ECOLOGICAL IMPLICATIONS FOR SUSTAINABLE STORMWATER SYSTEMS IN THE TALLGRASS PRAIRIE REGION

by

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Abstract

Urban stormwater is one of the leading causes of water quality impairment and stream channel degradation in the United States. In an effort to address the negative effects of stormwater runoff on receiving aquatic systems, Best Management Practices for stormwater, including ecologically-designed stormwater systems, are becoming more common across the urban landscape. Throughout eastern Kansas and the rest of the Midwestern United States, prairie grasses are beginning to receive attention for their potential to enhance infiltration within these systems. However, the function of vegetated stormwater systems and the influence of factors such as vegetation age on infiltration and system performance are not well understood because monitoring data for these systems is limited. When performance data is collected, it often pertains only to the hydraulic and water quality aspects of the system but neglects any assessment of the integrity of the ecosystem functions on which the system's performance is dependent. The objective of this study was to address the need for an assessment tool that considers the ecological integrity, or health, of ecologically-designed stormwater systems, as well as to fill the gap in the literature regarding the function of ecologically-designed stormwater systems in the tallgrass prairie region. Since many of the eco-based stormwater practices in the region rely upon the establishment of native prairie grasses to enhance infiltration on the site, the specific focus of this study was to gain a better understanding of infiltration processes in ecologically-designed systems and the extent of our ability to regain these processes through prairie restoration in previously disturbed urban sites. To address these objectives, two stormwater systems at different stages of vegetative maturity were examined. In general, ecosystem health scores were higher for the more mature system and could be used to guide future management decisions at both sites. Results from the hydraulic analysis indicate the function of the system may improve over the course of the growing season, but statistical relationships between system age and infiltration rate could not be established.

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CHAPTER 1 - Introduction

Urbanization is one of the most rapidly growing forms of land use change (Paul and Meyer, 2001). Although the total amount of land occupied by urban areas remains small, the ecological footprint of urban land uses is disproportionately large. Among the most significant changes associated with urbanization are increases in impervious surface cover and the efficiency with which water is transmitted from the surface to the receiving water body (Booth and Jackson, 1997; Dunne and Leopold, 1978). As a result of these changes, infiltration is restricted while the volume and flow rate of surface runoff generated by precipitation increases. In addition to altering the predevelopment hydrologic regime, urbanization also impacts the quality of runoff flows. Stormwater picks up contaminants such as sediment, petroleum products, heavy metals, and excess nutrients as it flows over the urban landscape (Paul and Meyer, 2001).

Since the ultimate destination of stormwater flows is a stream, lake, or other water body, the impacts of urban runoff on water quality are of special concern. The connection between urbanizing watersheds and the degradation of downstream water bodies is well-established in the literature (Booth and Jackson, 1997; Dunne and Leopold, 1978; McCrea, 1997). In the United States alone, nearly 81,000 miles of streams and rivers have been impaired by urbanization (Paul and Meyer, 2001). Stream channels in urbanizing areas are degraded as they become a means of conveyance for urban stormwater flows. Often, in the interests of reducing flooding in urban areas, channels are intentionally widened, straightened, or lined with concrete to increase the efficiency with which they transport floodwaters away from the urban populace (Booth and Jackson, 1997). Small ephemeral streams are often either graded over or placed in a pipe during development so that many of the headwater channels that would otherwise play a role in attenuating flood peaks and providing channel storage are destroyed (Dunne and Leopold, 1978). The level of development at which the degradation of aquatic systems is readily observable is quite low; Booth and Jackson (1997) reported significant changes in stream channel morphology and water quality after the effective impervious surface area of the watershed was increased by only 10% in urbanizing watersheds in western Washington State.

The degradation of aquatic systems in urbanized watersheds, even at low levels of development, has driven the evolution of best management practices (BMPs) to mitigate the impacts of urban stormwater runoff on receiving waters. Of the suite of BMPs that have been developed to manage urban stormwater, most can be grouped into one of two categories: storage-based and infiltration-based. The following sections discuss detention, one of the most widely used storage-based mitigation method, and its shortcomings, followed by a review of infiltration-based mitigation methods and their potential to improve stormwater management efforts.

Detention for Stormwater Management

Detention basins are perhaps the most common storage-based stormwater practice (Perez-Pedini et al, 2005). Detention basins are designed to reduce peak flow rates by temporarily storing a design volume of stormwater and then releasing it at a slower rate. In doing so, detention basins aim to more closely maintain the predevelopment timing and peak rate of flows and, in effect, reduce both flooding and the impacts of high runoff rates on receiving water systems (WEF and ASCE, 1998).

Detention basins came into use at about the same time as the Clean Water Act of 1972 (WEF and ASCE, 1998). By this time, the undesirable effects of urbanization on receiving waters had been recognized, and stormwater detention basins arose as the hopeful champion of stormwater mitigation. Initially, stormwater detention basins were designed to control peak flow rates from storms with a 10- to 100-year return frequency (WEF and ASCE, 1998). However, studies of fluvial systems have shown that channel formation and maintenance is most strongly influenced by smaller, more frequent flows. In most natural systems, the threshold channel-forming flow, also called the bankfull flow, corresponds to the stream discharge that occurs, on average, once every 1.5 years (Dunne and Leopold, 1978). Therefore, early detention basins provided little protection against channel erosion from these more frequent flows. In the particularly environmentally-sensitive Chesapeake Bay region, the need to control peak rates from smaller storms to protect water quality prompted the state of Maryland to require detention to limit the peak flow rate from the 2-year storm to predevelopment conditions in the late 1970's (WEF and ASCE, 1998). Since that time, detention facilities have become the most common engineering approach taken to control the impacts of urban runoff (Perez-Pedini, 2005) and are often implemented as large, end-of-the-pipe facilities to manage runoff flows from the upstream development (USEPA, 2000). The use of detention basins to control the peak runoff rate from the 2-year storm has become an acceptable mitigation practice across most of the United States (WEF and ASCE, 1998).

Limits of Detention

Although detention basins are designed to maintain post-development peak flows to the 2-year return frequency discharge for pre-development conditions, studies have found that stream channel degradation persists (Beyerlein, 2005; Booth and Jackson, 1997; MacRae, 1997). The inability of detention to adequately mitigate the impacts of urban runoff stem from the simplified design approach taken to size detention structures (Beyerlein, 2005; Wulliman and Urbonas, 2005). Detention basins are typically designed using a single design storm as the criterion for storage volume and peak discharge requirements (MacRea, 1997). While such an approach simplifies design equations and calculations, it does not address the duration or frequency of flows. As noted previously, as the percentage of impervious surface cover in the watershed increases, so does the volume of surface runoff produced. Therefore, even when the peak flow rate is reduced to predevelopment levels, the larger runoff volume prolongs the duration of higher flow rates. Because detention basins trap sediment, the effect of extended flow duration from these facilities may be exacerbated as the flows leaving the basin are "sediment-starved," and thereby remove sediment from the within the channel to reach equilibrium between flow rate and sediment load. As a result, stormwater discharged from detention basins may accelerate channel degradation as it conducts geomorphic work on stream channels over a longer period of time (Beyerlein, 2005). In addition, reducing flows to the standard 2-year peak flow rate may not be adequate. The frequency of bankfull discharge has been found to decrease from a 1.5-year return frequency to a return frequency closer to the 1-year return frequency as urban development progresses (MacRae, 1997). Thus, it is the range of flows below the 2-year design criterion used in sizing detention basins that typically do the most erosional work in urban streams. A detention design which does not address this shift in bankfull discharge cannot be expected to adequately reduce stream degradation (MacRae, 1997).

The intended function of detention basins is further limited by the effect of multiple detention basins. Wulliman and Urbonas (2005) have found that while it may be possible to control peak rates directly downstream of a single detention basin, the flows released from individual basins have an additive effect in receiving streams served by multiple detention systems. As a result, peak flow rates in receiving waters are often not reduced to predevelopment levels.

Infiltration-based Strategies for Stormwater Management

The shortcomings of detention-based mitigation strategies indicate that simply controlling peak flow rates will not satisfactorily reduce the impact of stormwater flows on streams in urbanizing watersheds. Rather, volume control is also needed to maintain the duration of stormwater flows to predevelopment conditions (Wullimon and Urbonas, 2005). The need for volume control is slowly initiating a shift in the present stormwater management paradigm from conveying, storing, and discharging from a detention structure to one of collecting and infiltrating stormwater on the site in order to reduce both peak flows and volume. This new approach to stormwater management is part of the Low Impact Development (LID) site design strategy developed in Prince George's County, Maryland in the early 1990's. LID practices include flatter grades in developments, open grassed swales for stormwater conveyance, depression storage, and functional bioretention landscaping, all of which are aimed at preventing stormwater from running off to encourage infiltration on the site (USEPA, 2000). The use of LID practices to reduce runoff volume through infiltration is supported by infiltration studies found in the literature; studies citing infiltration rates in soils report that many undisturbed soils are able to completely infiltrate low intensity rainfall (Gregory et al., 2006; Pitt et al., 1999). An additional advantage of LID and infiltration-based stormwater management practices over conventional drainage systems and regional detention basins is that they typically offer greater water quality improvements. Because LID practices aim to infiltrate stormwater onsite rather than allow it to run off, the physical, chemical, and biological processes that remove contaminants from stormwater are able to take place (Rushton, 2001). Oftentimes, LID practices can be more easily and cost-effectively integrated into the existing stormwater infrastructure as compared to more conventional structural stormwater management practices (USEPA, 2000; WEF and ASCE, 1998).

As mentioned above, LID and infiltration-based practices rely largely on naturally-occurring ecosystem processes- including infiltration, sedimentation, pollutant sorption to soils, microbial degradation of organics, nutrient cycling, and evapotranspiration- in order to successfully manage runoff quantity and quality. In the interest of enhancing these processes and ensuring their long-term sustainability, vegetation is a key component in infiltration-based stormwater practices. The role of vegetation in reducing surface runoff is well known. Vegetation can enhance infiltration rates by increasing macropore flow within the soil (Perrygo et al, 2001). Between storm events, vegetation drives evapotranspiration rates so that the moisture stored in the soil is returned to the atmosphere and the storage capacity of the soil profile to infiltrate the next storm is increased (Dunne and Leopold, 1978). When the magnitude of the storm exceeds the landscape's capacity to infiltrate precipitation, the roughness and irregularity of vegetated surfaces reduces runoff velocity, thus delaying flood peaks to receiving channels (Gregory et al, 2006). As runoff velocity is slowed, vegetation helps improve runoff quality by promoting sedimentation. Plants also play a direct role in nutrient cycling, removing excess nitrogen and phosphorus from stormwater after it has infiltrated into the soil profile through biological uptake (USEPA, 2000). Indirect water quality improvements are facilitated in the region of soil surrounding the plants' roots, which is rich in sugars and other root exudates which support the bacterial populations that play a role in nutrient cycling and pollutant degradation (Bradshaw, 1997).

As a consequence of the reliance of LID performance upon naturally-occurring ecosystem processes, the specific set of environmental conditions- including climate, soil type, and rainfall patterns- native to the area in which the practice is being implemented will determine design factors such as vegetation type and soil media (USEPA, 2000). Therefore, the same vegetation types and soil media used in Maryland, where LID practices were pioneered, may not transfer directly to other regions of the country with different climatic regimes. Herein lies one of the challenges to successfully implementing ecologically-based systems for stormwater management: designing systems that are ecologically appropriate for the climatic and environmental setting in which they are located.

Another challenge to the successful design and implementation of eco-based stormwater systems is the long term monitoring and maintenance of these systems. Despite their growing popularity and use, there is relatively little data available to assess the performance of eco-based stormwater systems. Vegetated systems require more time than conventional systems to establish; the root systems of some prairie plants continue to develop over a period of three or more years, meaning that the performance of the system will also continue to develop over this time (Weaver and Zinc, 1946). Since many of these systems have been installed within the past few years, the time required for extensive monitoring to evaluate long-term performance has not been available. Monitoring data from infiltration-based systems in Kansas and the Great Plains region is nearly absent from the literature even as these systems are being designed and implemented more frequently in the area. Many of the bioretention and other ecologically-based stormwater systems are managed by city municipalities that have come under the mandate of Non-point Pollution Discharge Elimination System (NPDES) permits which regulate the quality of urban runoff entering natural water bodies. After installation, funding for long-term monitoring is typically not available. Of the systems that have been monitored, the primary focus has been on their performance in terms of water quantity and quality. While both are certainly important metrics in assessing the performance of ecologically-designed stormwater systems, they do not consider the ecological component of these systems, which, as established earlier, is closely linked to the overall function of the system and has important implications for their long-term maintenance.

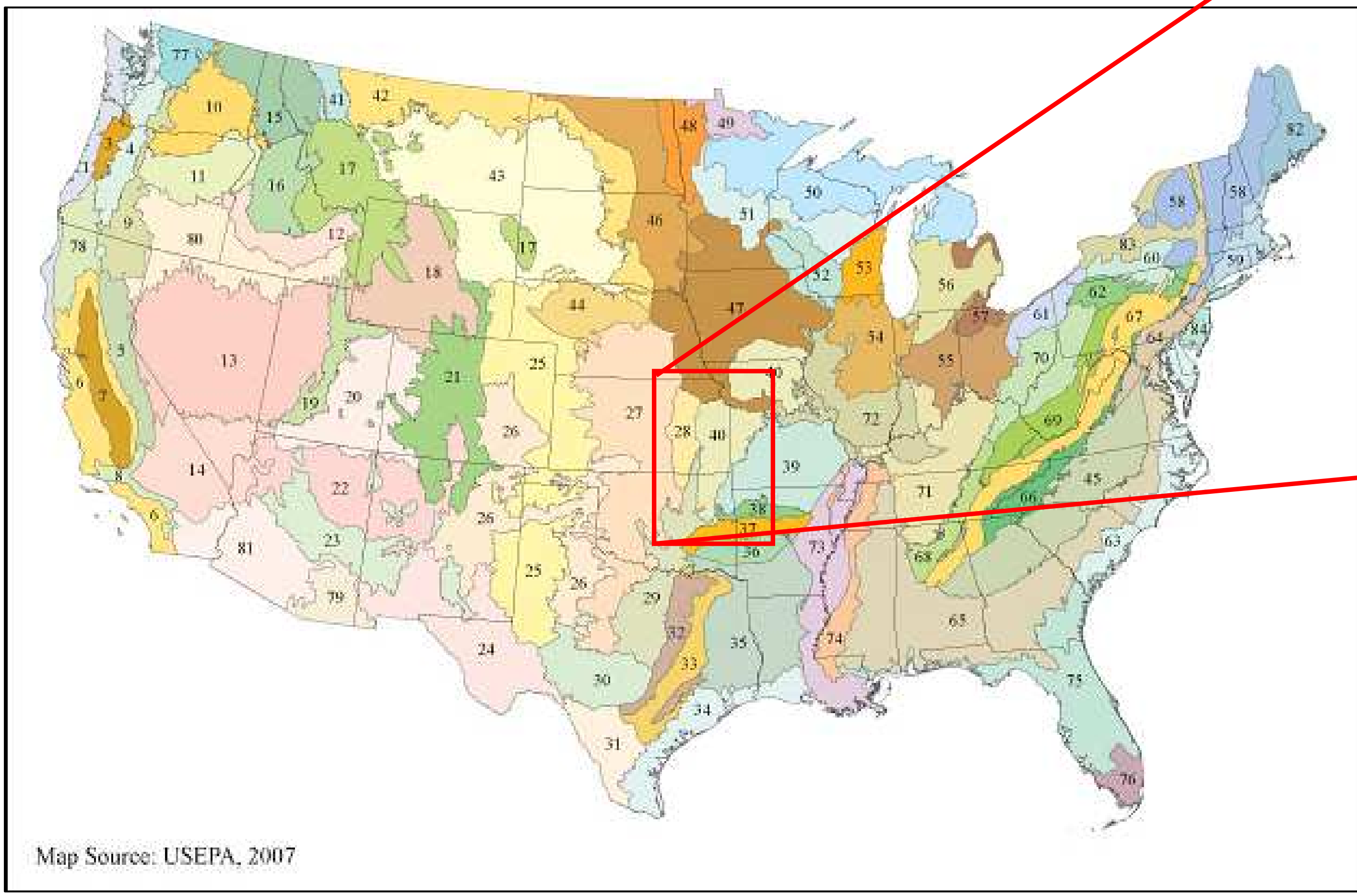
Objectives

The objective of this study was to address two of the needs presented by ecologically-designed stormwater systems. First was the need to develop an assessment tool that considers the ecological components of these systems. In addition to providing an assessment of the ecological aspects of the stormwater system, this tool should be one that could easily be applied by municipal stormwater managers with minimal time and financial input while providing meaningful contributions to the long-term maintenance decisions for these systems. The second need addressed by this study was to fill the gap in the literature regarding ecologically-based stormwater systems designed after the tallgrass prairie ecosystem. As noted previously, the design of eco-based stormwater systems must be appropriate to the ecoregion, which is an area defined by specific climatic and environmental conditions, in which they are located. The sites chosen for this study were both located in Northeast Kansas in the region defined as the Osage Cuestas Ecoregion by the EPA level III ecoregion classification system (Figure 1.1). Although this ecoregion classification is based on the specific vegetation types, soils, landforms, and climatic patterns of the region in which the study sites were located, it is believed that the findings of this study are applicable to the rest of the historic extent of the tallgrass prairie, shown in Figure 1.2, which also shares similar vegetation types and climate patterns

Since many of the eco-based stormwater practices in the region rely upon the establishment of native prairie grasses to enhance infiltration on the site, the specific focus of this study was to gain a better understanding of infiltration process in ecologically-designed systems and of the extent of our ability to regain these processes through prairie restoration in previously disturbed urban sites. This need was examined by combining field-measured values of saturated infiltration rates with inlet and outlet flow rates and volumes collected at an ecologically-based stormwater system designed specifically after the tallgrass ecosystem in Northeastern Kansas.

The remainder of this thesis is organized into two chapters that address the objectives of this study. The second chapter describes the development of an ecological health assessment tool for application to ecologically-designed stormwater systems. The third chapter examines infiltration and hydraulic function in a stormwater basin restored with native prairie grasses in northeastern Kansas.

Figure 1.1. EPA Level III Ecoregion mapping for the continental United States. Inset displays Osage Cuestas Ecoregion in which study sites were located. Image from USEPA, 2007.



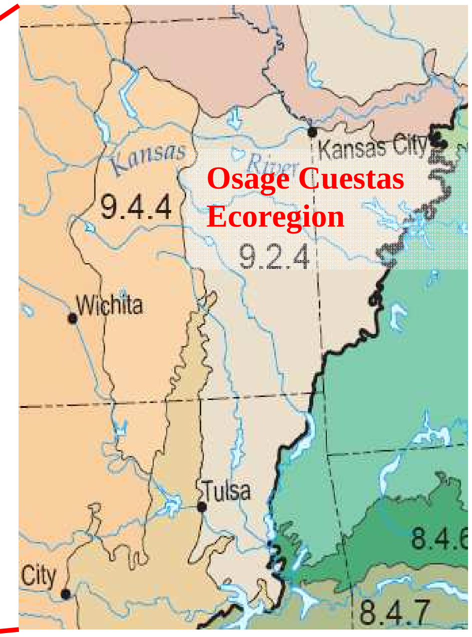
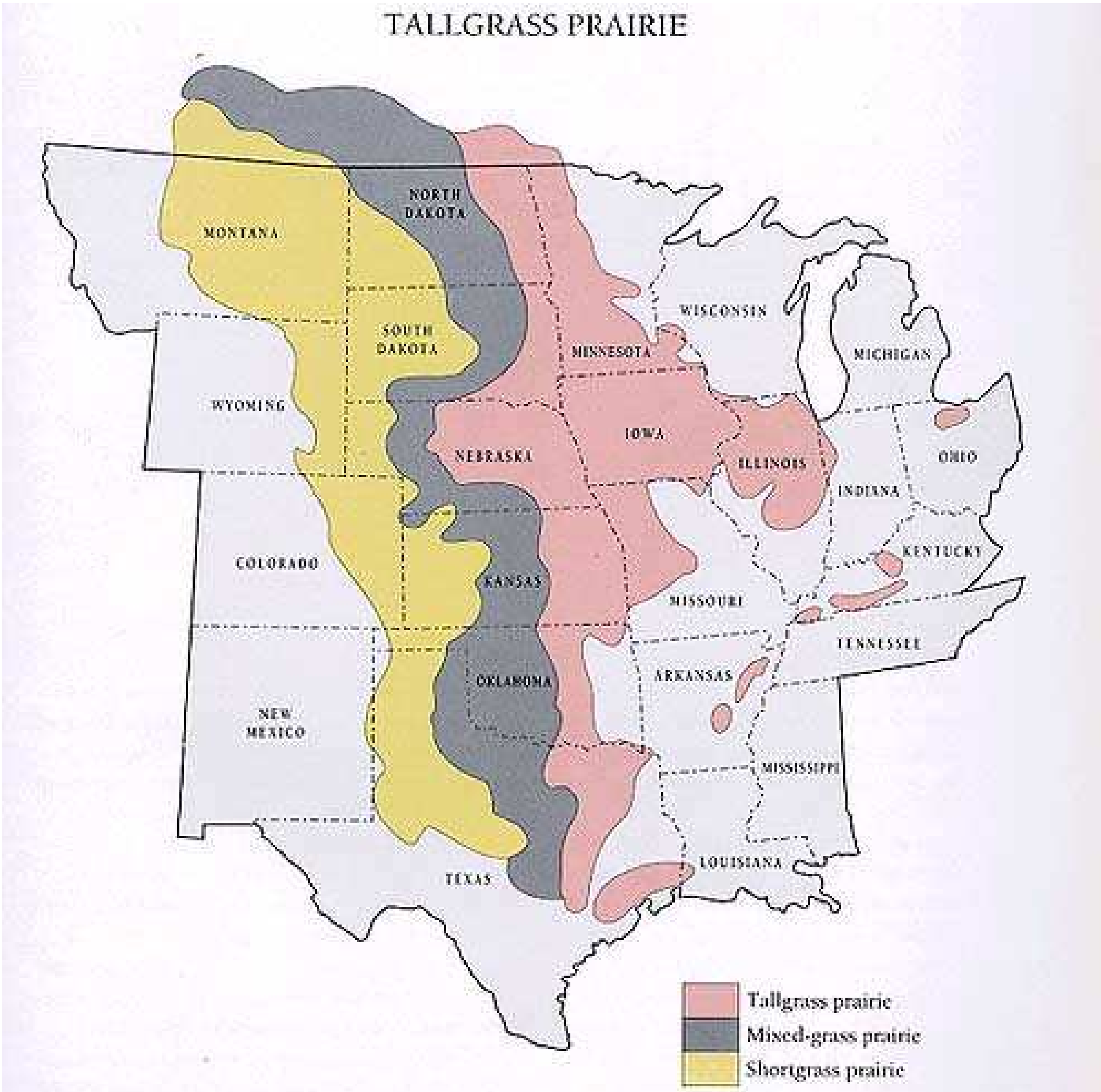


Figure 1.2. Map of the original extent of the tallgrass prairie, the potential area to which this research is applicable. The extent of the mixed-grassand shortgrass prairies is also shown. Image from www.earlparkindiana.com/



CHAPTER 2 - Development of an Ecological Health Assessment Tool for Ecologically-based Stormwater Systems

The ability to understand and improve the design of ecologically-based stormwater systems depends on the proper assessment of their current performance. Typically, the same metrics used to assess traditionally-designed stormwater structures are applied to ecologically-designed systems, and include measures of both water quality (such as total suspended solids, contaminant concentrations, and BOD) and water quantity (such as peak flow and volume reduction). However, unlike the concrete-lined channels and detention facilities traditionally used to manage stormwater, the performance of ecologically-designed stormwater systems hinges upon ecosystem processes - including sedimentation, infiltration, sorption, biological degradation, nutrient transformation, and evapotranspiration - occurring within the system. For this reason, the overall health of the ecosystem should be considered when assessing the performance of these systems. While physical and chemical indicators such as water quality and quantity are important assessment tools, additional indicators which integrate the biological and ecological aspects of the system are needed to provide a more complete picture of overall health and performance potential of ecological stormwater systems (Watzin and McIntosh, 1999). The following section includes a review of the literature describing some of the metrics developed to assess ecosystem health, and how these metrics could be applied to ecologically-based stormwater systems.

Literature Review

A healthy ecosystem is one that maintains its organization and autonomy over time and is resilient to stress (Doran and Parkin, 1996). Many attributes of ecosystem health, such as the integrity of nutrient cycles, energy flowpaths, and resilience, are difficult to directly measure (Pyke et al., 2002). As a result, ecological and biological indicators representing components of these difficult-to-measure attributes were developed in the late 1980’s to aid in assessing ecosystem health (Jørgenson et al., 2005). Since that time, a variety of indicators have been suggested for use across a wide spectrum of both aquatic and terrestrial ecosystems. Indicators can be as simple as a general presence or absence of a species, or as complex as detailed energy balances (Jørengson et al., 2005). Due to the inherent complexity and heterogeneity of natural systems, ecologists recognize that it is not feasible to use a single indicator, or even a few, as a general assessment of ecosystem health. Rather, sets of indicators tailored to a particular ecosystem are used in concert to assess ecosystem health (Jørgenson et al., 2005).

Biological indicators developed to assess the health of aquatic ecosystems are documented extensively in the literature. The United States Environmental Protection Agency (EPA) has established detailed methodologies for assessing wetland health and function based on the resident plant, microbial, and macroinvertebrate communities (Adamus, 1996). Jørgenson et al. (2005) presented a compilation of ecological indicators developed for a wide range of ecosystems in The Handbook of Ecological Indicators for Assessment of Ecosystem Health, the majority of which were specific to aquatic ecosystems. In addition, the majority of case studies presented in the literature to demonstrate the use of ecological indicators were conducted in aquatic ecosystems (Jørgenson et al., 2005; Watzin and McIntosh, 1999).

Many of the metrics presented in the scientific literature have been developed for a specific ecosystem and require large amounts of data and time to apply (Jørgenson et al, 2005). However, more generalized and simplified assessments of aquatic ecosystem health have been developed, particularly for use by lay groups to assess the integrity of lakes, streams, and wetlands.

Quantitative measures have also been developed to assess the health of terrestrial ecosystems, including forests, rangeland, desert shrubland, and agricultural landuses. Soil quality is the predominant focus of many terrestrial assessments because it is the soil that provides vital ecosystem services such as supporting plant growth, filtering water and cycling nutrients. Doran and Parkin (1996) present a minimum data set for assessing soil quality, including soil texture, rooting depth, infiltration, bulk density, water holding capacity, nutrient content, microbial biomass, and respiration. Many of the quantitative indicators developed to assess terrestrial ecosystem health can be just as complex and time-consuming as those developed for aquatic ecosystems. So, as simpler, qualitative measures of health were developed for aquatic ecosystems, simple tools that can be rapidly applied to assess the integrity of terrestrial ecosystems have also been developed. For example, Manske (2002) presents a nonquantitative procedure for assessing rangeland health that can be easily conducted by rangeland managers. While such a qualitative assessment does not provide the level of precision of more complex, scientific methods, the results of a qualitative assessment can still provide valuable insight to changes in ecosystem health over time when conducted via valid scientific procedures. Manske's method combines an annual inventory of the major plant species present with a qualitative health assessment to evaluate the level at which ecosystem processes are operating and examine interactions among climate, soil, vegetation, and biota (Manske, 2002). A standardized scoring rubric is used to assign a score of one to four to each component of the health assessment based on the condition of the site. After completing this assessment, rangeland managers can make decisions based on the scores for each ecosystem component to ensure that the rangeland ecosystem as a whole will perform at its peak potential.

Clearly, ecological health indicators have already been developed for and applied to naturally occurring aquatic and terrestrial ecosystems- the same systems upon which ecological stormwater systems are engineered. As noted, many quantitative ecological indices which have been developed are relatively involved and may require large amounts of data, time, and expertise to calculate. While such a degree of complexity is desirable for some ecological studies, it is less likely to be practical for the general assessment of ecological stormwater systems, the monitoring of which is typically left to municipalities with limited time and finances. The objective of this study was to examine the feasibility of applying a qualitative ecological assessment tool to ecologically-designed stormwater systems in order to provide municipal stormwater managers and consultants with an easy-to-use and inexpensive means of quickly assessing the general system health and performance of the system.

The assessment tool developed in this study was based on ecologically-designed stormwater systems in Northeastern Kansas. The following section briefly describes ecological and infiltration-based stormwater management practices in Kansas and throughout the Midwest, and how the development of an ecological assessment tool could aid in the monitoring and management of these systems.

Eco-based Stormwater Management in the Tallgrass Prairie Ecosystem

Throughout Northeast Kansas and much of the mid-continental United States, infiltration-based stormwater management practices are beginning to be adopted as a means of reducing stormwater volume while improving stormwater quality. The majority of these practices focus on incorporating vegetation to reduce peak runoff rates and improve infiltration. Prior to the advent of agriculture and urbanization, the tallgrass prairie ecosystem dominated most of the Midwest. In addition to their extensive root systems that enhance the infiltration properties of the soil, these robust prairie grasses are also adapted to the region's climatic patterns, enabling them to survive intense storm events followed by extended periods of drought. Studies have shown that prairie grasses can be very effective in intercepting rainfall before it reaches the soil, thus reducing runoff. For example, Weaver and Rowland (1952) found that thick stands of big bluestem (*Andropogon gerardii*) were capable of intercepting 97% of the rainfall during very light showers and about 66% during storms in which 3 cm to 4.5 cm (1.2-1.8 inches) of rain fell. Stated in other terms, a well-developed stand of big bluestem with fully-developed foliage may intercept over 1.3-cm of water per acre when 2.5-cm of rain falls in 1 hour (Weaver and Rowland, 1952). Infiltration rates under well-developed prairie grasses are also impressive. In a study comparing infiltration rates under the tallgrass prairie native big bluestem and Kentucky bluegrass (*Poa pratensis*), a shallower-rooted cool-season grass which has been domesticated for use in lawns, big bluestem exhibited infiltration rates up to 480% greater than bluegrass (Weaver and Rowland, 1952). By virtue of their ability to substantially improve infiltration and reduce surface runoff, native prairie grasses are ideal candidates for vegetated stormwater systems in the Midwest.

Although vegetated stormwater systems are being implemented as a means of controlling urban non-point source pollution, the overall impact of these systems on receiving aquatic ecosystems still requires much study. Typical metrics used to evaluate the performance of these systems include peak flow rate and stormwater volume reduction, as well as pollutant removal capabilities. While such hydraulic and chemical analyses provide valuable insight to the system's performance, collection of these data typically require specialized equipment or costly laboratory analysis. In addition, these measures do not incorporate the ecological processes upon which the system's pollutant removal mechanisms are based. The limitations of hydraulic and chemical measures present a challenge to the long-term monitoring of vegetated stormwater systems. In this study, traditional hydraulic and chemical performance assessments were applied in addition to a quantitative ecological health rubric to determine if such an ecological health assessment tool can provide some indication of system performance. The development of the ecological health rubric is discussed in the following sections. The rubric developed for this study is intended to provide a better understanding of ecological function and assessment in ecologically-designed stormwater systems and how these functions can improve system performance.

Methods

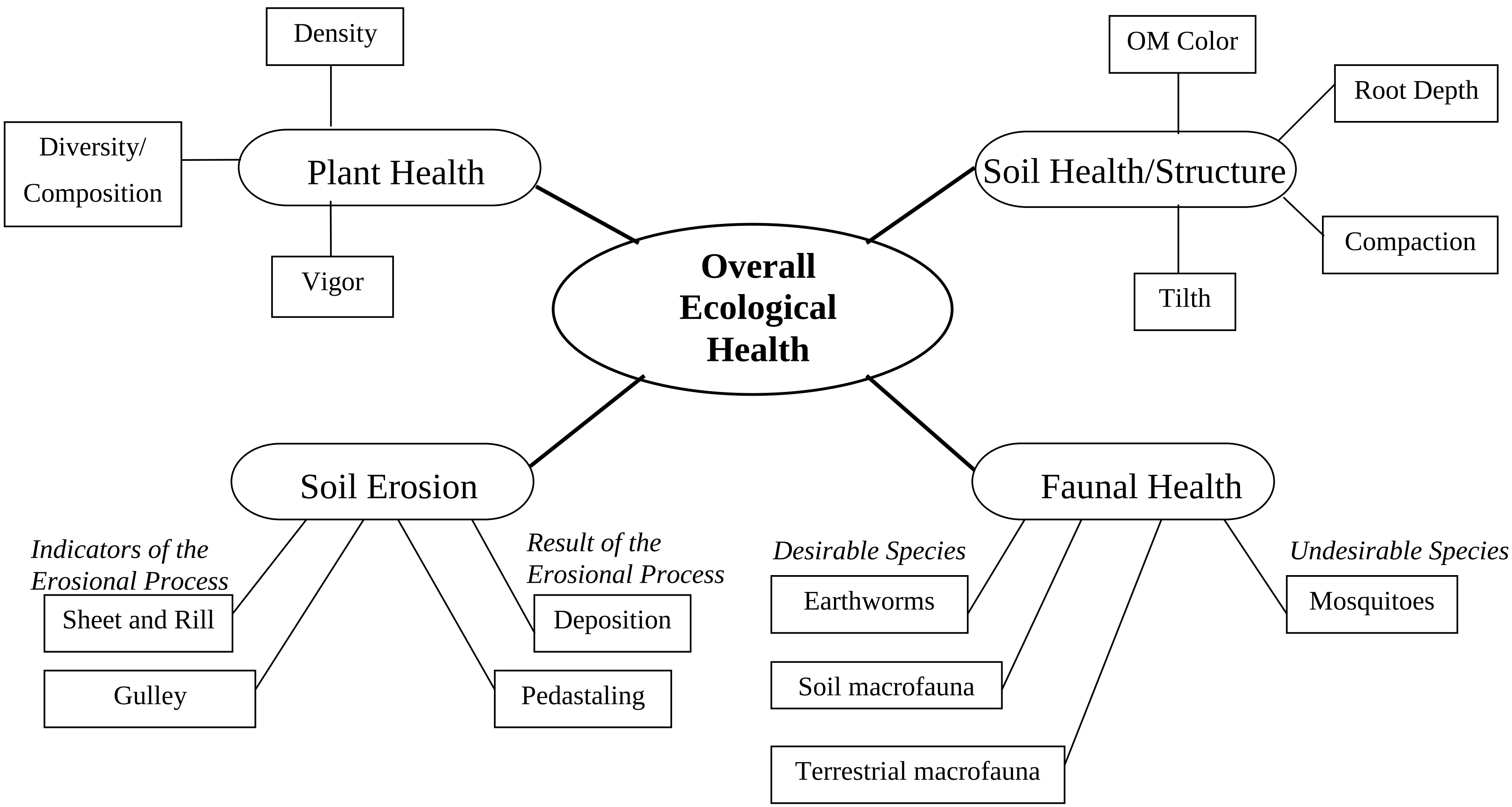
Two study sites at different stages of vegetative maturity were selected for the development of the ecological assessment tool. Both were located in Northeastern Kansas, in an area once dominated by the tallgrass prairie ecosystem. The first of these sites was a stormwater detention basin located in a medium-density residential neighborhood in Topeka, Kansas. The basin, which will be referred to as the Quinton Heights basin after the neighborhood in which it was built, was constructed in 2004 to relieve flooding in the area. The 1,550 square-meter basin was designed to receive runoff from a 6,000 square-meter area, the majority of which flows down a street and into the basin through a grated opening placed in the middle of the street (Spaar, 2004). After excavation, the basin was replanted with grasses native to the tallgrass prairie, including big bluestem (*Andropogon gerardii*), Indiangrass (*Sorghastrum nutans*), switchgrass (*Panicum virgatum*), and sideoats gramma (*Bouteloua curtipendula*). At the time of the study, the Quinton Heights basin was in its third year of operation, so the grasses were well-developed and approaching vegetative maturity as defined by Weaver and Zink (1946). The second site, located in Johnson County, Kansas, was a prairie restoration project intended to intercept runoff from an adjacent municipal building. The yard surrounding the building was originally a traditional fescue lawn. The lawn was sprayed with Roundup® in the spring of 2007 and a mix of mid-height prairie grasses, including sideoats gramma (*Bouteloua curtipendula*) and hairy grama (*Bouteloua hirsute*), was seeded directly into the lawn in June 2007. To help prevent erosion, a cover crop of annual rye (*Lolium multiflorum)* was planted after the native grass was seeded. Soil samples were taken from both sites and analyzed by the Kansas State University Soil Testing Lab. Both sites were classified as silty clay loams, the expected saturated hydraulic conductivity of which is 0.15 cm/hr (Rawls, et al., 1982). Double-ring infiltrometers were used to conduct infiltration studies to examine changes in effective saturated hydraulic conductivity (K_{eff}) throughout the study period in addition to differences in K_{eff} between the two systems at different stages of vegetative maturity. Infiltration tests were conducted at both sites on a monthly basis in replicates of three. In addition to monitoring infiltration rates, both sites were instrumented with ISCO automated water samplers (ISCO Inc., Lincoln, NE) to measure peak runoff rates and total runoff volume entering and leaving the site. Water samples were also analyzed for total suspended solids (TSS), phosphorus and nitrogen content, as these are among the pollutants of greatest concern in the region. An ecological health rubric, which will be described in the following section, was also developed to assess system health on a monthly basis.

Development of the Ecological Assessment Rubric

Using qualitative assessments already developed for soil and rangeland health as a guide, an ecological assessment rubric was developed for application to the study sites described above, both of which were modeled after prairie ecosystems. The rubric was broken into four main categories: vegetation health, soil structure, soil erosion, and faunal health. Each of these categories contained three to four indicators which were given a score of one through four, with one being poor condition, two being fair, three being good, and four being excellent condition. These scores were assigned based upon how the conditions at the site compared with the conditions described in the rubric for each ranking. Indicators within each category were chosen based on their relevance both to ecosystem health and desired function of a stormwater management system, namely to soil stability, hydrologic function, and biotic integrity. To determine the overall score for each category, the individual scores assigned to each indicator within the category were summed. A final rating for overall ecosystem health was determined by summing the scores for each of the four categories. A diagram of the rubric developed for this study is presented in Figure 2.1 to graphically illustrate the indicators within each category and their contribution to the overall health score

It should be noted that the scoring system used for this assessment rubric assumed that all indicators contribute equally to the overall health of the ecosystem and did not account for the relative importance of one indicator to another. Still, an equally-weighted rubric was selected as the best model for the assessment developed for this study as it is simple and its use is supported

Figure 2.1. Ecological health assessment rubric diagram.



by similar assessments in the literature which report satisfactory response despite being equally weighted (Manske 2002; Romig, 1996).

The following sections describe each of the indicators included in the rubric in greater detail and provide reasoning for their inclusion in an assessment for the function of an ecological stormwater system. Descriptions of the “Excellent” and “Poor” ranking are given for each indicator to illustrate the full range of possible conditions. The complete ecological health rubric developed for this study includes descriptions for the “Good” and “Fair” rankings as well, and is displayed in Table 2.1. A Student’s t-test was performed using the data analysis function in Microsoft® Office Excel® to compare average scores between the two sites in each of the categories to determine statistical significance at the 95% confidence level ($p < 0.05$).

Table 2.1, part 1. Ecological Health Rubric developed to assess health of ecologically-designed stormwater systems.

	4 - Excellent	3 - Good	2 - Fair	1 - Poor	Total
Plant Health -Density -Diversity/ Composition -Vigor	-Plants closely spaced with even distribution pattern. Less than 20% soil surface exposed. -Diverse plant community with no invasive species. -Plants are vigorous with balanced mix of young and mature growth.	-Plants closely spaced with somewhat even distribution pattern. 20-40 % soil surface exposed. -Diverse plant community with a few less desirable species. -Plants are vigorous with no deformed growth patterns	- Patchy plant spacing and distribution pattern. 40-60% soil surface exposed. -Reduced diversity with some less desirable and invasive species. -Plants pale green or yellowing, deformed growth patterns or are developing close to the ground.	- Fragmented/clumped plant spacing and distribution pattern. Over 60% soil surface exposed. -Restricted diversity with many undesirable or invasive species. -Plants appear stressed , are developing close to the ground. Most are dead or dying.	
Comments: (record presence of desirable and/or invasive species and whether appropriate to ecoregion and stormwater system goals.)					
Soil Erosion Indicators -Sheet & Rill -Gullies -Deposition -Pedestaling	-Soil removal by wind or water is not evident -No bare soil deposits. Plants have colonized soil deposits. -Recent gully formation is not evident. If gullies are present, they are small and vegetated. -Plant pedestaling is not evident.	-Small rills developing. Transported soil remains on site. -A few bare soil deposits. Plants are stabilizing recent deposits. -Very little recent gully formation. If some gullies present, they are small and vegetated. -Very little plant pedestaling.	-Sheet and/or rill erosion occurring in small areas. Most soil remains on site.. -Several small soil deposits due to deposition. Plants have not stabilized. -Some recent gully formation but still small and unbranched. -Plant pedestaling is evident but not so severe that roots are exposed.	-Sheet and rill erosion occurring in large areas. Much of the soil transported off site. -Deposited soil inhibiting plant growth or present as large, bare deposits. -Well-developed, active gullies present. -Plant pedestaling has exposed plant roots.	
Comments:					

Table 2.1, part 2. Ecological Health Rubric developed to assess health of ecologically-designed stormwater systems

	4 – Excellent	3 – Good	2 – Fair	1 - Poor	Total
Soil Health/ Structure -OM Color -Roots/Residue -Subsurface Compaction -Soil Tilth	-Topsoil clearly defined, darker than subsoil -Roots penetrate over 15 cm deep; surface residue abundant -Wire flag easily inserted 20 cm or more -Soil crumbles well and is easy to slice through	-Topsoil somewhat darker than subsoil -Roots 10 cm to 15 cm deep; surface residue abundant -Wire flag inserted 20 cm deep or more, but with some effort -Soil crumbles fairly well but some clods persist	-Topsoil only slightly darker than subsoil -Roots 5 cm to 10 cm deep; some surface residue -Considerable effort required to insert flag up to 20 cm -Soil is cloddy	-No difference between color of topsoil and subsoil -Roots less than 5 cm deep; no or little surface residue -Wire cannot be inserted to 20 cm depth without bending or breaking -Soil very cloddy; clods hard like brick	
Comments: (Structure assessment can be performed while excavating area for earthworm count or with a soil bulk density corer.)					
Living Organisms -Desirable Species -earthworms -soil macrofauna -others (birds, dragonflies, butterflies, ect.) -Undesirable Species (mosquitoes)	-10 or more earthworms in excavated area with many casts and holes -several other species of soil macrofauna (ie, beetle and cicadae larvae, millipedes, etc) present -several different organisms present in/ around the system -No mosquitoes or mosquito larvae present	-5 to 9 earthworms in excavated area with some casts and holes -a few other species of soil macrofauna present -several different organisms present in/ around the system -No mosquito larvae, though a few mosquitoes	-1-4 earthworms in excavated area with few casts and holes -1 other species of soil macrofauna present -One species of organism present in/ around the system -Mosquito larvae present in standing water	-neither earthworms or their casts and holes in excavated area -no other species of soil macrofauna present -No organisms present in/ around the system -Both mosquitoes and mosquito larvae prevalent	
Comments/ Record organisms observed here:					

Plant Health

The category for plant health considered plant density, diversity, and overall vigor. In an ecological stormwater system, vegetation health is perhaps one of the easiest characteristics of the system to observe. Let us consider each of the components used to assess plant health. In an ecological stormwater system, as well as in most other vegetated systems, establishing a dense stand of vegetation is extremely important in order to prevent soil erosion from the system. In addition to holding soil in place, vegetation also helps to prevent the occurrence of soil sealing and the formation of soil crusts by protecting soil from the impact of raindrops (Holman-Dodds, 2006). An additional advantage of closely spaced vegetation in vegetated stormwater systems is that dense vegetation slows the flow of water moving through the system better than fragmented clumps of vegetation, thus improving the removal of suspended sediments. The rubric developed for this study assessed density on the basis of plant distribution and the amount of soil surface exposed. Systems in which plants were distributed such that less than 20% of the soil surface was exposed received a score of “Excellent” (4) while those with a clumped or fragmented distribution pattern which left over 60% of the soil surface exposed received a ranking of “Poor” (1). Percent cover partitions for the plant distribution category were based on a similar partitioning scheme set forth by Manske (2002) for rating plant distribution in rangeland health assessments.

In addition to plant density, plant diversity is another important component of vegetation health. Diversity is desirable for many reasons, the foremost of which is that diverse systems are usually more resilient against stressors such as the introduction of disease, pests, or toxic chemicals, and unfavorable climatic conditions (Collins, et al. 1998). In the case of prairie ecosystems, plant biodiversity is an important feature of ecosystem function. Different species of grasses and forbs exhibit different rooting structures to extract water and nutrients from different depths within the soil profile (Weaver, 1958). Such diversity in root structure is a desirable feature for ecological stormwater systems because it allows the soil profile to be dried out more completely between rainfall events so to increase the storage capacity of the soil for the next storm. Plant diversity is also important when considering the hydrologic aspects of a vegetated stormwater system as some plants tolerate frequent inundation with water while others flourish in drier environments. In addition to the aforementioned functional aspects, the aesthetic appeal of a diverse plant community offers another advantage to vegetated stormwater systems. For example, the various heights, colors, and flowering parts of different grasses, forbs and wildflowers can be exploited to design a stormwater system that is both functional and pleasing to the public eye. For the purposes of this assessment, "Excellent" sites are those with a diverse plant community with no invasive species while sites with greatly restricted diversity or many undesirable or invasive species was considered "Poor."

While a diverse vegetative community is desirable, not all plants are as desirable as others. In the case of prairie ecosystems, invasive species such as *Sericea lespedeza* can crowd out native grasses and transform the system into a monoculture (Ohlenbusch et al., 2001). In a stormwater system designed after a prairie ecosystem, other less desirable plants include weedy annuals such as foxtail (*Setaria spp.*), crabgrass (*Digitaria spp.*), and stinkgrass (*Eragrostis cilianensis*). These grasses have a shallow root system and therefore contribute minimally to improving soil structure or infiltration properties. Shallow-rooted plants are also unable to access water once it has percolated below the top few inches of soil, rendering these plants relatively ineffective at drying out deeper portions of the soil profile between rain events. The desired plant composition will depend upon both ecosystem type and the goals of the system. In the case of prairie ecosystems after which the stormwater systems in this study were modeled, a ranking of "Excellent" was assigned for systems with a diverse plant community of grasses, forbs, and wildflowers with no invasive species. A "Poor" ranking was given to systems with relatively little diversity and many invasive or undesirable species such as weedy annuals.

The final indicator of plant health used in the rubric was the overall appearance of the plant, including color, vigor, and growth patterns. Plant appearance can be used as an indicator of plant stress due to environmental conditions such as nutrient or water limitations. A high proportion of dead or dying vegetation indicates that recruitment is not occurring and that the site is at risk of being overtaken by undesirable plants, such as weedy annuals and invasive species (USDA NRCS, 1997). In the assessment developed for this study, sites at which the vegetative community exhibited vigorous growth and a balanced mixture of young and mature plants were given an "Excellent" score. Sites at which the majority of the plants appeared stressed, were developing close to the ground, or were dead or dying were given a "Poor" classification.

Soil Erosion Indicators

Since the goal of most stormwater systems is to provide some degree of water quality improvement, the loss of soil from the system by erosion- and subsequent addition of soil to stormwater leaving the system- is typically not desirable. Soil erosion indicators have been incorporated into other assessments of terrestrial ecosystem health (Manske, 2002; Romig et al., 1998) and were also included in the rubric developed for this study. To assess the prevalence of erosional activity in the system, erosional indicators including sheet and rill erosion, gully formation, plant pedestals and terracettes, and excessive deposition were considered.

Soil loss by sheet or rill erosion is evidenced by the presence of linear streamlets cut by flowing water. The frequency and spatial distribution of these streamlets was used as an indicator to assess rill erosion at the site. An “Excellent” rating was given to sites with no evidence of soil removal by either wind or water. A “Poor” rating was assigned to sites at which rills were widely distributed across the area and accompanied by evidence of soil transport off the site.

If the streamlets formed by sheet and rill erosion are not revegetated, continued erosion may transform these small channels into gullies. Sites at which gullies were present and actively eroding were given a “Poor” rating. If recent gully formation was not evident and gullies that were present were small and vegetated, an “Excellent” rating was assigned.

Plant pedestals and scouring can be used as additional indicators of soil erosion (Pyke, 2002) and were also included in the rubric. Plant pedestalling occurs when soil is removed from around the base of plants via erosion and can be observed by checking for exposed roots (Manske, 2002).

Scour around and beneath rocks or other hard surfaces, including the inlet and outlet pipes of stormwater systems, is another indicator of soil loss and excessive erosional energy. If neither pedestalling nor scour was present at the site, an “Excellent” rating was assigned. Sites at which plant roots were exposed and/or severe scouring had occurred received a rating of “Poor.”

In keeping with the goal of water quality improvement, deposition is expected in ecologically-designed stormwater systems as stormwater flows are slowed and sedimentation occurs. However, excessive deposition can have negative effects on the system by decreasing capacity or clogging of pores by fine sediments. These potential negative impacts are countered by the establishment of vegetation on soil deposits as root development and associated biological activity will improve porosity and the overall structure of deposits. To assess the impact of deposition on the health of the system, the presence and extent of soil deposits were observed. Sites in which soil deposits were being revegetated were given an "Excellent" rating. If deposited soil appeared to be inhibiting plant growth or occurred in large, bare deposits, a "Poor" rating was assigned.

Soil Health/Structure

Soil health indicators included in the assessment rubric were the organic matter layer, the density of roots and residue, the presence of compacted soil layers, and soil tilth. Unlike the indicators for plant health, which could be determined from the surface, the indicators for soil health require a look below the ground. Other qualitative assessments of soil health have employed either a shovel (Manske, 2002) or soil core (Arshad et al., 1996) to observe soil health indicators. In this study, a both tools were used to observe soil characteristics. The methods used to score each of the aforementioned soil health indicators, along with the reasoning behind their inclusion in the assessment developed for this study, follows.

A well-developed organic layer is beneficial for soils in stormwater management systems as this nutrient rich layer provides nutrients to support the growth of vegetation at the site while helping to improve infiltration properties of the soil (Holman-Dodds, 2006). Soils rich in organic matter tend to be darker in color than the subsoil beneath, and the relative color of the upper soil layer has been used as a simple qualitative indicator of healthy soil (Manske, 2002). The rubric developed for this study incorporated a measure of organic matter by comparing the color of the topsoil with that of the subsoil. A standard soil bulk-density core was used to remove soil from the upper 8 cm of the profile to examine the coloration of the topsoil and subsurface layers. Soils in which the topsoil was clearly darker than the subsoil were given an "Excellent" rating. If topsoil was light in color and could not be differentiated from the subsoil, a rating of "Poor" was assigned.

The abundance of roots and residue are also important components of a healthy soil ecosystem. The degree of root development will vary depending on vegetation age and type, soil conditions, and climate. For stormwater management, deep, well-developed root systems are desirable as they have a greater impact on infiltration and stormwater volume than do shallow-rooted grasses (Perrygo et al. 2001; Weaver and Rowland, 1952). This is partly due to the effect of roots on soil physical properties. Over time, root penetration has been found to contribute to increased soil porosity and the development of stable soil aggregates, both of which promote higher infiltration rates (Holman-Dodds, 2006). Roots continue to enhance soil infiltration properties as they decay by increasing the organic matter content and creating macropores through which high infiltration rates have been observed due to preferential flow (Linden et al., 1991). The other advantage of deeply-rooted plants over those with shallow root systems is that deeply-rooted plants are able to consume water from a greater portion of the soil profile for transpiration in between storms, thus increasing the storage capacity of the soil for the next runoff event (Holman-Dodds, 2006). Rooting depth and density were observed using a shovel to expose the top 15 to 20 cm of soil. Although the root systems of many tallgrass prairie species, including those planted at the study sites, can grow to depths greater than 1.5 meters, sites at which roots penetrated 15 cm or more were given an "Excellent" rating for the purposes of this study. A dense root system at this depth would still impact infiltration for a typical rain event in the Midwest, and the required sampling depth of 15 to 20 cm makes observation of this indicator more practical for the intended user of this assessment. A "Poor" rating was given to sites at which roots were sparsely distributed or did not penetrate beyond 5 cm.

Root development can be adversely impacted by the presence of compacted soil layers. Subsurface compaction can also restrict infiltration and nutrient cycling processes (USDA NRCS, 1997) and is therefore undesirable in ecological stormwater systems. Compacted soil layers can be detected through the use of a penetrometer, or, more simply, by probing the soil with a sharp rod or shovel (USDA NRCS, 1997). To check for the presence of compacted layers in this study, the ease with which a wire flag could be inserted was gauged as suggested by Romig (1996). An "Excellent" rating was assigned to soils in which the flag is easily inserted to a depth of 15 cm (6 inches) or more, while a "Poor" rating was given to soils in which the flag cannot be inserted to this depth without considerable bending or breaking. The 15 cm depth was selected as a threshold for determining the presence of compacted layers based on literature reports that cite compaction layers typically occur in the upper 15 cm of the soil profile (Pyke, 2002; USDA NRCS, 1997).

Soil tilth is a qualitative indicator commonly used in agronomic applications to describe the soil's suitability for supporting plant and root growth. Good tilth pertains to soils which are friable with a stable assemblage of aggregates, and is a function of soil texture, soil structure, and organic matter content (Hillel, 1998). The pore spacing in soils with good tilth is large enough to allow adequate air and water movement through the soil. Soil tilth can be assessed by crumbling a fistful of soil in one's hand and observing the ease with which the soil crumbles and the size of the aggregates, or soil crumbs, into which the soil breaks (Manske, 2002).

Faunal Health

Although largely ignored in stormwater management literature, the ecological literature focuses heavily upon the interaction between biota and the physical and chemical components of the ecosystem, including infiltration and nutrient cycling. In the soil, earthworms are among the most important components of the soil biota due to their role in the formation and maintenance of soil structure and fertility (Edwards, 2004). As earthworms burrow through soil, they ingest mineral particles, which are then mixed with organic matter in the earthworm gut and form stable aggregates when excreted. The soil aggregates formed during passage through the earthworm gut contribute to improved drainage and moisture-holding capacity of the soil profile (Edwards, 2004). Of particular interest to stormwater management is the affect of earthworms on infiltration. Earthworms are a major source of biological macropores in many soils, including those in the Midwest, and thus impact the rate at which water is transmitted through the soil profile (Linden, et al., 1991). Studies that have quantified infiltration due to earthworm burrows and have reported a wide range of infiltration rates, but all conclude that the flow rate in burrows is much greater than in the surrounding soil matrix (Edwards, 2004; Shipitalo and Butt, 1999; Weiler and Naef, 2003).

In addition to their influence on soil properties and processes, earthworms also respond to ecosystem disturbances, such as urbanization. For example, in a study of different-aged urban systems, Smetak (1998) observed significantly fewer earthworms in urban yards less than 10 years old (26 worms per square meter) than in urban yards greater than 75 years old (121 worms per square meter). Earthworm population densities of 300 per square meter have been reported for the Konza Prairie, a native tallgrass prairie reserve in northeastern Kansas (Ransom et al., 1998). Coupled with the relative ease of observing earthworms over other soil organisms, these important soil macroinvertebrates have been identified as excellent biological indicators of soil health and, therefore, were included in the health assessment developed for this study.

The sampling method chosen to quantify earthworms depends on the basic life histories of the earthworms found in the study area (Blair et al., 1996). Earthworms are classified into one of three groups based upon their feeding and burrowing strategies: (1) Epigeic species, which live in or near the surface litter, (2) Endogeic species, which live within the soil profile in temporary, horizontally-oriented burrow systems that are filled with cast material as the earthworm moves through the soil, and (3) Anecic species, which create permanent, vertically-oriented burrow systems extending up to 2 m into the soil profile. Since both shallow and deeper dwelling earthworms have been reported in the tallgrass prairie ecoregion (Ransom, et al., 1998), a combination of sampling procedures was chosen according to sampling methods outlined in the literature (Blair et al., 1996). First, vegetation and residue at the surface was cleared from an approximately 50 cm square area (2.7 ft^2) and examined for surface-dwelling earthworms. A shovel was then used to excavate the area to a depth of 20 cm. All excavated soil was hand-sorted to determine the presence of earthworms living in the upper region of the soil profile. A solution containing 5 g/L of dry mustard dissolved in tap water was then applied to the bottom of the excavated area to elicit any deeper-dwelling anecic earthworms present in the excavation area from their burrows to the surface. Earthworms were sampled a total of three times at each site: once in late spring and twice in the fall. These sampling times were chosen to correspond with the seasonal height of earthworm activity (Blair et al., 1996). Due to the relative invasiveness of the excavation method, only one excavation was made at the site per sampling event. Sites at which 10 or more earthworms were found were given an “Excellent” rating. If neither earthworms nor their casts or burrows were found in the excavated area, a “Poor” rating was assigned.

In addition to earthworms, other species of macrofauna inhabit the soil and contribute to biological diversity. Beetle and cicadae larvae, millipedes, and centipedes are among the most common arthropods present in prairie ecosystems (Ransom, et al. 1998) and their presence or absence was included in the rubric developed for this study. Soils in which four or more other species of soil macrofauna were found received an “Excellent” rating while soils devoid of soil macrofauna received a “Poor” rating.

Above-ground species diversity was also considered in the rubric. While not directly vital to the performance of the stormwater system, insects such as bees and butterflies can aid in pollinating wildflowers planted for aesthetic value in the stormwater system. Other insects, such as beetles, grasshoppers, and crickets attract birds to make ecological stormwater systems more of a public amenity. Sites at which several different species of insects, birds, or other wildlife were observed were given an “Excellent” rating while sites at which no organisms were observed were assigned a “Poor” rating.

While species diversity was used as an indicator of system health, some organisms are not desirable in ecologically-designed stormwater systems. Mosquitoes are perhaps the foremost of these, primarily for reasons concerning public health and relations. Systems in which no adult mosquitoes or their larvae were observed were given an “Excellent” rating, while a “Poor” rating was assigned to sites at which both mosquitoes and mosquito larvae were prevalent.

RESULTS AND DISCUSSION

The ecological health rubric developed for this study was applied to the two study sites once every three to four weeks over the course of the growing season. The results and overall assessed health of the stormwater system are summarized in Table 2.2. The complete data set and statistical analysis for the health assessment is included in Appendix A. The Total column in the table represents the sum of the health scores from each of the four categories and is used as an indicator of the overall ecological health of each system. In basing the total score on the summation of category scores, this rubric assumes that the indicators in each category contribute equally to the overall ecological health of the system. Although this assumption overly simplifies the complex relationships related to ecosystem health and function, it was deemed acceptable for the purposes of this assessment.

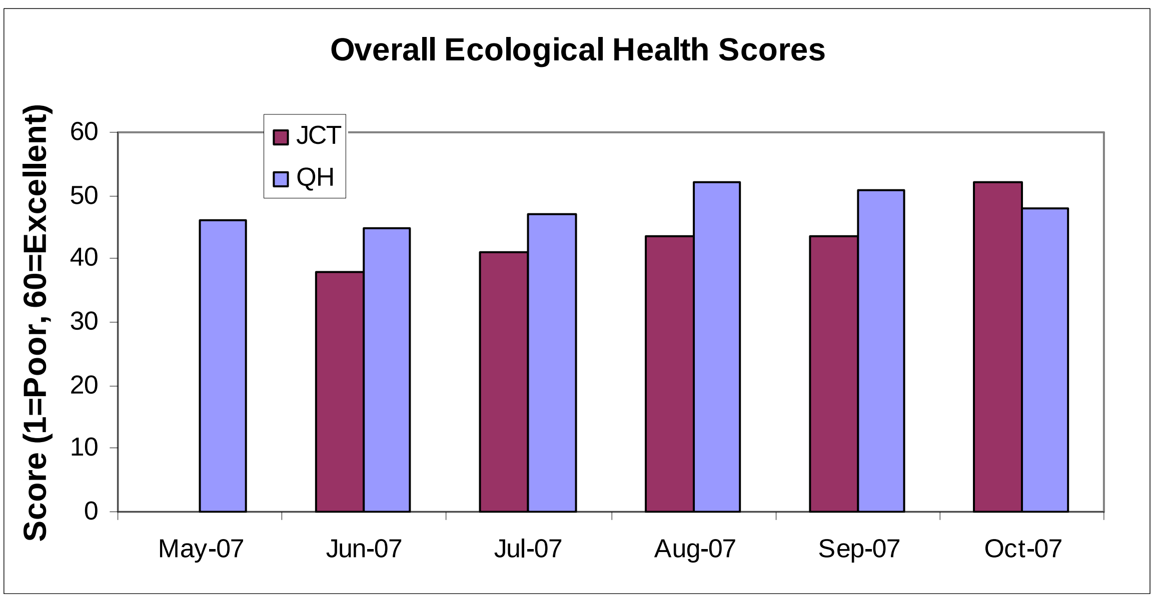
Table 2.2 Summary of scores from ecological health assessment conducted at Johnson County (planted in June 2007) and Quinton Heights (planted in summer of 2004).

Ecological Health Assessment											
Johnson County						Quinton Heights					
Date	Plant Health	Soil Erosion	Soil Health	Faunal Health	**Total**	Date	Plant Health	Soil Erosion	Soil Health	Faunal Health	**Total**
6/26/2007 [a]	6	15	10	7	**38**	5/8/2007	10	11 [b]		21	26
7/10/2007	8	15	9	9	**41**	5/31/2007	11	14	13	38	46
8/6/2007	9	16	9	9	**43**	6/22/2007 [a]	9	14	13	36	45
8/24/2007	10	16	7	11	**44**	7/20/2007	10	15	13	38	47
9/5/2007	9	16	7	10	**42**	8/7/2007	11	16	14	41	52
9/28/2007 [a]	10	16	10	9	**45**	9/7/2007	11	16	14	41	51
10/10/2007	10	16	11	15	**52**	10/2/2007 [a]	10	13	14	37	48

[a] denotes dates on which earthworms were sampled.
[b] soil health not assessed due to overly wet soil conditions

A more graphic comparison of the changes in the ecological health scores at the two sites during the study period is provided in Figure 2.2. As seen from the figure, the overall health score generally increased throughout the growing season at both sites. This was expected because many of the indicators used in the rubric, particularly vegetation density and vigor, improve as the growing season progresses. Numerically, the average overall score at Quinton Heights was higher than that at Johnson County. However, the Student's t-test conducted between the overall health scores at the two sites indicated the differences in overall health scores at the two sites were not statistically significant ($p = 0.71$). So, although the vegetative community at Quinton Heights was more mature than that at the Johnson County site, its overall ecological health was not significantly greater than that of a recently revegetated site. The following sections will discuss observations for each of the four assessment categories and their contribution to the overall ecological health score at each site.

Figure 2.2 Overall scores from the ecological health assessment conducted for the stormwater systems at Johnson County and Quinton Heights.



Plant Health

Observed scores for plant health at the Quinton Heights and Johnson County sites are displayed in Figure 2.3. In general, plant heath scores at Quinton Heights were consistently higher than that at Johnson County, primarily because the Quinton Heights site was in its third growing season at the time of the study, whereas the grasses at Johnson County were seeded at the start of the study period. The difference between the vegetative maturity of the sites was most apparent in the density and types of plants that dominated either site. The grasses at the Quinton Heights site covered approximately 80% of the ground surface from the start of the growing season, whereas the emerging vegetation remained sparsely distributed for the first two months at the Johnson County site. The grass community at the Quinton Heights basin was dominated by perennial prairie grasses, including big bluestem, sideoats gramma and indian grass. The wet, central channel of the basin was dominated by the moisture-loving species barnyardgrass (*Echinochloa crus-galli*), switchgrass, and prairie cord grass (*Spartina pectinata*). Plant health at the Johnson County site was lowest at the beginning of the observation period due to the sparse distribution of the emerging vegetation, but gradually increased as the growing season progressed and the grasses began to fill in the site. Although grasses were healthy in appearance and covered approximately 80% of the soil surface by early August (both of which earned the site "Excellent" scores in the plant health category) the plant community was dominated by weedy annuals, including yellow foxtail, stinkgrass, and crabgrass. Dominance by such less desirable plants is, however, expected in the early stages of prairie restoration projects, and the site at Johnson County proved to be no exception. It is expected that perennial prairie grasses originally planted at the site, namely sideoats and hairy grama, will become established over the next one to two years to move the site closer to its ecological potential. The photographs in Figure 2.4 depict the Quinton Heights and Johnson County sites toward the peak of the growing season.

Figure 2.3 Scores for the plant health category at the Johnson County (JCT) and Quinton Heights (QH) sites.

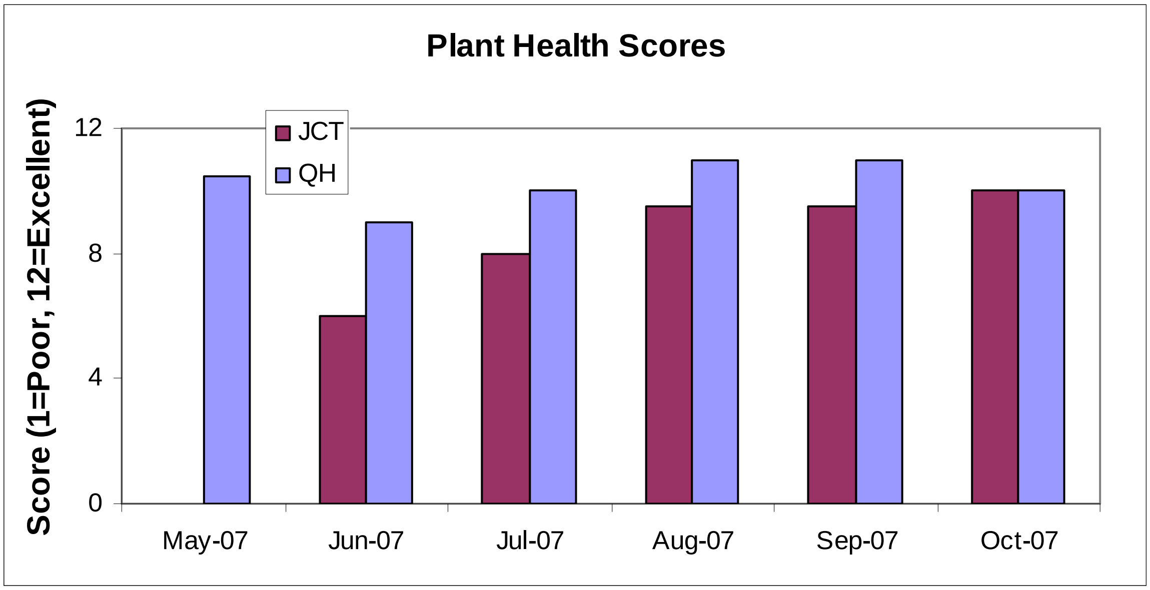


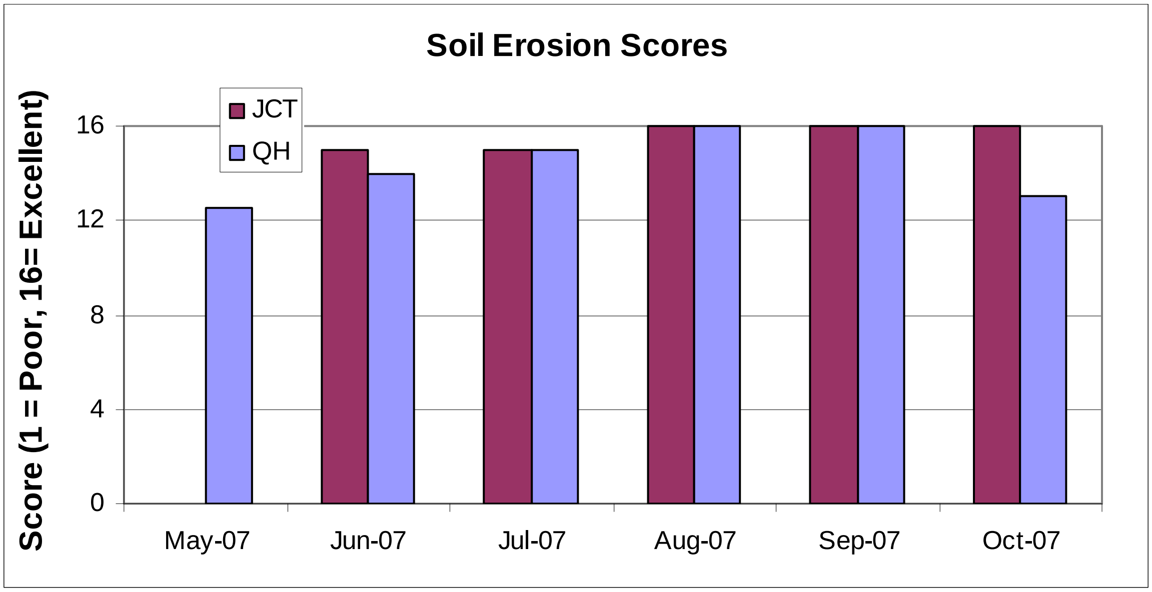
Figure 2.4 Photograph of the Quinton Heights (left) and Johnson County (right) study sites in late July 2007.

Given the differences in plant density and species composition, differences in the overall plant health scores at the two sites were statistically significant by the Student's t-test ($p = 0.047$) at the 95% confidence level. Although plant health was significantly higher at Quinton Heights, scores at the 3-year old site were lower than expected, primarily as a result of two mowing occurrences. The basin was mowed once in early June and again in early October, both of which are reflected by a drop in plant health scores. While periodic mowing can be used as an alternative to burning to maintain a healthy grass community (Diboll, 1982), inappropriate timing and frequency of mowing may adversely affect the health of prairie grasses. Studies of grasslands and restored prairies suggest mowing on an annual or semi-annual basis either in early spring or at the end of the growing season (Dale, 1984; Diboll, 1982). In addition to the less than desirable timing and frequency of mowing in the basin, both mowings left a considerable thatch layer in the basin, particularly along the central channel where grass growth is most vigorous. While a moderate mulch covering is desirable to promote infiltration, maintain plant-water relations, and help prevent erosion, excessive mulch has been found to retard emergence of regrowth, reduce the amount of biomass produced throughout the growing season, and reduce plant diversity (Weaver and Rowland, 1952). The mowing in early October was potentially damaging to the development of the grass below the ground as well; since temperatures were still warm enough to support above-ground biomass growth, the grasses may have pulled reserves from the roots to support recovery growth aboveground, thus weakening the root system (Knapp et al., 1999). Due to the potentially negative impacts of over-mowing on the growth of tallgrass species, mowing treatments are recommended either in the spring or late fall after the first killing frost (Schach et al., 1996).

Soil Erosion

Scores for soil erosion remained fairly constant at each site throughout the observation period and, although numerically higher at the Johnson County site, were not found to be statistically different ($p = 0.058$). Soil erosion scores are displayed in Figure 2.5. The Johnson County site maintained scores in the “Excellent” range, a result which can most likely be attributed to the sites’s relatively flat topography and small watershed. Erosive and depositional forces were more prevalent at the Quinton Heights site where small gullies had formed along the steep sides of the basin and incoming stormwater flows from the streets provided an abundant supply of sediment and debris to the basin. The erosional and depositional features of the Quinton Heights basin did not, however, present a great concern to the stability of the site as the grasses in the basin began revegetating the gullies and sediment deposits. Accordingly, the relationship between soil erosion indicators and vegetation health is evidenced by the increase in soil erosion scores throughout the growing period at Quinton Heights, despite heavy rainfall events that also occurred during this period. The drop in the erosional indicator score for the October observation at Quinton Heights is related to the late-season mowing of the basin, which removed vegetation along gullies and sides of the basin, thus leaving the basin more susceptible to erosion.

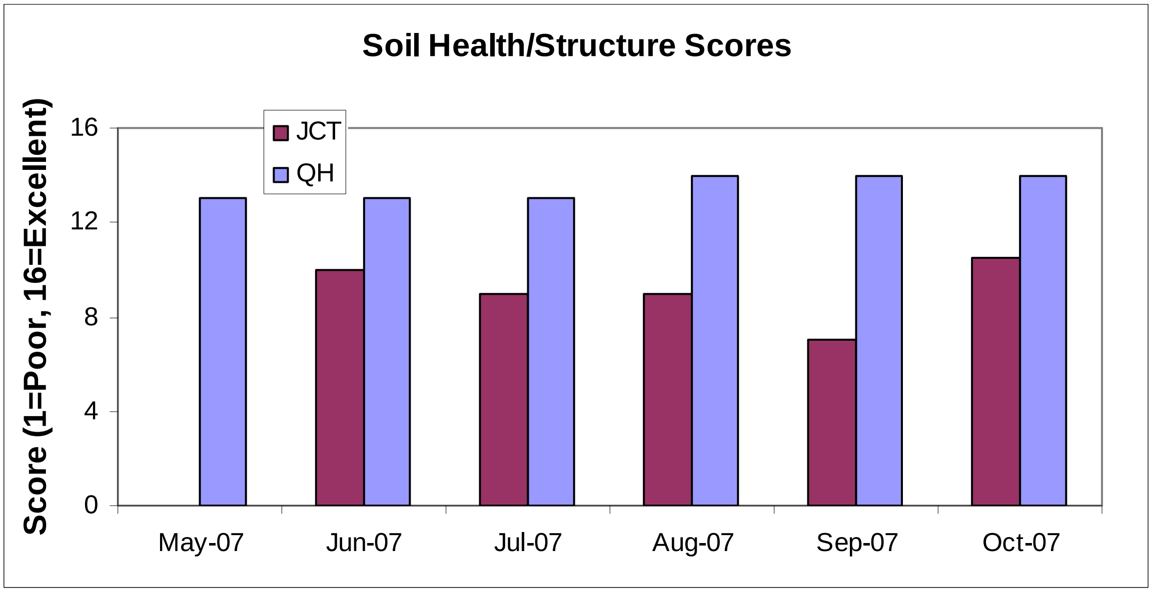
Figure 2.5 Scores for the soil erosion category at the Johnson County (JCT) and Quinton Heights (QH) sites.



Soil Health and Structure

The scores for the soil health and structure category are displayed in Figure 2.6. As seen in the figure, soil health and structure scores also remained fairly constant throughout the observation period, particularly at the Quinton Heights site where the coefficient of variation (C.V.) was 4.1%. Scores were found to be statistically higher at the Quinton Heights site ($p = 8.65 \times 10^{-5}$). The observed stability in scores was anticipated since the majority of the indicators upon which the soil health score was based- namely the depth of the organic matter layer, soil compaction, and tilth- are related to the structure of the soil and would not be expected to change significantly over a single growing season (Vogel, et al., 2001). Because the grasses at Quinton Heights were further along in their development than those at Johnson County, the rooting depth and density at Quinton Heights was greater and contributed to higher soil health scores for the site.

Figure 2.6 Scores for the soil health and structure category at the Johnson County (JCT) and Quinton Heights (QH) sites.



In comparison to the relatively constant soil health scores at Quinton Heights, greater variability was observed in the scores at the Johnson County site (C.V.= 17%). The spatial variability of soil properties across the Johnson County site, particularly with regards to the organic matter layer, is believed to have contributed to the increased variability in scores. An additional source of variability was the soil moisture at the time the assessment was completed. The soil moisture was found to affect the ease with which the wire flag, used to determine the presence of compacted soil layers, could be inserted into the soil so that drier soils appear to be more compacted. Skewed compaction scores due to soil moisture content could be avoided by conducting the health assessment when the soil moisture is about the same, for instance, 48 hours after rainfall. Other physical features could also be incorporated into the rubric to confirm the presence of suspected soil compaction, including blocky, dense soil structure over less dense soil layers or horizontal root growth (USDA NRCS, 1997). Another adjustment that could be made to the assessment developed for this study would be to measure the depth of the granular, organic matter rich layer in the upper-most region of the soil profile rather than simply compare the color of the topsoil to that of the subsoil. The depth of this layer was easily observed while with the soil core and conducting the earthworm survey. At Johnson County, the upper 1 to 3 cm of the soil profile exhibited a granular organic layer compared to the top 6 cm at Quinton Heights. Differences in the development of this layer are most likely related to the depth and activity of the roots of the grasses at each of the sites and their impact on soil structure. A quantified measure such as this would provide a better indication of changes in the organic content and structure of the soil over time than the more subjective color-comparison method.

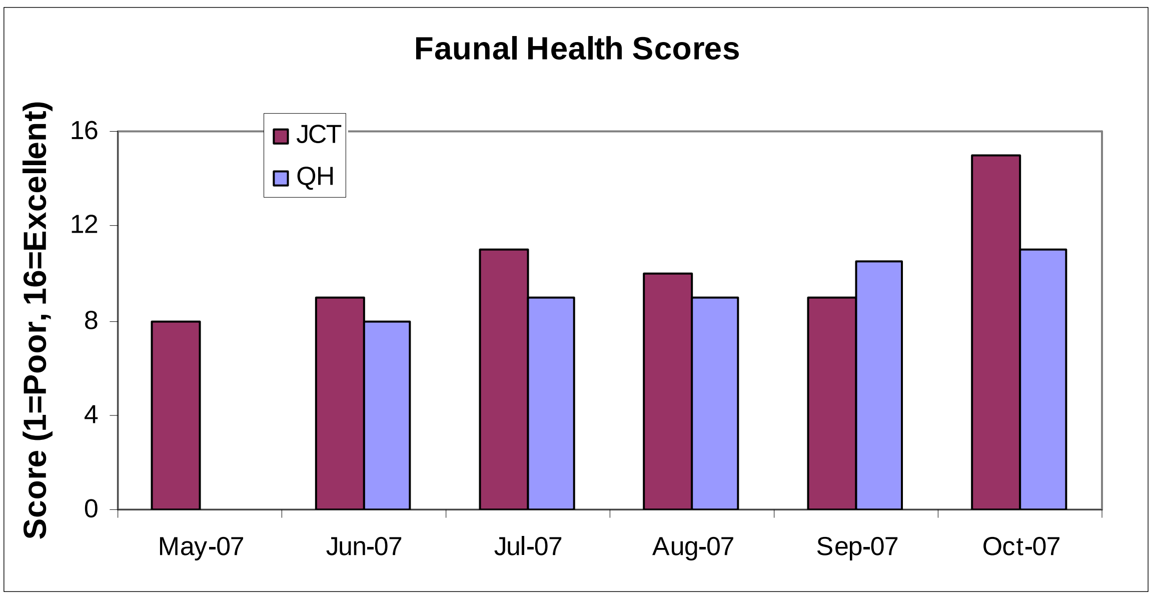
Faunal Health

Of the four categories used to assess the ecological health of the study sites, the scores for the faunal health category were the most variable over time. The inconsistent nature of scores for this category can be observed from the values displayed in Figure 2.7. This result could also be expected since the biological indicators upon which scores were based are mobile, and, unlike the grasses or soils in the system, may or may not be present at the time and location at which the assessment was conducted. Due to the variable nature of the Fauna scores at each site, the Student's t-test returned no significant differences in scores between the two ($p = 0.35$). Both sites played host to a variety of birds and insects and received scores of "Good" to "Excellent" at each assessment date. Earthworms were found at the Johnson County site in each of the three earthworm sampling trials, but none were observed at the Quinton Heights site. The earthworms recovered at the Johnson County site were predominantly of the epigeic or endogeic variety and resided near the soil surface. Because the earthworm species encountered at the site were found near the surface, the earthworm sampling procedure was adjusted to more effectively account for the earthworms present in the sampling area. During the sampling trials, it was found that more worms could be found when the sampling area was cleared of vegetation and the dry mustard solution was applied directly to the soil surface prior to excavation. After sufficient time was allowed for shallow-dwelling earthworms to navigate to the surface, the area was excavated to the 20-cm depth, hand-sorted, and the dry mustard solution was then applied to the bottom of the excavated area. Earthworm densities at the Johnson County site ranged from 15 to 25 individuals per square meter, which is in line with values reported in the literature for recently developed urban lawns (Smetak, 1998). The absence of earthworms from the sampling trials at the Quinton Heights sites does not preclude the presence of earthworms in the basin. It does, however, suggest that a migrational barrier or other condition may exist to hinder the establishment of a substantial earthworm population in the basin.

As explained previously, the Fauna category also accounted for the presence of nondesirable species. The primary organism of concern to stormwater systems is the mosquito.

Throughout the study period, only one mosquito was observed at the Johnson County site, and its presence there was not necessarily linked to the site itself. Mosquito larvae were, however, a regular inhabitant of the stormwater system at Quinton Heights. The presence of mosquitoes at the site can be attributed to ponded water which remained standing in the central channel of the basin for several days after rainfall and it provided a place for the larvae to grow. Mosquito larvae could be observed in this channel and were occasionally present in water samples taken at the outlet of the system. Despite the presence of mosquito larvae at the site, adult mosquitoes were never observed, even during lengthy visits to the basin to conduct infiltration tests. Full development of the larvae to adult mosquitoes was probably somewhat controlled by the drying out of the channel before larvae emerged or by washout of the larvae during storms.

Figure 2.7 Scores for the faunal health category at the Johnson County (JCT) and Quinton Heights (QH) sites.



CONCLUSIONS

One of the challenges to the successful implementation of ecologically-engineered stormwater systems is their continued maintenance and monitoring after installation. The application of an easy-to-use ecological health rubric could help in making post-installation monitoring efforts more successful. Although qualitative in nature, the assessment rubric developed in this study can still provide valuable information about a stormwater system's condition and potential performance, especially when quantitative data is not available. The health assessment developed in this study was applied to two urban stormwater sites planted with native prairie grasses. One of the sites, Quinton Heights, was in its third season of operation and approaching vegetative maturity. The second site, Johnson County, had just been converted from a traditional turf lawn to prairie grasses at the start of the study period. Although the scores for the overall ecosystem health were generally higher at the more mature Quinton Heights site, differences between the two sites were not found to be statistically significant. Statistically significant differences in the scores between the two sites were observed for the plant health, soil erosion, and soil health and structure categories. The most evident difference between the two sites was in the soil health and structure category. The significantly higher soil health scores at the Quinton Heights site are attributed to greater root development and density, which have in turn contributed to soil tilth and structure at the site. One of the most obvious visual differences between the two sites was the vegetation. The vegetative community at Quinton Heights, which was composed primarily of perennial tallgrass prairie species, received higher scores than the Johnson County site, which was dominated by weedy annuals. Still, the difference between vegetation health scores were not as high as expected. Lower than anticipated vegetation health scores at the Quinton Heights site are attributed to two mowing events in which the grasses were cut very short and a thick layer of potentially growth-inhibiting grass clippings was left.

Ecological Implications to Stormwater Management

The function and sustainability of ecologically-designed stormwater systems depends in part on the maintenance of ecosystem processes. Metrics used to assess the performance of ecologically-designed systems are incomplete apart from the consideration of the health of the ecosystem. Maintaining a healthy ecosystem has important implications for the long-term management of ecologically-designed systems. For example, in the case of the integration of a tallgrass prairie ecosystem with stormwater management, a proper burning or mowing regime should be followed to ensure the continued health of the vegetation, and thus the root structures which positively impact soil infiltrative properties. If a mowing regime is adopted, care should be taken to mow either early in the growing season or at its conclusion, and to remove at least part of the grass clippings to make way for regrowth.

Despite the lack of statistical significance, the trend in overall health scores at the sites supports the hypothesis that ecological health of a site improves over time. These improvements are likely to be most evident over the span of years rather than a single growing season. The time to establish a healthy, functioning ecosystem also has important implications for the design of ecologically-based stormwater systems. Unlike traditionally constructed systems, ecologically-designed systems will not be fully functional at the time of their installation, but will have the potential to improve over the first few growing seasons.

The next step in this research is to establish a correlation between the ecological health score and the actual hydraulic and pollutant removal functions of the system. Although hydraulic and water quality data were collected at the Quinton Heights basin, the amount of data collected was not sufficient to make any firm connections between the ecological health score from the rubric developed for this study and the true functionality of the system. In order to make this ecological assessment tool more useful to the stormwater management community, the rubric will be applied over the next few growing seasons at these sites. Coupling the health scores with continued hydraulic and water quality monitoring data will allow closer comparisons between ecological health and system function to be established. Therefore, practicality of using an ecological health rubric to rapidly assess the potential of the system to fulfill its intended hydraulic and water quality functions may be determined.

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CHAPTER 3 - Ability to reclaim ecological infiltration processes in urban environments through prairie restoration

Introduction

The success of Low-Impact Development (LID) strategies in mitigating the impacts of increased runoff volume and flow rate following urbanization hinges upon infiltration. In order to meet the goals of LID, it is important to understand the processes and properties that govern infiltration. This chapter discusses these properties and their application in infiltration-based stormwater management in eastern Kansas and in urban areas throughout the historical extent of the tallgrass prairie region.

Properties Governing Soil Infiltration

Infiltration in soils has long been a topic of interest. Henry Darcy set the stage with his column studies of the movement of water through saturated soils in the 1850s (Kirkham, 2005). Several decades later, Horton (1940) suggested that infiltration rates in soils were a function of both the physical and biological properties of the soil, as well as the vegetation type and land use. Physical soil properties that govern infiltration include soil texture, structure, bulk density, and the size and distribution of pores. In an extensive review of the literature published concerning hydrologic soil properties, Rawls et al. (1982) provides a summary of values reported for saturated hydraulic conductivity based on soil texture. In general, soils classified as sands or loams exhibit the highest hydraulic conductivities as a result of the coarser particle size of these soils. Silty clay loams, including the soils and subsoils commonly found in Eastern Kansas (USDA SCS, 1970), are typified by saturated hydraulic conductivities in the 0.15 cm/h range, or over 100 times less than rates expected for sandy soils (Rawls, 1982). In addition to texture, the soil structure is also important in determining potential infiltration rates. Soil structure refers to the arrangement of soil particles. In the interest of infiltration, aggregated soils, or those in which individual grains are flocculated together through a combination of physical, biological, and chemical processes in the soil, are the most desirable (Hillel, 1998). The size and interconnectedness of the pores between soil aggregates are important components of soil structure. The diameter of these pores typically range from 0.2 μm to 10 mm. While smaller pore sizes are important for water retention in the soil, pores 75 μm or more in diameter, referred to as macropores, are most important in consideration of the infiltration process (Kirkham, 2005; Linden et al, 1991). Soil bulk density is also related to soil structure. Defined as the ratio of the mass of soil solids to the total soil volume (Hillel, 1998), bulk density is typically inversely related to infiltration rates (Holman-Dodds, 2006). This relationship can be explained by the reduced pore space typical of soils with high bulk densities, which inhibits the transmission of water through the soil.

Beyond the physical properties of the soil itself, the biological components of the ecosystem can have profound impacts on infiltration characteristics. Higher rates of infiltration on vegetated areas versus bare ground cover have been well documented (Holman-Dodds, 2006, Perrygo et al. 2001; Weaver and Rowland, 1952). Infiltration gains observed in the presence of vegetative cover have been attributed to changes in soil properties as a direct result of interactions between soil and plant roots. As discussed in the previous chapter, root penetration contributes to those soil properties which enhance infiltration, namely increased soil porosity and the development of stable soil aggregates. Roots continue to enhance soil infiltration properties as they decay by increasing the organic matter content and creating macropores through which high infiltration rates have been observed due to preferential flow (Linden et al., 1991). In addition to enhancing soil infiltrative properties through macropore development, actively respiring vegetation also alter the soil moisture regime by using water stored in the soil for transpiration. In effect, the antecedent moisture content of the soil prior to the next precipitation event will be lower in soils with vegetation than in those without, increasing initial infiltration rates and prolonging the time to soil saturation (Hino et al., 1987).

In addition to vegetation, other biological forces at work within the soil alter infiltration characteristics. Earthworms are among the most influential of the soil biota with respect to infiltration (Edwards, 2004). In addition to contributing to the formation of infiltration-promoting soil aggregates, earthworms are a major source of biological macropores through which significantly higher infiltration rates have been observed (Linden, et al., 1991). Microbial activity, which is promoted by exudates released by plant roots in the root zone, also plays a role in improving soil structure and, thus, infiltration properties (Holman-Dodds, 2006).

From the preceding discussion, the intimate tie between both the physical and biological soil environment in the infiltration process is evident. As such, both should be considered when designing infiltration-based stormwater management systems. In much of the Midwest United States, where the tallgrass prairie ecosystem historically predominated, the soil environment that formed under the cover of tallgrass prairie species was ideal for rapid infiltration. Although many of the soils in the region are classified as silty-clay loams, high rates of infiltration have been observed in prairie systems. This observation attests to the role of vegetation, particularly of deeply-rooted prairie grasses, in improving soil infiltration characteristics. Prairie soils boast well-developed soil aggregates and rich organic matter layers to enhance movement of water into and through these soils (Jastrow, 1987).

Impacts of Urbanization on Soil Infiltrative Properties

Soil infiltration properties, which have been developed through hundreds of years of root growth and decay, can be lost quickly. The changes in soil structure and infiltration processes following the conversion of virgin or long-term grassland to agricultural cropland have been well documented. Multiple studies report significant losses of water-stable aggregates as well as changes in the distribution of aggregate size classes (Jastrow, 1987; Mohanty et al., 1991). Urbanization, likewise, alters the soil structure and significantly reduces the infiltrative capacity of the landscape. One of the most obvious causes for reductions in infiltration as an area is converted to urban uses is the addition of streets, parking lots, buildings, and other impervious surfaces to the landscape. Compaction during construction activities has also been identified as a leading source of infiltration reductions associated with urbanization. Compaction restricts water and air movement through the soil by increasing the bulk density, decreasing porosity, and forcing a smaller pore size distribution (Gregory, et al., 2006). Both the deliberate compaction of a site to increase the structural strength of the soil to support subsequent developments and unintentional compaction caused by the use of heavy equipment and lot grading reduce infiltration in soils. In a study of the effects of compaction on both wooded and pasture lots before and after construction, Gregory et al. (2006) reported significant reductions in infiltration rates post construction activities. Furthermore, infiltration loses occurred regardless of the intensity of compaction; infiltration rates were reduced whether light or heavy compaction occurred.

In addition to soil compaction that results as heavy equipment is driven over development sites, typical construction processes involve scraping the top 10 cm (3.9 in.) of soil from the entire site prior to construction. The process of scraping, flipping, and refilling soil during development results in infiltration rates that lag far behind predevelopment rates even after vegetation is reestablished (Gregory, et al., 2006).

Infiltration-based Stormwater Management

As part of the effort to mitigate the increase in runoff flow rate and volume following urbanization, infiltration-based stormwater management strategies are being adopted across the nation. In eastern Kansas, some ecologically-engineered stormwater systems have been designed after the landscape that has proven to be one of the most effective in infiltrating precipitation in the ecoregion: the tallgrass prairie. The prairie species native to the region offer several infiltration-related advantages over the shallow-rooted turf grasses introduced in most urban open spaces. Infiltration studies conducted by Jastrow (1982) suggest higher infiltration rates under warm season, prairie grasses than under cool season grasses such as brome. Similarly, Perrygo et al. (2001) reported higher infiltration rates associated with Eastern Gamma grass, a native of the tallgrass prairie, than with fescue, a cool-season pasture grass that has also been adapted for lawn plantings. The extensive root systems of prairie grasses are believed to contribute to higher infiltration rates under these grasses. In contrast to the shallow root systems of turf grasses, which are typically concentrated in the upper 10 to 15 cm due to frequent mowing (Qian and Follett, 2002), the fibrous root systems of tallgrass prairie species can penetrate to depths exceeding 1.2 m (4 ft) (Weaver and Zinc, 1946), thus improving the hydraulic properties of a greater portion of the soil profile. However, regaining predevelopment infiltration processes following development is not as easy as simply restoring prairie vegetation. The soils that originally developed under the prairie formed over hundreds of years, and regaining full ecological function following disturbance and subsequent restoration of prairie grasses may require years. Although substantial increases in infiltration rates have been observed within a few months of the establishment of grasses with extensive rooting systems in mesocosm studies (Hino et al., 1987; Culbertson and Hutchinson, 2004), field-scale studies have reported periods of 30-50 years to regain soil infiltrative properties after revegetating disturbed soil (Jastrow, 1989).

Regardless, incorporating native tallgrass prairie species into stormwater systems poses an attractive option for ecological stormwater management in the Midwest. Still, important questions remain to be answered. To what extent will land restored to prairie recover its ability to infiltrate? Can infiltration gains obtained through the root structure of these grasses supersede the need for engineered soil media- a potentially cost-prohibitive means of increasing infiltration rates, especially over large areas, in stormwater systems? The objective of this study was to examine the potential to enhance infiltration through the incorporation of prairie grasses into stormwater systems and to explore how the design of these systems could capitalize upon infiltration gains to further reduce peak runoff rates and volume.

Methods and Materials

Site Descriptions

To investigate how infiltration is effected by the reintroduction of prairie grasses in urban sites, two study sites were selected. The first site was located at the Johnson County Transit Center in Johnson County, Kan. The yard surrounding the transit center was converted from a traditional fescue turfgrass to a mix of mid-height prairie grasses, including hairy grama (*Bouteloua hirsute*) and sideoats grama (*Bouteloua curtipendula*), in June 2007 as a stormwater management demonstration site. The native grass mix was seeded directly into the lawn after killing the fescue with Roundup earlier in the spring. Therefore, the soil at the site had not undergone significant disturbance since the construction of the transit center in 2001. Since this site was in its first growing season at the time of the study, it represented a baseline with regards to vegetation development.

The second site was a retention basin located in a medium-density neighborhood in Topeka, Kan. and will be referred to as Quinton Heights after the name of the neighborhood in which it is located. The Quinton Heights basin occupies a 1,500 square meter area and was constructed in 2004 to relieve flooding in the neighborhood. The basin and its drainage area are depicted in Figure 3.1 with flow boundaries overlaid as depicted in the drainage plan produced by the Topeka Public Works (Spaar, 2004). The majority of the 6000-m^2 (1.5-acre) watershed draining to the basin inlet is paved, consisting of a 1,600-m^2 (0.4-acre) parking lot and an additional 1,000 m^2 of streets. Runoff from the parking lot in the southern most portion of the watershed enters a subsurface sewer network and is delivered to the basin's 45.7 cm (18 in.) diameter inlet pipe. Runoff from the rest of the drainage area runs down the street on the eastern border of the basin and enters the basin area through a grated opening placed in the middle of the street. In addition to the 6,000 m^2 that drains to the basin's inlet, an adjacent hillslope and residence to the south of the basin also contributes runoff to the basin via overland flow. After excavation, the basin was replanted with grasses native to the tallgrass prairie, including big bluestem (*Andropogon gerardii*), indiangrass (*Sorghastrum nutans*), switchgrass (*Panicum virgatum*), and side-oats grama. A shallow, sinuous channel was constructed through the center of the basin to link the inlet to the outlet. The total length of the channel was approximately 35.5 m and its average width was 2 m. For most storms, runoff delivered to the basin through the inlet pipe is contained in this channel; overflow to the remainder of the basin occurs during larger storms. Since this channel remained wetter than the rest of the basin following precipitation, vegetation growth was most vigorous here and was dominated by more hydric plants, including switchgrass, prairie cordgrass (*Spartina pectinata*), and barnyard grass (*Echinochloa crusgalli*). At the time of the study, the vegetation in the basin was in its fourth growing season and, as such, was nearing the time frame at which the vegetation's rooting structure is approaching maturity as suggested by Weaver and Zinc (1946).

Figure 3.1 Aerial view of Quinton Heights basin, outlined in yellow, and its attendant watershed. Drainage boundaries, areas, and curve numbers were taken from the drainage mapping produced by the Topeka Public Works Department (Spaar, 2004). Image from Google Earth (retrieved March 31, 2008).

Although the Quinton Heights and Johnson County sites represent different stages of vegetative maturity, they are both relatively young from the viewpoint of soil development. In consideration of the question of our ability to reclaim ecological infiltration processes in urbanized sites through prairie restoration, the saturated infiltration rates from the Quinton Heights and Johnson County sites were compared with those measured in a more established native grassland system. Native grass buffer strips were established at the Fort Riley Military Installation in Fort Riley, Kan. in 2001 (Ramirez, 2006). Prior to this date, the site had been grazed until the 1960s when it was procured by Fort Riley and used for training maneuvers. The vegetative community and soil types at the Fort Riley site were very similar to those at Johnson County and Quinton Heights.

The soil texture at the Johnson County and Quinton Heights sites were determined by the hydrometer method at the Kansas State University Soils Testing Lab. The soil composition at the Fort Riley site was taken from the textural classification previously conducted at the site (St. Clair, 2007). The soils at all three sites were classified as silty clay loams (Table 3.1).

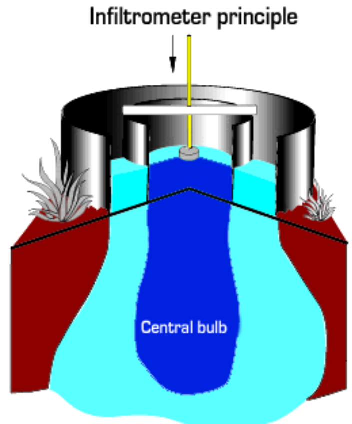
Table 3.1 Soil composition at study sites.

Site	Sand	Silt	Clay	Textural Classification
Johnson County	13	54	33	Silty clay loam
Quinton Heights	15	55	30	Silty clay loam
Fort Riley	18	50	32	Silty clay loam

Infiltration Tests

Double-ring infiltrometers were used to measure steady-state infiltration rates at the Johnson County and Quinton Heights sites on a monthly basis from May to October 2007, with one measurement in late March at the Johnson County site. Infiltration measurements at the Fort Riley site were conducted in 2006 by Amy St. Clair as part of her M.S. research at Kansas State University using the same method (St. Clair, 2007). The double-ring infiltrometer (Figure 3.2) is commonly used for the field measurement of infiltration rates. As seen in the figure, a constant head is maintained so that a bulbous wetting front forms under the outer ring. The water level in the inner ring is allowed to draw down and is measured at regular time intervals to determine the rate of water movement into the soil. Although the bulb of water that infiltrates below the outer ring moves both laterally and horizontally, it promotes one-dimensional, vertical flow beneath the inner ring (ASTM, 2006).

Figure 3.2 Photograph of double-ring infiltrometer used in study (on left) and illustration of the bulbous wetting front and central bulb that develop under a double-ring infiltrometer (on right, from Ramirez, 2006).



The infiltrometers used in this study were constructed of 0.64-cm (0.25-in.) thick steel pipe. The outer rings were 16.5 cm (6.5 in) tall with an inner diameter of 34.3 cm (13.5 in). The inner rings stood 35.5-cm (14-in) tall with an inner diameter of 20.3 cm (8 in). A 4.5-kg (10-lb) sledge hammer was used to drive the rings to a depth of 7.6 cm (3 in.) below the ground surface. All vegetation within the infiltrometer area was left in place. The ends of both the inner and outer rings were beveled in order to minimize disturbance as the rings were driven into the ground. After placement, the outer ring was filled with tap water to the top of the ring. This water level was maintained throughout the infiltration measurement period. Next, the inner infiltrometer was filled by pouring tap water into a bucket with holes emanating radially from its center to distribute the inflowing water over the area of the inner ring and minimize surface compaction due to a concentrated stream of water. The water level inside the inner ring was measured with a ruler at 10-minute intervals until the movement of water into the soil profile reached an approximately steady rate (usually three to four hours, depending upon the initial soil moisture). In between measurements, the inner ring was covered to prevent evaporation from contributing to infiltration rate measurements. The inner ring was refilled as needed, typically after the depth of water in the ring had fallen to 15 cm.

Infiltration tests were conducted in replicates of three. Infiltrometer locations were selected to best represent the average conditions at the testing site. To determine the saturated, steady-state infiltration rate, the cumulative depth of water infiltrated was plotted against time.

The slope of this line represents the steady-state infiltration rate, which is taken to be the effective saturated hydraulic conductivity (K_{eff}). A Student's t-test was performed in Microsoft® Office Excel® to compare the mean saturated infiltration rate measurements at each site and to determine if differences in measured rates were significant. One-way analysis of variance (ANOVA) tests were performed to test for significant differences among the three sites- the two stormwater sites and the Fort Riley native grass filter strips- with Microsoft® Office Excel®. A significance level of $\alpha = 0.05$ was used for both ANOVA and Student's t-tests.

Water Budget

Infiltration tests with the double-ring infiltrometer allow estimations of the infiltration rate under saturated conditions. The amount of time from the onset of precipitation until saturated conditions are reached depends on the initial moisture content of the soil profile. Although daily soil moisture was not monitored throughout the study period, it was estimated using a water balance spreadsheet developed for the Konza Prairie Biological Station (KPBS) in Riley County Kansas (Hutchinson et al., 2008). The spreadsheet calculates the volumetric soil water content using a check-book accounting method. The method assumes that the maximum water content corresponds to field capacity, which is approximately 30% by volume for silty clay loam soils. The soil water content is a function of the evapotranspiration (ET) rate calculated using the modified Penman-Monteith method (Ham, 2000), a warm season grass water usage coefficient (Hutchinson et al., 2008), and the amount of water applied via precipitation or irrigation. While this spreadsheet is calibrated for the soil and vegetation at the KPBS, the soil, vegetation, and climatic conditions at the Topeka study site were assumed to be similar enough to obtain reasonable estimates of the soil water content. Estimates of soil-moisture from the check-book accounting method were compared with field-measured values of gravimetric moisture content, which were taken at the same time as the infiltration measurements, in order to check the accuracy of values predicted from the water balance spreadsheet. Gravimetric moisture content was determined by oven-drying soil samples taken from the top 8-cm of the soil profile (excluding the uppermost layer of organic matter) for 24 hours at 105°C. Samples were taken in replicates of three using the standard soil core method (ASTM, 1992).

Based on daily estimates of the actual ET for prairie grasses calculated with the check-book balance method, the difference between the water use by the grass in a fully-watered situation and under the actual precipitation that fell was determined. This difference is of interest to stormwater system design because it indicates the available water storage in the soil profile. It is believed that deep-rooted prairie grasses will deplete the moisture stored in the soil more rapidly than a traditional manicured urban landscape through ET, thus increasing the storage capacity of the system before the next precipitation event. The difference between the fully-watered situation, which represents a potential maximum depth of water that the grasses could use, and the actual rainfall was used to make recommendations as to the potential ponding height that could be maintained at the Quinton Heights basin to further reduce stormwater volume outflow while allowing all ponded water to infiltrate or be lost through ET within 48-hours.

Hydraulic Monitoring

ISCO automated water samplers (Model 6712, Isco, Inc.) were installed at both the inlet and outlet of the Quinton Heights basin to measure the flow rate of stormwater entering and exiting the system. The inlet consisted of a 46-m (18-in.) diameter concrete pipe, while flow at the outlet was controlled by a 90-degree V-notch weir. Flow was determined with an ISCO bubbler module (Model 720, Isco, Inc.), which measures the height of the stormwater above the bubbler tube through a pressure transducer located inside the flow meter (Grant and Dawson, 2001). The ISCOs were programmed to take a depth-measurement reading every five minutes. To relate the height measured by the ISCO to a flow rate, discharge equations for a pipe (at the inlet) and a free-flowing 90-degree V-notch weir (at the outlet) were used. Flow, Q, (in L/s) through the circular inlet pipe was calculated by multiplying the Manning's Equation (Equation 3.1) for velocity by the area of the pipe:

$$Q = \frac{1000}{n} R^{2/3} S^{1/2} A \qquad \text{(Equation 3.1)}$$

Here, n is Manning's roughness coefficient (selected as 0.012 to correspond with the value for concrete), R is the hydraulic radius in meters, S is the slope of the pipe (measured as 0.035 m/m), and A is the cross-sectional area of the flow through the pipe in square meters (Haan et al., 1994). Discharge at the outlet was calculated using the weir equation (Equation 3.2).

$$Q = KH^{2.5} \qquad \text{(Equation 3.2)}$$

Here, K is a constant dependent upon the angle of the weir (90°) and the units of Q (L/s), and was taken to be 1380 as reported by Grant and Dawson (2001). H is the head above the weir in meters (Grant and Dawson, 2001) and was taken as the depth measurement recorded by the ISCO at the outlet.

Runoff hydrographs for the inlet and outlet were constructed using the calculated flow rates. The total volume of stormwater entering and exiting the system for a given storm was calculated by integrating and summing the areas under the hydrograph for each 5-minute period.

Rainfall at the site was recorded with a HOBO data logging rain gauge (Model RG2M, Onset). In addition to recording the total amount of rain that fell during a given event, data from this tipping-style rain gauge was used to construct site-specific rainfall hyetographs for each storm.

Results and Discussion

Infiltration

Measured saturated infiltration rates at the Topeka and Johnson County study sites are displayed in Table 3.2. Measured effective saturated infiltration rates vary considerably across replicate measurements taken on the same day. The seasonal average was 4.4 cm/hr with a standard deviation of 7.7 cm/hr and 3.7 cm/hr with a standard deviation of 6.0 cm/hr for Quinton Heights and Johnson County, respectively. Such variability was expected with the double-ring infiltrometer method (ASTM, 2006) and was consistent with the variability reported by other infiltration studies using similar methods (Gregory et al., 2006; Holmann-Dodds, 2006). The variability in measurements was probably due to differences in soil composition and structure at the site. Soils in urbanized areas- including those at the Quinton Heights and Johnson County Transit Center study sites- have been scraped, inverted, and otherwise disturbed and can exhibit unexpected subsurface heterogeneities. For example, the impact of soil heterogeneity on observed infiltration rates was evident at the Quinton Heights study site during the infiltration measurements taken on May 31, 2007. The measured value of 32.8 cm/hr was taken along the central channel in the basin where the upper soil layer gives way to a shallow gravel pack which enhanced drainage through the channel. The uncharacteristically high infiltration rate of 22.5 cm/hr measured on September 28, 2007 at the Johnson County site was thought to have resulted from inadvertently placing the infiltrometer over an underground sprinkler line. While such rapid infiltration rates are not characteristic of the entire site, they do attest to the heterogeneity of the soil profile.

Table 3.2 also contains effective saturated infiltration rates measured along native grass filter strips at the Fort Riley military base. As noted earlier, the Fort Riley site offers a comparison between infiltration rates measured at Quinton Heights and Johnson County with those measured at a more mature site. As seen from the calculated standard deviations and coefficient of variance, the effective saturated infiltration rates are substantially more consistent at Fort Riley than at either of the less mature stormwater sites.

Table 3.2 Effective saturated infiltration rates measured at the Quinton Heights and Johnson County stormwater sites and along grass filter strips at Fort Riley military base.

QUINTON HEIGHTS						
	Infiltration Rate, cm/hr					
Date	Replicate 1	Replicate 2	Replicate 3	Average	Std. Dev.	%CV
5/31/2007	32.8	7.9	1.7	14.1	16.4	116
7/20/2007	1.5	2.8	1.3	1.9	0.8	41.7
8/8/2007	8.7	2.3	1.9	4.3	3.8	89.0
9/7/2007	1.5	0.8	0.5	0.9	0.5	57.0
10/2/2007	1.9	0.2	9.8	4.0	5.1	129
10/17/2007	0.4	2.4	0.1	0.9	1.2	134
Seasonal Averages				4.4	7.7	176

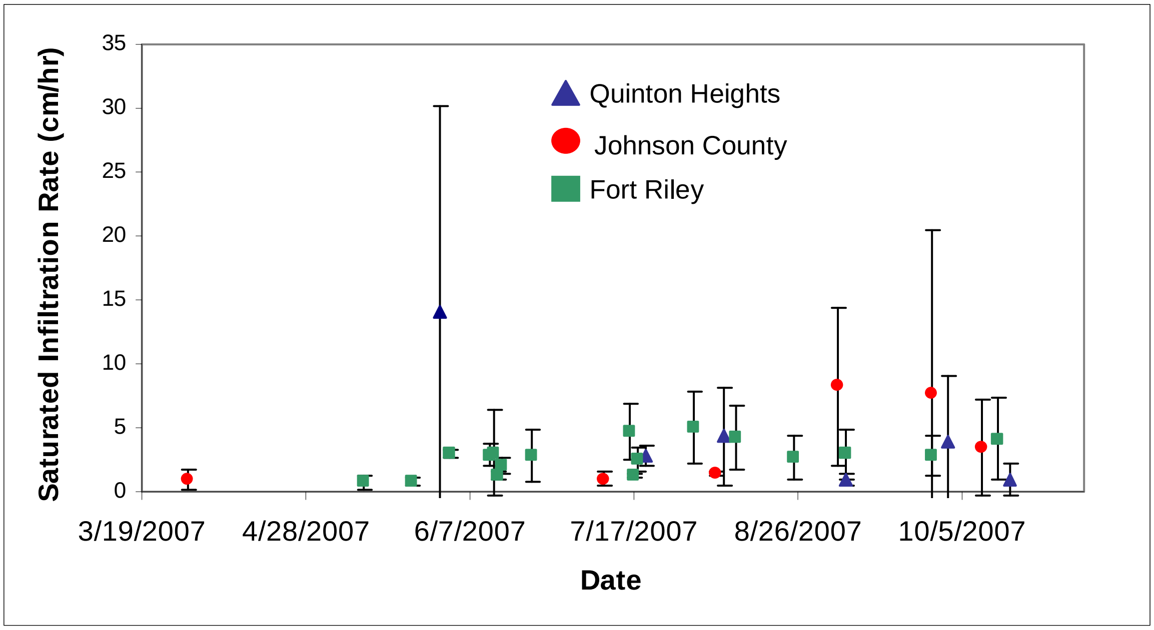
JOHNSON COUNTY						
	Infiltration Rate, cm/hr					
Date	Replicate 1	Replicate 2	Replicate 3	Average	Std. Dev.	%CV
3/30/2007	1.8	0.9	0.2	1.0	0.8	80.7
7/10/2007	0.9	1.6	0.6	1.0	0.5	52.6
8/6/2007	1.5	1.3	1.3	1.4	0.1	9.7
9/5/2007	1.9	14.2	8.7	8.3	6.2	74.8
9/28/2007	0.1	0.2	22.5	7.6	12.9	170
10/10/2007	7.7	2.5	0.4	3.5	3.8	107
Seasonal Averages				3.8	6.0	158

FORT RILEY						
	Infiltration Rate, cm/hr					
Date	Replicate 1	Replicate 2	Replicate 3	Average	Std. Dev.	%CV
5/12/2007	0.9	0.1	0.8	0.6	0.4	68.6
5/24/2007	0.8	1.2	0.5	0.8	0.4	43.4
6/2/2007	3.2	2.7	3.0	3.0	0.3	8.5
6/12/2007	3.0	3.7	1.9	2.9	0.9	31.7
6/13/2007	6.9	1.2	1.0	3.0	3.4	110
6/14/2007	1.6	1.2	1.0	1.3	0.3	24.1
6/15/2007	2.0	2.7	1.5	2.1	0.6	29.2
6/22/2007	3.9	5.8	3.4	4.4	1.3	29
7/16/2007	6.7	4.7	1.9	4.4	2.4	54.4
7/17/2007	1.2	1.2	1.3	1.2	0.1	4.7
7/18/2007	3.5	2.1	1.8	2.5	0.9	36.8
8/1/2007	1.8	6.3	7.0	5.0	2.8	56.1
8/2/2007	1.8	1.6	1.0	1.5	0.4	28.4
8/4/2007	1.0	0.3	0.4	0.6	0.4	66.8
8/10/2007	9.4	2.4	3.8	5.2	3.7	71.2
8/11/2007	6.8	1.8	4.2	4.3	2.5	58.6
8/25/2007	4.6	1.4	2.1	2.7	1.7	62.3
9/7/2007	2.7	2.4	0.7	1.9	1.1	55.8
9/28/2007	1.9	4.2	1.2	2.4	1.6	64.5
10/7/2007	1.0	1.6	0.5	1.0	0.6	53.1
10/14/2007	7.4	1.1	3.8	4.1	3.2	77.1
Seasonal Averages				2.6	1.4	49.3

Figure 3.3 provides a more graphic comparison of the saturated infiltration rates measured at the sites. Here, the average rate (shown by the symbol) and ±1 standard deviation (shown by the bar) for each infiltration test are shown. Between the two stormwater sites, the overall mean infiltration rate was higher at Quinton Heights. However, mean saturated infiltration rates between the two sites were not statistically different as determined by the Student's t-test ($p=0.82$) and as is visually demonstrated by the overlap of the error bars (Figure 3.3). Differences between infiltration rates at the sites remain insignificant even when extreme infiltration measurements are removed from the data set.

The ANOVA analysis of mean saturated infiltration rates between the relatively young study sites (Quinton Heights and Johnson County) with the more ecologically-mature site at Fort Riley also yielded no significant difference ($p=0.26$). However, as can be seen from the graph and in Table 3.2, the variance of infiltration measurements at the Fort Riley site is much lower than that observed at either of the urban stormwater sites. The lack of significant differences among the infiltration rates measured at the three sites supports previous research stating that reclamation of infiltration processes following soil disturbance may require several decades (Jastrow, 1987). Reduced variability at the Fort Riley site could indicate that older, more mature grassland systems have a broader impact over the entire area as the root system is denser in both vertical and lateral directions. It should be noted that the saturated infiltration rates measured at all three sites were well above the predicted rate of 0.15 cm/hr for silty clay loams as predicted by Rawls (1982) based on soil properties alone.

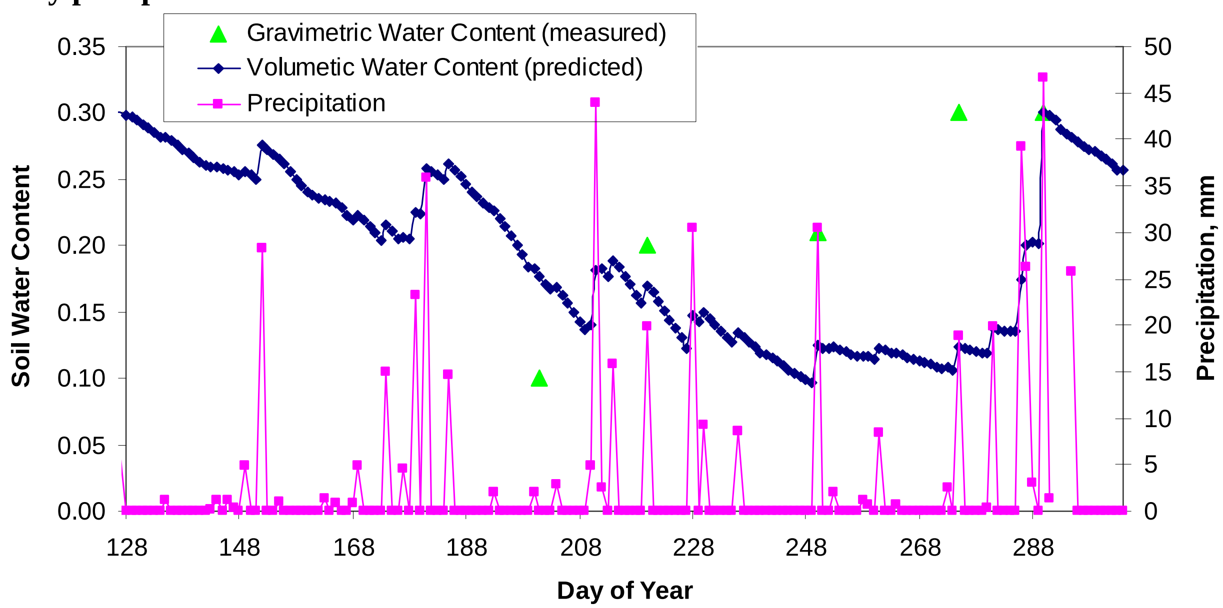
Figure 3.3 Comparison of effective saturated infiltration rates measured at stormwater study sites (Quinton Heights and Johnson County) with those measured on established native grass filter strips, Fort Riley Military base.



Water Budget and Hydraulic Analysis

Based on the precipitation recorded at the Quinton Heights basin, soil moisture was predicted over the duration of the growing season (May 8, 2007 to Oct. 23, 2007) using the check-book balance method developed for tallgrass prairie at the KPBS. Predicted values for volumetric soil moisture, along with the daily precipitation values, are displayed in Figure 3.4. Field-measured values of gravimetric moisture content are also overlaid in the figure. Ideally, the offset between the measured gravimetric moisture content and the predicted volumetric moisture content would be constant, representing the bulk density of the soil. However, as can be seen in the figure, a constant offset does not exist. The first measurement, taken the afternoon following a light, 2-mm shower, fell below the predicted volumetric measurement by a factor of 1.77, indicating a bulk density of 1.77 g/m^3. This value is much higher than expected; soil bulk density typically ranges from 1.2 to 1.3 g/m^3 (Hillel, 1998). The remainder of the gravimetric soil moisture measurements, all of which were taken after precipitation events ranging from 20 to 45 mm, were well above predicted values. This result implies that the model under-predicted soil moisture for the Quinton Heights basin. Under-prediction by the model was possible for several reasons. First, the upper 10 cm of soil in the basin was underlain by a thick clay layer which would impede water movement through the soil, thus preventing the upper soil layer from draining as rapidly as the deeper soils for which the model was developed. Secondly, because the basin received runoff from its 7,900-m^2 watershed, more water actually entered the basin that was accounted for in the model, which considered only precipitation. Differences between measured and predicted moisture levels were probably accentuated by the fact that the majority of gravimetric measurements were taken following precipitation events. This is perhaps one of the reasons that the first measurement, which was taken following a relatively dry period with only light rain showers, was much closer to predicted values.

Figure 3.4 Comparison of volumetric water content as predicted by the Konza Prairie Biological Station spreadsheet model and actual measured gravimetric water content. Daily precipitation is also included.



Although the values for soil moisture shown in Figure 3.4 do not represent actual field-measured values, they do provide a new vantage from which to examine the hydrologic data collected at the Quinton Heights basin. Combining the predicted soil moisture values with the HOBO rain gauge data provides a better understanding of the hydrologic response of the basin to individual storm events. A summary of the hydraulic data collected at the site- including precipitation, flow volume into and out of the basin, and the peak inlet and outlet flow rates- are presented alongside modeled values of antecedent moisture content in Table 3.3. The percent water retained by the basin was also calculated based on the total volume of water delivered to the basin (rainfall plus inflow) and the volume of water discharged through the outlet.

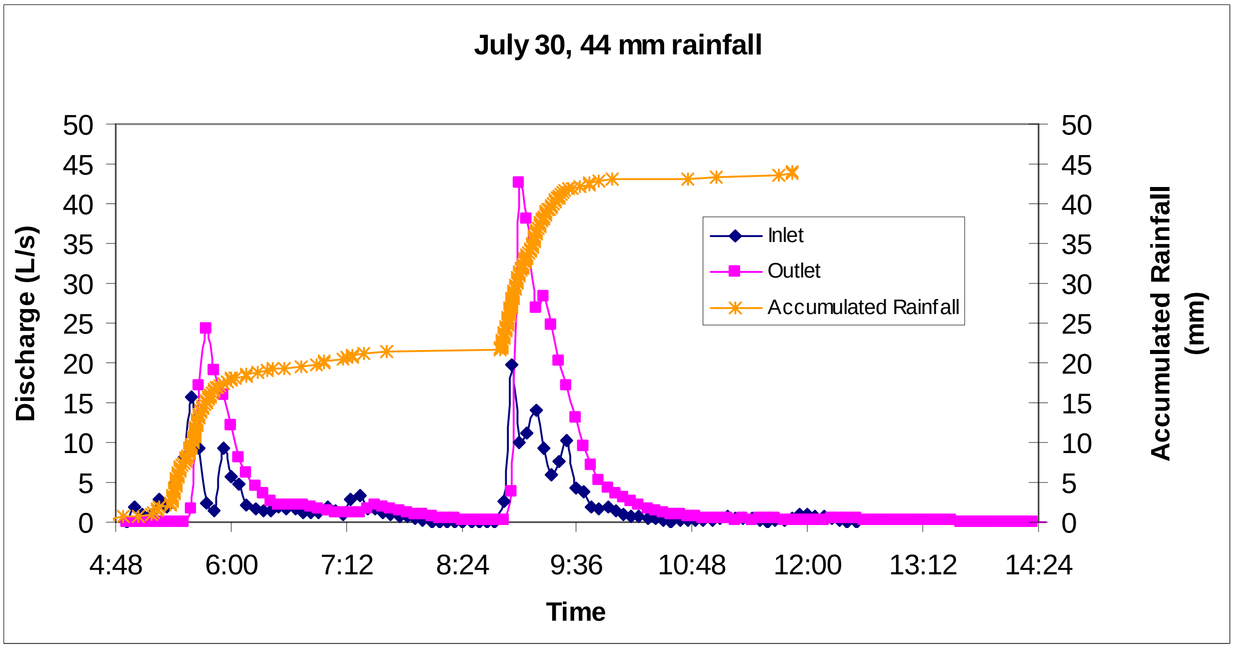
Table 3.3 Summary of hydrologic data measured at inlet and outlet of Quinton Heights basin.

Date	Anteced-ent Soil Moisture	Precipitation		Inlet		Total In	Outlet	% Vol. Retained	Peak Flow	
		Depth mm	Volume[1] m^3	Volume m^3	Depth[2] mm	Volume[3] m^3	Volume m^3		Inlet L/s	Outlet L/s
6/1/2007	0.28	28	44	57	10	101	92	9	9	45
6/4/2007	0.27	1	2	4	1	6	5	24	6	1
6/18/2007	0.22	5	7	9	2	16	0	100	9	0
6/23/2007	0.22	15	23	32	5	55	56	-1	9	13
6/27/2007	0.21	23	36	51	9	87	93	-6	10	20
6/29/2007	0.22	36	55	108	18	163	224	-37	9	9
7/4/2007	0.26	15	23	26	4	49	63	-29	11	26
7/12/2007	0.23	2	3	4	1	7	0	100	6	0
7/20/2007	0.18	2	3	1	0	5	0	100	1	0
7/23/2007	0.17	3	4	1	0	5	0	100	1	0
7/29/2007	0.14	5	7	9	1	16	0	100	17	0
7/30/2007	0.18	44	68	67	11	135	126	6	17	45
7/31/2007	0.18	3	4	2	0	6	0	100	3	0
8/2/2007	0.19	16	24	16	3	40	43	-10	9	34
8/8/2007	0.17	20	31	26	4	57	60	-5	13	23
8/16/2007	0.15	30	47	22	4	69	2	97	9	0
8/18/2007	0.15	9	14	9	2	23	0	100	7	0
8/24/2007	0.13	9	13	16	3	29	0	100	9	0
9/10/2007	0.12	2	3	1	0	4	0	100	1	0
9/18/2007	0.12	8	13	15	3	28	21	25	6	3
9/30/2007	0.11	3	4	4	1	8	0	100	10	0
10/2/2007	0.12	19	29	13	2	42	66	-58	3	71
10/7/2007	0.2	0	1	8	1	9	0	100	7	0
10/8/2007	0.14	20	31	44	7	75	93	-25	20	34
10/13/2007	0.17	39	61	*	*	*	232	*	*	45
10/14/2007	0.19	29	45	*	*	*	220	*	*	37
10/17/2007	0.24	46	72	*	*	*	352	*	*	35
10/22/2007	0.25	26	40	*	*	*	167	*	*	7

[1]Precipitation volume calculated by multiplying the depth measured at the basin with the HOBO rain gauge by the area of the basin (1,500 m^2).
[2]Runoff depth calculated by dividing the volume measured at the inlet by the area of the inlet's watershed (6,000 m^2, as given by Spaar (2004)).
[3]Total In includes volume contributions from direct precipitation and inlet pipe flow.
*indicates storms for which inlet bubbler was not working properly; therefore, data is not reported.

From the table, it can be seen that the percent of runoff retained by the basin varied widely. In general, the basin either completely retained runoff, particularly for precipitation events less than 10 mm, or retained less than 25% of runoff, particularly for storms in which more than 10 mm of rain fell. For a number of storms, retention values less than zero were calculated, indicating that more water was flowing out of the basin than was actually entering. A comparison of the peak flow rates entering and leaving the basin revealed very limited reduction, if any, in peak flows. Aside from the events that were completely retained by the basin, peak flows exiting the basin were much greater than those recorded at the inlet. This is illustrated by the inlet and outlet hydrographs generated during the storm that occurred on July 30, 2007 (Figure 3.5). Although this storm was one of the largest recorded during the study period (44 mm), the inlet and outlet hydrographs produced during this storm were representative of those from the rest of the study.

Figure 3.5 Inlet and outlet hydrographs for July 30, 2007 storm at Quinton Heights basin. The peak flow rate and volumes at the outlet were higher than those measured at the inlet.



Both the peak flow and percent retention calculations indicate that the basin did not effectively manage storms greater than 10 mm. However, data collected at the inlet provokes the question of why the inlet readings were so low, and often even lower, in comparison to the outlet. There are several reasons to explain why inlet flow measurements are lower than expected, the foremost being the addition of unaccounted for overland flow to the basin and measurement error due to the configuration of the bubbler in the inlet pipe. These reasons will be examined further in the following discussion.

Contribution by Overland Flow

One of the reasons that the inlet volumes and flow rates are low in comparison to those measured at the outlet is that runoff also entered the basin via overland flow which was not accounted for in the quantity of water that was measured at the inlet pipe. As seen in Figure 3.1, the flow boundary of the area lying directly south of the basin was delineated in the drainage plan for the area created by the City of Topeka. However, since this area was not hydraulically connected to the inlet pipe, it was not included in the drainage area calculated for the basin. This area does, however, contribute runoff directly to the basin through overland surface flow. To account for this area, the NRCS curve number method was applied to calculate the depth of runoff expected from this area for each precipitation event. The depth of runoff was calculated using equation 3.3 (Schwab et al, 1993).

$$Q = \frac{(I - 0.2S)^2}{I + 0.8S} \qquad \text{Equation 3.3}$$

Here, Q is the direct surface runoff depth (mm), I is the storm rainfall (mm) and S is the maximum potential difference between rainfall and runoff at the time the storm begins (mm). The calculation for S is given by Equation 3.4, where N represents the curve number.

$$S = \frac{25400}{N} - 254 \qquad \text{Equation 3.4.}$$

A curve number of 85 was selected based on the land uses and soil types of the area from the table of curve number values presented in the TR-55 Urban Hydrology for Small Watersheds handbook (USDA NRCS, 1996). The S value calculated with a curve number of 85 was 44.8 mm. It should be noted that the NRCS method assumes an initial abstraction of 0.2S, so if the total amount of precipitation that falls does not exceed this amount (9 mm) then runoff is not expected and Q equals zero.

The depth of runoff as calculated by the NRCS curve number method was converted to a volume by multiplying it by the area of the hillside (1,900 m^2) as given by the drainage plans for the Quinton Heights basin (Spaar, 2004). The calculated depth and volume of runoff contributed through overland flow from this area is displayed in Table 3.4. When the overland flow component is accounted for, the basin’s water balance improves slightly, with only five events producing more volume at the outlet than entered the basin through overland flow, pipe flow at the inlet, or direct precipitation. Still, a negative percent retention by the basin cannot be explained physically. The following section discusses the second reason by which the discrepancy between flow coming into and exiting the basin is thought to occur.

Table 3.4 Summary of volume into (including direct precipitation, flow from inlet, and overland flow from adjacent hillside) and out of the basin.

Date	Precipitation[1] Depth mm	Precipitation[1] Volume m^3	Runoff In Inlet[2] Volume m^3	Runoff In Inlet[2] Depth mm	Overland Flow[3] Depth mm	Overland Flow[3] Volume m^3	Total In[4] Volume m^3	Outlet Volume m^3	% Vol. Retained
6/1/2007	28	42	57	10	6	11	110	92	17
6/4/2007	1	2	4	1	0	0	6	5	18
6/18/2007	5	7	9	2	0	0	16	0	100
6/23/2007	15	23	32	5	1	1	56	56	0
6/27/2007	23	35	51	9	3	7	92	93	-1
6/29/2007	36	54	108	18	10	19	181	224	-24
7/4/2007	15	22	26	4	1	1	49	63	-28
7/12/2007	2	3	4	1	0	0	7	0	100
7/20/2007	2	3	1	0	0	0	4	0	100
7/23/2007	3	4	1	0	0	0	5	0	100
7/29/2007	5	7	9	2	0	0	16	0	100
7/30/2007	44	66	67	11	15	29	162	126	22
7/31/2007	3	4	2	0	0	0	6	0	100
8/2/2007	16	24	16	3	1	2	41	43	-4
8/8/2007	20	30	26	4	2	4	60	60	0
8/16/2007	30	45	22	4	7	13	80	2	98
8/18/2007	9	14	9	2	0	0	23	0	100
8/24/2007	9	13	16	3	0	0	29	0	100
9/10/2007	2	3	1	0	0	0	4	0	100
9/18/2007	8	13	15	3	0	0	28	21	24
9/30/2007	3	4	4	1	0	0	8	0	100
10/2/2007	19	28	13	2	2	3	45	66	-48
10/7/2007	0	1	8	1	0	0	9	0	100
10/8/2007	20	30	44	7	2	4	78	93	-20

[1]Precipitation volume calculated by multiplying the depth measured at the basin with the HOBO rain gauge by the area of the basin (1,500 m^2).
[2]Runoff depth calculated by dividing the volume measured at the inlet by the area of the inlet's watershed (6,000 m^2, as given by Spaar (2004)).
[3]Overland flow depth calculated with NRCS Curve Number equation using a curve number of 85. Overland flow volume calculated by multiplying depth by area of the contributing area (1,900 m^2).
[4]Total In includes volume contributions from direct precipitation, inlet pipe flow, and overland flow.

Inlet Measurement Error

In addition to the additional contribution of runoff to the basin through overland flow, error in the inlet measurement is believed to have produced inlet flows that were much lower than expected when compared to the outlet. Error in the inlet flow measurements is validated by an examination of the percent runoff generated in the basin's watershed. Based on the inlet

ISCO flow data, the percent runoff from the drainage area ranged from 9% to 50%, with an average value of 26%. Such low levels of runoff indicate that a large portion of rainfall was being infiltrated or intercepted upstream of the basin. However, observation of the land cover in the watershed, the primary components of which are a parking lot and streets, and the drainage plans for the Quinton Heights basin indicates that this result is not realistic. To examine the possibility of under-measurement of stormwater runoff at the inlet pipe by the ISCO, the NRCS curve number method was applied to the 6,000 m^2 area which drained to the inlet pipe, similarly as it was applied to calculate overland flow. Based on the land uses and soil types in the delineated watershed, curve numbers were selected for each of the sub-watersheds from the table of curve number values presented in the TR-55 (USDA NRCS, 1996). A weighted curve number of 95 was calculated for the drainage area by multiplying the curve number for each sub-watershed by the percent of the total drainage area it occupied. The runoff depth and volume calculated for the area draining to the basin inlet are presented in Table 3.5. It should be noted that in almost every case, the amount of runoff calculated for the site was greater than that measured at the basin inlet with the ISCO. Exceptions are for storms in which less than 8 mm of rain fell because these small storms fell below the 0.2S initial abstraction threshold assumed by the NRCS method. Although the runoff calculated for these storms was zero, inlet measurements indicate than some runoff was generated, which could indicate that the 0.2S initial abstraction assumption is too high for this site.

Using the runoff depth predicted by the NRCS curve number method, the percent retention by the basin was recalculated. The recalculated retention rate includes volume expected from this calculated, along with inputs from direct rainfall and overland flow (as discussed in the previous section. Use of the NRCS equation to predict the amount of runoff at the inlet pipe also enabled comparisons of inlet and outlet flow volumes for storms that occurred from October 13 to 22 to be made. Actual inflow measurements for this time period are not available due to a malfunctioning inlet bubbler module, so measured inlet data is not presented. Runoff predictions from the NRCS curve number method, however, allowed estimates of the percent retention for this period to be made. Estimated percent retention rates are also presented in Table 3.5.

Table 3.5 Percent retention by the basin as determined by actual flow measurements and runoff calculations from the NRCS curve number method.

		Measured Inflow[1]			Calculated Inflow (CN=95)[2]			% Retained[3]	
Date	Precip mm	Volume m^3	Depth mm	% Runoff	Depth mm	Volume m^3	% Runoff	Measured %	Calculated %
6/1/2007	28	57	9	34	17	101	59	9	40
6/4/2007	1	4	1	46	0	0	0	24	-119
6/18/2007	5	9	2	31	0	2	6	100	100
6/23/2007	15	32	5	36	6	35	39	-1	6
6/27/2007	23	51	9	37	12	75	54	-6	20
6/29/2007	36	108	18	50	24	142	66	-37	-4
7/4/2007	15	26	4	30	6	34	39	-29	-12
7/12/2007	2	4	1	30	0	0	0	100	100
7/20/2007	2	1	0	12	0	0	0	100	100
7/23/2007	3	1	0	7	0	0	0	100	100
7/29/2007	5	9	1	30	0	2	6	100	100
7/30/2007	44	67	11	25	31	187	71	6	55
7/31/2007	3	2	0	14	0	0	0	100	100
8/2/2007	16	16	3	16	7	39	41	-10	33
8/8/2007	20	26	4	22	10	58	49	-5	34
8/16/2007	30	22	4	12	19	111	61	97	99
8/18/2007	9	9	2	17	2	13	23	100	100
8/24/2007	9	16	3	30	2	11	21	100	100
9/10/2007	2	1	0	11	0	0	0	100	100
9/18/2007	8	15	3	30	2	10	20	25	7
9/30/2007	3	4	1	26	0	0	0	100	100
10/2/2007	19	13	2	11	9	53	47	-58	22
10/7/2007	1	8	1	334	0	0	0	100	100
10/8/2007	20	44	7	37	10	58	49	-25	-2
10/13/2007	39	*	*	*	27	160	68	*	48
10/14/2007	29	*	*	*	17	104	60	*	24
10/17/2007	46	*	*	*	33	200	72	*	35
10/22/2007	26	*	*	*	15	88	57	*	32

[1]Measured inflow depth calculated by dividing the volume measured at the inlet by the area of the inlet's watershed (6,000 m^2, as given by Spaar (2004)).
[2]Calculated inflow determined using NRCS curve number method and a weighted curve number of 95. The runoff depth calculated from this method was converted to a volume by multiplying by area of inlet watershed (6,000 m^2, as given by Spaar (2004)).
[3]Measured % retained considers only volume inputs by direct precipitation and measured inflow; calculated percent retained considers direct precipitation and NRCS curve number method calculations for overland flow and runoff to the inlet pipe.
* indicates storms for which inlet bubbler was not working properly; therefore, data is not reported.

The measurement error at the inlet was most likely a result of the configuration of the ISCO bubbler in the inlet pipe. Unlike the bubbler at the outlet, which measured outflow over a 90° v-notch weir, the inlet bubbler was placed inside a 45.7-cm (18 in.) diameter concrete pipe. While the ISCO can measure low flows over a v-notch weir with a fair degree of accuracy (Grant and Dawson, 2001), low flows in the large diameter pipe are believed to have produced substantial measurement errors. Measurement errors at lower flows would account for inlet flow rates and runoff volumes that were consistently lower than the volume and flow rates measured at the outlet. Such error would also explain why peak discharge rates at the inlet were lower than expected when compared with the outlet.

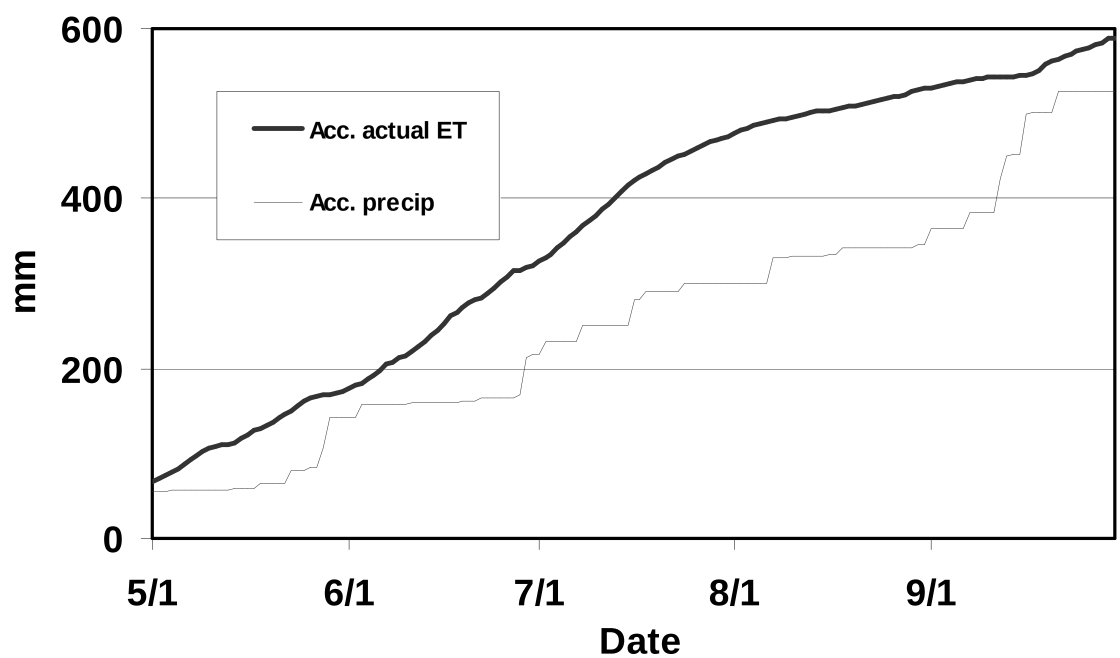
An examination of the recalculated percent volume retention indicates that, in general, the basin either completely retained the runoff event or retained very little of it. The basin completely retained storms in which less than 10 mm of rain fell. Such small showers accounted for 11 of the 28 precipitation events recorded during the study period. For storms in which more than 10 mm of rain fell, retention rates were lower than desirable, ranging from 0% to 55%. The following section discusses probable reasons for the percent retentions observed at the basin, with particular emphasis on the role of ET in the hydraulic performance of the basin.

Evapotranspiration and percent runoff retained

Low retention rates were especially characteristic of the period from June 1 to July 4 and from October 8 to October 22. The inability of the basin to retain a substantial portion of the runoff during these periods can be partially explained by the short duration between rain events, which likely resulted in saturated conditions in the basin. Figure 3.5 helps illustrate this point. The figure displays the accumulated depth of precipitation that fell on the basin along with the modeled accumulated ET of the grasses. During periods in which the soil profile was near field capacity, there was little difference between the two curves. As the soil moisture was depleted through ET, the gap between the two curves increased. As seen from the figure, the modeled ET of the grasses was nearly the same as the depth of rain that fell on the basin from June 1 to July 4. Since the water added to the basin does not reflect runoff contributions from the rest of the watershed, it can safely be assumed that the soil profile was full at this time. Therefore, water added to the basin was not able to infiltrate, and less was retained by the basin. A similar result was observed during the series of storms in mid-October as frequent storms refilled the soil profile, reducing the infiltrative capacity and the volume retention of the basin.

In between these two periods, the gap between water use and water supplied increased, indicating that the soil profile was being depleted of water even while multiple rain events during this period attempted to refill the profile. Looking back to the modeled soil moisture values in Figure 3.3, it was evident that this drying out of the basin occurred even as multiple storms greater than 25 mm (1 in.) fell on the basin. The deficit that developed between the amount of water stored in the soil profile and that used by the grasses during this period was largely a result of high rates of water use by the grasses in the basin; warm season grasses in this region enter the reproductive stage of plant growth during late July and early August, during which time water demand is high. Due to higher rates of water use by the grasses during this period, it was surmised that ET by the grasses played a large role in drying out the basin despite multiple rain events.

Figure 3.5 Accumulated actual ET (reflecting water use by the grass) and precipitation for Quinton Heights basin as predicted by the check-book water balance method for June 1 through October 23, 2007.



Although the curve for the accumulated ET in the basin is a modeled prediction, it can be used to estimate the storage available in the soil profile. The difference between the summation of the accumulated water use (actual ET for fully-watered grass) and water supplied (precipitation) curves represents the storage available in the profile. The gap between the curves in Figure 3.5 indicates that there was excess storage available during the latter half of the growing season. It was also during this period that the basin was most successful in retaining and reducing the volume of runoff generated during each precipitation event, as evidenced in Table 3.5. The amount of storage available in the profile will change seasonally depending upon the timing and amount of rainfall and the actual ET rates of the vegetation. Thus, it is difficult to predict how the deficit between plant water use and water supply will change from year to year. Still, this concept can be applied to recommend outlet configurations to improve the system's effectiveness in managing stormwater runoff flows. Observations of the basin during and after rainfall indicate that that the majority of the runoff entering the basin through the inlet remained confined to the central channel. In order to utilize the soil storage available in the rest of the basin, the outlet should be reconfigured to force runoff to spread across a greater portion of the basin. By temporarily ponding runoff across a larger area, more time would be allowed for infiltration processes to occur, thus further reducing the total volume of runoff to exit the basin. As the grasses and their root systems continue to mature, thus improving soil structure and hydraulic properties, the outlet configuration could be further adjusted to take advantage of higher infiltration rates.

Conclusions

In order to successfully implement infiltration-based stormwater management practices, an understanding of infiltration processes and how these processes can change over time is needed. This study examined infiltration at three sites planted with tallgrass prairie species at different stages of vegetative maturity. Two of these sites, Johnson County and Quinton Heights, functioned for urban stormwater management and were in their first and fourth seasons of growth, respectively. Statistically significant differences in saturated infiltration rates between the two sites were not detected, probably as a result of the high variability in rates measured at each site. However, the numerical mean effective saturated infiltration rate at the more established Quinton Heights site was higher than at the newly planted Johnson County site, suggesting that slight gains in infiltration may result over relatively short periods of time (four years). Comparisons of infiltration rates at the urban sites with a grass filter strip in its seventh year of growth also yielded no significant differences, although the variability of saturated infiltration rates measured at the older site were substantially less than at the Quinton Heights or Johnson County sites. These results suggest that increases in root development and density may help promote infiltration over more of the area, thus decreasing the spatial variability of infiltration rates as encountered at the less established urban sites. However, since saturated infiltration rates were not significantly different among any of the sites, the findings of this study support previous work stating that long periods of time may be required to regain predisturbance soil characteristics, including the size distribution of water-stable soil aggregates (Jastrow, 1987).

Actual infiltration rates, and thus stormwater volume reductions, realized by infiltration-based stormwater management systems depend upon more than the saturated infiltration rate of the soil profile; rainfall amount, intensity, and antecedent moisture conditions also affect the performance of infiltration-based systems. In order to better understand the interplay of these factors on actual stormwater volume and peak flow reductions, stormwater inflow and outflow was monitored at the Quinton Heights basin with ISCO automated water samplers. Upon examination of measured inlet flows, it was determined that the inlet data did not capture all of the runoff delivered to the basin due to the configuration of the bubbler in the large inlet pipe, which is believed to have led to substantial measurement errors at low flows, and additional runoff inputs via overland flow from a hillside adjacent to the basin. To adjust inlet flow measurements to reflect these additional runoff inputs, the NRCS curve number method was used to calculate the depth of runoff from the basin's drainage area. The percent of runoff captured by the basin was calculated using the adjusted inlet flows and outlet discharge. Frequent rainstorms through the months of June and October kept the basin wet and reduced the infiltrative capacity of the system. Less frequent storms coupled with the active growth and higher ET rates of the grasses from July to September to deplete the soil moisture content of the profile, thus increasing the deficit between plant-water use and supply. As a result, the basin was most successful in capturing storms during this period.

While the basin performed moderately well in terms of percent volume retained, field observations indicated that the majority of runoff delivered to the basin remained confined in the central channel. By reconfiguring the outlet, runoff could be forced to spread over a larger area of the basin, thus allowing a greater portion of the soil's storage capacity to be utilized and further reducing the volume of outflow leaving the system.

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CHAPTER 4 - Conclusions

Stormwater management remains one of the greatest challenges to urban planners and engineers in terms of both runoff quantity and quality. In the interest of curtailing flooding in urban areas, stormwater management has traditionally focused on the issue of quantity, targeting practices that quickly convey runoff from the site. The most common management methods encountered in the urban landscape- including straightened, concrete-lined channels, subterranean stormwater pipes, and occasional regional detention basins- reflect this management philosophy. While such management methods reduce flooding in and directly adjacent to urban areas, the erosive energy, high volume, and poor quality of urban runoff has led to stream channel and water quality degradation in receiving water bodies. To combat the negative effects of urban stormwater runoff, ecologically-designed stormwater systems are being implemented. These systems capitalize upon natural ecosystem functions, particularly infiltration, to reduce stormwater volume while improving stormwater quality.

Although these systems are becoming more common across the urban landscape, there are challenges involving the maintenance and monitoring of these systems that must be overcome in order to more effectively mitigate the effects of urban runoff. Because the desired operation of these systems hinges on the maintenance of ecosystem processes, assessment of theses systems should include an ecological health component in addition to the more traditional hydrologic and chemical metrics. In addition, maintenance regimes which will ensure the sustainability of these functions should be adopted. The first objective of this study was to examine the feasibility of developing an ecological health rubric that could be used to assess the ecological health of an ecologically-designed stormwater system. A simple rubric containing biological and ecological health indicators which were relevant to stormwater system performance was developed and applied to two stormwater systems in eastern Kansas designed after the tallgrass prairie ecosystem. The data presented in this study represents the preliminary development of the ecological health assessment. The next step is to tie the health scores with hydrologic and water quality data from the site in order to establish a correlation between ecological health and system function and, if necessary, to further refine the scoring system. The overall goal for the ecological health assessment tool developed in this study is to provide municipal stormwater managers and employees- who are often responsible for the maintenance of these systems- with a tool that would allow them to easily track the function of the system over time without the need for time consuming or expensive hydraulic and chemical analysis. The initial application of the assessment developed for this study indicates that the ecological health assessment could be used to help guide maintenance decisions, such as mowing or burning regimes, to maintain the health of the ecosystem.

Many stormwater BMPs in the region occupying the historic extent of the tallgrass prairie incorporate tallgrass prairie species, which were the dominant vegetation type prior to the rise of agriculture and urbanization, in order to improve infiltration rates at the site and reduce stormwater volume. However, the paucity of monitoring data for these systems, especially for those in eastern Kansas and the rest of the region, creates challenges to improving system design and to quantifying actual infiltration gains in the system. The second objective of this study was to examine infiltration in a stormwater retention basin vegetated with prairie grasses in order to determine the ability to regain infiltration processes in the disturbed urban landscape. A comparison of the effective saturated infiltration rates measured at sites at different stages of vegetative maturity yielded no significant differences. This result suggests that the time required to realize significant improvements in infiltration rates may be longer than the three to five years required for the extensive root systems of prairie grasses to fully develop. Rather, regaining the hydraulic properties of the soil after disturbing may require decades, as suggested by Jastrow (1987). This result has important implications for the planning and construction of infiltration-based stormwater systems. Since a period of years may be required to regain pre-disturbance infiltration rates even after vegetation is established, infiltraton-based practices will be most effective on land that has not been disturbed. In the context of urban development, this would require a paradigm shift away from the current practice of razing the entire area prior to development and reestablishing vegetation afterwards to preserving undisturbed sites within the development area which can serve as infiltration-based BMPs.

Implementing infiltration-based stormwater systems in undisturbed sites is not always possible, especially in the case of urban retrofits or in the construction of retention basins similar to the Quinton Heights site. To better understand the hydraulic performance of a disturbed site after being revegetated with deeply-rooted prairie grasses, the flow into and out of the basin was monitored. The basin completely contained storms less than 10 cm in depth, and retained 20% to 64% of storms larger than this, depending on the size of the event and the antecedent soil moisture. Field observations indicate that the majority of runoff remained in the central channel of the basin rather than spreading over the entire basin floor. By incorporating an orifice plate or other outlet reconfiguration, it is believed that runoff could be forced to temporarily pond over the remaining area of the basin, thus utilizing more of the available soil storage and further reducing the volume exiting the system. This brings about another important point for the implementation and improved design of these systems. Because of the dynamic nature of biological systems, eco-based systems will continue to develop and, if maintained properly, improve in function after construction. By gaining a better understanding of how the functions of these systems, including infiltration, change with time, designs- including adjustable outlet orifices- can be adopted to have a greater impact on stormwater runoff as the system matures.

Taking a step back from the single stormwater system monitored in this study allows bigger picture to come into focus. Questions related to the overall impact of ecologically-designed stormwater systems on the quality of receiving streams and water bodies are still in need of answers. Based on the results of this study, eco-based stormwater systems such as the Quinton Heights basin do have the potential to significantly reduce stormwater volume, especially if steps are taken to increase the ponding area in the basin. The basin was especially effective in capturing the runoff volume from smaller, more frequent storms. Studies of fluvial systems, and particularly those in urbanizing areas, have demonstrated that it is the smaller, more frequent discharges that conduct the majority of work in stream channels. In urban areas, this discharge has been equated with the runoff from precipitation events with a 1.1- to 1.5-year return frequency (McCrea, 1997). The ability of the basin to significantly reduce the volume of runoff from these smaller flows indicates that these systems could protect receiving streams from further degradation in urbanizing watersheds.

Of course, one system cannot be expected to significantly reduce the volume or rate of runoff delivered to receiving streams and water bodies; the Quinton Heights basin was only designed to control runoff from a 7,900 m^2 (2 acre) area. In order to have a significant impact, similar infiltration-based systems must be distributed across the landscape. This gives rise to questions of regarding the optimum number and location of stormwater BMPs. The results from this study adds to the foundation from which the answers to these questions, and hence our ability to truly protect the quality of receiving streams and water bodies in urban watersheds, will come.

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Appendix A - Ecological Health Assessment

This appendix includes all data recorded for the ecological health assessment, including scores for individual indicators within each category, total scores, and results from statistical analysis. Scores and statistical analysis can also be found at L:\WaterQuality\UrbanSites\Trisha\EcoAssessCompile.exe.

Table A.1 Ecological Health Assessment scores and statistics.

Johnson County					Quinton Heights				
Date	**Plant Health**				**Date**	**Plant Health**			
	Density	Diversity	Vigor	**Total**		Density	Diversity	Vigor	**Total**
6/26/2007	1	2	3	**6**	5/8/2007	4	3	3	**10**
7/10/2007	2	2	4	**8**	5/31/2007	4	3	4	**11**
8/6/2007	3	2	4	**9**	6/22/2007	3	3	3	**9**
8/24/2007	4	2	4	**10**	7/20/2007	3	4	3	**10**
9/5/2007	3	2	4	**9**	8/7/2007	4	3	4	**11**
9/28/2007	4	2	4	**10**	9/7/2007	4	3	4	**11**
10/10/2007	4	2	4	**10**	10/2/2007	4	4	2	**10**
Average				**8.9**	Average				**10.3**
Standard Deviation				**1.5**	Standard Deviation				**0.8**
Coefficient of Variation				**16.5**	Coefficient of Variation				**7.3**
t-test: p= 0.047									

Johnson County						Quinton Heights					
Date	**Soil Erosion**					**Date**	**Soil Erosion**				
	Sheet/Rill	Deposition	Gullies	Pedestaling	**Total**		Sheet/Rill	Deposition	Gullies	Pedestaling	**Total**
6/26/2007	3.0	4	4	4	**15**	5/8/2007	4.0	2	2	3	**11**
7/10/2007	3.0	4	4	4	**15**	5/31/2007	4.0	3	3	4	**14**
8/6/2007	4.0	4	4	4	**16**	6/22/2007	4.0	3	3	4	**14**
8/24/2007	4.0	4	4	4	**16**	7/20/2007	4.0	4	3	4	**15**
9/5/2007	4.0	4	4	4	**16**	8/7/2007	4.0	4	4	4	**16**
9/28/2007	4.0	4	4	4	**16**	9/7/2007	4.0	4	4	4	**16**
10/10/2007	4	4	4	4	**16**	10/2/2007	4	3	2	4	**13**
Average					**15.7**	Average					**14.1**
Standard Deviation					**0.5**	Standard Deviation					**1.8**
Coefficient of Variation					**3.1**	Coefficient of Variation					**12.5**
t-test: p= 0.058											

Table A.1 (cont.). Ecological Health Assessment scores and statistics.

Johnson County						Quinton Heights					
Date	**Soil Health** OM Color	Roots	Tilth	Comp-action	**Total**	**Date**	**Soil Health** OM Color	Roots	Tilth	Comp-action	**Total**
6/26/2007	3	1	3	3	**7**	5/8/2007	*	*	*	*	*
7/10/2007	3	1	2	3	**6**	5/31/2007	4	3	3	3	**10**
8/6/2007	3	2	2	2	**7**	6/22/2007	4	3	3	3	**10**
8/24/2007	1	2	2	2	**5**	7/20/2007	4	3	3	3	**10**
9/5/2007	1	2	2	2	**5**	8/7/2007	4	3	4	3	**11**
9/28/2007	3	2	2	3	**7**	9/7/2007	4	3	4	3	**11**
10/10/2007	4	2	2	3	**8**	10/2/2007	4	3	4	3	**11**
Average					**6.4**	Average					**10.5**
Standard Deviation					**1.1**	Standard Deviation					**0.5**
Coefficient of Variation					**17.64**	Coefficient of Variation					**5.2**
t-test: p= 8.65E-05											
						* assessment not conducted- saturated conditions					
Date	**Faunal Health** Earth-worms	Soil fauna	Other	Mosqu-itoes	**Total**	**Date**	**Faunal Health** Earth-worms	Soil fauna	Other	Mosqu-itoes	**Total**
6/26/2007	1	1	1	4	**7**	5/8/2007	1	1	1	2	**1**
7/10/2007	2	1	3	3	**9**	5/31/2007	1	1	4	2	**8**
8/6/2007	1	1	3	4	**9**	6/22/2007	1	1	3	4	**9**
8/24/2007	2	2	3	4	**11**	7/20/2007	1	1	3	4	**9**
9/5/2007	1	1	4	4	**10**	8/7/2007	1	2	4	4	**11**
9/28/2007	1	1	3	4	**9**	9/7/2007	1	1	4	4	**10**
10/10/2007	3	4	4	4	**15**	10/2/2007	1	3	4	3	**11**
Average					**10.0**	Average					**8.5**
Standard Deviation					**2.5**	Standard Deviation					**3.4**
Coefficient of Variation					**25.17**	Coefficient of Variation					**39.8**
t-test: p= 0.35											
Date	**Overall**					**Date**	**Overall**				
6/26/2007					**38**	5/8/2007					26
7/10/2007					**41**	5/31/2007					46
8/6/2007					**43**	6/22/2007					45
8/24/2007					**44**	7/20/2007					47
9/5/2007					**42**	8/7/2007					52
9/28/2007					**45**	9/7/2007					51
10/10/2007					**52**	10/2/2007					48
Average					**43.6**	Average					**45.0**
Standard Deviation					**4.4**	Standard Deviation					**8.8**
Coefficient of Variation					**10.0**	Coefficient of Variation					**19.5**
t-test: p= 0.71											

Appendix B - Infiltration

This appendix includes a sample data sheet used to record effective saturated infiltration rates. The final statistical analysis used to test for significant differences in mean saturated rates among sites is also included. All data for infiltration measurements can be found at L:\WaterQuality\UrbanSites\Trisha\Infiltration.exe.

Table B.1 Double-ring infiltrometer measurements taken May 31, 2007 in Quinton Heights stormwater basin.

Site:	Quinton Heights
Date:	5/31/2007

Infiltrometer 1

Time	Δ Time	Cum. Time minutes	Cum. Time hours	Time Between minutes	Refill Reading cm	Depth Reading cm	Cum. Infil. cm	Rate cm/hr
2:43:50		0.00	0.00	0.00		3.3	0.00	
2:55:40	0:11:50	11.83	0.20	11.83	9.0	12.5	9.20	46.65
3:06:00	0:10:20	22.17	0.37	10.33	5.2	14.2	14.40	30.19
3:20:40	0:14:40	36.83	0.61	14.67	3.9	13.5	22.70	33.95
3:35:15	0:14:35	51.42	0.86	14.58	5.1	11.8	30.60	32.50
3:49:50	0:14:35	67.13	1.12	15.71	5.6	13.0	38.50	30.16
4:03:00	0:13:10	80.30	1.34	13.17	7.1	13.5	46.40	36.00
4:17:00	0:14:00	94.30	1.57	14.00	4.4	14.6	53.90	32.14
4:30:10	0:13:10	107.46	1.79	13.17		11.5	61.00	32.35
4:34:30	0:04:20	111.80	1.86	4.33	3.3	13.5	63.00	27.69
4:53:10	0:18:40	130.46	2.17	18.67	3.1	13.5	73.20	32.79
5:14:00	0:20:50	151.30	2.52	20.83		14.6	84.70	33.12
5:18:30	0:04:30	155.80	2.60	4.50	7.7	16.6	86.70	26.67
5:31:00	0:12:30	168.30	2.80	12.50	3.6	14.2	93.20	31.20
5:47:50	0:16:50	185.13	3.09	16.83		13.6	103.20	35.64

Infiltrometer 2

Time	Δ Time	Cum. Time minutes	Cum. Time hours	Time Between minutes	Refill Reading cm	Depth Reading cm	Cum. Infil. cm	Rate cm/hr
2:45:10		0.00	0.00	0.00		3.0	0.00	
2:57:30	0:12:20	12.33	0.21	12.33		8.9	5.90	28.70
3:06:40	0:09:10	21.50	0.36	9.17		11.0	8.00	13.75
3:15:40	0:09:00	30.50	0.51	9.00	5.3	12.0	9.00	6.67
3:27:50	0:12:10	42.67	0.71	12.17		8.9	12.60	17.75
3:38:20	0:10:30	53.17	0.89	10.50		10.6	14.30	9.71
3:51:50	0:13:30	66.67	1.11	13.50		11.8	15.50	5.33
4:05:40	0:13:50	80.50	1.34	13.83	5.1	12.4	16.10	2.60
4:20:30	0:14:50	95.33	1.59	14.83		8.8	19.80	14.97
4:31:00	0:10:30	105.83	1.76	10.50		10.3	21.30	8.57
4:56:30	0:25:30	131.33	2.19	25.50		11.8	22.80	3.53

5:14:40	0:18:10	149.50	2.49	18.17		12.1	23.10	0.99
Time	Δ Time	Cum. Time minutes	Cum. Time hours	Time Between minutes	Refill Reading cm	Depth Reading cm	Cum. Infil. cm	Rate cm/hr
5:20:50	0:06:10	155.67	2.59	6.17	5.2	12.3	23.30	1.95
5:33:50	0:13:00	168.67	2.81	13.00		8.5	26.60	15.23
5:49:10	0:15:20	184.00	3.07	15.33		10.5	28.60	7.83

Infiltrometer 3

Time	Δ Time	Cum. Time minutes	Cum. Time hours	Time Between minutes	Refill Reading cm	Depth Reading cm	Cum. Infil. cm	Rate cm/hr
2:46:50		0.00	0.00	0.00		2.3	0.00	
2:58:20	0:11:30	11.50	0.19	11.50		4.5	2.20	11.48
3:07:40	0:09:20	20.83	0.35	9.33		5.5	3.20	6.43
3:18:10	0:10:30	31.33	0.52	10.50		6.5	4.20	5.71
3:28:40	0:10:30	41.83	0.70	10.50		7.0	4.70	2.86
3:39:30	0:10:50	52.67	0.88	10.83		7.5	5.20	2.77
3:52:42	0:13:12	78.27	1.30	25.60		8.0	5.70	1.17
4:07:48	0:15:06	93.37	1.56	15.10		8.5	6.20	1.99
4:22:00	0:14:12	107.57	1.79	14.20		8.9	6.60	1.69
4:32:30	0:10:30	118.07	1.97	10.50		9.1	6.80	1.14
4:59:00	0:26:30	144.57	2.41	26.50		9.7	7.40	1.36
5:16:00	0:17:00	161.57	2.69	17.00		10.0	7.70	1.06
5:35:40	0:19:40	181.23	3.02	19.67	4.0	10.3	8.00	0.92
5:50:10	0:14:30	195.73	3.26	14.50		4.8	8.80	3.31
6:04:40	0:14:30	210.23	3.50	14.50		5.5	9.50	2.90
6:24:40	0:20:00	230.23	3.84	20.00		6.5	10.50	3.00

Figure B.1 Plot of cumulative infiltration with time. Effective saturated infiltration rate calculated as slope of line when curve becomes linear.

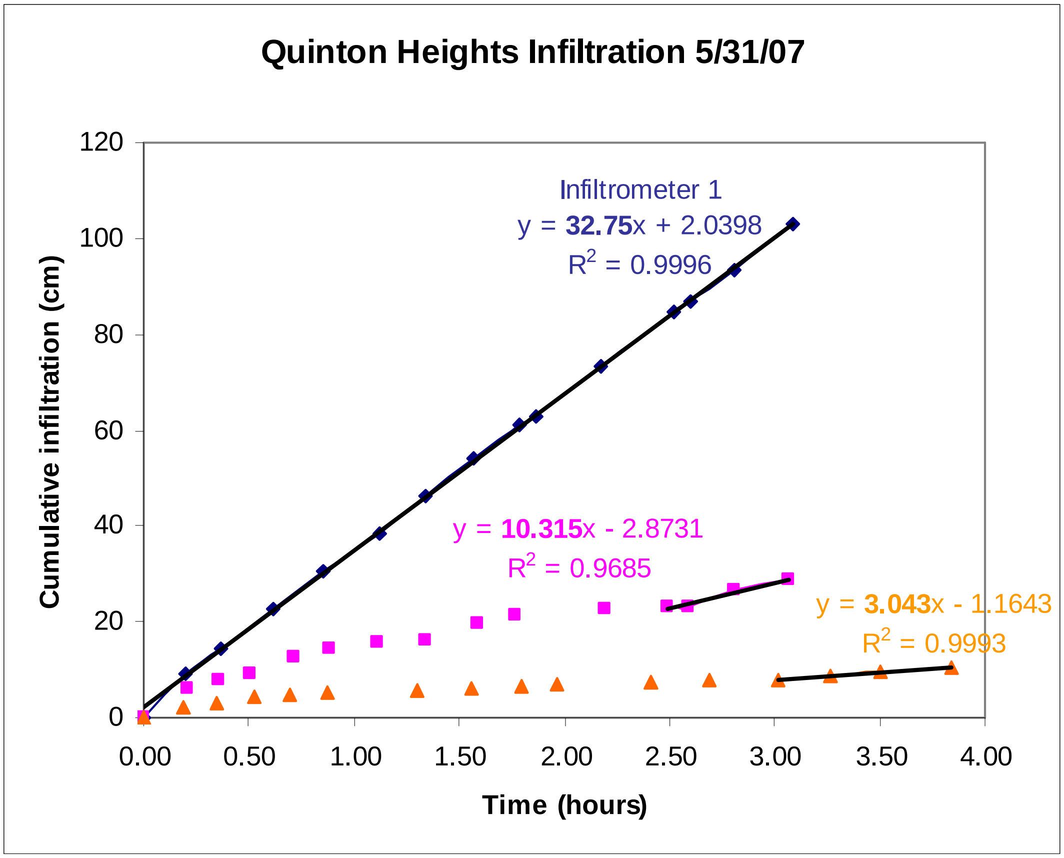

Table B.2 Results of t-test used to test for significant differences in average infiltration rates between Quinton Heights (Q.H.) and Johnson County (J.C.).

t-Test: Paired Two Sample for Means		
	Q.H.	*J.C.*
Mean	4.355	3.778611
Variance	25.09133	11.23239
Observations	6	6
Pearson Correlation	-0.4352	
Hypothesized Mean Difference	0	
df	5	
t Stat	0.197824	
P(T<=t) one-tail	0.425487	
t Critical one-tail	2.015048	
P(T<=t) two-tail	**0.850974**	
t Critical two-tail	2.570582	

Table B.3 Results of ANOVA used to test for significant differences in mean effective saturated infiltration rates among Quinton Heights (Q.H.), Johnson County (J.C.), and Fort Riley sites.

ANOVA: Single Factor — **Mean Effective Saturated Infiltration Measurements**

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
QH	6	26.13	4.355	25.09133
JCT	6	22.67167	3.778611	11.23239
FtRiley	23	58.06767	2.524681	1.978826

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	19.75007	2	9.875033	**1.403496**	**0.260457**	**3.294537**
Within Groups	225.1528	32	7.036024			
Total	244.9028	34				

Appendix C - Hydraulic Monitoring

This appendix contains a sample of inlet and outlet data from ISCO automated water samplers, along with discharge calculations. A sample of the HOBO rain gauge data is also included. The complete data set can be found at L:\WaterQuality\UrbanSites\ Trisha\ QuintonFlow_2007.

Table C.1 Inlet flow measurement data at Quinton Heights basin.

Quinton In								
Round Concrete Culvert								
Slope								0.035
Diameter (ft)								1.5
Manning's n								0.012
Bubbler Offset (ft)								0
dtTimeStamp	**dReading** m	dReading ft	Actual Depth ft	θ radians	Area ft^2	Hydraulic Radius ft	Quinton In Flow cfs	Quinton In Flow L/s
6/1/2007 1:20	0.0018	0.0059	0.0059	0.2511	0.0007	0.0039	0.0004	0.0121
6/1/2007 1:25	0.001	0.0033	0.0033	0.1871	0.0003	0.0022	0.0001	0.0034
6/1/2007 1:30	0.001	0.0033	0.0033	0.1871	0.0003	0.0022	0.0001	0.0034
6/1/2007 1:35	0.0375	0.1230	0.1230	1.1618	0.0687	0.0789	0.2935	8.3060
6/1/2007 1:40	0.0385	0.1263	0.1263	1.1777	0.0714	0.0809	0.3103	8.7809
6/1/2007 1:45	0.0328	0.1076	0.1076	1.0846	0.0564	0.0693	0.2210	6.2555
6/1/2007 1:50	0.026	0.0853	0.0853	0.9632	0.0400	0.0553	0.1349	3.8174
6/1/2007 1:55	0.0375	0.1230	0.1230	1.1618	0.0687	0.0789	0.2935	8.3060
6/1/2007 2:00	0.032	0.1050	0.1050	1.0710	0.0544	0.0677	0.2098	5.9363
6/1/2007 2:05	0.0255	0.0837	0.0837	0.9537	0.0388	0.0543	0.1294	3.6627
6/1/2007 2:10	0.03	0.0984	0.0984	1.0362	0.0494	0.0636	0.1829	5.1761
6/1/2007 2:15	0.0298	0.0978	0.0978	1.0326	0.0489	0.0632	0.1803	5.1030
6/1/2007 2:20	0.0295	0.0968	0.0968	1.0273	0.0482	0.0626	0.1765	4.9945
6/1/2007 2:25	0.023	0.0755	0.0755	0.9049	0.0333	0.0491	0.1039	2.9391
6/1/2007 2:30	0.0197	0.0646	0.0646	0.8364	0.0265	0.0422	0.0746	2.1108
6/1/2007 2:35	0.0185	0.0607	0.0607	0.8102	0.0241	0.0397	0.0652	1.8452
6/1/2007 2:40	0.0185	0.0607	0.0607	0.8102	0.0241	0.0397	0.0652	1.8452
6/1/2007 2:45	0.0158	0.0518	0.0518	0.7479	0.0191	0.0340	0.0465	1.3158
6/1/2007 2:50	0.0138	0.0453	0.0453	0.6985	0.0156	0.0298	0.0348	0.9841
6/1/2007 2:55	0.013	0.0427	0.0427	0.6777	0.0143	0.0281	0.0306	0.8656
6/1/2007 3:00	0.0128	0.0420	0.0420	0.6724	0.0139	0.0276	0.0296	0.8372
6/1/2007 3:05	0.012	0.0394	0.0394	0.6509	0.0127	0.0259	0.0258	0.7288
6/1/2007 3:10	0.01	0.0328	0.0328	0.5937	0.0096	0.0216	0.0174	0.4923
6/1/2007 3:15	0.0078	0.0256	0.0256	0.5240	0.0067	0.0169	0.0102	0.2882
6/1/2007 3:20	0.0083	0.0272	0.0272	0.5406	0.0073	0.0180	0.0116	0.3295

Table C.2. Outlet flow measurement data at Quinton Heights basin.

Quinton Out V-Notch Weir: 90 degrees				
dtTimeStamp	**dReading** m	**dReading** ft	Flow cfs	Flow L/s
6/1/2007 1:20	0	0.0000	0.000	0.000
6/1/2007 1:25	0	0.0000	0.000	0.000
6/1/2007 1:30	0	0.0000	0.000	0.000
6/1/2007 1:35	0.0096	0.0315	0.000	0.012
6/1/2007 1:40	0.1101	0.3612	0.196	5.548
6/1/2007 1:45	0.2521	0.8271	1.555	44.017
6/1/2007 1:50	0.2301	0.7549	1.238	35.033
6/1/2007 1:55	0.1758	0.5768	0.632	17.875
6/1/2007 2:00	0.1473	0.4833	0.406	11.487
6/1/2007 2:05	0.1268	0.4160	0.279	7.897
6/1/2007 2:10	0.1113	0.3652	0.201	5.701
6/1/2007 2:15	0.1051	0.3448	0.175	4.940
6/1/2007 2:20	0.1036	0.3399	0.168	4.765
6/1/2007 2:25	0.1018	0.3340	0.161	4.561
6/1/2007 2:30	0.0968	0.3176	0.142	4.021
6/1/2007 2:35	0.0893	0.2930	0.116	3.287
6/1/2007 2:40	0.0831	0.2726	0.097	2.746
6/1/2007 2:45	0.0788	0.2585	0.085	2.404
6/1/2007 2:50	0.0748	0.2454	0.075	2.111
6/1/2007 2:55	0.0708	0.2323	0.065	1.840
6/1/2007 3:00	0.0666	0.2185	0.056	1.579
6/1/2007 3:05	0.0628	0.2060	0.048	1.363
6/1/2007 3:10	0.0596	0.1955	0.042	1.196
6/1/2007 3:15	0.0558	0.1831	0.036	1.015
6/1/2007 3:20	0.0528	0.1732	0.031	0.884

Table C.3. Rainfall data recorded by HOBO tipping rain gauge.

Date	Cum. Rain Depth (cm)	Storm Depth (cm)
5/29/2007 15:58	0.40	0.40
5/29/2007 15:58	0.42	0.42
5/29/2007 16:08	0.44	0.44
5/29/2007 16:16	0.46	0.46
5/29/2007 16:31	0.48	0.48
6/1/2007 1:28	0.50	0.02
6/1/2007 1:28	0.52	0.04
6/1/2007 1:30	0.54	0.06
6/1/2007 1:31	0.56	0.08
6/1/2007 1:31	0.58	0.10
6/1/2007 1:31	0.60	0.12
6/1/2007 1:32	0.62	0.14
6/1/2007 1:32	0.64	0.16
6/1/2007 1:32	0.66	0.18
6/1/2007 1:32	0.68	0.20
6/1/2007 1:33	0.70	0.22
6/1/2007 1:33	0.72	0.24
6/1/2007 1:33	0.74	0.26
6/1/2007 1:33	0.76	0.28
6/1/2007 1:33	0.78	0.30
6/1/2007 1:33	0.80	0.32
6/1/2007 1:34	0.82	0.34
6/1/2007 1:34	0.84	0.36
6/1/2007 1:34	0.86	0.38
6/1/2007 1:34	0.88	0.40
6/1/2007 1:34	0.90	0.42
6/1/2007 1:34	0.92	0.44
6/1/2007 1:34	0.94	0.46
6/1/2007 1:35	0.96	0.48
6/1/2007 1:35	0.98	0.50
6/1/2007 1:35	1.00	0.52
6/1/2007 1:35	1.02	0.54
6/1/2007 1:35	1.04	0.56
6/1/2007 1:36	1.06	0.58
6/1/2007 1:36	1.08	0.60
6/1/2007 1:36	1.10	0.62
6/1/2007 1:36	1.12	0.64
6/1/2007 1:37	1.14	0.66
6/1/2007 1:37	1.16	0.68
6/1/2007 1:37	1.18	0.70
6/1/2007 1:37	1.20	0.72
6/1/2007 1:38	1.22	0.74
6/1/2007 1:38	1.24	0.76
6/1/2007 1:38	1.26	0.78
6/1/2007 1:38	1.28	0.80