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Determining the Influence of Land Use Change and Soil Heterogeneities on Discharge, Sediment and Phosphorus

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ABSTRACT. Southern Quebec's Missisquoi Bay, a freshwater body in the northeastern portion of Lake Champlain is threatened by algal blooms arising from excess nutrient inputs contributed by agricultural watersheds which have their outlets in the bay. A version of the Soil and Water Assessment Tool (SWAT) model, calibrated in a previous study to estimate annual runoff, sediment and total phosphorus (TP) fluxes from the Castor subwatershed into the Pike River watershed, which, in turn, flows into the Missisquoi Bay, used static landscapes and single land uses to arrive at its predictions. However, in reality, farmers do rotate crops. Therefore, the present study's objective was to quantify the impact of soil heterogeneities on land use change patterns in the Castor subwatershed from 1999 to 2011. Data from a 24-point soil survey within the Castor subwatershed were partitioned and regionalized into 5, 10, 15, 20 and 24 heterogeneous regions or configurations. Using the standard soils map (with mean properties) employed in several prior studies in the subwatershed, a sixth configuration termed "Reference," was also developed. All 6 configurations were factorially combined with either 1999 or 2011 land use data to yield 12 different versions of the SWAT model and quantify the heterogeneities and uncertainty of soil properties on land use change. For hydrology, it was discovered that there were no marked differences in the predictions, which was attributable to the use the SCS-CN subroutine which masks the physical properties of soil parameters within the same hydrologic group. We evaluated all the models for two periods i.e. 1991-1999 and 2000-2007. All the 1999 land use SWAT configurations underestimated runoff, sediment and TP whereas all the 2011 land use SWAT set ups gave higher and more accurate values. For both land use periods, the 5 Region models both showed higher and more accurate estimates, than those set ups with a greater number of regions, but were similar in accuracy to the Reference model set-ups. Since the 5-region configurations showed the highest within-zone heterogeneity, it can be concluded that having many regions (many sampling points as regions) does not necessarily increase SWAT's prediction accuracy.

Keywords: Curve Number, total phosphorus, partitioning, land use

1. Introduction

The Lake Champlain Basin Programme (Winslow, 2012) was an initiative undertaken to restore and protect Lake Champlain's 1,127 km^2 of surface area and the 21,000 km^2 of its contributing watersheds for future generations. The lake is bordered by the states of New York and Vermont in the United States and the province of Quebec, in Canada. New York-, Vermont- and Quebec-encompassed watersheds contribute 37, 56 and 7% of Lake Champlain's waters, respectively. Southern Quebec's Missisquoi Bay, located in the northeast portion of Lake Champlain is threatened by excessive algal growth, and has been reported as one of the main contributors to pollution in Lake Champlain (Giroux et al., 1996; Jamieson et al., 2003; Michaud et al., 2008). The Missisquoi Bay accounts for 24.1% of the total phosphorus (TP) load entering Lake Champlain (LCBP, 2004; Winslow, 2012). This situation is largely due to the persistent seasonal application of fertilizers and manures in agricultural watersheds in Quebec. This has, in turn, led to a worsening of the health of aquatic ecosystems in the Bay. In this impaired freshwater environment, algal growth has turned the water green, reduced transparency, depleted available oxygen for aquatic life and creates odour problems. The high nutrient loads and concentrations [nitrogen (N) and phosphorus (P)] entering the lake have been reported to come from Non-Point Sources (NPS). Vermont and Quebec have been able to make significant reductions in P loadings from point sources, such as sewage treatment plants, but there remains a great challenge in providing strategies to reducing TP coming from NPS. The High levels of P coming from NPS, largely arise from farmers' excessive manure and fertilizer applications on limited land resources. Moreover, there is an increase in animal density and intensive agriculture in the area. This is a serious challenge in terms of water health and socio-economic activities for the people living in this area. It is reported that several lakeside parks

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have been closed in the Missisquoi Bay over recent years due to outbreaks of cyanobacterial blooms (Priskin, 2008). This has had a significant impact on aquatic life and also reduces revenue generation from tourism (Priskin, 2008). Most of the P promoting these blooms comes from agricultural lands, of which 30% account for 80% of the TP load (LCBP, 2004; Winslow, 2012).

The areas on the watershed where P loadings are higher are referred to as critical source areas (CSA). These are areas that combine a P source, associated with the particular soil type or management practices, so positioned in the watershed as to allow the contaminant to be delivered to freshwater (LCBP, 2004; Winslow, 2012). The watershed location is such that it includes hilly terrain where water runs off quickly, in proximity to the lake or rivers. Most CSA were identified to be on agricultural lands (LCBP, 2004; Winslow, 2012).

Several studies in this area have used hydrologic models, in particular the Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998; Arnold et al., 1999), to quantify the impact of management practices on runoff, sediment and TP fluxes from NPS (Deslandes et al., 2007; Michaud et al., 2007; Michaud et al., 2008). However, these results were based on a static landscape (single land use pattern). The inferences drawn from these studies have been based on the application of a single land use or static landscape, whereas, in reality, farmers rotate their crops. In the first paper of the research in this watershed, we developed SWAT models that integrate and quantify the various spatial resolutions of soil heterogeneities only and calibrate the processes of SWAT (Boluwade and Madramootoo, 2013). However, the objective of this paper is to quantify the accuracy of SWAT modeling for the Castor watershed using two different land use patterns combined with 6 different soil maps of varying complexities. The research question we attempt to address in the paper to know how much information is gained or lost using “old” (1999 land use dataset) or “recent” (2011 land use dataset)”.

2. Materials and Methods

2.1. Land Use Change in Castor Watershed, Quebec

Castor watershed land use data for 1999 was derived from LANDSAT ETM+ imaging from 5 July 1999 (Cattaï, 2004; Deslandes et al., 2007; Michaud et al., 2008). This was obtained in a processed format from the Institut de recherche et de développement en agroenvironnment (IRDA). The 2011 data was obtained through a field survey campaign carried out in July 2011. By imposing a regular grid of 100 by 100 m on the study area and with the aid of a global positioning system (GPS), each grid was surveyed and the current land use pattern noted. This information was reflected in a shapefile that was derived from the 1999 land use data. We edited this 1999 land use data to reflect the current condition using ArcGIS 10.1 software (ESRI, 2012).

2.2. Regionalization and Partitioning of the Soil Data

In the first part of this study (Boluwade and Madramootoo, 2013), a basic soil survey was undertaken to measure properties needed by SWAT for prediction. The Regionalization with Dynamically Constrained Agglomerative Clustering and Partitioning (REDCAP) (Guo, 2008) method was used to create different configurations of the soil data. This procedure involves the division of the spatial objects into a number of contiguous regions while optimizing an objective function (Guo, 2008). We applied the Full-order Complete Linkage algorithm which basically is a two level procedure involving:

- Clustering of the data with contiguity constraints to produce spatially contiguous trees and
- Partitioning the tree to generate regions while optimizing an objective function.

The major advantage of using a full-order complete linkage is that it can be applied to achieve complexity without having to use a binary search and sorting tree, since there is no need to re-examine prior visited edges after each merge (Guo, 2008). The further description, procedure and concepts regarding REDCAP are given in Guo, (2008).

In all, we set up twelve SWAT models using the soil-clustered configurations and also the 1999 and 2011 land use data sets (Table 1).

2.3. SWAT Modeling

SWAT is a semi-distributed physically-based model which can predict the impact of management decisions on surface flow, sediment and nutrient loads from point and non-point sources, on hourly, daily, monthly and annual timescales (Arnold et al., 1998; Arnold et al., 1999). In SWAT, a watershed may be partitioned into a number of sub-basins. This is very beneficial when different areas of the watershed are dominated by land use or soils dissimilar enough in properties to impact hydrology (Neitsch et al., 2009). The user is able to spatially reference different areas of the watershed to each other (Neitsch et al., 2009). Input information is classified into unique groups (within different sub-basins) called hydrologic response units (HRUs). These HRUs are composed of unique properties of soil, land use, slope and management combinations. As rain falls towards the soil surface, it can be held within the vegetation canopy or otherwise intercepted. Water reaching the soil surface can infiltrate through the soil profile or can contribute to overland flow (Neitsch et al., 2009). The infiltrated water may be later returned to the atmosphere through evapotranspiration or it can slowly flow to surface waters through underground pathways (Neitsch et al., 2009). Runoff is predicted separately for each HRU and routed to obtain the total runoff at the outlet of the watershed (Neitsch et al., 2009).

Hydrology is the driving force for the movement of nutrients and sediments from the watershed. The SWAT modeling process is based on HRU units. These processes can be divided into two phases: (i) the land phase, which deals with the amount of water, sediment and nutrient fluxes to the main channel in each sub-basin of the watershed, (ii) the routing phase, which deals with the movement of water, sediment and

Table 1. Resulting Twelve SWAT Configurations

1999 SWAT Configuration	Description	2011 SWAT configuration	Description
Reference_1999	This is the SWAT model set up using the soil map that has been used in the previous studies in the study area and 1999 land use data set.	Reference_2011	This is the SWAT model set up using the soil map that has been used in the previous studies in the study area and 2011 land use data set.
Region_5_1999	This is a SWAT configuration set up using the partitioned survey soil properties into 5 heterogeneous regions and 1999 land use dataset.	Region_5_2011	This is a SWAT configuration set up using the partitioned survey soil properties into 5 heterogeneous regions and 2011 land use dataset.
Region_10_1999	This is a SWAT configuration set up using the partitioned survey soil properties into 10 heterogeneous regions and 1999 land use data set.	Region_10_2011	This is a SWAT configuration set up using the partitioned survey soil properties into 10 heterogeneous regions and 2011 land use data set.
Region_15_1999	This is a SWAT configuration set up using the partitioned survey soil properties into 15 heterogeneous regions and 1999 land use data set.	Region_15_2011	This is a SWAT configuration set up using the partitioned survey soil properties into 15 heterogeneous regions and 2011 land use data set.
Region_20_1999	This is a SWAT configuration set up using the partitioned survey soil properties into 20 heterogeneous regions and 1999 land use data set.	Region_20_2011	This is a SWAT configuration set up using the partitioned survey soil properties into 20 heterogeneous regions and 2011 land use data set.
Region_24_1999	This is a SWAT configuration set up using the partitioned survey soil properties into 24 heterogeneous regions 1999 land use data set.	Region_24_2011	This is a SWAT configuration set up using the partitioned survey soil properties into 24 heterogeneous regions 2011 land use data set.

nutrients through the channel tributaries to the outlet of the watershed (Neitsch et al., 2009).

These two phases can be further described as:

- Land Phase of the Hydrologic Cycle

The hydrologic cycle in SWAT is simulated based on the water balance equation (Neitsch et al., 2009):

$$SW_t = SW_o + \sum_{i=1}^{t}(R_{day} - Q_{surf} - E_a - W_{seep} - Q_{gw}) \quad (1)$$

where,

t, time (days);

Q_{gw}, amount of return flow on day i (mm H_2O);

Q_{surf}, quantity of surface runoff on day i (mm H_2O);

R_{day}, quantity of precipitation on day i (mm H_2O);

SW_t, the final soil water content (mm H_2O);

SW_o, the initial soil water content on day i (mm H_2O);

W_{seep}, amount of water entering the vadose zone from the soil profile on day i (mm H_2O).

- Routing Phase of the Hydrologic Cycle

A command structure that is similar to that of the Hydrologic Model (HYMO; Williams and Hann, 1972; Neitsch et al., 2009) is used to route the loadings of water, sediment and nutrient to the main channel through the watershed's stream network (Neitsch et al., 2009). The routing process in the main channel can be divided into four different parts (Neitsch et al., 2009)

1. *Flood Routing*: The variable storage coefficient method developed by Williams (1969) or the Muskingum routing method can be used to route the flow through the main channel.
2. *Sediment Routing*: This is done through a simultaneous process of degradation and deposition. Former versions of SWAT used the stream power to quantify deposition and degradation (Neitsch et al., 2009); however, the current version (SWAT2009) uses the maximum amount of sediment that can be transported from a reach as a function of peak channel velocity (Neitsch et al., 2009).
3. *Nutrient Routing*: This is controlled by the in-stream water quality component of the model (Neitsch et al., 2009). This is adapted from the QUALE model (Brown and Barnwell, 1987; Neitsch et al., 2009). This model tracks both nutrients adsorbed to the sediment and those dissolved in the stream.
4. *Channel Pesticide Routing*: Total pesticide load in SWAT modeling is divided into dissolved and sediment-attach-

ed pesticides. Only one type of pesticide may be modelled at one time due to the complexity of the process (Neitsch et al., 2009). The transformation of the pesticide in the dissolved and sorbed phases is controlled by a first-order decay relationship (Neitsch et al., 2009).

Concerning sediment loading, the prediction of sediment in the watershed's channel is controlled through simultaneous processes of deposition and degradation (Neitsch et al., 2009). William's (1980) definition of stream power in the Bagnold equation was used in the determination of degradation, as a function of channel slope and velocity (Neitsch et al., 2009).

In terms of nutrient loadings, phosphorus (P) has been discovered to be the dominant nutrient in this watershed (Deslandes et al., 2007; Michaud et al., 2008). Two P pools are modelled by SWAT: Mineral P and Organic P. These can be applied to the soil either as a fertilizer, manure or compost. SWAT models P by grouping its different forms into six pools (Chaubey et al., 2006). In SWAT, subsurface drainage is not considered a pathway for P loss, which is one of the model's major limitations. It has been reported in several studies that significant amounts of P can be found in subsurface drainage water (Heckrath et al., 1995). Soluble P and Particulate P are modeled for surface runoff. Top soil (up to the depth of 10 mm) is the only portion of the soil considered to interact with overland flow. The only loss of soluble P occurs from this layer through surface runoff (Chaubey et al., 2006).

The following section defines how P movement through surface runoff is modeled (Chaubey et al., 2006).

Soluble P movement through surface runoff is calculated as:

$$P_{surface} = \frac{P_{(solution,surface)} Q_{surface}}{\rho_b depth_{surface} K_{(d,surface)}} \tag{2}$$

where,

$K_{(d,surface)}$ is the P soil partitioning coefficient (m^3 Mg^{-1});

$P_{surface}$ is the soluble P transported by surface runoff (kg P ha-1);

$P_{(solution,\ surface)}$ is the labile P in the top 10 mm of soil (kg P ha^{-1});

$Q_{surface}$ is the surface runoff in a given day (mm);

$depth_{surface}$ is the depth of the "surface" layer (10 mm);

ρ_b is the bulk density (Mg/m^3).

In terms of applications, Gassman et al., (2007) summarized the applications of SWAT globally. Many of the applications are from North America. Eckhardt et al. (2002); Conan et al. (2003) a,b; Bouraoui et al. (2004); Gikas et al. (2005); El-Nasr et al. (2005); Bärlund et al. (2007); Glavan and Pintar (2012) are all work in Europe using SWAT.

In this SWAT modeling, the simulation period for the study area was from January 1971 to December 2007. The periods from January 1971 to December 1976 (5 years) were used as a warm-up period. This was done to allow the model to "stabilize", especially for model parameters such as soil moisture, which are dependent on prior conditions. The time periods from April 2001 to December 2002 were used for the evaluation of flow, sediment and TP fluxes. The selection of the data period was based on the period for which observed data was available. Observed discharges, suspended sediments and TP data sets were obtained from the Ministère du Développement Durable, Environnement et Parcs du Quebec (MDDEPQ). Automatic subbasin delineation, based on given threshold areas (10% for soil map and 0% for landuse) along with manual input of the subbasin outlet generated 4 subbasins for the watershed. SWAT then divided each subbasin into smaller units (HRUs) based on these thresholds represented as percentages of each land use, soil type, and slope. These thresholds and the initial labile P soil concentration (of 31 mg kg^{-1}) were chosen based on previous hydrologic modeling studies undertaken in the same area (Michaud et al., 2008). The spatial distribution of crops, derived from the classification of the 1999 land use (Cattaï, 2004) was maintained throughout the modelling process (Deslandes et al., 2007).

Daily precipitation and temperature data were obtained from the three closest weather stations in the study area: Philipsburg, Farnham and Sutton (MDDEPQ, 2003; 2005). All other parameterizations were done as outlined in Michaud et al. (2008).

2.4. Model Performance

The evaluation was based on comparing the simulated results for individual SWAT configurations and corresponding observed data. We compared the simulations (uncalibrated) from each of the configurations with the observed dataset for flow sediment and nutrient loading. This was to avoid any bias in the evaluation. Researchers such as Rosenthal et al. (1995); Heathman et al., (2008); Srinivasan et al. (2010); Niraula et al. (2012) have all used the SWAT model without calibration for specific purposes. In this case, we used the pre-calibrated results to avoid any form of bias due to adjustment of model parameters. However, interested readers can look at Boluwade and Madramootoo (2013) for the calibration procedure and model performance using the six soil configurations for another purpose. Previous efforts to assess goodness of fit in hydrologic modeling (Moriasi et al., 2007; Radcliffe et al., 2009; Mukundan et al; 2010; Najedhashemi et al., 2011; Qiu et al., 2012), the Nash Sutcliffe (*NSE*), mean difference (*MD*), root mean standard difference (*RMSD*), and coefficient of determination (R^2) were used to compare the simulations of each SWAT configuration, (i.e., with both 1999 and 2011 land use data sets). The period of April 2001 to December 2002 was selected as the evaluation period. These indices can be defined as:

$$MD = \frac{1}{n}\sum_{i=1}^{n}(O_{pi} - O_{bi}) \tag{3}$$

$$RMSD = \left[\frac{1}{n} \sum_{i=1}^{n} (O_{pi} - O_{bi})^2 \right]^{0.5} \quad (4)$$

$$NSE = 1 - \frac{\sum_{i=1}^{n} (O_{bi} - O_{pi})^2}{\sum_{i=1}^{n} (O_{bi} - \bar{O}_{bi})^2} \quad (5)$$

$$R^2 = \left\{ \frac{\sum_{i=1}^{n} (O_{bi} - \bar{O}_{bi})(O_{pi} - \bar{O}_{pi})}{\sqrt{\sum_{i=1}^{n} (O_{bi} - \bar{O}_{bi})^2} \sqrt{\sum_{i=1}^{n} (O_{pi} - \bar{O}_{pi})^2}} \right\}^2 \quad (6)$$

where,

n is the number of samples used in the computation;

O_{pi} is the i^{th} prediction from a given SWAT configuration;

O_{bi} is the i^{th} measured value at the outlet of the watershed;

$\bar{O}_{bi}$ is the measured mean value at the outlet of the watershed; and

$\bar{O}_{pi}$ is the mean prediction from a given SWAT configuration.

The *NSE* is an index that measures how well the predicted and measured values agree. An *NSE* of 1 indicates a perfect fit between model and measured values, while an *NSE* = 0 indicates the model predicts no better than the mean of the measured variable. Increasingly more negative values of *NSE* indicate poorer and poorer model performance. Values of *NSE* > 0.5 (Moriasi et al., 2007) are generally considered to represent good model performance. The R^2 is an indicator that varies between 0 and 1. Depending on the objective and precision desired by the user, an $R^2 \geq 0.5$ (Santhi et al., 2001; Van Liew et al., 2003) can be considered representative of a good model fit. Using this statistics, we tested the performance of the models using the streamflow, sediment loads and TP fluxes measured between April 2001 and December, 2002 (MDDEPQ, 2003; 2005). We also evaluate the performance of the model for two periods (1991 ~ 1999 and 2000 ~ 2007). This is to quantify the impact of land use change and soil heterogeneities on runoff, sediment and TP loads. These simulations were compared to a simulation with the "Reference" data set and a regionalized soil data set.

3. Results and Discussion

3.1. Land Use Change

Land use data sets from 1999 and 2011, illustrated in Figures 1a and b, respectively, show noticeable differences in land use patterns. The relative changes in terms of the total land area are shown in Table 2. There was a considerable change in pasturelands, with a reduction from 29 to 6% of the total area. This was confirmed by personal interviews with farmers in the area. Between 1999 and 2011 some large animal farms were converted to mixed-crop farming. Also, there was considerable increase in corn (Zea mays L.) production, from 43 to 60% of the total area. Soya bean [Glycine max (L.) Merr.] farming also increased from 2% in 1999 to 19% of the area in 2011. Oat cropping also declined from 20% in 1999 to 11 % in 2011.

3.2. Hydrologic Response Units (HRUs) under Different Soil and Land Use Configurations

The smallest unit of the watershed, the HRU is where all the hydrologic processes and modeling begin. SWAT delineates HRUs with thresholds (10% for soil map and 0% for landuse) represented as percentages of each land use, soil type, and slope. These thresholds were chosen based on previous hydrologic modeling done in the area (Michaud et al., 2008). The HRUs are aggregated together to form a sub-basin. Subbasins are networked together and streamflow routed. Estimates of runoff, sediment and nutrients reaching the outlet of the basin are then computed. Table 3 shows the HRUs under the two-land use pattern, using the different soil configurations. Slightly higher numbers of HRUs were derived for 1999 than 2011 land use data.

3.3. Model Results and Assessment

To avoid bias during model calibrations (parameter adjustments), we used the un-calibrated results of the configurations. This follows the report of Srinivansan et al. (2010)

Table 2. Land Use Percentages (% of Total Area) for Different Land Uses in the Castor Sub Watershed in 1999 and 2011

Land use Pattern	Land Use Percentage of Total Area (%)	
	1999	2011
Transportation	2.93	2.96
Oats	20.66	11.14
Forest-Mixed	1.74	1.68
Corn	43.15	59.67
Pasture	29.03	5.87
Soyabean	2.5	18.69

Table 3. Hydrologic Response Units (HRUs) for the Castor Watershed, Quebec in 1999 and 2011

SWAT Configuration	Hydrologic Response Units (HRUs)	
	1999	2011
Reference	60	64
Region_24	120	140
Region_20	110	124
Region_15	112	96
Region_10	80	78
Region_5	53	55

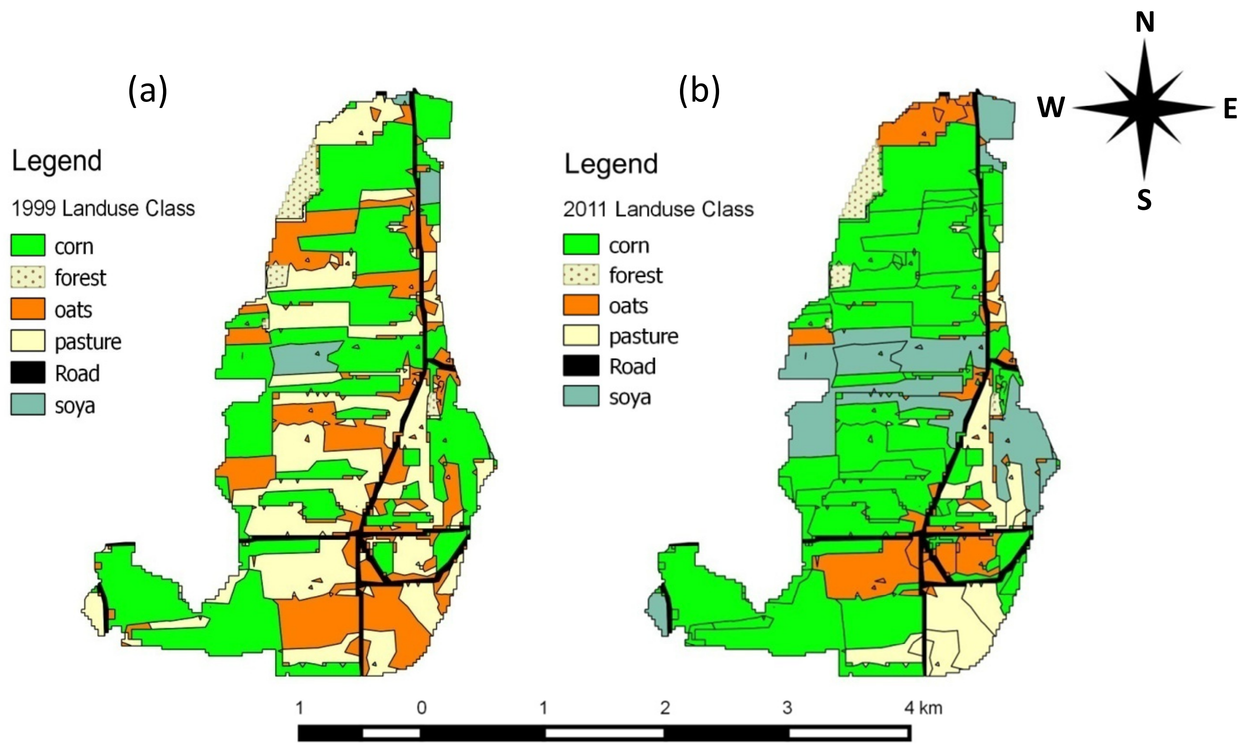

Figure 1. Land use pattern in the Castor watershed, Quebec (a) 1999, (b) 2011.

that *"... In addition, it is worth noting that uncertainties associated with input data and measured hydrologic variables may lead to biased estimation of parameters calibrated using one or several stream gauges. For example, under typical conditions, errors ranged from 6 to 16% for streamflow measurements (Harmel et al., 2006). A case study in Reynolds Creek Experimental Watershed showed that a parameter set with high streamflow simulation performance at the watershed outlet can have much lower performance at some internal points within the watershed (Zhang et al., 2008). Very frequently, the calibrated model is user-dependent; as it is based on the model user's experience and knowledge about the watershed, model, chosen parameters, and their ranges. Therefore, calibrated models may be limited to their intended purpose".*

In the same vein, Niraula et. al., (2012) also collaborated this: *"... In many watershed-modeling studies, due to limited data, model parameters for flow, sediment, and nutrients are calibrated and validated against observed data only at the watershed outlet. Model parameters are adjusted systematically for the entire watershed to obtain the closest match between the model-simulated and observed data at the watershed outlet (lumped calibration). It is hypothesized that the relative loadings of pollutants and/or sediments contributed by each computational unit are not affected by this calibration procedure. In other words, areas generating relatively higher pollutant loads with an uncalibrated model will still generate relatively higher loads after calibration".*

However, the calibrations and assessment of the model's performance using the same soil configurations have been performed in our earlier paper (Boluwade and Madramootoo, 2013).

3.3.1. Hydrology

In terms of predicted hydrology parameters, Table 4 summarizes the goodness of fit achieved with 1999 and 2011 land use datasets. There are no significant differences in accuracy between using the use of the 1999 or 2011 land use data in terms of average monthly discharges from April, 2001 to December, 2002. The coefficient of determination (R^2) and *NSE* were satisfactory for all the SWAT configurations. However, the Reference configuration with both land use data sets performed slightly better, while those of the equivalent 24 region configurations (regions created with the sample points without any averaging of the parameters) performed somewhat more poorly. In other words, the latter configuration showed the lowest within region heterogeneity (Guo, 2008). The lack of differences between model configurations might be due to the SCS-CN subroutine, which masks or covers the soil physical properties (Ye et al., 2009).

3.3.2. Sediment

Performance statistics for the impact of soil heterogeneity and land use changes for sediment load are given in Table 4, for both 1999 and 2011 land use data sets. Sediment loads were poorly predicted in all cases (*NSE* values). The R^2 values were similar for the two different land use datasets;

Table 4. Resulting Twelve SWAT Configurations

Model	Reference		Region									
			5		10		15		20		24	
	1999	2011	1999	2011	1999	2011	1999	2011	1999	2011	1999	2011
Monthly discharge (cms)												
R^2	0.72	0.60	0.67	0.57	0.69	0.60	0.69	0.59	0.69	0.59	0.63	0.41
MD	0.65	0.54	0.64	0.59	0.65	0.59	0.65	0.59	0.65	0.59	0.60	0.48
RMSD	0.20	0.19	0.18	0.19	0.17	0.18	0.17	0.18	0.17	0.18	0.21	0.22
NSE	0.49	0.57	0.57	0.55	0.60	0.60	0.59	0.59	0.59	0.59	0.40	0.40
Sediment load (Mg)												
R^2	0.58	0.45	0.56	0.54	0.58	0.58	0.58	0.58	0.58	0.58	0.45	0.45
MD	247	420	273	396	228	317	223	296	217	288	101	130
RMSD	217	446	238	397	204	315	201	285	195	274	167	168
NSE	-0.363	-4.56	-0.64	-3.51	-0.20	-1.84	-0.17	-1.33	-0.10	-1.15	-2.94	0.18
Total phosphorus (kg)												
R^2	0.83	0.74	0.78	0.78	0.85	0.84	0.85	0.84	0.85	0.85	0.88	0.88
MD	364.44	411.80	342.40	381.63	210.03	236.72	278.92	313.38	296.50	333.94	167.82	182.37
RMSD	592.85	598.62	640.10	601.85	753.50	719.97	664.47	623.43	640.37	596.20	774.44	751.89
NSE	0.59	0.59	0.53	0.58	0.34	0.40	0.49	0.55	0.52	0.59	0.31	0.34

however, there were huge differences in *MD* and *RMSD*. This indicates that by using the 1999 land use data set one underestimates sediment loads. This will be further evaluated during the analyses of the periods of 1991 ~ 1999 and 2000 ~ 20 07.

3.3.3. Total Phosphorus

For TP load at the watershed outlet, the performance statistics are summarized in Table 4 for the 1999 and 2011 land use datasets. All SWAT configurations performed well predicting P fluxes, as the R^2 and *NSE* values attest. As was the case with sediment loads, the *MD* and *RMSD* for the 2011 land use data were higher than for the 1999 land use data. It may be that the 1999 land use dataset leads to an underestimation of TP movement from the agricultural watershed.

Since in all cases, the configurations using the 2011 land use data set performed better than those using the 1999 dataset, time series of the observed and predicted values for flow, sediment and TP loads were plotted using the 2011 dataset. Using this dataset, flows were underestimated during major storm events, but overestimated during low flows. However, Deslandes et al. (2007) reported underestimations of streamflow by SWAT in the winter periods of January to March, 20 02 were due to the exceptionally unseasonable nearzero and above zero temperatures (as high as 10 °C) that were recorded at that time. In other words, all SWAT configurations had difficulty differentiating between rainfall and snowfall due to its weather generating subroutine (Deslandes et al., 2007). The discrepancies between the observed and the predicted water quality have often been linked to the differences in temporal scale (Radcliffe et al., 2009). The measurements from outlets of the watershed are grab samples or instantaneous measurements, whereas SWAT outputs (runoff, sediment and TP) are mean values (Radcliffe et al., 2009). This is an important issue especially when the grab samples or instantaneous measurements are higher than the average monthly predicted values (Radcliffe et al., 2009).

We further explore the relevance of using current land use patterns by considering the flow, sediment and TP fluxes for two long term intervals (1991 ~ 1999 and 2000 ~ 2007). Figure 2 shows the average monthly runoff estimates for the two land use datasets. There is no significant difference between the two periods, but the 1999 land use data set was underestimated. Figure 3 also presents the sediment loads for the two periods using the two land use periods. It can also be seen that SWAT configurations using 1999 land use data underestimated the sediment loading from agricultural fields. The SWAT configurations using the 2011 dataset also overpredicted TP leaving the watershed as shown in Figure 4 for the 1999 and 2011 land use datasets, respectively.

Some form of consistency appears when using the Region_5 configurations (Region_5_2011 and Region_5_ 1999) for both land use periods (2011 and 1999). Their accuracy is always at par with the Reference sets (Reference_2011 and

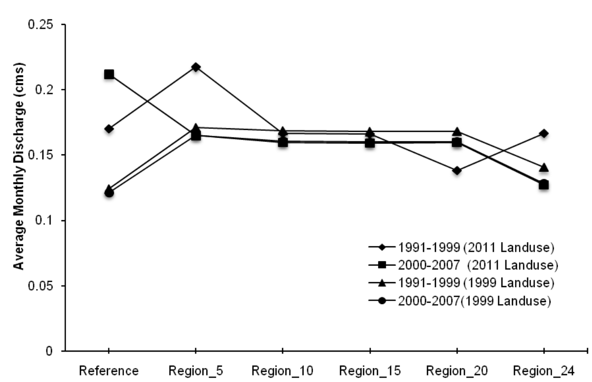

Figure 2. Plot of average monthly discharge for SWAT configurations using the 1999 and 2011 land sue datasets.

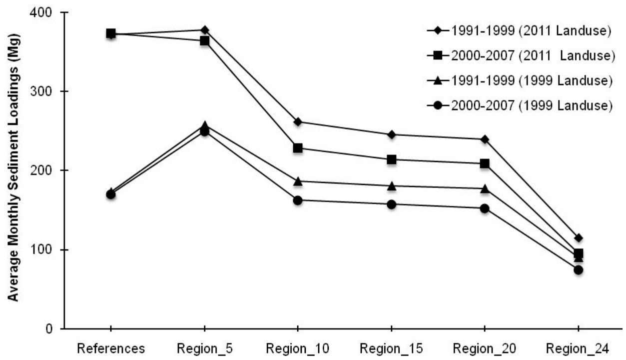

Figure 3. Plot of average monthly sediment loading for SWAT configurations using the 1999 and 2011 land use datasets.

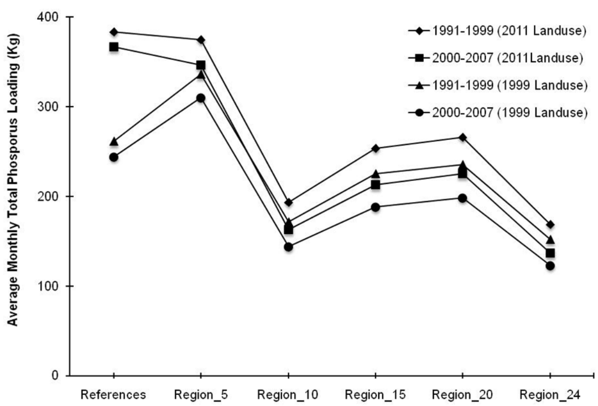

Figure 4. Plot of average monthly phosphorus loading for SWAT configurations using the 1999 and 2011 land use datasets.

Reference_1999). During the regionalization, this configuration (Region_5) has the highest within region heterogeneity, and the largest units averaged together. Drawing inference from this may be subjective, but it seems that having many regions (less within-region heterogeneity) does not necessarily increase prediction accuracy.

4. Conclusions

Many hydrologic modeling results are based on a static landscape, whereas in reality farmers rotate their crops. The objective of the study was to quantify the impact of soil heterogeneities on land use changes (e.g., 1999 vs. 2011). Regio nalized clustered datasets were used to evaluate the impact of soil heterogeneities. Between 1999 and 2011 an increase in corn cropping and a decrease in pasture land occurred in the Castor watershed.

The soils data set was partitioned and regionalized by creating soil maps having 5, 10, 15, 20 to 24 heterogeneous regions. Furthermore, prior modeling activities performed in this watershed have usually used average soil properties. An attempt was made to establish how much information was gained by doing field measurements. Therefore an additional SWAT configuration using average properties soil data was tested. The six SWAT configurations were tested with each of the land use datasets (1999 and 2011), resulted in a total of twelve SWAT configurations being tested.

There were no marked differences in predictions of runoff, which can be attributable to the use SCS-CN in SWAT. As mentioned previously (Section 3.3.1), this masks or covers up the physical properties of soil parameters within different soil hydrologic groups (Ye et al., 2009). As reported by Deslandes et al. (2007), underestimations of runoff occurred in the months of January to March, 2002. These underestimations are attributable to the exceptionally unseasonable near-zero and above zero (as high as 10 °C) temperatures that were recorded in the study area. Under such conditions SWAT has difficulty differentiating between rainfall and snowfall due to its weather generating subroutine.

For sediment, the SWAT configurations did not predict sediment loads very well; however, for TP loadings all the SWAT configurations performed well. For these water quality parameters, all the SWAT configurations captured the high flow loadings, but underestimate loadings during low flow periods. Sometimes discrepancies between measured and predicted values were likely based on temporal scale issues (Radcliffe et al., 2009).

Simulation results were evaluated for two periods, i.e., 1991 ~ 1999 and 2000 ~ 2007. All 1999 land use SWAT configurations underestimated runoff, while 2011 land use SWAT configurations gave higher values. For both sets of land use data, Region_5 (i.e., 5 regions during regionalization) gave the highest estimates of runoff, sediment and TP while being of a similar magnitude to the Reference SWAT configuration for either 1999 or 2011 land use data. This configuration has the highest within zone heterogeneity. Thse results indicate that having many sampling points do- es not necessarily increase the prediction accuracy.

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