
soluble Kjeldahl nitrogen and total Kjeldahl nitrogen. These parameters were used because they appeared to vary most distinctly with antecedent dry period in days. These plots are seen in figures 17 and 18. As may be seen, SKN load reaches a maximum of 0.2 mg/m^2 to 0.3 mg/m^2 after an antecedent dry period of 7 or more days. TKN load reached a maximum of 0.1 mg/m^2 to 0.2 mg/m^2 during an antecedent dry period of 8 days or more. Ammonia-N and total phosphorus accumulations versus antecedent dry period are illustrated in Figures 19 and 20.

Total Rain and Flow Balances.

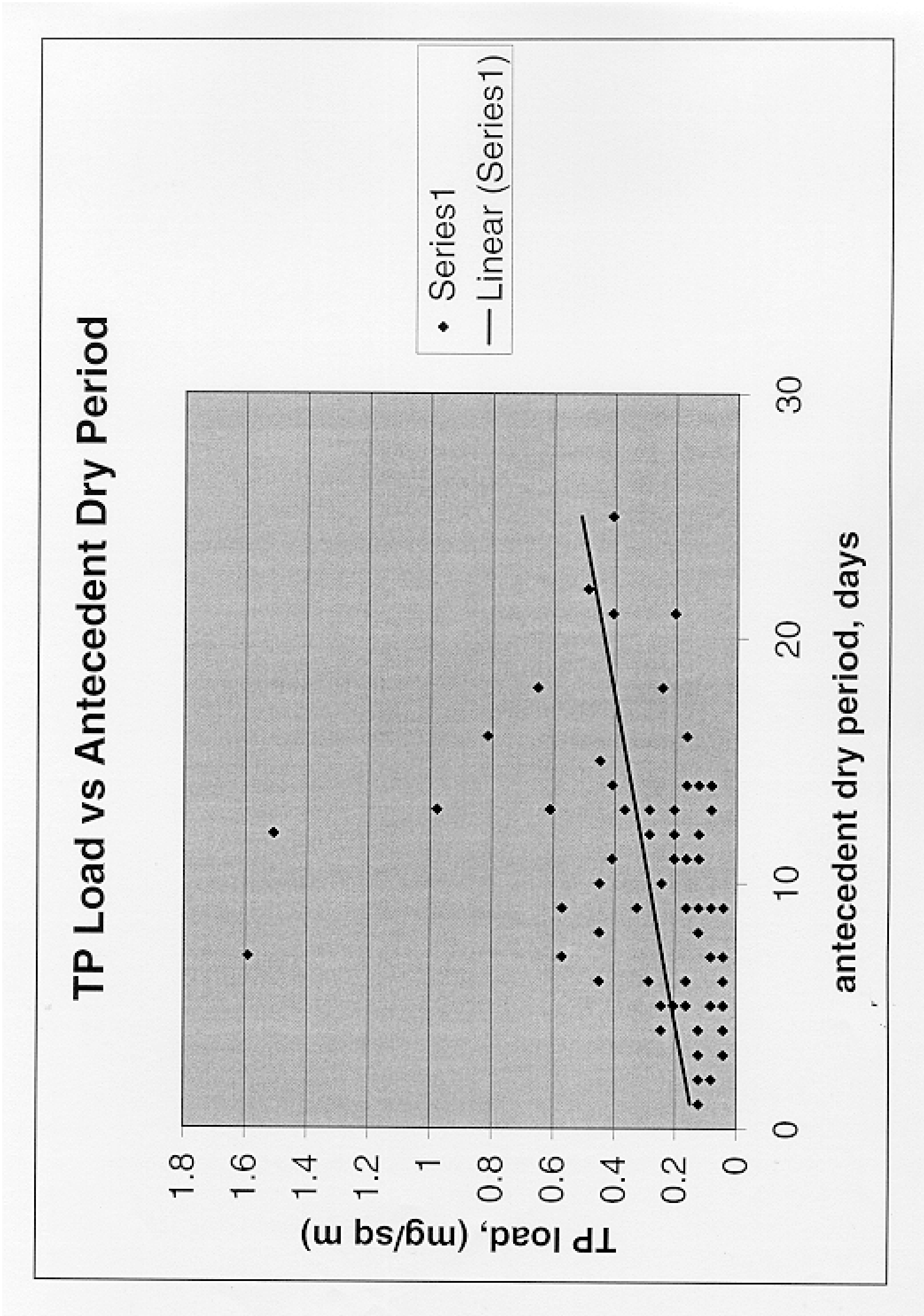
Although BMP performance was based on 1) representative synoptic storms in which all data were complete and 2) the entire data set, hydrologic performance was based on the entire database. Flow and rainfall monitoring equipment were operated continuously, and hydrologic data were successfully collected for 51 runoff events.
In this analysis storms which were complete in rainfall and hydrologic data were used as well as storms in which the BMP's were 100 percent efficient in reducing runoff. The average rainfall for these 51 storms was 0.86 inches. Figure 21 shows the relationship between total rainfall and runoff for the control catchment (PP01) in storms for which the flow and rainfall data were collected. In the figure, runoff is expressed in inches and is calculated by:

$$\text{Runoff, inches} = \frac{\text{total rainfall (in}^3\text{)}}{\text{catchment area (in.}^2\text{)}}$$

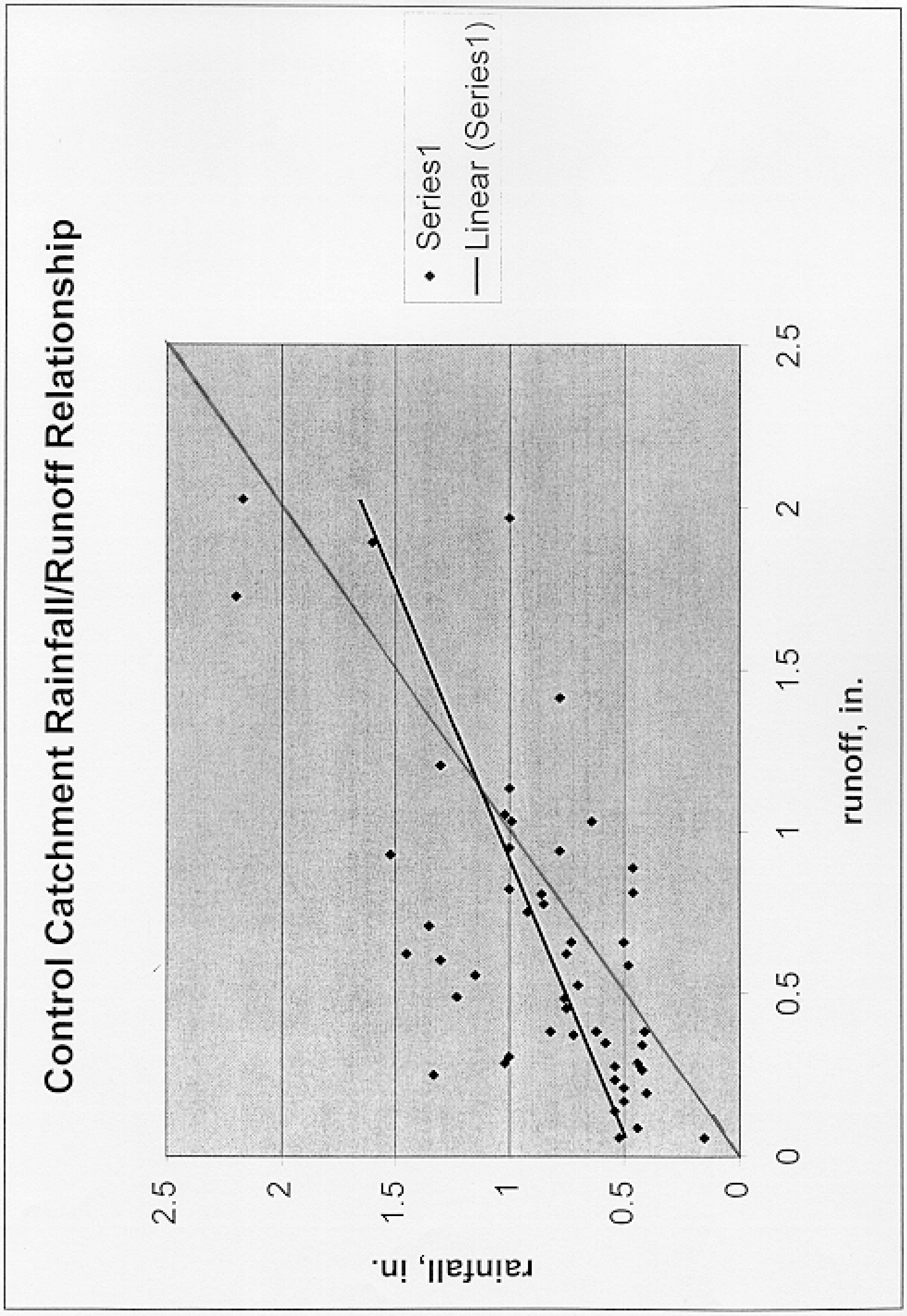
The figure illustrates that the conventional pavement efficiently transmitted nearly all of



NH4_N vs Antecedent Dry Period
NH4_N load, (mg/sq m)
3
2.5
2
1.5
1
0.5
0
0
10
20
30
antecedent dry period, days
Series1
Linear (Series1)



TP Load vs Antecedent Dry Period
TP load, (mg/sq m)
1.8
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
0
10
20
30
antecedent dry period, days
Series1
Linear (Series1)

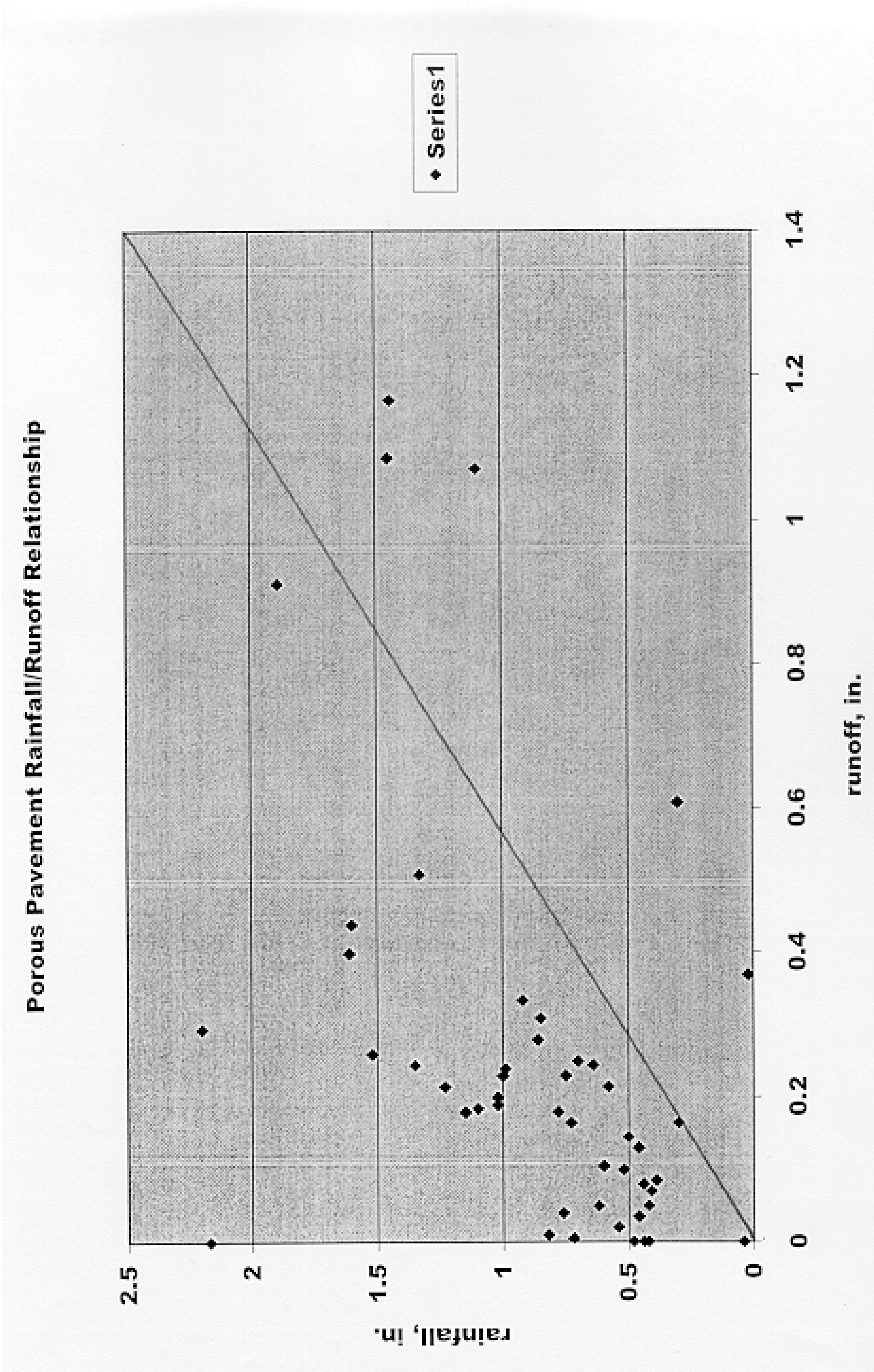


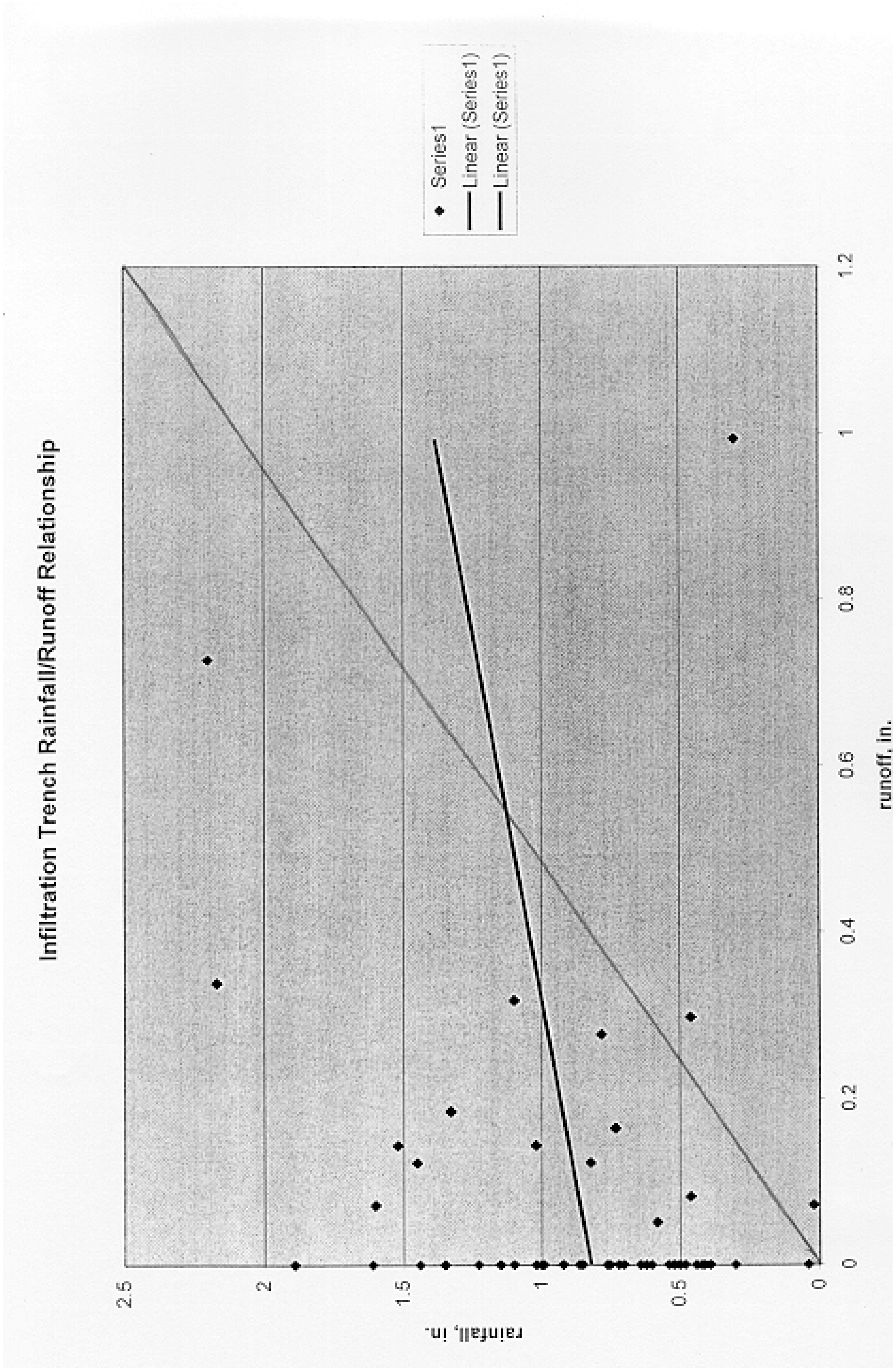
Control Catchment Rainfall/Runoff Relationship
rainfall, in.
2.5
2
1.5
1
0.5
0
runoff, in.
0
0.5
1
1.5
2
2.5
Series1
Linear (Series1)

the incident rainfall offsite. The hydraulic efficiency of the catchment area may be calculated by the slope of the line which is described by the linear regression of runoff as a function of rainfall. The slope at the control catchment was 0.83, indicating an efficiency of 83 percent. The remainder may be assumed to be lost by evaporation and to some slight storage.

Because infiltration trenches and porous pavement are volume control BMP's, it was expected that a large fraction of stormwater runoff would be lost to infiltration to the soil. Figures 22 and 23 show this relationship between runoff and rainfall at the BMP sites. From the slope of the line in Figure 22 it may be seen that the total flow at the porous pavement catchment was 21 percent that of the control. This is in agreement with the 54 percent total flow reduction, found later in this report, for the entire data set of porous pavement flows in comparison to the control catchment flows.

The relationship between total flow volume and rainfall for the infiltration trench site is shown in Figure 23. The slope of the line indicates that total runoff was 25 percent of that for the control catchment. For the entire data set total flows at the infiltration trench were 7 percent control site total flows. This retention is primarily a factor of the storage volume in relation to the surface drainage area. The variability may also be attributed to the variation in soil moisture content at the trench between storm events. In contrast, porous pavement subsoil, being under the pavement and gravel layer would maintain a more stable level of soil moisture retention than the infiltration trench.


Porous Pavement Rainfall/Runoff Relationship
rainfall, in.
2.5
2
1.5
1
0.5
0
runoff, in.
0
0.2
0.4
0.6
0.8
1
1.2
1.4
Series1


Infiltration Trench Rainfall/Runoff Relationship
rainfall, in.
2.5
2
1.5
1
0.5
0
runoff, in.
0
0.2
0.4
0.6
0.8
1
1.2
Series1
Linear (Series1)
Linear (Series1)

Peak and Total Flow Reductions.

An analysis was conducted of the peak and total flow information in the hydrologic data set. Peak flows were significantly reduced by both BMP's in comparison to the control, as may be seen in Figures 24 and 25.

The median peak flow was reduced 68 percent by the porous pavement BMP and 83 percent by the infiltration trench BMP. These peak flow reductions would result in reductions of the shock load effect described earlier in this report. For the entire data set, the median time to peak was retarded 5 minutes at the infiltration trench site and 6 minutes at the porous pavement BMP.

The median total flow of 0.51 in. runoff /in rainfall at the control was reduced to 0.19 in. runoff/in rainfall at the porous pavement site and 0.00 in. runoff/in rainfall at the site of the infiltration trench. These are reductions in total flow of 63 percent for porous pavement and 100 percent for the infiltration trench. This 63 - 100 percent flow reduction would indicate that the two BMP's would also be expected to achieve at least a 63 percent - 100 percent removal efficiency for the soluble pollutant loads in the stormwater. Total runoff was 100 percent reduced at the infiltration trench for 17 storms and for 4 storms at the porous pavement site. Total flow reduction is illustrated in Figure 26.

The two BMP's had a significant effect on storage time of stormwater. Storage time

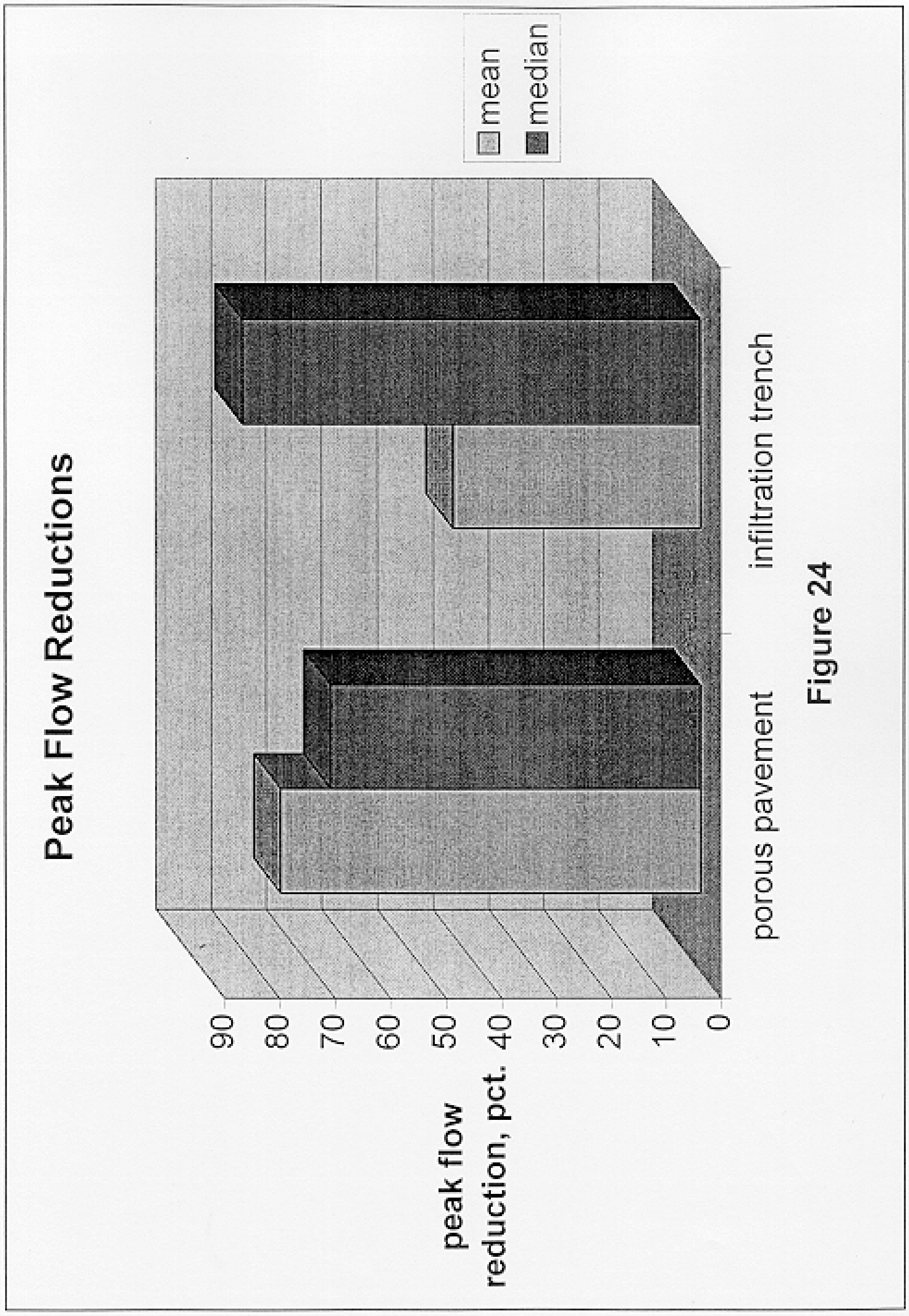

Peak Flow Reductions
peak flow reduction, pct.
90
80
70
60
50
40
30
20
10
0
porous pavement
infiltration trench
mean
median

Figure 24

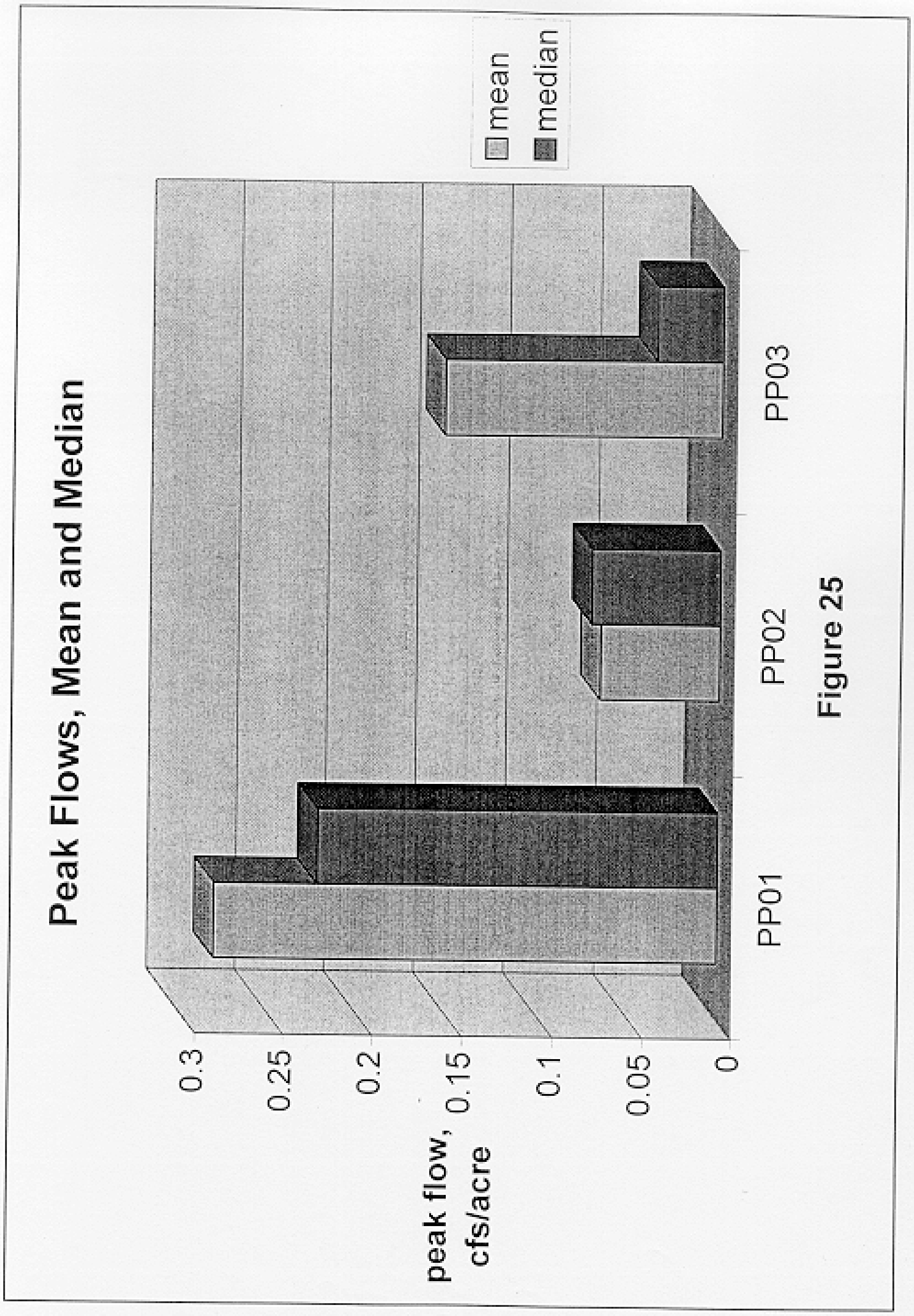

Peak Flows, Mean and Median
peak flow, cfs/acre
0.3
0.25
0.2
0.15
0.1
0.05
0
PP01
PP02
PP03
mean
median

Figure 25

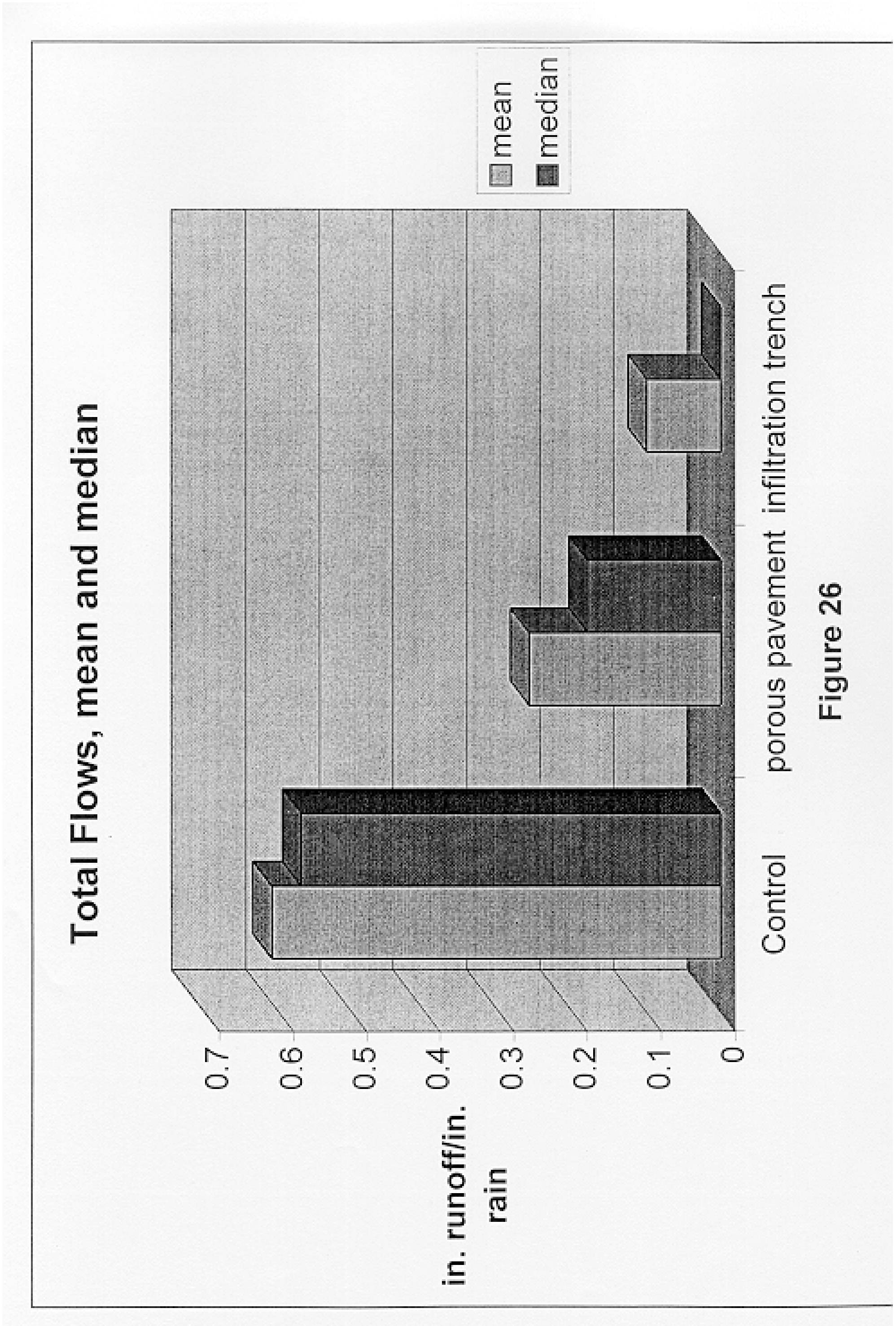

Total Flows, mean and median
in. runoff/in. rain
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
Control
porous pavement
infiltration trench
mean
median

Figure 26

may be defined as the time difference between the beginning of flow at PP01 (control) and the beginning of flow from the two BMP sites. A median value of storage time was calculated as 4.66 hours for porous pavement and the median value of storage time for the infiltration trench was 5.17 hours.

Due to the sensitivity of the data logger in receiving a signal (voltage) from the flow meters, for many storms the preflow voltage was recorded on the cassette tape not as a 0, but as 0.01. This made it difficult to determine the time that flow actually began. In these cases a beginning flow time was decided as 1 hour before the obvious indication of flow.

Mean and median values of storm duration expressed in hours are shown in Figure 27. These figures were obtained by calculating the time between 1 hour prior to and 1 hour after initial and final flow indications. Storm duration, based on median values, was reduced 58 percent by the porous pavement site and 69 percent by the infiltration trench. This also illustrates the total flow reduction by infiltration.

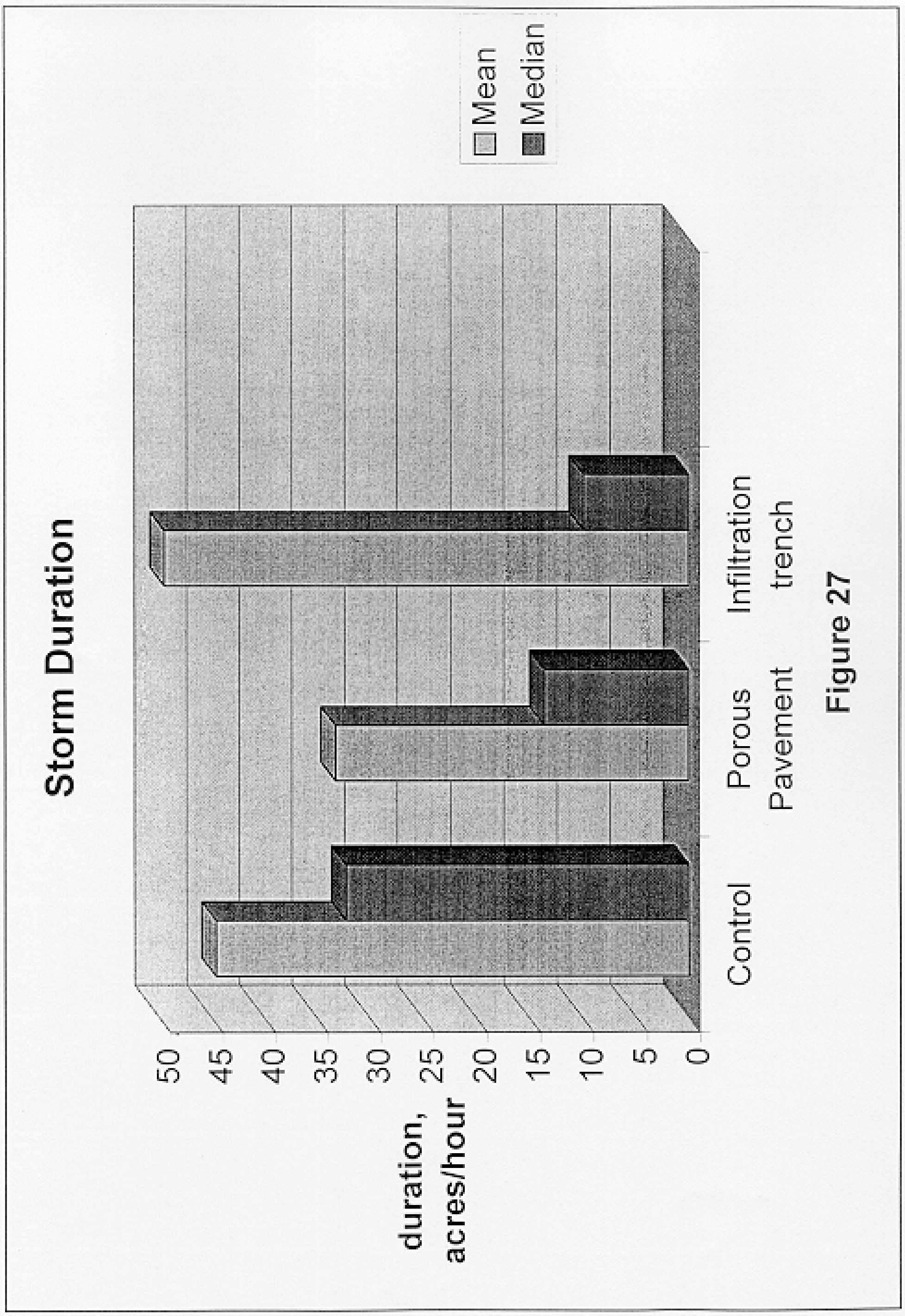

Storm Duration
duration, acres/hour
50
45
40
35
30
25
20
15
10
5
0
Control
Porous Pavement
Infiltration trench
Mean
Median

Figure 27

Stormwater Characteristics

Comparisons to Other Studies

There have been a number of major examinations of the characteristics of urban stormwater in communities around the U.S. and throughout the world. In order to place the data from the current study in historical perspective, it would be useful to examine extracts from some of these earlier studies, and compare them to the results of the current study. Figures 28 - 31 utilize a common abbreviation system for identifying data from a number of studies. The abscissa value abbreviations, and an explanation of the data sources are given below:

Abbreviation	Study Identifier
208	Water Quality Management Planning Study in Northern Virginia carried out under auspices of PL 92-500 Section 208, 1975-1976. Data from townhouse/garden apartments only (NVPDC/Virginia Tech, 1978).
MWURP	Data from Metropolitan Washington Urban Runoff Project, performed under the auspices of the EPA NURP, 1980-1981. Data from townhouse/apartments only. (Virginia Tech, 1983; NVPDC, 1983; MWCOG, 1983).
LC	Data from London Commons Urban BMP Demonstration Project, 1985-1987.
PP	Data from current study, Porous Pavement Urban BMP Demonstration Project, 1985-1988.

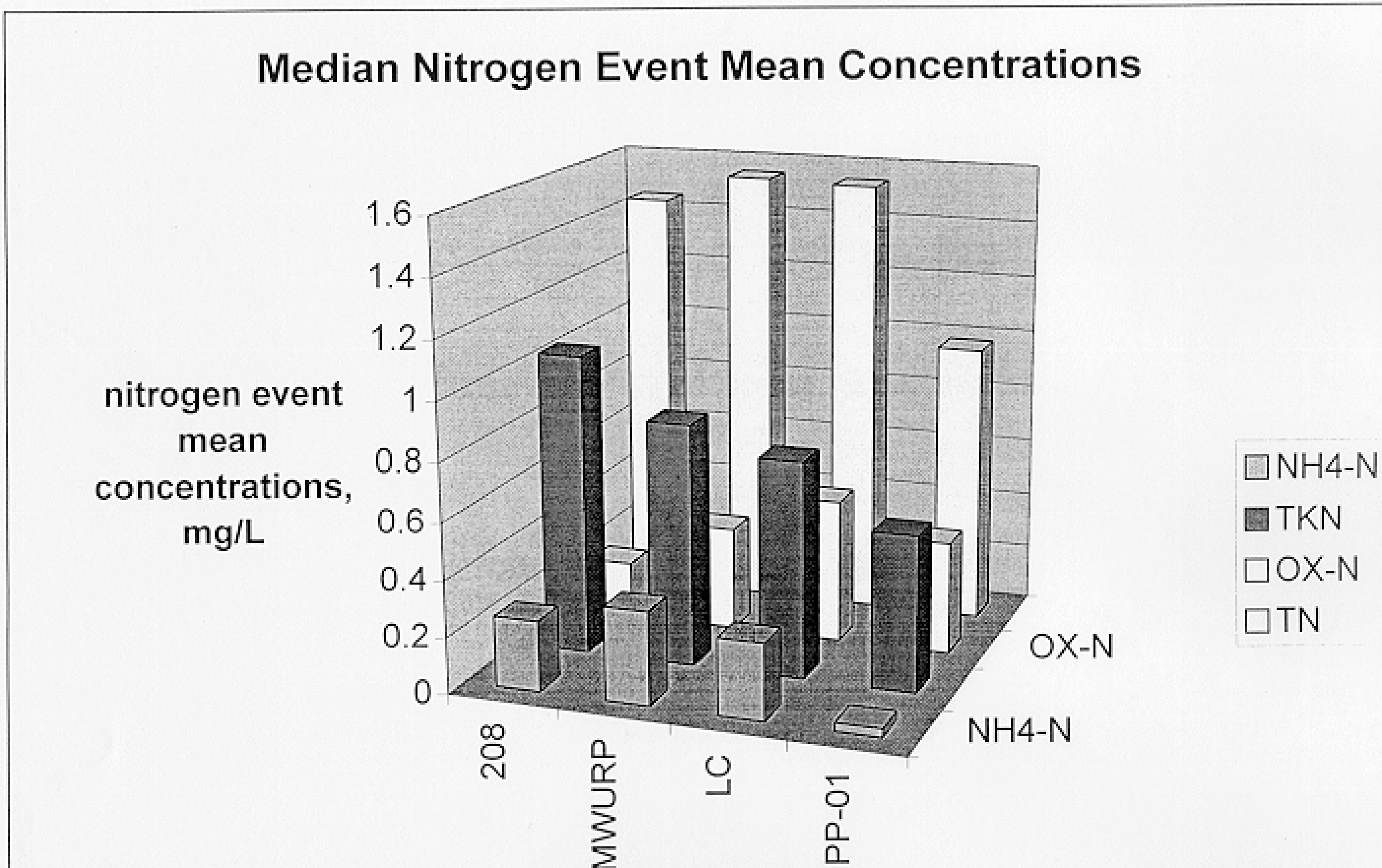

Median Nitrogen Event Mean Concentrations
nitrogen event mean concentrations, mg/L
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
208
MWURP
LC
PP-01
NH4-N
OX-N
NH4-N
TKN
OX-N
TN

Figure 29

Median Trace Metal Event Mean Concentrations

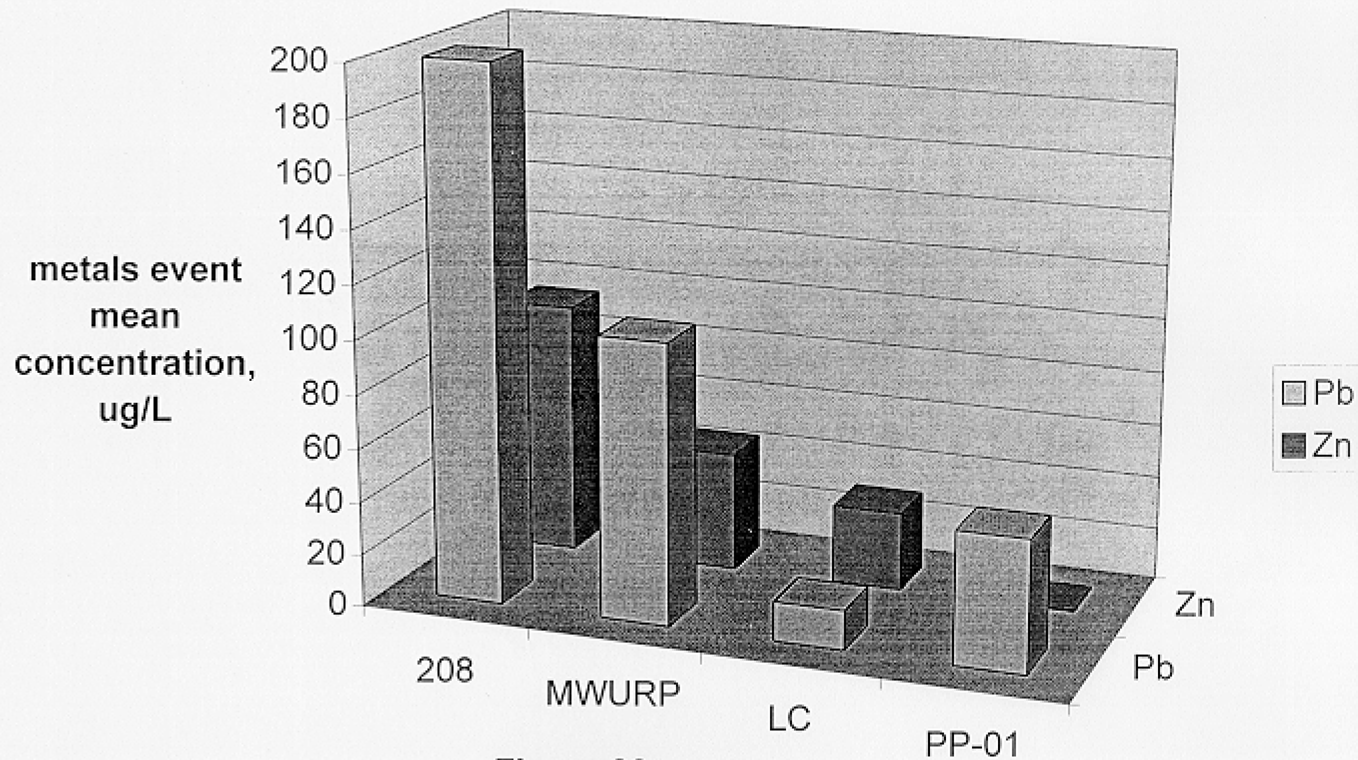


Figure 30

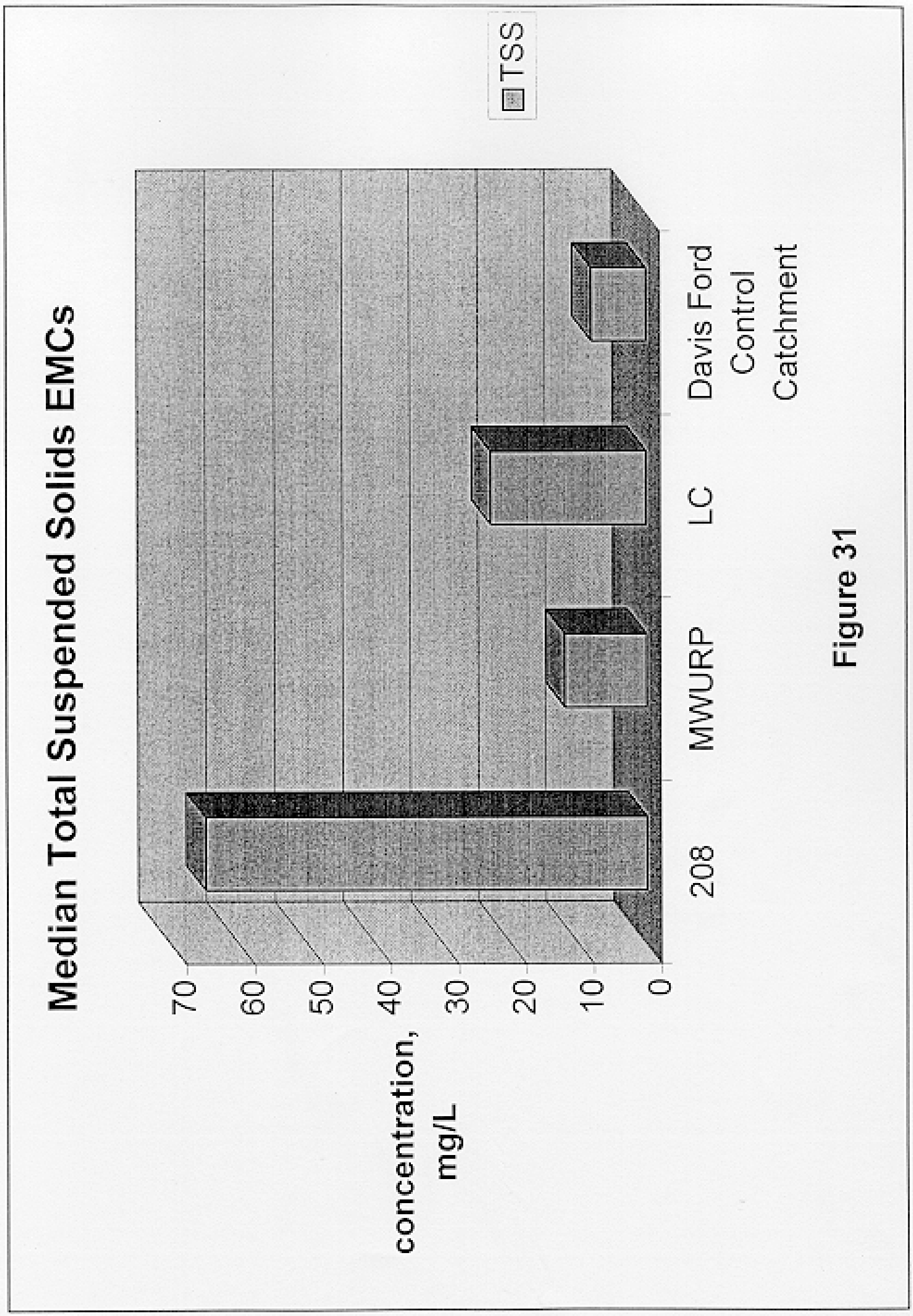

Median Total Suspended Solids EMCs
concentration, mg/L
0
10
20
30
40
50
60
70
208
MWURP
LC
Davis Ford Control Catchment
TSS

Figure 31

Phosphorus.

Very small differences in median EMC's of OP were found for Section 208, MWURP, London Commons, and Porous Pavement. The median Ortho-phosphate phosphorus (OP) EMC for the current study was calculated to be 0.01 mg/L. Values for the three comparative studies range from 0.08 to 0.14 mg/L. These values are comparatively low in relation to Sylvester's Seattle study which found an average OP concentration to be 9.0 mg/L (Weibel et al, 1964).

Sylvester reported a value of 1.4 mg/L total phosphorus TP for typical stormwater runoff (Weibel et al, 1964). For the Davis-Ford project, the median total phosphorus EMC was found to be 0.03 mg/L at the control site, which is much less than the values for the comparative studies.

Median EMC's of TP for the comparative studies were 0.43mg/L, 0.33 mg/L, and 0.24 mg/L for 208, MWURP, and LC respectively. The comparative values can be seen in Figure 28.

These concentrations of TP are well below the 1.4 mg/L average that Sylvester reports as typical for urban stormwater and yet they are well above the 0.01 mg/L level which induces the nuisance algae problem (Weibel et al, 1964). However the degree of the problem depends on the volume (diluting power) of the receiving water and the volume of the runoff which enters that receiving water

.

Nitrogen.

Median EMC's for ammonia nitrogen (NH4_N), total Kjeldahl nitrogen (TKN), and oxidized nitrogen ([NO_2 + NO_3]_N) at the control catchment are displayed in Figure 29.

The concentration of 0.18 mg/L NH4_N found at the control is within the concentration range (0.1- 1.9 mg/L) found by Weibel in a study of runoff in Cincinnatti (Weibel et al, 1964).

The median EMC value of TKN was found to be 0.49 mg/L for the current project, while those of projects 208, MWURP, and LC ranged from 1.10 mg/L to 1.48 mg/L. Colston, in an urban runoff study found a mean Kjeldahl nitrogen value of 0.96 mg/L (Colston, 1974).

The value of oxidized nitrogen of 0.55 mg/L at the control site is much smaller than the 1.2 to 2.9 mg/L range of oxidized nitrogen in urban runoff found by Whipple in Morristown, New Jersey (Whipple et al, 1974). Values of median nitrogen EMC's in Washington area runoff are compared in Figure 29.

Trace Metals.

The majority of all extractable lead concentrations were below the detection limit of 50 ug/L or 0.05 mg/L and yet a median EPb value was found to be .0375 mg/L. Lead concentrations of other areas studies ranged from 14.0 ug/L at the LC project to greater than 300 ug/L for the 208 study.

These values can be seen in Figure 30. It is important to note that the Section 208 study was conducted between 1976 and 1977, prior to the widespread use of unleaded motor fuels. This would account for the high concentration of lead in the stormwater runoff of this study.

The median zinc EMC at the control site was found to be 5 ug/L or 0.005 mg/L. This is small in comparison to the mean concentration of 0.361 mg/L found in stormwater runoff in Lancaster, Pennsylvania (Huber et al, 1979). The maximum concentration for the current project during the entire monitoring period was 0.222 mg/L.

Zinc deposition has been found to be related to the direct automobile use in the specific catchment. The low concentration of zinc for the current study is accounted for by the very light usage of the parking lot by automobiles.

Total Suspended Solids

The median total suspended solids (TSS) EMC for the control catchment was found to be 4.8 mg/L. As with nutrient and trace metal EMC's the median EMC's for total suspended solids from the Davis-Ford project were much smaller than those found in the literature. EMC's for the comparative area studies ranged from 24 mg/L (MWURP) to greater than 100 mg/L (208). These values are shown in Figure 31.

The median value of 4.8 mg/L at the control site is small compared to the 5.0 to 1200 mg/L range found by Weibel in Cincinnatti runoff (Weibel et al, 1964). Whipple found a median value of 84.0 mg/L TSS for a residential site; the lowest value of four land use types (Whipple, 1977).

Determination of Pollutant Removal Efficiency

Calculation of pollutant removal involves two components. The first component illustrates the reduction of constituent concentrations in the runoff. The second illustrates the BMP's reduction of loading of pollutants, due to flow reduction, which has been shown to be substantial.

The calculation strategy was to analyze pollutant removal based on matched storms which involved the entire population of data from all three catchments (including those storms in which the BMP's removed 100 percent of the runoff, but not including those storms in which equipment malfunctions occurred).

The change in pollutant EMC may be assumed to be related to alterations due to physical, chemical, or biological process in the sub-base such as:

1. association of pollutants with particles by physisorption or by chemisorption.
2. physical filtration by the soil particles.
3. biological processes which may alter the pollutant of interest.

Runoff pollutant concentrations are expressed as event mean concentrations (EMC's). The EMC may be taken to be the concentration that would have been determined had the entire stormwater flow been captured, mixed and a single sample abstracted for analysis. Median concentrations of all nutrients are displayed in Figure 32. Mean concentrations of these same nutrients are shown in Figure 33. EMC's which were below the detection limit for that parameter were replaced with a value which was half of the detection limit value. EMC reductions were calculated for the synoptic data set and the entire data set.

The efficiency calculation based on a determination of EMC change between control and BMP does not consider hydrologic balance and is expressed as:

$$\text{percent removal} = \frac{EMC_{control} - EMC_{BMP} * 100}{EMC_{control}}$$

where,

$EMC_{control}$, EMC_{BMP} = event mean concentration in mass/length^3

Because these are volume control BMP's, efficiency determination derived from the loading of pollutants at each site takes into account the hydrologic performance of the sub-base in detaining stormwater as well as physical/biological/chemical processes occurring in the sub-base.

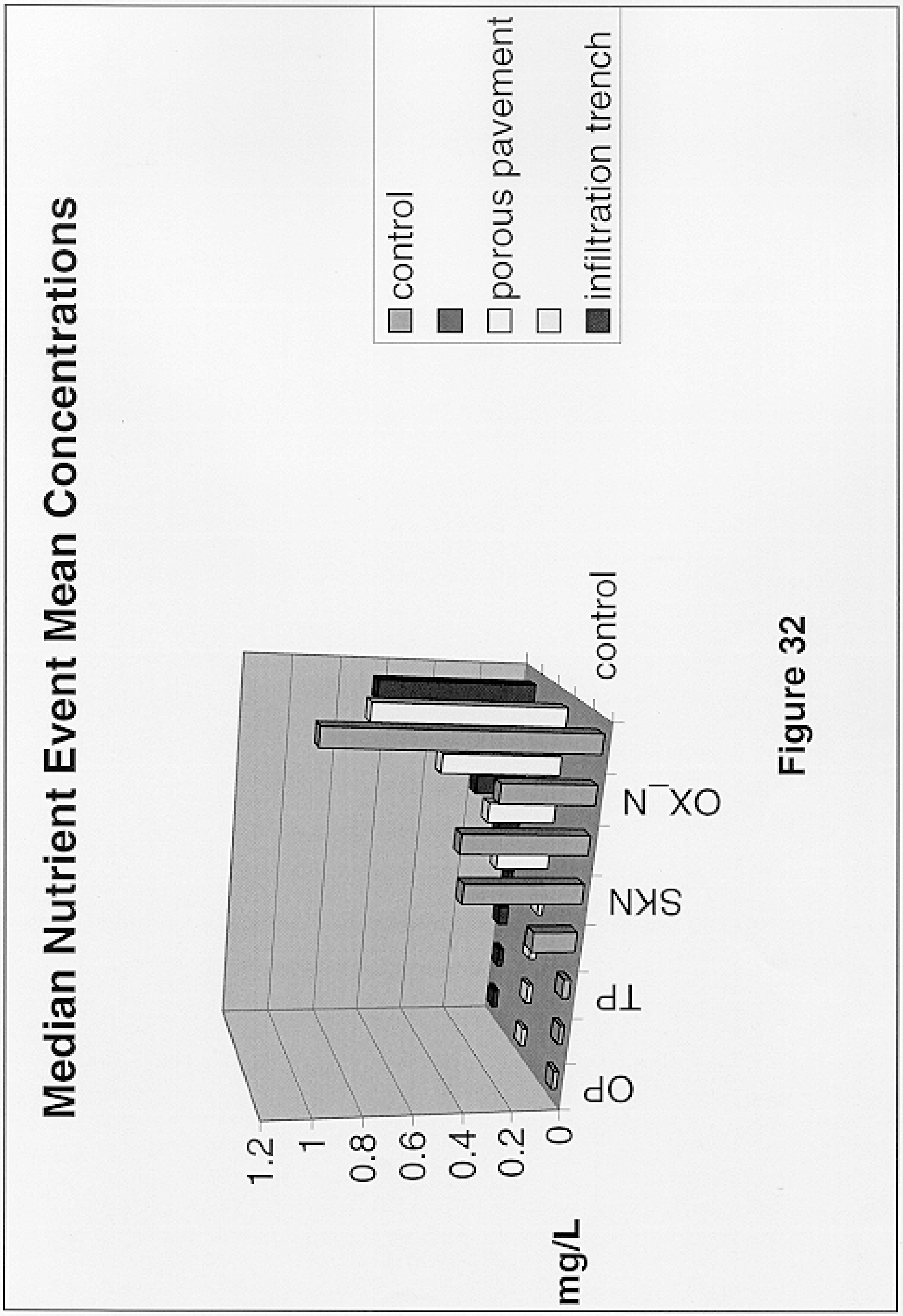
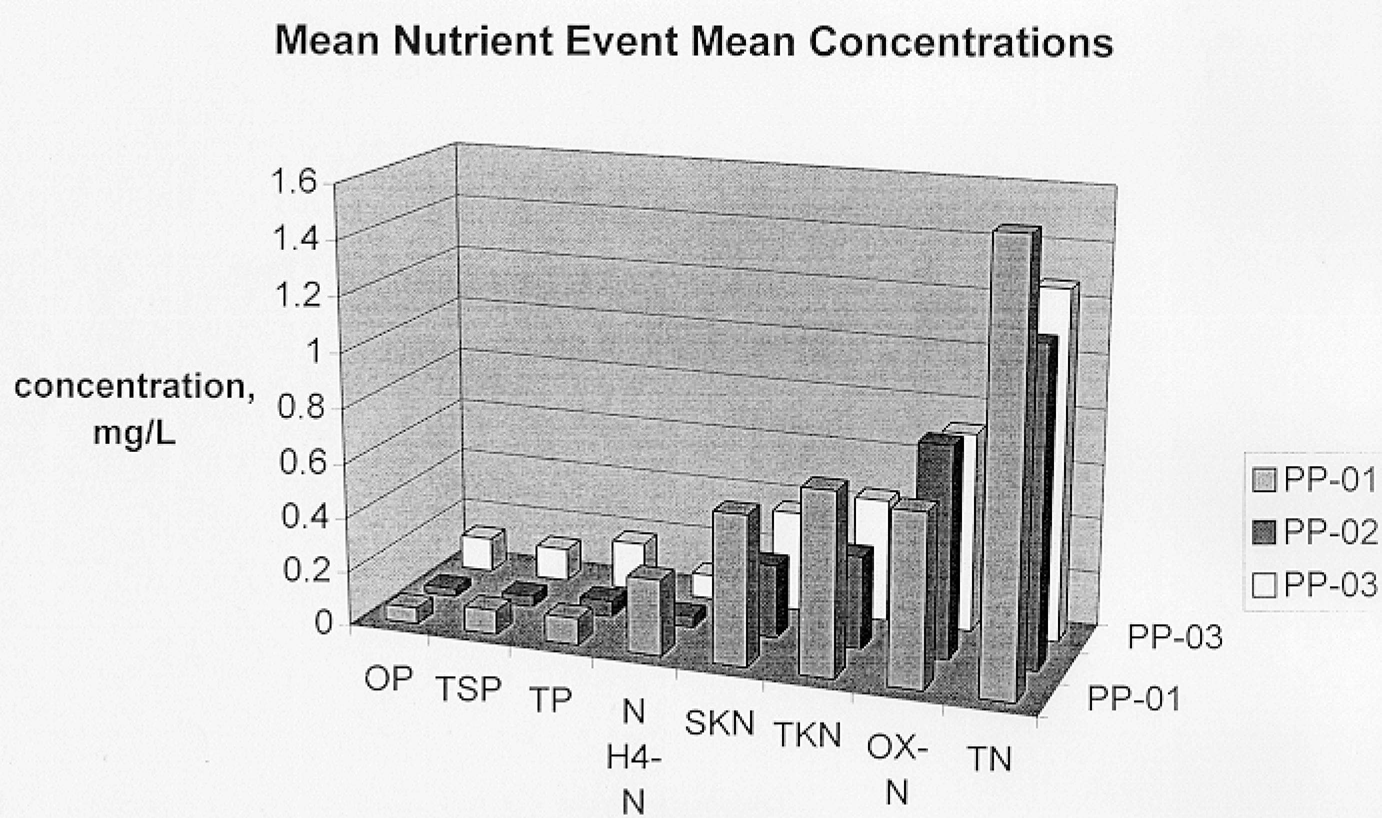

Median Nutrient Event Mean Concentrations
mg/L
1.2
1
0.8
0.6
0.4
0.2
0
OP
TP
SKN
OX_N
control
control
porous pavement
infiltration trench

Figure 32



Mean Nutrient Event Mean Concentrations
concentration, mg/L
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
OP
TSP
TP
N H4- N
SKN
TKN
OX- N
TN
PP-01
PP-03
PP-01
PP-02
PP-03

Figure 33

Calculation of pollutant load reduction was made by two separate strategies. They are:

1. the median loading rate reduction and
2. the long term removal efficiency.

These strategies are illustrated in Figure 34.

The calculation of loads for the BMP's based on median loading rate is expressed as:

$$\text{percent removal} = \underline{1 - \frac{\text{outflow load}}{\text{inflow load}} \times 100}$$

where outflow load = $EMC_{BMP} \times V_{BMP}$ and inflow load = $EMC_{control} \times V_{control}$

and where:

$load_{control}$, $load_{BMP}$ = total load in units of mass (lbs/acre)

$EMC_{control}$, EMC_{BMP} = event mean concentration (mg/L)

$V_{control}$, V_{BMP} = total storm volume in units of $length^3$

In this case, the removal efficiency for the BMP's are taken to be the median (50th percentile, i.e., half of the measured values lie above this point and half below) value of all the storm event efficiencies calculated. The median is often used for comparisons of events that are not normal in the statistical sense, because arithmetic averages naturally weight results toward the larger numbers. Therefore, in general, the median is

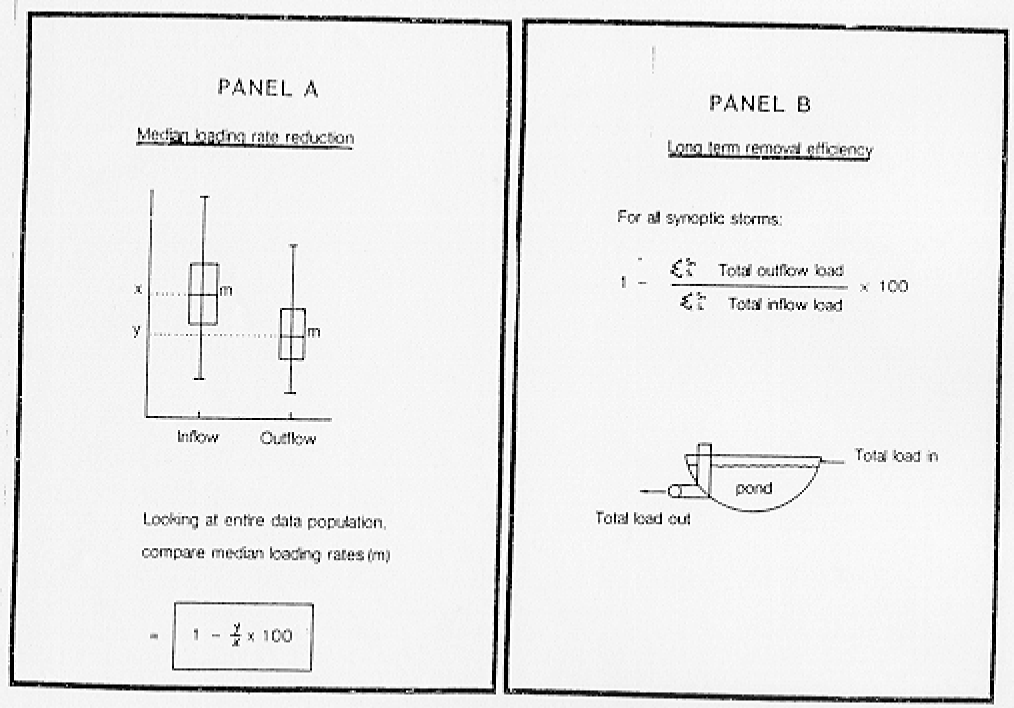


Figure 34. Methods Used to Calculate the Pollutant Removal Efficiency of Urban BMPs [69]

taken to be a much better estimator of the central tendency of a distribution than the mean.

The second calculation strategy was to express loading removal as a long term removal efficiency:

1 - total outflow load x 100 total inflow load

It should be noted that the substantial difference in drainage area to each of these stations was taken into account in calculating loads on a unit acre basis for both methods of calculation.

CHAPTER V

POLLUTANT REMOVAL EFFICIENCY

Phosphorus. Ortho-phosphate phosphorus (OP) EMC's ranged from 0.01 mg/L to 1.75 mg/L with median values of 0.01, 0.02, and 0.08 mg/L for PP01, PP02, and PP03 respectively. These values are comparatively low in relation to Sylvester's Seattle study which found an average OP concentration to be 9.0 mg/L (Weibel et al, 1964).

For the Davis-Ford project, total phosphorus (TP) EMC's ranged from 0.01 mg/L to 2.65 mg/L at the control catchment. The median EMC's for TP were observed to be 0.03 mg/L, 0.04 mg/L, and 0.10 mg/L for control, porous pavement, and infiltration trench respectively.

The calculated results in Table 11 should be regarded with some discretion because of the fact that the concentrations are minute and a value of 0.01 mg/L (PP01) in comparison to 0.03 mg/L (PP03) represents a 300 percent difference. This only suggests that the runoff filtering through the soil media was more apt to contact all three P forms than was the runoff from the conventional pavement.

When the entire data set is considered (synoptic and nonsynoptic storm events), median EMC's for OP are found to be 0.02 mg/L for porous pavement and 0.08 mg/L for the infiltration trench catchment resulting in a net negative reduction. Median values for TP were 0.03 mg/L for control and porous pavement while a median value of 0.10 mg/l was calculated for TP at the infiltration trench site. This indicates that TP is negatively reduced at the infiltration trench site. Table 10 illustrates these reduction efficiencies.

Table 10
Percent Reduction of Phosphorus Forms
Based on Median EMC's of Entire Data Set

BMP	OP	TSP	TP
Porous pavement (PP02)	-100	0	-33
Infiltration trench (PP03)	-700	-300	-233

Storm loading of all pollutants were expressed as lbs/acre. Median values of loading of OP, TSP, and TP for the entire data set at the control catchment were 0.002 lbs/acre, 0.003 lbs/acre, and 0.005 lbs/acre respectively.

Based upon median storm load reduction, porous pavement displayed a removal efficiency very nearly equal to that of the infiltration trench when the entire database was analyzed. The efficiencies ranged from 61 percent - 72 percent removal of the phosphorus species at the porous pavement BMP and 56 percent - 82 percent removal at the infiltration trench. These efficiencies are shown in Table 11 and Figures 35 and 36.

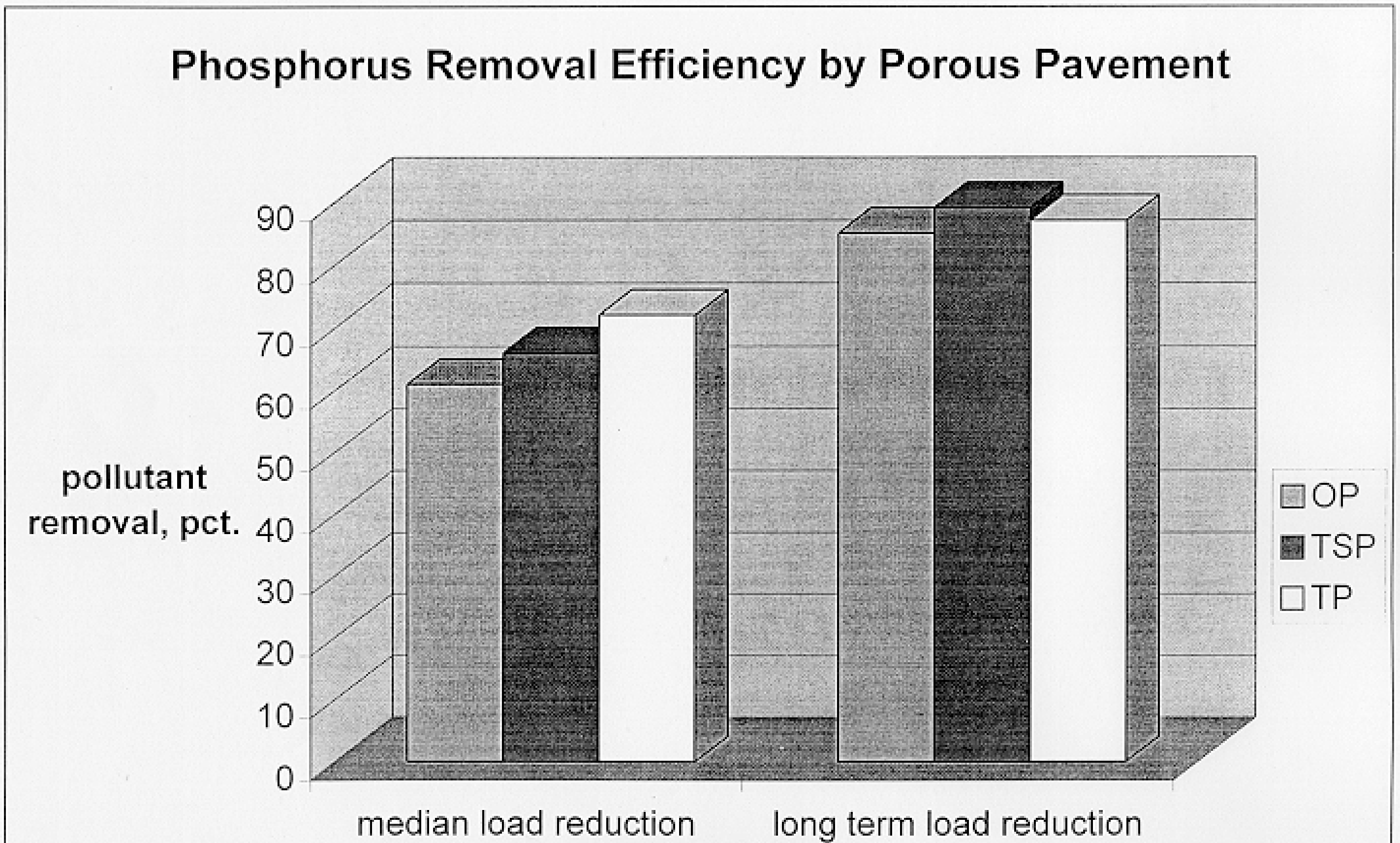
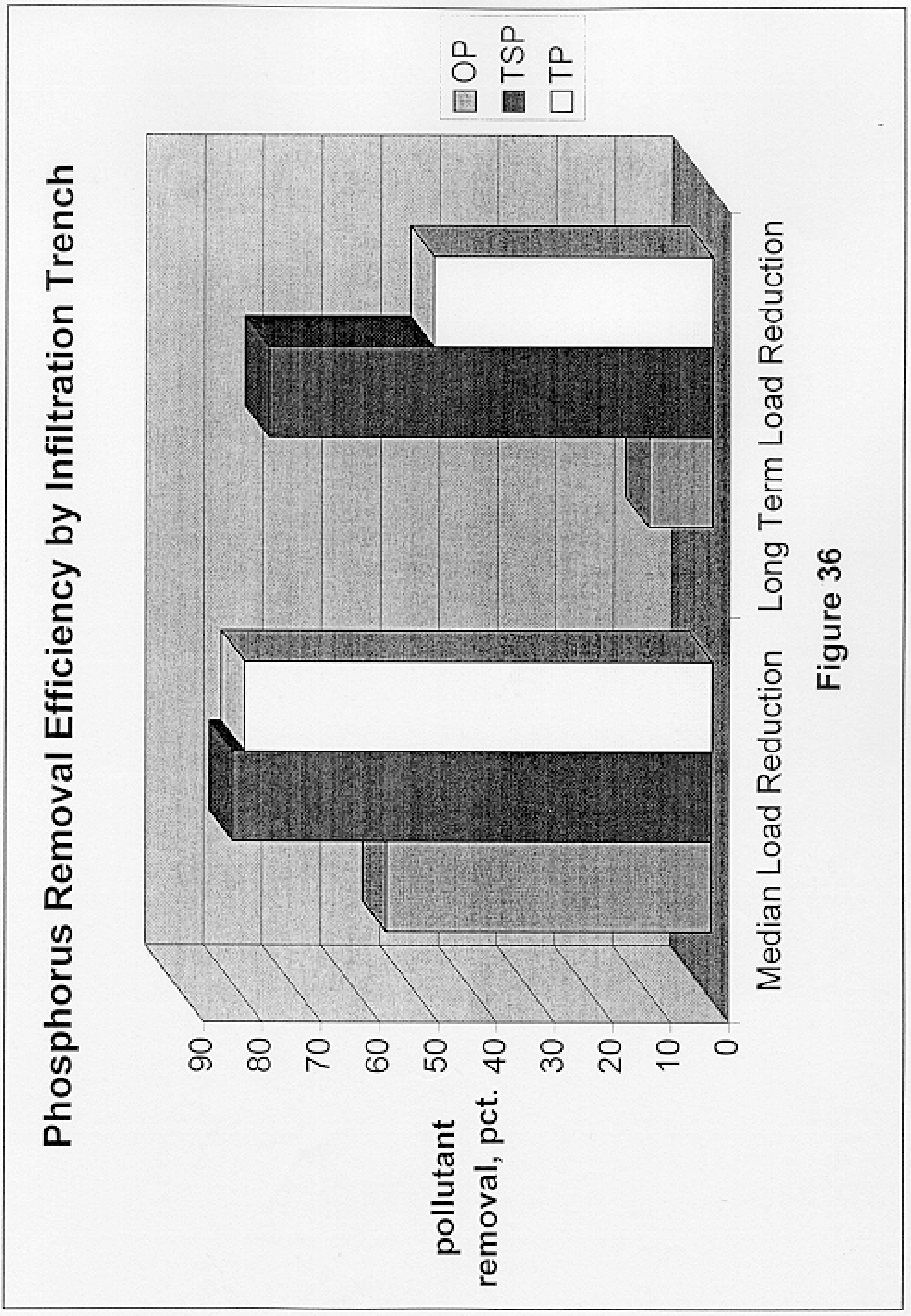

Phosphorus Removal Efficiency by Porous Pavement
pollutant
removal, pct.
0
10
20
30
40
50
60
70
80
90
median load reduction
long term load reduction
OP
TSP
TP

Figure 35



Phosphorus Removal Efficiency by Infiltration Trench
pollutant removal, pct.
90
80
70
60
50
40
30
20
10
0
Median Load Reduction
Long Term Load Reduction
OP
TSP
TP

Figure 36

Table 11
Percent Reduction Of Phosphorus Forms
Based on Median Load Reduction (lbs/acre)

BMP	OP	TSP	TP
Porous Pavement (PP02)	61	66	72
Infiltration Trench (PP03)	56	82	80

For the entire data-set, calculations by the long term load reduction method yielded similar removal values with the exception of ortho-phosphorus at the infiltration trench site. These figures may be seen in Table 12 and Figure 36.

The effectiveness of the infiltration and the volume control aspects of the two BMP's are clearly exhibited in the near 90 percent reduction in P loadings by porous pavement and the 11 percent - 76 percent reduction by the infiltration trench. These results are consistent with the phosphorus reduction efficiencies found by MWCOG (trench, 40 percent - 60 percent ; porous pavement, 60 percent - 80 percent) (MWCOG, 1982).

These removal efficiencies are a direct result of the 63 percent - 100 percent total flow reduction exhibited by the two BMP's and calculated earlier in this report.

Table 12
Percent Reduction of Phosphorus Forms
Based on Long Term Load Reduction (lbs/acre)

BMP	OP	TSP	TP
Porous Pavement (PP02)	85	89	87
Infiltration Trench (PP03)	11	76	48

Nitrogen. Median EMC's for ammonia nitrogen (NH4_N), total Kjeldahl nitrogen (TKN), and oxidized nitrogen ([NO_2 + NO_3])_N are displayed in Figure 29. For the entire data set NH4_N EMC's ranged from 0.01 mg/L to 1.30 mg/L with median concentrations of 0.18 mg/L, 0.03 mg/L and 0.05 mg/L for PP01, PP02, and PP03 respectively.

The concentration of 0.18 mg/L at the control is relatively small compared to the study of runoff in Cincinnatti, in which Weibel found NH4_N to range from 0.1 to 1.9 mg/L (Weibel et al, 1964).

OX-N EMC's ranged from 0.05 mg/L to 2.81 mg/L with median concentrations of 0.55 mg/L, 0.76 mg/L, and 0.61 mg/L for PP01, PP02, and PP03 respectively. The value of 0.55 mg/L at the control catchment is much smaller than the 1.2 to 2.9 mg/L range for oxidized nitrogen in urban runoff found by Whipple in Morristown, New Jersey (Whipple, 1974).

Median EMC values of SKN for the entire data base at PP02 and PP03 were equivalent at 0.25 mg/L. Median EMC's of TKN at PP02 and PP03 were comparable at 0.29 mg/L and 0.30 mg/L. The median EMC for TKN at the control site was 0.49 mg/L. Colston, in an urban runoff, study found a median Kjeldahl nitrogen value of 0.96 mg/L (Colston, 1974).

For the entire data set both BMP's were efficient in positively reducing EMC's of NH4_N, SKN, and TKN and yet both failed to positively reduce oxidized nitrogen and total nitrogen species. This is due to the action of nitrification by soil microbes in the subsoil. The exchange capacity of the soil may also play a role in this finding.

With regard to the entire data set, median values of removal of nitrogen forms based on observed reductions in EMC's are shown in Table 13 and in Figure 37. NH4_N was the most highly reduced nitrogen species; 83 percent and 72 percent reductions by porous pavement and infiltration trench. Both BMP's exhibited negative reductions of OX-N; porous pavement -39 percent, and infiltration trench -12 percent. The infiltration trench reduced 34 percent of the SKN and 27 percent of the TKN while porous pavement reduced 46 percent of the SKN and 41 percent of the TKN EMC.

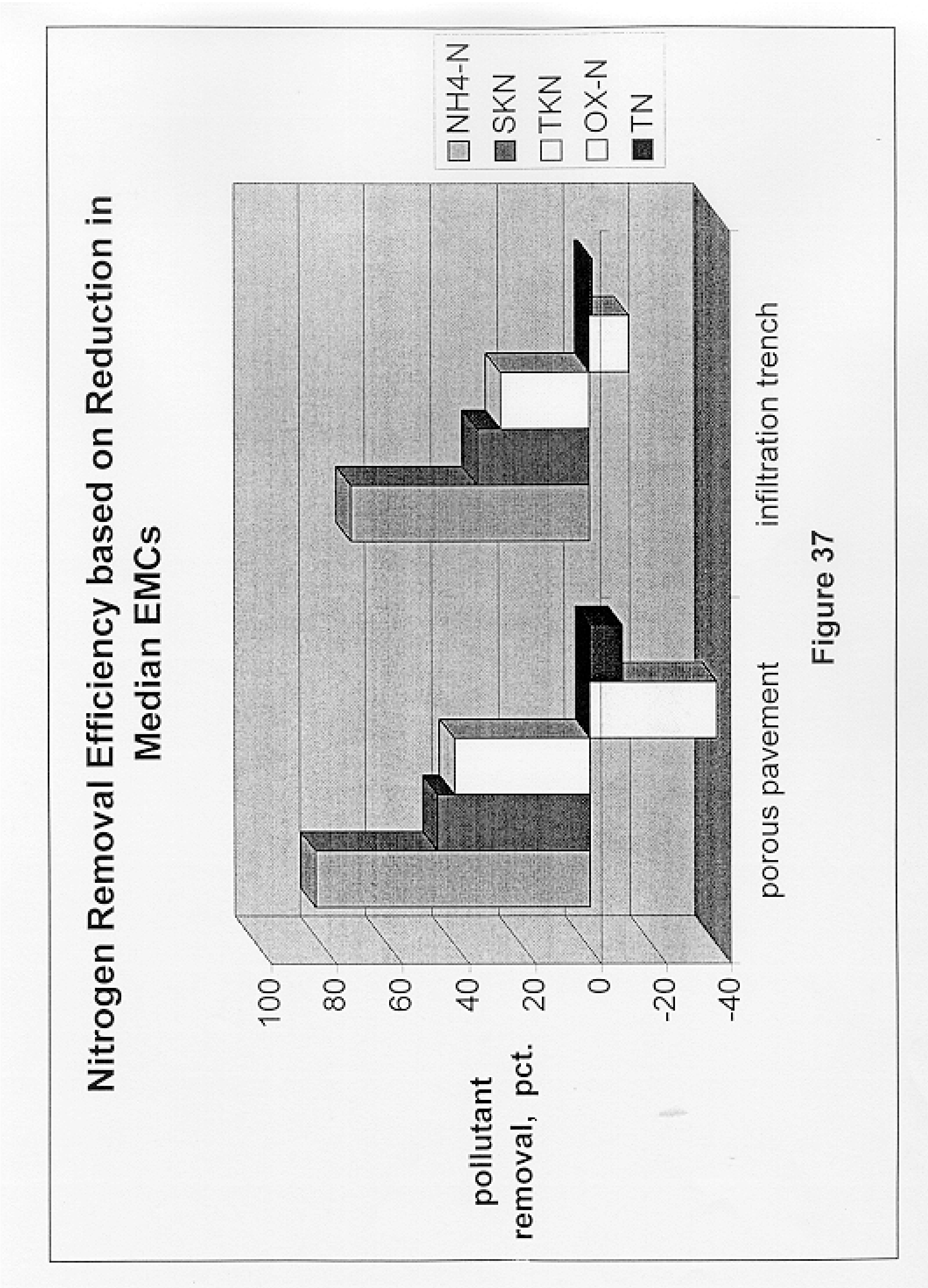

Nitrogen Removal Efficiency based on Reduction in Median EMCs
pollutant removal, pct.
100
80
60
40
20
0
-20
-40
porous pavement
infiltration trench
NH4-N
SKN
TKN
OX-N
TN

Figure 37

Table 13
Percent Reduction of Nitrogen Forms
Based on Median EMC's of Entire Data Set

BMP	NH_4^+-N	SKN	TKN	OX-N	TN
Porous pavement (PP-02)	83	46	41	-39	-10
Infiltration trench (PP-03)	72	34	27	-12	-1

The reduction of loadings (expressed as a percentage) by the two BMP's, by median value and long range removal calculations for the entire data set are shown in Tables 14 and 15 respectively.

Based on median storm loadings (lbs/acre) of the entire data set porous pavement removal efficiencies ranged from 54 percent reduction of oxidized nitrogen (OX-N) to 96 percent reduction of ammonium nitrogen (NH_4^+-N). As may seen in Table 16, NH_4^+-N was the N form most effectively reduced by both BMP's; 96 percent at porous pavement and 99 percent at the infiltration trench which is consistent with the findings of EMC reduction for both BMP's. Of all N forms, the reduction of OX-N by the two BMP's exhibited the largest difference; 54 percent reduction at PP02 and 82 percent at PP03.

Table 14
Percent Reduction of Nitrogen Forms
Based on Median Load Reductions (lbs/acre)

BMP	NH_4^+-N	SKN	TKN	OX-N	TN
Porous pavement (PP-02)	96	81	81	54	68
Infiltartion trench (PP03)	99	97	93	82	88

MWCOG found that TN loading was reduced 40 percent - 60 percent at an infiltration trench and 60 percent - 80 percent at a porous pavement site (MWCOG, 1982).

When load removal was calculated by the long term load reduction method it was found that the efficiencies were improved an average of six percentage points at porous pavement, and yet were less effective by an average of eight percentage points at the infiltration trench. These values may be seen in Table 15 and Figures 38 and 39.

Table 15
Percent Reduction of Nitrogen Forms
Based on Long Term Load Reduction (lbs/acre)

BMP	NH_4^+-N	SKN	TKN	OX-N	TN
Porous pavement (PP02)	97	88	88	62	76
Infiltration trench (PP03)	92	90	86	71	82

Trace Metals.

Trace metal removal efficiencies were the highest of all pollutant reduction efficiencies based on both median EMC removal and median storm load reduction expressed as lbs/acre. The majority of lead concentrations were below the detection limit of 50 ug/L. These concentrations were replaced with values which were half of the detection limit; in this case 0.025 mg/L. This was done to calculate the loading removal efficiency of lead.

Nitrogen Removal Efficiencey Based on Median Load Reduction

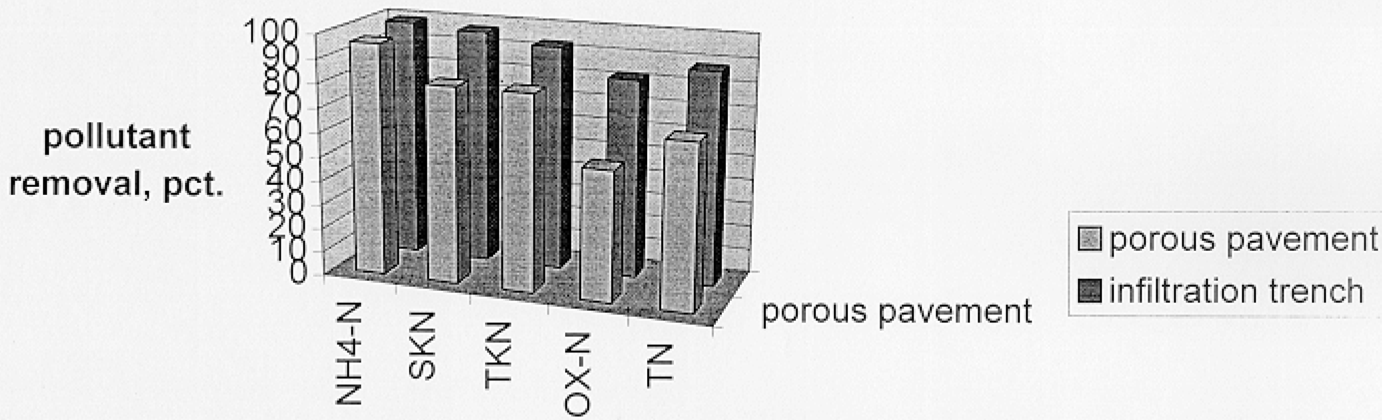


Figure 38

Nitrogen Removal Based on Long Term Load Reduction

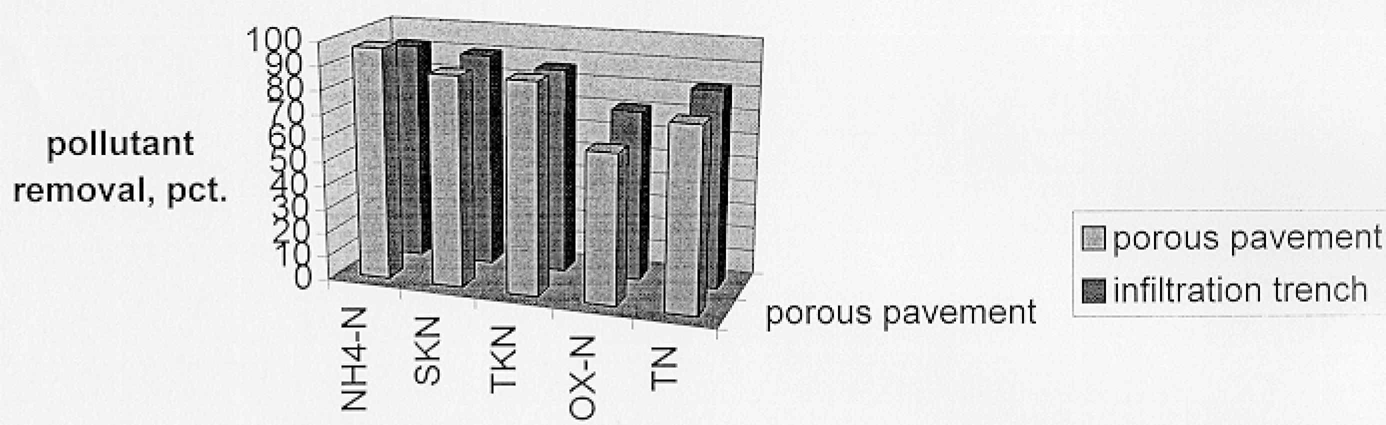


Figure 39

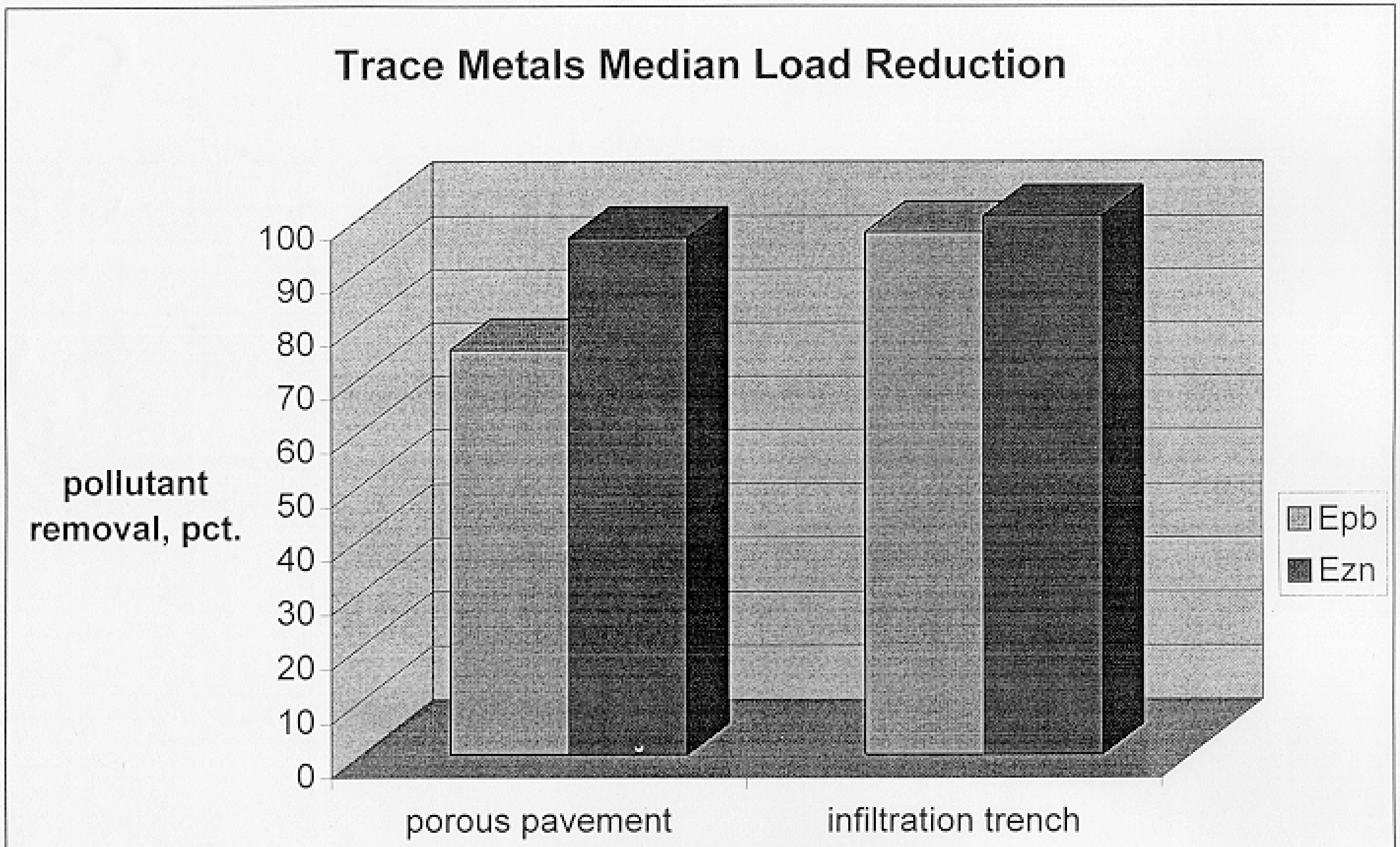
The median zinc EMC at the control site was found to be 31 ug/L or 0.031 mg/L. This is small in comparison to the mean concentration of 0.361 mg/L zinc found in stormwater runoff in Lancaster, Pennsylvania (Huber et al, 1979). The maximum EMC for the entire monitoring period was 0.222 mg/L.

For the entire data base the zinc EMC was reduced 84 percent at porous pavement and 67 percent at the infiltration trench.

Due to the volume control aspect of the BMP's and expressed as lbs/acre the zinc removal efficiency improved to 96 percent at the porous pavement and increased to 100 percent at the infiltration trench for the median value calculation method. Metropolitan Washington Council of Governments found porous pavement to reduce zinc loading at nearly 100 percent (MWCOG, 1983). They also found in a different study that an infiltration trench reduced 80 percent - 100 percent of the zinc loading (MWCOG, 1982).

Calculated zinc removal efficiencies were not as high for porous pavement and infiltration trench when considering the entire data set on a long term load reduction method; 94 percent and 88 percent.

Lead removal efficiencies ranged from 75 percent - 80 percent for porous pavement and from 83 percent - 97 percent for the infiltration trench. Trace metal removal efficiencies may be seen in Tables 17 and 18 and in Figures 40 and 41.



Trace Metals Median Load Reduction
pollutant removal, pct.
100
90
80
70
60
50
40
30
20
10
0
porous pavement
infiltration trench
Epb
Ezn

Figure 40

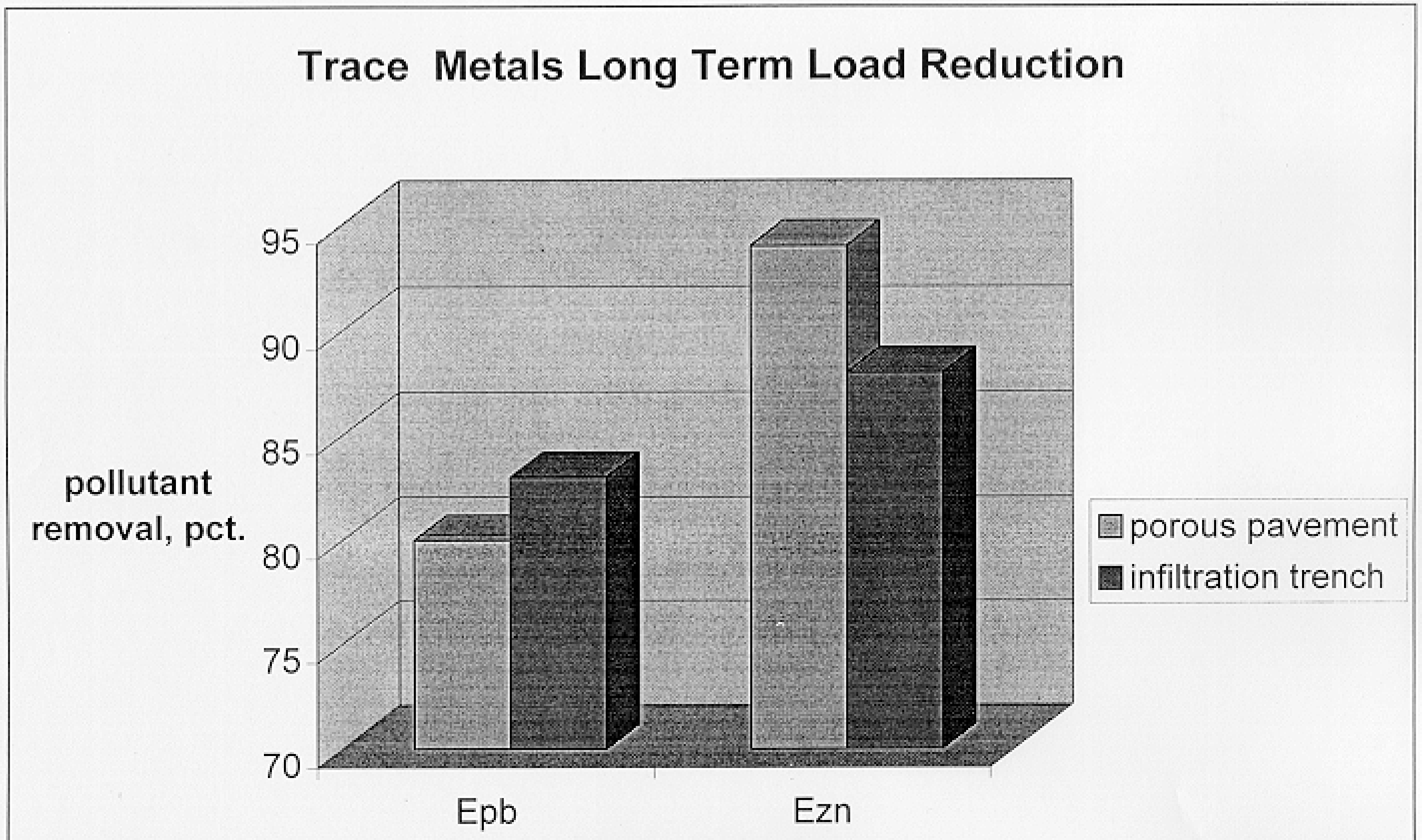

Trace Metals Long Term Load Reduction
pollutant
removal, pct.
95
90
85
80
75
70
Epb
Ezn
porous pavement
infiltration trench

Figure 41

Table 16
Percent Reduction of Trace Metals
Based on Median Load Reductions (lbs/acre)

BMP	EPb	Ezn
Porous pavement (PP-02)	75	96
Infiltration trench (PP-03)	97	100

Table 17
Percent Reduction of Trace Metals
Based on Long Term Load Reductions (lbs/acre)

BMP	EPb	EZn
Porous pavement (PP02)	80	94
Infiltration trench (PP03)	83	88

Total Suspended Solids

Complete data set TSS EMC's ranged from 1.0 mg/L to 69 mg/L for the three catchments. Median EMC's were 6.7 mg/L, 3.6 mg/L, and 6.4 mg/L for PP01, PP02, and PP03 respectively.

As with nutrient and trace metal EMC's, the median EMC's for total suspended solids from the Davis Ford Project were much smaller than those found in the researched literature. Whipple found a median value of 84 mg/L TSS for a residential site; the lowest value of four land use types (Whipple, 1974). The median value of 3.6 mg/L at the control site is small compared to the 5.0 to 1200 mg/L range found by Weibel in Cincinnatti, runoff (Weibel, 1964).

Based on EMC values, reduction efficiencies for the complete data set are calculated as -33 percent for porous pavement and 46 percent for the infiltration trench.

The median value of TSS loading at the control site was found to be 0.786 lbs/acre. Loading reductions by the BMP's are consistent with MWCOG's (MWCOG, 1982) findings and are as follows:

Table 18
Percent Reduction of TSS
Based on lbs/acre:

BMP	Median Load Reduction	Long Term Load Reduction
Porous pavement (PP02)	54	65
Infiltration trench (PP03)	100	83

The values in Table 18 above are also shown in graph form in Figures 40 and 41.

The efficiency of porous pavement to reduce TSS loading was lower than that found by MWCOG (96 percent) and yet the efficiency of the infiltration trench was higher than that found in a different MWCOG study (50 percent) (MWCOG, 1983).

Lysimeter collected soil pore water.

Around the perimeter of the infiltration trench, the three working lysimeters collected soil pore water from a depth of around 1.45 meters. Lysimeters were allowed 24-48 hours following a storm event to collect soil pore water. After this allotted time, the samples were transferred to individual samples bottle and transported to the lab. PP33 failed to collect samples for any of the storms. Because this lysimeter continued to hold a charge as long as the others, yet failed to collect any water it is believed that the soil in this area was so fine as to not allow the retention of soil pore water.

Storms for which soil interstitial water samples were collected were:

87331 88086 88118

87344 88097

Table 19
Soil Pore Water EMC Comparisons

Station	Storm number	OP	TSP	NH_4^+-N	SKN	OX-N
PP-31	87344	0.01	0.03	0.02	0.21	1.11
PP-31	88086	-	0.01	0.01	0.04	-
PP-31	88097	0.01	0.01	0.03	0.14	0.72
PP-31	88118	-	0.01	0.02	0.05	-
PP-32	87331	-	0.02	-	0.19	0.41
PP-32	87344	0.01	0.04	0.02	0.16	0.53
PP-32	88097	0.01	0.01	0.04	0.15	0.58
PP-32	88118	-	0.01	0.04	0.10	-
PP-34	87331	-	0.03	-	0.17	0.56
PP-34	87344	0.01	0.01	0.02	0.18	0.90
PP-34	88086	-	0.01	0.02	0.06	-
PP-34	88097	0.01	0.01	0.02	0.09	0.74
PP_34	88118	-	0.01	0.02	0.05	-

Phosphorus in Soil Pore Water

In soils, organic phosphorus undergoes cycles of mineralization and immobilization. Phosphate may be removed from runoff by several mechanisms:

1. Rapid removal or sorption
2. slow mineralization and insolubilization
3. plant uptake
4. biological immobilization.

Because of the strong soil retention, phosphorus migrates little in the soil profile and quantities are low. Although most rapid infiltration systems do well in phosphate removal, it is possible to saturate the soil and get phosphate breakthrough.

Median values of ortho-phosphorus were 0.01 mg/L for all three collection sites. This is 100 percent of the OP which was carried out with the runoff at PP03 based on median EMC's. Soil pore water contained 60-67 percent of the total soluble phosphorus which was carried away with the discharge. This value of TSP is the amount which is unavailable for removal by sorption, the most important removal mechanism.

Nitrogen in Soil Pore water.

Soil pore water analyzed for NH_4^+-N from PP31, PP32, and PP34 contained 25, 40, and 25 percent of the median EMC's for the same storms at PP03. The oxidized nitrogen fractions were 177, 102, and 142 percent of the median EMC's for the same 5 storms. The combination of these two parameters, low NH_4^+-N and high OX-N are a good indication that favorable conditions are present for nitrifying bacteria in converting NH4_N to OX-N in soil interstices. This can be seen in the -16 percent removal efficiency of OX-N at PP03 in comparison to the median EMC's at PP01.

Median Values for soluble Kjeldahl nitrogen ranged from 0.09 mg/L to 1.24 mg/L. This was 33 percent - 56 percent of the runoff median EMC's displayed for the 5 storms at PP03.

Chapter VI
Summary and Conclusions

1. In the course of this study 60 synoptic storm events from a total of 64 storm events in which all data were complete, were analyzed for both hydrology and pollutant transport. These synoptic storms were compared to the analysis of the entire data set for the project. The total number of events successfully monitored exceeded the total in the work plan by over 50 percent.

2. Atmospheric depositions (wetfall and dryfall) were small in comparison to the deposition rates found by The Water Resources Planning Board and Metropolitan Washington Council of Governments. Atmospheric wetfall loading rates, expressed as pounds/acre, were consistently higher than dryfall loading rates for all pollutants of interest. The sum of dryfall loading rates preceding representative storms and wetfall loading rates during the same storms exceeded 100 percent of pollutants found in the runoff at the control for several constituents. Therefore, it is hypothesized that incomplete washoff occurred for many of the storm events.

3. Seasonal aspects of atmospheric fallout show that dryfall loadings of all pollutants of interest except zinc, orthophosphorus, and total soluble phosphorus were heaviest during the months of spring. Zinc loadings were largest during the summer months. Wetfall NH_4_N loadings were heaviest during the spring.

4. A definite pattern was seen when the author compared the length of antecedent dry period with dryfall accumulation. With increasing antecedent dry period pollutants accumulate from dryfall sources but only to a certain point at which the amount is not exceeded.

5. Hydrologic data were collected for a total of 57 storm events. The infiltration trench was 100 percent efficient in flow reduction for 17 storms. Porous pavement and the infiltration trench were simultaneously 100 percent efficient in reducing flow for 4 storm events.

6. Total flows at the porous pavement catchment for the entire data set of storm events were found to be to be 47 percent of the total flows at the control catchment. The flows at the infiltration trench for the same set were much more variable in comparison to the control, yet exhibited a 100 percent reduction in total flow when expressed on a median value basis.

6. Median peak flows were reduced 68 percent by the porous pavement BMP and 83 percent by the infiltration trench BMP. Peak flows were delayed a median value of five minutes at the infiltration trench and six minutes at the porous pavement catchment. Storm duration was reduced 58 percent by porous pavement and 69 percent by the infiltration trench.

7. Following two years of monitoring the infiltration trench became clogged and required constructive modifications. Modifications included removal and replacement of the stone bed. During this time separate data loggers were installed at each station to improve data collection performance.

8. Pollutant removal performance was determined in three ways for the two BMP's:

- Reduction in median EMC's
- Reduction in median total load (lb/acre)
- Reduction in long term load (lb/acre)

These three expressions were calculated for the entire data set. For the entire data set median EMC and load reductions were as follows:

Pollutant	Porous pavement	Infiltration trench	Porous pavement	Infiltration trench
OP	-100	-700	61	56
TSP	0	-300	66	82
TP	-33	-233	72	80
NH-3N	83	72	96	99
SKN	46	34	81	97
TKN	41	27	81	93
OX-N	-39	-12	54	82
TN	-10	-1	68	88
TSS	-33	28	54	100
Pb	-33	0	75	97
ZN	84	67	96	100

When the reduction efficiencies are calculated using the long term removal efficiency method the following values are found:

	Percent load reduction (lbs/acre)	
Pollutant	Porous pavement	Infiltration trench
OP	85	11
TSP	89	76
TP	87	48
NH4_N	97	92
SKN	88	90
TKN	88	86
OX-N	62	71
TN	76	82
TSS	65	83
Pb	80	83
Zn	94	88

In comparison to the literature review all pollutants of interest were in the very light range. This would be analogous to a rural or light residential classification.

9. Lysimeter pore water analyses proved that conditions in the infiltration trench soil were suitable for the nitrificatrion process.

CONCLUSIONS

In general, concentrations of all pollutants were small in comparison to the concentrations of pollutants from urban areas in the literature. In the cases of orthophosphorus, total phosphorus, and oxidized nitrogen the event mean concentrations were two orders of magnitude smaller than the concentrations found in the literature. Lead and zinc concentrations at the Davis Ford project were also two orders of magnitude less than those found in the literature. This would be analogous to a rural or light residential classification.

Wetfall concentrations of all nitrogen species were highest during the spring months. Extractable lead and extractable zinc concentrations were heaviest during the winter and summer months respectively. Dryfall concentrations of orthophosphorus and total soluble phosphorus were highest during the summer months while all of the nitrogen species were highest during the spring months. Lead was equally high during spring and summer. Extractable zinc was highest during the summer months.

Based on the median load reduction evaluation pollutants were reduced from 54 percent to 96 percent at the porous pavement catchment and from 56 percent to 100 percent at the infiltration trench catchment. Reduction rates were equally efficient based on the long term removal method.

Antecedent dry period analysis proved that beyond several days of pollutant build-up upon a surface there was a point where the build up began to level off and accumulation ceased. For individual parameters soluble Kjeldahl nitrogen ceased to build up after seven days and total kjeldahl nitrogen ceased to build up after about eight days. This effect is caused by air currents which may remove pollutants after they have built up to a particular point.

Soil conditions on the site are favorable for nitrifying bacteria in converting NH4_N to oxidized nitrogen as indicated by the low concentrations of NH4_N and the relatively high oxidized nitrogen values within the soil pore water.

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