

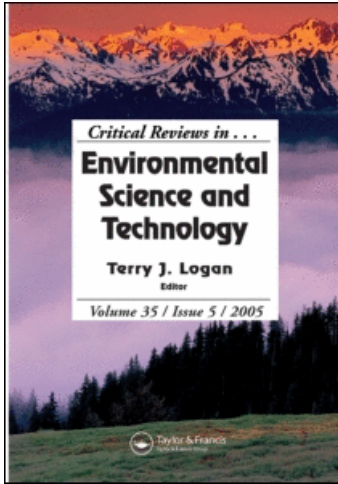
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### *Escherichia Coli* and Fecal Coliforms in Freshwater and Estuarine Sediments

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## ***Escherichia Coli* and Fecal Coliforms in Freshwater and Estuarine Sediments**

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*It has been known for some time that substantial populations of fecal coliforms and E. coli are harbored in freshwater bottom sediments, bank soils, and beach sands. However, the relative importance of sediments as bacterial habitats and as a source of waterborne fecal coliforms and E. coli has not been recognized until recently, when a large number of publications have shown that in many cases the resuspension of sediment, rather than runoff from surrounding lands, can create elevated E. coli concentrations in water. This review is an attempt to develop the first comprehensive single source of existing information about fecal coliforms and E. coli in sediments and adjacent soils and to outline the knowledge gaps and research needs. The authors summarize available information on variability and environmental correlations of E. coli and FC concentrations in sediments, genetic diversity of E. coli in sediments, survival of E. coli and FC in sediments, release with resuspended sediment and settling of E. coli and FC, modeling of sediment effects on fate and transport of E. coli in surface waters, and implications for monitoring and management of microbiological water quality. The demonstrated role of pathogenic E. coli strains in food and water quality challenges reinforces the need in better understanding ecological and hydrological factors that affect functioning of sediments as E. coli reservoirs.*

**KEY WORDS:** *Escherichia coli*, bottom sediment, survival, release, water quality

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## 1. INTRODUCTION

Fecal coliform bacteria, and more recently *E. coli*, have attracted special attention in environmental research due to the use of these microorganisms as indicators of fecal contamination and microbiological impairment of water. Present water quality regulations for drinking, irrigation, and recreational uses are primarily based on *E. coli* concentrations. Mandatory water quality improvement programs, such as the U.S. Environmental Protection Agency (EPA) Total Maximum Daily Load, target *E. coli* concentrations in water. *E. coli* is also frequently used as the source identifier in microbial source tracking methods. The existence of various pathogenic *E. coli* (e.g., enteropathogenic, enterotoxigenic, enterohemorrhagic *E. coli* strains) makes the choice of *E. coli* even more appropriate.

It has been known for some time that substantial populations of fecal coliforms (FC) and *E. coli* are harbored in freshwater bottom sediments, bank soils, and beach sands. Testing sediments to evaluate bacterial pollution was first proposed more than 100 years ago (Savage, 1905). In the seminal work by Geldreich (1970) establishing the need for testing water for FC, it was stated that the water-sediment interface of a stream or lake bottom can serve as a reservoir for fecal pollution 'fallout' from overlying water. Although the reservoir concept perpetuated, the relative importance of sediments as bacterial habitats and as a source of water-borne FC and *E. coli* has not been recognized until recently, when a large number of publications have shown that in many cases the resuspension of sediment, rather than runoff from surrounding lands, can create elevated *E. coli* concentrations in water. This has far-reaching consequences for the detection, monitoring, and control of microbiological pollution of freshwater sources. As sediments may act as a reservoir for pathogens, it is important that they, too, be evaluated to determine if they pose a potential risk to human health (Donovan et al., 2008).

This review is an attempt to develop the first comprehensive single source of existing information about FC and *E. coli* in sediments and adjacent soils and to outline the knowledge gaps and research needs.

## 2. VARIABILITY AND ENVIRONMENTAL CORRELATIONS OF *E. COLI* AND FC CONCENTRATIONS IN SEDIMENTS

### Magnitude of Observed Concentrations

Examples of observed concentrations of *E. coli* and FC in sediments are presented in the Table 1. Very large variations have been recorded in bacteria concentrations in sediments from different sources as well as within a single stream or water body. Literature reports for values of *E. coli* concentrations in sediment vary from 1 to 500000 MPN or CFU per gram of dry weight.

**TABLE 1.** Examples of concentrations of *E. coli* and fecal coliforms observed in freshwater sediments

Source	Region	Medium	Range or average	Units <sup>c</sup>
Alm et al., 2003; <i>E. coli</i>	Lake Huron and St. Clair River	Beach sand, 60% sand, 0–5 cm	12–80	CFU GDW <sup>-1</sup>
An et al., 2002 <i>E. coli</i>	Lake Texoma, located on the Oklahoma and Texas border.	Lake sediment, 0–1 cm	(0.6–5.0)·10 <sup>5</sup>	CFU GDW <sup>-1</sup>
Anderson et al., 2005 <i>E. coli</i>	Hillsborough River, Fla	River sediment, SDA <sup>b</sup>	5 · 10 <sup>2</sup>	CFU GWW <sup>-1</sup>
Atwill et al., 2007, <i>E. coli</i>	Northern California	River sediment, SDA	5–13	CFU GDW <sup>-1</sup>
Bergstein-Ben Dan and Keppel 1992, <i>E. coli</i>	Lake Kineret, Israel, 0–8 m depth	Lake sediment, 0–1 cm	1–1000	CFU GDW <sup>-1</sup>
Bergstein Ben-Dan and Keppel 1992, <i>E. coli</i>	Lake Kineret, Israel, 8–17 m depth	Lake sediment, 0–1 cm	1–100	CFU GDW <sup>-1</sup>
Berry et al., 2007, <i>E. coli</i>	Nebraska	Basin to collect filtered discharge from pens. Bottom sediment, 0–15 cm	100–1000 in 5 samples 0 in another four samples	CFU GDW <sup>-1</sup>
Buckley et al., 2007, total coliforms	Subtropical rainforest, Queensland, Australia	Streambed, 1–1.5 cm	0.5·10 <sup>4</sup> —dry season 1.2·10 <sup>4</sup> —wet season	CFU ml <sup>-1</sup>
Byannapali et al., 2003	Michigan	Creek sediment, NA	70–230 25–75th percentiles	MPN GDW <sup>-1</sup>
Byannapali et al., 2003	Michigan	Bank sediment, 6 cm	10–210 25–75th percentiles	MPN GDW <sup>-1</sup>
Crabill et al., 1999, fecal coliforms	Arizona	Creek bottom 10 cm	(0–3.5) 10 <sup>7</sup> Mean 10 <sup>7</sup>	CFU 100 ml <sup>-1</sup>
Davies and Bavor, 2000	Australia	Constructed wetland	10–10000	CFU GDW <sup>-1</sup>
Desmarais et al., 2002	Florida	Embankments, 0–5 cm	700	CFU GDW <sup>-1</sup>
Donovan et al. 2008, fecal coliforms	Pennsylvania	River sediments, city, 0–15 cm	80–12100	CFU GWW <sup>-1</sup>

(Continued on next page)

**TABLE 1.** Examples of concentrations of *E. coli* and fecal coliforms observed in freshwater sediments (Continued)

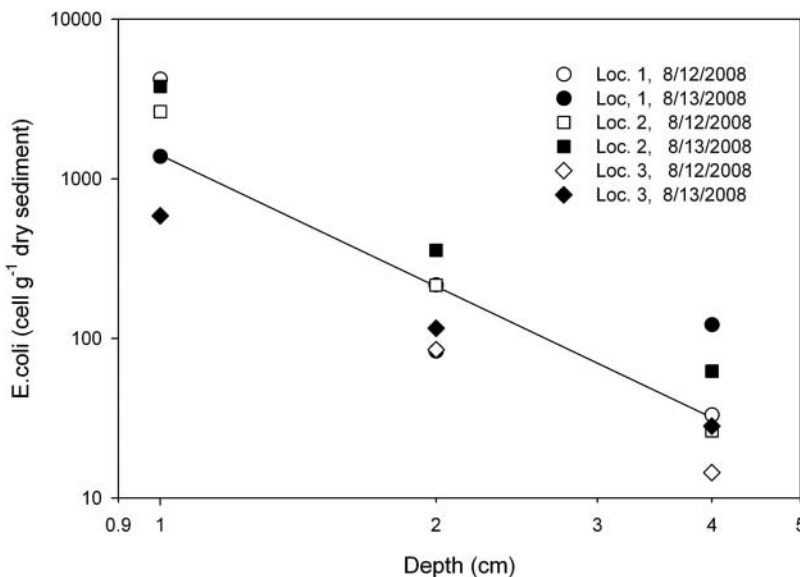
Source	Region	Medium	Range or average	Units <sup>c</sup>
Donderski and Wilk, 2002	Poland, Vistula River	Bottom sediment, Ekman dredge <sup>a</sup>	120–2400	CFU GWW <sup>-1</sup>
Doyle et al., 1992, fecal coliforms	Arizona	Reservoir bottom, 0–5 cm	200–10000	CFU 100 ml <sup>-1</sup>
Erkenbrecher, 1981	Virginia,	Estuary bottom, upper few centimeters	1–200000, geometric mean = 807	MPN GDW <sup>-1</sup>
Evanson and Ambrose, 2006, <i>E. coli</i>	California	Coastal wetlands, sediment, 0–2 cm	0–3000	MPN GDW <sup>-1</sup>
Fries et al., 2008, <i>E. coli</i>	North Carolina, Neuse River estuary	Estuary bottom, 1 cm	10–1000	MPN GDW <sup>-1</sup>
Giddings and Oblinger, 2004, <i>E. coli</i>	Newfound Creek basin, NC	Stream sediment, 2 cm	390–12000	CFU GDW <sup>-1</sup>
Goyal et al., 1977, fecal coliforms	Coastal canals, Texas	Bottom sediment Ekman dredge	2–92000	MPN 100 ml <sup>-1</sup>
He et al., 2007, <i>E. coli</i>	San Diego region, California	Stream stagnant zones or ponds, SDA	220–90000	MPN GDW <sup>-1</sup>
He et al., 2007, <i>E. coli</i>	San Diego region, California	Streams with flowing water, SDA	130–5000	MPN GDW <sup>-1</sup>
Hussong et al., fecal coliforms	Chesapeake Bay	Bay bottom, Ekman dredge	13–3500	CFU 100 ml <sup>-1</sup>
Karim et al., 2004, <i>E. coli</i>	Arizona	Constructed wetland, hyacinth pond (Ekman dredge)	5.5·10 <sup>4</sup> –3.0·10 <sup>6</sup>	MPN GDW <sup>-1</sup>
Karim et al., 2004, <i>E. coli</i>	Arizona	Constructed wetland, duckweed pond (Ekman dredge)	(1.0–4.3)·10 <sup>5</sup>	MPN GDW <sup>-1</sup>
Muirhead, et al., 2004, <i>E. coli</i>	New Zealand	Creek bottom sediment	10 <sup>8</sup>	CFU m <sup>-2</sup>
Niewolak, 1998, Fecal coliforms	Poland, Czarna Haricza River	Bottom sediment, 1 cm	4–16600	CFU GDW <sup>-1</sup>
Smith et al., 2008, total coliforms	NC, Charlotte, urban creeks	Bottom sediment, 1 cm	(0.4–8.5)·10 <sup>4</sup>	CFU/1 cm stream crosssection
Lipp et al., 2001, <i>E. coli</i>	Florida, Charlotte Harbor	Bottom sediment, 7–10 cm	20–4700	CFU 100 GDW <sup>-1</sup>
Wu et al., 2009, <i>E. coli</i>	Massachusetts, Blackstone River Watershed	Bottom sediment	2–335	GDW <sup>-1</sup>

<sup>a</sup>Up to 15 cm.<sup>b</sup>SDA = sampling depth unknown.<sup>c</sup>GDW = gram dry weight; GWW = gram wet weight.

Strongly asymmetric distribution functions are found where the replications have been taken (Berry et al., 2007; Erkenbrecher, 1981). It is not uncommon to find differences of 2–5 orders of magnitude between maximum and minimum concentrations observed at the same site or in the same watershed. Anderson et al. (2005) observed higher variability between replicate sediment samples as compared to water samples in a laboratory study. The authors commented that this variability had multiple contributing factors, including patchy distribution of organisms in sediments and difficulty in dissociating bacteria from sediment particles.

### Distributions of *E. Coli* and FC in Sediment Profile

Data on FC/*E. coli* concentrations in sediments can be influenced by the thickness of the sediment layer studied (e.g., data of Bergstein-Ben Dan and Keppel [1992] in Table 1). Alm et al. (2003) observed a twofold decrease in *E. coli* content in beach sand with 5 cm increments of depth within the first 15 cm. The coefficient of variation was about 30% and did not change with depth. A much steeper decline in sediment *E. coli* concentrations with depth was observed by Garzio (2009) in a rural creek in Maryland (Figure 1). Concentrations of FC in the top 2 cm of sediments were significantly ( $p < .001$ ) higher than in the 2–10 cm layer (Ferguson et al., 1996). Haller et al. (2009) observed the fast (about 1 order of magnitude per centimeter) decrease of *E. coli* concentrations with depth within first 5 cm of sediment in



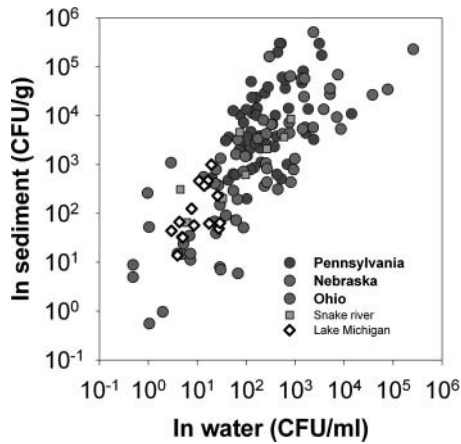
**FIGURE 1.** Dependence of *E. coli* concentration of depth in sediment at three locations along the creek in agricultural settings, Maryland (after Garzio, 2009).

Lake Geneva. No significant variation was observed within the 5–10 cm layer in this work. Babinchak et al. (1977) monitored changes in FC concentration in Thames River water caused by the deposition of sediment dredged elsewhere. These authors did not see an increase of FC concentrations in water at the disposal sites; they explained that FC appeared to occur only in the surface sediment material and were diluted by the subsurface material during the dredging operation. Since storm flow resuspension comes mostly from the top thin layers of sediments (in the order of 1 cm as reported by Pachepsky et al., 2009), the potential effect of sediment bacteria on the concentration in water should be estimated only from this upper layer.

### *E. Coli* and FC Contents in Sediment and Water Columns

Comparisons of *E. coli* and FC contents in sediment and in the water column above it have inevitably led to the conclusion that sediments are the dominant reservoir for microorganisms. Numerous authors have observed that concentrations of FC in sediments are multiple-fold higher than in the water column. For example, Van Donsel and Geldreich (1971) noted that concentrations of sediment FC were 100–1000 times greater than that of overlying waters in various aquatic environments. Goyal et al. (1977) found that FC in sediment were from 1 to 383 times higher than in water with a median value of 10. Similar differences were observed by Erkenbrecher (1981) in an estuary in the Chesapeake Bay. Doyle et al. (1992) reported ratios of mean sediment FC densities to mean water FC densities from between 10 and 100 to 1. The sediment populations of FC were on average 2200 times greater than the water counts in the study by Crabill et al. (1999) set in Arizona. Davies-Colley et al. (2007) analyzed data from agricultural streams in New Zealand and concluded that most of the time the water in the agricultural streams contained only a tiny fraction (about 1/1000) of the total FC contamination in the stream; the rest resided in the streambed from where it could be released by floods.

Correlations between *E. coli* and FC concentrations in the sediment and in the water column have, in most cases, been reported to be weak. Byannapali et al. (2003) noted a relatively low correlation between concentrations of *E. coli* in stream water and in the sediment with  $r = .49$  ( $p = .064$ ). A similar low correlation ( $r = .28$ ) was reported by Crabill et al. (1999) for concentrations of FC in water and sediment along a creek in Arizona. Trends in FC populations in sediments could not be used to predict those in waters or vice versa according to two-month-long observations of Doyle et al. (1992). An attempt to correlate *E. coli* densities in water and in sediment was not successful for Lake Texoma, Oklahoma (An et al. 2002). These authors observed no correlation between concentrations of *E. coli* in sediment and water; *E. coli* in sediment ranged from  $6 \cdot 10^4$  to  $5 \cdot 10^5$  CFU  $\text{g}^{-1}$  dry sediment while concentrations of *E. coli* in water were less than 1 CFU  $100 \text{ ml}^{-1}$ . Data

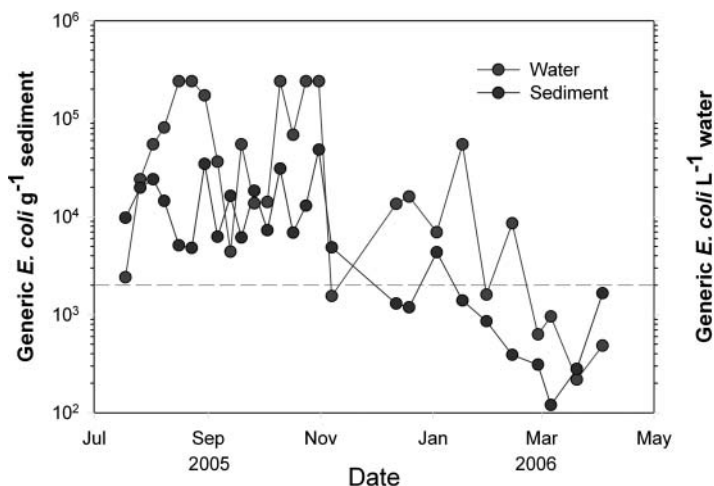


**FIGURE 2.** Examples of relationships between *E. coli* concentrations in water and sediment. ● = Cove Mountain Creek, Pennsylvania (Shelton et al., 2008); ○ = basin to collect filtered discharge from pens (Berry et al., 2007); ● = Maumee Bay and Maumee River, Ohio (Francy et al., 2005); □ = Snake River and Boise River, Washington (Stephenson and Rychert, 1982); ◇ = Lake Michigan (Whitman et al., 2006).

from several locations in the U. S. (Figure 2) support the conclusion that correlations between *E. coli* and FC concentrations in sediment and in the water column are weak. The lack of correlation between sediment and water is to be expected during periods of base flow. In the absence of turbulence and resuspension, sediments contribute very little of the bacterial load to surface waters. Rather, bacterial inputs into waters during base flow are likely due primarily to fecal deposition from wildlife, waterfowl, or agricultural animals, or leaching from septic systems or sewer lines (depending on adjacent land use). Note that at any given sampling site, the measured bacterial concentrations are derived from multiple locations upstream of the sampling site. An additional factor is stream depth versus sampling depth. Even during periods of sediment resuspension, if sediment-borne organisms are not distributed throughout the entire water column, water samples taken near the surface may not be representative of the total load. Finally, the diurnal oscillations in *E. coli* concentrations in stream water may be up to 1.5 orders of magnitude (Meays et al., 2006), and this may compromise correlations with *E. coli* concentrations in sediment, although nothing is known about the diurnal oscillations of *E. coli* concentrations in sediments.

Although sediment-borne bacterial concentrations are not reliable short-term indicators of water quality, they may provide a relatively stable indicator of long-term bacterial concentrations in a watershed, as proposed by Franson (1992). Longer periods of observations indicate that seasonal trends in *E. coli* concentrations in water and sediment generally coincide (Figure 3).





**FIGURE 3.** Seasonal dynamics of *E. coli* concentrations in water and sediment of Cove Mountain Creek (Shelton et al., 2008).

Averaging of concentrations over longer periods of time improves correlations between water column and sediment concentrations. Although the water and sediment concentration trends were different within 2–3 month periods, seasonal trends in water and sediment *E. coli* concentrations were similar for this site in Appalachia. Erkenbrecher (1981) observed that average annual FC concentrations in water and sediment were highly correlated ( $r = .86$ ) over 10 observation sites.

### Temporal Variations in *E. Coli* and FC Concentrations

Sediment-borne bacterial concentrations are highly variable over time. In fact, this variability in conjunction with resuspension (Chawla et al., 2003; Giddings and Oblinger, 2004) is a likely explanation for the erratic variations in FC indicator organism concentrations frequently observed in water quality monitoring (Jawson et al., 1982; Sherer et al., 1988; Sherer et al., 1992). Temporal variability is due to relative rates of bacterial growth/die-off and to episodic resuspension and redistribution of sediments due to rainfall events, while spatial variability is due to stream bed heterogeneity. Seasonal dynamics in *E. coli* concentrations in sediments have been documented by several authors. Goyal et al. (1977) observed higher numbers of FC in canal sediments in winter than in summer, and attributed these differences to lower die-off rates in winter months. On the other hand, Crabill et al. (1999) encountered differences of 3 orders of magnitude between sediment FC concentrations in summer versus winter. The frequent flushing of sediments during the winter melt has been suggested as a possible cause of the decrease of the sediment FC population in winter (Crabill, 1999). Buckley et al. (1998) observed

more than a twofold difference between *E. coli* sediment concentrations in wet and dry periods for a subtropical rainforest creek. Lipp et al. (2001) reported that sediment *E. coli* concentrations tended to be greatest in August and again in December through February in a Floridian estuary. These seasonal differences were larger than the seasonal differences in total bacteria numbers obtained from direct scanning electron microscope observations of intertidal sediments (DeFlaun and Mayer, 1983). In the latter work, the temporal dynamics of bacterial biomass directly followed the temperature changes. *E. coli* O157 was not found in summer sediment samples. Based on a large-scale survey of the Rio Grande basin, Hartke et al. (2005) found no *E. coli* O157 in sediments during the summer; however, its prevalence varied from 0 to 80% in fall, winter, and spring samples. The temporal variability of FC concentrations in reservoir sediment was higher than in water as documented by Doyle et al. (1992). In this work, the instability of sediment FC populations may have been influenced by daily water level fluctuations due to hydroelectric operations and by the coarse composition of the sediments that permitted rapid interflow of water during tidal fluxes, induced by reservoir operations, resulting in highly variable bacterial residence times in these sediments.

The year-to-year variations in *E. coli* concentrations in sediment have been attributed to climatic conditions. Cinotto (2005) published data on *E. coli* sediment for two consecutive years and noted that drought conditions in 2002 resulted in lower overall bacteria concentrations than the more typically wet year of 2003. *E. coli* concentrations in fluvial sediment along the study reach in 2002 had a median concentration of 92 CFU (g<sup>-1</sup> wet sediment); in 2003, the median concentration had risen to 4,752 CFU (g wet sediment)<sup>-1</sup>.

### Sediment Properties

The spatial variability of *E. coli* and FC concentrations has often been attributed to the differences in sediment particle size distributions. Regression analysis has confirmed a significant direct relationship between the percentage of clay and silt particles and FC and *E. coli* concentrations in estuarine and riverine sediment samples in Northern California (Atwill et al., 2007). Garzio (2009) observed an increase in sediment *E. coli* concentrations with increasing silt content in the sediment of a Maryland creek. The same trend was observed by Niewolak (1998) across 10 observation sites on a river in Poland. On the other hand, Cinotto (2005) reported the highest median concentration of *E. coli* (2160 CFU g<sup>-1</sup> wet) in the 0.125–0.5 mm size range of natural sediments. This maximum was attributed to the possibly unfavorable conditions for bacteria in fine sediments, such as reduced porosity and permeability. Doyle et al. (1992) did not find a significant relationship between sediment coliform concentrations and sediment textural fractions

contents; sediments were very coarse (>90% sand and gravel) in this work and might not have provided a sufficient range of particle sizes to establish a relationship.

The contradictory reports on the effect of sediment texture on the size of *E. coli* and FC populations are probably related to the multiplicity of ways in which the particle size distribution can affect the persistence of these organisms. Coarse sediments may not provide sufficient protection from the environment to allow the persistence of a substantial concentration of bacteria; for example, with too much exposure, bacteria may be subject to the effects of sunlight inactivation or protozoan grazing. On the other hand, availability of nutrients may be better in coarse sediments (Cinotto, 2005).

Differences in sediment organic matter contents have been postulated as a possible explanation of the spatial variation in *E. coli* and FC populations, with mixed results. No correlation between organic matter content and total coliform concentration was found in streambed sediments from a tropical rainforest (Buckley et al., 1998). On the contrary, FC concentrations in the sediments were significantly higher in the presence of organic matter in the work of Ferguson et al. (1996). High correlations were found around locations where bacteria entered the stream with the influx of organic matter (Irvine and Pettibone, 1993). DeFlaun and Mayer (1983) noted that organic matter accumulation may be a cause and a consequence of bacterial biomass accumulation; the search for correlations between coliform concentrations and organic matter content may be not successful in such cases.

### Electrolytes in Water

Differences in water salinity have been invoked as a possible explanation for variations in sediment *E. coli* and FC concentrations in environments with brackish water (Erkhenbrecher, 1981; Goyal et al., 1977; Lipp et al., 2001). The negative effect of salinity was attributed to faster die-off, which in turn was thought to be associated with the decrease in attachment of bacteria to sediment particles (Goyal et al., 1977). Average *E. coli* concentrations in ponded water with conductivity greater than 5 mS/cm were significantly lower than that with conductivity lower than 5 mS/cm in the longitudinal study of He et al. (2007) in Southern California.

### Plant–Microbe Interactions

The presence of aquatic plants responsible for release of carbon substrates may affect the *E. coli* populations in sediments. Cinotto (2005) studied an impoundment area in a Pennsylvania creek and observed that elevated aquatic growth resulted in sharp increases in *E. coli* concentrations from upstream to downstream throughout the impoundment area in 2002 and 2003. In 2003, *E. coli* concentrations within the waters column increased from 940 CFU/100

ml to 6,000/100 ml at the dam crest. In laboratory experiments by Ksoll et al. (2007), *E. coli* readily colonized periphyton (from the Lake Superior shoreline) and persisted for several weeks; in addition, cells were released to the overlying water. Field data of these authors showed a significant linear relation between FC concentrations and periphyton ash-free dry weight ( $R^2 = .72$ ).

Numerous studies suggest that bacteria and algae coexist in an association that benefits both groups of organisms (e.g., Carr et al., 2005). Based on observations in constructed wetlands, Karim et al. (2004) suggested that attachment of microbes to plant root surfaces could be an important mechanism to reduce settling of microorganisms.

### Proximity to Sources

The distance from water input sources has often been described as the factor controlling *E. coli* and FC sediment concentrations. *E. coli* numbers in water and sediment were higher close to the shoreline of the Kinneret Lake (Bergstein-Ben Dan and Keppel, 1992). Similarly, when the data representing permanently flooded and tidal zones were compared, sediment FC were found to occur in significantly greater densities in the tidal than in the flooded zone in one of the studied reservoirs (Doyle et al., 1992). However, the opposite was true for another reservoir. The bottom topography, causing different bottom agitation by the surf, could be a reason for the different trends in sediment *E. coli* and FC concentrations changes with the distance from the shore in one of the reservoirs.

Bacterial loads associated with human or animal presence and 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 ( $r = .79$ ) 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).

## Adjacent Soils, Wetlands, and Beaches as Sources

Autochthonous (indigenous) *E. coli* have been found in soils of various environments, first in tropical (Byappanahalli and Fujioka, 1998; Fujioka, 2001; Fujioka et al., 1999), then in subtropical (Solo-Gabriele et al., 2000; Desmarais et al., 2002), and finally in temperate regions (Byappanahalli et al., 2006; Ishii et al., 2006). In the latter work conducted with soils of several coastal Lake Superior watersheds, PCR-based DNA fingerprint analyses indicated a 92% similarity level between soil-borne *E. coli* genotypes that overwintered in frozen soil. Soil-borne *E. coli* strains had DNA fingerprints that were unique to specific soils and locations, suggesting that these *E. coli* strains became naturalized members of the soil microbial community. Growth at high temperatures (30–35°C) and survival at medium temperatures (25°C) was observed in this work.

The distance from the edge of water affects the density of *E. coli* populations in soils and beaches. Byannapali et al. (2003) noted that *E. coli* decreases rapidly with distance from the streambed. A similar pattern was reported by Desmarais et al. (2002), who observed 1 order of magnitude in *E. coli* concentrations in soil at the distance of 90 cm from the edge of water near the creek in Florida. Along the Lake Michigan shore *E. coli* was present in highly variable counts in beach sand to depths just below the water table and distances of at least 5 m inland from the shore, which provides a large potential area of input to beach (Whitman et al., 2003). The comprehensive analysis of beach sands in this region led to the conclusions that *E. coli* may be able to sustain population density in temperate beach sand during summer months without external inputs (Whitman and Nevers, 2003). This research presented evidence that foreshore beach sand plays a major role in bacterial lake water quality, is an important nonpoint source of *E. coli* to lake water rather than a net sink, and is possibly capable of supporting an autochthonous, high density of indicator bacteria for sustained periods, independent of lake, human, or animal input. A very strong correlation was observed between *E. coli* counts in the foreshore sand and aqueous *E. coli* monthly geometric means at beaches at the Lake St. Clair in Michigan (Macomb County Public Health, 2004).

Indirect but convincing indications of the important role of the wetland sediment as a fecal bacteria transport medium in tidal wetlands were presented by Huang (2005) for the Southern Chesapeake Bay. According to the analysis in this work, sediment transport seemed to control FC transport in the tidal wetlands during dry periods. During wet weather, sediments from tidal wetlands as well as from upland were distributed to the adjacent water in a very short time and the risk of fecal contamination could increase dramatically. The work of Wolfert (1998) supports these observations.

*E. coli* appears to be able to utilize constituents of the soil organic matter for population support and growth (Tate, 1978). The presence of

organic litter does not necessarily imply elevated populations of *E. coli*. Byappanahalli et al. (2003) surveyed banks of a stream in Michigan and found that *E. coli* was highest in relatively clean moist sands and much lower in litter-laden sandy soils.

Soil water content affects the survival and regrowth of *E. coli*. The greatest survival of coliforms was noted with anaerobically grown cells amended into flooded soil as compared to moist soil in the work of Tate (1978). However, temporary drying did not eliminate the *E. coli* population in soil taken at 50 cm distance from the edge of water (Desmarais et al., 2002). Laboratory experiments of Solo-Gabriele et al. (2000) confirmed that upon soil drying, *E. coli* is capable of multiplying by several orders of magnitude. The authors hypothesized that *E. coli* can survive at lower soil moisture than its predators, and therefore, upon soil drying, conditions are suitable for *E. coli* growth. Under this assumption, it is likely that the outer fringes of the channel banks, which experience the most extreme drying conditions, dominate the contribution of *E. coli* to the water column. Based on a comprehensive review, Zaleski et al. (2005) concluded that *E. coli* does not seem to enter a viable but nonculturable state in soils.

There are indications that the naturalized *E. coli* population can move downslope toward streams by erosion or runoff events (Ishii et al., 2006) or as a result of tidal events (Solo-Gabriele et al., 2000). The ubiquity of *E. coli* along the banks and forested soils clearly suggests that increased *E. coli* loading in the creek water was mostly attributable to contributions from nonpoint (possibly nonfecal) sources held and subsequently released by soil and sediment erosion (Byannapali et al., 2003).

### 3. GENETIC DIVERSITY OF *E. COLI* IN SEDIMENTS

Relatively little is known about the diversity of *E. coli* inhabiting sediments and *E. coli* population dynamics. Since the gastrointestinal tract of warm-blooded animals is the primary habitat for *E. coli*, all sediment-borne *E. coli* are initially derived from fecal runoff and deposition. However, the relative diversity of *E. coli* strains found in sediments may be different from those in the overlying water column or feces of local animal populations. Matches between strains isolated from bottom sediments and from the water column were rare in a survey of isolates undertaken by Atwill et al. (2007). The authors, using the Box-PCR method, found that the median value match was 0%. The authors speculated that either the *E. coli* located within estuarine sediments were independent from the *E. coli* transiting the estuary in the water column and whose source was somewhere upstream or, if most of the *E. coli* did in fact originate from sources upstream of the estuary, then high levels of genetic diversity for this population of *E. coli* resulted in very low probabilities of a DNA match. Poff and Tecele (2002) reported a substantial

mismatch between genotypes of *E. coli* in water and in sediment in Slide Rock State Park, Arizona; 84% of the water-borne *E. coli* were derived from raccoons (31%), humans (16%), skunks (11%), elk (11%), beaver, dogs, or white-tailed deer (6% each), whereas the *E. coli* proportions in the sediments were from horses (16%), humans (12%), raccoons and white-tailed deer (11% each), elk and skunk (10% each), and cows and mule deer (9% each). These data illustrate the fact that, due to resuspension and redistribution of sediments within a watershed, the diversity of sediment-borne strains may differ from input strains in the immediate vicinity. On the other hand, Vogel et al. (2007) observed a remarkable similarity in host library-based source tracking classification of water-isolated *E. coli* and sediment-isolated *E. coli* for the Plum Creek watershed in south-central Nebraska. Isolates that could be identified as cattle-related constituted 57% in water and 58% in sediment samples. Similarly, isolates identified as having a wildlife origin comprised 29% of water and 29% of sediment samples.

### Diversity Due to Different Growth/Die-Off Rates

Diversity of indigenous sediment-borne *E. coli* strains can be different from other introduced strains due to different rates of growth, die-off, parasitism, or predation. Sediments are not simply a repository for introduced strains; they are a secondary habitat with their own unique physical and chemical characteristics and microbial food chains and in which selection pressures determine which strains of *E. coli* will persist. For example, Anderson et al. (2005), using *E. coli* ribotyping, demonstrated that the *E. coli* strains originally present in water and sediment were much more persistent than strains introduced with inoculation of freshwater mesocosms, containing dog feces, hospital wastewater, and soil microcosms; furthermore, the distribution of ribotypes sampled from mesocosm waters was dissimilar from the distribution in fecal material. The prolonged survival of enteric bacteria in sediments has long been recognized (Carrillo et al., 1985; Davies et al., 1995; Fish and Pettibone, 1995; Hendricks, 1971; Hendricks and Morrison, 1967; McFeters et al., 1974; Sherer et al., 1992). This may be because sediments, depending on the environment, offer a more favorable chemical and biological environment (Gannon et al., 1983), supplying the osmoprotecting substances allowing bacteria to withstand the elevated salinity (Ghoul et al., 1990), heavy metal toxicity (Jones, 1964), UV radiation (Bitton et al., 1972), or protection from parasitism by bacteriophages (Roper and Marshall, 1974) or predation from protozoa (Davies et al., 1995). Genetic differences appear to effect the attachment of *E. coli* to particles of specific sizes (Pachepsky et al., 2008).

Two hypotheses have been proposed to describe the manner in which *E. coli* respond to the transition from its primary to secondary habitat. Savageau (1983) suggested that *E. coli* cells cope with the transition

by possessing a dual regulation system, where genes with products in high demand are under positive control, whereas genes with products in low demand are under negative control; the controls systems alternate depending on the specific demands imposed by the primary and secondary environments. Savageau (1983) presented numerous examples of physiological functions in enteric bacteria that are consistent with the demand theory, and it has been demonstrated that in *E. coli* some genes appear to be preferentially expressed in the aquatic environment (Espinosa-Urgel and Kolter, 1998). If *E. coli* possesses a dual regulation, this would lead to the prediction that the clonal composition of *E. coli* in host populations would reflect that in the external environment. If selection were the dominant force determining the outcome of the transition, then the assumption underlying efforts to trace the source of coliform contamination would be invalid (Gordon et al., 2002). Anderson et al. (2005) noted that, so far, typing of *E. coli* populations in primary and secondary habitats has shown substantial shifts in the dominant isolates over time, suggesting that only certain members of the population in the primary habitat remain viable in the secondary habitat.

#### 4. SURVIVAL OF *E. COLI* AND FC IN SEDIMENTS

##### Survival Kinetics

A wide range of *E. coli* die-off rates have been reported in the literature. Interpretation of these data, however, is problematic because, as previously discussed, die-off rates are highly dependent on the ability of introduced strains to adapt to and persist in specific sediment habitats. In an early study, Van Donsel and Geldreich (1971) reported a 90% die-off of both FC and *Salmonella spp.* in 7 days in various sediments. By comparison, in Davies et al.'s (1995) study, 85 days was required to reach 90% die-off (or inactivation) of FC.

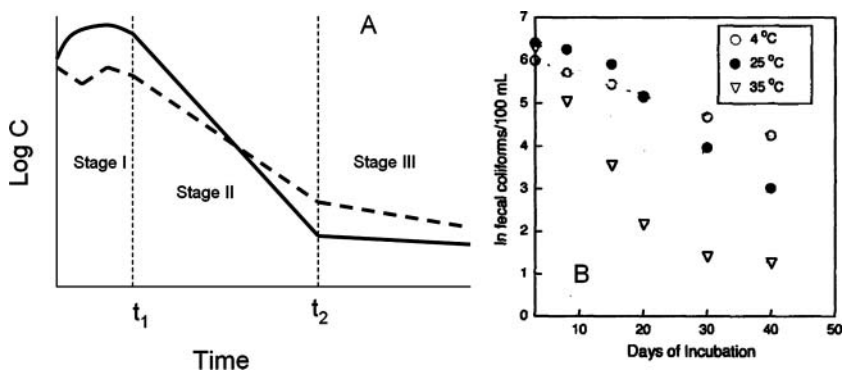
The common way to generalize data on FC and *E. coli* survival data in sediments is to use the exponential die-off model (Chick, 1908)

$$\ln C = \ln C_0 - \mu t \quad (1)$$

where  $C$  and  $C_0$  are initial and present concentration, respectively;  $\mu$  is the inactivation rate; and  $t$  is time. The inactivation rates found in the literature vary significantly. The inactivation rates of *E. coli* in inoculated sediment from lakes of the eastern United States were about  $0.54 \text{ d}^{-1}$ . Jamieson et al. (2004a) reported a value of  $\mu = 0.15 \text{ d}^{-1}$  in sediments of Southern Ontario creeks. A FC inactivation rate constant of  $0.07 \text{ d}^{-1}$  was published for sediment from Hillsborough River, Florida (Anderson et al., 2005).

Differences in the shapes of die-off curves complicate the comparison of *E. coli* die-off rates in sediments. Equation (1) was in many cases found





**FIGURE 4.** Observed shapes of *E. coli* and fecal coliform die-off curves. (a) Schematic representation of three survival stages observed in experiments; (b) examples of die-off curves with two survival stages present; data on *E. coli* survival in manure–sediment mixtures from Howell et al. (1996).

inadequate to simulate FC or *E. coli* die-off in sediments (e.g., Davies et al., 1995; Howell et al., 1996; Jamieson et al., 2004a). These authors attributed that to the effects of predation and growth on the overall population decrease. Several distinctly different behaviors have been observed that can be summarized in schematics of survival shown in Figure 4. The first stage (0 to  $t_1$  in Figure 4) is an adaptation period when regrowth, slow inactivation, or oscillations in the population numbers occur. The second stage ( $t_1$  to  $t_2$  in Figure 4) is the stage of the exponential inactivation when Model 2 is applicable given the time is counted from  $t_1$ . The third stage is the stage of slow inactivation or stabilization of the population at the low population level. All three stages are not necessarily represented in measured *E. coli* and FC die-off in sediments, as illustrated in Figure 4b. Neither Stage I nor Stage III have been observed at a low temperature of 4°C, Stages I and II have been seen at the temperature of 25°C, whereas Stages II and III have been observed at 35°C.

The initial regrowth during Stage I has been documented by Desmarais et al. (2002) and Solo-Gabriele et al. (2000). On the other hand, Anderson et al. (2005) did not observe the growth of *E. coli* populations when introduced with dog feces or wastewater into soil in mesocosms with freshwater and sediment.

*E. coli* survival has been observed to be longer in sediments than in water (e.g., Craig et al., 2004). Coupling *E. coli* or FC inactivation data in water and sediment has led to the conclusion that inactivation rates in sediment are an order of magnitude lower than those for the water column (Anderson et al., 2005; Jamieson et al., 2004a; Mallin et al., 2007). Karim et al. (2004) reported a similar phenomenon for a constructed wetland, although the differences were smaller; the FC inactivation rates in water and in sediment were 0.51 and 0.35  $d^{-1}$ , respectively.

### Temperature Effects

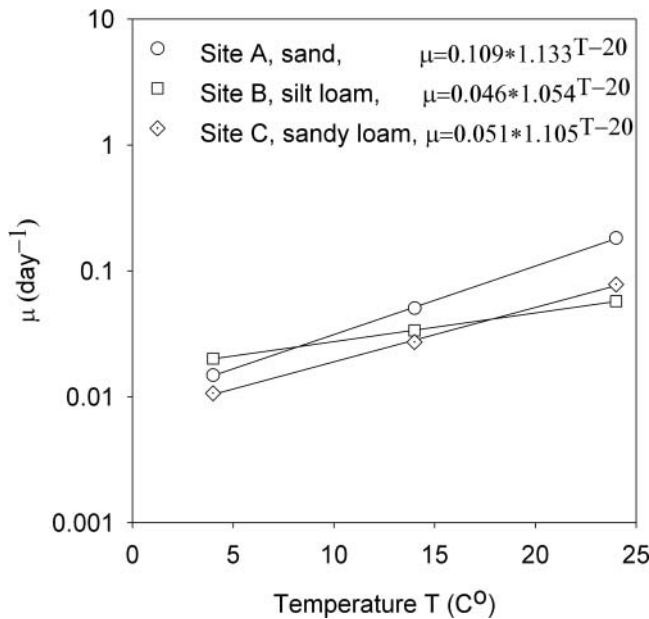
Temperature is a major factor controlling *E. coli* and FC die-off in sediment, comparable to other environmental matrices. Garzio (2009) tested the applicability of the often-used  $T_{20}$  model

$$\mu = \mu_{20}\theta^{T-20} \tag{2}$$

to determine the survival in sediment of *E. coli* introduced with manure. We found that Equation (2) performed very well (Figure 5) when the value  $\mu$  was found from the exponential decay part of the survival curve (Stage II in Figure 4). The parameter  $\theta$  increased as the texture become finer (i.e., the survival rate was less sensitive to temperature in the finer soil). The die-off rates at the temperature of 4°C were very low, indicating that *E. coli* is able to survive in sediments through winter. An et al. (2002) noted that lower temperature at the bottom of deep freshwater reservoirs favored survival of *E. coli* in sediments.

### Salinity and Pollutants

Saline water has been reported to accelerate *E. coli* and FC die-off (Anderson et al., 2005). Atwill et al. (2007) observed a trend of decreasing *E. coli* concentrations with increasing salinity and depth. On the other hand, Fries



**FIGURE 5.** Dependence of *E. coli* die-off rate  $\mu$  on temperature in flow-chamber experiments with manure-amended sediment from a Maryland creek (Garzio, 2009).

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et al. (2008) found a positive correlation between salinity and *E. coli* concentrations in water. However, this correlation has been interpreted as an artifact; the increased persistence of *E. coli* in sediments was deemed to be the real cause of the concentration increase.

The role of pollutants in solution and sediment in FC survival is not known. Simmons (quoted in U.S. Environmental Protection Agency, 2004) hypothesized that because *E. coli* may be consistently more pollution-tolerant than bacteriophages, restoring stream health can then encourage greater predation of *E. coli* by bacteriophages.

### Organic Matter and Nutrients

Presence of suitable organic substrate may enhance FC survival. Survival of *E. coli* from manure slurry mixed with sediment was much longer in clay sediments than in sand and loamy sediments (Howell et al., 1996; Sherer et al., 1992); the half-life in clays was about 2 times higher than in sand and loam. Overall, data on the effect of the nutrient levels on *E. coli* survival in sediments were inconclusive. In the absence of specific information regarding nutritional status or inputs, interpretation is problematic. Banning et al. (2003) suggested that an increase in available nutrients may have little effect on *E. coli* persistence due to competition for nutrients by other microflora. In fact, Hartz et al. (2008) noted that the addition of nutrients to sand in their study did not increase the growth of *E. coli*, indicating that indigenous *E. coli* were out competed. Cahoon and Toothman (2004) did not observe significant correlations between phosphate levels and *E. coli* concentrations in North Carolina creeks, whereas sediment phosphate levels were important in controlling FC survival and growth in estuarine sediments (Rowland, 2002).

Conversely, Jeng et al. (2005) reported that storm events increased *E. coli* populations in sediment; they hypothesized that the increase was due to the availability of fresh, nutrient-rich sediments deposited onto the surface of the creek bed. Presumably, these fresh sediments provide new surfaces for colonization, thus regenerating the supply of bacteria available for suspension and redistribution during the next storm flow event (Jamieson et al., 2005a). Evanson and Ambrose (2006) observed that wetland sediment-associated *E. coli* levels remained elevated longer than wetland water after rainfalls; at one of their observation sites, sediment *E. coli* levels declined by 48% from peak levels whereas wetland water declined by 93%. They attributed this difference to nutrient inputs from wildlife, as well as the vulnerability of water-borne microbial populations to predation and UV radiation.

### Sediment Texture

Survival studies, in general, demonstrate slower die-off of *E. coli* and FC in sediments containing larger amounts of clay and silt particles (e.g., Hoyle

et al., 1996). The better survival of *E. coli* in sediments containing at least 25% clay observed by Burton et al. (1987) was interpreted by Davies and Bavor (2000) as the result of better protection from bacterial predators. It was suggested that the location of soil bacteria in small pores, from which the predators were excluded due to their larger size, provided the bacteria with significant protection from predation (Decamp and Warren, 2000). On the other hand, direct SEM observations of bacteria in intertidal sediments (DeFlaun and Mayer, 1983) showed that bacteria inhabited shallow depressions on sand and silt grains; they were not found on grains smaller than about 10  $\mu\text{m}$  or inside smaller pores such as those on weathered feldspar grains. Three hypotheses were proposed to explain this pattern. First, unless the particle is appreciably larger than the bacterium, it will not provide protection against resuspension by fluid shear forces in a high-energy environment. Second, there will be little room for colony formation on small grains. Third, clay grains in these sediments have very smooth surfaces, with virtually no shallow depressions apparently so favored by bacteria.

The presence of smectite clay minerals appear to stimulate growth and survival of *E. coli*. Montmorillonite clay minerals have been shown to provide protection to *E. coli* from phage attack at various salinity levels, and a physical protection (Roper and Marshall, 1974). In laboratory experiments with dialysis chambers, Faust et al. (1975) observed a 40% increase in the *E. coli* survival (half-life;  $t_{50}$ ) in the estuarine water from Rhode Island river when 50  $\text{mg L}^{-1}$  of montmorillonite was added. An increase of the montmorillonite concentration to 500  $\text{mg L}^{-1}$  did not further change the  $t_{50}$  in this work. Among the reasons for the effect, the authors mentioned sedimentation and flocculation may play an important role in the removal of bacteria from the water column, and that attachment of bacteria to particulates can aid in their preservation.

A sheer presence of fine solid material devoid of biota may stimulate initial *E. coli* growth. This effect was discovered by Gerba and McLeod (1976) with marine sediments. Laliberte and Grimes (1981) showed that freshwater sediments had the same effect. They observed a more than two-log increase in *E. coli* inoculum population in autoclaved sand and silty clay sediments bags submerged for three days in Lake Onalaska. Adding sterile sediment to river water caused the *E. coli* population growth, which was more pronounced with larger amount of sediment added (Desmarais et al., 2002). Growth of about 3 orders of magnitude was observed during the first day of incubations. *E. coli* populations were about an order of magnitude larger than the original populations after 4 days of incubations. Comparison of *E. coli* growth rates in two different sediments indicated that the higher organic content and a greater fraction of fines were more conducive for the initial growth observed in this work.

## Association of *E. coli* and FC With Particles in the Sediment Bed

The persistence of *E. coli* in sediments is strongly influenced by the ability of strains to attach to sediment particles and subsequently form biofilms. For example, Schillinger and Gannon (1985) noted that broth-grown *E. coli* cells more readily attached to sediment particles than nutrient agar-grown pure cells. Based on electron micrographs of cells, the authors suggested that the presence of fimbriae (pili) on broth-grown was responsible for enhanced attachment.

The association of *E. coli* with mineral particles apparently takes time to develop. Hartz et al. (2008) observed the tenfold increase in the number of *E. coli* cells attached to silica grains and CaCO<sub>3</sub> grains increased significantly from Day 1 to Day 4 of the incubation.

Biofilm formation can be an important factor for increased survival of bacteria in sediments. Alimova et al. (2006) demonstrated that the presence of smectite clays enhances the formation of *E. coli* biofilms; *E. coli* populations in the clay mixtures were greater than the respective populations in media without clay. Smectite-bearing clay slurries developed bacteria–clay aggregates with a substantial biofilm component within 24 hr, whereas the exclusively bacterial suspensions did not develop any observable biofilm component. The biofilm–clay aggregates varied in size from tens of micrometers to several millimeters. Banning et al. (2003) studied the interactions between *E. coli* inocula and biofilms of indigenous microorganisms introduced with ground water. *E. coli* colonized all layers of the mixed population in this study. Experiments on the development and persistence of biofilms in stream beds (Packman et al., 2002) should provide a better understanding of the role of biofilms in *E. coli* survival.

## Diversity of *E. coli*

Survival studies by Anderson et al. (2005) have demonstrated very significant changes in *E. coli* population structures associated with sediment. Strain-dependent variability of *E. coli* growth and survival has been shown in soils (Topp et al., 2003). Smallbeck and Bromel (1975) showed that *E. coli* survival in sediment after reinoculation was much better than after the first inoculation.

Strain-specific *E. coli* survival may be a significant issue with regard to pathogenic *E. coli*. Very little is known so far about the prevalence of pathogenic *E. coli* strains in freshwater sediments. An intensive survey of generic and pathogenic *E. coli* at numerous sites throughout the Salinas Valley, CA watershed, conducted just prior to a major spinach-related outbreak of *E. coli* O157:H7, resulted in only one instance of *E. coli* O157:H7 detection in sediment samples (Cooley et al., 2007). The detected strain was not similar to clinical strains associated with the outbreak. The authors concluded

that, in general, generic *E. coli* is a poor indicator of the *E. coli* O157:H7 occurrence.

## 5. RESUSPENSION AND SETTLING OF *E. COLI* AND FC

### Impact of Resuspension on Water Column

As previously described, resuspension of sediments may account for much of the measured water-borne FC and *E. coli* during or shortly after rainfall events. The concentrations of *E. coli* and FC in streams during storm events are usually 2–3 orders of magnitude higher than in the base flow conditions (e.g., Hunter et al., 1992). Mechanical disturbance of bottom sediments can cause increased *E. coli* concentrations in the overlying waters as a result of their resuspension (e.g., Lopez-Torres et al., 1987; Seyfried and Harris, 1986; Stephenson and Rychert, 1982). At least three resuspension mechanisms can act during the high flow events (Wilkinson et al., 2006). A steep-fronted wave, with wave height much greater than the preceding water depth, can effectively suck organisms from the bottom sediment and hold them in the turbulent wave front. A less steep front or falling wave can lift organisms but not draw them in the wave overrun. Finally, the steady-flow stochastic erosion of bed and bank sources, resulting from high flow turbulence, can maintain FC concentrations elevated above those encountered at lower rates of flow.

Time series of *E. coli* concentrations in stream water and discharge are not necessarily strongly correlated, and such correlations do not give a definite answer on the contribution of sediments in the bacteria budget of the stream. Gannon et al. (2005) observed a strong increase of water-borne bacterial concentration after one rainfall event, but much lower increases during subsequent events in the Oldman River Basin, Alberta, Canada. They concluded that the relatively weak correlation between discharge and *E. coli* counts (Pearson's correlation of 0.594) suggested that the peaks in *E. coli* did not originate solely within in-stream sediments.

The direct method to observe the effect of sediment resuspension on bacteria concentration in water was proposed by McDonald et al. (1982) who carried out multiple releases of water to releasing water from reservoir to a stream in North Yorkshire; water releases from reservoirs is a common water management technique (Henson et al., 2007). Experiments were conducted after rainless periods to isolate the effects of stream channel entrainment from those induced by rainfall on the land surface. They observed 1–2 orders of magnitude increase of bacteria concentration during the water release due to *E. coli* resuspension, and noted the delay between the hydrograph and bacterial response. No correlation between turbidity and FC concentration was found in this work. Creating a pulse of water to observe resuspension is referred to as *pulsed flow event* or *artificial flood*.

Artificial flood experiments were carried out by Nagels et al. (2002) and Muirhead et al. (2004) in New Zealand. Muirhead et al. (2004) observed that *E. coli* concentrations peaked ahead of the flow peak, consistent with the entrainment of FC into the water column from underlying contaminated sediments by accelerating currents on the rising limb of the hydrograph. An increase of 2 orders of magnitude was observed during the event. *E. coli* concentrations correlated with turbidity over the flood event ( $R^2 = .92$ ) and both variables. However, while the turbidity returned to base levels between each flood, the *E. coli* concentrations remained somewhat elevated. Bacterial peak concentrations and yields declined systematically through a triplicate artificial flood series in this work. By projecting this decline to the infinite number of floods the authors arrived at a total in-channel *E. coli* population of  $10^8$  CFU  $m^{-2}$  of streambed area. An attempt to validate this estimate by direct measurements of *E. coli* in sediment was only partially successful, as only few sites (associated with cattle crossings) had areal concentrations as high as  $10^8$  CFU  $m^{-2}$ , consistent with flood yields. Davies-Colley (2007) described storm chasing studies that showed that FC pollution in streams typically peaks well ahead of hydrograph peaks. This timing reflects bacteria being resuspended from sediments by accelerating currents rather than wash in (which arrives later).

Jamieson et al. (2005a, 2005b) seeded a section of the stream with a specific strain of *E. coli* and observed its transport downstream. The presence of the tracer bacterium within the water column was directly linked to stormflow events. *E. coli* NAR was recovered at all sampling stations during the rising limb of storm hydrographs at 225, 550, and 600 hr after the source cell was seeded. The appearance of the tracer bacterium coincided with increases in total suspended solids, indicating that the *E. coli* NAR that were being resuspended were sediment-associated; observed resuspension rates were in the order of  $10^4$  cells  $m^{-2} s^{-1}$ .

Observations of storm-related bacteria loading in beach areas of lakes led to the conclusion that resuspension rates are smaller than in rivers and streams. Measurements by Palmer (1988) at three Toronto waterfront beaches resulted in FC coliform loadings in the range of 0–1410 CFU  $m^{-2} s^{-1}$ .

Dramatic increases in water-borne FC and *E. coli* due to resuspension, however, is dependent on the initial microbial populations in sediment. Gary and Adams (1985) observed FC in pools in the Hash Fork watershed in Wyoming; they found that while the FC concentrations in water ranged from 0 to 5 CFU per 100 mL of water, raking pool sediments never led to increase in bacterial densities more than 2 CFU per 100 mL.

### Impact of Resuspension on Temporal and Spatial Variability

The redistribution and subsequent settling of sediments accounts, in part, for the temporal and spatial variability of FC in stream sediments. Cinotto (2005)

monitored a stream with well-defined point sources of *E. coli* and found that, during storm flow, *E. coli* populations were similar along the entire length of the study reach: five samples from upstream, eight from the center of the reach, and seven from the downstream end of the reach. These data indicated that point sources (e.g., culverts and pipes, septic systems, wastewater treatment facilities) were not likely the origin of *E. coli* contamination during stormflow. However, the reserves of *E. coli* in sediment that can be activated by rainfall flushes may be limited. Jamieson et al. (2005b) noted that the supply of bacteria available for resuspension was depleted on the rising limb of the storm hydrograph in their experiments, although populations regenerated between storm events. A similar depletion was documented by Henson et al. (2007) at some but not all observation sites during a controlled-pulse event on a California river. McDonald et al. (1982) monitored two consecutive high-flow events and noted that there was a substantial peak of bacteria concentrations after the first but not after the second event. In experiments in New Zealand, Nagels et al. (2002) and Muirhead et al. (2004) reported that *E. coli* yields produced by the second of two consecutive high-flow events were 25% and 52% of the first event yield, respectively. Evanson and Ambrose (2006) observed a substantial increase of *E. coli* concentrations in water and sediment during two consecutive rainfalls after a long drought, but no increase after the third rainfall. They concluded that after the flush of *E. coli* from the surface layers during the first two rain events) there was insufficient time for *E. coli* regrowth, resulting in subsequent low-level flushes during later periods of rainfall.

Roper and Marshall (1974) demonstrated that FC (especially *E. coli*) can desorb from benthic sediments under conditions of reduced salinity. Because dramatic fluctuations in salinity occur in the estuarine environments, desorption has become a highly plausible explanation for why FC counts often increase after significant freshwater flow into these areas (Erkenbrecher, 1981).

### Microbial Attachment to Suspended Sediments

It is essential to know what fraction of bacteria is attached to suspended sediment because this fraction will settle with the sediment. Two contradictory views exist with respect to the numbers of *E. coli* or FC associated with suspended sediments. A large number of researchers have reported relatively small proportions of bacteria attached to suspended solids. For example, Schillinger and Gannon (1985) reported that about 16% of suspended coliforms were attached to suspended solids in storm water from an urban drain. Atwill et al. (2007) determined that during storm flow conditions, about 90% of the total amount of bacteria in a composite water sample were associated with the water fraction and only about 10% were attached to the suspended solids. Jamieson et al. (2005b) reported a range from 20



to 44% of *E. coli* associated with the suspended sediment during an artificial flow event. Jeng et al. (2005) found that 20–30% of FC and *E. coli* were associated with sediment particles after a stormflow release to a lake with brackish water. However, Auer and Niehaus (1993) estimated the number of bacteria associated with suspended solids to be about 90%.

The methods used to arrive at estimates of bacterial attachment to suspended sediments may be responsible for the discrepancies observed in different studies. One way to estimate the number of bacteria associated with particles is to use filters. Gannon et al. (1983) and Schillinger and Gannon (1985) demonstrated that a significant fraction of FC in the water column were retained on 52  $\mu\text{m}$  or 30  $\mu\text{m}$  screens. Auer and Neuhaus (1993) applied the same method and used seven filters of different mesh sizes to determine the differences in bacteria attachment to particles of different sizes; FC were predominantly associated with particles in the size range 6  $\mu\text{m}$  to 10  $\mu\text{m}$  in this work. Characklis et al. (2005) suggested that free and particles bound bacteria could be separated using centrifugation. To find the appropriate regime of centrifugation, they used latex particles (10  $\mu\text{m}$  mean diameter, density of 1.05  $\text{g}/\text{cm}^3$ ) as a surrogate for free-phase microbes and organic particles, and glass beads (5–50  $\mu\text{m}$  size range, density of 2.5  $\text{g}/\text{cm}^3$ ) as a surrogate for inorganic particles (e.g., clays, silicates). Their final procedure involved spinning the samples at 1164g for 10 min followed by braking for 4 min. After applying this regime to water samples, they concluded that on average 20–35% and 30–55% of total *E. coli* cells were associated with suspended particles in background samples and in storm samples, respectively. Using the same centrifugation technique, Cizek et al. (2008) determined the percentage of solids-associated *E. coli* to be between 10 and 35% with the 95% quantiles from 0 to 70%. Sampling that was not timed to storm flow events resulted in the average percentage of bacteria associated with suspended sediment varying in the wide range from 4 to 95% in the work of Sayler et al. (1975).

The effect of concentration of suspended solids on the percentage of bacteria associated with suspended solids remains unclear. Sayler et al. (1975) did not find a significant correlation between the total suspended sediment concentration and the percentage of FC associated with the suspended solids. On the other hand, George et al. (2004) showed an approximately linear increase of the percentage of *E. coli* associated with suspended particles > 5  $\mu\text{m}$  from 0 to 80% as suspended solids increased from 0 to 80  $\text{mg L}^{-1}$ . If the preferential attachment of *E. coli* to specific textural fractions occurs in streams and lakes, correlations may exist between specific textural fraction contents and the percentage of bacteria associated with suspended solids.

Turbidity should be correlated with the *E. coli* counts in water if the majority of bacteria are associated with suspended solids, no other water quality parameters affect the association, and the microbiological analysis does not involve filtering off suspended solids with bacteria associated with

them. Strong and very weak relationships have been observed by various authors between *E. coli* concentrations and turbidity, and between *E. coli* concentrations and total suspended solids concentrations. The absence of correlation between turbidity and FC concentrations observed by Goyal et al. (1977) across a network of coastal channels was explained by the fact that in shallow water bacteria could be resuspended due to decrease in salinity whereas in deep water the actual suspension of sediment material would be required to cause a concurrent increase in the number of organisms in the water column.

Experiments on attachment of *E. coli* to soil particles bear relevance to the attachment of these bacteria to suspended sediment particles. The commonly used approximation assumes the partitioning according the linear dependence

$$S = K_d C \quad (3)$$

where  $S$  is the amount of microorganisms associated with solid particles, count  $\text{g}^{-1}$ ;  $C$  is the concentration in runoff, count  $\text{mL}^{-1}$ ; and  $K_d$  is the partitioning coefficient. Partitioning of microorganisms in batch experiments is usually studied using centrifugation to separate solid and liquid particles. Duration and speed of centrifugation affects the distributions of *E. coli* between solids in sediment and supernatant (e.g., Ling et al., 2003). It is presently unclear whether the kinetics of attachment or the kinetics of straining of bacteria bombarded by settling soil particles is the factor affecting the solid-liquid distribution. With the long centrifugation, the attachment of bacteria to soil particles can be efficiently simulated with Equation (5) when  $K_d$  is assumed to be a function of the clay content in soil. Data of Ling et al. (2003) have been approximated (Pachepsky et al., 2006) as

$$K_d = A \cdot \text{CLAY}^B \quad (4)$$

where CLAY is the percentage of clay particles  $< .002$  mm in soil, Equation (4) was developed for  $2 < \text{CLAY} < 50$ , and parameters  $A = 10^{-1.6 \pm 0.9}$ , and  $B = 1.98 \pm 0.7$  are the slope and the intercept of the regression in log-log coordinates. Although clay is often assumed to be the leading variable of the attachment, properties of particle surfaces may substantially modify clay effect. Guber et al. (2007) studied the attachment of *E. coli* to soil particles of different sizes and found that FC attachment to soil, silt, and clay particles having no organic matter coating was much higher than to similar sand particles, whereas the attachment to organic matter coated sand particles was comparable to the attachment to coated silt and clay particles.

Schematic subdivision of resuspended bacteria into free-floating and attached to individual solid particles may be misleading. Bacteria frequently persist in the environment in heterogeneously distributed biofilms. Very little is known about the relative stability of sediment biofilms. Pettibone et al.

(1996) observed that a few large flocculated particles accounted for most of the volume of resuspended sediment-borne FC in a study conducted with waters from the Buffalo River before and after ship passage. Particle size distribution in the upper water column was determined by automated image analysis; the median particle size class, by volume, before and after the ship passed, was 48.9–56.6  $\mu\text{m}$  and 366.5–391  $\mu\text{m}$ , respectively. Resuspension of flocculated, bacteria-laden particles affects bacteria transport and deposition as well as survival. Sediment flocs can be a dominant form of sediment transport in freshwater fluvial systems (Droppo and Ongley, 1994).

### Settling Rates and Mechanisms

Because bacteria are small, their settling rates are extremely slow (1.6  $\text{m d}^{-1}$  as estimated by Cizek et al., 2008). However, settling rates of sediment-associated bacteria are significantly higher due to the density of sediment particles (Gannon et al., 1983). In laboratory experiments, Stott et al. (2008) observed that *E. coli* settling rates were similar to the decline of turbidity induced by kaolin clay suggesting that the exchange of *E. coli* in the stream subsurface reflects that of fine sediment. Clearly, as sediments and attached bacteria settle back to the creek and stream bottoms, FC and *E. coli* concentrations in the water column will necessarily decrease (Matson et al., 1978). Gannon et al. (1983) assessed the effectiveness of storm water impoundment for removal of FC. They observed much higher percentages of bacteria attached to particles (greater than 5  $\mu\text{m}$  in size) where the river entered the impoundment as compared with sites further down the impoundment. They concluded that sedimentation was an important element in the overall FC decline in the upper end of the impoundment. Davies and Bavor (2000) directly related settling to the higher *E. coli* removal efficiency of constructed wetlands as compared with the pollution control pond. They observed that bacterial concentrations did not change while silt particles settled; they assumed that bacteria were predominantly adsorbed to clay (<2  $\mu\text{m}$ ) particles and settled with them. Given that 2  $\mu\text{m}$  is the characteristic size of *E. coli* cell per se, mechanisms other than adsorption need to be researched to explain these differences.

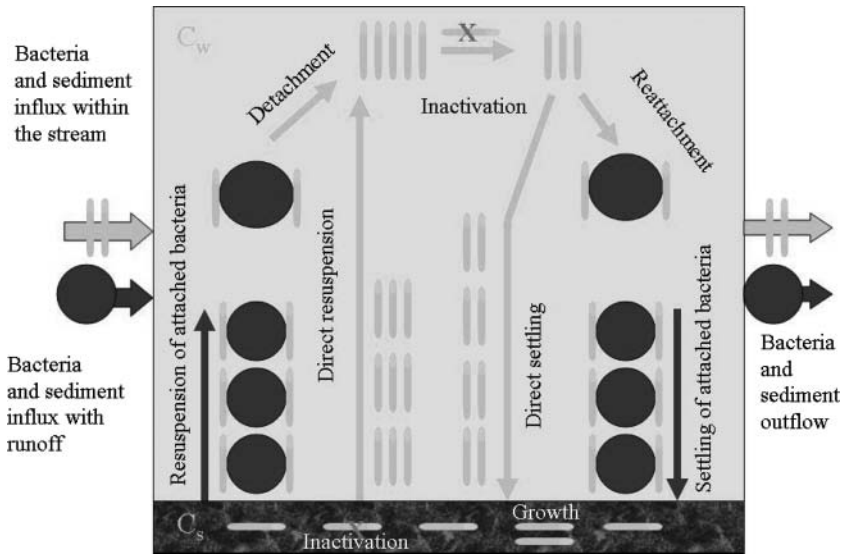
Estimates of settling rates for suspended *E. coli* have been obtained with several methods. Auer and Neuhaus (1993) suggested that, for bacteria associated with suspended particles, the settling rates of bacteria should be related to the size of particles they are associated with. They suggested two values of settling velocity: 2.40  $\text{m d}^{-1}$  for large particles (>10  $\mu\text{m}$ ) and 1.17  $\text{m d}^{-1}$  for small particles (0.45–10  $\mu\text{m}$ ). Canale et al. (1993) used the value of settling velocity of 1.38  $\text{m d}^{-1}$  to simulate the FC distribution in Onondaga Lake. Using similar assumptions, Jamieson et al. (2005b) arrived at the values of the settling velocity in the range from 2.6 to 25  $\text{m d}^{-1}$ . Cizek et al. (2008), using a centrifugation method, estimated settling velocities to be in the range

4.2 m d<sup>-1</sup> to 13.1 m d<sup>-1</sup>. Jeng et al. (2005) found that *E. coli* and FC were attached predominantly to particles and computed settling velocities of 6.5 and 7.9 m d<sup>-1</sup>, respectively. Similar assumptions led Liu et al. (2006) to a settling velocity value of 5 m d<sup>-1</sup>. These estimates are 0.5–1 order of magnitude larger than possible gravity-driven settling of bacteria. Disturbance by dredging, on average, doubled FC concentrations in Mississippi water (Grimes, 1975); concentrations remained above the initial value 5 days after dredging ended. This indicates that FC settling velocities can be much less than most recently reported values between 4.2 m d<sup>-1</sup> and 13.1 m d<sup>-1</sup> (Cizek et al., 2008).

The straightforward representation of settling in streams based on Stokes law (i.e., as the pure effect of the effects of viscosity and gravity) has been recently questioned (Ren and Packman, 2002; Salehin et al., 2004). Cooley et al. (2007) indicated that the hyporheic exchange (i.e., exchange through the a subsurface volume of sediment and porous space adjacent to a stream) can lead to high rates of suspended particle deposition in sediment beds, even when the suspended particles are very small and have no appreciable settling velocity. On the other hand, Jamieson et al. (2005b) remarked that the calibrated settling velocities observed in their study were 2 orders of magnitude lower than the corresponding Stokes fall velocities. They suggested that the high shear stresses occurring near the bed limited the number of particles that can actually bond to the bed without being reentrained. The rate of transfer of *E. coli* from surface water to stream gravel sediments depended strongly on the biomass of epilithic biofilms in the work of Stott et al. (2008). More needs to be learned to develop a coherent concept of settling, which is the essential process controlling pathogen and indicator bacteria in freshwater sources.

## 6. MODELING OF THE SEDIMENT EFFECTS ON FATE AND TRANSPORT OF *E. COLI* IN STREAMS AND WATER BODIES

A conceptual model of in-channel processes proposed by Matson et al. (1978) and substantiated by Yagow and Shanholtz (1998) provides a framework in which to view quantitative approaches to *E. coli* transport. Their model includes storage of bacteria in the channel sediment that achieves a relatively steady-state with respect to water column concentrations during base flow conditions. As runoff begins, discharge and velocity increase, scouring bacteria from the benthic surfaces. Water column concentrations increase from bacteria in the surface runoff and from the scoured sediment, whereas bottom sediment concentrations decrease. As soon as peak discharge occurs, bacteria concentrations in the water column decrease at a faster rate than discharge, depositing sediment further downstream, where the sediment



**FIGURE 6.** Processes that have been considered in stream *E. coli* fate and transport modeling. Yellow rods are symbols for bacteria, brown circles are symbols for sediment particles.

concentrations increase and water column concentrations decrease to approximately background levels.

Processes that have been considered in describing *E. coli* fate and transport models are depicted in Figure 6. Resuspension and settling, die-off, attachment, and detachment affect concentrations of free-floating bacteria and concentrations of bacteria in sediment. Influx with runoff and from the neighboring stream sections has been considered for free-floating and sediment-attached microorganisms. Similarly, bacteria can leave a section of stream either as free-floating or attached.

Modeling *E. coli* fate and transport in the Charles River, Massachusetts, without including resuspension and settling mechanisms, successfully captured spatial trends but appeared to be incapable of explaining changes in *E. coli* concentrations in water during and after rainfall events (Hellweger and Masopust, 2008). McCorquodale et al. (2004) simulated *E. coli* fate and transport following storm water discharges in the coastal Lake Pontchartrain, Louisiana, and included in the model the description of settling, but not resuspension. They assumed that a constant fraction of total bacteria count is attached to suspended solids and this fraction settles at a constant rate. The authors indicated that the possibility of resuspension of these bacteria 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 FC 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.

Other models simulated resuspension and settling at various levels of complexity. Earlier models did not distinguish between transport of free-floating and sediment-associated bacteria. Wilkinson et al. (1995) simulated an artificial high flow event and proposed to partition the sediment bacteria storage into a number of substorages. Which substorages are undergoing resuspension and which are enriched by settling is determined from the present flow rate, maximum observed flow rate, and minimum observed flow rate. The resuspension rates for a given time interval  $\Delta t$  are equal to the number of resuspension storages multiplied by the organism contents in these storages and the washout fraction parameter. Settling occurs at a rate proportional to total concentration. The total settled amount is equally distributed between substorages that are enriched by settling at the present flow rate. The transport was simulated as the joint effect of convection and dispersion, the latter attributed to the existence of mixing zones with the stream.

Tian et al. (2002) simulated *E. coli* fate and transport at the watershed scale. The authors did not distinguish between free-floating and sediment-associated *E. coli* and defined a threshold flow rate value that separated the depositional and the resuspension flow regimes. The fraction of deposited *E. coli* grew as the flow rate was below the threshold and approached zero. The resuspension rate was proportional to the concentration of *E. coli* in the sediment and, the proportionality coefficient nonlinearly grew as the flow rate increased.

Steets and Holden (2003) simulated the fate of runoff-associated FC through a coastal lagoon in California. They used a single value of FC concentration in water and relied on the description of deposition as a function of flow rate and total suspended solids. The authors assumed that about 90% of suspended FC cells were associated with suspended sediment and used the settling velocity for fine-grained sediments to estimate the deposition of FC. The estimate of resuspension rate was made by assuming the sediment erosion rate and the depth of erodable sediment of 0.05 m for a storm event. Steets and Holden (2003) addressed the question of redistribution of settled bacteria assuming that the fraction of FC that are particle-associated become concentrated in the sediment, beginning at the sediment–water interface. Specifically, they assumed that all particle-associated FC enter the sediment via a Stoke's Law process where they become immediately concentrated in proportion to the ratio of sediment bulk density to total suspended solids in the water column. No die-off for bacteria in sediment was assumed, but a removal was suggested based on the constant thickness of the sediment layer where the resuspendable FC reside.

Collins and Rutherford (2004) simulated *E. coli* watershed-wide fate and transport in the Northern New Zealand. They suggested that the resuspension rate should be simulated as a power function of the mean daily flow rate, and the deposition rate could also be estimated as the power function of

that rate with the negative exponent. With this approach, a threshold value of the flow should be used at which no deposition would occur.

In a thoughtful review, Jamieson et al. (2004b) noted that to date no attempt has been made to assess the movement of microorganisms by directly modeling the sediment particles to which they are attached. The pattern and magnitude of bacteria resuspension and deposition in streams should be related to the sediment particles to which they are attached. This has been addressed in models developed recently. Bai and Lung (2005) simulated the artificial high flow events, and, following the suggestion of Chapra (1997), modeled the partitioning of bacteria between water and suspended sediment as the result of the linear reversible adsorption according to Equation (3). Transport of both sediment and free-floating bacteria was simulated as convective dispersion. The settling of free-floating bacteria was neglected. The resuspension of bacteria from the bottom sediments and the deposition of sediment with bacteria attached were defined by two values of the bottom shear stress. The deposition was simulated if the bottom shear stress was lower than the critical shear stress for deposition. The resuspension was simulated if the bottom shear stress is higher than the critical shear stress for resuspension. Yang et al. (2008) suggested that partitioning of sediment into cohesive and non-cohesive groups may further improve simulation results.

Jamieson et al. (2005b) also considered the transport of free-floating and sediment attached bacteria separately when simulating the injection of sediment associated *E. coli* during the artificial high flow event. Unlike the previous authors, they assumed the adsorption of *E. coli* to the sediment to be assumed to be irreversible. Transport of both sediment and bacteria was simulated as the joint effect of convection and dispersion, assuming that settling of sediment-associated *E. coli* occurred with the rate proportional to the depth of water, and that the free-floating *E. coli* had a negligible settling velocity.

Modeling of bacteria resuspension and settling is hampered by substantial uncertainties in parameters. The majority of authors, with the exception of Steets and Holden (2003), have agreed that after being resuspended bacteria remain predominantly free floating. In such a case, distinguishing between the models for *E. coli* attachment to suspended solids may be difficult given the accuracy of available experimental data and multiplicity of possible mechanisms of *E. coli* attachment to solid surfaces (Guber et al., 2005).

Another uncertainty in modeling is related to the value of the critical shear stress at the bottom above which sediment becomes resuspended and sediment bacteria enter flowing water. Values of the critical stress proposed by different authors vary substantially (e.g.,  $1.7 \text{ N m}^{-2}$  [Jamieson et al. 2005a],  $0.4 \text{ N m}^{-2}$  [Bai and Lung, 2005],  $0.02\text{--}0.1 \text{ N m}^{-2}$  [Steets and Holden, 2003]).

Lack of robust methods to predict sediment transport and lack of information with respect to the partitioning of bacteria between water and

sediment have been also mentioned as the impediment for explicit modeling differences in transport of free-floating and adsorbed bacteria separately (Jamieson et al., 2004b). None of the existing models quantifies *E. coli* exchange between water and sediment in the absence of sediment resuspension; the rate and impact of this process remain essentially unknown. No tested submodel of *E. coli* persistence in sediment has been proposed to date.

## 7. IMPLICATIONS FOR MANAGEMENT OF MICROBIOLOGICAL WATER QUALITY

The presence of bottom sediments containing large, unquantified reservoirs of fecal pollution indicator bacteria introduces substantial uncertainty in detection, monitoring, and control of microbiological water quality and stream impairment. Cinotto (2005) used data from two years of exhaustive monitoring of water and sediment to conclude that the bacteria concentrations observed during stormflow events probably result from remobilized sessile bacteria stored within fluvial sediments. In this case, these bacteria should not be considered indicators of current fecal contamination. Meays et al. (2006) arrived to similar conclusions based on data from three streams that were sampled for *E. coli* each 15 min for 24 hr at three different days.

### Microbiological Monitoring of Sediments

There are no established methods for the microbiological monitoring of sediments. Unless the sediment is reworked by surf, bacteria can be resuspended only from the first centimeter of sediment. Therefore, sampling this particular thin skin layer of sediment appears to be of utmost importance. Large *E. coli* populations in sediment may enhance the feasibility and ease of the direct determination of sediment-bound pathogens by molecular biologic techniques (i.e., techniques based on the analysis of the nucleic acid content [DNA and RNA] of pathogens [Toze, 1999]). Chawla et al. (2003) and Rose et al. (2003) demonstrated successful recovery and amplification of *E. coli* DNA from the river sediment. Ongoing research on the improvement of the extraction methods (chemical dispersion, sonication, or homogenization) should result in further advances. Another aspect of sediment analysis for *E. coli* has been addressed by Fish and Pettibone (1995), who demonstrated that enumeration of *E. coli* sediments using plate methods were essentially equivalent to direct count; therefore, growth-dependent methods of enumeration may be an accurate and economical practice to use in evaluating densities of enteric bacteria in sediment. A similar conclusion was reached by Bogosian et al. (1996) in experiments with river water and soil. This observation may not hold, however, for the survival experiments. Hood and Ness (1982) observed



in experiments on survival of *E. coli* in estuarine sediments that although *E. coli* most probable numbers were statistically the same as the viable counts, the direct counts were much higher. All attempts to resuscitate the intact *E. coli* cells by using lactose enrichment failed in this work.

Sandbag samplers (Nix and Merry, 1990) have shown promise as a means of evaluating indicator bacteria storage in sediments. Sandbag samplers rely on the ability of bacteria to attach to surfaces and use the larger sediment-associated bacteria populations instead of the more commonly used free-floating bacteria populations. Cinotto (2005) reported that *E. coli* concentrations observed in the sandbag samplers, after 1 week in place, were similar to those found within natural sediments collected concurrently.

### Sediment Bacteria and Microbial Source Tracking

The presence of *E. coli* bacteria populations in sediment substantially complicates attempts at microbial source tracking. Differences in survival rates between different strains conflicts with the underlying assumption of microbial source tracking that all characterized strains in the collection libraries have comparable survival rates (Anderson et al., 2005). The results of Gordon et al. (2002) suggest there are strains of *E. coli* that are better adapted to conditions found in the external environment compared to strains isolated from the gastrointestinal habitat. Further, the finding that the numerically dominant clones and clonal diversity in secondary habitats can differ substantially from those found in the source populations will confound efforts to identify the sources of fecal pollution in the environment. Therefore, comparisons of sediment samples in which significant differences were not found must be interpreted cautiously (i.e., real differences in persistence may exist that were not detected in these experiments). Differences in survival of indigenous and introduced *E. coli* strains in stream sediments have been demonstrated by Anderson et al. (2005). More work is needed to define the role of sediments in the interpretation of source tracking data.

### Wetlands and Flow Control Structures

The effect of wetlands on bacterial sediment storage needs to be further evaluated. Byannapali et al. (2003) compared *E. coli* levels in streams of different orders in delineated wetlands and concluded that extensive ditching of the wetlands had increased the stream order of Dunes Creek and altered drainage patterns, which presumably increased potential inputs to and subsequent *E. coli* loadings in the creek. The efficiency of constructed wetlands was directly related to the reduction of suspended particles in the work of Stenström and Carlander (2001). The authors indicated that resuspension should be prevented in order not reintroduce bacteria into the water.

Flow control structures demonstrate the strong effect of bacteria accumulation in sediments. Cinotto (2005) documented the effect of a large (20 feet high) dam on *E. coli* concentrations. The capacity of the larger dam to impede flows, combined with nutrients entering the reach, resulted in increased biologic activity throughout the impoundment area. Within this larger impoundment, and *E. coli* bacteria populations were observed to increase sharply as flow approached the dam crest. All bacteria levels were then observed to drop to background levels, in the water column and fluvial sediment, immediately downstream from the dam crest.

The practice of aquifer storage and recovery (ASR), which involves the injection of surface water into an aquifer, can create an environment which allows the rapid growth of indigenous and/or introduced bacteria in the aquifer, particularly when the injected water is high in nutrients (Pavelic et al., 2007). ASR provides a means of storing reclaimed wastewaters prior to reuse for purposes such as irrigation. During an ASR pilot project in South Australia using treated sewage effluent, bacterial growth during injection resulted in biofilm formation within the aquifer matrix immediately surrounding the injection well (Rinck-Pfeiffer et al., 2000).

### Best Management Practices

Evaluation of best management practices should factor in the effect of sediments on microbiological water quality. Agouridis et al. (2005) analyzed the case where streams impacted by grazing, or by greatly reduced sediment loading due to BMPs, may become unstable resulting in erosion. The creation of a bacterial reservoir may then occur as a result of the inability of unstable streams to transport sediment through the watershed. A significant reduction in peak flow associated with selective structural BMPs may also increase stream bed depositional processes depending on the sediment transport capacity of the stream. Removal of riparian vegetation can increase temperature of water and sediment and facilitate FC and *E. coli* bacteria growth and survival. Controlling the input of *E. coli*-bearing sediments to the Bay Maumee State Park, Ohio, was advocated after an extensive survey (Toledo Metropolitan, 2003).

Most of water conservation measures have the potential to decrease the sediment storage of pathogenic and indicator coliforms. Reduction of nutrient loads from agricultural and urban areas can decrease aquatic growth within impoundment areas, thereby reducing the potential for bacterial regrowth. Restoration of wetland and riparian zones may allow for infiltration of surface water runoff, thereby filtering out most bacteria by way of bank filtration. Implementing storm water runoff structures that promote infiltration and do not allow direct input of storm water from urban areas to enter the stream may reduce bacteria levels by means of sediment filtration. Reducing streambank and streambed erosion may reduce bacteria concentrations

during storm flows by minimizing the amount of sessile bacteria washed into the stream as the streambank and streambed are eroded. The explicit recognition of the importance of the sediment as the pathogen bacteria reservoir makes imperative the development of treatments specifically aimed on the reduction of the pathogen concentrations in the sediment. The efficiency of such treatments is evaluated by the changes of pathogen concentrations in water and sediment (Scholes, 2008).

### Irrigation and Flood Water Quality

Microbiological water quality has attracted attention as the substantial factor for food safety with regard to produce. Crohn and Bianchi (2008) advocated the need in comanagement of water quality and food safety with respect to water-borne pathogens and emphasized the research need and paucity of data on pathogen fate as related to storm flows, floods, and impoundments. Practices need to be identified to help growers determine when and if sediment from catch basins can be safely reincorporated onto the fields, when and if tailwater can be reintroduced to the field, and what are effects and feasibility of treating irrigation water and irrigation reservoirs to reduce pathogens. Progress in these directions hardly can be achieved without understanding of sediment function in microbial ecology of waters in agricultural use.

Overall, freshwater and estuarine sediments are important microbial habitats that may be critical contributors to water contamination; as such, they have received much less attention that they deserve. The demonstrated role of pathogenic *E. coli* strains in food and water quality challenges reinforces the need in better understanding ecological and hydrological factors that affect functioning of sediments as *E. coli* reservoirs.

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