Uncertainty Evaluation of Coliform Bacteria Removal from Vegetated Filter Strip under Overland Flow Condition

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Vegetated filter strips (VFS) have become an important component of water quality improvement by reducing sediment and nutrients transport to surface water. This management practice is also beneficial for controlling manure-borne pathogen transport to surface water. The objective of this work was to assess the VFS efficiency and evaluate the uncertainty in predicting the microbial pollutant removal from overland flow in VFS. We used the kinematic wave overland flow model as implemented in KINEROS2 coupled with the convective-dispersive overland transport model which accounts for the reversible attachment-detachment and surface straining of infiltrating bacteria. The model was successfully calibrated with experimental data obtained from a series of simulated rainfall experiments at vegetated and bare sandy loam and clay loam plots, where fecal coliforms were released from manure slurry applied on the top of the plots. The calibrated model was then used to assess the sensitivity of the VFS efficiency to the model parameters, rainfall duration, and intensity for a case study with a 6-m VFS placed at the edge of 200-m long field. The Monte Carlo simulations were also performed to evaluate the uncertainty associated with the VFS efficiency given the uncertainty in the model parameters and key inputs. The VFS efficiency was found to be <95% in 25%, <75% in 23%, and <25% in 20% of cases. Relatively long high-intensity rainfalls, low hydraulic conductivities, low net capillary drives of soil, and high soil moisture contents before rainfalls caused the partial failure of VFS to retain coliforms from the infiltration excess runoff.

Microbial pathogens rank first and second among five leading pollutants in estuaries and rivers, respectively, in the United States by the frequency of being the cause of water quality impairment (EPA, 2004). Vegetation filter strips are among the best management practices commonly used to decrease the pollutant loads from agricultural fields and pastures, where animal manures are being applied. A wide range of contradicting opinions exists on the VFS efficiency and function with respect to pathogens and/or indicator organisms removal (Pachepsky et al., 2006). Initial management tools to evaluate the efficiency of VFS and select its VFS parameters with respect to manure-borne pathogens removal have been developed in the 1980s. They include: Agricultural Runoff Management II, Animal Waste Version (ARM II) model (Overcash et al., 1983); Utah State (UTAH) model (Springer et al., 1983); MWASTE model (Moore et al., 1988); and COLI model (Walker et al., 1990).

Recent interest with regard to the fate and transport of manure-borne pathogens has generated a substantial increase in monitoring data on the fate and transport of pathogens and indicator organisms under VFS conditions. Several excellent reviews have been published (Jamieson et al., 2002; Ferguson et al., 2003; Tyrrel and Quinton, 2003; Unc and Goss, 2004; Oliver et al., 2005). The existing knowledge base shows that the efficiency of VFS, as barriers for pollutants, depends to a large extent on the soil infiltration capacity in the VFS, antecedent soil moisture (soil moisture content before the rain event), vegetation status and microtopography, and rainfall intensity and duration (Muñoz-Carpena et al., 1999; Helmers et al., 2006). The spatial scale of the available information about VFS's soil and vegetation properties, and its management usually does not warrant details of soil properties for specific VFS locations. This information along with the natural variability in weather patterns creates an uncertainty in the efficiency of VFS for microbial pollutant removal.

The objective of this work was to use monitoring and modeling approaches for evaluating the uncertainty in the VFS efficiency in

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Abbreviations: CFU, colony forming units; FC, fecal coliform; PDF, probability distribution function; VFS, vegetated filter strip.

microbial pollutant removal from the overland flow as related to variability in the key parameters controlling this efficiency. We used the overland flow and transport model that we had previously calibrated and tested with our experimental data on microbial transport in runoff for this uncertainty assessment.

Materials and Methods

Modeling Overland Transport and Removal of Bacteria

Several models for simulating transport of dissolved chemicals by the overland flow have been developed during past decades (Havis et al., 1992; Wallach et al., 2001). The overland flow component in these and other models is usually well described by the kinematic wave equation (Woolhiser et al., 1990),

$$\frac{\partial h}{\partial t}+\frac{\partial q}{\partial x}=R-I \qquad [1]$$

where h is the depth of ponding (cm), q is the water discharge per unit width (cm^2 h^{-1}), R is the precipitation rate (cm h^{-1}), I is the infiltration rate (cm h^{-1}), t is time (h), and x is the distance along the slope (cm).

The depth-discharge relationship used is normally expressed as

$$q=\alpha h^{5/3} \qquad [2]$$

where the parameter α is computed according the Manning hydraulic resistance law as

$$\alpha=n^{-1}S^{1/2} \qquad [3]$$

where S is the surface slope, (cm cm^{-1}) and n is the Manning's roughness coefficient for overland flow.

The Parlange et al. (1982) equation is used for calculating infiltration rate through the soil surface

$$I=K_s\left[1+\frac{\sigma}{\exp\left(\sigma I^*/B\right)-1}\right] \qquad [4]$$

where K_s is the saturated hydraulic conductivity (cm h^{-1}), σ is the dimensionless parameter that represents the soil type, I^* is the infiltrated depth (cm), $B = (G + h)(\theta_s - \theta_i)$, G is the net capillary drive (cm), θ_s and θ_i are the soil water contents at saturation and initial condition (cm^3 cm^{-3}), respectively.

The mass balance of the overland movement of bacteria has the form

$$\frac{\partial hC}{\partial t}+\frac{\partial S_s}{\partial t}+\frac{\partial qC}{\partial x}=\frac{\partial}{\partial x}\left(a_L q\frac{\partial C}{\partial x}\right)+k\left(C_m-C\right)-\left(1-k_{str}\right)IC+RC_R \qquad [5]$$

where C and C_m are the cell concentration in runoff water and soil mixing zone (cell cm^{-3}), respectively, S_s is the cell concentration in the manure applied to the soil surface (cell cm^{-2}), a_L is the dispersivity (cm), k is the rate of mass transfer (cm h^{-1}), k_{str} is the straining coefficient, and C_R is the cell concentration in rainwater (cell cm^{-3}).

The first and the second terms in the left-hand side of (5) describe change of bacteria concentration in the overland flow and the surface-applied manure, while the third term accounts for the bacteria advection with the velocity calculated by (2). In the right-hand side, the first term describes the bacteria dispersion assuming its linear dependence on the advective flux; the second term accounts for the exchange of microorganisms between runoff water and the mixing zone of soil at the surface; the third term simulates straining of microorganisms from the infiltrating water by plant litter and vegetation layer (assumed to be proportional to the infiltration rate I, the coefficient k_{str} $0 \le k_{str} \le 1$ is a lumped parameter that accounts for the rate of straining, $k_{str} = 0$ no straining occurs); and the last term describes influx of bacteria, if present, from rain or irrigation water.

The irreversible release of bacteria from the surface applied manure is described as

$$\frac{\partial S_s}{\partial t}=-C_sR \qquad [6]$$

where C_s is the concentration of released bacteria calculated according Bradford and Schijven (2002) as

$$C_s(t)=\frac{dM_w}{Rdt}=\frac{M_0\alpha_m}{R}(1+\alpha_m\beta_m t)^{-(1+1/\beta_m)} \qquad [7]$$

where M_w is the cumulative manure mass released into the aqueous phase (g), α_m (h^{-1}) and β_m (dimensionless) are fitting parameters defining the shape of the release curve, and M_0 is the initial content of bacteria in the applied manure (cell cm^{-2}).

The mass balance equations of bacteria transport in the mixing zone of the topsoil are computed as:

$$d\frac{\partial\left(\theta C_m\right)}{\partial t}+\rho d\frac{\partial S_m}{\partial t}=-k\left(C_m-C\right)+\left(1-k_{str}\right)IC \qquad [8]$$

$$\rho\frac{\partial S_m}{\partial t}=\theta k_{am}C_m-k_{dm}\rho S_m \qquad [9]$$

where S_m is the cell concentration on the solid phase of soil mixing zone (cell g^{-1}), k_{am} and k_{dm} are the attachment and detachment rates of bacteria at the solid phase (h^{-1}), respectively, ρ is the topsoil bulk density (g cm^{-3}), and d is the thickness of the mixing zone, that is, topsoil layer that actively interacts with the overland flow (cm).

First and second terms in the left-hand side of Eq. [8] describe change of bacteria concentration in the liquid and solid phases of the mixing zone, respectively; first and second term in the right-hand side account for exchange of microorganisms with the overland flow and their influx with infiltrated water, respectively. Equation [9] describes reversible attachment and detachment of bacteria (first and second term in the right-hand side, respectively) to the solid phase in the mixing zone.

The set of equations [1–9] was solved numerically using the implicit finite different scheme. Initial and boundary conditions were set as described below. The KINEROS2 numerical code (Woolhiser et al., 1990) was used to solve the flow equation. The front limitation algorithm (Haefner et al., 1997) was applied for solving the solute transport equations.

Experimental Methodology

Overland flow experiments with the manure-borne fecal coliforms were performed in October 2003 on a two-sided lysimeter of 21.34 m long and 13.2 m wide located at the Patuxent Wildlife Research Refuge (Beltsville, MD). The slope of the lysimeter surface was 20%. A detailed description of the lysimeter is given in Roodsari et al. (2005). The lysimeter soil texture is clay loam on one side and sandy loam on the other side. Approximately half of the areas on each side of the lysimeter were planted with blue fescue (*Festuca ovina* L. '*Glauca*') and white clover (*Trifolium repens* L.) mixture while the rest of the area maintained bare. Twelve plots of 2 m width and 6 m length were established on both sides of the lysimeter, in the bare and the vegetated areas, resulting in total of four treatments: vegetated clay loam, bare clay loam, vegetated sandy loam, and bare sandy loam. All plots were isolated with metal plates (10 cm width), inserted into the soil to a depth of about 5 cm, to prevent cross contamination. Rainfall was applied at water pressure of 100 kPa using a rainfall simulator equipped with four TeeJet 1/4 HH SS 14 WSQ nozzles (Spraying Systems Co., Wheaton, IL) at 3 m above the soil surface. The average simulated rainfall rate measured was 5.8 ± 1.9 cm h^{-1}. The vegetated plots were mowed 1 wk before the experiment to maintain uniform grass height of about 5 cm. Plots were irrigated for 30 min with simulated rainfall 1 d before the experiment at the same rate as in the runoff study. Manure slurry was collected from the dairy barn at the Dairy Research Unit at the USDA/ARS-Beltsville Agricultural Research Center (BARC), Beltsville, MD. Some selected properties of manure are provided in Table 1. Soil water content was measured by taking gravimetric samples in triplicates from each plot at 0 to 5, 5 to 10, 10 to 15, and 15 to 20 cm depth before the manure application (Table 1). The manure slurry was uniformly applied on the top of the plots in a 30-cm wide strip at the rate of 11.7 L m^{-2}. The uniformity was ensured by placing a wire mesh on top of subplots and pouring the predetermined amount of the manure to each of the 5- by 10-cm mesh cells. Irrigation was started immediately after manure application. Rainfall was simulated for 1 h after the initiation of runoff. Rainfall rates were measured with rain gauges installed adjacent to each subplot. Runoff samples were collected in troughs, located at the bottom of plots for 2 min, at 5-min intervals. The combination of soil properties and slope at the subplots ensured the unobstructed removal of the released manure constituents to the troughs.

Runoff volume and fecal coliform (FC) concentrations were measured for each runoff sample from each plot. FC concentrations were determined by plating 50-μL subsamples of runoff, after centrifugation at 100 × *g* to remove sediment, onto MacConkeys Agar using an Autoplate 4000 spiral platter (Spiral Biotech, Bethesda, MD). After overnight incubation at 44.5°C, pink to red colonies typical of FC were counted using a Protocol plate reader (Synoptics, Cambridge, UK). The FC counts were expressed as colony forming units (CFU).

Model Calibration

The overland flow and bacteria transport model (Eq. [1–9]) was calibrated for each plot by solving the inverse problem with the PEST software (Doherty, 2005), which is based on the least square minimization using the Levenberg-Marquardt algorithm. The overland flow component was calibrated on data of the runoff rate time series for each plot separately. The boundary condition of $h = 0$ at $x = 0$ was used. The Manning roughness coefficient n was set to 0.09 for the grass plots and 0.035 for bare plots (Knisel, 1980). The saturated hydraulic conductivity (K_s), net capillary drive (G), and the parameter σ were found by fitting Eq. [1–4] to the runoff experimental data.

The transport model was calibrated using FC breakthrough curves measured at the runoff plots outlet. We accounted for the bacteria and manure release through the boundary condition at the edge of the manure strip. For FC transport problem the boundary concentration was obtained by simulating kinetics of FC release from manure. The release of bacteria from the manure was studied at separate subplots during the same experiments (Guber et al., 2006). In this work, the release model of Bradford and Schijven (2002) was found to perform best for manure borne bacteria. No significant correlations between α_m and β_m were observed. Values of α_m were closely related to the irrigation rate R according to the linear regression equation

$$\alpha_m = 0.0036 + 0.860\,R,\ R^2 = 0.988 \qquad [10]$$

in the range of R from 2.51 to 6.93 cm h–[1]. Values of β_m varied widely, generally increased with the irrigation rate, and were markedly larger for vegetated plots than for bare plots (Guber et al., 2006). Thus, Eq. [7] was used to provide the boundary condition at $x = 30$ cm for the overland transport model (Eq. [6–9]). The Neumann boundary condition of zero concentration gradient was set at the bottom of the runoff collector. Other input parameters such as the thickness of the active soil layer d was set to 5 cm. Initial zero concentrations of FC were assign for this layer. The dispersivity (a_L), the rate of mass transfer (k), and the straining coefficient (k_{str}) for FC were fitting parameters.

Monte Carlo Simulations to Evaluate the Vegetated Filter Strip Efficiency

The VFS efficiency has been defined as the percentage of bacteria that were retained by the VFS.

The Latin Hypercube sampling technique was used to evaluate the uncertainty of the bacteria removal in the VFS. This method allows one to sample a large-dimension parameter space sparsely (Press et al., 1992). We considered the VFS efficiency being affected by nine parameters and input values of the model, namely, by duration of rainfall, t_R, rainfall intensity R, initial soil water content θ_i, the Manning coefficient, n, hydraulic conductivity, K_s, net capillary drive, G, bacteria release parameter, β_m, and dispersivity, a_L. Values of each parameter were defined at 20 levels of probability: 0.025, 0.075, 0.125, …, 0.925, 0.975. The probability distributions functions of parameters are defined below. The Latin Hypercube sampling was repeated 1000 times. In each sample, each model parameter was set at only one of 20 probability levels (Press et al., 1992). The FORTRAN code is available by request from the corresponding author.

Table 1. Selected properties of the manure, volumetric soil water content at depth of 0 to 20 cm prior to the irrigation, irrigation time, and rate used in the study.

Plot	Manure solid/liquid ratio	Fecal coliform content in the manure	Initial soil water content	Irrigation time
	g g^{-1}	10^6 CFU mL^{-1}	cm^3 cm^{-3}	h
Clay loam, vegetated	0.158	1.70	0.280 ± 0.035†	1.217
Clay loam, bare	0.082	1.71	0.227 ± 0.036	0.950
Sandy loam vegetated	0.079	0.67	0.234 ± 0.029	1.267
Sandy loam, bare	0.075	2.05	0.243 ± 0.014	1.029

† The " ± " sign separates the estimates of the average value and the standard error.

Results

Model Calibration Results

The hydraulic component of the model after the calibration performed satisfactorily is shown in Fig. 1 and in Table 2. Calibrated values of the hydraulic parameters are presented in Table 3. The average soil hydraulic conductivity at the vegetated plots was 3 to 10 times higher than at bare plots of the same soil texture. Values of the capillary drive parameter (G) for sandy loam varied from 0.1 to 6.4 cm, which was lower than those of clay loam (7.5–26.3 cm). The estimates of this parameter obtained using mean texture class parameters and Brooks and Corey water retention equation are 7.0 and 26 cm for sandy loam and clay loam, respectively (Woolhiser et al., 1990). Values of σ in Eq. [4] varied from 0.07 to 0.75. Smaller values of this parameter indicate that Eq. [4] approaches to the Green-Ampt approximation (Woolhiser et al., 1990).

The performance of the calibrated transport model is shown in Fig. 2. Results are generally satisfactory (Table 2) except the simulations of concentrations in the first portions of runoff where measured values of these concentrations are much larger than the modeled ones. The variability of values of model performance indicators at replicated grass plots was higher than at bare plots (Table 2).

The value of the mass transfer rate (k) converged to zero in all calibration runs. This means that for the conditions, when initial concentration of bacteria in the top soil layer is zero, the exchange between this layer and the overland flow is negligible compared to the amount transported with the infiltrating water. Therefore, we were not able to find the rates of FC attachment and detachment k_{am} and k_{dm}, since they did not affect the bacteria concentration in the overland flow phase, when infiltration dominated the mass exchange between overland and soil water. The values of the straining parameter (k_{str}) were found to be 0 for all plots except the sandy loam vegetated plot where k_{str} = 0.9, 0.96 and 0.94 for the three replicates A, B, and C, respectively. The calibrated dispersivity values (Table 3) varied in a relatively large range: from 0.1 to 51.9 cm.

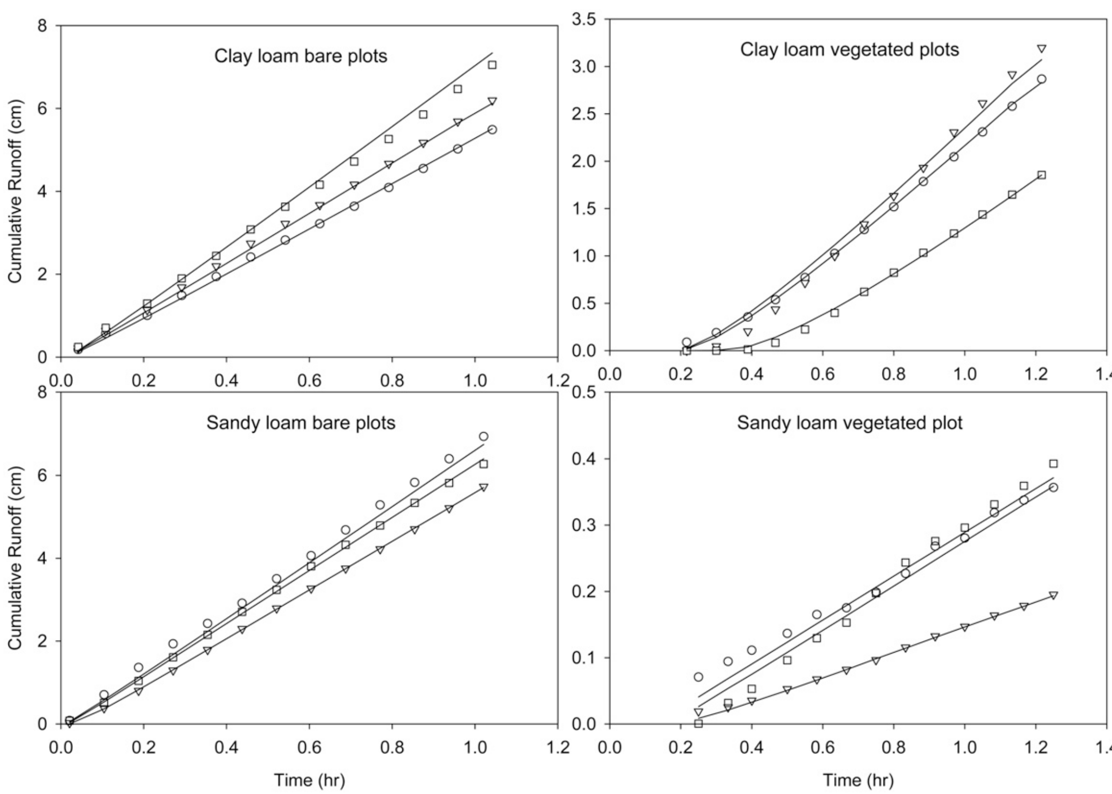

Fig. 1. Results of the runoff and infiltration module calibration. Different symbols show replications in measurements, lines show simulation results.

Table 2. Model performance statistics† from the model applications at slopes with different soil and vegetation cover.

Plot	R^2_{runoff}	R^2_{FC}	RMSE $_{Runoff}$ (cm)	$RMSE_{FC}$
Clay loam, bare	1.000 ± 0.000‡	0.926 ± 0.018	0.077 ± 0.033	0.017 ± 0.002
Clay loam, grass	0.999 ± 0.000	0.793 ± 0.051	0.055 ± 0.026	$(0.028 \pm 0.002)\cdot10^{-2}$
Sandy loam, bare	0.999 ± 0.000	0.552 ± 0.004	0.045 ± 0.014	0.015 ± 0.003
Sandy loam, grass	0.999 ± 0.000	0.623 ± 0.077	0.013 ± 0.006	$(0.028 \pm 0.014)\cdot10^{-2}$

† The determination coefficient, R^2, and the root-mean-squared error, RMSE, are given for the cumulative runoff and the relative fecal coliform (FC) concentration.

‡ The " ± " sign separates the average value and the standard error from three replications for each plot.

Table 3. Model parameters calibrated on data of the runoff plot experiments.

Plot	Replication	K_s	G	σ	a_L
		cm h^{-1}	cm		cm
Clay loam, vegetated	A	2.70	17.9	0.36	1.3
	B	1.96	26.1	0.07	1.4
	C	3.03	26.3	0.10	23.6
Clay loam, bare	A	0.03	10.0	0.21	23.1
	B	0.03	7.5	0.30	50.0
	C	0.10	9.4	0.12	51.9
Sandy loam, vegetated	A	7.33	0.1	0.16	23.6
	B	7.02	0.7	0.31	44.2
	C	7.00	0.7	0.08	7.0
Sandy loam, bare	A	0.08	6.4	0.75	26.6
	B	2.99	2.3	0.23	0.10
	C	2.60	1.0	0.06	0.10

Sensitivity of the Vegetated Filter Strip Efficiency to Model Parameters and Input Values

The effect of model parameter values on the simulated VFS efficiency was assessed in a set of simulations in which bacteria were released from manure applied on a 200-m long field and were moved with runoff water flowed through a 6-m long vegetated buffer strip set at a homogeneous clay loam soil profile. The base set of model parameters and initial and boundary values were as follows: the rainfall intensity was 2.5 cm h^{-1}; the rainfall duration was 1 h; the net capillary drive G = 26 cm; parameter σ = 0.85; the initial water content θ_i = 0.35 cm^3 cm^{-3}; the saturated hydraulic conductivity K_s = 1.45 cm h^{-1}; the Manning's n = 0.035 for the field and 0.090 for the vegetated buffer strip; the dispersivity a_L = 65 cm; the mass exchange rate k = 0; the straining parameter k_{str} = 0; and the thickness of the active soil layer d = 5 cm. Bacteria transport parameter values were chosen based on the calibration results for clay loam plots. The value of α_m was computed according to the Eq. [10] from the rainfall intensity value. The rainfall intensity was selected at 90% probability level for 1 h rainfall durations in Maryland conditions. The sensitivity for a given parameter was assessed by running the model with the parameter of interest varying in a reasonable range and all other parameters taken constant at or close to the base set values. Results of the simulations are shown in Fig. 3. The efficiency of the filter strip decreased after the rainfall intensity exceeded 2.3 cm h^{-1} (Fig. 3a). Long rainfall could exhaust the filtering capability of the VFS (Fig. 3b). High initial soil water contents caused a decrease in the VFS efficiency (Fig. 3c). The increase in the roughness of the soil

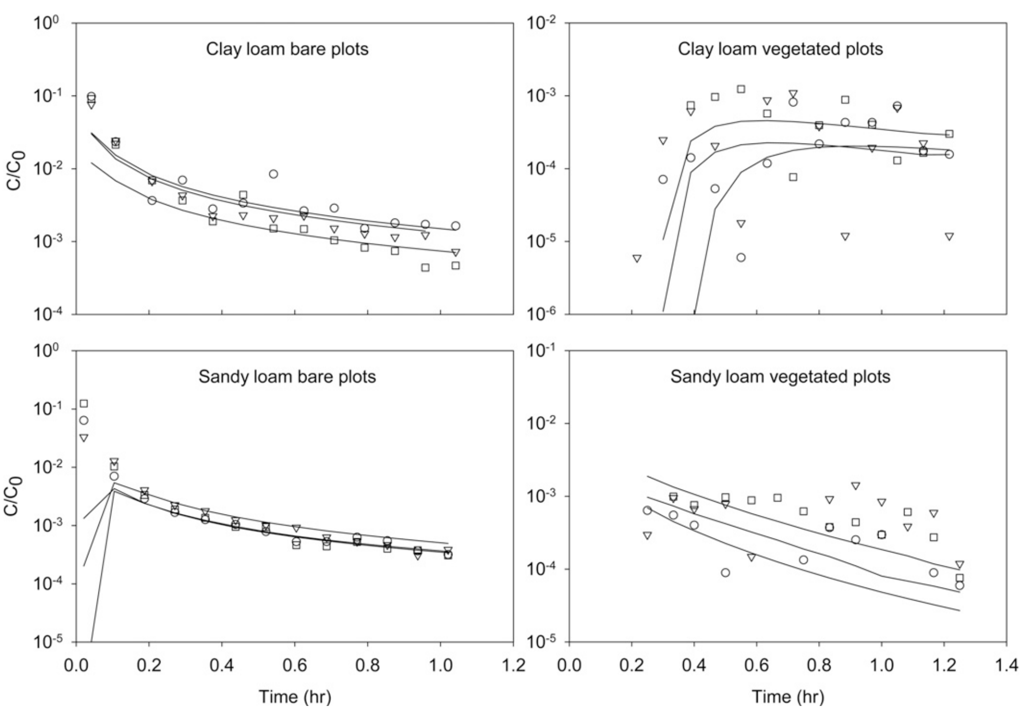

Fig. 2. Results of the bacteria transport module calibration. Different symbols show replications in measurements, lines show simulation results.

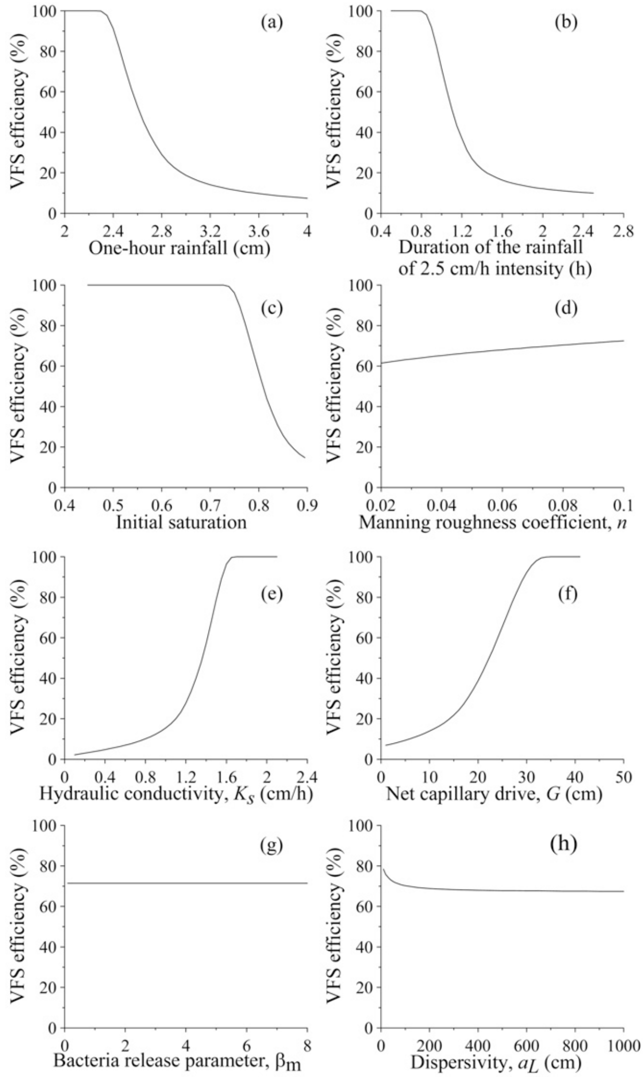

Fig. 3. Relationships between the vegetated filter strip efficiency and model parameters and rainfall characteristics.

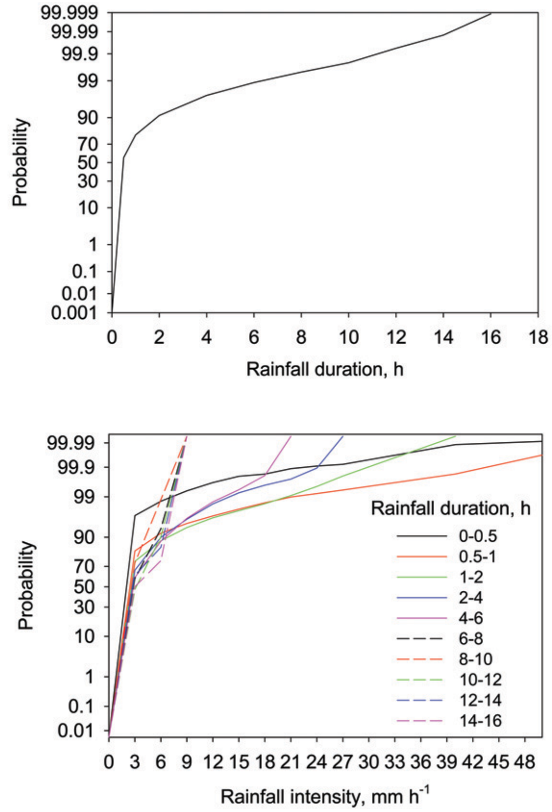

Fig. 4. Empirical probability distribution functions of rainfall durations and rainfall intensity for different rainfall durations.

surface in the filter strip slightly improved its efficiency (Fig. 3d). Low soil saturated hydraulic conductivity and/or small net capillary drive tended to decrease the efficiency (Fig. 3e and 3f). The changes in the bacteria release parameter β_m did not affect the efficiency of the strip (Fig. 3 g) probably because no appreciable retention in the soil active layer has occurred with the parameters used in simulations. Increase in dispersivity caused a small decrease in the strip efficiency (Fig. 3h).

Estimating Uncertainty in Model Inputs and Parameters

Probability distribution functions (PDFs) were defined for parameters and input values of the model: precipitation depth and intensity, initial water content, the Manning roughness coefficient, saturated hydraulic conductivity, net capillary drive, the bacteria release parameter, β_m, and the dispersivity. Probability distribution functions for the rainfall durations and intensities associated with each specific rainfall durations were developed from rainfall data collected for 11 yr with 15 min frequency at six automated weather stations across the ARS experimental station in Beltsville, MD (Fig. 4). Uniform probability distributions were used for the initial water contents (ranged from 0.2 to 0.4 cm^3 cm^{-3}) and for the Manning roughness coefficient (ranged from 0.05–0.10) to encompass published grassland values.

Statistical distributions of soil water retention and soil hydraulic conductivity parameters for soils of different textures have been summarized by Meyer et al. (1997) and by Rawls et al. (1998), respectively (Table 4). They found the normal PDF N(a,b)

$$f(x)=\frac{1}{\sqrt{2\pi}b}\exp\left[-\frac{1}{2}\left(\frac{x-a}{b}\right)^2\right],$$

lognormal PDF L(a,b)

$$f(x)=\frac{1}{\sqrt{2\pi}bx}\exp\left[-\frac{1}{2}\left(\frac{\ln(x)-a}{b}\right)^2\right]$$

and scaled β-distribution PDF B(a,b,A,B)

$$B(a,b,A,B)=\frac{\Gamma(a+b)}{\Gamma(a)\Gamma(b)}\frac{(x-A)^{a-1}(B-x)^{b-1}}{(B-A)^{a+b-1}}$$

should be used for hydraulic properties depending on the soil textural class (Table 4). In the latter equation, a and b are shape parameters, A and B are the cutoff boundaries within which the scaled β distribution is defined.

With limited data on the bacteria overland transport parameters, we have used the lognormal distributions L(0.9,0.6)

Table 4. Statistical distributions of soil parameters by soil textural classes.

Soil textural class	The van Genuchten's soil water retention parameters (Meyer et al., 1997)				K_s (cm h^{-1}) after Rawls et al. (1998)		
	θ_s cm^{3}cm^{-3}	θ_r cm^{3}cm^{-3}	α_{VG}, cm^{-1}	n_{VG}	$P = 0.25$†	$P = 0.5$	$P = 0.75$
Sand	N(0.430, 0.060)‡	L(−3.09, 0.224)‡	N(0.147, 0.025)	L(0.978, 0.099)	9.6	18.2	26.6
Loamy sand	N(0.410, 0.090)	N(0.057, 0.015)	N(0.125, 0.04)	L(0.816, 0.091)	8.4	12.3	19.5
Sandy loam	N(0.410, 0.090)	B(2.89, 2.30, 0.017, 0.102)‡	B(1.82, 4.41, 0.008, 0.202)	L(0.634, 0.082)	3	5.6	13
Sandy clay loam	N(0.390, 0.070)	B(2.20, 2.01, 0.086, 0.114)	L(−3.04, 0.639)	L(0.388, 0.086)	0.2	0.77	5.05
Loam	N(0.430, 0.099)	B(3.64, 2.65, 0.037, 0.107)	B(1.58, 3.62, 0.003, 0.113)	L(0.442, 0.073)	0.16	0.39	2.8
Silt loam	N(0.450, 0.080)	B(3.35, 2.57, 0.024, 0.099)	L(−4.10, 0.554)	L(0.343, 0.085)	0.76	1.44	3.71
Clay loam	N(0.410, 0.090)	N(0.095, 0.010)	L(−4.22, 0.72)	N(1.32, 0.097)	0.22	0.42	1.31
Silty clay loam	N(0.430, 0.070)	N(0.088, 0.009)	L(−4.72, 0.56)	N(1.23, 0.060)	0.23	0.37	1.04
Sandy clay	N(0.380, 0.500)	B(4.00, 1.49, 0.056, 0.117)	L(−3.77, 0.56)	L(0.241, 0.065)	0.03	0.09	0.25
Silty clay	N(0.360, 0.070)	N(0.071, 0.023)	L(−5.66, 0.58)	L(0.145, 0.043)	0.05	0.18	0.75
Clay	N(0.380, 0.090)	B(1.50, 1.58, 0.001, 0.14)	L(−5.54, 0.89)	B(0.80, 1.55, 1.04, 1.36)	0.09	0.2	0.6

† Saturated hydraulic conductivities are given at three probability levels.

‡ N(a,b), L(a,b), and B(a,b,A,B) denote the normal distribution, the lognormal distribution, and the beta distribution with the probability density functions $f(x)=\frac{1}{\sqrt{2\pi}b}\exp\left[-\frac{1}{2}\left(\frac{x-a}{b}\right)^2\right]$, $f(x)=\frac{1}{\sqrt{2\pi}bx}\exp\left[-\frac{1}{2}\left(\frac{\ln(x)-a}{b}\right)^2\right]$, and $f(x)=\frac{\Gamma(a+b)}{\Gamma(a)\Gamma(b)}\frac{(x-A)^{a-1}(B-x)^{b-1}}{(B-A)^{a+b-1}}$, respectively.

and L(3.0,1.111) for the bacteria release coefficient, β_m, and the dispersivity a_L, respectively. These distributions included the values found from the model calibration (Table 3). The distribution of the net capillary drive, G, was found using the distributions of the Van Genuchten's (Van Genuchten, 1981) soil water retention parameters (Table 4) and the Latin Hypercube sampling technique.

The Latin Hypercube sampling of soil, rainfall, and transport parameters has been used to evaluate the probability of failure of the above mentioned vegetated clay loam plots to prevent the transport of pathogen indicator organisms from a 200-m stretch of a manured field under weather conditions of Beltsville, MD. No correlation was assumed in the random sampling of model parameter distributions.

The probability distribution function of the VFS efficiency was very steep. The simulated efficiency was close either to 100% or to 0% in the majority of cases. The VFS efficiency was found to be <95% in 25% of cases, <75% in 23% of cases, and <25% in 20% of cases. Relatively long high-intensity rainfalls, low hydraulic conductivities, low net capillary drive of soil, and high soil moisture contents before the rainfall were the main sources of the VFS low efficiency.

Discussion

The relatively simple overland flow and transport model was shown to be reasonably successful in explaining the observed experimental data on manure-borne bacteria transport study at a vegetated plots that emulated vegetated filter strips. However, values of model parameters (Table 3) and fitting results (Fig. 2 and Table 2) indicate that more complex description of the transport may be needed. The value of dispersivity was very large at some plots indicating that a very fast bacteria breakthrough has occurred that most likely can be attributed to the preferential overland flow pathways. The same indication can be deduced from the differences between simulated and measured concentrations in the very first portions of runoff studies. The VFS integrity may be compromised by the lack of vegetation, the differences in vegetation densities and patchiness, or other features that allow the bypass flow through the VFS that can substantially decrease its efficiency. Although successful models of flow and transport with bypass have been developed (Helmers et al., 2005), obtaining parameters for such models remains a challenge. The differences between velocities of different parts of the overland flow are represented as the hydrodynamic dispersion in the model in this work, but this may not be sufficient for cases with dominant bypass (concentrated and/or preferential) flows.

Controversy exists in literature on what portion of manure-borne fecal coliforms or *E. coli* moves in the overland flow being attached to solids. Edwards et al. (1996) considered full bacteria suspension in the liquid phase, while Schillinger and Gannon (1985) have reported up to 25% of bacteria were attached to the solids. Because vegetated buffers have been found to be most effective in trapping particulate pollutants (Dillaha et al., 1989; Schmitt et al., 1999; Helmers et al., 2006; Dabney et al., 2006), the model may need to be amended by including some modules to simulate the sediment transport and the bacteria attachment to sediment. However, no sediment was found in the runoff samples in our experiments, and such amendment might not be needed in this particular study. Whether attachment of manure borne coliforms to suspended sediment is important may depend on the sediment and runoff water properties. Previous experimental data have shown a drastic decrease in *E. coli* attachment to soil-borne solids in the presence of manure particulates (Guber et al., 2005). It remains to be investigated whether and how manure particulates in the overland flow may affect the overland coliform bacteria transport.

The soil under VFS soils may be or may become rich in organic matter which can substantially affect soil hydraulic properties (Rawls et al., 2004). Relatively little is currently known about hydraulic properties of VFS soils as compared with soils in other land uses.

The efficiency of the Monte Carlo simulations with the Latin Hypercube sampling substantially increases when the correlations among parameters are taken into account (Reckhow, 1994). Correlations among some of the model parameters are not known, even though correlations among others, for example, hydraulic properties, are known in general but not for typical VFS conditions, and correlations among the rainfall parameters have to be defined locally. Also, further research is needed to analyze the VFS efficiency for sequences of rainfall events during the periods when the upland field remains a source of microorganisms for overland flow. In this case, more has to be known about the redistribution and accumulation of infiltrating water in subsurface, because soil saturation after first rainfall event may prevent infiltration in the subsequent events. To account for this effect, simulations of coupled surface-subsurface flow, used in models such as VFSMOD model (Muñoz-Carpena et al., 1999) or STIR model (Kouznetsov et al., 2006) are needed. Modeling in this work uses the simple version of the Parlange infiltration model embedded in the KINEROS and is limited to the runoff generation due to infiltration excess. An analysis of the applicability of the surface transport with experimental data from landscapes with the saturation excess overland flow presents an interesting avenue to explore, and Monte Carlo simulations similar to ones in this work have to be done with a model that handles the saturation excess runoff.

Although the superiority of the Bradford-Shijven model (7) in simulations of bacteria release from manure has been demonstrated (Guber et al., 2006), little is known so far about the values of this model parameters for different manure types. In particular, we did not study the sensitivity of the VFS efficiency to the parameter α_m in this work, because the available information indicated that this parameter was strongly related to the value of irrigation or rainfall intensity as shown in Eq. [10]. The value of α_m may strongly affect the VFS efficiency and obtaining field information about the parameters of the model (7) presents an important avenue for research.

Although a substantial uncertainty is found for the values of parameters that control efficiency of VFS, many parameter combinations provide the simulated VFS efficiency of 100%. Most of these parameters cannot be measured for a specific site but their statistical distributions can be estimated for any specific site, using publicly available information. Therefore, the uncertainty analysis for VFS efficiency can be done without site-specific survey. The survey may be needed only if the uncertainty analysis will show a high probability of the VFS failure, and the change of the VSF design is not desirable. Then the survey may narrow the parameter ranges and affect the uncertainty analysis outcome.

Conclusions

Results of this study demonstrate that the vegetative filter strip efficiency is sensitive to the environmental parameters of soil, vegetation, and weather. The mechanistic modeling of the flow and transport of fecal coliforms in the VFS has provided a satisfactory description of experiments at plots with differing soil and vegetation cover. Using such modeling in Monte Carlo simulations of rainfall-runoff events with distributions of the environmental parameters of the rainfall runoff events can be useful for risk-informed site-specific decisions on the VFS design and placement. Further work is required to establish distributions of the coliform release and retention parameters as dependent on manure properties and vegetation properties.

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