WATER RESEARCH 46 (2012) 4750–4760

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The modified SWAT model for predicting fecal coliforms in the Wachusett Reservoir Watershed, USA

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ARTICLE INFO

Article history:
Received 22 November 2011
Received in revised form
18 May 2012
Accepted 27 May 2012
Available online 28 June 2012

Keywords:
Fecal coliform
SWAT model
Solar intensity
Wildlife

ABSTRACT

This study assessed fecal coliform contamination in the Wachusett Reservoir Watershed in Massachusetts, USA using Soil and Water Assessment Tool (SWAT) because bacteria are one of the major water quality parameters of concern. The bacteria subroutine in SWAT, considering in-stream bacteria die-off only, was modified in this study to include solar radiation-associated die-off and the contribution of wildlife. The result of sensitivity analysis demonstrates that solar radiation is one of the most significant fate factors of fecal coliform. A water temperature-associated function to represent the contribution of beaver activity in the watershed to fecal contamination improved prediction accuracy. The modified SWAT model provides an improved estimate of bacteria from the watershed. Our approach will be useful for simulating bacterial concentrations to provide predictive and reliable information of fecal contamination thus facilitating the implementation of effective watershed management.

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1. Introduction

Freshwater resources are susceptible to various fecal contaminations. Fecal coliform, also known as thermotolerant coliforms (Ashbolt et al., 2001), are a subgroup of total coliforms associated with fecal contamination. Most fecal coliform bacteria are not pathogenic, but they indicate the possibility of the presence of microbial pathogens, which are detrimental to public health (Noble et al., 2003). Fecal coliform sources include: agricultural runoff, sewage, and wild and domestic animal feces (Howell et al., 1995; Alderisio and DeLuca, 1999; Gerba, 2000; Guber et al., 2006; Servais et al., 2007; Cho et al., 2010a). Another potential source of fecal contamination is the release of fecal coliform from streambeds. Previous studies have demonstrated that sediments can contain one to three orders of magnitude more fecal coliform than the overlying water column (Goyal et al., 1977; Doyle et al., 1992; Buckley et al., 1998; Crabill et al., 1999; Smith et al., 2008; Rehmann and Soupir, 2009; Cho et al., 2010a; Pachepsky and Shelton, in press).

Several studies found that the levels of fecal coliform are substantially affected by meteorological conditions (Gannon

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http://dx.doi.org/10.1016/j.watres.2012.05.057

and Busse, 1989; Marsalek and Rochfort, 2004; Paul et al., 2004; Petersen et al., 2005; Tani et al., 1995; Kim et al., 2007, 2010; Cho et al., 2010b). Previous studies demonstrated that solar radiation has a substantial influence on the survival of fecal coliform in natural waters (Bellair et al., 1977; Mancini, 1978; Fujioka et al., 1981; McCambridge and McMeekin, 1981; Auer and Niehaus, 1993; Sinton et al., 1999, 2007; Cho et al., 2010b). Mancini (1978) found a strong relationship between die-off rates and solar radiation from *in-situ* studies. Solic and Krstulovic (1992) observed an increase in die-off rate in response to high intensity of solar radiation in laboratory experiments. Cho et al. (2010b) reported that in-stream levels of fecal coliform are sensitive to solar radiation and therefore the solar radiation intensity is one of control factors to model fecal coliform in surface waters.

Modeling, in conjunction with laboratory experiments and field observations, can help improve the understanding of the fate of fecal coliform in water. In addition, this approach can be used to provide predictive decision-support information for effective public health management. The Soil and Water Assessment Tool (SWAT), a widely used watershed model that operates on a continuous daily time-step, was further expanded by adding a bacteria module (Sadeghi and Arnold, 2002). Baffaut and Benson (2003) used the bacteria module to predict flow rates and fecal coliform concentration. Parajuli (2007) and Parajuli et al. (2009) applied this module to modeling nutrient and fecal coliform in two different sub-basins in Kansas. Coffey et al. (2010) used the original version of the SWAT to predict fecal coliform in Irish catchments, showing satisfactory prediction accuracies in the calibration step. In addition, the SWAT bacteria module was modified to consider streambed fecal coliform release and deposition (Kim et al., 2010). The effect of solar radiation on bacterial die-off, however, has not been considered in simulations of fecal coliform concentration in a water body.

The objectives of this study are (a) to develop and evaluate a solar radiation-associated bacteria module in the SWAT model and (b) to calibrate and validate the model to predict fecal coliform concentrations. This will provide an understanding of the fate of bacteria, which may lead to the implementation of more effective management decisions.

2. Materials and methods

2.1. Study area

The Wachusett Reservoir is one of the primary drinking water sources to the residents of the Greater Boston area in Massachusetts. The Stillwater River is one of the two largest tributaries of the Wachusett Reservoir and drains approximately 30% of the watershed (Fig. 1). The basin has a very low percentage of impervious cover (<10%) and high percentage of protected land with active watershed protection program in place.

2.2. Bacteria module in SWAT model

In this study, the upgraded SWAT bacterial module (Kim et al., 2010) was modified by adding the solar radiation-associated die-off. Fig. 2 shows the proposed in-stream bacteria algorithm in the SWAT model, including resuspension of fecal coliform and die-off by solar intensity. When current sediment concentration is less than the maximum sediment concentration, resuspension of sediment will occur. Otherwise, sediment deposition will occur. Total die-off is calculated by summing the base mortality, removal by deposition, and die-off due to solar radiation.

Previous studies had used solar intensity and temperature to estimate bacterial die-off (Mancini, 1978; Chapra, 2007; Cho et al., 2010b). The current version of SWAT does not include the effect of solar intensity on die-off rate. The bacteria in-stream module (rtbact.f) was modified by adding a new parameter (SOLLPCH) to observe the effect of solar intensity, and also to improve prediction accuracy. This parameter has been widely used to estimate the bacteria die-off by solar intensity (Mancini, 1978; Mayo, 1995; Canale et al., 1993; Xu et al., 2002; Cho et al., 2010b). The natural die-off rate is needed to calculate the total die-off rate. The natural, the die-off rate at 20 °C can be calculated by using Equation (1) below:

$$K = K + I(t) \cdot K_s. \tag{1}$$

where K_n is the natural die-off rate [d^{-1}], which indicates NATDIELP parameter in the model; $I(t)$ is the solar radiation [MJ m^{-2} d^{-1}]; and K_s is the solar radiation coefficient [m^2 MJ^{-1}], which represents SOLLPCH parameter in the model.

A first-order decay equation (Chick's law) is used to determine the amount of bacteria removed by die-off and regrowth as described by Sadeghi and Arnold (2002). The total die-off rate is estimated assuming that temperature is constantly 20 °C. Therefore, the total die-off rate is recalculated by using a temperature adjustment factor as follows:

$$C_t = C_0 e^{-KtA(T-20)} \tag{2}$$

where C_t is concentration at time t; C_0 is initial concentration; K is decay rate [d^{-1}]; t is time [days]; A is temperature adjustment factor [THBACT]; and T is temperature [°C].

When streambed sediments are resuspended, the amount of the released bacteria ($M_{B,res}$, CFU) is determined as follows:

$$M_{B,res} = M_{S,res} \cdot C_{B,B}. \tag{3}$$

where $M_{S,res}$ is the mass of resuspended sediment (ton); and $C_{B,B}$ is the fecal coliform concentration in streambed sediments (CFU/g sediment). Here, sediment bacteria concentration ($C_{B,B}$) has not been monitored in this watershed, but it was estimated by calibration process within a range of reported concentrations (Pachepsky and Shelton, 2011). The fecal coliform suspended in stream water ($M_{B,W}$) are divided into free-floating ($M_{B,free}$) and sediment-associated bacteria (Bai and Lung, 2005). The sediment associated fecal coliform are further divided into bacteria attached to the suspended sediments ($M_{B,sus}$) and fecal coliform attached to the deposited sediments ($M_{B,dep}$), as follows:

$$\frac{M_{B,free} + M_{B,sus} + M_{B,dep}}{M_{B,W}} = \frac{1 + K_p \cdot conc_{sed,sus} + K_p \cdot conc_{sed,dep}}{1 + K_p \cdot conc_{sed,i}}. \tag{4}$$

where $conc_{sed,sus}$ is the concentration of the suspended sediments; $conc_{sed,dep}$ is the concentration of the deposited

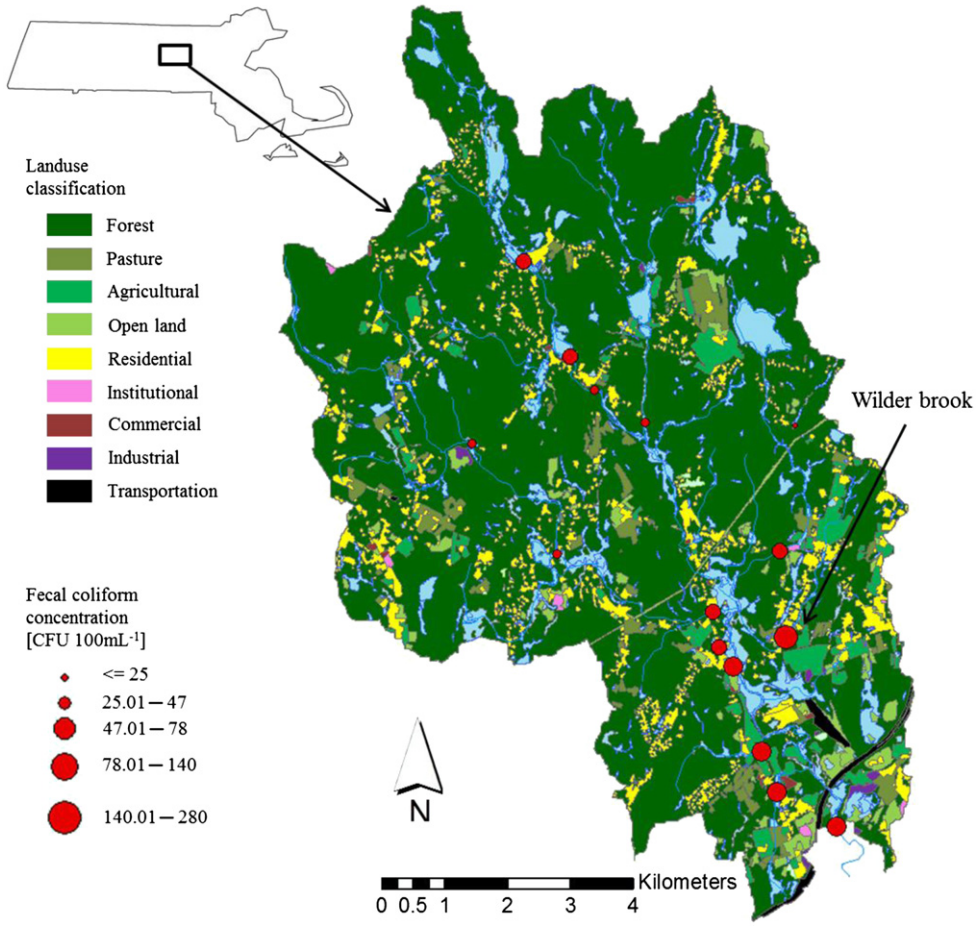

Fig. 1 – The land use and fecal coliform concentration in Stillwater River Basin.

sediments; $conc_{sed,i}$ is the initial concentration of sediment in a stream segment; and K_p is the partitioning coefficient of fecal coliform between the sediments and water (m^3/ton or mL/g). Detailed information on bacteria resuspension module can be found in Pachepsky et al. (2006b) and Kim et al. (2010).

Massachusetts Department of Conservation and Recreation (DCR, 2010) found a significant beaver population downstream of Stillwater River, which could be one of possible sources for relatively high fecal coliform and conductivity. Beaver impact on the water quality has been addressed in previous studies (Maret et al., 1991; Harthun, 2000). Skinner et al. (1991) indicated that variation of fecal coliform in a stream could be partially explained by beaver activity. Fecal coliform concentration could be affected by their feces and scoured bacteria from streambeds in the basin. The effect of beaver activity was simulated based on the assumption that beaver activity is more apparent in summer than that in winter, having a direct relation with temperature. A simple nonlinear equation was applied

$$FL = aA \times Tb. \quad (5)$$

where a and b are the coefficients; T is water temperature [°C]; and FL is fecal coliform loading due to beaver activity [CFU d^{-1}].

2.3. *Data acquisition*

GIS layers of digital elevation map (DEM), land use, streamlines, and ponds were acquired from Massachusetts Office of Geographic Information (MassGIS, http://www.mass.gov/mgis). The Soil Survey Geographic (SSURGO) database of the U.S. Department of Agriculture (USDA) was used to characterize the soil properties of the watershed (USDA, 2011). The layer was converted by SSURGO processing tool (Arc GIS extension) for the compatibility with the SWAT model (Sheshukov et al., 2009). Grazing operations were assumed to be one of the fecal contamination sources and livestock was assumed to be evenly distributed in all land area in the watershed. It is because that small size farms are randomly spread out within the watershed and that the SWAT model is a semi-distributed model which does not fully consider the spatial distribution of bacteria sources. A GIS layer associated with farm animal was obtained from MassDCR and used to estimate fecal contamination loading in the basin. (See supplementary information Table S1 that shows the total manure production and the number of fecal coliform in manure used in this study).

Daily measured data for precipitation, temperature, and wind speed at the Worcester Regional Airport Station were used in this study (NCDC, 2011; NESDIS, 2011). Since solar

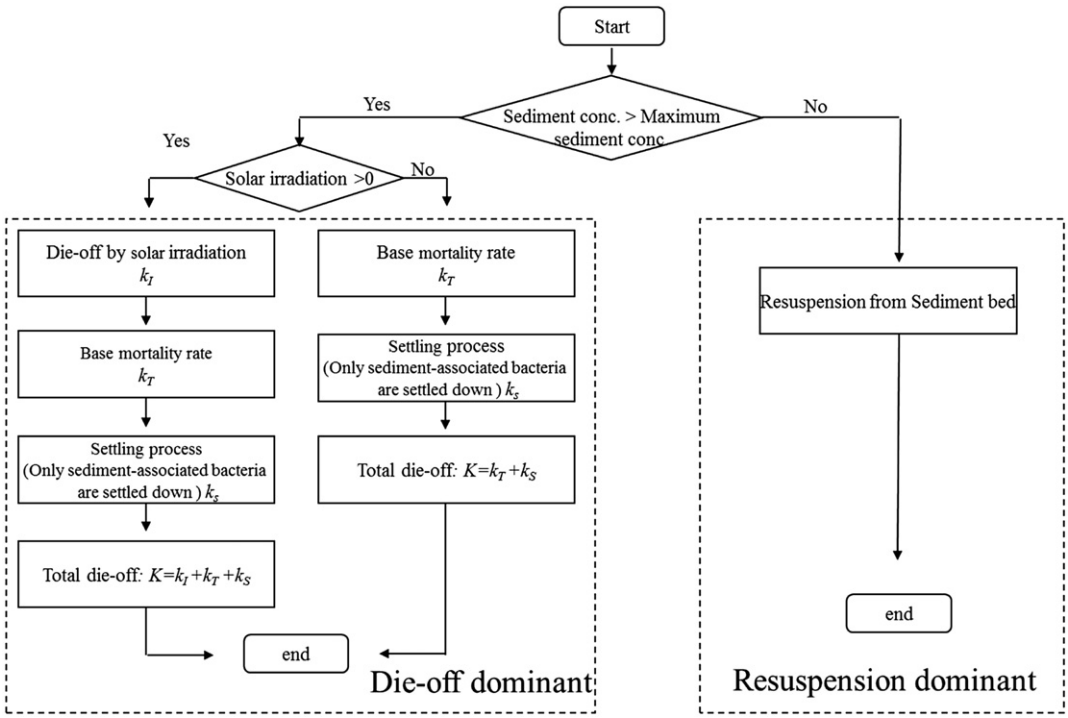

Fig. 2 – The proposed algorithm for determining resuspension, die-off by solar intensity, and natural die-off of fecal coliform.

radiation data were not available for the Worcester Regional Airport Station, data from Kingston station which is the closest (approximately 80 km distance) to the watershed were used in this study.

Fecal coliform sampling program has been conducted at 15 stations by MassDCR (2010), as shown in Fig. 1 (MassGIS). The 6-year data of flow rates, sediment, and fecal coliform from 2004 to 2009 were used to calibrate and validate the modified module of the SWAT. Flow rates from 2007 to 2009 were so varied that they were used to calibrate the module, whereas flow rates from 2005 to 2006 were used to validate the model. Data collected in 2004 were used for model stabilization (spin-up period). The Latin Hypercube-One-factor-At-Time (LH-OAT) method (Van Griensven et al., 2006) was used to investigate parameter sensitivity of the SWAT model. The objective function of sensitivity analysis was set to be the sum of squared errors (SSE) computed as differences between the observed and predicted flow rates. Ranking parameters from the sensitivity analysis makes it computationally efficient to calibrate the model with the most significant parameters only. The degree of sensitivity S is defined using the LH-OAT (see supplementary information). The auto-calibration tool embedded in SWAT model, which is an automatic shuffled complex evolution (SCE) optimization scheme, was applied to calibrate the hydrologic module (Van Griensven et al., 2006). The highly ranked parameters ($S > 0.1$) were used to calibrate the hydrologic module of SWAT within their default ranges, except for the watershed morphology parameters using flow measurement at the Stillwater monitoring station.

2.4. *Model evaluation*

Nash-Sutcliffe model efficiency coefficient (NSE) between the predicted and observed data (Nash and Sutcliffe, 1970) was used to evaluate model prediction accuracy with its criteria proposed by Moriasi et al. (2007). Furthermore, the Williams–Kloot test was used to compare the model performance for predicting fecal coliform concentration (Williams and Kloot, 1953). The test has been widely used to identify the best model between two (or more) models in various fields (Brassard and Correia, 1977; Pachepsky et al., 2006a) using the following equation:

$$\left[Y - \frac{1}{2}(\overline{Y}_1 + \overline{Y}_2)\right] = \lambda(\overline{Y}_2 - \overline{Y}_1). \tag{6}$$

where Y is the measurement, $\overline{Y}_1$ is the predicted valued by model 1 and $\overline{Y}_2$ is the predicted valued by model 2, and λ is the slope of this relation. If the slope λ is significantly different from zero and positive, it demonstrates that model 2 is better than model 1.

3. Result and discussion

3.1. *The relationship between land use and fecal coliform concentrations*

Table 1 shows land use in the basin, which is dominated by forest area (73.61%). This implies that the contribution of wildlife in the forest area could be significant for the fecal

Table 1 – Land use in the Stillwater River Basin.

Land use	Area [ha]	Percentage [%]
Forest	5637.11	73.61
Pasture	212.19	2.72
Agriculture	271.75	3.49
Open land	231.01	2.97
Residential	581.60	7.46
Institutional	12.66	7.46
Commercial	14.00	0.18
Industrial	17.41	0.22
Transportation	42.41	0.54
Water	674.10	8.65

Table 2 – Soil properties in the Stillwater River Basin.

Properties	Unit	Value
Moist Bulk density	[g cm^{-3}]	1.04
Available water capacity	[mm/mm]	0.18
Saturated hydraulic conductivity	[mm/h]	139.06
Sand	[%]	51.01
Silt	[%]	31.67
Clay	[%]	17.32

contamination in this watershed. The combined fraction of agricultural and pasture area is approximately 6%, which also can be fecal contamination sources in the watershed. Fig. 1 shows the spatial variation of fecal coliform in the watershed with land use information. Fecal coliform concentration downstream is greater than those upstream, which might be related to significant beaver population downstream of the Stillwater basin.

Table 2 shows soil texture and properties in the watershed. The dominant soil texture is coarse loamy with higher percentage of sand. It might result in low sediment bacteria concentration, relatively high decay rate of bacteria and low fraction of sediment-associated bacteria (Pachepsky, 2006b; Garzio-Hadzick et al., 2010; Pachepsky and Shelton, in press).

3.2. *Flow sensitivity and calibration*

Table 3 shows 27 hydrologic parameters with their ranges for the sensitivity analysis and calibration. The result shows that the CN2 (initial SCS runoff curve number for moisture condition II) was the most sensitive factor to affect the SWAT hydrologic cycle simulation for the watershed, and followed by CH_K2 (effective hydraulic conductivity in main channel alluvium), SURLAG (surface runoff lag coefficient), ALPHA_BF (baseflow alpha factor-baseflow recession constant) and

Table 3 – Sensitivity analysis results for hydrologic parameters.

Parameter	Min	Max	Rank	S	Value	Definition	Process
CN2	35	98	1	3.75E+00	66.87	Initial SCS runoff curve number for moisture condition II	Runoff
CH_K2	0	150	2	1.61E+00	146.29	Effective hydraulic conductivity in main channel alluvium (mm/hr)	Channel
ALPHA_BF	0	1	3	1.43E+00	0.97	Baseflow alpha factor-Baseflow recession constant	Runoff
SOL_Z	−50	50	4	5.19E-01	–	Depth from soil surface to bottom of layer (mm)	Groundwater
SMTMP	0	5	5	3.73E-01	0.00	Snow melt base temperature (°C)	Soil
TIMP	0.01	1	6	3.60E-01	0.87	Snow pack temperature lag factor	
SMFMN	0	10	7	2.78E-01	0.57	Melt factor for snow on December 21(mm H_2O/°C-day)	
SMFMX	0	10	8	2.64E-01	1.84	Melt factor for snow on June 21 (mm H_2O/°C-day)	
SFTMP	0	5	9	1.87E-01	1.51	Snowfall temperature (°C)	Channel
SURLAG	0	10	10	1.63E-01	0.13	Surface runoff lag coefficient	Evaporation
BIOMIX	0	1	11	1.39E-01	–	Biological mixing efficiency	Soil
CANMX	0	15	12	1.36E-01	–	Maximum canopy storage (mm H_2O)	Geomorphology
CH_N	−20	20	13	1.35E-01	–	Manning's "n" value	Runoff
SLOPE	−50	50	14	9.93E-02	–	Average slope steepness (m/m)	Geomorphology
ESCO	0	1	15	9.63E-02	–	Soil evaporation compensation factor	Soil
SOL_AWC	−50	50	16	9.56E-02	–	Available water capacity of the soil layer (mm H_2O/mm soil)	Soil
SOL_ALB	0	1	17	4.94E-02	–	Moist soil albedo	Evaporation
SOL_K	−50	50	18	4.58E-02	–	Saturated hydraulic conductivity (mm/hr)	Snow
GWQMN	0	5000	19	4.32E-02	–	Threshold depth of water in the shallow aquifer for return flow (mm H_2O)	Soil
SLSUBBSN	−50	50	20	1.60E-02	–	Average slope length (m)	Snow
RCHRG_DP	0	1	21	1.32E-02	–	Deep aquifer percolation fraction	Groundwater
EPCO	−50	50	22	9.78E-03	–	Plant uptake compensation factor	Snow
GW_REVAP	0.02	0.2	23	1.56E-03	–	Groundwater "revap" coefficient	Snow
GW_DELAY	0	100	24	5.80E-04	–	Groundwater delay time (days)	Groundwater
REVAPMN	0	500	25	1.91E-04	–	Threshold depth of water in the shallow aquifer for percolation to the deep aquifer (mm H_2O)	Evaporation
TLAPS	−50	50	26	0.00E+00	–	Temperature lapse rate (°C/km)	Snow
BLAI	−50	50	27	0.00E+00	–	Maximum potential leaf area index	Groundwater

SOL_Z (depth from soil surface to bottom of layer). The sensitivity ranking of hydrologic parameters in this study agreed with the sensitivity ranking in previous studies in that the curve number CN2 was usually ranked as either the first or the second (Arnold and Allen, 1996; Srinivasan et al., 1998; Spruill et al., 2000; White and Chaubey, 2005; Holvoet et al., 2005; Van Griensven et al., 2006; Kim et al., 2010).

Predicted and observed flow rates in both the calibration (2007–2009) and validation step (2005–2006) are compared in Fig. 3. The results show acceptable prediction accuracy for modeling flow rates (NSE > 0.5, Moriasi et al., 2007). However, the model underestimated the peak flow values. The SWAT model is limited when simulating the high peak flow (Kirsch et al., 2002; Chu et al., 2004) because it only simulates daily flow rates, when actual peak flows occur within hours (Atkinson et al., 2003).

3.3. *Sediment sensitivity and calibration*

Table 4 shows the results of sensitivity analysis for sediment modeling. CH_COV (Channel cover factor) and CH_EROD (Channel erodibility factor) were identified as the first and fourth sensitive parameters, respectively. Their contributions were more significant than previous study by Kim et al. (2010, 16th and 17th respectively). These parameters are related to the amount of sediment re-entrained to the stream, implying that resuspension of sediment highly contributes to sediment loading in the watershed. In addition, SPCON (Linear parameter for calculating the channel sediment routing) and SPEXP (Exponent parameter for calculating the channel sediment routing) were ranked the second and the fifth, respectively, showing a strong contribution to sediment modeling. These parameters are also indirectly associated with the re-entrainment of sediment.

Sediment dynamics provides useful information for modeling fecal coliform (Bai and Lung, 2005). Particularly, resuspension and deposition of fecal coliform are directly associated with sediment. The amount of resuspension and deposition of sediment was calculated using SWAT in order to predict resuspension and deposition of fecal coliform. (See supplementary information Figure S1 that compares the predicted and observed sediment concentrations in both the calibration and validation steps). The simulated concentration slightly underestimated the sediment observation. Very low levels of sediment were measured during the observation period, even during rainfall events. Fecal coliform concentration, however, showed relatively high levels up to 4000 cfu 100 mL^{-1} and fluctuated significantly, as shown in Figs. 4 and 5. This difference between sediment and bacteria implies that the proportion of the particle-associated fecal coliform is low in the watershed, which might be due to the coarse soil texture (Garzio-Hadzick et al., 2010). In addition, a series of dams constructed downstream has slowed the flow of water and resulted in the deposition of large quantities of sand and silt (MassDCR, 2010) because they tend to increase residence time of the water body and decrease the stream velocity. As shown in Table 2, the combined total percentage of silt and sand is approximately 83%, implying low sediment bacteria concentrations with high decay rates (Garzio-Hadzick et al., 2010). Coarse sediments may not provide sufficient protection from the environment to allow the persistence of a substantial concentration of bacteria (Pachepsky and Shelton, in press). In addition, the coarseness of sediment texture is directly related to a low partition coefficient of fecal coliform between sediment and water as it has been shown for soils (Pachepsky et al., 2006b). Mahler et al. (2000) have reported 5–100% attachment rates of fecal coliform and enterococci to particles in a karst area, showing a high variability. It is well known, however, that clay-dominant soil results in higher partition coefficient of fecal coliform to sediment (Pachepsky et al., 2006b). Consequently, sand-dominated soil texture in the watershed may result in poor bacterial habitat and low partitioning coefficient of bacteria to sediment.

3.4. *Fecal coliform sensitivity and calibration*

The result of sensitivities of bacteria-associated parameters was similar to the result of sediment because parameters ranked from first to fourth were identical for both. However,

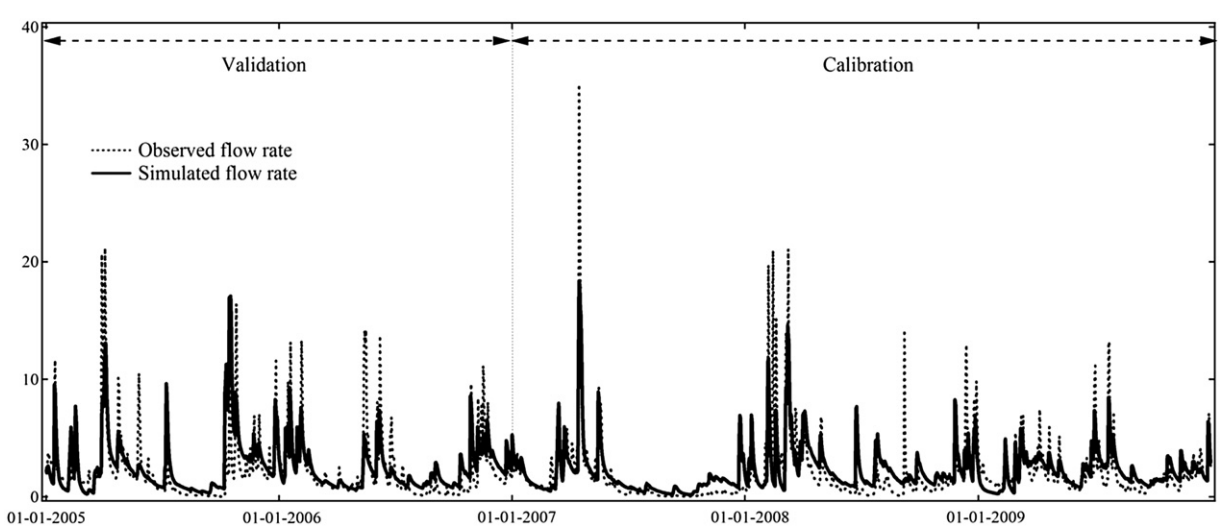

Fig. 3 – Observed and simulated daily flow rates for model calibration and validation.

Table 4 – Sensitivity analysis results for bacteria and sediment.

Parameter	Min	Max	Ranking		Value	Definition
			Sediment	Bacteria		
CH_COV	0	1	1	1	0.50	Channel cover factor
SPCON	0.001	0.01	2	2	0.11	Linear parameter for calculating the channel sediment routing
PRF	0	1	3	3	0.11	Peak rate adjustment factor for sediment routing in the main channel
CH_EROD	0	1	4	4	0.22	Channel erodibility factor
ADJ_PKR	0	1	–	–	1.00	Peak rate adjustment factor for sediment routing in the subbasin (tributary channels)
BACT_SWF	0	1	–	–	0.61	Fraction of manure applied to land areas that have active colony forming units
BACTKDDB	0	1	–	–	0.75	Bacteria partition coefficient in manure
BACTKDQ	150	200	–	–	166.14	Bacteria soil partitioning coefficient (m^3 mg^{-1})
BACTMIX	7	20	–	–	18.31	Bacteria percolation coefficient (m^3 mg^{-1})
BIO_EAT	10	100	11	16	46.34	Dry weight of biomass consumed daily [kg ha^{-1} day^{-1}]
BIO_INIT	10	500	10	14	134.68	Initial dry weight biomass [kg ha^{-1}]
BIO_MIN	10	300	8	9	33.00	Minimum plant biomass for grazing [kg ha^{-1}]
BIO_TRMP	10	100	–	15	51.49	Dry weight of biomass tramped daily [kg ha^{-1} day^{-1}]
CLAY	10	30	–	13	17.32	Percentage of clay in sediments [%]
FILTERW	0	2	7	10	0.50	Width of edge-of-field filter strip [m]
LAI_INIT	0	1	9	12	0.81	Initial leaf area index
NATDIELP	0	1	–	11	0.74	Natural die-off rate [1/day]
PHU_PLT	10	2000	–	–	747.41	Total number of heat units or growing degree days needed to bring plant to maturity [days]
SED_CON	10	10E3	–	5	50	Sediment bacteria concentration [CFU/g]
SPEXP	1	1.5	5	7	1.19	Exponent parameter for calculating the channel sediment routing
USLE_P	0.1	1	6	8	0.11	USLE equation support practice factor
SOLLPCH	0	1	–	6	0.99	Solar intensity coefficient [m^2/MJ/day]
WOF_LP	0	1	–	–	0.15	Wash-off fraction for bacteria

sediment bacteria concentration was ranked the fifth, which implies that bacteria released from streambeds were significant in this modeling despite its uncertainties. Pachepsky and Shelton (in press) reported large variations in sediment bacteria concentrations from different water bodies such as lakes, rivers and streams. Generally, the variation of sediment bacteria concentration was 2–3 times greater than the variation of bacteria in water column. SOLLPCH, newly added to the SWAT model in order to consider the effect of solar intensity on the die-off, was ranked the sixth, while NATDIELP was ranked the eleventh. This result shows that the effect of solar intensity on bacteria survival was more significant than natural die-off, verifying our hypothesis on the SWAT bacteria module.

The modified bacteria module has been calibrated to predict fecal coliform bacteria. Fig. 4(a) compares predicted and observed fecal coliform concentrations for both calibration and validation steps without considering beaver activity. The prediction accuracy of the module is not satisfactory for bacteria, showing reduced RMSEs in both calibration and validation steps (Table 5). The module well predicts low bacteria concentrations but shows a limitation to reproduce peaks in bacteria concentrations. Fig. 4(b) compares the observed and predicted fecal coliform bacteria, considering the contribution of wildlife. The model captured the seasonal variation of measured bacteria concentration although the accuracy of the model was not substantially improved in terms of the estimated NSE value. In addition, the model did not show significant impact of non-point sources related to stormwater runoff. Fig. 5 illustrates the relationships between simulated and observed bacteria, flow rate, and water temperature, which explain why the module had a limitation to reproduce the peak bacteria. Simulated bacteria concentrations were related to flow rates because storm events resulted in the elevated fecal coliform concentrations (Kim et al., 2010). Observed bacteria, however, show a better correlation with water temperature rather than with flow rates, indicating the annual periodicity during the observation period. MassDCR (2010) reported that fecal coliform concentrations had been relatively low during the winter and early spring; increased during the late spring and summer; and remained elevated concentrations through the fall. This could not be explained by either surface runoff or resuspension induced by rainfall intensity. Resuspension is not commonly occurring during baseflow and no correlation was reported between the levels of coliforms in sediments and in water column during baseflow (Pachepsky and Shelton, in press). Therefore, the elevated concentrations of fecal coliforms in this study are likely to be explained by the influx from wildlife or animal.

Beaver activity had been reported to be one of possible fecal coliform sources in the watershed (Jellison et al., 2002; MassDCR, 2010). Jellison et al. (2002) applied microbial source tracking to identify the sources of fecal contamination and found that the contamination was mainly caused by wildlife and dairy farming especially in the summer. Another source could be groundwater contamination from failed septic systems (Tache, 2009; MassDCR, 2010). The temporal pattern

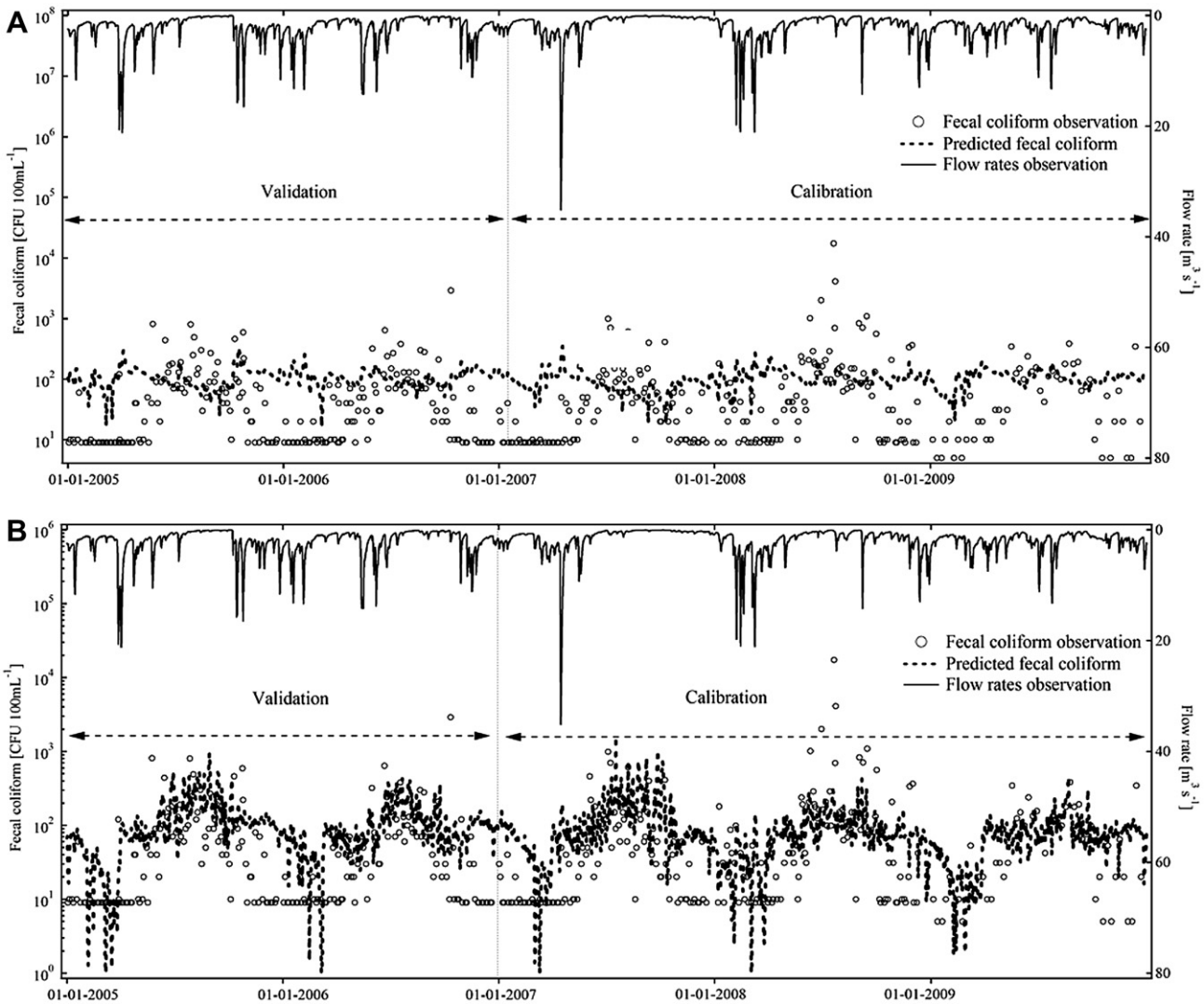

Fig. 4 – Observed and simulated values of fecal coliform for model calibration and validation; (A) without the temperature-associated function, (B) with the temperature-associated function.

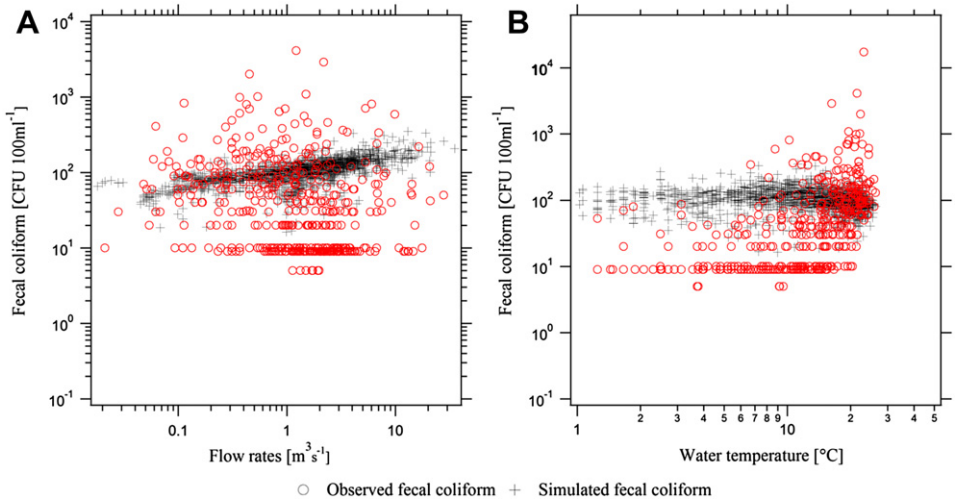

Fig. 5 – Relationships among flow rates, water temperature and observed and simulated fecal coliforms. Note that the simulated fecal coliform does not include the contribution of wildlife.

Table 5 – Accuracy results of SWAT simulation.

	Flow rate		Sediment	Fecal coliform without wildlife	Fecal coliform with wildlife
	RMSE [$m^3 s^{-1}$]	NSE [–]	RMSE [$mg L^{-1}$]	RMSE·10^2 [CFU]	RMSE·10^2 [CFU]
Calibration	1.73	0.57	2.50	3.44	1.15
Validation	1.83	0.53	3.08	2.61	0.86

Note: that RMSE stands for Root-mean-squared error and NSE for Nash-Sutcliffe model efficiency.

of the observed fecal coliform can be explained by the fact that continuous bacteria input from groundwater could be diluted by precipitation. Tache (2009) found that human-associated bacteria in the Gate Brook mostly came from residential areas. However, our result did not show correlation between residential areas and fecal contamination. Figure S2 (See Supplementary information) shows the result of the Williams–Kloot test to identify the best model to predict fecal coliform. The slope of the regression is 0.439 with an error of 0.020; the probability that the slope is not different from zero is 0.001, which means that the hypothesis that the model was enhanced by adding the temperature-associated function cannot be rejected at the significance level of 0.001. Consequently, the statistics clearly shows the improvement after introducing the temperature dependencies (Eq. (5)).

Previous SWAT applications to fecal coliform shows prediction accuracies ranged from −0.70 to 0.73 (Table S2 in supplementary information; Baffaut and Sadeghi, 2009; Chin et al., 2009). The NSE value from Coffey et al. (2010) was 0.59 using relatively small number of bacteria observations. Most of the previous studies on modeling bacteria have been conducted in pasture/grassland dominated watersheds. For example, the pastures, grasslands and agricultural areas were 86% and 85–93% in Coffey et al. (2010)'s study watershed and Parajuli et al. (2009)'s study watersheds, respectively. Our study area consists mostly of forest and thereby the contribution of wildlife could be greater compared to previous studies. Parajuli et al. (2009) used animal road kill data to estimate the contribution of wildlife to fecal coliform. This method can only be applied to certain watersheds with a high portion of transportation area without considering seasonal behavior of wildlife. Kim et al. (2010) discussed insufficient information of wildlife in a forest and pasture dominant watershed, Pennsylvania. Therefore, the contribution of wildlife and its seasonal effect should be considered in order to accurately estimate fecal contamination.

4. Conclusions

This study provides a modeling approach to quantify sources and sink of fecal contamination, which can provide predictive information of microbial water quality. The modified SWAT model considering the effect of solar intensity and wildlife was applied to predict fecal coliform. The major findings in this study are as follows:

1) The SWAT model showed an acceptable accuracy to predict flow rates in both the calibration and validation steps, showing satisfactory NSE values.
2) Very low sediment concentrations found in both the model predictions and measurements during the observation period could be resulted from the coarse soil texture in the watershed.
3) The sensitivity analysis results show the significant effect of solar intensity on bacteria survival
4) The model prediction was improved by considering wildlife contribution. This emphasizes the significance of wildlife contribution to model fecal coliform in this watershed. For more accurate simulations, sufficient information of wildlife activity such as the number and the location of beaver activity is needed to quantify its effect on fecal contamination.

The SWAT model can generally provide reliable simulation of flow and pollution from a watershed and however it could be improved by considering wildlife contribution to fecal contamination. It could be improved by incorporating wildlife database in the SWAT simulation for fecal coliform bacteria. This study shows that modeling bacteria still suffers from substantial uncertainty and insufficient information. This study will provide predictive and scientific information for improving surface water quality.

Acknowledgments

This study was funded by Massachusetts Department of Conservation and Recreation (MassDCR). The authors would like to thank Ms. Patricia Austin, Mr. Lawrence Pistrang and Mr. Craig F. Fitzgerald at MassDCR for sharing data and their review.

Appendix A. Supplementary material

Supplementary data related to this article can be found online at http://dx.doi.org/10.1016/j.watres.2012.05.057.

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