THE INFLUENCE OF PAST AND FUTURE URBANIZATION ON WATERSHED NITROGEN EXPORT AND HYRDROLOGY DYNAMICS IN TWO MID-ATLANTIC WATERSHEDS IN FAIRFAX, VIRGINIA

by

Ryan J. Albert
A Dissertation
Submitted to the
Graduate Faculty
of
George Mason University
In Partial fulfillment of
The requirements for the Degree
of
Doctorate of Philosophy
Environmental Science and Public Policy

Committee:

	Dr. R. Chris Jones, Dissertation Director
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Date: 4/11/2007	Spring Semester 2007 George Mason University Fairfax Virginia

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The Influence of Past and Future Urbanization on Watershed Nitrogen Export and Hydrology Dynamics in Two Mid-Atlantic Watersheds in Fairfax, Virginia

A dissertation submitted in partial fulfillment of the requirements for the degree of Doctor of Philosophy at George Mason University

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Abstract

THE INFLUENCE OF PAST AND FUTURE URBANIZATION ON WATERSHED NITROGEN EXPORT AND HYRDROLOGY DYNAMICS IN TWO MID-ATLANTIC WATERSHEDS IN FAIRFAX, VIRGINIA

Ryan J. Albert, PhD

George Mason University, 2007

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This study examines urban land use change and its impact on watershed hydrology and nutrient loading in the Accotink and Pohick watersheds in Fairfax County, Virginia. The dissertation explored the amount of urbanization in the watersheds over the past 30 years and the impact of that urbanization on nitrogen loadings and stream hydrology. Further, it examined different projections of future urban development in the watersheds and how urbanization may affect nitrogen loadings and hydrologic changes. It was hypothesized that deterioration in hydrologic conditions and increases nitrogen loadings would be notable.

Land use was estimated from 1975 to 2004 using a combination of remote sensing and demographic data, which was given the name of the Household Method. Adjusted land use projections from an existing study generated by the SLEUTH model and

projections using the household method were used to estimate future land use. These land use estimates were input into several analytical tools, including the hydrologic component of HSPF, L-THIA, and export coefficient-based approaches. Water quality data collected by George Mason University and the Norman M. Cole Jr. Pollution Control Plant are available at four sites from 1984-1992. Data were also collected on 24 occasions in 2005 at four sites in the watersheds as part of this dissertation. Nitrate-N, ammonia-N, total phosphorus and soluble reactive phosphorus were compared between watersheds using statistical techniques and a multiple regression loading model (LOADEST); focus was given to nitrogen. Physical parameters, including conductivity, dissolved oxygen, pH, and temperature were also analyzed.

Modeled and observed results indicate that significant changes correlated with increased urbanization have occurred to the hydrology of these watersheds. Furthermore, without implementation of effective Best Management Practices (BMPs), significant alterations in hydrology will continue into the future. Nitrogen loadings have also increased and will likely continue to increase without effective BMPs, although the increases in nitrogen loading do not pose a significant a risk to the streams themselves. However, these increased nitrogen loadings may pose a potential risk to the Chesapeake Bay ecosystem.

1. Introduction

The Chesapeake Bay is the largest estuary in the United States. The watershed, spanning six states and Washington D.C., contains one of the most heavily urbanized areas in the United States. Fast paced urbanization has resulted in significant changes for many of the Bay's formerly rural or forested watersheds. These land use changes have had marked impacts on numerous watersheds, everything from small first order creeks to the main channels of the Bay. During this period of increasing population, housing and demographic patterns have changed and average suburban population density per developed acre has decreased, resulting in greater consumption of land for urban and suburban uses. Cities continue to develop outward, consuming land for urban purposes at a faster rate than the rising urbanized population would seem to indicate. Nationwide, between 1950 and 1990 metropolitan areas have almost tripled in size (Dwyer *et al.*, 2000). As of the year 2000, urbanized areas were found on 3.5 percent of total land area in the lower 48 states and account for 75% of the population. In the Chesapeake Bay watershed, developed land increased by 39 percent from 1986 to 2000 (Jantz *et al.*, 2004).

Fairfax County, Virginia is situated to the west of Washington DC. The county is located in the Potomac River Basin and has numerous impacted watersheds typical of urbanized and urbanizing watersheds draining to the bay. Fairfax has already been largely developed, and most remaining new development will occur in the few remaining forested areas or in the less populated southwest part of the county. As of 1996, of Fairfax's 1025 km^2 land area, 67.0% was urban, 21.9% was forested, and 5.8% was agriculture (Dwyer *et al.*, 2000). The population of the county was 964,712 as of the year 2000, an increase of 112% from 454,275 in the year 1970. The estimated population of the county increased to 1,022,298 in 2004, an increase of 6% over 2000 levels in the relatively short time period (Fairfax County, 1975-2004)

Natural and minimally developed land in Fairfax is currently owned by private landowners, or contained in federal parks (Great Falls National Park, Mason Neck Wildlife Refuge), county (Hideway, Burke Lake, Accotink Stream Valley Park), state (Mason Neck) and regional parks (Bull Run, Hemlock Overlook). However, developmental pressure is placed on remaining undeveloped lands because of increased housing prices and demand in Northern Virginia, and many lands with existing structures are being redeveloped. The projected population of Fairfax County is 1,182,000 for the year 2025 (Fairfax County, 1975-2004). In order to accommodate this growth, either population density will have to increase in existing structures, which goes against the historical and national trend, or new developments must be built to accommodate the new residents. This development must occur on the remaining private natural habitat or where existing structures currently stand. Hence, the county will continue to urbanize by infill development and intensification of land use. Additionally, these projections predate the military base realignment decisions to relocate thousands of workers to Fort Belvior in the South of the County: they therefore likely underestimate future county development.

As the county continues to urbanize, we must reconcile that many of our current urban land use practices are detrimental to watersheds and water quality, a result of increasing forest fragmentation, hydrological modifications, increased non-point source pollution, and air and thermal pollution. McElfish and Wilson (2000) state that when forest cover drops to below 75 percent, watersheds sustain some damage. Schueler (1994) notes that damage occurs in watersheds when impervious cover exceeds 10%. Urbanized areas are more likely to have more intense runoff episodes and, consequently, more polluted water bodies. Imperviousness increases in urbanized areas, which leads to higher levels of peak runoff with resulting erosion and habitat degradation. Furthermore, urban runoff often contains pollutants from industry, atmospheric deposition, and automobiles. Documented increases in loadings from heavy metals, phosphorus, nitrogen, sediment, and pathogens occur in urban and suburban watersheds (Jones & Holmes, 1985). The United States Environmental Protection Agency (EPA) (1997a) has listed pollution from sources other than traditional point sources, which include runoff from urban and suburban areas, to be the most significant source of contamination to the country’s water bodies. In short, stormwater from urban areas can increase runoff volume, reduce groundwater recharge, reduce water quality, reduce habitat quality, decrease base flow, and lead to increase channel erosion and widening.

Two watersheds in Fairfax County, Accotink and Pohick, have undergone significant development and continue to be developed and redeveloped (Figure 1.1). The watersheds are located in the southern half of the county. Accotink, the watershed closest to DC, experienced peak urbanization from the 1950s to the 1990s. The Pohick watershed has undergone peak development from the 1980s to the present. Both streams flow into Gunston Cove, a small tidal embayment on the Potomac River. Both watersheds are currently considerably urbanized, with the remaining undeveloped land occurring in the southernmost parts of the watershed. Accotink, in particular, is undergoing considerable in-fill development and redevelopment in the northern reaches and headwaters of the watershed. Both streams are considered degraded (Fairfax County, 2001) and parts of each stream are listed as not supporting their designated uses in the Virginia 303(d)/305(b) consolidated report (Virginia Department of Environmental Quality, 2006).

County officials must consider three primary issues regarding water quality and development in these watersheds. The first issue, and perhaps most pressing in the case of these two streams, is pollutant loadings to the Chesapeake Bay. The major pollutant of concern for the mesohaline stretches of the bay (the “main” Bay) is nitrogen. The bay has undergone significant eutrophication. In the past, the nutrient loadings contributing to this eutrophication could be attributed to traditional point sources such as sewage treatment plants, known as Publicly Owned Treatment Works (POTW) and agriculture. Point source loadings have decreased with the successful implementation of the National Pollution Discharge Elimination System (NPDES) program. However, as urbanization in the bay has increased, so has loading from urban stormwater, and these increases threaten to offset gains made in POTW reductions. For example, in the Gunston Cove watershed (Accotink and Pohick), increased nitrogen loadings from urban areas are estimated to have offset approximately 9.3% to 13.6% of the decreases in nitrogen loading from the

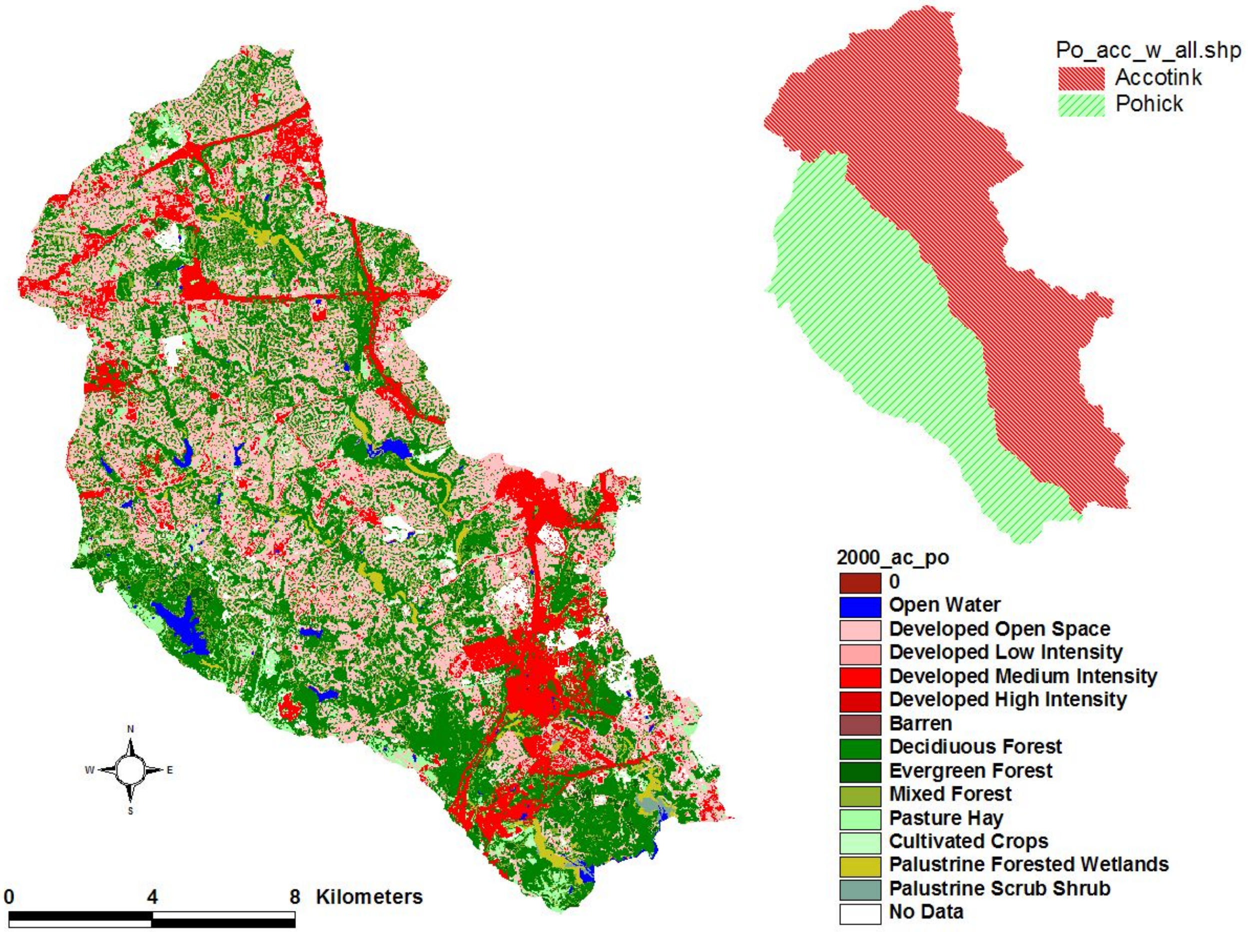

Figure 1.1. Land use in the year 2000 in the Accotink (red 45 degree slant) and Pohick (green 225 degree slant) watersheds, located in Fairfax County, Virginia. Land use data come from the Multi-Resolution Land Characteristics Consortium (MRLC) dataset and have a 30 meter resolution.

Noman Cole facility (discussed in detail in chapter 4). If this estimate is accurate, and the same patterns hold true across other urbanizing watersheds, then the cumulative impact of increasing nutrient loadings from urban area could be significant.

The second concern to county officials is the impact of peak flow increases to watersheds and higher total annual runoff volume. Whereas nutrient loading and eutrophication are major issues facing the bay, they are not the gravest concerns for smaller streams. Frequent high flows from watersheds with significant impervious cover and limited Best Management Practice (BMP) coverage has led to scoured streams and embedded stream bottoms. Erosion in the streams is severe, benthic communities are heavily impacted, sediment loadings to downstream waters are great, and riparian systems suffer due to disconnectivity between the streams and the floodplains.

The third major issue that confronts both the stream and the upper reaches of downstream receiving waters is the loading of toxic and bioaccummulative compounds. For example, coal tar sealed parking lots have recently been shown to discharge high quantities of polycyclic aromatic hydrocarbons (PAHs). Polychlorinated biphenyls (PCBs), pesticides, mercury, and other heavy metals also cause problems in many urban streams. These toxics impair aquatic communities and pose threats for terrestrial species, human, and domestic animal populations.

In January of 2001, Fairfax County published its baseline study of its Stream Protection Strategy (Fairfax County, 2001). Several sites within the two watersheds were sampled. In Accotink, all of the sites scored either poor (6 of 12) or very poor (6 of 12) on the site condition index. Pohick's watersheds had only three sites that scored either poor (2of 11) or very poor (1 of 11), with the rest scoring Fair (4 of 11), Good (3 of 11) or Excellent (1 of 11). The baseline study was designed, in part, to decide how to allocate resources to stream protection and where to focus stream protection efforts. The county designated watershed land area with one of three designations: Watershed Protection Areas, Watershed Restoration Level I Areas, and Watershed Restoration Level II Areas. Watershed Protection Areas are designated to "preserve biological integrity by taking active measures to identify and protect, as much as possible, the conditions for current high quality rating of these streams." Watershed Level I areas are designated to "re-establish healthy biological communities by taking active measures to identify and remedy causes of stream degradation." Watershed Restoration Level II management goals are to "maintain areas to prevent further degradation and to take active measures to improve water quality to comply with Chesapeake Bay Initiatives, Total Maximum Daily Load (TMDL) regulations, and all other existing water quality standards." The county designated the entire Accotink watershed as a Watershed Restoration Level II Area. In contrast, Pohick has received all three designations, with most of the eastern and northern most portions of the watershed as watershed Restoration Level II Area. Hence, county officials intend to protect and restore parts of Pohick. However, their goals for Accotink and much of Pohick are to simply to prevent further degradation to the stream itself and to comply with necessary Chesapeake Bay agreements to reduce the county's impacts on the bay.

Though policy makers and planners understand the general concepts that lead to water quality degradation, land use planning often ignores the exact impacts of urbanization on specific watersheds and cumulative impacts of development on watersheds. Ideally, decision makers should have the best available information in order to make value-based decisions that meet the needs of their communities and other stakeholders. Policy makers need detailed analyses of the impact that projected land use changes will have on local water quality to better evaluate trade-offs, or set additional requirements for development and redevelopment projects to maintain aquatic resources. They will be confronted with decisions such as whether to promote infill development and smart growth, more diffuse ‘conservation development’ patterns, or both. Understanding how land use patterns are likely to change will also help in determining how stringent the requirements for the protection of water quality must be, what costs are acceptable, and what strategies they may want to pursue. Forecasts based on more than estimates of population growth and coupled with tools that approximate changes in water quality could provide such analysis and serve as a valuable tool in land use planning and water quality protection. A study that attempts to integrate coarse quantification of forecasted impacts of urbanization on water quality and watershed dynamics is necessary. Specifically, such a study could serve to aid regional planners in the Pohick and Accotink watersheds. This study is particularly timely with Fairfax County watershed managers starting the process of completing watershed plans for Accotink in Pohick in 2007. With projections of increased loading, planners can identify what level of protective steps must be taken if they are to maintain or improve upon current conditions. Furthermore, techniques learned in this study might be applicable in other watersheds.

This dissertation focused on two of the listed watershed and water quality concerns: nutrient loading to the bay and changes in runoff quantity on the streams themselves. A detailed study of current and historical conditions in the watersheds estimated relative loading of nutrients from both watersheds in relation to land use. The study also looked at key flow data from a USGS gauge station on Accotink, other studies on the hydrology of the area, and key indicators associated with increases in peak flow and stream scouring. A wide variety of both simple and more complex modeling tools were used to characterize past and present watershed conditions. Furthermore, projections from land use models were used in combination with simple water quality models to estimate future loadings of nutrients that the watersheds are projected to discharge to Gunston Cove, the portion of the Chesapeake Bay into which these two watersheds discharge. Additionally, both a simple and a more complex hydrologic model were used to characterize changes to watershed hydrology associated with land use change. Lastly, the dissertation discussed the implications of the projections of these water quality models on nutrient loading to the bay and runoff quantity to the streams themselves and how this impacts Fairfax County’s commitments to the Chesapeake Bay Agreement and their goals of preventing further degradation in Accotink and Pohick.

We know that continuing urbanization without effective BMPs will lead to additional degradation of both Accotink and Pohick creeks and increased loadings to the Potomac River and Chesapeake Bay. This dissertation has sought to quantify the problem so we can identify the scale and scope of solutions needed to best confront the issues we are facing. Often changes brought on by urbanization are either irreversible or very expensive to reverse: hence, knowing what changes need to be made could help protect the remaining quality of these aquatic ecosystems.

The questions this dissertation asks are:

1. How much have the Accotink and Pohick watersheds urbanized over the last 30 years? (chapter 3)
2. What has been the impact of urbanization on the Accotink and Pohick watersheds in the last 25 years, specifically on flow and nitrogen? (Chapters 3 & 4)
3. How urban are these watersheds likely to become with future development? (Chapter 5)
4. What will this urbanization mean for future water quality? What changes need to be made to keep conditions at approximately their current level (to not allow degradation)? What changes are necessary to improve water quality? (Chapters 5 & 6)
5. Are these changes technically feasible with current approaches? If so, what level of efficiency needs to be obtained? (Chapter 6)
6. What are the specific threats faced by each aquatic ecosystem? How do these specific threats integrate with county watershed managers' goals? (chapter 6)

These questions are important because:

1. Fairfax County is spending considerable money on protecting aquatic resources, in large part in response to the Chesapeake Bay Agreements and their NPDES permit requirements. The money spent needs to target the most relevant problems for the county's watersheds and its downstream receiving waters. The county must also consider antidegredation[1] requirements, water quality standards, and its impaired waters and how it is going to meet requirements relevant to these issues.
2. Fairfax County has identified preventing the continued degradation of Accotink and Pohick as goals in their watershed management plan (Fairfax County, 2001).
3. Pohick and Accotink discharge into Gunston Cove, eventually flowing into the Potomac River, and the Chesapeake Bay. Any increase in pollutant loadings from stormwater sources have direct ecological and economic implications for these waterbodies.
4. Despite their current degraded state, both watersheds support a somewhat diverse assortment of benthic macroinvertebrates and vertebrates and provide recreational opportunities for area residents. Continued degradation may result in eliminating many of these tolerant species and limit recreational use.

[1] Regulations promulgated as required by the Clean Water Act state that states (and subsequently permittees such as Fairfax County) must minimize any lowering of quality of waters that currently meet or exceed water quality standards. Though this requirement is not commonly applied in stormwater, numerous environmental groups have noted that these requirements are applicable to stormwater and any new stormwater discharger must not cause backsliding or degradation of water quality standards.

The hypothesis of this dissertation is: as the Accotink and Pohick watersheds have urbanized, noticeable deterioration in water quality and hydrologic conditions was observable. Modeling, monitoring, and statistical approaches will be used to examine historical and current water quality and quantity data. This hypothesis would be supported if water quality conditions are found to be comparable between the Accotink and Pohick sites as urbanization in Pohick approached the levels present in Accotink. The alternative hypothesis is: with increasing implementation of BMPs since 1993, continued urbanization resulted in minimal degradation of the Accotink and Pohick watersheds and the downstream loading of nitrogen did not increase significantly. Supporting evidence for this hypothesis would be if the models and statistical analyses show that there has been no significant increase in loadings to either Accotink or Pohick after significant Best Management Practice (BMP) implementation and select watershed retrofits (starting in earnest around 1993). Furthermore, increased BMP implementation in the newer developments of the Pohick watershed would have kept loadings relatively lower for Pohick than for Accotink. In exploring these hypotheses, this dissertation has examined historical records of flow, nitrate-N, ammonia-N, total phosphorous, soluble reactive phosphorus, and total suspended solids.

Assuming water quality and hydrology continue to be degraded with increasing urbanization, land use forecasts plugged into water quality and hydrology models should estimate continued alteration in the watersheds. These models would show increased annual flow, increased peak flow, decreased base flow, and increased nitrate-N loading to Gunston Cove as a result of development. However, if there is increasing effectiveness of BMP implementation over the last 14 years and into the future, continued urbanization will result in little to no net degradation of water quality in the Pohick and Accotink watersheds.

This dissertation uses output from land use models with modeling tools to develop specific forecasts of the impact of urbanization on the hydrology of the Accotink and Pohick watersheds. It also examines nitrate-N loadings into Accotink and Pohick creeks, and consequently the implications for their loading to the bay. The dissertation then discusses the necessary steps to maintain current (2005) hydrologic conditions in the Accotink and Pohick streams themselves and to maintain (or reduce) their loading to the bay. In short, the goals of this dissertation are to obtain a solid understanding of current conditions and sources of water quality degradation in these watersheds, to show future estimations of water quality and stream conditions based on current trends, and to provide analysis that could be used as a tool for policy makers to experiment with policy decisions.

2. How Urban Runoff Impacts Watersheds, is Quantified, and is Managed.

2.1 Why is the Study of Urbanization and Urban Runoff Relevant?

Urbanization has significant and lasting impacts on watershed integrity and water quality. As regions urbanize, impervious surface area, such as roofs, parking lots, and roads increases. These surfaces exacerbate problems with pollutant transport, groundwater recharge, and altered rates of surface water flow. Impervious levels as low as 10 to 20 percent can lead to stream degradation (Schueler, 1994). Corbett et al. (1997) found that increases in impervious surface area increased runoff volumes linearly and peak flow rates exponentially. Urban land coverage is increasing, and consequently, the overall impact that urban land has on aquatic ecosystems is also increasing dramatically. Urban area coverage in the United States increased from 18.6 to 74 million acres from 1954 2001 (US EPA, 2001a), resulting in both increased numbers of stormwater outfalls and increased volume contribution to existing outfalls.

Since passage of the Clean Water Act in 1972, non-point source pollution and stormwater have become a larger contributor of pollutants to waters in the United States than point source pollution. Despite passage of the 1987 Clean Water Act amendments to regulate stormwater, stormwater from urban areas plays a progressively larger role in pollutant loadings simply because of the increased coverage of urban land. Non-point source pollution including urban runoff is the major cause of impairment of US waters (Baker, 1992; US EPA, 2002b). Runoff episodes are likely to be more intense, nutrient loadings are likely to be higher, and water bodies are likely to be more polluted in urban than in forested watersheds (Wahl *et al.*, 1997). Land use plays a crucial role in determining the intensity of non-point source pollution (Allan & Flecker, 1993). Several studies have been completed in the Fairfax area examining the impacts of land use on various aquatic indicators such as water quality, soil constituents, or biological health (Albert *et al.*, 2005; Arcisewski, 1999; Buchino, 2004; Hogan, 2005; Jones *et al.*, 1996; Maher, 1999; Via, 2003), all of which found that urbanization led to either increased pollutant loadings or decreased biological integrity.

There are multiple changes to aquatic water bodies associated with urbanization: significant urban land use increases runoff volume; reduces groundwater recharge; reduces water quality; increases toxic, nutrient, and sediment loadings; increases temperature; reduces habitat quality; decreases base flow; and leads to increased channel erosion and stream channel widening (Booth, 1990; Davis *et al.*, 2001; Hogan, 2005; Jones & Clark, 1987; Nelson & Booth, 2002; Schueler, 1994). Changes in flow patterns and hydrology as a result of urbanization are among the primary concerns faced by watershed managers. The three pollutants[2] most often associated with impaired waters are nutrients (nitrogen, phosphorus, and others), suspended solids (TSS), and pathogens (US EPA, 2002a). This dissertation primarily examines the impacts of urbanization on

[2] Pollutants as defined by the EPA for regulation under the Clean Water Act. At this time, flow is not considered a pollutant.

flow and two of those pollutants, nitrogen and suspended solids, and their expected variation with changes in land use over time.

As a watershed urbanizes, increased impervious cover leads to higher peak flows and can lead to lower groundwater infiltration. Imperviousness can be defined as the amount of impermeable surfaces present on a landscape and is generally associated with urbanization. Impervious cover includes rooftops, roads, driveways, and even compacted soils or exposed bedrock. Schueler (1994) states that total runoff volume can be 16 times greater from a one acre parking lot than from a 1 acre meadow. Examining a watershed that urbanized in the 1950s and 60s, Sala and Inbar (1992) noted an increase in total runoff, peak discharges, and shortening of lag times between precipitation and runoff. Jennings and Jarnagan (2002) noted a statistically significant increase in flow relative to precipitation which can directly be attributed to increases in impervious cover associated with urbanization. Increased runoff volume and peak flow can contribute to flooding, overly frequent flushing of aquatic ecosystems, and channel scouring.

Nitrogen loading has been shown to be higher from urban watersheds than from forested watersheds. In a study in the Neuse River watershed, Stow et al. (2001) found that total nitrogen loading had increased since the mid-1980s in an urbanizing subwatershed mostly due to increases in nitrate-N concentration. Wahl et al.(1997) found that loadings of dissolved inorganic nitrogen were 34 kilograms per year from a 11 hectare urban watershed versus 14 kilograms per year in a 37 hectare forested watershed. These rates equate to about a 7.5 fold increase in loading per hectare from the urbanized watershed. The difference between nitrate loading from the urban versus forested watershed was even larger with the urban stream discharging 11 times more nitrate. In a study done in the larger Great Lakes watershed, Glandon et al. (1981) found that total nitrogen export rates from an urban watershed were 3.688 kg/ hectare in the urbanized watershed, versus 0.585 kg/hectare in marshland and 5.965 kg/hectare in an agricultural watershed[3]. These increases in nitrogen loading could have significant consequences for downstream areas. For instance, nitrogen can cause significant problems with eutrophication of estuaries and offshore ecosystems (Baker, 1992; Bay *et al.*, 2003; Hollanda *et al.*, 2004; United States Environmental Protection Agency, 1997b). Mallin et al. (2004) noted that concentrations of nitrate-nitrogen as low as 50 micrograms per liter were enough to stimulate algal growth in three saltwater tidal creeks with largely urban and suburban watersheds. They further found that nitrate stimulated growth in the freshwater portions of the tidal creek, though these areas were also occasionally stimulated more by phosphorous inputs.

Higher flows associated with urbanization lead to increased erosion in channel beds and sedimentation. Additionally, construction activities increase loads of sediment to waters. Corbett et al. (1997) found that both runoff volume and sediment yield were over five fold in urban versus forested watersheds in South Carolina. Jones and Clark (1987) noted that Northern Virginia watersheds which were urbanized exported two to three times more sediment than rural sites and up to four times greater than forested sites. Furthermore, sediment export begins at lower rainfall amounts.

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[3] Note that export of nitrogen is higher from agricultural areas than urban areas. This can lead to agricultural areas masking nitrogen export from urbanization in mixed land use watersheds.

Increased suspended solids and resultant sedimentation destroys benthic habitat, which has implications for higher trophic organisms and other ecological processes. The ecological integrity of streams and other aquatic systems is compromised by these changes in sediment loads that occur as a result of changes to watershed dynamics (Nietch et al., 2005). Suspended solids decrease the photic zone by increasing turbidity. This increased turbidity reduces the visibility for living organisms, and can reduce the growth of periphyton and other algae and inhibit the growth and reproduction of aquatic macrophytes (Jha, 2003). Increased sediment loading also increases drinking water costs, impacts navigation, affects recreation, fishing, and agriculture. Once sediment enters a lotic aquatic system, it can impact multiple locations in the stream system since it can often be resuspended and impact downstream habitat. In short, as the country continues to urbanize, we will note changes in local hydrologic processes and may face an increased percentage of impairments caused by resultant erosion and channel scouring.

A few studies have attempted to predict impacts of future land use changes on water quality. Conway and Lathrop (2005) used impervious cover estimations to forecast changes in environmental indicators, including non-point source pollution loading, as a result of urbanization in the Barnegat Bay Watershed. They used a spatially explicit build-out model to develop four possible scenarios (current regulations scenario, down-zoning scenario, large buffer scenario, and open space scenario). The differences in urbanized area at build-out were relatively slight, ranging from 39% urban in the large buffer scenario to 43% percent urban in the current regulations scenario. They estimated that impervious cover would increase from about 8% in 1995 to 12 to 13% at build out.

Certain catchments were estimated to have the larger increases in impervious cover, the biggest one being from 6% to 30%. Based on work by Schuler (1994) and Arnold & Giboons (1996), Conway and Lanthrop determined that noticeable negative effects of urbanization would be noted in water quality based on a 10% impervious cover threshold. However, they did not attempt to quantify the impact of urbanization on loading and only relied on one technique for predicting eventual build-out.

A complication for projecting impacts of future urbanization on watersheds is the role that climate change will play. Polsky et al. (2000) discuss multiple climate change scenarios' impacts on the Mid-Atlantic region, including results from the Hadley Model and the Canadian Climate Center (CCC) model. These models tend to predict future trends that are not consistent with each other for the mid-Atlantic. The author's proceed to discuss the Hadley model because it is more consistent with empirical observations and multiple other models. According to their results, Fairfax County will be both wetter and warmer in 2025-2034 compared the period 1984-1993 (see table 2.1).

Table 2.1 Climatic variation difference comparing 2025-2034 to 1984-1993 in the cell over Fairfax County based on the Hadley Center Climate Change Model.
(adapted from results from Polsky et al. (2000))

Month	Max Temp (°C)	Min Temp (°C)	Precipitation (mm)
Jan	0.1-0.2	0.6-0.7	8-12
July	0.9-1.1	1.5-1.6	22-32

Neff et al. (2000) note that, based on the output of the Hadley model, stream flow at the mouth of the Susquehaqanna River would increase during the winter and spring months comparing simulated flow from 1985-1994 to 2025 to 2034. Use of the CCC model output produced a decrease in spring but an increase in winter stream flow. Scenarios using both models estimate that nitrate-N loadings will increase in winter and late spring, but that they will decrease in late summer. Though Polsky et al. (2000) emphasized the results of the Hadley model, stream flow results using the CCC model output illustrate the difficulties of predicting future hydrologic impact of climate change. Neff et al. (2000) state that though "it would be ideal to predict the impacts of climate change on water resources accurately, no methodology can achieve that goal." Hence, they comment that these efforts produce plausible results, but should not be taken as firm predictions.

Nonetheless, efforts are underway to include a climate assessment tool in the new final release of BASINS 4.0 (Johnson *et al.*, 2006). This feature will allow users to look at the impacts of future climate change scenarios on watersheds based on user input. This feature will allow researchers to look at water quality and hydrological characteristic such as stream flow or nutrient concentration. It was released in late February, 2007 in an updated BASINS 4.0 package. Though this tool was not completed in time to use as part of this dissertation, it increases the likelihood of producing results that account for both future land use change and climate change scenarios, in effect improving the plausibility of any future watershed model results.

As urban land use coverage continues to increase, one should expect that the role urban areas have in influencing watershed dynamics and conditions will increase. Based on numerous studies, empirical evidence suggests that if urban areas are developed in the future as they have been in the past, we can expect numerous newly degraded aquatic ecosystems and continued degradation of our large water bodies such as the Chesapeake Bay. Furthermore, climate change will play a role in future watershed changes, but this role is difficult to quantify. However, there is some hope that future urbanization may not have as large an impact on watershed deterioration. Passage of the National Pollution Discharge Elimination System (NPDES) stormwater regulations should somewhat limit new urban area's role in water quality degradation in those areas where these regulations apply[4]. In order for these regulations to be effective, one must continue to document historic and potential future impacts of urbanization on watershed features, and then identify alternatives that could most successfully prevent degradation caused by needed and expected urbanization.

2.2 How are urban stormwater issues diagnosed?

2.2.1 Monitoring

Though we know that urbanization will have certain impacts on aquatic ecosystems, it is often hard to exactly quantify what those impacts will be. Monitoring is the best option in many cases, as it gives observers an actual snapshot of what is happening in the watershed at that time. Monitoring can be short term or long term, depending upon the goals of the monitoring regime. Trend assessment tends to be a

[4] Unfortunately, areas are rarely regulated when they are undergoing the first development in watersheds due to the implementation of the NPDES program. This is a problem because it is at this point that watersheds will likely be damaged irreversibly.

common goal for monitoring programs, as do characterization of current ambient water quality conditions and meeting regulatory compliance goals (Dixon & Chiswell, 1996). Monitoring can involve measuring chemical components such as toxics, nutrients, or metals; physical components such as temperature, flow, and sediment; biological components such as fish, macroinvertebrates, or plants; or general habitat components using various metrics including stream embeddedness[5] or riparian buffer[6] width. Chemical monitoring can involve grab samples, which involves taking discrete water samples at a particular place and time. It can also involve automated sampling, which takes multiple samples that are either time weighted or flow weighted. Sampling can also involve composite sampling, in which individual water samples from different locations or different times are combined to make one sample for analysis. Monitoring programs can focus primarily on monitoring water quality or can focus more on watershed monitoring. Watershed monitoring involves a more comprehensive approach that also examines watershed conditions such as land use and wetland coverage that impact water quality (US EPA, 2003).

There are problems associated with monitoring alone. Professional monitoring can be very expensive, we often have little baseline data with which to compare results, and it is impossible to monitor all aquatic systems at all sites. Volunteer monitoring is an alternative to professional quality monitoring, but the types of data collected by volunteers can be limited due to equipment, funding, and training requirements. Though

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[5] Embeddedness can be defined as the extent to which the hard stream substrate is buried by fine sediment.
[6] A riparian buffer is the zone immediately adjacent to a water feature that is left in a natural or semi-natural vegetated state.

significant correlations have been found with professional and volunteer biomonitoring data (Fore *et al.*, 2001), many scientists have expressed concerns with the quality of other volunteer monitoring data. For example, Meyer and Albert (2004) found that data collected by high school students could be used as a prescreen for further analysis, but it did not appear robust enough for use in precise analysis. Furthermore, though monitoring can be used to create empirical models as to what we might expect with future urbanization, it is very difficult to prove causation by specific drivers unless we have a well-designed experiment with controls. Though monitoring is an excellent option when available, statistical and deterministic water quality models have been developed to look at watershed processes in those watersheds where extensive and long term monitoring is not feasible.

2.2.2 Modeling

The power of monitoring can be increased dramatically with statistical tests and modeling approaches. These approaches can look at historical conditions in the same watersheds, use reference watersheds, or use regional analyses. Models are most powerful when they are calibrated to local conditions, and they become increasingly powerful the larger the time series or the more macro scale at which a researcher might want to study a watershed and its processes. Models are a tool to aid researchers in better understanding watersheds, how those watersheds function, and how those watersheds impact downstream receiving waters. Hence, water quality modeling is a technique used to represent complex processes in a manageable and interpretable form.

Numerous water quality models have been developed in the last thirty to forty years that examine loadings and water quality in watersheds. All water quality models are designed to focus on specific processes to be represented: many have strengths and all have limitations. Some models are particularly data intensive while others are more easily used by a wide variety of audiences. Models need to be calibrated and should be validated before their output should be considered useful. Calibration entails adjusting the coefficients for parameter values to get better fit between output and known data. Validation involves comparing the model results with an independent data set to see how well the model output compares with real world results. The following discussion first focuses on examples of models used primarily in Agricultural watersheds, followed by discussion of two multi-use models, and then finishes with discussion of three commonly used simple models.

Many water quality models have focused primarily on agriculture and pollution from agricultural sources. ANSWERS (Areal Nonpoint Source Watershed Environment Response Simulation) was a model developed for use as a planning tool to analyze loadings from agricultural runoff (Beasley *et al.*, 1980). It incorporated several elements including a sediment transport component, a hydrologic model, and elements to examine whether water flow is overland, subsurface, or channelized. GLEAMS (Groundwater Loading Effects of Agricultural Management Systems) looks at the impacts of agricultural management practices on the movement of agricultural chemicals in the upper soil layer using pesticide and hydrological components (Leonard *et al.*, 1987).

These agriculturally based models have proven effective in simulating real world conditions of agricultural systems.

Another model used primarily in agricultural watersheds is the Soil and Water Assessment Tool (SWAT). SWAT is a physically based, GIS integrated model designed to simulate loadings from point and nonpoint source pollution (Arnold *et al.*, 1995). SWAT requires spatial land use data, input on weather, soil properties, topography, vegetation, and land management practices in the watershed (Neitsch *et al.*, 2001). The model has the ability to simulate watersheds without extensive data input such as stream flow data and can study the effects of alternative scenarios such as changes in land use or climate. It is a continuous time model meant to look at long term impacts, not single flood events: the model was specifically designed to study long term impacts with a possible decadal running time. SWAT appears to have been used in numerous studies examining water quality and loadings. For example, Tripathi et al. (2003) found that the SWAT model could be used to identify those sub-watersheds the at most need to be protected by best management practices. The model has proven versatile in its uses, for example, using a modified version of the SWAT model known as SWAT-G, Eckhardt and Ulbrich (2003) found that groundwater recharge and streamflow could be reduced by as much as 50% in summer in a central European watershed due to global climate change. Varanou et al. (2002) noted that SWAT did not appear sensitive to many different sub land use classes. In other words, the calibration seemed to be sensitive to groundwater or soil data, but not sensitive to more than three land use classes.

A modeling package called BASINS, a software tool created by the US EPA, (2005a; 2007) incorporates several water quality loading, hydrological, and in stream process models, along with data mining tools and a spatial interface. This software package was designed to be an 'all in one stop' package for modelers. Two of the water quality models available in BASINS 4.0 are PLOAD (Pollutant Loading Application) and HSPF (Hydrologic Simulation Program Fortran) (US EPA, 2007). Both of these models were used for the analysis component of this dissertation. Previous versions of BASINS included the Soil and Water Assessment Tool (SWAT) (US EPA, 2005a), although it is not included in BASINS 4.0 at this time.

An example of a complex deterministic model that could be used for both urban or agricultural systems is the Hydrologic Simulation Program Fortran (HSPF) (Bicknell *et al.*, 1997). HSPF uses rainfall data, climatic records, and land use data to estimate hydrographs and pollutant loadings to streams. In addition to simulating hydrographs, the model can be used to estimate nutrient loadings, sediment, toxic pollutants and metals, and pathogens such as fecal coliform. The HSPF modeling framework contains numerous modules that operate in a hierarchical structure. These modules include functions designed to best simulate hydrology, nutrients, toxics, sediment and other water quality processes. HSPF can be used to examine point and nonpoint source loadings of pollutants and to explore what-if scenarios. As a result, HSPF is a good platform to use to examine impacts of land change. It is one of the most commonly used models in Total Maximum Daily Load (TMDL) development, and can give results at a very short time scale. Since HSPF has been included in EPA's BASINS system (US EPA, 2005a; 2007), it is considerably easier for users to track down some of the necessary data inputs the model requires.

Numerous studies have used HSPF to simulate watershed conditions. Bicknell et al. (2006) are using HSPF to model nearly 1,000 km^2 of small catchments in the Seattle area to examine the impact of future development and infrastructure scenarios. Linker et al. (2000) used HSPF in conjunction with a regional acid deposition model to estimate nutrient loads at different time periods and under differing scenarios. They found that loadings from urban areas were the greatest source of loadings to the Chesapeake and that currently available technological approaches could reduce the discharge of pollutants by between 42-64% for the pollutants TSS, phosphorus, and nitrogen. Nasr et al. (In Press) used HSPF and SWAT to look at phosphorus loadings from agricultural lands in Ireland. They found that HSPF and SWAT were comparable in estimating annual total phosphorus loading, but that HSPF did a better job of estimating mean discharge while SWAT did a better job of estimating mean daily load. Singh et al. (2004) found that calibrated hydrologic HSPF and SWAT models compared favorably to actual streamflow data on a large Illinois watershed. HSPF has also been used as the primary model in countless watershed simulations for TMDLs, including streams in Virginia for parameters including fecal coliform, nutrients, and sediment. Streams for which these TMDLs have been developed include Accotink (fecal) (Moyer & Hyer, 2000), Beaver Creek (benthic/sediment and bacteria) (George Mason University & Tetra Tech Inc., 2004b), Four Mile Run (fecal) (Northern Virginia Regional Commission, 2002), Goose Creek (fecal) (Interstate Commission on the Potomac River Basin, 2003), Smith Creek

(Benthic/Sediment and bacteria) (George Mason University & Tetra Tech Inc., 2004a), an unnamed tributary to the Chickahominy River (phosphorus) (George Mason University & Tetra Tech Inc., 2004c). In summary, HSPF has been used on numerous occasions to accurately simulate watershed conditions and it has been used as a foundation for watershed planning purposes. HSPF is used in chapter 3 and chapter 5 of this work for flow dynamics of watersheds.

As can be seen, there are numerous strengths with HSPF: a calibrated and validated HSPF model yields results that are realistic simulations of real world conditions. However, there are several drawbacks of HSPF. Primarily, the model has a steep learning curve that means only dedicated modelers use the program. Additionally, the data requirements are relatively significant: a fact that is somewhat aided by use of EPA's BASINS program. Significant monitoring data should be collected for both calibration and validation of the water quality model for available watersheds. Though this increases the realism of the model, it requires significant expense and requires a broad range of expertise to generate all of the required data. Hence, where resources are not limiting and expertise is readily available, HSPF is an excellent multifunctional model for simulating watershed loading and hydrology.

Whereas HSPF is a deterministic watershed model, some models determine loadings via statistical approaches. An example of a statistical modeling tool that uses multiple regressions to develop loadings from existing data sources is the LOAD ESTimator program (LOADEST) (Runkel et al., 2004). A USGS model, LOADEST creates regression models that can be used to estimate constituent loads in streams and rivers. The model needs a fairly rich monitoring set through which the user can calibrate historical loadings to a point in the watershed for which the user has monitoring data. Data inputs required for LOADEST include streamflow, concentration of constituent, and any user defined variables that the modeler wishes to include in the calibration (such as point source loadings or land use). The model gives output in the form of monthly flow or seasonal loading, and individual output files estimate daily loading. LOADEST is used for estimating parameter loads in chapter 4 of this dissertation.

Other modeling approaches have used more simplified approaches based on generalized principles. These models tend not to be highly calibrated or validated in specific watersheds. Models such as City Green (American Forests, 2000) have focused on benefits provided to urban aquatic ecosystems by urban forests, planted trees, and soils. Citygreen is based primarily on the TR-55 model and an engineering approach using curve numbers. The curve number approach, developed by the USDA Natural Resources Conservation Service, looks at the relative imperviousness of a land surface based on its cover and its soil type: the more impervious an area, the higher its curve number. The City Green model can be used to model the decreased intensity of stormwater from forested or partially forested watersheds in urban areas. The model also can calculate other benefits provided by forests such as carbon sequestration, air pollution mitigation, and energy conservation.

PLOAD is a simple water loading model available in BASINS through which annual pollutant loadings can be calculated by export coefficients or using what is called the simple method. Export coefficients are reported as the mass of pollutant per unit of area per year (US EPA, 2001b). The simple method is an empirical approach using impervious cover and precipitation that estimates pollutant loading for watersheds of less than one square mile. The PLOAD model has been used to look at loadings of total suspended solids, nitrate-N, ammonia-N, BOD, total Kjehldahl nitrogen, phosphorous, fecal coliform and some metals such as lead and zinc. For example, Cui et al.(2003) used the model to evaluate loadings from numerous pollutants in the Xinshan sub-watershed of the Taihu watershed in China using a GIS interface. The model uses GIS land use data, GIS watershed data, pollutant loading rate date tables, and impervious terrain factor data tables, in addition to optional inputs of BMP site and area data, BMP pollutant reduction tables, and point source location and loads (US EPA, 2001b). The model is simplistic and is designed so that it could be used in a wide variety of projects including meeting the needs of stormwater permitting and watershed management.

The model requires a delineated watershed and land use data. In its calculations, the model overlays the watershed delineation with the land use data to calculate the total of each land use in the watershed. The pollutant loading rates, impervious cover factors, and BMP efficiency rates are put into pollutant loading tables for use in PLOAD. These pollutant loading tables include the export coefficient and the event mean concentration; the impervious factor table includes the relative percentage of imperviousness for each land use type, and the BMP table that identifies the effectiveness of various BMP types. The export coefficient method uses the following equation:

$$L_P = \sum_U (L_{PU} * A_U)$$

Where: L_P = Pollutant load, lbs;

U = Land Use Type

L_{PU}= Pollutant loading rate for land use type u, lbs/acre/year; and

A_U = Area of land use type u, acres (US EPA, 2001b)

For the simple method, the following equation is used:

$$LP = \frac{\sum U(P * PJ * RVU * CU * AU * 2.72)}{12}$$

Where:

LP = Pollutant load, lbs

P = Precipitation, inches/year

PJ = Ratio of storms producing runoff (default = 0.9)

RVU= Runoff Coefficient for land use type u, inches run off/inches rain =

$RVU = 0.05 + (0.009 * IU)$

CU = Event Mean Concentration for land use type u, milligrams/liter

AU = Area of land use type u, acres

and IU = Percent Imperviousness (US EPA, 2001b)

With both the export coefficient approach and the simple method, it is possible to refine the coefficients of the various parameters to aid in matching output to observed data and customize the model to each watershed. PLOAD is used in this dissertation with the export coefficient method applied in chapter 5.

Modeling using export coefficient approaches appears to be a valid approach compared to estimating observed loadings. Johnes (1996) constructed a simple export coefficient model to examine total nitrogen and total phosphorus loadings to two catchments. The author found that modeled nitrogen loading was within 2% of observed nitrogen loading and phosphorus loading was within 0.5% of observed total phosphorus loading for the year 1989. For a second watershed, the author found similar results for a different year. Hence, in this scenario, use of the simplified export coefficient approach gives plausible results.

The Long Term Hydrologic Impact Assessment model (L-THIA) is a simple model based on the curve number approach (similar to what is used in City Green) and use of the simple method (similar to part of what is available in PLOAD). L-THIA is available as both a web-based tool and as a downloadable GIS extension. The web-based tool includes historical climate data, and allows the user to input land use data and soil type in a simple to enter form. Both the web based and GIS based model estimate annual flow and pollutant loadings.

L-THIA has significant advantages in that is simple to use, and has been used in other studies examining water quality. Bhaduri et al. (2000) used L-THIA to model historic land use scenarios for select Midwestern watersheds. They found that annual runoff volume increased by more than 80% with an 18% increase in urban area coverage. The authors also used the simple method and event mean coefficients to model zinc, lead, and copper, and found that estimated average loads increased by 50% with the same 18% increase in urban coverage. Bhaduri et al. (2001) compared L-THIA to a commonly used complex stormwater model: the Storm Water Management Model (SWMM). The authors found that SWMM and L-THIA gave comparable annual runoff estimates, with L-THIA overestimating runoff by 1.1 to 23.7% compared to SWMM. The models appear to perform more similarly when watershed size increases. L-THIA is considerably easier to use than SWMM, and appears to be a valid tool for use by watershed managers to examine the impacts of land use change.

Other studies have used L-THIA to look at current conditions in watersheds using a simplified approach or to forecast water quality conditions by coupling L-THIA with other models including land use models and climate change models (Hwang & Foster, 2006; Wang et al., 2005). The significant strengths of L-THIA include that it was available in low to medium complexity and it gave plausible results. The simplest version of L-THIA uses the basic web based interface to allow the user to input land use. Average precipitation data, default curve number values and default event mean concentrations are used in the model calculations. A more complex version (that used in this dissertation) allows the user to assign custom land uses, define curve numbers, and if the user wants to calculate pollutant loadings, to use custom event mean concentrations based on monitored data or local values from the literature. However, the model has some shortcomings. For starters, the model does not currently include a function whereby users can look at the impact of BMPs[7][8]. Furthermore, until recently, the model

[7] There have been discussions to include BMP performance in L-THIA. EPA is currently working on a project to document 'green' BMP results and the creators of L-THIA have expressed interest in including theses results in the next model release.

[8] A user who is familiar with curve numbers can adjust them downward or lower the event mean concentration coefficient for a particular performance to simulate the impact of a BMP, but this is a crude approach to simulating BMP performance.

could not be truly calibrated or validated, though output can be compared to other modeling or monitoring data. A paper by Lim et al. (2006b) discussed the development of a program that automatically used millions of curve number combinations to improve the performance of the model. The authors note that the calibrated model predicted 43% more annual runoff, 24% more total nitrogen, 22% more total phosphorus, and 43% more total lead than the uncalibrated model. However, by using such a complex calibration approach that is not readily available, one risks losing the prime benefit of L-THIA: that it is simple to use and gives plausible results. Unless such a calibration framework is made available for individual unskilled model users, it seems that such an approach undermines many of the goals of the original model: to be a useful tool for the everyday practitioner. For automated calibration to add significantly to the utility of L-THIA, it must be user friendly. L-THIA is used in this dissertation in chapter 5, and it is not automatically calibrated. However, the output of L-THIA is compared to the hydrologic output from the HSPF model.

Monitoring and modeling provide tools that are useful for characterizing and managing watersheds. There are multiple approaches to both monitoring and modeling watersheds that vary from reasonably simplistic to complicated and detailed. Researchers or watershed managers must determine their goals and resources available before selecting the approach they will use to examine their targeted watersheds. In an ideal world, the most sophisticated monitoring and modeling would be employed on all aquatic water bodies. In the resource-constrained world in which we live, even simple level screening models are not used to examine many watersheds. Hence, there is a balance between sophistication and simplicity in how we study and manage watersheds.

2.3 How do we treat urban runoff and stormwater?

As part of the Clean Water Act amendments in 1987, EPA was directed to regulate most stormwater under NPDES programs (Water Environment Federation, 1997). The most recent regulations were promulgated in 1999: as of this point, all discharges must receive NPDES permit coverage from many municipalities, some industries (designated by SIC code), all construction sites greater than 1 acre (or belonging to a common plan of development of greater than 1 acre), and any other significant sources of stormwater designated by the permitting authority (US EPA, 1999). All of these regulated entities must employ best management practices (BMPs) to reduce the impact of their discharge on receiving water quality. Fairfax County was regulated under the first phase of the stormwater regulations, and runoff is regulated stormwater from both the entire Pohick and Accotink watersheds. BMPs may also be employed for other voluntary programs, such as agricultural or school programs, and are generally low cost (and sometimes more efficient) approaches to treating effluent than heavily engineered approaches.

Municipalities are left with great latitude in terms of how to manage their stormwater programs. A few examples of BMPs that municipalities and construction site operators employ include dry detention basins, wet retention ponds, constructed wetlands, silt fences, underground detention, bioretention (rain gardens), riparian buffers, and grass swales (US EPA, 2005b). These BMPs tend to emphasize either settling, infiltration, or filtering. To this point, Fairfax County has relied heavily on the use of dry detention ponds and wet detention ponds for most of their newer development and requires silt fences for construction sites. All of these techniques rely on settling mechanisms. The county is encouraging employing newer management techniques on a smaller scale such as using low impact development (LID) or better site design.

Results are mixed on the long-term performance of BMPs, and there are questions as to how BMPs perform to protect watersheds. Pollutant removal efficiency rates are recorded as high as 90% for some pollutants (particularly those that settle) for certain BMPs (US EPA, 2005b). BMPs may also mitigate changes in hydrology caused by urban runoff and minimize the intensity of the peak flow discharge. Some BMPs, particularly those that encourage infiltration or evapotranspiration, can reduce the total annual flow volume to near pre development conditions. Based on results put into two BMP databases, O' Shea et al. (2002) reported removal performance between -100% and 100%; however, for most studies, results indicate positive removal efficiency. Different BMP approaches have variable impact on different parameters: for example, wet retention ponds and grass swales appear considerably more effective at removing nitrate-N than dry detention ponds[9]. Lack of performance for BMPs based on settling

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[9]The following definitions are quoted from EPA's Menu of BMPs (2005):

"Wet ponds (a.k.a. stormwater ponds, wet retention ponds, wet extended detention ponds) are constructed basins that have a permanent pool of water throughout the year (or at least throughout the wet season).

In the context of BMPS to improve water quality, the term swale (a.k.a. grassed channel, dry swale, wet swale, biofilter, or bioswale) refers to a vegetated, open-channel management practices designed specifically to treat and attenuate stormwater runoff for a specified water quality volume.

mechanisms is particularly notable for dissolved constituents (O'Shea *et al.*, 2002), of which nitrate is an example. On a watershed scale, this means that the favored BMPs such as dry detention ponds and wet retention ponds may not be adequately protecting downstream water bodies from nutrient loading. Further complications arise when hydrology is altered. For example, Groffman et al. (2002) found that changes to hydrology caused by urbanization alter the manner in which nitrogen in the form of nitrate is produced and sequestered and denitrified in urban riparian areas. In effect, these urban riparian areas produce greater quantities of nitrate and store lower quantities, reducing their ability to remove nitrate from urban runoff.

Low Impact Development (LID) and Integrated Management Practices (IMPs) are showing some promise in maintaining predevelopment hydrology, and may offer promise for mitigating urban development more effectively (Prince George's County (Md.), 2000a, b; US EPA, 2000c). Some studies have been conducted that examine the effectiveness of techniques associated with LID. For example, the US EPA (2000b) published a study looking at the effectiveness of bioretention cells or rain gardens (one of the most promising technologies for pollutant loading reduction when used properly) for a retrofitted parking lot in Prince Georges County, Maryland. They used a landscaped island to filter runoff from a parking lot using gravel and a bioretention soil mixture of 50 percent construction sand, 20 to 30 percent topsoil, and 20 to 30 percent compost. In a short term study, they found the facility to have approximately a 87 +/- 2% removal

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Dry detention ponds (a.k.a. dry ponds, extended detention basins, detention ponds, extended detention ponds) are basins whose outlets have been designed to detain stormwater runoff for some minimum time (e.g., 24 hours) to allow particles and associated pollutants to settle. Unlike wet ponds, these facilities do not have a large permanent pool of water.”

efficiency of phosphorous, 67 +/- 9 removal efficiency of Total Kjeldahl Nitrogen, a 15 +/- 12 percent removal efficiency of nitrate, and that the temperature of the water was 12 degrees centigrade lower. Metals had removal efficiencies of 43 to 79 percent. In short, the bioretention cells may serve as a viable option for retrofitting older urbanized areas. Through modeling approaches, Williams and Wise (2006) showed that using land preservation practices and infiltration based stormwater techniques (such as bioretention) minimized the impact of development on the local hydrology. In summary, there are approaches that are more effective (and offer greater ecosystem services) than simply using dry detention basins to control the impacts of urbanization.

One additional commonly used approach to protect streams that can be considered a BMP is the use of forested or vegetated riparian buffers. Riparian buffers play major roles in the characteristics of neighboring smaller order streams (Knight & Bottorff, 1984). The cover from the vegetation serves to shade the water body from solar radiation and minimizes erosion in the riparian area. As a result, low order streams that are shaded will be cooler (Wilkerson *et al.*, 2006). Furthermore streams with effective riparian buffers are less likely to be alkaline and more likely to have lower turbidity and higher average dissolved oxygen (DO) concentrations. They also have a higher abundance of benthic macroinvertebrates that are shredders to breakdown allochthynous products and an increase in aquatic species that need moderated conditions (an increase in pollutant intolerant species) (Horne & Goldman, 1994). In contrast, first order streams that do not have forested riparian buffers are likely to be more alkaline if production is high and have greater variations in temperature. They are also more likely to be influenced by erosion and have higher peak runoff events and lower low flow marks, have higher daytime DO concentrations and lower DO concentrations at night, have an increased abundance of benthic macroinvertebrates that are grazers to eat autochthynous products, and species that prefer warmer temperatures (Matteo *et al.*, 2006). Forest buffers serve to remove pollutants from surface overland flow and soil drainage (US EPA, 2005b). Nitrates are more likely to go through the process of dentrification; in small quantities, oil and grease are likely to be trapped and eventually degraded by bacteria; phosphorous is used by plant species and stored and even occasionally transported away from the streams. The effectiveness of riparian buffers have been shown in multiple studies, including Montreuil and Merot (2006), who found that vegetated riparian bottomlands (including wetland components) removed up to 30% of nitrates from agricultural catchments.

Fairfax County has implemented strategies including requirements of riparian buffers and BMPs throughout their watersheds. The County designated any land within 100 feet of a feature including water bodies with perennial flow and their riparian buffers as resource protection areas, or areas to be protected (Fairfax County, 2005). The County began requiring best management practices for detention control in 1972, and for water quality county wide in 1993 (Kumar *et al.*, 2005). Kumar et al. (2005) estimate that approximately 25 to 50 percent of urbanized area in the Accotink watershed is uncontrolled by stormwater BMPs and 10 to 25% in the Pohick watershed is uncontrolled; however, the authors note that the amount of urban area controlled is likely overestimated. Furthermore, many of these controls are for detention purposes only, and do not significantly reduce the total volume of stormwater runoff nor do they offer significant water quality improvements. Hence, one would expect to see some water quality and hydrology benefits offered from these BMPs, but benefits offered by many of the older BMPs (and hence, many of those BMPs in Accotink) are less significant.

2.4 How this dissertation compares to previous literature, analyzes urban runoff, and contributes to the field

This dissertation seeks to utilize multiple tools to quantify recent historic water quality and hydrologic regime in the Accotink and Pohick watersheds. Flow analysis using observed data for the upper portion of Accotink is examined back to 1948. Modeled flow data using HSPF is examined back to 1975. Water quality parameter analysis starts from 1983, when George Mason and Noman Cole water quality monitoring data are first available. Based in part on the framework laid in these analyses, many of the tools discussed above, L-THIA, HSPF, PLOAD (and variants exported to spreadsheets), are used to better estimate current and future impacts of urban land use. The water quality and hydrology modeling tools used are listed with key characteristics and capabilities in Table 2.2. Additionally, the table displays time period (past, current, and future) for which each tool is used. The logic of how these tools are integrated with one another is displayed in Figure 2.1

Table 2.2. A table with how four water quality and hydrology modeling tools are used in this dissertation.

Model Characteristics/ Capabilities	LOADEST	PLOAD (export Coefficient)	HSPF	L-THIA
Type	Statistical	Deterministic	Deterministic	Deterministic
Time Series	Daily	Annually	Hourly*	Annually
Hydrology	No	No	Yes	Yes
Nutrients	Yes	Yes	No**	No***
Complexity	Complex	Simple	Complex	Simple - Intermediate
Significant Learning Curve	Yes	No	Yes	No
Tool for Non-modelers	No	Yes	No	Yes
Required Data				
Land Use	No	Yes	Yes	Yes
Flow	Yes	No	Yes	No
Precipitation	No	No****	Yes	Yes
Monitoring Data	Yes	No	Yes**	No
Time Series				
Past	XX	X	XX	X
Present	XX	X	XX	X
Future	No	XX	XX	XX

***** HSPF has the capability to model on an hourly time series, but daily output are used to report results for this dissertation.
** The HSPF model can model nutrients. Nutrient modeling with HSPF was not completed for this dissertation. Further work might be pursued post graduation.
*** L-THIA automatically models nutrients. Results are reported in the dissertation, but not given significant emphasis.
****PLOAD needs precipitation data for the simple method; however, not for the export coefficient method.
X Used during this time period to check the plausibility of results.
XX A primary tool during this time period and heavily emphasized.

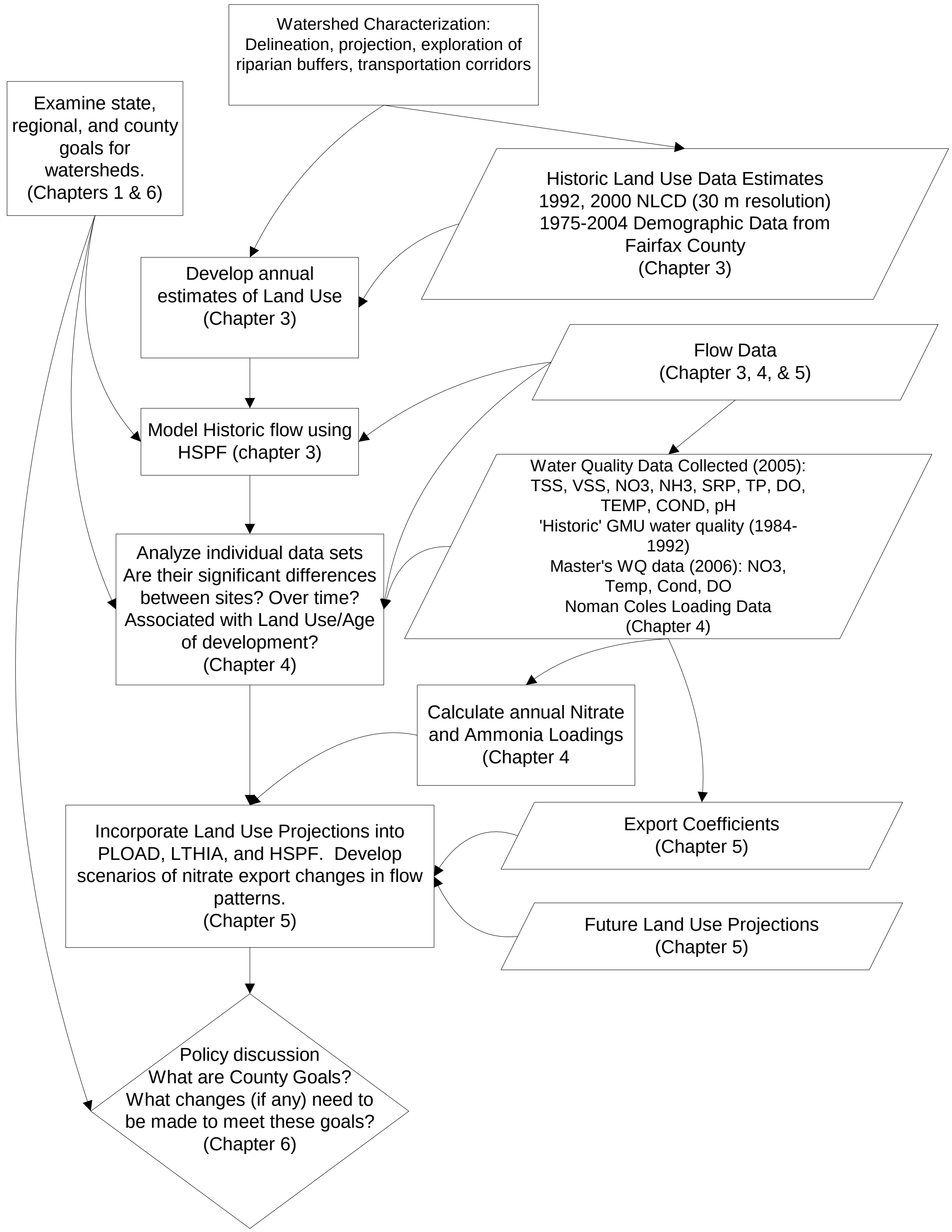

Figure 2.1 The logic and key questions this dissertation asks presented in a flow chart.

In the last few years, some studies have attempted to examine combinations of historic water quality conditions associated with land use and predicted future watershed land use conditions or climatic conditions to aid in watershed management (Arthur, 2001; Bhaduri *et al.*, 2000; Conway & Lathrop, 2005; Solecki & Oliveri, 2004; Wang *et al.*, 2005). These analyses are useful in giving planners watershed specific information and in exploring tools that can be used by other researchers. In addition to the location specific analyses being conducted, this study uses unique approaches that may prove exportable to other studies. Hence, the study completes a holistic water quality and flow analysis on two watersheds while exploring multiple tools available for watershed researchers. Based on these results, other users can pick and choose among these methods for future studies. However, most importantly, the research sought to consolidate some information currently available in Accotink and Pohick and provide additional analyses that could result in the creation of usable information for county decision makers.

3. Combining Demographic and Remote Sensing Data to Improve Performance of Watershed Applications

3.i - Abstract

Demographic household data and remote sensing data from the National Land Cover Data Set (NLCD) and the Multi-Resolution Landuse Consortium (MRLC) were used to devise annual landuse estimates for two watersheds in Fairfax County, Virginia. These estimates, here referred to as Household Land Use (HLU) estimations, were created to increase temporal resolution of land use estimates for examining historical water quality data sets. Our generated land use estimates were compared to other independent remotely sensed, temporally variant land use data for the same watersheds. Finally, the hydrologic module from the Hydrologic Simulation Program Fortran (HSPF) was employed to produce a watershed model that was used with the remotely sensed and classified land use data set (NLCD 1992) to generate flow estimates for multiple time periods. These results were compared to HSPF model runs generating flow using HLU estimates. The HLU estimates generated plausible results consistent with the other independent remote sensing data sets. Furthermore, use of these HLU data improved the performance of the hydrologic component of the HSPF model compared to the model performance using the static remote sensing land use data set.

3.1 Introduction

Land use plays an integral role in water quality, fluvial geomorphology, stream habitat, and aquatic biological diversity (Brown & Peake, 2006; Ierodiaconou *et al.*, 2005; Jones & Clark, 1987; Jones & Holmes, 1985; Roth *et al.*, 1996; Schueler, 1994). However, readily available land-use data are often limited temporal period scales. This lack of easily interpretable data can pose challenges for researchers trying to analyze impacts of land use change. To examine land use change, researchers need satellite data, aerial photography, or pre-classified spatial land use data from two or more time periods. Creation of these data can be expensive and time consuming, and remote sensing data is often limited. Many water research projects therefore use existing land use datasets or land use values from a single year and do not incorporate land use change components.

Water researchers or land use planners can more closely examine the impacts of land use on historic water quality if they have finer temporal resolution land use data. Commonly used water quality models including HSPF, L-THIA, PLOAD, and SWAT all have land use data requirements. Total Maximum Daily Loads (TMDLs) often use water quality models as tools to examine watershed dynamics; however, they often look at land use as a static term. For example, National Land Cover Data Set (NLCD) 1992 land use classifications are commonly used for multi year model calibrations. The maximum allowable load and waste allocations are often then calculated using static land use assumptions.

Remote sensing techniques, including orthorectification and then classification of aerial photography or classifying satellite images are common techniques used to determine land use at different temporal scales. These analyses are then often used to look at land use change. However, these techniques are both time consuming and expensive. In the case of researchers or practitioners who want additional temporal resolution, but are not remote sensors and do not have the expertise nor budget to complete or fund land use classifications, an approach that makes use of readily available data sets would be valuable.

Some studies that combine data sources to increase overall spatial data quality have been conducted. Ryznar and Wagner (2001) looked at social changes and vegetation in the urban environment and proposed developing a tool to use these results to track change in cities. Vogelmann et al. (1998) attempted to use remote sensing data with topographic data, US census data, wetlands inventory data, and other data sources to more accurately determine land use classifications for the Mid-Atlantic region of the United States. Kerr and Cihlar (2003) used SPOT[10] satellite imagery and Canadian Census of Agriculture data to improve classification of agricultural areas. Stefanov et al. (2001) developed an export system that used spatial social data such as water rights, and spatial physical data, such as land use and city and territory boundary data, to develop a system that more accurately classified land use for an existing LANDSAT image. These efforts focus on improving the spatial quality, accuracy, and usefulness of existing spatial data. However, these studies do not attempt to estimate land use for time where there is not accurate spatial data for any given year. Hence, these studies attempted to use demographic and other data to support existing remotely sensed spatial data and as a

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[10] SPOT imagery has a resolution of up to 2.5 meters, is taken by the French Space Agency, and is packaged and delivered by the commercial company Spot Image.

result spatial land use classification improved significantly. This study, in contrast, uses spatial land use data derived from remotely sensed data to supplement demographic datasets to derive spatial urban land use estimates at a finer temporal scale than what is available from the initial spatial dataset alone. Furthermore, I attempted to derive a simple technique that would be easily exportable for use by other practitioners until readily available, high quality temporal data sets are freely available.

In this paper, I discuss using remote sensing data combined with demographic data to obtain urban land estimates at a watershed scale on an annual basis. These estimates were generated to better look at characteristics within and between watersheds for the Accotink and Pohick watersheds in Fairfax County, Virginia. I hypothesize that such land estimates can be done accurately and that they are useful for examining watershed processes, including historical flow and water quality regimes. The approach used in this paper is summarized in Figure 3.1.

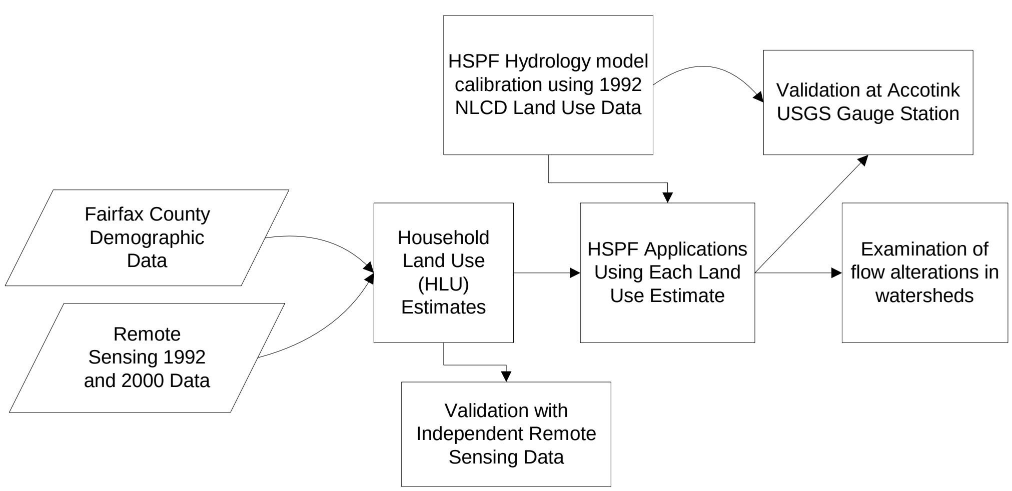

Figure 3.1 Generation of urban land use estimates and their use in the Hydrologic Simulation Program Fortran (HSPF) hydrology application. This flow chart illustrates the integration of HLU land use data and HSPF as used in this paper to simulate improved surface water hydrology for historic watershed analysis.

3.2 Estimating Land Use

This study examines historical land use in Pohick and Accotink watersheds in Fairfax County, Virginia. Both watersheds drain to Gunston Cove, a 6 km^2 embayment on the Potomac River south of Washington DC. The watersheds were segmented based upon locations of current and historical water quality monitoring sites collected by George Mason University and the Noman Cole Sewage Treatment Facility (see Figure 3.2). (Jones & Kelso, 2005b).

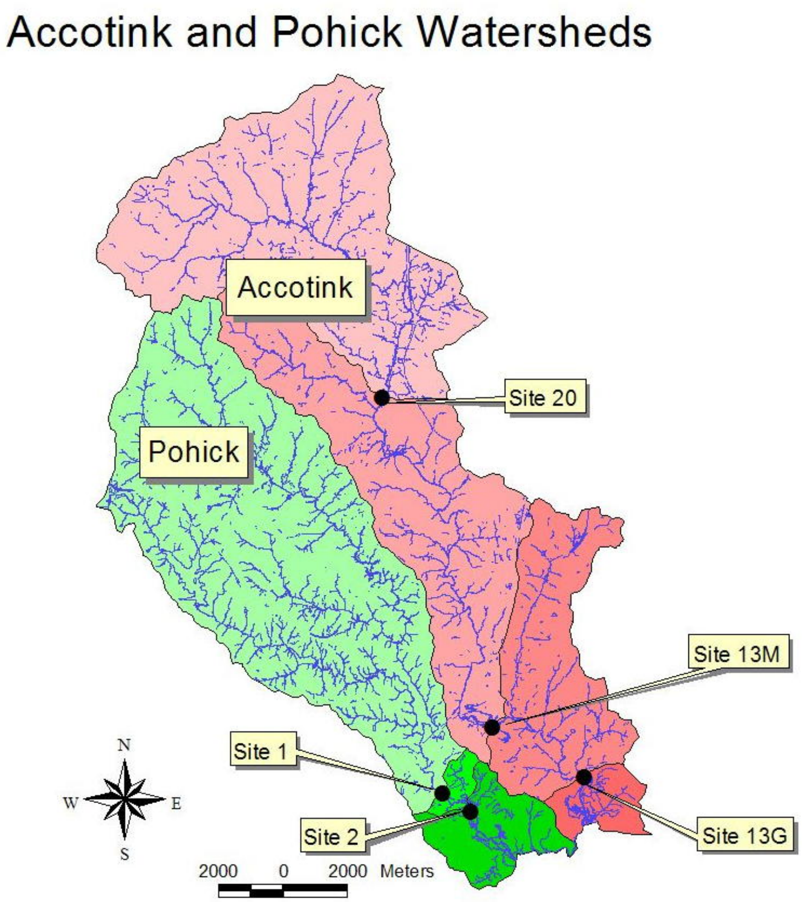

Figure 3.2. Accotink and Pohick watersheds with George Mason University Sampling Sites

3.2.1 Land Use Estimation Methods

Two pre-classified land use data sets were used: NLCD (1992) and Multi-Resolution Landuse Consortium (2000), both of which are in raster format. Both of these data are projected into Albers Conical Equal Area NAD83. The NLCD 1992 data are derived from LANDSAT TM imagery and have a spatial resolution of 30 meters. Each 30 meter pixel was classified into one of 21 classes consistent with Anderson level II classification schemes (Anderson *et al.*, 1976). The MRLC data are also derived from the LANDSAT TM satellite images, which have a 30-meter spatial resolution. The classification is somewhat different than the NLCD 1992 data, with 29 different land use classes, but they are also consistent with Anderson Level II classification. For the purposes of these analyses, these data were reclassified into 6 land use classes: urban, forest, grasses and pasture, wetlands, transition or barren, and water that are consistent with Anderson level I classification.

The land use data were reclassified into a system similar to Anderson level I classification for four main reasons. First, this approach is standardized and can be used with multiple data sets. Secondly, this work will be used with future land use projections, some of which would not be compatible with Anderson level II classification. Third, these results will ultimately be used with export coefficients, a simple approach for estimating precipitation driven loadings of pollutants to water bodies. These export coefficients are commonly available for Level I classifications. Lastly, and most importantly, this methodology aggregates land use at a watershed scale using household demographic data to determine land use at frequent temporal scales. Urban land use that is industrial or commercial in nature is not directly captured by residential household data, and one must assume that there is a relationship between the amount of residential households and commercial, transportation, and industrial data. On a watershed scale, I felt that these assumptions were warranted because I was calibrating the estimates derived to each individual watershed based on the amount of urban area produced by each household. In effect, this calibration indirectly takes other land uses into account at the watershed scale.

The Accotink and Pohick watersheds were manually delineated using USGS topographic maps and Fairfax County topographic data using ESRI Arcview 3.3. The watersheds were projected in Virginia North NAD 83. Several water quality samples were taken at sites within the watersheds; subwatersheds draining to these sample points were also delineated. The watershed files were then reprojected to Albers Conic Equal area NAD83. A script was used to clip the 1992 and 2000 land use data to the watersheds. The total area of each land use for each subwatershed for each time period was then summed.

Fairfax County has a rich demographic dataset that includes annual estimates for households by census and subcensus tract, sewersheds, and planning districts (Fairfax County, 1975-2004). These data sets are currently available for 1975 to 2004. Fairfax County has specific estimates for the number of townhouses, multi-family, and single-family housing units per administrative unit. The housing units are calculated based on real estate tax assessments, building permits, utility hook up information, and land use codes. For the purpose of this analysis, I elected to aggregate data simply into total households because I felt this was likely to be most compatible with level I classification. Additionally, in order to use demographic data, I had to be sure that the demographic data and the spatial land use data could be used at the same scale.

I elected to use household data based on sewersheds for this analysis since county sewersheds nearly overlap with the watershed boundaries. For subwatersheds, I estimated the percentage of each sub-sewershed area that was present in each subwatershed. This approach effectively assumes that population is distributed evenly across the sub-sewershed and likely results in some estimation error. Census or subcensus tract data could also be used with this approach, particularly since census data are the most commonly available and accurate demographic data in the United States. I elected not to use the data at the subcensus and census scale for this analysis because I would have had to overcome two minor obstacles: the census tracts do not directly overlap with the watersheds and census and subcensus tract boundaries change periodically. These limitations could be overcome by using census tract GIS layers and determining how much of each census tract is in the watershed for each time period. If only a portion of the census tract were in a watershed, then researchers could simply prorate the number of households in the watershed relative to the area within the watershed boundaries. Minor data irregularities might be created when census tract boundaries change due to differences in how households are prorated. If sewershed data were not available, or if I were seeking to use a different demographic data source such as US census data, these steps would need to be taken. However, since annual county demographic estimates were available at the sewershed scale, I elected to use those data.

Next, for each subwatershed, I estimated the density of households per pixel of observed urban land use based on the 1992 and 2000 remote sensing land use data. Such density observations were made to yield a ratio that can be used with household data to derive estimated urban area. For both 1992 and 2000, the density of households per square kilometer was calculated by:

$$Density_n = \frac{Households_n}{UA_n}$$

Where Density_n = the density of households to urban area at time period n (either 1992 or 2000)

Households_n = the number of households at time n in the subwatershed

UA_n = the Urban area in km^2 at time n in the subwatershed

For all subwatersheds examined, the density between 1992 and 2000 varied slightly. For instance, the density varied from about 761 households/km^2 urban area in 1992 to 772 households/km^2 urban area in 2000 in the entire Accotink watershed. In the overall Pohick watershed, the observed density varied between 1190 households/km^2 urban area in 1992 to 981 households/km^2 urban area in 2000. This decline in density per unit urban area could be attributable to an increase in industrial, commercial or transportation infrastructure relative to residential units; building more single family homes relative to townhouses or other high density development, or simply from classification error. In order to account for this change between 1992 and 2000, a linear change in density was estimated annually by:

$$Density_n = \frac{Density_{1992} - Density_{2000}}{8} + Density_{n-1}$$

Where $Density_n$ = a year between 1993 and 1999

Next, I needed to back-calculate density to 1975, the start of our demographic data series. We know that household density per acre was likely lower in 1975 due to the likelihood of larger lot sizes per family residence in 1975 than in the 1990s or 2000s; hence, I used the lower of 1992 or 2000 density values for 1975. This may result in a slight underestimate of urban land use, particularly pre 1980, but lower residential density was likely offset somewhat by increased area devoted to transportation infrastructure and impervious commercial areas per household. Therefore, density from 1975 to 1991 was estimated by:

$$Density_n = Density_{n+1} - \frac{\left|\left(Density_{1992} - Density_{2000}\right)\right|}{16}$$

This approach resulted in a density estimate of 761 households/km^2 urban area in the Accotink watershed and 981 households/km^2 in the Pohick watershed in 1975. It is quite possible, based on historic housing patterns, that these are over estimates of density. Typical construction in these watersheds from the 1950s to 1970s resulted in the development of primarily low-density, single-family houses. This issue is somewhat counteracted by the fact that it takes greater transportation and parking infrastructure to support these lower density developments: hence, on a watershed scale, greater automobile related impervious urban area would have been present. Nonetheless, it is possible that the higher density results in overestimation in urban land use in the earlier years of this analysis.

For years post 2000, density was assumed to remain constant because there is not adequate information using this approach to make changes to the density assumptions.

The meters squared per house is then determined by:

$$\frac{meters^2{}_n}{House_n} = \frac{1}{density_n}$$

Urban land Use (ULU) was then calculated for each year using year specific household estimates from the Fairfax County demographic data by:

$$ULU_n = Households_n * \frac{meters^2{}_n}{House_n}$$

I then generated other land use estimates based in large part on the results of urban area estimates generated with the demographic data. Barren or transitional land use was estimated by subtracting the number of households in the demographic data set from the current year minus the following year and assuming that the difference was houses constructed. There was assumed to be a base ‘barren’ component coupled with land dedicated to construction work. The base barren component would account for any long-term exposed area and classification error. The barren component for each year was estimated by:

$$BC = \frac{\left(BT_{1992} - \frac{1}{d_{1992}} * \left(H_{1993} - H_{1992}\right)\right) + \left(BT_{2000} - \frac{1}{d_{2000}} * \left(H_{2001} - H_{2000}\right)\right)}{2}$$

Where BC = the Barren Component

BT_n = The total of Barren and Transitional Land sensed in year n

D_n= Density of Households in year n in the watershed

H_n= Households in year n in the watershed

Transitional land use was estimated directly from the remote sensing data for 1992 and 2000. Transitional land use in years other than 1992 or 2000 was calculated by:

$$TLU_n = BC + \frac{1}{density_n} * \left(Households_{n+1} - Households_n\right)$$

Where TLU_n is Transitional Land use in year n

BC_n is the Base barren component in year n

Wetlands area and water area differed little between 1992 and 2000. These areas were assumed to remain constant from 1975 to 1992. It is likely the case that some wetlands were lost; however, with the passage of the Clean Water Act in 1972 and subsequent provisions to limit wetland destruction, I assumed that wetland loss was slowed considerably. Furthermore, Fairfax County has protected considerable areas of riparian area in these watersheds, and consequently, I can assume that these areas still exist in undeveloped form.

Forest area and agricultural area provided the largest challenges. I did not locate appropriate demographic data to support a specific quantified approach. Remotely sensed land use data were available from 1972 using Anderson Level I classification. These land use data were not used for the quantified part of this analysis, because it was evident that dominant land uses were highly overestimated in mixed-use watersheds, both of which included Accotink and Pohick. However, I used these data to qualitatively estimate that the change in urban land use came primarily from forested area from 1972 to 1992.

The Mid Atlantic Regional Science Application Center (RESAC) (2003) examined change in forest cover on a small parcel in Northern Virginia that is partly located in the headwaters of the Pohick watershed. Their results, using remote sensing of aerial photography, combined with a Markov model, found that the majority of urbanized land use change came from previously forested watersheds. Furthermore, despite a continuously increasing population, they noted that forest cover remained relatively stable through about 1978, before urbanization caused direct and significant loss in forest cover. Hence, though there is likely a slight loss in open area/agricultural area from 1972 to 1992 due to urban conversion and field abandonment, I assumed that all urban area came from forested area during this time period. Agricultural area was left as a constant from 1975 to 1992, with the acknowledgement that some urban grass area offset agricultural loss and that agricultural area in the early years of the estimation period are possibly underestimated. I assumed that significant agricultural and pasture area was abandoned in our watersheds leading up to the start of this study time period. Any loss of forest due to urbanization at this point was likely offset, at least in part, by regeneration of forests on pastured lands and regeneration of small forest cover in low density suburban areas. This assumption is consistent with nationwide trends of the abandonment of smaller farms from the 1910 to 1950s documented by Hart (1968). Many of these farms were allowed to regenerate to forest, which, in turn, were lost to urbanization. Furthermore, much of the grassland area classified in 1972 is located at the Lorton Federal Prison complex, an area that remained relatively unchanged until the early 21st century when the prison was shut down. For watersheds where urban land is converted from agricultural area as well as forested area, equations should be modified so that urbanization occurs on both forested and agricultural land.

Observed grassland and agricultural area increased from 1992 to 2000 in the MRLC classifications, likely due to opening up the forest canopy for urban grasses. Therefore, agricultural area for the time period from 1992 to 2000 is estimated by:

$$ALU_n = \frac{ALU_{2000} - ALU_{1992}}{8} + ALU_{n-1}$$

Where ALU_n = the grass, agricultural land, or open land use in year n

Finally, forested area for the full time period is estimated by assuming it is the remainder of all unclassified land:

$$FL_n = TWA - ULU_n - ALU_n - TLU_n - W_n$$

Where FL_n is Forested land in year N

TWA is Total Watershed Area

TLU_n is Transitional Land use in year n

W_n is area covered by water in year n.

To validate these results, generated urban land use estimates were then compared to impervious area data estimated by Jennings and Jarnagin (2002), which overlapped exactly with the northern third of the Accotink watershed (one of the subwatersheds for this study). Jennings and Jarnagin used aerial photography from Fairfax County to categorize impervious area for 6 time periods. Four of those time periods, 1971, 1979, 1988, and 1994, roughly paralleled the time period of this analysis. There were not enough data points to do standard analysis such as goodness of fit examinations; however, I compared the relative slopes of urbanization, in other words, how rapidly urbanization was occurring between time periods, between the 'household' method data set and the independent remote sensing observations. This validation approach assumes that impervious area roughly correlates with urban area.

Furthermore, Jantz et al. (2004) used remote sensing estimations of urban area in the Chesapeake Bay region for years 1986, 1990, 1996, and 2000 for calibrating a land use model known as SLEUTH[11] (discussed in chapter 5). The authors had resampled these data from a 30 to 45 meter resolution and had two land use classes: 0 for non urban and 1 for urban. As such, they are not directly comparable to the land use data here. However, I looked at these results to see if there were similar rates of urbanization

[11] The name SLEUTH comes from the data input requirements of the model: Slope, Land cover, Exclusion, Urbanization, Transportation, and Hillshade.

between the Jantz observations and the ‘household’ method for all of the subwatersheds. I reprojected the watershed files to NAD83 UTM 18 and analyzed the amount of each land use for each time period using Arcview 3.3. After exporting these results to Microsoft Excel and S-Plus 2000, I plotted the observed (household method) versus the expected (observations used by Jantz) and compared the relative slopes using linear regressions.

3.2.2 Land Use Estimation Results

The Accotink watershed, particularly at its northern headwaters, completed a greater percentage of its urbanization before the time period of this study. Pohick appears to have experienced peak urbanization in the 1980s and 1990s. From 1975 to 2004, the percentage of urban land use in Accotink is estimated to have increased from 36.3% of the watershed to 61.9% of the watershed, whereas it is estimated to have increased from 11.1% to 48.6% in the Pohick watershed. Urbanization followed expected trends, with urban area increasing in an outward direction from Washington DC (see Figures 3.3 and 3.4). There was more urban area than forested area in Accotink in 1985; in Pohick, there was more urban land use than forested land cover in 1997.

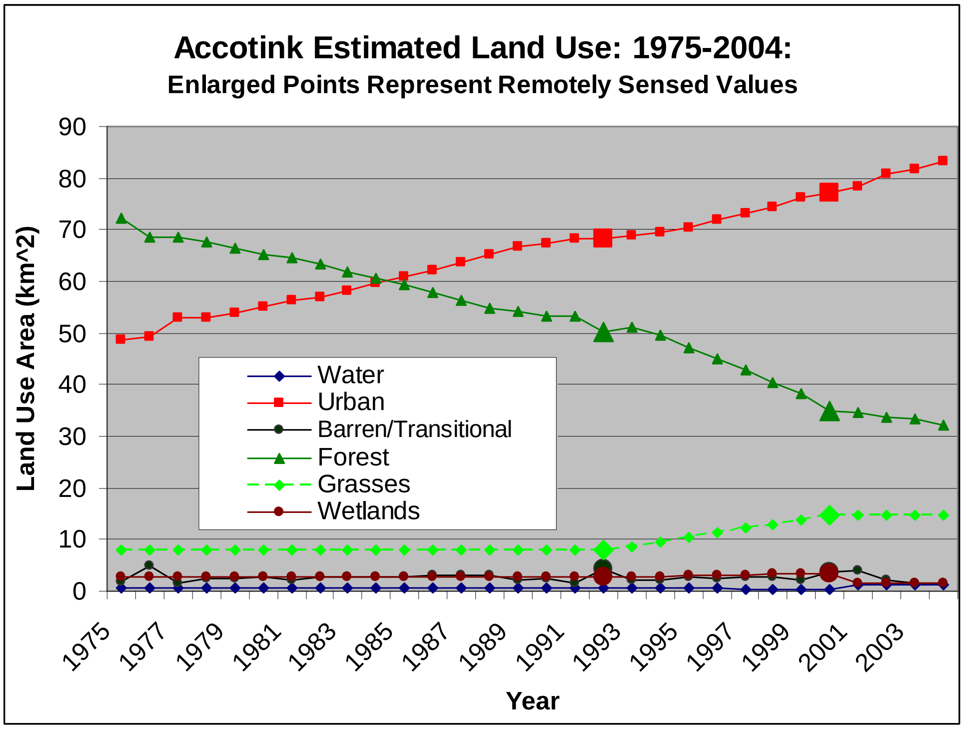

Figure 3.3 Estimated land use for the full Accotink watershed.

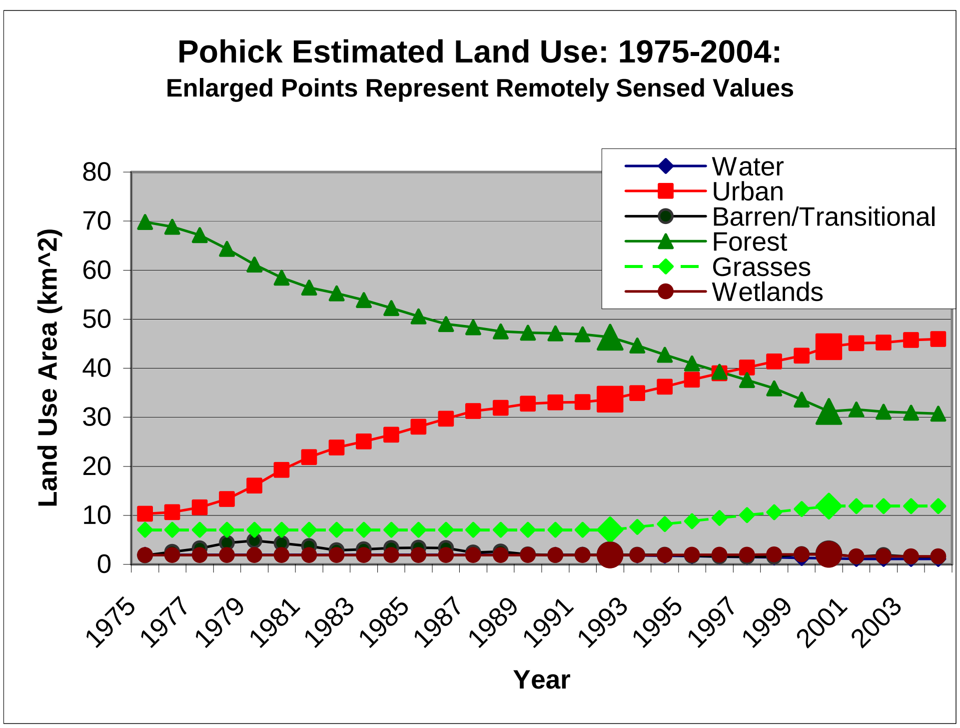

Figure 3.4 Estimated land use for the full Pohick watershed.

Results from the 'household' method compared favorably to the impervious area estimates obtained by Jennings and Jarnagin. In Figure 3.5, one can note that the 4 impervious area estimates recorded show a similar rate of urbanization compared to the 28 annual estimates based on the household method. A linear regression through each of these data sets has very similar relative slopes. The relative increase of impervious area from 1971 to 2004 based on the slope from a regression is approximately 82.7% (y = 0.549x + 21.363 R^2 = 0.9317) or a 2.51% increase per year for the remotely sensed Jennings and Jarnagan data. These results compare favorably to a relative increase of 85.4% (y = 0.535x + 20.412 R^2 = 0.9878) or 2.55% increase per year of urban area for the household method. These results compare less favorably to an increase of 63.6% (y =

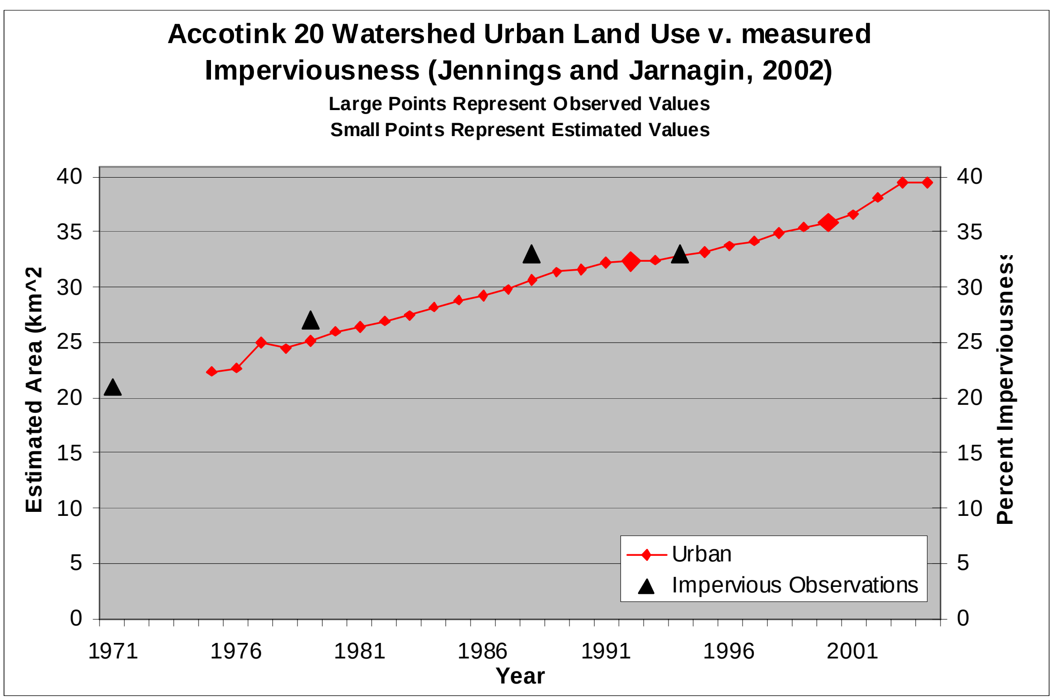

Figure 3.5. Estimated urban area with the household method compared to impervious area estimates generated by Jennings and Jarnagin (2002).

0.4439x + 22.575) if one were to use the slope of a trendline based on the NLCD (1992) and MRLC (2000) landuse datasets alone.

Results also compared favorably to the Jantz et al. (2004) estimations. The remotely sensed Jantz et al. values tended to estimate urban area higher than those values generated in the household method. The mean urban area estimate for 1986 was 41.7% using the household approach, but 49.1% for the Jantz observations. In 2000, which in the household method was based completely on the MRLC remotely sensed data, the results were 55.3% for the household approach and 62.0% for the Jantz observations. Hence, Jantz's observed urban area is somewhat higher than that classified by the MRLC. This difference could be due to any number of factors: classification differences (Jantz et al. mapped impervious area and used a 10% impervious threshold to classify a pixel as urban), alteration of results during resampling, or use of images taken during different seasons. Therefore, the Jantz results might not be directly comparable to those results from the household method if attempting to develop loading coefficients or other watershed applications, but the general urbanization trend is comparable. Nonetheless, the relative increase in urban area from 1986 to 2000 is comparable between the Jantz and HLU urban land use estimates (Table 3.1)

Table 3.1. The estimated increase of total watershed area in urban land use from 1986 to 2000.

Increased Percentage of Total Watershed in Urban Land Use:1986-2000		
Watershed	HLU	Jantz
1 (Pohick)	18.0%	15.3%
2 (Pohick)	11.1%	11.9%
Full Pohick	11.5%	12.0%
13 (Accotink)	10.1%	10.3%
13m (Accotink)	17.5%	15.5%
20 (Accotink)	10.9%	11.0%
Full Accotink	15.8%	14.4%

Next, I plotted the estimated values in the household method vs. the observed values from the Jantz et al. observations. Two different approaches were taken. The first approach was to look only at the full Pohick and full Accotink watersheds of the four time periods to allow for maximum independence of data. Linear regression between the 8 comparable data points in the two data sets were used to examine the ratio of estimated HLU urban area to Jantz estimated urban area: the R^2 explained the percent of variance explained by the regression. In this regression, the slope of the x coefficient equals the ratio of estimated HLU urban area to the Jantz urban area estimates minus the coefficient. This approach further clarified that the Jantz data estimated a higher percentage of urban land use for each time period, but the majority of the difference was explained by the regression ($y = 1.0878x-0.1108$ R^2 of 0.980, yielding a P value of <0.0001 (see Figure 3.6)). Hence, other than the issues discussed with underestimation, the urbanization estimates of the two methods are highly comparable.

The second approach looked at urban land use in all of the subwatersheds delineated based upon the location of the sampling station. This approach was used to determine if the results appeared valid for estimating urban area for smaller watersheds. Each of the urban area estimates for the household approach were compared to the observations used by Jantz et al. These subwatersheds are not completely independent of one another; in other words, the watershed for Accotink 20 drains into Accotink 13. I used the approach of including all land in upstream areas because I suspect that the household method would lose its validity at smaller and smaller scales. For example, attempting to draw broad conclusions based on a small tributary to Accotink alone might not be as trustworthy. However, the household approach also seemed to correlate quite well with the observed values used by Jantz ($y = 1.0227x-0.874$; $R^2 = 0.9539$ with 28 total observations, $P<0.0001$) when applied to these smaller watersheds (see Figure 3.7).

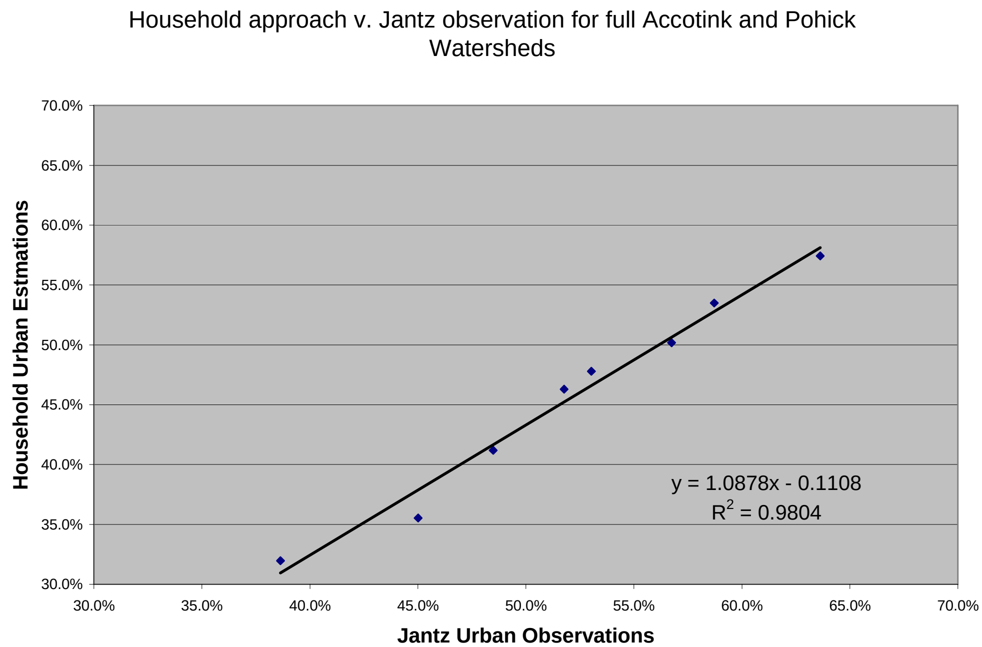

Figure 3.6. Comparison of household urban area estimates to those generated from Landsat images by Jantz et al. (2004) for 1986, 1990, 1996 and 2000 for the Accotink and Pohick Watersheds.

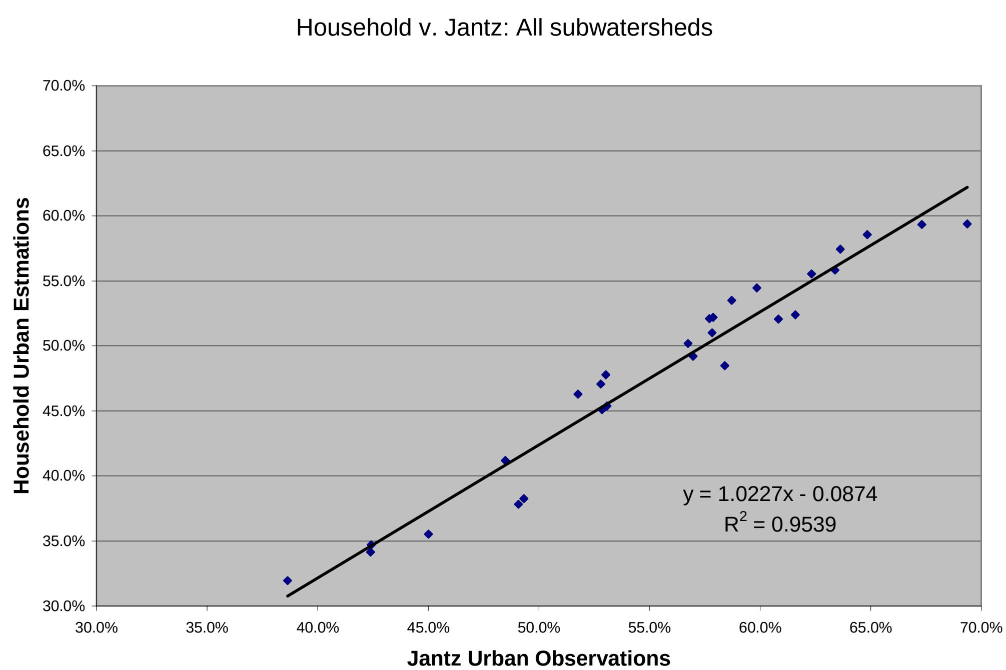

Figure 3.7. Comparison of household estimates to Jantz et al. (2004) estimates for all subwatersheds

3.3 Watershed Application: Use of the Hydrologic Simulation Program Fortran (HSPF) hydrology model with derived land use estimates

I sought to explore the utility of using these generated land use estimates in a watershed application. As such, I applied the hydrologic component of the commonly used water quality model Hydrologic Simulation Program Fortran (HSPF) to the northern third of the Accotink watershed. I compared observed flow to modeled flow using the static 1992 NLCD land use data and to modeled flow using 'household' estimates. I choose this approach for several reasons. First, HSPF is a widely used and accepted model in the watershed community. Second, changes in flow regime are the most obvious and easily quantified alterations to urban watersheds and as such, I could easily examine changes caused by shifts in land use. Third, the upper Accotink watershed corresponded to the United States Geological Survey gauge station on Accotink Creek that has been monitoring flow continuously since 1947. Hence, use of this model with the generated 'household' land use data would illustrate if the enhanced temporal data gave plausible results, improved model performance, and could have real watershed applications.

HSPF is a model funded by EPA and is a core part of the EPA Better Assessment Science Integrating Point & Nonpoint Sources (BASINS) tool (Bicknell *et al.*, 1997). The hydrologic component of the model is based on the Stanford Watershed Model, which estimates surface and shallow groundwater flow. The model requires precipitation data and evapotranspiration data to simulate stream flow; other meteorological data increase the number of functions that can be used in the model. The user also needs to use flow data to calibrate the hydrograph. HSPF adds numerous other modules to this model including various water quality components to increase the utility and functionality of the model. Because many of the goals of this project were to keep tasks as simple as possible (as would be done by practitioners), I only used data such as climatic and flow data that are easily downloadable from the BASINS system, other than the land use data derived from this method. Hence, my goals were to calibrate the model efficiently and avoid many of the problems associated with HSPF including seeking, modifying, and formatting substantial quantities of additional data.

The HSPF model was manually calibrated from 1991 to 1993 using NLCD 1992 data. As a starting point for calibration, other modeling efforts in the area were reviewed, including those on the Patuxent and Rappahannock Rivers included in the HSPFarm extension available in the BASINS 4.0 package. I also referred to a TMDL for fecal coliform for the upper Accotink watershed completed in 2000 that had a hydrologic component (Moyer & Hyer, 2000). Lastly, I used technical advice given in an EPA publication with suggested calibration parameters (US EPA, 2000a).

Calibration parameters were adjusted first using values from these reports as a starting point. Compared modeled versus observed flow at the Accotink USGS gauge station was used as the calibration metric. During the calibration period, a threshold was set where an acceptable calibration would model annual flow and seasonal flow within 10% of observed values. Furthermore, daily observed were compared to modeled flow at this USGS gauge station. Calibration parameters were fine tuned to capture daily maximum and low flow values. Furthermore, parameters were adjusted that maximized the correlation coefficient while maintaining calibration parameters that were realistic for the watersheds. Lastly, I used the advice available from the HSPFexp users manual to make final adjustments to calibration parameters (Lumb *et al.*, 1994), which resulted in adjusting seasonal parameters and shallow groundwater infiltration parameters. The snow component of the model was also included, which improved the model's performance considerably during the winter months. Model calibration used climatic data from National Airport, approximately 15 miles from the watershed being modeled. Though I am aware of a rain gauge station located within watershed at Vienna Woods, I made the conscious decision to only use data accessible in BASINS.

Several of these calibration parameters differed somewhat from other efforts in the area. For example, the infiltration capacity (INFILT) is relatively low for urban land use. The decision to use a low infiltration capacity was made for three reasons: first, assuming compacted impervious soil surfaces for developed urban areas makes empirical sense because urban soils are often highly compressed during the construction and development process. Second, these lower infiltration rates in urban areas are consistent with observations in the literature. Third, decreasing infiltration increased daily peak flow and improved model performance during the calibration period. Other approaches differed somewhat, including assigning small areas of effective impervious area to grasses and transitional areas. This decision was made because heavy compaction of surfaces by land development equipment is common, and installation of stormwater infrastructure such as collection grates in areas of low topography can make small portions of these apparently pervious areas behave more like impervious surface.

Though I calibrated the initial state variables as accurately as possible, based on advice in the BASINS technical note 6 (US EPA, 2000a), the model was run from 1990 to 1993 and the results of the first year were ignored. I felt that the approach of removing the first year's output would lead to better results using the model with different land use scenarios. The advantage of removing the first year's model output is that it allows most of the model parameters to reach a dynamic equilibrium since some of the model parameters can take months to reach equilibrium (US EPA, 2000a). Though I followed this approach of removing the first year's data throughout the calibration, validation, and simulation periods, I compared model outputs both removing the first year's output and running the model without a stabilization period. Hence, I ran the model from 1991-1993 to see if the initial calibration parameters were representative of watershed conditions and compared them to the model run of 1990-1993. I felt that this was important to confirm that our initial calibration parameters were set appropriately and that they would not have negative long-term impacts on the model run since soil storages can take months to years to stabilize. Whether the model was run from 1990-1993 or from 1991-1993, output for 1991, 1992, and 1993 were nearly identical in both model runs. The correlation coefficients between observed and modeled flow in both scenarios remained nearly constant, as did total annual flow and high flow and low flow values. Therefore, initial conditions were appropriate. Nonetheless, running the model for four years and removing the first year's output maximized confidence in the model's results for validation scenarios.

I first validated the model with the results from 1990 and 1994, the years closest to the remotely sensed land use of 1992 that were not used in the calibration. I compared the modeled versus observed annual flows, correlation coefficients, and hydrographs of these years. I then used the calibrated model to examine historical flow for three time periods: 1974-1976, 1979-1981, and 1984-1986. For these time periods, I compared observed flow, modeled flow with the static 1992 NLCD land use, and modeled flow using HLU estimates. Correlation coefficients were tabulated and annual flow and monthly flow were compared between the two model estimations and observed flow. As mentioned, in order to minimize the impact of initial conditions, the model was started 1 year before the period of time I was examining.

I used the household land use estimate of 1975 for 1974-1976, 1980 for 1979-1981, and 1985 for 1984-1986. I recognize that this is not an ideal approach; however, I merely sought to explore the feasibility and usefulness of increasing annual land use temporal resolution using this method on watershed applications. The most current version of HSPF (12) has a utility that can be utilized to allow incorporation of land use changes via an indirect method. A more ideal approach would be to use this method to adjust land use at the maximum temporal scale warranted both during the calibration and validation phases.

3.3.1 HSPF results with Household land use estimates

3.3.1.1. Calibration

Final calibration parameters for the hydrologic portion of the model follow in table 3.2.

Table 3.2[12] Select calibration parameters used in the HSPF hydrology model.

Land Use	EI	CEPSC	UZSN	INTFW	IRC	LZETP	DEEPFR	AGWETP
Water/Wetlands	0%	Varies	0.7	1.5	0.3	Varies	0.1	0.3
Urban	50%	Monthly	0.3	.5	0.3	Monthly	0.1	0
Barren/Transitional	10%	Monthly	0.3	.8	0.3	Monthly	0.1	0
Forest	0%	Monthly	0.7	1.5	0.3	Monthly	0.1	0
Pasture/Grasses	5%	Monthly	0.5	.8	0.3	Monthly	0.1	0

Land Use	LZSN	INFILT	LSUR	SLSUR	AGWRC
Water/Wetlands	8	0.07	350	0.02	0.985
Urban	5	0.01	200	0.02	0.94
Barren/Transitional	6	0.015	350	0.02	0.95
Forest	8	0.07	350	0.02	0.985
Pasture/Grasses	7	0.02	350	0.02	0.97

Overall, the calibrated hydrologic model performed well during the in-sample calibration period. Modeled flow slightly underpredicts observed flow at the USGS

[12] The following descriptions of parameters are paraphrased or exactly from the HSPF users manual (Bicknell et al., 1997).
EI- Effective impervious determines the amount of each land use that is modeled under the impervious model area
CEPSC is the interception storage capacity.
UZSN is the upper zone nominal storage.
INTFW is the interflow inflow parameter.
IRC is the interflow recession parameter.
LZETP is the lower zone evapotranspiration parameter.
DEEPFR is the fraction of groundwater inflow which will enter deep (inactive) groundwater
AGWETP is the fraction of remaining potential E-T which can be satisfied from active groundwater storage if enough is available (applicable to water, marshes & wetlands).
LZSN is the lower zone nominal storage.
INFILT is an index of the infiltration capacity of the soil.
LSUR is the length of the assumed overland flow plane.
SLSUR is the slope of the overland flow plane.
AGWRC is the basic groundwater recession rate.

gauge station, but there is strong correlation these flows. As can be seen in Figure 3.8, modeled flows tended to slightly underpredict peak storms (0.826x+6.197). The x coefficient represents relative increase in the quantity of predicted to observed flow. The high correlation coefficient 0.896 (R^2 = 0.803, p<.0001) indicates significant predictive ability of the model for hydrologic simulation. The model slightly underpredicts flow for the highest 2% volume of flows, tends to slightly overpredict for 2% to 30%, underpredict from 30 to 98%, and overpredict from 98 to 100% (Figure 3.9). There is a slight overprediction of low flows during the August to September months in 1991 and somewhat in the same period in 1993 (see Figure 3.10). Monthly calibration parameters were not adjusted, however, because the model performed well during this time period in 1992. It is possible that evapotranspiration losses during these late summer months are underestimated. Modeled versus observed total annual flow volume also compared well during this time period (see Table 3.3), well below the commonly used 10% difference threshold; however, all three years slightly overpredicted total volume.

Table 3.3. Annual Difference between Observed and Modeled Flow

Year	Percent Difference
1991	2.09%
1992	1.38%
1993	5.48%

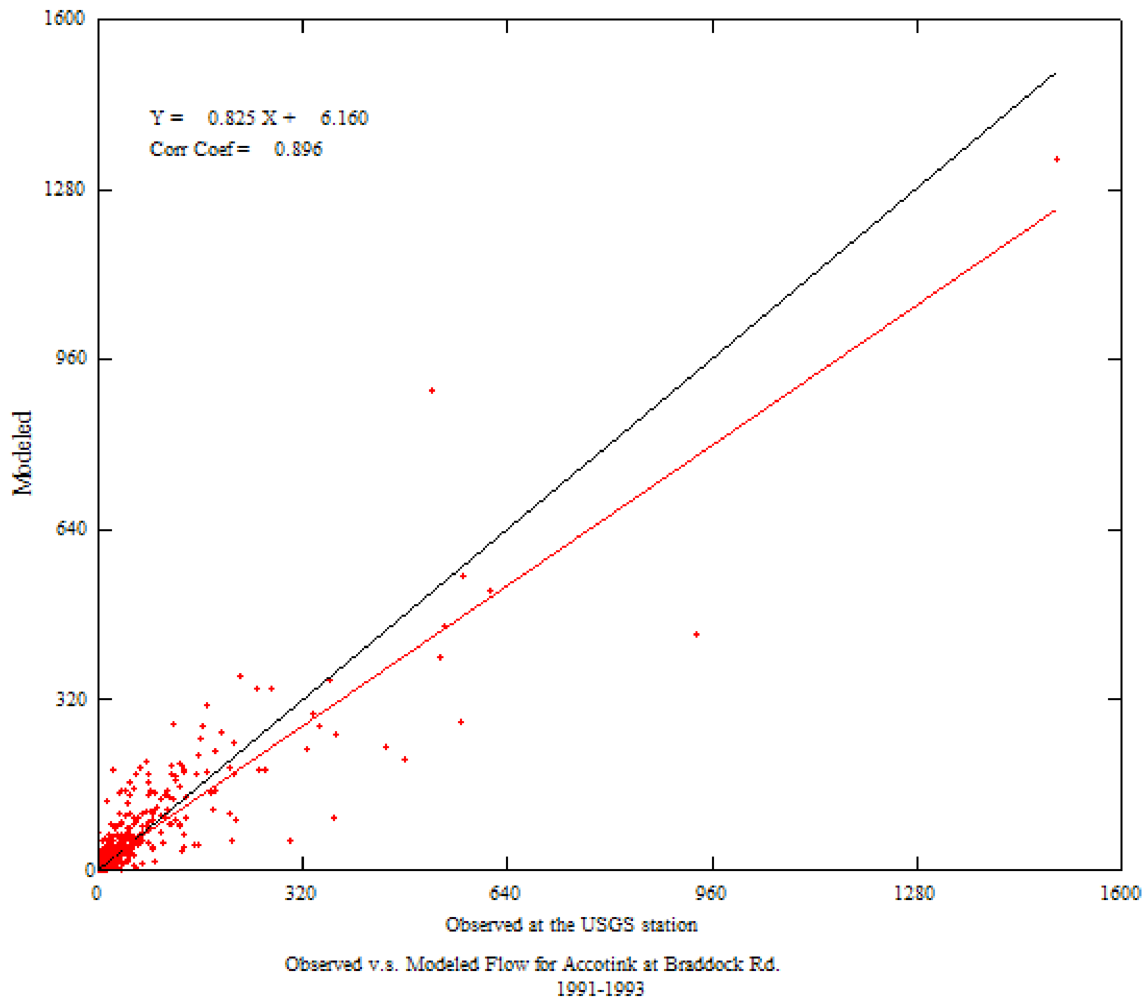

Figure 3.8. Modeled vs. Observed Flow for Accotink Creek 1991-1993. (Black line indicates equality between model results and observed data, red line indicates regression line).

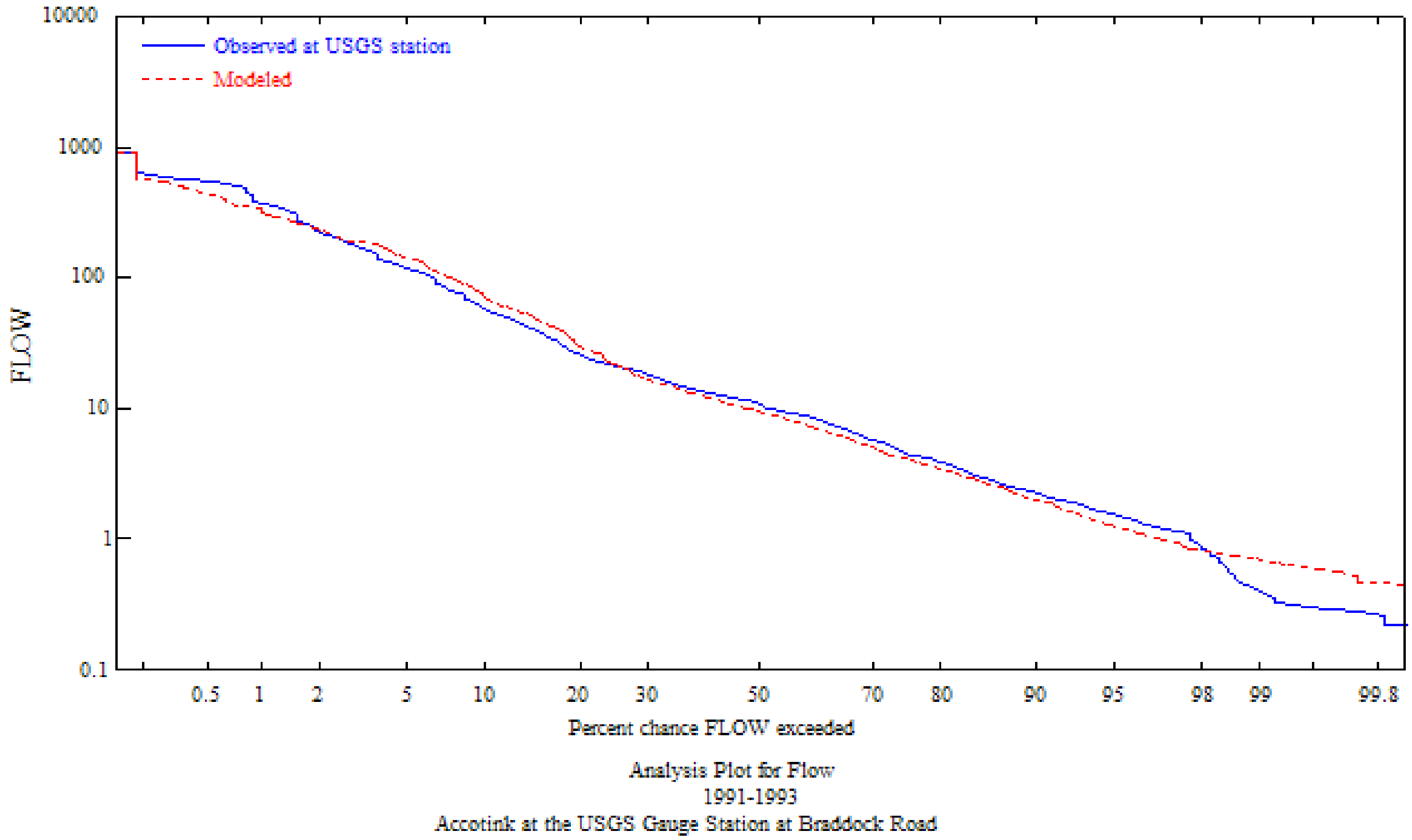

Figure 3.9. Flow duration curve of modeled (red-dashed line) vs. observed (blue-solid line) flow. A flow duration curve is a cumulative density distribution of daily flow volume.

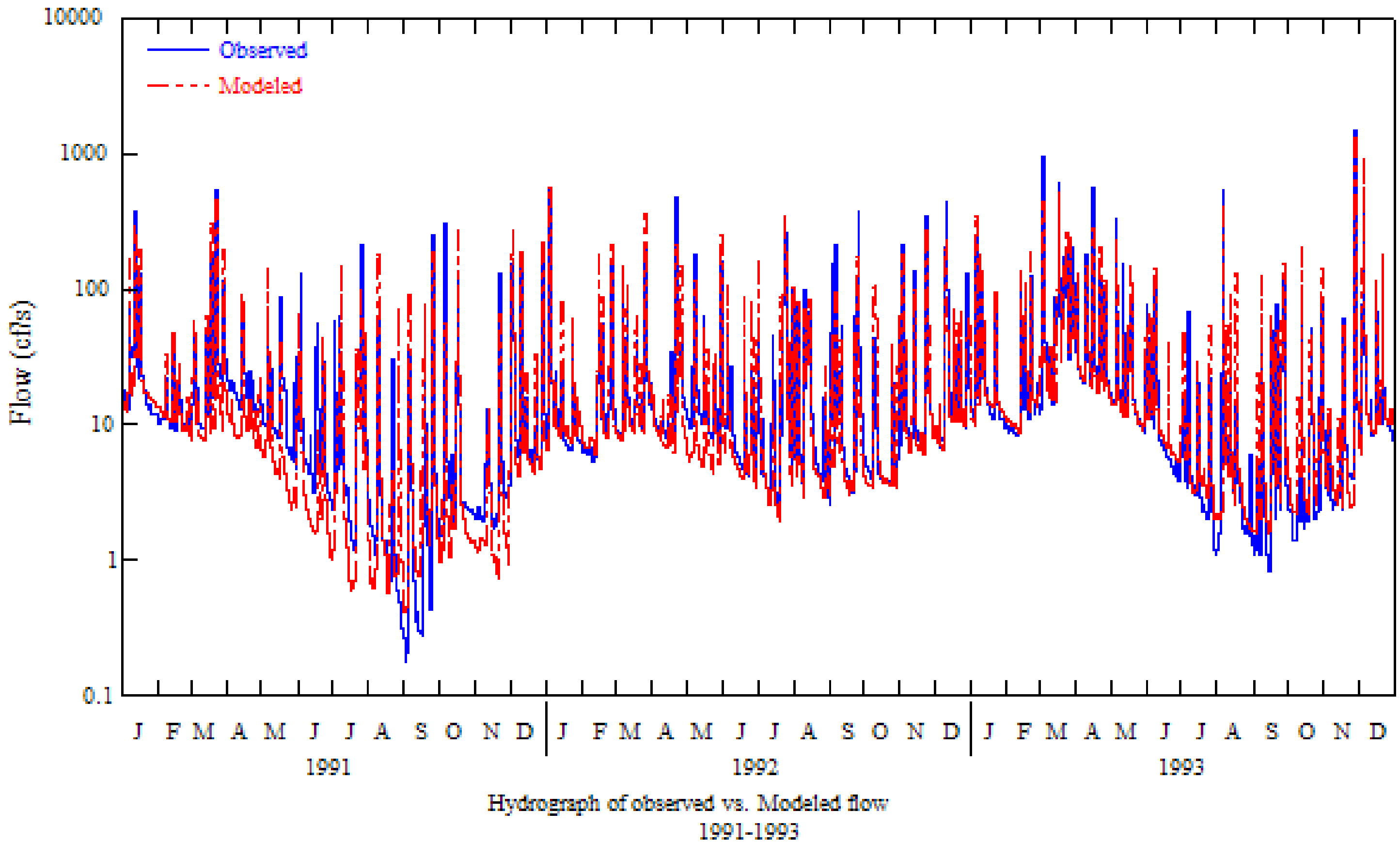

Figure 3.10. Hydrograph of modeled (red-dashed line) v. observed (blue-solid line) flow.

A significant source of scatter is caused by occasional high flows being modeled where not observed and high flows being observed where not modeled. These complications are likely due to the distance of National Airport (site of the climatic data) from the watershed. I compared predicted flow, observed flow, and precipitation, and noted absence of precipitation or precipitation higher than expected based on observed flow on several occasions. At least four occasions where only small precipitation events occurred but high observed flow was measured were noted at the USGS gauge station; hence, one can speculate that higher precipitation actually occurred in the watershed during these time periods, and these irregularities account for some of the model underperformance. The Northern Virginia region often has isolated thunderstorms that result in heavy precipitation in localized areas, primarily during the summer months. During the calibration period, there were 28 days of significant model underperformance, (defined here as the flow differing more than fivefold between observed and modeled daily flow); 26 of these events occurred between the 28th of May and the 15th of October, or the time period most likely to have isolated thunderstorms.

Figure 3.11 shows modeled and observed flow compared to precipitation from August 1st to September 30th, 1991. A likely thunderstorm (.081 inches in one hour) occurred at National Airport on August 9th. The model predicted an average daily flow of 179 cf/s, whereas the observed average daily flow was only 50 cf/s. Much of the precipitation that fell at National Airport likely did not fall in the Accotink watershed above Braddock. A similar event occurred on August 27th, 1991, when modeled flow was estimated at 71.5 cf/s and observed flow was only 0.5 cf/s. Quite clearly, observed precipitation at National Airport did not fall in the Accotink watershed. Conversely, on August 20th and 21st, observed flow exceeds modeled flow by more than seven fold. As evidenced by the precipitation graph, little precipitation was recorded at National Airport; however, precipitation likely fell in the Accotink watershed. Finally, a storm from the 24th to 26th of September was modeled reasonably well, indicating that similar quantities of precipitation likely fell at both National Airport and in the Accotink watershed.

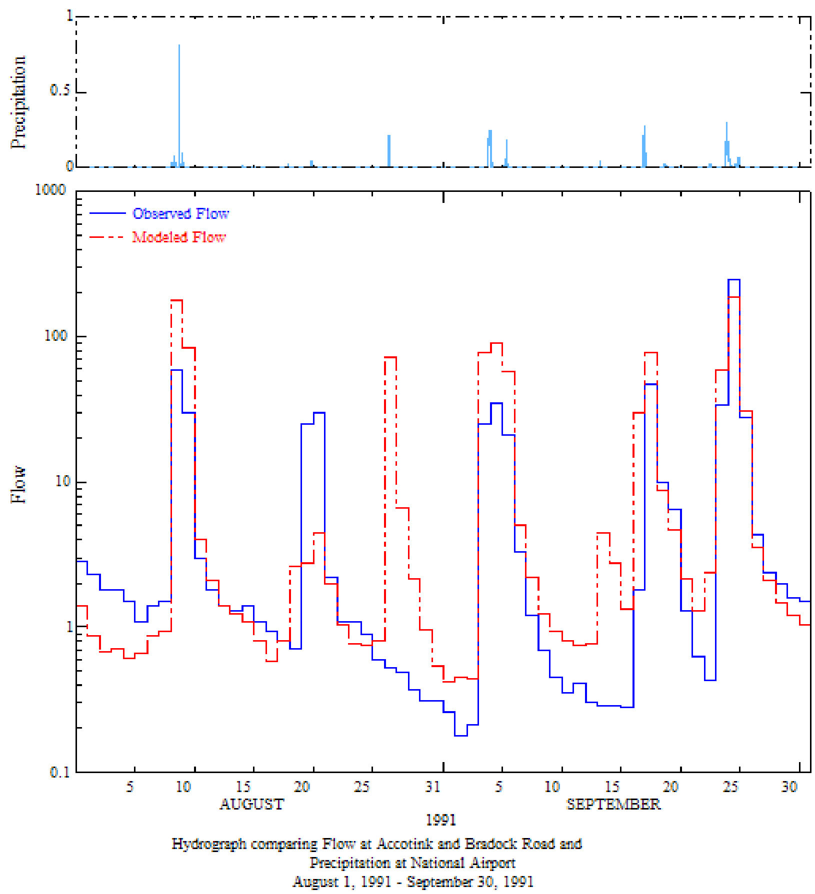

Figure 3.11. Hydrograph of modeled (red-dashed line) v. observed (blue-solid line) flow compared to precipitation from August 1st, 1991 to September 30, 1991. Note that precipitation is plotted on an hourly basis whereas flow is plotted on a daily basis. Hence, a value of 0.8 inches is a hourly precipitation rate.

3.3.1.2. HSPF Validation

The model performed well in both 1990 and 1994, years used for out of sample validation (not used during calibration). Validation was completed with the 1992 NLCD land use data set. The difference between the modeled and observed annual flow was less than 10% for each year (see table 3.4). The model appears to significantly underestimate large flow events in 1990 (.501x+20.132; R^2 = .423, P<0.0001), despite the annual flow being reasonably comparable. However, after further exploration, a single missed precipitation event caused the depression in the slope of the regression and depression of the R^2. If the high flow observed event is excluded, the model performance is adequate (.924x+8.6975; R^2 = .612, P<0.0001). As shown in Figure 3.12, the model predicted average intensity of storms well in 1994 and had a higher correlation among daily flows (1.037x+.461; R^2 = .736, P<0.0001), although there was at least one major flow event modeled with substantially lower flow observed and vice versa. This is likely due to issues with precipitation differing between the watershed and the rain gauge station as discussed above.

Table 3.4. Annual Difference between Observed and Modeled Flow in Validation Scenarios

Year	Percent Difference
1990	9.13%
1994	4.73%

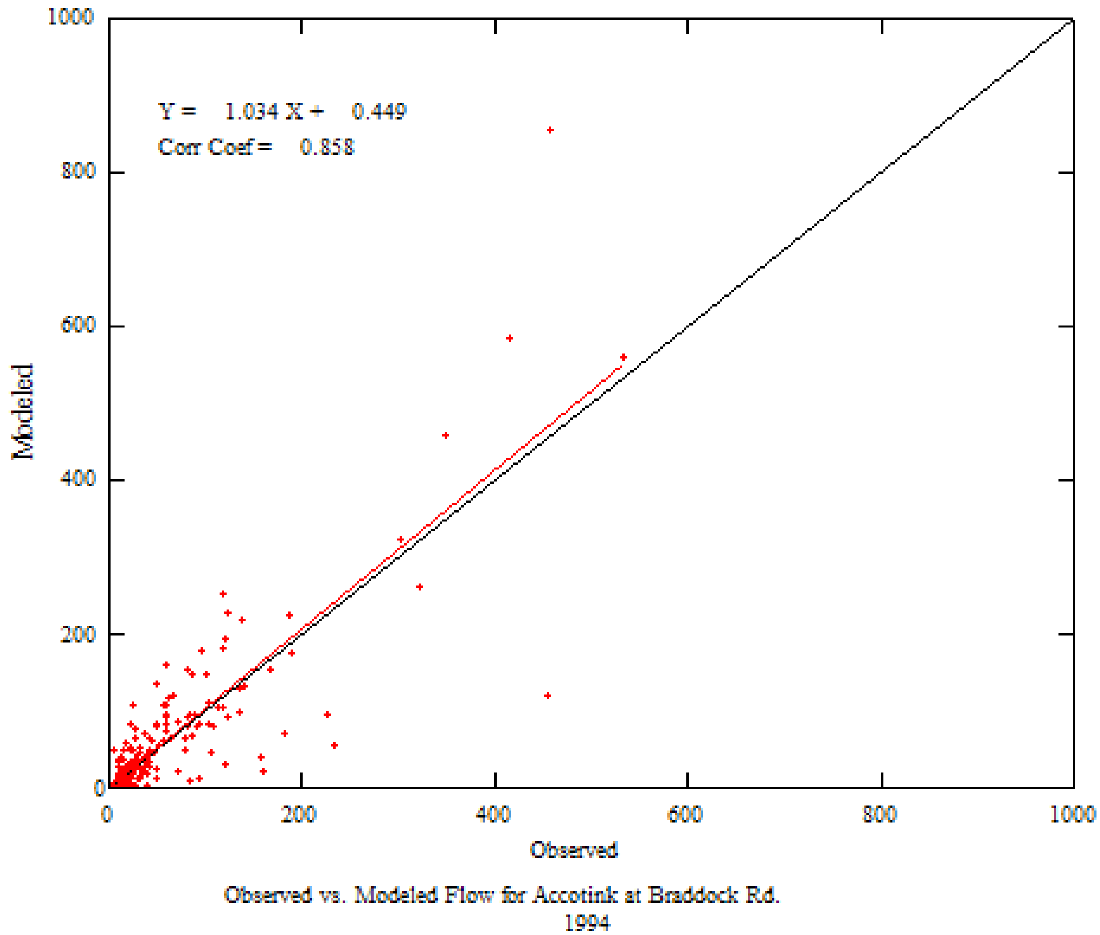

Figure 3.12. Modeled flow vs. Observed Flow in the validation scenario 1994 using NLCD 1992 land use data. (Black line indicates equality between model results and observed data, red line indicates regression line).

3.3.2 HSPF model results using NLCD data compared to household method land use estimates.

Model performance was compared in for time periods from 1974-1976, 1979-1981, and 1984-1986 using the HLU estimates and the static NLCD 1992 land use data. Running the HSPF model using NLCD data was not as accurate in years further removed from the calibration time period. This is because the NLCD land use data were static and not as representative of land use data during simulation time periods. The primary concern was an overestimation in annual flow for multiple years. However, there was still a reasonable correlation coefficient between all modeled and observed flow events.

Use of the Household Land Use (HLU) estimates versus the NLCD remotely estimated values increased HSPF model performance for flow during the 1974-1976, 1979-1981, and 1984-1986 time periods (table 3.5). In all scenarios modeled, the R^2 between the observed and modeled flow was somewhat higher with the HLU estimates than with the NLCD data and can be considered in a fair to good range[13]. Furthermore, HLU modeled annual flow was closer to observed annual flow in 7 of 9 years than the NLCD flow (table 3.5). The average annual volume was within 10% of the observed volume in only 2 of the 9 years using the NLCD data. In contrast, it was within the 10% range 5 of 9 years using the Household method land use data.

[13] As a reminder, the R^2 for observed versus modeled flow is negatively impacted by the distance to the observed precipitation gauge station. This R^2 could be improved by using precipitation data closer to the sites or triangulating between available precipitation stations.

Table 3.5. Percent difference between modeled and observed annual flow using HLU and NLCD 1992 land use scenarios. Note the R^2 corresponds to daily flow for the three-year time period run. P values are less than 0.0001 for all runs.

		HLU		NLCD (1992)	
HLU year Used	Year	% difference	R^2	% difference	R^2
1975	1974	3.12%	0.601	22.52%	0.579
	1975	44.55%		60.37%	
	1976	12.18%		36.28%	
1980	1979	-4.57%	0.759	1.54%	0.726
	1980	-4.73%		2.99%	
	1981	4.52%		19.53%	
1985	1984	7.29%	0.608	10.10%	0.599
	1985	44.72%		51.17%	
	1986	44.66%		51.78%	

The greatest difference between the two results occurred during the 1974 to 1976 model run. The actual 1975 urban land use is most overestimated by using the NLCD 1992 data for this model run (22.4 km^2 estimated urban land use in the HLU scenario and 32.3 km^2 estimated urban land use in the NLCD scenario). For both model scenarios, the calibrated model overestimated observed annual flow and peak flow; however, modeled flow was substantially closer to the observed flow in the HLU scenario. In comparing Figure 3.13 to 3.14, one can note the higher percentage of predicted flows exceeding observed flows and the lower correlation coefficient in the NLCD scenario. Figures 3.15 and 3.16 show that the HLU scenario has peak flows substantially closer to observed values than the NLCD scenario. Furthermore, the NLCD scenario seems to capture the lowest flow events better than the HLU scenario, but at the expense of the 30-99% (mid range) flow group, which the HLU scenario models quite accurately. In short, the HLU model simulated observed conditions substantially better than the static land use data set.

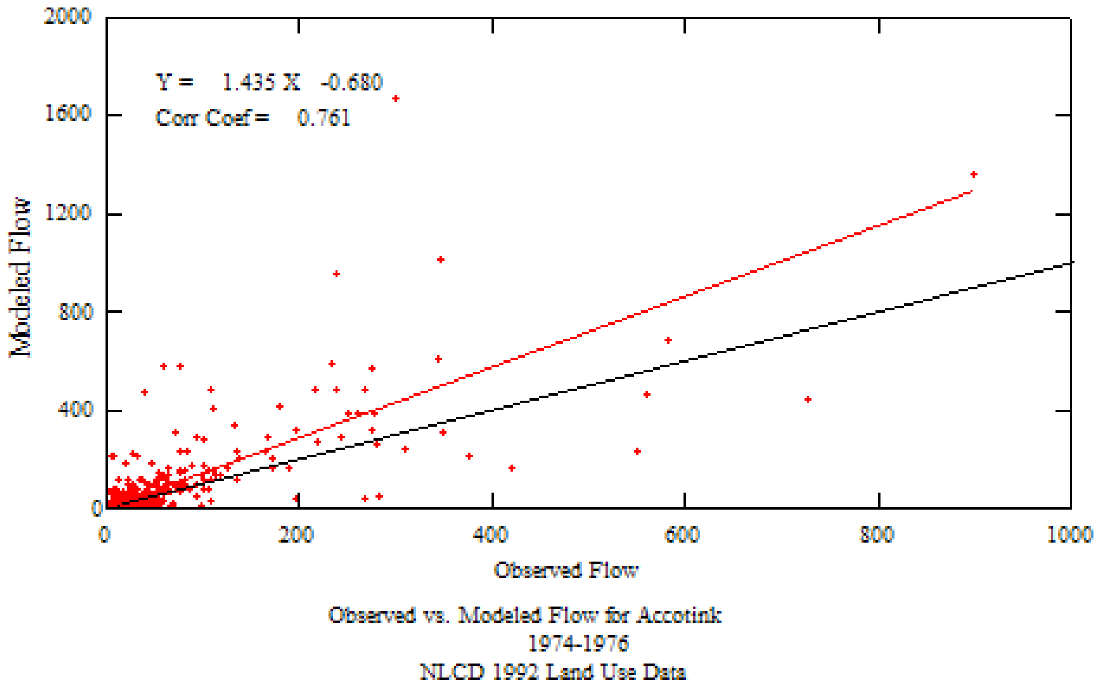

Figure 3.13 Both total annual and daily flow are significantly overestimated compared to observed flow in the NLCD land use scenario for the 1974-1975 model run. Note that the x and y axis are on different scales for easier visual interpretation. (Black line indicates equality between model results and observed data, red line indicates regression line).

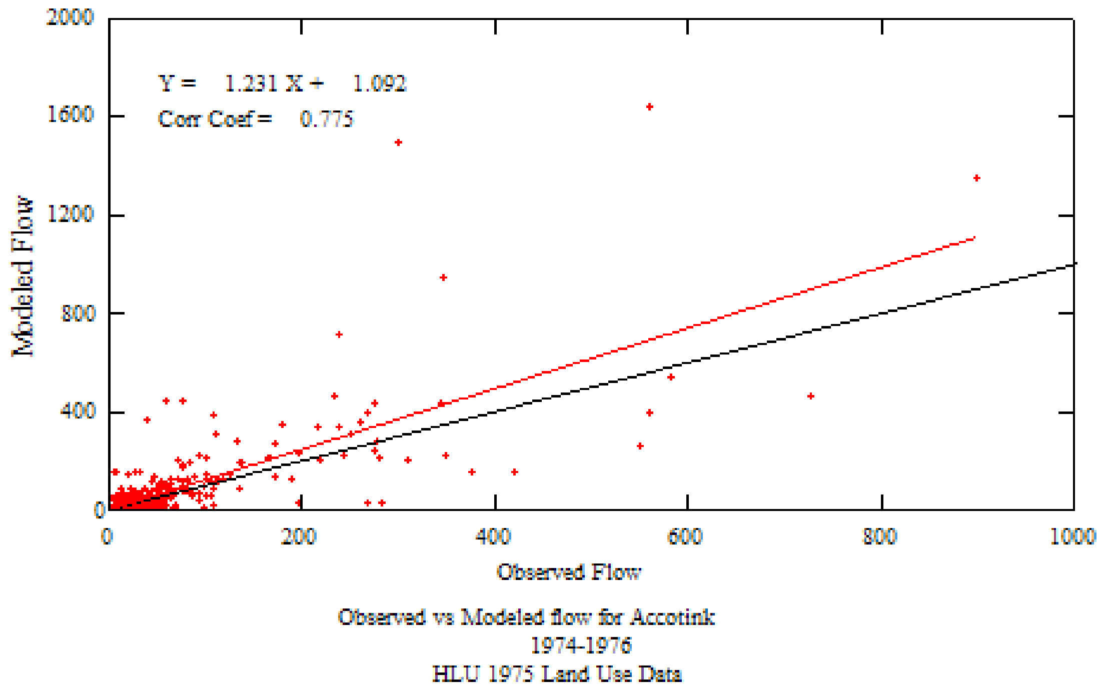

Figure 3.14. HLU 1975 scenario vs. observed flow. Note that daily flows are considerably more comparable to observed flows than in Figure 3.13. Note that the x and y axis are on different scales for easier visual interpretation. (Black line indicates equality between model results and observed data, red line indicates regression line).

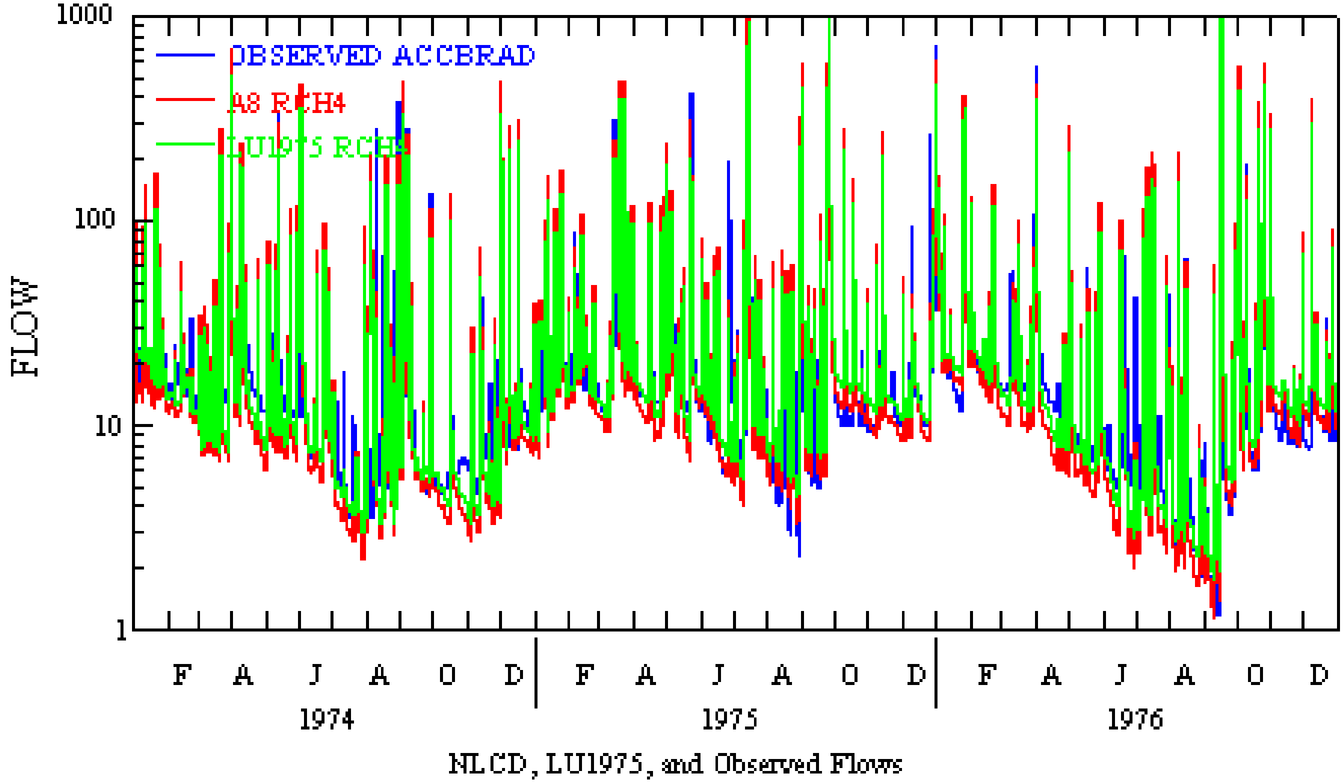

Figure 3.15 Flows for the observed flow (blue), HLU flow (Green), and static NLCD flow (red) for 1975. Except for the lowest observed flows, the HLU scenario generally does a better job simulating observed flow.

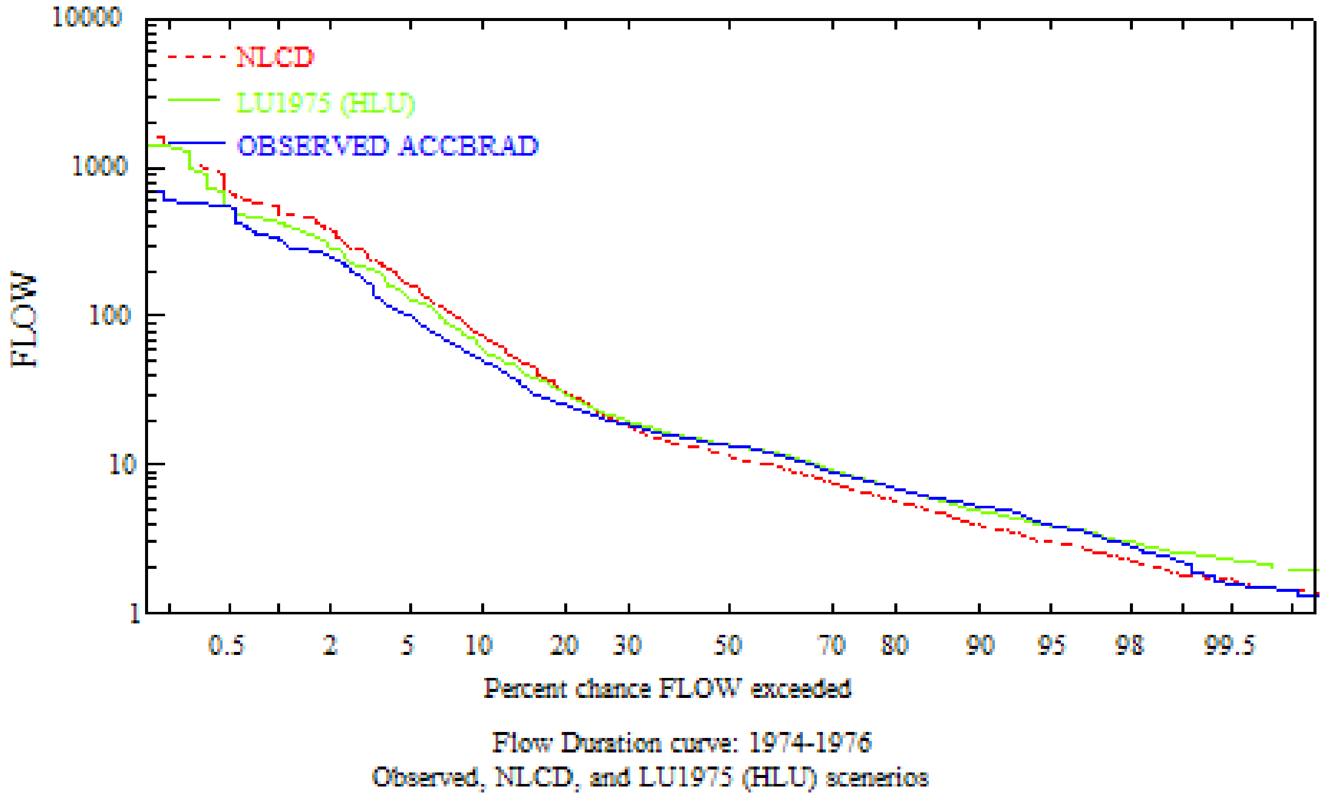

Figure 3.16. Flow duration curve showing that HLU (green dashed) is closer to observed flow (blue solid) than the static land use (red dotted) for 99% of flows.

3.4 Discussion

Inclusion of land use data improved the performance of the HSPF hydrologic model in the upper Accotink watershed. From 1975 to 1992 the watershed increased from an estimated 37.1% urban to 53.5% urban, a relative increase of 44.2%. Though significant, these changes are not as dramatic as many rapidly urbanizing watersheds. Where there are substantial rapid changes in urban land use, as in many North American suburbs and exurbs, the importance of having timely land use data is magnified. In these rapidly urbanizing areas, if one were to examine historic water quality conditions in the 1970s using 1992 data, he or she would have more difficulty finding meaningful correlations between land use and water quality.

The Pohick watershed above GMU sampling station 1 increased from 12% to 38.6% urban from 1975 to 1992, a relative increase of 321.7%. After 1992, the watershed continued to urbanize, increasing to 53.6% urban in 2004. For this portion of the Pohick watershed, the calibrated model flow output differs appreciably between the HLU 1975 land use scenario and the NLCD observed land use when run from 1974 to 1976 (Figure 3.17). Both daily peak flow and base flow are more extreme under the NLCD static land use scenario. Annual volume was estimated 29.0%, 19.1%, and 18.3% higher for 1974, 1975, and 1976 respectively for the static NLCD land use data compared with the HLU estimates. Peak flows were considerably higher, short-term interflow discharge decreased, and base flows were substantially lower when the static land use data was utilized. I propose that the HLU scenario is more representative of actual flow conditions for the watershed and that use of the static land use data set produces greater inaccuracies in model output. In summary, in order to most accurately model watershed dynamics, predict pollutant loadings, look at impacts from in-stream scouring, or estimate impacts to benthic or aquatic communities, researchers improve accuracy of their model by having the most accurate and time appropriate estimates of urban area.

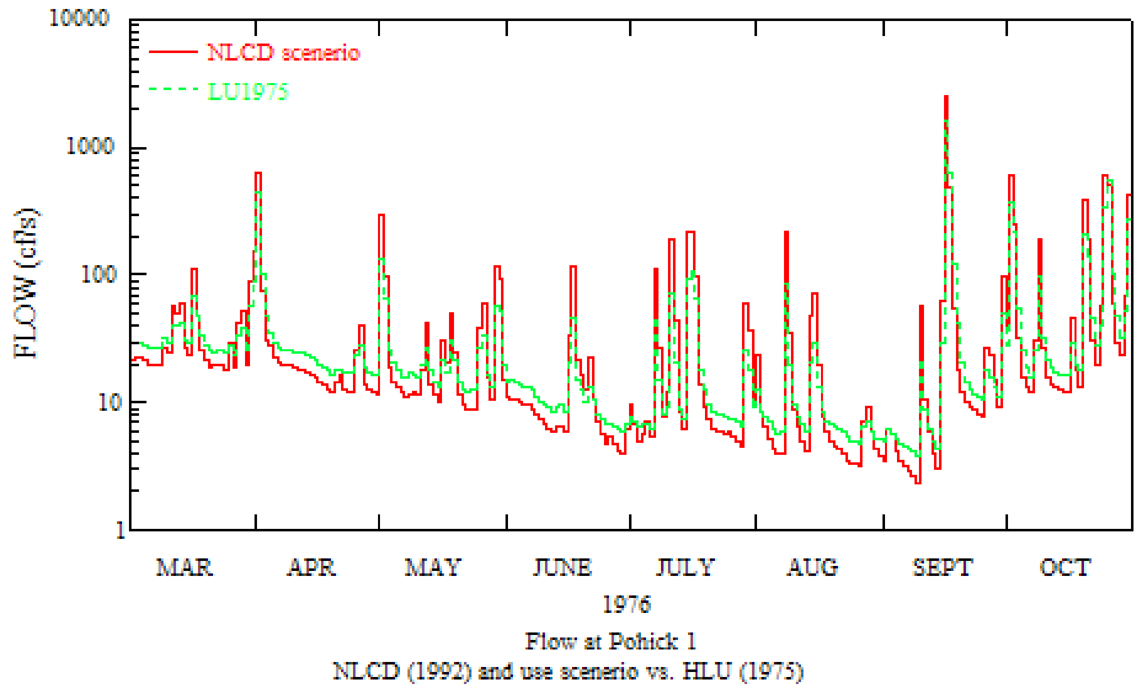

Figure 3.17. A modeled hydrograph from March 1, 1976 to October 31, 1976 for the Pohick watershed above the George Mason University Sampling Site 1.

Table 3.6.Comparison of the ten highest daily peak flows and 10 lowest daily base flows for the NLCD (static) land use output compared to HLU output. The NLCD peak daily flow is 29.9% higher on average than the HLU, while the base flow is 48.9% lower on average. Hence, the NLCD model output is likely over-predicting the hydrologic impacts of urbanization for this time period.

Modeled Peak Flow and Low Flow at Pohick 1					
10 Highest Peak Flows (cf/s)			10 Lowest Daily Flows (cf/s)		
NLCD	HLU	Difference	NLCD	HLU	Difference
2530	1590	37.2%	3.5	5	-42.9%
2270	1850	18.5%	3.5	4.7	-34.3%
2170	2130	1.8%	3.3	5	-51.5%
1560	1320	15.4%	3.3	5	-51.5%
1050	511	51.3%	3.1	4.8	-54.8%
812	505	37.8%	3.1	4.5	-45.2%
812	706	13.1%	3.1	4.3	-38.7%
677	361	46.7%	2.9	4.4	-51.7%
666	340	48.9%	2.6	4.1	-57.7%
634	454	28.4%	2.3	3.7	-60.9%

If a TMDL were to be conducted on Pohick using the land use data from 1992, the resultant model output would grossly underestimate the contribution from urban land use. Substantial resources would be put into TMDL development and environmental management decisions would be based on these reports. Not incorporating the impacts of land use change could have significant implications for accuracy of the output, assigned waste load allocations (WLAs), regulatory decisions, and allocation of resources and responsibilities. Hence, if land use changes, and consequently changes in loadings were not taken into account, wasteload allocations (WLAs) would not appropriately reduce overall loadings to maintain or improve aquatic health. Therefore, it is crucial that accurate, timely land use estimations are used in areas that are experiencing rapid land change. Ideally, TMDLs should attempt to anticipate potential alterations in scenarios; maximum loads that an aquatic ecosystem can tolerate are fairly constant.

The methods proposed in this paper for estimating land use may not be as exact for determining historic land use as using frequent historic remote sensing images and classifying those images into appropriate land use categories. For instance, in recent years, as the watersheds have started to have more in-fill development which have higher density per household, the methodology of estimating area under construction and barren area may not be as accurate. However, the reality is that very few high quality classified remote sensing land use data sets are available and the budgets of regulatory authorities are often limited. In watershed applications, most researchers and practitioners rely on commonly available, pre-classified land use sets such as NLCD (1992) or MRLC (2000). The approach utilized in this paper can increase accuracy of land use estimates in rapidly changing watersheds in which remote sensing data is not available for the desired time of analysis. This approach can also be used to examine the impacts of land use change on water quality. In another example of potential applications, the estimates generated here have been used to examine water quality data at different sampling sites throughout the Accotink and Pohick watersheds and were found to increase the understanding of watershed dynamics. Consequently, combining demographic data with classified remote sensing data sets is a resource-friendly approach to increasing the tools available for scientists and practitioners who rely on land use data at fine temporal scales.

3.5 Conclusion

Use of the household method appears to be an accurate approach to increasing the temporal land use estimates in areas where there is good demographic data. These data are then easily exportable for other analyses. This study shows that using these estimates in combination with the calibrated hydrologic component of the HSPF modeling framework increases the correlation coefficient between predicted and observed flow in validated data sets. Furthermore, use of these data result in more accurate simulation of annual flow volume. As part of this study, these land use estimates are later used in statistical analyses to compare a historic water quality data set to land use data. In short, this household method appears accurate at the watershed scale and it can reasonably argued that the method gives valid results for use in watershed analysis.

4. Accotink and Pohick Watershed and Water Quality Analysis

4.1 Introduction

Accotink and Pohick watersheds have experienced significant urbanization over the last 50 years. Accotink, the watershed closer to Washington DC, experienced peak urbanization from the 1950s to the 1990s, with urbanization occurring first in the headwater areas in Fairfax City and Vienna and later in the southern part of the county. The Pohick watershed experienced peak urbanization from the 1980s to the present, although significant urbanization occurred earlier in the headwaters near Fairfax City. The urbanization in both watersheds has significantly impacted stream hydrology, geomorphology, water quality, and loadings to downstream receiving waters.

Urban land use tends to increase runoff volume; increase toxic, nutrient, and sediment loadings; increase temperature; reduce habitat quality, water quality, and groundwater recharge; decrease base flow; and lead to increased channel erosion and stream channel widening (Booth, 1990; Davis *et al.*, 2001; Hogan, 2005; Jones & Clark, 1987; Nelson & Booth, 2002; Schueler, 1994). In urban watersheds, runoff episodes are likely to be more intense, nutrient loadings are likely to be higher, and water bodies are likely to be more polluted than in forested watersheds (Wahl *et al.*, 1997).

These profound changes are caused by the modification of the land surface, changes in impervious cover, increased loadings of nutrients, increased use of pesticides, increased toxic spills in the watershed, decreases in native vegetation, and reduced soil stability. Urbanization also impacts aquatic water bodies further downstream from the urbanized area. Multiple studies have shown that urban runoff contributes to the impairment of large rivers, lakes, estuaries and offshore ecosystems (Bay *et al.*, 2003; Hollanda *et al.*, 2004; United States Environmental Protection Agency, 1997b).

Accotink and Pohick are both small, low order streams: though Accotink has a larger watershed area, it is a 3^{rd} order stream, while Pohick is a 4^{th} order stream. Low order streams are influenced by many physical and biological factors; for example, the slope of the watershed influences the direction, speed, and amount of surface runoff. This, in turn, affects habitat, erosion, sedimentation, substrate, hydroperiod, and the riparian and terrestrial environment surrounding the stream. Hence, low order streams are very much affected by their terrestrial environment, which is why they are sensitive to urbanization.

Disturbed streams such as Accotink and Pohick differ significantly in their appearance than more 'natural' or undisturbed streams. In undisturbed conditions, low order streams tend to have dissolved oxygen (DO) near saturation because of the consistent contact of the stream with the air. Unless exposed to solar radiation, they normally do not have high peaks in DO during the day (and as a result, it is rare for them to have a high pH). Low order streams very rarely have anoxic conditions for the reasons described above unless there is a substantial anthropogenic impact. Rainwater input highly influences pH, the productivity of the water, and the alkalinity of the water.

Extremes in pH levels can occur in smaller order streams due to lack of buffering ability and acid rain (Fairfax County's rainfall is near a pH of around 4) or high productivity (pH can reach over 9 in hypereutrophic ponds). Alkalinity in streams is highly influenced by the amount of limestone in the watershed. Smaller watersheds and, consequently, smaller streams, more often vary widely in alkalinity. Additionally, toxic chemicals can be a problem in some lower order streams if there is a discrete source of polluted runoff entering the system.

Nutrient availability is highly variable in lower order streams and depends upon land use in the watershed, presence of point sources, historic watershed uses, and stream function. In streams such as Accotink and Pohick, nutrients tend to come from the terrestrial ecosystem or from external inputs. For subwatersheds with intensive urban or agricultural land use, nutrient concentrations can be diluted from other subwatersheds with less urban or agricultural land use. Nutrient concentrations can also be more elevated in these subwatersheds than in larger watersheds in which they discharge. Nutrient concentrations are sometimes seasonally affected in lower order streams due to changes in vegetation and climate (for instance dropping of leaves into the streams) and human impact (fertilizing at the start of the growing season).

As lotic aquatic ecosystems, streams loosely spiral nutrients downstream. Some nutrients are sequestered as they settle into the sediment (as in the case of phosphorous) or undergo denitrification (as in the case of nitrogen). However, in streams such as Accotink and Pohick, anything stored in the sediment is likely to be resuspendend in significant precipitation events. Both streams drain into Gunston Cove, whose waters ultimately make their way to the Chesapeake Bay. Correll (1987) estimates that 65% of annual nitrogen and 22% of total phosphorus comes from ‘land discharges’ (all terrestrial pollution excluding traditional point sources such as sewage treatment facilities or publicly owned treatment works (POTWs)). Tuttle et al. (1987) state that anoxic conditions were due in large part to increasing nutrient loadings. Hence, for the Accotink and Pohick watersheds, nutrients and other pollutants discharged from the mouth of the stream likely impact the economically important and environmentally threatened Chesapeake Bay.

This chapter contains a thorough analysis of water quality conditions in both Accotink and Pohick. The study examined historical sets, land use, point source discharge history, comparisons between these streams, and numerous existing sources of information to determine the primary stressors to Accotink and Pohick and downstream receiving waters, focusing on total suspended solids (TSS), nitrate, and flow. TSS was monitored because sediment is a common pollutant associated with urbanization. Furthermore, many pollutants sorb to sediment; hence higher sediment loadings can be indicative of higher loadings of other pollutants, including phosphate and toxics. Flow was analyzed because numerous studies have shown that changes in flow are the greatest threat to low order urban streams and because a few pollutants such as nitrate move in a dissolved state. Nitrate was monitored because it is one of the primary targets for reduction by the Chesapeake Bay Program and is the primary limiting nutrient in much of the Chesapeake. Furthermore, many BMPs are designed with reduction of nutrients in mind and significant resources are spent attempting to prevent the introduction of nitrogen to receiving waters. The hypothesis of this chapter is that there is distinct spatial variation in water quality parameters due primarily to differences in levels of urban land use. Furthermore, one would expect to see changes in flow, nitrate, and TSS discharges as the watersheds urbanized.

4.2 Methods

4.2.1 Monitoring, Flow, and Water Quality Analysis

4.2.1.1. George Mason University Monitoring Sites

The study area encompasses two Mid-Atlantic watersheds located near Washington DC (Figure 4.1). Monitoring sites were selected primarily due to the availability of historic data sets. Two of the sites, 1 and 2, directly overlap with previous George Mason University field stations on Pohick Creek. Site 1 is located above the Noman Cole sewage treatment facility and is influenced primarily by nonpoint source pollution and stormwater. Site 2 is located below that facility and is heavily influenced by discharge. George Mason University had a site on Accotink Creek (site 13) near the creek's outlet to the Gunston Cove. Due to problems accessing this site, an alternate site was selected upstream (site 13m). The fourth site, site 20, is located at the USGS gauge station. This site was added because of the

Figure 4.1. Landsat image of the Potomac River bisecting Virginia from Maryland, with the study area

location of the USGS flow station. Key characteristics of these sites are summarized in Table 4.1. Other sites were sampled sporadically, including several samples from a site at Accotink at Prosperity Rd. (site 21), and several grab samples were made at other sites on Accotink, Pohick, and from a construction site.

Figure 4.2. Taking a grab sample at Site 13M during the 2005 sampling season.

Field sampling consisted of monitoring these sites once every two weeks from March through December 2005. Additional monitoring days were added to sample wet weather events. Sites 1, 2, and 13M were monitored a total of 24 times in 2005. Site 20 was monitored 21 times and site 21 was monitored 3 times. Furthermore, sites 1, 2, 13m, and 20 were monitored along with other Fairfax County streams from February to December 2006 as part of another student's Masters thesis. This student also monitored immediately upstream of the old GMU 13 site (adding a fifth sampling point). These data were used to support this work's analysis.

Table 4.1. Key characteristics of sampling sites and the full Accotink and Pohick watersheds.

Site	Watershed Area (km^2 draining to point)	% Urban (1992 NLCD/ HLU)	% Urban (2004 HLU)	# Samples in 2005	Other Notes
1	82.0	38.6%	53.6%	24	
2	83.8	39.0%	53.9%	24	Below sewage treatment facility
Pohick All	92.8	36.3%	49.5%	N/A	
20	60.3	53.5%	65.4%	21	At USGS Gauge Station
13M	104.8	52.7%	59.3%	24	
13G	129.4	51.8%	63.2%	N/A	
Accotink All	134.2	51.0%	61.9%	N/A	

4.2.1.2. Sample Collection, Preparation, Analysis

Data collected at each sampling site included pH, conductivity, and temperature using a Hydrolab Minisonde (Hydrolab Corporation, 1997). The MiniSonde was calibrated for these parameters at least once every two weeks. Additionally, a larger Hydrolab DataSonde had to be used when the MiniSonde had reliability issues. The DataSonde functioned in the same way as the MiniSonde and was calibrated once every two weeks. Additionally, in order to prepare for sampling, bottles were cleaned and washed with hydrochloric acid (HCl) to remove any residual nutrients or sediment from bottle walls. These bottles were rinsed, dried and sealed until sample collection.

Other parameters monitored included total suspended solids (TSS), volatile suspended solids (VSS), nitrate-N, ammonia-N, total phosphorus, and soluble reactive phosphorus (SRP). All samples were analyzed using methods from Wetzel and Likens (2000). Total suspended solids (TSS) was analyzed gravimetrically. A glass fiber Whatman 4.25 cm filter was wrapped in a small piece of aluminum foil, predried and tared. A premeasured volume of well-mixed sample was filtered through the glass fiber filter using a vacuum apparatus. The filtrate was saved for the analysis of nitrate-N, ammonia-N, and SRP. Filtration was completed in the field. In the laboratory, the filter was then dried at 100^{o}C and reweighed. TSS was then calculated as the difference in weight between the tared filter and the dried filter divided by the volume of filtered.

Volatile suspended solids (VSS) was derived by incinerating the filter inside the aluminum foil at 500 degrees centigrade for one hour. The filter was then reweighed. Volatile suspended solids was calculated by the difference between the weight of the dried filter and the ashed filter divided by the sample volume run through the filter. VSS serves as an indicator of suspended particulate organic matter.

Nutrients were analyzed in the lab using acid washed glassware according to standard analytical techniques. All samples were prepared and measured using techniques and methods from Wetzel and Likens (2000). Using the filtered samples, nitrate was analyzed in the lab using cadmium reduction via the Hach NitraVer 5 method. Ammonia nitrogen was analyzed using the Solarzano method using filtered samples. Orthophosphate was determined by using the ascorbic acid method. Total phosphorus was analyzed by using persulfate digestion in an autoclave to digest total phosphorus to orthosphospate. The sample was then filtered through Whatman 16 cm filters in acid washed funnels to remove particles and then analyzed using the ascorbic acid method. Results of tests were stored in Excel spreadsheets. All test results were stored as individual tests and as final results by date.

4.2.1.3. Fairfax County Nitrate Data

The Fairfax County Health Department collected nitrate, total phosphorus, and fecal coliform data at many county streams from 1986-2002 (Fairfax County Health Department, 1986-2002). The total phosphorus data were considered unusable because the detection limit of the tests used was not sensitive enough for the majority of samples. The nitrate data, on the other hand, appeared as if they might offer additional insights, but must be considered carefully due to relatively high detection limits. These data were examined to see if there were any spatial or temporal trends between the sites, primarily through visual inspection using box plots and the Kruskal-Wallis test. A subset of these nitrate data were also used to validate the nitrate results from the LOADEST loading model.

4.2.1.4. Nitrogen Deposition

Nitrogen in the form of nitrogen oxides is mainly introduced into the air from the burning of fossil fuels such as coal, oil or gas. Ammonia is released from industrial activities and livestock, particularly animal feeding operations such as feedlots and dairy farms. Air deposition of this nitrogen accounts for a significant portion of nitrogen loadings to aquatic bodies. The Chesapeake Bay Program estimates that 25% of all nitrogen loadings into the Bay come from wet and dry deposition (Chesapeake Bay Program, 2005; Fisher & Oppenheimer, 1991). For this study, we wanted to explore whether increased nitrogen deposition would play a major role in increasing nitrogen export from the watersheds.

Nitrogen deposition data for the mid-Atlantic region were obtained from US EPA's Clean Air Markets CastNet database (US EPA, 2006a). These data were available for two locations in the mid-Atlantic region: Beltsville Maryland (to the east), and Shenandoah National Park (to the west) for the years 1989-2005. Data examined included total deposition, wet deposition, and dry deposition for nitrogen, although not all data were available for all years. Results were plotted and a linear regression was used to examine if there were any trends. Research was also sought that cited older nitrogen deposition estimations to attempt to explain variance in older data. Furthermore, a literature search of the scientific and policy literature was conducted to see if results over these past years are consistent with results plotted.

4.2.1.5. Flow

Flow data are available for the upper portion of Accotink for 1947-2006 at USGS Gauge Station 1654000. The USGS considers the data provisional until 18 months after their collection. Flow was transformed using the natural log and normality was examined. Total annual flow was plotted against time for 1948 to 2006 and percent urban land use for 1975-2004 to examine trends in flow volume. For these charts, total flow volume was converted to km^3 of water per year. Untransformed flow data were plotted on a scatterplot to examine changes in flow patterns for the time period in question. The frequency of high flows and low flows over time were examined by establishing a high flow threshold (100 cf/s) and a low flow threshold (2 cf/s) for the period of study, and counting the number of days in the year that the data were above or below these values. These thresholds were defined by examining the flow rate at which approximately 5% of all flows exceeded a flow rate or where approximately 5% of all flows were below a given flow rate. The data were presented in two ways: first, the sum of the count of high flow days and low flow days were aggregated, to de-emphasize the importance of wet flow vs. dry flow years. Secondly, the data were disaggregated to examine trends more closely.

Flow data were used as the basis for estimating loads and comparing concentrations of water quality parameters across the watersheds. The natural log of flow was found to be normal. In order to have a coarse estimate of the flow coming from downstream reaches, a ratio of the relative watershed area of downstream reaches over the watershed area of the USGS gauge station was multiplied by the flow. For the site located below the sewage treatment facility, the flow discharge reported for the Noman Cole plant were added to the estimated in-stream flow using this watershed area ratio method to determine flow downstream of the Noman Cole. This approach likely overestimated flow in early years of this analysis, since relative urbanization of the lower parts of Accotink and almost all of Pohick were significantly lower. It also may overestimate current total flow because Best Management Practices have been more widely used in the last 20 years of development. Additionally, the hydrograph further downstream would react somewhat differently to precipitation events. However, this method should be sufficient for examining loadings and comparing water quality parameters in the watersheds to each other and across time.

Flow data were also used in the calibration of an HSPF hydrology model discussed in chapter 3. That discussion explored the impact of urbanization on flow, to which these results will refer.

4.2.2 Constituent Data Analysis and Loading Calculations

Samples were compared both temporally and spatially. Temporal results were plotted in box plots and statistical analysis was done using the Kruskal-Wallis nonparametric test to test for differences in nitrate-N, ammonia-N, total phosphorus, SRP, TSS, and VSS. For the purposes of this analysis, site 13M was compared to the original site 13, which has limitations that will be discussed. Many parameters, particularly nutrients, have been shown to have variations in seasons due in large part to human influence and photosynthetic activity. The seasonality of nitrate was examined by separating nitrate samples into their respective seasons for each year and plotting against flow.

For 1983-1992 George Mason Data, flow data were ranked from most flow (100%) to least flow (0%). Nitrate concentrations were plotted against the relative rank of flow for each season, examined empirically, and summary statistics were generated. The same was done with 2005 data. Secondly, samples were compared across seasons using the Kruskal-Wallis test.

Furthermore, a regression was run using seasons and flow, and a dummy coefficient for each season to further examine seasonality:

$$Log(conc_{ps}) = \alpha \log \sum flow_s \quad + \beta \log \sum flow_{su} + \delta \log \sum flow_f + \varepsilon \log \sum flow_w$$

where:

$conc_{ps}$ = concentration of nitrate at a specific site and

α, β, δ, ε = dummy coefficients for each season, for example, α is the dummy coefficient for Spring.

and $flow_{x\,=}$ a given flow event during a defined season (s=spring, su = summer, f = fall, and w = winter).

Winter was defined as December to February, spring was March to May, summer was June to August, and fall was September to November. Summary statistics, the Kruskal Wallis nonparametric test, and visual plots also were used to examine seasonality.

Samples were also compared spatially, using paired-t tests between sites for those parameters mentioned above. The differences between the paired samples were found to be normal between sites 1, 13, and 20; use of a paired-t test was, therefore, appropriate. Differences were not as normal between all of the stations and site 2 due to the different drivers of pollutant loadings (nonpoint source and/or stormwater driven vs. sewage treatment driven). The paired t test was still used, with the understanding that the skewing may have somewhat altered the p values; however, since p values were extremely low, this was not a concern. Samples were also plotted in scatter plots by date, with constituent concentration vs. flow, and using box plots.

Physical parameters, including conductivity, temperature, DO, and pH were plotted using scatterplots, line graphs, and box plots to see if any of these parameters could be causing concern in the watersheds. Based on results with conductivity, follow up monitoring during a snow event was conducted at a separate site. This site, located on the northern Long Branch, was immediately south of Interstate 66. The site was selected because of its ease of access during a snow event and proximity to I-66 to see to what extent deicing activities on the interstate played a role in contributing ions to the waters.

Various parameters were compared to relative land use in each watershed both quantitatively and qualitatively. Land use estimates from the household method (discussed in chapter 3) were used for these analyses. Furthermore, the percentage of each type of land use in the riparian buffer was estimated as a static term for the years 1992 and 2000. The percentage of each land use in the riparian buffer (defined as 30 meters from each side of the stream for this analysis) was created using the delineated watershed layers and a written script was used to clip or extract total land use for each watershed. Fairfax County (1999) created hydrology layers that map streams, lakes, and wetlands that were intersected with the watershed layers. The find distance tool, a tool available in Arcview to capture distance and properties of a layer near a selected feature(s), was used to calculate the distance from the water features in the intersected layer. The output was converted to a grid and reclassified into multiple zones: 0-30 meters from the water feature, 30-60 m, 60-120 m, 120-200 and 200+. The thirty-meter zone was the only one used for quantitative analysis and considered the riparian buffer zone. Arcview's map calculator tool allows the user to input conditions into a query-based format to return a true/false spatial layer. The buffer grid layer created was used with the map calculator tool to calculate the amount of each land use in the buffer zone throughout the full watershed layer. Separate layers were created for wetlands, forested area, different categories of urban area, water, and grasses. The urban area was later reclassified to one category for analysis. Finally, each different land use in buffer area for each subwatershed was calculated by using the tabulate areas function. These forested buffer analyses were used both qualitatively, and as a parameter in calculating pollutant loadings.

Calculating pollutant loads is important for estimating the amount of a pollutant being added to a stream over time. Many current regulatory requirements are based on calculated or estimated loads. For example, Total Maximum Daily Loads (TMDLs) calculate the size of a load of a given parameter that an aquatic ecosystem can assimilate and still meet its designated use. Additionally, National Discharge Elimination Permit

(NPDES) requirements are based on reasonable potential analysis that examines what concentration or loading can be discharged to a water body without violating water quality criteria or standards.

Load for any given day for which there is an observation or sample can be estimated by:

$$Load_n(kg/day) = Q_n(cf/s) * \frac{28.3168466\,\text{liters}}{\text{cubic foot}} * \frac{C_n(mg/L)}{1000000(kg/mg)} * \frac{60\sec * 60\min * 24hrs}{day}$$

where

Q_n = average daily flow at the USGS gauge station at day n, and

C_n= concentration measured on day n

In areas in which loading is primarily driven by terrestrial input from precipitation events (such as stormwater and agricultural runoff), calculated loadings serve as a tool to look at typical loadings at various flows since these loadings are primarily precipitation driven. These calculated loads then serve as a basis point for estimating weekly, monthly, seasonal, or annual flows. Total load over a given time (t) can be estimated by:

$$Load_t = \sum_0^t QCdt$$

Total loadings can be calculated using these equations if continuous monitoring or daily monitoring results are available. However, such continuous data sets are rare and we must often extrapolate based on a more limited data set. Therefore, three different approaches were used to estimate loads in the watersheds. The first was by conducting linear regressions of nitrate and TSS values of sampled data for each year using the least squares method. Load at any given time could be calculated by:

$$Load_t = m * f_t + b$$

f_t = Flow at time t
m = slope of the line regressed between observed Load v. Flow
b = intercept (often forced to zero IF there is an undue influence from a single high flow value that pushes the intercept to a value that is not consistent with the many lower data points).

This simple approach plotted instantaneous points with load versus flow. The slope of the regression line was examined, as was the R^2, which was used to examine the percentage of the variance that could be explained by the regression. This approach appeared to be reasonably accurate for nitrate, but not as accurate for TSS. The approach is highly simplified and does not take into account numerous drivers of nutrient concentrations, including seasonality. In the nonpoint source driven watersheds, results were, nevertheless, plausible.

Point source loadings for the site downstream of the Noman Cole Sewage Treatment facility were estimated from data collected from the Chesapeake Bay Program and EPA's Permit Compliance System Database (Chesapeake Bay Program, 2006; US EPA, 2006b). The Chesapeake Bay Program had data available from 1984-2004; data from 2005-2006 were obtained from the PCS system. Data available included minimum, average, and maximum concentrations for monthly discharges. Point source loadings of nitrate-N, ammonia-N, and phosphorus were calculated. For some periods, data were not available for monthly average ammonia-N discharge in PCS. The Chesapeake Bay

Program data used a percentage estimate times total nitrogen to obtain ammonia-N loadings. Where ammonia-N measurements were missing for 2005 and 2006 data, I used the same ratio as applied by the Chesapeake Bay Program (7/20 TKN). Comparing those data to measured ammonia-N concentrations reveals likely overestimates of the ammonia-N contribution.

Average monthly concentration for each parameter was used, multiplied by the average flow per day for the month, multiplied by the days per month, and converted to the metric system to obtain monthly load:

$$L_m = F_d * C_d * N * 3.7854118$$

Where L_m = monthly load
F_d = Daily average flow for the month (million gallons/day)
C_d = Average Concentration for the month
N = number of days in the month
3.7854118 = conversion factor from gallons to liters

Annual loads were calculated to look at relative loadings coming from the sewage treatment facility. Monthly loadings were compared to ambient water quality conditions in stream to look at the impact of these loadings and changes in loadings on nutrient concentrations. For use in the LOADEST analysis discussed below, daily load by month was calculated.

The next two modes of analysis to estimate annual loadings involved use of the statistical LOAD ESTimator program (LOADEST). LOADEST is a pre-packaged statistical software package written in FORTRAN released by USGS that develops regression models for estimating constituent loads in streams and rivers (Runkel *et al.*, 2004). The model can automatically select one of 9 statistical models with the best fit (or the user can select one model option). See Appendix A for examples of input data needed and output data generated with the default LOADEST model. The user can also select a user-defined model option, in which he or she defines the variables chosen and how they are transformed. See Appendix B for examples of input data needed and output data generated with a user defined LOADEST model. At a minimum, the program needs stream flow (independent variable), concentration of constituent (dependent variable), and any user-defined variables that the modeler wishes to include in the calibration. The model then gives estimations of monthly and seasonal flow and seasonal loading, and daily loading and annual loading can be derived using spreadsheet software. The commonality of the LOADEST approach with the linear regression method is the presence of concentration and flow as the primary drivers. Once a user understands the nature of the drivers, he or she can force the model to add additional parameters.

LOADEST was used for multiple time-series for the same sites. The model automatically selected the best fit. After about 100 such variations of these runs, the best approach developed two major ‘product’ groups. The first grouping consisted of sites 1 and 13 data using groupings of 1983-1987 and 1988-1992, and using sites 1, 13, and 20 for 2005. I cannot use this approach with site 2 because of the known addition of load associated with the sewage treatment facility. Also, because these automatic model selections often include a time component, they cannot be validated. This approach also cannot accurately be used to estimate loadings for time periods for which no data was available. Nonetheless, these models produced very high R^2 and can be assumed to give reasonably accurate estimated loadings for years that data was collected.

The second group used user-defined models for the same time periods. These components included flow, concentration, and percentage urban land use from the household method. Seasons were added to the model; however, substantial complications were encountered and seasonality was ultimately dropped (more in discussion section 4.4.4). For site 2, the model had an additional component, addition of nitrate nitrogen and ammonia nitrogen from the Noman Cole POTW. For this site, total monthly flow was prorated per day to modify the flow values. Finally, all sites were grouped for the time periods 1987-1992 and 2005, and a user-defined model was run using flow, concentration, total urban land use in the watershed, point source loadings, and percent urban infringement on the riparian buffer zone. This model run was used to look at the relative influence of these various factors.

4.2.3 Duration Curves

Flow duration curves were created to examine changes in flow patterns in the upper portions of Accotink. Flow duration curves are cumulative density distributions that look at the percentage of flow events that are above or below a certain threshold value. They can be useful for characterizing flow, calibrating hydrology and water quality models, and determining how one might estimate loadings in portions of a watershed that do not have a flow gauge station. For this analysis, daily discharge values from the Accotink gauge station were used.

Load duration curves, on the other hand, are useful for examining how much loading of a parameter a given water body can handle for any given flow value or characterizing loading into a given water body. The simplest way one can use load duration curves to examine existing water quality criteria is to multiply flow times the water quality criteria:

$$Load_n = Q_n * WQC_p$$

Where $Load_n$ = the load over given time period n
Q_n equals flow for time period n
And WQC_p = equals the water quality criteria for the selected parameter.

Hence, the amount of total loading that a watershed can take and still meet its water quality criteria for any given time period is:

$$Load_t = \sum_{0}^{t} Q_n * WQC$$

Load duration curves could potentially prove useful in areas where regulators are using statically driven TMDLs. They are relatively easy to use and make use of existing water quality data in an interpretable format. For example, percent ranges of flow can be used to determine the average loading coming from observed data for each flow frequency (the same can be done for absolute flow values).

Load duration curves can also be a useful tool if one understands loading targets of a downstream receiving water body from any given stream or river. If a Fairfax County watershed planner were to determine that the county's goal was to deliver no more than a total of 50,000 kilograms per year of total nitrogen to the Potomac river from the Accotink and Pohick watersheds, load duration curves could be a useful tool in determining whether the source of nitrogen was primarily point or nonpoint source in nature and where to target in order to reduce or maintain loading to hit target goals.

For this dissertation, load duration curves were created to explore their feasibility for setting tentative loading delivery targets to Gunston Cove and to help characterize the primary source of loadings in the watersheds. I did not set water quality standard or variable water quality criteria concentrations since no firm water quality criteria exist for TSS or nitrate. Furthermore, firm loading targets for the downstream receiving waters were not set. Load duration curves were created for nitrate and total suspended solids with the 1984-1992 George Mason Data, USGS flow data, and data collected during the 2005 field season. I used the slopes of exponential regressions of individual nitrate loading on a logarithmic scale relative to percent flow to simply indicate whether loadings were point or nonpoint source in nature. These regressions were calculated by:

$$LCL_t = \sum_{1}^{t} \alpha e^{(\beta n)}$$

Where LCL_s = Loading concentration lines for time period t

α and β = coefficients and

n = relative time (from 1 to 365 or 366 if using a year; the cumulative distribution point of the flow date could also be used here)

These lines were plotted on a double axis scatter plot that also included a flow duration curve and the individual estimated daily loadings. Total flow was plotted on the left y-axis, percent relative flow plotted on the x axis, while loading was plotted on the right y axis. The percentage high flow events were positioned closest to the intersection between the x and y axis. In order to create a visual tool, the lines were positioned so that a portion of the regression line crossed was near to the flow duration curve. If the regression line was near to or below the flow duration curve during low flow events, but loading increased at a higher rate relative to flow, precipitation driven pollution (in this case stormwater) was determined to be the primary cause of loading. The logic behind these assumptions are that if precipitation driven pollution is the primary contributor of loadings, the majority of loading will occur during high flow events. Concentrations of in-stream parameters may be equal to or higher than concentrations in low flow events (dependent upon the nature of the parameter; however, the concentration is often higher if there are substantial sources of precipitation driven pollution). If the opposite were true - if the loadings were higher relative to the flow duration curve under lower flow conditions and the slope of the loading for the regression line increased at a slower pace than flow - then the pollutant source is likely driven by traditional point sources. The logic behind these assumptions are if point sources are the main contributor of loading, loading will be higher independent of flow and the relative concentration of a in-stream parameter will be higher during low flow events. Data were plotted as a group for the full original observation period, split into 1984-1992 and 2005. Data were also plotted annually for each site where data were available.

The loading regression lines were examined to see if, by summing the loadings of the line over time (*t*), the lines would give decent estimates of annual loadings compared to other methods such as LOADEST or linear regressions. One potential advantage using this method is that a single outlier would have less influence on the slope of the line. Furthermore, if the loading regression lines were already created to use as a visual tool, the extra effort needed to estimate the annual load would be minimal. Shortcomings of this approach include overestimating the loadings in the low end of the flow range and overestimating them in the high end of the flow range. Therefore, this would not be viable for a daily estimator, but there is promise for use with an annual load.

Due to the absence of additional flow data, flow patterns found downstream of the USGS station in the Accotink watershed and overall in the Pohick watershed were assumed to resemble those found in the portion of the Accotink watershed monitored by the USGS station. A ratio of flow based on the size of the watershed was used: for instance, daily flow at Accotink 20 (the location of the USGS gauge station) was multiplied times 1.36 for the Pohick 1 watershed. Similar ratios were then utilized for all sites.

4.3 Results

4.3.1 Flow

Based on observations, both Pohick and Accotink exhibited significant scouring as a result of changes in flow. The streams were extremely flashy, and multiple high flow events were noted during sampling periods and at other times when the streams were visually observed. As a side effect of this flashiness, it is possible that much of the TSS loading is coming from in-stream scouring.

The only USGS gauge station in either of these watersheds was used as a general indicator for flow for both watersheds. The upper reaches of the Accotink watershed, those included in the watershed of the gauge station, were urbanized primarily before the passage of EPA’s Phase I regulations in 1991 and Fairfax County’s 1993 requirement that all new development have BMPs designed to protect water quality. Few Best Management practices were used on the majority of the developments built in this time period. Hence, the many hydrological changes that occurred in the watershed as a result of urbanization occurred without steps being taken to mitigate the impact of that urbanization. This is likely a direct result of a high quantity of impervious cover not mitigated by sufficient or well-designed BMPs.

Average annual flow volume increased from the start of the first full year of the monitoring period (1948) to 2006 (Figure 4.3). A multiple linear regression model with annual precipitation, year, and flow (dependent variable) was run for the years 1948 to

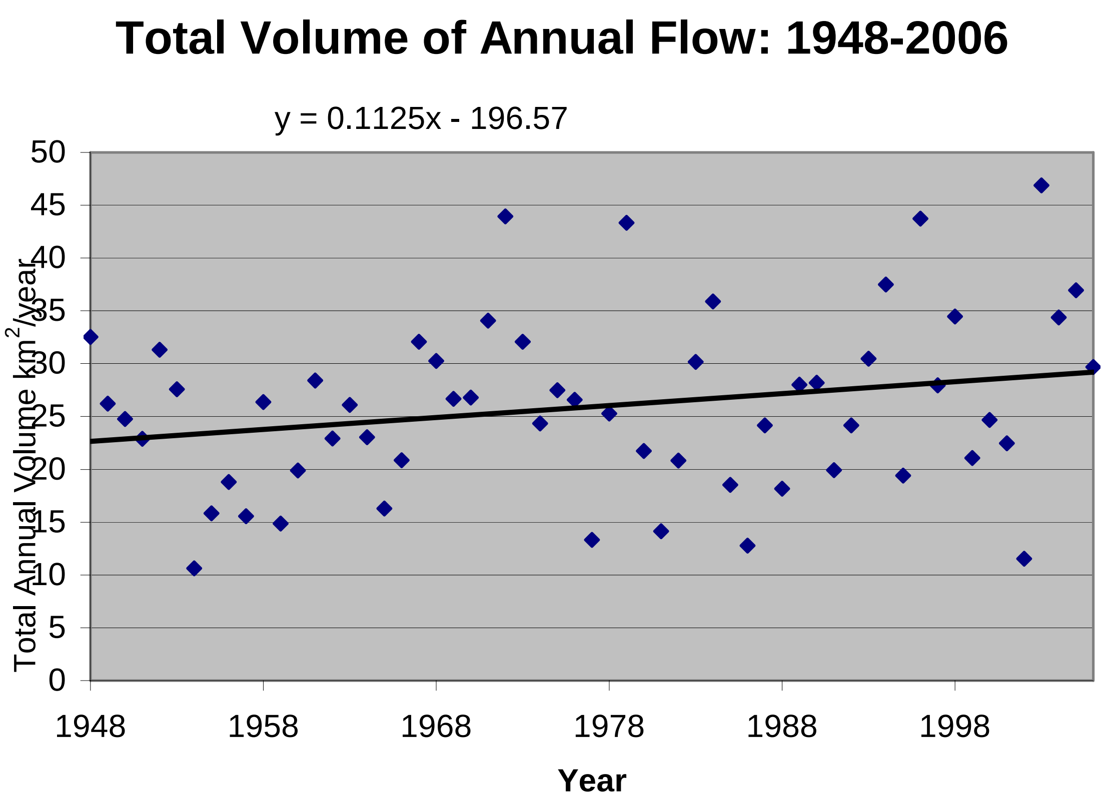

Figure 4.3. Annual flow volume 1948-2006 where y equals flow and x equals time. There is clearly a trend toward increasing annual flow volume.

2006. Both year and precipitation have a positive coefficient associated with flow, meaning that later years (more developed watersheds) and higher precipitation correlate with higher flow (R^2 = .512, p <0.0001). A similar regression from 1975 to 2004 with percent urban area, annual precipitation, and flow (dependent variable), also shows that increased urban area result in higher annual flow volume (R^2 = .509, p <0.0001). There is a positive correlation between percent urban land use and flow (R = 0.237) and precipitation and flow (R = .694). These values highlight the obvious: that years with increased precipitation have higher annual flow; however, these values also show that increased urbanization resulted in increased annual flow. The R^2 values explain about half of the variation. The remaining variation is likely explained by factors including type of storm events, temperature, antecedent soil conditions, and/or engineered changes to watershed drainage.

Individual flow event dynamics were also significantly impacted by urbanization. Both daily peak flow (those above 100 cf/s) and daily base flow (those below 100 cf/s) extremes were increasing over the time period that the watershed urbanized. In addition to being influenced by land use, flow volume is most heavily impacted by precipitation and climate. By combining these two metrics into one graph, we can somewhat compensate for the impact of wet versus dry years (see Figure 4.4). The R^2 is relatively low (.3537) because the primary cause of this variation is precipitation; nonetheless, the impact of time (and consequently urbanization) is significant (p<0.0001). This trend signals increasing extremes for hydrologic flow and is likely contributing substantially to degraded stream conditions.

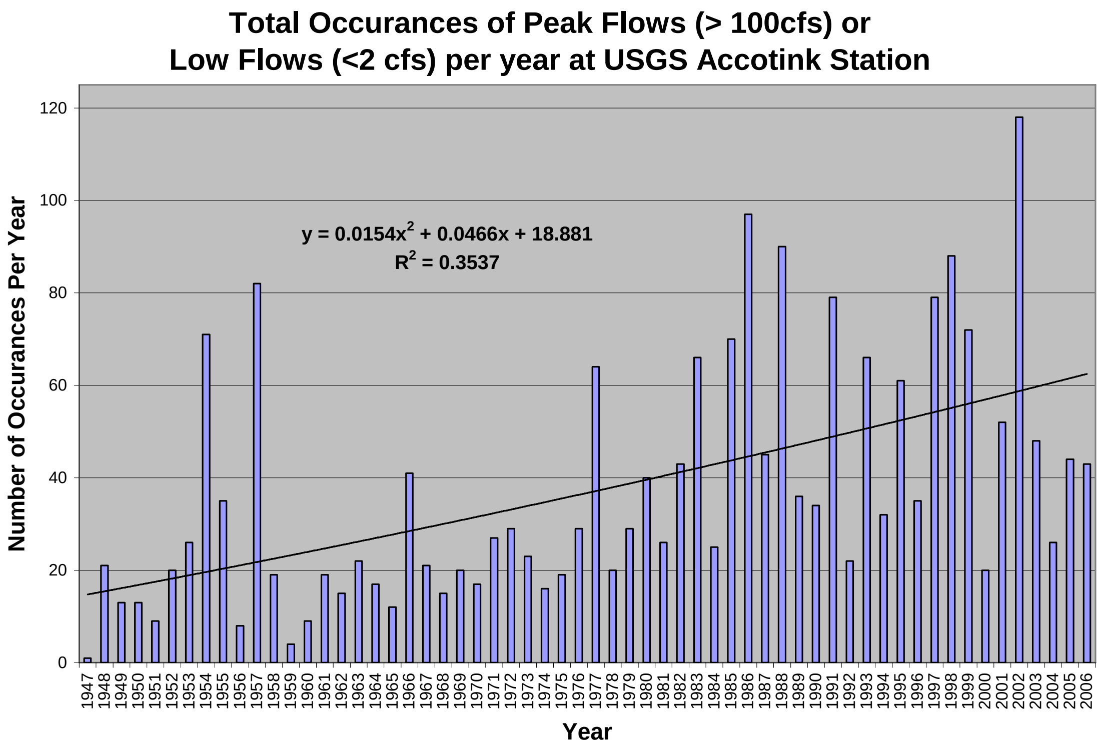

Figure 4.4. The number of total high and low flow events increased during the monitoring time period.

Both the number of low flow events and high flow events have increased over the monitoring time period as the watershed urbanized, although their response has been somewhat different (Figure 4.5). There was at least one recorded low flow day (less than 2 cf/s) event for 4 years in the 1950s, 2 years for the 1960s, and 2 years for the 1970s. In contrast, 8 of the 10 years in the 1980s had at least one low flow event, 7 of the 10 years in the 1990s, and 4 of 7 years thus far in the 2000s. Such an increase in the frequency of low flow events is clear evidence that the shallow groundwater is not being recharged as quickly and base flow is lower.

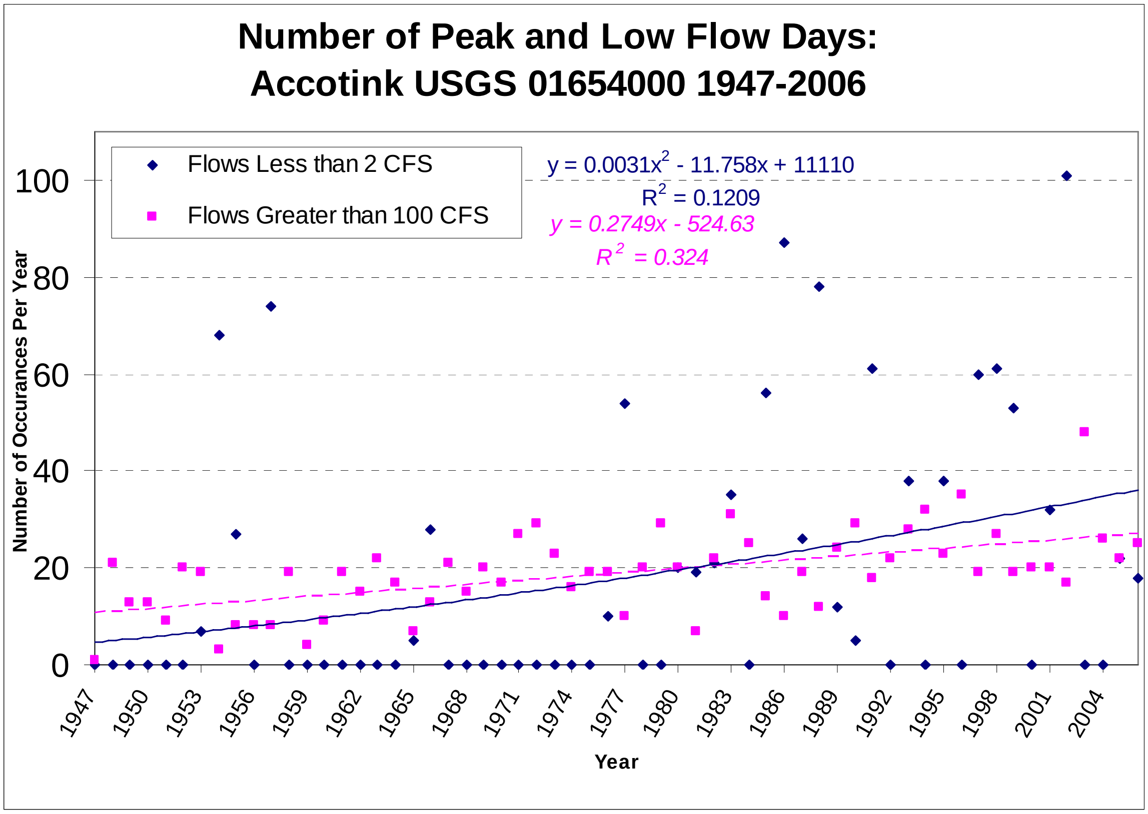

Figure 4.5. The number of high and low daily flow events per year. The pink squares represent the number of daily high flow events (greater than 100 cf/s average discharge) in any given year while the blue diamonds represent the number of daily low flow events in a year (less than 2 cf/s average discharge). In the regression equation, y equals flow and x equals time.

Changes in the frequency of high flow events have been equally dramatic. In the 23 years on record before 1970 (part of 1947 to 1969), there were only 5 years that had at least 20 days of high flow events (greater than 100 cf/s). The next 20 years, 1970-1989, had 10 years with at least 20 days of high flow events. The 1990s had 7 of 10 years with at least 20 high flow events: those three years that are excluded had 18, 19, and 19 events. Five of 6 years in the 2000s have had greater than 20 days of high flow events, the exception being the 17 events recorded in 2002, which is significant considering 2002 was a heavy drought year with over 101 days having flows below 2 cf/s, the highest number recorded for the full monitoring period.

Interestingly, as Figure 4.5 shows, the best-fit regression for the number of high flow events is linear, while the best-fit regression for low flow events is non-linear. Even though they occur in greater frequency in later years, periodic high flow events were recorded in almost every year. However, before 1975 low flow events were rare. The rapid increase indicates a significant lowering of base flow, which has substantial implications for aquatic life and nutrient and sediment delivery. Though the R^2 values are relatively low, the results are significant ($p < .0001$ for high flow days; $p = .007$ for low flow days). Furthermore, other causal agents, such as precipitation, explain a significant portion of the variation. For instance, the highest 5% of daily precipitation events (defined as greater than 0.7 inches/day) were counted and summed by year from 1970 to 2005. A multiple regression model was run using these counts, year, and the number of peak flow events (defined as greater than 100 cf/s), with the latter data set defined as the dependent variable. This model improves the R^2 from .072[14] to .4498 ($p<.0001$). In the period from 1975 to 2005, when percent urban land use is substituted for year, the R^2 improves to .505 ($p<0.0001$), compared to an R^2 of .1182 ($p=0.06$) when a simple linear regression is run with urban land use alone. Though changes in land use do cause an increase in flow, they are not the only causal agent, and explain only a portion of the variation.

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[14] This R2 value does not include the analysis from 1948-1969; hence, the R2 is substantially lower than in Figure 4.5, which includes analysis from 1948-2006.

Hence, extreme flow events have become significantly more commonplace as the upper Accotink has urbanized and this has likely contributed to the benthic and aquatic life impairments. These extreme flows have likely resulted in violations of the State's water control standards of "man-made alterations in stream flow shall not contravene designated uses including protection of the propagation and growth of aquatic life" (Virginia State Water Control Board, 2006) .

A second issue worth noting is the percentage of flow below the Noman Cole sewage treatment facility that comes from upstream, precipitation driven sources versus from the POTW. Flow from the Noman Cole facility consistently averages between 57.75 cf/s and 73.00 cf/s (an increase from an average of 50.99 cf/s during 1984 to 1985). Multiplying the Accotink stream gauge station times the watershed area of the Pohick 2 watershed gives us a daily flow of between 1.30 cf/s and 2304.5 cf/s. During most low-to-mid-flow events, the flow below the sewage treatment facility is dominated by that from the POTW, making the Noman Cole facility effluent the majority of stream flow during low to moderate flow days. However, during high flow events, the POTW makes a much lower percentage of the flow. On one hand, this means that the POTW must continue to maintain the quality of its discharge, particularly during low and mid flow events, since Pohick is an effluent dominated stream during these time periods. On the other hand, the facility serves to mitigate many of the issues with unnaturally low base flows due to lack of groundwater recharge and may very well serve as a positive influence during high flow events once the stream geomorphology stabilizes to accommodate a consistent 60-80 cf/s dry weather flow. Additionally, during the peak flow events, the relative facility contribution is small enough that their contribution toward issues such as peak flow should be minimal.

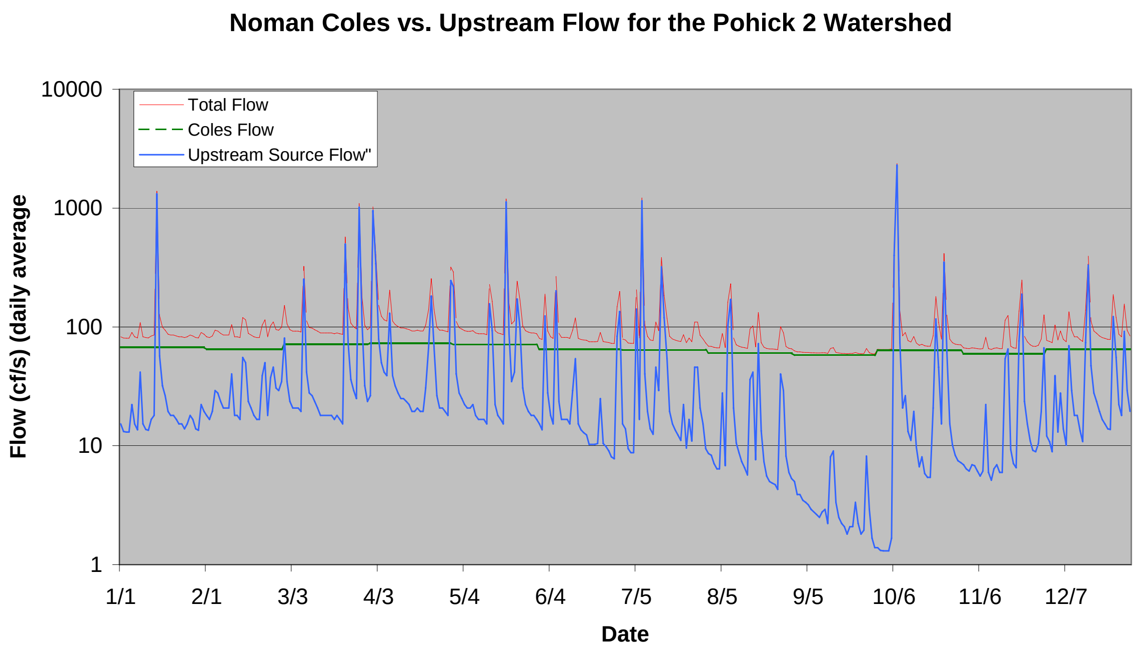


Figure 4.6. Noman Cole effluent flow vs. estimated upstream flow.

Lastly, the natural log of flow for the time period of 1983 to 1992 and 2005 was found to be normal (see Figure 4.7). This proves important later for conducting statistical tests and use in statistical analysis.

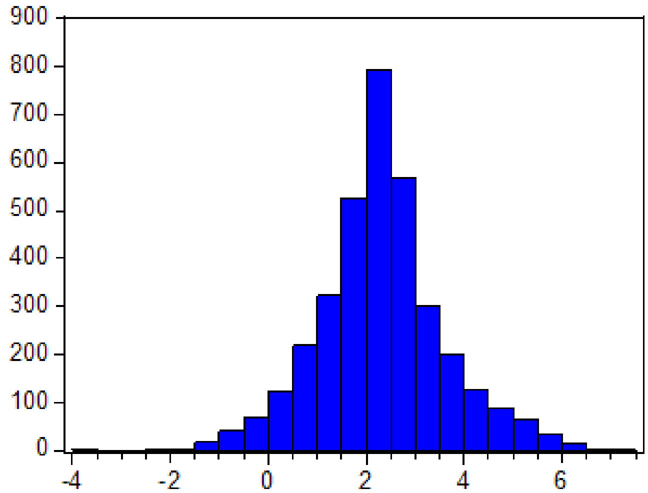


Figure 4.7. The number of flow events times the natural log of flow for the time period from 1983 to 1992. The natural log of flow for all time periods was found to be normal with a skewness of only .04732.

4.3.2 Physical Parameters

4.3.2.1. Conductivity

Of the physical parameters (conductivity, Temperature, Dissolved Oxygen, and pH) conductivity is the parameter most likely to be having significant environmental impact. For the 2005 monitoring year, conductivity concentrations in Accotink and Pohick varied with flow, season, and between the two watersheds and, for the most part, were within an acceptable range. As shown in Figure 4.8, Concentrations were significantly higher below the sewage treatment facility on Pohick due to the discharge of ions in the effluent ($P<.0001$). However, concentrations of conductivity exceeded acute water quality standards on at least two occasions in Accotink. These exceedences were each noted in multiple locations along the course of the waterway. Both of these events were in winter and followed a snow event. Presumably, these high levels of conductivity are directly caused by application and subsequent runoff of road salt to surface waters. The first exceedence was nearly one week following the snow event and after several days of melting. Therefore, one can assume that the high levels of conductivity in this part of the stream can last longer than a day and could theoretically have impacts on aquatic organisms.

It is interesting to note the high conductivity concentrations in Accotink versus the Pohick watershed upstream of the sewage treatment facility (site 1) (see table 4.2). Though the Pohick watershed has a high surface area of roads, many of these roads are residential or secondary in nature. In contrast, the Accotink watershed has significant portions of Interstates 495 and 66, which may receive substantially higher applications of road salts.

Table. 4.2. Average values of physical parameters in Pohick and Accotink for 2005.

Site	Cond.	DO (mg/l)	pH	Temp C
Pohick 1	201	9.61	6.3	16.3
Pohick 2	476	8.45	6.2	20.1
Accotink 13	421	9.18	6.2	16.5
Accotink 20	513	10.06	6.3	15.4

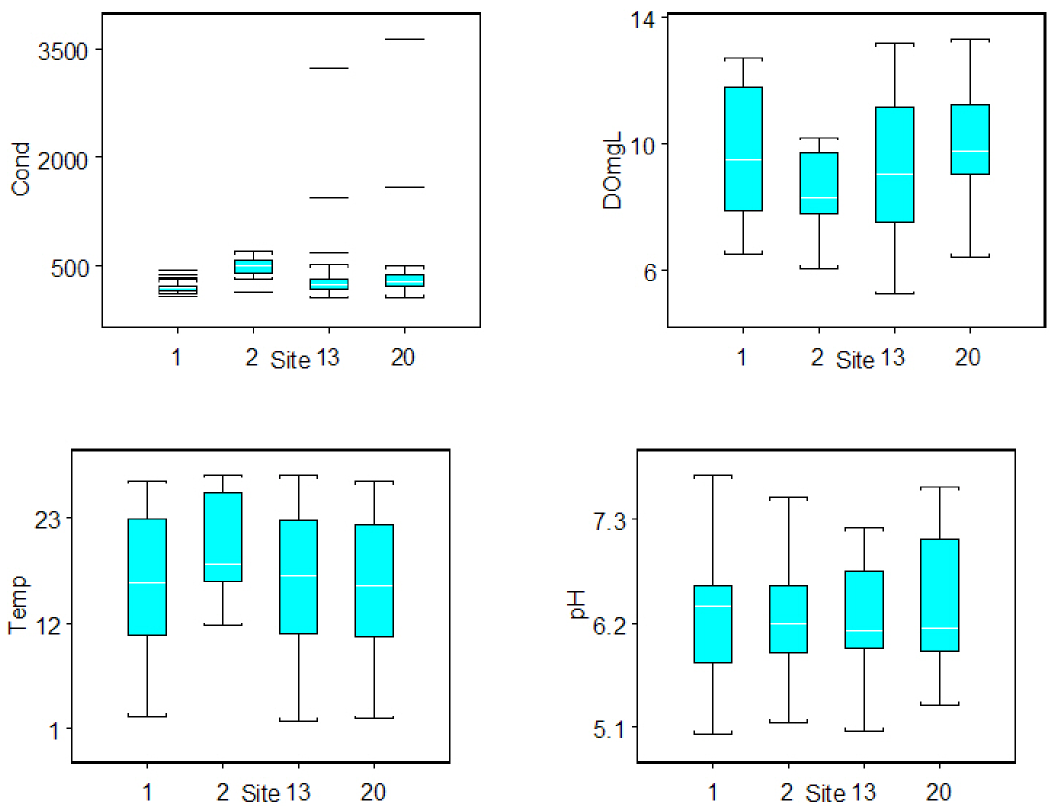


Figure 4.8. Conductivity, dissolved oxygen, temperature, and pH for the 4 monitoring sites for the 2005 monitoring year. Percent saturation for dissolved oxygen is displayed in Figure 4.13.

In order to further explore conductivity exceedences, the Northern Long Branch of Accotink Creek, was surveyed for conductivity immediately below interstate 66 in February, 2006. This site was selected because of its ease of access during the snowstorm relative to the author's home. The site was monitored immediately at the start of snow, once every 12 hours for the following three days, and once every two days until 9 days had passed since the start of snowfall. Additionally, on two occasions, Eakin Park (site 21) was monitored around the same time to see if results correlated. Results indicate a surge in conductivity immediately before snow started falling. Intensely elevated periods of conductivity lasted close to three days, but at the conclusion of monitoring, conductivity still remained elevated over pre-snow conditions. Furthermore, results at this site correlated well with two samples taken downstream at site 21. Conductivity reached nearly 3,000 microsiemens over the monitoring period in this headwater, but was still lower than what was observed in grab samples on other occasions lower in the watershed (Figure 4.9).

On February 26th, 2007, a period during the initial days of a snowmelt, conductivity was recorded at 4950 microsiemens at Site 20. Upstream, at the intersection of Little River Turnpike and Accotink Creek, conductivity remained elevated at 4,985

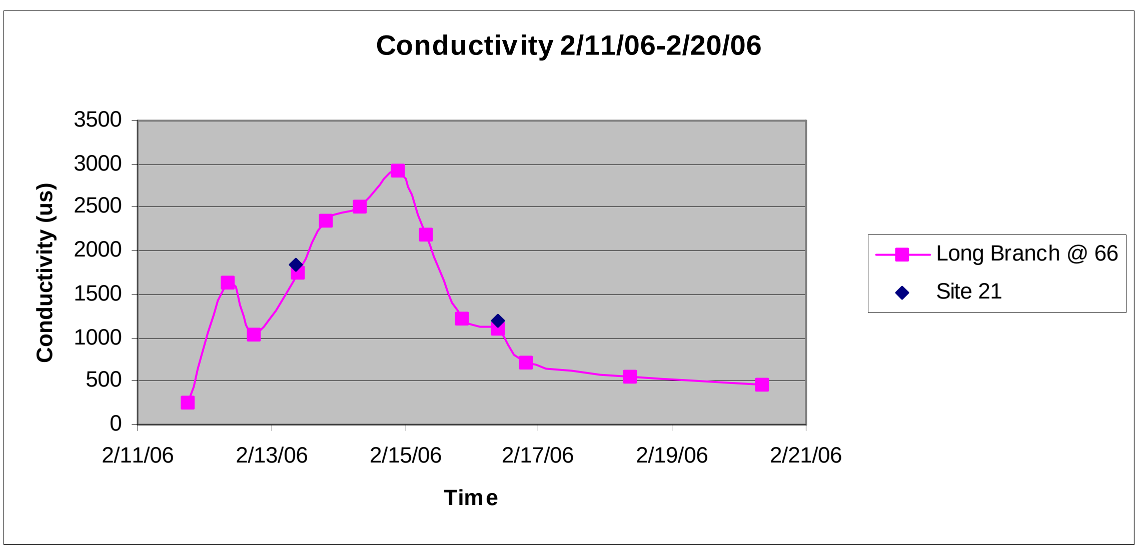


Figure 4.9. Conductivity from 2/11/2006 to 2/21/2006 at Long Branch Creek south of I-66 during a moderate snow event.

microsiemens. Continuing upstream to Accotink at Old Lee Highway, Conductivity dropped to 1705 microsiemens. Quite clearly, there is a significant source of conductivity loading between Accotink at Little River Turnpike and Accotink at Old Lee Highway.

4.3.2.2. Temperature

At all sites, temperature ranged from 1.7 to 25 degrees. There is a significant difference in temperature values between the Pohick site upstream of the sewage treatment facility and downstream of the facility during winter months using the Paired-t test (see Figure 4.10). The lowest recorded temperature at Accotink 2 is 11.71 °C while

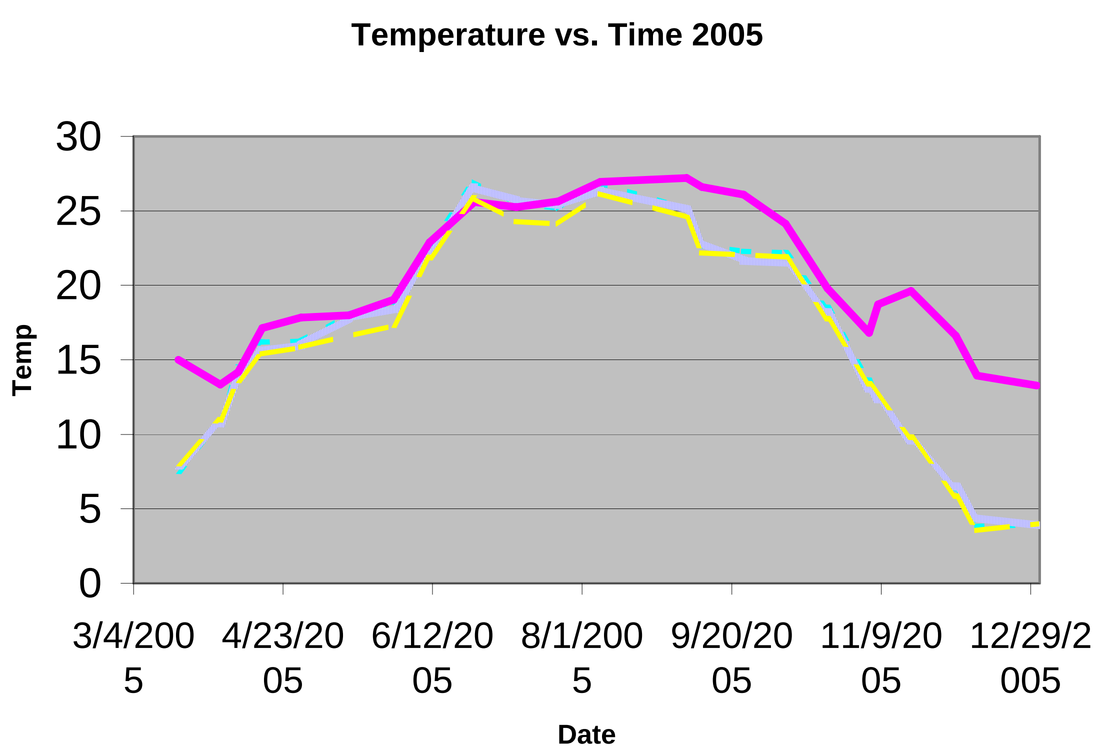


Figure 4.10. Temperature vs. date for 2005. Site 2 (below the sewage treatment facility) is solid pink, site 1 is patterned blue, site 13 is turquoise dotted, and site 20 is (yellow dashed).

it is 2.17 °C upstream at site 1. Summer temperatures do not exceed the state water quality standard of 32 °C for any site on any sampling event (Virginia State Water Control Board, 2006).

There is empirical evidence that stream temperature is increasing at site 1 over the full range of data availability (1984-2005), but not increasing as significantly at site 13 (see Figures 4.11 and 4.12). The 2005 monitoring year appears to be one of the warmest of the records for site 1 (2^{nd} of 10 with a mean temperature of 16.3 °C; however, though it is warmer than average, it does not appear to be one of the warmest years for site 13 (4^{th} of 10 years with a mean temperature of 16.5 °C). Furthermore, the difference in average temperature between site 1 and 13 has decreased from 1.06 degrees cooler in 1984 to 0.19 degrees cooler in 2005. Though this evidence is not overwhelming, it does appear the temperature of Pohick is warming, likely as a result of increased warm urban surfaces, and possibly due to occasional loss of streamside shading or the overall urban heat island effect.

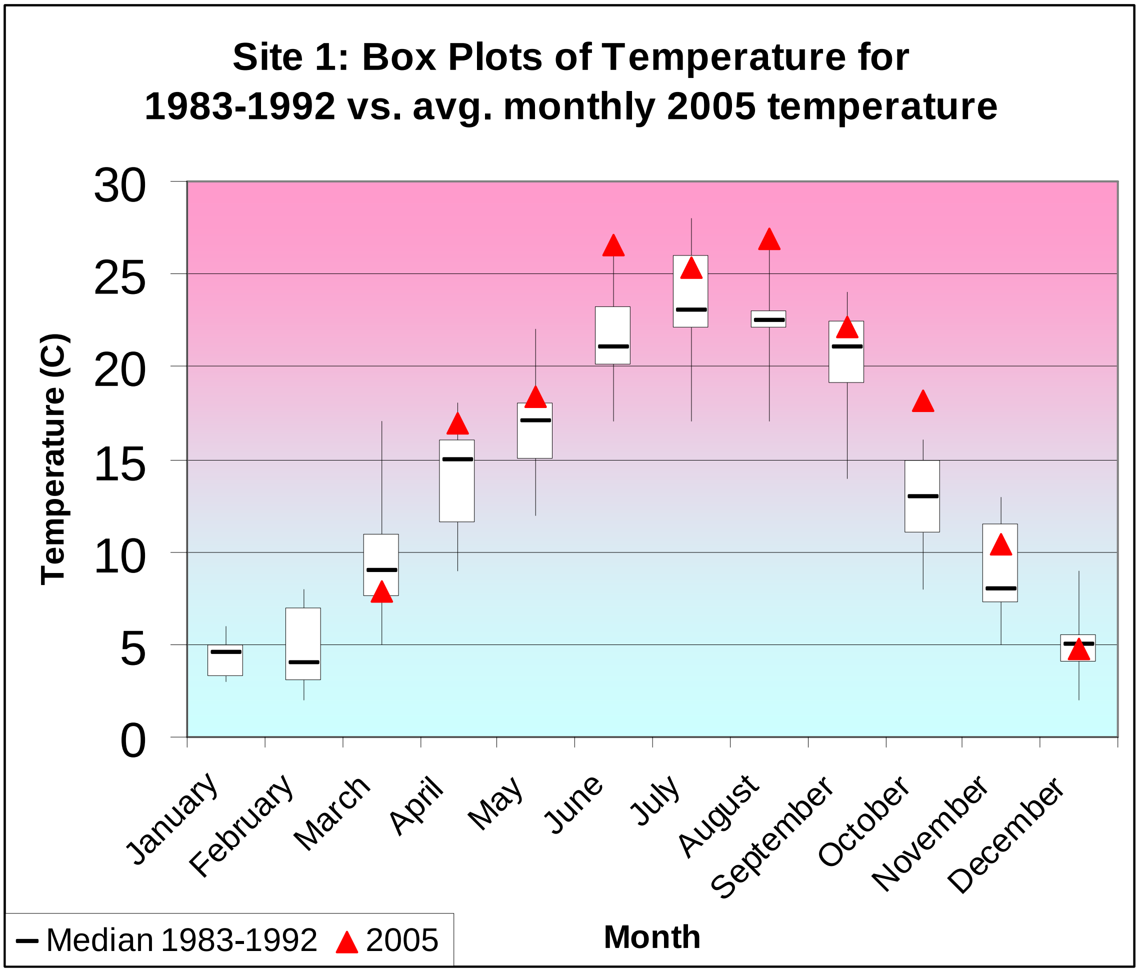


Figure 4.11. Box plot of temperature for 1983 to 1992, with the average temperature by month for 2005 for Pohick 1.

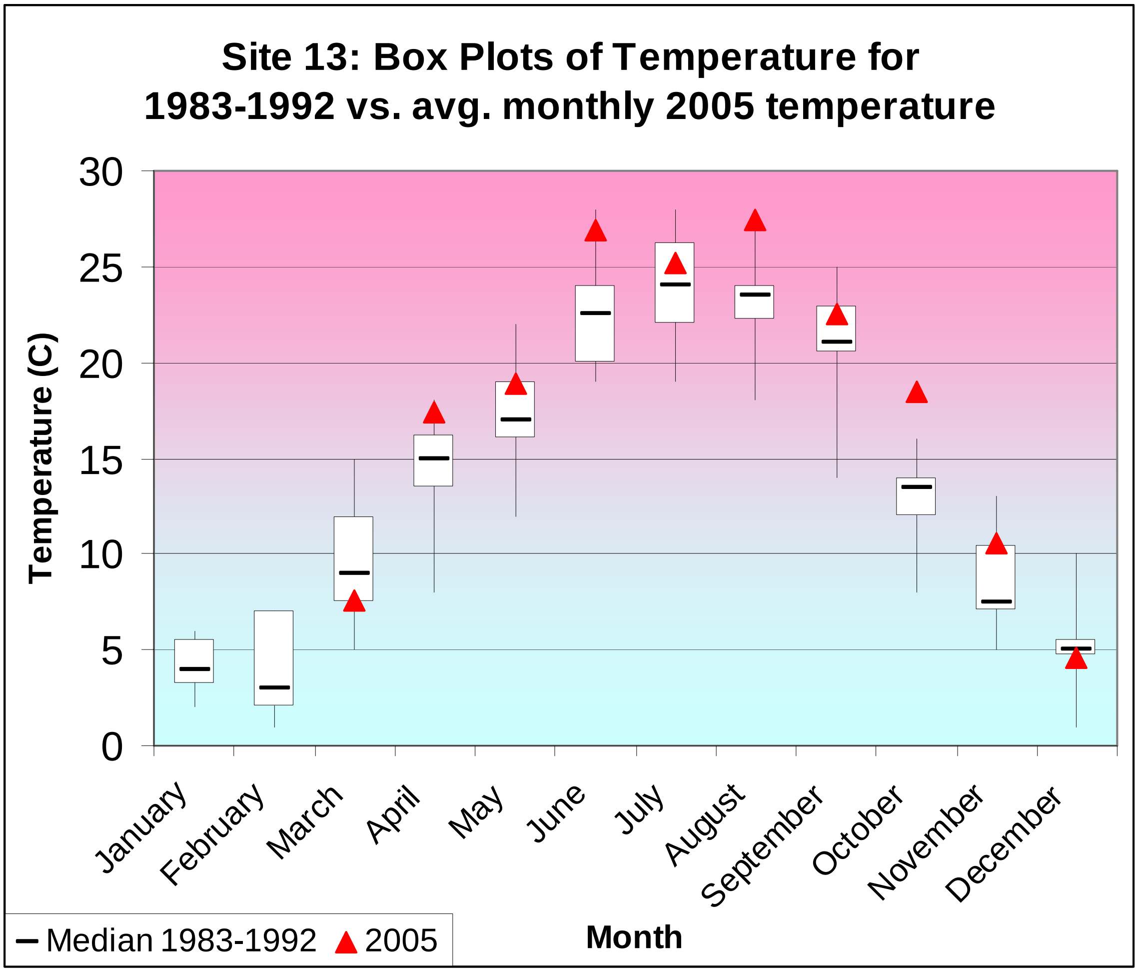


Figure 4.12. Box plot of temperature for 1983 to 1992, with the average temperature by month for 2005 for Accotink 13.

4.3.2.3. Dissolved Oxygen

Dissolved oxygen (DO) levels for Accotink and Pohick were found to range between 5.2 mg/L and 13.3 mg/L for 2005. Average values for the absolute concentration of dissolved oxygen are significantly lower at site 2 than any of the other sites (p=.0002 between 1 and 2, .015 between 2 and 13, and .002 between 2 and 20); however, these values are likely an artifact of higher winter temperatures (see Figure 4.13). This apparent difference is likely due to the warmer waters present in winter when the greatest difference in oxygen levels between Pohick 2 and the other sites is found. Warm waters do not hold as much oxygen, therefore, the significant difference between the sites is not surprising. Unfortunately, DO levels were not taken during every sampling event due to problems with the equipment. Some samples were taken when the streams were the warmest, which is also where we would expect to see the lowest oxygen readings. Additionally, samples were taken at various points in the day and never immediately before sunrise, when one would expect to see the lowest DO readings. Nonetheless, based on these data, it appears that DO concentrations are not significant stressors for either of these streams. Though close, none of the measurements dip below the state water quality standard of 5.0 mg/L in either the 2005 or historic data sets (Virginia State Water Control Board, 2006). This is consistent with historical records (Figure 4.14), when no samples were recorded with DO values less than 5 mg/L.

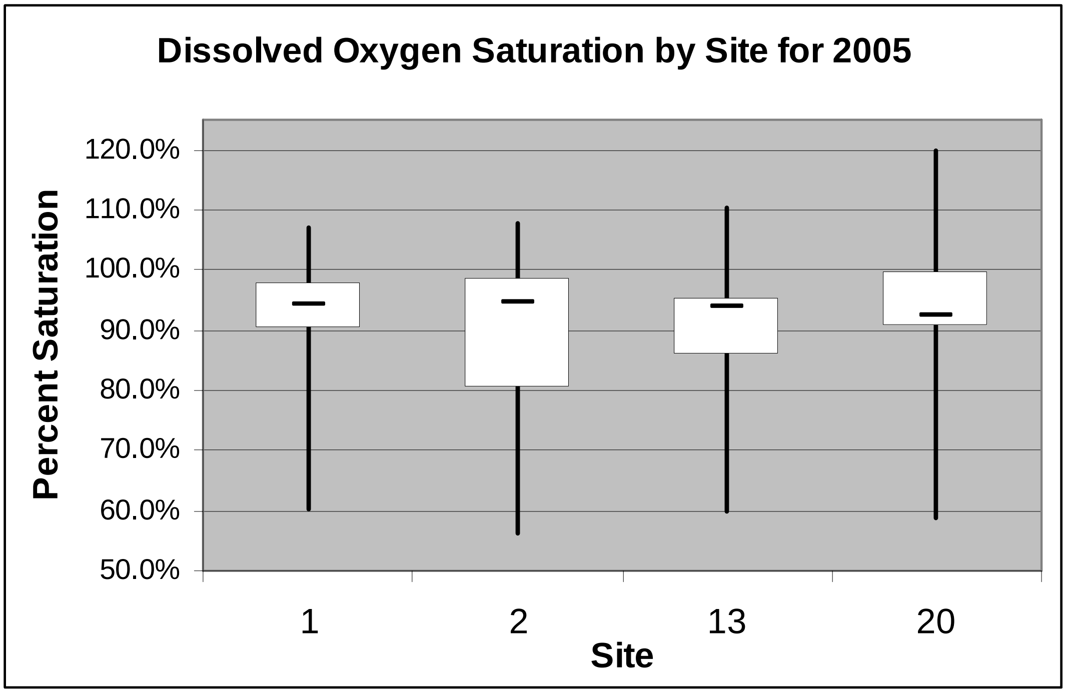


Figure 4.13 Percent saturation of dissolved oxygen by site.

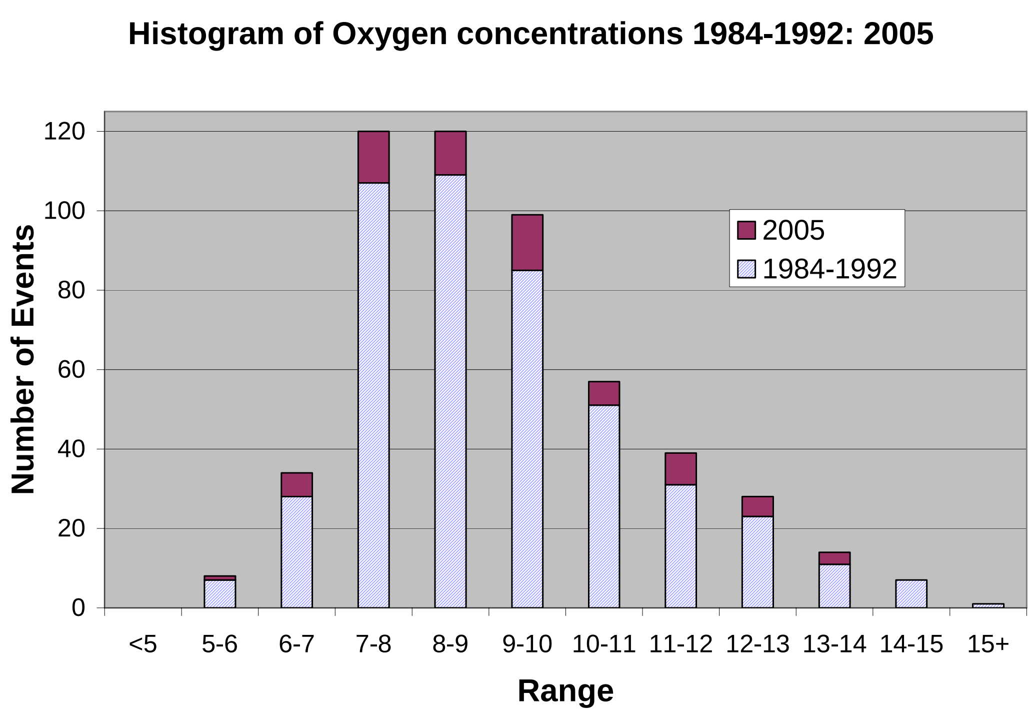


Figure 4.14. Oxygen concentrations over time in Accotink and Pohick

4.3.2.4. pH

The pH levels in Accotink and Pohick varied from 5.0 to 7.8 (see Figure 4.8 on pg. 123). The lowest pH values were often found during high flow events, indicating the influence of acid rain in the watersheds. Precipitation in the mid-Atlantic region tends to be about 4.5. Since it is precipitation driven, the low pH appears to be episodic, with an average pH of between 6.2 and 6.3 dependent upon site. The streams appear able to somewhat buffer the acidic rain of the northern Virginia area, although the lowest measurements are below the state standards of 6.0 (Virginia State Water Control Board, 2006).

4.3.3 Nitrogen

Nitrate concentrations varied significantly between all sites for all years ($p<.0001$ Kruskal Wallis two sided test, chi-square = 44.0374). Historically, there is an unexplained high mean in observed nitrogen concentrations from about 1984-1986 for Noman Cole/George Mason University data so these data were not utilized for all tests and model calibrations. From 1987 through 1992, results are consistent with expectations. Those watersheds with the highest percentage of urban area had the highest concentrations of nitrate. There appears to be an increase in nitrate concentrations in 2005 from 1987 to 1992 levels. However, 2005 levels of nitrate are lower than 1984-1985 data.

In 2005, nitrate concentrations showed significant variation between all stations (from p=.01 between site 13 and site 20 to p<.0001 for site 2 to site 13, site 2 to site 1, and site 2 to site 20). Concentrations are highest in Pohick 2 due to the Cole facility discharge, followed by the Accotink 20 and 13M sites, with Pohick 1 having the lowest consistent concentration (see table 4.3).

Table 4.3. 2005 Nitrate summary statistics

Site	n	Avg. (mg/L)	Max (mg/L)	Min (mg/L)
Pohick 1	24	0.57	1.14	0.17
Pohick 2	24	1.80	2.77	0.56
Accotink 13M	24	0.78	1.47	0.22
Accotink 20	20	0.93	1.73	0.31

Sites 1, 13, and 20 showed a relationship between increased concentration and increased flow. Low flow events tended to correlate with lower concentrations in all three sites. This quite clearly indicates that the majority of nitrate at these sites is associated with wet weather and likely stormwater. Since the largest flows coincide with the largest concentrations, the greatest loadings are associated with the nonpoint sources in these watersheds. The opposite results were observed at site 2 for one primary reason: the point source discharge nitrate was diluted from less concentrated precipitation driven flow. In lower flow conditions, nitrate concentrations increased due to decreased dilution of the POTW effluent. This indicates a relatively consistent loading of nitrate from the point source.

The following five Figures help to illustrate the relationship between flow and concentration at the sampling sites for 2005. Figure 4.15 shows that, for the most part, concentration of nitrate is still highest below the Noman Cole plant relative to other sites, except during high flow events. Figure 4.16 illustrates how concentrations decrease at site 2 as the impact of the sewage treatment facility is diluted, but increase at the other sites as a result of precipitation driven pollution. Figures 4.17 through 4.19 show the flow duration curves and regression lines of nitrate loading calculated for 2005 for each site. At sites 1 (Figure 4.17) and 13 (Figure 4.19), we can determine that the majority of loading is nonpoint source related and that the majority of loading occurs during few high flow events based on the slope of the line. At site 2, total loading does not increase substantially with increased flow illustrating that loading is driven more by point source pollutants (Figure 4.18). Hence, the slope of the lines further support what is intuitive based on previous analysis; that loadings from sites 1 and 13 are stormwater dominated while nitrate loadings at site 2 are dominated by the Noman Cole discharge.

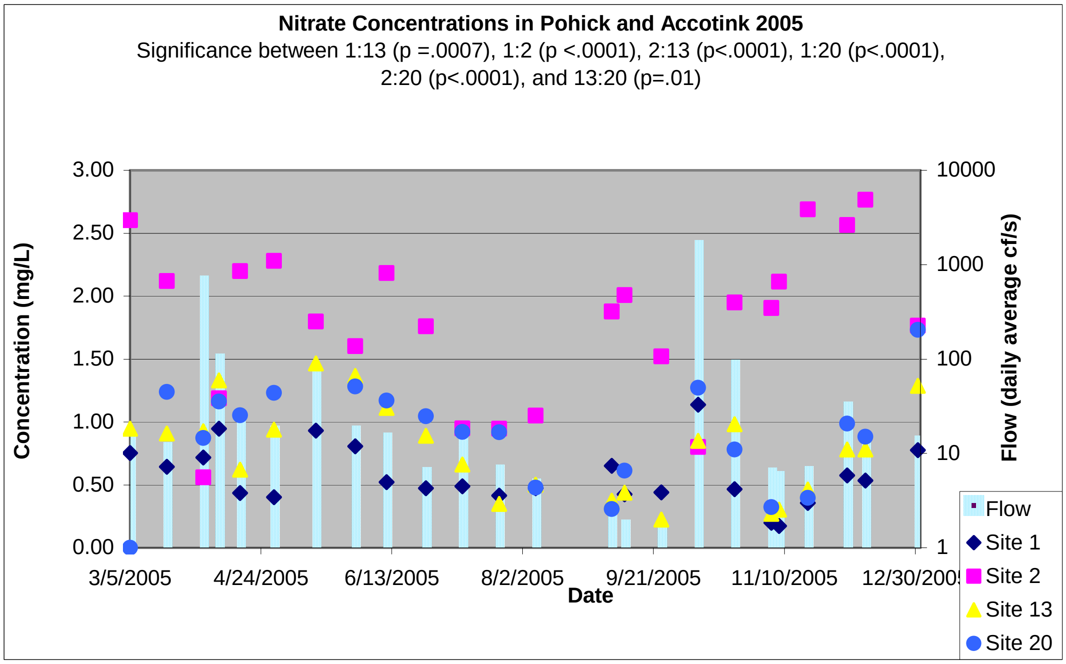


Figure 4.15. Nitrate concentration and flow v. time during 2005 sampled events.

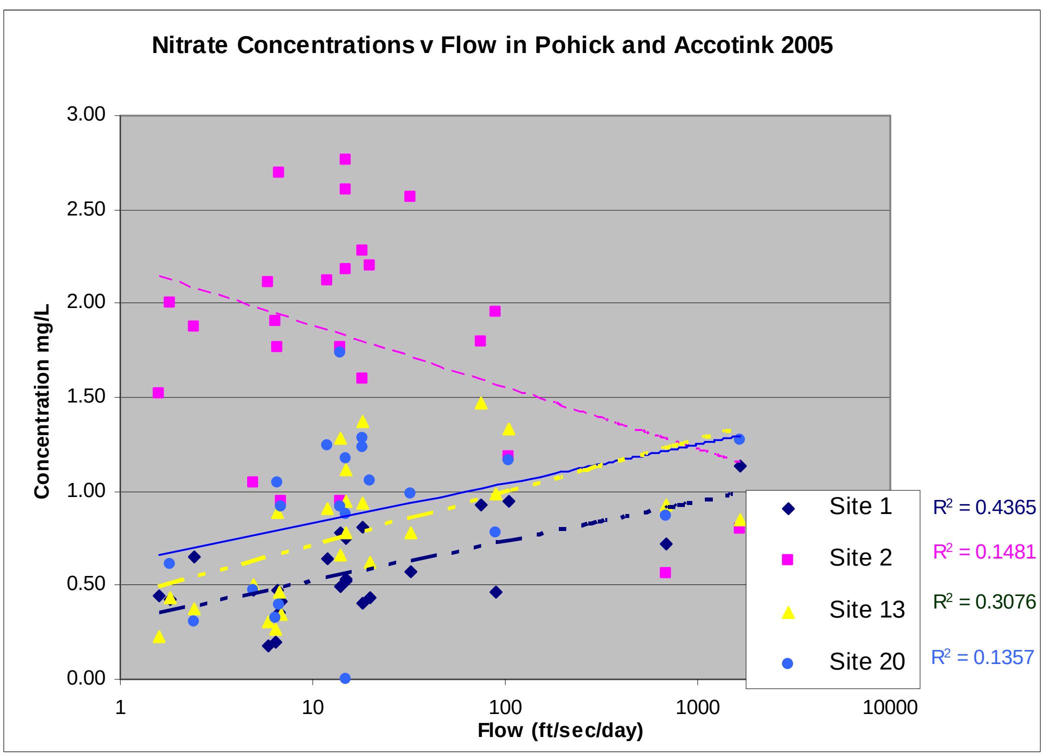


Figure 4.16. Nitrate concentration vs. flow. Of the point source dominated dischargers, nitrate appears highest at site 20 (upper Accotink) and clearly lower at site 1 (Pohick).

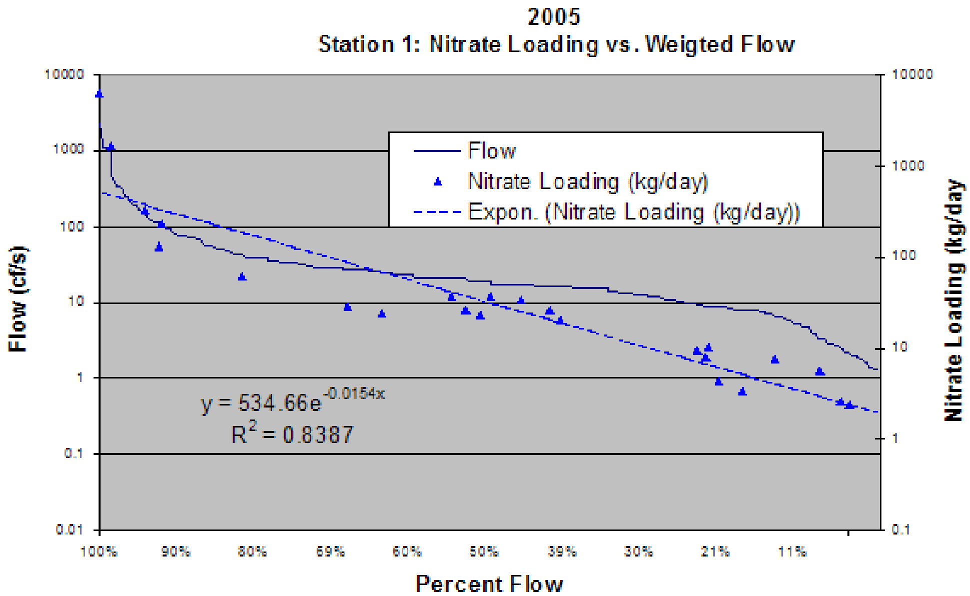


Figure 4.17. Flow duration curve for site 1 with Nitrate Loading. This graph consists of calculated daily loads (blue triangles), a flow duration curve (dark blue solid line), and a logarithmic regression of the daily loads versus the relative percent flow.

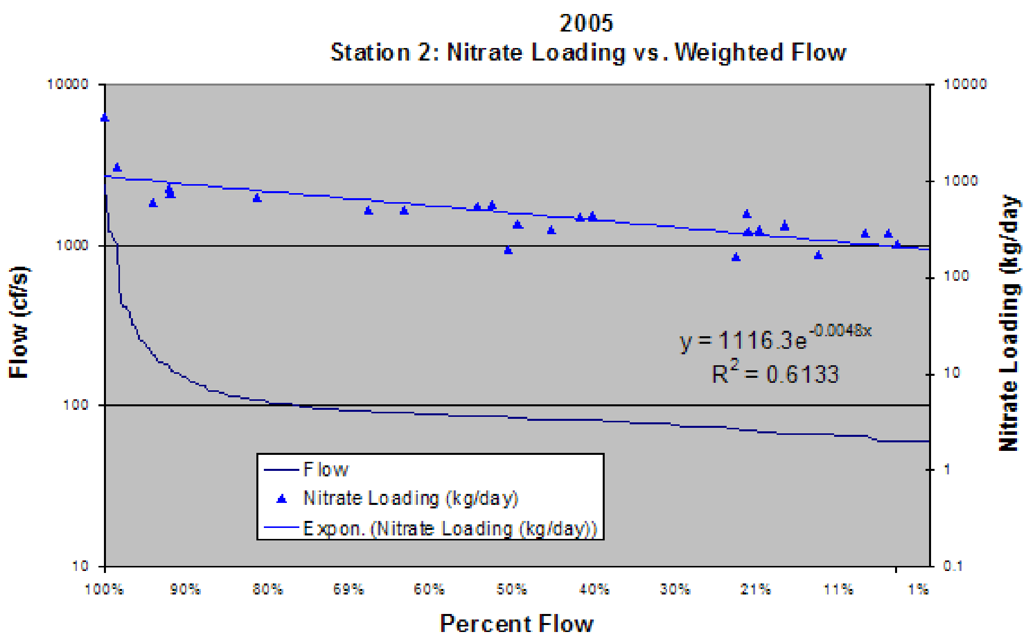


Figure 4.18. Nitrate loading and weighted flow for site 2 for 2005. Note that the values on the flow axis are on a different scale than in Figures 4.17 and 4.19.

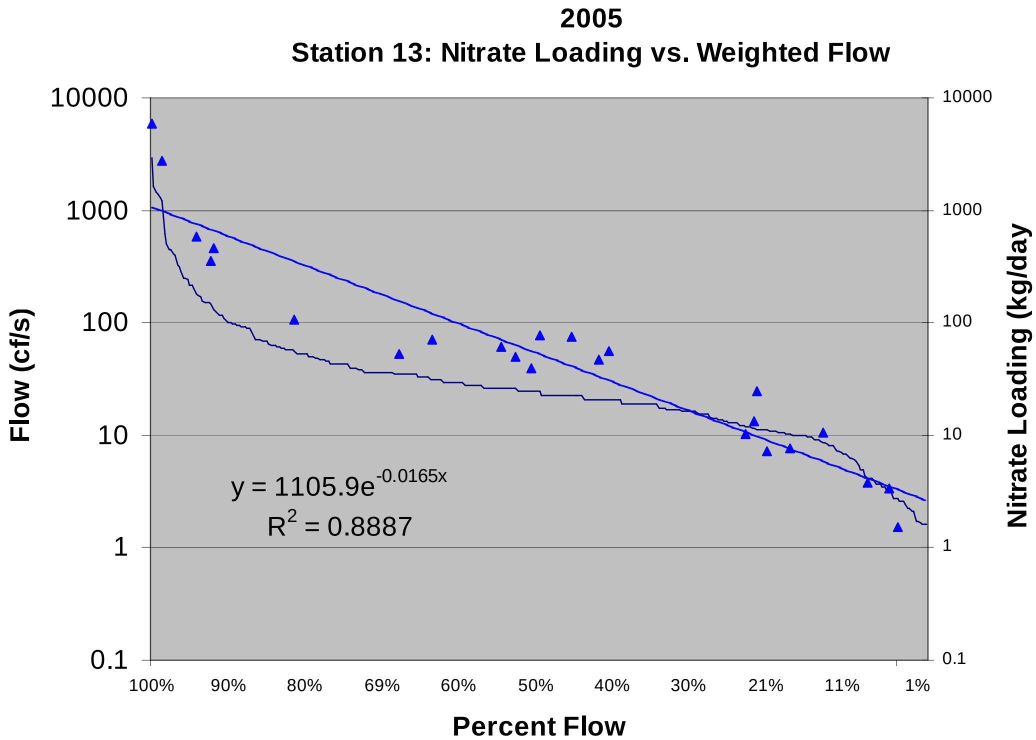


Figure 4.19. Nitrate loading and weighted flow for site 13 for 2005.

Temporal analysis of the data for nitrogen indicates that there has been a significant decrease in loadings from the Noman Cole sewage treatment facility, while there has been an increase in concentration and subsequent loadings from stormwater during the 1987-2005 time period. Examining the load duration curves for 1984-1992 and mean concentrations for these years, there is a marked decrease between these years and 2005 in nitrogen loadings for site 2. However, there is an apparent slight increase in loadings at sites 1 and 13. Additional load duration curves are available in Appendix B. These results are far from conclusive with the 12 year gap in data between 1993 and 2005, the lack of many high flow events for some of the earlier years, and climatic variability. Additionally, there is pattern of higher concentrations of nitrate primarily in

1984, but continuing to a lesser degree through 1986. There are several possible explanations for these higher concentrations: highly elevated atmospheric deposition, different sampling methodologies, differences in watershed dynamics or fertilizer usage. After initial unsuccessful attempts to explore the causes of the higher concentrations, the data for the 1983-1986 time period were often grouped into a second data set for analysis.

Atmospheric nitrogen deposition has undergone a slight decrease from 1989 to 2005 (Figure 4.20). Correll (1987) cited that 10.6 total-N ha yr-1 of total deposition were deposited at a site 20 km south of Annapolis for a 7 and a half year period in the 1970's. Two sites on opposite sides of the watershed for which monitoring data were available from 1989 to 2005 were located at Shenandoah National Park and Beltsville Maryland. Both the Shenandoah site and the Beltsville site had a total nitrogen deposition of 10.2 total-N ha yr-1 and 10.39 total-N ha yr-1 in 1989 respectively. Both sites had less than 7 total-N ha yr-1 in 2005; the year in which the average for the two sites was the lowest. The apparent decrease in nitrogen deposition indicates that watershed planners may get a small piece of good news in that nitrogen deposition appears to have decreased in recent years. This decrease is possibly due to two major factors: decreases in nitrate emissions from coal fire power plants and increasing reduction in nitrous oxides from automobiles. These results were taken as sufficient evidence to leave assumptions for nitrogen deposition steady in the forecasting models in chapter 5. Assuming that atmospheric nitrogen deposition may decrease, this leaves a margin of safety for any impacts that might be associated with climatic change, or increasing livestock grazing, policy changes, or coal power generation upwind of the Accotink and Pohick airsheds.

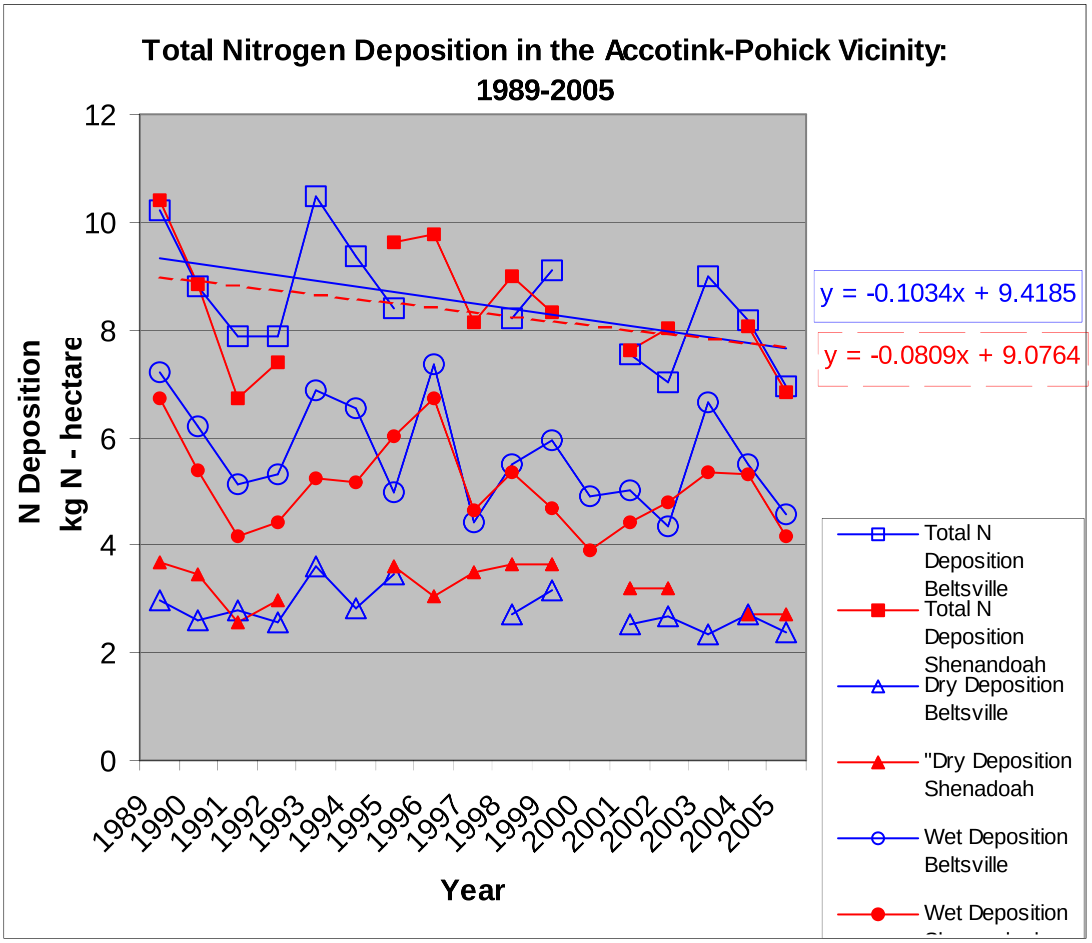


Figure 4.20. Both wet and dry nitrate deposition appear to be trending downward from 1989 to 2005.

Interestingly, there is a clear seasonal component to nitrate concentrations in Accotink above site 13, but the seasonality is not as clear in Pohick (though it was present, particularly in 2005) (Figures 4.21 and 4.22). For Accotink, mean concentrations are highest in spring, followed by concentrations in winter and summer (table 4.4). In Accotink, nitrate concentrations clearly increase with flow, particularly in Spring and Summer. In fall, concentrations are highest with low flow events and are relatively low with high flow events. Average concentration of nitrate is higher in winter but does not seem to vary with flow. Seasonality is statistically significant for most time periods using the nonparametric Kruskal Wallis test of the median (1983-1988 $p = 0.07$, 1989-1992 $p = 0.0002$, 2005 $p = 0.018$). Using the regression of concentration vs. the log of flow and a seasonal dummy variable for 1983-1992 and 2005, the coefficients for winter and spring were positive, while the coefficients for summer and fall were negative (table 4.5). This means that nitrate concentrations are higher relative to flow for winter and spring than for summer and fall. Hence, this regression illustrates seasonality, though there are clearly other factors between 1983-1992 that also explain variability. A reasonable explanation for the driver of this variance is the unexplained high values present in the 1984 and 1985 data.

Empirically, Pohick appears to have similar relationships regarding flow and concentration; however, the patterns are not so clear. Furthermore, mean concentration in Pohick is considerably lower than for Accotink in winter and spring, but it is comparable in summer and fall. Hence, the seasonality is not clear during the 1983-1992 time period, only somewhat present during the 2005 time period, and it is never statistically significant (1983-1988 $p = 0.90$, 1989-1992 $p = 0.06$, 2005 $p = 0.116$). There is likely a source of nitrate in Accotink in the winter and spring that is not present in Pohick. The Accotink results are consistent with past studies: Corell (1987) notes that nitrate loadings from Bay headwaters are seasonal and that these headwaters have higher concentrations in winter and spring.

Table 4.4. Mean concentrations of nitrate in Accotink and Pohick: 1983-1992; 2005

	1983-1992				2005			
	Winter	Spring	Summer	Fall	Winter	Spring	Summer	Fall
Site 1	0.54	0.59	0.61	0.55	0.58	0.70	0.47	0.48
Site 13	0.80	0.84	0.64	0.52	0.95	1.06	0.70	0.49

Table 4.5. Coefficients for the seasonal variable for the nitrate regression in Accotink and Pohick: 1983-1992 and 2005 using the equation:

$$Log(conc_{ps}) = \alpha \log \sum flow_s \quad + \beta \log \sum flow_{su} + \delta \log \sum flow_f + \varepsilon \log \sum flow_w \ .$$

	1983-1992						2005					
	Winter	Spring	Summer	Fall	R^2	p	Winter	Spring	Summer	Fall	R2	p
Site 1	0.113	-0.250	-1.383	0.000	0.051	=.07	0.272	0.356	-0.748	-0.145	0.251	=.116
Site 13	0.300	0.297	-1.285	-0.172	0.146	<.0001	0.353	.461	-.0434	-0.421	0.487	=.003

The influence of seasonality can be seen if one compares the equation listed in Table 4.5 to an equation without season as a component:

$$Log(conc) = \alpha \log \sum flow$$

For 1983 to 1992, omission of the seasonal component in the regression decreases the R^2 from 0.146 to 0.11 for site 13. For site 1, the R^2 decreases from 0.051 to .0035 and the p value increases from 0.070 to 0.145. These results further support the assertion that seasonality is responsible for some of the variance in nitrate concentrations, more-so in Accotink than in Pohick. Additionally, these results show that nitrate concentrations are more dependent upon flow in the historic data set in Accotink than they were in Pohick above the sewage treatment facility for the 1983-1992 data set. However, based on the higher R^2 in 2005 in Pohick, nitrate concentrations are more heavily influenced by flow, which means they are more heavily influenced by precipitation driven pollution.

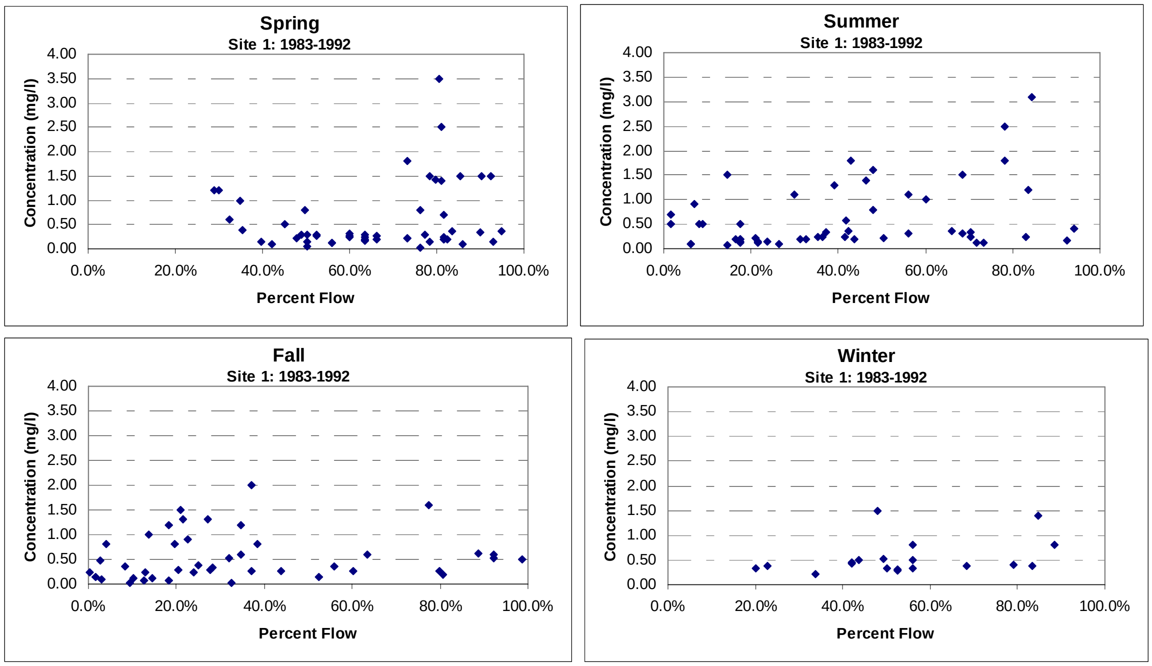


Figure 4.21. Relative concentration of nitrate vs. flow by season at site 1 from 1984-1992.

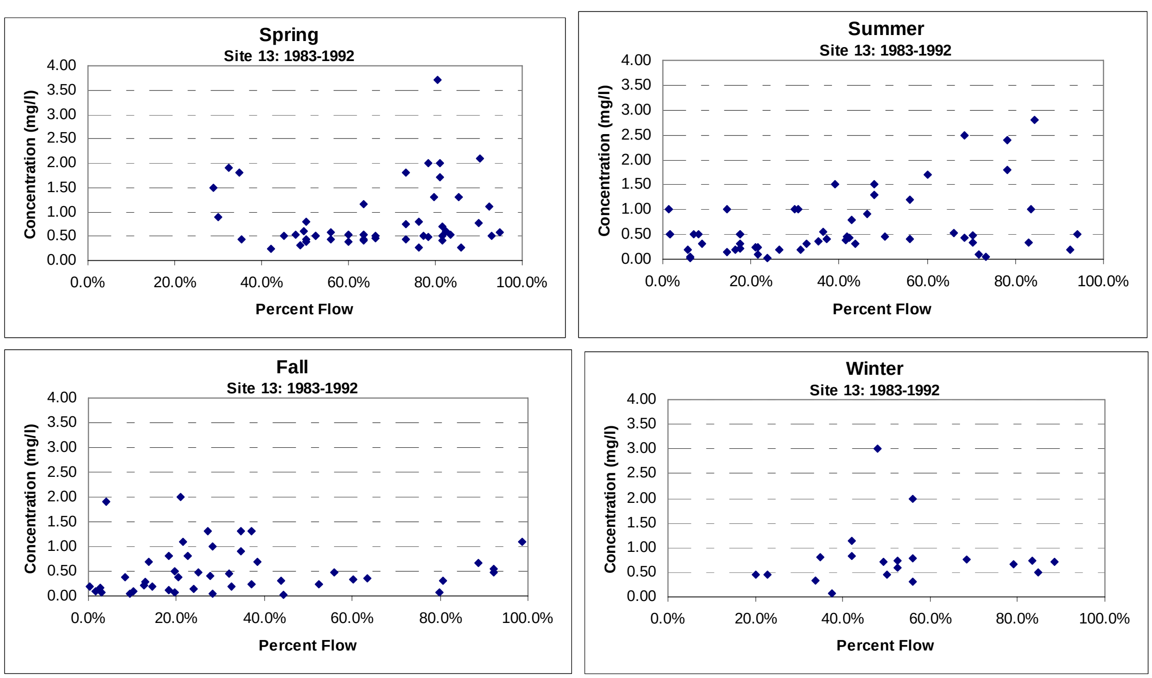


Figure 4.22. Relative concentration of nitrate vs. flow by season at site 13 from 1984-1992.

The Noman Cole sewage treatment facility began more advanced treatment for ammonia nitrogen around 2002, resulting in much lower discharges of nitrogen to Pohick. As the primary point source discharger in either watershed, this reduction is clearly significant. In Figure 4.23, one can notice that the data collected from 1984 to 1992 are clearly higher for all years for ammonia-N; however, they are also higher at

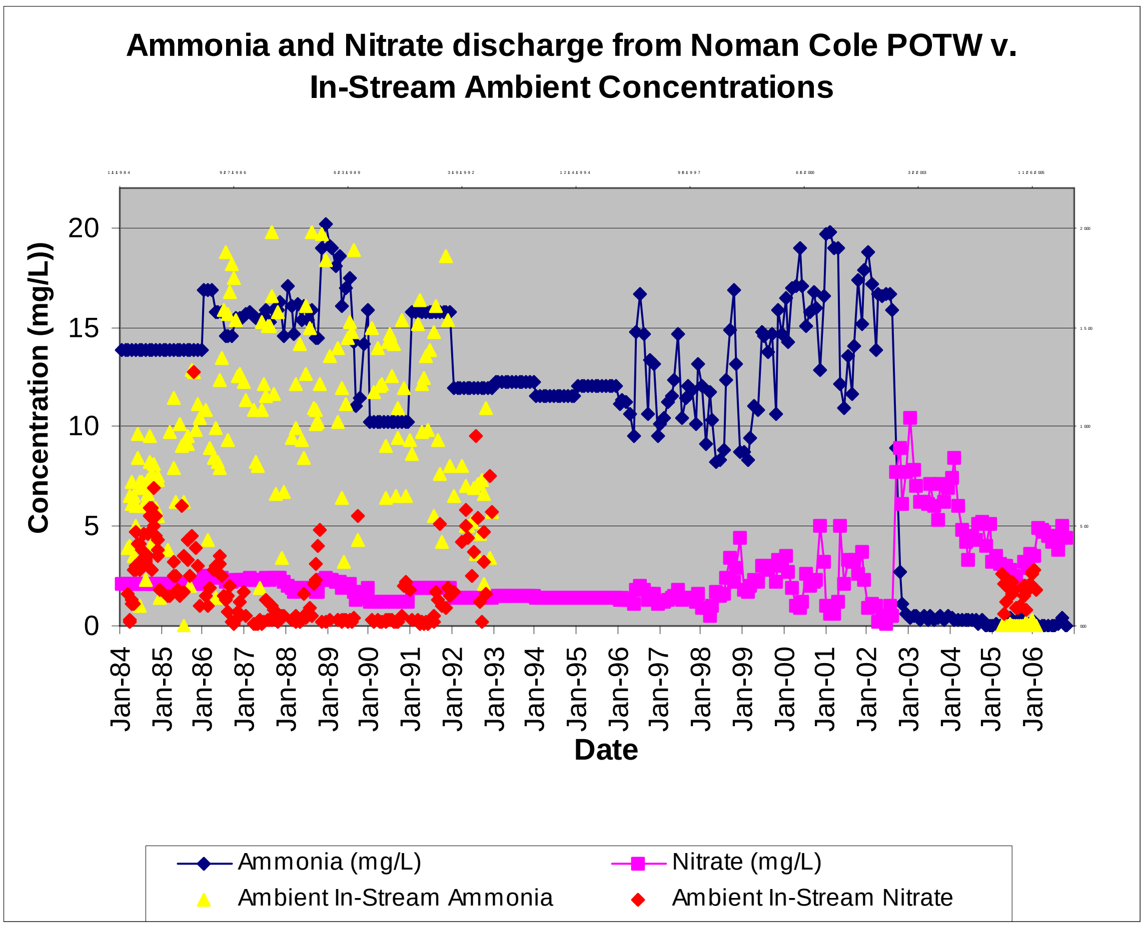


Figure 4.23. Ammonia discharge from the Noman Cole sewage treatment facility decreased sharply in 2002, resulting in significant reductions in the ammonia concentrations in Pohick Creek. As a result in decreasing nitrogen discharge, nitrate concentrations also stayed relatively low.

times for nitrate-N. Removal of ammonia from the effluent has clearly reduced ammonia-N concentrations in the streams, which no doubt has significant implications for downstream receiving waters and their freshwater species (due to ammonia toxicity and oxygen demand) as well as overall nitrate concentrations.

Like nitrate, high concentrations of ammonia appear to be flow driven. During 2005, all but four recorded concentrations were below 0.1 mg/L and the average concentrations and minimum concentrations for ammonia-N are both low for all sites (table 4.6). Furthermore, in 2005, ammonia-N concentrations were only significant during wet weather events (Figures 4.25 and 4.26). This may indicate a decrease in treatment efficiency at the Noman Cole facility during these time periods at site 2, or it may indicate a septic system overflow or similar leak immediately upstream of any of the monitoring sites. During low flow events, ammonia-N concentrations are typically low, indicating that ammonia is likely primarily converted to nitrite or nitrate, with some ammonia possibly being converted via denitrification. As can be seen previously in Figure 4.23, ammonia-N concentrations over the full study period (1983-2005) have decreased substantially below the Noman Cole sewage treatment plant due to enhanced nitrogen treatment controls. Ammonia-N concentrations are significantly lower than nitrate-N concentrations and account for far less of the nitrogen loading in both watersheds (Figure 4.24).

Table 4.6. Average, minimum, and maximum Values for ammonia-N in 2005.

Site	n	Avg. (mg/L)	Max (mg/L)	Min (mg/L)
Pohick 1	24	0.019	0.214	<0.001
Pohick 2	24	0.037	0.290	<0.001
Accotink 13	24	0.017	0.086	<0.001
Accotink 20	20	0.018	0.166	<0.001

Figure 4.25 shows that with the exception of two late autumn high ammonia-N concentrations, ammonia-N concentrations are consistent and relatively low. There is an October storm that caused widespread flooding and resulted in higher ammonia-N concentrations. These elevated concentrations are likely caused from human sources such as sanitary sewer overflows or from leaky septic systems. Nonetheless, ammonia-N concentrations are low across the board and are not the primary driver of nitrogen loading in these watersheds during 2005. In these regards, the increased nutrient treatment at the Noman Cole facility is highly successful. Figure 4.26 illustrates the relationship between ammonia-N concentrations and flow. Concentrations relative to flow are generally higher at site 2 than the other sites (p=.059 between sites 1 and 2 and p = .051 between sites 1 and 13), but there is not a clear distinction within the other three sites.

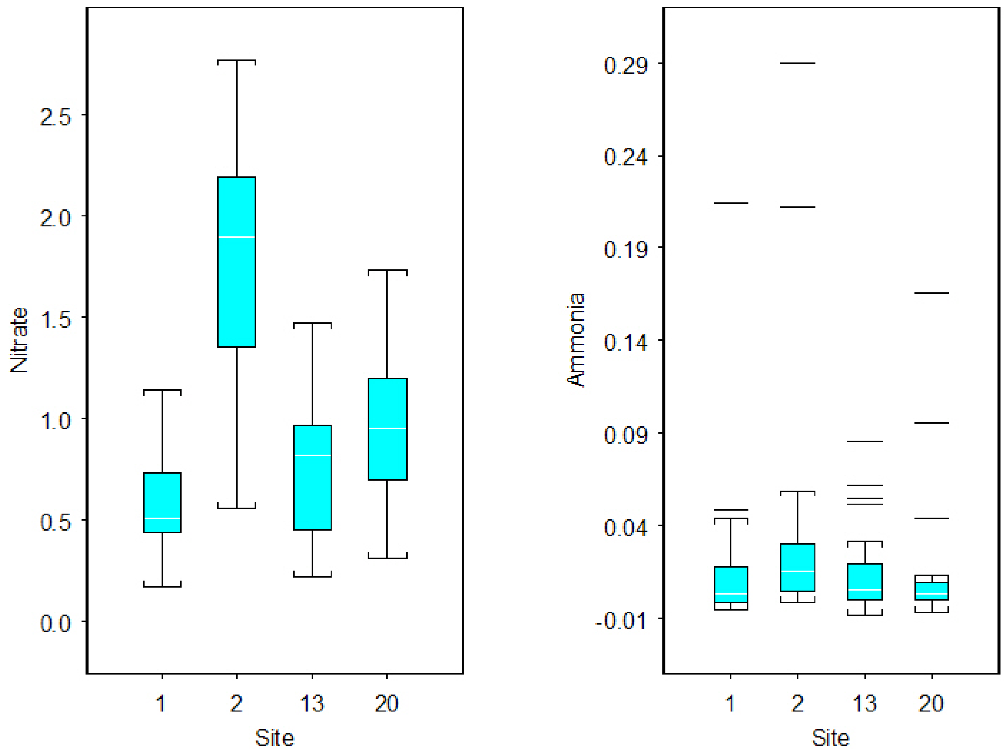


Figure 4.24. Boxplots of ammonia-N and nitrate-N concentrations for the four monitored sites for 2005.

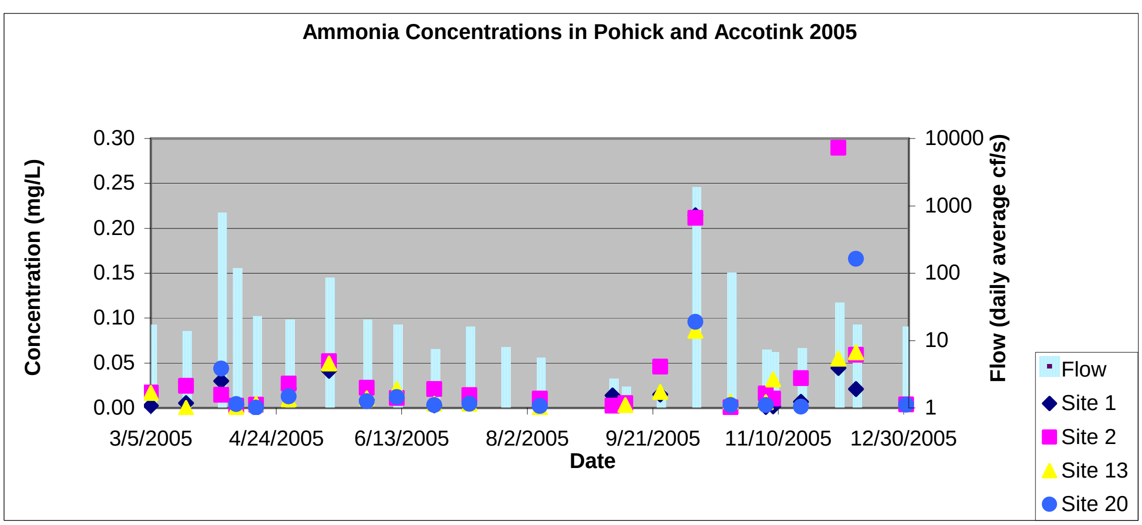


Figure 4.25. Ammonia-N concentrations and flow over time in Accotink and Pohick.

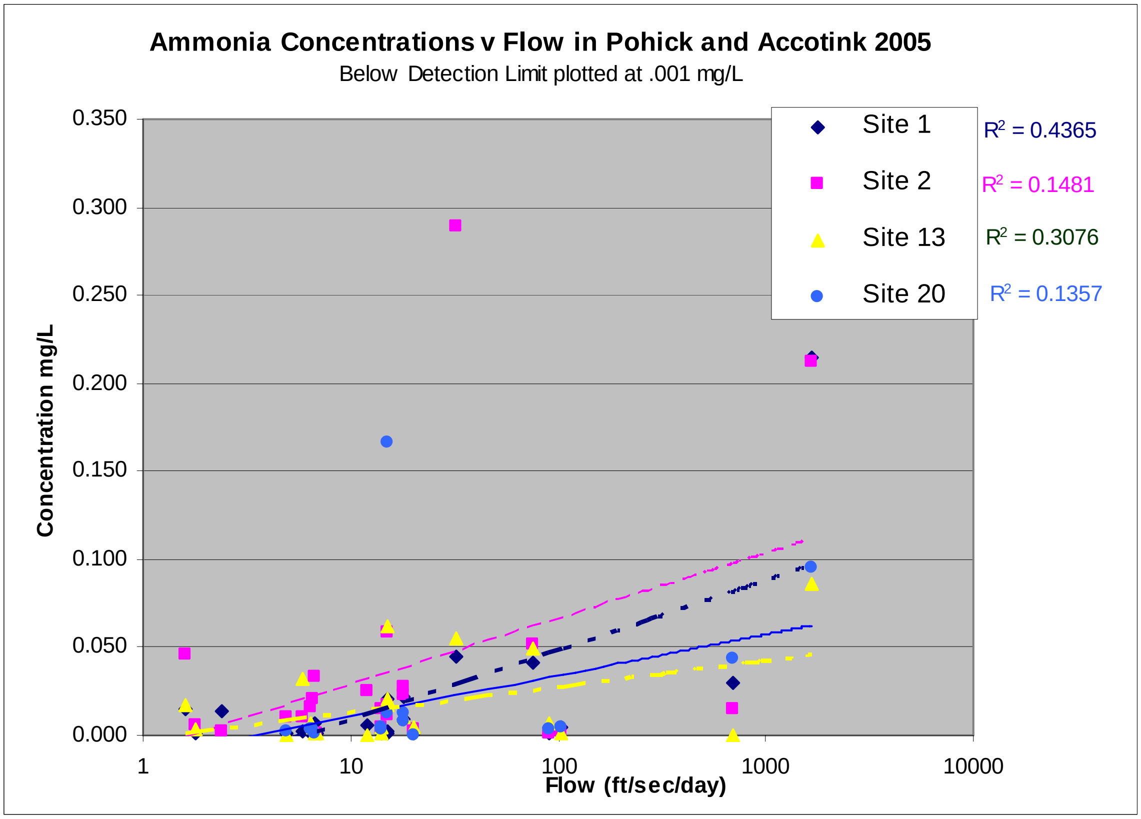


Figure 4.26. Ammonia-N concentrations versus flow in Accotink and Pohick.

4.3.4 Phosphorus

Like nitrogen, total phosphorus loadings in Accotink and Pohick were clearly flow driven, with the exception of historic loadings of phosphorus from the Noman Cole POTW. In stream phosphorus concentrations were lower in 2005 at site 2 than for previous monitoring years, indicating lower discharges from the Noman Cole facility (table 4.7). Furthermore, for 2005, average concentrations were marginally lower than for previous years, but the slope of the loading concentration lines for each year indicate that a higher percentage of total annual loadings may be coming from wet weather events than in previous years (Figure 4.27). These results are not conclusive, due to the under-representation of wet weather samples collected in previous sampling years. There are statistically significant differences between all sites, except between sites 1 and 13 (Figure 4.28). In 2005, low flow concentrations were still elevated below the sewage treatment facility relative to other sites, and once again, site 2 has the highest average concentrations (table 4.8). Table 4.8 also shows a much higher total phosphorus average concentration for 1990 than in any other year, driven by three extremely high concentrations in the winter of that year (19, 14.1 and 16.5 mg/L respectively). Site 13 also had one high concentration during 1990, raising its average value. Based on the total volume of loading, it is clear that the majority of phosphorus loads by weight comes from large flow events that are driven by stormwater. For example, in Figure 4.30, the slope of the regression line for 2005 is considerably higher than any of the previous monitoring years (1984-1992), indicating a higher percentage of loading coming from precipitation driven sources. Total phosphorus concentrations had a high correlation with flow at sites 1, 13, and 20, with each site having an R^2 of greater than 0.6. The relationship between phosphorus and flow was far weaker at site 2 due to the input from the treated sewage effluent (Figure 4.29). Site 20 had the highest average concentration between the sites that were primarily driven by stormwater loadings.

Table 4.7. Average, minimum, and maximum values for total phosphorus in 2005

Site	N	Avg. (mg/L)	Max (mg/L)	Min (mg/L)
Pohick 1	20	0.031	0.166	0.003
Pohick 2	20	0.064	0.318	0.015
Accotink 13	20	0.036	0.173	0.003
Accotink 20	15	0.047	0.203	0.011

Table 4.8. Average total phosphorus concentrations by site.

Year	Site 1	Site 2	Site 13
1983	0.035	0.163	0.068
1984	0.039	0.105	0.043
1985	0.053	0.124	0.058
1986	0.065	0.127	0.055
1987	0.043	0.132	0.052
1988	0.031	0.074	0.035
1989	0.040	0.063	0.034
1990	0.098	2.829	0.217
1991	0.043	0.080	0.048
1992	0.041	0.097	0.035
2005	0.031	0.064	0.036

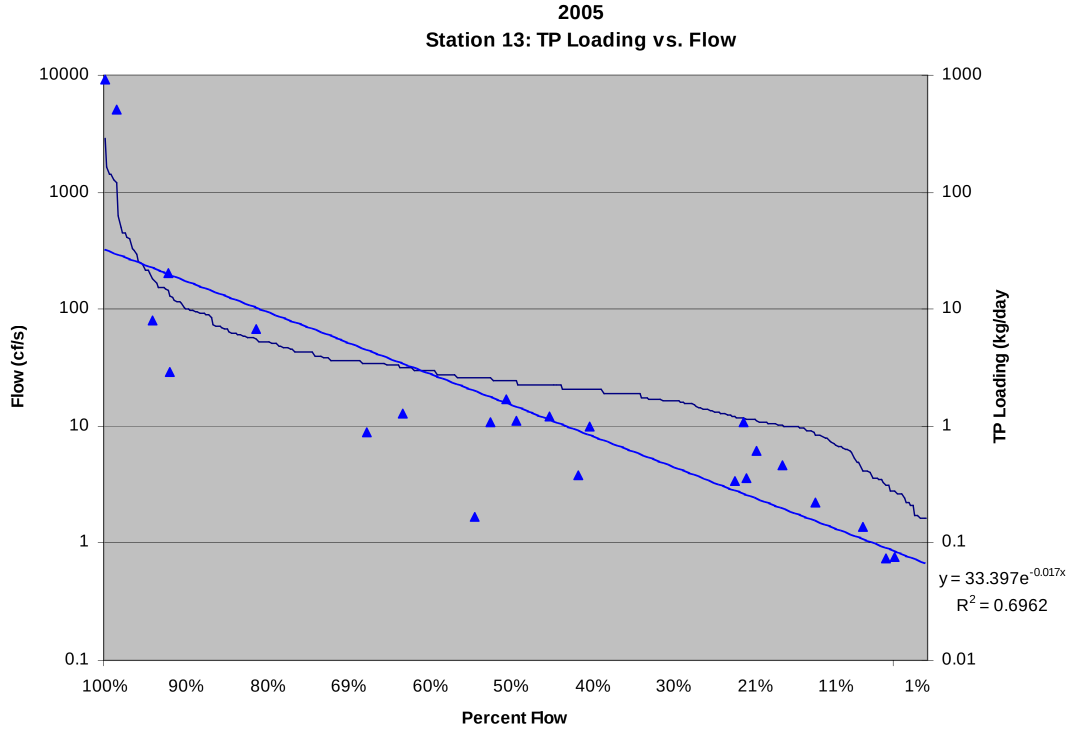


Figure 4.27. Flow duration curve with total phosphorus loading at site 13.

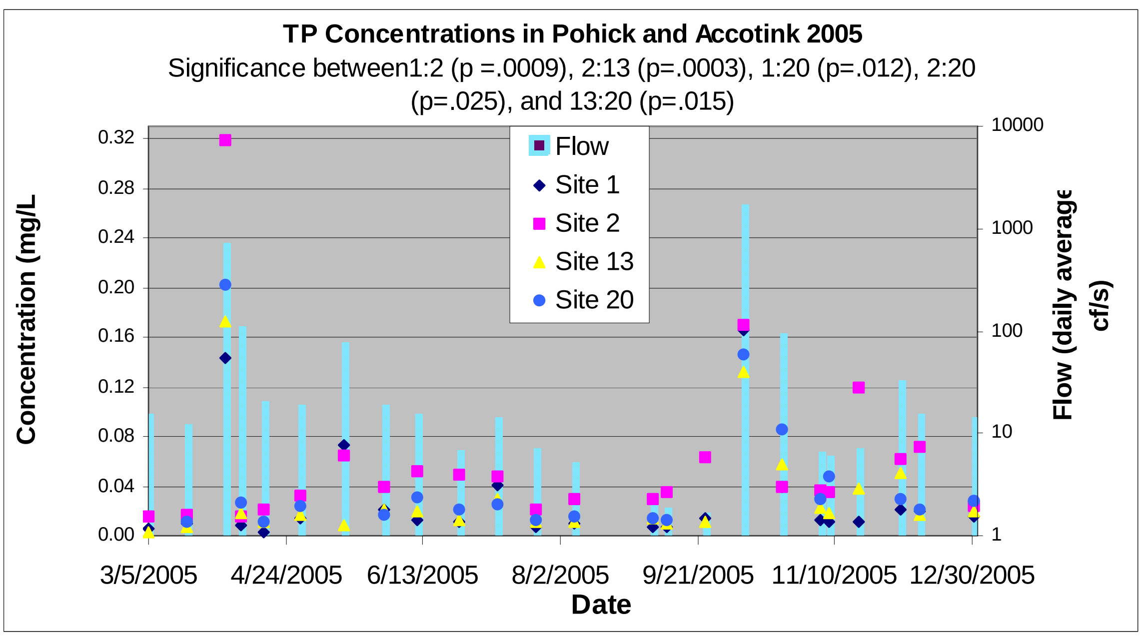


Figure 4.28. Total phosphorus concentrations and flow over time in Accotink and Pohick.

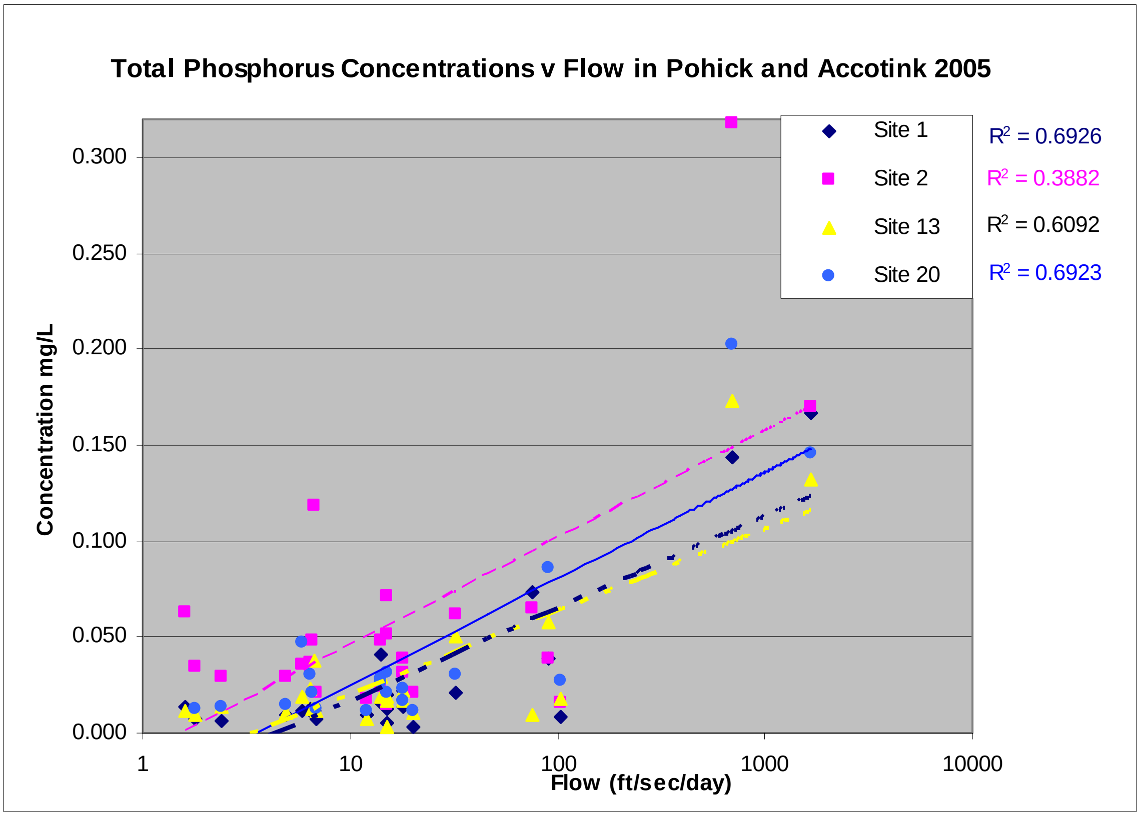


Figure 4.29. Total phosphorus concentrations versus flow in Accotink and Pohick.

Relationships with soluble reactive phosphorus (SRP) are far less clear. There is a significant difference (p=.047) between site 1 and site 2 with site 2 having a considerably higher average concentration, indicating that there is some discharge coming from the Noman Cole plant; however, the magnitude and clarity of this difference are far reduced relative to other nutrients. Average concentrations of SRP are higher in Accotink at sites 13 and 20 than at Pohick site 1 (table 4.9). At site 13 and 20, there is some correlation between flow and SRP concentration, with higher flow tending to have higher SRP concentrations (Figures 4.30 and 4.31). However, these higher concentrations are primarily from the highest flow event in October. Concentrations tend to be higher in the Accotink sites than at the Pohick 1 site, however, differences are not significant (1 to 13: $p = .13$, 1 to 20 $p = .27$). There are lower flow events that have SRP concentrations above 30 ug/L, three of which occur at site 2. These events primarily occur in late summer or autumn. Because of the isolated incidence of these events, there is either an isolated upstream discharge (more likely at site 2 and site 1 or 13) or possible sample contamination.

Table 4.9. Average, minimum, and maximum values for soluble reactive phosphorus in 2005

Site	n	Avg. (ug/L)	Max (ug/L)	Min (ug/L)
Pohick 1	22	6.4	35.8	2.0
Pohick 2	22	19.4	125.3	2.0
Accotink 13	20	13.4	70.8	2.0
Accotink 20	16	12.1	97.1	2.0

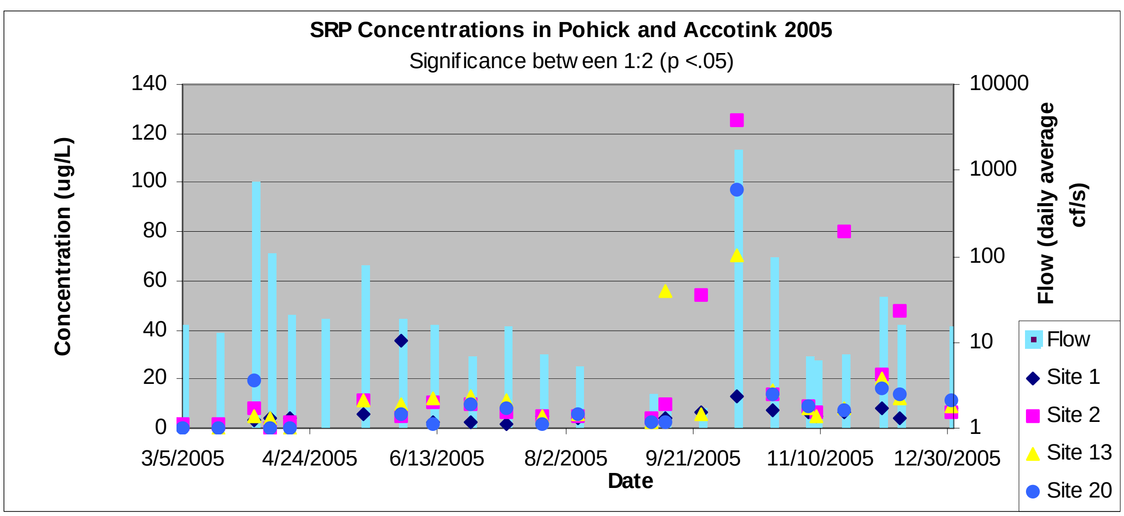


Figure 4.30. Concentration of SRP and flow vs. time.

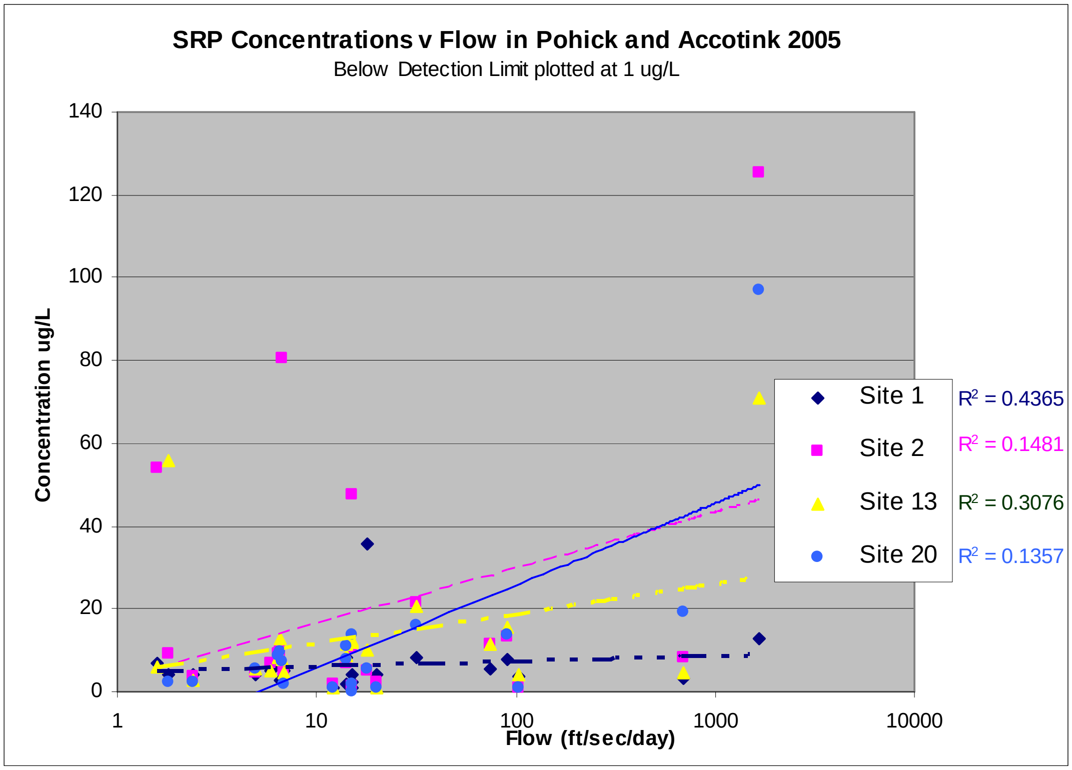


Figure 4.31. Orthophosphate (SRP) concentrations vs. flow for 2005. Relationships between flow and concentration are less clear than with nitrate, ammonia, or total phosphorus.

4.3.5 Suspended Solids

Total suspended solids (TSS) exhibited a strong relationship with flow at all sites in 2005 (Figures 4.32 and 4.33). Average concentrations of TSS were highest at the Pohick 1 site, followed by the Accotink 20, Accotink 13, and Pohick 2 (Table 4.10). Even excluding the highest flow dates (where samples at sites 2, 13, and 20 were taken from the floodplain and the sample at site 1 was taken from the channel), the average concentration of TSS is highest at site 1. These results are somewhat contradictory to what was expected since Accotink has a higher percentage of urban land use. There are three possible explanations: the first is that samples for the highest flow event had to be taken from outside the main channel at sites 2, 13, and 20 due to safety issues, and measured TSS was clearly lower than in the main channel. Excluding the measurements for this highest flow day, average TSS was still higher at Pohick 1 than the other sites, but the difference is far smaller, particularly between site 1 and site 20. The second explanation is that Pohick has had increased construction activity for the last 15 years relative to Accotink: hence, it is quite likely that Pohick is receiving a greater contribution of construction site runoff. Third, Accotink underwent urbanization earlier, and as such, the stream has had longer to scour. With the relatively recent urbanization of Pohick, the stream channel is not as wide or scoured and it may have increased pressure during the large flow volume events that come with urbanization. Therefore, more sediment may be eroded from stream banks in Pohick during these events. Concentrations decrease somewhat below the sewage treatment facility for two reasons. First, there is a large riparian floodplain/wetland that has capacity to handle excess water. Second, the channel receives a consistent volume of flow from the Noman Cole facility for which the stream channel may have accommodated. Additionally, effluent from the Noman Cole facility is not very turbid and, even though the facility is only responsible for a small percentage of flow during high flow events, the effluent dilutes the stream water somewhat.

Table 4.10. Mean, maximum, and minimum concentrations for total suspended solids (TSS) and Volatile Suspended Solids (VSS) (mg/L).

Site	n	TSS Avg.	VSS Avg.	TSS Avg. (minus highest flow day)	TSS max	VSS max	TSS min	VSS min
Pohick 1	24	66.7	10.9	47.8	909	85	3	3
Pohick 2	24	30.7	7.7	28.2	434	50	3	2
Accotink 13	24	35.9	8.8	33.5	510	66	2	2
Accotink 20	21	50.6	10.3	46.4	692	70	3	2

The fact that Site 1 and Pohick had higher TSS during the highest measured concentration events is consistent with historical measured TSS values. From the 1984-1992 time period, a total of 20 samples were measured with concentrations of greater than 50 mg/L. Nine of these were at site 1 (Pohick) while only 4 were at site 13 (Accotink). Furthermore, of the 9 sample days that had at least one sample from site 13 or site 1 with greater than 50 mg/L, 8 of those days had greater measured concentrations at site 1. The Pohick watershed was growing at a much faster pace during these years than the Accotink watershed: hence, these results are consistent with previous explanations.

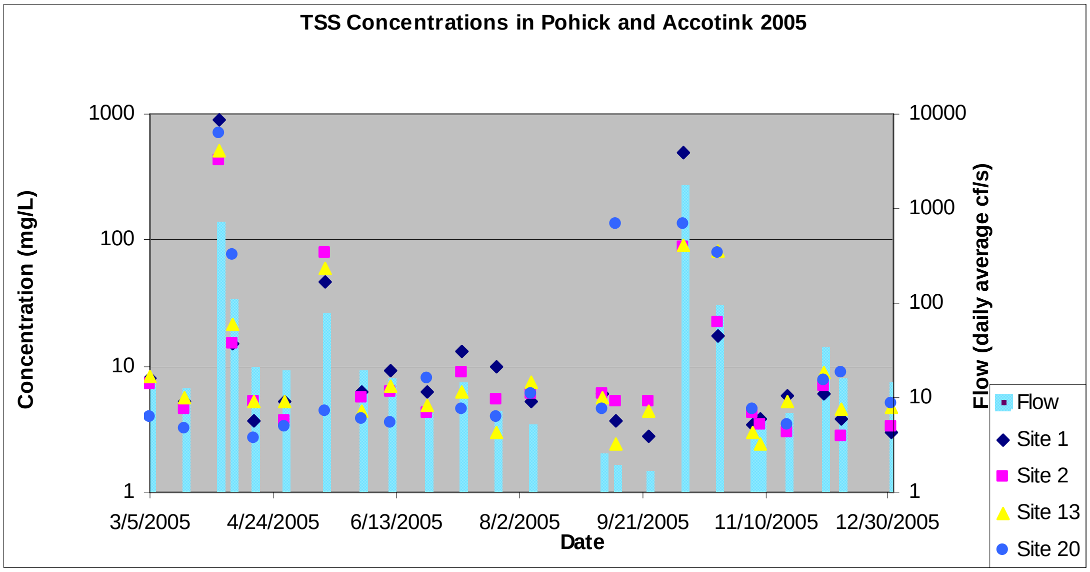


Figure 4.32. Concentration of TSS (mg/L) and flow vs. Date. There appears to be a slight trend toward the highest flow events appear to cause higher TSS concentrations in Pohick

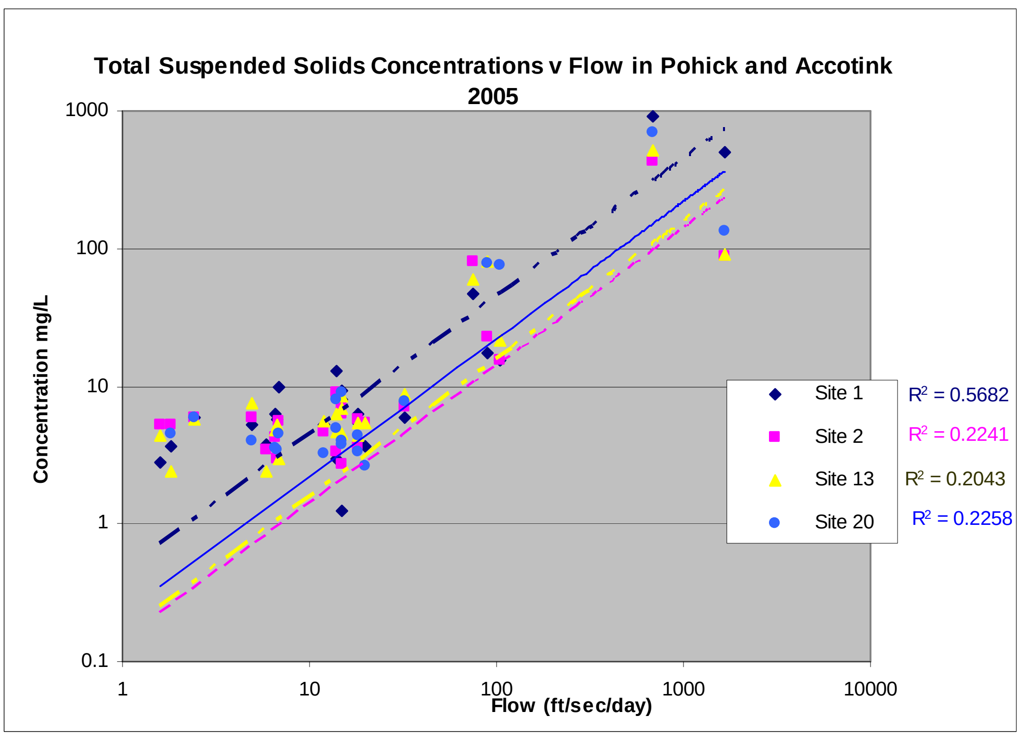


Figure 4.33. Concentration of TSS vs. flow in 2005 with a linear regression with the intercept forced to 0. The final lower TSS concentration for the highest flow day lowers the R^2 considerably for the linear regressions, but the relationship between flow and concentration is clear. Excluding the highest flow event raises the R2 from .623 (site 1) to .9941 (site 20).

Average TSS concentrations are higher in 2005 than previous years (table 4.11); however, this is likely due in part to more wet weather samples being taken in 2005 compared to other years and 2005 being a wet year. For example, from 1984 to 1992 at site 13, 2 samples (1.2%) were taken when flows equaled or exceeded 100 cf/s at the Accotink gauge, 65 (42.5%) were taken between 10 and 100 cf/s, and 86 (56.2%) were taken with flow less than 10 cf/s. In 2005, 3 samples (12.5%) were taken when flows equaled or exceeded 100 cf/s at the Accotink gauge, 12 (50.0%) were taken between 10 and 100 cf/s, and 9 (37.5%) were taken with flow less than 10 cf/s.

Table 4.11. Average TSS concentration (mg/L) at sites 1, 2, and 13.

Year	Site 1	Site 2	Site 13
1984	8.9	5.1	10.6
1985	31.0	26.1	33.2
1986	29.3	26.9	23.6
1987	18.8	13.0	11.3
1988	5.8	4.0	8.1
1989	10.3	8.7	12.6
1990	5.6	7.8	3.9
1991	10.3	7.3	9.7
1992	8.7	5.4	6.8
2005	66.7	30.7	35.9

Volatile suspended solids (VSS), or roughly the organic fraction of solids, exhibited a strong relationship with flow; however, the increase in concentration was substantially lower as flow increased than with TSS. As flow volume increased, the percentage of organic solids decreased as can be seen by the VSS/TSS ration in Figure 4.34, indicating that the majority of sediment is inorganic in high flow events. Organic solids make up a lower percentage of solids in both Pohick sites during flow events lower than 20 cubic per second, perhaps indicating different loading sources or sediment contributed to these streams being finer and taking longer to settle out. The percentage of organic solids is likely overestimated in the highest flow events due to samples being taken from the floodplain from sites 2, 13, and 20. Since inorganic sediment is denser, it likely settles out quicker. Figure 4.34 clearly shows that in high flow events, the majority of additional sediment is inorganic; i.e., from either in stream bank scouring or run-off from construction sites and is not a result of biomass wash-in.

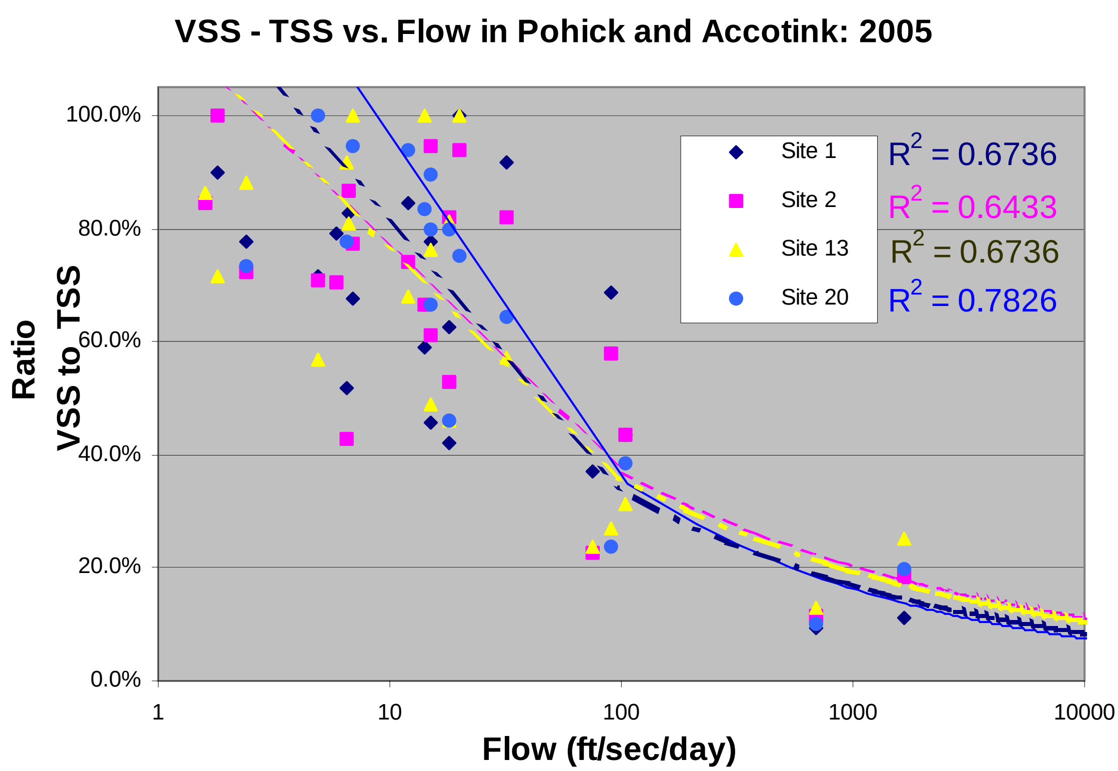


Figure 4.34. Relationship of VSS to TSS in measured samples.

4.3.6 Vegetated Buffer Protection and Relative to Urban Development

As noted in chapter 3 of this work, urban land cover has increased significantly during the course of this study period. Chapter 3 did not discuss the protection of riparian buffers, or what Fairfax County has designated as resource protection areas. In 1993, the county designated multiple areas as resource protection areas, including those bordering perennial streams. In 2003, the county designated additional resource protection areas based on field sites. These resource protection areas are defined as:

> "(b) RPAs shall include any land characterized by one or more of the following features:
> (1) A tidal wetland;
> (2) A tidal shore;
> **(3) A water body with perennial flow;**
> (4) A nontidal wetland connected by surface flow and contiguous to a tidal wetland or water body with perennial flow;
> (5) A buffer area as follows:
> (i) Any land within a major floodplain;
> **(ii) Any land within 100 feet of a feature listed in Sections 118-1-7(b)(1)-(4).**" (Fairfax County, 2005)

Based on the 1992 and 2000 NLCD data, it appears that these original resource protection area designations have been reasonably successful in limiting most urban land use intrusion into the riparian buffers. In 1992 (1 year before the county designated resource protection areas), urban land use in the 30 meter buffer zone varied from approximately 22.3% to 37.6% (table 4.12). In 2000, urban land use in the 30 meter buffer zone had only increased to between 26.0% to 38.3% urban (table 4.13). Note that the relative amount of urban area in the buffer zones decreased compared to the overall urban area watershed wide (tables 4.12 and 4.13). Furthermore, as shown in table 4.13, the watersheds that underwent urbanization earliest (starting with Accotink 20) have the highest percentage of urban intrusion into the buffer area relative to those that underwent urbanization later (Pohick).

It seems the county policy has been effective in protecting most county riparian buffers since its implementation. There is a slight increase in urban area in the region within 30 meters of streams between 1992 and 2000 in Pohick and the upstream areas of Accotink (Accotink 20), but a slight decrease is noted in the rest of Accotink. The ratio of new urban area in the buffer zone compared to new urban area watershed wide is between -14.0% and 27.7% (table 4.14). Unless this difference is due solely to classification error, which is unlikely, these statistics mean that development is being guided away from riparian areas. Where there was development in the 30 meter buffer zone between 1992 and 2000, this urbanization is likely due to a combination of exceptions being granted to allow development in the buffer area, small scale violators of the county policy, differences in classification between the 1992 and 2000 remote sensing data, and some streams may not be classified as perennial that are on the county's hydrologic map layer (and as such do not get RPA legal protection).

Table 4.12. Urban area in each watershed compared with urban area in the region 30 meters (98.43 feet) from the edge of the watersheds water features for 1992.

Watershed	Percent Urban in Full Watershed	Percent Urban in Buffer Zone	Ratio of Urban Land Use in Buffer zone to Overall Watershed
Pohick 1	38.60%	22.29%	57.74%
Pohick 2	38.99%	23.07%	59.17%
Pohick-all	36.29%	21.77%	59.99%
Accotink all	50.97%	34.64%	67.96%
Accotink 13G	51.80%	35.77%	69.05%
Accotink 13M	52.69%	34.63%	65.73%
Accotink 20	53.81%	37.61%	69.89%

Table 4.13. Urban area in each watershed compared with urban area in the region 30 meters (98.43 feet) from the edge of the watersheds water features for 2000.

Watershed	Percent Urban in Full Watershed	Percent Urban in Buffer Zone	Ratio of Urban Land Use in Buffer zone to Overall Watershed
Pohick 1	52.10%	26.03%	49.96%
Pohick 2	52.20%	26.59%	50.94%
Pohick-all	47.79%	24.42%	51.09%
Accotink all	57.44%	33.73%	58.73%
Accotink 13G	58.56%	35.09%	59.92%
Accotink 13M	59.34%	34.24%	57.71%
Accotink 20	59.39%	38.37%	64.61%

Table 4.14. Additional percentage of land use as urban area in the watershed (over the total watershed area) compared to the increase in urban area in the buffer zone (over the total buffer area in the watershed.

Watershed	Increase in urban area Watershed wide from 1992 to 2000	Increase in urban area in the buffer zone	Ratio of new urbanization in the buffer zone to compared to urbanization watershed wide
Pohick 1	13.5%	3.7%	27.7%
Pohick 2	13.2%	3.5%	26.6%
Pohick-all	11.5%	2.6%	23.0%
Accotink all	6.5%	-0.9%	-14.0%
Accotink 13G	6.8%	-0.7%	-10.1%
Accotink 13M	6.6%	-0.4%	-5.8%
Accotink 20	5.6%	0.8%	13.7%

4.3.7 Estimated Loadings of Parameters – Results

Annual precipitation derived loadings appear to increase from 1986-2005 for both watersheds, although they do not increase at rates directly proportional to the amount of urban land use (Figures 4.35 and 4.36). Point source nitrogen loadings from the Noman Cole sewage treatment facility decrease significantly during this time period (Figures 4.23 and 4.49). This is particularly important for ammonia-N loadings being delivered into Pohick and ultimately Gunston Cove. In short, loadings of ammonia-N and phosphorus to Gunston Cove have decreased significantly thanks to the improved treatment efficiency at the Noman Cole Plant.

Figures 4.37 and 4.38 show the annual loading estimations for each of the four loading estimation methods (simple linear regression approach, load duration linear regression, LOADEST default, and LOADEST urban land use). The linear regression and load duration approach were each run for individual years, and the slope of those regressions was used to estimate loads. The LOADEST default model was run for time periods 1983-1987, 1988-1992, and 2005 to minimize the impact of the unexplained high concentration events in 1984. The LOADEST model using Urban Land Use, urban intrusion into the buffered area, and estimated BMP implementation was run for 1983-1987 and 1988-2005. For years such as 1992, when all of the samples were taken in low flow conditions (less than the median flow day), the variance between the four methods is greatest, varying from 7,935 kg nitrate-N/year to 38,427 kg nitrate-N/yr. This means that the 1992 load estimates are likely low. For years such as 2006, when the samples were taken in a variety of flow conditions, the variation between the methods was much smaller as a percentage of total: all four approaches estimated loading between 67,409 kg nitrate-N/year and 81,857 kg nitrate-N/year. These load estimates are likely more representative of actual watershed conditions.

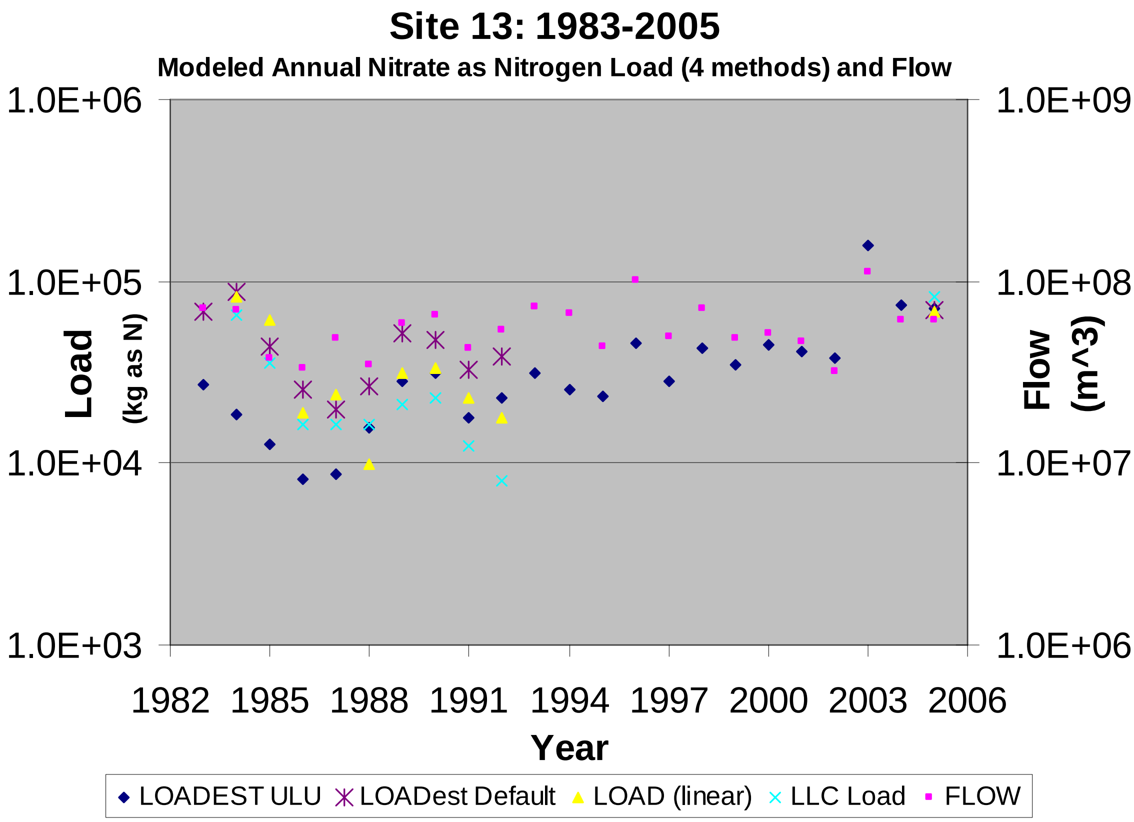


Figure 4.35. Estimated annual load of nitrate-N at site 13 (Accotink) using four methods compared to flow. For the LOADEST Method using urban land use, loading estimates from 1993-2004 were made.

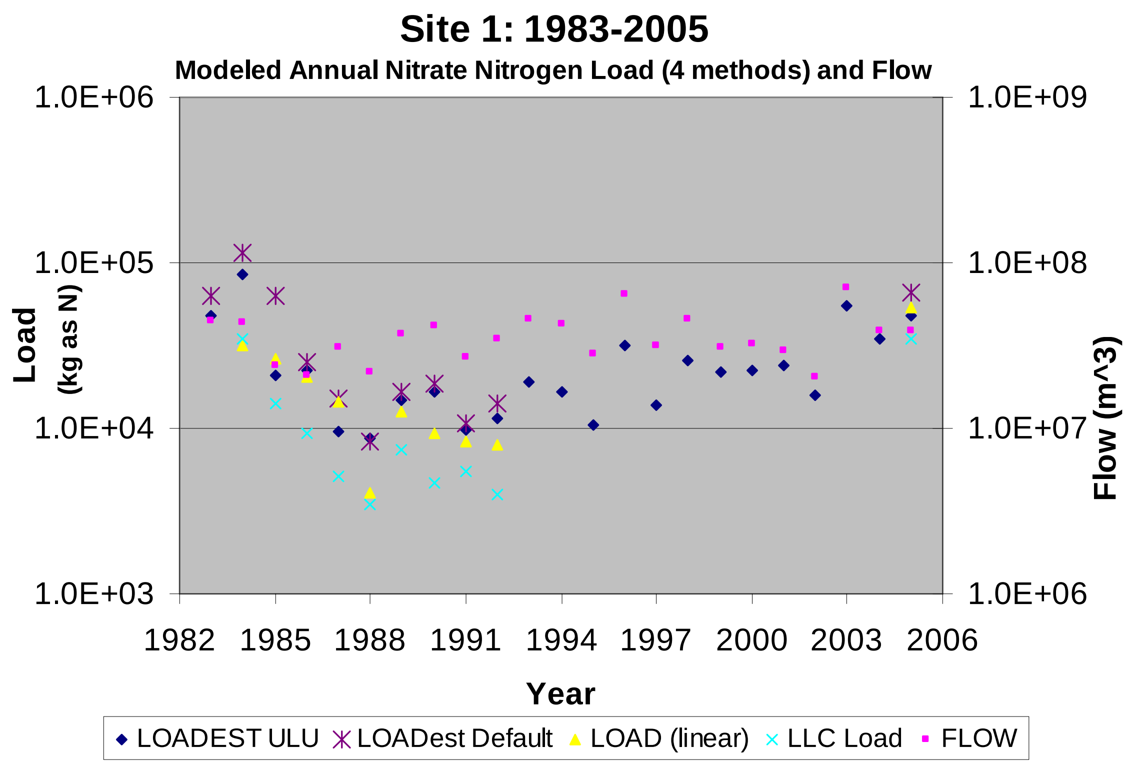


Figure 4.36. Estimated annual load of nitrate-N at site 1 (Pohick using four methods. For the LOADEST Method using urban land use, loading estimates from 1993-2004 were made.

The year 2005 had the 9th most annual flow volume during the time period 1983-2005. However, load delivered to the streams was relatively higher than its relative flow volume compared to previous years. The nitrate-N load in 2005 at site 13 was 3rd largest (of 23) using the LOADEST urban land use approach, 2nd (of 11) with the LOADEST default model, 2nd (of 10) with the linear regression, and 1st (of 10) with the LOAD duration approach. In other words, nitrate-N loading has increased at a more rapid rate than total flow volume. There are similarities in patterns at site 1, with the loading being 3rd of 23 with the LOADEST ULU approach, 2nd of 11 with the LOADEST default approach, 1st of 10 with the Linear approach, and 2nd of 10 with the Load duration approach. These results are particularly noteworthy when they are put in context that the LOADEST model ULU approach shows 2004 and 2005 as the 1st and 2nd largest load years. Furthermore, 1984 tended to have relatively high loadings attributed to it for most methods and was the only year higher than 2005 in some cases. This is striking since 1984 is the year with multiple unexplained high value measurements.

Another way of looking at these results is to compare 2005 loadings to those of previous years. The year 1990 had similar amount of annual flow to 2005, with an estimated 41.4 x 10^6 cubic meters of water flowing past site 13 compared to 38.7 x 10^6 for 2005. Nonetheless, estimated load at site 1 was between 289% to 740% higher for 2005 compared to 1990. The 1990 loading estimates are likely low, particularly for the load regression line method and the linear regression method, because there were no wet weather samples in 1990. Comparing 1989 to 2005, 1989 is a marginally drier year (37.3 x 10^6 cubic meters compared to 38.7 x 10^6 cubic meters in 2005), and relatively few wet weather samples were taken (the highest flow day sample was taken with a flow of 105 cubic feet per second (2.70 cubic meters per second)). Loadings in 1989 were estimated between 325% to 465% percent lower than 2005 dependent upon method.

Though there were increases in estimated load to site 13, these increases were not as substantial as at site 1. With the exception of 1984, the relative loading compared to flow volume seems to have increased at a greater rate in Pohick (site 1) than in Accotink (site 13). The increase for 2005 relative to 1989 is between 134% (LOADEST default) and 393% (load duration regression line). Between 1990 and 2005, the difference is 144% (LOADEST default), and 359% (load duration regression line). The high relative increase of the load duration regression line approach highlights the likelihood of underestimating load with this approach if samples taken are not distributed across the flow spectrum. Hence, the percent increase in loading between 1990 and 2005 is likely overstated since 1990 did not have significant wet weather sampling; however, what is most noticeable is that nitrate-N loading increased at a faster rate in Pohick than in Accotink. Nitrate-N loading increased at both site 1 (Pohick) and site 13 (Accotink), but it appears to have increased at a slightly higher rate in the Pohick watershed. This is consistent with an increased rate of urbanization present in Pohick vs. Accotink, as shown in Table 4.15.

Table. 4.15. Increase in estimated load from 1990 to 2005 in Accotink and Pohick at two representative sampling sites.

	Site 1	Site 13	% Difference between 1 and 13
1990 ULU (km2)	31.02	65.98	112.7%
2004 ULU (km2)	43.95	81.69	85.9%
% Change	37.5%	23.4%	
1990 Nitrate-N Load kg (ULU approach)	16540	31405	89.9%
2005 kg (ULU approach)	47820	70656	47.8%
% Increase from 1990 to 2005	356.8%	225.0%	
1990 (LOADEST default)	18722	47963	156.2%
2005 (LOADEST default)	66796	69043	3.4%
% Increase from 1990 to 2005	289.1%	144.0%	

As total annual flow increases, the annual nitrate-N loading increases in a polynomial fashion (Figure 4.37). However, the most recent data year (2005) tends to produce nitrate-N loadings that are higher than previous years relative to the amount of annual flow. In Figure 4.39, the large sample points represent estimated nitrate-N load for 2005. Hence, it is likely that nitrate-N loading meter^{-3} of water is increasing, which further supports that nitrate-N loading from precipitation driven sources in the watershed is increasing. Therefore, the increase in nitrate-N loading is likely attributable to the increase in urban area in the two watersheds.

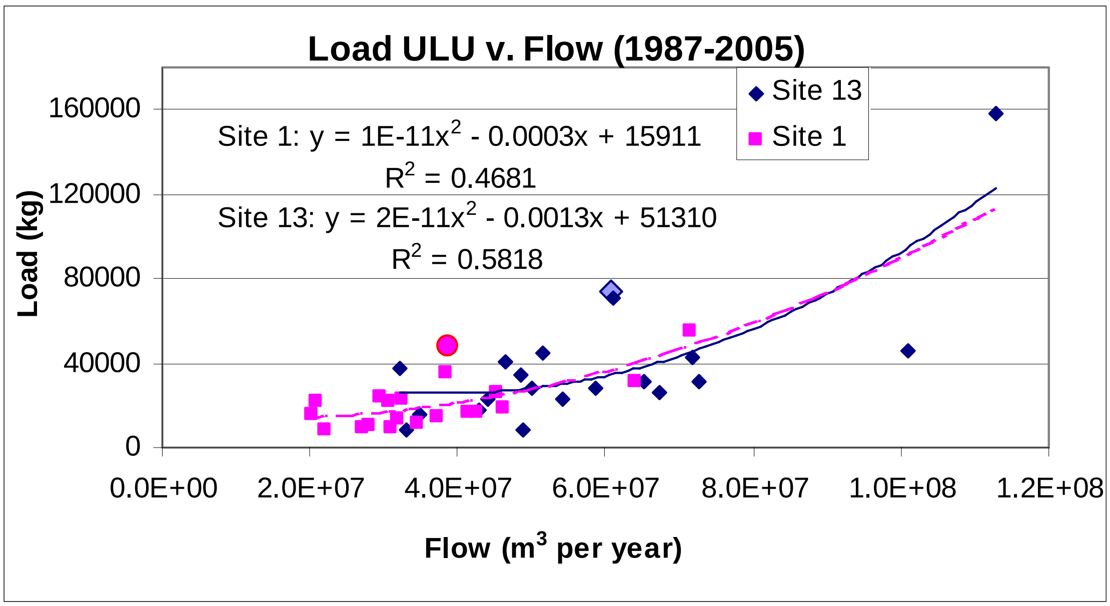


Figure 4.37. Nitrate-N loadings from 1987-2005 from site 1 and site 13 with a polynomial trend line. The trend line represents the increased loading that occurs in years with higher flow, where and x^2 and x are based on the annual flow volume.

The LOADEST model results for both the automatic selection and urban land use approach appeared to perform well. The automatic selection model performed particularly well for shorter model runs (less than 4 years) with those parameters that are most flow driven such as TSS, but also performed well for nutrients.

For on the urban land use user defined model in LOADEST, summary statistics for 1988 to 2005 show that traditional point source loadings from the Noman Cole facility made up a slight majority of nitrate-N loading, primarily due to a recent increase in nitrate-N discharge (Table 4.16). In contrast, point source loadings made up the vast majority of load for ammonia-N over the full study period; however, ammonia-N discharge decreased significantly from the Cole facility starting in 2002 (Table 4.17).

Table 4.16. Summary Statistics - Estimated nitrate-N loads [kg as N/day] for 1988-2005 (ULU forced model and reported Cole discharge data).

	Min.	25th Pct	Med.	75th Pct	90th Pct	95th Pct	99th Pct	Max.
Site 1	0.001	4.948	12.02	30.70	114.84	248.71	909.7	7017.5
Site 13	0.000	10.63	26.35	70.50	247.75	509.98	1686.9	8472.9
Cole	11.43	218.6	260.8	374.6	751.8	1048.0	1311.6	1608.6

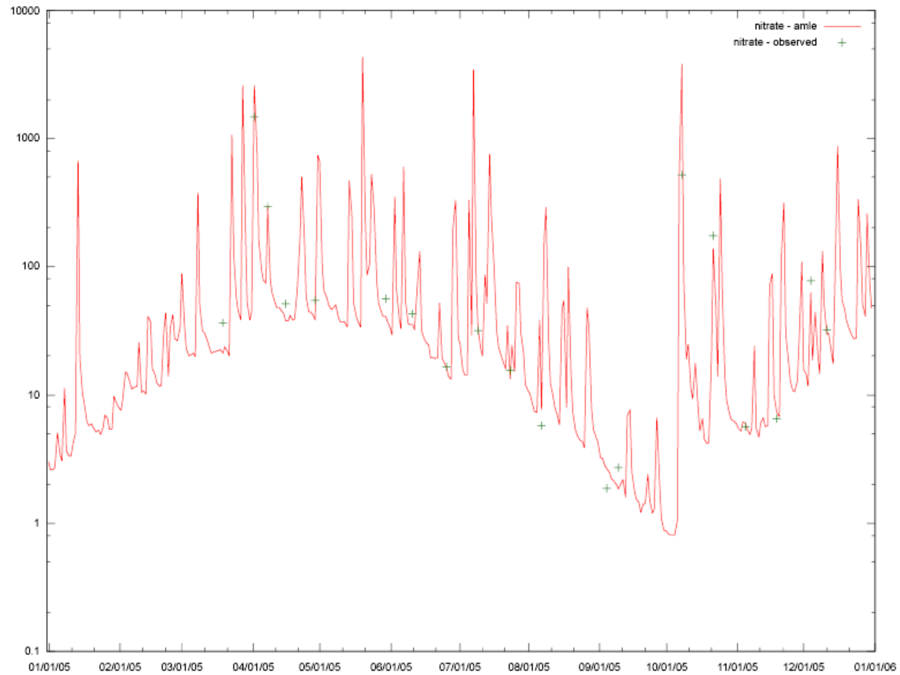
Table 4.17. Summary Statistics - Estimated ammonia-N loads [kg as N/day] (ULU forced model and reported Noman Cole discharge data).

	Min.	25th Pct	Med.	75th Pct	90th Pct	95th Pct	99th Pct	Max.
Site 1	0.000	0.365	1.240	3.617	15.73	44.70	255.2	3432.6
Site 13	0.000	0.995	2.800	7.078	27.33	60.38	218.6	1686.2
Cole	1.53	189.9	1815.5	2108.3	2503.6	2608.0	3151.0	3279.0

Based on the LOADEST results and Cole effluent discharge reports, it appears that nitrate-N loadings have jumped significantly from precipitation driven loadings and increased from the Cole discharge[15], but ammonia-N loadings have stayed relatively constant in the watershed, and decreased dramatically from the Cole discharge. The level of the increase of nitrate-N discharge from the Cole facility is less than the amount of ammonia-N decrease: hence, the facility is discharging considerably less nitrogen than it was previously. Furthermore, this discharge decrease more than offsets the increase in nitrate-N loadings noted from the precipitation driven loadings. Hence, for these individual watersheds, nitrogen pollution has decreased over the study period.

Figures 4.38 through 4.48 give additional details for the output from the land use user selected model (ULU) LOADEST output by graphing the loading of 5 parameters at site 1 and site 13 from 1988 to 2005. The urban land use model was selected for highlighting because it removed the time component dummy and replaced it with an urban land use component. For those years where both the LOADEST default model option was run and the ULU, estimates were generally within 25% of each other at site 1, but could be as far as 100% apart at site 13. Because there is a significant jump in estimated loadings in 2005 for nitrate-N and TSS at site 1 and in nitrate-N at site 13, there was bias in the output of the urban land use model so that each unit of flow contributed less loading in earlier years. In summary, the model predicted higher loadings of these parameters that coincided with increased urban land use.

[15] Nitrate-N discharge from the Cole effluent increased when the facility began treating for ammonia.



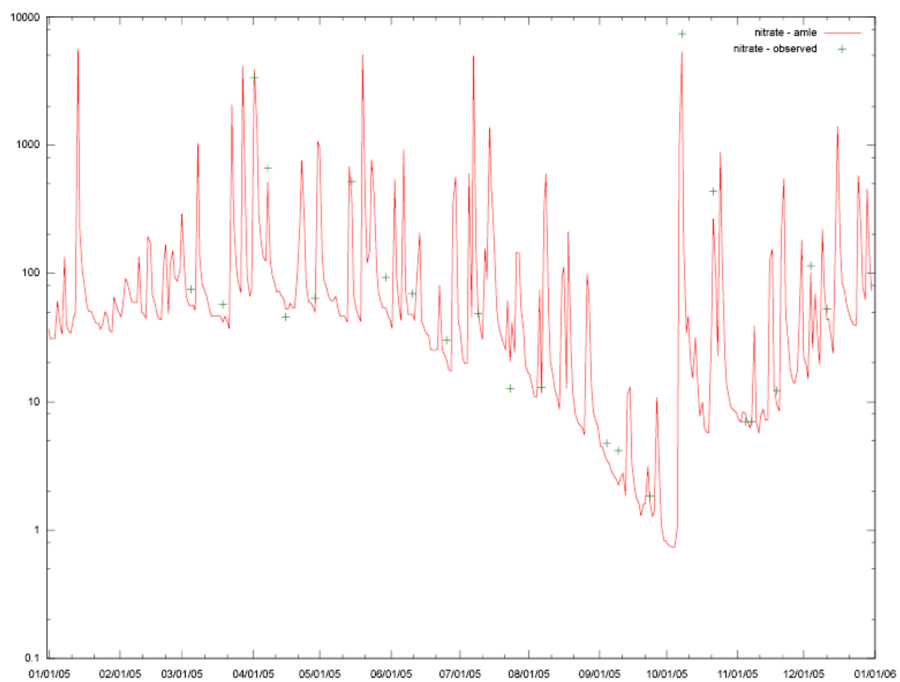


Figure 4.38. Site 20 (top): Nitrate-N loadings (red line) using automatic model selection compared to calculated loads based on daily measured loads (crosses). The automatic calibration appears sound, with an R^2 of 0.9785. For site 1 (bottom), the R^2 was 0.9804 and for site 13 (not shown) the R^2 was 0.9845.

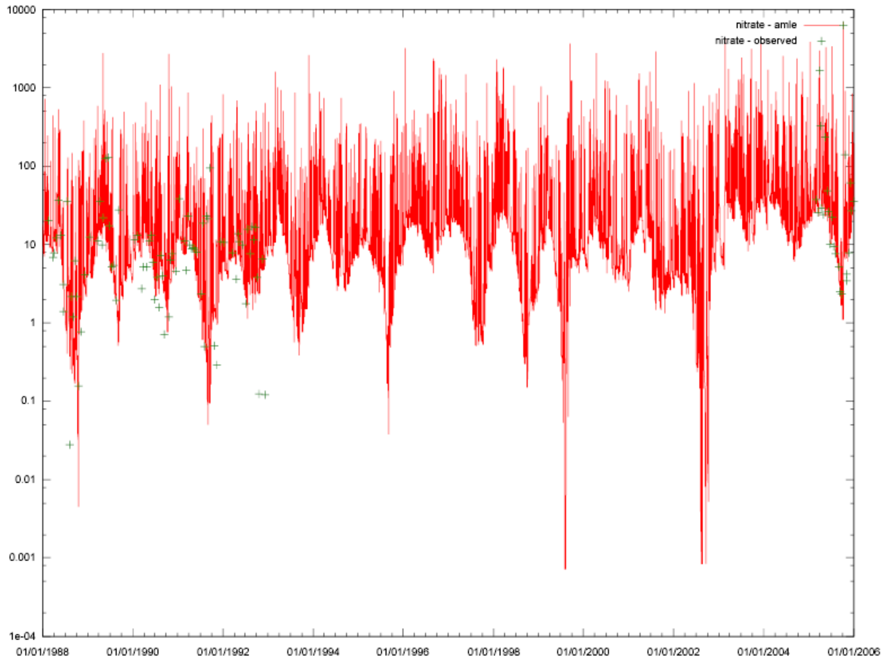


Figure 4.39. Site 1: 1988-2005 Nitrate-N - ULU, Buffer, and BMP estimate. R^2: 0.844

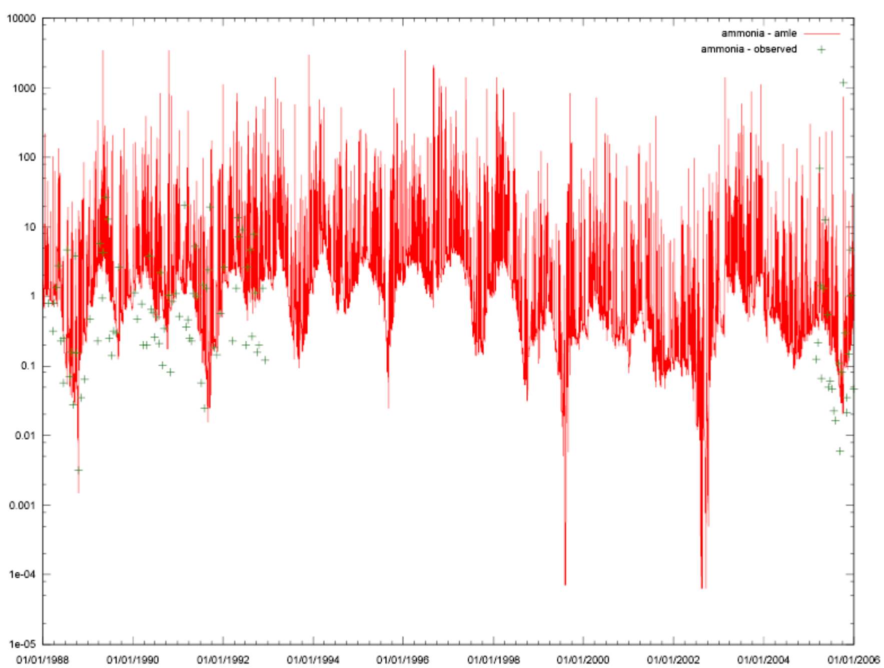


Figure 4.40. Site 1: 1988-2005 Ammonia-N - ULU, Buffer, BMP R^2: 0.6173.

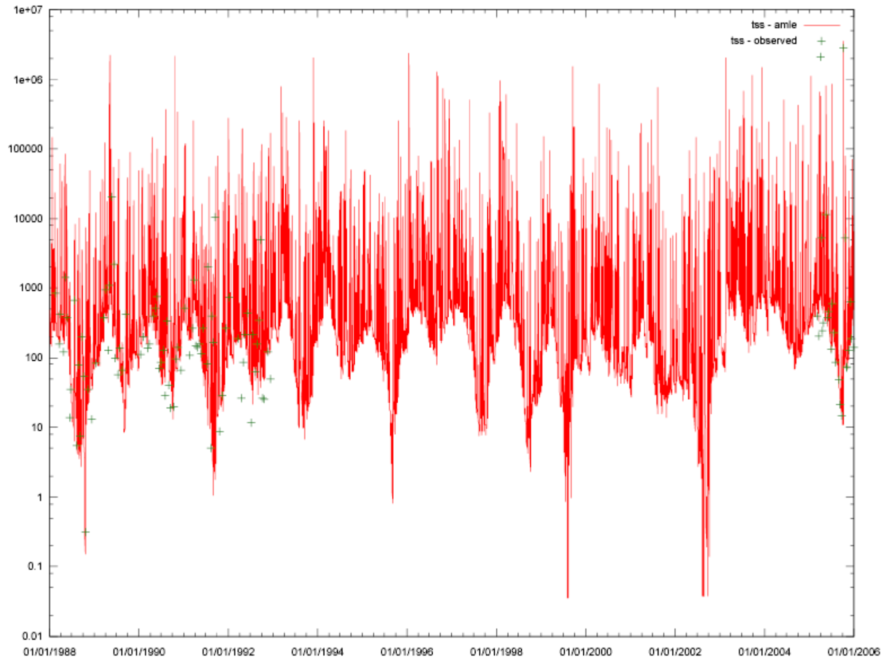


Figure 4.41. Site 1: 1988-2005 TSS - ULU, Buffer, and BMP estimate. R^2: 0.855

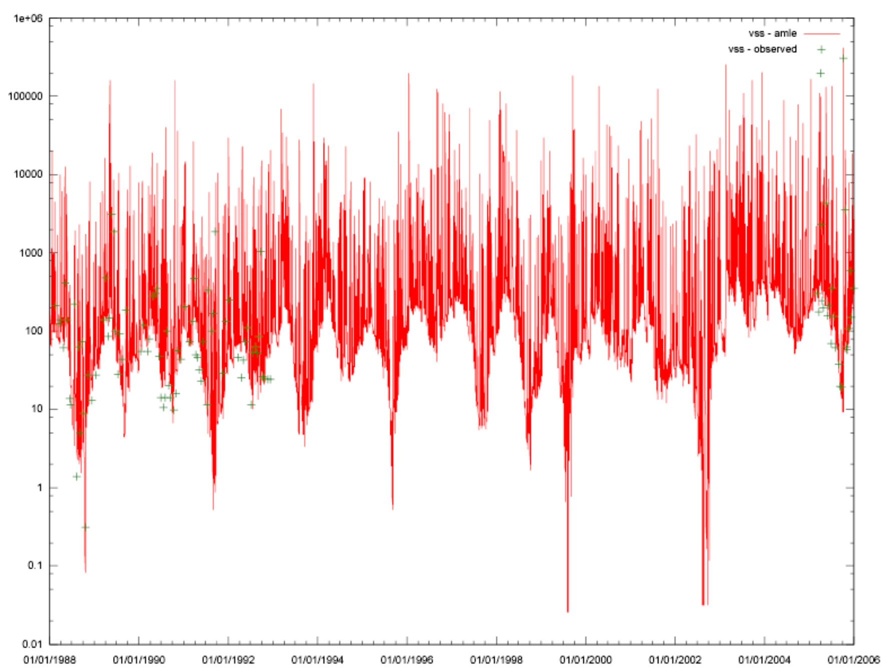


Figure 4.42. Site 1: 1988-2005 VSS - ULU, Buffer, and BMP estimate. R^2: 0.8897

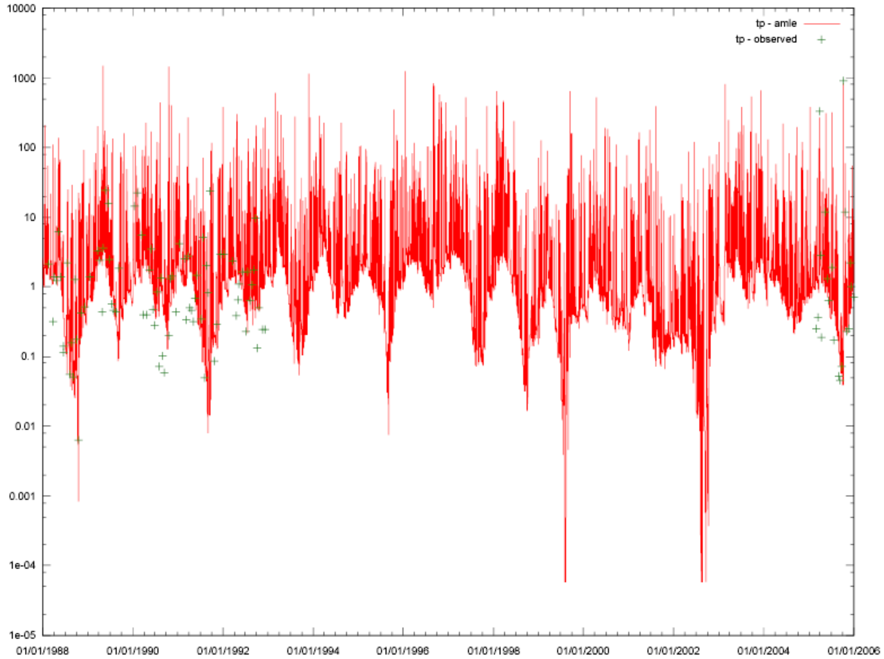


Figure 4.43. Site 1: 1988-2005 Total Phosphorus - ULU, Buffer, and BMP estimate. R^2: 0.7596

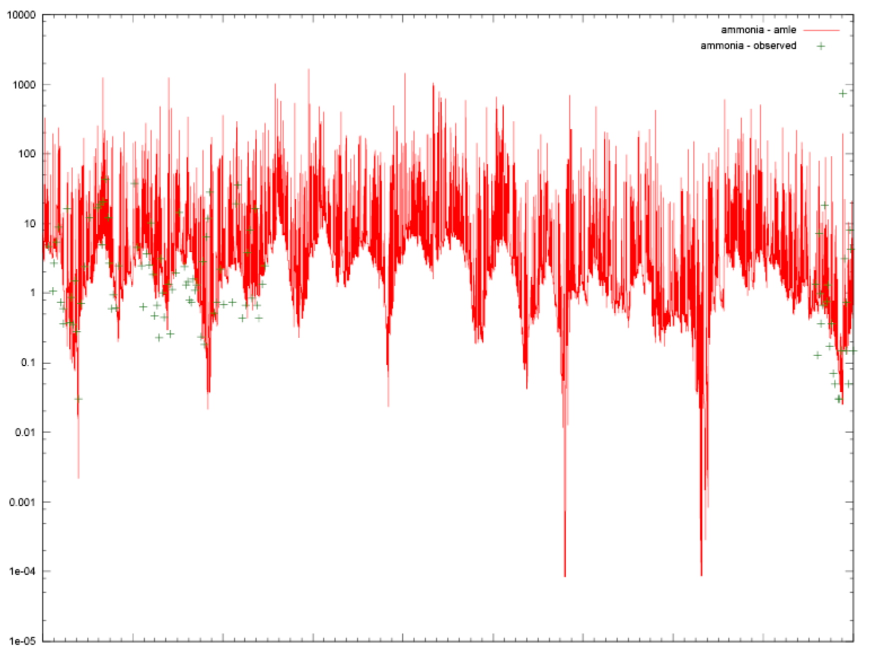


Figure 4.44. Site 13: 1988-2005 Ammonia-N- ULU, Buffer, BMP R^2: 0.6706.

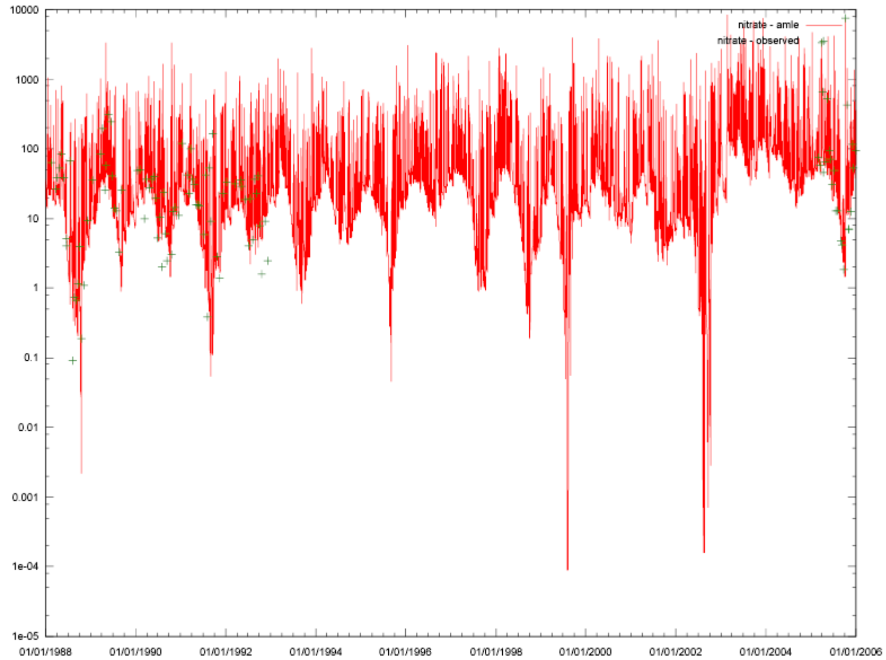


Figure 4.45. Site 13: 1988-2005 Nitrate-N - ULU, Buffer, and BMP estimate. R^2: 0.8277

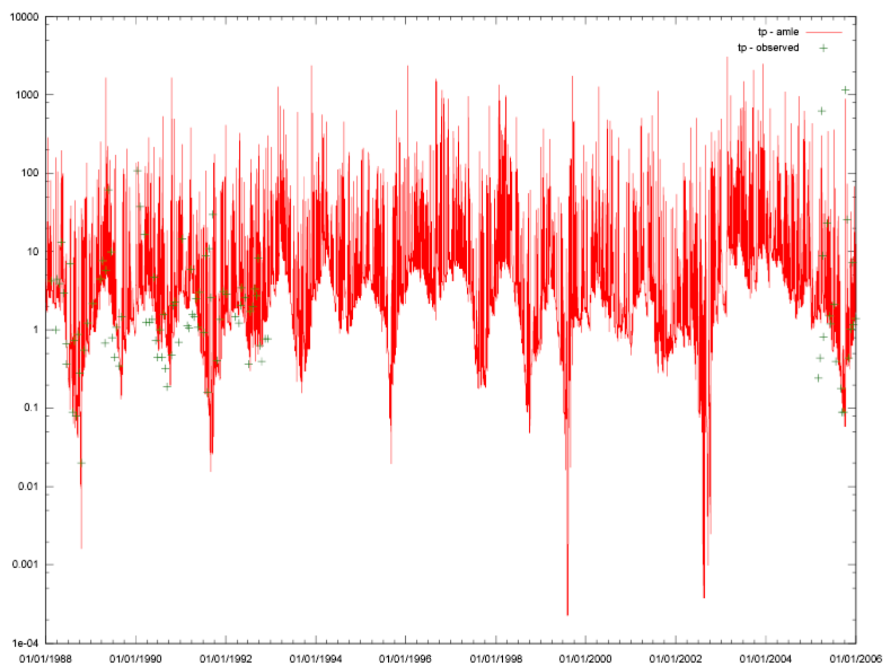


Figure 4.46. Site 13: 1988-2005 Total Phosphorus - ULU, Buffer, and BMP estimate. R^2: 0.7821

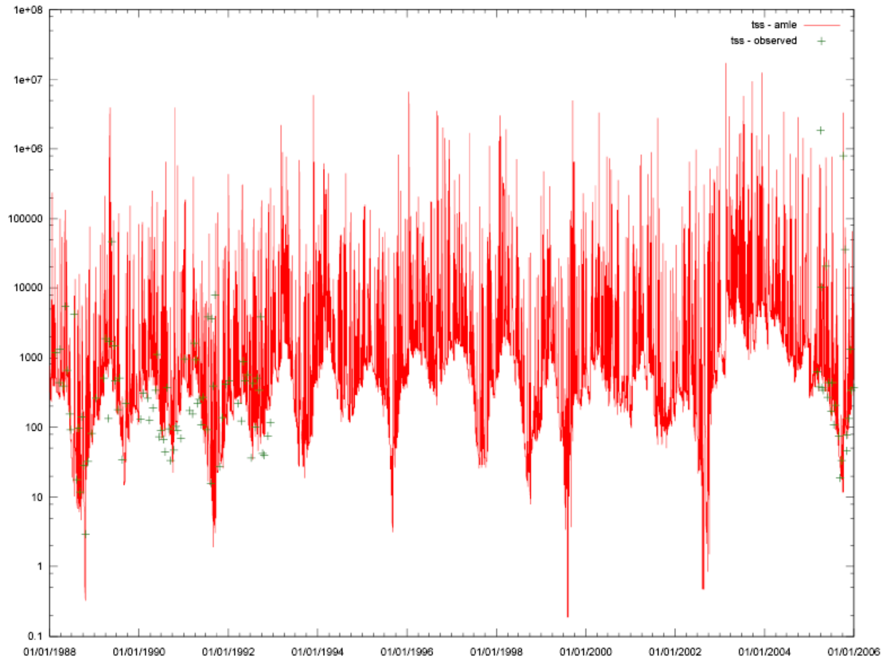


Figure 4.47. Site 13: 1988-2005 TSS - ULU, Buffer, BMP R^2: 0.8535.

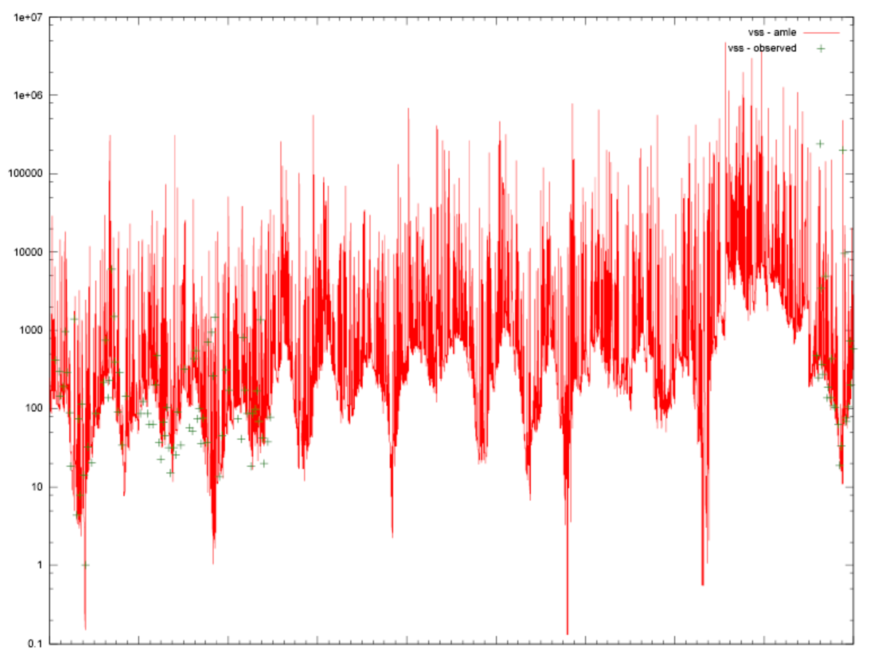


Figure 4.48. Site 13: 1988-2005 VSS- ULU, Buffer, BMP R^2: 0.868.

For the calibrated model output, the R^2 for the measured vs. modeled daily load were particularly high for nitrate, TSS, and VSS (R^2>0.8), particularly when one considers that the model was simulating conditions over a 17 year time period. Total phosphorus and ammonia-N values were not as high for the full time period (R^2 >0.6). The R^2 for ammonia-N is lower due to the episodic nature of ammonia events and the apparent tendency to overpredict ammonia-N loadings in low flow events. Where the automated selection model simulated one year's loading, it calibrated very effectively to the data available. For example, Figure 4.38 shows that in 2005, the single year nitrate-N calibration produced a R^2 values for sites 1, 13, and 20 greater than 0.97.

The user-defined model was validated for nitrate-N where possible with Fairfax County sampling data. Fairfax County Station 17-08 was located near site 1 and samples where taken periodically from 1986 to 2002. These data were compared to the model output for site 1 for 1988 to 2002. Daily load of the County sites were calculated by multiplying the measured concentrations times estimated flow at site 1 for the day. These data were compared to the output from the LOADEST model in SPLUS, where a linear regression of the expected (LOADEST) vs. observed (county data) was calculated and summary statistics generated. Results were positive: for all 308 days sampled during the 1988 to 2005 time period, summed daily loading for all sample days was estimated at 6778 kg for the observed county data versus 6774 kg for the modeled results, a difference of 0.06%. Furthermore, the predicted vs. observed values correlated well with a R^2 of 0.7963 and a p value of <0.0001 (Figure 4.49).

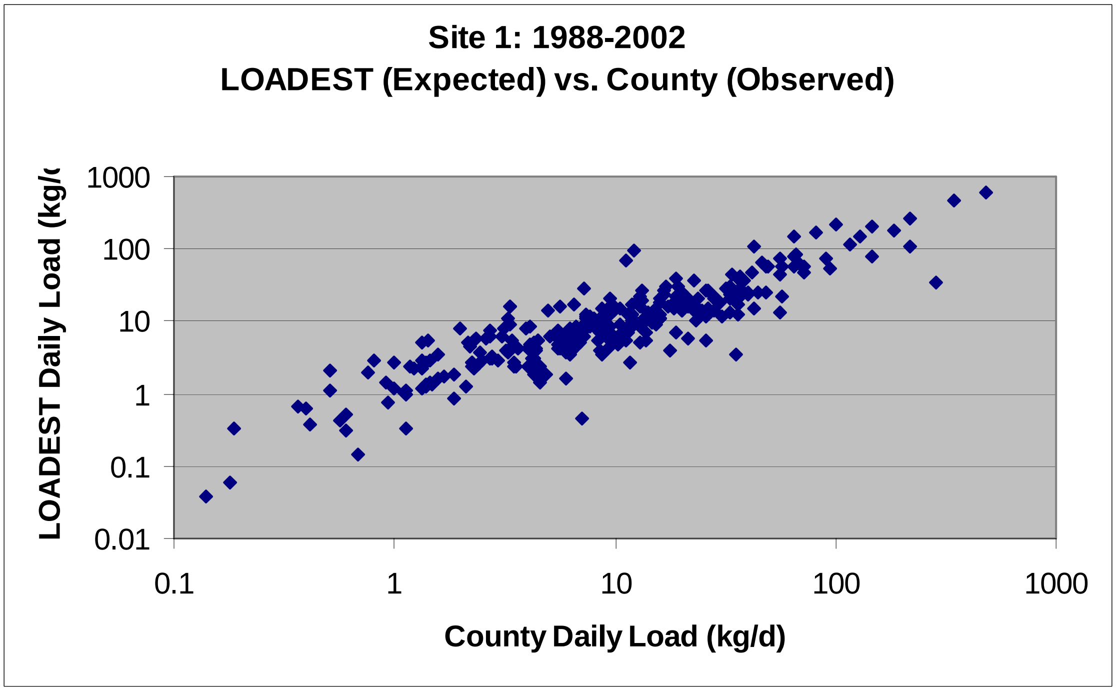


Figure 4.49. Results of the validation of LOADEST ULU model for Pohick 1 using county sampling data from 1988-2002: R^2 = .7963; p<0.0001. The plot contains expected (LOADEST ULU) vs. observed (County Data) values.

Similar validation was completed using the county data from county site 16-28, which was located close to GMU site 13 on Accotink. The data set were only available for 2000-2002: the 41 measured samples compared reasonably well to the LOADEST output with an R^2 of the daily load calculated at .7971 with a p value of less than 0.0001 (Figure 4.50); however, the model overpredicted total loadings significantly (2995 kg/(41 days) compared to 1255 kg/(41 days)), primarily due to four wet weather days. Without these wet weather days, the model's overprediction is significant, although not as dramatic (1023 kg/(41 days) compared to 623 kg/(41 days)). There are two possible explanations: first, this likely indicates that the model is overpredicting nitrate-N loading for the latter years of the calibration (1998-2004), with the exception of 2005, because of the impact of the 2005 data. Second, it is possible that the nitrate-N collections were made before or after the bulk of a storm event. The overprediction is the most likely explanation considering the more accurate predictions at the Pohick site.

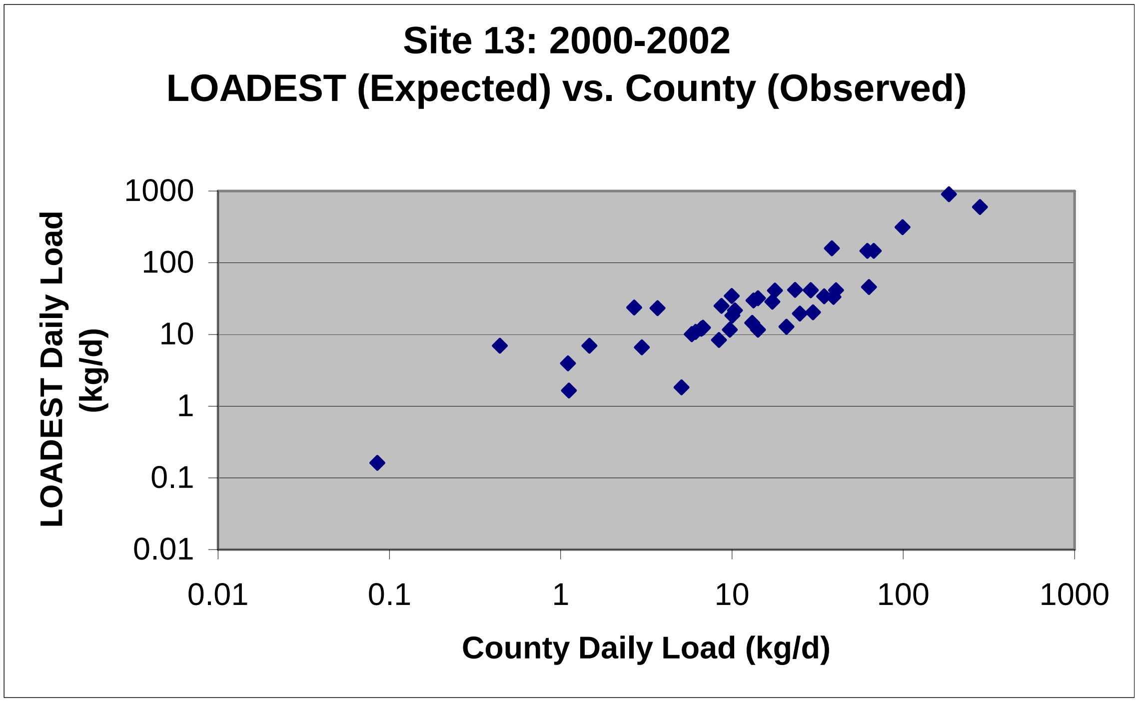


Figure 4.50. Results of the validation of LOADEST ULU model for Accotink 13 using county sampling data from 2000 to 2002: R^2 = .7971; p<0.0001. The plot contains expected (LOADEST ULU) vs. observed (County Data) values.

Each of the four methods used to compute loadings (Linear Regression, Load Duration Regression Line, LOADEST automatic calibration, and the LOADEST user defined model), each have their strengths. The linear regression model is the simplest to set up and execute. The user can run regressions using basic statistical software or Microsoft Excel for either one-year terms or multi-year time periods and use the regression formula to calculate annual loading. However, this is a highly simplified model and does not take into account other drivers for loadings. Furthermore, if there is one concentration that is either very high or low at a high flow level, it can have undue influence on the slope of the regression. Hence, this method should only be used if there are a significant number of data points and they are spread over the entire range of interest. For parameters such as nitrate, this approach seemed to work reasonably well. For parameters such as TSS, it did not work as well, perhaps because of TSS concentrations increase with flow more strongly than nitrate.

Use of the load regression line worked surprisingly well, if there were data points available for both high flow and low flow scenarios. The approach predicted annual loadings that were generally within reasonable ranges of other loading estimates. For those years in which there were no high flow events, this approach worked least well and most underestimated loading. Where there were data points that were representative of the full flow situation (such as 2006), this approach gave a loading estimate that was very close to the other loading estimates. This model has the benefit over the linear regression method in that a single outlier point at the extreme of the flow spectrum will influence the load estimation for that flow level, but will not skew other flow level concentrations. If one were to use this approach, years should be pooled to maximize the number of data points available, provided physical conditions such as land use do not change substantially between years. However, unless an automated technique is developed, just as much effort goes into creation of these loading estimates as the other loading estimates generated by the LOADEST approaches. As such, the primary benefit offered, making use of existing analysis to calculate loadings without significant extra work, is only partially realized. Additionally, because the approach does not accurately simulate watershed conditions on a daily basis (and thus accurately simulate daily loading), its potential applications are somewhat limited in the regulatory role for looking at impacts of a given parameter on an individual water body.

The two LOADEST approaches both appeared to increase the R^2 of the daily estimator, and consequently, were used as the primary basis for computing load. However, these results must be visually inspected. Validation of these results is difficult, but not impossible. For the default model, one would need to reserve some of the data from the middle of the iteration run. For a user-defined model that does not incorporate a time component, the validation can occur with data on either end.

With the LOADEST combined with urban land use results, automatic calibration of the multiple regression model resulted in a high R^2 and gave plausible results (Figures 4.38 through 4.48). However, the coefficients for the variables in the model were sometimes contrary. For sites 1, 13, and 20, the model had five variable inputs: the natural log of flow (lnQ) and the natural log of flow squared (lnQ^2), urban land use percent in the watershed, percent urban intrusion into the buffer zone (defined as 30 meters from the closest water edge, and a BMP dummy variable (as previously mentioned, we attempted to input seasonality, but this approach failed due to model limitations). As suspected, the three latter variables had notable multicollinearity, which is a result of how the estimates were derived and the fact that urbanization is the primary driver of all three variables. Furthermore, during data conversion, the urban land use percent figures were rounded to single percentages, which removed some of the precision of the estimate. Hence, the coefficients on the variables were determined to not accurately represent the driving processes of those particular variables, but the model as a whole seemed to represent the system processes well. The reason this is likely the case is that the BMP dummy variable did not start being a nonzero value until the mid 1980s, at which point it began increasing very gradually[16]. There was a considerable jump in loadings between 1992 and 2005, which the model explained by assigning loading to the BMP dummy variable. In effect, this produced two 'urban land use' drivers, one for earlier development and one for later development. Increases in loadings were best predicted by using the later urban development variable. In short, this calibration was not ideal, but it does appear to give accurate loadings, and in most cases, gave R^2 values that were higher than the default model runs for iterations greater than 4 years.

4.4 General Discussion

Accotink and Pohick are streams clearly impacted by urbanization and urban land use. Both streams have high concentrations of TSS, have highly impacted benthic communities (Fairfax County, 2001), have impacted hydrologic regimes, exhibit strong signs of stream scouring, and have elevated nitrate-N and phosphorous loadings from what would be expected from a forested watershed. Between 1975 and 2004, estimated urban land use increased from 11.1% to 48.6% in the Pohick watershed and from 36.3% to 61.9% in the Accotink watershed. Results show increased flow with increased

[16] The BMP variable was originally 10% of all new urban land in the mid 1980s, 50% from 1992 to 1998, and 80% from 1995 to the present.

urbanization; nitrate-N is somewhat more ambiguous, and total suspended solids have surprising results, yet are consistent with the notion of urbanization having immediate and profound impacts on a stream.

The hypothesis of this chapter was that there is distinct spatial variation in water quality parameters due primarily to differences in levels of urban land use. This hypothesis was supported for nitrate, and partially supported for total phosphorus. However, the hypothesis is contradicted by results with total suspended solids, perhaps due to the age of development or disproportionate levels of construction activity. I also expected to see changes in flow, nitrate, and TSS discharges as the watersheds urbanized. Changes in flow and nitrate-N were observable and correlated with increased urban area. The following discussion will focus on the four most significant issues from the results section: the implications of high conductivity; nitrogen loading and delivery to the cove and subsequently the bay; flow to the streams themselves in the context of TSS and changes in hydrology; and further limitations and shortcomings of the various approaches used here.

4.4.1 Conductivity

High levels of chloride, which directly correlate with conductivity, can be detrimental to freshwater aquatic biota. The EPA has established a freshwater Criteria Maximum Concentration (CMC) acute toxicity level of 860 mg/L, or about .86 ppm, and a Criteria Continuous Concentration (CCC) chronic toxicity level of 230 mg/L, or .23 ppm. Environment Canada (2000) found that high levels of salinity stressed periphyton communities. Conductivity concentrations of sodium and chloride have been shown to increase as a result of deicers in soils that border roads (Backstrom *et al.*, 2004). Other issues, such as increased mobilization and concentration of zinc, cadmium, and copper have been noted in association with high conductivity. As can be seen in Figure 4.51, portions of Accotink Creek very closely border interstates and commercial areas, many of which are directly drained into the stream.

With only two observed exceedences in 2005, it is difficult to make broad generalizations about the exact impact of road salts on water quality and the biota of Accotink. However, based on these results, this issue deserves further exploration. First, it should be assumed that there is an exceedence any time there is a significant snow event unless shown otherwise. This evidence is supported by exceedences noted in both 2006 in Long Branch and in 2007 at site 20 and at Accotink at Little River Turnpike. In colder winters, Accotink could be suffering from elevated conductivity on multiple occasions for an extended period. In terms of exploration, it would be interesting to note how long the salt ions take to flush from the system, whether these high concentrations are a chronic problem in addition to an acute problem, and whether the high conductivity plays a significant role in benthic impairment of the creek itself. It is plausible to suggest that one or more exceedences could result in severe depauperization of the benthic fauna on an annual basis which could not be easily recovered from the following spring.

If it is found that the stream is, in fact, suffering from impairment due to road salt application, the county and VDOT might want to consider the application of other materials such as Calcium Magnesium Acetate (CMA) or Potassium Acetate (KA) for major arteries during some or all snow storms. Though more expensive, a National

Research Council Report (1991) found that CMA is an acceptable alternative to sodium chloride solutions and that it will not have the same negative environmental impacts to stream ecosystems. Though road salt application is less frequent in Northern Virginia than in more northern climates, it is a potential environmental problem to receiving waters in select watersheds as evidenced by the two high and potentially lethal concentrations in Accotink. Furthermore, it might only take one high dose of conductivity to cause a reduction in benthic populations.

Figure 4.51. Accotink Creek near the Little River Turnpike interchange, looking at a culvert crossing under Interstate I-495.

4.4.2 Receiving Body Nitrogen Loadings

Correll (1987) estimates that as of 1987, 'land discharges' accounted for approximately 85,800 metric tonnes (1 metric tonne = 1000 kg) of loading of total N on an average year to the Chesapeake Bay. With a watershed size of approximately 106,841 km^2, this is an average of 803 kg/km^2. From 1986 to 1989, the Accotink and Pohick watersheds upstream of their sampling points (most of the watersheds) are estimated to have contributed a combined annual average load between 29.0 metric tonnes (LOADEST ULU) and 47.0 metric tonnes (LOADEST default) of nitrate-N per year, while the Cole plant was discharging an annual average of 89.4 tonnes of nitrate-N per year. During the same time period, Pohick and Accotink discharged an estimated average of between 9.9 (LOADEST ULU) to 36.6 (LOADEST default) metric tonnes of ammonia-N and the Cole plant discharged an annual average of 749.5 tonnes of ammonia nitrogen. Since the Accotink and Pohick watersheds are approximately 227.1 km^2, average nitrate-N loading between 1986 and 1989 is estimated at approximately 127.8 and 207.0 kg/km^2 and average ammonia-N loading is estimated to be between 43.6 and 161.2 kg/km^2. These two sources, which do not include organic nitrogen, combined for a total of between 171.4 and 368.2 kg/km^2. CH2M Hill (2000; US EPA, 2001b) found that event mean concentrations (average concentration in runoff during a precipitation event) of nitrate-N were between 29-36% of Total Nitrogen concentrations and those of ammonia-N were between 8-18% of TN. Considering that nitrate-N and ammonia-N are more bioavailable than organic nitrogen, it is reasonable to assume that they are taken up or transformed at least as quickly, if not quicker, than organic nitrogen. Hence, it is safe to assume that nitrate-N and ammonia-N nitrogen likely make up between 37% and 54% or less of total nitrogen. Hence, average annual total nitrogen loading was likely between 317.4 and 995.1 kg/km^2 for 1986-1989.

During 2005, the estimated load of nitrogen from the watersheds was 118.5 to 135.1 metric tonnes of nitrate-N and 3.6 to 5.6 metric tonnes of ammonia-N, with total N loading certainly above this level due to organic nitrogen contribution (unmeasured for this study). Average nitrate-N loading was between approximately 521.8 and 594.9 kg/km^2 for nitrate-N and 15.6 to 24.7 kg/km^2 for ammonia. The sources combine for a total loading of between 527.4 and 622.6 kg/km^2 of nitrogen, once again not including organic nitrogen components. Using the same ratios as those above, total nitrogen contribution is estimated to be approximately between 976.7 and 1682.7 kg/km^2 for the year.

In 2005, loadings of ammonia-N from the Noman Cole treatment facility decreased dramatically to 6 metric tonnes, but nitrate-N increased to 218.1 metric tonnes (Figure 4.52). Jones and Kelso (2005b) documented a decline in nitrate-N concentrations from an average of about 1.5 mg/L in 1983 to 0.6 mg/L in 2003 for Gunston Cove, the receiving waters for Accotink and Pohick[17]. They also noted a decrease in average ammonia-N concentrations from a peak of 0.3 mg/L in 1989 to 0.05 mg/L in 2003. Efforts at controlling point source pollution from the Noman Cole plant were successful in reducing the quantity of nitrogen being discharged to the cove, but some of these gains

[17] The data collected for the Gunston Cove report by Jones and Kelso is from the same project used in this study, but with different sites. The 1984 data for Gunston Cove also show the unexplained high values of nitrate-N concentration in Gunston Cove. Either there was some large scale, unexplained ecological process occurring during this year, or there could have been issues with laboratory testing.

were offset by increases in nitrogen loadings from urban stormwater sources. Even with the inclusion of the Noman Cole point source, nitrogen loading is decreasing in these individual watersheds. However, most small watersheds do not have traditional point sources from which to offset their increasing urban discharge. This means that as they urbanize, these watersheds are increasing nitrogen loading from stormwater sources without having offsetting decreases in loading from a POTW. Considering that there are numerous such watersheds in Northern Virginia alone, the increases in nitrogen loading from each watershed may result in a large cumulative increase. Hence, the results in Accotink and Pohick illustrate that increases in nitrogen loading as a result of increasing urbanization could likely result in substantial cumulative increase in nitrogen loadings to the Chesapeake Bay, provided that significant quantities of this nitrogen are not volatilized or otherwise removed in freshwater streams and rivers draining to the Bay.

Jaworski *et al.* (1992) used water quality and flow data from a monitoring station for the Potomac River located at Chain Bridge, the area immediately at the fall line between Washington, DC and Virginia, to calculate the nutrient mass balance for the time period from 1983 to 1986 for the watershed immediately upstream of Washington DC. The authors found that the river exported only 17% of total nitrogen that was input into the system via watershed loading and atmospheric deposition at the study area's mouth (at Chain Bridge). Furthermore, the authors found that over 66% of nitrogen was retained by the watershed or lost by volatilization, denitrification, or other methods. Additionally, nutrients are transformed once they are part of streams: for instance phytoplankton take up and dentrify nitrogen and phosphorus can settle in the sediment.

Furthermore, Boynton et al. (2002) found that nutrient losses were lower in the low salinity reaches of the Chesapeake Bay such as the Potomac than in the mesohaline reaches, and that 13% of total nitrogen was lost in the Potomac River. Furthermore, they found that 40% of the TN input into the Potomac system was exported to the Chesapeake.

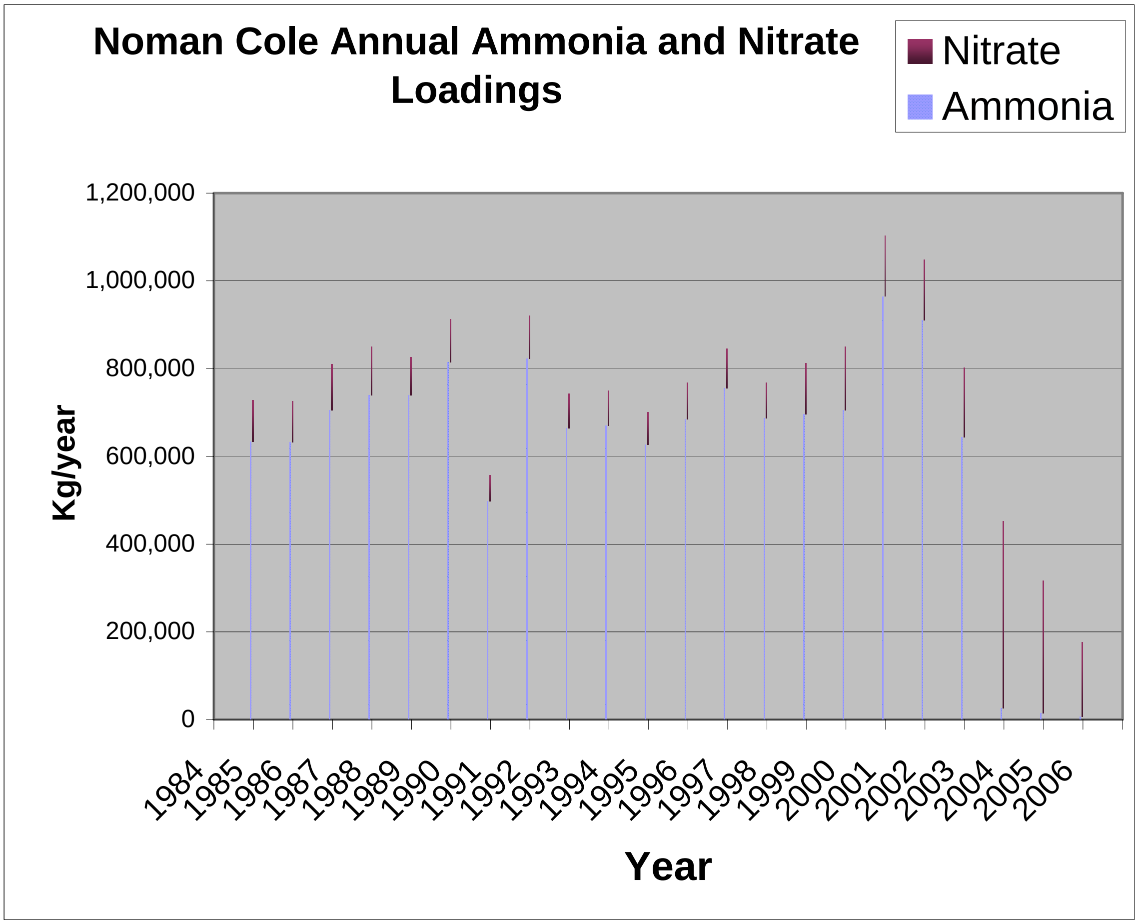


Figure 4.52. Annual loadings of ammonia and nitrate nitrogen from the Cole's facility. The vertical columns represent total annual nitrate-N (red solid) and ammonia-N (blue patterned) loadings.

Therefore, all nitrogen being discharged from Accotink and Pohick to Gunston Cove does not ultimately reach the Chesapeake Bay. After being discharged into Gunston Cove, some of the nitrogen from Accotink and Pohick is used by phytoplankton, volatilized, harvested, or stored. Nitrogen is not the limiting nutrient most times of the year in Gunston Cove; hence any excess nitrogen that is not lost via geochemical processes is exported to the Bay. Hence, the cove and the upstream freshwater Potomac have an assimilation capacity whereby they remove some of the nitrogen added to the aquatic system, but a significant portion is exported to the Potomac and consequently the Bay. Nonetheless, provided the Noman Cole facility lowers its nitrogen discharge as required by its next NPDES permit to 3.0 mg/L and the county works to maintain and restore watershed hydrology and maintain current levels of nitrogen discharge, these watersheds may be able to meet their Chesapeake Bay agreement obligations since Gunston Cove will serve to remove some nitrogen loading.

Accotink and Pohick are illustrative of the problems and opportunities faced by Fairfax County and other jurisdictions in controlling nitrogen pollution. On one hand, progress is being made in controlling traditional point source dischargers. Atmospheric deposition in the mid-Atlantic appears to be on a downward trend, and if current policies and market forces pushing cleaner technologies continue, atmospheric deposition of nitrogen should remain stable or continue declining. Streams and rivers have significant assimilative capacity of nitrogen, protection of riparian buffers in the form of resource protection areas has been successful, and the county has an aggressive stormwater program to attempt to mitigate the impact of continued urbanization. These issues are likely mitigating the loading of nitrogen somewhat. For instance, Schnabel (1986) noted nitrate-N was reduced by about 50 percent when groundwater percolated through a riparian vegetated buffer. Accotink still has statistically significantly higher concentrations of nitrate-N in its discharge than Pohick and the upper reaches contribute higher loads of nitrate-N per km^2 of the watershed[18]. Perhaps this is indicative that, though Pohick's rate of urbanization is approaching that of Accotink, conservation measures and/or BMPs are having a positive effect.

On the other hand, urban stormwater sources are numerous and their impact is expanding. Though the net nitrogen discharge from Accotink and Pohick is decreasing because of reductions at the Noman Cole plant, most small watersheds do not have such significant traditional point source reductions. In other words, increases in discharge from stormwater sources are not being offset in these watersheds. As noted in numerous peer-reviewed publications, the Environmental Protection Agency, and the Chesapeake Bay Program, the cumulative impact of increased nitrogen loading from urban sources is significant (Booth *et al.*, 2002; Carle *et al.*, 2005; Schueler, 1994; Tang *et al.*, 2005; US EPA, 1983, 2002b).

Therefore, increases in nitrogen loading are a real issue for the health of the bay. Policy makers in the county are allocating vast sums of county and private resources into strategies that minimize nitrogen pollution. Based on evidence of substantial nitrogen reductions from the Cole plant, however, the reduction of nitrogen pollution should be

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[18] There are some confounding issues with looking at total loadings in the watersheds relative to each other: the primary being that the flood samples were taken in the flood plain for all sites but Pohick 1 which makes Accotink's loading estimates lower than they would have been if the sample were taken from the main channel.

one of many goals of the stormwater program, but in my opinion, it should not be a primary goal. It makes sense to design traditional point source pollutant reduction programs around a single parameter: however, such an approach is not necessarily appropriate for precipitation driven pollution. Protecting riparian buffers accomplishes reductions of nitrogen while serving other environmental functions. The stormwater program should also take such an approach, focusing on solutions that serve multiple environmental benefits. As far as the health of the Accotink and Pohick streams themselves, nitrogen is no longer one of the most significant threats. The current primary threats are changes to the hydrologic regime and sedimentation, and any stormwater strategy in the county must address these issues while providing multiple services on limited land with limited resources, all while attempting to mitigate other pollutant loadings, including nitrogen.

4.4.3 Flow driven impacts

As evidenced by the analysis of flow data from the USGS gauge station in Accotink at the start of this chapter and the modeling of hydrology in Chapter 3 for the full watershed, significant changes in flow patterns and total volume impact Accotink and Pohick. This study has also shown that TSS concentrations are significant in both Accotink and Pohick. Changes in hydrology are playing a substantial role in affecting TSS concentrations, sediment deposition and stream scouring, and these changes result in stress to aquatic organisms from progressively lower groundwater flow and increasing power and frequency of flood events. Based on the results of this study, increased emphasis should be placed on groundwater recharge and redirecting water from the surface water hydrologic cycle for both watersheds.

One of the most significant causes of water quality impairment in the United States is the loading of sediments to aquatic ecosystems (United States Environmental Protection Agency, 2003). Construction and other land disturbances increase soil erosion and alter rates of sediment transport in aquatic water bodies, and are therefore the most notable drivers of sediment impairments in aquatic systems (Nietch et al., 2005; Waters, 1995). Sediment impacts both biological habitat and physical characteristics of streams, and, in Accotink and Pohick, is likely one of the primary drivers of the benthic impairments noted on the 303(d) list. The EPA estimates that approximately 40% of all assessed rivers and streams in the US were impaired due to excessive sediment loadings and deposition.

Because these increases in sediment are likely driven by both changes in hydrology and loadings from urban land use (as discussed in the TSS section of the results), it is critically important to utilize those techniques that best mitigate the impacts of watershed urbanization for reduction or moderation of volume and filtering of pollutants. Dry detention basins, for example, are appropriate in some situations but are not the one-size-fits-all solution which they are sometimes used in the watersheds. In addition to not providing other ecosystem or recreational services, they do little to allow groundwater recharge or mitigate total flow volume. It is these hydrologic drivers that are posing the greatest threat to these water bodies. Nitrogen and other pollutants do threaten downstream receiving waters, but the county should be focusing its management strategies on mitigating impacts to its streams as well as downstream receiving bodies. It is cost prohibitive to fully retrofit most stormwater infrastructure. The county should focus on requiring a reduction from its current redevelopment opportunities and requiring stricter controls on new development, thereby allowing no new net increase in storm peak flow and no decrease in groundwater recharge. Technologies such as green roofs, bioretention, and porous pavement would allow gradual transformation of the county's hydrology so that the watershed geomorphology might eventually stabilize, and continued ecological and economic impacts can be mitigated. Approaches that reduce total runoff volume should also reduce the nitrogen contribution from the county's watersheds to downstream receiving waters, freeing county stormwater managers and planners from having to choose between protecting the bay or the county's streams.

In this same vein, there should be increased enforcement of construction sites and other land disturbing activities. Though it was beyond the scope of this study to examine construction sites in detail, multiple construction sites without adequate BMPs were noted in the Pohick and Accotink watersheds, and substantial sediment laden runoff was noted. Sediment tracked onto roadways was common at many sites, despite the initial construction of gravel entrances to minimize such sediment tracking. These lapses are due to inadequate maintenance of sediment control structures and practices. As can be seen in Figure 4.53, construction runoff diluted by runoff from the streets was entering the storm sewer with concentrations of TSS greater than 4000 mg/L in this October 2005 storm. Figure 4.54 shows that sediment runoff was substantial from the construction site.

Though no official load was calculated, back of the envelope estimations put the volume of sediment lost from this construction site alone to be in the hundreds to the thousands of kilograms for this October storm. (160,000 liters of runoff per acre, 4,000-10,000 mg TSS/L, 2-5 acre site).

Figure 4.53. Sediment runoff from a construction site without appropriate BMPS in Lorton, Virginia, the southern portion of the Pohick watershed.

Figure 4.54. A cleared construction site with no BMPs in the Pohick Watershed. This site allows sediment to completely cover the sidewalk and run into the street, where it quickly is carried into a nearby storm drain and then Pohick Creek.

Changes in the flow pattern are causing significant ecological and economic damage to Fairfax County resources and those of its citizens. In a June 25th and 26th, 2006 storm, a portion of the Huntington subdivision in south Fairfax flooded causing over $10 million in damages (Turque, 2007). The flooding was caused by increased sedimentation in the channel and a high peak flow caused by the highly urbanized upstream watershed of Cameron Run. The same storm extensively damaged the Accotink Creek stream valley trail. The stream scoured the side of the trail and tore down railings and fences from bridges on the newly constructed portion of trail, causing over $1 million worth of damage. These are very real economic problems that will continue to magnify if source control is not more aggressively pursued.

Continued analysis and quantification of TSS in these watersheds is warranted. However, it is quite clear that these ecosystems are impaired from substrate embeddedness and symptoms common to almost all urban streams. On one hand, the county has a very smart philosophy laid out in its stream protection strategy (Fairfax County, 2001) to keep Accotink and Pohick from facing further degradation. However, based on the results of this study, I would argue that the approaches currently being implemented are not succeeding in preventing further degradation. The county must be more aggressive in the policies it adopts if it truly wants to meet these goals while being able to meet its Chesapeake Bay obligations.

4.4.4 Successful approaches, limitations and shortcomings

One of the original goals of this dissertation was to find the simplest ways to derive accurate information about streams that were practical to inform watershed mangers' decisions. This chapter described multiple approaches that were used to derive information: monitoring, searching other sources of data, statistical analyses, calculating loadings via a readily available multiple regression program (LOADEST), linear regressions, and using regression lines that can be easily generated in the process of making load duration curves. The primary benefit of the information in this chapter was compilation of a comprehensive analysis of water quality dynamics and related issues for the two watersheds. The results generated will be helpful in validating the simple approaches used in chapter 5 discussing nutrient loading and future land use scenarios. However, with the exception of the linear regression load estimations and the statistical analyses, these approaches were not simple and required time and specialized knowledge.

Linear regressions appeared accurate as a simple approach to calculate loads, provided there were samples from a variety of flow conditions. In terms of calculating loadings, linear regressions generated the highest R^2 for nitrate. For TSS, calculating load worked well with a linear regression if plotted against the log of flow. LOADEST, both in the automated form and the user-generated model, performed nicely and appeared to give more reasonable estimates even for years when wet weather samples were lacking. However, there were some major complications with this approach. First, the interface for LOADEST was not easy to use and setting up each data file for running in the model was time consuming. The LOADEST model is extremely functional and could generate quick loading estimates for users if it were more user friendly. However, based upon the dated user interface and the relatively high learning curve to use and operate the model, the model would not be a good tool for an untrained modeler or computer programmer. Furthermore, the model was limited to 6 explanatory variables. Though I tried to manipulate the code to allow for additional variables, the recompiled model did not work. Additionally, rewriting of code in Fortran was well beyond the scope of this study. Additional weaknesses included not being able to transform variables by other variables within the model: for instance, one could not multiply season times flow in the model: it had to be done in a previous conversion. Hence, being constrained to only 6 explanatory variables, creation of user-defined models provided limited utility, even if users could identify multiple drivers or needed to transform a variable into multiple functions (i.e. flow and flow squared). Despite multiple attempts, a user defined model could not be successfully created that used a dummy variable for each season, dummy variables for season times flow, and urban land use and riparian buffers, and a BMP dummy variable. Hence, the abilities of the model to perform were somewhat limited by the hard-wired capabilities of the system. This being said, if the creators of the model were to add a very simple user interface, this model could be extremely useful for the watershed community as a tool for calculating accurate loadings with very little training, provided the users understood basic statistics and watershed dynamics. Furthermore, with a few changes in the coding, the model could be significantly more flexible and, as a result, have enhanced performance.

If this dissertation were only a study of historical conditions in Accotink and Pohick, it would have been useful to have additional full data sets around 2005 to see if the patterns noted here held for all years. Monitoring of these watersheds periodically could result in enhanced understanding of loading in these mid-Atlantic watersheds. Enhanced analysis of other existing data sources might also be useful: particularly if more comprehensive validation analysis could be done with Fairfax County data. Furthermore, there is at least one Virginia Department of the Environment Monitoring site at Accotink 20 (Accotink and Braddock Road) and numerous volunteer monitoring sites within the watersheds.

Another issue with the analysis is the unexplained high measured values of nitrate-N in 1984. Apparently, they were an anomaly; however, perhaps there is some as yet unexplained driver that caused a spike in nitrate-N concentrations in both watersheds. Regardless, these data raise several questions as to the cause of the high measured nitrate-N values that are currently unanswered.

Lastly, this study attempted to look at multiple water quality and hydrologic conditions to gain a firm understanding of the watersheds. Using tools such as those used here could allow planners to consider more than the single parameter approach common in many TMDLs or 303d listed waters. For example, a bacterial TMDL was completed for Accotink, yet an informed watershed scientist could easily make the argument that this is not the most significant nor relevant cause of impairment in this stream. Increased degradation or loadings of pollutants are often correlated. Setting maximum daily loads for one parameter while ignoring the others has significant shortcomings. Though we may lower that particular pollutant, odds are another pollutant will quickly become limiting and improvements in habitat and water quality limits could be slowed or stopped. Furthermore, considerable resources are being spent on analysis of a watershed; with very little supplemental resources, a more comprehensive analysis could be completed. In short, an analysis that examines multiple facets of watershed conditions has significant strengths. Fairfax County did this somewhat with its stream protection strategy and is continuing to do so with their watershed management plans. Hopefully, this analysis will be useful when such plans are designed for Accotink and Pohick.

5. The Impact of Future Urbanization on the Accotink and Pohick Watersheds

5.1 Introduction

Fast paced suburbanization has resulted in significant changes for many formerly rural or forested watersheds. As the nation continues to increase in population, housing and demographic patterns may continue to change and average suburban population density will quite possibly continue to decrease. As a result, cities will continue to develop outward and urbanize many metropolitan counties, consuming land for urban purposes at a faster rate than the rising urbanized population would seem to indicate. This urbanization has had and will continue to have significant implications for water quality and the hydrology of watersheds. Urban land use has been shown to increase runoff volume, increase peak flow, decrease base flow, reduce groundwater recharge, and lead to increased channel erosion and stream channel widening. Furthermore, urban land use also reduces water quality, reduces habitat quality, increases toxic, nutrient, and sediment loadings, and increases temperature (Booth, 1990; Davis *et al.*, 2001; Hogan, 2005; Jones & Clark, 1987; Nelson & Booth, 2002; Schueler, 1994). Hence, absent effective Best Management Practices (BMPs) or other mitigation approaches, urbanization will play a progressively larger role in the impairment of streams, rivers, lakes, and estuaries.

Plausible land use projections used in combination with appropriate watershed analytical tools can help researchers and scientists quantify the scope of impact that urbanization might have for any given watershed or region. Such predictions could then prove useful to watershed managers and policy makers in weighing policy options to meet watershed goals. This chapter integrates numerous tools to project the impacts of land use change on Accotink and Pohick watersheds, and loadings to Gunston Cove as the receiving waters. Those tools include two approaches for generating future land use projections and three models that examine hydrology and/or water quality. The two approaches for forecasting land use are:

- Modified household land use method projections, and
- SLEUTH land use output

Three approaches are used with these land use projections to forecast impacts to hydrology and nutrient loadings:

- The Hydrologic Simulation Program Fortran (HSPF)
- Long-Term Hydrologic Impact Assessment Tool (L-THIA)
- The Export Coefficient Method (with PLOAD and spreadsheet output)

The two land use approaches include a modified version of the household method (discussed in chapter 3) and adapted model output from a preexisting cellular automata model known as SLEUTH (Jantz *et al.*, 2004). The background and adaptation necessary to include these models is discussed in section 5.2. These land use projections were then incorporated into three water quality diagnostic tools in Sections 5.3 and 5.4. The first water quality tool, the hydrologic component of the Hydrologic Simulation Program Fortran (HSPF), was used to project hydrological changes in future land use scenarios and is discussed in Section 5.3.1. Output from HSPF in future land use scenarios were compared to land use conditions in 1992 to illustrate hydrological changes that are projected to occur without effective implementation of Best Management Practices (BMPs). The HSPF modeling framework was selected because it is well tested, is included in EPA's BASINS system, and can be used to generate both daily flow and annual flow results (Lumb *et al.*, 1994). Furthermore, if this work is pursued in the future, HSPF can be calibrated for use with nutrients, and the results can be compared to the results of the other approaches used to estimate those nutrients in this dissertation. The second tool, the Long-Term Hydrologic Impact Assessment (L-THIA) tool, was used to compare flow under different land use change scenarios in Section 5.3.2. L-THIA was selected because it is simple to use, can be operated from a web based interface or a GIS extension, and is being promoted is a viable tool for watershed managers to examine the impacts of land use change. In this chapter, L-THIA was used to examine changes in annual runoff. Model performance of the more complex HSPF and the simpler L-THIA are compared in Section 5.3.2.6 to see how L-THIA's annual flow results compare to the more complex HSPF annual flow results. The third water quality prediction approach, discussed Section 5.4, used the export coefficient method to generate estimates of changes in nutrient loadings as a result of land use change. The export coefficient method was used in both the form of the model PLOAD and a created spreadsheet model. The export coefficient method is a simple level screening approach can give plausible estimates of loading to water bodies (US EPA, 2001b). In section 5.4.4.1, nutrient loadings were generated by the export coefficient method for past years to compare to loadings generated by the LOADEST model in chapter 4 of this dissertation to validate the export coefficient approach. Section 5.4.4.2 then uses this approach to predict changes in nitrogen loading under various projected land use scenarios. Key characteristics of these land use and water quality modeling tools are outlined in Table 5.1.

The final three sections of this chapter are discussion sections. The first section (5.5) discusses the relative threat to aquatic ecosystems from increases in nutrient loadings compared to changes in hydrology. Section 5.6 discusses all of the tools used in this chapter, and to what extent these approaches are useable or exportable by watershed managers, policy makers, or researchers. The final section, 5.7, discusses future research to expand upon the findings in this chapter. The framework used in this chapter is highlighted in Figure 5.1.

Table 5.1. Key characteristics of the land use and water quality models used in this chapter.

	Land Use Models		Water Quality and Hydrology Models		
	SLEUTH*	Household	HSPF	L-THIA	Export Coefficient Method, (PLOAD, Spreadsheet model)
Discussed in section:	5.2.2	5.2.3	5.3.1	5.3.2	5.4
Key Input Data Required	Physical characteristics of the watershed, transportation network, exclusion zones, defined rules	Demographic housing data, spatial remote sensing data (to establish density)	Land use, watershed area, watershed characteristics, flow, climatic data (precipitation, dew point, potential evapotranspiration, etc.), ~ 25 calibration parameters, point source loadings	Land use, land use characteristics, precipitation, soil type	Land use, export coefficients, point source loadings
Type of Model	Stochastic	Deterministic	Deterministic	Deterministic	Deterministic
Calibrated	By Jantz et al.	Yes	Yes	No	No
Validated	By Jantz et al.	Yes	Yes	Yes	Yes
Output generated for use in this chapter:	Future Land Use Estimates	Future Land Use Estimates	Hydrology: Peak and low daily flow: annual flow	Annual Flow	Nutrient Loadings
Time Period for modeled output used in this chapter	2030	Projected 2025* Treated alongside Jantz et al. 2030 estimates	1991-1993 (Baseline) 2025;2030 (Projected Scenarios)	1991-1993 (Baseline) 2025;2030 (Projected Scenarios)	1991-1993 (Baseline) 2025;2030 (Projected Scenarios)
Other output potentially generated by model (not used in this chapter)		Past/Current Land Use	Bacteria, Nutrient, Sediment, and Toxic Loading	Nutrient Loadings	

* Model was not run for this chapter; rather, existing model output from the Jantz et al. (2004) study were used.

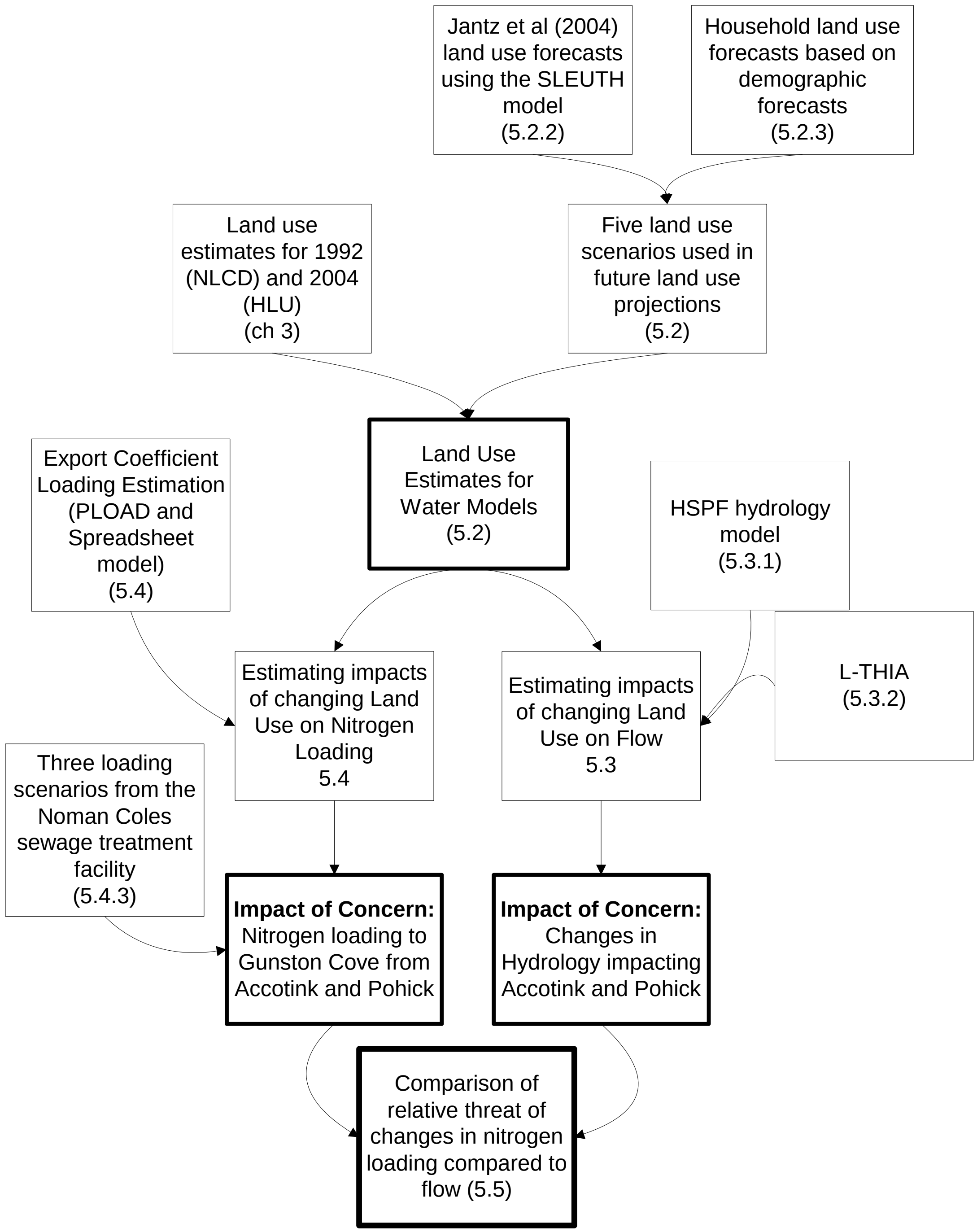

Figure 5.1. Schematic of logic and tools used throughout this chapter to examine the impacts of land use change on hydrology and nitrogen loading.

5.2 Future Land Use Change Scenarios

5.2.1 Background

Future urban land use can be projected using a range of tools. One common method is to combine zoning overlays with growth projections. Projecting future land use based on zoning allows researchers or practitioners to incorporate current policy makers' decisions and values in their forecasts. Furthermore, for the short term, these approaches are likely to be more accurate in predicting the exact physical location where development will occur. However, using zoning data alone for forecasting future land use has substantial limitations. Regulations and zoning locations are often changed and the likelihood of these regulations remaining static through build-out of the urbanizing area is unlikely (Conway & Lathrop, 2005). Though zoning and population growth estimates can be used by city planners to predict build-out scenarios, current zoning fails to take future adjustments to economic realities and values into account and, as such, produces long term forecasts that may deviate from actual build-out.

A second common method is to use land use change models. These models can estimate land use change based on historical data, economic data, physical constraints, and/or logical transition rules (Verburg *et al.*, 2006). Land use change models are simplified re-creations of real world development. Multiple techniques are used to model land use, including, but not limited to, statistical models, econometric models, multi-agent system models, and cellular automata models (Batty, 1997; Calkins, ; Clarke *et al.*, 1997; Irwin & Bockstael, 2002; Parker *et al.*, 2003; Pijanowski *et al.*, 2002; US EPA, 2000d; White & Engelen, 1993). These models are currently being used to explain urban form and deforestation patterns, predict urbanization, study the most financially beneficial use of land, and address other related land use questions (Batty *et al.*, 1999; Clarke *et al.*, 1997). All of these modeling techniques have their strengths and weaknesses, ranging from being overly simplistic to being data hungry, or ignoring social, economic, or physical variables. When it comes to predictive capabilities, a model is not a crystal ball, and thus it is virtually impossible that a model will be completely correct. However, models do provide some indication of what future land use may look like and can serve as a valuable tool for policy makers and scientists.

An example of a spatial model is one developed by Irwin and Bockstael (2002) to model urban development in Maryland counties surrounding Washington, DC. The model incorporates economic conditions, parcel size, density, and commuting distance, cost, and other important characteristics, such as 'no development zones' in water features such as rivers or in parks, public policy decisions, and value based/economic decisions. The decision to 'urbanize' a land parcel occurs when the greatest profit will be obtained by the landowner (thereby assuming that there is a rational actor). The Irwin and Bockstael model appears to accurately model current conditions and predict where development is likely to occur in the near term with the range of its calibration data. Furthermore, the model appears to do a reasonably good job of replicating development patterns when compared to actual historical development patterns in those Maryland suburbs. However, the model is data intensive; therefore, its greatest strengths appear to be examining development patterns on the small to meso scale. Additional strengths appear to include simulating and studying the effects of existing land development patterns, short-term projections, and examining the effects of negative externalities.

The projections used in this chapter are based on a cellular automata (CA) model developed by Clarke et al. (1997), now known as SLEUTH. White and Engelen (1993) showed that using cellular automata in modeling urban land use patterns created realistic spatial results at the macro-scale. Cellular automata models are important tools for forecasting how urbanization might shape future land cover patterns (Candau & Clark, 2000). These models can use relatively simple mathematical equations and rules of development to model the rather complex processes of urban development. A simple cellular automata model contains space represented by cells that are in a certain state at any given time. The status of each cell is determined by a predefined number of states; for example, in the case of an urban development model, the states could be vacant, residential, commercial and industrial. Torrens (2000) states that cellular automata have many advantages for modeling urban phenomena, including flexibility, a dynamic approach, compatibility with aspects of Geographic Information Systems, and a decentralized approach (particularly important when considering certain aspects of current urbanization patterns). The data needed for CA models are often easily accessible and available readily and cheaply from public sources. In contrast, an econometric land use change model, such as that developed by Irwin and Bockstael (2002), requires locating, using, and then interpreting a significant amount of localized economic data. Furthermore, CA models have produced good results in areas that cross jurisdictional boundaries.

Cellular automata models incorporate physical and sociological features, including transportation networks, topography, distance from city center, value placed on certain developments and amenities (i.e., distance to parks or lake shores). Development is modeled by using transition functions, which can use conditions or states in that cell and neighboring cells to determine what the cell will be in the next transition period. These transition rules are used to determine the future of cells at a time t+1 (O'Sullivan & Torrens, 2000). These rules will depend upon the situation in the given cell, neighboring cells, and the programmer's weights given to each variable.

As mentioned, the neighborhood of a given cell will influence its transition potential. Batty et al. (1999) state that objects in spatial systems cellular automata models are defined as "cells which can take on various states and which are influenced by what is happening in other cells in their immediate neighborhood" (pg. 206). The distance in this neighborhood is arbitrarily based upon what the author views as maximum distance to directly affect a given cell's state. For example Clarke et al. (1997) set the distance at six 100 meter cells (or .6 km). Barredo et al. (2003) state that .8 km is what "residents of a city commonly perceive to be their 'neighborhood.'" In almost all cellular automata models, nearby cells have greater influence than those cells which are farther apart. Throughout the literature, similar themes can be found characterizing neighborhood impacts. For example, industrial cells always repel residential cells, and are often attracted to one another. Commercial cells may repel residential cells when they are directly next to each other, yet attract them when they are one or two cells away.

The SLEUTH model, whose output is used in this work, was originally developed by Clarke et al. (1997) to examine development in the San Francisco Bay region and to generate scenarios of future growth. Historical land use data from maps from the years 1850, 1900, 1940, 1954, 1962, 1974, and 1990 were scanned and rasterized, although only data from maps from the 20th century were ultimately used in the model's creation. The created model is complex, incorporating many features and variables. The authors assembled a digital database and used geographic information systems to support modeling of the urban transformations and animation. They also collected four types of data: land cover, slope, transportation, and protected lands. The map scale varied by the specific data layer. For example, land cover was taken from 1:24,000 and 1:62,500 maps; topography at 1:250,000; and transportation at 1:24,000, 1:62,500, and 1:2,000,000. The model did not incorporate zoning or focus on socioeconomic issues such as income or job growth; however, these drivers are factored implicitly into historical data. The model used seed cells based on historical data and incorporated the data on a grid of 300 meter cells; statistical and graphical tests were used to calibrate the model. Each cell acted independently, imitating the actual city expansion, simulating "the result of hundreds of individual personal decisions, made one at a time, but susceptible to the physical, social, economic, cultural, and political landscape." (Clarke et al., 1997).

The SLEUTH model has five major factors or coefficients which control the behavior of the system and thereby predict future land use. The BREED factor determines how likely a newly detached settlement will form; the SPREAD coefficient directs outward ‘organic’ expansion; the SLOPE_RESISTANCE factor decreases likelihood of development on steep slopes, the ROAD_GRAVITY factor increases the likelihood of a new settlement near the existing road system; and the DIFFUSION factor determines dispersiveness of grid development and spreads it outward through the road system. Implemented as innovative features, the diffusion, spread, and breed factors are increased once they reach a critical value by using a multiplier greater than one to simulate the concept of an increasing pace of urban growth as cities grow larger. Exponential growth is prevented by slightly decreasing the multiplier applied to these factors each year. The system further modifies itself by adjusting the slope resistance factor as density increases, allowing development on steeper slopes, increasing the road gravity feature as the road system enlarges, and decreasing the diffusion, spread, and breed factors when the system falls below a critical value.

The Clarke et al. (1997) San Francisco calibration of the model suggests that organic growth is the most prevalent type of growth, with spontaneous growth playing the second largest role. Organic growth can be defined as growth surrounding existing growth, either spreading outward from existing growth or infill development. Spontaneous growth occurs anywhere on a landscape and does not rely on existing infrastructure. This type of growth can trigger sprawl, leapfrog growth, or new urban centers. If the model incorporates historical road data, the impact influence of road-influenced growth increases. This result shows that growth is predicted in a relatively dispersed pattern and the predicted growth form and location match up reasonably well with actual growth.

Clarke's model was further developed in the project Gigapolis developed by Candau and Clark (2000), which coupled the pre-SLEUTH 'Clarke Urban Growth Model" with another model, the Deltatron Land Use/Land Cover Model. The model was tested and calibrated using 1-km resolution data sets in the Mid-Atlantic Region, then the land cover change was simulated beginning in 1992 and continuing to 2050. For the purposes of this work, the scale is too large for useful analysis. However, the analysis gives a plausible prediction of land cover change in the Mid-Atlantic States through 2050, illustrating the feasibility of designing such a model with cellular automata as its base. Furthermore, additional work has been done with this model, such as Silva and Clarke's (2002) calibration for select European cities, illustrating the adaptability of this model to differing situation when correctly calibrated.

Jantz et al. (2004) used SLEUTH to model land use in the Washington DC-Baltimore metropolitan region under various policy scenarios. They modeled a large portion of the Chesapeake Bay Watershed and projected growth to the year 2030. The researchers produced three different scenarios: growth under current trends (they referred to it as the current conditions scenario), growth under managed conditions (referred to as smart growth), and ecologically sustainable growth (referred to as sustainable development). They used Landsat data from 1986, 1990, 1996, and 2000 at a resolution of 30 meters to estimate historical and existing land use, but due to computing power limitations, resampled their data to a 45-meter resolution. The authors calibrated their model and completely excluded water from development, which means that there can be no development in lakes, rivers, or other large bodies of water. They also put an 80%

level of development exclusion on local, state, and national parks, making it unlikely that these areas would develop. Their rationale for placing the 80% development exclusion instead of 100% exclusion on parks is that there is occasionally development within parks: buildings, labs, or parking lots are built. Using the historic land use data, the authors calibrated the model against actual development locations. For this calibration, they produced 100 Monte Carlo iterations, from which they calculated a probability map of cells urbanizing by the year 2000. They classified everything that had a greater than 50% chance of urbanizing as urban, and compared these results to actual observed data at the pixel, watershed, and county level. The authors disabled the SLEUTH model’s self-modification function. This action may have resulted in higher growth predictions, since disabling this function allows for a linear growth rate. Accuracy was not determined to be terribly high at the pixel level. However, at the 11-digit hydrologic unit code level (HUC11) the model performed reasonably well and accuracy was acceptable. Hence, they concluded that these results would be useful on a regional analysis, but not necessarily for predicting the exact location of development. HUC11 watersheds range from approximately 40,000-250,000 acres. The Accotink watershed is approximately 32,800 acres while the Pohick watershed is approximately 21,900 acres, combining for a total combined area of 56,100 acres. Hence, this approach was deemed appropriate for use in these watersheds.

5.2.2 Jantz et al. (2004) Model Land Use Forecasts

The Jantz et al. (2004) model produced three scenarios of growth under different policy assumptions. The first scenario, referred to as the base or the projected current conditions[19] scenario, was calibrated under the assumption that current policies would remain in effect. The second approach was based upon Smart Growth policies, and assumes that existing development centers would grow into high density hubs. The third iteration was based on a sustainable approach which severely limited growth sprawling away from Washington, DC. The sustainable development approach assumes more aggressive policies are implemented that severely limit outward sprawl and total urban area development (thereby assuming higher density) relative to the Smart Growth approach. Jantz et al. (2004) simulated these scenarios by using an exclusion factor in areas that they deemed more consistent with alternative approaches. Areas that were deemed less likely to be developed under different policy scenarios due to their current land ownership (i.e., parks) or because of physical constraints (i.e., open water) were given a high exclusion factor, which means that they are less likely to be developed. In the sustainable development scenario, areas far away from existing urban centers had a high exclusion factor, which significantly limited the likelihood of urban development. Furthermore, spontaneous growth was limited in this scenario, concentrating new development around existing urban centers. In the Jantz projected current conditions scenario, these areas were given no special exclusion factor. In summary, the sustainable

[19] Current Conditions is referring to current policy conditions and current development trends. To reduce confusion, this scenario will be referred to as Jantz projected current conditions (JCC) land use scenario, or projected current conditions scenario.

growth scenario limited new growth much more significantly than the smart growth scenario, particularly growth further from urban areas.

5.2.2.1. Data Format

Many of the data produced in the Jantz et al. study were made available for the current study by Dr. Claire Jantz (Jantz, 2005). The land use predictions for 2030 were shared as probability maps of the likelihood that any given cell would urbanize. A land use cell had between a 0 and 100 percent chance of being urban in 2030. The delineated Accotink and Pohick watershed files were reprojected to Albers Conical Equal Area NAD83 and overlaid on top of the land use data files. Cells were grouped based upon likelihood of urbanizing and then summed in each subwatershed. As can be seen in Figures 5.2-5.4, the three different scenarios do not produce dramatically different outcomes in terms of percent urban area because of their proximity to Washington, D.C. and existing development. The policies that would be implemented in the sustainable development and smart growth scenarios would direct development from outlying suburbs to more inner suburbs such as those in Accotink and in the headwaters of Pohick. The greatest differences in the scenarios are noted in outlying watersheds or areas that do not currently have urban growth centers such as watersheds to the south and west in Prince William and Loudoun counties. However, in the far western portion of Pohick, a significant percentage of area is urbanized in the current trends scenario that is not urbanized in the sustainable growth scenario. Additionally, there is a notable difference in the areas near streams and in the southern portions of the watersheds, where the Jantz projected current conditions scenarios allow for greater urban development in these areas than the other two scenarios. Hence, the three scenarios do result in differences in predicted urban area in each watershed (Figure 5.5).

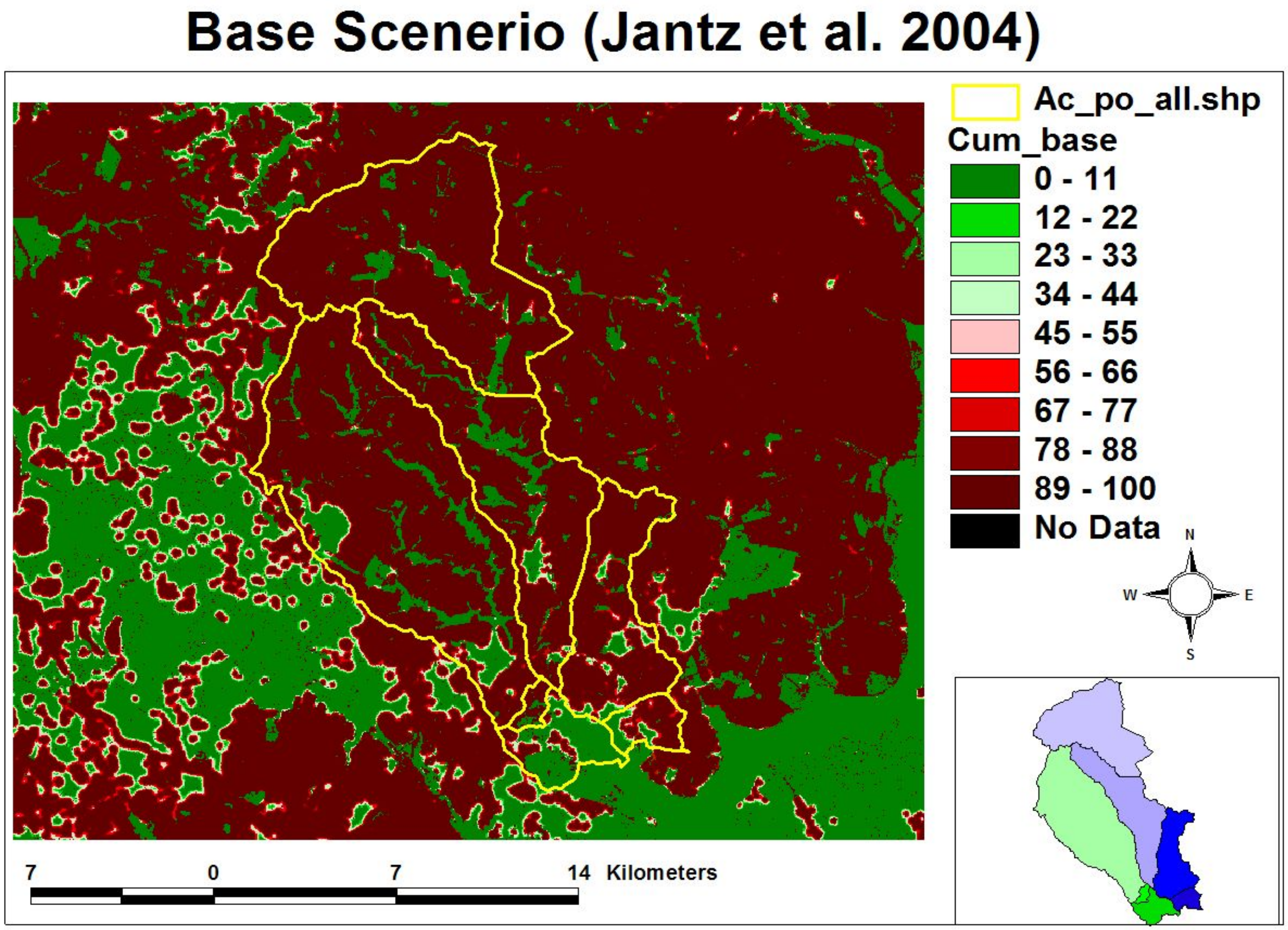

Figure 5.2. Projected land use in 2030 using the Jantz et al. projected current conditions (JSCC) scenario in the Accotink and Pohick watersheds. The darker the red color is, the greater the likelihood of urbanizing. The drawing in the bottom right shows the watersheds and subwatersheds by color: Accotink is blue, Pohick is green, with lighter shades indicating upstream areas in that watershed.

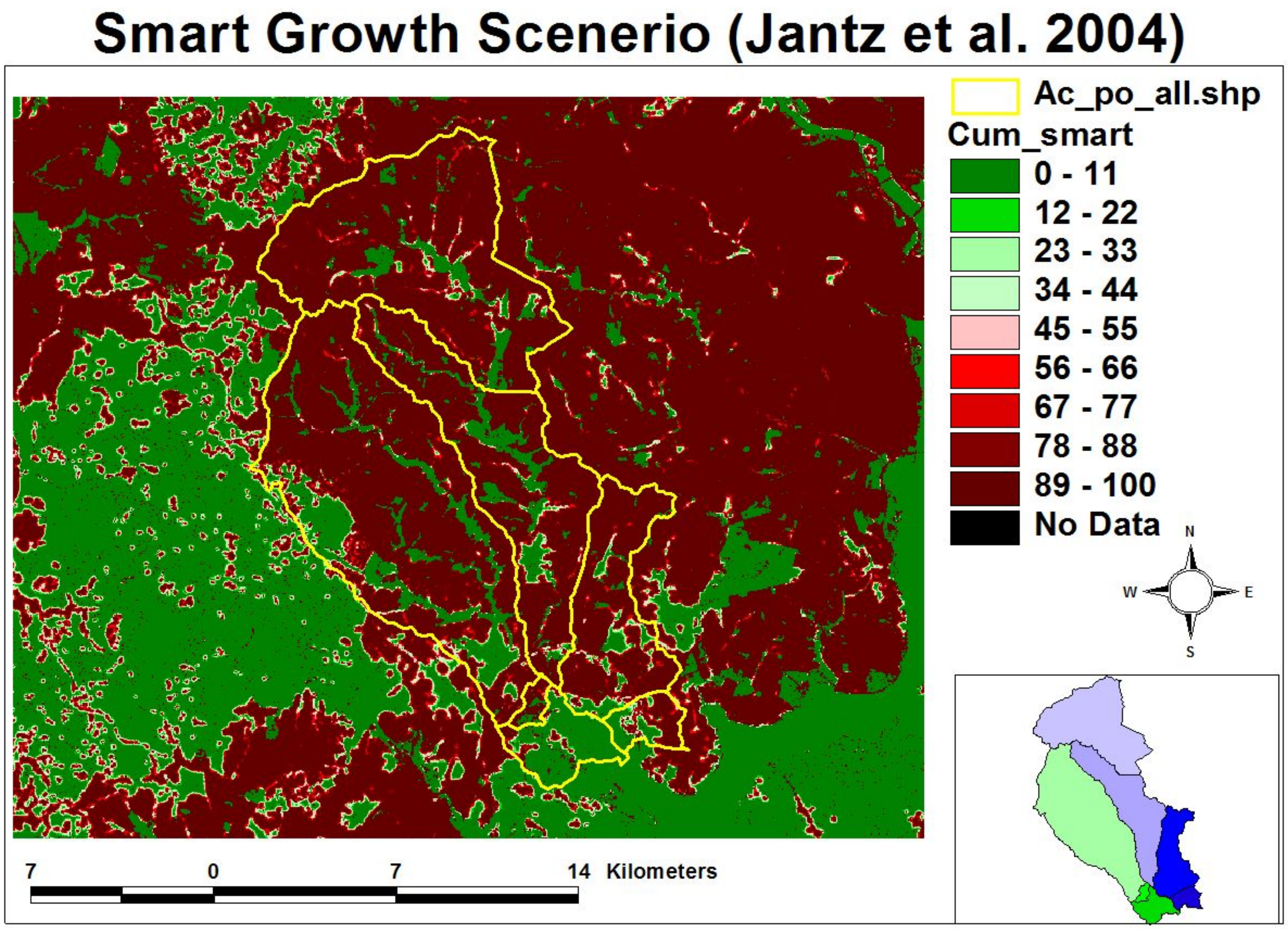

Figure 5.3. Projected land use in 2030 using the Jantz et al. smart growth scenario in the Accotink and Pohick watersheds.

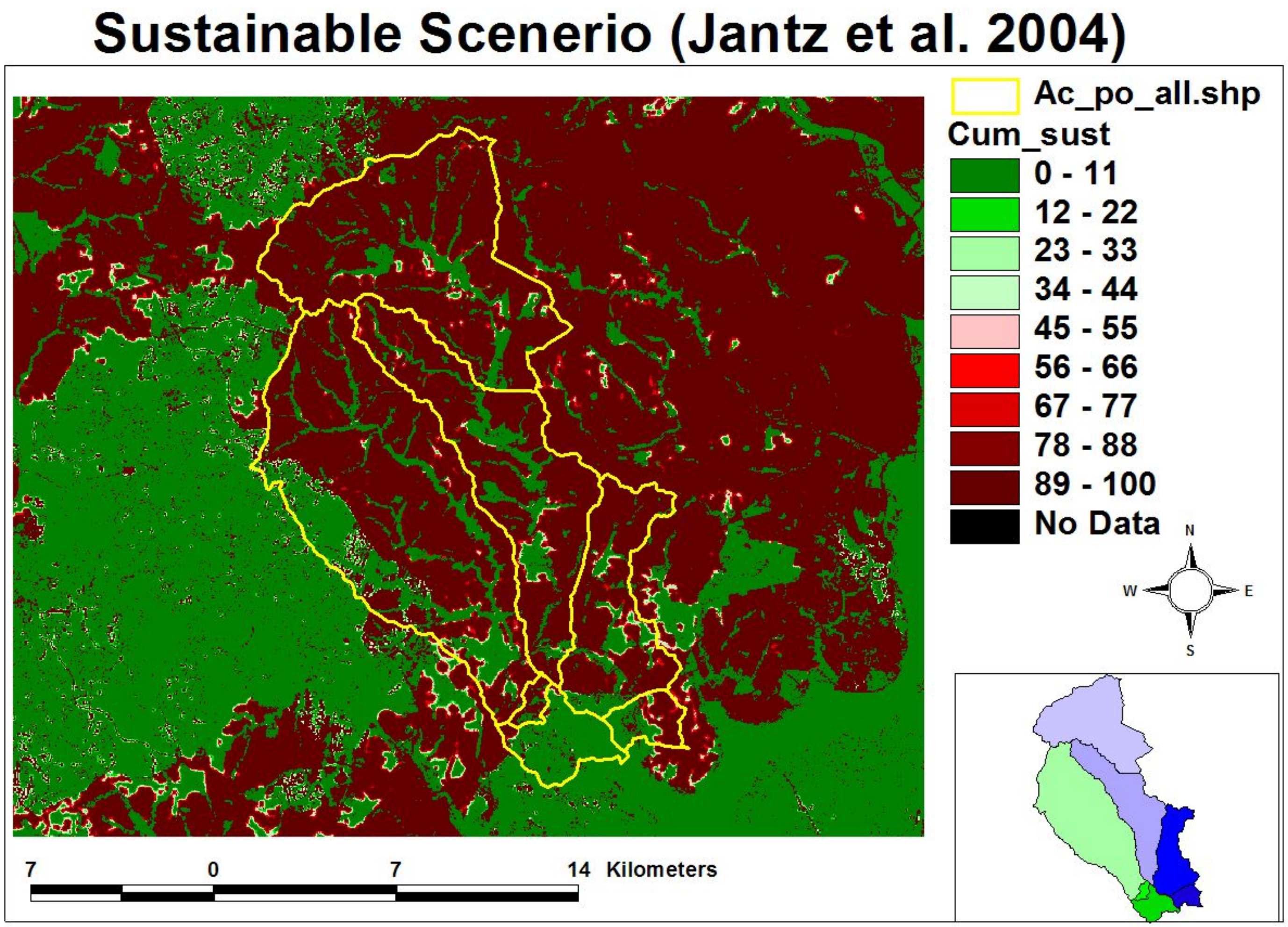

Figure 5.4. Projected land use in 2030 using the Jantz et al. sustainable scenario in the Accotink and Pohick watersheds.

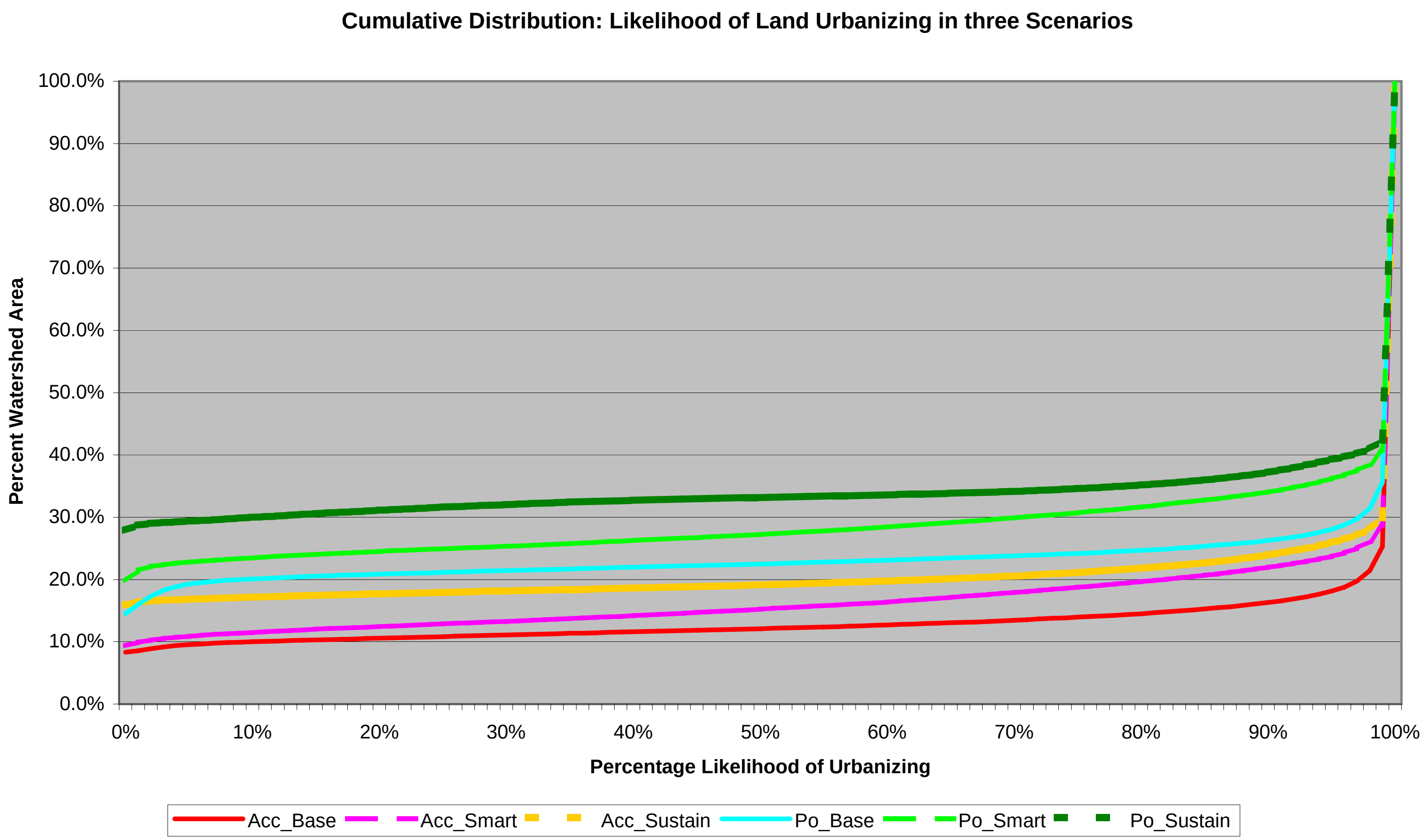

Figure 5.5. A cumulative distribution curve showing the percentage of each watershed and how likely it is to urbanize by 2030 in three scenarios. If the curve representing a scenario has a low percentage of watershed area that is not likely to be urban, such as in the Accotink base scenario (solid red), then a significant portion of the watershed is likely to be urban. Likewise, if a scenario has a higher percentage of watershed area that is unlikely to urbanize, such as the Pohick Sustainability scenario (dashed green), then less of the watershed is predicted to be urban.

5.2.2.2. Original Weighted Jantz Results

The weighted urban land use prediction for the year 2030 was then produced using the Jantz projected current conditions or base scenario (JCC), the smart growth scenario, and the sustainable development scenario. These estimates were produced by the equation[20]:

$$TUL_{s\_sw} = \sum_{i=1}^{N} p_i * cellarea_i$$

Where
TUL = Total urban land use for a given scenario in a subwatershed
P_i = Probability of a cell urbanizing
S_SW = The selected scenario in a given subwatershed
N = the number of cells in the watershed

This weighted approach resulted in land use estimates that were between 80.2 and 87.3% urban for the full Accotink watershed and 66.7 to 77.2% urban for the full Pohick watershed (Figure 5.6). In the model calibration, Jantz et al. (2004) estimated 2000 urban land coverage using remote sensing data. They estimated that Accotink was 63.6% urban and Pohick was 53.0% urban. The three projected land use scenarios represent an absolute increase of 16.6 to 24.3% urban coverage in the full Accotink watershed and of between 13.7 to 24.2% in the Pohick watershed. Hence, the sustainable development scenario resulted in just over half the urbanization in the Pohick watershed compared to

[20] An alternative and more sophisticated approach would have been to use the original Jantz et al. (2004) output for each of the 100 Monte Carlo distributions. This approach would have produced a range of possible outcomes and a distribution curve to be used in the water quality and hydrology models instead of the single outcome that is produced by using the above approach. This alternative approach is discussed in somewhat greater detail in section 5.6.1The approach used here effectively uses the mean value of each land use scenario in the water models.

the urbanization projected in the Jantz projected current conditions scenario. In the Accotink watershed, the sustainable development scenario resulted in about 70% of the urbanization projected in the Jantz projected current conditions scenario. Therefore, progressive land use policies stand to make a greater impact limiting the amount of new urbanization in Pohick than in Accotink.

The urban estimates produced, particularly for the Jantz projected current conditions scenario, are likely overestimating future urban land use due to the disabling of the SLEUTH self modification feature. Additionally, for past land use, the Jantz data estimate higher percentages of urban land use than the NLCD (1992) or MRLC (2000) data for which the HSPF hydrology model is calibrated (see section 3.3). Nonetheless, the Jantz projected current conditions (JCC) land use scenario for year 2030 is used in its existing form as a high end estimate of potential urban land development as one of the five land use scenarios modeled in the water quality application section. These results are presented with the other four land-use scenarios used in Table 5.3.

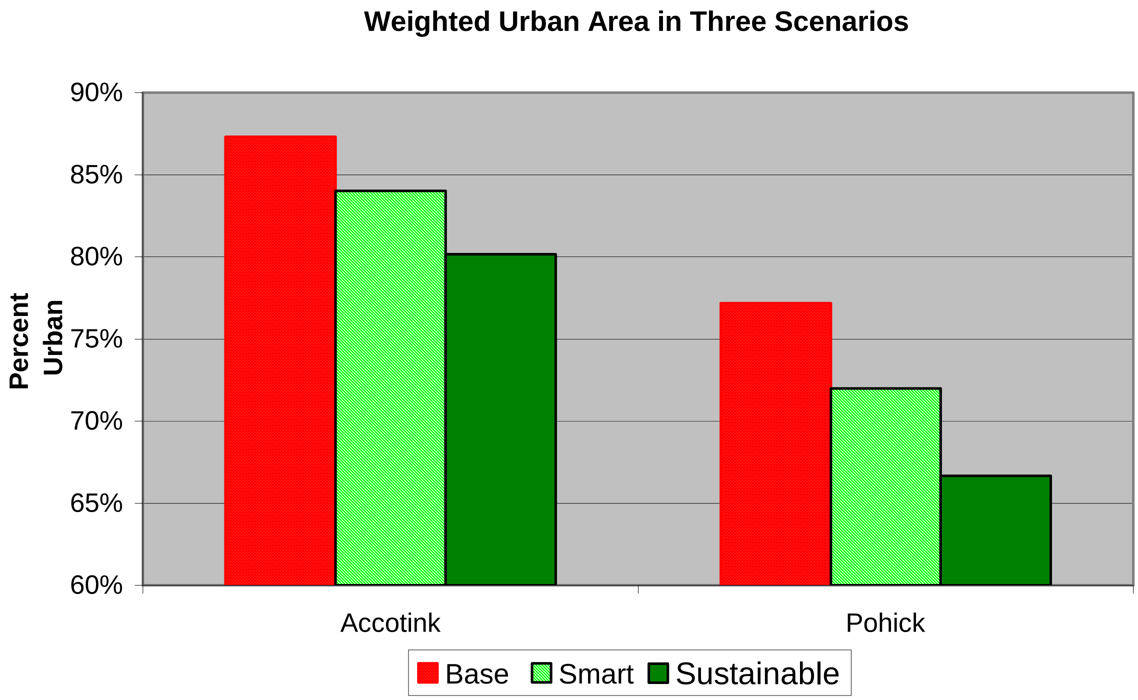

Figure 5.6. Weighted urban area estimates for the full Accotink and Pohick watersheds in three land use scenarios adapted from Jantz et al. (2004).

5.2.2.3. Adjusted Jantz Results

As was discussed Chapter 3 and shown in Figures 3.6 and 3.7, the Jantz et al. estimates tend to predict higher levels of urban area than the National Land Cover Database (NLCD) dataset for which the hydrologic model (HSPF) model is calibrated and validated and the HLU estimates. The relative differences between the Jantz estimates and the HLU estimates have strong linear relationship. These differences can be accounted for in the initial land use data set used for calibration of the Jantz et al. (2004) model compared to the NLCD data an the HLU estimates produced in chapter 3. This relative overprediction might be attributable to different classification regimes; the

Jantz et al. (2004) classification could have been slightly more sensitive than the corresponding NLCD data to sensing and classifying an area as urban. The differences could also be due to issues associated with scale: the Jantz data is based on a 45-meter resolution resampled landsat image based on a 30-meter resolution. The NLCD data is based on a 30-meter resolution. Urban area may have become more dominant in the resampling process. As a result of this disparity, adjustments were made to the Jantz predictions. These adjustments produced three adjusted Jantz scenarios: the adjusted Jantz projected current conditions scenario (ADJ-CC), the adjusted Jantz smart growth scenario (ADJ-SG), and the adjusted Jantz sustainable scenario (ADJ-Sust). These adjustments, based exclusively on the relationship between the historic data sets, were made to assure that results were most appropriate for comparison with the water quality models. It was assumed that the model's overestimation relative to the NLCD dataset for predicted urbanization would be comparable to those data on which it was calibrated. Hence, the linear regression shown in Figure 5.7 was used to generate an equation to allow for compatibility between the Jantz urban land data and the NLCD and HLU data. The equation is:

$$CUL = 1.0227WUL - .0874$$

Where
CUL = compatible urban land use
WUL = weighted urban land use estimates from Jantz

This approach downscaled the amount of urban area in each watershed, producing results that ranged from 73.2% to 80.6% urban for the Accotink watershed and 59.4-70.2%

urban for the Pohick watershed. These results are compared to those from the modified HLU method and the unadjusted Jantz projected current conditions land use in Table 5.3 and Table 5.4.

5.2.2.4. Generation of Non-Urban Land Use Estimates

The Jantz et al. model produced two classes of land use: urban and non-urban. In order to be compatible with the water quality and hydrology models, I generated estimates for other land uses including those for forest area, grassland, water and wetlands, and barren/transitional. I derived a ratio from the 2000 MRLC data, which are not the same dataset from which Jantz et. al (2004) calibrated their model. First, area that was classified as wetlands or water was kept constant to the amount in the MRLC 2000 since it was assumed development would not occur in flooded areas or wetlands. The proportion of each watershed that was in wetland or water land use classes was relatively small. For forested, grasslands, and barren/transitional area, I assumed that the proportion of each land use excluding urban land use, water, and wetlands would remain constant. Hence, in the 2000 MRLC data 68.1% of the area in these land use classes was forested, 27.1% grasses, and 4.8% barren or transitional, this relationship was maintained in the forecasted land use model. This approach requires us to make the assumption that the relative balance of grasses and forest remain somewhat constant, so these assumptions are a possible cause of error. However, using ratios based on existing remotely sensed data appears to be the most plausible way to produce estimates that are consistent with past development trends without generating refined, calibrated, multi-land use class models. Results from this approach are presented in Table 5.3 and Table 5.4.

5.2.3 Land Use Projections using the Household (HLU) method.

The HLU approach from chapter 3 was used to create a fifth development scenario. The household projections were used from the most recent year that data were available at the sewershed level (2004) from Fairfax County's demographic reports (Fairfax County, 1975-2004). The amount of urban land required by each household (and relevant support services) was assumed to remain constant. According to the 2004 report, the county uses a different approach for short-term and long-term forecasting. The short-term forecasts go five years into the future and are based upon active residential development of houses in the construction or planning process. The long-term forecasts are based upon areas reaching build-out, defined as filling all capacity that exists for residential development using existing planned land uses under the county comprehensive plan. According to the demographic report, additional development may be predicted if recent development activity indicates that there may be additional capacity beyond the build-out scenario. Development scenarios based on build out typically underestimate actual future development.

The Fairfax demographic report projects that build-out may occur as early as 2010 or 2015 in areas with medium to high density developments, which include both Accotink and Pohick. Using these household forecasts in the household method shows the development of new urbanized area flattening dramatically between 2010 about 2015. Results from this approach are presented in Table 5.2.

Table 5.2. Results of urban land use projections using the HLU method with county planners' projections of household development.

Site	2005	2010	2015	2020	2025
Site 1	54.7%	59.7%	60.9%	61.3%	61.4%
Site 2	55.0%	60.1%	61.3%	61.7%	61.8%
Pohick Full	**50.5%**	**55.2%**	**56.3%**	**56.6%**	**56.7%**
Site 20	66.0%	71.1%	73.8%	74.7%	74.9%
Site 13 Modified	64.2%	68.1%	70.3%	71.0%	71.3%
Site 13 Original	63.7%	67.8%	70.3%	71.1%	71.4%
Accotink Full	**62.4%**	**66.5%**	**69.0%**	**69.8%**	**70.0%**

Use of the household method as a prediction technique gives an alternative growth prediction to the Jantz model; however, there is likely slight underprediction to what should be actual or real growth due to rezoning and in-fill development not being considered in most cases. The county admits to underpredicting future urbanization due to inability to predict changes in zoning. How a given parcel is zoned can be changed based upon modifications to county comprehensive plans or based on appeals from property owners. Therefore, the county's approach likely predicts fairly accurate estimates for the 5-year time horizon; but it is quite likely that these results underpredict urban area in 2025. On the flip side, as vacant land becomes scarcer, urban development will occur more as infill development or redevelopment of older structures (commonly called tear-downs). In the case of low density household structures, this may result in the loss of established yards with trees as higher density areas or houses with larger footprints are constructed: these yards may have shown up as grasses and forests in the

NLCD data. This loss is likely to be relatively small due to the scaling issues with the remote sensing data; unless significantly large, these areas would not be picked up with the large 30 meter pixel scale. The other noteworthy point is that the household method was designed to estimate past land use at a higher temporal resolution. The inherent assumptions, calibrated based upon density of the number of households to urban land use at a watershed scale may not hold true in real world future land use scenarios.

The projected urban land use from the household method slightly underpredict those of the adjusted Jantz approaches (Table 5.3). The results of the household method in 2025 are compared to those for the Jantz data for 2030. It was deemed to have little consequence as the rate of change in the household projections is near 0 between 2020 and 2025. Hence, since build-out was reached, the methodology would result in few changes in 2030. While all five land use scenarios are discussed in combination, the reader is reminded that the HLU estimates are from 2025. The values presented in Table 5.3 are the urban land use estimates used in all water quality modeling approaches.

Table 5.3. Percent future urban land use estimates with the original Jantz projected "Current Conditions" (JCC) scenario, three adjusted Jantz scenarios (ADJ-CC, ADJ-SG, and ADJ-Sust.) and household method (HLU) projections based on the number of household projections by Fairfax County planners.

Site	JCC	ADJ-CC	ADJ-SG	ADJ Sustainable	HLU
Site 1	83.0%	76.1%	71.1%	65.6%	61.4%
Site 2	82.9%	76.0%	71.0%	65.6%	61.8%
Pohick All	77.2%	70.2%	64.9%	59.4%	56.7%
Site 20	90.6%	83.9%	80.9%	77.6%	74.9%
Site 13 Modified	87.9%	81.1%	78.3%	75.3%	71.3%
Site 13 Original	88.0%	81.3%	78.1%	74.2%	71.4%
Accotink All	87.3%	80.6%	77.2%	73.2%	70.0%

For the household method, estimates for forested, agricultural and transitional land use are estimated using the same approach discussed in Section 3.2.

Table 5.4. Urban land use for one scenario using original output from the Jantz et al. (2004) model, three scenarios based on the adjusted output of the Jantz et al. (2004) model, and the HLU projections with other land uses estimated from land use ratios observed in 2000 MRLC remote sensing data.

Site	Water	Urban	Barren/ Transitional	Forest	Grasses	Wetlands
Jantz original projected "Current Conditions" Scenario (JCC)						
Site 1	1.3%	83.0%	0.7%	9.3%	3.7%	2.0%
Site 2	1.3%	82.9%	0.7%	9.4%	3.8%	2.0%
Pohick All	1.3%	77.2%	0.9%	13.3%	5.1%	2.2%
Site 20	0.0%	90.6%	0.4%	4.1%	2.1%	2.7%
Site 13 Modified	0.2%	87.9%	0.6%	6.0%	2.7%	2.6%
Site 13 Original	0.2%	88.0%	0.7%	6.1%	2.7%	2.4%
Accotink All	0.2%	87.3%	0.7%	6.5%	2.7%	2.5%
Adjusted projected "current conditions" scenario (ADJ-CC)						
Site 1	1.3%	76.1%	1.0%	14.0%	5.5%	2.0%
Site 2	1.3%	76.0%	1.0%	14.1%	5.6%	2.0%
Pohick All	1.3%	70.2%	1.2%	18.1%	6.9%	2.2%
Site 20	0.0%	83.9%	0.8%	8.3%	4.3%	2.7%
Site 13 Modified	0.2%	81.1%	1.0%	10.4%	4.7%	2.6%
Site 13 Original	0.2%	81.3%	1.1%	10.5%	4.6%	2.4%
Accotink All	0.2%	80.6%	1.2%	10.9%	4.6%	2.5%
Adjusted projected smart growth scenario (ADJ-SG)						
Site 1	1.3%	71.1%	1.2%	17.5%	6.9%	2.0%
Site 2	1.3%	71.0%	1.2%	17.5%	7.0%	2.0%
Pohick All	1.3%	64.9%	1.5%	21.8%	8.3%	2.2%
Site 20	0.0%	80.9%	0.9%	10.2%	5.3%	2.7%
Site 13 Modified	0.2%	78.3%	1.2%	12.3%	5.5%	2.6%
Site 13 Original	0.2%	78.1%	1.4%	12.5%	5.5%	2.4%
Accotink All	0.2%	77.2%	1.4%	13.1%	5.5%	2.5%
Adjusted projected sustainable development scenario (ADJ-Sust)						
Site 1	1.3%	65.6%	1.5%	21.2%	8.4%	2.0%
Site 2	1.3%	65.6%	1.5%	21.2%	8.4%	2.0%
Pohick All	1.3%	59.4%	1.7%	25.6%	9.7%	2.2%
Site 20	0.0%	77.6%	1.1%	12.2%	6.4%	2.7%
Site 13 Modified	0.2%	75.3%	1.4%	14.2%	6.4%	2.6%
Site 13 Original	0.2%	74.2%	1.6%	15.0%	6.6%	2.4%
Accotink All	0.2%	73.2%	1.7%	15.7%	6.6%	2.5%
Projected HLU						
Site 1	1.3%	61.4%	1.7%	21.5%	12.0%	2.0%
Site 2	1.3%	61.8%	1.7%	21.1%	12.1%	2.0%
Pohick All	1.3%	56.7%	1.5%	25.4%	12.8%	2.2%
Site 20	0.0%	74.9%	0.1%	14.9%	7.4%	2.7%
Site 13 Modified	0.2%	71.3%	1.3%	13.6%	11.0%	2.6%
Site 13 Original	0.2%	71.4%	1.1%	14.0%	11.0%	2.4%
Accotink All	0.2%	70.0%	1.1%	15.1%	11.0%	2.5%

5.3 Modeling Flow

As watersheds urbanize, higher peak flows and decreased groundwater infiltration result from increased impervious cover (Schueler, 1994). These changes to hydrology can lead to increased flooding, increases in sediment delivery, substantial decreases in stream base flow, and overall impairment to stream ecosystems. For this dissertation, the impacts of projected land use change on hydrology for Accotink and Pohick were forecast with both the hydrology component of the Hydrologic Simulation Fortran Program (HSPF) and the Long-Term Hydrologic Impact Assessment tool (L-THIA).

5.3.1 Flow simulations using projected land use scenarios used in the hydrology component of the Hydrologic Simulation Fortran Program (HSPF)

5.3.1.1. HSPF calibration

The HSPF model was manually calibrated from 1991 to 1993 using NLCD 1992 data. For the HSPF calibration, other modeling efforts in the Mid-Atlantic region were reviewed, including those on the Patuxent and Rappahannock Rivers included in the HSPFarm extension available in the BASINS 4.0 package. Hydrology calibration parameters were also referenced from a Total Maximum Daily Load (TMDL) for fecal coliform using HSPF, completed in 2000 for the upper Accotink watershed (Moyer & Hyer, 2000). Technical advice given in Basins technical note 6 was also examined (US EPA, 2000a). Refer to chapter 3 for additional information on the HSPF calibration.

5.3.1.2. Isolating potential impacts of land use change and caveats

The HSPF calibrated model was run under 1991 to 1993 climatic conditions using the five different projected land use components and compared to results using 1992 land use conditions. I ran this 1992 scenario in order to establish a baseline before widespread water quality BMP implementation[21]. The model was also run using 2004 land use data (HLU) in the 1991 to 1993 climatic period to explore changes that are attributable to land use change that has already occurred (not including the potential impacts of BMPs).

The 1991 to 1993 time period was selected for several reasons. First, the 1991 to 1993 time period was when the HSPF model was calibrated and the NLCD data from 1992 most reflects existing land use conditions. Second, this was before Fairfax County started requiring water quality BMPs, and the calibrated model would not be 'confused' by the impact of numerous BMPs. Third, keeping everything static except for land use change conditions allows a researcher to examine only the impacts of land use change.

Use of HSPF with the original Jantz et al. (2004) projected base scenario (JCC) land use data must be qualified. Primarily, the historical Jantz data (from years 1986, 1990, 1996, and 2000) on which the Jantz land use model was calibrated likely have a lower threshold for classifying an area as urban compared to the NLCD 1992 dataset on which the HSPF water quality model was calibrated. This means that the calibration parameters are adjusted for a mean urban land use that is "more urban" (higher density or higher impervious cover) than that in the NLCD data. As a result, these Jantz data must

[21] Water quality BMPs were not required on all new development in Accotink and Pohick watersheds until 1993.

be considered a high-end estimate of the potential range of impacts caused by forecasted land use change.

5.3.1.3. HSPF data entry

Land use change scenarios were manually entered into winHSPF for all scenarios. The model was run separately for each land use scenario and generated flow results were compared to historic land use scenarios to gain a better understanding of the full impact of urbanization on hydrology in these watersheds. Metrics compared included total annual flow volume, peak and low daily flow, flow duration curves, hydrographs, and plots of expected (modeled) vs. observed values.

The resultant modeled scenarios do not include the effects of BMPs. These BMPs were not included so that we could examine the changes in hydrology that result due to land use change without any mitigation measures. Furthermore, L-THIA does not allow the direct inclusion of BMPs at this time[22], and comparing model performance between HSPF and L-THIA is one of the goals of this project. Based on these ‘static’ hydrology projections, the impact of land use change using traditional urban designs can be explored, so that the magnitude of treatment practices necessary can be discussed. Such an approach allows the discussion of whether the method of detention and slower release to streams, which is the basis of dry detention basin design, is the best approach to protecting the Accotink and Pohick watersheds.

[22] Curve Numbers or Event Mean Concentrations can be manipulated in L-THIA to roughly simulate the impact of BMPs, thereby allowing the indirect inclusion of BMP effects.

5.3.1.4. Hydrological changes predicted by HSPF

5.3.1.4.1. Annual Flow

The modeled results show profound changes in hydrology between land uses. Site 20 at Accotink illustrates these differences well since modeled flow can be compared to observed values, however, since the watershed above Accotink 20 is currently near build-out, the changes are not as significant as watersheds with greater urbanization differential between land use scenarios. Using climatic conditions from 1991-1993, Table 5.5 shows that annual flow increases 18.7% to 25.0% in all of the future land use projections scenarios except the unadjusted Jantz data compared to modeled annual volume with 1992 land use. The increase is a substantial 32.8% for the unadjusted Jantz data, once again, with the caveat that the original Jantz data had relatively higher estimates of observed land use than the NLCD data set to which the HSPF model was calibrated. The modeled annual volume using 1991-1993 data compares well to observed annual flow for this time period, only varying between 1.3% and 5.6%. Comparing the adjusted Jantz future land use projections and the HLU projections, total annual volume had a maximum variation of 4.5%, indicating that different land scenarios are projected to play a small difference in total annual runoff volume. There is a substantial difference of 11.5% between the unadjusted Jantz CC data (which is the high land use estimate) and the HLU data (which we know is likely a low estimate).

Table 5.5. Total annual flow volume (m^3) for climatic data from 1991 to 1993 for site Accotink 20, with five different land use scenarios and observed values. Unadjusted projected Jantz "current conditions" (JCC), adjusted current conditions (ADJ-CC), adjusted smart growth (ADJ-SG), adjusted sustainable (ADJ-Sust), and HLU scenarios are all projected scenarios. The 1992 NLCD modeled scenario uses NLCD 1992 land use with the HSPF model. The observed column represents recorded values at the USGS gauge station. All else being equal, the new land use scenarios, without BMPs, would produce the total stated volume of runoff compared to the 1992 land use scenario.

Year	JCC	ADJ-CC	ADJ-SG	ADJ-Sust.	HLU	92 NLCD Modeled (baseline)	Observed
1991	2.71×10^7	2.53×10^7	2.54×10^7	2.48×10^7	2.43×10^7	2.04×10^7	2.00×10^7
1992	3.42×10^7	3.20×10^7	3.21×10^7	3.21×10^7	3.21×10^7	2.57×10^7	2.53×10^7
1993	4.43×10^7	4.21×10^7	4.22×10^7	4.15×10^7	4.08×10^7	3.57×10^7	3.39×10^7

5.3.1.4.2. Daily Flow

Annual flow is not distributed evenly across all storm events. Table 5.6 shows that the highest flow observed in the unadjusted Jantz CC scenario is about 15.7% percent higher than in the modeled scenario with 1992 land use and only about 3% higher than that of the actual 1991-1993 observed conditions at Accotink 20. These changes are not all that substantial, perhaps due to the already heavily developed status of the Accotink 20 watershed in the year 1992. More substantial changes are noted in the total number of peak flow events between the modeled scenario using 1992 land use and all projected land use scenarios, as shown in Table 5.7. The sum of events greater than 100 cf/s daily average flow and the magnitude of those events increase noticeably from the NLCD 1992 land use scenario to the JCC scenario (from 111 to 124). The number of storm events greater than 500 cf/s daily average flow increases from 4 to 11 and those events greater than 200 cf/s daily average flow increase from 27 to 51 between the NLCD

1992 land use scenario and the JCC projected scenario. There are fewer storms with between 100 cf/s and 200 cf/s daily average flow in the 1992 land use scenario compared to the JCC projected scenario. This means that storms that were producing significant runoff (defined here as enough runoff to cause a daily average flow of greater than 100 cf/s in the 1992 modeled land use scenario) are more likely to cause a rapid peak in flow volume, thereby leading to a substantial increase in conditions causing significant scouring events.

These results are particularly noteworthy for two reasons: first, the calibrated model is slightly underpredicting daily peak flow in the calibration scenario compared to observed values. Secondly, 1991-1993 were not particularly wet years: hence, the significant increase in intensity of moderate to high flow events occurred in years where massive rain storms were not the norm. What this means is that in situations where extreme precipitation would occur (which happens cyclically and is predicted to happen more frequently with climate change scenarios), the increased volume and resultant impacts on the stream channel (and perhaps flooding) will be even more significant.

Table 5.6. The five greatest average flow volumes for given days (cf/s) for three years of climatic data, with different land use scenarios and observed values at Accotink 20. Unadjusted projected Jantz "current conditions" (JCC), adjusted current conditions (ADJ-CC), adjusted smart growth (ADJ-SG), adjusted sustainable (ADJ-Sust), and HLU scenarios are all projected scenarios. The 1992 NLCD modeled scenario uses NLCD 1992 land use with the HSPF model. The observed column represents recorded values at the USGS gauge station. All else being equal, the new land use scenarios, without BMPs, would produce the average daily peak flow listed in the table.

Year	JCC	ADJ-CC	ADJ-SG	ADJ-Sust.	HLU	92 NLCD Modeled (baseline)	Observed
11/28/1993	1550	1510	1510	1490	1480	1340	1500
3/4/1993	649	602	604	588	571	442	936
3/17/1993	771	716	718	700	680	524	612
1/4/1992	853	788	790	768	744	552	570
4/16/1993	395	367	368	359	350	280	566

Table 5.7. The total number of events with average daily flows exceeding 500 cf/s, 200 cf/s, and 100 cf/s using three years of static climatic data, with different land use scenarios and observed values for the Accotink 20 watershed. Unadjusted projected Jantz "current conditions" (JCC), adjusted current conditions (ADJ-CC), adjusted smart growth (ADJ-SG), adjusted sustainable (ADJ-Sust), and HLU scenarios are all projected scenarios. The 1992 NLCD modeled scenario uses NLCD 1992 land use with the HSPF model. The observed column represents recorded values at the USGS gauge station. The observed column represents recorded values at the USGS gauge station.

Flow (cf/s)	**JCC**	**ADJ-CC**	**ADJ-SG**	**ADJ-Sust.**	**HLU**	**92 NLCD Modeled (baseline)**	**Observed**
>500	11	10	10	9	9	4	8
>200	51	44	44	41	41	27	20
>100	62	62	62	65	62	80	47
Sum	**124**	**116**	**116**	**115**	**112**	**111**	**75**

Low flow events increase more dramatically than high flow events. The total number of low flow events from 1991-1993, defined as flow at or below 2 cf/s increases from 99 observed events or 113 modeled events using 1992 land use data, to a high of

256 events in the adjusted Jantz ADJ-CC scenario and 298 events in the unadjusted JCC scenario. Table 5.8 shows that not only do the number of events increase, but the severity of those events increase. Whereas there are no modeled events with flow volume of 0.3 cf/s using 1992 land use data and only 10 monitored events, the number of these extreme low flow events increases to 31 using household method projections (HLU), 40 using the adjusted projected sustainable land use projections (ADJ-Sust), 61 using both the adjusted projected smart growth (ADJ-SG) and the adjusted current conditions (ADJ-CC) projections, and 96 extreme low flow events using the unadjusted Jantz current conditions projections (JCC).

This increase is particularly profound when one notes that the most significant model underperformance is accurately predicting low flows. The calibrated model overpredicts low flow events that comprise approximately 2% of all flow dates. This means that the actual number of low flow events that would occur in these various land use scenarios, absent BMPs that increase groundwater infiltration, is significantly underestimated by these model results. Without significant infiltration in new urban development, low flow severity and number of extreme low flow events will increase in future years. In other words, aquatic organisms at and above Accotink at site 20 will face increasing stress from extreme low flow conditions. Additionally, those organisms that can survive these conditions will have to confront more peak flow events. The changes in flow distribution in the five projected land use scenarios compared to the 1992 NLCD modeled scenario and observed values are shown in Figure 5.7.

Table 5.8. The number of events with average daily flows between 0.1 cf/s and 2.0 cf/s using three years of static climatic data, with different land use scenarios and observed values for Accotink 20. Unadjusted projected Jantz "current conditions" (JCC), adjusted current conditions (ADJ-CC), adjusted smart growth (ADJ-SG), adjusted sustainable (ADJ-Sust), and HLU scenarios are all projected scenarios. The 1992 NLCD modeled scenario uses NLCD 1992 land use with the HSPF model. The observed column represents recorded values at the USGS gauge station.

Daily Average Flow (cf/s)	JCC	ADJ-CC	ADJ-SG	ADJ-Sust.	HLU	92 NLCD Modeled (baseline)	Observed
0.1	51	5	5	0	0	0	0
0.2	29	29	29	26	16	0	2
0.3	16	27	27	14	15	0	8
0.4	17	13	15	23	13	2	4
0.5	16	13	13	13	17	2	2
0.6	17	15	14	15	17	4	2
0.7	11	14	13	14	11	7	2
0.8	12	12	13	12	15	10	2
0.9	12	9	10	9	10	5	3
1	11	6	5	12	9	9	0
1.1	19	7	9	5	11	5	8
1.2	8	9	7	4	6	9	7
1.3	13	8	10	9	7	6	5
1.4	14	12	11	9	4	11	7
1.5	10	17	18	12	12	6	10
1.6	6	17	15	11	9	8	6
1.7	10	11	10	15	12	8	5
1.8	17	10	12	16	11	4	9
1.9	2	14	13	12	11	7	6
2	7	8	8	9	15	10	11
Sum							
Less than 1 cf/s	192	143	144	138	123	39	25
Less than 2 cf/s	298	256	257	240	221	113	99

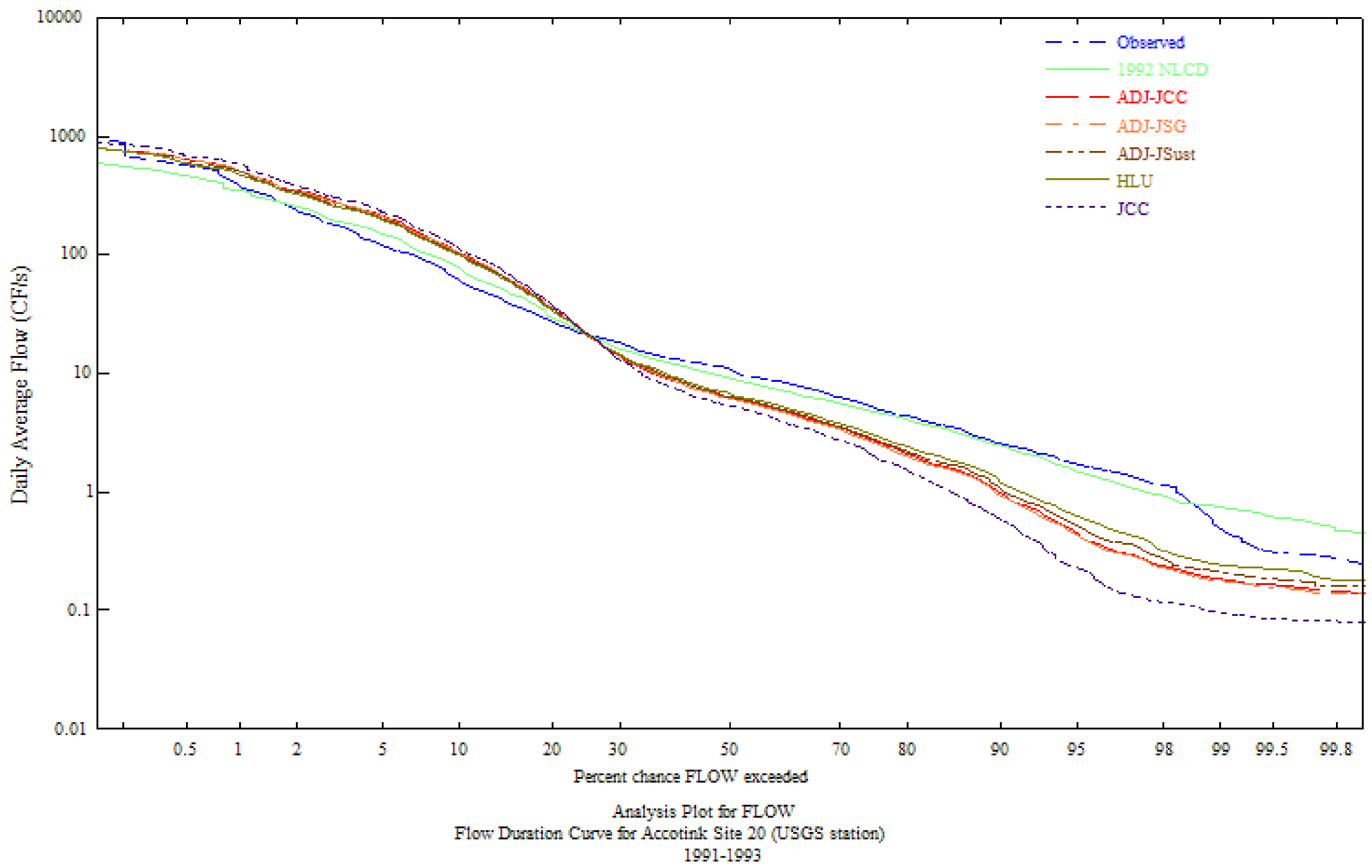

Figure 5.7. Flow duration curve for Accotink Site 20 using climatic conditions from 1991-1993 comparing observed flow (blue dot-dashed) to modeled flow using NLCD 1992 land use data (green solid) and five projected land use scenarios (adjusted Jantz projected current conditions (ADJ-JCC) - red dash; Jantz adjusted smart growth (ADJ-JSG) – orange dash dot; Jantz adjusted sustainable (ADJ-Jsust)– light brown dot-dash; HLU – olive solid; and unadjusted Jantz projected “Current Conditions” (JCC)- Purple Dot).

The Pohick watershed is projected to undergo substantially greater hydrologic changes than Accotink. The hydrograph in Figure 5.8 illustrates the substantial increase in intensity of both peak flows and low flow events between the modeled 1992 NLCD land use scenario and the adjusted projected ADJ-JCC land use scenario absent BMPs that infiltrate, reduce total volume, and retain peak flow events. The flow duration curve in Figure 5.9 shows that there is projected to be either greater or less runoff in the future adjusted projected ADJ-JCC land use scenario than in the scenario using 1992 land use (in other words, flow extremes will increase). This means that moderate flow conditions: those defined here as having a daily average flow of between 5 and 100 cf/s, decrease from a total of 74.5% of all days with 1992 land use to 58.9% of all days with the adjusted ADJ-JCC land use.

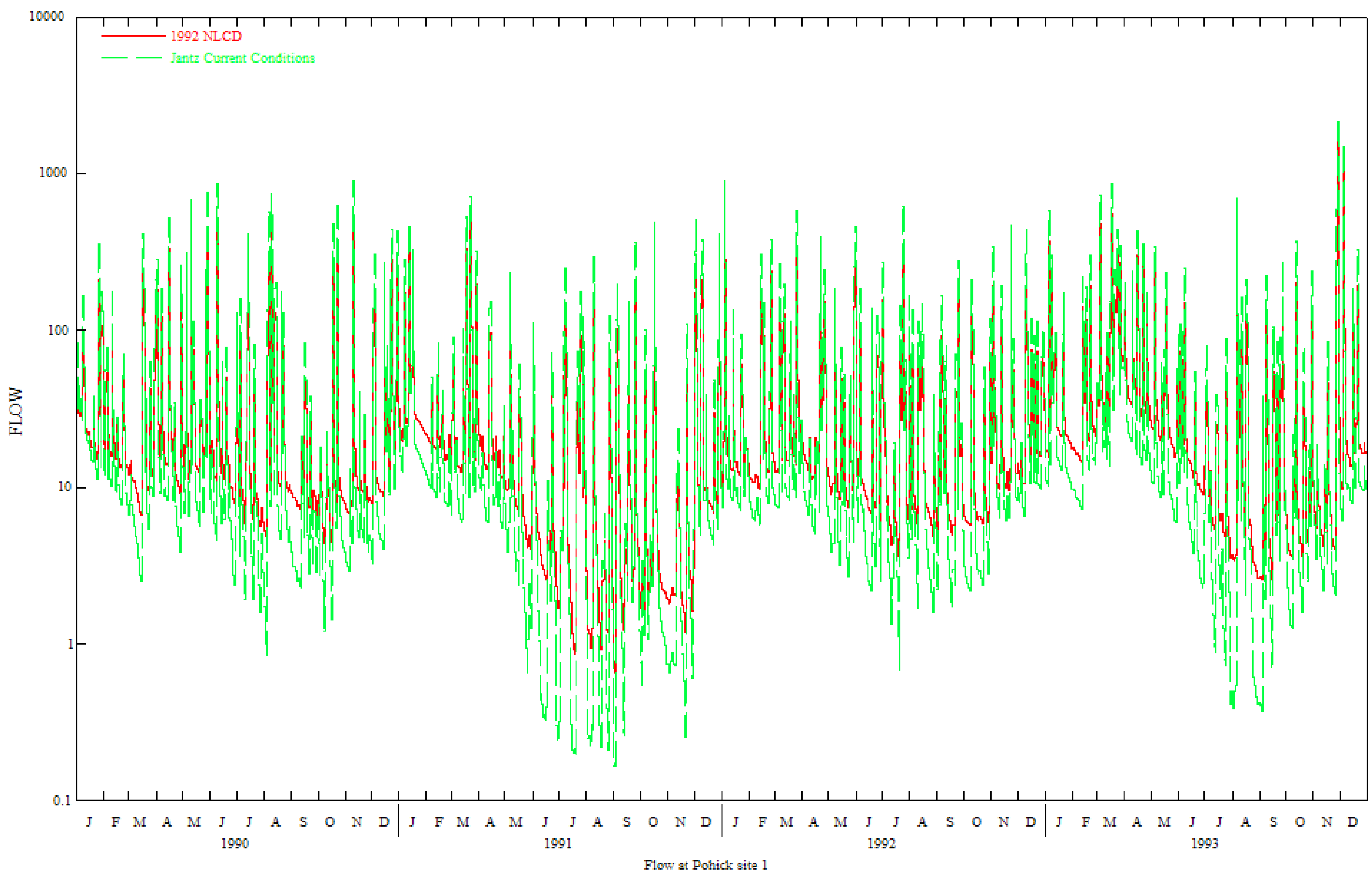

Figure 5.8. Hydrograph for Pohick 1 comparing 1992 land use conditions (red solid) to projected ADJ-JCC land use conditions in 2030 (green dashed) modified from Jantz et al. (2004).

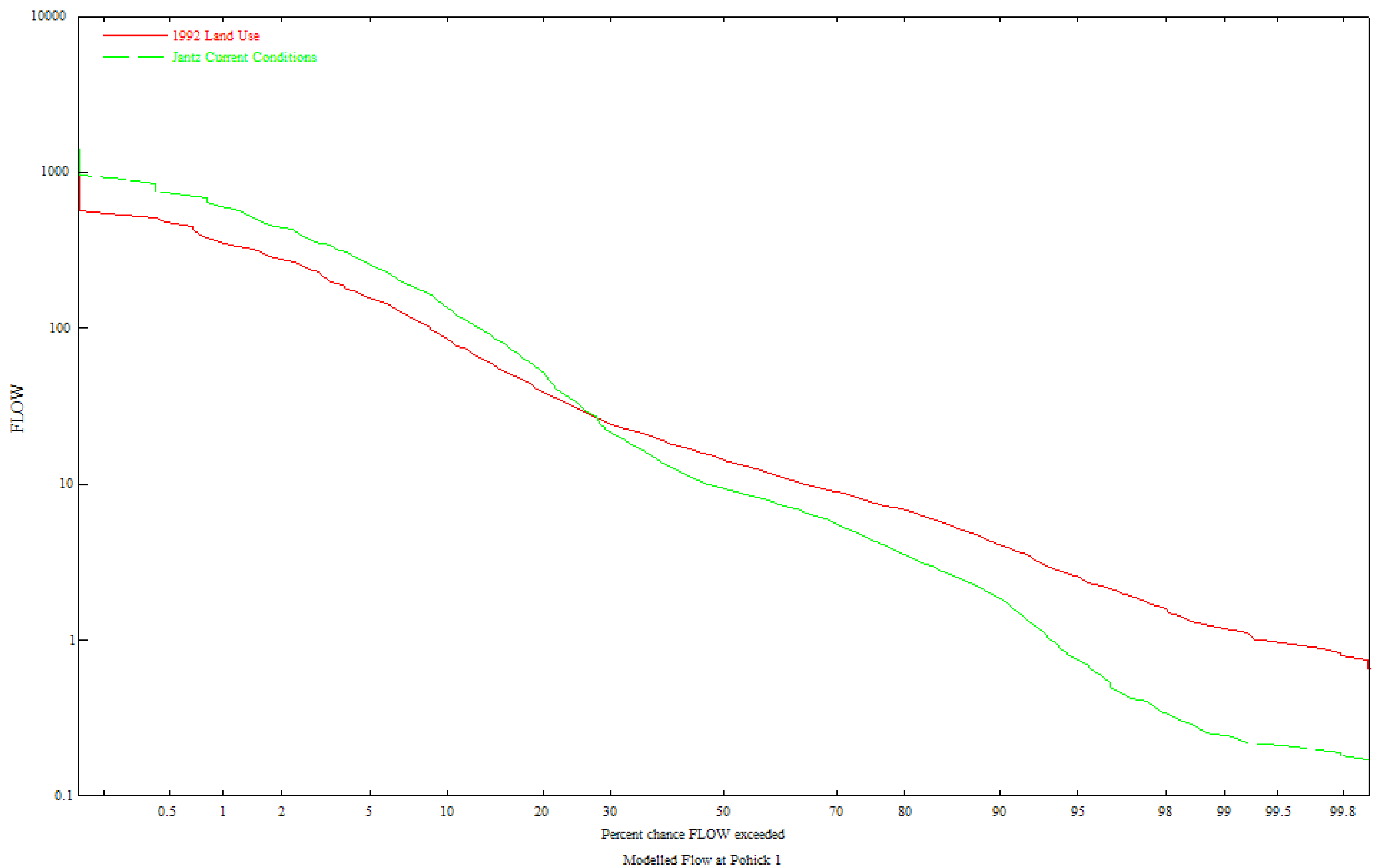

Figure 5.9. Flow duration curve for Pohick 1 comparing 1992 land use conditions (red solid) to projected ADJ-JCC land use conditions in 2030 (green dashed) modified from Jantz et al. (2004) using climatic conditions for 1991-1993.

In the Pohick watershed, different land use projections produce significantly different results compared to each other. This is mostly a result of less land being developed as of the final calibration period for the land use change models (2000 for Jantz et al. and 2004 for the HLU approach) and more undeveloped land is projected to develop. Furthermore, different land use policies result in real differences in projected land development in Pohick. Additionally, the growth that would occur in outlying Pohick regions is steered toward inner communities, such as the Accotink headwaters, in the sustainable development scenario. As can be seen in Table 5.9, annual flow volume increases from 17.4% to 45.7% between modeled 1992 conditions and the various projected land use scenarios. If we assume that the ADJ-CC is the expected scenario, annual flow volume is projected to be 38.0% higher in 2030 than in 1992. Total annual flow volume is projected to be as high as 11.3% higher in the ADJ-CC compared to the HLU projections, and as much as 8.0% higher between the ADJ-Sust projections and the ADJ-CC projections.

Table 5.9 Total annual flow volume (m^3) for three years of climatic data for Pohick site 1, with different land use scenarios and observed values. Unadjusted projected Jantz "current conditions" (JCC), adjusted current conditions (ADJ-CC), adjusted smart growth (ADJ-SG), adjusted sustainable (ADJ-Sust), and HLU scenarios are all projected scenarios. The 1992 NLCD modeled scenario uses NLCD 1992 land use with the HSPF model. This means that all else being equal, the new land use scenarios, without BMPs, would produce the total stated volume of runoff.

Year	JCC	ADJ-CC	ADJ-SG	ADJ-Sust.	HLU	92 NLCD Modeled (baseline)
1991	3.49E+07	3.31E+07	3.20E+07	3.07E+07	2.98E+07	2.41E+07
1992	4.40E+07	4.20E+07	4.05E+07	3.88E+07	3.77E+07	3.02E+07
1993	5.77E+07	5.56E+07	5.42E+07	5.25E+07	5.15E+07	4.39E+07

Variation in the number of peak flow events with daily average flow greater than 500 cf/s and 200 cf/s is noteworthy between land use projections, increasing by more than 10% from both the HLU and ADJ-Sust scenarios compared to the ADJ-CC scenario (Table 5.10). Unlike in the Accotink watershed, the ADJ-SG land use scenario produces a marginal decrease in the number of peak flow events. The number of low flow events less than 5 cf/s also decreases significantly in Pohick, particularly those extreme low flow events less than a daily average flow of less than 1 cf/s (Table 5.11). The number of those most extreme low flow events, less than 0.5 cf/s more than doubles from the HLU and ADJ-Sust to the ADJ-CC and JCC scenarios. As with peak flow, there is a noticeable difference between low flow conditions between the ADJ-SG and ADJ-CC projections. Though there is an increase in high flow and low flow events between the NCLD 1992 land use data and the HLU and ADJ-Sust scenarios, these results clearly show that modifying land use policies alone could lessen the negative impact of hydrological change on Pohick.

Table 5.10. The total number of events with average daily flows exceeding 500 cf/s, 200 cf/s, and 100 cf/s using three years of static climatic data with different land use scenarios for the Pohick watershed at site 1. Unadjusted projected Jantz "current conditions" (JCC), adjusted current conditions (ADJ-CC), adjusted smart growth (ADJ-SG), adjusted sustainable (ADJ-Sust), and HLU scenarios are all projected scenarios. The 1992 NLCD modeled scenario uses NLCD 1992 land use with the HSPF model.

Flow Range (cf/s)	JCC	ADJ-CC	ADJ-SG	ADJ-Sust.	HLU	92 NLCD Modeled (baseline)
500+	14	13	12	10	10	5
200-5499.99	79	71	67	63	59	33
100-199.99	147	134	129	124	118	89
Sum	**240**	**218**	**208**	**197**	**187**	**127**

Table 5.11. The total number of events with average daily flows within specified low flow ranges using three years of static climatic data and different land use scenarios for the Pohick watershed at site 1.

Flow Range (cf/s)	JCC	ADJ-CC	ADJ-SG	ADJ-Sust.	HLU	92 NLCD Modeled
<0.5	79	56	40	25	24	0
0.51-1.0	112	96	84	62	55	11
1.01-1.5	140	122	111	97	93	28
1.51-2.0	190	151	138	123	119	48
2.01-5.0	337	316	308	287	280	190
Sum	**858**	**741**	**681**	**594**	**571**	**277**

5.3.1.5. Hydrological change from 1975-2030

A final approach to obtaining an historical perspective on hydrologic change in Pohick is to compare the flow results using 1975 HLU land use (see chapter 3 for additional details) to the adjusted Jantz (ADJ-CC) projections for 2030 using climatic conditions from 1974-1976. The reason these analyses were performed was to examine the potential full range of development impacts from 1975 to 2030. As previously discussed, only an estimated 11.6% of the Pohick watershed was urbanized in 1975. The adjusted Jantz ADJ-CC scenario projects approximately 70.2% of the watershed will be urbanized in 2030. Results were most striking. For the model runs, the minimum flow decreased by more than 10 fold in the projected land use, the number of low flow events (less than 5 cf/s) increased by more than 30 times, the median flow value was half compared to 1975, and yet the number of high flow events more than doubled (Table 5.12 and Figure 5.10). In summary, without implementation of effective hydrologically focused BMPs (which did not occur from 1975-1992), the hydrological character of the watershed will change completely. Based on simple empirical assumptions that detention approaches are not allowing adequate infiltration and shallow groundwater recharge, a simple detention approach using dry detention basins will not significantly maintain the character of the stream and the watershed.

Table 5.12. Pohick 1 hydrologic conditions using HLU 1975 land use data and ADJ-CC projections (2030) with 1974-1976 climatic conditions. This table contains the total number of events with average daily flows within specified ranges and summary statistics.

Flow Range (cf/s)	HLU 1975	ADJ-CC
<0.5	0	8
0.51-1.0	0	23
1.01-1.5	0	43
1.51-2.0	0	67
2.01-5.0	11	213
Sum of low flow events	**11**	**354**
500+	9	28
200-499.99	39	73
100-199.99	69	133
Sum of high flow events	**117**	**234**
Minimum	3.7	0.3
1st quartile	15.8	6.275
Median	25.1	10.9
3rd quartile	35.025	32.35
Max	2130	3240

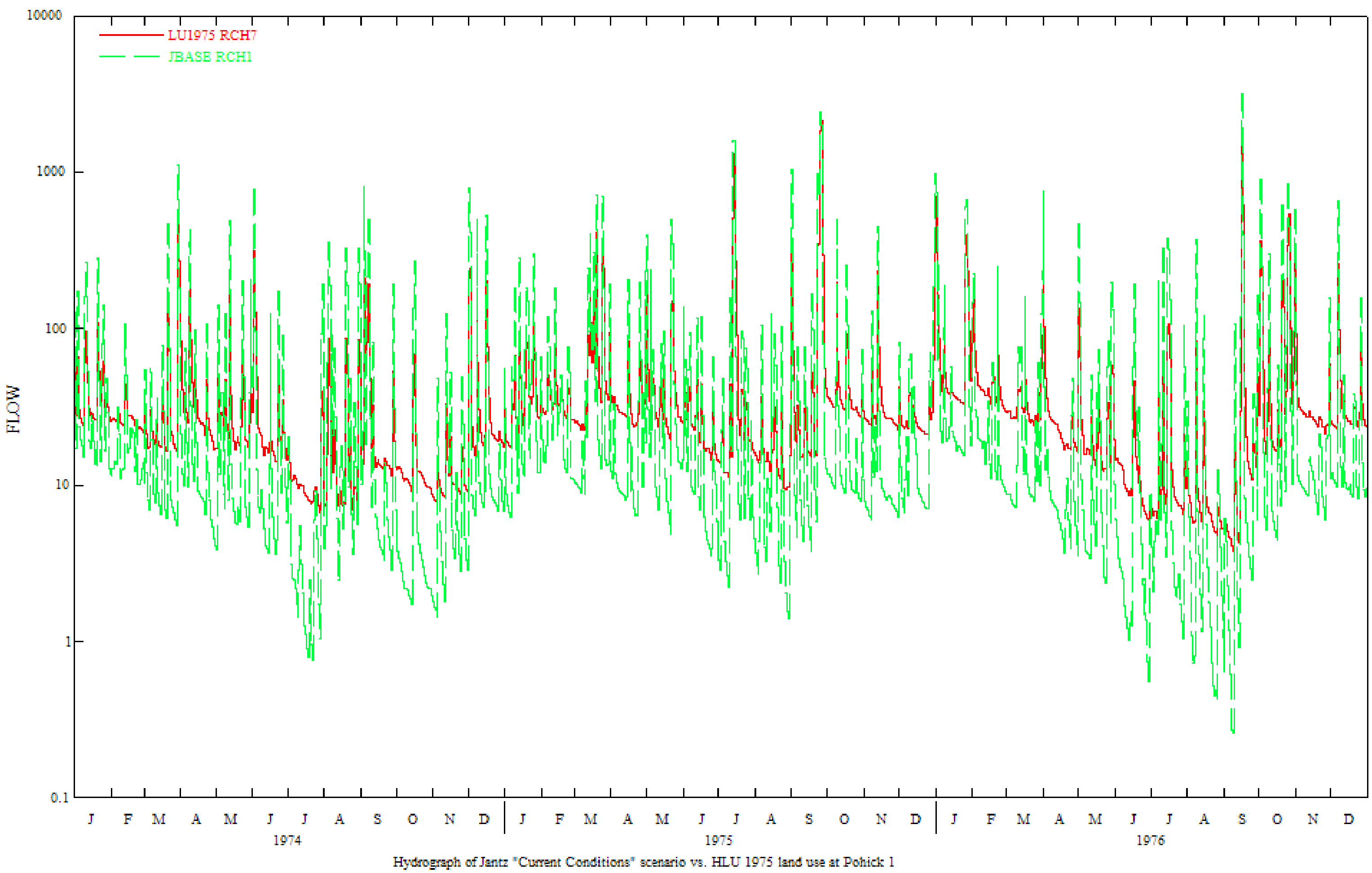

Figure 5.10. Hydrograph for Pohick 1 comparing 1975 land use conditions (red solid) to projected land use ADJ-CC conditions in 2030 (green dashed) modified from Jantz et al. (2004) using 1974-1976 climatic conditions.

5.3.2 Flow simulations with the Long-Term Hydrologic Impact Assessment (L-THIA) tool.

5.3.2.1. The Curve Number Approach

The Long-Term Hydrologic Impact Assessment (L-THIA) tool is a simple model based on the curve number method using the same procedure found in the Technical Release 55 (TR-55) approach (Bhaduri *et al.*, 1997; Bhaduri *et al.*, 2000). L-THIA is operated by Purdue University and is available as either a web-based interface or a GIS extension. As the name implies, the tool was designed to help community planners and watershed managers study the long-term impact of land use change. The web-based tool allows the user to estimate annual urban runoff, while the GIS model allows the user to estimate single storm runoff or annual runoff.

In its simplest form, the only user required input parameters for the model are land use data. In its more complex forms, the user may need to provide land use, custom curve numbers, custom event mean concentrations, and/or manipulated precipitation data (GIS model only). The curve number method was developed by the Soil Conservation Service in 1964 (National Soil Conservation Service, 2003; Soil Conservation Service, 1964). It is a simple approach generally used on smaller development sites, although in L-THIA, it is applied to larger watersheds. The curve number is a parameter that characterizes runoff response and is influenced by an area's soil type, antecedent soil moisture, land use, and impervious cover. It uses daily rainfall values to calculate flow or runoff volume depth from a given land area. The higher the curve number, the more impervious an area, and consequently, the more runoff will occur during a storm event. Flow using the curve number approach is determined by the following equation:

$$Q = \frac{(P - I_a)^2}{(P - I_z) + S}$$

where

Q = flow or runoff volume
P = the amount of runoff in inches
I_a = an initial abstraction or the amount of rainfall required to initiate runoff and
S = maximum retention after runoff begins to flow

The initial abstraction is typically set to 0.2 as recommended in its original development, which makes the equation simplify to:

$$Q = \frac{(P - 0.2S\ )^2}{(P - 0.8S)}$$

And $S = \frac{1000}{CN} - 10$

Where CN = the curve number

Curve numbers have been compiled on a variety of land use types (National Soil Conservation Service, 2003; Purdue Research Foundation, 2004). They vary from about 30 in a forested land use with more permeable soil to 95 in urban commercial districts with less permeable soil when using the initial abstraction of 2. The National Soil Conservation service (1964) catalogs soil as type A, B, C, or D. These soils are defined as:

> A. (Low runoff potential). Soils having high infiltration rates even when thoroughly wetted and consisting chiefly of deep, well to excessively drained sands or gravels. These soils have a high rate of water transmission.
> B. Soils having moderate infiltration rates when thoroughly wetted and consisting chiefly of moderately deep to deep, moderately well to well drained soils with moderately fine to moderately coarse textures. These soils have a moderate rate of water transmission.
> C. Soils having slow infiltration rates when thoroughly wetted and consisting chiefly of soils with a layer that impedes downward movement of water, or soils with moderately fine to fine texture. These soils have a slow rate of water transmission.
> D. (High Runoff potential). Soils having very slow infiltration rates when thoroughly wetted and consisting chiefly of clay soils with a high swelling potential, soils with a permanent high water table, soils with a claypan or clay layer at or near the surface, and shallow soils over nearly impervious material. These soils have a very slow rate of water transmission (Soil Conservation Service, 1964).

According to the soil classification tool in L-THIA, Accotink and Pohick watersheds are dominated by soil type B.

5.3.2.2. Curve Numbers Chosen for this Application

I assigned appropriate custom curve numbers to each land use in L-THIA. L-THIA has defaults for residential, commercial, and forested land uses etc., but it does not have defaults for Anderson level I classified land uses. Land use curve numbers for land uses in these watersheds were determined by taking the default for soil type B using a curve number of 60 for forest in fair condition, a value of 50 for wetlands and water[23], and the default value for open space (69) listed in the National Engineering Handbook (National Soil Conservation Service, 2003). The curve number for urban land use was originally determined by taking a blend of 25% residential quarter acre lots (75), 25% high density residential (1/8 acre lots)(85), 20% commercial (92), 25% paved parking

[23]L-THIA default is 0; however, it is likely that precipitation that runs directly into open water will make its way downstream in watersheds this large.

(for transportation infrastructure and parking) (98) and 5% industrial (88). This produced a weighted curve number of 87.3, which was rounded up to 88 since residential density is likely higher for portions of these watersheds. These ratios were determined based on approximate existing land use in the County as of 2004 (Fairfax County, 1975-2004).

5.3.2.3. Nutrient Loading with the "Simple Method"

This dissertation did not attempt to use L-THIA extensively to calculate nutrient loads; however, nutrient loading calculated by L-THIA were briefly noted in comparison to those produced using the export coefficient method. L-THIA calculates pollutant loadings using the simple method (Purdue Research Foundation, 2004). The simple method uses the following equation:

$$LP = \sum U(P * PJ * RVU * CU * AU * 2.72 / 12)$$

Where:
U = Land Use area in class U
LP = Pollutant load, lbs
P = Precipitation, inches/year
PJ = Ratio of storms producing runoff (default = 0.9)
CU = Event Mean Concentration for land use type u, milligrams/liter
AU = Area of land use type u, acres
RVU= Runoff Coefficient for land use type u, inchesrun/inchesrain, defined as
$RVU = .005 + (0.009 * IU)$
Where IU = Percent Imperviousness (US EPA, 2001b).

The results automatically produced by L_THIA were not analyzed in detail: however, the straightforward loading estimates produced were compared to those produced in the following section using export coefficients and the historic loadings produced by LOADEST in Chapter 4.

5.3.2.4. L-THIA Data Entry

For this project, both the GIS and web-based approaches were used in the exploration phase, but only the results from the web-based interface were used in the final phases. The GIS extension has the advantages of being able to convert spatial data and allowing the user to manipulate meteorological data. It also can be used to create visual diagrams and can calculate both annual and daily runoff. It has the disadvantages of requiring Arcview, having a maximum of 8 land use classes that must have specific names, and requiring a small to medium time commitment for the user to learn how to operate the tool. The web based extension allows manual entry of summed land use data and easily allows the user to explore what if scenarios. The SLEUTH land use projections for this project were converted to a non-orthorectified data format: hence, use of the web-based tool was deemed most appropriate for use with land use projections. If a user were to simulate climate change (by adjusting precipitation numbers), or wanted to use the spatial analysis capabilities in the GIS extension, this tool would be more appropriate.

The web-based interface has three settings: basic, detailed, and advanced. The form-based approach allows the user to simply input the amount of land use for each watershed. In the basic mode, the user inputs state and county and up to 8 default land uses, their soil type, and their area occupied in acres, hectares, square kilometers or square miles. The model then uses thirty years of climatic data, default curve number values, and event mean concentrations to give expected annual runoff depth and pollutant loadings of numerous parameters including nitrate, total phosphorus, and several metals. Additionally, the user can examine variation that would occur during high and low precipitation years in order to understand the range of impacts that could occur as a results of various land use change scenarios.

As in the basic mode, in the detailed mode the user inputs the state and county and elects how he or she will input land use area. The detailed input also allows the user to input one of 13 default land uses and create a custom land use in which the user defines a curve number. Furthermore, the user can label the custom land use as being similar to one of the 8 land uses in the basic input. Event mean concentrations are automatically assigned to the custom land use based upon user provided information. The user must identify which default land use the custom land use most resembles. The model then produces average annual runoff volume and pollutant loadings and allows the user to examine annual variability based on historic climatic data.

In addition to the capabilities in the detailed mode, the advanced mode allows the user to modify event mean concentrations for any standard land use or custom land use. The user can select what land use the custom land use resembles for default event mean concentrations or he or she can modify the event mean concentrations based on the literature or other information. Land use values were input in the detailed and advanced modes for this work, although event mean concentrations were not significantly modified since L-THIA was not used as a primary tool to examine pollutant loadings.

I input land use values into the L-THIA web form as custom land use for 7 land use scenarios. Estimated 1992 and estimated 2004 land use were used to quasi-validate the flow estimates produced compared to observed data at the USGS gauge station at Accotink and to modeled HSPF flow data. Projected land use scenarios including the HLU, ADJ-Sust, ADJ-SG, ADJ-CC, and JCC scenarios were input to examine the range of potential changes that could result in these projected land use scenarios.

5.3.2.5. Hydrologic Changes Predicted by L-THIA

L-THIA clearly shows an increase in annual direct runoff from 1992 conditions to the various projected land uses. Table 5.13 shows the estimated average annual flow using 30 years of climatic data for Pohick site 1, and minimum and maximum flow (low and high precipitation years respectively). High and low range years were determined by looking at average annual runoff depth graphs for high and low years in the model output and multiplying times watershed area. According to the model, in high runoff years, these values could be twice as high; in low flow years they could be half. The model output clearly shows that annual direct runoff will increase as a result of urbanization.

Table 5.13. Pohick 1 annual runoff volume in meters cubed in five projected land-use scenarios and two baseline scenarios as predicted by the L-THIA model. Unadjusted projected Jantz "current conditions" (JCC), adjusted current conditions (ADJ-CC), adjusted smart growth (ADJ-SG), adjusted sustainable (ADJ-Sust), and HLU scenarios are all projected scenarios. The 1992 NLCD modeled scenario uses NLCD 1992 land use with the L-THIA model. The 2004 HLU land use establishes an alternate baseline, using conditions that assume there were no BMPs that contributed to evaporation or infiltrations in the years between 1992 and 2004 for new development[24].

Annual Flow Volume (m3)	JCC	ADJ-CC	ADJ-SG	ADJ-Sust.	HLU	2004 HLU	1992 NLCD Modeled
Average	1.60×10^7	1.49×10^7	1.41×10^7	1.33×10^7	1.27×10^7	1.16×10^7	8.85×10^6
Minimum	1.13×10^7	8.34×10^6	7.92×10^6	7.50×10^6	6.68×10^6	6.25×10^6	5.22×10^6
Maximum	3.11×10^7	2.67E x 10^7	2.61E x 10^7	2.54E x 10^7	2.50E x 10^7	2.29E x 10^7	2.09E x 10^7

5.3.2.6. Comparing L-THIA to HSPF Results and Observed Conditions

The HSPF and L-THIA output both show annual flow increasing. HSPF simulates total flow while L-THIA only simulates direct runoff. In other words, L-THIA does not take into account groundwater recharge and interflow; hence, the model does not capture all of a streams flow. Logically, HSPF and L-THIA differ in magnitude of flow volume predicted. L-THIA underpredicts flow at site 1 by about 65% to 73% in all scenarios relative to HSPF. Even after accounting for only surface flow in HSPF, the L-THIA underprediction is about 60-70% (comparing surface flow runoff in inches generated by HSPF to runoff results generated by L-THIA). Furthermore, the total flow volumes predicted in HSPF are not captured by the minimum and maximum range inferred from the model. Similar results are noted for Accotink site 20 at the USGS gauge station: L-THIA predicts annual average direct runoff volume of 8.82×10^6 m^3 for

[24] Some BMPs that allowed some evaporation (such as wet retention ponds) or infiltration (such as grass swales) clearly were constructed, however, the influence of BMPs is ignored in this analysis.

1992 land use and 9.98 x 10^6 for 2004 land use. These values underpredict both the modeled 1992 HSPF values and the observed USGS gauge values by about 60 percent (these values were not adjusted to account only for surface runoff). Surface runoff alone at site 20 was estimated to be an average of 33.4 cm/year (13.15 inches/year) for the HSPF model during 1991-1993, compared to 10.8 cm/year (4.25 inches/year) predicted by L-THIA. For the 1992 land use scenario, the maximum value in the range, about 1.52 x 10^7 m^3, does not capture the observed or HSPF flow estimates for any year in the early 1990s.

The relative increase of average runoff at site 1 estimated from the NLCD 1992 to ADJ-CC projections increases by about 68% compared to an increase of 38% in average flow in HSPF. For site 20, that increase is about 44% compared to about 20%. Hence, because of the large difference in curve numbers between land uses, and the fact that L-THIA is looking only at direct surface runoff, L-THIA is predicting larger differences in runoff volume with changing land use than is predicted in total annual volume with HSPF or observed annual flow at the USGS gauging station. The disparity between L-THIA and HSPF values is greater if the default curve numbers are used for water (0), forest (55), open space (69) and ¼ acre residential (75). For instance at site 1, L-THIA predicts a relative increase in annual runoff of 51% from the 1992 to the projected ADJ-CC land use scenario. This compares to the 68% difference with the customized curve numbers, meaning that L-THIA is producing even lower annual runoff estimates.

5.3.2.6.1. Reasons for L-THIA underperformance

This underprediction is due to L-THIA output not generating enough runoff that reaches the modeled sites in the watershed. The additional magnitude of runoff can be generated so that results are consistent with the literature, but changes in assumptions must be made. There are three reasons for L-THIA underpredicting runoff compared to flow volumes for HSPF and observed flow. The first, as already discussed, is that L-THIA is only considering surface runoff, whereas HSPF and observed values[25] are total flow. The second reason is that the initial abstraction, set at 0.20, may be more appropriate at a lower number. Lim et al. (2006a) lowered the initial abstraction to 0.05 using the curve number approach. The authors found that the 0.05 abstraction produced more runoff and more accurate results. The third reason is that soils may not be properly accounted for. The HSPF calibration for this dissertation (chapter 3) used soil infiltration rates that are consistent with soil type C (forest) to D (urban). This makes logical sense as areas where there has been constructed often have compacted soils due to the grading and construction process (even in the pervious portions). The soil type listed in the state soil geographic (STATSGO) database list the soils in Accotink in Pohick as type B. However since urban soils undergo significant alteration and compaction: a less pervious soil classification may be appropriate. Lim et al. (2006a) used this logic to adjust their soil types to raise the curve numbers of land use in their model, and found that doing so improved model performance considerably.

[25] It is possible to use methods to manipulate the observed data to separate direct runoff from the flow, however, that analysis was not completed as part of this dissertation. Future research may include such analysis.

5.3.2.6.2. Improving Performance by Adjusting Curve Numbers

The L-THIA model is extremely sensitive to curve numbers: in the advanced modes, the user can estimate them based on land use conditions and soil type. For the runoff volumes to be in the same range, curve numbers for each land use must be elevated. This can be done by assuming that each land use has a higher curve number based on local watershed conditions such as increased compaction of soils or higher percentages of impervious cover. As previously discussed, soil class B may be inappropriate label for these watersheds, and soil class C or D could be more appropriate. For instance, if we assume that soil class C more accurately represents local conditions, we would increase the curve numbers of forest in fair condition to 73, open space to 79, and urban would use an averaged value of 92 (25% residential quarter acre lots or low density (80), 25% high density residential (1/8 acre lots)(90), 20% commercial (94), 25% paved parking (for transportation infrastructure and parking) (98) and 5% industrial (91)). At site 1, these values increase the estimated annual runoff to 1.44×10^7 m^3 for 1992 conditions to 2.24×10^7 m^3 for the ADJ-CC scenario or 2.37×10^7 m^3 for the projected JCC scenario. Based on the 1992 HSPF baseline model output, the L-THIA model still underpredicts annual flow, but this underprediction is lessened significantly to approximately 48-55%.

5.3.2.7. Appropriateness of using L-THIA

When comparing L-THIA to HSPF or observed flows, the L-THIA model gives reasonable values that are within the same order of magnitude as observed conditions or to those produced by HSPF, but the model underpredicts total runoff volume. This underprediction could be due to a number of causes: the relative large size of these watersheds causes underprediction or the model is not accounting for shallow water groundwater recharge to the streams (effectively only looking at runoff volume). These results show that L-THIA can be used as a planning tool or screening tool when looking at hydrologic changes caused from land use change because the model generates *plausible* results. The relative increases in runoff volume between land use scenarios are reasonable and the estimates of runoff volume generated are within the same order of magnitude as HSPF and observed flow values. However, the user must be cautious that curve numbers accurately reflect watershed conditions. Furthermore, the user must understand that the output only includes direct runoff and not flow, and so the user is only grasping a subset of hydrologic changes in the watershed (the user cannot estimate changes in base flow). Furthermore, if the user seeks to use L-THIA for more complicated analysis, for instance, studying mass balance of annual water flow over time, then the model could be stretched beyond its limitations. Hence, L-THIA appears to be a valid tool when used for its intended purpose as a screening tool for studying the impact of potential land use changes on watershed hydrology, but its accuracy and range of results are lower than that of HSPF.

5.4 Modeling Nitrogen Loadings to Gunston Cove

Nitrogen is the primary nutrient of concern associated with eutrophication in the mesohaline portions of the Chesapeake Bay. As discussed in chapter 4, historical loadings of nitrate-N from precipitation driven pollution have apparently increased from the 1987 to 2005 period. Export coefficients were used in combination with five land use projections and historical land use estimates to examine how land use change in these watersheds could impact nitrogen loading to Gunston Cove.

5.4.1 The Export Coefficient Method

The export coefficient method was based on that used in the model PLOAD. PLOAD is a simple loading model whereby annual pollutant loadings can be calculated by export coefficients or using the "simple method" discussed above with L-THIA. The model has been used to look at loadings of total suspended solids, nitrate, ammonia-N, BOD, total Kjehldahl nitrogen, phosphorous, fecal coliform and some metals such as lead and zinc. For example, Cui *et al.*(2003) used the model to evaluate loadings from numerous pollutants in the Xinshan sub-watershed of the Taihu watershed in China using a GIS interface. The PLOAD model uses GIS land use data, GIS watershed data, pollutant loading rate data tables, and impervious terrain factor data tables, in addition to optional inputs of BMP site and area data, BMP pollutant reduction tables, and point source location and loads (US EPA, 2001b). For data that is not available in an explicitly spatial format (i.e., land use data by subwatershed as calculated for 4 of 5 land use scenarios), the export coefficient method can be easily applied in a spreadsheet model or similar program. PLOAD is simplistic and designed so that it could be used for a wide variety of purposes as a watershed screening model.

The PLOAD model requires a delineated watershed and land use data. In its calculations, the model overlays the watershed delineation with the land use data to calculate the total of each land use in the watershed. The pollutant loading rates, impervious cover factors, and BMP efficiency rates are put into tabular files for use in PLOAD. Pollutant loading tables include the export coefficient and the event mean concentration. The impervious factor table includes the relative percentage of imperviousness for each land use type, and the BMP table identifies the effectiveness of various BMP types. The PLOAD model calculates all results in English units (lbs/acre). The export coefficient method uses the following equation:

$$LP = \sum U(LPU * AU)$$

Where: LP = Pollutant load, lbs;
LPU= Pollutant loading rate for land use type u, lbs/acre/year; and
AU = Area of land use type u, acres (United States. Environmental Protection Agency, 2001).

Relative to many modeling techniques, both the “simple method” and the “export coefficient” method are relatively easy to execute using existing information. It is possible to refine the coefficients of the various parameters to aid in matching output to observed data and customize the model to each watershed. The export coefficient approach is a good screening method to use in conjunction with other techniques because of its flexibility and ability to customize, the coarse resolution of its output and the consequent low sensitivity to individual events. Furthermore, this approach via the PLOAD model is readily available in the BASINS package and is available to the general public and watershed managers free of charge.

5.4.2 Determination of Export Coefficients

As described earlier, the land use projections were summed and tabulated by subwatershed; however, they were not left in a raster spatial format. Hence, after using PLOAD to generate estimated loadings for 1992 and 2000, a spreadsheet model using export coefficients was created. The user is required to input the type, quantity, and export coefficient rates of each land use. A range of export coefficient values were used to capture annual and spatial variability of nutrient loadings. Therefore, four export coefficient rates were determined: a high rate, an average year rate, a Chesapeake Bay program modeled rate, and a low rate.

The high rate is needed to capture years with higher precipitation, higher atmospheric deposition, and/or higher loading rates from anthropogenic activity or natural variability. With the impact of climate change, more frequent, intense, and warmer storms may also result in higher rates of nitrogen loading. The low rate is needed to capture dry weather **y**ears or other natural and anthropogenic activities that result in lower nitrogen export. The "average year" conditions were selected to simulate what should be the average export for normal climatic conditions, without significant changes in human behavior or extremes in natural variability. These loading rates for each land use were determined primarily by values from the literature (Beaulac & Reckhow, 1982; CH2M Hill, 2000; Dodd *et al.*, 1992; Hodge & Armstrong, 1993; Lin, 2004; McFarland & Hauck, 1998; Reckhow *et al.*, 1980; US EPA, 2001b),. Finally, export coefficients from the Chesapeake Bay HSPF based model were used to produce an alternative 'average year' conditions (Chesapeake Bay Program Office, 2006). The logic behind why each of these different loading values is needed is presented Figure 5.11. Furthermore, each of these four scenarios are further discussed in sections 5.4.2.1 through 5.4.2.4. Final export coefficient values used with land use scenarios are presented in Table 5.14.

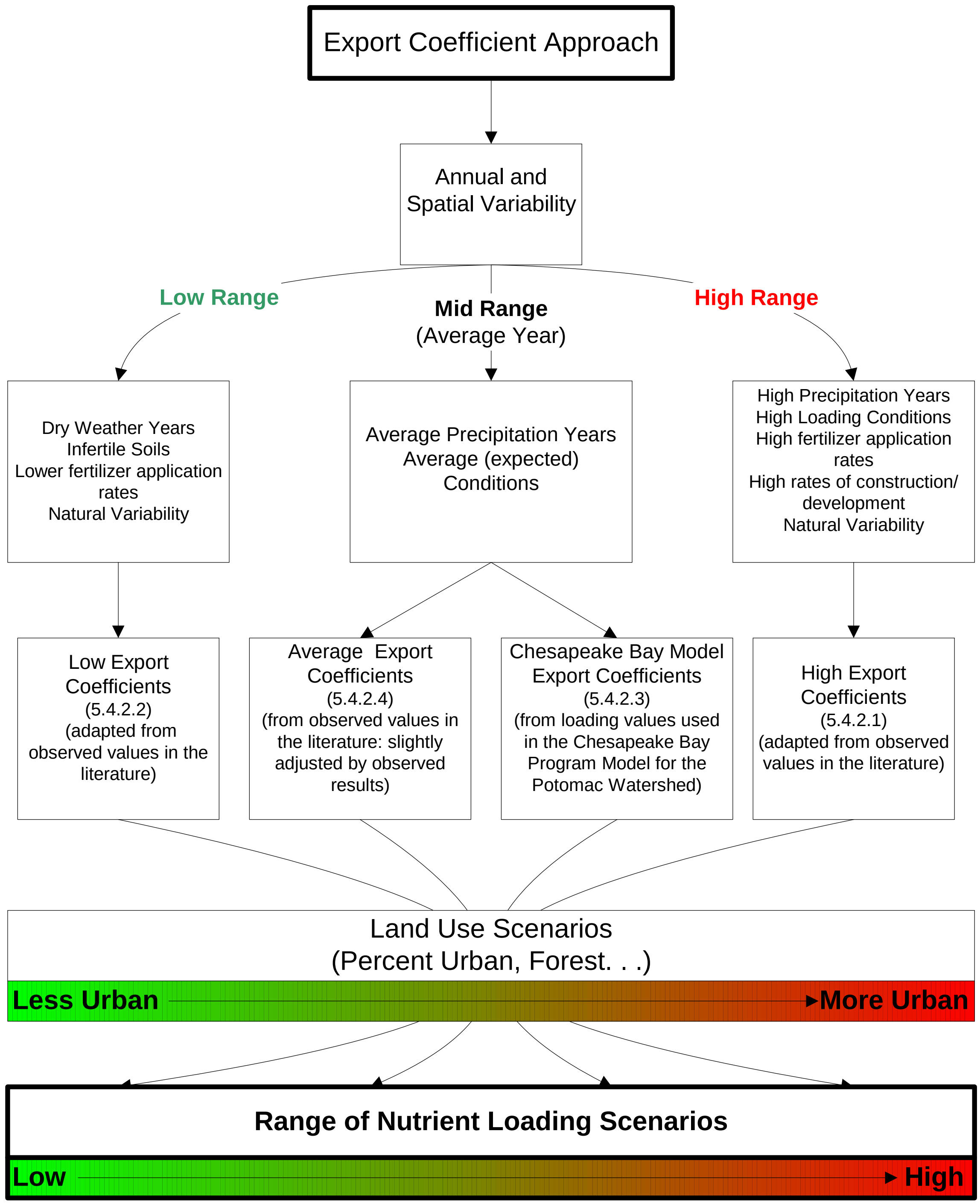

Figure 5.11. The logic behind why a range of export coefficients are used and how they are applied.

Table 5.14 Export Coefficients in kg/ha used for Accotink and Pohick in four scenarios for Total Nitrogen (TN), Nitrate-N (NO_3), and Ammonia-N (NH_3).

		Water	Urban	Barren/ Transitional	Forest	Grasses	Wetlands
TN (kg/ha)	Average	2.242	9.011	9.011	1.681	2.242	0.560
	High	10.188	16.028	16.028	2.802	8.967	2.802
	CBP	8.074	6.076	6.076	1.081	3.760	1.081
	Low	0.000	5.604	3.362	0.695	1.681	0.000
NO_3-N (kg/ha)	Average	1.121	4.506	4.506	0.841	1.121	0.280
	High	7.234	11.380	11.380	1.989	6.366	1.989
	CBP	5.639	4.209	4.209	0.606	2.647	1.081
	Low	0.000	2.802	1.345	0.278	0.672	0.000
NH_3-N (kg/ha)	Average	0.224	0.451	0.451	0.084	0.112	0.028
	High	1.019	2.404	2.404	0.140	0.897	0.280
	CBP	0.725	0.390	0.390	0.038	0.169	0.038
	Low	0.000	0.280	0.168	0.017	0.084	0.000

5.4.2.1. The High Export Coefficient Rate

Scenarios with high export coefficients should capture the high-end annual loading coming from a watershed during a given year. These higher loadings could result from higher precipitation values[26], higher relative loading per land unit in one watershed versus another, or other conditions. The high value export coefficients for urban, forest, and grasses were determined directly from values from the literature (Frink, 1991; Reckhow *et al.*, 1980). For example, the high total nitrogen export coefficient used was determined as one standard deviation above the mean given by Frink (1991). The high scenarios export coefficient for barren and transitional was assumed to be the same as for urban since some areas may be truly barren and have lower nitrogen export and some areas may be construction sites and have higher export levels. In the high scenario,

[26] Though precipitation is not directly included in the model, it plays a major role in nutrient export, and consequently is reflected in export coefficient rates.

wetland export was assumed to be the same as for forests since no published export coefficients for wetlands were found. This assumption seems reasonable since wetlands, like forests, are generally undisturbed areas. Though nitrogen is often denitrified in wetland environments, some of the wetlands in Accotink and Pohick likely denitrify similar quantities of nitrogen as riparian forests due to their disconnectedness from the stream. The water value was estimated by assuming that the water body would export 100% of atmospheric deposition of nitrogen.

No export coefficients for nitrate-N or ammonia-N were found in the literature. High values for nitrate-N were estimated by assuming that 71% of total nitrogen was exported in the form of nitrate. This ratio was determined by exploring reported event mean concentrations to export coefficients of total nitrogen and nitrate-N in the literature and the Chesapeake Bay program model output: 71% was one of the higher ratios found (CH2M Hill, 2000; Chesapeake Bay Program Office, 2006). High value export coefficients for ammonia-N were determined for urban and barren/transitional land uses by using 15% of the total nitrogen value, 10% of TN was used for wetlands, grass, and water, and 5% was used for forests. Once again, these values were determined by looking at event mean concentration to export coefficient ratios in the literature (CH2M Hill, 2000; Dodd *et al.*, 1992; Line *et al.*, 2002; Smullen *et al.*, 1999; US EPA, 1983). Best professional judgment and empirical assumptions were also used to estimate how much of the total nitrogen values consisted of ammonia-N and nitrate-N for each land use. For instance, organic nitrogen export would be higher from forested areas than from urban areas, and so ammonia-N and nitrate-N values are lower relative to total nitrogen.

5.4.2.2. The Low Export Coefficient Rates

Loading using low export coefficients should reflect scenarios in which there is lower annual precipitation, lower loading due to watershed conditions or behavior of inhabitants, or other conditions. Low export coefficient values were determined in a similar fashion to high export values, with the exception of using lower values in the range in the literature for nitrogen for urban, grasses, and forest. Additionally, barren/transitional was set lower than urban, thereby assuming that these lands are effectively less fertile, little overland runoff is occurring, and they are not being fertilized. Low export coefficient values for nitrate-N were estimated as 40% of the TN values for all land uses except urban, which was assumed to be 50% of TN in the low scenario. Ammonia-N was set at 5% of TN from all land uses except forested areas where it was assumed to be 2.5%. These values reflect the lower end of event mean concentration ratios between ammonia-N and TN or Total Kjeldahl Nitrogen (TKN) (Dodd *et al.*, 1992; Line *et al.*, 2002; Smullen *et al.*, 1999; US EPA, 1983). Water and wetlands were both listed as 0, assuming that they denitrified 100% of all nitrogen added to the system from the atmosphere; hence, they contributed no net nitrogen loading.

5.4.2.3. The Adapted Chesapeake Bay Program Export Coefficient Rates

The Chesapeake Bay Program values are adapted from the export coefficients used in the Chesapeake Bay Program HSPF model version 4.3 for the Potomac watershed

(Chesapeake Bay Program Office, 2006). This model scenario generated total loading by land use for areas both above and below the fall line for forested, impervious urban, pervious urban, mixed open, pasture, high and low till agriculture, and deep water. The total area in each land use above and below the fall line was divided by the total acreage in the watershed to generate export coefficients in the model. The calibrated HSPF model for Accotink and Pohick used in this dissertation, assumed that 50% of urban area was effective impervious area, while 50% was pervious. Hence, the loading rates of these two areas were averaged. Finally, the final export coefficient was calculated by averaging the export coefficient above the fall line and below using an 80/20 ratio for each land use. The mixed open value was used for grasses (open area), and deep water was used for water. There are shortcomings to using this open water value (namely, it is assumed that little of the atmospheric nitrogen deposition is transformed before being exported). However, these values were still below the high end estimates and assumed to be reasonable. The forest value was used for both forest and wetlands.

5.4.2.4. The "Average Year" Value Export Coefficient Rates

Like other values, the "average year" export coefficient values were generated by examining the mean values of total nitrogen loading in the literature (Beaulac & Reckhow, 1982; CH2M Hill, 2000; Dodd *et al.*, 1992; Lin, 2004; McFarland & Hauck, 1998; Reckhow *et al.*, 1980; US EPA, 2001b). After they were set, they were adjusted based on nitrate-N loadings in the actual watersheds. Mean export coefficient values for all land uses in the Accotink and Pohick watersheds for all land uses were referenced.

Accotink annually exported a mean nitrate-N load of 3.19 kg/hectare (2.84 lbs/acre) while Pohick exported a mean of 2.67 kg/hectare (2.38 pounds/acre) based upon results from the LOADEST model in chapter 4. Several multiple regressions examining the coefficients that contributed to nitrate-N loading were run examining most land use types and how they related to the LOADEST estimated annual loading of nitrate-N from 1987-2005. These regression models included examining various components of land use components, flow, and a dummy variable for site. Care had to be taken when interpreting coefficients because the land use data resulted in some multicollinearity; hence at least one land use was excluded for all regressions. Two of the regressions created coefficients for urban land use that were consistent with the literature: one that examined urban, forested, grassed, and transitional land use with flow and a site dummy and one that examined urban, forested, grassed, and transitional land use and wetland area. These coefficients amounted to export coefficients for nitrate-N of 4.51 kg/ha and 4.34 kg/ha (4.02 lbs/acre and 3.83 lbs/acre), both of which were consistent with the Chesapeake Bay Program estimated loadings of 4.21 kg as N/ha (3.76 lbs as N/acre) nitrate-N. Land use in watersheds were not uniform enough to produce proper export coefficients for all land use types by this method, nor were there enough sites to definitively determine export coefficients for the area. Hence, these tests were completed more as a way of adjusting export coefficient values from the literature than creating new export coefficients. The export coefficient of 4.02 chosen for nitrate-N in the average value is consistent between two regressions and those from the Bay program. The “average year” value for total nitrogen (TN) was then determined as 2 times the nitrate-N loading value and the average value of ammonia-N was considered as 5% of TN, based on ratios between event mean concentration of nitrate-N and total nitrogen.

5.4.3 Projected Nitrogen Loadings from the Noman Cole Sewage Treatment Facility

Annual historical nitrogen loading from the Noman Cole sewage treatment facility was estimated for 1984 to 2005 using data from the Chesapeake Bay Program and EPA's Permit Compliance System Database (Chesapeake Bay Program, 2006; US EPA, 2006b). The Chesapeake Bay Program had data available from 1984-2004; data from 2005-2006 were obtained from the PCS system. Annual loading rates were calculated by multiplying average monthly concentration times discharge volume per day times days in the month summed over 12 months.

Three scenarios (high, middle, and low) were calculated for future loading rates from the Noman Cole facility (Table 5.15). These scenarios were estimated looking at ranges of flow and average concentration scenarios. As of October 2006, the greatest average daily volume for any month at the POTW was 51.74 mg/d (June 2003). Values between 2003 and 2006 averaged 42.71 mg/d. In the low scenario, it was assumed that significant water conservation measures allow the average value to increase to 51.74 mg/day. The facility recently underwent an upgrade to support treating 67 mg/d. In the middle scenario, water conservation approaches are expected to somewhat limit the increase in treatment needs, and the average discharge is expected to increase to 65 mg/d[27]. In the high scenario, it is assumed that average water discharge increases to 90 mg/d.

Noman Cole's National Pollution Discharge Elimination Permit and historical discharge concentrations were referenced for determining appropriate nitrogen loadings for the three scenarios (Chesapeake Bay Program, 2006; US EPA, 2006b; Virginia Department of Environmental Quality, 2003). The current permit places an effluent limit of 1.0 mg/L ammonia-N for April to October and 2.2 mg/L for November to March. There is no current effluent limit for nitrate-N or total nitrogen. In the high scenario, it was assumed that the average discharge concentration for nitrate as nitrogen would be 5.0 mg as N/L (remaining at approximately 2005 conditions), an average of 1.6 mg as N/L of ammonia-N (just below existing permit limits), and an average total nitrogen discharge of 7.0 mg/L. In the middle scenario, VA DEQ is expected to issue a more stringent NPDES permit in subsequent years with an effluent limit of 3.0 mg as N/L for nitrate-N. Furthermore, the Noman Cole facility is expected to keep up their current level of treatment for ammonia-N, averaging approximately 0.2 mg/L. Hence, the facility is expected to discharge an average of 2.8 mg as N/L nitrate-N, 0.2 mg as N/L ammonia-N, and 3.3 mg/L TN. For the low scenario, the facility would continue its high performance months for ammonia-N removal (average concentrations have been as low as 0.01 mg as N/L) and discharge an average of 0.8 mg as N/L, nitrate-N concentration would average 1.8 mg as N/L and total nitrogen would be 2.0 mg as N/L. Hence, to generate the low loading scenario, the low flow scenario was multiplied times the low concentration

[27] Average values so close to a capacity of 67 mg/d are unlikely due to variability in the daily flows without a small facility upgrade.

scenario: likewise to generate the high loading scenario, high flow was multiplied times high concentration. This approach allows the representation of potential loading in the full range of conditions, while still emphasizing loading that will occur in the middle scenario. These results are presented in Table 5.15.

Table 5.15. Estimated nitrogen loading from the Noman Cole Sewage Treatment Plant in three Scenarios.

Scenario	Nitrate-N (tonnes)	Ammonia-N (tonnes)	TN (tonnes)
High	597.29	199.10	871.05
Middle	**251.64**	**17.97**	**296.57**
Low	128.77	5.72	143.07

5.4.4 Estimated Nitrogen Loadings using the Export Coefficient Approach

5.4.4.1. Validation of the Export Coefficient Approach using Historic Land Use

Nitrate-N and ammonia-N loadings using the four export coefficient scenarios (Low, CBP, average, and high) were compared to calculated loadings using the LOADEST model results from chapter 4 for 1992 and 2005 (Table 5.16 (nitrate-N) and Table 5.17 (ammonia-N). This analysis was performed to assure that the export coefficient produced plausible result compared to other estimated loadings using the LOADEST model (results from chapter 4).

5.4.4.1.1. Historic Nitrate-N loadings

The LOADEST annual loading estimates for 1992 were near the low range of the predicted loadings using export coefficients for nitrate-N. This is expected as 1992 was a somewhat drier year, and LOADEST loading was likely underestimated as samples were only taken during low flow days (Figure 5.12). Figure 5.13 shows the loading rates of nitrate in lbs/acre[28] generated by PLOAD by subwatershed. The export coefficient approach predicts nitrate-N loadings that range from 1.17 lbs as N/acre in the most undeveloped portions of the Pohick watershed to 2.62 lbs as N/acre in the most developed portions of Accotink. These values are slightly higher than the LOADEST ULU values and LOADEST default values. For example, in 1992, PLOAD predicts nitrate-N export loading rate 2.12 of lbs as N/acre using the average export coefficient rate and 1.42 lbs as N/acre using the low export coefficient rate for Pohick above site 1, compared to 1.25 lbs as N/acre in the LOADEST ULU output and 1.54 lbs as N/acre in the LOADEST default model (Figure 5.12). In contrast, 2005 LOADEST values are closer to the higher range of those predicted using the export coefficient approach as shown in Table 5.16. This is expected since 2005 was a wet year and there were many large intense storms that were sampled to produce the LOADEST loading estimates. In the watershed above Pohick above site 1, the LOADEST ULU model predicted a loading of 5.20 lbs as N/acre and the LOADEST default model predicted a loading of 7.27 lbs as N/acre. This compares to an average loading rate of 2.64 lbs as N/acre in the 'average year' scenario and 7.03 lbs as N/acre in the high scenario. In summary, both the 1992 and 2005 results show that the loadings estimated by the export coefficient approach are

[28] For PLOAD projections, the values are always displayed in lbs/acre.

within the same range as those by the more complicated LOADEST model, provided that a range of possible export coefficient loading scenarios are used to capture potential annual variability.

Table 5.16. Annual estimated nitrate-N loading in metric tonnes as N at 4 sites for 1992 and 2005. The LOADEST loading was calculated using multiple regression approaches in the LOADEST program (chapter 4) and is based on actual climatic conditions for the given year. The low, CBP, average, and high scenarios were calculated using export coefficients, land use estimates for the given year, and the actual point source (Noman Cole Facility) contribution for site 2. Loadings include the entire watershed upstream of each site.

Site	1992					
	LOADEST		Export Coefficients			
	(ULU)	Default	Low	CBP	Average	High
Site 1	11.50	14.17	10.62	19.13	19.31	51.24
Site 2	89.59	93.97	90.73	99.45	99.63	132.41
Site 20	N/A	N/A	10.01	16.29	17.24	44.47
Site 13	22.77	38.43	21.19	35.43	37.11	96.23
	2005					
Site 1	47.82	66.80	13.93	24.28	24.07	64.65
Site 2	205.18	237.04	184.54	195.13	194.91	236.50
Site 20	N/A	39.51	12.06	20.10	20.29	53.94
Site 13	70.66	69.04	24.95	41.33	41.96	111.12

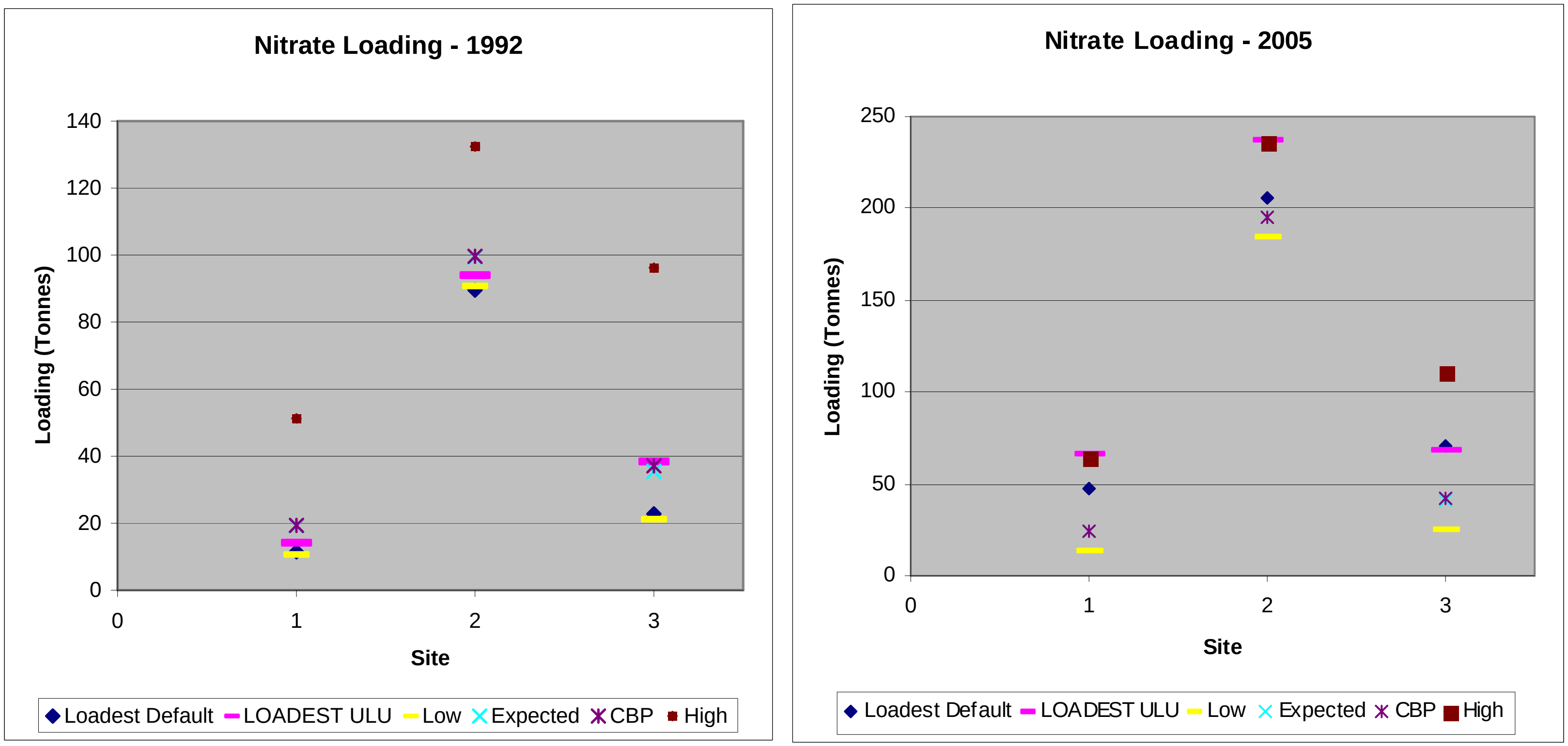

Figure 5.12. Estimated LOADEST 1992 nitrate-N loadings (left) and 2005 loadings (right) (for Pohick sites 1 and 2 and Accotink 13 (listed as 3) compared to loadings for the four export coefficient scenarios with observed 1992 and 2004 land use (2005 land use is not yet available, hence, 2004 was used as a surrogate). All units are in metric tonnes (1000 kg) as N.

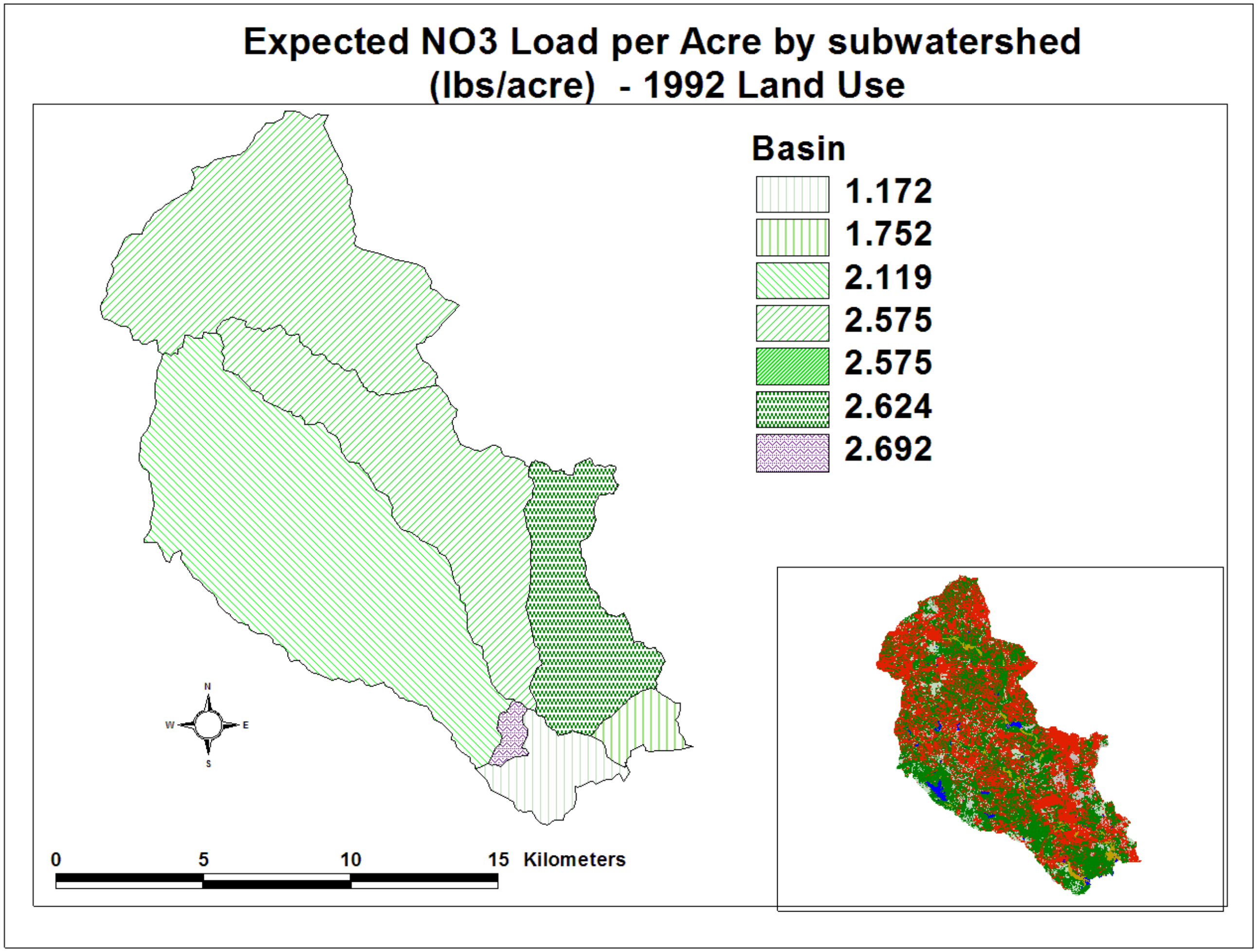

Figure 5.13. Loading rates of nitrate-N per acre in lbs/acre (default value in PLOAD) from stormwater using average year export coefficients in 1992. The 1992 land use is in the lower right corner (red – urban, dark green – forest, light brown – wetlands, blue – water, light green – grass, grey – barren/transitional).

5.4.4.1.2. Historic Ammonia-N loadings

Use of export coefficients with ammonia-N performed reasonably well when compared to loading estimates from LOADEST; however, estimated ammonia-N loading from LOADEST tended to be on the higher end of the range of the four export coefficient scenarios for 1992 (Figure 5.14) and on the lower end for 2005 (Figure 5.15). The decrease in ammonia-N loadings from 1992 to 2005 signal that either ammonia-N was overestimated for 1992 or that there has been elimination of some significant ammonia-N sources such as from sanitary sewer overflows (SSOs). The largest contributor of ammonia-N, the Noman Cole sewage treatment facility (POTW), decreased its discharge substantially. As seen in Table 5.17, the ammonia-N contributions from precipitation-driven pollution pale in comparison to that from the POTW. By 2005, the loadings were reduced by nearly two orders of magnitude. Therefore, thanks to the substantial decrease of ammonia-N by the POTW, ammonia-N loadings are rather a small net contributor of nitrogen to Pohick Creek, and consequently Gunston Cove (see chapter 4 for additional discussion). Based on these loadings, ammonia-N loading has ceased to be a primary concern for these ecosystems and unless the POTW decreases treatment efficiency substantially or deteriorating infrastructure results in numerous near stream SSOs, ammonia-N loading should never approach historical loadings with increased watershed development.

Table 5.17. Annual Ammonia-N Loading at 4 sites for 1992 and 2005. The LOADEST loading was calculated using multiple regression approaches in the LOADEST program. The low, CBP, average, and high scenarios were calculated using export coefficients plus the actual point source (Noman Cole Facility) contribution for site 2.

Site	1992					
	LOADEST		Export Coefficients			
	(ULU)	Default	Low	CBP	Average	High
Site 1	8.51	3.37	1.04	1.68	1.95	9.34
Site 2	672.98	667.83	665.53	666.20	666.47	674.07
Site 20	N/A	N/A	0.98	1.44	1.73	8.59
Site 13	4.67	4.01	2.10	3.14	3.72	18.59
	2005					
Site 1	2.48	1.20	1.39	2.15	2.42	12.49
Site 2	9.75	8.49	7.48	8.25	8.53	18.86
Site 20	N/A	0.00	1.21	1.78	2.03	10.76
Site 13	3.74	1.86	2.50	3.66	4.20	22.01

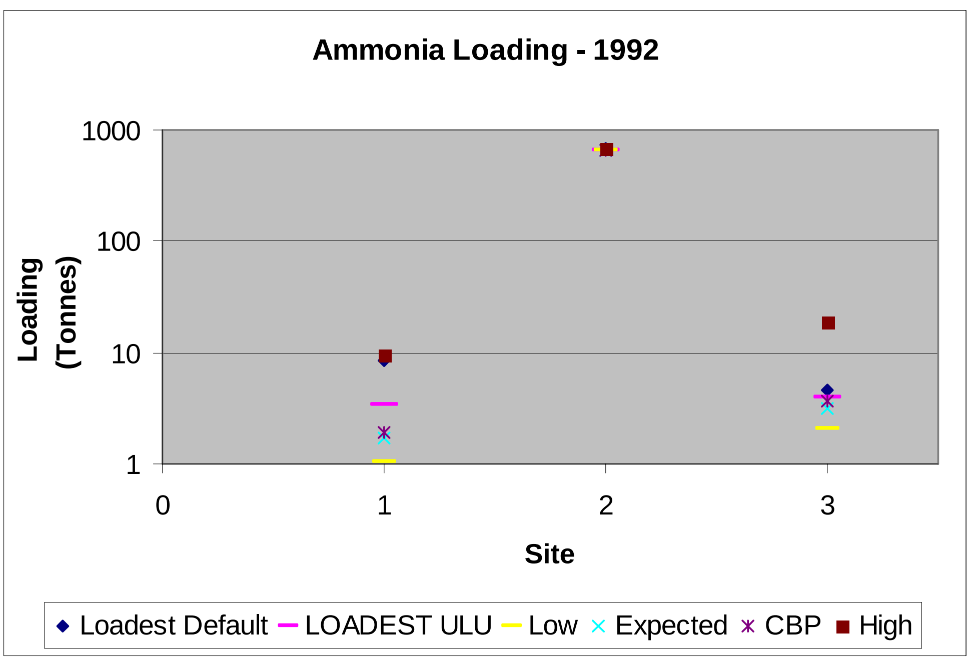

Figure 5.14. Estimated LOADEST 2005 ammonia-N loadings (tonnes as N) for Pohick sites 1 and 2 and Accotink 13 (listed as 3) compared to loadings with the four export coefficient conditions. Note that due to the significant loadings from the Noman Cole Facility, the y axis is on a logarithmic scale.

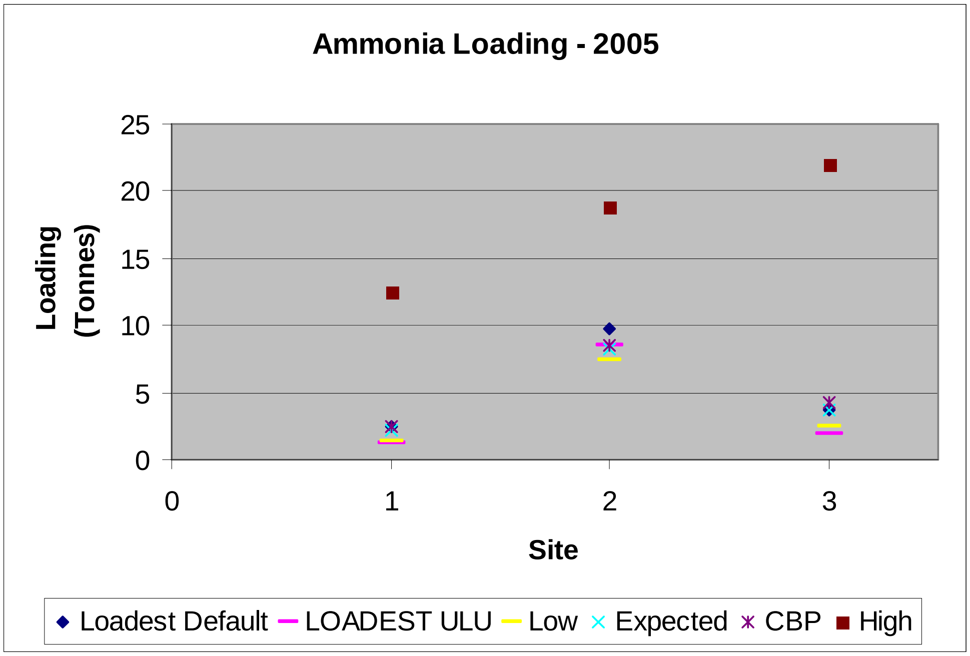

Figure 5.15. Estimated LOADEST 2005 ammonia-N loadings for Pohick sites 1 and 2 and Accotink 13 (listed as 3) compared to loadings with the four export coefficient conditions.

5.4.4.2. Forecasting nitrogen loading using the Export Coefficient Approach

The five future land use scenarios, the unadjusted projected Jantz "current conditions" (JCC), adjusted current conditions (ADJ-CC), adjusted smart growth (ADJ-SG), adjusted sustainable (ADJ-Sust), and HLU scenarios, were used in combination with the export coefficients for each scenario with each site to determine changes in nitrate-N, ammonia-N, and total nitrogen loadings. Examples of the spreadsheets used to determine loading by watershed are presented in Appendix C.

5.4.4.2.1. Inclusion of Point Source Loadings from the Noman Cole Plant

For Pohick site 2, below the Noman Cole sewage treatment plant, and the total Pohick watershed, the three projected scenarios of nitrogen loading from the POTW were also included. The high POTW nitrogen discharge scenarios were used with the JCC and ADJ-CC land uses and high export coefficient scenario to generate a high-end range of nitrogen loading to Gunston Cove. The middle nitrogen discharge from the POTW was used for the JCC and ADJ-CC land use scenarios with the average, CBP, and low export coefficient scenarios, and with the smart growth and HLU projections for high, average, and CBP export coefficients. The low nitrogen discharge from the POTW was used with the low export coefficients for the ADJ-SG and HLU land use scenarios, and with all export coefficient scenarios for the adjusted sustainable development (ADJ-Sust) scenario. Using such an approach yields the appropriate expected range, but also shows the full range of nitrogen loading for nitrate-N, ammonia-N, and total nitrogen.

5.4.4.2.2. Projected Nitrate-N Loading

This subsection discusses projected loadings of nitrate at each site and total loading into Gunston Cove using five projected land use scenarios. It answers the question of how nitrate loadings are projected in each land use scenario without effective BMPs. Table 5.18 shows projected nitrate-N loadings at each site, and compares the five future land use scenarios to two baseline scenarios, 1992 and 2004. The 2004 baseline may be a relative underestimation of nitrate loading, since the influence of BMPs are not considered in this subsection.

Table 5.18. Projected annual nitrate-N loading (tonnes) using five land use scenarios (row labels), four export coefficient approaches (column labels) and three POTW loading scenarios (at sites Pohick 2 and Pohick total). The high POTW loading scenario has a yellow background, the middle loading scenario has an orange background, and the low loading scenario has a purple background.

Site 1	High	Average	CBP	Low
JCC	82.65	32.07	30.94	19.56
ADJ-CC	78.28	30.15	29.32	18.23
ADJ-SG	75.07	28.73	28.13	17.25
ADJ-Sust.	71.54	27.18	26.82	16.18
HLU	71.64	26.85	26.84	15.70
Site 2	High	Average	CBP	Low
JCC	681.59	284.34	283.19	271.59
ADJ-CC	677.14	282.38	281.54	270.23
ADJ-SG	328.25	280.96	280.34	146.37
ADJ-Sust.	201.84	156.53	156.15	145.29
HLU	325.20	279.21	279.19	144.90
Pohick Total	High	Average	CBP	Low
JCC	686.65	286.08	285.11	272.50
ADJ-CC	681.44	283.85	283.17	270.95
ADJ-SG	331.94	282.16	281.74	146.91
ADJ-Sust.	209.23	159.32	158.94	146.94
HLU	329.09	280.49	280.66	145.48
Site 20	High	Average	CBP	Low
JCC	64.09	25.16	23.76	15.52
ADJ-CC	61.13	23.78	22.69	14.57
ADJ-SG	59.74	23.15	22.17	14.13
ADJ-Sust.	58.26	22.48	21.63	13.66
HLU	60.76	23.02	22.57	13.73
Site 13	High	Average	CBP	Low
JCC	134.90	52.80	50.06	32.47
ADJ-CC	128.48	49.94	47.69	30.42
ADJ-SG	125.44	48.58	46.57	29.47
ADJ-Sust.	123.57	47.74	45.87	28.88
HLU	125.38	47.63	46.54	28.21
Accotink Total	High	Average	CBP	Low
JCC	139.25	54.46	51.70	33.47
ADJ-CC	132.51	51.47	49.20	31.33
ADJ-SG	129.09	49.96	47.94	30.26
ADJ-Sust.	128.03	49.46	47.55	29.91
HLU	128.43	48.75	47.69	28.80

Using the average nitrate-N loading values, nitrate-N loading from both Pohick and Accotink to Gunston Cove is expected to be between 209 and 341 metric tonnes as N for all land use scenarios (Table 5.19). The full range of loading possibilities to the cove varies from 174 to 826 tonnes. There is considerable variation in the amount of nitrate-N that may be discharged to Gunston Cove; however, much of this variation is a result of the variation at the Noman Cole sewage treatment plant. In most situations, the Cole plant continues to make up the majority of the nitrate-N loading to Gunston Cove. In the ADJ-CC land use scenario, with average export coefficients, precipitation driven pollution makes up only 25.0% of nitrate-N loading to Gunston Cove (Table 5.20). Point source dominated nitrate-N loading from the Noman Cole plant is consistent with historical values. In 1992, non Noman Cole nitrate-N loading accounted for 42.5% of nitrate-N loading and in 2005 it accounted for 28.7% of nitrate-N loading.

Table 5.19. Total projected nitrate-N loading (tonnes) to Gunston Cove under five projected land use scenarios with varying assumptions. See Table 5.18 for explanation of color-coding.

Nitrate Loading	High	Average	CBP	Low
JCC	825.90	340.54	336.81	305.97
ADJ-CC	813.95	335.32	332.37	302.28
ADJ-SG	461.03	332.12	329.67	177.17
ADJ-Sust.	337.26	208.78	206.49	176.85
HLU	457.51	329.24	328.35	174.29

Table 5.20. Projected percent nitrate-N loading from stormwater sources versus that from the Noman Cole plant for Accotink and Pohick in five land use scenarios, four export coefficient rates, and three point source loading scenarios. See Table 5.18 for explanation of color-coding.

Stormwater Loading/ Noman Cole Loading	High	Average	CBP	Low
JCC	27.7%	26.1%	25.3%	17.8%
ADJ-CC	26.6%	25.0%	24.3%	16.8%
ADJ-SG	45.4%	24.2%	23.7%	27.3%
ADJ-Sust.	61.8%	38.3%	37.6%	27.2%
HLU	45.0%	23.6%	23.4%	26.1%

If the loading from the Noman Cole facility is excluded, the expected variation decreases significantly. The 'average' nitrate-N loading from precipitation-driven sources (stormwater and nonpoint sources) varies from 77.6 tonnes to 88.9 tonnes with projected land use scenarios, with a total range of 45.5 to 228.6 tonnes in all scenarios. These expected loadings increase from the value of 68.6 tonnes in 2004 using the average export coefficients (excluding the influence of BMPs) (Table 5.21). Hence, in an average conditions year, using the export coefficient approach, expected nitrate-N loadings are estimated to increase from 13 to 27% with future land use scenarios if no effective BMPs are used.

Table 5.21. Precipitation driven (nonpoint source and stormwater) projected nitrate-N loading (tonnes as N) for both Accotink and Pohick using five projected land use scenarios compared to historical land use as a baseline.

Nitrate Loading	High	Average	CBP	Low
JCC	228.61	88.91	85.18	54.33
ADJ-CC	216.66	83.68	80.73	50.64
ADJ-SG	209.39	80.49	78.04	48.41
ADJ-Sust.	208.50	80.02	77.72	48.08
HLU	205.88	77.60	76.71	45.52
2004 Land Use	183.34	68.64	68.46	40.26
1992 Land Use	154.93	59.09	57.41	33.15

5.4.4.2.3. BMP Nitrate-N Removal Targets to Maintain Current Precipitation Driven Loading Conditions

On a Chesapeake Bay wide scale, the 13 to 27% increase is significant. However, such a reasonably small increase can be handled with effective BMP treatment to at least keep nitrate-N discharges at 2004 conditions. In order to best gauge the impacts of land use change on nutrient loadings, no direct removal efficiencies were included for BMPs in the projected scenarios discussed in Section 5.4.4.2.2. If effective BMPs are used, then projected nutrient loading will be lower. However, there is question about the long-term efficiency of BMPs currently being most used by the County (dry detention basins) for nutrient removal. O'Shea et al. (2002) found that the dry detention basin median nitrate removal rate is only 5%. If more effective BMPs are used that can remove 20% nitrate-N for all new development, the 'average year' nitrate-N loading can be reduced by 3.99 tonnes under the ADJ-CC scenario (Table 5.22). Encouraging BMPs with higher effectiveness (including BMP treatment trains) in combination with varying degrees of watershed retrofitting can decrease nitrate-N added, so no net increase in nitrate-N loading is feasible with various combinations of BMP stringency and land use policies. BMPs are available that have higher nitrate removal efficiency: for instance, Oshea et al. (2002) documented a median nitrate removal efficiency of 40% for wetlands, 51% for vegetated swales, and 82% for infiltration practices in reported BMP performance values from two BMP databases.

A simple spreadsheet approach was used to look at examine net change of nitrate-N loading in the 'average year' and 'high' export coefficient scenarios. Several scenarios were examined to compare changes in nitrate-N loading if less effective BMPs are implemented or in several more effective BMP scenarios. A “static impact” class was created that assumed all future urban land use contributes loading at the same rate as past land use. Four categories for targeted BMP removal efficiencies were created: “20” assumes all new development has BMPs or design criteria that remove 20% of nitrate; “30/5” assumes new development has BMPs or design criteria that remove 30% of nitrate-N and watershed retrofits and changes in behavior remove 5% of nitrate-N from existing development; “40/10” assumes new development has BMPs or design criteria that remove 40% of nitrate-N and watershed retrofits and changes in behavior remove 10% of nitrate-N from existing development ; “40/15” assumes new development has BMPs or design criteria that remove 40% of nitrate-N and watershed retrofits and changes in behavior remove 15% of nitrate-N from existing development. Land use loading coefficients (before BMP removal) are projected to stay the same in each scenario.

Table 5.22. Additional nitrate-N added in future land use scenarios compared to 2004 land use conditions under 'average year' nitrate-N export coefficient rates using various BMP effectiveness scenarios.

Additional NO_3 Tonnes under different BMP effectiveness: "Average Year" Scenario Export Coefficients						
	Watershed	**Static Impact**	**20**	**30/5**	**40/10**	**40/15**
JCC	Pohick All	8.69	6.37	4.18	1.98	0.95
	Accotink All	11.57	8.48	5.07	1.65	-0.22
	To Cove	20.27	14.86	9.24	3.63	0.73
ADJ-CC	Pohick All	6.46	4.72	2.82	0.92	-0.12
	Accotink All	8.59	6.33	3.33	0.33	-1.54
	To Cove	15.04	11.05	6.15	1.25	-1.66
ADJ-SG	Pohick All	4.77	3.48	1.80	0.12	-0.91
	Accotink All	7.08	5.23	2.43	-0.36	-2.23
	To Cove	11.85	8.71	4.23	-0.24	-3.15
ADJ-Sust	Pohick All	4.81	3.51	1.83	0.14	-0.89
	Accotink All	6.57	4.86	2.14	-0.59	-2.46
	To Cove	11.38	8.37	3.96	-0.44	-3.35
HLU	Pohick All	3.10	2.49	1.15	-0.19	-1.23
	Accotink All	5.86	4.88	2.52	0.15	-1.72
	To Cove	8.96	7.37	3.67	-0.04	-2.94

Under the "average year" export coefficient scenario, nitrate-N loadings decrease, although not dramatically, based upon BMP effectiveness. The greatest potential change is less than 20 metric tonnes of nitrate-N per year reduced in the JCC scenario. However, under the high export coefficient range, nitrate-N decreases are more significant (Table 5.23) with reduction of nearly 50 metric tonnes in the JCC land use scenario. Under these circumstances, BMPs effectively play a greater role in nitrate-N removal than land use scenarios. It must be noted that 2005, the most recent year of data used for this analysis, resembles loading conditions found in the high scenario. If nitrate-N loading has increased in a non-linear fashion with urban land use, then the high export coefficient scenarios might be more likely than the "average year" values. Furthermore, under these higher export coefficient scenarios, BMP effectiveness in terms of percent removal would likely decrease since these situations likely represent wetter years. Hence, practices that may result in a "40/15[29]" situation in "average year" conditions, may only result in "30/5[30]" conditions in for those high precipitation years in which land exports higher nitrate loadings per unit urban area. Though the BMPs may be removing as much nitrate, there may be higher loadings in the influent and faster flushing times through the BMPs; hence, a lower percentage of nitrate-N is removed.

[29] 40/15 scenarios assume that 40% of nitrate is removed from stormwater from new development and 15% of nitrate is removed from stormwater from existing development.
[30] 30/5 scenarios assume that 30% of nitrate is removed from stormwater from new development and 5% of nitrate is removed from stormwater from existing development.

Table 5.23. Additional nitrate-N added in future land use scenarios compared to 2004 land use conditions under high nitrate-N export coefficient conditions using various BMP effectiveness. Static Impact assumes that future urban land use loads at the same rate as past land use; "20" assumes all new development has BMPs or design criteria that remove 20% of nitrate; "30/5" assumes new development has BMPs or design criteria that remove 30% of nitrate-N and watershed retrofits and changes in behavior remove 5% of nitrate-N from existing development; "40/10" assumes new development has BMPs or design criteria that remove 40% of nitrate-N and watershed retrofits and changes in behavior remove 10% of nitrate-N from existing development ; "40/15" assumes new development has BMPs or design criteria that remove 40% of nitrate-N and watershed retrofits and changes in behavior remove 15% of nitrate-N from existing development. All other land use export coefficients remain the same.

Additional NO_3 Tonnes under different BMP effectiveness: "High" Scenario Export Coefficients						
	Watershed	**Static Impact**	**20**	**30/5**	**40/10**	**40/15**
JCC	Pohick All	19.71	13.85	8.30	2.76	0.15
	Accotink All	25.57	17.76	9.13	0.50	-4.23
	To Cove	45.27	31.61	17.43	3.26	-4.08
ADJ-CC	Pohick All	14.50	10.12	5.31	0.51	-2.10
	Accotink All	18.82	13.13	5.55	-2.03	-6.75
	To Cove	33.32	23.25	10.87	-1.52	-8.86
ADJ-SG	Pohick All	10.65	7.39	3.15	-1.09	-3.70
	Accotink All	15.40	10.74	3.68	-3.38	-8.11
	To Cove	26.05	18.13	6.83	-4.47	-11.82
ADJ-Sust	Pohick All	10.82	7.54	3.29	-0.96	-3.57
	Accotink All	14.34	10.03	3.14	-3.74	-8.47
	To Cove	25.16	17.57	6.44	-4.70	-12.04
HLU	Pohick All	7.80	6.26	2.88	-0.50	-3.12
	Accotink All	14.74	12.26	6.29	0.32	-4.41
	To Cove	22.54	18.52	9.17	-0.18	-7.53

5.4.4.2.4. Projected Ammonia-N Loading

As previously discussed, ammonia-N loadings from the Noman Cole POTW have decreased significantly and, provided there are no major reductions in treatment efficiency or new sources, ammonia-N loadings are projected to remain low in all future land use scenarios (Table 5.24). With all five land-use situations, ammonia-N loading from stormwater pollution to Gunston Cove is estimated to vary between 10.5 tonnes and 246.3 tonnes (Table 5.25). The greatest variability in ammonia-N loading is caused by the scenarios with the Noman Cole facility: if the facility continues treating ammonia-N as effectively as it does today, then ammonia-N loading should be nowhere near the high estimates. However, if treatment efficiency decreases to levels approaching permit limits, then scenarios nearer the high estimates would be likely. The stormwater driven loading is expected to account for approximately 31.8% of ammonia-N loading in the "average year" export coefficient, ADJ-CC scenario (Table 5.26). Though ammonia-N loading from stormwater is not increasing rapidly, the relative contribution of ammonia-N loading from these sources is an increase from less than 1% in 1992, when there was substantial ammonia-N loading from the Noman Cole Facility. The stormwater driven loading under "average year" conditions is not expected to increase dramatically from 2004 conditions, ranging from 7.8 to 8.9 tonnes in all five scenarios (Table 5.27). However, these values assume that sanitary sewer infrastructure is maintained effectively.

Table 5.24. Projected annual ammonia-N loading using five land use scenarios (row labels), four export coefficient approaches (column labels) and three POTW loading scenarios (at sites Pohick 2 and Pohick total). The high POTW loading scenario has a yellow background, the middle loading scenario has an orange background, and the low loading scenario has a purple background.

Site 1	High	Average	CBP	Low
JCC	17.03	3.22	2.84	1.96
ADJ-CC	15.93	3.03	2.67	1.82
ADJ-SG	15.13	2.89	2.55	1.72
ADJ-Sust.	14.24	2.73	2.41	1.62
HLU	13.92	2.70	2.38	1.57
Site 2	**High**	**Average**	**CBP**	**Low**
JCC	216.47	21.26	20.87	19.97
ADJ-CC	215.35	21.06	20.70	19.83
ADJ-SG	33.41	20.92	20.58	7.48
ADJ-Sust.	20.27	8.51	8.19	7.37
HLU	32.27	20.74	20.42	7.33
Pohick Total	**High**	**Average**	**CBP**	**Low**
JCC	217.33	21.43	21.03	20.06
ADJ-CC	216.03	21.21	20.83	19.90
ADJ-SG	33.94	21.04	20.68	7.53
ADJ-Sust.	21.72	8.79	8.44	7.54
HLU	32.84	20.87	20.53	7.39
Site 20	**High**	**Average**	**CBP**	**Low**
JCC	13.39	2.52	2.18	1.55
ADJ-CC	12.64	2.38	2.06	1.46
ADJ-SG	12.28	2.32	2.01	1.41
ADJ-Sust.	11.91	2.25	1.95	1.37
HLU	12.08	2.30	2.00	1.37
Site 13	**High**	**Average**	**CBP**	**Low**
JCC	28.07	5.28	4.59	3.25
ADJ-CC	26.45	5.00	4.34	3.04
ADJ-SG	25.68	4.86	4.22	2.95
ADJ-Sust.	25.20	4.78	4.15	2.89
HLU	24.88	4.77	4.13	2.82
Accotink Total	**High**	**Average**	**CBP**	**Low**
JCC	28.95	5.45	4.73	3.35
ADJ-CC	27.25	5.15	4.47	3.13
ADJ-SG	26.39	5.00	4.34	3.03
ADJ-Sust.	26.11	4.95	4.30	2.99
HLU	25.43	4.88	4.23	2.88

Table 5.25. Total projected ammonia-N loading (tonnes) to Gunston Cove under five land use scenarios with varying assumptions. See Table 5.24 for explanation of color-coding.

Ammonia-N Loading	High	Average	CBP	Low
JCC	246.27	26.88	25.76	23.40
ADJ-CC	243.27	26.36	25.30	23.03
ADJ-SG	60.32	26.04	25.02	10.56
ADJ-Sust.	47.83	13.74	12.73	10.53
HLU	58.27	25.75	24.76	10.27

Table 5.26. Projected percent ammonia-N loading from stormwater sources versus that from the Noman Cole plant for Accotink and Pohick in five land use scenarios, four export coefficient rates, and three point source loading scenarios. See Table 5.24 for explanation of color-coding.

Stormwater/PS Load.	High	Average	CBP	Low
JCC	19.2%	33.1%	30.2%	23.2%
ADJ-CC	18.2%	31.8%	29.0%	22.0%
ADJ-SG	70.2%	31.0%	28.2%	45.8%
ADJ-Sust.	88.0%	58.4%	55.1%	45.6%
HLU	69.2%	30.2%	27.4%	44.3%

Table 5.27. Total projected ammonia-N loading (tonnes) for both Accotink and Pohick under five land use scenarios compared to historical land use excluding the Noman Cole Point Source.

Ammonia-N Loading	High	Average	CBP	Low
JCC	47.18	8.91	7.79	5.43
ADJ-CC	44.18	8.39	7.33	5.06
ADJ-SG	42.35	8.07	7.05	4.84
ADJ-Sust.	42.11	8.02	7.01	4.81
HLU	40.29	7.78	6.79	4.55
2004 Land Use	35.72	6.88	6.03	4.02
1992 Land Use	29.16	5.94	5.06	3.26

5.4.4.2.5. BMP Ammonia-N Removal Targets to Maintain Current Precipitation Driven Loading Conditions

Specific BMP removal efficiency needed to maintain ammonia-N concentrations at 2004 conditions are not presented here since increases in ammonia-N loadings from stormwater do not play a foreseeable role in water quality degradation in Accotink or

Pohick or increasing nutrient loadings to Gunston Cove. Like with nitrate, a strategy of removing 40% of ammonia from new development and either 10 or 15% from existing development will keep ammonia loadings at about 2004 levels. However, these levels are so low that a single large sanitary sewer overflow can more than double these loading rates. Hence, maintenance of sewer system infrastructure will likely play as large a role as construction of efficient BMPs on impacting ammonia-N loadings (keeping in mind that increases in nitrate-N loadings will increase ammonia-N concentrations in anoxic and hypoxic conditions).

5.4.4.2.6. Projected Total Nitrogen Loading

Total nitrogen loading follows similar patterns to that of nitrate. The expected total nitrogen loading from Accotink and Pohick to Gunston Cove is expected to be between 303.1 and 474.4 metric tonnes for all land use scenarios (Table 5.28). The full range of loading possibilities to the cove varies from 236.3 to 1193.0 tonnes. As with nitrate, much of the variation in the amount of nitrogen discharged to Gunston Cove is a result of the variation in scenarios at the Noman Cole sewage treatment plant. In most situations, the Cole plant continues to make up the majority of the total nitrogen loading to Gunston Cove (Table 5.29). Under “average year” conditions, total nitrogen loading should increase from approximately 133.3 tonnes using 2004 land use to between 155.2 to 177.8 tonnes, an increase of between 16-33% (Table 5.30).

Table 5.28. Total projected total nitrogen loading (tonnes) to Gunston Cove under five land use scenarios with varying assumptions. See Table 5.24 for explanation of color-coding.

Total Nitrogen Loading	High	Average	CBP	Low
JCC	1193.04	474.39	419.63	405.92
ADJ-CC	1176.21	463.94	413.41	398.89
ADJ-SG	591.49	457.54	409.64	241.14
ADJ-Sust.	590.23	303.11	255.69	240.53
HLU	586.54	451.77	408.24	236.32

Table 5.29. Projected total nitrogen loading from stormwater sources versus that from the Noman Cole plant for Accotink and Pohick in five land use scenarios, four export coefficient rates, and three point source loading scenarios. See Table 5.24 for explanation of color-coding.

Stormwater/ Point Source Loading	High	Average	CBP	Low
JCC	27.0%	37.5%	29.3%	26.9%
ADJ-CC	25.9%	36.1%	28.3%	25.6%
ADJ-SG	49.9%	35.2%	27.6%	40.7%
ADJ-Sust.	75.8%	52.8%	44.0%	40.5%
HLU	49.4%	34.4%	27.4%	39.5%

Table 5.30. Projected total nitrogen loading (tonnes) for both Accotink and Pohick under five land use scenarios compared to historical land use excluding the Noman Cole Point Source.

Total Nitrate Loading	High	Average	CBP	Low
JCC	321.99	177.82	123.06	109.34
ADJ-CC	305.15	167.37	116.84	102.31
ADJ-SG	294.92	160.97	113.06	98.06
ADJ-Sust.	447.16	160.04	112.62	97.46
HLU	289.97	155.20	111.66	93.25
2004 Land Use	249.82	133.25	96.24	80.34
1992 Land Use	209.62	113.88	81.08	66.32

5.4.4.2.7. BMP Total Nitrogen Removal Targets to Maintain Current Precipitation Driven Loading Conditions

Total nitrogen loadings can maintained at 2004 conditions using a mix of effective BMPs on newer developments and watershed retrofits using “average year” conditions (Table 5.31). For these moderately developed inner suburban watersheds, changes in land use policies do not play a major role in impacting total nitrogen loading since much of the watersheds were already developed as of 2004. It should be noted that with current technologies, it is sometimes easier to remove more of the total nitrogen than nitrate-N alone since organic nitrogen can often be more easily filtered or settled out. Hence, reaching target removal efficiencies used as examples in Table 5.31 and Table 5.32 may completed more easily than with nitrate. Like with nitrate, total nitrogen loadings are significantly higher using the high export coefficient scenarios: hence, BMPs have a greater impact on the mass of total nitrogen removed Table 5.32.

Table 5.31. Additional total nitrogen added in future land use scenarios compared to 2004 land use conditions under "average year" total nitrogen export coefficient conditions using various BMP effectiveness scenarios. Static impact assumes that future urban land use loads at the same rate as past land use; "20" assumes all new development has BMPs or design criteria that remove 20% of total nitrogen; "30/5" assumes new development has BMPs or design criteria that remove 30% of total nitrogen and watershed retrofits and changes in behavior remove 5% of total nitrogen from existing development; "40/10" assumes new development has BMPs or design criteria that remove 40% of total nitrogen and watershed retrofits and changes in behavior remove 10% of total nitrogen from existing development ; "40/15" assumes new development has BMPs or design criteria that remove 40% of total nitrogen and watershed retrofits and changes in behavior remove 15% of total nitrogen from existing development. All other land use export coefficients remain the same.

Additional TN Tonnes under different BMP effectiveness: ""average Year"" Scenario Export Coefficients						
	Watershed	**Static Impact**	**20**	**30/5**	**40/10**	**40/15**
JCC	Pohick All	17.39	12.75	8.36	3.97	1.90
	Accotink All	23.15	16.97	10.13	3.30	-0.45
	To Cove	40.53	29.71	18.49	7.26	1.45
ADJ-CC	Pohick All	12.91	9.44	5.64	1.84	-0.23
	Accotink All	17.17	12.66	6.66	0.66	-3.08
	To Cove	30.08	22.11	12.30	2.50	-3.32
ADJ-SG	Pohick All	9.54	6.96	3.60	0.24	-1.83
	Accotink All	14.15	10.46	4.87	-0.72	-4.47
	To Cove	23.69	17.42	8.47	-0.48	-6.30
ADJ-Sust	Pohick All	9.61	7.02	3.65	0.29	-1.78
	Accotink All	13.14	9.73	4.28	-1.18	-4.92
	To Cove	22.76	16.75	7.93	-0.89	-6.70
HLU	Pohick All	6.19	4.98	2.30	-0.38	-2.45
	Accotink All	11.72	9.76	5.03	0.31	-3.44
	To Cove	17.92	14.74	7.33	-0.08	-5.89

Table 5.32. Additional total nitrogen added in future land use scenarios compared to 2004 land use conditions under high total nitrogen export coefficient conditions using various BMP effectiveness scenarios.

Additional TN Tonnes under different BMP effectiveness: "High" Scenario Export Coefficients						
	Watershed	**Static Impact**	**20**	**30/5**	**40/10**	**40/15**
JCC	Pohick All	27.75	19.50	11.69	3.89	0.20
	Accotink All	36.01	25.02	12.86	0.71	-5.96
	To Cove	63.76	44.52	24.56	4.59	-5.75
ADJ-CC	Pohick All	20.42	14.25	7.49	0.72	-2.96
	Accotink All	26.51	18.49	7.82	-2.85	-9.51
	To Cove	46.93	32.74	15.30	-2.13	-12.48
ADJ-SG	Pohick All	15.00	10.41	4.44	-1.54	-5.22
	Accotink All	21.69	15.12	5.18	-4.76	-11.42
	To Cove	36.70	25.54	9.62	-6.30	-16.64
ADJ-Sust	Pohick All	15.23	10.63	4.64	-1.35	-5.03
	Accotink All	20.20	14.12	4.43	-5.27	-11.93
	To Cove	35.43	24.75	9.06	-6.62	-16.96
HLU	Pohick All	10.98	8.82	4.05	-0.71	-4.39
	Accotink All	20.76	17.27	8.86	0.45	-6.21
	To Cove	31.75	26.09	12.91	-0.26	-10.60

5.5 Short Discussion: Comparison of Changes in Hydrology to Changes in Nutrient Loading

In the absence of effective BMPs, land use change affects hydrology within Accotink and Pohick more significantly than it will impact nitrogen loadings to Gunston Cove, and ultimately, the Chesapeake Bay. The actual ecosystems of Accotink and Pohick Creeks will undergo extreme changes due to these hydrological alterations. Remaining tolerant benthic species in Pohick will be subjected to increasingly extreme flow events, and sedimentation and embeddedness will proliferate as the increasing power of storm flows causes elevated scour. Those species that are able to tolerate the additional sediment deposition and flushing events will have to contend with lower flows and consequently less habitat in the summer dry months when shallow groundwater inflow makes up the majority of low flow events. In short, it is reasonably clear from these results that these hydrological changes are a significant threat to both streams.

The changes to nitrogen loading in Gunston Cove, on the other hand, are more than offset by decreases in loading to the cove from the sewage treatment facility. Hence, in terms of nitrogen loading to the cove alone, the actual availability of nitrogen should decrease over time. Furthermore, nitrogen is rarely the limiting nutrient in Gunston Cove (Jones & Kelso, 2005a), meaning excess nitrogen is either lost via volatilization or denitrification, or is passed through to the Potomac and the Chesapeake. The increasing nitrogen loading from stormwater sources will therefore not greatly alter the cove's ecology, nor should it significantly impact the ecology of the streams.

The net delivery of nitrogen from Accotink and Pohick to the mesohaline stretches of the Potomac and the Chesapeake Bay has decreased and should continue to do so with tighter NPDES requirements for the Noman Cole facility. However, Accotink and Pohick do not exist in isolation. There are numerous watersheds being influenced by stormwater pollutants that do not have reductions from a sewage treatment facility to offset increasing stormwater loads. Thus, the increase in nitrogen loadings from stormwater sources from Accotink and Pohick has consequences, albeit far less immediate and more clearly identifiable than those dealing with hydrology. In order to prevent continued expansion of the Chesapeake Bay dead zone, nitrogen loading must continue to be reduced (Chesapeake Bay Program Office, 2004). Based on these results, it is likely that despite continued development, watersheds that have had some previous development can have stable nitrogen export as they further develop. These objectives can be reached with a two-prong approach. First, there needs to be effective watershed retrofits on small portions of existing development. Second, stringent requirements for BMP effectiveness must be required for any new development and the County. Furthermore, the County must continue its inspection and maintenance regime (and expand it as more BMPs come on line).

In order to prevent continued degradation to local and downstream ecosystems, any stormwater program must address the issues of both hydrology and nutrient loadings. Fairfax County and Fairfax City have taken significant steps toward monitoring streams, restoring streams, and implementing BMPs. However, the results from this dissertation further support the conclusions from numerous publications (Booth *et al.*, 2004; Prince George's County (Md.), 2000b; US EPA, 2000c; Williams & Wise, 2006) that continued degradation will only be prevented on small streams if management plans prevent major changes to hydrological regimes. Plans that address these hydrological changes and result in a decrease in de facto nutrient loading (either via decreased total flow contribution or via percent reduction of runoff) are the only way in which local watershed planners can meet the complementary goals of preventing continued degradation in stream watersheds, while meeting obligations to reduce nitrogen loading in impacted downstream estuarine environments.

5.6 Evaluation of modeling tools used in this chapter for use by planners and policy makers

One of the original goals of this dissertation was to explore tools used to assess the impacts of urbanization on water quality for use by planners, policy makers, or others in resource-limiting environments. This chapter manually integrated two tools to project land use change with three tools to assess water quality. Two of those water quality models are relatively simple, though one is more complex. Policy makers have several options for tool development. The first option, to be used as a screening level approach, is to take land use projections from an existing model and plug them into a simple water quality model. The second option is to create an appropriate tool that allows direct examination of land use projections in an existing water quality modeling system. These two options are discussed in detail in the following sections.

The third option, one not discussed in this dissertation, is to have a more advanced research group use land use model projections with more complicated models like HSPF and distribute those results to local communities. This option requires a well organized and knowledgeable research group to model results on a sufficiently small enough scale for those results to be useful for local watershed managers and/or land use planners. For example, the Chesapeake Bay Program is currently working toward including forecasts from land use change models (based on SLEUTH) to use in its Chesapeake Bay wide HSPF based water quality model (Peter Claggett, personal communication, Spring, 2006 and 3/29/07).

5.6.1 Land Use tools

This chapter showed that land use change projections can be integrated with all three water quality tools used here. Because the real strength of the HLU method is estimating historic land use, HLU projections are not ideal for planners to use in forecasting land use. HLU projections are based on modified build-out projections. Projections based on build out scenarios tend to underestimate future land use and it is more resource friendly to use a more simple build-out model. On the other hand, the output from a model such as SLEUTH does appear to be potentially useful for policy makers.

In the case of the SLEUTH model output, multiple approaches could have been taken to integrate these data into water quality models. These options are summarized below:

A. Use of the probability summarized data; maintain data in its spatial orthorectified form; useable for direct input into BASINS (HSPF and PLOAD) and the GIS version of L-THIA.
    - Pros:
        - Single, intuitive land use data file can be used for the analysis.
        - Maintains visual tools provided by many WQ models.
    - Cons:
        - Would have to assign appropriate calibration parameters for each pixel with a different probability of urbanizing. This would

effectively result in more than 100 land uses being defined in the model. This would be too time-consuming a process for planners.
        - Only performs analysis for the mean land use scenario of all the Monte Carlo simulations.
        - The GIS version of L-THIA relies on proprietary software (Arcview).

B. Use of the probability summarized data. Tabulate and summarize land use data, manually enter data; useable in HSPF, the web-based input for L-THIA, and an export coefficient based spreadsheet – process used in this chapter.
    - Pros:
        - Single, intuitive land use data file can be used for the analysis.
        - Allows for easy export of data into web-based L-THIA, manual insertion into HSPF, or an export coefficient spreadsheet model.
    - Cons:
        - Only performs analysis for the ‘mean’ land use scenario of all the Monte Carlo simulations.
        - Unable to effectively utilize all of the visual tools in the Arcview version of L-THIA and PLOAD for future land use scenarios.

C. Use every Monte Carlo simulation output of the SLEUTH model (or other land use model output); maintain data in its spatial orthorectified form; useable for direct input into BASINS (HSPF and PLOAD) and the GIS version of L-THIA. –
    - Pros:
        - Produces a distribution of land use scenarios to generate a robust range of probable water quality outcomes.

    - Makes use of existing, high quality tools that already facilitate data collection for watershed analysis.
    - Maintains visual tools provided by many WQ models.
  - Cons:
    - Process must be automated. Manual entry and loading of each data set would be very time consuming.
    - Original full data output from land use model must be made available.
    - Need to find a model with more than just urban/non-urban land use or find another method to assess non-urban.
    - The GIS version of L-THIA relies on proprietary software (Arcview).

D. Use of every Monte Carlo output of the SLEUTH model, tabulate and summarize land use data, manually enter data; useable in HSPF, the web-based input for L-THIA, and an export coefficient based spreadsheet.

  - Pros:
    - Land Use can be easily summed from each iteration and plugged into models.
  - Cons:
    - Should be automated. If automated, the intermediate step offers no additional benefit where data is left in full spatial format.
    - Original full data output from land use model must be made available.

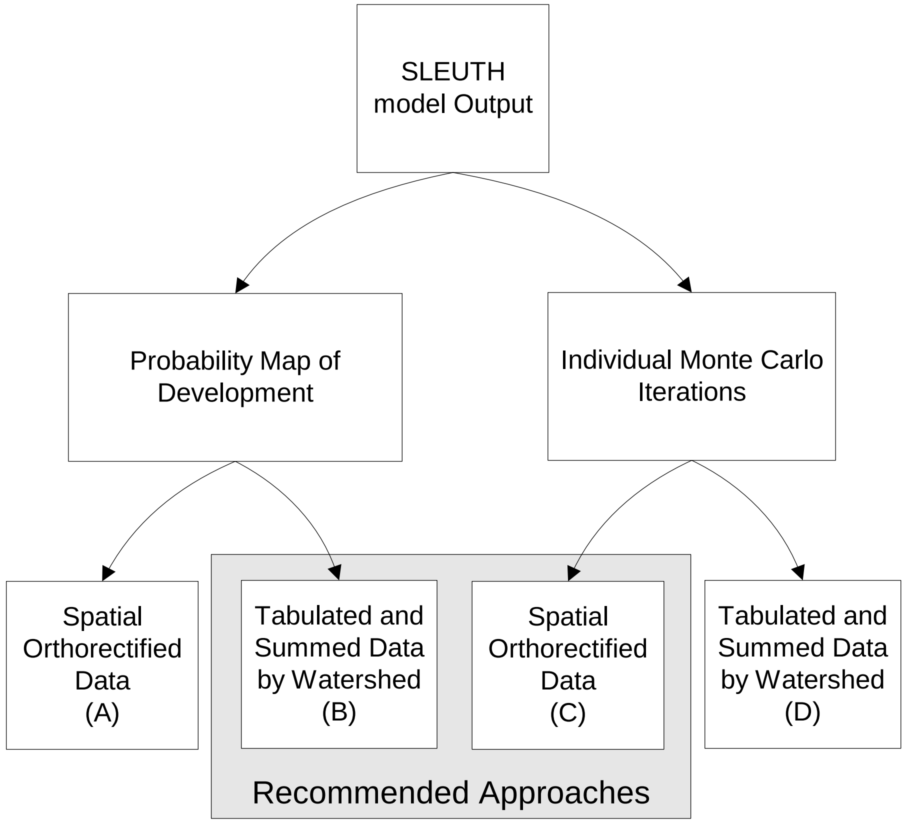

Figure 5.16. A flow chart illustrating the data form necessary for each option, and recommended approaches.

Options B and C are recommended for use with land use forecasts. Option B was used for this dissertation due to data format availability, the need to adapt original SLEUTH data to account for other land use types, and the need to generate alternate land use scenarios that take into account the disparity between density of urban land use observed in the NLCD and MRLC data (upon which the HSPF model was calibrated) and the remote sensing land use classifications generated by Jantz et. al (2004). Option B allowed the use of all tools, but it prevented the user from employing the spatially explicit tools in PLOAD and L-THIA GIS that auto sum results and create visual images and maps. It allowed the user to account for differences in observed urban density between the land use data utilized to calibrate the HSPF model and the land use data set used to calibrate the land use data model. It also allowed the land use projections in SLEUTH to be compared to other land use projections. In summary, this was a good approach that requires an understanding of GIS, spreadsheet models and functions, and basic drivers of land use models, land use change, and drivers that impact water quality.

In an ideal world, Option C is preferable. This option would require an original land use data set that included each Monte Carlo iteration and had more than two land use types (urban/non-urban). If used with a more complicated model such as HSPF, the user would need to make sure that the original land use data set on which the water quality model was calibrated is consistent with the land use data set for which the land use model was calibrated. Preferably, an automated script could be used to automatically load and run each Monte Carlo iteration and automatically export results to a data file for the user. The user could then sum results, have a maximum, minimum, and middle range of water quality outcomes, and produce probability distributions using the visual tools available in L-THIA GIS, BASINS, or another appropriate water quality model. However, this option is not viable for planners unless the appropriate land use model projections are already available. Additionally, automated, intuitive tools that facilitate inclusion of these land use projections would greatly enhance the viability of using this approach by planners and researchers alike.

5.6.2 Water Quality Tools

All of the water quality tools used in this dissertation could be used in a created tool for policy makers with caveats. In terms of using these tools for policy makers, there are several potential issues of concern, including, but not limited to, model reliability, the amount of data needed to run the model, time needed to learn how to use the model, model interface friendliness, and whether the model is available in the public domain. These attributes are discussed with each water quality model used in this dissertation in Table 5.33.

Table 5.33. Comparison of attributes of water quality models for use by planners.

Metric	HSPF in BASINS	L-THIA	PLOAD in BASINS (Export Coefficient)
Confidence in Accuracy of Hydrology Output	**High** (can comfortably reach annual runoff rates within 10% of observed values)	**Low-Medium** (Runoff rates within 50% using default values)	N/A
Confidence in Accuracy of Nutrient Loading Output	**N/A** (Model has capabilities but not completed for this dissertation)	**Low** (Uses simple method which is not initially recommended for watersheds larger than a square kilometer).	**Low-Medium** using single value or **Medium** using range of values (captured both wet and dry year values in historic land use scenarios)
Data Input Requirements	**High**	**Low**	**Low**
Parameters Required	Future land use, climatic data, flow, hydrologic parameter calibration, Point Source Data (if applicable)	Future land use, Precipitation, soil type, land use, custom curve numbers (optional)	Future land use, Export Coefficients, Point Source Data (if applicable)
Parameters or information available via the modeling system	Climatic data, flow, point source data (if applicable)	Precipitation, soil type, default curve numbers	Point source data (if applicable)
Parameters provided by the user	Future land use, hydrologic parameter calibration,	Future land use, Custom curve numbers	Future land use, Export Coefficients
Learning Curve	**High**	**Low** (Web based), **Medium** (Arc GIS version)	**Low**
Interface friendliness	**Medium**	**Extremely High** (web based), **Medium** (GIS)	**Medium**
Public availability	BASINS 4.0 with HSPF is free; Basins 3.1 requires Arcview 3	L-THIA is free: GIS based version requires Arcview	BASINS 4.0 with PLOAD is free; Basins 3.1 requires Arcview 3
Other notes			PLOAD in Basins 3.1 contained bugs that would limit implementation, doubt this is an issue in 4.0

Refer to sections 5.3 and 5.4 for additional details on the water quality tools used in this chapter. The metrics highlighted as important in the table are those that are most important to policy makers. Questions such as “where can I find the data,” “how long will I have to invest to use the system” and “are results reliable enough for decision making” are key elements that drive use of data systems and tools by many practitioners. Time and resources for natural resource protection and conservation agencies are extremely limited, and as such any tools used or developed must produce reliable results, and preferably should offer a range of options for sophistication of the models. Some planners, and many modeling specialists and researchers, would, however, like a land use projection tool that could also be used with more complicated models.

Provided BASINS 4.0 has eliminated the discouraging and undocumented bugs in PLOAD in BASINS version 3.1, BASINS is an ideal tool to integrate a land use forecast module that accomplishes all of these goals. Like flow data, meteorological data, existing land use data (NLCD 1992), STORET data (water quality data), Point Source Data, and other data sources, land use forecast data could be stored on the EPA server and downloaded by users on request. The modeling system already has the tools and data retrieval systems necessary to easily allow for other management of information. Modules to examine ‘what-if’ scenarios have already been successfully developed, as evidenced by the recent (February 2007) release of the climate change assessment module within BASINS. Land use forecasting data could either be stored as Monte Carlo simulations or probability distributions.

Training with BASINS typically has focused on HSPF. Such trainings are an excellent option for those who have the time to learn how to use the more complicated model. For those practitioners who do not, development of a one or two day training seminar that shows how to use the data retrieval tools in BASINS with a simple approach such as PLOAD would be exceptionally helpful. For these seminars, training materials specific to the simpler approaches should be developed with examples of applications of the PLOAD model. These materials could be made available on the web and advertised, much in the same way EPA currently makes available training materials for the HSPF system in BASINS. Lack of clear and intuitive training materials is a major obstacle for those wishing to use BASINS specifically for the PLOAD capabilities.

The web-based or GIS version of L-THIA could also easily integrate such data, but would likely need to use a probability distribution map rather than individual Monte Carlo iterations. For use with the GIS version, a link to download the data with specific instructions on inclusion and caveats could be provided. Furthermore, users of this system would need to either assign a different curve number for each urbanization probability class of cells or come up with alternative approach. However, the majority of planners will likely be inclined to use the web-based version of the model, and those who would think to come up with a viable system in L-THIA might be just as inclined to use a more complicated model like HSPF (which if resources and time are available, yields more accurate results). If viable land use projections are available, a pop-up tool, similar to that currently available with the web based L-THIA to identify soil types, could be created with projected land use data. The tool would need to have some more complicated features to allow for easy use by planners. One option would be to have a layer with predefined watershed delineations, such as those available with the new NHD plus data set (US EPA & USGS, 2005), that would automatically sum each land use type within a specific subwatershed(s) identified by the user for a single or multiple land use scenarios. The tool could then enter the data into the form fields and the user could manually type other land use options if they wanted to compare different water quality outcomes.

The manner in which data was retrieved, dissected, and manipulated for this dissertation is not feasible for most practitioners. It was resource intensive, required specialized knowledge, and significant time invested in exploration. However, the research clearly shows that the approaches used here do produce forecasts of water quality outcomes that are plausible with various land use scenarios, that these forecasts are consistent with past trends and theory, and that these results could be useful for planning purposes. Elements of the approaches used here could easily be completed by these practitioners. However, they would have to have access to other data sources, including local monitoring and/or loading data, to tailor their results to local watersheds. Ideally, a tool will be developed that supplements one of the viable modeling packages currently in use. Development of a user-friendly tool that automatically or semi-automatically retrieves and integrates the data into a framework that allows for appropriate analysis would serve a useful function for those practitioners and researchers alike who invest the time to learn to use the systems.

5.7 Future Research and Work

Specific results from the model output for the Accotink and Pohick watersheds have been examined in the context of Fairfax County's stormwater management program and are discussed in the following chapter. These results are being shared with Fairfax County for consideration in development of the Accotink and Pohick watershed management plans. Additionally, results from the approaches and models used in this chapter will be applied in a George Mason project examining how land manager behavior can positively affect impacts caused by land use change. Furthermore, ideally the HSPF model will be calibrated to compare results to PLOAD output. This work will either be completed by this researcher or by collaborating with other students or researchers. These future efforts could also make use of the newly released climate assessment model, which is far more sophisticated than the initial coarse manipulation approaches tried by this researcher at the start of this project. Lastly, these thoughts will be shared with EPA's BASINS team. If a tool with a viable, user-friendly interface can be created by a qualified programmer, planners and modelers alike will be better equipped to model and consider the impacts of land use change on water quality. This tool could be a key component for developing Total Maximum Loads, local watershed plans, local ordinances or regulations regarding site design or required BMP treatment, or simply studying future watershed conditions that incorporate elements of climate change and anthropogenic impacts.

6. The Existing Framework for the Management of Accotink and Pohick: Goals, Obligations, and Needs

6.1 Introduction

Fairfax County has two primary goals in managing the Accotink and Pohick watersheds: first, to maintain or reduce loadings of nitrogen to the Chesapeake Bay, and second, to prevent further degradation of the two watersheds. Based upon current and projected development and site design and best management practice (BMP) requirements for new development, Fairfax County is not on track to meet either of these goals. Fairfax County's public sector efforts are a good start, but more must be expected and/or required from the private sector if these goals are realistically achievable. The following chapter summarizes the results of this dissertation, and explores the implications of those results in the context of Fairfax County regulatory and policy objectives.

6.2 Summary of Results

This dissertation has shown that changes in hydrology and nutrient loadings have occurred in both the Accotink and Pohick watersheds over the last thirty years. Changes in watershed conditions as a result of urbanization help explain why all nine sites in these watersheds sampled by Fairfax County in 2005 scored a benthic rating of fair (two of nine), poor (three of nine), or very poor (four of nine)[31] (Fairfax County, 2006a). The dissertation has demonstrated that, absent effective BMPs, these watersheds will face further significant transformations over the next 20 to 25 years.

Chapter 5 explains that changes in hydrology will impact Accotink and Pohick more than changes in nitrogen loading. Using the climatic conditions from 1991-1993, modeling efforts indicate that, without effective offsets from Best Management Practices, projected peak flow could increase by as much as 60% for future land use scenarios in the Accotink watershed above Braddock Road. At Pohick above Route 1, the lowest base flow conditions are expected to decrease by 80% over 1991-1993 levels without effective infiltration practices in new urban areas. These figures do not include the impacts of climate change, which are expected to result in wetter conditions for the mid-Atlantic and more intense storms, leading to yet higher peak flows. This combined with warmer summer temperatures may lead in turn to greater evapotranspiration and lower base flows.

The greater extremes in daily flow and overall increase in annual flow will stress biological communities and result in continued rapid alteration of stream geomorpohology. Sediment from both stream scouring and construction sites will settle on the streambeds, clog watershed lakes, and ultimately settle in Gunston Cove, where flushing events and tidal flows can transport them to the Potomac channel. These sediments are likely to contain phosphorus, some of which will be released to the water column in the hypoxic conditions in lakebeds or resuspended during intense storm events.

[31] Scores of good or excellent are awarded in other streams in the county: however, no sites ranked this high in Pohick or Accotink.

In contrast, increases in nitrogen loading from precipitation-driven pollution will have greater impact on the downstream receiving waters in the mesohaline stretches of the Potomac and Chesapeake.

This dissertation presented six questions which have been answered throughout the course of the study or are discussed in this chapter. The questions and summary answers follow:

1. How much have the Accotink and Pohick watersheds urbanized over the last 30 years? (Answered from chapter 3)

From 1975-2004, urbanization in the watersheds has been significant. In the Pohick watershed, total urban area is estimated to have increased from 11.1% to 49.5%. The fastest rates of urbanization were estimated to have occurred from 1978-1987 and from 1993-2000. The Accotink watershed has increased from an estimated 36.3% to 61.9% urban land cover. The rate of urbanization was slower than in Pohick since a greater percentage of the Accotink watershed had been urbanized prior to 1975. There was a slight increase in the rate of urbanization in Accotink from 1996-2003, but the urbanization rate was slower than in Pohick until 2002. This recent increase in urbanization rate in Accotink could be due to increased in-fill development.

2. What has been the impact of urbanization on the Accotink and Pohick watersheds in the last 25 years, specifically on flow and nitrogen? *(Answered in chapter 4)*

Observed flow has increased significantly at the USGS gauge station on Accotink at Braddock Road. Increases have a correlation with time, which directly correlates with percentage of urban land use. Modeled results show peak flow and total annual flow volume increasing dramatically in Accotink and even more so in Pohick since 1975.

Nitrate loadings from stormwater appear to have increased in both Accotink and Pohick from the late 1980s to 2005. Ammonia loadings have decreased slightly. Overall nitrogen loadings to Gunston Cove have decreased substantially due to the enhanced treatment at the Noman Cole sewage treatment facility.

Nitrate concentrations are significantly different between the two Accotink stations and Pohick above the sewage treatment facility for 2005 despite only a 10% difference in urban area coverage. This could be due to a number of factors, including the increased coverage of forested and grassed riparian buffers removing a higher percentage of nitrogen; the increased coverage of BMPs on urban development in Pohick removing a greater percentage of nitrogen; the 10% of additional urban area present in Accotink; the older urban infrastructure in Accotink; and/or possible lower impervious area coverage per unit of 'urban' area in Pohick on average (lower density of impervious area for urban area).

3. How urban are these watersheds likely to become with future development? (Answered in chapter 5)

The Pohick watershed is predicted to be between 56.7% and 70.2% urbanized by 2030. The Accotink watershed is projected to be between 70.0% and 83.9% urban area. These results are based on several different types of models bracketing a range of assumptions.

4. What will this urbanization mean for future water quality? What changes need to be made to keep conditions at approximately their current level (to not allow degradation)? What changes are necessary to improve water quality? *(Answered in Chapter 5)*

Without the implementation of effective BMPs, hydrology will continue to be greatly altered by new development. Furthermore, nutrient loading will increase. Since few BMPs are 100% effective, a strategy that uses highly effective BMPs for new development should be combined with watershed retrofits in the rest of the County. In order to actually improve watershed conditions, more watershed retrofits will have to occur in combination with continued protection of sensitive habitats and enhanced requirements for BMP effectiveness.

Maintaining or reducing nutrient loadings may be easier than maintaining or restoring the hydrologic regime with available resources, currently available BMP technology, and benefits offered by resource protection areas.

5. Are proposed changes technically feasible with current approaches? If so, what level of efficiency needs to be obtained? *(Discussed in chapter 5)*

   Yes. To maintain present hydrology, no net change in pre-development hydrology can be allowed. This means that a combination of effective BMPs that incorporate both infiltration from clean sites and evapotranspiration must be more widely encouraged. Options for reaching these goals include using treatment trains (multiple BMPs with complementing strengths), charging significant impervious surface fees, offering credits for the most effective or greenest site designs, and/or setting up trading mechanisms for stormwater credits. By encouraging reductions in existing development, the County can maintain current hydrologic conditions. Though these conditions are currently negatively affecting the channel, we can mitigate the scale of future impacts.

   In terms of nitrogen reduction, if approximately 40% of nitrate is removed from loadings from new development and approximately 10% of nitrate is removed from loadings from existing development, expected nitrate loadings would remain at approximately 2005 levels in future development scenarios. Removing 15% of nitrate from already developed watersheds reduces nitrate loading for all watersheds in all future land use scenarios. The same is true for estimates of total nitrogen loadings. However, using existing BMP approaches, it is typically easier to remove a higher percentage of total nitrogen than nitrate.

6. What are the specific threats faced by each aquatic ecosystem? How do these specific threats integrate with county watershed managers' goals? *(Discussed in chapters 5 and 6)*

    Changes in hydrology are more likely to impact the Accotink and Pohick streams and headwaters. Furthermore, the increased sediment loadings that will accompany these changes will impact sediment deposition in all watershed lakes and in Gunston Cove. Increases in nitrogen loading pose little risk to Gunston Cove and Accotink since these ecosystems are phosphorus limited for most of the year. However, increases in nitrogen loadings would pose the most risk to the Chesapeake Bay where hypereutrophication is contributing to the expansion of the Chesapeake Bay dead zone. Additionally, Accotink is currently receiving high levels of road salt in the winter months which are likely impacting aquatic communities.

    These threats of hydrologic alteration increased nutrient loading greatly impact two of the primary drivers of Fairfax County's stormwater program: not allowing future degradation of streams and reducing loadings of nitrogen from urban stormwater sources to the Chesapeake Bay. The threat caused by road salt application is in conflict with the required public need of safe roadways during winter. Application of a substitute chemical such as CMA instead of road salt by Fairfax County and the Virginia Department of Motor Vehicles is a viable alternative to minimize the threat from road salt application.

6.3 Restatement of Hypotheses

The hypothesis of this dissertation is: as the Accotink and Pohick watersheds have urbanized, noticeable deterioration in water quality and hydrologic conditions was observable. Modeling, monitoring, and statistical efforts were used to quantify past degradation. The hypothesis was supported by observed increases in estimated loadings of nitrogen to both Accotink and Pohick relative to a historic data set. Furthermore, as urbanization increased, observed changes in hydrology at the Accotink gauge station were noted. Two modeling tools showed that historic changes to hydrology associated with land use were substantial for both watersheds. This hypothesis would have been further supported if water quality conditions had been comparable between the Accotink and Pohick sites as urbanization in Pohick approached the levels present in Accotink. Relative TSS concentrations are higher in Pohick than in Accotink, indicating increased stream scouring and likely increased runoff from construction sites. However, expected results related to increasing nutrient concentrations from stormwater-related pollution, particularly nitrogen, did not occur. In fact, nutrient concentrations in the Pohick watershed are statistically significantly lower ($p=0.0007$) than in Accotink. This is a possible indication that BMPs, including protection of buffers, are being somewhat effective in nutrient removal in this more newly urbanized watershed. Other possible explanations include age of development or influence of septic systems or sanitary sewer infrastructure. Nonetheless, it appears that some increase in relative loadings from stormwater sources in both Pohick and Accotink has occurred (although loadings from traditional point sources have decreased).

Modeling indicates that future urbanization will continue to degrade these watersheds. The primary cause of degradation will be changes to hydrology; however, nutrient loadings increase slightly as well. If effective BMPs are encouraged on a large scale – those that encourage groundwater infiltration and evapotranspiration – and sensitive areas (resource protection areas) continue to be protected, then future impacts to the watersheds will not be as significant as modeled.

The alternative hypothesis of this dissertation is: With increasing implementation of BMPs since 1993, continued urbanization resulted in minimal degradation of the Accotink and Pohick watersheds and the downstream loading of nitrogen did not increase significantly. As discussed, the results indicate that water quality concentrations of nitrate in Pohick are significantly lower than those in Accotink. Because Pohick reached nearly the same level of relative urbanization as Accotink by 2005, each unit of urbanized area may be exporting less nitrogen. If this were attributable to BMP performance, this alternative hypothesis is partially supported. However, estimates do show that nitrogen loading has increased from Pohick with increasing urbanization. Hence, this alternative hypothesis is only weakly supported.

Based on modeling efforts, which do not take BMP effectiveness into account, one can infer that aggressive countermeasures need to be made to prevent the extreme changes in hydrology expected from urbanization. Less significant, but still substantial efforts need to be made to maintain nitrogen loadings at current levels. Using these theories and published BMP performance data, informed discussion of the current and past approaches can be made. Based on the performance data of dry detention basins, one of the favored BMPs of county developers, implementation of these BMPs alone, or even in combination with protected riparian buffers, will not suffice to prevent further degradation of Accotink and Pohick. More aggressive measures are needed to protect watershed hydrology and to prevent the negative effects of changes to this hydrology. Current BMPs may be removing some nitrogen from stormwater coming from urban land use. This removal may be enhanced by other policies currently being implemented, such as riparian buffer protection. Nonetheless, based on the results of this dissertation, this nitrogen removal may not be enough to meet the goals of the County, the Commonwealth, or the region.

6.4 Nonpoint Source and Stormwater Related Regulatory Requirements and County Goals applicable to Accotink and Pohick

The United States Environmental Protection Agency (EPA) began regulating urban stormwater pollution following the addition of section 402(p) to the Clean Water Act in 1987 (Water Environment Federation, 1997). In 1990, the EPA passed the Phase I regulations, which required that operators of municipal separate storm sewer systems in jurisdictions with populations greater than 100,000 get NPDES permit coverage. Under these regulations, Fairfax County was required to get an NPDES permit and start a regulated stormwater program.

Fairfax is currently on their 3rd iteration of permit coverage (issued 01/24/2002). The permit requires that the county set the goal of not allowing any “discharge of pollutants in quantities that would cause a violation of State Water Quality Standards,” that they “limit increases in the discharge of pollutants from storm water as a result of development and significant re-development,” and that they maintain roadways to “minimize the discharge of pollutants, including those pollutants related to deicing or sanding activities” (Virginia Department of Environmental Quality, 2002). The permit specifically outlines that Fairfax must complete watershed management plans, must complete outreach activities, conduct monitoring, and must strive to reduce nutrient and pollutant loadings. However, nowhere in the permit does it require that Fairfax maintain pre-development hydrology. The permit does not discuss flow rate, discharge volume, or other major hydrological concerns, all of which are quite clearly impacting the streams and quite likely contributing to (or causing) impairments of water quality standards. This permit expired as of January 24, 2007 and the county has submitted an application for a new permit to the Department of Conservation and Recreation (Danielle Wynne, personal communication, 3/13/2007). The work of this dissertation has shown that, for the county watersheds themselves, changes in hydrology are more significant than changes in nutrient loading. As part of its regulated stormwater (MS4) program, based upon the wording of the existing Virginia Discharge Elimination Permit (VPDES) permit, Fairfax will get minimal credit if it undertakes actions to meet these goals, which will improve watershed conditions and meet water quality standards.

Nonetheless, based in part on the work of numerous scientist-practitioners, regulations are shifting to encourage the maintenance of pre-development hydrology, using popular concepts such as low impact development. The Commonwealth of Virginia passed House Bill 1177, which charges municipalities to encourage low impact development to the maximum extent practicable (General Assembly of Virginia, 2004). Furthermore, it allows municipalities to adopt fees for permit applications for land disturbing activities (construction) on greater than 1 acre or reduced fees for disturbing activities on greater than 2500 square feet in Chesapeake Bay Preservation Act localities. The bill also allows localities to adopt regulations that are stricter than state standards if appropriate conditions are met:

> "Localities are authorized to adopt more stringent stormwater management ordinances than those necessary to ensure compliance with the Board's minimum regulations, provided that the more stringent ordinances are based upon factual findings of local or regional comprehensive watershed management studies or findings developed through the implementation of a MS4 permit or a locally adopted watershed management study and are determined by the locality to be necessary to prevent any further degradation to water resources or to address specific existing water pollution including nutrient and sediment loadings, stream channel erosion, depleted groundwater resources, or excessive localized flooding within the watershed and that prior to adopting more stringent ordinances a public hearing is held after giving due notice" (10.1-603)

For Fairfax County, more stringent regulations can be written to minimize impacts from flow driven effects to implement their MS4 permit or as part of the watershed plans. Watershed plans must therefore contain a detailed rationale the most appropriate solutions to wet weather related issues for the watersheds.

In July of 2006, the Virginia stormwater management program (chapter 60) was modified to require that new development maintain the existing predevelopment site hydrology as nearly as practicable (Virginia Department of Conservation and Recreation, 2006). However, the code specifies that these goals can be met if the peak flow is not higher than predevelopment conditions, and the flow is detained for 48 hours. The resultant stormwater volume will still increase, since the total volume of additional runoff will increase, despite the mitigation of the peak flow event. Erosion, scouring, and associated impacts will continue at elevated levels, due to the increase in frequent ‘light’ flooding events. Rosgen (1996) noted that frequent floods, such as the 1 to 2 year storm, have a more significant impact on channel formation because of their relative frequency. The ordinance seeks to protect water quality and prevent erosion and flooding, and it may help somewhat, but the number of powerful channel forming events will increase.

Fairfax is also charged to meet the goals of the Virginia Chesapeake Bay Preservation Act. This act, originally passed in 1988, set the goal of reducing the precipitation-driven nutrient pollution, including stormwater and other nonpoint source pollution, to the Chesapeake Bay (Code of Virginia, 1988). These regulations encourage the conservation of resource protection areas as one of the primary mechanisms for meeting these goals. The Act created the Chesapeake Bay Local Assistance Board, which had the authority to assist localities and make legally binding policy decisions. The Act clearly states that local governments are responsible for land use decisions: hence, responsibility for establishing land use requirements that meet the goals of the Act currently rests with the county. In 2001, the Local Assistance Board made recommendations to broaden the scope of resource protection areas. Fairfax originally designated resource protection areas, areas that include streams and riparian buffers, in 1993. In 2003, the County Board revised the Chesapeake Bay Preservation Ordinance to expand the resource protection areas (Fairfax County, 2005). According to the county's stormwater status report (2006c), 17.4 square miles of additional county area was added to an existing 55.3 square miles for a total of 72.7 square miles in resource protection areas.

The final regulatory requirement for the county is to protect impaired waters and those with Total Maximum Daily Loads (TMDLs) from additional degradation from their pollutant of concern. Both Accotink and Pohick streams have multiple impairments, as do Accotink Bay, Pohick Bay, and Gunston Cove (Fairfax County, 2006a). The impairments' causes include toxics such as high PCB concentrations in fish, bacterial contamination from *E. coli* and fecal coliform, low dissolved oxygen concentrations in portions of Pohick Bay, low levels of submerged aquatic vegetation in the bays, and benthic macroinvertebrate impairments in multiple waters. While this dissertation does not address most of the root causes of these impairments, it does address one of the primary drivers of benthic impairments: flow and sedimentation. Fairfax County watershed managers must integrate the regulatory requirements of the TMDL program with their other regulatory obligations. They are legally obligated to establish watershed management plans to restore these waters to their designated uses. When combined with the other regulatory drivers, implementing policies that will allow these waters to reach their officially designated use will be no small feat. However, a comprehensive approach may be more effective in the long run.

Finally, though not a regulatory driver, the county has clearly laid out goals for the watersheds. In 2001, Fairfax completed its stream protection strategy, in which the county conducted initial assessments of watersheds and laid out the initial goals for each watershed (Fairfax County, 2001). As discussed in Chapter 1, the County stated that all of Accotink and most of Pohick were watershed restoration II areas, which means that policies should be implemented which prevent further degradation and actively improve the watersheds to comply with Chesapeake Bay initiatives, TMDL regulations, and other water quality standards. The county has also set out to protect the few non-degraded areas in the northwest of Pohick and to actively remedy causes of stream degradation in other portions of Pohick. In short, the county has established goals of both complying with all of the requirements of the regulations listed above and preventing further degradation of these streams. To reach these goals, the county will require strong political support and a substantial financial investment, and will have to take a more proactive role in demanding more effective structural BMP construction from county stakeholders, particularly those wishing to significantly modify or develop land within the county.

Fairfax County has historically been reasonably proactive by establishing policies and the management infrastructure attempting to protect water quality. Fairfax required water detention BMPs county-wide in 1972 and water quality control devices such as dry detention basins in 1993 (Kumar *et al.*, 2005). At the end of 2005, there were 1178 county maintained and 2251 privately maintained stormwater control facilities (structural BMPs) (Fairfax County, 2006c). As stream restorations began to show greater potential for improving watershed conditions, the county restored several tributaries in the Accotink watershed[32] (Schagrin *et al.*, 1998). The stormwater planning division is creating watershed plans as one of the requirements of their MS4 permit. In fiscal year 2005, the county created seven ecologist positions in its stormwater management division to aid in assessing the condition of the county's streams and increased that to nine ecologist positions in the 2007 fiscal year (Fairfax County, 2004, 2007). The budget of the stormwater planning division increased from $628,065 to $1,849,250(Fairfax County, 2002, 2006b), and the scope and purpose of the division has expanded to focus on a more inclusive approach to stormwater management. In 2006, the county set aside 1 penny per hundred dollars of assessed valuation of its real estate tax to guarantee adequate funding for stormwater management in the County (Fairfax County, 2006c). In terms of managing public areas and public assets, the county is addressing stormwater issues progressively on many stormwater related issues including management, finance, monitoring, stream restoration, small scale watershed restoration (on public sites), public planning, inspecting and maintenance of existing facilities, and establishment of the resource protection areas.

The county's incentive and mandatory regulatory programs for adoption and implementation of the most effective Best Management Practices by the private sector

[32] Fairfax City is also restoring portions of the Accotink stream channel, the most recent restoration occurring between Route 50 and Old Lee Highway in 2006. The United States Environmental Protection Agency, United States Geological Survey, and George Mason University are all engaged in monitoring below this site.

and local residents are not as well developed. The vast majority of the 3400 BMPs identified in the County are underground or rooftop detention or ponds designed for water detention (Fairfax County, 2006c). These BMPs are not primarily designed for shallow groundwater recharge or for enhanced evaporation. Furthermore, many of these might not be the most effective BMP approaches for nutrient removal[33]. No incentive or fee reduction programs are in place for installation of stormwater infrastructure by local individuals. Most large scale commercial projects still include plans for underground detention rather than techniques that infiltrate or allow for evaporation such as bioretention or green roofs. Furthermore, structural BMP construction often means one dry detention basin in the corner of a development, hidden away from the rest of the development and not integrated into the public space or usable by the local community. Without either voluntary or mandatory involvement from the private sector and the local community, those commendable efforts at the public county level will be more than offset by changes in the landscape as a result of increasing urbanization.

6.5 Will goals be met using the existing approach?

Implementing only partially effective regulations will have substantial economic and ecological costs. Historic changes to watersheds as a result of land use changes have already proven quite costly both economically and ecologically. Examples include sedimentation in Lake Accotink, which reduced the average depth of the lake to 4 feet.

[33] It is difficult to categorize all of the BMPs. The stormwater report categorizes most county maintained stormwater ponds as ‘on-site ponds;’ however, wet retention ponds or created wetlands within dry detention ponds have considerably enhanced nutrient removal capacity compared to traditional dry detention basins.

Fairfax County has spent $6.15 million, in large part to dredge the lake (Fairfax County Environmental Quality Advisory Council, 2006). Furthermore, this is not the first time the County has had to drain the lake (R Chris Jones, Personal Communication, 3/26/07). Heavy rains in June of 2006 caused over $1 million in damage to newly constructed portions of the Accotink stream valley trail that had to be repaired (Fairfax County Park Authority field employee, personal communication, 11/14/2006). Continued scouring and erosion threatens existing homes, eroding steep banks under yards, and, in some cases, reaching dangerously near the residences themselves. Much of the county sanitary sewer infrastructure lies near streams; with increased scour, the risk of this vital infrastructure being damaged increases. Separation of the streams from the floodplain results in less nutrient removal and other ecosystem services provided by the riparian zone, compounding the concern with loadings from urban areas (thus mitigating some of the effectiveness of protecting riparian buffers as resource protection areas). Streams with extremely low base flow will have increased difficulty supporting aquatic species like the catadromous American eel, which directly impacts both the ecosystem and fisheries in the Bay. Jowett et al. (2005) found that reductions in fish populations were noted with the lowest flows, and the greatest reductions occurred in the year where there was the lowest flow. Direct impacts to the Chesapeake Bay are equally well documented, with nitrogen loading suspected as the primary contributor to the expanding dead zone (Correll, 1987).

These economic and ecological costs will only increase if the changes caused by new urbanization to hydrology and nutrient loadings are not completely mitigated or offset. Even with effective action, some future impacts will be felt due to the lag effect seen in channel widening to accommodate flow with current levels of urbanization. If effective approaches are instituted immediately, the magnitude of future damage to Accotink and Pohick themselves will be minimized. In the long-term, this may allow for the restoration of county streams.

Fairfax County has made significant steps toward establishing an effective program to meet the goals of the county. The county's decision to dedicate $0.01 per $100.00 of assessed valuation from its real estate taxes toward stormwater management produces a sizable, stable funding source to maintain existing stormwater infrastructure and to plan for the future. Use of demonstration projects allows the county to lead by example, while creating infrastructure that is both ecologically friendly and cost effective in the long term. Establishment of resource protection areas and minimizing development in those areas will greatly contribute to the goals of reducing nitrogen and phosphorus loadings to the Chesapeake Bay. However, both green field and in fill development continue in Fairfax County and in Accotink and Pohick watersheds. The County has countless stormwater drains that discharge directly into streams, effectively bypassing the riparian buffers and much of the nutrient removal that would occur in those buffers. While the County may spend more than $20 million dollars in fiscal year 2007 to maintain existing stormwater infrastructure, restore streams, retrofit watersheds, and monitor results, new developments are still being constructed with minimal post-construction BMP requirements. Without more stringent stormwater design requirements implemented and supported by the majority of the community, efforts by the County will continue to be offset by increasing impacts from new urban development. In short, passively encouraging more efficient BMPs will not be enough to meet the regulatory requirements or watershed goals of the County.

In order to meet these goals and regulatory requirements, county planners must use more effective and aggressive techniques to achieve greater effectiveness of BMPs on private development. As illustrated in previous chapters of this dissertation, if each acre of new urban area in the future has similar impact on stream characteristics as urban area in the past, by 2030 county planners may be dealing with a 25 to 40% (or more) increase in annual flow and an 80% decrease in lowest base flows relative to 1992 land use conditions.

With the increasing cost of land in the County in recent years, the use of land intensive dry detention basins alone have a higher opportunity cost to developers (Sample *et al.*, 2003; Thurston, 2006), not to mention that they provide little benefit in terms of additional ecosystem or recreational services. The median removal efficiency rate of dry detention basins has been documented to be a mere 5% of nitrate and 26% of total nitrogen (O'Shea *et al.*, 2002). Stormwater detention alone is not enough to protect stream channels (MacRae, 1996), nor is it enough to recharge shallow groundwater for maintenance of base flow. As land becomes progressively scarcer, increased pressure will require stormwater BMPs that use less land, provide greater ecosystem services, and/or provide greater services to county residents. Numerous papers, articles, and manuals have been completed on techniques that provide increased performance and/or enhanced services (Booth *et al.*, 2004; Brattebo & Booth, 2003; Casey Trees Endowment

Fund and Limno-Tech Inc., 2005; Davis, 2005; Holman-Dodds *et al.*, 2003; Hood & Clausen, 2006; Kane, 2005; Matteo *et al.*, 2006; Minnesota Stormwater Steering Committee, 2006; O'Shea *et al.*, 2002; Pollock, 1992; Prince George's County (Md.), 2000a; US EPA, 2000c, 2005b; Villarreal *et al.*, 2004; Williams & Wise, 2006). The trick is to broaden the recognition among all stakeholders that the dry detention basin as a one-size-fits-all approach for private development will not work. Instead, the County must design a combination of stick and carrot policy approaches that rapidly changes the paradigm, and consequently, the design of stormwater mitigation in new development.

Fairfax County is actively retrofitting many older dry detention basins to function more as constructed wetlands (Danielle Wynne, personal communication, 3/13/07). However, the County's abilities are limited by resources. Despite their impressive maintenance plan and regular BMP inspections at least once every permit term (every 5 years), Fairfax cannot possibly increase the efficiency or performance of every underperforming dry detention basin. The gains the County may be making are more than offset by new development using dated techniques. In order to reach the County's regulatory obligations and goals, improved stormwater BMP performance and/or better site design is required for all new urban development. Designs and approaches to mitigate stormwater must allow some redundancy for failure, remain cost effective, and be properly designed and constructed, while maximizing other ecosystem and/or resident services. In addition to requiring or strongly encouraging BMP treatment train approaches or better performing BMPs, there must be increased integration of environmental planning into the design and construction phases in order to have sustainable post-development urban land use.

Achieving the goal of maintaining or improving watershed conditions in Accotink and Pohick will require strong commitment by Fairfax County watershed managers, board members, and developers, but this goal is attainable. Incentive programs, regulatory requirements, and enhanced assistance are all techniques that have increased the participation of sustainable design in other jurisdictions, including Chicago, Portland, and Seattle. High stormwater utility and construction permit fees, combined with credits that genuinely reward property owners and developers with significant price reductions for sustainable or green designs, reduce the cost disparity between the traditional ‘business as usual’ approach and an approach which meets long term environmental needs. Grant programs funded by these fees can encourage existing owners to complete small-scale retrofits, which can cumulatively make a significant and long-term positive impact. Hence, there are financing approaches that work and can be revenue neutral to both the County and good actors. Results from this study and others have shown significant action is needed to meet County goals and requirements. A strong body of scientific literature indicates that there are more effective approaches than what is currently commonly implemented for new private development. Furthermore, the County has a legal mechanism in the form of watershed planning and other programs to create the legal framework and detailed planning necessary to implement effective strategies. Finally, Fairfax County’s board of supervisors have shown through funding initiatives that they support effective stormwater management that can be combined with continued economic growth. Though this study has shown that continued urbanization will significantly impact Accotink and Pohick without effective management of both hydrological alterations and water quality constituent loadings, with effective watershed management and planning and continued strong political backing from the board of supervisors, the goals laid out by the County and required by various statute and regulations are achievable.

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Appendix A: Example LOADEST Input and Output using the Default Model

<u>Control File:</u>
header.txt
calibration.txt
estimation.txt

<u>Header File:</u>

#Title
Site 13 2005 Calibration
#PRTOPT
1
#SEOPT
3
#LDOPT
0
#MODNO
0
#NCONST
5
nitrate 1 1
ammonia 1 1
tp 1 1
tss 1 1
vss 1 1

<u>Calibration File:</u>

20050305	1200	32.16414793	0.95	0.017	0.003	8	6
20050319	1200	25.73131835	0.91	<.002	0.007	6	4
20050402	1200	1470.973699	0.93	<.002	0.173	510	66
20050408	1200	201.5619937	1.33	<.002	0.018	21	7
20050416	1200	30.0198714	0.62	0.005	0.011	5	5
20050429	1200	27.87559487	0.94	0.010	-100.0005		4
20050515	1200	143.6665274	1.47	0.052	0.065	59	14
20050530	1200	27.87559487	1.37	0.011	0.022	4	2
20050611	1200	25.73131835	1.11	0.021	0.019	7	3
20050626	1200	13.93779744	0.89	0.005	-100.0005		4
20050710	1200	30.0198714	0.66	0.005	0.029	6	6
20050724	1200	14.79550805	0.35	<.002	0.011	3	3
20050807	1200	10.50695499	0.50	<.002	-100.0008		4
20050905	1200	5.146263669	0.38	<.002	0.014	6	5
20050910	1200	3.859697752	0.44	0.003	0.010	2	2
20050924	1200	3.430842446	0.22	0.018	0.011	4	4
20051008	1200	3559.499038	0.85	0.086	0.132	91	23
20051022	1200	180.1192284	0.98	0.007	0.058	81	22
20051105	1200	10.50695499	0.27	0.006	-100.0003		3
20051108	1200	9.434816727	0.30	0.032	0.019	2	3
20051119	1200	10.9358103	0.46	<.002	0.038	5	4
20051204	1200	60.03974281	0.78	0.055	0.050	9	5
20051211	1200	27.87559487	0.78	0.062	0.017	5	3
20051231	1200	30.0198714	1.29	<.002	0.019	5	8

Example Estimation File Text (Excerpt):

20050101	1200	23.58704182	20050218	1200	27.87559487
20050102	1200	20.37062702	20050219	1200	25.73131835
20050103	1200	20.15619937	20050220	1200	25.73131835
20050104	1200	20.15619937	20050221	1200	60.03974281
20050105	1200	34.30842446	20050222	1200	77.19395504
20050106	1200	23.58704182	20050223	1200	27.87559487
20050107	1200	21.01390998	20050224	1200	57.89546628
20050108	1200	64.32829586	20050225	1200	70.76112545
20050109	1200	23.58704182	20050226	1200	47.17408363
20050110	1200	21.22833763	20050227	1200	45.0298071
20050111	1200	20.79948233	20050228	1200	53.60691322
20050112	1200	25.73131835	20050301	1200	124.3680387
20050113	1200	27.87559487	20050302	1200	53.60691322
20050114	1200	2037.062702	20050303	1200	36.45270099
20050115	1200	87.91533768	20050304	1200	32.16414793
20050116	1200	49.31836016	20050305	1200	32.16414793
20050117	1200	40.74125405	20050306	1200	32.16414793
20050118	1200	30.0198714	20050307	1200	30.0198714
20050119	1200	27.87559487	20050308	1200	390.2583282
20050120	1200	27.87559487	20050309	1200	64.32829586
20050121	1200	25.73131835	20050310	1200	42.88553058
20050122	1200	23.58704182	20050311	1200	40.74125405
20050123	1200	23.58704182	20050312	1200	36.45270099
20050124	1200	21.44276529	20050313	1200	32.16414793
20050125	1200	23.58704182	20050314	1200	27.87559487
20050126	1200	27.87559487	20050315	1200	27.87559487
20050127	1200	25.73131835	20050316	1200	27.87559487
20050128	1200	21.44276529	20050317	1200	27.87559487
20050129	1200	20.79948233	20050318	1200	27.87559487
20050130	1200	34.30842446	20050319	1200	25.73131835
20050131	1200	30.0198714	20050320	1200	27.87559487
20050201	1200	27.87559487	20050321	1200	25.73131835
20050202	1200	25.73131835	20050322	1200	23.58704182
20050203	1200	30.0198714	20050323	1200	771.9395504
20050204	1200	45.0298071	20050324	1200	113.646656
20050205	1200	42.88553058	20050325	1200	55.75118975
20050206	1200	36.45270099	20050326	1200	45.0298071
20050207	1200	32.16414793	20050327	1200	38.59697752
20050208	1200	32.16414793	20050328	1200	1573.898972
20050209	1200	32.16414793	20050329	1200	167.2535692
20050210	1200	62.18401933	20050330	1200	49.31836016
20050211	1200	27.87559487	20050331	1200	36.45270099
20050212	1200	27.87559487	20050401	1200	40.74125405
20050213	1200	25.73131835	20050402	1200	1470.973699
20050214	1200	85.77106115	20050403	1200	525.3477495
20050215	1200	77.19395504	20050404	1200	122.2237621
20050216	1200	36.45270099	20050405	1200	77.19395504
20050217	1200	32.16414793	20050406	1200	64.32829586

Example Summary Output File (Site 1: 1986-1992)

```
                    LOADEST
        A Program to Estimate Constituent Loads
     U.S. Geological Survey, Version: MOD36 (Sep 2004)
     --------------------------------------------------
Site 13 2005 Calibration 13

Constituent: nitrate
----------------------------------------------------------------------
  Constituent Output File Part Ia: Calibration (Load Regression)
----------------------------------------------------------------------

Number of Observations          :    24
Number of Uncensored Observations:    24
"center" of Decimal Time        :  2005.581
"center" of Ln(Q)               :    4.5366
Period of record                :    2005-2005

Model Evaluation Criteria Based on AMLE Results
-----------------------------------------------
Model #    AIC       SPPC
----------------------------------
 1        1.328     -17.120
 2        0.786     -11.193
 3        1.277     -17.086
 4        1.001     -14.363
 5        0.804     -12.003
 6        0.778     -12.287
 7        0.785     -12.369
 8        0.738     -12.395
 9        0.708     -12.620

Akaike Information Criterion (AIC) and Schwarz Posterior Probability
Criteria (SPPC) did not select same best fit model.  Model # 9
selected on basis of AIC. (Model # 2 would have been selected based on SPPC)

Selected Model:
---------------
Ln(Load) = a0 + a1 LnQ + a2 LnQ^2 + a3 Sin(2 pi dtime) + a4 Cos(2 pi dtime)

         + a5 dtime + a6 dtime^2
where:
    Load  = constituent load [kg/d]
    LnQ   = Ln(Q) - center of Ln(Q)
    dtime = decimal time - center of decimal time

    Model Coefficients
     a0      a1      a2      a3      a4      a5      a6
    -----------------------------------------------------------------------
AMLE   4.7198   1.1831  -0.0423  -0.5336   0.7471   1.0380   9.4630
MLE    4.7198   1.1831  -0.0423  -0.5336   0.7471   1.0380   9.4630
LAD    4.6338   1.1948  -0.0444  -0.5467   1.0580   1.0120  12.0498
```

AMLE Regression Statistics

R-Squared [%] : 98.45
Prob. Plot Corr. Coeff. (PPCC) : 0.9689
Serial Correlation of Residuals: 0.1480

Coeff.	Std.Dev.	t-ratio	P Value
a0	0.4587	10.29	5.589E-12
a1	0.0424	27.94	7.236E-22
a2	0.0241	-1.76	4.518E-02
a3	0.2441	-2.19	1.473E-02
a4	0.5453	1.37	1.129E-01
a5	0.6744	1.54	7.680E-02
a6	6.1826	1.53	7.837E-02

Correlation Between Explanatory Variables

Explanatory variable corresponding to:

	a1	a2	a3	a4	a5
a2	0.0000				
a3	-0.2083	0.4473			
a4	-0.2872	0.3700	0.0060		
a5	-0.2164	0.2507	0.9143	0.0030	
a6	0.2348	-0.3791	-0.0114	-0.9809	0.0000

Additional Regression Statistics

	Residual Variance	Turnbull-Weiss Stat	DF	PL
AMLE	0.089	8.32	1	3.926E-03
MLE	0.089	8.32	1	3.926E-03

Constituent Output File Part Ib: Calibration (Concentration Regression)

AMLE Regression Statistics

Model # 9 was selected for the load regression (PART Ia) and is used here:

Ln(Conc) = a0 + a1 LnQ + a2 LnQ^2 + a3 Sin(2 pi dtime) + a4 Cos(2 pi dtime)

+ a5 dtime + a6 dtime^2

where:

Conc = constituent concentration
LnQ = Ln(Q) - center of Ln(Q)
dtime = decimal time - center of decimal time

Concentration Regression Results

R-Squared [%] : 78.11
Residual Variance : 0.0888

Coeff.	Value	Std.Dev.	t-ratio	P Value
a0	-0.7115	0.4587	-1.55	7.470E-02
a1	0.1831	0.0424	4.32	2.446E-05
a2	-0.0423	0.0241	-1.76	4.518E-02
a3	-0.5336	0.2441	-2.19	1.473E-02
a4	0.7471	0.5453	1.37	1.129E-01
a5	1.0380	0.6744	1.54	7.680E-02
a6	9.4630	6.1826	1.53	7.837E-02

Constituent Output File Part IIa: Estimation (test for extrapolation)

Load Estimates for 20050101-20051231

Streamflow Summary Statistics [cfs]

Data	Mean	Minimum	10th Pct	25th Pct	Median	75th Pct	90th Pct	Maximum
Cal.	248.	3.	5.	11.	28.	53.	836.	3559.
Est.	85.	2.	8.	15.	28.	45.	123.	3559.

The maximum estimation data set steamflow does not exceed the maximum calibration data set streamflow. No extrapolation is required.

Constituent Output File Part IIb: Estimation (Load Estimates)

Load Estimates for 20050101-20051231

Load Estimates [KG/DAY]

AMLE Load Estimates

	N	Mean Load	95% Conf.Intervals Lower	95% Conf.Intervals Upper	Std Error Prediction	Standard Error
Est. Period	365	233.51	123.80	402.36	71.83	70.59

MLE Load Estimates

	N	Mean Load	Standard Error
Est. Period	365	233.63	70.22

LAD Load Estimates

	N	Mean Load	Standard Error
Est. Period	365	274.71	128.21

Summary Statistics - Estimated Loads [KG/DAY]

	Min.	25th Pct	Med.	75th Pct	90th Pct	95th Pct	99th Pct	Max.
AMLE	0.91	22.99	57.56	112.15	340.08	914.49	6211.98	6877.22
MLE	0.91	22.99	57.56	112.17	340.09	914.51	6212.01	6912.05
LAD	0.87	22.83	66.43	132.22	414.14	917.86	6801.94	16064.93

Summary Statistics - Estimated Concentrations [MG/L]

	Min.	25th Pct	Med.	75th Pct	90th Pct	95th Pct	99th Pct	Max.
AMLE	0.18	0.57	0.88	0.98	1.18	1.37	1.49	1.63
MLE	0.18	0.57	0.88	0.98	1.18	1.37	1.49	1.63
LAD	0.18	0.57	0.91	1.15	1.47	1.80	2.84	3.22

WARNING: Maximum estimated concentration exceeds twice the maximum calibration concentration of 1.470 MG/L

<u>Individual Daily Load Calculations (abbreviated):</u>

Date	Time	Flow	Loads Estimated by: AMLE	MLE
20050101	1200	2.359E+01	4.5713E+01	4.6428E+01
20050102	1200	2.037E+01	3.8362E+01	3.8902E+01
20050103	1200	2.016E+01	3.8230E+01	3.8720E+01
20050104	1200	2.016E+01	3.8588E+01	3.9038E+01
20050105	1200	3.431E+01	7.6047E+01	7.6894E+01
20050106	1200	2.359E+01	4.7933E+01	4.8403E+01
20050107	1200	2.101E+01	4.1678E+01	4.2045E+01
20050108	1200	6.433E+01	1.6725E+02	1.6873E+02
20050109	1200	2.359E+01	4.8908E+01	4.9268E+01
20050110	1200	2.123E+01	4.2992E+01	4.3276E+01
20050111	1200	2.080E+01	4.2087E+01	4.2338E+01
20050112	1200	2.573E+01	5.5442E+01	5.5748E+01
20050113	1200	2.788E+01	6.1590E+01	6.1900E+01
20050114	1200	2.037E+03	6.8772E+03	6.9121E+03
20050115	1200	8.792E+01	2.5107E+02	2.5220E+02
20050116	1200	4.932E+01	1.2613E+02	1.2663E+02
20050117	1200	4.074E+01	1.0006E+02	1.0041E+02
20050118	1200	3.002E+01	6.8548E+01	6.8755E+01
20050119	1200	2.788E+01	6.2531E+01	6.2699E+01
20050120	1200	2.788E+01	6.2606E+01	6.2757E+01
20050121	1200	2.573E+01	5.6610E+01	5.6732E+01
20050122	1200	2.359E+01	5.0687E+01	5.0783E+01
20050123	1200	2.359E+01	5.0697E+01	5.0782E+01
20050124	1200	2.144E+01	4.4849E+01	4.4916E+01
20050125	1200	2.359E+01	5.0675E+01	5.0743E+01
20050126	1200	2.788E+01	6.2668E+01	6.2744E+01
20050127	1200	2.573E+01	5.6562E+01	5.6622E+01
20050128	1200	2.144E+01	4.4718E+01	4.4759E+01
20050129	1200	2.080E+01	4.2937E+01	4.2972E+01
20050130	1200	3.431E+01	8.1107E+01	8.1169E+01
20050131	1200	3.002E+01	6.8464E+01	6.8509E+01

Appendix B: Select LOADEST Input and Output using the User Defined Model

Header File:

```
#Title
Site 1 1988 to 2005 Calibration ULU, Buffer, and BMP
#PRTOPT
1
#SEOPT
3
#LDOPT
3
#NSEAS
4
#SBEG -  SEND
1201     0229
0301     0531
0601     0831
0901     1130
#MODNO
99
#NADDL(8)
3
#NEXPL(9)
5
#DVNAME TRANS (I=1,NEXPL)
#################################################################
Q       LN
Q       LNSQ
ADDL1   NONE
#ADDL1 is percent urban land use #
ADDL2 NONE
#ADDL2 is percent urban buffer infringement on urban buffer
ADDL3   NONE
#ADDL3 is estimated BMP implementation
#NCONST
5
nitrate                              1   1
ammonia                              1   1
tp                                   1   1
tss                                  1   1
vss                                  1   1
```

Summarized Output File (select portions abbreviated):

```
                         LOADEST
               A Program to Estimate Constituent Loads
           U.S. Geological Survey, Version: MOD36 (Sep 2004)
           -------------------------------------------------

 Site 1 1988 to 2005 Calibration ULU, Buffer, and BMP
```

Constituent: nitrate

Constituent Output File Part Ia: Calibration (Load Regression)

Number of Observations : 105
Number of Uncensored Observations: 101
"center" of Decimal Time : 1997.714
"center" of Ln(Q) : 3.0924
Period of record : 1988-2005

Model Evaluation Criteria Based on AMLE Results

Model #	AIC	SPPC
99	2.208	-123.901

Model #99 selected

Selected Model:

Ln(Load) = a0
+ a1 LnQ
+ a2 LnQ^2
+ a3 ADDL1
+ a4 ADDL2
+ a5 ADDL3

where:
Load = constituent load [kg/d]
LnQ = Ln(Q) - center of Ln(Q)
dtime = decimal time - center of decimal time

Model Coefficients

	a0	a1	a2	a3	a4	a5
AMLE	5.0662	1.1983	-0.0084	-10.3792	1.4654	24.0745
MLE	5.0659	1.1983	-0.0084	-10.3790	1.4657	24.0746

AMLE Regression Statistics

R-Squared [%] : 84.40
Prob. Plot Corr. Coeff. (PPCC) : 0.9187
Serial Correlation of Residuals: 0.0544

Coeff. Std.Dev. t-ratio P Value
----

a0	******	0.23	8.106E-01
a1	0.0571	20.97	1.294E-39
a2	0.0210	-0.40	6.793E-01
a3	******	-0.47	6.306E-01
a4	******	0.08	9.352E-01
a5	******	1.06	2.777E-01

Correlation Between Explanatory Variables

Explanatory variable corresponding to:

	a1	a2	a3	a4
a2	0.0000			
a3	0.1983	0.1468		
a4	-0.2154	-0.1168	-0.9902	
a5	0.1879	0.1670	0.9965	-0.9795

Additional Regression Statistics

	Residual Variance	Turnbull-Weiss Stat	DF	PL
AMLE	0.475	41.75	18	1.197E-03
MLE	0.477	41.75	18	1.197E-03

Constituent Output File Part Ib: Calibration (Concentration Regression)

AMLE Regression Statistics

Model #99 was selected for the load regression (PART Ia) and is used here:

Ln(Conc) = a0
+ a1 LnQ
+ a2 LnQ^2
+ a3 ADDL1
+ a4 ADDL2
+ a5 ADDL3

where:
Conc = constituent concentration
LnQ = Ln(Q) - center of Ln(Q)
dtime = decimal time - center of decimal time

Concentration Regression Results

R-Squared [%] : 33.09
Residual Variance : 0.4754

Coeff.	Value	Std.Dev.	t-ratio	P Value
a0	1.0791	******	0.05	9.593E-01
a1	0.1983	0.0571	3.47	5.220E-04
a2	-0.0084	0.0210	-0.40	6.793E-01
a3	*******	******	-0.47	6.306E-01
a4	1.4654	******	0.08	9.352E-01
a5	24.0745	******	1.06	2.777E-01

Constituent Output File Part IIa: Estimation (test for extrapolation)

Load Estimates for 19880101-20051231

Streamflow Summary Statistics [cfs]

Data	Mean	Minimum	10th Pct	25th Pct	Median	75th Pct	90th Pct	Maximum
Cal.	51.	0.	4.	7.	14.	20.	56.	2253.
Est.	43.	0.	3.	7.	15.	28.	90.	2253.

The maximum estimation data set steamflow does not exceed the maximum calibration data set streamflow. No extrapolation is required.

Constituent Output File Part IIb: Estimation (Load Estimates)

Load Estimates for 19880101-20051231

Load Estimates [KG/DAY]

AMLE Load Estimates

	N	Mean Load	95% Conf.Intervals Lower	Upper	Std Error Prediction	Standard Error
Est. Period	6575	60.86	35.79	97.00	15.73	15.56
Season 1	1625	60.69	34.55	99.16	16.62	15.94
Season 2	1656	75.75	46.68	116.41	17.90	17.34
Season 3	1656	49.62	29.52	78.37	12.55	11.97
Season 4	1638	57.34	29.68	100.57	18.29	17.42
Jan. 1988	31	40.48	13.71	93.91	21.13	10.10

Feb. 1988	29	33.80	17.78	58.58	10.52	6.55
Mar. 1988	31	31.88	13.09	65.57	13.68	6.84
Apr. 1988	30	26.32	11.96	50.59	10.02	5.13
May 1988	31	65.77	30.96	123.38	23.94	15.41
June 1988	30	6.62	3.78	10.81	1.81	1.08
July 1988	31	16.91	8.18	31.12	5.93	3.25
Aug. 1988	31	10.03	3.87	21.45	4.59	1.91
Sep. 1988	30	5.99	2.35	12.69	2.70	1.07
Oct. 1988	31	6.54	2.20	15.23	3.43	1.22
Nov. 1988	30	33.82	14.25	68.38	14.09	7.55
Dec. 1988	31	7.15	3.79	12.30	2.19	1.17

MLE Load Estimates (monthly data abbreviated)

	N	Mean Load	Standard Error
Est. Period	6575	60.89	15.58
Season 1	1625	60.72	15.97
Season 2	1656	75.79	17.37
Season 3	1656	49.64	11.99
Season 4	1638	57.37	17.45
Jan. 1988	31	40.50	10.12
Feb. 1988	29	33.82	6.56
Mar. 1988	31	31.90	6.84
Apr. 1988	30	26.34	5.14
May 1988	31	65.80	15.43
June 1988	30	6.63	1.08
July 1988	31	16.92	3.25
Aug. 1988	31	10.03	1.91
Sep. 1988	30	5.99	1.08
Oct. 1988	31	6.54	1.22
Nov. 1988	30	33.84	7.56
Dec. 1988	31	7.15	1.17

Summary Statistics - Estimated Loads [KG/DAY]

	Min.	25th Pct	Med.	75th Pct	90th Pct	95th Pct	99th Pct	Max.
AMLE	0.001	4.948	12.018	30.700	114.836	248.714	909.701	7017.460
MLE	0.001	4.951	12.025	30.720	114.905	248.856	910.185	7018.424

Summary Statistics - Estimated Concentrations [MG/L]

	Min.	25th Pct	Med.	75th Pct	90th Pct	95th Pct	99th Pct	Max.
AMLE	0.030	0.257	0.345	0.485	0.651	0.728	0.940	1.273
MLE	0.030	0.257	0.345	0.486	0.651	0.729	0.940	1.273

Appendix C: Example Export Coefficient Loadings Calculated by land use scenario

Nitrate: "Average Year" loading

	"AVERAGE YEAR"											
PER SITE BASIS	Water	Urban	Barren/ Tra	Forest	Grasses	Wetlands	Sum (lbs)	Sum(kg)	**Sum(tonnes)**	Totals	w/o ps	w/ps
JCC												
Site 1	269.5425	67622.46	534.9557	1416.952	750.915	100.5225	70695.34	32066.86898	**32.06687**	Po	32.06687	32.06687
Site 2	1.556764	1308.621	18.50094	49.00397	25.9697	1.445566	1405.098	637.3418167	**0.637342**		32.70421	284.34
Pohick All	35.36078	2272.24	298.5986	790.9066	419.1416	24.74143	3840.989	1742.243164	**1.742243**	ALL	34.44645	286.0836
Site 20	5.115081	54411.04	186.3722	493.6492	261.6098	100.2445	55458.04	25155.3418	**25.15534**	Acc	25.15534	25.15534
Site 13 Modified	46.92531	37187.69	274.5265	727.1462	385.3518	66.66285	38688.31	17548.72095	**17.54872**		42.70406	42.70406
Site 13 Original	5.559871	21656.31	114.6566	303.6943	160.943	20.90512	22262.07	10097.90396	**10.0979**		52.80197	52.80197
Accotink All	16.67961	3367.173	49.83645	132.0032	69.95522	22.68427	3658.331	1659.391233	**1.659391**	ALL	54.46136	54.46136
Current Condition: ACRES												
Site 1	269.5425	62037.5	800.1651	2131.088	1122.054	100.5225	66460.87	30146.14371	**30.14614**	Po	30.14614	30.14614
Site 2	1.556764	1187.503	28.60793	57.76146	41.9077	1.445566	1318.783	598.1898203	**0.59819**		30.74433	282.38
Pohick All	35.36078	1526.708	301.866	952.6684	388.1022	24.74143	3229.447	1464.852317	**1.464852**	ALL	32.20919	283.8463
Site 20	5.115081	50307	451.9578	925.2956	640.9218	100.2445	52430.53	23782.09051	**23.78209**	Acc	23.78209	23.78209
Site 13 Modified	46.92531	34164.14	597.2493	1119.677	533.8271	66.66285	36528.48	16569.03888	**16.56904**		40.35113	40.35113
Site 13 Original	5.559871	19934.94	413.8991	492.7803	262.5912	20.90512	21130.67	9584.712853	**9.584713**		49.93584	49.93584
Accotink All	16.67961	3007.806	85.94688	204.7295	53.39889	22.68427	3391.245	1538.242812	**1.538243**	ALL	51.47409	51.47409
Smart Growth Scenario												
Site 1	269.5425	57930.13	996.5721	2654.181	1397.471	100.5225	63348.42	28734.36195	**28.73436**	Po	28.73436	28.73436
Site 2	1.556764	1154.489	30.47197	61.52508	44.63832	1.445566	1294.127	587.006102	**0.587006**		29.32137	280.96
Pohick All	35.36078	764.776	334.8727	1056.835	430.538	24.74143	2647.124	1200.715188	**1.200715**	ALL	30.52208	282.1592
Site 20	5.115081	48453.56	556.8242	1139.989	789.6329	100.2445	51045.36	23153.78734	**23.15379**	Acc	23.15379	23.15379
Site 13 Modified	46.92531	33049.85	673.3534	1262.351	601.8496	66.66285	35700.99	16193.69741	**16.1937**		39.34748	39.34748
Site 13 Original	5.559871	18858.84	522.2465	621.7766	331.3303	20.90512	20360.66	9235.439538	**9.23544**		48.58292	48.58292
Accotink All	16.67961	2548.542	114.1825	271.988	70.9417	22.68427	3045.018	1381.197155	**1.381197**	ALL	49.96412	49.96412
Sustainable Development Scenario												
Site 1	269.5425	53414.96	1212.48	3229.211	1700.234	100.5225	59926.95	27182.40895	**27.18241**	Po	27.18241	27.18241
Site 2	1.556764	1130.059	31.85133	64.31011	46.65895	1.445566	1275.882	578.7302915	**0.57873**		27.76114	156.53
Pohick All	35.36078	5369.851	135.3824	427.2575	174.058	24.74143	6166.651	2797.146041	**2.797146**	ALL	30.55829	159.3246
Site 20	5.115081	46477.52	668.6268	1368.883	948.1803	100.2445	49568.57	22483.92632	**22.48393**	Acc	22.48393	22.48393
Site 13 Modified	46.92531	33249.69	659.7047	1236.763	589.6504	66.66285	35849.39	16261.01178	**16.26101**		38.74494	38.74494
Site 13 Original	5.559871	18115.54	597.0857	710.8787	378.8108	20.90512	19828.78	8994.184635	**8.994185**		47.73912	47.73912
Accotink All	16.67961	3542.761	53.05772	126.386	32.96482	22.68427	3794.534	1721.17148	**1.721171**	ALL	49.46029	49.46029
HLU Scenerio												
Site 1	269.5425	50013.94	1737.095	4626.424	2435.891	100.5225	59183.42	26845.14674	**26.84515**	Po	26.84515	26.84515
Site 2	1.556764	1416.126	40.23123	81.22972	58.93463	1.445566	1599.523	725.5316255	**0.725532**		27.57068	279.21
Pohick All	35.36078	909.8652	339.7304	1072.166	436.7835	24.74143	2818.647	1278.516685	**1.278517**	ALL	28.8492	280.4863
Site 20	5.115081	44897.57	1284.717	2630.208	1821.859	100.2445	50739.71	23015.14577	**23.01515**	Acc	23.01515	23.01515
Site 13 Modified	46.92531	29281.32	1142.567	2141.996	1021.237	66.66285	33700.71	15286.38474	**15.28638**		38.30153	38.30153
Site 13 Original	5.559871	17529.89	1063.892	1266.65	674.9683	20.90512	20561.87	9326.706281	**9.326706**		47.62824	47.62824
Accotink All	16.67961	1680.278	188.6398	449.3488	117.2021	22.68427	2474.833	1122.565165	**1.122565**	ALL	48.7508	48.7508
Current Land Use (2004 HLU)												
Site 1	269.5425	43658.34	2301.084	4293.635	2435.891	100.5225	53059.02	24067.16482	**24.06716**	Po	24.06716	24.06716
Site 2	1.556764	1182.917	27.52533	47.04976	58.93463	1.445566	1319.429	598.4827497	**0.598483**		24.66565	194.9135
Pohick All	35.36078	788.611	0	1110.18	436.7835	24.74143	2395.677	1086.660822	**1.086661**	ALL	25.75231	196.0002
Site 20	5.115081	39169.79	1766.203	1872.236	1821.859	100.2445	44735.44	20291.65603	**20.29166**	Acc	20.29166	20.29166
Site 13 Modified	46.92531	22598.56	661.0815	2899.968	1021.237	66.66285	27294.43	12380.54593	**12.38055**		32.6722	32.6722
Site 13 Original	5.559871	19381.36	28.68616	358.646	674.9683	20.90512	20470.12	9285.090858	**9.285091**		41.95729	41.95729
Accotink All	16.67961	1411.245	17.94183	467.4304	117.2021	22.68427	2053.183	931.3083032	**0.931308**	ALL	42.8886	42.8886
Land Use (1992)												
Site 1	405.8706	31448.3	1785.372	7375.892	1490.713	67.05205	42573.2	19310.8799	**19.31088**	Po	19.31088	19.31088
Site 2	9.118189	990.5822	13.41041	87.56797	30.46809	5.671068	1136.818	515.6519448	**0.515652**		19.82653	99.62907
Pohick All	64.7169	1034.39	148.4085	1127.709	213.2767	42.9222	2631.422	1193.593161	**1.193593**	ALL	21.02013	100.8227
Site 20	25.57541	32067.86	789.4261	4309.845	762.3695	49.87204	38004.95	17238.75643	**17.23876**	Acc	17.23876	17.23876
Site 13 Modified	104.3032	22807.53	1450.112	3023.347	610.6962	55.20952	28051.2	12723.80916	**12.72381**		29.96257	29.96257
Site 13 Original	15.79003	11690.3	2003.515	1572.22	459.9125	19.62634	15761.36	7149.234872	**7.149235**		37.1118	37.1118
Accotink All	16.45722	1392.894	165.395	390.9701	99.85528	44.53457	2110.107	957.1283139	**0.957128**	ALL	38.06893	38.06893

Nitrate: High Export Coefficient Loading

	High Export Coefficient Scenario											
PER SITE BASIS	Water	Urban	Barren/ Tra	Forest	Grasses	Wetlands	Sum (lbs)	Sum(kg)	**Sum(tonnes)**	Totals	w/o ps	w/ps
JCC												
Site 1	1739.601	170788.8	1351.096	3353.453	4265.197	713.7095	182211.8	82649.88863	**82.64989**	Po	82.64989	82.64989
Site 2	10.0472	3305.082	46.72639	115.9761	147.5079	10.26352	3635.604	1649.082017	**1.649082**		84.29897	681.59
Pohick All	228.2149	5738.818	754.1472	1871.812	2380.725	175.6641	11149.38	5057.274261	**5.057274**	ALL	89.35624	686.6488
Site 20	33.01222	137421.7	470.7056	1168.303	1485.944	711.7358	141291.4	64088.71298	**64.08871**	Acc	64.08871	64.08871
Site 13 Modified	302.8513	93922.06	693.3502	1720.913	2188.798	473.3063	99301.27	45042.30012	**45.0423**		109.131	109.131
Site 13 Original	35.88285	54695.65	289.5793	718.7431	914.1564	148.4263	56802.44	25765.15113	**25.76515**		134.8962	134.8962
Accotink All	107.6486	8504.205	125.868	312.4076	397.3457	161.0583	9608.533	4358.357355	**4.358357**	ALL	139.2545	139.2545
Current Condition:	ACRES											
Site 1	1739.601	156683.3	2020.914	5043.576	6373.268	713.7095	172574.3	78278.39951	**78.2784**	Po	78.2784	78.2784
Site 2	10.0472	2999.185	72.25282	136.7021	238.0357	10.26352	3466.486	1572.371572	**1.572372**		79.85077	677.14
Pohick All	228.2149	3855.886	762.3995	2254.648	2204.42	175.6641	9481.234	4300.615396	**4.300615**	ALL	84.15139	681.4439
Site 20	33.01222	127056.5	1141.475	2189.866	3640.436	711.7358	134773	61131.99768	**61.132**	Acc	61.132	61.132
Site 13 Modified	302.8513	86285.69	1508.426	2649.902	3032.138	473.3063	94252.31	42752.13067	**42.75213**		103.8841	103.8841
Site 13 Original	35.88285	50348.12	1045.353	1166.247	1491.518	148.4263	54235.55	24600.82956	**24.60083**		128.485	128.485
Accotink All	107.6486	7596.58	217.0693	484.5265	303.3057	161.0583	8870.188	4023.449792	**4.02345**	ALL	132.5084	132.5084
Smart Growth Scenario												
Site 1	1739.601	146309.6	2516.964	6281.563	7937.638	713.7095	165499.1	75069.12477	**75.06912**	Po	75.06912	75.06912
Site 2	10.0472	2915.803	76.96067	145.6094	253.5457	10.26352	3412.23	1547.761348	**1.547761**		76.61689	328.25
Pohick All	228.2149	1931.535	845.7617	2501.176	2445.456	175.6641	8127.808	3686.711668	**3.686712**	ALL	80.3036	331.9407
Site 20	33.01222	122375.4	1406.327	2697.974	4485.115	711.7358	131709.5	59742.438	**59.74244**	Acc	59.74244	59.74244
Site 13 Modified	302.8513	83471.42	1700.636	2987.563	3418.506	473.3063	92354.29	41891.19986	**41.8912**		101.6336	101.6336
Site 13 Original	35.88285	47630.3	1318.997	1471.538	1881.956	148.4263	52487.1	23807.74855	**23.80775**		125.4414	125.4414
Accotink All	107.6486	6436.654	288.3818	643.705	402.9488	161.0583	8040.397	3647.062724	**3.647063**	ALL	129.0884	129.0884
Sustainable Development Scenario												
Site 1	1739.601	134906	3062.265	7642.465	9657.33	713.7095	157721.4	71541.21153	**71.54121**	Po	71.54121	71.54121
Site 2	10.0472	2854.102	80.44442	152.2006	265.0228	10.26352	3372.081	1529.550093	**1.52955**		73.07076	201.84
Pohick All	228.2149	13562.21	341.9247	1011.176	988.6495	175.6641	16307.84	7397.113371	**7.397113**	ALL	80.46787	209.2342
Site 20	33.01222	117384.6	1688.699	3239.69	5385.664	711.7358	128443.4	58260.9687	**58.26097**	Acc	58.26097	58.26097
Site 13 Modified	302.8513	83976.14	1666.165	2927.007	3349.214	473.3063	92694.68	42045.60062	**42.0456**		100.3066	100.3066
Site 13 Original	35.88285	45753.01	1508.013	1682.413	2151.645	148.4263	51279.39	23259.94064	**23.25994**		123.5665	123.5665
Accotink All	107.6486	8947.675	134.0037	299.1135	187.2402	161.0583	9836.739	4461.86997	**4.46187**	ALL	128.0284	128.0284
HLU Scenerio												
Site 1	1739.601	126316.3	4387.245	10949.2	13835.86	713.7095	157941.9	71641.2532	**71.64125**	Po	71.64125	71.64125
Site 2	10.0472	3576.598	101.6089	192.2437	334.7487	10.26352	4225.51	1916.658914	**1.916659**		73.55791	325.20
Pohick All	228.2149	2297.976	858.0304	2537.458	2480.93	175.6641	8578.274	3891.039452	**3.891039**	ALL	77.44895	329.0861
Site 20	33.01222	113394.3	3244.71	6224.826	10348.16	711.7358	133956.7	60761.74604	**60.76175**	Acc	60.76175	60.76175
Site 13 Modified	302.8513	73953.55	2885.692	5069.39	5800.627	473.3063	88485.41	40136.3083	**40.13631**		100.8981	100.8981
Site 13 Original	35.88285	44273.88	2686.99	2997.738	3833.82	148.4263	53976.74	24483.43581	**24.48344**		125.3815	125.3815
Accotink All	107.6486	4243.747	476.4327	1063.459	665.7078	161.0583	6718.053	3047.257689	**3.047258**	ALL	128.4287	128.4287
Current Land Use (2004 HLU)												
Site 1	1739.601	110264.5	5811.668	10161.6	13835.86	713.7095	142526.9	64649.11563	**64.64912**	Po	64.64912	64.64912
Site 2	10.0472	2987.6	69.51859	111.3511	334.7487	10.26352	3523.529	1598.245896	**1.598246**		66.24736	236.4952
Pohick All	228.2149	1991.733	0	2627.427	2480.93	175.6641	7503.969	3403.743186	**3.403743**	ALL	69.6511	239.899
Site 20	33.01222	98928.07	4460.76	4430.96	10348.16	711.7358	118912.7	53937.89115	**53.93789**	Acc	53.93789	53.93789
Site 13 Modified	302.8513	57075.41	1669.642	6863.257	5800.627	473.3063	72185.09	32742.60777	**32.74261**		86.6805	86.6805
Site 13 Original	35.88285	48949.98	72.4504	848.7956	3833.82	148.4263	53889.35	24443.79886	**24.4438**		111.1243	111.1243
Accotink All	107.6486	3564.272	45.31427	1106.252	665.7078	161.0583	5650.253	2562.911469	**2.562911**	ALL	113.6872	113.6872
Land Use (1992)												
Site 1	2619.448	79426.52	4509.176	17456.28	8467.248	476.0695	112954.7	51235.40883	**51.23541**	Po	51.23541	51.23541
Site 2	58.84788	2501.836	33.86962	207.2442	173.0588	40.26459	3015.121	1367.635959	**1.367636**		52.60304	132.4056
Pohick All	417.6764	2612.477	374.8238	2668.91	1211.411	304.7477	7590.047	3442.787239	**3.442787**	ALL	56.04583	135.8484
Site 20	165.0611	80991.3	1993.792	10199.97	4330.259	354.0915	98034.47	44467.68756	**44.46769**	Acc	44.46769	44.46769
Site 13 Modified	673.1623	57603.2	3662.435	7155.254	3468.755	391.9876	72954.79	33091.73596	**33.09174**		77.55942	77.55942
Site 13 Original	101.9073	29525.28	5060.122	3720.921	2612.303	139.347	41159.88	18669.80736	**18.66981**		96.22923	96.22923
Accotink All	106.2132	3517.925	417.7253	925.296	567.178	316.1954	5850.533	2653.757045	**2.653757**	ALL	98.88299	98.88299

Nitrate: Low Export Coefficient Loading

	LOW EXPORT COEFFICIENT SCENARIO											
PER SITE BASIS	Water	Urban	Barren/ Tra	Forest	Grasses	Wetlands	Sum (lbs)	Sum(kg)	**Sum(tonnes)**	Totals	w/o ps	w/ps
JCC												
Site 1	0	42053.77	159.6883	468.5388	450.549	0	43132.54	19564.59233	**19.56459**	Po	19.56459	19.56459
Site 2	0	813.8192	5.52267	16.20398	15.58182	0	851.1276	386.0650072	**0.386065**		19.95066	271.59
Pohick All	0	1413.084	89.13392	261.5265	251.485	0	2015.23	914.092809	**0.914093**	ALL	20.86475	272.5019
Site 20	0	33837.71	55.63348	163.2333	156.9659	0	34213.55	15519.00395	**15.519**	Acc	15.519	15.519
Site 13 Modified	0	23126.68	81.94822	240.443	231.2111	0	23680.28	10741.19341	**10.74119**		26.2602	26.2602
Site 13 Original	0	13467.85	34.22586	100.4216	96.56582	0	13699.07	6213.792203	**6.213792**		32.47399	32.47399
Accotink All	0	2094.013	14.87655	43.64907	41.97313	0	2194.512	995.4137194	**0.995414**	ALL	33.4694	33.4694
Current Condit ACRES												
Site 1	0	38580.53	238.8552	704.6799	673.2325	0	40197.3	18233.18905	**18.23319**	Po	18.23319	18.23319
Site 2	0	738.4971	8.539681	19.09979	25.14462	0	791.2812	358.9191252	**0.358919**		18.59211	270.23
Pohick All	0	949.4451	90.10927	315.0157	232.8613	0	1587.431	720.0467561	**0.720047**	ALL	19.31215	270.9493
Site 20	0	31285.45	134.9128	305.9644	384.5531	0	32110.88	14565.24925	**14.56525**	Acc	14.56525	14.56525
Site 13 Modified	0	21246.35	178.2834	370.2398	320.2963	0	22115.17	10031.27373	**10.03127**		24.59652	24.59652
Site 13 Original	0	12397.35	123.552	162.946	157.5547	0	12841.4	5824.762395	**5.824762**		30.42129	30.42129
Accotink All	0	1870.526	25.65579	67.69723	32.03933	0	1995.918	905.3333093	**0.905333**	ALL	31.32662	31.32662
Smart Growth Scenario												
Site 1	0	36026.2	297.4842	877.6493	838.4829	0	38039.82	17254.57182	**17.25457**	Po	17.25457	17.25457
Site 2	0	717.9659	9.09611	20.34429	26.78299	0	774.1893	351.1663756	**0.351166**		17.60574	146.37
Pohick All	0	475.6069	99.96199	349.4601	258.3228	0	1183.352	536.7593748	**0.536759**	ALL	18.1425	146.9088
Site 20	0	30132.81	166.2162	376.9564	473.7798	0	31149.76	14129.29425	**14.12929**	Acc	14.12929	14.12929
Site 13 Modified	0	20553.39	201.001	417.4173	361.1098	0	21532.92	9767.166962	**9.767167**		23.89646	23.89646
Site 13 Original	0	11728.13	155.8945	205.6008	198.7982	0	12288.43	5573.937241	**5.573937**		29.4704	29.4704
Accotink All	0	1584.914	34.08432	89.93738	42.56502	0	1751.501	794.4675541	**0.794468**	ALL	30.26487	30.26487
Sustainable Development Scenario												
Site 1	0	33218.26	361.9343	1067.792	1020.14	0	35668.13	16178.79085	**16.17879**	Po	16.17879	16.17879
Site 2	0	702.7731	9.507861	21.26521	27.99537	0	761.5416	345.4294385	**0.345429**		16.52422	145.29
Pohick All	0	3339.46	40.41266	141.2798	104.4348	0	3625.587	1644.538626	**1.644539**	ALL	18.16876	146.9351
Site 20	0	28903.93	199.5901	452.644	568.9082	0	30125.07	13664.50388	**13.6645**	Acc	13.6645	13.6645
Site 13 Modified	0	20677.67	196.9268	408.9564	353.7902	0	21637.34	9814.532308	**9.814532**		23.47904	23.47904
Site 13 Original	0	11265.88	178.2345	235.0639	227.2865	0	11906.47	5400.683816	**5.400684**		28.87972	28.87972
Accotink All	0	2203.21	15.83813	41.79164	19.77889	0	2280.618	1034.471073	**1.034471**	ALL	29.91419	29.91419
HLU Scenerio												
Site 1	0	31103.2	518.5358	1529.804	1461.534	0	34613.07	15700.22572	**15.70023**	Po	15.70023	15.70023
Site 2	0	880.6751	12.00932	26.85996	35.36078	0	954.9051	433.1376857	**0.433138**		16.13336	144.90
Pohick All	0	565.8366	101.412	354.5294	262.0701	0	1283.848	582.3437116	**0.582344**	ALL	16.71571	145.482
Site 20	0	27921.37	383.4977	869.7222	1093.115	0	30267.71	13729.20132	**13.7292**	Acc	13.7292	13.7292
Site 13 Modified	0	18209.78	341.0647	708.2867	612.7423	0	19871.87	9013.728942	**9.013729**		22.74293	22.74293
Site 13 Original	0	10901.67	317.5798	418.8389	404.981	0	12043.07	5462.646422	**5.462646**		28.20558	28.20558
Accotink All	0	1044.949	56.31037	148.5847	70.32125	0	1320.165	598.8169189	**0.598817**	ALL	28.80439	28.80439
Current Land Use (2004 HLU)												
Site 1	0	27150.71	686.8908	1419.762	1461.534	0	30718.9	13933.85721	**13.93386**	Po	13.93386	13.93386
Site 2	0	735.6446	8.216518	15.55779	35.36078	0	794.7797	360.5060101	**0.360506**		14.29436	184.5422
Pohick All	0	490.4297	0	367.0996	262.0701	0	1119.599	507.8417702	**0.507842**	ALL	14.8022	185.0501
Site 20	0	24359.32	527.2246	619.0862	1093.115	0	26598.75	12064.9883	**12.06499**	Acc	12.06499	12.06499
Site 13 Modified	0	14053.83	197.3378	958.9227	612.7423	0	15822.83	7177.11575	**7.177116**		19.2421	19.2421
Site 13 Original	0	12053.08	8.563034	118.5923	404.981	0	12585.22	5708.559002	**5.708559**		24.95066	24.95066
Accotink All	0	877.64	5.355769	154.5637	70.32125	0	1107.881	502.5262272	**0.502526**	ALL	25.45319	25.45319
Land Use (1992)												
Site 1	0	19557.4	532.947	2438.962	894.4276	0	23423.74	10624.82911	**10.62483**	Po	10.62483	10.62483
Site 2	0	616.0337	4.003107	28.95581	18.28086	0	667.2735	302.6701622	**0.30267**		10.9275	90.73004
Pohick All	0	643.2771	44.30105	372.8957	127.966	0	1188.44	539.0672203	**0.539067**	ALL	11.46657	91.2691
Site 20	0	19942.7	235.6496	1425.122	457.4217	0	22060.9	10006.65365	**10.00665**	Acc	10.00665	10.00665
Site 13 Modified	0	14183.79	432.8693	999.72	366.4177	0	15982.79	7249.673469	**7.249673**		17.25633	17.25633
Site 13 Original	0	7270.087	598.0642	519.8809	275.9475	0	8663.98	3929.915225	**3.929915**		21.18624	21.18624
Accotink All	0	866.2279	49.37166	129.2808	59.91317	0	1104.794	501.1259155	**0.501126**	ALL	21.68737	21.68737

CURRICULUM VITAE

Ryan Albert was born in Atlanta, Georgia in 1977. He graduated from Emory University with a Bachelor of Arts in 1999 before spending two years living and teaching in Austria. In 2001, he commenced studying in the Environmental Science and Public Policy Program at George Mason University. He is currently employed at the United States Environmental Protection Agency, where his work focuses on stormwater.