Release of *Escherichia coli* from the bottom sediment in a first-order creek: Experiment and reach-specific modeling

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**S U M M A R Y**

*Escherichia coli* release from streambed sediments may substantially affect microbial water quality. Models of *E. coli* release and transport commonly use a single set of parameters for the whole stream or reservoir, yet little is known about the magnitude and sources of the variability of parameters of the streambed bacteria release. The objectives of this work were: (a) to obtain and compare parameters of streambed *E. coli* resuspension in three stream reaches with distinctly different bottom sediment textures, and (b) to see whether the modeling of streambed *E. coli* resuspension with reach-specific parameters could provide substantially better accuracy than modeling with a single set of parameters. Sediment particle size distributions and the streambed *E. coli* concentrations were measured along a first-order creek in the USDA-ARS OPE3 experimental watershed in Maryland. Afterwards, 80 m³ of water were released into the creek at a rate of 60 L per second in four equal allotments separated by 1–3 min intervals. Flow rates and *E. coli* concentrations were monitored with automated samplers at the ends of the three reaches with a total length of 630 m. A high concentration of streambed *E. coli* (‘‘hotspot’’) resuspended within the first reach caused a pulse of high *E. coli* concentrations that propagated along the creek without substantial attenuation; inputs of sediment-borne *E. coli* from the next two reaches were relatively small. The *E. coli* transport model included one-dimensional Saint–Venant and advective–dispersive equations. The calibrated roughness coefficient values were comparable for the three reaches, whereas the critical stress and the entrainment rate differed among reaches by a half order and an order of magnitude, respectively. Overall, better accuracy was observed when the model contained reach-specific parameters. Additional research is needed to understand which and how sediment properties affect parameters of streambed *E. coli* release into the water column.

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**1. Introduction**

Microbiological impairment of drinking, irrigation, or recreational waters is commonly monitored using concentrations of fecal indicator bacteria (FIB). The 2000 US National Water Quality Inventory (NWQI) reported that approximately 93,000 rivers and streams in this country contain elevated levels of FIB (USEPA, 2002). Mandatory water quality improvement programs, such as the USEPA Total Maximum Daily Load (TMDL), have been established in an attempt to decrease the fecal contamination of surface waters. Similarly, mandatory FIB concentration levels have been established for European coastal waters (Council of the European Communities, 1976, 2002). *Escherichia coli* is the FIB commonly used to evaluate microbiological water quality. Studies of fate and transport of *E. coli* in water bodies have shown that concentra-

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Artificial high-flow events caused a drastic two-order of magnitude increase in E. coli concentrations in stream water without any substantial input from any land source, such as runoff or animal feces (McDonald et al., 1982; Muirhead et al., 2004).

Modeling fate and transport of E. coli in streams and lakes without accounting for the release of E. coli from bottom sediments successfully captured spatial trends but appeared to be incapable of explaining dynamic variations of E. coli concentrations during and after rainfall events (Hellweger and Masopust, 2008). McCormquodale et al. (2004) simulated E. coli fate and transport following storm water discharges into the coastal Lake Pontchartrain, LA, USA; the model included a description of settling, but not resuspension. The authors indicated that the possibility of bacterial resuspension due to waves, currents, or swimming activity was a concern that had not been addressed in the model. Dortch et al. (2008) modeled concentrations of fecal coliform bacteria in the same lake without consideration of bacteria release from sediment and concluded that there must have been other source loadings into the lake that were not accounted for in the model.

The conceptual framework for including sediment–water column interactions into in-stream bacteria fate and transport modeling was first proposed by Matson et al. (1978). This framework has been implemented to varying extent in several recent modeling projects (Tian et al., 2002; Steets and Holden, 2003; Jamieson et al., 2005; Collins and Rutherford, 2004; Bai and Lung, 2005; Yang et al., 2008; Cho et al., 2010). In the majority of these works, the number of streambed bacteria released into the water column was simulated as the bacteria content in sediment multiplied by the mass of the resuspended sediment. Similarly, the number of bacteria deposited to the streambed was simulated as the settling sediment concentration in water multiplied by the fraction of bacteria thought to be associated with sediment particles. A single set of parameters of sediment resuspension and bacteria entrainment was used in all abovementioned modeling works.

Differences in sediment properties can cause large differences in parameters that are important for predicting resuspension of E. coli from streambeds. For example, considerable differences have been found in the values for the critical shear stress defining the onset of resuspension in the modeling work on E. coli transport. Jamieson et al. (2005) reported the value of 1.7 N m$^{-2}$; Bai and Lung (2005) have used the value of 0.4 N m$^{-2}$, whereas substantially lower values from 0.02 to 0.1 N m$^{-2}$ can be found in the work of Steets and Holden (2003). Yang et al. (2008) suggested that partitioning of sediment into a cohesive group and a non-cohesive group, and using different parameters for these groups, may improve simulation results. This suggestion can be substantiated by the fact that bacteria are generally thought to be associated with fine sediment particles within aquatic environments (Gannon et al., 1983; Auer and Niehaus, 1993).

The objectives of this work were: (a) to obtain and compare parameters of bottom E. coli resuspension in three stream reaches with distinctly different sediment particle size distributions, and (b) to determine whether modeling streambed E. coli resuspension with reach-specific parameters could provide substantially better accuracy than modeling with a single set of parameters.

2. Materials and methods

2.1. Description of study area

The study area (Fig. 1) is located at the Optimizing Production Inputs for Economic and Environmental Enhancement (OPE3) watershed research site, USDA-BARC on the mid-Atlantic coastal...
plain of Maryland, USA (http://www.ars.usda.gov/Research/doc- s.htm?docid=8438). The entire watershed is about 70 ha, with 75% employed in agricultural crop production while 15% is under deciduous forest. The riparian corridor, comprising 10% of the total land area, runs along the entire creek (~1100 m) with the width from about 65 m at its narrowest to more than 100 m at its widest point. Four fields (A, B, C, and D in Fig. 1, total area of 22.5 ha) have been under continuous corn production for the last 12 years. Field A receives only chemical fertilizers. The creek shown in Fig. 1 is the Beaver Dam Creek Tributary described in detail by Angier et al. (2005).

2.2. Sampling design

Four sampling stations were instrumented with weirs and automated refrigerated samplers (Sigma 900 Max All Weather Refrigerated Sampler, Hach Company, Loveland, CO) to measure depth of water and to sample water in the creek (Fig. 1). The weirs have been calibrated to convert depth of water to flow rate (Hively et al., 2006). The sections of the creek between stations 1 and 2, 2 and 3, and 3 and 4 are referred below as reach 12, reach 23, and reach 34, respectively.

The Trimble GeoXM 2005 Series global positioning system was used to determine elevations of the creek bottom at incremental distances along the creek. Creek sediment was sampled at 20-m increments along the creek to measure particle size distribution in the top 1 cm. Fifty grams of sediment were collected at four positions across the creek at each sampling location to represent the texture variation across the width of the streambed.

The artificial high flow experiment was carried out on August 12, 2008. Sediment was sampled equidistantly in four replications within each reach 1 h before the high-flow event. Composite samples were taken across the creek from the top 1 cm layer. The artificial high-flow event was created by releasing 80 m³ of city water on a tarp-covered stream bank 11 m upstream from station 1 at a rate of 60 L per second in four equal 20-m³ allotments. Water was delivered in trucks, and time intervals between allotments (1 min, 3 min, and 1 min) were determined by logistics of truck maneuvering. The sampling reservoirs were placed on the bottom of the stainless steel weir. The weirs were cleaned before the experiment, so there was no sediment lying in the weir. Water samples were collected every 2 min at station 1, and every 5 min at stations 2, 3, and 4. Water depth in weirs was measured every minute.

2.3. Physical and microbiological analysis

2.3.1. Water

Samples were transported to the laboratory on ice within 1 h and analyzed for E. coli concentrations using Colilert-18® (IDEXX Laboratories, Inc., Westbrook, Maine) with Quanti-tray™/2000 trays (Olstad et al., 2007). Five milliliters of the supernatant was added to IDEXX 100 mL bottles (two bottles per subsample for replication) containing 95 mL of sterile distilled water. One packet of Colilert-18 reagent was then added to each bottle and thoroughly shaken. After the reagent dissolved, bottle contents were poured into IDEXX trays and sealed with the IDEXX Quanti-Tray Sealer model 2X. The trays were then incubated at 37 °C for 18 h. To determine the Most Probable Number (MPN) of E. coli, positive, fluorescent wells within each Quanti-tray™/2000 tray were counted under 365 nm UV lighting (Spectroline CM-10, Spectro Corporation, Westbury, NewYork, USA). Turbidity was measured with the Orbeco-Hellige Digital Direct Reading Turbidimeter (Sarasota, FL).

2.3.2. Sediment

Sediment samples were scooped from the upper 1 cm of sediment, put into tubes with lids to prevent water loss, and immediately transported to the laboratory. The wet sediment and 90 mL of sterile distilled water were blended (blender model 34BL97, Waring) at high speed for 2 min in order to produce a homogeneous slurry (Garzio-Hadzick et al., 2010). After allowing the slurry to settle for 1 h, the supernatant was analyzed using the same procedure applied to water (Section 2.3.1). Note that potential IDEXX quantification errors due to particle-bound organisms have been previously explored by Fries et al. (2006) and determined to be minor in most scenarios. Sediment particle size was determined with the pipette method (Gee and Or, 2002). Sediment water content was determined in subsamples dried at 40 °C for 24 h. E. coli concentrations were expressed in MPN per gram of dry weight of sediment (gdw).
The explicit finite-difference scheme was used to solve the Saint Venant equations (Keskin and Agiralioglu, 1997), while the Crank–Nicolson method was applied to solve the transport equation (Fletcher, 1991). The observed time series of flow rate and E. coli concentrations at station 1 were used as the upstream boundary conditions in the hydrodynamic and water quality modules. Distributions of flow rates and E. coli concentrations derived from initial measurements along the creek were used as initial conditions. Spatial and temporal increments were 10 m and 0.1 s, respectively. Halving these values changed the solution less than 1%, and the Courant criterion was met.

The pattern search tool (patternsearch.m) in MATLAB software was applied to calibrate the model (Lewis and Virginia, 2002). For the flow model, the objective function was the root-mean-squared difference between predicted and observed flow rates. For the transport model, the objective function was the root-mean-squared difference between predicted and observed cumulative numbers of E. coli cells measured at stations 2, 3, and 4.

The model performance was evaluated using the root-mean-squared error (RMSE) and the Nash–Sutcliffe model efficiency coefficient (NSE) (Nash and Sutcliffe, 1970).

2.5. Parameter sensitivity analysis with The Latin Hypercube-One-factor-At-Time (LH-OAT)

The Latin Hypercube One-factor-At-a-Time (LH-OAT) method is a combination of the Latin Hypercube (LH) sampling and One-factor-At-a-Time (OAT) sampling (van Griensven et al., 2006). A range of variation is defined for each parameter of the model. Then the number of the parameter that has been varied, \( i = 1, 2, \ldots, P \), \( e_{ijk} \) is the fraction by which the parameter \( e_i \) is increased in the \( j \)-th LH sample in the \( k \)-th replication, \( k = 1, 2, \ldots, K \), and \( f_i \) is the fraction by which the parameter \( e_i \) is changed.

The sensitivity \( S_{ijk} \) for the parameter “i” is computed by averaging partial sensitivities over all LH sampling points and replications:

\[
S_{ijk} = \frac{100}{\sum_{j=1}^{J} f_{ijk}} \frac{1}{J} \sum_{j=1}^{J} S_{ijk}
\]

where \( M \) is the model output, sensitivity of which to parameters is investigated, \( j \) is the number of the LH sample \( (j = 1, 2, \ldots, J) \), \( i \) is the number of the parameter that has been varied, \( i = 1, 2, \ldots, P \), \( e_{ijk} \), \( e_{ijk} \), \( e_{ijk} \), \( e_{ijk} \), ..., \( e_{ijk} \), model parameters, for the \( j \)-th LH sample in the \( k \)-th replication, \( k = 1, 2, \ldots, K \), and \( f_i \) is the fraction by which the parameter \( e_i \) is changed.

The sensitivity \( S \) for the parameter “i” is computed by averaging partial sensitivities over all LH sampling points and replications:

\[
S_i = \frac{1}{K} \sum_{k=1}^{K} S_{ijk}
\]

where the larger \( S_i \) is, the more sensitive the parameter \( e_i \) is.

3. Results and discussion

3.1. Sediment composition

The distribution of sediment particle fractions and bottom slope along the creek are shown in Fig. 2. Sediment was predominantly sandy at reach 12; the amount of silt and clay were generally higher in reaches 23 and 34 (Fig. 2a). The percentage of clay was variable in reach 34 with the largest values at distances of 380–460 m. Variations in sediment particle size along the creek are at least partially attributable to the channel slope of the creek (Fig. 2b). The relative contents of clay and silt were greatest in reach 34 where the channel slope was lowest.

E. coli concentrations in sediment are shown in Table 1. The concentration of total sediment-borne E. coli was highest in reach 12; concentrations decreased in reaches 23 and 34. This was attributable to clay- and silt-borne E. coli concentrations 20-fold to 30-fold higher in reach 12 than in reaches 23 or 34. Our results illustrate that clay- and silt-borne E. coli concentrations can vary substantially within relatively short distances. One possible reason for the increased E. coli concentrations at site 12 could be the effect of manure applications on field A (Fig. 1); runoff from this field could reach the creek between stations 1 and 2, thereby contributing to the total E. coli load within reach 12. Another reason for the high concentrations of E. coli could be enhanced nutrient availability, due to the high prevalence of sandy sediments, promoting streambed bacterial growth (Ciarnoto, 2005).

An increase in E. coli concentrations associated with fine sediment particle fractions has often been observed in longitudinal studies. For example, Atwill et al. (2007) found a significant direct relationship between fine particle (clay and silt) contents, and E. coli concentrations in estuary and river sediment samples in Northern California. Our results are consistent with these prior observations; E. coli concentrations per unit mass of finer (silt + clay) particles were on average 2–6 times greater than E. coli concentrations per unit of total sediment (Table 1). Association of E. coli with clay and silt particles has been shown to be stronger than with sand particles (Guber et al., 2007; Pachepsky et al., 2006). Assuming that the resuspended sediment consisted mainly of clay and silt, and that the sediment resuspension was the main mechanism of E. coli release to the water column resuspension, we used concentrations of E. coli in sediment \( C_s \) determined per unit mass of fine (silt + clay) particles.

3.2. Initial flow parameters

The reach-average length, width, and calculated addition to flow between stations are shown in Table 2. The most substantial
addition to flow occurred in reach 23. This is in agreement with the results of the study by Angier et al. (2005), who found a small (4.8 m²) upwelling area that comprised only about 0.006% of the total riparian area (or approximately 0.001% of the entire catchment), yet supplied on average 4% of total stream flow.

3.3. Artificial high flow experiment

Fig. 3 shows spatiotemporal variations of flow rates, turbidity, and E. coli concentrations at the four different creek stations during the artificial high-flow event, as well as flows, turbidity and E. coli concentrations prior to the flow event. Interruptions in water release caused several rising and falling limbs at the station 1 hydrograph. The release interruptions were less discernible on the hydrographs as the water pulse moved downstream. Turbidity and E. coli pollutographs, i.e., dependences of turbidity and E. coli concentration on time at the locations where hydrographs were measured, had long tails indicating low settling rates.

Table 1

<table>
<thead>
<tr>
<th>Reach</th>
<th>Total sediment-based $\times 10^5$ (MPN kg$^{-1}$)</th>
<th>Clay and silt-based $\times 10^5$ (MPN kg$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reach 12</td>
<td>48.93</td>
<td>307.80</td>
</tr>
<tr>
<td>Reach 23</td>
<td>30.37</td>
<td>75.75</td>
</tr>
<tr>
<td>Reach 34</td>
<td>8.39</td>
<td>54.20</td>
</tr>
</tbody>
</table>

Table 2

Stream reach geometry and flow added for the Beaver Dam Creek Tributary, Maryland.

<table>
<thead>
<tr>
<th>Reach 12 (stations 1–2)</th>
<th>Reach 23 (stations 2–3)</th>
<th>Reach 34 (stations 3–4)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Length of reach (m)</td>
<td>140</td>
<td>140</td>
</tr>
<tr>
<td>Average width (m)</td>
<td>1.25</td>
<td>1.02</td>
</tr>
<tr>
<td>Calculated groundwater addition to flow, $\times 10^5$ (m² s$^{-1}$)</td>
<td>0.46</td>
<td>1.80</td>
</tr>
</tbody>
</table>

Fig. 3. Flow rates, turbidity, and E. coli concentrations during an artificial high-flow event at four monitoring sites (stations 1, 2, 3, and 4).
The cumulative numbers of E. coli cells that passed sampling stations, 1, 2, 3, and 4 during the observation periods were $0.6 \times 10^5, 7.1 \times 10^5, 8.3 \times 10^5$, and $6.3 \times 10^6$, respectively. A marked difference could be observed in sediment-borne E. coli inputs between reaches 12, 23 and 34. Reaches 12 and 23 had similar lengths and hydrographs, yet 4–5 times less E. coli were released from bottom sediments in reach 23 than in reach 12 during the 2.5 h observation period.

The dependence of the cumulative number of cells on time at station 2 seemed to approach an asymptotic value; therefore, the cumulative number of cells measured at stations 1 and 2 could be used to estimate the thickness of the effective disturbed layer from which bacteria were released into the water column. The approximate expression for the cumulative number of cells $N_i$, leaving the reach can be derived from the mass balance as

$$N_t = dAB_c(1-n)\rho_i + N_i - B$$  \hspace{1cm} (7)

where $d$ is the thickness of the effective disturbed layer, $m$, $A_B$ is the bottom area of the reach, $m^2$, $C_i$ is the initial number of cells in sediment, cell kg$^{-1}$, $n$ is the sediment porosity, $\rho_i$ is the specific density of solids in the sediment material, $N_i$ is the cumulative number of cells that entered the reach, and $B$ is the number of cells that settled back to bottom sediments. The assumption for the applicability of Eq. (7) is that the concentration of bacteria in resuspended sediment is the same as the disturbed layer, and is equal to $C_i$. For the reach 12, assuming $A_B = 175$ m$^2$, $C_i = 6 \times 10^6$ cell kg$^{-1}$, $n = 0.6$, $\rho_i = 2650$ kg m$^{-3}$, and $B = 0$, the thickness of the disturbed layer was about 6 mm. The thickness of the disturbed layer was probably similar or smaller for the two other reaches.

The average sediment bacteria release rate $R$ could be approximated as

$$R = \frac{(N_2 - N_i - B)}{t_R}$$  \hspace{1cm} (8)

where $t_R$ is the duration of the release. Eq. (8) is based on the applicability of (7). Taking $t_R = 40$ s as the approximate duration of the rising limb, we obtained a rate $R$ of about 15,000 cells m$^2$ s$^{-1}$ for the reach 12. This number is comparable with the values found by Jamieson et al. (2005) in their study of nalidixic acid-resistant E. coli transport in the Swan Creek in Canada. The bacterial release rates were of the order of 10,000 cells m$^2$ s$^{-1}$ for three the storms that they documented.

Turbidity and E. coli were only weakly correlated ($r = 0.41$), although the shapes of their pollutographs had some similarity (Fig. 3). The most obvious discrepancies in turbidity and E. coli concentrations were observed at stations 1 and 4, where turbidity peaked prior to E. coli concentrations and returned to lower levels more rapidly. A possible reason for the weak correlation could be that sediment flocculation caused turbidity to peak before E. coli, i.e., the enhanced settling velocity of particles resulted in lower turbidity levels. Pettibone et al. (1996) observed that a few large flocculated particles accounted for most of the volume of resuspended sediment-borne FC (before and after ship passage) in a study conducted with waters from the Buffalo River. Sediment flocs can be a dominant form of sediment transport in freshwater fluvial systems (Droppo and Ongley, 1994). Higher suspended solid concentrations can increase settling velocity due to the occurrence of large particles entraining other particles (Wolanski et al., 1995). In addition, inorganic particles can potentially attach to organic components from both living organisms, such as algae, ciliates, zooplankton and bacteria, as well as detritus from plants and animals in the presence of large amounts of organic riverside material, thereby leading to an increase in settling velocity (Uiterwijk Winkel, 1975). Note that both strong and weak correlations between bacterial concentrations and turbidity have been established in various studies. Whereas Muirhead et al. (2004) found a significant correlation between turbidity and E. coli, only a weak relation between E. coli and turbidity was observed by McDonald et al. (1982) and Goyal et al. (1977). McKergow and Davies-Colley (2009) noted that turbidity can be a useful surrogate for E. coli in small catchments; this conclusion was reached based on the rainfall events in which both runoff and sediment resuspension took place. Site-specific sediment and water properties within a given watershed may have an important impact on the E. coli/turbidity relationship.

3.4. Flow modeling

The correspondence between simulated and measured hydrographs is shown in Fig. 4. Only roughness coefficient values were calibrated. Inspection of the graphs and values of RMSE and Nash–Sutcliffe coefficients (Table 3) show that, overall, predicted flow rates as to the Saint–Venant equation were in accordance with observed values. The arrival times were correct. The modeling accuracy at station 3 was relatively poorer than those at stations 2 and 4; the model overestimated peak flow rate and underestimated the hydrograph in the tail section. Presence of large wooden debris could cause large deviations from the assumptions of the model given by Eqs. (1) and (2) that could not be simulated by varying values of the roughness coefficient.

![Fig. 4. Simulated and observed values of flow rates during the artificial high-flow event at three monitoring sites (station 2, 3, and 4).](image)

Table 3

<table>
<thead>
<tr>
<th>Parameters</th>
<th>Range</th>
<th>Final effect, $S$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rank 1 Entrainment coefficient ($R_e$)</td>
<td>0.10E-6 – 0.10E-2</td>
<td>1150.72</td>
</tr>
<tr>
<td>Rank 2 Critical shear stress ($\tau_c$)</td>
<td>0.10E-5 – 1.00</td>
<td>401.30</td>
</tr>
<tr>
<td>Rank 3 Dispersion coefficient ($D$)</td>
<td>0.10E-3 – 2.00</td>
<td>74.10</td>
</tr>
<tr>
<td>Rank 4 Settling velocity ($v_s$)</td>
<td>0.10E-6 – 0.10E-2</td>
<td>0.01</td>
</tr>
</tbody>
</table>
The calibrated values of the roughness coefficient (Table 4) were between 0.1 and 0.12. These values are similar to those reported for some streams, e.g., n = 0.11 for Cypress Creek (Arcement et al., 1979) or n = 0.12 for Yockanookany River (Colson et al., 1979), although they were 2–3 times greater than the values reported by Chapra (1997). According to Cowan (1956), the roughness coefficient can be determined by considering four factors: base values for straight, uniform and smooth channels of natural materials; surface irregularities; obstruction effects; and vegetation and flow condition effects. According to Barnes (1967), the roughness coefficient for stable channels ranges from 0.024 to 0.075. Channel geometrical irregularities can increase the roughness coefficient by as much as 0.02 (Chow, 1959; Benson and Dalrymple, 1967). Furthermore, maximum increases in the roughness coefficient of 0.003 can be caused by the channel curvature. Obstructions to flow increase roughness; this increase is related to both shape and size of the obstruction, the size of the obstruction with respect to the size of the cross-sectional area, as well as the number, arrangement and spacing of obstructions. Various obstructions such as riparian vegetation and tree branches were present within the creek boundary. One cannot exclude some effect of the weirs at the sampling stations on the flow.

3.5. E. coli transport modeling

The sensitivity of the root-mean square-error, used to predict cumulative numbers of E. coli for the four model parameters (entrainment coefficient ($R_e$), critical shear stress ($\tau_c$), settling velocity ($v_s$), and dispersion coefficient ($D$)), was investigated using the LH-OAT method. Table 5 represents the sensitivity rankings of the four E. coli concentration parameters during the high-flow event. The most significant parameter for simulating E. coli was the entrainment coefficient ($R_e$), followed by the critical shear stress ($\tau_c$) and dispersion coefficient ($D$). The sensitivity of the settling velocity ($v_s$) was very low, probably because the observation and simulation time were relatively short. Based on the sensitivity analysis results, the model was calibrated for (a) values of $R_e$ and $\tau_c$ individual for each reach, and (b) the same values of $R_e$ and $\tau_c$ for the whole creek section between stations 1 and 4. The value of $D$ was found only for the whole creek section between stations 1 and 4. The settling velocity was not calibrated, but was computed from Stokes law as suggested by Auer and Niehaus (1993).

Results of the E. coli transport model calibration are shown in Table 3. Overall, the entrainment coefficient determined for the whole creek section between stations 1 and 4 was greater than the entrainment coefficients determined for individual reaches. The reach 12 had the largest resuspension rate since the entrainment coefficient was the largest and the critical shear stress was the lowest. On the contrary, the lowest E. coli resuspension rates were found in reach 23. A possible explanation for the differences in parameter values between the reaches can be the difference in access to fine particles for the flowing water. Large pore spaces present in the predominantly sandy sediments in reach 12 may have provided better conditions for the initial resuspension and further mobility of fine particles and associated microorganisms. Of the three reaches, reach 12 contained the best conditions for the enhancement of hyporheic exchange, i.e., circulation of cells that move from stream to sediment bed (i.e., downwelling) and back again (i.e., upwelling); in particular, upwelling can provide nutrients to streams by mobilizing fine particles in sediment beds (Valett et al., 1994). The hyporheic exchange has been previously suggested as the reason of very slow settling of E. coli back to sediments after their release into the water column (Jamiesson et al., 2005). The differences in calibrated values of $\tau_c$ for reaches 23 and 34 were more consistent with the observation that increases in particle size generally increase the critical shear stress for fine sediments (Lick, 2009).

Fig. 5 compares observed and simulated cumulative numbers of E. coli that passed water sampling stations in the creek during the artificial high-flow event. Overall, the model with reach-specific parameters demonstrated better accuracy than the model that assumed the single set of values for $R_e$ and $\tau_c$ for the whole creek (between stations 1 and 4). The latter model provided reasonable predictions at stations 3 and 4, but, on the whole, underestimated the cumulative E. coli numbers. The model performance parameters in Table 4 support the superior performance of the reach-specific model. In particular, the predictive power of the reach-specific model was notable for station 2 where the NSE increased from 1.89 to 0.97.

3.5.1. Patterns of the sediment resuspension

Fig. 6a shows spatiotemporal variations of velocity while Fig. 6b and c show the relative shear stress and sediment resuspension

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**Table 4**

<table>
<thead>
<tr>
<th>Parameter used in the flow and E. coli transport modeling for the Beaver Dam Creek Tributary.</th>
<th>Unit</th>
<th>Calibrated value</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>A. General parameters</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Manning's roughness coefficient for Reach 12, n</td>
<td>–</td>
<td>0.13</td>
</tr>
<tr>
<td>Manning's roughness coefficient for Reach 23, n</td>
<td>–</td>
<td>0.12</td>
</tr>
<tr>
<td>Manning's roughness coefficient for Reach 34, n</td>
<td>–</td>
<td>0.10</td>
</tr>
<tr>
<td>Dispersion coefficient, D</td>
<td>m$^2$ s$^{-1}$</td>
<td>0.56</td>
</tr>
<tr>
<td>Particle setting velocity, $v_s \times 10^3$</td>
<td>m s$^{-1}$</td>
<td>2.66</td>
</tr>
<tr>
<td>Fraction of E. coli associated with suspended solids, f</td>
<td>–</td>
<td>0.50</td>
</tr>
<tr>
<td><strong>B. Reach-specific parameters</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Critical shear stress for Reach 12, $\tau_c, \times 10^2$</td>
<td>N m$^{-2}$</td>
<td>3.4</td>
</tr>
<tr>
<td>Critical shear stress for Reach 23, $\tau_c, \times 10^2$</td>
<td>N m$^{-2}$</td>
<td>18.7</td>
</tr>
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<td>Critical shear stress for Reach 34, $\tau_c, \times 10^2$</td>
<td>N m$^{-2}$</td>
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<td>Entrainment coefficient for Reach 12, $R_e, \times 10^6$</td>
<td>kg m$^{-2}$ s$^{-1}$</td>
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</tr>
<tr>
<td>Entrainment coefficient for Reach 23, $R_e, \times 10^6$</td>
<td>kg m$^{-2}$ s$^{-1}$</td>
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<td>Entrainment coefficient for Reach 34, $R_e, \times 10^6$</td>
<td>kg m$^{-2}$ s$^{-1}$</td>
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<td><strong>C. Non-specific parameters</strong></td>
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<tr>
<td>Critical shear stress, $\tau_c \times 100$</td>
<td>N m$^{-2}$</td>
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</tr>
<tr>
<td>Entrainment coefficient, $R_e \times 10^6$</td>
<td>kg m$^{-2}$ s$^{-1}$</td>
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**Table 5**

<table>
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<tr>
<th>Station</th>
<th>Parameter set</th>
<th>Flow rate</th>
<th>RMSE $\times 10^3$ (m$^2$ s$^{-1}$)</th>
<th>NSE (–)</th>
<th>E. coli</th>
<th>RMSE $\times 10^6$ (MPN/100 mL)</th>
<th>NSE (–)</th>
</tr>
</thead>
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<tr>
<td>2</td>
<td>The same for all reaches</td>
<td>4.56</td>
<td>0.84</td>
<td>42.33</td>
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<td></td>
<td>Reach-specific</td>
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<td>–</td>
<td>5.32</td>
<td>0.98</td>
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<tr>
<td>3</td>
<td>The same for all reaches</td>
<td>5.88</td>
<td>0.52</td>
<td>7.69</td>
<td>0.98</td>
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<td></td>
</tr>
<tr>
<td></td>
<td>Reach-specific</td>
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<td>–</td>
<td>7.77</td>
<td>0.98</td>
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<tr>
<td>4</td>
<td>The same for all reaches</td>
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<td>0.78</td>
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<td>–</td>
<td>3.19</td>
<td>0.99</td>
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</table>
rates, respectively. Comparison of Figs. 6a and 2b shows that the channel slope controls spatiotemporal variations of the longitudinal velocity; velocity increases as the streambed becomes steeper. The spatiotemporal pattern of the resuspension rate, in general, replicated the velocity pattern because shear stress is a function of longitudinal velocity. However, the value of the critical shear stress modified the relation between velocity and resuspension. Specifically, the hydrodynamic shear force was the lowest at reach 23 (Fig 6b) where the velocity was relatively high and the critical shear stress was also the highest. As shown in Fig. 6b and c, both hydrodynamic shear force and resuspension rate were largest at reach 12, and this, along with the largest concentrations of E. coli in sediment, resulted in the creation of a large pulse of E. coli in the water column that propagated through the subsequent reaches without substantial changes.

4. General comments

The spatial and temporal scale of the research reported in this work is comparable to the scale of other experimental studies of sediment E. coli release into the water column during artificial or natural high-flow events; although our experimental data were collected over a shorter distance and at more frequent time intervals. For example, McDonald et al. (1982) examined bacteria concentrations entrained from a stream bed along an approximated 3-km stretch. Artificial flood experiments by Muirhead et al. (2004) were conducted in a stream which fed a reservoir (water supply) with an observation reach length of about 2.5 km with a peak flow of 4 m$^3$ s$^{-1}$. Most recently, the experiment by Jamieson et al. (2005) was conducted in a 1.7 km-long creek section. In our work, the total length of the creek section was 630 m with a peak flow of 0.4 m$^3$ s$^{-1}$.

Values of sediment bacteria release parameters may be affected by the E. coli enumeration procedure that includes blending which, in principle, can cause cell injury. Effect of blending on the E. coli enumeration is not certain, and currently blending for 2 min (United States Food and Drug Administration, 2002), or for 1 min (Boehm et al., 2009) is commonly used in E. coli determination in various porous media. The recent method comparison study by eight laboratories in USA and Canada (Boehm et al., 2009) showed that blending as compared with the preferred handshaking method did not affect enumeration in two of the three studied sands and produced lower E. coli than hand shaking by 0.2 log units in the third sand. We compared the enumeration results with two blending speeds and two blending times for the sediments in three locations at the creek in this study and did not see statistically significant difference ($P > 0.02$, data not shown).

Our experimental results show how a high concentration (“hot spot”) of sediment-borne E. coli can affect the E. coli pollutographs downstream. The hot spot at reach 12 caused a pulse of E. coli that propagated along the creek without substantial attenuation and with relatively small inputs from the sediment of the downstream reaches. Spatial variability of E. coli concentrations in sediment is notoriously high (Pachepsky and Shelton, 2011). Literature reports for values of E. coli concentrations in sediment vary from 1 to 500000 MPN or CFU per g dry weight. Strongly asymmetric distribution functions have been found where multiple replicate samples have been taken (Erkenbrecher, 1981, or Berry et al., 2007). It is not uncommon to find two to five orders of magnitude differences between maximum and minimum concentrations observed at the same site or in the same watershed. With this type of

Fig. 5. Observed (circles) and simulated (lines) cumulative numbers of E. coli cells that have passed monitoring sites during artificial high-flow event; solid line – simulated with the same parameters for all reaches, dash line – simulations with reach-specific parameter sets.
variability, hot spots can be missed with grab sampling. However, as our data demonstrate, the presence of hot spots can be revealed in pollutographs. Therefore, *E. coli* pollutographs can provide important information on sediment-borne *E. coli* reservoirs and their availability for being released into the water column.

The spatial variability of *E. coli* concentrations in sediments may present the greatest difficulty in setting initial conditions for the *E. coli* transport model (3). An error in $C_s$ estimation may substantially affect the value of $Re$ found from the calibration. One approach to avoid this uncertainty might be to use the product $RsC_s$ as a single parameter $R_s$ to characterize the bottom sediment source in the transport equations

$$\frac{\partial C}{\partial t} + u \frac{\partial C}{\partial x} = \frac{D}{\tau} \frac{\partial^2 C}{\partial x^2} + \frac{Rs}{h} \frac{\partial C}{\partial x} - \frac{q}{A} C, \quad R_s = \begin{cases} R_s\left(\frac{q}{A} - 1\right), & \tau \geq \tau_c, \\ 0, & \tau < \tau_c \end{cases} \quad (9)$$

so long as settling does not change sediment concentrations significantly, e.g., for short events, such parameterization can be viable.

This study focused on short and dynamic *E. coli* concentration variations in response to a single high-flow event. We are not aware of models that can simulate survival, replenishment, and bacterial exchange between the sediment and water column. Development of such models would represent the essential complement to resuspension release modeling in order to understand and predict temporal dynamics of *E. coli* in stream water that is used for recreational and irrigation purposes.

The detailed sampling undertaken in this work is hardly feasible at coarser (catchment) scales at which monitoring, management and policy decisions are usually made. Further studies are needed to scale up the creek bacteria transport modeling to the watershed scale. Two possible avenues for research are of interest here. One is to retain the model of this work and to strive to relate the critical shear stress and entrainment coefficients to soil and landscape properties that have been obtained from surveys and can be found in public sources. The second is to modify the description of release, transport, and settling as the scale of research changes. One example for this second approach is presented in the work of Kim et al. (2010) where release and settling of the bottom sediment *E. coli* were effectively tied to the resuspension and settling of the sediment in the second order stream. No matter which of the two above modeling approaches will be taken, estimating the initial concentrations of *E. coli* in the sediment will remain the most challenging part of the modeling projects. The proximity to *E. coli* sources remains the most promising indicator of the possible level of *E. coli* in water. Bacterial loads associated with human or animal presence and/or activities could, in some cases, be related to the elevated concentrations of *E. coli* in sediments. Locations of creek recreational use (activity in water) coincided with increased concentrations of FC in sediments in the study of Crabill et al. (1999). That led the authors to the conclusion that recreational use served as the FC distribution system. Giddings and Oblinger (2004) suggested that the high *E. coli* densities at one of their sites was the result of the many animal operations and home sites upstream, and the particularly large depositional area at the site where sediments accumulated from upstream sources. The distance from the source of pollution affects concentrations of FC and *E. coli* in sediments when the source of fecal pollution is clearly defined. Goyal et al. (1977) observed an inverse relationship between FC in sediment and the distance from the sewage outfalls in the canals of the Texas West Coast. A similar strong dependence was documented by Haller et al. (2009) near the water treatment outlet at the Lake Geneva. Time spent by birds at the observation sites at the Chesapeake Bay strongly ($r = 0.79$) correlated with FC concentrations in sediments (Hussong et al., 1979). FC numbers increased 100-fold in the sediments of water bodies following their colonization by water fowl in Poland (Niewolak, 1989). More needs to be learned about the potential effect of a particular *E. coli* discharge on the *E. coli* concentration in sediments.
5. Conclusions

Resuspension of sediments during or shortly after rainfall events can cause sharp increases in water-borne fecal indicator bacteria, e.g., *E. coli*. In this study, both monitoring and modeling were performed to investigate *E. coli* resuspension in response to an artificial high-flow event. The major conclusions from this study are the following:

1. The artificial high-flow event that did not create runoff inflow to the experimental creek, caused a substantial increase in *E. coli* concentrations in stream water. This indicates that the release of *E. coli* from bottom sediments can be the major factor of microbiological water quality in streams.

2. During and after the artificial high-flow events, the bacteria breakthrough at observation stations along the creek stations exhibited long tails which indicated low settling rate values allowing bacteria to be transported far from the release sites.

3. A high concentration (“hot spot”) of sediment-borne *E. coli* can affect the *E. coli* pollutographs downstream. The hot spot at the first reach in this work caused a pulse of *E. coli* that propagated along the creek without substantial attenuation and with relatively small inputs from the sediment of the downstream reaches. Given the high spatial variability of *E. coli* concentrations in sediments, the hot spot could be missed if sediment was sampled, but it manifested itself in pollutographs. Observing pollutographs appears to be more reliable method of presence of *E. coli* in sediments as compared to direct sediment sampling, although the pollutography is not really able to identify the location of the hot spot.

4. The bacteria transport model with reach-specific parameters showed better performance in prediction of *E. coli* resuspension than the model with a single set of parameters, although the model with the single set of parameters was able to capture dynamics of bacteria transport. Differences in sediment properties among the three stream reaches affected both the critical shear stress, controlling the onset of bacteria release, and the entrainment coefficient.

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References


