Evaluating WASCoBs, Vegetative Filter Strips and Road-side Ditches in a Rural Watershed

by

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ABSTRACT

EVALUATING WASCOBS, VEGETATIVE FILTER STRIPS AND ROAD-SIDE DITCHES IN A RURAL WATERSHED

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University of Guelph, 2018

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Ramesh P. Rudra,
Bahram Gharabaghi

Modeling the spatial dynamics of certain BMPs like WASCoBs (Water and sediment control basins), road-side ditches, and vegetative filter strips at a watershed scale could be a challenging issue since these BMPs are implemented at a field-scale at specific locations within a watershed. Due to these challenges, very few attempts have been made to develop models for quantifying the effect of some of these BMPs upon hydrology and sediment yield at a watershed scale. Further for some BMPs, no models are available to investigate their performance at a watershed scale at all.

This research is divided into four main sections. The first section discusses how a toolbox named CoBAGNPS was developed to model WASCoBs through AGNPS model. The developed toolbox utilizes the inputs from AGNPS, through the launching of an application for execution of the WASCoB module. The toolbox was tested for the Gully Creek watershed and one of its sub-basins located in Ontario, Canada. In the second section, the performance of the toolbox for modeling two types of surface inlets, namely pipe risers, and blind inlets is investigated, and their impact upon sediment yield was explored. Analysis showed that blind inlets were more efficient in detaining sediment
within the berm of the WASCoBs compared to pipe risers. The third section includes modeling the impact of road-side ditches through KINEROS 2 model. Various scenarios pertaining to grass cover lining and its impact upon downstream flow and sediment yield were investigated for the watershed. In the fourth section, a modeling approach was developed using AGNPS, AGNPS_VFS toolkit, and a regression equation to evaluate edge of fields vegetative filter strips at a watershed scale. This modeling approach demonstrates a simple method to predict the impact of edge of fields vegetative filter strips upon sediment reduction at a watershed scale. This research will help to locate the most appropriate place to constructing vegetative filter strips as targeted BMPs to control sediment yield.
DEDICATION

I would like to dedicate this thesis to the person I love the most in my life, my son Reyansh Gupta, and my brother-in-law Deepak Jain.
ACKNOWLEDGMENTS

I would like to express my sincere thanks and appreciation to my thesis supervisors, Profs. Ramesh P. Rudra and Bahram Gharabaghi, for their guidance, support and constructive criticism throughout the period of this Ph.D. program. A profound debt is acknowledged to Dr. Prasad Daggupati for his unfailing patience and guidance during this study.

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<tr>
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<tbody>
<tr>
<td>ABCA</td>
<td>Ausable Bayfield Conservation Authority</td>
</tr>
<tr>
<td>AGNPS</td>
<td>Agricultural Non-Point Source Pollution Model</td>
</tr>
<tr>
<td>AMC</td>
<td>Antecedent moisture condition</td>
</tr>
<tr>
<td>AMLE</td>
<td>Adjusted maximum likelihood estimation</td>
</tr>
<tr>
<td>APEX</td>
<td>Agriculture Policy/Environmental eXtender</td>
</tr>
<tr>
<td>CN</td>
<td>Curve number</td>
</tr>
<tr>
<td>DFTILE</td>
<td>Tile outlet monitoring station at the sub-basin outlet</td>
</tr>
<tr>
<td>DNDP</td>
<td>Drainage network/drainage pattern</td>
</tr>
<tr>
<td>DNDP_M</td>
<td>Road-side ditches with the same Manning’s n</td>
</tr>
<tr>
<td>DNDP_MV</td>
<td>Road-side ditches lined with medium vegetation</td>
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<tr>
<td>DNDP_HV</td>
<td>Road-side ditches lined with thick vegetation</td>
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<tr>
<td>ENS</td>
<td>Nash–Sutcliffe efficiency</td>
</tr>
<tr>
<td>GUL_RSD</td>
<td>A sub-basin outlet</td>
</tr>
<tr>
<td>GULGUL 5</td>
<td>Watershed outlet</td>
</tr>
<tr>
<td>HSPF</td>
<td>Hydrologic Simulation Program FORTRAN</td>
</tr>
<tr>
<td>KINEROS 2</td>
<td>Kinematic Run-off and Erosion Model</td>
</tr>
<tr>
<td>K2</td>
<td>KINEROS 2</td>
</tr>
<tr>
<td>LAD</td>
<td>Least absolute deviation</td>
</tr>
<tr>
<td>LOADEST</td>
<td>LOAD ESTIMATOR</td>
</tr>
<tr>
<td>L-THIA</td>
<td>Long Term Hydrologic Impact Analysis</td>
</tr>
<tr>
<td>MLE</td>
<td>Maximum likelihood estimation</td>
</tr>
<tr>
<td>OMAFRA</td>
<td>Ontario Ministry of Agriculture, Food, and Rural Affairs</td>
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<tr>
<td>OMNRF</td>
<td>Ontario Ministry of Natural Resources and Forestry</td>
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<tr>
<td>PLOAD</td>
<td>GIS Pollutant Load Application</td>
</tr>
<tr>
<td>R²</td>
<td>Coefficient of determination</td>
</tr>
<tr>
<td>RSD</td>
<td>Road-side ditches</td>
</tr>
<tr>
<td>STEPL</td>
<td>Spreadsheet Tool for Estimating Pollutant Load</td>
</tr>
<tr>
<td>SL</td>
<td>Sediment loads</td>
</tr>
<tr>
<td>SRE</td>
<td>Sediment reducing efficiency</td>
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<tr>
<td>Acronym</td>
<td>Description</td>
</tr>
<tr>
<td>---------</td>
<td>-----------------------------------------------</td>
</tr>
<tr>
<td>SWAT</td>
<td>Soil and water assessment tool</td>
</tr>
<tr>
<td>VFS</td>
<td>Vegetative filter strips</td>
</tr>
<tr>
<td>VFSMOD</td>
<td>Vegetative Filter Strip MODel</td>
</tr>
<tr>
<td>WASCoB</td>
<td>Water and Sediment Control Basin</td>
</tr>
<tr>
<td>WHAT</td>
<td>Web-based hydrograph analysis Tool</td>
</tr>
<tr>
<td>Bi</td>
<td>Blind inlets</td>
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<tr>
<td>OSL</td>
<td>Observed sediment load</td>
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<tr>
<td>SSL</td>
<td>Simulated sediment load</td>
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<tr>
<td>OSR</td>
<td>Observed surface runoff</td>
</tr>
<tr>
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Chapter 1

1 Introduction

“The battle for water is as old as the life on the Earth. A man certainly first fought for water, and then against floods. That battle for water actually continues today, with a conclusion that it has never stopped. Today’s battle is more complex and difficult and will be even harder for future generations. Once sufficient supplies of water for human use could be found in nature, but the situation is different today. There is virtually no water left suitable for use that has not been upgraded by a human action, thus water has become a commodity, a commodity of special significance.” (Djekovic et al., 2016). Furthermore, water in past, present and in the future will always be an irreplaceable food item (Mihailovic, 2007).

During the world summit on Sustainable Development in 2002, all nations argued for framing up a national integrated water resources management (IWRM) plan to initiate a launching pad towards worldwide water security (Global Water Partnership, 2008). Adequate management and planning of water resources at a watershed scale is the basis of IWRM, under which surface and groundwater should be examined properly to attain sustainable development (Global Water Partnership, 2008). To meet the thriving demand for food, agriculture expansion through land use changes (Taniwaki et al., 2017), and enhancing crop productivity on currently cultivable land is gaining momentum. Transforming plant phenotypes, where plants could thrive under inhospitable land
(Reynolds and Borlaug, 2006), improving irrigation techniques (Fereres and Soriano, 2007), increased use of pesticides and fertilizers are a few practical approaches adopted to accomplish the objective.

The increased application of pesticides and fertilizers for enhancing crop productivity has become a universal practice. An escalation of 88 and 3150 % respectively, in the employment of 2,4-D and glyphosate has been observed in the past two decades (Gonzalez et al., 2016). Herbicides, insecticides, and fungicides in excess of half a billion pounds were used per year between 1992 and 2011 for enhancing crop production and mitigate insect-borne disease (Stone et al., 2014). These fertilizers and pesticides are subsequently transported through the watershed within the aquatic ecosystems, thereby impairing water quality significantly (Rodriguez-Lloveras et al., 2015; Worrall and Burt, 1999). Elevation in macronutrients loads, particularly phosphorus (P) and nitrogen (N) could elevate the danger of eutrophication in freshwater ecosystems (Heathwaite et al., 2005). Furthermore, degradation of water quality also has severe financial consequences, for instance, an increase in water treatment costs (Cunha et al., 2016). This type of pollution commencing primarily from diffuse sources (mainly agricultural activities) is often labelled as non-point source (NPS) pollution. NPS pollution is a predominant source of water quality deterioration in agricultural watersheds (Hu and Huang, 2014). Consequently, United States Environmental Protection Agency (USEPA) was setup in the U.S to provide guidelines for water quality including benchmarks for aquatic life for the abatement of NPS pollution. In Canada, however, water-pollution is mainly responsibility of the provinces. Nonetheless, there are certain federal legislations
which regulate the discharge of pollutants in Canada. Noticeably, the Fisheries Act of 1985 and the Canadian Environmental Protection Act of 1999 (CEPA) (Poulton, 2016).

In Canada, in different parts of the country, distinct threshold limits have been setup for nutrient concentration (P and N) in streams viz, [(Atlantic Maritime region: 0.03 mg L\(^{-1}\) for total P and 0.87–1.2 mg L\(^{-1}\) for total N), (Montane Cordillera region: 0.02 mg L\(^{-1}\) total P and 0.21 mg L\(^{-1}\) total N), (Mixedwood Plains region: approximately 0.03 mg L\(^{-1}\) total P and 1.1 mg L\(^{-1}\) total N), (interior prairies of Canada region: approximately 0.10 mg L\(^{-1}\) total P and 0.39–0.98 mg L\(^{-1}\) total N)] (Chambers et al., 2012). Further, for surface water bodies, a threshold of 3 mg L\(^{-1}\) has been established for NO\(_3^-\) N in Canada (Canadian Council of Ministers of the Environment, 2003).

Adoption of appropriate management practices commonly referred to as “Best Management Practices (BMPs)” is an effective methodology for minimizing NPS pollution, originating primarily from agricultural landuse activities for achieving adequate ecological and chemical conditions for maintaining water quality standards. Although research pertaining to BMPs has focused primarily on water quality, their impact upon watershed hydrology is also significant and more attention should be paid towards its investigation (Sloan et al., 2017).

Some of the BMPs increasing becoming popular in North America in recent times are water and sediment control basins (Fiener et al., 2005; Her et al., 2017; Kovacic et al., 2006; Verstraeten and Poesen, 1999), blind inlets (Feyereisen et al., 2015; Gonzalez et al., 2016; Smith et al., 2015), pipe risers (King et al., 2015; Li et al., 2017; Oolman and Wilson, 2003), agricultural drains (Needelman et al., 2007; Skaggs et al., 1994) and
vegetative filter strips (Parajuli et al., 2008; Park et al., 2011; Rudra et al., 2010; Sebti and Rudra, 2010).

WASCoBs are defined as “an earth embankment or a combination of ridge and channel constructed across the slope of minor water-courses” (USDA-NRCS, 2018). The drainage outlet of the WASCoB is referred to as a surface inlet (Li et al., 2017). Further, surface inlets can be categorized as perforated pipe risers, and blind inlets (Li et al., 2017). Whereas a perforated pipe riser is a hollow cylindrical pipe which has open holes around its circumference, a blind inlet is a fixed area of permeable cross-section filled with grading gravel, sand, and limestone.

Agricultural drainage ditches are a manually constructed watercourse employed to commute water (like direct runoff and tile-drainage fed effluents) from agricultural fields to receiving ditches or streams (Ahiablame et al., 2010; Smith and Pappas, 2007). Based upon their construction scheme and location, drainage ditches could also be classified into “road-side ditches” if constructed along the periphery of roads.

Vegetative filter strips (VFS) is a another commonly adopted BMP (Inamdar et al., 2001; Parajuli et al., 2008; Park et al., 1994; SWCS, 2001) comprising of an area of planted vegetation, designed to improve surface runoff quality (Lobo and Bonilla, 2017).

Proper assessment of BMPs upon water resources could be made either through field experimentation or using sophisticated watershed models. Although more realistic in its data procurement, field experimentation is an expensive, onerous and time-consuming process (Dardashti, 2010). On the contrary, watershed models can be employed to simulate the hydrologic or hydraulic processes with reduced costs, and are also much
less time consuming (Rong, 2009). Therefore, empirical and process-based hydrological models operating at a watershed scale are actively being employed for watershed management and planning (Daggupati, 2012).

These BMPs are therefore becoming an integral part of their watershed planning system. Hence, their importance cannot be ignored during hydrologic modeling. However, research pertaining to the impact of the above mentioned BMPs upon water quality and quantity using sophisticated hydrologic models is understudied. Hydrologic and environmental conditions are an important aspect to consider when selecting a model. A plethora of hydrologic models can be used for simulating hydrological processes pertaining to a catchment or watershed. [HSPF (Bicknell et al., 1996); MIKE SHE (Refsgaard and Storm., 1995); ANSWERS (Beasley et al., 1980); AnnAGNPS (Bingner and Theurer, 2005); WARMF (Chen et al., 1998); APEX (Williams et al., 2008); SWAT (J G Arnold et al., 1998)] are a few extensively used models by researchers and conservation authorities. Each model has its own strengths and weaknesses.

Further hydrology plays a significant role in the amount of runoff generated, ultimately impacting sediments transported along with it. Also since rainfall events change from season to season, an event-based modeling approach would be better than a continuous modeling approach for simulating peak flows and sediment loads. AGNPS is one such event based model operating at the watershed scale employed widely for NPS pollution estimation (Cho et al., 2008; Liu et al., 2008; Miklanek et al., 2004; Mohammed et al., 2004; Parajuli et al., 2007; Sebti and Rudra, 2010). Given the robustness of the model, there has been an increasing demand for developing a modeling approach through which WASCoB could be simulated using the AGNPS model. Nonetheless,
AGNPS is an event-based model with no subsurface flow component thus making it unsuitable to simulate blind inlets. Few studies have investigated the impact of WASCoBs [specially with surface inlets (pipe risers and blind inlets)], road-side ditches and vegetative filter strips on the transport of flow and sediment through the use of watershed models, specially in Canada.

Henceforth, in this study these four knowledge gaps are addressed. In this present study, a modeling approach is developed to simulate WASCoBs through the AGNPS model. A toolbox named CoBAGNPS is developed to accomplish this modeling approach. The fully integrated toolbox, CoBAGNPS is an effective decision-making tool for simulating WASCoBs along with its surface inlets (pipe risers and blind inlets). Further, in this research, a modeling exercise was executed to investigated the impact of road-side ditches upon downstream flow and sediment loads. The final research gap was addressed to simulate the effect of vegetative filter strips constructed along edge of fields upon sediment loads. Further these enhanced modeling techniques could be employed to regions having paucity of monitored data.
1.1 Objective

In this study four knowledge gaps are addressed. This will aid policy makers and stakeholders such as conservation authorities in developing and implementing an economically more viable plans for maximizing environmental benefits.

1. Develop a modeling approach to simulate WASCoBs at a watershed scale: WASCoBs are an important BMP. Further the type of surface inlet employed in the WASCoB also plays a pivotal role in the transport of water and pollutants downstream of the watershed.

2. Develop a modeling approach to investigated the impact of road-side ditches upon downstream flow and sediment loads: Drainage ditches could be a potential source for the transport of water and sediments form agricultural fields to downstream streams and creeks. Therefore, knowledge about the impact of drainage ditches upon flow and sediment transport (specially there vegetative lining) is required for maximum environmental benefit.

3. Develop a modeling approach to investigate the impact of edge of the fields vegetative filter strips upon sediment loads: Vegetative filter strips are an important BMP used extensively for abating sediment loads in a watershed. Therefore, knowledge about the impact of in field vegetative filter strips upon the transport of sediment is necessary for environmental benefit.

Based on the above discussion, great improvement towards abatement of NPS pollution could be attained by adequately adopting the above mentioned BMPs. Since, field investigation is expensive and cumbersome, development of a modeling approach capable of simulating the impact of these BMPs at a watershed scale is necessary. Thane
main objective of this study are to develop a proper modeling system capable of simulating WASCoBs, drainage ditches, and edge of field VFS at a watershed scale.

Specific objectives are:

2. Simulation the impact of different surface inlets (pipe risers and blind inlets) through the toolbox developed.
4. Modeling the effect of rural roadside ditches upon flow and sediment using KINEROS 2.
5. Model the impact of VFS applied along the edge of fields at a watershed scale.

This study was conducted in a 2611.52-acre watershed located in eastern Ontario, Canada. This watershed lies in a cold and humid region of eastern Canada and drains into Lake Huron. This watershed is fully tile drained and with 70 % agricultural land.

1.2 Thesis Organization

This thesis is written in the form of seven chapters. Chapter one, and two include the Introduction and Literature review section for the thesis. Further, chapter three, four, five and six are written as separate papers in a peer-reviewed article format. These individual papers elaborate upon different aspects of the research and include a section for introduction, methodology, results and discussion, and conclusion.
Chapter 1 - Introduction

This thesis begins with chapter 1 which includes introduction of the problem, research objectives, and thesis organization.

Chapter 2 - Literature Review

In this chapter, a detailed literature review of past research, pertaining to best management practices (BMPs) and its application through hydrologic models is discussed.

Chapter 3 - CoBAGNPS: A toolbox for Simulating Water and Sediment Control Basin, WASCoB through AGNPS model

In this chapter, a toolbox is developed to simulate WASCoBs through an event-based model AGNPS. Also, the application of the toolbox is tested for a rural watershed. Further, it addresses several scenarios, where the impact of different type of pipe risers and drainage pipes upon routing of flow through WASCoBs is analyzed. This paper has been submitted to the Catena which is a peer-reviewed journal.


Chapter 4 - CoBAGNPS: A Toolbox to Estimate Sediment Removal Efficiency of WASCoBs–Pipe Risers and Blind Inlets

This chapter discusses the sediment routing module of the toolbox developed. This module of the toolbox is tested for a calibrated and validated watershed. Also, a comparison of the sediment routing efficiency for two different types of surface inlets used
by WASCoBs, pipe risers and blind inlets is analyzed. This paper has been published in the peer-reviewed Journal of Environment and Natural Resources Research.


**Chapter 5- Predicting the Impact of Drainage Ditches upon Hydrology and Sediment Loads Using KINEROS 2 Model: A Case Study in Ontario**

This chapter investigates the impact of different drainage network/drainage patterns (DNDP) on a rural watershed using the KINEROS 2 model. Event-based modeling approach is used to estimate the impact of different DNDP upon downstream flow and sediment yield. This paper has been published in the peer-reviewed Journal of Canadian Biosystems Engineering.


**Chapter 6- A modeling approach for evaluating watershed-scale water quality benefits of vegetative filter strip - A Case Study in Ontario**

This chapter develops a modeling framework to investigate the impact of edge of the field vegetative filter strips (VFS) at a watershed scale. It identifies the critical areas
within the watershed where VFS could be located to achieve maximum sediment removal efficiency. This paper has been submitted for publication in the peer reviewed Journal of Applied Engineering in Agriculture.


**Chapter 7 - Conclusion**

This final summarizes up the thesis by describing overall contributions made by this research and provides future recommendations pertaining to this research.
Transition to chapter 2

This chapter includes a detailed literature review of past research pertaining to water quality problems, their sources, and their possible solutions. Further special emphasis has been made to discuss the application of best management practices (BMPs) to mitigate water quality issues through the use of hydrologic models.
Chapter 2

2 Review of literature

A detailed literature review was structured based on the objectives of this study. Specific emphasis was on the following four research areas:

1. Importance of freshwater and water quality.
2. The significance of non-point source (NPS) pollution in water quality impairment.
3. Best management practices (BMP) to mitigate non-point source (NPS) pollution.
4. Review of appropriate hydrologic models.

2.1 Hydrologic Cycle

Figure 2.1 illustrates the essential hydrological processes pertaining to the hydrologic cycle. The hydrologic cycle comprises of 6 fundamental constituents: precipitation (P), evapotranspiration (ET), infiltration (I), groundwater flow (G) and surface runoff (R) (Lewis and Viessman, 2003). Water balance is the basic principle of the hydrologic cycle and is independent of the extent of the study (Gollamudi, 2006). Evaporation from water bodies and transpiration from plants escalates the moisture content in the atmosphere. Consequently, water vapor condenses and forms clouds, which finally occurs as precipitation: snow, dew or rain upon the earth’s surface (Golmohamadi, 2014).
Subsequently, upon descending onto the earth’s surface, precipitation: a) is intercepted by the canopy of plants and trees, b) infiltrates and possibly stored into the soil profile, c) appears as surface runoff and subsurface lateral flow into the streams and rivers, and d) percolates into deep aquifers (Linsley et al., 1982). Surface runoff is triggered when the rate of precipitation descending upon the ground exceeds the infiltration capacity of the soil. Further, spring snowmelt along with rainfall have been established as major nutrient transport mechanisms from agricultural farmlands (Husk et al., 2017; Jamieson, 2001; Ren et al., 2016). Considering that hydrological cycle plays a significant part in the transport of pollutants, an explicit computation and assessment of water flows are paramount for estimating the magnitude of the pollutant loads from various sources (Golmohamadi, 2014).

![Graphic depiction of the land phase of the hydrologic cycle](Source: Neitsch et al., 2005)

Figure 2. 1: Graphic depiction of the land phase of the hydrologic cycle

(Source: Neitsch et al., 2005)
2.2 Fresh Water

Distinct activities pertaining to human life: washing, drinking, growing crops, cooking, and personal hygiene along with the existence of almost all categories of aquatic and terrestrial ecosystems living on planet earth are supported by freshwater (Millennium Assessment, 2005). However, nearly 96.5% of water found on earth is saline encompassed within the oceans, while 2.5%, 0.07%, and only 0.93% of the remaining water is fresh water, saline water in lakes and saline groundwater, respectively (USGS, 2016). Even with such a minuscule proportion, the quantity of fresh water is still greater than the water required for agricultural, industrial, and household purposes combined (Srebotnjak et al., 2012). Further uneven distribution of fresh water around the world (UNEP-FI and S.I.W.I, 2005) along with rapid growth of populations in certain regions of the world has resulted in an ever-increasing demand for fresh water (Morrison and Gleick, 2004).

Henceforth it could be asserted that although resources for fresh water are fixed, its demand is constantly escalating. For example, 64 billion cubic meters of additional fresh water is needed to accommodate an increase of nearly 80 million in the world’s population every year (UN-Water, 2013). Consequently, managing freshwater resources for ever-increasing population and also maintaining water quality standards for aquatic life is the biggest challenge encountered by researchers, policymakers, and politicians during the 21st century (Postel, 2000).

2.3 Surface Water

Rivers, Lakes, streams, and wetlands constitute surface waters. Surface water quality plays a decisive part in the flourishing of urban and suburban settlements
Surface water provides many benefits: it ensures drinking water, water for irrigating arable land, food for survival, and a comfortable environment for sports and recreation. Control of surface water quality from the aspect of sustainable development has been actively executed in Serbia and many other countries following the announcement of the objectives of the "sustainable policy in the water area" by the Council of the European Union and the European Parliament in 1996 (Djekovic et al., 2016). Henceforth, it is crucial to preserve both the water quantity/quality of surface water resources.

2.4 Water Quality

Deterioration of surface water quality is predominantly an outcome of the escalated concentration of pollutants such as nutrients, undesired rise in water temperature, and excessive sediment loads transported into aquatic environments (Allinger and Reavie, 2013; Conley, 2012; Yáñez-Arancibia et al., 2013). Agriculture and urbanization (Schindler et al., 2012), wildfire and climate change (Tang et al., 2013; Hoque et al., 2016), emerging pollutants (Simazaki et al., 2015), waterborne parasites (Richard et al., 2016) and disinfection by-product (DBP) formation (Goslan et al., 2009) could adversely impact drinking water security, thereby increasing treatment costs and adversely impacting human health. Expansion in agriculture specifically to meet growing food demand has resulted in increased use of pesticides, fertilizers and excessive land use changes thereby enhancing the degradation of water quality of rivers and streams (Taniwaki et al., 2017). A waterbody is identified as impaired if it is incapable of being utilized for certain purposes; for instance, drinking, recreation, fishing, navigation, and wildlife (USEPA, 2009). Appropriate identification of sources of water quality pollution is
necessary to implement adequate strategies for enhancing water quality (Srebotnjak et al., 2012). A plethora of water quality standards could be utilized for assessing water quality (Moermond and Smit, 2016; Tango and Batiuk, 2013; Zhao et al., 2018; Su et al., 2017). Further, physical and chemical parameters could be utilized to specify a threshold of a required parameter for pollutants; whereas biological parameters could be employed for illustrating possible community attributes (USEPA, 2012).

To maintain water quality specifications in Canada, the Canadian Environmental Protection Act 1999 (CEPA) and the Fisheries Act 1985 (Poulton, 2016) were introduced to regulate pollutant discharges into waterbodies. Many studies pertaining to water quality have been carried out; explicitly how to preserve water quality by either establishing certain water quality indexes, tracing the origin of pollution, and adopting appropriate BMPs (Asadzadeh et al., 2015; Espejo et al., 2012; Kovacsa et al., 2012; Liu et al., 2016; Zhoua et al., 2012). Taniwaki et al., (2017) explored the effect of land-use change upon downstream water quality. Their study revealed that changing pasture to sugarcane fields has a detrimental impact upon the water quality of a small tropical stream, indicated by high concentrations of nitrate and suspended solids. Espejo et al., (2012) concluded that certain critical areas of their study watersheds contributed relatively high loads of phosphorus to the draining stream. These loads could be controlled by implementing appropriate BMPs in these critical areas. Liu et al., (2016) evaluated the potential impact of certain BMPs using the SWAT model. Their study revealed nutrient management and wetland restoration to be most efficient BMPs in minimizing nutrient loads at the watershed outlet.
2.5 Sources of Water Pollution

Water quality degradation is a severe problem throughout the world (Taniwaki et al., 2017). Consequently, regulating water pollution is vital for improving water quality. Henceforth, an adequate identification of the origin of water pollution is necessary. Water pollution is primarily initiated due to point and nonpoint source pollutants. Whereas point-source pollutants are simple to regulate since they are discharged from an identifiable source (Huang and Xiang, 2015), it is challenging to detect and control nonpoint-source pollutants considering its complicated diffuse processes (Li et al., 2015; Liu et al., 2016; Zhang et al., 2016).

2.5.1 Point Source

Point-source pollution is a threat to regional watercourses. For example, it is contamination that discharges directly into a river from a centralized sewage treatment plant and wastewater discharges from industries (Yao et al., 2015). Therefore, point-source pollution could be monitored and guarded systematically at a single location (Carpenter et al., 1998). The discharged pollutants have the potential to impair aquatic ecosystems, and eventually influence regional human health and socioeconomic status. Discharge of point-source pollution into water resources is regulated by government agencies.

Sources of point-source pollution are generally easy to identify and control compared to nonpoint sources. In Ontario, Canada, the “Ontario Environmental Protection Act” prohibits the discharge of pollutants into the waters of the province. Further, it is a requisite requirement for industries and sewage plants discharging point-source pollutants into the environment to apply for “Environmental Compliance Approvals
(permits)” within Ontario; severe penalties are imposed if these approvals are breached (Ontario Ministry of the Environment, 2012).

2.5.2 Non-Point source (NPS)

During the latter period of the 20th century, since NPS pollution has been identified as a dominant threat to water resources, it has drawn the concern and attention from both research organizations and law enforcement agencies (Wang et al., 2012). The growing significance about the origin of NPS pollution, for example, the pollution of lakes in watersheds by means of agricultural activities, has stimulated new theoretical and empirical studies intended at designing relevant instruments for dealing with such problems (Camacho-Cuena and Requate, 2012). Henceforth, numerous programs for regulating water quality were established [e.g., Clean Water Act and Total Maximum Daily Loads (TMDL)] (Houck, 2002). NPS pollution commences from precipitation, atmospheric deposition, infiltration, groundwater flow, runoff and drainage primarily under agriculture (Shi et al., 2012). Further research has established that factors such as rainfall intensity, soil characteristics, topography, and land use significantly impact the magnitude and quality of NPS pollution (Wang et al., 2012). Problems concerning NPS pollution are characterized by the dominant feature; information for which can only be gathered by pollution control authorities (Giri, 2013). This might be since it is technically impractical to monitor the emissions of each individual polluter, or because monitoring those emissions is excessively expensive and cumbersome (Camacho-Cuena and Requate, 2012). Due to spatial variation of NPS pollutants in a watershed (Shen et al., 2015), pollutants from a certain area in the landscape, identified as critical source area (CSAs), generally contribute major portion of the pollution load in the watershed and therefore have a
definitive impact upon receiving water quality (Ou and Wang, 2008). Surface runoff is the dominant mechanism for the transport of pollutants within a watershed. For example, when surface runoff moves over the land surface, pollutants such as sediments, organic matter, nutrients, oil, bacteria, and metals are commuted along with it to the accepting waterbodies.

In Ontario, Canada, NPS pollution is identified as a source for polluting the Great Lakes (Allinger and Reavie, 2013). Escalated use of fertilizers and pesticides to enhance crop productivity for an ever-increasing population are bound to have serious repercussions pertaining to NPS pollution. Henceforth, parity amidst crop productivity and the detrimental impact of NPS pollutants (associated with enhanced crop production) upon water quality should be maintained. Therefore appropriate understanding regarding the fate of NPS pollutants is required (Wali et al., 2011). In Canada, there are several federal legislations which regulate the discharge of pollutants into the environment. Noticeably, the Canadian Environmental Protection Act 1999 (CEPA) and the Fisheries Act 1985 (Poulton, 2016) are important in taking initiatives to implement appropriate BMPs for NPS pollution abatement (Benidickson, 2009).

2.6 Water Quality Impairment due to NPS pollutants

2.6.1 Impairment due to Sediments

Sediments are a significant source for polluting rivers and streams worldwide (Pandey et al., 2007). Adverse impact of sediments upon aquatic ecosystems includes higher water temperature, reduction in spawning areas for many fish species, decline in habitat for macroinvertebrates, etc. (Robertson et al., 2006). Furthermore, excessive sediments escalate turbidity and movement of toxic materials, including fertilizer and
pesticide residues (bound to clay and silt particles) in water bodies (Tegtmeier and Duffy, 2004). Sediment contributes 88% and 86% of total nitrogen and phosphorus respectively to the national waterbodies in the U.S (Holmes, 1988). Moreover, sediments could constantly discharge pollutants into the water column, thereby increasing the recovery time of eutrophic environments (Søndergaard et al., 2007). Resistance levels for soil erosion differ vastly throughout the world. The mean tolerance level for soil erosion is estimated to be 6 tons/hectare/year or less in Canada (Wall et al., 2002). Further, excessive sediment also reduces the storage capacity of reservoirs. Huge economic losses have been reported due to sediments transported through waterways (Tegtmeier and Duffy, 2004).

In the US, external cost of approximately $13.4 billion is due to excessive sedimentation (Tegtmeier and Duffy, 2004). In addition, suspended sediment increases turbidity and reduces photosynthesis thereby affecting the aquatic life in degraded water bodies (Robertson et al., 2006). Further, Bosch et al. (2011) investigated sediment and nutrient loads along with discharge in the Lake Erie (US) watershed using the SWAT model. Researchers concluded SWAT to be reliable in modeling discharges and sediments. However, nutrient simulations were not as accurate. Alternative studies have reported excessive sediment and nutrients transported into Lake Erie to cost the federal government $143 million for its improvement and management (Forster and Rausch, 2002; Richards et al., 2002).

2.6.2 Impairment due to Nutrients

Excessive nutrients in surface waters is a dominant water quality problem, concerning numerous local, regional, and national entities around the world. For over a
decade, excess nutrients have been rated amongst the top five elements for impairment of watercourses in the U.S (USEPA, 2008). Decrease in the oxygen level of coastal, marine, and freshwater bodies continues to increase globally both in magnitude and frequency; thereby arousing concern on the detrimental impact upon tourism, fisheries, and ecosystems in Gulf of Mexico (Yáñez-Arancibia et al., 2013), Baltic Sea (Conley, 2012), Black Sea (Diaz, 2001), Chesapeake Bay (Boesch, 2006), China Sea (Chen et al., 2007), the Great Lakes (Allinger and Reavie, 2013), Lake Winnipeg (Schindler et al., 2012), and throughout the United Kingdom and Europe (Haygarth et al., 1999; Heathwaite and Dils, 2000).

2.6.2.1 Phosphorous

Eutrophication is a serious environmental concern in freshwater ecosystems worldwide (Schindler et al., 2012). Phosphorous is the most important nutrient since it is a constraint for the growth of algae in freshwater and lakes (Carpenter, 2008; Herath, 1997; Lee, 1973), thereby impairing water quality in a plethora of ways: fish kill, growth in harmful algal blooms, hypoxia (low dissolved oxygen), reduction in wildlife diversity, and degradation in habitat (Dodds, 2006). Several studies have investigated the budgets of P in lakes and reservoirs (Boggess et al., 1995; Hiscock et al., 2003; Le et al. 2014).

In the United States, authorities at the state level regulate the water quality standards regarding P. However, only three states: Minnesota, Illinois, and Wisconsin have established criteria at the state level for regulating critical P concentrations for individual surface water bodies (USEPA, 2014). In Canada, a limit of 0.03 mg L−1 for total P has been set in Quebec and Ontario for protecting fresh waterbodies against eutrophication (Environment Canada, 2004). Also, a limit between 0.01 and 0.087 mg/L
for total P has been setup for explicit water bodies extending across several ecozones in Canada (Chambers et al., 2009). Several studies have rejected point-source pollution to be the major cause of phosphorus pollution in water bodies (Orderud and Vogt, 2013). Researchers have instead identified agricultural activity to be the significant cause of P pollution in freshwater ecosystems (Heathwaite et al., 2005; Lü et al., 2016; Lu and Tian, 2017; Orderud and Vogt, 2013; Withers and Haygarth, 2007).

A plethora of researchers have studied the adverse impact of phosphorous upon water quality. Poor and McDonnell, (2007) and Hoare, (1982) reported 42 % of annual phosphorous load to be exported from a catchment during a three-day storm event period. Therefore, it could be inferred that phosphorus loads are predominantly generated from non-point sources (diffuse pollution), essentially soil erosion, and is strongly correlated with the precipitation volume (Scholz 2010). Bowes et al., (2003) studied phosphorous pollution of an English plain using export coefficients. Their study revealed that domestic wastewater was associated with the greatest pollution proportion. Ren et al., (2016) reported that 80 % treatment of wastewater reduced the total P loads by approximately 52 %.

### 2.6.2.2 Nitrogen

Reactive-N causes eutrophication, prolific growth in toxic algae, hypoxia, damage in the biodiversity and degradation of animal habitat in ecosystems along the coast of seas and oceans contaminating aquatic environments and is therefore considered an essential pollutant adversely impacting riverine water-systems globally (Husk et al., 2017). Also, reactive-N contaminates drinking water, thereby having a detrimental impact upon the health of humans (Doering et al., 2011; Howarth et al., 2012; Rabalais, 2002).
In comparison to any other economic sector, agricultural activities contribute the most towards reactive-N loss to the environment, predominantly as nitrate-N (NO₃-N), since it readily dissolves with stormwater runoff and leaches into the soil profile (Husk et al., 2017). Fertilizers are used to enhance the productivity of grain and non-leguminous crops. Approximately 60% of their N input leaches into the soil rather than consumed by crops directly (Cassman et al., 2002; Conant et al., 2013; Drinkwater and Snapp, 2007).

Further, research in the past associates nitrate-nitrogen to be the major cause for hypoxic regions in coastal waters; however, recent studies suggest that P may also play an important role (Dodds, 2006). Furthermore, recreational activities, for instance, swimming, fishing, and boating are impaired as a result of excessive plant growth, aroma and taste issues pertaining to water (USEPA, 1999). Studies have also revealed that serious environmental hazards like annihilation and deformity in amphibians (Rouse et al., 1999) along with the creation of dead zones such as in the Gulf of Mexico (Ebionews, 2010) could be initiated due to the movement of nitrogen from agricultural lands to surface watercourses.

Moreover, nitrate is expected to be a hazard for human health specifically for children. Nitrate could decrease oxygen in the bloodstream causing blue baby syndrome (USEPA, 2012). As per USEPA, $200 million is needed to preserve the federal threshold of nitrate in drinking water by virtue of nitrate contamination of drinking water resources (USEPA, 2012). In Canada, the critical environmental threshold for Canadian surface waters is set to 3.0 mg NO₃-N L⁻¹ (Canadian Council of Ministers of the Environment, 2003). Previous studies reveal that agricultural activities substantially contribute to nitrate loads; however, they have little impact on phytoplankton growth (Hutchins, 2012). Husk
et al., (2017) conducted a field-scale study for three years to evaluate the denitrification capacity of bioreactors in Quebec, Canada. They found that bioreactors reduced total-nitrogen and nitrate-nitrogen median concentrations by 72 and 99 % respectively in the subsurface drainage outflows.

2.6.3 Impairment due to Pesticides

Pesticides are originally employed to enhance crop productivity; however, they may impact watercourses adversely. During the last two decades, the application of total pesticide on corn and soybean hectares has increased from 104,231 to 141,788 Mg (NASS 2015). During the same period, use of 2,4-D and glyphosate elevated by 88 and 3150 %, respectively (Gonzalez et al., 2016). More than 200 million kgs of herbicides, insecticides, and fungicides were used annually from 1992 to 2011 to escalate crop production and mitigate insect-borne disease (Stone et al., 2014). In the U.S, USEPA has set-up benchmarks for pesticide concentration below which they are considered to be safe for aquatic life (USEPA, 2015) known as “aquatic life benchmarks”. Although modern farming equipment and management practices are constructed to assist in the application of pesticides at a controlled rate and at specific locations to limit the amount of pesticides that could be transported with surface run-off, pesticide levels are still above the established criterion in streams (Gonzalez et al., 2016).

For example, in the U.S, between 2002 and 2011, approximately 61 and 90 % of rural and urban streams, had at least one pesticide above the threshold limit established for aquatic life (Stone et al., 2014). Henceforth there is an ever-increasing requirement for adequate development and employment of conservation practices. These practices
would mitigate contaminant losses from agricultural farmlands, which would otherwise pollute waterbodies and impair the environment.

2.7 Mechanism to Abate Agricultural Non-Point Source Pollution

Transportation of NPS pollutants initiated due to agricultural activities could be restrained by implementing several management exercises like BMPs and adopting alternative agricultural practices (USEPA, 2012). Subsequent section briefly reviews how BMPs could be adequately employed to abate NPS pollution.

2.8 Best Management Practices

Effective and efficient control of NPS pollution is a key issue for protecting water quality (Chen et al., 2014; Sanders et al., 2013). Therefore adopting BMPs on agricultural farmlands for improving water quality is currently a topic of increasing interest amongst researchers and conservation authorities. (Giri et al., 2014). BMPs are management practices and operations which attenuate the load of pollutants in runoff and leaching while being economically feasible for the farmers (UNEP, 1998). Nonetheless, the performance of BMPs differs from one location to another within a watershed and with the characteristic of BMP implemented (Giri et al., 2012). BMPs attenuate NPS pollution through three mechanisms: 1) reducing the mass of the pollutant by adopting source control management, 2) minimizing the transport of pollutants to surface water resources, and 3) implementing biological, chemical, or physical processes for remediation of contaminated water bodies (Cunningham, 2003). Typically structural or non-structural management practices could be used as BMPs for stormwater management (Kaplowitz and Lupi, 2012).
Structural BMPs are physical constructed operations (e.g., construction of wetlands, dry basins) designed towards minimizing the impact of runoff generated during severe storms [American Society of Civil Engineers (ASCE), 2000; Center for Watershed Protection [CWP], 2000]. Non-structural BMPs are a modification of agricultural practices through educational efforts aimed towards improving certain aspects of water quality; for example usage of less fertilizer, minimizing littering etc.(Taylor et al., 2007). Some of the commonly utilized and newly adopted BMPs reported in USDA NRCS conservation practice technical document are described in the subsequent section. Table 2.1 characterizes the type of BMP, its specification, definition, and condition for the application of BMP. In the U.S, USDA offers distinct conservation programs for promoting BMP implementation like Conservation Reservation Program, the Environmental Quality Incentive Program, and the Conservation Stewardship Program (Sommerlot et al., 2013). Similarly, in Canada, different agencies such as provincial level conservation authorities, the ministry of environment at provincial level work along with landowners for implementing agricultural BMPs in their fields to minimize NPS pollution.

The impact of a BMP in reducing pollution is site-specific (Giri et al., 2012). Further, application of BMPs is cumbersome and expensive due to opposing environmental, economic, and institutional activities (Arabi et al., 2008). Also, a single BMP could be insufficient to control certain types of pollutants (USEPA, 1999). Henceforth, adequate selection, design, and application of BMPs are necessary to estimate the impact of the BMP in the abatement of NPS pollution.
Table 2. 1: Common BMPs adopted as described in the NRCS technical booklet

<table>
<thead>
<tr>
<th>Type of BMP</th>
<th>Definition of BMP</th>
<th>Purpose of the BMP</th>
<th>The condition for BMP application</th>
<th>Specification for the BMP</th>
<th>Literature</th>
</tr>
</thead>
</table>
| Conservation Crop Rotation (NRCS Code 328) | Crops are grown in a planned sequence on the same field (USDA-NRCS, 2018)    | • Minimize sheet and rill or wind erosion.  
  • Improve soil quality.  
  • To conserve water. | • The practice could be applied to all cropland land where annually-planted crops make up at least one-third of the crop sequence (time basis).  
  • For the purposes of this practice, a cover crop is considered a crop in the rotation. | • Fallow land should not cover more than 25% of the planned crop sequence during the uncropped period (Giri, 2013).  
  • A planned two crop sequence should contain a warm and cool season crop (Giri, 2013). | (Douglas-Mankin et al., 2010; Tuppen et al., 2010) |
<p>| TERRACE (NRCS code 600)               | An earth embankment, or a combination ridge and                                  | • Minimize soil erosion by reducing slope length. | • If the area has a problem due to excessive slope, runoff.                                                          | • Capacity to control 10-                                                            | (Tuppen et al., 2010; Schwab et al., 1995; Alberts et al., 1978; |</p>
<table>
<thead>
<tr>
<th><strong>Channel</strong></th>
<th><strong>Constructing a channel across the field slope (USDA-NRCS, 2018)</strong></th>
<th><strong>Detain runoff in the field for moisture conservation</strong></th>
<th><strong>Topography and soils are suitable for terrace construction and a suitable outlet could be provided</strong></th>
<th><strong>Year 24-hour storm</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>• The ridge should have a minimum width of 3 ft.</td>
<td>• The maximum allowable slope should be 2:1 H: V.</td>
<td>• The maximum length of the terrace should be 3500 ft. for preventing potential failure.</td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td><strong>Dano and Siapno, 1992)</strong></td>
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<tr>
<td><strong>Strip Cropping</strong></td>
<td><strong>Growing row crops, forages, small grains, or fallow in a systematic arrangement of equal width strips across a field in rotation</strong></td>
<td><strong>Minimize soil erosion from water and wind</strong></td>
<td><strong>Applied on lands where crops are grown.</strong></td>
<td><strong>Strips of the crops should be placed at an angle perpendicular to the water and wind erosion forces.</strong></td>
</tr>
<tr>
<td><strong>(NRCS code 585)</strong></td>
<td></td>
<td>• Protect growing crops from damage by wind-borne soil particles</td>
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| **CONTOUR FARMING**  
**(NRCS code 330)** | **Ridges and furrows formed by farming operations such as tillage, planting etc. Thereby altering the direction of runoff from directly downslope to around the** | **Minimizes sheet and rill erosion, and reduces the transport of sediment and contaminants attached to it.** | **Applied on sloping land where annual crops are grown.** | **At least 50% of the cover consists of erosion resistant crops or sediment trapping cover**  
**Strip boundary should be parallel to each other and in proximity to the contour**  
**Minimum ridge height of two inches should be maintained during the rotation period for row spacing greater than 10 inches**  
**Minimum ridge height (Panagopoulos et al., 2011; Sahu and Gu, 2009).** |
<table>
<thead>
<tr>
<th>CONSERVATION COVER (NRCS Code 327)</th>
<th>Implementing and preserving permanent vegetative cover (USDA-NRCS, 2018)</th>
<th>Reduce soil erosion and sedimentation.</th>
<th>Applied on all type of lands requiring a permanent vegetative cover.</th>
<th>Perennial crop vegetation should provide full ground coverage in the pathway during mowing and harvesting. Combination of grasses, forbs, legumes, and (Rittenburg et al., 2015; Kaspar et al., 2001; Zhu et al., 1989).</th>
</tr>
</thead>
<tbody>
<tr>
<td>hillslope (USDA-NRCS, 2018).</td>
<td>of one inch should be maintained for close-grown crops having a row spacing of maximum 10 inches.</td>
<td>The maximum limit for row grade is 0.2%</td>
<td></td>
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</tbody>
</table>
| CONSTRUCTED WETLAND (NRCS Code 656) | Artificially constructed wetland with hydrophytic vegetation for biological treatment of water (USDA-NRCS, 2018) | • Treatment of wastewater or contaminated runoff from agricultural processing, livestock, or aquaculture facilities  
• Improving the water quality of stormwater runoff. | • For an agricultural wastewater management system where the constructed wetland is a component.  
• Construction of an auxiliary spillway or inlet bypass to control peak flow of 25-year, 24-hour storm.  
• Provide a suitable inlet control structures to avoid entering debris into the wetland. | shrubs shall be planted to promote biodiversity (Rittenburg et al., 2015; Lee et al., 2009; Brix, 1994). |
| RESIDUE AND TILLAGE MANAGEMENT, NO TILL (NRCS Code 329) | Minimizes disturbance in the soil to manage the amount, orientation and • Reduce sheet, rill and wind erosion along with tillage-induced | • This practice applies to all cropland.  
• Soil tillage intensity rating value should be within 20 | (Rittenburg et al., 2015; Ghidey et al., 2005) |
| FILTER STRIP (NRCS Code 393) | A strip of herbaceous vegetation that removes contaminants from overland runoff (USDA-NRCS, 2018) | • Trap suspended solids and dissolved contaminants in runoff.  
• Reduce suspended solids and associated contaminants in irrigation water and excessive sediment in surface waters. | • Established where environmentally sensitive areas need to be protected from sediment, suspended solids, and dissolved contaminants in runoff.  
• Minimum width of the filter strip should be 20 feet  
• The slope of one percent or greater is preferable for the area upstream of filter strip.  
• Maximum of four inches plant spacing (Muñoz-Carpena et al., 2010; Sebti and Rudra, 2010; Shan et al., 2014). | • Crops having row spacing less than 15 inches should have a minimum 10 inches crop stubble height.  
• One to three inches deep soil disturbance is preferable to release less CO₂.  
• Enhances plant-available moisture.  
• Crops having row spacing less than 15 inches should have a minimum 10 inches crop stubble height.  
• One to three inches deep soil disturbance is preferable to release less CO₂.  
• Enhances plant-available moisture. |
| **GRASSED WATERWAY** (NRCS Code 412) | The graded channel which is constructed with suitable vegetation to convey surface water at a non-erosive velocity using a broad and shallow cross-section to a stable outlet (USDA-NRCS, 2018). | • Convey runoff from terraces, diversions, or other water concentrations without causing erosion or flooding.  
• Prevent gully formation and improve water quality. | • This BMP is applied in areas where added water conveyance capacity and vegetative protection is required to control erosion and improve the water quality of surface runoff resulting from concentrated flow. | is preferable in the filter strip  
• Can control the peak runoff from 10-year 24-hour rainfall.  
• Bottom width of the grass waterway should be less than 100 feet.  
• Slide slope should be less than 0.5.  
• To avoid damage a freeboard should be provided above the designed depth. |
<p>| <strong>SEDIMENT BASIN</strong> (NRCS Code 350) | Artificially constructed basin with an outlet, formed by an embankment, | • Detain sediment in runoff, or other debris for a sufficient period to allow it to | • Constructed in urban areas, agricultural land, | Basin can at least store 3600 ft 3/acre |</p>
<table>
<thead>
<tr>
<th>Drainage Management Practice</th>
<th>Riparian Forest Buffer (NRCS Code 391)</th>
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<tbody>
<tr>
<td></td>
<td>excavation or a combination of the two (USDA-NRCS, 2018).</td>
<td>settle out in the basin.</td>
<td>and construction sites.</td>
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<td></td>
<td></td>
<td></td>
<td>of drainage area.</td>
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<td></td>
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<td>• Length to width ratio of the basin should be 2 to 1 or greater.</td>
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<td>• Porous baffles in the entire basin should be constructed to control the turbulence within the basin.</td>
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<td></td>
<td>• Sheet flow is preferred through the riparian buffer (Rittenburg et al., 2015)</td>
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<td></td>
<td>Area essentially comprising of trees and/or shrubs located adjacent and upslope from water bodies (USDA-NRCS, 2018).</td>
<td>• Improve riparian aquatic life by controlling water temperature.</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>• Attenuate sediment, organic material, nutrients and pesticides in surface runoff.</td>
<td></td>
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<tr>
<td></td>
<td></td>
<td>• Reduce pesticide drift penetrating</td>
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<tr>
<td></td>
<td></td>
<td>• Constructed in forested areas adjacent to streams, creeks, ponds, and wetlands.</td>
<td></td>
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<tr>
<td></td>
<td></td>
<td>• Minimum width of the riparian forest buffer should be 35 ft.</td>
<td></td>
</tr>
</tbody>
</table>
| WATER AND SEDIMENT CONTROL BASIN (NRCS Code 638) | “An earth embankment or a combination ridge and channel constructed across the slope of minor watercourses to form a sediment trap and water detention basin with a stable outlet” (USDA-NRCS, 2018) | • Trap sediment  
• Reduce watercourse, gully erosion, onsite and downstream runoff  
• Topography is largely irregular and gully erosion is an issue  
• Runoff and sediment damages land.  
• Appropriate outlets could be provided.  
• Installed as part of a conservation system that adequately addresses issues pertaining to both above and below the basin.  
• Installed in areas where treatment of the upper portion of a slope is not possible. Hence, WASCoBs are employed to separate this area from and permit treatment of | • Native and non-invasive trees and shrub species are in the riparian buffer  
(Her et al., 2017; Kovacic et al., 2006; Yang et al., 2013) |
|---|---|---|---|
| into water bodies.  
Enhance carbon storage | | | |
| UNDERGROUND OUTLET (NRCS Code 620) | A conduit or system of conduits installed beneath the surface of the ground to convey surface water to a suitable outlet (USDA-NRCS, 2018). | • Route water from diversions, waterways, subsurface drains to a suitable outlet without causing damage by erosion or flooding. | • Removing excess runoff is required to minimize ponding.  
• A buried outlet is needed for subsurface drain or similar practices.  
• A surface outlet is considered impractical due to site-specific conditions. | • The design capacity of the underground outlet is based on structure requirements.  
• The underground outlet can be designed to function as the principal outlet for the structure.  
• The capacity of the underground outlet for constructed basins shall provide adequate drainage for the intended purpose. | (Kovacic et al., 2006; Her et al., 2017; Li et al., 2017) |
| SURFACE DRAIN, FIELD DITCH (NRCS Code 607) | A graded ditch for collecting excess water in a field (USDA-NRCS, 2018). | • Interception of excess subsurface water and conveyance to an outlet.  
• Collection or interception of excess surface water, such as sheet flow from natural and graded land surfaces or channel flow from furrows, and conveyance to an outlet.  
• Drainage of surface depressions. | • Have soils that are slowly permeable (low permeability) or are shallow over barriers such as rock or clay, which hold or prevent ready percolation of water to a deep stratum.  
• Have surface depressions or barriers that trap rainfall.  
• Have insufficient land slope for ready movement of runoff across the surface.  
• Receive excess runoff or seepage from uplands.  
• Require the removal of excess irrigation water.  
• Require control of the water table. | • Field ditches shall be planned as integral parts of a drainage system for the field served and shall collect and intercept water and carry it to an outlet with continuity and without ponding. | (Ahiablame et al., 2010; Ahiablame et al., 2011; Herzon and Helenius, 2008; Kleinman et al., 2007) |
A detailed literature review was performed to review a few BMPs that are relevant to this thesis. This selection identifies BMPs which have the potential to efficiently manage agricultural stormwater. The identified BMPs and their impact on hydrology and nutrient mitigation are described below.

2.8.1 WASCoBs

A water and sediment control basin (WASCoB) is installed to mitigate transport of sediment and nutrients initiated from agricultural farmlands; thereby retarding the development of gullies by interrupting a watercourse (Her et al., 2016). Berm, surface inlets, and tile drains together constitute a WASCoB. Impact of WASCoBs upon flow and nutrient transport have been evaluated by a few researchers. Her et al. (2016) used the SWAT model for evaluating the impact of several conservation practices (including WASCoBs) in mitigating sediment and nutrient loads at field and watershed scales in the St. Joseph River watershed. Simulation results demonstrated that WASCOBs increased soluble P and N loads. Her et al. (2017) studied the spatial and temporal variability in sediment and nutrient loads in an agricultural watershed in the Midwest U.S using the SWAT model. Also, nine conservation practices (including WASCoB) were implemented to compute the reduction in the nutrient amounts. The authors concluded that WASCOBs often elevated soluble nutrient amounts depending upon changes in hydrology, mixing rate of fertilizer, and cycles of nutrients in residue left on the field.

Kovacic et al. (2006) constructed experimental wetlands along with a WASCoB for intercepting surface and tile drainage upslope of Lake Bloomington for reducing pollutants before it entered the lake. The network of wetlands and WASCoB reduced the NO$_3$–N concentrations by 42% and 31% respectively. Yang et al. (2013) developed a WASCoB
module and linked it with the SWAT model to examine water quantity and quality effects of WASCoBs in the Gully Creek watershed, Ontario Canada. WASCoBs were represented as a sub-basin outlet in the module. Furthermore, the stage-storage and stage-discharge relationships for the WASCoBs were used to route water through the tile drain. Further surface inlets associated with the drainage system of the WASCoBs are broadly classified into four main types based upon their function and dimensionality viz., perforated pipe risers, open inlets, rock inlets, and blind inlets (Li et al., 2017). A few “surface inlets” used in this research, and their hydrologic impact is described in the sections below.

2.8.2 Surface Inlets

Draining farmlands artificially is gaining widespread momentum in humid regions (Goswami et al., 2008; Vlotman et al., 2007); thereby altering the movement of water and nutrients to alternative waterways (Dinnes et al., 2002). Tile drains are commonly adopted for artificial irrigation and drainage purposes. Tile drains escalate infiltration, thereby minimizing overland routing of surface runoff, simultaneously creating an alternative flow path for nutrient-enriched water, which generally drains directly into surface water bodies. However, impact of installing tile drains upon the movement and fate of N is questionable and not well documented. In certain regions surface inlets are connected to tile drains to boost the drainage efficiency of regions with flat topography. Henceforth, with the aid of these surface inlets, overland flow could enter the drainage system directly with or without filtering by soil or buffers. Hickenbottom and catch basins are common types of surface inlets employed. Also, blind inlets (Bi), commonly known as “French drains”, have widely been adopted (Wright and Sands, 2001), since they are more suitable than other types
of inlets while managing field operations, thereby providing a better substitute to surface inlets which are open at the surface.

### 2.8.2.1 Pipe Risers

Pipe risers (also referred to as tile risers) are the most widely adopted surface inlets employed for drainage systems (King et al., 2015; Li et al., 2017; Oolman and Wilson, 2003). Pipe risers of several dimensions are used for routing surface runoff into buried underground drains thereby affecting the water balance (usually peak discharge) and sediment nutrient loads at the outlet tile drain network (Figure 2.2). Few researchers have studied the impact of different types of surface inlets upon flow and sediment characteristics. King et al. (2015) severely reviewed research pertaining to P transport through subsurface drainage systems. Several factors affecting P transport such as type of surface inlets were analyzed in their review. The authors asserted that transport of pollutants increased with the use of traditional type of surface inlet.

Li et al. (2017) studied the flow and sediment transport characteristics of four surface inlets—QuickDrain low profile inlet (QD), AgriDrain wick inlet (AD), Hickenbottom standard inlet (HB), and Ag-Solutions standard inlet (AS)—under laboratory and field settings using a rainfall simulator. Results from the study revealed that average sediment concentration and load for HB and QD inlets were higher among the four inlets with values of 3532.70 mg/L, 64919.05 mg/min, and 3104.31 mg/L, 24880.69 mg/min respectively. Further, AD inlet was the most effective surface inlet in reducing sediment [66% concentration and 23.2% load compared to (HB)] (p<0.01); however, it removed water at a much slower rate.
Ball Coelho et al., (2012a) in Ontario, Canada studied the impact of subsurface drainage, along with the type of surface inlet (Hb, Bi) and management practice (tillage and manure application method), upon N loads in surface water measured both for tile drains and overland runoff monitored for a one-year period. Results revealed surface inlets to have a minimal impact upon N movement through the drainage system. In an additional study conducted in Ontario, Canada (Ball Coelho et al., 2012b) explored the impact of subsurface drainage (BI or HB), and a few management practice like tillage and manure application method upon partitioning of P and sediment to surface water. Authors reported that combination or permutation of surface inlet had a negligible impact.
on P and sediment loading. Feyereisen et al. (2015) determined the concentrations of total suspended sediment (TSS) and P when pipe risers are replaced with blind inlets. Results demonstrated that blind inlets were more effective in minimizing sediment and P loads compared to surface inlets. Tomer et al. (2010) estimated the kinetics of nitrate-N, total P, Escherichia coli, and sediment during a single event in Tipton Creek, Iowa. Approximately 13% of the total flow generated at the watershed outlet originated due to discharge from tile risers. Further tile risers delivered nearly 50% of the total P load and 33.33% of the E. coli load observed at the watershed outlet.

Ginting et al., (2000) evaluated the characteristics of surface runoff along with pollutants lost through surface tile inlets draining natural depressions of lacustrine landscapes. Results indicate that surface inlet conveyed less than 5% of annual precipitation into the tile drains. Also, loss of particulate pollutants through surface tile inlets were less than expected with 138 kg/ha total solids (TS), 20 kg/ha chemical oxygen demand (COD), 363 g/kg TP, 205 g/ha dissolved molybdate reactive P (DMRP), 1342 g/ha total dissolved inorganic N (TDIN), and 1126 g/ha nitrate N lost between 1995 to 1998 in a Southern Minnesota River watershed.

2.8.2.2 Blind Inlets

Pipe risers are a regular and effective practice for draining potholes (Smith and Livingston, 2013). However, pipe risers are not very effective in mitigating the transport of sediment and contaminants from the depression area to the draining watercourses. Henceforth; blind inlets (Figure 2.3) which is a relatively modern and state-of-art conservation practice, have demonstrated significant reduction in sediment and nutrient losses from potholes (Feyereisen et al., 2015; Smith and Livingston, 2013; Smith et al.,
They could be an alternative to pipe riser for routing surface water ponded within a depression area into the underground drainage system. Results from studies conducted between 2013 and 2015 have reported that blind inlets are more competent in alleviating sediment and nutrient losses from potholes to drained watercourses than pipe risers (Smith and Livingston 2013; Feyereisen et al. 2015; Smith et al. 2015).

![Blind Inlet Diagram](image)

**Figure 2.3:** Blind inlet consisting of sand, limestone, and geotextile connected to a subsurface drain

Further blind inlets could be designed and installed conveniently. Also, compared with pipe risers, farm equipment can pass directly over them without disturbing daily farming practices (Li et al., 2017). A blind inlet consists of two layers of sand and limestone gravel excavated at the lowest elevation point of the WASCoB. Detailed characteristics and installation dynamics of a blind inlet considered for this research have been described in Smith and Livingston, (2013). Although potentially more effective in
reducing sediment than pipe risers, blind inlets require more maintenance than a pipe riser. Further, sediment trapped needs to be ejected frequently from the system culminating in higher labor cost. Hence, the lifespan of a blind inlet is uncertain depending upon several factors including the composition of the inlet (gravel/sand and limestone), type of soil, area of the catchment drained, and farm management practices (Li et al., 2017). Previous studies pertaining to blind inlets have reported them to be operational for more than 10 years with minimal complications (Feyereisen et al., 2015). However, due to difficulty in monitoring runoff and sediment losses through blind inlets, there is a paucity of published research. Smith et al. (2015) reported that there was no significant difference in median phosphorus concentration between tile riser (0.37 mg/L) and blind inlet (0.30 mg/L) ($p = 0.59$) based on a ten-year study in northeast Indiana.

Feyereisen et al. (2015) conducted a study to analyze whether blind inlets could minimize suspended sediment and P loads in drainage effluent relative to tile risers or open inlets. Results demonstrated that TSS, total P and soluble reactive P loads were 64.35, 66 and 50% less for the blind inlets, relative to open inlets. Gonzalez et al. (2016) investigated the effectiveness of blind inlets relative to tile risers during a 6-year study in the midwest US. Autosamplers were used to collect data followed by analyses for several pesticides. Results from the study revealed blind inlet to be more effective in mitigating pesticide losses i.e., atrazine (57 %), 2,4-D (58 %), metolachlor (53 %), and glyphosate (11 %) respectively. McKague (2017) assessed that blind inlet constructed with a layer of red sand was approximately 70% effective at removing the total phosphorus that passed through it and about 50% effective at removing the soluble reactive phosphorus that passed through it.
2.8.3 Agricultural Drains/Road-Side Ditches

Hydrology and water quality of a watershed is greatly impacted by effluents from agricultural runoff and tile drains (Sloan et al., 2017; Moriasi et al., 2012) which are transported to the watershed outlet through a network of agricultural drains. It is therefore pivotal to appropriately understand the water routing mechanism within a watershed. Agricultural drainage ditches are an important BMP used to transport water from agriculturally dominated areas to creeks and streams (Ahiablame et al., 2010; Smith and Pappas, 2007).

Plenty of researchers have studied nutrients transported through drainage ditches (Gentry et al., 2007; Kleinman et al., 2007; Smith et al., 2005; Strock et al., 2007). Strock et al., (2007) investigated issues related with usage of agricultural N, transport of N from artificially drained agricultural land to drainage ditches, cycling of N within ditches, and other associated management scenarios.

In another study, Soana et al., (2017) monitored nitrate removal capacity in vegetated and unvegetated ditches nourished by nitrate-rich groundwater. A significant removal of N was reduced by vegetated ditches (38–84 mmol N m\(^{-2}\) d\(^{-1}\)) compared to unvegetated ditches (12–45 mmol N m\(^{-2}\) d\(^{-1}\)). Denitrification removed the maximum amount of N, followed by plant uptake. Furthermore, Iseyem et al., (2016) investigated nutrient removal efficiency in experimental drainage ditches lined with different vegetation. Mowed and unmowed ditches reduced \(NO_3^-\) - N by 79 % and 94 % and \(PO_4^{3-}\) by 95 % and 98 %, respectively. Thereby indicating no significant difference amongst their reduction capacities. Based upon their construction and location, drainage ditches could also be classified as “road-side ditches” if directly feeding runoff effluents (tile flow...
and runoff) from agricultural fields to streams. While several studies have analyzed the impact of agricultural drains upon hydrology, the process is understudied for road-side ditches (RSD) and canals (Needelman et al., 2007; Skaggs et al., 1994). RSD could be similar in form and function to drains. However, less attention has been devoted towards understanding the impact of RSD specifically (e.g. Diaz-Robles, 2007; Falbo, 2013) perhaps due to difficulty in monitoring runoff and sediment losses through RSD there is a paucity of published research in this topic.

Falbo et al. (2013) measured the concentration of fecal indicator organisms and sediments in RSD and reported that concentrations exceeding the recommendations of US EPA and New York State (in excess of 241,960 MPN of E. coli/100 mL) were observed during a 1-year study period in central New York. Further, the authors concluded that RSD potentially harbors bacteria, which could be re-suspended and transported with storm-runoff. Buchanan et al., (2013) applied the distributed direct hydrograph model, and investigated several scenarios for evaluating the impact of roadside ditches upon hydrology at multiple spatial resolutions. Results demonstrated that roadside ditches heavily remodeled the watersheds water routing framework, elevated peak flow, and stream discharge, and accelerated the transport of NPS pollutants, by-passing natural degradation mechanisms that would contrarily have alleviated their effects.

Smith, (2009) assessed the in-stream processing of soluble P (SP) in agricultural drainage ditches. Soluble P was injected into the ditch water, thereby increasing its concentration by 0.25 mg L⁻¹. Further, the results were monitored at seven locations along three drainage ditches and one location on a third-order stream. Results revealed phosphorus uptake lengths to vary between 40 and 1900 m, and SP uptake rates between
0.4 and 52 mg m\(^{-2}\) h\(^{-1}\) respectively indicating that agricultural drainage ditches have the potential to process nutrients and could, therefore, be used to minimize the export of soluble P.

Surfleet et al., (2010) used a generalized likelihood uncertainty estimation procedure with distributed hydrology soil vegetation model (DHSVM) for two streamflow and 11 road ditch-flow locations. Model’s performance decreased as the size of the modeled area decreased. Further, the sensitivity of parameters and the range of that sensitivity varied across simulations for road-side ditch and streamflow. DHSVM simulations for two streamflow locations varied outside the uncertainty bounds between 10%-22% for storm volumes and 12%-22% for peak flows, respectively. Twenty-eight percent to 52% of storm volumes and 28%-48% of peak flows were outside the uncertainty bounds for the six-road ditch-flow locations.

**2.8.4 Vegetative Filter Strips**

Vegetative filter strip (VFS) is a prominent BMP employed for trapping sediment, nutrient, and bacteria, thereby reducing NPS pollutants transported into streams/creeks and lakes (Inamdar et al., 2001; Parajuli et al., 2008; Park et al., 1994). Typically, a VFS comprises of a zone of thick vegetation designed along the buffer of a stream. VFS intercepts the pollutant loads by (a) trapping the sediment and associated pollutants via particle settlement and (b) by expediting the infiltration of the runoff containing dissolved pollutants (Sebti and Rudra, 2010). Since the construction of VFS is economical and eco-friendly, they are used world-wide. Further, VFS are usually constructed along the edge of agricultural fields and along the buffer of stream or creeks. Therefore, there is a need to simulate VFS effectiveness before their construction.
The use of Vegetative filter strip has been documented by several researches and conservation authorities due to its environment-friendly and economical aspects. Lerch et al., (2017) conducted a experimental study to estimate the impact of width, type of vegetation of the VFS along with the season of the year upon the transport of atrazine (ATR), metolachlor (MET), and glyphosate along with runoff. Vegetation type along with the width of VFS had the most impact upon the transport of herbicide loads.

Muñoz-Carpena et al. (1999) and Helmers et al. (2006) asserted that length of VFS is the most important factor affecting the sediment removal efficiency. Further, a few other studies concluded that enhancing the length of VFS beyond 10 m does not complement its trapping efficiency (Abu-Zreig et al., 2004; Lee et al., 2003). Lobo and Bonilla (2017) developed a SCVFS model for estimating the trapping efficiency of VFS for particles like clay, silt, and sand. The results of the novice model were further compared (calibrated and validated) with WEPP results.

Chen et al. (2016) used a meta-regression approach to develop a model for predicting the efficiency of VFS in removing pesticides. Several factors such as characteristic of pollutants, infiltration, sedimentation etc. formulated the framework of the model. Results indicated that comparatively infiltration had a bigger impact than sedimentation on pesticide retention. Also, the new model outperformed (Q² of 0.81) the existing pesticide retention module of VFSMOD (Q² of 0.72).

Further, a few researchers have also used existing models or developed new interfaces (by integrating models) to simulate VFS at a watershed-scale. Parajuli et al. (2008) used the soil and water assessment tool model to investigate the effectiveness of
VFS in a Kansas watershed. Constructing VFS at various locations along the watershed was found to be efficient at reducing sediment loads at the watershed outlet and along overland flow. However, the BMP was effective at abating fecal coliform bacteria (FCB) only for overland flow.

Sebti and Rudra (2010) developed an interface (GDVFS) to evaluate the performance of vegetative filter strip at a watershed-scale by integrating a vegetative filter strip model (VFSMOD) with a watershed-scale hydrology and sediment transport model (AGNPS). The toolbox however could not be validated with existing data.

Park et al., (2011) programmed a new VFS module within the SWAT models source code for better simulation of overland flow between subwatersheds. The altered approach was adopted to analyze the impact of a diversion channel and VFS upon reduction of sediments. Results demonstrated that there was minuscule difference in simulated flow at the watershed outlet.

Park et al., (2013) developed a web-based GIS oriented VFSMOD interface to provide a user friendly and scientifically appropriate VFS modeling system. The basic purpose of the Web-based system was to determine the appropriate width of a VFS before installing it. The Web GIS-based VFSMOD uses the UH and VFSM executable programs from the VFSMOD-w model as core engines to simulate rainfall-runoff and sediment trapping.

### 2.9 Hydrologic Models

The effectiveness of a BMP within a watershed could be estimated via the aid of computer models, considering they accurately represent site-specific characteristics
Several watershed models including Soil and Water Assessment Tool (SWAT), Annualized Agriculture Non-Point Source (AnnAGNPS), Hydrologic Simulation Program FORTRAN (HSPF), Agriculture Policy/Environmental eXtender (APEX), GIS Pollutant Load Application (PLOAD), Spreadsheet Tool for Estimating Pollutant Load (STEPL) and High Impact Targeting (HIT) are available and are intensively employed for estimating NPS pollution and effectiveness of BMPs. The subsequent sections briefly describe each model, its components, methodology, and finally the utilization of the model.

2.9.1 SWAT

SWAT (Arnold et al., 1998) is a spatially distributed model capable of simulating hydrology at a watershed scale. SWAT is explicitly employed to adumbrate the impingement of management practices pertaining to different land use upon sediments, water, and pesticides within a compound watershed with contrasting soil, land use, and management practices (Boscha et al., 2011; Parajuli et al., 2009; Zhang et al., 2011). The watershed is subdivided into sub-watersheds which are further bifurcated into hydrologic response units (HRUs) having the same feature for soil, landuse, and management practices. Readily available input data such as precipitation, temperature etc. need to be provided to SWAT upon which the model simulates flow, nutrient, sediment, and pesticide yield within the watershed. Further, SCS curve number equation or the Green and Ampt infiltration equation are used to compute the surface runoff. The Modified Universal Soil Loss Equation (MUSLE) is used to estimate the sediment yield for each hydrologic response unit (HRUs). Manning’s equation is employed for calculation related to overland flow (Neitsch et al., 2011).
2.9.2 HSPF

USEPA developed the HSPF model for simulating parameters pertaining to hydrology and water quality (Bicknell et al., 1996). The model has been employed for analyzing the impact of land use changes along with the impact of point and NPS pollution sources at a watershed scale. The continuous watershed-scale model has the potential to calculate several water quality parameters. Both in-stream and overland flow processes are taken into consideration for computing pollutant loads. The methodology of hydrological response units is considered for overland flow routing (Bhaduri et al., 2000). Major components of the model are sediment, phosphorus, nitrogen, dissolved oxygen, pesticides, zooplankton, temperature, and pH. HSPF utilizes several theoretically and empirically developed equations for simulating physical, chemical, and biological processes.

2.9.3 AnnAGNPS

The USDA developed the AnnAGNPS model to address the compound problems pertinent to NPS pollution. The model could be simulated at both event and continuous basis (Bosch et al., 1998). The model is distributed in approach in which the watershed is bifurcated into uniform sized square areas called cells (Polyakov et al., 2007). The hydrological parameters like sediment, runoff, and nutrients (N and P) are computed at a cell basis (Finn et al., 2003). Twenty-two sets of input parameters including DEM, land cover, and soil properties are required by the model (Finn et al., 2003). Hydrology, sediments, nutrients, irrigation, precipitation, and snowmelt are the main building blocks of the model (Bosch et al., 1998). Darcy’s equation is used to compute the lateral subsurface flow while Hooghoudt’s equation is used to calculate flow through tile drains.
2.9.4 APEX

APEX is employed to analyze the impact of different strategies for land management, erosion, soil quality, and plant competition for small watersheds and farmlands (Williams et al., 2008). APEX model could be employed to simulate field level management practices such as furrow diking, terraces, waterways, tillage, pesticides, grazing, buffer strips, crop rotation and manure management (Williams and Izaurralde, 2005). Dominant components of the model include hydrology, sediments, nutrient cycling, pesticides, crop growth, tillage, and weather simulation (Williams et al., 2008). Surface runoff is computed using the SCS curve number method. Peak runoff is calculated using the TR-55 model while USLE, RUSLE, and MUSLE are used to compute soil erosion (Williams et al., 2008).

2.9.5 STEPL

STEPL is a model capable of computing pollutant loads along the reach of a stream at a watershed-scale. Effect of LIDs and BMPs upon sediment and nutrient loads, under different land, uses is analyzed by the model (Tetra Tech, 2006). USLE equation is used for quantifying the annual sediment load. Outputs of the model include surface runoff, nutrient load, and oxygen demand (Nejadhashemi et al., 2011).

2.9.6 PLOAD

The PLOAD model was developed to quantify the impact of BMPs upon pollutant loads on a yearly basis. Some of the input parameters required by the model for simulation includes watershed dimensions, land use, BMP information, data tables pertaining to pollutant loads, precipitation and impervious terrain factors (Nejadhashemi et al., 2011). Several researchers have employed the PLOAD model to appraise the
impact of NPS pollution upon water quality (Nejadhashemi et al., 2011; Endreny and Wood, 2003).

2.9.7 L-THIA

L-THIA is a lumped parameter model. SCS curve number methodology is utilized for computing annual runoff (Bhaduri et al., 2000). Researchers at Purdue University integrated the model with GIS (Bhaduri et al., 1997). Input variables required by the model are land use, soil, and climate while the outputs procured from the model are runoff and NPS pollution (Bhaduri et al., 2000). Numerous researchers have used L-THIA for estimating NPS pollution in different watersheds (Nejadhashemi et al., 2011). Henceforth, it is evident that watershed models are pivotal in the assessment of water quality and implementation of BMPs. Results from watershed simulations have been used to identify specific regions within the watershed where soil and water conservation practices should be employed. However, they have been less efficient at making field-level recommendations due to their watershed discretization techniques. Also, minimal progress has been executed in utilizing the theoretical results of these models for practical purposes at a field scale. Currently, available watershed models fail to locate or simulate ephemeral gullies, even though they are practically a significant source of sediment transport. There is a need to spatially locate ephemeral gullies so that estimates of ephemeral gully sediment load can be achieved using process-based models. Further, it would also aid in estimating the adequate length of VFS that would be most effective in trapping sediment generated or transported from these ephemeral gullies.
2.9.8 **KINEROS 2**

KINEROS2 is an advanced version of the KINEROS model (Woolhiser et al., 1990). KINEROS2 is an event-based, physically oriented, distributed watershed model, competent in simulating runoff and erosion for small-scale rural and urban watersheds (Borah et al., 2006). A watershed is subdivided into cascades of planes and channels. Flow and sediments are routed from one plane to another and finally to the channels and to the outlet of the watershed. Surface runoff is computed by the model using the Hortonian concept employing an upgraded version of the Smith- Parlange infiltration model (Smith and Parlange, 1978).

KINEROS presumes one dimensional flow for each plane. Further, it solves the kinematic wave approximations of the overland and channel flow equations using finite differences approach. For overland planes, KINEROS simulates both splash and hydraulic erosion. The capability of the model to simulate both splash and hydraulic erosion, small detention reservoirs along with its ability to account for spatial variability of precipitation are major advantages of the model (Shoemaker et al., 2005). On the contrary, since the model employs numerical solutions, it is a computationally-comprehensive model which restrains its utilization (Borah et al., 2006). Henceforth, it was suggested by the pioneers of the model (Smith et al., 1995) that watersheds with a maximum area of 10 km$^2$ should be simulated using the model for best results.

2.10 **Targeting Approach**

Efficient implementation of BMPs like detention ponds (WASCoBs), drainage outlets (surface inlets: pipe risers and blind inlets) and tile drains require an adequate design mechanism before construction of the BMP in practice. Prioritizing the
dimensionality of the WASCoBs requires appropriate modeling approach at a watershed scale. The impact of WASCoBs and drainage structures are performed by watershed and water quality models and tools such as AGNPS and GIS.

2.11 Summary

As discussed in the above sections, substantial efforts have been made to evaluate the impact of new BMPs, especially surface inlets (pipe risers and blind inlets) upon hydrology and water quality. However, no modeling approach is available to explicitly compute the effectiveness of surface inlets in an agricultural watershed.

Recognizing the requirement for a comprehensive modeling approach capable of “explicitly examining the performance of surface inlets upon water quality and quantity over a period of events” a new toolbox (CoBAGNPS) is developed which could simulate the impact of pipe risers and blind inlets at a watershed scale. Further, in addition to the increasing interest in the application of surface inlets for mitigating the transportation of pollutants within agricultural watersheds, this research study was initiated to fill an existing lacuna by developing an event-based modeling approach capable of evaluating the NPS pollution abatement efficiency of surface inlets at a watershed scale.
A WASCoB constituting a modest detention pond (berm), surface inlets and tile drains; designed to capture the flow and release it gradually into the drainage system is an efficient watershed BMP. Henceforth; a toolbox, CoBAGNPS for the AGNPS model, is developed to simulate WASCoBs through the AGNPS model. The toolbox utilizes the inputs from AGNPS, through the launching of an application for execution of the WASCoB module. Finally, the output files are generated after routing flow through WASCoBs. The toolbox was applied with a case study in the Gully creek watershed and one of its sub-basins (DFTILE sub-basin) located in Ontario, Canada. The toolbox reproduced the required outputs successfully. Henceforth, significantly enhancing the capability of the AGNPS model to simulate flow through a WASCoB and a network of WASCoBs. Furthermore; the efficiency of the drainage system is also analyzed under different scenarios of pipe risers and tile drains. Also, few scenario analyses were considered in which different diameter drainage pipes were considered to route flow for extreme events for WASCoB3 in the DFTILE sub-basin. A 375-mm diameter drainage pipe is efficient in routing flow for a 10-year 24-hour design storm without it overtopping the berm. Finally, another component of the toolbox was tested, where the flow from the DFTILE sub-basin was directly routed to the outlet of the Gully creek watershed (GULGUL 5) through a drainage pipe of 200 mm with 1% slope, assuming a lag time. The hydrograph at the watershed outlet increased marginally due to the small size of the DFTILE sub-basin. This paper has been submitted to the journal of Catena (Accepted with minor revisions).

and Sediment Control Basin, WASCoB through AGNPS model. Submitted to Catena (Accepted with minor revisions).
Chapter 3

3 CoBAGNPS: A Toolbox for simulating Water and Sediment Control Basin, WASCoB through AGNPS model

3.1 Introduction

Non-point source (NPS) pollution initiated by virtue of agricultural activities is a major contributor towards water pollution in many regions of the world (Das et al., 2006; McKague et al., 2006; Babin et al., 2016; Fraga et al., 2016; Fu et al., 2012; Malawska, 2008; Wang et al., 2016). Significant agricultural NPS pollutants include sediment, nitrogen, phosphorus, pesticides, and pathogens (Gharabaghi et al., 2006; Ahmed et al., 2007a, b; Rudra et al., 2010; Stang et al., 2016). Interrial and rill erosion, along with ephemeral and classical gullies are the three primary mechanisms for the transport of NPS pollutants from agricultural prevailing regions (Daggupati et al., 2013).

Typically, ephemeral and classical gullies are channels culminating from concentrated overland flow, repeatedly reappearing at an identical spatial position after
major precipitation events (Capra et al., 2005; Daggupati et al., 2013; Nachtergaele et al., 2001a; Nachtergaele et al., 2001b; Poesena et al., 2003). Gullies are therefore an important source of pollutants and also play a pivotal part in transporting NPS pollutants. Henceforth; adequate identification of these concentrated flow paths, followed by implementation of an appropriate BMP is pivotal towards mitigating NPS pollutants (Daggupati et al., 2011, 2013). BMPs can be placed in an agricultural watershed either randomly or through strategic targeting process (Tomer et al., 2008; Daggupati et al., 2011).

Numerous studies have demonstrated that random selection and placement of BMPs are not equally efficient in abating NPS pollutants in a given watershed (Dillaha et al., 1988; Sprague and Gronberg, 2012; Tomer et al., 2008). Henceforth, strategic targeting and prioritization of areas and selection of appropriate BMPs is a key for effective watershed management (Diebel et al., 2008; Daggupati et al., 2011; Douglas-Mankin et al., 2013). Conservation tillage, adequate management, and planning for nutrients, and adoption of cover crops are typical BMPs which are widely implemented in agricultural watersheds in many countries around the world to abate NPS pollutants.

Water and Sediment Control Basin (WASCoB) is another useful BMP designed specifically for impeding development of concentrated flow paths or gullies (Fiener et al., 2005; Her et al., 2017; Kovacic et al., 2006; Verstraeten and Poesen, 1999), and is increasingly being adopted in agricultural watersheds.

By definition, a water and sediment control basin (WASCOB) (Figure 3.1) is an “earth embankment or a combination of ridge and channel constructed across the slope of minor watercourses” (Foster and High fill 1983; USDA NRCS code 638). Further,
WASCoBs are constructed to alleviate transport of sediment and associated nutrients generated within farmlands; thereby impeding the formation of gullies by interrupting a watercourse (Her et al., 2016).

Figure 3.1: Location of Gully Creek watershed and the DFTILE sub-basin in Ontario, Canada

Further, WASCoB slowly diverts the runoff ponded behind the berm through surface inlets into an underground drainage network, thereby regulating flow and trapping sediments and nutrients within the ponding area (Fiener et al., 2005; Kovacic et al., 2006; USDA NRCS code 638). The combined system of berms, surface inlets, and tile drains are generally referred to as a WASCoB (Figure 3.2).
Figure 3. 2: (A) Grid discretization of the Gully creek and the DFTILE sub-basin in the AGNPS model, (B) Pipe riser consisting of a standpipe connected to a subsurface drain

There are many types of surface inlets, and selection of appropriate types and size of the inlet is critical to ensure proper functioning of the WASCoBs (Li et al., 2017). Perforated pipe risers (also referred to as tile risers) are the most widely used surface inlets for drainage systems (King et al., 2015; Li et al., 2017; Oolman and Wilson, 2003). Pipe risers of several dimensions are employed for routing flow ponded behind the berm through the buried tile drains thereby affecting the water balance (usually peak discharge) and sediment nutrient loads at the outlet tile drain network.

Few studies have examined the impact of WASCoBs on flow and nutrient transport. Her et al., (2017) evaluated the effect of nine field-scale BMPs (including WASCOBs) in alleviating sediment and nutrient loads using the SWAT model in the St.
Joseph River watershed, U.S.A. Results revealed that WASCOBs were effective in decreasing the amount of surface runoff and increasing the soil water content. Further results indicated that WASCoBs are efficient in minimizing sediment and nutrients (phosphorus and nitrogen) losses, except soluble nitrogen. Kovacic et al. (2006) investigated the combined impact of a WASCoB and two wetlands, which intercepts surface runoff and tile drainage in reducing NPS pollution in Lake Bloomington Illinois, USA. Concentrations of NO$_3^-$-N were reduced by 42% and 31% in the two wetlands, respectively. Further, combined phosphorus (P) and combined total organic carbon (TOC) mass retention was 53% and 9%, respectively. Further, the impact of the different type of surface inlets upon flow and sediment characteristics has been understudied by researchers.

King et al. (2015) reviewed several studies about surface inlets and asserted that traditional surface inlets (typically surface tile inlets) are potent in increasing the transport of pollutants into receiving water bodies. Further, Li et al. (2017) analyzed the flow and sediment transport characteristics of four pipe risers. Feyereisen et al. (2015) determined the impact of replacing pipe risers with blind inlets upon TSS and P concentrations. Results demonstrated that blind inlets were more effective in alleviating sediment and P loads. Tomer et al. (2010) studied the dynamics of nutrients, Escherichia coli, and sediment during a runoff event in Tipton Creek, Iowa. Results revealed that 13 % of the flow in the watershed was due to discharge from tile risers. Ginting et al. (2000) evaluated the characteristics of surface runoff flowing via surface inlets draining natural depressions in two watersheds in Minnesota. Results revealed that loss of particulate pollutants through surface inlets were smaller than expected. However, additional research is
recommended to compute the impact of WASCoBs with different combination and permutation of pipe risers and tile drain pipes upon flow, sediment and nutrient characteristics. Due to financial and time constraints related to experimental in agricultural studies, mathematical modeling with the aid of hydrological models is recommended (Golmohammadi et al., 2016).

Although a few researchers have developed modeling approaches to simulate WASCoBs using SWAT their approaches have severe lacunas. For example, Her et al. (2017) evaluated the impact of WASCoBs on sediment and nutrient loads. To simulate WASCoBs with the SWAT model, they used an approach similar to simulating terraces on the contour. This approach does not consider routing flow through surface inlets like pipe riser or blind inlets. Hence, a more authentic approach about WASCoB simulation is needed. Further, Yang et al. (2013) developed a WASCoB module and linked it with the SWAT model to examine the impact of WASCoBs upon water quantity and quality in the Gully Creek watershed, Ontario Canada. WASCoBs were represented as a subbasin outlet in the module. Furthermore, the stage-storage and stage-discharge relationships for the WASCoBs were used to route water through the tile drain. However, their modeling efforts were focused upon continuous timestep. The authors simulated the modeling exercise on a daily interval, thereby assumed a constant height of water in the berm for each simulation day, which is an unrealistic assumption. Moreover, no analysis was performed using perforated pipe risers of different dimensions.

Borah and Bera (2004) stated that continuous modeling is reliable for making predictions on a yearly or monthly basis even though the models can run at a daily time step. However, a continuous modeling approach would be less accurate for
modeling stream flows which are characterized by severe storm events (Borah and Bera, 2004). Therefore event-based models perform well in simulating storm events, including severe storms (Borah et al., 2007). Further, hydrology plays a significant part in the amount of runoff generated, ultimately impacting sediments transported along with it. Also, since the pattern of rainfall changes significantly between seasons, an event-based modeling approach would be better than a continuous modeling approach. Also, WASCoBs are typically designed with a minimum capacity to detain runoff generated during a 24-hour storm event having a return period of 10-years, thereby; employing a combination of berm (to control flood storage) and discharge through the outlet (USDA NRCS code 638). Henceforth, since the detention time for the designed WASCoBs is less than 24 hrs, modeling hydrology of WASCoBs at sub-daily/hourly interval is paramount. Therefore, flow simulation at hourly or sub-hourly basis is required which could only be accomplished via an event-based modeling system.

Although SWAT can be modeled at a sub-daily or hourly basis (Boithias et al., 2017; Yang et al., 2016) making it capable of simulating individual events, but to simulate flow through surface inlets results on a finer resolution (15, 30 min, etc.) are needed. Also, flow cannot be routed between individual HRUs in SWAT. Due to all these lacunas, the AGNPS model seems to be an appropriate model which could be used for simulating WASCoBs at a watershed scale. Further, AGNPS can also route flow between individual cells. AGNPS is an event-based hydrological model operating at the watershed scale employed widely for NPS pollution estimation (Cho et al., 2008; Liu et al., 2008; Miklanek et al., 2004; Mohammed et al., 2004; Parajuli et al., 2007; Sebti and Rudra, 2010). However, the current version of the AGNPS model does not have the capacity to simulate
the impact of WASCoBs on hydrology and sediment loads. Her et al. (2017) and Yang et al. (2013) did use a modeling approach to simulate WASCoBs at a watershed scale. However, their modeling approach was simplistic and utilized a limited set of physical characteristics pertaining to WASCoBs. Moreover, no attempt was made to emphasize the importance of surface inlets in their modeling exercise. Therefore, a novel approach was developed in this study through which WASCoBs can be simulated using the AGNPS model. This paper describes the development of a toolbox, named CoBAGNPS which would enhance the lacuna of the AGNPS model, thereby, making it capable of simulating WASCOBs. Further, the capabilities of the improved model are tested for an agricultural watershed in Ontario, Canada. Specific questions associated with novice modeling approach addressed in this paper are (1) can the CoBAGNPS simulate the hydrology of WASCoBs? (2) will there be a difference is the simulated water yield using pipe risers of different dimensions?

3.2 Materials and Methods

3.2.1 AGNPS Model

Version 5.0 of the AGNPS model (Young et al., 1989) was used for this study. AGNPS is an event-based watershed model developed by USDA-Agricultural Research Service (ARS) (Jianchang et al., 2008), capable of estimating surface runoff, sediment yield, and nutrient loading for a watershed. The watershed is subdivided into discrete square cells. Further, the SCS curve number method (SCS-USDA, 1972, 1985) is used for computing surface runoff for each cell separately by the model.
\[ I_a = 0.2S \quad (3.1) \]

\[ S = \left( \frac{1000}{CN} - 10 \right) \quad (3.2) \]

\[ Q_D = \frac{(P - I_a)^2}{P + S - I_a} \quad (3.3) \]

Where,

\[ Q_D = \text{Surface runoff (mm)} \]

\[ P = \text{Storm precipitation (mm)} \]

\[ S = \text{Potential maximum retention (mm)} \]

\[ I_a = \text{Initial abstraction (mm)} \]

\[ CN = \text{Curve number} \]

Curve number is an arbitrary number assigned to a unique combination of land use, soil and the antecedent soil moisture condition (AMC) (Neitsch et al., 2002). Curve number method evaluates surface runoff, which further computes channel runoff, and subsurface flow. Antecedent soil moisture conditions (AMC) for the soil is calculated based upon preceding five-day precipitation amount before a particular storm.

Retention factor \( S \) is constant for a storm event. Infiltration, interception, depression storage and AMC processes preceding surface runoff initiation are accounted for in initial abstraction \( I_a \) (Eq. 1). Further, \( Q_D \) (Eq. 3) refers to surface runoff in the AGNPS model (Young et al., 1989). A unique curve number is assigned to each cell depending upon its predominant landuse and soil type. Hence, surface runoff is computed
for each cell. Further, the surface runoff computed is routed through the watershed based upon gradient difference between cells until it reaches the outlet cell. In addition, the model estimates peak flow rate using the surface runoff, drainage area, slope and size of the channel, and maximum length of the flow path into the cell for each cell (Smith and Williams, 1980). It is to be noted that the model primarily calculates surface water flow and sediment and nutrient (like nitrogen and phosphorus) yield for single storm events. Detailed description of the processes used in the model could be found in the AGNPS manual (Young et al., 1989; http://www.waterbase.org/docs/MWAGNPS%20Setup.pdf).

### 3.2.2 Study Area

The study area chosen for this research is the Gully creek watershed and one of its subbasin, the DFTILE sub-basin (having WASCoBs) which is located in the eastern part of the Gully creek watershed, Ontario, Canada (Figure 3.1). The Gully creek watershed encompasses an area of 1056.84 hectares, while the DFTILE sub-basin envelops an area of 18.79 hectares. The landform is characterized by undulating topography ranging between 217 m of 281 m. Further, the DFTILE sub-basin comprises six WASCoBs which finally drain to the DFTILE outlet through a network of tile drains.

Nearly 60% of the total precipitation occurs as rainfall from April to October, while the remaining as snow during November to March. During 2001-2011, the average annual precipitation is 1,055 mm with a standard deviation of 165 mm. Upper reaches of the watershed are dominated by clay loam soil while the lower reaches are mostly sandy loam (Table 3.1).
Table 3. 1: Name and extent of each soil type in the Gully Creek watershed

<table>
<thead>
<tr>
<th>Soil code</th>
<th>Soil type</th>
<th>Area (ha)</th>
<th>Area (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>ZAL</td>
<td>Bottom Land</td>
<td>84.57</td>
<td>8.08</td>
</tr>
<tr>
<td>PTH</td>
<td>Perth Clay Loam</td>
<td>101.06</td>
<td>9.66</td>
</tr>
<tr>
<td>HUO</td>
<td>Huron Clay Loam</td>
<td>797.11</td>
<td>76.21</td>
</tr>
<tr>
<td>BKN</td>
<td>Brookston Clay Loam</td>
<td>63.26</td>
<td>6.05</td>
</tr>
</tbody>
</table>

Landuse of the watershed is primarily agriculture dominated with 70% agricultural land, and the remaining 25% dominated by trees, shrubs, and grasses (Figure 3.3). Soybean, corn, and winter wheat are the major crops grown in the watershed (Table 3.2).

Figure 3. 3: (A) Digital Elevation model, (B) Soil, and (C) Landuse of the Gully Creek watershed
Table 3.2: Land use types and res in the Gully Creek watershed

<table>
<thead>
<tr>
<th>Soil Code</th>
<th>Soil Type</th>
<th>Area (ha)</th>
<th>Area (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>CORN</td>
<td>Corn</td>
<td>305.13</td>
<td>29.17</td>
</tr>
<tr>
<td>SOYB</td>
<td>Soybean</td>
<td>208.62</td>
<td>19.94</td>
</tr>
<tr>
<td>WWHT</td>
<td>Winter Wheat</td>
<td>234.89</td>
<td>22.46</td>
</tr>
<tr>
<td>BARL</td>
<td>Spring Barley</td>
<td>7.18</td>
<td>0.69</td>
</tr>
<tr>
<td>HAY</td>
<td>Hay</td>
<td>19.47</td>
<td>1.86</td>
</tr>
<tr>
<td>PAST</td>
<td>Pasture</td>
<td>20.02</td>
<td>1.91</td>
</tr>
<tr>
<td>ORCD</td>
<td>Orchard</td>
<td>2.31</td>
<td>0.22</td>
</tr>
<tr>
<td>BROM</td>
<td>Meadow Bromgrass</td>
<td>9.42</td>
<td>0.90</td>
</tr>
<tr>
<td>FESC</td>
<td>Tall Fescue</td>
<td>7.47</td>
<td>0.71</td>
</tr>
<tr>
<td>FRSE</td>
<td>Forest-Evergreen</td>
<td>17.82</td>
<td>1.70</td>
</tr>
<tr>
<td>FRSD</td>
<td>Forest-Deciduous</td>
<td>48.89</td>
<td>4.67</td>
</tr>
<tr>
<td>FRST</td>
<td>Forest-Mixed</td>
<td>128.99</td>
<td>12.33</td>
</tr>
<tr>
<td>RNGB</td>
<td>Range-Brush</td>
<td>0.81</td>
<td>0.08</td>
</tr>
<tr>
<td>WETN</td>
<td>Wetlands-Non-Forested</td>
<td>0.66</td>
<td>0.06</td>
</tr>
<tr>
<td>WATR</td>
<td>Water</td>
<td>0.58</td>
<td>0.06</td>
</tr>
<tr>
<td>URLD</td>
<td>Residential-Low Density</td>
<td>23.89</td>
<td>2.28</td>
</tr>
<tr>
<td>UTRN</td>
<td>Transportation</td>
<td>9.85</td>
<td>0.94</td>
</tr>
</tbody>
</table>

3.2.3 Input Data

Geospatial data used for setting up the AGNPS model included DEM (5 m resolution), soil, land use, streamflow network. The data layers were obtained from Ontario Ministry of Agriculture, Food, and Rural Affairs (OMAFRA), Ontario Ministry of Natural Resources and Forestry (OMNRF), and the Ausable Bayfield Conservation Authority (ABCA). Precipitation data (April 2013 to May 2014) needed to simulate the model was procured from a weather station installed within the Gully creek watershed which had been established as a part of watershed based BMP evaluation project by Yang et al. (2013) initiated in April 2011. The Stage (water level) in each DFTILE sub-basin (having WASCoBs) is monitored with the use of the water level logger, VanEssen brand, model D1501.
3.2.4 Model Setup

In the AGNPS model, the Gully creek watershed is discretized into uniform grids of 250 m, except for the DFTILE sub-basin which is further discretized into grids of smaller units to appropriately represent each berm sub-basin (Figure 3.2). Upon successful calibration and validation of the AGNPS model for surface runoff and peak flow at the outlet of the watershed (GULGUL 5), flow directions of the cells within the DFTILE sub-basin are adjusted manually to represent each berm sub-basin as an individual identity. The flow at the outlet cell of each berm sub-basin represents the inflow to the pipe riser of the berm. Further, surface runoff and other hydrological parameters computed for these outlet cells (for all sub-basins) are routed through the pipe riser using the toolbox developed to calculate the outflow from the pipe riser.

3.2.5 Base Flow Separation

Surface runoff and base flow compound to generate streamflow. While surface runoff is a fast contributor to stream flow, base flow is a slow contributor to streamflow (Kalin and Hantush, 2006). Since AGNPS is an event-based model, it only simulates surface water hydrology. Therefore, base flow is separated from streamflow at the outlet of the Gully creek watershed (GULGUL5 station) for adequate calibration and validation of events. WHAT program(https://engineering.purdue.edu/mapserve/WHAT/) was used to separate base flow from observed stream flow based upon the methodology described by Kyoung et al. (2005). Signal analysis and processing which bifurcates high-frequency signals from low-frequency signals form the basis of the methodology (Ladson et al., 2013) where high, and low-frequency waves are correlated with direct runoff and base flow respectively (Eckhardt and Arnold, 2001).
3.2.6 CoBAGNPS Development

3.2.6.1 Conceptual Design

An amalgamation of a modest detention pond (referred to as a berm), surface inlet and drainage system is referred to as a WASCoB (Figure 3.1). WASCOBs are usually designed to capture flow and nutrients along the concentrated flow paths or gullies (Fiener et al., 2005; Kovacic et al., 2006). Therefore; WASCoBs are typically constructed along ephemeral and classical gullies within a watershed (Liu et al., 2013). In addition, WASCoBs are usually designed as per the criteria of a severe storm selected for a given study area (e.g., 2-year, 5-year, and 10-year 24-hour storms).

Water flowing through the berm of the WASCoB stems from the drainage area above the WASCoB. Therefore, a detailed LiDAR/geographic DEM data was used to prepare a stage-volume (storage) relationship for the ponding area of each WASCoB berm (Figure 3.4), in the DFTILE sub-basin.

The AGNPS model is used to compute runoff volume stored behind the berm for the drainage area of each WASCoB. Furthermore; the stage-storage relationship of the runoff volume ponded behind the berm combined with the stage-discharge relationship for pipe risers (R. Wilson, 2016) is utilized to determine the discharge from the WASCoB into the tile drain. Under ideal conditions, water enters a pipe riser and is conveyed through an underground tile drain to the mainstream channel or the road-side ditch.
However, if the water stored behind the WASCoB berm exceeds the storage volume, then the excess water overtops the berm. This conceptualization forms the basis of the WASCoB module design and consists of five main steps:

1. The AGNPS cell delineation was performed such that the drainage area of each WASCoB represented as a sub-basin and the WASCoB location represented as the outlet of that sub-basin. The flow direction of the cells in the sub-basin above the WASCoB are adjusted using the flow editor in tool in the AGNPS model.

2. Procuring a stage-storage relationship for the WASCoBs ponded area and stage-discharge relationships for various pipe risers.

3. Route the water ponded through the pipe riser into the tile drain.
4. A similar process is repeated for each WASCoBs.

5. Finally, flow routed from each WASCoB is routed through the drain pipes assuming a lag time (Irwin, 1985) to the outlet of the sub-basin (DFTILE).

3.2.6.2 Detailed description of the Toolbox

CoBAGNPS is written in C# (www.learncs.org) which is a state-of-art high-level programming language. The design of CoBAGNPS is divided into two parts: (1) CoBAGNPS flow, which contains the code for routing the flow computed at the outlet cell of the sub-basin cell (representing the WASCoB or berm) procured from the AGNPS model through a combination of pipe risers and tile drains and (2) CoBAGNPS sediments, which contains code for routing sediment procured at the outlet cell of the sub-basin through pipe riser and blind inlet. The performance of the latter module will be discussed in another manuscript.

CoBAGNPS can be installed as a stand-alone application. The conceptual architecture of the toolbox is demonstrated in (Figure 3.5). Each berm within the DFTILE sub-basin is represented as a separate watershed in the AGNPS model. The “DownstreamRunoff” parameter for the outlet cell for each “berm” is fed into the toolbox.

Henceforth; an input hydrograph is generated for the surface inlet (pipe riser in the present scenario) based on the “dimensionless triangular hydrograph” methodology (U.S. Soil Conservation Service, 1972). Further, the hydrograph generated is routed through the WASCoB (a combination of berm and surface inlets) using the “Level pool” routing method (Chow et al., 1988).
The storage-discharge and the stage-discharge relationship of the berm and the pipe risers combined are employed for computing the outflow hydrograph from the pipe riser. Finally, flow for each of the sub-basin is routed to the DFTILE outlet assuming an appropriate lag time for flow through the tile drain (Irwin, 1985). Figure 3.6 shows the main components of CoBAGNPS toolbox and its functionalities.
3.2.6.3 Model Calibration and Validation

Complementary to most distributed models, the AGNPS model also constitutes certain empirical components. Values for a few variables in the AGNPS model, viz. SCS curve number (discussed earlier), crop cover and management factor in the modified universal soil loss equation (MUSLE), are not fixed physically (Cho et al., 2008; Choi and Blood, 1999; Liu et al., 2008). Henceforth, results procured from the AGNPS model should be calibrated and validated with the observed data. In this study, the calibration was performed for peak flow, and surface runoff for nine storm events, followed by validation of nine storm events from June 2012 to April 2014. As per guidelines of (Wischmeier and Smith, 1978), an event is procured if it exceeds precipitation of 12.5 mm and separated from another precipitation event by more than 6 h or else as much as 6 mm of precipitation occurs within 15 min.
Observed flow data was procured from Gully Creek conservation authority at the outlet of the watershed (GULGUL 5). Model simulated results were evaluated for both calibration and validation periods using commonly used performance measures which such as coefficient of determination ($R^2$) and Nash–Sutcliffe efficiency ($E_{NS}$). Nash–Sutcliffe efficiency (Moriasi et al., 2015) is the measurement of how well the plot of observed and predicted values fit the 1:1 line, while the coefficient of determination is a measure of the strength between observed and simulated values. A value greater than 0.5 for the Nash-Sutcliffe coefficient and coefficient of determination is considered acceptable for model calibration and validation (Santhi et al., 2002).

3.2.6.4 Pipe Riser and Tile Drain Evaluation

Statistics from calibrated and validated AGNPS models were used to compute the performance of WASCoBs particularly pipe risers and tile drains. For this study, several scenarios were considered to route water ponded behind the berms through a combination of different pipe risers and tile drains, finally to the sub-basin outlet (DFTILE) using CoBAGNPS toolbox. Furthermore, routing efficiency of the drainage system was also evaluated considering tile drains of different diameter. A couple of storm events (18 April 2013 and 1 August 2013) were selected to perform these scenarios. Under the first set of scenario analysis, the performance of the different type of pipe risers (HB, HBS, and PI) in routing flow to the sub-basin outlet (DFTILE) was analyzed. Further, a separate analysis was performed for assessing the impact of different pipe risers (HB, HBS, and PI) upon flow characteristics for only one WASCoB (WASCoB 3).

In another scenario analysis, the impact of different diameter of tile drains upon flow characteristics for extreme storm events (2-year, 5-year 24-hour storm duration) was
considered. For this modeling exercise, WASCoB 3 along with HB pipe riser and tile drains of 150, 200, and 375 mm diameter were selected. The selection was made as per the recommendation of Ausable Bayfield Conservation Authority watershed report (Ross Wilson, 2016). Purpose of this modeling exercise is to recommend the most appropriate diameter for the tile drain pipe which could route flow at a rapid rate allowing least amount of water to over topple the berm for the storm event selected.

Under the final scenario analysis, the performance of another important module of the toolbox is investigated: “The direct routing module.” Execution of this module directly routes the hydrograph generated at the DFTILE outlet to the GULGUL 5 outlet using a time lag methodology. Purpose of this module is to analyze the impact of WASCoBs upon flow at the watershed outlet. A portion of surface runoff which under natural conditions could have been lost due to evapotranspiration or infiltration during overland routing, would now positively contribute to the watershed outlet. Henceforth, in this scenario the flow from the DFTILE outlet is directly routed to the GULGUL 5 outlet through an artificially assumed tile drain of 150 mm in diameter, assuming an appropriate lag time.

3.3 Results and Discussion

3.3.1 Calibration and Validation of Flow at GULGUL 5 Outlet

Rainfall-runoff models forecast streamflow is utilizing accessible climate information (e.g., precipitation, temperature, solar radiation, wind speed) and adjusting the model parameters to replicate certain components of the watershed. However, under general circumstances, these model parameters cannot be estimated precisely by measurement of watershed characteristics. Henceforth, the value of these parameters is estimated through the aid of model calibration (Kim and Lee, 2014). According to
(Madsen, 2000) the objective of model calibration is the “selection of model parameters so that the model simulates the hydrological behavior of the catchment as closely as possible.” Calibration and validation of individual storm events is a complicated process due to the variability in model parameters and the observed values. Further, the optimal value for a model parameter significantly depends upon the season of the storm event considered for calibration. Therefore, a degree of uncertainty will exist when determining the optimal value of a model parameter for a storm event.

In this study, data for individual seasons was utilized to calibrate and validate the model on a seasonal basis. Six storm events were considered for spring, summer and fall season respectively. Further, the events were bifurcated into three events for calibration and validation. Also, model parameters were adjusted for the calibration events for each season. Statistical parameters were however computed for the total calibration and validation events compounded for all the seasons.

### 3.3.1.1 Seasonal Calibration

Table 3.2 shows the events selected for calibration and validation for fall, spring and summer respectively. As described in the section above, six events are selected for each season, where three events are used for model calibration, and the latter three events are utilized for the model validation. Figure 3.7 demonstrates the distribution of SCS curve number and soil hydrologic group for each grid employed in the AGNPS model for the Gully creek watershed.
Figure 3. 7: Curve number and soil hydrologic group distribution for each grid employed in the AGNPS model
Table 3.3: Events selected for calibration and validation based upon different seasons

<table>
<thead>
<tr>
<th>Season</th>
<th>Date</th>
<th>Precipitation</th>
<th>Duration</th>
<th>AMC</th>
<th>manning's</th>
<th>$Q_{Peak}^{Observed}$</th>
<th>$Q_{Peak}^{Simulated}$</th>
<th>Surface Runoff</th>
<th>Surface Runoff</th>
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<td></td>
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<td>(hours)</td>
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<td>(m$^3$/s)</td>
<td>(m$^3$)</td>
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<td></td>
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<td>II</td>
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<td>0.28</td>
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<td>Date</td>
<td>Runoff</td>
<td>SAC</td>
<td>30-Year PDD</td>
<td>Average Peak Flow</td>
<td>Direct Runoff</td>
<td>Average</td>
<td>5-Year PDD</td>
<td>Average</td>
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<td></td>
</tr>
<tr>
<td>12-Jun-13</td>
<td>19</td>
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<td>II</td>
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<td><strong>5368</strong></td>
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<td></td>
</tr>
</tbody>
</table>

**Calibration**
- $R^2$: 0.97
- $E_{NS}$: 0.89

**Validation**
- $R^2$: 0.91
- $E_{NS}$: 0.83

AMC: Antecedent moisture condition  
$E_{NS}$: Nash-Sutcliffe efficiency  
$R^2$: Coefficient of determination
During the calibration process, model parameter values were altered to get a good match between the model-predicted peak flow, surface runoff and were the observed peak flows, surface runoff values. Parameters varied for calibration was the SCS curve number (CN2), and the manning’s n, which have considerable impact upon surface runoff and peak flow. Therefore, SCS curve number and manning’s n were manually manipulated for each grid and each channel pertaining to the grid until peak flow and direct runoff simulated at the outlet grid of the AGNPS model closely matches the observed value for parameters simulated. The curve number is altered based upon the antecedent moisture conditions (AMC-I, II, and III) by considering the previous 5-day precipitation amounts. Antecedent moisture conditions also vary with the season of the storm as the precipitation depends upon the season of the storm. The threshold used for AMC I, II, and III were based on criteria suggested by (Jianchang et al., 2008; Wischmeier and Smith, 1978) which is listed in Table 3.4.

Table 3. 4: Criteria suggested by (Jianchang et al.,2008; Wischmeier and Smith, 1978) for computing SCS curve number for AMC I, II, AND III conditions

<table>
<thead>
<tr>
<th>AMC</th>
<th>5-day antecedent rainfall</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0-13 mm</td>
</tr>
<tr>
<td>2</td>
<td>13-28 mm</td>
</tr>
<tr>
<td>3</td>
<td>&gt; 28 mm</td>
</tr>
</tbody>
</table>

Adjustment of the SCS curve number (CN2), is deemed necessary during model’s calibration and validation for surface runoff and peak flow in many AGNPS studies (Cho et al., 2008; Liu et al., 2008; Parajuli et al., 2007). Further, the CN values for the most predominant landuse and soil type for each grid within the Gully creek watershed is shown in Figure 3.7. Curve number is varied under this prescribed range during the model calibration and validation (Neitsch et al., 2002). The logic behind altering SCS curve
number and manning’s n for each grid is that once a close agreement is reached between observed and modelled values for peak flow and direct runoff at the watershed outlet, the adjusted values for SCS curve number and manning’s n would be realistic and matching the real-world parameters for each grid within the watershed. Henceforth, the SCS curve number and manning are n values within the DFTILE sub-basin would also be close to reality thereby simulating the real-world dynamics. Therefore, it could be assumed that the hydrologic parameters observed the outlet cell for each berm (Figure 3.2) would match the real-world hydrologic parameters. Thus, the flow procured after routing water ponded behind the berms through the pipe risers and the tile drainage network to the DFTILE outlet would be realistic and match the real-world dynamics (although we do not have real data to match the simulated flow at the DFTILE outlet).

### 3.3.1.2 Individual Season’s

For the summer season, three events were selected for model calibration (28 June 2013, 16 June 2013, 1 August 2013) and model validation (30 August 2013, 12 June 2013, 12 June 2012) respectively (Table 3.3). Similarly, for the fall season, three events were selected for model calibration (14 October 2012, 14 September 2012, 20 September 2013) and model validation (20 October 2012, 17 October 2013, 12 November 2012) respectively. Also, for the spring season again three events were selected for model calibration (18 April 2013, 12 April 2014, 28 May 2013) and validation (29 April 2014, 24 April 2013, 31 May 2013).

Calibration was executed manually by statistically comparing computed and observed peak flow and surface runoff values. Table 3 & 5 shows the calibrated parameters for each storm event selected where CN is the SCS curve number and n
represents manning's n. Furthermore, Table 3.3 reveals the coefficient of determination ($R^2$) and Nash–Sutcliffe efficiencies ($E_{NS}$) (Nash and Sutcliffe, 1970) for both surface runoff and peak flow for the combined events, used for calibration and validation. Further, the mean value for peak flow and surface runoff is also computed for each season individually for the calibration and validation events.

The measured average peak flow of 0.4863 m$^3$/s was slightly lower than the predicted peak flow of 0.6975 m$^3$/s over the summer calibration period. Similarly, for the fall calibration period, the measured average peak flow of 0.1766 m$^3$/s underpredicted the simulated peak flow of 0.2726 m$^3$/s. Complementary to the summer and fall season, the average peak flow of 1.5227 m$^3$/s underpredicted the simulated value of 1.6258 m$^3$/s for the spring season. This suggests that AGNPS slightly over-predicted peak flow for the calibration period for all the seasons (Figure 3.8). This was further confirmed by high statistical parameter values with $R^2$ 0.97 and $E_{NS}$ 0.89 computed for the calibration events combined.

Further, Table 3.3 also demonstrates peak flow for all the validation events. The results suggest that AGNPS prediction of peak flow for the validation period was adequate (Figure 3.8). The estimated peak flow of 0.27 m$^3$/s was slightly higher than the predicted peak flow of 0.2137 m$^3$/s over the fall validation period. During the spring season, the estimated peak flow (0.48 m$^3$/s) overpredicted the simulated peak flow (0.456 m$^3$/s). Similarly, the estimated average peak flow of 0.286 m$^3$/s was slightly higher than the simulated average peak flow of 0.266 m$^3$/s for the summer season for the validation events considered.
Figure 3.8: Observed and simulated peak flow at the GULGUL 5 outlet (A) Summer season, (B) Fall season, and (C) Spring season.
This suggests that AGNPS slightly underpredicted peak flow for the validation period, contradictory to the calibration events for all the three seasons (fall, spring, and summer). Statistical parameters $R^2$ and $E_{NS}$ were 0.91 and 0.83, respectively for the validation events combined for all the seasons.

Graphical comparison of direct runoff for the calibration period for individual seasons (Figure 3.9) and statistical parameters (for all the calibration events combined), with $R^2$ 0.87 and $E_{NS}$ 0.73, suggest that the direct runoff was adequately simulated by the AGNPS model. Estimated average direct runoff for the fall season of 4372.9 m$^3$ was about the same as the predicted average direct runoff of 4472.87 m$^3$ over the entire validation period. For the spring season, predicted average direct runoff of 25048.12 m$^3$ under-predicted the estimated average direct runoff of 27761.28 m$^3$. However, for the summer season, the predicted average direct runoff of 13418.62 m$^3$ over-predicted the estimated average direct runoff of 8164.02 m$^3$.

Table 3.3 also demonstrates direct runoff for all the seasons, for the validation period respectively. The results suggest that AGNPS prediction of direct runoff for the validation period was satisfactorily (Figure 3.9). The same is validated by the statistical parameters $R^2$ and $E_{NS}$ of 0.91 and 0.46, respectively, suggesting that the direct runoff was adequately simulated by the AGNPS model for the validation events combined for all the seasons. For the summer season, the estimated monthly average direct runoff of 5212.74 m$^3$ was about the same as the predicted direct runoff of 5367.45 m$^3$ over the entire validation events.
Figure 3.9: Observed and simulated direct runoff at the GULGUL5 outlet (A) Summer season, (B) Fall season, and (C) Spring season.
However, for the fall events, the predicted direct runoff of 6262 m$^3$ under-predicted the estimated direct runoff of 9075.8 m$^3$. Similarly, for spring events, predicted direct runoff of 9840.33 m$^3$ under-predicted the estimated direct runoff of 17616.24 m$^3$. Figure 9 demonstrates the direct runoff for all the calibration and validation seasons combined. Since peak flow and direct runoff flow predictions used for WASCoB simulation were very good, the AGNPS model was considered to be adequately describing the hydrology of the study watershed.

3.3.2 Flow Routed through WASCoB 3 using different Type of Pipe Risers

Under this investigation, the impact of the different type of pipe risers for a single WASCoB (WASCoB 3) upon the outflow hydrographs is considered. A currently installed tile drain of 150 mm diameter is considered for simulation. Upon procuring the inflow hydrograph for the WASCoB through the AGNPS model, flow is routed through three different type of pipe risers namely, HB, HBS, and PI) (Figure 3.10). The purpose of this investigation is to precisely measure any difference in flow characteristics routed through different pipe risers. Two storm events: 18 April 2013 and 1 August 2013 are used for simulating the outflow hydrographs.

Flow characteristics for the two storm events are fairly similar when routed through a particular pipe riser. The primary distinction being in the magnitude of hydrographs routed during the storm events. Graphical comparisons demonstrate no substantial difference in flow characteristics simulated from an HB and PI pipe riser. However, the time required to drain the WASCoB through an HB pipe riser is slightly more.
Figure 3. 10: Flow simulated at the outlet for WASCoB 3 for different type of pipe risers (A) 18 April 2013 HB pipe riser, (B) 18 April 2013 HBS pipe riser, (C) 18 April PI pipe riser, (D) 1 August 2013 HB pipe riser, (E) 1 August 2013 HBS pipe riser, (F) 1 August 2013 PI pipe riser
Figure 3.

Figure 3. 11 (cont): Flow simulated at the outlet for WASCoB 3 for different type of pipe risers (A) 18 April 2013 HB pipe riser, (B) 18 April 2013 HBS pipe riser, (C) 18 April PI pipe riser, (D) 1 August 2013 HB pipe riser, (E) 1 August 2013 HBS pipe riser, (F) 1 August 2013 PI pipe riser
Figure 3. 12 (cont): Flow simulated at the outlet for WASCoB 3 for different type of pipe risers (A) 18 April 2013 HB pipe riser, (B) 18 April 2013 HBS pipe riser, (C) 18 April PI pipe riser, (D) 1 August 2013 HB pipe riser, (E) 1 August 2013 HBS pipe riser, (F) 1 August 2013 PI pipe riser
This could be attributed to the smaller number/area of the holes along the circumference of the cylindrical pipe of the HB pipe riser draining water into the tile drain. Further, flow routed through the HBS surface inlet demonstrates flow characteristics distinct to the other two pipe risers with a much shorter peak discharge. Henceforth, water is drained at a slightly slower rate through the HBS surface inlet.

3.3.3 Flow at the sub-basin outlet (DFTILE) routed through WASCoBs using different type of pipe risers

After successfully investigating the impact of different pipe risers upon the drainage efficiency and the flow characteristics for a single WASCoB (WASCoB 3); it is worthwhile to investigate the impact of different pipe risers upon flow characteristics at the outlet of the DFTILE sub-basin, i.e., DFTILE, routing flow for all the WASCoBs combined. Utilizing the procedure described earlier, results obtained from the AGNPS model for the outlet cell of the WASCoBs were transferred through the CoBAGNPS toolbox to simulate the event-based flow at the outlet of the DFTILE sub-basin, i.e., DFTILE. The following scenarios pertaining to WASCoBs were performed. Inflow hydrograph for the WASCoB procured through the AGNPS model, is routed through three different type of pipe risers namely, HB, HBS, and PI) (Figure 3.11). Two storm events: 18 April 2013 and 1 August 2013 are used for simulating this scenario.

3.3.3.1 Scenario1: Routing Flow through an HB pipe riser

Graphical plots of the observed discharges (m$^3$/s) procured from the toolbox for the two-storm event: 1 August 2013 and 18 April 2013, are plotted in Figure 3.11 a, b respectively after adequate calibration and validation of the AGNPS model. Visual scrutiny of these results reveals that the overall simulation appears to be reasonable. It
can be concluded from these results that the model simulated flow in the tile drain (DFTILE outlet) as per the expected lines. The predicted amount of maximum discharge transmuted through the tile drain (DFTILE outlet) for 1 August 2013 is 0.01265 m³/s. Storm event for 18 April 2013 drains the entire DFTILE sub-basin faster with a lower peak discharge of 0.0197 m³/s.

### 3.3.3.2 Scenario 2: Routing Flow through an HBS pipe riser

Flow characteristics at the DFTILE outlet routed through an HBS pipe riser across all the WASCoBs in the DFTILE sub-basin are investigated in this scenario. Observed discharge (m³/s) simulated form the toolbox for the two-storm events: 1 August 2013 and 18 April 2013, are graphically represented in Figure 3.12 a, b. Visual analyses of the graph reveal that the predicted amount of maximum discharge transmitted at the DFTILE outlet for 1 August 2013 is 0.008535 m³/s with the water from all the berms being drained through the DFTILE outlet in approximately four hours.

Although the drainage pipes are not flowing to its maximum capacity, the peak flow is quite close to the maximum flow capacity of 0.02499 m³/s for the 200-mm diameter pipe connecting WASCoBs 1, 2 & 5 to the DFTILE outlet. Henceforth, the diameter of the drainage pipe for one or more of the WASCoBs could be increased for more efficient drainage. On the contrary, storm event for 18 April 2013 which has much higher precipitation, flows the DFTILE with a peak of 0.01718 m³/s, draining the entire network of WASCoBs in approximately five hours. Graphical comparisons indicate that flow characteristics at the DFTILE outlet are analogous for the two storm events with a minor difference in the magnitude of flow routed.
Figure 3. 13: Flow simulated at the DFTILE outlet for HB pipe riser (A) 1 August 2013, (B) 18 April 2013 storm events
Figure 3.14: Flow simulated at the DFTILE outlet for HBS pipe riser (A) 1 August 2013, (B) 18 April 2013 storm events
3.3.3.3 Scenario 3: Routing Flow through an PI pipe riser

Flow is routed through PI pipe risers across all the WASCoBs to the DFTILE outlet in this scenario. Storm event for 1 August 2013 is drained to the DFTILE outlet in approximately 4 hours and thirty minutes with a peak flow of 0.01 m³/s. Graphical representation of the 18 April 2013 event reveals that the DFTILE outlet flows to a maximum of 0.01897 m³/s Figure 3.13 a, b. Henceforth, since the water ponded behind the berms is routed efficiently for these two intense storm events, the pipe risers and the diameter of the drainage pipes is deemed adequate. However, a modification in the drainage system for the sub-basin is recommended to reduce further the time the water is ponded behind the berms, thereby minimizing the adverse effect upon the crop productivity and alleviating financial losses (Ross Wilson, 2016).
Figure 3. 15: Flow simulated at the DFTILE outlet for PI pipe riser (A) 1 August 2013, (B) 18 April 2013 storm events
3.3.4 Flow at a WASCoB outlet under extreme events for different diameter of Tile Drains

Construction of hydraulic structures whose primary purpose is to manage rainfall-runoff volume requires adequate knowledge about the time distribution of rainfall (hyetograph) in the region (Powell et al., 2008). A storm event of high intensity could suddenly yield a huge quantity of runoff in a brief period. Henceforth, runoff volume produced depends upon the intensity and shape of the rainfall hyetograph (Powell et al., 2008). Consequently, a detention pond’s design conclusively depends upon the characteristics of the hyetograph. For example, if the hyetograph produces a large amount of precipitation during a short time period, a large volume of runoff has to be stored behind the detention pond. Henceforth, a fairly large sized pond is required. Alternatively, a moderately sized pond is needed if the hyetograph yields the same amount of precipitation over a longer period of time. Henceforth, while designing WASCoBs (which is technically a detention region), analysis pertaining to extreme rainfall patterns is of paramount importance. Therefore, scenario analysis is investigated where the flow is routing for extreme events for the various diameter of tile drain for WASCoB 3. In the present-day condition, a 150-mm diameter tile drain receives water routed through the surface inlet for WASCoB 3. Events with a return period of 2-year, 5-year, 10-year with a 24-hour storm (Table 3.4) duration are selected for analysis.
### Table 3.5: Precipitation for 2-yesr, 5-year, and 10-year storms for various time periods

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<thead>
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<th>Rainfall Depth (mm)</th>
<th>5-min</th>
<th>10-min</th>
<th>15-min</th>
<th>30-min</th>
<th>1-hr</th>
<th>2-hr</th>
<th>6-hr</th>
<th>12-hr</th>
<th>24-hr</th>
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<tr>
<td>2-year</td>
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<td>18.7</td>
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<tr>
<td>10-year</td>
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<td>23.4</td>
<td>28.8</td>
<td>35.5</td>
<td>43.7</td>
<td>60.9</td>
<td>75</td>
<td>92.4</td>
</tr>
</tbody>
</table>

### 3.3.4.1 Scenario 1: 2-year 24-hour Design Storm

In this scenario, a 2-year design storm based upon a 24-hour return period was utilized to simulate the effectiveness of the currently employed 150 mm diameter drainage pipe for WASCoB 3. For this exercise, tile drains of 200 mm diameter and 375 mm diameter were also considered for the scenario analysis based upon OMAFRA criteria’s (Ross Wilson, 2016) to route flow through the sub-basin respectively.

The present 150 mm diameter pipe can route water at a maximum capacity of 0.01099 m³/s. The drainage pipe flows to its optimum capacity for approximately five hours and fifteen minutes. Drainage pipe flowing to its optimum capacity for such an extended period is not recommended (Wilson, 2016). Therefore, a novice scenario is considered where the tile drain is replaced with a drainage pipe of 200 mm and 375 mm diameter drainage pipes respectively. Graphical comparison between the three pipes (Figure 3.16) reveals that contrary to the 150-mm diameter pipe, larger diameter pipes (200 and 375 mm) are potent in draining water ponded behind the berm at a much rapid rate. Further; a 375-mm drainage pipe never flowing to its optimum capacity due to its larger surface area and volume. Furthermore, a 375-mm diameter pipe is cheap and is in
the regimen of the OMAFRA guidelines (Ross Wilson, 2016) for WASCoB design. Henceforth; a 375-mm diameter pipe is recommended when designing a WASCoB (with dimensions similar to WASCoB 3) for draining a 2-year 24-hour storm event.

3.3.4.2 Scenario 2: 5-year 24-hour Design Storm

Under this scenario analysis, a 5-year 24-hour design storm was utilized to identify the effectiveness of the 150 mm (present case), 200 mm and 375 mm diameter tile drains for WASCoB 3. The 150-mm drainage pipe routes water, flowing at its optimum capacity (0.01099 m³/s) for 10 hours and 10 minutes approximately. Furthermore, there is significant overtopping of water from the berm, which is not recommended for the safety of the crops and erosion purposes. Considering the sluggish efficiency of the currently installed tile drain of 150 mm in draining water ponded behind the berm, analysis where the diameter of the outlet pipe is increased to 200 mm and 375 mm respectively is considered. The altered drainage pipe could route water at a maximum capacity of 0.02499 m³/s due to its greater volume. The 200 mm drainage pipe flows to its optimum capacity only for approximately three hours and forty-five minutes respectively. Henceforth; although the modified pipe (200-mm) is more efficient than the currently installed pipe of 150-mm diameter in draining the water ponded behind the berm, it is not efficient enough to route water without it overtopping the berm (Figure 3.17).
Figure 3.16: Flow simulated through WASCoB 3 for 2-year 24-hour storm event routed through a (A) 150 mm, (B) 200 mm, and (C) 375 mm drainage pipes.
Figure 3. 17: Flow simulated through WASCoB 3 for 5-year 24-hour storm event routed through a (A) 150 mm, (B) 200 mm, and (C) 375 mm drainage pipes.
Henceforth another scenario of employing a 375-mm drainage pipe is considered. Graphical comparison between the three pipes (Figure 3.17) reveals that the 375-mm drainage pipe is potent in draining water ponded behind the berm at a much rapid rate without it never flowing to its optimum capacity. Henceforth; a 375-mm diameter pipe is recommended when designing a WASCoB (with dimensions similar to WASCoB 3) for draining a 5-year 24-hour storm event.

### 3.3.4.3 Scenario 2: 10-year 24-hour Design Storm

In this scenario, a 10-year design storm based upon a 24-hour return period was utilized to simulate the effectiveness of the currently employed 200 mm diameter drainage pipe for WASCoB 3. Drainage pipes of three distinct diameters: 150 mm (present case), 200 mm and 375 mm are considered in this scenario analysis (Figure 3.18). Similar to the 5-year-24 hr design storm the 150 mm drainage pipe drains water ponded behind the berm with an optimum capacity of 0.01099 m³/s. The drainage pattern is similar to the 5-year-24 hr design storm with water overtopping the berm for a considerable period of time.

When the drainage system is replaced by a 200-mm diameter tile drain water is drained at a faster rate of 0.02499 m³/s. Furthermore, the pipe flows to its maximum capacity for four hours and 35 minutes approximately. Another scenario where drainage pipe of 375- mm is also considered. The modified drainage system is the most effective. The tile drain never flows to its maximum capacity. Moreover, the can drain the entire water in approximately six hours. Therefore; for a 10-year 24 -hour design storm a 375-mm diameter pipe is recommended when designing a WASCoB with dimensions similar to WASCoB 3.
Figure 3. 18: Flow simulate through WASCoB 3 for 10-year 24-hour storm event routed through a (A) 150 mm, (B) 200 mm, and (C) 375 mm drainage pipes.
3.3.4.4 Routing of Flow from DFTILE outlet

For this study, water was routed from the DFTILE outlet to the outlet of the entire Gully creek watershed (GULGUL 5). This was done to study a realistic water routing scenario where WASCoB/WASCoBs are directly routed to streams. In this investigation, a 200-mm diameter drainage pipe with 1% slope is assumed connecting the DFTILE outlet to the GULGUL 5 outlet. Utilizing the pipe diameter and the slope gradient as per the OMAFRA guidelines a lag time of 70 minutes is procured to route water from the DFTILE to GULGUL 5 outlet.

Graphical comparison of the hydrographs at the watershed outlet (GULGUL 5) with and without lag reveals the hydrological impact of directly routing water from a sub-area of the watershed to a stream (watershed outlet in this case). The area of the DFTILE sub-basin is approximately 2% of the area of the Gully creek watershed contributing a very small percentage of runoff volume for the storm events considered in this scenario analysis. Henceforth, directly routing water from the DFTILE outlet to the GULGUL 5 outlet does not have a significant impact upon the peak flow of the hydrograph. However, the recession limp of the hydrograph is impacted due to the routed hydrograph, and a slight increase is observed (Figure 3.19).
3.4 Summary and Conclusions

This study presents a new toolbox for simulation of WASCoBs through the aid of the AGNPS model. The toolbox utilizes the inputs from AGNPS, followed by the launching of the application for execution of the CoBAGNPS toolbox. Finally, the output files are generated.

In this study, various combinations and permutations of pipe risers and tile drains are considered. The performance of several pipe risers [(HB), (HBS), and (PI)] upon flow characteristics was evaluated. Initially, the toolbox is simulated for a single WASCoB (WASCoB 3). Results of visual interpretation show that the toolbox is efficient in routing flow through pipe risers and the drainage system. Furthermore, upon successful execution of the toolbox for a single WASCoB, flow through the network of WASCoBs...
and tile drains was routed to the DFTILE outlet. Peak flow and the time to drain the network of WASCoBs for the drainage system (DFTILE outlet) varied slightly, depending upon the storm events and the pipe risers selected. This finding is substantial because it suggests that, no matter how which pipe riser is selected, it only changes the peak flow at the DFTILE miniscually.

Another important practical scenario simulated in this study was to design an efficient drainage system capable of draining extreme events adequately using the toolbox developed. The efficiency of the drainage system was investigated under extreme events (2-year, 5-year, and 10-years with a 24-hour) for a single WASCoB (WASCoB 3). We found that presently installed drainage pipe of 150-mm diameter does not drain a 2-year, 5-year, and 10-year storm events in a timely manner. Replacing the tile drains with a pipe of 200 and 375-mm diameter pipes would significantly enhance the efficiency of the drainage system and reduce the time of ponding of water behind the berms. However, a 375-mm diameter pipe would be the most efficient to drain the WASCoB. Finally, the last and most important component of the toolbox was tested, where the flow routed through one of the DFTILE outlets was directly routed to the outlet of the Gully creek watershed (GULGUL 5) through a 200-mm diameter drainage pipe with 1 % slope, assuming a lag time. The graphical comparison demonstrates that impact of directly routing water from a sub-basin of the watershed to a stream (watershed outlet in this case) impact the recession limb of the original hydrograph but only slightly.

In the present study, only flow is routed through the WASCoBs; future research should focus on the impact of the different type of pipe risers on the nutrient and sediment transport through the WASCoBs. Further, the efficiency of WASCoBs for trapping
sediments and phosphorous within the ponded area of the WASCoB could be evaluated for individual storm events. Furthermore, since berm area of the WASCoB, type of pipe riser and the drainage system significantly impact the hydrology of the sub-basin, research needs to be conducted to determine if altering the berm area of the WASCoB in combination with pipe risers have an impact upon the flow and nutrients (sediment and phosphorous) transported at the outlet of the sub-basin. Also, the impact of directly routing larger sub-basins (which generates substantial runoff) directly to the outlet of the Gully creek watershed should be studied.
**Transition to Chapter 4**

Water and Sediment Control Basin (WASCoB) is an important BMP constructed along concentrated flow-paths (gullies etc.) to control the movement of water and sediment within a watershed. A WASCoB constitutes of a berm, surface inlets, and a drainage pipe to route water into a ditch. Direct runoff ponded behind the berm is routed through surface inlets into an underground drainage pipe. Therefore, surface inlets are an exceedingly important constituent of a WASCoB. Further pipe risers and blind inlets are the two most common type of surface inlets used. Therefore, maximum sediment removal efficiency of WASCoBs at a watershed-scale can be attained by the appropriate selection of a surface inlet, since the efficiency of a WASCoB is greatly impacted by the quantity of runoff and sediment leaving the surface inlet. In this study a toolbox was developed viz., CoBAGNPS to compute the sediment removal efficiency of pipe risers and blind inlets. A watershed-scale model (AGNPS) was integrated within the toolbox. Output files of the AGNPS model are fed as input files into the toolbox where a sediment routing module is programmed separately for pipe risers and blind inlets to obtain the sediment removal efficiency for each type of surface inlet. Further, the sediment routing module programmed for blind inlets integrates the AGNPS model with the HYDRUS 1-D model. The toolbox developed was applied to the Gully Creek watershed in Ontario, and the sediment load routed through pipe risers and blind inlets were compared. This paper has been published in the peer reviewed Journal of Environment and Natural Resources Research.

Chapter 4

4 CoBAGNPS: A Toolbox to Estimate Sediment Removal Efficiency of WASCoBs–Pipe Risers and Blind Inlets

4.1 Introduction

Non-point source (NPS) pollution culminating due to agricultural practices extensively pollutes water bodies globally (Das et al., 2006; Babin et al., 2016; Fraga et al., 2016; Fu et al., 2012; Malawska, 2008; Wang et al., 2016). Sediment is a significant NPS pollutant (Gharabaghi et al., 2006; Sebti & Rudra, 2010). Transport of NPS pollutants over the land surface from agriculturally prevailing regions eventuates essentially by three primary mechanisms: rills, ephemeral gullies, and classical gullies (Daggupati et al., 2013). Therefore; precisely locating these gullies, followed by implementation of appropriate BMP is cardinal for the abatement of NPS pollution (Daggupati et al., 2011, 2013). BMPs could be constructed in a watershed either randomly or by strategic targeting (Tomer et al., 2010; Daggupati et al., 2011). Several studies have revealed that constructing BMPs randomly is not ideal for obtaining their maximum efficiency in mitigating NPS pollutants (Dillaha et al., 1988; Sprague & Gronberg 2012; Tomer et al., 2010). Therefore, strategically identifying and prioritization of regions, followed by implementation of adequate BMPs is a crucial step for an effective
watershed management (Diebel et al., 2008; Daggupati et al., 2011; Daggupati et al., 2013).

Water and Sediment Control Basin (WASCoB) is one such BMP designed specifically for impeding the development of concentrated gullies (Fiener et al., 2005; Her et al., 2017; Kovacic et al., 2006; Verstraeten & Poesen, 2001). WASCoBs are constructed with an aim to alleviate transport of sediment and nutrients generated within farmlands; thereby impeding the formation of gullies by interrupting a watercourse (Her et al., 2016). Also, WASCoBs are normally constructed in agricultural watersheds. Berms (ponded area), surface inlets, and drainage system combined are commonly referred to as a WASCoB (Figure 4.1).

WASCoBs slowly divert surface runoff ponded behind the berm through surface inlets into an underground drainage system, thereby regulating flow and trapping sediments and nutrients (P and N) within the berm (Fiener et al., 2005; Kovacic et al., 2006; NRCS code 638). Henceforth surface inlets which route water into the drainage system are an exceedingly essential constituent of a WASCoB. Surface inlets are constructed at the point of lowest elevation within the berm and connected directly to the subsurface drainage system (usually tile drains) (Ayars & Evans, 2015). Based upon their design and function surface inlets are categorized into four types; pipe risers, open inlets, rock inlets, and blind inlets (Li et al., 2017). Amongst them, pipe risers and blind inlets are the most commonly used surface inlets. A pipe riser is a hollow cylindrical tube, with holes encompassing its circumference (Figure 4.1). Also, perforated pipe risers (also referred to as tile risers) are the most widely adopted type of surface inlets for drainage systems (King et al., 2015; Li et al., 2017; Oolman & Wilson, 2003).
Figure 4.1: Location of (a) Gully creek watershed and the DFTILE sub-basin in Ontario, Canada, (b) a typical pipe riser, (c) Pipe riser consisting of a standpipe connected to a subsurface drain

Whereas blind inlet is a permeable area with sections of gravel, sand, and limestone usually constructed at the point of minimum elevation in the berm designed to minimize transport of sediment into the drainage pipe (Gonzalez et al., 2016; Smith & Livingston, 2013). Since surface inlets are directly coupled with the drainage pipe, they provide an uninterrupted pathway for transportation of runoff and sediment loads (Smith et al., 2008). King et al. (2015) analyzed several studied pertaining to surface inlets and asserted that pollutants could be transported through traditional type of surface inlets. Tomer et al. (2010) studied an Iowa watershed and concluded that discharge from tile risers constituted 15% of the total flow. Ginting et al. (2000) reported that in a Minnesota
watershed, surface inlets transported less than 5% of annual precipitation into the subsurface drainage system between 1995 and 1998. Authors further reported that dissolved pollutants were transported primarily with snowmelt, and particulate pollutants with severe storm-events. Although pipe risers are the most commonly used surface inlets for draining huge potholes (Smith & Livingston, 2013), they significantly accelerate the transport of sediment and other pollutants.

On the contrary, blind inlets, a relatively modern and state-of-art surface inlet having an underground drainage system (NRCS Code 620), have demonstrated a significant reduction in the transport of sediment and nutrient losses (Feyereisen et al., 2015; Smith & Livingston, 2013; Smith et al., 2015). Hence, it could be a possible alternative to pipe riser for sediment and nutrient. Although potentially more effective in reducing sediment than pipe risers, blind inlets require more maintenance. Sediment trapped needs to be removed frequently culminating in high maintenance costs. Therefore, the lifespan of a blind inlet is uncertain depending upon a plethora of factors such as composition of the inlet (gravel/sand and limestone), type of soil, area of the catchment drained, and farm management practices (Li et al., 2017). Several studies conducted in the past have reported the operational life of blind inlets to be greater than 10 years with little to no complication (Feyereisen et al., 2015). However, due to difficulty in monitoring runoff and sediment losses through blind inlets, there is a paucity of published research in this regard. Smith et al. (2015) reported the difference in P concentrations routed through tile risers and blind inlets not to be statistically significant.

Further, Feyereisen et al. (2015) reported that blind inlets produced less TSS compared to open inlets during a three-year investigation in Minnesota. Gonzalez et al.
(2016) reported that blind inlets were effective in reducing transport of atrazine, 2,4-D, metolachlor, and glyphosate by 57%, 58%, 53%, and 11% respectively. In another study in Ontario, Canada it was revealed that the presence or type of surface inlet (blind inlet or pipe riser) had little impact upon P and sediment loading along with N movement through the drainage systems (Ball Coelho et al., 2012a; Ball Coelho et al., 2012b). In another instance in Ontario, Canada McKague, (2017) estimated that blind inlets (constructed with a layer of red sand) were 70% and 50% effective respectively, at removing total phosphorus and soluble reactive phosphorus.

Additional research, through sophisticated modeling techniques, is required to quantify the performance of pipe risers and blind inlets in mitigating sediment loads at a watershed scale. Further unlike field experimentation, computer models, are both labor and cost-effective (Golmohammadi et al., 2016). Nonetheless, due to the complexity in simulating flow and sediment transport through surface inlets, no study has investigated modeling flow and sediment transport through blind inlets and pipe risers at a watershed level in Ontario, Canada. Further, hydrologic conditions are an essential aspect to be examined when selecting an appropriate model. Borah et al. (2007) concluded that event-based models perform better in simulating stream flows, constituted by severe storm events. Consequently, since the transport of sediments along with surface runoff is essentially influenced by storm intensity, an event-based modeling approach is more appropriate for modeling pipe risers and blind inlets. AGNPS is one such event-based model, potent in simulating the hydrology of a watershed, routing flow and sediment between cells to the watershed outlet (Cho et al., 2008; Liu et al., 2008; Miklanek et al., 2004; Mohammed et al., 2004; Parajuli et al., 2007; Sebti & Rudra 2010). However,
AGNPS is an event-based model with no groundwater or subsurface module. Therefore, to overcome these limitations a toolbox (CoBAGNPS) was developed which could simulate the movement of flow and sediment both through pipe risers and blind inlets. A sediment module is incorporated in the toolbox to simulate the movement of sediment through pipe risers. Further, the toolbox utilizes the results from AGNPS and integrates it with HYDRUS 1-D model (Meng et al., 2014; Shang et al., 2016) to replicate the movement of flow and sediments through blind inlets. The HYDRUS 1-D model (Meng et al., 2014; Shang et al., 2016) which simulates flow through a porous media is used to duplicate the geometry of a blind inlet. Therefore, the objective of this research was to develop and exhibit through a case study the sediment routing module of the toolbox; routing sediment through pipe riser and blind inlets. The following important question and objective associated with the toolbox developed were addressed in this paper:

- Can CoBAGNPS toolbox simulate the sediment reducing efficiency of WASCoBs when direct runoff is routed through a pipe riser and blind inlets?
- Compare the sediment removal efficiency of pipe risers and blind inlets procured from the toolbox?

4.2 Method

4.2.1 Study Area

Gully Creek watershed located in Ontario, Canada along with one of its sub-basins, the DFTILE sub-basin (having WASCoBs) were selected for this study (Figure 4.1). This watershed drains into Lake Huron. The watershed encompasses an area of 2611.52 acres with elevation ranging from 281 m to 217 m at its outlet. The DFTILE sub-basin comprises of six WASCoBs which drains to the DFTILE outlet through a network of tile
drains (Figure 4.1). Precipitation occurs mostly as rainfall between April and October (nearly 60 %), while remaining occurs as snow during the winter months. Average annual precipitation occurring in the region is approximately 1,055 mm (monitored between 2001-2011) (Golmohammadi et al., 2017). Detailed description regarding the geographical extent for each soil type in the watershed is provided in Table 4.1. Further, the landuse distribution of the watershed is presented in Figure 4.2. Nearly 70 % of the watershed is agricultural land dominated by corn, soybean and winter wheat. Remaining portion of the watershed is mostly under natural vegetation, including trees, shrubs, and grasses.

Figure 4. 2: (A) Landuse, (B) Soil, and (C) Digital Elevation model of the Gully Creek watershed
Table 4.1: Name and extend of each soil type in the Gully Creek watershed

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<th>Soil code</th>
<th>Soil Type</th>
<th>Area (ha)</th>
<th>Area (%)</th>
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<td>BKN</td>
<td>Brookston Clay Loam</td>
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Source: OMAFRA, Ontario Ministry of Natural Resources (OMNR), and Ausable Bayfield Conservation Authority (ABCA).

4.2.2 CoBAGNPS Development

CoBAGNPS is written in C# (www.learncs.org) which is a state-of-art high-level programming language. The design of CoBAGNPS is divided into two parts: (1) CoBAGNPS flow, which consists of the code for routing the flow computed at the outlet cell of the sub-basin cell (representing the WASCoB or berm) procured from the AGNPS model through a combination of pipe risers and drainage pipes and (2) CoBAGNPS sediments, which contains code for routing sediment procured at the outlet cell of the sub-basin through pipe riser and blind inlet. CoBAGNPS can be installed as a stand-alone application. The conceptual architecture of the toolbox is depicted in Figure 4.3.

4.2.2.1 Conceptual Design: Routing of Sediment: Pipe Risers

WASCoBs are constructed along the pathway of concentrated flow-paths like ephemeral and classical gullies within a watershed (Liu, 2013). Also, water flowing through the berm of the WASCoB branches from the drainage area above it. Therefore, a detailed LiDAR DEM data was utilized to procure a stage-volume (storage) relationships for the ponding area behind each berm in the DFTILE sub-basin (Figure 4.4). AGNPS model is used to estimate runoff volume and SL generated behind each berm.
Further; the stage-storage relationship combined with the stage-discharge relationship for pipe risers (Wilson, 2016) is utilized to determine the discharge from the WASCoB into the drainage pipe. To route sediment amassed behind the berm through the pipe riser, a plug-in sediment routing module was programmed in the toolbox.
Figure 4. 4: Stage-Storage relationship for all six berms

This conceptualization forms the basis of the sediment module for pipe risers and consists of the following main steps:

- The AGNPS cell delineation was performed such that the drainage area of each WASCoB is represented as a sub-basin and the surface inlet location is represented as the outlet of that sub-basin.
- AGNPS model simulates flow and SL generated at the outlet cell of a WASCoB.
- Further, the AGNPS model provides the SL for the outlet cell of the WASCoB subdivided into five classes:
  1. Large Aggregates
  2. Small Aggregates
3. Sand
4. Silt
5. clay
6. Total

- The flow and sediment load thus generated is routed through a pipe riser into the drainage pipe.
- “Plug-flow” method (Wilson & Barfield 1984) is used to route SL through the pipe riser into the tile drain.

4.2.2.2 Conceptual Design: Routing of Sediment-Blind Inlets

In order to estimate the performance of blind inlets at a watershed scale in removing SL, this study concentrated upon coupling a model which could replicate blind inlets with a watershed scale model. AGNPS model (Young et al., 1989) was used to simulate runoff and SL generated for the drainage area of the WASCoB, while HYDRUS 1-D (Šimůnek et al., 1998) was selected to replicate the blind inlets. Further, a blind inlet (4.25 m×4.25 m× 1 m) located at the lowest elevation point of the WASCoB sub-basin is considered for this modeling exercise. Top 50 cm of the excavation constitutes of coarse-textured soil; typically sand to aid infiltration. A Geotextile fabric is placed below the soil layer. Further, coarse limestone/gravel is placed below the fabric up to a depth of 50 cm (Figure 4.5). The purpose of geotextile fabric placed between limestone gravel and sand is to prevent sedimentation within the limestone layer. Further, it reduces the hydraulic efficiency of the layer, thereby preventing drainage.
Figure 4.5: Schematic diagram of a blind inlet

Detailed description of the blind inlet simulated in this study (for scenario analysis) is presented in Smith & Livingston, (2013). HYDRUS 1-D is used to model infiltration of water ponded behind the berm (computed from AGNPS) through the sand and the limestone layer of the blind inlet. However, HYDRUS 1-D does not simulate the movement of sediment within porous media. To overcome this limitation, an equation (Equation 4.1) has been proposed to route sediment ponded behind the berm through the sand and the limestone layer of the blind inlet (Neitsch et al., 2002).

\[
\text{Sediment Mass}_{\text{Blind Inlet}} = \frac{K \times \text{Area}_{\text{Blind Inlet}} \times \text{Concentration}_{\text{Clay+silt}} \times Q_{\text{Blind Inlet}}}{1000}
\]  

(4.1)

\(K\): Geotextile Filtering coefficient
Area_{Blind\,\text{Inlet}}: Area\,of\,the\,blind\,inlet\,(m^2).

Concentration_{Clay+Silt}: Concentration\,of\,clay\,and\,silt\,in\,the\,sediment\,(kg/m^3)

Q_{Blind\,\text{Inlet}}: Flow\,of\,water\,through\,the\,Blind\,Inlet\,into\,the\,tile\,drain.

This conceptualization forms the basis of the blind inlet module in the toolbox and consists of the following main steps:

- Cell delineation was executed such that the drainage area for each WASCoB is represented as a sub-basin and the blind inlet location as the outlet of that sub-basin.

- Thereafter, the total quantity of runoff and SL generated behind the berm for each event is computed using the AGNPS model (The AGNPS model is executed within the toolbox).

Further, the hydrology data (flow and sediment) for the outlet cell of each WASCoB sub-basin is procured from the output file of the AGNPS model. The geometry of the blind inlet is modeled by the HYDRUS 1-D model. Finally, the movement of water and sediment through the blind inlet is simulated using the HYDRUS 1-D model and equation 4.1 programmed in the blind inlet module of the toolbox.

4.2.3 AGNPS Model

AGNPS model (Young et al., 1989) is used to simulate the Gully Creek, Ontario, Canada watershed in this study. AGNPS is a physically-based, event simulation, watershed model. The entire watershed is divided into discrete square cells. The hydrology component of the model computes runoff volume for each cell, using the curve
number approach (McCuen, 1982), and peak flow using an empirical equation (Knisel, 1980). Further, the erosion component of the model computes sediment load for each cell using the Bagnold stream power equation (Bagnold, 1966). Comprehensive information pertaining to the model can be procured from the AGNPS manual (Young et al. 1989; http://www.waterbase.org/docs/MWAGNPS%20Setup.pdf).

4.2.4 HYDRUS 1-D Model

In this study, the HYDRUS-1D code (Šimůnek et al., 1998) is used to simulate 1-D movement of water through a blind inlet. One-dimensional uniform (equilibrium) water movement in partially saturated rigid porous medium is described by modified form of the Richards equation using the assumptions that the air phase plays an insignificant role in the liquid flow process and that water flow due to thermal gradients can be neglected:

\[
\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial x} \left[ K \left( \frac{\partial h}{\partial x} + \cos \alpha \right) \right] - S
\]  

Where

H: water pressure head

\( \theta \): Volumetric water content

t: Time

x: spatial coordinate (upwards)

S: sink term

\( \alpha \): angle between flow direction and vertical axis

K: unsaturated hydraulic conductivity function
4.2.5 Input Data

DEM (5-m resolution), soil, land use, precipitation data, and stream network is the input data required to model the watershed with the AGNPS model. The required dataset is procured from several sources such as Ontario Ministry of Agriculture, Food, and Rural Affairs (OMAFRA), Ontario Ministry of Natural Resources (OMNR), and the Ausable Bayfield Conservation Authority (ABCA).

4.2.6 Base Flow Separation

AGNPS being an event-based model computes only surface runoff. Henceforth; baseflow was separated from streamflow at the outlet of the Gully Creek watershed (GULGUL 5). Surface runoff and base flow compound into streamflow. The former’s contribution to streamflow is brisk, on the contrary, base flow’s contribution to streamflow is slow (Kalin & Hantush, 2006). The WHAT program (https://engineering.purdue.edu/mapserve/WHAT/) was used to separate base flow from observed flow (Srivastava et al., 2010; Gupta et al., 2018) based upon signal analysis and processing (Eckhardt & Arnold, 2001; Kyoung et al., 2005; Lyne & Hollick, 1979).

4.2.7 Sediment Load Estimation

Observed SL was available for only a few grab samples. Henceforth; sediment loads for these grab samples need to be extrapolated to obtain observed sediment load for each event. Thereafter, observed sediment load for each event was compared with AGNPS simulated sediment load for that event. To extrapolate sediment load of the grab samples on an event basis, LOAD ESTimator (LOADEST) a web-based tool (https://engineering.purdue.edu/mapserve/LOADEST/) was engaged (Park et al., 2015;
Runkel et al., 2004). The tool estimates monthly SL using observed streamflow, observed sediment concentration data (for grab samples), and regression model coefficients.

### 4.2.8 Model Calibration and Validation

Similar to most distributed watershed models (Dile et al., 2016), AGNPS also has a few empirical parameters. Certain variables in AGNPS (CN2, K, C, and P), are not fixed physically (Cho et al., 2008; Choi & Blood, 1999; Liu et al., 2008). Therefore, the model is calibrated and validated against observed data. In this study, calibration was performed separately for peak flow, direct runoff and sediment yield for nine storm events between June 2012 and August 2013. Further, validation was performed using nine storm events between August 2013 and June 2015. Seasonal calibration and validation were also performed for peak flow, direct runoff, and sediment yield. Observed flow data were obtained from the Ausable Bayfield Conservation Authority (ABCA) at the watershed outlet. Calibration process was accomplished by changing AGNPS parameters to match the model-predicted peak flow, direct runoff and sediment yield with its counterpart. The parameters calibrated were the curve number (CN2) manning’s n, C, and K, which have considerable influence upon peak flow, surface runoff, and sediment yield. CN is altered based upon antecedent moisture conditions. Statistical parameters used to evaluate model’s performance are elaborated in Table 4.2.
Table 4.2: Statistical parameters used to assess the model’s performance

<table>
<thead>
<tr>
<th>Statistical Parameter</th>
<th>Formula</th>
</tr>
</thead>
<tbody>
<tr>
<td>$R^2$</td>
<td>$\frac{\sum_{i=1}^{N}(O_{obs,i} - \bar{O}<em>{obs})(O</em>{sim,i} - \bar{O})}{\sum_{i=1}^{N}(O_{obs,i} - \bar{O}<em>{obs})^2}^{0.5} \left[ \frac{\sum</em>{i=1}^{N}(O_{sim,i} - \bar{O}<em>{sim})^2}{\sum</em>{i=1}^{N}(O_{sim,i} - \bar{O}_{sim})^2} \right]^{0.5}$</td>
</tr>
<tr>
<td>$E_{NS}$</td>
<td>$1.0 - \left[ \frac{\sum_{i=1}^{N}(O_{sim,i} - O_{obs,i})^2}{\sum_{i=1}^{N}(O_{obs,i} - \bar{O}_{obs,i})^2} \right]$</td>
</tr>
</tbody>
</table>

Where
- $O_{sim,i} =$ Simulated flows
- $O_{obs,i} =$ Observed flows
- $N =$ Number of observations
- $\bar{O}_{obs} =$ Mean observed flow
- $\bar{O}_{sim} =$ Mean simulated flow

4.2.9 Routing Sediment: Pipe Riser and Blind Inlet

For this study, a set of scenario analyses were simulated using the CoBAGNPS toolbox developed. Statistically calibrated and validated AGNPS results were used to compute the sediment reducing efficiency of pipe risers and blind inlets through the toolbox. Furthermore, sediment removal efficiency was evaluated for three different types of pipe risers (HB, HBS, and PI). A couple of storm events (18 April 2013 and 1 August 2013) were selected to perform scenario’s pertaining to pipe risers. Further, a currently installed drainage pipe of 150 mm diameter is considered for simulation. Under the first set of scenario analysis, the performance of the different type of pipe risers (HB, HBS, and PI) in routing sediment load from the berm into the tile drains was analyzed for WASCoB 3 for the selected storm events. Additionally, the sediment routing efficiency of the HB pipe risers was investigated for two synthetic design storm events (5 and 10-year, 24-hour) for various diameters of drainage pipes. In another scenario analysis, the impact of blind inlet upon SL routed into the drainage pipe for WASCoB 3 was assessed.
Sediment constantly gets trapped within the blind inlet reducing its efficiency and lifespan (Li et al., 2017). Therefore, analysis pertaining to extreme storm events was not investigated for blind inlets. Storm events with moderate intensity (18 April 2013 and 1 August 2013) were used to test the performance of the blind inlet module of the toolbox.

4.3 Results

4.3.1 Calibration and Validation of Flow and Sediments at the GULGUL 5 Outlet

In this study, data for individual seasons were utilized to calibrate and validate the model on a seasonal basis. A calibration method based upon seasonal calibration is proposed for calibration and validation of storm events. Six storm events were considered for spring, summer and fall season. Further, the events were bifurcated into three events for calibration and validation respectively. Model parameters were adjusted for the calibration events for each season. Statistical parameters were however computed for total calibration and validation events respectively, compounded for all the seasons.

4.3.1.1 Flow: Seasonal Calibration

Seasonal calibration was performed for peak flow and direct runoff at the watershed outlet (GULGUL 5 station). Table 4.3 summarizes the events selected for calibration and validation for fall, spring and summer respectively. Overall the model performed well. For peak flow, \( R^2 \) and ENS values of 0.97 and 0.89 for calibration events and 0.91 and 0.83 for validation events were obtained. Also, \( R^2 \) and ENS values of 0.87 and 0.73 respectively for calibration events and 0.91 and 0.46 for validation events were procured for direct runoff.
Table 4.3: Events selected for calibration and validation based upon different seasons [surface runoff (OSR, SSR) and peak flow]

<table>
<thead>
<tr>
<th>Season</th>
<th>Date</th>
<th>Precipitation (mm)</th>
<th>Duration (hours)</th>
<th>AMC</th>
<th>manning’s (n)</th>
<th>$Q_{\text{Peak}}^{\text{Observed}}$ m³/s</th>
<th>$Q_{\text{Peak}}^{\text{Simulated}}$ m³/s</th>
<th>OSR(m³)</th>
<th>SSR(m³)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fall (Cal)</td>
<td>14-Oct-12</td>
<td>18.4</td>
<td>20</td>
<td>II</td>
<td>0.02</td>
<td>0.12</td>
<td>0.2</td>
<td>6317</td>
<td>5368</td>
</tr>
<tr>
<td></td>
<td>14-Sep-12</td>
<td>17.2</td>
<td>7</td>
<td>II</td>
<td>0.02</td>
<td>0.07</td>
<td>0.13</td>
<td>792</td>
<td>2684</td>
</tr>
<tr>
<td></td>
<td>20-Sep-13</td>
<td>24.8</td>
<td>5</td>
<td>II</td>
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<td>0.34</td>
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</tr>
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<td></td>
<td></td>
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</tr>
<tr>
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<td>12-Apr-14</td>
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</tr>
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<td>25048</td>
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<td>Summer (Cal)</td>
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<td>15</td>
<td>I</td>
<td>0.04</td>
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<tr>
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<td>0.09</td>
<td>0.08</td>
<td>3500</td>
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<td>28-Jun-13</td>
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<td>II</td>
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<td>0.08</td>
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<td>12-Nov-12</td>
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<td>II</td>
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<td>6015</td>
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<td></td>
<td>0.48</td>
<td>0.46</td>
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<td>9840</td>
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<td>30-Aug-13</td>
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<td>II</td>
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<td>12-Jun-12</td>
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<td>II</td>
<td>0.02</td>
<td>0.34</td>
<td>0.28</td>
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<td>5368</td>
</tr>
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<td></td>
<td>12-Jun-13</td>
<td>19</td>
<td>3</td>
<td>II</td>
<td>0.02</td>
<td>0.24</td>
<td>0.26</td>
<td>6829</td>
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</tr>
<tr>
<td></td>
<td>Average</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.29</td>
<td>0.27</td>
<td>5213</td>
<td>5368</td>
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</table>

Average Peak Flow: 0.29
Direct runoff: 0.27
<table>
<thead>
<tr>
<th></th>
<th>R²</th>
<th>Eₜₙₜ</th>
<th></th>
<th></th>
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<tr>
<td>(Cal)</td>
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<tr>
<td></td>
<td>0.89</td>
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<tr>
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<tr>
<td></td>
<td>0.83</td>
<td>0.46</td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

AMC: Antecedent moisture condition
OSR: Observed surface runoff
SSR: Simulated surface runoff
4.3.1.2 Sediment: Seasonal Calibration

Table 4.4 shows the events selected for calibration and validation of SL during fall, spring and summer seasons respectively. As described in the section above, six events are selected for each season, where three events are used for model calibration and three events for model validation. The calibration process was completed by altering parameters within the AGNPS model to match the model-predicted SL with the observed SL. It is a prerequisite to complete calibration and validation for flow before initiating the process for sediment calibration. The parameters varied during calibration are shown in Table 4.5.

These parameters were manually manipulated for each grid and its channel until peak flow, direct runoff and SL simulated at the outlet grid of the AGNPS model closely matches the observed value for parameters simulated. Adjustment of the SCS curve number (CN2), has been found necessary for calibration and validation for surface runoff and peak flow in many AGNPS studies (Cho et al., 2008; Liu et al., 2008; Parajuli et al., 2007). Likewise, alteration of K, C, and P factors have been found necessary for calibration and validation for SL generated (Choi & Blood, 1999). The logic behind altering CN, n, K, C, and P for each grid is that once a close agreement is reached between observed and model-simulated values at the watershed outlet, the adjusted values for these model parameters would be realistic and match the real-world parameters for each grid within the watershed.
Table 4.4: Events selected for calibration and validation based upon different seasons [surface runoff (OSR, SSR) and sediment yield (OSL, SSL)]

<table>
<thead>
<tr>
<th>Season</th>
<th>Date</th>
<th>PCP (mm)</th>
<th>Duration (hours)</th>
<th>AMC</th>
<th>manning’s (n)</th>
<th>OSR (m³)</th>
<th>SSR (m³)</th>
<th>OSL (tons)</th>
<th>SSL (tons)</th>
</tr>
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<td>Fall (Cal)</td>
<td>14-Oct-12</td>
<td>18.4</td>
<td>20</td>
<td>II</td>
<td>0.02</td>
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<td>II</td>
<td>0.02</td>
<td>792</td>
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<td>0.023</td>
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<td>5</td>
<td>I</td>
<td>0.02</td>
<td>6011</td>
<td>5368</td>
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<td>Average</td>
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<td></td>
<td>4373</td>
<td>4473</td>
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<td>1.34</td>
</tr>
<tr>
<td>Spring (Cal)</td>
<td>18-Apr-13</td>
<td>30.6</td>
<td>10</td>
<td>II</td>
<td>0.02</td>
<td>37471</td>
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<td>19.49</td>
<td>16.8</td>
</tr>
<tr>
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<td>12-Apr-14</td>
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<td>7</td>
<td>II</td>
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<td>5</td>
<td>II</td>
<td>0.02</td>
<td>1068</td>
<td>536</td>
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<td>27761</td>
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<td>59</td>
<td>15</td>
<td>I</td>
<td>0.04</td>
<td>19174</td>
<td>34888</td>
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<td>8164</td>
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<td>Fall (Val)</td>
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<td>15.8</td>
<td>15</td>
<td>II</td>
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<td>5016</td>
<td>5368</td>
<td>1.29</td>
<td>1.22</td>
</tr>
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<td></td>
<td>17-Oct-13</td>
<td>18</td>
<td>12</td>
<td>II</td>
<td>0.02</td>
<td>16196</td>
<td>8051</td>
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<td></td>
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<tr>
<td></td>
<td>12-Nov-12</td>
<td>16.4</td>
<td>9</td>
<td>II</td>
<td>0.02</td>
<td>6015</td>
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<td>9076</td>
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<td>9</td>
<td>II</td>
<td>0.02</td>
<td>20830</td>
<td>13419</td>
<td>12.42</td>
<td>9.31</td>
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<td></td>
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<td>21</td>
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<td>18716</td>
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<td>20.85</td>
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<td>31-May-13</td>
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<td>9</td>
<td>III</td>
<td>0.02</td>
<td>13303</td>
<td>8051</td>
<td>2.584</td>
<td>2.61</td>
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<td></td>
<td></td>
<td>17616</td>
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<td>6.76</td>
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<tr>
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<td>18</td>
<td>3</td>
<td>II</td>
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<td>5368</td>
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</tr>
<tr>
<td></td>
<td>12-Jun-12</td>
<td>18.2</td>
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<td>II</td>
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<td>19</td>
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<td>II</td>
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<td>6829</td>
<td>5368</td>
<td>0.533</td>
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<td>5213</td>
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</tr>
<tr>
<td></td>
<td></td>
<td>Calibration (Cal)</td>
<td>$E_{NS}$</td>
<td>0.83</td>
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<td></td>
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<tr>
<td>Validation (Val)</td>
<td></td>
<td>$R^2$</td>
<td></td>
<td>0.92</td>
<td></td>
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<tr>
<td></td>
<td></td>
<td>$E_{NS}$</td>
<td></td>
<td>0.56</td>
<td></td>
<td></td>
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<td></td>
</tr>
</tbody>
</table>

AMC: Antecedent moisture condition  
OSR: Observed surface runoff  
SSR: Simulated surface runoff  
OSL: Observed sediment load  
SSL: Simulated sediment load
Henceforth, the model parameters values within the DFTILE sub-basin would represent the real-world dynamics. Therefore, it could be assumed that the hydrologic parameters applied at the outlet cell for each WASCoB would match the real-world hydrologic parameters. Hence, flow and SL procured after routing water ponded behind the berms through the blind inlet would be realistic. Instantaneous SL concentrations were available at the watershed outlet (Figure 4.1) between 2012 and 2014. This data was transformed into monthly SL using LOADEST as described in the above section. Estimated monthly SL was further converted to event-based loads depending upon the month of the event. These events were used for calibration. Also, the model was validated against observed SL in the next phase. Three events each for summer, spring, and fall season were used for SL calibration and validation respectively (see Table 4.4). The observed average SL of 6.7 tons was slightly lower than the simulated load of 8.52 tons over the calibration period for the summer season. Comparable results were observed for calibration events during the spring and fall period where again the model overestimated the observed SL (Figure 4.6).

Table 4.4 shows that, except for 14 September 2012 and 28 May 2013 events, the model was able to capture SL for most of the calibration events (2012–2014). The AGNPS

<table>
<thead>
<tr>
<th>Parameters adjusted during model calibration</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>CN</td>
<td>SCS curve number</td>
</tr>
<tr>
<td>n</td>
<td>Manning’s n</td>
</tr>
<tr>
<td>K</td>
<td>Soil erodibility factor</td>
</tr>
<tr>
<td>C</td>
<td>Crop/vegetation and management factor</td>
</tr>
<tr>
<td>P</td>
<td>Support practice factor.</td>
</tr>
</tbody>
</table>
model substantially overestimated SL for these two events. The overestimation in these 2 events may be due to the lack of observed grab samples during this period. Hence, the observed SL for these events could have been substantially under-predicted. Except for a few events, the model was able to simulate SL accurately during the calibration period. Performance statistics \( R^2 = 0.94 \) and \( E_{NS} = 0.83 \) (for all the calibration events combined)] further confirms that simulated SL adequately matched observed SL at the watershed outlet for the calibration events (see Table 4.4).

During the validation period, pertinent results were procured. The measured average SL of 1.87 tons was slightly higher than the simulated load of 1.55 tons for the validation events during the summer season. Comparable results were observed for the spring and fall period where again the model underestimated observed SL (Figure 4.6). Overall, the model underestimated the SL for the validation events (see Table 4.4).

Further, except for the 17 October 2013 event where the model substantially underestimated the SL, the model was successful in capturing SL for most of the events during the validation period (2012–2014). The underestimation in this event is due to some very large storms, which might not have been captured accurately by the weather station. The estimated rainfall recorded was 18 mm during the event (which might have been measured inaccurately). To obtain a better model performance during the validation period, SL for this event was removed. Thereafter, appropriate model statistics \( R^2 = 0.92 \) and \( E_{NS} = 0.56 \) were obtained indicated that AGNPS successfully simulated SL for the watershed (see Table 4.4).
Figure 4.6: Observed and simulated SL at the GULGUL 5 outlet (A) summer season, (B) fall season, and (C) spring season.

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4.3.2 Routing Sediment: Pipe Riser

4.3.2.1 Scenario 1: Sediment Routed Through WASCoB 3 Using Different Type of Pipe Risers

Under this investigation, the impact of the different type of pipe risers (HB, HBI, PI) upon the sediment routed into the drainage pipe for WASCoB 3 is analyzed. Sediment removal efficiency for the HB pipe riser is 94.31% for 18 April 2013 storm event, however, for 1 August 2013 storm event an increased removal efficiency of 96.14 % was procured (see Table 4.6). Further upon routing flow through the HBS pipe riser, the sediment removal efficiency increased marginally to 94.87 % for the 18 April 2013 storm event and 96.42 % for 1 August 2013 storm event (see Table 4.6). Adopting the HBS pipe riser, resulted in an increase in drainage time for the storm, thereby allowing more time for sediment to settle within the berm thereby increasing the sediment removal efficiency (Figure 4.7 and 4.8). Also, another scenario where water is routed through a PI pipe riser is also considered. Under this investigation, sediment removal efficiencies of 95.23% and 97.21 % were obtained for 18 April 2013 and 1 August 2013 storm event respectively. Henceforth results reveal that adopting a PI pipe riser is more efficient and practical in optimizing sediment removal efficiency.

Table 4.6: Sediment removal efficiency for various pipe risers

<table>
<thead>
<tr>
<th>Storm event</th>
<th>Pipe Riser</th>
<th>Inflow load (kg)</th>
<th>Outflow load</th>
<th>Sediment Removal Efficiency</th>
</tr>
</thead>
<tbody>
<tr>
<td>18-Apr-13</td>
<td>HB</td>
<td>160</td>
<td>9.1</td>
<td>94.31</td>
</tr>
<tr>
<td></td>
<td>HBS</td>
<td>160</td>
<td>8.21</td>
<td>94.87</td>
</tr>
<tr>
<td></td>
<td>PI</td>
<td>160</td>
<td>7.64</td>
<td>95.23</td>
</tr>
<tr>
<td>01-Aug-13</td>
<td>HB</td>
<td>150</td>
<td>5.79</td>
<td>96.14</td>
</tr>
<tr>
<td></td>
<td>HBS</td>
<td>150</td>
<td>5.37</td>
<td>96.42</td>
</tr>
<tr>
<td></td>
<td>PI</td>
<td>150</td>
<td>4.19</td>
<td>97.21</td>
</tr>
</tbody>
</table>
Figure 4. 7: Flow simulated at the outlet for WASCoB 3 for different type of pipe risers (A) 18 April 2013 HB pipe riser, (B) 18 April 2013 HBS pipe riser, (C) 18 April 2013 PI pipe riser
Figure 4.8: Flow simulated at the outlet for WASCoB 3 for different type of pipe risers (A) 1 August 2013 HB pipe riser, (B) 1 August 2013 HBS pipe riser, (C) 1 August 2013 PI pipe riser
4.3.2.2 Scenario 2: 5-Year 24-Hour Design Storm

Drainage pipes with a diameter of 150 mm (present case), 200 mm and 375 mm were selected to route sediment through an HB pipe riser for a 5-year 24-hour design storm. Sediment removal efficiencies of 76.06 %, 63.07 %, and 46.15 % were observed for 150 mm, 200 mm and 375 mm drainage pipes respectively. A 5-year 24-hour design storm generates a much higher quantity of SL of 260 kg. Also, the sediment removal efficiency of this design storm is lower than 18-Apr-13 and 01-Aug-13 storm events. This is expected because a heavy storm event (for example a 5-year 24-hour or 10-year 24-hour design storm) would generate a large amount of fine sediment particles (clay, silt, and sand). These finer particles are difficult to settle compared to large sediment particles like small aggregates and large aggregates and hence it is expected to have lower sediment removal efficiency as compared to real design storms. Furthermore, a discrepancy in the sediment removal efficiency is primarily due to the time for which water is ponded behind the berm. The 150-mm drainage pipe routes water, flowing at its optimum capacity (0.01099 m3/s) for 10 hours and 10 minutes approximately. Henceforth, there is sufficient time for fine sediment particles like silt and clay to settle within the berm. When the diameter of the outlet pipe is increased to 200 mm and 375 mm a substantial change in the flow pattern is observed in the routing efficiency of the WASCOb (Figure 4.9). The 200 mm drainage pipe is more efficient in draining the water ponded behind the berm. The pipe flows at its optimum capacity only for approximately three hours and forty-five minutes respectively; thereby significantly reducing the ponding time. However, with an increase in the draining efficiency, finer sediment particles (silt, clay etc.) have less time to settle within the berm; thereby significantly reducing the sediment removal
efficiency of the pipe riser (see Table 4.7). Another scenario of employing a 375-mm drainage pipe was also considered. Graphical comparison between the three pipes (Figure 4.9) reveals that the 375-mm drainage pipe is potent in draining water at a much rapid rate. Also, it is never flowing at its optimum capacity. Henceforth; since its ponding time is the least amongst the three scenarios of drainage pipe considered, sediment particles get minimum time to settle within the berm resulting in the least sediment removal efficiency.
Figure 4. 9: Flow simulated through WASCoB 3 for 5-year 24-hour storm event: routed through a (A) 150 mm (B) 200 mm and (C) 375 mm drainage pipes
4.3.2.3 Scenario 4: 10-Year 24-Hour Design Storm

A scenario for routing a 10-year 24-hour design storm through a drainage pipe of 150 mm, 200 mm and 375 mm diameter was also considered in this study. This storm produces a SL of 360 kg for WASCoB 3 drainage area. Tile drains of various diameter have sediment removal efficiency between 49.96 and 76.48 % (see Table 4.7). A 150 mm drainage pipe removes 275.36 kg of SL (see Table 4.7) and therefore, has the maximum sediment removal efficiency. This is expected because a 150 mm diameter drainage pipe takes the maximum amount of time to drain stormwater ponded (Figure 4.10). Therefore, sediment particles take a longer time to settle compared to other drainage pipes (200 and 375 mm).

Table 4. 7: Sediment removal efficiency for various design storms

<table>
<thead>
<tr>
<th>Extreme Event</th>
<th>Pipe Riser</th>
<th>Tile Drain</th>
<th>Inflow Load (Kg)</th>
<th>Outflow Load (kg)</th>
<th>Sediment Removal Efficiency</th>
</tr>
</thead>
<tbody>
<tr>
<td>5-year-24-hour</td>
<td>HB</td>
<td>150 mm</td>
<td>260</td>
<td>62.22</td>
<td>76.07</td>
</tr>
<tr>
<td></td>
<td></td>
<td>200 mm</td>
<td>260</td>
<td>96</td>
<td>63.08</td>
</tr>
<tr>
<td></td>
<td></td>
<td>375 mm</td>
<td>260</td>
<td>140</td>
<td>46.15</td>
</tr>
<tr>
<td>10-year-24-hour</td>
<td>HB</td>
<td>150 mm</td>
<td>360</td>
<td>84.64</td>
<td>76.49</td>
</tr>
<tr>
<td></td>
<td></td>
<td>200 mm</td>
<td>360</td>
<td>125.47</td>
<td>65.15</td>
</tr>
<tr>
<td></td>
<td></td>
<td>375 mm</td>
<td>360</td>
<td>180.11</td>
<td>49.97</td>
</tr>
</tbody>
</table>
Figure 4. 10: Flow simulated through WASCoB 3 for 10-year 24-hour storm event: routed through a (A) 150 mm (B) 200 mm and (C) 375 mm drainage pipes
4.3.2.4 Routing Sediment: Blind Inlets

In another scenario analysis, the sediment removal efficiency of blind inlets was investigated for 18 April 2013 and 1 August 2013 storm events. WASCoB 3 was again used for simulation. Sediment removal efficiency for the HB pipe riser was 81.25% for 18 April 2013 storm event; however, for 1 August 2013 storm event the sediment removal efficiency increased marginally to 86.67% (see Table 4.8). Sediment routing module for blind inlets programmed in the toolbox has a simple empirical approach (Equation 4.1) where it assumes heavy particles like sand, small aggregates, and large aggregates are settled within the berm. Only fine particles like silt and clay, which are difficult to settle and require a long time to settle would be in dissolved form and hence cannot be removed through blind inlet. Since the two storm events considered for analysis (18 April 2013 and 1 August 2013) generate almost the same quantity of sediment load their sediment removal efficiencies are also similar.

Table 4.8: Sediment removal efficiency for blind inlets

<table>
<thead>
<tr>
<th>Storm event</th>
<th>2013-04-18</th>
<th>2013-08-01</th>
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</thead>
<tbody>
<tr>
<td>Sand (kg)</td>
<td>10.00</td>
<td>10.00</td>
</tr>
<tr>
<td>Silt (kg)</td>
<td>10.00</td>
<td>10.00</td>
</tr>
<tr>
<td>Clay (kg)</td>
<td>20.00</td>
<td>10.00</td>
</tr>
<tr>
<td>Small Aggregates (kg)</td>
<td>80.00</td>
<td>80.00</td>
</tr>
<tr>
<td>Large Aggregates (kg)</td>
<td>40.00</td>
<td>40.00</td>
</tr>
<tr>
<td>Total Inflow load into the blind inlet (kg)</td>
<td>160.00</td>
<td>150.00</td>
</tr>
<tr>
<td>Total Outflow load into the tile drain (kg)</td>
<td>30.00</td>
<td>20.00</td>
</tr>
<tr>
<td>Removal Efficiency</td>
<td>81.25</td>
<td>86.67</td>
</tr>
</tbody>
</table>

4.4 Conclusion

Modeling sediment through surface inlets (viz. pipe risers and blind inlets), through hydrological models has not yet been documented in Canada. We developed a user-
driven stand-alone graphical user interface toolbox, called CoBAGNPS, to evaluate the sediment removal efficiency of surface inlets, which are an exceedingly important constituent of a WASCoB. This paper provides details of the toolbox tested using a case study. The AGNPS and HYDRUS 1-D model are integrated with the interface of the toolbox for routing the SL through blind inlets. Once the AGNPS model is successfully developed, calibrated, and validated for the watershed, the toolbox was executed. Three different pipe risers (HB, HBS, and PI) and various diameter of tile drains were used to simulate the sediment removal efficiency of the toolbox for 12 April 2014 and 18 April 2013 storm events. Also, the sediment removal efficiency for two extreme events (5-year and 10-year with a 24-hour) for one WASCoB was evaluated. It must be stressed that the toolbox simply utilizes the outlet results of the AGNPS model and feeds it as an input file into the HYDRUS 1-D model for routing sediment through blind inlets. Results revealed that pipe risers were more effective in detaining sediment load within the berm. It should be noted that the sediment routing module programmed into the toolbox is based upon a simple empirical equation, where fine sediment particles like silt and clay are not settled at all within the berm thereby yielding lower sediment removal efficiency for blind inlet compared to pipe risers. This is a limitation of this study. Also, the results obtained from the toolbox need to be validated with real experimental data collected to assess the accuracy of the toolbox.
Transition to Chapter 5

Hydrologic models are calibrated and validated with an existing drainage network/drainage pattern (DNDP). However, in present times water could be routed through alternative DNDPs. The main objective of this paper was to explore the performance of KINEROS 2 model in predicting streamflow and sediment yield in response to alterations in DNDP. Adopting the existing DNDP as an input, the model was calibrated for three events (18 April 2013, 12 June 2012, and 12 June 2013) and validated for two events (12 April 2014, and 30 August 2013) for flow at the watershed outlet. Further, the model was calibrated for eight events and validated for seven events for sediment content at the watershed outlet. Thereafter, the model was driven with a modified DNDP, and its response upon peak flow, direct runoff and sediment yield were investigated for two events (12 April 2014 and 18 April 2013) and a synthetic design storm (2-year-24 hour) at a sub-basin outlet (GUL_RSD). Three DNDPs: DNDP_M (road-side ditches with the same Manning’s n), DNDP_MV (road-side ditches lined with medium vegetation), and DNDP_HV (road-side ditches lined with thick vegetation) were considered. KINEROS 2 results revealed that peak flow, direct runoff, and sediment yield increased by 47.36 %, 31.39 %, and 26.96 % respectively for 12 April 2014 event for DNDP_M. Similar results were obtained for 18 April 2013 and synthetic design storm events. However, when road-side ditches were lined with a thicker vegetation (DNDP_MV and DNDP_HV), a reduction in peak flow, direct runoff, and sediment yield was observed. This paper has been published in the peer reviewed journal Canadian Biosystems Engineering.
Chapter 5

5 Predicting the Impact of Drainage Ditches upon Hydrology and Sediment Loads Using KINEROS 2 Model: A Case Study in Ontario

5.1 Introduction

Agricultural runoff and subsurface drain effluents have a momentous impact upon the hydrology and water quality of a watershed (Sloan et al. 2017; Moriasi et al. 2012). Therefore, an appropriate understanding of the direct water routing scheme, across the watershed is of pivotal importance. Moreover, state-of-the-art agricultural drainage techniques have extensively modified the landscape topography at the watersheds-scale through the development of artificial canal networks (Soana et al. 2017). These modest watercourses, whether natural or artificial, act as interfaces amidst agricultural lands and draining rivers and streams (Soana et al. 2017). Such canals are symbolized by numerous interaction amidst water, sediment and vegetation (Marion et al. 2014; Pinay et al. 2015). These watercourses are sometimes also referred to as “agricultural drainage ditches”. Therefore, agricultural drainage ditches act as a water management practice employed to commute water (direct runoff and tile-fed drainage effluents) from agricultural fields to receiving ditches (Ahiablame et al. 2010; Smith and Pappas, 2007). Agricultural drainage ditches essentially act as 1st or 2nd order streams which provide a linkage amidst
agricultural farmlands and draining water bodies (Ahiablame et al. 2010; Smith and Pappas, 2007). In the 21st century, issues concerning the transport of nutrients through agricultural drainage ditches to receiving ecosystems have been addressed by a plethora of researchers (Gentry et al. 2007; Kleinman et al. 2007; Smith et al. 2005; Strock et al. 2007).

For instance, Alexander (Alexander et al., 2008) asserted that rural watersheds in midwest U.S are dominant cause for nitrogen (N) and phosphorous (P) transport in the Gulf of Mexico. Ahiablame et al. (2010; 2011) reported nutrient uptake by benthic sediments in managed agricultural drainage ditches and later they found its potential impact on the quality of downstream waters in Indiana region. In another study in Italy, (Soana et al. 2017) studied the nitrogen mitigating efficiency of ditches nourished by spring water which is polluted by $NO_3^-$ with and without emergent vegetation. Further, (Iseyemi et al., 2016) investigated the nutrient removal efficiency of mowed and unmowed agricultural drainage ditches during an experimental study. The authors reported no significant difference between the nutrient removal capacity for the two treatments.

Additionally, based upon their construction scheme and location, agricultural drainage ditches could also be classified into road-side ditches (RSD) if directly feeding runoff effluents (tile flow and runoff) collected from agricultural fields to downstream watercourses. Also, RSD are always constructed along the periphery of roads. Therefore, any alteration in the DNDP where water is routed through RSD is expected to increase the quantity of water and sediment routed downstream of a watershed since it provides a short pathway for routing water. Water which would be have been routed through elongated gullies and creeks under normal drainage pattern would now be routed directly
through short RSD to downstream watercourses. Therefore, increase in peak flow and runoff volume is expected under the new drainage pattern. While many studies have investigated impacts of agricultural drainage ditches on hydrology and water quality, the process is understudied for RSD (Birgand et al. 2007; Cuo et al. 2006; Needelman et al. 2007). RSD analyzed in previous studies have revealed them to be a conduit for E. coli and sediments under both agricultural and forested settings (Falbo et al. 2013). Smith (2009) found phosphorus uptake lengths between 40 and 1900 m within agricultural drainage ditches in Northeast Indiana. These values generally increased with watershed size for a single drainage ditch due to SP uptakes proportion. They also exposed SP uptake rates (U) to be greater at smaller sites. However, due to difficulty in monitoring runoff and sediment losses through RSD, there is a paucity of published research in this topic. Additionally, field scrutinization in agriculture, through experimentation, is largely empirical and site specific (Ma et al. 2001). On the contrary, computer models (viz., SWATDRAIN, SWAT, AGNPS etc.), could be employed efficiently and cost-effectively for simulating the biogeochemical processes of agro-ecosystems at different scales (Golmohammadi et al. 2016). However, due to the complexity in simulating flow and sediments through RSD, few researchers have tried to model the influence of RSD in transmuting flow and sediment transport through the aid of hydrological models. Buchanan et al. (2013) used the SDDH-VSA (Buchanan et al. 2011) model and concluded
that RSD substantially modifies the watershed morphometry and natural flow pathways, thereby expediting the transport of agricultural pollutants, that would have otherwise mitigated considerably under natural degradation processes.

In Canada, (Surfleet et al. 2010) used the DHSVM model along with the generalized likelihood uncertainty estimation procedure in two streams and 11 RSD. The sensitivity of parameters and the range of the sensitivity varied across simulations for flow in the ditches and stream. Further, water routed through man-made agricultural drainage ditches or RSD significantly alters the DNDP of a watershed. Therefore, investigating the impacts of DNDP changes on hydrology or water quality, which are either projections or purely scenario analysis, need to be assessed through watershed modeling. Hydrologic conditions are of paramount importance while setting up a model. For example, Borah and Bera (2004) stated that a continuous modeling approach is less accurate for modeling stream flows, which are characterized by impulsive storm events.

Therefore event-based models would perform better in simulating storm events, including storms that generate runoff (Borah et al. 2007). Additionally, sediment and nutrient transported along with direct runoff are predominantly influenced by the intensity of the storm. Henceforth; event-based modeling is deemed more appropriate for simulating RSD as compared to a continuous approach. KINEROS 2 is an event-based, distributed and dynamic model that predicts direct runoff, erosion losses, infiltration amount, and interception depth from the catchments, produced by predominantly overland flow (Smith et al. 1999; Semmens et al. 2007). Also, the model operates on a cascade modeling approach. Given the robustness of the model, it could be used for simulating flow through RSD. Therefore, the objective of this paper is to evaluate the
performance of the KINEROS 2 model in predicting impacts of DNDP changes upon water quantity and sediment yield in a watershed in Ontario, Canada, specifically if water is routed through an alternate drainage ditch like RSD.

The following two important questions associated with RSD modeling with KINEROS 2 model were addressed in this paper:

• What would be the impact of a routing flow through RSD upon downstream flow and sediment yield?

• What would be the impact of lining the RSD with different vegetation upon downstream flow and sediment yield?

5.2 Materials and Methods

5.2.1 KINEROS 2 Model

KINEROS 2 (K2) model is an advanced version of KINEROS model (Woolhiser et al. 1990). K2 is a physically based, distributed model operating on an event basis capable of simulating direct runoff, erosion, infiltration, and interception depth from watersheds (Smith et al. 1999). Also, watersheds and their sub-basins are represented by cascades of overland flow planes (OFP), channels and impoundments in the model. OFP can be split into several constituents with distinct slopes, roughness, soils, etc. Further, contiguous planes can have different widths in the model (Semmens et al. 2007). Moreover, the K2 intercalates micro-topography in the simulation. K2 may be used to determine the effects of various artificial features, such as urban development, small detention reservoirs, or lined channels on flood hydrographs and sediment yield. Upon
rainfall rate’s exceedance of the infiltration capacity, overland flow is initiated. K2 could be used to analyze the impact of several design structures and artificial scenario’s, such as small reservoirs, urban development, or lining channels with different vegetation upon flood hydrographs and sedimentographs. Upon rainfall rate’s exceedance of the infiltration capacity, overland flow is initiated. Further, the K2 model assumes one-dimensional flow in each plane and solves the continuity equation with a kinematic wave approximation to compute flow in planes and channels as shown in equation (5.1).

\[
\frac{\partial h}{\partial t} + \alpha m^{m-1} \frac{\partial h}{\partial x} = q_L(x, t) \tag{5.1}
\]

where

\[h = \text{flow depth,}\]

\[t = \text{time,}\]

\[x = \text{the distance along the slope direction,}\]

\[q_L\] is the lateral inflow rate, and \(\alpha\) and \(m\) are parameters related to slope, direct roughness, and flow regime. Unit flow discharge (q) is correlated to flow depth with the expression \(Q = \alpha h^m\).

Also, the mass balance equation is used to compute sediment transport in planes and channels (Equation 5.2).

\[
\frac{\partial (AC)}{\partial t} + \frac{\partial (QC)}{\partial x} - e(x, t) = q_s(x, t) \tag{5.2}
\]

where
A = cross-sectional area of the channel

C = volumetric sediment concentration,

Q = channel discharge,

e = sediment erosion rate, and

qs = rate of lateral sediment inflow for channels.

5.2.2 Study Area

The Gully creek watershed (43°30'26" N- 43°35'26" N Lat., 81° 37'42.19" W-81° 39'51.9" W Long) draining into Lake Huron, Ontario Canada; along with one of its sub-basin outlet (GUL_RSD) is selected for this study (Figure 5.1). Gully Creek watershed encompasses an area of 1056.84 hectares. The landscape is characterized by undulating terrain with a maximum elevation of 281 m and minimum elevation of 217 m. Further, the GUL_RSD outlet drains a sub-basin comprising an exhaustive network of RSD. These ditches, if operational under a future drainage scenario for routing water, would ultimately drain at the GUL_RSD outlet. Therefore, GUL_RSD is considered for comparison analysis amongst various scenarios for this study (Figure 5.1).
Nearly 60% of the precipitation occurs as rainfall from April to October while the remaining as snow/rainfall during the five winter months from November to March. The average annual precipitation is 1,055 mm over 2001 - 2011 with a standard deviation of 165 mm (Liu 2013). Upper reaches of the watershed are dominated by clay loam soil while the lower reaches are mostly sandy loam. The name and areal extent of each soil type are presented in Table 5.1. Nearly 70% of the watershed is agricultural with corn, soybean and winter wheat being the primarily grown crops (Figure 5.2). Remaining 25% of the watershed is under natural vegetation, including trees, shrubs, and grasses.
Naturally vegetated areas primarily buffer along the main channel.

Figure 5. 2: (A) Soil, (B) Landuse, and (C) DEM of the Gully creek watershed
Table 5.1: Name and extent of each soil type in the Gully Creek watershed

<table>
<thead>
<tr>
<th>Soil code</th>
<th>Soil Type</th>
<th>Area (ha)</th>
<th>Area (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>ZAL</td>
<td>Bottom Land</td>
<td>84.57</td>
<td>8.08</td>
</tr>
<tr>
<td>PTH</td>
<td>Perth Clay Loam</td>
<td>101.06</td>
<td>9.66</td>
</tr>
<tr>
<td>HUO</td>
<td>Huron Clay Loam</td>
<td>797.11</td>
<td>76.21</td>
</tr>
<tr>
<td>BKN</td>
<td>Brookston Clay</td>
<td>63.26</td>
<td>6.05</td>
</tr>
</tbody>
</table>

Source: OMAFRA, Ontario Ministry of Natural Resources (OMNR), and Ausable Bayfield Conservation Authority (ABCA)

5.2.3 Baseflow Separation

The K2 could be used to simulate direct runoff and erosion for moderate-size watersheds on an event basis (Smith et al. 1995). Therefore, base flow is removed from the total streamflow at the watershed outlet (GULGUL 5 station) for competent calibration and validation of events. Direct runoff and base flow unify to compound total stream flow. While direct runoff is a fast contributor to stream flow, base flow is a slow contributor to streamflow (Kalin and Hantush, 2006b). WHAT software developed at Purdue University (https://engineering.purdue.edu/mapserve/WHAT/) was used to separate base flow from observed flow in this study. The methodology of the WHAT program is described by Kyoung et al. (2005). This methodology is based upon signal analysis and processing which splits high frequency signals (associated with direct runoff) from low frequency signals (associated with base flow) (Eckhardt and Arnold 2001; Lyne and Hollick 1979).

5.2.4 Sediment Load Estimation

Sediment loads are considered as a parameter for assessing water quality in this study. Therefore, sediment loads for the selected rainfall events considered need to be compared with the K2 simulated sediment loads for calibration and validation. Henceforth,
sediment loads should have an identical temporal resolution with observed streamflow data. Further observed streamflow data used during this study was available for every 15 minutes; from which direct runoff was separated and events prepared accordingly. However, due to collection and analysis costs involved, observed sediment loads were available for only some grab samples. Henceforth; sediment loads for these grab samples were extrapolated to procure observed sediment load for each event. Thereafter, observed sediment load for each event was compared with K2 simulated sediment load for that particular event. To extrapolate sediment load of the grab samples on an event basis, LOAD ESTimator (LOADEST) a web-based tool was employed (Park et al. 2015; Runkel et al. 2004). The tool estimates monthly sediment loads using observed streamflow, observed sediment concentration data (for grab samples), and regression model coefficients. There are eleven regression models in the LOADEST model. Further, three statistical methods: adjusted maximum likelihood estimation (AMLE), maximum likelihood estimation (MLE) and least absolute deviation (LAD) are used for calibrating its coefficients (Runkel et al. 2004). The AMLE method was used to compute monthly sediment loads. Further, the monthly sediment loads were extrapolated to procure sediment loads for each event based upon the month of the event and the duration of the storm event.

5.2.5 Input Data

Geospatial data used to setup the K2 model included a digital elevation model, DEM (5 m resolution), soil, land use, and streamflow network (Figure 5.2). The data layers were obtained from Ontario Ministry of Agriculture, Food, and Rural Affairs (OMAFRA), Ontario Ministry of Natural Resources and Forestry (OMNRF), and the Ausable Bayfield
Conservation Authority (ABCA). Precipitation data (April 2013 to May 2014) for K2 was obtained from a weather station installed within the watershed which had been established as a part of Watershed Based BMP Evaluation project by Yang et al. (2013) initiated in April 2011. Two different drainage network patterns (natural and RSD) were used as input for creating two different models. The natural drainage pattern was the most representative dataset for the entire watershed and hence used to calibrate and validate the K2 model. Modified drainage network, which was specifically developed for this study (details are given, Figure 5.1), was used as input for the second model (modified DNDP) to perform post-validation for the events.

5.2.6 Model Calibration and Validation

Prior to model calibration, detailed review of the literature was conducted to determine parameters which are sensitive to the model (Kalin and Hantush, 2006a; Memarian et al., 2013). Consequently, a sensitivity analysis was performed to identify the most sensitive parameters (see Table 5.2).

Henceforth only the identified sensitive parameters (Table 5.2) were used for model calibration. Further, manual calibration was performed in this study, based upon previous experience with the K2 model (i.e. expert knowledge) (see, for example, Kalin and Hantush, 2006a; Memarian et al. 2013; Nguyen et al. 2016; Acde et al. 2017). Also, calibration and validation of the model were performed using a multiplier approach. Quantitative and qualitative measures were coupled together during the calibration of the model.
Table 5. 2: Calibrated KINEROS 2 parameters

<table>
<thead>
<tr>
<th>Process</th>
<th>Parameter</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow</td>
<td>$K_s$</td>
<td>Saturated hydraulic conductivity</td>
</tr>
<tr>
<td>Flow</td>
<td>$n_c$</td>
<td>Channel's manning’s roughness</td>
</tr>
<tr>
<td>Flow</td>
<td>$n_p$</td>
<td>plane’s manning’s roughness</td>
</tr>
<tr>
<td>Flow</td>
<td>$S_i$</td>
<td>$S_i$ is initial relative saturation</td>
</tr>
<tr>
<td>Flow</td>
<td>$G$</td>
<td>Effective capillary drive</td>
</tr>
<tr>
<td>Sediment</td>
<td>$d_{50}$</td>
<td>Median particle size diameter.</td>
</tr>
<tr>
<td>Sediment</td>
<td>$\lambda$</td>
<td>pore size distribution index</td>
</tr>
<tr>
<td>Sediment</td>
<td>$\phi$</td>
<td>porosity</td>
</tr>
</tbody>
</table>

A conscious effort was made to maximize the statistics which demonstrate the performance of the model (more on these below) on an event basis. Statistics computed for estimating the efficiency of the model in simulating flow were Nash–Sutcliffe coefficient ($E_{NS}$), model bias (MB) and modified correlation coefficient ($r_{mod}$) (Safari et al. 2012; Moriasi et al. 2015; Nash and Sutcliffe 1970) while coefficient of determination ($R^2$), Nash–Sutcliffe coefficient ($E_{NS}$) were used to estimate the models performance in predicting sediment loads. A lower value for these statistical parameters indicates a poor prediction of the model. On the contrary, a higher value suggests a good prediction of the model. For assessing the size, shape, and volume of simulated hydrographs/sedimentographs, an aggregated measure (AM) was calculated using equation 5.3.

$$AM = \frac{r_{mod} + E_{NS} + (1 - |MB|)}{3}$$  \hspace{1cm} (5.3)

where

$E_{NS} = $ Nash–Sutcliffe coefficient,

$MB = $ model bias,
\( r_{\text{mod}} = \) modified correlation coefficient.

Model validation was performed based on the parameters determined during the calibration of the model. For flow, we investigated the peak flow and total direct runoff of the events to make sure that the model is performing well in predicting the hydrology of the watershed. The flow component of the K2 model was calibrated for three events (18 April 2013, 12 June 2012, and 12 June 2013) and validated for two events (12 April 2014 and 30 August 2013) respectively at the GULGUL 5 outlet (Table 5.3).

Since sedimentographs for observed events were not available due to lack of grab samples (n = 280 between April 2012 to June 2014), only a total sediment load could be estimated for the entire event using the methodology described in the earlier section. Therefore, eight events were used to calibrate K2 simulated sediment load with the observed sediment loads. Also, seven events were used to validate the sediment loads (Table 5.4). Value of parameters altered during the calibration process is shown in Table 5.5. Further, Nash–Sutcliffe coefficient (\( E_{\text{NS}} \)) and coefficient of determination (\( R^2 \)) were used to compute the model’s efficiency in simulating sediment loads.
Table 5.3: KINEROS-2 model parameters and outputs for calibration and validation events for flow

<table>
<thead>
<tr>
<th>Event date</th>
<th>Precipitation (mm)</th>
<th>Duration (hrs)</th>
<th>$n_{\text{channel}}$</th>
<th>$n_{\text{plane}}$</th>
<th>$K_s$</th>
<th>$Q_{\text{Peak}}^{\text{Observed}}$ (mm/hr)</th>
<th>$Q_{\text{Peak}}^{\text{Simulated}}$ (m$^3$/s)</th>
<th>Direct Runoff (Observed) (m$^3$)</th>
<th>Direct Runoff (Simulated) (m$^3$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2013-04-18</td>
<td>30.6</td>
<td>10</td>
<td>0.1</td>
<td>0.25</td>
<td>0.82</td>
<td>2.38</td>
<td>2.26</td>
<td>33390</td>
<td>22434</td>
</tr>
<tr>
<td>2012-06-12</td>
<td>18.2</td>
<td>2</td>
<td>0.1</td>
<td>0.25</td>
<td>0.82</td>
<td>0.34</td>
<td>0.46</td>
<td>4491</td>
<td>5378</td>
</tr>
<tr>
<td>Calibration</td>
<td>2013-06-12</td>
<td>19</td>
<td>0.1</td>
<td>0.25</td>
<td>0.82</td>
<td>0.24</td>
<td>0.46</td>
<td>5562</td>
<td>5558</td>
</tr>
<tr>
<td>2014-04-12</td>
<td>27.39</td>
<td>7</td>
<td>0.1</td>
<td>0.25</td>
<td>0.82</td>
<td>1.79</td>
<td>1.94</td>
<td>35131</td>
<td>19658</td>
</tr>
<tr>
<td>Validation</td>
<td>2013-08-30</td>
<td>18</td>
<td>0.1</td>
<td>0.25</td>
<td>0.82</td>
<td>0.28</td>
<td>0.27</td>
<td>2804</td>
<td>2892</td>
</tr>
</tbody>
</table>
Table 5.4: KINEROS-2 model parameters for calibration and validation events for sediment

<table>
<thead>
<tr>
<th>Event</th>
<th>Precipitation (mm)</th>
<th>Duration (hours)</th>
<th>Observed Sediment (Kg)</th>
<th>KINEROS 2 Simulated (Kg)</th>
<th>( R^2 )</th>
<th>( E_{NS} )</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Calibration</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2012-10-14</td>
<td>18.40</td>
<td>20</td>
<td>740</td>
<td>810</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2013-09-20</td>
<td>24.80</td>
<td>5</td>
<td>270</td>
<td>1040</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2013-04-18</td>
<td>30.60</td>
<td>10</td>
<td>19490</td>
<td>27230</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2014-04-12</td>
<td>27.39</td>
<td>7</td>
<td>12790</td>
<td>16320</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2013-08-01</td>
<td>59</td>
<td>15</td>
<td>19490</td>
<td>14180</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2013-06-16</td>
<td>14</td>
<td>6</td>
<td>300</td>
<td>1710</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2013-06-28</td>
<td>15.80</td>
<td>8</td>
<td>320</td>
<td>2290</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2012-10-20</td>
<td>15.80</td>
<td>15</td>
<td>1290</td>
<td>600</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Average</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>6840</td>
<td>8026</td>
</tr>
<tr>
<td><strong>Validation</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2012-11-12</td>
<td>16.40</td>
<td>9</td>
<td>910</td>
<td>1090</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2014-04-29</td>
<td>22.40</td>
<td>9</td>
<td>12420</td>
<td>13510</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2013-04-24</td>
<td>20.20</td>
<td>21</td>
<td>20850</td>
<td>8600</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2013-08-30</td>
<td>18</td>
<td>3</td>
<td>4945</td>
<td>6280</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2012-06-12</td>
<td>18.20</td>
<td>2</td>
<td>133</td>
<td>1010</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2013-06-12</td>
<td>19</td>
<td>3</td>
<td>530</td>
<td>1046</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Average</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>6630</td>
<td>5260</td>
</tr>
</tbody>
</table>
Table 5.5: Soil parameters calibrated during KINEROS-2 simulations

<table>
<thead>
<tr>
<th>Soil Code</th>
<th>Soil Type</th>
<th>G(cm)</th>
<th>λ</th>
<th>φ</th>
<th>d_{50}(μm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>ZAL</td>
<td>Bottom Land</td>
<td>31.14</td>
<td>0.21</td>
<td>0.53</td>
<td>1</td>
</tr>
<tr>
<td>PTH</td>
<td>Perth Clay Loam</td>
<td>40.04</td>
<td>0.18</td>
<td>0.45</td>
<td>1</td>
</tr>
<tr>
<td>HUO</td>
<td>Huron Clay Loam</td>
<td>45.03</td>
<td>0.17</td>
<td>0.41</td>
<td>1</td>
</tr>
<tr>
<td>BKN</td>
<td>Brookston Clay Loam</td>
<td>42.04</td>
<td>0.23</td>
<td>0.42</td>
<td>1</td>
</tr>
</tbody>
</table>

5.2.7 Post-Validation

The K2 model was calibrated and validated under the natural existing DNDP. Further, the model parameters adjusted during the calibration of the model were transferred to the model with modified DNDP (for RSD) during the post-validation phase. To accomplish the required task, modified DNDP was used as an input to K2 in place of the previously used DNDP_N for the Gully Creek watershed. While the calibration process is used to determine and adjust the sensitive parameters in order to bring model simulated flow values close to their observed counterparts, it is equally important to transfer these calibrated model parameters in a systematic way when the model is set up with a different DNDP. This was the case during post-validation of the K2 model, where DNDP_N was replaced by the modified DNDP. For this purpose, calibrated model parameters were transferred according to the soil and landuse combinations for planes and channels (see Figure 5.3).
In other words, there was no new model calibration. The same parameter values obtained during the calibration period were used during the post-validation period, too. For example, effective capillary drive $G$ was altered for different soil types (Table 5.5) and Manning’s $n$ was changed for planes and channels. Further, the model was also run using the modified DNDP to visualize the differences in model simulations using various DNDP
for two events (12 April 2014, and 18 April 2013). This was done to demonstrate the significance of draining flow through RSD in modeling. Further, various scenarios for the modified DNDP (road-side ditch) were considered for scenario analysis: DNDP_M (road-side ditches with the vegetation as DNDP_N), DNDP_MV (road-side ditches lined with medium vegetation), and DNDP_HV (road-side ditches lined with thick vegetation). Basically, Manning’s n was changed for these scenarios. Thereafter model simulations for various DNDP scenarios were compared with the existing DNDP at the GUL_RSD outlet to demonstrate the performance of the model.

5.3 Results and Discussion

5.3.1 Calibration and Validation

5.3.1.1 Flow

Calibration and validation were executed manually by comparing observed and K2 simulated direct runoff hydrographs. Model parameters such as effective capillary drive G, pore size distribution index λ, Manning’s n and porosity φ were adjusted during the calibration events which have considerable impact upon direct runoff and peak flow (Table 5.4 and 5.6).
Table 5.6: Model statistics for calibration and validation flow events

<table>
<thead>
<tr>
<th></th>
<th>Direct Runoff Simulation</th>
<th></th>
<th></th>
<th>Statistical fitting</th>
<th>Direct Runoff Simulation</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>MB</td>
<td>-0.25</td>
<td>0.24</td>
<td>-0.01</td>
<td>MB</td>
<td>-0.34</td>
<td>0.17</td>
</tr>
<tr>
<td>$r_{mod}$</td>
<td>0.89</td>
<td>0.57</td>
<td>0.68</td>
<td>$r_{mod}$</td>
<td>0.9</td>
<td>0.76</td>
</tr>
<tr>
<td>$E_{NS}$</td>
<td>0.8</td>
<td>0.19</td>
<td>0.6</td>
<td>$E_{NS}$</td>
<td>0.72</td>
<td>0.6</td>
</tr>
<tr>
<td>Calibration Events</td>
<td>AM</td>
<td>0.81</td>
<td>0.51</td>
<td>0.76</td>
<td>Validation Events</td>
<td>AM</td>
</tr>
<tr>
<td>Goodness of fit</td>
<td>Good</td>
<td>Good</td>
<td>Good</td>
<td>Goodness of fit</td>
<td>Average</td>
<td>Below average</td>
</tr>
<tr>
<td>Observed peak flow (m$^3$/s)</td>
<td>2.38</td>
<td>0.34</td>
<td>0.24</td>
<td>Observed peak flow (m$^3$/s)</td>
<td>1.79</td>
<td>0.28</td>
</tr>
<tr>
<td>Simulated peak flow (m$^3$/s)</td>
<td>2.26</td>
<td>0.46</td>
<td>0.46</td>
<td>Simulated peak flow (m$^3$/s)</td>
<td>1.94</td>
<td>0.27</td>
</tr>
</tbody>
</table>
Therefore, model parameters were manually manipulated for each overland flow plane (OFP) and each channel until peak flow and direct runoff simulated at the outlet channel of the K2 model (GULGUL 5) closely matched the observed values. The logic behind altering parameters like Manning’s n for each plane and channel is that once a close agreement is reached between observed and modelled values for peak flow and direct runoff at the watershed outlet, the adjusted values for model parameters (Manning’s n, G, Ks etc.) would be realistic and match the real-world parameters for each plane and channel within the watershed. Hence, these parameters could be transferred to another model with similar conditions during post-validation. Table 6.6 reveals the model statistics computed for the calibration and validation events respectively. K 2 model simulated values for peak flow and direct runoff were compared with their subsequent observed values during model calibration and validation respectively. Also, Table 5.6 reveals the observed and computed direct runoff and peak flow for calibration events. The model performs fairly well for calibration events, with decent statistical parameters (Nash–Sutcliffe efficiencies) as categorized in Table 5.6. Positive values are generally considered to be acceptable (Safari et al., 2012; Memarian et al., 2013) with values above 0.5 being good. The best performance is for 18 April 2013, which corresponds to spring season characterized by wet soil condition with no crops on the field. Also, the model estimated peak flow very closely resembles the observed data for the event 18 April 2013. The model fit for the event 12 June 2012 is also good, but it is just acceptable for 12 June 2013. Also, Table 5.6 shows simulated and observed direct runoff for validation events, along with peak flow. The best performance is observed for the event on 12 April 2014, for flow ($E_{NS} = 0.68$). The performance of the model for the event 30 August 2013 is just
acceptable for flow (ENS = 0.48). It is not possible to assert that the calibrated parameters are optimal since calibration was carried out manually rather than automated. Figure 5.4 shows the observed and computed hydrographs for calibration events. The model performs quite well for calibration events, with acceptable Nash–Sutcliffe efficiencies especially for 18 April 2013 and 12 June 2013 storm events. Model simulated values underestimated peak flow for 18 April 2013 event. However, the model overestimated peak flow for 12 June 2012 and 12 June 2013 storm event. Also, Figure 5.5 shows simulated hydrographs for validation events, along with observed data. The best performance is observed for the event on 12 April 2014, where both the peak flow and direct runoff is simulated fairly well.
Figure 5.4: Computed and observed hydrographs with KINEROS-2 for calibration events
Figure 5: Computed and observed hydrographs with KINEROS-2 for validation
5.3.1.2 Sediment

Observed and simulated sediment load for eight and seven events respectively, at the watershed outlet (Figure 5.1) were used for K2 calibration and validation. The parameters calibrated and their final values are shown in Table 5.5. Graphical comparison of sediment load for the calibration period (Figure 5.6a) and statistical parameters, with $R^2$ 0.93 and ENS 0.80, suggest that the sediment loads were adequately simulated by the K2 model. The observed average sediment load of 6.84 tons (Table 5.4) was slightly lower than the K2 simulated average sediment load of 8.02 tons over the entire calibration events (14 October 2012 -12 April 2014). Further, Table 5.6 indicated no significant overprediction or underprediction of flow during this simulation period. Graphical comparison of sediment loads for the validation period (Figure 5.6b) and statistical parameters, with $R^2$ 0.79 and $E_{NS}$ 0.56; suggest that the sediment loads were adequately simulated by the K2 model. The estimated average sediment load of 6.63 tons was about the same as the predicted sediment load of 5.26 tons over the entire validation period. Since total sediment loads predictions used for post-validation analysis (changed DNDP) were very good, the K2 model was considered to be adequately describing the hydrology of the study watershed.
Figure 5. 6 (A) Events for calibration, and (B) validation for sediment loads.
5.3.2 Post validation

In this section, the calibrated/validated K2 model was further tested for its capabilities in predicting effects of changes in DNDP on flow and sediment loads. Combination of landuse and soils for the Gully Creek watershed remained the same for both DNDP. Only the water routing network was different under the scenario considered for modified DNDP. Further, a storm event of high intensity could suddenly yield a huge quantity of runoff in a brief period. Henceforth, while analyzing the performance of K2 model for RSD, analysis pertaining to extreme rainfall patterns is of paramount importance. Therefore, a scenario analysis is investigated where the flow is routing for extreme events and its performance assessed at the GUL_RSD outlet. Event with a return period of 2-year with a 24-hour storm duration is selected for analysis. Comparison of the model simulations for flow is presented in the section below.

5.3.2.1 Flow-DNDP_M (road-side ditches with the same manning’s n)

In this scenario, simulated flow values obtained from the model, (DNDP_M: with the same Manning’s ‘n’ as DNDP_N for RSD) were compared with the simulated hydrograph procured under the DNDP_N at the GUL_RSD outlet (Figure 5.7). Unaltered Manning’s n value of 0.1 estimated during the model calibration was used in this scenario for the channel representing the RSD in this scenario (Table 5.7).
Figure 5.7: Simulated hydrographs with KINEROS-2 for post-validation events for DNDP_M scenario.
Table 5.7: Manning’s n for various drainage network/drainage pattern (DNDP)

<table>
<thead>
<tr>
<th></th>
<th>2014-04-12</th>
<th></th>
<th></th>
<th>2013-04-18</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>DNDP_N</td>
<td>DNDP_M</td>
<td>DNDP_MV</td>
<td>DNDP_HV</td>
<td>DNDP_N</td>
<td>DNDP_M</td>
</tr>
<tr>
<td></td>
<td></td>
<td>(same n =</td>
<td>(n = 0.15)</td>
<td>(n = 0.2)</td>
<td>(same n =</td>
<td>(n = 0.15)</td>
</tr>
<tr>
<td>Q_p (m³/s)</td>
<td>1.14</td>
<td>1.68</td>
<td>1.31</td>
<td>1.16</td>
<td>1.85</td>
<td>2.26</td>
</tr>
<tr>
<td>Runoff (m³)</td>
<td>8367.52</td>
<td>10995</td>
<td>9458</td>
<td>9093</td>
<td>13382</td>
<td>16011</td>
</tr>
<tr>
<td>Sediment load (Kg)</td>
<td>13283.20</td>
<td>16866</td>
<td>15786</td>
<td>14427</td>
<td>21775</td>
<td>25766</td>
</tr>
</tbody>
</table>
As mentioned in the section above, 12 April 2014 and 18 April 2013 storm events were considered for estimating the model’s performance under this scenario analysis during the post-validation phase. Graphical representation of hydrograph for the 18 April 2013 event reveals that the GUL_RSD outlet flows to a maximum of 2.26 m$^3$/s and 1.85 m$^3$/s for the DNDP_M and the DNDP_N respectively (Figure 5.7). Likewise, the volume of direct runoff escalates from 13382.13 m$^3$ to 16010.77 m$^3$ under the revised DNDP. Furthermore, storm event for 12 April 2014 is drained to the GUL_RSD outlet with a peak flow of 1.68 m$^3$/s for the altered DNDP_M. Also, Figure 5.7 shows that there is a considerable increase in the flow of water routed, with the volume of direct runoff increasing to 10994.35 m$^3$ compared with 8367.52 m$^3$ under the DNDP_N. Henceforth, it could be concluded that the amount of water routed to the GUL_RSD outlet increases considerably under the DNDP_M. Although this drainage network is convenient for routing water from tile drains etc. to the downstream ecosystems (streams/creeks), it substantially increases the quantity of water routed downstream. Therefore, it is recommended to make certain modifications in the DNDP_M to reduce the quantity of water routed to the watershed outlet, thereby minimizing their adverse effect upon downstream water quantity and sediment yield.
5.3.2.2 Flow-DNDP_MV (Road-side ditches lined with medium vegetation)

A scenario where road-side ditches are lined with medium vegetation (DNDP_MV) was also considered. For this exercise, manning’s n was changed to 0.15 for the channel representing the road-side ditch (Table 5.7). Simulated flow values obtained from the model for DNDP_MV were compared with the simulated hydrograph procured for the DNDP_N at the GUL_RSD outlet (Figure 5.8). Similar to the earlier scenario, 18 April 2013, and 12 April 2014 storm events were again considered for analysis. Graphical representation of hydrographs for the 18 April 2013 event reveals that the GUL_RSD outlet flows to a maximum of 2.14 m³/s and 1.84 m³/s for the DNDP_MV and the DNDP_N respectively (Figure 5.8).

However, considering DNDP_N as a baseline, the percentage increase in peak flow reduces from 22.16 % to 15.67 %. Likewise, the percentage increase in direct runoff volume reduces from 22.16 % to 15.67 % under this revised scenario. However, compared to DNDP_N; the direct runoff volume is still higher (Table 5.7). Furthermore, the 12 April 2014 storm event drains to the GUL_RSD outlet with a peak flow of 1.31 m³/s. Also, Figure 5.8 reveals that the volume of water routed is still considerably higher than DNDP_N. However, compared with DNDP_M the percentage increase in peak flow diminishes from 47.36 % to 14.91% keeping DNDP_N as the baseline. Similarly, compared with DNDP_M, the percentage increase in direct runoff volume reduces from 31.34 % to 13.03 % respectively (Table 5.7).
5.3.2.3 Flow-DNDP_HV (road-side ditches lined with thick vegetation)

In this scenario, flow is routed through the modified DNDP_HV considering a lining of thick vegetation over the drainage ditches. Henceforth manning’s n is marginally increased (0.2) for the RSD considered (Table 5.7). A procedure similar to the above-mentioned scenario’s is considered, where model simulated flow routed for the DNDP_HV scenario is compared DNDP_MV, DNDP_M and DNDP_N at the GUL_RSD outlet for 12 April 2014, and 18 April 2013 storm events respectively (Figure 5.9).

Model simulated hydrograph at GUL_RSD for the 12 April 2014 event (Figure 5.9) reveals that percentage increase in peak flow, reduces to 1.75 % for this scenario, which is substantially lower than 47.36 % for DNDP_M, 14.91 % for DNDP_MV compared to the DNDP_N baseline respectively (Table 5.7). Also, the percentage increase in direct runoff volume reduces to 8.67 % compared with 31.39 % for DNDP_M and DNDP_MV respectively (Table 5.7).

Similarly, flow simulated at the GUL_RSD outlet for 18 April 2013 storm reveals a peak flow of 1.83 m3/s for the DNDP_HV which is significantly less than the peak flow procured under DNDP_MV drainage scenario, a reduction from 15.67 % to -1.08 % when compared with the DNDP_N scenario (Table 5.7 and Figure 5.9). Subsequently, Table 5.7 demonstrates a modest increase in the quantity of direct runoff simulated; a percentage increase of 1.55 % compared with DNDP_N. However, a considerable reduction in direct runoff volume is observed when compared to DNDP_M (19.64 %) and DNDP_MV (11.29 %) respectively. Therefore, since the flow simulated at the GUL_RSD outlet decreases for DNDP_HV compared with DNDP_M and DNDP_MV, it is recommended to line the road-side ditches with a heavy vegetation like thick grass-cover.
It would increase the manning’s n value which would further reduce the quantity of water routed.

5.3.2.4 Sediment-DNDP_M (road-side ditches with the same manning’s n)

Upon successful calibration and validation of the model for flow (peak flow and direct runoff) and sediment, the K2 model was run for 12 April 2014, and 18 April 2013 storm events under the modified DNDP scenario. Manning’s n was kept unchanged to 0.1 for the channel representing the RSD under the modified DNDP (Table 5.7). Further results were compared at the GUL_RSD outlet. The results indicate that DNDP-M increased sediment loads by 26.97, and 18.32 % for 12-Apr-14, and 18-Apr-13 storm events respectively. Table 5.7 shows detailed sediment loads at the GUL_RSD outlet. Results clearly demonstrate that the change in DNDP significantly increases the amount of sediment loads routed.
Figure 5.8: Simulated hydrographs with KINEROS-2 for post-validation events for DNDP_MV scenario
Figure 5.9: Simulated hydrographs with KINEROS-2 for post-validation events for DNDP_HV scenario
5.3.2.5 Sediment-DNDP_MV (road-side ditches lined with medium vegetation)

In this scenario sediment routed under modified DNDP_MV; where the road-side ditches are lined with medium vegetation is considered. Manning’s n representing the RSD was altered to 0.15 in this scenario. Sediment loads increased by 18.83, and 13.34% for 12-Apr-14, and 18-Apr-13 storm events respectively compared with the DNDP_N. However, when compared with the DNDP_M, the sediment loads reduced (Table 5.7).

5.3.2.6 Sediment-DNDP_HV (road-side ditches lined with thick Vegetation)

Under this scenario analysis, the sediment load routed decreases further. Increase in Manning’s n to 0.2 reduces flow and sediment load for 12-Apr-14 storm event (Table 5.7). Compared to DNDP_M (26.95%) and DNDP_MV (18.84 %), the percentage increase in sediment load reduces to 8.6 % considering the DNDP_N as the benchmark. However, the sediment loads are still higher compared with the sediment loads procured under DNDP_N. Similar results are obtained for 18-Apr-13 storm event where an increase in sediment load reduced to 4.18 % compared with 18.33 % and 13.33 % for DNDP_M and DNDP_MV respectively. Therefore, since the amount of sediment load simulated at the GUL_RSD outlet decreases further for the DNDP_HV compared with the DNDP_MV, it is recommended to line the RSD with a thick vegetation like grass-cover since it increases manning’s n value which would reduce the quantity of water and sediment routed at the GUL_RSD outlet.

5.3.2.7 2-Year 24-Hour Design Storm

In this scenario model simulated flow routed for the DNDP_M, DNDP_MV and DNDP_HV scenarios at the GUL_RSD outlet are compared for a synthetic 2-year 24-hour design storm (Table 5.8, Figure 5.10). Also, model-simulated hydrographs for the design
storm (Figure 5.10) reveals that percentage increase in peak flow is 15.97%, 9.25%, and 1.99% respectively for DNDP_M, DNDP_MV, and DNDP_HV scenarios compared with DNDP_N (Table 5.8). Furthermore, the percentage increase in direct runoff volume is 9.86 %, 4.53 %, and 1.6 % respectively for DNDP_M, DNDP_MV and DNDP_HV scenarios (Table 5.8). Sediment routed under these scenarios indicate that RSD lined with thick vegetation cover (DNDP_HV) is successful in mitigating the sediment loads and getting the values close to the natural conduction (DNDP_N) (Table 5.8).
Figure 5. 10: Simulated hydrographs (A) DNDP_M (B) DNDP_MV and (C) DNDP_HV for 2-year-24-hour design storm
Table 5. 8: Manning’s n for various drainage network/drainage pattern (DNDP)

<table>
<thead>
<tr>
<th></th>
<th>DNDP_N</th>
<th>DNDP_M (same n = 0.1)</th>
<th>DNDP_MV (n = 0.15)</th>
<th>DNDP_HV (n = 0.2)</th>
</tr>
</thead>
<tbody>
<tr>
<td>$Q_p (\text{m}^3/\text{s})$</td>
<td>5.51</td>
<td>6.39</td>
<td>6.02</td>
<td>5.62</td>
</tr>
<tr>
<td>Runoff (m$^3$)</td>
<td>41503.87</td>
<td>45596.29</td>
<td>43387.58</td>
<td>42168.83</td>
</tr>
<tr>
<td>Sediment load (Kg)</td>
<td>25238.08</td>
<td>32344.45</td>
<td>29592.45</td>
<td>27680.73</td>
</tr>
</tbody>
</table>
5.4 Conclusions

Watershed models are mostly calibrated and validated for a study area with the existing natural DNDP. However, in modern times due to urbanization and alteration of drainage network due to tile drains, there may be a significant change in the DNDP of watersheds. The main objective of this paper was to explore the performance of the KINEROS 2(K2) model in predicting water quantity and sediment yield in response to alterations in DNDP within an environmentally-sensitive watershed in Ontario, Canada. Results demonstrated that the model was able to simulate flow (peak flow and direct runoff) and sediment loads satisfactorily for both calibration and validation events. The calibrated model then was fed with a modified DNDP. The model-produced outputs for peak flow, direct runoff, and sediment yield were compared for two rainfall events (12 April 2014 and 18 April 2013) and a synthetic design storm (2year-24hour) at the GUL_RSD outlet under three distinct DNDP (natural vegetation, medium, and thick vegetation). The results procured for the altered DNDP scenario were as expected. The following observations and conclusions are drawn:

1. K2 performed well for DNDP_N, both for the calibration and validation events. Also, for altered DNDP, keeping the natural vegetation lining on the RSD (DNDP_M) the model-simulated peak flow, direct runoff and sediment yield increased as expected for the for post-validation events.

2. Further, when RSD were lined with a thicker vegetation (DNDP_MV and DNDP_HV), a reduction in peak flow, direct runoff, and sediment yield was observed compared to DNDP_M.
3. A DNDP with thick vegetation cover, which would have a higher value for manning’s n, is more efficient in reducing the peak flow, direct runoff volume and sediment loads, both for the natural rainfall events and the synthetic design storm (2year-24-hour).

In summary, this study showed that K2 could be a useful model in simulating flow and sediment within watersheds undergoing severe alterations in their drainage pattern due to development or implementation of specific watershed management plans. However, it should be noted that this is just an isolated study conducted in a cold region. Similar studies under different physiographic and climatic regions can further reinforce conclusions of this study.
Transition to Chapter 6

Vegetative filter strips (VFS) are globally recognized as an effective BMP in reducing non-point source pollution. Maximum effectiveness of a VFS at a watershed-level could be achieved by adequately installing and sizing a VFS along the edge of the field. Existing watershed models have limitations in appropriately representing and modeling VFS at the watershed scale. Therefore, in this research, a new modeling approach consisting of the Agricultural Non-Point Source (AGNPS) model, AGNPS_VFS toolkit, and a regression equation is developed to explore the effectiveness of VFS applied along the edge of fields. AGNPS cells are identified as locations where the edge of the field VFS is to be installed. Further, the approach was tested with a case study. The model was calibrated and validated for a flow and sediment load at the watershed outlet. Thereafter, the modeling approach is used to compute sediment reducing efficiency (SRE) for the edge of the field VFS. Objectives of this study were to test the effectiveness of uniform VFS (5m by 18 m) lengths located at several locations (draining an upstream area of 3 ha, 4 ha, 6 ha, and at spatially variable locations) within a watershed to demonstrate the ability of the developed approach to evaluate effectiveness of VFS application in sediment abatement. Maximum SRE was observed for VFS placed at spatially variable locations; the developed approach reduced nearly 23.03% of sediment yield, while VFS placed along cells draining an upstream area of 3 ha, 4 ha and 6 ha removed 9.59 %, 12.39 %, and 5.91% of sediment loads respectively.
Chapter 6

6 A modeling approach for evaluating watershed-scale water quality benefits of vegetative filter strip - A Case Study in Ontario

6.1 Introduction

Implementation of appropriate Best Management Practices (BMPs) can substantially abate non-point source (NPS) pollution to meet desired water quality criteria (Chen et al., 2016; Lobo and Bonilla, 2017). However, a critical source area (CSA) within a watershed needs to be identified before implementation of BMPs. A CSA in a watershed is usually the minimum area required by a stream for permanent existence (Xiaoyan & Qinhui, 2011). Several studies have proved that random selection and installation of BMPs are not equally effective in mitigating NPS pollutants (Dillaha et al., 1988; Sprague and Gronberg, 2012; Stang et al., 2016; Singh et al., 2017). Henceforth; determination of critical source areas (CSAs) for an impaired water body, followed by placement of best management measures is pivotal for effective improvement of water quality (Giri et al., 2012; Panagopoulos et al., 2011; Trevisan et al., 2010).

A Vegetative filter strip (VFS) is one such globally adopted BMP used for trapping sediment (Chen et al., 2016; Inamdar et al., 2001; Gharabaghi et al., 2002 & 2006; Lobo
VFSs are areas of natural or planted vegetation aimed towards improving the quality of surface runoff. VFSs prevent pollutants from entering the stream by (a) trapping sediment transported with runoff through the settlement of particles and (b) by promoting infiltration of the surface runoff containing suspended clay particles. The impact of VFSs upon flow and sediment reduction at a watershed scale have been evaluated by a few researchers (Chen et al., 2016; Lobo et al., 2017). Lobo and Bonilla (2017) using nonlinear relationships and the WEPP model to develop a Vegetative Filter Strip (SCVFS) model to forecast the trapping efficiency of VFS for clay, silt, and sand in central Chile. Further, Chen et al. (2016) utilized the meta-regression approach for developing a model capable of evaluating VFS pesticide retention efficiency. Muñoz-Carpena et al. (1999) and Helmers et al. (2006) found the length of the VFS to be the most significant criterion influencing its sediment removal efficiency. Some investigations asserted that increasing a VFS length more than 10 m does not enhance its efficiency significantly (Abu-Zreig et al., 2004; Lee et al., 2003). However, Gharabhagi et al. (2001) determined that finer sediment particles take longer to separate out and therefore need a longer length. They determined that the first 5m of VFS play a significant role in removing suspended solids and aggregates larger than 40 mm.

Additional research is recommended for a more cost-effective and efficient manner of placing VFSs. The installation of a VFS along the edge of the fields could be one such scenario where the maximum efficiency of VFS could be achieved. Parajuli et al. (2008) used the soil and water assessment tool (SWAT) model for investigating the effectiveness of different VFS lengths applied along the edge of fields to reduce non-point source
pollution in the Upper Wakarusa watershed in northeast Kansas. Further, the overland flow routing module of the SWAT model was modified by Park et al. (2011) to better simulate the impact of overland flow from upper sub-basins as a variable to the installed VFS. Their modeling efforts were focused upon a continuous timestep; however, since sediment transport along with surface runoff is primarily dependent upon storm intensity, event-based modeling would be more appropriate.

Therefore AGNPS (which is an event-based model) is used to simulate watershed hydrology in this study. AGNPS is a grid-based model having the capability to route flow between cells. Also, location of channel networks can also be more explicitly represented. Henceforth, in this study, a new modeling approach is developed using an event-based model (AGNPS) and the AGNPS_VFS toolkit. Further, the approach is tested for a watershed in Ontario, Canada. Specific objectives associated with the developed modeling approach are as follows:

1. Development of a new modelling approach for more accurate simulation of the edge of the field VFS at a watershed scale using the AGNPS model.
2. Compare the sediment removal efficiency of the edge of the field VFS (constructed at several locations within the watershed).

6.2 Materials and Methods

6.2.1 AGNPS_VFS toolkit

AGNPS_VFS toolkit developed by Rudra and Sebti (2010) is used for analysis in this study. Design of AGNPS_VFS is programmed to integrate two models, AGNPS and VFSMOD. A user interface is developed to link the two models (Sebti and Rudra, 2010) where the AGNPS model utilizes the topography and land surface information to compute
sediment load, and runoff for each dominant flowpath. Subsequently, AGNPS output is fed as input to the VFSMOD model, where the amount of sediment loads leaving the VFS is estimated. A detailed description of the processes used in the toolkit can be found (Sebti and Rudra, 2010).

The AGNPS model (Young et al., 1989) developed by USDA-Agricultural Research Service was setup for Gully Creek watershed in this study for computing its sediment load. AGNPS is a physically-based, event simulation, watershed model where the entire watershed is divided into square cells. Runoff is computed for each cell in the model using the SCS curve number approach (McCuen, 1982). Also, peak flow is estimated using an empirical relationship proposed by Smith and Williams (1980). Further, the sediment yield is calculated from a modified form of the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978). Comprehensive information pertaining to the model can be procured from the AGNPS manual (Young et al., 1989).

To simulate the transport of flow and sediment through VFS, the VFSMOD model (Muñoz-Carpena et al., 1999) is selected. The latest version of the program and the related documentation are available on the VFSMOD website. VFSMOD model uses a finite element approach for simulating runoff through the VFS (Muñoz-Carpena et al., 1993a), which is further coupled to mechanistic infiltration equations (Muñoz-Carpena et al., 1993b) making it more suitable than other VFS models. The solution of the one-dimensional kinematic wave equation for overland flow (mass conservation and momentum conservation) is linked to a sediment filtration model to simulate the transport and deposition of sediment based on the hydraulics of flow within grass filter media.
Further, Muñoz-Carpena and Parsons (2004) incorporated a field-scale upland hydrology model (UH) component to the VFSMOD, developing a user interface called VFSMOD-W.

The VFSMOD model requires several input parameters as it considers distinct field and VFS conditions. VFSMOD is a mechanistic, event-based model operating at a field-scale designed to reproduce the hydrograph and sediment inflow from an adjacent upper field. The model computes the outflow, infiltration, and sediment reduction transported via VFS from an adjacent upper field (Muñoz-Carpena et al., 2010). Also, the model could be used to predict sediment transported through VFS. Outflow and sediment reduction through a VFS is based upon flow hydraulics and infiltration into the soil layer.

6.2.2 Development of the Modeling Approach

Almost all the hydrological models operating at a watershed scale are capable of estimating runoff and sediment yield at any point of interest within a watershed. However, some of the commonly used models like AGNPS cannot be used directly to assess the impact of VFS in reducing sediment loading within a watershed. Other field scale models such as the VFSMOD have been primarily developed to evaluate VFS performance at a field scale. Therefore, to overcome the limitations of the AGNPS model, a methodology was developed where the AGNPS model is indirectly linked with the VFSMOD model to evaluate the efficiency of VFS in reducing sediment loads at the watershed level (Sebti and Rudra, 2010). The authors evaluated the impact of VFS placed along the buffer zone of a stream in reducing sediment loads at a watershed scale. In this study, their approach is modified to simulate the impact of the edge of the field VFS using the AGNPS model.
In the modified approach, a VFS is installed at planned locations within the watershed. The AGNPS model calculates runoff and sediment load downstream for every cell. When a VFS is installed at a certain cell in the AGNPS model, it is expected to reduce the sediment load. This reduction in sediment yield through a VFS could be simulated efficiently using the VFSMOD model. Henceforth, a reduced sediment yield would be transported to the downstream cell in the AGNPS model. Alteration in the source code of the AGNPS model to simulate the above process would be the most efficient method. However, such a comprehensive programming exercise was beyond the scope of this research.

Alternatively, if the AGNPS output for a cell (where the VFS is to be placed) is routed through the VFSMOD model, a reduced sediment yield for that particular cell would be observed (where VFS is to be placed). Subsequently, in the AGNPS model, if the sediment yield for that cell (where the VFS is to be placed) could be made to match the VFSMOD output by adjusting the parameters that impact the sediment load for a particular cell like USLE K, C, P, and n of all the upland catchment cells (draining into the current cell) the impact of VFS could be indirectly incorporated into the AGNPS output. However, despite a simple solution provided by this methodology, setting up a separate VFSMOD model for every VFS installed within a watershed would be impractical and cumbersome.

Further to simplify the process and eliminate the requirement for hydrologic modeling of every VFS within the watershed, the VFSMOD model was set up and run for 32 representative catchments (simulated from an earlier validated AGNPS model, see above section). Then a simple regression analysis was performed on the results to
develop a regression equation for VFS removal efficiency as a function of drainage area. Thirty-two catchments selected for the study are sub-divided into drainage classes having an upland drainage area of 4, 6, 8, and 12 acres respectively. Then eight catchments are selected for each drainage class. Further, a VFS is placed at the end of the last cell for each drainage class so that the sediment removal efficiency of VFS for different catchments areas could be analyzed and a regression function could be developed accordingly. These representative catchments represent approximately 16% of the total watershed area. Note that concentrated flow significantly reduces the VFS effectiveness. Hence VFS catchment areas were limited to 6 ha for the considered scenarios. For each representative catchment, the following information is available (simulated from an earlier validated AGNPS model):

- Event data (rainfall depth, rainfall duration, and pattern)
- Geometry (drainage area, the average slope of the upland area, flow length)
- Land Use/Landcover data (SCS curve number, dominant soil texture, soil erodibility factor, crop factor, and practice factor)

The above data was used as input data to set-up the VFSMOD model for each representative catchment. Further, the AGNPS-VFSMOD toolkit interface was used in setting up the VFSMOD model. To estimate VFS effectiveness, VFSMOD requires a hydrograph with an average sediment concentration as input data. To ensure input data match with AGNPS outputs for selected catchments, the VFSMOD model was calibrated to match runoff volume, peak flow, and total sediment yield. By calibrating the VFSMOD hydrologic model, the total sediment yield would automatically match the AGNPS output
as both AGNPS and VFSMOD use the Universal Soil Loss equation for sediment transport modeling.

Also, the VFSMOD model requires the sediment median particle size ($d_{50}$) as input data. AGNPS output provides the total weight of sediment yield for each particle class (clay, silt, sand, large aggregate, and small aggregate) at any grid cell. Assuming a nominal particle size for each particle class (2 µm for clay, 10 µm for silt, 100 µm for sand, 30 µm for small aggregate, and 300 µm for large aggregate) and considering the percentage of each particle class in the total sediment yield, the $d_{50}$ of the sediment at the outlets of representative catchments were estimated.

Finally, the VFSMOD model was setup for each representative catchment using the AGNPS outputs at the downstream end of each catchment with the following assumptions:

- A 5m wide filter strip was assumed downstream of each representative catchment
- The length of the filter strip was assumed to be 20% of the AGNPS grid cell, i.e., the flow and sediment downstream of each representative catchment is distributed along 20% of the grid cell

The outcome of VFSMOD (i.e., VFS removal efficiency) for each representative catchment is represented in Table 1. For each scenario considered, the geometric mean of the removal efficiencies was calculated, and a power trendline was determined as follows:

$$RE = 1.0196A^{-0.0935}$$

$$R^2 = 0.99$$
Where:

RE = Removal Efficiency (ratio)
A = Drainage Area (ha)

It must be noted that equation 1 is not valid for drainage areas less than 1.62 ha (4 ac).

The regression relationship developed is represented in Figure 6.1.

![Regression Equation](image)

Figure 6.1: Regression equation developed for the modeling approach
Table 6.1: VFSMOD output

<table>
<thead>
<tr>
<th>Area (ha)</th>
<th>VFSMOD Outputs for Representative Catchments</th>
<th>Geometric Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>4.86</td>
<td>87.4% 90.8% 86.9% 81.8% 81.6% 87.2% 88.8% 98.5%</td>
<td>87.7%</td>
</tr>
<tr>
<td>3.24</td>
<td>92.7% 92.2% 88.8% 92.4% 94.5% 92.9% 92.2% 86.0%</td>
<td>91.4%</td>
</tr>
<tr>
<td>2.43</td>
<td>97.0% 94.6% 93.0% 95.5% 94.0% 93.0% 93.0% 95.1%</td>
<td>94.4%</td>
</tr>
<tr>
<td>1.62</td>
<td>96.6% 96.4% 97.3% 95.8% 98.4% 97.6% 97.3% 97.4%</td>
<td>97.1%</td>
</tr>
</tbody>
</table>
In this study, several scenarios were considered, and VFS was assumed to be installed at various locations within the watershed arbitrarily. For each installed VFS, the power function was used to estimate the VFS trapping efficiency based on the VFS drainage area (Equation 1). Further, the sediment transport parameters for the catchment were adjusted to match the AGNPS output for sediment yield \((SY_i)\) with sediment yield for the downstream cell of VFS \((SY_o)\) computed using equation 6.2.

\[
SY_o = (1 - RE)^* SY_i
\]  

(6.2)

Where:
- \(RE\) = Removal Efficiency (ratio) as estimated by Equation 6.1
- \(SY_i\) = Sediment yield estimated by AGNPS at a point of interest (upstream of the VFS)
- \(SY_o\) = Sediment yield downstream of the VFS

### 6.2.3 Study Area

The new modeling approach was tested for the Gully Creek watershed (43°30'26" N- 43°35'26" N and 81°37'42.19" W-81°39'51.9" W) with a targeted area of 982 hectares (Figure 6.2). The watershed is a significant source of pollution to Lake Huron and has been investigated by several researchers in Canada (Gupta et al., 2018; Golmohammadi et al., 2017). The entire Gully Creek watershed encompasses an area of about 1056.85 hectares. However, part of the watershed has exceedingly flat topography and could not be modeled through the AGNPS-VFS toolkit. Elevation of the watershed varies from 281 m to 217 m at the watershed outlet (Figure 6.2). Mean annual precipitation in the watershed is approximately 1,055 mm (recorded between 2001 and 2011) (Liu, 2013).
The land use in the watershed is mainly dominated by agricultural cultivation. Approximately 41.7% of the watershed is agricultural, 19.41% forest, 28.96% FAL-crops, 6.9% agricultural reserve, 1.3% wetland, 0.95% meadow, and 0.68% urban. The soils in the watershed are comprised mainly of ZAL [Bottom Land (16.58%)], BKN [Brookston Clay Loam (6.53%)], HUO [Huron Clay Loam (55.67%)], and PTH [Perth Clay Loam (21.21%)] (Golmohammadi et al., 2017).

### 6.2.4 Input Data

Geospatial data required to setup the AGNPS and AGNPS-VFS model are: a) Digital Elevation Model (DEM with 5-m resolution), b) Soil data, c) Landuse data, and d) Streamflow path network. These datasets were collected from the Ontario Ministry of Agriculture, Food, and Rural Affairs (OMAFRA), Ontario Ministry of Natural Resources (OMNR), and Ausable Bayfield Conservation Authority (ABCA). Precipitation data (April 2013 to May 2014) used for preparing events to simulate the AGNPS model, was obtained
from a weather station installed within the watershed. (Station established during the Watershed Based BMP Evaluation (WBBE) project in April 2011).

6.2.5 Base Flow Separation

AGNPS being an event-based model computes only surface runoff. Therefore, base flow was separated from streamflow at the outlet of the Gully Creek watershed (GULGUL5) to estimate surface runoff. The WHAT program was used to separate base flow from observed streamflow based upon the method described by Lim et al. (2005). The WHAT program has been used by several researchers for separation of base flow from stream flow (Gupta et al., 2018; Srivastava et al., 2010).

6.2.6 Sediment Load Estimation

Observed sediment loads were available only for a few samples. Therefore, sediment loads for these samples needs to be extrapolated to obtain observed sediment loads for each event. Subsequently, observed sediment loads were compared with AGNPS simulated sediment loads. LOAD ESTimator (LOADEST) a web-based tool was used (Park et al., 2015; Runkel et al., 2004) for this purpose.. The tool estimates monthly sediment loads using observed daily streamflow, observed sediment concentration data (for grab samples collected), and regression model coefficients. Further, the monthly sediment load (concentration) computed using the tool is multiplied by the runoff volume for a particular event. Hence, observed sediment load is computed for that particular event. Here an assumption is made that the sediment concentration for a particular event will be constant over a month. Only the sediment load will change.
LOADEST has eleven regression models, and its coefficients are calibrated using three statistical methods: adjusted maximum likelihood estimation (AMLE), maximum likelihood estimation (MLE) and least absolute deviation (LAD). The AMLE method within LOADEST was used to compute monthly sediment loads. The monthly sediment loads were extrapolated to procure sediment loads for each event based upon the monthly event and storm event durations.

6.2.7 Model Calibration and Validation

The AGNPS model needs to be calibrated and validated with observed data before any scenario analysis. For this study, the calibration was performed separately for peak flow, direct runoff and sediment yield on a seasonal basis for nine storm events from June 2012 to August 2013. Also, validation was performed on a seasonal basis using nine storm events from August 2013 to June 2015. Observed flow data were obtained from the watershed outlet (GULGUL 5). The calibration process was completed by varying AGNPS parameters to match the model-predicted peak flow, direct runoff and sediment yield with its counterpart. The parameters calibrated were the curve number (CN), Manning’s n, C (the crop/vegetation and management factor), and K (the soil erodibility factor), which have considerable influence upon peak flow, surface runoff and sediment yield. The curve number is adjusted based on the antecedent moisture conditions (AMC-I, II, and III) by considering the previous five-day rainfall volumes. Statistical parameters used to evaluate model prediction against observed values were the coefficient of determination ($R^2$) and Nash–Sutcliffe efficiency ($E_{NS}$) (Santhi et al. 2002).
6.3 Post validation

The AGNPS model was calibrated and validated for flow (direct runoff and peak flow) and sediment under the natural conditions. Further, the validated model was used for investigation during the post-validation phase. An approach described in the above section based on AGNPS_VFS toolkit, the regression equation and the AGNPS model is used to accomplish the required task. During the post-validation phase, several scenarios were considered where the VFSs were placed along the edge of the cells draining an upstream area of 3ha, 4ha, and 6 ha. Also, another scenario is considered where VFSs are placed along the spatially distributed buffer cells. The model was run using the example scenarios to visualize the differences in the sediment removing efficiency of the VFS for an event (18 April 2013, Rainfall Event: 10h, 35.6mm rainfall). This was done to demonstrate the significance of placing VFSs at several locations within the watershed upon their sediment reducing efficiency at a watershed level through event-based modeling.

6.4 Results and Discussion

6.4.1 Calibration and validation of flow and sediments at the GULGUL 5 outlet

In this study, events from individual seasons were used to calibrate and validate the model on a seasonal basis. The purpose of seasonal calibration is to determine whether events for certain season are simulated better by the model compared to other seasons. Therefore, a method based upon seasonal calibration is proposed in this study for the calibration and validation of storm events. Six storm events each were selected for spring, summer and fall season. Further, the events were bifurcated into three events for calibration and validation respectively. Model parameters shown in Table 6.2 were
adjusted for the calibration events for each season. Statistical parameters were however computed for the total events considered (compounded for all the seasons) during the calibration and validation phase respectively.

Table 6.2: Calibration of AGNPS parameters

<table>
<thead>
<tr>
<th>Process</th>
<th>Parameter</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow</td>
<td>CN</td>
<td>SCS curve number</td>
</tr>
<tr>
<td>Flow</td>
<td>nc</td>
<td>Channel's Manning’s roughness</td>
</tr>
<tr>
<td>Sediment</td>
<td>K</td>
<td>K_factor</td>
</tr>
<tr>
<td>Sediment</td>
<td>C</td>
<td>C_factor</td>
</tr>
<tr>
<td>Sediment</td>
<td>P</td>
<td>P_factor</td>
</tr>
</tbody>
</table>

6.4.1.1 Flow-Seasonal calibration

Seasonal calibration was performed for peak flow and surface runoff volume at the watershed outlet (GULGUL 5). Table 3 summarizes the events selected for calibration and validation for fall, spring and summer respectively. Surface runoff volume was well reproduced by the model with coefficient of determination \( R^2 \) of 0.87 for calibration and 0.91 for validation. Similarly, Nash–Sutcliffe efficiency \( E_{NS} \) of 0.73 for calibration and 0.46 for validation was observed.
Table 6.3: Events selected for calibration and validation based upon different seasons (surface runoff and peak flow rate)

<table>
<thead>
<tr>
<th>Season</th>
<th>Date</th>
<th>Precipitation</th>
<th>Duration</th>
<th>AMC</th>
<th>Manning’s</th>
<th>$Q^\text{Observed/Peak}$</th>
<th>$Q^\text{Simulated/Peak}$</th>
<th>Observed Surface Runoff (m$^3$)</th>
<th>Simulated Surface Runoff (m$^3$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fall (calibration)</td>
<td>14-Oct-12</td>
<td>18.4</td>
<td>20</td>
<td>II</td>
<td>0.02</td>
<td>0.12</td>
<td>0.20</td>
<td>6317</td>
<td>5368</td>
</tr>
<tr>
<td></td>
<td>14-Sep-12</td>
<td>17.2</td>
<td>7</td>
<td>II</td>
<td>0.02</td>
<td>0.07</td>
<td>0.13</td>
<td>792</td>
<td>2684</td>
</tr>
<tr>
<td></td>
<td>20-Sep-13</td>
<td>24.8</td>
<td>5</td>
<td>II</td>
<td>0.02</td>
<td>0.34</td>
<td>0.49</td>
<td>6011</td>
<td>5368</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.18</td>
<td>0.27</td>
<td>4373</td>
<td>4473</td>
</tr>
<tr>
<td>Spring (calibration)</td>
<td>18-Apr-13</td>
<td>30.6</td>
<td>10</td>
<td>II</td>
<td>0.02</td>
<td>2.38</td>
<td>2.78</td>
<td>37471</td>
<td>37572</td>
</tr>
<tr>
<td></td>
<td>12-Apr-14</td>
<td>27.39</td>
<td>7</td>
<td>II</td>
<td>0.02</td>
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<td>1.62</td>
<td>35128</td>
<td>26837</td>
</tr>
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<td></td>
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<td>21</td>
<td>5</td>
<td>II</td>
<td>0.02</td>
<td>0.40</td>
<td>0.48</td>
<td>10685</td>
<td>10735</td>
</tr>
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<td></td>
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<td></td>
<td>1.52</td>
<td>1.63</td>
<td>27761</td>
<td>25048</td>
</tr>
<tr>
<td>Summer (calibration)</td>
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<td>15</td>
<td>I</td>
<td>0.04</td>
<td>1.29</td>
<td>1.92</td>
<td>19174</td>
<td>34888</td>
</tr>
<tr>
<td></td>
<td>16-Jun-13</td>
<td>14</td>
<td>6</td>
<td>II</td>
<td>0.02</td>
<td>0.09</td>
<td>0.08</td>
<td>3500</td>
<td>2684</td>
</tr>
<tr>
<td></td>
<td>28-Jun-13</td>
<td>15.8</td>
<td>8</td>
<td>II</td>
<td>0.02</td>
<td>0.08</td>
<td>0.09</td>
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<td></td>
<td></td>
<td></td>
<td>0.49</td>
<td>0.70</td>
<td>8164</td>
<td>13419</td>
</tr>
<tr>
<td>Fall (validation)</td>
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<td>15</td>
<td>II</td>
<td>0.02</td>
<td>0.13</td>
<td>0.09</td>
<td>5016</td>
<td>5368</td>
</tr>
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<td>18</td>
<td>12</td>
<td>II</td>
<td>0.02</td>
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<td>0.35</td>
<td>16196</td>
<td>8051</td>
</tr>
<tr>
<td>Date</td>
<td>Peak Flow</td>
<td>Runoff</td>
<td>II</td>
<td>III</td>
<td>Average</td>
<td>Direct runflow</td>
<td>6015</td>
<td>5368</td>
<td></td>
</tr>
<tr>
<td>--------------</td>
<td>-----------</td>
<td>--------</td>
<td>-----</td>
<td>-----</td>
<td>---------</td>
<td>----------------</td>
<td>------</td>
<td>------</td>
<td></td>
</tr>
<tr>
<td>12-Nov-12</td>
<td>16.4</td>
<td>9</td>
<td>II</td>
<td>0.02</td>
<td>0.20</td>
<td>0.20</td>
<td>6015</td>
<td>5368</td>
<td></td>
</tr>
<tr>
<td>12-Jun-12</td>
<td>18.4</td>
<td>9</td>
<td>II</td>
<td>0.02</td>
<td>0.28</td>
<td>0.20</td>
<td>6015</td>
<td>5368</td>
<td></td>
</tr>
<tr>
<td>12-Jun-13</td>
<td>19</td>
<td>9</td>
<td>II</td>
<td>0.02</td>
<td>0.24</td>
<td>0.20</td>
<td>6015</td>
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<tr>
<td>Average</td>
<td>0.27</td>
<td>0.21</td>
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<td>6262</td>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Spring (validation)</td>
<td>29-Apr-14</td>
<td>22.4</td>
<td>9</td>
<td>II</td>
<td>0.02</td>
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<td>0.39</td>
<td>18716</td>
<td>8051</td>
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<td>31-May-13</td>
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<td>III</td>
<td>0.02</td>
<td>0.39</td>
<td>0.33</td>
<td>13303</td>
<td>8051</td>
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<tr>
<td>Average</td>
<td>0.48</td>
<td>0.46</td>
<td>17616</td>
<td>9840</td>
<td></td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Summer (validation)</td>
<td>30-Aug-13</td>
<td>18</td>
<td>3</td>
<td>II</td>
<td>0.02</td>
<td>0.26</td>
<td>3203</td>
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</tr>
<tr>
<td>12-Jun-12</td>
<td>18.2</td>
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<td>II</td>
<td>0.02</td>
<td>0.34</td>
<td>0.26</td>
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<td>19</td>
<td>3</td>
<td>II</td>
<td>0.02</td>
<td>0.24</td>
<td>0.26</td>
<td>6829</td>
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<tr>
<td>Average</td>
<td>0.29</td>
<td>0.27</td>
<td>5213</td>
<td>5368</td>
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</tbody>
</table>

<table>
<thead>
<tr>
<th>Calibration</th>
<th>Peak Flow</th>
<th>Direct runoff</th>
</tr>
</thead>
<tbody>
<tr>
<td>R²</td>
<td>0.97</td>
<td>0.87</td>
</tr>
<tr>
<td>ENS</td>
<td>0.89</td>
<td>0.73</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Validation</th>
<th>Peak Flow</th>
<th>Direct runflow</th>
</tr>
</thead>
<tbody>
<tr>
<td>R²</td>
<td>0.91</td>
<td>0.91</td>
</tr>
<tr>
<td>ENS</td>
<td>0.83</td>
<td>0.46</td>
</tr>
</tbody>
</table>
AGNPS underestimated runoff volume for four of the nine storm events selected for model calibration. Comparatively, five out of the nine storm events selected for model validation was underestimated by the model. However, the model efficiency is still better for validation since one of the storm events selected for model calibration (14 September 2012) was highly overestimated by the model. In case of peak flow, coefficient of determination ($R^2$) was 0.97, and Nash–Sutcliffe efficiency ($E_{NS}$) was 0.89 for calibration. The corresponding value for validation was 0.91 and 0.83 respectively. AGNPS overestimated peak flow for events selected for calibration and underestimated for validation. However, the model’s performance was similar for the estimation of peak flow during calibration and validation.

### 6.4.1.2 Sediment-Seasonal calibration

Six events are chosen for each season, three each for model calibration and validation for sediment calibration and validation as shown in table 4. During the model calibration, AGNPS parameters were adjusted such that the model-simulated sediment loads matched with the observed sediment loads. Also, it is a pre-requisite to calibrate and validate the model for flow before initiating the process for any water quality parameter (above section). The parameters varied during calibration for flow, and sediments are represented in table 4. These parameters were manually adjusted for each cell and channel pertaining to that cell until peak flow, direct runoff amount and sediment load simulated at the outlet cell of the watershed (GULGUL 5) matched the observed value.
Table 6.4: Events selected for calibration and validation based upon different seasons [surface runoff (OSR and SSR) and sediment yield]

<table>
<thead>
<tr>
<th>Season (Cal)</th>
<th>Date</th>
<th>Precipitation (mm)</th>
<th>Duration (hours)</th>
<th>AMC</th>
<th>Manning's n</th>
<th>OSR (m$^3$)</th>
<th>SSR (m$^3$)</th>
<th>OSL (tons)</th>
<th>SSL (tons)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fall</td>
<td>14-Oct-12</td>
<td>18.4</td>
<td>20</td>
<td>II</td>
<td>0.02</td>
<td>6317</td>
<td>5368</td>
<td>0.74</td>
<td>1.05</td>
</tr>
<tr>
<td></td>
<td>14-Sep-12</td>
<td>17.2</td>
<td>7</td>
<td>II</td>
<td>0.02</td>
<td>792</td>
<td>2684</td>
<td>0.02</td>
<td>1.27</td>
</tr>
<tr>
<td></td>
<td>20-Sep-13</td>
<td>24.8</td>
<td>5</td>
<td>I</td>
<td>0.02</td>
<td>6011</td>
<td>5368</td>
<td>0.27</td>
<td>1.71</td>
</tr>
<tr>
<td></td>
<td>Average</td>
<td></td>
<td></td>
<td></td>
<td></td>
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<td>4473</td>
<td>0.34</td>
<td>1.34</td>
</tr>
<tr>
<td>Spring</td>
<td>18-Apr-13</td>
<td>30.6</td>
<td>10</td>
<td>II</td>
<td>0.02</td>
<td>37471</td>
<td>37572</td>
<td>19.49</td>
<td>16.8</td>
</tr>
<tr>
<td></td>
<td>12-Apr-14</td>
<td>27.39</td>
<td>7</td>
<td>II</td>
<td>0.02</td>
<td>35128</td>
<td>26837</td>
<td>12.79</td>
<td>15.11</td>
</tr>
<tr>
<td></td>
<td>28-May-13</td>
<td>21</td>
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<td>II</td>
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<td>10735</td>
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<tr>
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<td>Average</td>
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<td></td>
<td></td>
<td></td>
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<td>10.97</td>
<td>13.67</td>
</tr>
<tr>
<td>Summer</td>
<td>01-Aug-13</td>
<td>59</td>
<td>15</td>
<td>I</td>
<td>0.04</td>
<td>19174</td>
<td>34888</td>
<td>19.49</td>
<td>22.89</td>
</tr>
<tr>
<td></td>
<td>16-Jun-13</td>
<td>14</td>
<td>6</td>
<td>II</td>
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<td>2684</td>
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<td>1.25</td>
</tr>
<tr>
<td></td>
<td>28-Jun-13</td>
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<td>8</td>
<td>II</td>
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<tr>
<td></td>
<td>Average</td>
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<td></td>
<td></td>
<td>8164</td>
<td>13419</td>
<td>6.7</td>
<td>8.51</td>
</tr>
<tr>
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AMC: Antecedent moisture condition
OSR: Observed surface runoff
SSR: Simulated surface runoff
OSL: Observed sediment load
SSL: Simulated sediment load
Curve number is an important parameter impacting direct runoff and peak flow and is hence considered to be necessary for model calibration studies (Cho et al. 2008; Liu et al. 2008; Parajuli et al. 2007). Similarly, K (soil erodibility factor in USLE), C (crop/vegetation and management factor in USLE), and P (support practice factor in USLE) are considered necessary for calibrating sediment loads (Choi and Blood, 1999).

For the summer season (event model), measured average sediment loads of 6.7 tons were slightly lower than the simulated sediment loads of 8.52 tons for the calibration period. Similar results were observed for calibration events during the spring and fall period whereas the model overestimated the sediment loads (Figure 6.3). The model was successful in simulating sediment loads for the majority of the calibration events (2012-2014), except for 14 September 2012 and 28 May 2013, where the model substantially overestimated sediment loads.

The overestimation in these two events could be due to the lack of observed samples during this period resulting in under-prediction of observed sediment loads values. Overall, except for some extreme events, the model was able to simulate sediment loads accurately during the calibration period. Performance statistics with \( R^2 \) (0.94) and \( E_{NS} \) (0.83) for all the calibration events combined which, confirms that simulated sediment loads adequately matched observed sediment load at the GULGUL 5 represented in Table 6.4.
Figure 6.3: Observed and simulated sediment loads at the GULGUL 5 outlet (A) Summer season, (B) Fall season, and (C) Spring season
During validation, decent statistical results with coefficient of determination (R²) of 0.92 and Nash-Sutcliff coefficient (E_{NS}) of 0.56 were obtained. Measured average sediment loads of 1.87 tons was slightly higher than the simulated load of 1.55 tons for the validation events during the summer season. During spring and fall periods, the model slightly underestimated the sediment loads represented in Figure 6.3, and Table 6.4. The model was successful in simulating sediment loads for most of the events during the validation period (2012–2014), except for 17 October 2013, where the model substantially underestimated sediment loads. The underestimation in this event is due to large storms, which might not have been recorded accurately by the weather station. Rainfall recorded for this event was 18 mm (which might have been measured inaccurately). Therefore, to get a better understanding about the performance of the model, data for this event was removed from the analysis.

6.4.2 Post-Validation

In this section, results procured from the calibrated/validated AGNPS model was further tested using the modeling approach developed to simulate a few scenarios. Four different scenarios were considered where VFSs were placed at different locations within the watershed.

6.4.2.1 Scenario S1

Abatement of sediment loads by overland flow process was affected due to the different scenarios of VFS adopted. When a VFS of constant length of 5-m width were applied along cells having a draining area of 3 hectares in the watershed, reduction in sediment load was about 9.59 % (Table 5), reducing 4.07 tons of sediment entering into the stream system. The total amount of sediment entering the stream network of the
watershed is reduced from 42.46 tons (without VFS) to 38.39 tons. Graphical representation of this scenario upon sediment yield within the watershed is represented in Figure 6.4 a. The target approach removed sediment load with a maximum efficiency of 100% mostly when the VFS is adopted along buffer cells to the stream (Figure 6.4 a).

Further, the total area of the VFS constructed under this scenario is 0.315 hectares, which is approximately 0.032 % of the total watershed area. Applying VFS under the current scenario yielded decent results with an overall SRE of 9.59 %. However, since the area of the VFS to be constructed is only 0.315 hectares, it’s proves to be a good BMP option; considering the cost of the VFS constructed and its SRE procured at a watershed scale.

6.4.2.2 Scenario S2

A scenario of constructing a VFS along cells draining a catchment area of 4 hectares was also considered. SRE within the watershed is revealed in Figure 6.4 b. The maximum sediment removal efficiency is reduced to 92%, while the overall sediment removal efficiency increases to 12.39 % under this scenario [from 42.46 tons (without VFS) to 37.2 tons under scenario S2]. (Table 6.5). The total amount of sediment loads entering the stream system reduces by 5.26 tons. Results demonstrate that the SRE under this scenario has increased compared to scenario S1. This is expected since VFSs are constructed along cells draining an area of 4 hectares. Hence, the VFSs would be intercepting more sediment loads, thereby removing a better chuck of the load. Further, the total area of VFS constructed also reduces to 0.252 hectares (approximately 0.026 % of the watershed area). Adoption of VFS under this scenario increases the overall SRE from 9.59 % to 12.39 %. Also, the area of the VFS to be constructed decreases from
0.315 hectares to 0.252 hectares. Therefore, this is a better BMP; accounting for the cost of the VFS constructed and its SRE obtained at a watershed scale.

### 6.4.2.3 Scenario S3

Under this scenario, VFSs are constructed along cells draining an upstream area of 6 hectares. The maximum sediment removal efficiency reduces further to 88.46% and the overall SRE to 5.91% reducing the total amount of sediments entering the streams from 42.46 tons (without VFS) to 39.95 tons. Although the VFS constructed along cells draining an area of 6 hectares are expected to intercept more sediment load and increase the SRE, the total number of cells having a drainage area of 6 hectares are less in this watershed. Therefore, the SRE under this scenario is less than that of scenarios S1 and S2.

Further, the area of the VFS constructed under this scenario is 0.144 hectares (approximately 0.0146% of the watershed area). Although, the area of the VFS to be constructed decreases from 0.252 to 0.144 hectares but its SRE is also the least amongst the three scenarios: S1, S2, and S3. Therefore, this BMP in not recommended considering the cost of the VFS constructed and its SRE obtained at a watershed scale.

### 6.4.2.4 Scenario S4

The final scenario considered where 5-m constant width of VFSs are constructed along spatially variable locations within the watershed. The spatially variable location includes cells located along the stream buffer and cells draining an area of 8 hectares. The maximum SRE obtained under this investigation was 100 % with an overall SRE of 23.03 % which is the maximum amongst all the four scenarios simulated. Therefore, the
total sediment load entering the stream network reduces to 32.68 tons from 42.46 tons (without VFS). A detailed comparison of the results provided in Table 6.5 reveals that within this watershed, the overall SRE predicted for scenarios (S1 to S4) differ substantially. However, the overall SRE procured is the finest amongst all the scenarios evaluated in this study. These results are expected since VFS are mostly constructed along buffer cells in this scenario which drain directly into the stream network. Therefore, VFS would be intercepting a good quantity of sediment load accumulated over the concentrated flow-path. Also, the total area of VFS constructed would increase to 0.819 ha (approximately 0.083 % of the watershed area).

Table 6.5 shows the area of the VFS considered for these scenarios differs significantly. It is observed that construction of a VFS with a uniform width along the spatially variable cells for the entire watershed (S4) would be the most expensive, since it occupies the most area. Also, because the VFS would be adopted along buffer cells, it is more efficient compared to the construction of filter strips along the edge of the field (S1, S2, and S3). Analyzing the SRE and the area of the VFS constructed, this scenario appears to be a best BMP amongst all the scenarios considered and is therefore recommended within this watershed.
Figure 6.4: Spatial distribution of VFS sediment removal efficiency based on the optimized filter width (A) Scenario S1, (B) Scenario S2, (C) Scenario S3, and (D) Scenario S4.
Table 6.5: Scenarios and results

<table>
<thead>
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<th>S. No</th>
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<th>Total Sediment Entering the Stream System (ton)</th>
<th>Maximum Removal Efficiency (%)</th>
<th>Overall Removal Efficiency (%)</th>
<th>Total VFS Area (ha)</th>
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<td>N/A</td>
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<td>2</td>
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<td>38.39</td>
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<td>9.59</td>
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<td>3</td>
<td>S2</td>
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<tr>
<td>4</td>
<td>S3</td>
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<td>88.46</td>
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<tr>
<td>5</td>
<td>S4</td>
<td>32.68</td>
<td>100</td>
<td>23.03</td>
<td>0.819</td>
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</table>
6.5 Conclusions

This article describes a watershed-scale water quality modeling approach, referred to as the AGNPS-VFS toolkit, to simulate the performance of the edge-of-the fields vegetative filter strips. The results procured using our new modeling approach for a range of storm events closely matched observed flow and water quality data. The following observations and conclusions are drawn:

1. AGNPS performed well, both for the calibration and validation events. Also, for the scenarios considered during the post-validation phase, the model-simulated sediment loads closely matching expected water quality benefits.

2. Further, for VFS simulated under the 3\textsuperscript{rd} maximum SRE was achieved.

In summary, this study showed that the modeling approach developed could be useful in simulating the SRE at a watershed scale and to optimize the strategic placement of the VFS along the edge of the field to achieve intended water quality improvements. However, it should be noted that this is just one study conducted in the cold climate of Canada. Also, in this study a simple regression model is developed which considers the SRE as a function of the drainage area of the cells. Similar studies under different regions of the world with varying climatic regions and with a few more independent variables like sediment particle size, etc. This would further strengthen the conclusions of our study.
7 Conclusion and Recommendations

This study focused on (1) development of a toolbox, CoBAGNPS for simulating the impact of WASCoBs upon flow at a watershed scale (2) comparing the impact of two type of surface inlets used for WASCoBs, pipe risers and blind inlets upon flow and sediment yield at a watershed scale using the toolbox developed (3) analyzing the impact of drainage ditches upon downstream flow and sediment yield using the KINEROS 2 model (4) developing an approach to model the impact of edge of field vegetative filter strips at a watershed scale.

7.1 Conclusions

The outcome of this research has provided a methodology to analyze impact of WASCoBs (especially surface inlets), road-side ditches, and VFS at a watershed scale. The application of the results of this study will help in properly analyzing the impact of several BMPs [WASCoBs along with surface inlets (pipe risers and blind inlets), road-side ditches, and vegetative filter strips] in routing flow and sediment loads. Adequately analyzing the impact of these BMPs will aid in the development of economically viable and environmentally friendly management plans for NPS pollution abatement.

The following conclusions have been drawn from this study.

1. A successful modeling approach was developed (though the development of a toolbox, CoBAGNPS) using the AGNPS model which could simulate WASCoBs at a watershed-scale.
2. The developed modeling approach was successful in routing flow and sediments through different type of surface inlets and drainage pipes.
3. For the watershed investigated, pipe risers were more effective than blind inlets in detaining sediments within the berm. Results demonstrated that the presently installed drainage pipes cannot drain a 2-year, 5-year, and 10-year storm events efficiently.

4. The toolbox was efficient in routing sediments through pipe risers and blind inlets. Results confirmed that pipe risers were more effective compared to blind inlets in detaining sediments within the berm.

5. The modeling approach developed using the KINEROS 2 model was successful in simulating flow and sediment loads under distinct drainage patterns. Results confirmed that when road-side ditches when lined with a thicker vegetation result in reduction of direct runoff, peak flow, and sediment yield.

6. A successful modeling approach was developed using AGNPS, AGNPS_VFSMOD for simulating edge of the field VFS at a watershed scale.

7. The Results indicated that when vegetative filter strips were placed at spatially variable locations along the edge of fields, maximum sediment removal efficiency is observed.

### 7.2 Recommendations for future research

1. Development of a nutrient transport module is required for the CoBAGNPS toolbox which could also simulate the impact of WASCoBs in transporting phosphorous and nitrogen at a watershed scale.

2. Similar toolbox needs to be developed which could simulate the impact of WASCoBs, along with their surface inlets especially pipe risers and blind inlets through continuous models.
3. A software program should be developed which could automate the methodology developed to analyze the impact of edge of the field vegetative filter strips at a watershed scale.

4. A methodology should be developed to simulate the impact of the edge of field vegetative filter strips in removing phosphorous at a watershed scale.

5. This research focussed on analyzing the impact of RSD upon flow and sediment. Impact of RSD should be analyzed upon nutrient transport like phosphorous and nitrogen.

6. Further, in this research, an event-based modeling approach was used to simulate impact of RSD upon downstream flow and sediment. A continuous modeling approach should also be used for the same analysis pertaining to RSD.
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