Impacts of deforestation on water quality and quantity in a Canadian agricultural watershed

By

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A Thesis

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Abstract

Around the world, many forested areas have been and continue to be cleared for expanding agriculture. Canada's remaining forested lands account for around 9% of the world's forest cover. Although only a fraction is lost to deforestation annually (0.02%, 2013), Statistics Canada reports that conversion to agriculture is the most significant driver of forest loss. As climate changes and agricultural demand expands, this trend is expected to continue, and ecosystems will continue to be impacted by resulting habitat loss and hydrological changes that can impact infrastructure and communities. Additionally, changes to sediment and nutrient loadings can harm ecosystems and affect the downstream usability of freshwater supplies.

The impact of increased sediment and nutrient concentrations in freshwater systems has been extensively documented in the literature. In some extreme cases, it can lead to anoxic 'dead zones' in riverine, lacustrine, and marine habitats. Many river systems in Canada have shown elevated nutrient levels in recent years, often tied to the expansion of agricultural land use and destruction of natural forests to increasing nutrient levels in downstream rivers, lakes, and oceans.

This study applies numerical modelling to quantify the influence of forest loss, agricultural expansion and the application of best management practices (BMPs) on water quality and quantity in the South Nation Watershed in eastern Ontario, Canada. The land use in the watershed is mainly agricultural (over 60%) with forest (27%) that is unevenly distributed in the basin. Aerial photography surveys from 2008 and 2014 show a steady decline in forest cover. Recent water quality monitoring has shown nutrient concentrations at or above Canadian water quality standards in many parts of the basin. The Soil and Water Assessment Tool (SWAT) was used to model the

watershed because of its capacity to simulate comprehensive land management scenarios and assess their impact on a variety of water quantity and parameters quickly and effectively.

The work was performed in four steps:

1. Recent land use configurations (2008-2014) in the watershed were acquired, and simplified land use projections based on the direct substitution of cropland for forest land were developed.
2. A numerical model was calibrated and validated for the initial land use scenario.
3. These land use scenarios, as well as more hypothetical scenarios representing more extensive deforestation and reforestation, were used as the basis for hydrological modelling using 31 years of real-world meteorological observations.
4. Idealized vegetated filter strips (VFSs) and grassed waterways (GWWs) were added to the cropped land packages to study the potential of these practices to contribute to the management of water quality.

Analysis of the 33 output datasets derived from simulations of the suite of land use scenarios with and without VFSs and GWWs leads to several conclusions, while also raising some questions. Generally, forests significantly reduce sediment, nitrate and phosphorus outputs to streams as well as slightly reducing water yield compared to cropped areas due to an increase in surface runoff, groundwater and lateral flow combined with the absence of tile drainage. Across subbasins, this translates to significant reductions in sediment, nitrate and total phosphorus loadings entering the river reaches and a slight increase in water yield. At the basin outlet near Plantagenet, Ontario, streamflow and sediment loading show to have little sensitivity to changes in forest and crop cover, while increased forest cover leads to significantly reduced nutrient loadings, particularly in late spring and early winter.

It is clear from this work that continued deforestation will continue to drive further nutrient enrichment in the South Nation River, while VFSs seems to have a significant potential for offsetting some of this enrichment. Streamflow and sediment loadings, however, are not significantly impacted by foreseeable deforestation. The influence of land use change and BMPs was much more significant in the runoff than in exports from the basin, suggesting there would be value in further examination of water quality and quantity at a higher spatial density to expand on assumptions of in-stream processes made here.

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Abbreviations

AAFC	Agriculture and Agri-food Canada
BMP(s)	Best management practice(s)
CCME	Canadian Council of Ministers of Environment
DRAPE	Digital Raster Acquisition Project – East
ECCC	Environment and Climate Change Canada
GWW	Grassed waterways
HRU(s)	Hydrological responses unit(s)
NSE	Nash-Sutcliffe Efficiency
PBIAS	Percent bias
RSR	Root mean square error
SWAT	Soil and Water Assessment Tool
TP	Total phosphorus
VFS	Vegetated Filter Strips

1. Introduction

1.1. Overview

In many regions of the world, large tracts of natural forests have been and continue to be cleared for crop planting and livestock grazing. This is particularly true in Canada since European settlement. This change has a direct impact on ecosystems through the loss of habitat and changes in the hydrological regime. As this trend will likely continue in the coming decades, and given that climate is changing and the population is growing (increasing demand on water resources as well as agricultural produce), we must understand the influence of these modifications on both water quality and quantity. Changes in river flows from their historical regime can force changes to the design of infrastructures such as dams and waterfront developments, while increases in sediment and nutrient flows can have harmful effects on natural ecosystems and fish stocks as well as impacting downstream use for recreation and drinking water supplies.

At over 348 million hectares, Canada's remaining forested lands account for around 9% of the world's forest cover. As in many parts of the world, parts of these lands are slowly being converted to agricultural and urban uses. Deforestation is defined as the permanent removal of forest cover and conversion to other uses (sustainable forestry and natural disturbances such as insect and fire damage contribute to forest change but are not considered deforestation). The Canadian Forest Service (CFS) estimates that in 2013, 45,900 ha (about 0.02% of the total forested areas) were lost to deforestation nationally (Natural Resources Canada 2016). Positively, that rate of deforestation represents a decrease from around 60,000 ha/year circa 1990. This rate is expected to continue to decrease in the future, although national statistics likely do not capture regional variations that would affect particular watersheds (Masek et al. 2011). Statistics Canada (2018) reports that over the period 1990-2015 conversion to agriculture (clearing for pasture or cropping) accounted for 42% of deforestation, followed by natural resource development (24%), urban/transport/recreation development (16%), hydroelectricity development (13%) and forestry roads (6%).

Excessive concentration of nitrogen and phosphorus nutrients in freshwater systems can lead to increasing eutrophication of naturally oligotrophic water bodies. Eutrophication leads to the development of undesirable algae and weeds and associated shortages of dissolved oxygen, which altogether restricts water use for fishing, recreation, industry and domestic consumption. In Canada, nutrient enrichment in river systems has led to severe algal blooms in Lake Winnipeg and Lake Simcoe, and eutrophication problems re-emerging in the lower Great Lakes (Environment and Climate Change Canada 2011). Of 75 assessed monitoring sites, 27 were classified as eutrophic or hyper-eutrophic, corresponding to total phosphorus (TP) concentrations over 0.035 mg/L (ibid.). In a more recent report about the St Lawrence River between Ottawa and Quebec City (downstream of the watershed covered in this study), Environment and Climate Change Canada (ECCC) reported phosphorus and nitrogen levels above water quality guidelines at 8 out of 9 monitoring stations over the 2015-2017 period (Environment and Climate Change Canada, 2019).

In an extensive review, Allan (2004) noted that landscape-scale deforestation activities could be a significant threat to the ecological integrity of watersheds by modifying the geomorphological processes that maintain the river system, frequently resulting in degraded habitats with reduced biodiversity and pollutant buffering capacity. Allan (2004) also notes that further research is warranted to examine hydrological responses to land use under various management strategies.

Other studies that have found that forest cover and riparian areas generally reduce nutrient and contaminant concentrations, and improve invertebrate diversity (Paula et al. 2018, Kändler et al. 2017, Connolly et al. 2016).

In many humid-temperate regions of the world, subsurface (or tile) drainage is installed to remove excess water from agricultural land to optimize agricultural productivity (Hofstrand 2010). To underscore the regional importance of tile drainage, in Ontario, over 1.7 million hectares of tile-drainage have been installed (Ontario Ministry of Agriculture, Food and Rural Affairs 2015), while over 17 million hectares are artificially drained across the midwestern United States (Jaynes and Isenhart 2014). Once tiled, the land is unlikely to be returned to its natural state, given the investment in subsurface drainage infrastructure. Fenelon and Moore (1998) identified elevated nitrate concentrations in river flows as a side effect of tile drainage. Impacts include contamination of downstream public water resources, economic loss due to fertilizer/nutrient leaving the soil profile, stresses to fish communities and eutrophication of receiving waters downstream. A five-fold increase in nitrogen fertilizer application in the Mississippi River Basin from the 1950s to the mid-1990s is thought to be a major contributor to the doubling of nitrate export from the basin and resultant hypoxic conditions in the Gulf of Mexico. Similarly, nutrient loadings in the northeast US, contributing to eutrophication in Chesapeake Bay, are highly correlated with land use changes (Scanlon et al. 2007).

Best management practices (BMPs) are a suite of agricultural and structural measures aiming to reduce the negative impacts of agricultural water management on water bodies. They include specialized or reduced tillage, planting of cover crops and/or vegetative filter strips, and development of grassed waterways. Since the scale of BMP implementation is generally significantly smaller than that of the watershed, field studies of BMP impacts are sparse and show mixed results (Bosch et al. 2013). Numerical modelling approaches are more common and flexible as they allow the consideration of the wide-scale application of BMPs. The Soil and Water Assessment Tool (SWAT) used in this study is able to simulate a range of alternative management practices so users can quickly examine and compare the impact of various BMP scenarios on water quality.

Although others (e.g. El-Khoury et al. 2015a) have studied land use and climate change within this basin, no previous study has made a specific watershed-scale study of the impact of deforestation and expanding agriculture on the flow and quality of water in this watershed. SWAT allows for straightforward modelling of alternative land use and land management scenarios and quantitative estimates of the expected water quantity and quality in the South Nation River.

1.2. Objectives

The main objective of this study is to examine the watershed-scale impact on streamflow and water quality from clearing forest for conversion to cropland within the South Nation watershed. The specific objectives are the following:

1. Characterize trends in forest cover in the South Nation watershed and define a suite of land use scenarios representing de- and reforestation,
2. Recalibrate the SWAT model of the basin developed by Que (unpublished work, 2016),
3. Compile and analyze water quantity and quality results from executing SWAT simulations for the aforementioned suite of land use scenarios,

4. Develop simple empirical relations between land cover and selected water quantity and quality parameters.
5. Additionally, examine the influence of selected best management practices on the results and predictions from the deforestation and reforestation simulations.

1.3. Overview of the Methodology

The study uses SWAT to simulate the streamflow and sediment and nutrient loadings in the South Nation River for a sequence of land use scenarios based on recent deforestation trends within the watershed. Further, more hypothetical scenarios reflecting more significant forest gain and loss were included, as well as considering the additional influence of particular BMPs within the suite of land use scenarios. Initially, a land use model and a trend in forest cover were developed by integrating land use data produced by Agriculture and Agri-Food and Canada (Agriculture and Agri-Food and Canada, 2015) with recent forest cover surveys by the Ontario Ministry of Natural Resources and Forestry (Ontario Ministry of Natural Resources and Forestry, 2009, 2014). This trend was used to develop a series of potential future scenarios, as well as some more hypothetical scenarios which were then simulated with SWAT using 31 years of real-world meteorological observations both with and without the integration of BMPs. From the results of these simulations, changes in streamflow and loadings of sediment, nitrate, mineral phosphorus, total nitrogen and total phosphorus were analyzed to make recommendations about forest reserve management and application of BMPs within the watershed.

1.4. Thesis organization

Following this introduction (Chapter 1), this thesis includes six chapters, followed by references. Chapter 2 looks at some of the existing literature around water quality in river systems and the influence of land use and best management practices on water quality. Chapter 3 introduces the study area considered for this thesis. Chapter 4 covers the materials and methods, including data sources, model configuration, calibration and simulation scenarios. Results and observations of these simulations are then presented in chapter 5 and discussed in greater depth in chapter 6. Chapter 7 is a conclusion with a summary of the implications for land management in the South Nation and the potential for further study.

2. Literature Review

This literature review covers essential topics for the problem under investigation: watershed modelling, water quality issues due to excessive nutrient loading in rivers as driven by land use change and BMPs (best management practices) as a way to limit water course impairment.

2.1. Watershed modelling

A watershed is a geographic area contributing streamflow to a downstream receiving body. Watershed models can simulate some of the hydrological, erosional and depositional processes in the basin, along with as flows of chemicals, nutrients and microbial organisms. These models allow for the quantification of anthropogenic impacts on river systems. They play a significant role in addressing environmental and water resources problems, including flooding, non-point-source pollution, erosion and sedimentation, land degradation and aquatic ecosystem degradation (Novotny 2009; Singh and Frevert 2010). Watershed models can be mechanistic, i.e. developed *a priori* based on known hydrological, chemical and biological processes, or black-box, i.e. developed *a posteriori* using empirical relationships fitted on measured data. The model can be lumped or spatially distributed. Lumped models consider a watershed as a homogeneous unit within which all parameters have a set value everywhere. In distributed models, subwatersheds are further divided into significantly smaller homogeneous units which may be rectangular, triangular or based on consistent morphological and/or hydrological characteristics.

In recent years, rapidly increasing accessibility of computer processing capacity has allowed for a significant expansion in modelling-based studies of land use, climate change, and hydrological and chemical impacts thereof. Numeric modelling also allows researchers to consider changes and interventions that would otherwise take years to implement and test.

2.2. Water quality issues due to excessive nutrient loading in rivers

Eutrophication, the excessive enrichment of nitrogen and phosphorus in freshwater ecosystems, has become a significant issue internationally. Anthropogenic pollution in the form of sewage and industrial discharges, and runoff from agricultural and urban areas are primarily responsible for this nutrient enrichment (Khan and Mohammad 2014). Dorgham (2014) outlined numerous deleterious effects of eutrophication in freshwater systems including, but not limited to:

- Increasing occurrence of algal blooms
- Decreasing water clarity leading to the destruction of benthic habitat
- Hypoxic bottom waters increase mortality of benthic invertebrates
- Increased acidification due to CO_2 production from the decomposition of excess organic matter
- Harm to aquatic fauna leading to decreasing fish stocks with associated ecological and economic impacts

As oligotrophic water bodies are progressively transformed to mesotrophic, eutrophic or even hypertrophic states, the growth of algae and weed is accelerated. Their use for fishing, recreation, industry and drinking then becomes severely restricted. The decomposition of algae leads to increased oxygen demand.

A five-fold increase in nitrogen fertilizer application in the Mississippi River Basin from the 1950s to the mid-1990s is thought to be a major contributor to a doubling of nitrate export from the basin and resultant hypoxic conditions in the Gulf of Mexico (Scanlon et al. 2007). Similarly, nutrient loadings in the northeast US that are responsible for the eutrophication in Chesapeake Bay were found to be highly correlated with land use changes (ibid.). In temperate settings, changes in nutrient loadings have been tied to fertilizer inputs (e.g. Bosch et al. 2013) with agricultural regions facing a tension between the health of aquatic systems and viable agriculture (Withers et al. 2014).

2.3. Influence of land use on stream health

Several studies (e.g. Wang et al. 1997, Richards et al. 1996, Sponseller et al. 2001, Johnson et al. 1997, Roth et al. 1996, Ding et al. 2015, Meneses et al. 2015, Liu et al. 2015) have tied declining water and habitat quality, as well as reduced biological assemblages, to increasing agricultural land use in watersheds, particularly at the watershed scale, with riparian buffers having a weaker influence. Other studies that have considered the influence of forest cover on the health of stream systems have found that forest cover and riparian areas generally reduce nutrient and contaminant concentrations, and improve invertebrate diversity (Paula et al. 2018, Kändler et al. 2017, Connolly et al. 2016). Many studies of the particular influence of woodland in agricultural landscapes consider the influence of riparian buffers (e.g. Schultz et al. 2004, Dosskey et al. 2010, Vigiak et al. 2016), hedgerows (Benhamou et al. 2013, Burel 1996) and/or combinations (Herzog 2000). The conclusion is that these features reduce overall and peak streamflow, as well as trapping sediments in runoff and reducing nutrient content. Conversely, due to the increased interception and evapotranspiration of forests, removal of forest cover has been tied to increases in baseflow and streamflow (Maes et al. 2009, Price 2011, Nagy et al. 2011).

In many developed catchments, agriculture accounts for the largest single fraction of the watershed land use. In North America, agriculture varies from near zero in the north to around 66% in the Upper Mississippi (Allan 2004). Allan noted a range of mechanisms whereby land use can influence stream ecosystems, including sedimentation, nutrient enrichment, pollution by contaminants, hydrologic alteration, riparian clearing and loss of woody debris. These overlap generally result in the degradation and homogenization of stream habitat as well as changes to the physical stability of the channel.

Zhang et al. (2017) reviewed global studies conducted on 312 watersheds and examined the hydrological impacts (primarily runoff) of forest cover change. Those authors analyzed the respective contributions of spatial scale, climate, forest type and hydrological regime to these impacts. A majority of the watersheds reviewed showed that loss of forest cover could increase annual runoff across multiple spatial scales. However, the effect of forest cover gain was more complicated and inconsistent.

Given the significance of the Amazon system and much-publicized agricultural encroachment into the basin, Brazil has recently been a focus of a significant body of research considering the impact of various land use patterns on water quality in streams and rivers in Brazil. Mori et al. (2015) examined physical and chemical properties (turbidity, suspended solids, nutrients) of several agricultural catchments in the Corumbataí basin. They found a difference in the influence of sugarcane versus pasture areas, but overall that the degraded and fragmented forests in this region do not effectively contribute to protecting aquatic ecosystems. In a study of 9 years of pH, dissolved oxygen, electrical conductivity and total suspended solids in the Purus river, Ríos-Villamizar et al. (2017) found no relationship between these parameters and deforestation rates at four observation sites but found correlations between accumulated total deforestation (ATD) and each parameter for at least one observation site. A significant negative relationship between ATD and pH was found at all four locations. The authors explained the relationship by the increased release of hydrogen ions resulting from increased nitrate concentration and nitrification in deforested areas. A paired catchment study of headwater streams in southeastern Brazil conducted by Taniwaki et al. (2017) considered the value of forests in offsetting the negative impacts of intensive agriculture. This study associated the preservation of forests with better water quality overall, including lower nitrate concentrations and stream temperature, concluding that the protection of forests in headwater catchments is essential for maintaining water quality in tropical streams affected by intensive agriculture. In another tropical context, Connolly et al. (2015) measured water quality along four tropical Australian streams in sugarcane growing areas, with varying riparian vegetation including introduced weeds and native rainforest. The authors noted increasing NO_x loadings with distance downstream along with significantly lower NO_x loadings in streams with greater riparian vegetation. However, the loading reductions due to riparian vegetation were not sufficient to meet regional water quality guidelines. In another study by the same lead author, Connolly et al. (2016), also tied the presence of riparian vegetation to improved invertebrate diversity in streams.

Closer to home, Fanelli et al. (2018) studied sources of orthophosphate (PO_4)in tributaries to Chesapeake Bay in the eastern United States and the contribution of PO_4 to total phosphorus (TP) fluxes. Agricultural watersheds were found to export more TP than those dominated by urban or forest, with PO_4 making up 50%. Further, although many sites exhibited declining PO_4 exports, some agricultural areas showed increases sufficient to drive TP trends. Manure application and conservation tillage strongly predicted PO_4 concentrations during high flows, while improvements to loads from point sources accounted for decreasing PO_4 at many sites. In an examination of the impact of riparian vegetation on streambank erosion in an Oklahoma watershed and phosphorus contributions from this erosion, Miller et al. (2014) also linked the presence of riparian vegetation with significantly reduced streambank erosion and accompanying TP contributions to the stream. Basing on two soil microcosm experiments and a field case study, Neilen et al. (2017) attempted to directly compare the influence of woody versus grassed riparian vegetation for the interception of N and P. For P export. Woody vegetation was found to be superior, irrespective of rainfall. At the same time, the results for N export were more complex – grassed riparian zones were superior to woody vegetation under high rainfall conditions while soil conditions had more influence than vegetation type in low rainfall conditions.

Many recent modelling studies have considered the influence of different land use and land management scenarios on watershed hydrology and water quality (Álvarez-Cabria et al. 2016, El-Khoury et al. 2015, Fan and Shibata 2015, Gong et al. 2019, Hlásny et al. 2015, Lin et al. 2015, Neupane and Kumar 2015, Phung et al. 2019, Que et al. 2015). Álvarez-Cabria et al. (2016) modelled spatial and seasonal variability of water temperature and concentrations of nitrates and phosphates for a large area of northern Spain using the random forest technique with 12 water quality parameters and 15 environmental attributes (such as climate, topography, land use and hydrology). They found that land uses were important predictors for variations in the three key variables. Specifically, water temperature decreased with increasing forest cover, while nitrate concentration was mostly related to agricultural land use. Phosphate concentration was mostly related to urban land uses upstream and proximity of upstream effluent sources. Fan and Shibata (2015, 2016) predicted future land use in the forest-dominated Teshio watershed in northern Hokkaido, Japan to be a conversion of forest and rice paddy areas to farmland. They modelled the impact of these land use changes on water yield and sediment and nutrient exports using SWAT. These studies found that although there is a reduction in sediment and nutrient loads related to the conversion from paddy to farming, these loads were mainly supplied from agricultural land. The studies confirm the value of forests for stream health and suggested that forest or riparian zone management and improved fertilizer management may be of value. Gong et al. (2019) alternately studied two differing future land use scenarios for the Dongliao River Basin, China, using SWAT: one based on historical trends and a second on an ecological protection scenario. Again, the expansion of dryland and urban areas at the expense of forest and grassland were significant contributors to increasing non-point contamination of the river. Hlásny et al. (2015) also found deforestation has a significant influence on water balance components of a small forest-dominated Slovakian watershed. The authors simulated water balance with SWAT based on real meteorological records for 2001-2013 with two extreme scenarios – the current status with over 84% forest cover and an extreme deforestation scenario where almost all forest and shrubland is converted to pasture. Under the deforestation scenario, interception and evapotranspiration were significantly decreased, while runoff, baseflow and daily discharge increased significantly. Lin et al. (2015) considered a complex exchange of land uses (increases in bioenergy crops at the expense of wheat, forest and pasture) which confounds the influence of any single land use type. Nonetheless, this study found that the land use conversion results in increased variability of downstream streamflow (and associated uncertainty with flood forecasting) and increased sediment and nutrient loadings (most significantly from phosphorus).

Several studies have also considered the impacts of climate, land use and drainage management on the health of the South Nation River (e.g., El-Khoury et al. 2015; Que et al. 2015; Sunohara et al. 2015; Wilkes et al. 2011). Of particular relevance here is the work of (El-Khoury et al. 2014, 2015b) who considered future land use trends and the influence of these trends on streamflow and loadings of nutrients in the river. In their first paper, the authors calibrated and applied the Conversion of Land use and its Effects (CLUE) model to develop land use projections for the South Nation watershed from 2012-2050. These projections suggest the expansion of the Ottawa metropolitan area will encroach further into the northwest of the watershed, while across the rural areas, cropland will continue to encroach on forest areas, with many small patches likely to disappear. In the second paper, the authors considered the water quality and quantity in the South Nation using the SWAT model for the future period 2025-2050 considering climate change projections and land use projections separately and together, as compared to the historical 1985-2010 reference period. In summary, streamflow was increased by climate change, but only slightly increased by land use change when isolated from climate change (within the inter-annual variation), while increases in loadings of nitrogen species were mostly driven by land use and partly balanced by climate change. Increases in loadings of phosphorus species were driven cumulatively by both climate change and land use change.

2.4. BMPs and stream water quality

Brooks et al. (2012) define BMPs as non-structural (vegetative) or structural (engineering) methods, measures or practices that reduce the flow of sediment, nutrients, pesticides and other pollutants into stream systems.

2.4.1. Types of nutrient BMPs

In parallel with various studies of land use and climate change impacts, there is a continually expanding body of literature examining the economic and ecological costs and benefits of BMPs in diverse locations, most frequently through numerical modelling (e.g. Bosch et al. 2013; Haas et al. 2017; Hanief and Laursen 2019; Kaini et al. 2012; Motsinger et al. 2016), and occasionally through observational studies of test watersheds (Baker et al. 2018). Several different BMPs have been studied for their capacity to reduce runoff volume and velocity and reduce the loadings of sediment the nutrients in river systems. These approaches include:

- "No-till" farming – where tillage is reduced, or abandoned after autumn harvest, minimizing soil disturbance and reducing erosion and flushing of nutrients;
- Planting of cover crops – cover crops are planted after autumn harvest and ploughed into the soil before spring planting, thus improving soil stability during winter and spring and improving organic content in the soil before planting;
- Reduction of fertilizer and/or manure applications – reducing the application of these nutrient sources inherently reduces nutrient flows, but potentially at a cost to yield;
- Construction of detention ponds – these pools can reduce peak flows of runoff and sediment and nutrients therein;

- Terracing and contour cropping – can reduce surface runoff velocity, and hence erosion and flushing of nutrients, but less effective with shallow slopes;
- Controlled tile drainage – controlling the outflows from drainage tiles allows for control of water and nutrient flows similar to detention ponds;
- Vegetated filter strips or buffer strips – planting of dense vegetation downslope of cropland to reduce surface runoff velocity and allow greater infiltration and settling of particulates;
- Grassed waterways – these built channels, planted with grass or other vegetation collect and direct surface runoff to a stable outlet while reducing flow to non-erosive velocity and allowing settling of particulates and take-up of nutrients by the vegetation.

2.4.2. Effects of BMPs on water quantity and quality

Most relevant to this study is the work of Parker et al. (2008) which used the Annualized Agricultural NPS (AnnAGNPS) model to examine water quality impacts of a suite of BMPs. The authors found that adjusting fertilizer application rates and the installation of vegetated filter strips out-performed other options but were nonetheless unable to meet water quality standards without a reduction in agricultural activity within the basin.

In an observational study, Baker et al. (2018) used a field-scale, paired watershed approach to study sediment and nutrient runoff concentrations related to four management systems in two small watersheds in the Mississippi Alluvial Valley. The authors found variability in concentration from their sampling data suggested a strong influence from environmental and management variables, with no significant improvements in water quality from areas where BMPs have been implemented. Nonetheless, it is suggested that there is potential for better capture of nutrient dynamics at field scale to improve monitoring and reduction of NPS pollution.

These modelling studies primarily use SWAT simulations of alternative BMPs with a varying approach to results analysis and different suites of BMPs applied. Bosch et al. (2013) and Hanief and Laursen (2019) both used SWAT to examine watersheds draining into western Lake Erie, which has seen extensive algal blooms and hypoxia in recent decades. Bosch et al. studied six watersheds comprising 53% of the Lake Erie Basin. They found that hypothetical pristine nutrient yields without anthropogenic influence may be an order of magnitude lower than current yields. However, this study found that even in combination, the application of cover crops, filter strips and no-tillage of corn and soy crops reduced sediment and nutrient yields by only 0-11%. Selective targeting of these BMPs found the best value for nutrient reduction was achieved when placed in high yield areas, while the best value for sediment reduction occurred when BMPs were placed near the river outlet. Alternately, Hanief and Laursen tested wide buffer strips in conjunction with conservation tillage, bank stabilization and grassed waterways. They found that all three approaches significantly reduced sediment loads (by up to 38%, from bank stabilization) and TP loading (by up to 50%, from buffer strips) while having very little influence on streamflow. These authors also considered idealized reforestation and restoration of wetlands by adjusting land use fractions within subbasins where these land uses coexist with agriculture. They found relatively modest impacts on sediment and TP: converting 15% of agriculture to forest in each subbasin decreased both loadings by 9%. The impact of wetland restoration was limited as only a few subbasins contain both agricultural lands and wetlands, so the area converted was small. Haas et al. (2017) also modelled the effectiveness of four BMP approaches in a northern German watershed dominated by farming (48% agriculture, 31% pasture) to reduce nitrate loads at the basin outlet. This watershed is also extensively tile-drained. These authors tested a few configurations of buffer strips, as well as converting crop areas to pasture, converting corn grown for biogas to rye, and reduced fertilizing. Of these processes, simulated separately, some impacts seemed intuitive: reducing fertilizer applications by 15-30% had the greatest impact on nitrate loads, reducing loadings by a similar percentage, while buffer strips achieved 3-9% reductions, with wider strips having more influence. At the other end of the scale, increasing pasture area had little influence, while changing half the corn cropland to rye only reduced nitrate loadings by 3%. Combining these BMPs had approximately cumulative effects, where maximizing all of them decreased loadings by over 38%. The authors also considered costs of applying BMPs, finding reduced fertilizer applications to be significantly more cost-effective than the other approaches since yield reductions were partly offset by reduced expenditure on fertilizer.

3. Study Area

The South Nation watershed covers 3753 km^2 between the St Lawrence River and the Ottawa River in eastern Ontario, Canada (Figure 1). Eastern Ontario's climate is classified as humid continental with significant seasonal variation between cold, snowy winters and warm, humid summers. Temperatures are consistently below freezing from December to March and most precipitation falls as snow in that period. The river then experiences significant spring flooding in March and April as rainfall combines with snowmelt and high soil water content. The river flows generally north-east from headwaters near Brockville, Ontario, descending approximately 80 m over approximately 110 km before discharging into the Ottawa River about 50 km east of Ottawa (South Nation Conservation 2014). Except for the Russell and Prescott Sand Plains along the

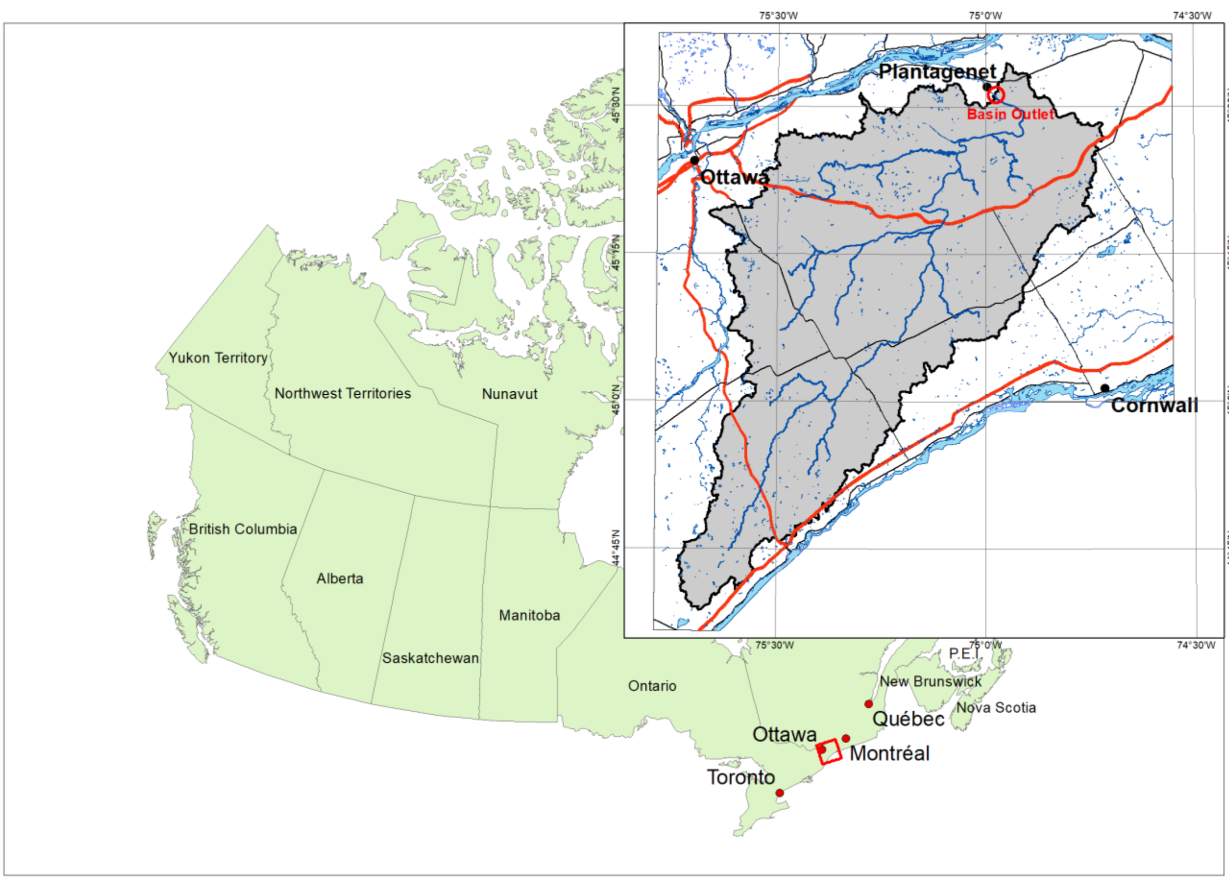

Figure 1 Location of the South Nation watershed and layout of river system.

northern margin, the watershed is dominated by clays and loam soils with poor drainage (South Nation Conservation 2014). The soil characteristics coupled with the region's low relief mean the watershed is at significant risk of flooding (particularly in the lower reaches) and riverbank and topsoil erosion. At the same time, agricultural areas feature extensive artificial drainage (as open drains and buried tile drains) to reduce pooling and improve crop management. Land use in the South Nation watershed is predominantly tile-drained agriculture (62% in 2008) and forests (28% in 2008) with the remainder classified as waterways/wetlands or built-up areas. There are approximately 101,100 ha of forested land within the watershed (Figure 2). However, 5,124 ha of forests were lost between 2008 and 2014, representing 4.8% of total forest cover (Ontario Ministry of Natural Resources and Forestry 2009, 2014, Figure 3). At just 26.9% (in 2014), the total forested cover is below the "high-risk" threshold of 30% recommended by Environment and Climate Change Canada (ECCC). Even at 30%, less than half of all potential species richness and only marginally healthy aquatic systems may be supported (Environment and Climate Change Canada 2013). Additionally, total phosphorus exceeds the provincial standard (0.035 mg-P/L) throughout the monitoring stations, while nitrate is below the national standard (3.0 mg-N/L) for all but two monitoring stations in the watershed (South Nation Conservation 2014). South Nation Conservation's most recent State of the Nation report card rates most surface water in the watershed to be of poor to fair quality based on Canadian Council of Ministers of Environment (CCME) standards for TP and benthic macroinvertebrate populations (South Nation Conservation 2018).

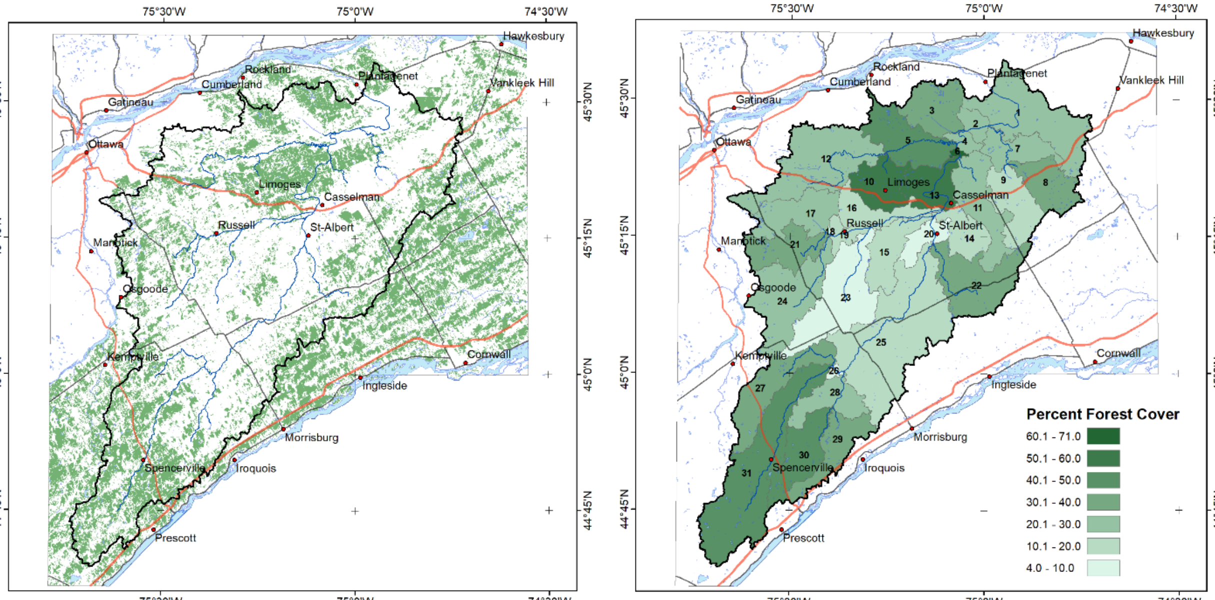

Figure 2 DRAPE forest cover distribution (left) and the breakdown by subbasin for simulation (right)

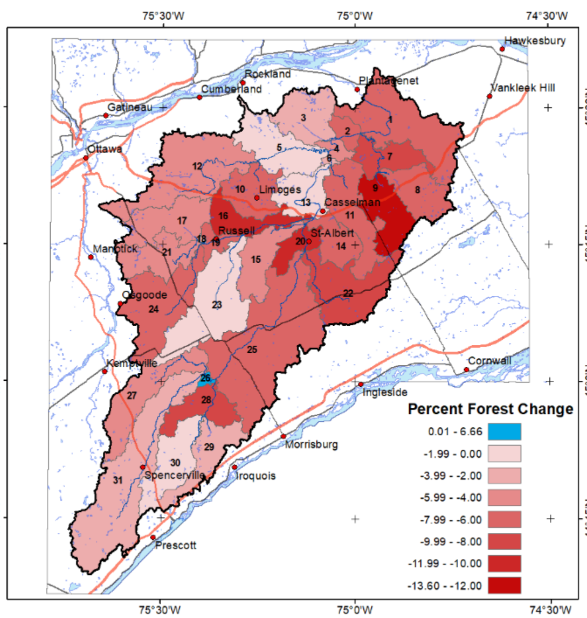

Figure 3 Forest cover change from 2008 to 2014 as a percentage of 2008 levels.

4. Materials & Methods

4.1. Soil & Water Assessment Tool (SWAT)

SWAT is a deterministic, semi-distributed and continuous-time model developed by the United States Department of Agriculture (USDA) to assist with assessments of the impact of land management on water supply and quality in watersheds (Arnold et al. 1998). The model allows for the assessment of complex watersheds by dividing the system into subwatersheds or subbasins, which are generally delineated such that the entire area drains to a single outlet. The components of the model for each subbasin are hydrology, weather, sedimentation, soil temperature, crop growth, nutrients, pesticides and agricultural management. Each subbasin contains one or more hydrologic response units (HRUs), which define a fraction of the subbasin with consistent land use, management and soil characteristics. Loadings (runoff with sediment, nutrients, etc.) are calculated for each HRU then summed to define total subbasin loadings. Although computations are based on a daily time step, SWAT is intended to be a long-term yield model for simulation of conditions over months and years, and not suited to simulation of single storm events. SWAT's hydrological simulation is based on the water balance in two phases. The land phase is based on the fundamental water balance equation:

$$SW_t = SW_0 + \sum_{i=1}^{t} \left(R_{day} - Q_{surf} - E_a - w_{seep} - Q_{gw} \right)$$

Where *SW* is soil water content at the final time, *t*, and time 0, R_{day} is the amount of precipitation on day *i*, Q_{surf} is the amount of surface runoff on day *i*, E_a is the amount of evapotranspiration on day *i*, w_{seep} is the amount of water entering the vadose zone from the soil profile on day *i*, and Q_{gw} is the amount of return flow on day *i*, all in mm H_2O. Secondly, the calculated loadings of water, sediment and nutrients from the HRUs are routed through the stream network where SWAT tracks mass flow as well as modelling the transformation of chemicals in the stream and streambed (Neitsch et al. 2011).

4.2. The South Nation Watershed Model

4.2.1. Model structure and sources

For the South Nation watershed, the watershed limits and 31 subbasins were defined from Shuttle Radar Topography Mission (SRTM) elevation data available from the United States Geological Survey (USGS) combined with the stream geography extracted from the National Topographic Database (NTDB) maintained by Natural Resources Canada.

Soil properties were derived from the Soil Landscapes of Canada version 3.2 (Soil Landscapes of Canada Working Group, 2010), including hydrologic soil group, soil layers, texture, hydraulic conductivity and organic content. Soil chemistry for forested areas was generalized from sampling carried out by Agriculture and Agri-food Canada.

The land use distribution was derived by combining data from the Agriculture and Agri-food Canada (AAFC) Crop Inventory for 2015 (2015 version was selected for its superior resolution) and forest coverage from the Digital Raster Acquisition Project Eastern Ontario 2008 and 2014 (DRAPE, Ontario Ministry of Natural Resources and Forestry, 2009, 2014). DRAPE is part of a 5-year plan to acquire high-resolution, leaf-off aerial photography across all of Ontario and was last acquired in spring 2014 under ideal conditions. Land uses across the watershed have been generalized as water bodies, wetlands, urban/built-up areas, forest or cropland. Table 1 outlines the fractions in the model of each land use across the entire watershed.

Table 1 South Nation Watershed delineation for SWAT simulation

Watershed area	3753.3 km^2
Number of subbasins	31
Number of HRUs	6040
Simulation period	1981-2011
Crop area (2008)	61.9% (2323.4 km^2)
Forest area (2008)	28.3% (1061.3 km^2)
Urban/built-up (2008)	5.7% (215.1 km^2)
Wetlands and water bodies (2008)	4.1% (153.0 km^2)
Forest change (2008-2014)	51.2 km^2 (net loss)

4.2.2. Crop selection and management

The management of units defined as cropland was standardized to an 8-year rotation of corn-corn-corn-soy-alfalfa-red clover-timothy-alfalfa, with the starting point in the cycle randomized for each cropland HRU. The modelled crop and land management cycle is shown in Table 2. Timing and fertilizer specifications were based on personal communication with staff at AAFC and represent a simplified typical cropping rotation for farming in the South Nation watershed.

Table 2 Crop rotations and management operations

Crop/Planting	**Operation**	**Date**	**Details**
Corn	Tillage	1 May	Coulter-Chisel Plow, 50% mixing efficiency, 150 mm depth
	Fertilizer application	5 May	Elemental N 180 kg/ha, Elemental P 30 kg/ha

	Planting	7 May	
	Harvest and kill	15 October	
	Tillage	20 October	Moldboard Plow 24-6, 95% efficiency, 150 mm depth
Soybean	Tillage	1 May	Coulter-Chisel Plow, 50% mixing efficiency, 150 mm depth
	Fertilizer application	18 May	Elemental N 7.0 kg/ha, Elemental P 20 kg/ha
	Planting	20 May	
	Harvest and kill	30 September	
	Tillage	5 October	Moldboard Plow 24-6, 95% efficiency, 150 mm depth
Alfalfa (year 5)	Planting	1 May	
	Fertilizer application	1 June	Elemental N 60 kg/ha, Elemental P 20 kg/ha
	Harvest only	3 June	
	Harvest only	30 July	
	Harvest only	1 September	
	Kill/end of growing season	30 October	
Red Clover	Planting	1 May	
	Fertilizer application	1 June	Elemental N 60 kg/ha, Elemental P 20 kg/ha

	Harvest only	3 June	
	Harvest only	30 July	
	Harvest only	1 September	
	Kill/end of growing season	30 October	
Timothy	Planting	1 May	
	Fertilizer application	1 June	Elemental N 60 kg/ha, Elemental P 20 kg/ha
	Harvest only	3 June	
	Harvest only	30 July	
	Harvest only	1 September	
	Kill/end of growing season	30 October	
Alfalfa (year 8)	Planting	1 May	
	Fertilizer application	1 June	Elemental N 60 kg/ha, Elemental P 20 kg/ha
	Harvest only	3 June	
	Harvest only	30 July	
	Harvest only	1 September	
	Kill/end of growing season	15 October	
	Tillage	20 October	Moldboard Plow 24-6, 95% efficiency, 150 mm depth

4.2.3. Best Management Practices

From the suite of BMPs packaged with SWAT, terracing, contour planting and strip cropping are likely of limited value in the very-low-relief geography of the South Nation watershed. Residue management was considered outside the scope of this study. However, vegetated filter strips (VFSs, Figure 4) and grassed waterways (GWWs, Figure 5) have been simulated to examine their potential to contribute to streamflow and loadings of sediment and nutrients.

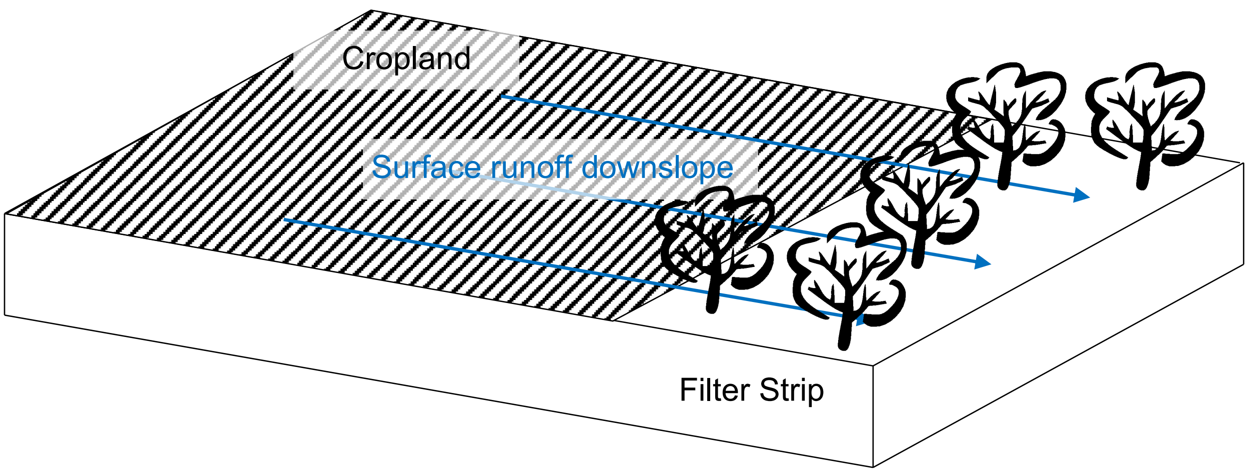

Figure 4 Layout of Vegetated Filter Strip (VFS), planted downslope of crop growing land to intercept runoff and contaminants.

Both VFSs and GWWs can remove agricultural and urban pollutants from runoff before it reaches lakes and rivers. Vegetated filter strips are maintained strips of planted and/or indigenous vegetation located downslope from non-point pollution sources such as agricultural fields, while GWWs are artificial or natural vegetated channels that reduce and regulate flow (Kaini et al. 2012). The first versions of SWAT utilized a simplistic trapping efficiency for VFSs calculated from the width of the filter strip, which was applied to all soluble and insoluble components in the runoff. However, after a review of the literature found considerable variation in the efficiency of filter strips to capture different species, White and Arnold (2009) developed empirical models for inclusion in SWAT for the reduction of runoff, sediment, total nitrogen, nitrate, total phosphorus and soluble phosphorus from Vegetated Filter Strip Model (VFSMOD) simulations and published data.

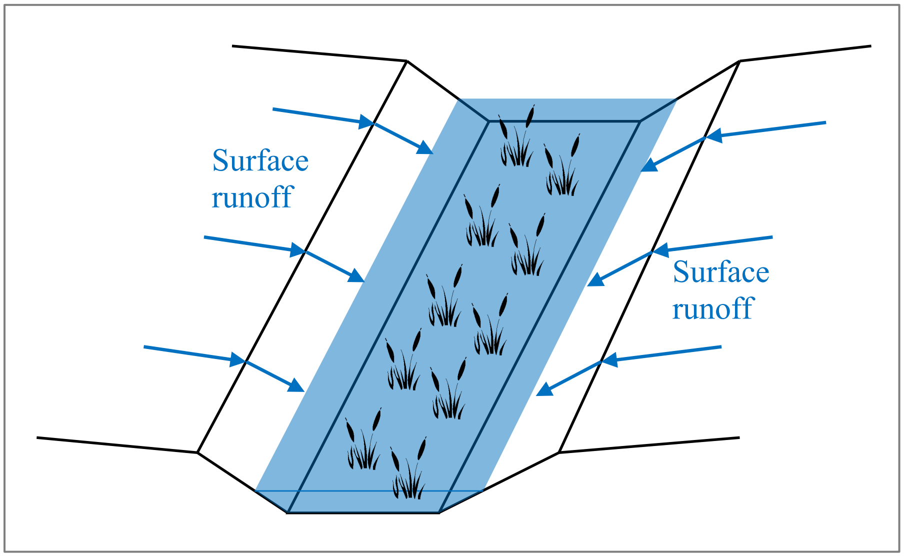

Figure 5 Layout of Grassed Waterway (GWW), installed between plots to collect surface runoff and capture contaminants.

GWWs are simulated as trapezoidal channels and generally treated in the same way as subbasin tributary channels, based on a simplified version of Bagnold's definition of stream power. However, the sediment transport capacity is defined specifically for GWWs as a function of the sediment transport coefficient and flow velocity, while the unsubmerged portions of the GWWs are treated as filter strips, and the removal of soluble and particulate pollutants is based on simplified forms of the VFS equations.

4.3. Hydrometeorological data

Meteorological inputs required for SWAT simulations include daily precipitation, maximum and minimum air temperatures, solar radiation, wind speed and humidity. For this study, precipitation, temperature and wind speed data were drawn from Environment and Climate Change Canada's archives for four stations in or near the watershed at Russell, St Albert, South Mountain and Morrisburg. Solar radiation and humidity data were extracted from the National Centers for Environmental Prediction/National Center for Atmospheric Research (NCEP/NCAR) reanalysis archive.

Monthly water quality and streamflow data for calibration and validation were sourced from monitoring stations maintained by Environment and Climate Change Canada (for streamflow) and the Ontario Provincial (Stream) Water Quality Monitoring Network (PWQMN) near the watershed outlet in Plantagenet, Ontario. Calibration and validation were limited to a whole-basin approach using the Plantagenet stations due to poor monitoring coverage (across space and/or time) elsewhere in the basin.

4.4. Calibration and validation

The calibration of the model was performed using SWAT-CUP (Abbaspour et al. 2007a) with the SUFI2 (Sequential Uncertainty Fitting) algorithm, based on:

1. monthly nutrient and sediment observations from the PWQMN for 2000-2011;
2. monthly averaged streamflow for 2000-2011;
3. daily meteorological data for 2000-2011
4. the baseline (2008) land use and cropping scenario described in Tables 1 and 2

The software operates by iterating the simulation using parameters within a user-defined range with the best simulation based on selected statistical measures including many of the figures in (Moriasi et al. 2007). In this case, Nash-Sutcliffe Efficiency was maximized.

The following relevant observations are directly available from the ECCC and PWQMN monitoring stations: monthly average streamflow (m^3/s) and concentrations (mg/L) of ammonium, nitrite, nitrate, total nitrogen, and soluble and total phosphorus. Concentration observations were converted to monthly loadings assuming individual measurements are representative of average monthly conditions and negligible change in flow and water quality between the two monitoring locations. While streamflow sampling is almost continuous over the calibration period (only one month missing, January 2007), water quality monitoring is consistently interrupted by winter and spring conditions such that calibration was limited to the period April-November.

The calibration process with SWAT-CUP was based on the recommendations in Abbaspour et al. (2007b) and Abbaspour et al. (2015). Firstly streamflow is refined, before adding further parameters that control sediment loadings and finally nutrient loadings. Validation was based on equivalent datasets for the years 2012-2017. Final calibration and validation metrics were calculated for months where observations were available and evaluated based on the recommendations of Moriasi et al. (2007). Moriasi et al. (2007) recommended three statistical measures and corresponding thresholds for evaluation of hydrological models: the classic Nash-Sutcliffe efficiency (NSE) (Nash and Sutcliffe 1970); Percent bias (PBIAS), proposed by Gupta et al. (1999) and the authors' own Root Mean Square Error-observations standard deviation ratio (RSR). In a perfect simulation, NSE=1 and PBIAS=RSR=0. Conversely, NSE=0 represents a simulation no better than the average of observations, RSR increases without limit for poor simulations, while positive and negative deviations from PBIAS=0 represent model under- or overestimation bias, respectively. Further, those authors recommended thresholds for each measure to classify model performance as unsatisfactory, satisfactory, good or very good.

4.5. Forest change scenario generation

To investigate the impact of forest loss on water quality properties in the South Nation river basin, simulations were repeated for a series of forest cover scenarios. The initial land use balance, representing the land use profile of the watershed in 2008, is outlined in

Table 1, based on sources listed in section 4.2.1. The second land use scenario was developed using forest cover data from the 2014 DRAPE survey (DRAPE, Ontario Ministry of Natural Resources and Forestry, 2014). In that scenario, all changes in forest cover from 2008 are assumed to be converted to tile-drained cropland, with other covers (urban, wetland, water) unchanged. Tile drainage was defined within SWAT using the following assumptions:

1. All tiles are buried at 1 m depth
2. Drains are spaced at 15 m;
3. Drain effective radius is 40 mm;
4. LATKSATF (multiplication factor to determine lateral k_{sat} from SWAT k_{sat} input value) set to 1.0
5. Time to drain to field capacity and drain lag time calibrated to 39.9 hours and 69.7 hours, respectively.

The changes in the SWAT model structure were as follow:

- Since each HRU represents a fraction of the surrounding subbasin, these fractions were updated by defining the change in forest cover (and the corresponding change in crop cover) for each subbasin. This was done by determining multipliers for the crop and forest fractions which were applied to adjust all the HRUs within each subbasin.

- Once the factors required to adjust from 2008 to 2014 forest cover scenarios were estimated, the multiplier used to adjust those HRU fractions was reapplied to simulate a sequence of future forest cover scenarios, assuming similar rates of forest change. This resulted in a relatively narrow range of overall forest cover (22.2-28.3% across the watershed).
- Five further scenarios were also developed with overall forest cover at 10%, 20%, 30%, 40% and 50%. Land cover profiles for these scenarios were calculated by scaling the 2008 forest and crop cover HRUs for each subbasin by the value required to adjust the overall 2008 cover to each level. In this way, the variation of forest cover between subbasins was maintained. The last three also correspond to the forest cover thresholds considered high, medium and low risk, respectively, for overall ecosystem health by ECCC (Environment and Climate Change Canada 2013).

The 11 scenarios are outlined in Table 3. SWAT was then used to simulate the watershed behaviour for each of these 11 scenarios with 31 years (1981-2011) of real-world meteorological observations. This period includes a range of weather variations that may not be adequately sampled by a shorter simulation. This range of climate observations also includes a wide range of annual rainfall totals, which allows the consideration of precipitation as a predictor of water quality and flows.

Table 3 Forest and crop cover scenarios simulated. The remainder of each scenario is a fixed balance of urban areas, wetlands and water bodies

Scenario	**Forest cover**	**Crop cover**
0: 2008 DRAPE	28.3%	61.9%
1: 2014 DRAPE	26.9%	63.3%
2: 2008 - 9.4% forest (+6 years)	25.6%	64.7%
3: 2008 - 13.6% forest (+12 years)	24.4%	66.2%
4: 2008 - 17.6% forest (+18 years)	23.3%	67.7%
5: 2008 - 21.3% forest (+24 years)	22.3%	69.2%
10% forest cover	10%	80.2%
20% forest cover	20%	70.2%
30% forest cover	30%	60.2%
40% forest cover	40%	50.2%
50% forest cover	50%	40.4%

4.6. BMP simulations

For both BMPs, control variables were defined based on watershed characteristics or SWAT defaults. For the simulation of VFS, the key parameter is the ratio of field area to filter area. Using an average field size of approximately 15ha in South Nation, a VFS of width 5.0 m along two sides of a square field yields a ratio very close to the SWAT default ratio of 40 which was used here. For the simulation of GWW, the average width of the waterway was set to 5.0 m and depth to 0.5 m with other variables allowed as defaults. These BMPs were simulated separately across all forest cover scenarios.

Table 4 Variable definitions for BMP simulations

BMP	Variable	Value
Vegetated Filter Strip	FILTER_RATIO	40
	FILTER_CON	0.5 (default)
	FILTER_CH	0.0 (default)
Grassed Waterways	GWATN	0.35 (default)
	GWATSPCON	0.005 (default)
	GWATD	0.5 m
	GWATW	5.0 m
	GWATL	Single side of square HRU (default)
	GWATS	0.75 × HRU slope (default)

5. Results

5.1. Model calibration and validation

Statistical measures for the calibration period for key variables are shown in Table 5, along with the performance classifications based on Moriasi et al. (2007). As mentioned previously, the authors recommended the use of Nash-Sutcliffe Efficiency (NSE), Percent Bias (PBIAS) and RMSE-observations standard deviation ratio (RSR), and published performance ratings for each statistic for monthly time-step. Figure 6 shows the observed and simulated time series for the nine calibrated variables over the period 2000-2011. Based on the statistical measures alone, calibration can be considered at least satisfactory for the key parameters of interest to this study.

Table 5 Selected statistical measures of model calibration from Moriasi et al. (2007).

Variable	**NSE**	**PBIAS***	**RSR**
Streamflow	0.76 (very good)	-1.9 (very good)	0.49 (very good)
Sediment loading	0.66 (good)	-30.9 (satisfactory)	0.58 (good)
Nitrate loading	0.61 (satisfactory)	6.9 (very good)	0.63 (satisfactory)
Mineral phosphorus loading	0.51 (satisfactory)	-39.3 (good)	0.70 (satisfactory)
Total Nitrogen	0.69 (good)	8.7 (very good)	0.56 (good)
Total Phosphorus	0.66 (good)	-26.9 (good)	0.58 (good)

* Moriasi et al. (2007) recommend different rating thresholds for each of streamflow, sediment loading and nutrient loadings.

Visual assessment of observed and simulated parameters through the calibration and validation leads to similar conclusions about model calibration. However, the lack of sediment and nutrient observations during winter and early spring means the calibration of these parameters is limited at best. Generally, the model slightly underestimates the highest spring flows and loadings and often slightly overestimates the summer flows and loadings. Nonetheless, the simulation appears to reasonably characterize these major parameters, particularly the streamflow and total nutrient loadings, while the calibration of sediment, nitrate, and mineral or soluble phosphorus is acceptable, with the caveat that the percentage bias indicates a notable overestimation of sediment and phosphorus loadings.

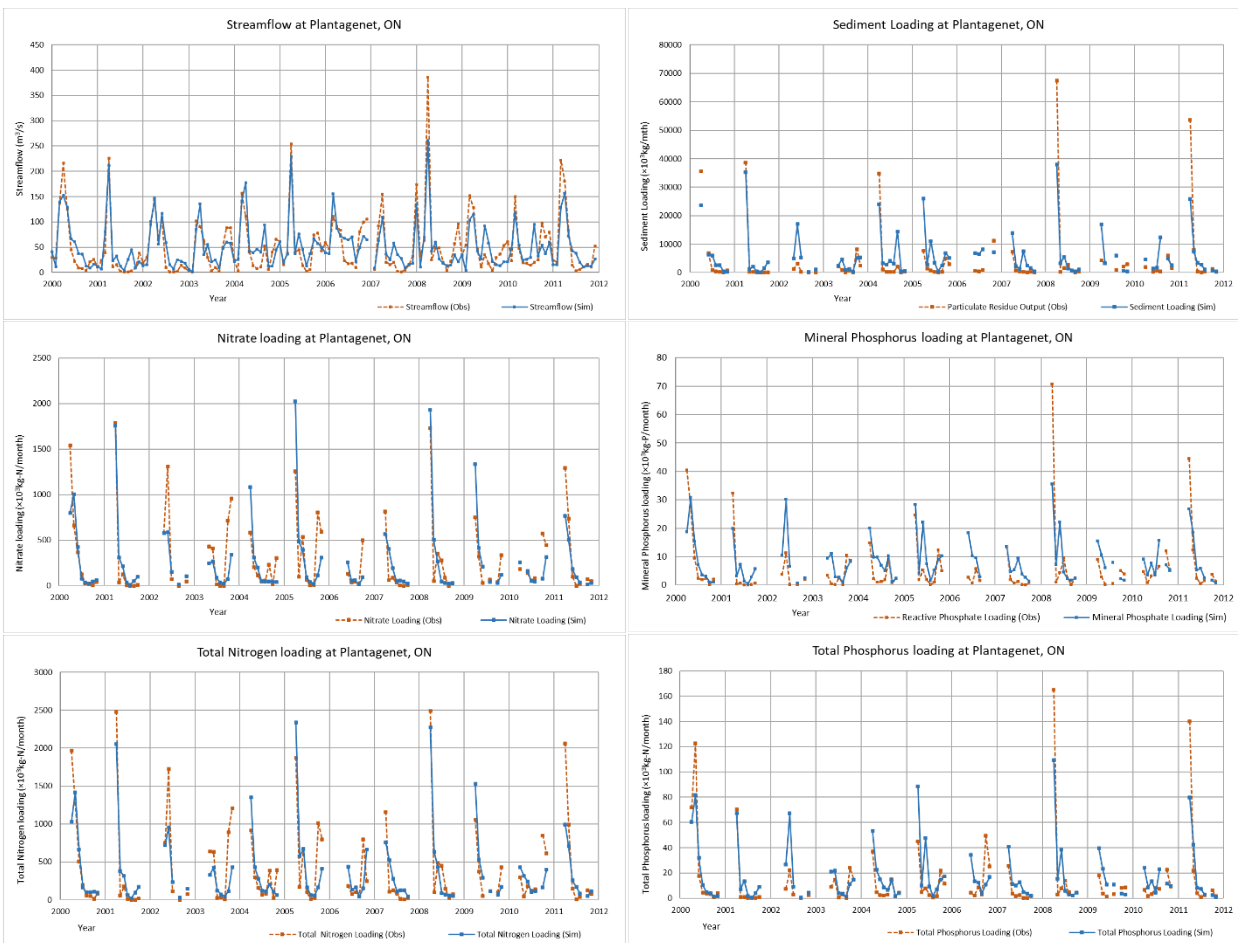

Figure 6 Comparison of monthly observed and simulated flow and quality parameters for the calibration period.

Table 6 and Figure 7 show equivalent profiles and statistics for the validation period 2012 to 2017. Similarly to the calibration period, the statistical performance ratings are very good for streamflow and satisfactory to very good for sediment and total nitrogen loadings while ratings for nitrate and phosphorus loadings are poorer, in this period generally unsatisfactory with phosphorus again showing significant overestimation bias. As with calibration, interpretation of validation is limited by the lack of winter-spring water quality sampling while the shorter period and the use of a different weather station for temperature and precipitation data for the validation period may contribute to the poor statistics. Nonetheless, since calibration was based on the same data sources as the planned simulations, those results were considered more relevant.

Table 6 Selected statistical measures of model validation from Moriasi et al (2007).

Variable	**NSE**	**PBIAS***	**RSR**
Streamflow	0.77 (very good)	3.0 (very good)	0.48 (very good)
Sediment loading	0.65 (satisfactory)	-44.3 (satisfactory)	0.60 (good)
Nitrate loading	0.27 (unsatisfactory)	3.8 (very good)	0.85 (unsatisfactory)
Mineral phosphorus loading	-0.54 (unsatisfactory)	-79.4 (unsatisfactory)	1.24 (unsatisfactory)
Total Nitrogen	0.52 (satisfactory)	-4.1 (very good)	0.69 (satisfactory)
Total Phosphorus	0.47 (unsatisfactory)	-42.5 (satisfactory)	0.73 (unsatisfactory)

* Moriasi et al. (2007) recommend different rating thresholds for each of streamflow, sediment loading and nutrient loadings.

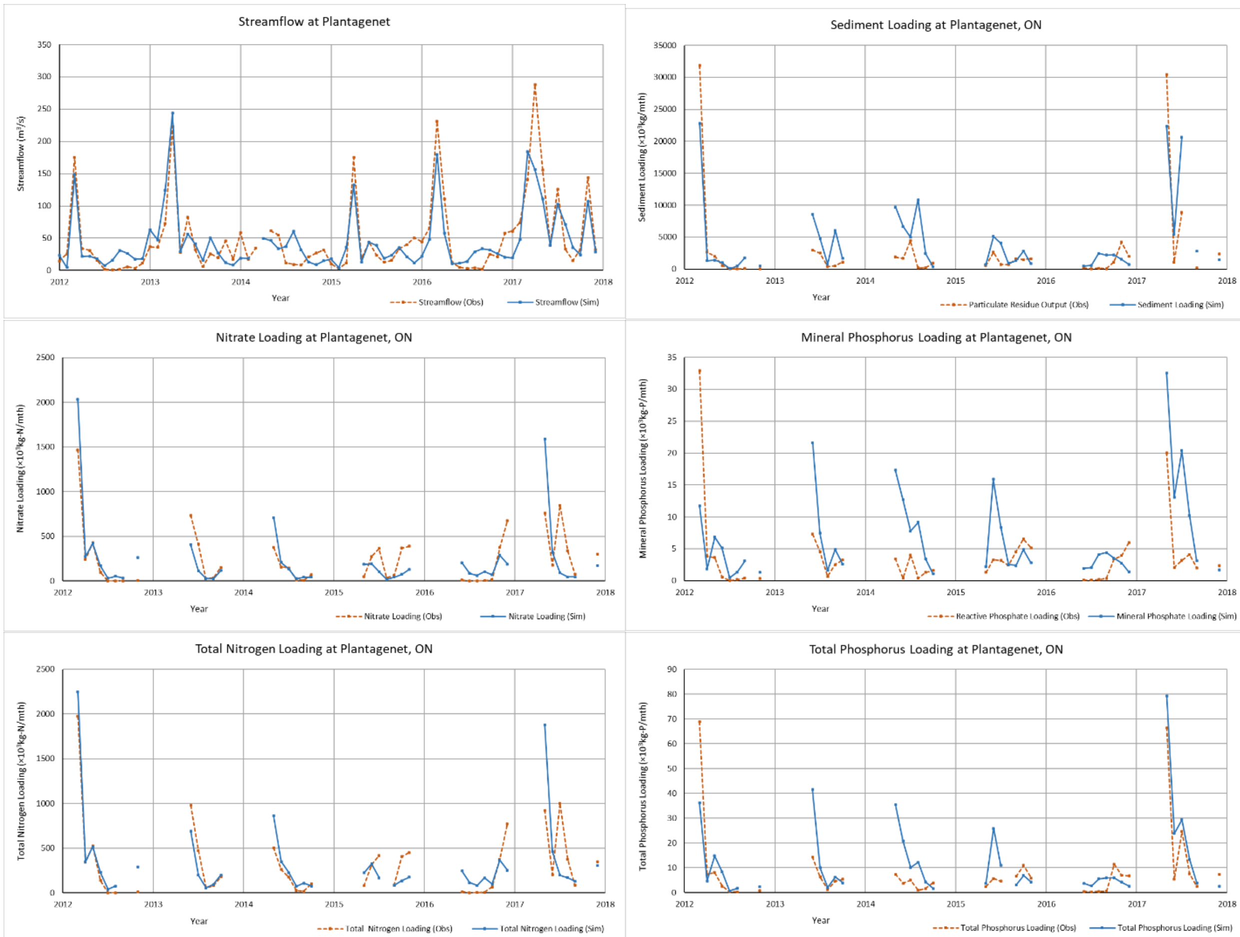

Figure 7 Comparison of monthly observed and simulated flow and quality parameters for the validation period.

5.2. Impacts of forest conversion to cropland

As described in section 4.1, SWAT first calculates flows of water, sediment and nutrients (and pesticides, not included in this study) to the main channel in each subbasin. Comparing flows and loadings at the subbasin scale is likely confounded by localized variables external to this study (e.g. soils, topography). These variables will be examined first at the HRU scale where the specific influence of distinct cover types can be separated, and secondly at the watershed scale. Contributions to the reach as well as flows and loadings leaving the basin at Plantagenet are examined at monthly and annual steps. It is important to note that without observation data for flows and loadings into each reach, values for these parameters should be considered as representative at best, particularly for the idealized, ubiquitous (to cropland) tile drainage simulation. However, since flow and loadings from the basin at Plantagenet have been calibrated and validated with river observations, these can be assessed with more confidence in the ranges and values produced.

5.2.1. Flows and loadings from HRUs

Before looking at water flows and sediment and nutrient loadings into and out of the reach, it is useful to consider the typical quality, quantity and route of water exported from HRUs. Since SWAT considers HRUs as discrete units with individually calculated flows and loadings, and this study varied land use profiles by adjusting HRU scales, the area-normalized results do not vary between the land use scenarios in this study. These results are considered as averages of annual flows and loadings by land use type since intra-annual weather conditions and landscape characteristics of individual HRUs vary significantly.

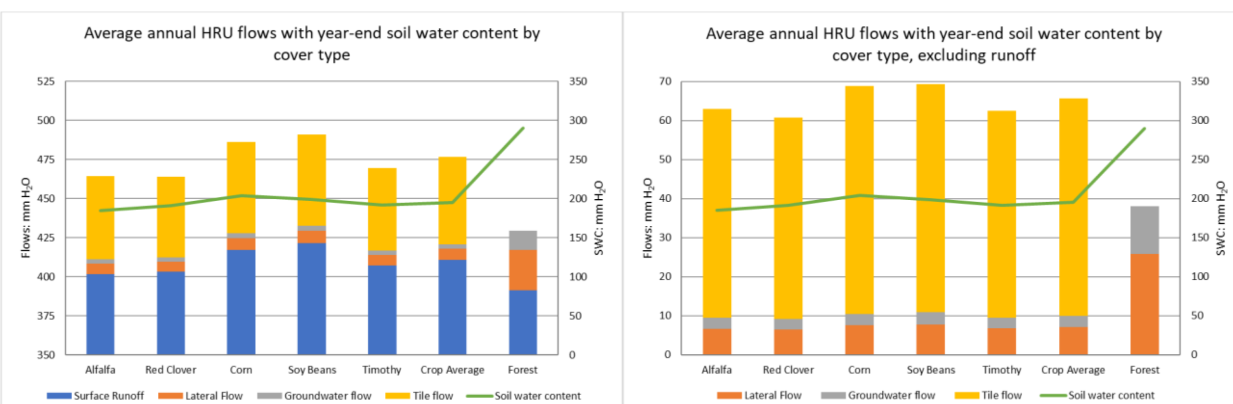

Figure 8 Average annual flows to reach by HRU and land use, partitioned as surface runoff and lateral, groundwater and tile flows (left), and excluding surface runoff (right) to facilitate comparison of subsurface flow components.

Figure 8 shows average annual flows to the reach by HRU for the six cover types included in this study as well as the average of all crops (since cropped areas rotated through all five crops). Surface runoff from forested HRUs is notably lower, while the absence of tile drainage allows for significantly higher retained soil water content, lateral flow and groundwater flow.

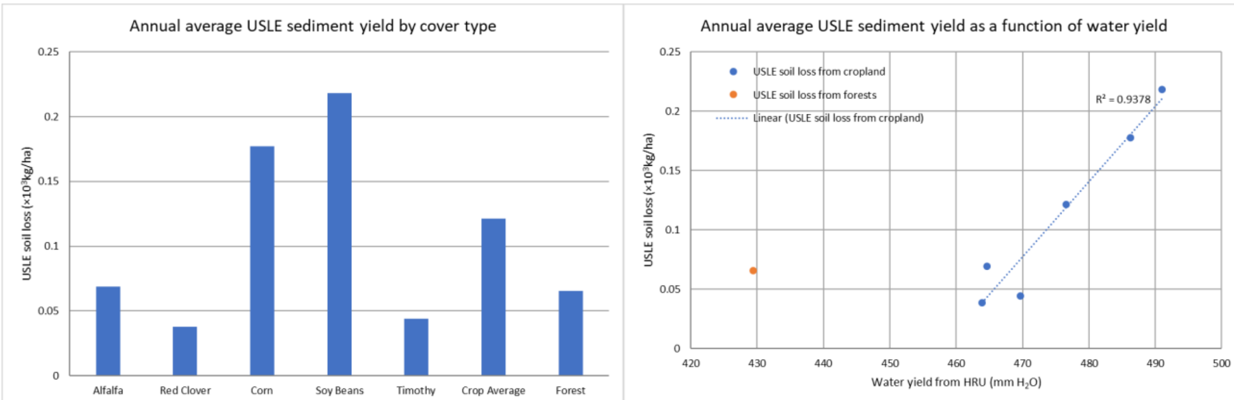

Figure 9 USLE sediment yield by cover type (left), and as a function of water yield from the HRU (right).

Across cropped HRUs, it can be seen in Figure 9 that sediment yield is closely related to the water yield, while the sediment yield from the forested areas is greater relative to water yield, likely primarily due to the non-zero MUSLE cover factor in the SWAT plant database. SWAT calculates the erosion due to rainfall and runoff using the Modified Universal Soil Loss Equation (MUSLE) (Williams 1975):

$$sed = 11.8 \cdot \left(Q_{surf} \cdot q_{peak} \cdot area_{hru}\right)^{0.56} \cdot K_{USLE} \cdot C_{USLE} \cdot P_{USLE} \cdot LS_{USLE} \cdot CFRG$$

where sed is sediment yield on a given day ($\times 10^3$ kg), Q_{surf} is surface runoff volume (mm H_2O/ha), q_{peak} is peak runoff rate (m^3/s), $area_{hru}$ is the area of the HRU in question, K_{USLE} is the USLE soil erodibility factor, C_{USLE} is the USLE cover and management factor, P_{USLE} is the USLE support practice factor, LS_{USLE} is the USLE topographic factor and $CFRG$ is the coarse fragment factor (Neitsch et al. 2011). Of these terms, most are calculated within SWAT, while only K_{USLE} and P_{USLE} were calibrated in this study. The minimum value of C_{USLE}, $C_{USLE,mn}$ is defined in the SWAT plant database. In some studies, this value is also adjusted (e.g. Abbaspour et al. 2007b), however for this study, this parameter was kept unchanged in the SWAT plant database. In the standard SWAT plant database, $C_{USLE,mn} = 0$ for the crop types studied here, while $C_{USLE,mn} = 50$ for mixed forest.

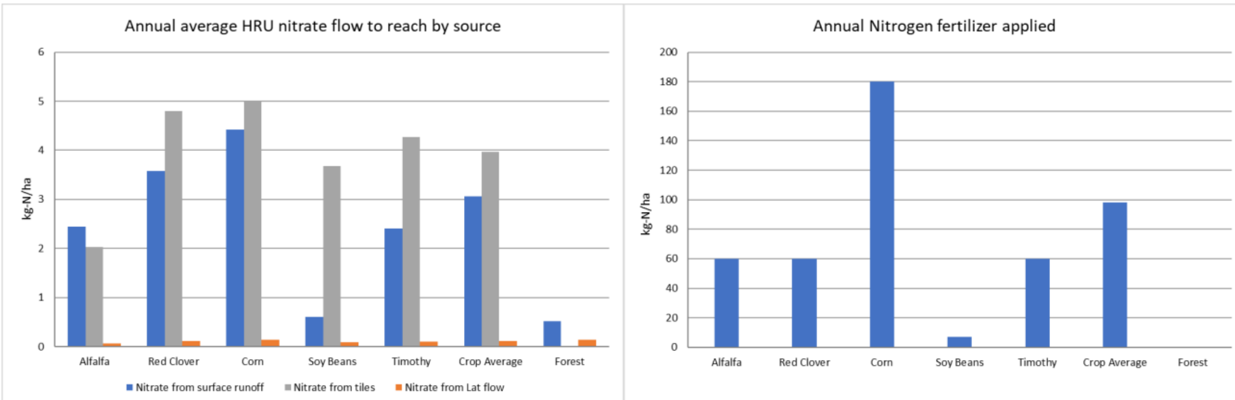

Figure 10 Nitrate exports to the South Nation River for each cover type simulated, as transported with surface runoff, lateral flow and tile drainage flow (left), and Nitrogen fertilizer applied to each crop as elemental Nitrogen (right). Nitrate transported in groundwater is negligible.

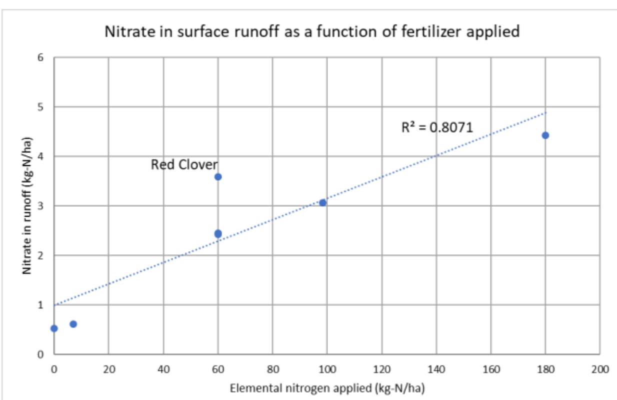

Figure 11 Annual nitrate exported in surface runoff as a function of the annual total elemental nitrogen fertilizer applied to each cover type.

A few key points are evident from figures 10 and 11, illustrating nitrate exports to the South Nation River:

- Nitrate exports are largely proportional to the application of nitrogen to each cover type, with the exception of red clover,

- Nitrate flushing by lateral and groundwater flow is generally insignificant, even in the absence of tile drainage (in forested areas) and
- Nitrate transported by tile drainage is unrelated to fertilizer application rates and may represent leaching from soils.

Although SWAT does not generate an equivalent breakdown of the phosphorus routing, P is partitioned into organic P, soluble P and P sorbed to sediment. The annual breakdown of these species, by cover type, is shown in Figure 12. Since the rate of phosphorus fertilizer application in the simulation was largely consistent between the crop types (30 kg-P/ha/yr for corn, 20 kg-P/ha/yr for other crops), most of the variation can be attributed to variations in the growth properties of the different plant types. Notably, however, the unfertilized forest HRUs generate significantly less soluble and sediment-bound phosphorus while releasing more organic P than the cropped areas.

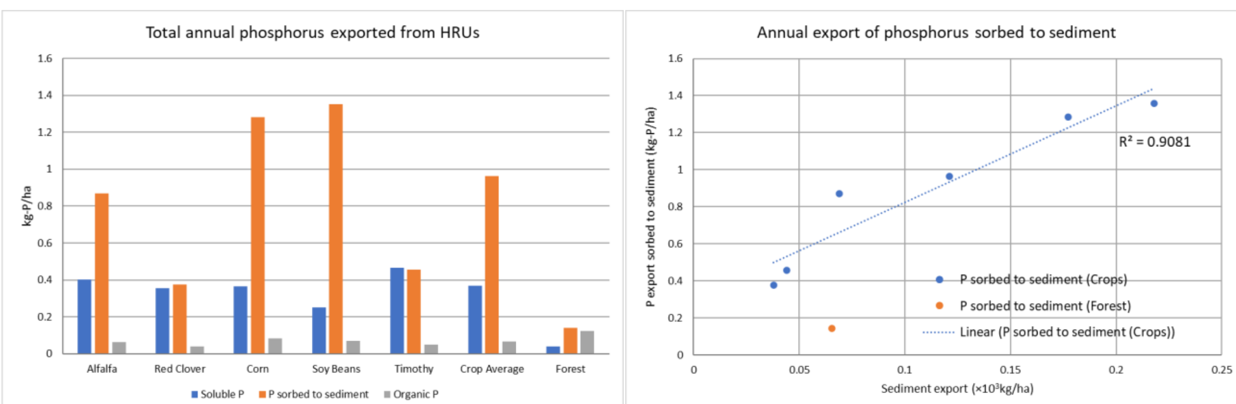

Figure 12 Breakdown of annual phosphorus exports from HRUs to the South Nation River by phosphorus form and HRU cover type.

To summarize these observations in the context of this study of changing the balance of forest and cropland:

- Forests increase soil water content and lateral and groundwater flow while eliminating tile flow and slightly reducing surface runoff and water yield.
- Forested HRUs export similar sediment loads per unit area, but more per unit of water yield due to higher $C_{USLE,mn}$.
- Nitrate in runoff is approximately proportional to fertilizer application, hence is significantly reduced in forests while nitrate in tile flow is largely unrelated to fertilizer and non-existent in forests. Loading in lateral and groundwater flows is negligible.
- Forests reduce soluble and sediment-bound phosphorus while releasing more organic phosphorus.

SWAT sums flows and loadings from HRUs for each subbasin to establish overall flows and loadings into and through the reach. It can hence be expected that decreasing forest cover will result in decreased soil water and subsurface flow as well as increased runoff, water yield and tile flow. Overall, forests export less sediment than the crop average, so decreasing forest cover should increase sediment loadings, although the cover crops release comparable or lower sediment than forest. Decreasing forest should increase nitrate and phosphorus exports due to an increase in overall fertilizer application and increased nutrient leaching to tile drainage.

5.2.2. Flows and loadings to reach, monthly averages

Monthly averages of the runoff and groundwater, lateral and tile flows (Figure 13) show, at a glance, that runoff and lateral and tile flow peak with the spring snowmelt. In contrast, groundwater flow varies slowly between a February minimum and July maximum. Additionally, runoff shows the largest intra-annual variation and is by far the most significant contributor to overall water yield to the reach. (Water yield in SWAT is the sum of surface runoff, groundwater flow, lateral flow and tile drain flow.) Figure 14 shows the related loadings of sediment and nutrients transported with surface runoff and tile flow (nitrate carried by lateral flow is insignificant compared to runoff and tile flow loadings, while nitrate in groundwater flow is zero). As with the flows, sediment, total phosphorus and tile flow nitrate loadings peak with the spring snowmelt and are largely stable through the remainder of the year. Nitrate in the runoff, however, peaks in May-June due to the application of fertilizer to crops in those months. Further, it is immediately apparent that the influence of changing land use scenarios on runoff is minor, while the other flows are much more sensitive, at least relatively. Varying erosion and interception of contaminants lead to more significant variations in the sediment and nutrient loadings. These simulations suggest that further reduction to 20% forest cover across the watershed is likely to increase the maximum monthly loadings of nitrate and total phosphorus by over 10%, and sediment by almost 7%. Since runoff makes up the majority of the water yield, it makes sense that the peak coincides with the peak snowmelt and soil water content in March, as seen in Figure 15. Early winter flushes of tile flow and nitrates in tile and surface flow coincide with early winter snowmelt after the harvest of crops. Table 7 shows the peak monthly and annual figures for selected land use scenarios and percentage variations compared to the 2008 calibration land use scenario. Here, too we can see very little variation in the runoff, even at peak flow, while the others all vary significantly, with loadings increasing with decreasing forest cover. At least part of the increase in tile flow and nitrate in tile flow is likely due to the increase in the area that is fertilized and drained by tiles as forest cover is converted to tile-drained agriculture in this study.

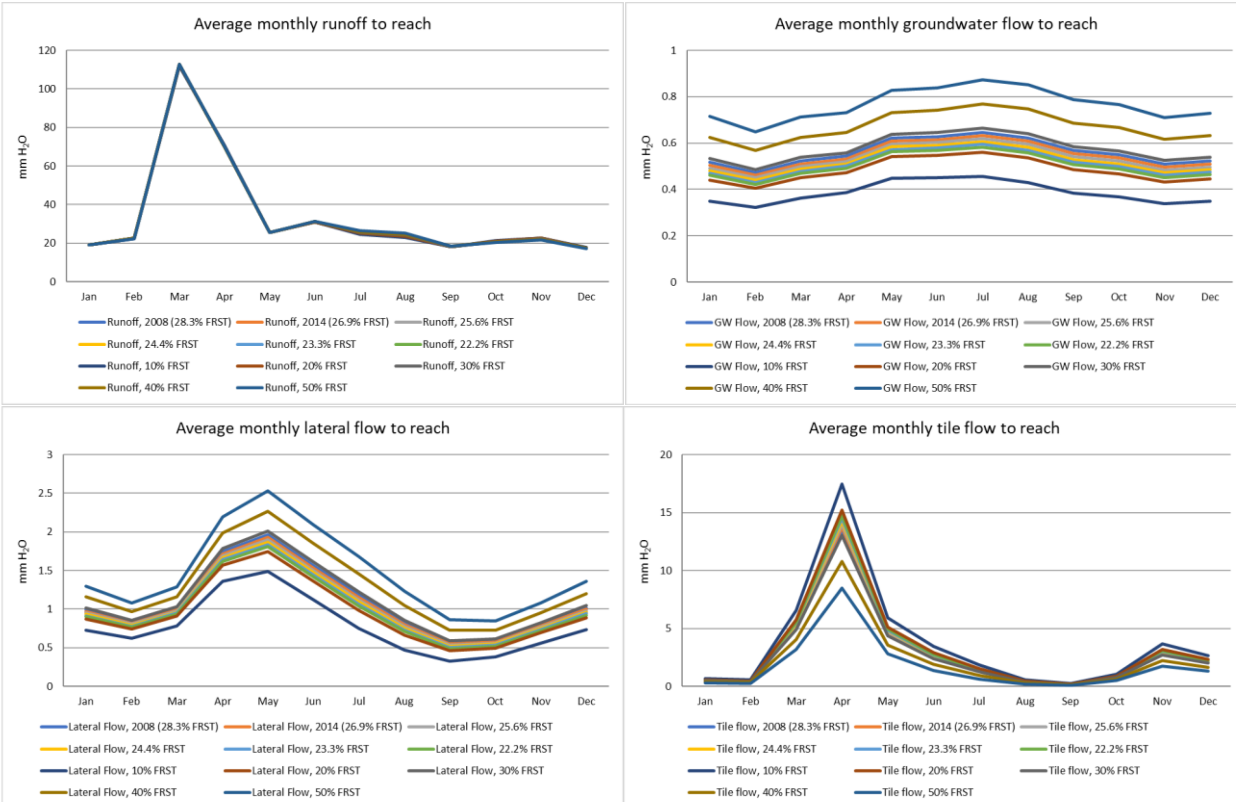

Figure 13 Watershed average monthly runoff and groundwater, lateral and tile flows into the South Nation River.

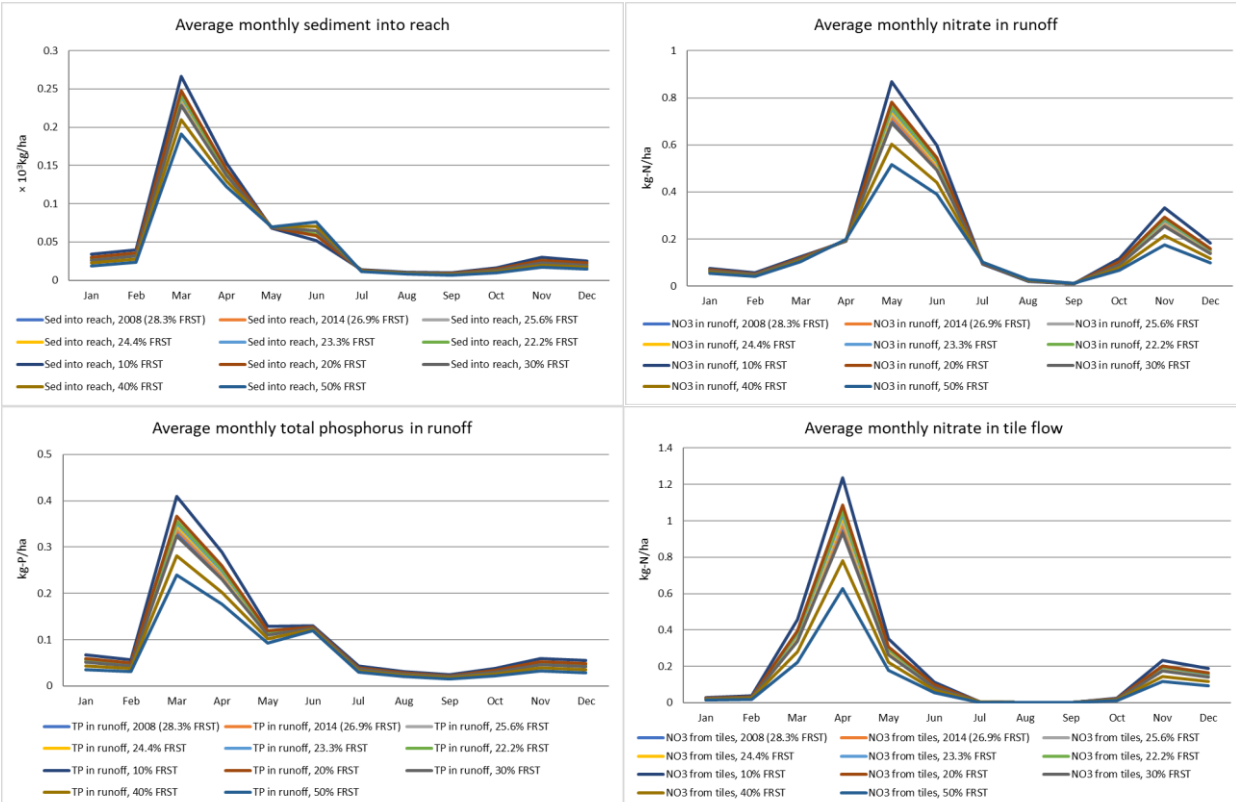

Figure 14 Watershed average monthly loadings of sediment and nutrients to the South Nation River.

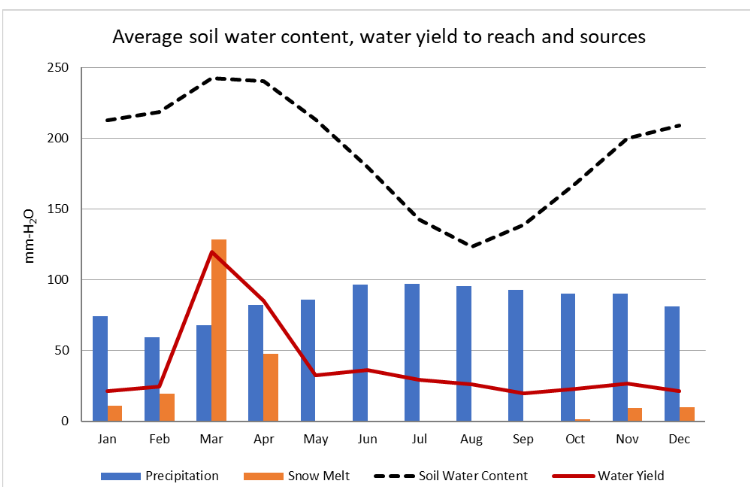

Figure 15 Comparison of average simulated soil water content, runoff, snowmelt and tile flow (2008 land use scenario) with observed precipitation.

5.2.3. Flows and loadings to reach, annual averages

It can be seen again, in figures and tables of the average annual flows and loadings into the South Nation River (Figures 16 & 17, Table 7), that groundwater, lateral and tile flows are relatively far more sensitive to land use change than surface runoff. This could be expected from the observations in section 5.2.1 that runoff varies relatively little between crops and forest, while the other components are significantly different. With this model configuration, the decrease in tile flow with increasing forest cover overrides the increasing trend from the other three components, leading to a decrease in water yield to the reach with increasing forest cover (Figure 18). In mm, the variation in runoff, groundwater flow and lateral flow due to deforestation alone is negligible; the entire range of scenarios produces a total range of variation of up to 1.0 mm in these variables, while the greater variation in tile flow, with an opposing trend, leads to slightly more significant variation in total water yield.

Variations in the runoff loadings, however, are quite significant as could be expected from some of the observations in section 5.2.1 – reducing forest cover from the 2008 starting condition to 20% is likely to increase annual contributions of sediment, nitrate and total phosphorus from runoff by 5-10%. Total tile flow and nitrate from tiles again appear to be very sensitive to the land use conversion since tile flow and nitrate therein is significant from cropland, but absent from forests. This sensitivity should be noted, however, when considering future land use planning.

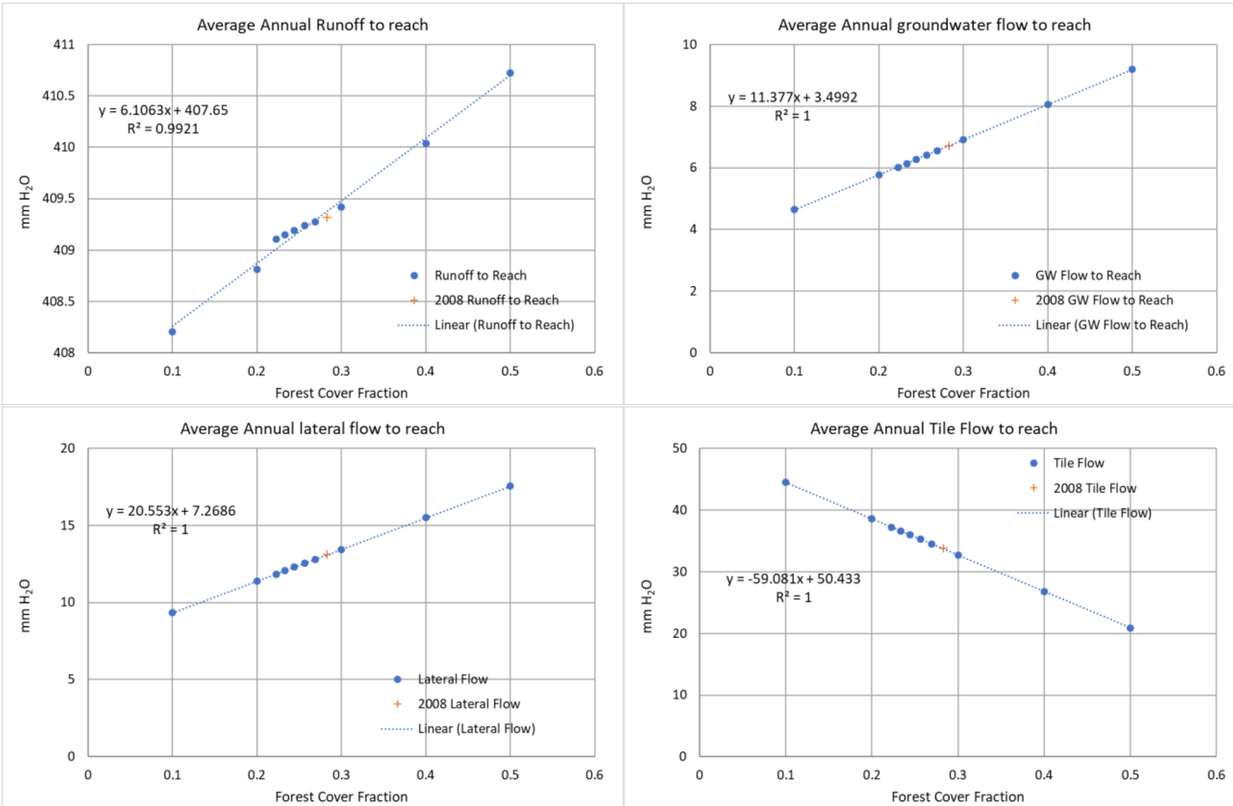

Figure 16 Watershed average annual runoff and groundwater, lateral, and tile flows into the South Nation River.

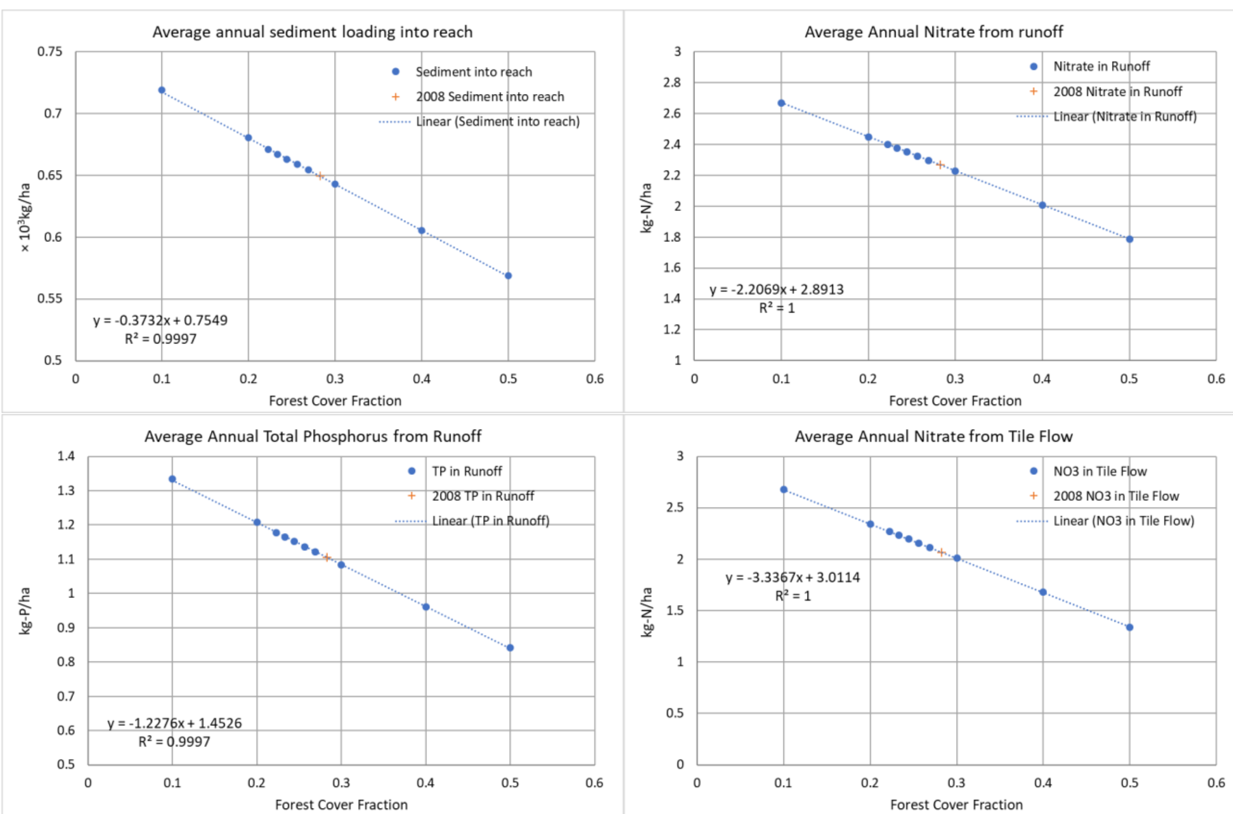

Figure 17 Watershed average annual loadings of sediment and nutrients to the South Nation River.

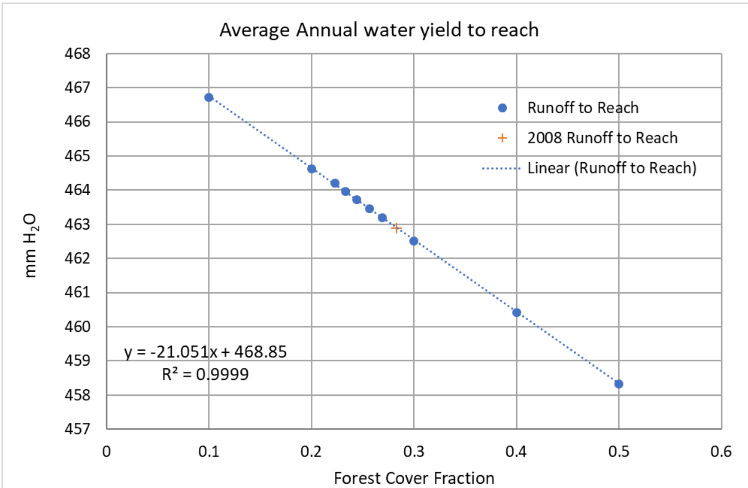

Figure 18 Watershed average annual water yield into the South Nation River

Table 7 Summary of runoff and other flows, and loadings of sediment and nutrients to the South Nation River and changes due to 20%, 10% and 50% forest cover (compared to 2008 baseline)

Variable		**2008 DRAPE (28.3% forest)**	**10% forest**	**20% forest**	**50% forest**
Runoff (mm)	Maximum monthly Ave	112.6	111.9	112.4	112.7
			-0.6%	-0.2%	0.1%
	Average Annual	409.3	408.2	408.8	410.7
			-0.3%	-0.1%	0.3%
GW Flow (mm)	Maximum monthly Ave	0.647	0.457	0.561	0.873
			-29.3%	-13.3%	35.0%
	Average Annual Total	6.72	4.65	5.78	9.19
			-30.9%	-14.0%	36.8%
Lateral Flow (mm)	Maximum monthly Ave	1.96	1.49	1.75	2.53
			-24.3%	-11.0%	28.7%
	Average Annual Total	13.1	9.32	11.4	17.5
			-28.8%	-13.0%	34.0%

Tile Flow (mm)	Maximum monthly Ave	13.4	17.5	15.2	8.49
			30.5%	13.8%	-36.5%
	Average Annual Total	33.8	44.5	38.6	20.9
			32.0%	14.5%	-38.1%
Water Yield (mm)	Maximum monthly Ave	119.2	119.7	119.5	117.9
			0.4%	0.2%	-1.1%
	Average Annual Total	462.9	466.7	464.6	458.3
			0.8%	0.4%	-1.0%
Sediment ($\times 10^3$ kg/ha)	Maximum monthly Ave	0.232	0.267	0.248	0.191
			15.1%	6.8%	-17.5%
	Average Annual Total	0.654	0.719	0.681	0.569
			10.7%	4.8%	-12.4%
Nitrate (kg-N/ha)	Maximum monthly Ave	0.708	0.869	0.781	0.516
			22.9%	10.4%	-27.0%
	Average Annual Total	2.27	2.67	2.45	1.79
			17.8%	8.1%	-21.1%
Total P (kg-P/ha)	Maximum monthly Ave	0.331	0.410	0.367	0.240
			23.7%	10.7%	-27.5%
	Average Annual Total	1.10	1.33	1.21	0.84
			20.7%	9.4%	-23.8%

Tile Nitrate (kg-N/ha)	Maximum monthly Ave	0.960	1.24	1.09	0.628
			28.9%	13.1%	-34.6%
	Average Annual Total	2.07	2.68	2.34	1.34
			29.4%	13.3%	-35.2%

5.2.4. Flows and loadings at basin outlet, monthly averages

Figure 19 shows that, similarly to runoff, streamflow at the basin outlet is mostly insensitive to forest cover variations in this study. Also, in keeping with the previous observations, nutrient loadings are significantly impacted by forest cover, particularly during the spring flood. Of particular interest is the sediment loading. Despite our previous finding that forest cover has a significant influence on the sediment carried to the reach with runoff, sediment loading in the river at Plantagenet shows relatively small variations between land use scenarios (deforestation to 20% cover increases sediment in runoff by ~7%, and sediment at Plantagenet by less than 1%). This suggests that the sediment loading leaving the basin is primarily controlled by reach processes (scour and deposition).

Although part of the spring surge of sediment and nutrients leaving the basin is related to the significantly higher streamflow in March and April, the plots of average nutrient concentrations in Figure 20 indicate that the highest sediment concentration occurs in March-April and the highest nutrient concentrations in May-June. Although the sediment flush coincides with the spring freshet, the nutrient flush slightly lags the spring flood, which likely reflects the application of fertilizer to cropped HRUs between May 5 and June 1. As with the runoff loadings, there is a second, smaller peak in nitrate and total nitrogen in November-December that coincides with early winter snowmelt, perhaps leading to flushing of soil nutrients after crop harvest. Figure 15 also indicates that soil water content and tile flow increase after the end of the growing season.

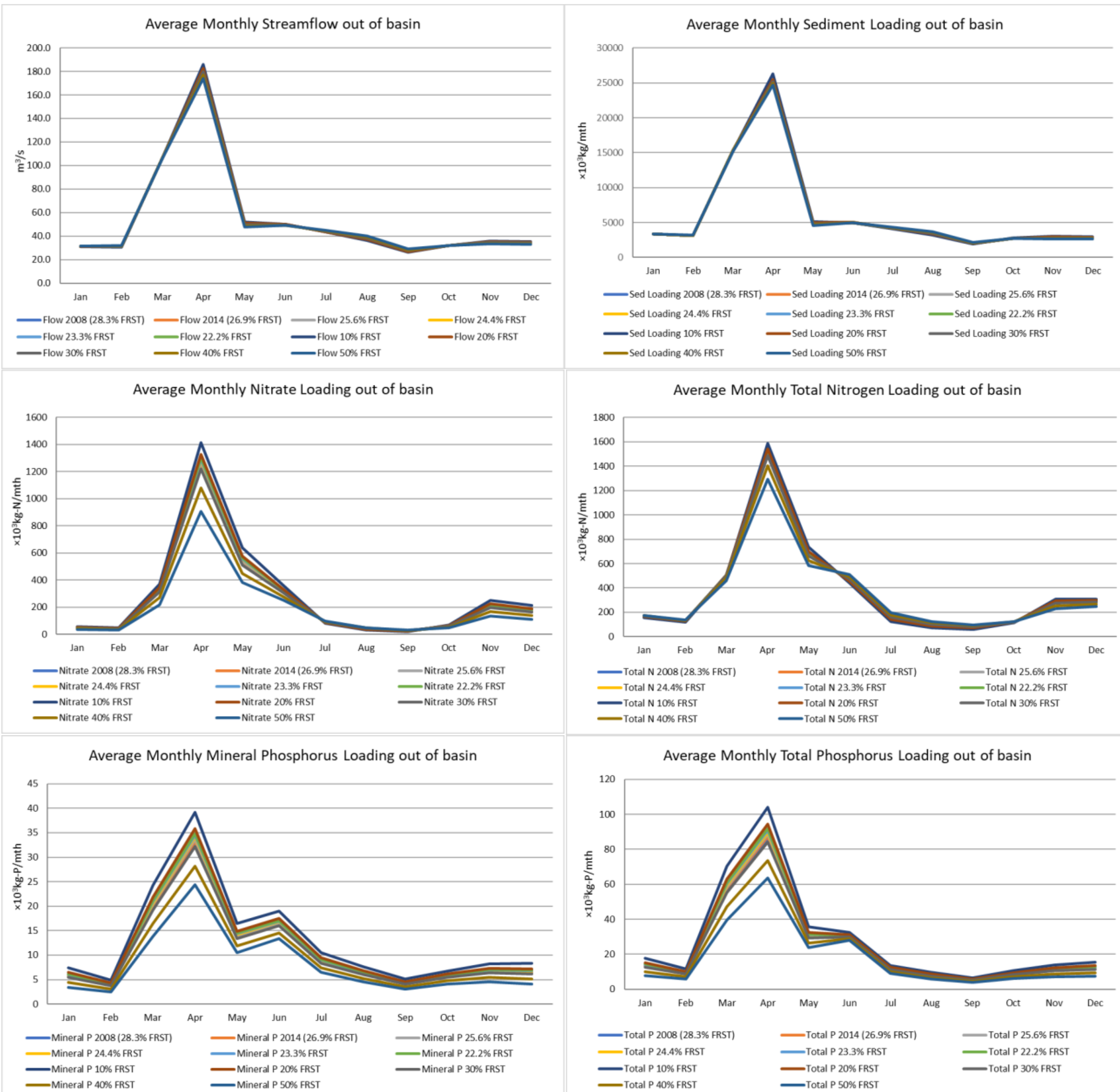

Figure 19 Monthly average streamflow and sediment and key nutrient loadings at the outlet of the South Nation watershed.

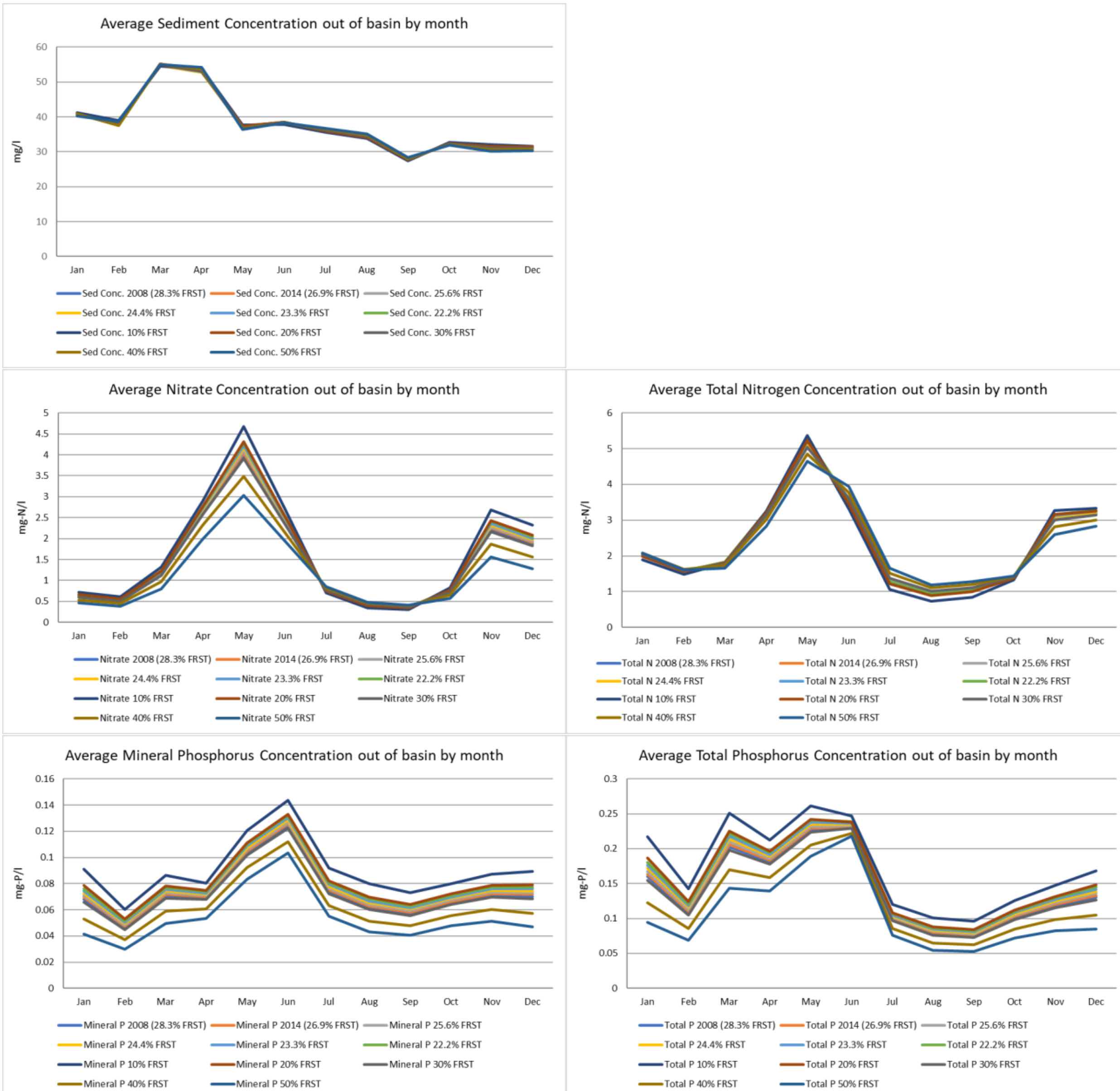

Figure 20 Monthly average simulated concentrations of sediment and key nutrient species at the outlet of the South Nation watershed.

5.2.5. Flows and loadings at basin outlet, annual averages

The annual average loadings at the basin outlet, shown in Figure 21, similarly show only small variations in streamflow and sediment across the simulated scenarios, while there is a more significant change in nutrient loadings. Total nitrogen loading shows a slightly different trend from the other nutrients, with progressively more deforested scenarios showing less dramatic increases in nitrogen loading than scenarios with greater forest cover. This appears to be related to nitrite-N and ammonia-N, which were not calibrated but show a slight *decrease* with diminishing forest cover.

Flow and loading variables under the 2008 land use scenario and the 10%, 20% and 50% forest cover scenarios are compared in Table 8. In summary, flows and loadings are inversely related to forest cover, although nutrient loadings are much more sensitive than streamflow or sediment loading. Changes related to recent forest loss (and near-term projected scenarios) are relatively small. More extreme, but not improbable, scenarios such as reducing to 20% or even 10% cover can be expected to trigger significant increases in nutrient outputs, with associated risks to river ecosystems.

Although variations in streamflow between scenarios are quite small, part of the increase in loadings with increasing deforestation can be assumed to be related to increased flow in the same cases. Figure 22 shows average annual concentrations for sediment and key nutrients at the basin outlet, and Table 9 lists results for the same parameters as Table 8 (excluding streamflow), again for the 2008 and 10%, 20% and 50% forest cover scenarios. It can be seen in Figure 22 that there is no coherent variation in sediment concentration. The concentrations of all four nutrient species increase with decreasing forest cover in very similar character to the loading relationships. Additionally, Table 9 reinforces this observation – variation in sediment between the extreme 10% and 50% scenarios is less than 1%, while the various forest scenarios have a very large influence on nutrient concentrations.

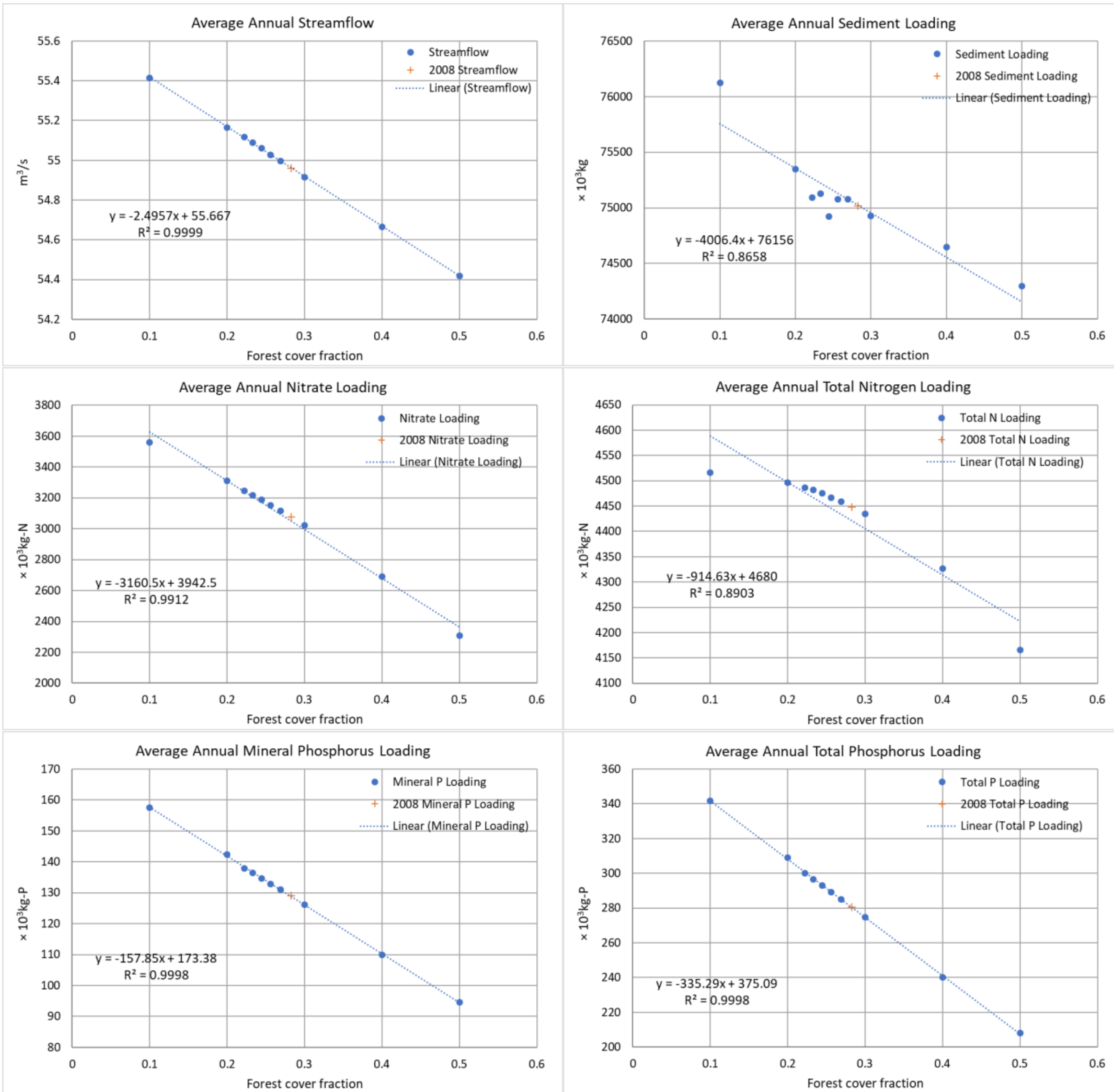

Figure 21 Annual average streamflow and sediment and key nutrient loadings.

Table 8 Summary of streamflow and sediment and nutrient loadings and changes at basin outlet due to 20%, 10% and 50% forest cover (compared to 2008 baseline)

Variable		**2008 DRAPE (28.3% forest)**	**10% forest**	**20% forest**	**50% forest**
Flow (m^3/s)	Maximum monthly Ave	180.5	185.9	182.8	174.0
			3.0%	1.3%	-3.6%
	Average Annual	55.0	55.5	55.2	54.5
			0.8%	0.4%	-1.0%
Sediment ($\times 10^3$ kg)	Maximum monthly Ave	25242	26284	25478	24720
			4.1%	0.9%	-2.1%
	Average Annual Total	75018	76125	75350	74293
			1.5%	0.4%	-1.0%
Nitrate ($\times 10^3$ kg-N)	Maximum monthly Ave	1239	1415	1327	908
			14.2%	7.1%	-26.7%
	Average Annual Total	3075	3559	3311	2310
			15.8%	7.7%	-24.9%
Total N ($\times 10^3$ kg-N)	Maximum monthly Ave	1497	1586	1544	1292
			6.0%	3.1%	-13.7%
	Average Annual Total	4448	4516	4496	4166
			1.5%	1.1%	-6.3%
Mineral P (kg-P)	Maximum monthly Ave	32833	39203	35864	24341
			19.4%	9.2%	-25.9%
	Average Annual Total	129000	157552	142324	94619
			22.1%	10.3%	-26.7%

Total P (kg-P)	Maximum monthly Ave	86057	103897	94479	63567
			20.7%	9.8%	-26.1%
	Average Annual Total	280548	341742	309096	208114
			21.8%	10.2%	-25.8%

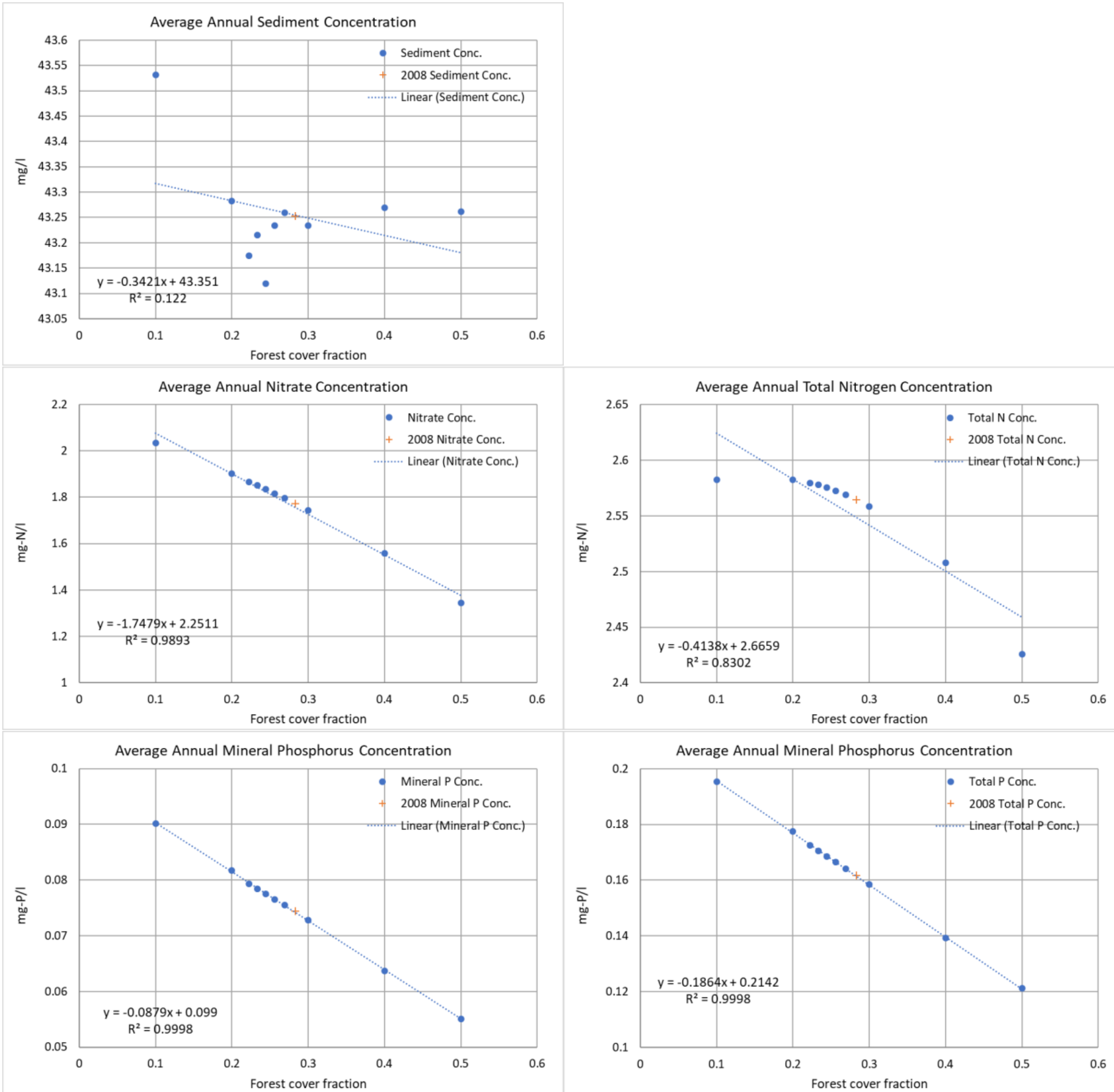

Figure 22 Annual average sediment and key nutrient concentrations

Table 9 Summary of sediment and nutrient concentrations and changes at basin outlet due to 20%, 10% and 50% forest cover (compared to 2008 baseline)

Variable		**2008 DRAPE (28.3% forest)**	**10% forest**	**20% forest**	**50% forest**
Sediment (mg/L)	Maximum monthly Ave	55.01	54.59	54.94	54.70
			-0.7%	-0.1%	-0.6%
	Average Annual	37.95	38.10	37.99	37.91
			0.4%	0.1%	-0.1%
Nitrate (mg-N/L)	Maximum monthly Ave	3.98	4.67	4.31	3.02
			17.3%	8.3%	-24.1%
	Average Annual	1.47	1.68	1.57	1.14
			14.2%	6.9%	-22.1%
Total N (mg-N/L)	Maximum monthly Ave	5.08	5.37	5.22	4.64
			5.8%	2.9%	-8.6%
	Average Annual	2.36	2.31	2.34	2.31
			-2.0%	-0.5%	-2.0%
Mineral P (mg-P/L)	Maximum monthly Ave	0.124	0.144	0.133	0.103
			15.7%	7.2%	-16.8%
	Average Annual	0.073	0.090	0.081	0.054
			23.0%	10.6%	-26.9%
Total P (mg-P/L)	Maximum monthly Ave	0.231	0.261	0.242	0.218
			13.1%	5.1%	-5.7%
	Average Annual	0.143	0.174	0.157	0.106
			22.1%	10.3%	-25.7%

5.3. Simple and multiparameter regression including precipitation

Annual precipitation totals also affect basin outputs. In this section, the relationship between forest cover, annual precipitation and annual basin outputs is quantified, initially through simple regression, similar to the regressions of forest cover and flow/loading already shown, then through multivariate regression with precipitation and forest cover as predictors to examine the two expected drivers together as potential predictors of stream and nutrient flows.

Examples of the simple regression for the 2008 land use scenario are shown in Figure 23. Although there is a general pattern across these parameters (and most others simulated) of slightly increased flow or loading with higher precipitation, the pressure from this input is not as significant as that from the simulated changes in forest cover. Nonetheless, the influence of changing precipitation needs to be considered in land use and climate planning, particularly where pressures from precipitation and forest change complement, and in the context of forecast increasing precipitation in this region this century (Bush and Lemmen 2019).

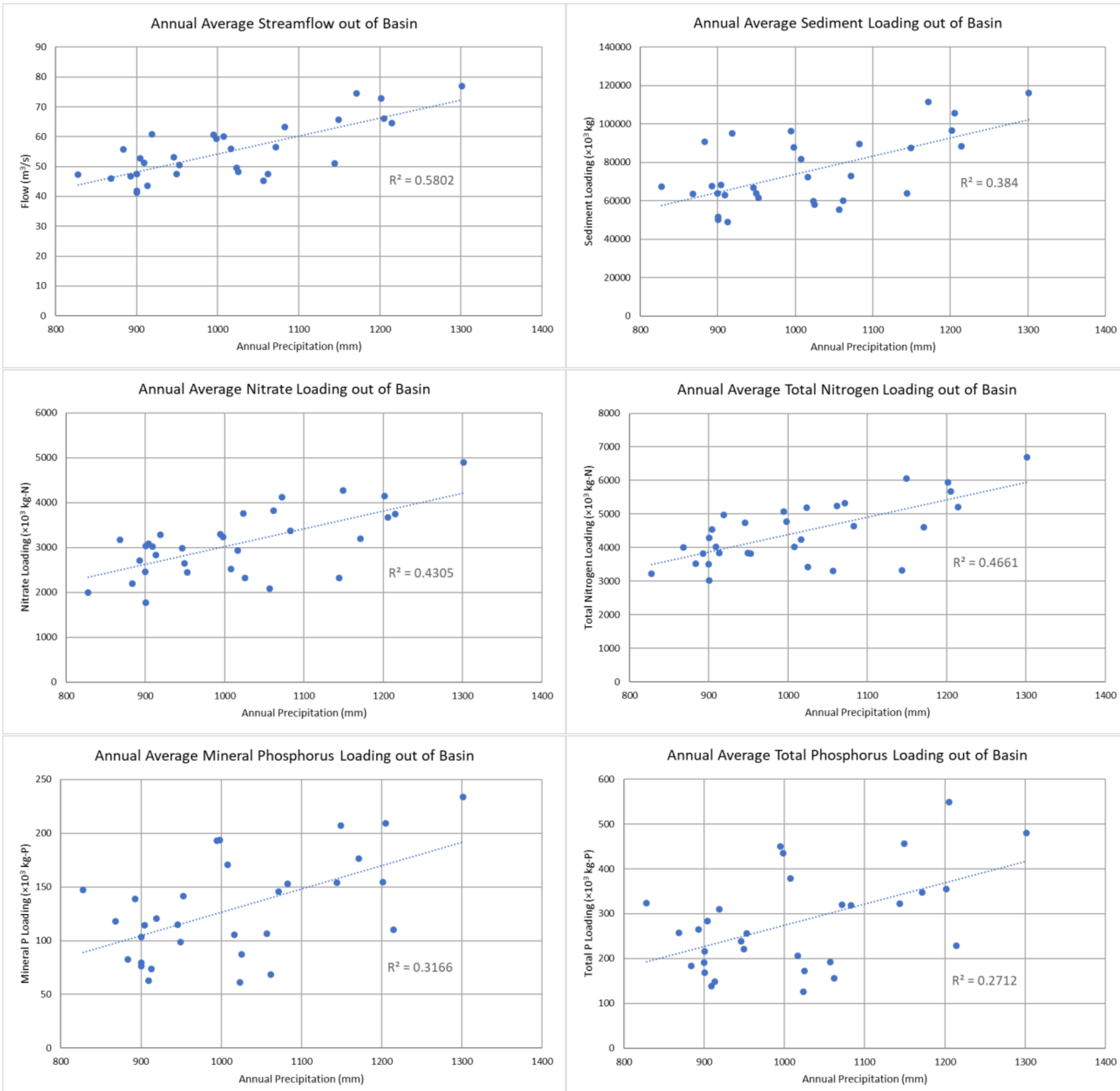

Figure 23 Simple regression of streamflow and loadings with annual precipitation

Further to these simple regressions, forest cover and precipitation were considered together with a stepwise regression approach using the Matlab *stepwisefit* function. Stepwise regression is an automated process where the selection of predictive variables for a regression model is based on a prespecified criterion such as F-tests, t-tests or p-values. The *stepwisefit* function in Matlab uses a p-value limit of 0.05. Figure 24 shows scatter plots of simulated flows and contributions to the reach with the results predicted by the multivariate regressions and the regression equation for each parameter, and Figure 25 shows the same for streamflow and nutrient loadings out of the basin. Qualitatively, these results appear to back up the observations from simple regression with forest cover and precipitation for most parameters: a clear negative relationship with forest cover, and a more scattered positive relationship with precipitation. As noted previously runoff, streamflow and sediment loading (at the exit) are not very sensitive to forest cover changes; indeed, the *stepwisefit* function did not include forest cover for streamflow or sediment loading due to high *p* values.

While calculating multivariate regression results, estimates of each predictor's contribution to variance reduction were determined for each variable, shown in Table 10, based on the following equation:

$$percent\ contribution = \frac{Var\big(X \cdot diag(B)\big)}{Var(Y)} \cdot 100$$

Where *X* is the matrix of predictors (in this study: forest cover, precipitation), *B* is a vector of regression coefficients for each variable and *Y* is the response vector.

It can be seen from these figures that generally precipitation is the dominant predictor of runoff, water yield, streamflow and sediment loading at the basin exit while forest has some significance in sediment yield into the reach. Forest cover has a more significant role in predicting tile flow (since tiles and forest do not coexist in any HRUs) and nutrient contributions and loadings, particularly nitrate since nitrogen fertilizer application is significant in most years of the crop cycle. However, even taken together, these predictors account for at most around 60% of the explained variance, suggesting that other drivers (e.g. topography, soil types) also play a significant role or that other variations in these predictors such as the distribution of forest and/or the intra-annual timing of precipitation are also significant.

Table 10 Percent contributions of Forest Cover and Precipitation to variance reduction in stepwise regressions

	Percent variance reduction		
Parameter	**Forest Cover**	**Precipitation**	**Total**
Surface runoff	N/A*	55.0	55.0
Tile flow	12.6	13.2	25.8
Water yield	N/A*	57.3	57.3
Sediment yield	4.0	23.6	27.5
Nitrate in runoff	12.8	17.2	30.0
Nitrate in tile flow	10.8	24.0	34.8
Total P contribution to reach	7.3	24.3	31.6
Streamflow	N/A*	58.0	58.0
Sediment loading	N/A*	39.8	39.8
Nitrate loading	15.5	35.8	51.3
Mineral Phosphorus loading	9.9	27.8	37.7
Total Nitrogen loading	1.0	45.6	46.6
Total Phosphorus loading	8.2	24.4	32.6

* Matlab did not include forest cover in the model for these parameters since $p > 0.05$.

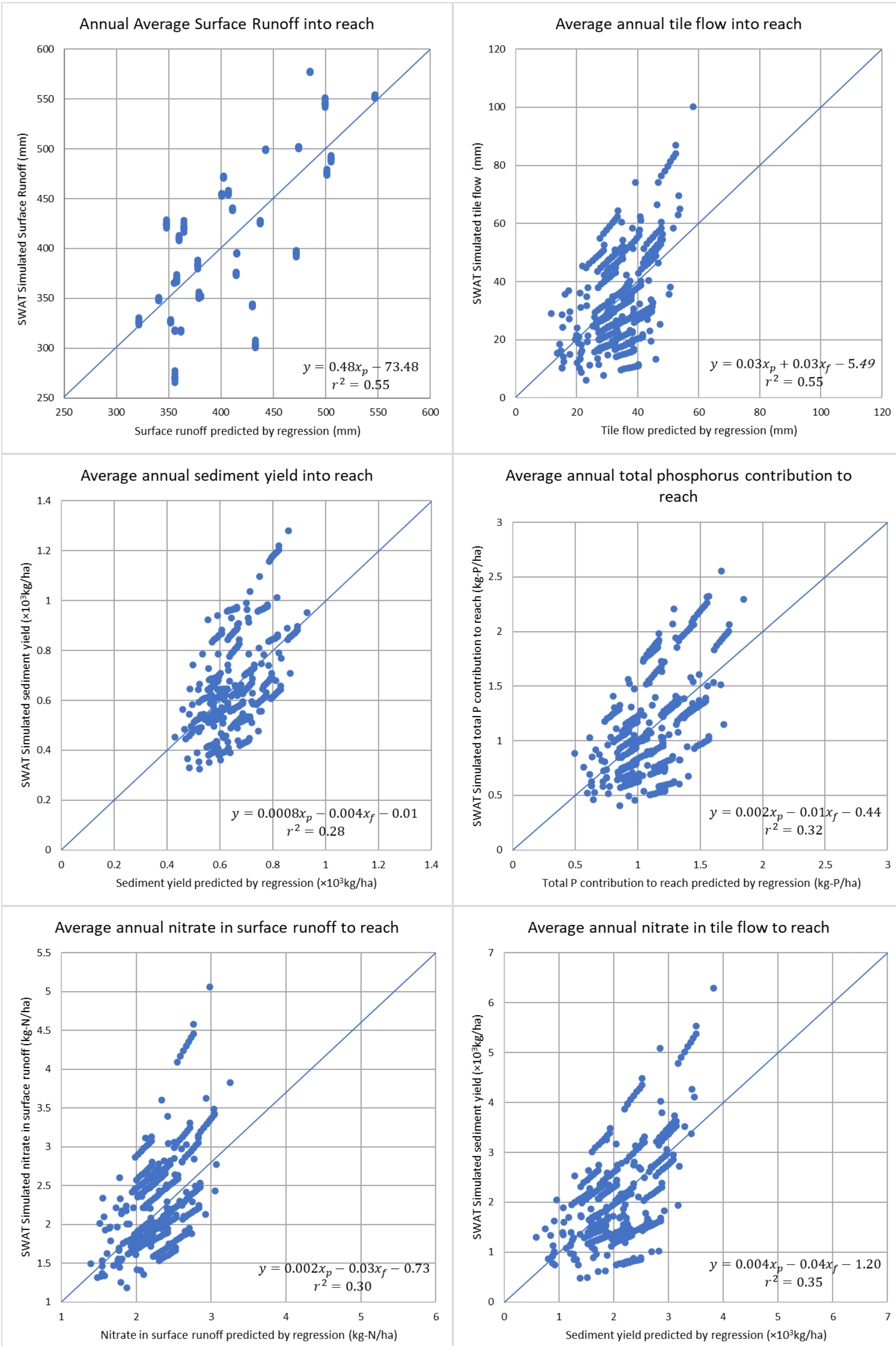

Figure 24 Scatter plots of SWAT-simulated and multivariate regression predicted flows and contributions from subbasins to the reach. In the regression equations, x_f is forest cover (percent) and x_p is precipitation (mm/y). Clusters of points relate to a range of LU scenarios for a given year and precipitation total

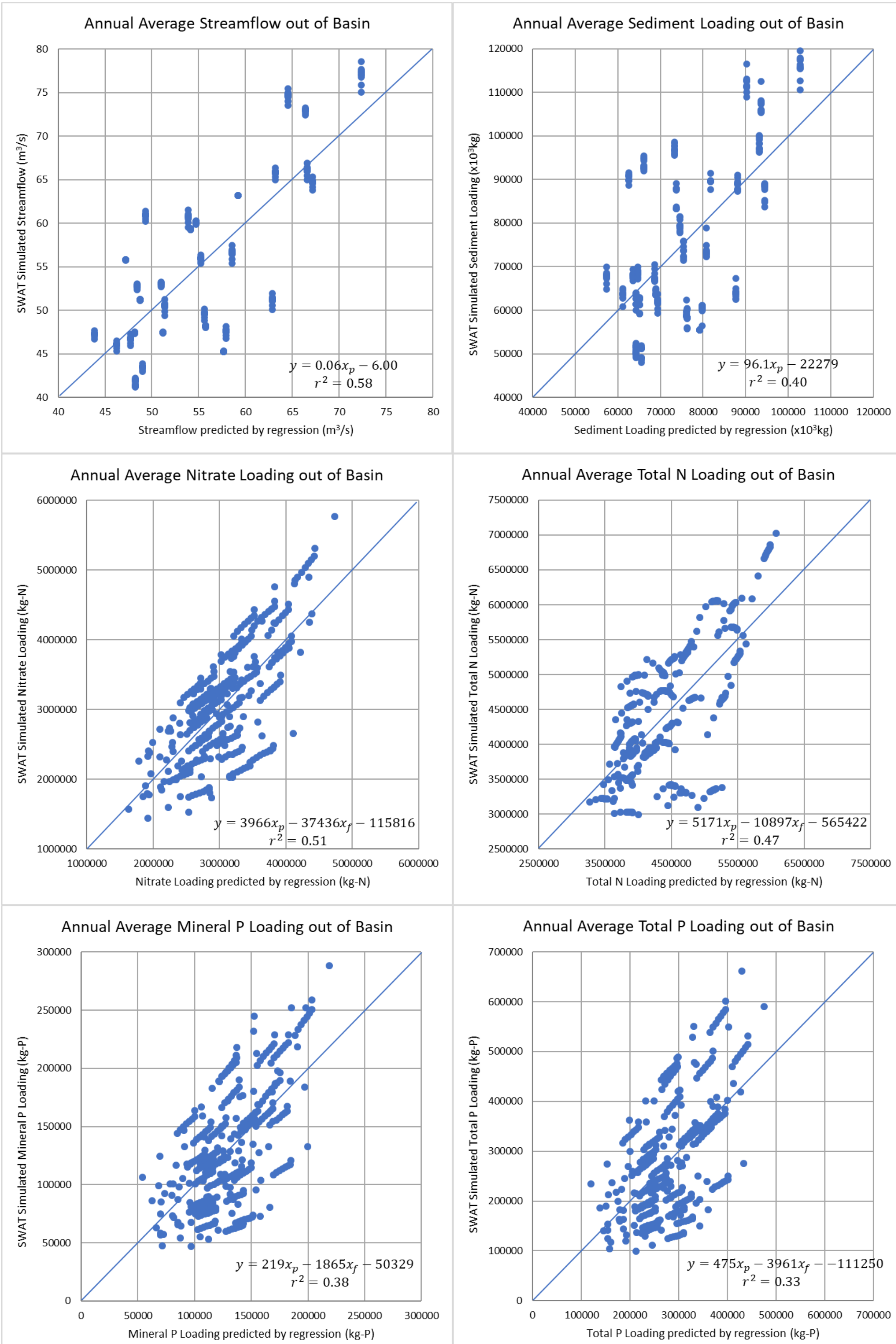

Figure 25 Scatter plots of SWAT-simulated and multivariate regression predicted streamflow and loadings at the basin outlet. In the regression equations, x_f is forest cover (percent) and x_p is precipitation (mm/y). Clusters of points relate to a range of LU scenarios for a given year and precipitation total

5.4. Quantitative impact of BMPs

In addition to examining the impact of converting forested land to cropland (or vice versa), the same land use scenarios were simulated with the addition of nutrient management BMPs. As mentioned previously, idealized representations of vegetated filter strips and grassed waterways were included in separate simulation series to provide some indication of the potential of such approaches to aid in the management of stream and nutrient flows in the South Nation.

5.4.1. Monthly averages

Due to limitations of the SWAT model, the influence of these BMPs on the components of water yield to the reach (and hence streamflow) are not calculated (White and Arnold 2009), although for tile flow this may be reasonable as soil drainage by tiles is likely to reduce the volume of surface flow reaching the BMPs such that each may have minimal influence on the other. Further, as we concluded from examination of the average sediment flows to the reach and the sediment loading carried out of the basin at Plantagenet, stream or reach processes dominate the balance of sediment in the river, and this effect is observed again here, where sediment carried in runoff is significantly reduced by BMPs, but sediment export from the basin is unaffected. However, both BMP options have a notable influence on nutrients transported to the reach and exported from the basin.

Figures 26-28 and Table 11 show monthly averages for the 2008 land use scenario for sediment and nutrients in runoff, and mass exports and concentrations, respectively, at the basin outlet near Plantagenet. Figure 26 indicates that the installation of VFSs has the potential to significantly reduce the sediment and nutrient loadings in runoff (by 50% or more for these simulations), particularly during peaks. GWWs are less effective, particularly for nitrates. Similarly, GWWs have a negligible impact on nitrate and total nitrogen loading exported from the basin, while VFSs have little influence for most of the year except for May-June, when fertilizer is applied in these simulations, and October-December after the harvest. Mineral and total phosphorus loadings are reduced fairly consistently year-round, with VFSs once again leading to a greater reduction than GWW. Although not shown here, simulations of both BMPs together showed negligible changes from the simulations of VFSs only.

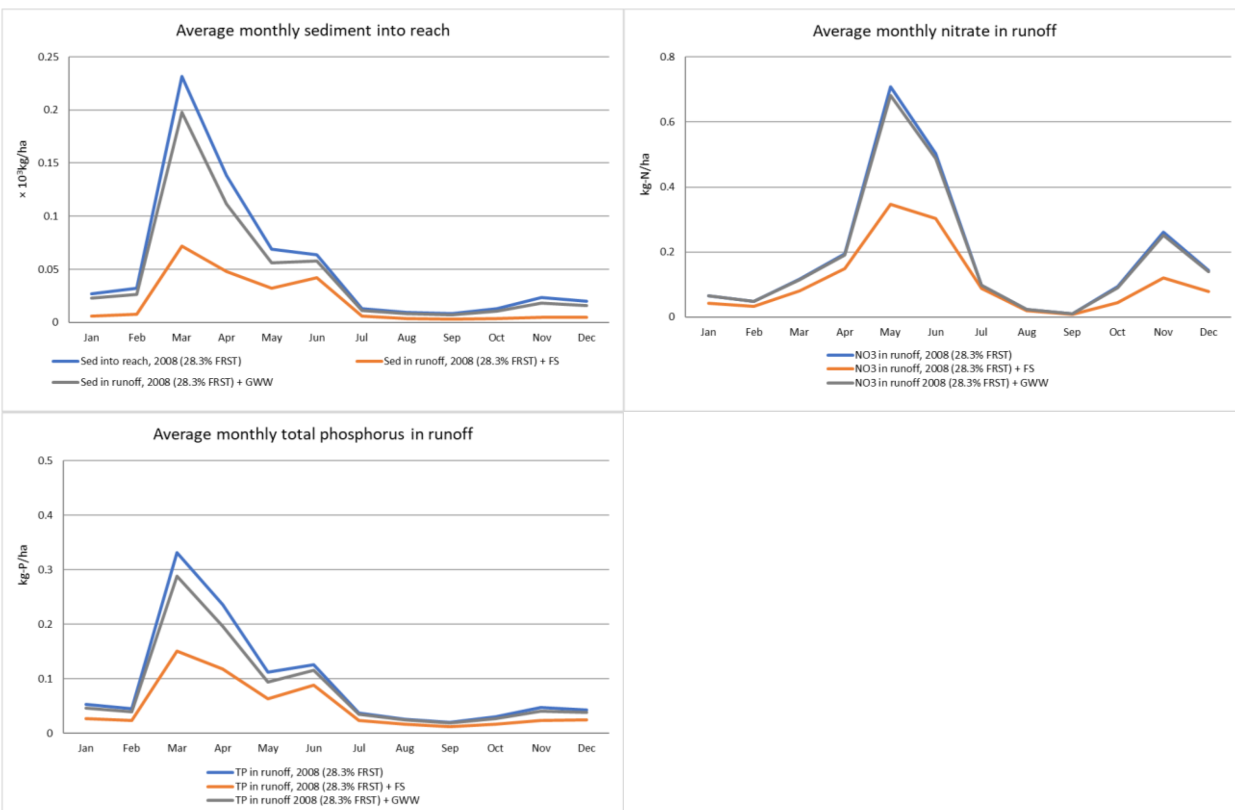

Figure 26 Monthly average loadings to the reach for 2008 LU scenario with and without BMPs simulated

Figure 27 Monthly average exports of sediment and nutrients from the basin, 2008 LU scenario with and without BMPs simulated

Figure 28 Monthly average nutrient concentrations at the South Nation watershed outlet, 2008 LU scenario, with and without BMPs simulated.

Table 11 Summary of watershed average sediment and nutrient loadings into reach, and exported from basin under 2008 LU scenario with and without BMPs

Variable		2008 DRAPE (no BMPs)	With VFS		With GWW	
Sediment in runoff	Maximum monthly	0.232	0.072		0.198	
($\times 10^3$ kg/ha)	Ave			-69.1%		-14.6%
	Average Annual	0.649	0.234		0.543	
				-64.0%		-16.4%
Nitrate in runoff	Maximum monthly	0.708	0.346		0.681	
(kg-N/ha)	Ave			-51.1%		-3.7%
	Average Annual	2.266	1.313		2.199	
				-42.1%		-2.9%
Total P in runoff	Maximum monthly	0.331	0.151		0.288	
(kg-P/ha)	Ave			-54.5%		-13.0%
	Average Annual	1.105	0.588		0.963	
				-46.8%		-12.9%
Nitrate export	Maximum monthly	1239	1215		1236	
($\times 10^3$ kg-N/year)	Ave			-7.8%		-0.2%
	Average Annual	3075	2834		3043	
				-7.8%		-1.0%
Total N export	Maximum monthly	1497	1455		1482	
($\times 10^3$ kg-N/year)	Ave			-2.8%		-1.0%
	Average Annual	4448	4156		4386	
				-6.6%		-1.4%

Mineral P export ($\times 10^3$ kg-P/year)	Maximum monthly Ave	32.8	23.9	-27.1%	29.9	-8.9%
	Average Annual	129.0	100.3	-22.2%	120.4	-6.7%
Total P ($\times 10^3$ kg-P/year)	Maximum monthly Ave	86.1	58.7	-31.7%	73.6	-14.5%
	Average Annual	280.5	204.7	27.0%	248.3	-11.5%

5.4.2. Annual averages

Also included in Table 11 and plotted in Figures 29-31 are average annual total loadings into the reach and out of the basin for all land use scenarios as simulated with and without the installation of VFSs and GWW. We can see in these displays again that the impact of these BMPs on runoff loadings is much more significant than on the loadings transported out of the basin. The impact on phosphorus transport is greater than on nitrogen transport. Figures 29-31 also allow for a visualization of the capacity of these BMPs to (partially) compensate for the changes in sediment and nutrient. Even the most optimistic reforestation scenario cannot match the reduction produced by VFSs in loadings to the reach; due to its weaker impact, GWWs are more easily matched. In terms of loadings exported from the basin (Figure 30), the comparison is more complicated. For example, the average annual loading of nitrate with 20% forest and VFSs is comparable to the loading with 30% forest cover and no BMPs. Alternately, the export of TP with 10% forest with VFSs is comparable to the export of TP with 40% forest and no BMP. What is clear is that VFSs are consistently more effective than GWWs in this context and that only quite significant reforestation has any potential to match the nutrient export reductions from the installation of VFS.

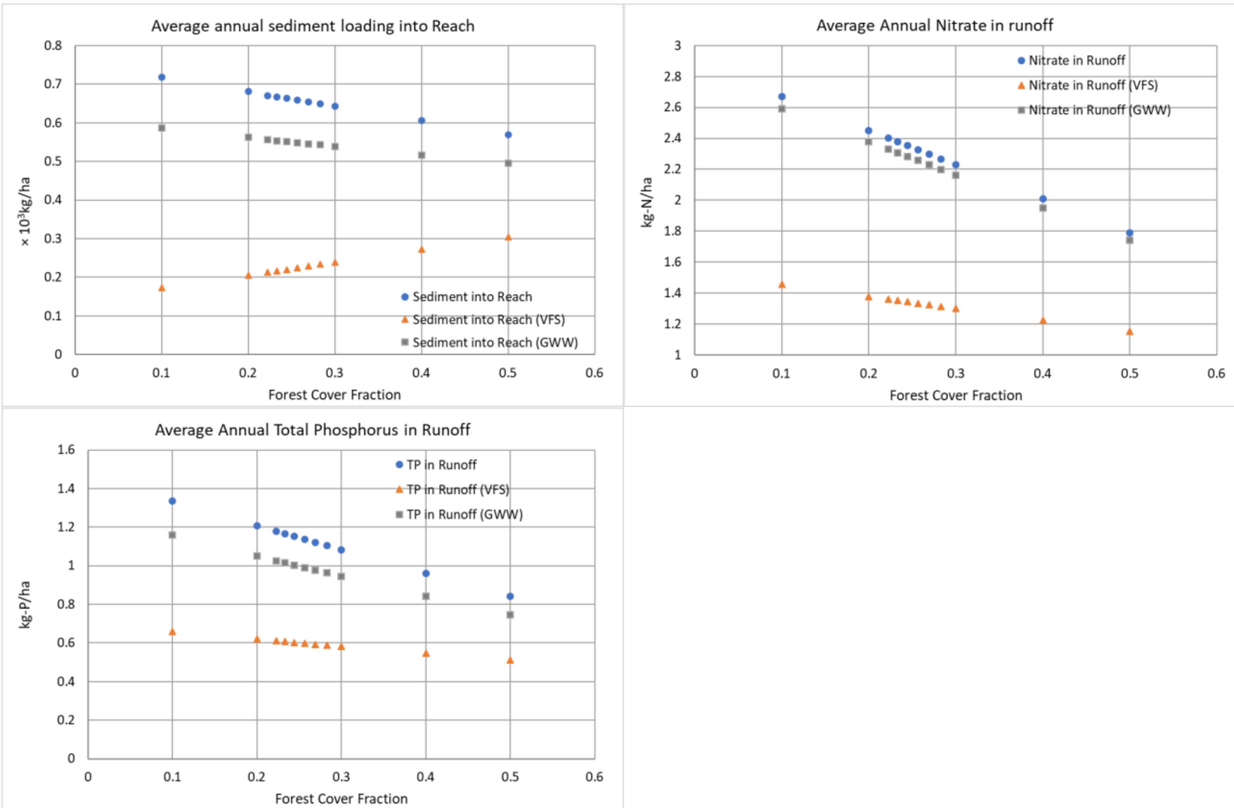

Figure 29 Watershed average annual sediment and nutrient loadings to the reach, with and without BMPs simulated

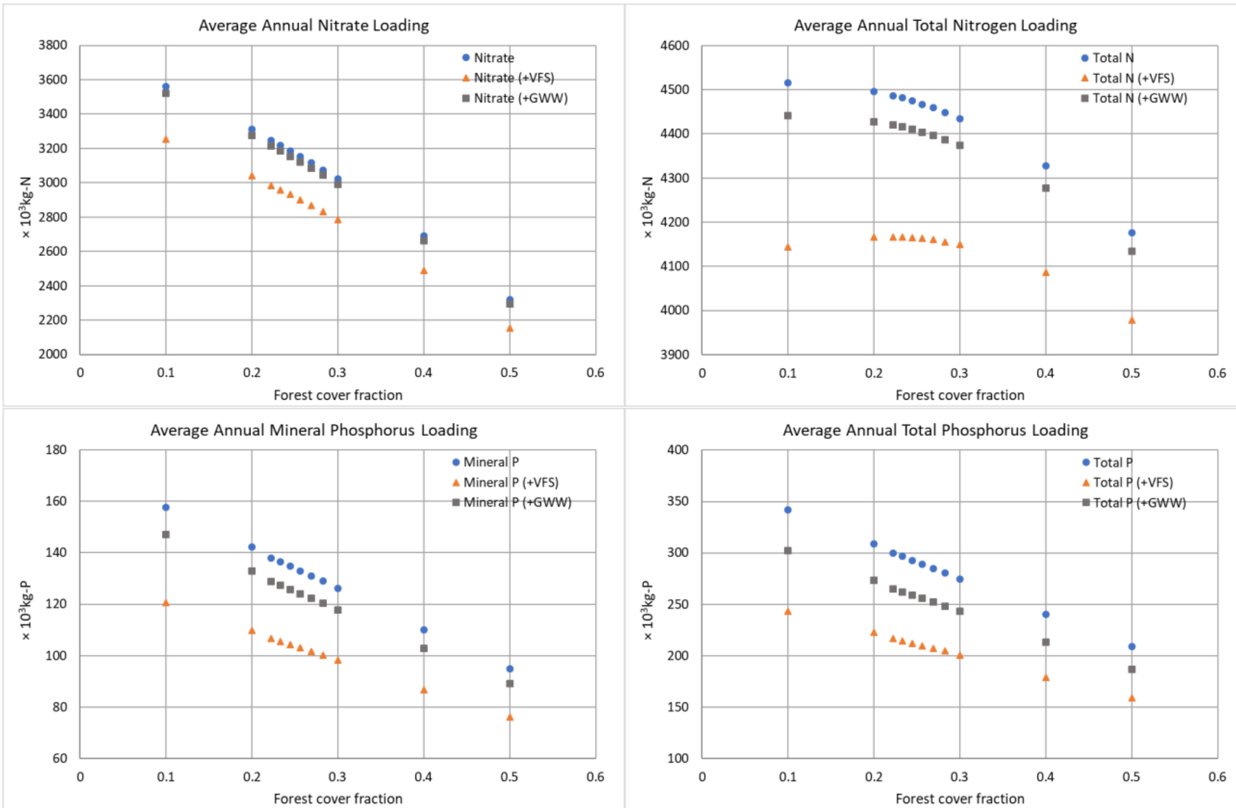

Figure 30 Average annual nutrient exports from the South Nation watershed, with and without BMPs simulated

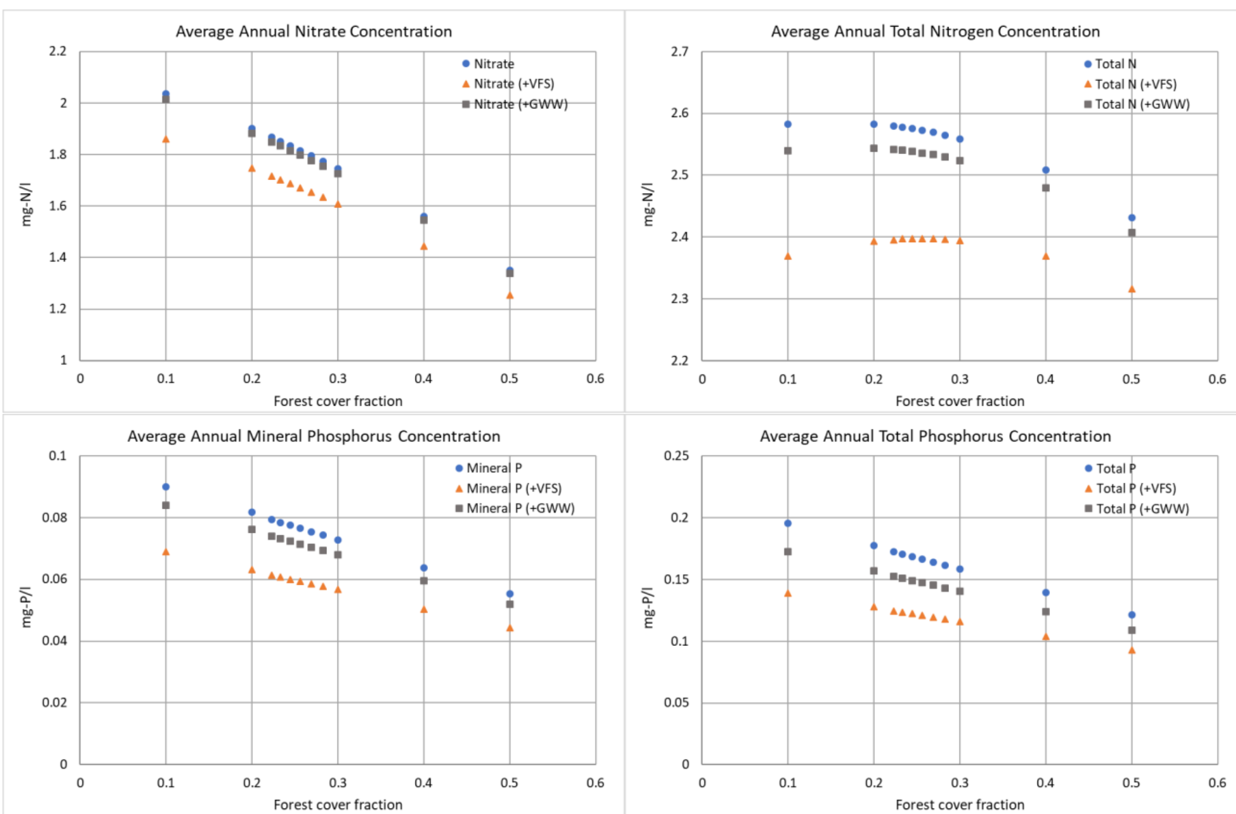

Figure 31 Average annual nutrient concentrations at the South Nation watershed outlet, with and without BMPs simulated

6. Discussion

6.1. Forest cover change

It was shown in 5.2.1 that flow from forested HRUs is quite distinct from the cropped HRUs, and that nutrient exports are significantly lower from forest HRUs due to the absence of fertilizer and tile drainage in those zones. The following key points can be drawn from the overall analysis of these simulations of forest cover change across the South Nation Watershed:

- The annual cycle of net water yield to the River and streamflow is dominated by spring snowmelt and freshet;
- The timing of peak nitrate loading to the reach corresponds to fertilizer applications;
- The timing of peak phosphorus loading to the reach closely match sediment peak;
- The peak loadings at outlet occur with spring freshet, while nutrient concentrations occur later, near fertilizer applications;
- Runoff from subbasins to reaches and streamflow at the basin outlet are largely insensitive to forest cover changes;
- Sediment loading to the reach is heavily influenced by forest change while sediment loading at the basin outlet changes very little;
- Nutrient loadings to the reach are more sensitive to forest change than loadings at the basin outlet;
- Increasing deforestation drives increases in nutrient flows and loadings at the reach and watershed scale; and
- Stream or reach processes appear to reduce the nutrient loading changes observed at the basin outlet.

Further forest clearing in the South Nation from current levels to 20% forest cover (a further reduction of 260 km^2 beyond the 51 km^2 lost between 2008 and 2014) would lead to a negligible change in water yield to the river system and annual average streamflow but increases in annual runoff nitrate and total phosphorus loadings to the reach of 8.1% and 9.4%, respectively, and increases at the basin outlet of 7.7% (NO_3) and 10.2% (TP). Maximum monthly concentrations of total nitrogen and phosphorus at the basin outlet are set to increase by 2.9% to 5.22 mg/L (TN) and by 5.1% to 0.133 mg/L (TP), further above current CCME benchmarks of 3.0 mg-N/L nitrate and 0.035 mg-P/L total P.

At the other end of the range of land use scenarios, even the most ambitious reforestation scenario considered here is insufficient to reduce maximum monthly averages to CCME standards. The 50% forest cover scenario represents a gain of over 860 km^2 from the 2014 DRAPE forest cover scenario. Although this simulation returns an annual average nitrate concentration of 1.1 mg-N/L, the maximum monthly average nitrate remains around 3.0 mg-N/L, while both annual and maximum monthly average TP concentrations are significantly above the 0.035 mg-P/L threshold defined as eutrophic by CCME. It is clear from these results that forest expansion and management alone is not sufficient to meet water quality standards, and that other approaches may be required in concert with forest management.

It is apparent from this study that streamflow out of the basin is influenced by, but not highly sensitive to changes to forest cover, and that for many of the flow and loading parameters examined, there is a linear relationship with forest cover. Although this may simply reflect the simple influence of replacing forest with tile-drained crops (or vice-versa) rather than a more complex ecological change, the influence of that replacement is real and must be considered, particularly the significant increases in nitrogen and phosphorus loadings. These observations are in line with the results published by El-Khoury et al. (2015) for the South Nation River and Tu (2009) for eastern Massachusetts. Tong et al. (2012), however, saw significant streamflow increase and only moderate nutrient increases, but with a land use projection that included significant urbanization and forest increase. Notably, the work of El-Khoury et al. (2015), although similar to this thesis, applied more complex land use change projections and future climate conditions while we have isolated the change from forest to cropland with real-world climate data from recent decades. Canada's Changing Climate Report (Bush and Lemmen, 2019) projects less than ±10% change to 2050 in annual and summer precipitation, and only slightly higher winter precipitation under low carbon emission scenarios. Even high carbon emission scenarios lead to only slightly higher projections, all of which suggests the use of real-world historical meteorological data is a valid approach. More significant is perhaps the projected reduction in snow accumulation as an increasing proportion of winter precipitation falls as rain which likely will significantly influence the timing and quantity of spring flooding with peak runoff potentially occurring earlier in the spring, and and/or not reaching historical levels.

Questions remain regarding the transport of sediments and the contribution of tile drainage to flow and nutrient loadings. Regarding the transport of sediments, it is notable that despite considerable changes in sediment carried by runoff between land use scenarios, changes in loading at the basin outlet are negligible. In section 5.2.1, the Modified Universal Soil Loss Equation (MUSLE) (Williams 1975) was introduced:

$$sed = 11.8 \cdot \left(Q_{surf} \cdot q_{peak} \cdot area_{hru}\right)^{0.56} \cdot K_{USLE} \cdot C_{USLE} \cdot P_{USLE} \cdot LS_{USLE} \cdot CFRG$$

which defines the sediment exported from each HRU. Of these terms, most are calculated within SWAT, while only K_{USLE} and P_{USLE} were calibrated in this study. However, due to the dominance of reach processes, they did not affect the sediment export. SWAT calculates sediment routing in the reach using the peak channel velocity to define a maximum concentration above which deposition is the dominant process, while below that value degradation is the dominant process. The maximum sediment concentration of sediment out of any reach segment is defined by:

$$conc_{sed,max} = c_{sp} \cdot v_{peak}{}^{spexp}$$

where $conc_{sed,max}$ is the maximum concentration of sediment ($\times 10^3$ kg/m^3), v_{peak} is the peak channel velocity in the reach, and c_{sp} and $spexp$ are parameters defined (calibrated) by the SWAT user. After the limiting effect of routing was noted, alternative configurations of erosion parameters and routing parameters were tested with SWAT-CUP, however, no configuration was found where routing processes did not override most variations in runoff sediment. Additionally, TP export was far more sensitive to changes to P_{USLE} and is a higher priority for this study. While this sediment routing behaviour is not impossible, there is certainly an opportunity for further studies to examine the bed and bank stability and sediment transport characteristics of the South Nation River and its tributaries. Such studies, through simulation and/or observation, would benefit from more extensive and consistent monitoring of water quality within the watershed.

6.2. Regression

It was shown that this set of simulations indicate that increased annual precipitation unsurprisingly applies an upwards pressure on streamflow and nutrient loadings out of the basin. Canada's Changing Climate Report (Bush and Lemmen, 2019) indicates 5-6% increases in annual precipitation to 2050 across Canada, even under conservative emission scenarios, so this increase in flow and loadings with increased precipitation will only become more significant in coming decades.

Multiparameter regression found that forest cover and precipitation together only account for up to 60% of the variance in the results, suggesting there is an opportunity for closer analysis of other drivers such as runoff, subsurface flow and percolation.

6.3. Installation of BMPs

The reader needs to keep in mind that the effects of the BMPs examined in this study were not calibrated using field data. All the conclusions are conditional on the accuracy of the algorithms implemented in SWAT for these particular BMPs. The following key conclusions can be drawn from the analysis of simulations executed with the inclusion of vegetated filter strips (VFSs) and grassed waterways (GWWs):

- VFSs are more effective than GWWs at reducing sediment and nutrient flows to reach and at the basin outlet
- Wide-scale adoption of VFSs is more effective at reducing nutrients than most reforestation scenarios
- GWWs are more effective at reducing sediment and phosphorus than nitrate or TN, likely due to improved trapping of sediment and attached phosphorus
- Again, reach processes counteract gains made to runoff loadings, particularly sediment
- Combining VFSs and GWWs makes negligible gain over VFSs only

- These BMPs are not sufficient to reach nutrient concentration standards with current LU balance

Although there has been limited comparable study made of the impact of BMPs on the South Nation River, other studies have used various approaches with similarly variable results. Many previous studies also used the older SWAT filter strip implementation, which defined a uniform trapping efficiency value for sediment, nutrients and pesticides based on a filter strip width defined for each HRU. This study rather implemented the newer algorithm developed by White and Arnold (2009) based around a ratio of field area to filter strip area. Parker et al. (2008) used an alternative modelling tool to examine BMPs in the South Nation watershed but similarly found that vegetated filter strips have significant potential with the management of nutrient loadings, but without changes to the scale of agriculture in the basin water quality standards could not be met. The results here also compare favourably with those published by Hanief and Laursen (2019) for the Grand River in Southern Ontario, although that work combined VFSs with conservation tillage, and found that both VFSs and GWWs had the potential to significantly reduce sediment transported from the watershed. Working on the Treene catchment in Germany, Haas et al. (2017) and Lam et al. (2011) (working on a tributary catchment) also found VFSs provide for a significant reduction in TN and NO_3.

In terms of the sediment and nutrient loadings carried by runoff to the reach, VFS installation is capable of achieving loading reductions under all forest cover scenarios that are greater than even the ambitious 50% forest cover scenario considered here, both at peak flows and in annual totals. Nonetheless, the lowest nutrient loads are delivered by scenarios with VFS *and* 50% forest cover.

At the basin outlet, the influence of the BMPs is somewhat attenuated, such that VFSs only partly offset the effects of deforestation. While 50% forest cover with VFSs remains the best-case scenario for nutrient loadings, partial deforestation (to 20% cover) combined with VFSs achieves similar TP loadings to 45-50% forest cover, and nitrate loadings similar to the 30% forest scenario. Although adding VFSs to cropland in the South Nation has the potential to offset some further deforestation, and/or maintain the water quality status quo, other approaches are necessary to meet CCME standards: the combination of VFSs with foreseeable forest cover scenarios reduces average annual nitrate concentration sufficiently, while spring and autumn peaks remain above thresholds, while even significant reforestation combined with VFSs delivers TP concentration almost three times the eutrophic threshold.

Also of note, again, is the attenuation in the reach of gains made to nutrient and sediment levels in the runoff. Although VFSs significantly reduce loadings in runoff to the reach, exports of sediment from the watershed are still apparently limited by deposition/degradation processes, which also potentially partially accounts for the attenuation of TP gains while the attenuation of nitrate reductions may be related to changes in the balance of nitrogen species leading to different trends in the nitrogen routing calculations. As with the forest cover changes, understanding of these processes could be improved by denser water quality sampling and improved characterization of concentrations of various nitrogen and phosphorus species.

7. Conclusions

This study examined the influence of deforestation in the South Nation Watershed, a predominantly agricultural watershed using hydrological modelling based on real-world land use and meteorological data. The land use was derived from national data published by AAFC, combined with previous eastern Ontario studies. Meteorological data spanning 1981-2001 for stations within the watershed was collected from Environment and Climate Change Canada. Using land use data for 2008 and 2014, trends for forest change were developed for each of the 31 subbasins within the watershed; for all but one very small subbasin, this trend is characterized as forest loss. More dramatic forest cover scenarios were developed to allow for the consideration of accelerated forest loss, as well as the possibility of stabilization and reforestation. Besides, the addition of idealized vegetated filter strips and grassed waterways were added to cropland land packages to consider the potential of these best management practices to offset increases in sediment and nutrient loadings related to forest cover loss.

The behaviour of water yield to the River, streamflow, and loadings to the reach and the basin outlet under varying forest cover and precipitation were examined. Results showed a consistent inverse relationship between forest cover and all flow and loading parameters, although the sensitivity of water yield, streamflow and sediment export was low. However, this study shows that further loss of forest cover is likely to drive significant increases in nutrient loadings while reversing the recent land conversion trend would decrease nutrient loading. Furthermore, this study indicates that considered in isolation, significant reforestation (500 km^2) would be required to reduce total nitrogen concentrations below federal standards, and even the most optimistic scenario tested returns total phosphorus levels well above provincial standards.

The introduction of BMPs to agricultural land packages showed significant potential, particularly for VFS, to reduce sediment and nutrient loadings into the stream system, as well as the export of nutrients, particularly phosphorus, from the watershed. However, even under the most optimistic reforestation scenarios combined with VFS, concentrations of nitrate and total phosphorus remain above benchmarks recommended by the Canadian Council of Ministers of the Environment.

Since the most ambitious simulations here (significant reforestation and VFSs on all agricultural land) fail to meet water quality standards, there is potential for further research to consider other management approaches that have been considered for other watersheds. Examples include reduced fertilizer applications, cover crops and alternative tillage approaches. It was also observed that despite large variations in sediment loading in the runoff between various land use and BMP scenarios, there was little variation in sediment export from the basin. Improved water quality sampling within the watershed would allow for confirmation or correction of this behaviour in the model, as well as potentially improving the constraints on nutrient transport.

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[dataset] Flow and Level for South Nation River near Plantagenet Springs (station number 02LB005). Extracted from the Environment and Climate Change Canada Historical Hydrometric Data web site (https://wateroffice.ec.gc.ca/mainmenu/historical_data_index_e.html) on 5 October 2018

[dataset] Water Quality for South Nation River (Station ID 18207002002, Cnty Rd 17, dwnstrm Plantagenet). Extracted from the Ontario Provincial (Stream) Water Quality Monitoring Network web site (http://www.ontario.ca/environment-and-energy/provincial-stream-water-quality-monitoring-network) on 7 May 2020