Characterization of Performance Predictors and Evaluation of Mowing Practices in Biofiltration Swales

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PREFACE

The values and measurements in this report are presented in the units used in the field and laboratory. In general, these are English units, however, where SI units were used the English equivalent is shown in parentheses. The conversions from English to SI units are given in Appendix 6.

INTRODUCTION

Runoff from urbanized areas, unable to infiltrate through impermeable surfaces, often drains to urban streams or lakes and is recognized as a source of toxins, nutrients, sediment, and other pollutants. The implications to the health of receiving waters of directly discharging stormwater are now better understood, and current regulations require new developments to store and treat their runoff. Common methods of treatment include detention or retention ponds, biofiltration swales, filter strips, and infiltration facilities. This report is focused on identifying the design and maintenance factors that enhance biofiltration swale performance.

Biofiltration swales are characterized as shallow, relatively wide vegetated channels that remove pollutants through vegetation filtration and water retention. Pollutant removal is most dependent on the length of time water remains in the swale (hydraulic residence time) and the amount of contact the flow has with the vegetation and soil surface (Metro, 1992). Fine, dense herbaceous plants, such as grass species, are generally recommended where conditions allow their establishment and growth (Horner et al., 1994).

It is generally assumed that greater grass densities remove more pollutants, and that mowing promotes denser growth. Schultz (1998) found that most vegetation maintenance and mowing practices have been developed from general observations based on these assumptions, despite a lack of research to support this notion. Even without evidence to support the benefits of current practices, mowing operations can make up more than 50 percent of the vegetation maintenance budget in stormwater facilities (Schultz, 1998).

General site maintenance is important for keeping a well functioning swale; however, it is not always easy to determine how well a site is currently functioning. Swale performance can be evaluated by measuring its hydraulic residence time (HRT), because a HRT of at least nine minutes has been found to be most effective at removing stormwater pollutants (Metro, 1992). Yet, determining the performance of an existing facility based on its HRT is also not easily accomplished. The first part of this study, therefore, attempts to develop an easy method for evaluating performance by relating easily measured swale characteristics to HRT.

The second part of this study is intended to remedy the lack of information on stormwater maintenance practices by examining and quantifying the benefit of mowing on pollutant removal in swales. This study also explored whether mowing more than once during the growing season produces better removal efficiencies. Determining the effect of mowing on swale water quality can help guide planning for future maintenance needs and potentially save money that could be used to address other maintenance concerns.

Identifying the practices and characteristics that influence biofiltration swale performance is important to modifying current practices to alternatives that could provide greater benefit. Determining the real value, if any, of mowing for vegetation cover and pollutant removal could save much time, money, and effort without compromising swale performance. In addition, being able to quickly evaluate a swale's current performance based on a few readily measurable features could focus retrofitting efforts on the sites that are likely performing the worst.

CHAPTER 1 – PREVIOUS STUDIES

Extensive urbanization of watersheds has resulted in significant impacts to local stream channels and the quality of receiving waters. These consequences have resulted primarily from the replacement of areas capable of infiltrating rainfall and moderating runoff with impervious cover. Because this replacement is so common, the effects of urbanization on aquatic systems have been studied at length. Efforts to minimize future impacts from uncontrolled runoff have evolved from this extensive information base and resulted in development of methods and regulations for stormwater mitigation.

Biofiltration swales are a common, relatively cheap option for treating stormwater. However, many appear to deteriorate in time if not well maintained (King County, 1995). Maintenance methods have developed over the years, but they are generally based on observations and not actual studies. There has been little systematic research into the effectiveness of maintenance practices for enhancing pollutant removal.

In addition to maintenance practices, design parameters for biofiltration swales are not well based, and so the performance of swales may not match expectations. Design calculations for biofiltration swales use open channel flow equations; however, the poor applicability of Manning's equation, which was developed for deep, wide channels, to the relatively small and shallow biofiltration swales is generally ignored. In addition, a constant resistance coefficient is generally used, despite evidence that it varies with depth.

Stormwater

The significance of stormwater to aquatic systems has grown as a consequence of urbanization. It has even been classified as a "non-point source" because of the pollutants it picks up as it runs over impervious surfaces and dumps into lakes and streams. This section will characterize the effects and quality of stormwater and discuss methods developed for mitigating its impacts to receiving waters.

Watershed Urbanization

The Puget Sound Basin was at one time primarily forested. This provided both a canopy cover to intercept rainfall, which reduced erosion and sediment deposition in streams, and a forest duff layer, which absorbed a large percentage of the runoff and released it slowly to the streams through interflow in the soil (Burges et al., 1989). As the area became developed, forests were cleared, soil was compacted, and impervious area increased. The resulting changes in stream and lake conditions have included:

- increased peak flows and velocities;
- reduction of groundwater recharge which can diminish summer season streamflows (Canning, 1985);
- increased bank erosion, streambed scouring, and sediment movement; enrichment with nutrients;
- elevated levels of suspended solids and heavy metals; higher concentrations of fecal coliform bacteria;
- depressed levels of dissolved oxygen;
- lower survival of salmonid eggs and fry;
- decreased stream production of andromonous salmonids; and
- habitat degradation or destruction (Hubbard et al., 1989; DOE, 1992; Osborne et al., 1988).

These effects of urbanization were evaluated systematically by Bellevue’s Urban Runoff Program (Pitt et al., 1984), which compared the conditions of a relatively undisturbed watershed, Bear Creek, to an urbanized watershed, Kelsey Creek. Kelsey Creek had all the characteristics of an urbanized watershed: its environmental quality was more degraded with higher total suspended solids (TSS) and nutrient concentrations, lower dissolved oxygen concentrations in gravel waters, and higher water temperatures. Only a small unhealthy salmonid population was supported and a shift had occurred from coho salmon to the less disturbance-sensitive cutthroat trout. In contrast, Bear Creek supported a higher level of coho salmon, and generally had more sensitive indicator species.

Non-Point Source Pollution

A non-point source (NPS) is typically defined as pollution that is discharged from diffuse, scattered sources rather than through pipes (PSWQA, 1986). It includes bacteria, sediments, nutrients, and toxicants picked up by rainwater or that are discharged from boats or other water-based sources (PSWQA, 1991). The Nonpoint Source Advisory Subcommittee of the PSWQA determined six major categories of NPS pollution with potential to affect at least some areas of Puget Sound:

1) Agricultural practices, including both commercial and noncommercial farms;
2) Forest practices;
3) On-site sewage disposal systems (septic tanks and drainfields);
4) Urban runoff, including runoff from commercial and industrial sites in both urban and rural areas, and construction erosion;
5) Marinas and recreational boating; and
6) Other concerns: household hazardous wastes, landfills, pesticides, log storage, and ground water (PSWQA, 1986).

Urban runoff's potential to act as a NPS and negatively impact water quality has been recognized more slowly than its recognition as a water-quantity problem that caused extensive flooding. Most early studies that did focus on runoff impacts to water quality addressed agricultural runoff. In 1978 the Environmental Protection Agency established the Nationwide Urban Runoff Program (NURP) to investigate the quality characteristics of urban runoff, its contribution to water quality problems, and the effectiveness of current practices to control pollutant loads from urban runoff (EPA, 1983). The NURP results found that heavy metals, primarily copper, lead, and zinc, are the most prevalent priority pollutant constituents in urban runoff. In some cases these concentrations were higher than EPA ambient water criteria and drinking water standards. Coliform bacteria are generally present in high levels, in most cases exceeding EPA water quality criteria. Nutrients are present but generally not at high levels compared with other possible sources, but they can still be in concentrations high enough to accelerate eutrophication. Total suspended solids are found in high concentrations relative to treatment plant discharges. Oxygen demanding substances are present at concentrations similar to those in secondary treatment effluent, but this condition was rarely viewed with concern since urban runoff is generally well oxygenated and has a sufficient travel time to reduce the oxygen demand before reaching sensitive waters.

The impact of runoff constituents on receiving waters is generally damaging, but varies widely based on the constituent and its concentration. Toxicants in runoff discharged into streams and lakes can become a threat to aquatic life by creating high toxicity levels in the water or by accumulating within the sediments or organisms (Field, 1981) with potential long-term effects. The 1991 Puget Sound Water Quality Management Plan reported that stormwater samples in the Seattle area typically exceed EPA water quality criteria for cadmium, copper, lead, nickel, and zinc (PSWQA, 1991). Bacterial/virus contamination from sources such as pet waste can cause decertification of shellfish beds and closing of swimming beaches (EPA 1983; PSWQA, 1986).

An increase in the use of lawn and agricultural chemical fertilizers over the last few decades has elevated nutrient concentrations in stormwater and accelerated eutrophication in lakes and shallow estuaries (Novotny et al., 1994). Extended eutrophic conditions lead to the depletion of dissolved oxygen in the water, which can cause a shift fish and shellfish populations to poorer quality species (Welch, 1992; Novotny et al., 1994). Phosphorus is the most growth-limiting nutrient in freshwater (Welch, 1992) and its concentration in runoff can often be enough to create eutrophic conditions in lakes (Field, 1981; EPA, 1983).

High concentrations of suspended solids elevate turbidity levels and reduce clarity in receiving waters. The sediment particles can also settle in streams and smother incubating fish eggs, fill the spaces between rocks used for cover by some organisms, and reduce the aquatic insect habitat (PSQWA, 1986).

The short-term effects of increased pollutant loads, which result in poor water quality, are restrictions on contact recreation, stressed aquatic organisms, and damaged shellfish beds and fish habitats. The long-term effects include alterations in the hydrologic regime and corresponding changes in stream morphology, and the accumulation of pollutants in the receiving waters which leads to impacts such as eutrophication, polluted groundwater, and contaminated sediments (DOE, 1992).

Runoff from residential areas will generally have lower concentrations and loadings of most pollutants than commercial areas or highways (Livingston et al., 1997). Compared to a predominately industrial catchment, a predominantly residential catchment’s pollutant exports are approximately half as much (Weeks, 1981). Streets and parking lots will have higher concentrations of heavy metals, petroleum pollutants, and nutrients (Pitt, 1984; Livingston et al., 1997), with busy, high-traffic areas being more polluted in solids, COD, and lead than residential streets and sidewalks (Reinertsen, 1981). Sidewalks were determined to often have higher pollutant concentrations than adjacent roadways and to be a significant source, despite their smaller dimensions (Reinertsen, 1981). Although roads appear to be more important than residential areas for most pollutants, a Bellevue study found that the majority of total solids originated from front and back yards (Pitt, 1984).

Best Management Practices

The term Best Management Practices (BMPs) refers to the actions taken and facilities constructed to control and/or treat urban runoff. DOE defines BMPs as physical, structural, and/or managerial practices that when used singly or in combination, prevent or reduce pollution of water (DOE, 1992). Initially, stormwater management paid little attention to pollution; its focus on flood prevention was accomplished by channelizing streams or placing them in culverts while collecting stormwater in pipes to be discharged directly to streams and lakes (Fok, 1981; PSWQA, 1986). The role of stormwater management has expanded from this traditional role of managing conveyance and flooding to include reduction of NPS pollution and erosion (DOE, 1992; Livingston et al., 1997). In addition, control of the stormwater volume, not just its peak discharge rate, is being required. Peak discharge rate control is evolving further from a single-frequency storm to multiple-frequency storms to minimize erosion of stream channels (Livingston et al., 1997).

DOE (1992) lists BMP types as “source control,” “runoff treatment,” and “streambank erosion control.” *Source control* includes methods such as mulching and covering disturbed soil, putting roofs over outside storage areas, and tracking down and eliminating illicit connections to storm drains. *Runoff treatment* BMPs are facilities that remove pollutants by simple gravity settling of particulate pollutants, filtration, biological uptake, and soil adsorption. *Streambank erosion control* practices moderate the rate, frequency, and flow duration of stormwater into streams. Runoff treatment and streambank erosion control practices include detention ponds, biofiltration swales, infiltration trenches and dry vaults. Ideally, stormwater systems in new developments would be designed so that the post-development rate, volume, timing, and pollutant load do not exceed the pre-development levels. However, this goal is virtually always unattainable because of intrinsic limitations in BMP design and siting (Livingston et al., 1997).

Biofiltration Swales

“Biofiltration” is the simultaneous process of filtration, infiltration, adsorption, ion exchange, and biological uptake of pollutants from runoff as it flows through a vegetated stormwater management system (Livingston et al., 1997). Biofiltration swales, the most common facility for this type of treatment, are generally open, gently sloped, vegetated channels that promote sedimentation and trapping of pollutants by grass blades (Dillaha et al., 1987; King County, 1998). Biofiltration swales are also referred to as grassy swales, bioswales, or biofilters. This section provides an overall picture of biofiltration swales by discussing the general mechanisms for pollutant removal, presenting studies relevant to their performance, and outlining current criteria used in their design.

Removal Mechanisms

Grass swales remove pollutants primarily through sedimentation (Canning, 1985); in contrast, pollutant uptake is not a very important removal mechanism (Horner et al., 1994). Because a large percentage of phosphorus loads (Mulhern et al., 1988), heavy metals, and toxic materials are attached to the sediment particles (Field, 1981), sedimentation is effective at removing these pollutants. This method of treatment generally requires shallow flow depths through vegetation, which retards the water flow by causing a loss of energy through turbulence and by exerting additional drag forces on the water (Kao et al., 1978). As the sediment-laden flow loses velocity, its transport capacity is also reduced (Barfield et al., 1977). The fraction of suspended sediment filtered by a rigid grass media depends on media spacing, flow depth and velocity, sediment concentration and particle size, and section length (Tollner, 1976). Van Dijk et al. (1996) found that the outflow sediment concentration can be described rather simply as a function of the inflow concentration and flow distance over vegetation.

Numerous studies have noticed the accumulation of sediment at the point of entry with a subsequent increase in the water-surface slope at this point, along with an increase in velocity and sediment transport (Tollner, 1976; Barfield et al., 1977; Deletic, 1999). As additional sediment settles out, the zone of deposition moves down the length of the swale (Dillaha et al., 1989). The accumulation of sediment tends to reduce the effectiveness of vegetated filter strips to remove sediment, although vegetation generally will be able to grow through the sediment (Dillaha et al., 1989).

Vegetation in biofiltration should be dense and is generally recommended to include terrestrial grasses and other fine herbaceous plants that will provide dense, uniform cover (Horner et al., 1994). The performance of vegetated filters strips diminishes as the ratio of vegetated to unvegetated area decreased (Magette et al., 1987). In a 1992 study, Metro noted that pollutant removal effectiveness is related to the density and stiffness of the grass blades, which provides a "scrub brush" effect, and to the amount of surface per unit area provided by the individual blades.

Performance Studies

The pollutant-removal performance of swales has been evaluated over several decades with varying results. Metro (1992) completed an extensive study of swale pollutant-removal effectiveness and design characteristics, comparing two similar biofiltration swales of varying lengths (100 feet and 200 feet). The 200-foot swale was found to consistently remove TSS (average 83 percent removal), turbidity (65 percent), and metals (63-72 percent) of largely particulate matter (lead, zinc, iron, and aluminum). About 74 percent of the oil and grease were removed by adhering to grass surfaces. Dissolved metals were less consistently removed. Nutrients were removed in varying rates, with total available phosphorus and total phosphorus (TP) showing the best results at 40 and 29 percent removal, respectively. Poor or negative removals were reported for dissolved nutrients. Removal of fecal coliform bacteria was highly variable.

Evaluation of the characteristics influencing swale performance determined that a hydraulic residence time (HRT) of at least 9 minutes for the 200-foot configuration was needed for good removal of particulate pollutants, oil and grease, and TPH. The HRT for the 100-foot swale was only 4.6 minutes, resulting in poorer and more variable pollutant removals. Manning's n was not found to vary with changes in slope between 3 and 4 percent, but it did decrease with increased flow rate. Variation in Manning's n was also reported with changes in vegetation height, increasing from 0.20 to 0.24 for heights of 6-inches and 12-inches, respectively. A velocity of 0.93 fps was observed to flatten vegetation, changing both its hydraulic characteristics and pollutant-removal performance.

King County Surface Water Management evaluated the condition and effectiveness of 32 biofiltration swales in the Issaquah/East Lake Sammamish basin (King County, 1995). Their evaluation did not involve extensive chemical analyses; instead, it used visual inspections only. The survey found that 28 percent of swales were in good condition, with relatively complete and uniform vegetation cover; 41 percent were considered fair, requiring some repair; and 34 percent were in poor condition, having little vegetation cover or extensive channelization. The swales identified as "fair" were considered to provide some water quality treatment, while the swales identified as poor were considered to provide none. The study observed that most of the swales incorporated into landscaped areas appeared to be more successful than swales hidden away. The most common problems observed were poor vegetative cover resulting from inadequate planting and early care practices (28%), continual saturation from poor grading or excavation below the water table (28%), channelization (15%), and loss of vegetation due to shading (8%). A monitoring of one 350-foot long swale dominated by wetland vegetation found removal rates of 67 and 39 percent for TSS and TP, respectively. However, the site appeared to act as a source of soluble forms of phosphorus. It was observed that soluble nutrients seemed to be retained more effectively in the spring and summer than during the fall and winter. The swale showed poor removal for metals and was a possible source for copper.

Mazer (1998) examined the factors limiting the establishment of good vegetation cover in biofitration swales, motivated by the widely held belief that vegetation cover in swales is a predictor of treatment efficiency. This study monitored eight swales, three of which were regraded and reseeded, and conducted a series of greenhouse experiments comparing four turfgrasses grown under four different moisture regimes. Long inundation periods after seeding resulted in poor vegetation establishment, although some volunteer wetland species did appear in less erosive, shallower areas. The proportion of inundation time showed high negative correlation with plant and organic litter biomass. Vegetation biomass at sites that experience summer drought was strongly dependent on soil depth. In general, biomass in the swales was not well correlated with hydraulic loading rate (HLR). If HLR is a more direct measure of pollutant-removal effectiveness, then vegetation density may <u>not</u> be a good indicator of swale performance.

The greenhouse experiment showed significant differences in growth among grasses. *Festuca arundinacea* (Tall Fescue) had higher biomass accumulation and *Agrotis alba* var. *stolonifera* produced significantly more leaf blades than the other species. Field surveys found *A. alba* was one of the primary species in three of the eight swales, even though it wasn't seeded in any of the swales.

Grass filter strips along highways, an alternative to swales, have been evaluated for effectiveness by several studies. A study of six 53-m roadside swales found that the removal of metals was greater in earthen channels than in grassed ones (Yousef et al., 1985). This result presumably resulted from negatively charged particles in the soil and organic litter that adsorb cations from the water. Phosphorus concentration reduction varied and sometimes increased. Reduced orthophosphorus concentrations were related to a decrease in flow velocity, probably due to a longer contact time between soluble phosphorus and the soil. The removal of heavy metals, nitrogen, and phosphorus species (on a mass basis) was directly related to the infiltration losses through the swales.

Another study of two filter strips located along two different highways (Yu et al., 1995), U. S. Routes 29S and 29N in Virginia, found that TSS concentrations were reduced by 28 percent and 49 percent, respectively. The site along 29S had a slope of 2 percent and achieved "removals" of TP, zinc, and COD of –0.4, 11.1, and –5.6 percent, respectively. In contrast, the 29N site removed TP, zinc, and COD by 33, 13, and 3 percent, respectively, with a 5 percent slope and downstream weir that acted as a checkdam. The report suggests that the large contrast in concentration reductions was a function of the weir, which allowed pollutants to settle out. The report noted that vegetated buffer strips preceded both sites, removing many of the particulate pollutants before the stormwater reached the swales. The buffer strip preceding the swale along 29S reduced concentrations by 64 percent for TSS, -21 percent for TP, 88 percent for zinc, and 59 percent for COD. It was concluded that runoff characterized by large suspended particles can easily be treated by flow through vegetation, but that smaller suspended particles or dissolved pollutants are more difficult to remove by this method.

A study of three 4.6 m x 6.1 m highway vegetated shoulder strips in Washington found that TSS was reduced by 72 percent on average, and that most of the TPH samples were significantly reduced to levels below detection limit of 1 mg/l (Yonge, 2000). Metals showed concentration reductions, but due to problems during collection, the actual value of these reductions could not be determined.

A study in the southwestern United States looked at the use of grass filtration specifically for sediment removal from flood waters (Wilson, 1967). Seven 25-foot wide and 1000-foot long grass strips with a slope of 0.10 percent, each seeded with different grass species, were used as the test plots. Results concluded that grass filtration is an effective economical treatment method for reducing sediment, with removals at low flow between 60 and 95 percent for a flow length of 700 feet. Bermudagrasses appeared to be the most effective at sediment removal. The study also found that application rate, grass height, and particle-size distribution in the incoming sediment affect the effective length of filtration.

Dillaha et al. (1989) compared the effectiveness of 9.1 and 4.6-meter-long vegetative filters in removing sediment, nitrogen, and phosphorus from agricultural runoff. They found that vegetated filter strips were effective for controlling some NPS pollutants, particularly sediment and sediment-bound constituents. With shallow uniform flow, the 9.1 and 4.6 meter strips removed an average of 84 and 70 percent of the incoming suspended solids, 79 and 61 percent of the incoming phosphorus, and 73 and 54 percent of the incoming nitrogen, respectively. The soluble nutrients were found in some cases to be higher than the incoming levels, however, due to lower removal efficiencies and release of nutrients previously trapped in the filter. This study also noted that most of the sediment removed was deposited just upslope or in the first few meters of the vegetated filter strip, and that the zone of deposition seemed to move downstream as areas just upstream became buried. As a result of this sediment accumulation and loss of treatment length, the effectiveness of the vegetated filter strip decreased with time. A good correlation was found between sediment and TP; over 90 percent of the phosphorous leaving the strips was sediment-bound. A similar correlation was found between sediment and sediment-bound total nitrogen (TN), with 66, 65, and 78 percent of the TN leaving plots of 9.1, 4.6, and zero meter (control), respectively.

Biofiltration Swale Design

Poor design of biofiltration swales can lead to undesirable conditions, such as low vegetation cover, standing water, and channelization, and poor pollutant removal. As a result, agencies have developed criteria to guide proper swale design and to prevent construction of poorly functioning swales. The 1998 King County Surface Water Design Manual states that biofiltration swales can be designed for both conveyance and treatment of stormwater flow for areas with less than five acres of impervious surface. It also states that proper design of biofiltration swales should promote uniform flow of stormwater across the entire width of a densely vegetated area. Concentrated flow can make vegetative filtration ineffective for sedimentation and nutrients removal (Dillaha et al., 1986; Dillaha et al., 1987) and can be the most significant factor degrading their performance.

Biofiltration swale sizing is based on several variables, including peak water quality design flow, longitudinal slope, vegetation height, bottom width, side slope, target hydraulic residence time, and design flow depth (Metro, 1992; King County, 1998). Several organizations and agencies have developed design criteria that generally incorporate the same principals, but with some differences in values. Table 1.1 lists the criteria developed in the Pacific Northwest.

Table 1.1. Biofiltration swale design criteria specified by various agencies and studies.

Design Criteria	King County (1998)	Mazer (1998)	DOE (1992)	Metro (1992)
Slope	1-6%: <2%, underdrain required; >6% install checkdams	0.1-1.5%, 1.5-2% requires obstructions (check dams)	2-4%: <2% requires underdrain; >4% requires check dams	2-4%; <2% requires underdrain; > 6% requires check dams
Bottom width	2-16 ft: > 10 ft requires length-wise divider	5 ft	Not specified	2-8 ft
$Length_{min}$	100 ft	200 ft	200 ft	125 ft
Design depth	2 in for frequently mowed grass, 4 in for unfrequently mowed	1.8 in	< 5 in	½ the grass height, up to 3 in; 1/3 height for tall grasses (9-12 in)
Side slope (H:V)	3:1	2:1	3:1	3:1
Design flow	60% of the developed 2-yr peak flow rate* or 2-yr release rate from detention facility**	Not specified	6-month, 24-hour	2-year, 24-hour
Manning's n	0.20	Not specified	0.07	0.20 - 6 in grass; 0.24 - 12 in grass
$Velocity_{max}$	< 1 fps	0.2 fps	<1.5 fps	0.9 fps
HRT_{min}	9 min	15 min	Not specified	5 min; 9 min recommended
Shape	Trapezoid	Not specified	Trapezoid	Trapezoid
HLR_{max}	Not specified	Recommends HLR_{max} of 13 ft/d	Not specified	Not specified

*Flow rate determined from KCRTS model; for application to facilities preceding detention
** Applies to facilities downstream of detention.

The significance of each of the design criteria specified in Table 1.1 to biofiltration swale performance is discussed below.

Slope

A biofiltration swale requires an adequate slope to prevent prolonged inundation, but it cannot be so steep as to create erosive flows leading to channelization. Several studies have noted the limiting effects standing water can have on vegetation establishment and growth (Horner, 1988; Minton et al., 1996; Mazer, 1998). Low slopes can be acceptable where infiltration is a goal, but it should be discouraged in the Puget Sound region where most of the soils are underlain by impermeable glacial till (Canning, 1985).

Bottom Width

Metro (1992) recommends widths no larger than eight feet due to observations that swales with larger widths were more susceptible to channelized flow. King County (1992) maintains a minimum width of two feet to facilitate future maintenance. In a review of studies on grass strips, Van Dijk et al. (1996) found that reduction of sediment discharge varied between 50-60 percent for strips 1 meter wide; 60-90 percent for strips 4-5 meters wide; and 90-99 percent for strips 10 meters wide.

Length

Length is generally chosen based on what is most likely to provide the greatest pollutant removal. At lengths less than 100 feet, channelization is more likely (Metro, 1992; Horner et al., 1994). Several references (King County, 1992; Horner et al., 1994) suggest the following formula for calculating an appropriate length(ft):

$$L = V*t*60 \text{ s/min}$$

Where, V = velocity, fps

t = hydraulic residence time, min

Design Depth

The design depth is determined to ensure that the runoff flows through the vegetation at depths less than the vegetation height. It has been reported that sediment retention decreases substantially when the grass becomes submerged (Van Dijk et al., 1996). Observations of 12 in. high grass note that it does not remain upright when water depths approach one-third its height (Metro, 1992). As it flattens, sedimentation is prevented (Horner et al., 1994).

Side Slopes

Flatter side slopes are generally recommended because they are easier to maintain, reduce erosion potential, and increase conveyance flow area (Metro, 1992). They will also increase the surface area available for treatment. If steeper slopes are required due to area limitations, slope stability may become a concern.

Design Flow

The design flow is generally the maximum flow that a site is designed to treat effectively and is based on the criteria specified in Table 1.1 for the watershed the swale drains.

Manning's n

Manning's *n*, also referred to as the resistance coefficient, is a measure of the flow resistance provided by the channel features and vegetation. A report that completed a comprehensive review of related literature found that for grass channels where water remained below the top of the grass, values ranged from 0.25 to 0.55 (Minton et al., 1996). Unfortunately, flows with such high roughness generally violate many of the hydrodynamic assumptions that were made in the original development of Manning's equation, and so the physical validity or utility of such high *n* values is uncertain. This report questions the conclusions of the 1992 Metro study that found an increase in *n* for less frequently mowed grass. Minton et al. (1996) anticipate that more frequently mowed grass would have a higher stem density and should therefore have a higher *n*, although no data were presented to support this statement. The original report (Metro, 1992) speculated that the higher *n* was due to thatch development as unmowed vegetation accumulated.

Velocity$_{max}$

The flow velocity in the swale should be selected such that erosion is prevented and treatment is maximized. The Metro (1992) study observed that at velocities approaching 0.93 fps the grass was laid flat and corresponding channelization occurred. King County notes that it is desirable to have the design velocity as low as possible to improve treatment effectiveness and to reduce swale length requirements (King County, 1998). A velocity of 1.5 fps was found to permit sedimentation of most particles, except the smallest fractions, found in typical urban runoff (DOE, 1992).

Hydraulic Residence Time

The hydraulic residence time (HRT) refers to the amount of time that water remains in a facility. For biofiltration swales, pollutant removal is most dependent on the HRT and the extent to which it contacts vegetation and soil surface (Metro, 1992; Livingston et al., 1997). The HRT can be calculated for a given flow velocity and swale length.

Shape

Generally, swales are designed as trapezoids because the shape is easier to construct and maintain. However, it has been observed that trapezoidal shapes tend to evolve toward a more parabolic configuration over time (Horner, 1988) without maintenance.

Hydraulic Loading Rate

The hydraulic loading rate (HLR) is the volume of water flowing into a facility per unit area per unit time (Mitsch et al., 1993). This value is not typically used as a basis for design. However, Mazer (1998) concludes that this would be a more appropriate variable to design for than HRT because it accounts for the likely variability between swales in regard to channel length, width, water depths, and inflow discharge rate. He also believed that it indicates the sedimentation potential of the swale.

Underdrains

Underdrains are required in low gradient swales to prevent periods of extended inundation. Maryland's stormwater design manual specifies that they ensure a maximum ponding of 48 hours (Center for Watershed Protection et al., 1997). Construction specifications require they be constructed from six-inch PVC perforated pipe, laid parallel to the swale bottom, and covered with three to six inches of gravel (Center for Watershed Protection et al., 1997; King County, 1998).

Basic Swale Modifications

In cases where longitudinal slope is less than one percent, water tables are high, or there is continuous base flow, wet swales may be appropriate. King County (1998) states that design of these facilities is primarily the same as for design of basic biofiltration swales, except maximum bottom width can be increased to 25 feet and underdrains are not required. The increased treatment area is necessary to compensate for the less dense species that tend to establish in saturated soil conditions (King County, 1998). Maryland allows swale inundation by the water table, but not above the "design bottom" of the channel (Center for Watershed Protection et al., 1997)

In cases where water tables are high, but a wet swale is not desired, liners may be used to prevent the water table from intersecting the swale bottom. King County (1998) provides specifications for liners designed to prevent stormwater in treatment facilities from infiltrating into the groundwater without adequate treatment. However, the same specifications can be applied to prevent groundwater from seeping into swales and inundating them for extended periods when water tables rise.

Vegetation

Grass species are generally the desired vegetation for biofiltration swales because of their dense coverage and finely divided blades, which provide a good filtering effect. The selection of a grass species needs to consider its ability to establish and grow, and its ability to perform in a desirable manner. For sites that have long periods of standing water, Mazer (1998) identified several species of turfgrasses that could establish quickly and successfully in episodically inundated areas. Frequently inundated sites unable to support terrestrial grasses may perform more successfully by establishing wetland vegetation (Metro, 1992; King County, 1995).

However, Minton et al. (1996) noted that a swale with wetland vegetation was less consistent at removing pollutants than grass lined swales. In addition, they noted that past research has shown that performance is affected by stem density, and that wetland species generally provide lower stem density.

Grass height recommendations vary, but are generally recommended between 4-9 in. (DOE, 1992; King County, 1998). Grass heights much more than the recommended tend to bend over, which reduces overall performance.

Swale Placement and Treatment Trains

A treatment train refers to the use of more than one type of stormwater control, which can provide additional treatment and often decreases the overall size and cost of the stormwater management system. Some consider biofiltration swales not to be useful alone but good in treatment trains (Livingston et al., 1997), but there appears to be conflicting views on where to place them relative to detention facilities. Placing swales downstream of detention ponds may degrade their performance ability due to extended periods of water flow (Minton et al., 1996; King County, 1998). Yet placing swales downstream of detention facilities should also minimize potential erosion (Metro, 1992; Livingston et al., 1997).

Maintenance

The appropriate level of maintenance required to keep biofiltration swales working well has not been well studied. Current practices are generally based on observations and the assumption that higher grass density will remove more pollutants (Schultz, 1998). Failure to perform adequate maintenance of stormwater management systems can result in reductions in performance levels and may even create more damaging conditions than if the facility had not been constructed (Livingston et al., 1997). For example, an eroding swale could be the source of excessive sediment. Regular mowing to maintain the desired grass height, especially during the growing season, removal of sediment and debris build-up, reseeding bare areas, and general site repairs are general requirements of most maintenance plans (DOE, 1992; Metro, 1992; King County, 1998). In addition, it is generally recommended to remove the grass clippings to prevent nutrient release during their decomposition and to reduce clogging of outlets and clumping along the bottom. One study recommends mowing for weed control to reduce shading out of desirable species, which could reduce ground cover within the first few inches, and promote thicker grass growth (Dillaha et al., 1986).

In 1998, Schultz completed a study that investigated current maintenance practices in stormwater management facilities. He found a significant lack of information on which practices or vegetation types provide the greatest water quality benefits. Most agencies mow 2-3 times a year, primarily during the growing season, but most do not remove the clippings. Despite existing assumptions and recommendations, it is still unknown exactly what impact these practices have on overall water quality.

Open-Channel Flow

During the end of the last century, many equations to describe open-channel flow were being explored. Currently, however, only a single equation is generally used. This equation is attributed to a single person, R. Manning, despite the fact that it is not in the form originally presented and a number of similar equations were developed around the same time. This equation was developed for large natural stream channel, but it is often applied to any open-channel flow. In addition, its resistance coefficient (Manning's *n*) is often used without considering the accompanying caveats. A brief background of the development of Manning's equation and a review of studies investigating the variability of the resistance coefficient are discussed in the following section. This is presented to better understand the underlying assumptions and validity of applying Manning's equation to biofiltration swale design.

Manning's Equation

In its general form (in English units), Manning's equation for uniform open channel flow is:

$$V = \frac{1.49}{n} R^{2/3} S^{1/2} \tag{1.1}$$

Where, V = mean velocity (fps)

n = coefficient of roughness ($L^{1/6}/T$)

S = energy slope (ft/ft)

R = hydraulic radius (ft), A/P

A = cross sectional area (ft^2)

P = wetter perimeter (ft)

This equation was originally developed for steady uniform flow, although this condition rarely exists in natural streams. However, for practical purposes a uniform-flow condition is frequently assumed in flow computations under normal conditions, which excludes conditions such as the rapidly rising limb of a flood flow or markedly varied flows caused by channel irregularities (Chow, 1959).

It is interesting to note that the equation attributed to Manning was not presented by him in its current form. Several papers have noted that Manning was not the first to present an equation in simple form with the exponents 2/3 and 1/2 for R and S, respectively (Rouse, 1956; Chow, 1959; Powell, 1968; Williams, 1970). The formula $V = lR^{2/3}S^{1/2}$ was first presented by P. G. Gauckler in 1867 (Powell, 1968; Williams, 1970). In 1891, R. Manning presented the same formula, derived from tests completed by Bazin, Kutter, and Ftely and Stearns, but recommended instead $V = C\sqrt{gS}\left[R^{1/2} + \frac{0.22}{m^{1/2}}(R - 0.15m)\right]$, where C is a factor which varies with the nature of the surface, and *m* is the atmospheric pressure in meters of mercury (Rouse, 1956; Powell, 1968). Manning noted that the value of the reciprocal of λ corresponded closely to *n* (Kutter's n), but he never suggest its use (Rouse, 1956).

Roughness Coefficient (*n*)

The *n* coefficient is referred to by several names including "Manning's n," the "roughness coefficient," the "resistance coefficient," and the "retardance coefficient." It is an important and influential part of Manning's equation, and although it is used in biofiltration swale design, few studies have investigated values of *n* for this purpose. Its choice will have the strongest influence over predicted depth in low-flow conditions. An *n* that does correctly reflect the resistance at a site may underpredict the flow depth, creating a less efficient swale where the water is deeper than the vegetation height. Many studies have attempted to characterize *n* and find equations that describe it for particular situations, but no equation has been developed yet for *n* in biofiltration swales.

A discussion of whether or not *n* is a dimensionless term still continues (Rouse, 1956; Chow, 1959), but Rouse states that

> "*n* must continue to be recognized in either system as an absolute (that is, dimensional) rather than a relative (that is, non-dimensional) roughness measure. Under these conditions it must be conceded to have the same dimension as the quantity to which it is related: length to the one-sixth power."

Manning's equation and other similar equations were mainly intended for use in fully turbulent, deep flows (Maheshwari, 1992), although they are often used in the estimation of shallow flow. If the range of depth is not large and the depth of flow is large relative to the roughness element, a constant resistance coefficient generally produces satisfactory results (Harbaugh, 1967). Maheshwari notes that when presenting his equation, Manning mentioned that the value of resistance coefficients should not be taken as constant even for the same channel because it varies with several factors including the size of the channel and hydraulic radius (Maheshwari, 1992). Apparently, most users have ignored this remark and use a constant value. In addition, in turbulent flows the effect of viscosity can safely be ignored and equations such as Manning's can provide adequate results. In low-flow conditions, where both laminar and turbulent conditions may exist, the effect of viscosity may be significant in determining total hydraulic resistance (Maheshwari, 1992). A number of studies have questioned the validity of Manning's equation for shallow flow calculations by showing that the resistance coefficient is not constant: the significance of the roughness elements increases as depth decreases (Ree et al., 1949; Fenzl et al.,1964; Harbaugh et al., 1967; Ree et al., 1977).

The selection of *n* is a very subjective process and never exact. Chow (1959) lists surface roughness, vegetation, channel irregularity, silting and scouring, obstructions, size and shape of channel, stage and discharge, seasonal change, and suspended material and bed load as the factors that exert the greatest influence upon *n* in both artificial and natural channels. Many of these conditions generally will also apply to biofiltration swales. The effect of vegetation on *n* depends on the depth of flow, the percentage of the wetted perimeter covered by vegetation, the density of vegetation below the high-water line, the degree to which the vegetation is flattened by high water, and the alignment of vegetation relative to flow (Arcement et al., 1989). Vegetation provides resistance from stalks, stems, and foliage and blocks out a large proportion of the cross-sectional area when flow is at small depths (Ree et al., 1949).

Many studies have examined the value of *n* for flow through vegetation-lined channels, although primarily for agricultural or erosion applications. In 1949, Ree and Palmer conducted a series of tests on flow at various depths through a variety of vegetation species and heights. The tests were through 1 to 4-foot-wide vegetation lined channels with most slopes between 3-6 percent, and side slopes between 1.5:1 and 4:1. They found that *n* varies widely with species of vegetation, seasonal condition, height, and density of individual species stands. The major result of their tests found that the product of velocity (V) and hydraulic radius (R) holds a certain relationship with *n* suitable for its estimation in submerged vegetation lined channels and independent of slope and length. A number of curves were developed for grasses providing different levels of retardance for use in channel design, including applications to other vegetation species with similar retardance levels.

Cowan (1956) states that *n* is used to indicate the net effect of all factors causing retardation of flow in a reach of channel under consideration. He developed a procedure for estimating *n* based on the recognition of five primary factors modifying the basic value: irregularity of the surfaces on the sides and bottom, variations in cross sectional size and shape, obstructions, vegetation, and channel meandering. This procedure starts with the selection of a basic *n* value assuming a straight, uniform, smooth channel in its natural materials. He recommends a basic value of 0.02 for earth channels. Next, for each of the listed modifying factors, except channel meandering, a value representative of the additional turbulence and retardance produced is added to the basic value. His paper lists a range of appropriate values for each factor. The value for meandering is based on the ratio of meander length of the channel to the straight length of the reach mulitiplied by the sum of the other factors. The value of *n* for the reach is the summation of all the *n* values. Limitations include use on small to moderate unlined channels, with hydraulic radii under 15 feet (Chow, 1959).

Ree and Crow (1977) conducted a series of tests to determine *n* values for vegetated channels of small slope, approximately 0.1%. They found that *n* did not remain constant, but varied with discharge. In addition they noted that for upright vegetation, undisturbed by the flow, *n* was not related to the product of velocity and hydraulic radius. The effect of different vegetation on velocity reduction was related to its shape and its leaf size and distribution. Due to the wide variation in results, it was concluded that the measured values should not be applied directly, unless a situation arose where conditions were exactly the same as during the original measurement.

A relationship for shallow overland flow through non-submerged vegetation was developed to account for the dominance of vegetation blade drag in such conditions (Kao et al., 1978). This study examined the application of Ree and Palmer's earlier results for relatively large flow depths. It was determined that at small depths Ree and Palmer's *n*-VR curves were inappropriate and that channel slope, not considered previously, was a dominating factor. It was also shown that the peak of the *n* value occurs generally at the point when flow is just over the top of vegetation blades, at which point it begins to decrease. The developed equation was based on the momentum balance in the system and determination of the coefficient of blade resistance, plotted in terms of blade width and flow depth Reynold's number. A nomograph method is proposed to provide a simple approach for determining flow depth.

A study of high-gradients streams (slope > 0.002) found a wide variation in *n* with depth (as expressed by hydraulic radius). The value of roughness generally decreased as R increased. It was also determined that slope is a significant factor, with streams of high gradient having a higher value of roughness than similar streams of low gradient. Multiple-regression analyses produced a value of *n* for steep streams, $n = 0.39S^{0.38}R^{-0.16}$. This equation is limited to natural main channels with stable beds and banks, slopes from 0.002 to 0.04, and R ranges from 0.5 to 0.7 feet.

Almost no one has studied the characteristics of hydraulic flow in biofiltration swales, generally assuming the application of open channel flow equations is satisfactory. The above studies indicate that a roughness coefficient would be a factor of flow depth, vegetation stem density and height, and slope. It seems evident that the design criteria need to recognize that *n* varies significantly over small changes in depth and individual site characteristics.

Biofiltration swales are generally designed using a specified *n*, but if this value is significantly different than its actual value, facilities can be over- or under-designed. Based on the steps listed in King County's design manual (1998), bottom width is calculated first, based on Manning's equation. As the chosen value for *n* increases, so does the design bottom width, which in turn results in significant decreases in design length and HRT. Use of an *n* that is too low will cause flow depth to increase above both the design depth and possibly the vegetation height, potentially reducing the filtration effects of the vegetation.

CHAPTER 2 - SITE DESCRIPTIONS

A number of sites were studied and compared in order to account for natural and constructed variability that might impact performance in stormwater swales. All of the sites chosen for the characterization of biofiltration swale performance indicators are located within King County, which were built as part of residential subdivisions' stormwater treatment facilities. As part of these facilities, almost none of the swales are the sole source of stormwater treatment, and they either drain to or from a detention pond. Most are separated from any road and appear to have little human-induced disturbance or damage. Lengths range from 57 to 222 feet and widths between 10.4 and 2.3 feet. Most are generally trapezoidal in shape, but all seem to be rectangular at low depths, probably due to scouring or poor design or construction. Surface bed material ranges from silt/sand to a mixture with very coarse grains and large boulders. Side slopes and surrounding banks are generally grass mixtures with some occurrences of woody and herbaceous vegetation. Rip-rap was found in about half of the sites, but check dams only occurred in two sites. Shading was not a major problem but did occur in a few instances.

Table 2.1 contains site-specific information. Each "Swale Number" in the first column, when preceded by D9, is the official project number given each treatment facility by King County. The location is given as the closest road intersection.

Table 2.1. Site locations and general information.

Swale No.	Location	Length, ft.	Width, ft. (avg.)	Slope, ft/ft (avg)	Bed Material	Other Facilities	Channel Protection	Observed Problems
2434	East KC; SE 35th Way & 212th Ave. SE	120	8.89	0.01	Silt/sand and cobbles	Drains pond	Rip-rap at inlet, check dams	Poor drainage
2135	East KC; SE 35th Pl. & 234th Ave. SE	57	9.80	0.005	Silt/sand	Drains to woods	None	
2335	East KC; SE 26th St. & 226th Ave. SE	150	10.24	0.00315	Silt/sand	Drains pond	None	Poor drainage
1977	North KC; end of 189th Pl. NE (S of NE 183rd St.)	100	15.15	0.03525	Silt/sand	Drains to pond	Rip-rap at inlet	
1703	North KC; NE 194th St. & 218th Pl. NE	119	5.22	0.0081	Silt/sand	Drains pond	None	High % of woody vegetation, low % herbaceous cover

1705	North KC; end of NE 187th (E of 222nd Way NE)	222	4.00	0.0117	Silt/sand and cobbles	Drains pond	Rip-rap at inlet	Poor drainage
1824	North KC; NE 121st St. & 189th Ave. NE	131	4.68	0.057	Cobbles	Drains to pond	Rip-rap at inlet and outlet, check dams	Channeled
1984	North KC; NE 139th St. & 217th Pl. NE	180	7.96	0.01356	Silt/sand and pebbles	Drains to pond	None	Lower half channeled/ downcut
2007	North KC; NE 166th St. & 224th Ave. NE	220	6.53	0.0215	Silt/sand	Drains pond	Rip-rap at inlet	
2008	North KC; NE 160th Pl. & 223rd Ave NE	170	6.60	0.01983	Silt/sand and cobbles	Drains pond	Rip-rap at inlet	Poor drainage
2009	North KC; Saybrook Dr. NE & NE 157th Ct.	176	6.82	0.00358	Silt/sand and cobbles	Drains pond	Rip-rap at inlet	Poor drainage, some shading
2091	North KC; NE 61st St. & 214th Ave. NE	95	6.22	0.0166	Silt/sand	Drains pond (?)	None	Poor drainage
2124	North KC; end of 224th Ave. NE (S of NE 62nd Pl.)	75	4.78	0.01813	Silt/sand	Drains pond	None	
2216	North KC; NE 72nd St. & 245th Way NE	190	7.26	0.001	Silt/sand	Drains pond	Rip-rap at inlet	Poor drainage
1819	South KC; SE 171st Pl. & Fairwood Blvd.	200	3.40	0.03763	Silt/sand and cobbles	Drains to pond	Rip-rap at inlet	Channeled/ downcut, poor drainage
1750	South KC; SE 174th Way & 190th Ave. SE	80	6.20	0.01867	Silt/sand	Drains to pond	None	Poor drainage
1784	South KC; SE 219th Pl. & 120th Ave. SE	100	10.44	0.04967	Silt/sand and cobbles	Drains to pond	Rip-rap at inlet and outlet	Very channeled, poor drainage
1903	South KC; SE 203rd Pl. & 132nd Ave. SE	85	8.21	0.02275	Silt/sand	Drains to woods	None	Poor drainage
1891	South KC; SE 271 St. & 110th Ave. SE	180	2.29	0.00575	Silt/sand	Drains to pond	Rip-rap at inlet	Poor drainage, small slope
1933	South KC; SE 267th Pl. & 164th Ave. SE	203	4.59	0.00905	Silt/sand	Drains to woods	None	Poor drainage in first half

The Pine Lake swale, officially a part of the Todd's Landing Ponds (Project Number D92335), is located on the Sammamish Plateau at the east-end of Pine Lake. It follows two detention ponds and is the final treatment before the stormwater is released into Pine Lake. This site was also included in the swale characterization study, and its specifics are included in Table 2.1.

CHAPTER 3 - APPROACH AND METHODOLOGY

Characterization of Swale Performance Predictors

This study measured the hydraulic residence time, geometry, and several vegetation parameters in order to characterize indicators of swale performance. The first step in this study occurred in late winter 1999 and involved identifying sites that provided varying types of vegetation cover and were close to a fire hydrant. In spring 1999, hydraulic residence time (HRT) tests were performed at each of 20 identified sites. The HRT tests were done at a time when it was assumed that there would be little water in the swale, but that the soil would still be saturated. In the summer of 1999, vegetation assessments were made and soil cores and biomass samples taken at each site. A follow-up site assessment was made in the spring of 2000.

Identifying Sites

A sample population of 20 sites was desired for the characterization of varying vegetation types and coverage. A pool of potential sites was developed by reviewing the results of a King County swale survey (Koon, 1995), which included pictures and site status and description sheets. The list of potential sites was supplemented by comments from a more recent, but less extensive swale survey, conducted by Dalius Gilvydis in 1998. The list of sites was narrowed down to 20 after examining each site in the field for current vegetation conditions, accessibility, hydrant location, and length. During the site examinations, length and width were measured and slope estimated with a hand level.

Fire hydrants were critical to supply water to the swales for the HRT tests, needing to be within 600 feet of the swale inlet. The use of water trucks was considered briefly, but they are not capable of supplying the water capacity needed for this study and would be difficult to negotiate in residential areas.

An attempt was made to choose sites throughout King County to account for any variations in soil composition or maintenance practices and to provide unbiased results. Some clumping within subdivisions did occur for convenience where individual sites showed varying characteristics.

Hydraulic Residence Time Tests

The purpose of the HRT tests was to evaluate the period of time that water, applied at a swales design flow, was in contact with swale vegetation. The design discharge is the rate of water flow that the swale was designed to treat effectively. This value was researched for each swale but in no cases could be found. Therefore, an estimate was made based on the current swale geometry, measured during site identification, and by assuming an optimal design depth of flow of three inches (Metro, 1992). These values were used in Manning's equation with a flow resistance coefficient of 0.20 for grassy swales (Metro, 1992).

The HRT was measured by adding a fluorescent dye to flow inlet and measuring dye concentrations at the outlet against time. The HRT was measured at several sites were checked with existing (non-augmented) flow, but the concentrations were inconsistent and the flow meter (Marsh-McBirney, Inc., Model 2000) readings fluctuated considerably at low flow depths. As a result, all but two sites that were found to have acceptable concentration versus time curves were retested with a hydrant water source, which provided much higher discharges.

The primary materials required to carry out these tests included: 50-foot fire hose segments, Rhodamine WT fluorescent dye, a stopwatch, 125-ml plastic sample bottles, and a hydrant wrench.

Setting up for each dye test involved attaching the water meter to the hydrant, connecting fire hose segments, and attaching a diffuser to the hose end, which prevented scouring, spread the flow across the swale, and most accurately reflected normal operating conditions. Using the water meter to measure the flow into the swale, the gate valve was opened slowly to release the estimated design discharge. Although applied discharges were calculated based on a depth of three inches, the discharge was not adjusted when actual depths in the swales did not meet this criterion. In general, they were close to three inches.

The assumption of saturated soil proved true in every case but one. It was important for the soil to be saturated before carrying out the HRT tests to reduce possible effects of infiltration. The loss of water from infiltration could skew the results toward predicting a longer residence time than actually occurs during most of a site's operation.

Approximately one cap full of Rhodamine WT dye, a biodegradable, non-toxic, fluorescent dye, was added at the inlet or at the point near the inlet most likely to spread the dye evenly over the flow path in the swale. Because this test was based on relative concentrations over time, the exact amount of dye added was not important. The stopwatch was started at the same instant that the dye was added. The dye was generally easy to see from its bright pink color; when it appeared to be approaching the swale's end, sampling was started. Sampling involved dipping sequentially numbered 125-ml plastic bottles at the outlet, just before the water entered a culvert or at the point determined to be the end of the swale. The time on the stopwatch was recorded for each sample taken. Samples were taken for a period of time after dye was visible at the outlet in order to be sure all dye had passed.

After completion of the HRT test and with the water still flowing, depth and width of flow were measured at 10 or 20-foot intervals, depending on the length of the swale. At each interval, a measuring tape was stretched across the water surface and the width recorded. Depth was recorded at half or 1-foot intervals across the swale, depending on width.

The samples were stored in a refrigerator and then analyzed for relative dye concentrations within a day or two of sampling. If not analyzed immediately, the samples were brought back to room temperature to reduce any errors from cool samples, condensation on the culture tubes, or the testing of samples at varying temperatures. The first few samples were analyzed for relative dye concentrations as a function of light absorbance using a Spectrophotometer (Shimadzu UV-1601) with lenses for analyzing Rhodamine WT dye samples. However, the spectrophotometer readings varied excessively for samples with relatively high suspended matter concentrations. The first set of samples with poor curves were saved and tested for fluorescence concentrations using a fluorometer (Turner Designs 10-AU) set up to analyze Rhodamine WT dye samples. The fluorometer provided higher quality results and showed no interaction effects from suspended matter. Therefore, it was used for all remaining sample analyses.

To determine each site's hydraulic residence time (HRT), the relative concentrations (measured as absorbance or fluorescence) were plotted against the recorded times (Appendix 1). The time that had 50% of the area under the curve on either side was determined as the average HRT. The times corresponding to 10% and 90% of the area under the curve were also calculated.

Vegetation Assessments and Soil Samples

Assessment of swale vegetation took place during the summer following the dye tests and began with an inventory of the location and size of bare spots and extended bare channels. They were noted systematically starting at the inlet and moving down the swale, measuring dimensions and recording the location relative to the inlet.

Vegetation assessments were made at each site to gather characteristics that could be related to the determined HRT and potentially be useful for developing a relationship that described functionality of swales. This portion of the study was completed in the summer, when sites were relatively dry and vegetation had flowers and seed heads, making them easier to identify.

Species composition and relative cover were recorded for each site based on the methods employed by Mazer (1998). Two quarter meter squared quadrats were laid side by side, 10 meters (32.8 ft) from the inlet and then every 15 meters (49.2 ft) from the previous station along the length of the swale. At those few swales that were especially short, these distances were modified to include at least two stations. At each station, the species in each quadrat were recorded or sampled for later identification, along with its relative cover, which was determined according to the Daubenmire cover class system (Barbour et al., 1987). In addition, the overall vegetation cover in each quadrat was recorded, also based on the Daubenmire system.

Vegetation cover was recorded at two levels – individual quadrat and individual species within each quadrat. To get an estimate of the area of vegetation cover within each quadrat, the average of the recorded percent cover range was multiplied by 0.25 m^2 (2.69 ft^2). At each site, the calculated areas of vegetation cover were summed and divided by the total quadrat area, equal to 0.25 m^2 (2.69 ft^2) multiplied by the number of quadrats at the site.

The area occupied by each species was computed in a similar manner. The average percent cover was multiplied by the previously computed quadrat vegetation cover, giving the average area covered by each particular species in a quadrat. The areas for each species were summed over all the quadrats in a site and divided by the total vegetated area to give percent of vegetation cover by each species and total cover by each species.

Plant and organic litter biomass samples were taken in each quadrat, following the protocol used by Mazer. A cylinder, measuring 10.5 cm (4.13 in) in height and 11 cm (4.33 in) in diameter, was placed in the center of each quadrat. The standing plant matter within the cylinder (only up to 10.5 cm (4.13 in)) was collected as plant biomass; the organic matter on the ground within the cylinder, was collected as organic litter. Only the plant biomass in the lower height range was collected because it is least influenced by mowing and is most affected by (and most significantly affects) flow. The organic litter provides an indication of the whether the flow through the swale generally has the tendency to carry or to drop out material - the presence of litter indicates lower energy, non-erosive flow. These samples were dried in a 105° C oven and weighed.

Two soil samples between 7 and 15 cm (2.76 and 5.90 in) in depth were taken in each quadrat after the completion of the vegetation assessment at each station. Initially, soil sampling was attempted with a soil corer, but the ground was generally too rocky to get sufficient depth, so a 3-cm-diameter (1.18-in) hand auger was used instead. In all cases, a 15-cm (5.90 in) depth was attempted, but many sites were too rocky to get that deep. The two soil samples from each individual quadrat were combined to provide average, representative samples and air-dried. All samples were tested for organic content by heating a representative portion in a 550° C muffle furnace until no further reduction in weight was detected. The net change in weight provided a measure of the organic content. Each soil sample was also tested for particle size distribution using a standard sieve test (US Standard sieve numbers 10, 40, 80, 140, 200). Generally, only the first and last station samples at each site were analyzed due to time constraints and the high volume of samples.

Additional Data

During the course of this study, additional data were collected as needed. The longitudinal slopes measured in the initial stage of this study were determined to be inaccurate because the hand level used initially was easily misread within small ranges. Since most of the slopes are very small, less than 4 percent, the inaccuracy of the hand level was more pronounced than if the slopes had been more significant. As a result, all slopes were recalculated based on surveyed measurements.

In addition, since the HRT tests were conducted during spring and the vegetation assessments were completed in the summer, the vegetation cover at many of the sites was drastically different at the time of the two visit. In an attempt to gain additional information on how vegetation may influence HRT, each site was visited in the spring of 2000, around the same time as the HRT tests were completed the previous year, and the general vegetation cover noted.

Hydraulic Variables

Hydraulic variables were computed from the following equations:

- Flow velocity, $V = \frac{Q}{A}$ (fps);

- Reynolds Number, $\mathrm{Re} = \frac{Vd}{n}$ (unitless), ν = kinematic viscosity ($1.2\text{x}10^{-5}$ ft^2/s);
- Shear stress, $t = gdS$ (psi), γ = specific weight (62.4 lb_f/ft^3);
- Unit stream power, $P = tV$ (lb_f/ft-s);
- Froude Number, $Fr = \frac{V}{\sqrt{gd}}$ (unitless), g = acceleration of gravity (32.2 ft/s^2);
- Hydraulic loading rate, $HLR = \frac{Q}{Lb}$ (ft/day), b = bottom width

Data Analysis

Scatter plots were examined for relationships between variables. Outliers were identified and investigated for potential exclusion from further analysis because of unique or confounding conditions. Potential correlations were further explored using linear regression analysis and variable transformations to find stronger relationships. Comparison of means for significant ($p<0.10$) differences was done using single factor ANOVAs. Application of ANOVA tests assumes that the data is normally distributed and has the same variance, although it is fairly robust and performs well with deviations from the underlying assumptions (Zar, 1999). Each data set was checked for normality by comparing the distribution of its points against a normal probability plot in SPSS 10.0. The Tukey Honestly Significant Difference multicomparison test (Zar, 1999) was applied where significant differences were detected from the ANOVAs. The single factor ANOVAs were computed on Microsoft Excel 97. All other tests used the SPSS 10.0 statistical package.

The Effect of Mowing Regimes on Swale Treatment Efficiency

This part of the overall study focused on the effects of different mowing regimes on stormwater treatment efficiency in biofiltration swales. Two different mowing regimes were tested which reflect the range of maintenance most commonly performed on these facilities: 1) mowing once early in the growing season and once late in the growing season, and 2) mowing only once late in the growing season. With two treatments and a control, three sites were required. Because this study sought to evaluate the consequences of each treatment, sites had to be almost identical in rainfall, geometry, and vegetation makeup, including plant species composition and cover.

Site Selection

The initial investigation into locating three relatively similar sites was completed by Dalius Gilvydis in early summer 1998. However, he was unable to find three sites that were sufficiently similar. It was decided to use a single, relatively wide site, and divide it into three channels. The Pine Lake swale (Swale No. 2335 from Table 2.1) was chosen because it averaged 10 feet in width and had good vegetation cover.

Site Setup

Creating three channels in the swale was accomplished using four rows of 0.5 inch x 15 inch x 8 foot plywood sheets laid end-to-end on their sides and sunk into the ground about three inches deep. Because the ground was too rocky to hammer in the boards or to dig a trench manually, a power trencher was used to create the channels for boards to be set in. The swale was divided ahead of time into three channels of equal width using strings as guides, but because the ground was so rocky and the trencher heavy and awkward to maneuver, it was difficult to follow the straight line of the strings. As a result, the channels are not exactly equal in width, varying by as much as 35%.

After all of the channel boards were installed, stakes were placed at each location were two boards came together and at a mid-point location along each board to provide stability to the channel walls. Silicon caulk was applied along the seams between boards to prevent leakage between channels and to give additional support.

The three resulting divisions were labeled PL1, PL2, and PL3, from left to right looking downstream. Their respective channel surface areas are: PL1-448.98 ft^2 (32% of total area), PL2-421.88 ft^2 (30%), and PL3-528.18 ft^2 (38%). However, the inundated widths at average flow (“effective widths”) are more equal. This approximate equality occurs because the widest channel (PL3) also has areas along its outer edge that are on higher ground and often not under water. A photograph of the divided swale, with channel designations, is shown in Figure 3.1.

Figure 3.1. Pine Lake swale divided into three channels. Channel designations (from the left) PL1, PL2, and PL3.

Flow Measurement

A preceding wet pond regulated flow into the swale. The flow volume into and out of the swale was measured with a 60-degree V-notched weir at the inlet and a two-foot high H-flume at the outlet, in conjunction with Isco Model 3230 flow meters. The inlet weir, built by Luke Bloedel of King County, is made from plywood and composed of three 60-degree V-notches centered in each channel to distribute the flow equally among channels (Figure 3.2). The invert of each V-notch is 0.43 feet off the ground, ponding the inflow before allowing water to overflow into the channels. The height of the V-notch is 0.75 feet, after which the weir becomes a 10-foot wide rectangular weir. The edges were all covered with aluminum to allow smooth flow and prevent distortions or damage from weathering. The weir was placed across the width of the channel about 10 feet from the location of the inlet pipe discharge. It was also sunk into the soil by digging a channel. It then was lined with geotextile fabric along the ground seams and sealed with bentonite to prevent any seepage. Two-foot long plywood splash plates were placed on the bed of each channel just downstream of the weir to prevent scouring from the concentrated flows over the weir.

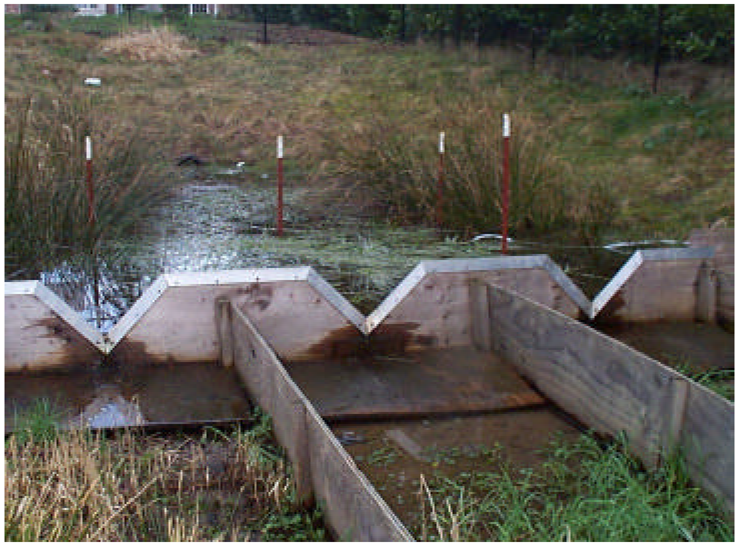

Figure 3.2. Pine Lake inlet weir setup.

Outlet flow was assumed to be equal in each channel and was recombined upstream of a two-foot-high fiberglass H-flume (Figure 3.3). It was laid in the center of the swale, about 150 feet downstream of the weir. Because the flume did not extend the width of the channel, plywood boards were attached at 90-degree angles and extended to the swale banks to prevent flow from passing around the flume. The plywood seams and flume bottom seam was also lined with impervious fabric and bentonite.

Figure 3.3. Two-foot H-flume at Pine Lake outlet.

The flow meters used with the controls at the inlet and outlet were place in sheds on the nearby banks, above the water line (illustrated in Figure 3.1). They were programmed to calculate flow based on a measured head value determined through a bubbler system. This system works by detecting changes in the level of flow by measuring the amount of air pressure required to force an air bubble from the end of a submerged tube (Isco, 1990). As the level of flow increases, so does the amount of pressure required to eject a bubble. Small vinyl tubes (inside diameter of 1/8-inch), carried air between the flow meter and the swale. The tube end in the swale was threaded through a cut-off and capped plastic pipe. The hole through which the tube entered was sealed with caulk to prevent the tube from falling out and to prevent any water from leaking out of or into the tube. The capped pipe provided a reservoir that the tube could use to measure level.

At the inlet, the pipe was cut off so that it was the same height as the bottom of the V-notches and attached to a stake for support. The flow meter was then calibrated so that it recorded zero head when the pipe was just full, and the water was just about to overflow into the channels. As water came in from the ponds, the level above the pipe was the same as the level above each weir. The values programmed into the flow meter use the measured head value with the standard weir equation to translate head into flow volume:

$$cfs = 2.5H^{2.5}$$

Since there were three V-notches, the calculated discharge value was tripled.

The outlet had a similar set up, except the pipe was attached to the outside of the flume for support. A small hole was drilled between the flume and pipe so that the level of water in the reservoir was the same as in the flume. The flow meter had built-in computations for flumes and only required specification of type and size. The flow meter at the outlet was also calibrated to read zero head when no flow was in the flume.

Sampling

The construction required to set up the site caused much more damage than anticipated. As a result, sampling was delayed for a year and to let the grass grow back. During this period, flow records were collected during the first wet season (November 1998 – June 1999), which provided valuable information on flow volumes and guidance in developing the sampling plan for the next year.

In early June 1999 a standard grass seed mix was scattered along the swale, particularly in the bare areas where the check dams had been located. The first mowing did not occur until the beginning of August because the site didn't dry sufficiently until late in the season. The vegetation in the north channel (PL-3) was mowed to about 8 inches, the average height of flow in the channel determined during the HRT study conducted in the spring. In late September, PL-3 was mowed again to the same height. Channel PL-1 was mowed to about an 8-inch height at the same time. Following each mowing, the clippings were removed to the sides where they would not be able to enter the swale. PL2 was not mowed at all.

The samplers were programmed to collect flow-proportional composite samples over a week for testing conductivity, turbidity, and period mean concentrations (PMCs) for total suspended solids (TSS) and total phosphorus (TP). The technique of sampling over an extended time period, instead of sampling an isolated storm event, was used successfully in a road construction monitoring program (Reese, 1997). A week was chosen as the sample period to provide information on the average extended treatment ability of the site, not just its performance during one storm. In addition, the ability to collect a large number of samples by the proposed method compared favorably to storm chasing, allowing better representation of channel treatments.

The flow meters were used as the sampling controllers, signaling the samplers to draw water after a specified volume of flow (flow quantity increment) had either entered or left the channels. The flow quantity increment (FQI), the number of samples to collect, and volume to collect each time, were all developed based on the following considerations: (1) obtaining adequate sample for laboratory analyses, (2) avoiding overfilling the composite containers, and (3) sampling at points spaced sufficiently close to represent the runoff hydrograph relatively well.

The samplers were programmed to collect a maximum of 200 samples of 50 ml each. The flow meters were initially programmed with FQIs of 58 ft^3 and 12 ft^3 at the inlet and outlet, respectively. The lower value at the outlet reflects the loss of volume in the swale to infiltration early in the season. This value was increased gradually to equal the inlet volume when no further loss was occurring. The values were adjusted periodically based on professional judgment and daily King County weather forecasts to best capture the flow over the entire week. The development of the FQIs and sampling procedure are detailed in Appendix 2.

The samplers collected water from the inlet and each channel's outlet into four-gallon carboys, which were replaced each week. The samples in the removed carboys were tested on site for conductivity and turbidity, and then taken to the King County Environmental Laboratory for TP and TSS analysis.

The project budget could cover analysis costs for up to 15 sampling episodes. Therefore, a criterion was developed for determining whether the sample episode should be submitted for analysis or not. Generally, if at least two outlet samplers and the inlet sampler collected successfully then the samples were submitted. In the spring, as the number of samples submitted approached 15, the flow volume for the sample episode was also considered in an attempt to get a number of samples from substantially different flow volume ranges.

At the time the carboys were switched out, the flow data in the flow meter was also uploaded and any spent desiccant replaced. In addition, the water level was checked to verify the reading by the flow meter and any problems were noted and fixed if possible. The deep-cycle marine batteries that powered all the equipment were also exchanged if their charge was running low.

Sample Analysis

The conductivity and turbidity of each sample was tested on-site using a portable Hanna Instruments 9033 Conductivity Meter and a Hach 2100P Turbidimeter. TP and TSS were analyzed at King County Environmental Laboratory using the Standard Methods 4500-P and 2540, respectively (APHA et al., 1992). A Perstop Flow Solution III autoanalyzer and Amsco autoclave (Eagle 10+) was used in TP analysis, and a Mettler AT200 analytical balance for TSS.

Data Analysis

King County Environmental Laboratory reported the inlet and each outlet channel TP and TSS PMCs for each episode. Loads were calculated by multiplying each PMC by the associated flow volume and dividing by the number of sample days. The PMCs and loads in each channel, along with turbidity and conductivity readings, were tested for significant ($p<0.10$) differences in their means using a single-factor ANOVA on Microsoft Excel and SPSS 10.0. Each data set was checked for normality by comparing the distribution of its points against a normal probability plot in SPSS 10.0. The Tukey Honestly Significant Difference multicomparison test (Zar, 1999) was applied to determine which channels had significant ($p<0.10$) differences in their means.

Computation and analysis of TP and TSS mass loadings provides the most accurate account of what is coming in and leaving the swale. By only looking at concentrations, any infiltration losses are not considered. Substantial infiltration will reduce the flow volume and overall pollutant load leaving the site, even though concentrations may not be that different from inflow concentrations.

CHAPTER 4 - RESULTS

Characterization of Swale Performance Predictors

The measured HRTs covered a wide range of values, from 3.0 minutes to 19.8 minutes, due to variations in applied discharges and in swale width, length, and slope. The depth and width of flow measurements recorded at intervals along the length of the channel were used to compute cross-sectional areas. The average of the areas was applied to the continuity equation to calculate the average water velocity during the test:

$$V = \frac{Q}{A} \qquad (4.1)$$

Where, V = average velocity (fps)

Q = average discharge (cfs)

A = average cross sectional area of flow (ft^2)

Using the values discussed above, Manning's *n* was calculated by rearranging Manning's equation for open channel flow. Values ranged from 0.19 to 0.53. The results of the dye test and the above calculations are given in Table 4.1.

Table 4.1. Calculated HRT and measured geometry and hydraulic variables.

Swale No.	Measured Discharge (Q_m), cfs	HRT_m, min	Average Depth (d_m), ft.	Average Width (w), ft.	Average Area (A), ft^2	Average Velocity (V_m), fps	Roughness Coefficient (n), unitless
1703	0.30	16.20	0.28	5.22	1.45	0.21	0.27
1750	0.45	3.00	0.34	6.20	2.11	0.21	0.45
1784	0.50	4.00	0.21	10.44	2.24	0.22	0.53
1819	0.30	6.60	0.21	3.40	0.72	0.42	0.24
1824	0.33	5.40	0.19	4.68	0.89	0.37	0.31
1903	0.51	4.70	0.23	8.21	1.92	0.26	0.32
1933	0.38	19.80	0.34	4.59	1.58	0.24	0.28
1977	0.80	7.60	0.17	15.15	2.52	0.32	0.28
1984	0.58	9.20	0.38	7.96	3.04	0.19	0.44
2007	0.79	7.50	0.30	6.53	1.98	0.40	0.24
2009	0.42	10.00	0.31	6.82	2.14	0.20	0.19
2124	0.30	3.30	0.20	4.78	0.97	0.31	0.23
2335	0.77	15.00	0.39	10.24	4.02	0.19	0.23
2434	0.62	8.20	0.38	8.89	3.39	0.18	0.41

Problematic Sites

Initial review of the data revealed a number of potential irregularities. These were investigated further before any extensive analysis was completed. Six sites were identified: 2135, 1705, 2008, 2091, 2216, and 1891. They were excluded based on the following rational:

- 2135 – The actual slope during the HRT test was unavailable due to regrading and site modifications that occurred after the HRT test but prior to Spring 2000. Slope has a significant impact on calculated values, and therefore the preliminary, inaccurate slope measurement invalidated this site's data.
- 1705 and 2008 – These two sites were not retested using flow from a hydrant because their pattern of dye concentration versus time from the initial tests, which used the existing flow, looked reasonable. However, discharge values measured with a current meter at such a low flow depth were found to be generally inaccurate.
- 2091 – The dye test flow rate (0.17 cfs) was very small, especially relative to the swale's design flow rate (0.56 cfs). With such a small velocity, the calculated 'n' was unreasonably large and skewed the scaled HRT.
- 2216 –The dye test (0.42 cfs) was run at about two and a half times its design flow (0.16 cfs) and was not able to be scaled to give a reasonable design HRT.
- 1891 – There was a hole in one of the hoses that lost a significant and unmeasurable amount of water, invalidating the measured discharge.

Scaling Measured HRTs

Dye tests were all run at values considered near each site's calculated design flow. However, use of inaccurate initial slope measurements in calculating the design flows, and field complications that precluded certain flows, resulted in most sites not being tested at design flow. As a result, the measured HRTs are not valid for direct comparisons between swales. Therefore, a method was developed to scale up or down the measured HRTs to a value closer to their actual value at each swale's design discharge.

An initial investigation was made into the relationships between the factors that might influence HRT. If the median value of the dye travel time reflects the average travel time of the water down the swale, then the following relationship should hold:

$$HRT = \frac{L}{V * 60} \qquad (4.2)$$

Where, HRT = hydraulic residence time (min)
L = length (ft)
V = average flow velocity (fps)

To determine how closely this equation predicts actual values, the measured HRTs (HRT_m) were plotted against the HRTs calculated from equation 4.2 (HRT_{calc}) (Figure 4.1). A 1:1 plot line was also included for reference.

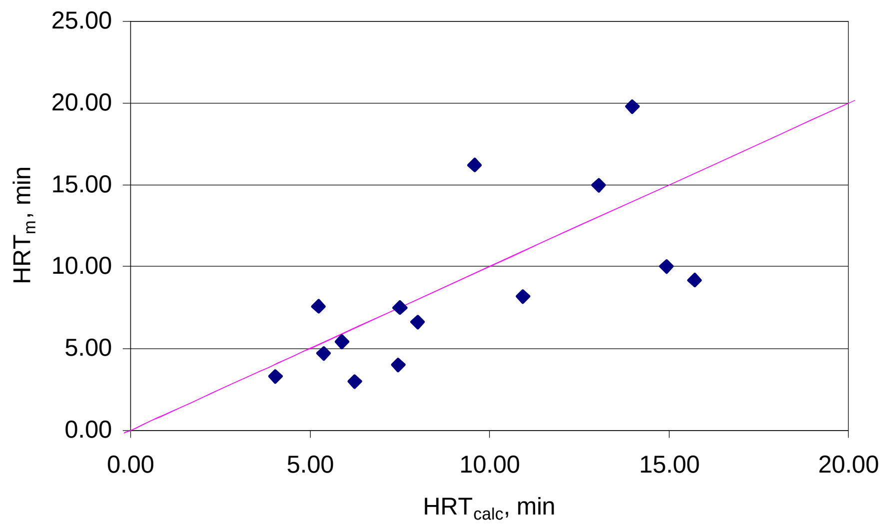

Figure 4.1. Measured HRT against HRT calculated from equation 4.2 (R^2=0.683, R_a^2=0.423).

Figure 4.1 supports reasonably well the expectation of an HRT relationship to velocity scaled by the swale lengths shown in equation 4.2. As a result, scaling the measured HRTs for the design discharge (HRT_d) was done by applying a scaling factor based on velocity. The design discharges (Q_d) were recalculated using Manning's equation ($Q_d = \frac{1.49}{n} AR^{2/3} S^{1/2}$) with *n* equal to 0.2, and depth equal to 0.25 ft., and surveyed slope values instead of the initial hand-level estimates were used, producing a more accurate number. Using this new design flow with the *n* values tabulated in Table 4.1, a rearranged Manning's equation was iteratively solved for design depth (d_d). Applying the continuity equation (Equation 4.1) with this calculated design depth and discharge, a design velocity (V_d) was calculated.

The ratio of V_m to V_d was computed and used as a scaling factor. HRT_d was computed from the product of HRT_m and the scaling factor. The measured *n* values, slope, measured and design velocities, measured and design depths, length, and measured and design HRTs are presented in Table 4.2.

Table 4.2. Scaled HRTs.

Swale	n	Slope (ft/ft)	V_m (fps)	V_d (fps)	$Depth_m$ (ft)	$Depth_d$ (ft)	Length (ft)	HRT_m (min)	HRT_d (min)
1703	0.27	0.0081	0.21	0.20	0.28	0.31	119	16.20	16.68
1750	0.45	0.01867	0.21	0.23	0.34	0.41	80	3.00	2.75
1784	0.53	0.04967	0.22	0.35	0.21	0.45	100	4.00	2.53
1819	0.24	0.03763	0.42	0.45	0.21	0.29	200	6.60	6.13
1824	0.31	0.057	0.37	0.51	0.19	0.32	131	5.40	3.90
1903	0.32	0.02275	0.26	0.32	0.23	0.33	85	4.70	3.83
1933	0.28	0.00905	0.24	0.21	0.34	0.31	203	19.80	22.72
1977	0.28	0.03525	0.32	0.45	0.17	0.3	100	7.60	5.36
1984	0.44	0.01356	0.19	0.20	0.38	0.41	180	9.20	8.75
2007	0.24	0.0215	0.40	0.37	0.3	0.28	180	7.50	8.16
2009	0.19	0.00358	0.20	0.17	0.31	0.25	176	10.00	11.69
2124	0.23	0.01813	0.31	0.33	0.2	0.28	75	3.30	3.08
2335	0.23	0.00315	0.19	0.15	0.39	0.27	150	15.00	19.40
2434	0.41	0.01	0.18	0.18	0.38	0.4	120	8.20	8.46

A plot of design HRT against slope (Figure 4.2) revealed an inverse relationship with a suggestion of clustering of the data into two groups: swales with slopes less than 1.0% and HRTs at or above nine minutes, and slopes greater than 1.0% with HRTs less than nine minutes.

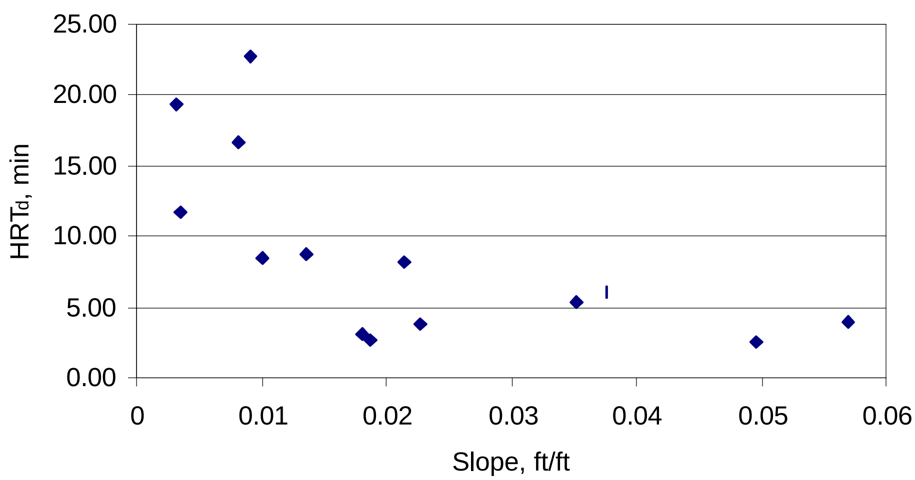

Figure 4.2. Relationship between design HRT and slope.

An ANOVA of the data revealed slope having a significant (p=0.001) influence on HRT, with only slopes less than 1.0 percent achieving an HRT greater than 9 minutes.

Vegetation Assessment

The results of the sieve analysis (Table 4.3) indicate that the majority of the soils were sandy loam, with the rest falling into the closely related category of loamy sand.

Table 4.3. Average site soil composition.

Swale	Soil Composition (Percent)			Total Not Gravel	Composition of Non-gravel Soil (percent)		Soil Type
	Gravel	Sand	Silt/clay		Sand	Silt/clay	
1703	9.0	52.6	38.4	91.0	57.8	42.2	sandy loam
1750	13.6	61.0	25.4	86.4	70.6	29.4	sandy loam
1784	17.6	57.9	24.5	82.4	70.2	29.8	sandy loam
1819	20.6	63.7	15.8	79.4	80.2	19.8	loamy sand
1824	6.6	63.0	30.5	93.4	67.4	32.6	sandy loam
1903	3.4	54.9	41.7	96.6	56.9	43.1	sandy loam
1933	23.2	57.7	19.2	76.8	75.1	24.9	loamy sand
1977	28.9	56.6	14.5	71.1	79.6	20.4	loamy sand
1984	15.6	50.5	33.9	84.4	59.8	40.2	sandy loam
2007	24.9	58.2	16.8	75.1	77.6	22.4	loamy sand
2009	14.7	61.1	24.1	85.3	71.7	28.3	sandy loam
2124	23.1	55.0	21.9	76.9	71.5	28.5	sandy loam
2335	23.8	53.8	22.4	76.2	70.6	29.4	sandy loam
2434	11.3	58.8	29.9	88.7	66.3	33.7	sandy loam

Biomass, soil organic content and percent bare areas are given in Table 4.4. Plant biomass, organic litter, and total biomass average dry weights per area range between 1.3 to 8.4 g/m^2 (0.0003 to 0.0017 lb/ft^2), zero to 11.0 g/m^2 (zero to 0.0022 lb/ft^2), and 4.1 to 18.4 g/m^2 (0.0008 to 0.0038 lb/ft^2), respectfully. Average percent soil organic content at each site ranges from 0.4 to 5.7 percent. The percent of spot bareness and channel bareness, calculated from the size of each bare spot or channel recorded during the vegetation assessment, vary between zero to 96.7 percent and zero to 54 percent, respectively.

Table 4.4. Biomass, percent organics in soil, and percent bare areas measured.

Swale	Plant Biomass (g/m^2)	Organic Litter (g/m^2)	Total Biomass (g/m^2)	Soil Organics (%)	Bare Areas (%)	Bare Channel Area (%)
1703	4.1	11.0	15.1	1.1	10.1	54.0
1750	4.0	0.0	4.0	0.4	1.0	4.1
1784	8.4	10.0	18.4	3.9	0.0	13.4
1819	4.8	1.0	5.7	2.8	0.0	9.6
1824	6.0	0.0	6.0	2.7	0.0	21.0
1903	5.2	3.8	9.0	5.7	0.0	0.0
1933	4.2	2.1	6.3	3.9	0.0	0.0
1977	4.9	1.1	6.0	1.9	2.3	0.0
1984	5.4	4.9	10.3	2.6	13.7	12.0
2007	5.3	6.8	12.1	2.4	0.5	0.0
2009	6.2	9.0	15.2	3.7	0.0	0.0
2124	3.1	1.1	4.2	1.9	4.2	0.0
2335	5.5	3.4	8.9	2.4	3.0	4.8
2434	1.3	4.9	6.2	1.3	96.7	0.0

Overall percent cover at each site ranged from 10 to 94 percent. However, the vegetation assessment took place during summer months, which can produce very different vegetation cover conditions than in the spring when the HRT tests were done. Vegetation cover was not recorded during HRT testing, but average percent cover for the site was recorded during site visits in spring 2000, around the same time as the tests were done the previous year.

Percent cover varied widely in both spring and summer, but in general summer coverages were similar or higher than in the spring. A few sites showed marked improvements from spring to summer. Spring coverage, which probably reflects wintertime conditions most closely, varied widely: over one-third of the swales (5 of 13) had less than 50 percent cover, but an equal amount had greater than 75 percent cover. Table 4.5 presents the contrasting vegetation cover between spring and summer.

Table 4.5. Comparison of vegetation covers in the spring and summer.

Swale	Average Spring Percent Cover	Average Summer Percent Cover
1703	15	28
1750	98**	94
1784	38	62
1819	15	68
1824	63	64
1903	85	68
1933	63	86
1977	85	91
1984	38	42
2007	85	81
2009	15	75
2124	85	83
2335	63	56
2434	N/A*	10

* Not able to quantify due to recent revegetation, but previous observations noted similar percent cover between seasons.
** Observed approximately 10% cover previous year.

A plot of the summer and spring percent cover against slope (figure 4.3) examines the data in Table 4.5 from another perspective and suggests a correlation between vegetation cover and slope ranges. Swales with slopes between 1.5-2.5% slope (site nos. 2124, 1750, 2007, and 1903) retained the most consistent cover between the spring and summer, with most sites at least 75-percent covered. Swales with slopes less than 1.5% (site nos. 1703, 1933, 1984, 2009, 2335, and 2434) all had spring cover less than 75 percent, with only two swales varying widely between seasons and having greater than 75 percent cover in the summer. Only one swale of four swales with greater than 2.5 % slope (Site nos. 1784, 1819, 1824, and 1977) retained cover greater than 75 percent during both seasons. The other three swales had less than 75 percent cover during both seasons.

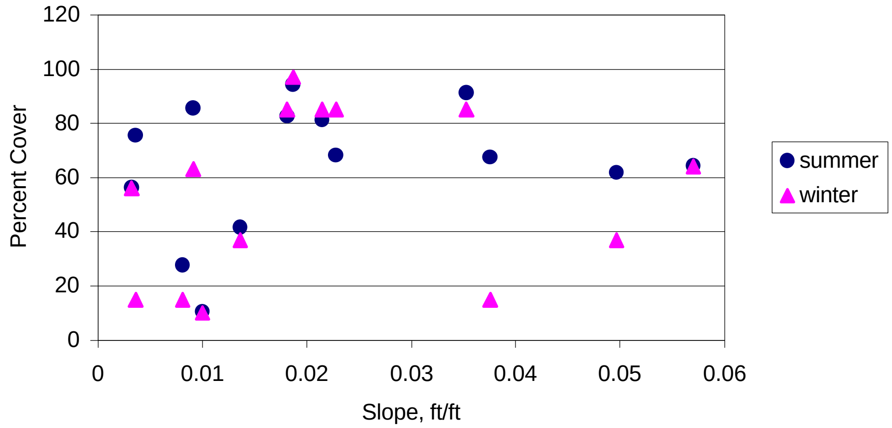

Figure 4.3. Percent summer and spring vegetation cover relative to slope.

ANOVA and Tukey tests revealed a significant difference between percent cover (squared to induce normality) within slope categories, but only between swales with slopes less than 1.5% and slopes 1.5-2.5% for both summer and spring (p=0.0927 and p=0.002, respectively) and between swales with slopes greater than 2.5% and slopes 1.5-2.5% for spring (p=0.025).

ANOVA and Tukey test results also revealed that slope is inversely related to organic litter and above ground biomass (both natural log transformed to induce normality), but only between slope categories less than 1.5% and greater than 2.5% (p=0.042 and p=0.066, respectively).

The vegetation assessment recorded 53 different species over all sites. *Agrotis spp.* (bentgrass), *Holxus lanatus* (velvet grass), *Festuca arundinacea* (tall fescue), and *Ranunculus repens* (Buttercup) were most prevalent, occurring in 55, 26, 26, and 19 percent of the quadrats, respectively. *Agrotis spp.* appeared in 12 of 14 swales, and *R. repens* and *H. lanatus* were present in seven of 14 swales. All other species occurred in less than five swales. Mazer (1998) also found *Festuca arundinacea, Agrotis spp*. and *R. repens* in high frequencies.

Agrotis spp. was most abundant, covering the greatest percent of the total area in the quadrats, 17 percent, followed by *Alopecurus aequalis* (Shortawn foxtail) and *Agropyron repens* (Quackgrass) with 9 and 6 percents, respectively.

Results from the six problematic sites found high frequencies of *Juncus effusus* (common rush), *Typha latifolia* (cattail), *Agrotis spp.*, and *Eleocharis palustis* (creeping spike-rush), occurring in 36, 33, 28, and 28 percent of the quadrats, respectively. The most abundant species were *J. effusus*, *Callitriche heterophylla* (diversed-leaved aquatic-starwort), and *Agrotis spp.*, covering 60, 29, and 25 percent of the total area in the quadrats.

Species occupying more than 10 percent of an inundated site were identified as water tolerant plants. The following species fit this classification: *Sparganium angustifolium* (narrow-leaved bur-reed), *Agrotis spp.*, *A. repens*, *R. repens*, *Phalaris arundinacea* (reed canary grass), *C. heterophylla*, *J. effusus*,and *F. arundinacea*.

Each site's species richness and individual species frequency is tabulated in Appendix 3, along with percent vegetation cover and percent cover of each species. No correlation existed between species richness and percent cover. No significant difference was found for species richness between slope categories, but over twice as many species provided greater than 10 percent cover in swales with slopes less than 1.5% than in any other slope category.

Data Analysis

ANOVA tests determined that slope had no effect on *n*, percent passing the 200 sieve (silt/clay soil fraction), and the percent of channel development and spot bareness in the summer. Observations noted that although no difference in percent of bare area from channelization was detected, the channels in swales steeper than 2.5% tended to be longer and convey more concentrated flow than those in the low grade swales, which tended to be extended bare areas. The average channel length in sites with slope greater than 2.5% was 53 feet compared with 16 feet in sites with less than 1.5% slope. A chi-squared goodness of fit analysis ($p<0.10$) detected no significant differences in summer water presence between slope categories. Water was present in one of four swales within the 1.5-2.5%, in two of four swales greater than 2.5% slope categories, and in four of six swales with less than 1.5% slope. Although any water in steeper swales appears to be a result of continued baseflow, water in flatter swales is more a result of poor drainage.

The percent of silt/clay material in the soil was not correlated to either spot or channel bareness or standing water in the summer. Plant biomass had no relationship to flow resistance.

One of the primary focuses of this study was to determine factors influencing swale performance or ability to remove pollutants by investigating the variables that might be anticipated to control HRT. Obviously swale length and flow velocity influence HRT, but the only other significant relationship with HRT was slope. No correlation was found between HRT and the following variables that might be expected to influence HRT, or that could be used as a surrogate to characterize HRT more easily: coefficient of flow resistance, percent vegetation cover, species richness, depth of flow, density of organic litter or above ground biomass, and percent bare area.

Regression Model

A goal of this study was to develop a model to predict a site's HRT as an indicator of swale functioning, based on measurable variables. It was expected that such a model would be in part a function of percent cover; however, no correlation existed with HRT. Development of a regression equation used measured values that correlated with the measured HRT. The final model was chosen based on the R_a^2 and $PRESS_p$ values. The R_a^2 is a better measure of the percent of variation in the response variable explained by the predictor variables because it adjusts for the number of parameters and can only increase if the amount of error decreases (Neter et al, 1996). The $PRESS_p$ is a measure of how well the fitted model would predict the measured response variables (Neter et al, 1996). Large R_a^2 and a small $PRESS_p$ values indicate good candidate models. The following equations were determined as best fits:

$$HRT = 0.025 L^{0.904} S^{-0.311} \qquad R^2=0.824,\ R_a^2=0.621 \qquad (4.3)$$

$$HRT = 0.014\left(\frac{L}{V}\right)^{1.003} \qquad R^2=0.749,\ R_a^2=0.525 \qquad (4.4)$$

Equation 4.3 describes a functional relationship for HRT, while equation 4.4 provides a mechanistic relationship between HRT and velocity.

Hydraulic Variables

Reynolds Number (Re), shear stress (τ), unit stream power (P), Froude Number (Fr), and hydraulic loading rate (HLR) were calculated for each site based on the measured values. No significant relationships were found between these variables and other swale characteristics that might explain swale performance.

Reynolds Number ranged from 3900 to 9974, well into the turbulent range. Shear stress provides a measure of erosion potential and ranged from 0.07-0.68 psi. Unit stream power varied very little between sites, with values from 0.014 to 0.025 lb_f/ft-s. Froude numbers well under 1.0 classified all flows as subcritical, as would be expected given the low velocities. Swale Froude numbers ranged from 0.052 to 0.160, indicating low erosive potential. Hydraulic loading rates, 0.897 to 1.004 ft/day, are much less than Mazer's (1998) recommended maximum of 13 ft/day.

Pond Influence

Swales are often used in treatment trains, but there is no clear consensus on whether to locate swales upstream or downstream of detention ponds. Nine of the swales in this study (including problematic sites) followed a detention pond. Eleven stood alone or preceded a detention pond. None of the swales draining detention ponds showed major signs of erosion or channelization, but seven had either flowing or standing water during the spring and four never dried out in the summer. Five of the swales receiving runoff directly had channelized areas, nine had flowing or standing water in the spring, and five contained water in the summer.

One consideration is that ponds preceding swales dampen the flow and reduce erosion. This would appear to be the case in this study, since all of the channelized swales receive runoff directly. However, of the five swales with channelization, the three with the greatest degree of channelization also had slopes greater than 2.5 percent, suggesting that slope, not direct discharge, could be the primary cause of channelization. The other side of the argument suggests that placing ponds before swales can create long-term saturation conditions as the pond drains slowly. This theory is also not well supported by the studied swales, since approximately equal numbers of sites draining ponds as preceding ponds did not dry up in the summer.

The Effect of Mowing Regimes on Swale Treatment Efficiency

Sample Collection

Fifteen sample episodes, out of 20 collected, were submitted to the King County Environmental Laboratory for TSS and TP analysis. The decision to submit was based on having an adequate sample volume in the inlet carboy and samples from at least two of the outlet carboys. Only four episodes lacked one outlet sample: PL1 missed one episode, PL2 missed 2 episodes, and PL3 missed one episode, generally as a result of power failure. Each battery's voltage was checked once a week, but some of the equipment, particularly the flow meters, required the batteries to be charged to at least 10 volts. Some of the earlier missed samples resulted from the displacement of the tubing that delivers the water. Over time, the pumping action appeared to cause the tube to come out of the carboy mouth and fill the sampler bottom instead. Checking the tubing positioning at each visit eliminated this problem.

Sampling episodes generally lasted a week and covered a variety of flow volume ranges. The site became saturated quickly, allowing for very little infiltration after the first week of testing. It remained wet for the duration of the sampling period. Vegetation was not significantly affected by the continual saturation, except in a depression about two-thirds down the channel length which became a constant pool of water. The depression resulted from the removal of a checkdam.

As the number of sample episodes collected for analysis approached 15, an attempt was made to balance the distribution of samples from different flow volume ranges. The number of sample episodes collected per flow volume range is given in Table 4.6.

Table 4.6. Number of sampling episodes per flow volume range.

	Number of Samples	
Volume (cf) Range x 1000	Inflow	Outflow
0 – 10	3	4
10 – 20	5	7
20 – 30	4	2
30 – 40	0	2
40 – 50	3	0
Total	15	15

The details of each sampling period, including number of samples drawn by the samplers, the dates of the sample episode, and the episode volume, are given in Appendix 4. Some of the early sample episodes did not span a week as planned, but instead collected full samples within a few days. The sampler's program allowed a maximum of 200 samples to prevent overflowing; however, during periods with heavy rainfall and a FQI set too low, all the samples were drawn within a shorter period of time. The episode volume for these instances was calculated only for the period over which the sampler took the 200 samples.

The initial FQIs used in this study are discussed in the methodology section and Appendix 2. The FQI was adjusted based on weather predictions and/or if the current value resulted in either too few samples or the samples being taken too quickly. The resulting FQIs and their corresponding sampling period are listed in Table 4.7.

Table 4.7. Programmed FQIs and associated sample periods.

Sample Period	FQI, ft^3 – Inlet	FQI, ft^3 – Outlet
10/28/99 – 11/4/99	58	12
11/4/99 – 11/10/99	58	44
11/10/99 – 11/23/99	115	86
11/23/99 – 12/21/99	230	172
12/21/99 – 1/10/00	115	86
1/10/00 – 1/24/00	230	172
1/24/00 – 2/7/00	115	86
2/7/00 – 3/14/00 (end)	230	230

Data Analysis

Appendix 5 gives each episode's average TSS and TP PMCs for the inlet and three channels, along with corresponding flow volumes and number of sample days. TSS inlet concentrations ranged from 0.9 to 17.2 mg/L, with a mean of 2.9 mg/L and a median of 1.8 mg/L, and outlet concentrations ranged from 2.3 to 96.1 mg/L, with a mean of 14.8 mg/L and a median of 10.7 mg/L. TP inlet concentrations ranged from 0.0066 to 0.0546 mg/L, with a mean of 0.0212 mg/L and a median of 0.0163 mg/L, and outlet concentrations ranged from 0.0117 to 0.0863, with a mean of 0.0322 mg/L and a median of 0.0289. For comparison, average residential runoff median event mean concentrations for TSS and TP are 101 mg/L and 383 μg/L (EPA, 1983).

The TSS and TP loads for each episode were calculated by multiplying each concentration by its corresponding flow volume and dividing by the sample days. Figures 4.4 and 4.5 present the inlet and outlet TSS and TP loads (g/day) for each episode, showing the unexpected result that a majority of the inlet loads were less than the outlet loads.

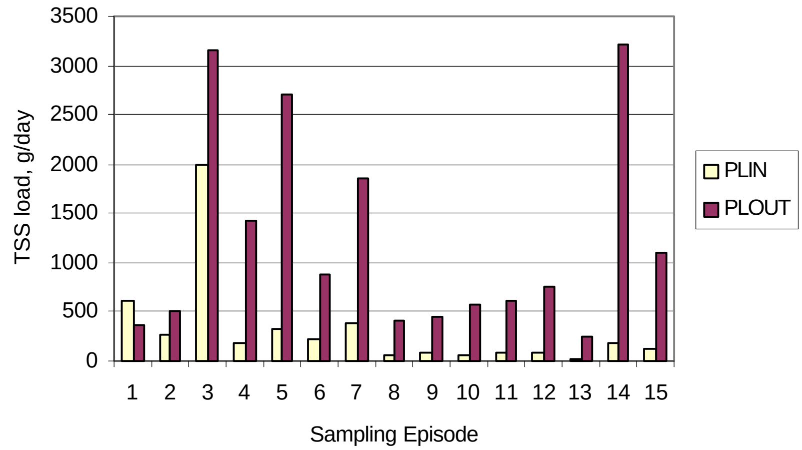

Figure 4.4. TSS inlet and outlet loads.

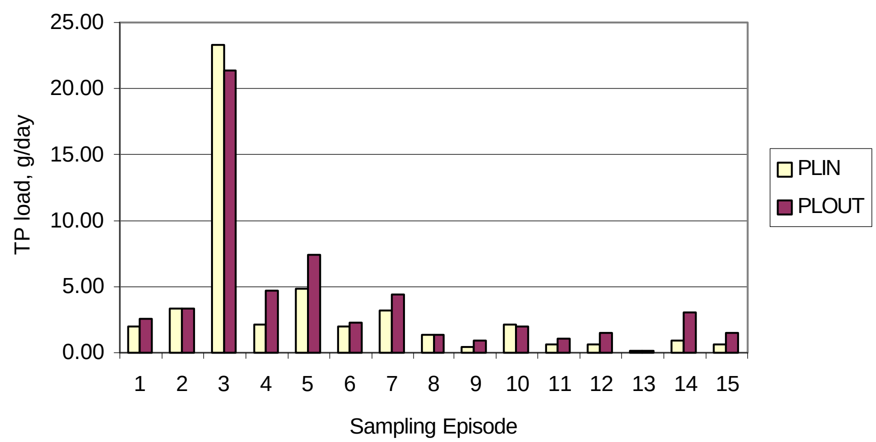

Figure 4.5. TP inlet and outlet loads.

Turbidity and conductivity readings for the inlet and three channels, given in Figures 4.6 and 4.7, show that in general turbidity increased in the channels while little change occurred in conductivity. Turbidity inlet values ranged from 1.19 to 8.01 NTUs, with a mean of 2.6 NTUs and a median of 2.04 NTUs, and outlet values ranged from 2.41 to 42.73 NTUs, with a mean of 7.4 NTUs and a median of 5.51 NTUs. Conductivity varied between 37.7 and 128.1 μS/cm at the inlet, with a mean of 90.9 μS/cm and a median of 95.9 μS/cm, and values between 36 and 127.7 μS/cm at the outlet, with a mean of 91.0 μS/cm and a median of 95.8 μS/cm.

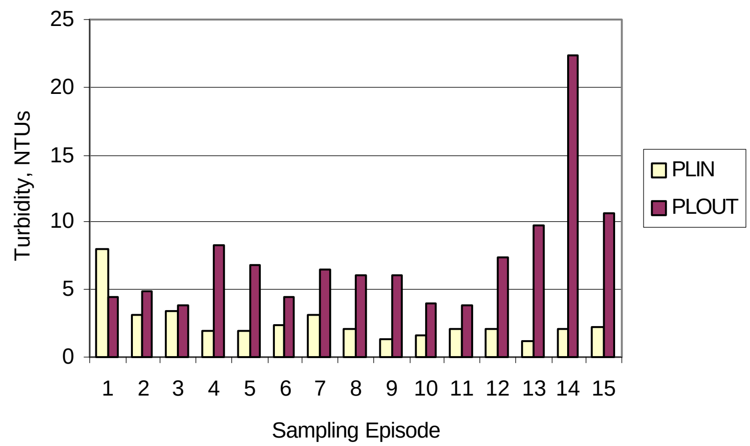

Figure 4.6. Turbidity inlet and outlet values.

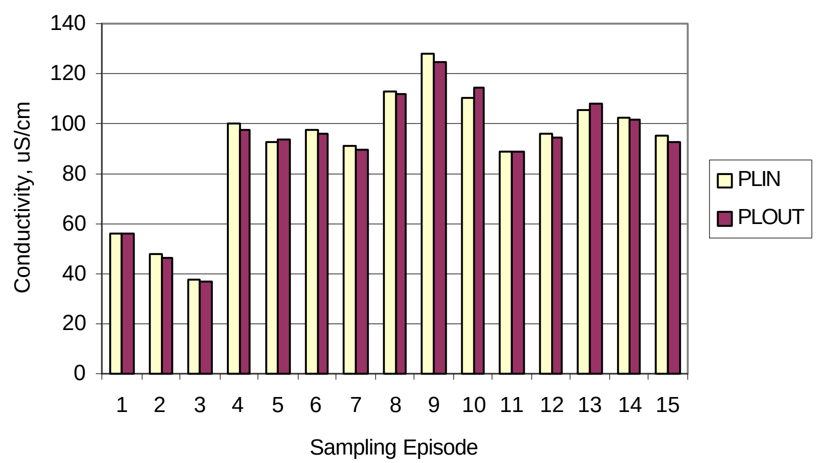

Figure 4.7. Conductivity inlet and outlet values.

A single-factor ANOVA was used to test the equality of the means between channels for each measured water quality constituent. TSS, TP, and turbidity were natural log transformed to induce normality. Conductivity required a cubic transformation.

ANOVA results on the transformed data found no significant differences ($p<0.10$) between channels for the mean values of TP and conductivity (Figures 4.8 and 4.9), but it did detect significant differences in mean values for TSS and turbidity (Figures 4.10 and 4.11). Further analysis using the Tukey multicomparison test found differences in the channel means for TSS and turbidity between channels PL2 and PL3 (p=0.047 and p=0.045, respectively) and between channels PL1 and PL2 (p=0.046 and p=0.012, respectively). The channel means for TSS and turbidity were lower in PL2, which was the control, or untreated channel.

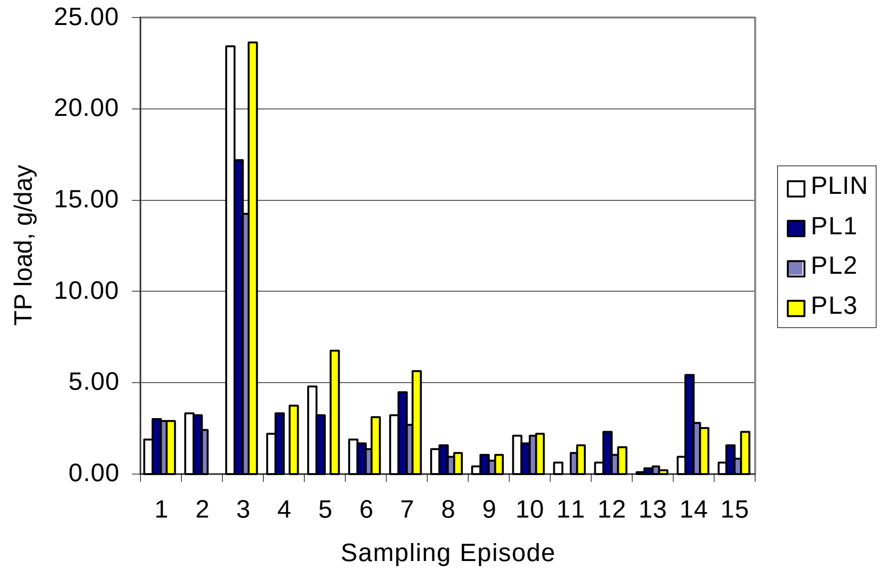

Figure 4.8. TP loads for Pine Lake biofiltration swale.

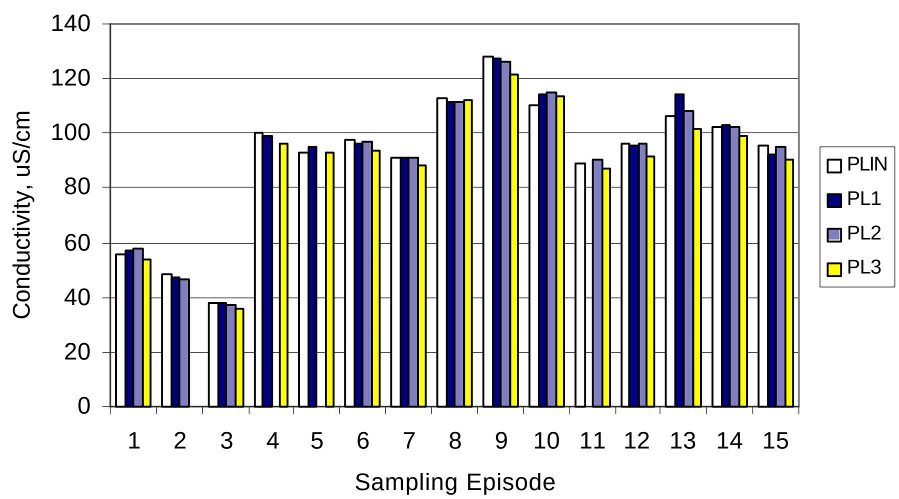

Figure 4.9. Conductivityat Pine Lake biofiltration swale.

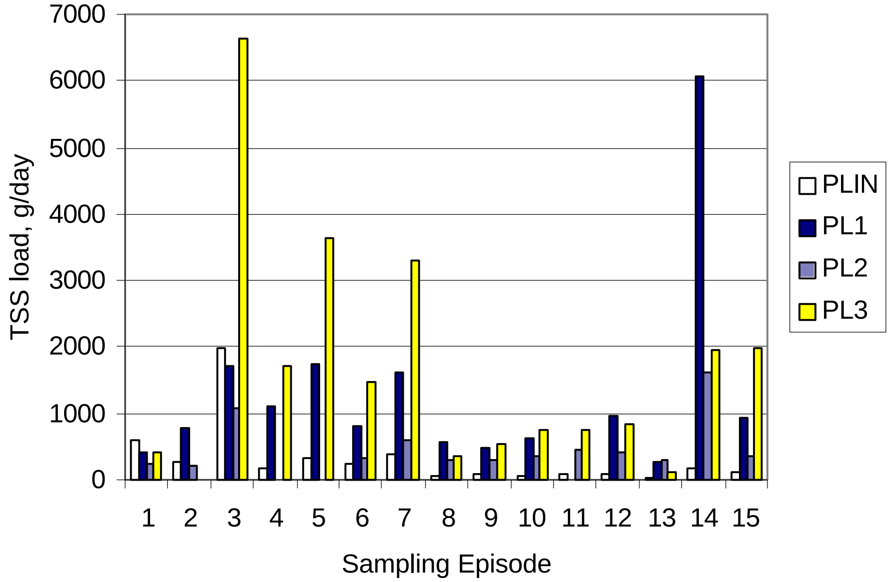

Figure 4.10. TSS loads for Pine Lake biofiltration swale.

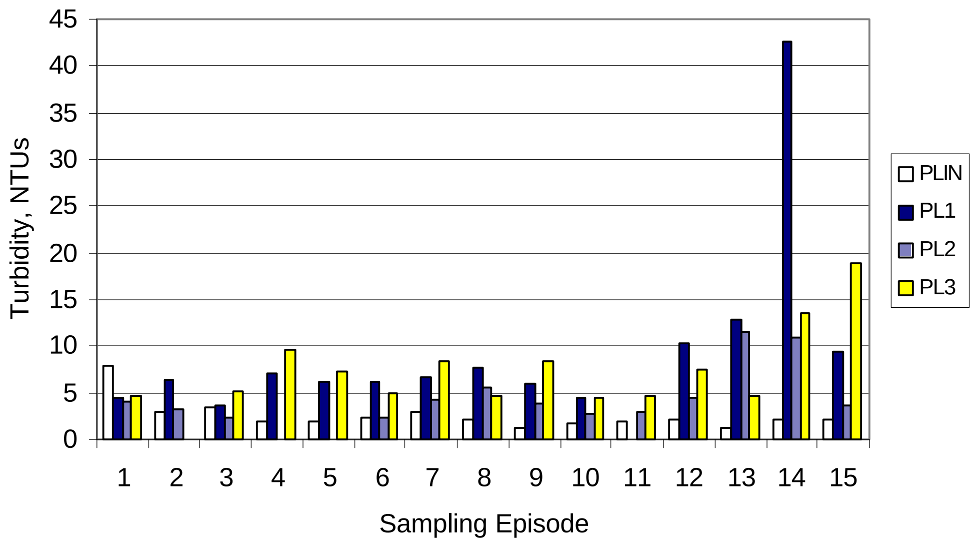

Figure 4.11. Turbidity readings for Pine Lake biofiltration swale.

High concentrations and readings produced the peaks in TSS loads and turbidity, respectively. A very high flow volume resulted in the peak in TP loads during episode 3.

Although significant differences were detected in pollutant loads for TSS and turbidity, overall, the presented results indicate that mowing had no measurable effect on swale treatment efficiency. Visual site observations support this statement, noting no obvious changes in vegetation density that might influence the treatment ability of the swale.

CHAPTER 5 - DISCUSSION

Characterization of Swale Performance Predictors

Evaluation of biofiltration swale performance is generally based on the key assumptions that a long HRT and good vegetation cover are the factors promoting good pollutant removal, and that these two factors are generally well-correlated. An HRT of at least nine minutes is posited as the minimum requirement to achieve high pollutant removals (Metro, 1992). This study measured HRT and characterized swale conditions at 14 sites. Swale geometry and vegetation cover were compared to hydraulic variables in order to find easily measurable characteristics that could be used as predictors of swale performance in place of directly measuring HRT. In addition, the general validity of the above key assumptions was explored.

Hydraulics and Geometry Criteria

Comparison of measured geometry to design criteria showed that most swales did not conform. None of the eight swales with less than two percent slope had underdrains, as required by most of the listed criteria. This could explain the higher percent of long-term standing water in low-slope sites. Most swales met King County's recommended range of 1 to 6% slopes, but only four met DOE and Metro requirements for 2 to 4% slopes; and six met Mazer's (contradictory) recommendation of 0.1 to 1.5% slopes. Observations during this study show that the swales with slopes greater than 3.5 % were susceptible to channel incision extending the length of the channel. It is recommended that slopes be less than 2.5 percent and that check dams be constructed when this conditions is not possible.

Most widths were within King County's range of two to 16 feet, but only four were close to Mazer's recommended five feet, and only one is within Metro's range of seven to eight feet. Eleven swales were greater than King County's recommended 100-foot length, seven were longer than Metro's 125-foot requirement, and only two fit Mazer's and DOE's 200-foot criterion. None of the measured *n* values were as low as DOE's 0.07 design value, and most were greater than King County and Metro's 0.20 design value.

Comparing the calculated design values to the design criteria specified in Table 1.1, only four out of 14 sites met the nine-minute minimum HRT criteria, and only three met Mazer's

(1998) 15-minute recommendation. The HRT was less than five minutes at five sites. All design velocities are under 0.9 fps and meet most guidelines for this criterion, but only five met Mazer's recommendation of 0.2 fps and these all had slopes less than 1.5 percent. None of the swales had design depths less than 0.15 ft and only one was less than 0.25 ft. Most (13 out of 14), however, fit DOE's criteria of design depths less than 5 in. (0.42 ft).

Because actual site design flows could not be located, values were computed based on the current geometry, a recommended depth of 0.25 ft., and a value of *n* equal to 0.20. As a result, the calculated values may be different from those that each site was actually designed to treat, giving inflated or deflated HRTs. In any subsequent studies, a more accurate method of testing would be to run flow at the same depth for all sites. This would allow results to be compared directly and would eliminate error arising from scaling the HRTs and hydraulic variables.

The calculations and methods used to determine design values were developed for evaluating relative swale functioning, and they are reasonably accurate for this purpose, but they should not be taken out of context. The design depth and flow were based on a constant *n* even though this parameter has been shown to vary notably with depth. In addition, the *n* value recommended for biofiltration swales and used to compute design flow was developed for a particular site condition and vegetation composition. The *n* values calculated in this study varied widely, however, with a range from 0.19 to 0.53, a mean of 0.32, and a median of 0.28. This suggests that the recommended value of 0.2 may not be generally applicable. The measured *n* values came from a wide variety of swale sizes and conditions, and showing no correlation to vegetation cover or species. Therefore, it would appear that a more accurate value for design would be 0.30.

Evaluation of Swale Design

The design and treatment consequences of alternative *n* values can be explored by calculating the design parameters for a swale that must treat a given discharge and occupy a specified slope. The design of swales based on an *n* that is too low will be discussed first and is illustrated by a hypothetical swale in Table 5.1. Designing with an *n* lower than actually exists will result in a smaller bottom width than needed to convey the discharge at design depth, forcing the flow depths to increase. Velocity would decrease from its expected value due to higher depths, resulting in a longer length and HRT than required. The values in Table 5.1 that change from those expected are shaded.

Table 5.1. Comparison of design and actual parameters in swales designed with too small *n*.

Swale Parameters	Slope = 0.025		Slope = 0.01	
	Design	Actual	Design	Actual
Q (cfs)	0.41	0.41	0.39	0.39
b (ft)	3.2	3.2	5.0	5.0
L (ft)	225	225	145	145
v (fps)	0.41	0.31	0.27	0.21
d (ft)	0.25	0.32	0.25	0.32
n	0.20	0.30	0.20	0.30
HRT (min)	9.1	12.0	8.9	11.7

Conversely, applying an *n* too high would lead to an overly wide bottom and overly short length. This would result in higher velocities but shallower flow. Table 5.2 compares velocity, depth, and HRT in a swale designed based on an *n* representing rougher conditions than actually exist to true conditions.

Table 5.2. Comparison of design and actual parameters in swales designed with too large *n*.

Swale Parameters	Slope = 0.025		Slope = 0.01	
	Design	Actual	Design	Actual
Q (cfs)	0.41	0.41	0.41	0.41
b (ft)	5.0	5.0	8.0	8.0
L (ft)	155	155	101	101
v (fps)	0.28	0.37	0.19	0.24
d (ft)	0.25	0.20	0.25	0.20
n	0.30	0.20	0.3	0.20
HRT (min)	9.1	7.0	9.1	7.0

The benefits of under- or over-designing swales relative to pollutant-removal performance were evaluated through investigation of sedimentation potential. The theory of ideal particle settling compares the detention time in a facility (equal to the volume of water in the facility divided by the inflow discharge, Q) to the time it takes a particle to settle through the full depth of flow (equal to the depth divided by particle settling velocity) (Metcalf & Eddy, 1991). Setting the detention time equal to settling time specifies a criterion for the minimum settling velocity for sedimentation within the facility, Q/A, where A represent the bed area (the product of width and length). For every situation illustrated in Table 5.1 and Table 5.2, the minimum computed settling velocities were all approximately 0.0004 fps. Even with allowances for the actual non-ideal mechanics of particle settling, this finding suggests that biofiltration swales will generally remove all sand particles (which have settling velocities greater than about 10^{-3} fps) regardless of length and depth (within the ranges specified in the design criteria), but not necessarily silt particles.

Based on the values presented in Tables 5.1 and 5.2, using an inaccurate *n* value will not result in significantly different conditions for sedimentation. Therefore, swales should be over-designed for width (but only up to a maximum of 10 feet, since this and other studies (Metro, 1992) have noted that flow does not necessarily spread over the full width of a wide swale), and under-designed for length using an *n* of 0.3. DOE's recommendation of 0.07 is too low and would produce an unreasonably narrow design bottom width and length, except at very low slopes.

These calculations also establish the maximum flow capacity that swales can effectively treat under design capacity. A swale designed with a 10-foot bottom width, flow depth of 0.25 feet, 2.5% slope and *n* equal to 0.3 can only treat up to 0.80 cfs. The same swale with an *n* of 0.20 can only treat up to 1.19 cfs. Greater design discharges will require multiple swales for treatment.

Vegetation

Significant vegetation cover is considered an important element of swale functioning because of its presumed ability to filter out pollutants. It is also generally assumed that vegetation reduces flow rates, but study findings did not support this assumption. It should be noted that percent cover is a misleading measurement in the very channelized swales (sites 1824, 1819, and 1784). At these sites the flow generally does not move through the vegetation as sheet flow, but instead is concentrated in the (mostly unvegetated) incised channel. In addition, the computed velocities at these sites are minimum values because they are based on the width of the entire swale instead of just the incised channel width. The velocities calculated using actual channel width could approach erosive flows, which would explain the channel formation.

The ANOVAs show that low slopes in biofiltration swales negatively affect vegetation cover and positively affects organic litter and plant biomass. Low gradient swales also have a greater tendency to retain water for a long period of time and into the growing season. This suggests that wet sites should have lower cover; however, the two dry sites in the low-slope category had low percent cover while two of the wet sites had greater than 75% cover. In addition, no significant differences were found for percent cover between swales with less than 1.5 % slope and those greater than 2.5 % slope. Low cover is probably a function of a variety of conditions and not just long-term inundation or slope. The low-slope sites do retain more organic litter, probably a function of the corresponding low velocities which are unable to flush out the loose organic material. Organic litter indicates a plausible source of soluble nutrients and re-suspendable organic matter from the decaying vegetation.

Swale Performance Predictors

This study does not support the assumption that a long HRT is well-correlated with good vegetation cover. Swales with the longest HRTs do not necessarily have high percent cover, and many swales with high percent cover have very short HRTs. Instead, results showed that HRT was not correlated with percent cover and instead strongly correlated only with swale length and slope. Table 5.3, sorted by HRT, illustrates this finding.

Table 5.3. Correlation between HRT, percent cover, slope and length.

Site No.	HRT_{design}	Summer Cover (%)	Winter Cover (%)	Slope (%)	Length (ft)
1784	2.5	62	37	5.0	100
1750	2.8	94	97	1.9	80
2124	3.1	83	85	1.8	75
1903	3.8	68	85	2.3	85
1824	3.9	64	64	5.7	131
1977	5.4	91	85	3.5	100
1819	6.1	68	15	3.8	200
2007	8.2	81	85	2.2	180
2434	8.5	10	10	1.0	120
1984	8.8	42	37	1.4	180
2009	11.7	75	15	0.4	176
1703	16.7	28	15	0.8	119
2335	19.4	56	56	0.3	150
1933	22.7	86	63	0.9	203

Three of the of the four sites with an HRT greater than nine minutes, have lengths greater than 150 feet and all four have slopes less than 1.0 percent. Design depths at these sites range between 0.25 to 0.31 ft; design velocities range from 0.15 to 0.21 fps, respectively. Manning's *n* values fall between 0.19 and 0.28. Spring vegetation cover varied from 15 to 63 percent and summer vegetation cover from 28 to 86 percent. The constant variables for swales with long HRT appear to be:

- long length,
- low slope, and

- low velocity.

The degree of vegetation cover, however, is *not* predictive of long HRT.

Since vegetation is assumed to enhance pollutant removal by providing a filtering effect, sites with long residence time but low percent cover may not be very effective. Alternatively, since sedimentation is widely assumed to be the primary removal mechanism for stormwater pollutants, perhaps a swale's performance is primarily a function of HRT and so not dependant on vegetation cover.

It is beyond the scope of this study to resolve the conundrum of whether HRT or vegetation cover is most important to swale performance. There is evidence in the published literature for both factors. Good grass cover has been shown to promote sedimentation (Wilson, 1967; Dillaha et al., 1989; and Yu et al., 1995) and a HRT greater than nine minutes has demonstrated good pollutant removal (Metro, 1992). For purposes of evaluating general swale performance, a site with long HRT and good vegetation cover can be assumed to have high pollutant-removal performance. Conversely, a site with short HRT and low vegetation cover would perform poorly. Sites with intermediate values of either HRT or vegetation cover or both, or that are "good" in one parameter but not in the other, cannot be evaluated. Most of the swales in this study, unfortunately, fit one of these descriptions.

Regression Models

The model that best characterizes HRT was expected to be a function of either vegetation cover directly or an *n* that reflected vegetation resistance. Instead, the most predictive equations for HRT included only length, slope, and velocity demonstrating that HRT is overwhelmingly determined by channel geometry and flow continuity.

Equation 4.3 provides a functional method of evaluating HRT in an existing site because both slope and length are easily measured. Equation 4.4 is more applicable to design situations where a target velocity or length is specified and values of the other variable are tested by substitution into equation 4.3 and evaluated for adequacy based on the resulting HRT. Both equations were developed from data spanning a wide variety of vegetation covers, and so HRT should be reasonably well estimated by these equations over a range of vegetation covers and species.

Coefficient of Flow Resistance

The value of *n* incorporates the effects of obstructions and channel irregularities that contribute to overall flow resistance. Numerous studies show *n* to vary widely with depth in small, low-flow channels. However, a constant *n* value is still traditionally used for biofiltration swales to set design criteria and solve for hydraulic variables.

In an ideal facility, the vegetation will be approximately 6 in high, is not fully submerged, and provides significant flow resistance. Under these conditions it is assumed that the resistance effect of vegetation cover dominates the value of *n*. Yet the results here found no such connection between values of *n* and percent vegetation cover; similar values of *n* spanning the range of percent cover were observed for both spring and summer. In fact, no significant relationships were found among the variables measured to indicate the influencing factors of *n* in low-flow vegetated channels.

The sensitivity of *n* to changes in slope and discharge or depth was illustrated at site no. 2135, the only site where data were recorded at two different discharges. Although its data had to be excluded from the results because recent retrofitting prevented the measurement of an accurate slope, they can be used to look at variations in *n* and design depth and velocity (Table 5.4).

Table 5.4. Variations in *n* with depth and slope.

Slope Estimate	Variables	Q = 0.40 cfs	Q = 0.75 cfs
0.0025	*n*	0.70	0.38
	$depth_{design}$ (ft)	0.55	0.37
	$velocity_{design}$ (min)	0.11	0.10
0.005	*n*	0.99	0.53
	$depth_{design}$ (ft)	0.67	0.45
	$velocity_{design}$ (min)	0.08	0.11
0.01	*n*	1.41	0.75
	$depth_{design}$ (ft)	0.85	0.57
	$velocity_{design}$ (min)	0.08	0.12

Review of the discharge columns shows that *n* will increase considerably with a doubling of slope. The other values are better compared across rows where slope is constant and the computed design discharge is the same. As a result of *n*'s sensitivity to changes in depth and slope, it would seem that the use of a constant value in hydraulic computations should be avoided. In addition, this study's large range of *n* values questions the designation of a single design value.

However, it is generally difficult to either design with a varying *n* or without using Manning's equation and avoiding the use of an *n* value altogether. The previous discussion regarding evaluation of swale design noted that the recommended *n* value for design should be 0.30 instead of 0.20, although the consequences of designing with a "wrong" *n* value are not severe. Figure 5.1 compares the actual depths for each swale computed from two different calculated design discharges based on an *n* equal to 0.20 and an *n* equal to 0.30. The results show a smaller range around 0.25 ft. for design flows calculated using *n* equal to 0.30, supporting the recommendation of a design *n* value equal to 0.30. The graph presents the values of the design depths as the difference from the design depth criterion (0.25 ft). The depths computed from the design discharge based on *n* = 0.20 are all above the 0.25 ft, while the depths computed from the design discharge based on *n* equal to 0.30 vary less widely from the design depth criterion.

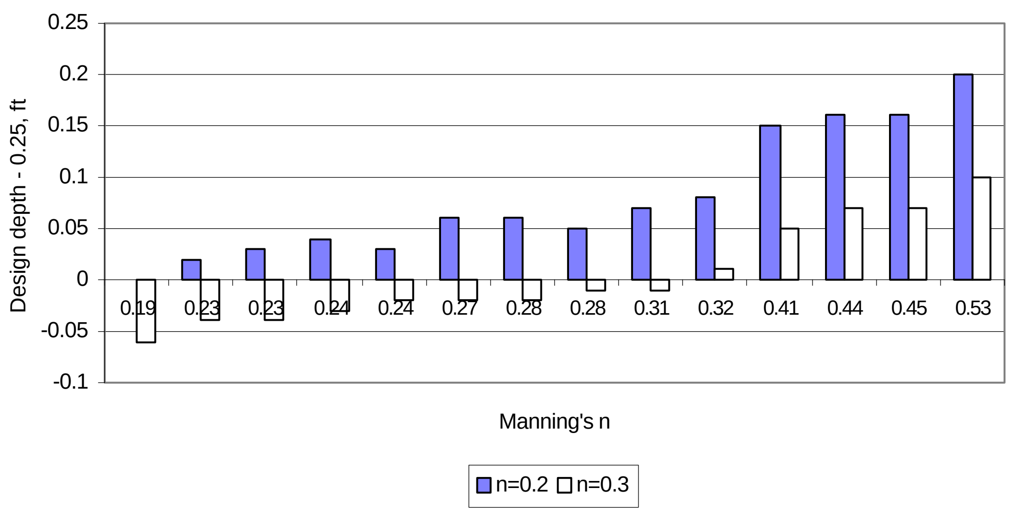

Figure 5.1. Comparison of design depths based on *n*=0.20 and *n*=0.30.

The Effect of Mowing Regimes on Swale Treatment Efficiency

Mowing has been characterized as an important maintenance practice for establishing and maintaining good treatment efficiency in biofiltration swales. By this thinking, mowing promotes thicker grass growth and reduces weed populations that might shade out desirable species. This theory was tested at the Pine Lake biofiltration swale. Channel PL1 was mowed once during the growing season, while PL3 was mowed twice. Channel PL2 was left unmowed and so acted as a control.

TSS and TP PMCs averaged 2.9 mg/L and 21.2 μg/L at the inlet, respectively, while outlet concentrations averaged 14.8 mg/L and 32.2 μg/L, respectively. Although both water quality constituents had the unexpected result of increasing in concentration by passing through the swale, the outlet concentrations are quite comparable to other reported outlet concentrations in other water-quality facilities (King County, 1995; Comings et al., 2000). However, any increase in concentrations obviously defeats the purpose of the swale for water quality treatment.

Among the three mowing treatments, TSS loads, TP loads, and turbidity showed no significant differences. They all followed an unexpected pattern by mostly showing concentration increases. Conductivity measurements showed relatively small and inconsistent effects between the three swale treatments. Based on these results, the different mowing treatments did not affect pollutant removal.

Irreducible Concentrations

The negative reductions in TP, TSS, and turbidity are probably a result of low initial concentrations. Low inflow concentrations are expected because the site follows two large wetponds; and the creation of an additional stilling basin by the upstream weir most likely reduces pollutant levels even further. It has been suggested that pollutants generally have an “irreducible concentration,” meaning that after a certain level additional BMPs will not reduce a pollutant’s concentration further (Schueler, 1996). Schueler’s suggested irreducible concentrations in stormwater BMPs for TSS and TP are 20 to 40 mg/L and 0.15 to 0.20 mg/L, respectively, which are consistent with the reported values from other recent studies. All of the inflow concentrations are below the irreducible concentrations.

CHAPTER 6 - CONCLUSION

This study was carried out in two parts to examine the impact of site characteristics and maintenance practices on biofiltration swale performance. One effort sought to characterize a swale's performance and to identify measurable indicators for general application and use by maintenance personnel. The second part used Pine Lake swale (site 2335) to evaluate whether mowing frequency during the growing season impacted pollutant removals.

Swale Performance Predictors

Studies have shown that shallow flow through vegetation, primarily grass, provides a good filter mechanism and reduces pollutant loads. However, identification of "well-performing swales" was predicated on an alternative criterion, namely an HRT greater than nine minutes. It was assumed this study would find a relationship between vegetation cover and HRT, thus confirming the importance of both factors in pollutant removal and providing a simple, visual technique for estimating pollutant-removal performance. Instead, results showed that HRT was not correlated with percent cover and instead strongly correlated only with swale length and slope.

Four out of 14 swales had an HRT greater than nine minutes, with summer vegetation cover ranging between 28 and 86 percent and winter cover ranging between 15 and 63 percent. All lengths were greater than 100 ft and most were at least 150 ft. Slopes were all less than 1.5 percent. In general, sites with slope between 1.5 and 2.5 percent had consistently higher cover in the summer and winter than other slope categories (less than 1.5% and greater than 2.5%) and although their HRTs were less than 9 min, this appears to be primarily because they are too short.

The fact that a significant trend could not be detected between HRT and vegetation cover, and that HRT depends primarily on length and slope, raises two questions: is HRT is a valid measure of swale performance when little vegetation is present? Is vegetation in fact important to swale performance?

Forty-eight different vegetation species were identified in the swales, but no significant trends were found relating species to swale HRT, percent cover, slope, soil type, frequency of water, or *n*. It was noted that *Agrotis spp.*, which has fine dense blades that can provide good filtration, was present at almost every site and appeared to thrive in all conditions. In addition, it is included in King County's (1998) recommended grass seed mixes.

Coefficient of Flow Resistance

Manning's equation and coefficient of resistance was originally developed for use in deep, turbulent flows. However, it is widely used as a general equation for all types of open channel flow. Results of this study support the findings of earlier studies that n varies with flow depth and site-specific conditions. Manning's n ranged from 0.19 to 0.53, with no general associations between general vegetation cover or type and depth of flow; at the one site where data were available, however, results show that n varies significantly with changing depth. This suggests that n is very site-specific and should not be generalized, especially in low flow conditions where boundary conditions are so influential. Yet, in cases where a value for n is required for design purposes the results of this study suggest a value of 0.30 is more appropriate than 0.20. However, design with an assumed value of n that is either too large or too small will not have severe consequences to a swales potential for sedimentation, which basically reflects its ability to remove pollutants.

Swale Maintenance and Location

Results at the Pine Lake swale did not support the theory that regular mowing creates thicker vegetation cover or leads to better pollutant removal. The lack of differences in vegetation density between channels was most likely due to low initial vegetation cover and a significant percent of non-grass species. Results from the composite sampling indicated that the swale was a source for TSS, TP and turbidity during most sample periods.

Mowing in biofiltration swales that have low percent vegetation cover and/or the presence of non-grass species may not enhance either the growth of "desirable" species or its pollutant-removal performance. Mowing is important for aesthetics, but minimal maintenance may be sufficient to prevent sites from becoming overgrown. The fact that no significant difference was noted after a year between the unmowed channel and the mowed channels, and that none of the vegetation was excessively high or unsightly, indicates that maintaining swales for aesthetics may only be necessary every few years. Swales that cannot sustain grass species should be seeded with low-growing non-grass species, such as *R. repens*.

The results also suggest that sites following retention ponds might not provide any additional water quality benefit because pollutant concentrations are already below an irreducible level. Not only may swales be an unnecessary facility downstream of adequately sized retention ponds, but also they may in fact elevate pollutant concentrations.

The results from this investigation of mowing regimes raise an important issue to investigate further. A large percent of maintenance budgets are spent on mowing. Determining if and under what conditions there are any real benefits from this practice could allow funds to be spent for other maintenance problems, such as retrofitting poorly functioning swales.

Operation and Maintenance Recommendations

Although the relative importance of HRT and vegetation cover to pollutant removal has not been established, the following criteria can be applied for general evaluations of whether a swale's performance is good or poor. Intermediate performance cannot be effectively evaluated at this time.

- Long HRT (greater than 9 minutes) and good vegetation cover (greater than 75%) identifies a well-functioning swale.
- Short HRT (less than 5 minutes) and poor vegetation cover (less than 25%) indicates a poor-functioning swale.

HRT can be estimated from equation 4.3. An estimate of vegetation cover can be made by applying the method described in the Methods and Materials chapter from Mazer (1998).

Another method for evaluating a swale's performance is by estimating its sedimentation potential. Comparison of a swale's minimum settling velocity (Q/A) against various particle settling velocities would indicate the smallest particle size ideally capable of settling out and the swale's potential to remove pollutants.

It was noted that all swales with slope greater than 3.5% were incised for the length of the channel. A maximum slope of 2.5 % is recommended. Where higher slopes are unavoidable, include checkdams.

Future Research Recommendations

The following issues should be explored further to better understand the swale characteristics that promote greatest treatment and to guide efficient maintenance practices.

- What is the relative importance of HRT and vegetation cover in achieving effective pollutant removal?
- If vegetation is identified as relatively important, does species matter and is there a threshold percent cover under which vegetation provides little or no benefit?
- Do long HRTs have negative effects on either vegetation growth or pollutant removal, especially during periods of low flow and where decaying vegetation is present?
- Is Manning's equation applicable to swales, especially given the apparent sensitivity of n to changing depth? If not, what alternative should be used?
- Should swale design equations include a safety factor that compensates for potential under-designed facilities based on an inaccurate n?
- What effect do retention ponds have on biofiltration swales that drain them, and what remaining treatment do downstream swales provide?
- What is the effect of mowing on swales with a good grass cover and that do no follow retention ponds? (Application of the methods and sampling regime used in this study is recommended as a cost-effective way to research this question.)

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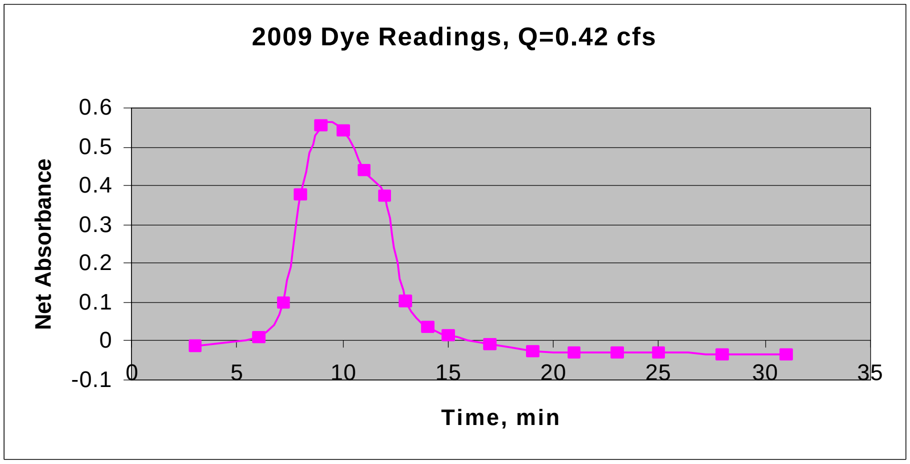
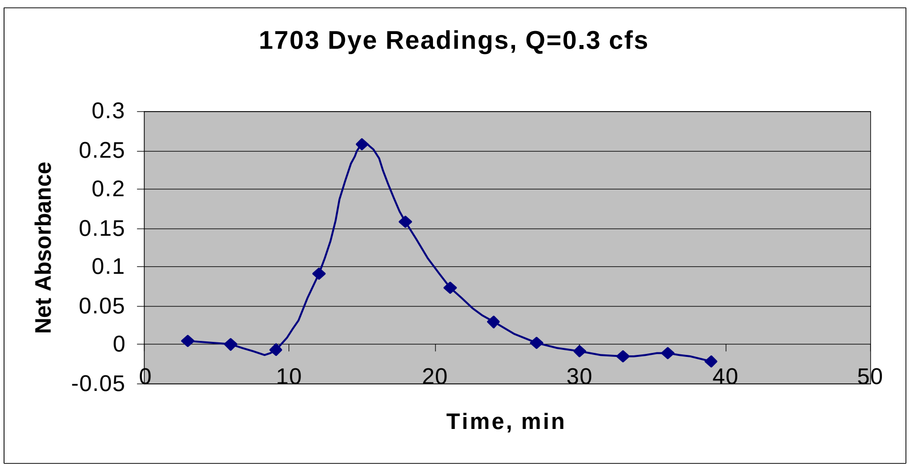
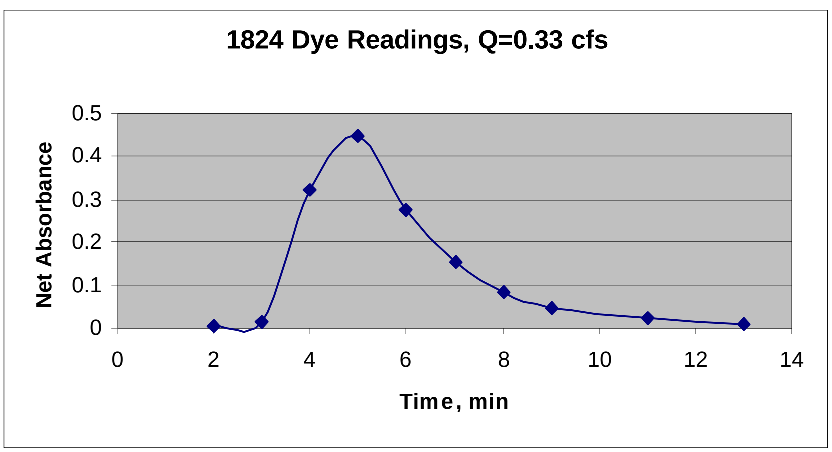
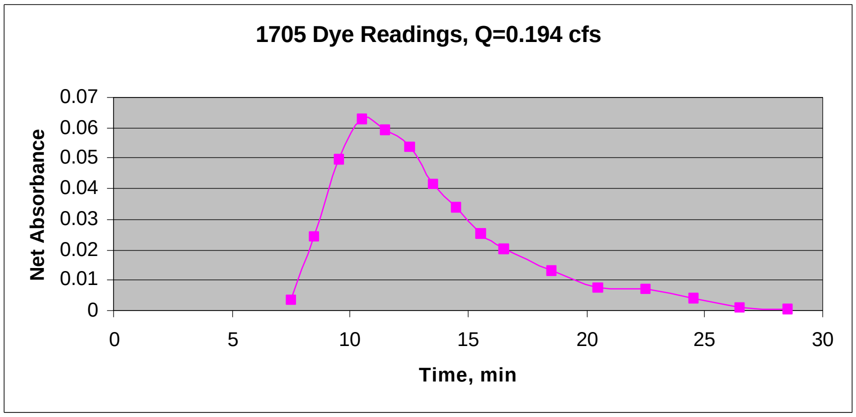
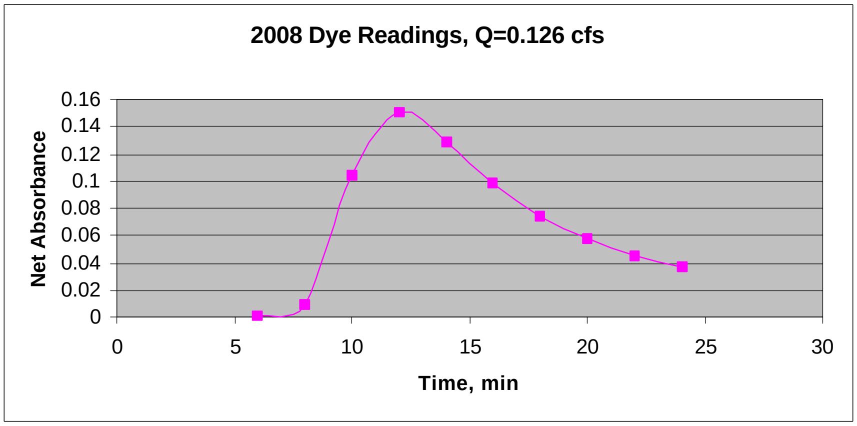
APPENDIX 1 – Dye concentration versus time curves for computing HRT.

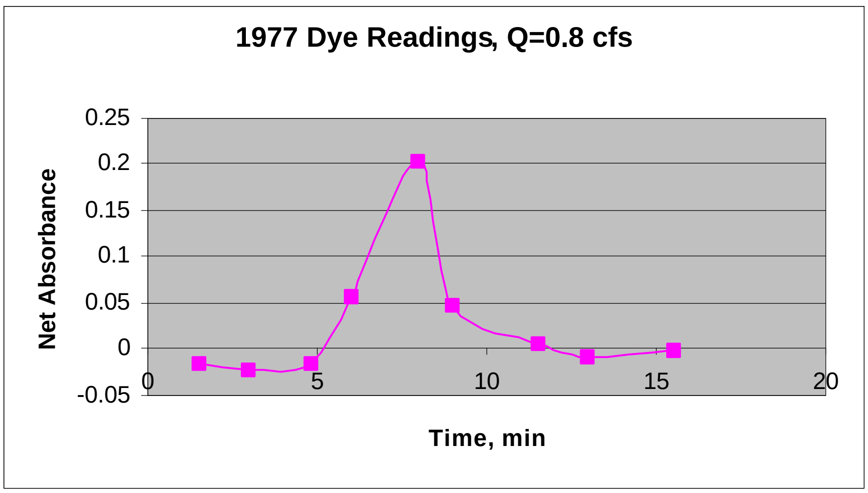


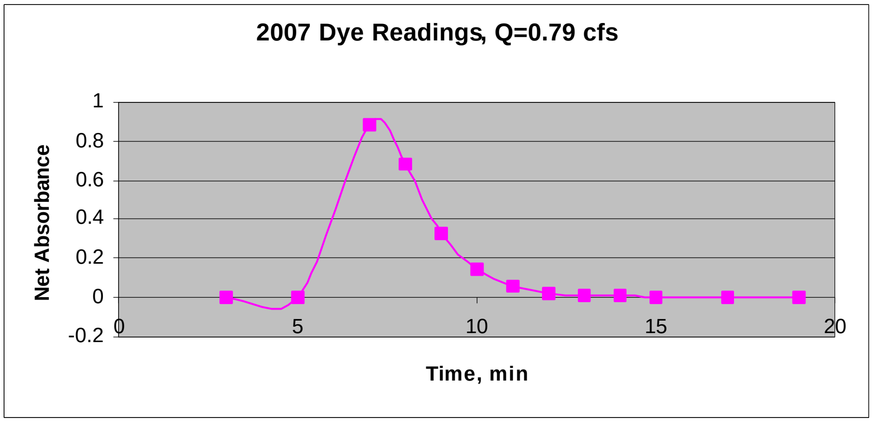


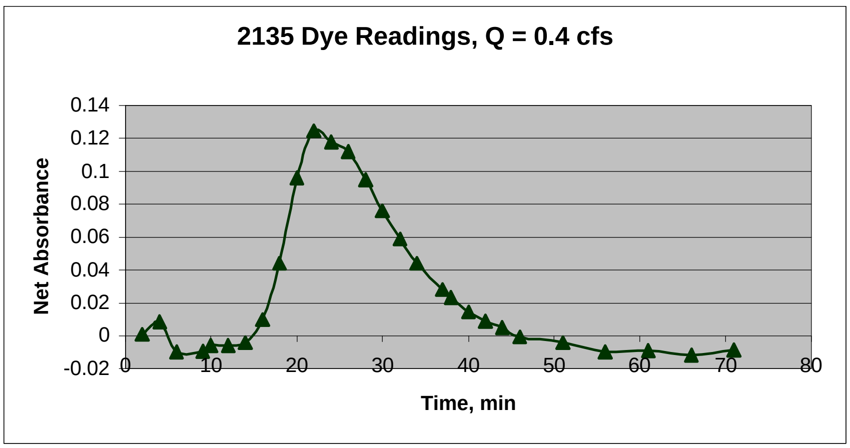


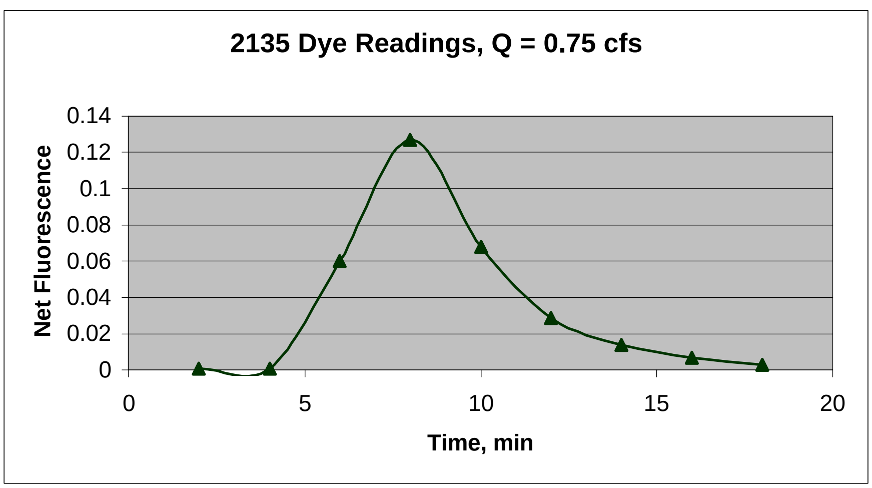

1705 Dye Readings, Q=0.194 cfs
Net Absorbance
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
0
5
10
15
20
25
30
Time, min


2008 Dye Readings, Q=0.126 cfs
Net Absorbance
0.16
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
0
5
10
15
20
25
30
Time, min


1977 Dye Readings, Q=0.8 cfs
Net Absorbance
0.25
0.2
0.15
0.1
0.05
0
-0.05
0
5
10
15
20
Time, min


2007 Dye Readings, Q=0.79 cfs
Net Absorbance
1
0.8
0.6
0.4
0.2
0
-0.2
0
5
10
15
20
Time, min


2135 Dye Readings, Q = 0.4 cfs
Net Absorbance
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
-0.02
0
10
20
30
40
50
60
70
80
Time, min


2135 Dye Readings, Q = 0.75 cfs
Net Fluorescence
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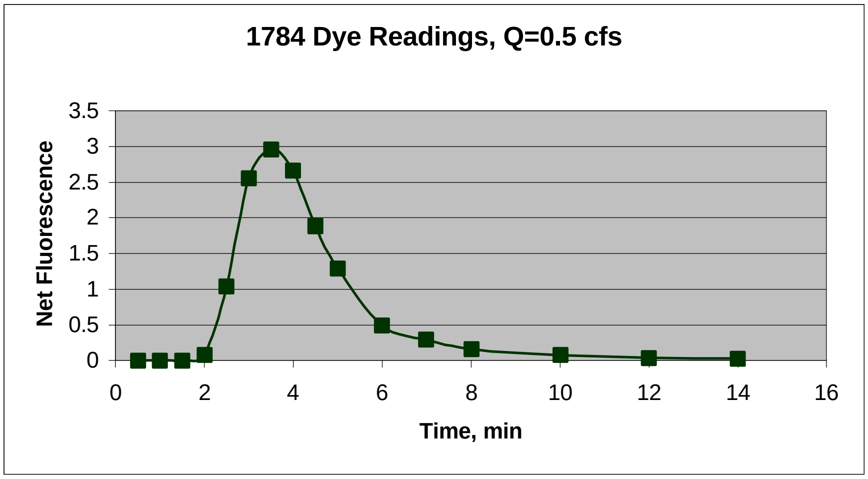
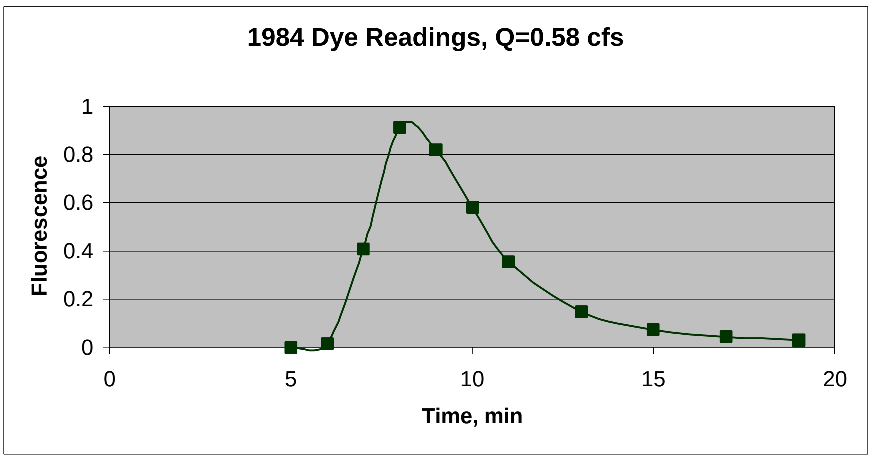
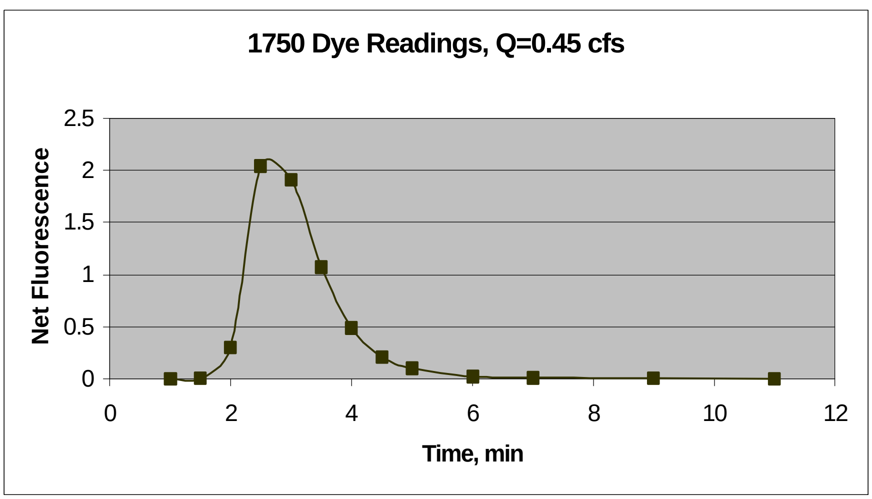
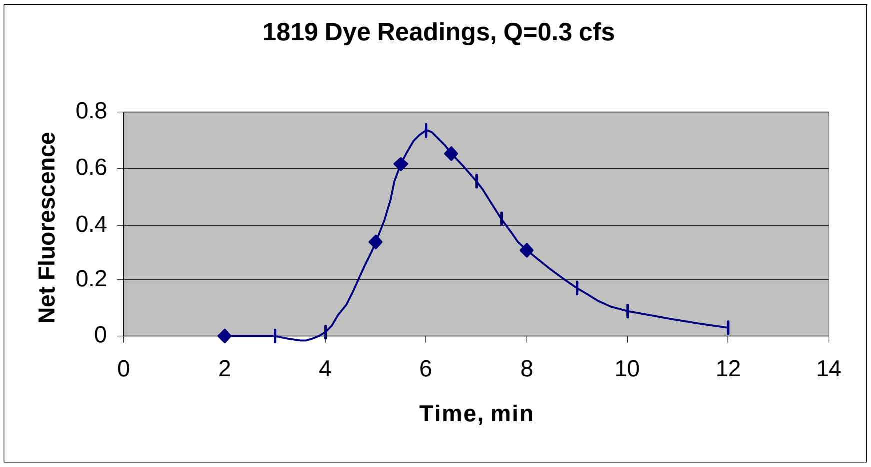
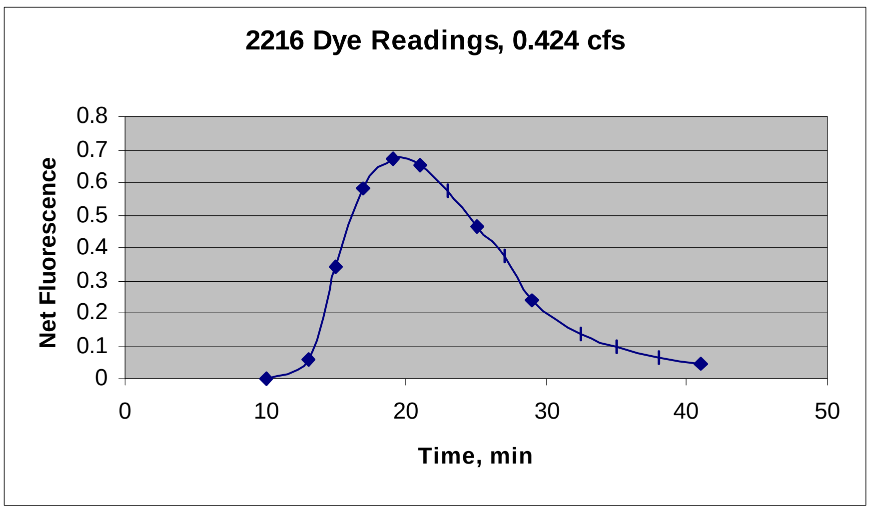
0.12
0.1
0.08
0.06
0.04
0.02
0
0
5
10
15
20
Time, min

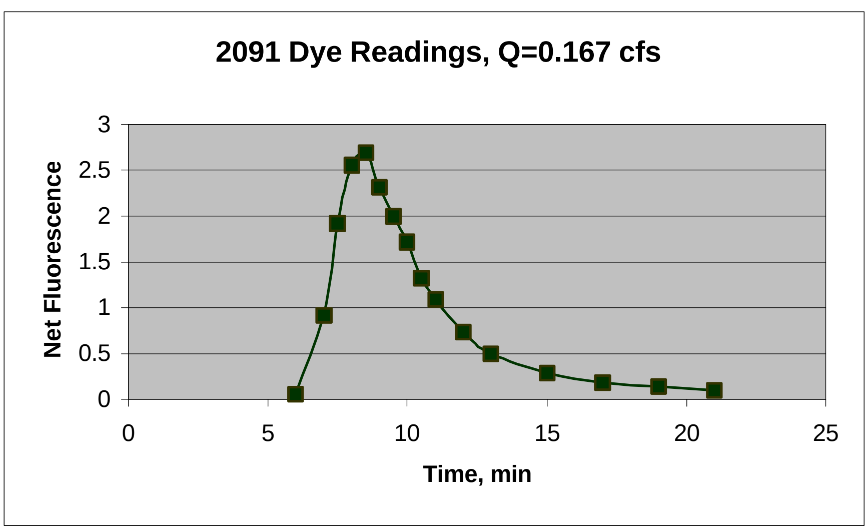

1784 Dye Readings, Q=0.5 cfs
Net Fluorescence
3.5
3
2.5
2
1.5
1
0.5
0
0
2
4
6
8
10
12
14
16
Time, min

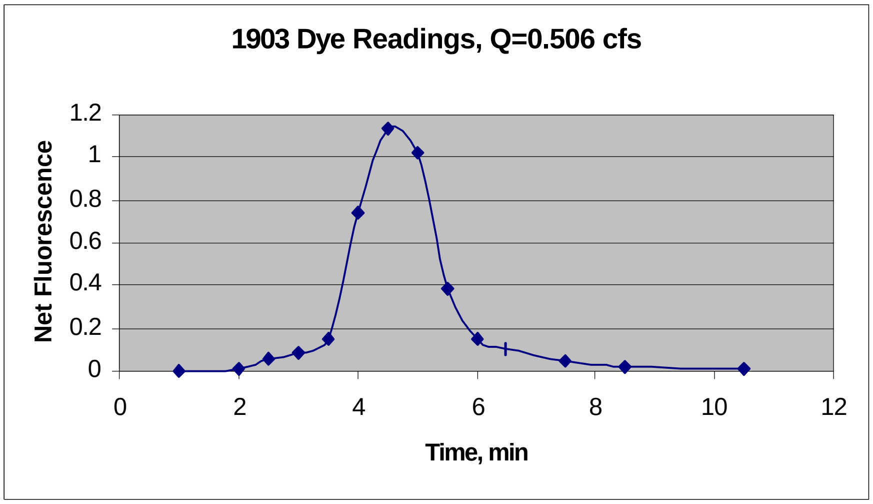

1984 Dye Readings, Q=0.58 cfs
Fluorescence
1
0.8
0.6
0.4
0.2
0
0
5
10
15
20
Time, min

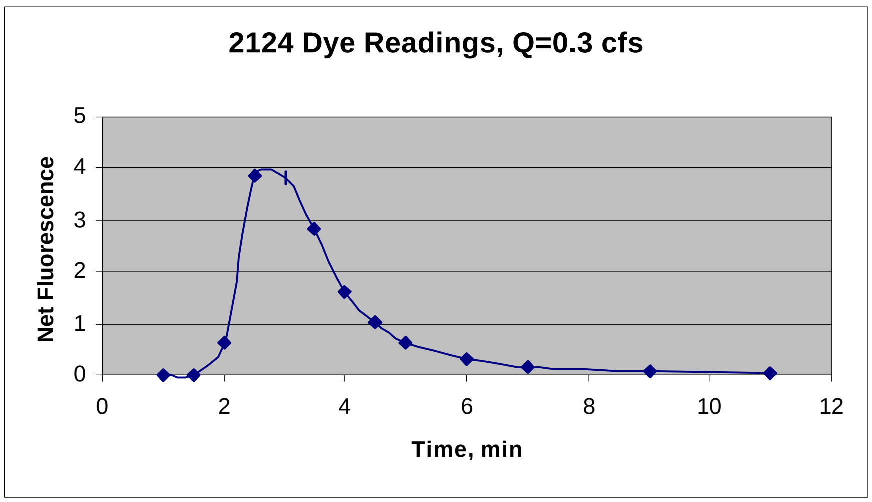

1750 Dye Readings, Q=0.45 cfs
Net Fluorescence
2.5
2
1.5
1
0.5
0
0
2
4
6
8
10
12
Time, min

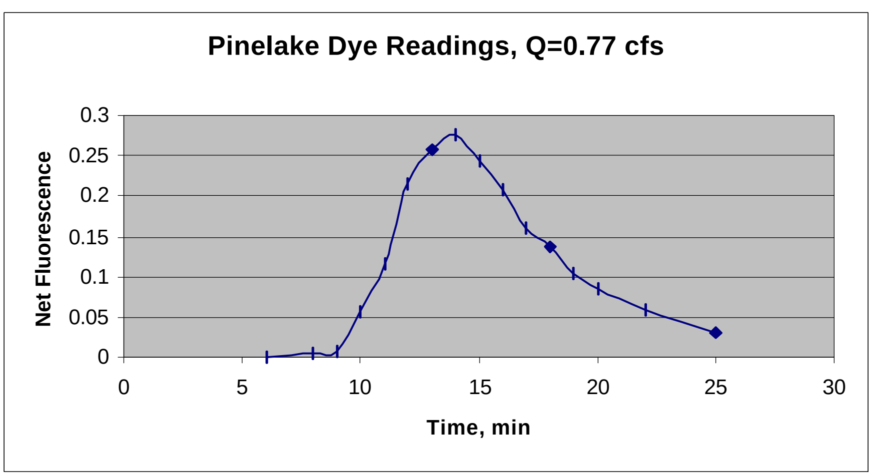

1819 Dye Readings, Q=0.3 cfs
Net Fluorescence
0.8
0.6
0.4
0.2
0
0
2
4
6
8
10
12
14
Time, min

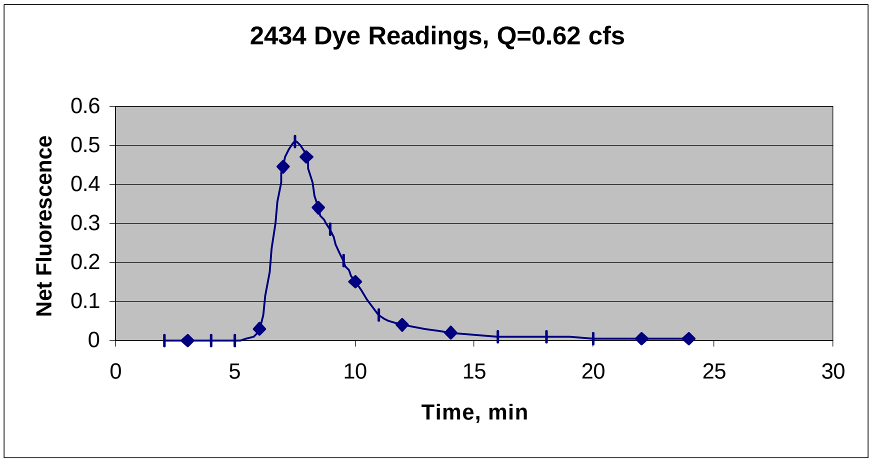

2216 Dye Readings, 0.424 cfs
Net Fluorescence
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
0
10
20
30
40
50
Time, min


2091 Dye Readings, Q=0.167 cfs
Net Fluorescence
3
2.5
2
1.5
1
0.5
0
0
5
10
15
20
25
Time, min


1903 Dye Readings, Q=0.506 cfs
Net Fluorescence
1.2
1
0.8
0.6
0.4
0.2
0
0
2
4
6
8
10
12
Time, min


2124 Dye Readings, Q=0.3 cfs
Net Fluorescence
5
4
3
2
1
0
0
2
4
6
8
10
12
Time, min


Pinelake Dye Readings, Q=0.77 cfs
Net Fluorescence
0.3
0.25
0.2
0.15
0.1
0.05
0
0
5
10
15
20
25
30
Time, min


2434 Dye Readings, Q=0.62 cfs
Net Fluorescence
0.6
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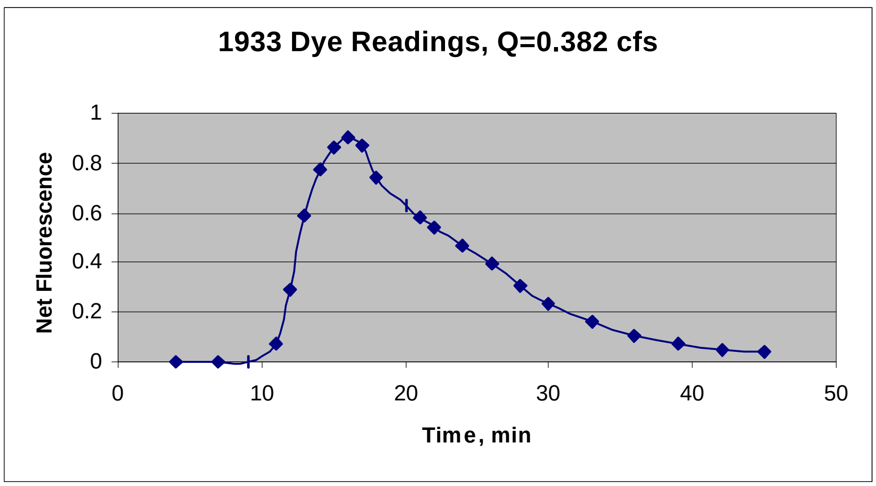
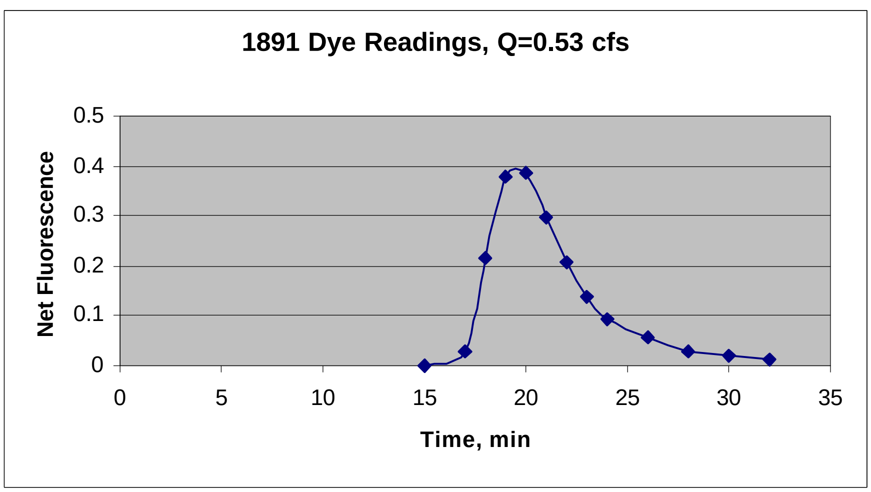
0.4
0.3
0.2
0.1
0
0
5
10
15
20
25
30
Time, min


1933 Dye Readings, Q=0.382 cfs
Net Fluorescence
1
0.8
0.6
0.4
0.2
0
0
10
20
30
40
50
Time, min


1891 Dye Readings, Q=0.53 cfs
Net Fluorescence
0.5
0.4
0.3
0.2
0.1
0
0
5
10
15
20
25
30
35
Time, min

APPENDIX 2 – FQI and Sampling Program Determination

VEGETATED STORMWATER FACILITIES MAINTENANCE PROJECT

INVESTIGATION OF EFFECTS OF SWALE MAINTENANCE OPTIONS ON WATER QUALITY PERFORMANCE THROUGH TEST SWALE MONITORING

Swale Monitoring Procedures and Criteria

GOAL: To make the best possible estimates of TSS and TP mass transport into the swale system and exiting each of the three swales.

Specific Objective: To assemble a complete flow record and to collect and analyze sufficient samples to make estimates of the mean and confidence limits of TSS and TP concentrations over the monitoring interval.

General Method

Mass transport will be determined as the product of total flow volume and mean concentrations. Mean concentrations and their confidence limits will be determined statistically from Period Mean Concentrations (PMCs) measured in a series of flow-proportional composite samples collected over sampling periods. PMCs are analogous to event mean concentrations (EMCs) but will represent periods of up to a week in length that may, and in general will, encompass more than one rainfall event. It is assumed that statistical calculations will be based on log-normal probability distributions of TSS and TP, but this assumption will be tested with the data.

EQUIPMENT PROGRAMMING

General Considerations

The samplers must be programmed with the volume to collect each time an individual sample is drawn. and the total number of samples to collect. The flow meters must be programmed with the flow pace or flow quantity increment; i. e., the increment of flow volume that will signal the samplers to draw water. These three quantities must be balanced according to three considerations: (1) obtaining adequate sample for laboratory analyses, (2) avoiding overfilling the composite containers, and (3) sampling at points spaced sufficiently close to represent the runoff hydrograph relatively well.

The first consideration is the smallest issue in this monitoring program, because little volume is needed for the few laboratory analyses specified and the extended monitoring periods will offer ample opportunity to obtain sample. On the other hand, these extended periods will raise the risk of overfilling, especially the inlet carboy. The more closely spaced in time the sampling points are, the better PMC estimates will be; but this consideration must be balanced with battery charge life and anticipated rainfall duration, as well as overfilling risk. There is no certain way of specifying the three quantities at the outset, but programming can be improved as experience with the site accrues.

The initial strategy in this program will be to use the smallest sample volume found to work well and the largest available composite carboys, not with the idea of filling them but to add assurance against overfilling. Specifying flow pace and number of samples is hindered by lack of information on flow volumes produced at the site by typical runoff events. The outlet structure geometry is known, however, which permits estimation of flows for selected scenarios.

Individual Sample Volume (V_I)

Samplers will be programmed, at least initially, to collect 50 mL at each sampling point. This volume is lower than some advice but has worked well in the Lakemont Boulevard construction project monitoring program, and should yield adequate sample for analyses in all cases. There appears to be no inherent reason why an even lower volume would not work; thus, a reduction could be considered, for example to allow more frequent sampling while not producing excessive collected volume.

$$\mathbf{V_I = 50\ mL}$$

Number of Samples (N)

With a 4-gallon (15-L) carboy, 300 samples each 50 mL can be taken before the container fills. Based on the Lakemont Boulevard experience, deep-cycle marine batteries are capable of drawing at least that many samples before needing a recharge. The region averages roughly 80 events over the six-month wet season, or about three per week, of average duration about 21 hours. Using these averages, the typical winter week would have about 63 hours of runoff, assuming the pond stage before the swales is full at the outset. Sampling 300 times would provide a sample approximately every 13 minutes, which would offer excellent hydrograph coverage. However, a somewhat longer interval between samples would still give good coverage and would leave carboy volume available when lengthier rainfall occurs. Also, 4 gallons of water weighs more than 33 lb. and would be somewhat hard to handle. Therefore, N will be set initially at 200, giving a sample on average about every 19 minutes. The setting will be reconsidered with experience for possible adjustment downward to lower the overfilling risk and reduce weight, so long as hydrograph coverage remains thorough.

$$\mathbf{N = 200}$$

Flow Quantity Increment (FQI)

FQI is the total flow volume expected over a sampling period (V_T) divided by N. Obviously, volume will differ with conditions, but variation will be attenuated in our system by the ponds. It is likely that the variation will be low enough to use one value for programming purposes, but it will be possible to modify the programming if experience shows that V_T varies substantially in a somewhat predictable way. When an estimate is established, that value will be used initially to program both the inlet and outlet flow meters. An adjustment will be made if experience shows that inlet and outlet flows differ systematically (e. g., if infiltration reduces outflow).

As mentioned, measurements during runoff are lacking at this point, and the initial setting must be made from estimates based on outlet geometry. According to a calculation using that geometry and the standard orifice equation, flow rates up to 2.74 cfs could be delivered to the swale via the outlet structure (pond overflow via the emergency spillway would be at a higher rate, beyond the scope for planning sampling). This high rate would occur only if the water surface were at its maximum at the overflow elevation in the outlet structure. It is very unlikely that any rate nearly this large would persist very long. Much more typical would be a consistent low flow rate with minimal head build up in the outlet structure. The flow rate then would be a fraction of a cfs. This scenario should be used for programming to ensure good coverage of the hydrograph in the most frequent, small storms. If a flow rate averaging 0.1 cfs over 63 hours of runoff in a week were assumed, V_T would be 22680 ft^3 and FQI = V_T/N = 22680/200 = 113.4 ft^3. It is recommended that the sampling program start with FQI = 115 ft^3 and adjust according to experience.

The selected flow pace, number of samples, and sample volume would fill 2/3 of the inlet carboy if, in fact, the week had 63 hours of runoff averaging 0.1 cfs. A period of relatively heavy rainfall could require suspending the sampling, retrieving the sample, and restarting sampling to avoid overfilling. Experience will be used to adjust the programming parameters to try to avoid irregular sampling periods that will be harder to manage than uniform ones.

FQI = 115 ft^3

SAMPLING PROCEDURES AND CRITERIA

General Considerations

An ideal program to reach the monitoring program's goal would be to sample and analyze all water that flows in the swales during the monitoring interval. However, achieving the ideal would entail having no problems that cause loss of sampling opportunity, which is rather unlikely, and would require a larger laboratory budget than anticipated. It is probable that, despite the likelihood of some problems, it will be possible to collect more composites than can be analyzed. Therefore, means are needed to decide whether or not to analyze a set of samples.

This section gives criteria to make that decision based on the objective of analyzing the samples that best represent the key conditions known to affect runoff water quality.

We originally anticipated a laboratory budget of $3400 split approximately equally between conventional analyses (TSS and TP) and metals. This budget would permit analyzing samples for conventionals and metals from 15 sampling periods, or over half of the wet season weeks, if sampling periods are a week in length. Sampling is getting a late start this year because of the monitoring station construction. Also, questions have arisen about analyzing extended composites for metals that make this portion of the anticipated program experimental instead of routine. To the extent metals analysis is done experimentally, it will be on fewer samples than originally considered. Therefore, it is likely that the budget can cover conventional analyses for more than 15 periods and for extension of sampling beyond the wet season. The program will start out by analyzing all valid samples to raise the chances of good wet season coverage. Continuing with full analysis will depend on what is decided to do with metals experiments and with consideration of saving some budget for dry season storms. It is likely that swale effluent will not occur in the driest part of the year, but sampling should be possible through the spring and early summer. The selection criteria here will be applied if and when it is decided to limit sample analyses.

Initiating Sampling

1. Program samplers with the initial parameter values given above.

2. At least every two days obtain an Internet local weather forecast giving enough information to estimate the total amount of rain expected during the sampling period.

3. Collect composite samples for one week, but not longer. If the inlet and at least two outlet samples have been successfully collected, immediately deliver the samples for analysis (otherwise discard all samples). The sampling period should be shortened from a week if the available experience and forecast rain suggest that the inlet carboy could overfill, and if a programming adjustment is not possible or may not avoid it. It is best to be conservative in this regard to preserve the integrity of the sample.

4. During the first several weeks visit the site at least twice a week to check equipment operations and need for maintenance or adjustments. Download flow data on each visit. Note the flow totals, numbers of samples collected, and time lapse since the last visit.

5. Obtain rainfall totals for the same lapsed time from the King County rain gauge closest to the site that can provide information quickly, preferably via Internet (probably, Yellow Lake). If no nearby King County gauge can provide information quickly enough, use Sea-Tac Airport data.

6. Compare flow totals and duration and the number and volume of samples collected to the programmed values. Correlate these parameters with rainfall to develop the experience necessary to adjust programming to get the best hydrograph coverage without too much risk of overfilling carboys. Adjust settings according to findings if new settings would improve program logistics.

Routine Sampling

1. Once some experience accrues to give more comfort with the equipment and its settings, site visits can be reduced to one a week.

2. Continue to get regular weather forecasts.

3. Continue the weekly sample collection and analysis as specified above (unless it is necessary to shorten the period, as previously described).

4. Each week download flow data; note flow totals, numbers of samples collected, and time lapse since the last visit; and collect rainfall data.

5. Continue to compare readings to settings and what is expected with the rainfall experienced. Readjust settings as advisable.

6. Consider if analyzing all samples should continue. If a decision is made to discontinue full analysis, determine if the samples should be analyzed according to the Sample Acceptance

Criteria below. To prepare to apply the criteria, compute for each sampling period thus far: (1) total runoff volume at the swale inlet within the period, (2) total runoff duration in the period, and (3) average length of time between runoff events in the period (define an "event" as an interval with a break in runoff at the swale inlet no longer than 2 hours; i. e., a new event would be declared if the flow meter records no runoff for longer than 2 hours). Determine the 10th, 25th, 75th, and 90th percentiles of each of the three distributions (note: the formulas for this calculation will be selected from statistical methods for the appropriate distribution type, which is expected to be log-normal, when data are available).

Sample Acceptance Criteria

Note: These criteria are to be applied only after a decision is made not to analyze all samples. They are intended to guarantee a series of samples representative of the hydrologic conditions at the site. The criteria are not directly representative of meteorological conditions, because response to rainfall is damped by the ponds, whose output nevertheless reflects meteorological events.

1. If the sampling period includes snowmelt runoff and if no other sample in the winter has represented snowmelt, analyze the sample. If snowmelt runoff was analyzed earlier in the winter or if the period had only rain runoff, go to 2.

2. Compute runoff volume at the swale inlet, total runoff duration, and average length of time between runoff events for the sampling period. Determine where the period fits in each of the three distributions. Analyze the sample if:

> The values for all three quantities are within the 25th-75th percentiles for the respective distributions, or;
>
> The values for two quantities are within the 25th-75th percentiles for the respective distributions and the third is within either the 10th-25th or 75th-90th percentiles.

Otherwise, discard the sample without analysis. These criteria may be changed depending on experience, opportunities, and budget.

APPENDIX 3 – Vegetation Composition and Percent Cover

Swale	Vegetation Cover (%)	Swale Species Richness	Species	Percent of Swale Vegetation	Percent of Total Vegetation	Percent of Swale Area
PL	56.25	7	Agrotis spp. (Bentgrass, Redtop)	5.46	0.38	3.07
			Ranunculus repens (Buttercup)	37.41	2.58	21.04
			Juncus effusus (Common rush)	1.67	0.12	0.94
			Phalaris arundinacea (Reed canary grass)	9.33	0.64	5.25
			Alopecurus aequalis (Shortawn foxtail)	49.76	3.44	27.99
			Festuca occidentalis (Western fescue)	0.11	0.01	0.06
2009	75.42	8	Sparganium angustifolium (Narrow-leaved bur-reed)	30.44	2.82	22.96
			Carex stipata (Sawbeak sedge)	0.21	0.02	0.16
			Agrotis spp. (Bentgrass, Redtop)	34.63	3.21	26.11
			Ranunculus repens (Buttercup)	1.01	0.09	0.76
			Myosotis laxa (Forget-me-nots)	1.49	0.14	1.13
			Alopecurus aequalis (Shortawn foxtail)	12.06	1.12	9.09
			Holxus lanatus (Velvet grass)	0.35	0.03	0.26
			Eleocharis palustris (Creeping spike-rush)	7.51	0.70	5.67
			Salix lucida ssp. Lasiandra (Pacific willow)	13.55	0.61	3.73
1703	27.5	13	Trifolium repens (White clover)	0.43	0.02	0.12
			Taraxacum officinale (Common dandelion)	1.45	0.07	0.40
			Festuca rubra (Red fescue)	34.55	1.56	9.50
			Carex spp. (Unknown)	2.98	0.13	0.82
			Spiraea douglasii spp. Douglasii (Hardhack)	1.45	0.07	0.40
			Alnus rubra (Red alder)	0.60	0.03	0.16
			Blechnum spicant (Deer fern)	0.43	0.02	0.12
			Lotus corniculatus (Birds-foot trefoil)	0.17	0.01	0.05
			Equisetum arvense (Horsetail)	0.60	0.03	0.16
			Agrotis spp. (Bentgrass, Redtop)	2.56	0.12	0.70
			Juncus effusus (Common rush)	7.67	0.35	2.11
			Holxus lanatus (Velvet grass)	1.02	0.05	0.28
1933	85.63	7	Geranium robertianum (Geranium)	0.36	0.05	0.30

			Taraxacum officinale (Common dandelion)	0.58	0.08	0.50
			Agropyron repens (Quackgrass)	73.10	10.25	62.59
			Crepis capillaris (Smooth hawksbeard)	0.54	0.08	0.46
			Agrotis spp. (Bentgrass, Redtop)	28.14	3.95	24.09
			Ranunculus repens (Buttercup)	2.09	0.29	1.79
			Holxus lanatus (Velvet grass)	0.58	0.08	0.50
2434	10.42	3	Typha latifolia (Cattail)	71.20	0.91	7.42
			Equisetum arvense (Horsetail)	4.50	0.06	0.47
			Agrotis spp. (Bentgrass, Redtop)	24.00	0.31	2.50
1984	41.67	6	Equisetum arvense (Horsetail)	0.38	0.02	0.16
			Agrotis spp. (Bentgrass, Redtop)	1.95	0.10	0.81
			Ranunculus repens (Buttercup)	27.38	1.40	11.41
			Veronica beccabunga ssp. Americana (American brooklime)	0.05	0.00	0.02
			Phalaris arundinacea (Reed canary grass)	66.30	3.39	27.63
			Scirpus microcarpus (Small-flowered bulrush)	0.85	0.04	0.35
2124	82.5	4	Agrotis spp. (Bentgrass, Redtop)	59.30	4.01	48.92
			Ranunculus repens (Buttercup)	0.74	0.05	0.61
			Phalaris arundinacea (Reed canary grass)	4.60	0.31	3.80
			Festuca arundinacea (Tall fescue)	43.92	2.97	36.23
1750	94.38	1	Callitriche heterophylla (Aquatic-starwort)	97.50	7.53	92.02
2007	81.25	9	Medicago lupulina (Black medic)	0.87	0.09	0.71
			Dactylis glomerata (Orchard grass)	1.63	0.16	1.32
			Taraxacum officinale (Common dandelion)	1.63	0.16	1.32
			Vicia americana (American vetch)	0.87	0.09	0.71
			Rumex acetosella (Sheep sorrel)	0.87	0.09	0.71
			Agrotis spp. (Bentgrass, Redtop)	6.54	0.65	5.31
			Ranunculus repens (Buttercup)	0.87	0.09	0.71
			Holxus lanatus (Velvet grass)	1.74	0.17	1.42
			Festuca arundinacea (Tall fescue)	77.15	7.70	62.69

1903	68.13	2	Phleum pratense (Timothy)	52.27	2.92	35.61
			Agrotis spp. (Bentgrass, Redtop)	47.73	2.66	32.52
1977	91.25	8	Geranium robertianum (Geranium)	1.25	0.09	1.14
			Medicago lupulina (Black medic)	1.25	0.09	1.14
			Trifolium repens (White clover)	5.17	0.39	4.72
			Dactylis glomerata (Orchard grass)	20.00	1.49	18.25
			Phleum pratense (Timothy)	18.75	1.40	17.11
			Agrotis spp. (Bentgrass, Redtop)	7.50	0.56	6.84
			Holxus lanatus (Velvet grass)	50.00	3.74	45.63
			Festuca arundinacea (Tall fescue)	7.50	0.56	6.84
			Medicago lupulina (Black medic)	15.14	1.67	10.22
1819	67.5	7	Trifolium repens (White clover)	0.17	0.02	0.12
			Dactylis glomerata (Orchard grass)	12.67	1.40	8.55
			Taraxacum officinale (Common dandelion)	0.84	0.09	0.57
			Agrotis spp. (Bentgrass, Redtop)	60.89	6.73	41.10
			Holxus lanatus (Velvet grass)	3.24	0.36	2.19
			Festuca arundinacea (Tall fescue)	1.62	0.18	1.09
1784	61.88	5	Salix lucida ssp. Lasiandra (Pacific willow	1.64	0.08	1.02
			Ranunculus repens (Buttercup)	0.63	0.03	0.39
			Juncus effusus (Common rush)	30.40	1.54	18.81
			Veronica beccabunga ssp. Americana (American brooklime)	0.63	0.03	0.39
			Festuca arundinacea (Tall fescue)	60.13	3.05	37.20
1824	64.38	4	Agrotis spp. (Bentgrass, Redtop)	89.13	4.70	57.38
			Veronica beccabunga ssp. Americana (American brooklime)	0.95	0.05	0.61
			Holxus lanatus (Velvet grass)	4.37	0.23	2.81
			Cardamine oligosperma (Few-seeded bitter-cress)	2.14	0.11	1.38

Problematic Sites

Swale	Vegetation Cover (%)	Swale Species Richness	Species	Percent of Swale Vegetation	Percent of Total Vegetation	Percent of Swale Area
1705	52.5	7	Agrotis spp. (Bentgrass, Redtop)	0.73	0.20	0.38
			Juncus effusus (Common rush)	8.93	2.49	4.69
			Myosotis laxa (Forget-me-nots)	1.46	0.41	0.77
			Phalaris arundinacea (Reed canary grass)	1.52	0.42	0.80
			Eleocharis palustris (Creeping spike-rush)	63.27	17.66	33.22
			Polygonum aviculare (Common knotweed)	2.54	0.71	1.34
			Rubus ursinus (Trailing blackberry)	1.01	0.28	0.53
1891	20.63	6	Typha latifolia (Cattail)	17.52	1.92	3.62
			Equisetum arvense (Horsetail)	0.45	0.05	0.09
			Agrotis spp. (Bentgrass, Redtop)	15.89	1.74	3.28
			Juncus effusus (Common rush)	20.24	2.22	4.18
			Phalaris arundinacea (Reed canary grass)	40.29	4.42	8.32
			Holxus lanatus (Velvet grass)	0.23	0.02	0.05
2008	50	9	Typha latifolia (Cattail)	12.81	2.56	6.41
			Alisma plantago-palustris (Water-plantain)	4.69	0.93	2.34
			Equisetum arvense (Horsetail)	1.63	0.32	0.81
			Ranunculus repens (Buttercup)	1.63	0.32	0.81
			Scirpus microcarpus (Small-flowered bulrush)	0.13	0.02	0.06
			Holxus lanatus (Velvet grass)	1.13	0.22	0.56
			Eleocharis palustris (Creeping spike-rush)	12.75	2.54	6.38
			Callitriche heterophylla (Aquatic-starwort)	59.63	11.89	29.81
			Rubus leucodermis (Black raspberry)	0.31	0.06	0.16
2091	20	6	Typha latifolia (Cattail)	8.20	0.44	1.64
			Torreyochloa pauciflora (weak alkali grass)	2.42	0.13	0.48
			Ranunculus repens (Buttercup)	73.75	3.92	14.75
			Juncus effusus (Common rush)	1.17	0.06	0.23
			Myosotis laxa (Forget-me-nots)	1.17	0.06	0.23
			Veronica beccabunga ssp. Americana (American brooklime)	2.34	0.12	0.47

2135	67.5	10	Taraxacum officinale (Common dandelion)	8.19	1.47	5.53
			Douglas fir	0.79	0.14	0.53
			Brachythecium frigidum (Golden short-capsuled moss)	34.95	6.27	23.59
			Polytrichum juniperinum (Juniper haircap moss)	4.26	0.76	2.88
			Veronica scutellata (Marsh-speedwell)	0.79	0.14	0.53
			Agrotis spp. (Bentgrass, Redtop)	32.85	5.90	22.17
			Ranunculus repens (Buttercup)	3.66	0.66	2.47
			Rubus discolor (Himalayan blackberry)	4.24	0.76	2.86
			Juncus effusus (Common rush)	12.59	2.26	8.50
			Holxus lanatus (Velvet grass)	1.37	0.25	0.92
2216	65.83	7	Typha latifolia (Cattail)	0.54	0.14	0.35
			Equisetum arvense (Horsetail)	2.91	0.76	1.92
			unidentified	9.38	2.46	6.18
			Polygonum lapathifolium (Willow weed)	14.00	3.68	9.22
			Rubus discolor (Himalayan blackberry)	0.40	0.10	0.26
			Juncus effusus (Common rush)	63.29	16.63	41.69
			Holxus lanatus (Velvet grass)	0.54	0.14	0.35

APPENDIX 4 – Pine Lake Biofiltration Swale Sampling Episode Details

Inflow

Sample Episode	Number of Samples	Beginning		Ending		Episode Days	Episode Volume, cf	Total Week Volume, cf
		Date	Time	Date	Time			
1	118	30-Oct	22:25	4-Nov	5:23	6	7403	7403
2	200	6-Nov	0:20	9-Nov	16:59	4	11555	27697
3	200	11-Nov	12:05	12-Nov	6:07	1	22701	100926
4	199	18-Nov	14:02	23-Nov	8:43	5	23743	23743
5	200	23-Nov	12:53	29-Nov	19:00	6	45685	54821
6	187	7-Dec	11:38	14-Dec	11:51	7	43108	43108
7	200	14-Dec	14:47	20-Dec	6:49	6	45839	50557
8	117	21-Dec	12:03	28-Dec	10:39	7	13583	13583
9	115	28-Dec	17:22	4-Jan	12:19	7	13290	13290
10	137	4-Jan	14:00	10-Jan	11:35	6	15870	15870
11	103	10-Jan	15:24	17-Jan	7:38	7	23969	23969
12	52	17-Jan	12:07	24-Jan	0:26	7	12058	12058
13	21	24-Jan	16:18	31-Jan	12:02	7	2431	2431
14	88	14-Feb	14:45	21-Feb	9:47	7	20409	20409
15	40	6-Mar	15:55	14-Mar	7:25	8	9342	9342

Outflow

Sample Episode	Number of Samples	Beginning		Ending		Episode Days	Episode Volume, cf	Total Week Volume, cf
		Date	Time	Date	Time			
1	200	31-Oct	0:00	31-Oct	5:35	1	2400	3691
2	200	6-Nov	0:17	9-Nov	23:20	4	8562	16582
3	200	11-Nov	11:50	12-Nov	7:00	1	16770	65333
4	155	18-Nov	14:14	23-Nov	9:18	5	13770	13770
5	194	23-Nov	13:00	30-Nov	9:36	7	33507	33507
6	154	7-Dec	12:03	14-Dec	10:52	7	26661	26661
7	200	14-Dec	14:56	20-Dec	17:21	6	34275	36235
8	105	21-Dec	12:21	26-Dec	22:09	5	9084	9084
9	131	28-Dec	13:55	4-Jan	12:22	7	11323	11323
10	200	4-Jan	14:01	9-Jan	0:30	5	17137	26346
11	142	10-Jan	15:15	17-Jan	9:32	7	24583	24583
12	63	17-Jan	11:33	23-Jan	16:36	6	10951	10951
13	33	24-Jan	16:53	31-Jan	6:08	7	2921	2921
14	67	14-Feb	14:01	21-Feb	9:41	7	15648	15648
15	49	6-Mar	15:15	14-Mar	7:28	8	11486	11486

APPENDIX 5 – TSS and TP Period Mean Concentrations

Sampling Episode	TSS, mg/L				Inflow Volume, cf	Outflow Volume, cf
	PLIN	PLOUT1	PLOUT2	PLOUT3		
1	17.2	6.3	3.7	6.2	7403	2400
2	3.2	12.8	3.7		11555	8562
3	3.1	3.6	2.3	14	22701	16770
4	1.3	14.4		22	23743	13770
5	1.5	12.9		26.9	45685	33507
6	1.3	7.5	3	13.7	43108	26661
7	1.8	10	3.8	20.4	45839	34275
8	1.2	10.9	6	7	13583	9084
9	1.5	10.7	6.4	11.6	13290	11323
10	0.88	6.5	3.8	7.6	15870	17137
11	0.9		4.6	7.5	23969	24583
12	1.8	18.6	8.4	16.4	12058	10951
13	2.1	23.5	25	11.1	2431	2921
14	2.3	96.1	25.7	30.9	20409	15648
15	3.7	22.8	9.2	48.8	9342	11486

Sample Episode	TP, mg/L				Inflow Volume, cf	Outflow Volume, cf
	PLIN	PLOUT1	PLOUT2	PLOUT3		
1	0.0546	0.0444	0.0422	0.0436	7403	2400
2	0.0406	0.0528	0.0403		11555	8562
3	0.0364	0.0362	0.03	0.0497	22701	16770
4	0.0163	0.0427		0.0486	23743	13770
5	0.0224	0.024		0.0501	45685	33507
6	0.011	0.0155	0.0122	0.0289	43108	26661
7	0.0147	0.0278	0.0169	0.0349	45839	34275
8	0.0248	0.0312	0.0187	0.0214	13583	9084
9	0.0085	0.0232	0.0162	0.0232	13290	11323
10	0.0284	0.0177	0.0212	0.0229	15870	17137
11	0.0066		0.0117	0.0157	23969	24583
12	0.0133	0.0437	0.0196	0.0276	12058	10951
13	0.0104	0.0265	0.0363	0.0185	2431	2921
14	0.0112	0.0863	0.0439	0.0398	20409	15648
15	0.0186	0.0386	0.0214	0.0553	9342	11486

APPENDIX 6 – Conversion Factors

multiply	by	to obtain
ft	0.3048	m
ft^2	0.0929	m^2
ft^3	0.0283	m^3
ft/sec (fps)	0.3048	m/sec
ft^3/sec (cfs)	0.0283	m^3/sec
in	2.54	cm
gallon	3.785	l
lb_f/ft-s	14.594	N/m-s
lb_m/ft^2	4.883×10^{-3}	g/m^2
lb_f/in^2 (psi)	6894.8	Pa