The Impact of Spatial Decision Variables Influencing Crop Rotation on Phosphorus Load Reduction: A Hydrologic Modeling Approach

By

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Author's declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

Non-point source anthropogenic nutrient loading through intensive farming practices is a global source of water quality degradation by creating harmful algal blooms in aquatic ecosystems. Phosphorus, as the key nutrient in this process, has received much attention in different studies as well as conservation programs aimed at mitigating the transfer of polluting nutrients to freshwater resources. Central to conservation initiatives developed to maintain and improve water quality is the application of the Conservation Practices (CPs), introduced widely as practical, cost-effective measures with overall positive impacts on the rate of nutrient load reductions from farmlands to freshwater resources.

Crop rotation is one of the field-based BMPs applied to maintain the overall soil fertility and preventing the displacement of the topsoil layers by surface water runoff across the agricultural watersheds. The underlying concept in the application of this particular BMP is a deviation from the monoculture cropping system by integrating different crops into the farming process. This way, cultivated soils do not lose key nutrients, which are necessary for crop growth, and the overall crop productivity remains unchanged in the landscape. The successful implementation of crop rotation highly depends on planning the rotation process, which is influenced by a variety of environmental, structural, and managerial factors, including the size of farmlands, climate variability, crop type, level of implementation, soil type, and market prices among other factors. Each of these decision variables are subject to variation depending upon the variability of other factors, complexity of watersheds upon which this BMP is implemented, and the overall objectives of the BMP adoption.

This study aims to investigate two of these decision variables and their potential impacts on phosphorus load reductions through a scenario-based hydrologic modeling framework developed to assess the post-crop rotation water quality improvements across the Medway Creek Watershed, situated in the Lake Erie Basin in Ontario, Canada. These variables are the spatial pattern of crop rotation and its level of implementation, assessed at the watershed scale through the modifications made to the delineation of the basic Hydrologic Response Units (HRUs) in the modeling process as well as certain assumptions in the management schedules, and decision rules required for the integration of crop rotation into the proposed modeling framework and optimal placement of this non-structural BMP across the watershed. The main modeling package utilized in this study is the Soil and Water Assessment Tool (SWAT), used in conjunction with the ArcGIS and IBMSPSS tools to allow for spatial assessment and statistical analyses of the proposed hydrologic modeling results, respectively.

Following in-depth statistical analyses of the scenarios, the results of the study elicit the critical role of both factors by proposing optimal ranges of application on the watershed under study. Accordingly, to achieve optimal implementation results compared to the baseline scenario, which has the zero rate of implementation, conservation initiatives in the watershed are encouraged to consider the targeted placement of crop rotation on half of the lands under cultivation. Despite, having a statistical significant impact on water quality compared to the baseline scenario, the random distribution scenario is less effective than the targeted scenario in mitigation of total phosphorus load. Similarly, compared to medium rate of implementation the targeted placement in a higher proportion of the cultivated areas did not lead to statistically significant results but may be considered depending upon the purpose and scope of implementation.

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List of Abbreviations

AAFC........Agriculture and Agri-Food Canada

APF........Agricultural Policy Framework

BMP........Best Management Practice

CSA........Critical Source Area

DEM........Digital Elevation Model

DIP........Dissolved Inorganic Phosphorus

DOP........Dissolved Organic Phosphorus

DP........Dissolved Phosphorus

HAB........Harmful Algal Bloom

HRU........Hydrologic Response Unit

LPUAI........Load Per Unit Area Index

LULC........Land Use Land Cover

MNRF........Ministry of Natural Resource and Forestry

MUSLE........Modified Universal Soil Loss Equation

N........Nitrogen

NAB........Nuisance Algal Bloom

NPS........Non-Point Source

OMAFRA........Ontario Ministry of Agriculture, Food and Rural Affairs

P........Phosphorus

PIP........Particulate Inorganic Phosphorus

POP........Particulate Organic Phosphorus

PP........Particulate Phosphorus

PTF .... PedoTransfer Function

SWAT .... Soil and Water Assessment Tool

SWOOP .... Southwestern Ontario Orthophotography

TN .... Total Nitrogen

TP .... Total Phosphorus

USDA-NRCS .... United States Department of Agriculture-Natural Resources Conservation Service

UTRCA .... Upper Thames River Conservation Authority

VFS .... Vegetative Filter Strip

Chapter 1. General Introduction and Problem Statement

Phosphorus is a critical element required by all organisms throughout their life cycle (Graham & Duce, 1979). Crops are no exception in this regard, and as such, inputs of P are integral to profitable crop production (Sharpley et al., 2000). When compared to the other essential nutrients in the lifecycle of plants, phosphorus is less mobile and bioavailable in different soil conditions, including soil pH level and the frequency of anions and metal elements such as Fe, Ca, and Al in the soil. Thus, it is perceived as the most critical limiting factor in the lifecycle of plants (Hinsinger, 2001)

However, excessive phosphorus is highly detrimental to freshwater ecosystems (Pierrou, 1976), and some of these aquatic ecosystems have been heavily influenced by excessive P loadings, mainly from agricultural runoff induced by the growing intensity of farming practices (Reid et al., 2018). These modern farming practices, derived heavily by the monoculture cropping system, are recognized as the main source of the diffuse phosphorus loads, ultimately washed to standing and flowing bodies of water (Withers & Jarvis, 1998; Vadas et al., 2005). This excessive loads, in turn, degrade freshwater resources by creating nuisance algal blooms and hypoxic zones across aquatic ecosystems (Reid et al., 2018), leading to financial implications for fishing, recreation, and industrial sectors (Haygarth et al., 2005).

The Great Lakes Region of North America is an ecological complex highly influenced by the anthropogenic activities disturbing the natural P cycle. Alterations in the land cover in this area during the past century have led to the excessive transport of polluting nutrients to Lake Erie, as the most sensitive great lake in this watershed (Hanief & Laursen, 2019). The last 60 years, in particular, has been critical in the episodic degradation of aquatic quality in this lake due to the accelerated rate of cultural eutrophication (Curl, 1957; Dolan & McGunagle, 2005).

Given the importance of phosphorus as a contaminating nutrient, and the difficulties of reducing algal blooms through remedial measures, preemptive mechanisms are highly preferred when it comes to dealing with causes of eutrophication across agricultural watersheds (Sharpley et al., 2000; Reid et al., 2018). Therefore, a variety of farm-based control measures or Best Management Practices (BMPs), have been initiated to mitigate excessive phosphorus loadings to freshwater resources from agricultural watersheds. As different landscapes, represent different types of physical, environmental, and economic challenges which are time and space-dependent, the optimum implementation of the BMP measures requires a thorough appreciation of the spatiotemporal factors influencing these variabilities from field dynamics (Gburek et al., 2000) to the broader watershed-scale processes (Withers et al., 2000). For instance, failure to identify the most suitable BMP for a given area would result in less observed impacts across the watershed. Similarly, if critical sources of pollution are not correctly identified, BMPs might only be applied to portions of the landscape with smaller contributions to the nutrient contamination, thus resulting in less tangible impacts when these initiatives are assessed at the larger watershed scale. Lastly, failure to assess the total implementation level would result in insufficient rates of the application when small areas across the watershed are considered for BMP implementation (Liu et al., 2017).

This study was designed by considering the potential impacts of the pattern of spatial distribution and the level of implementation of crop rotation, as the most widely used BMP for sustainable crop production across the Medway Creek Watershed - an agricultural landscape in the Thames River Basin in southwestern Ontario - on the phosphorus loadings originating within this farming landscape and ultimately flowing to the Lake Erie.

Chapter 2. Review of Literature

Both point and diffuse sources of pollution bring about the contamination of freshwater resources. Agricultural lands as non-point sources of contamination, contribute a significant share of the nutrients, particularly phosphorus, transferred to surface water systems. From this perspective, understanding the interaction between agricultural BMPs and P transfer is crucial in mitigating the P loss from farming areas. In this chapter, a description of the terrestrial P cycle, including sources and types of soil P and its role in the agricultural system, is provided. This description is then followed by an explanation of phosphorus transport pathways (i.e., surface and subsurface runoff). Following these initial concepts, BMPs are introduced and discussed concerning the factors affecting their functionality and impact on P loading into water bodies. A detailed review of the literature is then presented on crop rotation as a widely adopted practice, and its role is delineated concerning the cropping system. Finally, the current gaps in the literature are identified and discussed as a basis for the following chapters of the thesis.

2.1 Terrestrial P Cycling

As a non-bioavailable element with limited transport capacity, phosphorus (P) can be found, in small amounts, in the Earth's crust (Filippelli, 2008). It is a non-substitutable component of biological systems (Pierrou, 1976) and an essential nutrient for both terrestrial and aquatic life, often functioning as a nutrient controlling the productivity of plants (Graham & Duce, 1979). There is a limited concentration of plant-accessible phosphorus in soil (free orthophosphate ions PO_4^{3-}) because it mainly exists as insoluble forms (e.g., apatite and other metal complexes). Leaching of phosphorus as phosphate ions and erosion of soil containing phosphorus are the processes responsible for diminishing the amounts of plant-accessible phosphorus (Pierrou, 1976). As P is an indispensable element of the natural ecosystem, understanding the terrestrial P cycling is of the utmost importance.

To better understand the P cycle on Earth, the human impact on the availability of P for ecosystem functioning must be studied. In the pre-human era, phosphorus was present in small concentrations in a few P-bearing minerals, including igneous rocks like fluorapatite and small euhedral crystals associated with ferromagnesian minerals. "In sedimentary rocks, P is typically associated with authigenic carbonate-fluorapatite" (Filippelli, 2008, p.90). The daily accessibility of terrestrial plants to phosphorus becomes limited when liberated P is sequestered in recalcitrant phases (Filippelli, 2008). The anthropogenic terrestrial P cycle, however, dramatically differs from the pre-human cycle as its dissolved form is abundant in rivers (Compton et al., 2000), and there are higher loads of P bearing particulates (Filippelli, 2008). The former is highly influenced by human activities producing agricultural and industrial runoff, which results in Dissolved Inorganic Phosphorus (DIP) concentrations of 100 to 700 μg/L (Meybeck, 1993; Compton et al., 2000). This amount, as Filippelli (2008) argued, is twice the natural quantities.

Phosphorus has no stable atmospheric gas phase (Filippelli, 2008). Thus, the majority of sources of atmospheric phosphorus have been regarded to be high-temperature combustion of organic material. The eroded soil particles that could be found in the spores, fungi, and pollen, which are resulted from biological activities, could be considered as another potential source of atmospheric phosphorus. Also, sea-salt particles derived from the oceans and emissions, due to the anthropogenic activities, are considered as other sources of atmospheric P (Pierrou, 1976; Graham & Duce, 1979; Schlesinger, 1997). Past experiments (Bertine & Goldberg, 1971) illustrated that the significant share of atmospheric P exists as dust and sea-spray. Therefore, unlike other critical bio-nutrients, such as carbon and nitrogen, ecosystems rely on the aquatic transfer of phosphorus

(Pierrou, 1976; Filippelli, 2008). Consequently, "weathering of phosphorus and its delivery and fate in the oceans are important aspects of the phosphorus cycle to understand" (Compton et al., 2000, p.21). The transport of P in aquatic environments is dominated by four forms of phosphorus, namely Dissolved Organic Phosphorus (DOP), Particulate Organic Phosphorus (POP), Dissolved Inorganic Phosphorus (DIP), and Particulate Inorganic Phosphorus (PIP) (Compton et al., 2000). Compared to the soil environment, lower concentrations of iron, aluminum, and often calcium in water results in less immobilization of phosphate ions in aquatic environments, including lakes, rivers, and oceans (Pierrou, 1976). This is one of the reasons why eutrophication occurs and causes global environmental problems. Phosphorus, mostly as phosphate ions from industrial wastes, sewage, and detergents, is often directly discharged into aquatic environments where the existing plants and phytoplankton quickly utilize it. When this phosphorus loading occurs at an extensive rate, nitrogen becomes the limiting factor for the plants and algae for a short period. To recreate the balance in the environment with elevated phosphorus amounts, blue-green algae and other nitrogen-fixing organisms start to multiply. The result of this aquatic balancing reaction, according to Pierrou (1976), is an accelerated plant and plankton growth in the waters - the phenomenon known as eutrophication. From release to burial at aquatic ecosystems, phosphorus undergoes a convoluted journey which is mainly associated with interaction with particles that exist in Earth's biosphere (Compton et al., 2000). On the other hand, the interventions made by human activities on the natural ecosystems have an enormous influence on the terrestrial P cycling, thus adding extra complexity to its cycle on the Earth.

2.1.1 Source and Forms of P in Soil

In nature, phosphorus occurs exclusively as phosphate in all known minerals with an ionic form of PO_4^{3-} (Holtan et al., 1988). It can be found at levels of 400 – 1200 mg/kg of soil (Rodriguez &

Fraga, 1999). "Soil P bioavailability is determined by reaction with hydrous oxides, amorphous and crystalline complexes of Al, Fe and Ca, and organic matter" (Sharpley, 1995, p.261), each of which could be considered as a contributing factor to different forms of P in soil.

Soil P exists in organic and inorganic forms. The latter is "dominated by hydrous sesquioxides, amorphous, and crystalline Al and Fe compounds in acidic, noncalcareous soils and by Ca compounds in alkaline, calcareous soils. Organic P forms include relatively labile phospholipids, inositols and fulvic acids, while more resistant forms are comprised of humic acids" (Sharpley, 1995, p.264). While a large proportion of inorganic phosphorus exists in diverse soil minerals, soil solution has a lower concentration of inorganic phosphorus (Hinsinger, 2001). According to Holtan et al. (1988), a significant part of the phosphates in the soil is incorporated into soil organic matter or sorbed to soil particles. Several processes are responsible for P released by weathering in soil. First, biochemical respiration releases CO_2, which in turn creates an acidic environment around degrading organic matters and root hairs. This acidic environment can easily dissolve poorly crystalline P-bearing minerals and, as a consequence of this chemical reaction, P is released to the root pore spaces. Second, organic acid exudates from plant roots create a ground for dissolving apatite, which then releases P to soil pore spaces (Schlesinger, 1997). Upon dissolution, P is processed into forms less immediately bioavailable, which makes it an integral part of plant tissue where it is transformed into the organic form (Filippelli, 2008). All of these processes influence the P content on the soil profile.

Compared to subsoil levels, the sorption of added P coupled with the aggregation of organic material at surface soil layers results in a higher concentration of P in most soil types. Other factors influencing soil P distribution at different soil layers are texture, parent material, management practices including the level and type of phosphorus, and soil cultivation. Although the proportion of inorganic P can vary in different soil types this rate is above 50%-75% in most soils (Sharpley, 1995). By contrast, organic forms of P may comprise 30% to 50% of the total phosphorus in most soils (Paul & Clark, 1988; Rodriguez & Fraga, 1999). Surface and subsurface runoff decrease the amount of available phosphprus in the soil (McDowell & Sharpley, 2001; Filippelli, 2008). Nevertheless, interpreting soil P dynamics could be very tricky as temporal differences in the soil only represent net P losses and overlook other mechanisms affecting P over time (Frossard et al., 2000). Furthermore, the degree of chemical reactions contributing to the P bioavailability are influenced by soil management and drainage. Soil P dynamics could provide meaningful insights when developing effective management plans as they are critical factors influencing crop production and eutrophication (Sharpley, 1995).

2.1.2 P in the Agricultural System

P-management programs are integral parts of profitable crop production processes. For instance, fixation of phosphorus in organic and inorganic forms unavailable for crop uptake necessitate P amendments to soil in different forms such as manure and chemical fertilizer as well as crop residue material to maintain crop yield objectives. Thus, P amendment has become a critical phase of crop production targeted toward producing adequate food and fiber resources (Sharpley & Smith., 1994). However, substantial amounts of phosphorus have been accumulated in agricultural lands due to the applications of P as mineral fertilizers and manure (particularly in intensive agricultural production) (Richardson, 1994), and this accumulation is at the rate higher than the crop uptake capacity (Holtan et al., 1988; Sharpley et al., 1994; Rodriguez & Fraga, 1999; McDowell & Sharpley, 2001). This amount of P in soil dramatically influences the biogeochemical cycling of phosphorus (Pierrou, 1976) and increases the possibility for P loss through erosion, runoff and leaching (Frossard et al., 2000; McDowell & Sharpley, 2001) which creates several environmental problems such as negative impacts on surface water quality (Foy & Withers, 1995; Sharpley et al., 2000).

The phosphorus in manure and fertilizer applied to agricultural landscapes is partially removed by harvesting plants; some is washed out in the water, while the remaining amounts are accumulated in the soil (Holtan et al., 1988). Once there is an excessive amount of P, identified through Soil Test P (STP) in agricultural fields, the potential for P loss in runoff and drainage water "is greater than any agronomic benefits of further P applications" (Sharpley, 1995, p.269). Over time, even higher amounts of phosphorus are washed out in watercourses as soluble phosphates and erosion products (Holtan et al., 1988). Considerable time is required for the removal of accumulated phosphorus in soil (Sharpley, 1995).

The surface runoff entails sediment-bound P and dissolved P forms (Haygarth & Sharpley, 2000), which are functions of, but not exclusively include, factors such as topography, soil type, soil test phosphorus concentration, and soil hydrology (McDowell & Sharpley, 2001). Sediment-bound or particulate P includes phosphorus attached to soil particles and large molecular-weight organic matters eroded by runoff across the landscape. This includes significant proportions of P transport from most cultivated landscapes (60%-90%) (Pietilainen & Rekolainen, 1991) which is significantly higher compared to runoff from undisturbed ecosystems such as grassland, forest land, or non-erosive soils (Walker & Syers, 1976; St. Arnaud et al., 1988; Sharpley et al., 2000). These non-cultivated sources of surface runoff, however, constitute amounts of dissolved P - although P transport attached to colloidal material may also be important - especially where land is overstocked (Haygarth & Jarvis, 1997; Simard et al., 2000; Sharpley et al., 2000).

Accelerated soil erosion accounts for a significant share of sediment loads to rivers (Berner & Berner, 1987; Compton et al., 2000). Erosion control measures would mitigate particulate and soluble forms of P loss (Withers & Jarvis, 1998). However, mitigating the soluble P loss alone requires more effort compared to measures developed to control the particulate P loss (McDowell & Sharpley, 2001). Although there is insufficient data regarding P amounts in surface soil, surface runoff, and subsurface drainage (McDowell & Sharpley , 2001), identifying soils and management systems with higher potential of P loss and understanding the underlying drivers controlling soil P dynamics are essential to achieving a sustainable agricultural system (Sharpley, 1995).

2.2 P Transport Pathways

Mitigating P loss from agricultural lands cannot be achieved without a full comprehension regarding the forms, transport pathways, and sources of phosphorus across these landscapes (Reid et al., 2018). According to Haygarth and Sharpley (2000), classifying pathways of P transport requires consideration of the scalar as well as spatiotemporal variations of water flow across the landscape. Moreover, conduits along which P is transmitted do not simply act as transport means. Rather, they are a series of reactive hydrologic pathways facilitating P flux transformations by recycling and retention processes (Jarvie et al., 2014; Sharpley et al., 2013). Based on these considerations, Reid et al. (2018) classified P transport pathways into two distinct categories, namely overland flow, also known as surface runoff, and subsurface runoff, which encompasses lateral flow and runoff through tile drainages. These hydrological conduits are functions of site topography and geology (Sharpley et al., 2013). Air serves as another pathway for P transport. However, compared to surface and subsurface runoff, the amount of P transported through the air (in the form of P attached to soil particles) is negligible. That is the reason why this transport pathway is disregarded in many studies (Reid et al., 2018).

2.2.1 Surface Runoff (Overland Flow) and Subsurface Runoff (Tile Runoff and Lateral Flow)

Soil has a limited water absorption capacity. Sometimes, water generated from precipitation and snowmelt exceeds this absorption capacity. This excess amount of water is referred to as surface runoff or overland flow (Reid et al., 2018). Two reasons are contributing to the generation of overland flow across the landscape. Surface runoff could occur due to an imbalance between the precipitation rate and soil's infiltration rate, also known as Hortonian flow (Liu et al., 2004), or it could occur across saturated soils, which is known as saturation excess runoff (Beven, 2001). Therefore, it is possible to argue that runoff across any landscape is a combination of both processes, albeit climate, topography, and soils are factors affecting the dominant process (Reid et al., 2018).

The Hortonian flow - sometimes referred to as the infiltration excess runoff - occurs across landscapes where soil permeability is low, and extreme storm events are the major contributors to precipitation. This type of runoff is sensitive to soil cover (crop type in agricultural lands) and soil management practices, such as the tillage system (Garen & Moore, 2005). Saturation excess flow, in contrast, occurs in landscapes where the water table is higher and closer to the soil surface due to topographic condition or impermeability of soil layers at the surface level (Reid et al., 2018). In contrast, the saturated excess runoff is not sensitive to soil management practices or types of cultivated crops. Rather, "the proportion of precipitation that becomes runoff is related to the proportion of the landscape with saturated conditions" (Reid et al., 2018, p.5). While in models developed based on infiltration excess, the dominant runoff occurs in the upland areas of watersheds, in models considering the saturation excess lower reaches are considered as points where most of the surface runoff originates. Thus, when the goal is to control P loss to surface water, understanding the relative importance of each of these processes across the landscape is critical in identifying the sources of P loss and the design of the associated mitigation efforts (Reid et al., 2018).

Due to multiple contributions from surface and subsurface runoff, the hydrological pathways along which P is transported represent high levels of complexity (Gburek & Sharpley, 1998; Heathwaite & Dils, 2000). In general, most of the studies about the P transfer routes in agricultural landscapes (e.g., Schuman et al., 1973; Kronvang, 1992; Nash & Murdoch, 1997; Fleming & Cox, 1998, 2001; Svendsen et al., 1995; Stevens et al., 1999; Sharpley et al., 1999) have documented surface runoff (overland runoff) as the primary and dominant route for P transport (dissolved and particulate) to surface water. However, according to recent studies (e.g., Heathwaite & Dils, 2000; Chapman et al., 2001; Reid et al., 2012; King et al., 2015a; King et al., 2015b; Kleinman et al., 2015; Zimmer et al., 2016; Jarvie et al., 2017; Macrae et al., 2019), P transport in assisted subsurface (tile) drainage could also be considered as a significant pathway for both water and nutrient loss from agricultural fields. Tile drainage is considered as a useful tool, particularly in many areas with humid summers and cold winters to manage excess soil moisture. Since these systems influence the hydrology of areas in which they have been installed, they could be considered as major modifiers of dominant runoff processes (Macrae et al., 2019). Tile drainage is effective in influencing the water table in the soil and serves as a mitigating factor for the volume of surface runoff through alterations of the time and duration of peak runoff flows (Sloan et al., 2016), all of which influence P transport mechanisms.

2.3 Best Management Practices

Best management practices (BMPs), also known as Green Infrastructure (GI) and Low Impact Development (LID) practices in urban areas, have been widely used since the 1960s (Logan, 1993) as management mechanisms to address issues concerning hydrology and water quality in agricultural as well as urban environments (Gilroy & McCuen, 2009; Ahiablame et al., 2012; Andrews et al., 2013; Liu et al., 2015a, 2015b; Mwangi et al., 2015; Liu et al., 2017). Initially developed to reduce the negative environmental impacts of different urban and agricultural land uses, BMPs are targeted toward maintaining the productivity of lands in these areas (Kincheloe, 1994; Mostaghimi et al., 1997; Merriman et al., 2009). Agricultural BMPs, as practical and cost-effective mechanisms (Hanief & Laursen, 2019), are mainly developed to address a wide variety of issues including protection or restoration of the biophysical and chemical condition of water bodies by, for instance, reducing dissolved pollutant concentration or load, changing hydrology, improving vegetative habitat, reducing particulate/adsorbed pollutant concentration or load, and improving physical habitat (Meals et al., 2010). These functions are applied through reductions in the delivery of agricultural water pollutants (i.e., nitrate and phosphorus and modern pesticides) to surface waters (Logan, 1993).

Tillage management, crop residue management, grassed waterways, terraces, and contouring are among the very first instances of BMP applications addressed in the literature (Walter et al., 1979; Schaller & Bailey, 1983; Clark et al., 1985). These BMPs were primarily aimed at reducing soil erosion (Logan, 1993; Rao et al., 2009). Since the 1970s, BMP applications began to address the load of sediments entering waterways (Walter et al., 1979). Although these BMPs were successful applications of sedimentation reduction, they failed to prevent the dissolved pollutants discharge to water bodies from agricultural runoff (Walter et al., 1979, 2000, 2003; Novotny, 2003). Considering these implementations, Logan (1990) classified BMPs as structural BMPs which remove pollutants after they leave their sources (Getahun & Keefer, 2016), and cultural or management BMPs, also referred to as non-structural conservation practices or source impact BMPs (Novotny, 2003), applied to limit the pollutants transport from their sources (Getahun & Keefer, 2016). Structural BMPs reduce the risk of transferring nutrients to water bodies by either decreasing the dissolved P (DP) contribution from these P source areas (e.g., manure storage ponds) or altering hydrologic pathways away from these areas (e.g., interceptor drainage ditches or subsurface tile drains). In contrast, non-structural BMPs decrease nutrient transport from areas prone to high runoff by reducing or redistributing the potential P load to less vulnerable areas to P loss (e.g., nutrient management) (Rao et al., 2009). However, BMP application per se does not guarantee a desirable outcome (Sharpley, 2015) as there are other factors involved in BMP effectiveness.

2.3.1 Factors Affecting BMP Functionality

Many of the watershed-based conservation programs initiated during the past years have failed to deliver water quality improvement targets (Mulla et al., 2008; Meals et al., 2010; Jarvie et al., 2013). Examples include water quality programs to mitigate the eutrophication impacts in the Mississippi River basin (Dale et al., 2010), the Chesapeake Bay watershed (Reckhow et al., 2011), and the Lake Erie basin (Sharpley et al., 2012). These failures in water quality conservation programs led to fundamental questions concerning the intensity and scale of BMP applications across the watersheds (Sharpley et al., 2009, 2013). Despite the increasing number of research projects studying the implemented BMPs with respect to the watershed management projects (e.g., Hunt et al., 2006; Ahmed et al., 2015; Lewellyn et al., 2016) as well as their impacts on P loads reduction (both PP and DP) (Rao et al., 2009), there are still uncertainties regarding their temporal impact on water quality (Liu et al., 2017). The results of previous studies (e.g., Hunt et al., 2006; Dietz, 2007; Emerson & Traver, 2008; Hoffmann et al., 2009; Emerson et al., 2010; Mitsch et al., 2012, 2014; Ahiablame et al., 2012; Paus et al., 2015; Lewellyn et al., 2016) illustrate significant variations among short and long-term BMP performances. However, most modeling and management efforts leaves out this variability as some of them utilize reduction efficiency factors (Liu et al., 2017). In other words, functionality as a factor of vegetation type, degradation of structures, and pollutant accumulation is not addressed in these efforts (Jackson-Smith et al., 2010; Meals & Dressing, 2015). Other overlooked issues in these management and modeling efforts are the processes involved in BMP implementations.

The variability of BMP performances in short-term may be resulted from the watershed characteristics including landscape, land use, topography, metrology, and hydrology as well as other factors such as local design standards and installation quality (Rao et al., 2009; Bosch et al., 2011, 2013; Liu et al., 2017). The long-term effects of BMPs on hydrology and water quality have been assessed by few empirical studies (e.g., Komlos & Traver, 2012; Mitsch et al., 2012, 2014; Chen et al., 2015), mainly due to time and resource limitations (Dietz, 2007; Koch et al., 2014). Some of these studies (e.g., Bracmort et al., 2004; Uusi-Kämppä, 2005; Uusi-Kämppä & Jauhiainen, 2010) found that the impact of BMPs decreases over long periods. Considering these variabilities in BMP effectiveness are essential in reaching environmental conservation goals (Liu et al., 2017).

Water quality monitoring programs in a non-point source pollution mitigation project in a watershed may not illustrate definitive results even when management changes are well planned and fully implemented (Liu et al., 2017). This could be caused by various factors including the program design, monitoring period, and sampling frequency which may be insufficient to address the lag time, (i.e. the time between the adoption of BMP and the moment the first improvements could be measured in water quality in the target water body) (Meals et al., 2010). "The main components of lag time include the time required for an installed practice to produce an effect, the time required for the effect to be delivered to the water resource, the time required for the water body to respond to the effect, and the effectiveness of the monitoring program to measure the response" (Meals et al., 2010, p. 85). There are some factors affecting lag time. Scale, for instance, is an essential factor that may influence the response rate (e.g., BMP implementations in smaller scales illustrate a more rapid response than the ones applied in a larger setting). Chemical processes (e.g., sorption kinetics in soils or aquatic sediments), physical processes (e.g., sediment transport in streams), and groundwater movement and the type of management (e.g., selection of appropriate BMPs and application of BMPs to critical source areas) are other important factors need to be considered with reference to the nature and speed of response in water quality (Meals et al., 2010).

2.3.2 Impact of BMPs on P Loading into Water Bodies

Management plans should prioritize surface runoff, and particulate transport strategies as P transport generally occurs in the surface runoff pathway of watersheds by mobilizing sediments together with P applied as fertilizer or manure across farming landscapes. P loss resulting from this mobilization could be mitigated by management practices controlling the velocity of surface runoff (e.g. through terracing or contour cropping implemented to reducing the land slope or lengthening the flow path the water takes), reducing the overall volume of surface runoff, or contributing to sediment trapping and infiltration (Walter et al., 1979; Gburek et al., 2000). Therefore, any management plan needs to be carried out, considering not only source areas and transport pathways but also the reasons for the P loss across farming areas. This requires control over both P inputs, which is also referred to as nutrient management plans and control over P transport, which is considered as land management practices. The former is required to mitigate the losses of soluble P in land runoff and unpredicted shocks such as storms, particularly after the application of manure or fertilizer to the fields. The latter is required to manage the loss of particulate P before it enters the bodies of water due to soil erosion (Withers et al., 1998).

When phosphorus load reduction is considered, BMP-led water quality improvement measures may require several years before an impact could be measured across agricultural watersheds (Boesch et al., 2001; Wang et al., 2002; Rao et al., 2009). This is primarily due to the slow rate of landscape P transport, as well as the possibility of P accumulation in soil and stream sediments (Bishop et al., 2005). Sharpley et al. (2013) also argued that the limited responses of water bodies to the conservation measures to reduce P losses in many cases (e.g., Meals et al., 2010; Hamilton, 2012; Spears et al., 2012) have been due to legacies of past management activities, where the accumulated P along the land–freshwater continuum impedes the impact of management practices. For instance, In Manitoba, Canada, some vegetated buffer strips contained more surface soil P levels (33%) than source fields, and the runoff P level was higher (18%) in these instances (Sheppard et al., 2006). Similarly, while no-till management was successful in decreasing the erosion and particulate P runoff across wheat fields in Oklahoma (Sharpley & Smith, 1994) and soybean fields in Ohio (Richards et al., 2009; Sharpley et al., 2012), P in surface soils was identified as the source of dissolved P in the agricultural runoff in these areas. Therefore, despite these measures, soil and fluvial sediment P stores, which were accumulated ten to twenty years before the BMP implementations, could postpone the success of the present-day conservation measures (Sharpley, 2015).

2.4. Crop Rotation as an Agricultural BMP

As a fundamental agronomic mechanism with a long history of practice across farming landscapes, crop rotation plays an indispensable role in the sustainability of agricultural systems (Chambers & Mingay, 1966; Vandermeer et al. 1998; Castellazzi et al., 2008; Schönhart et al., 2011). The USDA Natural Resources Conservation Service (USDA-NRCS) defines crop rotation as a planned sequence of annual or perennial crops (Merriman et al., 2009) cultivated in a specific agricultural land. Likewise, Bullock (1992), Wibberley (1996), and Castellazzi et al. (2008) described crop rotation as a cyclical and temporal arrangement with pre-defined order for each crop, which has a fixed length. Crop rotation, therefore, is characterized by a cycle period/pattern (Leteinturier et al., 2006) and interval variability (Castellazzi et al., 2008). Based on these characteristics, Castellazzi et al. (2008) proposed four types of rotations based on the flexibility of their structure. The range of considered typologies varies from crop successions with a pre-defined fixed structure to an unformed rotation that is responsive to potential fluctuating environmental or market circumstances (Tsai et al., 1987; Klein Haneveld & Stegeman, 2005; Detlefsen & Jensen, 2007; Castellazzi et al. 2008). These types are (1) cyclical and fixed (static rotation) with a fixed rotation length (return period) in which each crop follows a pre-defined order with no potential deviation (Castellazzi et al., 2008), (2) cyclical with fixed rotation length and a flexible crop choice at least for one year of the rotation, (3) cyclical with variable rotation length and (4) high variable rotational length with less structured cycle (Dury et al., 2012). The only limitation associated with succession in the last type is that each year's crop should be different from the preceding year's crop (Castellazzi et al., 2008).

2.4.1. Role of Crop Rotation in Cropping Systems

According to Mignolet et al. (2004) and Leenhardt et al. (2010), cropping systems can be defined at two levels. The first level is associated with crops and their order on a particular field over time, referred to as crop rotation. The second level is the crop management system characterized by an organized series of cultivation techniques including tillage, fertilizer and pesticide application, and harvesting applied to a crop to obtain a given product (Salmon-Monviola et al., 2012). The cropping system analysis could, therefore, begin by crop rotation (Reckling et al., 2016).

The agro-economic and environmental performance of crops is influenced by the changes in cropping systems (Schönhart et al., 2011). "Fertilization, nitrogen mineralization, nitrate leaching, greenhouse-gas emissions, infestations with pests, diseases and weeds, and eventual crop yield are all affected not only by the management of the individual crops but also by long-term processes that are influenced by crop sequence" (Reckling et al., 2016, p.186). For example, continuous cultivation of a pre-defined crop or group of crops under similar management practices would only contribute to the entry and propagation of certain weed species, which gradually becomes difficult to control

(Chauhan et al., 2012). Consequently, rotation plans built around one or two core crops with different characteristics, followed by other crops, are of utmost importance to the overall health of the system (Zegada-Lizarazu & Monti, 2011). The choice of a sequence is, first and foremost, a management decision. It is usually based on optimization prospects targeting agricultural (e.g., yield maximization), environmental (e.g., pesticide use minimization), and financial objectives (e.g., profit maximization), which can be challenging to realize simultaneously (Castellazzi et al., 2008).

2.4.2. Crop Rotation as a Mitigative Measure

The earliest application of crop rotation was largely motivated by financial profit maximization associated with crops with high economic yields (i.e., cash crops) (Bullock, 1992). These applications were associated with enhanced agroecosystem functioning in terms of increased soil fertility (Varvel, 2000; Kelley et al., 2003; Kollas et al., 2015), increased nitrogen supply (Miglierina et al. 2000; Galantini et al., 2000; Schönhart et al., 2011), maintenance of soil structure, disruption of pest cycles and weed suppression (Bullock 1992; Berzsenyi et al., 2000; Carter et al. 2002; Tilman et al., 2002; Carter et al. 2003; Smith et al., 2008). For instance, increased crop yields (Magdoff & Weil, 2004; Ball et al., 2005; Bennett et al., 2012) and decreased soil erosion (Gantzer et al., 1991) leading to runoff based eutrophication (Drinkwater et al., 1998; Gardner & Drinkwater, 2009; Blesh & Drinkwater, 2013; Tomer & Liebman, 2014) could be achieved through increasing the length of corn-and soybean-based rotation systems with forage crops and fine grains while applying organic matter amendments (Hunt et al., 2019). In a broader context, crop rotation, and consequently crop diversification, have been identified as pillars of mitigation-based adaptation strategies to climate change (Olesen et al., 2011), which in turn enhance ecosystem services (Hauck et al., 2014), and the overall resilience of the agricultural system (Smith et al., 2008; Reidsma et al., 2009; Lin, 2011).

Due to emerging soil ecological processes, crop rotation can have a significant impact on soil health (Chauhan et al., 2012). Some studies (e.g., Power et al., 2000; Karlen et al., 2006; Smith et al., 2008) illustrated the impact of different rotation plans on soil nutrients which is mediated mainly by soil microorganisms through complicated biochemical processes (Parkinson & Coleman, 1991; Kennedy & Smith, 1995; Kennedy, 1999). The study by Venter et al. (2016), for instance, concluded that higher microbial richness and diversity scores could be achieved under a higher diversity of crops in rotation. Soil water-use efficiency and physical structure are also influenced by crop rotations (Kollas et al., 2015; Chauhan et al., 2012). These applications are also associated with stabilizing temperature fluctuations of soil through increased soil organic matter content and ground cover (Kennedy, 1999).

Six different categories for agricultural BMPs are defined by Merriman et al. (2009). Based on this classification, crop rotation - as an agricultural BMP - is an erosion control BMP. Crop rotation influences surface water quality (Vaché et al., 2002) by reducing the movement of nutrients caused by water erosion on both the surface and in the shallow profile of the soil (Ni & Parajuli, 2018). The results of the study by Logan (1990) on the evaluation of a wide range of BMPs as pollution control mechanisms, acknowledged crop rotation as a cultural (non-structural) BMP which controls and reduces soil erosion by maintaining soil organic matter (soil quality), thus influencing the nutrient and sediment losses (Broussard & Turner, 2009; Kollas et al., 2015; Smith et al., 2015), and in particular sediment P accumulation in surface water (Logan, 1993).

Soil water balance among specific crop sequences was investigated through crop rotation modeling in several studies (e.g., Post et al., 2007; Salado-Navarro & Sinclair, 2009). This modeling framework was also employed for estimating the amount of nitrate leaching over long periods (Kovács et al., 1995; Kersebaum & Beblik, 2001; Beaudoin et al., 2008), the carbon storage capacity of soils and crops in farming landscapes (Li et al., 1994; Blombäck et al., 2003; Hlavinka et al., 2014), the above/belowground biomass and yields (Berntsen et al., 2006), and the nitrogen uptake (Nendel et al., 2013). Crop rotation has only been considered in conjunction with other BMPs in a limited number of modeling studies on diffuse sources of pollution, soil erosion or water quality to date (e.g., Liu et al., 2016; Getahun & Keefer, 2016; Hanief & Laursen, 2019). However, none of these studies investigated the impact of this BMP as a variable component of the model on the water quality characteristics.

Most of the modeling approaches for crop rotation concentrate primarily on optimizing, explaining, or even predicting a well-structured crop rotation with impacts on soil properties. Venter et al. (2016), for instance, investigated the impact of decreased aboveground crop diversity on belowground microbial biodiversity in different soil layers by conducting a meta-analysis of studies comparing monocultures and crop rotations. They examined a total number of 27 peer-reviewed articles from which 14 reported a neutral effect of rotation on microbial diversity, while nine studies reported a positive effect. This study concluded that increasing crop diversity has a positive effect on soil microbial diversity and richness. In another study, Teixeira et al. (2018), develop a catchment-scale assessment to investigate the interaction of crop rotations and climate change. The results of the study illustrated that due to different timing of sowing and harvesting, the direction and magnitude of climate change impact on different crop rotations are spatially variable. These results provide more in-depth insight into the dynamics of climate change impacts for crop rotation systems.

However, due to the complex interrelationships inherent in environmental systems, a landscape-level approach could potentially lead to significant advantages when issues such as soil erosion, water resources, disease and pest control, crop coexistence, and food safety are the subject of investigation

(Castellazzi et al., 2010). Despite the positive effects of crop rotation on sustainable agricultural practices, mechanisms, and processes associated with this agricultural BMP are still unknown/unclear to researchers. Such uncertainties are even more noticeable when the spatial pattern/distribution impact of crop rotation on water quality is the subject of study. This study aims to fill this essential gap by investigating the implementation level and spatial distribution of crop rotation on P loss in surface runoff from the agricultural watershed.

These factors are investigated in chapter 3 of this thesis.

Chapter 3. The Impact of Implementation Level and Spatial Distribution Pattern of Crop Rotation on P Transfer in Surface Runoff

Abstract

Intensive farming is globally known as the non-point source contributor of eutrophication in aquatic ecosystems through nutrient loadings to freshwater resources. Among all polluting nutrients, phosphorus is recognized as the primary cause of eutrophication across agricultural watersheds. Farming practices highly influence the amount, type, and timing of phosphorus loss to freshwater resources, and as a result, control measures known as BMPs are applied across farming areas to mitigate the excessive P loss to bodies of water in different periods. Crop rotation, as an erosion control mechanism, is one of the most widely applied BMPs implemented across agricultural watersheds to maintain soil quality and crop yield. In this study, we investigated the influence of the spatial pattern of crop rotation placement and its level of implementation, as two critical decision variables, on phosphorus reduction rates in an agricultural watershed in southwestern Ontario, Canada, through a multi-stage, scenario-based hydrologic modeling process using the Soil and Water Assessment Tool (SWAT). The results of this research illustrate that placing crop rotations based upon spatial prioritization and increased level of implementation are positively associated with the overall phosphorus reduction rates. However, the optimal placement of crop rotation is achieved when half of the fields are considered for targeted placement of this BMP by producing statistically significant reduction rates compared to a zero level of implementation.

Keywords: Crop rotation, Phosphorus Load, Level of implementation, Spatial distribution

3.1 Introduction

Eutrophication, a globally critical issue threatening freshwater resources and coastal marine ecosystems, is caused by increased anthropogenic nutrient loading to water resources (Dupas et al., 2015; Michalak et al., 2013). This phenomenon is contributing enormously to the emergence of Nuisance Algal Blooms (NABs) and Harmful Algal Blooms (HABs) (Diaz & Rosenberg, 2008; Seitzinger et al., 2010; Blaas & Kroeze, 2016), leading to ecosystem health degradation, reduced biodiversity, creation of hypoxic zones, and significant economic consequences, particularly in water-dependent economies (Roegner et al., 2014; Bingham et al., 2015; Blaas & Kroeze , 2016; Bullerjahn et al., 2016).

The contamination of freshwater resources can be the result of both point and non-point sources of pollution (Mayorga et al., 2010; Howarth et al., 2011; Blaas & Kroeze, 2016). Eutrophication in surface water systems, to a large extent, results from non-point source (NPS) phosphorus (P) and nitrogen (N), mainly originating from lands under cultivation (Bouraoui & Grizzetti, 2011; Windolf et al., 2012 ; Scavia et al., 2014; Dupas et al., 2015). Due to difficulties in controlling the exchange of N between the atmosphere and water bodies and the impact of blue-green algae on N fixation, much attention has been focused on phosphorus. It is introduced by many studies as the major driving force of accelerated eutrophication in freshwater and marine ecosystems globally (Sharpley, 1995; Schindler et al., 2008; Kane et al., 2014).

The intensity of farming practices in the agricultural landscape of southern Ontario, Canada, can be considered as one of the major causes of eutrophication contaminating freshwater resources in the Lake Erie basin. Eutrophication due to agricultural activities, therefore, is considered as one of the most significant human-induced changes to this ecosystem (ECCC & OMAFRA , 2018). To date, some studies (e.g., Michalak et al., 2013; Kane et al., 2014; Bingham et al., 2015; Bullerjahn et al.,

2016) have shown that the eutrophication of Lake Erie is a recurring phenomenon over the past two decades. This has led to the initiation of a new bilateral agreement between the U.S and Canada in 2012, which was fully enacted in 2016, for P load reduction by 40% from the loading rates recorded in the year 2008 in both nations. (Hanief & Laursen, 2019).

Water quality degradation due to intensive farming practices can be mitigated through the implementation of various Best Management Practices (BMPs), which are recognized widely as mechanisms to control water contamination and nutrient transfer into aquatic environments (Hanief & Laursen, 2019). The impact of these mechanisms on water quality, however, is highly dependent on the characteristics of the watershed, mainly the crops and the lands on which these practices are being implemented (Panagopoulos et al., 2012). All agricultural activities in the Canadian farming landscape are subject to conservation programs based upon the Canadian Agricultural Policy Framework, known as APF (Liu et al., 2016). Currently, a wide variety of BMPs, including crop rotation, cover crops, fragile land retirement, Vegetative Filter Strip (VFS), conservation tillage, windbreaks, and erosion control structures, are being implemented across the agricultural landscape of southern Ontario. Due to the integrated hydrological, ecological, economic and social functions, which interact with one another in a hierarchical structure (Cheng et al., 2014), watersheds are considered as complex systems (Johnson et al., 2002; Weible et al., 2010; Guo Dong & Xin, 2015). Consequently, successful implementation of any of these BMPs depends profoundly on fundamental decision variables, including the location, spatial distribution, and the type of BMPs considered for mitigating the non-point source pollution (Fleifle et al., 2014). Appropriate selection of BMP type and its implementation on agricultural fields across a watershed are two crucial steps for successful mitigation efforts (Getahun & Keefer, 2016). According to Lennox et al. (1997), farming practices would drastically influence the form, amount, and timing of P losses originating from several source areas within watersheds dominated by croplands. Crop rotation, as the primary pillar of farming practices, is a preemptive mechanism limiting excessive P loading to the bodies of water.

As one of the most frequently implemented BMPs in the agricultural watersheds of southern Ontario, crop rotation could target all types of phosphorous accumulations (i.e., sediment P, particulate P, and soluble reactive P) washed to the bodies of water through erosion or agricultural runoff in this region (Holeton et al., 2013). The importance of implementing optimal and economically viable P transport mechanisms is also discussed in a number of studies investigating the strategic targeting of the management strategies to the regions with potential P accumulation (Chang et al., 2009; Qi & Altinakar, 2011; Shen et al., 2013; Rao et al., 2009). In reality, BMP implementation, including crop rotation, occurs at the field scale, and since these implementations are voluntary conservation measures, installing BMPs on a significant proportion of farming landscapes is deemed impossible (Sommerlot et al., 2013). Considering this condition, understanding whether higher rates of implementation, as well as the spatial patterns of BMPs, truly influence the total phosphorus (TP) load reduction across agricultural watersheds is crucial to guide informed decision-making for conservation programs.

The use of computer models is critical in identifying areas with high potential for P transport across watersheds (Vadas et al., 2005). Nevertheless, despite an improved understanding of the sources and transport pathways of pollutant P in the last decades (Gburek et al., 2000; Sims et al., 2000), this approach has not been reflected in many simulation frameworks (Sharpley et al., 2002), leading to the dearth of knowledge in the application of computer models in this topic. Most of the studies (e.g., Panagopoulos et al., 2012; Kurkalova, 2015; Geng et al., 2015) about the location of BMP implementations and their frequency of application are concentrated mainly on evaluating these variables based on the monetary values such as low costs of implementation and maximization of profit from agricultural lands without considering their impact on mitigating the nutrient loads across the landscape. Likewise, a significant number of studies (e.g., Ron Vaz et al.,1993; Heckrath et al., 1995; Haygarth et al., 1997; Chardon et al., 1997; Haygarth & Jarvis, 1999; Daly et al., 2001; Turner & Haygarth, 2001; Maguire & Sims, 2002; Blake et al., 2002) were mainly focused on BMP applications and their impact on a single agricultural plot. These studies rarely investigated plot locations and, consequently, their spatial distribution in the landscape. Therefore, designing and implementing watershed-scale nutrient management programs concerning these variables is yet to be addressed (Kurkalova, 2015).

The goal of this study is to investigate the association of the implementation level and spatial distribution of crop rotation, as a BMP, with the resulting water quality (i.e., TP load) using the Soil and Water Assessment Tool (hereafter SWAT) across an agricultural watershed in the Thames River Basin in southwestern Ontario. The main objective is to create a model representing how the impact of crop rotation as a preventive mechanism for water quality degradation will be influenced by i) spatial distribution, and ii) level of implementation across the selected landscape. Due to the delayed response of water quality to practice implementation, a scenario-based modeling approach is selected to evaluate the impact of these two factors on water quality. Scenarios of crop rotation arrangement developed for this purpose are based on the different implementation levels of crop rotation and targeted versus random distribution of this BMP at the watershed scale.

3.2 Methods

3.2.1 Study Area

As one of the 28 watersheds of the Upper Thames River, the Medway Creek , located in southwestern Ontario, Canada (43° 0' 58.12" to 43° 12' 53.77" N; 81° 16' 7.28" to 81° 19' 5.52" W) is a

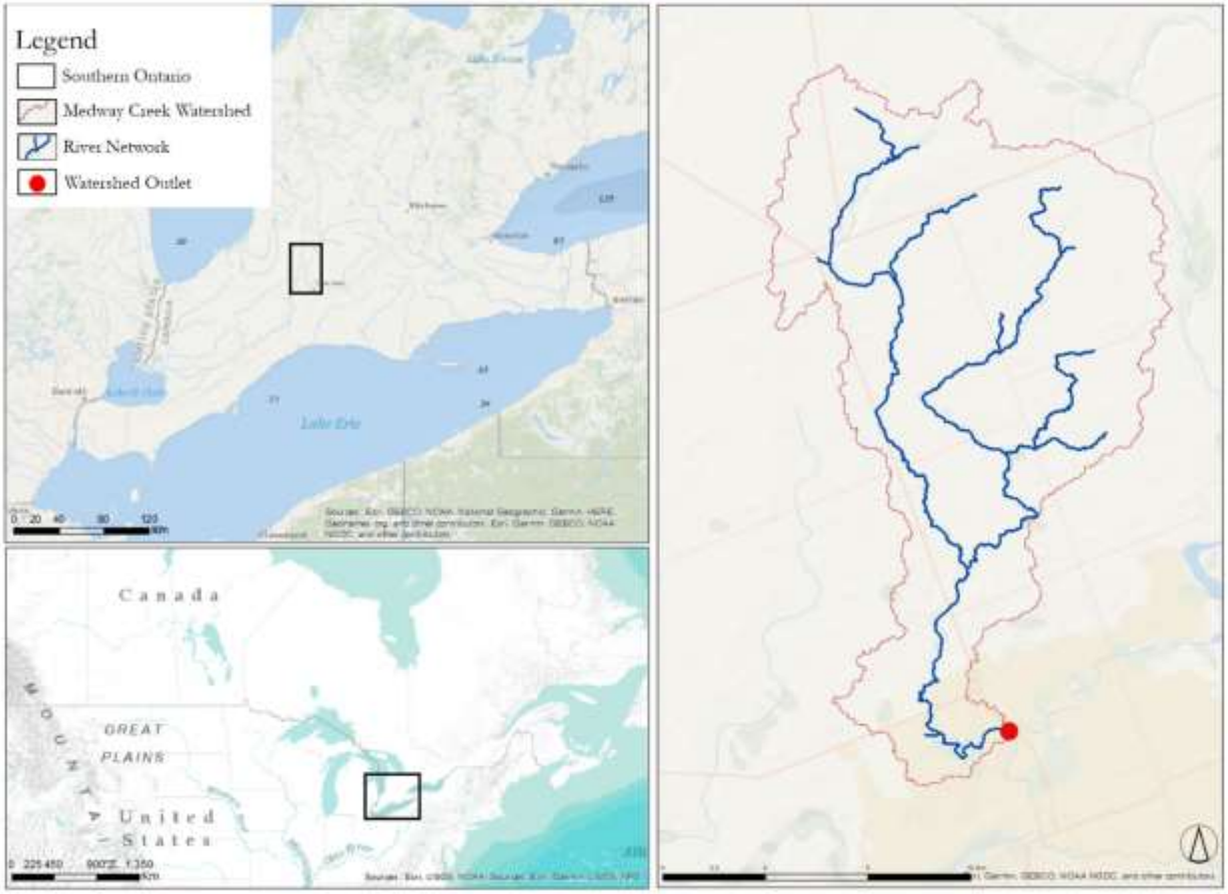

Figure 1. Location of Medway Creek Watershed in southwestern Ontario, Canada

major tributary of the Thames River which, at the larger scale, is part of the Lake Erie watershed (Figure 1).

The Thames River originates northeast of London, Ontario, and flows southwesterly to Lake St. Clair. Water from Medway Creek enters the North Thames River in London and then flows through London and Chatham into Lake St. Clair, which is connected to Lake Erie through the Detroit River (UTRCA, 2017a).

With the total length of 218 km of watercourses, the Medway Creek Watershed covers an area of 205 Km^2 or approximately six percent of the Upper Thames River watershed. The Medway Creek watershed is a predominantly agricultural watershed spanning across portions of four urban areas of southwestern Ontario including Middlesex Centre (65%, 133 km^2), Lucan-Biddulph (20%, 40 km^2), London (10%, 20 km^2), and Thames Centre (6%, 12 km^2) (UTRCA, 2017a). The waters and forests in this region face ongoing pressure from land use changes in the urban-rural gradient of this area (UTRCA, 2017b).

The climate associated with the watershed is classified as humid continental with dramatic seasonal variation. The growing season during which almost 60% of the precipitation occurs as rainfall is marked with an average of 160 frost-free days per year beginning in mid-April and ending in late October. The remaining precipitation throughout winter occurs as snow, with occasional rain events. From 2001 to 2016, the average annual precipitation was approximately 999 mm with a standard deviation of 168 mm and a range of 592 mm with a maximum of 1,262 mm in 2008 and a minimum of 670 mm in 2016. With the standard deviation of 0.8°C, the average annual temperature of the area is 8.4°C which includes the maximum temperature of 10.1°C and the minimum temperature of 6.7°C, recorded in 2012 and 2014, respectively (Watershed Evaluation Group, 2018). The elevation ranges from 230 to 336 m above the sea level with a mean slope of 2 degrees. The northwestern part of the watershed is increasingly sloped due to rolling topography. On the contrary, the southern part is mostly flat urban areas. The soil type in this watershed is mainly clay loam, silty loam, silty clay loam, bottomland, and coarse sand covering areas of 33%, 32%, 20%, 6%, and 3%, respectively. The remaining 6% soil type percentage is not mapped as it is situated in urban land use. Cultivated croplands comprise the majority of land uses, covering approximately 82% of all the watershed area. More than 98% of the cultivated non-fallow crop areas are dedicated to the cultivation of corn, soybean, and winter wheat. Approximately 12% of the watershed is forest, and the remaining is nonagricultural areas (i.e., roads, urban, and water bodies) (UTRCA, 2017a).

The anthropogenic activities of the past century have been degrading the water quality across the Thames River watershed. Among these activities, farming practices, wastewater treatment in urban areas, stormwater runoff, and industrial wastewater are the major sources for surface water quality degradation in the basin (UTRCA, 2017b). This watershed has been recognized as a polluting source to the Great Lakes, particularly Lake Erie, through accelerating the growth of harmful algal blooms, resulting in water quality degradation across these aquatic ecosystems (UTRCA, 2017b). This has resulted in recognition of this watershed as a priority watershed in Canada, according to the updated 2012 Great Lakes Water Quality Agreement for reducing the phosphorus load to water bodies (UTRCA, 2017a). Controlling P export from the Medway Creek Watershed, therefore, would potentially lead to improved water quality in Thames River and subsequently in the Lake Erie.

3.2.2 Data Sources

The SWAT model requires three primary layers of spatial data, including topography, Land Use Land Cover (LULC), and a soil typology. The Digital Elevation Model (DEM) provided by the Upper Thames River Conservation Authority (UTRCA) was used as the source of the topographic information. Created by the Ontario Ministry of Natural Resource and Forestry (MNRF) in 2010, this 10-meter elevation data layer is the product of the Southwestern Ontario Orthophotography Project (SWOOP) (MNRF, 2019). Obtained from the Agriculture and Agri-Food Canada (AAFC) crop inventory for the year 2014, the annual crop type digital map was used as the source of the LULC information. A decision tree algorithm combined with satellite imagery and ground-truthing of data were used in the development of this data layer. To better identify the crop types, optical images were taken during the key stages of reproduction, seed development, and senescence with the spatial resolution of 30 m and the accuracy of 85% (AAFC, 2018).

The soil layer and its physical parameters, including soil texture, soil depths, and bulk density, at a scale of 1:50,000 was obtained from Land Information Ontario (LIO) as a joint project between the Ontario Ministry of Agriculture, Food and Rural Affairs (OMAFRA), Agriculture and Agri-Food Canada (AAFC), and the Ministry of Natural Resources and Forestry (OMAFRA, 2019). The remaining parameters of the soil layer, including soil available water capacity and soil albedo, were obtained from a recent study in the watershed carried out by Hanke (2018). This study utilized PedoTransfer Function (PTF) developed by Saxton and Rawls (2006) and the ranges mentioned by Dobos (2003) to estimate soil available water capacity and soil albedo, respectively. Required climate data, including precipitation, temperature, solar radiation, wind speed, and relative humidity, were obtained from the above mentioned study.

3.2.3 Documentation of the Dominant Crop Rotation Pattern by Time Series Analysis

Since field-specific crop rotation data was not available, a trajectory analysis method (Wang et al., 2012) was utilized for determining the LULC changes in multiple time nodes for the area under study. More specifically, this method analyzes the entire LULC maps in five different time nodes to identify variability and dynamic changes, which are represented as time series. To create these time series, each LULC was assigned numeral/letter trajectory codes. Time series analysis allows one to understand the spatiotemporal variability in the LULC and to determine the trend and pattern of changes in the considered LULC (Wang et al., 2012). The total number of LULC types is a critical factor considered when defining the formula for calculating the trajectory value. When trajectory codes are smaller than or limited to 9, the following formula is applied to calculate the trajectory value for each pixel in the raster layer.

$$T_{ij} = (G_1)_{ij} \times 10^{n-1} + (G_2)_{ij} \times 10^{n-2} + (G_3)_{ij} \times 10^{n-3} + \ldots + (G_n)_{ij} \times 10^{n-n}$$

In this formula, T_{ij} is the trajectory value assigned to a pixel at row i and column j, n is the number of time nodes, and $(G_n)_{ij}$ is the trajectory code of each LULC type (Wang et al., 2011).

Units of application are modified from pixel base to agricultural field boundary to define the crop rotation in this study. The following formula is the modified version of the preceding mathematical illustration used for the calculation purposes in this study.

$$T_i = (G_1)_i \times 10^{n-1} + (G_2)_i \times 10^{n-2} + (G_3)_i \times 10^{n-3} + \ldots + (G_n)_i$$

In this formula, T_i is the trajectory value for agricultural land i, G_n is the trajectory code of LULC type, and n is the number of time nodes.

To apply this method, two different data layers, including agricultural field boundaries and LULC map of the study area for five consecutive years starting from 2011, were obtained from the Ontario Ministry of Agriculture, Food and Rural Affairs (OMAFRA) and Agriculture, Agri-Food Canada (AAFC), respectively. The source data was projected into the same coordinate system, NAD_1983_UTM_Zone_17N, to reduce the error before proceeding to the classification of crop types (Wang et al., 2012). A major change made to the LULC layer was producing the field level representation of land cover for each time node. This generalized LULC data layer allowed for a direct, field-by-field examination of crop rotations. Each LULC map was reclassified to seven

Table 1. The land use/cover classes and assigned trajectory codes

Land use class	Crop type	Trajectory Value
Pasture and forage	-	1
Winter Wheat	-	2
Corn	-	3
Soybean	-	4
Pulses	Beans, peas	5
Vegetables	Potato, Tomato, Sugerbeet	6
Fruits	Watermelon, Orchards, Vineyard, Strawberry	7

general LULC types for five-time nodes to reduce the number of potential land use trajectory combinations (Table 1).

Since changes in the crop types are considered in this project, only agricultural lands were analyzed. In light of this consideration, LULC was identified with numeric codes from 1 to 7 (Table 1); the dominant crop type in each cropland was identified using the zonal statistics function in ArcGIS 10.4.1 and considered as the sole crop type in each field. The trajectory value was calculated using the raster calculator in ArcGIS. The landscape metrics of trajectories were analyzed through the given time series to detect the spatiotemporal changes. Values with identical numerals (e.g., 3333, 44444) represent trajectories that undergo no LULC change over time, whereas others like 3311, 5111 and 6644, illustrate alterations in land cover in a specific period (Zomlot et al., 2017). The trajectory values with a particular pattern represent crop rotation in a specific field (Wang et al., 2012). The 23232 pattern, for instance, illustrates a two-year crop rotation of winter wheat and corn, while the 34234 pattern shows a three-year crop rotation of corn, soybean, and winter wheat. In this study, crop rotation was analyzed in four different categories based upon the interval of the rotation

Table 2. The crop rotation categories based on rotation interval

Crop Rotation (CR)	Category
Continuous crop	Pasture (P); Corn (C); Soybean (S)
2-year rotation	Corn-Soybean (C-S); Winter wheat-Soybean (W-S)
3-year rotation	Pasture-Corn- Winter wheat (P-C-W);
	Pasture-Corn-Pasture (P-C-P); Pasture-Pasture-Corn (P-P-C); Pasture-Pasture-Soybean (P-P-S);
	Corn-Corn-Soybean (C-C-S); Corn-Corn-Pasture(C-C-P); Corn-Soybeans-Corn (C-S-C); Soybeans-Corn-Corn (S-C-C);
	Winter wheat-Corn-Soybean (W-C-S); Corn-Soybeans-Winter wheat (C-S-W); Soybean-Winter wheat- Corn (S-W-C); Soybeans-Corn-Winter wheat (S-C-W);
	Soybean-Soybean-Corn (S-S-C); Soybean-Soybean-Winter wheat (S-S-W); Soybean-Winter wheat-Soybean (S-W-S); Corn-Soybeans-Soybeans (C-S-S); Soybeans-Corn-Soybean (S-C-S)

(Table 2). The resulting dominant crop rotation from these five LULC datasets was applied to a hydrologic model (SWAT) in the Medway creek watershed.

3.2.4 Model Setup

3.2.4.1 SWAT Fundamentals

Developed by the USDA Agricultural Research Service, SWAT is a physical and procedural spatially distributed watershed model (i.e., it accounts for spatial variability of input variable), widely used by water quality experts around the world (Arnold et al., 1998; Gassman et al., 2007; Williams et al., 2008; Arnold et al., 2012). SWAT is capable of predicting the influence of long-term, point, and non-point sources of pollution on variables such as pesticide loads, nutrients, and sediments, even across large complex watersheds recognized by different soil and land use conditions (Arnold et al., 1994). Furthermore, it has been widely used for investigating the effects of the current and potential future climate trends, LULC change, and farming management practices on quality of aquatic environments and resources across agricultural watersheds (Arnold & Fohrer, 2005; Eckhardt, 2005; Sheshukov et al., 2011; Wang et al., 2014).

The required input data to run the model consists of LULC type, topography, soil properties, metrological data, and land management practices (Neitsch et al., 2005). The model runs by dividing a watershed into smaller areas, known as sub-watersheds (subbasins), and then into Hydrologic Response Units (HRUs), comprising unique combination of soil, land use, and slope in the area under study. HRUs are critical units in the modeling process as fluxes of water, sediment and nutrient are simulated and computed at this level, and then aggregated to the higher sub-watershed (subbasin), and ultimately to the watershed level through routing processes (Singh et al., 2005; Arnold et al., 2001). In addition to hydrology, other essential building blocks of a SWAT model are nutrients, agricultural management, plant growth, chemicals, and weather data (Gassman et al., 2007). Since the focus of this modeling study is on changes in water quality by a special focus on sediment and nutrient transports (particularly total phosphorus), a more in-depth explanation of sediment and nutrient runoff as well as transport estimations are discussed in the following sections.

Detachment, transport, and degradation-deposition are the three stages defining sediment erosion. In the first stage, SWAT uses the Modified Universal Soil Loss Equation, known as MUSLE, which utilizes the energy of surface runoff instead of rainfall to estimate sediment yields, to calculate erosion and sediment yield for each HRU within the boundaries of the watershed (Williams, 1975). Daily sediment yield in metric tons is calculated using the following formula:

$$sed = 11.8 \times (Q_{surf} \times q_{peak} \times area_{hru})^{0.56} \times K_{usle} \times C_{usle} \times P_{usle} \times LS_{usle} \times CFRG$$

where the Q_{surf} is the volume of surface runoff (mm/ha), q_{peak} is the peak runoff rate (m^3/s), $area_{hru}$ is the area of the HRU (ha), K_{usle} is the USLE soil erodibility factor (0.013 metric ton m^2 hr/ m^3-metric ton cm), C_{usle} is the USLE land cover factor, P_{usle} is the USLE support practice factor, LS_{usle} is the USLE topography factor, and CFRG is the course soil fragment factor (Neitsch et al., 2005). The second stage is the transport of the detached sediment from the first stage. The following equation is used to calculate the amount of sediment transported and released to the main channel.

$$sed = (sed' + sed_{stor,i-1}) \times (1 - \exp[- surlag/t_{conc}])$$

where sed is the sediment discharged into the main channel on a given day (metric tons), sed′ is the amount of sediment load from an HRU on a given day (metric tons), $sed_{stor,i-1}$ is the mass of stored sediment from the preceding day (metric tons), surlag is the surface runoff lag coefficient (user-defined), and t_{conc} is the time of concentration (h). The third stage of erosion is degradation and deposition. Deposition occurs when sediment settles on streambed due to high concentration after it leaves the streamflow and settles on the streambed. "Meanwhile, degradation occurs while the sediment concentration is low, allowing the streambed sediments to be suspended, and travel with streamflow. The threshold value for these processes, maximum sediment concentration, is a function of the peak runoff rate of the stream" (Neitsch et al., 2005, p.380).

3.2.4.2 Hydrologic Response Units (HRU) Definition Based on the Agricultural Field Boundary

The hydrologic response units (HRU) are defined as the smallest units of the SWAT model, which are not spatially diffused, can be non-continuous, and contain no routing among them. In other words, similar basin characteristics such as soil, land use, and slope are aggregated based upon user-defined thresholds in the standard definition for HRU (Kalcic et al., 2015; Pai et al., 2012). Therefore, HRUs are spatially situated within the watershed boundaries, but their response may not be attributed to a specific field in the watershed. This could be interpreted as the elimination of spatial connectivity across the landscape, as discussed by Merriman et al. (2019).

Depending upon simulation objectives, HRUs could be defined by specific spatial locations and field boundaries. At the small watershed to field scale, for instance, field-based management might be an important consideration, rendering field-based outputs and potentially inputs to become the essential components of the model. This feature, however, is not defined in the ArcSWAT interface by default. Consequently, a different approach should be considered to apply it to the landscape under study. Daggupati et al. (2011) and Kalcic et al. (2015) presented techniques to create spatially accurate HRUs aligned to field boundaries within the SWAT modeling framework.

In this study, HRUs were defined based on the field boundary by adopting the method outlined in Kalcic et al. (2015). The field boundary layer used for HRUs definition is farm parcels obtained from

Land Information Ontario (LIO). Non-field areas of the watershed, including roads, developed areas, and wooded areas, were assigned a specific field boundary or allocated to nearby crop fields. This layer was prepared in ArcMap using the Arctoolbox integrate tool (to remove roads), union tool (to fill areas missing from field boundary layer), and figure to polygon tool (to separate non-continuous features formed by union tool), respectively. Polygons smaller than 1 ha were removed by using the eliminate tool and then merged with the larger polygons that share the longest boundary with these features.

One land use, one soil type, and one slope are required to define each HRU at the field scale. Since the slope of the study area is relatively low, only a single slope class was used for the HRU definition in the model setup; this consideration prevented the original field boundaries from becoming fragmented. Within each field boundary, the land use and soil type with the maximum number of

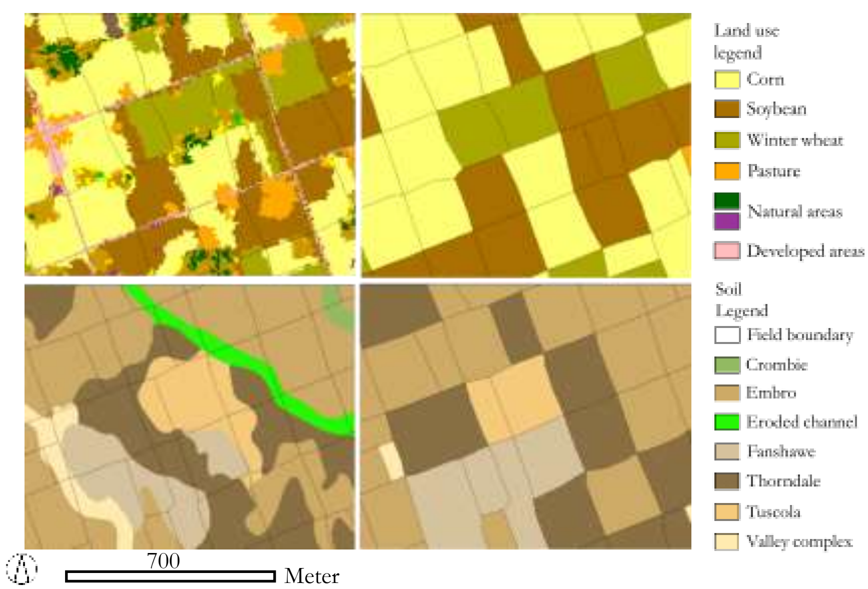

Figure 2. Land use and soil type maps before (left) and after (right) modification based on the agricultural field boundary

pixels were considered as the main crop type and soil type in each field, respectively (Figure 2). Zonal Statistics as Table tool was used to define the majority of land use and soil type within each

Table 3. The proportion of the land uses and soil types before and after application of field boundary

Land use class	Before application of field boundary	After application of field boundary	Soil class* (Hydrologic groups)	Before application of field boundary	After application of field boundary
Corn	32.26%	34.37%	A	1.59%	0.98%
Soybean	38.59%	41.46%	B	30.09%	28.96%
Winter wheat	14.19%	13.48%	C	60.47%	61.82%
Pasture	14.22%	9.96%	D	7.85%	8.24%
Pulses	0.34%	0.24%			
Vegetables	0.23%	0.37%			
Fruits	0.17%	0.12%			

*Soil classes are based on runoff and infiltration rate
A: Low runoff potential and high infiltration rate when thoroughly wetted;
B: Moderate infiltration rate when completely wetted;
C: Slow infiltration rate when thoroughly wetted;
D: High runoff potential and very slow infiltration rate when thoroughly wetted.

field boundary. As listed in Table 3, the percentage of the land use and soil type layers slightly differ before and after reconfiguration based on the majority of pixels in each field.

Despite being a critical initial step for delineating HRUs by field boundaries, considering the dominant land uses and soil types is not a sufficient requirement for defining HRUs based on this method, as fields with identical soil and land use types in a given subbasin would still aggregate into one HRU. To prevent this aggregation, soils or land uses can be assigned unique names. In the method used in this study, HRUs were defined through the addition of the user-defined unique soil names to the SWAT usersoil database. In this regard, lookup tables were created using the attribute tables of soil type and field boundary to define, and then map unique soil names to each field boundary. Upon entry to the usersoil table, each unique name was then added as the only altered characteristic of the considered soil layer. That is, the remaining attributes in the usersoil table remain unchanged. This way, HRUs were defined in the SWAT setup with a unique soil type for each field, which is required for the definition of HRUs based on the agricultural field boundary.

3.2.4.3 SWAT Model Development

For this study, we run the SWAT model version 2012 (664) using the ArcSWAT 2012.10_4.19 interface. Based on a 10 m Digital Elevation Model (DEM), the Medway Creek Watershed was divided into 19 subbasins with a 393 ha threshold value for flow accumulation which resulted in a stream density similar to the dataset provided by Upper Thames River Conservation Authority (UTRCA) for the watershed under study (Figure3). Watershed model for the Medway Creek was set up for HRU definition by agricultural field boundaries. Definition of land use, soils, and slope were the only different aspects compared to the standard HRU definition. As noted in section 3.2.4.2, when defining the HRUs based on the field boundary consideration, the pre-processed field boundary layer with the majority of land use and soil types was utilized. Based on the unique combination of threshold values of 0% for land use, 0% for soil type, and a single slope class, 906 HRUs were created. Modelling based upon these values allow analysts to investigate the efficacy of conservation efforts in a simulated landscape consisting of the maximum number of identified HRUs. More specifically, applying the 0% threshold for HRU delineation increases the accuracy of model and prevents loss of valuable information regarding the unique landscape and soil feature in the model representation (Her et al., 2015).

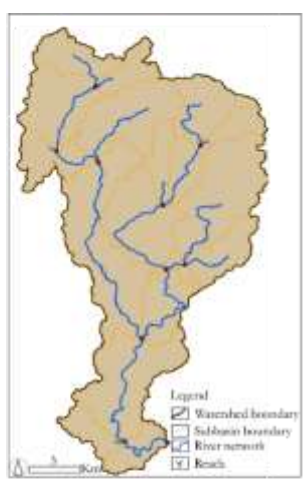

Figure 3. River network and subbasins created by SWAT

Each HRU represents an agricultural field that can be modified to reflect land use variation and different management practices. In this model, only corn, soybean, and winter wheat HRUs, as the dominant cultivated crops, were subject to the considered changes. These changes were based on the local knowledge of agriculture and the information provided by UTRCA for the Upper Medway Sub-watershed. The dominant crop rotation was estimated based on the time series analysis of land cover data layer from 2010 to 2015. Based upon the method outlined in Bosch et al. (2011), the assigned land use of each row crop HRU was considered as the starting crop of the multi-year crop rotations to ensure a more realistic annual distribution of crops across the watershed. The percentage of lands for application of both tillage systems (primary and secondary) and fertilizer (phosphorus and nitrogen) for each crop was developed according to the dominant rotation in the study area (Table 4). Also, the UTRCA information was considered to calculate the amount of nitrogen and phosphorus applications as fertilizer.

Table 4. The field-based percentage of tillage (primary and secondary) and fertilizer (N and P) application for management schedule

Crop type	Tillage (%)		Fertilizer-Nitrogen (%)			Fertilizer-Phosphorus (%)		
	Primary*	Secondary	Total	Spring	Fall	Total	Spring	Fall
Corn	100	95	100	100	-	100	46	54
Soybean	100	34	39	95	5	43	91	9
Winter wheat	100	-	100	100	-	100	4	96

* First tillage of the growing season

While fertilizers can be applied in SWAT in many forms of chemical fertilizer or manure, we considered all phosphorus and nitrogen fertilizers as elemental phosphorus and elemental nitrogen, respectively. Accordingly, two methods of fertilizer applications (banding and broadcast) were considered in this study. The rates and methods of fertilizer applications are illustrated in Table 5. The fraction of fertilizer applied to the top 10 mm of soil (frt_surf) for banding and broadcasting of fertilizers was set to 0.01 (Merriman et al., 2019) and 0.8, respectively. The timing for fertilizer and tillage application -which is typically determined based on the schedule for crop planting and harvesting - was considered, to be based on the most typical window of action in the five years of historical crop progress in the study area, according to the information provided by UTRCA.

Table 5. The rate and method of N and P applications as Fertilizer

Crop	Corn		Soybean		Winter wheat	
Method of application	Broadcast	Band	Broadcast	Band	Broadcast	Band
N Fertilizer-Spring application (Kg/ha)	186	205	17	-	139	-
N Fertilizer-Fall application (Kg/ha)	-	-	13	-	-	-
P Fertilizer-Spring application (Kg/ha)	43	95	54	77	46	-
P Fertilizer-Fall application (Kg/ha)	151	-	56	-	81	15

To estimate evapotranspiration, the Penman-Monteith method was incorporated into the model. This consideration is complemented by integrating the SCS curve number approach for computing surface runoff. Since in-stream water quality processes are beyond the scope of the current study, this process was excluded from our model setup. Processes such as evapotranspiration, water runoff, and nutrient runoff were simulated considering a month-by-month basis for each HRU, lumped at the subbasin scale, and aggregated into an overall watershed response at the watershed outlet.

These decisions were considered as a baseline scenario with no implemented crop rotation for calibrating and validating the model. The calibration of the model is done in a study in the same area by Hanke (2018) with a primary focus on estimating the impact of climate change on water quantity and quality. The calibrated parameters and their best fits are listed in the Appendix. The model simulations were carried out for the 35 years from 1979 to 2013. Accordingly, the first five years were the warm-up period and were not included in the analyses and conclusions. The warm-up period is used for equilibrating the hydrologic condition of the model before the simulation period (Daggupati et al., 2015).

The baseline scenario is used to draw conclusions based on comparisons with other scenarios using different considerations regarding the implemented crop rotation as the best management practice in the area under study.

3.2.5 Crop Rotation Incorporation in SWAT

Crop rotations in SWAT can be entered as a sequence of planting and harvesting operations within the same HRU supplemented by management operations that are available from the included database of crop classes and management practices. Thus, the first step of incorporating crop rotation into the SWAT model is to develop management schedules for each crop, which were defined based on the dominant crop rotation as corn, soybean, and winter wheat. The management schedules were applied on an annual basis to reflect the rotation of the crops in the study area.

Based on the categorization of Castellazzi et al. (2008), the crop rotation in this study is fixed and cyclical with variable rotation length, as illustrated in Figure 4. While HRUs with row crops as corn and soybeans were considered to be in a three-year rotation, HRUs containing winter wheat were considered to be in a four-year rotation. More specifically, a fixed management schedule was used for each crop upon the implementation of the scenarios defined in this study.

In defining the management schedules, dates were used in management operations instead of the default accumulated heat units in the SWAT model. The rotation includes tillage operations (primary

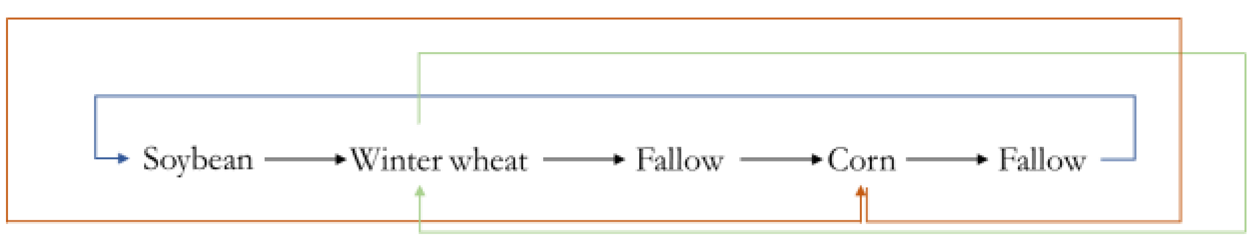

Figure 4. Fixed crop rotation with different lengths of rotation. The orange line illustrates the corn-fallow-soybean-winter wheat-fallow rotation. The blue line illustrates the soybean-winter wheat-fallow-corn-fallow rotation and the green line illustrates the winter wheat-fallow-corn-fallow-soybean rotation.

tillage and secondary tillage) with two fertilizer applications of nitrogen and phosphorus with fixed annual rates, applied in fall and spring, representing the post-harvesting and pre-planting dates, respectively. While farmers could potentially adjust fertilizer application in a given year based on the previous year's crop, we assumed no changes regarding the fertilizer application. The fertilization rates applied in the management schedules are based on the information given by farmers in the watershed survey by conservation specialists of Upper Thames River (M. Funk, personal communication, June 10, 2019).

The SCS curve number is the main parameter used to represent different tillage systems in the SWAT model (Kirsch et al., 2002). The method used by Feyereisen et al. (2008) and Arabi et al. (2008), as well as the SWAT input/output documentation (version 2012), were utilized in this study to assign the SCS curve number to each management schedule. Based on these studies, the curve number for conservation tillage is six units lower than the conventional tillage.

The crop of year 1 in the rotation represents the land use class in each HRU. The rotation was specified for a certain number of years, and if the simulation period exceeded a period of crop rotation, the same rotation was repeated consecutively within the same HRU. The graphical illustration of the three management schedules used in the application of crop rotation is shown in Figure 5.

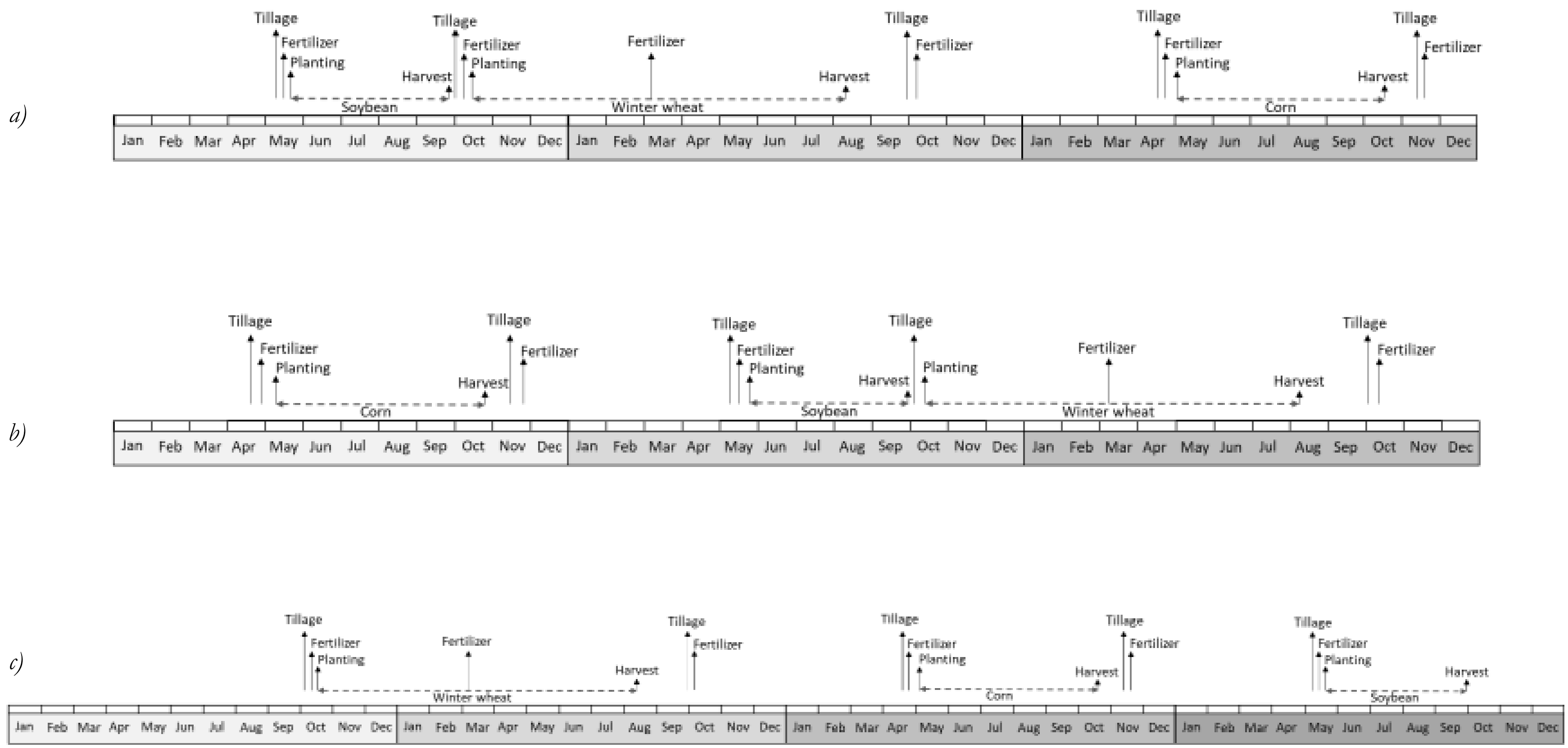

Figure 5. Management schedules a) soybean-winter wheat-corn b) corn-soybean-winter wheat c) winter wheat- corn-soybean

3.2.6 Scenario Development

3.2.6.1 Spatial Targeting Method

Watershed-scale applications of BMPs are both infeasible and costly. Furthermore, each area has a disproportionate share of polluting water resources across any given watershed (Maringanti et al., 2009). Therefore, identifying Critical Source Areas (CSAs) – zones with the highest contribution to water quality degradation per unit of area – is of the utmost importance to successful BMP implementation (Giri et al., 2012).

To develop scenarios in the current study, the Load Per Unit Area Index (LPUAI) technique (Giri et al., 2012) was used to prioritize BMP placement within the watershed. More specifically, the LPUAI was utilized to identify the high priority areas by considering the average sediment and nutrient loads per unit area of each subbasin. To calculate total phosphorus loss from surface runoff, the following equation was used:

Total phosphorus runoff (kg P ha^{-1}) = Σ (particulate P, dissolved mineral P)

where particulate P consists of organic P and sediment P, and soluble P is a component of dissolved mineral P.

Accordingly, subbasins were categorized into high, medium, and low priority areas using the natural breaks method of classification in ArcGIS (minimizing within-class variance and maximizing between-class differences). As a result, the subbasins with the similar range of P load will be grouped in the same level of priority. Moving from the highest priority area to the lowest priority area, subbasins that are closer to the main stream were considered as the areas representing the highest priority for BMP placement. The same approach was implemented at the HRU level for identifying the most preferred HRUs within each subbasin for BMP placement. The use of CSAs as a mechanism to identify high priority areas for BMP placement is referred to as the targeting approach, which was utilized in this study to develop the scenarios.

3.2.6.2 Description of Scenarios

As illustrated in Figure 6, two different groups of scenarios were considered in this study. The first group (the spatial distribution) was developed based upon the concept of the spatial distribution of crop rotation in the watershed by applying two distinct approaches for BMP implementation. These are a) the random distribution of BMP, using the random generator function in Microsoft Excel 2016, to select HRUs, and b) the targeting approach, which places the BMPs in the high priority areas defined in section 3.2.6.1. The second group of scenarios (level of implementation) was developed based on the different rates of implementation of crop rotation as the BMP in the

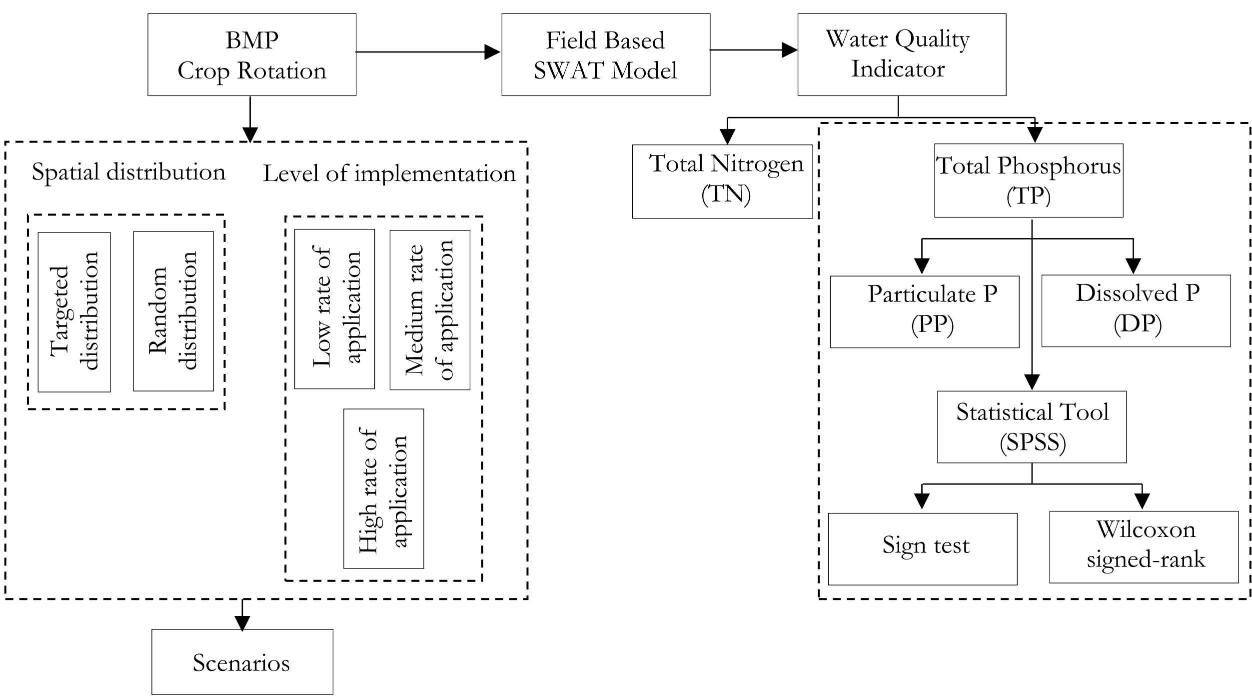

Figure 6. Modeling framework and scenario development

watershed. The first group of scenarios was modeled considering a constant level of implementation equal to 50% of the HRUs, on a LULC consisting of soybean, winter wheat, and corn.

Similarly, the second group of scenarios was modeled considering the targeted, rather than the random distribution of crop rotation to assess the efficacy of different levels of implementation, ranging from low to high, on the landscape under study (Table 6). The low, medium, and high scenarios were considered to be implemented by the rates of 30%, 50%, and 70%, respectively, to the agricultural HRUs (fields) prioritized by the LPUAI method.

Table 6. The explanation of scenarios

Modeled Scenarios	Percentage of fields with CR*	Method used for spatial distribution	Number of fields with CR	Area of watershed with CR (ha)
Baseline	-	-	0	0
First Group of Scenarios				
Targeted Distribution	50	Targeted	345	7419
Random Distribution	50	Random generator	345	8269
Second Group of Scenarios				
Low rate of implementation	30	Targeted	208	4433
Medium rate of implementation	50	Targeted	345	7419
High rate of implementation	70	Targeted	486	10746

* Crop Rotation

Figure 7 illustrates a conceptual diagram of the scenario groups considered in this study. The amount of TP load reduction from the farming landscape was considered to investigate the effectiveness of each group of scenarios at the watershed scale. Accordingly, each of these hypothetical scenarios was simulated from 1984 to 2013, using the same precipitation and temperature data utilized in the baseline scenario setup. To evaluate the impact of crop rotation in each case, a comparison was made with the baseline scenario, which represents a zero level of BMP implementation.

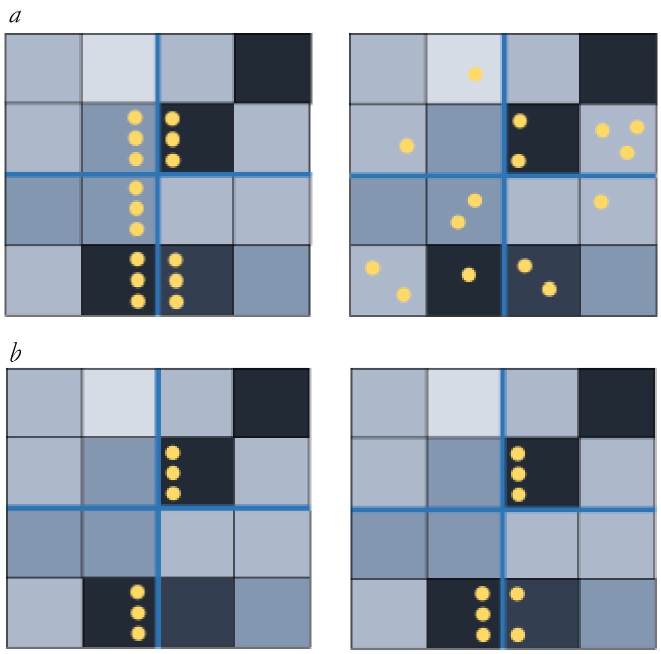

Figure 7. A hypothetical watershed in which the blue lines are representative of the river network, each square is considered as a subbasin with the colors illustrating the degree of TP load to the main stream, and the yellow circles are the implemented BMPs. The TP load is considered to be higher in the darker subbasins.

a. Spatial distribution scenarios. The number of implemented crop rotation is constant.
Left: Targeted distribution of BMP
Right: Random distribution of BMP

b. Level of implementation scenarios. The spatial distribution of crop rotation is targeted.

3.3 Results and Discussion

3.3.1 Dominant Crop Rotation

All possible crop rotations were determined following the LULC processing from 2010 to 2015. Although the generalization of the LULC layer disregarded small-scale crop rotations, the dominant agronomic culture in the Medway Creek Watershed stayed unchanged by maintaining soybean, corn and winter wheat as part of several rotations in the region.

The three-year rotation period was the most widely practiced rotation regime applied to approximately 45% of agricultural lands within the watershed. As the main agronomic culture in the study area, corn, soybean, and winter wheat were found to be the dominant rotation pattern within a three-year rotation period in 39% of the rotations in the Medway Creek agricultural watershed (Table 7). The different combinations of pasture with soybean and corn comprise the remaining (i.e. 6%) three-year rotations, which were not included in this modeling study. Other crop rotations in the area, namely the continuous crop, two-year rotation, and four-year rotation are practiced in 13.48%, 14%, and 27.52% of the agricultural fields, respectively.

Table 7: The dominant crop rotation and the corresponding area percentage in Medway Creek Watershed

Primary crop	Crop for rotation year			Area covered with crop rotation (%)
	1	2	3	
Corn	Corn	Soybean	Winter wheat	18
Soybean	Soybean	Winter wheat	Corn	10
Winter wheat	Winter wheat	Corn	Soybean	11

3.3.2 Spatial Targeting Method

Critical source areas of pollution and placement of BMPs were two fundamental stages of this study. The spatial targeting method utilized here was based on the load per unit area index. Based on the

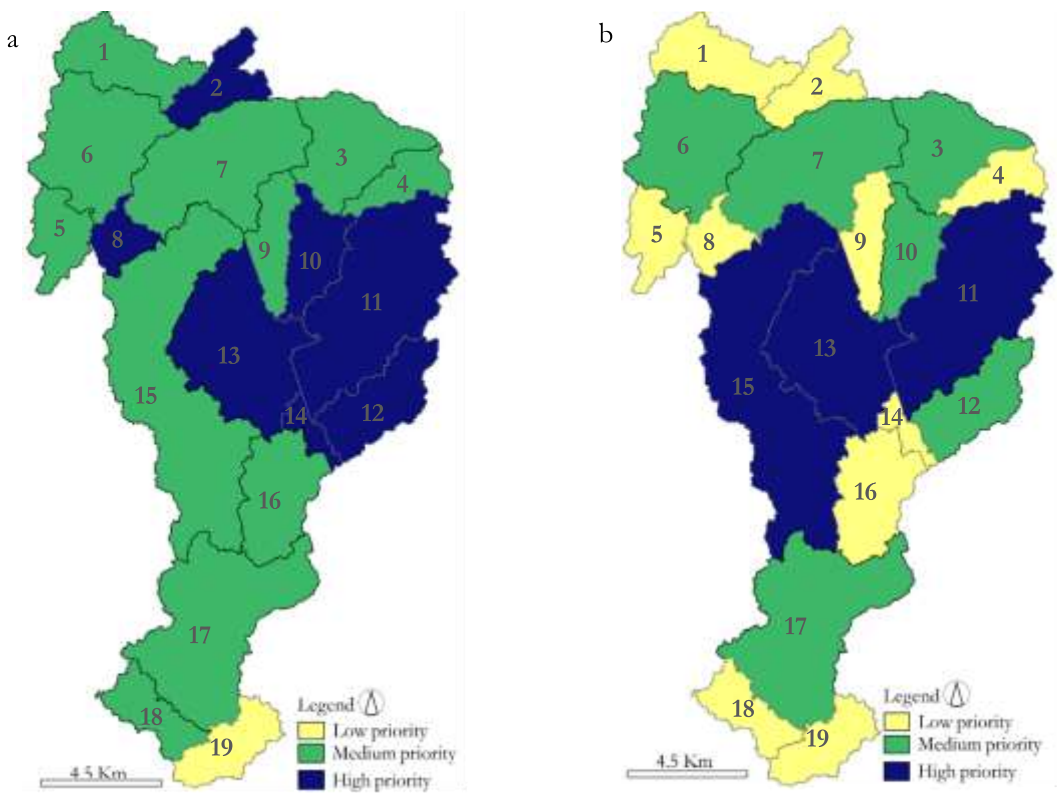

Figure 8. CSAs for TP load in the Medway Creek Watershed based on (a) the load per unit area (kg/ha/year) (b) the load per subbasin area (kg/subbasin/year)

TP results produced by the SWAT model for the baseline scenario, the watershed was divided into high, medium, and low priority areas. TP concentration were 0.0 – 1.08 kg/ha, 1.09- 25.86 kg/ha, and 25.87- 35. 62 kg/ha, for low, medium, and high priority areas, respectively. As depicted in Figure 8a, seven subbasins were identified as the high priority areas for BMP implementation. These are subbasins 2, 8, 10, 11, 12, 13, and 14.

The comparison of the priority areas delineated based on the per unit area index (Figure 8a) with the priority areas determined by the subbasin area index (Figure 8b) illustrated that the area of high priority zones in the LPUAI method was smaller than the area of high priority zones in the subbasin area index method as some of the larger subbasins were disregarded when the per unit area index was utilized to identify the CSAs. For instance, subbasin 15 would have been identified as a priority zone if its area was the only criterion considered for prioritization. However, when ranked based on the unit area contribution, this subbasin was considered as a medium priority zone for BMP implementation.

The high priority areas were mostly concentrated in the upstream and central sections of the watershed. A majority of the watershed subbasins were characterized as medium priority areas. There was only one subbasin with the low priority, which was covered by the urban land use.

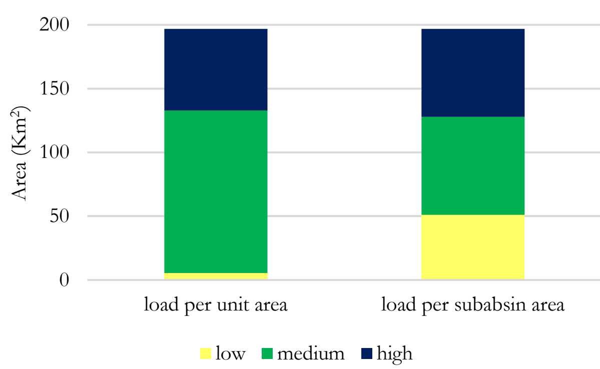

Figure 9. Distribution of high, medium and low priority areas for TP load

Figure 9 compares the total areas for the different categories of the priority areas based on the two delineation methods described above.

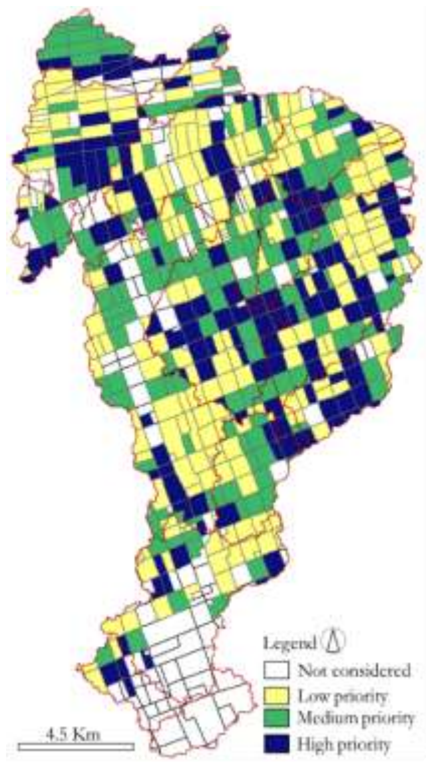

Figure 10. CSAs for TP load at HRU level (kg/ha/year)

The extension of the LPUAI method to the field (HRU) level as the basic unit for implementation of crop rotation determined the critical source areas at this scale, which were also classified into the low, medium, and high priority fields, as shown in Figure 10.

3.3.3 Impact of the Implementation Rate of Crop Rotation on Phosphorus Load in the Watershed

The performance of the high, medium, and low scenarios in the considered 30-year period was determined in terms of percent reduction of TP compared to the baseline scenario, which has no BMPs. To investigate the significance of the annual changes in the TP load reduction between the simulated and baseline scenarios, the non-parametric statistical tests of Sign and Wilcoxon signed-rank were used. In this respect, the Wilcoxon signed-rank test was utilized when the distribution of the differences of TP load between the baseline and the considered scenarios was symmetrical. Otherwise, the sign test was selected as the most suitable statistical test. The IBM SPSS Statistics 23 was used to perform the analysis. As illustrated in Table 8, at $\alpha = 0.05$, the difference between the low scenario (30% rate of implementation) and the baseline scenario did not illustrate a statistically significant change in TP load reduction in the watershed ($p = 0.062$), whereas the medium scenario (50% rate of implementation) represented a statistically significant TP load reduction ($p = 0.004$), when compared to baseline at the same α level. As the medium scenario represented statistically significant results for the TP load reduction compared to the baseline scenario, the association between the considered high scenario (70% rate of implementation) and the baseline scenario was also considered to be statistically significant. Furthermore, the Wilcoxon signed-rank test was performed between the medium and high scenarios. The results of this test illustrated that the change in the TP load created by the 20% increase to the implementation rate applied at the medium scenario was not sufficient to create statistically significant change in the TP level in the area.

Table 8. The results of the statistical tests

Test	Test Statistics	Asymptotic Sig. * (2-sided test)
Baseline and low scenario	1.789	.062
Baseline and medium scenario	2.667	.004
Medium and high scenario	1.183	.237

* The significance level is .05

Figure 11 illustrates the annual TP, PP, and DP reduction in each subbasin considered for the BMP implementation for all the simulated scenarios. Given that the LPUAI method was utilized to identify the priority areas, the total number of subbasins with crop rotation increased by increasing the rate of implementation from the low scenario to the high scenario. Accordingly, the higher rates of implementation resulted in higher rates of reduction among all the considered subbasins.

Subbasin 12 in the low scenario could be considered as an instance of the described trend. In this subbasin, both TP and PP illustrated a zero rate of reduction when only one field was considered for crop rotation in the low scenario, whereas by increasing the number of crop rotations to 35 fields in the medium scenario, a reduction in the phosphorus levels (1.4% for TP and 2% for PP) was observed. The variability of TP, PP, and DP in the high scenario and among those subbasins

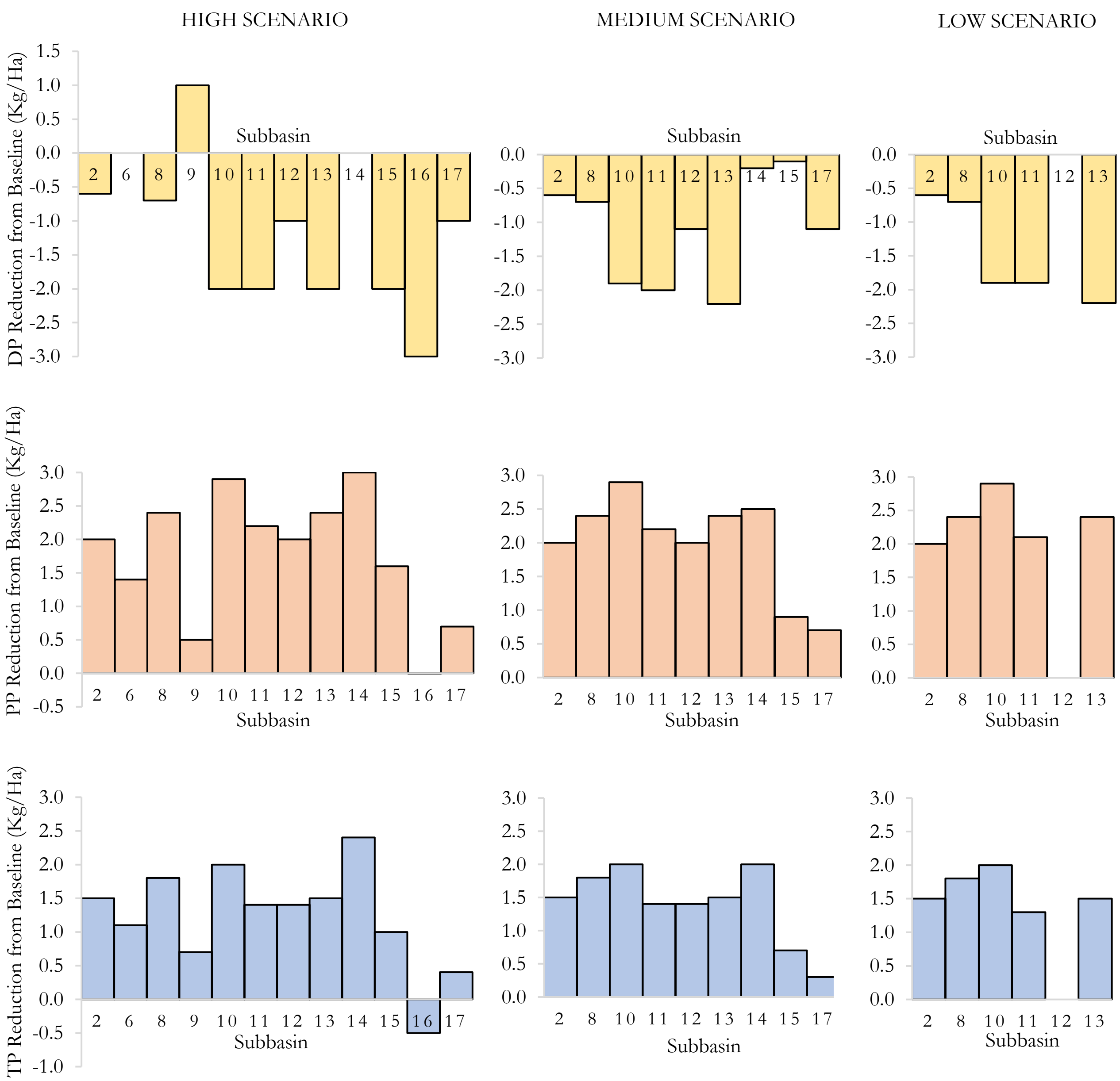

Figure 11. Annual TP, PP and DP reduction rate for each subbaisn in the low, medium and high scenarios

designated for BMP implementation is illustrated in Figure 12 for three-year intervals starting from 1984.

Figure 13 illustrates the rate of the watershed-scale phosphorus reduction ranged from 12% in the low scenario to 21% in the high scenario for PP and from 8% in the low scenario to 17% in the high scenario for TP. The low scenario has the least TP and PP reductions rates which is consistent with having the minimum number of fields with crop rotation. There is a 20% increase in the number of fields with crop rotation in the medium scenario, corresponding to a 14% TP reduction and 18% PP reduction, both with a 6% increase compared to the low scenario. Therefore, the increase in the implementation rate of crop rotation gradually increased the model performance at reducing the PP load by an average reduction rate equal to 16%. Similarly, the TP showed a gradual improvement in the rate of reduction from 8% to 17% across the watershed. However, this increase in the rate of TP reduction has a descending trend from one scenario to a higher-level scenario. More specifically, in the low scenario, the TP reduction rate is 8%, compared to the baseline scenario, whereas this rate in the high scenario is only equal to 2% when compared to the medium scenario. In both of these scenarios, the C-S-W, S-W-C, and W-C-S have the same proportion of implementation equal to 44%, 44%, and 12%, respectively, showing no association between the aforementioned descending trend in the TP reduction rate and the proportion of the type of the crop rotation among these scenarios. Thus, it is possible to argue that there is a potential threshold level beyond which any increase in the implementation level of crop rotation creates only minor changes in the level of phosphorus reduction across the watershed.

Considering the share of each type of phosphorus loads in the baseline scenario, the results of this modeling study illustrated that PP constituted a significant proportion with the 81% of the total phosphorus load, whereas DP had a much smaller share of 19% across the study area. The higher

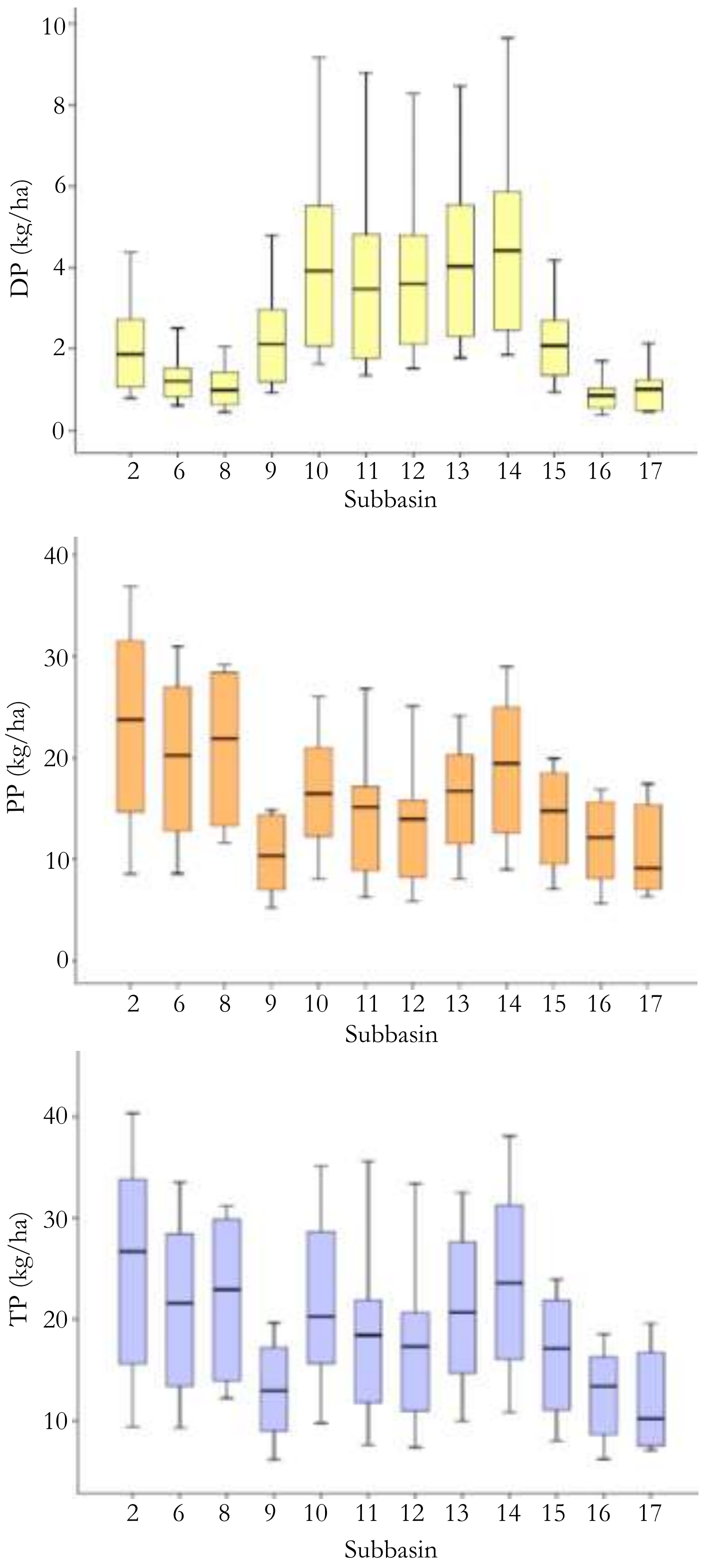

Figure 12. Variability of TP, PP, and DP loads across the watershed in the high scenario

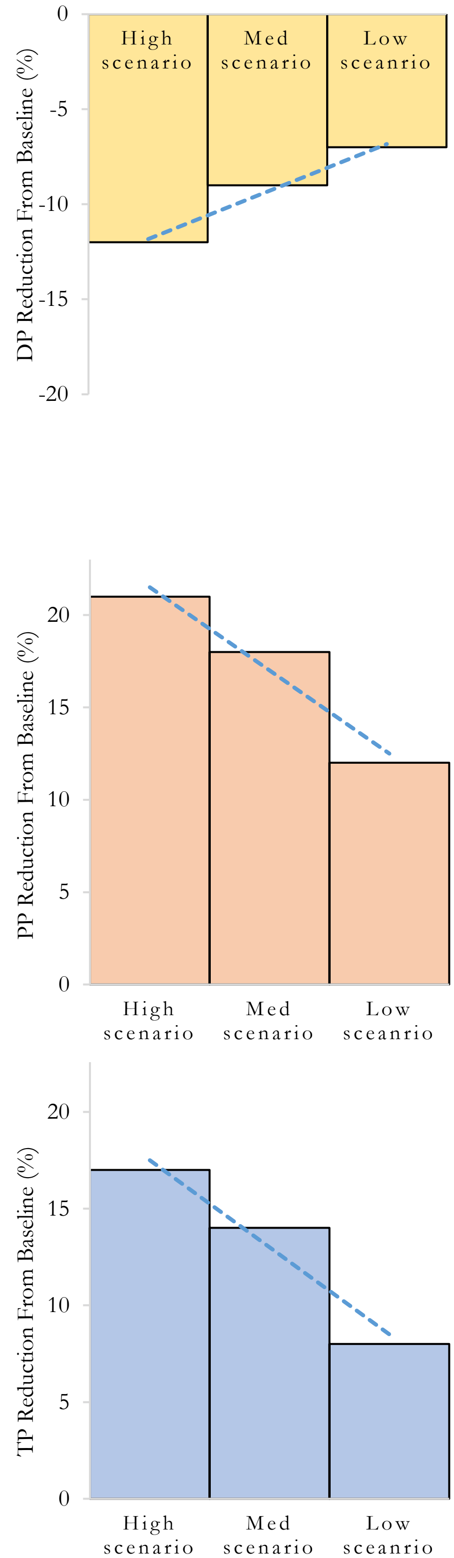

Figure 13. TP, PP, and DP rates of reduction in the high, medium and low scenarios

share of PP accumulation, however, was decreasing in each scenario as a result of BMP implementation. In contrast, the change in the rate of DP accumulation showed an increasing trend, changing from 7% in the low scenario to 12% in the high scenario. Similar results were observed by Bundy et al. (2001) where they found an increased dissolved phosphorus concentration across cornfields with no-till, and by Merriman et al. (2018b) where they found that crop rotation increased DP by an average rate of 17% across a watershed, when applied individually as a BMP, and by 18% when applied with cover crop, no-tillage, and nutrient management plan.

The increase in the DP level indicates that the crop rotation as a BMP is likely to heavily influence the PP level primarily due to the erosion control effect of crop rotation as a BMP (Logan, 1990; Bullock, 1992; Kollas et al., 2015; Fahmy et al., 2010). More specifically, crop rotation influences the level of PP, which is mainly transported to water bodies by surface runoff and soil erosion (Kronvang et al., 1999). Therefore, measures to stop erosion are more effective in PP reduction goals (Withers & Jarvis, 1998), and as such, decreasing DP loss alone requires different sets of considerations that specifically address dissolved phosphorus loads in a watershed (McDowel & Sharpley, 2001).

Taking into account the type and source of P, some studies (e.g., Bundy et al., 2001; Merriman et al., 2019) discussed the importance of correctly identifying these features upon the BMP implementation because the optimal BMP to reduce one pool of phosphorus may not necessarily be as effective in reducing the other pools. Thus, a combination of BMPs is required to increase the efficacy of these measures (Hanief & Laursen, 2019). In their research on the efficacy of BMP combinations and their impact on reducing different phosphorus pools at the watershed scale, Yang et al. (2012) showed that the most effective sediment reduction is achieved when crop rotation, flow diversion terraces, and no-till are combined, whereas to reduce the amount of soluble P, the best combination of BMPs comprises crop rotation, flow diversion terraces, and fertilizer reduction.

3.3.4 Impact of the Spatial Distribution of Crop Rotation on Phosphorus Load in the Watershed

To compare the impact of spatial patterns of distribution on the phosphorus load reduction in the watershed, the results of the targeted distribution were assumed to be identical to the results attained from the medium scenario with the 50% rate of implementation, as noted in section 3.3.3. The results of the statistical test between the baseline scenario and the random distribution of crop rotation with a 50% rate of implementation illustrated a statistically significant change, at $\alpha=0.05$, in the TP load across the watershed ($p=0.001$) (Table 9).

Table 9. The results of the statistical tests

Test	Test Statistics	Asymptotic Sig. * (2-sided test)
Baseline scenario and Random distribution scenario	3.332	.001
Baseline scenario and Targeted distribution scenario	2.667	.004

* The significance level is .05.

Compared to the randomly distributed scenario, the results of the targeted distribution of crop rotation exhibited higher reductions in the total level of phosphorus. This difference could be attributed to the total number of the BMPs (crop rotations) applied in the high priority areas across the watershed. More specifically, in the targeted distribution scenario, 77% of the BMPs are applied in high priority areas, whereas this proportion was only 39% of the high priority areas in the random distribution scenario. When compared to the baseline scenario, the random placement of crop rotation illustrated a 9% TP reduction rate, which was only 1% higher than the low scenario with the

30% rate of targeted BMP implementation. However, the TP reduction based on the random placement of crop rotation illustrated a 5% lower reduction rate, when compared to the targeted scenario with the same rate of BMP implementation.

Figure 14 illustrates the difference of the TP, PP, and DP reduction rates between the random and targeted scenarios of crop rotation at the subbasin level.

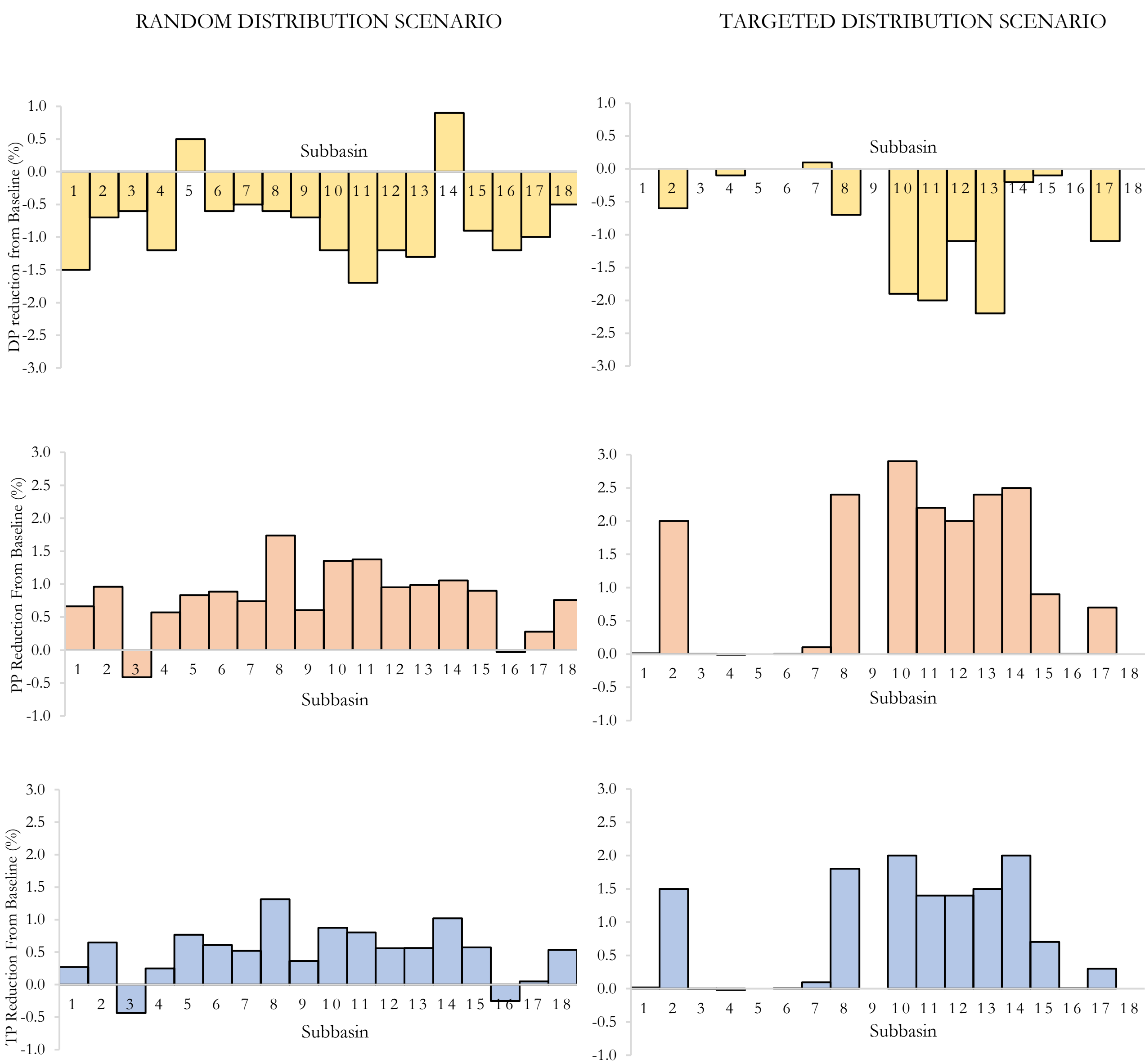

Figure 14: Annual TP, PP and DP reduction rate for each subbaisn in the random distribution and targeted scenarios

As can be seen, while both TP and PP reductions occurred in a relatively consistent way across the watershed upon the random placement of crop rotation, these rates differed significantly among individual subbasins when the targeted approach was adopted. Furthermore, the consistent rates observed in the random approach was lower compared to the ones observed in the targeted scenarios, reflecting the higher efficacy of crop rotation when applied based on the targeted approach across the watershed. Overall, the results indicated that the emphasis should be on the placement of crop rotations to meet the objectives of the BMP implementation project as different spatial distribution patterns produced varying levels of reduction, which resulted in fluctuating efficiency rates at the watershed scale.

Furthermore, the simulated model yielded varying reduction results regarding the types of phosphorus pools across the watershed, as depicted in Figure 16. Accordingly, the PP reduction rate was 14% upon the random placement of crop rotation, which was only 2% higher than the low scenario with the targeted placement of BMPs across the 30% of the identified HRUs. Nevertheless, the total reduction rate attained from the random application of BMP was 5% lower than the targeted scenario (Figure 15).

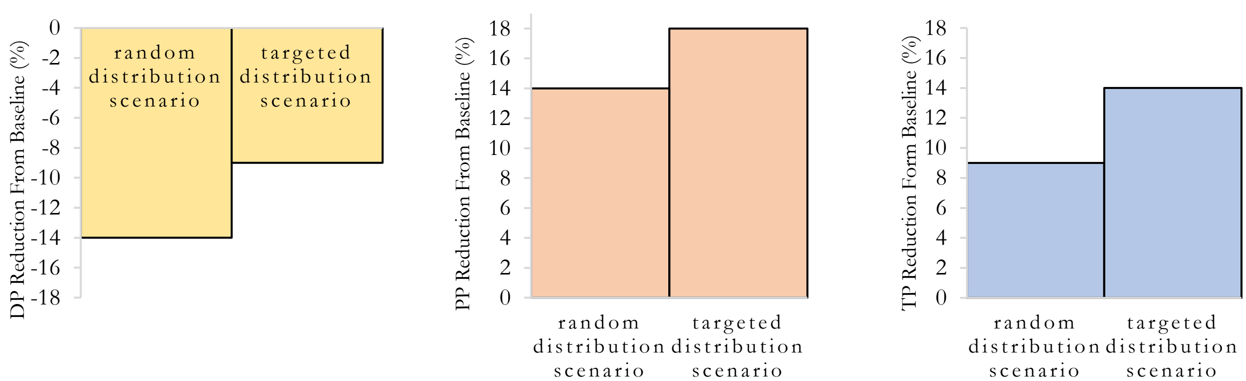

Figure 15. TP, DP and PP rate of reduction in the random distribution and targeted distribution scenarios

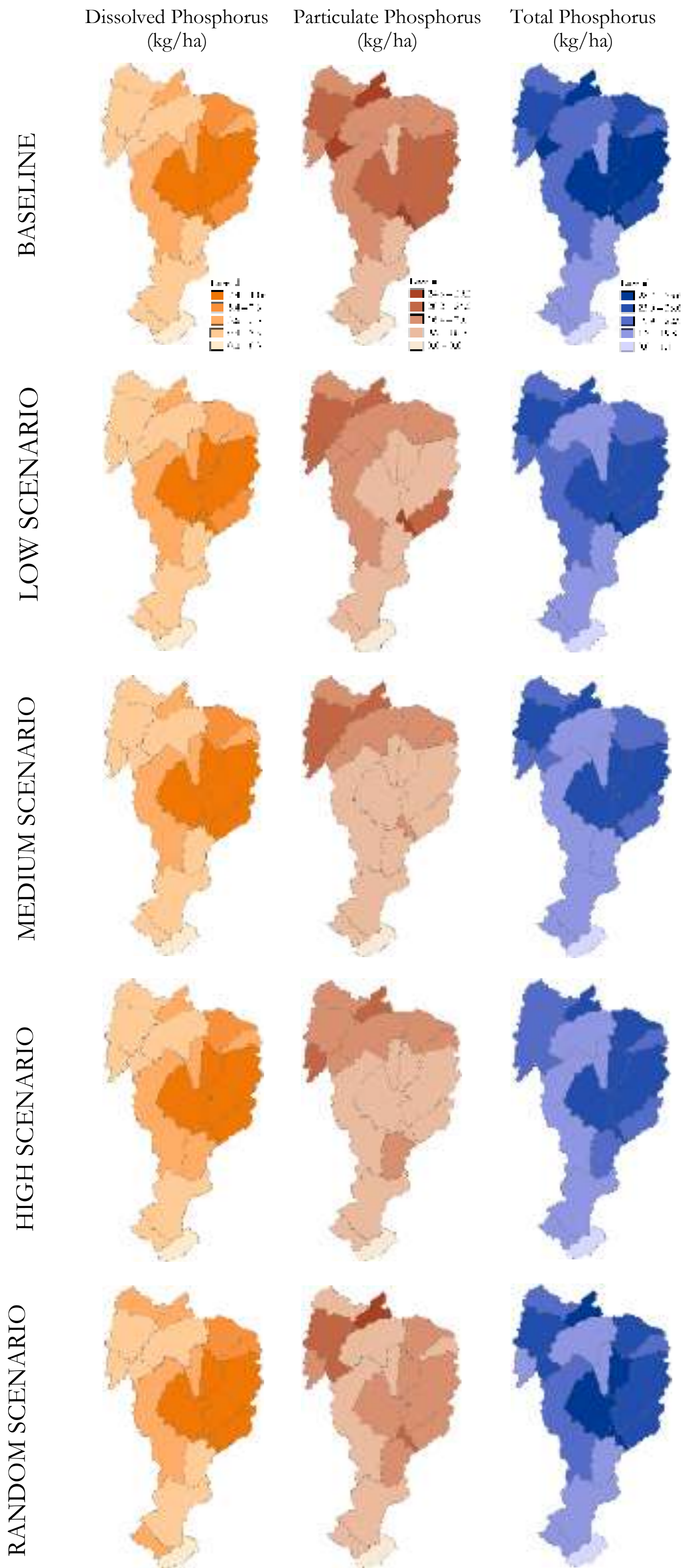

Figure 16. A watershed-level TP, PP and DP in different scenarios

DP pool, however, showed a negative rate of reduction in both random and targeted scenarios with the results illustrating a more accumulation of DP in the watershed in the random distribution scenario. These results also corroborate the findings of Geng et al. (2015) and Tuppad et al. (2010), which acknowledged the higher impact of the targeted placement of BMPs on water quality improvement in the watershed outlet.

In their study on the effectiveness of the different targeting methods on water quality in the Great Lakes watershed, Giri et al. (2012), investigated the difference between the concentration based targeting method and the subbasin load targeting method. Their findings illustrated that the former method was most effective in reducing the total nitrogen and phosphorus loads, whereas the latter method influenced particulate phosphorus loads in the area (Giri et al., 2012). Therefore, when targeted BMP placement is selected in a watershed, the emphasis should also be placed on the targeting strategy to meet the primary goals of BMP implementation project.

3.4.5 Limitations

The simulations of this study were carried out with some assumptions and generalizations. First, the fertilizer and tillage data was only available for the Upper Medway Creek Watershed and was generalized to the whole watershed. Secondly, there was missing weather data for some of the days in the study area. This was generated using the SWAT weather generator. Consequently, precipitation, which influences the amount of surface runoff and TP load across the watershed, might have been misestimated. Thirdly, there is a generalization of LULC and soil type maps in order to define the HRUs based on the agricultural field boundaries. Although there was a minimum variability between the modified LULC and soil type maps (discussed in section 3.2.4.2), this could provide some accuracy problems in the model performance. Finally, essential to any hydrologic modeling is the recognition of the changing patterns in crop planting and selection of the relevant dominant crop rotations to the years for which simulation is conducted (Gao et al., 2017). Due to limited availability of multi-year, crop-specific LULC data for the entire simulation period (1984 - 2013), the dominant crop rotation was defined for a five-year period (2010-2015), and then generalized for the entire simulation period (30 years).

Complex spatiotemporal factors are influencing the agricultural management decisions at the field scale (Merriman et al., 2018a, b), and the existing versions of the SWAT modeling framework are not capable of taking into account some of these complex features. BMP management, for instance, was simplified in this study to model the crop rotation. In this respect, we modeled the most predominant crop in each field as this version of SWAT could consider and simulate one crop on each HRU at a time. More specifically, each HRU represents one field in the Medway Creek Watershed model, and as such, was considered for one crop at a time. Furthermore, planting, harvesting, fertilizer application, and tillage dates were assumed to be identical for all fields and the entire simulation period within the Medway Creek Watershed. In reality, though, management would change on an annual basis and from one field to another to fit soil conditions, weather, and the market price of the crops. Currently, the SWAT modeling framework is not capable of considering operation-delaying mechanisms for events. Precipitation, for instance, leads to delaying the seeding or tillage operation in real-world applications. Nevertheless, this delay mechanism is not integrated into the SWAT model structure (Merriman et al., 2019). These are instances of the real world agricultural situations for which the model does not provide flexibility.

One of the major factors influencing the model accuracy is the number of different crop rotations included in the model. In the current study, we only implemented one type of crop rotation, including corn, soybean, and winter wheat. Although this rotation is the dominant rotation in the study area, the results of the study by Gao et al. (2017) indicated that including higher numbers of different crop rotations in the SWAT model would better represent the farming condition at the watershed scale, which in turn may positively affect the model performance. Nevertheless, this improvement was positive up to a certain threshold represented by the point of marginality (i.e., stasis) and depended, to a large extent, to the size and geographical condition of a given watershed, and as such could yield varying results from one area to another.

3.4 Conclusions

BMPs are one of the primary considerations utilized for maintaining surface water quality. Informed placement of BMPs and optimum rate of implementation play an indispensable role in achieving the desired goals of maximizing the pollution reduction and minimizing the implementation costs. This study was aimed at evaluating the potential impacts of the rate of implementation and spatial distribution of crop rotation on mitigating phosphorus load to the water bodies in an agricultural watershed in southwestern Ontario. Four hypothetical scenarios were simulated to identify the impact of these factors on water quality, considering the phosphorus load as a determining factor.

The results showed that the increased implementation rate of crop rotation improved water quality by decreasing the amount of total phosphorus load (TP). However, this increase was not effective in reducing the dissolved phosphorus (DP) as this pool has a negative rate of reduction, which is consistent with the findings of other studies (Merriman et al., 2018b; Bundy et al., 2001). A 50% implementation rate of crop rotation in the medium scenario was found to be sufficient to elicit a significant change in the total phosphorus load at the watershed scale. Nevertheless, it was proven that this change experienced a negative growth when the implementation rate was increased to 70%, indicating a potential threshold level beyond which no significant change is expected in the TP reduction rates. Therefore, more studies in other watersheds with farming landscapes are required to identify appropriate implementation rates of crop rotation, which would ultimately help to better plan for optimum implementation of this BMP to prevent water impairment and to maintain the expected crop yields at the watershed scale.

The SWAT modeling indicated that the most considerable reductions in phosphorus load were achieved upon the targeted placement of the crop rotation with high rates of implementation. Furthermore, this study has shown that more than a single type of BMP is required to reduce nutrient loads. These results will be helpful to stakeholders when prioritizing future BMPs and action plans.

Chapter 4. Major Conclusions of the Thesis

The modeling research conducted in this thesis aimed to investigate how the spatial distribution and rate of implementation of crop rotation would influence the phosphorus load indicator of water quality in the Medway Creek Watershed, an agricultural landscape dominated by soybean, corn, and winter wheat crops in southwestern Ontario. As a unique BMP with low costs of implementation, crop rotation can be influenced by both environmental and economic factors, thus playing a critical role in the agronomic culture. Consequently, understanding the optimum levels of implementation as well as the spatial pattern of crop rotation, as critical decision rules, is necessary when planning for uncertainties upon the application of this BMP. The results of this study illustrated that the increase in the level of crop rotation implemented across this agricultural watershed positively influences the total phosphorus reduction rates in the watershed outlet. This influence was illustrated to be more effective when the placement of crop rotation is targeted in the fields located in areas with higher contributions to the phosphorus runoff.

Given that crop rotation is a voluntary preemptive measure (Smith et al., 2009), and as such, cannot be applied throughout the entire study area, a scenario-based modeling approach was selected to investigate the optimum allocation of this BMP across the Medway Creek watershed. Accordingly, the SWAT modeling framework was utilized to simulate the considered scenarios. To date, studies on the efficacy of the BMP implementations, assessed the impact of crop rotation as part of a broader set of BMP implementations considered at the field (e.g., Sommerlot et al., 2013; Merriman et al., 2019) and watershed scales (e.g., Panagopoulos et al, 2012; Liu et al., 2016). The scenarios in this research study, however, were designed to assess the watershed-level impacts of the decision variables affecting the efficacy of crop rotation when applied individually. To this end, a set of recently developed techniques were combined to address the limitations associated with the data and modeling framework considered for this study. First, the trajectory analysis method (Wang et al., 2012) was used to ascertain the crop rotation, considering the temporal changes of the land cover in the area. Secondly, as crop rotation is a field-based BMP, this scale of implementation was redefined for the SWAT tool through the set of changes applied in the HRU delineation process - adapted from Klacic et al. (2015) - and modifications made to the SWAT usersoil database. These changes were also necessary to increase the applicability of the model to the actual condition. The next step in the process was the delineation of critical source areas by the LPUAI method (Giri et al., 2012) at the subbasin level. To rank the equally prioritized subbasins, the proximity of each subbasin to the major tributaries was considered as a critical spatial factor affecting the placement decisions. Finally, to identify the critical source areas for the placement of crop rotation at the field scale, an extension of the LPUAI method was applied to the HRUs defined in the model. A distinct advantage of this approach is the prioritization of each field within the prioritized subbasins, which ultimately led to a more optimum placement strategy.

The results of this study signify the difference among the different levels of implementation for crop rotation when this BMP is applied based on the proposed targeted allocation strategy. These results should be of interest to conservation initiatives as well as farmers considering the application of crop rotation in the Medway Creek Watershed by proposing a range of BMP placement proportions to attain desirable phosphorus reduction goals when compared to the zero application level. More specifically, the analysis conducted in this study proved that a 30% application rate would not serve as a minimum effective phosphorus reduction threshold, and to obtain a desirable (statistically significant) reduction goal, practitioner must opt for a higher rate equal or less than 50% implementation rate, considered as the medium rate of application in this study. Going beyond the medium threshold implementation rate will positively influence the total phosphorus reduction rates, but was proven to be statistically insignificant at the 70% rate of implementation as the highest application rate considered in this study, when compared to the proposed 50% implementation rate in the medium scenario.

Considering the significance of the practical aspects of this study regarding the proposed rates of implementation for crop rotation in the Medway Creek Watershed, further research is required to assess the lowest threshold beyond which statistically significant results will be observed for the first time and the upper threshold beyond which the expected rates of total phosphorus reduction can be considered negligible. Based upon the aforementioned results, the former threshold is expected to be between the 30-50% rate of implementation, and the latter threshold is expected to be higher than the 70% application rate considered in this study. Furthermore, considering the method utilized in this study, future scholarly work is required to assess an alternative approach whereby crop rotation can be integrated into the model by changes applied through the primary LULC consideration. More specifically, future research should investigate the impact of the dynamic input of the LULC layer to the model to allow the assessment of time as an independent variable, the consideration of which would potentially influence the results of the total phosphorus reduction rate at the watershed scale.

As a management practice influenced by the spatial variables of pattern and level of implementation, crop rotation is a popular BMP application that positively influences the overall soil productivity. The results of this research study shed light on these decision variables by testing their association with the total phosphorus reduction levels in the Medway Creek Watershed. Accordingly, the results of this study can be considered by the conservation initiatives in the area when crop rotation is part of a larger set of BMP applications. In other words, the share and the type of reduction induced by this particular BMP can help conservation programs when defining the combination of nutrient management measures in the area. The method used in this study can be applied to other watersheds to define the optimum combinations of the two decision variables when crop rotation is considered as a target BMP.

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Appendix

SWAT-CUP is used to calibrate the model. The flow, suspended sediments, nitrate and total phosphorus were calibrated, respectively. Nash-Sutcliffe coefficient was used to measure the efficiency of the calibration.

Calibrated parameters for Flow

Parameter	Definition	Best fit
v_GW_REVAP.gw	Groundwater revap coefficient	0.29
v_ALPHA_BF.gw	Baseflow alpha factor (1/days)	0.89
v_GW_DELAY.gw	Groundwater delay time (days)	6.45
v_GWQMN.gw	Threshold depth of water in the shallow aquifer required for return flow to occur (mm H2O)	490
v_RCHRG_DP.gw	Deep aquifer percolation fraction	0.38
v_TIMP.bsn	Snow pack temperature lag factor	0.71
v_SMTMP.bsn	Snow melt base temperature (˚C)	-0.56
v_SFTMP.bsn	Snowfall temperature (˚C)	0.72
v_SMFMX.bsn	Melt factor for snow on June 21 (mm H2O/˚C-day)	4.2
v_SMFMN.bsn	Melt factor for snow on December 21 (mm H2O/˚C-day)	8.9
v_SNOCOVMX.bsn	Minimum snow water content for 100% snow cover (mm H2O)	23.3
v_SNO50COV.bsn	Fraction of snow volume represented by SNOCOVMX that corresponds to 50% snow cover	0.13
v_CH_K(2).rte	Effective hydraulic conductivity in main channel alluvium (mm/hr)	268
v_CH_N(2).rte	Manning's "n" value for the main channel	0.15
v_CH_K1.sub	Effective hydraulic conductivity in tributary channel alluvium (mm/hr)	106
v_ESCO.hru	Soil evaporation compensation factor	0.37
v_EPCO.hru	Plant uptake compensation factor	0.3
r_OV_N.hru	Manning's "n" value for overland flow	0.18
v_CANMX.hru	Maximum canopy storage (mm H2O)	49
r_CN2.mgt	Initial SCS runoff curve number for moisture condition II	0.05
r_SOL_BD().sol	Soil bulk density	-0.37
r_SOL_AWC().sol	Available water capacity of the soil layer (mm H2O/mm soil)	-0.21

Calibrated parameters for suspended sediment

Parameter	Definition	Best fit
v_ADJ_PKR.bsn	Peak rate adjustment factor for sediment routing in the subbasin	1.9
v_SPEXP.bsn	Channel re-entrained exponent parameter	1.4
v_SPCON.bsn	Channel re-entrained linear parameter	0.0015
v_PRF.bsn	Peak rate adjustment factor for sediment routing in the main channel	0.42
r_USLE_K( ).sol	USLE equation soil erodibility (K) factor	-0.49
v_USLE_P.mgt	USLE support practice factor	0.42
v_CH_COV2.rte	Channel cover factor	0.43

Calibrated parameters for Nitrate

Parameter	Definition	Best fit
v__CDN.bsn	Denitrification exponential rate coefficient	1.03
v__SDNCO.bsn	Denitrification threshold water content	0.85
v__NPERCO.bsn	Nitrate percolation coefficient	0.6
v__N_UPDIS.bsn	Nitrogen uptake distribution parameter	12.9
v__ANION_EXCL.sol	Fraction of porosity (void space) from which anions are excluded	0.37
v__CH_ONCO.rte	Organic nitrogen concentration in the channel (ppm)	57.8

Calibrated parameters for Total P

Parameter	Definition	Best fit
v__BC4.swq	Rate constant for mineralization of organic P to dissolved P in the reach at 20° C (day-1)	0.37
v__ERORGP.hru	Phosphorus enrichment ratio for loading with sediment	2.12
v__P_UPDIS.bsn	Phosphorus uptake distribution parameter	96.6
v__PHOSKD.bsn	Phosphorus soil partitioning coefficient (m3/Mg)	144.9