ASSESSING THE NONPOINT SOURCE POLLUTANT REMOVAL EFFICIENCIES OF A TWO-BASIN STORMWATER MANAGEMENT SYSTEM IN AN URBANIZING WATERSHED

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Abstract

Monitoring of a regional stormwater management facility, located on the Virginia Tech campus in Blacksburg VA, was conducted in order to assess its efficacy in reducing nonpoint source pollutant losses downstream. The facility design includes both an upper water quality (wet) pond and a lower 100-yr-event quantity (dry) pond. These on-stream ponds capture both baseflow and storm runoff from the southern portion of the Virginia Tech campus and surrounding lands, and release the water back to the unnamed stream shortly above its conjunction with Stroubles Creek, a tributary of the New River. Monitoring sites for flow measurement, water quality sampling, and biotic assessments (habitat evaluation and rapid bioassessment of benthic macroinvertebrates) were located above and below each of the ponds.

Both grab samples and automated samples were collected at these stations. Between 1997 and 1999, water quality grab samples included 35 baseflow samples and 22 stormflow samples. The grab samples were analyzed for concentrations of total suspended solids (TSS), metals, bacteria, and nutrients as well as temperature, pH, dissolved oxygen, conductivity, total organic carbon (TOC), and chemical oxygen demand (COD). Automated flow-weighted sampling was initiated in February of 1999 and results are reported through the end of October 1999. Thirty-three storms in 1999 were monitored for flow and various water quality parameters (TSS, TOC, COD, and nutrients). Pollutant loads and pollutant removal estimates were calculated with regard to the wet pond, dry pond, and the combined facility. Two types of pollutant removal efficiencies were calculated: (1) the EMC efficiency, based on pollutant concentrations from individual storms; and (2) the SOL efficiency, based on pollutant loads, to estimate long-term performance over the study period. Benthic macroinvertebrate sampling and habitat assessment were performed in both 1997 and 1999. In addition, a preliminary investigation of pond characteristics was conducted, including measurements of water quality and composition, sediment deposition and composition, and residence time.

As a system, the stormwater management facility appears to have minimum impact on improving the downstream water quality. Pollutant concentrations and loads both appear to increase downstream of the facility as compared to upstream, during both storm event and baseflow periods. Monitoring results of the benthic assemblages showed evidence of moderate to high impairment at all sampling locations, and habitat assessments showed evidence of high sedimentation levels within the stream, even after installation of the stormwater management facility. Total suspended solids (TSS) concentration removal efficiency was 10% for the combined wet pond and dry pond system, much lower than the 80 to 90% TSS removal expected for properly functioning stormwater management facilities (Hartigan, 1989). There is some evidence of sedimentation within the ponds because of a slight reduction in sediment-bound constituent export, but the dissolved nutrient constituents had either very low and most often

negative (indicating pollutant export) removal efficiencies. Concentrations of metals measured in the stream often exceeded their respective acute and chronic water quality criteria at all sampling locations.

Pollutant removal efficiencies measured in the wet pond are atypical of those reported in the literature (Schueler, 1993). Insufficient residence time (two days compared to the optimal two weeks), and wet pond embankment failure are likely the principal causes of the wet pond's inadequate performance and thus, the inadequate performance of the overall facility. TSS removal efficiencies were low in the wet pond (19% for concentrations and 33% for loads) compared to the 80 to 90% expected for similar ponds. Nevertheless, the wet pond reduced the concentrations of several pollutants typically associated with TSS and not likely to be associated with the fill material for the wet pond embankment. Zinc concentrations in sediment cores were highest near the pond inlet, where the majority of sedimentation occurs. During storm events, the following results were noted. Copper and zinc concentrations in 1998 were lower at the pond outlet as compared to the pond inlet, and TOC concentrations and loads were reduced by the wet pond (13% for concentrations and 12% for loads). However, sedimentation is also expected to remove phosphorus, and wet pond phosphorus loads were only reduced by 10% and 3% for orthophosphorus and total phosphorus, respectively.

Because the wet pond is undersized with respect to the watershed it serves (surface area less than 1% of the watershed area (0.87 ha), as compared to the 3% ratio often recommended for optimal pollutant removal (Athanas, 1988)), higher removal efficiencies were found during baseflow periods. The greatest reductions in baseflow concentrations were for ammonia (67%), nitrate (57%), total nitrogen (54%), and COD (45%). However, the residence time of two days appears to be insufficient to reduce fecal coliform concentrations in the stream, and over 40% of the fecal coliform samples collected exceeded the water quality standard for contact recreation (DEQ-WQS, 1997). Furthermore, the wet pond did not appear to reduce TSS or TOC during baseflow periods. Export of TSS (-29% EMC efficiency) and TOC (-44% EMC efficiency) from the wet pond during baseflow periods is likely due to the wet pond embankment failure as well as pond eutrophication. Eutrophication processes are favored by the water temperature increase as flow passes through the shallow wet pond. The wet pond increased downstream temperatures by approximately 8°C above inflow temperatures during the summer, and to levels above 21°C that cannot be tolerated by sensitive coldwater species (Schueler, 1987).

The dry pond did not remove dissolved nutrient constituents or other pollutants during baseflow periods, but there is some evidence of sedimentation within the dry pond during storm events. During storm events, the dry pond was effective in removing TSS, with a concentration removal efficiency of 69% (EMC efficiency) and loading removal efficiency of 43% (SOL Efficiency). Removal of TKN and total phosphorus (36% and 37% respectively for concentrations) within the dry pond is further evidence of sedimentation within the dry pond.

The wet pond embankment was built in 1997. Monitoring occurred during a potential stabilization period when evidence of water quality benefits is slow to appear, especially with respect to downstream habitat and aquatic communities. Some benefits which could have been observed more immediately may have been negated or masked by the progressive erosion of the wet pond embankment because of a design flaw. Further complicating the results is the appearance; based on observations of extended drawdown time and results from a water budget analysis in the wet pond (where inflow substantially exceeds inflow); that groundwater interacts with the pond in a complicated fashion, possibly including both recharge and discharge.

To fully understand the impact of the stormwater management facility on the water quantity and quality within this tributary of Stroubles Creek, monitoring efforts should continue after the wet pond embankment is repaired and is fully operational. If biotic community improvement is desired, the stabilization period could be defined by the time necessary to flush out accumulated sediment within the channel. Monitoring efforts should also expand to include the investigation of the groundwater regime and water level fluctuations within the wet pond. Further measurements of pollutant removal processes and influences upon those processes within the wet pond should also be considered. Last, the influence of the stormwater management facility on downstream flow regimes should be investigated to assess the adequacy of its performance with regard to flow control and prevention of stream channel degradation.

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CHAPTER 1.0 INTRODUCTION

1.1 The Impacts of Urban Runoff

Urbanization replaces the natural vegetative cover in a watershed with impervious surfaces and causes runoff to be the dominant hydrologic factor. Humans seek to manage the increased runoff to reduce the threat of flooding, and the combination of this management and the increased energy associated with water entering the stream channels causes substantial modification of the physical state of the receiving streams. Furthermore, sediment and pollutants generated by human activities are deposited on the impervious surfaces between storm events and can be washed into the streams by storm runoff (Delleur, 1982). These inputs to the stream modify physical, chemical, and biological pathways and processes in the aquatic ecosystem (Imhof and Annable, 1993). Human influences impact stream ecology with respect to flow regime, habitat structure, water quality, and biotic interaction (Karr et al., 1986; Karr, 1999; Gibson et al., 1996; Barbour et al., 1999).

The increase of impervious land cover in urban environments causes two significant hydrologic impacts in a watershed. First, the hydrologic cycle is altered through decreased interception, evapotranspiration, depression storage, and infiltration, thereby decreasing the volume of baseflow and increasing the volume of direct runoff (Delleur, 1982; Imhof and Annable, 1993). For any given rainfall intensity and duration, urbanization could increase peak discharge by a factor of 2 to 5, duration of any given flow magnitude by a factor of 5 to 10, and the frequency of damaging flows moving downstream by a factor of 10 or more (Booth and Jackson, 1997). Watersheds with 20-30% impervious cover were reported to produce 10-15 times the frequency of small (1-yr recurrence) and double the volume of large (100-yr) flood events (Maxted and Shaver, 1996). In watersheds with more than 50% impervious surface, pervious surfaces affect runoff volumes but have little impact on the magnitude of flood peaks from moderate-size storms (Aron, 1982). Second, the smoother land surface (due to both impervious surfaces and storm drain networks) increases the hydraulic conveyance efficiency and thus the velocity of stormwater runoff, which in turn decreases the time to peak flow and increases the energy associated with the flow. As a result of these high-energy flows, the rate of streambed and streambank erosion and scouring increases and changes stream geometry (Van Buren, 1994). In small streams (watersheds less than 500 km^2), the most common form of erosion is channel incision, which is often followed by channel widening as the stream banks become overly steep and fail (Knight et al., 1998). This form of erosion is particularly destructive as it produces large amounts of sediment in the stream (Knight et al., 1998).

The aquatic systems' biotic conditions are limited by the quality of their physical habitat (Barbour et al., 1999). Urbanization increases the number of physical and chemical barriers to the migration of aquatic organisms; reduces channel and floodplain complexity; and simplifies, modifies, or eliminates physical habitat features essential to aquatic ecosystems (Imhof and Annable, 1993). Aquatic system degradation is readily observed when the impervious surfaces total 10% or more of the watershed area (Booth and Jackson, 1997; Schueler and Claytor, 1996). Above 25% impervious cover in the watershed, most indicators of stream quality consistently shift to a poor condition (Schueler and Claytor, 1996).

Urban runoff, due to its volume and energy, transports pollutants from areas of concentrated human activities to nearby streams. The amount of pollutants carried by stormwater runoff or snowmelt into the streams is influenced by the extent of impervious surfaces within the watershed and the mass of available pollutants. Pollutants found in urban runoff include sediment, heavy metals, nutrients, pesticides, bacteria, and viruses. The largest sediment loads originate from construction sites and other land disturbing activities that leave little to no vegetative cover on the land. Nonpoint sources of metals in urban environments primarily derive from automobile use, including fossil fuel combustion, tire wear, fluid leaks, and metal alloy corrosion. Excess nutrients may originate from excessive applications of fertilizers, applications of de-icing chemicals, and decomposition of fecal material. Bacteria and viruses may be traced to septic systems and illicit connections with sanitary sewers, or fecal material deposited in the watershed by pets, domestic animals, and wild animals (Van Buren, 1994). The extent of environmental damage by pollutants is influenced by characteristics of each watershed including soil types, topography, and quantity, frequency, and intensity of precipitation (Kibler et al., 1998). The effects of pollutants in receiving waters may include health risks; aesthetic degradation due to eutrophication and foul odors; high biological oxygen demand (BOD); and acute (lethal) or chronic damage to fish, other aquatic organisms, or their food sources from toxic metals, petroleum products, and ammonia (Van Buren, 1994).

Nonpoint sources of pollution are by definition diffuse and variable in nature. The cumulative and synergistic effects of nonpoint pollution are evidenced in the condition and function of the biotic community.

> Chemical, physical, or biological stressors impact the biological characteristics of an aquatic ecosystem (Gibson et al., 1996). For example, chemical stressors can result in impaired functioning or loss of a sensitive species and a change in community structure. Ultimately, the number and intensity of all stressors within an ecosystem will be evidenced by a change in the condition and function of the biotic community. The interactions among chemical, physical, and biological stressors and their cumulative impacts emphasize the need to directly detect and assess the biota as indicators of actual water resource impairments (Barbour et al., 1999).

The biological condition of a site can be reflected in the structure of the benthic macroinvertebrate community (Kibler et al., 1998). Benthic macroinvertebrates are considered good indicators of water quality because they occur in almost all types of freshwater habitats; they have a broad range of tropic levels and pollution/stress tolerances. Furthermore, benthic macroinvertebrates are relatively immobile and seldom move away from the influence of pollution; they live long enough that they are likely to be exposed to pollution or environmental stress; and the community cannot recover very quickly (generally only one generation per year) (Kibler et al., 1998). Damage to the benthic community also has repercussions along the food chain since macroinvertebrates are a significant source of food to fish and other aquatic and terrestrial animals. In summary, benthic macroinvertebrates are suitable for interpreting biotic condition that may be impaired due to cumulative pollutant effects and site-specific impacts (Voshell et al., 1989, Barbour et al., 1999).

Since 1996, Stroubles Creek has been listed on the 303(d) Total Maximum Daily Load (TMDL) Priority List of the Virginia Department of Environmental Quality due to partial impairment of aquatic life use along approximately 5 miles of the stream (DEQ-WQA, 1998), located predominantly downstream of the confluence with the unnamed tributary investigated in this study. Pollutants are suspected to originate from nonpoint sources associated with agricultural and urban activities in the Stroubles Creek watershed (DEQ-WQA, 1998). In particular, pH exceedances have been noted which might contribute to the benthic community impairment (DEQ-WQA, 1998).

1.2 Best Management Practices

According to the 1996 National Water Quality Inventory, a biennial summary of nationwide water quality, 36 percent of surveyed U.S. waterbodies were impaired by pollution and did not meet water quality standards (http://www.epa.gov/OW/resources/9698/chap2.html, March 17, 2000). The U.S. EPA mandates reductions in non-point source pollution associated with urban and agricultural stormwater runoff and promotes the development of new and improved methods for reducing degradation of water quality in urban areas (Allan et al., 1997). There are four objectives to improve urban stormwater quality: prevention, source control, source disposal and treatment, and follow-up treatment (Urbonas, 1994). The last objective, follow-up treatment, includes Best Management Practices and is addressed by this study.

A Best Management Practice (BMP) is a structural or nonstructural practice that is designed to minimize the impacts of development on surface and groundwater systems (DCR, 1998b). One category of BMP often used in urban settings are stormwater management facilities, defined as devices that control stormwater runoff and change the characteristics of that runoff including, but not limited to, the quantity and quality, the period of release, or the velocity of the flow (DCR, 1998b). Water quality stormwater management facilities described and defined in the 1998 Virginia Stormwater Management Regulations and Act include vegetated filter strips, grassed swales, constructed wetlands, extended detention, extended detention-enhanced, retention basins, bioretention basins, bioretention filters, infiltration, and sand filters (DCR, 1998b). This study focuses on evaluating the performances of detention basins, extended-detention basins, and retention basins. An extended detention basin temporarily impounds runoff (the basin is normally dry during non-rainfall periods) and discharges it through a hydraulic outlet structure over a specified period of time to a downstream conveyance system for the purpose of water quality enhancement or stream channel erosion control (DCR, 1998b). Detention basins discharge impounded runoff at faster rates than extended detention basins and thus provide very little water quality enhancement. Retention basins impound runoff for more than one day (Martin and Smoot, 1986). A wet pond is a retention basin with a permanent pool for enhancing water quality (Martin and Smoot, 1986). Wet ponds often have additional storage volume for reducing flooding and downstream channel erosion (DCR, 1998b).

Due to the difficulties of determining the impact of any particular facility on receiving waters (Livingston, 1989), Virginia stormwater management regulations (4VAC3-20) specify technical criteria and performance-based criteria for local and state stormwater management programs (DCR, 1998b). The Virginia Stormwater Management Regulations and Act in the Code of Virginia Law as amended through 1998 applies to every locality that establishes a local stormwater management program and to every state project (DCR, 1998b). State agencies intending to develop large tracts of land such as campuses are encouraged to develop regional stormwater management facilities where practical (DCR, 1998b). A regional stormwater management facility is a facility or series of facilities designed to control stormwater runoff from an entire watershed, though land development may only occur on a small portion of it (DCR, 1998b). Regional stormwater management facilities are considered to be economical and efficient, and are anticipated to not only mitigate the impacts of new development, but also to remediate erosion, flooding, or water quality problems caused by existing development within the watershed (DCR, 1998b). Site variations, the performance variability of stormwater management facilities, risk analysis, cost-benefit comparisons, and implementation feasibility are considered when determining which BMP should be used to meet water quality standards (Livingston, 1989).

In 1997, Virginia Tech built a regional stormwater management facility, located on the southeast tributary of Stroubles Creek in Blacksburg VA. Construction projects on the VPI&SU campus contributing to the facility include an athletic center, a parking lot, a track & soccer facility, a softball field, practice fields, a health & fitness facility, and associated road relocation. The intended function of the regional facility was to control flooding by means of the lower pond (2933-ha-mm. quantity control) and to remove major non-point source pollutants such as sediment, nitrogen, phosphorus, metals, bacteria, and organic compounds primarily by means of the upper pond (592-ha-mm quality control, 1615-ha-mm total volume). The primary reason the water quality pond was located upstream of the dry pond was to capture the first flush of pollutants off the parking lots (J.B. Sutphin, personal communication, May 25, 1999). Both water quantity and water quality objectives are addressed by such a system (Kibler et al., 1998). For example, the wet pond area is often maximized to increase the hydraulic residence time and allow for the interaction of pollutants with treatment processes. This may not allow sufficient volume for stormwater quantity control and thus a dry pond, in series, can be used to fulfill the water quantity regulatory obligations. If needed to meet water quality objectives, other BMP components could be added to the system.

1.3 Goal and Objectives

Stormwater runoff control ponds are intended to minimize the impact of hydrologic changes caused by urbanization on receiving waters; but few studies have documented their impact upon receiving-water quality (Van Buren, 1994). Site-specific monitoring is needed to determine stormwater volumes, pollutant loads, and performance of various types and combinations of management practices (Veenhuis et al., 1989).

The overall goal of this study was to assess the efficacy of the stormwater management system, consisting of both a wet pond and a dry pond in series, on reducing the loss of nonpoint source pollutants downstream and enabling recovery of the impaired stream biotic condition. The objectives included (1) assessing the nonpoint source pollutant removal efficiencies of the wet pond, dry pond, and the overall system; (2) conducting bioassessment and habitat quality surveys; and (3) comparing results to other urban stormwater control practices reported in the literature and investigating potential influences upon the performance of the stormwater management facility.

CHAPTER 2.0 LITERATURE REVIEW

2.1 Overview of Stormwater Management Facilities

A stormwater management facility is defined as a device that controls stormwater runoff and changes the characteristics of that runoff including, but not limited to, the quantity and quality, the period of release or the velocity of flow (DCR, 1998b). From the 1960's to the 1970's, storage of urban stormwater in dry detention basins was the primary means used to reduce peak flows and provide flood control (Van Buren, 1994; Schueler and Helfrich, 1989). This form of stormwater management was a means of protecting downstream urban developments from flooding, and did not examine the effects of stormwater on receiving waters or downstream portions of the watershed (Delleur, 1982). During the early 1980's, extended detention dry ponds and wet ponds were used to address additional stormwater management objectives of water quality improvement (Schueler and Helfrich, 1989). Late in the 1980s the need for greater streambank erosion protection was realized, and storm event frequency control was initiated using extended detention wet ponds for control of storm volumes less than the 2-year event but greater than the first flush (Schueler and Helfrich, 1989). Furthermore, the environmental impacts of wet ponds themselves were realized and special pond designs were sought to minimize the impacts of anoxic pond water releases, thermal loading, and freshwater wetland disturbance upon sensitive downstream aquatic life (Schueler and Helfrich, 1989).

2.1.1 Stormwater Discharge Regulations

The technical criteria for the design of BMPs in Virginia include: (1) the determination of impacts from land development projects using either a 24-hour storm according to U.S. Soil Conservation Service methods, or if other methods are used, a storm of critical duration that produces the greatest required storage volume at the site, (2) impounding structures not covered by Impounding Structure Regulations (4 VAC 50-20-10) are required to be engineered for structural integrity during the 100-year storm event, (3) pre-development and post-development rates must be calculated assuming that all pervious surfaces prior to development are in good condition, and (4) outflows from a stormwater management facility shall be discharged to an adequate channel, using velocity dissipaters as necessary to provide a non-erosive velocity of flow from the basin into the channel (DCR, 1998b).

Either performance-based criteria or technology-based criteria may achieve compliance with the water quality criteria. Both approaches refer to a target pollutant removal efficiency based on the proportion of impervious cover in the watershed. In the Virginia Stormwater Management Regulations and Act, the target pollutant is specified to be phosphorus. BMPs not specified by the regulations and/or alternate target pollutants may be allowed at the discretion of the local program administrator or of the Department of Conservation and Recreation in Virginia. Performance-based criteria include the following: (1) the pollutant discharge after development shall not exceed 90% of the pollutant discharge under pre-existing conditions or the pollutant discharge based on the average land cover conditions, whichever is greater, and (2) a BMP shall be located, designed, and maintained to achieve its target pollutant (phosphorus) removal efficiency based on proportion of impervious cover (DCR, 1998b) as listed in Table 2.1 below. The water quality volume (WQ Vol) is defined as the volume equal to the first 1.27-cm (½-in.) of runoff multiplied by the impervious surface of the land development project (DCR, 1998b).

Table 2.1 Virginia regulations for sizing retention basins (DCR, 1998b)

Water Quality BMP	**Target Phosphorus Removal Efficiency**	**% Impervious Cover**
Retention basin I (3 x WQ Vol)	40%	22-37%
Retention Basin II (4 x WQ Vol)	50%	38-66%
Retention Basin III (4 x WQ Vol with aquatic bench)	65%	67-100%

The technology-based criteria require the selection of an appropriate BMP as specified in the regulations. Design standards and specifications must be followed, and the BMP must perform at the target pollutant removal efficiency (Table 2.1).

Virginia Stormwater Management Regulations further state that properties and receiving waterways downstream of any land development project shall be protected from erosion and damage due to increases in volume, peak velocity, and peak flow rate of stormwater runoff (DCR, 1998b). Depending upon the watershed or receiving stream system, the land development project may need to provide reduction of the 2-yr post-development peak rate of runoff, or alternatively an enhanced criterion of 24-hr extended detention of the runoff generated by a 1-yr, 24-hr duration storm (DCR, 1998b). To protect downstream properties and waterways from flooding, the 10-yr post-development peak runoff rate from the developed site should not exceed the 10-yr pre-development peak runoff rate (DCR, 1998b).

2.1.2 Water Quality Standards and Criteria

The national objective stated by the Federal Clean Water Act is to “restore and maintain the chemical, physical, and biological integrity of the nation’s water” (Livingston, 1989). In particular, sections 303 and 304 of the CWA require states to protect biological integrity as part of their water quality standards (Barbour et al., 1999). Biological integrity is defined as “the ability of an aquatic ecosystem to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats of a region” (Karr and Dudley, 1981; Kibler et al., 1998). Section 319 of the CWA requires states to develop management programs (BMPs and implementation schedule) for any water body in which water quality standards cannot be met by point source controls alone, if they wish to be considered for federal appropriations (Hodges, 1997).

Standards are developed to protect designated beneficial uses from degradation. They may use water quality criteria for regulation and enforcement, but other factors are also considered including local conditions, the importance of the waterway, economic considerations, and the desired degree of safety (Livingston, 1989). Virginia water quality standards require all state waters to be “free from substances…in concentrations, amounts, or combinations which contravene established standards or interfere directly or indirectly with designated uses of such water or which are inimical or harmful to human, animal, plant, or aquatic life” (DEQ-WQS, 1997). “As a minimum, existing instream water uses and the level of water quality necessary to protect the existing uses shall be maintained and protected” (DEQ-WQS, 1997).

A few standards exist for nutrient and bacteria levels in receiving waters. These include a maximum contaminant level for nitrate of 10-mg/L, an alga promotion concentration for nitrate of 0.1-mg/L, and an algae limiting concentration for phosphorus of 0.01-mg/L (Mostaghimi et al., 1989). The standard for fecal coliform bacteria is designated to be no more than a geometric mean of 200 fecal coliform bacteria per 100 ml of water (for recreational activities) for two or more samples from a given water body over a 30-day period (9 VAC 25-260-170: DEQ-WQS, 1997).

Water quality criteria are constituent concentrations or levels “associated with a degree of environmental effect upon which scientific judgement may be based” (Livingston, 1989). Below the concentration or level specified (which includes the incorporation of a degree of safety), the stream is thought to remain healthy, based on knowledge of the stream assimilative capacity and the effects of site-specific effects of specific pollutants (Livingston, 1989).

“Water quality criteria traditionally have been based on acute and chronic toxicity bioassay tests that were derived for continuous pollutant sources” (Livingston, 1989). This is likely not appropriate for application to systems with urban runoff which is characterized as intermittent and variable and described as short duration shock loading with long times between exposures (Livingston, 1989). Long-term and cumulative impacts due to sediment loads and sediment bound toxics are also not generally accounted for by traditional water quality criteria (Livingston, 1989).

Some water quality criteria have been designated for specific regions of Virginia. In the mountainous zones of Virginia, the minimum dissolved oxygen (DO) criterion is 4.0-mg/L, with a daily average of 5.0-mg/L (9 VAC 25-260-50) (DEQ-WQS, 1997). Urban runoff can depress the DO levels within receiving waters (Osborne and Herricks, 1989). Dissolved oxygen is affected by carbonaceous and nitrogenous oxygen demand, sediment oxygen demand, reaeration, and plant and algae photosynthesis and respiration (Weatherbe et al., 1993). In all waters of Virginia, including the mountainous zones, the water quality criteria set for pH is a range, from 6.0 to 9.0, which should support all designated uses; exceptions are made for wetlands when the natural pH differs from this range (9 VAC 25-260-50: DEQ-WQS, 1997). The maximum temperature allowed in the New River and its tributaries from the Montgomery-Giles County line upstream to the Virginia-North Carolina state line is 29°C (9 VAC 25-260-310: DEQ-WQS, 1997), and any rise above natural temperature shall not exceed 3°C without the influence of any point-source discharge (9 VAC 25-260-60) (DEQ-WQS, 1997).

Water quality criteria for toxic substances, applicable across the state of Virginia, recognize that multiple parameters may influence the concentration at which each substance becomes toxic. In general, Virginia regulations require that instream water quality conditions shall not be acutely or chronically toxic, except as allowed for mixing zones (9 VAC 25-260-140: DEQ-WQS, 1997). The methods for the determination of toxicity for several metals and ammonia are specified in the Virginia State Water Control Board Water Quality Standards (DEQ-WQS, 1997). Freshwater aquatic life criteria for cadmium, copper, and lead are expressed as a function of total hardness as $CaCO_3$ (mg/L) and as a function of the pollutant's water effect ratio (WER) (9 VAC 25-260-140F: DEQ-WQS, 1997). The water effect ratio (WER) is the ratio of the toxicity of a metal to standard test organisms as determined simultaneously in receiving water and laboratory water (assumed to be 1.0 unless otherwise determined) (DEQ-WQS, 1997). The equations and tables used to determine the acute and chronic criteria for metals and ammonia in freshwater systems can be found in Appendix A. (DEQ-WQS, 1997).

Criteria-based management tends to address the concentration, duration of exposure, and return frequency of specific pollutants; however, it fails to incorporate temporal and spatial issues important to habitat and water quality as influenced by urban stormwater runoff. Some of the temporal issues not included in typical water quality criteria include the timing, magnitude, and return frequencies of flow conditions (Osborne and Herricks, 1989).

> …the variation in the composition, concentration, amplitude, rate of change and mass loading of toxicants in urban runoff prevents lotic communities from acclimating to urban runoff. The expected effect would be a community in constant flux and an ecosystem lacking essential stability (Osborne and Herricks, 1989).

The stream taxa that remain in an urban stream must be able to tolerate some level of poor water quality and habitat instability (Osborne and Herricks, 1989). These taxa must also be able to utilize transient low-quality food sources because of the shorter retention time of food sources within the stream (Osborne and Herricks, 1989).

In 1994, a toxicity test was conducted using flow-through aquaria on Lincoln Creek, an urban stream located in Milwaukee Wisconsin (Crunkilton et al., 1996). While the "base flow demonstrated the same magnitude of toxic effects in test organisms as high flow, stormwater runoff was clearly responsible for the many potentially toxic contaminants detected in stream water during the study" (Crunkilton et al., 1996). Crunkilton et al. (1996) suggests that there is a “mechanism where urban runoff delivers contaminants to the stream channel with the toxic effects being manifested after base flow has resumed.” They further suggest “resuspended sediments may be more important than short-term discharges from first flush runoff and contribute to delayed toxicological effects” (Crunkilton et al., 1996). Not enough is known about toxicity effects within receiving waters, especially with regard to duration of exposure (involving event specific responses, responses to multiple events, and responses to contaminant residuals after storm events), and exposure to multiple stressors at once (Herricks et al., 1996). The variable concentrations and duration of exposure within receiving waters makes the results of typical laboratory-scale tests, and therefore standards based on those results, inapplicable to receiving waters.

Spatial considerations are also not typically accounted for by water quality criteria and associated management strategies (Osborne and Herricks, 1989). Several examples of spatial criteria worthy of consideration follow. In the absence of pre-impact habitat conditions, empirical relationships have been sought to relate location within a watershed to expected species richness, abundance, and biotic indices. Spatial location could specify the metrics, criteria, or models appropriate for determining habitat and water quality specifications for urban stream management in a particular stream (Osborne and Herricks, 1989). For instance, fish abundance patterns have been shown to correlate well with drainage area (even better than their correlation with stream order and link number) (Osborne and Herricks, 1989). Impact assessments can use these correlations to assess a stream according to its biotic potential (Osborne and Herricks, 1989).

Fish abundance also spatially correlates with habitat availability measures including depth, velocity, and substrate type (Osborne and Herricks, 1989). Frequencies of use curves based on these three habitat measures have been generated for many species and geographic regions. This information may not be very useful in urban environments unless data predicting how pulse runoff events affect these measures in specific stream reaches are available (Osborne and Herricks, 1989).

Location within a watershed should be a consideration in the development and application of water quality criteria. While most water quality criteria are based upon the pollutant concentration, pollutant loads may be more appropriate for regulatory use in certain circumstances (Osborne and Herricks, 1989). For upstream waters, where dilution capacity is limited, criteria based on concentrations may be applicable (Osborne and Herricks, 1989). However, downstream dilution capacities and stream assimilative capacities may be greater, and thus large loads may have limited acute effects and yet causes detrimental cumulative impacts (Osborne and Herricks, 1989). This relationship also applies when comparing streams vs. ponds or reservoirs; concentrations may be of greater importance in the evaluation of stream impacts (lower dilution potential), while loads may be more important in the evaluation of water quality within ponds and reservoirs (Osborne and Herricks, 1989). Another issue associated with choosing the method of water quality criteria determination is that load measurement is inherently more difficult than measurement of concentrations, and increases both cost and potential sources of error associated with monitoring programs (Osborne and Herricks, 1989).

The development of biocriteria from multimetric indices adjusted for stream classes and designated aquatic life uses can account for some of the spatial and temporal concerns missing from traditional water quality criteria. A quantitative regional biocriterion is generally chosen as a percentile along the distribution of indicator scores representative of minimally impaired sites (Barbour et al., 1995). Though more than 85% of state water quality agencies in the United States use some form of multimetric biocriteria to monitor their aquatic resources (90% of those use benthic macroinvertebrates in the assessment) (Evans, 1997), biological criteria are not referenced in the regulatory standards of many states, including Virginia, except in narrative form as required by the U.S. EPA.

2.2 Design of Stormwater Management Facilities

BMPs in urban areas are installed with the expectation that they will minimize the impact of either the quantity or quality of urban runoff on downstream receiving waters (Driscoll et al., 1986). Usually BMPs are located as close to the source of human impact as possible to minimize upstream damage. In order to minimize erosion of upstream channels and minimize embankment height, Virginia and Maryland have used a maximum drainage area of 40 to 120-ha when designing regional detention basins; the lower end of this range is more typically assigned to highly impervious watersheds (Hartigan, 1989).

Although there are many BMP options available, often the selection comes down to a choice between a wet pond and a dry pond. Dry ponds are usually less expensive, but other considerations may come into play when choosing between extended dry ponds and wet ponds: (1) the permanent pool of a wet pond is considered to be aesthetic and can raise property values; furthermore it can hide the accumulated sediment and debris, and thus require less frequent cleaning, (2) the drainage area and the local soils may not be able to sustain a permanent pool, (3) existing wetlands may be destroyed by creation of a pond, and (4) receiving waters downstream of the pond outlet may be critical habitat for organisms that are sensitive to pond effluent (Hartigan, 1989).

In Northern Virginia, the wet pond storage volume requirements are based on an average hydraulic residence time of two weeks, while the dry detention pond volume requirement is based on capturing the first flush of pollutants (designated as the first 20 to 23-mm of runoff for each impervious acre (0.4047 ha)) (Hartigan, 1989). The first flush refers to the higher concentrations of pollutants that are expected to occur in the earliest portions of the runoff as compared to that in the latter runoff; this effect will vary depending on land use, the drainage basin characteristics, and the pollutant of interest (Livingston, 1989). Research has indicated that 95% of pollutants are washed off in the first 12.7-mm of rainfall from small watersheds (40 ha) (Licsko et al., 1993). Usually this effect is not as important in larger watersheds or where impervious surfaces do not predominate (Licsko et al., 1993; Livingston, 1989). While dry detention basins may take up less storage space and cost less, wet detention basins designed for a detention time of two weeks may achieve two times greater average nutrient removal with a relatively small increase in cost (Hartigan, 1989). Because of their potential to remove dissolved bioavailable nutrients from incoming flows, wet ponds may also be more appropriate when the receiving water of interest has high eutrophication potential (lakes for example) (Hartigan, 1989).

Detention volumes for BMPs are typically designed to contain the equivalent volume of a given depth of water over the area of land disturbed by construction activities. The detention volume stores runoff from a post-development design storm (based on depth-duration-frequency analysis of historical data) and discharges it at pre-development flow rates (Van Buren, 1994), hopefully producing a better quality outflow than inflow. Often this design methodology produces inadequate water detention time, thus few associated water quality benefits.

Various researchers have presented several modifications to this design methodology. First, due to the variable nature of stormwater runoff and pollutant transport, it is generally recognized that design of any BMP should be based on a long-term assessment of climate and BMP performance, rather than on a single design storm approach (Yu et al., 1993). Second, the relative size of the basin to the watershed is often recommended as a design parameter. Yousef and Wanielista (1993) produced graphs relating the removal of suspended solids and total phosphorus in detention ponds to the ratio of detention basin volume divided by the runoff volume of the mean storm event for a given location. Removal rates of 50% or more were associated with pond volumes 3 or 4 times the average runoff volume, or alternatively a surface area of at least 3% of the watershed (Athanas, 1988). Maristany (1991) suggested that a pond surface area of 2% of the watershed area could achieve high removal rates for total phosphorus when the pond is deep. A permanent pool volume of 4 to 6 times the runoff volume from the local mean storm event was recommended for maximum removal efficiency by the U.S. EPA Nationwide Urban Runoff Program (NURP) which studied nine wet ponds, among other stormwater best management practices (Yousef and Wanielista, 1993). According to the NURP study, removal of both nitrogen and phosphorus increases as pond surface area (SA) increases, relative to watershed surface area (DA) (Athanas, 1988). Wu (1989) found in his study of Piedmont North Carolina wet detention ponds that a SA/DA ratio of 1% to 2% would be necessary to achieve the following removal efficiencies; TSS (70%), iron (60%), zinc (40%), TKN (30%), and TP (45%). A minimum of a 2% SA/DA ratio would be required to achieve 80% or more of TSS removal (Wu, 1989). Third, detention time itself can be modeled and used to modify BMP design. In one modeling study of detention ponds, it was concluded that at least 72 hours of detention time was needed to remove over 95% of suspended solids and 30-70% of nutrients and heavy metals (Hvitved-Jacobsen et al., 1989). Hvitved-Jacobsen et al. (1989) suggested that to achieve a minimum detention time, statistical analysis of rainfall should be used to design pond volumes based on the inter-event dry period and the desired pollutant removal effectiveness (Hvitved-Jacobsen et al., 1989). Rainfall volume – inter-event dry period – frequency curves can be developed and used to calculate design storms (Hvitved-Jacobsen et al., 1989). This approach would also reduce the risk of cumulative effects from successive storms with short inter-event dry periods (Hvitved-Jacobsen et al., 1989).

There are two typical approaches to wet detention basin design. The first approach relies upon solids settling theory, which includes the processes of advection, turbulent diffusion, particle aggregation and disaggregation, deposition, and scouring, to account for the majority of pollutant removal (Hartigan, 1989; Krishnappan et al., 1999). For non-residential watersheds with relatively high levels of imperviousness, the pond surface area to watershed area ratio should be in excess of 3% in order to achieve high levels of sedimentation based on the solids settling design model (Hartigan, 1989). The other approach views the wet pond "as a lake achieving a controlled level of eutrophication, in an attempt to account for biological and physical/chemical processes that have been documented as the principal nutrient removal mechanisms" (Hartigan, 1989). This more conservative approach (yielding a basin approximately three times the size designed by the sedimentation model) is recommended when nutrient control is the primary water quality goal (Hartigan, 1989). This second design approach usually incorporates a detention time of at least 2 weeks, or a ratio of permanent pool storage (VB) to mean storm runoff (VR) greater than or equal to four (Hartigan, 1989).

One of two different flow models is generally assumed in the design of a wet detention basin: completely mixed, or plug flow. The choice between the two models usually depends on the ratio of design storm runoff volume to pond storage volume (Martin, 1989). When the entire flow from the majority of storms can be contained within the basin, plug flow is desired as a model since it ensures that all the water will be retained and subject to removal processes within the basin (Martin, 1989). When the storm runoff volume, relative to the storage volume, is large it is possible that the entire contaminant load could be passed downstream with little pollutant reduction under the plug-flow model (Martin, 1989). However, if the flow is completely mixed under similar circumstances, the peak concentrations could be reduced below the receiving water quality standard by means of dilution by the pond water as well as the incoming runoff (Martin, 1989).

Using the RESPOND model, Wu and Yu (1995) examined hydraulic behavior in ponds and found that the larger the length to width ratio, the higher the removal efficiency and the longer the detention time (Wu and Yu, 1995). Their results showed an increase in the average pollutant removal efficiency of 15-30% and an increase in detention time of 8-12% for a length to width (L/W) increase from 1 to 5. The majority of the pollutant removal improvement (10-20%) occurred with the increase of L/W ratio from 1 to 3 (Wu and Yu, 1995). Other researchers have found that a L/W ratio between 2 to 4 minimizes short-circuiting, enhances sedimentation, and helps to prevent vertical stratification (Ellis, 1989; Hartigan, 1989).

The ideal basin flow of wet ponds is primarily compromised by short-circuiting. Short-circuiting reduces both the hydraulic and trap efficiencies (Ellis, 1989). Short-circuiting is influenced by four factors: (1) currents induced by the basin shape and position of the inlet and outlet, (2) turbulence dispersion, (3) currents caused by density or temperature stratification, and (4) wind-induced surface currents (Ellis, 1989).

Perhaps the most effective prevention of short-circuiting can be achieved in the inlet design (Ellis, 1989). The inflow needs to be distributed across the cross-sectional area of the settling zone (Ellis, 1989). Some of the designs that include this characteristic use submerged weirs, gradually expanding inlets, or the use of inlet baffle walls or stepped inlets. Stepped inlets also provide aeration to the pond waters (Ellis, 1989). Furthermore, the inflow velocity should not exceed 0.3 m/s to ensure deposition (Ellis, 1989).

A second design alternative, the addition of islands or baffles, can improve flow patterns, prevent short-circuiting, improve overall mixing within the pond, increase hydraulic residence time, aid with re-oxygenation of the water, and reduce 'dead' zones (Ellis, 1989; Wu and Yu, 1995; Anderson et al., 1996). Baffles are particularly effective with storms of high rainfall intensity (Wu and Yu, 1995). Islands impede short-circuiting currents and, if graded properly, have the additional benefits of being aesthetically pleasing and acting as a natural wildlife refuge for wildlife and plants (Ellis, 1989).

The removal rate of pollutants depends primarily upon the surface overflow rate, but pond depth may affect the removal rate, if settling is not the only removal mechanism (Wu and Yu, 1995). While deeper areas of stormwater management ponds will increase the amount of sediment storage, and thus decrease the dredging rate, shallow areas "promote the growth of emergent and submerged vegetation, and produce higher water temperatures with greater sunlight intensities, factors that aid the growth of algae and bacteria. Shallow water also means that a higher percentage of the water column will be exposed to plant and soil surfaces, important locations for the occurrence of biological reactions" (Athanas, 1988). When the perimeter of a pond is gradually sloped and planted from 0.6-m below to 0.3-m above the permanent pool, the wetland vegetation can remove nutrients from the pond water and help keep algae growth down (Hartigan, 1989). Furthermore, a mean depth of 1 to 3-m should be maintained in the permanent pool of a wet pond, shallow enough to minimize the risk of thermal stratification (no greater than 4 to 6-m), but deep enough to discourage algal blooms and to minimize sediment re-suspension (Hartigan, 1989). In one study by Cunningham (1993), comparing a 2.7-m pond with a 1.1-m pond in Florida (adjacent stormwater ponds constructed for the study), it was found that the deeper 2.7-m pond became stratified frequently during warm and cold weather periods, while the shallower 1.1-m pond only stratified during extended periods of hot weather.

2.2.1 Dry and Extended Dry Detention Basins

A dry detention basin has an outlet smaller that its inlet and thus reduces the rate of stormwater entering the receiving water. An extended dry detention basin releases water at a slower rate than typical dry detention basins. Due to the temporary detention of the water, some sedimentation of particulate and adsorbed pollutants can be achieved (Athanas, 1988). Dry extended detention ponds, even with short (6 to 12-hrs) detention times, can be moderately effective in removing particulate pollutants, such as TSS, trace metals and organic nutrients (Schueler and Helfrich, 1989). Based on a number of studies, a properly designed extended detention basin can be expected to achieve the following long-term removal rates: TSS (50-70%), total phosphorus (10-20%), nitrogen (10-20%), organic matter (20-40%), lead (75-90%), zinc (30-60%), hydrocarbons (50-70%), bacteria (50-90%), Cu (41%), and COD (35%) (Horner and Wonacott, 1985; Yu et al., 1994; Urbonas and Stahre, 1993). The major disadvantage of the short-term detention of dry ponds is that it does not remove much of the dissolved pollutant load of stormwater runoff due to insufficient contact time of the pollutants with biological mechanisms of removal (Athanas, 1988). Other disadvantages include: (1) smaller more frequent storm events flowing into a dry detention basin may not be retained long enough for significant settling of pollutants to occur, (2) the first flush of larger storms may not be detained significantly, and (3) often sediment and its adsorbed pollutants are resuspended during the subsequent storms and lost downstream (Adams and Dove, 1984; Kibler et al., 1998).

2.2.2 Wet Detention Basins

Wet ponds (also called retention ponds) typically have a permanent pool with temporary storage above the permanent pool elevation. Primary benefits of a wet detention basin include its capture of first flush pollutants in the permanent pool, and its longer hydraulic retention time that should promote processes of sedimentation, pollutant degradation, and transformation. The permanent pond may also prevent resuspension and loss of previously deposited sediments and associated pollutants during subsequent storm events.

“The EPA Nationwide Urban Runoff Program (NURP) studies…demonstrated that wet detention basins exhibit some of the highest pollutant removal efficiencies of any Best Management Practice (BMP)” (Hayes et al., 1993). “Monitored average pollutant removal efficiencies for wet detention basin BMPs are on the order of 2 to 3 times greater than extended dry detention BMPs in the case of total P (50%-60% vs. 20%-30%) and 1.3 to 2 times greater in the case of total N (30-40% vs. 20%-30%)” (Hartigan 1989). These pollutant removal efficiencies were estimated by assuming an average hydraulic residence time of 2 weeks or greater for the permanent pool of a wet detention basin, and 12 to24 hours of detention time for the extended dry detention basin with a storage capacity of 25.4-mm of runoff per impervious acre (0.4047-ha) (Hartigan, 1989).

Wet ponds may remove dissolved phosphorus and dissolved nitrogen (Hartigan, 1989) particularly if free-floating, submerged, and emergent vegetation colonize the site (Hartigan, 1989). In one evaluation of wet detention basins by Hartigan (1989), removal efficiencies for dissolved bioavailable nutrients ranged from 50% to 70%. For all other pollutants measured the removal efficiencies of wet ponds and extended dry ponds were similar: 80%-90% for TSS, 70%-80% for lead, 40%-50% for zinc, and 20%-40% for chemical oxygen demand (COD) (Hartigan, 1989).

Though wet ponds have been promoted heavily for urban water quality improvement, there are still problems in using them as a BMP. “Stormwater ponds can adversely impact the aquatic ecosystem by regulating the flow regime, increasing thermal loads, and changing patterns of production in streams” (Schueler and Helfrich, 1989). Research continues to investigate the ecological impacts of wet ponds in order to mitigate their impacts (Schueler and Helfrich, 1989). Some examples of wet pond deficiencies are described here.

Removal rates of pollutants decrease when the incoming runoff volume is either very large or very small compared to the permanent pool volume (Schueler and Helfrich, 1989). When runoff volume is large, there may be insufficient hydraulic residence time to treat the incoming pollutants (Schueler and Helfrich, 1989). When runoff volume is small, or during periods of baseflow, the export of pond seston (fine-grained organic matter generated internally to the pond) and associated pollutants may exceed the inflow concentrations or loads for those pollutants (Schueler and Helfrich, 1989).

Excessive nitrogen and phosphorus from decay of organic matter or leachate runoff from high-fertilizer-use areas, generally found in soluble and bioavailable forms, can lead to eutrophication of surface waters (Van Buren, 1994). Eutrophication effects include the decrease of species diversity; increase in plant productivity; change of the dominant biota; increase in turbidity and the rate of sedimentation; and the increased frequency of anoxic conditions (Van Buren, 1994; Mason, 1996). The organic matter that accumulates in the pond or enters it via runoff is oxidized by the degradation processes of bacteria and discoloration of the water, odors, and depressed oxygen levels may result (Van Buren, 1994). The pond is not intended to be high quality habitat since it is designed to trap pollutants and prevent their movement downstream (Van Buren, 1994). However, oxygen depletions in the pond may create depressed oxygen levels in its outflow and thus adversely affect aquatic life downstream (Van Buren, 1994). Furthermore, the oxygen depletion of the pond may create anoxic zones within the pond that can promote the release of toxic substances from the deposited sediment (Van Buren, 1994). If excessive nitrogen and phosphorus are not removed from the water prior to discharge from the stormwater pond, these nutrients promote the growth and formation of dense mats of green algae in the stream channel (Van Buren, 1994) that increase BOD and decrease dissolved oxygen concentrations as decay progresses. Blue-green algae in particular do not provide a good substrate with respect to habitat or chemistry. The algae attach to the gravel substrate of the streams in shallow areas and interfere with the riffle habitat’s ability to support macroinvertebrate communities (Van Buren, 1994).

2.2.3 Extended Detention Wet Ponds

The extended detention wet pond system is designed to minimize the impacts of a typical retention pond and maximize its potential benefits. The extended detention wet pond has three storage components (permanent pool, extended detention storage, and stormwater storage) and a shallow fringe marsh that together work to maximize urban pollutant removal, reduce pond maintenance, and provide pond safety and aesthetics (Schueler and Helfrich, 1989). Due to its treatment of a wide range of runoff volumes, the extended detention wet pond stormwater management system is considered by Schueler and Helfrich (1989) to "provide greater pollutant removal and downstream erosion protection than can be achieved by an individual wet pond or dry extended detention pond alone."

Schueler and Helfrich (1989) recommend a standard design and several modifications for different situations. In the standard design, the permanent pool is sized for the first flush, extended detention storage is set to provide 24 hours of detention for the next ½-inch of runoff, and stormwater management storage is provided to reduce the post development peak discharge rate to pre-development levels for a two-year storm event (Schueler and Helfrich, 1989). "The water level in the permanent pool is established by a reverse-sloped pipe…that withdraws water from the pool one to three feet below the normal pool elevation, and has an invert at the top of the permanent pool. The diameter of the pipe is set by an adjustable gate valve within the concrete riser to provide the required extended detention release rate for the pond" (Schueler and Helfrich, 1989). Control of large storms (two and ten year design storms) is provided by a weir in the reinforced concrete riser, while a 100-year storm event is passed through an emergency spillway (Schueler and Helfrich, 1989). Two wetland areas are created around the permanent pool; one is designed as an aquatic bench 0.3-m below the permanent pool, the second is formed due to frequent inundation of the extended detention storage (Schueler and Helfrich, 1989). Basin landscaping (both aquatic and terrestrial) is important to stabilize slopes, create wildlife habitat, achieve pollutant removal performance, gain acceptance from adjacent property owners, and minimize maintenance (Schueler and Helfrich, 1989). The species chosen should be regionally native; able to withstand extremely compacted soils, full sun, and exposure; perform pollutant removal as desired; provide food and cover for wildlife, birds, and predaceous insects, and have few maintenance requirements (Schueler and Helfrich, 1989).

The weaknesses of each individual component of the extended detention wet pond system are addressed by the other components. Large storms that would normally pass through wet ponds receive extended detention treatment; small storms that would normally pass through dry ponds receive treatment in the permanent pool; the pool prevents significant resuspension of deposited sediments and prevents export of plant nutrients in the form of detritus; and the wetland perimeter provides additional pollutant uptake and degradation and helps to stabilize sediments and promote settling (Schueler and Helfrich, 1989).

Four approaches have been used in the design of extended detention wet ponds to minimize the thermal impacts of urbanizing watersheds (caused by the impervious surfaces warming the runoff and the pool acting as a heat sink during the summer) (Schueler and Helfrich, 1989). Mitigation of thermal impact is especially necessary for watersheds with cold-water trout streams. The four mitigation options are as follows: (1) prohibit the use of a permanent pool, (2) bypass the pool with baseflow, (3) bypass the pool with baseflow and use an undersized permanent pool, or (4) use a deep permanent pool and position the extended detention pipe deep to release cooler water from the pond (Schueler and Helfrich, 1989). Furthermore, when low-oxygen content water is released from the permanent pools (either from deep stratified pools or shallow pools with high sediment oxygen demand), its impact downstream can be minimized by increasing the roughness of the outlet structure (including rip-rap cascades or surge stones within the barrel) to promote re-aeration of the outflow water (Schueler and Helfrich, 1989).

2.2.4 Wetlands

Wetlands are generally used to polish the effluent quality from other BMP systems or to enhance the removal of wet ponds if incorporated into the design appropriately. Wetlands provide biological uptake of pollutants via the plant roots, which as a result may free some sediment sorption capacity (Van Buren, 1994). Emergent plants both help to stabilize the sediments and improve settling characteristics in the pond through their action as baffles to flow (Van Buren, 1994, Schueler and Helfrich, 1989). Biofilms may form on the plant surfaces and further enhance pollutant removal through bacterial degradation processes (Van Buren, 1994). Aquatic plants and bacteria may also aid in the transformation of pollutants from a toxic or unstable form to a more benign state (Van Buren, 1994).

Efficiencies can be moderate to high if the wetlands are large enough relative to the size of the watershed (Schueler and Helfrich, 1989). One study of constructed wetlands for stormwater management (Yu et al., 1998b) showed average removal rates as high as 90% for TSS, 65% for COD, 70% for total phosphorus and orthophosphate, and 50% for zinc. Mean Event Mean Concentration (EMC) reductions are more conservative, and were found by Yu et al. (1998b) to be as high as 57% for TSS, 50% for COD, 68% for total phosphorus, 81% for orthophosphate, and 43% for Zn. Due to complex biogeochemical cycling within wetlands, nutrient removal is highly variable, but well designed wetlands may remove 25% of total nitrogen and 45% of total phosphorus over the long-term (Yu et al., 1998b). Peak runoff reductions by wetlands were measured to be about 40%, but in combination with a detention basin, peak attenuation greater than 90% can be achieved (Yu et al., 1998b). Of concern are the low and negative removal rates typical during senescence between growing periods due to export of plant detritus, nutrients, and other organic matter (Schueler and Helfrich, 1989; Wotzka and Oberts, 1988).

2.2.5 Maintenance

The growth of algae and macrophytes within ponds is beneficial to the goals of reducing not only nutrients in ponds but also solids, bacteria, and toxins; furthermore, the plant growth can promote dissolved oxygen recovery (Ellis, 1989). However, these benefits will not be fully realized unless a maintenance program is followed that includes regular plant harvesting (preferably by mechanical means) (Ellis, 1989). It has been shown that as much as 30 to 50-kg/ha/yr of phosphorus and 400 to 500-kg/ha/yr of nitrogen can be removed by harvesting reed ponds (Ellis, 1989). Without harvesting, the degradation of plant material after senescence will release many pollutants back to the system.

Often maintenance of stormwater management facilities is overlooked, especially if the property changes owners or the owners are not committed to a maintenance program. Regardless of ownership, ease of maintenance should always be considered in the BMP design.

Settled solids reduce the amount of detention capacity, which has implications both for performance over time as well as maintenance frequency. Several reasons exist to make the sediment holding capacity of BMPs as large as possible. First, the rate of sediment accumulation appears to decline geometrically with increasing pond surface area as a percentage of total drainage area (Yousef et al., 1994a). Second, “excavating sediment during construction is approximately five times cheaper than dredging it later” (Schueler and Helfrich, 1989). Since over 50% of sediment can settle within the first 1 to 2-hrs of detention, it appears that even small forebays can trap appreciable pollutants. More frequent cleaning is required when using smaller forebays (Driscoll, 1989). When the performance of the BMP is degraded by excessive sedimentation, maintenance is made easier and less expensive by (1) designing a drain pipe to completely draw down the permanent pool within 24 hours, (2) providing direct access to the forebay (including a maintenance right-of-way), and (3) reserving a site for on-site sediment disposal which could reduce sediment removal costs by as much as 50% (Schueler and Helfrich, 1989).

2.3 Contaminant Removal

Pollutant removal processes in ponds include biological incorporation, adsorption, bio-degradation (aerobic and anaerobic), biotransformation (including nitrification and denitrification), volatilization, immobilization (including precipitation), and burial. Biological transformations of pollutants carried by stormwater runoff include transformations of pollutants as a byproduct of other processes, uptake and incorporation into living cells or volatilization through the organism, and the loss of nutrients from organisms due to secretion and decomposition (Athanas, 1988). For biological processes to act on the pollutants, there must be contact between the responsible organisms and the pollutants, and there must be time for the processes to occur. Maintaining a pool of water allows for populations of aquatic plants (macrophytes, algae, phytoplankton), fungi, bacteria, and zooplankton to develop (Athanas, 1988). The retention time of the pollutants and water can be increased by increasing the volume of the pond and slowing the rate of flow through the basin (Athanas, 1988).

Some pollutant removal mechanisms, in particular nitrification and denitrification, require both aerobic and anaerobic environments. Wetland vegetation can create aerobic microsites around their roots (Mitchell et al., 1995). Wetland vegetation increases the variety of biogeochemical reactions that may occur in a given area (Athanas, 1988) and thus facilitates improvement of water quality (Mitchell et al., 1995). Vegetative growth (emergent, submerged, and floating) uses incoming dissolved and adsorbed nutrients, and may uptake some heavy metals. As mentioned before, the plants themselves are a substrate for microorganisms, and physically impede water flow, thus promoting sedimentation.

Sorption and desorption reactions occurring at the interface between the water column and the bottom sediment can greatly affect the removal of nutrients and metals from ponds (Van Buren, 1994). Aerobic conditions and a two-week detention time will allow pond sediments to adsorb orthophosphorus, total phosphorus, and ammonia at a significant rate; these could however be slowly released from sediment under anaerobic conditions (Van Buren, 1994).

2.3.1 Analytes

2.3.1.1 Sediment

"Siltation is the most common pollutant affecting surveyed rivers and streams" and "contributes to 51% of all the water quality problems" (http://www.epa.gov/OW/resources/9698/chap2.html, March 17, 2000). Sediment in streams may originate from erosion of stream banks and beds, erosion of soil by overland flow (especially over construction sites and unpaved driveways and streets), and erosion of accumulated deposits on impervious surfaces (Marsalek, 1997).

Sedimentation is considered the most important process enhancing water quality in stormwater ponds (Van Buren, 1994). This process is affected by factors including pond size (depth and volume) relative to runoff volumes and watershed area, frequency of runoff events, flow conditions in the pond (overflow rate, residence time, short-circuiting, and turbulence), and physio-chemical properties of the local runoff and particulates (Driscoll, 1989; Van Buren, 1994).

The bottom sediments in the Kingston stormwater pond studied by Marsalek et al. (1997) accumulated at a temporally and spatially averaged rate of 0.02 m/year as measured by the average length of sediment cores divided by the period of accumulation. This sedimentation rate represents a loss in the permanent wet pond volume of approximately 13% over the ten-year period. Gravel and sand accumulated at the inlet while silt and clay (45% and 54% of the total sediment respectively) were distributed throughout the pond (Marsalek et al., 1997).

Laboratory tests have indicated that sedimentation can account for following removal rates of sediment-associated pollutants: 50% of total phosphorus, 65-85% of lead, 30-45% or zinc, and 40% of copper (Van Buren, 1994). One field study found that 60% of the Pb, 20 to 70% of the Cd, and 30 to 80% of the Cu were bound to sediments (Felstul and Montgomery, 1991) and thus susceptible to the process of removal by sedimentation. Yousef and Lin (1990) found that concentrations of nutrients in bottom sediments are high particularly in the first 20-cm, below which concentrations attenuate rapidly with depth (Yousef and Lin, 1990). Total nitrogen concentrations, mostly TKN, were much higher than total phosphorus, but both attenuated rapidly and declined exponentially with sediment depth (Yousef and Lin, 1990). When sediment samples are not available, the settling distributions of many pollutants can be estimated using the settling velocity distributions of associated sediment size classes. Table 2.2 shows how different pollutants vary with respect to their associated sediment size classes (Driscoll, 1989).

Table 2.2 Proportion of pollutant associated with each sediment particle size (% by weight) (Hodges, 1997).

Sediment Particle Size (μm)	**>2000**	**840-2000**	**246-840**	**104-246**	**43-104**	**<43**
TSS	24.4	7.6	24.6	27.8	9.7	5.9
TKN	9.9	11.6	20	20.2	19.6	18.7
Nitrate	8.6	6.5	7.9	16.7	28.4	31.9
Phosphate	0	0.9	6.9	6.4	29.6	56.2
Copper	22.5	20	16.5	19	22	
Zinc	4.9	25.9	16	26.6	26.6	
Lead	1.7	2.6	8.7	42.5	44.5	

2.3.1.2 Metals

While many toxic substances are potentially present in runoff waters, including metals, pesticides, herbicides, and hydrocarbons, this study focuses on selected metals (copper, cadmium, zinc, and lead) for several reasons. First, 47% of the watershed above the wet pond in the stormwater management facility of interest is designated as commercial and residential which indicates that this is an urban watershed, though agricultural land uses remain in the watershed. Second, it is much more difficult and expensive to routinely test water samples for pesticides, herbicides, and hydrocarbons than it is to test for metals. Third, concentrations of pesticides, herbicides, and synthetic organic compounds (including plasticizers and wood preservatives) in runoff from residential and commercial areas rarely exceed current water quality criteria (Schueler, 1987; Dennison, 1996). Fourth, few toxicity tests have been performed to examine the effect of urban runoff hydrocarbon loads on aquatic communities under the typical exposure conditions found in urban streams (Dennison, 1996), while numerous studies have been performed for heavy metals. Finally, the heavy metals typically found to have the highest concentration in urban runoff are copper, lead, zinc, and cadmium (Dennison, 1996); other metals are found less often and usually do not exceed human health or aquatic life criteria (Schueler, 1987).

Metals most commonly detected in urban runoff include copper, lead, zinc, and to a lesser extent, cadmium, chromium, nickel, iron, manganese, arsenic, silver, and beryllium (Van Buren, 1994). Lead, zinc, and copper amount to more than 76% of total metals detected in urban and highway runoff, excluding iron (Yousef et al., 1994b). Zn and Cd are more likely than Pb and Cu to be reactive and biologically available in solution (Yousef et al., 1985; Pitt, 1995). Metal mobility appears to be high in runoff from roads and car parks, but metals (particularly lead) are transformed to more stable forms due to their association with solids that undergo structural modifications during transport through the sewer network (Flores-Rodriguez et al., 1994). The mechanisms of metal toxicity include: (1) competitive blockade of a functional group of a macromolecule, (2) displacement of essential ions by toxic ions, and (3) conformational change in proteins (Gerhardt, 1993). Both respiration and photosynthesis of plants are inhibited by the presence of elevated levels of heavy metals (Hill, 1997). Decreases in diatom and aquatic vascular plant species richness, and shifts in algal dominance have been documented as the result of metal contamination (Hill, 1997). Algal growth inhibition, as studied using algal bioassays with *Selenastrum*, occurs when the concentration of copper reaches 50-μg/L, when zinc reaches 30-μg/L, and when cadmium reaches 50-μg/L (Harper, 1985). Complete inhibition of algal growth was found at 90-μg/L for copper, 120-μg/L for zinc, and 80-μg/L for cadmium (Harper, 1985).

There are five metal fractions of interest in assessing interactions with environmental conditions and aquatic organisms: bound to carbonates, residual, bound to iron and manganese oxides, bound to organic matter, and exchangeable (Marsalek, 1997). “Hardness, alkalinity, and organic complexes appear to significantly reduce the toxicity of Cd, Zn, Pb, and Cu in natural water" (Yousef et al., 1985). Increasing hardness is thought to ameliorate metal toxicity through competition of the Ca^{2+} and Cd^{2+} ions at the membrane binding sites. Furthermore, other metals form unavailable complexes with $CaCO_3$ (Gerhardt, 1993). Residual trace metals are held within the crystal structure of the sediment and are generally immobile (Marsalek, 1997). Marsalek (1997) found lead to be primarily (52%) bound to iron and manganese oxides, with only 8% in residual form. Anoxic conditions can cause thermodynamic instability of iron and manganese oxide coatings (which enable particle scavenging of trace metals), and thus cause the potential release of the trace metals (Marsalek, 1997). In saturated soil, the anaerobic environment allows the immobilization of toxic metals by sulfides and other complexing ligands (Athanas, 1988). Marsalek (1997) found the majority of total copper (59%) as bound to organic matter. Trace metals bound to organic matter may become mobile under oxidizing conditions (Marsalek, 1997). Dissolved oxygen concentrations were important to the mobility of zinc due to its high distribution in both organic matter (33%) and iron and manganese oxide (29%) fractions.

The exchangeable fraction of a metal "is likely to be affected by changes in water ionic composition which affect sorption-desorption processes" (Marsalek, 1997). Measures of conductivity indicate the ionic activity of the water, which may therefore indicate the availability of metals in the water for potentially toxic interactions with aquatic life (Urbonas, 1995).

Electrical conductivity is a measure of the ability of water to conduct an electrical current (good conductors include inorganic acids, bases, and salts) and is often used as a surrogate for total dissolved solids (Marsalek, 1997). If conductivity increases with depth in ponds, it would indicate the presence of densimetric stratification (Marsalek, 1997). The maximum TDS for potable water is usually 500-mg/L, corresponding to a conductivity of approximately 1000-μmhos/cm (Karch, 1999). Certain plant and animal species are known to be intolerant beyond certain thresholds of electrical conductivity (Karch, 1999). Specific conductance has also shown to be correlated with chloride concentrations (Mayer et al., 1996), which may be higher in winter runoff due to de-icing practices within the watershed.

Changes in pH affect the sorption processes, complexation, and solubility of metals (Gerhardt, 1993). Changes in metal speciation are affected by pH; most metals are complexed at a pH of seven or greater, but concentrations of free metal ions increase when pH falls below five (Marsalek, 1997). The free metal ion is one of the most toxic species and is generally taken up directly from the water by organisms (Gerhardt, 1993). "The influence of pH on metal speciation decreases in the following order: Cu>Pb>Cd>Zn" (Gerhardt, 1993). Depending upon the metal and the animal species, a lowering of pH may also decrease the Cd, Cu, and Zn toxicity, probably because of competition for the binding sites with H^+ (Gerhardt, 1993). Therefore, the actual chemical availability of a metal species is not identical to bioavailability (Gerhardt, 1993).

Hodge and Armstrong (1993) found a statistically significant relationship for storm water concentrations of certain metals and land use (however this relationship explained less than 20% of the sampling data variability). Their results showed that while open areas were the largest land use in the study, they produced less than 2% of the total pollutant load due to low runoff volume and pollutant concentration. Residential areas produced 63% of total runoff, but only 40% of the total load for copper and 48% for lead. Though only 20% of the urban land use area, the combined land use categories of transportation, commercial, and industrial areas represented 50 to 60% of the pollutant load due to high metals concentrations and high runoff volumes (Hodge and Armstrong, 1993).

"Copper is the major aquatic toxic metal found in stormwater and is quickly accumulated by both plants and animals" (Marsalek, 1997). Its toxicity is largely attributed to the Cu^{2+} ion (Hellawell, 1986). Factors that influence its toxicity include hardness, the presence of organic matter, temperature, oxygen concentration, and pH. Copper forms a complex with a wide range of substances, and is readily absorbed onto suspended particles (Hellawell, 1986). Copper comes from many sources in the urban environment including automobiles (thrust bearings, bushings, brake linings, tires), combustion of lubricating oils, and leaching and corrosion of building materials including copper pipes and brass fittings (Cohn-Lee and Cameron, 1992; Hodges, 1997; Marsalek, 1997).

Lead salts generally have low solubility and therefore acute toxicity is unlikely (Hellawell, 1986). However, lead causes chronic effects including bioaccumulation in bottom dwelling fish and shellfish, production of spinal deformities, retardation of growth, and reduced photosynthesis

(Hellawell, 1986; Cohn-Lee and Cameron, 1992; Marsalek, 1997). Lead toxicity is affected by pH, hardness, and the presence of organic material (Hellawell, 1986). Lead in the urban environment derives primarily from motor oil, transmission babbit metal bearings, and tires since it has been removed from gasoline and paint (Hodges, 1997; Marsalek, 1997). The effects of highly toxic organo-lead compounds used as anti-knock additives in petroleum are not well known (Hellawell, 1986).

Zinc bioaccumulates in all organisms and is toxic to fish and aquatic macroinvertebrates (though it may not be as toxic as other heavy metals) (Hellawell, 1986; Cohn-Lee and Cameron, 1992; Marsalek, 1997). Filter feeders, however, are able to tolerate moderate Zn levels (Clements, 1994). Often zinc is found in concentrations that exceed the U.S. EPA's freshwater criteria (Marsalek, 1997). In freshwater environments impacted by urban runoff, zinc derives from combustion of lubricating oils, wear of tires and brake pads, and corrosion of building materials and galvanized iron and steel (Hodges, 1997; Marsalek, 1997).

Cadmium is another metal often found in urban runoff. Cadmium produces both mutagenic and carcinogenic effects, as well as renal (kidney) damage (Cohn-Lee and Cameron, 1992). Cadmium is toxic to many organisms after exposure to low concentrations (<1-µg/L) over extended periods (Hellawell, 1986). Cadmium accumulates in tissues and is thought to damage ion-regulating mechanisms (Hellawell, 1986). Large amounts may be absorbed rapidly and lost slowly; this loss may explain the survival of populations exposed to intermittent doses (Hellawell, 1986). "In field studies of cadmium-stressed plankton communities, phytoplankton photosynthesis and primary production were reduced by very low concentrations (0.2-µg Cd per litre) and zooplankton community structure was affected by concentrations of 5-µg Cd per litre or less" (Hellawell, 1986). Cadmium in freshwater streams may derive from paints or atmospheric deposition from smelters and metal finishing industries (Cohn-Lee and Cameron, 1992).

Because different metals are bound to different size fractions of the sediment, they may require different retention times to achieve similar removal rates. For example, lead has a stronger affinity for solids than cadmium, and thus may settle out faster and have a higher removal rate for a given retention time (Felstul and Montgomery, 1991). Several studies have examined metals within pond sediments. Marsalek et al. (1997) examined bottom sediments of a stormwater pond in Kingston Ontario and found, by sequential analysis of sediment samples, that 40-90% of the retained metals was in potentially mobile forms. Zinc and lead were the dominant metals accumulated in the wet detention pond sediments, and the attenuation rates were in the order of Zn>Pb>Cu (Yousef and Lin, 1990). Yousef and Lin (1990) found that concentrations of metals in the first 20-cm of bottom sediments, similar to nutrients, are high initially but attenuate rapidly with depth (Yousef and Lin, 1990; Yousef et al., 1994b). If the sediment has a large capacity to retain metals, it can take many years to saturate the top 150 to 200-mm with lead, zinc, and copper (Yousef et al., 1994b). The higher the clay content (which has a high affinity for metal uptake) and organic matter (which may complex with and immobilize metals) in sediments, the lower the fractions of metals that can be extracted (Yousef and Lin, 1990). In general, if aerobic conditions are maintained, the potential release of trace metals to solution in natural waters is low (Yousef et al., 1985). Whether the bottom sediments are considered hazardous waste and can be land applied is determined by use of the U.S. EPA toxicity characteristics leaching procedure (TCLP) (Yousef et al., 1994b; Yousef and Lin, 1990).

2.3.1.3 Phosphorus

Erosion is the primary cause of phosphorus entering freshwater streams (Mason, 1996). Both particulate and dissolved forms of phosphorus are found in retention ponds. Because phosphorus is readily absorbed to sediment, particularly to amorphous ferrous hydroxide (Reddy and Reddy, 1993), sedimentation is often considered to be the primary removal mechanism of phosphorus within stormwater detention basins. The forms of particulate phosphorus include inorganic phosphorus attached to soil particles, organic detritus (tree leaves may be a large source (Hodges, 1997)), and phytoplankton phosphorus (Wu and Yu, 1995). In general, total phosphorus loading is directly correlated to the volume of runoff with the majority of phosphorus contributed during storm events (Wulliman et al., 1989). The research by Wulliman et al. (1989) in Shop Creek Basin implied that initial high phosphorus concentrations in runoff from a developing basin would later decrease to baseline levels as the disturbed lands are stabilized by vegetative growth or other erosion retardants.

In one study by Wu et al. (1996) in the Piedmont region of North Carolina, the predominant form of phosphorus in storm runoff was soluble and thus its removal was largely controlled by dilution or utilization capacity (by microorganisms, algae, and aquatic plants). Biotic factors may largely control exchangeable phosphorus uptake (greater than 80%) by stream sediments and organic matter (Hill, 1997).

The cycle of dissolved or available phosphorus within a pond primarily begins with utilization by phytoplankton (because phosphorus is the rate-limiting factor for primary production in freshwaters) or interaction with particulate inorganic phosphorus through sorption and desorption (Wu and Yu, 1995). Due to respiration and mortality, phosphorus is returned from the phytoplankton biomass pool to dissolved and particulate organic phosphorus and dissolved inorganic phosphorus (Wu and Yu, 1995). Organic phosphorus is also converted to dissolved inorganic phosphorus at a temperature-dependent rate (Wu and Yu, 1995).

In general, phosphorus concentrations increase during late summer and fall as compared to concentrations in spring and summer (Mulhern and Steele, 1989). In a study of a wet pond by Mulhern and Steele (1989), it was found that between August and November 1987, algal growth in the pond was substantial and had the effect of remobilizing phosphorus into the water column when oxygen demand was high and anaerobic conditions were created at the pond bottom (average depth of 1.22-m). These releases however were insignificant with respect to the total annual loading from the adjacent golf course and the phosphorus removal during storm events due to deposition of suspended sediments (Mulhern and Steele, 1989).

Phosphorus levels within ponds remain stable due to equilibrium between the water column and the sediments (Lewis and Wang, 1997). Under aerobic conditions, the pond may absorb orthophosphorus and total phosphorus at high rates (18.9 and 20.5-mg of $P/m^2/day$) (Marsalek et al., 1992). Under anaerobic conditions, a pond can release phosphorus albeit at low rates (0.3 to 0.9-mg of $P/m^2/day$) (Marsalek et al., 1992). Low pH conditions also result in a release of phosphorus from the sediment to the water column (Lewis and Wang, 1997). Furthermore, Reddy and Reddy (1993) found that sediments in their study showed a greater sorption capacity under reduced conditions as compared to oxidized conditions. The amount of phosphate adsorption-desorption by the sediment varies with soil sorption potential (recent exposure of soils in new ponds may have high phosphorus uptake initially but the system may become saturated with decreased uptake after several years), ambient pH, pore-water P concentration, and presence of competing ions (Wotzka and Oberts, 1988; Reddy and Reddy, 1993). It has been shown in studies of lakes that nitrate concentrations above 0.001-mg/L are able to buffer the redox potential of the surface sediment so that phosphorus remains bound to the sediments (Bayley, 1985).

2.3.1.4 Nitrogen

Nitrate concentration in rivers is closely linked to the volume of flowing water (Mason, 1996). Nitrate levels in streams follow a seasonal pattern, increasing in autumn and winter due to leaching from soils when transpiration and evaporation decline, decreasing in late winter when the soluble nitrate reserves have been depleted and nitrification rates decline, and increasing again in spring when temperatures rise and applications of fertilizer occur along with spring rainfall (Mason, 1996).

Greater concentrations of nitrogen are typically found within the water column as compared to the sediment, and these concentrations fluctuate more than phosphorus levels (Lewis and Wang, 1997). Total nitrogen includes four major components: organic, ammonia, nitrite, and nitrate forms (Wu and Yu, 1995). The organic nitrogen can be found in both particulate and dissolved forms. The inorganic nitrogen is used by phytoplankton for growth (Wu and Yu, 1995). Nitrogen removal for particulate nitrogen (organic detritus particles and phytoplankton) in ponds can be very good via settling in the pond (Wotzka and Oberts, 1988). Soluble nitrate removal occurs to a lesser extent primarily as loss of nitrogen gases due to denitrification, and mineralization of nitrate into organics (Wotzka and Oberts, 1988). Some losses of nitrate-nitrogen occur in the stream channel itself, on the order of 0.01 to 0.91-g/m^2/day (Bachmann et al., 1991). In general, though, the soluble nitrogen (dissolved organic, ammonia, and nitrate) reacts among themselves but remain soluble and exit a pond or wetland with little reduction (Wotzka and Oberts, 1988).

2.3.1.5 Bacteria

Urban runoff, almost without exception, contains bacterial levels that exceed public health standards for water contact recreation (Schueler, 1987). However, several problems exist when analyzing bacterial results. First, bacterial measurements are made with indicator organisms, rather than of the more potent pathogens and viruses, and it is unclear whether the kinds of bacteria found in urban runoff are a severe health hazard (Schueler, 1987). Second, the variability in results is very high and displays substantial seasonality. Since “bacteria multiply faster during warm weather, it is not uncommon to find a twenty-fold difference in bacterial levels between summer and winter” (Schueler, 1987). Third, bacterial removal by die-off is not well understood in the context of variable loading and flow through stormwater-receiving ponds (Van Buren, 1994).

A few studies have investigated potential influences on bacterial variability. Wet weather fecal coliform (FC) yields from impervious surfaces, defined as the FC generated per m^2 of surface per cm of rainfall, were found by Weiskel et al. (1996) to be related to the surrounding land use, with the highest FC yields generated by high-density residential areas. “Bacterial yields (FC m^{-2} cm^{-1} of rainfall) from impervious surfaces served by storm drains were 300-8000 times higher than those from areas of low-intensity land use drained by streams” (Weiskel et al., 1996). Weiskel et al. (1996) also found that storm events, though infrequent, had FC densities about 6 times higher than dry-weather flows. Yet Weiskel et al. (1996) also found no significant difference in FC density between first-flush samples and those collected after 0.6-cm of rainfall.

Dry weather sources of fecal coliforms are also important considering the larger volume of water they influence. Dry weather sources of fecal coliforms may include groundwater discharge, minor rain events, and periodic releases due to in-stream production (Weiskel et al., 1996).

The major source of fecal coliforms to stormwater is likely the feces of domestic animals and wildlife, especially if accumulated on paved surfaces (Weiskel et al., 1996). Weiskel et al. (1996) estimates that the fecal production rate per dog is 450-g/day. Furthermore, “dog feces exposed to ambient environmental conditions in the study area showed no detectable declines in FC densities after 7, 14, and 30 days (30-day median density = 10^6 FC g^{-1})” (Weiskel et al., 1996). Geese and ducks also may contribute substantially to the fecal coliform (FC) loads of a water body, particularly from over-wintering populations (Weiskel et al., 1996). Daily fecal coliform production of geese and ducks is 8-253$\times 10^6$ and 1000-11000$\times 10^{6-}$cfu/head/day respectively (Yagow, 2000). The seasonality of resident populations and the amount of time they spend away from the water body influence estimates of total waterfowl FC inputs to a water body (Weiskel et al., 1996). Waterfowl numbers tend to peak mid summer to late fall on the

Virginia Tech campus (Dr. James Parkhurst, personal communication, April 25, 2000). Waterfowl fecal bacterial inputs may be reduced by the tendency of fecal pellets in quiescent settings to remain intact and thus limit bacterial dispersal prior to die-off (Weiskel et al., 1996). Though not investigated here, other wildlife including rodents and birds could also contribute fecal material to streams.

Some general estimates have been made with regard to wet ponds and bacterial contaminant treatment. Pond performance in terms of fecal coliform (FC) removal or number of exceedances appears to deteriorate if the dry weather flow rate fills the pond in less than 5 days (Droste et al., 1993). According to results from Droste et al. (1993), pond volumes between 9 and 12-mm of runoff for the entire watershed appear to provide optimal performance for fecal coliform removal. Because bacteria tend to adsorp on suspended particles in the runoff water, sedimentation could be a significant removal mechanism (Hvitved-Jacobsen, 1986). However, removal of the bacteria from the water column to the pond bottom may enhance its survival if conditions within the sediment are conducive to growth and reproduction (Hvitved-Jacobsen, 1986). Furthermore, fecal coliforms have shown a lower tendency to adsorp to suspended particles than other bacterial species and thus may not be the optimum group for assessing health risks (Hvitved-Jacobsen, 1986).

2.3.1.6 Temperature

The increased water temperature of urban streams is due to three factors: (1) runoff is heated as it passes over the urban landscape, (2) fewer trees on the streambank shade the stream channel, and (3) water stored in shallow wet ponds and other impoundments is heated between storms (Schueler, 1987). Typically, a watershed impervious increase of 12% will raise the mean stream temperature by 1°C (Weatherbe et al., 1993).

A rise in water temperature of few degrees Celsius over ambient conditions can reduce or eliminate sensitive stream insects and fish (Schueler, 1987). Sustained summertime water temperatures in excess of 21°C are considered stressful, if not lethal, to many cold-water organisms (Schueler, 1987). The optimal temperature range for diatoms is 15 to 25°C, while green algae prefer a range of 25 to 35°C and blue-green algae 30 to 40°C (Lewis and Wang, 1997). Blue-green algal blooms are the most undesirable, especially if composed of *Anabaena* spp. and *Microcystis* spp that produce chemicals toxic to both aquatic and terrestrial life (Lewis and Wang, 1997).

Water temperature affects a wide range of pond characteristics. High temperature is associated with low oxygen solubility and high oxygen consumption (Marsalek, 1997). Temperature increases typically increase the toxicity of zinc and cadmium, probably due to increased rates of uptake (Gerhardt, 1993). Temperature also produces densimetric effects within a pond, as water reaches its maximum density at 4°C (Marsalek, 1997). Unless there is thorough mixing, a temperature gradient with an associated density gradient will develop within the pond (Marsalek, 1997) and affect flow circulation patterns. Wind may produce some mixing and may even entrain previously deposited sediment if pond depths are less than 1-m. High inflows may also contribute to mixing of the pond, and at least temporarily destroy thermal stratification (Marsalek, 1997).

2.3.1.7 Chemical Oxygen Demand

Chemical oxygen demand (COD) is used to estimate oxygen consumption in receiving waters due to oxidation of organic matter by chemical processes (Van Buren, 1994). Urban surface waters may have high levels of COD due to automobile leakage of hydrocarbons (Williams and Feltmate, 1992). COD measurements include some organic matter that does not ordinarily contribute to oxygen demand, and is weakly correlated to measurements of biological oxygen demand (BOD) (Schueler, 1987). However, COD is recommended over BOD in urban runoff measurements because trace metals may inhibit bacterial growth and thus interfere with the BOD test (Schueler, 1987).

2.3.1.8 Total Organic Carbon

Total organic carbon (TOC) has been used to estimate the amount of oil and grease in runoff (Hodges, 1997). “Oil and grease contain a wide array of hydrocarbon compounds, some of which are known to be toxic to aquatic life at low concentrations” (Schueler, 1987). TOC is used as a surrogate for petroleum hydrocarbons because they are difficult to assay but the measure may be greatly influenced by the decomposition of organic matter (Hodges, 1997).

2.3.2 Removal Efficiencies of BMPs

The ideal measurement of pollutant removal from stormwater would be performed on a specific unit volume of water as it moves through a stormwater management facility. Since this would be very difficult and expensive, if not impossible to achieve, pollutant removal is estimated from concentration and flow data using removal efficiency calculations. Pollutant removals by stormwater management practices are calculated by different methods in the literature. The methods differ in their use of either pollutant concentrations or loads, and by the means of data summary and time period for which they are estimated. Several methods are reviewed here including event mean concentration (EMC) efficiency, mass removal efficiency (MRE), Pollutant Removal (PR) efficiency, Sum of Loads (SOL) efficiency, and regression efficiency.

The concentration removal efficiency calculation, often used in the literature, is called the event mean concentration efficiency, which compares the percent difference in the basin's inflow and outflow pollutant concentrations to the inflow pollutant concentration. It is generally used for individual storm events and is considered a conservative method since it assumes no retention of flow (Yu et al., 1998a). The median of the EMC efficiencies may produce the lowest efficiency value since other calculations give more weight to larger storms (Rushton and Dye, 1991). This method is not appropriate if dilution of the pollutant concentrations occurs due to unmeasured flow sources (including groundwater exfiltration or rainfall) (Yu et al., 1998b). Contaminated rainfall, evaporation, and/or infiltration to groundwater may also influence pollutant concentrations.

$$\text{EMC Efficiency (\%)} = (1-(\frac{outletEMC}{inletEMC}))*100 \qquad (2.2)$$

Individual storm events can also be assessed by the mass removal efficiency (MRE), using Equation 2.2 modified to compare the basin's inflow and outflow loads (Yu et al., 1998b).

With respect to pollutant removal, ponds "exhibit variable performance characteristics, depending on the size of the storm being processed. In general, basins perform more poorly for the larger storms than for the smaller ones" (Driscoll, 1989), due to decreased detention times. Calculations of removal efficiencies of basins are also influenced by residual stormwater from previous events (Driscoll et al., 1986). Depending on storm frequency and size, the effluent displaced during a particular event represents both some volume of the current storm and some volume due to antecedent storms (Driscoll et al., 1986). Therefore, comparing influent and effluent loads for individual storms may be less appropriate than comparing overall influent and effluent loads over all completely monitored storms (Driscoll et al., 1986). Several removal efficiency calculation methods seek to address this issue by assessing long-term performance with respect to pollutant removal. The Sum of Loads (SOL) efficiency and the Pollutant Removal (PR) efficiency are extensions of the mass removal efficiency calculation. The difference lies in the way that pollutant loads are calculated prior to their use in the efficiency calculation. The Sum of Loads (SOL) efficiency is based on the mass entering and leaving the system over all completely monitored storms (Yu et al., 1998b).

$$\text{SOL Effic.}(\%) = \frac{\sum(\text{Inflow Volume} * \text{Inflow EMC}) - \sum(\text{Outflow Volume} * \text{Outflow EMC})}{\sum(\text{Inflow Volume} * \text{Inflow EMC})} * 100 \qquad (2.3)$$

The Pollutant Removal (PR) efficiency modifies the MRE equation by substituting average volumes and average EMCs into Equation 2.2 modified for loads (Glick and Chang, 1998).

The regression efficiency is another method of calculating long-term performance for pollutant removal. It is calculated as the percent slope of the line fitted by least squares regression, constrained to a zero intercept, for the relation between event loads retained by and loads entering each treatment unit (Martin and Smoot, 1986; Wotzka and Oberts, 1988; Gain and Miller, 1989). This method is considered more robust against the influence of a single event (Wotzka and Oberts, 1988).

Assessment of BMP performance should also consider potential limits to stormwater treatment efficiency, as discussed by Schueler (1996). There may be irreducible concentrations with respect to stormwater BMPs, possibly due to the internal production of nutrients and turbidity, or limitations of removal pathways. Existence of irreducible pollutant concentrations has implications not only for BMP performance assessment but also for understanding cumulative watershed impacts and reconsidering the necessity of multiple BMP systems (Schueler, 1996).

2.3.2.1 Pond Influences

Pollutant reductions during baseflow events may be substantial. During baseflow periods, Van Buren (1994) found positive removal rates for organic contaminants and metals, and negative removal rates for nutrients and COD. Even in wet ponds, dissolved constituents may pass through with little attenuation during either baseflow or storm conditions, though all other contaminants may show positive removal rates under storm conditions (Van Buren, 1994). Negative removal rates of nutrients and COD under baseflow conditions could also be due to the release of nutrients and high loading of COD due to the decay of organic matter that originated in the pond (Van Buren, 1994).

Inconsistent removal efficiencies of phosphorus and nitrogen in a study of urban wet detention ponds by Wu et al. (1996) was attributed to waterfowl droppings into the pond. "The daily excrement contribution from geese was estimated at an average of 1.22-kg (2.7-lb) of dry excrement per 100 geese with an average dry weight content of 4.5% nitrogen and 1.0% phosphorus (Wu, 1989; Wu et al., 1996). The total phosphorus and total kjeldahl nitrogen (TKN) loads from geese droppings were calculated to be less than 10% of the total load to the pond, but their influence could be significant if the geese were localized near the pond outlet (Wu et al., 1996). Waterfowl influence would be more significant during baseflow periods, since during storm events the incoming loads of organic matter and COD would dominate over in-pond loads, and positive removal rates should be observed (Van Buren, 1994).

Inconsistent removal rates may also be attributed to seasonal influences; ponds may not prove as efficient in pollutant removal during the winter with regard to melting snow and winter ice cover. Melting snow may contribute large loads of pollutants to receiving waters that had accumulated over the winter within the snow pack. The winter accumulations are a result of snowflake scavenging of aerosol and particulate pollutants, less efficient operation of motor vehicles, and application of deicing chemicals (Marsalek, 1997). Some studies have shown that up to 60% of the annual runoff load of some pollutants may be produced by snowmelt runoff (Marsalek, 1997). Four stormwater ponds monitored in Minnesota showed a marked reduction in their performance when treating snowmelt runoff, particularly with regard to nutrients and lead (Oberts, 1994). Winter ice cover on ponds has been found by Mayer et al. (1996) to be associated with low DO levels within ponds (<1-mg/L) and releases of ammonia from pond sediments. Ice cover may degrade the pollutant removal capacity of wet ponds for several reasons. First, the ice layer may eliminate part of the permanent storage volume (Oberts, 1994). Second, flow conditions below the ice layer may be turbulent and potentially cause scouring and resuspension of bottom sediments (Oberts, 1994). Third, meltwater flow may be forced to flow over the top of the ice, which greatly reduces the amount of treatment possible (Oberts, 1994). Last, treatment capacity within the stormwater ponds is reduced due to cold temperatures limiting biological activity, including biological pollutant removal processes (Oberts, 1994).

2.3.2.2 Residence Time

Retention basins are designed to capture a certain volume of runoff and/or achieve a particular rate of pollutant removal based on a minimum detention time. Effective improvement of water quality parameters is directly related to the detention time of raw water in the system. Pond design features that appear to enhance settling include long detention times and good mixing conditions (Van Buren, 1994). Factors that do not favor settling include short-circuiting of the flow, high flow velocities, and secondary currents (Van Buren, 1994).

Hydraulic residence time, or volumetric residence time, is often computed for ponds as the constant volume of water in the basin divided by the constant flow through the basin (Nix, 1985). This type of calculation may be slightly improved for estimating conditions in a plug-flow basin by dividing the volume of water in the basin by the average of the inflow and outflow rates (Nix, 1985). However, this steady state definition of residence time is considered a poor indicator of a basin's pollution control ability (Nix, 1985) because it is only applicable to basins with plug or completely mixed flow conditions (Matthews et al., 1997).

Basin flow is often assumed to follow a plug-flow model flow (concentrations are uniform in the cross-section with no longitudinal mixing) or a completely mixed-flow model (concentrations uniform throughout the basin) (Martin, 1989). The residence time of the plug-flow basin applies to each particle while the residence time in the completely mixed flow basin will be steady state (the mean of the flow particles) (Martin, 1989). These opposite assumptions are idealized and suggest longer residence times than that typically seen in the field due to moderately mixed flow conditions (Martin, 1989). Basins typically deviate from these ideal conditions; the deviation is computed as the hydraulic efficiency, defined as the ratio of measured to volumetric retention time (Matthews et al., 1997).

The moderately mixed flow basin exhibits deviations from either flow model as a result of short-circuiting; wind influences; shear stresses along the pond boundaries; dispersion; inlet and outlet effects; circulation patterns (recycling); and stagnant zones within the ponds (Martin, 1989; Shaw, 1995). Such situations are difficult to model and therefore mixing characteristics and residence times may have to be measured in the field (Martin, 1989).

Short-circuiting of flow through the pond can decrease detention time, and is related to degree of stratification and the potential for mixing within the pond (Allan et al., 1997). Profiles of temperature, dissolved oxygen, pH, conductivity, and flow can be used to estimate the effects of stratification upon pollutant removal efficiencies. Seasonal changes in stormwater volume, inflow rates, and stratification (thermal gradients controlling vertical mixing) (Gain, 1996) will need to be considered in any measurement of retention time.

Flow patterns can be measured to better understand hydraulic residence time. The flow pattern determined for the Kingston stormwater management facility showed the pond had three different flow zones (Shaw et al., 1997). The first zone was the main advective flow from inlet to outlet, characterized by the highest velocities, the second zone was the mixed zone characterized by significant recirculation, and the third zone was the dead zone which had both limited recirculation and low velocities (Shaw et al., 1997).

Wind stress was found by Shaw et al. (1997) to greatly influence baseflow circulation patterns in the stormwater pond. Under conditions of low wind stress, the circulation pattern in the pond was predominantly circular in the horizontal plane, dependent upon pond shape and inflow momentum (Shaw et al., 1997). Wind speeds of 2 to 4 m/s generated a vertical circulation pattern in the pond that could have caused resuspension of previously deposited sediment (Shaw et al., 1997). Furthermore, high wind stress under low flow (baseflow) conditions generated a current with shear velocities sufficiently large to impede particle settling (Shaw et al., 1997). In still water in a flume, wind-induced current extended to 35% of the entire depth and the return current occupied the bottom 65% (Shaw et al., 1997). In a pond, the wind current could potentially oppose or enhance the inflow current, and thus increase or decrease residence time (Shaw et al., 1997). Under well-mixed baseflow conditions, the fine suspended particles may be kept in suspension and experience little attenuation.

Dye-response curves (time-concentration) are used to determine time-of-travel and dispersion characteristics of on-stream basins (Shaw, 1995). These breakthrough curves correspond to the distribution of retention times of influent water at the time of dye injection (Matthews et al., 1997). The median residence time is estimated as the elapsed time for 50% of the dye mass to be recovered (Martin, 1989). The spread of the base of the breakthrough curve indicates the extent of dispersion (Thackston et al., 1987). If the flow in the pond could be described as plug flow, the measured tracer dye concentration at the outlet would be unimodal; thus describing the dye slug appearing at the outlet at its hydraulic residence time, with limited trace of dispersion and mixing (Anderson et al., 1996). A bimodal distribution would indicate the existence of short-circuiting currents in the pond (the first peak due to the fast direct flow and the second peak as a result of the dye's incorporation into the mixed flow zone) (Anderson et al., 1996). Indication of short-circuiting within the pond can also be seen on a graph of measured dye concentration as compared to the concentration of dye assuming complete mixing had occurred (Martin, 1989). Measured concentrations greater than the maximum concentration with complete mixing may indicate that at least part of the dye is short-circuited across the pond (Martin, 1989). Higher concentrations may also indicate plug flow with diffusion, but the time to peak concentration would then be the same as the mean residence time (Martin, 1989). The extent of short-circuiting can be calculated as the portion of flow that exits the basin in 0.3 to 0.4 of the volumetric residence time (Thackston et al., 1987). The optimum residence time for wet detention basins varies according to the design criteria; when the lake eutrophication assumption is made for pollutant removal, the optimum residence time may be 2 to 3 weeks as compared to 1 to 2 weeks for the solids settling design assumption (Hartigan, 1989). Though sedimentation may improve with increased residence times, a detention time greater than 2 weeks runs the risk of thermal stratification (and thus the risk of short-circuiting) and anaerobic bottom waters (with the associated risk of release of nutrients and toxics from the bottom sediments) (Hartigan, 1989). Hartigan (1989) advises minimizing the average residence time to that which can ensure adequate nutrient removal (2 weeks as opposed to 3).

2.3.2.3 Comparable Studies

One objective of this study was to compare the VPI&SU facilities' pollutant removal efficiencies with those obtained by other researchers. Schueler (1993) reviewed 58 performance-monitoring studies of stormwater ponds and wetlands including extended detention ponds, wet ponds, wet extended detention ponds, constructed wetlands, extended detention wetlands, and pond/wetland systems. The results Schueler (1993) compiled are shown in Table 2.3. Total suspended solids and lead showed consistently high removal rates, but the stormwater ponds and wetlands were less effective for phosphorus, nitrogen, and zinc. Of the 58 studies, the best performers were pond systems that combined treatment techniques (the pond/marsh system and the wet extended detention ponds). Wet ponds and constructed wetlands also showed consistently high removal rates, while dry extended detention ponds showed consistently low removal rates (Schueler 1993). Often the low performing systems had poor internal design geometry including low length to width ratios, lack of forebays, lack of structural complexity, inadequate treatment volumes, and deep flow paths (Schueler 1993).

Table 2.3 Percentile of stormwater ponds and wetlands performance monitoring studies where indicated removal rates were achieved (Schueler, 1993)

Range in Reported Removal Rate (%)	**Total Suspended Solids (%) N=58**	**Total Phosphorus (%) N=58**	**Total Nitrogen (%) N=29**	**Extract. Lead (%) N=32**	**Extract. Zinc (%) N=32**
81 to 100	43	9	4	31	16
61 to 80	28	25	14	38	28
41 to 60	14	22	7	13	16
21 to 40	9	22	54	19	28
1 to 20	5	19	18	0	3
Negative	1	3	4	0	10

The nine wet ponds studied by the U.S. EPA Nationwide Urban Runoff Program (NURP as mentioned previously) indicated that wet ponds could achieve particulate removal (TSS and Pb) in excess of 90% (Wu, 1989). Soluble pollutant fractions had lower potential reductions in these studies: TP (65%), COD (50%), TKN (50%), Cu (50%) and Zn (50%) (Wu, 1989; Wu et al., 1996).

Martin (1989) reported that in a fairly well mixed system with some short-circuiting at high flows, the removal efficiencies were found to be from 50 to 80% for total lead, zinc, and solids while the total nitrogen and phosphorus efficiencies ranged from 30 to 40%. Because of short-circuiting within the basin, these removal efficiencies were expected to increase had practices been installed to reduce short-circuiting (Martin, 1989).

Extensive research into the performance of a stormwater management facility located in Kingston Ontario has contributed much to the understanding of the factors influencing performance of these facilities (Shaw, 1995; Anderson et al., 1996; Marsalek, 1997). The system consists of a 5200-m^2 wet pond (1-m average depth) with a dry pond located immediately upstream for flood control (Anderson et al., 1996). The pond is small in relation to its catchment area of 4.4-km^2, and its length to width ratio is 1.5:1 (Anderson et al., 1996). Outflow composition appears to be influenced by both the inflow composition and by processes that generate internal loading (Anderson et al., 1996). During baseflow periods, Anderson et al. (1996) found no removal of dissolved constituents (total dissolved solids, sulphate, and chloride), negative removal of nutrients (nitrogen, TKN, phosphorus), COD, and suspended solids, and positive removal of metals and organic contaminants (Anderson et al., 1996). During storm events, their results showed negative removal of dissolved constituents, positive removal of nutrients and suspended solids, and positive removal of metals and organic contaminants (Anderson et al., 1996).

It is interesting to compare the performance results from another stormwater BMP located at VPI&SU, an extended dry detention basin, to the results from this study. The extended dry detention basin was installed in 1993 along the main stem of Stroubles Creek, above the confluence of Stroubles Creek with the unnamed tributary of this study. It was designed to contain 1.27-cm (½-in.) of water from an impervious parking lot and detain it for 40 to 50-hrs. (Hodges 1997). Comparisons can be made between EMC removal rates (Table 2.4) from the six-month study of the extended dry detention basin as reported by Hodges (1997) to the wet pond/dry pond system reported in this study as well as to other stormwater management facilities in the literature. Hodges (1997) reported removal rates in the extended dry detention basin of over 70% for TSS, 30-50% for heavy metals, and almost 60% for total nitrogen; however as expected the removal rates for dissolved constituents were low (Hodges, 1997).

A highly recommended regional stormwater management practice involves combinations of ponds and wetland systems that can achieve high pollutant removal rates (Schueler, 1994a). The results from two such systems are reported in Table 2.5 as an estimate of potential pollutant removal. In particular, the pond/wetland system investigated by Leersnyder (1993) had a large treatment volume, excellent internal geometry, and redundant treatment mechanisms (Schueler, 1994a).

Table 2.4 EMC removal rates for several extended dry detention basins (Hodges, 1997)

Analyte	**EMC Removal (%) Zariello and Sherwood (1993)**	**EMC Removal (%) Hodges (1997)**
TSS	83.8	65.7
TOC	47.4	38.2
NH_4	21.5	96
NO_3	35.2	70.4
TP	32	27.3
FTP	11.1	-136
Pb	37.6	27.3
Zn	66.1	36.4

Table 2.5 Pollutant removal rates during storms for pond/wetland systems (Schueler (1994a) adapted from Leersnyder (1993); and Schueler (1994b), adapted from Urbonas et al. (1993))

Parameter	**Average Removal Rate, Urbonas et al. (1993): Residential Land Use, Colorado**			**Leersnyder (1993): Industrial Land Use, New Zealand**
	% Removed by Pond	**% Removed by Wetland**	**% Removed by System**	**% Removed by System**
TSS	78	29	72	78
TP	49	3	51	79
NO_3	-85	5	-76	62
NH_4	NA	NA	NA	-43
COD	44	21	56	2
Total Copper	57	2	57	84
Total Lead	NA	NA	NA	93
Total Zinc	51	31	66	88

2.4 BMP Assessment Using Habitat and Benthic Macroinvertebrate Metrics

The U.S. EPA, states, and tribes are working together to develop new water quality criteria and standards programs across the country. The water quality criteria and standards program will fully integrate biocriteria, nutrient criteria, and microbial pathogen control with improved chemical-specific and whole effluent toxicity criteria (http://www.epa.gov/OST/standards/

planfs.html, Water Quality Criteria and Standards Plan – FACTSHEET - Priorities for the Future, March 17, 2000). The U.S. EPA has published rapid bioassessment protocols for use in streams and rivers to measure indicators of stream health with respect to periphyton, benthic macroinvertebrates, fish, and their habitat (Barbour et al., 1999).

Rapid Bioassessment Protocols (RBPs), predictive qualitative sampling procedures standardized according to level of effort, are often used to assess aquatic life impairments (Barbour et al., 1999). They expedite assessment of water quality problems; expending the minimum amount of effort to get reproducible, scientifically valid results; and thus they expedite management decisions (Lenat and Barbour, 1994). The means of achieving a "rapid" assessment is to use shortcut techniques relative to traditional methods, usually qualitative or semi-quantitative sampling, or processing a targeted number of organisms per site (Lenat and Barbour, 1994). Replication is not emphasized (Evans, 1997). Results are typically presented in the form of metrics and multimetric indices. A metric is a measure made from a sample of the community. Metrics allows the investigator to use meaningful indicator attributes to assess the response of assemblages and communities to perturbation (Barbour et al., 1999). The use of each metric is based on a hypothesis about the relationship between instream condition and human influence (Barbour et al., 1995). For a metric to be useful, it must have the following attributes: (1) ecologically relevant to the biological assemblage or community under study and to the specified program objectives; (2) sensitive to stressors; (3) provides a response that can be discriminated from natural variation; (4) environmentally benign to measure; and (5) cost-effective to sample (Barbour et al., 1995; Barbour et al., 1999). The level of effort in Rapid Bioassessment Protocols is not enough to determine the cause of impairment, so the use of other studies is required to show how specific pollutants including suspended solids, metals, nutrients & organic matter impair biological and habitat potential.

2.4.1 Habitat

Assessment of the physical structure of the habitat in terms of its support of the regional biota includes the evaluation of the variety and quality of the substrate, channel morphology (sinuosity, point bars), bank structure, and riparian vegetation (Barbour et al., 1999). Stream habitat can be degraded by conditions along the streambank created by human activities, and by changes in the stream flow regime due to alteration of the watershed by humans.

Streamside vegetation is usually removed when a watershed is converted to agricultural or urban land uses. This causes several impacts to the stream. First, the lack of tree canopy above the stream causes temperatures in the stream to increase. Macroinvertebrates utilize diatoms and green algae as food, but the less desirable blue-green algae are promoted with higher temperatures. Second, reduced amounts of leaf litter and large woody debris enter the stream channel. Leaf packs are used both as food and substratum by aquatic invertebrates, while large woody debris provides the aquatic community with shelter and habitat, dissipates flow energy, and protects the streambed and streambank (Booth and Jackson, 1997).

Soil compaction and the addition of impervious surface to a watershed change the stormwater flow path from subsurface dominated to surface dominated. The increased surface flow causes the movement of fine sediment fractions into the stream channels from the land surface and substantially alters the sediment size distribution of previously gravel bed streams (Booth and Jackson, 1997). The fine sediment is not preferred as a substrate by the macroinvertebrate community, and sediment fills in the stream sequences of riffles and pools used by macroinvertebrates and fish for feeding and reproduction. The most common form of erosion within streams is channel incision, followed by channel widening, which produces large amounts of sediment in the stream (Knight et al., 1998). Cumulative changes in geometry and composition within human-impacted streams degrade the habitat quality for aquatic life (Knight et al., 1998).

In urbanized watersheds, streams are "observed to have a particular visual signature including eroded banks along both bends and runs, uniform and shallower depth, wider channel, and newly deposited sediment in the channel" (Maxted and Shaver, 1996). Additional streambed instability characteristics include a flatter gradient channel, increased braiding and head-cutting, reduced pool frequency and less diverse habitat, reduced large organic debris, and reduced algal community diversity (Sovern and Washington, 1996). Aquatic system degradation is readily observable when the impervious surfaces cover 10% or more of the watershed area (Schueler and Claytor, 1996). Above 25% impervious cover, most indicators of stream quality consistently shift to a poor state or condition (Schueler and Claytor, 1996) with a higher frequency of small flood events (Maxted and Shaver, 1996).

The rapid and qualitative habitat assessment matrix prescribed by the U.S. EPA Rapid Bioassessment Protocols was developed to rate the ability of the habitat to support the optimal biological condition of the region (Barbour et al., 1999). The matrix is composed of ten metrics listed and described below. Increases in the scores of the habitat metrics indicate a potential for improved biotic conditions.

(1) Epifaunal Substrate / Available Cover

Percentile of the quantity and variety of natural structures in the stream, including riffle and run habitats produced by cobble, large rocks, fallen trees, logs and branches, and undercut banks. These natural structures are used for refuge, nurseries, feeding, and spawning (Barbour et al., 1999).

(2) Embeddedness

Percentile amount that gravel, cobble, and boulder particles within riffles are surrounded by fine sediment. The surface area available to macroinvertebrates and fish for shelter, spawning and egg incubation decreases as embeddedness increases (Barbour et al., 1999).

(3) Velocity-Depth Combinations

Measurement of the number (up to four) and quality of possible velocity-depth combinations (slow-deep, slow-shallow, fast-deep, and fast-shallow). A stream reach with all four patterns tends to provide and maintain a stable aquatic environment as well as a diverse habitat. The general guidelines are 0.5-m depth to separate shallow from deep, and 0.3-m/sec to separate fast from slow (Barbour et al., 1999).

(4) Sediment Deposition

Defined as the percentile of sediment accumulation and change to the stream bottom because of large-scale movement of gravel, sand, or fine sediment. Such conditions are due to an unstable and continually changing environment, unsuitable for many organisms. Evidence of sediment deposition may be seen in the formation of islands, point bars or shoals, and the filling of runs and pools (Barbour et al., 1999).

(5) Channel Flow Status

Percentile of how much of the channel is filled with water, especially as related to exposed substrate that would be suitable habitat for aquatic organisms if covered by water (Barbour et al., 1999).

(6) Channel Alteration

Measures the presence and amount of channel alteration by humans, including straightening, deepening (dredging), or diverting water (especially through concrete channels); or the addition of dams, bridges, artificial embankments, riprap, or other stabilization measures (Barbour et al., 1999).

(7) Frequency of Riffles (or Bends)

Measures the distance between riffles divided by the width of the stream in order to estimate the frequency of riffles, which are a source of high-quality habitat and diverse fauna. A highly sinuous stream with many bends will also provide for diverse habitat and fauna as the bends protect the stream from excessive erosion and flooding and provide refuge for benthic macroinvertebrates and fish (Barbour et al., 1999).

(8) Bank Stability

Percent measure of the extent of erosional conditions along the stream banks (steepness, crumbling, lack of vegetation, exposed tree roots and soil). Each bank is evaluated separately on a scale of 0-10, and the combined score is used for the parameter (Barbour et al., 1999).

(9) Bank Vegetative Protection

Measures the percent cover of the streambank and riparian zone by naturally growing native vegetation, including a mixture of trees, understory shrubs, and herbaceous species. Each bank is evaluated separately (0-10) and the combined score is used for the parameter. Vegetation increases the erosive resistance of the streambanks, helps to control instream scouring, provides nutrients to aquatic species, reduces contamination of runoff water, controls runoff volumes and velocities entering the stream, provides wildlife habitat, and controls extremes in water temperature (Mahood and Zukovs, 1993; Barbour et al., 1999).

(10) Riparian Vegetative Zone Width

Measure of the width of natural vegetation from the edge of the streambank out to substantial impact by human activities. The riparian zone vegetation controls erosion, provides habitat and nutrient input into the stream, and buffers the stream from runoff pollutants. Each bank is evaluated separately (0-10) and the combined score is used for the parameter (Barbour et al., 1999).

2.4.2 Benthic Macroinvertebrates

The U.S. EPA Rapid Bioassessment Protocols (Rev. 1) also prescribes a benthic macroinvertebrate sampling method for use in wadeable streams and rivers (Barbour et al., 1999). Macroinvertebrates are generally considered to include the invertebrates that are large enough to be seen with the unaided eye for most of their life history (Voshell et al., 1997). Benthic refers to organisms spending most of their time on the bottom of surface waters or on objects protruding above the bottom. A high proportion of freshwater benthic macroinvertebrates are insects that are only aquatic in the immature stage, while the adult stage is spent in the terrestrial environment (Kibler et al., 1998).

Multimetric indices are developed from data collected in a given region with similar ecological characteristics, and are not necessarily applicable outside the region. The ecoregion classification system developed by Omernik (1987) was based on comparable land surface form, underlying geology, land use, and potential natural vegetation. This classification system is the principal means used in the U.S. for organizing streams into homogeneous groups (Evans, 1997).

The purpose of a multimetric index is to provide a means of integrating information from the various measures of biological attributes (or metrics) and comparing the result to minimally disturbed reference conditions within the ecoregion (Barbour et al., 1999). To measure biological condition, patterns and processes from individual to ecosystem levels must be examined (Barbour et al., 1995). The results from the integration of different metrics, chosen to provide information on the response of diverse biological attributes to diverse stressors, can provide an indication of biological condition (Barbour et al., 1995). This is because of two main reasons. One, the weakness of any individual metric is minimized by the combined strengths of the metrics (Barbour et al., 1995). Two, the multimetric approach, adjusted for a particular bioregion, integrates information from individual, population, community, and ecosystem levels as required to assess biological condition (Barbour et al., 1995). The metrics chosen for the multimetric index will have some overlap in the ranges of sensitivity to help reinforce final conclusions but will also include metrics able to differentiate responses at the extremes of the range of impairment (Barbour et al., 1995). The single index value (a sum of the standardized metrics) is used to judge the biological condition of a stream as "acceptable" or unacceptable." The biological condition categories are based upon reference data used in the development of the multimetric index. Some states use multimetric indices for regulatory purposes. This is not the case in Virginia but multimetric indices are used for screening and as guidelines for management decisions (Dr. Voshell, personal communication, Dec. 7, 1999).

The taxonomic level for identification of macroinvertebrates is chosen based upon time, money, training, and study objective. Less taxonomic training is required to identify macroinvertebrates at the family level, as compared to the species level, and thus costs and sampling times are reduced (Lenat and Barbour, 1994). The use of family-level data requires that assumptions about the species composition for a particular family be made, particularly with reference to their ecological sensitivities; these assumptions are generally based upon regional expertise. The use of family-level data may not be appropriate when examining changes over time or the length of a recovery zone (Lenat and Barbour, 1994). Though subtle impacts may escape detection when using family-level identification or subsampling techniques, it is believed that almost any rapid bioassessment method will be able to identify severe impact in a typical upstream-downstream survey (Lenat and Barbour, 1994).

Nine metrics are used to calculate the Macroinvertebrate Aggregated Index for Streams (MAIS), developed by Smith and Voshell (1997) for the Central Appalachian Ridges and Valleys ecoregion (among others) within the mid-Atlantic highlands. These metrics are described below.

(1) EPT Index

These three taxa are all pollution-sensitive and thus useful as measures of perturbation in the environment (Kibler et al., 1998).

(2) No. Ephemeroptera Taxa

The number of Ephemeroptera taxa: this metric is used because Ephemeroptera are considered especially sensitive to pollution (Kibler et al., 1998).

(3) % Ephemeroptera

This metric is calculated as the number of organisms within this pollution-sensitive order as compared to the total number of organisms, expressed as a proportion (Kibler et al., 1998).

(4) % 5 Dominant Taxa

This metric is calculated as the number of organisms in the five most abundant taxa as compared to the total number of organisms, expressed as a proportion. In undisturbed locations, there are usually many evenly proportioned taxa. When pollution is present, the few tolerant taxa can comprise a high proportion of the total number of organisms (Kibler et al., 1998).

(5) Simpson's Diversity Index (SDI)

This is calculated as: $1-\sum [(n_i*(n_i-1))/(N*(N-1))]$
where:

n = number of organisms in a particular taxa in the sample
N= total number of organisms in the sample
i = from 1 to the total number of taxa in the sample

The SDI Index is a general measure of diversity that integrates richness and evenness with no assumptions about sampling. The result ranges from 0 to 1 and high values indicate good biological condition as hypothesized using niche theory (undisturbed systems have many taxa with few organisms in most taxa) (Kibler et al., 1998).

(6) Modified Hilsenhoff Biotic Index

This is calculated as: $\Sigma\, x_I t_I / n$ (3.6)

where:

x_I = number of individuals within a taxon

t_I = pollution tolerance value of a taxon (0 intolerantto10 tolerant

n = total number of organisms in a sample

This biotic index contains information about the numbers and kind of organisms along with their tolerance to pollution. Though originally developed to monitor organic pollution in Wisconsin (Hilsenhoff, 1982; Hilsenhoff 1987; Hilsenhoff, 1988), it has been modified to reflect responses to agricultural and urban nonpoint source pollution (including sediment as well as organic pollution) by stream fauna in Virginia (Kibler et al., 1998; Evans, 1997). The index criteria were based on spring sampling and Hilsenhoff recommends subtracting 0.5 (on a 0-10 scale) from the index when calculated from a summer sample (Lenat and Barbour, 1994).

(7) No. of Intolerant Taxa

The number of macroinvertebrate fauna with tolerance values of five or less (Smith and Voshell, 1997; Kibler et al., 1998).

(8) % Scrapers

Species designated as scrapers are specialized feeders who feed by scraping periphyton from solid surfaces. Moderate nutrient enrichment can increase this metric, but heavy enrichment or sedimentation will decrease it (Kibler et al., 1998). These feeders are thought to be more sensitive to pollution and well represented in healthy streams (Barbour et al., 1999).

(9) % Haptobenthos

This metric is calculated as the number of organisms that require clean, coarse, firm substrate as compared to the total number of organisms, expressed as a proportion. Sedimentation or excessive growth of algae, bacteria, or fungi will cause this metric to decrease (Kibler et al., 1998).

The nature of stream impairment may be better understood through the analysis of the component metrics and raw data in association with local reference data and other ecological information (Barbour et al., 1999). The analysis of individual metrics involves looking for "response signatures" which are unique combinations of biological community or assemblage characteristics that identify one impact type over others (Barbour et al., 1995). Additional metrics may be used to aid stream assessment, including the five metrics below.

(1) % Shredders

Shredders are insect larvae that feed on coarse particulate organic matter primarily for the bacteria and fungi growing on the leaves and other detritus particles (Smith, 1992). Limited litterfall in the stream channel and reduced amounts of stored and transported organic matter may diminish the role and density of shredders in the stream (Delong and Brusven, 1998).

(2) % Collector-Filterers

Species designated as collector-filterers feed on fine particulate organic matter (and associated bacteria) moving through the water. Collector-filterers may be more tolerant of zinc (Clements, 1994) and may be found downstream of eutrophic water impoundments that release significant quantities of fine particulate organic matter, including phytoplankton and zooplankton (Clements, 1994; Delong and Brusven, 1998).

(3) % Collector-Gatherers

Species designated as collector-gatherers feed on fine particulate organic matter (and associated bacteria) deposited on the stream bottom. Similar to collector-filterers, they may be found downstream of eutrophic water impoundments that release significant quantities of fine particulate organic matter, including phytoplankton and zooplankton.

(4) % 1 Dominant Taxon

This metric is calculated as the number of organisms in the most abundant taxon as compared to the total number of organisms, expressed as a proportion. This metric indicates the extent of disturbance to species composition within the macroinvertebrate community or assemblage.

(5) Total No. of Taxa (Taxa Richness)

The number of distinct taxa represents the diversity within the sample and reflects the health of the ecosystem; the presence of different species may indicate that adequate niche spaces, habitat, and food sources are available to support their survival and propagation. Generally, this metric decreases with decreasing water quality and habitat suitability (Barbour et al., 1999; Kibler et al., 1998).

Metrics that may indicate metal-polluted streams are those that assess reduced macroinvertebrate abundance, reduced species richness, changes in the proportion of major groups, and a shift in assemblage composition from sensitive taxa to tolerant taxa (Clements, 1994). Three stages of response to metal contamination have been identified: (1) heavy pollution causes the normal macroinvertebrate fauna to be eliminated, (2) moderate pollution reduces the diversity of fauna but Chironomids and Trichoptera (caddis flies) remain, and (3) mild pollution allows a greater number of Chironomid species to exist as well as Ephemeroptera (mayflies) (Hellawell, 1986).

Often trace-metal sensitivity depends upon duration of exposure and life-history attributes such as life-span, body size, development stage, and feeding behavior (Johnson et al., 1993; Gerhardt, 1993). For example, "studies with the isopod *Asellus aquaticus* showed that embryonic development was sensitive to cadmium, whereas juvenile development was more sensitive to copper; moreover, juveniles were more sensitive than adults, and males were more sensitive than females (Johnson et al., 1993). Invertebrates differ in their sensitivity to heavy metals because of genetically based or acclimation mechanisms including decreased uptake, increased excretion, or induced metallothionein production (Gerhardt, 1993). Resistant groups to metal pollution include Diptera, some species of Plecoptera (Perlodidae), and Trichoptera; while Oligochaetes and Ephemeroptera are intermediate, and molluscs and malacostracan Crustacea seem to be the most sensitive (Hellawell, 1986; Clements, 1994). Significant declines of Plecoptera have also been noted at metal-polluted sites (Clements, 1994). At the Clinch River in Virginia, Tanytarsini Chironomids were also found to be highly sensitive to metals (Clements, 1994). Chironomids in particular are more sensitive to Cu than damselflies, caddisflies, and stoneflies (Gerhardt, 1993). "Generally, insects appear to be less sensitive than gastropods and crustaceans to metal exposure" (Johnson et al., 1993).

Sediment, of all pollutants, may have the most effect on stream biota (Appelboom et al., 1998). Many studies have reported 45% to 90% decreases in desirable macroinvertebrate populations, as well as reductions in species diversity, associated with sediment increases of only 20 to 80 mg/L (Appelboom et al., 1998). Chironomids and Oligochaetes, pollution tolerant macroinvertebrate families, increase within the macroinvertebrate assemblage when sediment becomes the dominant habitat in the stream (Dr. Voshell, personal communication, 1998; Rosenberg and Resh, 1993). Suspended solids can reduce prey capture for sight feeders, clog gills, reduce spawning, and destroy the habitat potential of the stream bottom (Van Buren, 1994). Concentrations of 80-100 mg/L are the maximum concentrations that fish have been shown to be able to tolerate on a continual basis without causing gill damage (Knight et al., 1998).

Other pollutants such as nutrients and excess organic compounds will also create conditions unfavorable to a diverse macroinvertebrate assemblage. First, as BOD increases and dissolved oxygen levels decrease, the assemblage structure shifts to organisms tolerant of low oxygen demands (Chironomids, Oligochaetes, Gammarus, and Asellids) and tolerant of the highly saprobic conditions (Chironomids and Oligochaetes) (Dr. Voshell, personal communication, 1998; Rosenberg and Resh, 1993). Chironimids have a higher oxygen storage capacity because they have hemoglobin (Williams and Feltmate, 1992). Second, eutrophication will occur downstream of the organic matter source, creating dense algal mats which prevent attachment of clingers and scrapers on clean substrate, and also prevent the formation of thin surface films of edible algae the scrapers feed upon (Dr. Voshell, personal communication, 1998). It is hard to determine which of these impacts may be the specific cause of impairment, since all of the pollution types may be present; thus, the RBPs are used to make a general assessment of biotic condition.

Proper assessment of the recovery of a system to disturbance is dependent on the characterization of the variability of the assemblage on a spatio-temporal scale (Hill, 1997). Recovery may be assessed by the return of a species to an ecosystem or a return of the species to previous levels of abundance, but should not be confused with seasonal changes in pollutant concentrations, migration patterns, or a shifting age structure (Clements, 1994; Hill, 1997). The relative abundance of functional feeding groups also changes seasonally, particularly in response to the availability of particulate organic matter (Delong and Brusven, 1998). For example, densities of filter-feeders at lake outlets are often better correlated with food quality and quantity rather than pollutant concentrations (Clements, 1994). Furthermore, not all benthic macroinvertebrates are susceptible to the same pollutants because of different coping strategies including the following: (1) the ability to burrow and avoid toxic episodes, (2) body and gill movements to enhance oxygen uptake, (3) breathing at the surface by means of tracheal tubes, (4) adjustment of life-cycle to avoid periods of pollution stress, and (5) generation times short enough to avoid stressful periods (Williams and Feltmate, 1992).

Typically, urban runoff pollutants cause chronic (cumulative) effects, rather than acute effects, in receiving waters (Van Buren, 1994). Examples of these types of effects on fish are gill damage and subsequent suffocation due to acute toxicity of metals; and long-term lethality, effects on reproduction and growth, and physical or behavioral abnormalities due to chronic effects of metals (Van Buren, 1994). The extent of damage depends upon the amount of the pollutant and the characteristics of its chemical form including availability, activity, and mobility. For example, the partitioning and speciation of metals in stormwater is controlled by a number of chemical processes including adsorption, inorganic and organic complexation, and solid precipitation / dissolution (Van Buren, 1994). The most bioavailable and toxic forms, the free metals and some of the weakly bound metal complexes, are only a small fraction of the total metal concentration entering the receiving waters (Van Buren, 1994). However, under the right chemical conditions, the particulate and strongly-bound forms can dissociate over to become bioavailable (Van Buren, 1994). The behavior and fate of many pollutants, including metals, will change in the presence of other constituents within the water (ex: major ions) or in accordance with properties of the water (ex: alkalinity, hardness, pH, and salinity) (Van Buren, 1994).

Once a watershed has 10-15% impervious cover, biological quality appears to decrease by about 50%, and about 90% of the sensitive organisms are eliminated from the community (Maxted and Shaver, 1996). Maxted and Shaver (1996) sought to determine the effectiveness of stormwater retention ponds in the protection of wadeable nontidal stream resources after urbanization. They used a rapid bioassessment approach because it is a simple and cost-effective method for long-term studies of stormwater facility performance (Maxted and Shaver, 1996). In Delaware 8 stormwater management pond facilities (of which the predominant land use of two sites was commercial; residential for the remaining six sites), sampled in the spring of 1996, were evaluated by means of physical habitat and benthic macroinvertebrate metrics (Maxted and Shaver, 1996). These results were compared to results from three reference condition sites in the region as well as to 21 non-BMP sites with similar land use conditions sampled in the fall of 1993 (Maxted and Shaver, 1996). Maxted and Shaver (1996) sought to test the assumption that water quality treatment and pollutant capture would be translated into receiving system protection. They found that both BMP and non-BMP sites were significantly different from the reference condition, yet no significant difference was found between the results from the BMP and non-BMP sites in terms of the overall macroinvertebrate assemblage or sensitive species (Maxted and Shaver, 1996). The conclusion was that the "BMPs did not prevent the almost complete loss of sensitive species (e.g. mayflies, stoneflies, and caddisflies) after development" (Maxted and Shaver, 1996). Though each BMP facility was at least 2-years old, the investigators thought the results could be due to insufficient time for recovery between construction of the BMP facility and sampling. This interpretation was supported by the fact that three of the BMP sites were appearing to show some signs of habitat recovery (they had better habitat quality than would have been expected using the biological information) (Maxted and Shaver, 1996), which tends to come before biological community recovery. Thus, the researchers emphasize that their study should not be used to derive definitive conclusions on the ability of stormwater controls to protect stream biota and habitat (Maxted and Shaver, 1996).

BMPs can produce their own negative effects upon the stream biotic community. Released water may contain significant amounts of fine particulate organic matter, phytoplankton, and zooplankton that may increase the numbers of filter-feeders downstream (Delong and Brusven, 1998) and thus disrupt the species balance in the ecosystem. Limited litterfall in the stream channel and reduced amounts of stored and transported organic matter may diminish the role and density of shredders in the stream (Delong and Brusven, 1998). The results from Jones et al. (1996) suggest "appropriately designed and properly sited BMPs can provide some mitigation of stormwater impacts on stream communities. However, the resulting communities differ greatly from those in undeveloped watersheds and reflect a fundamental alteration in stream biotic diversity, structure, and function."

2.5 Literature Review Summary

Stormwater management facilities are implemented in order to reduce negative impacts of urban stormwater on receiving systems including streams and lakes. Design standards and specifications for BMPs are used in order to achieve removal of targeted pollutants. Water quality criteria and standards are specified for a few pollutants but even those may not be fully protective of the health of receiving waters.

Most often wet ponds and extended detention dry ponds are the BMPs selected to manage stormwater. Sedimentation is the primary pollutant removal process for both wet ponds and dry pond, but the lower detention time and the lack of other pollutant removal mechanisms typically causes lower pollutant removal efficiencies by the extended dry detention pond as compared to wet ponds or other stormwater management facilities with multiple pollutant removal mechanisms.

Maximum pollutant removal by wet ponds occurs when their design meets certain specifications. First, the permanent pool surface area should be approximately 2 to 3% of the surface area of the watershed (Athanas, 1988). Second, the permanent pool volume should be 4 to 6 times the runoff volume from the local mean storm event (Yousef and Wanielista, 1993). Third, the detention time within the pond should be approximately 2 weeks (Hartigan 1989). Fourth, the mean depth within the pond should be 1 to 3-m (Hartigan, 1989). Care should be taken to insure that the effluent of the wet pond does not degrade the receiving water further because of altered flow regime, high temperatures, or low oxygen content. Habitat and benthic macroinvertebrate assessment within the stream can be used to indicate impairment or improvement of the receiving waters as a result of BMP implementation, although any improvement to a stream due to BMP installation may be slow to appear.

CHAPTER 3.0 METHODS

3.1 Site Description

The study site is located within a subwatershed of the Stroubles Creek basin in Montgomery County, Virginia. The tributary joins Stroubles Creek just upstream of the U.S. Route 460 Bypass, and Stroubles Creek's confluence with the New River is approximately 8 miles downstream of the Virginia Tech campus (Hayes, Seay, Mattern & Mattern, Inc., 1995). The stormwater management site is bounded by the Virginia Maryland Regional College of Veterinary Medicine to the north, by Southgate Drive to the south, by Duck Pond Drive to the east, and by U.S. Route 460 to the west.

The study area lies within the Valley and Ridge geologic provinces in southwestern Virginia (Wolter, 1996). Quaternary-aged, fine-grained alluvium overlay the Cambrian-aged Elbrock formation (Wolter, 1996). The Elbrock formation consists of inter-bedded dolomite and limestone with lesser amounts of shale and siltstone (Wolter, 1996). This carbonate bedrock leads to characteristic karst formations. Karst formation can be accelerated by human-induced changes in the soil stress and hydrologic regimes due to construction activities (including site grading, building construction, and water impoundments) (Wolter, 1996). No sinkholes or other karst features were observed during the subsurface exploration and geotechnical analysis conducted in the region of the wet and dry ponds prior to construction (Wolter, 1996). Primarily, the subsurface exploration encountered rock outcrops and alluvial soils (fine-grained silts and clays (OL, ML, CL, CH) and coarse-grained alluvium (SC, SM)) over disturbed residual soils (Wolter, 1996). The soils in the area of the wet pond and dry pond are predominantly McGary (40%) and Purdy (35%) soils with the remaining 25% composed of Guernsey soils on low terraces and Ross and Weaver soils on the flood plains (Anderson and Associates Inc., 1996). Measurable subsurface water was encountered in 3 of the 29 borings in the uncased boreholes immediately upon completion (Wolter, 1996).

3.1.1 Design and Construction of the Stormwater Management Facility

Hayes, Seay, Mattern & Mattern, Inc. was commissioned by Virginia Tech to improve drainage conditions on the Virginia Tech campus and to develop a regional stormwater management plan to bring the university into compliance with Virginia Stormwater Management Regulations (VR

215-02-00) (Hayes, Seay, Mattern & Mattern, Inc., 1995). Alternatives considered for stormwater quality control included infiltration, retention basins, and extended detention basins (Hayes, Seay, Mattern & Mattern, Inc., 1995). Alternatives considered for quantity control included detention with wet or dry basins (Hayes, Seay, Mattern & Mattern, Inc., 1995). Upstream stormwater management would have been a preferred option in order to take advantage of flow attenuation and thus lessen flooding and stream erosion upstream, but lack of vacant and sufficiently large land area, except downstream of most campus development, prevented the use of this option (Hayes, Seay, Mattern & Mattern, Inc., 1995).

The Vet School Stormwater Management quality basin (Figure 3.1) was originally proposed as an extended detention basin (Hayes, Seay, Mattern & Mattern, Inc., 1995) to treat 1.27-cm (½-in.) of water over the disturbed land area (the water quality volume) for which the Virginia Stormwater Management Regulations required treatment (DCR 1998b). However, a retention facility was eventually designed and installed to retain approximately three times the water quality volume as required by Virginia Stormwater Management Regulations (DCR 1998b). The wet pond was completed in September 1997. Construction was delayed by unforeseen excavation needs and rock blasting which extended from February to August 1997. Seeding of the area surrounding the wet pond occurred in the first half of September 1997. The wet pond has a 2-yr storm event storage capacity (1184-ha-mm including 592-ha-mm of extended detention and 592-ha-mm of permanent pool), and a total volume of 1615-ha-mm (13.09 acre-ft). (Anderson & Associates, Inc., 1996).

Downstream of the wet pond outlet, approximately 290-m of natural stream was maintained. This stretch of stream leading to the dry detention basin receives flow from natural seeps and an off-line detention basin. The stream reach below the wet pond was considered by Hayes, Seay, Mattern & Mattern, Inc. (1995) to have moderate erosion potential as determined by using the average cross-sectional velocity in that reach: 0.91-1.52-m/s during the 2-yr design storm event.

The stormwater quantity basin (dry pond) was intended to provide stormwater management, as required by 1990 Virginia stormwater management regulations, to ensure that the post-development peak runoff rates from a 2-yr storm and a 10-yr storm did not exceed their respective pre-development rates (Wolter, 1996). The dry detention pond was designed to

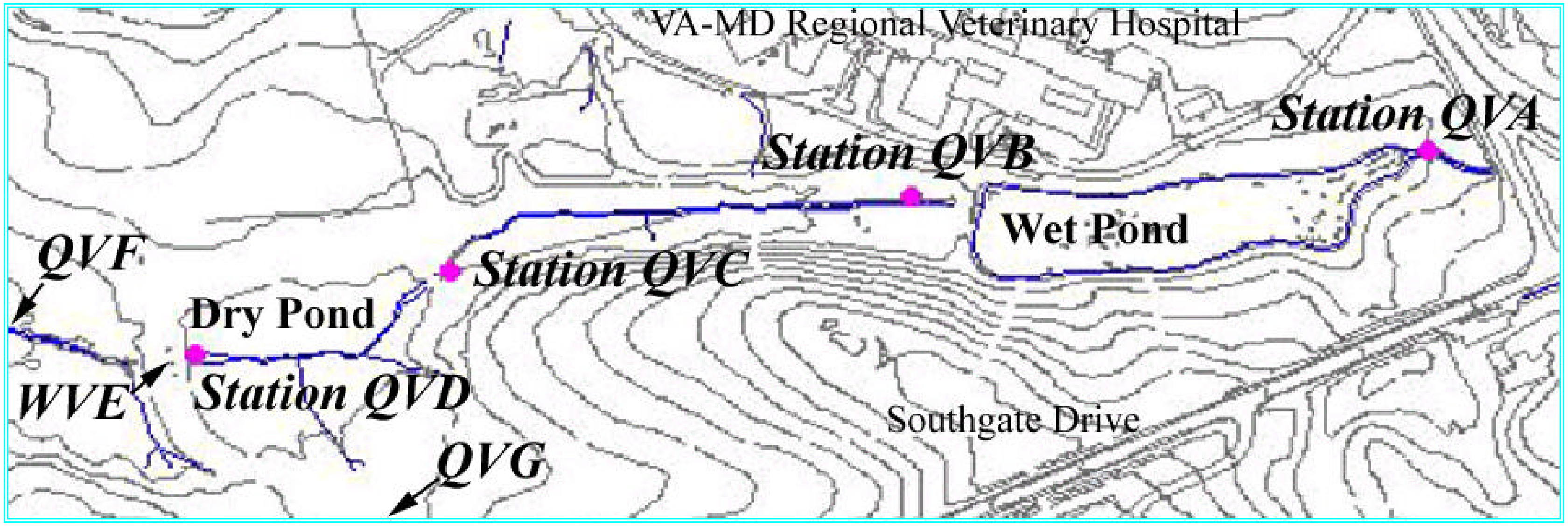


Figure 3.1. The VPI&SU stormwater management facility located adjacent to the Virginia-Maryland Veterinary School of Medicine

accomplish this objective considering all future growth in the watershed as planned by Virginia Tech (1994 Master Plan). Construction of the approximately 130-m long and 90-m wide dry pond was initiated in February of 1997 and completed in April of 1997. The dry detention basin also receives additional runoff from a subwatershed totaling approximately 44-ha, where the land uses include feed corn cultivation, dairy pasture, and manure spreading fields. The dry detention basin retains 2933-ha-mm (23.78-acre-ft) of stormwater, calculated as the volume of runoff from a 100-yr storm event from the watershed. Table 3.1 shows the design specifications for the dry pond.

Table 3.1 Quantity pond design flow rates for storm events of various sizes (Anderson & Associates, Inc., 1996)

Storm Event:	**2-yr**	**10-yr**	**100-yr**
Pre-installation peak flow (m^3/s)	9.3	13.8	18.0
Post-installation peak flow (m^3/s)	7.7	11.7	16.5
Reduction in peak flow (%)	17.0	15.1	8.3

During low-flow conditions, the stream meanders in an open channel through the dry detention basin. The dry detention basin embankment impounds higher flows to be released at a rate controlled by the outlet structures.

3.1.1 Watershed Characterization

The stormwater management facility composed of a dry pond and a wet pond (Figure 3.1) captures runoff from a 238-ha watershed, primarily the VPI&SU campus area surrounding Lane Stadium. The watershed encompasses a veterinary hospital; campus and office buildings; parking lots; construction sites; athletic facilities; lawn areas; agricultural activities including cornfields and dairy cattle pasture; a cross-country running track which runs around both ponds; a sewer line running along the stream the entire length of the stormwater management facility; wetlands below the wet pond and within the dry basin; and a vehicle maintenance facility. Land use delineation was included in the report written by Hayes, Seay, Mattern & Mattern, Inc. (1995), regarding Virginia Tech property within the Stroubles Creek watershed. Limited soils information for the watershed was obtained from a project report by Anderson and Associates, Inc. (1996) regarding the pavement of the Virginia Tech Lane Stadium parking lot. Anderson and Associates, Inc. report that the soils near the stadium are predominantly Udorthents (45%), with 25% other soils and 30% urban land. Digital topographic files of the Virginia Tech campus (created from 1998 aerial photography, 1”=50’) and of the Town of Blacksburg (created from 1991 aerial photography, 1”=100’) were acquired and projected into ArcView GIS 3.2 (Environmental Systems Research Institute, Inc., Redlands, CA) from Virginia State Plane, South Zone, North American Datum (NAD) 1983 coordinates (ft) to UTM 1927 Zone 17 coordinates (m). Using a hardcopy of the topographic layers and campus buildings from the Virginia Tech digital files, the sub-watershed boundaries from the five monitoring stations (including the QVG grab sample monitoring site) were identified and then verified in the field. The watershed outlines were digitized in ArcView GIS and the subwatershed areas were calculated.

In the upper portion of the watershed, grate inlets from the campus intercept the storm runoff and discharge it to a 1.83x1.52-m box culvert that runs beneath the ground (Anderson and Associates, Inc., 1996). The box culvert also accepts water from twin 1.22-m concrete pipes from the north, and one 1.22-m concrete pipe from the east (Anderson and Associates, Inc., 1996). The box culvert runs west beneath Duck Pond Drive and discharges to a short section of stream channel above the wet pond (Anderson and Associates, Inc., 1996). While the box culvert discharges continuous flow, the wet pond also receives intermittent flow from several culverts (runoff from storm events).

Various chemicals and nutrients are applied to Virginia Tech properties primarily for landscape maintenance, which may then show up in the runoff that enters the stormwater management facility. Summarized here are the applications of chemicals to roads, lawn areas, and the athletic facilities.

The road that runs parallel to the wet pond adjacent to the VA-MD Veterinary School receives applications of road salt (NaCl or CaCl) and fertilizer when ice and snow accumulations are expected (Jerry Dobbs, personal communication, Nov. 15, 1999). In general, road salt is applied to the roads on campus with a chemical spreader installed on a truck, at the rate of 227-kg/lane mile (Bill Swain, personal communication, Nov. 22, 1999). In a given winter, approximately 45-metric tons of road salt is used on the Virginia Tech campus (Bill Swain, personal communication, Nov. 22, 1999). Potassium Chloride (KCl) is applied as a slow release deicer to campus sidewalks, particularly in the areas of steps and ramps (Jerry Dobbs, personal communication, Nov. 15, 1999; Bill Swain, personal communication, Nov. 22, 1999).

Maintenance of grass (including the swale surrounding the wet pond) requires fertilizer applications. Virginia Tech uses slow-release (sulphur coated) fertilizer with a N-P-K ratio of 32-5-7 (Jerry Dobbs, personal communication, Nov. 15, 1999). The application rate is calculated as 0.45-kg of nitrogen / 93 m^2 of lawn (Jerry Dobbs, personal communication, Nov. 15, 1999). Many of the athletic facilities on campus are located in this watershed, including the football, soccer, softball, track, and cross-country track facilities. The cross- country track winds around both the wet and dry ponds. Information on fertilizers used by the Virginia Tech Athletic department on their grounds is summarized in Table 3.2 based on applied acreage and total chemical application (not % of active ingredient) (Casey Underwood, personal communication, Jan. 4, 2000).

Table 3.2 Virginia Tech Athletic Department fertilizer usage (Casey Underwood, personal communication, Jan. 4, 2000)

Fertilizer	**Rate (kg/ha)**	**Hectares**	**# Applications/yr**
Nitrogen	48	6	3
Urea	48	0.4	7
Phosphorus	15	6	3

In November 1994, Virginia Tech developed a nutrient management plan for the 748-ha of agricultural land that it manages. The plan was officially approved for a Virginia Pollution Abatement permit in January of 1996 (Dean Gall, personal communication, Feb. 29, 2000). The nutrient management plan is phosphorus-based; it includes liquid storage for 330 dairy cattle (storage volume for longer than 6-months), and accounts for 527 total dairy cattle (Dean Gall, personal communication, Feb. 29, 2000). Each field where liquid manure is applied (162-ha of corn, 65-ha of alfalfa, 24-ha of barley, 121-ha of grass/clover/hay, and 202-ha of pasture) receives one application a year, either in spring or fall, of an amount less than 101-kg/ha (Dean Gall, personal communication, Feb. 29, 2000). Manure is cleaned from around the feeding areas and redistributed over the pasture (Dean Gall, personal communication, Feb. 29, 2000). Dry stack manure (typically horse, beef cattle, and sheep manure) is hauled to outlying campus farms (Dean Gall, personal communication, Feb. 29, 2000).

With respect to fecal coliform within the watershed, there are potentially several major sources. First, there is a sewer line that runs parallel to the stormwater management facility; any leak in the pipe would likely discharge into the stream with little mitigation. Second, as noted by Weiskel et al. (1996), a major source of fecal coliforms to stormwater is the feces of domestic animals and wildlife accumulated on paved surfaces during dry periods (Weiskel et al., 1996). Impervious surfaces dominate the watershed upstream of the wet pond. While the area immediately around the stormwater management facility is of a more rural nature, it is also a popular location for people to take their dogs for a walk (with no encouragement or enforcement of feces removal), as well as an area with relatively abundant wildlife. Furthermore, the wet pond has provided a desirable location for waterfowl. Third, dairy cattle deposit manure on fields within a subwatershed of the dry pond and on at least one field that drains into the wet pond. Liquid manure is also applied to fields in the dry pond subwatershed; these applications may also contribute to the overall fecal coliform load.

3.1.2 Sampling Locations

Four flow and water quality monitoring stations have been established: one above and one below each of the water quality and quantity ponds (Figure 3.1). These were installed in order to measure hydrologic and flow-weighted pollutant concentrations coming into and leaving the basins. The first pair of stations includes site QVA, located between the 1.5 x 1.8 box culvert outfall and the wet pond entrance (6-m upstream from the culverts above the wet pond), and QVB, located 31-m downstream of the wet pond outlet. The second pair of stations was located at the entrance (QVC: 2.5-m from the culverts above the dry pond) and outlet (QVD) of the dry detention facility. Station QVD was relocated several times early in the study. Prior to construction of the dry detention basin, the monitoring was conducted in the natural stream channel. When construction began, QVD was relocated to the diversion channel that routed water around the embankment construction area. Once the quantity pond embankment was completed, the QVD site was moved to the outlet of the principal spillway of the quantity pond. Another major source of storm flow located to the south of the quantity facility is the runoff from dairy pastureland, routed through a culvert under Southgate drive. Site QVG was established at the outlet of this culvert in order to characterize the pollutant loads from the dairy pasture. QVG is located 200-m from the stream and its runoff joins with the stream in the center of the dry detention basin, 90-m from the dry pond dam, during low flow events. During high flow events, some of the runoff appears to join the stream close to the dry pond outlet.

3.2 Pond Characterization

3.2.1 Pond Configuration and Morphology

The inlet of the wet pond is protected by riprap. The riprap was placed on both sides of the earthen bridge that covers the corrugated metal pipes which discharge stream water into the wet pond. The inlet spillway for the wet pond is primarily bedrock and empties into a narrow section of the pond (Figure 3.1). During large storm events, the streamflow overtops the earthen bridge and flows into the pond along its banks. The permanent pool elevation was designed to be 615.43-m above sea level. The wet pond earthen embankment (1.2-m tall, total elevation 616.5-m, 6.1-m wide, 4:1 side-slopes) was designed to be overtopped during storms greater than the design storm; the top was designed as a trapezoidal weir with a 30.48-m width at 616-m elevation and 5:1 side slopes. Soil stabilization matting was installed over the embankment to protect it from overflow events, but the embankment was overtopped with almost every storm event due to the lack of sufficient drawdown rate expected according to the design. Riprap was used to protect the discharge site.

Several times during the winter and early spring of 1999, the wet pond was drained manually by about 0.26-m (the extended detention storage volume) because sufficient drawdown of the water level was not occurring within 48 hours of the storm as specified by the design (Leon Law, personal communication, March 1999). In April of 1999, the pond water level was permanently lowered (by approximately 0.26-m depth) in order to view erosional damage (piping failure) that was occurring to the wet pond embankment (Janet Smith, personal communication, March 1999). The elevation of the water at the wet pond embankment was measured to be 615.3-m (0.13-m lower than the design elevation) on June 18, 1999 (Anderson and Associates, Inc., Stormwater Facility Modifications Site Plan Document # 16889-002, 1999). The pond depth was designed to be approximately 0.65-m (a base elevation of 614.78-m on the plan drawings as compared to the permanent pool elevation of 615.43-m) and the pond can be drained by 0.5-m with the pipes fully open. Numerous large boulders were intentionally placed within the pond basin to enhance the aesthetics of the pond and change its flow characteristics. Using the Virginia Tech and Town of Blacksburg digital topographic files and survey data collected of the pond perimeter and interior depths, calculations were made of the wet pond surface area, volume, and length of the permanent pool. To calculate pond surface area, the pond outline was digitized in ArcView GIS from the Virginia Tech digital files, and ArcView GIS performed the area calculations. The survey data were also used to calculate pond surface area and volumes within ArcView GIS.

An off-line dry detention basin discharges into the stream reach between the wet pond and dry pond via a 91.44-cm diameter corrugated metal pipe with an invert elevation of 613.60-m. The small earthen bridge immediately below site QVC covers four 60.96-cm diameter corrugated metal pipes which discharge into the dry pond.

The dry pond flow is discharged through a 3-m tall earthen embankment (embankment elevation: 614.14-m). The inlet of the dry pond outflow culvert is a modified VDOT standard EW-11 with bar spacing reduced to 0.305-m. The invert of the 30-m long, 61-cm diameter reinforced concrete pipe, 0.6% slope, is at 611.09-m elevation, and the outlet is at 610.91-m elevation. The emergency spillway for the dry pond embankment, located at 612.65-m elevation, is a trapezoidal weir with a 3.66-m bottom width and 3:1 side slopes. Soil stabilization matting was installed over the emergency spillway (Anderson and Associates, Inc., 1997).

3.2.2 Physical Parameters

The pond perimeter was surveyed in June 1999 in preparation for pond measurements made using a 4.3-m long rowboat. Further surveying was performed in June 1999 of the pond interior sampling locations, and additional areas of pond inflow including overland flow channel erosional features and the outlet of the storm sewer pipes discharging to the wet pond.

3.2.2.1 Pond Water Measurements

On June 5, 1999, a field crew collected temperature, conductivity, and dissolved oxygen data from 20%, 40%, and 80% of the total depth at 13 sampling locations within the wet pond (Figure

3.2) in order to estimate stratification of the pond in the summer season. Temperature and dissolved oxygen were measured in the pond profile using a YSI 58 dissolved oxygen meter and a YSI 5700 Series dissolved oxygen probe in order to obtain an estimate of stratification. Conductivity was measured in μmhos with an YSI Model 33 S-C-T (salinity-conductivity-temperature) instrument. These probes were lowered to the appropriate depth of measurement and readings were recorded upon stabilization of the digital reading. Sampling locations were also surveyed. A different data collection protocol was used on Oct. 14, 1999 to obtain measurements of pond depth, pH, conductivity, dissolved oxygen, and temperature in order to obtain an estimate of stratification of the pond during the fall season. The depths at which measurements were taken were 18-cm, 55-cm, and 73-cm from the pond surface except where the depth was insufficient for these locations. These depths were based on 19.7%, 60%, and 80% of an average 91.4-cm. depth of the pond. A YSI 95 handheld dissolved oxygen probe with a MicroElectrode sensor was used to collect measurements of temperature and dissolved oxygen. In order to obtain readings of conductivity and pH using handheld instruments (Corning pH-30 instrument, and TDSTestr 3 instrument), water samples were collected with a LaMotte 1054-DO limnological water sampling body and 2-oz.glass collecting bottle at the different depths, then transferred to HDPE 250-ml bottles for measurement.

3.2.2.2 Sediment Composition

A preliminary determination of the location, extent, and type of sediment deposition was performed in the wet pond. This information is minimal, but can be used in the design of further sampling programs, as an aid in the selection of sediment sampling locations. The core samples were used in this study to obtain information about the hydraulic transport gradient within the wet pond.

Samples were collected at 2-3 locations across four latitudinal cross-sections of the pond, then at three additional locations in the narrow upstream neck of the pond. The location of each sample within the pond was surveyed. Hand cores were taken in the field on 9/24/1999, stored at 4°C, and frozen prior to cross-sectional analysis. Each sediment core was collected with a 5.1-cm inner diameter PVC tube, 100-cm in length. The PVC pipe was gently pressed into the sediment,

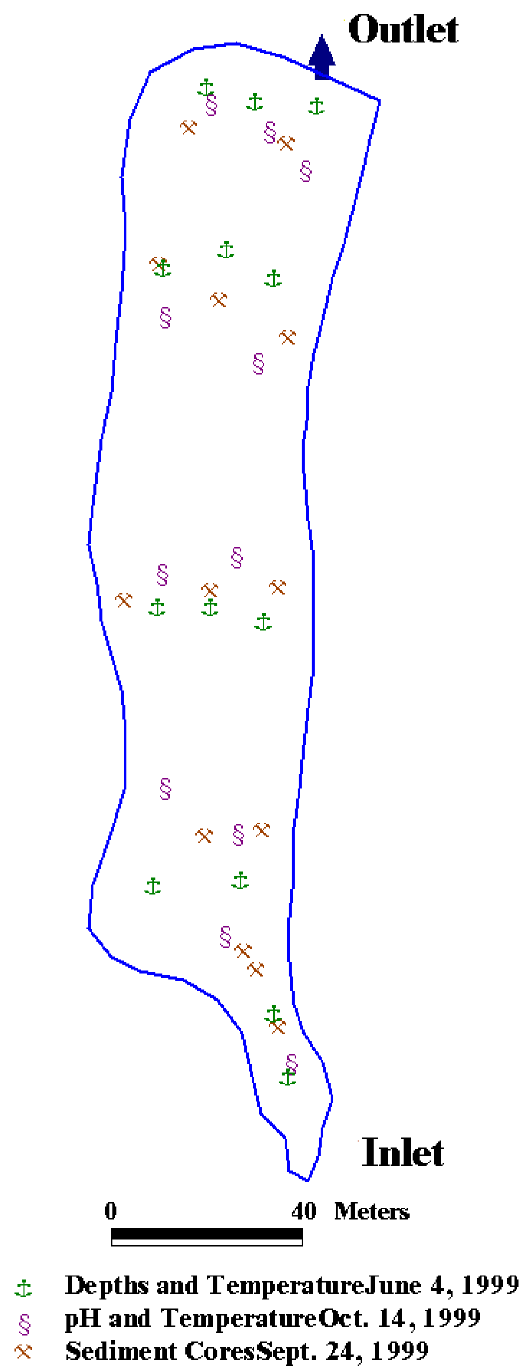


Figure 3.2 Sampling locations within the wet pond labeled according to collection date: June 5, Sept. 24, or Oct 14

then driven down as far as possible (until a bedrock or clay layer was met) using a rubber-mallet. The top of the tube was capped, the core was withdrawn from the sediment, and a cap was placed on the bottom prior to placing the sample upright in the boat. The tubes were kept in a vertical position to the extent possible during both transport and storage.

Each PVC tube sample was secured in a wooden frame prior to cutting them in half using a circular saw. The core sections were washed with distilled water to better view the frozen layers of the core. A picture was taken of each core, and the layers were described and measured with a ruler. The unconsolidated layers were stored for further analysis to assess settling characteristics of the pond. Because no liner was installed in the wet pond, the identification of the layers due to sedimentation after pond construction (versus construction debris) could not be very accurate. If there was sufficient sediment to allow separation of the retained layers (approximately 20-g was needed to perform all analyses), layers were separated by color and texture and placed in acid washed glass containers. Each core was thus divided into 1 to 4 subsamples. The subsamples were stored at 4°C until further analysis.

The subsamples were dried overnight in a VWR Scientific forced air oven at 80 to 90°F, then crushed by a rubber mallet in order to enable processing them through Fisher Scientific U.S. Standard Sieves #4 and #10 (Series according to ASTM specifications). The mass of the particles for each subsample that were larger or smaller than the #4 and larger than the #10 sieve were recorded. The portion of each sieved subsample that passed through the #10 sieve was used for particle size analysis. Following this analysis, the remaining portions of the subsamples were analyzed for pH, phosphorus, organic matter, copper, and zinc.

3.1.1.1 Residence Time

Dye tracing was performed to aid the understanding of flow through the wet pond. An inert dye (16972 Rhodamine WT 20%: Acid Red 388) was released in the stream above the pond inlet during two separate occasions (initiated 9/27/99 and 10/28/99), and then sampled at the pond outlet over time. The results allow for some assessment of residence time under two different flow conditions (rainfall occurred throughout the week of the first dye trace). Flow was measured by use of the Marsh-McBirney flow meter in conjunction with data collected by the QVB flow monitoring station during the course of sampling.

Fifty-ml of 20% rhodamine WT dye (0.238-g/mL) was mixed with 5-L of stream water within the dye dispenser on each occasion. A 20-ml elutant sample was removed and the remaining dye was released in a sheet through the 93.3-cm long, 2.6-cm wide longitudinal slot in the PVC tube dye dispenser (9.9-cm I.D., 148.3-cm long, capped at both ends) into the stream cross-section at the location of the QVA staff gage (approximately 6-m upstream of the pond inlet culverts) (Figure 3.1). This method of dye release was used in an effort to assure even distribution of dye across the middle 75% of the channel as recommended by the U.S. Geological Survey (Wilson et al., 1986). The somewhat turbulent conditions created by the rough bedrock surface at the pond inlet immediately below the culverts helped to ensure complete mixing of the dye in the stream water as it entered the pond basin.

All samples were collected in 25-ml Fisher Scientific Dilu-vials. Background samples were collected the day before and immediately prior to dye release both above and below the wet pond. Samples were collected from the outfall of the wet pond concrete outflow culvert on a regular interval (typically 30-min) for the length of time until the dye concentration fell to background levels (202-hrs for the first dye run and 169-hrs for the second dye run). Towards the end of the sample collection period samples were collected at longer time intervals (1-hr, 2-hr, and twice daily). All samples were stored in coolers to prevent photodegradation, and were analyzed as soon as possible for fluorescence characteristics.

Fluorescence of the samples was measured using a Cytofluor II multiwell fluorescence spectrometer (PerSeptive Biosystems, Bedford, MA). The appropriate filters were used to excite the sample and filter the emission for assessment of rhodamine WT concentration; the excitation filter used was rated for 530-nm ± 25-nm, and the emission filter was rated for 590-nm ± 35-nm. Three to four replicates were possible per sample due to the small (200-μL) subsample size needed for analysis. The subsamples were transferred from the collection vials into the 0.3-L wells on the Dynatech Immulon plates, using disposable pipette tips. Each Dynatech plate was assessed for background fluorescence prior to loading the samples. Samples were not placed into wells of the Dynatech plate where scratches were observed.

The relationship (rating curve) between fluorescence and dye concentration was developed using eight standard concentrations (1, 2, 4, 6, 10, 15, 20, 25-μg/L) created in the laboratory following procedures outlined by Wilson et al. (1986). The average fluorescence reading of the replications for each sample was used to calculate the dye concentration in the sample using the developed rating curve. The dye concentrations could then be plotted over time in order to visualize the dye movement through the pond. In this fashion, the dye peak was noted and sampling was terminated when the dye concentrations fell to background levels after the peak had passed.

The median residence time in the pond was calculated as the time at which 50% of the dye mass had been recovered (Martin 1989). In order to calculate the mass flux of dye through the pond, the background concentrations were subtracted from the measured concentrations (Martin, 1989). Hydraulic efficiencies were calculated to represent the pond's deviation from the ideal plug flow or completely mixed flow conditions (Matthews et al., 1997). The breakthrough curves were examined for evidence of short-circuiting, and the method outlined by Thackston et al. (1987) was used to calculate the degree of short-circuiting.

3.1.1.2 Water Budget

The wet pond and the dry pond are located in a region where karst features are often observed, though none were found in the subsurface exploration and geotechnical investigation conducted in the immediate vicinity of the ponds. Yet some concerns arose regarding the performance of the wet pond that might indicate some influence from the groundwater regime. Initial concerns arose when the wet pond took weeks rather than the expected 48-hours to accomplish drawdown of the 592.1-ha-mm extended detention volume (0.26-m depth over the pond) after a storm event (Leon Law and Janet Smith, personal communication, April 4, 2000). To investigate this matter, groundwater-monitoring wells were installed around the wet pond and at downstream locations. Dye traces from the inflow of the wet pond and from the upstream well were performed by Virginia Tech and DCR (in particular Terri Brown of the Dublin DCR office) to determine if the unlined wet pond was losing water to the karst aquifer under maximum hydraulic head, and to determine movement of the groundwater in the vicinity of the stormwater management facility (Terri Brown, personal communication, Sept. 4, 1998).

Furthermore, a water budget was constructed using information acquired from the flow monitoring stations above and below the wet pond as well as the rainfall and evaporation data collected at the weather station installed at the study site. Evaporation was measured from May 11 through Nov. 3 using a U.S. Weather Bureau Class A Land Pan. The pan was supported on a wooden frame 15.24-cm (6-in.) above a gravel surface. The water level, maintained between 5.08 to 7.62-cm (2 to 3-in.) below the rim of the pan, was measured daily with a micrometer hook gage in a brass stilling well, located as prescribed by McGuiness et al. (1979). Evaporation readings were adjusted as appropriate for water additions, rainfall, changes in water level due to cleaning, and duration of time between measurements. A pan coefficient of 0.7 was used to convert the observed pan evaporation to an estimate of pond evaporation (true values of this coefficient range from 0.5 to 0.8 influenced by the pan surroundings, fetch, relative humidity, and wind speed (Saxton and McGuiness, 1964)) (Veihmeyer, 1982). This choice of pan coefficient assumes conduction through the pan is negligible and the pan is located so that the exposure conditions are representative of the wet pond (Viessman et al., 1972).

Total incoming water volume to the pond is a combination of the stream inflow (including both spring-fed flow and runoff), side storm-sewer discharges (however not measured here except with regards to their contribution in the outflow), precipitation, and groundwater discharge. Outflows include evaporation extrapolated over the area of the pond, and seepage to the groundwater. Groundwater influences were estimated using Equation 3.1 (Watt and Marsalek, 1994) for the wet pond.

$$I + P - O - E = \Delta S \qquad (3.1)$$

Where:

I = inflow runoff volume
P = precipitation over the pond surface
O = outflow volume
E = evaporation from the pond surface
Δ S = change in volume due to groundwater recharge or discharge

3.2 Rainfall Measurement

A weather station designated as WVE (Figure 3.1) was installed to monitor meteorological variables important for measuring storm event magnitudes as well as for establishing a water budget for the quality basin. The Campbell Scientific MetData1 weather station (Campbell Scientific, Inc., 1997) is composed of a tower supporting multiple instruments connected to a CR10X datalogger for automated measurements. Data collected on a 10-minute basis included basic meteorological variables such as rainfall, air temperature, solar radiation, wind direction and speed, and relative humidity. Barometric pressure was also measured on a 60-minute interval, and pan evaporation measurements were made manually on a daily basis.

Rainfall was measured by a tipping-bucket raingage (Campbell Scientific, Inc., 1996) located 76-cm (30-in.) above the ground on the dry pond embankment (and sufficiently far away from tall objects). The number of 0.01-in. bucket tips (measured as electrical pulses) within each 10-min interval was recorded by the CR-10 datalogger installed at the weather station. From this rainfall data, calculations could be made including storm event antecedent conditions and rainfall amounts and durations.

The amount of rainfall that occurred in the 24-hr period prior to a given sample was used to discriminate between baseflow and storm grab samples. Baseflow samples had less than 0.254-cm. of rain within the previous 24-hr period, with the exception of one sample on 2/11/1998 that had 0.2-cm. of rainfall by the end of the sampling period.

When periods of needed rainfall record were missing (primarily prior to installation of the weather station at WVE), several other rainfall records were substituted. The secondary source of data came from datalogger records of a tipping-bucket raingage located at a dry detention facility on the Virginia Tech campus (approximately 0.8-km distance NW from the dry pond embankment). The remainder of the missing rainfall information (Feb., March, June, and Nov. 1997; April 1, 1998) was taken from records kept by the National Weather Service in Blacksburg, VA (Michael Gillen, personal communication, Jan. 14, 1999).

The design storm rainfall amounts used in the stormwater management models and calculations of Hayes, Seay, Mattern & Mattern, Inc. (1995) are shown in Table 3.3. These design values can be compared with the storm events that occurred during the monitoring period in order to aid the evaluation of stormwater management facility performance. The historical data for Blacksburg (1961 to 1990) show that the average precipitation of the region is 103.9-cm. (ranging from 7.01-cm in Jan. to 10.26-cm in May), the average snowfall is 58.7-cm., and the average temperature is 10.6°C (ranging from –1.33°C in Jan. to 27.9°C in July) (http://www.people.virginia.edu /~climate/Climate/normals/ 440766_30yr_norm. html, Jan. 22, 2000). The average monthly precipitation data for Blacksburg, provided by the National Weather Service Office located in Blacksburg, Virginia are given in Table 3.4.

Table 3.3 Design storm rainfall depths (Hayes, Seay, Mattern & Mattern, Inc., 1995)

Return Frequency (years)	**Total Rainfall (mm)**
2	76.2
10	124.46
25	139.7
50	152.4
100	177.8

Table 3.4 Monthly summary of Blacksburg rainfall (cm) for 1997-1999 and the thirty-year normal (Michael E. Gillen, personal communication, Jan. 24, 2000)

Month	**1997**	**1998**	**1999**	**Normal**
January	7.90	18.77	8.79	7.01
February	6.50	9.02	6.15	7.34
March	9.98	11.63	7.26	9.04
April	6.22	12.78	8.28	9.19
May	5.84	18.08	6.48	10.26
June	12.34	15.09	3.48	8.66
July	6.65	3.56	12.07	10.19
August	5.00	11.61	7.98	9.58
September	8.79	2.31	12.60	8.92
October	3.61	6.96	4.09	9.22
November	6.20	1.70	3.35	7.34
December	6.02	6.93	4.88	7.16
Annual	85.8	118.4	85.4	103.9

3.3 Runoff Characterization

3.3.1 Flow Measurement and Flow-Weighted Sample Collection

Each monitoring station was equipped with a staff gage to measure the stage in the stream, stage recording equipment (a KPSI pressure transducer and a Campbell Scientific CR-10 datalogger (Campbell Scientific, Inc., 1993)), and an automatic water sampler (ISCO 3700 Compact Portable Sampler (ISCO, 1993)). The pressure transducer used in this study was an isolated diaphragm sensor, with a silicon pressure cell in a stainless steel barrier diaphragm. It was wired as a full bridge, and calibrated by calculating the voltage response to current and head and performing a regression to staff gauge measurements over a range of responses.

An equation was developed for each pressure transducer to relate water depth as measured by the staff gage to the mV response from the pressure transducer. Sampling was initiated by a signal from the datalogger triggered by a change in stream stage. During a storm event the ISCO sampler collected a sample after a certain flow volume had passed (4500 for QVA, 17000 for QVB, 33500 for QV, and 7000 for QVD). This flow volume was calculated using Mannings equation (Rantz et al., 1982) (Equation 3.2) and adjusting the volume until the sample numbers were equivalent among sampling sites for a given storm, and the sampling intervals were frequent enough to catch small storm events but not too frequent that more than 4 - 8 bottles were needed for moderate storm events.

$$Q = (AR_h^{2/3}S_0^{1/2})/n \qquad (3.2)$$

Where Q = discharge (metric units) calculated by the Manning equation
A = cross-sectional area of the channel (m)
R_h= hydraulic radius (m) = A/WP
WP = wetted perimeter
S_0 = friction slope (m/m)
n = roughness coefficient

Flow was calculated within the datalogger program based upon a unitless adaptation of the Mannings equation (Equation 3.3) achieved by the removal of two constants in the equation -- the friction slope and the roughness coefficient.

$$Q = (A^{5/3})/(WP^{2/3}) \qquad (3.3)$$

Where Q = unitless discharge estimate
A = cross-sectional area of the channel (m)
WP = wetted perimeter of the channel (m)

A rating curve was developed at each sampling location to relate the observed stage as measured by the staff gage to the flow rate through the stream cross-section as measured by a Marsh-McBirney, Inc. Flo-Mate Model 2000 current meter. These rating curves in association with the equations relating pressure transducer output to stage were applied in data analysis when flow volumes were calculated from the pressure transducer record for the purpose of obtaining pollutant loads.

3.3.2 Water Quality Sampling and Analysis

Three different bottles were used to collect nutrient, metal, and biological samples. Nutrient and metal water quality samples were collected in 500-ml and 250-ml high-density polyethylene (HDPE) bottles respectively, which had been rinsed in a 10%-HCl acid bath after cleansing. Bacteriological samples were collected in sterilized (autoclaved) 250-ml bottles. Storm composite samples were collected in multiple 1-gallon sample containers (also rinsed in the 10%-HCl acid bath after cleansing) in the field and transferred to 250-ml sample containers for laboratory analysis.

Since February 1997 and prior to January 26, 1999 grab samples were taken at all four primary sampling sites (QVA, QVB, QVC, and QVD) on a monthly basis during baseflow periods and on a monthly basis during storm events when possible. Grab samples were also collected at site QVG during storm events in an attempt to characterize runoff from its subwatershed. Physical parameters of water quality including temperature, pH, conductivity, and dissolved oxygen (DO) were measured in the field when grab baseflow samples were collected and when grab storm samples were collected prior to automated sampler installation. Temperature, pH, and conductivity were usually measured with the hand-held instruments placed in the water sample bottle intended for nutrient analysis. Temperature was measured with a Fisher Scientific long-stem digital Multi-thermometer. Both a Hanna Instruments pH tester and a Corning pH instrument were used for pH readings. A TDSTestr 3 with ATC was used to measure conductivity and dissolved oxygen was measured using a CHEMetrics K-7512 acidic indigo carmine kit (CHEMets).

After January 26, 1999, when automated samplers were installed, grab-sampling frequency was increased to bimonthly for baseflow conditions and automated composite flow-weighted samples were collected during storm events. Because automated samples were not analyzed for bacteria and metals, separate grab samples were collected for two selected storm events during each month.

All nutrient and biological samples were taken to the Department of Biological Systems Engineering (BSE) Water Quality Laboratory at VPI&SU for analysis of total suspended solids (TSS), total kjeldahl nitrogen (TKN), filtered TKN (FTKN), ammonia (NH_4), nitrate (NO_3), total phosphorus (TP), filtered TP (FTP), orthophosphate (PO_4), total organic carbon (TOC), chemical oxygen demand (COD), total coliforms, fecal coliforms, and fecal streptococci.

Extraction and analytical procedures were followed by the Biological Systems Engineering Water Quality Laboratory as outlined in Standard Methods for the Examination of Water and Wastewater, 18th Ed. (APHA et al., 1992) and according to U.S. EPA approved standard operating procedures (Mostaghimi and McClellan, 1999). Nutrients were analyzed with a TRAACS 800 continuous flow wet chemistry analytical system using colorimetric techniques. The following methods are referenced in U.S. EPA (1983a) and modified for the TRAACS 800 system as specified in its manual: Ammonia: EPA 350.1, TRAACS 780-86; Nitrate: EPA 353.1, TRAACS 782-86; Orthophosphate EPA 365.1, TRAACS 781-86; TKN: EPA 351.2, TRAACS 786-86; Total-P: EPA 365.1, TRAACS 787-86; TSS: EPA 160 (Mostaghimi and McClellan, 1999). The methods followed in the analysis of the remaining parameters are found in Standard Methods (APHA et al. 1992): Total Hardness 2340 B, Total Organic Carbon 5310 C, Total Coliform 9222 B, Fecal Coliform 9222 D, and Fecal Streptococci 9230 C (Mostaghimi and McClellan, 1999). The limits of detection changed from 0.05 mg/L to 0.01 mg/L for TP, FTP, and NO_3 after January 1999. The remaining detection limits were 0.01-mg/L for PO_4 and NH_4; 0.1-mg/L for TN, TKN, and FTKN; 0.02-g/L for TSS; and 10-mg/L for COD (Mostaghimi and McClellan, 1999). In order to estimate potential ammonia toxicity, a subset of samples (four sets from the four monitoring stations; 2 sets representing storm events (9/21/1999 and 10/5/1999) and 2 sets representing baseflow periods (9/8/1999 and 10/19/1999)) was chosen for analysis of hardness (mg CaCO3/L).

The water samples collected for metals analysis were acidified promptly with 0.5-ml trace-metal grade nitric acid for preservation prior to analysis. All metals samples were taken to the

Department of Civil and Environmental Engineering (CEE) at VPI&SU for analysis of copper, lead, cadmium, and zinc. These metal analytes were selected because they are commonly found in runoff from roads and parking lots. Digestions were performed on 100-ml of each sample using 5-ml of trace metal grade nitric acid and following QA/QC procedures for replicates and blanks. Metals were analyzed for all 1997 samples and for one baseflow and one storm event sample set per month collected in 1999, but no analysis was performed on the 1998 samples due to lack of funding and a 6-month limit on sample storage. EPA approved graphite furnace atomic absorption spectrophotometric analysis was used to determine copper (Method 220.2), cadmium (Method 213.2), and lead (Method 239.2) concentrations, while flame atomic absorption spectrophotometric analysis was used to determine zinc (Method 289.1) concentrations (U.S. EPA 1983a). Graphite furnace analysis was performed using a Perkin Elmer 5100PC Atomic Absorption Spectrophotometer including the PE Graphite Furnace HGA 600 using Zeeman 5100 background correction. The flame analysis was performed using a Perkin Elmer PE703 Flame Atomic Absorption Spectrophotometer (Jody Smiley, personal communication, May 5, 2000). The QA/QC procedures for metal analysis included checking known standards and using spikes and duplicates. The nondetect values for copper and lead were 1-ug/L while the limit of detection was 0.05-ug/L for cadmium and 0.01-mg/l for zinc prior to 11/3/1999 (and 0.005-mg/L afterwards) (Jody Smiley, personal communication, January 1999).

3.3.3 Data Analysis

The impacts of the stormwater management facility on hydrology, water quality, habitat, and macroinvertebrate assemblages within the stream channel were investigated and compared to both regulatory standards and other studies of urban best management practices as summarized in the literature review. All spreadsheet manipulation was performed using Microsoft Excel 97 SR-2.

Pollutant loads were calculated by multiplying the flow volume for a given storm by the EMC (flow-weighted sample concentration) of the pollutant calculated for the same storm. In cases where the analyte concentration was below instrument & method detection limits (non-detect: ND), the value of one-half the detection limit was substituted, a common practice in the literature (Cunningham, 1993).

Two pollutant removal efficiency calculations were chosen, (Section 2.3.2), to analyze the results of this study. The medians of the EMC removal efficiencies were used to summarize the concentration pollutant removal efficiencies for each constituent from individual storm events. The median was used because of the large number of censored data points, because it makes no assumptions about the distribution of the data, and because it minimizes the effect of outlier data points (Kantrowitz and Woodham, 1995). The other four methods estimate loading removal efficiency. The MRE method was not selected due to its emphasis on individual storms, while the PR efficiency calculation was not chosen due to its use of statistics within the calculation that could introduce additional sources of error to the equation. The zero-intercept regression method is invalidated by unmonitored sources of flow such as the hypothesized interaction with groundwater, by the baseflow component of the stream flow, and by the additional sources of overland flow (Kantrowitz and Woodham, 1995). The SOL method was thus chosen to estimate the long-term loading efficiencies for pollutant removal during storm events for each constituent sampled at the stormwater management facility.

The data sets used for statistical testing, using Minitab 10.5 Xtra software (Minitab Inc., State College, PA) included the concentrations and loads for each water quality constituent at each sampling location, compiled by storm events and baseflow samples. Additional data sets tested were the pollutant removal efficiencies, calculated by the EMC Efficiency and SOL Efficiency methods, for each constituent sampled during storm events, and the differences (with and without seasonal averaging) between inflow and outflow loads and concentrations for the wet pond, the dry pond, and the overall facility. Similar calculations were made for baseflow using concentrations of discrete samples and average daily flow volumes.

In this study, nonparametric methods were used for describing the data (medians and inter-quartile-ranges), testing the data for independence, and testing measures of location for all data sets. The nonparametric test for independence used was the RVN ratio test ($p < 0.1$) (Chakraborti and Gibbons, 1992). Because its only assumption is that the data are independent, the test of location chosen for use with all data sets that met that assumption was the modified sign test (Putter, 1955; Coakley and Heise, 1996) ($p > 0.95$) shown below:

$$((N^+ - N^-)/(\sqrt{(N^+ + N^-)}) \qquad (3.4)$$

3.4 Biotic Parameters

The biological condition of this study's unnamed Stroubles Creek tributary was impaired prior to construction of the Virginia Tech wet pond detention facility. The impairment was due to multiple sources; including lack of riparian cover along the majority of the stream length; the conduit of the upper reach of the stream channel below campus parking lots and roads; livestock activity along the middle reach of the stream channel; urban runoff from the surrounding watershed; and runoff from agricultural activities along the lower reach of the stream channel (Kibler et al., 1998). The condition of the stream below the ponds was further impacted by sediment-laden runoff due to BMP construction activities that terminated in April 1997 for the dry pond and in August 1997 for the wet pond (Kibler et al., 1998).

3.4.1 Habitat

The high gradient (naturally coarse substrate) stream habitat assessment matrix prescribed by the U.S. EPA RBP protocols was appropriate for use in this study (Kibler et al., 1998). The ten parameters used in the high gradient assessment matrix include epifaunal substrate/available cover; embeddedness; velocity/depth regimes; sediment deposition; channel flow status; channel alteration; frequency of riffles (or bends); bank stability; bank vegetative protection; and riparian vegetative zone width. Assessments were made for the ten habitat metrics above and below each of the wet and dry ponds on Feb. 20, 1997, Apr. 29, 1997, Aug. 29, 1997, and Nov. 11, 1999. Each parameter for a sampled reach was rated on a numerical scale of 0-20, for a total possible score of 200 (Barbour et al., 1999). The 0-20 scale, representing a continuum of conditions, is divided into categories defined as optimal, suboptimal, marginal, or poor conditions based on criteria specific to each parameter (Barbour et al., 1999). The scores for each of the ten parameters (higher scores reflecting better habitat condition) were totaled and compared to a regional reference condition (Barbour et al., 1999). A percent comparability measure was calculated as the ratio between the test station score and the score for the reference condition (Barbour et al., 1999).

3.4.2 Benthic Macroinvertebrates

Macroinvertebrate biomonitoring samples were collected in the winter, spring, and summer seasons of 1997 and the spring, summer, and fall seasons of 1999 in order to assess the impact of the BMP facility on stream health. Winter 1997 samples (February 20, 1997) were collected prior to BMP construction, while the spring 1997 samples were collected during construction (April 29, 1997), and the summer 1997 samples were collected shortly after construction was completed (August 29, 1997). The macroinvertebrate samples were taken from riffle areas above and below each pond (sites QVA and QVB in reference to the wet pond and sites QVC and QVF in reference to the dry pond). In order to sample riffle habitat below the dry pond, the sample was taken approximately 90-m downstream of the dry pond embankment, and thus was labeled separately (QVF) from the embankment culvert (QVD) where the water quality samples were collected.

For each sampling date and site, a standardized sample was collected composed of seven subsamples taken via a D-frame dip net (0.305-m wide) (Barbour et al., 1999; Kibler et al., 1998). Two of the subsamples were 1-m sweeps of the terrestrial grasses and roots exposed to the current on the left and right banks, respectively. The remaining five subsamples were taken from fast and slow current conditions within riffles by collecting all material swept into the D-frame dip net when it was placed on the stream bottom immediately downstream of manual disturbance of the mineral substratum. The subsamples were combined in the field to constitute the sample for the site, then preserved in 95% ethanol, labeled, and returned to the Entomology laboratory on the Virginia Tech campus for further processing. In the laboratory, each sample was washed in a 500-μm soil sieve. Conspicuous invertebrates were removed and saved, with the remaining material evenly spread over a 500-μm rectangular gridded (5 x 5-cm^2 quadrats) sieve submerged in a few centimeters of water. The gridded sieve was removed from the water, and randomly selected squares were sorted until 200 organisms (±10%) were obtained. The material in each selected 5 x 5-cm^2 quadrat was transferred to a white enamel pan, covered with clean water, and thoroughly examined to count and remove all macroinvertebrates. All quadrats that were selected were sorted in their entirety (Kibler et al., 1998). All organisms were examined under a stereomicroscope at 5 to 45X magnification and identified to the family level with reference to taxonomic literature (Merritt and Cummins, 1996).

From each set of raw data (abundance by family), fourteen metrics were calculated. There are four primary categories of metrics: (1) richness measures; (2) composition measures; (3) tolerance measures; and (4) trophic/feeding strategies or habit measures (Barbour et al., 1999).

In this study, the richness metrics calculated from each standardized sample included total number of taxa, the EPT Index (number of taxa in the insect orders Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies)), and the number of Ephemeroptera taxa; the composition metric used was % of Ephemeroptera in the sample; the tolerance measures included the modified Hilsenhoff Biotic Index and the number of taxa intolerant to perturbation; the feeding strategy metrics were % of scrapers, % of shredders, % of collector-gatherers, and % of collector-filterers in the sample; and the habit metric was % of haptobenthos in the sample. Three other metrics used in this study are measures of balance: % 1 Dominant Taxon, % 5 Dominant Taxa and Simpson's Diversity Index. Only the modified Hilsenhoff biotic index and the % dominance metrics are expected to increase with increasing perturbation; the remaining eight metrics are predicted to decrease with increasing perturbation of the environment. Percent Collector-Gatherers and % Collector-Filterers may increase due to eutrophic conditions.

Nine metrics, listed in Table 3.5, were used to calculate the Macroinvertebrate Aggregated Index for Streams (MAIS), developed by Smith and Voshell (1997) for the Central Appalachian Ridges and Valleys ecoregion (among others) within the mid-Atlantic highlands. The MAIS index score was calculated as the sum of the nine metric scores after standardization to a unitless score of 2, 1 or 0 for each metric as specified in Table 3.5 (Barbour et al., 1999).

Table 3.5 Aggregation of metrics into the MAIS score (Smith and Voshell, 1997; Kibler et al., 1998)

Metrics	**Categories**	**Scores**		
		2	**1**	**0**
EPT Index	Richness	≥8	3 – 7	≤ 2
# Ephemeroptera	Richness	≥ 4	1 – 3	0
% Ephemeroptera	Composition	≥ 18	1 – 17	0
% 5 Dominant Taxa	Balance	≤ 79	80 – 99	100
SDI	Balance	≥ 0.83	0.67 – 0.82	≤ 0.66
Modified HBI	Tolerance	≤ 4.21	4.22 – 5.55	≥ 5.56
# Intolerant Taxa	Tolerance	≥ 10	2 – 9	≤ 1
% Scrapers	Trophic	≥ 11	1 – 10	0
% Haptobenthos	Habit	≥ 84	53 – 83	≤ 52

The MAIS index score was then compared to Table 3.6 to assess its implications for overall biological condition of the sampling site. The MAIS index has a maximum total score of 18 and a minimum of zero. The biological condition was judged as very good if the MAIS score was ≥ 17; good if the MAIS score was between 13 and 16; poor if the MAIS score was between 7 and 12; or very poor if the MAIS score was ≤ 6 (Kibler et al., 1998).

Table 3.6 MAIS score implications for biological condition (Kibler et al., 1998)

MAIS Scores	**Biological Condition Categories**	**Biocriteria**
≥ 17	Very Good	Acceptable
13 - 16	Good	Acceptable
7 - 12	Poor	Unacceptable
≤ 6	Very Poor	Unacceptable

Individual metrics were examined for change along the stream profile, and for potential response to the BMP facilities. Two sets of reference conditions were used for comparison to this study. A study in western Virginia by Evans (1997) found the mean values listed in Table 3.7 for metrics assessed within the valley and plateau subregions.

Table 3.7 Metric values for reference conditions in the Valley/Plateau region of Western Virginia (Evans, 1997)

Metric	Valley/Plateau Western Virginia Mean (±95% confidence interval)
No. of families	16.79 (2.01)
EPT index	11.95 (2.23)
% 5 most dominant taxa	61.61 (3.88)
Hydropsychidae / Trichoptera	0.77 (0.09)
Simpson diversity index	0.81 (0.05)
% Collector-gatherers	41.89 (6.18)
% Collector-filterers	32.16 (5.27)
% Scrapers	22.81 (5.91)
% Haptobenthos	62.83 (3.66)

Another study reported the means of metrics measured at minimally impacted sites in the Ridge & Valley ecoregion (listed in Table 3.8) which were also used as reference conditions in the

Kibler et al. (1998) report of data collected in the first year of monitoring this stormwater management facility.

Table 3.8 Metric mean and range from reference sites in the Ridge & Valley ecoregion (Kibler et al., 1998)

Metric	Mean	Range
Taxa Richness	23	18-34
% 5 Dominant Taxa	76	58 – 91
Modified HBI	4.1	3.23 - 5.5
% Haptobenthos	70	52 – 95
EPT Index	13	9 – 18
# Ephemeroptera	4	3 – 6
% Ephemeroptera	22	7 – 42
Simpson Diversity Index	0.83	0.74 - 0.9
# Intolerant Taxa	18	14 – 24
% Scrapers	25	2 – 69

CHAPTER 4.0 RESULTS AND DISCUSSION

The objectives of this study were to: (1) investigate potential influences upon the performance of the stormwater management facility; (2) assess the nonpoint source pollutant removal efficiencies of the wet pond, dry pond, and the overall system; (3) compare results to other urban stormwater control practices reported in the literature; and (4) conduct bioassessment and habitat quality surveys. The watershed and subwatersheds are characterized with respect to areal coverage and land use. The wet pond is characterized with respect to water quality measurements, sediment composition, residence time, and water budget. Rainfall and runoff measurements are summarized for each storm and pollutant concentrations of storm and baseflow samples are presented with respect to water quality standards and criteria. Pollutant loads and removal efficiencies are reported for all measured constituents during storm events and baseflow periods. The pollutant concentrations, loads, and removal efficiencies are compared to the results from other studies of stormwater dry ponds, wet ponds, and wetlands reported in the literature. To conclude the chapter, the results of the benthic and habitat bioassessments are also discussed.

4.1 Watershed Characterization

Information on the study area watershed and the four subwatersheds is given in Figure 4.1. The watershed above the wet pond (QVA watershed) covers an area of 142-ha. The central portion of the QVA watershed surrounds Lane Stadium on the Virginia Tech Campus. In the vicinity of Lane Stadium, soils are primarily Udorthents (45%) while 30% of the soil surface is impervious and other soils comprise the remaining 25% of the area (Anderson and Associates, Inc., 1996). Since installation of the Virginia Tech regional stormwater management facility, approximately 16-ha within the QVA watershed has been converted from pervious to impervious surface. The watershed above station QVB (Figure 4.1) includes the QVA watershed and an additional 25-ha that contributes runoff to the pond at various locations around the pond perimeter. The watershed above station QVC (Figure 4.1) includes an additional 14-ha of area in addition to the QVB watershed, and drainage waters from an off-line dry pond that primarily collects runoff from the area behind the VA-MD Regional Veterinary Medicine facility. The entire watershed of the stormwater management facility covers 238-ha (with sampling station QVD at the outlet

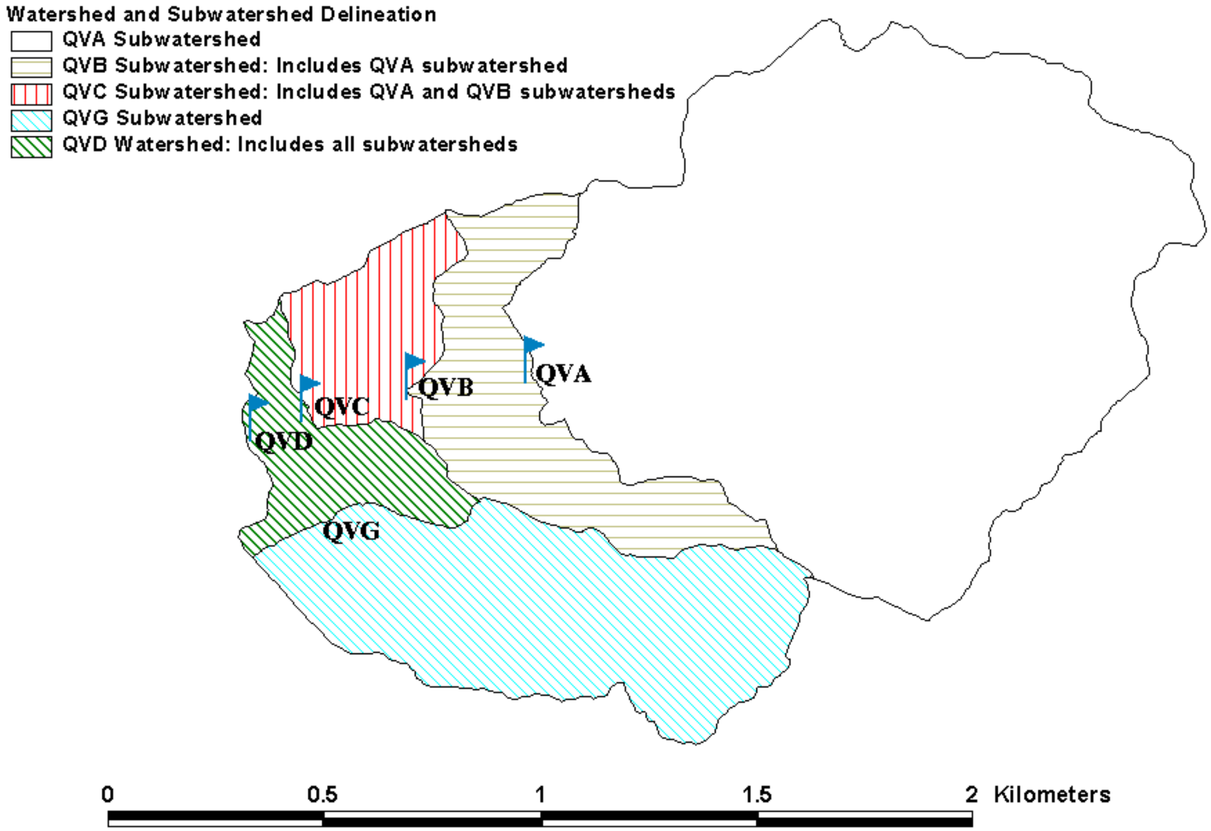


Figure 4.1 Watershed draining to the VPI&SU stormwater management facility; noting subwatershed additions with respect to sampling locations

and including 44-ha of the QVG subwatershed). Agricultural / open / and vacant land is the primary land use in the watershed and may be expected to contribute sediment, nutrients, and bacterial loads to surface runoff. While there is impervious surface within the area designated as agricultural / open / vacant land, most of the impervious surfaces are located within the land areas designated as commercial and residential. The commercial and residential areas are expected to contribute the majority of the runoff and the greatest pollutant load of metals, as compared to other land uses (Hodge and Armstrong, 1993). This watershed has been heavily impacted by human activities as evidenced in Figure 4.2 by the small amount of woodland area remaining. The majority of the residential land is located in the QVA watershed; however some large areas of commercial land use contribute runoff within the QVB and QVC watersheds and thus runoff from these areas could only be treated by the dry pond. The QVB and QVD watersheds also receive runoff from dairy pasture that may inhibit comparisons of upstream-downstream effects of the stormwater management facility with respect to nutrients and fecal coliform.

4.1 Pond Characterization

The wet pond is designed for an extended detention volume of 592-ha-mm, and a permanent pool volume of 592-ha-mm. The water quality volume (assuming the 16.25-ha of impervious surface increase in the QVA watershed according to Table 4.1) is 206-ha-mm. Thus, the permanent pool volume (592-ha-mm) is approximately three times the water quality volume (206-ha-mm) as required by Virginia Stormwater Management regulations (DCR, 1998b). During the majority of 1999, the water level in the wet pond was lowered in order to view damage to the embankment and to prevent frequent overtopping of the embankment. Because of the lowered pool stage, the outlet pipe discharged water from near the wet pond water level. Based on a survey performed of the pond water level in October 1999, the volume of the permanent pool was 455-ha-mm (4547-m^3) while the surface area of the pond was 0.87-ha. At 455-ha-mm, the wet pond was operating at 77% of its capacity for the majority of the period between spring 1999 and summer 2000.

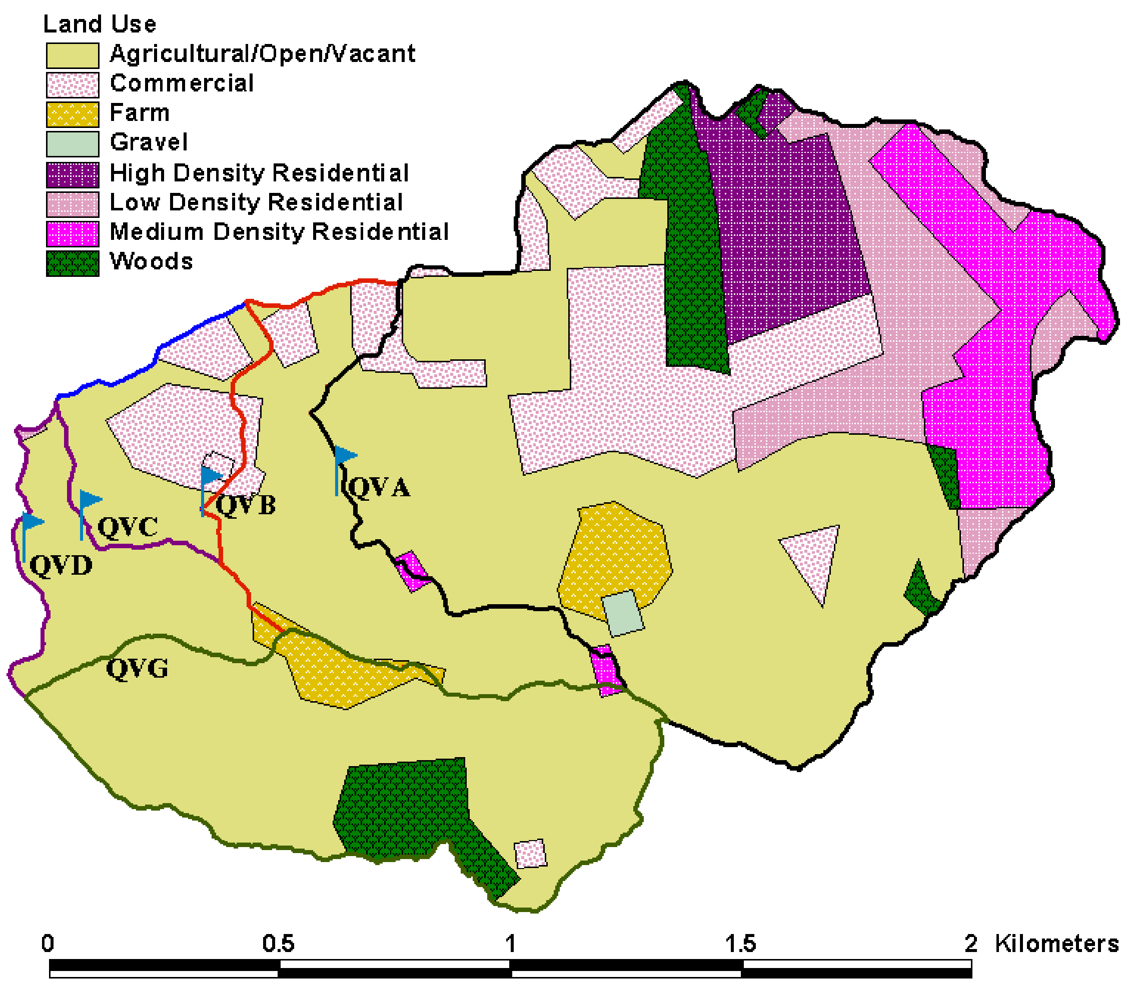


Figure 4.2 Land uses within the watershed of the VPI&SU stormwater management facility

Table 4.1. Subwatershed areas and land use characterization (compiled from information provided by John Fisher and Scott Edelman, Hayes, Seay, Mattern & Mattern, Inc., March 28, 2000).

Land Use	QVA	QVB	QVC	QVG	QVD
Area	142-ha	167-ha	181-ha	44-ha	238-ha
Commercial	22-ha (16%∗)	26-ha (15%)	31-ha (17%)	0.3-ha (1%)	32-ha (13%)
Residential	4- ha (31%)	45-ha (27%)	45-ha (25%)	0-ha (0%)	45-ha (19%)
Agricultural / Open / Vacant	6- ha (44%)	84-ha (50%)	91-ha (51%)	36-ha (80%)	140-ha (59%)
Farm / Gravel	5-ha (3%)	5-ha (3%)	5-ha (3%)	3-ha (6%)	8-ha (3%)
Woods	7-ha (5%)	7-ha (4%)	7-ha (4%)	6-ha (13%)	13-ha (6%)

∗ Land use as % of the subwatershed area

The ratio of detention basin volume (4547-m^3) to median storm runoff volume (1362-m^3 for all storm events monitored) was 3.3 for 1999. This ratio compares well to the desirable ratio of 3-4 although it is less than the 4-6 ratio recommended for maximum removal efficiency by Yousef and Wanielista (1993). If the pond had been operating at design capacity this ratio would have increased to 4.3.

The comparison of pond volume to runoff depth over the watershed has also been used to indicate pond performance. Pond volumes between 9 and 12-mm of runoff for the entire watershed appear to provide optimal performance based on fecal coliform and suspended solids results in Droste et al. (1993), yet the pond survey results indicated it contained approximately 4-mm of runoff from the watershed in June 1999, and only 3-mm in October 1999.

Another measure of pond performance involves the pond surface area. The optimal pond surface area is considered 3% or more of the watershed area that would result in a 4.3-ha pond surface area in this study. As mentioned before, the existing area for the wet pond surface area is 0.87-ha; this is well below the recommended optimal value for maximum pollutant removal efficiency (Yousef and Wanielista, 1993; Hartigan, 1989).

One general rule recommended for optimal performance of the wet pond was not violated. The length of the permanent pool is approximately 224-m. The permanent pool has a L:W ratio of 4.4 not including the narrow section of the pond near the inlet. This ratio meets the minimum recommended values (2 to 4) expected to minimize short-circuiting, enhance sedimentation, and help to prevent vertical stratification (Ellis, 1989; Hartigan, 1989).

4.1.1 Pond Water Measurements

Pond depth, as measured in June 1999, ranged from 0.6-m to 1.1-m (Figure 4.3). Hartigan (1989) suggests that a mean depth of 1 to 3-m is shallow enough to minimize stratification but deep enough to discourage algal blooms and minimize sediment resuspension. The shallow depth of this pond may have resulted from efforts to minimize bedrock excavation during construction, and algal growth within the pond has been substantial throughout the seasons. Nevertheless, as described below, some stratification does occur in the pond though it may not be significant. Wind and currents, storm event flow mixing effects, and water density differences due to temperature or water chemistry can affect stratification.

Unless there is thorough mixing, a temperature gradient with an associated density gradient will develop within the pond (Marsalek, 1997) and affect flow circulation patterns. As expected, the inflow temperature to the pond was lower than the outflow temperature (Figure 4.4) when measured within the pond in June of 1999. In general, the water temperature increases along the longitudinal profile of the pond. The temperature measurements show some initial temperature stratification near the inflow (temperature decreasing with depth), but this was not as striking near the outflow point which could indicate mixing within the pond (Figure 4.4).

The pH measurements, as shown in Figure 4.5, suggest that there may be an increase in pH value along the longitudinal profile of the pond from the inflow to the outflow. This is speculated to be a result of contact with the carbonate bedrock within the pond.

4.1.2 Pond Sediment Composition

Table 4.2 summarizes the results of sediment core sample analyses for the wet pond. The amount of sediment deposition within the pond was estimated using the core sample length that is the portion of the core estimated to be due to sediment deposition, rather than sediment accumulation during pond construction. Accurate deposition depths could not be estimated due to the lack of a clay liner or other indication of sediment accumulated since construction. It appears that significant sedimentation may be occurring in the initial narrow section of the pond

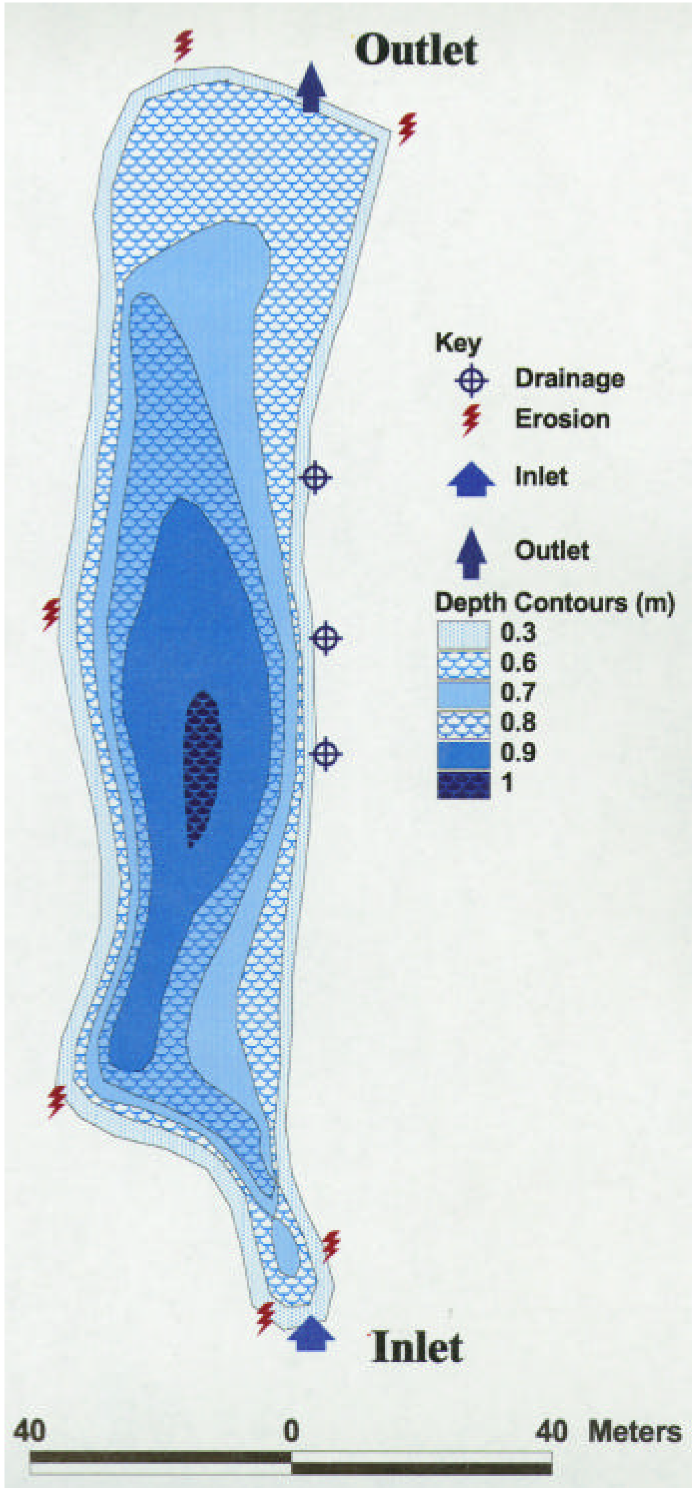


Figure 4.3 Depth contours of the wet pond extrapolated from a June 1999 survey. Note the additional sources of inflow to the pond; both planned drainages and overland flow as evidenced by erosional features.

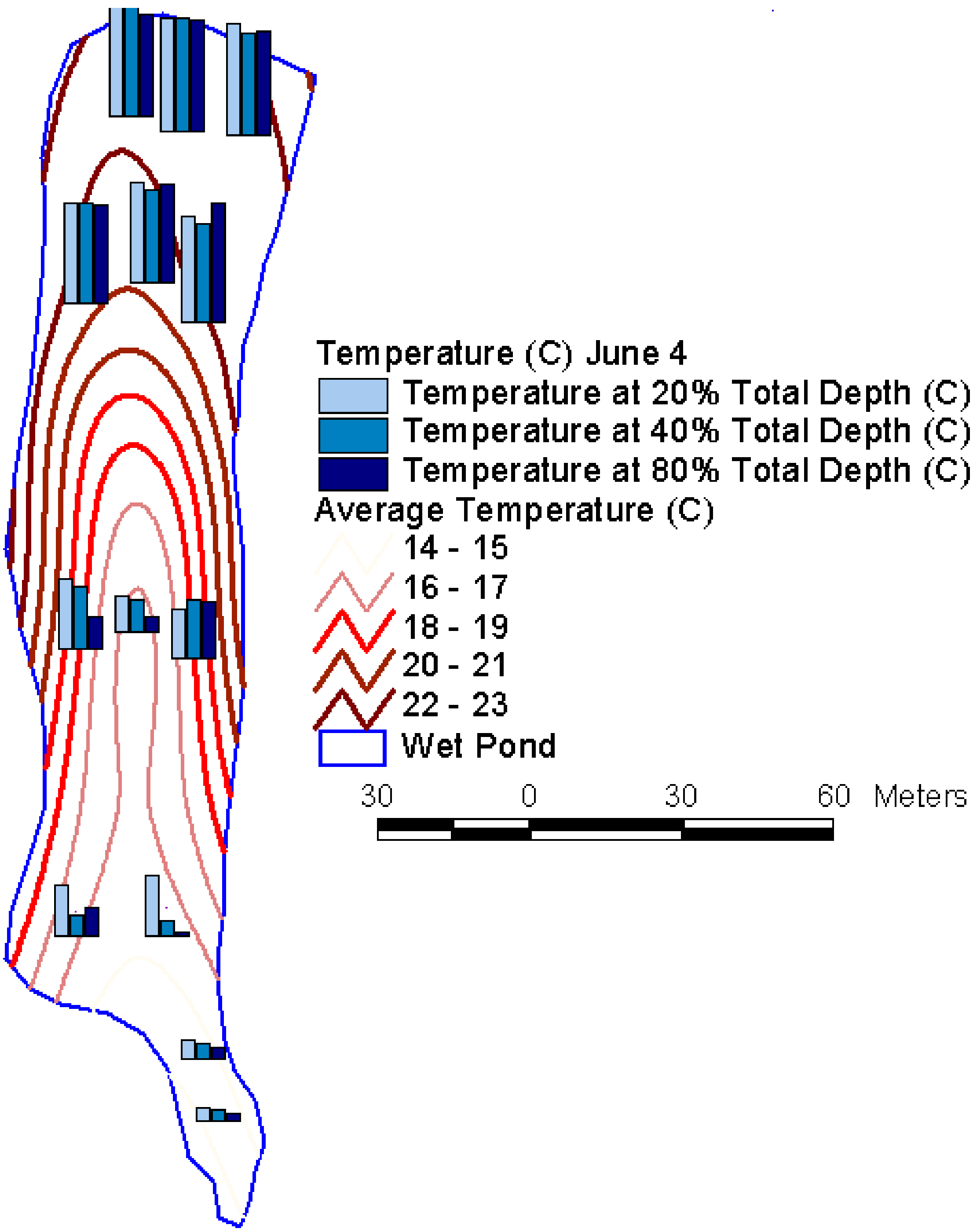


Figure 4.4 Temperature profile and contours of average temperature in the wet pond measured in June 1999.

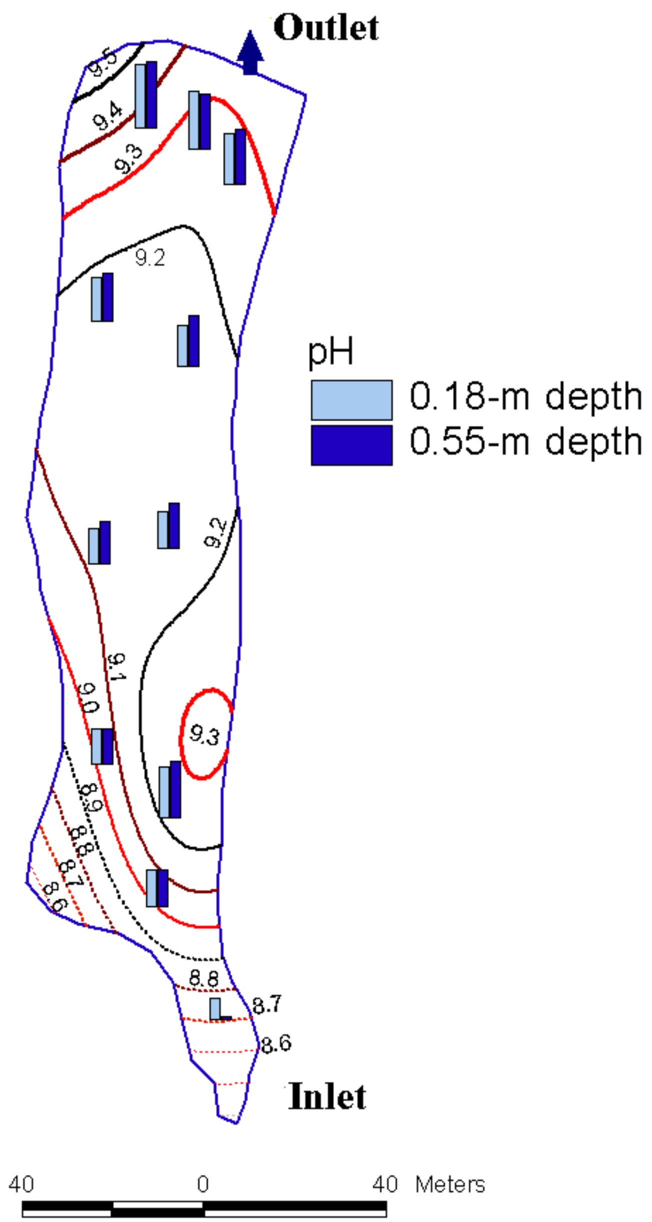


Figure 4.5 pH profile and contours of average pH in the wet pond measured in October 1999

where flow velocity drops substantially. In addition, the long core length collected at the downstream edge of the pond coincides with overland flow entry into the wet pond that has caused substantial erosion of the wet pond embankment. Some evidence that a particle size gradient is forming in the wet pond, with larger particles settling first, is shown by the results presented in Table 4.2. Comparison of the three pond sections as presented in Table 4.2 indicates that the section nearest the inlet (upstream) has the most sand as measured in the sediment cores (31%). Furthermore, the sediment cores from the middle section of the pond have the most silt (60%), and the sediment cores from the downstream pond section have the most clay (24%).

Table 4.2 Sediment core mean lengths, particle sizes, and composition as sampled in September 1999 and summarized by pond sections.

Pond Section	**Core Length (cm)**	**Sand (%)**	**Silt (%)**	**Clay (%)**	**OM (%)**	**pH**	**P (mg/L)**	**Zinc (mg/L)**	**Copper (mg/L)**
Upstream	3.0	31	53	16	2.5	7.5	0.39	0.44	0.03
Middle	4.2	25	60	15	2.5	7.6	0.47	0.14	0.04
Downstream	3.4	24	52	24	2.9	7.4	0.41	0.12	0.03

Sediment composition analyzed in the core samples suggests that organic matter is associated with sediment deposited near the points of inflow (Figure 4.6). Phosphorus and copper show no apparent trends throughout the pond, but zinc concentrations in the sediment are substantially greater in the first third of the pond (0.44-mg/L) compared to the rest of the pond (0.04 and 0.03-mg/L) (Table 4.2).

4.1.1 Residence Time

Dye-response curves (time series of concentrations or mass) are used to determine time-of-travel and dispersion characteristics of on-stream ponds (Shaw, 1995). The time series of dye concentrations corresponds to the distribution of retention times of influent water at the time of dye injection (Matthews et al., 1997). Figure 4.7 shows the time series plot of dye concentrations for the two dye traces performed for this study; the first during a rainfall event (flow 57% greater than baseflow) and the second without rainfall (baseflow conditions). A summary of conditions and statistics associated with the dye tracing experiments is presented in

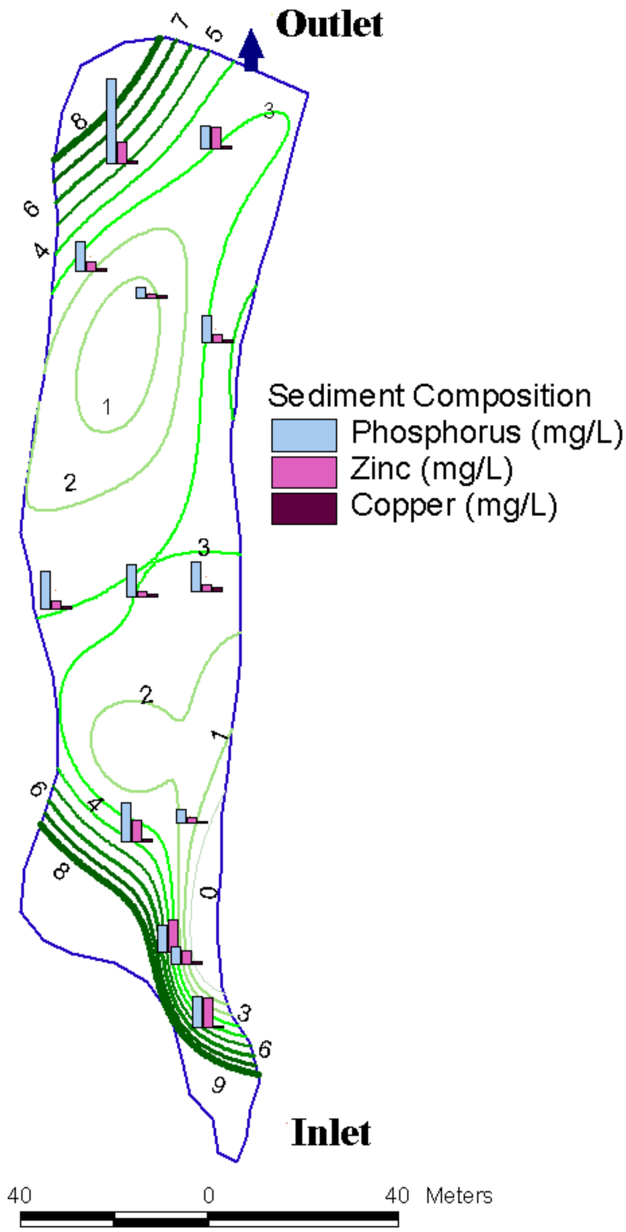


Figure 4.6 Sediment composition with respect to phosphorus, copper, zinc, and organic matter as measured in the cores collected within the wet pond

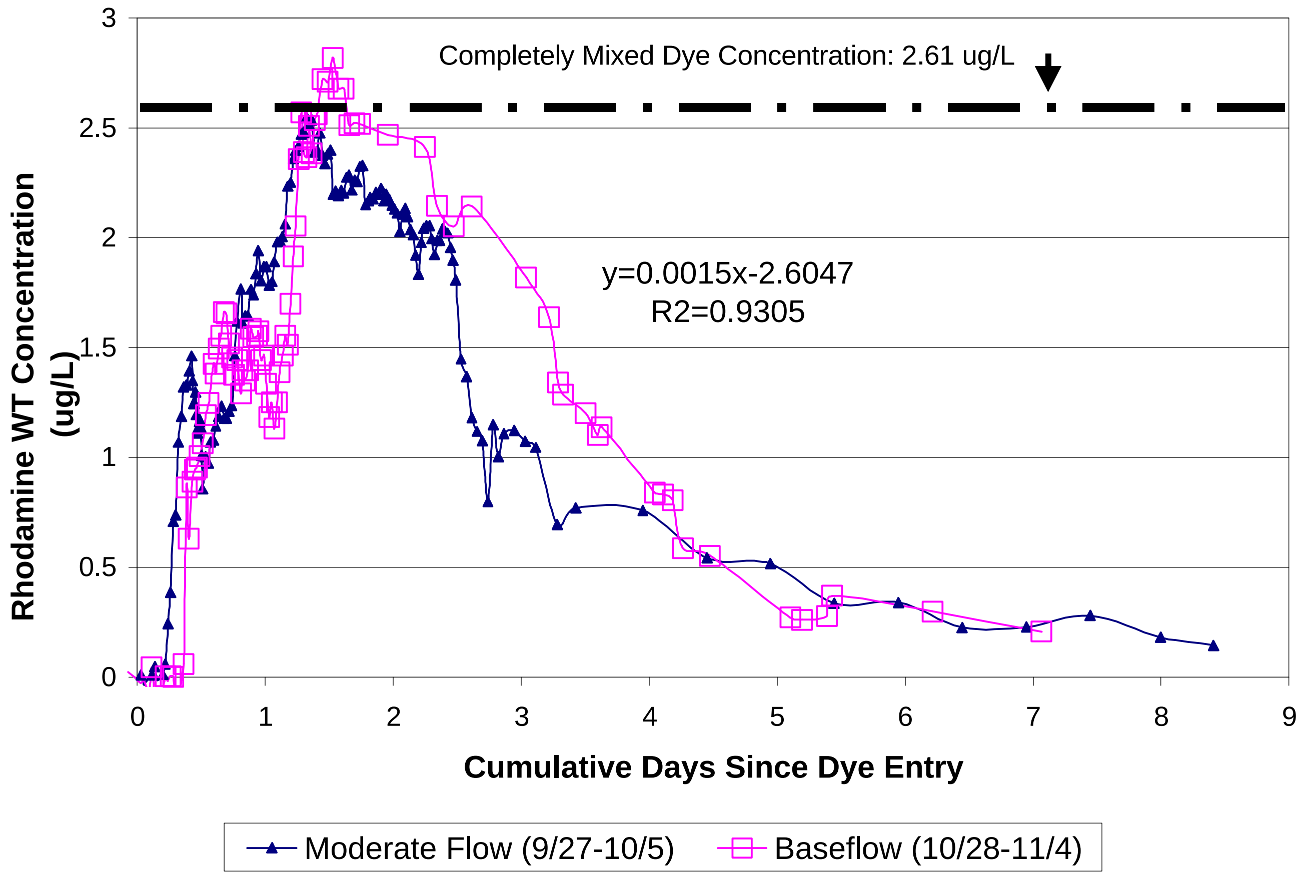


Figure 4.7 Dye concentrations vs. cumulative days since dye entry, for two different flow conditions

Table 4.3. The calibration curve and time series graphs of dye mass are presented in Appendix C.

Table 4.3 Hydraulic performance of the wet pond during two dye trace experiments as calculated from the breakthrough curves of dye concentration and mass

Statistic	**1st Dye Trace 9/27-10/5/1999**	**2nd Dye Trace 10/28-11/4/1999**
Rainfall (mm) *	20.1	0
Flow Volume (L) *	10,376,000	6,599,000
Dye Arrival at Pond Outlet (days)	0.19	0.35
Time to Peak Concentration (days)	1.32	1.53
Mass after 24-hrs (%)	16.5	12.1
Median Retention Time (days) **	2.05	2.09
Volumetric Residence Time (days) ***	2.24	2.42
Hydraulic Efficiency	0.91	0.86

* Amount measured during sample collection (202-hrs for the first dye trace and 169-hrs for the second dye trace)

** The time from initial dye injection at the pond inlet until 50% of the dye mass had been recovered at the pond outlet culvert

*** Calculated as the pond volume divided by the average of the pond inflow and outflow rates, where the inflow and outflow rates were the medians of their respective 10-min. interval flow rates during sample collection

If the flow in the pond could be described as plug flow, then the measured tracer dye concentration at the outlet would be unimodal with limited traces of dispersion and mixing (Anderson et al., 1996). As evident from Figure 4.7, the distribution of the dye concentration over time at the pond outlet shows that there is substantial dispersion of the dye within the pond. Furthermore, under plug flow conditions the time to peak concentration would be the same as the median residence time and this was not the case in this study (Table 4.3).

A bimodal distribution would indicate the existence of short-circuiting in the pond (a first peak due to the fast direct flow and a second (often smaller) peak as a result of the dye's incorporation into the mixed flow zone) (Anderson et al., 1996). Both flow conditions show a small initial peak within the first day of measurements (Figure 4.7) that may be evidence of some short-circuiting. Another indicator of short-circuiting within the pond is exceedance of the concentration of dye assuming complete mixing had occurred (Martin, 1989). In this case, the breakthrough curve for the baseflow condition exceeds the completely mixed dye concentration only slightly (Figure 4.7). The degree of short-circuiting, as calculated by the portion of flow that exits the wet pond within 0.3-0.4 of the volumetric residence time, was 10.7% to 14.4% for the first dye tracing experiment (moderate flow), and 10.4% to 16.5% for the second (baseflow). The time of first dye arrival at the pond outlet also indicates the degree of short-circuiting. While the higher flow during the first dye tracing experiment did not affect the median residence time substantially as compared to the second dye tracing experiment performed during baseflow conditions (a decrease of 0.04 days), it did move the dye faster through the pond by advection currents. This was evidenced by the reduction in time needed to observe the first dye arrival at the pond outlet (from 8.4 hours to 4.6 hours). The majority of the dye in each case remained in the pond after the first 24-hours (84% in the first dye tracing experiment and 88% in the second).

The volumetric residence time, defined as basin volume divided by flow rate through the basin, is only applicable to basins with plug or completely mixed flow conditions (Matthews et al., 1997). These flow conditions are idealized and result in longer residence times than the moderately-mixed flow condition typical in field conditions (Martin, 1989). The moderately-mixed flow basin exhibits deviations from either flow model as a result of mixing characteristics within the pond as affected by dye dispersion; short-circuiting; wind influences; shear stresses along the pond boundaries; inlet and outlet effects; circulation patterns (recycling); stratification; and stagnant zones within the ponds (Allan et al., 1997; Martin, 1989; Shaw, 1995).

As this basin appears to be a moderately mixed flow basin, estimation of the volumetric residence time is not applicable but is useful for comparison because this technique is often applied in pond designs. Hydraulic efficiency represents the deviation of the basin from ideal conditions (Matthews et al., 1997) and is defined as the ratio of measured to volumetric retention time (Matthews et al., 1997). The hydraulic efficiencies for the two dye tracing experiments are reported in Table 4.3. While the volumetric residence times were higher than the calculated median retention time as expected, the hydraulic efficiency ratios were quite high: 0.91 for the first dye tracing experiment and 0.86 for the second dye tracing experiment. The shallow nature of the wet pond may limit the degree of dead zones and stratification that could affect mixing within the pond; as a result, it may approximate the completely mixed flow condition.

Consideration must be made, however, that the pond is not operating under design conditions due to the water level drawdown for purposes of embankment investigation and repair. Seasonal changes in stormwater volume, inflow rates, stratification (thermal gradients controlling vertical mixing), and groundwater interactions (Gain, 1996) will need to be considered in a full analysis of wet pond performance with respect to detention time.

The optimal hydraulic residence time for wet detention basins will vary according to the design criteria. Forty hours of detention within wet ponds has been reported as sufficient for pollutant removal via sedimentation, but two weeks (Hartigan, 1989) of retention or a ratio of permanent pool storage to mean storm runoff greater than or equal to four (Van Buren, 1994), is recommended to maximize biological uptake within the pond (Schueler, 1987). Site-specific factors are important to residence time determination. One compromise recommended by Hvitved-Jacobsen et al., 1989) was for a minimum detention time of 72 hours. This detention time was considered sufficient to remove up to 95% of TSS and 30-70% of nutrients and heavy metals (Hvitved-Jacobsen et al., 1989). The median detention time measured in this study's wet pond was approximately 48 hours which allows for sedimentation to remove pollutants, but falls far short of 72 hours and far short of two weeks.

4.1.1 Water Budget

A water balance was constructed for the wet pond for the period between May 11 and October 31 of 1999 (Table 4.4). The results show excess input to the pond (direct rainfall on the pond surface and inflow from the QVA watershed) than output in the form of outflow from the pond to the stream channel and evaporation from the pond surface. Errors inherent to this type of analysis include instrumentation errors; flow calculation errors; assumption of the relationship between pan evaporation and pond evaporation; and the geographic location of the pond and weather station and associated topographic influences on rainfall and evaporation measurements. Furthermore, groundwater recharge could have a marked impact on the water balance in the pond.

Table 4.4 Water budget for the wet pond from May 11 through October 31, 1999

Water Balance	**Rainfall (P) (m^3/month)**	**Inflow (I) (m^3/month)**	**Outflow (O) (m^3/month)**	**Evaporation (E) (m^3/month)**	**Change in Storage I+P-O-E (m^3/month)**
May 11-31	494	56106	38354	628	17619
June	291	79023	53696	844	24775
July	944	85381	72610	908	12807
August	685	79463	69865	1122	9161
September	973	80803	59033	319	22424
October	311	70602	56347	374	14192

One possible explanation for the greater measured input than measured output is that the pond is recharging the underlying groundwater. The terrain surrounding the pond has karst formations that can be accelerated by construction activities including site grading, building construction, and water impoundment (Wolter, 1996). The dye trace study performed by the Department of Conservation and Recreation (Terri Brown, Dublin, VA DCR office, Sept. 17, 1999) showed that the pond water and the groundwater do interact, though the extent of the exchange is unknown (Terri Brown, personal communication, October 1999). Although the water balance in Table 4.4 shows a possible recharge of the groundwater by the pond, the original concern that prompted investigation into the pond by Virginia Tech and DCR personnel was the slow drawdown of the water in the pond after storm events. The water table is high in the region of the wet pond as evidenced by an area of wetland located between the wet pond and the dry pond. Interaction of the pond with the water table could explain why the pond level remained high for long periods after storm events (groundwater recharge), but appears to discharge to the groundwater between storm events.

4.2 Rainfall and Storm Event Characterization

Baseflow conditions were monitored by taking periodic grab samples at the monitoring stations. A total of 35 grab samples were collected during baseflow periods from each sampling station during the course of this study between 2/6/1997 and 10/19/1999; while 22 additional grab samples were collected during storm events between 2/11/1198 and 10/5/1999. Two each of the baseflow (2/6/1997 and 2/25/1997) and stormflow (3/3/1997 and 3/25/1997) samples were used to estimate baseline conditions prior to BMP installation. Flow-weighted water quality samples were collected by automatic samplers from 33 storm events in 1999 between February 1 and October 19. Seven samples were collected at site QVG for estimating the influence of the 44-ha subwatershed upon water quality downstream of the stormwater management facility (Appendix E). Rainfall and runoff information for the individual storms is reported in Table 4.5.

The rainfall-runoff relationship at each sampling location, presented in Figure 4.8, shows that the watersheds of QVA and QVD had similar hydrologic responses. Stations QVB and QVC also had similar hydrologic responses but different from QVA and QVD; perhaps due to the influence of groundwater interactions within the wet pond. Rainfall and inflow and outflow data for each storm are presented in Table 4.5. In general station QVB shows depressed flow as compared to QVA, while station QVC demonstrates the same pattern with respect to QVD, similar to the patterns observed in Figure 4.8.

None of the storms monitored during the course of this study attained the 76.2-mm design rainfall depth calculated by Hayes, Seay, Mattern & Mattern, Inc. to characterize the 2-year storm event for the Stroubles Creek watershed (Hayes, Seay, Mattern & Mattern, Inc., 1995). The storm on September 6, 1999 (Table 4.5), the largest storm monitored, produced 61.2-mm (2.4-in.) of rainfall. Furthermore, it should be noted that the thirty-year normal precipitation for the Blacksburg region is 104-cm/yr. In comparison, the 1997 and 1999 annual rainfall amounts of 86-cm and 85-cm respectively were low while the rainfall for 1998 was greater than normal at 118-cm. (Michael E. Gillen, personal communication, Jan. 24, 2000). Characterization of the stormwater management facility would benefit from data collection over multiple years during which time the stormwater management facility would be operational and stabilized with respect to performance.

4.3 BMP Impacts on Pollutants

4.3.1 Pollutant Concentrations

Pollutant concentrations were obtained through analyses of baseflow grab samples, storm event grab samples, and automated storm event mean concentration (EMC) samples. The first two relationships investigated were to determine the difference between baseline (pre-BMP) and subsequent (post-BMP) samples and to determine the degree of potential influence from

Table 4.5 Rainfall and flow associated with monitored storms, including total rainfall, duration of rainfall, total flow, baseflow, and peak flow rate at each monitoring station

Storm Date	*Rainfall (mm)*	*Duration (hr)*	*Rain Prior 48-hrs (mm)*	*Total Flow (watershed-mm)*				*Baseflow (watershed-mm)*				*Peak Flow Rate (m^3/s)*			
				QVA	QVB	QVC	QVD	QVA	QVB	QVC	QVD	QVA	QVB	QVC	QVD
2/2/99	16.3	15.7		3.5	1.6	2.3	3.0	2.0	1.1	1.1	1.5	0.09	0.03	0.08	0.11
2/8/99	5.3	9.2		0.8	0.6	1.0	1.4	0.4	0.5	0.7	1.0	0.23	0.04	0.08	0.11
2/12/99	2.3	2.5		0.4			0.9	0.3			0.7	0.05			0.05
2/18/99	22.1	23.5		3.3	1.9	4.2	3.9	0.9	1.1	1.5	1.3	0.22	0.08	0.30	0.27
2/28/99	7.4	21.3		1.3	0.9	1.5		0.7	0.8	1.1		0.08	0.03	0.05	
3/4/99	10.4	28.5		4.6	5.6	5.9	7.4	2.8	2.7	3.1	3.3	0.54	0.13	0.37	0.31
3/16/99	26.7	27.5		7.5	2.8	5.1	5.8	3.6	2.2	2.7	2.1	0.24	0.05	0.17	0.20
3/22/99	16.0	12.3		2.8	1.4	1.3	3.3	1.1	0.7	0.7	1.6	0.32	0.11	0.15	0.40
4/1/99	13.0	32.2		1.7	2.1	1.2	3.4	0.6	1.2	0.8	2.3	0.22	0.05	0.03	0.08
4/12/99	23.4	11.2	1.8	2.9	3.3	2.3	4.4	0.5	1.4	1.5	1.5	0.31	0.08	0.05	0.18
4/15/99	11.2	31.5		1.2	1.2	1.4	2.9	0.4	0.8	1.2	2.0	0.14	0.03	0.03	0.06
4/27/99	7.1	10.2		0.8			0.9	0.5			0.7	0.07			0.05
4/29/99	11.4	33.5	7.1	1.2			2.1	0.4			1.4	0.15			0.07
5/8/99	12.5	8.8		1.6	1.7	2.2		0.5	1.0	1.8		0.32	0.04	0.03	
5/14/99	19.8	19.7	4.6	3.5	4.3	3.9	5.9	1.4	1.6	2.3	2.3	0.33	0.17	0.16	0.34
5/19/99	27.2	7.8		3.1	3.3	2.9	4.1	0.5	1.2	1.8	1.5	0.97	0.08	0.06	0.16
6/17/99	13.7	25.7		1.7			1.7	0.8			1.1	0.14			0.05
6/21/99	7.1	8.0	2.3	0.6	1.4		1.0	0.3	1.0		0.6	0.14	0.04	0.04	
6/28/99	4.1	5.2	0.5	0.5			0.7	0.1			0.4	0.28			0.03
7/8/99	4.8	0.2		0.3	0.6			0.1	0.5			0.08	0.03		
7/11/99	13.0	18.8		1.2	2.1	2.1	1.8	0.4	1.5	1.7	1.0	0.26	0.04	0.04	0.06
7/12/99	40.9	30.8	13.0	5.7	6.1	5.7	7.4	0.8	1.9	2.3	2.0	0.70	0.19	0.21	0.35
7/29/99	29.0	24.0	0.3	4.0	5.2	4.7	5.2	0.8	1.8	2.5	1.6	2.01	0.13	0.10	0.16
8/2/99	9.4	0.5		1.1	2.3		1.3	0.1	1.1		0.8	1.64	0.13		0.06
8/14/99	8.1	12.0		0.7	1.8	2.1	1.6	0.1	1.2	1.7	0.9	1.09	0.04	0.04	0.05
8/20/99	10.4	2.3		1.2	1.6	2.9	3.3	0.3	1.1	2.4	2.2	1.08	0.04	0.05	0.06
8/24/99	46.5	60.5		12.7	9.2	9.5	10.1	3.6	3.7	5.0	3.6	7.05	0.13	0.14	0.22
9/6/99	61.2	39.2		10.4	8.6	12.7	12.5	2.0	2.2	3.6	2.8	1.44	0.51	0.91	0.68
9/21/99	26.9	7.2		3.7	3.2	3.6	5.0	0.5	1.0	1.5	1.9	1.22	0.10	0.13	0.18
9/28/99	11.9	26.0		2.2	1.8		3.0	1.5	1.2		2.2	0.15	0.03		0.06
9/30/99	7.9	17.0	11.9	0.8	0.5	1.0	0.9	0.2	0.3	0.8	0.5	0.91	0.03	0.05	0.07
10/5/99	7.9	16.3	0.3	0.6	1.3	2.2	1.8	0.2	1.0	1.9	1.3	0.32	0.03	0.05	0.06
10/9/99	15.8	33.7		2.2			4.6	0.9			2.4	0.11			0.09
10/19/99	10.4	39.5		1.1	1.0		3.4	0.6	0.7		2.5	0.12	0.03		0.03

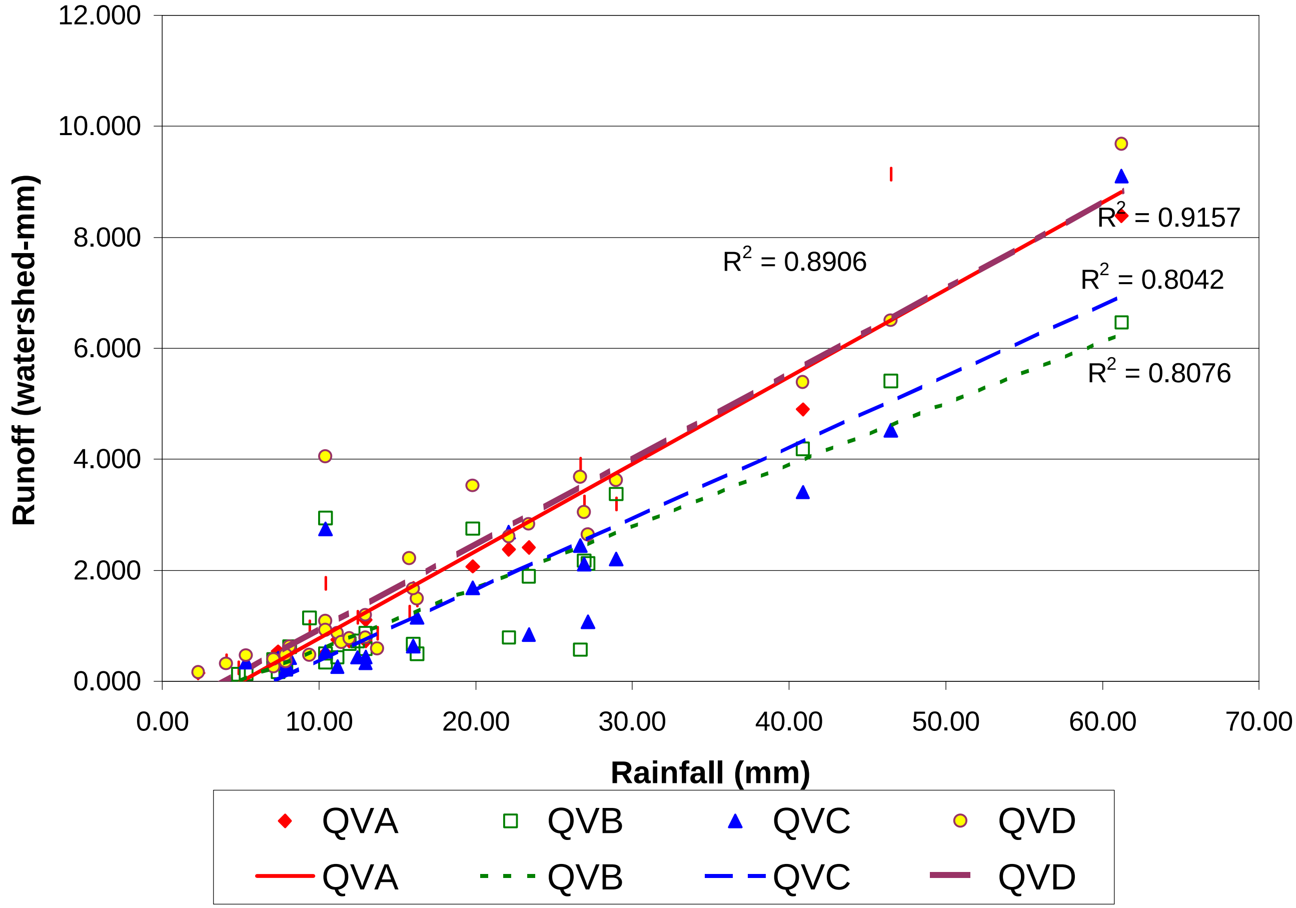


Figure 4.8 Rainfall - runoff relationships for each subwatershed

subwatershed QVG upon the results obtained at site QVD. Baseline conditions were represented by the first two storm samples and the first two baseflow samples collected from the study site, which were during the winter season. The only relevant items noted were that nitrate and temperature were higher at all stations for storm and baseflow baseline events, but especially high at sites QVB, QVC, and QVD. Therefore, it appears that installation of the wet pond decreased the nitrate concentrations released downstream, and decreased the temperature of the released water during the winter season. There were no other relevant trends noticed in the data with respect to how the wet or dry ponds influenced conditions as compared to baseline results. The results from the seven samples collected at QVG over the study period show that QVG may indeed have had an impact upon results seen at station QVD. The analytes for which station QVG had higher median concentrations as compared to QVD are as follows: TSS (27-mg/L vs. 14-mg/L), TOC (4.92-mg/L vs. 1.99-mg/L), COD (25-mg/L vs. 15 mg/L), Total P (1.25-mg/L vs. 0.10-mg/L), orthophosphorus (0.63-mg/L vs. 0.01-mg/L), filtered total P (0.99-mg/L vs. 0.03-mg/L), nitrate (2.26-mg/L vs. 1.33-mg/L), total N (3.75-mg/L vs. 2.07-mg/L), fecal coliforms (6000-col/100ml vs. 130-col/100ml), and fecal streptococci (700-col/100ml vs. 200-col/100ml). These results show that pollutant reductions investigated for the dry pond and for the entire facility are significantly affected by inflow from subwatershed QVG.

Figure 4.9 shows the annual median total suspended solids (TSS) concentrations measured at all sampling locations. In 1997 and 1998, no marked changes among the sampling stations were noted. Samples taken in 1999 tell a different story with regard to the effectiveness of the wet pond in removing TSS. In 1999, median TSS concentrations measured at all sampling stations were greater than previous years, but dramatically higher at station QVC compared to all other stations. The fact that TSS concentrations at QVC are greater than those measured at QVA and QVB indicates that the sediment most likely originated from the progressive failure of the wet pond embankment. It should also be noted that at QVC, sediment accumulated more rapidly near its sampler intake due to the location of a channel bend immediately upstream. The bend caused sediment deposition due to decreased water velocities near the sampler intake, which likely caused greater TSS concentrations in the samples. The TSS concentrations at QVD were much lower, primarily due to the deposition of TSS that occurred as runoff water traveled from site QVC to site QVD. Sedimentation is affected by pond size (depth and volume) relative

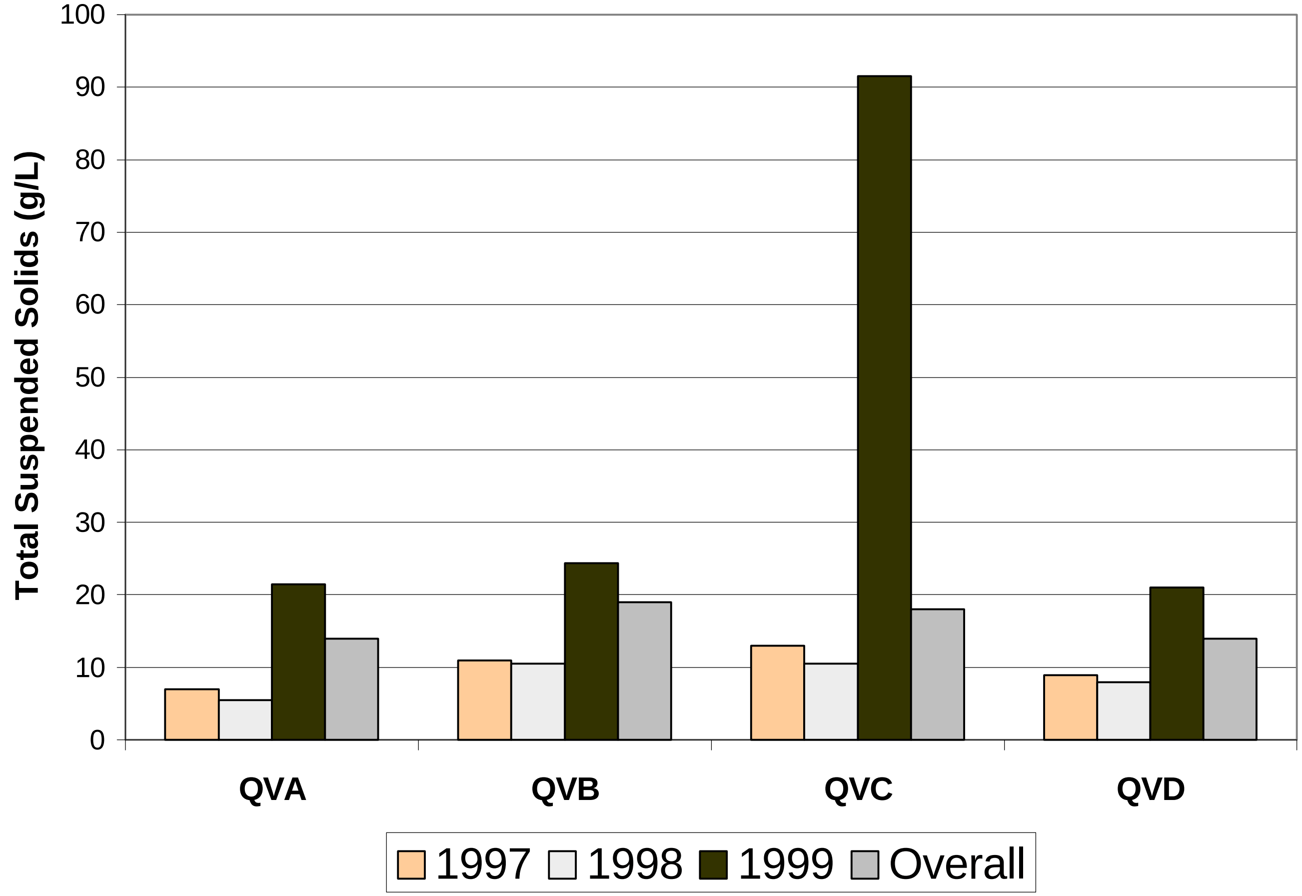


Figure 4.9 Annual medians of combined storm and baseflow concentrations of total suspended solids at all four sampling locations

to runoff volumes and watershed area, frequency of runoff events, flow conditions in the pond (overflow rate, residence time, short-circuiting, and turbulence) and physio-chemical properties of the local runoff and particulates (Driscoll, 1989; Van Buren, 1994). Evidence that sedimentation occurred in the wet pond was supported by a decrease in 1998 metals concentrations at station QVB, as compared to QVA for copper and zinc (Figure 4.10 and 4.11). Metals have a strong affinity with solids (Felstul and Montgomery, 1991) and thus are deposited in the pond attached to the sediment particles. One study reported by Felstul and Montgomery (1991) showed that metals were highly bound to sediment: Pb (65%), Cd (20-70%) of Cd, and Cu (30-80%). Van Buren (1994) reports that sedimentation can account for substantial removal of TP (50%), lead (65-85%), zinc (30-45%), and copper (40%). However, the results of the modified sign test show that of all the metals investigated in this study, only copper concentrations significantly decreased downstream of the wet pond. QVD was significantly lower than QVA over the entire study period, and concentrations measured at QVB were significantly lower than those at QVA were in a seasonal analysis (prior to statistical analysis, the data were summarized by the median of each season). The annual median metals concentrations at all sampling sites are presented in Figures 4.10 to 4.13. Exceedances of acute and chronic freshwater quality criteria are discussed in Section 4.6.2 with regard to macroinvertebrates. Several trends can be noticed from the results displayed in these figures. First, cadmium concentrations in 1997 appear to increase downstream, perhaps due to tire wear during construction activities in the wet pond and dry pond (Figure 4.12). Second, both copper and zinc concentrations in 1998 are reduced downstream of the wet pond (stations QVB, QVC, and QVD) as compared to concentrations in pond inflow (station QVA) (Figures 4.10 and 4.11). Only copper concentrations showed a statistically significant reduction by the modified sign test that was used to compare inflow and outflow for the dry pond and the entire system. The trend disappears in 1999 perhaps masked by the failure of the wet pond embankment. The median total organic carbon (TOC) concentrations (Figure 4.14) downstream of the wet pond generally increased. Station QVA always had lower TOC than station QVB. In addition, TOC has an increasing trend from upstream to downstream sites both in 1998 and for the entire sampling period (overall). These results may be due to organic matter contributions to the stream from seston in the wet pond and from vegetation along the streambanks below the wet pond.

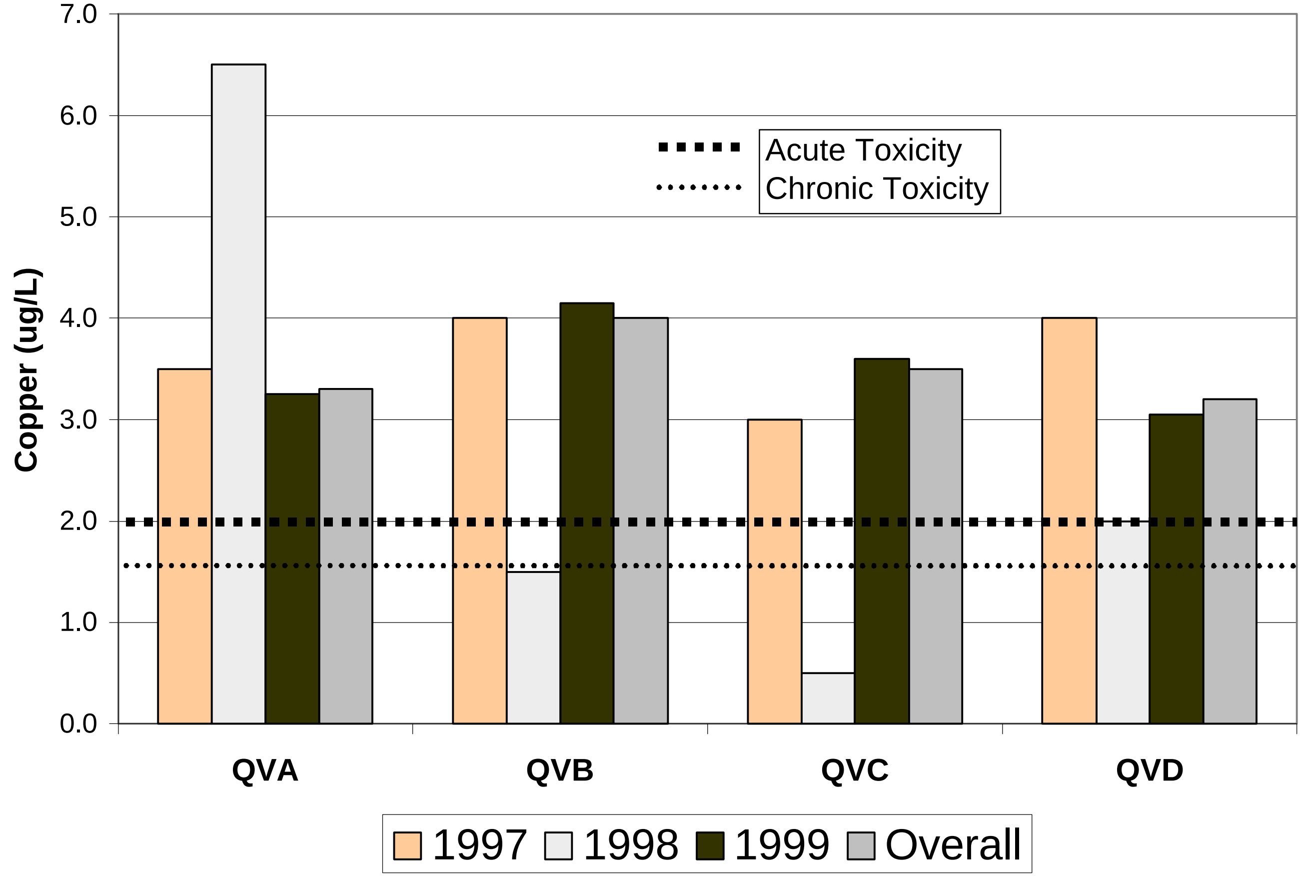


Figure 4.10 Annual medians of combined storm and baseflow concentrations for copper at all four sampling locations, compared against its acute and chronic water quality toxicity criteria (DEQ-WQS, 1997)

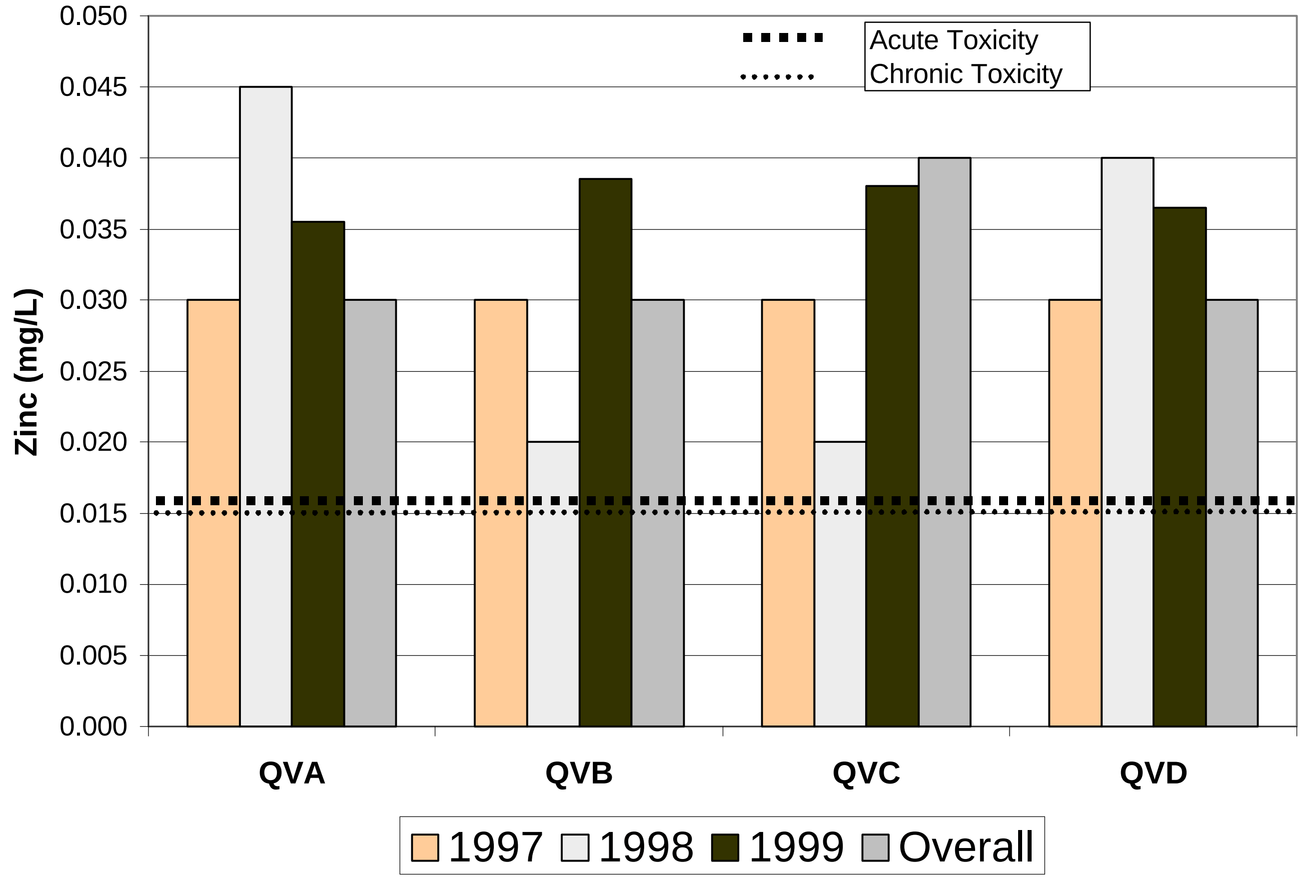


Figure 4.11 Annual medians of combined storm and baseflow concentrations for zinc at all four sampling locations

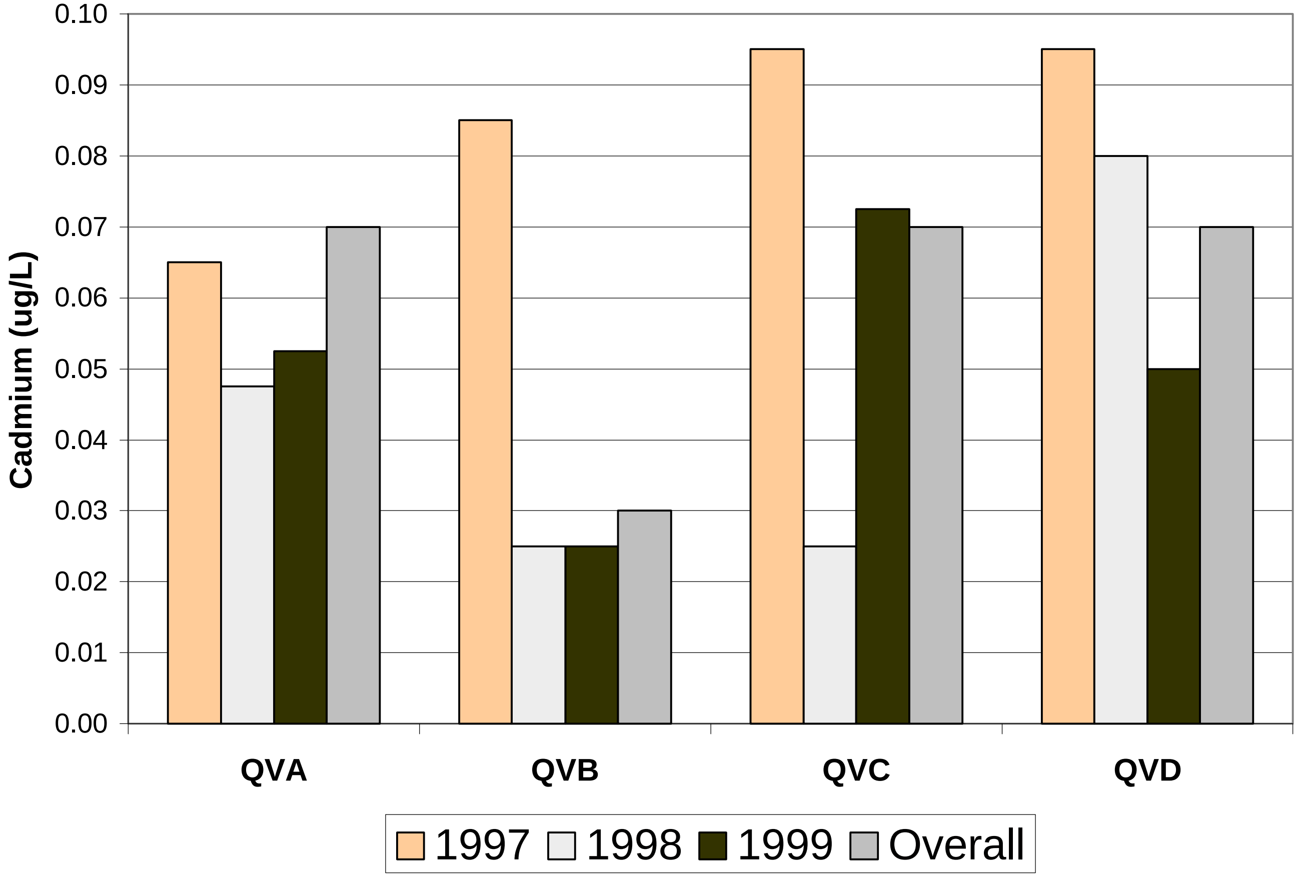


Figure 4.12 Annual medians of combined storm and baseflow concentrations for cadmium at all four sampling locations

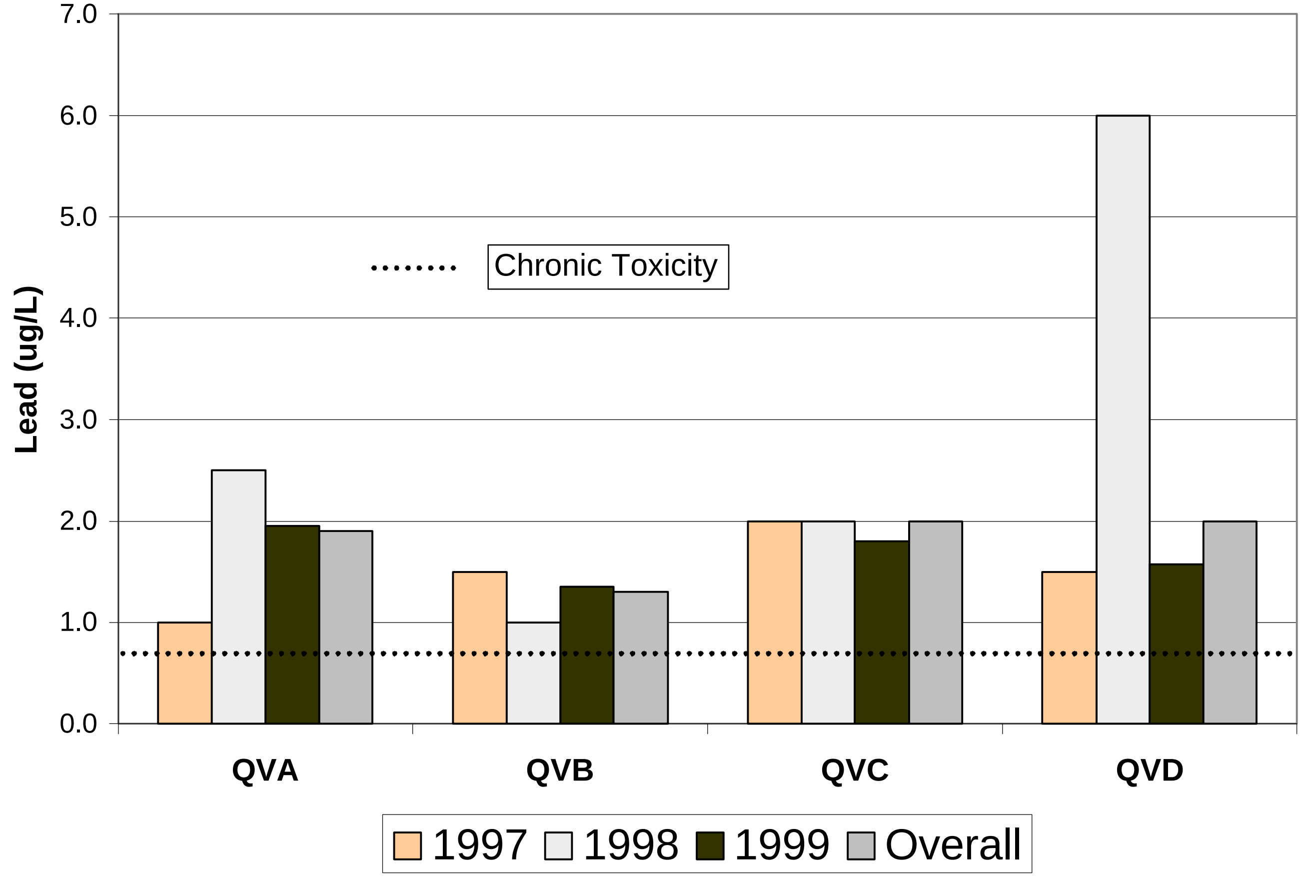


Figure 4.13 Annual medians of combined storm and baseflow concentrations for lead at all four sampling locations, compared against its chronic water quality toxicity criteria (DEQ-WQS, 1997)

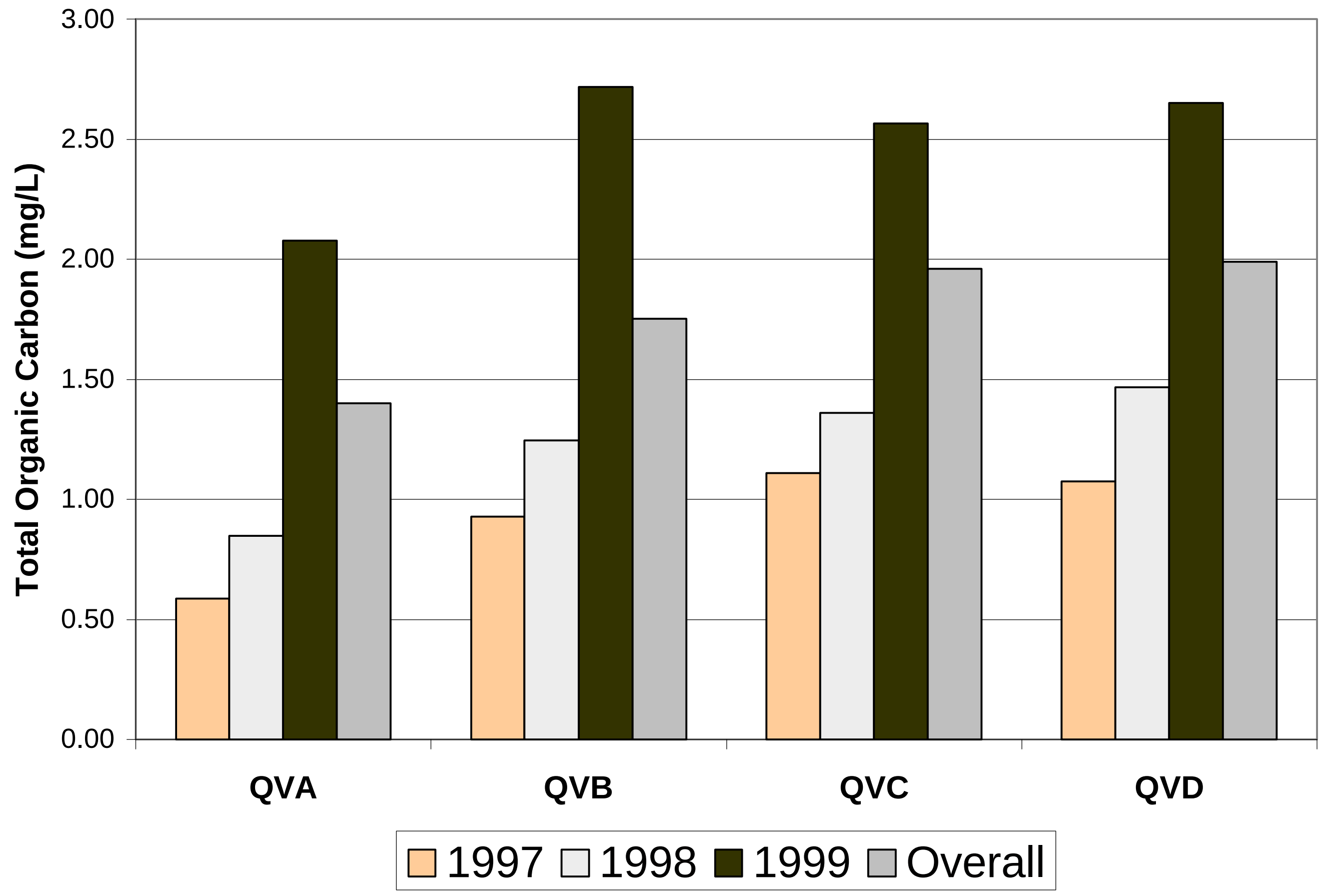


Figure 4.14 Annual medians of combined storm and baseflow concentrations for total organic carbon at all four sampling locations

Soluble nitrogen constituents may be removed from the pond by biological processes, may react among themselves, or may remain soluble and exit a pond or wetland with little reduction (Wotzka and Oberts, 1988). The annual and overall median nitrate concentrations for the sampling stations are displayed in Figure 4.15. This figure shows that median nitrate concentrations at QVA were greater than at the downstream sampling stations in 1998, 1999, and for the entire sampling period (overall). The concentration level (0.1-mg/L) that promotes algae growth is consistently exceeded at all stations, which may lead to degraded macroinvertebrate habitat quality within the stream. The maximum containment level for nitrate (10-mg/L) was not exceeded and only once was the chronic ammonia freshwater criteria exceeded at station QVA. The statistical analyses of differences between pollutant concentrations above and below the wet pond, as tested by the modified sign test, indicated significant nitrate removal from the wet pond. One possible explanation is the transformation of the dissolved constituents within the stream and pond to other forms of nitrogen such as TKN or inorganic N. This is supported by the evidence of decreased nitrate concentrations (Figure 4.16) and increased TKN concentrations (Figure 4.17) at all stations during the summer seasons. The statistical analyses also show that total nitrogen removal by the wet pond is significant and that removal of ammonia and TKN was significant in the dry pond. Figure 4.18 shows that total nitrogen decreases at stations QVB and QVC compared to QVA in 1998, 1999, and overall; but increases from QVC to QVD for all reported medians. This increase may be a result of nitrogen inputs to the dry pond from fertilizer, wetland vegetation degradation, or nitrogen inputs from cattle or wildlife fecal material. The statistically significant removal efficiencies of nitrate and total nitrogen within the wet pond are supported by evidence that the concentrations entering the basins are too low to be irreducible. With respect to potentially irreducible concentrations as discussed by Schueler (1996), total nitrogen concentrations at QVA (median 2.4-mg/L, IQR 1.03) and QVD (median 2.07-mg/L, IQR 1.14) exceed the irreducible limit given by Schueler as 1.9-mg/L, whereas QVB (median 1.88-mg/L, IQR 1.15) and QVC (median 1.74-mg/L, IQR 1.24) fall below this value. Nitrate concentrations are above the irreducible limit given as 0.7-mg/L for all stations (medians: QVA (1.69-mg/L), QVB (1.09-mg/L), QVC (1.08-mg/L), and QVD (1.33-mg/L).

The water temperature measured at sampling stations never exceeded the water quality criterion

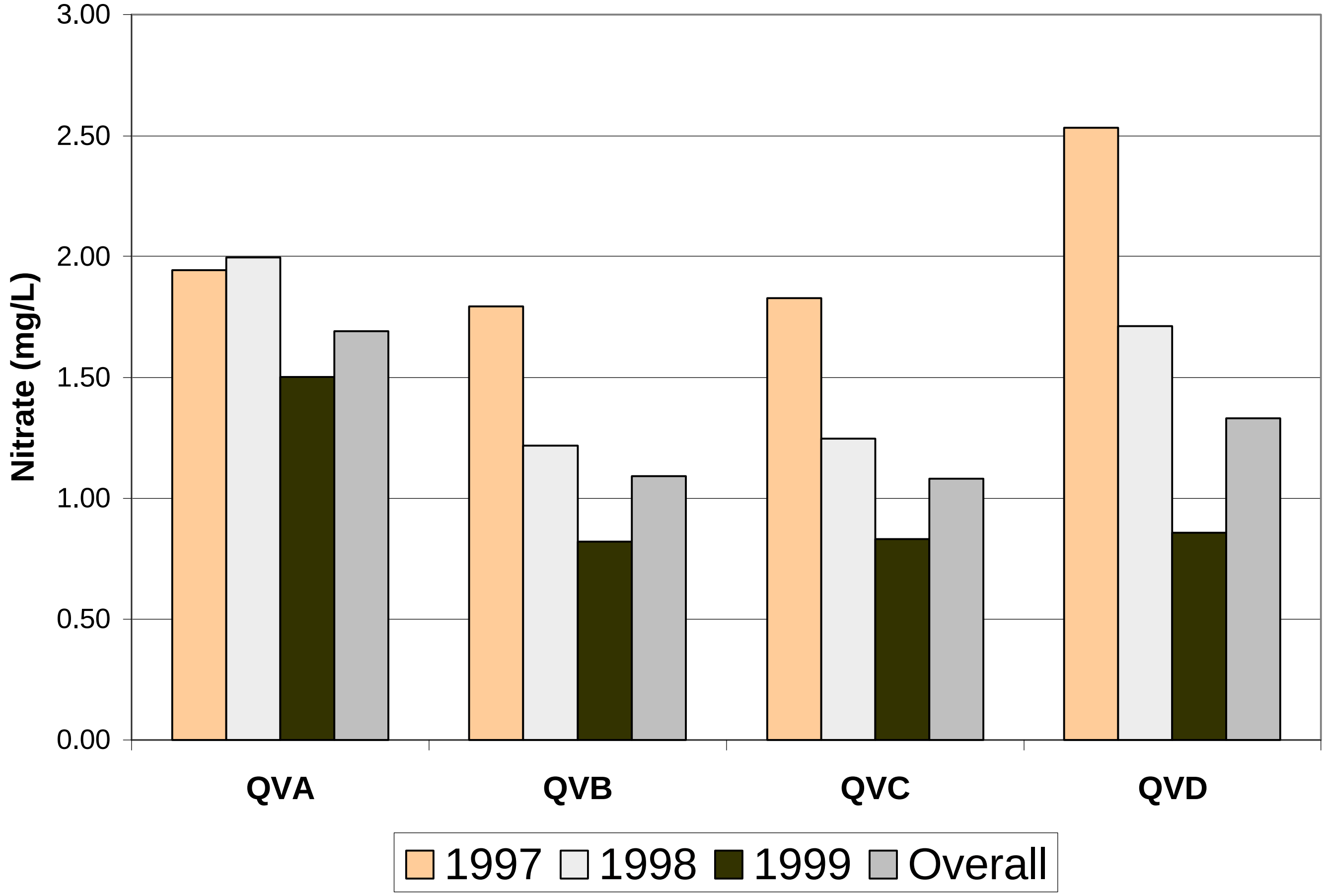


Figure 4.15 Annual medians of combined storm and baseflow concentrations for nitrate at all four sampling locations

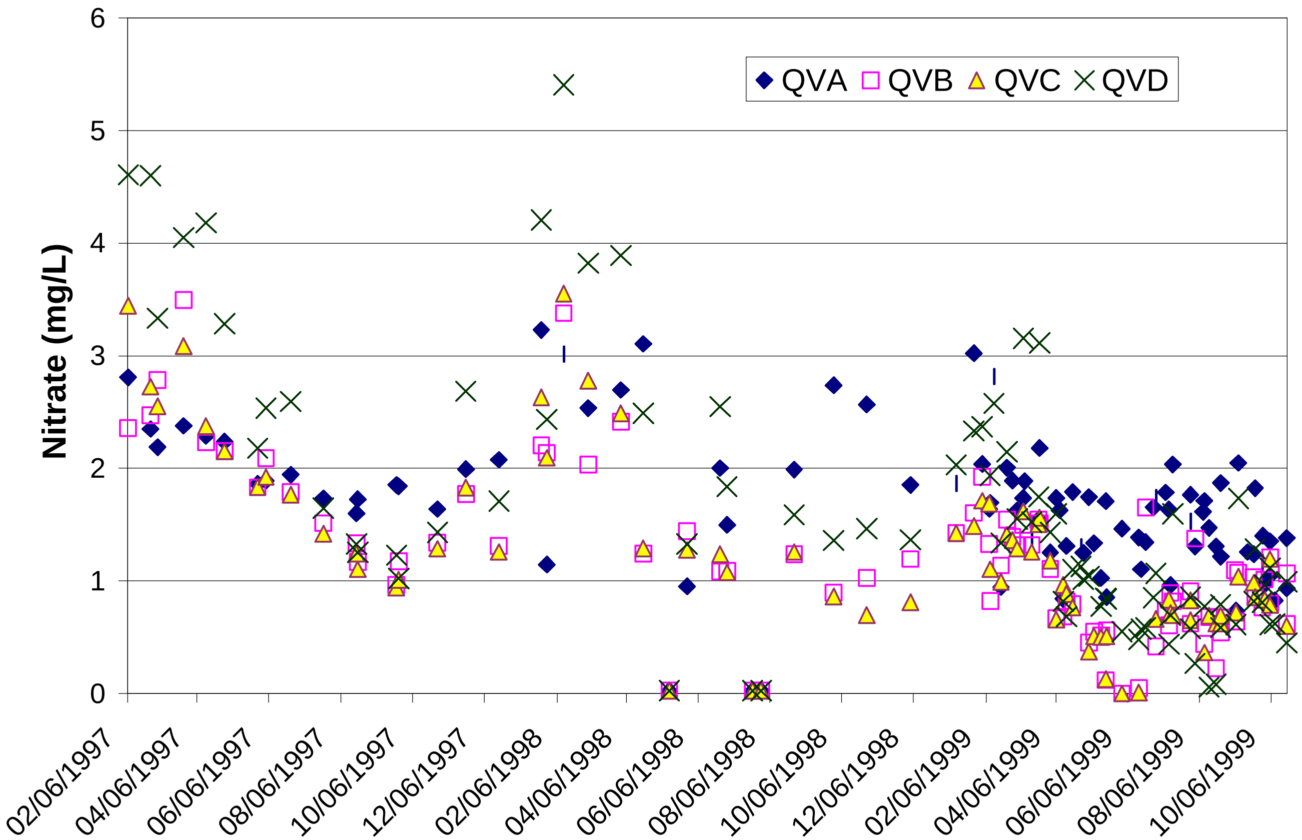


Figure 4.16 Time series of combined storm and baseflow measurements for nitrate at all four sampling locations

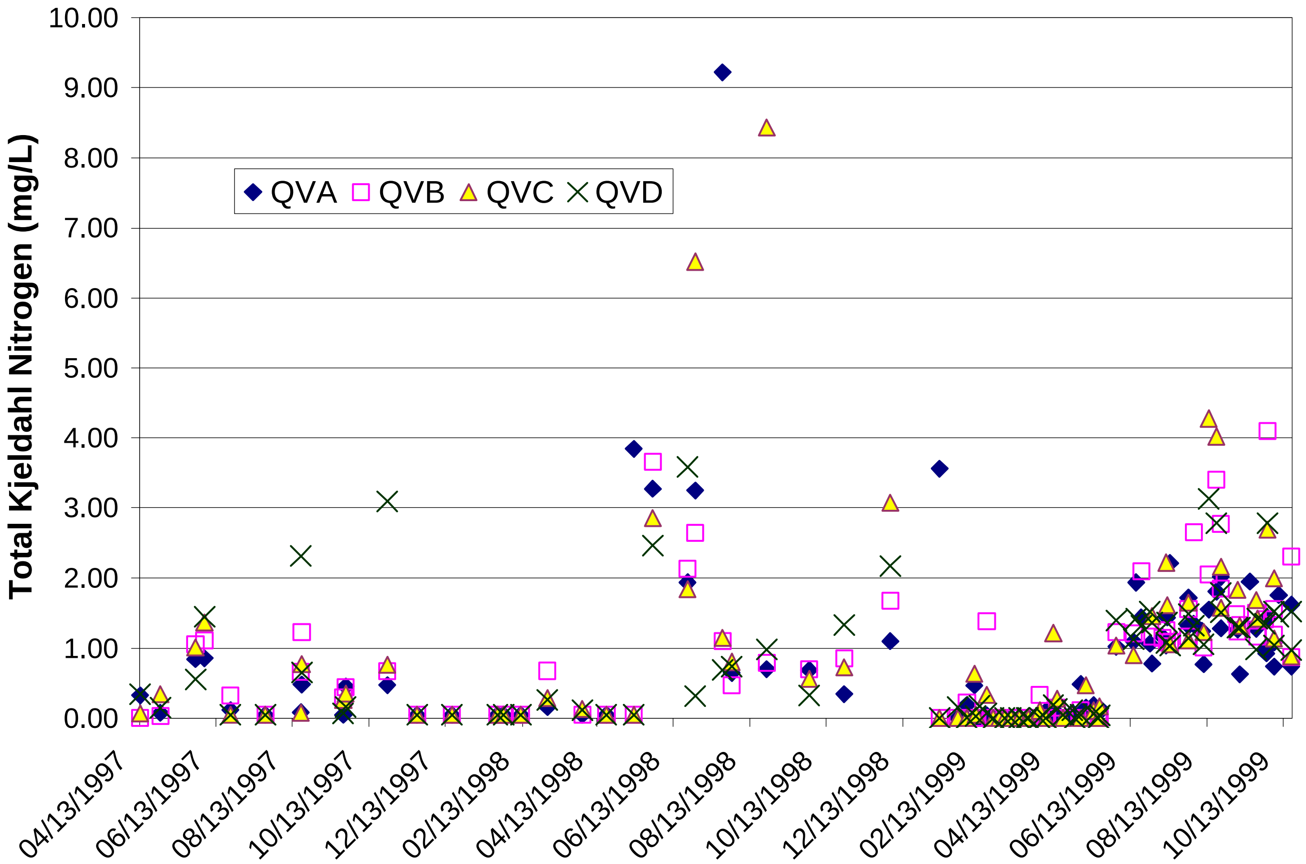


Figure 4.17 Time series of combined storm and baseflow measurements for TKN at all four sampling locations

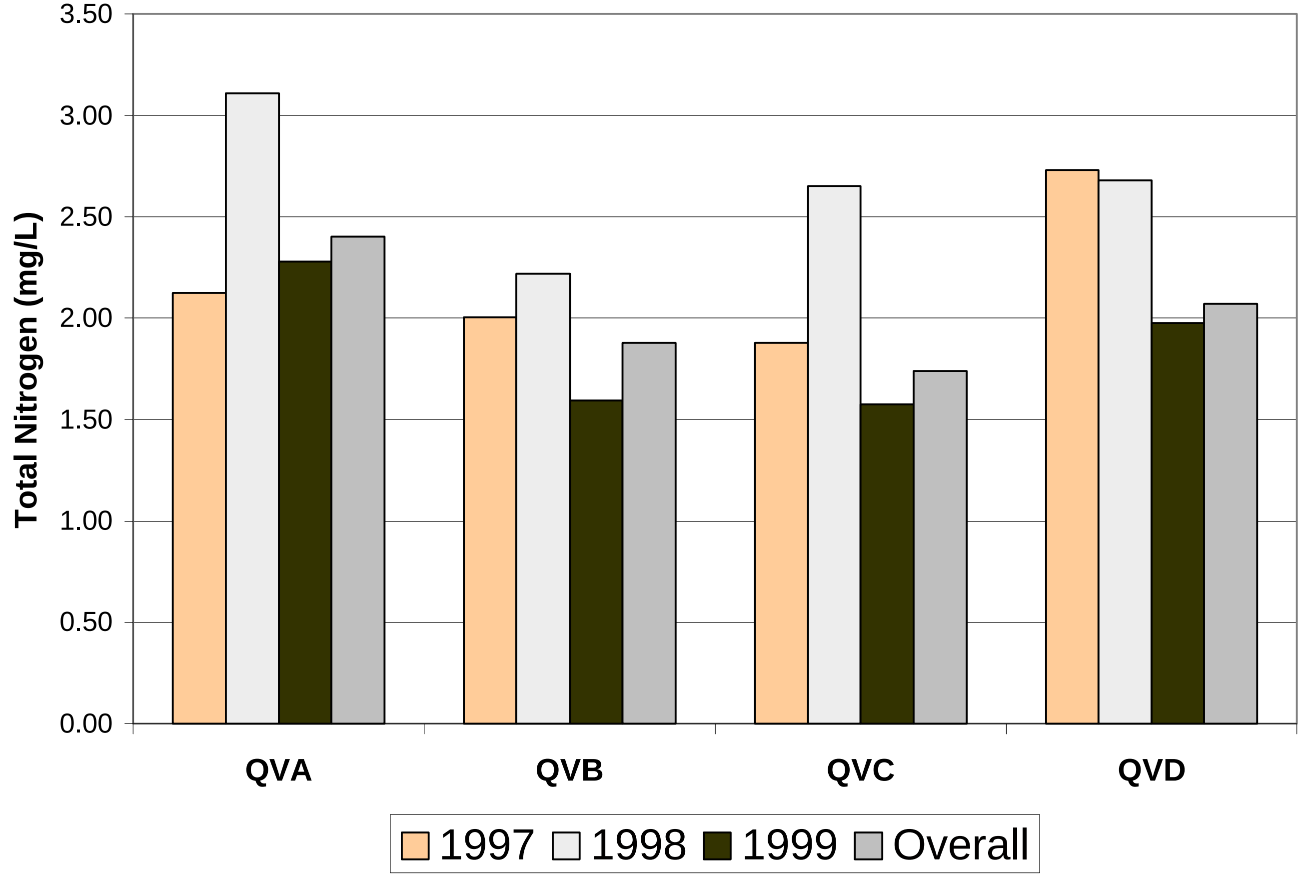


Figure 4.18 Annual medians of combined storm and baseflow concentrations for total nitrogen at all four sampling locations

of 29°C (Figure 4.19) specified by the state of Virginia for protection of aquatic life (DEQ-WQS, 1997). However, the influence of the pond can clearly be seen in the seasonal fluctuations of temperature at sites QVB, QVC, and QVD (while temperatures at QVA only fluctuate slightly) (Figure 4.19). The temperature fluctuation is a stressor to sensitive species within the aquatic community due to its effect upon their metabolism and reproduction (for example, egg production can decrease). According to Schueler and Galli (1995), “it is important to remember that stream temperature is one of the central organizing features of aquatic communities, affecting the rates of detrital processing, respiration, and bacterial growth, as well as the timing of reproduction, molting, and drift.” A rise in water temperature of just a few degrees Celsius over ambient conditions can reduce or eliminate sensitive stream insects (orders *Plecoptera* and *Trichoptera* in particular) and fish species (*Salmonid* species) (Schueler, 1987; Schueler and Galli, 1995). Virginia water quality standards specify that no rise more than 3°C above natural temperature shall occur in a tributary of the New River (DEQ-WQS, 1997). A rise of almost 8°C was seen every summer at stations QVB, QVC, and QVD, as compared to station QVA. In general, sustained summertime water temperatures in excess of 21°C (70°F) are considered stressful, if not lethal, to many cold-water organisms (Schueler, 1987). During the study period, every summer the temperature rose over 21°C for a substantial period of time.

Figure 4.20 shows that the minimum water quality criterion for pH (6-9) was exceeded at sites QVB and QVC for a short time in 1999. Though limited, such an exceedance is again detrimental to sensitive species within the community. Figure 4.21 also shows that a great number of samples fail to meet the dissolved oxygen water quality criteria at all sites (though this is less of a problem at station QVA). Furthermore, no trend is observed over time. The BMP practice of installing a wet pond may lower the dissolved oxygen in the receiving stream if the pond has high biological oxygen demands. Though not measured, substantial algal growth was observed in the wet pond and would not tend to suggest that dissolved oxygen would improve downstream.

The water quality standard for fecal coliforms is often exceeded in the sampled stream, typically during the summer months (Figure 4.22). The standard is specified as the geometric mean over a 30-day period which shall not exceed 200 colonies/100 ml for two or more samples from the

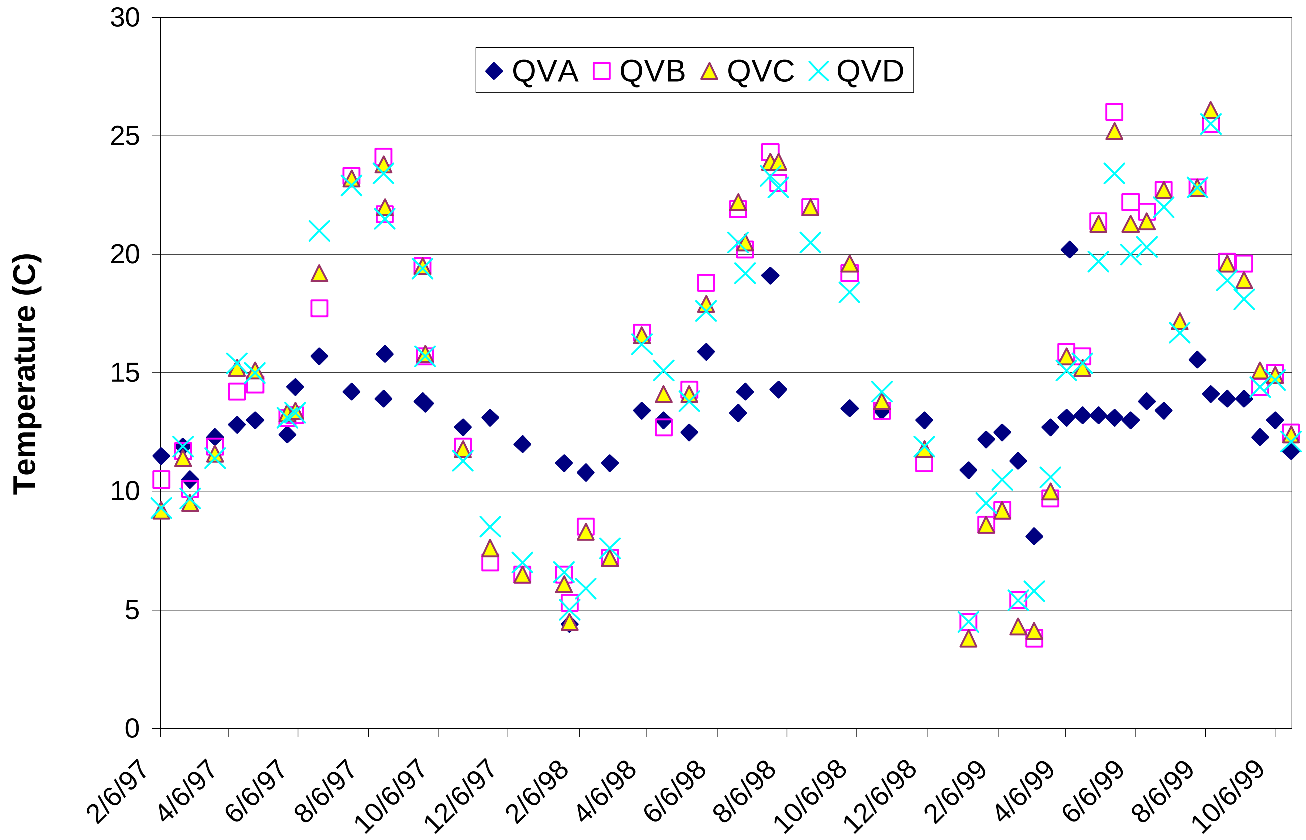


Figure 4.19 Time series of combined storm and baseflow measurements for temperature at all four sampling locations

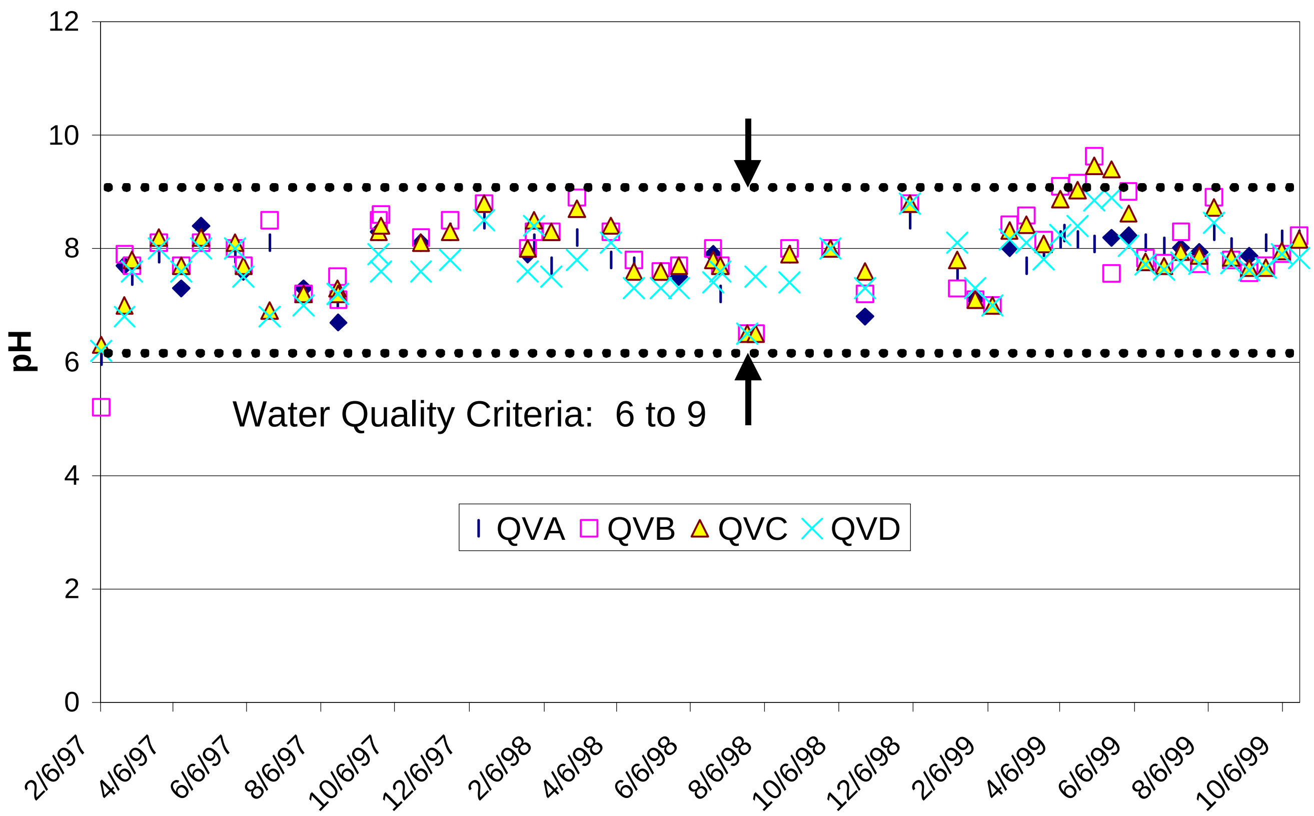


Figure 4.20 Time series of combined storm and baseflow measurements for pH at all four sampling locations, compared against its water quality criteria (DEQ-WQS, 1997)

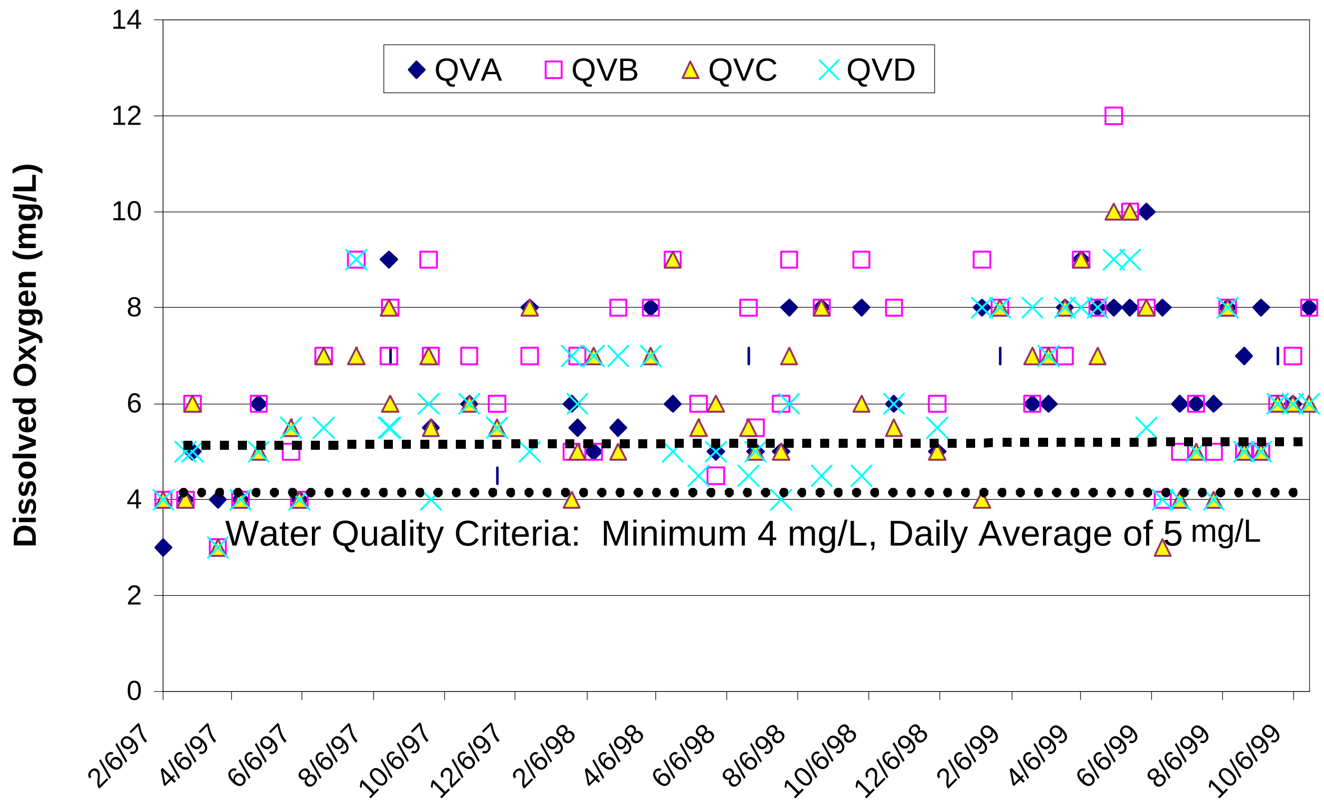


Figure 4.21 Time series of combined storm and baseflow measurements for dissolved oxygen at all four sampling locations, compared against its water quality criteria (DEQ-WQS, 1997)

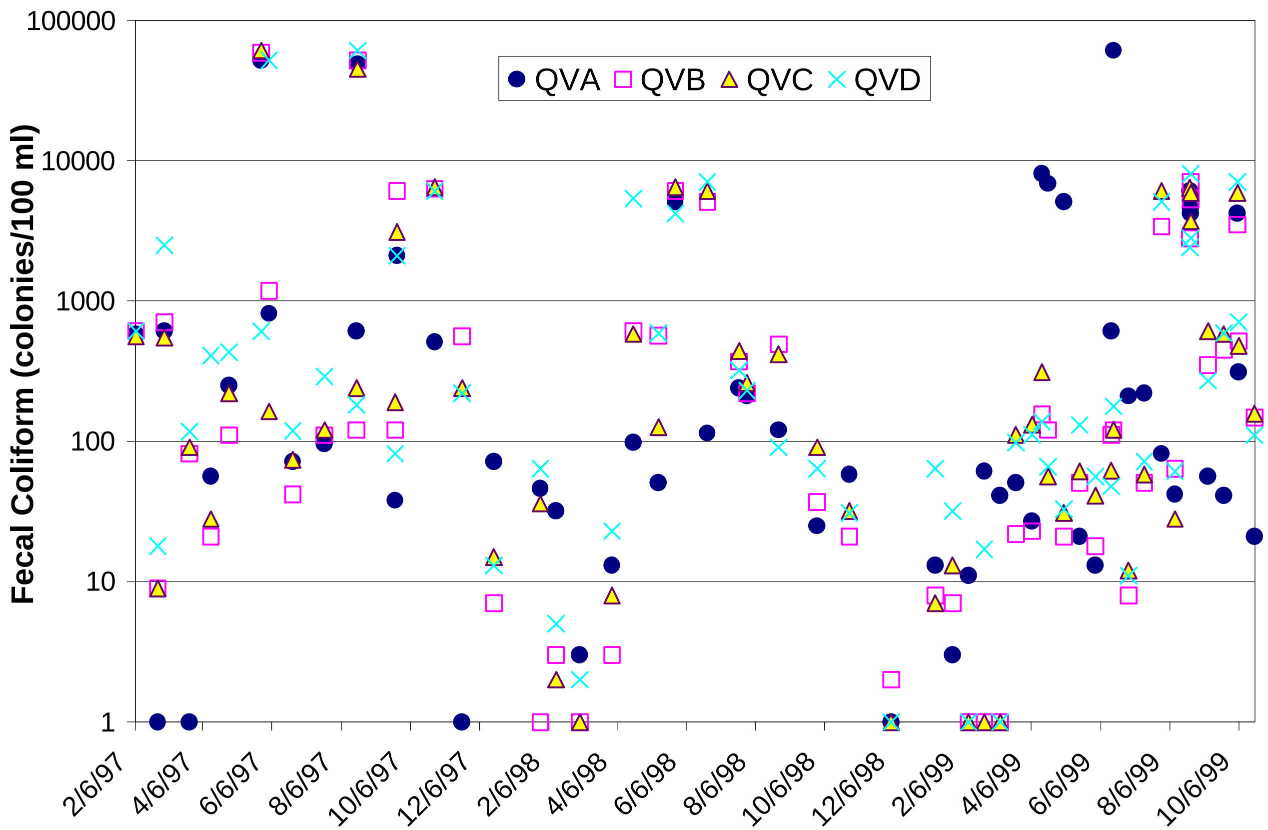


Figure 4.22 Time series of combined storm and baseflow measurements for fecal coliform bacteria at all four sampling locations

same water body. Growth rates of bacteria increase with temperature, and cattle and wildlife spend more time in the stream during hot summer periods. The combination of these factors could have resulted in high fecal coliform concentrations at the study site. Exceedance of the standard by water quality samples taken over the course of the study was 42% at QVA and QVB, 40% at QVC, and 47% at QVD; the exceedances are also evident in Figure 4.23, which shows the median annual and overall fecal coliform geometric means at all four sampling locations. Figure 4.24 shows that the fecal coliform results do not differ among sampling locations. The lack of fecal coliform reduction downstream of the wet pond may be a result of the limited residence time of the water in the pond.

4.1.1 Pollutant Loads

No statistically significant differences were detected when comparing pollutant inflow and outflow loads for the wet pond, dry pond, and overall system during the 1999 for the storm samples taken by the automatic samplers. However, examination of Figures 4.25 through 4.28 supports the hypotheses developed when examining pollutant concentrations in section 4.4.1. Median loads for chemical oxygen demand (COD) (Figure 4.25) and total phosphorus (Figure 4.26) mimic the total suspended solids (TSS) results (Figure 4.27) at the four sampling stations (QVC has the highest load and the other stations show increased load from upstream to downstream sites). This may be a result of both pond seston export and wet pond embankment failure. Total organic carbon (Figure 4.25) and nitrate loads (Figure 4.28) increase at site QVD compared to the other sites, perhaps due to the influence of wetland vegetation within the dry pond.

Possibly due to their soluble nature, orthophosphorus and ammonia loads do not change much among sites (Figures 4.23 and 4.24). Total nitrogen increases progressively downstream (Figure 4.28) and total kjeldahl nitrogen (TKN) appears to increase at each basin outlet, likely due to nitrogen transformations within the pond.

4.1.2 Pollutant Removal Efficiencies

Pollutant removal efficiencies were calculated based on basin inflow and outflow pollutant concentrations or loads. Positive removal efficiencies indicate removal by the stormwater management facility, while negative removal efficiencies indicate pollutant export due to

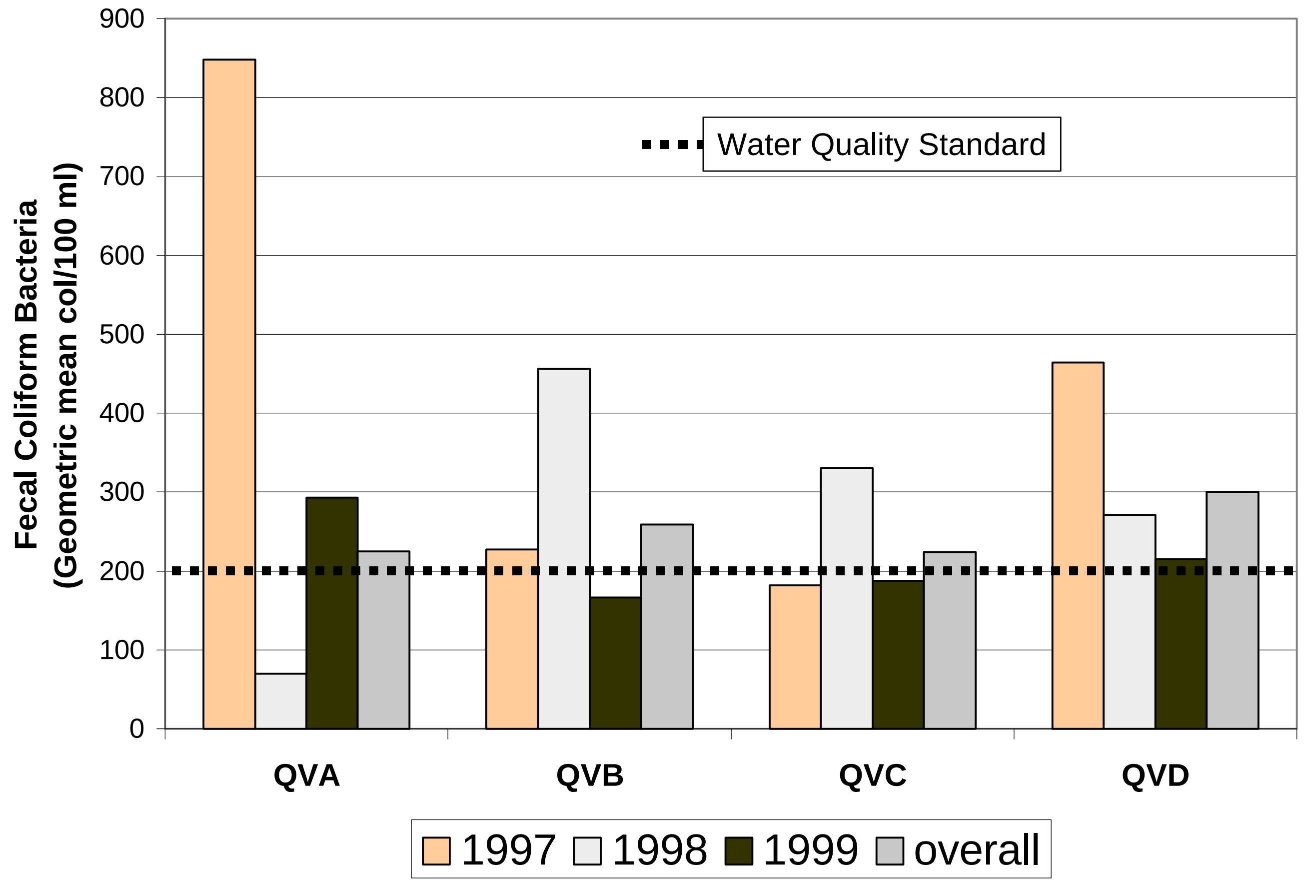


Figure 4.23 Annual medians of the geometric means of storm and baseflow measurements for fecal coliform bacteria concentrations at all four sampling locations, compared against its water quality standard (DEQ-WQS, 1997)

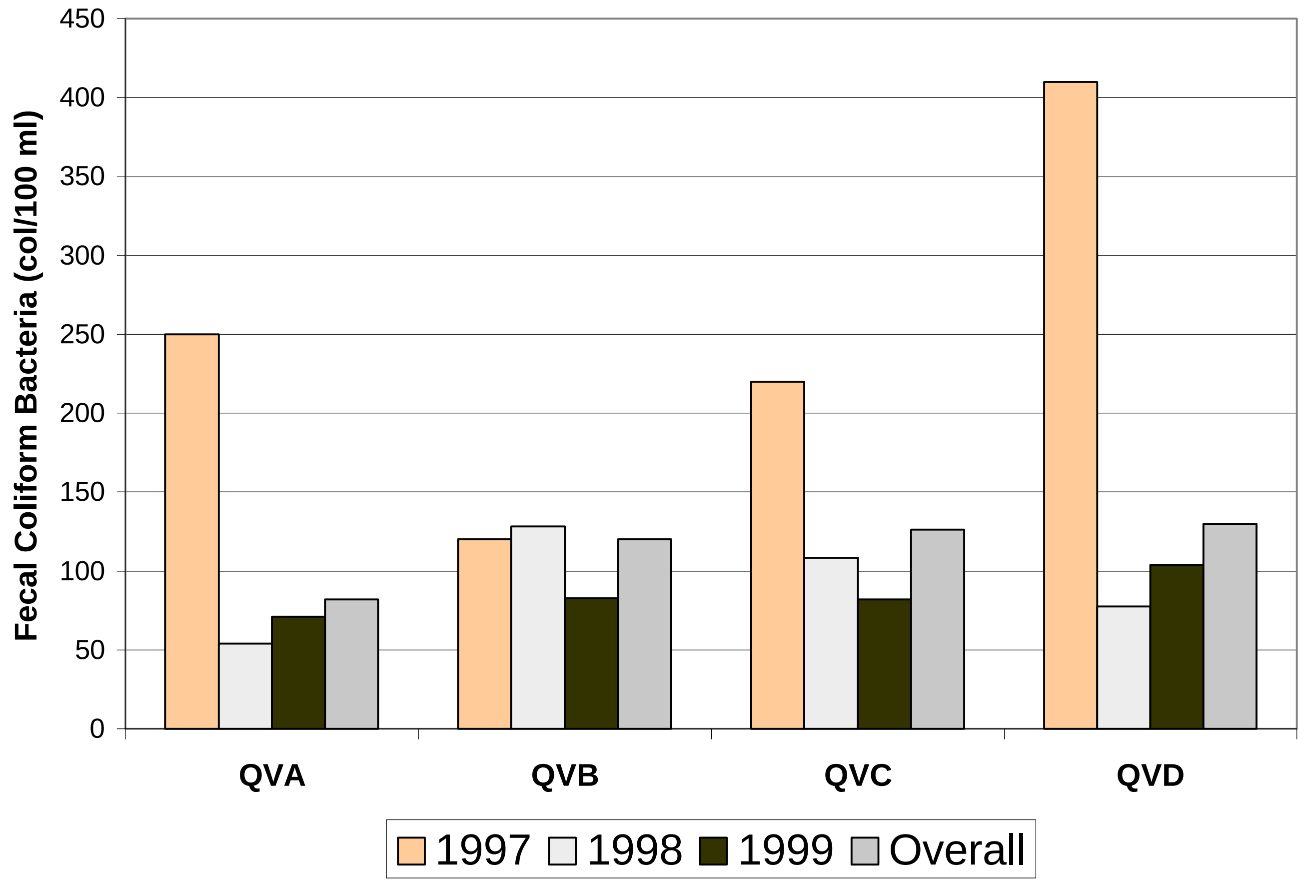


Figure 4.24 Annual medians of the fecal coliform bacteria concentrations at all four sampling locations

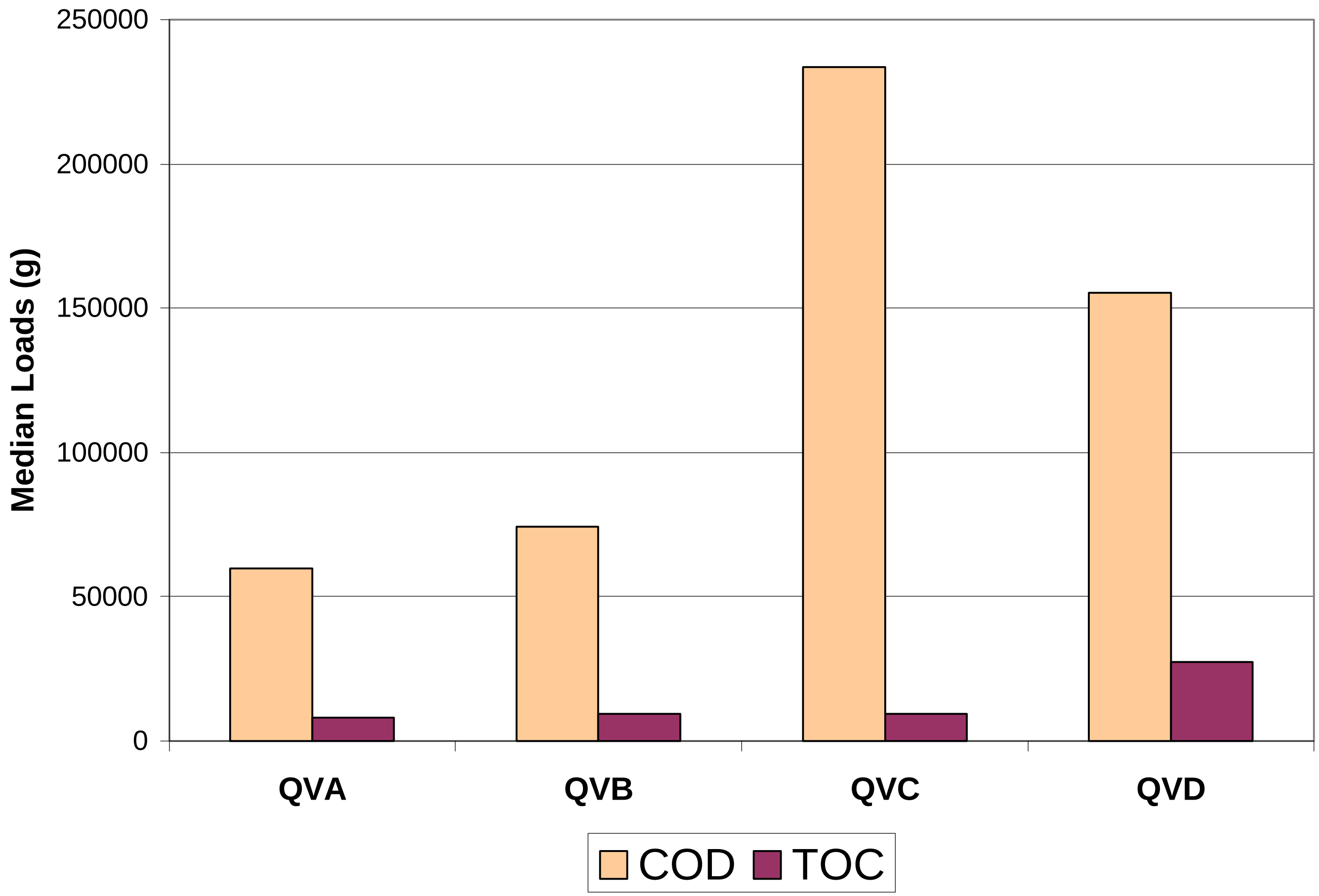


Figure 4.25 Median loads for chemical oxygen demand and total organic carbon as measured at all four sampling locations

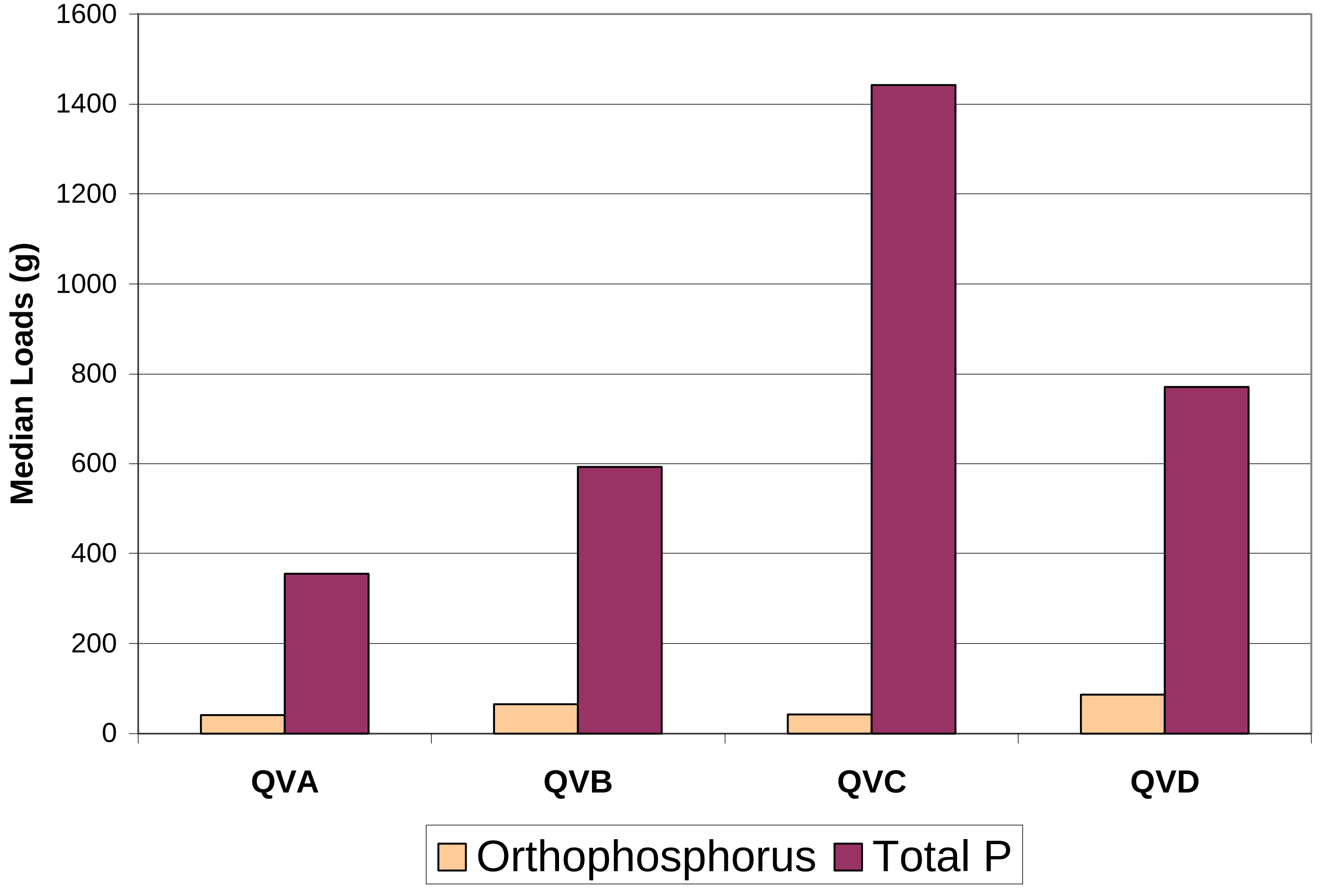


Figure 4.26 Median loads for orthophosphorus and total phosphorus as measured at all four sampling locations

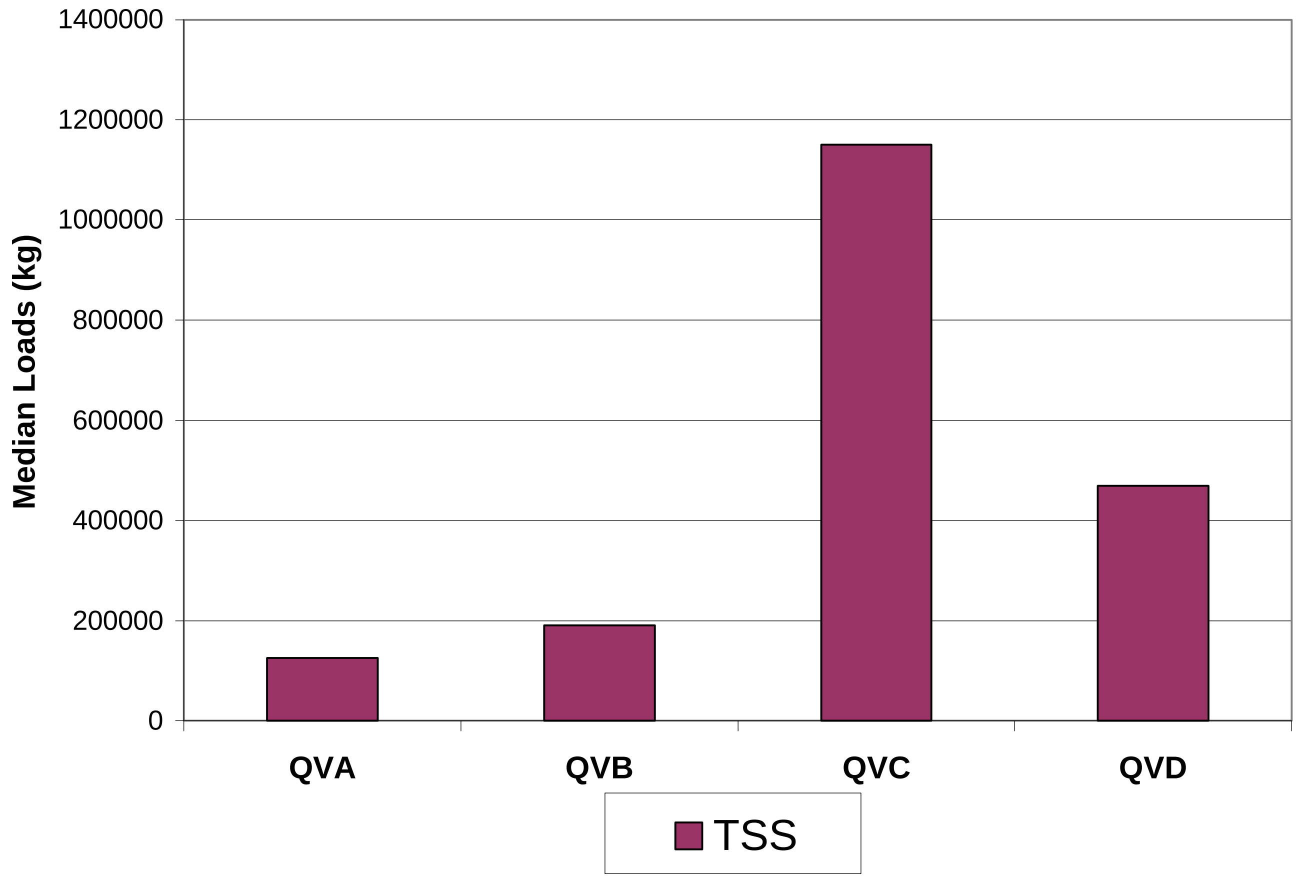


Figure 4.27 Median loads for total suspended solids as measured at all four sampling locations

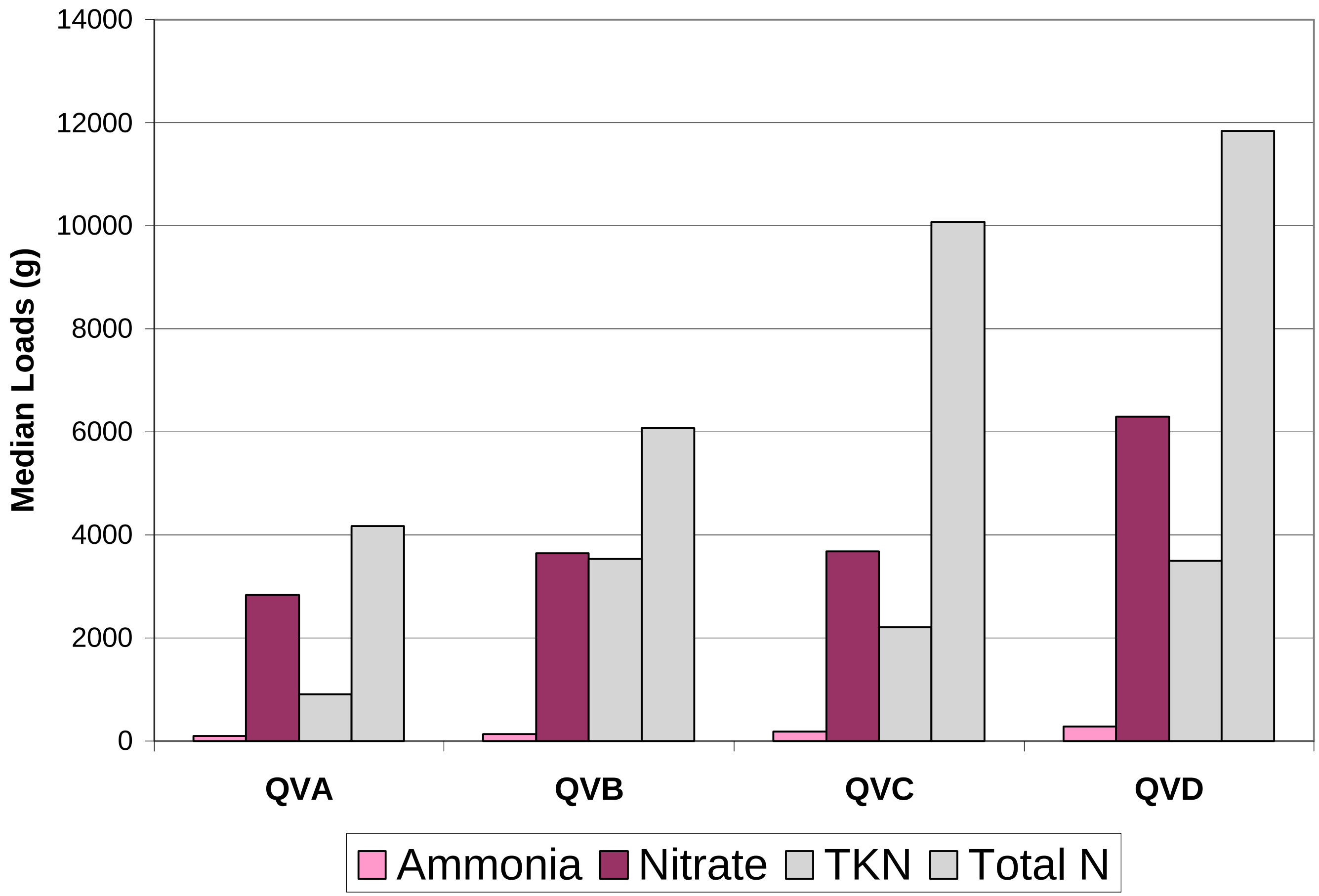


Figure 4.28 Median loads for ammonia, nitrate, total kjeldahl nitrogen, and total nitrogen as measured at all four sampling locations

pollutant generation within the basin or to additional pollutant sources. A best management practice is determined to be effective for stormwater quality control if it results in positive pollutant removal efficiencies.

The results from two different methods of calculating pollutant removal efficiencies are presented in this section. First, the event mean concentration (EMC) removal efficiencies were calculated because they are considered conservative unless the inflow volumes are diluted by other flow sources (groundwater or other surface flows) before they reach the pond outflow (Yu et al., 1998b). The EMC removal efficiency is a measure of a pond's performance in reducing the concentrations of pollutants. Flow volumes at station QVB were consistently less than that measured at station QVA and thus dilution was not considered a substantial problem. However, dilution might be a factor in the analysis of results from the dry pond because flow volumes at station QVC were consistently lower than flow volumes measured at station QVD.

$$\text{EMC Efficiency (\%)} = (1-(\frac{outletEMC}{inletEMC}))*100 \qquad (2.2)$$

The Sum of Loads (SOL) removal efficiencies (equation 2.3) account for flow volumes and were calculated in order to assess the long-term removal of pollutant mass (loads). Loads were summed over the entire study period prior to calculation of the removal efficiency values.

$$\text{SOL Effic.(\%)} = \frac{\sum(\text{Inflow Volume}*\text{Inflow EMC}) - \sum(\text{Outflow Volume}*\text{Outflow EMC})}{\sum(\text{Inflow Volume}*\text{Inflow EMC})} * 100 \qquad (2.3)$$

4.1.1.1 Baseflow

The pollutant removal efficiencies during baseflow conditions for the wet pond, dry pond, and overall facility (system) are presented in Table 4.6. Baseflow reductions were calculated using the Sum of Loads (SOL) method (Equation 2.3). However, the median of all baseflow grab sample concentrations at the respective sampling station was used for the inflow and outflow concentrations in the equation. The median baseflow volume at each sampling station, calculated from baseflow rates measured prior to the storm events sampled in 1999, was used for the inflow and outflow volumes in the equation.

Table 4.6 Pollutant removal efficiencies of the VPI&SU stormwater management facility during baseflow conditions

Analyte	Weekly Baseflow Reductions (%)		
	Wet Pond	Dry Pond	System
TSS	-29∗	-436	-1026
TOC	-44	-2726	-7215
COD	45	-436	-463
PO_4	26	-659000	-901049
TP	8	-71303	-74996
NH_4	67	-225382	-145247
NO_3	57	-3774	-2316
TKN	31	-15741	-12214
TN	54	-2631	-1612
Cd	26	-34581	-87440
Pb	2	-1201	-2818
Zn	4	-69262	-97167
Cu	-9	-594	-711

∗ Negative numbers indicate pollutant export rather than removal

The wet pond was very effective in removing baseflow concentrations of most pollutants. The greatest pollutant removal efficiencies were obtained for NH_4 (67%), NO_3 (57%), total nitrogen (54%), and chemical oxygen demand (COD) (45%) (Table 4.6). A statistical analysis (modified sign test) of the baseflow data, summarizing nitrate and total nitrogen data by season, resulted in significant differences between stations QVA and QVB and supports the pollutant removal efficiency results. These results are different from results from Van Buren (1994) that reported positive removal rates for the organic contaminants and metals, and negative removal rates for nutrients and COD during periods of baseflow. The wet pond was not effective for removing TSS (-29%) perhaps because of the wet pond embankment failure, or for removing TOC (-44%) likely due to organic matter decay within the pond.

The dry pond was not effective in reducing baseflow concentrations of pollutants nor was the facility taken as a whole (Table 4.6). However, one interesting result was obtained by the application of the modified sign test to the seasonal data set of pH measurements (Appendix B). Both the dry pond and the overall system show a significant reduction in pH value at QVD compared to stations QVC and QVA, respectively. These results would be of interest because

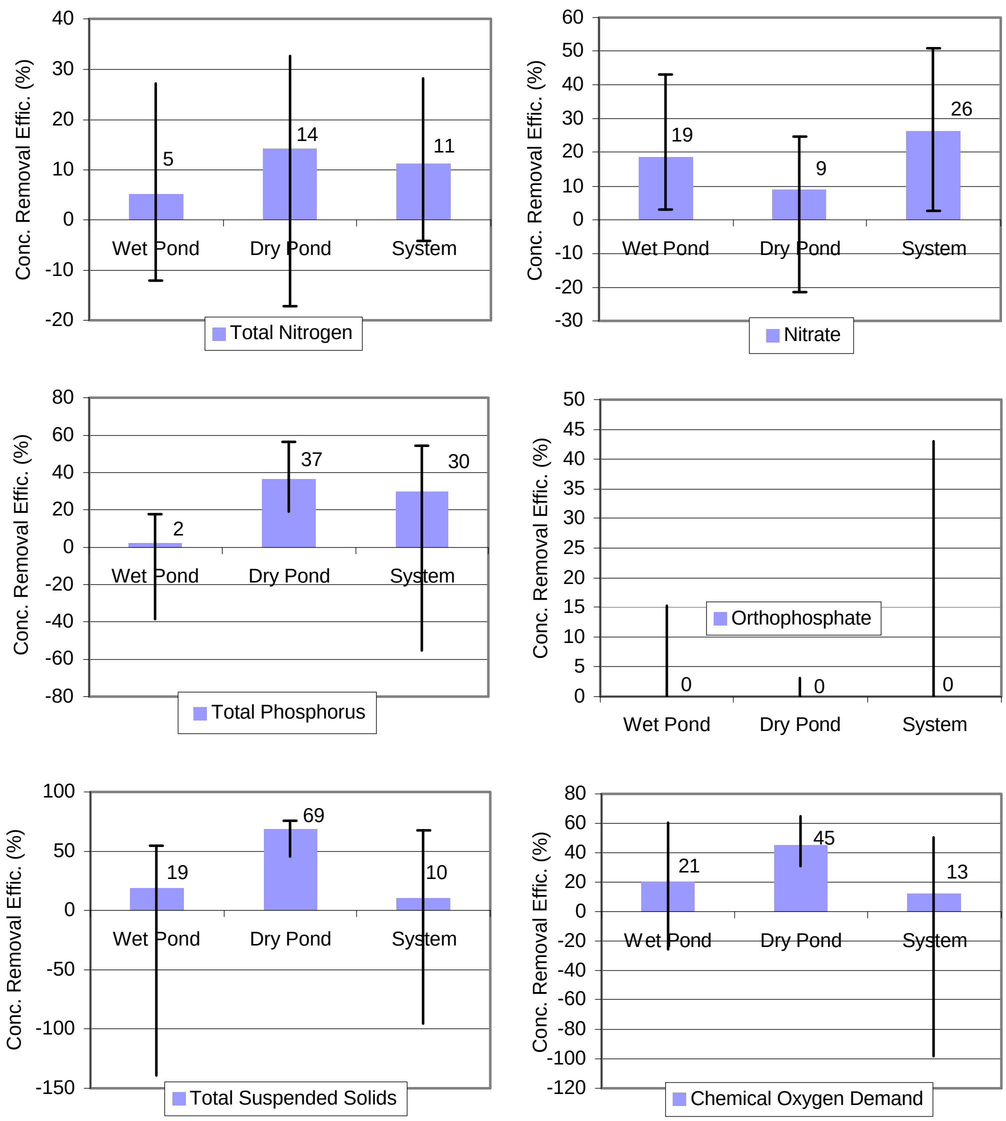


Figure 4.29 Median Event Mean Concentration (EMC) efficiencies of water quality parameters for the wet pond, dry pond, and stormwater management facility (entire system), shown with their interquartile ranges

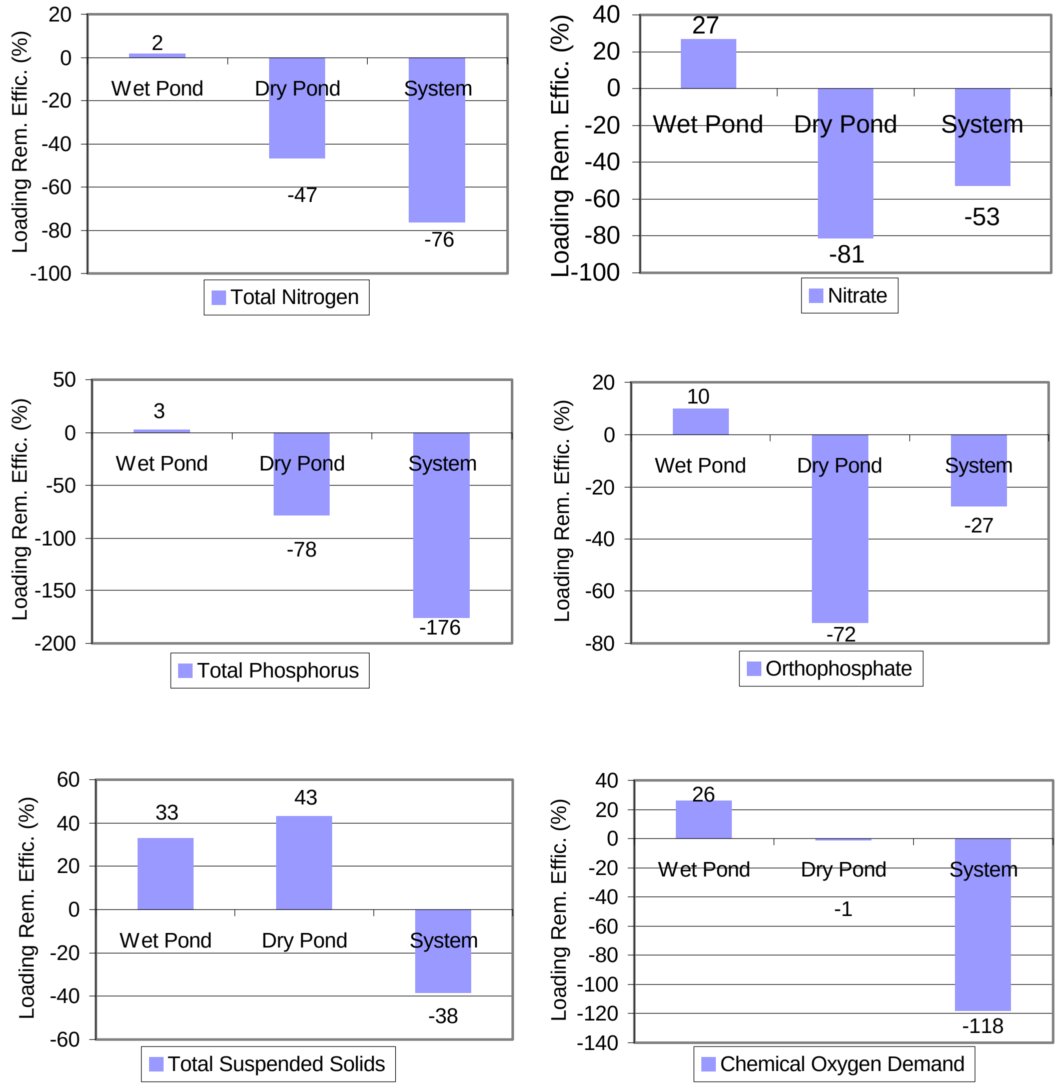


Figure 4.30 Loading removal (SOL) efficiencies of water quality parameters for the wet pond, dry pond, and stormwater management facility (entire system)

pH exceedances in particular are hypothesized to threaten the benthic assemblages in Stroubles Creek.

4.1.1.1 Stormflow

The storm event removal efficiencies, calculated for the each basin and for the stormwater management facility as a whole, are reported in Table 4.7. Selected pollutant removal efficiencies are presented in Figure 4.29 (EMC efficiencies) and Figure 4.30 (SOL efficiencies).

Table 4.7 Comparison of the median pollutant removal efficiencies (%) for each constituent sampled by automatic samplers during storm events

Analyte	EMC Removal Efficiency*			SOL Efficiency **		
	Wet Pond	Dry Pond	System	Wet Pond	Dry Pond	System
TSS	19	69	10	33	43	-38***
COD	21	45	13	26	-1	-118
TOC	13	-14	-4	12	-184	-172
NH_4	8	8	1	41	-85	-64
NO_3	19	9	26	27	-81	-53
TKN	-26	36	0	-32	-23	-108
TN	5	14	11	2	-47	-76
PO_4	0	0	0	10	-72	-27
TP	2	37	30	3	-78	-176

* The median of the EMC removal efficiency values from individual storm events

** Calculated as the sum of all outflow loads subtracted from the sum of all inflow loads and the result divided by the sum of all inflow loads

*** Negative values indicate pollutant export rather than removal

4.1.1.1.1 Pollutant Concentration Removal Efficiencies

The median storm event EMC efficiencies indicate higher pollutant removal of ammonia and nitrate in the wet pond compared with TKN (Table 4.7). EMC removal efficiencies for ammonia and nitrate were 8 and 19% respectively for the wet pond. However, the wet pond was not effective in removing TKN (-26%). One possible explanation is that dissolved constituents are transformed by biological processes within the wet pond to other forms of nitrogen such as TKN or inorganic N. Thus, there would be an increase in TKN at the pond outlet. Typically, the TKN would be expected to decrease at the pond outlet due to the sedimentation of organic matter within the basin (Cunningham, 1993). The higher removal of TKN within the dry pond (EMC removal 36% compared to 8 and 9% respectively for ammonia and nitrate) may be attributable to sedimentation since TSS removal is relatively high in the dry pond (69% EMC removal efficiency, and 43% SOL removal efficiency). However, it should be noted that dilution in the dry pond might have elevated these results. EMC TSS removal efficiency is also positive for the wet pond (19%) and the entire system (10%) but much less than the 80 to 90% removal expected for wet ponds (Hartigan, 1989).

These results are supported by results from statistical tests presented in Appendix B. Median EMC removal efficiencies for nitrate and ammonia in the wet pond are statistically significant (Appendix B) when using the modified sign test. Furthermore, Figure 4.29 shows the interquartile ranges for the analytes with respect to the median EMC removal efficiencies (also presented in Appendix D). The only analytes whose interquartile range does not include zero are nitrate (Figure 4.29) for the wet pond and for the entire system; and total suspended solids, chemical oxygen demand, and total phosphorus for the dry pond. The statistics support the hypotheses of nitrogen transformations within the wet pond and sedimentation within the dry pond. The EMC removal efficiency for total organic carbon (13%) in the wet pond is also statistically significant which may indicate that some sedimentation within the wet pond could be occurring. It is possible that this process is not as evident with respect to the other analytes (TSS, total phosphorus) due to the masking effect of wet pond embankment erosion.

The data presented in Table 4.8 show EMC removal rates for extended dry detention ponds (Hodges, 1997) as compared to the VPI&SU stormwater management facility (both basins and overall). The total phosphorus and TSS removal efficiencies of the VPI&SU dry pond show equivalent or greater removal efficiencies as compared to the other studies, which supports other evidence that pollutant removal by sedimentation is occurring in the dry pond. The system as a whole achieves 26% nitrate and 30% of total phosphorus removal, less than the extended dry detention basin results in Table 4.8. Considering the cost and size of the VPI&SU wet and dry pond stormwater management facility, it would be expected to perform better than extended dry detention basins (Table 4.8). According to Hartigan (1989), wet ponds, compared to extended dry detention ponds, should have 2-3 times greater removal of total P (50-60% vs. 20-30%) and 1.3-2 times greater removal of total N (30-40% compared to 20-30%).

Table 4.8 EMC removal rates (%) for several extended dry detention basins (Hodges, 1997) compared to results from this study

Analyte	Zariello and Sherwood (1993)	Hodges (1997)	VPI&SU Wet Pond	VPI&SU Dry Pond	VPI&SU System
TSS	83.8	65.7	19	69	10
TOC	47.4	38.2	13	-14	-4
NH_4	21.5	96	8	8	1
NO_3	35.2	70.4	19	9	26
TP	32	27.3	2	37	30
FTP	11.1	-136	2	13	16

Removal of bacteria from urban stormwater should be able to occur within wet ponds if residence time is sufficient (five or more days) to allow natural die-off of coliform bacteria (Schueler, 1987; Driscoll, 1989; Droste et al. 1993). While fecal coliforms did show a seasonal response within the stream (increases during warm weather), no marked reductions were noted over time (Figure 4.22) or due to the influence of the basins (Figures 4.23 and 4.24). EMC efficiencies were calculated for fecal coliforms using the combined baseflow and storm event data that showed fecal coliform exports from the wet pond (-46%), dry pond (-3%), and facility (-59%).

The removal efficiency results for the dry pond and for the facility as a whole may have been lowered due to the pollutant contributions from subwatershed QVG. This may have been the case for TSS, TOC, COD, total P, orthophosphorus, filtered total P, nitrate, total N, fecal coliforms, and fecal streptococci, but less important for ammonia, TKN, filtered TKN, total coliforms, and zinc.

4.1.1.1.2 Pollutant Load Removal Efficiencies

Load removal efficiencies for ammonia and nitrate were 41 and 27% respectively for the wet pond. These results, when compared to TKN removal (-32%), support the hypothesis developed in the previous section for nutrient transformation within the wet pond, and the 43% TSS load removal in the dry pond supports the previous evidence for sedimentation of TSS and associated pollutants within the dry pond. Some sedimentation may occur within the wet pond as shown by a 33% removal efficiency of TSS, still far lower than the expected 80 to 90%, and 26% COD removal efficiency expected for both dry and wet ponds (Hartigan, 1989). All of the load removal efficiencies for the stormwater management facility (stations QVA compared to QVD) are negative, indicating no net benefit due to installation of the facility. One possible reason for increased number of negative values of removal efficiencies calculated by the SOL method as compared to the EMC method was that dilution elevated the EMC removal efficiencies.

If the VPI&SU stormwater management facility had functioned as intended, it would be expected to achieve pollutant removal comparable to other combined facilities. Results from a combined pond/wetland system were reported by Urbonas et al. (1993) (Schueler, 1994a), while the results presented by Leersnyder (1993)(Schueler, 1994b) were from a combined system with large treatment volume, excellent internal geometry, and redundant treatment mechanisms. Table 4.9 shows these results as examples of what are considered high pollutant removal rates (Schueler, 1994a; and Schueler, 1994b); and are presented here for comparison with the VPI&SU mass pollutant removal efficiencies.

The VPI&SU facility has higher nitrate removal (27%) than Urbonas et al. (1993). No benefit of the combined system is realized however because all of the VPI&SU parameters show net pollutant export rather than reduction.

Table 4.9 Pollutant removal rates (%) for pond/wetland systems (from Schueler (1994a), adapted from Leersnyder (1993); and from Schueler (1994b), adapted from Urbonas et al. (1993)) compared to the VPI&SU facility

Parameter	**Urbonas et al. (1993)**			**Leersnyder (1993)**	**VPI&SU Facility (SOL)**		
	Pond	Wetland	System	System	Wet Pond	Dry Pond	System
TSS	78	29	72	78	33	43	-38
TP	49	3	51	79	3	-78	-176
NO_3	-85	5	-76	62	27	-81	-53
NH_4	NA	NA	NA	-43	41	-85	-64
COD	44	21	56	2	26	-1	-118

The pollutant removal efficiencies are expected to increase after the wet pond embankment is replaced and the system has had a chance to stabilize with respect to pollutant removal performance. A wet pond in Virginia is designed with the intent to reduce total phosphorus by 40 to 60%, based on the proportion of impervious cover in the watershed (Table 2.1) (DCR, 1998b). In this stormwater management facility, only the dry pond has statistically significant reductions in total phosphorus concentrations. A seasonal analysis also shows a significant reduction in orthophosphate concentrations by the dry pond. However, Table 4.7 shows that phosphorus is not removed from the system in large amounts, and the maximum phosphorus loading removal efficiency was 10% for orthophosphorus and only 3% for total phosphorus (Table 4.7).

4.1.1.1.3 Discussion of Pollutant Removal

The stormflow removal rates achieved in this study (Table 4.7) rank in the bottom percentiles of removal rates summarized by Schueler (1993) for 58 monitoring studies of stormwater ponds and wetlands (Table 2.3). Because the removal rates do not fall into a majority category, the results cannot be judged as typical for a wet pond system. The wet pond achieves 19% EMC efficiency removal of TSS that is comparable to 5% of the studies reviewed by Schueler, 2% EMC efficiency removal of total phosphorus comparable to 19% of the other studies, and 5% EMC efficiency removal of total nitrogen comparable to 18% of the studies reviewed.

The VPI&SU system may never perform as expected due to three factors. First, its two day residence time (compared to 2 weeks usual in wet pond design (Hartigan 1986)) may not increase substantially once the pond is fully operational and therefore optimal performance capability is lost. Second, the pond surface area to watershed surface area ratio of 0.6% does not meet the 1 to 2% ratio recommended to achieve pollutant removal efficiencies of 40 to 80% for TSS, 30% for TKN, and 45% for total phosphorus (Wu, 1989). Third, the pond is shallow and its volume is small based on its capacity to store runoff from the watershed area. The depth to the water table and the cost to excavate further in the presence of bedrock are factors that would prevent increasing the pond's depth. Entrainment of previously deposited sediment in the pond is possible when pond depth is less than 0.91-m (Marsalek, 1997), as it is in the majority of the wet pond. Winds with greater than 2 to 4-m/s velocities may cause vertical circulation patterns within the wet pond and resuspend sediment. Thus, the shallow pond depth may contribute to the low pollutant removal efficiencies measured in the wet pond.

The possible interaction of the wet pond with the water table could also have affected the pollutant removal efficiency results. During baseflow periods, water may infiltrate into the groundwater from the pond. Pollutant concentrations may increase in the pond if the pollutants are filtered out of the infiltrating water and remain in the pond sediment. Therefore, concentration pollutant removal efficiencies may be lowered during baseflow periods. During storm events, additional groundwater inflow could dilute pollutant concentrations within the pond sufficient to enhance concentration pollutant removal efficiencies. It is unknown how the loading removal efficiencies would respond to the competing influences of pollutant concentration during baseflow periods, and pollutant dilution during storm events. Furthermore, the water quality characteristics of the groundwater are unknown and could influence the results.

4.2 Stormwater Management Facility Impacts

4.2.1 Habitat

It is expected that after installation of a stormwater management facility, the habitat condition downstream will recover to some extent (Maxted and Shaver 1996) due to mitigation of erosive stormwater runoff velocities. This mitigation is required by Virginia stormwater management regulations to protect properties and receiving waters downstream of land development projects from erosion and damage due to increases in volume, peak velocity, and peak flow rate of stormwater runoff (DCR, 1998b). It would take time for the downstream habitat to recover, partly due to the limited frequency of flows that could carry the previously deposited sediment out of the reach. However, sedimentation within the wet pond should create low-TSS pond outflow conditions with high potential carrying capacity for sediment from the streambed. Twelve months after highway construction in a Virginia stream, fish were able to repopulate the impacted stream. The length of time will vary depending on the stream's self-cleansing ability to remove silt (Pitt, 1995). In some cases, in-place polluted sediments must be controlled and destroyed habitat must be restored within the receiving stream (Pitt, 1995). Furthermore, wet ponds are barriers to both high-intensity cleansing flows as well as to fish migration upstream (Schueler and Galli, 1995). Wet ponds also block the movement of coarse-particles downstream while exporting fine-grained particles, which can lead to increased embeddedness of downstream substrates (Schueler and Galli, 1995).

The results of the study (Tables 4.12 through 4.15) do not demonstrate habitat improvements after installation of the stormwater facility. Increased habitat quality, evidenced by most metrics in Table 4.10, may have been attained at station QVA, located upstream of the wet pond, and station QVF, downstream of the facility. Stations QVB and QVC, located downstream of the wet pond, showed decreases in habitat quality metrics primarily because of deposited sediment (Tables 4.13 and 4.14).

Two trends can be seen from the habitat metric results, displayed separately for each station in Tables 4.10 through 4.13. The first trend in habitat metric change occurred between the Feb. 20, 1997 sample and the April 29, 1997 sample. Decreases in the habitat metric scores at sites downstream of the wet pond (QVB, QVC, and QVF) occurred for the measures of epifaunal substrate and available cover (1), embeddedness (2), and sediment deposition (4). Likely, this is the result of sediment movement into the stream channel by the construction of the stormwater management facility. The second trend of habitat metric change involves the comparison of the 1997 samples to the Nov. 11, 1999 sample. In 1999, all stations downstream of the wet pond had increased sediment deposition (4) (Table 4.11 to 4.13), and station QVB had increased embeddedness (2) (Table 4.11). Both stations QVB and QVC had a decreased frequency of riffles (7) in 1999, also a potential result of sediment deposition (Table 4.11 and 4.12). Station QVB (Table 4.11) furthermore showed a decreased amount of substrate and available cover (1) likely the result of the additional sediment entering the stream channel due to the erosion of the wet pond embankment. Since the early spring of 1999, the wet pond embankment has been failing due to improper design as evidenced by large eroded areas on the upstream and downstream faces of the embankment. The embankment is scheduled for replacement by Virginia Tech in the summer of 2000 (Martha Wirt, personal communication, Feb. 25, 2000). Furthermore, the decreases in the metrics for bank stability (8a and 8b) and vegetative protection of the streambank (9a and 9b) in the 1999 samples at sites QVB and QVC (Table 4.11 and 4.12) show that degraded habitat conditions are partially a result of erosion occurring within the stream channel. This indicates that the wet pond has not effectively controlled flow velocities in the stream.

Table 4.14 gives the ratio of the habitat assessment score, for each site and sampling date, to a minimum cutoff score (160 on a scale of 200) used as reference for this study (Dr. Voshell, personal communication, 1998). These results show that overall habitat quality appears to increase slightly at stations QVA and QVF in 1999 as compared to 1997 (scores in the range of 70% of the reference condition as compared to 60%). However, overall habitat quality decreases at station QVB (score in the range of 50% as compared to 60%), and do not appear to change at station QVC.

It is unclear how much time is necessary after installation of a stormwater management facility before significant habitat recovery will be seen, assuming that the management facility, as intended, mitigates the increased stormwater volumes, peak velocities, and peak flows resulting from urbanization of a watershed. Two years is often assumed adequate for recovery results to be noticeable, but this may not be sufficient if the facility is not stabilized (Maxted and Shaver, 1996). The implication of these results is that habitat recovery may not be noticeable downstream of the wet pond until two or more years after the wet pond embankment is rebuilt.

4.2.2 Benthic Macroinvertebrates

It is hoped that stream health and potential uses would improve due to the installation of the wet pond, and thus results from the macroinvertebrate biomonitoring at sampling locations downstream of the wet pond were examined for evidence of biotic condition improvement. Biotic conditions at site QVA were not expected to improve because the stream above site QVA has been channeled through underground culverts with little natural habitat remaining.

Macroinvertebrate taxa and numbers for each site are presented by sampling date in Tables 4.17-4.20. The metrics that were calculated for each sampling date are presented in Tables 4.21-4.24 for each sampling location. It is important to note is that all of the metrics, though they reflect very different aspects of community ecology, show moderate to high impairment on all dates due to pollution or environmental stress. All reference comparisons are made to the information presented in Tables 3.7 and 3.8, unless otherwise noted. Table 4.23 summarizes the Macroinvertebrate Aggregated Index for Streams (MAIS) scores for each site.

Table 4.10 Habitat metrics at station QVA for the four sampling dates

QVA Metrics	Feb. 20, 1997	Apr. 29, 1997	Aug. 29, 1997	Nov. 11, 1999
(1) Epifaunal Substrate / Available Cover	11	11	11	14
(2) Embeddedness	13	13	13	15
(3) Velocity/Depth Regime	10	10	10	10
(4) Sediment Deposition	11	11	11	11
(5) Channel Flow Status	10	10	10	15
(6) Channel Alteration	11	11	11	10
(7) Frequency of Riffles (or Bends)	16	16	16	18
(8a) Bank Stability (right bank)	4	4	4	4
(8b) Bank Stability (left bank)	4	4	4	7
(9a) Vegetative Protection (right bank)	7	7	7	5
(9b) Vegetative Protection (left bank)	7	7	7	6
(10a) Riparian Zone Width (right bank)	1	1	1	3
(10b) Riparian Zone Width (left zone)	1	1	1	3
Total	**106**	**106**	**106**	**121**

Table 4.11 Habitat metrics at station QVB for the four sampling dates

QVB Metrics	Feb. 20, 1997	Apr. 29, 1997	Aug. 29, 1997	Nov. 11, 1999
(1) Epifaunal Substrate / Available Cover	8	6	6	1
(2) Embeddedness	8	6	6	1
(3) Velocity/Depth Regime	5	2	2	6
(4) Sediment Deposition	11	11	11	4
(5) Channel Flow Status	5	2	2	13
(6) Channel Alteration	9	9	9	16
(7) Frequency of Riffles (or Bends)	16	16	16	10
(8a) Bank Stability (right bank)	7	7	7	2
(8b) Bank Stability (left bank)	7	7	7	3
(9a) Vegetative Protection (right bank)	8	8	8	5
(9b) Vegetative Protection (left bank)	8	8	8	5
(10a) Riparian Zone Width (right bank)	5	5	5	9
(10b) Riparian Zone Width (left zone)	5	5	5	4
Total	**102**	**92**	**92**	**79**

Table 4.12 Habitat metrics at station QVC for the four sampling dates

QVC Metrics	**Feb. 20, 1997**	**Apr. 29, 1997**	**Aug. 29, 1997**	**Nov. 11, 1999**
(1) Epifaunal Substrate / Available Cover	8	6	6	7
(2) Embeddedness	8	6	6	7
(3) Velocity/Depth Regime	5	2	2	11
(4) Sediment Deposition	11	11	11	6
(5) Channel Flow Status	5	2	2	15
(6) Channel Alteration	9	9	9	14
(7) Frequency of Riffles (or Bends)	16	16	16	10
(8a) Bank Stability (right bank)	7	7	7	3
(8b) Bank Stability (left bank)	7	7	7	4
(9a) Vegetative Protection (right bank)	8	8	8	5
(9b) Vegetative Protection (left bank)	8	8	8	6
(10a) Riparian Zone Width (right bank)	5	5	5	9
(10b) Riparian Zone Width (left zone)	5	5	5	2
Total	**102**	**92**	**92**	**99**

Table 4.13 Habitat metrics at station QVF for the four sampling dates

QVF Metrics	Feb. 20, 1997	Apr. 29, 1997	Aug. 29, 1997	Nov. 11, 1999
(1) Epifaunal Substrate / Available Cover	11	10	10	11
(2) Embeddedness	11	11	11	8
(3) Velocity/Depth Regime	7	5	5	8
(4) Sediment Deposition	11	11	11	6
(5) Channel Flow Status	6	3	3	17
(6) Channel Alteration	11	11	11	19
(7) Frequency of Riffles (or Bends)	16	16	16	16
(8a) Bank Stability (right bank)	7	7	7	7
(8b) Bank Stability (left bank)	7	7	7	8
(9a) Vegetative Protection (right bank)	8	8	8	6
(9b) Vegetative Protection (left bank)	8	8	8	7
(10a) Riparian Zone Width (right bank)	5	5	5	9
(10b) Riparian Zone Width (left zone)	5	5	5	3
Total	**113**	**107**	**107**	**125**

Table 4.14 Habitat assessment summary: % similarity to reference conditions*

Sampling Date	QVA	QVB	QVC	QVF	Ridge & Valley Undisturbed
Feb. 20, 1997	66	64	64	71	>160
Apr. 29, 1997	66	58	58	67	>160
Aug. 29, 1997	66	58	58	67	>160
Nov. 11, 1999	76	49	62	78	>160

* Calculated as the total of the metrics measured at a given site (Tables 4.12 through 4.15) divided by the reference habitat total (160)

Table 4.15 Macroinvertebrate taxa and abundance at sampling location QVA

QVA Taxa	Feb. 20, 1997	April 29, 1997	Aug. 29, 1997	Apr. 29, 1999	Aug. 8, 1999	Sept. 28, 1999
Ephemeroptera (Mayflies)						
Baetidae						
Ephemerellidae						
Isonychiidae						
Trichoptera (Caddisflies)						
Hydropsychidae	2					
Hydroptilidae						
Plecoptera (Stoneflies)						
Diptera (True Flies)						
Chironomidae	146	142	20	243	129	206
Dixidae						
Empididae						
Hemerodromia						
Muscidae		3	6		6	13
Simuliidae	3	1				
Stratiomyiidae						
Tabanidae						
Tipulidae						
Odonata (Damsel-, Dragonflies)						
Aeshnidae						
Calopterygidae						
Coenagrionidae						
Hemiptera (True Bugs)						
Corixidae						
Gerridae						
Notonectidae						
Veliidae						
Coleoptera (Beetles)						
Dytiscidae		5	3		18	1
Elmidae	2	1	1			
Haliplidae						
Hydrophilidae						
Other Invertebrates						
Asellidae			2		11	
Cambaridae		1				
Collembola	1	1				
Corbiculidae						
Corduliidae						
Gammaridae						
Hirudinea						
Lumbriculidae					9	6
Oligochaeta	16	29	15			
Physidae						
Planariidae			7		1	
Planorbidae						
Pleuroceridae						

Table 4.16 Macroinvertebrate taxa and abundance at sampling location QVB

QVB Taxa	Feb. 20, 1997	April 29, 1997	Aug. 29, 1997	Apr. 29, 1999	Aug. 8, 1999	Sept. 28, 1999
Ephemeroptera (Mayflies)						
Baetidae			10			4
Ephemerellidae			8			
Isonychiidae						
Trichoptera (Caddisflies)						
Hydropsychidae				1		
Hydroptilidae						
Plecoptera (Stoneflies)						
Diptera (True Flies)						
Chironomidae	180	148	46	94	5	113
Dixidae						
Empididae						
Hemerodromia				1		
Muscidae						1
Simuliidae	10	19	5	103	2	9
Stratiomyiidae						
Tabanidae			1			
Tipulidae				10	3	
Odonata (Damsel-, Dragonflies)						
Aeshnidae						
Calopterygidae						
Coenagrionidae			5	9		7
Hemiptera (True Bugs)						
Corixidae						
Gerridae			4			
Notonectidae						
Veliidae			2			
Coleoptera (Beetles)						
Dytiscidae	1	3	4	3		1
Elmidae			1	3	7	
Haliplidae						3
Hydrophilidae						
Other Invertebrates						
Asellidae	1	2			1	
Cambaridae			17			
Collembola			1			
Corbiculidae						
Corduliidae						1
Gammaridae						
Hirudinea					1	7
Lumbriculidae				2		4
Oligochaeta	8	24	7			
Physidae					3	2
Planariidae			15		14	9
Planorbidae					163	3
Pleuroceridae			6	2		

Table 4.17 Macroinvertebrate taxa and abundance at sampling location QVC

QVC Taxa	Feb. 20, 1997	April 29, 1997	Aug. 29, 1997	Apr. 29, 1999	Aug. 8, 1999	Sept. 28, 1999
Ephemeroptera (Mayflies)						
Baetidae		1	1	7	32	13
Ephemerellidae			1			
Isonychiidae						
Trichoptera (Caddisflies)						
Hydropsychidae			1	73	13	9
Hydroptilidae						
Plecoptera (Stoneflies)						
Diptera (True Flies)						
Chironomidae	144	150	75	154	159	155
Dixidae						
Empididae						
Hemerodromia				1		
Muscidae						
Simuliidae	7	23	2	6	3	3
Stratiomyiidae						1
Tabanidae						
Tipulidae	1					
Odonata (Damsel-, Dragonflies)						
Aeshnidae					1	
Calopterygidae					1	1
Coenagrionidae					4	14
Hemiptera (True Bugs)						
Corixidae			12			7
Gerridae			1			
Notonectidae						
Veliidae			1			
Coleoptera (Beetles)						
Dytiscidae	1	1	2	1	3	
Elmidae	2			1	5	14
Haliplidae						
Hydrophilidae					2	1
Other Invertebrates						
Asellidae	19	6				
Cambaridae	6		11			
Collembola						
Corbiculidae		1				
Corduliidae						
Gammaridae			1			
Hirudinea						
Lumbriculidae						
Oligochaeta	13	28	17			
Physidae						
Planariidae			11			
Planorbidae						
Pleuroceridae				1		

Table 4.18 Macroinvertebrate taxa and abundance at sampling location QVF

QVF Taxa	Feb. 20, 1997	April 29, 1997	Aug. 29, 1997	Apr. 29, 1999	Aug. 8, 1999	Sept. 28, 1999
Ephemeroptera (Mayflies)						
Baetidae			9	5	7	5
Ephemerellidae			2			
Isonychiidae				1		
Trichoptera (Caddisflies)						
Hydropsychidae	1		28	30	79	91
Hydroptilidae		1	1	14		
Plecoptera (Stoneflies)						
Diptera (True Flies)						
Chironomidae	87	132	90	151	111	115
Dixidae			1			
Empididae				6		
Hemerodromia						
Muscidae		1	2			
Simuliidae	12	23	11	39	6	5
Stratiomyiidae						
Tabanidae						
Tipulidae	1		1	1	1	5
Odonata (Damsel-, Dragonflies)						
Aeshnidae						
Calopterygidae						
Coenagrionidae				2	1	6
Hemiptera (True Bugs)						
Corixidae						2
Gerridae						
Notonectidae			1			
Veliidae						
Coleoptera (Beetles)						
Dytiscidae	1	3		4	1	
Elmidae	2		1	3	17	7
Haliplidae						
Hydrophilidae						
Other Invertebrates						
Asellidae	32	15				1
Cambaridae	3		4	1	2	
Collembola						
Corbiculidae		2	1			
Corduliidae						
Gammaridae	41	37	19			
Hirudinea						
Lumbriculidae						
Oligochaeta		10				
Physidae						
Planariidae			12			
Planorbidae						
Pleuroceridae			21			

Table 4.19 Benthic macroinvertebrate metrics calculated for site QVA

QVA Metrics	**Feb. 20, 1997**	**April 29, 1997**	**Aug. 29, 1997**	**Apr. 29, 1999**	**Aug. 8, 1999**	**Sept. 28, 1999**
Taxa Richness	6	8	7	1	6	4
% 1 Dominant Taxon	85.9	77.6	37.0	100.0	74.1	91.2
% 5 Dominant Taxa	99.4	98.4	94.4	100.0	99.4	100.0
Modified HBI	6.2	6.3	7.1	6.0	6.3	6.2
% Haptobenthos	4.7	1.6	1.9	0.0	0.0	0.0
EPT Index	1	0	0	0	0	0
# Ephemeroptera	0	0	0	0	0	0
% Ephemeroptera	0.0	0.0	0.0	0.0	0.0	0.0
Simpson Diversity Index	0.25	0.37	0.77	0	0.43	0.17
# Intolerant Taxa	1	2	1	0	0	0
% Collector-Gatherers	95.3	93.4	81.5	100.0	86.2	93.8
% Collector-Filterers	2.9	0.5	0.0	0.00	0.0	0.0
% Shredders	0.0	0.0	0.0	0.0	0.0	0.0
% Scrapers	1.2	0.5	1.9	0.0	0.0	0.0
Mutimetric Index (MAIS) Categorization Evaluation						
MAIS Score	2	3	3	0	1	0
Biological Condition Category	Very Poor	Very Poor	Very Poor	Very Poor	Very Poor	Very Poor

Table 4.20 Benthic macroinvertebrate metrics calculated for site QVB

QVB Metrics	**Feb. 20, 1997**	**April 29, 1997**	**Aug. 29, 1997**	**Apr. 29, 1999**	**Aug. 8, 1999**	**Sept. 28, 1999**
Taxa Richness	5	5	15	10	9	13
% 1 Dominant Taxon	90.0	75.5	34.8	45.2	81.5	68.9
% 5 Dominant Taxa	100.0	100.0	72.7	96.1	96.0	88.4
Modified HBI	6.1	6.3	5.9	6.0	6.9	6.5
% Haptobenthos	5.0	9.7	27.3	52.2	4.5	12.2
EPT Index	0	0	2	1	0	1
# Ephemeroptera	0	0	18	0	0	4
% Ephemeroptera	0.0	0.0	13.6	0.0	0.0	2.4
Simpson Diversity Index	0.19	0.41	0.84	0.62	0.33	0.52
# Intolerant Taxa	0	0	5	3	2	1
% Collector-Gatherers	94.5	88.8	65.2	42.1	93.0	82.3
% Collector-Filterers	5.0	9.7	3.8	45.6	1.0	5.5
% Shredders	0.0	0.0	0.0	4.4	1.5	0.0
% Scrapers	0.0	0.0	5.3	2.2	3.5	0.0
Mutimetric Index (MAIS) Categorization Evaluation						
MAIS Score	0	0	9	4	3	4
Biological Condition Category	Very Poor	Very Poor	Poor	Very Poor	Very Poor	Very Poor

Table 4.21 Benthic macroinvertebrate metrics calculated for site QVC

QVC Metrics	**Feb. 20, 1997**	**April 29, 1997**	**Aug. 29, 1997**	**Apr. 29, 1999**	**Aug. 8, 1999**	**Sept. 28, 1999**
Taxa Richness	8	7	13	8	10	10
% 1 Dominant Taxon	74.6	71.4	55.1	63.1	71.3	71.1
% 5 Dominant Taxa	97.9	99.0	92.6	98.8	95.5	94.0
Modified HBI	6.3	6.3	6.6	5.9	5.9	6.1
% Haptobenthos	4.7	11.4	4.4	36.5	26.0	24.3
EPT Index	0	1	3	2	2	2
# Ephemeroptera	0	1	2	7	32	13
% Ephemeroptera	0.0	0.5	1.5	2.9	14.3	6.0
Simpson Diversity Index	0.43	0.46	0.66	0.51	0.47	0.48
# Intolerant Taxa	3	1	3	3	3	2
% Collector-Gatherers	91.2	88.1	77.9	66.0	85.7	77.5
% Collector-Filterers	3.6	11.4	2.2	32.4	7.2	5.5
% Shredders	0.5	0.0	0.0	0.0	0.0	0.0
% Scrapers	1.0	0.0	0.0	0.8	2.2	6.4
Mutimetric Index (MAIS) Categorization Evaluation						
MAIS Score	3	3	6	6	6	6
Biological Condition Category	Very Poor	Very Poor	Very Poor	Very Poor	Very Poor	Very Poor

Table 4.22 Benthic macroinvertebrate metrics calculated for site QVF

QVF Metrics	**Feb. 20, 1997**	**April 29, 1997**	**Aug. 29, 1997**	**Apr. 29, 1999**	**Aug. 8, 1999**	**Sept. 28, 1999**
Taxa Richness	9	9	16	12	9	9
% 1 Dominant Taxon	48.3	58.9	44.1	58.8	49.3	48.5
% 5 Dominant Taxa	97.2	96.9	83.3	93.4	97.8	94.5
Modified HBI	6.3	6.2	5.6	6.0	5.8	6.0
% Haptobenthos	31.1	27.2	45.6	38.9	48.9	48.1
EPT Index	1	1	4	4	2	2
# Ephemeroptera	0	0	11	6	7	5
% Ephemeroptera	0.0	0.0	5.4	2.33	3.1	2.1
Simpson Diversity Index	0.68	0.61	0.76	0.62	0.63	0.62
# Intolerant Taxa	3	0	7	5	4	3
% Collector-Gatherers	88.9	86.6	65.2	60.7	52.4	51.1
% Collector-Filterers	7.2	11.2	19.6	27.2	37.8	40.5
% Shredders	0.6	0.0	0.5	0.4	0.4	2.1
% Scrapers	1.1	0.0	10.8	1.2	7.6	3.0
Mutimetric Index (MAIS) Categorization Evaluation						
MAIS Score	4	1	9	7	6	6
Biological Condition Category	Very Poor	Very Poor	Poor	Poor	Very Poor	Very Poor

Table 4.23 MAIS scores for the sampling locations on the sampling dates

MAIS Score	QVA	QVB	QVC	QVF
Feb. 20, 1997	2	0	3	5
Apr. 29, 1997	3	0	3	1
Aug. 29, 1997	3	9	6	9
Apr. 29, 1999	0	4	6	7
Aug. 8, 1999	1	3	6	6
Sept. 28, 1999	0	4	6	6
Average 1997	2.7	3.0	4.0	5.0
Average 1999	0.3	3.7	6.0	6.3
Average Overall	1.5	3.3	5.0	5.7

The MAIS score for undisturbed similar-sized streams in the Central Appalachian Ridges and Valleys ecoregion averages 15 and ranges from 12-17 (Kibler et al., 1998). When comparing the minimum in this range (12) to the highest score in this study stream (9: Table 4.23) (with an average even lower for each site), it is clear that the biotic conditions of each of the sites are seriously impaired. Comparing the 1997 and 1999 MAIS averages for each site, biological condition appears to decline above the wet pond but improve slightly at the three stations downstream of the wet pond. The overall average also shows better biotic condition downstream of the wet pond. The MAIS scores improve for the downstream sites beginning with the Aug. 29, 1997 sample, which is the first sample taken after the wet pond construction was completed. However, if using the MAIS scores as biocriteria, the results for all sampling locations on all sampling dates would be considered unacceptable (Table 3.6). The biological conditions ranked as very poor (MAIS score ≤ 6) for all but two of the samples that, with a score of 9, still indicate poor (MAIS score 7-12) biological condition.

The values for taxa richness are below the minimum value expected, based on local reference data that had an average taxa richness score of 23 and ranged from 18 to 34 (Kibler et al., 1998). Only sites QVC (Table 4.21) and QVF (Table 4.22) on August 29, 1997 showed taxa richness even approaching the 17 families seen by Evans (1997) for reference conditions in the valley/plateau region of western Virginia. Furthermore, the total number of taxa seen throughout the stream reach (the number of taxa presented in Tables 4.17 through 4.20) was very low (38) compared to similar size streams in the Central Appalachian Ridges and Valleys ecoregion which usually have two or three times as many taxa (Kibler et al., 1998).

Very few of the EPT taxa, often sensitive to pollution and environmental stress, are seen in the stream at the study sites. EPT organisms were seen more frequently at stations QVC & QVF, and are practically nonexistent at site QVA. Stations QVB, QVC, and QVF all show an increase in metric values with respect to EPT taxa (EPT index, # Ephemeroptera, and % Ephemeroptera) starting with the Aug. 29, 1997 sample, which was the first to be collected after completion of the stormwater management facility. However the range of values for the EPT index metric for this stream reach (0-4) is far below those seen in the reference studies: 11.95±2.23 from Evans (1997) and 9-18 from Kibler et al (1998). Furthermore, of all the sites that had Trichoptera (caddisflies), only QVF had a family from this order other than Hydropsychidae, a fairly pollution tolerant family. Therefore, it is worth examining the # Ephemeroptera and % Ephemeroptera metrics. No Ephemeroptera were seen at QVA, but their numbers increased downstream after the wet pond was completed. Only the QVB summer 1997 samples and the QVC summer 1999 samples showed values for the % Ephemeroptera metric that fell within the range (though below the mean) for the metric as seen in the Kibler et al. (1998) reference samples.

Several patterns emerge when examining taxa that are tolerant or facultative with regard to pollution and environmental stress. First, the Simpson's Diversity Index (SDI) (Tables 4.21 to 4.24) demonstrates significant impairment of the benthic assemblage composition. The SDI generally increases from upstream to downstream sites but only one sample at each site had a SDI value that fell within the range reported for reference sites by Kibler et al. (1998). Second, the # Intolerant Taxa metric (Tables 4.21 to 4.24) also demonstrates impairment of the assemblage composition. The number of intolerant taxa observed at site QVA was 0 for all dates in 1999, though a few were seen in 1997; therefore poor water quality conditions (including sediment and/or toxics) appear to have caused this decline at QVA. Sites QVB and QVF show a general increase in the number of intolerant taxa for 1999 samples as compared to 1997, while site QVC does not change much over time (except perhaps seasonally). However, all sites show significant impairment (0-7 range for all sites combined), compared to reference conditions where the number of intolerant taxa ranges from 14 to 24 (Kibler et al., 1998). The third pattern examined involved the metrics % 1 Dominant Taxon and % 5 Dominant Taxa. These metrics demonstrated little balance in species composition among most of the samples taken in the stream. The values for the metric % 5 Dominant Taxa (Tables 4.21 to 4.24) were above the highest range seen in the reference samples (91%) except for two samples at QVB (summer 1999 and fall 1999) and one sample at QVF (summer 1997). Except for two samples at QVB (where Simuliidae (spring 1999) and Planorbidae (summer 1999) dominated), Chironomidae (midges) was the dominant taxon for all sites on all dates. Chironomids are tolerant of sedimentation, saprobic conditions, and low oxygen (Dr. Voshell, personal communication, 1998; Rosenberg and Resh, 1993). The order Diptera (including the families Chironomidae and Simuliidae) is also resistant to metal pollution (Hellawell, 1986; Clements, 1994).

The modified HBI metric does not appear to detect impairment along this stream, as almost all values calculated for this stream were above the values seen in the reference streams. This could be an artifact of the calculation due to its sensitivity to taxa richness and the limited taxa richness observed within these samples. Furthermore, taxa richness was calculated at the family level, and had the samples been identified to genus or species taxonomic level, the range of the modified HBI metric might have increased.

There are many potential causes of the benthic macroinvertebrate assemblage degradation at the study site. Sedimentation of the stream may be the primary impediment to stream health improvement. Two things need to occur prior to stream biota recovery. First, the wet pond needs to operate as designed, so that (1) no further erosion of the embankment occurs and (2) no further erosion of the streambanks and channel downstream of the wet pond should occur (the flow frequency must be managed effectively). Second, previously deposited sediment needs to be washed downstream to allow macroinvertebrate population recovery.

Metal pollution may also contribute to benthic macroinvertebrate depopulation and assemblage degradation. Copper is considered the most toxic metal to aquatic organisms (Yousef et al., 1996; Marsalek, 1997). Median copper concentrations exceeded both the acute and chronic water quality criteria at all sampling sites in 1997, 1999, and for the overall sampling record (Figure 4.10). Only in 1998 did there appear to be any improvement in copper concentrations below the wet pond (Figure 4.10). Furthermore, median lead concentrations exceeded the chronic water quality criterion every year at all sampling locations (Figure 4.13). Despite apparent sedimentation of zinc within the wet pond, median zinc concentrations exceeded both the acute and chronic water quality criteria every year at all sampling locations. If the degradation is a result of metal pollution, according to definitions by Hellawell (1986) detailed in Chapter 2, QVA has moderate metal pollution (no Ephemeroptera families represented) while all other sites are mildly polluted by metal contamination because they have more Ephemeroptera families. None of the sites had any organisms in the order Plecoptera, which has been noted to decline at sites with high concentration of metals (Clements, 1994). However, metal pollution cannot be said to be the only cause of the impairment of this stream. There are many other conditions within the stream, as well as other pollutants, that could influence assemblage composition and health.

Available food sources within the stream are impacted by urbanization activities, and effects can be seen when examining the feeding strategies (or trophic position) represented in the macroinvertebrate assemblage. The first metric examined here is % Scrapers (Tables 4.21 to 4.24). Eutrophication can lead to dense algal mats on the stream substrate that prevent the attachment of clingers and scrapers and prevent formation of thin surface algal films that the scrapers use as food (Dr. Voshell, personal communication, 1998). The expected value for this metric is 25% (range of 2-69) from Kibler et al. (1998) and 22.81 ± 5.91% from Evans (1997). In 1997, QVA values for this metric fell below the range presented by Kibler et al. (1998) in 1997, and scrapers disappeared from the site altogether in 1999 (0 for all three dates). Slight increases were seen in this metric at sites QVB, QVC, and QVF, especially after wet pond installation, but the metric values remained far below the reference mean conditions. The second group examined was the shredders (Tables 4.21 to 4.24), who utilize coarse particulate organic matter (CPOM) in their feeding strategy. Upstream of site QVA, the stream is channeled underground, and no shredders were seen at that site perhaps due to the lack of CPOM sources or the lack of conditions adequate to promote the associated fungal and or bacterial growth on the CPOM consumed by shredders (Rosenberg and Resh 1993). Very few shredders were seen at either station QVB or QVC, but QVF had noticeably more shredders, probably due to the greater amount of riparian cover along the streambank downstream of the wet pond. The third feeding strategy examined was the use of fine particulate organic matter (FPOM) by collector-gatherers and collector-filterers (Tables 4.21 to 4.24). Evans (1997) saw an equal partition between these two groups, with a 41.89 ± 6.18% representation in the assemblage by collector-gatherers and a 32.16 ± 5.27% representation by collector-filterers. This balance is not typically seen in the samples collected at sites QVA through QVF. In general, collector-filterers increase from upstream to downstream sites (perhaps due to the wet pond release waters), but the majority of the macroinvertebrate assemblages at all sites (and all dates) have the collector-gatherer feeding strategy. While both groups utilize the same food source, the likely cause of the imbalance between these two groups is the substrate; collector-filterers tend to require higher quality habitat in terms of substrate to attach to. However, collector-gatherers tend to be mobile, scavenging the stream bottom for their food, and thus can be more tolerant of sedimentation or excessive algal growth. A greater balance between these two groups is achieved at site QVF where the habitat is better. Only QVF shows a consistent increase in %-Collector-filterers over time, from 7.2% in the winter 1997 sample to 40.5% in the fall 1999 sample.

In general, a number of benthic macroinvertebrate metrics improve downstream and may indicate initial recovery of the benthic macroinvertebrate assemblage due to BMP installation. The maximum EPT index value seen at each site increases downstream (QVA, 1; QVB, 2; QVC, 3; and QVF, 4). The abundance of Ephemeroptera noticeably increases at QVF after the wet pond was installed. The % Scrapers metric increases at stations QVB and QVF after wet pond installation, while its value decreases at QVA. This trend is also apparent for the % intolerant metric, which increases at stations QVB and QVF in the 1999 samples, and decreases at station QVA. The % collector-filterers metric shows a consistent increase over time at station QVF and a better balance of collector-gatherers and collector-filterers at all stations downstream of the wet pond. Though there appears to be some improvement downstream, it must be remembered that even the highest values for these metrics are far below reference conditions, and the multimetric index for this region (the MAIS) indicate very poor conditions everywhere along the stream.

CHAPTER 5.0 SUMMARY AND CONCLUSIONS

Monitoring of a regional stormwater management facility, located on the Virginia Tech campus in Blacksburg VA, was conducted in order to assess its efficacy in reducing nonpoint source pollutant losses to downstream waters. Both grab samples and automated samples were collected above and below each of the basins (wet pond and dry pond) of the stormwater management facility. Between 1997 and 1999, water quality grab samples included 35 baseflow samples and 22 stormflow samples. The grab samples were analyzed for concentrations of total suspended solids (TSS), metals, bacteria, and nutrients as well as temperature, pH, dissolved oxygen, conductivity, total organic carbon (TOC), and chemical oxygen demand (COD). Automated flow-weighted sampling was initiated in February of 1999 and continued through October 1999. Thirty-three storms in 1999 were monitored for both flow and event-mean concentrations (EMCs) of various water quality parameters (TSS, TOC, COD, and nutrients). Pollutant loads and pollutant removal efficiency estimates were calculated with regard to the wet pond, the dry pond, and the combined facility. Two types of removal efficiencies were calculated: (1) the event mean concentration (EMC) efficiency based on concentrations for individual storms, and (2) the Sum of Loads (SOL) removal efficiency based on mass of the pollutants removed by the facility over the entire study period. In addition, benthic macroinvertebrates were sampled and habitat conditions were assessed in 1997 and 1999. Furthermore, a preliminary investigation of pond characteristics was conducted including measurements of water quality and composition, sediment deposition and composition, and residence time.

5.1 Combined Stormwater Management Facility

The stormwater management facility as a whole appears to have very low pollutant removal efficiencies (all of the mass pollutant removal efficiencies were negative indicating overall export of pollutants rather than reductions) and thus does not markedly improve the water quality. Pollutant concentrations and loads both appear to increase downstream of the facility as compared to upstream, during both storm event and baseflow periods.

Monitoring results of the benthic assemblage showed evidence of moderate to high impairment at all sampling locations. The evidence included fewer taxa, lower taxa richness, lack of balance in the assemblage composition, and more pollutant tolerant and facultative macroinvertebrates at each of the monitored sites as compared to reference conditions. The nature of the impairment is unknown but may include multiple factors such as metals toxicity, eutrophication, pond export of seston and organic matter, lack of coarse particulate organic matter (CPOM) in the stream, and, most predominantly, sedimentation. The largest storm monitored in 1999 produced 61.2-mm of rainfall, much smaller than the 2-year design storm event of 76.2-mm. Therefore, very little removal of accumulated sediment within the stream channels occurred which is a prerequisite to habitat and biotic community improvements. The stormwater management facility may eventually improve conditions for aquatic organisms, but may fundamentally alter the community composition partly because of increased stream water temperatures caused by heating of the wet pond water.

The following conclusions could be made from the results for the stormwater management facility as a whole:

- The facility produced a TSS concentration removal of 10%; much lower than the 80 to 90% TSS removal expected for properly functioning stormwater management facilities, especially those with more than one treatment mechanism (Hartigan 1989).
- The facility did not reduce fecal coliform bacteria levels in the stream. Overall, the concentration removal efficiency for fecal coliform was –59%, possibly because of direct fecal deposition by ducks.
- Metals concentrations often exceeded their respective acute and chronic water quality criteria. Median copper concentrations consistently exceed both criteria at all sampling sites in 1997 and 1999. Median lead concentrations exceeded the chronic water quality criterion every year at all sampling locations, while median zinc concentrations exceeded both the acute and chronic water quality criteria.

5.2 Wet Pond

Pollutant removal efficiencies measured in the wet pond are atypical of those reported in the literature (Schueler, 1993). Insufficient residence time and wet pond embankment failure are likely the principal causes of the wet pond's and thus the overall facility's performance.

Based on the data collected from the wet pond, the following conclusions could be made:

- In general, the wet pond is undersized with respect to the watershed it serves. It has a wet pond surface area less than 1% of the watershed area (0.87 ha), as compared to the 3% ratio often recommended for optimal pollutant removal (Athanas, 1988). Removal of pollutants increases as detention basin size relative to the contributing catchment area increases from less than 1% to 7.5% (Cappuccitti, 1993). While it is not feasible to enlarge the pond, and the length to width ratio of 4.4 is optimal (Ellis, 1989; Hartigan, 1989), other steps can be taken to increase residence time within the pond from 2 days to a period closer to the optimal residence time of two weeks (Hartigan, 1989).

- The detention basin volume is adequate compared to the median of the storm runoff volumes observed in 1999 (ratio of 3.3) (Yousef and Wanielista, 1993), but no large storms were recorded and the 85-cm of rainfall that occurred in 1999 was below the thirty-year normal of 104 cm/yr. The wet pond volume of 3-4 mm of runoff from the watershed is not adequate compared to the desired runoff volume of 9 to 12-mm (Droste et al., 1993)).

- Concentrations of TSS and sediment-bound pollutants including COD and total phosphorus were especially high in 1999 at station QVC. These results likely are due to the erosion of the wet pond embankment.

- Despite the masking effects of sediment from the dam and erosion around the perimeter of the pond, some evidence of sedimentation within the wet pond was found. TSS removal efficiencies were low in the wet pond (19% for concentrations and 33% for loads) compared to the 80 to 90% expected for similar ponds, but several pollutants typically associated with TSS, and not likely to be associated with the fill material for the wet pond embankment, experienced reduction within the wet pond. First, zinc concentrations in sediment cores were highest in the upstream third of the pond where the majority of sedimentation occurs. Second, copper and zinc concentrations in 1998 were lower at the wet pond exit (station QVB) as compared to the wet pond entrance (station QVA); these results were statistically significant for copper. Third, the reduction in TOC concentrations by the wet pond was statistically significant (EMC efficiency 13%) and TOC loads were reduced by 12% (SOL efficiency).

- Evidence of eutrophication of the wet pond and its negative downstream impacts was found. Nitrate and ammonia experienced statistically significant reductions (19% and 8%, respectively in concentration; 27 and 41%, respectively in loads), likely due their biological transformation within the pond. The TKN concentrations and loads increased by 26% and 32%, respectively, downstream of the wet pond. Total organic carbon concentrations and loads also increased downstream which may be evidence of flushing algae during storm events. Furthermore, the concentration of nitrate required to promote algae (0.1 mg/L) (Mostaghimi et al., 1989) was consistently exceeded at all sampling locations.

- The shallow nature of the wet pond (0.6 to 1.1-m) allows for significant heating of the pond water, promotes eutrophication (including low dissolved oxygen concentrations at the pond outlet), and may allow for resuspension of deposited material due to the influence of wind or currents. A seasonal fluctuation in temperature was observed downstream of the wet pond. The 8°C increase seen at stations QVB, QVC, and QVD as compared to station QVA during the summer would stress sensitive aquatic species, especially because sustained summertime water temperatures are greater than 21°C (Schueler, 1987).

- The residence time of 2 days appears to be insufficient to reduce fecal coliform concentrations in the stream, and over 40% of the samples collected exceeded the water quality standard for contact recreation (DEQ-WQS, 1997). The concentration removal efficiency calculated for the wet pond was –46%. However, this residence time appears adequate to reduce most other pollutants during baseflow conditions with the exception of TSS (-29% concentration efficiency) and TOC (-44% concentration efficiency). The greatest reductions in baseflow concentrations were obtained for ammonia (67%), nitrate (57%), total nitrogen (54%), and COD (45%). These reductions are likely due to slower processes within the pond among aquatic biota, mineral and organic bottom materials, and the aquatic chemical substrate (Gain and Miller, 1989). These processes may include chelation, flocculation, biological uptake and release, biologically mediated oxidation and reduction, ion exchange, and dissolution from bed materials (Gain and Miller, 1989).

- The limited residence time also may be the primary reason why phosphorus levels within the wet pond are only reduced by 10% and 3% for orthophosphorus and total phosphorus respectively. Retention basins according to Virginia regulations should remove at least 40% of the incoming phosphorus (DCR, 1998b) due to removal mechanisms including adsorption on bottom materials or precipitation with iron and aluminum oxides (Gain and Miller, 1989).

The lack of removal may also be attributed to association with fine organic particulates or to uptake by free-floating aquatic organisms (Gain and Miller, 1989).

- Benthic macroinvertebrate metrics increased downstream of the wet pond in 1999 samples, possibly due to some beneficial influences of the wet pond, but the values observed in the stream were all low compared to reference conditions. The highest taxa richness metric (17) for the stream was sampled at stations QVC and QVF. (Station QVF, located downstream of the dry pond in a riffle habitat, was sampled rather than QVD to avoid the immediate impacts of the dry pond flow). The highest taxa richness was still below the range observed for reference conditions (18 to 34, with an average of 23 (Kibler et al., 1998)). Total taxa for the entire study site was 38 while more pristine reference sites often have 2-3 times that number (Kibler et al., 1998). The pollution sensitive species were typically absent at station QVA and increased downstream of the wet pond at stations QVB and QVF after construction. Despite the increase, the EPT index never exceeded 4 from the samples compared to the range of 9-18 for the reference sites (Kibler et al., 1998). Also, the metric for intolerant species ranged from 0 to 7 in this stream compared to the reference condition range of 14 to 24 (Kibler et al., 1998). In addition, the samples were heavily dominated by pollution tolerant species, typically Chironomidae. The trophic measures also demonstrated a lack of diversity and a tolerance of degraded conditions, with some improvement downstream, especially at station QVF.

- Habitat conditions below the wet pond were further degraded by the wet pond embankment failure as evidenced by the metrics associated with sedimentation and the overall matrix assessment scores. The conditions at station QVB declined from approximately 60% of reference conditions to 50% of reference conditions (Kibler et al., 1998). Conditions at station QVC did not change. However, habitat conditions appear to have improved slightly (from 60% to 70% of reference conditions) both above the wet pond and further downstream below the stream reach where primary sedimentation of the TSS from the wet pond embankment occurs.

5.3 Dry Pond

The dry pond did not remove pollutants during baseflow periods but there is some evidence of sedimentation within the dry pond during storm events for TSS and associated pollutants. In general, the dry pond was not effective in removing dissolved nutrient constituents:

- During storm events, the dry pond was effective in removing TSS concentrations, with a pollutant removal efficiency of 69% (EMC efficiency), and TSS loads by 43% (SOL Efficiency). Removal of TKN and total phosphorus concentrations (36% and 37%, respectively) within the dry pond is further evidence of sedimentation within the dry pond. Reductions in total phosphorus and TSS concentrations were equivalent or greater than results reported by Hodges (1997) for extended dry detention basins. Only the dry pond had statistically significant removal efficiencies for total phosphorus and orthophosphorus. It must be noted that dilution may have elevated the EMC removal efficiency results.
- Positive EMC removal efficiencies for ammonia and export of TOC may indicate some biotic influences within the dry pond (wetland vegetation uptake and senescence).
- The dry pond (-3% concentration pollutant removal efficiency) did not reduce fecal coliform concentrations.

CHAPTER 6.0 RECOMMENDATIONS FOR FURTHER STUDY

To better understand the impact of the stormwater management facility on the water quantity and quality within this tributary of Stroubles Creek and its downstream waters, monitoring efforts should continue after the wet pond embankment is repaired. Monitoring should continue beyond any stabilization period needed for a stormwater management facility to reach its potential for reducing nonpoint source pollutants. If biotic community improvement is desired, the stabilization period, at minimum, could be defined by the time necessary to flush out accumulated sediment within the channel based, on storm event characteristics. Sediment may have the most impact on stream biota by reducing prey capture for sight feeders, clogging gills, reducing spawning, and destroying the habitat potential of the stream bottom (Van Buren, 1994; Appelboom et al., 1998). Some water quality benefits which could have been observed during the stabilization period may have been negated or masked by the progressive erosion of the wet pond embankment

It is recommended that monitoring efforts be expanded to include the following:

1. Efforts to monitor the interactions of groundwater with the wet pond should be pursued. This would include measurements of water level fluctuations within the wet pond and further investigation into the reasons for the smaller flows measured at station QVB as compared to station QVA, and at station QVC as compared to station QVD.

2. Further information regarding wet pond water quality, sediment quality and sedimentation rates, and residence time should be collected to investigate pollutant removal processes and their influences upon those processes within the wet pond.

3. Flow-weighted sampling should also take place at station QVG since it appeared to have significant pollutant levels that would influence any determination of removal efficiencies for the dry pond and for the system as a whole.

4. In future studies of sediment deposition and composition in the wet pond, a segmented gravity corer (described by Aanderaa Instruments, Victoria, British Colombia, Canada) should be used, if possible, to obtain the core samples of fine-grained sediments. The core tube of the sampler is a series of rings that can rotate and cut 1-cm thick sediment layers from the core (Mudroch and Azcue, 1995). Freeze-drying is the preferred method of sample

storage for determination of most organic and inorganic pollutants (Mudroch and Azcue, 1995). Particle size analysis should be carried out on the wet sediment.

5. Future monitoring should include the analysis of macrophytes within the wet and dry ponds and periphyton within the two basins and in the stream. The macrophytes may influence the water quality of the wet pond and of the dry pond through their influence on sedimentation and by direct uptake and transformation of pollutants including nutrients and metals. The periphyton assemblages may be more sensitive to particular contaminants due to their increased generational turnover. In addition, the pathways of contaminant effect are different from that of benthic macroinvertebrates. As a result, periphyton metrics can improve the detection capabilities of rapid bioassessment approached to monitoring and performance evaluation (Barbour et al., 1995). If time and money were not an issue, more rigorous sampling and sample analysis techniques could be employed for benthic macroinvertebrates and/or periphyton.

6. Experimentation with floating baffles in the pond to increase the residence time of the pond is recommended to improve pollutant removal efficiencies within the pond and therefore, of the stormwater management facility as a whole. However, it is not recommended that the length: width ratio is increased beyond 10 (Thackston et al., 1987).

7. Substantial erosion of the streambanks within the channel downstream of the wet pond indicates that a velocity dissipater should be used at the pond outflow according to DCR regulations (DCR, 1998b). Furthermore, the frequency and duration of flows from the wet pond and the sensitivity of the downstream boundary material scour should be investigated in order to recommend management strategies that would reduce the erosion of the channel downstream (MacRae, 1996).

8. Streambank, stream channel, and riparian buffer restoration may be required to speed up the recovery of the habitat and aquatic organism assemblages within the stream.

9. Accurate information on the amount of impervious surfaces in the watershed should be quantified in order to compare the performance of this stormwater management facility to others in the literature.

APPENDIX A

From DEQ-WQS (1997) the equations for aquatic life criteria of metals in freshwater are as follows:

Acute criterion = WER exp {m_A[In(hardness)]+b_A
Chronic criterion = WER exp {m_C[In(hardness)]+b_C
where WER = water effect ratio

Parameters for certain metals:

Metal	m_A	b_A	m_C	B_C
Cadmium	1.128	-3.828	0.7852	-3.490
Copper	0.9422	-1.464	0.8545	-1.465
Lead	1.273	-1.084	1.273	-3.259
Zinc	0.8473	0.8304	0.8473	0.7614

Results of selected samples analyzed for hardness:

Sampling Location	**Baseflow Hardness (mg $CaCO_3$/L)**		**Storm Hardness (mg $CaCO_3$/L)**		**Overall Average**
	9/8/1999	10/19/1999	9/21/1999	10/5/1999	
QVA	212.83	246.04	120.32	165.92	186.28
QVB	211.08	200.38	145.36	221.38	194.55
QVC	205.36	202.08	142.87	223.33	193.41
QVD	229.47	218.9	146.43	233.04	206.96

Application of the acute and chronic criterion equations according to average stream hardness:

Metal	**WER**	**Hardness: Stream Avg. (mg CaCo3/L)**	**Acute Criterion (µg/L)**	**Chronic Criterion (µg/L)**
Cadmium	1	195.3	0.29	0.18
Copper	1	195.3	2.00	1.64
Lead	1	195.3	6.25	0.71
Zinc	1	195.3	15.98	14.91

The acute ammonia criteria for freshwater are as follows (DEQ-WQS, 1997):

Total Ammonia (mg/L)
Temperature (°C)

pH	0 C	5 C	10 C	15 C	20 C	25 C	30 C
6.5	32	33	31	30	29	29	29
6.75	32	30	28	27	27	26	26
7	28	26	25	24	23	23	23
7.25	23	22	20	19.7	19.2	19	19
7.5	17.4	16.3	15.5	14.9	14.6	14.5	14.5
7.75	12.2	11.4	10.9	10.5	10.3	10.2	10.3
8	8	7.5	7.1	6.9	6.8	6.8	7
8.25	4.5	4.2	4.1	4	3.9	4	4.1
8.5	2.6	2.4	2.3	2.3	2.3	2.4	2.6
8.75	1.47	1.4	1.37	1.38	1.42	1.52	1.66
9	0.86	0.83	0.83	0.86	0.91	1.01	1.16

The chronic ammonia criteria for freshwater are as follows (DEQ-WQS, 1997):

Total Ammonia (mg/L)
Temperature (°C)

pH	0 C	5 C	10 C	15 C	20 C	25 C	30 C
6.5	3.02	2.82	2.66	2.59	2.53	2.5	2.5
6.75	3.02	2.82	2.66	2.59	2.53	2.5	2.5
7	3.02	2.82	2.66	2.59	2.53	2.5	2.5
7.25	3.02	2.82	2.66	2.59	2.53	2.5	2.5
7.5	3.02	2.82	2.66	2.59	2.53	2.5	2.5
7.75	2.8	2.6	2.47	2.38	2.35	2.3	2.4
8	1.82	1.71	1.62	1.57	1.55	1.56	1.59
8.25	1.03	.97	.93	.91	.9	.91	.95
8.5	.58	.55	.53	.53	.53	.55	.58
8.75	.34	.32	.31	.31	.32	.35	.38
9	.2	.19	.19	.2	.21	.23	.27

APPENDIX B

Results of the RVN Ratio and Sign tests

RVN ratio test: alpha =0.1, Ho: independent
if p<0.1 reject Ho; for exact p-values Bartels 1982 JASA v 77, pp42-44
Test of randomness:

$$\frac{N^{+}-N^{-}}{\sqrt{N^{+}+N^{-}}} \sim N(0,1)$$

Sign Test, modified for ties (Putter 1955)
Ho: (p+)<=(p-)
Hi: (p+)>(p-)

		Sample size	RVN ratio test stat	Num of RVNRTS	Std. RVNRTS	Approx. p-value	If p>0.1, run sign test		+	-	Sign test stat	Approx p-value
BQVTemp	QVA-QVB	33	0.5475	1637	-4.2371	0.00						
BQVTemp	QVC-QVD	33	0.556	1981.25	-4.3341	0.00						
BQVTemp	QVA-QVD	33	0.5712	1708.5	-4.1679	0.00						
BQVpH	QVA-QVB	35	1.9824	7037.5	-0.0528	0.48	BQVpH	QVA-QVB	9	21	-2.19	0.014
BQVpH	QVC-QVD	35	1.4127	5025.75	-1.7627	0.04						
BQVpH	QVA-QVD	35	1.5825	5636.75	-1.2532	0.11	BQVpH	QVA-QVD	18	14	0.71	0.76
BQVCond	QVA-QVB	35	1.7012	6040	-0.8969	0.18	**BQVCond**	**QVA-QVB**	23	10	2.26	**0.99**
BQVCond	QVC-QVD	35	2.0199	7057.5	0.0597	0.52	BQVCond	QVC-QVD	2	31	-5.05	<0.001
BQVCond	QVA-QVD	35	1.5789	5601	-1.264	0.10	BQVCond	QVA-QVD	16	17	-0.17	0.43
BQVDO	QVA-QVB	34	1.8273	5733	-0.5113	0.30	BQVDO	QVA-QVB	8	16	-1.63	0.05
BQVDO	QVC-QVD	34	1.8313	5601	-0.4993	0.31	BQVDO	QVC-QVD	13	8	1.09	0.86
BQVDO	QVA-QVD	34	1.5913	5102.5	-1.2096	0.11	BQVDO	QVA-QVD	19	11	1.46	0.93
BQVTSS	QVA-QVB	35	2.0629	7318	0.1887	0.57	BQVTSS	QVA-QVB	9	23	-2.47	0.007
BQVTSS	QVC-QVD	35	1.8879	6697.5	-0.3363	0.37	**BQVTSS**	**QVC-QVD**	23	12	1.86	**0.97**
BQVTSS	QVA-QVD	35	1.423	5070.75	-1.7319	0.04						
BQVTOC	QVA-QVB	35	1.3808	4928	-1.8586	0.03						
BQVTOC	QVC-QVD	35	1.7854	6373	-0.6441	0.26	BQVTOC	QVC-QVD	17	18	-0.17	0.43
BQVTOC	QVA-QVD	35	1.3552	4834	-1.9353	0.03						
BQVCOD	QVA-QVB	35	1.8204	6470.75	-0.5389	0.30	BQVCOD	QVA-QVB	13	18	-0.90	0.18
BQVCOD	QVC-QVD	35	2.2605	8036	0.7818	0.78	BQVCOD	QVC-QVD	16	16	0.00	0.50
BQVCOD	QVA-QVD	35	1.8235	6490.75	-0.5298	0.30	BQVCOD	QVA-QVD	12	19	-1.26	0.10
BQVTotP	QVA-QVB	35	1.5152	4891	-1.4552	0.07						
BQVTotP	QVC-QVD	35	2.7518	9189.5	2.2564	0.99	BQVTotP	QVC-QVD	5	16	-2.40	0.008
BQVTotP	QVA-QVD	35	2.1828	7291.5	0.5485	0.71	BQVTotP	QVA-QVD	3	18	-3.27	<0.001
BQVPO4	QVA-QVB	33	1.8024	3795.75	-0.5766	0.28	BQVPO4	QVA-QVB	7	4	0.90	0.82
BQVPO4	QVC-QVD	34	1.3803	3845.5	-1.8341	0.03						
BQVPO4	QVA-QVD	34	2.1432	5973	0.4237	0.66	BQVPO4	QVA-QVD	1	15	-3.50	<0.001
BQVFTotP	QVA-QVB	35	2.1145	6138.5	0.3438	0.63	BQVFTotP	QVA-QVB	6	9	-0.77	0.22
BQVFTotP	QVC-QVD	35	1.3552	4178	-1.9354	0.03						
BQVFTotP	QVA-QVD	35	2.0911	6449	0.2735	0.61	BQVFTotP	QVA-QVD	4	13	-2.18	0.015
BQVNO3	QVA-QVB	35	0.8907	3178.75	-3.3296	0.00						
BQVNO3	QVC-QVD	35	1.513	5400	-1.4616	0.07						
BQVNO3	QVA-QVD	35	0.7769	2773	-3.6712	0.00						
BQVNH4	QVA-QVB	35	1.617	5319	-1.1497	0.13	BQVNH4	QVA-QVB	6	14	-1.79	0.04
BQVNH4	QVC-QVD	35	1.8337	6281.25	-0.4992	0.31	BQVNH4	QVC-QVD	10	13	-0.63	0.26
BQVNH4	QVA-QVD	35	1.543	5225.5	-1.3715	0.09						
BQVTKN	QVA-QVB	33	2.151	6303.5	0.4405	0.67	BQVTKN	QVA-QVB	9	15	-1.22	0.11
BQVTKN	QVC-QVD	33	1.9649	5660.75	-0.1025	0.46	BQVTKN	QVC-QVD	14	8	1.28	0.90
BQVTKN	QVA-QVD	33	2.2853	6797.5	0.8321	0.80	BQVTKN	QVA-QVD	8	19	-2.12	0.02
BQVFLTKN	QVA-QVB	33	1.4685	3981.75	-1.5505	0.06						
BQVFLTKN	QVC-QVD	32	2.2328	5466	0.6692	0.75	BQVFLTKN	QVC-QVD	10	7	0.73	0.77
BQVFLTKN	QVA-QVD	32	2.0433	5109.25	0.1244	0.55	BQVFLTKN	QVA-QVD	7	11	-0.94	0.17
BQVTotN	QVA-QVB	33	1.3115	3924	-2.0084	0.02						
BQVTotN	QVC-QVD	33	2.3172	6933	0.9252	0.82	BQVTotN	QVC-QVD	3	29	-4.60	<0.001
BQVTotN	QVA-QVD	33	1.9422	5811	-0.1687	0.43	BQVTotN	QVA-QVD	19	14	0.87	0.81
BQVTotCol	QVA-QVB	35	1.0661	3806	-2.803	0.00						
BQVTotCol	QVC-QVD	35	1.6191	5772	-1.1433	0.13	BQVTotCol	QVC-QVD	14	20	-1.03	0.15
BQVTotCol	QVA-QVD	35	1.3953	4979.75	-1.815	0.03						
BQVFecCol	QVA-QVB	35	1.3625	4862.25	-1.9133	0.03						
BQVFecCol	QVC-QVD	35	1.3117	4677	-2.0658	0.02						
BQVFecCol	QVA-QVD	35	1.6834	6009	-0.9502	0.17	BQVFecCol	QVA-QVD	12	22	-1.72	0.04
BQVFecStrep	QVA-QVB	35	1.6319	5826	-1.1047	0.13	BQVFecStrep	QVA-QVB	19	15	0.69	0.75
BQVFecStrep	QVC-QVD	35	1.3288	4742	-2.0144	0.02						
BQVFecStrep	QVA-QVD	35	1.5218	5433	-1.4351	0.08						
BQVCd	QVA-QVB	17	2.3094	845.25	0.6578	0.74	BQVCd	QVA-QVB	6	3	1.00	0.84
BQVCd	QVC-QVD	17	1.6967	621	-0.6448	0.26	BQVCd	QVC-QVD	5	4	0.33	0.63
BQVCd	QVA-QVD	17	2.5771	978	1.2269	0.89	BQVCd	QVA-QVD	4	6	-0.63	0.26
BQVCu	QVA-QVB	17	1.889	766	-0.2359	0.41	BQVCu	QVA-QVB	6	9	-0.77	0.22
BQVCu	QVC-QVD	17	1.307	530	-1.4733	0.07						
BQVCu	QVA-QVD	17	2.4088	976.75	0.869	0.81	BQVCu	QVA-QVD	7	7	0.00	0.50
BQVPb	QVA-QVB	17	2.3337	940.5	0.7095	0.76	BQVPb	QVA-QVB	4	9	-1.39	0.08
BQVPb	QVC-QVD	17	2.5509	1014	1.1713	0.88	BQVPb	QVC-QVD	8	4	1.15	0.87
BQVPb	QVA-QVD	17	2.4895	1004.5	1.0406	0.85	BQVPb	QVA-QVD	4	10	-1.60	0.05
BQVZn	QVA-QVB	17	2.3386	808	0.7199	0.76	BQVZn	QVA-QVB	3	7	-1.26	0.10
BQVZn	QVC-QVD	17	2.8583	1069	1.8247	0.97	BQVZn	QVC-QVD	5	7	-0.58	0.28
BQVZn	QVA-QVD	17	2.7957	1033	1.6916	0.95	BQVZn	QVA-QVD	4	10	-1.60	0.05

RVN ratio test: alpha =0.1, Ho: independent

if p<0.1 reject Ho; for exact p-values Bartels 1982 JASA v 77, pp42-44
Test of randomness:

			Sample size	**RVN ratio test stat**	**Num of RVNRTS**	**Std. RVNRTS**	**Approx. p-value**	**If p>0.1, run sign test**		**+**	**-**	**Sign test**	**Approx p-value**
BQVTemp	QVA-QVB	Seasonal	12	2.0979	300	0.1773	0.57	BQVSeasTemp	QVA-QVB	4	7	-0.90	0.18
BQVTemp	QVC-QVD	Seasonal	12	1.7263	246	-0.4957	0.31	BQVSeasTemp	QVC-QVD	7	5	0.58	0.72
BQVTemp	QVA-QVD	Seasonal	12	2.2238	318	0.4053	0.66	BQVSeasTemp	QVA-QVD	4	8	-1.15	0.13
BQVpH	QVA-QVB	Seasonal	12	1.3099	186	-1.2501	0.11	BQVSeaspH	QVA-QVB	5	6	-0.30	0.38
BQVpH	QVC-QVD	Seasonal	12	1.8662	265	-0.2424	0.40	**BQVSeaspH**	**QVC-QVD**	10	1	2.71	**>0.999**
BQVpH	QVA-QVD	Seasonal	12	1.7836	245.25	-0.3919	0.35	**BQVSeaspH**	**QVA-QVD**	8	2	1.90	**0.97**
BQVCond	QVA-QVB	Seasonal	12	0.7964	109.5	-2.1802	0.01						
BQVCond	QVC-QVD	Seasonal	12	2.1926	310.25	0.3488	0.64	BQVSeasCond	QVC-QVD	0	11	-3.32	<0.001
BQVCond	QVA-QVD	Seasonal	12	1.1237	159	-1.5873	0.06						
BQVDO	QVA-QVB	Seasonal	12	1.5	206.25	-0.9057	0.18	BQVSeasDO	QVA-QVB	2	6	-1.41	0.08
BQVDO	QVC-QVD	Seasonal	12	2.1588	295.75	0.2876	0.61	BQVSeasDO	QVC-QVD	5	3	0.71	0.76
BQVDO	QVA-QVD	Seasonal	12	1.7893	250.5	-0.3817	0.35	BQVSeasDO	QVA-QVD	8	3	1.51	0.93
BQVTSS	QVA-QVB	Seasonal	12	1.0599	150.5	-1.7029	0.04						
BQVTSS	QVC-QVD	Seasonal	12	2.3011	326.75	0.5453	0.71	BQVSeasTSS	QVC-QVD	8	4	1.15	0.87
BQVTSS	QVA-QVD	Seasonal	12	2.4755	354	0.8613	0.81	BQVSeasTSS	QVA-QVD	4	8	-1.15	0.13
BQVTOC	QVA-QVB	Seasonal	12	1.4615	209	-0.9753	0.16	BQVSeasTOC	QVA-QVB	0	12	-3.46	<0.001
BQVTOC	QVC-QVD	Seasonal	12	2.3986	343	0.722	0.76	BQVSeasTOC	QVC-QVD	4	8	-1.15	0.13
BQVTOC	QVA-QVD	Seasonal	12	2.2238	318	0.4053	0.66	BQVSeasTOC	QVA-QVD	1	11	-2.89	0.002
BQVCOD	QVA-QVB	Seasonal	12	2.6144	371.25	1.1129	0.87	BQVSeasCOD	QVA-QVB	4	8	-1.15	0.13
BQVCOD	QVC-QVD	Seasonal	12	1.8345	260.5	-0.2998	0.38	BQVSeasCOD	QVC-QVD	6	6	0.00	0.50
BQVCOD	QVA-QVD	Seasonal	12	1.6632	237	-0.6101	0.27	BQVSeasCOD	QVA-QVD	5	6	-0.30	0.38
BQVTotP	QVA-QVB	Seasonal	12	1.4261	164	-1.0395	0.15	BQVSeasTotP	QVA-QVB	1	4	-1.34	0.09
BQVTotP	QVC-QVD	Seasonal	12	2.4887	331	0.8852	0.81	BQVSeasTotP	QVC-QVD	1	6	-1.89	0.03
BQVTotP	QVA-QVD	Seasonal	12	2.3764	326.75	0.6817	0.75	BQVSeasTotP	QVA-QVD	2	6	-1.41	0.08
BQVPO4	QVA-QVB	Seasonal	12	2.6957	310	1.26	0.90	BQVSeasP04	QVA-QVB	3	2	0.45	0.67
BQVPO4	QVC-QVD	Seasonal	12	1.3845	173.75	-1.1149	0.13	BQVSeasPO4	QVC-QVD	1	5	-1.63	0.05
BQVPO4	QVA-QVD	Seasonal	12	1.1056	138.75	-1.6201	0.05						
BQVFTotP	QVA-QVB	Seasonal	12	1.684	210.5	-0.5724	0.28	BQVSeasFTotP	QVA-QVB	1	5	-1.63	0.05
BQVFTotP	QVC-QVD	Seasonal	12	1.758	219.75	-0.4383	0.33	BQVSeasFTotP	QVC-QVD	1	5	-1.63	0.05
BQVFTotP	QVA-QVD	Seasonal	12	1.8386	230.75	-0.2923	0.39	BQVSeasFTotP	QVA-QVD	1	5	-1.63	0.05
BQVNO3	QVA-QVB	Seasonal	12	1.3077	187	-1.254	0.10	**BQVSeasNO3**	**QVA-QVB**	12	0	3.46	**>0.999**
BQVNO3	QVC-QVD	Seasonal	12	1.5734	225	-0.7727	0.22	BQVSeasNO3	QVC-QVD	0	12	-3.46	<0.001
BQVNO3	QVA-QVD	Seasonal	12	1.1329	162	-1.5706	0.06						
BQVNH4	QVA-QVB	Seasonal	12	1.1504	153	-1.5389	0.06						
BQVNH4	QVC-QVD	Seasonal	12	1.942	268	-0.105	0.46	BQVSeasNH4	QVC-QVD	2	6	-1.41	0.08
BQVNH4	QVA-QVD	Seasonal	12	1.1295	141.75	-1.5768	0.06						
BQVTKN	QVA-QVB	Seasonal	11	3.0502	334	1.8295	0.97	BQVSeasTKN	QVA-QVB	4	5	-0.33	0.37
BQVTKN	QVC-QVD	Seasonal	11	2.3704	256	0.6452	0.74	BQVSeasTKN	QVC-QVD	5	3	0.71	0.76
BQVTKN	QVA-QVD	Seasonal	11	2.1735	238	0.3023	0.62	BQVSeasTKN	QVA-QVD	3	6	-1.00	0.16
BQVFLTKN	QVA-QVB	Seasonal	11	1.5648	169	-0.7581	0.22	BQVSeasFlTKN	QVA-QVB	4	4	0.00	0.50
BQVFLTKN	QVC-QVD	Seasonal	11	2.05	205	0.0871	0.53	BQVSeasFlTKN	QVC-QVD	4	2	0.82	0.79
BQVFLTKN	QVA-QVD	Seasonal	11	1.731	181.75	-0.4687	0.32	BQVSeasFlTKN	QVA-QVD	5	2	1.13	0.87
BQVTotN	QVA-QVB	Seasonal	11	2.2636	249	0.4593	0.68	**BQVSeasTotN**	**QVA-QVB**	9	2	2.11	**0.98**
BQVTotN	QVC-QVD	Seasonal	11	2.5909	285	1.0294	0.85	BQVSeasTotN	QVC-QVD	1	10	-2.71	0.003
BQVTotN	QVA-QVD	Seasonal	11	2.6091	287	1.061	0.86	BQVSeasTotN	QVA-QVD	7	4	0.90	0.82
BQVTotCol	QVA-QVB	Seasonal	12	1.9231	275	-0.1393	0.44	BQVSeasTotCol	QVA-QVB	4	8	-1.15	0.13
BQVTotCol	QVC-QVD	Seasonal	12	2.007	287	0.0127	0.51	BQVSeasTotCol	QVC-QVD	6	6	0.00	0.50
BQVTotCol	QVA-QVD	Seasonal	12	1.6993	243	-0.5447	0.29	BQVSeasTotCol	QVA-QVD	2	10	-2.31	0.010
BQVFecCol	QVA-QVB	Seasonal	12	1.7552	251	-0.4433	0.33	BQVSeasFecCol	QVA-QVB	6	6	0.00	0.50
BQVFecCol	QVC-QVD	Seasonal	12	2.0699	296	0.1267	0.55	BQVSeasFecCol	QVC-QVD	4	8	-1.15	0.13
BQVFecCol	QVA-QVD	Seasonal	12	2.2448	321	0.4433	0.67	BQVSeasFecCol	QVA-QVD	2	10	-2.31	0.010
BQVFecStrep	QVA-QVB	Seasonal	12	2.3566	337	0.646	0.74	BQVSeasFecStrep	QVA-QVB	5	6	-0.30	0.38
BQVFecStrep	QVC-QVD	Seasonal	12	1.7343	248	-0.4813	0.32	BQVSeasFecStrep	QVC-QVD	6	5	0.30	0.62
BQVFecStrep	QVA-QVD	Seasonal	12	1.8951	271	-0.19	0.42	BQVSeasFecStrep	QVA-QVD	3	9	-1.73	0.04
BQVCd	QVA-QVB	Seasonal	8	2.2635	83.75	0.4	0.66	BQVSeasCd	QVA-QVB	3	1	1.00	0.84
BQVCd	QVC-QVD	Seasonal	8	1.4695	60.25	-0.8053	0.21	BQVSeasCd	QVC-QVD	3	3	0.00	0.50
BQVCd	QVA-QVD	Seasonal	8	3.488	144.75	2.2587	0.99	BQVSeasCd	QVA-QVD	2	4	-0.82	0.21
BQVCu	QVA-QVB	Seasonal	8	1.3333	56	-1.012	0.16	BQVSeasCu	QVA-QVB	3	4	-0.38	0.35
BQVCu	QVC-QVD	Seasonal	8	0.65	26	-2.0493	0.02						
BQVCu	QVA-QVD	Seasonal	8	2.1	84	0.1518	0.56	BQVSeasCu	QVA-QVD	4	4	0.00	0.50
BQVPb	QVA-QVB	Seasonal	8	2.4392	90.25	0.6667	0.75	BQVSeasPb	QVA-QVB	0	4	-2.00	0.02
BQVPb	QVC-QVD	Seasonal	8	2.0361	84.5	0.0549	0.52	**BQVSeasPb**	**QVC-QVD**	6	1	1.89	**0.97**
BQVPb	QVA-QVD	Seasonal	8	1.488	61.75	-0.7773	0.22	BQVSeasPb	QVA-QVD	2	4	-0.82	0.21
BQVZn	QVA-QVB	Seasonal	8	1.8446	68.25	-0.2359	0.41	BQVSeasZn	QVA-QVB	1	3	-1.00	0.16
BQVZn	QVC-QVD	Seasonal	8	2.4	96	0.6072	0.73	BQVSeasZn	QVC-QVD	2	3	-0.45	0.33
BQVZn	QVA-QVD	Seasonal	8	2.9085	119.25	1.3792	0.92	BQVSeasZn	QVA-QVD	1	5	-1.63	0.05

RVN ratio test: alpha =0.1, Ho: independent
if p<0.1 reject Ho; for exact p-values Bartels 1982 JASA v 77, pp42-44
Test of randomness:

$$\frac{N^+ - N^-}{\sqrt{N^+ + N^-}} \sim N(0,1)$$

Sign Test, modified for ties (Putter 1955)
Ho: (p+)<=(p-)
Hi: (p+)>(p-)

		Sample size	RVN ratio test stat	Num of RVNRTS	Std. RVNRTS	Approx. p-value	If p>0.1, run sign test		+	-	Sign test stat	Approx p-value
SQVTSS	QVA-QVB	47	1.4698	12706	-1.837	0.03						
SQVTSS	QVC-QVD	39	1.1585	5718.5	-2.662	0.00						
SQVTSS	QVA-QVD	49	1.6048	15722	-1.3977	0.08						
SQVTOC	QVA-QVB	47	1.723	14899.75	-0.9598	0.17	SQVTOC	QVA-QVB	27	20	1.02	0.85
SQVTOC	QVC-QVD	39	1.2745	6296	-2.2951	0.01						
SQVTOC	QVA-QVD	49	1.3782	13506	-2.1991	0.01						
SQVCOD	QVA-QVB	47	1.7605	15211	-0.8298	0.20	SQVCOD	QVA-QVB	25	20	0.75	0.77
SQVCOD	QVC-QVD	39	1.6226	8009	-1.194	0.12	**SQVCOD**	**QVC-QVD**	26	12	2.27	**0.99**
SQVCOD	QVA-QVD	49	1.7589	17228	-0.8528	0.20	SQVCOD	QVA-QVD	22	25	-0.44	0.33
SQVTemp	QVA-QVB	19	1.0931	622	-2.0313	0.02						
SQVTemp	QVC-QVD	19	2.4739	1396.5	1.0614	0.86	SQVTemp	QVC-QVD	12	6	1.41	0.92
SQVTemp	QVA-QVD	19	1.3686	778.75	-1.4142	0.08						
SQVpH	QVA-QVB	19	1.9797	1097.75	-0.0454	0.48	SQVpH	QVA-QVB	3	15	-2.83	0.002
SQVpH	QVC-QVD	19	2.0623	1158	0.1396	0.56	**SQVpH**	**QVC-QVD**	17	0	4.12	**>0.999**
SQVpH	QVA-QVD	19	1.6728	948.5	-0.7328	0.23	SQVpH	QVA-QVD	10	8	0.47	0.68
SQVDO	QVA-QVB	19	2.1766	1201.5	0.3956	0.65	SQVDO	QVA-QVB	6	9	-0.77	0.22
SQVDO	QVC-QVD	19	1.5201	737.25	-1.0749	0.14	SQVDO	QVC-QVD	6	3	1.00	0.84
SQVDO	QVA-QVD	19	1.4892	755	-1.1443	0.13	SQVDO	QVA-QVD	7	3	1.27	0.90
SQVCond	QVA-QVB	18	1.8493	889.5	-0.3291	0.37	**SQVCond**	**QVA-QVB**	12	5	1.70	**0.96**
SQVCond	QVC-QVD	18	2.3399	1067	0.7423	0.77	SQVCond	QVC-QVD	4	13	-2.18	0.015
SQVCond	QVA-QVD	18	1.5724	760.25	-0.9338	0.18	SQVCond	QVA-QVD	10	6	1.00	0.84
SQVTotP	QVA-QVB	47	1.5677	13456.75	-1.4981	0.07						
SQVTotP	QVC-QVD	39	2.3768	11470.25	1.1919	0.88	**SQVTotP**	**QVC-QVD**	22	6	3.02	**>0.999**
SQVTotP	QVA-QVD	49	1.4622	14062	-1.9019	0.03						
SQVPO4	QVA-QVB	44	2.1163	13601.25	0.3901	0.65	SQVPO4	QVA-QVB	13	11	0.41	0.66
SQVPO4	QVC-QVD	36	2.2836	7346.25	0.8628	0.81	SQVPO4	QVC-QVD	9	7	0.50	0.69
SQVPO4	QVA-QVD	43	1.6538	9847.75	-1.1485	0.13	SQVPO4	QVA-QVD	15	8	1.46	0.93
SQVFTotP	QVA-QVB	45	1.4982	9852	-1.7023	0.04						
SQVFTotP	QVC-QVD	39	1.8797	8373.25	-0.3805	0.35	SQVFTotP	QVC-QVD	13	8	1.09	0.86
SQVFTotP	QVA-QVD	48	2.0025	16668	0.0088	0.50	**SQVFTotP**	**QVA-QVD**	18	8	1.96	**0.98**
SQVNO3	QVA-QVB	47	1.6581	14339	-1.1848	0.12	**SQVNO3**	**QVA-QVB**	35	11	3.54	**>0.999**
SQVNO3	QVC-QVD	39	1.045	5162	-3.021	0.00						
SQVNO3	QVA-QVD	49	0.7014	6872	-4.5923	0.00						
SQVNH4	QVA-QVB	47	2.1014	17312.5	0.3514	0.64	SQVNH4	QVA-QVB	19	12	1.26	0.90
SQVNH4	QVC-QVD	39	1.7875	7806	-0.6723	0.25	**SQVNH4**	**QVC-QVD**	16	4	2.68	**>0.999**
SQVNH4	QVA-QVD	49	2.0914	18882.5	0.3233	0.63	SQVNH4	QVA-QVD	18	12	1.10	0.86
SQVTKN	QVA-QVB	45	1.5228	11493	-1.6189	0.05						
SQVTKN	QVC-QVD	37	1.9415	8134	-0.1803	0.43	**SQVTKN**	**QVC-QVD**	21	9	2.19	**0.99**
SQVTKN	QVA-QVD	47	1.6037	13822	-1.3733	0.08						
SQVFLTKN	QVA-QVB	45	1.8149	12565.5	-0.6279	0.27	SQVFLTKN	QVA-QVB	12	13	-0.20	0.42
SQVFLTKN	QVC-QVD	36	1.889	6810	-0.3376	0.37	SQVFLTKN	QVC-QVD	8	13	-1.09	0.14
SQVFLTKN	QVA-QVD	46	2.0418	15561.25	0.1432	0.56	SQVFLTKN	QVA-QVD	12	16	-0.76	0.22
SQVTotN	QVA-QVB	45	1.8073	13714	-0.6536	0.26	**SQVTotN**	**QVA-QVB**	29	16	1.94	**0.97**
SQVTotN	QVC-QVD	37	1.4538	6131.5	-1.6842	0.05						
SQVTotN	QVA-QVD	47	1.1831	10229	-2.8306	0.00						
SQVTotCol	QVA-QVB	23	2.8014	2835	1.9653	0.98	SQVTotCol	QVA-QVB	10	13	-0.63	0.26
SQVTotCol	QVC-QVD	23	2.5109	2538.5	1.2528	0.89	SQVTotCol	QVC-QVD	14	9	1.04	0.85
SQVTotCol	QVA-QVD	23	2.6057	2637	1.4855	0.93	SQVTotCol	QVA-QVD	12	11	0.21	0.58
SQVFecCol	QVA-QVB	23	1.7549	1776	-0.601	0.27	SQVFecCol	QVA-QVB	9	14	-1.04	0.15
SQVFecCol	QVC-QVD	23	2.4103	2438	1.0061	0.84	SQVFecCol	QVC-QVD	9	14	-1.04	0.15
SQVFecCol	QVA-QVD	23	2.3281	2356	0.8045	0.79	SQVFecCol	QVA-QVD	10	13	-0.63	0.26
SQVFecStrep	QVA-QVB	23	2.0791	2104	0.1939	0.58	SQVFecStrep	QVA-QVB	12	11	0.21	0.58
SQVFecStrep	QVC-QVD	23	1.9901	2013	-0.0242	0.49	SQVFecStrep	QVC-QVD	7	16	-1.88	0.03
SQVFecStrep	QVA-QVD	23	2.119	2141.25	0.2918	0.61	SQVFecStrep	QVA-QVD	9	13	-0.85	0.20
SQVCd	QVA-QVB	17	1.3399	544	-1.4034	0.08						
SQVCd	QVC-QVD	17	1.7758	714.75	-0.4767	0.32	SQVCd	QVC-QVD	6	7	-0.28	0.39
SQVCd	QVA-QVD	17	1.4646	589.5	-1.1383	0.13	SQVCd	QVA-QVD	8	5	0.83	0.80
SQVCu	QVA-QVB	17	2.2307	909	0.4904	0.69	SQVCu	QVA-QVB	10	6	1.00	0.84
SQVCu	QVC-QVD	17	2.14	871	0.2977	0.62	SQVCu	QVC-QVD	6	9	-0.77	0.22
SQVCu	QVA-QVD	17	2.6093	1062	1.2955	0.90	**SQVCu**	**QVA-QVD**	12	4	2.00	**0.98**
SQVPb	QVA-QVB	17	1.3477	546.5	-1.3868	0.08						
SQVPb	QVC-QVD	17	1.702	691	-0.6336	0.26	SQVPb	QVC-QVD	5	9	-1.07	0.14
SQVPb	QVA-QVD	17	1.2269	497.5	-1.6437	0.05						
SQVZn	QVA-QVB	17	1.3262	539.75	-1.4326	0.08						
SQVZn	QVC-QVD	17	1.3784	558.25	-1.3215	0.09						
SQVZn	QVA-QVD	17	2.6464	1075.75	1.3742	0.92	SQVZn	QVA-QVD	10	5	1.29	0.90

RVN ratio test: alpha =0.1, Ho: independent

if p<0.1 reject Ho; for exact p-values Bartels 1982 JASA v 77, pp42-44

Test of randomness:

			Sample size	RVN ratio test stat	Num of RVNRTS	Std. RVNRTS	Approx. p-value	If p>0.1, run sign test		+	-	Sign test	Approx p-value
SQVTSS	QVA-QVB	Seasonal	11	1.5091	166	-0.8552	0.20	SQVSeasTSS	QVA-QVB	3	8	-1.51	0.07
SQVTSS	QVC-QVD	Seasonal	11	0.4815	52	-2.6453	0.00						
SQVTSS	QVA-QVD	Seasonal	11	2.3105	253	0.5409	0.71	SQVSeasTSS	QVA-QVD	3	8	-1.51	0.07
SQVTOC	QVA-QVB	Seasonal	11	2.3364	257	0.586	0.72	SQVSeasTOC	QVA-QVB	7	4	0.91	0.82
SQVTOC	QVC-QVD	Seasonal	11	1.3273	146	-1.1719	0.12	SQVSeasTOC	QVC-QVD	2	9	-2.11	0.02
SQVTOC	QVA-QVD	Seasonal	11	1.5273	168	-0.8235	0.21	SQVSeasTOC	QVA-QVD	4	7	-0.91	0.18
SQVCOD	QVA-QVB	Seasonal	11	1.5434	169	-0.7954	0.21	SQVSeasCOD	QVA-QVB	5	5	0.00	0.50
SQVCOD	QVC-QVD	Seasonal	11	1.3364	147	-1.1561	0.12	SQVSeasCOD	QVC-QVD	8	3	1.51	0.93
SQVCOD	QVA-QVD	Seasonal	11	1.4727	162	-0.9185	0.18	SQVSeasCOD	QVA-QVD	4	7	-0.91	0.18
SQVTemp	QVA-QVB	Seasonal	10	2.3758	196	0.6276	0.73	SQVSeasTemp	QVA-QVB	3	7	-1.26	0.10
SQVTemp	QVC-QVD	Seasonal	10	2.7914	227.5	1.3218	0.91	SQVSeasTemp	QVC-QVD	7	3	1.26	0.90
SQVTemp	QVA-QVD	Seasonal	10	2.5091	207	0.8503	0.80	SQVSeasTemp	QVA-QVD	3	7	-1.26	0.10
SQVpH	QVA-QVB	Seasonal	10	1.9356	157.75	-0.1076	0.46	SQVSeaspH	QVA-QVB	2	8	-1.90	0.03
SQVpH	QVC-QVD	Seasonal	10	1.8628	152.75	-0.2291	0.41	**SQVSeaspH**	**QVC-QVD**	9	0	3.00	**>0.999**
SQVpH	QVA-QVD	Seasonal	10	1.872	153.5	-0.2139	0.42	SQVSeaspH	QVA-QVD	5	5	0.00	0.50
SQVDO	QVA-QVB	Seasonal	10	2.6164	208	1.0294	0.85	SQVSeasDO	QVA-QVB	2	6	-1.41	0.08
SQVDO	QVC-QVD	Seasonal	10	2.5	200	0.8351	0.80	SQVSeasDO	QVC-QVD	5	2	1.13	0.87
SQVDO	QVA-QVD	Seasonal	10	2.6494	204	1.0845	0.86	**SQVSeasDO**	**QVA-QVD**	5	1	1.63	**0.95**
SQVCond	QVA-QVB	Seasonal	10	1.7212	142	-0.4656	0.32	SQVSeasCond	QVA-QVB	7	3	1.26	0.90
SQVCond	QVC-QVD	Seasonal	10	2.2687	181.5	0.4489	0.67	SQVSeasCond	QVC-QVD	1	9	-2.53	0.01
SQVCond	QVA-QVD	Seasonal	10	1.589	129.5	-0.6865	0.25	SQVSeasCond	QVA-QVD	6	4	0.63	0.74
SQVTotP	QVA-QVB	Seasonal	11	2.5571	280	0.9704	0.83	SQVSeasTotP	QVA-QVB	5	4	0.33	0.63
SQVTotP	QVC-QVD	Seasonal	11	1.269	133.25	-1.2733	0.10	**SQVSeasTotP**	**QVC-QVD**	6	1	1.89	**0.97**
SQVTotP	QVA-QVD	Seasonal	11	2.0093	217	0.0161	0.51	SQVSeasTotP	QVA-QVD	5	3	0.71	0.76
SQVPO4	QVA-QVB	Seasonal	11	2.6946	249.25	1.21	0.89	SQVSeasPO4	QVA-QVB	1	3	-1.00	0.16
SQVPO4	QVC-QVD	Seasonal	10	1.8192	118.25	-0.3019	0.38	**SQVSeasPO4**	**QVC-QVD**	4	0	2.00	**0.98**
SQVPO4	QVA-QVD	Seasonal	10	2.2269	144.75	0.379	0.65	SQVSeasPO4	QVA-QVD	3	1	1.00	0.84
SQVFTotP	QVA-QVB	Seasonal	11	2.5459	235.5	0.951	0.83	SQVSeasFTotP	QVA-QVB	2	3	-0.45	0.33
SQVFTotP	QVC-QVD	Seasonal	11	1.2892	119.25	-1.2382	0.11	SQVSeasFTotP	QVC-QVD	4	1	1.34	0.91
SQVFTotP	QVA-QVD	Seasonal	11	3.2439	266	2.1669	0.98	SQVSeasFTotP	QVA-QVD	3	1	1.00	0.84
SQVNO3	QVA-QVB	Seasonal	11	1.4364	158	-0.9819	0.16	**SQVSeasNO3**	**QVA-QVB**	9	2	2.11	**0.98**
SQVNO3	QVC-QVD	Seasonal	11	1.0364	114	-1.6787	0.05						
SQVNO3	QVA-QVD	Seasonal	11	1.6	176	-0.6968	0.24	SQVSeasNO3	QVA-QVD	7	4	0.91	0.82
SQVNH4	QVA-QVB	Seasonal	11	1.7222	186	-0.4839	0.31	SQVSeasNH4	QVA-QVB	4	4	0.00	0.50
SQVNH4	QVC-QVD	Seasonal	11	1.4946	138.25	-0.8804	0.19	**SQVSeasNH4**	**QVC-QVD**	5	0	2.24	**0.99**
SQVNH4	QVA-QVD	Seasonal	11	1.8243	168.75	-0.306	0.38	SQVSeasNH4	QVA-QVD	4	1	1.34	0.91
SQVTKN	QVA-QVB	Seasonal	10	1.4024	115	-0.998	0.16	SQVSeasTKN	QVA-QVB	2	7	-1.67	0.05
SQVTKN	QVC-QVD	Seasonal	10	1.6364	135	-0.6073	0.27	**SQVSeasTKN**	**QVC-QVD**	7	2	1.67	**0.95**
SQVTKN	QVA-QVD	Seasonal	10	1.2848	106	-1.1944	0.12	SQVSeasTKN	QVA-QVD	4	5	-0.33	0.37
SQVFLTKN	QVA-QVB	Seasonal	10	1.5154	98.5	-0.8094	0.21	SQVSeasFLTKN	QVA-QVB	2	2	0.00	0.50
SQVFLTKN	QVC-QVD	Seasonal	10	2.3774	184.25	0.6304	0.74	SQVSeasFLTKN	QVC-QVD	4	2	0.82	0.79
SQVFLTKN	QVA-QVD	Seasonal	10	2.2484	174.25	0.4148	0.66	SQVSeasFLTKN	QVA-QVD	2	4	-0.82	0.21
SQVTotN	QVA-QVB	Seasonal	10	1.0424	86	-1.5993	0.05						
SQVTotN	QVC-QVD	Seasonal	10	1.2364	102	-1.2754	0.10	SQVSeasTotN	QVC-QVD	3	7	-1.27	0.10
SQVTotN	QVA-QVD	Seasonal	10	1.5636	129	-0.7288	0.23	SQVSeasTotN	QVA-QVD	7	3	1.27	0.90
SQVTotCol	QVA-QVB	Seasonal	10	2.7879	230	1.3159	0.91	SQVSeasTotCol	QVA-QVB	4	6	-0.63	0.26
SQVTotCol	QVC-QVD	Seasonal	10	1.4182	117	-0.9717	0.17	SQVSeasTotCol	QVC-QVD	6	4	0.63	0.74
SQVTotCol	QVA-QVD	Seasonal	10	2.9091	240	1.5183	0.94	SQVSeasTotCol	QVA-QVD	5	5	0.00	0.50
SQVFecCol	QVA-QVB	Seasonal	10	1.1515	95	-1.4171	0.08						
SQVFecCol	QVC-QVD	Seasonal	10	2.303	190	0.5061	0.69	SQVSeasFecCol	QVC-QVD	3	7	-1.26	0.10
SQVFecCol	QVA-QVD	Seasonal	10	1.8061	149	-0.3239	0.37	SQVSeasFecCol	QVA-QVD	3	7	-1.26	0.10
SQVFecStrep	QVA-QVB	Seasonal	10	2.497	206	0.83	0.80	SQVSeasFecStrep	QVA-QVB	6	4	0.63	0.74
SQVFecStrep	QVC-QVD	Seasonal	10	1.1879	98	-1.3564	0.09						
SQVFecStrep	QVA-QVD	Seasonal	10	2.4121	199	0.6883	0.75	SQVSeasFecStrep	QVA-QVD	3	7	-1.26	0.10
SQVCd	QVA-QVB	Seasonal	6	2.8571	50	1.1615	0.88	SQVSeasCd	QVA-QVB	4	1	1.34	0.91
SQVCd	QVC-QVD	Seasonal	6	1.4559	24.75	-0.7373	0.23	SQVSeasCd	QVC-QVD	4	2	0.82	0.79
SQVCd	QVA-QVD	Seasonal	6	2.5143	44	0.6969	0.76	SQVSeasCd	QVA-QVD	3	3	0.00	0.50
SQVCu	QVA-QVB	Seasonal	6	1.0857	19	-1.239	0.11	**SQVSeasCu**	**QVA-QVB**	5	1	1.63	**0.95**
SQVCu	QVC-QVD	Seasonal	6	2.2857	40	0.3872	0.65	SQVSeasCu	QVC-QVD	3	3	0.00	0.50
SQVCu	QVA-QVD	Seasonal	6	2.2206	37.75	0.2989	0.62	SQVSeasCu	QVA-QVD	4	1	1.34	0.91
SQVPb	QVA-QVB	Seasonal	6	2.7794	47.25	1.0562	0.85	SQVSeasPb	QVA-QVB	3	1	1.00	0.84
SQVPb	QVC-QVD	Seasonal	6	2.2857	40	0.3872	0.65	SQVSeasPb	QVC-QVD	2	4	-0.82	0.21
SQVPb	QVA-QVD	Seasonal	6	2.3235	39.5	0.4384	0.67	SQVSeasPb	QVA-QVD	2	3	-0.45	0.33
SQVZn	QVA-QVB	Seasonal	6	1.2571	22	-1.0066	0.16	SQVSeasZn	QVA-QVB	3	3	0.00	0.50
SQVZn	QVC-QVD	Seasonal	6	1.2	21	-1.0841	0.14	SQVSeasZn	QVC-QVD	3	2	0.45	0.67
SQVZn	QVA-QVD	Seasonal	6	1.5429	27	-0.6195	0.27	SQVSeasZn	QVA-QVD	4	1	1.34	0.91

RVN ratio test: alpha =0.2, Ho: independent
if p<0.2 reject Ho; for exact p-values Bartels 1982 JASA v 77, pp42-44
Test of randomness:

$$\frac{N^+ - N^-}{\sqrt{N^+ + N^-}} \sim N(0,1)$$

Sign Test, modified for ties (Putter 1955)
Ho: (p+)<=(p-)
Hi: (p+)>(p-)

		Sample size	RVN ratio test stat	Num of RVNRTS	Std. RVNRTS	Approx. p-value	If p>0.2, run sign test		Positi	egati	Sign test stat	Approx p-value
SLoadTSS	QVA-QVB	28	1.3952	2549	-1.6298	0.05						
SLoadTSS	QVC-QVD	20	1.2632	840	-1.6909	0.05						
SLoadTSS	QVA-QVD	30	1.0429	2344	-2.6661	0.00						
SLoadTOC	QVA-QVB	28	1.1385	2080	-2.3215	0.01						
SLoadTOC	QVC-QVD	20	0.409	272	-3.6511	0.00						
SLoadTOC	QVA-QVD	30	0.7715	1734	-3.4222	0.00						
SLoadCOD	QVA-QVB	28	2.2003	4020	0.5398	0.71	SLoadCOD	QVA-QVB	16	12	0.76	0.78
SLoadCOD	QVC-QVD	20	2.1729	1445	0.3969	0.65	SLoadCOD	QVC-QVD	13	7	1.34	0.91
SLoadCOD	QVA-QVD	30	2.1958	4935	0.5454	0.71	SLoadCOD	QVA-QVD	7	23	-2.92	0.0018
SLoadTotP	QVA-QVB	28	1.5227	2782	-1.2861	0.10						
SLoadTotP	QVC-QVD	20	2.0406	1357	0.0932	0.54	SLoadTotP	QVC-QVD	10	9	0.23	0.59
SLoadTotP	QVA-QVD	30	1.816	4077	-0.5125	0.30	SLoadTotP	QVA-QVD	6	21	-2.89	0.002
SLoadPO4	QVA-QVB	26	1.3641	1995	-1.6536	0.05						
SLoadPO4	QVC-QVD	19	1.4368	819	-1.2614	0.10	SLoadPO4	QVC-QVD	3	16	-2.98	0.0014
SLoadPO4	QVA-QVD	27	1.0482	1717	-2.5202	0.01						
SLoadFLTotP	QVA-QVB	27	1.1987	1961	-2.1219	0.02						
SLoadFLTotP	QVC-QVD	20	1.0241	681	-2.2396	0.01						
SLoadFLTotP	QVA-QVD	30	1.0283	2309	-2.707	0.00						
SLoadNO3	QVA-QVB	28	1.5041	2748	-1.3363	0.09						
SLoadNO3	QVC-QVD	20	1.3353	888	-1.5253	0.06						
SLoadNO3	QVA-QVD	30	0.8209	1845	-3.2846	0.00						
SLoadNH4	QVA-QVB	28	0.9616	1755	-2.798	0.00						
SLoadNH4	QVC-QVD	20	2.7496	1823	1.7203	0.96	SLoadNH4	QVC-QVD	4	13	-2.18	0.015
SLoadNH4	QVA-QVD	30	1.9206	4307	-0.2211	0.41	SLoadNH4	QVA-QVD	5	21	-3.14	<0.001
SLoadTKN	QVA-QVB	28	1.5107	2752.5	-1.3185	0.09						
SLoadTKN	QVC-QVD	20	2.3469	1556	0.7961	0.79	SLoadTKN	QVC-QVD	9	8	0.24	0.59
SLoadTKN	QVA-QVD	30	0.3527	792	-4.5889	0.00						
SLoadFTKN	QVA-QVB	28	0.6768	1139.75	-3.5655	0.00						
SLoadFTKN	QVC-QVD	20	1.0887	678.25	-2.0913	0.02						
SLoadFTKN	QVA-QVD	30	0.3645	789.25	-4.5559	0.00						
SLoadTotN	QVA-QVB	28	1.2474	2279	-2.028	0.02						
SLoadTotN	QVC-QVD	20	1.9684	1309	-0.0725	0.47	SLoadTotN	QVC-QVD	5	15	-2.24	0.013
SLoadTotN	QVA-QVD	30	1.7375	3905	-0.7313	0.23	SLoadTotN	QVA-QVD	1	29	-5.11	<0.001
Sflow	QVA-QVB	28	1.1938	2181	-2.1725	0.01						
Sflow	QVC-QVD	20	1.8947	1260	-0.2416	0.40	Sflow	QVC-QVD	1	19	-4.03	<0.001
Sflow	QVA-QVD	30	1.3375	3006	-1.8456	0.03						

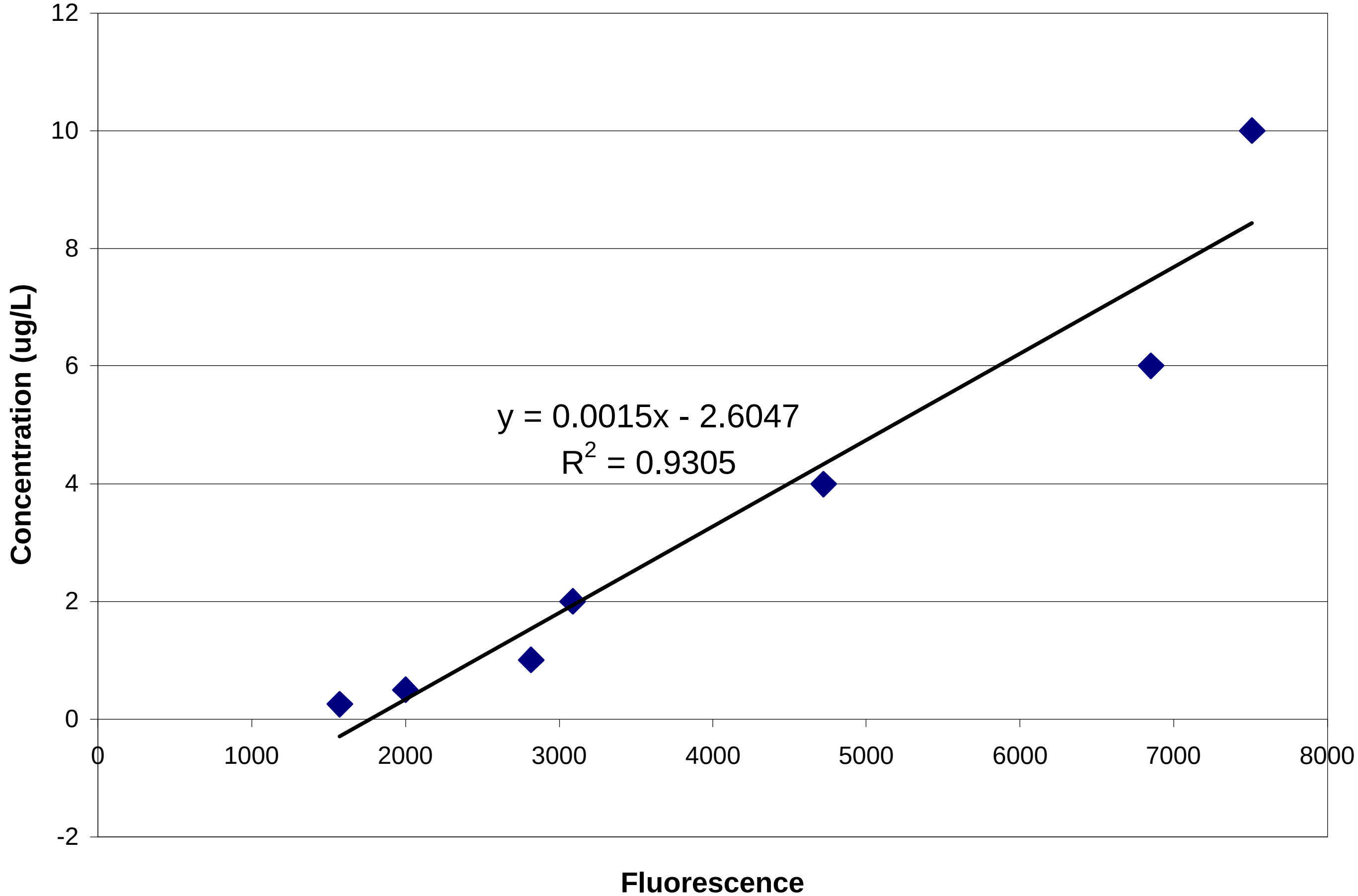
RVN ratio test: alpha =0.1, Ho: independent
if p<0.1 reject Ho; for exact p-values Bartels 1982 JASA v 77, pp42-44
Test of randomness:

Sign Test, modified for ties (Putter 1955)
Ho: (p+)<=(p-)
Hi: (p+)>(p-)

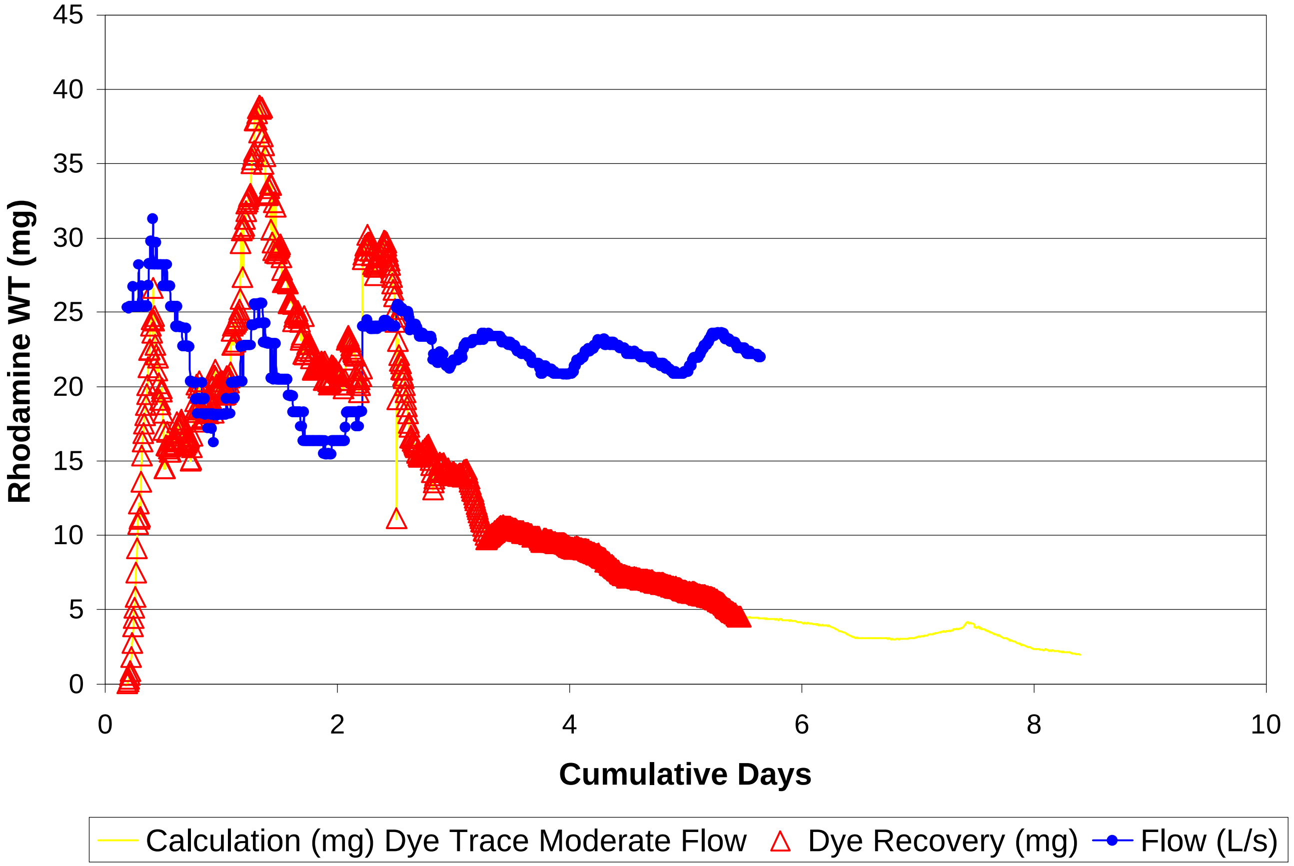
EMCEff		**Sample size**	**RVN ratio test stat**	**Num of RVNRTS**	**Std. RVNRTS**	**Approx. p-value**	**If p>0.1, run sign tes**		**+**	**-**	**Sign test stat**	**Approx p-value**
TSS	Wet Pond	28	1.3771	2516	-1.6784	0.05		TSSWet	15	13	0.38	
	Dry Pond	20	1.3594	904	-1.4701	0.07		**TSSDry**	20	0	4.47	>.999
	System	30	1.3391	3009	-1.841	0.03		TSSSys	16	14	0.37	
TOC	Wet Pond	28	1.5813	2889	-1.1283	**0.13**	**0.13**	**TOCWet**	20	8	2.27	0.9884
	Dry Pond	20	1.4105	938	-1.3528	0.09		TOCDry	5	15	-2.24	**0.01255**
	System	30	1.1008	2474	-2.505	0.01		TOCSys	14	16	-0.37	
COD	Wet Pond	28	2.0353	3716.5	0.0952	**0.54**	**0.54**	CODWet	17	9	1.57	
	Dry Pond	20	1.7023	1132	-0.6833	**0.25**	**0.25**	**CODDry**	19	1	4.02	>.999
	System	30	2.0436	4593	0.1215	**0.55**	**0.55**	CODSys	16	13	0.56	
TP	Wet Pond	26	1.2428	1817	-1.969	0.02		TPWet	13	12	0.20	
	Dry Pond	19	2.892	1647	1.9981	**0.98**	**0.98**	**TPDry**	16	2	3.30	>.999
	System	27	1.699	2783	-1.797	**0.21**	**0.21**	TPSys	16	10	1.18	
PO4	Wet Pond	26	2.5765	3399.75	1.4992	**0.93**	**0.93**	PO4Wet	10	4	1.60	
	Dry Pond	19	2.0261	932	0.0584	**0.52**	**0.52**	PO4Dry	5	3	0.71	
	System	26	1.6222	2194	-0.9825	**0.16**	**0.16**	**PO4Sys**	11	4	1.81	0.96485
FTP	Wet Pond	24	1.1746	1330.25	-2.0657	0.02		FTPWet	12	6	1.41	
	Dry Pond	19	1.775	992.25	-0.5039	**0.31**	**0.31**	FTPDry	10	4	1.60	
	System	26	1.6604	2398.5	-0.883	**0.19**	**0.19**	**FTPSys**	17	3	3.13	>.999
NO3	Wet Pond	28	1.5944	2913	-1.0929	**0.14**	**0.14**	**NO3Wet**	22	6	3.02	0.998736
	Dry Pond	20	1.1955	795	-1.8462	0.03		NO3Dry	11	9	0.45	
	System	30	0.7244	1628	-3.5536	0.00		**NO3Sys**	25	5	3.65	>.999
NH4	Wet Pond	25	1.5521	2009.25	-1.1429	**0.13**	**0.13**	**NH4Wet**	16	5	2.40	0.991802
	Dry Pond	17	1.8089	733.5	-0.4063	**0.34**	**0.34**	NH4Dry	11	3	2.14	
	System	26	1.5093	2191.5	-1.276	**0.10**	**0.10**	NH4Sys	13	8	1.09	
TKN	Wet Pond	24	1.8815	2160	-0.2965	**0.38**	**0.38**	TKNWet	8	16	-1.63	
	Dry Pond	16	0.5824	198	-2.9299	0.00		**TKNDry**	15	1	3.50	>.999
	System	26	1.7156	2509	-0.7397	**0.23**	**0.23**	TKNSys	13	13	0.00	
FTKN	Wet Pond	16	1.0382	353	-1.9877	0.02		FTKNWet	8	7	0.26	
	Dry Pond	10	1.8303	151	-0.2834	**0.39**	**0.39**	FTKNDry	5	4	0.33	
	System	18	2.2043	1068	0.4462	**0.67**	**0.67**	FTKNSys	9	8	0.24	
TN	Wet Pond	28	1.5599	2850	-1.1858	**0.12**	**0.12**	TNWet	17	11	1.13	
	Dry Pond	20	1.4421	959	-1.2803	**0.10**	**0.10**	TNDry	11	9	0.45	
	System	30	1.3766	3094	-1.7365	0.04		**TNSys**	20	10	1.83	0.96638

APPENDIX C

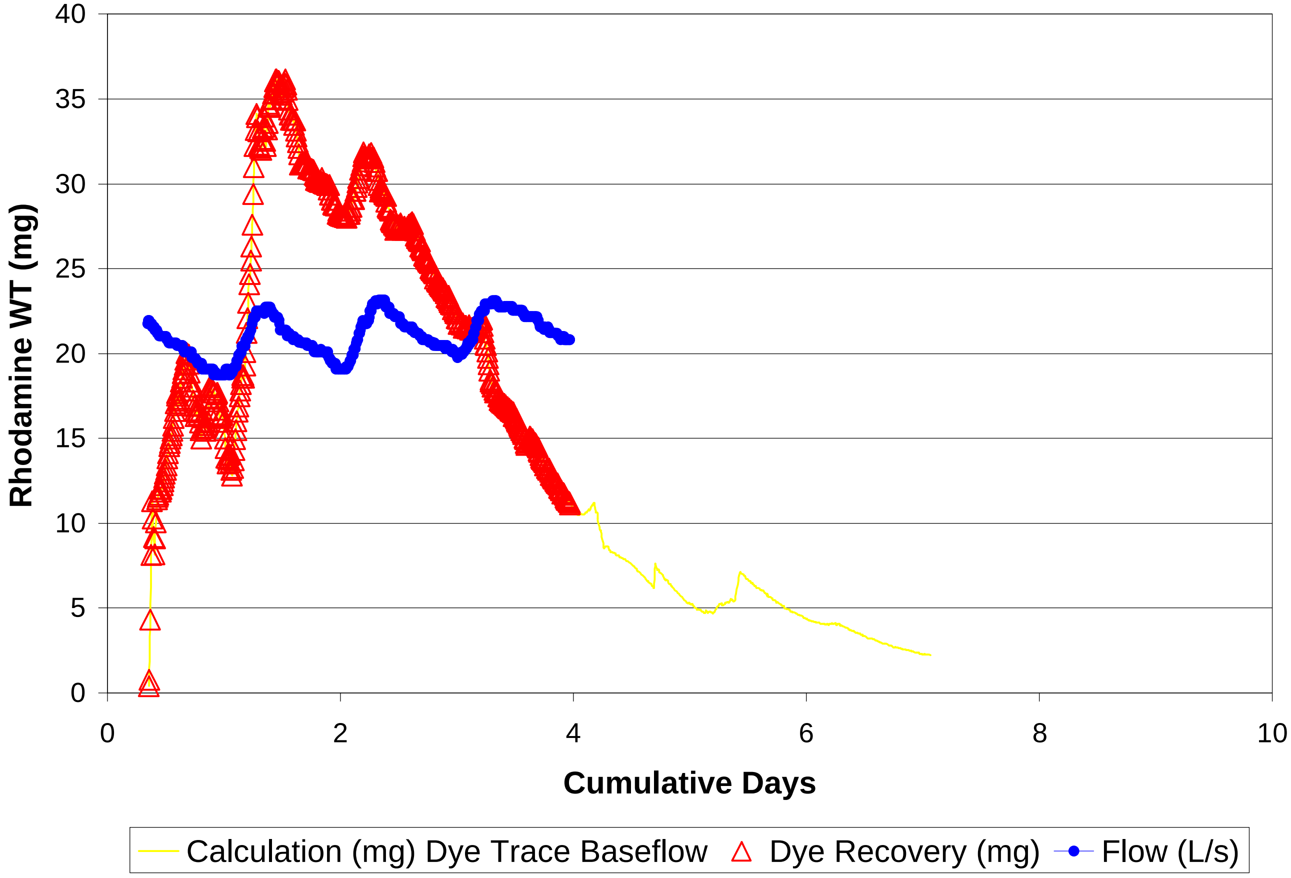
Rhodamine WT 20% fluorescence vs. dye concentration rating curve and dye mass recovery for the two dye traces: Baseflow condition (10/28/1999 though 11/4/1999), Moderate flow condition (9/27/1999 through 10/5/1999)



12
10
8
6
4
2
0
-2
Concentration (ug/L)
y = 0.0015x - 2.6047
R2 = 0.9305
0
1000
2000
3000
4000
5000
6000
7000
8000
Fluorescence



Rhodamine WT (mg)
45
40
35
30
25
20
15
10
5
0
0
2
4
6
8
10
Cumulative Days
Calculation (mg) Dye Trace Moderate Flow
Dye Recovery (mg)
Flow (L/s)



Rhodamine WT (mg)
0
5
10
15
20
25
30
35
40
0
2
4
6
8
10
Cumulative Days
Calculation (mg) Dye Trace Baseflow
Dye Recovery (mg)
Flow (L/s)

APPENDIX D

Descriptive statistics of all concentration and loading data sets including sample size, median, and interquartile range

Analyte		QVA			QVB			QVC			QVD		
		n	**median**	IQR	n	**median**	IQR	n	**median**	IQR	n	**median**	IQR
TSS (g/L)	1997	17	**7**	12	17	**11**	18	17	**13**	13	17	**9**	12
	1998	16	**6**	9	16	**11**	12	16	**11**	14	16	**8**	13
	1999 Total	60	**22**	51	50	**24**	50	44	**91**	235	51	**21**	61
	Baseflow	35	**4**	5	35	**7**	6	35	**8**	7	35	**7**	9
	Storm EMC	39	**42**	64	29	**46**	89	23	**242**	232	30	**56**	92
	Grab	39	**3**	5	39	**8**	7	39	**9**	8	39	**7**	11
COD (mg/L)	1997	17	**9**	9	17	**11**	14	17	**15**	16	17	**11**	25
	1998	16	**7**	17	16	**7**	6	16	**8**	10	16	**11**	10
	1999 Total	60	**18**	26	50	**16**	23	44	**27**	53	51	**21**	26
	Baseflow	35	**8**	14	35	**6**	14	35	**8**	11	35	**7**	17
	Storm EMC	39	**25**	22	29	**20**	20	23	**61**	56	30	**30**	29
	Grab	39	**5**	14	39	**8**	14	39	**8**	10	39	**7**	17
TOC (mg/L)	1997	17	**0.59**	0.23	17	**0.93**	1.01	17	**1.11**	1.68	17	**1.07**	1.21
	1998	16	**0.85**	0.72	16	**1.25**	0.66	16	**1.36**	0.66	16	**1.47**	0.68
	1999 Total	60	**2.08**	2.74	50	**2.72**	1.74	44	**2.57**	1.53	51	**2.65**	2.68
	Baseflow	35	**0.62**	0.33	35	**1.21**	1.55	35	**1.52**	1.77	35	**1.29**	1.11
	Storm EMC	39	**2.98**	2.07	29	**2.76**	1.77	23	**2.39**	1.47	30	**3.42**	4.86
	Grab	39	**0.64**	0.36	39	**1.30**	1.82	39	**1.58**	1.81	39	**1.29**	1.37
Temperature (C)	1997	17	**13.0**	1.6	17	**13.2**	6.0	17	**13.4**	7.8	17	**13.3**	8.1
	1998	15	**13.3**	2.0	16	**15.5**	10.1	16	**15.4**	10.0	16	**15.7**	8.7
	1999 Total	21	**13.1**	1.5	20	**15.8**	12.3	21	**15.7**	11.3	21	**15.4**	9.4
	Baseflow	33	**13.1**	1.3	34	**15.2**	12.0	35	**15.7**	11.8	35	**15.1**	9.9
	Grab	37	**13.1**	1.5	38	**15.8**	11.2	39	**15.7**	10.7	39	**15.4**	9.2
Conductivity (uOhm)	1997	17	**380**	110	17	**370**	100	17	**370**	100	17	**410**	100
	1998	15	**410**	150	15	**350**	110	15	**330**	115	15	**380**	115
	1999 Total	22	**400**	73	21	**340**	110	21	**340**	110	21	**360**	100
	Baseflow	35	**400**	105	35	**350**	100	35	**370**	90	35	**370**	80
	Grab	39	**400**	85	39	**350**	110	39	**360**	110	39	**370**	85
Dissolved Oxygen (mg/L)	1997	17	**5.5**	3.0	17	**6.0**	3.0	17	**5.5**	3.0	17	**5.0**	1.5
	1998	16	**5.8**	2.3	16	**7.5**	2.1	16	**5.5**	2.0	16	**5.3**	1.8
	1999 Total	20	**8.0**	2.0	20	**7.0**	2.3	20	**6.5**	3.0	20	**6.5**	3.0
	Baseflow	34	**7.0**	2.0	34	**7.5**	2.0	34	**6.5**	3.0	34	**5.8**	2.8
	Grab	38	**7.0**	2.0	38	**7.0**	2.0	38	**6.0**	2.8	38	**5.8**	2.8
pH	1997	17	**7.9**	1.0	17	**8.0**	0.8	17	**7.8**	1.0	17	**7.6**	0.7
	1998	16	**7.8**	0.5	16	**8.0**	0.6	16	**7.9**	0.7	16	**7.5**	0.6
	1999 Total	22	**8.1**	0.2	21	**7.9**	0.9	21	**8.0**	0.9	21	**7.8**	0.4
	Baseflow	35	**8.0**	0.5	35	**8.0**	0.9	35	**7.9**	0.8	35	**7.8**	0.8
	Grab	39	**8.0**	0.5	39	**8.0**	0.9	39	**7.9**	0.8	39	**7.8**	0.7

Analyte		QVA n	QVA median	QVA IQR	QVB n	QVB median	QVB IQR	QVC n	QVC median	QVC IQR	QVD n	QVD median	QVD IQR
NH4 (mg/L)	1997	17	**0.01**	0.00	17	**0.01**	0.00	17	**0.01**	0.00	17	**0.01**	0.02
	1998	16	**0.01**	0.05	16	**0.01**	0.01	16	**0.01**	0.00	16	**0.01**	0.00
	1999 Total	60	**0.05**	0.04	50	**0.05**	0.04	44	**0.05**	0.05	51	**0.05**	0.05
	Baseflow	35	**0.03**	0.05	35	**0.01**	0.05	35	**0.02**	0.05	35	**0.03**	0.06
	Storm EMC	39	**0.05**	0.04	29	**0.05**	0.03	23	**0.05**	0.05	30	**0.05**	0.04
	Grab	39	**0.03**	0.05	39	**0.01**	0.06	39	**0.02**	0.05	39	**0.03**	0.07
NO3 (mg/L)	1997	17	**1.94**	0.40	17	**1.79**	0.90	17	**1.83**	1.12	17	**2.53**	1.90
	1998	16	**2.00**	1.61	16	**1.22**	1.07	16	**1.25**	1.41	16	**1.71**	1.51
	1999 Total	60	**1.50**	0.50	50	**0.82**	0.67	44	**0.83**	0.57	51	**0.86**	0.92
	Baseflow	35	**1.87**	0.40	35	**1.09**	0.89	35	**1.11**	0.87	35	**1.60**	1.42
	Storm EMC	39	**1.31**	0.54	29	**1.01**	0.64	23	**0.88**	0.52	30	**0.79**	0.51
	Grab	39	**1.85**	0.34	39	**1.03**	0.87	39	**0.94**	0.80	39	**1.59**	1.38
TKN (mg/L)	1997	13	**0.12**	0.43	13	**0.33**	0.62	13	**0.26**	0.72	13	**0.17**	0.60
	1998	16	**0.67**	2.21	16	**0.69**	1.20	16	**0.64**	2.04	16	**0.32**	1.02
	1999 Total	60	**0.75**	1.28	50	**1.08**	1.40	44	**0.76**	1.49	51	**1.03**	1.35
	Baseflow	33	**0.35**	0.73	33	**0.33**	0.99	33	**0.26**	1.00	33	**0.16**	1.08
	Storm EMC	39	**0.98**	1.33	29	**1.24**	1.56	23	**0.64**	1.81	30	**1.18**	1.42
	Grab	37	**0.49**	0.97	37	**0.48**	1.14	37	**0.29**	1.07	37	**0.27**	1.09
FLTTKN (mg/L)	1997	13	**0.05**	0.00	13	**0.05**	0.00	13	**0.05**	0.06	13	**0.05**	0.10
	1998	16	**0.30**	1.04	16	**0.20**	0.83	16	**0.22**	0.72	14	**0.13**	0.78
	1999 Total	60	**0.69**	0.90	50	**0.74**	1.04	44	**0.00**	0.95	51	**0.74**	0.95
	Baseflow	33	**0.05**	0.68	33	**0.05**	0.82	33	**0.05**	0.89	32	**0.05**	0.93
	Storm EMC	39	**0.71**	0.93	29	**0.74**	0.86	23	**0.00**	0.85	30	**0.74**	0.90
	Grab	37	**0.05**	0.69	37	**0.10**	1.01	37	**0.06**	0.94	36	**0.05**	0.96
TOTN (mg/L)	1997	13	**2.12**	0.27	13	**2.00**	0.62	13	**1.88**	1.08	13	**2.73**	1.88
	1998	16	**3.11**	1.15	16	**2.22**	1.44	16	**2.65**	2.27	16	**2.68**	1.99
	1999 Total	60	**2.28**	1.02	50	**1.59**	1.13	44	**1.58**	1.09	51	**1.98**	0.83
	Baseflow	33	**2.48**	0.90	33	**1.56**	0.77	33	**1.48**	0.92	33	**2.43**	1.33
	Storm EMC	39	**2.23**	1.23	29	**1.97**	1.39	23	**2.18**	1.47	30	**1.90**	0.67
	Grab	37	**2.48**	0.95	37	**1.58**	0.77	37	**1.49**	0.92	37	**2.15**	1.10
PO4 (mg/L)	1997	16	**0.01**	0.00	17	**0.01**	0.00	17	**0.01**	0.00	16	**0.01**	0.00
	1998	16	**0.01**	0.03	16	**0.03**	0.04	16	**0.02**	0.04	16	**0.01**	0.08
	1999 Total	56	**0.01**	0.02	47	**0.01**	0.02	42	**0.02**	0.02	50	**0.02**	0.02
	Baseflow	34	**0.01**	0.01	34	**0.01**	0.02	34	**0.01**	0.02	35	**0.01**	0.03
	Storm EMC	36	**0.02**	0.02	27	**0.01**	0.04	22	**0.01**	0.03	30	**0.01**	0.02
	Grab	38	**0.01**	0.01	38	**0.01**	0.02	38	**0.01**	0.02	38	**0.01**	0.03
FLTOTP (mg/L)	1997	17	**0.03**	0.00	17	**0.03**	0.00	17	**0.03**	0.00	17	**0.03**	0.00
	1998	15	**0.03**	0.17	16	**0.03**	0.21	16	**0.09**	0.28	16	**0.03**	0.33
	1999 Total	60	**0.07**	0.08	49	**0.07**	0.08	44	**0.06**	0.08	51	**0.06**	0.08
	Baseflow	35	**0.03**	0.04	35	**0.03**	0.07	35	**0.03**	0.07	35	**0.06**	0.07
	Storm EMC	39	**0.10**	0.09	28	**0.10**	0.07	23	**0.10**	0.05	30	**0.05**	0.08
	Grab	39	**0.03**	0.05	39	**0.03**	0.07	39	**0.03**	0.08	39	**0.06**	0.10
TOT-P (mg/L)	1997	17	**0.03**	0.00	17	**0.03**	0.00	17	**0.03**	0.00	17	**0.03**	0.00
	1998	16	**0.35**	0.43	16	**0.36**	0.42	16	**0.38**	0.48	16	**0.36**	0.45
	1999 Total	60	**0.10**	0.12	50	**0.10**	0.16	44	**0.17**	0.19	51	**0.10**	0.12
	Baseflow	35	**0.06**	0.12	35	**0.08**	0.11	35	**0.06**	0.12	35	**0.10**	0.21
	Storm EMC	39	**0.13**	0.16	29	**0.14**	0.16	23	**0.23**	0.20	30	**0.13**	0.14
	Grab	39	**0.06**	0.11	39	**0.08**	0.15	39	**0.06**	0.17	39	**0.10**	0.21

Analyte		QVA n	QVA median	QVA IQR	QVB n	QVB median	QVB IQR	QVC n	QVC median	QVC IQR	QVD n	QVD median	QVD IQR
Fecal Coliform (col/100ml)	1997	17	**250**	544	17	**120**	1009	17	**220**	469	17	**410**	1882
	1998	14	**54**	89	14	**129**	547	14	**109**	421	14	**78**	498
	1999 Total	26	**71**	4170	26	**83**	407	26	**82**	527	26	**104**	460
	Baseflow	35	**50**	96	35	**42**	126	35	**62**	193	35	**72**	175
	Grab	39	**56**	185	39	**50**	277	39	**74**	226	39	**82**	219
Fecal Streptococci (col/100ml)	1997	17	**310**	740	17	**200**	4912	17	**260**	2673	17	**320**	2488
	1998	14	**105**	263	14	**174**	525	14	**310**	751	14	**230**	566
	1999 Total	26	**525**	5937	26	**135**	843	26	**171**	754	26	**187**	728
	Baseflow	35	**84**	242	35	**106**	155	35	**127**	231	35	**118**	234
	Grab	39	**99**	481	39	**118**	197	39	**160**	360	39	**122**	258
Total Coliform (col/100ml)	1997	17	**3100**	67727	17	**8000**	58300	17	**8000**	70100	17	**8000**	52000
	1998	14	**6550**	5800	14	**39000**	85495	14	**45500**	81325	14	**37000**	52875
	1999 Total	26	**8200**	58225	26	**33000**	59325	26	**40500**	73900	26	**34000**	45575
	Baseflow	35	**4400**	6646	35	**5600**	30650	35	**9000**	45900	35	**21000**	35300
	Grab	39	**4900**	6532	39	**8400**	43000	39	**26000**	50200	39	**25000**	38800
Cu (ug/L)	1997	12	**3.5**	3.8	12	**4.0**	4.5	12	**3.0**	3.5	12	**4.0**	3.0
	1998	4	**6.5**	9.5	3	**1.5**	0.5	3	**0.5**	0.3	3	**2.0**	4.5
	1999 Total	18	**3.3**	5.1	18	**4.2**	2.0	18	**3.6**	2.5	18	**3.1**	3.9
	Baseflow	18	**2.7**	2.9	17	**4.0**	3.0	17	**3.0**	2.0	17	**2.0**	3.0
	Grab	22	**2.6**	2.6	21	**4.0**	3.0	21	**3.0**	2.4	21	**2.3**	3.3
Cd (ug/L)	1997	12	**0.07**	0.11	12	**0.09**	0.41	12	**0.10**	0.29	12	**0.10**	0.37
	1998	4	**0.05**	0.05	3	**0.03**	0.04	3	**0.03**	0.00	3	**0.08**	0.04
	1999 Total	18	**0.05**	0.27	18	**0.03**	0.07	18	**0.07**	0.07	18	**0.05**	0.08
	Baseflow	18	**0.03**	0.07	17	**0.03**	0.05	17	**0.06**	0.08	17	**0.05**	0.07
	Grab	22	**0.03**	0.08	21	**0.03**	0.07	21	**0.06**	0.08	21	**0.05**	0.07
Pb (ug/L)	1997	12	**1.0**	1.5	12	**1.5**	1.0	12	**2.0**	1.8	12	**1.5**	1.6
	1998	4	**2.5**	2.3	3	**1.0**	2.0	3	**2.0**	1.0	3	**6.0**	2.0
	1999 Total	18	**2.0**	5.4	18	**1.4**	2.1	18	**1.8**	1.5	18	**1.6**	2.3
	Baseflow	18	**0.8**	1.5	17	**1.0**	1.5	17	**1.6**	1.0	17	**1.3**	1.0
	Grab	22	**1.0**	1.7	21	**1.3**	1.5	21	**1.8**	1.1	21	**1.3**	1.5
Zn (mg/L)	1997	12	**0.03**	0.01	12	**0.03**	0.01	12	**0.03**	0.02	12	**0.03**	0.01
	1998	4	**0.05**	0.02	3	**0.02**	0.02	3	**0.02**	0.01	3	**0.04**	0.01
	1999 Total	18	**0.04**	0.11	18	**0.04**	0.07	18	**0.04**	0.02	18	**0.04**	0.05
	Baseflow	18	**0.02**	0.01	17	**0.03**	0.02	17	**0.03**	0.02	17	**0.03**	0.02
	Grab	22	**0.02**	0.01	21	**0.03**	0.02	21	**0.03**	0.02	21	**0.03**	0.02

Analyte	Season	QVA n	QVA median	QVA IQR	QVB n	QVB median	QVB IQR	QVC n	QVC median	QVC IQR	QVD n	QVD median	QVD IQR
TSS		93	14	41	83	19	32	77	18	147	84	14	42
	Winter	20	16	26	18	20	29	18	14	124	17	16	57
	Spring	27	9	40	23	13	19	23	23	181	25	18	40
	Summer	29	20	85	27	26	81	23	19	162	26	9	39
	Fall	17	4	14	15	11	57	13	13	14	16	8	14
COD		93	14	22	83	11	18	77	16	35	84	15	25
	Winter	20	16	22	18	17	13	18	28	36	17	23	20
	Spring	27	11	39	23	5	12	23	16	38	25	13	26
	Summer	29	14	18	27	11	22	23	8	26	26	13	17
	Fall	17	17	15	15	11	11	13	13	15	16	21	26
TOC		93	1.40	2.70	83	1.75	1.73	77	1.96	1.74	84	1.99	2.19
	Winter	20	1.16	1.13	18	1.46	0.86	18	1.18	0.59	17	1.29	0.49
	Spring	27	1.00	1.97	23	1.68	1.74	23	1.79	1.50	25	1.99	1.67
	Summer	29	2.69	3.02	27	2.71	2.19	23	2.76	1.77	26	3.45	2.66
	Fall	17	2.18	3.12	15	2.08	2.07	13	2.41	1.65	16	2.68	3.60
Temperature		53	13.1	1.6	53	14.5	10.9	54	15.2	10.8	54	15.1	9.2
	Winter	12	11.2	0.9	12	7.9	4.1	12	7.8	4.8	12	7.1	3.9
	Spring	17	13.1	0.6	16	15.1	6.3	16	15.2	4.8	16	15.3	4.5
	Summer	13	14.2	1.7	14	22.4	2.7	15	22.2	3.5	15	21.5	3.0
	Fall	11	13.0	0.9	11	13.4	3.8	11	13.8	3.7	11	14.2	3.6
Conductivity		54	400	108	53	350	110	53	370	110	53	370	110
	Winter	11	450	115	11	440	140	11	430	95	11	450	75
	Spring	17	370	80	16	355	95	16	360	100	16	405	90
	Summer	15	370	105	15	290	140	15	280	150	15	330	140
	Fall	11	410	65	11	340	35	11	340	65	11	370	50
Dissolved Oxygen		53	6.0	2.5	53	7.0	3.0	53	6.0	2.0	53	5.5	2.0
	Winter	11	5.5	1.0	11	6.0	2.5	11	5.0	3.0	11	7.0	2.0
	Spring	16	7.0	2.6	16	6.5	3.9	16	6.5	3.5	16	5.3	3.6
	Summer	15	7.0	2.0	15	7.0	2.8	15	5.5	2.0	15	5.0	1.0
	Fall	11	6.0	1.8	11	7.0	1.5	11	6.0	0.5	11	6.0	0.8
pH		55	7.9	0.6	54	8.0	0.7	54	7.9	0.7	54	7.7	0.6
	Winter	12	7.7	0.5	12	8.0	1.1	12	7.9	1.3	12	7.6	0.9
	Spring	17	8.0	0.5	16	8.1	0.8	16	8.2	1.0	16	8.0	0.6
	Summer	15	7.9	0.9	15	7.7	0.7	15	7.7	0.7	15	7.5	0.5
	Fall	11	8.2	0.2	11	8.2	0.6	11	8.2	0.4	11	7.8	0.3

Analyte	Season	QVA n	QVA median	QVA IQR	QVB n	QVB median	QVB IQR	QVC n	QVC median	QVC IQR	QVD n	QVD median	QVD IQR
NH4		93	0.03	0.06	83	0.03	0.05	77	0.02	0.05	84	0.03	0.05
	Winter	20	0.03	0.04	18	0.03	0.03	18	0.03	0.04	17	0.03	0.03
	Spring	27	0.01	0.02	23	0.01	0.02	23	0.01	0.01	25	0.01	0.04
	Summer	29	0.05	0.07	27	0.05	0.05	23	0.05	0.07	26	0.05	0.07
	Fall	17	0.06	0.10	15	0.05	0.07	13	0.05	0.07	16	0.05	0.06
NO3		93	1.69	0.68	83	1.09	0.83	77	1.08	0.81	84	1.33	1.43
	Winter	20	1.95	0.92	18	1.57	0.84	18	1.65	1.24	17	2.44	1.79
	Spring	27	1.55	0.62	23	0.82	1.15	23	0.97	1.20	25	1.13	1.67
	Summer	29	1.61	0.46	27	0.89	0.52	23	0.72	0.44	26	0.80	0.95
	Fall	17	1.64	0.69	15	1.03	0.33	13	0.86	0.39	16	1.17	0.46
TKN		89	0.49	1.23	79	0.68	1.21	73	0.48	1.32	80	0.45	1.31
	Winter	17	0.00	0.05	15	0.00	0.05	15	0.00	0.05	14	0.04	0.06
	Spring	26	0.11	0.70	22	0.04	0.28	22	0.12	0.78	24	0.08	0.37
	Summer	29	1.28	0.95	27	1.24	0.87	23	1.58	0.92	26	1.29	0.53
	Fall	17	0.74	0.65	15	0.87	0.93	13	0.77	1.08	16	1.34	1.24
Filtered TKN		89	0.05	0.87	79	0.10	0.89	73	0.05	0.89	78	0.11	0.91
	Winter	17	0.00	0.02	15	0.00	0.03	15	0.00	0.03	14	0.00	0.04
	Spring	26	0.02	0.05	22	0.00	0.08	22	0.00	0.05	23	0.01	0.05
	Summer	29	0.93	0.37	27	1.02	0.34	23	0.94	0.38	25	0.90	0.22
	Fall	17	0.71	0.71	15	0.74	0.71	13	0.65	0.69	16	0.74	0.69
Total Nitrogen		89	2.40	1.03	79	1.88	1.15	73	1.74	1.24	80	2.07	1.14
	Winter	17	2.01	1.12	15	1.54	0.86	15	1.49	0.89	14	2.43	1.08
	Spring	26	2.01	1.28	22	1.24	1.43	22	1.22	1.68	24	1.60	1.82
	Summer	29	2.71	0.67	27	2.12	0.78	23	2.33	1.20	26	2.08	0.77
	Fall	17	2.29	0.47	15	2.00	0.86	13	1.88	0.86	16	2.15	0.83
PO4		88	0.01	0.02	80	0.01	0.02	75	0.01	0.02	82	0.01	0.02
	Winter	20	0.00	0.00	18	0.00	0.00	17	0.00	0.00	16	0.00	0.00
	Spring	27	0.03	0.03	22	0.03	0.03	23	0.03	0.03	25	0.03	0.04
	Summer	29	0.01	0.01	27	0.01	0.03	23	0.01	0.01	26	0.01	0.01
	Fall	12	0.01	0.03	13	0.01	0.03	12	0.01	0.03	15	0.01	0.01
Total P		93	0.10	0.20	83	0.10	0.22	77	0.10	0.28	84	0.10	0.20
	Winter	20	0.10	0.15	18	0.09	0.07	18	0.08	0.14	17	0.10	0.16
	Spring	27	0.10	0.13	23	0.10	0.17	23	0.10	0.24	25	0.10	0.15
	Summer	29	0.15	0.23	27	0.24	0.30	23	0.24	0.27	26	0.17	0.20
	Fall	17	0.03	0.09	15	0.03	0.25	13	0.03	0.35	16	0.03	0.15
Filtered Total P		92	0.03	0.07	82	0.03	0.07	77	0.03	0.07	84	0.03	0.07
	Winter	20	0.06	0.07	17	0.03	0.07	18	0.04	0.07	17	0.07	0.07
	Spring	27	0.04	0.07	23	0.03	0.07	23	0.04	0.07	25	0.05	0.07
	Summer	28	0.06	0.11	27	0.07	0.13	23	0.03	0.11	26	0.05	0.12
	Fall	17	0.02	0.03	15	0.03	0.03	13	0.03	0.03	16	0.02	0.03

Analyte	Season	QVA n	QVA median	QVA IQR	QVB n	QVB median	QVB IQR	QVC n	QVC median	QVC IQR	QVD n	QVD median	QVD IQR
Fecal Coliform		57	82	568	57	120	580	57	126	530	57	130	544
	Winter	11	32	47	11	3	9	11	7	25	11	18	61
	Spring	18	174	5042	18	106	444	18	112	149	18	134	476
	Summer	16	205	1408	16	360	3734	16	430	5817	16	280	3246
	Fall	12	49	329	12	299	1262	12	216	1109	12	165	970
Total Coliform		57	6000	35800	57	26000	67000	57	37000	74600	57	29000	52900
	Winter	11	4200	7337	11	2600	3186	11	3100	5077	11	3600	3818
	Spring	18	5500	76400	18	36000	67525	18	35500	69375	18	23000	42250
	Summer	16	7950	42936	16	82000	366750	16	88000	369250	16	52000	348250
	Fall	12	3800	35266	12	39000	59650	12	35000	64800	12	30500	58150
Fecal Strep		57	260	944	57	146	836	57	230	936	57	200	818
	Winter	11	84	546	11	10	95	11	13	127	11	86	232
	Spring	18	215	9018	18	135	478	18	162	750	18	165	680
	Summer	16	490	1763	16	400	5514	16	560	6800	16	377	7658
	Fall	12	70	423	12	208	2002	12	215	816	12	169	815
Cadmium		34	0.07	0.23	33	0.03	0.09	33	0.07	0.08	33	0.07	0.09
	Winter	8	0.04	0.04	7	0.03	0.00	7	0.03	0.00	7	0.05	0.04
	Spring	9	0.42	0.83	9	0.07	0.11	9	0.10	0.14	9	0.10	0.15
	Summer	9	0.08	0.09	9	0.13	0.15	9	0.10	0.19	9	0.12	0.28
	Fall	8	0.03	0.04	8	0.03	0.05	8	0.04	0.05	8	0.03	0.01
Copper		34	3.3	5.4	33	4.0	3.0	33	3.5	3.4	33	3.2	3.5
	Winter	8	3.0	3.4	7	2.0	1.3	7	2.0	3.5	7	4.0	2.9
	Spring	9	5.1	6.3	9	3.4	2.3	9	3.6	3.0	9	5.0	4.6
	Summer	9	3.7	5.0	9	4.8	2.1	9	4.2	4.0	9	4.5	2.4
	Fall	8	2.8	3.1	8	4.3	2.1	8	2.8	3.3	8	2.0	0.6
Lead		34	1.9	2.5	33	1.3	1.3	33	2.0	1.7	33	2.0	2.1
	Winter	8	2.0	0.9	7	1.3	1.6	7	2.0	1.2	7	2.7	3.5
	Spring	9	2.0	6.3	9	0.5	0.9	9	1.0	1.6	9	0.5	1.0
	Summer	9	1.7	2.5	9	3.0	3.7	9	2.8	1.2	9	2.3	1.5
	Fall	8	0.8	1.7	8	1.3	1.0	8	1.9	0.8	8	1.3	2.2
Zinc		34	0.03	0.03	33	0.03	0.03	33	0.04	0.02	33	0.03	0.02
	Winter	8	0.02	0.02	7	0.02	0.01	7	0.02	0.02	7	0.03	0.01
	Spring	9	0.13	0.29	9	0.07	0.13	9	0.04	0.02	9	0.08	0.12
	Summer	9	0.03	0.02	9	0.04	0.01	9	0.04	0.02	9	0.04	0.01
	Fall	8	0.02	0.01	8	0.02	0.01	8	0.03	0.01	8	0.02	0.01

Analyte	QVA			QVB			QVC			QVD		
	n	median	IQR	n	median	IQR	n	median	IQR	n	median	IQR
COD (g)	34	59647	107402	28	74310	82039	24	233588	322807	30	155346	338305
Flow (L)	34	2351301	3597691	28	3169236	3273604	23	4152913	4799662	31	7788545	7502999
FLTKN (g)	34	533	1677	28	744	3071	24	0	3887	30	2354	5865
FLTOTP (g)	34	199	421	27	273	567	24	406	758	30	237	839
NH4 (g)	34	100	160	28	133	167	24	189	360	30	276	409
NO3 (g)	34	2830	4047	28	3640	3153	24	3675	4547	30	6295	9705
PO4 (g)	31	41	82	26	65	145	23	42	104	30	87	139
TKN (g)	34	913	2779	28	3529	6593	24	2205	11122	30	3507	11284
TOC (g)	34	7924	12110	28	9359	10292	24	9315	11826	30	27223	36280
TotN (g)	34	4174	5166	28	6071	6370	23	10077	13920	30	11834	14489
TOT-P (g)	34	355	873	28	593	574	24	1442	1500	30	770	1549
TSS (kg)	34	125028	178720	28	189972	238410	23	1150547	1635965	30	469364	1001206

Baseflow	QVA			QVB			QVC			QVD		
Analyte	n	median	IQR	n	median	IQR	n	median	IQR	n	median	IQR
Temperature (C)	33	13.10	1.3	34	15.20	12.0	35	15.70	11.8	35	15.10	9.9
pH	35	8.01	0.5	35	8.00	0.9	35	7.94	0.8	35	7.80	0.8
Conductivity (uOhm)	35	400	105	35	350	100	35	370	90	35	370	80
Dissolved Oxygen (mg/L)	34	7.00	2.0	34	7.50	2.0	34	6.50	3.0	34	5.75	2.8
TSS (g/L)	35	4.00	5.0	35	7.00	6.5	35	8.00	7.5	35	7.00	9.2
TOC (mg/L)	35	0.62	0.3	35	1.21	1.5	35	1.52	1.8	35	1.29	1.1
COD (mg/L)	35	8.00	13.5	35	6.00	13.5	35	8.00	10.5	35	7.00	16.5
TOT-P (mg/L)	35	0.06	0.1	35	0.08	0.1	35	0.06	0.1	35	0.10	0.2
PO4 (mg/L)	34	0.01	0.0	34	0.01	0.0	34	0.01	0.0	35	0.01	0.0
FLTOTP (mg/L)	35	0.03	0.0	35	0.03	0.1	35	0.03	0.1	35	0.06	0.1
NO3 (mg/L)	35	1.87	0.4	35	1.09	0.9	35	1.11	0.9	35	1.60	1.4
NH4 (mg/L)	35	0.03	0.1	35	0.01	0.0	35	0.02	0.0	35	0.03	0.1
TKN (mg/L)	33	0.35	0.7	33	0.33	1.0	33	0.26	1.0	33	0.16	1.1
FLTTKN (mg/L)	33	0.05	0.7	33	0.05	0.8	33	0.05	0.9	32	0.05	0.9
TOTN (mg/L)	33	2.48	0.9	33	1.56	0.8	33	1.48	0.9	33	2.43	1.3
Total Coliform (col/100ml)	35	4400	6646	35	5600	30650	35	9000	45900	35	21000	35300
Fecal Coliform (col/100ml)	35	50	96	35	42	126	35	62	193	35	72	175
Fecal Streptococci (col/100ml)	35	84	242	35	106	155	35	127	231	35	118	234
Cd (ug/L)	18	0.03	0.1	17	0.03	0.0	17	0.06	0.1	17	0.05	0.1
Cu (ug/L)	18	2.70	2.9	17	4.00	3.0	17	3.00	2.0	17	2.00	3.0
Pb (ug/L)	18	0.75	1.5	17	1.00	1.5	17	1.60	1.0	17	1.30	1.0
Zn (mg/L)	18	0.02	0.0	17	0.03	0.0	17	0.03	0.0	17	0.03	0.0

Analyte	EMC Removal Efficiency (%)								
	Wet Pond			**Dry Pond**			**System**		
	n	**median**	IQR	n	**median**	IQR	n	**median**	IQR
Chemical Oxygen Demand	28	**20.8**	86	20	**45.3**	33	30	**12.5**	149
Filtered TKN	16	**0.4**	27	10	**0.6**	17	18	**1.3**	23
Filtered TP	24	**1.9**	51	19	**12.8**	35	26	**15.8**	54
Ammonia	25	**8.0**	27	17	**7.5**	25	26	**0.8**	39
Nitrate	28	**18.7**	40	20	**8.9**	46	30	**26.4**	48
Orthophosphate	26	**0.0**	15	19	**0.0**	3	26	**0.0**	43
Total Khedahl Nitrogen	24	**-26.3**	64	16	**36.3**	35	26	**-0.1**	67
Total Nitrogen	28	**5.1**	39	20	**14.2**	50	30	**11.2**	32
Total Organic Carbon	28	**13.1**	32	20	**-14.2**	153	30	**-3.8**	77
Total Phosphorus	26	**2.2**	56	19	**36.6**	37	27	**29.6**	110
Total Suspended Solids	28	**19.1**	194	20	**68.7**	30	30	**10.3**	163

APPENDIX E

Station QVG data and statistics

Sampling Location QVG Results	3/3/97	4/13/97	6/3/97	1/28/98	2/11/98	4/20/98	5/27/98	9/5/99	**Mean**	**Median**	**1st quartile**	**3rd quartile**	**n**
TSS (g/L)	40	4	27	23	51	4	176		46	27	14	46	7
TOC (ppm)	11.6	0.9	14.9	2.6	3.5	4.9	20.4		8.4	4.9	3.0	13.3	7
COD (ppm)	86	13	68	49	20	25	20		40	25	20	59	7
TOT-P (ppm)	1.25	0.03	1.76	0.01	0.13	1.33	2.71		1.03	1.25	0.08	1.55	7
PO4 (ppm)	0.85	0.01	1.57	0.16		0.42	2.36		0.89	0.63	0.23	1.39	6
FLTOTP (ppm)	0.99	0.03	1.57	0.03	0.08	2.03	1.90		0.94	0.99	0.05	1.73	7
NO3 (ppm)	2.91	2.26	1.14	0.64	6.16	3.08	0.98		2.45	2.26	1.06	2.99	7
NH4 (ppm)	1.09	0.01	0.13	0.01	0.01	0.01	0.01		0.18	0.01	0.01	0.07	7
TKN (ppm)	4.09	0.10	3.54	0.49	0.05	0.05	2.77		1.58	0.49	0.08	3.16	7
FLTTKN (ppm)	3.57	0.05	2.90	0.05	0.05	0.05			1.11	0.05	0.05	2.18	6
TOTN (ppm)	7.00	2.36	4.68	1.13	6.21	3.13	3.75		4.04	3.75	2.74	5.45	7
Temperature (C)	8.1	16.2	14	1.3	8.4	18.5	17.6		12.0	14.0	8.3	16.9	7
pH	7.9	7.4	7.4	7.3	7.1	7.9	7.5		7.5	7.4	7.4	7.7	7
Conductivity	430	470	410		510	520	330		445	450	415	500	6
Dissolved Oxygen	4	4	3	4	7	8	5		5	4	4	6	7
Total Coliform (col/100ml)	240000	7500	530000	8500	3000	29000	850000		238286	29000	8000	385000	7
Fecal Coliform (col/100ml)	60000	6000	60000	55	10	540	6800		19058	6000	298	33400	7
Fecal Streptococci (col/100ml)	89000	360	95000	450	166	700	9900		27939	700	405	49450	7
Cd (ug/L)				0.025	0.025			0.17	0.073	0.025	0.025	0.098	3
Cu (ug/L)				3	2			7.6	4.2	3.0	2.5	5.3	3
Pb (ug/L)				2.0	2.0			1.0	1.7	2.0	1.5	2.0	3
Zn (mg/L)				0.17	0.04			0.07	0.09	0.07	0.06	0.12	3

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