BIORETENTION FOR PHOSPHORUS REMOVAL:
MODELLING STORMWATER QUALITY IMPROVEMENTS

by

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We cannot solve our problems with the same thinking we used when we created them.

~Albert Einstein

Man is a complex being: he makes deserts bloom - and lakes die.

~Gil Stern

Cleaning anything involves making something else dirty, but anything can get dirty without something else getting clean.

~Laurence J. Peter

We forget that the water cycle and the life cycle are one.

~Jacques Cousteau
Abstract

Bioretention systems are best management practices (BMPs) that make use of the biogeochemical processes within a forest-type ecosystem to provide at-source stormwater retention and pollutant removal. Laboratory studies and field monitoring have shown great potential for water quantity and quality control through the use of bioretention, but reported nutrient removal has been inconsistent between these systems. In particular, the processes involved in the cycling of phosphorus within bioretention systems are not clearly understood. Some studies report high phosphorus removal from bioretention systems, while phosphorus leaching was observed in other systems.

Phosphorus is a macronutrient required by all forms of life. It is also an important water pollutant, as it controls algal growth in most freshwater environments. High phosphorus loadings to these aquatic ecosystems can lead to eutrophication, which has significant ecological, environmental and economical impacts.

The Bioretention Phosphorus Removal Model (BPRM), an event-based one-dimensional finite difference model, was developed to simulate phosphorus removal in bioretention systems. The model includes four completely-mixed layers to simulate hydrologic processes as well as both soluble and particulate phosphorus transport in a bioretention system. Model processes include
evapotranspiration, infiltration, overflow, exfiltration to native soils, underdrain discharge, soluble phosphorus sorption and vegetative uptake, and particulate phosphorus capture.

Monitoring data collected by the Toronto and Region Conservation Authority (TRCA) at a bioretention system installed on Seneca College’s King City campus, in Ontario, Canada, was used to evaluate the performance of BPRM. The model was found to overestimate total underdrain discharge volumes, but total phosphorus concentration and mass predictions were found to be useful for design purposes. BPRM correctly predicted phosphorus leaching from the Seneca College bioretention system for all storm events considered but one. The model can be used by practitioners to evaluate the potential for phosphorus leaching in a bioretention system.

A detailed sensitivity analysis revealed that BPRM phosphorus transport predictions are particularly sensitive to the drainage properties of bioretention soils, which highlights the importance of hydrologic transport processes for water quality control in bioretention systems. Modelling results suggested that soluble phosphorus desorption from bioretention soils was responsible for phosphorus leaching from the Seneca College bioretention system.
Co-Authorship

Chapters 2 through 5 included in this thesis have been submitted for publication in peer-reviewed scientific journals. The work presented was carried out by Audrey Roy-Poirier, with the assistance of the following co-authors, who provided comments and suggestions, reviewed, and edited the manuscripts for publication:

- Chapter 2: Pascale Champagne and Yves Filion
  - Manuscript submitted to the Journal of Environmental Engineering, ASCE.

- Chapter 3: Pascale Champagne and Yves Filion
  - Manuscript submitted to Environmental Reviews, National Research Council Canada (NRC) Research Press.

- Chapter 4: Yves Filion and Pascale Champagne

- Chapter 5: Yves Filion and Pascale Champagne
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Chapter 1

Introduction

1.1 Background information

Stormwater management strategies have evolved considerably over the past few decades. Until the 1970s, stormwater management had been largely focused on flood prevention. Sewers were considered a means to rapidly convey stormwater away from urban areas. The first sewerage works reported by archaeologists date from circa 3000 BC. Extensive sanitary-storm sewerage works were discovered in the ruins of the Indus civilization, the Mesopotamian Empire and the Minoan Empire, amongst other early civilizations (Burian et al. 1999). Through the middle ages, the early modern period and most of the modern era, sewer construction and design saw constant improvement, but flood control remained the primary objective of stormwater management. A formal debate on the selection of separate versus combined sewers began in the 19th century. At the beginning of the 20th century, new water quality standards required the treatment of sanitary wastewater, and the debate was settled on the choice of separate sewers for economic reasons related to the volumes of water to be treated (Burian et al. 1999). It was not until many decades later that the need to also control the quality of stormwater became evident.

In 1972, the United States federal government introduced the Clean Water Act (CWA), which established the National Pollutant Discharge Elimination Systems
(NPDES) permit program under which pollution from point sources is currently regulated in the United States. The initial point sources targeted by the NPDES program between 1973 and 1976 were industrial discharges. Municipal sources of sanitary wastewater were also required to conduct biological treatment by 1977 (United States Environmental Protection Agency (USEPA) undated). In 1990, following the Amendments of 1987 to the CWA, the need to control separate stormwater quality was legally recognized by the USEPA through the introduction of Phase I of the new NPDES Stormwater program (National Research Council 2009). Phase I required permit coverage for municipal separate storm sewer systems of municipalities with populations above 100,000. Phase II of the NPDES Stormwater program was introduced in 1999, which extended the permit coverage requirements to small municipal separate storm sewer systems located in urbanized areas.

Between 1970 and 1990, following the newly introduced water discharge regulations in the U.S., the focus of stormwater management in North America shifted from improving sewer design and construction to considering alternative solutions for stormwater management. Most large North American cities had designed and constructed their main sewer systems at the turn of the 20th century, many before the separate storm sewer era. Increased urban development led to an exceedance of capacity in combined sewers in many urban areas. With costly sewer system replacement and retrofit, engineers began to explore less expensive solutions to control flows and improve water quality. Detention ponds
and other storage structures became commonly used stormwater management practices during this period (Debo and Reese 2003).

In the past two decades, stormwater management has focused primarily on the development of best management practices (BMPs). Following the introduction of the NPDES Stormwater program, pollutant removal from stormwater in separate systems has become a priority. Previously used storage systems have proved insufficient to achieve this goal. Designers have focused on the development of management practices that can provide significant water quality improvements, such as swales, wetlands and manufactured treatment systems (Debo and Reese 2003). In 1997, the Department of Environmental Resources in Prince George’s County (PGC), Maryland, introduced Low Impact Development (LID), a comprehensive approach to stormwater management (PGC 1997). LID, which aims to maintain or recreate natural hydrologic conditions in a developed area, relies heavily on the use of BMPs. The concept rests on four key activities: reducing the impact of development by minimizing impervious areas; implementing on-site stormwater management systems to collect and treat stormwater; routing stormwater through the watershed to match post-development times of concentration to pre-development times of concentration; and educating the public on pollution prevention and on the maintenance of stormwater management systems (Coffman et al. 1999). Bioretention is one of the most extensively used BMPs in LID stormwater designs (Natural Resources Defense Council (NRDC) 1999).
Bioretention systems are stormwater BMPs that were developed in the early 1990s by PGC to provide at-source treatment. The systems rely on the ecological functions of a terrestrial system of soil, plants and microbes to retain and treat stormwater. The bioretention area is excavated and native soils are replaced with high permeability soil filtration media (generally composed of a mixture of sand and organic soil). The surface of the system is graded to provide depression storage, which is filled by influent runoff during large storm events. A layer of mulch is typically placed at the surface of the system, and native vegetation is planted to promote stormwater retention and pollutant cycling inside the system. Runoff from a drainage area is directed to the bioretention system through an inlet structure. Water ponds at the surface of the system until it infiltrates the bioretention soil layers. An overflow structure diverts flows above the ponding capacity of the system during large storm events. Typically, an underdrain structure is placed below bioretention systems constructed in low permeability native soils. This prevents stormwater from ponding at the surface of bioretention systems for extended periods of time.

1.2 Scope and Objectives

Bioretention systems have gained considerable attention in recent years because of their potential for stormwater retention and pollutant removal. They have been shown to considerably reduce influent runoff volumes and peak flow rates (Davis 2008). High pollutant removal levels have also been observed from laboratory and field monitoring of these systems (Hsieh and Davis 2005; Rusciano and Obropta 2007; Li and Davis 2008; Diblasi et al. 2009; Li and Davis 2009).
However, nutrient removal in bioretention systems has been inconsistent between systems. In particular, high discrepancies in phosphorus removal have been reported from the field monitoring of bioretention systems. In one study, influent phosphorus concentrations in two bioretention systems were reduced by an average of 74% and 68% (Davis 2007). Conversely, significant phosphorus leaching from bioretention systems has also been noted in a number of studies (Dietz and Clausen 2005; Hunt et al. 2006; Toronto and Region Conservation Authority (TRCA) 2008).

Phosphorus is one of the major macronutrients required for life. It is relatively abundant in soils, but it cycles slowly through the environment as it does not have a naturally-occurring gaseous form. Because it is often found to limit the productivity of freshwater environments (Schindler 1977; Correll 1999), phosphorus is also a water pollutant of concern around the world. Phosphorus-limited water bodies undergo eutrophication under high phosphorus loadings. Excessive algal growth and decay deplete oxygen in aquatic environments, such that water quality is degraded and fish habitat can no longer be sustained. Important economic and ecological consequences, including increased water treatment costs and a loss of biodiversity, arise from the eutrophication of aquatic environments.

It is now widely recognized that integrated phosphorus pollution management is required in order to protect sensitive water bodies from the consequences of eutrophication (Heathcote 1998). Nonetheless, urban stormwater remains
untreated in many municipalities, because of challenges associated with the treatment of non-point source pollution (Schindler and Vallentyne 2008). Bioretention systems, which are inexpensive at-source stormwater BMPs, could become a preferred treatment option for urban runoff. However, the inconsistent levels of phosphorus treatment reported in field monitoring studies of bioretention systems are a barrier to their widespread use for stormwater treatment. There is a need for a greater understanding of the phosphorus cycle within bioretention systems if these systems are to be used for phosphorus pollution abatement in stormwater runoff.

In this thesis, a model is developed to assess the potential for phosphorus pollution control in stormwater through the use of bioretention systems. The primary objective of the work presented is to provide a numerical tool which can be used to predict phosphorus transport and removal in bioretention systems. The purpose of the model will be to assist designers in ensuring that planned bioretention facilities can provide the required level of phosphorus pollution control. As such, the focus will be on the development of a simple and user-friendly model which is accessible to bioretention system designers. A secondary objective of the model will be to provide insight into the importance of different phosphorus cycling processes in bioretention systems.

In Chapter 2, the evolution of bioretention systems is reviewed, along with the current state of research on the performance of these systems. Bioretention design guidelines and implementation issues are also examined. A review of

6
bioretention modelling work is provided. The chapter concludes with an overview of research needs to facilitate the widespread adoption of bioretention systems for stormwater management.

Chapter 3 presents a review of the information and data required to build a phosphorus transport model for bioretention systems. The different forms of phosphorus that occur in environmental systems are discussed, along with classifications of phosphorus forms that should be considered in bioretention system modelling. The bioretention phosphorus cycle is introduced and bioretention processes are examined. The discussion concentrates on the significance of specific processes in bioretention systems and their mathematical representation. Finally, models currently available to simulate phosphorus transport within mitigation systems similar to bioretention are reviewed. The applicability of these models to simulate phosphorus transport within bioretention systems is discussed.

In Chapter 4, an event-based model developed to simulate both soluble and particulate phosphorus transport in bioretention systems is presented. Modelling objectives and model development are discussed in detail, followed by a presentation of the structure and mathematical framework of the model. The performance of the model is then evaluated against field data collected by the Toronto and Region Conservation Authority (TRCA) at a bioretention system constructed on the King City campus of Seneca College, in the Greater Toronto Area (Ontario, Canada). Both hydrologic and phosphorus transport modelling
predictions are compared to collected field measurements to assess the performance of the model.

In Chapter 5, the sensitivity of the model presented in Chapter 4 to input parameter selection is investigated. Both hydrologic and phosphorus transport predictions are examined through input parameter perturbations and a Monte Carlo Simulation (MCS) analysis. In addition, the importance of different bioretention system processes to phosphorus removal is explored and the significance of model results is discussed. A short discussion of potential model improvements is included.

Finally, Chapter 6 reviews the engineering contribution of the work presented in this thesis, and contains a summary of the findings and some recommendations for future work.
1.3 References


Chapter 2 – A Review of Bioretention System Research and Design: Past, Present, and Future

Manuscript submitted to the ASCE Journal of Environmental Engineering, currently under review.

2.1 Abstract

This chapter reviews the evolution of bioretention systems, a promising at-source stormwater best management practice (BMP). The introduction of bioretention systems in the 1990s by Prince George’s County, Maryland, is examined, along with the motivations behind the development of the systems. A summary of the research findings on the performance of bioretention systems is provided, including proposed design modifications to improve field performance. Also included is an overview of past and current bioretention design guidelines in North America, as well as a discussion of issues surrounding the public adoption and implementation of bioretention systems. Potential alternative uses for the systems are highlighted and a review of bioretention modelling work is provided. Finally, the chapter outlines research needs and anticipated future work necessary to bring about the widespread use of bioretention systems.

2.2 Introduction

Stormwater management strategies have evolved considerably in the past few decades. Until the 1990s, stormwater management largely focused on flood prevention – initially through the use of combined sewers and separate stormwater sewers, and then through the use of detention basins and master planning (Debo and Reese 2003). In 1990, the need to control stormwater quality
was legally recognized by the United States Environmental Protection Agency (USEPA) through the introduction of the National Pollutant Discharge Elimination Systems (NPDES) Stormwater program (National Research Council 2009). This led to the development and adoption of stormwater best management practices (BMPs) to treat storm flows from urban areas.

Stormwater BMPs are used to mitigate the impacts of urban development on the quantity and quality of stormwater. Two types of BMPs can be distinguished: structural and non-structural BMPs (USEPA 1999a). Structural practices include infiltration, filtration, detention and retention systems, wetlands and other vegetated systems, as well as water quality treatment devices. Non-structural BMPs consist of maintenance, housekeeping and disposal practices, such as street sweeping, outreach programs and land-use planning.

The first structural stormwater BMPs to be commonly employed were detention basins, storage tanks, and wet ponds. These presented the advantages of being simple to design and construct, providing effective peak flow reductions, and moderately improving water quality. In order to meet NPDES stormwater quality regulations, however, systems which could significantly reduce stormwater pollutant loads were required. Constructed wetlands and other systems which rely on ecosystem functions to improve stormwater quality gained popularity.

Wetlands act as buffers in landscapes, storing or releasing water in times of flood or drought. Wetland soil and plants naturally filter water by retaining pollutants through sedimentation, filtration, sorption, plant uptake and storage, and
microbial decomposition (Champagne 2008). An important limitation associated with the use of wetlands for stormwater treatment is the specific hydrologic conditions required to sustain vegetation adapted to wetland environments (USEPA 2004). The volume and intensity of storm flows vary greatly, which can hinder the ability of wetlands to maintain the specific soil moisture range required for wetland vegetation to survive.

A recent BMP which has gained considerable attention in the last decade is the bioretention system. Bioretention systems consist of small areas which are excavated and backfilled with a mixture of high-permeability soil and organic matter designed to maximize infiltration and vegetative growth, and are covered with native terrestrial vegetation. The vegetation is selected to be resistant to environmental stresses and can range from small plants and shrubs to large trees, depending on the size of the bioretention facility. A layer of mulch is often added to cover the soil media and retain solids. An inlet structure is created to route urban runoff from the surrounding area to the unit, while an overflow structure bypasses flows above the ponding capacity of the unit. In regions having native soils of low permeability, an underdrain structure is installed at the bottom of the facility to prevent water from standing in the unit for extended periods of time. Figure 2.1 shows a typical bioretention system cross-section with the main design elements labelled.
Bioretention is a promising technology that, like wetlands, relies on ecological interactions in a natural system to provide stormwater retention and pollutant removal. Unlike wetlands, however, bioretention systems rely on terrestrial forested ecosystems and are designed to drain within hours (Prince George's County (PGC) 2007). The systems have the potential to be used in a wide variety of environments, due to the high tolerance to changing hydrologic regimes of bioretention vegetation (Coffman et al. 1993b). Another advantage of bioretention systems is their ability to significantly reduce stormwater volumes through infiltration and evapotranspiration. The systems can thus be used in urban areas to counteract the stormwater volume increase associated with urban development.
This chapter describes the evolution of bioretention systems, from their initial development to potential future improvements. The chapter highlights the reasons that motivated the development of bioretention systems, the improvements made to system designs, and the work ahead before bioretention systems are used widely. Research activities on the performance, potential uses, and suggested design modifications of bioretention systems, as well as implementation issues, public adoption and design guidelines for these systems are examined.

2.3 Development of Bioretention Systems

Bioretention systems, also referred to as rain gardens, are at-source structural stormwater BMPs developed in the early 1990s by Prince George’s County, Maryland (Coffman et al. 1993b). Bioretention systems were designed to treat runoff as sheet flow before it would drain away from the site. The initial bioretention system design consisted of an excavated area backfilled with planting soil underlined by a thin layer of sand and planted with native grass, shrub and tree species. The systems were introduced as a means of treating the “first flush” runoff from urban areas, which is the initial portion of runoff in a storm that carries disproportionately large pollutant loads. The concept of bioretention systems for stormwater treatment was inspired by similar natural systems used to treat sewage effluents. The land application of effluents from wastewater treatment relies on natural processes within the soil and vegetation to trap and cycle nutrients. Bioretention is an attempt to “maximize all available
physical, chemical and biological pollutant removal processes found in the soil and plant complex of a terrestrial forested community” (Coffman et al. 1993b).

The development and early adoption of bioretention systems were motivated by a number of beneficial characteristics of the systems. Aside from their expected efficiency in reducing storm flows and retaining pollutants, bioretention units can be integrated in urban developments and can provide at-source treatment. The installation of these systems is inexpensive and little system maintenance is required after installation. The cost of the construction of a bioretention facility that treats runoff from a parking lot of 0.3 ha (0.8 acres) was estimated at $6,500 (1993 USD), which is approximately a third of the cost of an oil and grit separator to treat stormwater for the same area (Coffman et al. 1993a). Bioretention systems can improve site aesthetics, reduce noise, and provide shade and wind cover. The areas required for the systems to effectively reduce storm flows generally fall below site landscaping requirements. According to the initial bioretention design guidelines established by Prince George’s County (1993), a bioretention unit covering 5% of the impervious drainage area could allow the first 1.27 cm (0.5 in) of runoff from the area to infiltrate. Commercial and industrial site developments in Prince George’s County use an average of 6% of the total area as green space, such that bioretention can easily be incorporated into site landscaping (Coffman et al. 1993b).
2.4 Research on Bioretention System Performance

Extensive research has been conducted in both laboratory and field studies to assess the performance of bioretention systems in storm flow retention and pollutant removal. This section reviews the main findings of these studies, including proposed bioretention system design modifications and findings from optimization studies on bioretention system components.

Comparing the performance of stormwater BMPs installed in different geographic locations and receiving different hydraulic and stormwater pollutant loads can be complicated. BMP performance is often reported as a reduction percentage in pollutant concentrations. However, while simple and intuitive, this performance indicator can be misleading. Lower removal percentages are generally obtained when influent stormwater quality is good. This can be incorrectly interpreted as poor BMP performance. Mass pollutant removals are better performance indicators, as they account for both concentration and stormwater volume reductions. However, BMP hydraulic and pollutant loadings, as well as pollutant detection limits, can influence the reported performance of BMPs. Researchers have proposed different indicators to facilitate BMP performance comparisons (Strecker et al. 2001; Barrett 2005). In this review, mass pollutant removals are reported whenever available.

2.4.1 Hydrologic Impact

A series of column experiments was performed to study the infiltration behaviour of bioretention systems over long time periods (Le Coustumer et al. 2007). The
hydraulic conductivity of the soil media was found to decrease significantly over the first four weeks of the experiment, after which it tended towards a constant value. To represent undersized bioretention systems, storm inflow volumes to some columns were doubled. The decrease in hydraulic conductivity observed in all columns was accelerated in the undersized columns due to the high hydraulic loading rate.

The results of a study over 49 storm events of two field-scale bioretention facilities installed at the University of Maryland demonstrated that bioretention can effectively reduce the impacts of development on hydrologic regimes in urban areas (Davis 2008). Significant reductions in stormwater volumes were reported, with no outflow detected for 18% of the storm events monitored. Stormwater inflow from these smaller storm events (all smaller than 0.5 m³/m²) was entirely captured by the bioretention cells. Mean peak flow reductions of 49% and 58% were noted for the cells monitored, respectively. Significant increases in times to peak were also reported, by an average factor of 5.8 for one cell, and 7.2 for the second cell. Longer peaking times better mimic pre-development hydrology in drainage basins.

Hunt et al. (2006) reported similar results from the study of a field-scale bioretention cell in North Carolina over a 12-month period. The annual average ratio of outflow volume to estimated runoff volume was calculated as 0.22 for the bioretention cell. Volume ratios were also estimated for different seasons of the year in North Carolina, with a mean ratio of 0.07 for the summer months.
compared to 0.54 for the winter months. These results indicate that the hydrologic performance of bioretention systems is highly dependent on seasonal conditions. Higher outflow volume ratios are observed during cold months as the rate of evapotranspiration in bioretention systems is reduced at lower temperatures.

Dietz and Clausen (2005) reported overflows for only 0.8% of total inflow volumes during their study of two field-scale rain gardens in Connecticut over a period of 12 months. The study period included a cold winter with frequent soil frosts, which suggests that infiltration can effectively occur in bioretention cells even under cyclical freeze-thaw soil conditions. Further research is required to assess the behaviour of bioretention systems under frozen soil conditions.

2.4.2 Nitrogen and Phosphorus Removal

Nutrients, nitrogen and phosphorus in particular, are pollutants of primary concern for the protection of aquatic ecosystems. High nutrient loadings can lead to the eutrophication of receiving water bodies as excessive algal bloom and decay deplete dissolved oxygen in the water. In coastal and oceanic waters, nitrogen is generally found to be a limiting nutrient for algal growth, while phosphorus tends to become limiting in freshwater systems (Howarth and Marino 2006).

Bioretention box experiments were performed by Davis et al. (2006) to investigate the nutrient removal potential of bioretention systems. Total
phosphorus removals ranging from 70 to 85% were noted (corresponding to an average mass removal of 82%), while 55 to 65% of total Kjeldahl nitrogen (TKN) was removed (or an average of 86% on a mass basis). Poor nitrate reduction was observed, with occasional nitrate production. A series of tests performed with different runoff inflow characteristics showed that increases in runoff duration and intensity both resulted in a reduction in nutrient removal due to an increase in flow rate through the bioretention soil. Changes in runoff pH (both increases and decreases from neutral pH) resulted in a phosphorus release in the upper soil media portion, but the outflow phosphorus concentrations were not affected by the pH changes, as runoff buffering occurred within the soil depth. Runoff pH had little influence on TKN removal, but nitrate removal was significantly decreased under higher and lower pHs. There is no evidence in the current literature that aqueous nitrate speciation or microbial denitrification rates are significantly influenced by pH changes in the range tested, and more research is required to confirm the results reported.

Results from additional laboratory and pilot bioretention studies (Davis et al. 2001; Hsieh and Davis 2005a; Hsieh and Davis 2005b) showed moderate to poor ammonia and nitrate reductions, with nitrate production observed in some instances. Davis et al. (2001) suggested that the poor nitrate removal observed may be due to ammonification and nitrification processes occurring in the bioretention unit between storm events. Since bioretention systems are designed to drain rapidly, aerobic conditions should exist inside bioretention soils between storm events. Aerobic conditions are required for nitrification, while
ammonification can be carried out by both aerobes and anaerobes. However, under low soil moisture conditions, denitrification rates are considered negligible as they are favoured by anaerobic conditions (Stevenson and Cole 1999). Nitrate can thus accumulate inside unsaturated bioretention soils, to be released during subsequent storm events.

An increase in total phosphorus concentration was consistently observed in the underdrain of two rain gardens constructed in Haddam, Connecticut, over a monitoring period of 12 months (Dietz and Clausen 2005). An exponentially decreasing trend in underdrain phosphorus concentrations was observed over time, while phosphorus concentrations in the inlet decreased linearly with time. The phosphorus export was attributed to a disturbance of the soil, but a similar trend in nitrogen concentrations over time was not reported. Redox potential measurements in the rain gardens indicated conditions favourable to denitrification, but nitrate levels were not significantly reduced in the rain gardens. No significant difference between inlet and outlet concentrations of TKN and organic nitrogen were noted, but ammonia levels were significantly reduced at the rain garden outlets, with 84.6% mass removal.

Hunt et al. (2006) reported inconsistent nutrient removal efficiencies from their monitoring of bioretention cells in North Carolina. While total nitrogen removal efficiencies were similar for two of the bioretention cells (40% on a mass basis for both cells), removal levels for nitrogen components varied greatly. Nitrate mass removal in the first cell was 75%, but TKN and ammonia production were noted,
with increases of 4.9% and 0.99% on a mass basis, respectively. In the second cell, nitrate was reduced by 13%, but TKN and ammonia removals of 45% and 86% on a mass basis were noted, respectively. Small pockets of saturated soil, favourable to denitrification processes, were observed in the bioretention media of the first cell, which may explain its higher capacity for nitrate removal. The soil media of the second bioretention cell, a more uniform sand with low organic content, was more favourable to ammonia retention. Total phosphorus removal in one of the cells reached 65% on a mass basis, while total phosphorus in the outflow of the second cell was increased by 240%. The level of phosphorus present in the soil media used for each bioretention cell was measured prior to installation. The soil media of the first bioretention cell was found to have a low phosphorus index, indicative of a low level of soil phosphorus, while a high phosphorus index was measured in the media of the second cell. It was suggested that the presence of phosphorus in bioretention filtration media may greatly influence phosphorus transport within the media. Soils with high phosphorus levels have lower adsorption capacities, as phosphorus levels in the soil approach saturation.

Hatt et al. (2007b) investigated the performance of bioretention systems under variable wetting and drying cycles through bioretention column tests. Semi-synthetic stormwater (prepared using a mixture of stormwater basin sediment and chemicals) was used for testing to accurately represent the composition of pollutants in stormwater. An inverse relationship between the level of moisture in the system and its infiltration capacity was observed, as might be expected.
Removal of most stormwater pollutants (sediment, heavy metals and phosphorus) was unaffected by the hydrologic cycles, but higher concentrations of nitrogen were measured in the outflow following extended dry periods, which could be due to a build-up of ammonia in the soil during dry periods.

Henderson et al. (2007) observed nutrient removal in vegetated and non-vegetated bioretention columns under synthetic runoff applications. Partial organic carbon removal was observed in all systems (from 28% to 66% of total influent loading). Vegetation had little influence on organic carbon removal, but higher removal rates were observed in systems with loam as the soil media (58% carbon mass removal on average) compared to sand and gravel, which were nearly identical at an average carbon mass removal of 30%. Nitrogen and phosphorus leaching was observed from the non-vegetated systems when they were flushed with tap water, but little leaching occurred in vegetated systems. These results emphasize the importance of vegetation to nutrient removal in bioretention systems.

Kim et al. (2003) suggested that introducing a saturated soil zone at the base of a bioretention system could create an anoxic zone favourable to microbial denitrification. The anoxic zone is created by introducing an upward bend in the cell underdrain to prevent water from exiting the lower portion of the bioretention cell. Testing of potential solid substrates for bioretention cells indicated that newspaper shredding was a good electron donor to support denitrification. Column studies and pilot-scale bioretention tests confirmed that
this substrate could contribute to the reduction of nitrate levels in stormwater. Nitrate mass removals of up to 80% were noted.

The field experiments reported by Dietz and Clausen (2006) did not show any conclusive evidence of improvements in nitrate removal when introducing a saturated water zone in bioretention cells. Similarly, the results of the field-scale monitoring of two bioretention cells on the University of Maryland campus, one of which was designed to include an anoxic layer, showed no significant difference in the nitrate reductions provided by the two cells (Davis 2007). However, in this case, unexpectedly high mass removals of nitrate at 90% and 95% were reported, which were attributed to naturally occurring saturated zones with high denitrification potential within the soil of both cells, possibly created by the high hardwood mulch content of the soil media used. Total phosphorus removals of 79% and 77% on a mass basis were achieved for the two respective cells.

Hsieh et al. (2007) suggested that conditions favourable to nitrification and denitrification processes could be created by introducing layers of high and low permeability media in the bioretention soil, which would impact drainage conditions between storms. Column tests were performed on two different configurations of three-layered soil media: the first column used a layer of sand, covered by a layer of soil and a layer of mulch; while the second column was designed with a layer of soil covered by a layer of sand and topped by a mixture of mulch, soil, and sand. A higher ammonia removal efficiency was obtained with
the second column (59% on a mass basis compared to 13% for the first column), which was attributed to the high cation exchange capacity of the second soil configuration due to its higher mulch content. Nitrate export, attributed to nitrification processes occurring between runoff applications, was noted in both columns. The study confirmed that nitrification and denitrification processes can be promoted in a bioretention cell by introducing a low permeability layer below a higher permeability layer in the bioretention soil media.

The short-term effectiveness of bioretention media in retaining phosphorus was investigated through laboratory column studies by Hsieh et al. (2007). They reported a total phosphorus removal of 85% on a mass basis for high conductivity soil filtration media, while low conductivity media typically showed lower phosphorus retention potential (65% on a mass basis).

The influence of cold temperatures on nutrient removal in bioretention systems was investigated by Blecken et al. (2007) in a bioretention column study. High removals of particulate-bound nutrients was observed (total suspended solids (TSS) were reduced by 97%) independently of temperature. However, poor nitrogen removal was observed, with high levels of nitrate production. The removal of dissolved nitrogen constituents was highly dependent on temperature. Influent dissolved nitrogen concentrations increased from 1.16 mg/L to 1.33 mg/L at 2°C (an increase of 14.9%), and 3.94 mg/L at 20°C (a 240% concentration increase). Dissolved phosphorus removal was not significantly influenced by temperature.
Zhang et al. (2008) investigated the potential of using different materials as soil media in bioretention systems to improve phosphorus removal. Based on sorption tests performed, fly ash was identified as a material with significant potential for phosphorus sorption. Because of the low permeability of fly ash, the use of sand amended with 5% of fly ash on a mass basis was suggested as the optimal mixture to improve phosphorus retention in bioretention systems. Desorption tests indicated negligible amounts of phosphorus leaching from the mixture under low concentration influents, while 42% of previously sorbed phosphorus leached from a non-amended sand sample.

A series of bioretention column experiments was performed to determine the optimal characteristics of a bioretention system for nutrient removal (Bratieres et al. 2008). Certain plant species (*Carex appressa* and *Melaleuca ericifolia*) were found to provide significantly higher nitrogen removals than other species tested, which suggests that vegetation selection is an important parameter in bioretention design for nutrient removal. High phosphorus removal (around 85%) was consistently observed from the bioretention columns, but reduced removal efficiencies (around 40%) were noted for columns with soils high in organic content due to a net production of orthophosphate ($\text{PO}_4^{3-}$) in these columns.

In a similar study, Read et al. (2008) investigated the influence of 20 different Australian plant species on stormwater pollutant removal in bioretention systems. The concentration of nutrients in the effluent of bioretention test
columns varied greatly (by factors of 2 to 4 for total phosphorus and total nitrogen to factors of more than 20 for ammonia and nitrate) based on the plant species used in the sample. A comparison of the pollutant removal capacity per root mass of each plant revealed even greater variations (by factors of 18 to 50 for total phosphorus and nitrogen to a factor of 4150 for nitrate and ammonia).

2.4.3 Oil and Grease Removal

Bioretention systems are commonly used as stormwater treatment options in parking lots and roadways, where runoff can be collected and treated along medians and shoulders (USEPA 1999b). Designers are thus interested in understanding the potential of bioretention cells in removing stormwater pollutants emanating from vehicle emissions, in particular oil and grease, as well as heavy metals (USEPA 1999a).

Bioretention column tests were performed on 18 soil media of varying compositions to identify the optimal bioretention soil for pollutant removal (Hsieh and Davis 2005b). Oil and grease removal above 96% was reported for all soil media (based on a synthetic influent oil and grease concentration of 20 mg/L). The oil and grease removal capacity of eight existing bioretention cells in Maryland was also evaluated to corroborate laboratory results. Synthetic runoff with an oil and grease concentration of 20 mg/L was applied to six of the cells, while the last two cells were monitored during a storm event with high inflow oil and grease concentrations. For all cells, oil and grease removals above 99% were noted, similar to the results obtained in the laboratory study.
Because oil and grease tend to be persistent pollutants, their accumulation within bioretention cells is of concern, as the capacity of the cell for oil and grease retention may be lost once saturation is reached within the cell. Hong et al. (2006) investigated the potential for oil and grease retention and biodegradation in a mulch layer. They found removal levels between 83% and 97% for the hydrocarbon contaminants tested. Complete biodegradation of the hydrocarbon contaminants sorbed in the mulch layer was observed after 3 to 10 days, which suggests that the use of a mulch layer in bioretention systems could minimize the accumulation of the hydrocarbons investigated. However, to date, no research has been published on the potential for oil and grease biodegradation in bioretention media.

2.4.4 Heavy Metal Removal

Copper, lead and zinc were selected as representative heavy metal species for testing heavy metal removal in a series of bioretention pilot studies (Davis et al. 2001; Davis et al. 2003). Removal efficiencies greater than 97% on a mass basis were observed for each of the metals tested. Core samples extracted from one of the pilot-scale bioretention units after 31 synthetic runoff applications indicated that the mulch layer was responsible for significant levels of metal uptake. A series of bioretention tests with varying runoff characteristics demonstrated that runoff pH, duration, intensity, and heavy metal concentrations have little impact on metal removal efficiencies in bioretention systems. Synthetic runoff with copper, lead and zinc concentrations of 0.08 mg/L was also applied to two field-scale bioretention cells in the cities of Greenbelt and Largo, Maryland. The
Greenbelt cell, which showed a high infiltration capacity, removed 97%, 95%, and 95% of copper, lead and zinc inputs, respectively. The Largo cell, which was constructed more recently with coarser soil filtration media, removed only 43%, 70%, and 64% of copper, lead and zinc inputs, respectively. This was attributed to the lower filtration capacity of the coarse soil media, as well as the lower maturity of the vegetation in the more recently installed bioretention cell.

Glass and Bissouma (2005) measured inlet and outlet concentrations of an extensive array of heavy metals in a parking lot bioretention cell over a 3-month period. They reported removal efficiencies of 81% for copper, 66% for cadmium, 79% for zinc, 53% for chromium, 75% for lead, 17% for aluminum, 11% for arsenic, and 53% for iron. It was suggested that lower removal efficiencies were observed in this field experiment than in laboratory studies due to a lack of maintenance of the bioretention system. The differences in removal efficiencies observed for different metals suggest that certain metal species, in particular aluminum and arsenic, do not undergo the same removal processes as other heavy metals such as copper and zinc, which are often selected as representative heavy metal species. Aluminum and arsenic influent and effluent concentrations were near detection limits in this study, which may have influenced the accuracy of the concentration measurements reported. Also, only cadmium and lead influent concentrations exceeded the drinking water maximum contaminant levels prescribed by the USEPA (2003).
The efficiency of bioretention systems in removing heavy metals under cold climate conditions was examined by researchers from the Norwegian University of Science and Technology and Lulea University of Technology in Sweden. Bioretention box experiments using synthetic runoff were performed in the months of April and August, with average daily temperatures of 4.0°C and 13.2°C, respectively (Muthanna et al. 2007b). Metal retention was good during both months leading to a 90% removal of zinc on a mass basis in April and August. Warmer temperatures favoured removals of lead from 83% to 89% on a mass basis, and removals of copper from 60% to 75% on a mass basis. To test the capacity of bioretention cells for heavy metal removal from snowmelt, another series of bioretention box experiments was performed by applying snow collected from three sites with different expected pollutant concentrations resulting from roadway traffic (Muthanna et al. 2007c). Zinc, copper, lead, and cadmium removals in the range of 81 to 99% on a mass basis were reported in all cases. Generally, an increase in pollutant concentration compared to the average pollutant concentration in the snowpack can be measured in the first flush of snowmelt water (Schondorf and Herrmann 1987). Bioretention cells were found to reduce this enrichment ratio for heavy metals compared to enrichment ratios reported for untreated snowmelt.

One of the concerns related to heavy metal removal in bioretention systems is the limited capacity of the systems to store these metals. A potential solution involves the removal of metals from bioretention systems through the harvest of metal-saturated plant biomass. Sun and Davis (2007) studied the capture of heavy
metals in bioretention system components using planter-pot prototypes with three plant species selected to uptake large quantities of metals through high biomass yields. They reported that 88 to 97% of all input metals were captured within the soil media, while accumulation in plant tissue only accounted for 0.5 to 3.3% of the metals. Higher concentrations of metals were measured in the plant roots than in the shoots. The small fraction of metals accumulated in plant biomass was attributed to low biomass densities. This suggests that higher biomass yields than those obtained during this series of experiments are required to make plant harvesting a viable solution for heavy metal accumulation in bioretention systems.

Australian researchers studied the potential of bioretention systems as a treatment technology for stormwater reuse through columns tests with semi-synthetic stormwater applications (Hatt et al. 2007a). Six different filter media were tested to compare their pollutant removal potential, and total copper, lead and zinc removals above 90% were noted for all soil mixtures. Bioretention shows promise as a treatment technology for irrigation use (the most common application of stormwater reuse) since heavy metals are the pollutants of concern for this application.

### 2.4.5 Total Suspended Solids Removal

The presence of suspended solids in stormwater is generally a good indicator of the presence of organic matter, particulate nutrients, heavy metals and other pollutants. Suspended solids can also clog stormwater conveyance systems or

Hsieh and Davis (2005a) performed a series of 12 bioretention column tests using synthetic runoff. Sediment washout was observed during the first 6-hour run, but a total TSS mass removal of 91% was noted after the test series.

In comparison, field monitoring of two bioretention facilities over a period of 14 months at the University of Maryland showed low levels of TSS removal (Davis 2007). Event mean concentrations (EMCs) for 23 storm events were reduced by an average of 41% by one cell, while the second cell, which included an anoxic zone at its base, reduced influent EMCs by an average of 22%. The first cell retained 54% of influent TSS, while 59% of influent TSS was retained in the second cell on a mass basis. Sediment washout was also observed in the field study as TSS effluent concentrations exceeded influent concentrations, mostly during the first storm events following the initiation of the system. Early inputs to the bioretention units appeared to contribute to the stabilization of the soil media.

The filtration mechanism of bioretention cells was investigated by Li and Davis (2008b) through a series of bioretention column tests and field observations. It was found that, as particulate matter is captured by bioretention media, stratification tends to occur in the media. Finer soil particles were found in the upper media layer as suspended solids were trapped inside the media, altering its characteristics and reducing its hydraulic conductivity. Based on the results of the
study, it was concluded that the useful lifespan of bioretention filter media is limited by clogging, which is expected to occur prior to TSS breakthrough. Annual to biannual bioretention soil media replacement to a depth of 5 to 20 cm was suggested as a potential preventive measure for media clogging based on the laboratory results. However, the roles of the mulch layer and vegetation in bioretention media clogging were not considered in this study.

2.4.6 Removal of BOD and Pathogens

If discharged in a receiving aquatic environment, organic matter, measured in stormwater by its biological oxygen demand (BOD$_5$), can lead to oxygen depletion as aerobic digestion occurs. BOD$_5$ measurements of the influent and effluent of a field-scale bioretention cell in North Carolina were collected during 23 storm events (Hunt et al. 2008). The average EMC for all storm events was reduced from 8.54 mg/L BOD$_5$ in the inflow to 4.18 mg/L in the outflow, a 63% concentration reduction.

While pathogen counts may be significantly smaller in stormwater than in sanitary wastewater, the presence of pathogens in urban runoff is of concern to regulatory agencies because of the health hazards they present to both humans and aquatic species. Column tests performed by Rusciano and Obropta (2007) suggested that bioretention systems have the capacity to significantly reduce the presence of pathogens in stormwater. For 13 experiments containing a range of fecal coliform counts, an average reduction of 91.6% was obtained, with removal efficiencies ranging from 54.5 to 99.8%. The results of a two-year monitoring
project on a bioretention cell constructed in North Carolina confirmed the ability of bioretention cells to reduce pathogens (Hunt et al. 2008). An average reduction in fecal coliform counts of 69% was observed, along with a reduction in *E. coli* counts of 71%.

### 2.4.7 Removal of Polycyclic Aromatic Hydrocarbons (PAHs)

A study of 16 PAHs considered as priority pollutants by the USEPA was conducted by Dibiasi et al. (2009) on a bioretention cell installed at the University of Maryland, College Park campus. The mean total PAH concentration was reduced from 2.08 µg/L in the inflow of the bioretention system to 0.22 µg/L in the outflow, corresponding to a 90% mean reduction in concentration. Outflow concentrations were similar for all events and higher removal percentages were noted for events with high influent concentrations, suggesting that bioretention cells can decrease PAH concentrations to a specific threshold, independently of influent concentrations. Average PAH outflow concentrations may be a more indicative measure of the performance of bioretention systems. An average PAH mass load reduction of 87% was noted over the study period. Core analyses of the soil media suggested that shallow bioretention depths may be sufficient to reduce PAH loads, as soil PAH concentrations were an order of magnitude higher in the top 10 cm of the soil media compared to samples taken at greater depths.

### 2.4.8 Effect on Water Temperature, pH, and Dissolved Oxygen

A study of a parking lot bioretention system in Washington D.C. (Glass and Bissouma 2005) showed consistent increases in effluent dissolved oxygen
concentrations from the unit, even when influent dissolved oxygen concentrations were above saturation levels for the temperatures measured. This result was unexpected under slow groundwater flow conditions, and may be worthy of further study. The pH of the bioretention unit influent and effluent were also monitored. Although the region received acid rain (in the pH range of 5 to 6), near-neutral influent pHs were measured, indicating that rainfall pH was buffered while running off, and no significant changes in pH were observed in the effluent of the bioretention unit. Results from laboratory studies using different influent pHs showed that bioretention soil media has the capacity to significantly buffer pHs within the range of 6.0 to 8.0 (Davis et al. 2003; Davis et al. 2006).

The influent and effluent temperatures of two rain gardens in Haddam, Connecticut, were monitored by Dietz and Clausen (2005). An annual average increase in water temperature was measured in the effluent, but it was determined that the difference in temperature between influent and effluent was not statistically significant and fluctuations in effluent temperatures could be correlated to seasonal temperature variations. Jones and Hunt (2008) investigated the potential to reduce stormwater thermal pollution to receiving water bodies through the use of bioretention. Maximum influent stormwater temperatures to four bioretention systems in North Carolina were significantly reduced at the outflow of the systems after the water travelled through the bioretention soils. Median influent temperatures were also significantly decreased by the two bioretention systems with areas above 10% of their drainage area. Bioretention systems which cover large areas relative to their contributing
watershed benefit from a larger soil mass to absorb the heat from influent stormwater. Also, large bioretention areas can reduce surface ponding times, which are susceptible to increase the water temperature. The results show that bioretention can decrease, but not eliminate, stormwater thermal pollution.

2.5 Bioretention Design Guidelines and Implementation Issues

Bioretention guidance documents produced to assist planners, designers, contractors and landowners have evolved significantly since the introduction of bioretention systems in the 1990s. This section examines the evolution of bioretention design guidelines over the last two decades and provides an overview of current regulations pertinent to bioretention systems across North America. Political, regulatory and social issues surrounding the implementation of bioretention systems are also discussed.

2.5.1 Design Guidelines

As bioretention systems gain popularity across the United States, design guidelines are being developed and implemented by a number of states. A review of some of the most extensive guidelines reveals five main sizing methods for bioretention systems. The states of Georgia, Maryland, New York and Vermont require that bioretention facilities be sized based on a volume of runoff to be treated to meet water quality objectives, where filter bed sizing is based on Darcy’s law (Maryland Department of the Environment 2000; Atlanta Regional Commission 2001; Vermont Agency of Natural Resources 2002; New York Department of Environmental Conservation 2008). North Carolina adopted the
initial sizing guidelines proposed by Prince George’s County in 1993, which are based on the Rational Method for peak runoff (North Carolina Department of Environment and Natural Resources 2007). Bioretention design guidelines for the states of Virginia and Idaho require that bioretention areas cover a specific percentage of the total impervious drainage area (Virginia Department of Conservation and Recreation 1999; Idaho Department of Environmental Quality 2005). The state of Delaware requires bioretention system geometry to meet necessary loading rates (Delaware Department of Natural Resources and Environmental Control 2005). Finally, Wisconsin bioretention design guidelines recommend the use of the RECARGA model to size bioretention facilities (Wisconsin Department of Natural Resources (WDNR) 2006). This model was developed at the University of Wisconsin to determine the hydrologic impact of bioretention systems (Atchison and Severson 2004). Requirements for system configuration and the selection of bioretention components also vary across guidelines, but system designs remain fairly consistent overall. In most states, bioretention sizing requirements have been selected to be consistent with sizing requirements for other BMPs. However, the design approaches based on a fixed area percentage and on a specific loading rate may not appropriately address both water quantity and quality objectives.

In Canada, no current federal legislation oversees stormwater discharge quality to freshwater receiving bodies. Provinces are responsible to develop guidelines for discharge water quality requirements, and municipalities can adopt more stringent guidelines at their own discretion. Most Canadian provinces have no
guidelines on the implementation of stormwater management infrastructure. In Ontario, the Stormwater Management Planning and Design Manual (Ontario Ministry of the Environment 2003) offers some guidance on bioretention facility design, based on experience from other jurisdictions. Little field experience has been reported in the province.

The current design guidelines available from Prince George’s County, Maryland, build on the first bioretention guidelines published in 1993 and draw from extensive research work to develop and improve bioretention systems in the area (PGC 2007). A design storm is selected based on a specific Low Impact Development (LID) criterion for the area. The facility is sized to provide sufficient storage in order to maintain pre-development runoff volumes and peak runoff for the storm selected. Pre and post-development hydrologic conditions are determined based on the SCS curve number method (Soil Conservation Service 1969). Aside from introducing this new bioretention sizing method, some important changes in bioretention design from the initial 1993 PGC guidelines have been implemented. The requirement for a sand bed below the filtration soil has been removed, and the initial definition of a bioretention system has been expanded to include facilities with various design purposes, shapes, slopes and locations, and vegetation types. Different bioretention designs have been defined for particular applications such as groundwater recharge and filtration of highly contaminated stormwater, and bioretention themes have been introduced to facilitate the integration of bioretention in site landscaping.
A software tool has been developed to assist engineers and planners in the sizing of bioretention facilities according to design guidelines adopted by different states (T.E. Scott & Associates, Inc. 2008). In the development of this sizing software, four different sizing methods were compared based on a hypothetical project. A drainage area of 0.24 ha (0.60 acre) with 0.16 ha of impervious area (0.40 acre) was assumed for comparison purposes, and area sizing was based on a bioretention media depth of 0.76 m (30 in) and soil void ratio of 0.30. Large discrepancies between requirements for each sizing approach were found. While the fixed percentage sizing methods adopted by the states of Virginia and Idaho required bioretention filter beds to cover an area of 40.5 m² to 80.9 m² (436 ft² to 871 ft²), the sizing method described in the PGC bioretention guidelines required a bioretention surface of 140 m² (1508 ft²), which represents a difference in size of over 230%. The sizing methods based on Darcy’s law and on the rational method yielded areas of 110 m² (1180 ft²) and 85 m² to 119 m² (915 ft² to 1281 ft²), respectively.

Researchers in Norway and Sweden (Muthanna et al. 2007a) investigated eight current methods of sizing bioretention facilities under snow storage conditions and found that snow storage became an important sizing parameter, particularly in inland areas. The RECARGA model, along with field observations, was used to compare the performance of bioretention systems sized according to each method. A bioretention area corresponding to 8% of the impervious drainage area was found to suit both rainfall runoff and snow storage requirements for coastal climates. Larger snow storage requirements are expected in inland areas,
where intermittent snowmelt occurs less frequently throughout the cold season. This study suggests that current design guidelines may not be suitable for use in cold climates and that guidelines should take into account regional climatic factors such as snow accumulation and soil frost conditions.

2.5.2 Implementation Issues

Poor bioretention system construction practices have been a major implementation concern. Most contractors are unfamiliar with bioretention system construction, which has led to improper soil mixture selection or placement and poor vegetation establishment (Cosgrove and Bergstrom 2004; Toronto and Region Conservation Authority 2008). In addition, Carpenter and Hallam (2009) showed that infiltration rates in bioretention systems are highly influenced by construction techniques as well as bioretention soil mixtures. Poor construction techniques led to the development of preferential flow paths within the soil media of a bioretention cell (composed of 80% compost and 20% sand), which drastically affected the hydrologic performance of the cell. Stormwater volume retention in the cell for a 2.3 cm rainfall event decreased from an average of 92.0% to an average of 40.2% due to the establishment of preferential flow paths. Under the same conditions, a bioretention cell constructed with a soil media mixture of 20% compost, 50% sand and 30% topsoil displayed a fairly consistent stormwater volume retention capacity, at an average of 82%.

The process of integrating bioretention systems in the stormwater management plan of a new Wisconsin subdivision was studied by Morzaria-Luna et al. (2004).
The regulatory and political frameworks were identified as factors that encouraged the adoption of BMPs such as bioretention facilities. Bioretention practices were welcomed by governing bodies under pressures from NPDES regulations and resident concerns regarding the protection of local water bodies. It was found, however, that current stormwater management regulations did not provide sufficient flexibility for the inclusion of bioretention systems as standalone stormwater management structures. Due to a lack of knowledge and confidence in bioretention systems from the regulatory bodies, a great deal of redundancy was incorporated in the stormwater system as traditional components were added to meet established design guidelines. Current regulatory approaches which discourage the use of non-traditional stormwater management systems need to be revised if bioretention systems are to become widely used for stormwater management.

Another issue identified by Morzaria-Luna et al. (2004) was that of system ownership. While traditional centralized stormwater management systems can easily be maintained and monitored by the municipality, stakeholders believed it would be impractical to enforce proper maintenance of bioretention systems installed on privately-owned land. In the end, bioretention areas were not included in the final stormwater management design plans. However, a deed restriction requiring homeowners to construct a rain garden with the assistance of a landscaping company and to maintain the rain garden was added to all lots of the subdivision. In North Carolina, a rain garden implementation and public education project revealed that most landowners did not have a good
understanding of the installation and maintenance requirements associated with rain gardens (Woodward et al. 2008). In all, 73 rain gardens installed in the region were surveyed two years after installation. At that time, 23% of the gardens had been abandoned or were no longer functioning, while another 44% were in need of maintenance.

A related public adoption issue is the risk posed by bioretention systems to public health, particularly with regards to breeding of mosquitoes and other disease vectors. Research has shown that some mosquito species can complete their breeding cycle in less than a week (Metzger 2004). Consequently, current design guidelines require bioretention systems to drain within 72 hours to minimize mosquito breeding (PGC 2007). A survey of 37 BMPs in Southern California revealed the presence of immature mosquitoes at 4 of 6 biofiltration swales surveyed and 1 of 3 biofiltration strips surveyed. Mosquitoes were collected on 7% of the visits to these 5 sites (Metzger et al. 2008). Additional research may be required to confirm whether or not bioretention systems promote mosquito breeding.

2.6 Bioretention Modelling Research

As highlighted in the previous section, current bioretention design guidelines lack consistency across regions. While climatic and geographic factors can affect the performance of systems, and treatment objectives vary across jurisdictions, there is a need for modelling tools to accurately predict the hydrologic and water
quality performance of bioretention system designs and verify the suitability of current guidelines.

The RECHARGE model, developed by researchers at the University of Wisconsin (Dussaillant et al. 2004), is the most comprehensive model for bioretention system hydrologic performance currently available. The model includes surface ponding, a maximum of three soil layers and an optional underdrain and relies on the Richard’s equation to simulate infiltration. RECARGA, an alternative version of the RECHARGE model which uses the less complex Green-Ampt equation to represent infiltration (Dussaillant et al. 2003), has been adopted by the state of Wisconsin in its bioretention design guidelines (WDNR 2004). RECARGA is a MATLAB application which allows for both continuous and single-event modelling in bioretention facilities. Based on bioretention system characteristics and user-specified precipitation and evapotranspiration data, the model computes water balance terms including inflow, outflow, evaporation, underdrain flow, soil moisture, ponding times, and number of overflow events.

Heasom et al. (2006) developed a hydrological model within the HEC-HMS environment to predict the hydrologic impact of a bioretention facility. Sub-basins (permeable and impermeable basins) were introduced in the model to represent the bioretention system drainage area. Input rainfall is converted to runoff inflow series by the model based on the sub-basin characteristics. The model includes a diversion element which simulates variations in infiltration rates caused by changes in soil moisture and hydrologic conditions. The
bioretention system is represented by a reservoir combined with a weir, and a stage-discharge relationship is used to control the reservoir outflow, which corresponds to the bioretention system underdrain flow. The model was applied to monitoring data from a bioretention cell located on the Villanova University campus, and accurately predicted water levels in the cell. Similar models have been developed within the SWMM modelling software environment and using different programming languages (Poresky and Palhegyi 2008; Lucas 2009).

Bioretention water quality modelling has only addressed a very limited number of stormwater pollutants to date. One model was developed to simulate the one-dimensional filtration of suspended solids through bioretention media based on laboratory and field experiments on urban particulate capture (Li and Davis 2008c). Three layers (a cake layer, a working zone, and a pristine zone) of changing depth are included in the MATLAB model to represent stratification processes caused by TSS capture. The cake layer depth is the depth of the homogenous layer of solids formed at the bioretention media surface. The working zone depth is the depth of media containing any captured solids estimated with the suspension concentration profile. The model accounts for both cake layer filtration in the top layer and depth filtration in the lower layers. An equation to simulate bioretention media replacement was also included in the model. Input data includes time series of inflow rate and TSS concentration, as well as bioretention media characteristics such as depth, area, and hydraulic conductivity. Using laboratory results, the calibrated model accurately predicted
TSS removal and the hydraulic conductivity of the soil media. Media replacement simulation results also closely matched laboratory results.

A similar model was developed to investigate the spatial profile and partitioning patterns of heavy metal captured in bioretention soils (Li and Davis 2008a). The one-dimensional model combines a first-order filtration equation for particulate metals and advection, dispersion and adsorption transport equations for dissolved metals. Predicted heavy metal profiles were in good agreement with the measured metal concentrations in a bioretention system installed in the District of Columbia. A bioretention water quality model is also being developed by He and Davis (He and Davis 2009) to simulate the fate of two PAH compounds, naphthalene and pyrene, along with hydrologic processes within bioretention systems. The partial differential equations in the COMSOL Multiphysics modelling environment are being used to build this model.

2.7 Knowledge Gaps and Research Needs

Extensive research has been conducted on the performance of bioretention systems, both in the laboratory and in the field. Beyond field monitoring, future research should explore and attempt to optimize nutrient removal processes within bioretention cells. Design modifications to improve nutrient removal within these systems have been proposed: inclusion of an anoxic layer to improve nitrate removal (Kim et al. 2003); addition of a low permeability layer below a higher permeability layer in bioretention soil media to promote nitrification and denitrification processes (Hsieh et al. 2007); and fly-ash soil amendment to
increase phosphorus sorption (Zhang et al. 2008). However, little quantification of the improved efficiency or treatment benefits of these modifications has been reported to date. More work should be performed to identify how conditions favourable to nitrification, denitrification, or phosphorus sorption can be introduced in bioretention cells. A thorough investigation of the impact of such modifications on the removal of other pollutants within the system should also be conducted.

There is also some concern that pollutant accumulation in bioretention systems, in particular heavy metals and oil and grease accumulation, could negate the potential of the cells to retain stormwater pollutants over time, or even cause pollutant leaching as pollutant concentrations exceed system capacity (Davis et al. 2006; Sun and Davis 2007). Long-term field monitoring of bioretention cells is required to explore this issue.

A wide range of phosphorus removals has been reported for bioretention systems monitored in the field, and possible explanations related to soil disturbances and soil phosphorus content have been suggested (Dietz and Clausen 2005; Hunt et al. 2006). Additional research work is required to better understand the mechanisms for phosphorus removal within bioretention systems, to identify factors that significantly influence phosphorus removal, and to determine the cause of phosphorus leaching often observed in the field. Future bioretention design guidelines should be adapted based on the findings of this research to reduce the risks of phosphorus leaching in the environment.
A number of studies have been published on the performance of bioretention cells under cold climate conditions, including hydrologic performance of bioretention cells under cold climatic conditions, snow storage requirements in bioretention areas, and snowmelt treatment efficiencies for heavy metal removal (Dietz and Clausen 2005; Muthanna et al. 2007a; Muthanna et al. 2007b; Muthanna et al. 2007c). However, there is still uncertainty surrounding the implementation of bioretention cells in climates that differ significantly from that of the Eastern United States, where the systems were initially developed and tested. There is a lack of knowledge on the performance of bioretention systems in arid or semi-arid climates. Also, the infiltration capacity of bioretention areas under frozen soil conditions and the selection of vegetation for areas subject to severe frost should be investigated further. Further field work is also needed to confirm the potential of bioretention areas to treat snowmelt, particularly in areas with large snow accumulation.

No comprehensive models are available to date to predict the effect of bioretention systems on stormwater hydrology and effluent quality. There is a need for a model that allows designers to optimize the characteristics of a bioretention unit to meet specific hydrologic and water quality objectives. Such a model could be used to validate design procedures across regions by comparing the performance of systems designed according to different guidelines. Also, regions sensitive to nutrient loadings could benefit from a tool to predict nutrient removal in bioretention cells, since contradicting efficiencies have been reported in the literature. Models combining hydrologic and water quality components
would allow researchers to investigate specific processes within bioretention units and determine the relative importance of cell components and processes.

While extensive research work on the technical performance of bioretention cells has been published, little information is available on implementing bioretention systems, obtaining permits from regulatory bodies, promoting public adoption, and educating landowners on bioretention system maintenance. Locating bioretention systems on privately-owned land has been identified as a barrier to their adoption in stormwater management plans, as it requires the enforcement of proper installation, use and maintenance of the cells (Morzaria-Luna et al. 2004). Other implementation issues include mosquito breeding concerns and construction challenges. Research focused on solutions to these problems is needed to facilitate the use of bioretention systems as stand-alone stormwater management practices.

The potential of LID designs (some incorporating bioretention systems) for stormwater management (NRDC 1999; Clar and Rushton 2002; Hood et al. 2007; Zimmer et al. 2007) and combined sewer overflow management (Patwardhan et al. 2005; Weinstein and Jones 2005; Zhen et al. 2006) has been studied. However, the potential use of bioretention systems in urbanized areas to increase the capacity of combined sewer systems and reduce pollutant loads to water bodies has not been examined extensively.

Results from MODFLOW simulations performed by Endreny and Collins (2009) suggest that groundwater levels can be raised significantly over long time periods
by properly placed bioretention systems within a watershed. There is a need to examine in greater depth the ability of bioretention to increase groundwater recharge in areas that rely heavily on groundwater as a drinking water source. The related issue of flooding or damaging of infrastructure located near bioretention systems also requires further investigation.

While stormwater management systems are typically designed based on expected storm flows, urban dry weather discharges (from excessive irrigation, incidental discharges, and other urban activities) have been found to contribute a significant portion of the total runoff volume and pollutant load to be handled by stormwater infrastructure in some regions (Duke et al. 1999; McPherson et al. 2002). Numerical modelling performed by Tyagi et al. (2008) suggests that LID technologies can be used to manage dry weather flows in semi-arid climates. However, the performance of bioretention systems at managing urban dry weather flows has not been directly investigated.

Current design guidelines suggest that bioretention systems can be used in areas with high levels of runoff contaminants, often referred to as pollution “hotspots”. This can be done if an impervious liner is placed below the bioretention soil media to prevent groundwater contamination (PGC 2007). The use of bioretention systems for brownfield restoration has also been suggested (Russ 2000). However, lining a bioretention cell with an impermeable membrane was found to reduce its hydrologic performance considerably (Li et al. 2009). Additional research is required to evaluate the feasibility of treating runoff at
highly-contaminated sites through bioretention. Issues such as reduced hydrologic performance, concentration of stormwater pollutants within the soil and vegetation of the system, plant toxicity, liner tearing and groundwater contamination should be explored.

While bioretention is a recent innovation, stormwater management practices that share similarities with bioretention systems, such as swales, filter strips and detention ponds, have been used and studied extensively. Previous research on these similar systems may shed light on some of the performance and implementation issues for bioretention systems identified in this review.

2.8 Summary

Bioretention systems were introduced as a stormwater BMP by Prince George’s County in the 1990s. The performance of bioretention systems has been studied extensively, both through laboratory and field studies, and findings have confirmed the potential of bioretention systems for stormwater retention and treatment. However, uncertainty surrounding the use of bioretention systems under arid or cold climate conditions remains. High variability in reported nutrient removal efficiencies has also been noted. Design modifications have been proposed to improve the hydrologic and water quality performance of bioretention systems, but further research is required to optimize pollutant removal in bioretention systems. Current bioretention sizing guidelines were found to lack consistency. A need for a modelling tool that can compare and validate the appropriateness of current design guidelines was identified. Further
research on issues surrounding the implementation of bioretention systems is also necessary to facilitate the widespread use of bioretention systems. Limited modelling work has been performed on bioretention systems and there is a need for a comprehensive bioretention model including both hydrologic and water quality processes. Further research on the potential to use bioretention systems in applications such as dry weather flow control and combined sewer overflow management, brownfield restoration, and stormwater reuse is also required.
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Chapter 3 – Bioretention Processes for Phosphorus Pollution Control

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Chapter 3
Bioretention Processes for Phosphorus Pollution Control

3.1 Abstract

Phosphorus is a water pollutant of concern around the world as it limits the productivity of most shallow freshwater systems which can undergo eutrophication under high phosphorus inputs. Eutrophication has several undesirable ecological, environmental and economic consequences. The importance of treating stormwater as part of an integrated phosphorus pollution management plan is now recognized. Bioretention systems are stormwater best management practices (BMPs) which rely on terrestrial ecosystem functions to retain storm flows and reduce pollutant loads. Bioretention has shown great potential for stormwater quantity and quality control. However, phosphorus removal within bioretention systems has been inconsistent, with high phosphorus removals reported for some systems compared to occasional phosphorus leaching observed in other systems.

Previous research has identified some of the processes that affect phosphorus removal in bioretention systems. However, if bioretention systems are to be used for phosphorus pollution abatement, there is a need for a greater understanding of the phosphorus cycling processes within bioretention systems. Numerical simulation models are inexpensive tools that can be used to study complex systems such as bioretention facilities. The aim of this chapter is to identify and characterize bioretention phosphorus cycling processes that can be used to
develop a model for phosphorus transport within bioretention systems. The
different forms of phosphorus encountered in stormwater are described, and
classifications of phosphorus forms useful for modelling purposes are discussed.
The phosphorus cycle within bioretention systems is then presented, followed by
a discussion of bioretention processes and their mathematical representation.
Processes reviewed include dissolution and precipitation, sorption and
desorption, vegetative uptake, mineralization and immobilization, colloidal
filtration and mobilization, as well as sedimentation. Finally, approaches used to
model phosphorus transport in systems similar to bioretention are examined.

3.2 Introduction

3.2.1 Phosphorus and Eutrophication

Phosphorus is an essential nutrient for all life forms. It is required to form DNA
and cell membranes, amongst other functions (Jahnke 2000). Unlike other major
macronutrients (hydrogen, oxygen, carbon, nitrogen and sulphur), phosphorus
has no significant natural gaseous form. Because of this, phosphorus cycling
through the environment is a slow process, despite the relative abundance of the
element in soils (Botkin et al. 2006).

In many parts of the world, phosphorus is considered as a water pollutant of
concern to receiving water bodies. It has been identified as the limiting nutrient
for algal growth in most shallow freshwater environments (Schindler 1977;
Correll 1999). Phosphorus-limited lakes and rivers that receive high phosphorus
loads can become overly productive, in a phenomenon referred to as
eutrophication. Excessive algal growth and decay deplete dissolved oxygen levels (<2 mg/L) in the water body, such that water quality is degraded and fish habitats can no longer be sustained. The indicators of eutrophication include murky water with increased turbidity, the apparition of green algal mats at the water surface, and loss of biodiversity. Eutrophic water bodies have little appeal for recreation and treatment costs increase when drinking water sources become eutrophied (Wilson and Carpenter 1999).

As they age, some water bodies naturally undergo a slow increase in productivity, which can be viewed as natural eutrophication (Whiteside 1983). However, changes in global nutrient cycles associated with urban development around the world have led to a phenomenon that is often referred to as cultural eutrophication, or the rapid eutrophication of water bodies under anthropogenic influence. The problem of cultural eutrophication is not a recent one. Evidence of eutrophication has been observed in water bodies near populated regions since the beginning of the 20th century (Hasler 1947). In the early 1970s, the problem had grown to such an extent in North America that legislation was introduced to radically reduce the concentration of phosphate in detergents, which accounted for nearly 50% of the phosphorus in wastewater (Knud-Hansen 1994). The apparition of algal blooms in lakes and rivers is still a concern today (e.g. Government of Canada 2009). While phosphorus concentrations in wastewater discharges have been reduced, untreated non-point sources of phosphorus from urban developments have increased phosphorus loadings to receiving water bodies (Schindler and Vallentyne 2008).
Cyanobacteria, often referred to as blue-green algae, are recognized as one of the organisms that thrive under eutrophic conditions (Vasconcelos 2006). Toxins released by cyanobacteria have been responsible for chronic health effects and death in animals and, more rarely, humans (Fristachi and Sinclair 2008). Cyanobacterial toxins have also been identified as a carcinogenic substance (Nishiwaki-Matsushima et al. 1992) which has revived the need to control phosphorus loadings to receiving water bodies in many municipalities.

3.2.2 Bioretention for Phosphorus Control

Wastewater discharges have traditionally been recognized as the main source of pollution to receiving water bodies, while contributions from stormwater to the degradation of waterways have been mostly disregarded (Novotny 1995). As wastewater treatment technologies have improved and wastewater discharge regulations have become more stringent, the impact of stormwater discharges as a source of phosphorus to water bodies has become more evident (Ice 2004). Stormwater quality management is now recognized as an important part of integrated approaches to phosphorus pollution management (Heathcote 1998).

Bioretention systems are recent stormwater best management practices (BMPs) used to retain stormwater and reduce pollutant loads at or near their source. Bioretention systems were developed to take advantage of the ecological functions within a terrestrial ecosystem system of soil, microbes and plants. These systems use a layer of high permeability soil to promote stormwater infiltration while filtering and retaining pollutants. Woody and herbaceous native
vegetation that can tolerate drought and flood conditions is planted in the systems to promote pollutant cycling. The bioretention soil media is generally covered by a layer of mulch to encourage vegetative growth and to capture particulate pollutants. An underdrain structure is also installed below the systems to ensure prompt drainage when native soils have low permeabilities.

Bioretention systems are becoming increasingly popular in Low Impact Development (LID) designs, as they have been shown to provide effective at-source stormwater retention, peak flow attenuation, and pollutant removal. In particular, as discussed in Chapter 2, laboratory and field studies have shown high levels of suspended solid, heavy metal, oil and grease and fecal coliform removals through bioretention. In contrast, phosphorus removal has been highly inconsistent in bioretention systems, with occasional leaching observed (Dietz and Clausen 2005; Hunt et al. 2006; Line and Hunt 2009). Table 3.1 summarizes reductions in phosphorus concentrations and loadings obtained from the field monitoring of different bioretention systems.
<table>
<thead>
<tr>
<th>Location</th>
<th>Design characteristics</th>
<th>Average inflow concentration (mg/L)</th>
<th>Average outflow concentration (mg/L)</th>
<th>Load reduction (%)</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Garden 1, Haddam, CT</td>
<td>Lined</td>
<td>0.019</td>
<td>0.058</td>
<td>-117&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Dietz and Clausen (2005)</td>
</tr>
<tr>
<td>Garden 2, Haddam, CT</td>
<td>Lined</td>
<td>0.019</td>
<td>0.060</td>
<td>-104&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Dietz and Clausen (2005)</td>
</tr>
<tr>
<td>Cell A, College Park, MD</td>
<td>Lined, saturated</td>
<td>0.61&lt;sup&gt;c&lt;/sup&gt;</td>
<td>0.15&lt;sup&gt;c&lt;/sup&gt;</td>
<td>79</td>
<td>Davis (2007)</td>
</tr>
<tr>
<td>Cell B, College Park, MD</td>
<td>Lined</td>
<td>0.61&lt;sup&gt;c&lt;/sup&gt;</td>
<td>0.17&lt;sup&gt;c&lt;/sup&gt;</td>
<td>77</td>
<td>Davis (2007)</td>
</tr>
<tr>
<td>Cell G1, Greensboro, NC</td>
<td>Saturated</td>
<td>0.11</td>
<td>0.56</td>
<td>U/A</td>
<td>Hunt et al. (2006)</td>
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<td></td>
<td>0.10</td>
<td>3.00</td>
<td>-240</td>
<td>Hunt et al. (2006)</td>
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<td>U/A</td>
<td>65</td>
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<td>Cell L2, Louisburg, NC</td>
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<td>0.25</td>
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<td>0.24</td>
<td>69</td>
<td>Sharkey (2006)</td>
</tr>
<tr>
<td>Rocky Mount, NC</td>
<td>Grassed, saturated</td>
<td>0.18</td>
<td>0.06</td>
<td>67</td>
<td>Brown et al. (2009)</td>
</tr>
<tr>
<td>Charlotte, NC</td>
<td></td>
<td>0.19</td>
<td>0.13</td>
<td>U/A</td>
<td>Hunt et al. (2008)</td>
</tr>
<tr>
<td>King City, ON, Canada</td>
<td>Lined</td>
<td>0.115&lt;sup&gt;d&lt;/sup&gt;</td>
<td>5.089</td>
<td>U/A</td>
<td>Toronto Region and Conservation Authority (2008)</td>
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<tr>
<td>North Cell, Graham, NC</td>
<td>Grassyed</td>
<td>0.137</td>
<td>0.051</td>
<td>53</td>
<td>Passeport et al. (2009)</td>
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<tr>
<td>South Cell, Graham, NC</td>
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<td>0.137</td>
<td>0.058</td>
<td>68</td>
<td>Passeport et al. (2009)</td>
</tr>
</tbody>
</table>

<sup>a</sup>: *Lined* refers to a bioretention system enclosed in an impermeable membrane liner, *Saturated* refers to a bioretention system that includes a saturated zone below the underdrain to promote anaerobic processes, and *Grassed* refers to a bioretention system that is covered with grass rather than a layer of mulch and plants.

<sup>b</sup>: calculated based on the data reported

<sup>c</sup>: median concentration

<sup>d</sup>: concentration in runoff from a control area
Phosphorus removal in bioretention systems is not clearly understood. However, certain bioretention processes have been shown to significantly influence the transport of phosphorus within bioretention systems. The results of bioretention box experiments performed by Davis et al. (2006) emphasized the importance of hydrologic processes for phosphorus removal in bioretention systems. Phosphorus was removed to a lesser extent under increased influent runoff duration or intensity, due to a decrease in the stormwater residence time inside the bioretention boxes. Phosphorus leaching from bioretention systems has been attributed to the desorption of phosphorus from bioretention soil media with a high phosphorus content, which is considered near saturation (Hunt et al. 2006). Bratieres et al. (2008) also observed phosphorus leaching from bioretention columns when soils were amended with organic matter (with either 10% compost or 10% mulch). The column tests performed by Henderson et al. (2007) suggested that vegetation plays an important role in the removal of phosphorus in bioretention systems. When test columns were flushed with tap water, significant phosphorus leaching was observed from non-vegetated columns, while vegetated columns retained phosphorus well. Significant differences in phosphorus removal were also achieved in bioretention columns planted with different vegetative species (Bratieres et al. 2008; Read et al. 2008), suggesting that some plant species have a higher phosphorus retention capacity than others. The role of vegetation for phosphorus cycling in bioretention systems may extend beyond the uptake of nutrients for biomass production, as the improvements in phosphorus retention observed by Lucas and Greenway (2008) in vegetated bioretention mesocosms compared to unvegetated mesocosms were considerably
greater than anticipated vegetative uptake rates. This observation may be the result of increased microbial activity in vegetated mesocosms, since nutrients exuded by plant roots are known to stimulate microbial colonization in the soil rhizosphere (Bais et al. 2006).

To reduce the risk of phosphorus leaching from bioretention systems, which can have important environmental consequences in phosphorus-limited ecosystems, designers require a greater understanding of the phosphorus cycle within bioretention systems. Modelling is an inexpensive means of studying complex natural systems such as bioretention facilities. Simulation models can be used to identify critical bioretention processes and to predict the phosphorus removal potential of a bioretention system. A thorough literature review has shown that, currently, no model exists to simulate phosphorus transport in bioretention systems.

Extensive literature is available on carbon and nitrogen cycling in soil systems. In comparison, many aspects of the phosphorus cycle in soils are not clearly understood. A large number of the organic compounds found in soils have not yet been identified (Brady and Weil 2007) and measurements of the rate of microbial activity in soils are complicated by the tendency of released orthophosphate to quickly precipitate with minerals in solution or sorb onto soil particles (Stevenson and Cole 1999). Modelling the phosphorus cycle in soil-based systems is thus a challenging task. This chapter aims to collect the information required for the development of a phosphorus transport model for bioretention systems.
In this chapter, processes involved in the transport and cycling of phosphorus in bioretention systems are reviewed with an emphasis on process modelling. The main purpose of this chapter is to synthesize relevant information which can later be used to develop a numerical simulation model for phosphorus transport within bioretention systems. The forms of phosphorus present in stormwater are first reviewed. The phosphorus cycle within bioretention systems is then presented, followed by a discussion of individual bioretention phosphorus cycling processes. The discussion focuses on describing the importance of different processes within bioretention systems and presenting modelling equations to represent each process. Finally, a number of phosphorus transport models developed for systems that share similarities with bioretention facilities are reviewed.

This chapter focuses on defining biogeochemical phosphorus cycling processes within bioretention systems. While also critical to the development of phosphorus transport models, hydrologic processes will not be discussed at length in this chapter; nor will advection, dispersion, diffusion, and seepage pollutant transport processes. These are well described in other sources (e.g. Charbeneau 2000; Ramaswami et al. 2005; Kemna et al. 2006).

3.3 Forms of Phosphorus in Stormwater

Sources of phosphorus in stormwater include lawn fertilizers, atmospheric deposition, automobile exhaust, soil erosion, animal waste, and detergents (United States Environmental Protection Agency (USEPA) 1999). Phosphorus
from these sources appears in different chemical and physical forms. For modelling purposes, the following distinctions between phosphorus forms are useful:

1. **Soluble phosphorus vs. particulate phosphorus**

Processes involved in the transport and cycling of soluble and particulate phosphorus differ because of their physical form. Particulate phosphorus can be transported by stormwater under flows of sufficient velocity, but it is subject to sedimentation and filtration. In comparison, soluble phosphorus is found in solution in stormwater and undergoes sorption and vegetative uptake, amongst other reactions.

The size delimitation between soluble and particulate phosphorus is usually considered to be 0.45 µm because, for analytical purposes, the two phosphorus fractions are typically separated using 0.45 µm pore filters (van Loon and Duffy 2005). In reality, the division between particulate and soluble material is not absolute as colloids may also be present in a sample. Colloids are intermediate particles which cannot be seen by the naked eye and are suspended (rather than dissolved) in the liquid phase (Birdi 2008). Figure 3.1 shows the delimitations between soluble, colloidal and particulate phosphorus based on the particle sizes given by van Loon and Duffy (2005).
A further classification of phosphorus into soluble, colloidal and particulate phosphorus forms may be useful for modelling purposes, as the behaviour of colloids tends to deviate from that of macro-sized particles (Gustafsson and Gschwend 1997). This deviation is due to the high specific surface area of colloidal particles that is commonly associated to charge effects (Russel et al. 1989). Colloids thus tend to attract or repulse dissolved pollutants and other colloids, which influences their filtration and sedimentation behaviours.

2. **Organic phosphorus vs. inorganic phosphorus**

Organic and inorganic phosphorus forms have different origins and chemical compositions, which influences the cycling processes they typically undergo. Important inorganic phosphorus compounds in water include orthophosphate (HPO$_4$ and its dissociated forms), as well as condensed phosphates, which are formed by bonding a number of orthophosphate molecules (Snoeyink and Jenkins 1980). In natural systems, organic phosphorus originates from biological processes, such as plant and microbial biomass production. Soil organic matter contains
approximately 0.5% phosphorus by weight (Barber 1984). Some organic phosphorus pollutants present in stormwater may also have been synthesized, such as the organic phosphorus compounds used in some pesticides (Corbridge 2000).

The division between organic and inorganic phosphorus forms is particularly useful when describing biological processes in natural systems. Soil microbes are responsible for the conversion of organic phosphorus forms to inorganic forms, which then become available for plant uptake. Soluble orthophosphate is considered the most readily available form of phosphorus for vegetative uptake (Schachtman et al. 1998). On the other hand, plant and microbial biomass production converts inorganic phosphorus to organic compounds, which are returned to the soil after biomass death.

3. Available phosphorus vs. unavailable phosphorus

This differentiation is similar to that between organic and inorganic phosphorus forms. However, the definition of available and unavailable phosphorus is based on whether the element can be retrieved easily or not, particularly by vegetation. Phosphorus is considered bioavailable if it is either immediately available for vegetative uptake, or can be made available through naturally occurring processes (Boström et al. 1988). While organic phosphorus has typically been considered unavailable, it is
now becoming evident that it plays an important role in plant nutrition (McKelvie 2005).

Typical soluble and total phosphorus concentrations in stormwater for different land use types are found Table 3.2, as reported in the National Stormwater Quality Database (Pitt et al. 2004).

<table>
<thead>
<tr>
<th>Land use type</th>
<th>Total phosphorus concentration</th>
<th>Soluble phosphorus concentration</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Median (mg/L)</td>
<td>Coefficient of variation (unitless)</td>
</tr>
<tr>
<td>Residential</td>
<td>0.30</td>
<td>1.1</td>
</tr>
<tr>
<td>Commercial</td>
<td>0.22</td>
<td>1.2</td>
</tr>
<tr>
<td>Industrial</td>
<td>0.26</td>
<td>1.4</td>
</tr>
<tr>
<td>Institutional</td>
<td>0.18</td>
<td>1.0</td>
</tr>
<tr>
<td>Freeways</td>
<td>0.25</td>
<td>1.8</td>
</tr>
<tr>
<td>Open Space</td>
<td>0.31</td>
<td>3.5</td>
</tr>
</tbody>
</table>

Regulations on the total maximum daily load (TMDL) of phosphorus that may reach a water body are generally expressed in terms of total phosphorus (USEPA 2007), which is an unambiguous measurement. However, some phosphorus forms may be more detrimental to receiving water bodies than others. In particular, soluble orthophosphate can promote the rapid eutrophication of sensitive aquatic ecosystems, as it is immediately available for vegetative uptake. In comparison, some forms of unavailable particulate phosphorus may be buried by sediment before they can be converted into plant available form (Brenner et al. 2005).
For this reason, designers may be interested in differentiating different forms of phosphorus in the outflow of bioretention systems.

3.4 Phosphorus Processes in Bioretention Systems

The phosphorus cycle within a typical bioretention system is presented in Figure 3.2. Some of the processes included may not prevail in all bioretention systems because of differences in system designs. The bioretention system presented in Figure 3.2 includes an underdrain structure and a mulch layer, which may be omitted in some systems. Also, modifications to bioretention system design, such as the introduction of a saturated zone within bioretention soils (Kim et al. 2003), have been proposed to improve the performance of the systems. These may have an impact on the phosphorus cycle within the system.

Figure 3.2: Phosphorus cycle within a typical bioretention system.
The bioretention processes involved in the cycling of phosphorus in Figure 3.2 are described below, with some information on the mathematical representation of each process.

### 3.4.1 Dissolution and Precipitation

Orthophosphate is involved in a complex chemical equilibrium system in stormwater which includes acid dissociation, complexation and precipitation reactions. $\text{PO}_4^{3-}$ is the fully dissociated form of orthophosphoric acid ($\text{H}_3\text{PO}_4$), which has four aqueous protonation states. Table 3.3 shows the dissociation equilibrium constants for orthophosphoric acid in distilled water at 25 °C (after Snoeyink and Jenkins 1980). The actual dissociation constants of phosphoric acid in stormwater may be affected by the presence of other water pollutants – ionic species in particular (Stumm and Morgan 1996). Under typical stormwater pHs of 7 to 8 (Pitt et al. 2004), the dominant species are $\text{H}_2\text{PO}_4^-$ and $\text{HPO}_4^{2-}$.

<table>
<thead>
<tr>
<th>Chemical Equation</th>
<th>Dissociation equilibrium constant $pK_a$</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\text{H}_2\text{PO}_4^{(aq)} + \text{H}_2\text{O}^{(l)} \rightleftharpoons \text{H}_3\text{O}^{+ (aq)} + \text{H}_2\text{PO}_4^{- (aq)}$</td>
<td>2.1</td>
</tr>
<tr>
<td>$\text{H}_2\text{PO}_4^{- (aq)} + \text{H}_2\text{O}^{(l)} \rightleftharpoons \text{H}_3\text{O}^{+ (aq)} + \text{HPO}_4^{2- (aq)}$</td>
<td>7.2</td>
</tr>
<tr>
<td>$\text{HPO}_4^{2- (aq)} + \text{H}_2\text{O}^{(l)} \rightleftharpoons \text{H}_3\text{O}^{+ (aq)} + \text{PO}_4^{3- (aq)}$</td>
<td>12.3</td>
</tr>
</tbody>
</table>

Table 3.4 contains selected precipitation equilibrium constants for phosphate and some minerals at 25°C. Several additional solids can be formed with phosphates, and a number of complexation reactions with minerals can also be involved in the
chemical equilibrium system (Snoeyink and Jenkins 1980; Stumm and Morgan 1996). In addition, co-precipitation reactions can be observed (Reddy and D'Angelo 1994). Lastly, soil microorganisms are responsible for the production of an array of substances that increase the solubility of phosphates, thereby effectively increasing the fraction of phosphorus available to plants (Deubel and Merbach 2005).

Table 3.4: Chemical equilibrium between phosphate and selected minerals at 25°C (after Snoeyink and Jenkins 1980; Stumm and Morgan 1996).

<table>
<thead>
<tr>
<th>Solid formed</th>
<th>Chemical equation</th>
<th>Solubility equilibrium constant pKso</th>
</tr>
</thead>
<tbody>
<tr>
<td>Calcium hydrogen phosphate</td>
<td>CaHPO$_4$(s) $\rightleftharpoons$ Ca$^{2+}$ + HPO$_4^{2-}$</td>
<td>6.66</td>
</tr>
<tr>
<td>Calcium dihydrogen phosphate</td>
<td>Ca(H$_2$PO$_4$)$_2$(s) $\rightleftharpoons$ Ca$^{2+}$ + 2 H$_2$PO$_4^-$</td>
<td>1.14</td>
</tr>
<tr>
<td>Hydroxyapatite</td>
<td>Ca$_5$(PO$_4$)$_3$OH(s) $\rightleftharpoons$ 5Ca$^{2+}$ + 3PO$_4^{3-}$ + OH$^-$</td>
<td>55.9</td>
</tr>
<tr>
<td>Tricalcium phosphate (β crystal form)</td>
<td>Ca$_3$(PO$_4$)$_2$(s) $\rightleftharpoons$ 3Ca$^{2+}$ + 2PO$_4^{3-}$</td>
<td>24.0</td>
</tr>
<tr>
<td>Ferric phosphate</td>
<td>FePO$_4$(s) $\rightleftharpoons$ Fe$^{3+}$ + PO$_4^{3-}$</td>
<td>21.9</td>
</tr>
<tr>
<td>Aluminum phosphate</td>
<td>AlPO$_4$(s) $\rightleftharpoons$ Al$^{3+}$ + PO$_4^{3-}$</td>
<td>21.0</td>
</tr>
<tr>
<td>Magnesium ammonium phosphate</td>
<td>MgNH$_4$PO$_4$(s) $\rightleftharpoons$ Mg$^{2+}$ + NH$_4^+$ + PO$_4^{3-}$</td>
<td>12.6</td>
</tr>
</tbody>
</table>

Based on the chemical equilibrium equations presented in Table 3.4 and the dominant phosphate species in stormwater of typical pH (7 to 8), the concentration of calcium ions in stormwater tends to dictate the solubility of orthophosphate species. Mean stormwater calcium concentrations of 4.8 to 26.5 mg/L have been reported, with values ranging from 0.04 mg/L to 2113.8 mg/L (Makepeace et al. 1995). However, calcium is not generally considered as a pollutant in stormwater, and therefore, its concentration is infrequently monitored and reported in the literature. Because of this lack of information on
the concentration of calcium ions in stormwater, it is often difficult to predict the equilibrium concentration of phosphate.

The precipitation of orthophosphate from solution is not instantaneous. During the initial reaction phase, which is referred to as the induction period, there is nuclei formation and the rate of phosphate precipitation is negligible. After this period, rapid precipitation occurs, but at a decreasing rate as equilibrium concentrations are approached (Snoeyink and Jenkins 1980). The rate of precipitation of calcium phosphate in slightly alkaline waters (such as stormwater) is generally slow, and has a long induction period (Ferguson et al. 1973). For this reason, predicted phosphate equilibrium concentrations may not be reached at every time step when modelling over short durations.

Because of the complexity of the orthophosphate equilibrium system in stormwater, precipitation and dissolution reactions are commonly represented with empirical equations (Kadlec and Knight 1996). De Haas et al. (2004) reviewed a number of models developed to simulate chemical precipitation in activated sludge systems. The model presented by Briggs (1996) relies on a first-order reaction equation to predict phosphorus precipitation with metals dosed for treatment:

$$P_r = P_{p0} e^{-a (Me_0:P_{p0})}$$  \hspace{1cm} (3.1)

where $P_r$ is the residual orthophosphate concentration after precipitation [M/L^3]; $P_{p0}$ is the initial orthophosphate concentration [M/L^3]; $Me_0$: $P_{p0}$ is the ratio of metal dosed for treatment to initial orthophosphate concentration [M/M]; and $a$
is a constant related to removal stoichiometry [M/M]. Although this equation does not capture all of the phosphate equilibrium system complexities in stormwater, it could be adapted for use in bioretention systems.

### 3.4.2 Sorption and Desorption

Sorption refers to the attachment of a solute onto a sorbent. It encompasses the processes of physical sorption (both absorption and adsorption), chemisorption, and anion exchange (Sample et al. 1980). In bioretention systems, soluble phosphorus can be sorbed onto mulch (if present) and bioretention soil particles. Sorption of soluble phosphorus onto colloids suspended in stormwater or in soil water can also occur (Sharpley et al. 1981; Uusitaloa et al. 2001). Desorption is the reverse process by which previously sorbed phosphorus is released back into solution. This can occur when the phosphorus content of soil matrix is high relative to the concentration of phosphorus in the soil pore solution.

Phosphorus sorption is a complex phenomenon which is considered, along with precipitation, as the main phosphorus retention process in natural treatment systems (Reed et al. 1998). Sorption can normally be divided into two distinct phases: short-term and long-term sorption. Short-term sorption is thought to be a reversible process that occurs almost instantaneously, while long-term sorption is considered to be essentially irreversible (McGechan and Lewis 2002). Desorption is generally considered as the reverse of short-term sorption and can be modelled by similar equations.
Sorption is typically modelled using isotherms. Some of the isotherms most commonly used to represent phosphorus sorption are presented in Table 3.5, along with a short description of the advantages and disadvantages associated to their use.
Table 3.5: Isotherms commonly used to model phosphorus sorption.

<table>
<thead>
<tr>
<th>Isotherm</th>
<th>Equation</th>
<th>Description</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Linear</td>
<td>$Q = k_d C$</td>
<td>$k_d$ is the distribution coefficient $[L^3/M]$</td>
<td>Limousin et al. (2007)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Simple and easy to use</td>
<td>Only valid at low or trace phosphorus concentrations</td>
</tr>
<tr>
<td>Langmuir</td>
<td>$Q = Q_{max} \left[ \frac{k_L C}{1 + k_L C} \right]$</td>
<td>$k_L$ is the Langmuir sorption coefficient $[L^3/M]$; and $Q_{max}$ is the maximum soil sorption capacity $[M/M]$</td>
<td>McGechan and Lewis (2002)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>$Q_{max}$ provides a measure of the soil sorption capacity; two-surface Langmuir isotherms can be used for better accuracy</td>
<td>Assumes monolayer sorption, which is rarely representative of the true sorption behaviour</td>
</tr>
<tr>
<td>Freundlich</td>
<td>$Q = k_F C^b$</td>
<td>$k_F$ is the Freundlich sorption coefficient; and $b$ is an exponent</td>
<td>McGechan and Lewis (2002)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Easily fitted to most soils because of its form</td>
<td>Does not consider initial soil phosphorus content or maximum sorption capacity</td>
</tr>
<tr>
<td>Temkin</td>
<td>$Q = k_{T1} \ln (k_{T2} C)$</td>
<td>$k_{T1}$ and $k_{T2}$ are coefficients</td>
<td>Goldberg (2005)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Accounts for the energy required to sorb particles based on the amount of particles already sorbed</td>
<td>Only valid at intermediate phosphorus concentrations; parameters cannot be obtained through linear regression</td>
</tr>
</tbody>
</table>

1: $Q$ is the soil phosphorus content at equilibrium $[M/M]$ and $C$ is the soil water phosphorus concentration at equilibrium $[M/L^3]$. 
For most applications, phosphorus sorption reactions can be considered instantaneous. However, when modelling bioretention systems over very short time steps (on the order of minutes or hours), equilibrium may not be reached in the system at the end of each time step (McGechan and Lewis 2002). The kinetics of the sorption reaction should be considered in these cases. Some kinetic rate models commonly used to describe sorption are presented in Table 3.6. The complexity of the three equations presented is similar, and thus selection is generally made based on goodness-of-fit with available data. Kinetic versions of most isotherm equations can also be derived to account for the rate of the observed sorption process (McGechan and Lewis 2002).

Table 3.6: Commonly used kinetic sorption rate models.

<table>
<thead>
<tr>
<th>Kinetic model</th>
<th>Equation</th>
<th>Description</th>
<th>Source</th>
</tr>
</thead>
</table>
| Pseudo-first-order rate       | \[
\frac{dq}{dt} = k_1(q_e - q)\]                                         | \(k_1\) is the first order rate constant \([\text{T}^{-1}]\); and \(q_e\) is the soil phosphorus content at equilibrium \([\text{M/M}]\) | Azizian (2004)              |
| Pseudo-second-order rate      | \[
\frac{dq}{dt} = k_2(q_e - q)^2\]                                       | \(k_2\) is the second order rate constant \([\text{T}^{-1}]\); and \(q_e\) is the soil phosphorus content at equilibrium \([\text{M/M}]\) | Azizian (2004)              |
| Elovich                       | \[
\frac{dq}{dt} = a e^{(b-q)}\]                                         | \(a\) and \(b\) are constants (unitless)                                  | Polyzopoulos et al. (1986)  |

\(1\): \(t\) is time \([\text{T}]\); and \(q\) is the soil phosphorus content at time \(t\) \([\text{M/M}]\).

The capacity of a soil to sorb phosphorus varies greatly based on the organic and clay contents of the soil, as well as the presence of aluminum and iron ions in the soil. The phosphorus sorption characteristics of soils have been correlated to physical and chemical soil properties in a number of studies (e.g. Singh and Gilkes 1991; Brennan et al. 1994; Börling et al. 2001; Zhang et al. 2005; Samadi
2006). The short-term phosphorus sorption capacity of mulch and some typical
bioretention soils was also evaluated by Hsieh et al. (2007). Recently, it was
shown that organic phosphorus can be sorbed in greater quantities and with a
higher energy than inorganic phosphorus in soils containing hydroxide minerals
(Karathanasis and Shumaker 2009).

3.4.3 Vegetative Uptake

Plants require phosphorus as a nutrient for growth and typical plant biomass
contains 0.05 to 1.0% of phosphorus by plant weight (Corbridge 2000). Phosphorus is taken up by plant roots and assimilated as biomass through
mechanisms that were reviewed by Schachtman et al. (1998). Typical plant
biomass contains 0.05 to 1.0% of phosphorus by plant weight (Corbridge 2000).

The rate of vegetative uptake is typically defined by Michaelis-Menten kinetics
(Barber 1984):

\[
I_n = \frac{l_{max} (C_i - C_{min})}{K_m + C_i - C_{min}}
\]

(3.2)

where \( I_n \) is the ion influx [mol/L²/T]; \( l_{max} \) is the maximum phosphorus uptake
rate [mol/L²/T]; \( C_i \) is the soil water phosphorus concentration [mol/L³]; \( C_{min} \) is
the minimum soil water phosphorus concentration for uptake [mol/L³]; and \( K_m \)
is the phosphorus uptake rate [mol/L³]. \( I_n \) is also sometimes expressed on a
root length or root mass basis (Engels et al. 2000). Net efflux from the vegetation
roots occurs at concentrations below \( C_{min} \).
Table 3.7 contains phosphorus uptake parameters for a few vegetative species representative of bioretention system vegetation. Phosphorus uptake rates are most commonly reported for crop plants and trees that have significance in the forestry industry, as such, little data is available for native plants commonly used in bioretention systems.

<table>
<thead>
<tr>
<th>Type of plant</th>
<th>$I_{max}$ (µmol/cm²/s)</th>
<th>$K_m$ (µmol/L)</th>
<th>$C_{min}$ (µmol/L)</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tall fescue (<em>Festuca arundinacea</em>)</td>
<td>0.001</td>
<td>5</td>
<td>1</td>
<td>Barber (1984)</td>
</tr>
<tr>
<td>Reed Canary grass (<em>Phalaris arundinacea</em>)</td>
<td>0.003</td>
<td>4</td>
<td>1</td>
<td>Barber (1984)</td>
</tr>
<tr>
<td>Ryegrass (<em>Lolium perenne</em>)</td>
<td>$3.54 \times 10^{-7}$</td>
<td>1.3</td>
<td>0.1</td>
<td>Föhse et al. (1991)</td>
</tr>
<tr>
<td>Rape (<em>Brassica napus</em>)</td>
<td>$9.79 \times 10^{-7}$</td>
<td>0.4</td>
<td>0.1</td>
<td>Föhse et al. (1991)</td>
</tr>
<tr>
<td>Loblolly pine (<em>Pinus taeda</em>)</td>
<td>$2.68 \times 10^{-7}$</td>
<td>16</td>
<td>0.6</td>
<td>Kelly et al. (1992)</td>
</tr>
</tbody>
</table>

Factors that influence the rate of phosphorus uptake in plants include the proportion of plant roots that are exposed to phosphorus, plant and root age, as well as environmental conditions including temperature and soil pH (Barber 1984). In addition, increased phosphorus uptake rates are generally observed in plants that are colonized by mycorrhizal fungi, as a symbiotic relationship exists between plant roots and mycorrhizae which increases the availability of phosphorus to the plant (Schachtman et al. 1998).

### 3.4.4 Mineralization and immobilization

Mineralization is the process by which microbes convert organic phosphorus into orthophosphate, thus making it available for plant uptake (Stevenson and Cole...
Mineralization is closely related to the decomposition of organic matter, a process by which complex organic compounds are broken down into small organic molecules and inorganic constituents. Decomposition is accomplished mainly by soil microorganisms (Scheu et al. 2005). Particulate organic matter is deposited onto soils through litter fall after plant death. The decomposition process can be divided into decomposition of fresh litter, a rapid process, and breakdown of decomposed matter, which occurs over long time periods (Gregorich and Janzen 2000).

Because organic matter decomposition is a complex process which occurs over a number of stages, it is typically modelled through empirical relationships. Olson (1963) suggested the use of an exponential decay model to predict the release of nutrients through organic matter decomposition:

\[
\frac{dP}{dt} = L - k \cdot P \tag{3.3}
\]

where \( P \) is the mass of organic phosphorus to be decayed [M]; \( L \) is the rate of production of organic phosphorus for decay [M/T]; \( k \) is the decay rate constant [T\(^{-1}\)]; and \( t \) is time [T]. More sophisticated models account for different “pools” of phosphorus in soil which are more or less likely to undergo mineralization (Cole et al. 1977; Jones et al. 1984; Andren and Paustian 1987).

Immobilization is the reverse of mineralization, which occurs as soil microbes convert soluble orthophosphate to microbial biomass (Stevenson and Cole 1999). Microbial tissue generally contains a greater concentration of phosphorus than
plants. In a study of 15 soils of different types, a mean phosphorus content of 3.3% in soil microbial biomass was estimated (Brookes et al. 1984). Microbial phosphorus uptake is generally faster than plant uptake (Deubel and Merbach 2005). The rate of phosphorus uptake by microbes is governed by Michaelis-Menten kinetics (Healey 1980; Klausmeier et al. 2007), as is the rate of vegetative phosphorus uptake (see Eq. (3.2)). Typical phosphorus uptake parameters for different classes of microorganisms are reported in Table 3.8.

Table 3.8: Typical microbial phosphorus uptake parameters (Vadstein 2000).

<table>
<thead>
<tr>
<th>Type of microorganism</th>
<th>( U_m )</th>
<th>( K_m )</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(/hr)</td>
<td>(µg/L)</td>
</tr>
<tr>
<td>Heterotrophic bacteria</td>
<td>5.2</td>
<td>3.0</td>
</tr>
<tr>
<td>Cyanobacteria</td>
<td>14</td>
<td>36</td>
</tr>
<tr>
<td>Green algae</td>
<td>1.8</td>
<td>12</td>
</tr>
</tbody>
</table>

1: The value of \( C_{min} \) is zero in all cases.
2: \( U_m \) corresponds to \( I_{max} \) in Eq. (3.2).

The mineralization and immobilization processes occur simultaneously in soil, such that net changes in organic and inorganic phosphorus fractions may not be significant over short time periods (Stevenson and Cole 1999). However, Oberson et al. (2001) observed rapid turnover rates in soil, such that mineralization and immobilization processes can have a significant impact on the availability of phosphorus to plants in soils.

Daily mineralization rates of 0.2 to 3.8 mg P/kg-soil have been measured (Zou et al. 1992; Lopez-Hernandez et al. 1998; Oehl et al. 2001), while immobilization rates ranging from 0.0 to 4.3 mg P/kg-soil have been reported (Zou et al. 1992).
The rates of organic phosphorus mineralization and inorganic phosphorus immobilization are highly dependent on soil temperature and moisture conditions (Van Meeteren et al. 2007). Soil aeration and pH can also influence the rate of these microbial processes (Stevenson and Cole 1999).

Under anaerobic conditions, which can occur in bioretention systems when soils become saturated, a release of phosphorus into soil solution is commonly observed. This process has been explained as a reduction in ions which are associated to phosphate in soils (iron in particular). There is significant evidence that this process is actually mediated by soil microbes (Fleischer 1978).

3.4.5 Filtration and Mobilization

In bioretention systems, both particulate and colloidal phosphorus can be retained in soils through filtration. Influent particulate phosphorus is expected to be almost entirely captured within bioretention soils, but colloidal transport through soils may be significant (Morales et al. 2009). Filtration is believed to be an important phosphorus removal mechanism in bioretention systems, as high rates of suspended solid capture have been reported from bioretention experiments (Hsieh and Davis 2005; Blecken et al. 2007; Davis 2007). However, bioretention soil particles have been observed in the underdrain of a North Carolina bioretention system (Hunt et al. 2008), which suggests that particle mobilization may be significant in some bioretention soils.
As discussed by Li and Davis (2008b), traditional filtration theory is not directly applicable to particle filtration in bioretention systems as significant differences exist between bioretention systems and traditional sand filters. By design, bioretention systems drain rapidly, such that saturated conditions are not maintained in bioretention soils between storm events. Also, bioretention system inflow rates fluctuate significantly during storm events, while the hydraulic conditions in traditional sand filters are relatively constant. Lastly, typical bioretention media is finer than traditional filtration media and has a wider particle size distribution; as well as being biologically active.

Colloidal filtration in unsaturated soils involves a number of processes: physicochemical filtration, attachment to air-water interfaces, straining in water-saturated pores and thin water film straining (DeNovio et al. 2004). Similarly, colloidal mobilization can be caused by dispersion, chemical perturbation, expansion of water films, air-water interface scouring, and shear mobilization (DeNovio et al. 2004).

The kinetics of colloidal filtration and mobilization in unsaturated soils are often represented by a first-order rate model (Bradford et al. 2002), as given by Eq.(3.4):

\[
Q_{sw} = \theta_w k_{att} C - \rho_b k_{det} C \tag{3.4}
\]

where \(Q_{sw}\) is the transfer of colloidal mass from the aqueous to the solid phase [M/L³·T]; \(\theta_w\) is the volumetric water content (unitless); \(k_{att}\) is the colloid attachment coefficient [T⁻¹]; \(C\) is the aqueous phase colloid concentration [M/L³];
\( \rho_b \) is the soil bulk density \([M/L^3]\); \( k_{det} \) is the colloid detachment coefficient \([T^{-1}]\); and \( S \) is the solid phase colloid concentration \([M/M]\). Eq. (3.4) only accounts for the capture of colloids on the soil-water interface. However, attachment to the air-water interface has been shown to play a significant role in the filtration of colloids at low water contents (Wan and Wilson 1994). A more sophisticated version of the model presented in Eq. (3.4), which also accounts for the rate of particle attachment and detachment to the air-water interface, has been proposed by Corapcioglu and Choi (1996). However, because of the complexity of the processes involved in the filtration and mobilization of colloids in unsaturated soils, limited data is available at the moment on the rates of colloid attachment and detachment to the air-water interface.

The rate at which colloids are filtered or mobilized in soil depends mainly on the size of the colloids and the size of soil particles. In addition, charged colloidal particles are filtered at a significantly decreased rate compared to uncharged particles of similar size (Elzo et al. 1998). Based on the results of filtration experiments in saturated porous media, colloidal particle mobilization is likely to occur under decreased solution ionic strength and increased pH (DeNovio et al. 2004).

3.4.6 Sedimentation

Particulate phosphorus can settle at the surface of a bioretention system under ponding conditions. In wetlands, sedimentation is considered as the primary retention mechanism for particulate phosphorus (Braskerud 2002).
hydrologic regime in bioretention systems fundamentally differs from that in wetlands, however. While wetland soils are saturated over continuous time periods, often with significant depths of ponding water, bioretention systems are designed to drain rapidly to avoid issues associated with stagnant water (Prince George's County, 2007). For this reason, ponding depths in bioretention systems are typically only significant over short time periods during storm events.

Depending on the system inlet configuration, resuspension of particulate phosphorus can also occur under high velocity inflows. This process is not expected to be significant, however, as particles will settle once laminar conditions are re-established in the ponding layer. The presence of vegetation has also been shown to reduce the likelihood of particle resuspension in wetlands (Braskerud 2001), and this result can be expected to also apply to bioretention systems.

The sedimentation process is typically modelled based on the settling velocity of phosphorus particles, as predicted by Stokes’ law (Reddy and DeLaune 2008):

$$u = \frac{2}{9} \frac{(gr^2)(\rho_p - \rho_f)}{\mu}$$  \hspace{1cm} (3.5)

where $u$ is the sedimentation velocity [L/T]; $g$ is the gravitational acceleration [L/T²]; $r$ is the “equivalent radius” of the particle containing phosphorus [L]; $\rho_p$ is the density of the phosphorus particle [M/L³]; $\rho_f$ is the density of stormwater [M/L³]; and $\mu$ is the viscosity of stormwater [M/L·T]. Stokes’ law was developed for ideal spherical particles. For this reason, the settling velocity of a particle is
calculated using its “equivalent radius”, which is the radius of a spherical particle with identical density and settling velocity.

Stokes’ law is only valid under laminar flow or quiescent conditions, which implies that no turbulence is generated by the settling of particles (Washington 1992). This assumption is met in the ponding layer of bioretention systems for most phosphorus particles of interest in stormwater. However, under high runoff inflows, transitional or turbulent conditions can be induced in the ponding layer. Stokes’ law does not apply under these conditions, but the modelling errors introduced under semi-turbulent flow conditions should be acceptable for practical applications.

Because fine particles tend to have a greater affinity for phosphorus sorption than large particles (Scalenghe et al. 2007), the mass of phosphorus that settles at the surface of a bioretention system may not be well represented by the average settling velocity of particles. A more accurate representation would consider the phosphorus content of each particle size.

3.5 Phosphorus Removal Modelling

Bioretention modelling is an emerging research field, with an increasing number of models being developed to simulate hydrologic processes and the fate of various pollutants within bioretention systems. The most recently developed hydrologic model for bioretention systems is the R2D model (Aravena and Dussaillant 2009), which implements a two-dimensional version of the Richard’s
equation to model infiltration in rain gardens. A few simpler hydrologic models for bioretention systems are also available in the literature (Dussaillant et al. 2003; Dussaillant et al. 2004; Heasom et al. 2006; Poresky and Palhegyi 2008). Recently, water quality models have been developed to simulate the fate of suspended solids (Li and Davis 2008c), heavy metals (Li and Davis 2008a) and polycyclic aromatic hydrocarbons (PAHs) (He and Davis 2009) in bioretention systems. However, the role of microorganisms and vegetation in bioretention systems, which is expected to significantly influence phosphorus cycling, has not been included in these water quality models.

A number of numerical models have been developed to simulate phosphorus transport within systems that share some similarities with bioretention systems, such as wetlands and other stormwater best management practices. These models are reviewed, as they may provide a framework for the development of a bioretention phosphorus transport model. The characteristics of some phosphorus transport models currently available in the literature are summarized in Table 3.9 and discussed below. Nutrient cycling in wetlands has been studied extensively, and a large number of models of ranging complexity levels are available in the literature. This review considered three wetland models with considerably different structures.
<table>
<thead>
<tr>
<th>System modelled</th>
<th>Model</th>
<th>Model structure</th>
<th>Hydrologic structure</th>
<th>Phosphorus forms</th>
<th>Phosphorus transport processes</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetland</td>
<td>Autobiotic wetland model</td>
<td>Completely-mixed system; water enters through rainfall, leaves through evapotranspiration</td>
<td>Total phosphorus</td>
<td>First-order areal rate removal</td>
<td>Kadlec (1997)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>WETSAND</td>
<td>Flows obey diffusion wave theory; water enters through rainfall, leaves through infiltration, evapotranspiration, and groundwater recharge</td>
<td>Total phosphorus</td>
<td>Advection, dispersion, first-order areal rate removal</td>
<td>Kazezyilmaz-Alhan et al. (2007)</td>
<td></td>
</tr>
<tr>
<td>Grass filter strip and grass swale</td>
<td>GRAPH</td>
<td>Major flow component is in the horizontal direction; water enters through surface runoff and rainfall, leaves through infiltration</td>
<td>Dissolved and sediment-bound</td>
<td>Advection, adsorption of dissolved phosphorus onto sediment, desorption of phosphorus from ground, biological uptake, particle detachment, and deposition</td>
<td>Lee et al. (1989)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TRAVA</td>
<td>Major flow component is in the horizontal direction; water enters through runoff, leaves through infiltration</td>
<td>Particulate phosphorus</td>
<td>Advection, dispersion, particle deposition</td>
<td>Deletic (2001)</td>
<td></td>
</tr>
<tr>
<td>Wet detention pond</td>
<td>Pollutant removal model for wet ponds</td>
<td>Completely-mixed system; water enters through rainfall, watershed inflow and perennial inflow, leaves through evaporation and infiltration</td>
<td>Total phosphorus</td>
<td>First-order rate removal</td>
<td>Wang et al. (2004)</td>
<td></td>
</tr>
</tbody>
</table>
Autobiotic wetlands are self-regulating and biologically-evolving natural systems, not unlike bioretention systems. The autobiotic wetland model developed by Kadlec (1997) predicts phosphorus retention based on a lumped first-order rate term. Because large inconsistencies in phosphorus retention have been observed in bioretention systems, this model cannot accurately capture the phosphorus cycle in bioretention systems. Similarly, the WETSAND model (Kazezyilmaz-Alhan et al. 2007) agglomerates phosphorus retention into a single first-order rate term. WETSAND also incorporates complex hydrologic modelling to predict the flow behaviour in wetlands. However, as previously discussed, the hydrologic regime in wetlands differs significantly from the hydrologic regime in bioretention systems. For this reason, the hydrologic equations included in WETSAND are not directly applicable to bioretention systems.

The simple mechanistic model developed by Hafner and Jewell (2006) is different from all other models presented here. It focuses on the cycling of phosphorus in wetland soils through decomposition and assimilation and relies on the structure of typical soil cycling models in which phosphorus is placed in readily available and refractory pools. However, the model does not account for such processes as sorption, precipitation and filtration, which are believed to be important phosphorus removal processes in bioretention systems. In addition, the influence of the hydrologic regime on phosphorus transport within wetlands was omitted in this model. This has limited applicability to bioretention modelling, as the hydrologic regime within bioretention systems is highly variable and dynamic under storm flows.
Many of the processes believed to be of importance in bioretention systems are included in the GRAPH model (Lee et al. 1989). However, as GRAPH was developed to simulate the performance of grass buffer strips, it focuses on overland flow and sediment transport, which are not significant in bioretention systems. Stormwater is considered to leave the system when it infiltrates, while the main abatement component in bioretention systems is the downward flow through the soil media. Similarly, the TRAVA model (Deletic 2001) was developed mainly to simulate overland flow. TRAVA is also strictly a sediment transport model, which means that it accounts only for particulate phosphorus cycling processes.

The wet pond model developed by Wang et al. (2004) assumes that wet ponds behave like small lakes. Water and pollutants leave the system once they infiltrate the soil below the pond, such that the main flow component in bioretention systems is not represented by this model. In addition, phosphorus retention is predicted through a lumped first-order rate term. For these reasons, the wet pond model cannot be directly applied to bioretention systems. However, the model could be considered as a representation of the ponding layer in bioretention systems.

### 3.6 Conclusion

Phosphorus is an environmental pollutant of concern, as it controls the productivity of most freshwater systems. Eutrophication of these systems can result from increases in phosphorus availability, which in turn leads to significant
environmental and economic consequences. Stormwater discharges have become an increasingly important source of phosphorus in receiving water bodies. For this reason, the use of BMPs has become an important measure in the protection of water bodies from eutrophication. Bioretention systems, one of the most recent BMPs, have gained considerable attention due to their promising potential for stormwater retention and treatment. However, inconsistent levels of phosphorus removal have been reported in the systems. There is a need for better understanding of the phosphorus cycle in bioretention systems, and numerical simulation models are inexpensive tools that can be used to study these complex natural systems.

This chapter identified and reviewed the processes involved in phosphorus cycling in bioretention systems. The aim of the chapter was to define processes and equations that would be necessary in the development of a model to simulate phosphorus transport within bioretention systems. Some classifications of phosphorus forms that can be useful for modelling purposes were discussed. The bioretention phosphorus cycle was reviewed, and mathematical representations were provided for the most significant cycling processes. The structure of previously developed phosphorus transport models was examined, along with a discussion of the modelling components that could be appropriately applied to bioretention systems. By combining the information presented in this chapter with hydrologic and pollutant transport equations, a model could be developed to simulate the transport of phosphorus in bioretention systems. Such a model would assist designers in estimating the phosphorus removal potential of planned
bioretention systems, and could also be used to gain a better understanding of the importance of individual bioretention processes for phosphorus removal.
3.7 References


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Chapter 4 – Formulation and Evaluation of an Event-Based Simulation Model for Phosphorus Removal in Bioretention Systems

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Chapter 4
Formulation and Evaluation of an Event-Based Simulation Model for Phosphorus Removal in Bioretention Systems

4.1 Abstract

Bioretention systems are newly introduced stormwater management practices that have shown great potential at reducing stormwater volumes and improving stormwater quality. However, phosphorus removal in bioretention systems has been inconsistent in previous field studies. Phosphorus is a pollutant of concern in many regions of the world, as it is a limiting nutrient for algal growth in most shallow freshwater environments. High phosphorus loadings to these sensitive ecosystems can lead to eutrophication. To assist system designers in predicting phosphorus removal through bioretention, a one-dimensional finite difference model was developed to simulate both soluble and particulate phosphorus transport in a bioretention system over the duration of a storm event. The Bioretention Phosphorus Removal Model (BPRM) comprises four completely-mixed layers; the ponding layer, the mulch layer, the soil root zone, and the deep soil zone. Processes included in the model are evapotranspiration, overflow, infiltration, exfiltration to native soils, underdrain discharge, soluble phosphorus sorption and vegetative uptake, and particulate phosphorus capture. Model results were evaluated against monitoring data collected by the Toronto and Region Conservation Authority (TRCA) at a bioretention system installed on the King City campus of Seneca College, in the Greater Toronto Area (Ontario,
Model input parameters were mainly estimated from the literature due to limited field data availability. Measured underdrain flow rates and phosphorus outflow concentrations were compared to model predictions. Total underdrain discharge volumes were overestimated for most storm events, but the model was found to provide practical estimates of total phosphorus outflow concentrations and mass. When applied to carefully selected input parameters, BPRM can produce useful estimates of the potential for phosphorus removal in a bioretention system. In particular, the model should identify whether phosphorus leaching from the bioretention system is of concern.

4.2 Introduction

In the three past decades, the impact of discharging storm runoff directly into the natural environment has been recognized as one of the main sources of pollution to receiving water bodies (National Research Council 2009). Stormwater carries significant loads of urban pollutants, which include solids, biodegradable matter, nutrients, pathogens, metals, oil and grease, as well as synthetic organics (Horner 1994). Urban development also brings about important changes in the hydrologic cycle of a region by increasing runoff volumes and peak flows, reducing runoff travel times, and decreasing infiltration and groundwater recharge rates.

Bioretention systems, also referred to as rain gardens or biofiltration practices, are at-source stormwater BMPs that can be used to reduce the impact of urban development on receiving water bodies. The systems were developed in the 1990s by Prince George’s County (PGC), Maryland (Coffman et al. 1993), and have since
been adopted by many municipalities (Morzaria-Luna et al. 2004). Bioretention systems make use of the physical, chemical and biological processes in forested ecosystems to retain and treat urban runoff. Stormwater is directed to the system from its drainage area through an inlet structure. Water then ponds at the surface of the system and infiltrates through its engineered high permeability soil media. Large particulate pollutants are trapped within an optional mulch layer at the surface of the system, while flood and drought-tolerant vegetation helps to retain stormwater and cycle pollutants in the system. An overflow structure is used to bypass storm runoff when the ponding capacity of the system is exceeded. If low permeability native soils surround the bioretention system, rapid drainage of the system is ensured by an underdrain structure installed below the soil media.

As discussed in Chapter 2, bioretention systems have shown great potential for stormwater retention and removal of most urban pollutants through both laboratory testing and field monitoring. However, nutrient removal, and phosphorus removal in particular, has been inconsistent between bioretention systems. Davis (2007) reported mean total phosphorus mass removals of 79% and 77%, respectively, for two bioretention cells constructed on the University of Maryland campus. The median inflow concentration to the cells was 0.61 mg/L and median outflow concentrations of 0.15 mg/L and 0.17 mg/L were measured for each of the two cells. In comparison, Dietz and Clausen (2005) reported total phosphorus mass increases of 110.6% in the outflow of two rain gardens in Haddam, Connecticut. An exponentially decreasing trend in phosphorus concentrations over time was observed at the outlet of the rain gardens during
the 12-month monitoring period, suggesting that phosphorus leaching from the gardens stabilizes over time. In a study of three bioretention systems in North Carolina, one system was found to retain 65% of the total phosphorus inflow mass, while phosphorus leaching from another system increased the total phosphorus loading from the inflow by 240% (Hunt et al. 2006). In this case, the phosphorus content of the bioretention soils was measured for each system before installation. The soil used in the first system was found to have a low phosphorus content, while the phosphorus content of the soil used in the second system was high. This suggests that phosphorus removal in bioretention systems may be highly influenced by the capacity of bioretention soils to adsorb phosphorus.

Phosphorus is a pollutant of concern in many regions of the world, as it is a limiting nutrient for algal growth in most shallow freshwater environments (Schindler 1977; Correll 1999). Excessive algal bloom and eutrophication can arise from high phosphorus loadings to these aquatic environments. As a result, bioretention system designers are interested in better understanding and predicting phosphorus removal through bioretention.

Currently, most stormwater quality management guidelines require a specific fraction of the total volume of runoff from a drainage area to undergo treatment (e.g. Maryland Department of the Environment 2000; Atlanta Regional Commission 2001; Vermont Agency of Natural Resources 2002; New York Department of Environmental Conservation 2008). The level of treatment to be
achieved is not addressed by these regulations, and their strict application may not ensure that pollutant loads to sensitive ecosystems are reduced to an acceptable level. To allow designers to estimate the treatment levels achievable with the installation of a bioretention system, most design guidelines suggest expected pollutant removals based on the limited monitoring data available in the literature. However, it can be misleading to estimate pollutant load and concentration reductions based on reported bioretention performance. The level of pollutant removal achieved in a bioretention system can be influenced by many factors, including: stormwater inflow rates to the system, total stormwater volumes entering the system, pollutant inflow concentrations, as well as bioretention system and component design. There is a need for a tool to accurately predict pollutant removal in bioretention systems.

To date, bioretention modelling has received limited attention in the literature. A small number of models have been developed to simulate hydrologic processes within bioretention systems, most of them relying on the hydrologic functions included within modelling environments such as SWMM and HEC-HMS (Heasom et al. 2006; Poresky and Palhegyi 2008; Lucas 2009). The R2D hydrologic model developed by Aravena and Dussaillant (2009), which is effectively a two-dimensional extension of the RECHARGE and RECARGA hydrologic models developed by Dussaillant et al. (2003; 2004), is the most comprehensive model currently available. Recently, bioretention modelling has been extended to include water quality modelling. Models have been developed to simulate the fate of suspended solids (Li and Davis 2008c), heavy metals (Li
and Davis 2008a) and polycyclic aromatic hydrocarbons (PAHs) (He and Davis 2009) in bioretention systems. To the authors’ knowledge, no bioretention model is currently available to simulate both the hydrologic and water quality performance of bioretention systems.

The main objective of the work presented in this chapter is to develop a numerical tool to assist designers in predicting phosphorus transport in bioretention systems. The focus is on developing a user-friendly numerical model which can be used by bioretention system designers from different professional backgrounds (e.g., site landscapers, planners and engineers). The model developed will also provide insight into the relative importance of bioretention processes to phosphorus removal. The model is to meet the following modelling objectives:

1. To simulate hydrologic and phosphorus transport within a bioretention system;
2. To estimate the rates at which water, soluble phosphorus and particulate phosphorus exit the system;
3. To estimate the concentrations of soluble and particulate phosphorus in the outflow of the bioretention system; and
4. To provide insight into the processes that affect stormwater retention and phosphorus removal in bioretention systems.
This chapter presents the formulation and evaluation of the Bioretention Phosphorus Removal Model (BPRM), an event model that simulates phosphorus transport in bioretention systems. The model structure and development are discussed, as well as the numerical modelling methods selected. Field monitoring data collected by the Toronto and Region Conservation Authority (TRCA) at the Seneca College, King Campus bioretention site is presented and compared to modelling results. In Chapter 5, the sensitivity of the model to input parameters is assessed, the relative importance of model processes is examined and the significance of model results is discussed.

4.3 Model Development

BPRM is a one-dimensional finite difference model that simulates phosphorus transport in bioretention systems over the duration of a storm event. The model comprises four layers, which act as storage reservoirs for water volumes and both soluble and particulate phosphorus masses. Soluble phosphorus includes all forms of dissolved phosphorus, while colloidal and suspended phosphorus forms are considered as particulate phosphorus in BPRM. All model layers are assumed to be completely-mixed reservoirs, such that no gradients in water content or phosphorus concentration exist within the layers. This modelling assumption is reasonable as bioretention systems are designed to drain rapidly, such that two-dimensional horizontal flow within the systems is expected to be minimal. The first layer included in BPRM is the ponding layer, which is used to represent water that accumulates at the surface of the bioretention system during high intensity rainfall periods. The second model layer, the mulch layer, is an optional
layer in BPRM. As noted by Minervini (2007), while some design guidelines currently require bioretention systems to be covered by a layer of mulch, other guidance documents advise designers against the potential problems associated with the use of mulch. The third and fourth model layers make up the bioretention soil media. The soil root zone extends from the mulch layer to the end of the vegetation roots. The remaining layer is the deep soil zone, which drains to the underdrain structure, if included, or recharges the ground below the system if no underdrain structure is installed.

Figure 4.1 indicates the transport processes involved in the hydrologic, soluble phosphorus and particulate phosphorus components of the BPRM model. The direction of the arrows on the figure indicates whether flows enter or leave the system. Double-headed arrows are used to represent soluble phosphorus sorption and particulate phosphorus capture in Figure 4.1, as both processes can transport phosphorus from the aqueous phase into the soil matrix and from the soil matrix into the aqueous phase. This implies that soluble phosphorus sorption terms can become desorption terms, while particulate phosphorus capture terms can become release terms. The model diagrams are presented based on a bioretention system which includes both a mulch layer and an underdrain structure. If the mulch layer is excluded, water infiltrates directly from the ponding layer to the soil root zone, and both soluble phosphorus sorption to mulch and particulate phosphorus capture by mulch are removed from the model. If no underdrain structure is included in the bioretention system to be modelled, underdrain discharge in the deep soil zone is replaced by infiltration to native soils below the
system. In this case, an additional layer is included in the model to accurately simulate infiltration rates into native soils, which generally have lower permeabilities than bioretention soils.

Figure 4.1: Model schematic for BPRM.

The hydrologic portion of BPRM (Figure 4.1 a) is similar to the structure of the RECARGA model, which simulates hydrologic processes within bioretention systems (Dussaillant et al. 2003). However, the RECARGA model was mainly developed to investigate groundwater recharge through bioretention over long-term simulations. In contrast, BPRM was developed to simulate bioretention processes over a single storm event. This modelling time scale was selected to remain consistent with current design guidelines that require bioretention systems to be sized based on a design storm event (PGC 2007). Moreover, event models can be evaluated relatively easily, while evaluating the performance of continuous models requires the collection of field data over long consecutive time periods. Such data is not available for phosphorus removal in bioretention systems at present.
In the RECARGA model, a sand layer was included below the underdrain structure as per the initial bioretention design guidelines (PGC 1993). The structure of BPRM was adapted to better reflect current bioretention design guidelines (PGC 2007). The sand layer below the underdrain was removed, the underdrain discharge rate was defined based on unsaturated flow in the bioretention soils and weir equations were included to model the rate of overflow from the system. The bioretention soils were also divided in two layers for improved accuracy, and exfiltration from the system to surrounding native soils was included in the model.

Bioretention processes relevant to phosphorus cycling were discussed in detail in Chapter 3. Two types of phosphorus cycling processes can be distinguished: rapid (short-term) and slow (long-term) processes. Only short-term bioretention processes which can significantly impact phosphorus cycling over the duration of a storm event were included in BPRM. Results from previous field and laboratory studies suggest that both sorption and vegetation uptake can significantly impact phosphorus removal in bioretention systems (Hunt et al. 2006; Davis et al. 2006; Henderson et al. 2007; Lucas and Greenway 2008). Particulate phosphorus processes can also have a significant impact on total phosphorus removal in bioretention systems, as large fractions of phosphorus in stormwater are often found in particulate form (United States Environmental Protection Agency 1999; Bratieres et al. 2008). Particulate phosphorus removal in BPRM is achieved through capture of the phosphorus-containing colloidal particles on the solid matrix of bioretention media. The model does not account for sedimentation of
particulate phosphorus in the ponding layer, as ponding depths at the surface of bioretention systems are considered to be insignificant throughout most storm events. Movement of phosphorus between the soluble and particulate forms was not included in BPRM for simplicity. Phosphorus precipitation and dissolution rates are generally considered to be relatively slow (Snoeyink and Jenkins 1980).

4.4 Mathematical Framework

The change in water volume stored in the model’s ponding layer is given by Eq. (4.1):

\[
\frac{dW_1}{dt} = q_w + r_w - j_w - o_w - i_{w_1}
\]  

(4.1)

where \( W_1 \) is the volume of water in the ponding layer [L³]; \( q_w \) is the rainfall inflow rate [L³/T]; \( r_w \) is the runoff inflow rate [L³/T]; \( j_w \) is the rate of evapotranspiration [L³/T]; \( o_w \) is the ponding layer overflow rate [L³/T]; \( i_{w_1} \) is the rate of infiltration to the mulch layer [L³/T]; and \( t \) is time [T]. If no mulch layer is installed in the system, \( i_{w_1} \) is replaced in Eq. (4.1) by \( i_{w_2} \), which is the rate of infiltration to the soil root zone [L³/T].

\( q_w \) corresponds to the rate at which precipitation falls directly above the bioretention system. It is calculated by BPRM based on a time series of rainfall intensities for the storm event of interest provided as input data by the user. \( r_w \) is also provided to the model in the form of time series. \( j_w \) is calculated using \( j_w \), a constant user-specified evapotranspiration rate [L/T], and is adjusted based on
the volume of water available for evapotranspiration in the ponding layer at each time step.

The overflow structure in a bioretention system is treated as a weir in BPRM. For rectangular overflow weirs, the discharge rate is given by Eq. (4.2):

$$o_W = \frac{2}{3} C_d \sqrt{\frac{1}{2g}} L_o \left( \frac{W_i}{A_b} - D_p \right)^3$$

(4.2)

where $C_d$ is the weir discharge coefficient $[L^3/L^3]$; $g$ is the gravitational acceleration $[L/T^2]$; $L_o$ is the weir width $[L]$; $A_b$ is the bioretention system area $[L^2]$; and $D_p$ is the ponding capacity of the system $[L]$, that is the height of the overflow weir above the bioretention system bottom. BPRM also has the capability to model triangular overflow weirs for greater flexibility.

$SP_1$, the soluble phosphorus mass contained in the ponding layer $[M]$, varies over time in BPRM as given by Eq. (4.3):

$$\frac{dSP_1}{dt} = r_{SP} - o_{SP} - i_{SP1}$$

(4.3)

where $r_{SP}$ is the soluble phosphorus runoff inflow rate $[M/T]$; $o_{SP}$ is the soluble phosphorus overflow rate $[M/T]$; and $i_{SP1}$ is the infiltration rate of soluble phosphorus to the mulch layer $[M/T]$. $r_{SP}$ is calculated in BPRM based on the rate at which runoff enters the bioretention system and the concentration of soluble phosphorus in runoff, which is provided to the model as a time series. $o_{SP}$ is calculated as the product of the soluble phosphorus concentration in the ponding layer at time $t$ and the rate at which water overflows from the system.
Similarly, $i_{sp_1}$ is given by the product of the soluble phosphorus concentration in the ponding layer and the rate of infiltration to the mulch layer. If the mulch layer is omitted from the simulation, $i_{sp_1}$ becomes $i_{sp_2}$, the rate of soluble phosphorus infiltration to the soil root zone [M/T], in Eq. (4.3).

The change over time in $PP_1$, the mass of particulate phosphorus suspended in water in the ponding layer [M], is given by Eq. (4.4):

$$\frac{dPP_1}{dt} = r_{pp} - o_{pp} - i_{pp_1} \tag{4.4}$$

where $r_{pp}$ is the particulate phosphorus inflow rate in runoff [M/T]; $o_{pp}$ is the particulate phosphorus overflow rate [M/T]; and $i_{pp_1}$ is the infiltration rate of particulate phosphorus to the mulch layer [M/T]. $i_{pp_1}$ is replaced in Eq. (4.4) by $i_{pp_2}$, the rate of particulate phosphorus infiltration to the soil root zone [M/T], if no mulch layer is included in the bioretention system. A time series of particulate phosphorus concentrations in runoff is provided as input to the model and is used along with the rate at which runoff enters the system to calculate $r_{pp}$. Consistently with the assumption that model layers are completely mixed, $o_{pp}$ is calculated as the product of the concentration of particulate phosphorus in the ponding layer and the rate of overflow from the system, and $i_{pp_1}$ is calculated as the product of the particulate phosphorus concentration in the ponding layer and the infiltration rate to the mulch layer.
The volume of water stored in the mulch layer \([L^3]\), \(W_2\), varies in time according to Eq. (4.5):

\[
\frac{dW_2}{dt} = i_{W_1} - i_{W_2}
\]  

(4.5)

In BPRM, the mulch layer is treated as an additional layer of storage for stormwater above the bioretention soils. Water enters the mulch layer as soon as it is available in the ponding layer until the storage capacity of the mulch layer, which is given by Eq. (4.6), is met.

\[
\alpha_2 = M_2 \ D_2 \ A_b
\]  

(4.6)

where \(\alpha_2\) is the storage capacity of the mulch layer \([L^3]\), \(M_2\) is the effective porosity of the mulch layer (which is the difference between the saturated and the residual water contents of the layer) \([L^3/L^3]\); and \(D_2\) is the mulch layer depth [L].

The change in the mass of soluble phosphorus in solution in the mulch layer [M], \(SP_2\), is given by:

\[
\frac{dSP_2}{dt} = i_{SP_1} - i_{SP_2} - b_{SP_2}
\]  

(4.7)

where \(b_{SP_2}\) is the rate of soluble phosphorus sorption to mulch [M/T]. \(i_{SP_2}\) is calculated as the product of the soluble phosphorus concentration in the mulch layer and the rate at which water infiltrates into the soil root zone.
The mass of particulate phosphorus in the model’s mulch layer \([M]\), \(PP_2\), varies over time according to Eq. (4.8):

\[
\frac{dPP_2}{dt} = i_{pp_1} - i_{pp_2} - c_{pp_2}
\]

(4.8)

where \(c_{pp_2}\) is the net particulate phosphorus capture rate on the solid matrix in the mulch layer \([M/T]\). The concentration of particulate phosphorus in the mulch layer and the infiltration rate into the soil root zone are used to calculate \(i_{pp_2}\).

Eq. (4.9) defines the change over time in the volume of water contained in the soil root zone \([L^3]\), \(W_3\):

\[
\frac{dW_3}{dt} = i_{w_2} - i_{w_3} - e_{w_3}
\]

(4.9)

where \(i_{w_3}\) is the rate of percolation to the deep soil zone \([L^3/T]\); and \(e_{w_3}\) is the rate of exfiltration to native soils from the soil root zone \([L^3/T]\).

Dussaillant et al. (2003) compared the predictions obtained with two models developed to simulate hydrologic processes within a bioretention system. The first model, RECHARGE, was developed using Richard’s equation for infiltration, while infiltration in the second model, RECARGA, was simulated based on the less complex Green-Ampt equation (1911). Good agreement was observed between the RECHARGE and RECARGA model predictions. For this reason, infiltration into the bioretention soil media is represented in BPRM by the Green-Ampt infiltration equation.
Modifications to the traditional Green-Ampt equation (Mein and Larson 1973) were incorporated in Eq. (4.10) to better represent infiltration rates into bioretention soils. Generally, the depth of water ponding on the soil surface is assumed to be relatively small compared to the wetting front depth and the capillary suction head across the wetting front. In bioretention systems, however, this depth is not always negligible. To account for this, the depth of water ponding above the soil root zone, given by the depth of water in both the mulch layer and the ponding layer, is included in Eq. (4.10). Also, in the traditional Green-Ampt equation, the wetting front depth in a soil is represented by the cumulative infiltration depth. In bioretention soils, however, water exfiltrates from the system into surrounding native soils, such that the effective wetting front depth in the soil does not correspond to the cumulative infiltration depth. The effective wetting front depth is defined by Eq. (4.11) in BPRM.

\[
i_{W_2} = K_{sat_3} A_b \left[ 1 + \frac{\left( \theta_{sat_3} - \theta_{res_3} \right) \left( \psi_{s} + \frac{W_1 + W_2}{A_b} \right)}{W_e} \right]
\]

(4.10)

where \( K_{sat_3} \) is the saturated hydraulic conductivity of soil in the root zone [L/T]; \( \theta_{res_3} \) is the residual water content in the soil root zone [L³/L³]; \( \theta_{sat_3} \) is the saturated water content in the soil root zone [L³/L³]; \( \psi_{s} \) is the capillary tension in the soil root zone [L]; and \( W_e \) is the effective wetting front depth [L], as given by the following equation:
\[
W_e = \frac{(W_3 + W_4)}{A_b} - (\theta_{\text{res}_3} D_3 + \theta_{\text{res}_4} D_4) \frac{(\theta_{\text{sat}_3} - \theta_{\text{res}_3})}{(\theta_{\text{sat}_4} - \theta_{\text{res}_4})} 
\]  
(4.11)

where \( W_4 \) is the volume of water in the deep soil zone [L³]; \( D_3 \) is the depth of the soil root zone [L]; \( \theta_{\text{res}_4} \) is the residual water content in the deep soil zone [L³/L³]; and \( D_4 \) is the deep soil zone depth [L].

Many processes are involved in soluble phosphorus transport within the soil root zone, as shown by Eq. (4.12), which describes the change over time in the mass of soluble phosphorus in the soil root zone [M], \( SP_3 \):

\[
\frac{dSP_3}{dt} = i_{SP_3} - i_{SP_3} - b_{SP_3} - v_{SP} - e_{SP_3}
\]  
(4.12)

where \( i_{SP_3} \) is the rate of soluble phosphorus percolation to the deep soil zone [M/T]; \( b_{SP_3} \) is the rate of soluble phosphorus sorption to soil in the root zone [M/T]; \( v_{SP} \) is the soluble phosphorus vegetative uptake rate [M/T]; and \( e_{SP_3} \) is the soluble phosphorus exfiltration rate from the soil root zone [M/T]. \( i_{SP_3} \) and \( e_{SP_3} \) are calculated using the concentration of soluble phosphorus in the soil root zone at each time step and the rates of percolation to the deep soil zone and exfiltration from the soil root zone, respectively.

Vegetative uptake is represented in BPRM by a Michaelis-Menten relationship (Barber 1984), as given by Eq. (4.13). The equation is meant to lump both the rates of phosphorus uptake by vegetation and by microorganisms present in bioretention soils (these rates are difficult to quantify and model independently).
A term was added to the equation to account for the fraction of soluble phosphorus that is considered available to plants, which is generally defined as the fraction of soluble phosphorus in the form of orthophosphate (PO_4^{3-}). The equation was also modified to account for the portion of the roots that are in contact with soluble phosphorus at any time, based on the assumptions that both stormwater and vegetation roots are equally distributed within the soil layer, and that no phosphorus concentration gradients exist within the layer.

\[
v_{SP} = A_r \cdot f_{BP} \left( \frac{W_3}{D_3 \cdot \theta_{sat3} \cdot A_b} \right) \left( \frac{l_{max} \left( \frac{SP_3}{W_3} - C_{min} \right)}{v_M + \frac{SP_3}{W_3} - C_{min}} \right)
\]  

(4.13)

where \( A_r \) is the total root surface area \([L^2]\); \( f_{BP} \) is the fraction of soluble phosphorus that is available to plants \([M/M]\); \( l_{max} \) is the maximum phosphorus uptake rate \([M/L^2/T]\); \( C_{min} \) is the minimum soil water phosphorus concentration for vegetative uptake \([M/L^3]\); and \( v_M \) is the Michaelis constant for phosphorus uptake \([M/L^3]\).

The mass of particulate phosphorus in the soil root zone \([M]\), \( PP_3 \), varies over time according to Eq. (4.14):

\[
\frac{dPP_3}{dt} = i_{pp_3} - i_{pp_3} - c_{pp_3} - e_{pp_3}
\]  

(4.14)

where \( i_{pp_3} \) is the rate of particulate phosphorus percolation to the deep soil zone \([M/T]\); \( c_{pp_3} \) is the net particulate phosphorus capture rate on the soil matrix of the root zone \([M/T]\); \( e_{pp_3} \) is the particulate phosphorus exfiltration rate from the soil root zone \([M/T]\). The concentration of particulate phosphorus in the soil root
zone and the rates of percolation to the deep soil zone and exfiltration from the soil root zone are used to determine $i_{pp_3}$ and $e_{pp_3}$, respectively.

The change in water volume over time in the fourth model layer, the deep soil zone, is defined by Eq. (4.15):

$$\frac{dW_4}{dt} = i_{W_3} - u_W - e_{W_4}$$  \hspace{1cm} (4.15)

where $u_W$ is the underdrain discharge rate [L³/T]; and $e_{W_4}$ is the exfiltration rate to surrounding native soils from the deep zone [L³/T]. If no underdrain structure is included in the bioretention system, $u_W$ in Eq. (4.15) is replaced by $z_W$, which is the exfiltration rate to native soils below the system [L³/T]. An additional layer is also included in the model below the bioretention system to collect water that exfiltrates from the deep soil zone. Drainage from the additional native soil layer is limited by $z_W'$, the groundwater recharge rate [L³/T].

Percolation was defined in the RECARGA model based on the kinematic theory of unsaturated flow, which implies fully-gravitational vertical drainage and a pressure gradient of unity (Singh 1997; Dussaillant et al. 2003). Similarly, the kinematic theory of unsaturated flow is used in BPRM to define the rate of percolation to the deep soil zone, the underdrain discharge rate, the rate of exfiltration to native soils below the bioretention system and the groundwater recharge rate, as given by Eq. (4.16) through (4.19).
\[ i_{W_3} = A_b \, K_{unsat_3} \]  
(4.16)

\[ u_W = A_b \, K_{unsat_4} \]  
(4.17)

\[ z_W = A_b \, K_{unsat_4} \]  
(4.18)

\[ z_W' = A_b \, K_{unsat_5} \]  
(4.19)

where \( K_{unsat_i} \) is the unsaturated hydraulic conductivity of layer \( i \) [L/T].

As suggested by Dussaillant et al. (2003) for the RECARGA model, the van Genuchten drainage equation (1980) is used to define the unsaturated hydraulic conductivities used in Eq. (4.16) through (4.19):

\[
K_{unsat_i} = K_{sat_i} \, \theta_i^{1/2} \left[ 1 - \left(1 - \theta_i^{1/m_i} \right)^{m_i/2} \right] 
\]  
(4.20)

where \( K_{sat_i} \) is the saturated hydraulic conductivity of layer \( i \) [L/T]; \( \theta_i \) is the effective saturation [L³/L³], as defined by Eq. (4.21):

\[
\theta_i = \frac{(\theta_i - \theta_{res})}{(\theta_{sat_i} - \theta_{res})} 
\]  
(4.21)

and the \( m_i \) parameter (unitless) is given by:

\[
m_i = 1 - 1/n_i 
\]  
(4.22)

where \( n_i \) is the van Genuchten parameter for layer \( i \) (unitless).
Exfiltration from bioretention soils into surrounding native soils is defined in BPRM based on a modified version of the Green-Ampt equation for sloped soil surfaces (Chen and Young 2006). Infiltration into the vertical soil face that separates the bioretention system from the surrounding soil at layer $i$ is defined by Eq. (4.23):

$$ e_{Wi} = K_{satN} \frac{\psi_N}{y_{Ni}} (P_b \ D_i) $$

(4.23)

where $K_{satN}$ is the saturated hydraulic conductivity of surrounding native soils [L/T]; $\psi_N$ is the capillary tension in surrounding native soils [L]; $P_b$ is the bioretention system perimeter [L]; and $y_{Ni}$ is the wetting front depth inside native soils at layer $i$ [L], as given by Eq. (4.24):

$$ \frac{dy_{Ni}}{dt} = \frac{e_{Wi}}{M_N \ P_b \ D_i} $$

(4.24)

where $M_N$ is the effective porosity of native soils [L³/L³].

The mass of soluble phosphorus in solution in deep soil zone [M], $SP_4$, varies with time according to Eq. (4.25).

$$ \frac{dSP_4}{dt} = i_{SP3} - u_{SP} - b_{SP4} - e_{SP4} $$

(4.25)

where $u_{SP}$ is the soluble phosphorus underdrain discharge rate [M/T]; $b_{SP4}$ is the rate of soluble phosphorus sorption to soil in the deep zone [M/T]; and $e_{SP4}$ is the soluble phosphorus exfiltration rate to surrounding native soils from the deep zone [M/T]. When bioretention systems do not include an underdrain
structure, $u_{SP}$ is replaced by $z_{SP}$, the rate of soluble phosphorus exfiltration to native soils below the system [M/T], in Eq. (4.25). As the model layers are assumed to be completely-mixed reservoirs, $u_{SP}$, $e_{SP4}$ and $z_{SP}$ are calculated based on the concentration of soluble phosphorus in the deep soil zone and the rate of the corresponding hydrologic processes.

The change over time in mass of particulate phosphorus in suspension in the deep soil zone [M], $PP_4$, is given by Eq. (4.26):

$$
\frac{dPP_4}{dt} = i_{pp3} - u_{pp} - c_{pp4} - e_{pp4}
$$

(4.26)

where $u_{pp}$ is the particulate phosphorus underdrain discharge rate [M/T]; $c_{pp4}$ the net particulate phosphorus capture rate on the soil matrix of the deep zone [M/T]; and $e_{pp4}$ is the particulate phosphorus exfiltration rate to surrounding native soils from the deep zone [M/T]. If no underdrain structure is included in the bioretention system to be modelled, $z_{pp}$, the rate of particulate phosphorus exfiltration to native soils below the system [M/T], replaces $u_{pp}$ in Eq. (4.26). $u_{pp}$ and $e_{pp4}$, as well as $z_{pp}$ when no underdrain structure is included, are defined as the product of the concentration of particulate phosphorus in the deep soil zone and the rates of underdrain discharge, exfiltration from the deep soil zone and exfiltration to native soils below the system, respectively.

Soluble phosphorus sorption to mulch and soil is represented in BPRM using a kinetic version of the Langmuir isotherm (McGechean and Lewis 2002), as given by Eq. (4.27). The Langmuir isotherm equation was adopted in BPRM because of
its common use to model sorption in the literature and the availability of input parameters for this equation compared to other sorption models (Ho 2004; Zhang et al. 2005). If Eq. (4.27) yields a negative sorption rate for layer $i$, phosphorus desorption from the layer is occurring. An additional term was added to the equation to account for the level of saturation in layer $i$. This assumes that sorption can only occur if the soil or mulch within a layer is in contact with water. The rate of soluble phosphorus sorption in layer $i$ is thus defined as:

$$b_{SP_i} = \beta \left( \frac{W_i \rho_i}{\theta_{sat_i}} \right) \left( k_{l_i} \frac{SP_i}{W_i} - \frac{Q_i}{Q_{max_i}} \left( 1 + k_{l_i} \frac{SP_i}{W_i} \right) \right)$$  \hspace{1cm} (4.27)$$

where $\beta$ is the kinetic sorption rate constant [T-1]; $\rho_i$ is the soil density of layer $i$ [M/L³]; $k_{l_i}$ is the Langmuir sorption constant for layer $i$ [L³/M]; $Q_{max_i}$ is the maximum sorption capacity of soil in layer $i$ [M/M]; and $Q_i$ is the mass of phosphorus sorbed to soil per mass of soil in layer $i$ [M/M], as given by Eq. (4.28):

$$\frac{dQ_i}{dt} = \frac{b_{SP_i}}{(D_i \cdot A_b \cdot \rho_i)}$$  \hspace{1cm} (4.28)$$

Corapcioglu and Choi (1996) developed a model for colloid transport in unsaturated porous media which used first and second-order rate equations to define colloidal particle capture on the solid matrix and the air-water interface in the soil. The equation developed for particle capture on the solid matrix in a soil is adopted in BPRM to define particulate phosphorus capture. (Particulate phosphorus capture on the soil’s air-water interface is ignored in BPRM for
simplicity.) The net capture of particulate phosphorus on the soil matrix in layer $i$ is thus given by Eq. (4.29):

$$
c_{pp_i} = A_b \, D_i \left( h_{capl_i} \, \frac{PP_i}{W_i} - h_{rel_i} \, \sigma_i \right) \tag{4.29}
$$

where $h_{capl_i}$ is the rate of particulate phosphorus capture on the soil matrix in layer $i$ [T^-1]; $h_{rel_i}$ is the rate of particulate phosphorus release from the soil matrix in layer $i$ [T^-1]; and $\sigma_i$ is the mass of particulate phosphorus captured on the soil matrix of layer $i$ per volume of soil [M/L^3], which varies in time according to Eq. (4.30).

$$
\frac{d\sigma_i}{dt} = \frac{c_{pp_i}}{A_b \, D_i} \tag{4.30}
$$

### 4.5 Numerical Method Selection and Algorithm Design

BPRM uses a fully-explicit fourth order Runge-Kutta numerical integration scheme to solve the set of differential equations described in the mathematical framework section. A diagonally-implicit Runge-Kutta method was also considered for the model, but the improvements in model predictions obtained with this method were not sufficient to justify the increase in computational expense. Table 4.1 shows a comparison of the number of evaluations of the differential equations included in the model for the diagonally-implicit and the fully explicit Runge-Kutta numerical integration schemes for a selected storm event with a modelling length of 4020 min and a time step of 1 min. The diagonally-implicit method required 33 times the number of model evaluations.
required by the fully explicit method. The total underdrain discharge volume predicted by the two methods varied by only 8%, while the total phosphorus underdrain mass prediction varied by 5%.

Table 4.1: Comparison of diagonally-implicit and fully explicit Runge-Kutta numerical methods.

<table>
<thead>
<tr>
<th>Numerical Method</th>
<th>Number of evaluations of differential equations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Diagonally-implicit Runge-Kutta</td>
<td>531 683</td>
</tr>
<tr>
<td>Fully Explicit Runge-Kutta</td>
<td>16 080</td>
</tr>
</tbody>
</table>

Modelling run times are short for single storm events. Table 4.2 shows some typical modelling run times measured on a test computer (1.73 GHz Intel processor, 2.5 GB RAM).

Table 4.2: Modelling run times for typical simulation lengths and modelling time steps.

<table>
<thead>
<tr>
<th>Simulation length (min)</th>
<th>Modelling time step (min)</th>
<th>Number of modelling time steps</th>
<th>Computer run time (min:sec)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2000</td>
<td>0.1</td>
<td>20 000</td>
<td>0:50.50</td>
</tr>
<tr>
<td>2000</td>
<td>1</td>
<td>2000</td>
<td>0:05.38</td>
</tr>
<tr>
<td>2000</td>
<td>10</td>
<td>200</td>
<td>0:00.68</td>
</tr>
<tr>
<td>4000</td>
<td>0.1</td>
<td>40 000</td>
<td>1:39.64</td>
</tr>
<tr>
<td>4000</td>
<td>1</td>
<td>4000</td>
<td>0:10.63</td>
</tr>
<tr>
<td>4000</td>
<td>10</td>
<td>400</td>
<td>0:01.20</td>
</tr>
<tr>
<td>6000</td>
<td>0.1</td>
<td>60 000</td>
<td>2:28.82</td>
</tr>
<tr>
<td>6000</td>
<td>1</td>
<td>6000</td>
<td>0:14.77</td>
</tr>
<tr>
<td>6000</td>
<td>10</td>
<td>600</td>
<td>0:01.86</td>
</tr>
</tbody>
</table>

The model developed was coded in the Visual Basic 2008 programming language to facilitate the creation of a simple and attractive user interface. Since BPRM
was developed as a numerical tool for practitioners, ensuring the model’s ease of use was an important element of software design. Figure 4.2 shows a screen capture of the main BPRM program window. A detailed user manual was created with instructions on model use and input parameter selection and is accessible through the help menu of the program. Extensive checks were performed to ensure that model equations were coded correctly and that mass was conserved within the system. The sum of initial water volumes inside the bioretention system and total system inflow volume was compared to the sum of final water volumes in the system and total system outflow volume for a number of storm events. Similar checks were performed on soluble and particulate phosphorus mass. Maximum differences of 0.2% in water volumes and phosphorus mass were calculated. These small discrepancies were the result of round-off errors.

Figure 4.2: Screen capture of the main BPRM program window.
BPRM requires the user to provide an input file containing time series of rainfall intensities, runoff inflow rates, and both soluble and particulate phosphorus runoff concentrations throughout the storm event to be modelled. For each modelling time step, the output files produced by BPRM contain water volumes and both soluble and particulate phosphorus mass in all model layers, as well as flow rates and phosphorus mass transfer rates for all model processes. Phosphorus concentrations in the bioretention system inflow and outflow are also provided at each time step.

4.6 Model Evaluation

In this section, bioretention field data collected by the Toronto and Region Conservation Authority (TRCA) is presented and compared to modelling results. The hydrologic portion of the model is evaluated in a first step based on underdrain discharge rates from the bioretention system. Then, measured underdrain phosphorus concentrations are compared to the concentrations predicted by BPRM. This is done to assess the performance of the hydrologic equations used in the model separately from the performance of phosphorus transport equations.

4.6.1 Field data

The model results were evaluated against field monitoring data collected by the Toronto and Region Conservation Authority (TRCA) at a bioretention system constructed on the King City campus of Seneca College, in the Greater Toronto Area (GTA), in Ontario, Canada (TRCA, 2008). The data was collected as part of
the Sustainable Technologies Evaluation Program (STEP) in order to assess the performance of bioretention systems and porous pavement under environmental and climatic conditions representative of the GTA.

A campus parking lot was divided into three sections of equal dimensions for this study. Porous pavement was installed on one of the 286 m² parking lot sections, while the two other sections were paved with traditional asphalt pavement. Runoff from one of the asphalt parking lot sections was directed to a bioretention system constructed on site, while the remaining asphalt pavement section was used as a control for the study. Asphalt curbs were used to separate the pavement drainage areas. A partial layout of the asphalt-paved parking lot sections and the location of the instrumentation used is indicated in Figure 4.3.

![Figure 4.3: Parking lot design for the bioretention system field study (adapted from TRCA, 2008). Note: The permeable pavement section and its associated instrumentation are not shown on this figure.](image-url)
Construction of the bioretention system began in August 2004. The asphalt parking lot section adjacent to the bioretention area was graded to drain towards the bioretention system (Figure 4.3). The system area was excavated to an approximate depth of 1 m and filled with a mixture of screened garden soil and sand (at a 3:1 ratio). A texture analysis revealed that the bioretention soils contain 42% sand, 50% silt and 8% clay, which is classified as silt loam by the United States Department of Agriculture (USDA) classification system (2009b). An underdrain structure made up of a weeping tile wrapped in a filter sock was placed below the bioretention soils and covered with granular material. For monitoring purposes, an impermeable plastic membrane was installed around the bioretention soils to prevent exfiltration from the system. The soil surface in the system was lightly compacted and graded to provide depression storage. A layer of cedar mulch was placed at the surface of the system. Drought and flood-tolerant vegetative species, such as Andropogan gerardii, Aster puniceus, Aster laevis, Penstemon digitalis, Liatris spicata, and Cornus sericea, were then planted in the bioretention system. Figure 4.4 shows the bioretention system after a storm event with water ponding at the surface.
Field monitoring began on site in September 2005 and continued through to April 2008. A tipping bucket rain gauge was installed immediately adjacent to the monitoring site in 2006 to collect rainfall intensities at a 5-min interval. Surface runoff flows from the asphalt-paved control area and underdrain flows from the bioretention system were measured continuously during the study period using magnetic induction flow meters. The data collected was recorded at 1-min intervals. To prevent damage to the flow meters, a sediment trap was installed ahead of the measuring devices to settle coarse sediment. Water quality monitoring results may have been affected by this sediment-control measure. Starting in August 2006, water levels at the surface of the bioretention system were measured using pressure transducers installed at two locations within the system (Figure 4.3). These water level fluctuations reveal the occurrence of overflows during high-intensity rainfall events. The bioretention soil temperature was also measured throughout the study at the location of the water level sensors.
at a depth of 50 cm below the surface of the system. Water quality sampling was triggered during large storm events when flow rates reached 0.005 L/s. Sampling intervals were set at 2 min for runoff from the control pavement, and 60 min for underdrain flow from the bioretention system. 48 aliquots were collected per sampling event and flow-proportioned samples were prepared from the aliquots collected for laboratory analysis. The water samples were analysed by the Ontario Ministry of the Environment (MOE) based on standard MOE analytical methods. A wide range of water quality parameters were assessed, including total phosphorus concentrations, soluble orthophosphate concentrations, inorganic and organic dissolved carbon concentrations, and total suspended and dissolved solid concentrations. A chemical analysis was performed on sediment cores extracted from the bioretention system in the fall of 2005 after the construction of the system and in the summer of 2007. The average organic carbon content of the soil was measured as 66.7 mg/g of dry soil and an average total phosphorus content of 1.54 mg/g of dry soil was obtained.

Groundwater levels remained more than 3 m below the base of the bioretention system throughout the study period, such that native soils surrounding the system remained unsaturated. Runoff in excess of the bioretention system capacity overflowed from the system and was directed to surrounding grass swales (Figure 4.3). The bioretention system was observed to function properly during winter periods and soil temperatures near the surface of the system never reached the freezing point. Additional details on the construction of the
bioretention system, on the data collection methodologies and on the findings of the study are available in the STEP project report produced by the TRCA (2008).

### 4.6.2 Input parameter selection

A total of 10 storm events with different characteristics were selected to evaluate the performance of BPRM. The selection includes storm events from each season of the year. Events were also selected to represent a wide range of total rainfall depths, average and maximum rainfall intensities, storm durations, and antecedent conditions. As well, a number of the storm events selected produced overflows from the bioretention system. Table 4.3 summarizes the main characteristics of the storm events selected for modelling. For the purpose of this analysis, an interevent time definition (IETD) of 12 hours was used, such that underdrain flows from different storm events could be distinguished. The interevent time reported in Table 4.3 is based on the number of days with less than 2 mm of precipitation recorded.
Table 4.3: Characteristics of the storm events selected for modelling.

<table>
<thead>
<tr>
<th>Storm event date</th>
<th>Total rainfall duration (hr)</th>
<th>Total rainfall depth (mm)</th>
<th>Average rainfall intensity (mm/hr)</th>
<th>Maximum rainfall intensity (mm/hr)</th>
<th>Interval event time (days)</th>
<th>Peak runoff rate from control pavement (L/s)</th>
<th>Total runoff volume from control pavement (m³)</th>
<th>Overflow from bioretention system</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jul. 10, 2006</td>
<td>12.4</td>
<td>73.0</td>
<td>5.9</td>
<td>40.8</td>
<td>8.11</td>
<td>4.307</td>
<td>14.97</td>
<td>Yes</td>
</tr>
<tr>
<td>Jul. 12, 2006</td>
<td>9.4</td>
<td>15.8</td>
<td>1.7</td>
<td>36.0</td>
<td>1.65</td>
<td>0.496</td>
<td>1.29</td>
<td>Unknown</td>
</tr>
<tr>
<td>Sept. 22, 2006</td>
<td>46.2</td>
<td>16.8</td>
<td>0.4</td>
<td>19.2</td>
<td>3.75</td>
<td>2.029</td>
<td>1.09</td>
<td>No</td>
</tr>
<tr>
<td>Sept. 27, 2006</td>
<td>3.9</td>
<td>13.2</td>
<td>3.4</td>
<td>26.4</td>
<td>3.15</td>
<td>3.518</td>
<td>2.03</td>
<td>No</td>
</tr>
<tr>
<td>Nov. 7, 2006</td>
<td>24.5</td>
<td>11.6</td>
<td>0.5</td>
<td>4.8</td>
<td>9.09</td>
<td>0.138</td>
<td>0.41</td>
<td>No</td>
</tr>
<tr>
<td>Nov. 30, 2006</td>
<td>36.8</td>
<td>64.4</td>
<td>1.8</td>
<td>14.4</td>
<td>12.83</td>
<td>0.958</td>
<td>10.20</td>
<td>Yes</td>
</tr>
<tr>
<td>Jan. 4, 2007</td>
<td>42.7</td>
<td>17.6</td>
<td>0.4</td>
<td>7.2</td>
<td>3.79</td>
<td>0.246</td>
<td>3.13</td>
<td>No</td>
</tr>
<tr>
<td>Apr. 23, 2007</td>
<td>0.8</td>
<td>9.8</td>
<td>13.1</td>
<td>43.2</td>
<td>11*</td>
<td>3.017</td>
<td>2.26</td>
<td>Yes</td>
</tr>
<tr>
<td>Jun. 3, 2007</td>
<td>14.3</td>
<td>24.6</td>
<td>1.7</td>
<td>60.0</td>
<td>3.07</td>
<td>3.892</td>
<td>6.13</td>
<td>Yes</td>
</tr>
<tr>
<td>Sept. 25, 2007</td>
<td>16.1</td>
<td>18.6</td>
<td>1.2</td>
<td>74.4</td>
<td>10.67</td>
<td>4.162</td>
<td>4.38</td>
<td>No</td>
</tr>
</tbody>
</table>

* Estimated using precipitation data from the Toronto Buttonville Airport weather station (Environment Canada 2008).
The modelling time step was set to 1 min for all storm events to match the frequency of flow rate measurements. Flow rates from the control pavement were used to estimate runoff inflow rates to the bioretention system. Rainfall intensities, which were measured on site at a 5-min interval, were kept constant over a period of 5 modelling time steps. Water quality monitoring results did not directly provide separate concentrations for soluble and particulate phosphorus. However, the proportions of organic and inorganic carbon in dissolved form were found to remain fairly consistent throughout the monitoring period. Therefore, the ratio of inorganic to total dissolved carbon was assumed to also hold for phosphorus.

The influent and effluent concentrations of soluble phosphorus for each storm event were calculated using the orthophosphate concentrations measured for the event (taken as the total inorganic soluble phosphorus fraction) and the ratios observed between inorganic and total dissolved carbon. All total phosphorus concentrations measured were above the soluble phosphorus concentrations calculated with this method, except for 2 events out of 38 monitored for the control pavement and 1 event out of 46 monitored for the bioretention system. For the soluble phosphorus concentrations calculated, average soluble phosphorus to total phosphorus ratios of 23% and 25% were obtained, respectively, for runoff from the control pavement and underdrain flow from the bioretention system. Similar ratios for storm runoff were reported by Bratieres et al. (2008) and Smullen et al. (1999). Also, while soluble orthophosphate concentrations could not be correlated to dissolved solid concentrations, a correlation was observed between total phosphorus concentrations and suspended solid concentrations in both the control runoff (R²...
and the bioretention system underdrain ($R^2 = 0.78$). This correlation suggests that a large fraction of total phosphorus in the control runoff and bioretention underdrain is in particulate form. Concentrations measured in the control pavement runoff were used to estimate bioretention inflow concentrations for modelling. Table 4.4 contains the event mean concentrations of total phosphorus and soluble orthophosphate measured in the control pavement runoff, as well as the soluble and particulate phosphorus inflow concentrations estimated for modelling. Phosphorus concentrations were not available for the July 10th, 2006 storm event.

Table 4.4: Inflow event mean concentrations for the modelling storm events.

<table>
<thead>
<tr>
<th>Storm event date</th>
<th>Measured concentration from control pavement</th>
<th>Estimated bioretention inflow concentration</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Total phosphorus (mg/L)</td>
<td>Soluble orthophosphate (mg/L)</td>
</tr>
<tr>
<td>Jul. 12, 2006</td>
<td>0.028</td>
<td>0.003</td>
</tr>
<tr>
<td>Sept. 23, 2006</td>
<td>0.066</td>
<td>0.015</td>
</tr>
<tr>
<td>Sept. 27, 2006</td>
<td>0.105</td>
<td>0.006</td>
</tr>
<tr>
<td>Nov. 7, 2006</td>
<td>0.055</td>
<td>0.003</td>
</tr>
<tr>
<td>Nov. 30, 2006</td>
<td>0.042</td>
<td>0.003</td>
</tr>
<tr>
<td>Jan. 4, 2007</td>
<td>0.061</td>
<td>0.022</td>
</tr>
<tr>
<td>Apr. 23, 2007</td>
<td>0.457</td>
<td>0.092</td>
</tr>
<tr>
<td>Jun. 3, 2007</td>
<td>0.098</td>
<td>0.003</td>
</tr>
<tr>
<td>Sept. 25, 2007</td>
<td>0.226</td>
<td>0.006</td>
</tr>
</tbody>
</table>

Table 4.5 shows the values selected to define the bioretention system parameters in BPRM. Survey data was collected and used to determine the area and perimeter of the system and estimate its total storage and ponding capacity. The mulch layer depth was estimated based on visual observations. Root depth requirements reported in the USDA PLANTS database (2009a) for the vegetative
species planted in the system were used to estimate the soil root zone depth. *Andropogon gerardii* and *Cornus sericea* were both found to have the largest root depth requirement at 0.51 m (20 in). The deep soil zone depth was calculated based on a total bioretention soil depth of 1 m.

All modelling input parameters were assigned a level of uncertainty based on the expected range of each input parameter, the source of the data used, and the level of measurement accuracy. The input parameter uncertainties are discussed further in Chapter 5, where they are included in an in-depth sensitivity analysis of the model.

### Table 4.5: BPRM system parameters for Seneca College King City campus bioretention facility.

<table>
<thead>
<tr>
<th>BPRM input</th>
<th>Symbol</th>
<th>Value</th>
<th>Units</th>
<th>Uncertainty</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bioretention system area</td>
<td>$A_b$</td>
<td>20.2</td>
<td>m²</td>
<td>Low</td>
</tr>
<tr>
<td>Bioretention system perimeter</td>
<td>$P_b$</td>
<td>22.8</td>
<td>m</td>
<td>Low</td>
</tr>
<tr>
<td>System ponding capacity</td>
<td>$D_p$</td>
<td>106</td>
<td>mm</td>
<td>Low</td>
</tr>
<tr>
<td>Mulch layer depth</td>
<td>$D_z$</td>
<td>50</td>
<td>mm</td>
<td>Medium</td>
</tr>
<tr>
<td>Soil root zone depth</td>
<td>$D_z$</td>
<td>0.51</td>
<td>m</td>
<td>Medium</td>
</tr>
<tr>
<td>Deep soil zone depth</td>
<td>$D_4$</td>
<td>0.49</td>
<td>m</td>
<td>Medium</td>
</tr>
</tbody>
</table>

Table 4.6 indicates the hydrologic input parameters selected for modelling. An average discharge coefficient for the overflow weir was calculated using an empirical equation adapted from the work of Kindsvater and Carter (Crowe et al. 2005), based on a system ponding capacity of 156 mm and a head of water above the overflow weir varying from 0 mm to 50 mm. The overflow weir width was estimated based on site observations. The effective porosity of the mulch layer
was defined based on the porosity of compacted mulch reported by Huang et al. (2006). The hydraulic properties of the bioretention soils were obtained with the Rosetta model (Schaap et al. 2001) based on the percentages of sand, silt and clay particles in the soil. The bioretention soil hydraulic conductivity was estimated based on the underdrain flows measured on site and agreed well with the Rosetta prediction (6% difference). The residual water content reported by Aravena and Dussaillant (2009) for an organic soil layer (50% sand and 50% compost) placed in a rain garden was adopted. The soil root zone capillary tension parameter was determined using the chart produced by Rawls et al. (1990), which gives estimates of the wetting front suction in a soil based on the percentages of sand, silt and clay particles in the soil. The hydraulic properties of native soils were set to zero as exfiltration from the bioretention system is prevented by the impermeable membrane installed around the bioretention soils.

Table 4.7 indicates the soluble phosphorus transport parameters selected for modelling. The density of mulch selected was reported by Huang et al. (2006) for compacted cedar mulch, while the density of bioretention soils in the root zone and the deep zone was estimated using the relationship developed by Rawls (1983) that relates the bulk density of a soil to its texture and organic content. Data from batch phosphorus sorption tests reported by Hsieh et al. (2007) was used to determine the Langmuir sorption parameters for mulch. Langmuir sorption parameters for bioretention soils were defined using parameters reported by Brock et al. (2007) for a loamy soil with a similar composition to the bioretention soils on the Seneca Campus. The kinetic sorption rate constant was
defined by assuming an average of 24 hrs required for phosphorus to reach sorption equilibrium, which corresponds to the testing period used by Brock et al. (2007) and Hsieh et al. (2007) in their sorption experiments. Phosphorus uptake rate parameters have been studied mostly for different crop plants and for large trees. Vegetative uptake parameters reported by Barber (1984) for fescue were selected as input parameters for BPRM. Similar parameters were reported for reed canary grass (Barber 1984). For each storm event, the fraction of bioavailable soluble phosphorus was set to the ratio between the orthophosphate concentration measured on site and the soluble phosphorus concentration estimated for the event. The total plant root surface area was estimated based on the vegetation cover observed on site (Figure 4.4) and the average root area per soil volume reported by Dittmer (1938) for oat (Avena sativa), rye (Secale cereale), and Kentucky bluegrass (Poa pratensis).

Particulate phosphorus capture rates in the bioretention soils were calculated based on clean bed filtration theory (Yao et al. 1971) and values reported by Bradford et al. (2006) for colloid straining experiments with Ottawa sands. An average colloidal particle size of 1 µm representative of clay particles was assumed. The soil water content was taken at 50% saturation, and the pore water velocity was predicted based on the hydraulic conductivity of the soil at 50% saturation. Particulate phosphorus release rates were approximated based on values reported by Bradford et al. (2002; 2006) for 1 µm colloid particles and different soil particle sizes. BPRM has the capability to model particulate phosphorus capture in the mulch layer. However, based on the results presented
by Bradford et al. (2002; 2006), the mulch layer included in the Seneca College bioretention system was assumed not to capture or release any colloidal matter. The particulate phosphorus transport parameters selected for modelling are found in Table 4.8.

The evapotranspiration rate for each storm event was calculated using the Penman-Monteith equation, as described in the Food and Agriculture Organisation (FAO) guidelines (Allen et al. 1998). Meteorological data from the Toronto Buttonville Airport weather station was used for these calculations (Environment Canada 2008). The Buttonville Airport station is the closest weather station to the bioretention field site that collected hourly meteorological data over the study period. Table 4.9 contains the evapotranspiration rates calculated for all storm events selected. The evapotranspiration rates calculated are considered to have a low level of uncertainty.
Table 4.6: BPRM hydrologic parameters for Seneca College King City campus bioretention facility.

<table>
<thead>
<tr>
<th>BPRM input</th>
<th>Symbol</th>
<th>Value</th>
<th>Units</th>
<th>Uncertainty</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Weir discharge coefficient</td>
<td>$C_d$</td>
<td>0.61</td>
<td>---</td>
<td>Low</td>
<td>Based on Crowe et al. (2005)</td>
</tr>
<tr>
<td>Rectangular weir width</td>
<td>$L_a$</td>
<td>200</td>
<td>mm</td>
<td>High</td>
<td>Visual observation</td>
</tr>
<tr>
<td>Effective porosity of the mulch layer</td>
<td>$M_2$</td>
<td>0.76</td>
<td>---</td>
<td>Medium</td>
<td>Huang et al. (2006)</td>
</tr>
<tr>
<td>Saturated hydraulic conductivity of the soil root zone</td>
<td>$K_{sat}$</td>
<td>17.1</td>
<td>mm/hr</td>
<td>Medium</td>
<td>Measured on site</td>
</tr>
<tr>
<td>Capillary tension parameter of the soil root zone</td>
<td>$\psi_s$</td>
<td>180</td>
<td>mm</td>
<td>Low</td>
<td>Rawls et al. (1990)</td>
</tr>
<tr>
<td>Soil root zone van Genuchten parameter</td>
<td>$n_3$</td>
<td>1.578</td>
<td>---</td>
<td>Medium</td>
<td>From Rosetta (Schaap et al. 2001)</td>
</tr>
<tr>
<td>Residual water content of the soil root zone</td>
<td>$\theta_{res}$</td>
<td>0.246</td>
<td>---</td>
<td>Medium</td>
<td>Aravena and Dussaillant (2009)</td>
</tr>
<tr>
<td>Saturated water content of the soil root zone</td>
<td>$\theta_{sat}$</td>
<td>0.407</td>
<td>---</td>
<td>Medium</td>
<td>From Rosetta (Schaap et al. 2001)</td>
</tr>
<tr>
<td>Saturated hydraulic conductivity of the deep soil zone</td>
<td>$K_{sat}$</td>
<td>17.1</td>
<td>mm/hr</td>
<td>Medium</td>
<td>Measured on site</td>
</tr>
<tr>
<td>Deep soil zone van Genuchten parameter</td>
<td>$n_4$</td>
<td>1.578</td>
<td>---</td>
<td>Medium</td>
<td>From Rosetta (Schaap et al. 2001)</td>
</tr>
<tr>
<td>Residual water content of the deep soil zone</td>
<td>$\theta_{res}$</td>
<td>0.246</td>
<td>---</td>
<td>Medium</td>
<td>After Aravena and Dussaillant (2009)</td>
</tr>
<tr>
<td>Saturated water content of the deep soil zone</td>
<td>$\theta_{sat}$</td>
<td>0.407</td>
<td>---</td>
<td>Medium</td>
<td>From Rosetta (Schaap et al. 2001)</td>
</tr>
<tr>
<td>Saturated hydraulic conductivity of native soils</td>
<td>$K_{satN}$</td>
<td>0</td>
<td>mm/hr</td>
<td>N/A*</td>
<td></td>
</tr>
<tr>
<td>Capillary tension parameter of native soils</td>
<td>$\psi_N$</td>
<td>0</td>
<td>mm</td>
<td>N/A*</td>
<td></td>
</tr>
<tr>
<td>Effective porosity of native soils</td>
<td>$M_N$</td>
<td>0</td>
<td>---</td>
<td>N/A*</td>
<td></td>
</tr>
</tbody>
</table>

*: Parameters with a value of zero were not assigned a level of uncertainty, as the coefficient of variation of a distribution with an average of zero is undefined.
Table 4.7: BPRM soluble phosphorus transport parameters for Seneca College bioretention facility.

<table>
<thead>
<tr>
<th>BPRM input</th>
<th>Symbol</th>
<th>Value</th>
<th>Units</th>
<th>Uncertainty</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mulch density</td>
<td>( \rho_2 )</td>
<td>0.15</td>
<td>g/cm(^3)</td>
<td>High</td>
<td>Huang et al. (2006)</td>
</tr>
<tr>
<td>Density of soil in the root zone</td>
<td>( \rho_3 )</td>
<td>0.83</td>
<td>g/cm(^3)</td>
<td>Medium</td>
<td>Rawls (1983)</td>
</tr>
<tr>
<td>Density of soil in the deep zone</td>
<td>( \rho_4 )</td>
<td>0.83</td>
<td>g/cm(^3)</td>
<td>Medium</td>
<td>Rawls (1983)</td>
</tr>
<tr>
<td>Kinetic sorption rate constant</td>
<td>( \beta )</td>
<td>0.042</td>
<td>/hr</td>
<td>Medium</td>
<td>Based on Brock et al. (2007)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>and Hsieh et al. (2007)</td>
</tr>
<tr>
<td>Langmuir sorption constant for mulch</td>
<td>( k_{L2} )</td>
<td>0.2</td>
<td>L/mg</td>
<td>Medium</td>
<td>Hsieh et al. (2007)</td>
</tr>
<tr>
<td>Langmuir sorption constant for soil in the root zone</td>
<td>( k_{L3} )</td>
<td>0.03</td>
<td>L/mg</td>
<td>High</td>
<td>Brock et al. (2007)</td>
</tr>
<tr>
<td>Langmuir sorption constant for soil in the deep zone</td>
<td>( k_{L4} )</td>
<td>0.03</td>
<td>L/mg</td>
<td>High</td>
<td>Brock et al. (2007)</td>
</tr>
<tr>
<td>Maximum sorption capacity of mulch</td>
<td>( Q_{max} )</td>
<td>5</td>
<td>mg/kg</td>
<td>Medium</td>
<td>Hsieh et al. (2007)</td>
</tr>
<tr>
<td>Maximum sorption capacity of soil in the root zone</td>
<td>( Q_{max} )</td>
<td>3168</td>
<td>mg/kg</td>
<td>High</td>
<td>Brock et al. (2007)</td>
</tr>
<tr>
<td>Maximum sorption capacity of soil in the deep zone</td>
<td>( Q_{max} )</td>
<td>3168</td>
<td>mg/kg</td>
<td>High</td>
<td>Brock et al. (2007)</td>
</tr>
<tr>
<td>Total plant root surface area</td>
<td>( A_r )</td>
<td>150</td>
<td>m(^2)</td>
<td>High</td>
<td>Based on Dittmer (1938)</td>
</tr>
<tr>
<td>Bioavailable fraction of soluble phosphorus</td>
<td>( f_{BP} )</td>
<td>0.33 – 0.83</td>
<td>---</td>
<td>Low</td>
<td></td>
</tr>
<tr>
<td>Maximum vegetative uptake rate</td>
<td>( I_{max} )</td>
<td>0.001</td>
<td>µmol/cm(^2)/s</td>
<td>Medium</td>
<td>Barber (1984)</td>
</tr>
<tr>
<td>Minimum aqueous concentration for uptake</td>
<td>( C_{min} )</td>
<td>1</td>
<td>µmol/L</td>
<td>Medium</td>
<td>Barber (1984)</td>
</tr>
<tr>
<td>Michaelis constant for phosphorus uptake</td>
<td>( v_M )</td>
<td>5</td>
<td>µmol/L</td>
<td>Medium</td>
<td>Barber (1984)</td>
</tr>
</tbody>
</table>

*: Parameters with a value of zero were not assigned a level of uncertainty, as the coefficient of variation of a distribution with an average of zero is undefined.

Table 4.8: BPRM particulate phosphorus transport parameters for Seneca College bioretention facility.

<table>
<thead>
<tr>
<th>BPRM Input</th>
<th>Symbol</th>
<th>Value</th>
<th>Units</th>
<th>Uncertainty</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Particle capture rate in mulch</td>
<td>( h_{capt_2} )</td>
<td>0</td>
<td>/hr</td>
<td>N/A*</td>
<td>After Bradford et al. (2002; 2006)</td>
</tr>
<tr>
<td>Particle release rate in mulch</td>
<td>( h_{rel_2} )</td>
<td>0</td>
<td>/hr</td>
<td>N/A*</td>
<td>After Bradford et al. (2002; 2006)</td>
</tr>
<tr>
<td>Particle capture rate in soil of the root zone</td>
<td>( h_{capt_3} )</td>
<td>0.013</td>
<td>/hr</td>
<td>High</td>
<td>After Yao et al. (1971) and Bradford et al. (2006)</td>
</tr>
<tr>
<td>Particle release rate in soil of the root zone</td>
<td>( h_{rel_3} )</td>
<td>0.083</td>
<td>/hr</td>
<td>High</td>
<td>After Bradford et al. (2002; 2006)</td>
</tr>
<tr>
<td>Particle capture rate in soil of the deep zone</td>
<td>( h_{capt_4} )</td>
<td>0.013</td>
<td>/hr</td>
<td>High</td>
<td>After Yao et al. (1971) and Bradford et al. (2006)</td>
</tr>
<tr>
<td>Particle release rate in soil of the deep zone</td>
<td>( h_{rel_4} )</td>
<td>0.083</td>
<td>/hr</td>
<td>High</td>
<td>After Bradford et al. (2002; 2006)</td>
</tr>
</tbody>
</table>

*: Parameters with a value of zero were not assigned a level of uncertainty, as the coefficient of variation of a distribution with an average of zero is undefined.
Three categories of antecedent moisture conditions ‘dry’, ‘average’, and ‘wet’ were defined for the bioretention system with associated water contents. The ponding layer and mulch layer were assumed to contain no water in all cases. Dry bioretention soils (both in the root zone and the deep zone) were set to 20% saturation, while average and wet moisture contents were set to 50% and 70% saturation, respectively. For each modelling event, the category of antecedent moisture condition was selected based on the interevent time (see Table 4.3, the values are repeated in Table 4.9). The antecedent moisture category for each storm event is specified in Table 4.9. The antecedent moisture conditions in bioretention soils have a medium level of uncertainty.

Table 4.9: BPRM evapotranspiration rates and antecedent moisture conditions for Seneca College King City campus bioretention facility.

<table>
<thead>
<tr>
<th>Storm event date</th>
<th>Inter-event time (days)</th>
<th>Evapotranspiration rate (mm/d)</th>
<th>Antecedent moisture condition category</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jul. 10, 2006</td>
<td>8.11</td>
<td>1.50</td>
<td>Dry</td>
</tr>
<tr>
<td>Jul. 12, 2006</td>
<td>1.65</td>
<td>1.98</td>
<td>Wet</td>
</tr>
<tr>
<td>Sept. 22, 2006</td>
<td>3.75</td>
<td>1.43</td>
<td>Average</td>
</tr>
<tr>
<td>Sept. 27, 2006</td>
<td>3.15</td>
<td>1.18</td>
<td>Average</td>
</tr>
<tr>
<td>Nov. 7, 2006</td>
<td>9.09</td>
<td>0.26</td>
<td>Dry</td>
</tr>
<tr>
<td>Nov. 30, 2006</td>
<td>12.83</td>
<td>0.80</td>
<td>Dry</td>
</tr>
<tr>
<td>Jan. 4, 2007</td>
<td>3.79</td>
<td>0.43</td>
<td>Average</td>
</tr>
<tr>
<td>Apr. 23, 2007</td>
<td>11*</td>
<td>3.14</td>
<td>Dry</td>
</tr>
<tr>
<td>Jun. 3, 2007</td>
<td>3.07</td>
<td>1.58</td>
<td>Average</td>
</tr>
<tr>
<td>Sept. 25, 2007</td>
<td>10.67</td>
<td>1.70</td>
<td>Dry</td>
</tr>
</tbody>
</table>

The initial bioretention soil contents of sorbed and captured phosphorus were based on the average phosphorus content measured in the Seneca College bioretention soils. The proportion of phosphorus in particulate and soluble forms estimated for the control runoff was assumed to hold for phosphorus captured
and sorbed in the bioretention system, such that 23% of the phosphorus content in soils was assumed to be sorbed, with the rest in the form of captured particles. The initial sorbed phosphorus content in mulch was set to half its sorption capacity, which is a reasonable assumption since the material has a low sorption capacity. High uncertainty was assigned to the initial captured and sorbed phosphorus contents because of the assumptions involved in estimating these values. The initial particulate phosphorus content of mulch was set to zero, since the material is assumed not to retain any colloidal particles. Initial soil water concentrations of soluble and particulate phosphorus were set to meet sorption and capture equilibrium with the soluble and particulate phosphorus contents of the soils. This assumption is reasonable since water inside bioretention soils should have sufficient time to reach a sorption and capture equilibrium between storm events. Soil water concentrations were assigned a high level of uncertainty.

4.6.3 Hydrologic Performance

The structure of BPRM relies heavily on the assumptions that model layers are completely mixed and that soluble and particulate phosphorus travel at the same velocity as water inside the bioretention system. As a result, the hydrologic component of BPRM significantly influences phosphorus transport modelling. In this section, underdrain flows predicted by the model are compared to measured underdrain flows to evaluate the performance of the hydrologic portion of BPRM. Underdrain flow rates and volumes are of concern to designers when estimating the potential phosphorus removal of a bioretention system, as they contribute the largest portion of bioretention outflows for most storms. Overflows can become
significant during large storm events and should be considered along with underdrain flows in those cases. Because bioretention overflows were not collected at the Seneca College site, only underdrain discharge flows are considered here.

Table 4.10 contains both the field-measured and predicted peak underdrain discharge rate and total underdrain discharge volume for all events modelled. The percentage of difference between model predictions and measured values is also found in Table 4.10. Figure 4.5 to Figure 4.8 show measured and predicted underdrain discharge rates for selected storm events. Figure A.1 to Figure A.6 in Appendix A show measured and predicted underdrain discharge rates for the storm events that were not included in this chapter.
Table 4.10: Hydrologic BPRM model predictions with flow data from Seneca College King City campus bioretention facility.

<table>
<thead>
<tr>
<th>Storm event date</th>
<th>Simulation length (min)</th>
<th>Peak underdrain discharge rate</th>
<th>Total underdrain discharge volume</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Measured (L/s)</td>
<td>Predicted (L/s)</td>
</tr>
<tr>
<td>Jul. 10, 2006</td>
<td>3120</td>
<td>0.0687</td>
<td>0.0674</td>
</tr>
<tr>
<td>Jul. 12, 2006</td>
<td>4020</td>
<td>0.0456</td>
<td>0.0674</td>
</tr>
<tr>
<td>Sept. 22, 2006</td>
<td>3900</td>
<td>0.0241</td>
<td>0.0085</td>
</tr>
<tr>
<td>Sept. 27, 2006</td>
<td>3120</td>
<td>0.0351</td>
<td>0.0674</td>
</tr>
<tr>
<td>Nov. 7, 2006</td>
<td>2340</td>
<td>0.0101</td>
<td>0.0000</td>
</tr>
<tr>
<td>Nov. 30, 2006</td>
<td>4500</td>
<td>0.0922</td>
<td>0.0674</td>
</tr>
<tr>
<td>Jan. 4, 2007</td>
<td>3180</td>
<td>0.0087</td>
<td>0.0674</td>
</tr>
<tr>
<td>Apr. 23, 2007</td>
<td>2700</td>
<td>0.0160</td>
<td>0.0175</td>
</tr>
<tr>
<td>Jun. 3, 2007</td>
<td>2000</td>
<td>0.0137</td>
<td>0.0280</td>
</tr>
<tr>
<td>Sept. 25, 2007</td>
<td>3300</td>
<td>0.0660</td>
<td>0.0674</td>
</tr>
</tbody>
</table>
For most storm events, good agreement was found between predicted and measured peak underdrain discharge rates, but total outflow volumes tend to be overestimated. Based on site observations, it is possible that runoff from the drainage parking lot section was not entirely collected by the bioretention system. Actual runoff volumes entering the bioretention system would then be smaller than the volumes draining from the control parking lot section, which were used as inflow volumes in this analysis. Actual field evapotranspiration rates may also be greater than those predicted by BPRM, since evapotranspiration only occurs in the model when water is available in the ponding layer. This may explain part of the discrepancy between measured and predicted total outflow volumes. Discrepancies in overflow rates may also influence the accuracy of the total outflow volumes predicted for large storm events. No formal overflow structure was installed in the field bioretention system, such that accurately predicting overflow rates is difficult.

The largest differences in peak underdrain discharge rate and total underdrain discharge volume were obtained for the Jan. 4, 2006 storm event. BPRM greatly overestimated underdrain flows for this storm event (Figure 4.7). This seems to have been caused by a decrease in the infiltration capacity of bioretention soils under cold weather, which has been observed by others (Constantz and Murphy 1991; Lin et al. 2003). A cover of snow or ice may also have formed at the surface
of the bioretention system, considerably reducing the infiltration capacity of the soils. The average air temperature during this storm event was 6.4°C.

Bioretention system underdrain flows are delayed for most events modelled, which suggests that the water storage capacity of bioretention soils may be overestimated by the input parameters selected. The delay observed could also be a result of the model structure, which assumes that water infiltrates uniformly across the mulch layer and that infiltrated water is uniformly distributed across the bioretention system area. These assumptions may not accurately represent the flow behaviour in bioretention systems, in particular when no ponding exists at the surface of the system to distribute influent runoff. In these cases, influent water tends to infiltrate at the entrance of the bioretention system without occupying the entire bioretention area.

The model tends to underestimate peak underdrain flows for large storm events (e.g. Jul 10, 2006; Nov. 30, 2006; Sept. 25, 2007). This is due in part to the structure of BPRM, which limits the rate of underdrain discharge to the saturated hydraulic conductivity of the deep soil zone, based on the assumption that underdrain discharge is strictly driven by gravity (note the flat peak outflow rates predicted for the Nov. 30, 2006 storm event in Figure 4.6). Actual soil behaviour tends to deviate from this assumption under high runoff inflow rates, as flows become dominated by capillary forces inside the soil (Singh 1997).
No significant outflow was predicted by BPRM for the Nov. 7, 2006 storm event. Because of this, high relative differences in peak underdrain discharge rate and total underdrain discharge volume were calculated for the event (see Table 4.10). However, the absolute difference between modelled and predicted values is relatively small. Modelling results are more sensitive to the choice of input parameters and initial system conditions for small storm events than for large events. If design storm events are applied to bioretention systems in BPRM, the issues mentioned for small storm event modelling should not be of concern.

Figure 4.5: Measured and predicted underdrain discharge rate for Sept. 22, 2006 storm event.
Figure 4.6: Measured and predicted underdrain discharge rate for Nov. 30, 2006 storm event.

Figure 4.7: Measured and predicted underdrain discharge rate for Jan. 4, 2007 storm event.
4.6.4 Phosphorus transport

Measured and predicted phosphorus concentrations and total outflow mass are compared in Table 4.11. The Jul. 10, 2006 storm event does not appear in Table 4.11 as water quality monitoring results were unavailable for this storm event.
Table 4.11: BPRM phosphorus transport model evaluation with flow data from Seneca College King City campus bioretention facility.

<table>
<thead>
<tr>
<th>Storm event date</th>
<th>Underdrain event mean concentration</th>
<th>Total underdrain mass</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Measured (mg/L)</td>
<td>Predicted (mg/L)</td>
</tr>
<tr>
<td>Sept. 27, 2006</td>
<td>2.130</td>
<td>2.887</td>
</tr>
<tr>
<td>Nov. 7, 2006</td>
<td>0.877</td>
<td>4.133</td>
</tr>
<tr>
<td>Nov. 30, 2006</td>
<td>4.720</td>
<td>1.389</td>
</tr>
<tr>
<td>Jan. 4, 2007</td>
<td>1.440</td>
<td>2.652</td>
</tr>
<tr>
<td>Apr. 23, 2007</td>
<td>4.620</td>
<td>2.848</td>
</tr>
<tr>
<td>Sept. 25, 2007</td>
<td>3.360</td>
<td>2.308</td>
</tr>
</tbody>
</table>

\(a\): estimated based on the soluble orthophosphate concentrations measured and the organic fraction in dissolved carbon.

\(b\): estimated based on the concentrations and total outflow volumes measured.
For most of the storm events modelled, fair estimates of total phosphorus concentrations were obtained. The large difference in total phosphorus concentration observed for the Nov. 7, 2006 storm event is due to the small outflow volume predicted for this event.

BPRM predicted that soluble phosphorus would constitute the main form of phosphorus in the outflow of the bioretention system modelled, while field data suggests that a large fraction of this phosphorus is in particulate form. Soil microbial processes which are not currently included in BPRM may be responsible for the conversion of soluble phosphorus to particulate phosphorus in the Seneca College bioretention system. This discrepancy may also be a result of the high level of uncertainty associated with the input parameters involved in modelling phosphorus transport in bioretention systems. In particular, the assumption used to determine the proportions of sorbed and captured initial phosphorus contents in bioretention soils may not appropriately represent the soil conditions. The soluble and particulate phosphorus transport processes and their associated input parameters are examined further in Chapter 5.

Particulate phosphorus removal seems to be overestimated by the model. This could be due in part to the lack of knowledge associated with the particulate phosphorus capture and release rates in bioretention soils. High algae concentrations were also observed in the water collected from the bioretention system outflow. The development and transport of algal material in bioretention systems is not considered in BPRM, as it has not been reported in previous
bioretention studies. Additional research is required to better understand this process and how it affects phosphorus removal in bioretention systems.

A correlation can be observed between the differences in total underdrain volume reported in Table 4.10 and the differences in total phosphorus mass from Table 4.11. Better prediction accuracy of total phosphorus outflow mass was obtained for storm events that were accurately modelled by the hydrologic portion of BPRM, while poor predictions of phosphorus mass were obtained for storm events with a poor hydrologic fit. This reinforces the connection between the hydrologic and phosphorus transport components of the model. Accurately quantifying hydrologic processes within a bioretention system is critical to modelling phosphorus transport within the system.

Significant leaching of phosphorus was observed in the field from the Seneca College bioretention system, as can be seen from the inflow and outflow concentrations reported in Table 4.4 and Table 4.11. Ratios between predicted and measured reductions in total phosphorus mass and concentration are presented in Table 4.12. Ratios approaching a value of 1.0 indicate good prediction accuracy and the negative value for the Nov. 7, 2006 storm event indicates that phosphorus removal was predicted for this event rather than phosphorus leaching. BPRM correctly predicted increases in phosphorus concentrations and mass in the underdrain of the bioretention system for all storm events modelled, except one. Therefore, when used with carefully selected
input parameters, the model can effectively identify the potential for phosphorus leaching in a bioretention system.

Table 4.12: Measured and predicted total phosphorus concentration and mass reductions for storm events modelled with BPRM.

<table>
<thead>
<tr>
<th>Storm event date</th>
<th>Ratio of predicted phosphorus reduction to measured phosphorus reduction</th>
<th>Reduction in total phosphorus event mean concentration</th>
<th>Reduction in total phosphorus mass</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jul. 12, 2006</td>
<td>0.98</td>
<td>1.28</td>
<td></td>
</tr>
<tr>
<td>Sept. 23, 2006</td>
<td>0.95</td>
<td>0.61</td>
<td></td>
</tr>
<tr>
<td>Sept. 27, 2006</td>
<td>1.37</td>
<td>2.52</td>
<td></td>
</tr>
<tr>
<td>Nov. 7, 2006</td>
<td>4.96</td>
<td>-0.07</td>
<td></td>
</tr>
<tr>
<td>Nov. 30, 2006</td>
<td>0.29</td>
<td>0.35</td>
<td></td>
</tr>
<tr>
<td>Jan. 4, 2007</td>
<td>1.88</td>
<td>11.77</td>
<td></td>
</tr>
<tr>
<td>Apr. 23, 2007</td>
<td>0.57</td>
<td>3.21</td>
<td></td>
</tr>
<tr>
<td>Jun. 3, 2007</td>
<td>0.45</td>
<td>0.71</td>
<td></td>
</tr>
<tr>
<td>Sept. 25, 2007</td>
<td>0.66</td>
<td>0.93</td>
<td></td>
</tr>
</tbody>
</table>

4.7 Discussion

BPRM was developed as a simple tool for designers to estimate phosphorus removal in bioretention systems. The model provides information on the transport of water as well as both soluble and particulate phosphorus forms within bioretention systems. The BPRM model was developed to simulate discrete storm events and, for this reason, only phosphorus transport processes that have a significant influence on short-term phosphorus cycling in bioretention systems were considered. Processes involved in long-term phosphorus cycling should be included in the BPRM model if it is to be used to predict phosphorus loading reductions over long time periods.
Most input parameters required for the modelling work presented in this chapter were selected from the literature, as field measurements of these parameters were unavailable. This required extensive research, especially in the case of phosphorus sorption and capture parameters, which vary greatly between soil types. Colloidal particle filtration in unsaturated soils is an emerging research area, and understanding of this process is still limited. In particular, limited data is available for the rates of particle capture and release in soils, which are generally determined experimentally. Additional research is required to better understand and quantify particle straining in unsaturated bioretention media.

If at all possible, the physical measurement of modelling input parameters is preferable to the estimation of these parameters from published works. However, for most practicing bioretention system designers, measuring some or all of the input parameters required by the BPRM model is impractical. In these cases, bioretention system designers can expect similar model performance to that presented in this chapter. While BPRM only roughly estimates outflow phosphorus mass and concentrations from bioretention systems, the modelling results obtained are useful for design purposes. They provide a better assessment of the potential for phosphorus removal in a planned bioretention facility than field monitoring data currently available in the literature (which has shown high levels of variability).

The modelling input parameters presented in this chapter did not account for changes in bioretention system properties or process rates over time, as these
changes are not well understood. The infiltration capacity of bioretention soils may vary over time as the soils settle under their own weight. Also, the hydraulic conductivity of bioretention media can be expected to decrease over time as particulate matter is captured within the soil and reduces its porosity (Li and Davis 2008b). The development of preferential flow paths in bioretention media which greatly reduces the capacity of the bioretention system to retain water has also been observed in a previous study (Carpenter and Hallam 2009). Bioretention soil phosphorus contents can be expected to vary over time as phosphorus is retained in the system or released from the system. Also, the total surface area of vegetation roots can be expected to increase greatly over time, especially during the vegetation establishment phase after construction of the system. Long-term monitoring of bioretention systems could provide some valuable insight into the evolution of flow and phosphorus transport behaviours in bioretention systems over time.

As shown by the poor agreement between BPRM model predictions and field measurements for the January 4th, 2006 storm event (Figure 4.7), at the present time BPRM cannot accurately predict hydrologic processes within bioretention systems during the winter season. In order for the BPRM model to prove more useful for bioretention system design in cold climates, a temperature dependency should be introduced in the model process rates and input parameters. Snow accumulation, snowmelt and cycles of frost and thaw in bioretention soils should also be considered in the hydrologic portion of the model.
The BPRM model presented in this chapter could be further refined to improve the accuracy of model predictions. For example, the hydrologic portion of the model could be replaced by the one-dimensional (Dussaillant et al. 2004) or two-dimensional (Aravena and Dussaillant 2009) Richard’s equation models developed for bioretention systems. Also, additional bioretention processes involved in the cycling of phosphorus, such as sedimentation of particulate phosphorus in the ponding layer, phosphorus dissolution and precipitation, and particulate phosphorus capture on the air-water interface in soil, as well as long-term processes, such as microbial decomposition and long-term phosphorus sorption, could be included in the model developed. However, the BPRM model currently provides a useful estimate of the potential for phosphorus removal in bioretention systems. If several additional bioretention processes are included in the BPRM model, the input parameter requirements of the model could become prohibitive to practitioners. The structure of the model presented in this chapter could also be adapted to simulate the fate of other pollutants of concern in bioretention systems, such as nitrogen, heavy metals, oil and grease, biodegradable organic matter (measured as BOD) and pathogens.

4.8 Conclusions

An event-based finite difference model (BPRM) was created to simulate phosphorus transport in a bioretention system. The BPRM model relies on four completely-mixed layers to simulate stormwater, soluble and particulate phosphorus transport in bioretention systems. A user-friendly interface was
designed for the model to encourage its adoption by bioretention systems designers.

The BPRM model was applied to field monitoring data collected by the TRCA at the Seneca College, King City campus (Toronto, Canada) bioretention facility. Underdrain discharge rates and outflow phosphorus concentrations predicted by the model were compared to measured flow rates and phosphorus concentrations. The model was found to overestimate total underdrain discharge volumes, but measured and predicted underdrain discharge curves corresponded fairly well for most storm events. The accuracy of phosphorus removal predictions was found to be closely related to the accuracy of hydrologic modelling, which reinforces the importance of hydrologic processes in water quality modelling. Better agreement between predicted and measured total phosphorus concentrations and mass in the underdrain of the bioretention system was obtained for storm events which were more accurately modelled by the hydrologic portion of BPRM. The BPRM model correctly predicted phosphorus leaching from the Seneca College bioretention system for all storm events modelled, with the exception of one event.

BPRM could be used by bioretention system designers to gain insight into the potential for phosphorus removal in a planned or constructed bioretention system. In particular, the BPRM model can identify the potential for phosphorus leaching from a bioretention system when used with appropriate input
parameters. The sensitivity of the model to input parameter selection and the significance of model results are discussed at length in Chapter 5.
4.9 References


Prince George's County (PGC). (2007). "Bioretention Manual." Prince George's County, Maryland, Department of Environmental Resources, Environmental Services Division, Landover, MD.


Chapter 5 – Sensitivity and Significance of the Bioretention Phosphorus Removal Model Predictions

Manuscript submitted to the Elsevier Journal of Hydrology.

Chapter 5
Sensitivity and Significance of the Bioretention Phosphorus Removal Model Predictions

5.1 Abstract

Bioretention systems are best management practices which have shown great potential for stormwater quality and quantity control. However, phosphorus removal in bioretention systems has been inconsistent, with phosphorus leaching sometimes reported. Phosphorus is a limiting nutrient for algal growth in most shallow freshwater systems, which makes it a pollutant of concern in many parts of the world. Aquatic environments which receive high phosphorus loadings can undergo eutrophication, which has important ecological and economic consequences. In this chapter, an event-based one-dimensional finite difference model developed to simulate phosphorus transport in bioretention systems was presented. The Bioretention Phosphorus Removal Model (BPRM) was found to produce estimates of total phosphorus removal useful for design purposes. In Chapter 5, the sensitivity of the model to input parameter selection was assessed. Model predictions were found to be particularly sensitive to the drainage parameters and initial water content of bioretention soils, highlighting the strong relationship between hydrologic processes and water quality control in bioretention systems. The modelling results suggest that high rates of soluble phosphorus desorption from bioretention soils caused significant phosphorus leaching from the Seneca College bioretention facility, while particulate
phosphorus processes had little influence on phosphorus removal in the system. Future phosphorus transport modelling efforts should focus on improving the accuracy of hydrologic modelling within bioretention systems before refining the phosphorus cycling equations included in the model.

5.2 Introduction

Bioretention systems were developed by Prince George’s County in the 1990s as structural stormwater best management practices (BMPs) capable of providing at-source treatment. The soil and plant-based systems make use of terrestrial ecosystem functions to retain stormwater inflow volumes and treat pollutants. Field monitoring has shown great promise for bioretention systems (see Chapter 2), but phosphorus removal has been inconsistent with phosphorus leaching observed in some systems (Dietz and Clausen 2005; Hunt et al. 2006; Li and Davis 2009).

Phosphorus is an important water pollutant around the world, as it limits the productivity of most shallow freshwater environments (Schindler 1977; Correll 1999), which can undergo eutrophication under high phosphorus loadings. Green algal mats formed by cyanobacteria appear at the surface of eutrophic water bodies. As algal growth and decay consumes oxygen inside the water column, the frequency of anoxic events increases, with consequent fish kills and eventual losses in biodiversity. Toxins which pose a health risk to animals and humans are also released by some species of cyanobacteria (Vasconcelos 2006). Economic
losses associated to eutrophication include loss of recreational space and increases in drinking water treatment costs (Wilson and Carpenter 1999).

The processes involved in the transport and cycling of phosphorus in bioretention systems are not clearly understood. The presence of vegetation in bioretention systems has been shown to increase the potential of the systems for phosphorus removal (Henderson et al. 2007; Bratieres et al. 2008; Read et al. 2008). The results of previous field monitoring also suggest that phosphorus cycling in bioretention systems is greatly influenced by the phosphorus sorption capacity of bioretention soils (Hunt et al. 2006). Processes involved in the transport of particulate matter can be expected to play an important role for total phosphorus removal in bioretention systems, as particulate phosphorus often constitutes a large fraction of the influent total phosphorus to bioretention systems (Pitt et al. 2004; Bratieres et al. 2008). Finally, the results of bioretention box tests by Davis et al. (2006) highlighted the strong relationship between the hydrologic performance of a bioretention system and its potential for stormwater quality improvement. Less phosphorus was removed in bioretention boxes under increased influent runoff durations or intensities, as flowrate through the bioretention soils was increased.

In Chapter 4, the Bioretention Phosphorus Removal Model (BPRM) developed to simulate phosphorus transport in bioretention systems was presented. BPRM is a one-dimensional finite difference model which simulates both hydrologic processes and phosphorus transport within a bioretention system over the
duration of a storm event. The model was developed as a design tool for practitioners to estimate phosphorus removal in bioretention systems. BPRM relies on four completely-mixed model layers to represent different zones in a bioretention system: the ponding layer, the mulch layer, the soil root zone, and the deep soil zone. Model processes include evapotranspiration, overflow, infiltration, percolation, exfiltration from the system, underdrain discharge, soluble phosphorus sorption and vegetative uptake, and particulate phosphorus capture.

The BPRM model performance was evaluated using field data collected by the Toronto and Region Conservation Authority (TRCA) at the bioretention facility installed on the Seneca College King City Campus in Toronto (Ontario, Canada). For most storm events considered, fair agreement was observed between measured and predicted underdrain discharges. However, BPRM was found to overestimate total underdrain discharge volumes for most storm events considered. The results also showed that BPRM generated predictions of total phosphorus concentrations and mass in the bioretention underdrain that were useful for design purposes. The model correctly predicted phosphorus leaching from the Seneca College bioretention system for all storm events considered, except one. However, the results suggested that BPRM overestimated the fraction of soluble phosphorus in the underdrain of the bioretention system. A relationship was noted between the hydrologic performance of BPRM and the accuracy of its total phosphorus mass predictions. Better agreement between predicted and measured total phosphorus mass was obtained for storm events.
which were modelled accurately by the hydrologic component of BPRM, while poor predictions were obtained for the winter storm event considered. The results suggested that this was related to a reduction in the infiltration capacity of the bioretention soils under cold temperatures and the presence of snow or ice.

In this chapter, a sensitivity analysis of the BPRM model is presented, as well as a discussion of the significance of model results. The importance of different bioretention processes to phosphorus removal is examined. Finally, potential improvements to the BPRM model developed and future bioretention modelling work are discussed.

5.3 Sensitivity Analysis

The sensitivity of BPRM predictions to input parameter selection was assessed using field data collected by the TRCA at the Seneca College King City Campus bioretention facility in Toronto (Ontario, Canada). The field data used was presented in Chapter 4 and additional information on the field monitoring study is available in the project report produced by the TRCA (2008). A two-part sensitivity analysis was performed on BPRM to separate the parameters that influence water flow from the parameters that strictly influence phosphorus transport in bioretention systems. In the first part, the hydrologic component of the model only was examined. A deterministic sensitivity analysis was performed on the hydrologic input parameters and model sensitivity was determined based on predicted underdrain discharge rates. In the second part, the sensitivity of the total phosphorus mass predictions to input parameter selection was assessed. A
deterministic sensitivity analysis was first performed to identify critical input parameters, followed by a Monte Carlo Simulation (MCS) which considered the sensitivity of BPRM to combined input parameter uncertainties. For the purpose of the Monte Carlo sensitivity analysis, all modelling input parameters were assumed to be normally-distributed and independent.

5.3.1 Input Parameters

The selection of input parameters for BPRM was described in detail in Chapter 4. Each input parameter was assigned a degree of uncertainty (low, medium, or high) and an associated coefficient of variation (10%, 25%, or 50%) to reflect this uncertainty. Input parameter uncertainty levels and associated coefficients of variation are found in Table 5.2 through Table 5.12.

The coefficient of variation of each input parameter was used as a measure of its uncertainty in the deterministic sensitivity analyses presented below. Each input parameter underwent perturbations of ±5%, ±20%, and ±\( c_v \), which is the coefficient of variation of each input parameter. In the case of the ±\( c_v \) perturbations, input parameters with higher uncertainty values undergo larger perturbations than parameters with lower uncertainty values. While the first four perturbations (±5% and ±20%) can be used to identify the most influential input parameters, the last perturbations (±\( c_v \)) point to input parameters that have significant effects on BPRM predictions as a result of their level of uncertainty.
The coefficients of variation assigned to input parameters were also used to define normal probability density functions (PDF) for the parameters in the MCS-based sensitivity analysis. All modelling input parameters were considered to be independent. The average of the normal PDF defined for each input parameter was set to the value assigned to each input parameter in Chapter 4, referred to in this chapter as the expected parameter value. The standard deviation of each parameter distribution was defined based on its assigned coefficient of uncertainty, as given by Eq. (5.1):

\[ c_v = \frac{\sigma}{\mu} \]  

(5.1)

where \( c_v \) is the coefficient of variation for an input parameter; \( \mu \) is the expected value of the parameter; \( \sigma \) is the standard deviation for the parameter. Input parameters with a value of zero have an undefined coefficient of variation and have, therefore, been excluded from the sensitivity analysis presented in this chapter.

The Jul. 12, 2006 storm event was selected to assess the sensitivity of the model developed. Good agreement between measured and predicted underdrain discharge volumes and total phosphorus mass was obtained for this event. Further, although no water level data was available for this storm event from the Seneca College bioretention system, no overflow is believed to have occurred during the storm based on its characteristics. BPRM predicted no overflow during this storm event. This will likely minimise the impact of overflow events on the underdrain discharge volumes predicted by BPRM. The main
characteristics of the Jul. 12, 2006 storm event were presented in Table 4.3 in Chapter 4. Table 5.1 below summarizes model inputs and outputs for the storm event, where the bioretention system inflow parameters are based on the control runoff measurements at the Seneca College site, while the bioretention system outflow parameters were predicted by BPRM, as described in Chapter 4. Figure 5.1 shows the rainfall intensity and runoff inflow rate for the Jul. 12, 2006 storm event, as well as the predicted and measured underdrain discharge rates in the Seneca College bioretention facility.
Table 5.1: Summary of BPRM modelling inputs and outputs for the Jul. 12, 2006 storm event.

<table>
<thead>
<tr>
<th>Peak runoff inflow rate (L/s)</th>
<th>Total runoff inflow volume (m³)</th>
<th>Peak underdrain discharge rate (L/s)</th>
<th>Total underdrain discharge volume (m³)</th>
<th>Event mean concentration of total phosphorus in the inflow (mg/L)</th>
<th>Total phosphorus inflow mass (g)</th>
<th>Event mean concentration of total phosphorus in the underdrain (mg/L)</th>
<th>Total phosphorus underdrain mass (g)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.496</td>
<td>1.291</td>
<td>0.0674</td>
<td>1.786</td>
<td>0.028</td>
<td>0.036</td>
<td>3.159</td>
<td>5.642</td>
</tr>
</tbody>
</table>

Figure 5.1: Bioretention system inflows and outflows for the Jul. 12, 2006 storm event.

a) Measured rainfall and runoff inflows to the bioretention system

b) Measured and predicted underdrain flows from the bioretention system
5.3.2 Hydrologic Sensitivity Analysis

A deterministic sensitivity analysis was performed on the hydrologic input parameters of BPRM. The expected value of each hydrologic input parameter (see Chapter 4) was varied by ±5%, ±20%, and ±$c_v$ (Table 5.2 through Table 5.4). The ±5% and ±20% parameter perturbations were used to identify critical modelling parameters, while the ±$c_v$ parameter perturbations indicated which parameters introduced large uncertainties in the model predictions as a result of their assigned uncertainty level.

The sensitivity of BPRM hydrologic predictions to input parameters perturbations was assessed with Eq. (5.2):

$$\Delta U = \frac{U_{pert} - U_{exp}}{U_{exp}} \times 100\%$$

(5.2)

where $\Delta U$ is the difference in total underdrain discharge volume caused by the perturbation of a modelling input parameter (%); $U_{pert}$ is the total underdrain discharge volume predicted with the perturbed parameter value (L); and $U_{exp}$ is the total underdrain discharge volume predicted with the expected parameter values (L). The sensitivity to hydrologic input parameter perturbations of the total underdrain discharge volume predictions given by BPRM over the modeling period of 4020 min is reported in Table 5.2 through Table 5.4. As no overflow was predicted by BPRM for the Jul. 12, 2006 storm event, the overflow weir
width and drag coefficient are not included in the sensitivity analysis presented in this section.

The rate of evapotranspiration in Table 5.2 had little impact on hydrologic modelling predictions, since evapotranspiration is a slow process relative to the event modelling length. The depth of the soil root zone and deep soil zone in Table 5.2 appeared to have a moderate influence on model predictions. However, it should be noted that these two parameters are not truly independent, since the total soil depth in a bioretention system is generally well known. If both parameters undergo perturbations of similar magnitude, but opposite direction, model predictions should remain nearly unchanged. Model predictions were also moderately sensitive to the bioretention system area in Table 5.2, since it is used to calculate the rate of many processes in BPRM. However, the area of a bioretention system can generally be measured (or specified, if the system is being designed) with good accuracy, such that its uncertainty remains relatively low.

In Table 5.2, the system ponding capacity had no influence on the total volume of underdrain discharge predicted by BPRM, as no overflow was predicted for this storm event under any of the perturbed ponding capacities. Perturbations of the mulch layer depth (Table 5.2) and mulch layer effective porosity (Table 5.3) also did not affect the total underdrain discharge volume predicted, as the layer acts
as additional depression storage. The capillary tension parameter of the soil root zone in Table 5.3 had no effect on model predictions, which suggests that flow through bioretention soils is not restricted by the rate of infiltration in this simulation.

The hydrologic component of BPRM was found to be highly sensitive to the drainage parameters of bioretention soils in both the root zone and the deep zone. The model was particularly sensitive to the saturated water content of bioretention soils (Table 5.3), but important changes in total underdrain discharge volumes were also associated to perturbations of the van Genuchten parameter and the residual water content of bioretention soils (Table 5.3). Model predictions were also very sensitive to the initial water content of bioretention soils in Table 5.4. This suggests that flow through the bioretention system is limited by the rate of percolation from the soil root zone to the deep soil zone and by the underdrain discharge rate. Both of these processes are defined based on the van Genuchten drainage equation, which is particularly sensitive to changes in effective soil water content and van Genuchten parameter because of its exponential form.
Table 5.2: BPRM hydrologic sensitivity to perturbations in bioretention system parameters.

<table>
<thead>
<tr>
<th>Input Parameter</th>
<th>Symbol</th>
<th>Uncertainty</th>
<th>Difference in total underdrain discharge volume (%)</th>
<th>Input parameter perturbation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Evapotranspiration rate</td>
<td>$J_{ev}$</td>
<td>Low ($c_v = 10%$)</td>
<td>-0.1 0.1 -0.2 0.2 -0.1 0.1</td>
<td>+5% -5% +20% -20% +c_v -c_v</td>
</tr>
<tr>
<td>System area</td>
<td>$A_b$</td>
<td>Low ($c_v = 10%$)</td>
<td>-1.7 1.7 -6.9 6.0 -4.0 3.2</td>
<td></td>
</tr>
<tr>
<td>System ponding capacity</td>
<td>$D_p$</td>
<td>Low ($c_v = 10%$)</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td></td>
</tr>
<tr>
<td>Mulch layer depth</td>
<td>$D_2$</td>
<td>Medium ($c_v = 25%$)</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td></td>
</tr>
<tr>
<td>Soil root zone depth</td>
<td>$D_3$</td>
<td>Medium ($c_v = 25%$)</td>
<td>-1.8 1.6 -7.6 6.9 -9.7 8.6</td>
<td></td>
</tr>
<tr>
<td>Deep soil zone depth</td>
<td>$D_4$</td>
<td>Medium ($c_v = 25%$)</td>
<td>-1.6 1.5 -6.2 6.0 -8.8 7.7</td>
<td></td>
</tr>
</tbody>
</table>
Table 5.3: BPRM hydrologic sensitivity to perturbations in soil hydraulic parameters.

<table>
<thead>
<tr>
<th>Input Parameter</th>
<th>Symbol</th>
<th>Uncertainty</th>
<th>Difference in total underdrain discharge volume (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Input parameter perturbation</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>+5%</td>
</tr>
<tr>
<td>Effective porosity of mulch</td>
<td>M_2</td>
<td>Medium (c_r = 25%)</td>
<td>0.0</td>
</tr>
<tr>
<td>Soil root zone saturated hydraulic conductivity</td>
<td>K_{sat_4}</td>
<td>Medium (c_r = 25%)</td>
<td>0.6</td>
</tr>
<tr>
<td>Soil root zone capillary tension parameter</td>
<td>\psi_3</td>
<td>Low (c_r = 10%)</td>
<td>0.0</td>
</tr>
<tr>
<td>Soil root zone van Genuchten parameter</td>
<td>n_3</td>
<td>Medium (c_r = 25%)</td>
<td>2.4</td>
</tr>
<tr>
<td>Soil root zone residual water content</td>
<td>\theta_{res_4}</td>
<td>Medium (c_r = 25%)</td>
<td>1.5</td>
</tr>
<tr>
<td>Soil root zone saturated water content</td>
<td>\theta_{sat_4}</td>
<td>Medium (c_r = 25%)</td>
<td>−9.4</td>
</tr>
<tr>
<td>Deep soil zone saturated hydraulic conductivity</td>
<td>K_{sat_4}</td>
<td>Medium (c_r = 25%)</td>
<td>0.5</td>
</tr>
<tr>
<td>Deep soil zone van Genuchten parameter</td>
<td>n_4</td>
<td>Medium (c_r = 25%)</td>
<td>2.5</td>
</tr>
<tr>
<td>Deep soil zone residual water content</td>
<td>\theta_{res_4}</td>
<td>Medium (c_r = 25%)</td>
<td>1.3</td>
</tr>
<tr>
<td>Deep soil zone saturated water content</td>
<td>\theta_{sat_4}</td>
<td>Medium (c_r = 25%)</td>
<td>−8.1</td>
</tr>
</tbody>
</table>

1 This simulation was not performed as the perturbed parameter fell outside the range of acceptable values for this storm event.

Table 5.4: BPRM hydrologic sensitivity to perturbations in antecedent moisture conditions.

<table>
<thead>
<tr>
<th>Input Parameter</th>
<th>Uncertainty</th>
<th>Difference in total underdrain discharge volume (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Input parameter perturbation</td>
</tr>
<tr>
<td></td>
<td></td>
<td>+5%</td>
</tr>
<tr>
<td>Initial water content of the soil root zone</td>
<td>Medium (c_r = 25%)</td>
<td>4.7</td>
</tr>
<tr>
<td>Initial water content of the deep soil zone</td>
<td>Medium (c_r = 25%)</td>
<td>4.6</td>
</tr>
</tbody>
</table>

1 This simulation was not performed as the perturbed parameter fell outside the range of acceptable values for this storm event.

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The exfiltration modelling parameters were assigned a value of zero, as an impermeable liner prevents exfiltration from the Seneca College bioretention system. However, to assess the sensitivity of model predictions to the bioretention system perimeter and exfiltration parameters, a separate sensitivity analysis was conducted, in which the presence of the liner around bioretention soils was ignored. The exfiltration parameters were defined based on the characteristics of the native soils at the Seneca College monitoring site (TRCA 2008). The native soils were classified as clay loam, with a hydraulic conductivity ranging from $10^{-4}$ to $10^{-5}$ cm/s. Table 5.5 contains the exfiltration parameters used for this analysis. Figure 5.2 shows a comparison of the predicted underdrain flows with and without the impermeable liner surrounding the bioretention soils. The total underdrain discharge volume is reduced from 1.786 m$^3$ without exfiltration to 1.154 m$^3$ with exfiltration (35%). The sensitivity analysis results can be found in Table 5.6.
Table 5.5: Exfiltration parameters for the Seneca College bioretention system assuming that no liner prevents exfiltration from the soils.

<table>
<thead>
<tr>
<th>BPRM input</th>
<th>Symbol</th>
<th>Value</th>
<th>Units</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bioretention system perimeter</td>
<td>$P_b$</td>
<td>22.8</td>
<td>m</td>
<td>Site surveying</td>
</tr>
<tr>
<td>Saturated hydraulic conductivity of native soils</td>
<td>$K_{sat}$</td>
<td>1.8</td>
<td>mm/hr</td>
<td>TRCA (2008)</td>
</tr>
<tr>
<td>Capillary tension parameter of native soils</td>
<td>$\psi_N$</td>
<td>20.9</td>
<td>mm</td>
<td>Rawls et al. (1993)</td>
</tr>
<tr>
<td>Effective porosity of native soils</td>
<td>$M_N$</td>
<td>0.315</td>
<td>---</td>
<td>Rawls et al. (1982)</td>
</tr>
</tbody>
</table>

Table 5.6: BPRM hydrologic sensitivity to perturbations in exfiltration parameters.

<table>
<thead>
<tr>
<th>Input Parameter</th>
<th>Uncertainty</th>
<th>Difference in total underdrain discharge volume (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>+5%</td>
</tr>
<tr>
<td>Bioretention system perimeter</td>
<td>Low ($c_p = 10%$)</td>
<td>-2.6</td>
</tr>
<tr>
<td>Saturated hydraulic conductivity of native soils</td>
<td>High ($c_p = 50%$)</td>
<td>-1.3</td>
</tr>
<tr>
<td>Capillary tension parameter of native soils</td>
<td>Medium ($c_p = 25%$)</td>
<td>-1.3</td>
</tr>
<tr>
<td>Effective porosity of native soils</td>
<td>Medium ($c_p = 25%$)</td>
<td>-2.6</td>
</tr>
</tbody>
</table>
Figure 5.2: Comparison of predicted underdrain flows with and without an impermeable liner surrounding the bioretention soils.

The exfiltration parameters had a significant influence on modelling predictions. This result is not surprising, as exfiltration effectively decreases the total volume of water stored in bioretention soils, which influences both the volume of water available for underdrain discharge and the rate at which water travels through soil (as the hydraulic conductivity of soil decreases with soil water content reductions).

The sensitivity of BPRM hydrologic predictions to changes in modelling time step was assessed for the Jul. 12, 2006 storm event. The modelling time step used to evaluate the performance of the model (1 min) was divided and multiplied by factors of 5 and 10. For modelling time steps of 0.1 min and 0.2 min, rainfall intensities and runoff inflow rates were kept constant over the interval between field measurements (1 min for runoff and 5 min for rainfall). When field
measurements were available at a shorter interval than required by the modelling time step, an average of rainfall intensities and runoff inflow rates measured during each modelling time step was taken. Table 5.7 shows the results of the sensitivity analysis on modelling time steps and Figure 5.3 shows the variation in underdrain discharge rate with modelling time step.

<table>
<thead>
<tr>
<th>Time step (min)</th>
<th>Number of modelling time steps</th>
<th>Difference in total underdrain discharge volume (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.1</td>
<td>40200</td>
<td>1.1</td>
</tr>
<tr>
<td>0.2</td>
<td>20100</td>
<td>0.9</td>
</tr>
<tr>
<td>5</td>
<td>804</td>
<td>-5.2</td>
</tr>
<tr>
<td>10</td>
<td>402</td>
<td>-16.3</td>
</tr>
</tbody>
</table>

Figure 5.3: Predicted underdrain discharge rate comparison for different modelling time steps.
Peak flow rates were significantly higher under short modelling time steps than under long time steps. This is because, under shortened time steps, the time that water takes to travel through the bioretention system to the underdrain structure is decreased, which allows a greater volume of water to reach the underdrain in a short time period. As a result, total underdrain discharge volumes were also increased slightly under shortened modelling time steps. Little improvement in total underdrain discharge modelling predictions was found between time steps of 5 min and 0.1 min. However, a significant difference in modelling predictions was observed between time steps of 5 min and 10 min, which suggests that modelling time steps of 1 to 5 min provide good accuracy in total underdrain discharge volume predictions at minimal computational expense.

5.3.3 Sensitivity Analysis on Phosphorus Transport

The results of two different sensitivity analyses are presented in this section. First, a deterministic sensitivity analysis was performed on all modelling input parameters to identify the parameters which should be defined most carefully. Then, a MCS-based sensitivity analysis was performed to capture variations in the model outputs over the entire input parameter domain.

The deterministic sensitivity analysis was performed in a similar manner as the hydrologic sensitivity analysis presented above. In this case, the sensitivity of model predictions to input parameter perturbations was assessed based on
changes in total phosphorus mass in the bioretention system underdrain, as given by Eq. (5.3):

$$\Delta P = \frac{P_{\text{pert}} - P_{\text{exp}}}{P_{\text{exp}}} \times 100\%$$

(5.3)

where $\Delta P$ is the difference in total phosphorus underdrain mass caused by the perturbation of one modelling input parameter (%); $P_{\text{pert}}$ is the total phosphorus underdrain mass predicted with the perturbed parameter value (g); and $P_{\text{exp}}$ is the total phosphorus underdrain mass predicted with the initial parameter values (g). Results of the sensitivity analysis are found in Table 5.8 through Table 5.12.

BPRM predicted that increases in bioretention system area or bioretention soil depths in Table 5.8 would result in a decrease of the total phosphorus mass in the underdrain of the system. This result is consistent with the expectation that better water quality can be obtained in oversized bioretention systems. Increases in the rate of evapotranspiration also resulted in a reduction of the total phosphorus mass in the bioretention system underdrain. However, since evapotranspiration is a continuous process which occurs slowly, it has little impact on modelling predictions over the duration of a storm event.

Model predictions were insensitive to the system ponding capacity in Table 5.8, as no overflow was predicted by BPRM for this storm event under perturbed system ponding capacities. The mulch layer depth (Table 5.8) and mulch layer
saturated water content (Table 5.9) also did not influence modelling predictions, which suggests that soluble phosphorus sorption and particulate phosphorus capture in the mulch layer are secondary phosphorus transport processes in bioretention systems.
Table 5.8: BPRM phosphorus transport sensitivity to perturbations in bioretention system parameters.

<table>
<thead>
<tr>
<th>Input Parameter</th>
<th>Symbol</th>
<th>Uncertainty</th>
<th>Difference in total phosphorus mass (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Input parameter perturbation</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>+5%</td>
</tr>
<tr>
<td>Evapotranspiration rate</td>
<td>j_w</td>
<td>Low (c_v = 10%)</td>
<td>-0.1</td>
</tr>
<tr>
<td>Bioretention system area</td>
<td>A_b</td>
<td>Low (c_v = 10%)</td>
<td>-1.2</td>
</tr>
<tr>
<td>System ponding capacity</td>
<td>D_p</td>
<td>Low (c_v = 10%)</td>
<td>0.0</td>
</tr>
<tr>
<td>Mulch layer depth</td>
<td>D_2</td>
<td>Medium (c_v = 25%)</td>
<td>0.0</td>
</tr>
<tr>
<td>Soil root zone depth</td>
<td>D_3</td>
<td>Medium (c_v = 25%)</td>
<td>-1.7</td>
</tr>
<tr>
<td>Deep soil zone depth</td>
<td>D_4</td>
<td>Medium (c_v = 25%)</td>
<td>-1.2</td>
</tr>
</tbody>
</table>
Table 5.9: BPRM phosphorus transport sensitivity to perturbations in soil hydraulic parameters.

<table>
<thead>
<tr>
<th>Input Parameter</th>
<th>Symbol</th>
<th>Uncertainty</th>
<th>Difference in total phosphorus mass (%)</th>
<th>Input parameter perturbation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mulch saturated water content</td>
<td>$M_2$</td>
<td>Medium ($c_v = 25%$)</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Soil root zone saturated hydraulic conductivity</td>
<td>$K_{sat_3}$</td>
<td>Medium ($c_v = 25%$)</td>
<td>0.5</td>
<td>-0.4</td>
</tr>
<tr>
<td>Soil root zone capillary tension parameter</td>
<td>$\psi_5$</td>
<td>Low ($c_v = 10%$)</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Soil root zone van Genuchten parameter</td>
<td>$n_3$</td>
<td>Medium ($c_v = 25%$)</td>
<td>2.4</td>
<td>-2.7</td>
</tr>
<tr>
<td>Soil root zone residual water content</td>
<td>$\theta_{res}$</td>
<td>Medium ($c_v = 25%$)</td>
<td>1.5</td>
<td>-1.5</td>
</tr>
<tr>
<td>Deep soil zone saturated water content</td>
<td>$\theta_{sat_3}$</td>
<td>Medium ($c_v = 25%$)</td>
<td>-7.9</td>
<td>7.4</td>
</tr>
<tr>
<td>Deep soil zone saturated hydraulic conductivity</td>
<td>$K_{sat_4}$</td>
<td>Medium ($c_v = 25%$)</td>
<td>0.5</td>
<td>-0.4</td>
</tr>
<tr>
<td>Deep soil zone van Genuchten parameter</td>
<td>$n_4$</td>
<td>Medium ($c_v = 25%$)</td>
<td>3.0</td>
<td>-3.3</td>
</tr>
<tr>
<td>Soil root zone residual water content</td>
<td>$\theta_{res_4}$</td>
<td>Medium ($c_v = 25%$)</td>
<td>1.7</td>
<td>-1.7</td>
</tr>
<tr>
<td>Deep soil zone saturated water content</td>
<td>$\theta_{sat_4}$</td>
<td>Medium ($c_v = 25%$)</td>
<td>-8.2</td>
<td>8.5</td>
</tr>
</tbody>
</table>

$^1$This simulation was not performed as the perturbed parameter fell outside the range of acceptable values for this storm event.
BPRM phosphorus transport predictions were found to be particularly sensitive to the drainage parameters of bioretention soils (Table 5.9), which highlights the strong relationship between hydrologic processes and phosphorus transport in bioretention systems. The model also showed high sensitivity to the initial water content of bioretention soils (as noted in the hydrologic sensitivity analysis) in Table 5.12, and to the initial concentration of soluble phosphorus in the deep soil zone (Table 5.12). The initial concentration of soluble phosphorus in the soil root zone in Table 5.12, however, had no influence on the model predictions. This may be the result of the assumption that model layers are completely-mixed, such that the concentration of soluble phosphorus in the underdrain is taken as the concentration of soluble phosphorus in the deep soil zone.

As noted in the hydrologic sensitivity analysis, the soil root zone capillary tension parameter (in Table 5.9) had no effect on model predictions. Since hydrologic transport in the bioretention system was not significantly influenced by this parameter, its influence on phosphorus transport is also limited. This is due to the underlying model assumptions that layers are completely-mixed and that phosphorus is transported at the same velocity as water inside the bioretention system.

Most soluble phosphorus transport parameters in Table 5.10 had little to no influence on model predictions. Soluble phosphorus concentrations in the
underdrain appear to be affected mostly by hydrologic processes, perhaps because soluble phosphorus cycling processes have slow rates relative to the storm event duration. The predicted mass of soluble phosphorus in Table 5.10 was affected significantly only by perturbations of the sorption parameters in the deep soil zone. This seems to be an artefact of the model structure, as the underdrain discharge concentrations are taken as the concentrations of soluble phosphorus in the deep soil zone. While highly uncertain, none of the particulate phosphorus transport parameters in Table 5.11 and the particulate phosphorus initial concentrations in Table 5.12 significantly influenced the modelling predictions. The model predicted that soluble phosphorus was the main form of total phosphorus in the bioretention system underdrain, which may explain why modelling predictions were insensitive to changes in particulate phosphorus transport parameters.
<table>
<thead>
<tr>
<th>Input Parameter</th>
<th>Symbol</th>
<th>Uncertainty</th>
<th>Difference in total phosphorus mass (%)</th>
<th>Input parameter perturbation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mulch density</td>
<td>( \rho_2 )</td>
<td>High ((c_p = 50%))</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Density of soil in the root zone</td>
<td>( \rho_3 )</td>
<td>Medium ((c_p = 25%))</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Density of soil in the deep zone</td>
<td>( \rho_4 )</td>
<td>Medium ((c_p = 25%))</td>
<td>0.0 0.0 0.1 -0.1 0.1 -0.1</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Kinetic sorption rate constant</td>
<td>( \beta )</td>
<td>Medium ((c_p = 25%))</td>
<td>0.0 0.0 0.1 -0.1 0.1 -0.1</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Langmuir sorption constant for mulch</td>
<td>( k_{L_2} )</td>
<td>Medium ((c_p = 25%))</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Langmuir sorption constant for soil in the root zone</td>
<td>( k_{L_3} )</td>
<td>High ((c_p = 50%))</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Langmuir sorption constant for soil in the deep zone</td>
<td>( k_{L_4} )</td>
<td>High ((c_p = 50%))</td>
<td>-0.2 0.1 -0.5 0.6 -1.2 1.4</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Maximum sorption capacity of mulch</td>
<td>( Q_{max_2} )</td>
<td>Medium ((c_p = 25%))</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Maximum sorption capacity of soil in the root zone</td>
<td>( Q_{max_3} )</td>
<td>High ((c_p = 50%))</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Maximum sorption capacity of soil in the deep zone</td>
<td>( Q_{max_4} )</td>
<td>High ((c_p = 50%))</td>
<td>-0.2 0.2 -0.6 0.9 -1.1 3.5</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Total plant root surface area</td>
<td>( A_p )</td>
<td>High ((c_p = 50%))</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Bioavailable fraction of soluble phosphorus</td>
<td>( f_{BP} )</td>
<td>Low ((c_p = 10%))</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Maximum phosphorus vegetative uptake rate</td>
<td>( I_{max} )</td>
<td>Medium ((c_p = 25%))</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Minimum aqueous phosphorus concentration for uptake</td>
<td>( C_{min} )</td>
<td>Medium ((c_p = 25%))</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
<tr>
<td>Michaelis constant for phosphorus uptake</td>
<td>( v_M )</td>
<td>Medium ((c_p = 25%))</td>
<td>0.0 0.0 0.0 0.0 0.0 0.0</td>
<td>(+5% -5% +20% -20% +c_p -c_p)</td>
</tr>
</tbody>
</table>
Table 5.11: BPRM phosphorus transport sensitivity to perturbations in particulate phosphorus transport parameters.

<table>
<thead>
<tr>
<th>Input Parameter</th>
<th>Symbol</th>
<th>Uncertainty</th>
<th>Difference in total phosphorus mass (%)</th>
<th>Input parameter perturbation</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>+5%</td>
<td>−5%</td>
</tr>
<tr>
<td>Particle capture rate in soil of the root zone</td>
<td>( h_{cap_{r3}} )</td>
<td>High (( c_v = 50% ))</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Particle release rate in soil of the root zone</td>
<td>( h_{rel_{3}} )</td>
<td>High (( c_v = 50% ))</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Particle capture rate in soil of the deep zone</td>
<td>( h_{cap_{4}} )</td>
<td>High (( c_v = 50% ))</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Particle release rate in soil of the deep zone</td>
<td>( h_{rel_{4}} )</td>
<td>High (( c_v = 50% ))</td>
<td>0.0</td>
<td>0.0</td>
</tr>
</tbody>
</table>

Table 5.12: BPRM phosphorus transport sensitivity to perturbations in antecedent conditions.

<table>
<thead>
<tr>
<th>Input Parameter</th>
<th>Uncertainty</th>
<th>Difference in total phosphorus mass (%)</th>
<th>Input parameter perturbation</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>+5%</td>
<td>−5%</td>
</tr>
<tr>
<td>Initial water content of the soil root zone</td>
<td>Medium (( c_v = 25% ))</td>
<td>4.4</td>
<td>-4.5</td>
</tr>
<tr>
<td>Initial water content of the deep soil zone</td>
<td>Medium (( c_v = 25% ))</td>
<td>3.5</td>
<td>-3.6</td>
</tr>
<tr>
<td>Initial concentration of soluble phosphorus in the root zone</td>
<td>High (( c_v = 50% ))</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Initial concentration of soluble phosphorus in the deep zone</td>
<td>High (( c_v = 50% ))</td>
<td>1.7</td>
<td>-1.6</td>
</tr>
<tr>
<td>Initial concentration of particulate phosphorus in the root zone</td>
<td>High (( c_v = 50% ))</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Initial concentration of particulate phosphorus in the deep zone</td>
<td>High (( c_v = 50% ))</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Initial sorbed phosphorus content in mulch</td>
<td>High (( c_v = 50% ))</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Initial sorbed phosphorus content in soil of the root zone</td>
<td>High (( c_v = 50% ))</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Initial sorbed phosphorus content in soil of the deep zone</td>
<td>High (( c_v = 50% ))</td>
<td>0.2</td>
<td>-0.2</td>
</tr>
<tr>
<td>Initial captured phosphorus content in the root zone</td>
<td>High (( c_v = 50% ))</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Initial captured phosphorus content in the deep zone</td>
<td>High (( c_v = 50% ))</td>
<td>0.0</td>
<td>0.0</td>
</tr>
</tbody>
</table>

1 This simulation was not performed as the perturbed parameter fell outside the range of acceptable values for this storm event.
A separate analysis was performed to assess the sensitivity of BPRM phosphorus transport predictions to changes in bioretention system perimeter and exfiltration parameters. No liner was assumed to prevent exfiltration from the bioretention system and the exfiltration parameters reported in Table 5.5 were used for this analysis. The mass of total phosphorus in the bioretention system underdrain reduced from 5.642 g with the liner to 1.459 g without the liner (74% difference). Results of the sensitivity analysis can be found in Table 5.13.

As was noted for hydrologic modelling predictions, BPRM phosphorus transport predictions were significantly affected by changes in the exfiltration parameters. This is due to a significant decrease in the volume of water carrying phosphorus that reaches the underdrain structure. The predicted mass of total phosphorus in the bioretention underdrain increases under decreased exfiltration rates. Therefore, if the properties of the native soils surrounding a bioretention system are unknown, the assumption that exfiltration from the bioretention soils cannot occur should provide conservative results.
Table 5.13: BPRM hydrologic sensitivity to perturbations in exfiltration parameters.

<table>
<thead>
<tr>
<th>Input Parameter</th>
<th>Uncertainty</th>
<th>Difference in total underdrain discharge volume (%)</th>
<th>Input parameter perturbation</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>+5%</td>
<td>−5%</td>
</tr>
<tr>
<td>Bioretention system perimeter</td>
<td>Low (є_v = 10%)</td>
<td>-2.5</td>
<td>5.1</td>
</tr>
<tr>
<td>Saturated hydraulic conductivity of native soils</td>
<td>High (є_v = 50%)</td>
<td>-1.2</td>
<td>2.5</td>
</tr>
<tr>
<td>Capillary tension parameter of native soils</td>
<td>Medium (є_v = 25%)</td>
<td>-1.2</td>
<td>2.5</td>
</tr>
<tr>
<td>Effective porosity of native soils</td>
<td>Medium (є_v = 25%)</td>
<td>-1.2</td>
<td>3.8</td>
</tr>
</tbody>
</table>
The modelling time step used to evaluate the performance of BPRM (1 min) was varied to assess the sensitivity of phosphorus transport predictions to time step selection. Rainfall intensities and runoff flow rates were kept constant over the field measurement interval (1 min for runoff, 5 min for rainfall) for short modelling time steps. For long modelling time steps, averages of the field measurements within a modelling time step were calculated. Table 5.14 shows the results of the phosphorus transport sensitivity analysis on modelling time steps.

### Table 5.14: BPRM phosphorus transport sensitivity to changes in modelling time step.

<table>
<thead>
<tr>
<th>Time step (min)</th>
<th>Number of modelling time steps</th>
<th>Difference in total phosphorus underdrain mass (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.1</td>
<td>40200</td>
<td>0.7</td>
</tr>
<tr>
<td>0.2</td>
<td>20100</td>
<td>0.6</td>
</tr>
<tr>
<td>5</td>
<td>804</td>
<td>-4.7</td>
</tr>
<tr>
<td>10</td>
<td>402</td>
<td>-15.0</td>
</tr>
</tbody>
</table>

Increases in total phosphorus underdrain mass predictions were noted for short modelling time steps, while long modelling time steps yielded decreases in total phosphorus underdrain mass predictions. This follows the underdrain discharge volume predictions noted above. Phosphorus transport modelling predictions varied little under modelling time steps of 5 min. Modelling time steps of 10 min
were associated to significant changes in modelling predictions however, which corresponds to the observations that were made for hydrologic modelling predictions.

The initial portion of a storm event often carries disproportionately large concentrations of pollutants deposited on impermeable surfaces during dry periods, in a phenomenon referred to as the “first flush”. The sensitivity of BPRM to the occurrence of a “first flush” of phosphorus was assessed by increasing the influent phosphorus concentration during the first 100 minutes of the Jul. 12, 2006 storm event, while keeping the total inflow mass of phosphorus for the event constant. In this scenario, 80% of the total mass of influent phosphorus was carried by the first 25 L of the total 1,291 L influent runoff volume. No significant change in model predictions was observed, as the total phosphorus mass predicted increased by 1 mg, corresponding to a difference of 0.01%. Outflow concentrations were unaffected, as can be seen in Figure 5.4. Modelling results seem to be affected by the total mass of phosphorus in the bioretention system rather than by the concentration of phosphorus in the inflow, which is reasonable since the concentration of phosphorus in bioretention soils is typically significantly greater than the concentration of phosphorus in the bioretention system inflow.
A Monte-Carlo analysis was performed on the Jul. 12, 2006 storm event to assess the combined effect of uncertainty in multiple hydrological and phosphorus parameters on BPRM predictions. One thousand model runs were performed using randomly-generated input parameter sets. All modelling input parameters were assumed independent and normally distributed, with standard deviations defined by the coefficient of variation of each parameter (indicated in Table 5.8 through Table 5.12). Input parameter PDFs were truncated to ensure that values which are not physically-based (e.g., negative volumes) were not generated. The probability space underneath the truncated distributions was renormalized to 1.0. In the case of the saturated water contents in both the soil root zone and the deep soil zone, the minimum parameter values were allowed to fluctuate for each simulation based on the residual water content defined for the layer. The
saturated water content distribution was thus renormalized individually for each simulation based on the truncation applied for the simulation. In effect, the distribution assigned to the saturated water contents of bioretention soils varied from one simulation to the next to accommodate the residual water content generated for the soils. The initial water contents of bioretention soils in both the root zone and the deep zone were treated similarly. In this case, however, both the minimum and maximum distribution values were allowed to fluctuate based on the residual and saturated water content of the layers. Figure 5.5 illustrates the process by which normal PDFs for the initial and saturated water contents of bioretention soils were renormalized. The results of the Monte Carlo analysis are presented in a cumulative probability plot in Figure 5.6.

Figure 5.5: Examples of renormalized PDFs for the saturated and initial water content of bioretention soils.
Most model predictions of total phosphorus underdrain mass fell between 0.1 g and 5 g. Figure 5.6 was plotted using a logarithmic scale on the horizontal axis for clarity. The logarithmic scale does not indicate a value of zero, and total phosphorus was not present in the underdrain for predictions that fell below the 0.0001 g of total phosphorus plotted in Figure 5.6. Total phosphorus was absent in the underdrain in 25.5% of the model simulations. This occurred when all influent stormwater is retained by bioretention soils or evaporated from the system. The MCS-based sensitivity analysis results suggest that BPRM input parameter uncertainties could result in an underestimation of the total phosphorus mass in the underdrain of a bioretention system. Further investigation is required to verify whether this conclusion applies to all storm events and bioretention systems, but this should be taken into account when the
modelling results are used for design purposes. The analysis also reveals that modelling predictions can vary over several orders of magnitude depending on input parameter selection. For this reason, correctly estimating modelling input parameters is essential to obtain reliable model predictions. However, it should be noted that some BPRM input parameters; such as the initial, residual and saturated water contents of bioretention soils, as well as the soil root zone and deep soil zone depths; are not truly independent. For this reason, the MCS-based sensitivity analysis results presented may overestimate the actual sensitivity of the model to input parameter sets.

5.4 Bioretention Processes for Phosphorus Removal

A limited number of phosphorus transport processes were included in BPRM. Despite this limitation, the model can be used to develop a preliminary understanding of the importance of bioretention processes in phosphorus removal. A careful examination of the rate of individual bioretention processes can also provide a greater understanding of the sensitivity of model predictions to uncertainties in hydrologic and phosphorus transport parameters. Predicted rates of bioretention processes were examined and compared to identify important bioretention processes and explain significant model prediction sensitivities to uncertain input parameters identified and discussed above.
Figure 5.7 shows the rate of infiltration and percolation through the Seneca College bioretention system as predicted by BPRM for the Jul. 12, 2006 storm event. Infiltration to the mulch layer is only limited by the layer’s storage capacity, which was not exceeded during the Jul. 12, 2006 storm event, such that the rate of infiltration to the mulch layer closely followed the bioretention system inflow rate throughout the event. The rate of infiltration to the soil root zone followed the rate of infiltration to the mulch layer at the beginning of the storm event. As the soil root zone approached saturation, however, the rate of infiltration to the layer became limited by its storage capacity. The rate of percolation to the deep soil zone, which is defined by the drainage properties of the soil root zone, limited the flow of stormwater through bioretention soils. This explains the high sensitivity of hydrologic and phosphorus transport predictions produced by BPRM to the drainage parameters of soil in the root zone. The rate of evapotranspiration (not shown on Figure 5.7) was nearly insignificant for this storm event, and no overflow occurred during the event.
Figure 5.7: Predicted infiltration and percolation rates in the Seneca College bioretention system for the Jul. 12, 2006 storm event.

Figure 5.8 shows both the soluble and particulate phosphorus underdrain discharge rates predicted by BPRM. Soluble phosphorus was predicted to represent the main form of phosphorus in the bioretention system underdrain, as shown by the negligible particulate phosphorus underdrain discharge rate. This explains why phosphorus transport modelling predictions are relatively insensitive to particulate phosphorus cycling processes.
The predicted rates of soluble phosphorus transport processes for the Jul. 12, 2006 storm event at the Seneca College bioretention system are plotted in Figure 5.9, while Figure 5.10 shows the predicted rates of particulate phosphorus transport processes. The modelling results would suggest that the phosphorus leaching observed at the Seneca College bioretention system for the Jul. 12, 2006 storm was caused by soluble phosphorus desorption from the root and deep soil zones, as indicated by the negative values in Figure 5.9. This phenomenon has been observed in previous laboratory and field studies (Davis et al. 2006; Hunt et al. 2006). Some particulate phosphorus release from bioretention soils is also observed in Figure 5.10, but the release rates are insignificant compared to the soluble phosphorus desorption rates.
The vegetative uptake rates predicted for the Seneca College bioretention system are similar but opposite to the rates of soluble phosphorus desorption from the root zone. The results would indicate that vegetation takes up soluble phosphorus as it is made available to the soil solution through desorption, which stabilizes the soluble phosphorus concentrations in the soil root zone. This may partly explain the low sensitivity of phosphorus transport model predictions to soluble phosphorus processes inside the soil root zone. The total mass of phosphorus that can be taken up by vegetation over a long time period is not limited in the BPRM model at this time, which may not be representative of the behaviour of vegetation in bioretention systems. Note that high rates of soluble phosphorus desorption and vegetative uptake occur in the root zone during the first few modelling time steps, until stable concentrations are reached in the layer (see Figure 5.9). However, the sensitivity analysis revealed no change in predicted underdrain mass of total phosphorus under perturbations of the initial concentration of soluble phosphorus in the soil root zone.
Figure 5.9: Predicted soluble phosphorus process rates for the Jul. 12, 2006 storm event.

Figure 5.10: Predicted particulate phosphorus process rates for the Jul. 12, 2006 storm event.

The model predictions presented in Figure 5.9 and Figure 5.10 suggest that the mulch layer at the Seneca College bioretention system was not very effective at
sorbing soluble phosphorus or capturing particulate phosphorus. These results were expected since mulch has a low phosphorus sorption capacity and a coarse texture with limited straining capacity. This suggests that mulch has a limited role to play in the removal of phosphorus in bioretention systems.

The phosphorus content of soils in the Seneca College bioretention system was measured twice during the monitoring period. The first measurements were taken in the fall of 2005 after construction of the system. Another series of measurements was then made in the summer of 2007 to assess the degree of contamination of bioretention soils after 2 years of operation. Figure 5.11 shows the soil phosphorus contents measured in 2005 and 2007 at different depths. A t-test was performed on the measurements to determine whether the decrease in average soil phosphorus content observed between 2005 and 2007 was significant. The decrease in soil phosphorus content was found to be statistically significant at a 5% significance level, which supports the hypothesis that soluble phosphorus desorption is responsible for the phosphorus leaching observed at the Seneca College bioretention system.
5.5 Discussion

The analysis presented in this paper relied on a single storm event to assess the sensitivity of modelling predictions to input parameter selection. In Chapter 4, it was noted that modelling performance varies from one storm event to another due to the specific event characteristics. The sensitivity of the model should be assessed under a number of different storm events to verify whether the results presented in this chapter can be generalized to most storm events.

The mass of total phosphorus in the bioretention underdrain was chosen as model output in this analysis to evaluate model sensitivity to parameter uncertainties. Total phosphorus mass was selected because soluble and particulate phosphorus concentrations were not directly measured at the Seneca
College bioretention system. Maximum loading regulations for sensitive ecosystems are also generally defined in terms of total phosphorus (United States Environmental Protection Agency (USEPA) 2007). However, designers may be interested in the concentration or mass of a specific form of phosphorus in the bioretention system underdrain. In particular, soluble orthophosphate is generally considered as the main form of phosphorus readily available for vegetation uptake, such that it can promote rapid algal growth in sensitive water bodies. Other phosphorus forms normally need to undergo physical and chemical changes before they can be incorporated into plant biomass (Schachtman et al. 1998).

The results of the MCS-based sensitivity analysis suggest that parameter uncertainties on the order of 10-50% can produce significant modelling prediction uncertainties that range over 2-3 orders of magnitude. In particular, BPRM was found to be highly sensitive to the selection of drainage parameters for bioretention soils in both the root zone and the deep zone. If possible, laboratory or field testing should be performed on the bioretention soils to improve the accuracy of the soil drainage input parameters. However, since the van Genuchten soil drainage parameters are not physically-based parameters, but rather empirical values, designers could not directly specify required drainage parameters for bioretention soils. Many researchers have correlated physical and chemical soil properties to the van Genuchten drainage parameters
used in BPRM (Rawls et al. 1982; Carsel and Parrish 1988; van Genuchten et al. 1991; Schaap et al. 2001). These correlations can be used to specify physical and chemical properties for bioretention soils.

The model developed was found to be particularly sensitive to antecedent moisture conditions in bioretention soils. To improve the accuracy of modelling predictions, an effort should be made to quantify the bioretention soil moisture content before a storm event. In the future, BPRM could be modified to include both short-term and long-term modelling processes, such that it could be used for continuous modelling simulations. A continuous model would likely be less sensitive to antecedent system conditions, and could be used to estimate total phosphorus loadings to receiving ecosystems over long time periods.

5.6 Conclusions and Recommendations

A model to simulate phosphorus transport in bioretention systems over the duration of a storm event was introduced in Chapter 4. The BPRM model was found to produce estimates of total phosphorus removal appropriate for the design of a bioretention system. In this chapter, the sensitivity of model predictions was assessed using field data collected by the TRCA (2008) at a bioretention system constructed on the Seneca College King City Campus in the Greater Toronto Area (Ontario, Canada).
BPRM hydrologic and phosphorus removal predictions were found to be particularly sensitive to the drainage properties of bioretention soils. Uncertainties in initial water contents inside bioretention soils and uncertainties in the initial concentration of soluble phosphorus in the deep soil zone caused wide variations in predicted underdrain discharge volume and predicted total phosphorus underdrain mass. While most phosphorus transport parameters had medium to high uncertainty, they had little influence on the predicted total phosphorus underdrain mass. Depending on the input parameters selected, phosphorus removal predictions were found to range over several orders of magnitude, such that careful parameter selection is a crucial modelling step when using BPRM.

Modelling results suggest that soluble phosphorus sorption to bioretention soils and vegetative uptake are the two most important phosphorus cycling processes in bioretention systems for the storm event considered. BPRM predictions point to soluble phosphorus desorption as the cause of phosphorus leaching from the Seneca College bioretention system during the storm event considered. This theory is supported by the decrease in bioretention soil phosphorus content observed over the monitoring period.
5.7 References


Chapter 6
Conclusions and Recommendations

6.1 Engineering Contribution

The objective of the work presented in this thesis was to develop a simulation model for phosphorus transport in bioretention systems. The main purpose of the model would be to assist designers in ensuring that planned bioretention facilities provide the level of phosphorus treatment required to protect receiving water bodies. The model developed would help designers gain an understanding of phosphorus cycling in bioretention systems and the relative importance of individual bioretention processes. To encourage the adoption of the model by bioretention system by a variety of bioretention systems designers, the model should be made simple and user-friendly.

In Chapter 4, a model was developed to simulate phosphorus transport within bioretention systems. The Bioretention Phosphorus Removal Model (BPRM) simulates both soluble and particulate phosphorus transport within bioretention systems over the duration of a storm event. Efforts were made to keep the level of complexity of the BPRM model to a minimum, while capturing the important processes for phosphorus removal in bioretention systems. To encourage the adoption of BPRM by practitioners, a user-friendly interface was also built for the model, which is supplemented by a detailed user manual. In addition, the BPRM
model produces detailed output files that allow program users to examine the rate of each bioretention process included in the model at all modelling time steps. This allowed for a preliminary investigation of the relative importance of different phosphorus cycling processes within a bioretention system.

6.2 Summary of findings

Bioretention systems are promising stormwater BMPs which have shown great potential for the reduction of influent stormwater volumes and peak flow rates, as well as the retention of a large array of important stormwater pollutants, including total suspended solids, heavy metals, fecal coliforms, oil and grease, and polycyclic aromatic hydrocarbons (PAHs). However, nutrient removal has been inconsistent between bioretention systems. In particular, high discrepancies in phosphorus removal from bioretention systems have been noted in the field, with occasional phosphorus leaching observed (Dietz and Clausen 2005; Hunt et al. 2006). A number of studies have identified processes that significantly influence phosphorus removal in bioretention systems (Davis et al. 2006; Hunt et al. 2006; Henderson et al. 2007; Bratieres et al. 2008), but the lack of understanding of the overall phosphorus cycle in bioretention systems and the relative importance of different bioretention processes remains to be addressed.

Currently, a number of hydrologic models are available in the literature to simulate stormwater retention in bioretention systems (Dussaillant et al. 2003;
Dussaillant et al. 2004; Heasom et al. 2006; Aravena and Dussaillant 2009). Models have also been developed to simulate the fate of heavy metals (Li and Davis 2008a), total suspended solids (Li and Davis 2008b), and polycyclic aromatic hydrocarbons (PAHs) (Diblasi et al. 2009) in bioretention systems. A need for a numerical model which can predict phosphorus transport and cycling within bioretention systems was identified. A number of phosphorus transport models currently available in the literature for systems similar to bioretention have been reviewed, but the applicability of these models to bioretention systems is limited.

A model was developed to simulate phosphorus transport and cycling within bioretention systems throughout of a storm event. The Bioretention Phosphorus Removal Model (BPRM) comprises four completely-mixed model layers which represent different components of a bioretention system: the layer of water ponding at the surface of the system; the optional layer of mulch covering bioretention soils; the upper portion of bioretention soils which contain the vegetation rhizosphere; and the remaining bioretention soil portion. Soluble and particulate phosphorus forms are distinguished in BPRM, and processes included in the model are evapotranspiration, overflow, infiltration, exfiltration to native soils, underdrain discharge, soluble phosphorus sorption and vegetative uptake, and particulate phosphorus capture.
BPRM modelling predictions were evaluated using monitoring data collected by the Toronto and Region Conservation Authority (TRCA) at a bioretention system installed on Seneca College’s King City campus in the Greater Toronto Area (Ontario, Canada). Due to limited field data availability, most modelling input parameters were selected from values reported in the literature. In practice, bioretention system designers are often faced with similar issues with respect to field data availability. Measured and predicted underdrain discharge rates and phosphorus outflow concentrations and mass were compared for selected storm events. The BPRM model was found to overestimate underdrain discharge volumes, but fair agreement between measured and predicted underdrain discharge curves was noted for most storm events. BPRM predictions of total phosphorus concentration and mass in the bioretention outflow were found to be practical for design purposes. The accuracy of phosphorus transport modelling predictions was found to be closely related to the accuracy of hydrologic predictions, highlighting the strong relationship between water quality performance and stormwater retention in bioretention systems. BPRM correctly predicted phosphorus leaching from the Seneca College bioretention system for all but one of the storm events considered. Early results indicate that, when used with carefully selected input parameters, the BPRM model can identify the potential for phosphorus leaching from a bioretention system. Further testing is required to confirm this.
A detailed sensitivity analysis of the BPRM model revealed that hydrologic and phosphorus transport modelling predictions are particularly sensitive to the drainage properties of bioretention soils. Changes in the initial water content of bioretention soils and the initial concentration of soluble phosphorus in the deep soil zone also significantly influenced phosphorus transport predictions. In contrast, soluble and particulate phosphorus transport parameters had little influence on the predicted mass of total phosphorus in the bioretention underdrain. Modelling input parameters should be selected carefully, as the Monte Carlo simulation analysis revealed that the predicted mass of total phosphorus in the underdrain of a bioretention system can vary over 2 to 3 orders of magnitude under different input parameters.

BPRM predictions suggested that soluble phosphorus vegetative uptake and sorption to bioretention soils were the two most important phosphorus cycling processes within the Seneca College bioretention system during the storm event modelled. Modelling results also suggested that phosphorus leaching from the Seneca College bioretention system was caused by soluble phosphorus desorption from bioretention soils. This hypothesis agrees with the statistically-significant decrease in phosphorus content observed in the Seneca College bioretention soils over the field monitoring period.
6.3 Recommendations for future work

The accuracy of BPRM phosphorus transport predictions was also found to depend to a large extent on accurately modelling hydrologic processes within bioretention systems. Consequently, the hydrologic component of the BPRM model presented in this thesis could be refined to improve the accuracy of modelling predictions. Currently available hydrologic models for bioretention systems that rely on a one-dimensional (Dussaillant et al. 2004) or two-dimensional (Aravena and Dussaillant 2009) version of Richard’s equation could be incorporated into the BPRM model for this purpose. The Richard’s equation models are expected to predict hydrologic processes within bioretention systems with better accuracy than the Green-Ampt infiltration model used in the current version of BPRM (Dussaillant et al. 2003).

BPRM modelling accuracy could be improved by introducing a greater number of model layers in the bioretention soils, or by transitioning from completely-mixed model layers to plug flow modelling. These changes could be easily integrated into the current model structure and may improve modelling results considerably. Moreover, localized infiltration at the inlet of bioretention systems is expected to be significant under low inflow rates. For this reason, the inclusion of a two-dimensional flow component into BPRM could significantly improve modelling results for small storms. Modelling predictions for large storm events are not expected to be affected as appreciably by the inclusion of two-dimensional
transport within the soils, however. Because horizontal flow behaviour in bioretention systems is difficult to represent mathematically and can vary significantly based on system inlet design and soil properties, efforts to improve modelling results should first focus on improving the one-dimensional vertical component of flow in bioretention systems.

The focus of this thesis was to provide a simple model which could be used by practitioners for design purposes. While modelling results did provide some insight into the relative importance of bioretention processes for phosphorus removal, extensions of this work should consider the development of a more complex and comprehensive numerical model, with the purpose of studying the phosphorus cycle in bioretention systems in detail. Such a model could be used to supplement field monitoring and laboratory testing results in the improvement of bioretention system design guidelines.

While extensive research has been performed to identify and quantify phosphorus cycling processes in soils, the mathematical representations available to describe certain processes (mineralization and immobilization in particular) are limited. Further research is required to develop more accurate mathematical representations of phosphorus cycling processes in soils. In addition, limited data is available on the rate of phosphorus uptake by plants and microbes that are typically found in bioretention systems. Instead, research has focused mostly on
quantifying phosphorus uptake in crop plants, trees with significance to the forestry industry, and marine microorganisms. Further research is also required to determine the rate at which colloidal particles are captured in unsaturated bioretention soils. Until now, colloidal capture and release rates have been reported mostly for uniformly-sized sand media. Bioretention soils differ from these media as they are typically biologically-active fine-grained soils with a wide particle size distribution.

The framework of the BPRM model presented in this thesis could be used to develop water quality models for other important stormwater pollutants, such as nitrogen, organic matter (measured by its biochemical oxygen demand (BOD)), oil and grease, and pathogens. In particular, the nitrogen cycle in bioretention systems shares some important similarities with the phosphorus cycle in the systems (Prince George’s County (PGC) 2007). Consequently, the structure of the BPRM model could be easily adapted to simulate nitrogen transport within a bioretention system. As nitrogen cycling in soils is highly dependent on the hydrologic conditions in the soil (denitrification occurs under anaerobic conditions which develop when soils are saturated, while nitrification occurs under aerobic conditions), nitrogen transport models for bioretention systems can be expected to display a high sensitivity to hydrologic processes, as was observed with BPRM for phosphorus cycling.
6.4 References


Prince George's County (PGC). (2007). "Bioretention Manual." Prince George's County, Maryland, Department of Environmental Resources, Environmental Services Division, Landover, MD.
Appendix A
Hydrologic Modelling Results

This section contains hydrologic modelling results for the storm events that were not included in Chapter 4.

Figure A.1: Measured and predicted underdrain discharge rate for Jul. 10, 2006 storm event.
Figure A.2: Measured and predicted underdrain discharge rate for Jul. 12, 2006 storm event.

Figure A.3: Measured and predicted underdrain discharge rate for Sept. 27, 2006 storm event.
Figure A.4: Measured and predicted underdrain discharge rate for Nov. 7, 2006 storm event.

Figure A.5: Measured and predicted underdrain discharge rate for Jun. 3, 2007 storm event.
Figure A.6: Measured and predicted underdrain discharge rate for Sept. 25, 2007 storm event.