In Situ Sensors for Monitoring BMP Performance and In-Stream Pollutant Loading

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Dedication

This thesis is dedicated to my grandfather, Arthur V. Dienhart, the epitome of a professional engineer and, most importantly, a loving father and grandfather.

Abstract

For this research, a sensor network was used to evaluate pollutant loads and concentrations in the targeted portions of streams and to evaluate the performance of pond/wetland stormwater best management practices (BMPs). The study sites were chosen to examine whether BMPs achieve their intended purpose of improving surface water quality. This data was also collected with intent of examining whether a sensor network could provide more a more efficient and comprehensive means of monitoring surface waters and BMP systems.

Violations of water quality standards for turbidity, dissolved oxygen, and chloride (chronic) were periodically observed at both sampling locations. Measured concentrations of pollutants we converted to loading rates to determine flow profile influences on water quality. Dissolved oxygen and chloride violations were generally observed during periods of low flow, while those for turbidity were generally independent of flow.

Water quality downstream of BMPs was often (but not always) better than upstream water quality, suggesting that BMPs controlling stormwater inflow are not contributing to the degradation of downstream water quality. Analysis of water quality before and during rain events indicates that chloride and suspended solids are controlled in the pond/wetland systems studied. Evidence for phosphorus removal was also obtained.

Table of Contents

List of Tables

List of Figures

1. INTRODUCTION

1.1 Water Quality Monitoring in Minnesota

As early as 1965, many states began water quality monitoring programs in response to the passage of the Federal Water Quality Act. This legislation legally defined requirements for states to monitor water quality in an effort to manage the nation’s waters (MPCA, 2007). In 1972, the Federal Water Pollution and Control Act (the Clean Water Act, CWA) passed Congress and revolutionized water quality management. In waters with point source discharges, comprehensive water quality data collection was a legal mandate. In addition, Section 303d of the CWA required states to inventory water bodies not meeting water quality standards. Section 305b required periodic assessments of water quality conditions in each state to be submitted to the Environmental Protection Agency (EPA). Considerable effort was initially put toward controlling point source discharges, and attention turned toward non-point sources only within the last decade (MPCA, 2007).

Nonpoint source pollution generally results from land runoff, precipitation, atmospheric deposition, drainage, seepage or hydrologic modification. The term "nonpoint source" is defined to mean any source of water pollution that does not meet the legal definition of "point source" in section 502(14) of the CWA. That definition states:

> “The term "point source" means any discernible, confined and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, or vessel or other floating craft, from which pollutants are or may be discharged. This term does not include agricultural storm water discharges and return flows from irrigated agriculture.”

A shift in focus from point source to non-point source pollution also moved the focus of CWA enforcement to legally defensible information about impaired waterways in the form of Total Maximum Daily Loads (TMDLs). TMDLs are calculations of the maximum amount of a pollutant that a water body can receive and still meet water quality standards, and an allocation of that amount to the pollutant's sources (EPA, 2005). Due to the CWA, water quality data collected through periodic sampling was then used to determine compliance with water quality standards and for the development of TMDLs.

Monthly water quality monitoring within Minnesota (generally referred to as either fixed-station or shallow water monitoring), currently consists of field personnel collecting on-site data and water samples that are sent to laboratories where they are analyzed for concentrations of various chemical indicators (e.g., nutrients, carbon). Such data are then analyzed for compliance with thresholds established either through state legislation or the advice of expert panels convened for such purposes (Hall and Wazniak, 2004).

Once thresholds have been established for a water body, Federal and state regulations and programs also require implementation of restoration measures to meet TMDLs in the form of Best Management Practices (BMPs). The restoration measures are included in an Implementation Plan, along with metrics for determining effectiveness of the restoration measures and monitoring plans to assess water quality.

1.2 Development of TMDLs

TMDLs typically are developed and monitored using analysis of discrete water samples collected on a fixed schedule. Measurements are then compared to water quality standards based on each water body's intended use. For example, the health of a drinking water reservoir is evaluated against different criteria than a water body designated for fishing and swimming.

In the process of developing a TMDL, an attempt is made to define the individual components of the TMDL, at least to a necessary level of detail to assist with water quality planning and restoration. A TMDL is calculated using the following identified components of the loading capacity of given water body: $TMDL = WLA + LA + MOS$. The Waste Load Allocation, or WLA, for a water body is the portion of the receiving water's loading capacity that is allocated to one of its existing or future point sources of pollution. The waste load allocations are typically applied to individual point sources, but can also be applied to a category of point sources. The Load Allocation (LA) includes natural background and remaining non-point sources. Natural background is generally set equal to the existing natural background load for TMDL development purposes because modifications to natural conditions are generally not the intent of the TMDL process. The remaining LA applies to nonpoint sources. The LA is applied to nonpoint source categories to address the overall cumulative effects from a given category and to better identify potential pollutant reductions through similar applications of best management practices over a given source category. The Margin of Safety (MOS) margin of safety is typically identified in the TMDL equation to account for uncertainty in pollutant loads and receiving water quality. The margin of safety can be provided implicitly through analytical assumptions or explicitly by reserving a portion of the loading capacity (EPA, 2001 and CWA section 303(d) (1)(c)).

TMDLs estimated on the basis of discrete samples, however, may not adequately represent the daily, monthly, or annual constituent load variability in a watershed system.

Continuous water-quality monitoring and regression estimates are beneficial to TMDL programs because they describe variability in water-quality conditions better than discrete samples alone. Continuous values from in-stream sensors (turbidity, specific conductance, dissolved oxygen, pH, and water temperature) and regression models make it possible to estimate concentrations and loads for different time periods. It is possible to collect data by the minute, hour, day, week, month, or year. This higher frequency of sampling and measurements can help characterize load fluctuations under changing stream flow and seasonal conditions. Higher frequency data may also be able to characterize responses of different areas within a watershed to environmental conditions, further improving the quality of TMDL studies.

Sensor technology currently is not available to directly measure all chemicals of interest in streams. Regression models, therefore, are used to estimate stream chemical concentrations from the relationship between laboratory-analyzed samples and in-stream sensor measurements such as turbidity and specific conductance. Uncertainty in regression-estimated concentrations is defined using prediction intervals and typical regression diagnostic statistics including mean square error (MSE) and the coefficient of determination (R^2).

Continuous data provides the foundation for a more comprehensive evaluation of the variability in loading characteristics and water-quality degradation than provided by discrete water-quality samples, especially in determining the Load Allocation (LA) portion of a TMDL. Continuous concentration estimates can be used to construct cumulative frequency distribution (duration) curves to determine percentage of time that estimated concentrations exceed water-quality criteria. Estimated concentration and load duration curves can be used to evaluate current water-quality conditions and estimate the duration and magnitude of potential water-quality degradation. They are used frequently in TMDL studies. In situations where discrete samples and concentration data are necessary for TMDL efforts, monitoring by continuous sensor data allows regulatory agencies to optimize sampling efforts. When continuous monitoring is considered over the long term, it may be possible to identify changes in water-quality conditions as a result of a TMDL implementation plan (Rasmussen et al., 2008).

1.3 Stormwater

Non-point source (NPS) pollution comes from many diffuse sources. One source of NPS pollution that must be considered during development of TMDLs is stormwater runoff. Stormwater is an all-inclusive term that refers to any of the water running off of the land surface after a rainfall or snowmelt event. This runoff inevitably contains NPS pollution. Prior to development, stormwater is a small component of the annual water balance. As development increases, the increases in impervious land area that occur during urban development significantly alter watershed hydrology and water quality (Figure 1.1). Development reduces rainfall/snowfall pathways for storage, infiltration, and groundwater recharge, and increases surface runoff. Urban runoff also collects and accumulates significant pollutant loads from impervious areas, further deteriorating surface water quality (Davis and McCuen, 2005; Bauer et al., 2007).

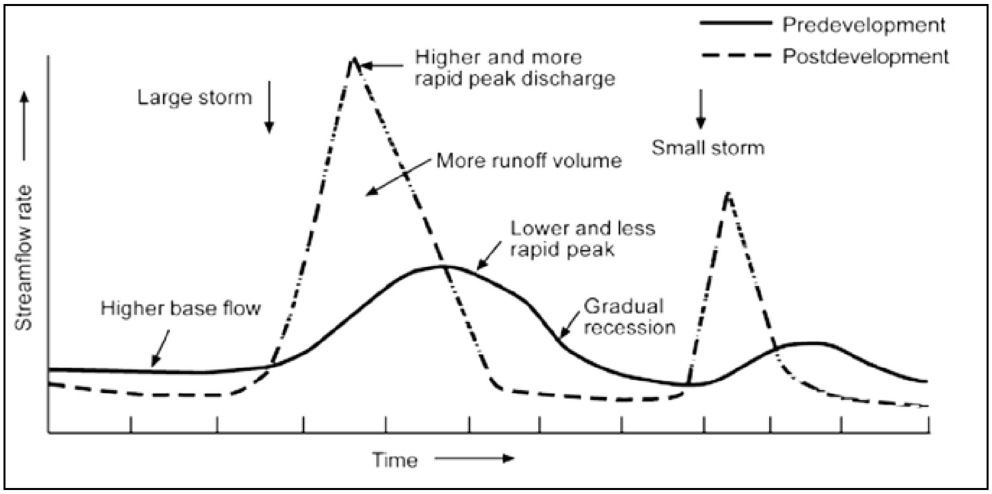
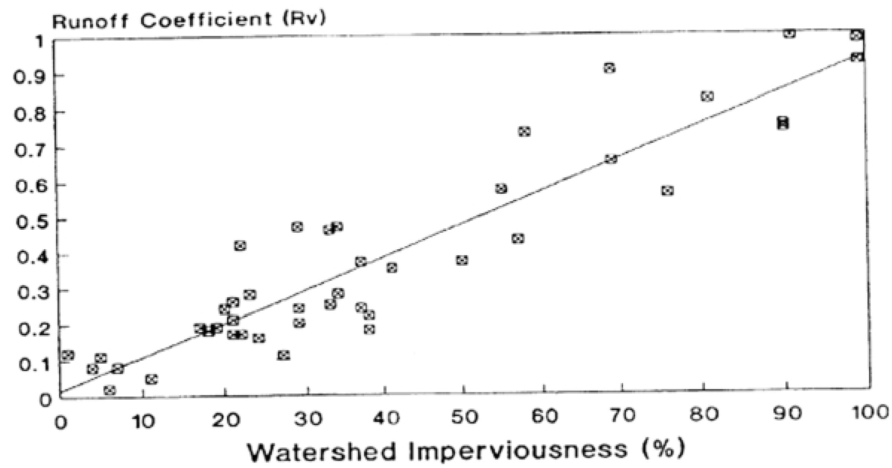


Figure 1.1: Changes in Stream Hydrology as a result of Urbanization (from Schueler, 1994)

Population growth and resulting urban sprawl have increased the impervious surface area throughout the state of Minnesota. Increased watershed imperviousness generally leads to increased stormwater flows and pollutant loads (Figure 1.2). From 1990 to 2000, impervious surface area increased in Minnesota by 44%. More importantly, this included increases of more than 100% in 20 of the 81 major watersheds in the state (Bauer et al., 2007). These increases in impervious surface area are worrisome because they prevent infiltration, which is an effective, natural method for treating stormwater and thus reducing its impact upon surface waters. Because most NPS pollution results from a lack of infiltration, rain events in areas with a high percentage of impervious surfaces cause water to accumulate a multitude of pollutants, including harmful anthropogenic organic compounds, nutrients, pathogens, and particulates, before washing off into nearby water ways. The concern stemming from NPS pollution has increased dramatically with increasing urbanization and as a result, multiple BMPs have been identified and established to minimize the impact of storm events on surface water quality (MPCA, 2005).



The runoff coefficient (R_v) expresses the fraction of rainfall that is converted into runoff. The data points reflect over 35 monitoring stations in the U.S.

Figure 1.2: Relationship between impervious cover and the volumetric runoff coefficient (Schueler, 1994). The runoff coefficient (Rv) expressed the fraction of rainfall that is converted into runoff. The data points reflect over 35 monitoring stations in the U.S.

NPS pollution is a leading remaining cause of water quality problems. The effects of nonpoint source pollutants on specific waters vary and may not always be fully assessed. We know, however, that these pollutants have harmful effects on drinking water supplies, recreation, fisheries, and wildlife. Urban runoff flows contain a variety of physical, chemical, and biological pollutants from human activities and natural processes, including suspended solids, oil and grease, organic carbon, nutrients such as nitrogen and phosphorus species, heavy metals, toxic organics such as pesticides and fuel residues, and pathogens (Davis and McCuen 2005).

More specifically, pollutants of concern in Minnesota surface waters include nitrate, phosphorus, chloride, turbidity (total suspended solids), and fecal coliforms. Dissolved oxygen levels are also of paramount concern for aquatic health in these creeks. Nitrate is an important nutrient when examining water quality. Normally only small amounts of nitrate are found naturally, but an increase in nitrate levels from many man-made sources such as septic systems, fertilizer runoff and improperly treated wastewater has caused eutrophication problems across the United States and especially in the Gulf of Mexico. In Minnesota, however, phosphorus, not nitrate, is the limiting nutrient in natural water bodies. Phosphorus is of concern in Minnesota's surface waters because it is the nutrient primarily responsible for the eutrophication of Minnesota's surface waters. Excess nutrients cause excessive algal blooms and reduced water transparency (MPCA, 2008). When excessive amounts of algae in surface waters die, the decomposition of algae by microorganisms consumes dissolved oxygen and consequently stresses the biological community in that water body. Stormwater BMP performance is essential in urban areas because phosphorus concentrations are related to intensity of land use, with loads being highest from urban lawns and streets. Generally speaking, NPS of phosphorus comprise a much larger fraction of the aggregate total phosphorus load to Minnesota surface waters during relatively wet periods, while point sources become more important during dry conditions (MPCA, 2008).

Phosphorus attached to suspended sediment promotes algal growth, which also results in another water quality index of concern: turbidity (MPCA, 2008). Turbidity is a measure of the degree to which the water loses its transparency due to the presence of suspended particulates. It is used as a surrogate to measure total suspended solids (TSS) in water. The higher the turbidity, the more suspended sediment is in water. Equation 2 in section 2.3.1 is used to convert turbidity measurements to a concentration of TSS. Sources of sediment in urban stormwater include new construction, sand from de-icing, runoff from lawns, and atmospheric deposition (MPCA, 2008). Natural creek sediments can also remain dispersed in the water column due to turbulent mixing and create turbid or cloudy conditions. Sediment not only can be a source of phosphorus loading, but it can also destroy habitat and threaten organism survival. The two most effective strategies to reduce sediment loading are decreasing impervious surface and disconnecting impervious surface from nearby water bodies. Structural BMPs, such as detention basins, can reduce turbidity in surface waters (MPCA, 2008). In addition, implementation of phosphorus BMPs will reduce suspended solids associated with biological growth.

Specific Conductance (SC) is a measure of how well water can conduct an electrical current. Specific conductance increases with increasing amounts and mobility of ions. These ions, which come from the breakdown of compounds (e.g. Cl^- from NaCl), conduct electricity because they are negatively or positively charged when dissolved in water. Therefore, SC is an indirect measure of the presence of dissolved solids, including chloride. The correlation between chloride and specific conductance for Minnehaha Creek is shown in Section 2.3, Equation 1. At high concentrations (acute) chloride is toxic to aquatic organisms and at lower levels (chronic), increased chloride concentrations in waters may affect community structure, diversity and productivity. Point sources of chloride in urban water creeks include discharge from wastewater treatment plants, and particulate matter from vehicle exhaust, while NPS include parking lots and roads where deicer is used and the natural weathering of rocks containing chloride. A common assumption is that chloride is conservative in soils and can be used as a groundwater tracer, but according to Bastviken et al. (2010) an increasing number of studies indicate that chloride can be retained in soils. The observation of chloride behavior in BMPs and urban streams sheds light not only on seasonal chloride concentrations, but possible chloride removal mechanisms in the form of stormwater BMPs (Trowbridge et al., 2010).

Fecal coliform bacteria are used as a biological indicator of fecal contamination. These bacteria do not necessarily cause a significant health risk on their own, but their presence signifies an elevated risk of the presence of pathogens within the water (Glassmeyer et al., 2005). For this reason contaminant limits are placed on fecal coliform bacteria based on the types of use of a water body. Impairments of water bodies due to exceedances in the contaminant limits for fecal coliform bacteria are some of the most common impairments in the United States (Surbeck et al., 2010). Fecal coliform contamination is often a result of stormwater runoff that accumulates these organisms as it passes over soils and other surfaces exposed to fecal matter. Stormwater runoff also tends to collect phosphorus, nitrogen and other nutrients such as dissolved organic carbon (DOC) that allow for the bacteria to grow. Both phosphorus and DOC have been found to correlate with fecal coliform populations (Surbeck et al., 2005). Thus fecal populations would be expected to be significantly higher during and after rain events due to both the input of new bacteria and the presence of sufficient nutrients to grow within the water body. These populations are due in large part to animal fecal matter, however, which is of a smaller hazard to human health than human fecal matter (Davenport et al., 1976).

1.4 Best Management Practices (BMPs)

As stated in Minn. Stat. §103F.711, Minnesota Clean Water Partnership Act: "Best Management Practices" are practices, techniques, and measures that prevent or reduce water pollution from nonpoint sources by using the most effective and practicable means of achieving water quality goals. BMPs include, but are not limited to, official controls, structural and nonstructural controls, and operation and maintenance procedures. BMPs can exist alone or as a part of a "BMP system". A treatment system "consists of a series of best management practices (BMPs). Such systems may include multiple management options, ranging from street sweeping and structures to open space and litter control laws" (MPCA, 2005).

Generally, implementation of a BMP or a BMP system is aimed at avoiding adverse environmental impacts on surface waters (e.g., avoiding damage in fragile ecosystems). As development occurs, BMP systems must then be used to help the modified site behave more similarly to predevelopment hydrologic conditions, including peak discharge, runoff volume, infiltration capacity, base flow levels, ground water recharge and maintenance of water quality. Finally, BMPs and BMP systems are used as treatment or "mitigation for unavoidable impacts" (MPCA, 2005). When determining BMP effectiveness, not only is the appropriate selection of BMPs necessary, but also the assessment of BMP performance by the ability to avoid adverse environmental impacts, minimize and mitigate unavoidable environmental impacts (Young et al., 2009).

Current practices to stem the effects of stormwater runoff include various forms of stormwater ponds, wetlands, channels and engineered systems involving filtration and settling techniques. Previous studies performed for the US EPA show high pollutant removal for phosphorus and total suspended solids in both pond systems and wetlands, with the ponds performing better (Winer, 2000). Neither BMP, however, was very effective at removing metals and nitrogen . Overall, stormwater ponds and wetlands tend to be the most effective BMPs when the removal of all major pollutants is assessed. Unfortunately these systems lose effectiveness with age and thus may become ineffective if they are not monitored and maintained correctly (Winer, 2000). The following sections summarize previous research on the BMPs relevant to this study, ponds and wetlands.

1.4.1 Ponds

Stormwater ponds are the most common "structural" method of regulating and treating stormwater runoff. Many of the BMPs and BMP systems that municipalities and watershed districts construct implement stormwater ponds to mitigate the impacts of urbanization. Research of stormwater treatment by detention pond BMPs indicates that treatment occurs primarily by dynamic and quiescent settling of sediment particles. In addition to settling, treatment may also be accomplished by biological and chemical action, plant uptake, evapotranspiration, infiltration and, in some cases, physical diversion to other systems (USEPA, 1983). If well designed, wet ponds and constructed wetland treatment systems are effective for removing sediment and associated pollutants, such as trace metals, nutrients and hydrocarbons. Ponds can also be effective at removing or treating oxygen-demanding substances, bacteria, and dissolved nutrients. Up to 90%

removal of total suspended solids (TSS) appears to be an attainable goal in stormwater-treatment ponds (MPCA, 2005). More specific pond removal data is show in Figure 1.3, along with data for other systems.

1.4.2 Wetlands

Constructed wetlands (CWs) are shallow marsh systems planted with emergent vegetation that are designed to treat stormwater runoff. While they are one of the best BMPs for pollutant removal, CWs can also mitigate peak rates and even reduce runoff volume to a certain degree. They also can provide considerable aesthetic and wildlife benefits. CWs use a relatively large amount of space and require an adequate source of inflow to maintain the permanent water surface. They can also be used effectively in series with other flow/sediment reducing BMPs that reduce the sediment load and equalize incoming flows to the CW. CWs are a good option for retrofitting existing detention basins. Nitrogen removal, however, may be limited (Campbell and Ogden, 1999). Perhaps the biggest disadvantage is the relatively low treatment capacities of subsurface flow CWs – they are generally only able to treat small flows. The advantage of these CWs is that they contain bioretention media to promote evapotranspiration, biological activity, and pollutant uptake, as well as to maintain soil porosity and permeability. Previous studies have indicated that bioretention effectively improves both

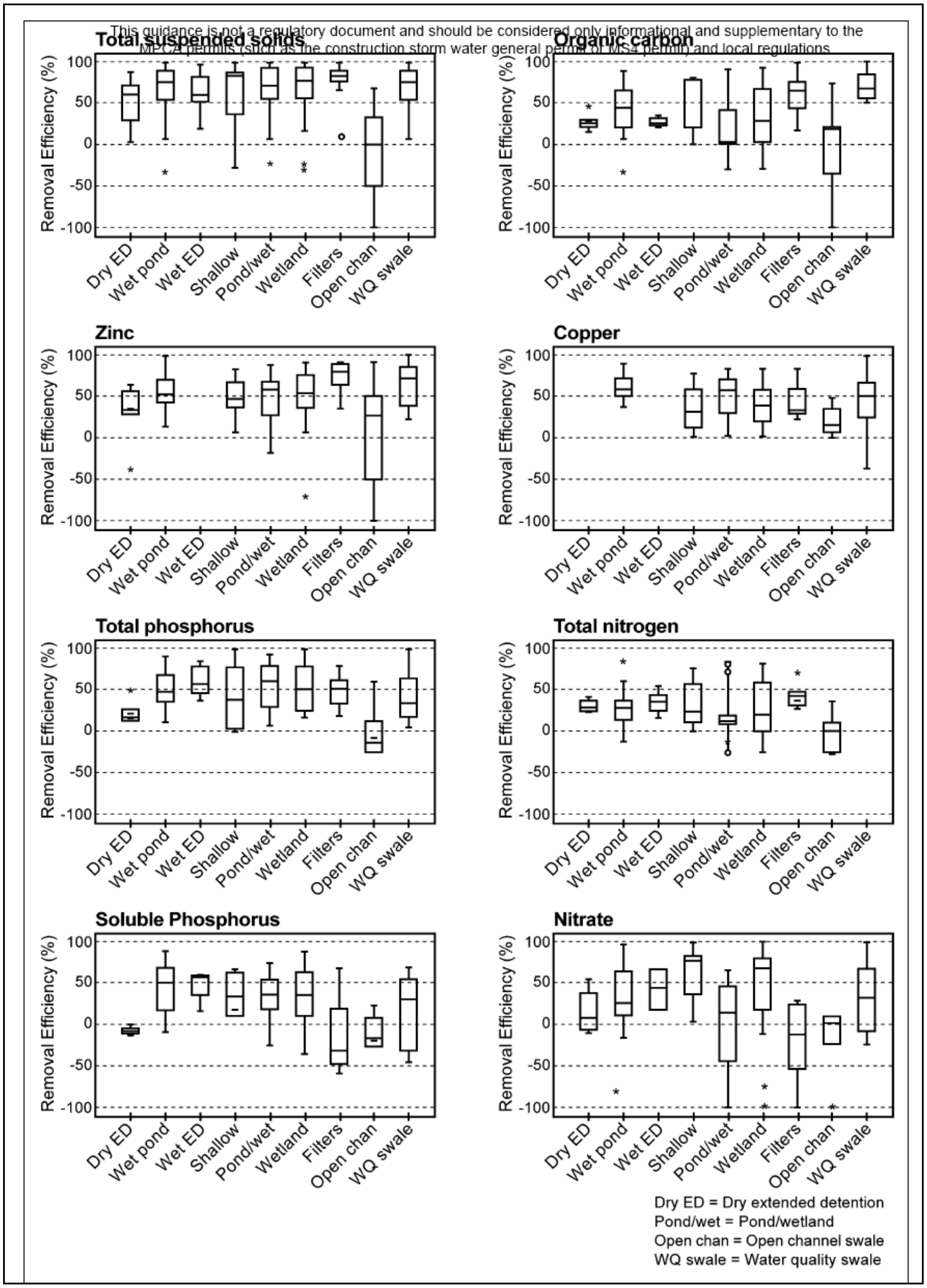


Figure 1.3: Pollutant removal by type of BMP system (MPCA Stormwater Manual) water quality and drainage area hydrology (Davis et al. 2001, 2003; Hunt et al. 2006; Davis 2007, 2008; Hunt et al. 2008).

BMP system performance varies drastically depending on environmental conditions. One of the important characteristics of a BMP is system response time, i.e., the expected time that it takes an average parcel of water to move through the transport and control system from the source to the outlet. Response times of a BMP vary from a few minutes for infiltration systems along highways to several days for regional storm-water detention ponds. Hydraulic response is faster than water quality response in general.

1.5 Monitoring BMPs

Effective management of environmental resources is predicated on the existence of well-conceived and appropriately implemented monitoring programs. In practice, environmental monitoring of pollutants often takes the form of manual sampling of an environmental matrix (water, air, or soil) followed by laboratory analysis. It is not uncommon, however, that the pollutant concentration varies faster than the turn-around-time of the sampling and analysis procedure. In such situations, the monitoring data may be of limited value for informing rapid management actions that might, for example, minimize exposure to toxins.

1.5.1 Discrete Sampling

Variable response times in BMP and BMP systems make evaluating BMP effectiveness necessary. This variability, compounded with changes in stream hydrology as a function of urbanization, further complicates monitoring BMPs and impacts on urban streams by BMPs. Generally grab sampling would be required to evaluate BMP performance. Interpretation of the grab sampling data from monitoring networks, however, is often problematic. With field based technologies, such as stormwater BMPs, the water flows (which are driven by rain storms) are not predictable or controlled. Thus, it is difficult to accurately measure the flows. Additionally, coordinating sampling for water quality parameters or for pollutants with the flow events is difficult if not impossible. Grab samples only provide "snapshots" of water quality at the moments of sampling, and frequencies are generally not sufficient to capture the dynamic behavior of shallow groundwater and surface water quality. Thus, obtaining data at the necessary spatial and temporal resolution to evaluate stormwater BMPs is challenging using traditional sampling methods. Even if data is collected during storm events, there are usually not enough samples at a given time (or under various conditions) to evaluate BMP performance with any statistical certainty. Lastly, such measurements are usually costly and time-consuming due to the necessary travel to the site, the labor associated with sample collection and processing, and the analytical costs.

1.5.2 Continuous Monitoring

Many researchers have tried to improve discrete sampling by developing and testing different methods for estimating loads from discrete grab sample data (Alexander et al. 1998). None of the methods tested to date, however, clearly outperformed the others, and the accuracy of the load estimates turned out to depend mainly on the sampling frequency. Increasing the frequencies of common sampling and laboratory analyses in regional monitoring programs is extremely laborious and expensive. Another option for increasing the measurement frequencies is the use of onsite automatic samplers and analyzers which can produce continuous water quality time series of many chemicals.

A continuous monitoring system provides intensive temporal data used to assess water quality on smaller timescales. Using *in situ* sensors, quantifying concentrations of target pollutants at high temporal and spatial density allows for the performance of stormwater BMPs to be determined in real-time.

1.6 Research Objectives

The objective of this portion of the project was to assess surface water quality and the performance of BMPs in two streams in Minnesota: Minnehaha Creek, an urban stream, and Painters Creek, a rural stream. The water quality measurements were taken using a real time water quality monitoring system called a Wireless Sensor Network (WSN). Using this system, in-stream water quality measurements were taken and pollutant loads were calculated. The system was also configured to determine BMP performance at three different constructed BMP systems. Collectively, these measurements were used to accurately estimate pollutant loads in an urban stream, BMP effectiveness, and the extent that local land use, hydrology and time influence BMP performance.

2. MATERIALS AND METHODS

The wireless Sensor Network was deployed in three separate locations during the duration of this monitoring project. The first and second monitoring locations were on Minnehaha Creek in Edina, MN and St. Louis Park, MN, respectively. Additional data collected in Painters Creek, in Mound, MN and along the Mississippi River are discussed in Part III. Water quality was measured in real-time at chosen BMP systems, which included constructed wetlands and detention ponds. The sensors were placed at the inflow and outflow of the BMPs and upstream and downstream of storm water inputs from a given BMP into Minnehaha Creek. Sensors were used to measure water temperature, depth, pH, specific conductance, turbidity/total suspended solids, dissolved oxygen, nitrate, and phosphate. All parameters were measured continuously, except for nitrate or phosphate, which have a maximum frequency of three measurements per hour. Additionally, grab samples (collected by an ISCO sampler) were used to cross check the sensor measurements (*e.g.,* nitrate, suspended solids) and allow measurement of selected pesticides and/or coliform bacteria via laboratory analysis. The ISCO sampler was triggered remotely at the beginning of storm events or other times to take samples at regular intervals to further analyze the impacts of stormwater and BMP systems on Minnehaha Creek.

2.1 Site Descriptions

2.1.1 Minnehaha Creek-Pamela Park

Pamela Park is a 64-acre city park located within Edina, MN which contains Pamela Lake and an 18.4-acre wetland designated as a Minnesota Department of Natural Resources Protected Wetland. It is part of the Minnehaha Creek Watershed District drainage area. Historically, existing wetland areas within the park were part of a much larger wetland complex that drained into Minnehaha Creek. With the gradual urbanization of the area, the more than 500 acres of fully developed residential areas that drained through Pamela Lake and eventually discharged into Minnehaha Creek caused substantial filling of wetland areas, sediment deposit from municipal storm sewers, and degrading water quality.

To improve water quality, sediment settling basins were constructed in 2001 at the two major storm sewer outlets to the lake/wetland at the southern end of Pamela Park. These basins were designed to capture and settle pollutants from the storm sewer runoff before it drains into the wetland, keeping pollutants from entering Pamela Lake and downstream, Minnehaha Creek. Three wetland-settling ponds were then built in the northern part of the park, starting in the spring of 2001, as an environmentally friendly, gravity-based method of cleaning surface water. The settling ponds receive storm water runoff from storm water drains with the intent that phosphorous would adsorb to the grit in the runoff and settle to the bottom of the ponds. The three stormwater ponds (Ponds 1, 2, and 3) treat runoff from 297 acres. The 297 acres includes 242 acres that currently discharge directly into Minnehaha Creek and 55 acres that currently drain into the wetland downstream of Pamela Lake. The primary goal in choosing this monitoring location is to determine the effectiveness of the three pond, BMP system in treating stormwater runoff that would eventually impact Minnehaha Creek.

The flow of water through the Pamela Park site and into Minnehaha creek is regulated by water levels in Pamela Lake and the three stormwater pond system north of the protected wetland as shown in Figure 2.1. The pond system was specifically monitored at this site to determine the effectiveness of the pond/wetland system at mitigation stormwater pollution from entering Minnehaha Creek. Stormwater runoff from the surrounding park and residential neighborhoods is routed to a storm sewer and into the first of the stormwater ponds. Water then flows from the first stormwater pond, to the second pond, then to the third pond, and finally into the wetland before reaching Minnehaha Creek. Monitoring stations were positioned at the inlet of the first stormwater pond, the outlet of the last stormwater pond, upstream of the BMP outlet to Minnehaha Creek, and downstream of the BMP outlet to Minnehaha Creek. These monitoring locations were specifically chosen to determine pollutant behavior in the BMP pond system, in the wetland system, and in the creek itself.

Land Use

The primary land use in Pamela Park is single family residential. Minnehaha Creek, at this monitoring location, is lined on the east side by single-family homes, and on the west side, there is a city park.

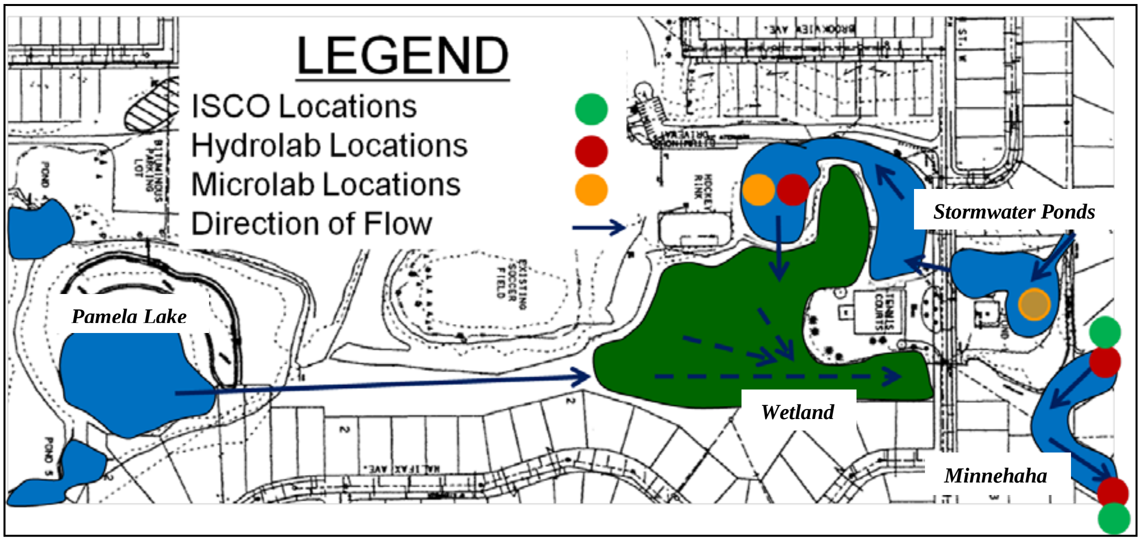


Figure 2.1: Pamela Park Site Map. The Microlab location with shading indicates that the Microlab was only function for a short time because the stormwater pond had no standing water after the first week of sampling.

Local Hydrology and Creek Impairments

Pamela Park, in the defined study area, is 26-50% impervious. Minnehaha Creek is also on the federal CWA 303(d) list of impaired waters for violating the Minnesota chloride and dissolved oxygen water quality standards for a Class 2 water body (MPCA, 2010). Additional impairments within the creek include fecal coliforms and fish biota, and multiple lakes along the creek are impaired because of excess nutrient concentrations (MPCA, 2008). The deadline for most of these TMDLs is 2012.

2.1.2 Minnehaha Creek-Saint Louis Park, MN

Saint Louis Park is a suburb of Minneapolis Minnesota. Starting in 2004, a business expansion constructed two stormwater ponds as a means of treating stormwater runoff from 15.51 acres of commercial land. The stormwater ponds receive direct runoff from a large parking lot. A constructed wetland, owned by the city, receives the water collected from the stormwater ponds for further treatment before the water is released into Minnehaha Creek. The area surrounding this site includes the shopping mall, as well as other commercial industries.

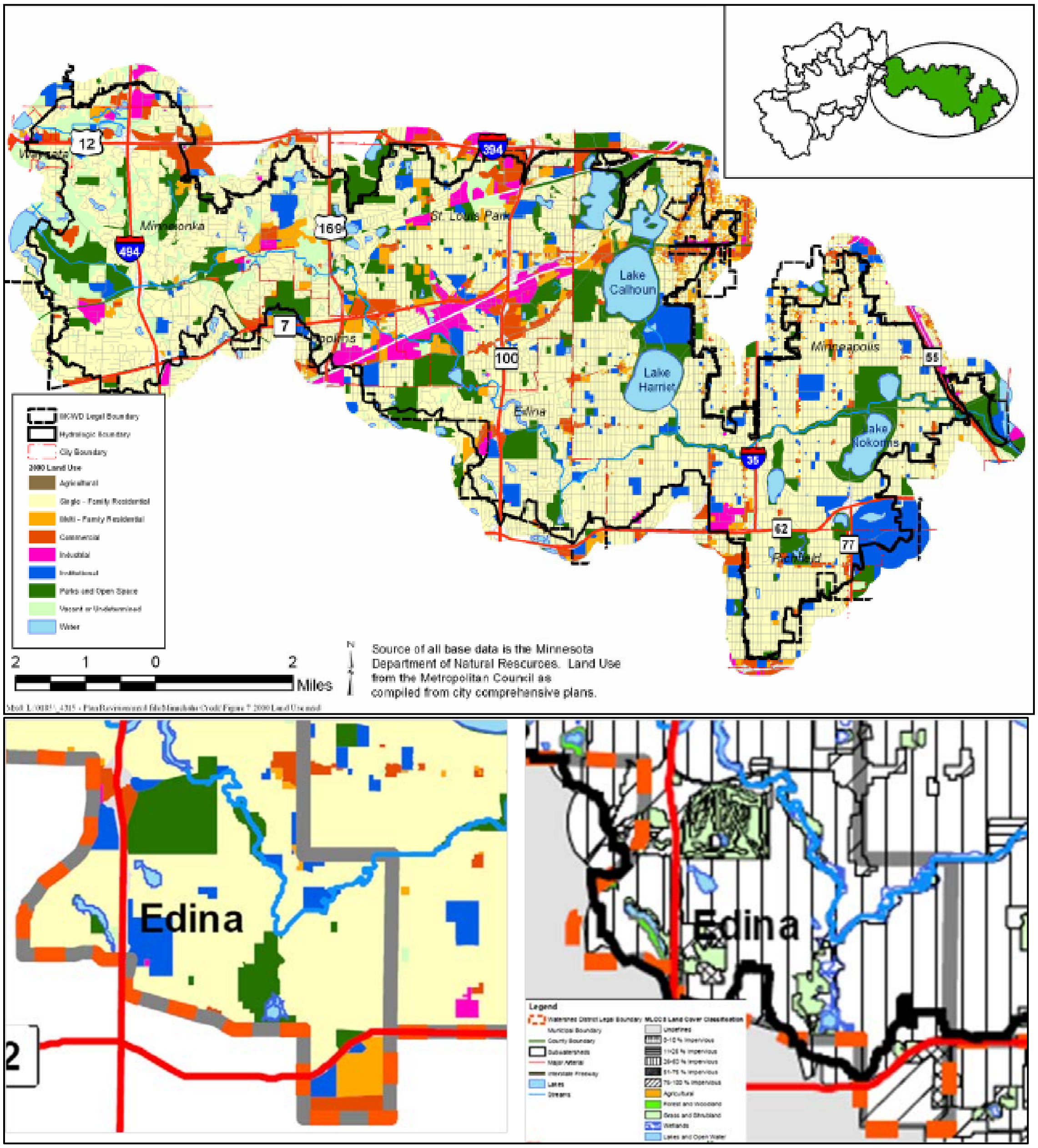


Figure 2.2: Minnehaha Creek land use and percent imperviousness in the Pamela Park area.

Flow Path

The flow of water through the Knollwood site and into Minnehaha creek is regulated by water levels in a series of stormwater ponds, a smaller pond that feeds into a second, larger one, as well as a constructed wetland system as shown in Figure 2.3. The pond system was specifically monitored at this site to determine the effectiveness of the constructed wetland, in particular, at reducing stormwater pollution from entering Minnehaha Creek. Stormwater runoff from the surrounding commercial buildings and parking lots is routed to storm sewers and into both of the stormwater ponds. Water can flow from the first stormwater pond into the second pond. Water is also collected directly by the second (larger) stormwater pond. Stormwater from the second stormwater pond then flows to the wetland before reaching Minnehaha Creek. Monitoring stations were positioned at the inlet of the second (larger) stormwater pond, upstream of the BMP outlet to Minnehaha Creek, and downstream of the BMP outlet to Minnehaha Creek. The first stormwater pond could not be monitored because of logistical issues associated with assembling a monitoring station at that location. These monitoring locations were chosen to determine pollutant behavior in the BMP pond system, in the wetland system, and in the creek itself.

Land use

The primary land use at the Knollwood site is commercial. Minnehaha creek is surrounded on both sides by commercial buildings.

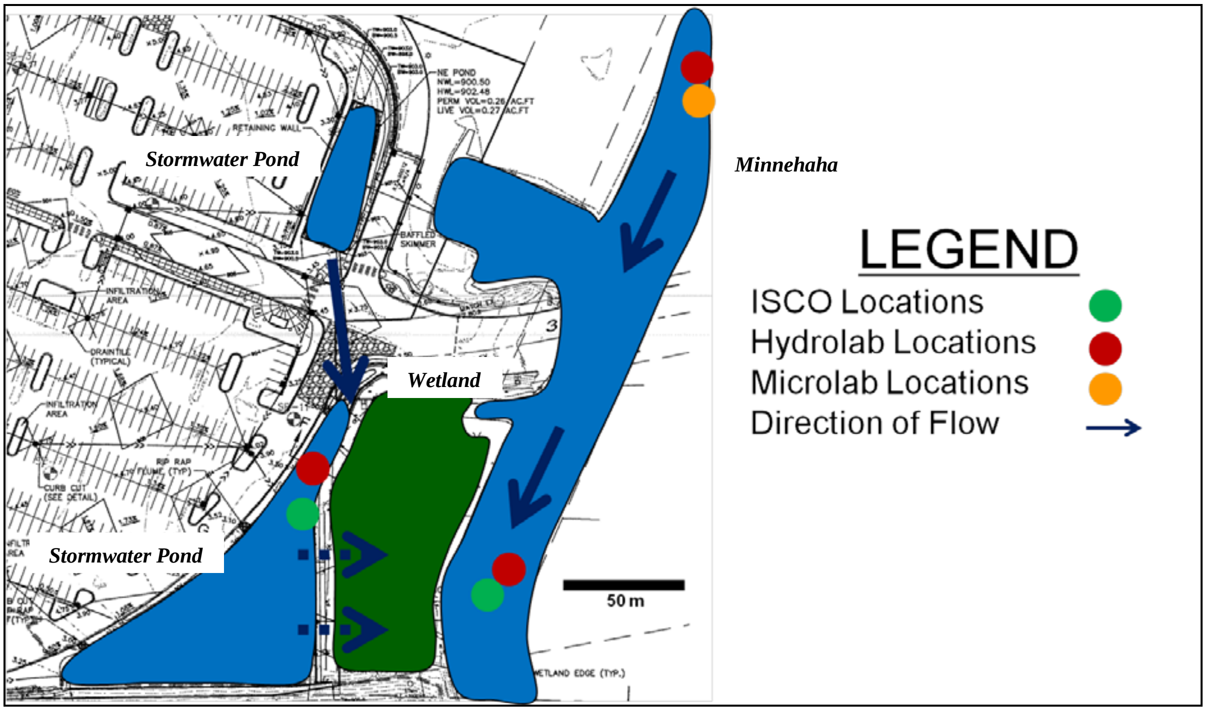


Figure 2.3: Knollwood Site Map.

<u>Local hydrology and Creek Impairments</u>

The sampling site, in the defined study area, is 76-100% impervious. Minnehaha Creek is also on the federal CWA 303(d) list of impaired waters for violating the Minnesota chloride and dissolved oxygen water quality standards for a Class 2 water body (MPCA, 2010). Additional impairments within the creek include fecal coliforms and fish biota, and multiple lakes along the creek are impaired because of excess nutrient concentrations (MPCA, 2008). The deadline for most of these TMDLs is 2012.

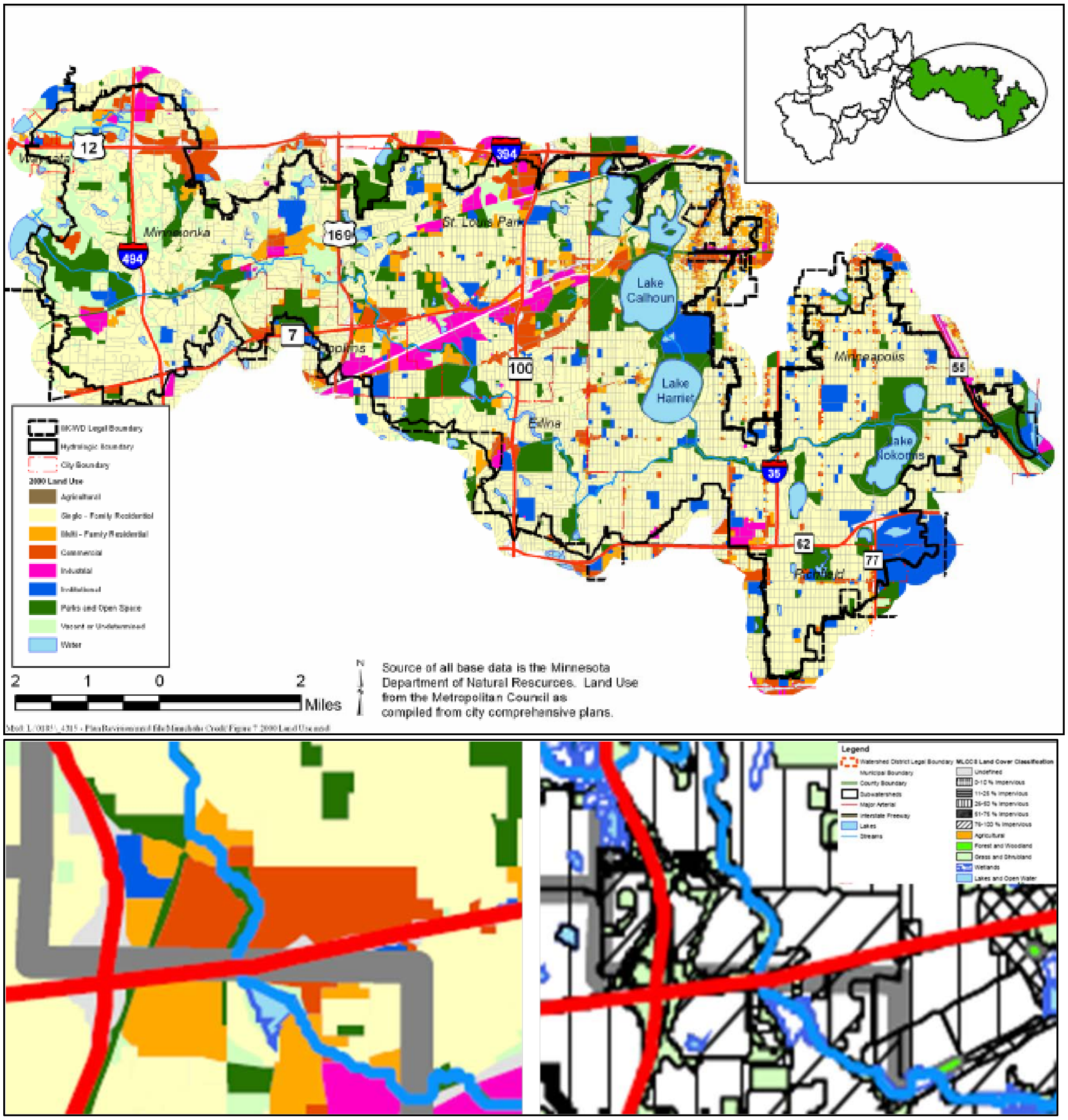


Figure 2.4: Minnehaha Creek land use and percent imperviousness at the Knollwood site.

2.2 Network and Monitoring Strategy

2.2.1 Stream Flow

Options for tracking water quality vary with the type of water resource. For example, a monitoring program for ephemeral streams can be different from that for perennial streams or large rivers. Freshwater streams can be classified on the basis of flow attributes as intermittent or perennial streams. Intermittent streams do not flow at all times and serve as conveyance systems for runoff. Perennial streams always flow and usually have significant inputs from ground water or interflow. For intermittent streams, seasonal variability is a very significant factor in determining pollutant loads and water quality. During some periods sampling may be impossible due to no flow. Seasonal flow variability in perennial streams can be caused by seasonal patterns in precipitation or snowmelt, reservoir discharges, or irrigation practices. For the purposes of this study, both sampling locations on Minnehaha Creek are considered perennial streams as the creek is not just a conduit for runoff. During the summer of 2009, however, Minnehaha Creek did not have consistent stream flow due to unusually low rainfall in the summer season. Gray's Bay dam, used as flow control for the creek, was also closed due to the dry conditions, further exacerbating the lack of stream flow in the creek.

2.2.2 Sampling Frequency

It was expected that the greatest concentrations of suspended sediment and other pollutants occur during spring runoff or snowmelt periods. Concentrations of both particulate and soluble chemical parameters have also been shown to vary throughout the course of a rainfall event in many studies. This seasonal and short-term variability was considered in developing monitoring the monitoring strategy for the two sampling locations. It was determined that sampling snowmelt and spring runoff were necessary for the analysis of stream behavior. Thus at both the Pamela Park and the Knollwood sites, monitoring equipment was deployed during the spring melt season. Grab samples were also collected in-stream at the Knollwood site to capture the change in stream pollutant concentrations from the winter through the snowmelt.

2.2.3 Spatial Variability

Spatial variability was also considered in the monitoring strategy for these sites. Spatial variability is largely lateral for both intermittent and perennial streams. Vertical variability does exist, however, and can be very important in both stream types, especially during runoff events. Intake depth is often a key factor in stream sampling. The effects of vertical and lateral variability were addressed by keeping the sensor locations at the same spot (vertically and laterally) in the streamflow at all times. Efforts to minimize the impacts of these variables, however, do not rid the data of variability due to spatial variability in the stream flow (Spooner et al., 1985).

2.2.4 Monitoring Design

The monitoring design for this study was a combination of an upstream-downstream design and an inflow-outflow design. Upstream-dowstream design was used specifically for monitoring Minnehaha Creek while inflow-outflow design was used for monitoring BMP systems (e.g., stormwater ponds).

In the upstream-downstream design, there is one station at a point directly upstream from the area where implementation of management measures will occur and a second station directly downstream from that area. Upstream-downstream designs are generally more useful for documenting the magnitude of a nonpoint source than for documenting the effectiveness of nonpoint source control measures (Spooner et al., 1985), but they have been used successfully for the latter. This design provides for the opportunity to account for covariates (e.g., an upstream pollutant concentration that is correlated with a downstream concentration of same pollutant) in statistical analyses. An upstream-downstream design was needed for this study because upstream activities that were expected to confound the analysis of downstream data.

Inflow-outflow, or process, designs are very similar to upstream-downstream designs. This monitoring design, however, lends itself to studies of individual management measures or practices. For the purposes of this study pollutant loading at the inflow and outflow of a BMP was measured to determine the pollutant removal efficiency of the BMP. In general, no inputs other than the inflow are present, and the only factor affecting outflow is the management measure (Spooner et al., 1985). When dealing with stormwater runoff, and, in particular, the analysis of rain events, the inflow/outflow dynamics are compounded by multiple stormwater inlets to the same BMP.

2.2.5 BMP Performance Analysis

An essential part of determining how a BMP system should be monitored is determining the set of management measures to be implemented. Management measures can generally be classified with respect to their modes of control. These modes of control include, source reduction, delivery reduction, or the reduction of direct impacts. Source reduction measures all rely on the prevention of nonpoint source pollution; trapping and treatment mechanisms are not relied upon for control. Delivery-reduction measures include those that rely on detention basins, filter strips, constructed wetlands, and similar practices for trapping or treatment prior to release or discharge to receiving waters. Delivery reduction measures have been implemented at the study locations, therefore BMP monitoring of the BMP systems must be tailored to determining trapping and treatment efficiency.

Delivery-reduction measures lend themselves to inflow-outflow (or process) monitoring to estimate the effectiveness in reducing loads. The simple experimental approach is to take samples of inflow and outflow at appropriate time intervals to measure differences in the water quality between the two points. Sampling of the BMP system was implemented based on this approach. Sensors, as logistically possible, were placed at the inflow and outflow of each BMP and grab samples, when taken, were also taken at the inflow and outflow of the systems.

2.2.6 Load Versus Water Quality Status Monitoring

Monitoring was conducted to determine the condition of the water resource and the pollutant load to the water resource. Loading is the rate of pollutant transport to the creek via overland, tributary, or ground-water flow. Load monitoring was used to assess the change in magnitude of major pollutant sources or to assess the change in pollutant export at a fixed station. Monitoring water quality status includes measuring a physical attribute or chemical concentration and was used to assess baseline conditions, trends, and the impact of treatment on Minnehaha Creek.

Load monitoring requires a complex, and typically expensive, sampling protocol to measure water discharge and pollutant concentration (Coffey and Smolen, 1990). Both discharge and concentration data are needed to calculate pollutant loading. For the purposes of this study, discharge parameters were determined using a USGS monitoring station on Minnehaha Creek where discharge was already being monitored.

Water quality status was evaluated for designated use attainment, standards violations and monitoring indicator parameters. Sensor parameters were specified by the State water quality standards including chloride, total suspended solids, and nutrients, as well as pollutants of interest to the watershed districts (Coffey and Smolen, 1990).

2.2.7 Wireless Sensor Network Description

A Wireless Sensor Network (WSN) was used to assess the effectiveness of stormwater BMPs in mitigating pollutant loads from urban and suburban sources. The wireless sensor network is capable of monitoring fundamental water quality parameters at high spatial and temporal resolution. Field studies were conducted at the previously mentioned Minnehaha Creek BMP systems from April to October of 2009 and March to October of 2010. Data collected during these field studies was used to calculate pollutant loads in-stream and in BMPs, as well as model stream behavior during dry and rain events.

A WSN consists of autonomous sensors that communicate with a base station, while the base station communicates with a server network. Through this network, data collected by the sensors can be transmitted back to the network remotely and in near real-time. The University of Minnesota WSN is currently comprised of five datalogger stations (*Campbell Scientific*, 4 CR1000, 1 CR206) which are equipped with water quality sensors, a radio for communication between the stations, and a single cell modem on the "base station" for communication with the server at the Saint Anthony Falls Laboratory. The data loggers run sampling programs created in CR Basic and are interfaced via the program LoggerNet. The five complete stations of the WSN also consist of tripods, watertight boxes, PS 100 power supplies and solar panels (*Campbell Scientific)*. A tipping bucket rain gauge is also attached to one of the stations.

Data Sondes

The Hach Environmental HydroLab Data Sonde is a measurement device designed to combine the sampling of multiple water quality parameters into one array of measurements which is then stored in the WSN station for retrieval. The HydroLabs were calibrated as specified by the manufacturer. The sensors were fully and thoroughly cleaned and calibrated on a monthly basis. The HydroLabs were programmed to measure temperature, pH, specific conductance, turbidity, depth, and dissolved oxygen at 1-minute intervals.

Nutrient Analyzer

The MicroLab is an *in situ* autosampler developed by EnviroTech, LLC, that measures concentration of nitrate or phosphorus, depending on which detector and reagents are installed. The device extracts a sample from the stream, performs the appropriate reagent chemistry on the sample and determines the inorganic chemical concentration based on comparative spectrophotometry with respect to a standard concentration. The MicroLab then outputs its measurement to the datalogger for storage and retrieval. The MicroLab automatically recalibrates itself to the on-board standard after every five water samples, ensuring data quality control. The MicroLab Nutrient Analyzers were either set up to measure nitrate/nitrate (as N) using EPA Method 353.4 / 4500-NO_3 (Arar et al., 1997) or phosphate (as P) using using the Drummond and Maher method (1995). The method for the determination of phosphorus in aqueous solution occurs via the formation of phosphoantimonylmolybdenum blue complex.

Automated Grab Sampling

Two Teledyne ISCO samplers were placed within at each sampling location. One was placed in the stormwater BMP (normally a pond) and another was placed either upstream or downstream of the pond outlet. With samplers in these locations the water quality in the pond and in the stream prior to being effected by water released by the stormwater pond was assessed. To analyze the impact of stormwater, these samplers were set to collect twelve 900 mL samples at two hour intervals and were triggered a few hours before a storm so that a baseline could be established. The 24 samples collected from the two stations were analyzed as soon as possible for fecal coliform bacteria contamination and nutrient concentrations. These samples were stored at 4 ^{o}C prior to analysis, with analysis for fecal coliform bacteria taking place within 24 hours of collection.

2.3 Data Collection and Analysis

Water quality data collected using the Hydrolabs was data quality controlled for the duration of the experimental period. The calibration of these sensors required Hydras 3 LT software. The Microlab sensor was calibrated using an internal standard.

Data collected were analyzed and plotted as raw concentrations. Contaminant loads were calculated by multiplying the raw concentrations of each pollutant by the daily average streamflow for the day as reported by the USGS at USGS station number 05289800. The station is located on Minnehaha Creek at Hiawatha Ave. in Minneapolis, MN. These loading values determine the mass of pollutant per unit time that passes the monitoring station in the stream. Pollutant load is determined by multiplying the total runoff volume by the average concentration of the pollutant in the runoff. As described in Section 2.2.6, status monitoring was conducted continuously using the WSN. Pollutant load determination was necessary because the objective of a TMDL is to reduce pollutant loading into the impaired water body. TMDL standards are quantified as loads; therefore the water quality data that was analyzed in this study was calculated as a pollutant load.

Calculated pollutant loads were then plotted in load duration curves. Load duration curves present the data as a function of the load duration interval. This interval describes the percent of loading rates that are less than the loading rate plotted.

2.3.1 Chloride and Total Suspended Solids Calculations

Pollutants of interest in this study included nitrate, specific conductance, and turbidity. The surrogate relationships (Equations 1 & 2) were used to estimate chloride concentration and loads from specific conductance measurements and total suspended solids (TSS) concentrations from turbidity measurements. These equations were developed for sites within close proximity to our monitoring sites (Christianson et al., 2007) and have been used in previous studies to describe stream behavior and BMP performance (Henjum et al., 2008).

Water quality monitoring within Minnehaha Creek is performed routinely by the USGS and the Minnehaha Creek Watershed District (MCWD). Several locations within the creek are monitored continuously for stage height and specific conductance. Grab samples collected on a bi-weekly basis were analyzed for a variety of parameters, including TSS, chloride, nutrients, and fecal coliforms (Christianson et al, 2007). A relationship between chloride and specific conductance was also developed using this data for Minnehaha Creek (Equation 1).

$$\boldsymbol{Cl^{-}\left(\frac{mg}{L}\right) = 0.22 \times Specific\ Conductance\ \left(\frac{\mu S}{cm}\right), R^{2} = 0.82} \qquad (\boldsymbol{Equation\ 1})$$

A regression analysis was also used in the development of an equation that relates turbidity to TSS (EPA, 2005, Fletcher et al, 2007)). Values for the coefficient, *k* in Equation 2 have been reported to range from 0.3 to 1.4, depending on the geology and land-use of the watershed as well as the creek characteristics (EPA, 2005, Fletcher et al, 2007). No studies have been performed at either Shingle or Minnehaha Creek to develop this correlation. Therefore, it was assumed that *k* was equal to 1.0. The significance of this assumption is minimal, as the primary function of this conversion is to determine a mass loading rate of TSS. The functionality of the WSN could also be used to determine *k* for a specific water body, or reach of a water body. By taking grab samples and turbidity measurements during the same time intervals, a correlation could be obtained between TSS and turbidity. This exercise, however, was not completed as part of this research.

$$\boldsymbol{TSS}\ \left(\frac{\boldsymbol{mg}}{\boldsymbol{L}}\right) = \boldsymbol{k} \times \boldsymbol{Turbidity}\ (\boldsymbol{NTU}) \qquad (\boldsymbol{Equation\ 2})$$

2.3.2 Load Duration Curve

A load duration curve facilitates characterizing concentrations/loads for different flow regimes while providing a visual display of the relation between flow and loading capacity. Load duration curves also account for how seasonal stream flow variation affects water quality. Conversely, this analytical method does not itself account for fate and transport mechanisms; load duration curve analysis by itself assumes that flow is the primary driver of water quality variation. Therefore, if processes other than flow affect the loading then use of additional assessment tools needs to be considered.

A load duration curve is developed, first, by creating a flow duration curve for the site in question. Stream flow is ranked from highest to lowest and then the percent of days these flows were exceeded (= rank ÷ number of data points) is determined. To compare water quality sample data, the pollutant concentration and streamflow are multiplied together for a particular corresponding data point to determine the pollutant load. These loads are then paired with the flow data point used to calculate that load to determine the value for “Percent of Days Flow Exceeded” which is equivalent to “Percent of Days Load Exceeded”. These load and percent data points are then plotted on the load duration curve. A curve can then be plotted by multiplying the water quality standard by the stream flow and plotting the corresponding data. If both sets of data are used, points above the curve (calculated loads using WSN collected water quality data) represent exceedances of the quality standards and the associated allowable loadings.

2.3.3 Cumulative Distributions

The water quality data that were collected over the duration of this study were also analyzed using cumulative distribution diagrams. For each pollutant loading rate computed, the value was ranked for the assessment period from largest to smallest. These values, as ranked, were then sequentially assigned percents as value that estimate percent of time. The paired percent of time and loading rate were then plotted. The resulting curve is the cumulative distribution diagram describing the percent of time that a loading rate would be less than the loading rate chosen. This cumulative distribution plot can then be compared to a reference cumulative distribution (a numerical water quality standard). If at any point the assessment cumulative distribution exceeds the reference cumulative distribution, a given level of noncompliance occurs more often than is allowed.

As defined above, the cumulative distribution is a data driven formulation. Because data from the WSN are collected continuously in time, a determination of the true state of the water body is possible. While the information needed to construct the ideal cumulative distribution, including continuous site-specific stream flow data, is not obtainable, it is important that the obtained data represent this ideal as a feasible monitoring scenario.

2.3.4 Efficiency Ratio

Sections 2.2.4 and 2.2.5 describe a monitoring system design based on inlet and outlet monitoring of BMP systems and stream flow. An efficiency ratio (ER) was used to calculate the relative change in concentration from the inlet to the outlet of a BMP based on the inlet pollutant concentration (Strecker et al., 2001) . The efficiency ratio is defined in terms of the average event mean concentration (EMC) of pollutants over some time period:

$$\boldsymbol{ER = 1 - \frac{average\ outlet\ EMC}{average\ inlet\ EMC}}$$

$$\boldsymbol{= \frac{average\ inlet\ EMC - average\ outlet\ EMC}{average\ inlet\ EMC}}$$

$$(\boldsymbol{Equation\ 3})$$

EMCs can be calculated from discrete measurements. The EMC for an individual event or set of field measurements, where discrete samples have been collected is defined as:

$$\boldsymbol{EMC = \frac{\sum_{i=1}^{n} V_i C_i}{\sum_{i=1}^{n} V_i}} \qquad (\boldsymbol{Equation\ 4})$$

where, V is the volume of flow during period I, C is the average concentration associated with period i, and n is the total number of measurement periods during the event.

The arithmetic average EMC is defined as,

$$\boldsymbol{average\ EMC = \frac{\sum_{f=1}^{m} EMC_f}{m}} \qquad (\boldsymbol{Equation\ 5})$$

Where, m is the number of events measured. This method weights EMCs from all storms equally regardless of relative magnitude of a storm. For example, a high concentration/high volume event has equal weight in the average EMC as a low concentration/low volume event. This method is most useful when loads are directly proportional to storm volume. For example, an EMC would not be as descriptive for chloride behavior in the summer, because a high volume event would generally also be a low chloride concentration event because of dilution. For work conducted on nonpoint pollution (i.e., inflows), the EMC has been shown to not vary significantly with storm volume (EPA/ASCE, 2002).

This method is taken directly from non-point pollution studies and does a good job characterizing inflows to BMPs but fails to take into account some of the complexities of BMP design (e.g., a media filter that treats to a relatively constant level that is independent on inflow concentrations). This analytical method assumes that if all storms at the site had been monitored, the average inlet and outlet EMCs would be similar to those that were monitored (Strecker et al., 2001, EPA/ASCE, 2002).

2.3.6 Relevant Period of Impact

The period of analysis used in an efficiency calculation is important. The period used should take into account how the parameter of interest varies with time. This allows for observation of relevant changes in the efficiency of the BMP on the time scale in which these changes occur. In addition to observing how factors, such as climate, affect efficiency as a function of time, it is important to relate the calculation period to the potential impact a given constituent would have on the receiving water. Analysis was conducted on time scales from an “intra-storm” basis to a seasonal basis (Table 2.1) (EPA/ASCE, 2002).

Table 2.1: Time scales for monitoring of different water quality parameters captured by the WSN.

Time Scale for Analysis	**Water Quality Parameter**
Short Term	DO
Long Term	Organics, Carcinogens
Both Short and Long Term	TSS, Nitrogen, Phosphorous, Temperature, pH

3. RESULTS AND DISCUSSION

3.1 Pollutant Loads and In-Stream Pollutant Concentrations

The ultimate goal of installing BMPs is to reduce the loads of pollutants to surface waters receiving the stormwater. Pollutant discharges are increasingly being set by limits on the total maximum daily load (TMDL). TMDLs are generally calculated by taking a grab sample at a specified interval (i.e., once a week or twice a month) and then assuming that sample is representative of the water quality over a period of time. A sensor network that can measure pollutants (*e.g.*, suspended solids, nitrate) with high temporal frequency will capture "event-driven" loads that would not be captured via grab samples. For example, first flush stormwater is known to have high levels of undesirable chemical/biological pollutants. While these loads will be missed by grab sampling, a sensor network that continuously measures these parameters will provide an integrated record of the loads entering the BMP, captured by the BMP, and the load to the receiving water. Thus, a more accurate TMDL will be determined and the specific portion of removal attributable to a targeted BMP can be determined.

Pollutant concentrations were measured at Pamela Park on Minnehaha Creek from April-June of 2009 and at the Knollwood site on Minnehaha Creek from August-October of 2009. The pollutant concentrations were compared against the water quality standards for Minnehaha Creek as shown in Table 3.1. Pollutant concentrations for specific conductance and turbidity were also calculated as chloride and total suspended solids using the correlations described in the equations in the Material and Methods section. Using these concentrations, pollutant loads were calculated by multiplying the in-stream pollutant concentrations by the stream flow. Calculating accurate pollutant loads are important because TMDLs are based on these loading calculations. As shown in Figures 3.1, 3.2, 3.8, and 3.9, pollutant loads are highly dynamic and fluctuate in large part because of storm events.

Table 3.1: Enforceable water quality standards for Minnehaha Creek including chloride, turbidity, and dissolved oxygen.

Pollutant	Units	Chronic Standard	Basis for Chronic Standard	Maximum Standard	Basis for Maximum Standard
Chloride	mg/L	230	Toxicity	860	Toxicity
Turbidity	NTU	25	NA	---	---
Dissolved Oxygen*	mg/L	NA	NA	NA	NA

*5.0 mg/L as a daily minimum. No site-specific standard shall be less than 5 mg/L as a daily average and 4 mg/L as a daily minimum. Compliance with this standard is required 50 percent of the days at which the flow of the receiving water is equal to the $7Q_{10}$.

3.1.1 Minnehaha Creek-Pamela Park

In-stream pollutant loads of chloride and total suspended solids for the upstream sampling location at Pamela Park are shown in Figures 3.1 and 3.2 though there is no significant statistical correlation between rainfall amount and chloride loading rate or TSS loading rate, it is clear that there are changes in pollutant loads when it rains. The chloride loading rate decreases with rainfall due to dilution with the additional water in the stream flow. That trend may be opposite, however, in the spring, when chloride remaining from road-salt application is flushed into the creek in very high concentrations. The increase in water volume may not be enough to dampen the affects of the high chloride concentrations in this case.

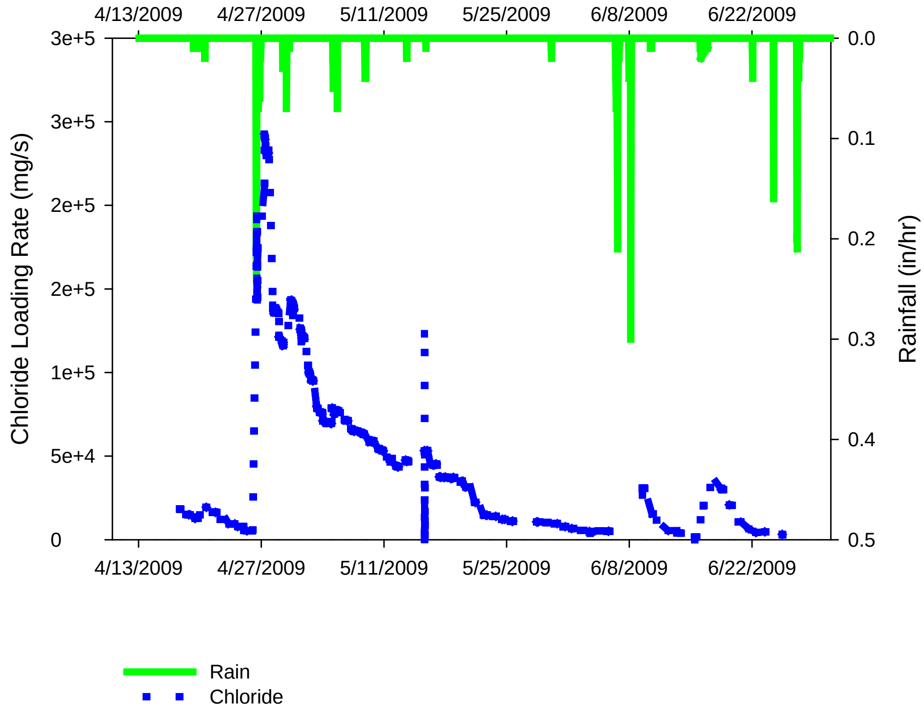


Figure 3.1: Chloride loading rate (mg/s) at the Pamela Park site with associated rainfall (in/hr). Parameter concentrations are displayed on the primary vertical axis and rainfall is displayed on the secondary vertical axis. Parameter concentrations are displayed for only the sampling location upstream of the BMP System. The inverted solid areas correspond to rainfall intensity.

Both TSS and chloride loading rates decrease starting in the early spring to the beginning of July. These decreases in loading rates are due in part to decreases in stream flow and the tapering effects of the snow melt as the summer progresses. The TSS loading rate is very high in the spring, in part due to debris associated with the snow melt. Significant increases in the TSS loading rate also occur with rainfall.

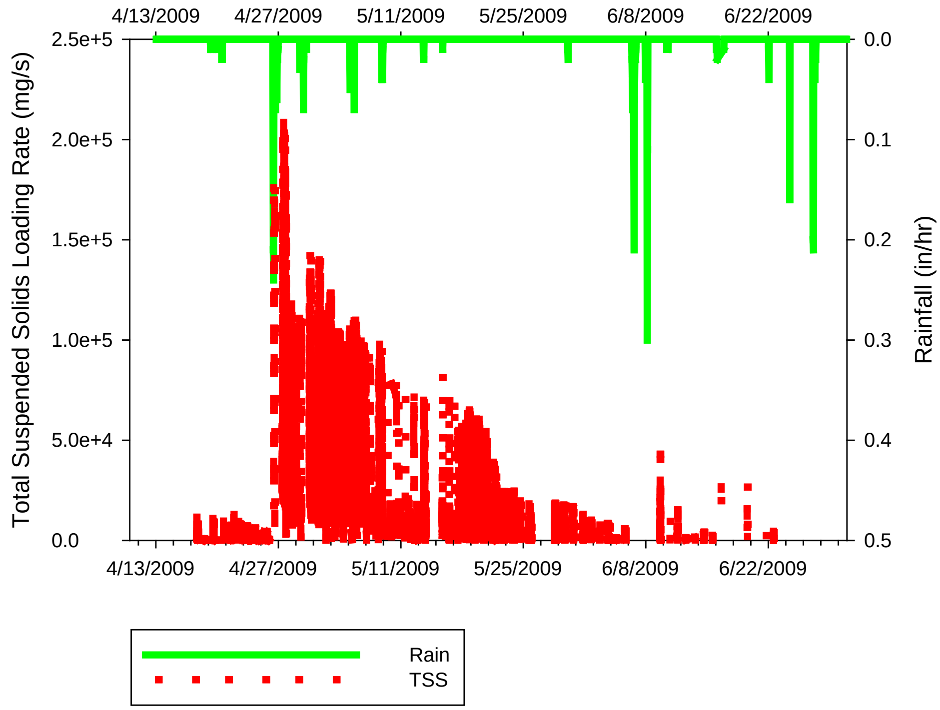


Figure 3.2: Total suspended solids loading rate (mg/s) at the Pamela Park site with associated rainfall (in/hr). Parameter concentrations are displayed on the primary vertical axis and rainfall is displayed on the secondary vertical axis. Parameter concentrations are displayed for only the sampling location upstream of the BMP System. The inverted solid areas correspond to rainfall intensity.

Temperature, pH, dissolved oxygen, specific conductance, and turbidity measurements that were taken with the WSN are presented in Figures 3.3-3.7. Diurnal fluctuations and seasonal increases in temperature are illustrated in Figure 3.3. The lack of nighttime temperature readings is due to the loss of power by the WSN at night. In-stream temperature and fluctuations in this temperature exerts a major influence on the biological activity and growth of aquatic organisms. Generally, the with higher water temperatures, the biological activity in the stream increases. Thermal pollution is less likely to occur in an area like Pamela Park because of the lack of elevated stormwater temperature because of minimal road and parking lot runoff. Water temperature fluctuations are also mitigated by the shade provided by the wooded areas in the park. High turbidity values may also increase water temperature through light absorption.

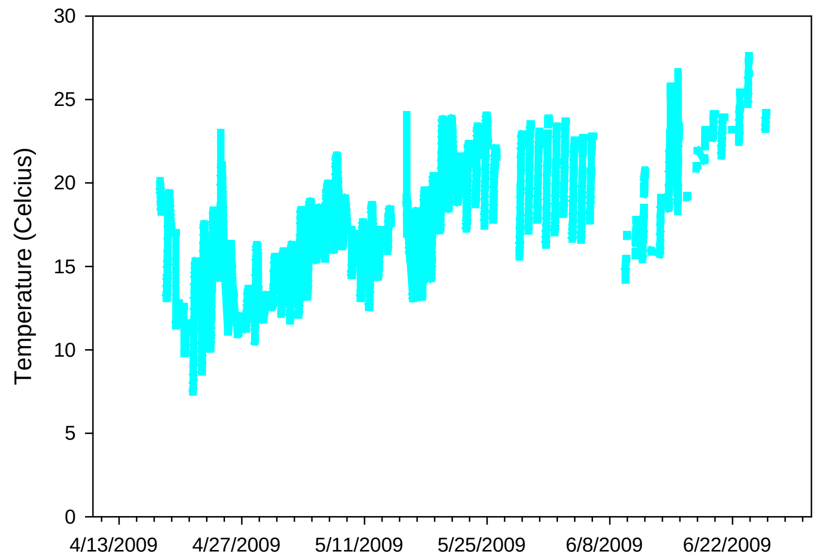


Figure 3.3: Temperature at the Pamela Park site for the monitoring period. Parameter concentrations are displayed on the primary vertical axis.

pH values (Figure 3.4) during the monitoring period also fluctuate on a diurnal cycle. Values are higher in the spring than the summer, while staying within the bounds of the pH standards for Minnehaha Creek. Values are missing at night because of the aforementioned power outages.

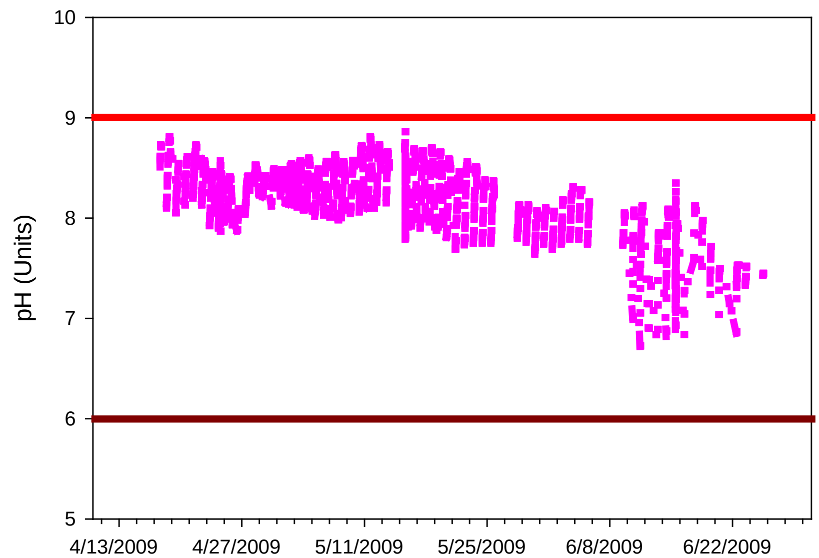


Figure 3.4: pH at the Pamela Park site for the monitoring period. The top solid line at pH 9 is the maximum standard for pH and the bottom solid line is the minimum standard for pH at pH 6. Parameter concentrations are displayed on the primary vertical axis.

Dissolved oxygen concentrations (Figure 3.5) in the creek reach a peak during the day and a minimum at night. The diurnal pattern of light intensity is associated with photosynthesis, aeration and diffusion during the day and an increase in the rate of utilization of dissolved oxygen (total respiration) during the night by the microbial populations in the creek. Overall, dissolved oxygen values decreased during the monitoring period. This decrease could be a result of a dry summer and therefore a decrease in stream flow and reaeration, as well as an increase in temperature. The inverse relationship between temperature and dissolved oxygen means that an increase in temperature causes a decrease in dissolved oxygen.

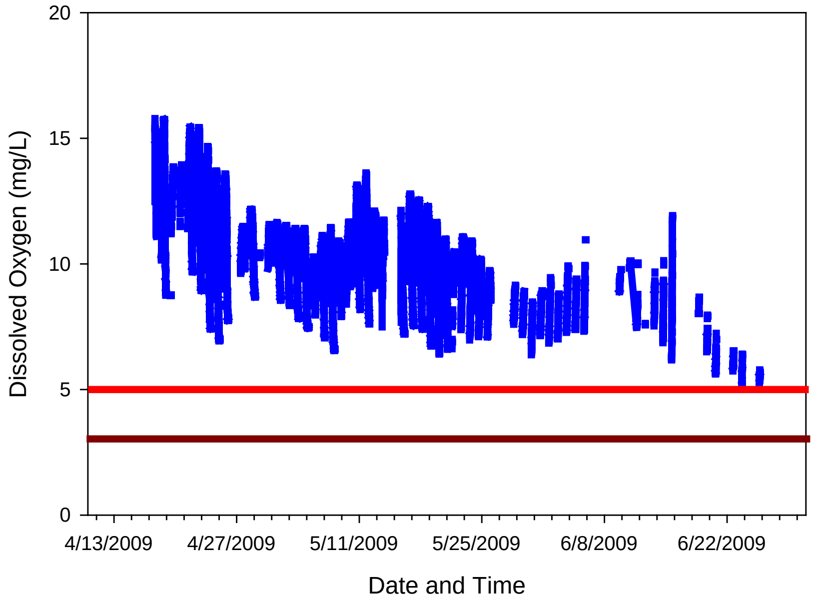


Figure 3.5: Dissolved Oxygen (mg/L) at the Pamela Park site for the monitoring period. The top solid line at 5 mg/L is the daily average standard and the bottom solid line is the daily minimum standard at 4 mg/L. Parameter concentrations are displayed on the primary vertical axis.

Monitoring of specific conductance revealed the highest values in the spring, most likely as a result of chloride in snow melt from road salt (Figure 3.6). Sharp decreases in specific conductance are a result of dilution by rainwater. Specific conductance is not evaluated directly against a water quality standard because the water quality standard for chloride is listed in mg/L of chloride. Equation 1 was used to calculate the chloride concentrations in the creek and whether or not the water quality standard was exceeded. The chloride concentration was above the chronic chloride standard only in April to early May. It is clear that the spring is the most important time in the monitoring period to mitigate the chloride entering the creek at this location.

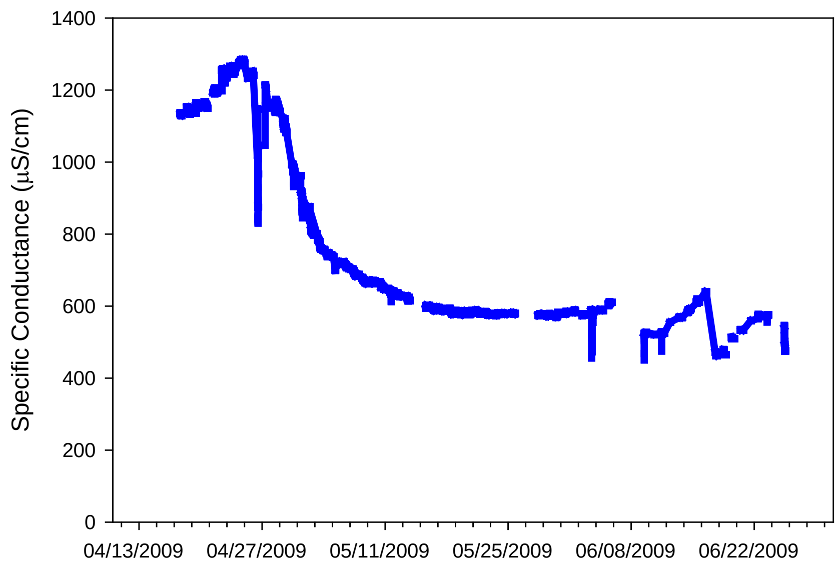


Figure 3.6: Specific conductance (µS/cm) at the Pamela Park site for the monitoring period. Specific Conductance is used as a surrogate for chloride. Parameter concentrations are displayed on the primary vertical axis.

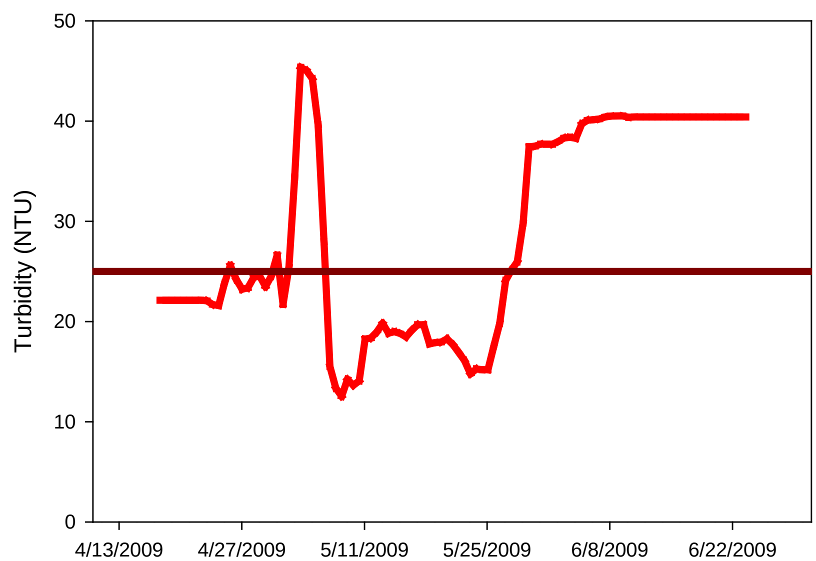


Figure 3.7: Turbidity (NTU) at the Pamela Park site for the monitoring period. The horizontal line is the Turbidity water quality standard at 25 NTU. Parameter concentrations are displayed on the primary vertical axis.

Turbidity, as shown in Figure 3.7, is reported as a moving average of the monitoring measurements. Scatter occurred in large part to debris collecting on the sensors, which was removed on at least a weekly basis. It is clear that turbidity, without being flow weighted, does not mimic the loading rates that were shown in Figure 3.2. During the month of May turbidity values are at their lowest values and in June the vales increase, most likely due to a lack of stream flow as well as fine sediment easily resuspended close to the stream bed. The turbidity data indicated that on many occasions the water quality standard for Minnehaha Creek is violated at this location.

3.1.2 Minnehaha Creek-Knollwood

Monitoring at the Knollwood site occurred between August and October 2009. The defining characteristic of this monitoring period is the lack of rainfall that occurred during the monitoring period. The below average seasonal rainfall, and below average stream flow, drastically impacted the water quality in Minnehaha Creek at this time and in this location. As seen in Figures 3.8 and 3.9 there was no official recorded stream flow in the Creek during a large portion of September. Thus, the pollutant loads are calculated as zero. Chloride loading rates during periods where there was measureable flow were greater than those recorded at the Pamela Park Site. This may be due to the higher amounts of impervious surfaces surrounding this sampling location as well concentration of chloride with minimal water flowing through the system.

The TSS loading rate was highly variable and appears to increase significantly with rainfall (Figure 3.9). The baseline TSS loading rate was similar to that of the Pamela Park site, but the increases in pollutant loading with storm events was much greater. This may also be a result of the lack of stream flow and greater disturbances to the stream bed during storm events.

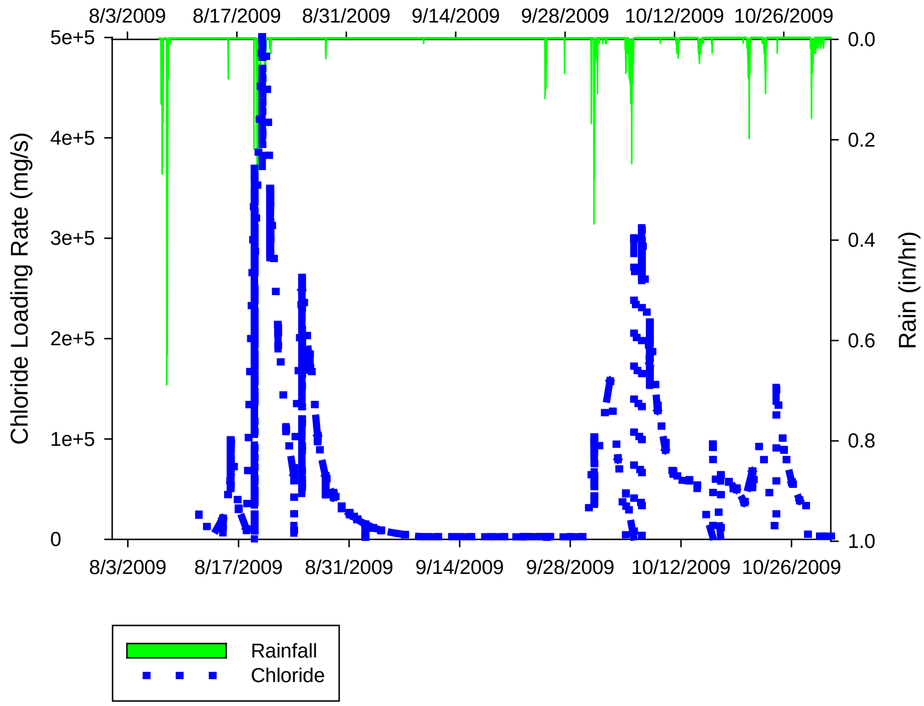


Figure 3.8: Chloride loading rate (mg/s) at the Knollwood site with associated rainfall (in/hr). Parameter concentrations are displayed on the primary vertical axis and rainfall is displayed on the secondary vertical axis. Parameter concentrations are displayed for only the sampling location upstream of the BMP System. The inverted solid areas correspond to rainfall intensity.

Temperature readings at the Knollwood monitoring site are presented in Figure 3.10. Temperature starts to decrease with the arrival of fall. The readings also show diurnal temperature fluctuations as discussed above. pH in Minnehaha Creek at this location was generally lower than the readings at the Pamela Park site, with smaller diurnal fluctuations shown particularly in October (Figure 3.11). Dissolved oxygen was very low in this section of the creek, with individual readings frequently dropping below the daily minimum numerical water quality standard of 4 mg/L of dissolved oxygen (see Figure 3.12). The ecological health of the creek greatly deteriorated as the flow dried up. Finally, specific conductance values reflect an increase in concentration due to the lack of rainfall and stream flow (Figure 3.13) and were routinely above the chronic chloride standard of

230 mg/L based on Equation 1. Maximum turbidity values also correspond with the driest part of the monitoring period as shown in Figure 3.14.

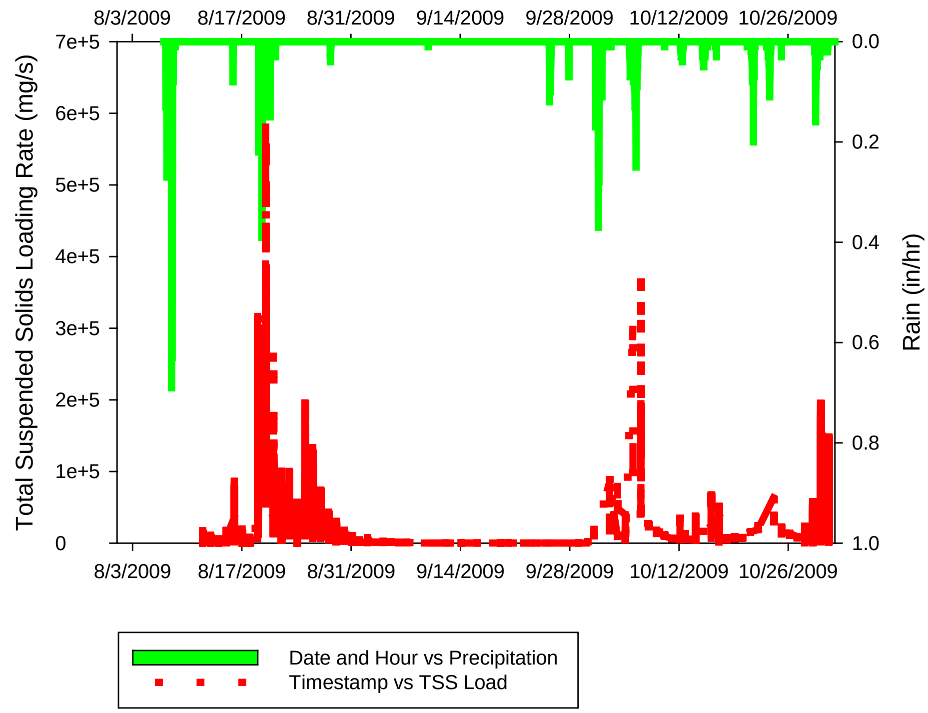


Figure 3.9: Total Suspended Solids loading rate (mg/s) at the Knollwood site with associated rainfall (in/hr). Parameter concentrations are displayed on the primary vertical axis and rainfall is displayed on the secondary vertical axis. Parameter concentrations are displayed for only the sampling location upstream of the BMP System. The inverted solid areas correspond to rainfall intensity.

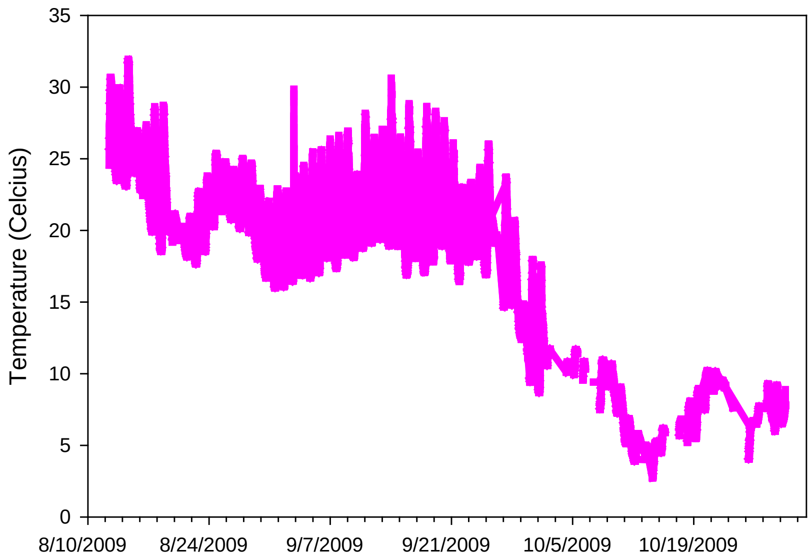


Figure 3.10: Temperature at the Knollwood site for the monitoring period. Parameter readings are displayed on the primary vertical axis.

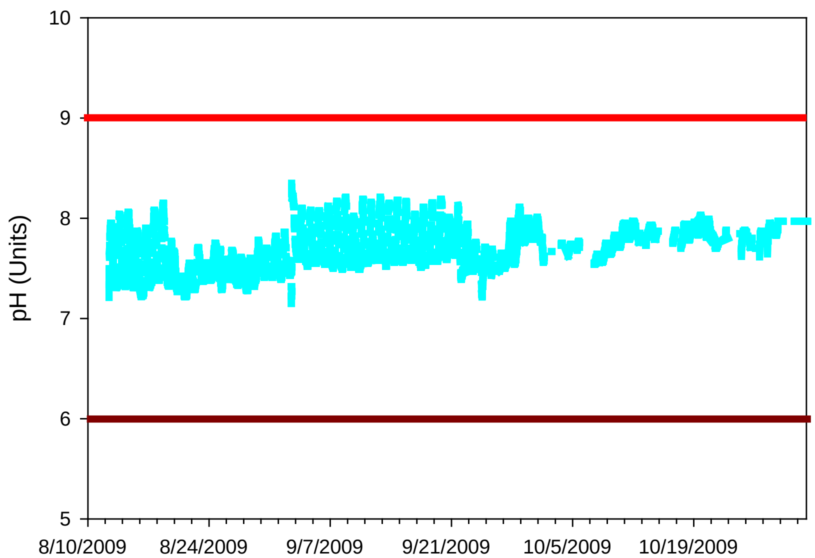


Figure 3.11: pH at the Knollwood site for the monitoring period. The top solid line at pH 9 is the maximum standard for pH and the bottom solid line is the minimum standard for pH at pH 6. Parameter concentrations are displayed on the primary vertical axis.

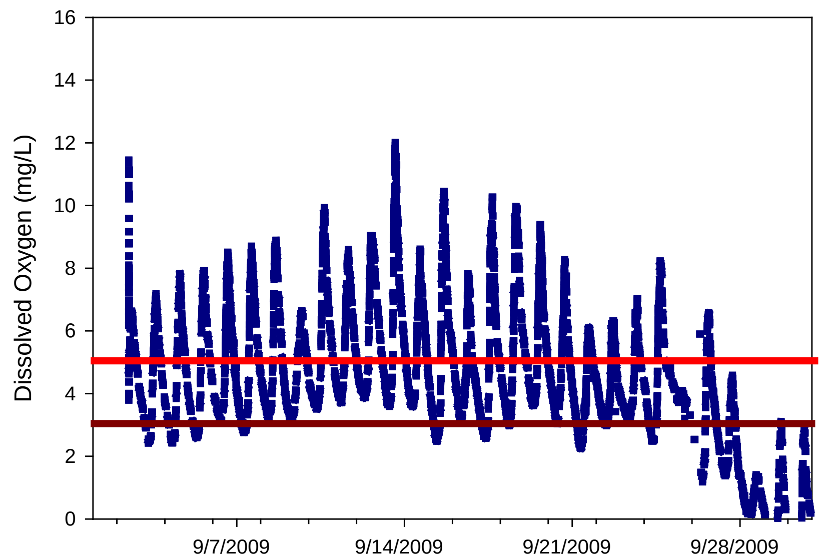


Figure 3.12: Dissolved Oxygen (mg/L) at the Knollwood site for the monitoring period. The top solid line at 5 mg/L is the daily average standard and the bottom solid line is the daily minimum standard at 4 mg/L. Parameter concentrations are displayed on the primary vertical axis.

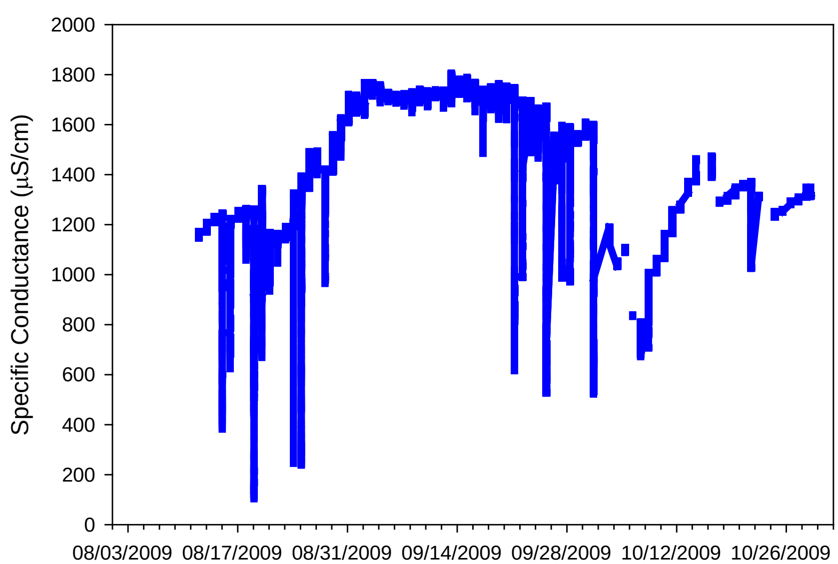


Figure 3.13: Specific conductance (µS/cm) at the Knollwood site for the monitoring period. Parameter concentrations are displayed on the primary vertical axis.

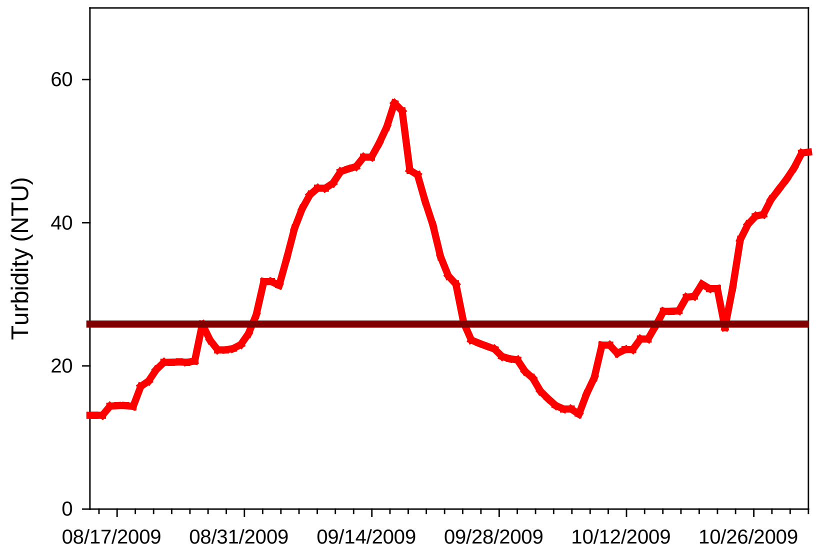


Figure 3.14: Turbidity (NTU) at the Knollwood site for the monitoring period. The horizontal line is the Turbidity water quality standard at 25 NTU. Parameter concentrations are displayed on the primary vertical axis.

3.1.3 Conclusions and Limitations

Pollutant inputs to urban streams are non-steady state in nature. It is also clear that these systems are variable with time and space (see Part II). Calculated in-stream pollutant loads were higher at the site with more impervious surface. A lack of stream-flow at the Knollwood location also greatly increased in stream specific conductance (and thus likely chloride), while decreasing the amounts of dissolved oxygen in the creek. Diurnal fluctuations of temperature, pH and DO were captured in the monitoring data, as well as seasonal fluctuations in monitoring parameters. The high frequency and complexity of the dynamics that impact our measurements of these parameters, makes statistical correlations between environmental dynamics and water quality parameters very difficult. Trends and overall patterns can be identified (see Part II), yet the dynamics of in-stream pollutant loads and concentrations are still very difficult to quantify. A breakdown of daily data collected by the WSN, which is an extensive undertaking given the frequency of data collection, would reveal more detailed patterns and trends than the longer timescales analyzed here.

3.2 Load Durations and Cumulative Distributions to Evaluate Water Quality and the Impact of BMPs on Water Quality

Data collected from April to June 2009 at the Pamela Park monitoring location and from August to October 2009 from the Knollwood monitoring location is shown in Figures 3.15 to 3.26. Based on the upstream, downstream, and in-pond design of the WSN set-up, BMP performance was evaluated using upstream and downstream water quality data comparisons. The data was also evaluated as compared to applicable water quality standards.

The physical design of the BMP systems in place at each monitoring location narrowed the applicable analysis of the BMP system by relevant pollutant and relevant monitoring time. Removal of TSS, nutrients, metals, pathogens, and toxins has been observed in stormwater ponds. The following analysis, therefore, focused on TSS and chloride removal as a function of the BMP's effectiveness, because of the relevance of these pollutants to Minnehaha Creek. Stormwater wetlands are similar in design to stormwater ponds and similar pollutants have been shown to be removed by the two BMPs. Because of the chloride impairment in Minnehaha Creek, it was relevant to determine if there was any effect of the BMP system on in-stream chloride loading. Chloride, as a mobile ion, generally does not change materially with passage through a

BMP. Infiltration, however, accomplishes the most “removal” of chloride from stormwater. This process of infiltration is aided by a wetland but may not be desirable if groundwater salt levels are high.

Conventional water quality parameters such as dissolved oxygen and pH may change within runoff as it passes through a BMP. The most common trend seen during the course of this study was the increase of pH and DO during the day because of photosynthesis and a decrease at night from respiration of plants and algae. During photosynthesis, carbon dioxide is removed from water, which causes a rise in pH. At night, with an increase in respiration there is a release carbon dioxide to the water, causing a decrease in pH. During low flow times, the depression of these trends was observed as a function of water stagnation. pH and dissolved oxygen behavior are not detailed in this section. The criteria used to evaluate BMP effectiveness focused on the effluent concentrations of the treated runoff and the ability of the BMP to provide flow-duration control.

3.2.1 Pamela Park

The calculated mean daily loading rates based on the collected data, as well as the applicable water quality standards for TSS and chloride, are discussed in this section. Both the monitoring data loading rates and the standard loading rates were calculated using the same mean daily stream flow. Consequently the monitoring data, as compared to the water quality standard, indicates if there was an exceedance of the water quality standard. This data also illustrates the frequency with which these exceedances occur and the stream flow profile that corresponds with exceedances.

TSS loading rates more frequently exceed the water quality standard during the sampling period at the upstream and downstream monitoring locations (Figures 3.15, 3.16) than they meet the standard. Loading rates are generally higher for the upstream monitoring location, indicating removal mechanism for particulates between the upstream and downstream sites. During higher flow periods (0%-20% of the flow duration interval) relatively fewer exceedances of the water quality standard occur at the downstream location than at the upstream location under the same flow conditions. During these periods, TSS concentrations at the downstream location were influenced by the BMP system and stormwater inflow from the antecedent snow melt. During low flow periods (80%-100% of the flow duration interval) there was no observed flow from the BMP system to the creek, even during small rainfall events. Therefore, TSS loading rates were independent of TSS removal in the BMP system at that time.

Chloride loading rates at the Pamela Park monitoring site never exceed the acute water quality standard for chloride. As seen in Figures 3.17 and 3.18, chloride loading rates behave similarly at the upstream and downstream sampling locations. The chronic water quality standard is exceeded during higher flow periods (0%-20% of the flow duration interval) which coincide with the snow melt and high chloride concentrations in stormwater due to road salt application. Low flow periods (80%-100% of the flow duration interval) are of more concern for chloride exceedances. Low flow means further concentration of chloride ions in a smaller amount of water, and without flow from the BMP system to the creek, there is no dilution of the chloride in the stream flow. Under unusually dry conditions, the chloride impairment is exacerbated.

BMP performance as a function of flow duration control is illustrated in Figures 3.19-3.22 using cumulative distribution diagrams. An effective BMP system would mitigate the impact of these events on the average loading rate of pollutants in-stream. The average loading rate for the upstream sampling location corresponded to a value greater than the 73rd percentile for TSS and greater than the 66th percentile for chloride. In other words, the mean loading is greater than the median loading. Not only does this result confirm that the daily pollutant load is disproportionately influenced by a small percentage of time intervals, it suggests that random sampling of the stream would, on average, result in an under-estimation of the true pollutant loading rate. Underestimation of pollutant loading rates may lead to false conclusions regarding attainment of basic water quality standards. The average loading rate for the downstream sampling location corresponded to a value greater than the 62nd percentile for TSS and greater than the 58th percentile for chloride. Though the mean loading rate for both pollutants is greater than the median loading rate, the pollutant loading is significantly less affected by high loading events indicating that the BMP system does not have a negative impact on downstream water quality. These results also may indicate the success of the BMP system in pollution mitigation efforts

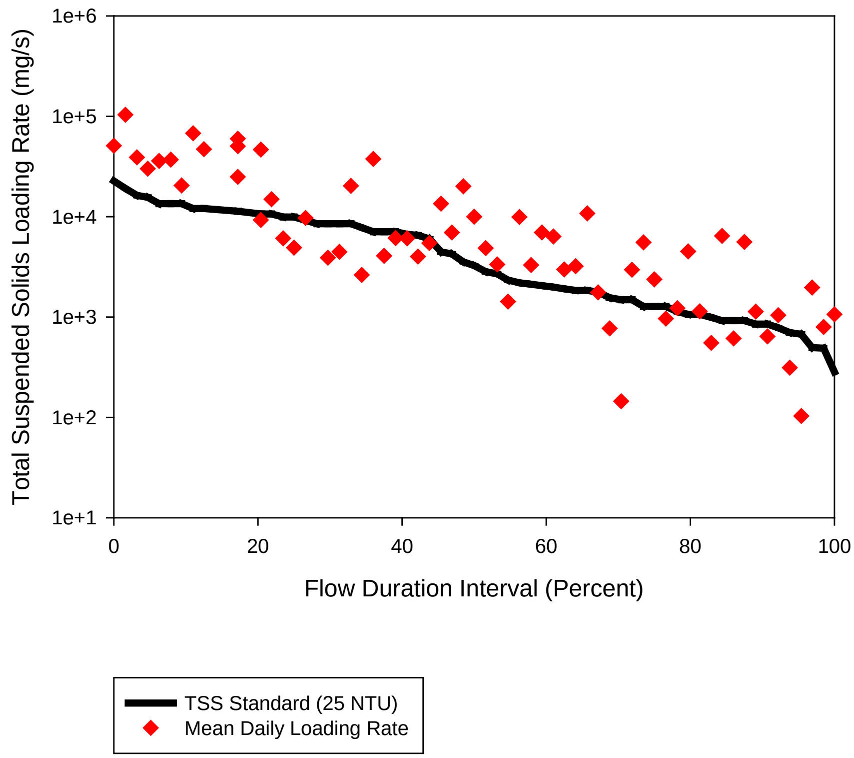


Figure 3.15: Pamela Park upstream sampling location load duration curve for TSS. The TSS standard of 25 NTU (or 25 mg/L as calculated based on Equation 2) was multiplied by the mean daily flow and plotted against the loading rate calculated based on monitoring data collected at the site. Loading rate is plotted on a log scale.

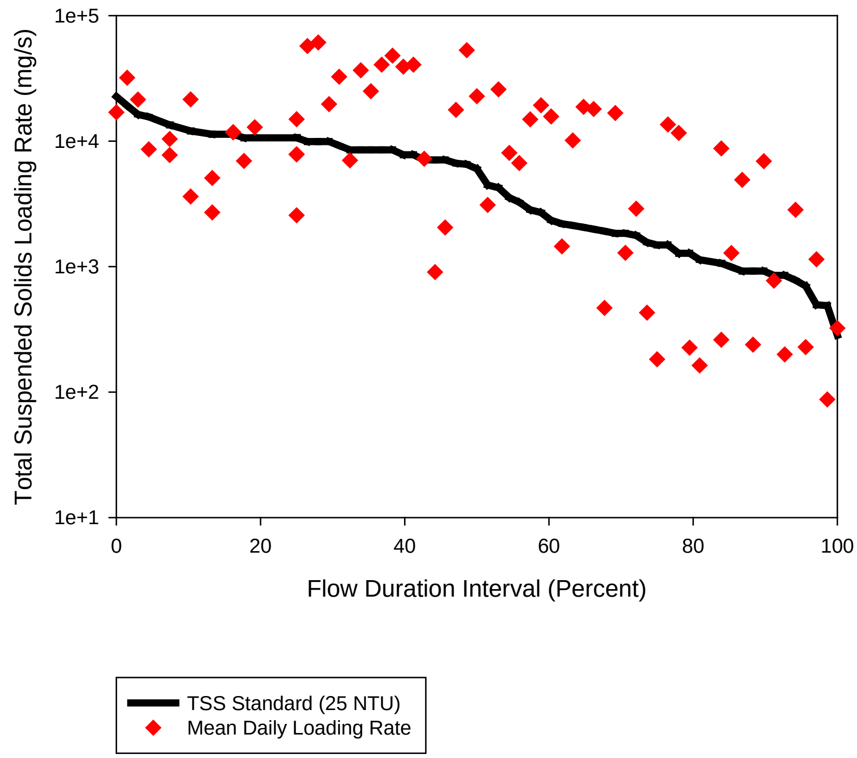


Figure 3.16: Pamela Park downstream sampling location load duration curve for TSS. The TSS standard of 25 NTU (or 25 mg/L as calculated based on Equation 2) was multiplied by the mean daily flow and plotted against the loading rate calculated based on monitoring data collected at the site. Loading rate is plotted on a log scale.

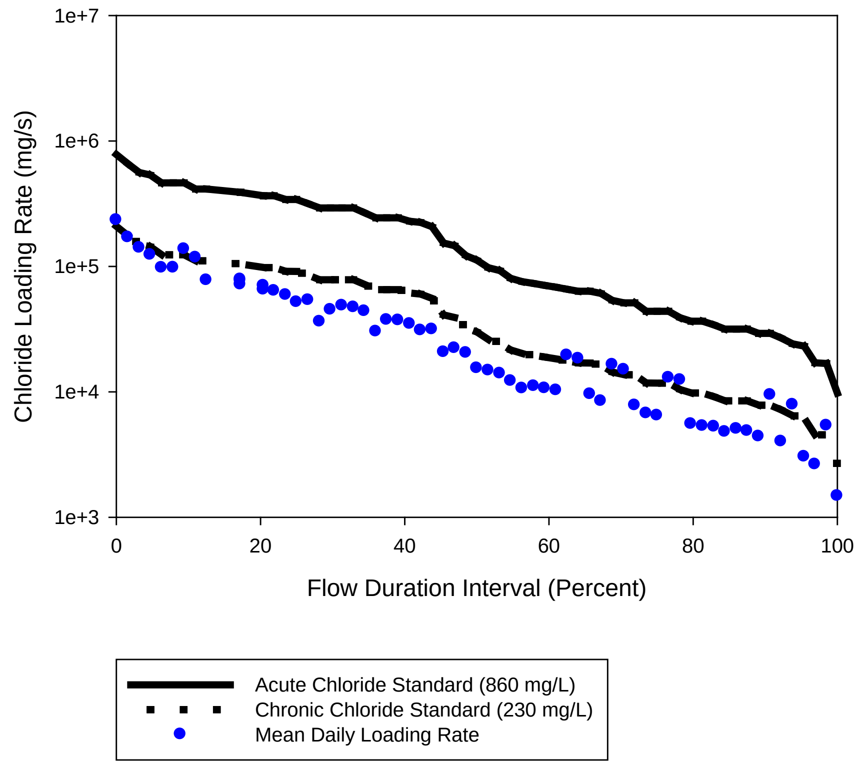


Figure 3.17: Pamela Park upstream sampling location load duration curve for chloride. The chloride standard of 230 mg/L (chronic) and 860 mg/L (acute) was multiplied by the mean daily flow and plotted against the loading rate calculated based on monitoring data collected at the site. Loading rate is plotted on a log scale.

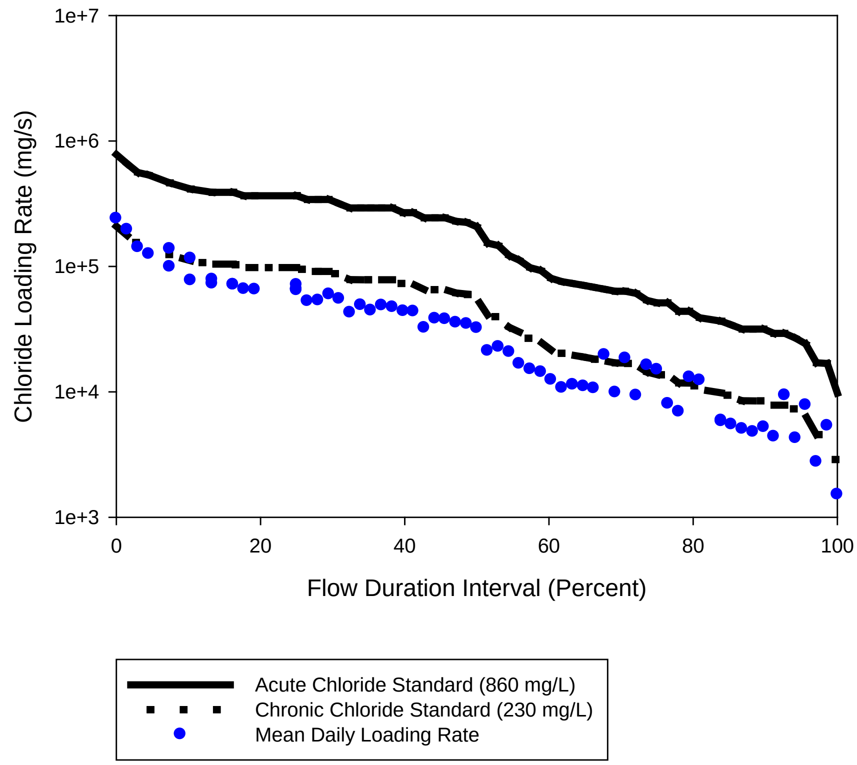


Figure 3.18: Pamela Park downstream sampling location load duration curve for chloride. The chloride standard of 230 mg/L (chronic) and 860 mg/L (acute) was multiplied by the mean daily flow and plotted against the loading rate calculated based on monitoring data collected at the site. Loading rate is plotted on a log scale.

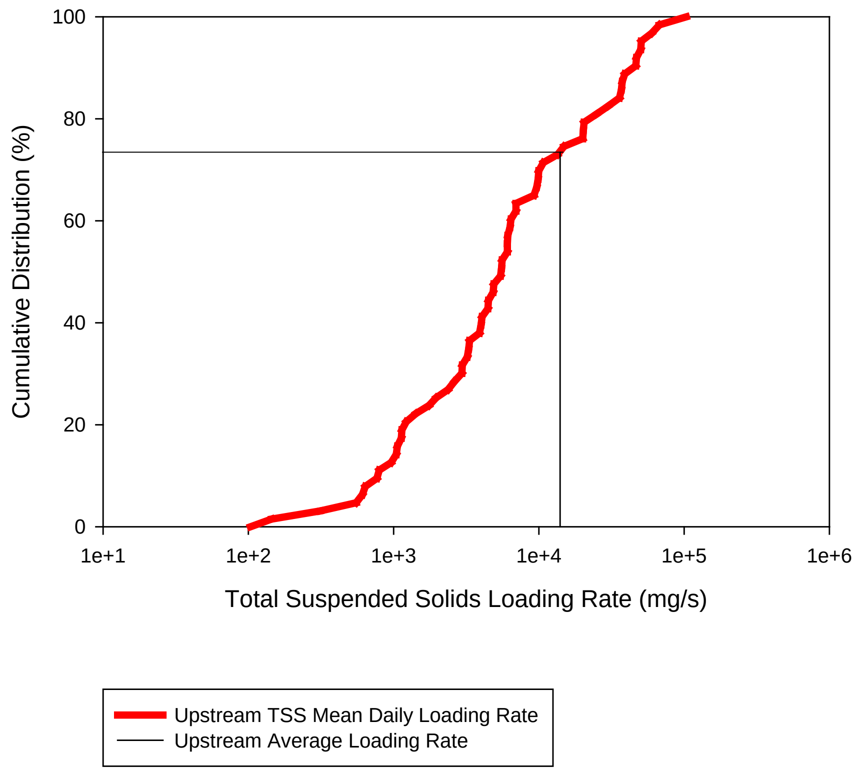


Figure 3.19: Pamela Park upstream sampling location cumulative distribution diagram for TSS. Overall average of the mean daily loading rate as calculated by the monitoring data collected at the site is plotted. The average loading rate corresponds with the 73rd percentile of all loading rates from the sampling location. Loading rate is plotted on a log scale.

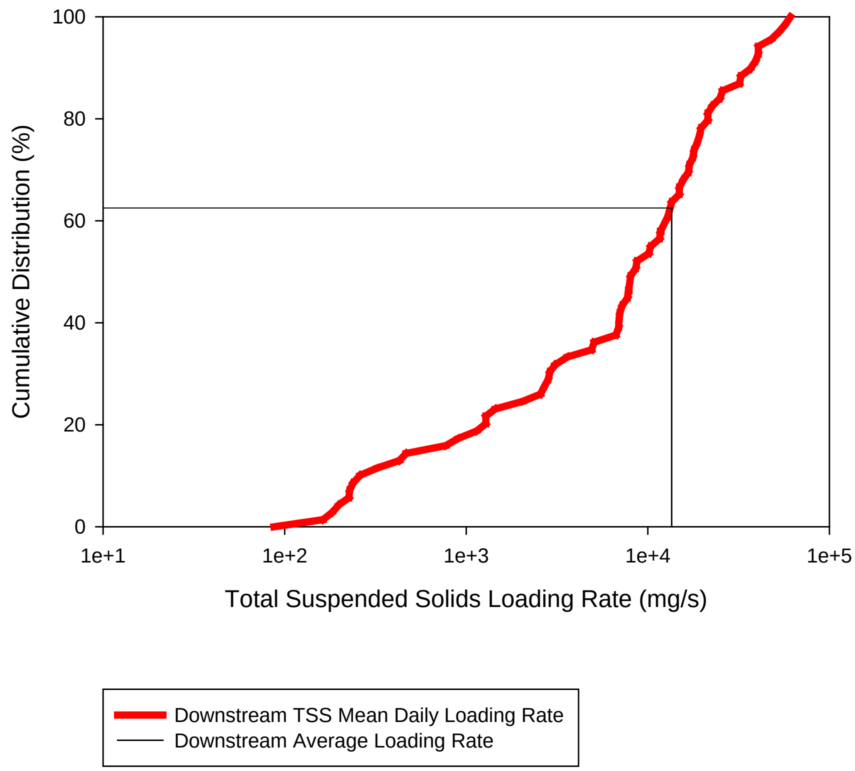


Figure 3.20: Pamela Park downstream sampling location cumulative distribution diagram for TSS. Overall average of the mean daily loading rate as calculated by the monitoring data collected at the site is plotted. The average loading rate corresponds with the 62nd percentile of all loading rates from the sampling location. Loading rate is plotted on a log scale.

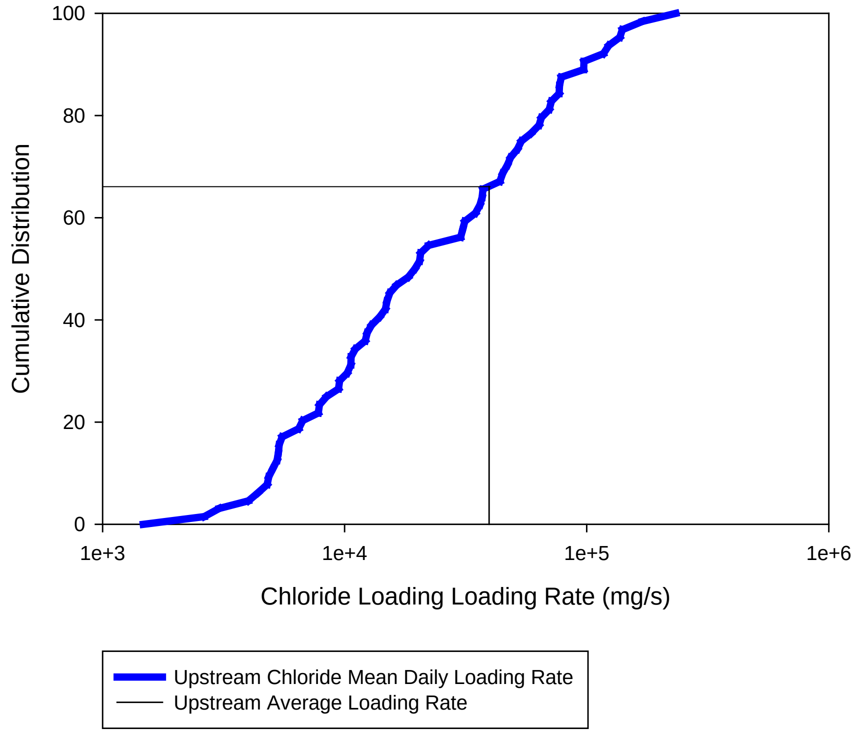


Figure 3.21: Pamela Park upstream sampling location cumulative distribution diagram for chloride. Overall average of the mean daily loading rate as calculated by the monitoring data collected at the site is plotted. The average loading rate corresponds with the 66th percentile of all loading rates from the sampling location. Loading rate is plotted on a log scale.

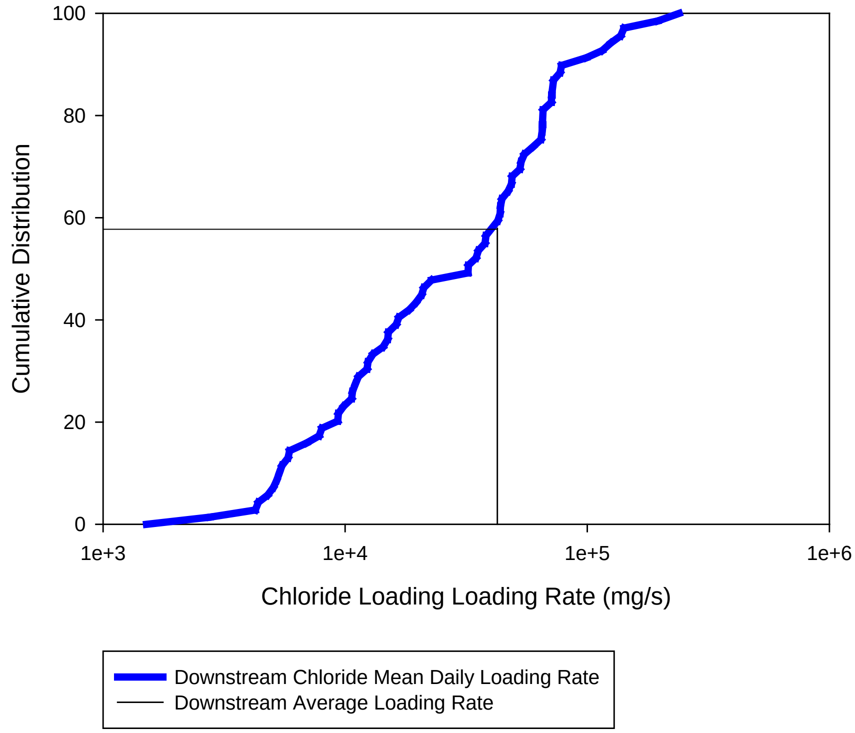


Figure 3.22: Pamela Park downstream sampling location cumulative distribution diagram for chloride. Overall average of the mean daily loading rate as calculated by the monitoring data collected at the site is plotted. The average loading rate corresponds with the 58th percentile of all loading rates from the sampling location. Loading rate is plotted on a log scale.

3.2.2 Knollwood

Figures 3.23 and 3.24 display the calculated mean daily loading rate of the monitoring data as well as the applicable water quality standards for TSS and chloride at the downstream sampling location. The methodology for calculating the loading rates is consistent with the procedure as described for the Pamela Park site previously. Only the downstream data is presented here because it is representative of the site behavior as a whole.

TSS loading rates almost always exceed the water quality standard during the sampling period at this location (Figure 3.23). Loading rates are clustered during low flow periods (80%-100% of the flow duration interval) as there was minimal or no recorded stream flow at the flow monitoring location on Minnehaha creek. On these days stream flow was assumed to be 0.001 m^3/s. During this time period there was also no observed flow from the BMP system to the creek, even during small rainfall events because of the long antecedent dry periods. TSS loading rates, therefore, were independent of TSS removal in the BMP system at that time.

Chloride loading rates at the Knollwood monitoring site never exceed the acute water quality standard for chloride, however they consistently exceed the chronic water quality standard. As stated previously, a complicating factor in determining BMP performance and general creek health at the site was the below average precipitation during the fall of that sampling season. Low flow, again, means further concentration of chloride ions in a smaller amount of water, and without flow from the BMP system to the creek, there is no dilution of the chloride in the stream flow. Under unusually dry conditions, the chloride impairment is exacerbated.

BMP performance as a function of flow duration control is illustrated in Figures 3.25 and 3.26. Flow duration control by a BMP system is realized by the mitigation of impacts from high stream flow events generally corresponding with high pollutant loading. Stormwater flow is necessary for this control to be tested. Due to dry conditions during the sampling period the average loading rate for each pollutant corresponded to a value greater than the 70th percentile within the sampling period for both chloride and TSS. In this case the mean loading is much greater than the median loading. Generally it would be assumed that the pollutant load is disproportionately influenced by a small percentage of time intervals associated with high pollutant concentrations and stormwater impacts. The absence of stormwater during the sampling period, however, makes this highly unlikely. Under the circumstances of the sampling period, it is difficult to determine where exactly the higher stream flow, high pollutant loading events come from.

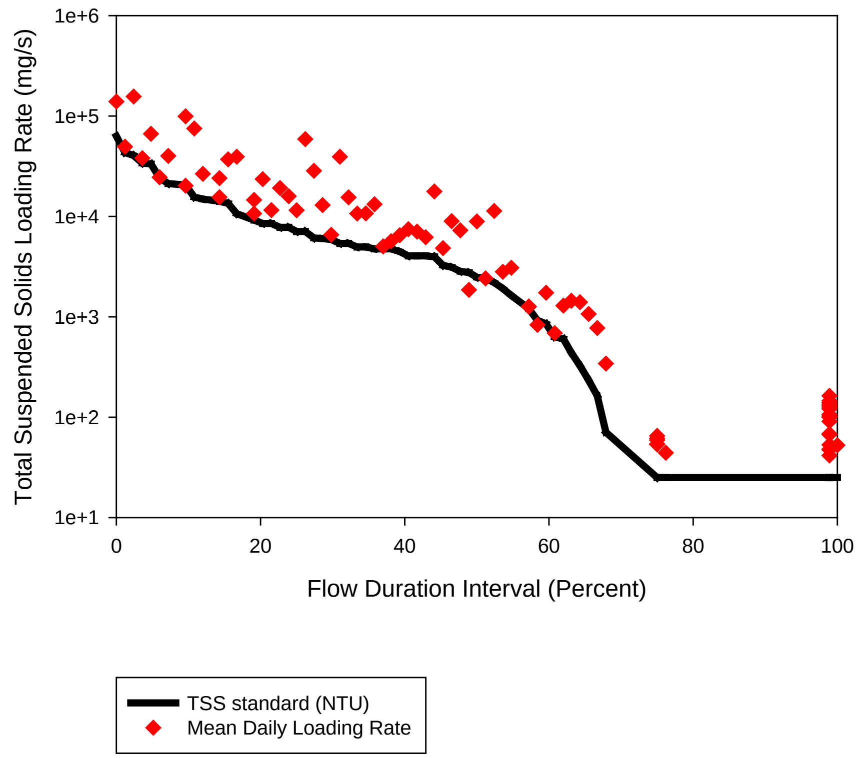


Figure 3.23: Knollwood downstream sampling location load duration curve for TSS. The TSS standard of 25 NTU (or 25 mg/L as calculated based on Equation 2) was multiplied by the mean daily flow and plotted against the loading rate calculated based on monitoring data collected at the site. Loading rate is plotted on a log scale.

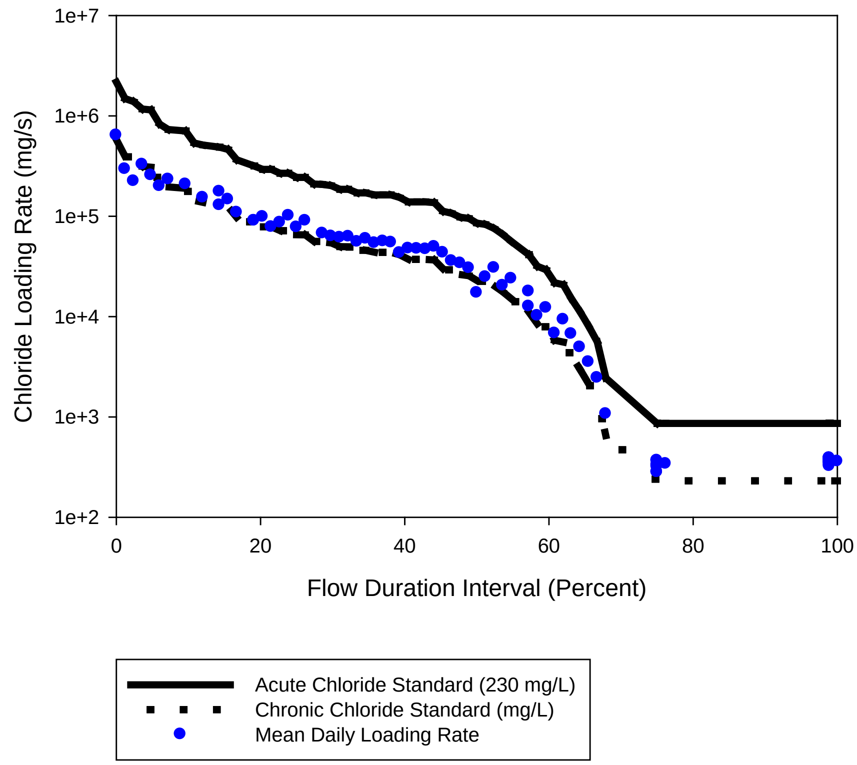


Figure 3.24: Knollwood downstream sampling location load duration curve for chloride. The chloride standard of 230 mg/L (chronic) and 860 mg/L (acute) was multiplied by the mean daily flow and plotted against the loading rate calculated based on monitoring data collected at the site. Loading rate is plotted on a log scale.

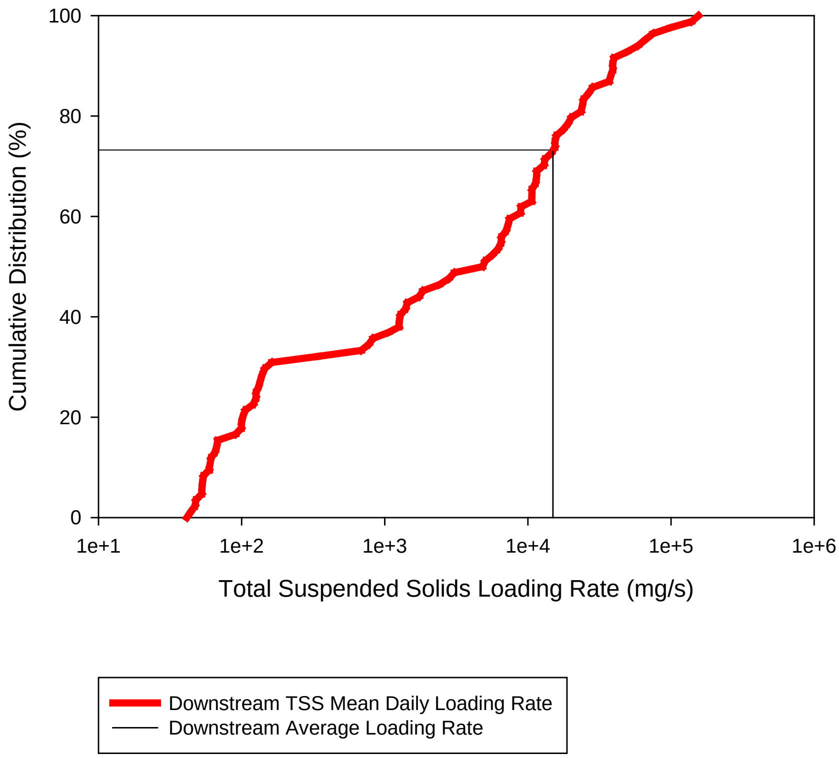


Figure 3.25: Knollwood downstream sampling location cumulative distribution diagram for TSS. Overall average of the mean daily loading rate as calculated by the monitoring data collected at the site is plotted. The average loading rate corresponds with the 73rd percentile of all loading rates from the sampling location. Loading rate is plotted on a log scale.

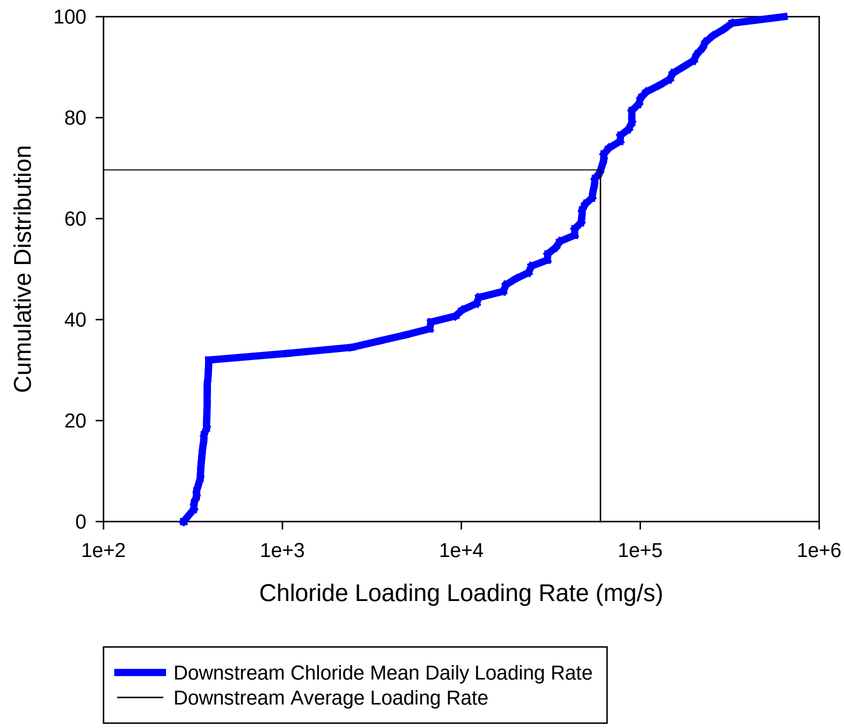


Figure 3.26: Knollwood downstream sampling location cumulative distribution diagram for chloride. Overall average of the mean daily loading rate as calculated by the monitoring data collected at the site is plotted. The average loading rate corresponds with the 70th percentile of all loading rates from the sampling location. Loading rate is plotted on a log scale.

3.2.3 Conclusions and Limitations

At the Pamela Park site, load duration and cumulative distribution diagrams illustrate that high flow, and high load events disproportionately skew the overall water quality picture at the sampling location. Though overall water quality is significantly less affected by high loading events at the downstream sampling location, general water quality data still suggests that the mitigation effects of the BMP system do not prevent the exceedance of water quality standards. The Knollwood sampling location revealed many of the same trends seen at the Pamela Park location; however pollutant load was not influenced by a small percentage of time intervals associated with high pollutant concentrations and stormwater impacts. To further understand the effectiveness of the BMP system at the Knollwood site, significant rainfall events would need to be studied and pollutant loading quantified for those events. Also, due to the lack of instantaneous flow data, the intricacies of changes in loading rates on shorter timescales could not be analyzed as part of this exercise. A more detailed picture of BMP performance at the two sampling locations could be painted with more frequently collected and more location specific flow data. Instantaneous flow data would allow for testing of stormwater volume control by the BMP systems during storm events. Finally, further monitoring of total metals concentrations, in addition to TSS monitoring at the sampling locations, could reveal variations in the correlations as seen in Helmreich et. al. (2010) by land use and season.

3.3 BMP Performance and Pollutant Sources

As stated previously, quantifying pollutant loads at the monitoring locations allows for the evaluation of the health of the water body. Using the WSN also allowed for the quantification of pollutant concentrations/loads entering the sampling location BMP systems, captured by the BMP systems, and discharged to the receiving water. Due to the unusually dry sampling season in the summer and fall of 2009, detailed BMP monitoring data was inconclusive because there was minimal flow of stormwater to the BMP systems and minimal flow from the BMP systems to the receiving body. To collect specific BMP monitoring data, the WSN was deployed at the Knollwood sampling site in the fall of 2010 to capture rain event impacts on the BMP system behavior. Detailed BMP information with analysis of pollutant sources is included in this section.

There is limited evidence documenting the effectiveness of BMPs at removing dissolved and particulate contaminants including nutrients from stormwater runoff (Revitt et al., 2004; Weiss et al., 2006). Pollutant removal efficiencies have been determined for some wet weather events. The use of discrete grab samples to characterize removal performance, especially during rain events, leaves doubt as to the accuracy of removal efficiency determinations. The use of high frequency *in situ* monitoring systems that afford high temporal resolution are beneficial in assessing the performance of stormwater BMPs.

When analyzing BMPs it is convenient to distinguish these systems by a calculation of control efficiency. This method of evaluating BMPs, however, can be misleading because many BMPs do not have defined inlets and outlets. For the purposes of our study, retention and infiltration from wetlands and ponds was the main means of pollutant removal before meeting the receiving body. Thus, simply calculating percent removal is not adequate for assessing BMP effectiveness. Typically BMPs will have an equal or lesser volume of outflow than of inflow on a mass basis. This affects removal, because volume (or flow) is used with concentration to determine mass for a storm event. As detailed in Section 2.3.4 an efficiency ratio and event mean concentrations were used as a method of evaluating BMP effectiveness to mitigate these complicating factors when evaluating BMP performance.

Chloride and turbidity are highlighted in the following section as Minnehaha Creek is impaired for both pollutants. They are of interest with regard to the removal efficiency of these pollutants in the systems that we monitored throughout the course of this study. BMP effectiveness is mainly discussed in the context of the Knollwood monitoring location. Because of minimal stream-flow throughout the course of the study at the Pamela Park site, there was little to no flow within the BMP system (i.e. pond to pond, or pond to wetland) itself or from the BMP system to the receiving water body. Quantification of pollutant removal was not possible other than at the very beginning of the study of Pamela Park.

3.3.1 Pamela Park

The removal of phosphate from the pond system at Pamela Park (Figure 3.27) was quantified until the pond monitored dried up due to weather conditions. Under dry conditions with no flow from storm water conduits to the pond and no flow out of the pond, a decrease in phosphate concentration was seen from the pond inlet to the pond outlet. Depending on pond residence time, it can be assumed that removal of phosphate would occur during storm events as well due to settling.

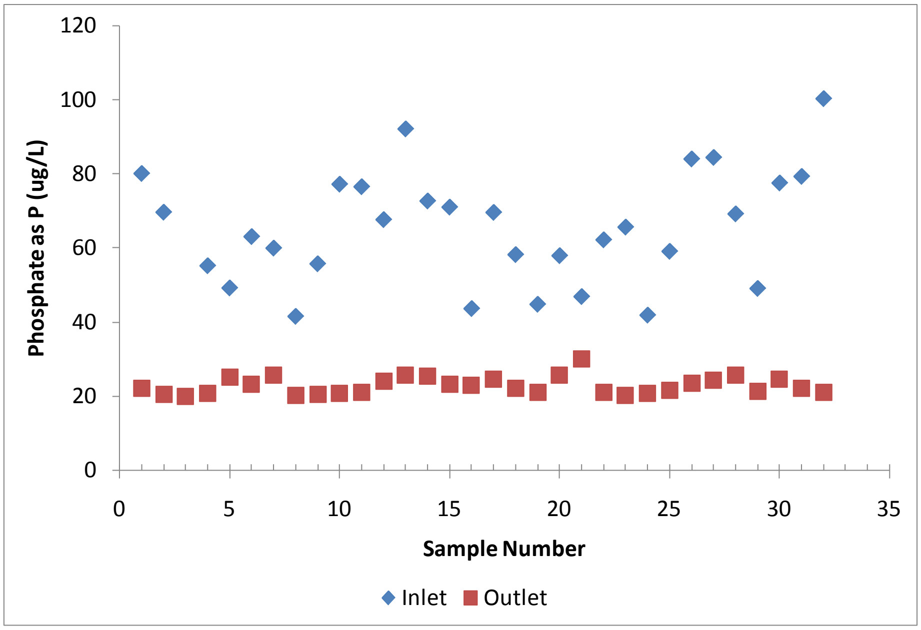


Figure 3.27: Phosphate removal in the pond at Pamela Park. Samples were collected over a period of 70 hours.

The main obstacle to extrapolating conclusions from the Phosphate monitoring data is the nonstationary behavior of the concentration relationships occurring in the pond itself. Especially in smaller catchments, event-scale concentrations are pollutant and catchment-specific. Though there were no rainfall event based findings from our monitoring, Rozemeijer et al. reported repetitive changes in NO_3 and P concentrations in response to rainfall events. The NO_3 concentrations dipped, while the P concentrations peaked during rainfall events. Similar event responses have been observed in other studies as well. The decrease in NO_3 concentrations during rainfall events is related to the dilution of NO_3 -rich stream discharge by NO_3 -poor precipitation water. Peaks in the P concentrations were usually attributed to the flushing of particulate P during rainfall events. During dry periods, the accumulation of particulate P occurs and during rainfall events this particulate P was detached and transported downstream (Rozemeijer et al., 2010).

3.3.2 Knollwood

Figures 3.28 and 3.29 capture the behavior of specific conductance and turbidity respectively for a rain event occurring on Sept 15, 2010. The antecedent dry period documented in the efficiency calculations is also included in these figures. For the period 24 hours prior to the rainfall event, an efficiency ratio was calculated for the BMP performance as reference for the efficiency ratio calculated during the rain event. Specific conductance, as illustrated in Figure 3.28, is directly impacted by rainfall. A decrease in specific conductance is seen with peaks of rainfall in the upstream, downstream and pond monitoring locations. These dips in specific conductance are generally associated with dilution as the quantity of chloride in the water remains the same but the volume of water in the system increases. The upstream and downstream specific conductance readings remain relatively steady when there is no rainfall. Pond specific conductance values steadily increase after the rainfall events. This would seem to indicate an increase in specific conductance associated with stormwater entering the pond system after the rainfall events.

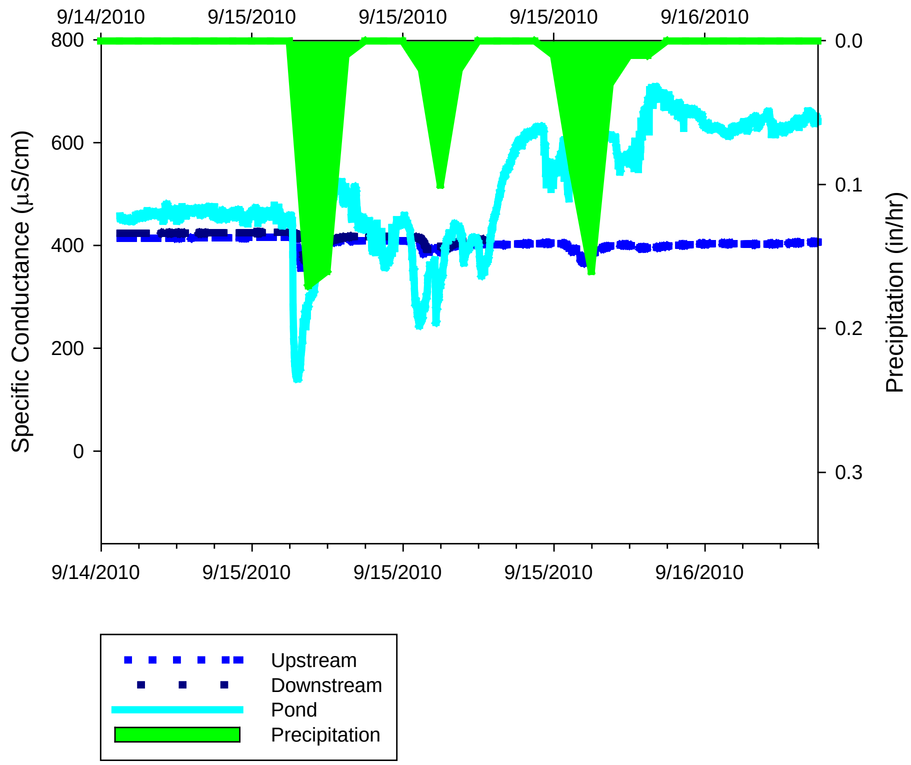


Figure 3.28: Specific conductance data from the upstream, downstream and pond monitoring locations and rainfall (in/hr).

When comparing all sampling locations the upstream and downstream specific conductance readings are very similar and do not increase in the same fashion after rainfall like the pond specific conductance readings. These trends point toward in stream processes controlling in-stream specific conductance as the in downstream sampling location was not impacted by the specific conductance in the pond. The minimal impact of higher specific conductance values at the pond sampling location show containment of higher specific conductance stormwater in the pond or a loss of specific conductance before the stormwater reaches the receiving water body. In either case, the BMP system does not exacerbate the impacts of the stormwater on the receiving water body.

Turbidity behavior varies greatly from specific conductance behavior in the BMP system. Figure 3.29 shows that turbidity in the pond is very high and appears to increase, not only with rainfall, but during the antecedent dry period. Downstream turbidity readings greatly increase with the first flush of stormwater (the first rainfall event) and remains elevated until an equipment malfunction midway through the rain event. The upstream monitoring location appears to experience some lag time in the turbidity response to rainfall. Turbidity increases after rainfall, but the increase is slightly delayed as compared to the instantaneous increase seen in the pond and downstream locations. Turbidity at the downstream location is generally higher than at the upstream location. Because of consistently elevated turbidity readings it is unclear as to whether the stormwater from the BMP system is the major contributing factor to the high turbidity levels at the downstream location. A steady increase in upstream turbidity could also be a function of debris being washed into the receiving water body and upstream disturbances reaching the sampling location.

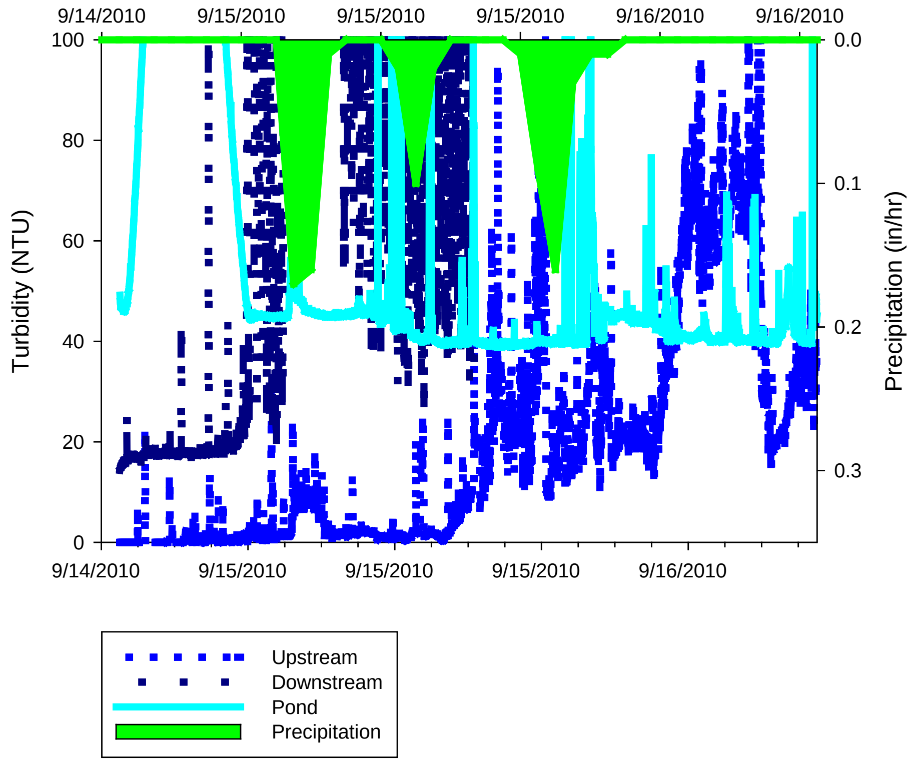


Figure 3.29: Turbidity data from the upstream, downstream and pond monitoring locations and rainfall (in/hr).

The efficiency ratio of the BMP system was calculated for the rain event. In Table 3.2 negative values indicate that the BMP outlet EMC is greater than the inlet EMC, generally describing ineffective removal of the pollutant in question by the BMP system. During both the rain event sampling and the antecedent dry period, the specific conductance efficiency ratio is positive due to a decrease in specific conductance at the outlet of the BMP as compared to the inlet. The turbidity efficiency ratio is negative during the rain event because of an increase in downstream turbidity with the rainfall. Turbidity EMC for both the inlet and outlet are elevated but the average for the outlet is consistently higher over the sampling period during rainfall. This suggests either the

BMP system is contributing to the particulate problem during rain events, or another source is increasing turbidity at the outlet (e.g., release of material from the wetland or in-stream suspension of material). The efficiency ratio for turbidity during the antecedent dry period is positive. Elevated pond turbidity as compared to the downstream sampling location demonstrates little contribution of the BMP to turbidity at the outlet of the BMP system during dry periods.

Table 3.2: Efficiency ratio calculations for the 9/14/2010-9/16/2010 rain event and antecedent dry period (1 day).

Pollutant	Precipitation Event	Dry Event
Specific Conductance	**0.33**	**0.07**
Turbidity	**-1.54**	**.84**

Detailed BMP monitoring also occurred at the Knollwood site on September 2, 2010. Figures 3.30 and 3.31 show specific conductance and turbidity measurements for the upstream, downstream, and pond monitoring locations. As the data sondes were running on battery power during this sampling period, the downstream chloride data is sparse due to the instrument trying to preserve battery power. General trends can still be extrapolated from the data that was collected. The rain event detailed below was also a higher intensity rain event than the one sampled on Sept 15, 2010, and behavior of the BMP system varied based on the rain event characteristics.

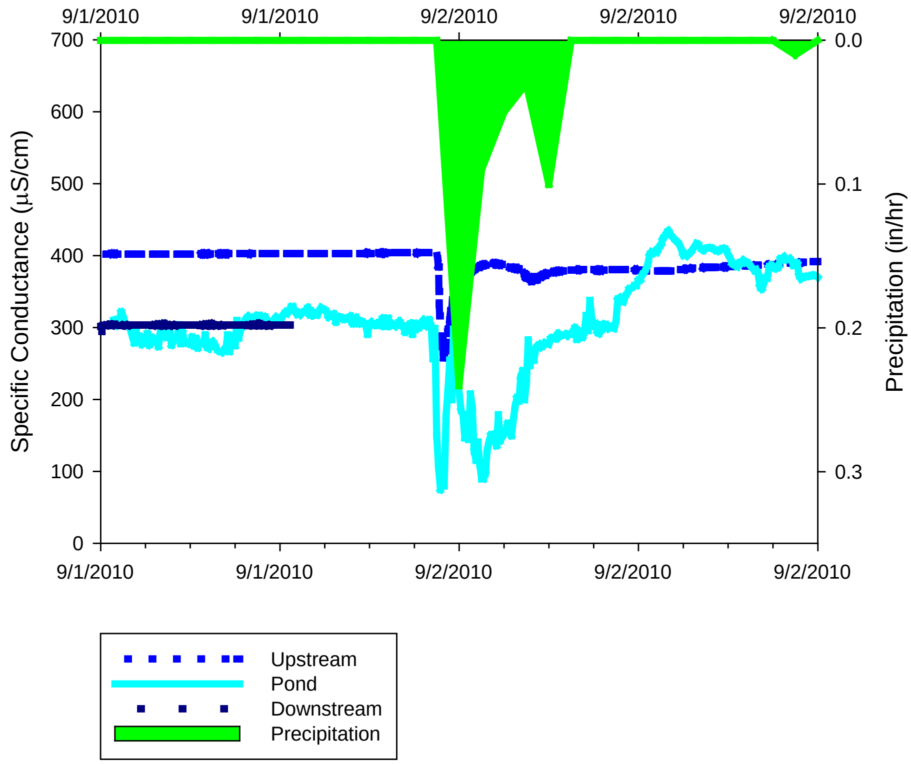


Figure 3.30: Specific conductance data from the upstream, downstream and pond monitoring locations and rainfall (in/hr).

Figure 3.30 shows specific conductance as a function of time. Specific conductance at the upstream sampling location is highest as compared to the downstream and pond locations during the event's antecedent dry period. Specific conductance values at the upstream and downstream locations are very steady at approximately 400 and 300 µS/cm respectively. Pond specific conductance values vary slightly throughout the dry period monitoring. During the rainfall event specific conductance values drop sharply for both the upstream and pond monitoring locations (data not available for the downstream location). A second decrease is observed at the upstream location with the second peak in rainfall. Specific conductance steadily increases in the pond after the first rainfall peak and the slope of the increase in specific conductance is dampened with the second rainfall peak, but, as observed during the September 15, 2010 monitoring event this would seem to indicate an increase in specific conductance associated with stormwater entering the pond system after the first flush of the rainfall event.

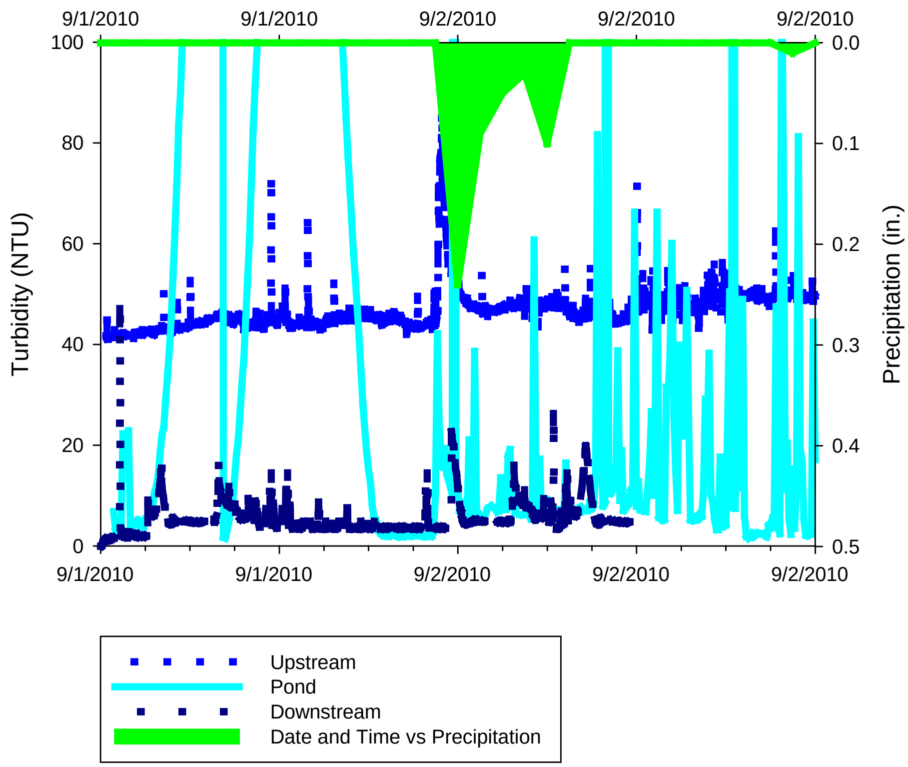


Figure 3.31: Turbidity data from the upstream, downstream and pond monitoring locations and rainfall (in/hr).

Turbidity data for the rain event is detailed in Figure 3.31. Turbidity in the pond varies greatly and appears to increase, not only with rainfall, but during the antecedent dry period. Downstream turbidity readings slightly increase with both rainfall peaks but remain relatively low throughout the duration of the sampling period. The upstream monitoring location turbidity response to rainfall is seen in a dramatic increase in turbidity with the first flush of stormwater and then a generally steadying of readings after the initial increase. Turbidity at the upstream location is higher than at the downstream location. Elevated turbidity levels in the stormwater pond associated with the rainfall appear to have little impact on the downstream sampling location. Turbidity at the downstream location, therefore, is not negatively affected by the BMP system . The BMP system may also have a positive impact on the stormwater entering the system from the BMP and decrease turbidity levels in exiting the outlet of the BMP.

Table 3.3: Efficiency ratio calculations for the 9/14/2010-9/16/2011 rain event and antecedent dry period (1 day).

Pollutant	Precipitation Event	Dry Event
Specific Conductance	**NA**	**-0.10**
Turbidity	**0.71**	**0.65**

The BMP effectiveness at mitigating the effects of turbidity is quantified in the efficiency ratio calculation in Table 3.3. The calculated turbidity efficiency ratio is similar for both the antecedent dry period and the rain event on September 2, 2010. For both scenarios the EMC was greater for the BMP system inlet than the outlet. The efficiency ratio indicates that there no increase in outlet turbidity as compared to the BMP system inlet during this rain. This result differs from what occurred on September 15, 2010. This could be, in part, due to the precipitation profile of the particular events. On September 2, the rain rate was of higher intensity but the duration of the rain event was shorter. Therefore, the initial stormwater inflow had a long residence time in the BMP system. During the September 15th event the total amount of rainfall was greater for the event, and thus it can be assumed that the volume of stormwater runoff routed to the BMP was also greater. This increased volume, increases the amount of stormwater bypassing treatment in the system and directly entering the creek. Elevated turbidity levels at the outlet of the BMP could be a result of shorter residence time in the BMP system.

An efficiency ratio for specific conductance during the rain event was not quantified in Table 3.3 because there was no data from the outlet of the BMP during the rain event. Specific conductance was slightly higher at the outlet of the BMP system than at the inlet, therefore the efficiency ratio for the dry event is slightly negative, indicating no effect of the BMP system on downstream specific conductance during the antecedent dry period.

3.3.3 Conclusions and Limitations

Efficiency ratio using EMC is a concise way of depicting the effect of a BMP system during a rainfall event. More rainfall and more complete rainfall event data is desirable to verify data trends and conditional BMP performance. For example, multiple high intensity/short duration rainfall events should be sampled to determine if a positive ER for the BMP system is observed during all monitored events. During the monitored events it appeared that overall specific conductance is unchanged by a BMP system during rainfall events. Monitored values were either greater at the downstream monitoring location or remained generally the same as the upstream location during monitored events. Turbidity efficiency ratios show mixed results and no definitive conclusions can be extrapolated from the data. The effects of the BMP system on turbidity may be identified and clarified further through further event monitoring and specific quantification of particulate removal as opposed to measuring turbidity.

4. CONCLUSIONS

Next generation environmental monitoring systems that utilize *in situ* sensor technology and transmit high frequency data in near real time are one possible approach for monitoring in-stream water quality, BMP effectiveness, and compliance with water quality standards. However, these next generation monitoring networks present a different challenge: how do managers sort through, in near-real-time, the large volumes of data generated by these systems to identify environmental events (e.g., pollution episodes)? And, is the upfront and maintenance cost worth it for the added benefit?

The situation is even more challenging in the case where easily measured environmental variables (e.g., turbidity and specific conductance) are used as surrogates for more difficult to measure pollutants (e.g., pesticides and pathogens) (Henjum, 2009). In these situations, water quality trends events may not be signaled by changes in the magnitude of the surrogate variable per, but rather by changes in the range of values over which the surrogate variable naturally fluctuates. In this way, a wireless sensor network can provide valuable information to watershed managers and regulatory agencies regarding the health of Minnesota's and the nation's waters. With a significant time and capital investment, these networks can also provide the needed detailed information to determine whether implementation of TMDLs is effective in mitigating pollution. More complete information can also be collected regarding whether BMPs have the intended effects on water quality, or if these practices have compounding environmental impacts that require more attention from environmental managers.

However, with an increasingly more complicated monitoring network, more sophisticated equipment, and added upfront and maintenance cost for a change in monitoring systems, the practicality of a WSN for environmental management is called into question. Running a WSN, even with robust sensors, requires extensive maintenance especially in impaired waters. Environmental and location hazards may deter widespread use of these systems. Sensors and associated network equipment were stolen as part of this project. Sensors were swept away by in-stream debris, and solar power was disrupted in highly vegetated areas of interest. For current environmental managers, the time of implementation alone may just not be available to dedicate to running such a system. However, high spatial and temporal resolution data is needed for improved understanding of impaired waters and BMP performance. The balance between functional technology and immediate watershed management needs creates conflict when determining whether the WSN provides enough benefits to be implemented on a large scale.

The benefits of a WSN are also abundantly clear when looking at the ever evolving needs of environmental managers. Government agencies, state agencies, environmental consultants and even "consumers" of our environment increasingly need more efficient, cost effective, and comprehensive means of obtaining environmental data. Water quality data informs decisions on how to manage our increasingly threatened water resources. Instantaneous data and robust data analysis capabilities only strengthens the ability to make informed environmental management decisions. WSN data collection that allows an environmental manager to spend more time drawing conclusions from data, rather than spending time collecting it, increases efficiency and, most importantly, allows managers to rapidly isolate areas of water quality concern and target these areas using real-time data. TMDLs created for impaired waters also benefit from this data as load allocations and compliance determinations would be based on more detailed and comprehensive data sets.

The future of monitoring Minnesota's water resources can most certainly be transformed by the WSN technology. Our understanding of pollution mitigation measures can also be further informed by the technology. The aforementioned difficulties with a WSN implementation and the obvious benefits to the monitoring technology will, hopefully, promote further discussion by environmental managers and researchers as to the role of the WSN in future water quality monitoring.

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