PERFORMANCE ANALYSIS OF THE ASHBY STORMWATER RETENTION POND IN THE CITY OF FAIRFAX, VIRGINIA

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ABSTRACT

Ashby Pond in the City of Fairfax, Virginia was retrofitted to treat runoff from 54.7 hectares of urban land of mixed use. The pond discharges into Accotink Creek, a highly urbanized tributary of the Potomac River and Chesapeake Bay that is listed on the State of Virginia 303(d) list for multiple impairments. The entire multi-state Chesapeake Bay Watershed is subject to Total Maximum Daily Load (TMDL) restrictions on sediment, phosphorus and nitrogen. Virginia and local municipalities assign pollutant reduction credits to retention ponds that meet certain design requirements. However, to actually meet existing and future water quality goals set by TMDLs, it must be proven that such ponds truly provide the water quality benefits for which they have been credited. The inflow and outflow water quality of Ashby Pond was examined over 7 months from fall 2012 to spring 2013. During that period, the pond provided statistically significant reductions of phosphorus, nitrogen and suspended sediment, but not organic carbon or oxygen demand. Ashby Pond had non-significant export of sodium, chloride and calcium. The pond underperformed when compared to state reduction credits for phosphorus load and concentration, but met and exceeded the credits for nitrogen load and concentration, respectively. The pond was under-sized compared to state design standards, and some underperformance should be expected.

DEDICATION

To my understanding wife Dana: Look, it's done! Now we can spend time with each other!

ACKNOWLEDGEMENTS

I'd like to thank all my committee members—Drs. Tom Grizzard, David Sample and Glenn Moglen—for their time, expertise and thoughtful suggestions. Your help was vital in crafting this thesis. I'd like to thank Drs. Sample and Grizzard for organizing the Ashby Pond project and thinking of me when it came time to choose a graduate student to analyze the data. Dr. Grizzard, my graduate advisor and committee chair, helped me get over the inevitable (and multiple) bumps in the road during my analysis, provided excellent edits to my thesis, and was also the faculty member who guided me into the Environmental Sciences and Engineering program to begin with. Drs. Sample and Moglen also provided careful edits and advice as this thesis evolved (and improved greatly) from its original rough draft to the final version presented here. I'd also like to thank all the staff and my fellow graduate students at the Occoquan Watershed Monitoring Lab.

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1. INTRODUCTION

Ashby Pond is a stormwater management pond in City of Fairfax, Virginia. It has a 54.7 hectare watershed of mixed urban and suburban land use. The Virginia Tech Biological Systems Engineering Department (BSE), the Occoquan Watershed Monitoring Laboratory (OWML) of the Virginia Tech Department of Civil and Environmental Engineering (CEE), and the City of Fairfax, using grant funds from the National Fish and Wildlife Foundation (NFWF), conducted an extensive retrofit of the pond. Based upon a design developed by BSE, the pond was dredged, the bottom re-graded, and a new concrete outflow structure was built. Construction work was performed by the City of Fairfax. Grant funds were also used to install all water quality and quantity monitoring equipment, and to provide funding for graduate student research assistantships. The retrofitted Ashby Pond was host to two studies: a performance analysis of floating treatment wetlands (FTWs) established in the pond, and this performance analysis of stormwater treatment provided by the pond itself. The overall project and the FTW sub-study were directed by Professor David Sample of the BSE Department; the pond performance sub-study was directed by Professor Thomas Grizzard of the CEE Department. Together, Professors Sample and Grizzard were responsible for the application, financial management, and reporting requirements of the grant, as well as project management of the pond retrofit and data collection. A BSE PhD student, C.Y. Wang, performed the FTW experiments and analysis, and an MS student in Environmental Sciences and Engineering (ESEN), Dan Schwartz, performed data analysis of the stormwater treatment provided by Ashby Pond. Design, installation, operation, and maintenance of the site instrumentation and water quality data collection were provided by faculty and staff of the OWML.

2. LITERATURE REVIEW

2.1 INTRODUCTION TO URBAN STORMWATER POLLUTION AND MANAGEMENT

The hydrologic cycle in natural landscapes is dominated by the infiltration of rainfall by soils, and the return of water to the atmosphere by transpiration from plants and evaporation from land and plant surfaces. The volume of precipitation remaining after these reductions is available to flow over the land surface as runoff. This runoff can pick up, entrain or dissolve loose materials on the land surface—both man-made and natural—and flush them into the nearest receiving waters: streams, lakes, rivers, bays, and eventually the oceans. In the natural environment, the total amount of runoff is small, as is its total pollutant load. In urbanizing landscapes, stormwater volumes increase markedly and the developed land surface acts as a source of potentially harmful materials to mobilize (Leopold, 1968). For this reason, and the increase in urbanization worldwide, stormwater is a particularly pressing issue in the control of nonpoint source pollution.

The term nonpoint source pollution signifies the diffuse nature of this form of pollution—it originates from multiple sources and areas at once—in contrast to point-source pollution whose source is centralized and easy to identify, such as the wastewater outfall of an industrial site (NVPDC and ESI, 1992). The seminal environmental law in the United States addressing water pollution—known commonly as the Clean Water Act (CWA)—targeted both point and nonpoint source pollution. However, it was many years after the 1972 passage of the CWA before the federal and state governments began to implement the nonpoint source pollution regulations authorized by the law (Novotny and Olem, 1994). Language in the CWA that requires states to regularly monitor and report the condition of surface waters within their jurisdictions (USEPA,

2009) has raised awareness in both the regulatory and the public sectors of the magnitude of the negative effects caused by diffuse stormwater.

> "Put simply, the vast majority of our nation's impaired waters have no possibility of being restored unless the nonpoint sources affecting those waters are effectively remediated. Moreover, unless nonpoint sources are more effectively addressed, we will continue to see the number of impaired waters grow over time." (USEPA, 2011a)

While once lightly regarded, stormwater is now rightly seen as one of the most important causes of water quality degradation, and its effective control is a major hurdle that must be overcome to achieve the ultimate goal of the CWA, restoration of the "chemical, physical and biological integrity" of US waters (33 U.S.C. 1251 *et seq*., 2002). In the sections that follow, the causes, sources, constituents and effects of stormwater will be discussed, as well as the regulations designed to control it. Later sections will address the design and effectiveness of the stormwater wet pond—or retention pond—in reducing the harmful effects of stormwater pollution.

2.2 HYDROLOGIC EFFECTS OF URBANIZATION

During the urbanization process, large areas of permeable soils are rendered impervious by a capping of roads, footpaths, rooftops and parking lots (Hollis, 1975). This results in a large decrease in infiltration, groundwater recharge, soil moisture and base flows; and a proportional increase in the volume of surface runoff . (Booth *et al*., 2002; Dietz and Clausen, 2008; Fletcher *et al.*, 2013; Hollis, 1975; Horner *et al.*, 1994; Jacobson, 2011; Leopold, 1968; Schueler, 1994; USEPA, 1992a; Wang *et al.*, 2001; Yang *et al*., 2011. The altered hydrology causes widespread impacts on receiving bodies, including in-stream erosion (Allmendinger *et* al., 2007; Nelson and

Booth, 2002), reductions in the populations of sensitive aquatic species and increases in the populations of tolerant species (Alberti *et al*., 2007), increased inputs of nutrients and toxins (Carey *et* al., 2013; Walsh *et al*., 2005), and increases in water temperature (Nelson and Palmer, 2007).

Evidence of the correlation between the amount of impervious surfaces in a watershed and the volume of runoff created abounds. After urbanization, average peak flows can be 2 to 4 times greater than pre-development conditions (Chin, 2006). Total runoff volumes have shown a positive linear correlation with increasing watershed imperviousness (USEPA, 1983; Novotny and Olem, 1994; Schueler, 1994), but for small watersheds with little ability to attenuate runoff spikes, the increase may be logarithmic (Dietz and Clausen, 2008). Results reported from the Nationwide Urban Runoff Program (NURP)—an EPA-funded study that was one of the first to provide in-depth characterizations of stormwater quality, quantity and treatment—presented the runoff coefficient, the ratio of runoff depth to rainfall depth, for each of the nationally distributed study sites (USEPA, 1983). While the storm-to-storm runoff coefficients from each site were lognormally distributed, the median site coefficients showed a strong linear correlation with the percent watershed imperviousness (USEPA, 1983). This relationship is shown in Figure 2.2.1. Novotny and Olem (1994) found that above 20% watershed imperviousness, each 10% increase in the amount of imperviousness results in a 0.1 increase in the runoff coefficient.

An explanation of the differences in correlation could be that the Dietz and Clausen study focused on a particularly small drainage area. Smaller watersheds respond more quickly to storm events, whereas larger watersheds have a greater capacity to prolong and dampen runoff spikes (Characklis and Wiesner, 1997). It would follow, then, that urbanization may cause especially large changes in the hydrology of smaller, headwater streams (Dietz and Clausen, 2008).

Figure 2.2-1 Watershed Imperviousness and Median Runoff Coefficients

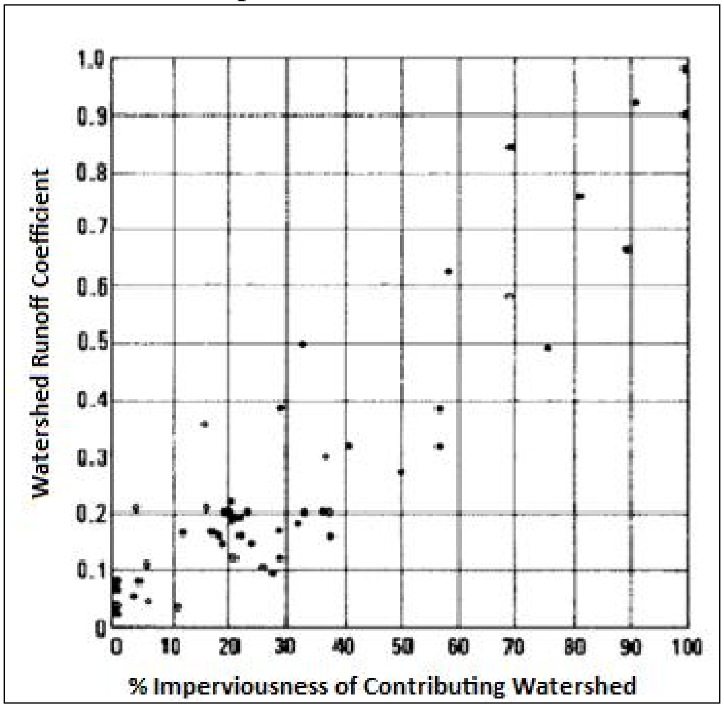


Image Credit: (USEPA, 1983). [Public Domain]

Imperviousness may the most visible cause of stormwater volume increases, but the effect of soil changes and grading should not be underestimated. In undeveloped landscapes, the topography is varied, and many small depressions are present that can trap and hold water. These depressions are graded out to create the smoother, free draining surface that characterizes the urbanized landscape, and the stormwater retention they provided is lost (Hollis, 1975). In all but the areas of most intense urban land use, significant pervious surface remains, such as lawns, medians and playing fields. Yet the re-graded soils found in these pervious surfaces are not natural; they have been compacted and are in marginal to poor condition, infiltrating less rainfall and creating more surface runoff than in their former state (Hancock and Chambers, 2010). In fact, the fundamental hydrologic effect of urbanization has been stated to be the loss of water storage in the soil column, caused by both the increase in impervious surfaces as well as the compaction of remaining soil (Booth *et al.,* 2002). In addition, changes in vegetative cover can also cause increases in runoff. The combination of soil compaction and the conversion of mature forest to suburban vegetation, mainly lawns, was shown to be far more significant in determining the resulting peak discharges of stormwater from developing rural areas than the small increases in impervious surfaces (Booth *et al.,* 2002).

All fresh water is part of the water cycle, and its total volume is finite and divided amongst only a few compartments: surface water, groundwater, atmospheric water and ice. Volume increases in one compartment must reduce the volume of water held in others. In the case of urbanization, stormwater and storage in soils and groundwater may have an antagonistic relationship: increases in runoff are matched by proportional decreases in groundwater storage and soil moisture (Horner *et al*., 1994; Jacobson, 2011; Leopold, 1968; USEPA, 1992a;; Wang *et al*., 2001). Since groundwater is the primary source for the maintenance of dry weather flows in streams and rivers, a reduction in groundwater may manifest as a similar reduction in base flow (Horner *et al*., 1994). In a study of US mid-Atlantic Piedmont streams, base flow typically declined to 30% of its pre-development average when the upstream watershed surpassed 30-45% imperviousness, and less than 10% of average when 65% imperviousness was reached (Klein 1979). However, other studies found inconsistent correlations between urbanization and base flow (Walsh *et al*., 2005), and a few even observed increases, likely due to pipe leakage and irrigation (Jacobson, 2011; Schueler, 2009).

Perhaps even more dramatic than the decline in base flow is the effect of urbanization on the peak flows of receiving waters. Peak flows are not just affected by the increased volume of runoff, but by the velocity of the runoff as well. Runoff velocity is increased by the man-made surfaces over which it flows in urbanized environments–sheet flow and concentrated flow over smooth, impervious paving and uniform grass rather than the more tortuous flow paths provided by the forested floor (Fletcher *et al.*, 2013; Jacobson, 2011). In addition, man-made conveyances (e.g., storm drains, pipes, concrete culverts) efficiently remove most runoff from streets and lawns (Horner *et al.*, 1994) and convey it to receiving waters at velocities much greater than could be achieved in natural channels (Hollis, 1975). This results in a reduced time of concentration—the time it takes for runoff from the most distant part of a watershed to reach the common outlet—and thus allows more runoff to flow into the channel in a condensed span of time (Horner *et al.*, 1994). A small developed watershed may now have a time of concentration of only a few minutes, whereas in its natural state there could have been a lag of hours, days or even weeks before all the storm flow reached the receiving channel (Booth *et* al., 2002). The net effect of increases in runoff volume and decreases in time of concentration is flood flows that are larger, flashier and more frequent in the receiving catchment than they were before urbanization (Chin, 2006).

Flashy stormwater flow has the effect of increasing both the size of floods and their rate of recurrence (Booth *et al*., 2002; Wang *et al*., 2001). In one of the early comprehensive analyses of the effect of urbanization on hydrology, Leopold (1968) found that streams construct their channels so as to pass the bankfull flood volume, and that flows at or above this level begin to cause changes in channel morphology. Others have concluded that sub-bankfull floods can also be responsible for morphological changes (Schueler, 1994). Nelson and Booth (2002) found that stream channel morphology directly correlated with the peak flow rate of the 2-year recurrence interval storm. In undeveloped watersheds, bankfull flows have a mean recurrence interval of 1.5 to 2 years (Leopold, 1968; Schueler, 1987). Urbanization greatly increases the frequency of bankfull events, and the increase has a near linear relationship with the level of imperviousness in the contributing watershed (Schueler, 1987). Bankfull flows have a mean recurrence interval of 1.5 to 2 years in undeveloped watersheds (Leopold, 1968; Schueler, 1987). Urbanization greatly increases the frequency of bankfull events, and the increase has a near linear relationship with the level of imperviousness in the contributing watershed (Schueler, 1987). Leopold (1968) found that a watershed that is only 20% impervious would double its bankfull frequency.

For larger flood events, as the recurrence interval and flood size grows larger, the effect of urbanization on increased flooding frequency grows less pronounced (Hollis, 1975; Yang *et al*., 2011). However, the flow volume of these storms, if not necessarily their recurrence intervals, can be greatly increased (Booth and Jackson, 1997), with peak flow rates for a 100 year storm potentially doubled in a watershed that is 30% or more impervious (Hollis, 1975).

The net effect of urbanization is often a flow regime that is much more variable, with higher peak flows and reduced base flows (Poff *et al*., 2006). This is well illustrated in a study by Bryan (1972) that examined the water quality and flow characteristics of a stream draining a heavily urbanized section of Durham, NC. The base flow of the stream averaged 0.1 ft^3/s, compared to the highest recorded storm flow of 700 ft^3/s. The factor of 7,000 over the range amply illustrates the flashiness of storm flow as well as the reduction of inter-storm base flow contributions to urban streams.

Increased erosion, both on land and within receiving channels, is an unfortunate side effect of the flashy nature of urban stormwater flows. The US Department of Agriculture-Natural Resources Conservation Service (USDA-NRCS) Universal Soil Loss Equation (USLE), Equation (1), presents a broadly-accepted way to estimate annual erosive losses of soil from land surfaces (Horner *et al*., 1994):

$$E=RKLSCP \tag{1}$$

The E variable represents annual erosion loss in tons per acre per year. All other variable are unitless: R is the rainfall erosion index, K is the soil erodibility factor, L and S are combined to create the slope length and steepness factor, C is the land cover factor and P is the erosion control factor. As may be seen from Equation (1), the above factors have a linear relationship with total erosive losses. In Horner *et al's* (1994) analysis of the equation, the C and P factors are the variables most affected by urbanization. The value of the C factor is determined by the type of vegetative cover applied to the land while the P factor relates to the compaction and smoothness of the soil. The C factor is crucially important during the active construction phase of urbanization, when land is being graded and soil is denuded. Bare soil has a C factor of 1; the value for lawns established either by seed or sod is 0.01, meaning that an active, denuded construction site will suffer 100 times the amount of erosion it would if it were grassed, all other factors being equal (Horner *et al.,* 1994). This is a dramatic increase, and other researchers have remarked upon the high concentrations of eroded sediment attributable to construction sites (Bryan,1972; GC and WWE, 2012a; Nelson and Booth, 2002; Sonzogni *et al.*, 1980). However, it is a temporary increase, assuming that the site is stabilized after completion of construction, so most of the effect of urbanization on the C value will be short term. The P value, however, reflects the porosity of the soil, and as has been demonstrated, compaction and reduced porosity can be permanent post-urbanization characteristics of soil (Booth *et al*., 2002; Hancock and Chambers, 2010). According to Horner *et al.* (1994), soil that has been machine compacted and smoothed has a P value of 1.3, whereas rough soil that has been loosened to deeper than 12 inches receives a value of 0.8. The compacted and smoothed condition is typical of post-construction soil, and the difference in P values shows that a soil such as this will suffer 40% more erosive losses as a result (Horner *et al*., 1994). While the increase is not as dramatic as the changes in pre- and post-construction C value, it is generally permanent.

In response to active construction, urban stream beds often fill with eroded sediment (Chin, 2006). However, after installation of impervious surfaces and stormwater conveyances is complete, this accumulated sediment is flushed out and stream beds and banks begin to scour to create the wider and deeper channel needed to accommodate the larger flow volumes and rates of the urban watershed (Chin, 2006; Wang *et al.,* 2001). The onset of in-stream erosion may be observed when there is as little as 10% imperviousness in the contributing watershed (Schueler, 1994). Abraded materials from the stream banks and bed and from the soils within the watershed become a pollutant in themselves (Nelson and Booth, 2002), contributing to water quality problems downstream.

2.3 THERMAL EFFECTS OF URBANIZATION

The thermal regime within receiving waters may also change significantly as a result of urbanization. As forest and fields are removed, impervious paving and rooftops take their place. To a much greater degree, these materials absorb and re-emit a significant amount of heat (Schueler, 1994). Runoff traversing these surfaces can become heated itself (Nelson and Palmer, 2007), as may the surrounding air (Galli, 1990). The loss of forest canopy also exposes surface water and soil to the direct irradiance of the sun (Hewlett and Fortson, 1982; Nelson and Palmer, 2007). Even when the riparian canopy is untouched by development directly, flashy urbanized streams expose themselves to increasing sunlight when they erode and widen their banks, causing the felling of near-bank trees and a widening gap in the canopy immediately above (Wang *et al*., 2001).

A high correlation between stream temperatures and watershed imperviousness has been noted (Nelson and Palmer, 2007). In Maryland Piedmont headwater streams, the relationship between the mean late spring to summer water temperature and watershed imperviousness has been represented by Equation (2) (Galli, 1990):

$$°F=60.4+0.14(\%\ Watershed\ Imperviousness) \tag{2}$$

The stream with the least amount of watershed imperviousness (1%) was on average 8.6°F cooler than the stream with the highest imperviousness (60%) throughout the duration of the study (Galli, 1990).

In an examination by Hewlett and Forston (1982) of thermal changes caused by the loss or riparian shading, two southeast piedmont watersheds of almost identical temperature characteristics were studied after one of the watersheds was clearcut for a lumber harvest. In the clearcut watershed, a narrow patchy riparian buffer of trees was left behind. The daily maximum water temperatures in the stream draining the cleared land rose in nearly all months compared to the forested watershed, but the winter daily minimum water temperatures also decreased, suggesting that a healthy forest overstory buffers stream temperatures in both directions. The total amplitude of temperature changes in the clearcut watershed, lacking this riparian buffer, increased by 21°F compared to the forested watershed. In Maryland, stream temperatures were observed to rise between 3.5° and 3.7°F after flowing by 150 to 300 feet of partially or completely open riparian buffer (Galli, 1990).

Different conclusions have been drawn as to the exact mechanisms of warming. A surprising result from the Maryland study was that base flow water temperatures in the monitored streams were actually higher than storm flow temperatures, leading to the conclusion that ambient air temperatures were a more important determinant of stream temperature than the inflow of heated stormwater (Galli, 1990). The author attributed this finding to the fact that during most storms, conditions were cloudy and ambient air temperatures were generally cooler than during base flow conditions. Contradicting this, other studies have found predictable stream temperature increases when rainstorms struck during warm summer months (Nelson and Palmer, 2007). Hewlett and Forston (1982) concluded that the heating observed in the clearcut watershed was due to changes in the temperature of the groundwater, although increased air temperature likely played a secondary role. Klein (1979) investigated the role of groundwater as well, but found that groundwater inflows in the mid-Atlantic generally ranged from only 10° to 11°C. This thermally stable input served as a buffer against both high and low temperatures and kept stream areas near the groundwater inflow in the general range of 7.8° to 20°C year round. Of course, urbanization, because of its ability to reduce groundwater recharge, could reduce this buffer and thus the ability of a stream to stay within this range (Klein, 1979).

2.4 URBAN STORMWATER QUALITY

2.4A Stormwater Pollutant Sources and Types

General nonpoint source runoff is the leading cause of water pollution impairment nationwide (USEPA, 2011a). While agriculture is the greatest source of impairments (USEPA, 2011a), on a per-unit-area basis, urban lands are a more concentrated source (USEPA, 1992a). Additionally, because of the growth of cities, the contribution of urban sources to total nonpoint source pollution is expected to increase significantly (NRC, 2009). Urban stormwater is currently thought to be the 5th most important source of impairment in US streams and rivers and the 10th most important source of impairment for estuaries (USEPA Attains Database. Accessed January, 2013, http://www.epa.gov/waters/ir/).

The Nationwide Urban Runoff Program (NURP) adopted the following as “standard pollutants characterizing urban runoff” (Strecker, 1999).

Total suspended solids (TSS)

Biochemical oxygen demand (BOD)

Chemical oxygen demand (COD)

Total Phosphorus (TP)

Soluble phosphorus (SP)

Total Kjeldahl nitrogen (TKN)

Oxidized nitrogen (OX-N): nitrite (NO_2^-) and nitrate (NO_3^-) combined

Copper (Cu)

Lead (Pb)

Zinc (Zn)

Other pollutants commonly found in elevated concentrations in urban stormwater include byproducts of automotive use and wear (Sartor and Boyd, 1972; Shaheen, 1975; USEPA, 1990; USEPA, 1992a; GC and WWE, 2012a); bacteria, such as fecal coliforms and streptococci (Shaheen, 1975; Hvitved-Jacobsen, 1990); and dissolved salts (Hawkins and Judd, 1972; Klein, 1979; USEPA, 1992a).

Trace metals are thought to be pollutants of particular concern because they are commonly found in urban runoff and may be toxic to aquatic life (USEPA, 1983; Strecker, 1999). Cu, Pb and Zn are naturally weathered, at slow rates and in small quantities, from exposed soils and mineral deposits (Strecker, 1999). Although direct emissions from automobiles is estimated to account for only 5% of the total accumulation of roadway detritus, this fraction is of particular importance because of the high percentages of toxics it contains, including metals (Shaheen, 1975; Novotny and Olem, 1994). The heavy metals found in roadway sediments, and their automotive sources, are as follows:

-Zinc: Used in tires and as a stabilizing agent in motor oil (Sartor and Boyd, 1972; Shaheen, 1975)

-Cadmium (Cd): Used in tires (USEPA, 1992a)

-Copper, Nickel (Ni), and Chromium (Cr): Found in metal platings, bushings, bearings and other moving parts in the engine. Cu especially found in brake linings (Shaheen, 1975)

-Lead: Almost all lead used in leaded gasoline, but some found in automotive batteries, and lead oxide found in tires (Shaheen, 1975; USEPA, 1990).

While roadways are responsible for a large amount of stormwater metal loadings, there are other sources. Zn is commonly used for roof flashing, gutters and downspouts, and has the benefit of greatly buffering the acidity that is common in urban rainfall (Novotny and Olem, 1994). However, acidic rainfall also causes a significant amount of Zn to solubilize, causing the concentrations of dissolved Zn in roof runoff to be enriched several hundred times when the pH of rainfall is less than 4 (Novotny andOlem, 1994). Rainfall in the Washington, DC metropolitan area has been reported to often have a lower pH (Grizzard, 2013).

Pb, Cd, Cu and Zn are not the only metals present in urban runoff, but they are commonly used to characterize pollutant metal loads because they are nearly ubiquitous and represent the general range of transport mechanisms (Strecker, 1998). It should be noted that Pb most likely poses a much less serious threat of water pollution in the US due to the ban on leaded gasoline. In a post-ban study of first flush highway runoff from the Washington, DC area, Pb was found to almost always be below detection limits despite the fact that first flush runoff should theoretically contain the highest percentages of pollutants (Dorman, 1996). A study from the late 1990s that sought to update the original NURP characterizations of stormwater quality also found that Pb concentrations had dramatically decreased (Smullen *et al*., 1999). A mid 1990s study in the Washington, DC suburbs found that the extractable lead median concentration was about 4% of the NURP median (Schehl, 1995). These reductions can likely be attributed to the phasing out of leaded gasoline starting in the mid-1970s (Schehl, 1995).

Nutrients, primarily nitrogen and phosphorus (N and P), also have a variety of sources in the urban environment which could lead to runoff enrichment. Common sources are listed in Table 2.4-1.

Table 2.4-1. Sources of Nutrients in Stormwater

Organic N and P	Inorganic N and P
Vegetative -Grass Clippings[1] -Leaves[1] -Pollen[2] -Seeds[2]	Fertilizers[3,4] Soil Erosion[3, 4] Fuel Combustion[3], * Flame Retardents[3], ** Corrosion Inhibitors[3],**
Sewage Wastes - Combined Sewer Overflows[3] -Pets[4] -Wildlife[3] -Septic Tank Leachate[3] -Cross Connection of Sanitary and Storm Sewers[4]	Plasticizers[3], **
Landfill Leachate[1]	**Sources** [1]Novotny and Olem, 1994 [2]Sartor and Boyd, 1972 [3]GC and WWE, 2012a [4]USEPA, 1992a **Symbols** *=N source only **=P source only
Soil Erosion[3]	

Bacteria–including fecal coliforms, E. coli, enterococci, and streptococci–may be found in elevated levels in urban runoff (USEPA, 1983; Williamson, 1985; Schueler, 1987; Hvitved-Jacobsen, 1990). While many stormwater studies focused their bacterial analysis on fecal coliforms, the EPA now recommends that E. coli or enterococci be used as indicators of water quality because of the better correlation with gastroenteritis in swimmers (USEPA, 2012). Bacterial analysis of stormwater can be hard to correlate directly to receiving water quality because many indicator bacteria die out quickly in the natural water environment (Sartor and

Boyd, 1972). Despite this uncertainty, stormwater is clearly a source of elevated bacterial concentrations (Struck *et al*., 2008; Winer, 2000).

Suspended sediments in stormwater may be divided into two general categories: eroded soil sediments and all other sources. Erosion of bare soil is by far the largest contributor to the sediment load of stormwater (Sartor and Boyd, 1972), and its major sources include construction sites, denuded landscape areas and eroding stream channels (Allmendinger *et al*., 2007; GC and WWE, 2012; Nelson and Booth, 2002). Road sanding is a potential source of sands during the winter months (GC and WWE, 2012), but the greatest source of particulates from the transportation sector is the road surface itself (Sartor and Boyd, 1972), the breakdown of which can be responsible for up to 15% of the total watershed sediment load (Nelson and Booth, 2002). A final major source of suspended sediments is atmospheric deposition: both wet fall and dry fall (GC and WWE, 2012). The common sources of windblown urban dust are unpaved roads and railroads, uncovered stockpiles, constructions sites, landfills, paved roads and industrial emissions (Novotny and Olem, 1994).

Oxygen demand, both biochemical (BOD) and chemical (COD), correlates directly to the amount of oxidizable organic matter that is present in stormwater (Strecker, 1999). Therefore, most urban sources of organic materials in runoff are also responsible for contributing to the high oxygen demand of stormwater. Vegetative detritus is thought to be a leading source (Sartor and Boyd, 1972).

Salt is a potential stormwater pollutant of particular concern during winter months (Sartor and Boyd, 1972; USEPA, 1992a). In areas of the United States which receive significant snowfall, an average of 200-400 pounds of salt is applied per lane mile, and nearly all will eventually be mobilized by runoff (Hawkins and Judd, 1972). A related concern is that by causing accelerated corrosion of metal vehicle parts, salt can also increase the levels of automotive metals in runoff (Novotny and Olem, 1994).

Other toxic materials of concern can be found in urban stormwater: aromatic hydrocarbons from gasoline, coal and charcoal burning; PCBs from electrical equipment and insulation; and the pesticides chlordane, dieldrin, lindane and aliphatic fumigants (USEPA, 1990).

2.4B Stormwater Pollutant Concentrations and Loads

Early studies of urban pollutants, prior to the Nationwide Urban Runoff Program (NURP), attempted to characterize the constituents and accumulative mass of roadway debris. The study by Sartor and Boyd (1972) described the total pounds of roadway pollutants per mile of street curb in the Washington, DC area, and provided baseline estimates of the total pollution accumulation on paved streets. The study did not attempt to separate individual sources, so the reported loadings are a combination of material from automotive sources as well as material deposited on streets from the surrounding drainage area and watershed. A study by Shaheen (1975) was designed to isolate and measure, insofar as possible, the contributions attributable only to automobile traffic. The approach allowed a loading rate measurement in terms of pounds of pollutant accumulation per axle mile driven. Traffic-related deposition was considered to include direct emission of materials from vehicles (exhaust, body wear and tear, collision debris, etc.) as well as sediments and dust tracked onto roadways by cars and trucks. Materials deposited by air or water from surrounding areas were removed from consideration to the extent practicable. These two studies provide interesting examinations of street loadings, but they cannot be directly compared because of the different measurement approaches. Selected findings from both studies are shown in Table 2.4-2 below.

Sartor and Boyd (1972) also found that while many pollutants of concern were present in roadway debris, the bulk of the debris consisted of inert geological material such as quartz and feldspar. The prime source of this material was found to be the road surface itself, and the variability in debris mass from one location to another was due to the type of paving material and the maintenance condition. Asphalt streets contributed 80% greater mass loadings than their concrete counterparts, and streets of either type in fair to poor maintenance condition contributed about 2.5 times the amount of material of a well maintained street.

Table 2.4-2. Pollutants Contributed by Roadways

Pollutant	**Shaheen, 1975** **lbs/axle-mile**	**Sartor and Boyd, 1972** **lbs/curb mile**
Dry Weight	2.30E-03	1,400
BOD	5.43E-06	13.5
COD	1.28E-04	95
Phosphate	1.44E-06	1.1
TKN	3.72E-07	2.2
NO_2	2.26E-08	--
NO_3	1.89E-07	9.4E-02
Tot. Cu	2.84E-07	0.2
Tot. Pb	2.79E-05	0.57
Tot. Zn	3.50E-06	0.65
Tot. Cr	1.85E-07	0.11
Tot. Ni	4.40E-07	0.05
Fecal Coliform (orgs./curb mile)	--	5.60E+09
Tot. coliform (orgs./curb mile)	--	9.9E+10

Street debris is not a direct indicator of water quality problems, but it may be reasonably assumed that greater mass loadings on streets will translate to greater pollutant loads in runoff. A first-order decay equation—Equation (3)—modeling the relationship between total roadway debris and the mass of debris mobilized by runoff was developed by Sartor and Boyd (1972) .

$$N_c=N_o(1-e^{-krt}) \tag{3}$$

N_c is the mass of roadway debris of a given particulate size that is mobilized per unit of street surface area (g/ft^2), N_0 is the total initial mass of such debris per unit area (g/ft^2), and k is a constant dependent on street surface conditions (hr/in. min.). The last two variables refer to characteristics of the storm, with r representing rainfall intensity (in/hr) and t representing rainfall duration (min.). Understandably, as rainfall intensities and durations increase, the amount of roadway debris mobilized also increases. What is surprising is that the roadway constant, k, is dependent on the paving type and maintenance condition of the roadway, but is independent of the particle size of the roadway debris (Sartor and Boyd, 1972). This means that large particles are just as likely to be mobilized as small particles during rainstorms, and therefore all roadway debris can be considered a potential stormwater pollutant. By this reasoning, the total accumulative mass of material on streets has a direct connection to stormwater quality.

Comparisons of roadway runoff quality to raw and treated sewage effluent can also clarify the importance of roadways as a source of water quality pollutants. The studies of Shaheen (1975) and Sartor and Boyd (1972) compared pollutant loading rates from streets and sewage effluent, but in very different ways. Shaheen calculated the per capita pollutant loading from vehicles as if it were added to waterways at a fairly constant rate, much like sewage effluent itself. Of course, in reality, runoff loads from streets contribute their pollutants to waterways in intermittent pulses

(Shaheen, 1975). Data from the Sartor and Boyd (1972) study were presented as pulse loadings. The street runoff loads were what were assumed to be mobilized during the first hour of a short duration, high intensity storm over a theoretical city of 100,000 inhabitants and 400 miles of curb. The raw and secondary effluent loads were estimates of typical hourly discharges for the same city. This approach resulted in a positive skew of the ratio and made street runoff appear as a much more potent source of contaminants, because the authors were, in effect, comparing the highest pulse loads from stormwater to the average loads of wastewater. It must be remembered that during inter-storm periods, the stormwater emission rate will be zero, while sewage flows in a large city are continuous, so in the long-term aggregate, the contributions from sewage will greatly converge with, if not overtake, stormwater.

Data from both studies is shown in Table 2.4-3.

Table 2.4-3. Pollutant Loading Rates of Roadways Compared to Sewage Effluent

Shaheen, 1975 (g/capita-day)				Sartor and Boyd, 1972 (lbs/hr)				
Pollutant	**Traffic Deposition**	**Secondary Effluent**	**Ratio: Traffic/ Secondary Effluent**	**Raw Effluent**	**Secondary Effluent**	**Street Runoff**	**Ratio: Street/ Raw Effluent**	**Ratio: Street/ Secondary Effluent**
TSS	26.3	9.08	2.9	1,300	100.1	560,000	431	5596
BOD	0.06	5.3	0.01	1,100	58.4	5,600	5.1	96
COD	1.41	7.57	0.2	1,200	83.4	13,000	10.8	156
Phosphate	0.016	2.64	0.01	50	29.2	440	8.8	15
TKN	0.004	1.14	0.0	210	12.5	880	4.2	70
Tot. Cu	0.011	0.011	1.0	0.17	0.13	80	471	639
Tot. Pb	0.31	0.011	28.2	0.13	0.13	230	1769	1839
Tot. Zn	0.039	0.03	1.3	0.84	0.33	260	310	779
Tot. Cr	0.002	0.004	0.5	0.17	0.04	44	259	1055
Tot. Ni	0.005	0.004	1.3	0.04	0.04	20	476	480
Tot. coliform (orgs/hr)	--	--	--	4.60E+15	4.6E+10	4.0E+13	8.7E-03	870

The pulse loading of stormwater is an important consideration when quantifying its relative importance as a pollutant source. Because its emission rate is most frequently zero, stormwater may do little to change the ambient quality of receiving waters over the long term, but during runoff pulses, short-term shock loading may do lasting damage to aquatic biota (Shaheen, 1975). As an illustration, compared to stream base flow, the storm flow concentrations of total trace metals, dissolved metals and polycyclic aromatic hydrocarbons (PAH) were found to be seven times, two times and twenty times as high (Crunkilton *et al*., 1997). On a per-storm loading basis, the bulk mass of many types of contaminants flushed into receiving waters during storm flow pulses can exceed weeks or even months of background contaminant loading from dry weather flows (Characklis and Wiesner, 1997).

Given the caveats above about pulse versus continuous loading, it is still instructive to look at the ratios in Table 2.4-3. In the per capita basis analysis by Shaheen (1975), street debris was found to be a more potent source of TSS and all metals except chromium. Conversely, TKN and phosphate concentrations were found to be much greater in the effluent. In the pulse loading analysis by Sartor and Boyd (1972), as would be expected, stormwater was a much more concentrated source of all pollutants when compared to secondary effluent, and trailed raw sewage only in the number of total coliforms. The stormwater:wastewater ratios for both raw and secondary wastewater were particularly high for TSS and all metals, suggesting that stormwater pulses are an important source of these pollutants. The ratios were lowest for phosphate and TKN, similar to the pattern found by Shaheen (1975).

Many researchers have quantitatively analyzed the concentrations of harmful constituents of urban runoff from the 1980s to the present. Table 2.4-4 presents a summary of the findings of several studies that have characterized concentrations, while Table 2.4-5 summarizes conclusions about examined pollutant mass loads expressed on a unit area basis over time.

From Table 2.4-4, an interesting pattern emerges: median concentrations are generally lower than mean concentrations. This is probably because most water quality data are right skewed, meaning that a few high concentrations resulting from large, runoff-rich storms may increase the average values (USEPA, 1983). Although mean values are affected in this way, medians are not similarly biased (Helsel and Hirsch, 2002). As a result, median values reported in the literature will almost always be less than means, even from similar datasets (Helsel and Hirsch, 2002).

Table 2.4-4. Pollutant Mean and Median Concentrations of Stormwater

Pollutant	Median Concentration			Mean Concentrations			
	NURP[1]	Smullen[2]	Williamson[3]	Bryan[4]	Camponelli[5]	Dorman[6]	Hvitved-Jacobsen[7]
TSS (mg/L)	100	54.5	52	2,730	--	578	30-100
BOD (mg/L)	9	11.5	6	14.5	--	--	--
COD (mg/L)	65	44.7	39	179	--	--	40-60
TP (mg/L)	0.33	0.259	0.104	0.58	--	0.92	0.5
SP (mg/L)	0.12	0.103	0.026	--	--	0.43	--
TKN (mg/L)	1.5	1.47	1.02	--	--	12.77	2
OX-N (mg/L)	0.68	0.533	--	--	--	4.11	--
NO_3 (mg/L)			1.55	--	--	--	--
Tot. Cu (μg/L)	34	11.1	23	--	11.3 - 70.2	114	5 - 40
Tot. Pb (μg/L)	144	50.7	95	320	--	295	50 - 150
Tot. Zn (μg/L)	160	129	190	--	142-389	1,312	300 - 500
fecal coliform (orgs/100mL)	--	--	$4.2x10^3$	$30x10^3$	--	--	--
fecal strep (orgs/100mL)	--	--	$20x10^3$	--	--	--	--
E. coli(orgs/100mL)	--	--	--	--	--	--	10^3 - 100^3
Tot. coliform (orgs/100mL)	--	--	$15x10^3$	--	--	--	--
Turbidity (NTU)*	--	--	21	--	--	--	--

1:(USEPA, 1983),
2: (Smullen *et al*., 1999).
3: (Williamson, 1985)
4: (Bryan, 1972)
5: (Camponelli *et al*., 2010)
6: (Dorman *et al*., 1996)
7: (Hvitved-Jacobsen, 1990)
Notes:
***NTU=Nephelometric Turbidity Unit**

Table 2.4-5. Pollutant Loading Rates of Stormwater

Pollutant (kg/ha-yr)	Marsalek[1]	Novotny[2]				Bryan[3]
		Dense Res.[a]	Low-Med. Dense Res.[b]	Comm.[c]	Ind.[d]	
TSS	240	942	707	549	834	17821
BOD	30.5	82	25	90	34	94
COD	--	--	--	--	--	1166
TP	1.53	2	0.9	1.5	2.2	3.8
TN	7.2	7.8	7.1	11.2	8.4	--
Tot. Cu	--	0.2	0.2	0.08	0.6	--
Tot. Pb	--	0.54	0.6	1	2.5	2.1
Tot. Zn	--	0.9	0.6	0.4	4.3	--

a=Dense Residential Land use
b=Low to Medium Density Residential Land use
c=Commercial Land use
d=Industrial Land use

1: (Marsalek, 1978)
2: (Novotnyand Olem, 1994)
3: (Bryan, 1972)

As with the previous studies of roadway pollutants, it was remarked that stormwater often exceeded the pollutant concentrations of secondary treatment effluent (Bryan, 1972; USEPA, 1983). In the case of BOD, stormwater was shown to match secondary effluent loadings on an annual basis, and stormwater TSS and COD loadings were actually larger than would be expected from the discharge of raw domestic sewage from a residential catchment of the same size (Bryan, 1972), although the author noted that the stormwater pollutant concentrations of his study were atypically high. In concurrence with Shaheen (1975), stormwater was found to be a relatively insignificant source of phosphates when compared to sewage effluent (Bryan 1972). Because of the potential toxicity to aquatic organisms, metal concentrations in stormwater are a particular concern. As shown in Tables 2.4-4 and 2.4-5, Zn, Cu and Pb are common constituents of urban runoff, with the concentration of lead declining as a result of the ban on leaded gasoline (Schehl, 1995). Zn is present in the highest concentrations, generally at an average concentration about 5 times greater than Cu, but Cu is more likely to be present at levels that exceed the EPA's chronic and acute exposure levels of 9 μg/L and 13 μg/L, respectively (Camponelli *et al*., 2010).

Table 2.4-4 shows that reported Cu levels from all included studies violated the chronic exposure criteria, and all but Smullen *et al*. (1999) exceeded the acute exposure. Results from NURP also showed that stormwater Cu concentrations nationwide, while variable, were frequently high enough to cause the impairment of aquatic biota (USEPA, 1983).

Since concentrations vary with time, it is beneficial to know if certain segments of the runoff hydrograph carry levels of Cu, Zn, Pb or other toxics that are above acute or chronic limits. The term “first flush” is used to describe the relative enrichment of the earliest pulse of runoff in a storm hydrograph, as runoff mobilizes sediments and other debris that have been accumulating on urban surfaces since the last storm (Viessman and Hammer, 1998). First flush effects have been reported in the literature, but not all pollutants may be affected (Batroney *et* al., 2010). Whenever pollutants are enriched in the first flush, the magnitude appears to be dependent on the size of the watershed being studied. Larger watersheds receive runoff from multiple locations, each with a unique travel time (Yang *et al*., 2011). As a result, there is staggering in the timing of flows delivered to downstream areas, and first flush effects from any one portion of the watershed may be attenuated in the mix with flows from other areas (Characklis and Wiesner, 1997; USEPA, 1993). This staggering effect is not likely to be seen in smaller watersheds, and it is more likely that elevated first flush concentrations can be observed (USEPA, 1993; Characklis and Wiesner, 1997). Shaheen (1975) calculated the percentage of total pollutant load contained in the first flush, and found it accounted for more than half of all OX-N and fecal coliforms, and more than one third of phosphate, TKN, BOD and fecal strep. Whipple *et al*. (1978) found a first flush correlation with Pb as well as phosphate. Batroney *et* al. (2010) found elevated TSS, nitrate, dissolved copper and dissolved cadmium in the first flush of a parking lot, but found no first flush effect for important pollutants like TN and TP. The presence of the first flush effect in smaller watersheds for some pollutants suggests that the earliest runoff flows during storm events may be the most damaging to receiving water biology, but even in those studies which documented this effect, pollutant concentrations continued to remain elevated in later parts of the storm, and every flow peak subsequent to the first one also correlated with a spike in pollutant concentration (Williamson, 1985).

From Table 2.4-5, some differences in stormwater loading rates were observed due to the surrounding land use. Industrial land seems to create greater pollutant loads (Novotny and Olem, 1994; Sartor and Boyd, 1972; USEPA, 1983). This is likely attributable to maintenance and aesthetic issues—for example, commercial and residential areas are more likely to have well-maintained paved roads that are swept more frequently (Sartor and Boyd, 1972). However, the land use itself may contribute in that industrial vehicles are more likely to be dirty and have significant fall-off of dust and debris during operation (Sartor and Boyd, 1972). Also, there is likely to be more atmospheric deposition, both wet and dry fall, over industrial areas and other high-density urban land uses (Novotny and Olem, 1994). On a monthly basis, these areas can experience anywhere from 7 to 30 tons of atmospheric deposition per square kilometer (Novotny and Olem, 1994). Additionally, because of the high amount of impervious surfaces in dense urban zones, the atmospheric fallout that lands is more likely to make its way to stream and rivers during storm events (USEPA, 1992a).

Another stormwater pollutant byproduct of urban land use is salt. In snowy areas of the US, salt use on roadways, 98% of which is sodium chloride (NaCl) (Novotny and Olem, 1994), can result in salt-laden runoff that has concentrations as high as 11,000 ppm, with smaller concentration spikes continuing as late as the middle of the following summer (Hawkins and Judd, 1972). In some cases, roadway runoff was found to have the same salt content as sea water: 20g/L (Novotny and Olem, 1994).

Natural landscape features also create significant pollutant loads in stormwater. In the fall months, organic detritus—particularly falling leaves—dominate stormwater nutrient loadings (Novotny and Olem, 1994). Each mature tree in a watershed can contribute 30 - 50 lbs. of organic matter and 0.013 - 0.15 lbs. of phosphorus to the watershed (Novotny and Olem, 1994). The soil type is also an important consideration, with clayey soils able to contribute significantly more nutrient loads to runoff than sandy, gritty soils (Sonzogni *et al*., 1980).

2.4C Stormwater Pollutant Variability

A goal of many urban runoff studies has been to find a link between land use and the concentration of pollutants in stormwater. If a link can be found, those areas that are prone to the most intense pollution can be made the focus of remediation efforts. The NURP report divided study sites up according to four general land use types, and isolated the median pollutant concentrations found in the runoff from each (USEPA, 1983). The summarized results are reproduced in Table 2.4-6 below.

While there are some clear differences in pollutant median concentrations between land uses, just as there were in the mass loading data from Table 2.4-5, differences were not found to be statistically significant except for open/non-urban land (USEPA, 1983). In fact, the defining

Table 2.4-6: NURP Land Use Effects on Median Stormwater Pollutant Concentrations

Pollutant		Land use			
		Residential	Mixed	Commercial	Open/Nonurban
BOD	mg/L	10	7.8	9.3	--
COD	mg/L	7.3	65	57	40
TSS	mg/L	101	67	69	70
Total Pb	µg/L	144	114	104	30
Total Cu	µg/L	33	27	29	--
Total Zn	µg/L	135	154	226	195
TKN	µg/L	1900	1288	1179	965
OX-N	µg/L	736	558	572	543
Total P	µg/L	383	263	201	121
Soluble P	µg/L	143	56	80	26

All data from USEPA, 1983.

characteristic of urban runoff seems to be its variability. Stormwater pollutant concentrations are quite variable (Strecker *et al*., 2001) Pollutant concentrations have been reported to vary:

- Within a storm event
- From storm event to storm event
- From site to site within an urban area
- From site to site within the same land use
- From site to site in different land uses
- From one urban area to another

(Sonzogni *et al*., 1980; USEPA, 1983; USEPA, 1992a)

Given the wide variability in stormwater concentrations, it can be concluded that urban land use alone is of limited predictive value when it comes to estimating stormwater quality, and that any differences are likely eclipsed by storm-to-storm variation (USEPA, 1983). The only significant difference found in the NURP studies were in the open/nonurban land category, which did have consistently better runoff quality than all urban land types (USEPA, 1983).

Storm size also did not correlate significantly with runoff pollutant concentrations. Larger storms did not exhibit higher concentrations than smaller storms (Sonzogni *et al*. 1980; USEPA, 1983), and many constituents showed a slight reduction in concentration as storm runoff volume increased.

Bacterial contamination also exhibited a lack of correlation with land use, but did show significant variability attributable to the season. During warmer months, coliform concentrations in urban areas increased significantly (Shaheen, 1975; USEPA, 1983), but this increase seemed to have no correlation to any urban activity, and was likely due to bacterial sources which do not pose human health risks (USEPA, 1983).

All in all, the differences in stormwater pollutant concentrations from site to site must be attributable to site-specific factors rather than any trait general enough to apply to a whole category of land use, such as commercial, residential or industrial land. These site specific factors might include the following: landform including topography, geology and soils; land use intensity; and materials usage within the land use (Sonzogni *et al*., 1980).

Land use intensity refers to how heavily the land is used for a specific purpose, with an interstate highway being a more intensive transportation use than a suburban cul de sac. It is not enough to classify land as industrial or residential, one must also quantify how much activity is associated with those land uses to begin to estimate the effect of the use on stormwater pollutant concentrations (Sonzogni *et al*., 1980). To illustrate materials usage, a golf course and a playground may both qualify as open land, but the golf course will likely receive much higher levels of fertilizer and pesticide application, and will be at higher risk of shedding stormwater enriched with those materials. However, of all three site specific factors mentioned, landform is probably the most important determinant of pollutant concentration, and is often more of a dominant factor than land use (Sonzogni *et al*., 1980). Of all the factors that fall within the characterization of landform, the most important is likely soil texture, or the percentage of sand, silt and clay of the native soils (Sonzogni *et al*., 1980). All soils that are not protected from stormwater erosion provide a nearly infinite pool of sediment and oxygen demanding materials (Novotny and Olem, 1994). Clayey soils in particular are a more potent source of runoff pollutants than sandy soils because they percolate water slowly, create more surface runoff, are easier to entrain, and have larger surface areas to which other pollutants can bind (Sonzogni *et al*., 1980). As an illustration, clayey soils, even under the same land use as sandy soils, will cause the leaching of 6.5 times more phosphorus (Sonzogni *et al*., 1980).

2.4D Stormwater Pollutant Loadings vs. Concentrations

While stormwater pollutant *concentrations* may not show any marked differences from land use to land use, their does appear to be a very clear and important correlation between land use and pollutant *loadings.* The reason is that, while different land uses may not affect concentrations in any predictable way, they do affect stormwater volume (Carey *et* al., 2013; Leopold, 1968; Schueler, 1987; Schueler 1994; USEPA, 1983). Loadings, and loadings are the product of both concentrations and volume. Therefore, the more runoff there is from a site, the more pollutant loadings there will be (Characklis and Wiesner, 1997).

As was shown earlier, runoff volume is strongly related to the amount of imperviousness in the watershed (Jacobson, 2011), and therefore pollutant loadings will have a direct relation to imperviousness as well (Sonzogni e*t al*., 1980; Schueler, 1994). Imperviousness alone, however, may be an inadequate predictor. Compacted, dense urban soil will be less permeable than natural soil (Hancock *et al*., 2010), and will also cause increased runoff volume and thus increased loadings (Sonzogni e*t al*., 1980).

Runoff studies have confirmed the relationship between increased runoff volume and increased pollutant loads. A study of BOD and P loads from low density single-family housing and high density multi-family housing found that the high density housing had 4.4 times the daily BOD loading and 6.6 times the P loading of the single-family community (as kg/km^2-day), despite the multi-family development being newly constructed and well maintained (Whipple *et al.*, 1978). In Houston, the metals strontium and barium, which were found in lower concentrations in stormwater than in base flows, nevertheless had short-term storm loadings that were the equal of weeks of base flow loading simply because of the increased volume of runoff (Characklis and Wiesner, 1997).

Using pollutant loading rather than pollutant concentration as the means of comparison, some authors found that land use could be used as a general predictor of runoff quality. Marasalek (1978), for example, divided urban land uses into four groups based upon their pollution potential:

1) Group 1: Low pollution loads - Low and medium density residential (<125 people/ha) and low intensity industrial (warehouses and wholesale).

2) Group 2: Intermediate Pollution Loads - High density residential (>125 people/ha) and commercial land use.

3) Group III: Highest Pollutant Loads - Medium and High Intensity industrial.

4) Group VI: Lowest Pollution Potential - Parks and playgrounds, undeveloped open space.

As may be seen, the groupings were not based upon traditional categorizations of land use (industrial versus residential, for example), but rather upon the density and intensity of land use, which may have a more direct correlation with imperviousness (Sonzogni *et al*., 1980).

Whether to examine pollutant concentrations or loadings as the runoff characteristic of concern may depend on the type of receiving body. In lotic streams or rivers, concentrations may be of more importance because pollutants will pass through the system fairly quickly, but if the receiving body is lentic and allows pollutants to accumulate, such as a lake or an estuary, loading may be the more applicable measure (Hartigan, 1989).

2.4E Physical and Chemical Characteristics of Stormwater Pollutants

Physical and chemical characteristics of stormwater pollutants may be just as important as the concentration or loading rate because they determine, to a large extent, the availability of the pollutant to aquatic biota. Pollutants that are readily available can cause rapid, intense, but short-term oxygen depletion and acute toxicity; pollutants that are more resistant to biological uptake may linger in the system and cause long term harm such as eutrophication, bioaccumulation of toxins, and chronic oxygen depression (USEPA, 1983).

One of the most important characteristics of stormwater pollutants is the fractionation between the dissolved and particulate forms. Strong, direct correlations have been reported between the majority of stormwater pollutants and suspended sediment concentrations (Bryan, 1972; Williamson, 1985). A result has been that TSS concentrations have often been treated as proxy measures of total stormwater pollutant loads (Strecker, 1999). While there certainly is a statistically significant connection, TSS concentrations alone are not enough to accurately predict the amounts of particulate pollutants (Strecker, 1999).

The distribution of pollutants between the dissolved and particulate phases has been examined by many researchers. Metals were found to be nearly evenly divided between the dissolved and particulate fractions in base flow, but during storm flow, total metal concentrations were seven times higher with the particulate phase accounting for 90% of the total (Crunkilton *et al*., 1997). Zinc was found to be primarily in the dissolved phase during base flow, but during storm flow it showed large increases in the macro-colloidal size fraction of suspended solids (Characklis and Wiesner, 1997). Iron was also strongly correlated with the macro-colloids, in both storm flow and base flow (Characklis and Wiesner, 1997). Total bacterial counts increased significantly in stormwater when the turbidity was over 100 NTU, suggesting a correlation with suspended sediments (Struck *et al*., 2008). However, while this correlation could have been attributable to bacterial attachment on sediment surfaces, it could also have been due to the reduction in UV light penetration afforded by the more turbid water (Struck *et al*., 2008).

Despite the findings above, many studies have found that a significant portion of stormwater pollutants are contained within the dissolved phase. A summary from relevant reports in the literature is shown in Table 2.4-7 below.

The toxicity of any pollutant is a function of its aqueous form (Tanizaki, Shimokawa et al. 1992). For metals, the dissolved phase is important because the free dissolved ion is the most toxic form (Novotny and Olem 1994), and for nutrients, the dissolved forms are the most readily available for biological uptake (Strecker 1999). However, it is not sufficient to quantify the percentages of a pollutant in the dissolved versus particulate phase, because within each phase there exist a

Table 2.4-7: Dissolved and Particulate Fractions of Stormwater Pollutants

Pollutant	Tanizaki[1]		NURP[2]*		Camponelli[3]		Crunkilton[4]**	
	Particulate	Dissolved	Particulate	Dissolved	Particulate	Dissolved	Particulate	Dissolved
TOC	25%	75%	--	--	--	--	--	--
Cu	--	--	50%	50%	56%	44%	--	--
Zn	39%	61%	50%	50%	69%	31%	--	--
Pb	--	--	90%	10%	--	--	--	--
TP	--	--	60%	40%	--	--	--	--
TKN	--	--	50%	50%	--	--	--	--
Fe	85%	15%	--	--	--	--	--	--
PAH	--	--	--	--	--	--	50%	50%

-Notes:
***NURP fractions not directly measured, but approximations calculated from limited settling column dataset.**
****Approximate values**

1: (Tanizaki *et al*., 1992)
2: (USEPA, 1983) and (USEPA, 1986)
3: (Camponelli *et al*., 2010)
4: (Crunkilton *et al*., 1997)

variety of substrates and ligands that can bind the pollutants and attenuate their availability and/or toxicity.

Within the dissolved phase, many metals bind to dissolved ligands such as sulfides, humic and fulvic acids, chloride ions and hydroxyl ions (Novotny and Olem, 1994). Zinc in particular was found to strongly associate with dissolved organic carbon (DOC) (Characklis and Wiesner, 1997), and this organically bound form, not the free dissolved ion, was the dominant dissolved form of zinc in urban streams (Tanizaki *et al*., 1992). Much of the DOC binding the zinc consisted of humic and fulvic acids (Tanizaki *et al*., 1992). Copper can also bind to DOC in significant amounts (Camponelli *et al*., 2010). Sulfides act as a powerful binding ligand for metals, but its effect is largely dependent on the pH and redox potential of the water (Novotny and Olem, 1994). Anaerobic or anoxic conditions must be present for sulfides to play an appreciable role, and these conditions are more often met in the pore water of sediments than in the water column itself (Novotny and Olem, 1994). Acidic conditions have the general effect of making most cations more soluble, as the increased concentration of H^+ ions compete with the positively charged metal ions for binding spaces on particulate surfaces (Brady and Weil, 1999).

The general importance of ligands is that all have the effect of reducing the free metal ion concentration of metals, and thus they also reduce the toxicity of the metal (Novotny and Olem, 1994).

Other dissolved ions, even if not acting as ligands, can affect the toxicity of aqueous metals. Aqueous ions will compete with dissolved Cu, Zn and Pb for uptake by aquatic biota, so the toxicity of metals is tempered by the natural hardness of the water (Strecker, 1999). This fact is recognized in Virginia water quality criteria: the values for metals vary as a function of hardness, with the acute and chronic concentrations being higher in harder waters than soft waters (Va. Admin. Code, 9VAC25-260-140).

In aqueous systems, if a pollutant is not in its free ionic or dissolved ligand-bound form, it must be attached to a particle (Novotny and Olem, 1994). In the particulate phase, as in the aqueous phase, there are many different materials with which the pollutants are associated. In the most obvious fashion, the pollutant is a particulate itself. This is true of most forms of organic nitrogen and phosphorus, which are contained within the bodies of living plants, animals and microbes as well as decaying organic detritus (GC and WWE, 2012a ; Novotny and Olem, 1994). Precipitates are additional forms of particulate pollutants. At high pH ranges, the solubility of many metal hydroxides becomes low enough that significant metal reduction in the water column can be achieved through the formation and settling of solid precipitates (Novotny and Olem, 1994). Small amounts of metal may form a precipitate with sulfides under anaerobic conditions (Novotny and Olem, 1994). Some phosphorus may precipitate with calcium at very high pH and with iron and aluminum at pHs of 6 to 8.5 (Casey, 1997; Droste, 1997).

Many pollutants in sediment are not particulates themselves, but rather are bound to the surfaces of sediments through various means. Iron (Fe) and manganese (Mn) oxides, organic matter and clay are all materials that can adsorb pollutants to sediment surfaces (Camponelli *et al*., 2010; Hogan and Walbridge, 2007; Novotny and Olem, 1994). Carbonates can also bind metals (Camponelli *et al*., 2010). In the sediments of a stormwater pond, 25-49% of particulate zinc was found adsorbed to Fe and Mn oxides, and 10-25% was bound to carbonates (Camponelli *et al*. 2010). The Fe oxide concentration of stormwater pond sediments was also found to correlate positively with reductions in stormwater P concentrations (Hogan and Walbridge, 2007). Clays, due to internal substitutions of high valence aluminum and silicon atoms within their crystal lattices by lower valence cations, carry a net negative charge and can attract and bind positive ions from solution (Brady and Weil, 1999). Organic matter and the weathered edges of clay particles have a variety of pH-dependent charged functional groups on their surface which can adsorb both cations and anions (Brady and Weil, 1999). Anion adsorption capacity increases with pH while cation adsorption dominates in more acid conditions (Brady and Weil, 1999). The adsorptive bonds of oxides, clay and organic matter are for the most part reversible, and ions can be released back into solution under the right conditions (Brady and Weil, 1999).

The balance between the dissolved and adsorbed phases is highly dependent on the natural solubility of the pollutant and the surrounding environmental conditions. Under reducing conditions, Fe and Mn oxides can be reduced, dissolve, and their bound pollutants can be released into the water column or the sediment interstitial water (Schueler, 1987; NVPDC and ESI, 1992). The pH of water is an important consideration as well. Phosphorus bonds with Fe and aluminum (Al) oxides and organic matter under acidic conditions and with calcium under basic conditions, but at near neutral pH it is much more soluble and will leach into the interstitial waters (Novotny and Olem, 1994).

Nitrogen behaves quite differently from phosphorus. Whereas phosphorus binds strongly to sediments and is washed into receiving waters attached to eroded soil particles (Novotny and Olem, 1994), nitrogen generally has a much higher dissolved concentration, mostly due to the high solubility of nitrate, which is almost always found in dissolved form (Hogan and Walbridge, 2007; Williamson, 1985). While nitrate is rarely bound to sediment surfaces, biological activity within the sediments can strongly affect its form in aquatic systems. Under anoxic conditions in bottom sediments, denitrifying bacteria can convert nitrate into gaseous forms that volatilize out of the system (Brady and Weil, 1999; GC and WWE, 2012a). Ammonium can bind to clay particles, volatilize into the atmosphere as gas at higher pH values, and in the top layer of sediment, which is generally oxygenated, undergo bacterial nitrification into nitrate (Novotny and Olem, 1994).

Reduction of nitrate through denitrification has proven to be one the best ways to remove nitrate from stormwater (Winer, 2000). Those stormwater treatment practices that are able to create the anoxic conditions needed for denitrification—wet ponds and treatment wetlands—generally have much better nitrate removal rates than stormwater facilities whose sediments remain oxidized, such as dry ponds and infiltration facilities (GC and WWE, 2012a; Landers, 2006; Winer, 2000). In fact, facilities maintaining oxidizing conditions often increase the amount of nitrate because they are able to create the conditions needed to convert ammonium to nitrate, but then cannot reduce nitrate into its gaseous forms (Winer, 2000).

Non-ionic organic chemicals—such as polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), other hydrocarbons, pesticides and herbicides—may also adsorb onto organic matter in sediments, but at a level controlled by their octanol-water partition coefficient (K_{OW}) and the organic carbon content of the sediments (Novotny and Olem, 1994). The K_{OW} can be normalized to create the K_{OC}, which is the partition coefficient for a hypothetical sediment that consists entirely of organic carbon, and one can then describe the total solid-liquid partitioning of this branch of pollutants using Equation (4) (Novotny and Olem, 1994).

$$\prod = K_{OC} * (\%OC)/100 \tag{4}$$

$\prod$ is the unitless solid-liquid partition coefficient of the pollutant and %OC is the organic carbon percentage of the sediment. As is evident, higher K_{OC} and higher organic carbon in the sediment will increase the amount of sediment bound pollutant. Some organic pollutants that occur in much higher concentrations in the sediment as compared to the water include PAHs, organochlorine pesticides (DDT, DDE, aldrin, chlordane) and PCBs (Novotny and Olem, 1994).

These pollutants—as well as the heavy metals such as Pb, Cu and Zn—are resistant to degradation and can therefore accumulate in sediments over many years, continually increasing their bound concentrations. This can result in sediment pollutant loads that are several orders of magnitude greater than in the water column, even in systems that have relatively clean stormwater inputs (Horner *et al*., 1994). The sediment load itself cannot be seen as a direct measure of toxicity (Williamson, 1985), because many of the bound sediments are not bio-available; rather the pore water concentration is taken as the best measure of sediment toxicity (DiToro, 1991; Novotny and Olem, 1994). Even with reduced bioavailability, long term accumulation of pollutants in sediments can be toxic to the benthic organisms at the base of many aquatic food chains, raising the risk of bioaccumulation at higher trophic levels (Horner *et al*., 1994).

Often the same traits that cause certain pollutants to be sediment-bound can lead them to accumulate within living tissue (Novotny and Olem, 1994). In a manner similar to calculating the K_{OC}, Novotny and Olem (1994) describe how a partition coefficient for bioaccumulation tendency may be computed by normalizing the K_{OW} by the fatty lipid content of an organism. The partition coefficient thus created symbolizes the equilibrium condition of the pollutant in living tissue and the ambient water. For nearly all aquatic organisms in all trophic levels, the lipid normalized bioaccumulation factor equals the K_{OW} up until log K_{OW}=5. Beyond 5, most relationships are still positively correlated, but not necessarily linear (Novotny and Olem, 1994).

Measurements of sediment toxicity are difficult, particularly given the great variety of binding sites and bioavailability of particulate pollutants. DiToro (1991) devised a measurement called the sediment toxic unit, or STU, which compares the concentration of a pollutant in interstitial water to the known lethal concentration in the bulk water column. When the concentration in the pore water equals the pollutant concentration that causes 50% mortality in test subjects in the bulk water (the LC_{50} concentration), the STU is given a value of 1. Pore water concentrations above the LC_{50} will have a value greater than 1, and concentrations below will have an STU less than 1.

MacDonald *et al*. (2000) created two sediment toxicity measures known as the threshold effect concentration, or TEC, and the probable effect concentration, or PEC. The TEC represents the pollutant concentration in sediment below which negative impacts on aquatic organisms are thought to be rare, and the PEC is the concentration above which negative impacts are common. Both TECs and PECs represent calculated consensus concentrations by the authors, created by reviewing the existing literature, converting all measurements to the mg pollutant/kg sediment scale, and deriving the geometric means for each pollutant. The TECs and PECS for several common stormwater pollutants are included in Table 2.4-8.

Table 2.4-8. Sediment Threshold (TEC) and Probable (PEC) Effect Concentrations

Pollutant	PEC	TEC
Cu	149	31.6
Zn	459	121
Pb	128	35.8
Ni	48.6	22.7
Cd	4.98	0.99
Cr	111	43.4

All data from MacDonald *et al*, 2000.
All units in mg pollutant/kg sediment.

Camponelli *et al.* (2010) examined sediment cores from the bottom of Maryland stormwater ponds. The cores were analyzed for the total concentration of Cu and Zn, as were samples from the top-most layer of sediments. All cores had Cu and Zn concentrations above the TEC values in Table 2.4-8, and the surface layer of sediments had Zn concentrations that mostly exceeded PEC values. Cu was mostly between the PEC and TEC concentrations. However, most of the Cu (92-98%) was found to be in fairly recalcitrant, unavailable form. Zn was mostly recalcitrant too, but one quarter to one half was determined to be bound to Fe and Mn oxides. The authors were concerned that the oxide-bound Zn could re-enter the water column if the bottom sediments were subjected to anoxic conditions.

In a similar Maryland study by Casey *et al*. (2005), stormwater pond sediment cores were examined for their concentrations of Cu, Zn, Ni, Pb, Cd and Cr. It was found that all examined metals but Cd were present in concentrations in excess of TECs in at least one of the studied ponds, and Ni was present above PEC levels in one third of the study sites. Despite the elevated concentrations in the sediments, the overlying water column had dissolved metal levels below the EPA-recommended water quality criteria for both acute and chronic exposure, except for a few instances of elevated Cu.

It is clear that sediments that settle to the bottom of stormwater ponds and other waterways can carry with them a significant load of associated pollutants. Because of this, the settling behavior of suspended sediments is of interest. Settling particulates convey their pollutant mass to the bottom and out of the water column, where a portion of pollutants will be stored in unavailable forms or biologically transformed into less harmful forms. Those that do not settle remain in the water column, or are flushed downstream through the watershed. Several investigations have used settling column studies to estimate the time in which portions of stormwater TSS will settle out of suspension. The EPA NURP study (1983) and a subsequent study by the Federal Highway Administration (Dorman *et al*., 1996) divided stormwater sediments into five subcategories based on the amount of settling, and assigned an average settling velocity to each, as shown in Table 2.4-9.

The results of the analyses are similar, and differences that exist may be explained by the fact that Dorman *et al*. (1996) examined the first flush of highway runoff, which can be enriched with larger particles and could have caused the increase in the fraction of sediments with the highest

Table 2.4-9. Average Settling Velocities of Stormwater Particulates

Average Settling Velocity (ft/hr)	Percentage of TSS Mass	
	NURP[1]	Dorman et al[2]
0.03	20	18
0.3	20	17
1.5	20	17
7	20	19
65	20	28

1: (USEPA 1983)
2: (Dorman, Hartigan et al. 1996).

settling velocities. It should be noted that the reported data were based on averages. The settling velocities for individually monitored storms, like most characteristics of stormwater, were highly variable, and the percentiles from storm to storm could vary by about one order of magnitude (USEPA, 1983).

Settling column tests were also performed to calculate the pollutant removal associated with TSS falling out of suspension. The results of such tests not only give estimates of the necessary time for significant sediment removal via settling to occur, but also provide an estimate of the percentage of pollutants that are associated with particulates (Dorman *et al*., 1996). The results from two such studies are shown in Table 2.4-10.

Some clear time patterns can be seen in the literature. Most TSS settling was rapid and completed within the first 6 to 16 hours (Whipple *et al*., 1978). Pb seemed to be the most particulate bound of the pollutants, and its settling rates and overall removal tracked closely with TSS (USEPA, 1983; Whipple and Hunter, 1981). While most settling of hydrocarbons and BOD occurred in the first 16 hours, settling was very slow in the initial hours of the tests, possibly due

Table 2.4-10. Stormwater Pollutant Removal Via Sediment Settling

Pollutant	% Removal From Water Column		
	Dorman et al[1]		Whipple et al[2]
	6 hours	48 hours	32 hours
TSS	70-78	87-92	70
TP	33-39	43-46	30-60*
Suspended P	58-68	77-81	--
TKN	19-24	24-31	--
Tot. Cu	43	55-58	30-50
Tot. Pb	60-65	72-82	60-85
Tot. Zn	34-35	40-42	17-36
Tot. Ni	--	--	30-50
Hydrocarbons	--	--	65
BOD_5	--	--	20-50

1: (Dorman *et al*., 1996) *** Measure of phosphate P, not TP**
2: (Whipple and Hunter, 1981)

to the need for smaller particles to flocculate together before settling could occur (Smullen *et al*., 1999). Most Cu, Zn and Ni was removed within the first 8 hours followed by slower removal rates, and total removal of these metals was much less than Pb (Whipple and Hunter, 1981). The most variability from test to test was found with BOD and phosphates (Whipple and Hunter, 1981).

In the study by Dorman et al. (1996), maximum removals by sedimentation were calculated for important stormwater pollutants, and represent the theoretical removal of pollutants that would occur if 100% of TSS could be removed by settling. The values also represent an estimate of the total mass of pollutants that are in particulate form. With 100% TSS removal, about 87% of Pb, 61% of Cu, 47% of Zn, 49% of P, and 30% of TKN would also be removed (Dorman *et al*., 1996). This again shows the high correlation between Pb and particulates. TKN numbers are likely lower due to the fact that nitrate is almost always present in soluble form (Novotny and Olem, 1994; Williamson, 1985).

Obviously, given the above data, particle size may be expected to have an important effect on settling rates and removal efficiencies. Larger particles will fall from suspension quickly and more completely, while smaller particles will do so slowly or not at all, at least within the normal time spans typical of stormwater treatment processes. This is important because certain pollutants seem to preferentially bind with certain particle sizes (Williamson, 1985). From the stormwater analysis of Characklis and Wiesner (1997), it was found that Zn seems to preferentially bond to macrocolloidal (0.45 – 20μm) particles of organic carbon, and spikes in concentration of one corresponds with a spike in concentration of the other. Iron also seems to be highly bonded to the macrocolloids, but to particles other than organic carbon.

In road dust, the smallest particulate fraction (<5μm), contained the highest concentrations of Cu and Zn, followed by the second smallest particulate fraction (63μm), even though combined, these two fractions represent only 3% of the mass of the bulk dust (Camponelli *et al*., 2010). The analysis of street debris by Sartor and Boyd (1972) concluded that organic matter was also found in higher concentrations in the smaller particle size ranges, likely due to the fact that organic material such as leaves, grasses, etc. can be ground down into finer and finer pieces before being washed away by stormwater. An EPA analysis (1986) assumed that Pb would be associated with all five particle settling velocities listed in Table 2.4-9, but that Cu, Zn, TKN, BOD and COD would be associated with the four slower velocities, correlating to the smaller particles. On a weight basis, 30% of Zn, 32% of Ni, 21% of TOC, and 25% of Cr were associated in urban streams with particles having molecular weights less than 500; these small particles are likely to be free ions, ion pairs and complexes of small molecular size (Tanizaki *et al*., 1992).

Many studies found a correlation between smaller particles and higher concentrations of stormwater pollutants. This is likely due to the fact that smaller particles have much greater surface area, which allows for more pollutants to sorb (Camponelli *et al*., 2010; Sartor and Boyd, 1972). In the case of metal enrichment of road dust, the smaller particles may also directly represent the size fraction of metal wear debris from automobiles (Camponelli *et al*., 2010). This is a problematic finding if the goal is to remove pollutants via settling. Smaller particles are more difficult to settle because of their slow settling velocities, and some may pass through conventional stormwater ponds without appreciable settling (Schueler, 1987). The size distribution of particles greatly affects the temporal and spatial distribution of stormwater pollutants (Characklis and Wiesner, 1997). As an illustration, at highway stormwater outfalls draining runoff with heightened concentrations of Pb and PAHs, the stream sediments directly below the discharge pipe were enriched with Pb, whereas PAH sediment enrichment occurred 60 yards downstream (Shaheen, 1975) . This was likely due to the fact that Pb was associated with larger particles but PAHs were mainly bound to smaller organic sediments with slower settling velocities (Shaheen, 1975).

Of course, the higher settling velocity of larger particles does not mean that the pollutants attached to these particles are permanently removed from the system. As discussed earlier, there are many ways by which pollutants can be liberated from sediments. Additionally, many stormwater sediments are actually loosely-bound aggregations of small mineral and organic particles that flocculate while suspended (Ellis *et al*., 1982). These flocculated aggregates may settle, but the floc particles may be broken apart and their constituent particles resuspended at a later time (Ellis *et al*., 1982). Turbulent flow through waterways that hold accumulated sediments is a common way in which settled pollutants can be resuspended, and their pollutants brought back into the water column (Schueler, 1987; Struck *et al*., 2008).

The varying chemical and physical aspects of pollutants must be considered when analyzing the negative impacts of stormwater as well as the best means of reducing the threat. A stormwater treatment facility must account for the great variability in stormwater pollutant types, sources, concentrations and behaviors—both in stormwater and in the receiving water body—as well as the variability in the sizes and frequencies of storms. An effective stormwater facility must, therefore, have the flexibility to treat pollutants by a variety of means if the goal is to reduce the whole array of potential impairments to receiving bodies.

2.5 EFFECTS OF URBAN STORMWATER ON RECEIVING BODIES

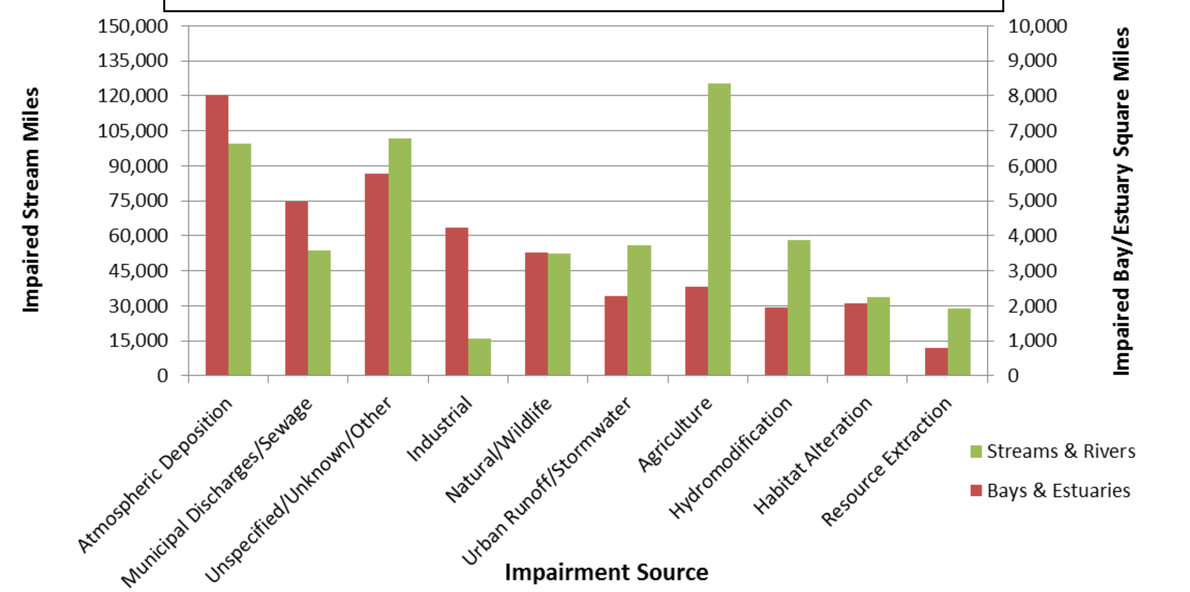
A succinct introduction to this section comes from the Environmental Protection Agency:

> "Non-point source pollution is the leading source of water quality impairment in the United States" (USEPA, 2011a).

Of all the water bodies in the US that have been flagged for not achieving mandatory standards of water quality, fully 76% (33,820 unique water bodies) had a nonpoint pollutant source as the primary causative factor (USEPA, 2011a). Of course, urban areas are not the sole source of nonpoint source runoff. Agriculture is, in fact, the leading source of nonpoint pollution (USEPA, 2011a), but urban areas are an important contributor, and their importance is rising because the extent of urban land is rapidly increasing as the United States population shifts from rural areas to suburbs and cities. Urban areas have increased about 20% every decade since WWII (USEPA, 1992b). Urban areas are also a proportionally more potent source of pollutants than agriculture. In the 1990s, urban areas accounted for 2.5% of the land area in the United States, but were thought responsible for 18% of impaired river miles, 24% of impaired lake acres, and were the leading cause of impairment in estuaries (USEPA, 1992a). Pollution wise, urban areas punch above their weight.

In the National Water Quality Inventory report to the US Congress, the self-reported assessments of the water resources of the US states found that urban runoff was the 5th leading cause of stream and river impairment (56,126 impaired stream miles) and the 10th leading cause of bay and estuary impairment (2,283 square miles impaired) (USEPA Attains Database. Accessed January, 2013, http://www.epa.gov/waters/ir/). The leading sources of impairment, and the length or area of waterway impaired by each, are summarized in Figure 2.5-1.

Figure 2.5-1. Causes of Water Body Impairment in the US



Data from (USEPA Attains Database. Accessed January, 2013, http://www.epa.gov/waters/ir/)

Of course, estimates of the damage caused by urban stormwater are probably understated, because the values in the figure only represent the impaired portions of the water bodies assessed in one reporting cycle, which is a fraction of the total water bodies in any state. In addition, many of the other listed sources of impairment have a relationship to urbanization, if not directly to stormwater. For instance, hydromodification, municipal/sewage discharges and habitat alteration are listed as the 4th, 6th and 9th leading causes of stream and river impairment (USEPA Attains Database. Accessed January, 2013, http://www.epa.gov/waters/ir/), and all can be partially attributable to urbanization.

It is difficult, if not impossible, to pinpoint a single reason for the decline of urbanized waterways, because it is the cumulative impact of many factors such as sedimentation, low base flow, high storm flow, scouring, pollutant discharge and thermal extremes that cause water bodies to decline in health (USEPA, 1992a). With the caveat in mind that the individual effects of urbanization will never be as important as the cumulative whole, one can still examine the damage done by these individual factors in affected receiving waters.

Increased sediment loads in stormwater, whether through erosion within the watershed or scouring of stream banks and beds, is a common symptom of urbanization (Chin, 2006; Nelson and Booth, 2002). Much of the material eroded consists of inert geologic mineral like quartz, feldspar, etc. (Sartor and Boyd, 1972), and the damage done is often more physical than chemical. Such damages include (all from Sartor and Boyd, 1972):

- burial of bottom plants, animals and habitats,
- alteration of bottom habitats,

- reduction of photosynthesis through increased turbidity,
- alteration of predator-prey relationships due to loss of light,
- transport of biological and chemical pollutants that use sediment as substrate,
- and abrasion of fine biological structures and clogging of gills, feeding and reproductive structures

A change in bottom sediments caused by urban erosion often creates a stream bed that consists of constantly shifting find sands or other small sediments, a much less suitable environment for benthic organisms that dwell or reproduce in stream bottoms (Klein, 1979).

An increase in the amplitude of thermal change is also a common effect of urbanization (Hewlett and Fortson, 1982). Aquatic organisms that are particularly sensitive to thermal stress include cold-water-loving organisms like the benthic invertebrate "EPT" species (Ephemeroptera, Pelcoptera, and Tricoptera – stoneflies, mayflies and caddisflies) (Curtis, 2012), as well as brook, brown and rainbow trout (Galli, 1990). Sculpins are moderately sensitive to heightened stream temperatures (Galli, 1990). Temperature changes do not have to cause organism death in order to change species diversity in streams. Thermal variation both hotter and colder than the ideal range for sensitive benthic species was found to cause reduced adult size and fecundity, which alone could cause a species to be eliminated from a stream due to reduced ability to compete (Sweeney and Vannote, 1978). Reduction in competitive ability was seen at temperature changes as low as 2-3°C (Sweeney and Vannote, 1978). This may be compared to the observation by Galli (1990) that the average summer temperature difference between an undeveloped stream and a highly developed stream in the Washington, DC suburbs was 8.6°F, or the observation by Nelson and Palmer (2007) that the average surge in stream urban temperatures during summer storms was 3.7°C.

The change in flow dynamics is also an important factor in the stress placed on urbanized streams. Since urbanization greatly increases the frequency of bankfull floods, the stresses on stream biota become more frequent and the recovery periods significantly shorter (Klein, 1979). Additionally, for benthic organisms, the increased frequency of high-flows translates to less stream bottom topography and more uniformly coarse bed material, both of which provide less variety of habitat (Chin, 2006; Cianfrani *et al*., 2006). Flashier storm flow can also increase the number of bottom-dwelling organisms that are physically flushed from the stream system (Klein, 1979).

The corollary to high storm flows may be lower base flows during antecedent dry periods (Horner *et al*., 1994; Leopold, 1968). Stream suitability for fish habitat is thought to become severely degraded when base flows drop below 10% of their pre-urbanized average, and fair to degrading conditions are maintained when base flow conditions are at 30% of average between April and September (Klein, 1979). These conditions will often be attained when the contributing watershed reaches 65% imperviousness and 30-45% imperviousness, respectively (Klein, 1979).

While many of the negative effects of urbanization on water quality are physical in nature (temperatures, flow volumes, etc.), pollutants are of course an important impairment as well. The most common cause of water quality impairment in the United States is bacterial pollution

(Struck *et al*., 2008). In the Commonwealth of Virginia since 1995, 661 water bodies have been listed as officially impaired due to bacterial pollution (USEPA Attains Database. Accessed January, 2013, http://www.epa.gov/waters/ir/). In the Washington, DC suburbs of Fairfax City and County, 150 miles of streamway are impaired by excess bacteria (Fairfax County GIS. Accessed April, 2014, http://www.fairfaxcounty.gov/maps/). Bacteria may originate from many sources in the urban environment, as is evidenced by the accumulation of bacteria in street debris (Sartor and Boyd, 1972; Shaheen, 1975) and in runoff (Hvitved-Jacobsen, 1990).

The NURP (USEPA, 1983) studies of stormwater from multiple sites around the US concluded that copper should be viewed as the most important runoff toxicant for aquatic life because:

- it is common,
- it is found more often at dangerous concentrations,
- where other metals are present at dangerous levels, copper is at even more dangerous levels,
- copper is a dangerous to a broader range of species than other metals,
- methods to control copper will likely control other metals as well.

In addition to the foregoing, copper and zinc have a synergistic effect on toxicity; when both are present together, they both become more toxic than if they were in the some concentration alone (Sartor and Boyd, 1972).

The results from NURP suggest that copper levels as low as 0.02 ppm can start to cause negative health effects in freshwater organisms, but because of the intermittent nature of stormwater, it contributes Cu (and all other pollutants) in variable pulses rather than in continuous doses

(USEPA, 1983). This raises the question of how one measures the long term effects of a variable system. The NURP study included an estimation of the metal concentrations that would cause toxicities for typical exposure durations of intermittent stormwater flows. Threshold value concentrations were estimated to cause mortality of the most sensitive members of the most sensitive species; significant mortality concentrations were based on mortality of the most sensitive individual of the 25th percentile species sensitivity (USEPA, 1983). Additionally, the water quality parameters for acute toxicity within Virginia regulations are based on an organism exposure time of one hour (Va. Admin. Code, 9VAC25-260-140), a reasonable facsimile of an intermittent stormwater pulse. The chronic exposure levels are based on an exposure time of four days (Va. Admin. Code, 9VAC25-260-140). Table 2.5-1 lists the threshold toxic levels published in NURP and the Virginia Code.

Table 2.5-1: Toxic Pollutant Concentrations for Freshwater Organisms

Pollutant	Water Hardness (mg/L CaCO3)	Concentration in µg/L			
		NURP[1]		Virginia Standards[1]	
		Threshold	Sig. Mortality	Acute	Chronic
Cu	50	20.0	50-90	--	--
	100	35.0	90-150	13.0	9.0
	200	80.0	120-350	--	--
	300	115.0	265-500	--	--
Zn	50	380.0	870-3,200	--	--
	100	680.0	1,550-4,500	120.0	120.0
	200	1200.0	2,750-8,000	--	--
	300	1700.0	3,850-11,000	--	--
Pb	50	150.0	350-3,200	--	--
	100	360.0	820-7,500	120.0	14.0
	200	850.0	1,950-17,850	--	--
	300	1400.0	3,100-29,000	--	--
Cr^{3+}	100	8,650	8650	570.0	74.0
Cd	50	3.00	7-160	--	--
	100	6.60	15-350	3.9	1.1
	300	20.00	45-1,070	--	--
Ni	100	--	--	180.0	20.0

1: (USEPA, 1983) 2: (Va. Admin. Code, 9VAC25-260-140)

The concentrations in Table 2.5-1can be compared to the stormwater concentrations of metal pollutants in Table 2.4-4 see if the median and means exceed toxic thresholds. The results of the comparison are shown in Table 2.5-2.

Table 2.5-2. Comparison of Stormwater Pollutant Concentrations to Toxicity Standards

Pollutant	Toxicity Standard	Median Concentration			Mean Concentrations			
		NURP[1]	Smullen[2]	Williamson[3]	Bryan[4]	Camponelli[5]	Dorman[6]	Hvitved-Jacobsen[7]
Cu	Va, Chronic[8]	x	x	x	x	x	x	x
	Va. Acute[8]	x		x	x	x	x	x
	NURP-Thresh.					x	x	x
	NURP-Sig. Mort						x	
Pb	Va. Chronic	x	x	x	x	x	x	
	Va. Acute	x						
	NURP-Thresh							
	NURP-Sig. Mort							
Zn	Va. Chronic	x	x	x	x	x	x	x
	Va. Acute	x		x	x	x	x	x
	NURP-Thresh						x	
	NURP-Sig. Mort							

X=median exceeds standard

1:(USEPA, 1983) **2: (Smullen *et al*., 1999)** **3: (Williamson, 1985)**
4: (Bryan, 1972) . **5: (Camponelli *et al*., 2010)** **6: (Dorman *et al*., 1996)**
7: (Hvitved-Jacobsen, 1990) **8: (Va. Admin Code, 9VAC25-260-140)**

A study by Crunkilton *et al*. (1997) attempted to test the toxic effects of stormwater while eliminating the effects of the physical damage urbanization causes within streams. This was done by subjecting D. magna and P. promelas to the ambient water quality of an urbanized stream in flow through aquaria within a protected housing next to the stream bank. One subset of test subjects was subjected to base flow only, and the other subset was subjected to base flow and storm flow. Surprisingly, no significant change in mortality was found between the two subsets.

The author's also found that standard acute toxicity tests of 48-96 hour duration and chronic toxicity tests of 7 day duration were not adequate to describe the toxicity of urban stream water. This is because the toxic effects of stormwater on D. magna did not yield significantly greater mortality than controls until after 7 days of exposure. P. Promelas only showed significant stunting of growth compared to controls after 7 days of exposure, and significant mortality increases only occurred after 17 days. The authors believe that these findings suggest that the negative effects of runoff are not primarily caused by acute exposure during storm flow, but are rather due to longer-term chronic exposure.

Salt is another pollutant that can be found in high concentration in stormwater. The salinity caused by road salts can be deadly to freshwater organisms (Hawkins and Judd, 1972). In addition, salty runoff can cause stratification in ponds and lakes because the heavier salt water sinks to the bottom and creates a density gradient (Hawkins and Judd, 1972). In a lake affected by runoff from road salt, the salinity and lack of mixing on the lake bottom caused the elimination all benthic species (Hawkins and Judd, 1972).

As stated earlier, it does not appear to be possible to assign responsibility for the composite water quality effects of urbanization to any single causative factor. However, there are clear correlations between stream impairments and the level of development within the contributing drainage area. This points to a possible development tipping point: a level of urbanization within a watershed that will cause serious injury to the receiving water. A widely accepted means of characterizing such a tipping point seems to be the percent imperviousness of the contributing drainage area.

Schueler (1994) found that streams would begin to scour their beds and banks at watershed impervious levels of just 10%. After imperviousness passes 10-15%, the EPT macroinvertebrates would be removed and replaced by more tolerant species such as aquatic worms, amphipods, chironimids and snails. Galli (1990) found that trout and other coldwater biota would be lost from streams at an impervious level of 12-15% due to the increased thermal loads. Klein (1979) concluded that average streams would be impaired at 15% watershed imperviousness, but for streams of particular sensitivity—such as those supporting trout—impairment would be seen at 10% imperviousness. Booth *et al.* (2002) and Wang *et al.* (2001) concluded that connected impervious surfaces were most strongly correlated with stream health, and that biotic impairments would be seen when these reached, respectively, 10% and 8-12% of the contributing watershed. Wang *et al.* (2001) also found connected imperviousness was the best measure of urbanization for predicting bank erosion and base flow. Alberti *et al*. (2007) concluded stream biological integrity was most strongly correlated with roadway density and the number of stream crossings, and not a particular level of imperviousness. Other studies have found that simple measures of imperviousness were too simplistic and that site-specific variables—such as riparian canopy condition (Cianfrani *et al*., 2006; Schueler *et al*., 2009), slope, geology (Chin, 2006), and location of impervious surfaces (Alberti *et al*., 2007; Booth *et al*., 2002)—were important determinants of stream biotic health. Despite this variety of conclusions, impervious surfaces can be seen as the primary causative factor of urbanization-induced hydrologic change (Chin, 2006).

On the amount of urbanization that marks the threshold of stream impairment, the literature surveyed is not yet fully in consensus. However, the studies surveyed support the conclusion that when the watershed attains 8-15% imperviousness or greater, a healthy, natural stream may be expected to turn into a degraded urban stream with an unstable channel, highly variable flows, amplified thermal variance, a wide array of potential pollutants, and a population of biota lacking the sensitive species that cannot cope with these changes.

2.6 URBAN STORMWATER REGULATIONS

The regulations that control urban stormwater are spread across all levels of government from the federal to the state and local levels. As the United States has become aware of the significance of the environmental degradation caused by urban stormwater, the laws and administrative rules to control it have become more restrictive and more thoroughly enforced. To obtain a sense of the extent of the current regulations, one need only look at the expected costs of stormwater control. In reports to Congress from the states detailing expected spending on water infrastructure, stormwater management costs have risen the most of any category, from $7.3 billion in 2000 to $25.4 billion in 2004 to $42.3 billion in 2008 (Landers, 2010; USEPA, 2008). Most of these costs are being incurred because of the enforcement of federal nonpoint source quality regulations (Landers, 2010), and despite their increase, the EPA believes that costs are being under-reported (USEPA, 2008). Due to the special needs of the Chesapeake Bay, states within that watershed have particularly high expected costs. The Bay states that reported data constituted three (Maryland, Pennsylvania, New York) of the top seven states in terms of total expected stormwater management costs (USEPA, 2008).

2.6A Federal Stormwater Regulations

Basic water laws have been on the federal books for over 100 years, but it was not until fairly recently that these laws began to focus on protection of water quality rather than just the right of navigation. The forefather of modern federal regulation of water quality was the Water Pollution Control Act of 1948; the importance and strength of this act increased greatly when it was amended in 1972, creating what is commonly known as the Clean Water Act (CWA) (Novotny and Olem, 1994). The CWA's ultimate goal was to ensure that the waters of the United States were conducive to recreation and the propagation of fish, shellfish and wildlife by 1983 (33 U.S.C. §1251 *et seq.*, 2002). That goal has not been met, but the CWA amendments of 1972 were successful in setting up a national regulatory scheme for controlling point source pollution. The enforcement structure is known as the National Pollutant Discharge Elimination Systems, or NPDES (Cox, 2012), and the enabling legislation is section 301 of the CWA (33 U.S.C. §1251 *et seq.*, 2002). Section 301requires that point sources meet effluent limitations created by the EPA that are based upon application of the "best practical control technology currently available" (33 U.S.C. §1251 *et seq.*, 2002).

The basis by which all NPDES effluent limitations are set is the Water Quality Standard (WQS) (USEPA, 2009). The WQS consists of three elements (USEPA, 2009):

- the designated uses assigned to water bodies, such as recreation, public water supply, shellfish harvesting, etc.;
- the water quality criteria or thresholds, expressed as quantitative pollutant levels or qualitative descriptions of the desired water body condition, that must be met to protect the designated uses;

- and the anti-degradation policy that prevents waters from being degraded below their current state of health, even if designated uses can still be met.

The over-riding goal of the CWA is to achieve water quality standards and thus ensure the protection of designated uses. All effluent limitations imposed by the CWA are done so to achieve this goal (Cox, 2012).

WQS numerical criteria can be broken down into two general categories: those that protect aquatic life and those that protect human health and recreation. The aquatic life criteria are further divided into two threshold concentrations: the criterion maximum concentration which defines the threshold for acute toxic effects, and the criterion continuous concentration which represent the threshold for chronic effects (USEPA, 2012). The acute threshold is derived from 48-96 hour toxicity tests of aquatic plants and animals that measures lethality or immobilization, while the chronic threshold is derived from longer term exposures (often 28 days or longer) that measure survival, growth or reproduction (USEPA, 2012). EPA recommends that the one hour average pollutant concentration in waterways not exceed the acute threshold and the four day average concentration not exceed the chronic threshold more than once every three years (USEPA, 2012). This allows for variation in the pollutant concentration, and even occasional spikes above the threshold values, as long as they are of limited duration (USEPA, 2012).

The 1972 CWA amendments were primarily aimed at controlling point source pollution, but some nonpoint source control measures were also specified, particularly in section 303(d) of the act. This section spelled out the process for establishing the Total Maximum Daily Load (TMDL), the maximum amount of a pollutant that could be discharged daily into a waterway while still achieving designated uses (33 U.S.C. 1251 *et seq*., 2002). TMDLs were implemented when the effluent limitations on point sources were not enough to achieve the desired water quality improvements (33 U.S.C. 1251 *et seq*., 2002). TMDLs applied to both point and nonpoint sources. While section 303(d) carried significant regulatory authority, it was not stringently enforced until 10 to 15 years after the passage of the 1972 amendments, and only then as a result of lawsuits filed by environmental groups attempting to prod the EPA into action (Cox, 2012).

Section 208 of the 1972 amendments also enacted a land use planning process that encouraged local municipalities to take a holistic approach to water quality planning that included nonpoint source pollution control (Novotny and Olem, 1994). However, since no formal compliance or penalty structure was set up to ensure nonpoint source controls were enacted, and no formal maintenance or implementation mechanisms were required for the plans themselves, many regional land-use plans that included progressive protections of water resources against stormwater were created and then simply ignored (Novotny and Olem, 1994).

In addition to the EPA taking a more assertive stance on TMDL enforcement, the 1987 amendments to the CWA really brought stormwater control under the regulatory auspices of the Act (NRC, 2009; Urbonas *et al*., 1990; USEPA, 1992a). Section 402(p) of the amended law required municipal stormwater drainage systems (labeled MS4s for Municipal Separate Storm Sewer System) that serve over 100,000 people to obtain an NPDES permit for their stormwater discharges, to prohibit the discharge of any materials except stormwater, and to implement management practices that remove pollutants from stormwater to the maximum extent practicable (33 U.S.C. 1251 *et seq*., 2002; NRC, 2009; USEPA, 2010). In addition to municipalities, certain facilities that are deemed to be special contributors of stormwater loads were also made subject to MS4 permitting (USEPA, 1992a). In Northern Virginia, some of these special contributors are the campuses of George Mason University and Northern Virginia Community College—large landowners with significant impervious infrastructure (Fairfax County GIS. Accessed May, 2013, http://www.fairfaxcounty.gov/maps/). As of 2008, in there were 7,080 communities and facilities in the US covered by an MS4 stormwater permit (USEPA, 2008).

One remaining portion of the CWA that has significant impact on urban stormwater control is section 319. This section requires governors of states to issue a report on the waters of their state that are not attaining their relevant WQS specifically because of nonpoint source pollution, and to submit a management plan detailing how and when that pollution will be controlled (33 U.S.C. 1251 *et seq*., 2002). States that submit section 319 reports deemed suitable by the EPA will be eligible for federal cost share funds that can cover up to 60% of the expense of the plan (33 U.S.C. 1251 *et seq*., 2002).

2.6B Commonwealth of Virginia and Local Stormwater Regulations

While the CWA sets the national framework for clean water laws, much of the regulation of stormwater occurs at the local level. In fact, most of the day-to-day management and enforcement of the provisions of the CWA were delegated from EPA to the states. For instance, the NPDES system for point sources in Virginia is administered by the state Department of Environmental Quality (DEQ) and known as the Virginia Pollutant Discharge Elimination System (VPDES) (Va. Admin. Code, 9VAC25-31). Stormwater regulations for individual properties are promulgated both at the local and state level in Virginia. The state stormwater regulations serve as a minimum standard that must be met by all localities, but any local government is free to enforce stricter rules if it so desires (Code of Va., § 62.1-44.15:33 *et seq*).

Local ordinances governing stormwater began appearing after the post-WWII suburban build-out and the realization that delivering stormwater quickly from roads and homes into streams often led to downstream flooding problems (NRC, 2009). To combat this, the earliest municipal rules regarding storm drainage generally focused solely on flood abatement, and practices such as channelizing and encasing streams were encouraged so that higher flows could pass without overflow into the floodplain (NRC, 2009). Because of the detrimental ecological effects of these practices, regulations began to change starting in the early 1970s, with the focus of stormwater management shifting to the temporary detention of stormwater on-site to reduce peak flows (NRC, 2009).

The stormwater laws in Virginia are primarily contained within three large acts: the Erosion and Sediment Control Act (Code of Va., §62.1-44.15:52 *et seq.*), the Chesapeake Bay Preservation Act (Code of Va., §62.1-44.15:67 *et seq.*), and especially the Virginia Stormwater Management Act (Code of Va. §62.1-44.15:25 *et seq*). Stormwater regulations in Virginia are applied in a stepwise process. The Erosion and Sediment Control Act governs management of stormwater on active construction sites, while the Stormwater Management Act takes over after structures are built and governs stormwater discharges in perpetuity.

The Erosion and Sediment Control Act (E and S Act) (Code of Va., §62.1-44.15:52 *et seq.*) stipulates that all construction sites in Virginia that disturb more than 10,000 square feet must have a permitted plan in place to reduce and control the erosion of sediments from the site, and this plan must meet minimum conservation standards stipulated by the state in order to receive a permit. For municipalities that are fully or partially located below river fall lines, the area of disturbance is reduced to 2,500 square feet (Code of Va., §62.1-44.15:67 *et seq.*) The major conservation standards of the E and S Act that apply to permitted sites include the following:

- Sediment barriers, trapping devices and diversion dikes must be built first before any up-slope clearing can take place.
- Soil stockpiles, borrow areas and any other denuded land must be stabilized within 7 days of vegetation removal if the area will not be reworked within 30 days.
- All cut and fill slopes shall be built to minimize erosion and no concentrated stormwater flows shall be allowed to pass over them.
- All manmade water diversions must be stabilized immediately after completion, and any outlet to a receiving body must have proper protection and lining to prevent erosion.
  (Va. Admin. Code, 9-VAC25-840 *et seq*.)

The overall goal of all E and S Act is to protect downstream properties from any damage caused by erosion and sediment deposition due to increased peak discharge, flows or velocity of stormwater from the site (Va. Admin. Code, 9-VAC25-840 *et seq*). This requirement is assumed to be met if it can be shown that the stormwater conveyances on the construction site will safely pass flow from the 10 year 24 hour storm, will not erode during the 2 year 24 hour storm and that the natural receiving channel that takes construction site runoff will not flood or erode during the 2 year 24 hour storm (Va. Admin. Code, 9-VAC25-840 *et seq*).

Central to stormwater regulation in Virginia is, of course, the Virginia Stormwater Management Act (Code of Va., §62.1-44.15:25 *et seq*). Recently it has undergone a major revision. Both the current (Va. Admin. Code, 9VAC25-870-63) and former (Va. Admin. Code, 9VAC25-870-96) versions of the Act require the maintenance of stormwater quality, as measured by the reduction of phosphorus in stormwater outflows, and the control of quantity, with the same minimum protections of downstream property provided in the E and S Act. Despite these similarities, the new Stormwater Management Act represents a significant tightening of stormwater controls.

The overall goal of the new Virginia stormwater regulations is to ensure that lands being developed "…maintain after-development runoff rates of flow and characteristics that replicate, as nearly as practicable, the existing predevelopment runoff characteristics and site hydrology, or improve upon the contributing share of the existing predevelopment runoff characteristics and site hydrology if stream channel erosion or localized flooding is an existing predevelopment condition." (Code of Va., §62.1-44.15:25 *et seq*). To accomplish the "characteristics" requirement, the total phosphorus load from new development cannot exceed 0.46 kg per hectare per year. For redevelopment sites that cause no increase in imperviousness over existing conditions, total P loads must be reduced 10-20% below the predevelopment load. For redevelopment sites that do increase imperviousness, the reduction criteria for new development shall be applied to the increased impervious areas and the loads from the remainder of the site must be reduced 10-20% below the predevelopment load (Va. Admin. Code, 9VAC25-870-63).

Compared to the old stormwater regulations, these required P reductions are significantly more restrictive. The former regulations (Va. Admin. Code, 9VAC25-870-96) compared the impervious surfaces of new and redevelopment sites to the average imperviousness of the watershed, which was assumed to be 16% unless a watershed-specific number was actually provided. For development that stayed below this average imperviousness, no stormwater quality enhancements were required. For development that exceeded the average imperviousness, the phosphorus discharge after development could not exceed the discharge from the same site with average imperviousness. For redevelopment of sites that already had above average imperviousness, the discharge of P had to be reduced by 10% or it had to equal the discharge from the site based on average imperviousness, whichever was greater.

Upon analysis, the old regulations (Va. Admin. Code, 9VAC25-870-96) actually allow a significant amount of development to create more pollution discharge than existed before. For instance, on an undeveloped site with no impervious surfaces, one could pave over portions of the site and not be forced to remediate the quality of any of the resulting stormwater as long as the total imperviousness was equal to or less than the watershed average. In contrast, the new regulations (Va. Admin. Code, 9VAC25-870-63) state that all new development is subject to a hard pollution cap of 0.41 lbs-P/acre-yr., and that redevelopment of existing built sites must adopt the same standards for all new imperviousness they create, while also reducing the P loadings from the existing imperviousness by 10-20% . In other words, the new regulations place a low ceiling on the discharge allowed from any new hard surfaces, while existing imperviousness must be improved over its existing conditions. The old regulations actually allowed some degradation in quality, as long as it did not exceed a certain benchmark.

The stormwater volume controls in the new regulations also represent a significant increase in restrictiveness. Both the old and new regulations must meet the previously mentioned standards of the Erosion and Sediment Control Act for protection of downstream properties (which focuses on flows from the 2-year and 10-year 24-hour storms) (Va. Admin. Code, 9-VAC25-840 *et seq*.), but in addition to this, the new stormwater regulations require that the peak flow rate from any development that is discharged into a natural receiving body must not exceed the rate obtained through Equation (5): (Va. Admin. Code, 9VAC25-870-66)

$$Q_{Developed} \leq I.F.*(Q_{Pre\text{-}developed} * RV_{Pre\text{-}Developed})/RV_{Developed} \tag{5}$$

$Q_{Developed}$ and $Q_{Pre\text{-}Developed}$ represent the allowable peak flow rate from the site post-development and pre-development, respectively, both in cubic feet per second. The I.F. is a unitless improvement factor that is equal to 0.8 for development sites greater than 1 acre and 0.9 for sites less than 1 acre. $RV_{Developed}$ and $RV_{Pre\text{-}Developed}$ equal the total volume of runoff, in cubic feet, from the site in the post-developed condition and pre-developed condition, respectively.

It may be seen that this puts a very restrictive cap on the allowable peak flow from any developed or redeveloped site discharging directly into a stream or river, especially if the site was initially in hydrologically good condition. While never stated explicitly, one can also see that this requirement pushes developers to reduce runoff volumes, because the closer $RV_{Pre\text{-}Developed}/RV_{Developed}$ is to 1, the less peak flow reduction will be needed to meet the standard. This ties in well with the new regulations goal of recreating or maintaining the pre-development hydrology of the land (Code of Va., §62.1-44.15:25). It is this hydrology based approach that is perhaps the biggest difference between the old and new stormwater regulations of Virginia.

In addition to managing the NPDES program locally, the EPA also grants states the right to set their own Water Quality Standards (WQS) (USEPA, 2009). Virginia has done this and has adopted a standard list of designated uses that apply to all of the state's waterways. They are as follows (Va. Admin. Code, 9VAC25-260-140):

1) Recreational uses
2) Propagation and growth of a balanced, indigenous population of aquatic life
    a. Sub-categories for Chesapeake Bay and tidal tributaries
        i. Migratory fish spawning and nursery
        ii. Submerged Aquatic vegetation
        iii. Open water aquatic life (above the metalimnion)
        iv. Deep water aquatic life (below the metalimnion)
3) Wildlife
4) Production of edible and marketable natural resources

To protect these designated used, Virginia has water quality criteria for 129 different substances (Va. Admin. Code, 9VAC25-260-140). Criteria have been formulated for freshwater acute and chronic toxicity, saltwater acute and chronic toxicity, drinking water, and general human health, although most pollutants have published criteria for only a subset of these uses (Va. Admin. Code, 9VAC25-260-140). A selection of water quality criteria from the Virginia regulations is included in Table 2.6-1. The freshwater aquatic life criteria for metals commonly found in urban runoff were included in Table 2.5-1.

Table 2.6-1. Selected Virginia Water Quality Criteria

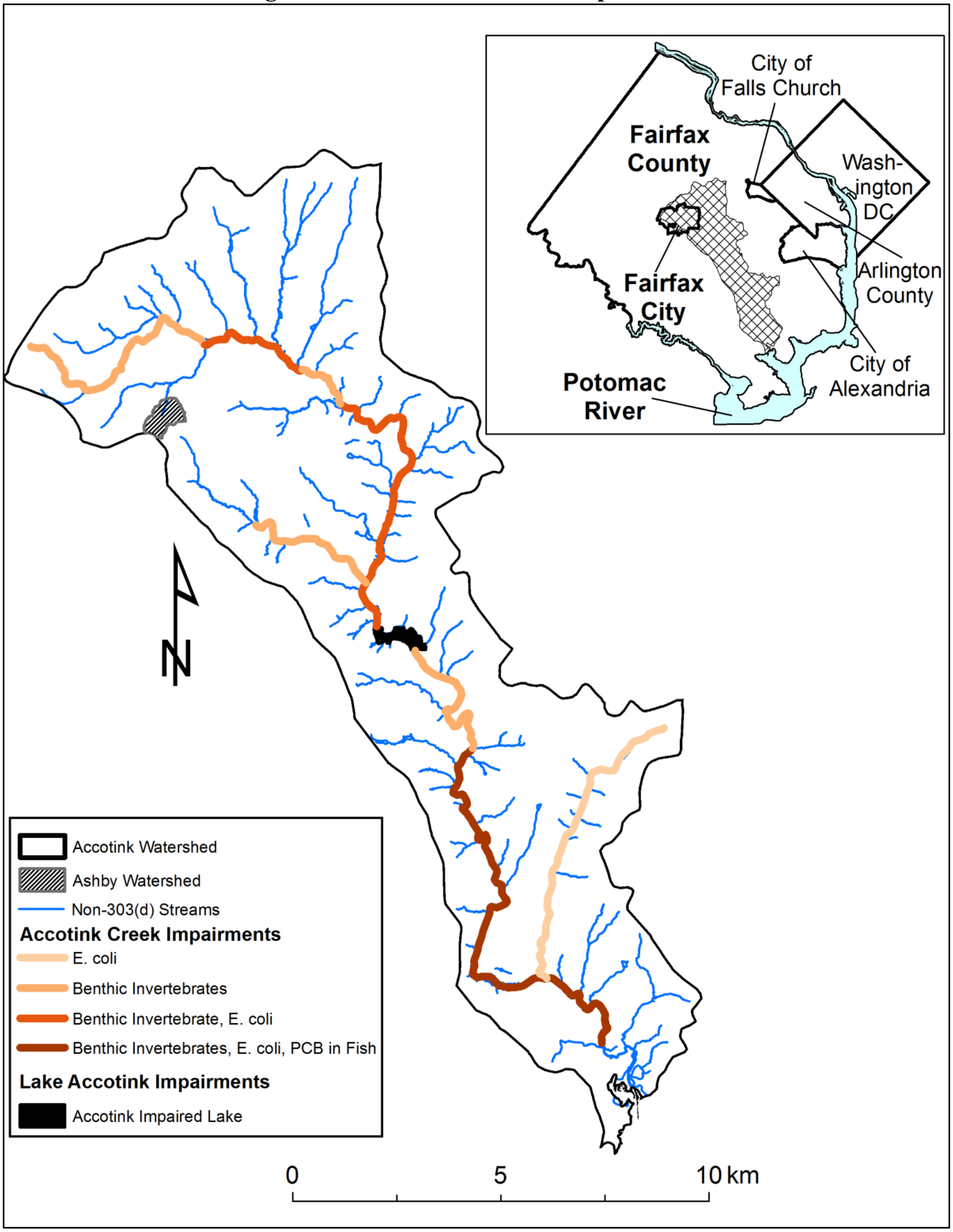
POLLUTANT (UG/L)	USE DESIGNATION					
	AQUATIC LIFE				HUMAN HEALTH	
	FRESHWATER		SALTWATER		PUBLIC WATER SUPPLY	ALL OTHER SURFACE WATERS
	Acute	Chronic	Acute	Chronic		
Arsenic	8	150	69	36	10	--
Benzo (a) anthracene	--	--	--	--	0.038	0.18
Chlordane	2.4	0.0043	0.09	0.004	0.008	0.0081
Chloride	860,000	230,000			250,000	--
Chromium VI	16	11	1,100	50		--
DDT	1.1	0.001	0.13	0.001	0.0022	0.0022
Mercury	1.4	0.77	1.8	0.94		--
Nitrate as N	--	--	--	--	10,000	--
PCB Total	--	0.014	--	0.03	0.00064	0.00064
Total Dissolved Solids	--	--	--	--	500,000	--

Source: (Va. Admin. Code, 9VAC25-260-140)

2.6C Northern Virginia Issues with Water Quality Standards and Total Maximum Daily Loads

Accotink Creek drains a highly developed watershed of Fairfax County and City that includes Ashby Pond and significant residential and commercial infrastructure (Fairfax County GIS. Accessed November, 2012, http://www.fairfaxcounty.gov/maps/; City of Fairfax GIS. Accessed November, 2012, http://www.fairfaxva.gov/). The creek has failed to meet the recreational use, propagation of aquatic life and edible natural resources designated uses and has been listed as officially impaired on the state of Virginia's 303(d) reports (USEPA, ATTAINS Database. Accessed December 29, 2013, http://www.epa.gov/waters/ir/). Outflows from Ashby Pond enter Daniels Run, a small tributary which flows northeast into an impaired segment of Accotink Creek. All of the impaired reaches of Accotink, and their reasons for impairment, are shown in Figure 2.6-1.

Figure 2.6-1: Accotink Creek Impairments



The benthic community impairments in Accotink are thought to be due primarily to excess sediment load in the creek, but rather than place limits on sediment itself, the EPA chose to use flow as a surrogate and placed a TMDL on storm flow (USEPA, 2011b). Through load and wasteload allocations, the TMDL required a 48.4% reduction in storm flow from the 1-year, 24-hour storm (USEPA, 2011b). In July of 2012, the Fairfax County Board of Supervisors and the Virginia Department of Transportation filed suit against the EPA, arguing the agency had overstepped its authority under the Clean Water Act by attempting to regulate a non-pollutant (Letter from Association of Clean Water Administrators National Office to all members, Subject: Federal Court Rules Against EPA's Flow TMDL for Sediment Control). In January 2013, the US District Court for the Eastern District of Virginia ruled in favor of the plaintiffs and found that the CWA only gave the EPA the authority to regulate pollutants. Stormwater, not being a pollutant itself, could not be a surrogate for the actual pollutant causing the impairments—sediment (Virginia Dept. of Transp., *et al*. v. USEPA, 2013 WL 53741, *5 (E.D.Va.)).

The case raised interesting questions about the future direction of stormwater management. As has been shown repeatedly in the literature, the increased volume of runoff caused by urban development does not just act as the transport mechanism for sediments, it is also the cause of sediment enrichment because the increased erosive force of the stormwater flows causes erosion within the watershed and of the banks and beds of streams (Allmendinger *et al*., 2007; Chin, 2006; Nelson and Booth, 2002). Controlling flow volume would certainly seem to be a reasonable way to control sediment loads. In the extant case, the impairment in question is benthic biota, and while neither party disputed that sediment is the likely leading cause of impairment (Virginia Dept. of Transp., *et al*. v. USEPA, 2013 WL 53741, *5 (E.D.Va.)), increased stormwater flows also harm the benthic ecosystem by changing the bed substrate and flushing out smaller individuals (Klein, 1979). Reducing stormwater flow could ameliorate all these damages, while controlling sediments only addresses one. As of now, the court decision applies only to the Eastern District of Virginia, and it has no bearing on the ability of states themselves to control stormwater flow as a pollutant, should they so wish (Letter from Association of Clean Water Administrators National Office to all members, Subject: Federal Court Rules Against EPA's Flow TMDL for Sediment Control).

2.7 STORMWATER BEST MANAGEMENT PRACTICES

A stormwater Best Management Practice (BMP) is "…a device, practice, or method for removing, reducing, retarding, or preventing targeted storm water runoff quantity, constituents, pollutants, and contaminants from reaching receiving water" (Strecker, *et al*., 2001). A popular and common BMP is the stormwater wet pond or retention pond.

2.7A Stormwater Wet/Retention Ponds

A wet pond maintains a permanent pool of water between storm events (Hvitved-Jacobsen, 1990). The permanent pool must hold enough volume to support the desired pollutant reduction mechanisms (NVPDC and ESI, 1992). An inflow structure brings stormwater to the pond while a series of precisely sized outlets allows it to leave at specified peak rates (Hvitved-Jacobsen, 1990). An earthen embankment generally constitutes the downstream perimeter of the pond, retains the water, and gives the pond its shape (USEPA, 1999). Typically, the outlets are located on the face of a riser, a concrete or corrugated metal cylinder that rises from the pond bottom to above the level of the permanent pool (ASCE and WEF, 1992). Pond water flows through the outlets, into the outlet pipe within the riser which then flows through the earthen embankment and outfalls into the receiving water body (ASCE and WEF, 1992). Rather than a riser, an outlet may also simply consist of a weir embedded in the embankment itself (Virginia DEQ, 2011).

The lowest elevation outlet releases low flows or base flows at a controlled rate (Virginia DEQ, 2011) and maintains the height of the permanent pool (ASCE and WEF, 1992). During storm flows, as water volume builds in the pond, the pool elevation rises until the higher elevation outlets are reached and become active. All outlets are designed to release water at a specified rate, and the higher elevation outlets are generally sized to draw down the water volume stored above the elevation of the permanent pool over 24 to 48 hours (ASCE and WEF, 1992; Schueler, 1987). This practice is known as extended detention (NVPDC and ESI, 1992) . For extremely high flows, an emergency spillway, usually a part of the earthen embankment, must be provided to safely pass flows from the 100-year storm without causing failure or erosion of the embankment itself (Virginia DEQ, 2011). All outlets with the exception of the emergency spillway must be protected by a trash rack or other device to prevent clogging (ASCE and WEF, 1992; Virginia DEQ, 2011;). There are no standard designs for such devices, but trash racks should generally have a cross sectional area ten times larger than the outlet it is protecting (ASCE and WEF, 1992).

Around the perimeter of the pool are frequently an aquatic bench and a safety bench (NVPDC and ESI, 1992). The aquatic bench is a shelf of relatively flat land that sits slightly submerged at the edge of the permanent pool and acts as a miniature littoral zone for the wet pond (Schueler, 1987). It provides habitat for rooted emergent and submerged aquatic vegetation, which thrive in the photic zone of water depths less than three feet (NVPDC and ESI, 1992). The vegetative ring of the aquatic bench has aesthetic benefits to the pond, and may also serve as wildlife habitat and prevent shoreline erosion (Schueler, 1987). The safety bench is a similarly shaped shelf of land, but sits just above the permanent pool perimeter (Virginia DEQ, 2011). It remains dry to allow for maintenance access and prevent falls into the pond (Schueler, 1987; Virginia DEQ, 2011).

A wet pond may also feature a forebay and a shelf wetland. The forebay is a small basin situated at the mouth of the inflow and is the first section of the wet pond to receive water (Virginia DEQ, 2011). The design provides some initial sediment settling before releasing the inflow into the rest of the pond (Hvitved-Jacobsen, 1990). A shelf wetland acts similarly to an aquatic bench: it is a shallow basin, generally with a maximum water depth of no more than one foot, that receives stormwater inflows after they pass through the forebay and sends overflows into the permanent pool (Virginia DEQ, 2011).

Wet ponds may or may not have aeration systems, such as internal fountains, to prevent stratification and the development of anoxic conditions at the bottom of the permanent pool (Virginia DEQ, 2011). A typical cross section of a stormwater wet pond is shown in Figure 2.7-1. A plan view of a wet pond with a forebay and shelf wetland is show in Figure 2.7-2

2.7B. Wet Pond Function

The permanent pool of the wet pond allows stormwater to be slowed and detained: the larger the pool volume in relation to the inflow volume, the longer the detention time (Hvitved-Jacobsen, 1990). The wet pond provides for turbulent settling of sediments during storms and quiescent settling between storms (Urbonas, 1995). Settled sediments will deposit and accumulate on the pond bottom (Yousef *et al*., 1994).

Figure 2.7-1. Wet Pond Typical Cross Section

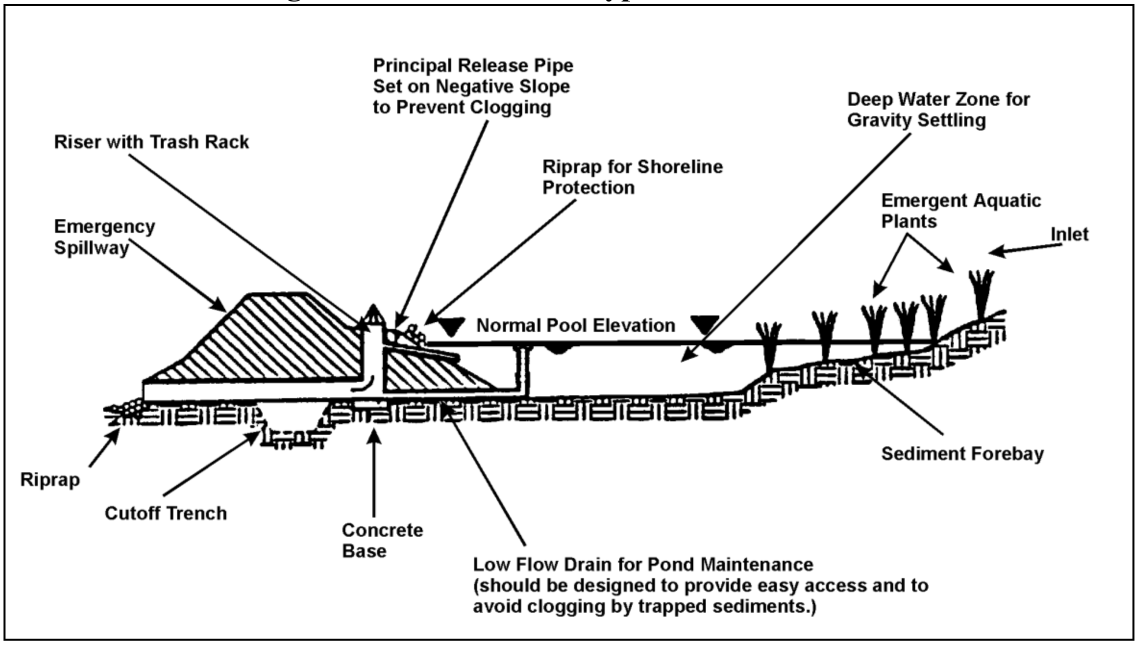


Image Credit: (USEPA, 1999) [Public Domain]

Figure 2.7-2. Wet Pond Typical Plan View

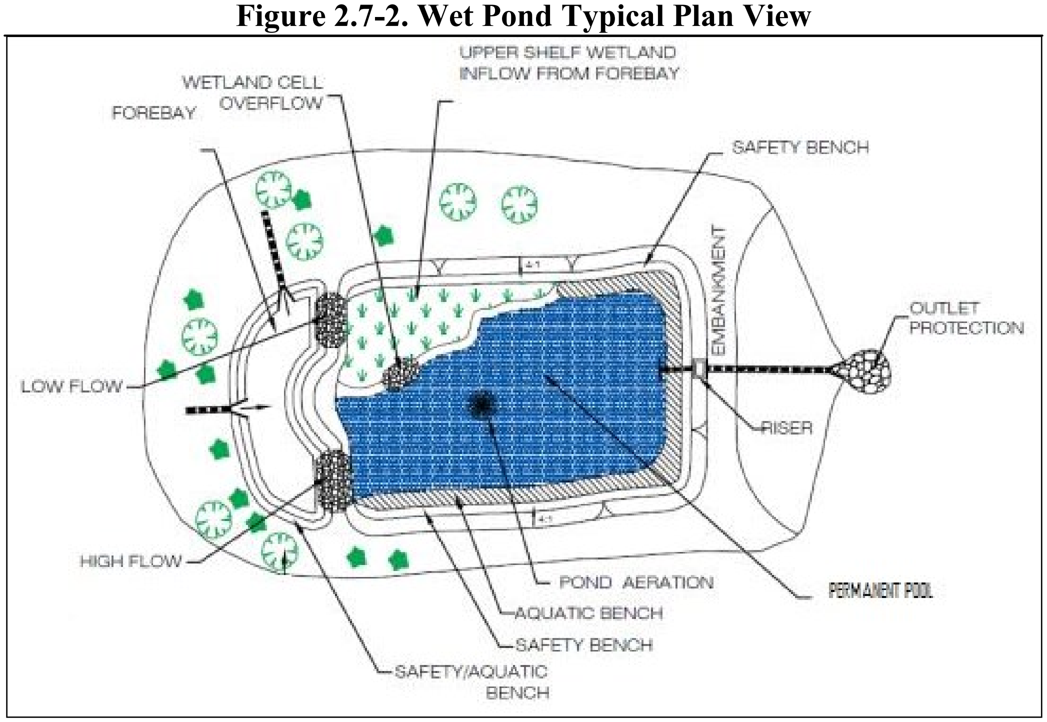


Image Credit: (Virginia DEQ, 2011) [Fair Use]

The extended detention storage that the pond provides on top of the permanent pool also has channel protection benefits (Virginia DEQ, 2011). The stormwater outlets let the extended storage out at a controlled rate that is meant to limit or, ideally, eliminate the bed and bank erosion caused by urban stormwater peak flows (ASCE and WEF, 1992). Significant pollution reduction via settling can also be achieved during drawdown of the extended detention volume (ASCE and WEF 1992; Schueler, 1987).

The benefit of wet ponds over the similar dry or detention ponds is that the stormwater is held during quiescent periods between storms, and during this time biological and chemical mechanisms can remove some dissolved or slowly settleable pollutants (Hvitved-Jacobsen, 1990; Urbonas, 1995). The aquatic plants and algae in a wet pond can take up dissolved nutrients and incorporate them into their biomass (Schueler, 1987). Upon death, the plants settle to the bottom as organic particulates (Schueler, 1987), microbial uptake and decomposition of which can cause further removal of pollutants (Hvitved-Jacobsen, 1990; Schueler, 1987). When the next storm occurs, the incoming storm flow displaces the treated water from the previous storm and sends it through the outlet into the receiving water body (Strecker, 1999).

2.7C Retention Pond Design Theories

The design of a stormwater wet pond is an important determining factor of pollutant removal efficiencies. Many studies have examined wet pond performance, and the reported data shows a great deal of variability, likely due to differing internal pollutant cycling dynamics (Winer, 2000). The varying characteristics of the pollutants themselves, including partitioning between dissolved and particulate phases, particle sizes, settling velocities, among others, may impact removal efficiencies (USEPA, 1993). A properly designed wet pond must account for the natural variability of stormwater pollutants and create an internal treatment process that adequately reduces all pollutant forms (USEPA, 1993).

There are two general design approaches that can be applied to wet ponds: the solids settling method and the controlled eutrophication method (Hartigan, 1989). In the solids settling approach, it is assumed that most pollutants are removed via Type I settling of sediments under plug flow conditions (Hartigan 1989). The controlled eutrophication model focuses on nutrients, and uses their removal as a proxy measurement of total pollutant removal (Hartigan, 1989; NVPDC and ESI, 1992). To do this, the model attempts to quantify biological uptake under mixed flow conditions . The controlled eutrophication model focuses on the cycling of nutrients, and uses their removal as a proxy measurement of total pollutant removal (Hartigan, 1989; NVPDC AND ESI, 1992). To do this, the model accounts for the biological removal of nutrients (Dorman *et al*., 1996; Hartigan, 1989). Both approaches attempt to find the pond volume that will maximize performance efficiency.

According to Hartigan (1989), the solids settling model relies on rainfall/runoff hydrographs and frequencies, the size distributions and settling velocities of stormwater particulates, and the assumed percentage of pollutants that are particulate bound. Given these data, one may calculate the total removal efficiencies at various pond overflow rates by assuming that plug flow conditions and Type 1 settling are maintained within the pond. The final removal rate for each pollutant is calculated by multiplying the TSS removal by the percentage of the pollutant that is particulate bound. Results of this type of analysis have shown that approximately 10% of all settling is accomplished during turbulent storm flow conditions, and 90% occurs during quiescent conditions between storms. Other literature states that settling can effectively remove sediments larger than 20 μm in diameter, but is inefficient at removing smaller particulates (GC and WWE, 2012a).

The controlled eutrophication model is based upon the assumption that nutrient cycling in wet ponds occurs in much the same way as natural lakes, with the principle controlling factors being the loading and decay rates of phosphorus, the hydraulic residence time and the mean depth of the pool (NVPDC AND ESI, 1992). It is a simpler model because it only includes annual phosphorus loads and does not take into account storm-to-storm pollutant variations (Hartigan, 1989). Although phosphorus is the only pollutant analyzed, it is assumed that a pond that can successfully remove phosphorus will remove other nutrient and non-nutrient pollutants as well (Hartigan, 1989).

According to Hartigan (1989), the controlled eutrophication model is based on the phosphorus coefficient model developed by Walker (1985). Phosphorus removal in a wet pond, assuming well-mixed conditions, can be characterized by Walker's second order decay reaction with a rate of K2 (m^3/mg-yr), as shown in Equation (6):

$$K2=(0.056)(QS)(F)^{-1} / (QS+13.3) \tag{6}$$

QS is the mean overflow rate (Z/T), Z is the mean pond depth (m), T is the average hydraulic residence time (years), and F is the ratio of Ortho P to Total P in the inflow.

The decay rate, K2, can then be used in Equation (7) to calculate the total phosphorus retention coefficient (R), which is the measure of the pond's overall efficiency.

$$R=1+91-(1+4N)^{0.5})/(2N) \quad (7)$$

The N variable is equal to (K2)(P)(T), P represents the inflow total phosphorus load (mg), and T is the average hydraulic residence time (yrs).

The recommended wet pond pool volume will generally be around 3 times larger when calculated according to the controlled eutrophication model as compared to the solids settling model (Dorman *et al*., 1996). This is because the solids settling approach assumes pollutants are reduced by sedimentation alone, and shorter detention times are needed to maximize settling processes under plug flow conditions than biological processes under mixed conditions (Hartigan, 1989).

Settling studies have shown that the percentage of TP and TKN that can be removed by settling alone is less than 50% (Dorman *et al*., 1996). The solids settling method is meant to estimate the pond size needed to maximize Type 1 settling, but if Dorman *et al*. are correct, a pond designed this way can only be expected to remove the less than half of nutrients that are particulate bound. Monitoring of wet pond performance has further shown that the reduction of pollutants cannot be due to settling alone (Dorman *et al*., 1996), and that the observed removals of $N0_2^-$, $N0_3^-$, Cu and Zn are strongly linked to biological cycling (USEPA, 1983). In comparing the estimated removals from both design methods to the actual results from Washington, DC area wet ponds monitored in the NURP study, the controlled eutrophication model did a better job of predicting phosphorus removal than the settling solids method (Hartigan, 1989). For these reasons, the controlled eutrophication model is thought to be the better design method for areas where nutrient pollution is a known problem (Hartigan, 1989). This is often the case when a BMP drains to a water body with a long residence time, like a lake, bay or estuary (Dorman *et al*., 1996). Since the Chesapeake Bay is a water body with nutrient pollution problems and a long residence time, the controlled eutrophication model may be the better design approach for stormwater wet ponds within the Bay watershed.

2.7D Maximizing Retention Pond Efficiency

Regardless of the design approach used, it is clear from the literature that the basic factor that determines wet pond pollutant removal efficiency is the size of the pond relative to the size of the influent stormwater volume. There are various ways to make this comparison, with the most direct being the ratio of the volume of the pond to the volume of the inflow. Other measures include the detention time, areal comparisons between the pond and the contributing drainage area, and the overflow rate.

For straight volume comparisons, NURP came to the conclusion that higher ratios of pond basin volume to inflow runoff volume (V_b/V_r) correlated with better sedimentation removal efficiencies (USEPA, 1983). Hartigan (1989) examined a subset of NURP data and found that V_b/V_r correlated well with phosphorus removal efficiency. In the subset, ponds with a V_b/V_r ratio greater than 4 had TP removal in the range of 50-80%. Smaller ponds with V_b/V_r ratios less than 2 could only achieve 10-25% TP removal, suggesting that larger pond volumes increased dissolved pollutant removal. Horner *et al*. (1994) found similar results, concluding that a V_b/V_r of at least 5 is needed to achieve the maximum possible phosphorus removal of 60%. Schueler (1987) noted a basic rule for wet pond volume: if the volume of incoming water is greater than the storage of the pond, some of the stormwater will pass through immediately and with minimal treatment. For this reason, wet ponds—especially smaller ones—do a poorer job at treating the runoff from large storms. Hvitved-Jacobson (Hvitved-Jacobsen, 1990) advocated for a slightly different means of determining the proper wet pond volume. He estimated that a pond should be sized to hold 250-300m^3 of water per hectare of effective drainage area (EDA). EDA is equal to the total drainage area times the runoff coefficient.

Detention time is calculated by dividing the wet pond storage volume by the incoming flow rate. While it is different for every storm, one can easily see how detention relates directly to pond volume: increasing the volume increases the detention time regardless of the inflow rate. Detention time is an important parameter because a wet pond needs to hold pollutants long enough to allow for sedimentation, biological uptake, adsorption, and ion exchange to occur (Hvitved-Jacobsen, 1990; USEPA, 1999). Hartigan (1989) applied the controlled eutrophication and solids settling models to a theoretical wet pond receiving runoff from the same drainage area under the same weather conditions and displayed the results graphically. The controlled eutrophication model predicted that pond phosphorus removal would increase rapidly as the detention time increased from 0 to 2 weeks. At greater than 2 weeks, P removals still increased but with diminishing returns as the relationship between became nearly asymptotic. He found a similar pattern for the solids settling model, but the inflection point occurred at one week. These findings illustrate the importance of detention time as well as the comparatively greater pond volumes required by the controlled eutrophication model. In a similar graphical analysis using NURP data and his own field measurements of pond performance plotted as a function of residence time, Hvitved-Jacobson (1990) found two inflection points for TP and TSS removal: the first occurred at a residence time of 5 days with diminishing reductions thereafter, and the second at 10 days, after which no more meaningful pollutant reductions could be obtained. Horner *et al.* (1994) found that a detention time of 2-3 weeks was needed for the maximum possible 60% phosphorus removal.

In a much earlier and more general analysis, Fitch (1958) concluded that detention time was not of great importance for the settling of larger particles, but it was the most important factor for smaller particles whose settling rate is slow compared to the rate of formation and settlement of floccules. Longer detention times provide more opportunities for collisions and thus floccule formation and settling. In stormwater, many particulates have been shown to be floccules of inorganics with a coating of fine organics (Ellis *et al*., 1982). Detention time seems to be of particular importance in the removal of nutrients by biological means. For ammoniacal-N reduction, the residence time must be long enough to allow for volatilization, microbial oxidation to nitrate and then denitrification to other gaseous forms, and/or biological uptake (GC and WWE, 2012a). Certain BMPs can be termed "nitrate keepers," meaning that they consistently reduce nitrate loads because they provide a long enough detention time to allow for the biological uptake of nitrate (Winer, 2000).

It is common in the literature to find areal comparisons between the size of the contributing drainage area and the size of the receiving wet pond. The EPA recommends that the ratio of the contributing watershed area to the surface area of the pond be 100 or less to ensure that the pond is large enough to provide adequate removal efficiencies (USEPA, 1999). Hartigan (1989) found that a good rule of thumb is for the wet pond surface area to be as large as 1.0% of its drainage basin if the watershed consists of relatively pervious residential development. If the catchment is highly urbanized and impervious, the pond area should expand to 3–5% of the watershed area. Horner *et al.* (1994), found the acceptable range to be between 3-7% of the watershed area. A pond of this area would generally provide the 2-3 week residence time that the authors also suggested was a criteria for maximum efficiency.

The surface overflow rate (SOR), calculated by dividing the stormwater inflow rate by the wet pond surface area, is another commonly used metric for estimating the general efficiency of a wet pond. Theoretically, a settling pond should be able to remove by Type I Sedimentation all the particulates with a terminal settling velocity greater than or equal to the pond SOR (Grizzard, 2013). If removal by Type I settling is the primary goal, then SOR is a more important design factor than detention time (Fischerström, 1958).

Empirical and theoretical relationships have been proposed to show the importance of SOR in the removal of suspended particulates. Fair and Geyer (1954) proposed Equation (8) to show the relationship between particulate settling and SOR.

$$R=1-[(1+(1/n) - (V_s/SOR)]^{-n} \tag{8}$$

R represents the decimal fraction of initial solids removed (R x 100=percent removal); n is a coefficient of turbulence and short circuiting, which ranges from 0 to 1; and V_s is the settling velocity of particles, in English or SI units. To simplify calculations, the continuous distribution of particle settling velocities may be lumped into several groups with each group assigned an average velocity, as was done by NURP (USEPA, 1983) and Dorman *et al.* (1996). These average velocities may be substituted into Equation (8) to estimate the total removal of each group (USEPA, 1993).

Lessard and Beck (1991) modeled the differential change in the concentration of suspended sediments in a settling pond and formulated Equation (9) to model the results:

$$dC_{ss}/dt = \textit{inflow loading rate} - \textit{outflow loading rate} - [(1\text{-}Scour\ Coefficient)(V_s)(A)(C)1/V] \quad (9)$$

C_{ss} is the concentration of suspended solids and C is the instantaneous concentration of suspended solids, both in g/m^3. The Scour Coefficient is a unitless measure of scour along the bottom and sides of the pond. A and V represent the pond area (m^2) and volume (m^3), respectively. V_s is the settling velocity of the sediments (m/h). The inflow and outflow loading rates are measured in $g/(m^3\ s)$.

While the SOR is not explicitly used in Equation (9), the particulate settling velocity, V_s, is the functional equivalent. The equation also reveals the importance of maximizing the area to volume ratio in order to reduce the concentration of suspended solids over time. The A/V ratio is analogous to the SOR: the larger it is, the larger the SOR. An interesting implication is that one can increase the ratio by maximizing surface area and/or minimizing volume. This implies that the shape of the pond is more important for suspended solids removal than the overall volume. As long as the surface area is maximized, the volume makes little difference in removal efficiency, and in fact, a larger volume will result in *less* efficiency. This can be due to the fact that shallower ponds require particles to settle a shorter distance before being removed in the bottom sediments (Horner *et al*., 1994; Schueler, 1987). This implication is attenuated by the inclusion of the scour coefficient. Ponds that are too shallow will suffer from re-suspension of sediments during turbulent flow (Hartigan, 1989; Schueler, 1987), so while Lessard and Beck

(1991) found increased volume to be detrimental to the initial removal by sedimentation, volume is important for keeping the settled sediments sequestered on the pond bottom.

Several other authors have endorsed the idea of shallower basins, even if not explicitly discussing SOR. Shallower basins tend to increase sunlight availability to the pond bottom, and thus increase aquatic plant growth and the biological uptake of pollutants (GC and WWE, 2012a). Shallower depths also increase the amount of oxygen available to the sediments which can increase microbial decomposition rates (Horner *et al*., 1994; GC and WWE, 2012a). Keeping the pond shallow generally will preclude stratification of the waters, which can cause anoxic conditions in the pond sediments and lead to the release of sequestered phosphorus and metals (Hvitved-Jacobsen, 1990; NVPDC and ESI, 1992: USEPA, 1999).

A study by Barrett (2004) arrived at a differing conclusion about the relation of pond size and treatment efficiency. He found that the only statistically significant predictor of wet pond mean outlet pollutant concentrations, with the exception of dissolved phosphorus, was the mean inlet concentration. This relationship held regardless of the pond size except for very large storms where the inflow volume was greater than the pond volume. He believed that most suspended solids settled in the first few hours, but that smaller particulates such as silt and clay hardly settle at all, regardless of the detention time.

With the exception of Barrett (2004), a synthesis of the design recommendations in the literature points to the importance of pond size (as measured by volume, surface, area, V_b/V_r, detention time, SOR, etc.) relative to stormwater inflows, regardless of whether pollutant settling is accomplished by sedimentation, biological processes or both. While the controlled eutrophication model recommends comparatively larger ponds than the solids settling model (Hartigan, 1989), both models correlate increased efficiency with increases in pond size. However, at a certain size, an inflection point is reached where larger ponds continue to increase pollutant removal, but at greatly diminished rates (Hartigan, 1989; Hvitved-Jacobsen, 1990). The inflection point for TSS removal is clearly illustrated in Figure 2.7-3. This graph from the NURP Final Report (USEPA, 1983) shows that for a wet pond with a constant depth and watershed runoff coefficient, increasing the pond surface area results in a greater reduction of TSS, but diminishing efficiency begins after the pond surface area exceeds 0.5% of the contributing drainage area.

Besides the pond size, there are other design parameters that can increase wet pond efficiency. Many of these parameters are meant to prevent short-circuiting of the inflow. There is no formal definition of short circuiting, but an operative definition is that it occurs when a portion of the inflow exits the pond after less than 30-40% of the theoretical detention time (Thackston *et al*., 1987). One common way to minimize short circuiting is to increase the length to width (L:W) ratio of the pond. Long, narrow ponds with a large L:W ratio reduce short circuiting and promote plug flow (Walker, 1998). For an inflow stormwater volume equal to the available pond storage, a pond with an L:W ratio of 0.5 retained 19% of the storm flow and allowed it to mix into less than half of the pond volume; a pond with an L:W ratio of 4 mixed the inflow with the majority of the pond volume and retained 65% after the storm (Walker, 1998). The horizontal short circuiting caused by a small L:W ratio may not have a significant effect on the settling removal or larger particles, but it will likely reduce the removal of smaller particles with slow settling velocities (Fischerström, 1958). Since smaller particles also carry a high concentration of bound pollutants (Camponelli *et al*., 2010; Sartor and Boyd, 1972; Tanizaki *et al*., 1992), their short-circuiting may cause significant reduction in pond efficiency. Recommended L:W ratios from the literature range from 2:1 (Hartigan, 1989) to 5:1 (Horner *et al*., 1994). If the space is not available to build such a pond, the effective length of the flow path can be increased by placing baffles or other obstructions within the permanent pool (Schueler, 1987). The inlet and outlet of the pond should be placed at the most hydraulically distant points from each other so as not to reduce the flow path provided by the ponds shape (Horner *et al*., 1994; Schueler, 1987)

Thackston *et al*. (1987) provided an empirical analysis of the relationship of settling pond efficiency and pond shape. They illustrated the correlation between pond dimensions and the hydraulic efficiency (the ratio of actual detention time to theoretical detention time) with Equation (10).

$$t/T=0.087(L/W)^{0.81}(L/D)^{0.25}(V_s/V_{adv})^{-0.093} \tag{10}$$

t/T is the unitless hydraulic efficiency while L/W is the ratio of the pond length to width. D refers to the depth of the pond. V_s is the particle settling velocity while V_{adv} is the advective velocity of the inflow. All distance and velocity measurements can be in either English or SI units.

The equation shows that the L:W ratio is the greatest determinant of hydraulic efficiency, but it also reveals that depth is inversely related to efficiency. The negative correlation between increased depth and wet pond efficiency was discussed in the previous section, but in this instance, the authors believed that greater depths produced more dead zones—areas of eddy

Figure 2.7-3: Correlation Between Pond Size and TSS Removal Efficiency

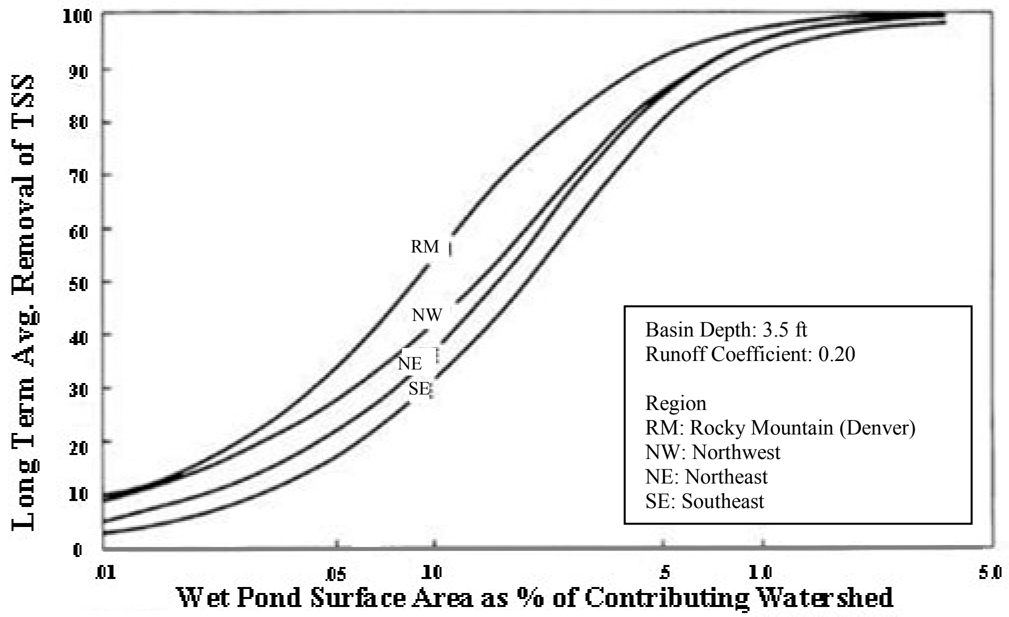


Image Source: (USEPA, 1983) [Public Domain]

recirculation that are hydraulically disconnected from the bulk flow and thus represent a decrease in the volume of the pond available for stormwater mixing and treatment.

The reduced volume leads to greater short circuiting. Hartigan (1989) came to a similar conclusion about depth, arguing against deeper ponds because the potential thermal stratification they underwent would produce a density gradient that would exclude the bottom of the pond from mixing with inflows. As was also noted previously, there is a limit to how shallow the ponds may be. A minimum depth of 2-8 feet was recommended by Thackston *et al*. (1987). Hartigan (1989) argued for a mean depth of 1-3 m, whereas Hvitved-Jacobson (1990) preferred the shallower 1-1.5 m range.

Inflow velocity and flow concentration are also important factors for design. The velocity should be reduced and the flow should be spread across the width of the pond as evenly as possible (Horner, 1994). High velocity concentrated flow into the pond can cause turbulent conditions and the re-suspension of bottom sediments. (Struck *et al*., 2008). Adequate depth at the inlet should be able to reduce the velocity of inflows (NVPDC AND ESI, 1992)

Over the long term, the dredging and removal of sediment is important to maintaining wet pond efficiency. Sediments will accumulate on the pond bottom, and over time will reduce the available storage volume and detention time. Accumulation rates are dependent upon the ratio of the pond surface area to the drainage area. Yousef *et al.* (1994) modeled the accumulation using Equation (11),

$$Accumulation = a[(100*pond\ area)/drainage\ area]^{-b} \qquad (11)$$

where a and b are empirically derived constants. It was found that accumulation rates increase greatly when the pond surface area is 2% or less of the contributing drainage age, and slow greatly above 4% (Yousef et al., 1994). Accumulation rates are also likely dependent on the soil characteristics of the drainage area. Silty soils are highly erodible and will likely determine the dredging frequency of wet ponds (NVPDC and ESI, 1992). Reported dredging cycles in the literature include a 10-20 year frequency (Hvitved-Jacobsen, 1990), and a 5-15 year recurrence for wet ponds in Northern Virginia (NVPDC and ESI, 1992). A sediment forebay that provides some pre-treatment settling before the inflow passes into the permanent pool can reduce the sediment accumulation rate and the need for dredging ((Hvitved-Jacobsen, 1990; NVPDC and ESI, 1992; Virginia DEQ, 2011;).

Rooted vegetation in the pond can removed pollutants from the sediment, aid in sediment settling, and protect the inlet and shoreline from erosion; algae can take dissolved material out of the water column; and both algae and rooted plants add organic seston to the pond that will increase the population and activity of the microbial population (Schueler, 1987). To increase the amount of vegetation in the pond, an aquatic bench around the pond perimeter can be provided to create the shallow, light filled littoral-type environment that will allow vegetation to thrive (NVPDC and ESI, 1992; USEPA, 1999; Virginia DEQ, 2011).

2.8 RETENTION POND PERFOMANCE ANALYSIS

It can be quite difficult to create standard performance ratings for wet ponds or any type of stormwater BMP because of the inherent variability of stormwater volumes and pollutant concentrations and the design variability of the BMPs themselves (Barrett, 2004; USEPA, 1993). A wet pond at a specific location has a fixed size and capacity, but the stormwater flowing to it will vary from storm to storm. As a result, the analysis of wet pond efficiency must account for the intermittent and variable nature of stormwater flows and make reasonable estimates of performance under the range of operating conditions likely to be encountered (USEPA, 1986). This will often mean that pollutant removal efficiencies must be reported as a range, rather than a single number (Maxted and Shaver, 1999).

2.8A Various Measurement Techniques

The variability of event mean concentrations (EMCs) of pollutants in stormwater at specific BMP sites across the country has been found to be best described by the lognormal distribution

(Helsel and Hirsch, 2002; Smullen *et al*., 1999; Strecker, 1999; USEPA, 1983). The median EMCs from site to site also follow the lognormal distribution (USEPA, 1983). This illustrates the variability of stormwater quality, but also gives a standard distribution transformation by which it may be statistically analyzed.

Stormwater data, both quality and volume, are generally right skewed: high outliers stretch out the right tail of the distribution, but the left tail remains fixed because concentrations and volume cannot be less than zero (GC and WWE, 2009; Helsel and Hirsch, 2002). These high outliers will cause the mean to inflate and become larger than the median (GC and WWE, 2009; Helsel and Hirsch, 2002). Because of this, if one seeks to compare the efficiencies of BMPs using a single measure of central tendency, it is advised to use the median rather than the mean (USEPA, 1983).

The variability in stormwater characteristics and wet pond performance is nearly matched by the variety of methods employed to measure efficiency. Unfortunately, each method appears to produce somewhat different results. This variability and lack of standardization is one reason it can be difficult to determine standard effectiveness estimates of various BMPs (Strecker *et al*., 2001; Urbonas, 1995;).

One of the most popular methods of reporting BMP performance is the efficiency ratio (ER) (GC and WWE, 2009), which is calculated according to Equation (12).

$$1\text{-}(\textit{Average outlet EMC}/\textit{Average Inlet EMC}) \tag{12}$$

While common in the literature, there are several problems inherent in the ER method. Since it uses average EMCs, the ER gives all storms equal weight, even though large storms carry a much greater pollutant load than small ones (Dorman *et al*., 1996; GC and WWE, 2009). Additionally, the removal efficiencies of all BMPs have been shown to be strongly dependent on the inflow pollutant concentrations (Barrett, 2004). As a result, efficiencies are often dependent on the characteristics of the storm, not just the BMP. Generally, ER performance is poor for storms with low pollutant concentrations and improves as inflow concentrations rise (Strecker *et al*., 2001; Urbonas, 1995). A BMP will often show poor removal only because it is treating cleaner storms (Winer, 2000). The storms with the lowest EMCs also tend to be the smallest volumetrically, and as previously stated, these storms already skew ER results because they are given the same weight as larger storms with larger loads. Additionally, in the rare cases in which stormwater volume is significantly reduced during passage through the wet pond, the resulting pollutant reduction cannot be quantified by the ER because it only considers concentration (Dorman *et al*., 1996).

The summation of loads (SOL) method is a common mass-based measure of efficiency shown in Equation (13).

$$\mathit{1\text{-}(total\ loads\ in/total\ loads\ out)} \tag{13}$$

The SOL, being based on the long term sum of loads, has the benefit of lessening the impact of storm-to-storm variability on overall BMP efficiency (USEPA, 1986). The accuracy of the method relies on the assumption that the monitoring period is representative of the entire population of potential storms and that there is not significant export of material during the dry periods between storms (GC and WWE, 2009). The SOL is often dominated by a few large storms with large pollutant loads (GC and WWE, 2009). Since only total load reductions are analyzed, this method cannot be used to determine if BMP outlet concentrations exceed benchmarks such as water quality standards (GC and WWE, 2009).

The efficiency ratio and summation of loads methods can also be applied to individual storms to determine concentration or mass reductions on a per-storm basis (GC and WWE, 2009; Winer, 2000). Reductions from each storm can be compared individually, or the average for all storms may be calculated. While similar in concept, the BMP performance calculated by averaging the efficiency ratio of each storm will differ from performance calculated by the normal ER, which compares the average EMC of all inflows versus the average EMC of all outflows (Winer, 2000). The same problems inherent in the ER apply equally to both the per-storm approaches. Both weight all storms as if they are equal, which ignores the fact that there are great variations in pollutant loads based on storm size (Strecker *et al*., 2001).

The performance of BMPs can also be compared to relevant benchmarks. For instance, outlet EMCs from a wet pond can be compared to the water quality standards that are applicable to the receiving water body (GC and WWE, 2009). Outlet concentrations can also be compared to the best possible effluent quality that can be expected from the BMP. Schueler (1996) created a set of "irreducible" concentrations that represented what he believed to be the best BMP pollutant reductions achievable. The effluent EMCs of a wet pond could be compared to these irreducible concentrations to see how close they come to matching the highest achievable efficiency (GC and WWE, 2009).

A performance measure called the "effluent probability method" was described following an analysis of data from the International Stormwater BMP Database (GC and WWE, 2009). The first step of a multi-step process is to determine if the BMP is providing treatment by using appropriate parametric or non-parametric statistical hypothesis tests to determine if inflow and outflow water quality are significantly different. In order to do this, the base data must first be analyzed to see if it adheres to a certain distribution, such as the lognormal (Helsel and Hirsch, 2002). Parametric hypothesis tests may be applied if the data do follow a standard distribution (normal, lognormal, gamma, etc.); otherwise, non-parametric tests should be applied (Helsel and Hirsch, 2002). The results of all statistical tests should be given with their confidence intervals, which are typically set at 95% (GC and WWE, 2009). The second step of the effluent probability method is to display the performance results graphically (GC and WWE, 2009). The cumulative distribution functions of the inflow and outflow EMCs or loads, placed side by side on probability or QQ plots, give clear and understandable visual evidence of the performance of the BMP (GC and WWE, 2009; Helsel and Hirsch, 2002).

2.8B Actual Field Measurements of Efficiency

The average performance of wet ponds from several multi-pond studies as well as the reduction credits assigned to wet ponds built according to Virginia state design specifications are given in Table 2-8-1. As may be seen from the table, the range of removal efficiencies is quite wide. Even within an individual study, where a range of pollutant removals is given for a single contaminant, that range is often large (for instance, the 7-88% ER ranges for lead removal reported by Dorman *et al*. (1996)). The two measurement approaches, ER and SOL, also resulted in quite different pollutant removals. None of the studies met the Virginia phosphorus

Table 2.8-1: Average Wet Pond Performance and Virginia Assumed Performance

CONSTIT-UENT	GC and WWE (2012a,b)[1]	WINER (2000)[2]	DORMAN *et al.* (1996)		NURP (1983)[4]		VIRGINIA REDUCTION CREDIT (2011)[5]	
	Conc.	-	Mass	Conc.[3]	Mass	Conc.	Mass	Conc.
TP	43	49	46	21	34	37	50	50
OP	54	70[7]	-	-	29[6]	38[6]	-	-
TN	30	32	-	-	-	-	30	30
OX-N	58	62	60	56-58	-	27	-	-
NH3+4	-	-			-	-	-	-
TSS	83	80	60	31	60	38	-	-
TOC	-	-	32	19	-	-	-	-
DOC	-	-	-	-	-	-	-	-
COD	-	-	-	-	15	15	-	-
Tot. Zn	-	-	62	52	-	30	-	-
Tot. Cu	-	-	47-57	43-72	-	33	-	-
Tot. Pb	-	-	22-77	7-88	62	66	-	-

[1]: Average reduction of median EMCs. TP and TSS data from ponds within or near Chesapeake. All other data from international database

[2]: The average of many individual studies that used different performance calculations

[3]: Average reduction of median EMCs

[4]: Average reduction of mean EMCs

[5]: Assumed performance of ponds built to Virginia DEQ (2011) Level 1 specifications

[6]: Reduction of soluble phosphorus

reduction credit of 50%, but the data reported by Winer (2000) were very close. All of the performance data in Table 2.8-1 was developed from ponds that were built significantly before the current Virginia design specifications were implemented, and most pond sites are not in Virginia, so their failure to attain the assumed 50% reductions of TP and TN does not necessarily mean they are underperforming for their intended purposes.

A review of wet ponds in the Chesapeake Bay Watershed found that wet ponds were providing statistically significant reductions of TP, TKN and TSS (GC and WWE, 2012a). In an analysis of the International Stormwater Database, the only BMPs to significantly reduce TKN were wet ponds and bioretention (GC and WWE, 2009). In a US-wide analysis of wet ponds, Winer (2000) found average bacterial reductions (combined removals of E. coli, fecal strep, enterococci and fecal and total coliforms) were 70% (Winer, 2000)

Struck *et al.* (2008) determined that wet pond bacterial reductions could be modeled by a standard first order decay reaction with a rate constant, $K_{overall}$ (h^{-1}), as shown in Equation (14).

$$K_{overall} = K_{20}\Theta_T^{T-20} + \Theta_L L + K_{other} \tag{14}$$

$K_{overall}$ is determined by K_{20}, the inactivation rate constant due to temperature at 20°C (h^{-1}); the unitless temperature coefficient, Θ_T; the temperature in degrees Celsius, T; the light proportionality coefficient, Θ_L, in units of cm^2/mWh; the light intensity, L, in mW/cm^2; and the inactivation rate (h^{-1}) due to other environmental factors such as adsorption, filtration, sedimentation, pH, DO and redox potential. While K_{other} demonstrated that the exact effects of individual environmental variables were difficult to segregate, the authors did calculate total k values for different bacterial species commonly found in stormwater. The k values varied with season and bacteria species and ranged between 0.044 and 0.335. The authors further believed that there was an irreducible concentration that represented the lowest possible bacterial concentration that could be reached by any BMP.

2.9 RETENTION POND PROBLEMS

Wet retention ponds have become a staple BMP in the United States, frequently recommended by regulators and used by developers (Hancock and Chambers, 2010). Their long residence times and assemblages of plants and microbes make them particularly well suited for removing dissolved pollutants like nutrients from stormwater, a feat that extended detention ponds cannot match (Hartigan, 1989). Yet, they are not without their problems. Despite their proven water quality benefits, they can also place added stress onto receiving waters.

2.9A Thermal and Dissolved Oxygen Issues

Thermal pollution is a problem that is often exacerbated by wet ponds (Galli, 1990; Struck *et al*., 2008). The same traits that make a wet pond efficient at sediment and dissolved pollutant removal—long detention times, large pool surface areas, and sunlight irradiance into the pool—also cause the temperature of the pond water to rise which can cause thermal stress in downstream biota (Galli, 1990; Roseen *et al*, 2010). Data from a wet pond in suburban Maryland showed that average water temperatures rose by 5.4°C during base flow and 4.7°C during storm flow, imitating the thermal changes caused by heavy urbanization of a watershed (Galli, 1991). Releases from wet ponds are sufficiently hot to stress or kill some sensitive downstream organisms like trout, sculpin and the EPT benthic community (Galli, 1990; Roseen *et al.*, 2010). Some of this effect may be minimized by choosing an outlet structure that draws cooler water from a foot or so below the permanent pool surface (Schueler, 1987), but even with precautions like this, wet ponds may not be suitable in watersheds that drain to streams that are not thermally stressed at the outset (Galli, 1990).

Connected to the thermal pollution issue is the related problem of dissolved oxygen (DO) in the pond discharge. Warmer waters contain less DO, and as a result, a pond can release oxygen-deprived water into its receiving body (Galli, 1990). In addition, the BOD within the pond can also depress DO outflow concentrations (Galli, 1990). DO restrictions were seen in the effluent of a thermally-stressed wet pond in Maryland, but it was noted that with turbulent flow in the receiving body, the DO levels would recover within 61-122 m of the pond outfall (Galli, 1990). Oxygen depression may be an issue in ponds that drain to relatively calm water bodies, but an adequate outfall that can create aerated, riffle-like flow should be able to reduce or eliminate low DO concerns in receiving waters.

2.9B Improper Sizing/Location

Wet ponds are generally designed to control the stormwater from a certain design storm of known frequency. For instance, Virginia stormwater laws require that wet ponds safely control the 2-year and 10-year 24-hour storms (Va. Admin. Code, 9-VAC25-840 *et seq*). The characteristics of such storms and the runoff they create is often estimated by using standard hydrologic models like the Rational Method or USDA-NRCS TR-55 (Hancock et al., 2010). Hancock et al. (2010) analyzed wet ponds designed according to the runoff estimates predicted by standard models. They found that actual inflow volumes and runoff coefficients were higher than model predictions 64% and 80% of the time, respectively. This caused the monitored wet ponds to frequently underperform their outflow peak shaving requirements. The authors proposed three reasons why this underestimation may have occurred:

1) Rainfall intensities during design storms likely exceeded the intensities predicted in USDA synthetic rainfall curves used for BMP design.

2) Runoff coefficients and TR-55 curve numbers likely underestimated actual conditions. Semi-pervious surfaces such as graded and grassed lawn and other open soil areas were likely not credited with creating as much runoff as they did.

3) "Piggybacking" by storms may have caused new inflows to enter the ponds before previous inflows were fully released. The pond storage volumes were not large enough to accommodate the combined flows.

In Virginia, the new state stormwater regulations specifically approve of the use of TR-55 and the Rational Method for estimating inflow runoff volumes and peak flow rates (Va. Admin. Code, 9VAC25-870-72). If Hancock et al. (2010) are correct in their findings, Virginia wet ponds and other BMPs may be consistently under-sized and unable to achieve their estimated pollutant and flow rate reductions.

The location of the wet pond within a watershed is also of concern. Ponds located near the outlet of the watershed may detain inflows long enough so as to release the stored water into the receiving body just as the upstream peak discharge reaches the same location (Schueler, 1987). Rather than reducing peak flow, the pond would actually add to it. To avoid this, comprehensive watershed modeling should be done before the proper location of a pond is determined (Schueler, 1987).

2.9C Limits of Effectiveness

Assuming that a pond is well positioned, properly designed for the storm flow conditions it will experience, well maintained, and otherwise functioning as intended, there is still a limit to what it can accomplish. The hydraulic changes wrought by urbanization can be so complete that "…even with more environmentally friendly urban development practices, urbanization probably will eventually degrade stream ecosystems once it exceeds a certain threshold level. All the urban best management practices can do is to raise such a threshold level" (Wang *et al*., 2001).

While wet ponds can reduce peak flows and filter out pollutants to improve water quality (NVPDC and ESI, 1992; Schueler, 1987; Schueler, 1994), because they are designed to simply hold and treat runoff, they do not reduce the significant increase in post-development stormwater volume (Maxted and Shaver, 1999), nor do they increase the low summer base flows of an urbanized watershed (Thomas *et al*., 2011). Retention and detention ponds have not consistently proven to reduce the number of bankfull floods (Schueler, 1987). While some studies have found that watersheds whose runoff was well controlled by retention and detention facilities suffered from lower peak flow s and less channel widening (Allmendinger *et al*., 2007; Meierdiercks *et al*., 2010), others have found that bank erosion still occurs downstream of these ponds (Booth *et al*. 2002). Even if wet ponds consistently provide flow attenuation and increase channel stability, that may not be enough to prevent biotic degradation because biological impairment often occurs before noticeable morphological changes (Chin, 2006). An analysis of stream habitats and biological communities downstream of retention ponds found that the ponds did not significantly improve the habitat conditions of the stream or the diversity of the benthic invertebrate population when compared to urbanized streams with no stormwater controls (Maxted and Shaver, 1999).

Wet ponds play an important role in reducing pollutant loads and concentrations, but they cannot recreate the natural hydrology that existed before development (Booth *et al*., 2002). In fact, altered site hydrology can be considered a built-in feature of the wet pond (Shaver, 1998) because the pond does not limit excess flow; it simply doles out stored stormwater over an extended time span. Stormwater controls in Virginia are shifting from an emphasis on traditional stormwater BMPs like detention and retention ponds to Low Impact Development (LID) practices that seek to recreate the pre-existing natural hydrology (Code of Va., §62.1-44.15:52 *et seq*). This does not mean that wet ponds will no longer have a role in stormwater management, but the state is now emphasizing their use as a last line of defense—a means of filtering and storing the excess stormwater that remains after upland practices have been employed that treat *and reduce* runoff volumes (DEQ, 2011). Beyond the new emphasis on LID techniques, a holistic, watershed-wide approach is most likely necessary to protect streams and other water bodies from the negative impacts of urbanization. This approach will need to include impervious surface limits, forest retention, riparian buffer preservation, protection of wetlands and unstable slopes, and BMPs such as the wet pond (Booth *et al*., 2002).

3. MANUSCRIPT

THE PERFORMANCE OF A STORMWATER WET POND IN TREATING URBAN RUNOFF IN THE CHESAPEAKE BAY WATERSHED

3.1 ABSTRACT

Ashby Pond in Fairfax City, Virginia was retrofitted to treat stormwater from 54.7 hectares of mixed urban land use. The pond flows into Accotink Creek, a highly urbanized tributary of the Potomac River and Chesapeake Bay that is on the State of Virginia 303(d) list for multiple impairments. The entire multi-state Chesapeake Bay Watershed is subject to Total Maximum Daily Load (TMDL) restrictions on sediment, phosphorus and nitrogen. Virginia and local municipalities assign pollutant reduction credits to retention ponds that meet certain design requirements, but to meet the water quality goals of existing and future TMDLs, it must be proven that such ponds truly provide the water quality benefits which they have been assigned. The inflow and outflow water quality of Ashby Pond was examined over 7 months from fall 2012 to spring 2013. The pond provided statistically significant reductions of phosphorus, nitrogen and suspended sediments, but not organic carbon or oxygen demand. Ashby Pond had non-significant export of sodium, chloride and calcium. The pond underperformed with regard to the state reduction credits for phosphorus loads and concentrations, but exceeded them for nitrogen loads and concentrations. The retrofitted pond was under-sized when compared to the state design standards and some underperformance should be expected.

3.2 AUTHORS

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3.3 INTRODUCTION

In the United States, urban stormwater is thought to be the 5th most important cause of impairments of streams and rivers and the 10th most important cause of impairments of estuaries (USEPA, ATTAINS Database. Accessed December 29, 2013, http://www.epa.gov/waters/ir/). As cities expand, the damage caused by stormwater may be expected to grow relative to other nonpoint sources (USEPA, 1992a). The harm done by urbanization is caused not just by the pollutants carried in stormwater, but also by the physical and hydrologic changes it brings about in receiving waters (Klein, 1979; USEPA, 1983; Wang *et al.*, 2001).

Hydrologic Effects

Urbanization has three related effects on hydrology: changes in peak flow, changes in total runoff and changes in the quality of water (Leopold, 1968). During development, the land is rendered impervious by a capping of roads, rooftops, footpaths and parking lots (Hollis, 1975); the permeability of remaining soil is lessened by compaction and grading (Hancock and

Chambers, 2010); and spongy native forests are replaced by suburban lawns (Booth *et al.*, 2002) Results of these changes are large increases in the volume and flow rate of surface runoff and a proportional decrease in the volume of soil moisture, groundwater and stream base flow (Booth *et al*., 2002; Dietz and Clausen, 2008; Fletcher *et al.*, 2013; Hollis, 1975; Horner *et al.*, 1994; Jacobson, 2011; Leopold, 1968; Schueler, 1994; USEPA, 1992a; Wang *et al.*, 2001; Yang *et al*., 2011). The altered hydrology causes widespread impacts on receiving bodies, including in-stream erosion (Allmendinger *et* al., 2007; Nelson and Booth, 2002), reductions in the populations of sensitive aquatic species and increases in the populations of tolerant species (Alberti *et* al., 2007), increased inputs of nutrients and toxins (Carey *et* al., 2013; Walsh et al., 2005), and increases in water temperature (Nelson and Palmer, 2007). After urbanization, average peak flows can be 2 to 4 times greater than pre-development conditions (Chin, 2006). Total runoff volumes have shown a positive linear correlation with increasing watershed imperviousness (USEPA, 1983; Novotny and Olem, 1994; Schueler, 1994), but for small watersheds with little ability to attenuate runoff spikes, the increase may be logarithmic (Dietz and Clausen, 2008). Urbanization also increases stormwater velocity as it flows over smooth asphalt, graded land, uniform turf grass; and through concrete pipes and culverts (Fletcher *et al*., 2013; Hollis, 1975; Horner *et al.*, 1994; Jacobson, 2011). Accordingly, the time of concentration is greatly reduced (Booth *et al*., 2002; Chin, 2006).

Increased watershed imperviousness can greatly increase the frequency (Schueler, 1987) while also decreasing the duration (Poff *et al.,* 2006) of bankfull events. For less frequent flood events, urbanization increases flow volumes (Booth and Jackson, 1997) but with diminishing effect as flood size increases (Yang *et al*., 2011). Since groundwater is the primary source of dry weather flows in streams and rivers, its reduction may cause a similar loss of base flow (Horner *et al.*, 1994). In a study of US Mid-Atlantic Piedmont streams, base flow typically declined to less than 10% of the regional pre-development average when watershed imperviousness passed 65% (Klein, 1979). However, other studies found inconsistent correlations between urbanization and base flow (Walsh *et al*., 2005), and a few even observed increases, likely due to pipe leakage and irrigation (Jacobson, 2011; Schueler, 2009). Urbanization tends to increase flow variability relative to undeveloped watersheds (Poff et al., 2006). Illustrating this, an urbanized headwater stream in Durham, NC had a more than 7,000 fold increase from its average base flow to its maximum storm flow (Bryan, 1972).

Physical Effects

In response to urbanization, streams first clog with sediment washed from active construction (Chin, 2006). After installation of impervious surfaces and stormwater conveyances, this accumulated sediment is flushed out and stream beds and banks begin to scour to create the wider and deeper channel needed to convey the larger flow volumes and rates of the urban watershed (Chin, 2006; Wang *et al.*, 2001). Channel scour is one the leading sources of suspended sediment in streams (Allmendinger *et al*., 2007; GC and WWE, 2012a; Nelson and Booth, 2002). In Northern Virginia, 64% of the annual sediment load in a typical urban stream was estimated to come from in-stream erosion (Medina and Curtis, 2011). Physical damage from suspended sediments includes burial of benthic plants, animals and habitat; reduction of photosynthesis through increased turbidity; transport of pollutants that use sediment as a substrate; and the abrasion and clogging of sensitive biological structures (Sartor and Boyd, 1972).

An increase in the amplitude of thermal change is also an effect of urbanization (Hewlett and Fortson, 1982; Nelson and Palmer, 2007). Impervious surfaces absorb and re-radiate heat that can be transmitted to runoff and the air (Galli, 1990; Schueler, 1994). The loss of forest canopy cover—by development or the erosive widening of stream channels—exposes surface water to the direct irradiance of the sun (Hewlett and Fortson, 1982; Schueler, 1994; Wang *et al.*, 2001) and the heated urban atmosphere (Galli, 1990). In the winter, the lack of a riparian canopy causes water temperatures to decrease (Hewlett and Fortson, 1982). These thermal variations are likely to lead to reduced populations of sensitive invertebrates and fish (Kumar, *et al*., 2013; Nelson and Palmer, 2007; Sweeney and Vannote, 1978).

Stormwater Pollutants

The mass of contaminants flushed into receiving waters during storm flow can exceed months of background loading from base flows (Characklis and Wiesner, 1997). Pollutants commonly characterizing urban runoff include suspended sediments; oxygen demand; forms of nitrogen (N) and phosphorus (P); the metals copper, lead and zinc (Strecker, 1999); bacteria (Hvitved-Jacobsen, 1990; Shaheen, 1975) and dissolved salts (Hawkins and Judd, 1972; USEPA, 1992a). Lead (Pb), cadmium (Cd), copper (Cu) and zinc (Zn) are commonly used to characterize the metals load of urban stormwater because they are almost always present in elevated concentrations (Strecker, 1999). Roadways are a particularly concentrated source of such metals (Davis and Birch, 2009; Novotny and Olem, 1994; Shaheen, 1975), as are residential areas (Davis and Birch, 2009). While Zn is often found in the highest concentrations (Camponelli *et al.*, 2010), Cu is thought to be more dangerous because it is frequently present at toxic levels (Camponelli *et al*, 2010; USEPA, 1983), and is harmful to the broadest range of aquatic species (USEPA, 1983). Concentrations of Cu, Zn and Pb in runoff (Bryan, 1972; Camponelli *et al.*, 2010; Dorman *et al.*, 1996; Hvitved-Jacobsen, 1990; Smullen *et al.*, 1999; USEPA, 1983;

Williamson, 1985) generally exceed the state of Virginia (Va. Admin. Code, 9VAC25-260-140) chronic and acute exposure limits.

The primary watershed sources of sediment include eroded soils, road surfaces, and atmospheric deposition (GC and WWE, 2012a; Nelson and Booth, 2002; Novotny and Olem 1994; Sartor and Boyd, 1972). Clays, organic matter, and iron (Fe) and manganese (Mn) oxides can sorb many pollutants to their surfaces (Camponelli *et al.*, 20120; Hogan and Walbridge, 2007; Novotny and Olem, 1994). In settling column studies by Dorman *et al*. (1996), effective removal of all suspended sediments correlated with the removal of 87% of Pb, 61% of Cu, 46% of Zn, 49% of P and 30% of total Kjeldahl nitrogen (TKN), suggesting these proportions were particulate bound. Smaller sediments tend to contain higher concentrations of pollutants because their relatively larger surface areas allow for greater sorption (Camponelli *et al.*, 2010; Sartor and Boyd, 1972) and because metal-rich automotive wear and tear is often a small particulate itself (Camponelli *et al.*, 2010).

For nutrients, stormwater sources of organic N and P include: vegetative detritus (Novotny and Olem, 1994), fecal material (GC and WWE, 2012a; USEPA, 1992a), landfill leachate (Novotny and Olem, 1994) and soil erosion (GC and WWE, 2012a). Organic N and P sources are also contribute oxygen demand (Strecker, 1999) and bacteria (Struck *et al.*, 2008). Sources of inorganic N and P include lawn fertilizers and eroded mineral soils (GC and WWE, 2012a; USEPA 1992a); nitrogen oxides from fuel combustion; and inorganic P in flame retardants, corrosion inhibitors, and plasticizers (GC and WWE, 2012a).

In areas of significant snowfall, salt used as road treatment is a pollutant of concern (Sartor and Boyd, 1972; USEPA, 1992a). In the US, up to 200-400 pounds of salt is applied per lane mile, and nearly all will eventually dissolve in runoff (Hawkins and Judd, 1972). Salt can increase the levels of metal in runoff by increasing automotive corrosion (Novotny and Olem, 1994).

Biological Effects

Using impervious surfaces as a proxy for urbanization, many studies have reported a tipping point at 10%-15% total watershed imperviousness beyond which stream biota become noticeably impaired (Galli, 1990; Klein, 1979; Schueler, 1994). Booth and Jackson (1997) and Wang *et al.* (2001) concluded that connected impervious surfaces were best correlated with stream health, and that biotic impairments would be seen when these reached 10% and 8-12%, respectively. Alberti *et al*. (2007) concluded biological health was most strongly correlated with roadway density and the number of stream crossings, and not a particular level of imperviousness. Other studies have found that simple measures of imperviousness were too simplistic and that site-specific variables—such as riparian canopy condition (Cianfrani, 2006; Schueler *et al*., 2009), slope, geology (Chin, 2006), and configuration of impervious surfaces (Alberti *et al*., 2007)—were important determinants of stream biotic health. A study by Crunkilton *et al.* (1997) that controlled for physical habitat degradation and investigated urban water quality found that D. magna and P. promelas showed increased mortality or growth stunting only after 7 days of exposure, suggesting chronic but not acute toxicity.

Retention Ponds

Retention ponds store runoff in a permanent pool between storms and release it a controlled rate through an outflow orifice (Hvitved-Jacobsen, 1990), or some other structure or device that limits peaks flows to protect downstream bed and bank stability (ASCE and WEF, 1992). The permanent pool allows for the settling of sediments (Urbonas, 1995) as well as biological uptake, adsorption, and ion exchange of dissolved or slowly settleable pollutants (Hvitved-Jacobsen, 1990; Urbonas, 1995). Two general theories explain pollutant reductions: the solids settling method assumes that most pollutants are removed by Type 1 settling (Hartigan, 1989) while the controlled eutrophication model treats the biological removal of phosphorus as a proxy for all pollutant reductions (Dorman *et al.*, 1996; Hartigan, 1989; NVPDC and ESI, 1992).

Larger ponds generally have greater pollutant reductions. To be effective, stormwater detention periods must be long enough to allow for removal processes to occur (Hvitved, Jacobsen, 1990; USEPA, 1999). Higher ratios of pool to inflow volume correlate with higher removals of suspended sediment (USEPA, 1983). A ratio of at least 5, or a detention time of two weeks, is needed to maximize phosphorus removal (Horner *et al.*, 1994; Hartigan, 1989). Detention time is also important for the removal of small particles—which must collide and flocculate before settling (Fitch, 1958)—and nutrients. NH_3-N + NH_4^+-N (ammoniacal-N) may sorb to clay or organic matter particles and settle (Brady and Weil, 1999; Novotny and Olem, 1994), volatilize, or oxidize to NO_3 (Brady and Weil, 1999; GC and WWE, 2012a). NO_3 must then be biologically consumed (Winer, 2000) or denitrified (GC and WWE, 2012a). Removal of larger particles can be maximized by decreasing the surface overflow rate (Fischerström, 1958; USEPA, 1993). There are rule-of-thumb reports in the literature suggesting that pond surface area should equal a certain percentage of its contributing drainage area, ranging from 1% (USEPA, 1999) up to 3-7% (Horner *et al.*, 1994), depending on the imperviousness of the catchment (Hartigan, 1989). Contrary to most findings, Barrett (2004) found that pond size has little effect on performance, and with the exception of dissolved P, the only significant predictor of a pond's outlet concentrations are its inlet concentrations.

Other design considerations include the length to width ratio, which should be large to prevent short circuiting (Thackston *et al.*, 1987; Walker, 1998). Pond depth should be deep enough to prevent re-suspension of bottom sediments (Schueler, 1987), but not so deep as to cause thermal stratification and anaerobic conditions which can cause the release of phosphorus and metals bound to sediments (Hvitved-Jacobsen, 1990; NVPDC and ESI, 1992). Vegetation within the pond will encourage biological uptake of pollutants, increase microbial activity, and protect against erosion (Schueler, 1987). An aquatic bench around the perimeter of the pond will provide the shallow littoral zone in which vegetation thrives (Virginia DEQ, 2011; Schueler, 1987; NVPDC and ESI, 1992).

Retention ponds can exacerbate the problems of thermal pollution and dissolved oxygen deficits (Galli, 1990; Schueler, 1987). Long detention times and large surface areas allow for significant solar energy absorption, which can raise water temperatures to levels that stress sensitive organisms (Roseen *et al*., 2010). Biochemical oxygen demand (BOD) within the pond, combined with the reduced oxygen saturation in heated water can lead to oxygen depression in the outflow (Galli, 1990). Standard methods for predicting inflow hydrographs may underestimate volume and flow rates of stormwater, leading to undersized ponds that do not provide assumed peak flow reductions (Hancock and Chambers, 2010). Even a retention pond in perfect working order cannot reduce the increased post-development stormwater volume (Maxted and Shaver, 1999) or restore degraded base flows (Schueler, 1987; Thomas *et al*., 2011).

3.4 SITE DESCRIPTION

The Ashby Retention Pond is located in park land in Fairfax City, VA, a suburb of the US National Capital, Washington, D.C. The 54.7 hectare watershed of Ashby Pond can be subdivided into three sub-watersheds (Figure 3.4-1), each with their own characteristic land use and inlet into the pond (City of Fairfax GIS. Accessed November, 2012, http://www.fairfaxva.gov/. Unless otherwise noted, all geographic data are from this source). Impervious surfaces for each sub-watershed represent the areal sum of roads, parking lots, driveways, rooftops and building additions such as decks and patios.

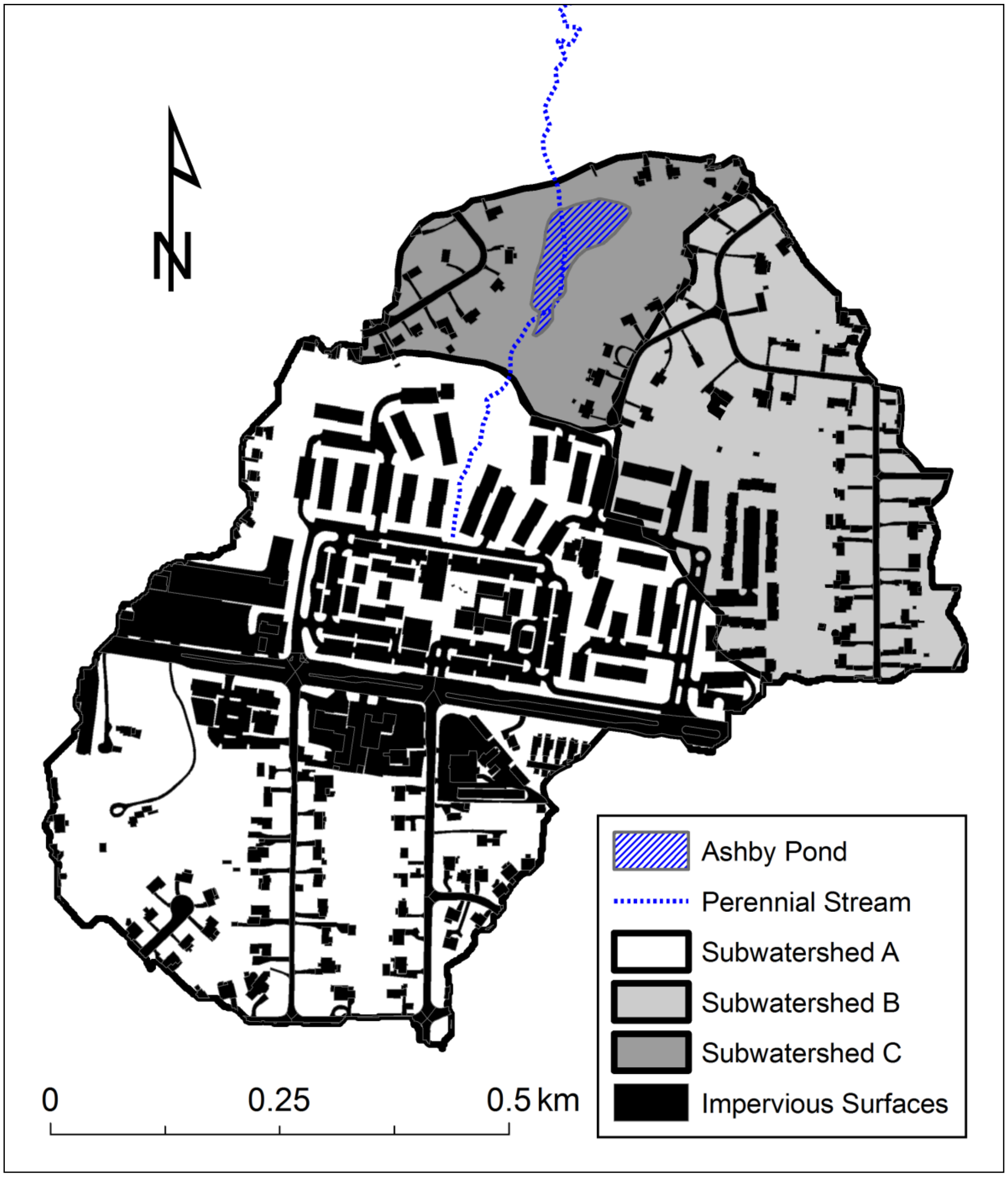
Sub-watershed A is 35.3 hectares with a total imperviousness of 42.8%. It drains to Ashby Pond through a small perennial stream that emerges from a concrete pipe approximately 750 feet upstream of the pond. The watershed is bisected east to west by State Route 236, a four lane divided highway. Highly impervious commercial and multi-family residential development borders each side of the road and dominates the northern half of the watershed. Development south of Route 236 tapers to single-family residences. The watershed is served by a traditional curb and gutter stormwater system.

Sub-watershed B is 13 hectares and 24.4% impervious. The watershed land use is approximately 80% low-density single-family residential, and 20% high density multi-family residential. The single-family residences are drained by a mix of grassed and concrete roadside swales, while the multi-family residences are served by a curb and gutter system. All stormwater enters Ashby Pond through a concrete storm sewer outfall pipe.

Sub-watershed C consists of the 6.4 hectares immediately adjacent to Ashby Pond that drains to it by overland flow. It is divided evenly between forested land and low-density, single-family residences. Total imperviousness is 13%.

Prior to beginning this study, the pond was subjected to an extensive retrofit. Dredging removed nearly 0.76 m of accumulated sediment, the bottom was re-graded to create a forebay at the sub-watershed A inlet, and a new concrete outflow structure was installed. The high flow outlet is a broad crested weir 3.79 meters in width. An 0.2m perforated PVC pipe passing through the concrete outflow structure serves as the low flow orifice. Wrapped in geotextile fabric, the pipe inlet collects water from below the surface of the permanent pool and releases it at the foot of the

Figure 3.4-1. Ashby Pond Sub-Watersheds and Imperviousness



weir. Outflows from the low flow orifice or over the weir enter a 3m wide by 15.8m long rectangular channel which discharges to Daniels Run, a tributary of Accotink Creek. Due to impairments to the benthic invertebrate community, nearly the entire length of Accotink Creek is included on Virginia's 303(d) impaired waters list, with several sections also impaired by E. coli and pollutants in fish tissue (Figure 3.4-2) (USEPA, ATTAINS Database. Accessed December 29, 2013, http://www.epa.gov/waters/ir/.) The initial Accotink TMDL approved by the EPA listed sedimentation caused by stormwater runoff as the most probable cause of benthic impairments. Total nitrogen (TN) and total phosphorus (TP) were considered possible stressors. The TMDL required a 48.4% reduction in flow volume during the 1-year, 24-hour storm (USEPA, 2011b). This requirement was struck down by a legal challenge brought by the Virginia Department of Transportation and Fairfax County (Virginia Dept. of Transp., *et al*. v. USEPA, 2013 WL 53741, *5 (E.D.Va.)), and a revised TMDL to address the excess sediment issue has yet to be produced.

3.5 STUDY DESIGN

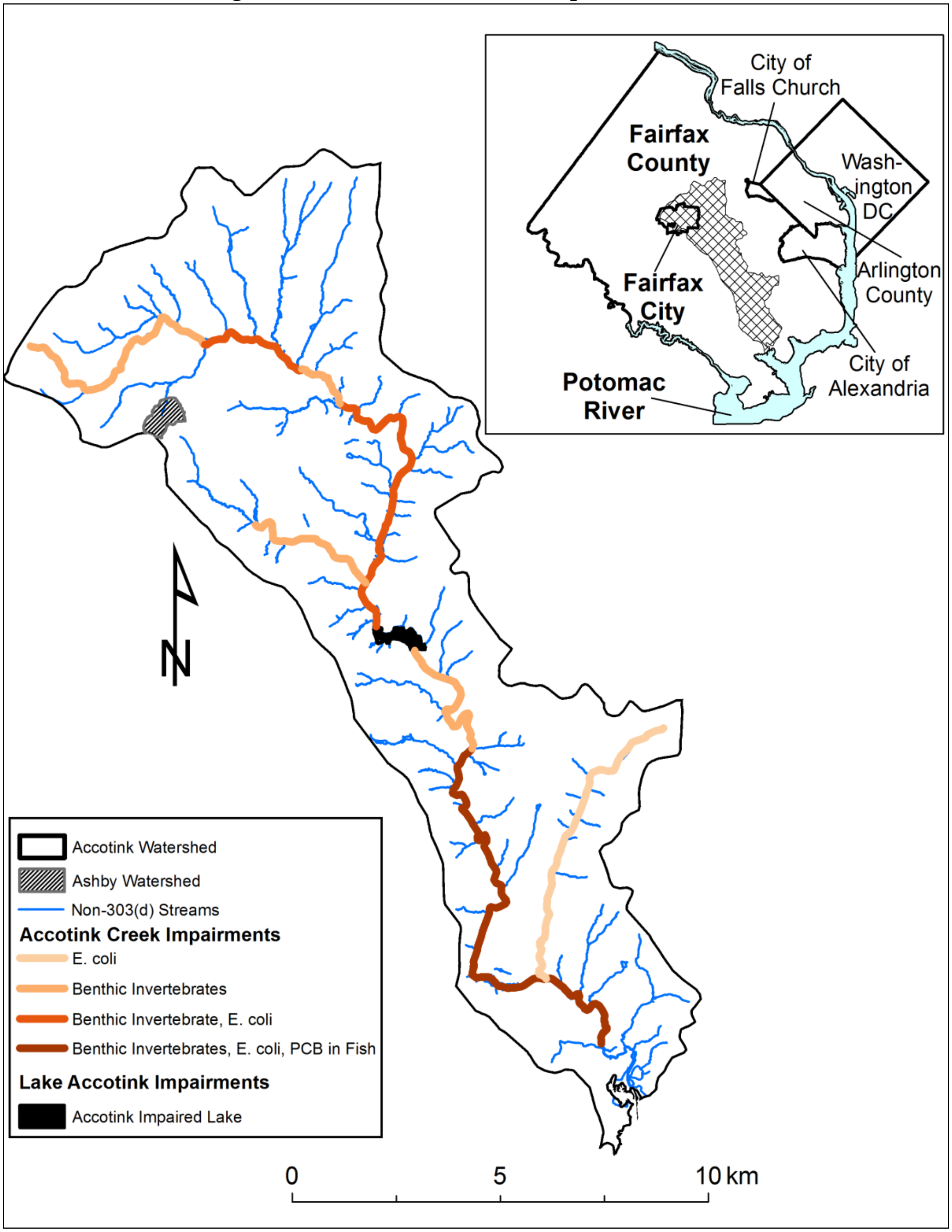
Data Collection

All monitoring devices were installed and calibrated and all water quality analyses were performed at the Occoquan Watershed Monitoring Laboratory (OWML) of the Virginia Tech Department of Civil and Environmental Engineering. Sampling and collection protocols followed generally accepted procedures (USEPA, 1992c) and were in accordance with the OWML Quality Assurance Plan (OWML, 2001) and the procedures required by its accreditation in the Virginia Environmental Laboratory Accreditation Program (Va. Admin. Code, 1VAC30-45).

Inflows were measured by an acoustic Doppler current profiler (ADCP) installed on the bed of the perennial stream draining sub-watershed A, and by a Palmer-Bowlus flume with a pressure transducer placed in the outfall of the 0.91 meter storm sewer pipe draining sub-watershed B. Sub-watershed C inflows consisted entirely of ungaged overland flow. The EPA Stormwater

Management Model, Version 5 (SWMM, USEPA, National Risk Management Research Laboratory. Accessed July 2013, http://www.epa.gov/nrmrl/wswrd/wq/models/swmm/) was used

Figure 3.4-2. Accotink Creek Impaired Reaches



to estimate the ungaged area flows. Inflow from direct rainfall onto the pond surface was also accounted for. To calibrate the Palmer-Bowlus flume rating curve, controlled flow from a fire hydrant was measured by both the flume and a handheld FlowTracker Acoustic Doppler Velocimeter (Son Tek, San Diego, California). The SWMM model was calibrated by modeling storm inflows from sub-watersheds A and B and comparing the SWMM hydrograph and total inflow volumes to those recorded by the ADCP and Palmer-Bowlus flume.

ADCP performance was problematic, possibly due to sediment accumulation caused by high stormwater flows and instability of the stream's banks. The ADCP measured stream cross sectional areas adequately, but flow readings were found to be biased high during storms and low, or even negative, during dry periods. To calibrate the ADCP, field measurements of storm flow were made using manual methods with the FlowTracker. A calibrated velocity was computed by dividing the field-measured flow by the ADCP-measured cross section. A velocity-velocity curve was plotted with the calibrated velocity on the y-axis and the ADCP-measured velocity on the x-axis. The linear relation between the two velocities is shown in Equation (15) with both y (calibrated velocity) and x (ADCP velocity) in m/s. Flow rates were recalculated using Equation (15).

$$y=0.5152x \tag{15}$$

Sub-watershed A had gaps of usable data during some storm durations, either due to the ADCP recording flows that could not be corrected by Equation (15) or more commonly failing to record any data at all. The SWMM model was used to backfill these gaps in flow data, which ranged in duration from minutes to hours.

For outflows, a pressure transducer within the permanent pool measured the head at the outlet weir. Manual ratings of low flows through the 8 inch PVC outlet pipe were calculated by capturing all flow over a set duration in a bucket and measuring the total volume. These "bucket" ratings were recorded at various pool depths and used to formulate Equation (16), correlating pipe flow (P_f) in m^3/s with the head on the pipe h_p (m). Pipe flow was very low due to clogging of the geotextile fabric with fine sediments, and most storm flow exited the pond over the weir. High flow rates over the weir (W_f) in m^3/s were calculated using the broad crested weir formula, Equation (17), using the width of the weir (3.79 m), a weir coefficient (C_v) in $m^{1/2}/s$, and the head on the weir (h_w) in m. C_v values varied with h_w and were taken from the Handbook of Hydraulics (Brater and King, 1976). Equations (16) and (17) were summed to create a rating curve for total outflows.

$$P_f=h_p(0.00060)-0.00040 \tag{16}$$

$$W_f= C_v(3.79)(h_w)^{1.5} \tag{17}$$

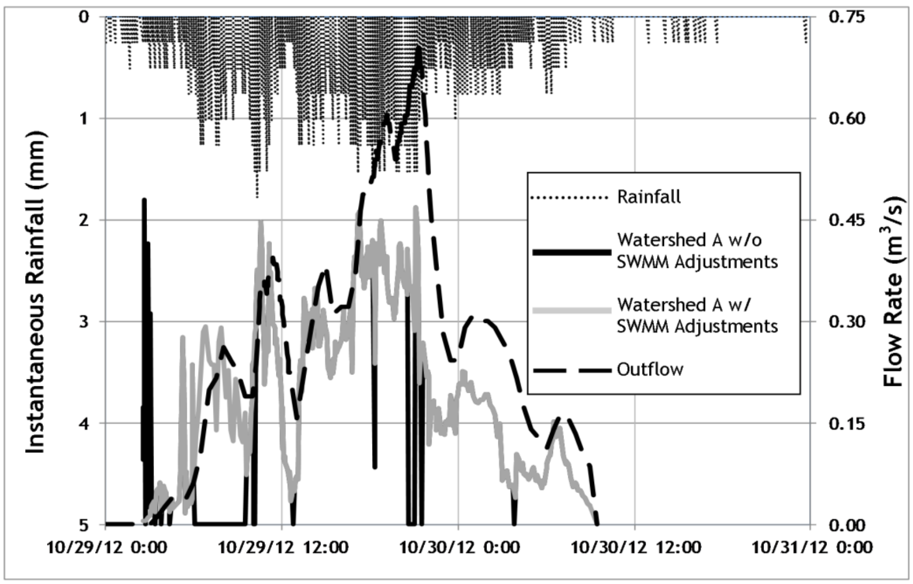
To calculate the volume of stormwater inflow that remained in the pond as storage after the duration of each storm, the difference in pond depth between the beginning and end of the storm, as recorded by the outlet pressure transducer, was multiplied by the surface area of the pond. Storage could be positive or negative and was of great enough magnitude to significantly alter flow balances of some smaller storms (i.e. 2012-09-26).

At each of the two gaged inlets and the outlet, a Sutron recorder housed in an on-site storage unit and programmed with the calibrated flow equations recorded incremental and total inflows and outflows for every storm. A Nalgene jug inside each storage unit collected water quality samples. The jugs were connected to the pond via plastic tubes installed just downstream of the inlets and outlet. When increased flow caused by a storm was detected, the Sutrons triggered water quality samples to be taken automatically at equal volume intervals (i.e., one sample for every $10m^3$ of flow). Samples were siphoned from the stream by a vacuum pump in the throat of each previously cleaned Nalgene and composited together until the storm flow ceased. Since each sub-sample of the composite represented an equal volume of storm flow, the pollutant concentrations of the composite sample were analogous to the event mean concentration (EMC) of the storm. The composite samples were analyzed at the OWML for concentrations, of nutrients (TP, orthophosphate (OP), TN, nitrate and nitrite (OX-N) and ammoniacal-N); sediment (total suspended sediment (TSS) and suspended sediment concentration(SSC)); organics (total organic carbon (TOC), dissolved organic carbon (DOC) and chemical oxygen demand(COD)); and soluble salts (sodium (Na^+), chloride (Cl^-), and calcium (Ca).

EMCs for the ungaged sub-watershed C, which contributed 2% or less of the total inflow volume for each storm, were estimated using the composite samples from sub-watershed B. The sub-watershed was divided into forested land and low-density single-family residential land and SWMM was used to estimate runoff from each. SWMM estimates showed that all storm flow from sub-watershed C originated from the residential area; the forested area contributed no flow. Comparison of all of sub-watershed B to the residential portion of sub-watershed C showed that the dominant land use (single family homes), soil types and slopes (USDA-NRCS Soil Survey of City of Fairfax, VA. Accessed August 2013, http://websoilsurvey.sc.egov.usda.gov/) were the same, and the impervious percentage (24.4% for sub-watershed B, 28.8% for the developed portion of sub-watershed C) was almost identical. While general land use categories are of little value in predicting stormwater EMCs (USEPA, 1983), site-specific factors such as soils, slopes, the intensity of land use, and materials usage within the watershed can be predictive (Sonzogni *et al.*, 1980). Given that all known site-specific factors are nearly identical, the EMCs of sub-watershed C were assumed to be equal to sub-watershed B.

The concentrations of TN, OX-N, ammoniacal-N, Cl, and Ca in direct rainfall were estimated using monthly average data from the National Atmospheric Deposition Program's National Trend Network study site in Fairfax County, VA (NADP-NTN, Station VA10, Mason Neck, VA. Accessed April 2013, http://nadp.isws.illinois.edu/NADP/networks.aspx). Rainfall concentrations of TOC and DOC were estimated from Willey *et al*. (2000). All other constituents were assumed to be negligible.

Figure 3.5-1. Hurricane Sandy Hydrograph with Unadjusted and SWMM Adjusted Inflows



A volume balance between inflows and outflows of ±10% was established as the criterion for storm inclusion in the performance analysis. Using the flow measurement methods previously described, all but three storms fell within this range. For the three non-conforming storms (2012-09-08, 2012-09-18, 2012-10-29), inflows for each sub-watershed were estimated using the SWMM model. Comparisons of SWMM to Sutron-recorded inflow data for the September 8th and 18th storms showed excellent matches of hydrograph morphology, and total inflow volumes differed by only .2% and 4.1%, respectively. Any error in the observed flow balances was therefore assumed to be with the outflow. The outflow volumes for both storms were set equal to the Sutron inflows. For the October 29th storm (Hurricane Sandy), the SWMM and Sutron inflow hydrograph morphologies matched well, but neither was within ±10% of the outflow volume. Several hours of inflow data were also missing for the same storm. SWMM was used to estimate the missing flows in the Sutron hydrograph, and this was assumed to be the best estimate of inflow (Figure 3.5-1). The error in the observed balance was assigned to the outflow, and outflow volume was set equal to the SWMM-adjusted inflow. A summary of all monitored storms, the constituent data measured for each, and the applicable flow volume adjustments are shown in Table 3.5-1.

Performance Analysis

Performance analyses were conducted on both mass and concentration bases. The flow-weighted inflow EMCs of all constituents were calculated for each storm. They and the outflow EMCs were multiplied by total inflow and outflow volumes to yield per storm and average inflow and outflow loads (kg). Constituent reductions were measured on a mass basis by the summation of loads method (SOL), Equation (18), and on a concentration basis by the efficiency ratio (ER) method, Equation (19) (GC and WWE, 2012a; Strecker, 1999; Strecker *et al.*, 2001). For censored observations, the SOL and ER were calculated twice, once assuming that all below-detection-limit concentrations were equal to 0 and again assuming they were equal to the detection limit. This showed the possible range of reductions without relying on substitution, which introduces bias into the analysis (GC and WWE, 2009; Helsel, 2012).

$$SOL=[1-(\text{sum of outlet loads}/\text{sum of inlet loads})]x100 \qquad (18)$$

$$ER=[1-(\text{avg. outlet EMC}/\text{avg. inlet EMC})]x100 \qquad (19)$$

For constituents with no censored values (TP, OP, TN, OX-N, TOC, DOC, COD, TSS, SSC), individual storm loads and EMCs were log transformed. The data and logs of data were ranked,

Table 3.5-1. Constituent Data and Applied Flow Modifications for All Monitored Storms

Storm Date	Missing Constituent Data[1]	Flow Modifications/Comments	Final Inflow Volume	Final Outflow Volume[2]
2012				
09-08	-	Inflows and outflow not within ±10%. SWMM estimates total inflow is 99% of recorded inflow and hydrographs match well. Assume error is with outflow and set outflow volume equal to inflow.	2,456	2,946
09-18	COD	Inflows and outflow not within ±10%. SWMM estimates total inflow is 96% of recorded inflow and hydrographs match well. Assume error is with outflow and set outflow volume equal to inflow.	2,469	1,796
09-26	COD	Good balance between inflows and outflow. Almost all inflow is stored in pond; little outflow.	568[3]	17[3]
10-02	COD	Fairly good balance between inflows and outflow. Used SWMM to backfill some data gaps from subwatershed A.	4,114	3,960
10-18	-	Good balance between inflows and outflow.	813	634
10-29	-	Inflows and outflow not within ±10%. SWMM total inflow is 88% of recorded inflow and hydrographs match well. Assume error is with outflow. Used SWMM to plug some data gaps in subwatershed A hydrograph and set the outflow equal to the SWMM-adjusted inflows.	24,426[3]	24,003[3]
12-20	OP	Fairly good balance between inflows and outflow. Used SWMM to backfill some subwatershed A inflow data gaps.	3,799	3,353
2013				
01-15	-	Good balance between inflows and outflow.	2,071	1,931
01-30	-	Fairly good balance between inflows and outflow. Used SWMM to plug some data gaps from subwatershed A.	8,444	9,566
02-26	-	Used SWMM to model second half of storm inflow from subwatershed A due to ADCP malfunction. SWMM adjusted inflows match outflow well.	3,105	2,931
03-12	TP, COD, Na^+, Cl^-, Ca	Good balance between inflows and outflow.	4,131	4,658

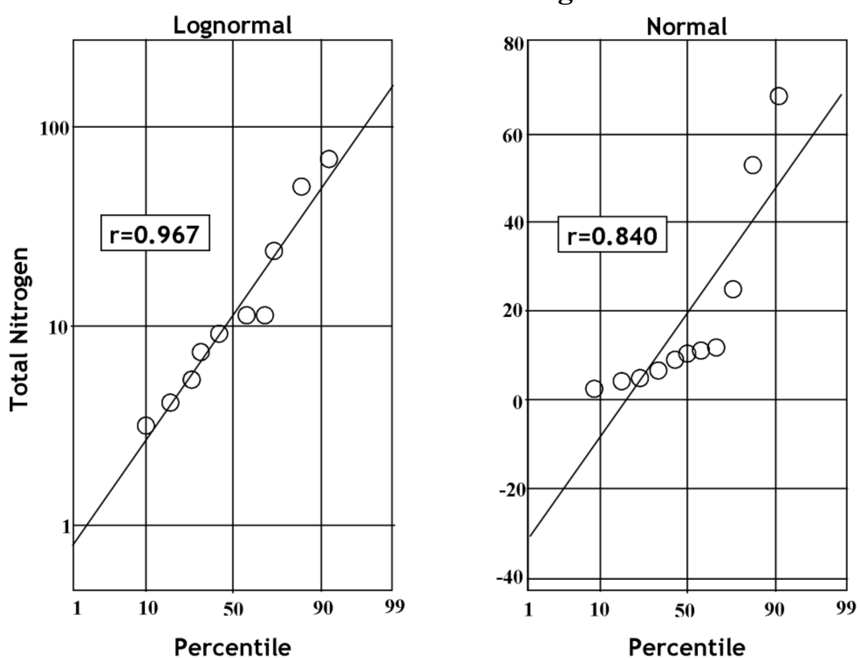
[1]Analyzed constituents include: TP, OP, TN, OX-N, Ammoniacal-N, TSS, SSC, TOC, DOC, COD, Na+, Cl- and Ca

[2]Some inflow and outflows seem unbalanced due to the effects of pond storage

[3]Considered extreme events: 2012-09-26 because nearly all inflow was captured as pond storage, 2012-10-29 because of the magnitude of inflows and outflows.

assigned percentiles using the Cunnane plotting position (Cunnane, 1978; Helsel and Hirsch, 2002), and the percentiles were converted to standard normal scores. Probability plots were created to visually test adherence of the data to either the lognormal or normal distribution. The closer the data are to a straight line, the better the adherence (Helsel and Hirsch, 2002). To provide a quantifiable measure of the adherence, Pearson's correlation coefficients (r) between the data and the standard normal scores were calculated using Minitab 16 statistical software (Minitab, Inc., State College, Pennsylvania). The probability plots and r values for TN inflow are shown in Figure 3.5-2.

Figure 3.5-2. Adherence of TN Inflow Loads to the Lognormal and Normal Distributions



The r values for all constituents, ranging between 0.891 (DOC outflow) and 0.977 (OX-N inflow) for loads and 0.916 (OP outflow) and 0.989 (OP inflow) for concentrations, justified the assumption of lognormality for the purposes of calculating median values. Medians of the uncensored constituents were calculated using the bias-corrected geometric mean (BCGM), a less biased estimate for lognormal data than the sample median or geometric mean (Parkin and Robinson, 1993). All other data (ammoniacal-N, Na^+, Cl^-, Ca) were interval censored with a non-zero lower bound (i.e. inflow ammoniacal-N loads for 2012-09-18 storm are 0.02–0.04 kg). Median estimates were made in Minitab using the Turnbull estimator, a survival analysis method (Helsel, 2012).

Because of the interval censored nature of some of the data, many standard statistical hypothesis tests could not be used because the difference between paired inflow and outflow loads is an interval itself (Helsel, 2012). The modified sign test (Fong, 2003) works for such data if there is one reporting limit per inflow-outflow pair (Helsel, 2012). The few data pairs that did not meet this standard were re-censored at the highest reporting limit (Helsel, 2012), and the modified sign test was run to determine if differences between inflow and outflow loads were statistically significant at a p-value≤0.05.

3.6 RESULTS AND DISCUSSION

The median inflow and outflow loads and flow weighted concentrations of Ashby Pond are compared to previous multi-pond studies in Table 3.6-1. Ashby's median influent concentrations of TP and TSS were higher; OP and OX-N were lower; and TN was the same as wet pond medians from the International BMP Database (GC and WWE, 2012a, b). Effluent median concentrations were higher for TP and TSS and lower for OP, TN and OX-N compared to the BMP Database (GC and WWE, 2012a, b) and the National Pollutant Removal Performance Database (Winer, 2000). State water quality standards do not exist for most of the measured constituents, but for those that do (ammoniacal-N, $N0_3$), Ashby effluent median concentrations were below limits (Va. Admin. Code, 9VAC25-260-140 *et seq*).

The pollutant removal performance of Ashby Pond, computed on mass and concentration bases and compared to the same multi-pond studies as well as the reductions credited by Virginia to ponds built to state design standards, is shown in Table 3.6-2. SOL and ER performance were calculated twice for Ashby: once for all storms and once with extreme events removed (2012-10-29 and 2012-09-26). Hurricane Sandy (2012-10-29) accounted for more than one third of the inflow mass load of OP, OX-N,TSS, COD, DOC and TOC; and more than one third of the outflow mass load of TP, OP, TN, OX-N, COD, DOC, TOC and Ca. The 2012-09-26 event was a small storm whose inflow was almost entirely captured by the pond: only 17m^3 of outflow was recorded. Ashby's performance with these storms excluded is likely more indicative of pollutant removal for the majority of storms.

Removing the extreme events had varied effects on performance. Mass removals are often dominated by a small number of large storms (GC and WWE, 2009), such as Hurricane Sandy. Without the extreme events, SOL reduction more than doubled for OP and nearly doubled for OX-N because of the exclusion of poor mass removals during Sandy. The small 2012-09-26 storm skewed mass removals slightly higher because the tiny outflow volume carried a load much smaller than the storm inflow, even though concentrations were not that different. Removing this storm caused a small decrease in most constituent SOL efficiency estimates. With both extreme events removed, mass removal of ammoniacal-N reduced enough to become statistically insignificant. For Na^+, Cl^- and Ca, the extreme events were the only storms that reduced loads; all other storms featured export or balance. Exclusion of the extreme events caused the mass export of each to increase to statistical significance.

For concentrations, ER values were not greatly affected by the removal of the extreme events since they rely on reduction in the average EMC and thus weight all storms the same, regardless of size (GC and WWE, 2009). An exception was ammoniacal-N. Removing the extreme events caused ammoniacal-N reduction to fall only a few percentage points, but it was enough to lose statistical significance.

Compared to Virginia's credited reductions for wet ponds, Ashby met or exceeded TN performance on a mass and concentration basis with or without the extreme event storms. The pond underperformed with respect to the TP reduction credits, with the exception of mass removal with the extreme events excluded. However, Ashby Pond is under-sized compared to

Table 3.6-1. Ashby Influent and Effluent Medians Compared to Prior Studies

CONSTITUENT	MEDIANS						
	ASHBY POND				GEOSYNTEC (2012a, b)[1]		WINER (2000)
	Influent Mass	Effluent Mass	Influent Conc.	Effluent Conc.	Influent Conc.	Effluent Conc.	Effluent Conc.
TP	0.67	0.27	0.23	0.15	0.14	0.08	0.11
OP	0.19	0.04	0.06	0.02	0.10	0.04	0.03
TN	5.3	2.3	1.8	1.2	1.8	1.3	1.30
OX-N	1.2	0.28	0.40	0.14	0.43	0.18	0.26
Ammoniacal-N	0.30	0.07	0.13	0.02	-	-	-
TSS	281	58	95	29	78	13	17.00
SSC	253	53	84	27	-	-	-
TOC	23	15	7.6	7.5	-	-	-
DOC	21	14	7.1	6.8	-	-	-
COD	129	107	36	31	-	-	-
Na^+	7.9	15	2.8	5.4	-	-	-
Cl^-	<2.63	31	<0.08	9.7	-	-	-
Ca	8.3	12	3.7	6.5	-	-	-

[1]TSS and TP data from ponds within or near Chesapeake watershed; all others from international dataset

the Virginia design specifications, so a direct comparison to state reduction credits may not be a fair evaluation. Ashby Pond holds 5,370m^3 of stormwater. Virginia has two separate design standards for wet ponds: A Level 1 pond serving Ashby's drainage area would have a treatment volume of 6,165m^3; a Level 2 pond would have 9,250m^3 of storage and provide aeration. For mass and concentration, Level 1 ponds receive the reduction credits listed in Table 3.6-2 while

Level 2 ponds are credited for a 75% reduction of TP and 40% reduction of TN (Virginia DEQ, 2011). Given the size difference, it may be more telling to compare Ashby's performance to the reductions credited by the Chesapeake Bay Program to infrastructure built before the implementation of current standards. Performance of these older ponds is calculated by discounting published removal rates to account for the age and variable maintenance of existing wet ponds (CSN, 2011). These older ponds are assumed to reduce 45% of TP, 20% of TN and

Table 3.6-2. Ashby Performance Compared to Prior Studies and State Reduction Credits

CONSTITUENT	ASHBY POND				GC & WWE (2012a, b)[2]	WINER (2000)[3]	NURP (1983)[4]		VIRGINIA REDUCTION CREDIT (2011)[5]	
	All Storms		Outliers Removed[1]							
	Mass	Conc.	Mass	Conc.	Conc.	-	Mass	Conc.	Mass	Conc.
TP	41*	41*	51*	39*	43	49	34	37	50	50
OP	30*	62*	61*	68*	54	70[7]	29[7]	38[7]	-	-
TN	30*	38*	38*	35*	30	32	-	-	30	30
OX-N	30*	62*	54*	64*	58	62	-	-	-	-
Ammoniacal-N	53*	60*	49 - 50	57 - 58	-	-	-	-	-	-
TSS	76*	68*	64*	66*	83	80	60	38	-	-
SSC	64*	68*	66*	68*	-	-	-	-	-	-
TOC	9	3	(3)	(4)	-	-	-	-	-	-
DOC	12	6	0	(2)	-	-	-	-	-	-
COD	11	13	2	13	-	-	15	15	-	-
Na^+	(57) - (56)	(91) - (87)*	(68) - (62)*	(93) - (88)*	-	-	-	-	-	-
Cl^-	(45) - (43)	(91) - (68)*	(64) - (55)*	(91) - (69)*	-	-	-	-	-	-
Ca	(18) - (16)	(55) - (51)*	(53) - (48)*	(60) - (55)*	-	-	-	-	-	-

1. Outliers are storms that were exceptionally large (Hurricane Sandy, 2012-10-29) or had almost no outflow volume (2012-09-26).
2. Calculated from median inflow & outflow EMCs of retention ponds. TSS and TP data from ponds within or near Chesapeake.
3. The average of many individual retention pond studies that used a variety of performance calculations.
4. The median SOL and ER of all studied retention ponds.
5. Assumed performance of retention ponds built to Virginia DCR (2011) Level 1 design specifications.
6. Zn reductions based on small (4 storms) and highly censored data set
7. Reduction of soluble phosphorous.

Notes:
*-Statistically significant Ashby data, p-values≤0.05
-Numbers in parentheses represent negative reductions, Signifying an increase from inflow to outflow.
-SOL and ER for constituents with interval censored data will often be a range of possible reductions.

60% of TSS on a mass basis (CSN, 2011). Ashby exceeded these reductions except for TP removal with the extreme events included. This underperformance was likely due to the skewing caused by Hurricane Sandy, which accounted for 43% of the total outflow load of TP for the study. Several weeks of detention is needed to maximize TP removal (Horner *et al.*, 1994;

Hartigan, 1989). The Hurricane Sandy inflow was 4.5 times larger than the total storage of Ashby Pond, and detention would have been less than one half day.

Probability plots of inflow and outflow loads and EMCs of TP, TN, TSS, SSC, TOC and DOC are found in Figure 3.6-1.

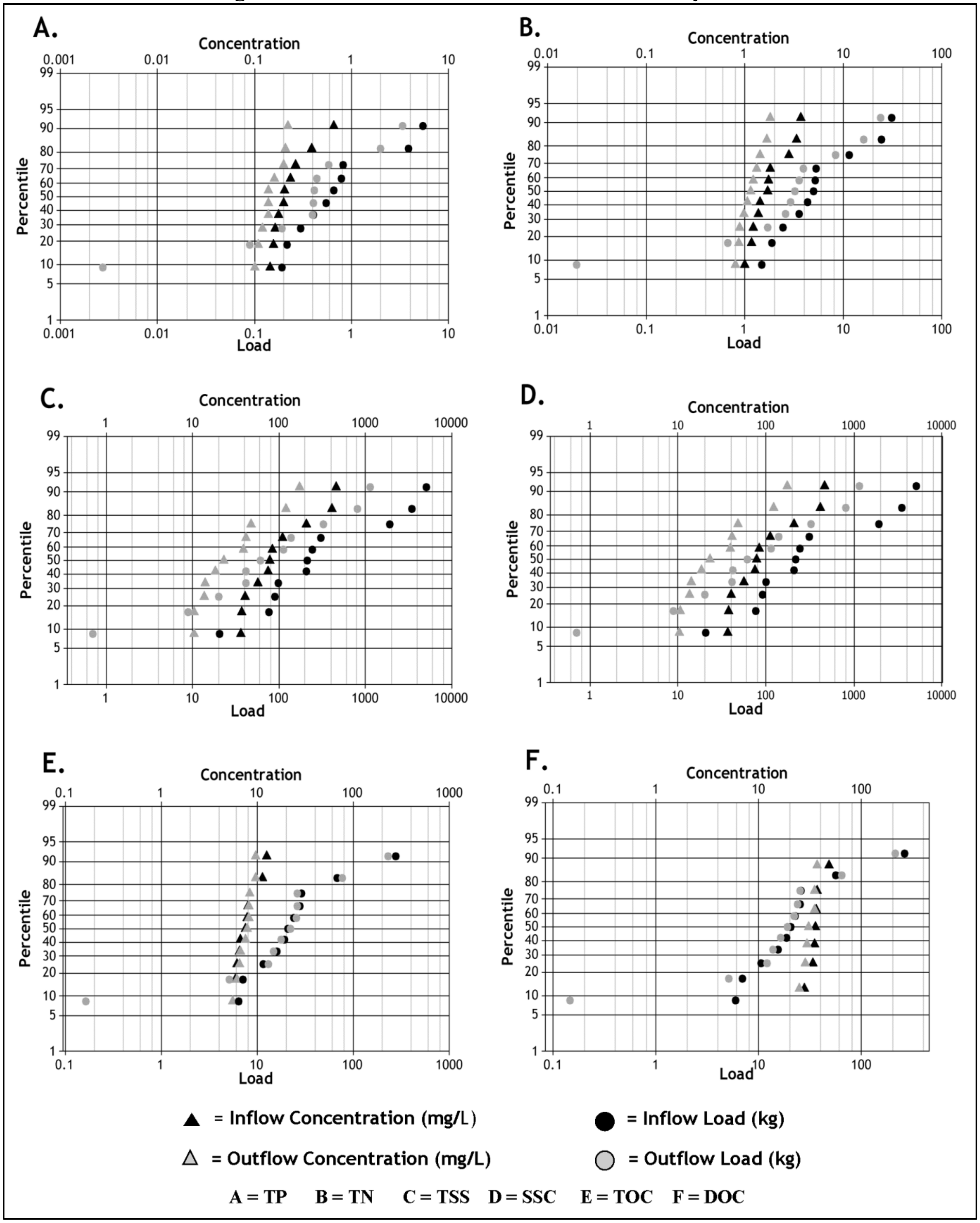
Analysis of the physical habitat of Daniels Run downstream of Ashby was not performed, but any improvements were likely marginal. Ashby was designed for peak flow attenuation by providing extended detention storage in the 0.33m separating the low flow outlet from the crest of the weir. Storage volume in this zone is equal to 43% of the average inflow of all storms, 66% if Hurricane Sandy is excluded. The low flow pipe was to slowly release this stored water over a period of hours. Clogging of the pipe brought the pool surface even with the crest of the weir, eliminated detention, and negated peak shaving.

Even fully functioning wet ponds do not provide significant stormwater volume reductions, and they will likely need to be used in concert with Low Impact Development (LID) practices that can in order to maximize the protection of downstream physical habitat (Maxted and Shaver, 1999). Virginia's new stormwater management laws recognize this by recommending that wet ponds only be used as the last in a sequence of BMPs if upstream LID practices cannot reduce stormwater peak flows and pollutant loads to the required level (Virginia DEQ, 2011).

3.7 CONCLUSIONS

Ashby Pond performance was measured on mass and concentration bases for all storms between September 2012 and March 2013, and again with the extreme event storms of 2012-9-26 and 2012-10-29 removed. For nutrients (forms of N and P) and sediment (TSS and SSC), Ashby Pond provided substantial mass and concentration reductions. All reductions were statistically significant except for ammoniacal-N with the extreme event storms excluded. Reductions of organics and oxygen demand (TOC, DOC, COD) were small or negative and not statistically

Figure 3.6-1. Inflow and Outflow Probability Plots



significant with or without the extreme events. Soluble salts (Na^+, Cl^-, Ca) were exported in all. Cases. Export on a mass basis was statistically significant only when the extreme event storms were removed. On a concentration basis, export was significant with or without the extreme events

Given that N, P and sediment are the primary pollutants of concern in Accotink Creek and the Chesapeake Bay, wet ponds appear to be a useful method of stormwater quality control in the Chesapeake Bay Watershed. However, wet ponds do not significantly reduce stormwater volumes, and because of its clogged low-flow pipe, Ashby Pond provided little peak flow attenuation. Since much of the pollutant load in urban streams is due to in-stream erosion of beds and banks caused by high stormwater flows, wet ponds will likely be more effective if they provide significant extended detention storage volume above their permanent pool, and most effective when used in combination with LID practices that reduce stormwater volumes, such as bioretention.

4. ENGINEERING SIGNIFICANCE

It is clear that stormwater retention ponds play a valuable role in the control of urban nonpoint source pollution. Unlike dry ponds that rely nearly exclusively on sedimentation, the long detention times and developed communities of algae, bacteria and macrophytes allow retention ponds to effectively remove dissolved pollutants. Ashby pond provided statistically significant reductions of TP, OP, TN, OXN, ammoniacal-N (with the extreme event storms included), TSS and SSC. The Chesapeake Bay TMDL requires that watershed states reduce Bay annual loadings of nitrogen, phosphorus and sediment. If retention ponds perform on average as well as Ashby, they could be a standard practice to achieve these goals. However, the other TMDL applicable to Ashby Pond—Accotink Creek—poses a tougher challenge.

The primary impairment of the benthic invertebrate community in Accotink Creek is likely excess suspended sediment caused by flashy stormwater flows scouring out the stream bed and banks. The design goals of Ashby Pond are to retain the first inch of runoff, trap sediment, and reduce nitrogen and phosphorous. The pond was not primarily designed to protect the stability of the downstream channel and benthic habitat. However, had the Ashby Pond low flow pipe not clogged, the pond could have provided extended detention and peak flow reduction that would have reduced flashiness, but it has been shown extensively in the literature that retention ponds do little to protect urbanized streams from damaging changes in morphology. This could be because wet ponds do little to address the hydrologic changes brought by urbanization. As porous soil is replaced by impervious paving and rooftop, more precipitation is converted into surface runoff and less percolates down to the groundwater table. This creates much higher storm flows and degraded base flows. Retention ponds release stored stormwater over an extended period of time, and have the ability to reduce peak flows. However, wet ponds do not convert urban runoff back into groundwater recharge, and for that reason cannot recreate the hydrology of a catchment in the pre-development state. At best, the so-called peak shaving function can partially mitigate some of the damaging characteristics of the new flow regime.

Retention ponds may also exacerbate some of the water quality issues caused by urbanization. Because of the long residence times that are crucial for dissolved pollutant removal, retention ponds expose their stored waters to the direct rays of the sun. The solar irradiance can exacerbate the thermal water pollution that is already common in developed areas due to the urban heat island effect and a lack of riparian shading. Retention ponds may be a poor BMP choice for watersheds feeding streams that are home to thermally sensitive species.

Oxygen depression is also a water quality concern in urban areas. The saturation concentration of dissolved oxygen decreases as water temperature increases, and the warm outflows from retention ponds can exacerbate this problem. While DO concentrations in lotic waters can quickly rebound due to naturally turbulent flow mixing in atmospheric oxygen, less turbulent lentic waters may not have this resilience. In addition to temperature, the oxygen demand from biochemical processes within the pond can also reduce DO. Ashby Pond showed little if any ability to reduce TOC, DOC or COD and in some cases showed increases in the outflow. While this may have little effect on DO within the naturally turbulent waters of a piedmont stream like Accotink, the oxygen demanding materials will eventually flow into the more quiescent tidal waters of the Potomac and Chesapeake Bay, where seasonal DO-depleted dead zones are a considerable problem.

Urbanization of watersheds causes a variety of deleterious effects that combine to reduce the habitat and species health of receiving water bodies. Some of these damages are caused by water quality concerns, while others are caused by hydrologic changes. On the water quality side, retention ponds are well suited to reduce nutrient and sediment loadings, as Ashby Pond did for all measured forms of nitrogen, phosphorus and suspended sediment, per its design goals. Since many pollutants are often attached to particulates, reductions in suspended sediments can correlate with reductions in other pollutants, including toxics like heavy metals. Retention ponds can often exacerbate oxygen depression and thermal pollution. Ashby provided almost no reductions in organic carbon or oxygen demand. On the hydrology side, retention ponds can at best provide some peak flow shaving, but cannot recreate the pre-urbanized hydrology. Due to its clogged low flow pipe, Ashby pond did neither.

If urban streams can only be restored by both improvements to water quality and reversion to a more natural flow regime, retention ponds can only be a partial solution. To best protect aquatic ecosystems—in Accotink Creek, the Chesapeake Bay and elsewhere—retention ponds will most likely have to be used in combination with Low Impact Development (LID) methods that mimic the pre-development hydrology, as well as concerted efforts to preserve or restore the natural lands whose hydrology LID emulates. LID facilities and the porous soils of natural environments do what retention ponds cannot: control runoff volume by allowing it to soak into the ground. The water percolates through the soil where it is cooled and filtered. The volume not claimed by absorption or plant uptake recharges the groundwater table which recharges stream base flow. The Commonwealth of Virginia recommends that retention ponds be used as the last facility in a train of treatment that incorporates upstream LID practices and land preservation (Virginia DEQ, 2011). This use of retention ponds—as a backstop to provide the needed pollutant and peak flow controls after reasonable efforts at runoff reduction—is likely the most attractive use of retention ponds going forward. For existing infrastructure like Ashby Pond, retrofitting LID techniques into the tributary watershed can provide the benefits of water quality and quantity control. If that is not feasible, spending the time, effort and funds needed for general maintenance, particularly periodic dredging to restore the pond's storage volume, will ensure continued and significant nitrogen, phosphorus and sediment reductions into the future.

REFERENCES

Alberti, M., D. Booth, K. Hill, B. Coburn, C. Avolio, S. Coe and D. Spirandelli, 2007. The impact of urban patterns on aquatic ecosystems: An empirical analysis in Puget lowland sub-basins. *Landscape and Urban Planning* 80:345-361.

Allmendinger, N. E., J. E. Pizzuto, G. E. Moglen and M. Lewicki, 2007. A Sediment Budget for an Urbanizing Watershed, 1951-1996, Montgomery County, Maryland, U.S.A. *Journal of the American Water Resources Association* (JAWRA) 43:1483-1498.

American Society of Civil Engineers and Water Environment Federtion (ASCE and WEF), 1992. Design and Construction of Urban Stormwater Management Systems. American Society of Civil Engineers, New York, New York, ISBN: 9780872628557

Barrett, M. E., 2004. Retention Pond Performance: Examples from the International Stormwater BMP Database. In: *Critical Transitions in Water and Environmental Resources Management: Proceedings of the World Water and Environmental Resources Congress,* Gerald Sehlke *et al.* (Editor). American Society of Civil Engineers, Reston, Virginia, pp.1-10. doi: http://dx.doi.org/10.1061/9780784407370.

Batroney, T., B. Wadzuk and R. Traver, 2010. Parking Deck's First Flush. *Journal of Hydrologic Engineering* 15(2):123-128.

Booth, D.B. and C.J Jackson, 1997. Urbanization of Aquatic Systems - Degradation Thresholds, Stormwater Detention, and the Limits of Mitigation. *Journal of the American Water Resources Association* (JAWRA) 33(5): 1077-1090. doi: 10.1111/j.1752-1688.1997.tb04126.x

Booth, D. B., D. Hartley and R. Jackson, 2002. Forest Cover, Impervious-Surface Area, and the Mitigation of Stormwater Impacts. *Journal of the American Water Resources Association* (JAWRA) 38(3): 835-845. doi: 10.1111/j.1752-1688.2002.tb01000.x

Brady, N. and R. Weil, 1999. The Nature and Property of Soils: 12th Edition. Prentice Hall, Upper Saddle River, New Jersey, ISBN: 0-13-852444-0.

Brater, E.F. and H.W. King, 1976. Handbook of Hydraulics for the Solution of Hydraulic Engineering Problems, 6th Edition. McGraw-Hill, New York, New York, ISBN: 0-07-007243-4.

Bryan, E. H., 1972. Quality of Stormwater Drainage From Urban Land. *Journal of the American Water Resources Association* (JAWRA) 8(3): 578-588. doi: 10.1111/j.1752-1688.1972.tb05180.x

Camponelli, K.M., S.M. Lev, J.W. Snodgrass, E.R., Landa, and R.E. Casey, 2010. Chemical Fractionation of Cu and Zn in Stormwater, Roadway Dust and Stormwater Pond Sediments. *Environmental Pollution* 158, 2143 - 2139. doi: 10.1016.j.envpol.2010.02.024

Carey, R. O., G. J. Hochmuth, C. J. Martinez, T. H. Boyer, M. D. Dukes, G. S. Toor and J. L. Cisar, 2013. Evaluating nutrient impacts in urban watersheds: Challenges and research opportunities. *Environmental Pollution* 173:138-149.

Casey, R.E, A.N. Shaw, L.R. Massal and J.W. Snodgrass, 2005. Multimedia Evaluation of Trace Metal Distribution Within Stormwater Retention Ponds in Suburban Maryland, USA. *Bulletin of Environmental Contamination and Toxicology* 74(2): 273-280. doi: 10.1007/s00128-004-0580-0

Casey, T.J., 1997. Unit Treatment Processes in Water and Wastewater Engineering. John Wiley and Sons, Inc., West Sussex, England, ISBN: 0471966932.

Cianfrani, C. M., W. C. Hession and D. M. Rizzo, 2006. Watershed Imperviousness Impacts on Stream Channel Condition in Southeastern Pennsylvania. *Journal of the American Water Resources Association* (JAWRA) 42(4):941-956.

Characklis, G.W. and M.R. Wiesner, 1997. Particles, Metals, and Water Quality in Runoff from Large Urban Watershed. *Journal of Environmental Engineering* 123(8): 753-759. doi: 10.1061/(ASCE)0733-9372(1997)123:8(753)

Chesapeake Stormwater Network (CSN), 2011. CSN Technical Bulletin No. 9: Nutrient Accounting Methods to Document Local Stormwater Load Reductions in the Chesapeake Bay Watershed. http://chesapeakestormwater.net/, *accessed* January, 2014.

Chin, A., 2006. Urban Transformation of River Landscapes in a Global Context. *Geomorphology* 79(4):460-487.

Crunkilton, R., J. Kleist, J. Ramcheck, and W. DeVita, 1997. Assessment of the Response of Organisms to Long-Term In Situ Exposures of Urban Runoff. In: *Effects of Watershed Development and Management of Aquatic Ecosystems,* Larry A. Roesner (Editor). American Society of Civil Engineers, New York, New York, ISBN: 9780784402320.

Cunnane, C., 1978. Unbiased Plotting Positions — A Review. *Journal of Hydrology,* 37(3–4): 205-222. doi: http://dx.doi.org/10.1016/0022-1694(78)90017-3

Curtis, Shannon, 2012. Personal communication.

Davis, B.S. and G.F. Birch, 2009. Catchment-Wide Assessment of the Cost-Effectiveness of Stormwater Remediation Measures in Urban Areas. *Environmental Science and Policy* 12(1): 84-91.

Dietz, M. E., and J.C. Clausen, 2008. Stormwater Runoff and Export Changes With Development in a Traditional and Low Impact Subdivision. *Journal of Environmental Management* 87(4): 560-566.

DiToro, D.M., 1991. Technical Basis for Establishing Different Sediment Quality Criteria for Nonionic Organic Chemicals Using Equilibrium Partitioning. *Environmetnal Toxicology and Chemistry* 10(12): 1541-1583.

Dorman, M.E., J.P. Hartigan, R.F. Steg, and T.F. Quasebarth, 1996. Retention, Detention and Overland Flow for Pollutant Removal From Highway Stormwater Runoff, Vol. 1: Research Report. Federal Highway Administration Report FHWA-RD-96-095, 166p. ftp://ftp.odot.state.or.us/techserv/Geo-Environmental/Environmental/Procedural%20Manuals/Water%20Quality/Sormwater%20Reference%20Materials/Dorman_etal_Vol1_1996.pdf, *accessed* October 2012

Droste, R., 1997. Theory and Practice of Water and Wastewater Treatment. John Wiley and Sons, Inc., New York, New York.

Ellis, J. B., R. Hamilton and A.H. Roberts, 1982. Sedimentary Characteristics of Suspensions in London Stormwater. *Sedimentary Geology* 33(2): 147-154.

Fair, G.M., and J.C. Geyer, 1954. Water Supply and Waste Water Disposal. John Wiley and Sons, New York, New York.

Fischerström, N. C. H., 1958. Overflow Rate Concepts. *Sewage and Industrial Wastes* 30(11): 1357-1360. doi: 10.2307/25033739

Fitch, E. B., 1958. Detention Time Concepts. *Sewage and Industrial Wastes* 30(11): 1361-1363. doi: 10.2307/25033740

Fletcher, T. D., H. Andrieu and P. Hamel, 2013. Understanding, Management and Modelling of Urban Hydrology and Its Consequences for Receiving Waters: A State of the Art. *Advances in Water Resources* 51:261-279.

Fong, D.Y.T., C.W. Kwan, K.F. Lam and K.S.L. Lam, 2003. Use of the Sign Test for the Median in the Presence of Ties. *American Statistician* 57: 237-240.
Galli, J., 1990. Thermal Impacts Associated with Urbanization and Urban Stormwater Management Practices. Metropolitan Washington Council of Governments Publication Number 91701, prepared for the Maryland Department of the Environment, 198p. http://www.dnr.state.md.us/irc/docs/00008608.pdf, *accessed* October 2012.
Geosyntec Consultants and Wright Water Engineers (GC and WWE), 2009. Urban Stormwater BMP Performance Monitoring. http://www.bmpdatabase.org/, *accessed* September, 2012.
Geosyntec Consultants and Wright Water Engineers (GC and WWE), 2012a. BMP Performance Summary: Chesapeake Bay and Related Areas: International Stormwater BMP Database. http://www.bmpdatabase.org/, *accessed* January, 2014.
Geosyntec Consultants and Wright Water Engineers (GC and WWE), 2012b. International Stormwater Best Management Practices (BMP) Database Pollutant Category Summary: Statistical Addendum: TSS, Bacteria, Nutrients, and Metals. http://www.bmpdatabase.org/, *accessed* January, 2014.
Grizzard, Thomas, 2013. Personal communication.
Hancock, G., and R. Chambers, 2010. A Field-Based Evaluation of Wet Detention Ponds: How Effective Are Ponds at Water Quantity Control. *Journal of the American Water Resources Association* (JAWRA) 46(6): 1145-1158. doi: 10.1111 / j.1752-1688.2010.00481.x
Hartigan, J. P., 1989. Basis For Design of Wet Detention Basin BMPs. In: *Design of Urban Runoff Quality Controls: Proceedings of Engineering Foundation Conference on Current Practice and Design Criteria for Urban Quality Control*, Larry A. Roesner *et al.* (Editor). American Society of Civil Engineers, New York, New York. pp. 122-143.
Hawkins, R. H., and J.H. Judd, 1972. Water Pollution as Affected by Street Salting. *Journal of the American Water Resources Association* (JAWRA) 8(6): 1246-1252. doi: 10.1111/j.1752-1688.1972.tb05268.x
Helsel, D.R., and R.M. Hirsch, 2002. Statistical Methods in Water Resources. United States Geologcacl Survey Techniques of Water Resources Investigations 04-A3, 523p. http://water.usgs.gov/pubs/twri/twri4a3/, *accessed* September, 2012.
Helsel, D. R., 2012. Statistics for Censored Environmental Data Using Minitab and R: 2nd Edition. John Wiley and Sons, Hoboken, NJ.
Hewlett, J. D., and J.C. Fortson, 1982. Stream Temperature Under an Inadequate Buffer Strip in the Southeast Piedmont. *Journal of the American* Water Resources *Association* (JAWRA) 18(6): 983-988. doi: 10.1111/j.1752-1688.1982.tb00105.x
Hogan, D. M., and M.R. Walbridge, 2007. Best Management Practices for Nutrient and Sediment Retention in Urban Stormwater Runoff. *J. Environ. Qual.* 36(2): 386-395. doi: 10.2134/jeq2006.0142
Hollis, G. E., 1975. The effect of urbanization on floods of different recurrence interval. *Water Resour. Res.* 11(3): 431-435. doi: 10.1029/WR011i003p00431
Horner, R., J. Skupien, E. Livingston and E. Shaver, 1994. *Fundamentals of Urban Runoff Management: Technical and Institutional Issues*. Terrene Institute, Washington, DC.
Hvitved-Jacobsen, T., 1990. Design Criteria for Detention Pond Quality. In: *Urban Stormwater Quality Enhancement: Source Control, Retrofitting and Combined Sewer Technology*, Harry C. Torno (Editor). American Society of Civil Engineers, New York, New York. pp. 111-130.

Jacobson, C. R., 2011. Identification and quantification of the hydrological impacts of imperviousness in urban catchments: A review. *Journal of Environmental Management* 92:1438-1448.

Klein, R. D., 1979. Urbanization and Stream Quality Impairment. *Journal of the American Water Resources Association* (JAWRA) 15(4): 948-963. doi: 10.1111/j.1752-1688.1979.tb01074.x

Landers, J., 2006. Test Results Permit Side-by-Side Comparisons of BMPs. *Civil Engineering* 76(4): 34-35.

Landers, J., 2010. Nearly $300 Billion Needed For Wastewater, Storm Water Over Next 20 Years, EPA Says. *Civil Engineering* 80(8): 20-22.

Leopold, L. 1968. Hydrology for Urban Land Planning: A Guidebook on the Hydrologic Effects of Urban Land Use. United States Geological Survey Circular 554, 18p. http://pubs.er.usgs.gov/publication/cir554, *accessed* 2012.

Lessard, P., and M.B. Beck, 1991. Dynamic simulation of storm tanks. *Water Research* 25(4): 375-391.

MacDonald, D.D., C.G. Ingersoll and T.A. Berger, 2000. Development and Evaluation of Consensus-Based Sediment Quality Guidelines for Freshwater Ecosystems. *Environmental Contamination and Toxicology* 39(1): 20-31.

Marsalek, J., 1978. Pollution Due to Urban Runoff: Unit Loads and Abatement Measures. Pollution from Land Use Activities Reference Group, International Joint Commission, 37p. http://www.gis.lrs.uoguelph.ca/agrienvarchives/download/pluarg_pollution_urban_runoff.pdf*, accessed* December, 2012.

Maxted, J.R., and E. Shaver, 1999. The Use of Retention Basins to Mitigate Stormwater Impacts to Aquatic Life. In: *National Conference on Retrofit Opportunities for Water Resource Protection in Urban Environments: Proceedings*. (EPA/625/R-99/002). U.S. Environmental Protection Agency, Cincinatti, Ohio. pp. 6-15.

Medina, D.E. and S. Curtis, 2011. Comparing LID and Stream Restoration: Finding Cost-Effective Ways to Reduce Sediment and Nutrient Loads. *Stormwater* 12(6): 10-23.

Meierdiercks, K. L., J. A. Smith, M. L. Baeck and A. J. Miller, 2010. Analyses of Urban Drainage Network Structure and its Impact on Hydrologic Response. *Journal of the American Water Resources Association* (JAWRA) 46(5):932-943.

National Research Council of the National Academies, Committee on Reducing Stormwater Discharge Contributions to Water Pollution (NRC), 2009. Urban Stormwater Management in the United States. National Academies Press, Washington, DC.

Nelson, E. J. and D. B. Booth, 2002. Sediment sources in an urbanizing, mixed land-use watershed. *Journal of Hydrology* 264:51-68.

Nelson, K. C. and M. A. Palmer, 2007. Stream Temperature Surges Under Urbanization and Climate Change: Data, Models, and Responses1. *Journal of the American Water Resources Association* (JAWRA) 43:440-452.

Novotny, V., and H. Olem, 1994. Water quality: Prevention, Identification, and Management of Diffuse Pollution. Van Nostrand Reinhold, New York, New York, ISBN-13: 978-0442005597

Northern Virginia Planning District Commission and Engineers and Surveyors Institute (NVPDC and ESI), 1992. Northern Virginia BMP Handbook.

http://www.novaregion.org/DocumentCenter/Home/View/1679, *accessed* December, 2012.

Occoquan Watershed Monitoring Laboratory (OWML), 2001. Quality Assurance Plan for the Occoquan Watershed Monitoring Laboratory. Manassas, VA**.**

Parkin, T.B., and J.B. Robinson, 1993. Statistical Evaluation of Median Estimators for Lognormally Distributed Variables. *Soil Science Society of America Journal* 57(2): 317 - 323. doi:10.2136/sssaj1993.03615995005700020005x

Poff, N. L., B. P. Bledsoe and C. O. Cuhaciyan, 2006. Hydrologic variation with land use across the contiguous United States: Geomorphic and ecological consequences for stream ecosystems. *Geomorphology* 79:264-285.

Roseen, R. M., N. DiGennaro, A. Watts, T. P. Ballestero and J. Houle, 2010. Preliminary Results of the Examination of Thermal Impacts from Stormwater BMPs. In: *World Environmental and Water Resources Congress 2010: Challenges of Change,* Richard N. Palmer (Editor). American Society of Civil Engineers, New York, New York. pp. 3424-3451.

Sartor, J.D., and G.B. Boyd, 1972. Water Pollution Aspects of Street Surface Contamination. U.S. Environmental Protection Agency Environmental Protection Technology Series Report EPA-R2-72-081, 236p. http://nepis.epa.gov/, *accessed* 2012.

Schehl, T. P., 1995. The Peformance of an Underground Storage Structure for Urban Stormwater Quality Management. Masters Thesis, Virginia Polytechnic Institute and State University, Merrifield, VA.

Schueler, T., 1987. Controlling Urban Runoff: A Practical Manual for Planning and Designing Urban BMPs. Metropolitan Washington Council of Governments, Washington, DC.

Schueler, T.R., 1994. The Importance of Imperviousness. *Watershed Protection Techniques* 1(3): 100 - 111.

Schueler. T.R., 1996. Irreducible Pollutant Concentrations Discharged from Urban BMPs. *Watershed Protection Techniques* 2(2): 369-372.

Schueler, T.R., L. Fraley-McNeal and K. Cappiella, 2009. Is Impervious Cover Still Important? Review of Recent Research. *Journal of Hydrologic Engineering* 14(4): 309-315.

Shaheen, D., 1975. Contributions of Urban Roadway Usage to Water Pollution. U.S. Environmental Protection Agency Environmental Protection Technology Series Report EPA-600/2-75-004, 352p. http://nepis.epa.gov/, *accessed* 2012.

Shaver, E., 1999. Conservation Design for Stormwater Management. In: *National Conference on Retrofit Opportunities for Water Resource Protection in Urban Environments: Proceedings*. (EPA/625/R-99/002). U.S. Environmental Protection Agency, Cincinatti, Ohio. pp 118-122.

Smullen J., A. Shallcross and K. Cave, 1999. Updating the U.S. Nationwide Urban Runoff Quality Database. *Water Science and Technology* 39(12): 9-16.

Sonzogni, W.C., G. Chesters, D.R. Coote, D.N. Jeffs, J.C. Konrad, R.C. Ostry, and J.B. Robinson., 1980. Pollution from land runoff. *Environmental Science and Technology* 14(2): 148-153. doi: 10.1021/es60162a003

Strecker, E.W., 1999. Considerations and Approaches for Monitoring the Effectiveness of Urban BMPs. In: *National Conference on Retrofit Opportunities for Water Resource Protection in Urban Environments: Proceedings*. EPA/625/R-99/002. U.S. Environmental Protection Agency, Cincinatti, Ohio. pp. 65-82.

Strecker, E.W., M. Quigley, B. Urbonas, J. Jones and J. Clary, 2001. Determining Urban Storm Water BMP Effectiveness. *Journal of Water Resources Planning and Management* 127(3): 144-149. doi:10.1061/(ASCE)0733-9496(2001)127:3(144)

Struck, S., A. Selvakumar and M. Borst, 2008. Prediction of Effluent Quality From Retention Ponds and Constructed Wetlands for Managing Bacterial Stressors in Storm-Water Runoff. *Journal of Irrigation and Drainage Engineering* 134(5): 567-578. doi: 10.1061/(ASCE)0733-9437(2008)134:5(567).

Sweeney, B., and R. Vannote, 1978. Size Variation and the Distribution of Hemimetabolous Aquatic Insects: Two Thermal Equilibrium Hypotheses. *Science* 200(4340): 444-446.

Tanizaki, Y., T. Shimokawa and M. Yamazaki, 1992. Physico-Chemical Speciation of Trace Elements in Urban Streams by Size Fractionation. *Water Research* 26(1): 55-63.

Thackston, E.L., D. Shields Jr. and P.R. Schroeder, 1987. Residence Time Distributions in Shallow Basins. *Journal of Environmental Engineering* 113(6): 1319-1332.

Thomas, B., J. Steidl, O. Dietrich and G. Lischeid, 2011. Measures to sustain seasonal minimum runoff in small catchments in the mid-latitudes: A review. *Journal of Hydrology* 408(3-4):296-307.

Urbonas, B., P. James, C. Guo and L. Tucker, 1990. Optimization of Stormwater Quality Capture Volume. In: *Urban Stormwater Quality Enhancement: Source Control, Retrofitting and Combined Sewer Technology*, Harry C. Torno (Editor). American Society of Civil Engineers, New York, New York. pp. 94-110.

Urbonas, B., 1995. Recommended Parameters to Report with BMP Monitoring Data. *Journal of Water Resources Planning and Management* 121(1): 23-34. doi:10.1061/(ASCE)0733-9496(1995)121:1(23)

U.S. Environmental Protection Agency (USEPA), 1983. Results of the Nationwide Urban Runoff Program: Volume 1 Final Report. EPA 832-R-83-112, 198p. http://nepis.epa.gov/, *accessed* 2012.

U.S. Environmental Protection Agency (USEPA), 1986. Methodology for Analysis of Detention Basins for Control of Urban Runoff Quality. EPA 440/5-87-001, 51p. http://nepis.epa.gov/, *accessed* November, 2012.

U.S. Environmental Protection Agency (USEPA), 1990. Urban Targeting and BMP Selection: An Information and Guidance Manual for State Non Point Source Program Staff Engineers and Managers. EPA 68-C8-0034, 54p. http://nepis.epa.gov/, *accessed* 2012.

U.S. Environmental Protection Agency (USEPA), 1992a. Environmental Impacts of Stormwater Discharges: A National Profile. EPA 841-R-92-001, 44p. http://www.epa.gov/nscep/index.html, *accessed* November, 2012.

U.S. Environmental Protection Agency (USEPA), 1992b. Managing Non-Point Source Pollution: Final Report to Congress on Section 319 of the Clean Water Act (1989). EPA-506/9-90, 197p. http://cfpub.epa.gov/ols/, *accessed* December, 2012.

U.S. Environmental Protection Agency (USEPA), 1992c. NPDES Storm Water Sampling Guidance Document. EPA 833-B-92-001, 185p. http://nepis.epa.gov/, *accessed* 2012.

U.S. Environmental Protection Agency (USEPA), 1993. Combined Sewer Overflow Control Manual. EPA 625-R-93-007, 102p. http://www.epa.gov/nscep/index.html, *accessed* 2012.

U.S. Environmental Protection Agency (USEPA), 1999. Stormwater Technology Fact Sheet: Wet Detention Ponds. EPA 832-F-99-048, 7p. http://nepis.epa.gov/, *accessed* 2013.

U.S. Environmental Protection Agency (USEPA), 2008. Clean Watersheds Needs Survey: 2008 Report to Congress. EPA-832-R-10-002, 40p. http://cfpub.epa.gov/ols/, *accessed* December, 2012.
U.S. Environmental Protection Agency (USEPA), 2009. National Water Quality Inventory: Report to Congress, 2004 Reporting Cycle. EPA 841-R-08-001, 37p. http://cfpub.epa.gov/ols/, *accessed* December, 2012
U.S. Environmental Protection Agency (USEPA), 2010. Chesapeake Bay Total Maximum Daily Load for Nitrogen, Phosphorus and Sediment, 352 p. http://www.epa.gov/reg3wapd/tmdl/ChesapeakeBay/tmdlexec.html, *accessed* May, 2014.
U.S. Environmental Protection Agency (USEPA), 2011a. A National Evaluation of the Clean Water Act Section 319 Program, 194p. http://water.epa.gov/polwaste/nps/cwact.cfm, *accessed* December, 2012.
U.S. Environmental Protection Agency (USEPA), 2011b. TMDL for Benthic Impairments in the Accotink Creek Watershed: Fairfax County, City of Fairfax and Town of Vienna, VA. http://www.epa.gov/waters/tmdldocs/va/VA_AccotinkCreekBenthic_AL_TMDL.pdf, *accessed* 2013.
U.S. Environmental Protection Agency (USEPA), 2012. Water Quality Standards Handbook: 2nd Edition. EPA-823-B-12-002, 136p. http://water.epa.gov/scitech/swguidance/standards/handbook/index.cfm, *accessed* 2013.
Viessman, W. and M. Hammer, 1998. Water Supply and Pollution Control, 6th ed. Addison-Wesley, Menlo Park, CA, ISBN: 0-321-01460-x.
Virginia Department of Environmental Quality (Virginia DEQ), 2011. Virginia DEQ Stormwater Design Specification No. 14: Wet Ponds, 23p. http://vwrrc.vt.edu/swc/ *accessed* February, 2013.
Walker, D.J., 1998. Modelling residence time in stormwater ponds. *Ecological Engineering* 10(3): 247-262. doi: http://dx.doi.org/10.1016/S0925-8574(98)00016-0
Walker, W., 1985. Empirical Methods for Predicting Eutrophication in Impoundments; Report 3, Phase II: Model Refinements. US Army Corps of Engineers Technical Report E-81-9, 321p.
Walsh, C.J., A.H. Roy, J.W. Feminella, P.D. Cottingham, P.M. Groffman and P.M. Raymond II., 2005. The Urban Stream Syndrome: Current Knowledge and the Search for a Cure. *Jornal of the North American Benthological Society* 24(3): 706-723.
Wang, L., J. Lyons, P. Kanehl and R. Bannerman, 2001. Impacts of Urbanization on Stream Habitat and Fish Across Multiple Spatial Scales. *Environmental Management* 28(2): 255-266. doi: 10.1007/s002670010222
Whipple, W. Jr., J.V. Hunter and S.L. Yu, 1978. Runoff Pollution From Multiple Family Housing. *Water Resources Bulletin* 14(2): 288 - 301.
Whipple, W. Jr., and J.V. Hunter, 1981. Settleability of Urban Runoff Pollution. *Journal (Water Pollution Control Federation)* 53(12): 1726-1731. doi: 10.2307/25041200
Willey, J.D., R.J. Kieber, M.S. Eyman and G.B. Avery Jr., 2000. Rainwater Dissolved Organic Carbon: Concentration and Global Flux. *Global Biogeochemical Cycles* 14(1): 139-148.
Williamson, R. B., 1985. Urban Stormwater Quality I. Hillcrest, Hamilton, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 19(3): 413-427. doi: 10.1080/00288330.1985.9516106
Winer, R. 2000. National Pollutant Removal Performance Database for Stormwater Treatment Practices: 2nd Edition. Center For Watershed Protection.

Yang, G., L. C. Bowling, K. A. Cherkauer and B. C. Pijanowski, 2011. The impact of urban development on hydrologic regime from catchment to basin scales. *Landscape and Urban Planning* 103(2):237-247.

Yousef, Y.A., T. Hvitved-Jacobsen, J. Sloat and W. Lindeman, 1994. Sediment Accumulation in Detention or Retention Ponds. *The Science of the Total Environment* 146/147: 451-456.

APPENDIX A: Storm Hydrographs

HYDROGRAPH CALIBRATION

In order to balance storm inflow and outflow volumes within ±10% of each other, modifications were made for all recorded storms. This was done for two reasons: data from the acoustic Doppler profiler (ADCP) recording inflows in sub-watershed A had obvious errors including negative flows, data gaps, runoff spikes not corresponding to rain patterns, and general overestimation of flows; and the perforated PVC drain pipe serving as the pond's low flow outlet quickly clogged with sediment, causing overestimation of outflow rates at all pond storage levels.

The outflow measurement was adjusted using field measurements from mid-December of flow through the clogged PVC pipe. The measurements showed a great reduction in flow compared to the unclogged state. A linear equation (1) was developed to relate the clogged pipe flow, P_f, in m^3/s to the depth of water in the pond, h_p, in m:

$$P_f=h_p(0.00060)-0.00040 \tag{1}$$

Y represent the clogged pipe flow while X represent the depth of water in the pond. The correction to sub-watershed A inflows was based on flow measurements and stream channel surveys taken in the field during the September 18, 2012 storm. Comparison of the field data to ADCP readings showed the device to be measuring cross sectional area accurately, but not flow. A calibrated flow velocity was obtained by dividing the field-measured flow by the ADCP-measured cross sectional area of the stream. A velocity-velocity curve was created to graphically compare the calibrated and ADCP velocities. A linear regression line (2) described the relationship between the two.

$$Y=0.5152X \tag{2}$$

Y is the calibrated velocity and X is the ADCP velocity, all in m^3/s. The adjusted flow at sub-watershed A is thus found by obtaining the calibrated velocity from each recorded ADCP velocity, and then multiplying the calibrated velocity by the ADCP cross sectional area.

A smaller adjustment to sub-watershed A inflows was needed to account for stream base flow after the bulk of storm flow had passed. Most storms featured a “long tail” of light but steady outflow for several hours after the end of rain. Even with the flow adjustment, zero or negative inflows were commonly recorded during these “long tails”. sub-watershed A feeds a perennial stream, and some base flow would have been present. To account for this, the head on the weir was analyzed during all “long tails,” and was found to be dropping at very slow rates, generally less than 1/100th of a meter every 2-3 hours. This suggested that inflow and outflow rates were nearly identical, and sub-watershed A base flow was set equal to the outflow rate for the extreme ends of storms.

The adjusted flows and base flow estimates brought sub-watershed A inflow volumes closer to outflow volumes for all storms, but there remained clearly apparent errors in many of the inflows; most notably unexplained runoff spikes, negative flow velocities in the middle of storms, and multi-hour gaps when the ADCP stopped recording. In addition, a significant ungaged area (sub-watershed C) around the perimeter of Ashby Pond contributed runoff during storms that was not measured. To estimate these flows, the EPA’s Stormwater Management Model Version 5 software (SWMM, U.S. EPA, National Risk Management Research Laboratory. Accessed July 2013, http://www.epa.gov/nrmrl/wswrd/wq/models/swmm/) was used to model inflows from sub-watersheds A and C. GIS analysis was first performed on all sub-watersheds to further sub-divide them based on land use and quantify the impervious surface cover within each land use (City of Fairfax GIS. Accessed November, 2012, http://www.fairfaxva.gov/). The land use patterns are summarized in Table A1 below.

TABLE A1: Ashby Pond Sub-Watershed Land Use

Sub-Shed	Land Use					Total Area	Total % Impervious
	Highly Impervious, Hydraulically Connected		Less Impervious, Unconnected Land		Natural Areas		
	Acreage	% Impervious	Acreage	% Impervious	Acreage		
A	51.6	61	21.1	27	14.5	87.2	43%
B	6.4	46	19	26	6.7	32.1	24%
C	0	NA	7.7	27	8.2	15.9	13%

This information was entered into SWMM and flows were estimated for the sub-watershed C for all storms. For sub-watershed A, the SWMM data was used to "plug the gaps" for those storms where data was missing or errors existed. GIS was used to measure the surface area of Ashby Pond, and the volume of direct rainfall on the pond was calculated.

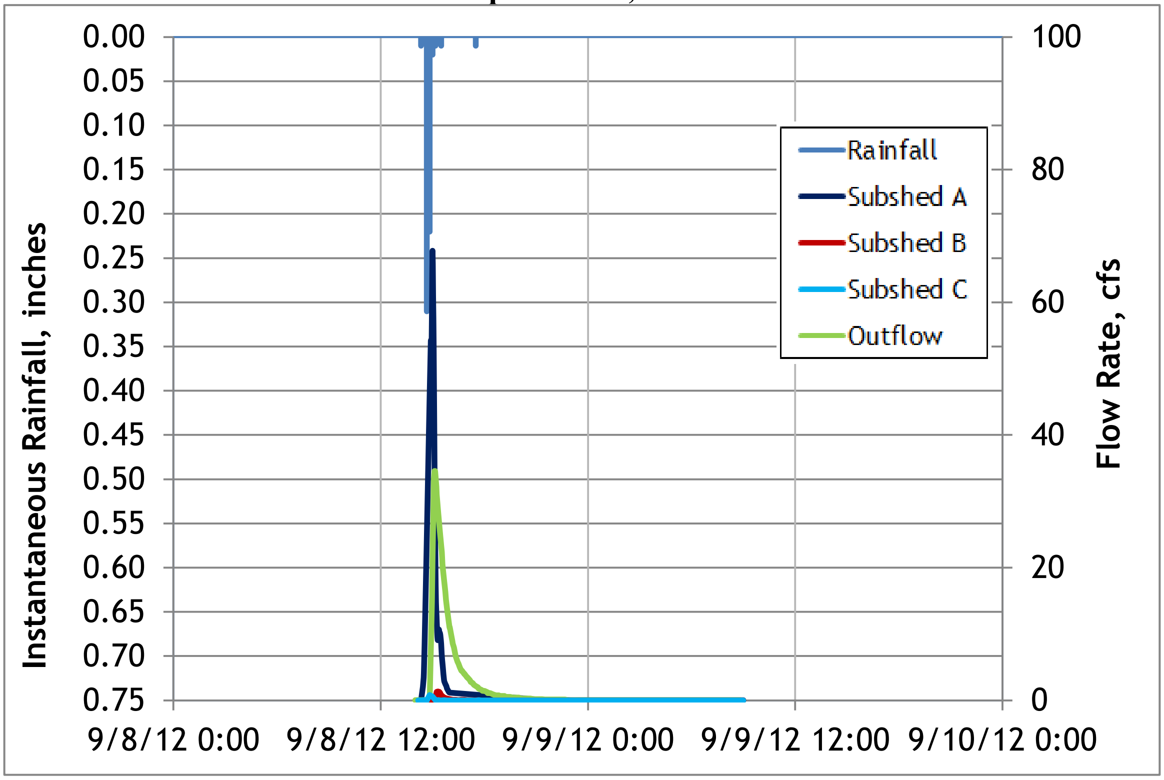
The adjustments to the outflow and inflows brought all but three storms to an acceptable balance, with the outflow volume and pond storage being within or very close to ± 10% of the inflow volume. For the three storms still outside of the acceptable range—September 8, September 18, and October 29, 2012—SWMM modeling was performed for all sub-watersheds. The SWMM inflow volumes and hydrographs were compared to the adjusted inflows and hydrographs. For the September 8th and 18th storms, SWMM inflow volumes were 99.8% and 95.9% of the adjusted inflows, respectively, and the hydrograph patterns matched very closely. It was assumed that the error was therefore with the outflow, so outflows were set equal to the adjusted inflows for these storms. For the October 29th storm, SWMM estimates of inflow were below the adjusted inflow volumes, and both were less than the adjusted outflow volume, but the inflow hydrographs matched very well. It was assumed that the error was with the outflow, and outflow volume was set equal to the adjusted inflow. Final flow balances are shown in Table A2.

TABLE A2: Flow Balances for All Storms

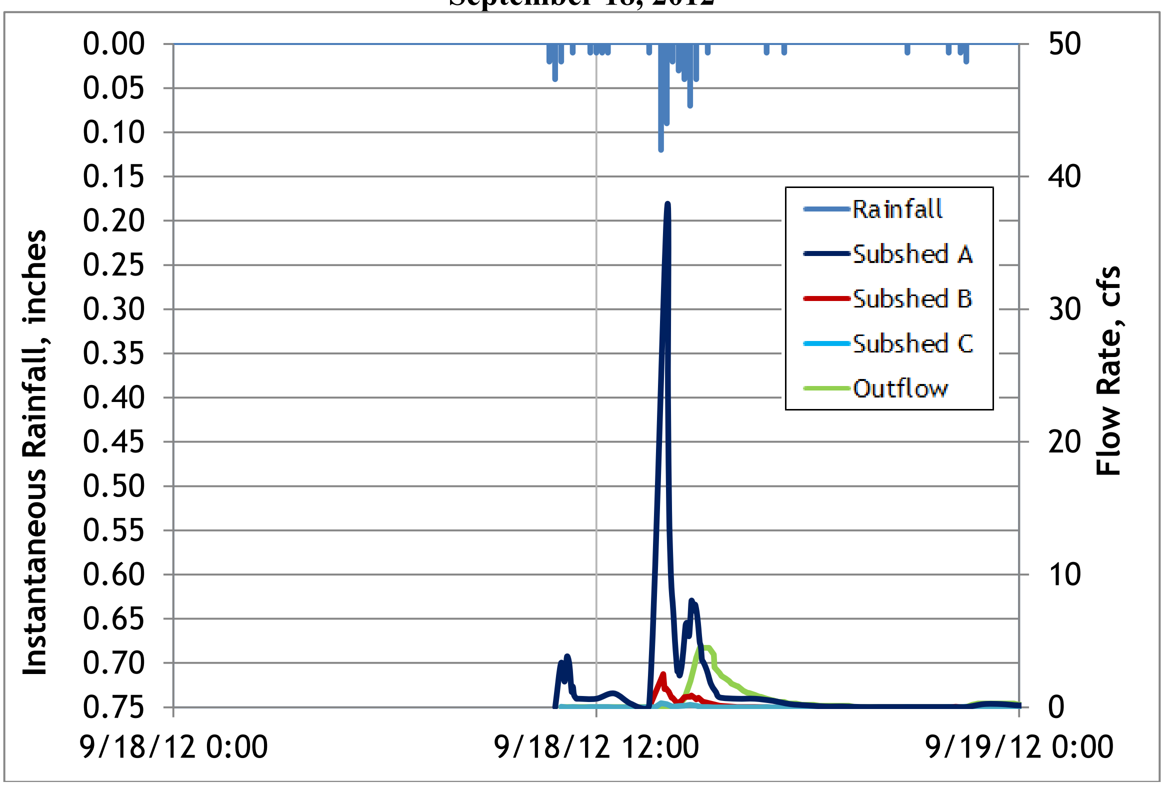
Storm	Outflow and Pond Storage as % of Inflow
8-Sep	100%
18-Sep	100%
26-Sep	101%
27-Sep	99%
2-Oct	105%
18-Oct	100%
29-Oct	100%
20-Dec	93%
26-Feb	92%
15-Jan	96%
30-Jan	113%
6-Mar	97%
12-Mar	113%

HYDROGRAPHS

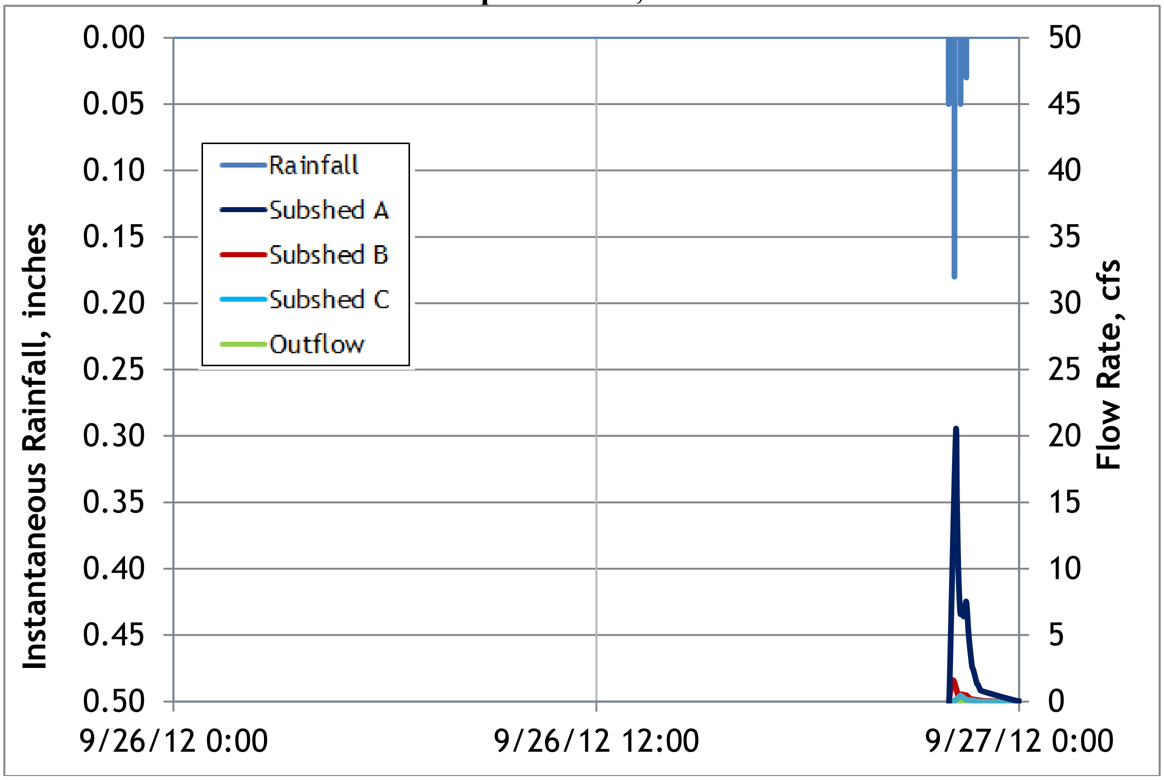
September 8, 2012



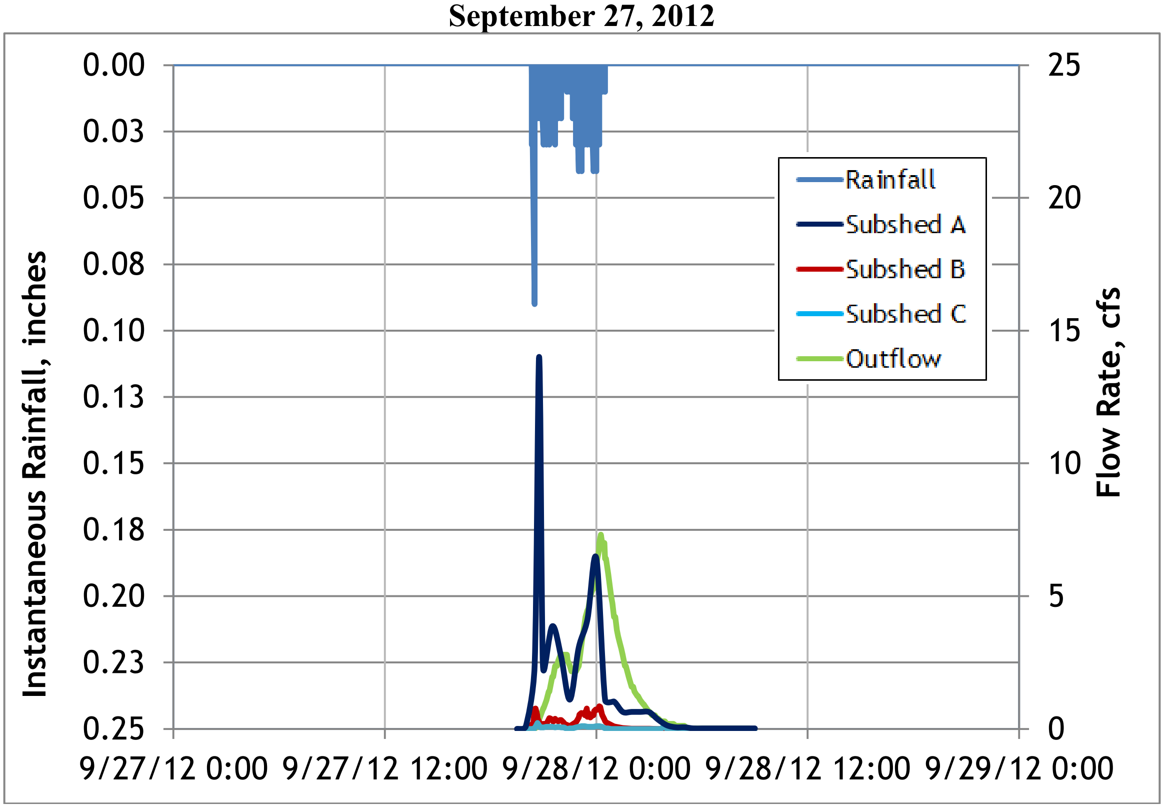
September 18, 2012



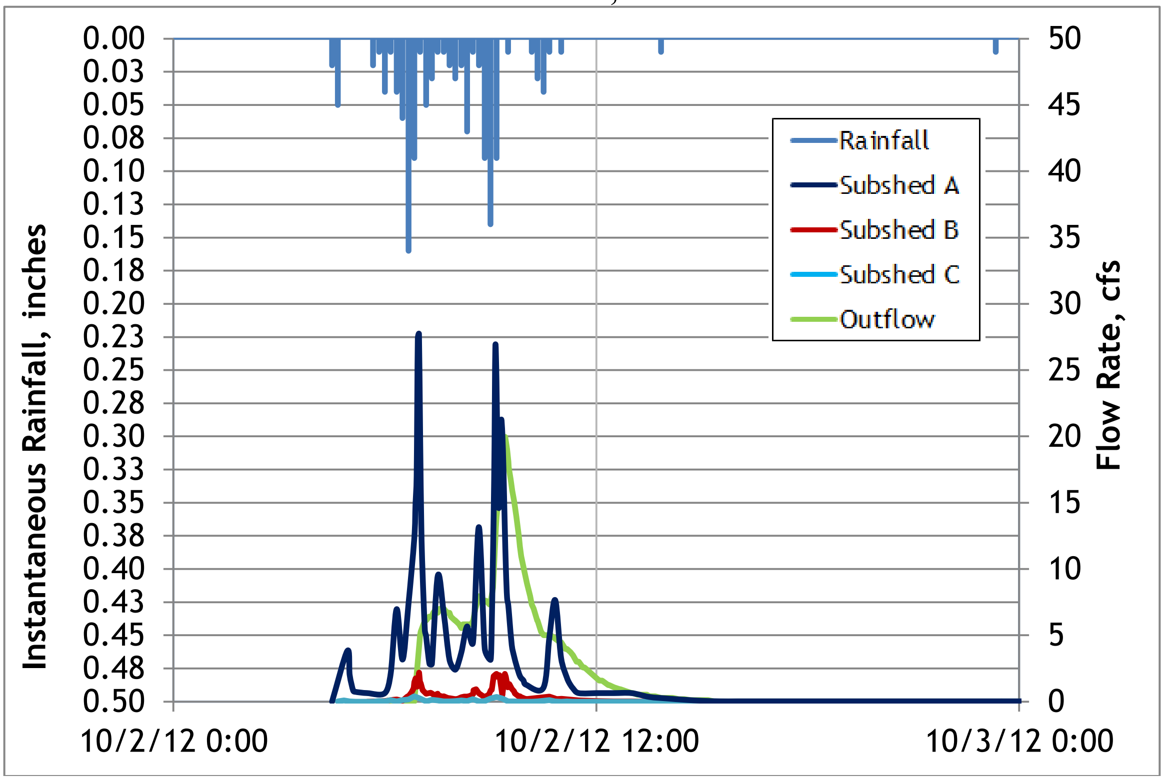
September 26, 2012



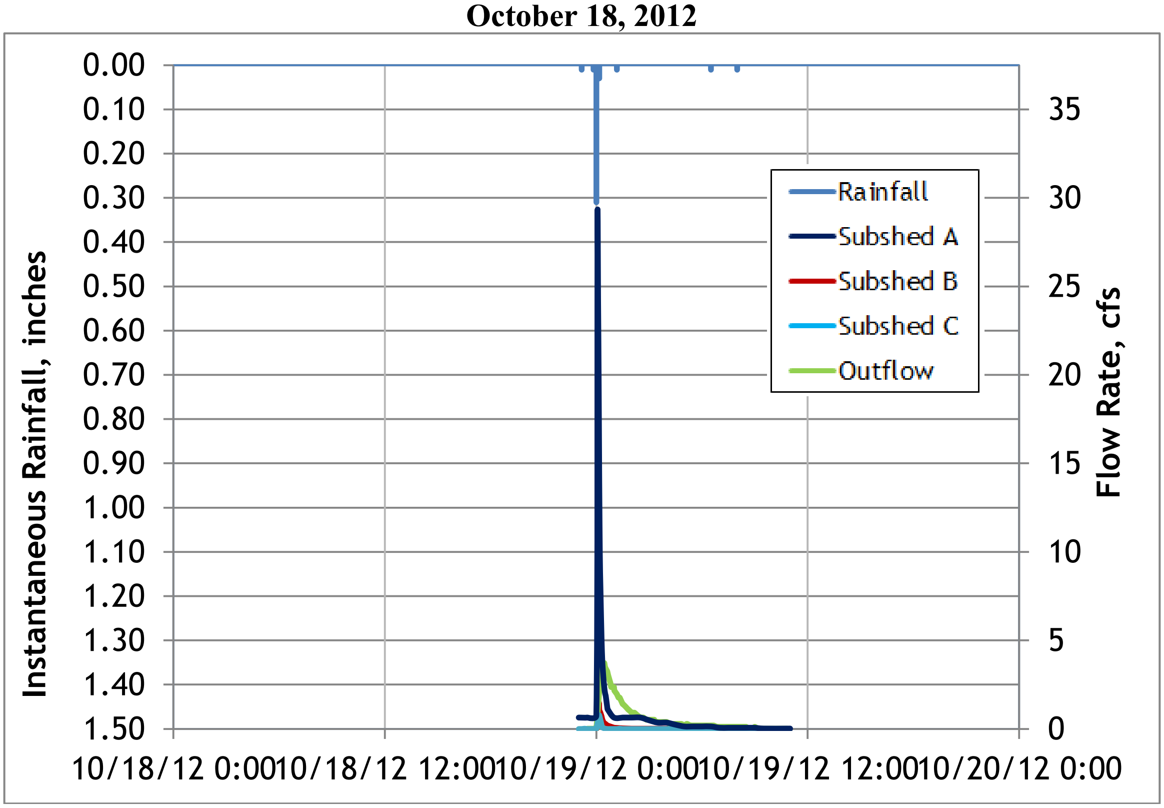
September 27, 2012



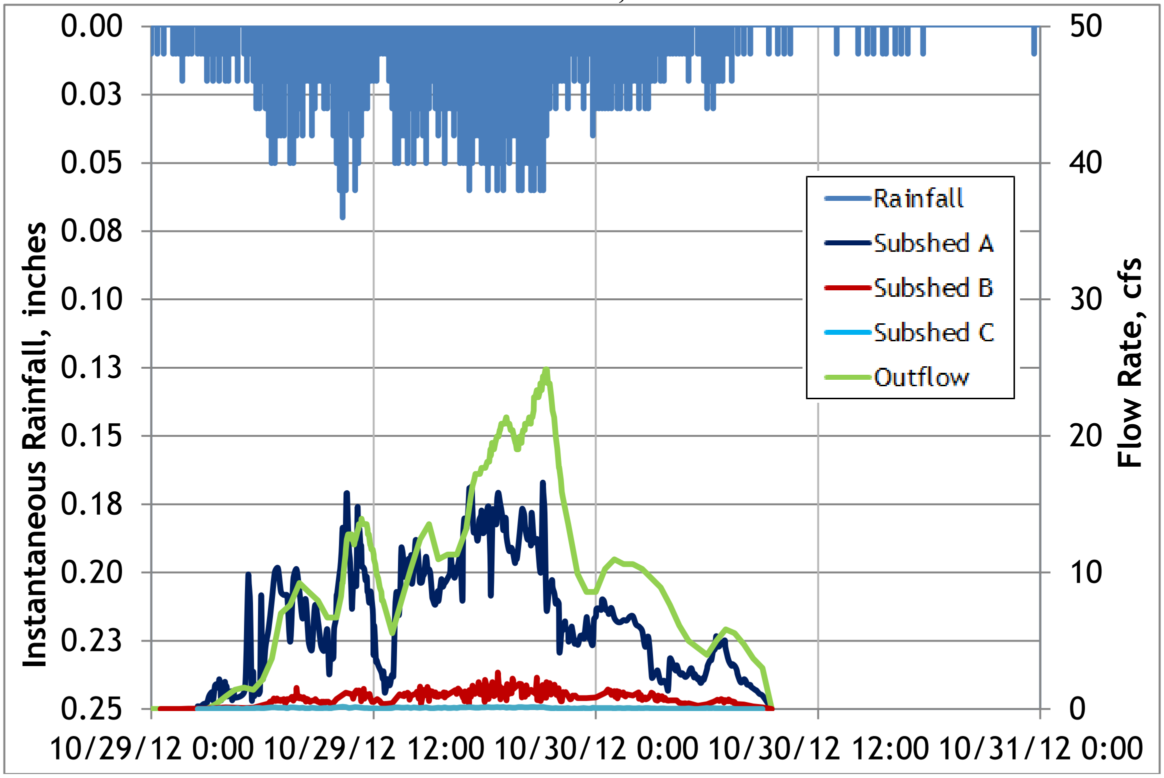
October 2, 2012



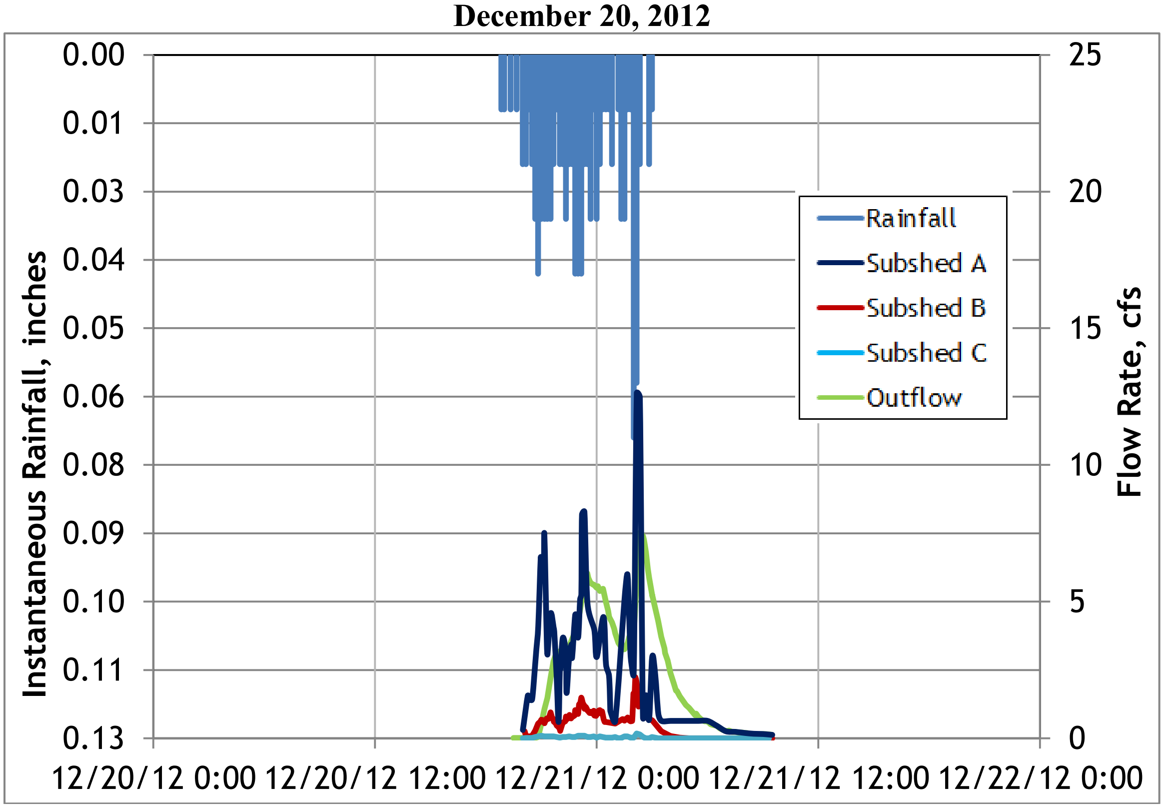
October 18, 2012



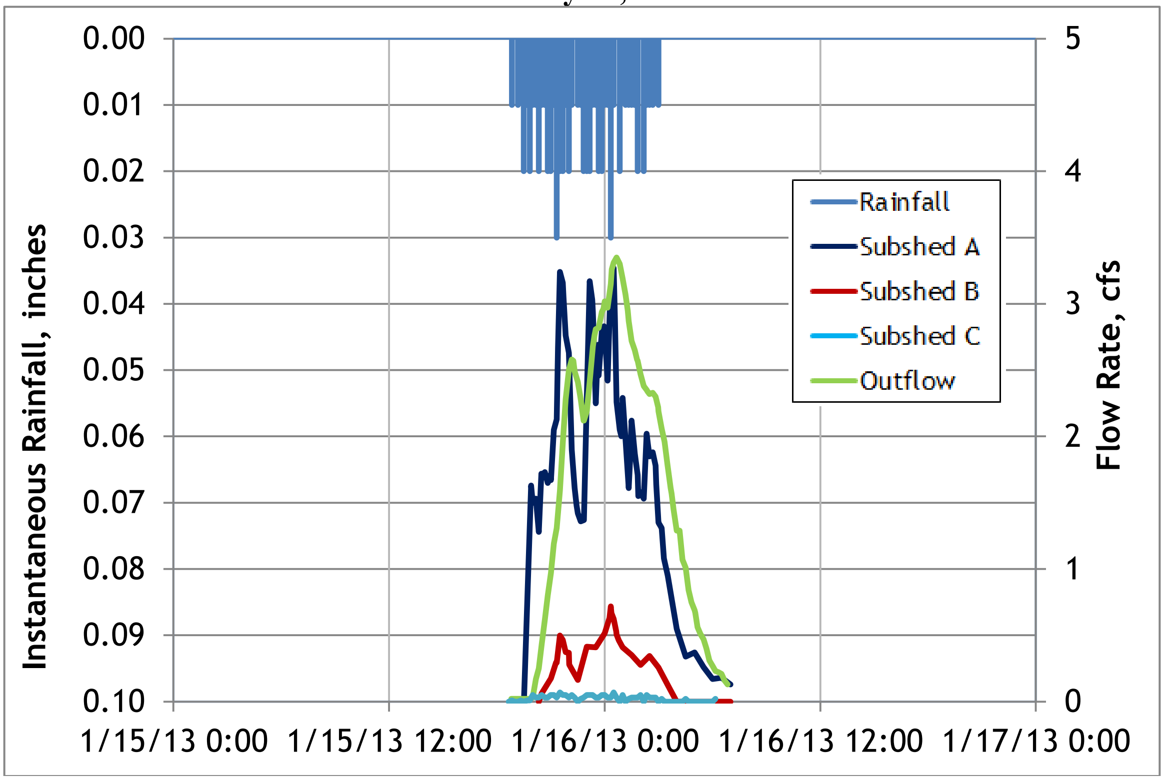
October 29, 2012



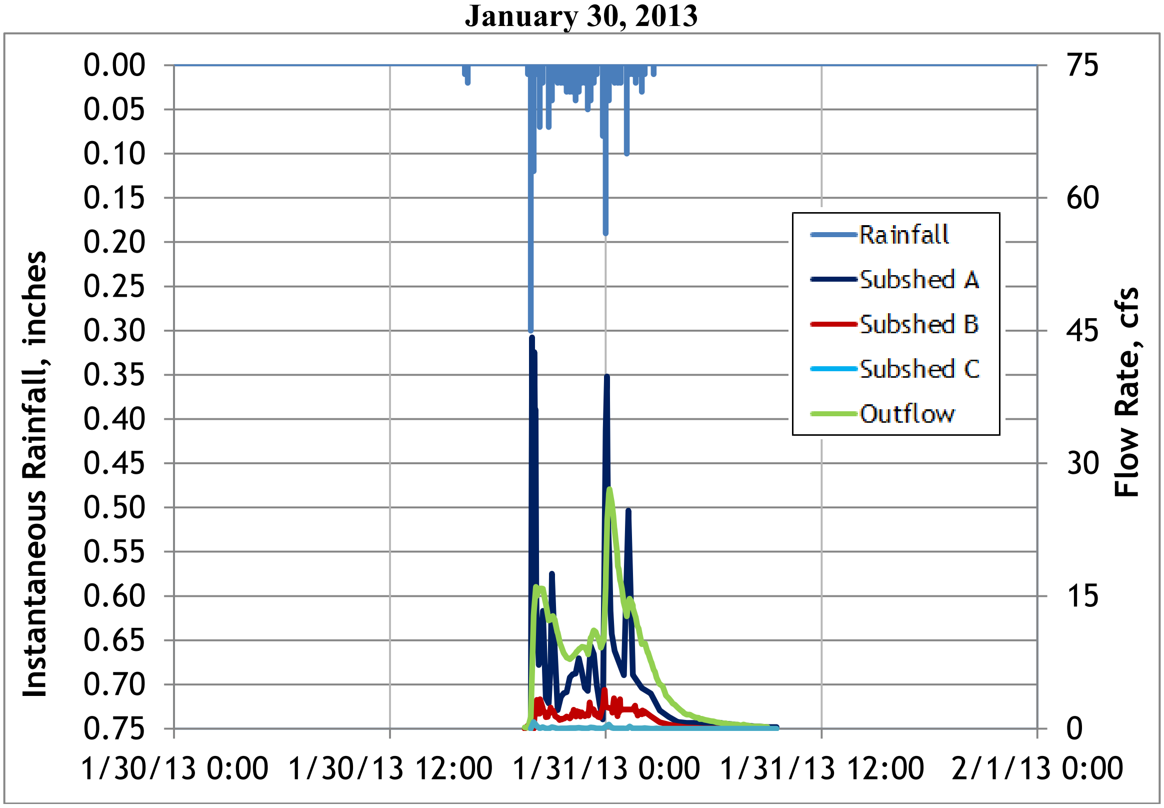
December 20, 2012



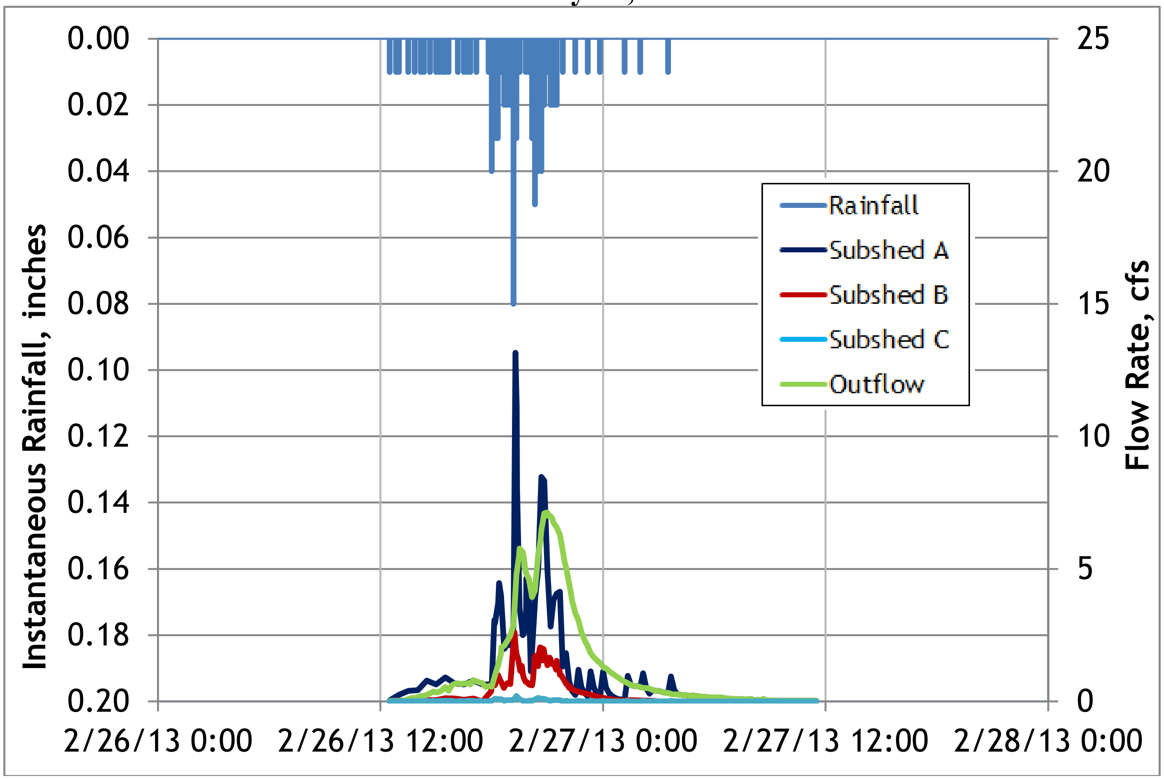
January 15, 2013



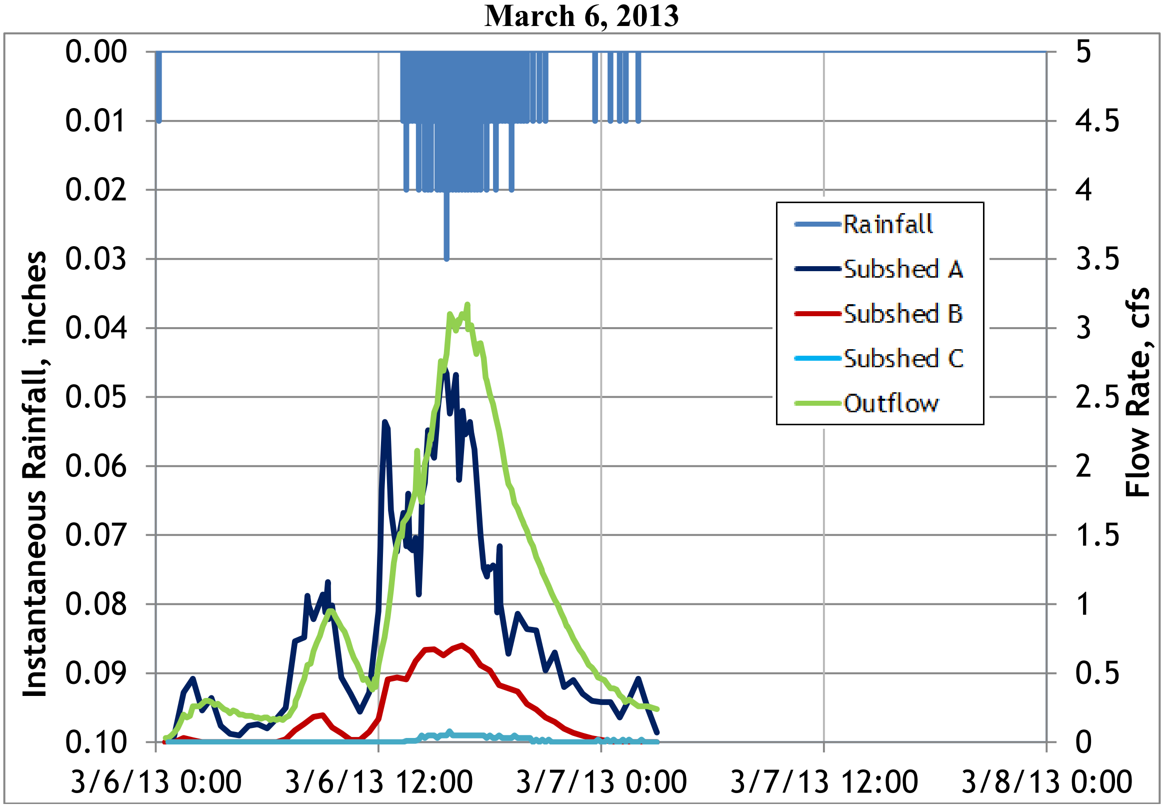
January 30, 2013



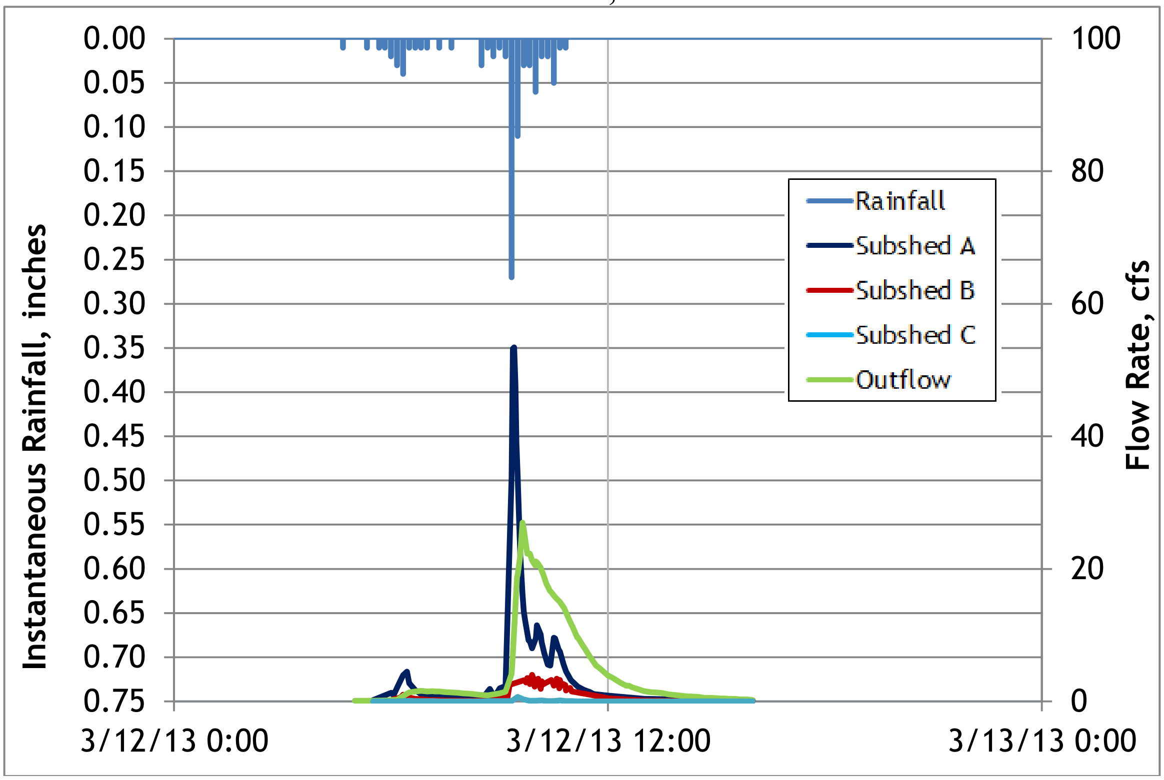
February 26, 2013



March 6, 2013



March 12, 2013



APPENDIX B: Pollutant EMCs and Loads

Storm	Constituent	OP		TP		Ammoniacal-N		OX-N	
		Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
9/8/2012	Conc. (mg/L)	0.12	0.020	0.27	0.20	0.031	0.020	0.61	0.11
	Load (Lbs)	0.67	0.13	1.4	1.3	0.17	0.13	3.3	0.71
9/18/2012	Conc. (mg/L)	0.064	0.02	0.16	0.22	0.008 - 0.017	0.020	0.33	0.050
	Load (Lbs)	0.35	0.079	0.89	0.87	.043 - .092	0.079	1.8	0.20
9/26/2012	Conc. (mg/L)	0.11	0.020	0.39	0.16	0.29	0.090	0.61	0.08
	Load (Lbs)	0.13	0.0	0.48	0.006	0.36	0.003	0.76	0.003
9/27/2012	Conc. (mg/L)	NA	0.02	NA	0.15	NA	0.0 - 0.010	NA	0.32
	Load (Lbs)	NA	0.088	NA	0.66	NA	0.0 - 0.044	NA	1.4
10/2/2012	Conc. (mg/L)	0.053	0.02	0.20	0.11	0.025	0.0 - 0.010	0.33	0.15
	Load (Lbs)	0.48	0.17	1.8	0.96	0.23	0.0 - 0.087	3.0	1.3
10/19/2012	Conc. (mg/L)	0.083	0.03	0.24	0.14	0.13	0.020	0.44	0.14
	Load (Lbs)	0.15	0.042	0.42	0.20	0.23	0.028	0.80	0.20
10/29/2012	Conc. (mg/L)	0.080	0.080	0.16	0.14	0.026	0.010	0.29	0.34
	Load (Lbs)	4.3	4.2	8.5	7.4	1.4	0.53	16	18
12/20/2012	Conc. (mg/L)	0.053	NA	0.21	0.12	0.17	0.020	0.41	0.17
	Load (Lbs)	0.44	NA	1.7	0.89	1.4	0.15	3.4	1.3
1/15/2012	Conc. (mg/L)	0.034	0.01	0.14	0.1	0.15	0.040	0.39	0.14
	Load (Lbs)	0.16	0.04	0.66	0.43	0.67	0.17	1.8	0.60
1/30/2012	Conc. (mg/L)	0.057	0.030	0.65	0.21	0.20	0.10	0.32	0.24
	Load (Lbs)	1.1	0.63	12	4.4	3.7	2.1	5.9	5.1
2/26/2012	Conc. (mg/L)	0.043	0.010	0.18	0.14	0.19	0.10	0.68	0.14
	Load (Lbs)	0.29	0.065	1.2	0.90	1.3	0.65	4.7	0.90
3/6/2012	Conc. (mg/L)	NA	0.01	NA	0.070	NA	0.10	NA	0.17
	Load (Lbs)	NA	0.058	NA	0.41	NA	0.58	NA	0.99
3/12/2012	Conc. (mg/L)	0.038	0.02	NA	0.26	0.078	0.090	0.27	0.21
	Load (Lbs)	0.35	0.21	NA	2.7	0.71	0.92	2.5	2.2

Storm	Constituent	TN		COD		DOC		TOC	
		Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
9/8/2012	Conc. (mg/L)	1.8	1.3	34	35	6.3	6.5	6.5	7.6
	Load (Lbs)	9.5	8.6	180	230	34	42	35	49
9/18/2012	Conc. (mg/L)	1.4	1.4	NA	30	8.3	7.6	8.4	8.3
	Load (Lbs)	7.7	5.7	NA	120	45	30	46	33
9/26/2012	Conc. (mg/L)	3.3	1.2	NA	31	12	8.5	12	9.6
	Load (Lbs)	4.2	0.043	NA	1.2	15	0.32	16	0.36
9/27/2012	Conc. (mg/L)	NA	1.4	NA	28	NA	8.1	NA	8.4
	Load (Lbs)	NA	6.2	NA	120	NA	36	NA	37
10/2/2012	Conc. (mg/L)	1.2	0.81	NA	21	6	5.5	5.9	6.6
	Load (Lbs)	11	7.1	NA	180	50	52	53	58
10/19/2012	Conc. (mg/L)	1.8	1.1	48	29	7.2	8.1	7.8	8.1
	Load (Lbs)	3.3	1.5	86	41	13	11	14	11
10/29/2012	Conc. (mg/L)	1.0	0.99	37	30	11	8.8	11	9.6
	Load (Lbs)	54	52	2000	1600	580	470	610	510
12/20/2012	Conc. (mg/L)	1.4	0.87	35	28	6.7	7.5	7.6	7.8
	Load (Lbs)	11	6.4	290	210	56	55	63	58
1/15/2012	Conc. (mg/L)	1.2	0.88	28	25	5.1	6.2	5.6	6.7
	Load (Lbs)	5.3	3.7	130	110	23	26	25	29
1/30/2012	Conc. (mg/L)	3.7	1.7	37	37	6.7	6.7	8.0	8.1
	Load (Lbs)	69	36	680	770	120	140	150	170
2/26/2012	Conc. (mg/L)	1.7	1.2	35	35	6.0	5.6	6.1	6.1
	Load (Lbs)	12	7.8	240	230	41	36	42	39
3/6/2012	Conc. (mg/L)	NA	0.86	NA	21.5	NA	5.8	NA	5.8
	Load (Lbs)	NA	5.0	NA	130	NA	34	NA	34
3/12/2012	Conc. (mg/L)	2.8	1.8	NA	NA	6.2	4.7	6.7	5.5
	Load (Lbs)	25	18	NA	NA	57	48	61	56

Storm	Constituent	TSS		SSC		Cl^-		Ca	
		Inflow	Outflow	Inflow	Outflow	Inflow	Outflow	Inflow	Outflow
9/8/2012	Conc. (mg/L)	84	48	100	58	0.016 - 4.8	11	2.9	6.4
	Load (Lbs)	460	310	570	370	0.085 - 26	69	16	42
9/18/2012	Conc. (mg/L)	40	23	39	19	0.017 - 4.8	7.9	3.7	6.4
	Load (Lbs)	220	92	210	77	0.091 - 26	31	20	25
9/26/2012	Conc. (mg/L)	37	41	45	12	0.036 - 4.6	9.7	3.6	7.7
	Load (Lbs)	46	1.6	57	0.46	0.045 - 5.8	0.36	4.5	0.29
9/27/2012	Conc. (mg/L)	NA	12	A	12	NA	5.9	NA	5.7
	Load (Lbs)	NA	51	NA	54	NA	26	NA	25
10/2/2012	Conc. (mg/L)	75	10	78	11	0.016 - 4.8	0 - 5.0	4.0	5.2
	Load (Lbs)	680	91	710	93	0.14 - 43	0 - 43	37	45
10/19/2012	Conc. (mg/L)	110	14	120	19	0.025 - 4.6	8.3	2.5	6.2
	Load (Lbs)	200	20	220	27	0.046 - 8.3	12	4.4	8.7
10/29/2012	Conc. (mg/L)	210	14	21	14	5.3 - 5.8	0.0 - 5.0	5.1	4.3
	Load (Lbs)	1100	720	1100	750	280 - 310	0.0 - 270	280	230
12/20/2012	Conc. (mg/L)	57	18	64	17	0.015 - 4.8	13	0.36 - 1.6	3.4
	Load (Lbs)	470	140	540	130	0.12 - 40	98	3 - 13	25
1/15/2012	Conc. (mg/L)	37	11	38	12	9.5	49	4.0	7.4
	Load (Lbs)	170	45	170	52	44	210	18	31
1/30/2012	Conc. (mg/L)	410	120	430	120	75	94	6.4	9.3
	Load (Lbs)	7700	2600	8000	2500	1400	2000	120	200
2/26/2012	Conc. (mg/L)	79	39	85	41	78	130	12	14
	Load (Lbs)	540	250	580	270	530	820	85	89
3/6/2012	Conc. (mg/L)	NA	9.2	NA	12	NA	240	NA	16
	Load (Lbs)	NA	54	NA	72	NA	1400	NA	93
3/12/2012	Conc. (mg/L)	460	174	530	180	NA	NA	NA	NA
	Load (Lbs)	4200	1800	4800	1800	NA	NA	NA	NA

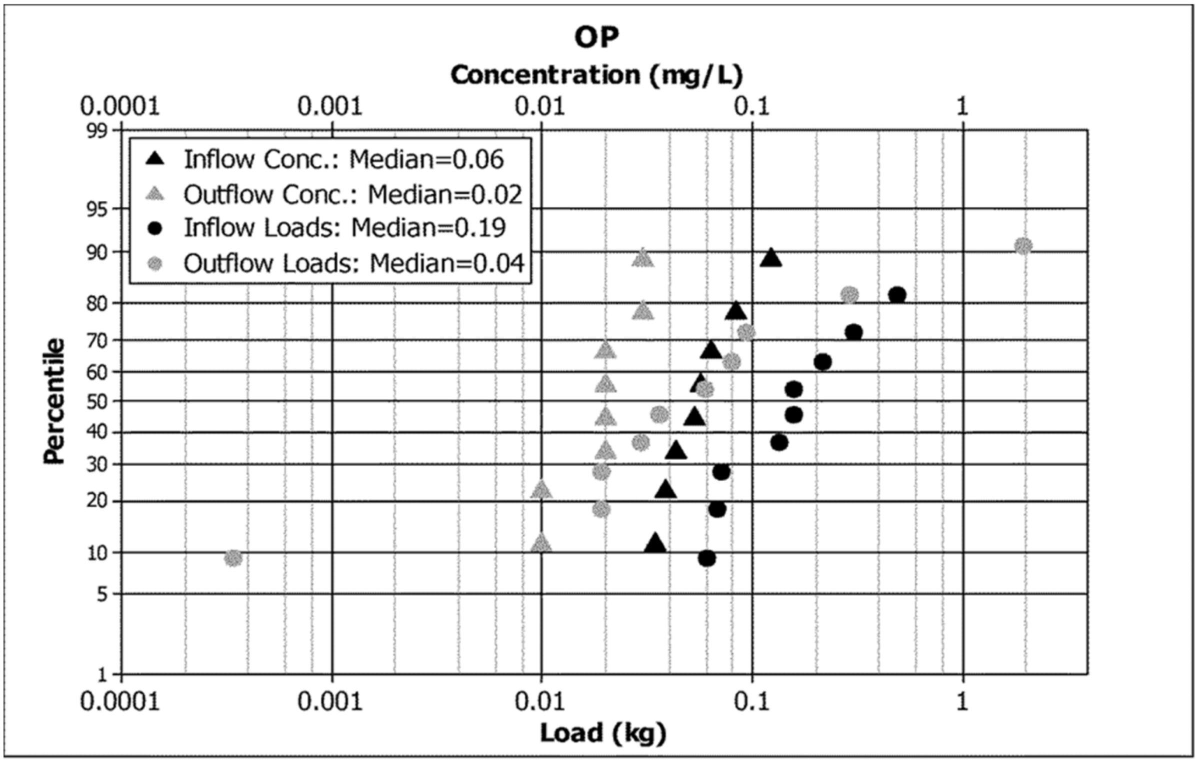
Storm	Constituent	Na^+		SZn*	
		Inflow	Outflow	Inflow	Outflow
9/8/2012	Conc. (mg/L)	1.9	5.9	NA	NA
	Load (Lbs)	10	39	NA	NA
9/18/2012	Conc. (mg/L)	3.2	5.4	25	11
	Load (Lbs)	17	21	0.14	0.042
9/26/2012	Conc. (mg/L)	2.8	6.3	NA	0.0 – 10
	Load (Lbs)	3.5	0.24	NA	0.0
9/27/2012	Conc. (mg/L)	NA	4.5	NA	12
	Load (Lbs)	NA	20	NA	0.051
10/2/2012	Conc. (mg/L)	3.0	3.8	14 – 15	0 – 10
	Load (Lbs)	27	33	0.13 – 0.14	0 – 0.087
10/19/2012	Conc. (mg/L)	0.13 – 1.4	4.2	18	0 – 18
	Load (Lbs)	0.24 – 2.5	58	0.032	0 – 0.051
10/29/2012	Conc. (mg/L)	1.6 – 1.7	0 - 1.5	18	19
	Load (Lbs)	85 – 93	0 – 79	0.98	1.0
12/20/2012	Conc. (mg/L)	0.25 – 1.4	2.6	NA	NA
	Load (Lbs)	2 – 12	19	NA	NA
1/15/2012	Conc. (mg/L)	7.7	30	NA	NA
	Load (Lbs)	35	130	NA	NA
1/30/2012	Conc. (mg/L)	47	58	NA	NA
	Load (Lbs)	880	1200	NA	NA
2/26/2012	Conc. (mg/L)	31	72	NA	NA
	Load (Lbs)	210	470	NA	NA
3/6/2012	Conc. (mg/L)	NA	140	NA	NA
	Load (Lbs)	NA	820	NA	NA
3/12/2012	Conc. (mg/L)	NA	NA	NA	NA
	Load (Lbs)	NA	NA	NA	NA

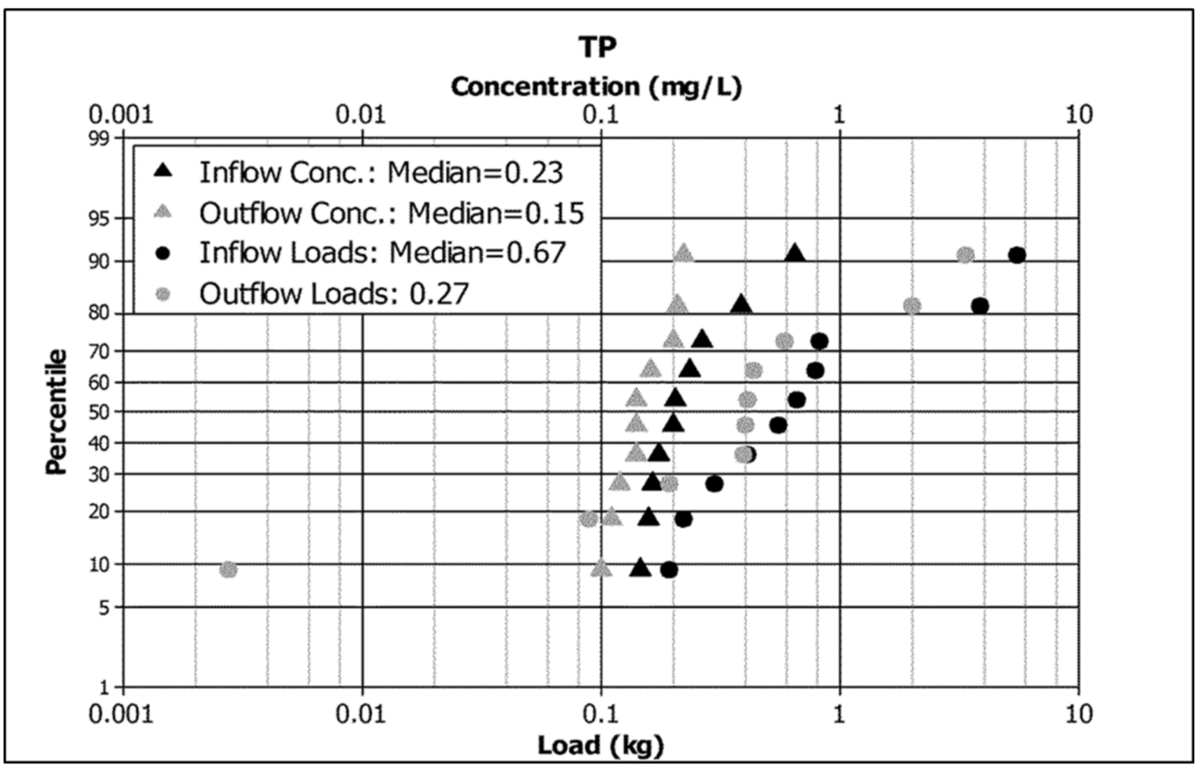
***SZn=Soluble Zinc**

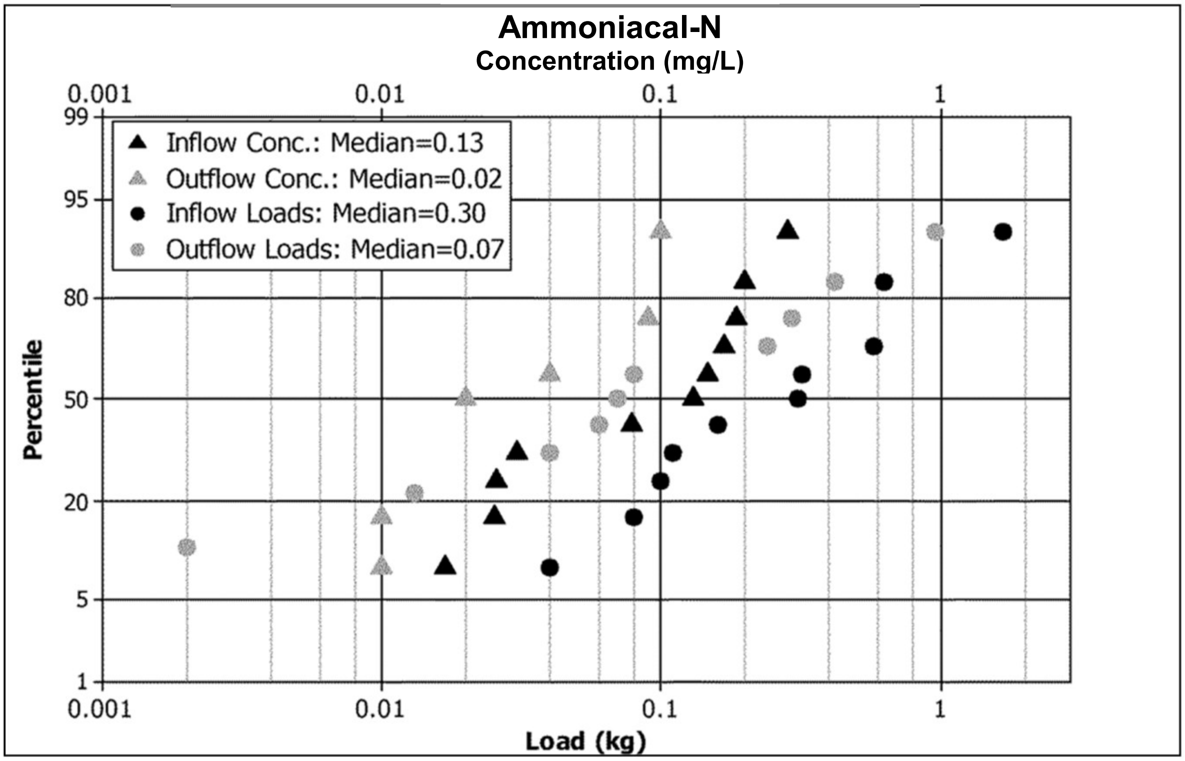
APPENDIX C: Probability Plots of Pollutant EMCs and Loads

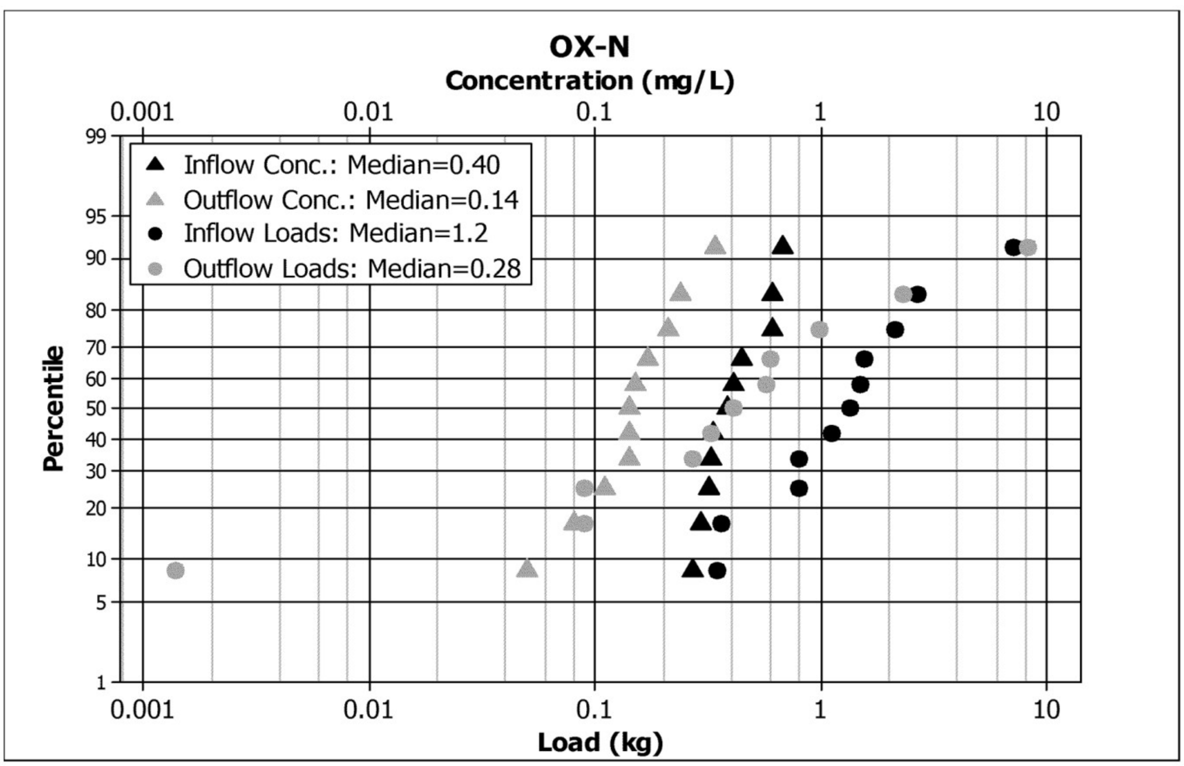
For constituents with no censored values (TP, OP, TN, OX-N, TOC, DOC, COD, TSS, SSC), the logs of storm loads and EMCs were ranked and assigned percentiles using the Cunnane plotting position (Cunnane, 1978; Helsel and Hirsch, 2002) and then converted to standard normal scores. Pearson's correlation coefficients (r) between the logs and the standard normal scores were calculated using Minitab 16 statistical software (Minitab, Inc., State College, Pennsylvania). The r values, ranging between 0.891 and 0.977 for loads and 0.916 and 0.989 for concentrations, showed that the data are lognormally distributed (Helsel and Hirsch, 2002). The bias-corrected geometric mean (BCGM), a less biased estimator of the median for lognormal data than the sample median or geometric mean (Parkin and Robinson, 1993) was calculated. All other data (ammoniacal-N, Cl^-, Na^+, Ca) is interval censored with a non-zero lower bound (i.e. inflow ammoniacal-N loads for 2012-09-18 storm are 0.02 – 0.04kg). Median estimates were made in Minitab using the Turnbull estimator, a survival analysis method similar to the commonly used Kaplan-Meier method, but applicable to interval censored data (Helsel, 2012).

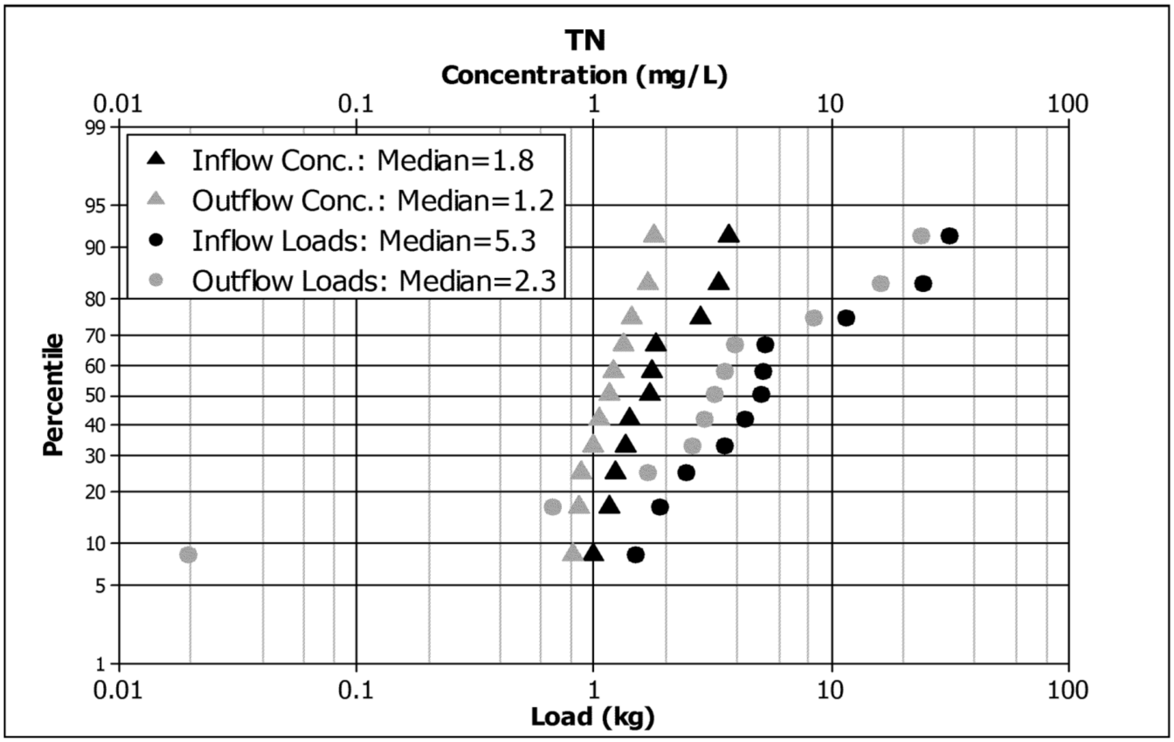
Probability plots of the loads and EMCs were created for each constituent to give graphical evidence of the changes from inflow to outflow. All plots were made using Minitab 16 statistical software (Minitab, Inc., State College, Pennsylvania). Calculated medians are listed in the legend of each graph. Were a trend line to be applied to each data type, the median would coincide closely with its intercept of the 50th percentile line. The horizontal distance between the inflow and outflow at each percentile gives an indication of the ponds performance for a storm of that size. Maintaining a relatively constant distance throughout the range of percentiles indicates that Ashby Pond provides consistent performance regardless of storm size, whereas varying distances show that the pond performs better for certain sized storms and worse for others.

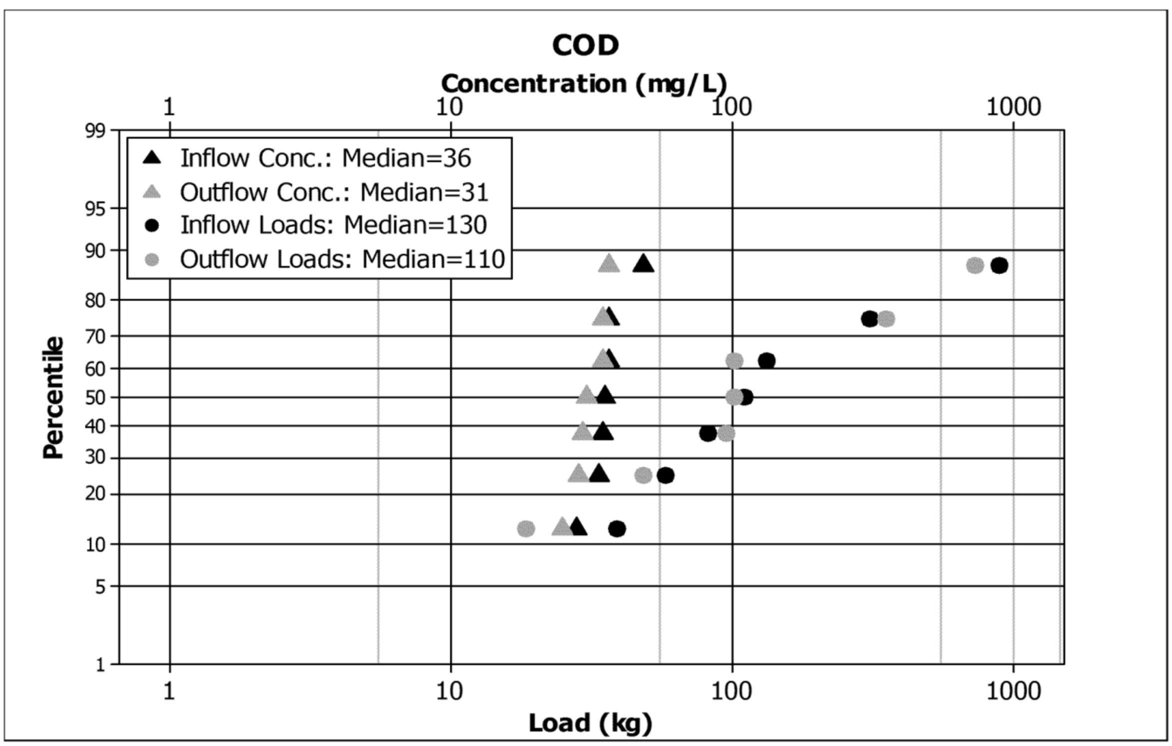

OP
Concentration (mg/L)
0.0001
0.001
0.01
0.1
1
Inflow Conc.: Median=0.06
Outflow Conc.: Median=0.02
Inflow Loads: Median=0.19
Outflow Loads: Median=0.04
99
95
90
80
70
60
50
40
30
20
10
5
1
Percentile
Load (kg)

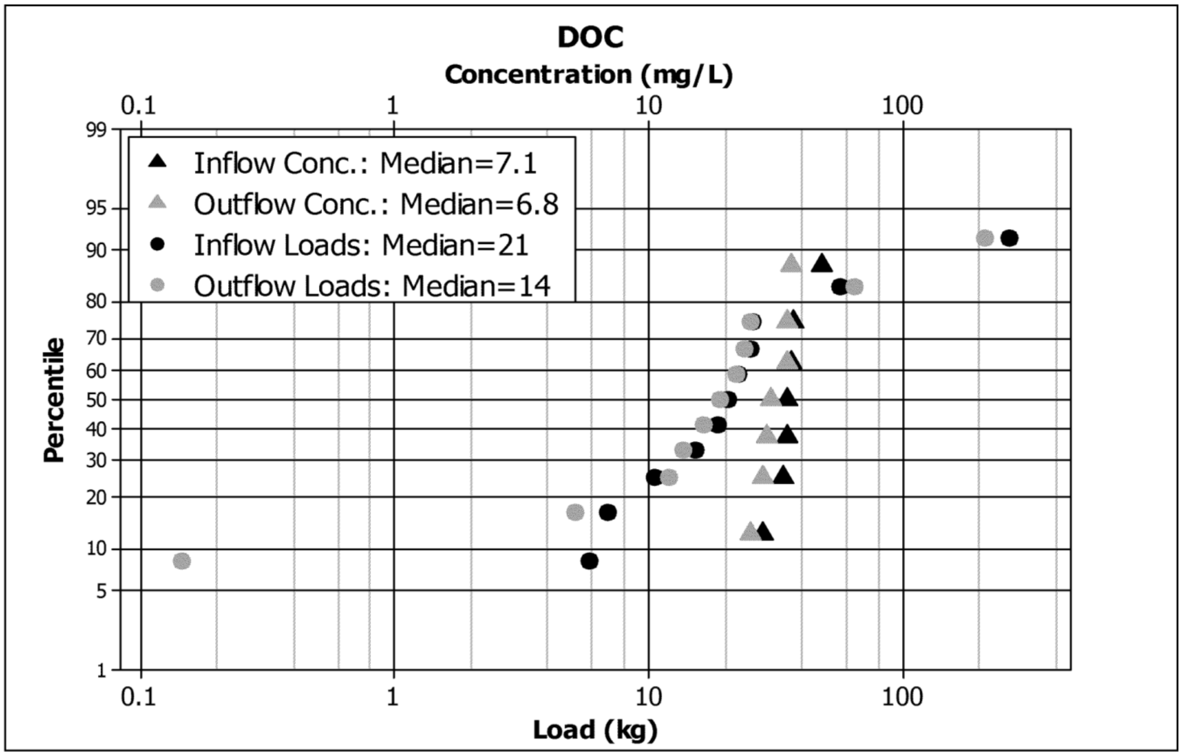

TP
Concentration (mg/L)
0.001
0.01
0.1
1
10
Inflow Conc.: Median=0.23
Outflow Conc.: Median=0.15
Inflow Loads: Median=0.67
Outflow Loads: 0.27
99
95
90
80
70
60
50
40
30
20
10
5
1
Percentile
Load (kg)

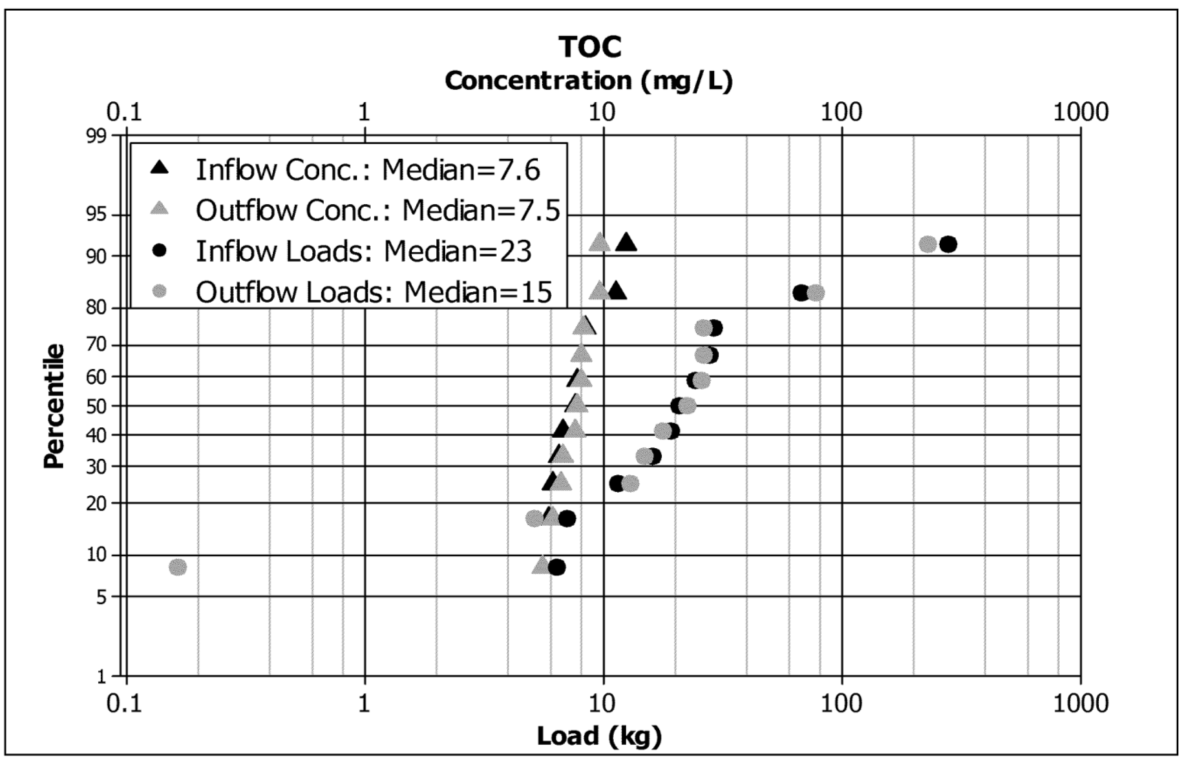

Ammoniacal-N
Concentration (mg/L)
Inflow Conc.: Median=0.13
Outflow Conc.: Median=0.02
Inflow Loads: Median=0.30
Outflow Loads: Median=0.07
Percentile
Load (kg)

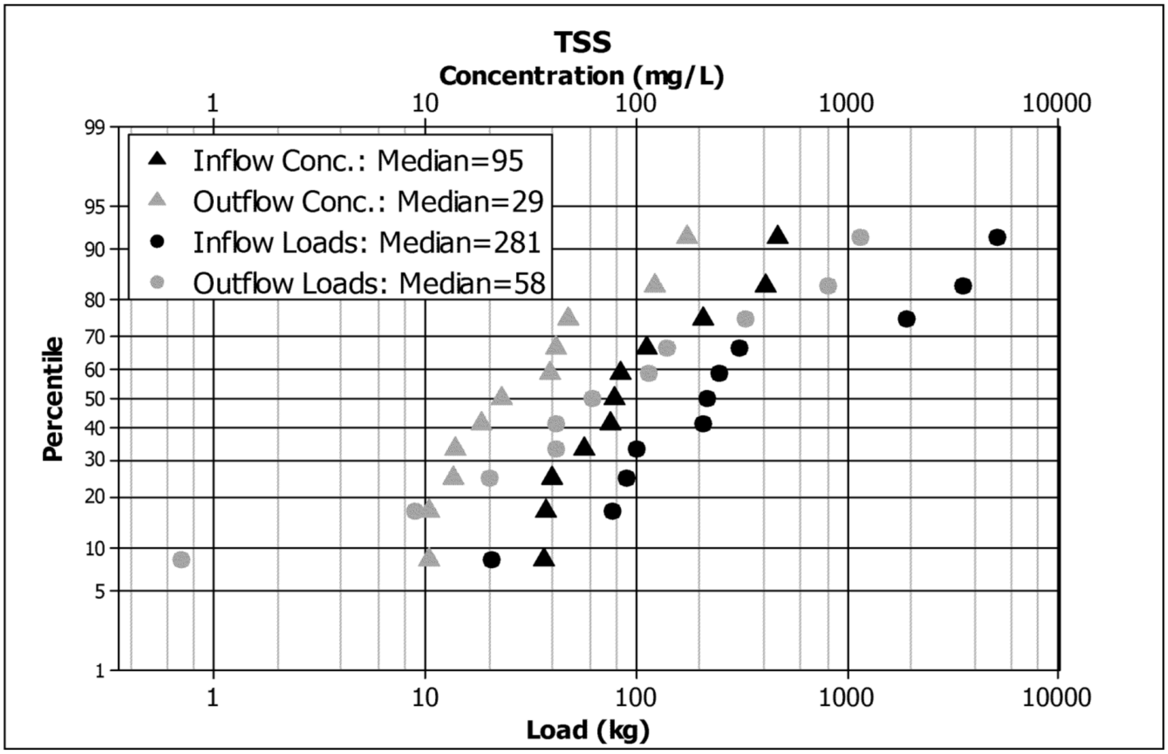

OX-N
Concentration (mg/L)
Inflow Conc.: Median=0.40
Outflow Conc.: Median=0.14
Inflow Loads: Median=1.2
Outflow Loads: Median=0.28
Percentile
Load (kg)

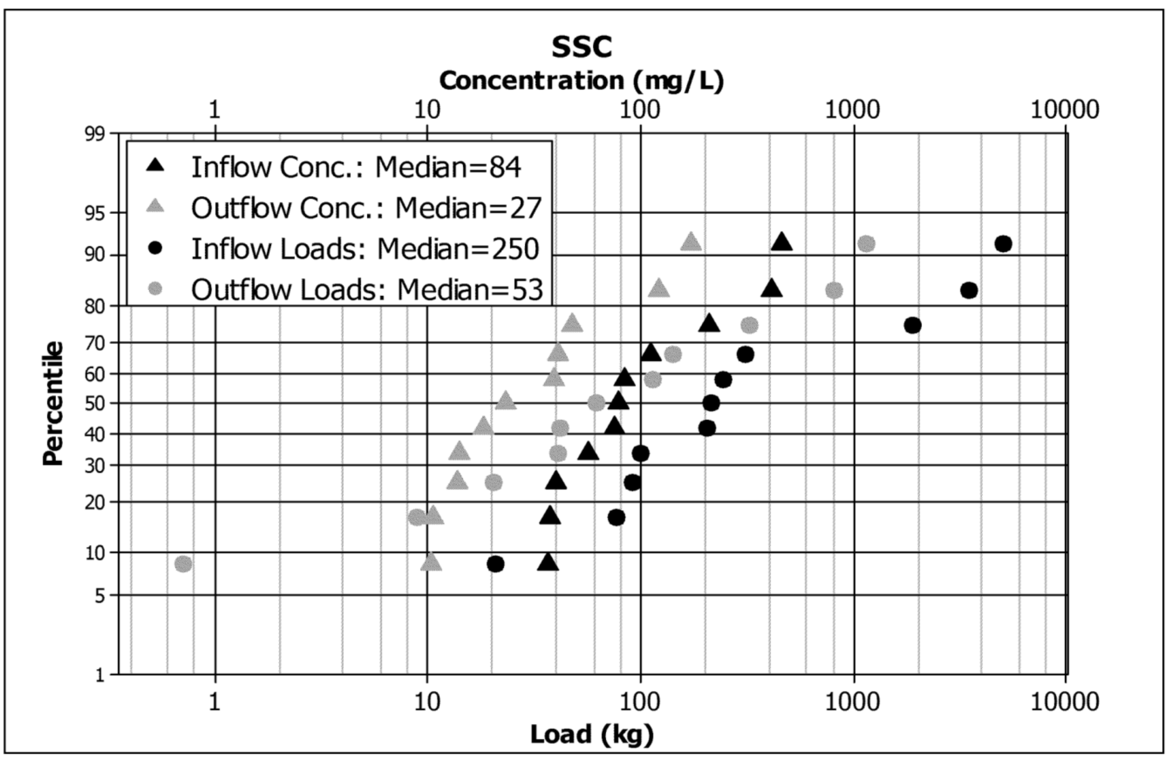

TN
Concentration (mg/L)
Inflow Conc.: Median=1.8
Outflow Conc.: Median=1.2
Inflow Loads: Median=5.3
Outflow Loads: Median=2.3
Percentile
Load (kg)

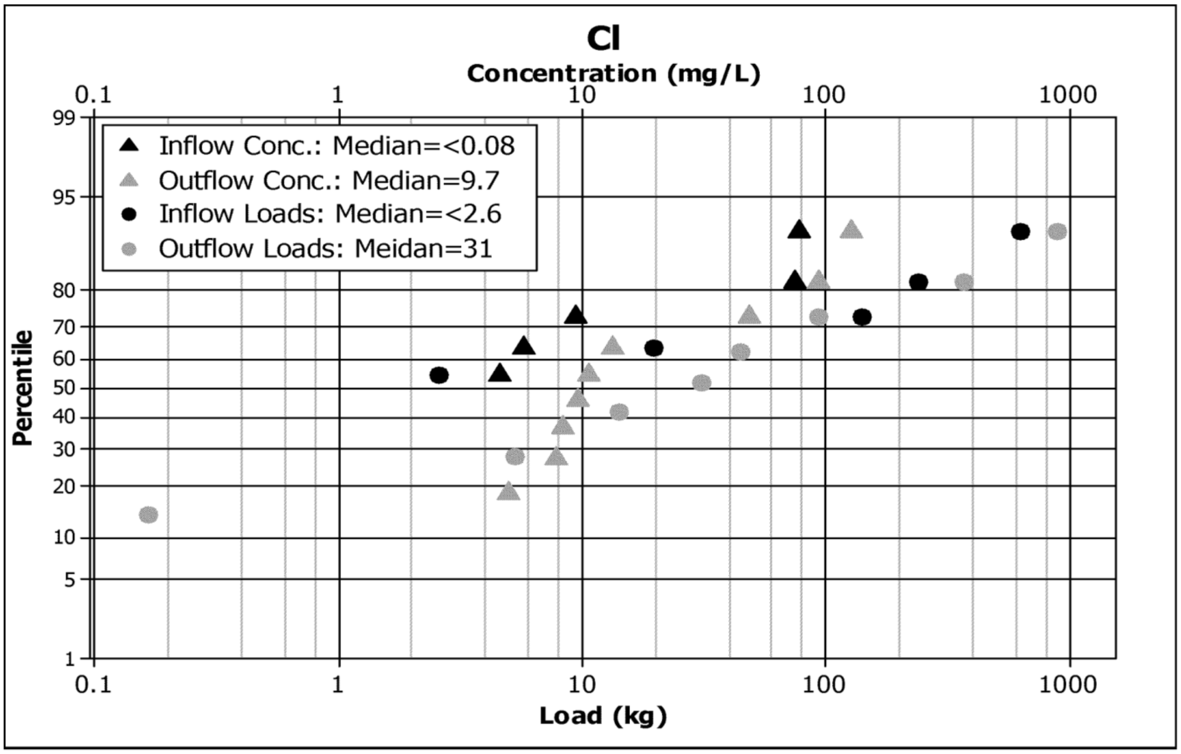

COD
Concentration (mg/L)
Inflow Conc.: Median=36
Outflow Conc.: Median=31
Inflow Loads: Median=130
Outflow Loads: Median=110
Percentile
Load (kg)

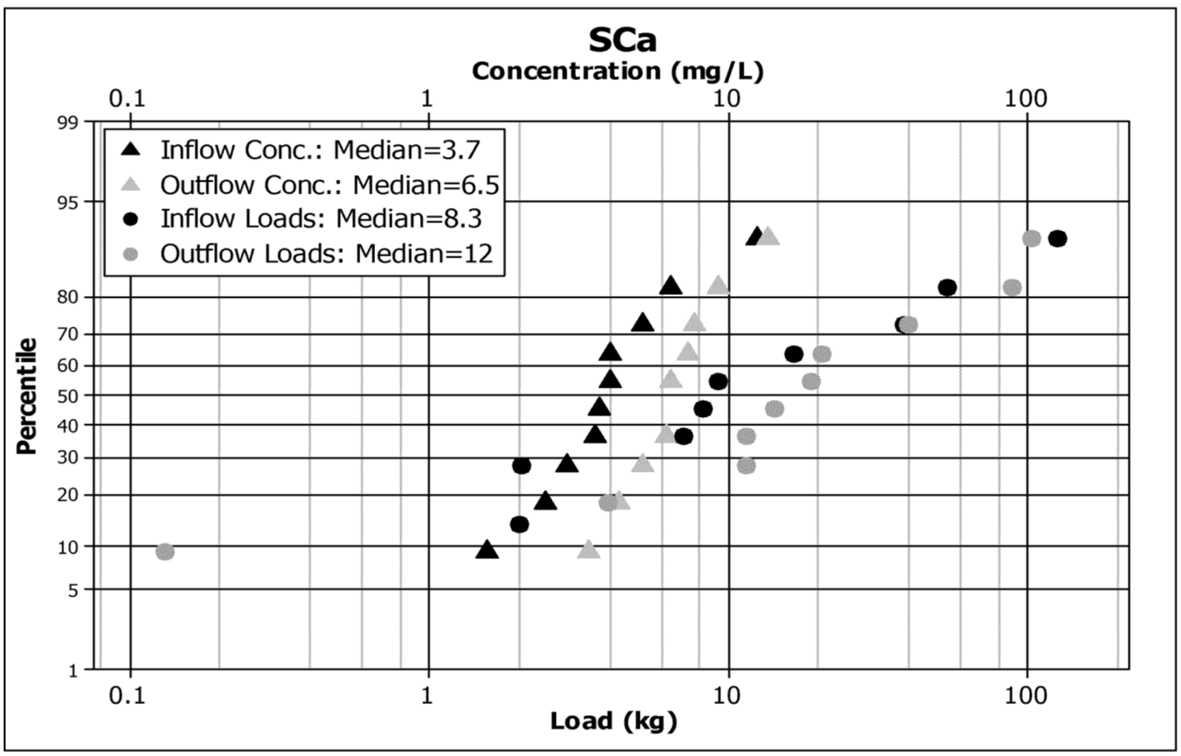

DOC
Concentration (mg/L)
Inflow Conc.: Median=7.1
Outflow Conc.: Median=6.8
Inflow Loads: Median=21
Outflow Loads: Median=14
Percentile
Load (kg)

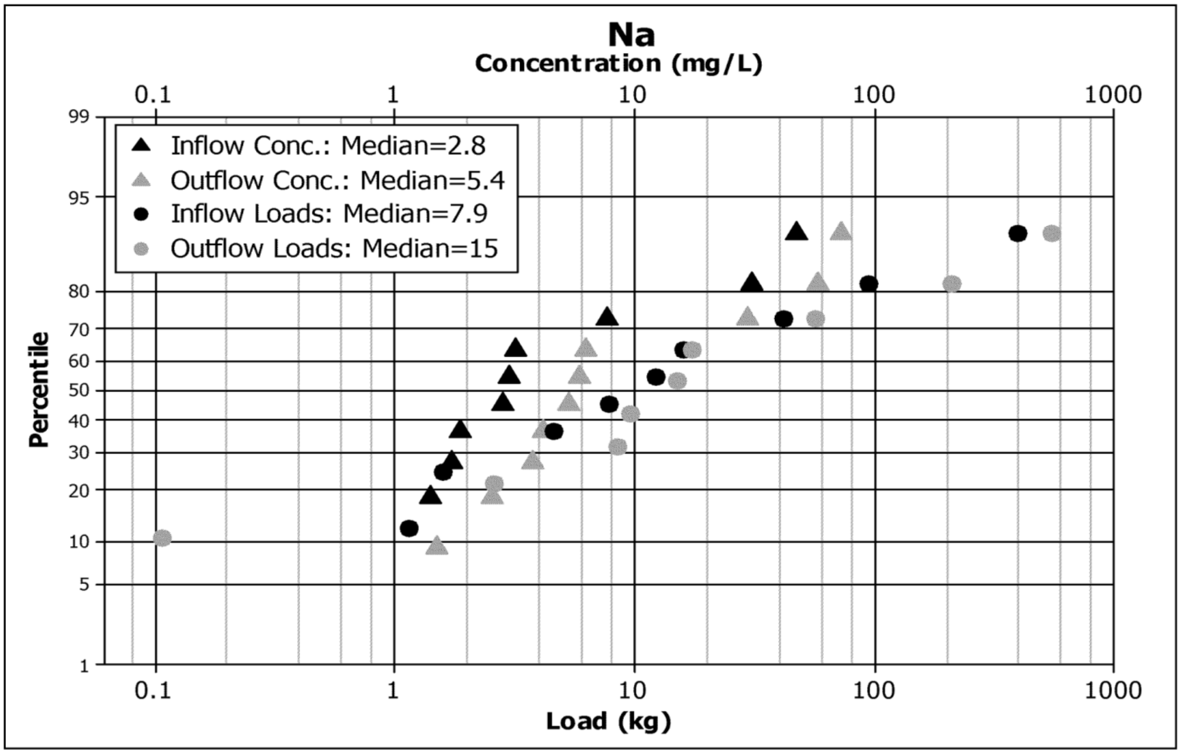

TOC
Concentration (mg/L)
Inflow Conc.: Median=7.6
Outflow Conc.: Median=7.5
Inflow Loads: Median=23
Outflow Loads: Median=15
Percentile
Load (kg)

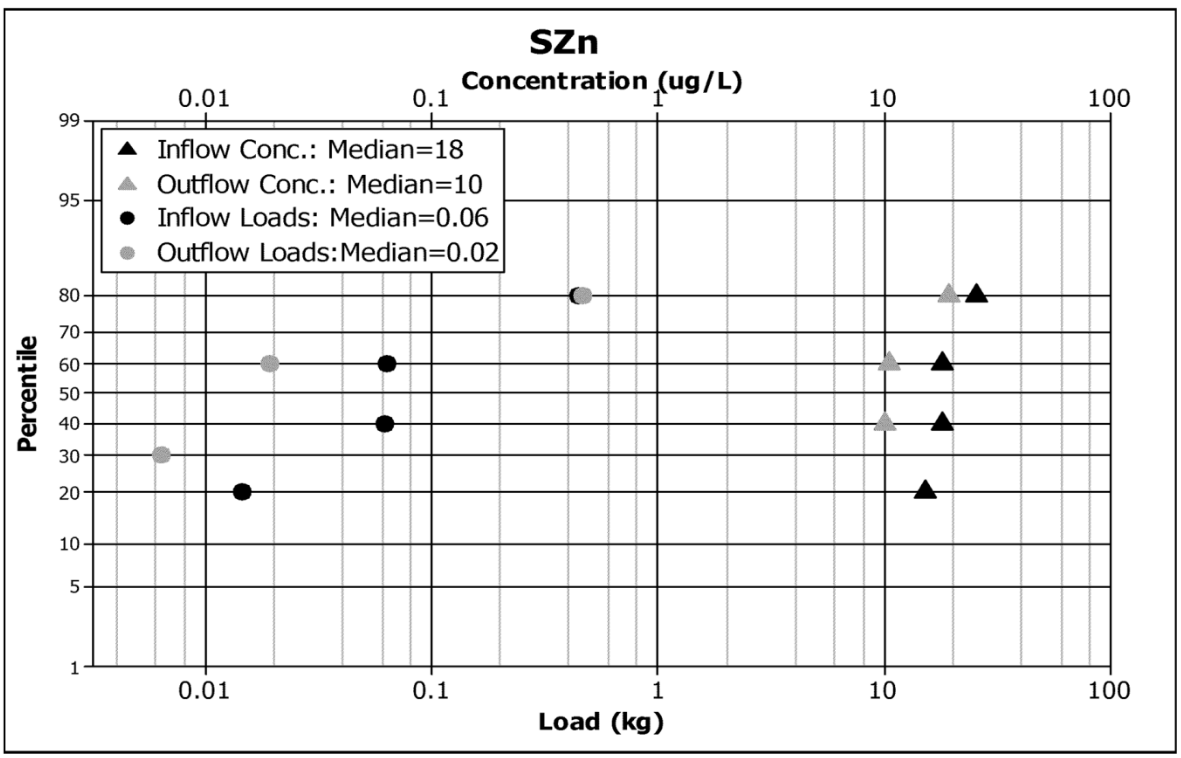

TSS
Concentration (mg/L)
Inflow Conc.: Median=95
Outflow Conc.: Median=29
Inflow Loads: Median=281
Outflow Loads: Median=58
Percentile
Load (kg)


SSC
Concentration (mg/L)
Inflow Conc.: Median=84
Outflow Conc.: Median=27
Inflow Loads: Median=250
Outflow Loads: Median=53
Percentile
Load (kg)


Cl
Concentration (mg/L)
Inflow Conc.: Median=<0.08
Outflow Conc.: Median=9.7
Inflow Loads: Median=<2.6
Outflow Loads: Meidan=31
Percentile
Load (kg)


SCa
Concentration (mg/L)
Inflow Conc.: Median=3.7
Outflow Conc.: Median=6.5
Inflow Loads: Median=8.3
Outflow Loads: Median=12
Percentile
Load (kg)


Na
Concentration (mg/L)
0.1
1
10
100
1000
Inflow Conc.: Median=2.8
Outflow Conc.: Median=5.4
Inflow Loads: Median=7.9
Outflow Loads: Median=15
Percentile
99
95
80
70
60
50
40
30
20
10
5
1
Load (kg)


SZn
Concentration (ug/L)
0.01
0.1
1
10
100
Inflow Conc.: Median=18
Outflow Conc.: Median=10
Inflow Loads: Median=0.06
Outflow Loads:Median=0.02
Percentile
99
95
80
70
60
50
40
30
20
10
5
1
Load (kg)

APPENDIX D: Sign Test p-Values

The sign test was run for the concentrations and loads of each analyzed constituent. The outflows were subtracted from the inflows for each storm and the differences between the two, Di, were recorded. All positive Dis were assigned a positive sign (+), all negative Dis were assigned a negative sign (-), and all zeros are a tie. The test statistics, S, is the sum of all positive (S^+) or negative (S^-) signs. Ties are not accounted for in the S value. For censored data (ammoniacal-N, Cl^-, CA, NA^+), inflows and outflows exist as an interval range. The lowest and highest possible Di values were calculated for these constituents by subtracting the largest possible outflow from the smallest possible inflow and the smallest possible outflow from the largest possible inflow. The Di values are presented as the range between these two values. If both ends of the Di range are positive, the inflow range is greater than outflow range and the sign is +; If both ends of the Di range are negative, the outflow range is greater than inflow range and the sign is -; If the low end of the Di range is negative and the high end positive, the inflow and outflow ranges overlap. The overlapping data were re-censored at the highest reporting limit (Helsel, 2012) and thus treated as ties.

The S statistic is used to calculate the one sided p-value. For constituents with no tied data, the p-value is calculated from the binomial distribution of exact probabilities found at www.vassarstats.net, accessed October 27, 2013. For constituents with ties, the Minitab macro from Helsel (2012) was used to compute Fong et al's (2003) modified one-sided p-value. Without this modification, ties cause the p-value to be lower than it should.

All Analyzed Storms

Constituent	Concentration				Load			
	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
OP	0.12	0.02	0.10	+	0.67	0.13	0.54	+
	0.064	0.02	0.04	+	0.35	0.08	0.27	+
	0.11	0.02	0.09	+	0.13	0.001	0.13	+
	0.053	0.02	0.03	+	0.48	0.17	0.30	+
	0.083	0.03	0.05	+	0.15	0.04	0.11	+
	0.080	0.08	0.00	Tie	4.3	4.2	0.1	+
	0.034	0.01	0.02	+	0.16	0.04	0.11	+
	0.057	0.03	0.03	+	1.1	0.63	0.4	+
	0.043	0.01	0.03	+	0.29	0.06	0.23	+
	0.038	0.02	0.02	+	0.35	0.21	0.14	+
			S+	9			S+	10
			p-value	0.009			p-value	0.00098
TP	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	0.27	0.20	0.07	+	1.4	1.3	0.1	+
	0.16	0.22	-0.06	-	0.89	0.87	0.02	+
	0.39	0.16	0.23	+	0.48	0.0060	0.48	+
	0.20	0.11	0.09	+	1.8	0.96	0.8	+
	0.24	0.14	0.10	+	0.42	0.20	0.23	+
	0.16	0.14	0.02	+	8.5	7.4	1.1	+
	0.21	0.12	0.09	+	1.7	0.89	0.8	+
	0.14	0.10	0.04	+	0.66	0.43	0.24	+
	0.65	0.21	0.44	+	12	4.4	8	+
	0.18	0.14	0.04	+	1.2	0.90	0.3	+
			S^+	9			S^+	10
			p-value	0.01			p-value	0.00098
Ammoniacal N	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	0.03	0.020	0.01	+	0.17	0.13	0.04	+
	0.008 - 0.017	0.020	-0.01 - 0	-	.043 - .092	0.079	-0.036 - 0.013	Tie
	0.29	0.090	0.20	+	0.36	0.003	0.35	+
	0.025	0.0 - 0.010	0.02 - 0.03	+	0.23	0.0 - 0.087	0.14 - 0.23	+
	0.13	0.020	0.11	+	0.23	0.028	0.20	+
	0.026	0.010	0.02	+	1.4	0.53	0.9	+
	0.17	0.020	0.15	+	1.4	0.15	1.3	+
	0.15	0.040	0.11	+	0.67	0.17	0.50	+
	0.20	0.10	0.10	+	3.7	2.1	1.6	+
	0.19	0.10	0.09	+	1.3	0.65	0.6	+
	0.078	0.090	-0.01	-	0.71	0.92	-0.21	-
			S^+	9			S+	9
			p-value	0.033			p-value	0.023

Constituent	Concentration				Load			
	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
OX-N	0.61	0.11	0.50	+	3.3	0.7	2.6	+
	0.33	0.05	0.28	+	1.8	0.2	1.6	+
	0.61	0.08	0.53	+	0.76	0.00	0.76	+
	0.33	0.15	0.18	+	3.0	1.3	1.7	+
	0.44	0.14	0.30	+	0.80	0.20	0.60	+
	0.29	0.34	-0.05	-	16	18	-2	-
	0.41	0.17	0.24	+	3.4	1.3	2.2	+
	0.39	0.14	0.25	+	1.8	0.6	1.2	+
	0.32	0.24	0.08	+	5.9	5.1	0.8	+
	0.68	0.14	0.54	+	4.7	0.9	3.8	+
	0.27	0.21	0.06	+	2.5	2.2	0.3	+
			S^+	10			S^+	10
			p-value	0.006			p-value	0.006
TN	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	1.8	1.3	0.42	+	9.5	8.6	0.9	+
	1.4	1.4	-0.02	-	7.8	5.7	2.1	+
	3.3	1.2	2.2	+	4.2	0.043	4.1	+
	1.2	0.81	0.41	+	11	7.1	4	+
	1.8	1.1	0.77	+	3.3	1.5	1.8	+
	1.00	0.99	0.01	+	54	52	2	+
	1.4	0.87	0.50	+	11	6.4	5	+
	1.2	0.88	0.29	+	5.3	3.7	1.6	+
	3.7	1.7	2.0	+	69	36	33	+
	1.7	1.2	0.50	+	12	7.8	4	+
	2.8	1.8	0.99	+	25	18	7	+
			S^+	10			S^+	11
			p-value	0.006			p-value	0.00049
COD	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	34	35	-1	-	182	226	-44	-
	48	29	19	+	86	41	45	+
	37	30	6	+	1977	1609	368	+
	35	28	7	+	291	208	82	+
	28	25	3	+	127	106	21	+
	37	37	0	?	681	772	-91	-
	35	35	1	+	242	225	17	+
			S^+	5			S^+	5
			p-value	0.15			p-value	0.23

Constituent	Concentration				Load			
DOC	**Inflow**	**Outflow**	**Di**	**Sign**	**Inflow**	**Outflow**	**Di**	**Sign**
	6.3	6.5	-0.2	-	34	42	-8	-
	8.3	7.6	0.7	+	45	30	15	+
	12	8.5	3.6	+	15	0.32	15	+
	5.5	6	-0.5	-	50	52	-3	-
	7.2	8.1	-0.9	-	13	11	2	+
	11	8.8	1.9	+	580	470	110	+
	6.7	7.5	-0.8	-	56	55	1	+
	5.1	6.2	-1.1	-	23	26	-3	-
	6.7	6.7	0.0	Tie	130	140	-10	-
	6.0	5.6	0.4	+	41	36	5	+
	6.2	4.7	1.5	+	57	48	8	+
			S^+	**5**			**S^+**	**7**
			p-value	**0.5**			**p-value**	**0.27**
TOC	**Inflow**	**Outflow**	**Di**	**Sign**	**Inflow**	**Outflow**	**Di**	**Sign**
	6.5	7.6	-1.1	-	35	49	-14	-
	8.4	8.3	0.1	+	46	33	13	+
	12	10	3	+	16	0.36	15	+
	5.9	6.6	-0.7	-	53	58	-4	-
	7.8	8.1	-0.3	-	14	11	3	+
	11	10	2	+	610	508	102	+
	7.6	7.8	-0.2	-	63	58	6	+
	5.6	6.7	-1.1	-	25	29	-3	-
	8.0	8.1	-0.1	-	149	171	-22	-
	6.1	6.1	0.0	Tie	42	39	3	+
	6.7	5.5	1.2	+	61	56	5	+
			S^-	**6**			**S^+**	**7**
			p-value	**0.5**			**p-value**	**0.27**
TSS	**Inflow**	**Outflow**	**Di**	**Sign**	**Inflow**	**Outflow**	**Di**	**Sign**
	84	48	37	+	457	309	148	+
	40	23	17	+	219	92	127	+
	37	41	-5	-	46	1.6	44	+
	75	10	64	+	679	91	588	+
	111	14	97	+	200	20	180	+
	208	14	195	+	11209	720	10489	+
	57	18	38	+	474	136	338	+
	37	11	27	+	170	45	125	+
	414	121	293	+	7708	2552	5156	+
	79	39	40	+	540	251	289	+
	463	174	289	+	4218	1787	2432	+
			S^+	**10**			**S^+**	**11**
			p-value	**0.006**			**p-value**	**0.00049**

Constituent	Concentration				Load			
	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
SSC	105	58	47	+	568	374	194	+
	39	19	20	+	213	77	136	+
	45	12	33	+	57	0.46	56	+
	78	11	67	+	707	93	614	+
	124	19	105	+	223	27	196	+
	21	14	7	+	1131	748	383	+
	64	17	47	+	539	128	410	+
	38	12	25	+	172	52	120	+
	428	118	310	+	7974	2491	5483	+
	85	41	44	+	584	267	317	+
	531	176	355	+	4840	1809	3031	+
			S^+	11			S^+	11
			p-value	0.0005			p-value	0.0005
CL	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	0.016 - 4.8	11	-11 - -6	-	0.085 - 26	69	-69 - -43	-
	0.017 - 4.8	8	-7.9 - -31	-	0.091 - 26	31	-31 - -5	-
	0.036 - 4.6	10	-9.6 - -5.1	-	0.045 - 5.8	0.36	-0.32 - 5.4	Tie
	0.016 - 4.8	0 - 5.0	-5 - 4.8	Tie	0.14 - 43	0 - 44	-44 - 43	Tie
	0.025 - 4.6	8	-8.3 - -3.7	-	0.046 - 8.3	12	-12 - -3	-
	5.3 - 5.8	0.0 - 5.0	0.3 - 5.8	+	280 - 310	0.0 - 270	19 - 312	+
	0.015 - 4.8	13	-13 - -8	-	0.12 - 40	98	-98 - -58	-
	10	49	-39	-	44	210	-165	-
	75	94	-19	-	1400	2000	-590	-
	78	130	-49	-	530	820	-289	-
			S-	8			S-	7
			p value	0.044			p value	0.1
CA	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	2.9	6.4	-3.5	-	16	42	-26	-
	3.7	6.4	-2.7	-	20	25	-5	-
	3.6	7.7	-4.1	-	4.5	0.29	4.2	+
	4.0	5.2	-1.1	-	37	45	-8	-
	2.5	6.2	-3.7	-	4.4	8.7	-4.3	-
	5.1	4.3	0.8	+	280	230	50	+
	0.36 - 1.6	3.4	-3.1 - -1.9	-	3 - 13	25	-22 - -12	-
	4.0	7.4	-3.4	-	18	31	-13	-
	6.4	9.3	-2.9	-	120	200	-77	-
	12	14	-2	-	85	89	-4	-
			S^-	9			S-	8
			p-value	0.01			p value	0.055

Constituent	Concentration				Load			
NA	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	1.9	5.9	-4.0	-	10	39	-28	-
	3.2	5.4	-2.2	-	17	21	-4	-
	2.8	6.3	-3.5	-	3.5	0.24	3.3	+
	3.0	3.8	-0.8	-	27	33	-6	-
	0.13 - 1.4	4.2	-4.0 - -2.7	-	0.24 - 2.5	58	-5.6 - -3.3	-
	1.6 - 1.7	0 - 1.5	0.1 - 1.7	+	85 - 93	0 - 79	5 - 93	+
	0.25 - 1.4	2.6	-2.3 - -1.1	-	2 - 12	19	-17 - -7	-
	7.7	30	-22	-	35	130	-90	-
	47	58	-11	-	880	1200	-347	-
	31	72	-41	-	210	470	-256	-
			S^-	**9**			**S^-**	**8**
			p-value	**0.01**			**p-value**	**0.055**
SZN*	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	25	11	15	+	0.14	0.042	0.10	+
	14 - 15	0 - 10	4 - 10	+	0.13 - 0.14	0 - 0.087	0.04 - 0.05	+
	18	0 - 10	8 - 18	+	0.032	0 - 0.051	0.018 - 0.032	+
	18	19	-1	-	0.98	1.0	0	Tie
			S+	**3**			**S+**	**3**
			p-value	**0.31**			**p-value**	**0.23**

***SZn=Soluble Zinc**

Extreme Event Storms (9-26-2012 and 10-29-2012) Excluded

Constituent	Concentration				Load			
OP	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	0.12	0.02	0.10	+	0.67	0.13	0.54	+
	0.06	0.02	0.04	+	0.35	0.079	0.27	+
	0.05	0.02	0.03	+	0.48	0.17	0.30	+
	0.08	0.03	0.05	+	0.15	0.042	0.11	+
	0.03	0.01	0.02	+	0.16	0.043	0.11	+
	0.06	0.03	0.03	+	1.1	0.63	0.4	+
	0.04	0.01	0.03	+	0.29	0.065	0.23	+
	0.04	0.02	0.02	+	0.35	0.21	0.14	+
			S^+	**8**			**S^+**	**8**
			p-value	**0.004**			**p-value**	**0.004**

Consti-tuent	Concentration				Load			
	Inflow		Di	Sign	Inflow	Outflow	Di	Sign
TP	0.27	0.20	0.07	+	1.4	1.3	0.1	+
	0.16	0.22	-0.06	-	0.89	0.87	0.02	+
	0.20	0.11	0.09	+	1.8	0.96	0.8	+
	0.24	0.14	0.10	+	0.42	0.20	0.23	+
	0.21	0.12	0.09	+	1.7	0.89	0.8	+
	0.14	0.10	0.04	+	0.66	0.43	0.24	+
	0.65	0.21	0.44	+	12.1	4.4	7.7	+
	0.18	0.14	0.04	+	1.2	0.90	0.3	+
			S^+	**7**			**S^+**	**8**
			p-value	**0.04**			**p-value**	**0.004**
Amm-oniacal N	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	0.031	0.020	0.011	+	0.17	0.13	0.04	+
	0.008 - 0.017	0.020	-0.012 - - 0.003	-	.043 - .092	0.079	-0.036 - 0.013	Tie
	0.025	0.0 - 0.010	-.015 - 0.025	+	0.23	0.0 - 0.087	0.14 - 0.23	+
	0.13	0.020	0.11	+	0.23	0.028	0.20	+
	0.17	0.020	0.15	+	1.4	0.15	1.3	+
	0.15	0.040	0.11	+	0.67	0.17	0.50	+
	0.20	0.10	0.10	+	3.7	2.1	1.6	+
	0.19	0.10	0.09	+	1.3	0.65	0.6	+
	0.078	0.090	-0.012	-	0.71	0.92	-0.21	-
			S^+	**7**			**S^+**	**7**
			p-value	**0.09**			**p-value**	**0.06**
OX-N		**p-value**	**0.09**		**p-value**	**0.06**	Di	Sign
	0.61	0.11	0.50	+	3.3	0.71	2.6	+
	0.33	0.05	0.28	+	1.8	0.20	1.6	+
	0.33	0.15	0.18	+	3.0	1.3	1.7	+
	0.44	0.14	0.30	+	0.80	0.20	0.60	+
	0.41	0.17	0.24	+	3.4	1.3	2.2	+
	0.39	0.14	0.25	+	1.8	0.60	1.2	+
	0.32	0.24	0.08	+	5.9	5.1	0.8	+
	0.68	0.14	0.54	+	4.7	0.90	3.8	+
	0.27	0.21	0.06	+	2.5	2.2	0.3	+
			S^+	**9**			**S^+**	**9**
			p-value	**0.002**			**p-value**	**0.002**

Constituent	Concentration				Load			
TN	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	1.8	1.3	0.4	+	9.5	8.6	0.9	+
	1.4	1.4	0.0	Tie	7.8	5.7	2.1	+
	1.2	0.8	0.4	+	11	7.1	4	+
	1.8	1.1	0.8	+	3.3	1.5	1.8	+
	1.4	0.9	0.5	+	11	6.4	5	+
	1.2	0.9	0.3	+	5.3	3.7	1.6	+
	3.7	1.7	2.0	+	69	36	33	+
	1.7	1.2	0.5	+	12	7.8	4	+
	2.8	1.8	1.0	+	25	18	7	+
			S^{+}	8			S^{+}	9
			p-value	0.013			p-value	0.002
COD	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	34	35	-1	-	182	226	-44	-
	48	29	19	+	86	41	45	+
	35	28	7	+	291	208	82	+
	28	25	3	+	127	106	21	+
	37	37	0	Tie	681	772	-91	-
	35	35	0	Tie	242	225	17	+
			S^{+}	3			S^{+}	4
			p-value	0.37			p-value	0.34
DOC	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
	6.3	6.5	-0.2	-	34	42	-8	-
	8.3	7.6	0.7	+	45	30	15	+
	5.5	6	-0.5	-	50	52	-3	-
	7.2	8.1	-0.9	-	13	11	2	+
	6.7	7.5	-0.8	-	56	55	1	+
	5.1	6.2	-1.1	-	23	26	-3	-
	6.7	6.7	0.0	+	125	141	-16	-
	6.0	5.6	0.4	+	41	36	5	+
	6.2	4.7	1.5	+	57	48	8	+
			S^{-}	5			S^{+}	5
			p-value	0.5			p-value	0.5

Constituent	Concentration				Load			
	Inflow	**Outflow**	**Di**	**Sign**	**Inflow**	**Outflow**	**Di**	**Sign**
TOC	6.5	7.6	-1.1	-	35	49	-14	-
	8.4	8.3	0.1	+	46	33	13	+
	5.9	6.6	-0.7	-	53	58	-4	-
	7.8	8.1	-0.3	-	14	11	3	+
	7.6	7.8	-0.2	-	63	58	6	+
	5.6	6.7	-1.1	-	25	29	-3	-
	8.0	8.1	-0.1	-	149	171	-22	-
	6.1	6.1	0.0	Tie	42	39	3	+
	6.7	5.5	1.2	+	61	56	5	+
			S^-	**6**			**S^+**	**5**
			p-value	**0.17**			**p-value**	**0.5**
	Inflow	**Outflow**	**Di**	**Sign**	**Inflow**	**Outflow**	**Di**	**Sign**
TSS	84	48	37	+	457	309	148	+
	40	23	17	+	219	92	127	+
	75	10	64	+	679	91	588	+
	111	14	97	+	200	20	180	+
	57	18	38	+	474	136	338	+
	37	11	27	+	170	45	125	+
	414	121	293	+	7708	2552	5156	+
	79	39	40	+	540	251	289	+
	463	174	289	+	4218	1787	2432	+
			S^+	**9**			**S^+**	**9**
			p-value	**0.002**			**p-value**	**0.002**
	Inflow	**Outflow**	**Di**	**Sign**	**Inflow**	**Outflow**	**Di**	**Sign**
SSC	105	58	47	+	568	374	194	+
	39	19	20	+	213	77	136	+
	78	11	67	+	707	93	614	+
	124	19	105	+	223	27	196	+
	64	17	47	+	539	128	410	+
	38	12	25	+	172	52	120	+
	428	118	310	+	7974	2491	5483	+
	85	41	44	+	584	267	317	+
	531	176	355	+	4840	1809	3031	+
			S^+	**9**			**S^+**	**9**
			p-value	**0.002**			**p-value**	**0.002**

Constituent	Concentration				Load			
	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
CL	0.016 - 4.8	11	-11 - -6	-	0.085 - 26	69	-69 - -43	-
	0.017 - 4.8	8	-7.9 - -3.1	-	0.091 - 26	31	-31 - -5	-
	0.016 - 4.8	0 - 5.0	-5.0 - 4.8	Tie	0.14 - 43	0 - 43	-44 - 43	Tie
	0.025 - 4.6	8	-8.3 - -3.7	-	0.046 - 8.3	12	-12 - -3	-
	0.015 - 4.8	13	-13 - -8	-	0.12 - 40	0.0 - 270	-98 - -58	-
	9.5	49	-39	-	44	98	-137	-
	75	94	-19	-	1400	210	-229	-
	78	130	-49	-	530	2000	-2049	-
			S^-	**7**			**S^-**	**7**
			p-value	**0.028**			**p-value**	**0.035**
	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
SCA	2.9	6.4	-3.5	-	16	42	-26	-
	3.7	6.4	-2.7	-	20	25	-5	-
	4.0	5.2	-1.1	-	37	45	-8	-
	2.5	6.2	-3.7	-	4.4	8.7	-4.3	-
	0.36 - 1.6	3.4	-3.1 - -1.9	-	3 - 13	25	-22 - -12	-
	4.0	7.4	-3.4	-	18	31	-13	-
	6.4	9.3	-2.9	-	120	200	-77	-
	12	14	-2.0	-	85	89	-4	-
			S^-	**8**			**S^+**	**8**
			p-value	**0.004**			**p-value**	**0.004**
	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
SNA	1.9	5.9	-4.0	-	10	39	-28	-
	3.2	5.4	-2.2	-	17	21	-4	-
	3.0	3.8	-0.8	-	27	33	-6	-
	0.13 - 1.4	4.2	-4.0 - -2.7	-	0.24 - 2.5	58	-5.6 - -3.3	-
	0.25 - 1.4	2.6	-2.3 - -1.1	-	2 - 12	19	-17 - -7	-
	7.7	30	-22	-	35	130	-90	-
	47	58	-11	-	880	1200	-347	-
	31	72	-41	-	210	470	-256	-
			S-	**8**			**S-**	**8**
			p-value	**0.004**			**p-value**	**0.004**
	Inflow	Outflow	Di	Sign	Inflow	Outflow	Di	Sign
SZN	25	11	15.0	+	0.14	0.042	0	+
	14 - 15	0 - 10	4 - 15	+	0.13 - 0.14	0 - 0.087	0.04 - 0.14	+
	18	0 - 18	8 - 18	+	0.032	0 - 0.051	0.018 - 0.032	+
			S+	**3**			**S^+**	**3**
			p-value	**0.125**			**p-value**	**0.13**

APPENDIX E: FAIR USE ANALYSIS

Virginia Tech ETD Fair Use Analysis Results

This is not a replacement for professional legal advice but an effort to assist you in making a sound decision.

Name: Daniel Schwartz

Description of item under review for fair use: Figure 14.1. Wet Pond Design Schematics - Plan View. Source: Virginia DEQ Stormwater Design Specification No. 14. Available at http://vwrrc.vt.edu/swc/. Image re-used in this thesis as Figure 2.7-2: Wet Pond Typical Plan View.

Report generated on: 05-11-2014 at : 21:48:02

Based on the information you provided:

Factor 1

Your consideration of the purpose and character of your use of the copyright work weighs: in favor of fair use

Factor 2

Your consideration of the nature of the copyrighted work you used weighs: against fair use

Factor 3

Your consideration of the amount and substantiality of your use of the copyrighted work weighs: in favor of fair use

Factor 4

Your consideration of the effect or potential effect on the market after your use of the copyrighted work weighs: in favor of fair use

Based on the information you provided, your use of the copyrighted work weighs: ***in favor of fair use***

University of Minnesota Fair Use Analysis

Schwartz, Daniel

From:	Apache <apache@side02.oit.umn.edu>
Sent:	Tuesday, April 08, 2014 6:52 PM
To:	Schwartz, Daniel
Subject:	Fair Use Analysis Summary

Work considered: Figure 14.1: Wet Pond Design Schematics. From Virginia DEQ Stormwater Design Specification No.14: Wet Ponds. Version 1.9, March 2011. Available at http://vwrrc.vt.edu/swc/documents/2013/1DEQ%20BMP%20Spec%20No%2014_WET%20PONDS_Final%20Draft_v1-9_03012011.pdf

Here's your "Thinking Through Fair Use" results! As a reminder, these results have not been analyzed or processed in any way, except to format the input you provided into one document. These results don't claim to tell you whether a proposed use is fair or not, and do not constitute legal advice. For formal legal advice, please contact an attorney.

You thought that purpose was <i>strongly favorable</i> towards fair use.

You highlighted these elements as relevant to your proposed use:
Favors Fair Use
* Educational, scholarly, and research uses, and/or news reporting

Weighs Against Fair Use

You thought that nature was <i>strongly favorable</i> towards fair use.

You highlighted these elements as relevant to your proposed use:

Favors Fair Use
* Factual or non-fiction source

Weighs Against Fair Use

You thought that amount was <i>strongly favorable</i> towards fair use.

You highlighted these elements as relevant to your proposed use:

Favors Fair Use
* Only as much as absolutely necessary for a favored purpose

Weighs Against Fair Use

You thought that market was <i>strongly favorable</i> towards fair use.

You highlighted these elements as relevant to your proposed use:

Favors Fair Use
* No impact on market for original work

Weighs Against Fair Use

When looking over your results, remember that no single factor is decisive of fair use. There also may be other relevant considerations, specific to your use, that do not appear in the results from this general-purpose tool. Many considerations are relevant, and only by looking at the whole picture, across all the issues, can you make a reasonable guess about whether your use is fair or not!