PATHOGENS
in Urban Stormwater Systems

Prepared by
Urban Water Resources Research Council
Pathogens in Wet Weather Flows Technical Committee
Environmental and Water Resources Institute, American Society of Civil Engineers

With Support from
Urban Drainage and Flood Control District, Denver, CO
Urban Watersheds Research Institute

August 2014

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Preface

This report was completed to serve as a technical resource for local governments working to address elevated fecal indicator bacteria (FIB) in urban areas, particularly with regard to National Pollutant Discharge Elimination System (NPDES) Municipal Separate Storm Sewer System (MS4) permit requirements arising from FIB total maximum daily loads (TMDLs). The authors are primarily water resources engineers and water resources scientists, as opposed to microbiologists and epidemiologists. For this reason, the report focuses primarily on information needed to develop and implement plans for addressing elevated FIB in MS4s, as opposed to discussion of the biological underpinnings of human health risks from pathogens.

Acknowledgements

This report was completed by a voluntary committee of stormwater experts associated with the Urban Water Resources Research Council of the Environmental and Water Resources Institute of the American Society of Civil Engineers. The committee would like to thank Ken Mackenzie, P.E., of the Urban Drainage and Flood Control District in Denver, CO and Ben Urbonas, P.E., of the Urban Watersheds Research Institute for providing supplemental funding to enable timely completion of the report.

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John Stover, U.S. Department of Energy (retired)
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Acknowledgements of Previously Published Works

One of the challenges associated with addressing FIB in urban areas is the breadth of knowledge and understanding needed on multi-disciplinary topics including microbiology, engineering, experimental design, statistics, modeling, water quality regulations, and other topics. This report does not cover any one topic in detail, but instead integrates multi-disciplinary information into one general reference. Several chapters draw upon previously published material, which can be referenced for more detail on specific topics. These references include:

Burton, G.A. Jr., and R. Pitt. (2002). *Stormwater Effects Handbook: A Tool Box for Watershed Managers, Scientists, and Engineers*. ISBN 0-87371-924-7. CRC Press, Inc., Boca Raton, FL. 2002. 911 pages. www.epa.gov/ednnrmrl/publications/books/handbook/index.htm www.epa.gov/ednnrmrl/publications/books/handbook/index.htm

Center for Watershed Protection and R. Pitt. (2004). *Illicit Discharge Detection and Elimination: A Guidance Manual for Program Development and Technical Assessments*. Center for Watershed Protection, Ellicott City, MD, and University of Alabama, Tuscaloosa, AL. http://www.cwp.org/online-watershed-library/cat_view/64-manuals-and-plans/79-illicit-discharge-detection-and-elimination

City of Santa Barbara, Creeks Division, in Partnership with University of California, Santa Barbara. (2012). *Tools for Tracking Human Fecal Pollution in Urban Storm Drains, Creeks, and Beaches*. September 1, 2012. Accessible at: http://www.santabarbaraca.gov/civicax/filebank/blobdload.aspx?BlobID=16722 and http://www.santabarbaraca.gov/civicax/filebank/blobdload.aspx?BlobID=16724

Griffith, J.F., Layton, B.A., Boehm, A.B., Holden, P.A., Jay, J.A., Hagedorn, C., McGee, C.D., and S.B. Weisberg. (2013). *The California Microbial Source Identification Manual: A Tiered Approach to Identifying Fecal Pollution Sources to Beaches.* Technical Report 804 - December 2013. Prepared for the Southern California Coastal Water Research Project. Accessible at: http://www.waterboards.ca.gov/board_info/agendas/2014/jan/012114_11_804_sipp_mst_manual.pdf

International Stormwater BMP Database, including various statistical summaries such as Wright Water Engineers and Geosyntec. (2010). *International Stormwater Best Management Practices (BMP) Database Pollutant Category Summary: Fecal Indicator Bacteria*, and 2012 statistical appendices. Project sponsors include: Water Environment Research Foundation, Federal Highway Administration, American Society of Civil Engineers Environmental and Water Resources Institute, U.S. Environmental Protection Agency and American Public Works Association. Accessible at: www.bmpdatabase.org

U.S. Environmental Protection Agency. (2012). *Recreational Water Quality Criteria*. Office of Water 820-F-12-058. http://water.epa.gov/scitech/swguidance/standards/criteria/health/recreation/

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Acronyms and Abbreviations

ANOVA	analysis of variation
BMP	best management practice (also known as stormwater control practice)
CCTV	closed circuit television
CDC	Centers for Disease Control
cfu	colony forming units
COV	coefficient of variation
CSO	combined sewer overflow
CLRP	comprehensive load reduction plan
ESD	environmental site design
EPA	U.S. Environmental Protection Agency
FIB	fecal indicator bacteria
GIS	geographic information system
LA	load allocation
LTCP	long-term control plan
LRP	load reduction plan
MCA	microbial community analysis
MOS	margin of safety
MPN	most probable number
MS4	municipal separate storm sewer system
MST	microbial source tracking
NEEAR	National Epidemiological and Environmental Assessment of Recreational Water
NPDES	National Pollutant Discharge Elimination System
NSE	natural source exclusion
NSQD	National Stormwater Quality Database
PCA	principal component analysis
QMRA	quantitative microbial risk assessment
WWTP	wastewater treatment plant
qPCR	quantitative polymerase chain reaction
RWQC	Recreational Water Quality Criteria
SCCWRP	Southern California Coastal Water Research Project
SSM	single sample maximum
SSO	sanitary sewer overflow
STV	statistical threshold value
TIL	tolerable illness level
TMDL	total maximum daily load
UAA	use attainability analysis
UV	ultraviolet
VBNC	viable but not culturable
WHO	World Health Organization
WLA	wasteload allocation

EXECUTIVE SUMMARY

The single most frequent cause of water quality impairment in the U.S. is elevated fecal indicator bacteria (FIB) (EPA 2014). FIB-related impairments can have significant and costly implications for local governments, businesses, and watershed stakeholders due to beach closures and total maximum daily load (TMDL) compliance and implementation requirements to address these impairments. TMDLs and associated municipal separate storm sewer system (MS4) NPDES permit requirements for FIB load reductions pose unique challenges relative to TMDLs for chemical constituents. FIB are living organisms that occur naturally in the environment and whose sources can move freely throughout watersheds and storm drain systems, even when anthropogenic sources of FIB are controlled. Furthermore, FIB are generally not a direct cause of human health impacts; instead, they are easy-to-measure surrogate parameters that are intended to infer that fecal wastes and associated pathogens may be present. Nonetheless, FIB are currently considered to be the best available practical alternative to monitoring for multiple pathogens associated with human and animal wastes. Although the human health risk associated with exposure to waters impacted by untreated or poorly treated human sewage is well documented, the health risk from recreational exposure to elevated FIB in urban runoff-impacted receiving waters is less well known.

The state of the art and practice in modeling transport and fate of FIB (and pathogens) involves significant uncertainty, more so than traditional water quality constituents. This uncertainty carries forward into evaluation of FIB management strategies, development of appropriate wasteload and load allocations for TMDLs, and regulatory decisions. Nonetheless, MS4 owners/operators are often assigned wasteload allocations in urban FIB TMDLs and may face significant wasteload reduction requirements, which are enforceable through MS4 discharge permits. Although management and correction of human sources of FIB (e.g., leaking sanitary infrastructure, illicit connections, dumpster drainage) to storm sewer systems can reduce FIB loads posing human health risk, many MS4s will need to reduce FIB from other sources as well to meet wasteload reduction targets. Identifying the sources of FIB and their relative contributions can be complex and costly. Load reductions are difficult, especially for the natural, non-human FIB sources, for multiple reasons (e.g., ubiquitous nature of FIB, current limits of technology related to urban stormwater controls, magnitude of reductions targeted). For these and other reasons, there are real questions regarding the attainability of FIB water quality standards in urban watersheds and in MS4 discharges. Depending on the sources of FIB affecting a particular receiving water and the manner in which MS4 permit compliance is assessed, dry weather standards may be attainable in some cases, but consistently attaining standards under wet weather conditions may be infeasible.

To support MS4 permit holders and watershed stakeholders in developing realistic goals and effective strategies for addressing pathogens in urban stormwater systems, this report consolidates information on many facets of FIB impairments, providing information on the following topics:

- Basic background related to regulatory context, pathogens in receiving waters and the use of FIB as surrogates for pathogens.

- Sources of pathogens in the urban environment.
- Transport and fate issues, along with the factors affecting survival of pathogens and FIB. Although an evaluation of models for FIB is beyond the scope of this report, understanding of transport and fate issues affects the ability of water resources scientists and engineers to develop models for FIB.
- Approaches for monitoring, source tracking and evaluating FIB and pathogen data, including a discussion of challenges associated with these activities.
- Source controls and treatment strategies, including expected effectiveness, data gaps and practical constraints related to source controls, structural stormwater controls, and disinfection.
- Case studies illustrating challenges and approaches to implementing and complying with FIB TMDL requirements in urban areas.
- Conclusions and recommendations for additional applied research needs related to pathogens in urban stormwater systems and complying with FIB and/or pathogen TMDLs.

A summary of key background information and findings of this report includes:

1. In 2012, EPA updated the Recreational Water Quality Criteria (RWQC), which established health-based water quality criteria intended to protect human health in the context of primary contact recreation in streams and lakes. These criteria serve as guidance for states for purposes of developing water quality standards. The criteria are based on epidemiological studies conducted primarily at lake and ocean beaches at locations affected by FIB and pathogens associated with sources mostly of sanitary (human) origin.

2. Epidemiological and quantitative microbial risk assessment (QMRA) studies regarding human health risks associated with recreational activities in urban runoff impacted receiving waters, particularly during wet weather, remain limited, and conclusions regarding human health risks associated with urban stormwater systems are mixed. Additionally, EPA-sponsored literature reviews and QMRA studies have shown that human health risks associated with zoonotic (animal) sources of FIB and pathogens may vary depending on a variety of factors. Although many experts agree that non-human sources of FIB and pathogens generally pose a lower risk of human illness than human sources, EPA did not have adequate information to provide national source-based exclusions in the 2012 RWQC, and instead developed risk-based criteria based on specific gastrointestinal illness rates.

3. Receiving waters with primary contact recreation use classifications in most urbanized areas must comply with standards based on the RWQC, regardless of the source of FIB. However, under the 2012 RWQC, EPA allows options for development of site-specific standards that provide equivalent protection to EPA's recommended criteria. These alternative standards generally become a viable option only after human sources of FIB have been controlled. Scientific methods that can be used to support alternative standards

generally include either epidemiologic studies or QMRA. Although QMRA is less costly than an epidemiological study, both approaches require significant scientific expertise and are expensive to implement. Sanitary surveys, possibly including microbial source tracking techniques, are also important evidence needed for developing site-specific standards in urban areas.

4. Sources of FIB in urban environments can include both human and non-human sources. A variety of source identification approaches can be used, depending on local conditions and budgets. The first step in addressing FIB impairments is to inventory the various FIB sources specific to the watershed, and prioritize human FIB sources first, given the greater public health risks they may present. Although municipal wastewater treatment plants (WWTPs) are not typically a significant source of elevated FIB in urban receiving waters, sanitary sewer collection systems can contribute human waste, particularly in areas with aging infrastructure (e.g., leaky sewer lines), sanitary sewer overflows (SSOs), or combined sewer overflows (CSOs). Other urban sources of human waste include homeless, RV discharges, and septic systems. The second management priority is control of non-human anthropogenic sources contributing to FIB loading, which include pet waste, fertilizers, trash, and dumpster leaks, to the extent that they are controllable. The third and lowest priority of FIB control is non-anthropogenic sources, which include urban wildlife, plants, soils, and decaying organic materials. Recent scientific advances in MST allow fecal sources to be more reliably and quantitatively identified, with validated source markers available for such categories as human, canine, gull, horse, pig, and ruminant. Such tools can be used to support a comprehensive source identification investigation, where conditions warrant advanced investigations.

5. FIB concentrations in wet weather urban discharges from separate storm sewer systems are typically orders of magnitude above primary contact recreation standards, regardless of the land use. FIB in dry weather urban runoff may also be elevated, depending on site-specific conditions. FIB in waters receiving runoff from natural areas may also sometimes exceed primary contact standards. Regulatory flexibilities based on high-flow recreational use suspensions and allowable exceedances frequencies based on reference (natural) watershed conditions vary depending on state regulations, but are not explicitly addressed in the federal RWQC.

6. FIB monitoring results, given their large variability, do not provide the statistical confidence or power necessary to make statistically significant conclusions, such as regarding spatial or temporal patterns, unless very large numbers of samples are available. FIB sources, fate, and transport dynamics contribute to this large variability in concentrations. FIB are living organisms that die-off, grow, and persist, depending on environmental conditions. For example, particle-associated FIB may settle out of the water column and persist (and reproduce) in sediments for long periods of time, then be resuspended in the water column during periodic high flows. Additionally, FIB sources vary seasonally and may change over short time periods. For example, illicit discharges may be intermittent, and stormwater discharges occur episodically. For this reason, it is critically important that decisions for TMDLs and proposed control strategies be based on robust data sets that represent each critical period. Monitoring to identify or confirm the absence of human sources should be a high priority. This typically includes dry-weather

sampling of storm drain outfalls, visual and/or CCTV inspection of storm drain networks, and receiving water monitoring programs to identify areas where more intensive source monitoring may be needed.

7. Urban stormwater quality mathematical/computer models, such as watershed models that are typically used for TMDL development and/or implementation, have more limited predictive capability for FIB than for other conventional urban stormwater pollutants. This is due to the relatively smaller input datasets (such as regional land use event mean concentrations), as well as the greater uncertainty regarding FIB sources, fate and transport (parameters which, unless directly measured, require calibration to match receiving water monitoring data). Robust monitoring datasets are needed for model setup, calibration, and verification; however, watershed-specific datasets are often costly to develop. Where regional or national datasets are used (such as for land-use based concentrations), interpretation of model results should carefully consider results of sensitively and uncertainty analyses, and should recognize current limitations of the state of the practice. Thus, watershed modeling studies for FIB should place an emphasis on the development of robust and representative input and calibration datasets, as well as on analysis of output sensitivity and uncertainty, wherever feasible. The same recommendations apply to the application of risk-based models (e.g., QMRA).

8. Based on stormwater control performance data from the International Stormwater BMP Database, consistent attainment of concentration-based primary contact recreational standards at end of pipe during all discharge conditions is unlikely for most passive stormwater controls (excluding disinfection). However, stormwater controls have many other water quality benefits and may still reduce FIB loads (especially through volume reductions), even if concentration-based limits are not consistently attainable. When selecting structural stormwater controls, both concentration and volume reduction benefits should be considered, focusing on practices with unit treatment processes that may be effective at reducing FIB loads.

9. Disinfection through chlorination, ultraviolet radiation, and ozonation are well documented to effectively reduce both FIB and pathogen concentrations in wastewater and drinking water. Chlorination and ozonation are typically impractical for urban stormwater applications due to needs for dechlorination (to prevent byproduct formation or discharge of toxic residuals) and risks of chemical storage. Ultraviolet radiation of dry weather MS4 discharges has been implemented in some locations, although long-term operation and maintenance costs can be significant. Examples of disinfection of urban low-flows are typically limited to MS4 discharges to receiving waters where recreational exposure (i.e., potential public health impact) and economic impacts of beach closures are significant. Generally, disinfection is considered an option when source controls and stormwater controls have not resulted in attainment of FIB standards and elevated human health risks are present. In some cases, disinfection has been effective at point of treatment, but FIB regrowth has been observed shortly downstream, thereby potentially reducing its benefits (at least in terms of compliance with FIB limits).

10. Although the primary focus of this report is not CSOs, urban stormwater controls (e.g., green infrastructure controls that emphasize infiltration) that provide volume reduction

can play a significant role in reducing the frequency and magnitude of CSO events and are often a component of long-term control plans (LTCPs). Additionally, principles of integrated planning of stormwater and sanitary municipal programs may be transferable to MS4 permits. Regulatory flexibilities that have been approved under LTCPs for CSOs may be helpful in formulating practical regulatory solutions to receiving water impairments once reasonable steps have been taken to reduce controllable sources of FIB. For example, some LTCPs have allowed use-attainability analysis (UAA) to modify the recreational designated use (classification) of a waterbody receiving wet weather discharges from CSOs during wet weather conditions. Even in the absence of LTCPs, some regulations allow high-flow suspension of recreational uses, which is conceptually similar to the use of a sizing criterion for an end-of-pipe retention or treatment system.

11. Given the issues and constraints described in this report, additional policy-level dialogue is needed to determine the most effective approach for developing and implementing urban FIB TMDLs and to determine TMDL “endpoints” that may differ from 100% compliance with RWQC, while still protecting public health. Once human sources of FIB are addressed, site-specific criteria, such as based on QMRA, are one alternative, particularly for large metropolitan areas with high exposure or high value recreational use waters; however, the cost of conducting these studies at multiple smaller waterbodies is beyond the reach of many smaller municipalities across the country. An alternative, cost-effective compliance approach that is protective of public health and that also recognizes economic constraints of local governments and practical limitations of technology and/or controllability of FIB sources is needed.

Recommendations for additional research and policy discussions needed to advance the science and policy on this complex issue are also provided with this report.

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1 INTRODUCTION

“Pathogens” are the single most frequent cause of water quality impairment in the U.S., with over 10,950 waterbodies listed as impaired on state 303(d) lists (EPA 2014). Pathogen impairments usually are identified based on elevated counts of fecal indicator bacteria (FIB), such as *Escherichia coli* (*E. coli*), enterococci or fecal coliform. Pathogens are disease-causing organisms, whereas FIB indicate the potential presence of such organisms. Determinations regarding impairment are based on comparison of FIB concentrations to applicable waterbody standards and classifications. In the majority of cases, this contamination cannot be traced to a single point discharge such as a wastewater treatment plant (WWTP) (Clark et al. 2010). FIB originate from warm blooded animals, but have also been associated with reptiles (Habersack et al. 2011) and naturalized environmental sources (Fujioka et al. 1999, Byappanahalli et al. 2006, Yamahara et al. 2007, Boehm et al. 2009). There are many natural and human-induced sources of FIB in receiving waters and stormwater systems, and identifying these sources and controlling them pose significant challenges. Unlike chemical pollutants, FIB and pathogens are living organisms that die-off, grow, or persist, depending on environmental conditions, which are mostly uncontrollable for all practical purposes. Additionally, even when human and non-human anthropogenic sources of FIB and pathogens (e.g., leaking sanitary sewers, pet wastes) are controlled, urban wildlife and other ubiquitous non-fecal sources may persist as on-going causes of elevated FIB.

This report focuses on urban stormwater management issues related to elevated FIB concentrations in receiving waters, particularly challenges faced by National Pollutant Discharge Elimination System (NPDES) Municipal Separate Storm Sewer System (MS4) permit holders and by watershed stakeholders. FIB and pathogen sources in urban stormwater systems can result from both dry and wet weather conditions. Under dry weather, FIB can be associated with flows to storm sewer systems that originate from groundwater, irrigation runoff from lawns, vehicle washwater, power-washing flows, leaking sanitary sewer lines, improper sanitary sewer line connections, and other sources. FIB and pathogens may be associated with the original water source itself or flows may transport previously deposited fecal material from urban wildlife (e.g., birds, squirrels, foxes) living in the urban area and in storm sewers (e.g., rats, raccoons). Under wet weather conditions, urban runoff mobilizes FIB and pathogens deposited on landscaped and impervious surfaces, collected in catchbasin sediment, or present in biofilms within the storm sewer system. Additionally, some communities face significant challenges associated with combined sewer overflows (CSOs) where wet weather conditions cause sewage overflows into receiving waters or where sanitary sewer overflows (SSOs) occur. SSO conditions may also occur during dry weather if sanitary sewers become clogged or are undersized. Although CSOs and SSOs are briefly discussed in this report, the primary emphasis is MS4s and their receiving waters in urbanized areas.

This report provides information on the following topics to support MS4 permit holders and watershed stakeholders in developing realistic goals and effective strategies for addressing FIB and pathogens in urban stormwater systems:

- Basic background related to regulatory context, pathogens in receiving waters and the use of FIB as surrogates for pathogens.
- Sources of pathogens in the urban environment.
- Transport and fate issues, along with the factors affecting survival of pathogens and FIB. Although an evaluation of models for FIB is beyond the scope of this report, understanding of transport and fate issues affects the ability of water resources scientists and engineers to develop models for FIB.
- Approaches for monitoring, source tracking, and evaluating FIB and pathogen data, including a discussion of challenges associated with these activities.
- Source controls and treatment strategies, including expected effectiveness, data gaps, and practical constraints related to source controls, structural stormwater controls, and disinfection.
- Case studies illustrating challenges and approaches to implementing and complying with FIB total maximum daily load (TMDL) requirements in urban areas.
- Conclusions and recommendations for additional applied research needs related to pathogens in urban stormwater systems and complying with FIB and/or pathogen TMDLs.

2 BASIC BACKGROUND

Concurrent to development of this report, the U.S. Environmental Protection Agency (EPA) sponsored significant research to support an update to its Recreational Water Quality Criteria (RWQC) (EPA 2007a,b&c, 2012). Many scientific reports related to epidemiology, risk assessment, test methods, and other topics were published as a result of this process (accessible at http://water.epa.gov/scitech/swguidance/standards/criteria/health/recreation/). The primary focus of EPA's effort was determining whether changes in numeric criteria and assessment methods were needed for the RWQC, rather than guiding how communities respond to FIB impairments (i.e., TMDL implementation plans). The purpose of this chapter is to provide a brief synopsis of the underlying basis of the key regulatory drivers pertaining to FIB in stormwater systems for MS4 managers, rather than to provide an exhaustive synopsis of the complex epidemiological and health risk-related decisions made during EPA's 2012 update of the RWQC.

In addition to regulatory context, other background provided in this section relates to human health risks from recreating in waters identified as impaired due to elevated FIB, human health risks associated with natural versus human sources of FIB, and additional background on the relationship between FIB and pathogens.

2.1 Regulatory Context for Recreational Water Quality Criteria

Both internationally and in the U.S., epidemiological and other health studies form the basis for RWQC based on the risk to swimmers of contracting disease from exposure to water containing a specified number of microorganisms (IAWPRC 1991, Jin et al. 2004, EPA 2012). Although the primary focus of this report is the U.S., the European Union (EU) and the World Health Organization (WHO) have also developed standards that are used in other countries. In the U.S., Section 304(a)(1) of the federal Clean Water Act requires the EPA to promulgate criteria for water quality. States with delegated Clean Water Act authority rely on EPA's criteria to promulgate numeric standards to protect human health in waterbodies with recreational use classifications. Such standards are also integrated into NPDES permits for wastewater treatment plants. EPA initially released RWQC in 1976, updated the criteria in 1986, and most recently updated the criteria in 2012. The RWQC include numeric criteria for FIB that are intended to be indicative of health risks associated with recreational use. The overall goal of the criteria is to provide public health protection from gastroenteritis (i.e., gastrointestinal [GI] illness) associated with exposure to fecal contamination during water-contact recreation. These criteria have evolved over time; therefore, there is variation among the criteria adopted by various states as water quality standards. EPA must approve the water quality standards adopted by the states with Clean Water Act authority, within the constraints of the following considerations (EPA 2012):

- The RWQC are intended as guidance to states in developing water quality standards to protect swimmers from exposure to water that contains organisms that indicate the presence of fecal contamination.

- States have the discretion to adopt other scientifically defensible water quality criteria.

- The RWQC are designed to protect primary contact recreation, including swimming, bathing, surfing, water skiing, tubing, water play by children, and similar water contact activities where a high degree of bodily contact with the water, immersion and ingestion are likely.

EPA relies on FIB, as opposed to pathogens, as the basis of water quality criteria because FIB are easier to identify and enumerate in water quality samples than the broad range of pathogens in human and animal feces. In the U.S., RWQC for FIB date back to the 1960s and 1970s based on work by the U.S. Public Health Service and subsequently EPA. This historic work formed the basis for the use of fecal coliform and associated numeric criteria for the protection of recreational water quality uses. (Total coliform and fecal streptococcus have also been used as FIB in the past, but are no longer recommended.) Many states still use fecal coliform in their water quality standards; however, EPA has recommended use of *E. coli* or enterococcus since 1986. EPA's 1986 criteria were derived from epidemiological studies conducted at marine (Cabelli 1983) and freshwater lake (Dufour 1984) locations with contamination from effluent discharged from single point-sources, essentially addressing the question: "Does swimming in sewage-contaminated water carry a health risk for bathers; and, if so, what type of illness?" Since that time, additional epidemiological studies have been completed (i.e., NEEAR studies during 2003-2009), but the numeric criteria established for primary contact based on the research by Cabelli (1983) and Dufour (1984) have generally been retained as the core of EPA's (2012) update of the criteria, with some modifications based on health-based considerations.

EPA's criteria are developed including three components: magnitude, duration, and frequency of exceedance. The magnitude component of the criteria refers to the numeric value and statistical measure (e.g., geometric mean) used in the criteria. Duration refers to the time period over which compliance with the criteria should be assessed (e.g., monthly, seasonal, annual) and frequency refers to the number of sample results that are allowed to exceed the numeric criteria. Currently, there is broad variation in how states have adopted these three components of historic RWQC into their individual water quality standards. All three components affect the stringency and attainability of the standards. For example, a standard of 126 cfu/100 mL assessed over a

National Epidemiological and Environmental Assessment of Recreational Water (NEEAR)

In support of the 2012 RWQC update, EPA conducted epidemiological investigations at U.S. beaches in 2003, 2004, 2005, 2007, and 2009. These investigations are collectively referred to as "the NEEAR study." The NEEAR study enrolled 54,250 participants, included nine locations, and collected and analyzed numerous samples from a combination of fresh water, marine, tropical, and temperate beaches (EPA 2010a, Wade et al. 2008, 2010). The general purposes of the studies included: 1) evaluate the water quality at one or two beaches per year; 2) obtain and evaluate a new set of health and water quality data for the new rapid, state-of-the-art methods; and 3) share results to support new state and federal guidelines and limits for water quality indicators of fecal contamination, so that beach managers and public health officials can alert the public about the potential health hazards before exposure to unsafe water can occur.

The NEEAR study reaffirmed an association of enterococcus and *E. coli* with gastrointestinal illness and was a key component of EPA's decision to retain these two indicators as the basis of the 2012 RWQC. For more information on NEEAR and for links to specific reports, see http://www.epa.gov/neear/.

30-day period can be more challenging to meet than the same numeric standard assessed on an annual basis because in many areas, cool-weather samples with lower *E. coli* concentrations tend to offset higher *E. coli* concentrations in warm-weather samples when a geometric mean is calculated (Hathaway et al. 2010, Selvakumar and Borst 2006).

Because there is typically a lag time between EPA publishing new criteria and states adopting the new criteria, both current and historically recommended criteria are summarized in Tables 2-1 and 2-2, respectively. EPA's 2012 RWQC include both geometric mean values that are not to be exceeded based on a 30-day assessment period, as well as Statistical Threshold Values (STVs) that may be exceeded in up to 10 percent of samples. The STV replaces the previously recommended Single Sample Maximum (SSM) values shown in Table 2-2, which varied based on use intensity, as promulgated under the Beach Act (EPA 2004). Prior to the 2012 RWQC, Beach Act regulated waters were required to implement both the geometric mean and SSM criteria. Some, but not all, inland states also chose to adopt the SSM as a component of their criteria.

Under the 2012 RWQC, EPA is recommending that all waters classified for recreational use adopt both the geometric mean and STV components of the criteria, with no differentiation of standards based on recreational use intensity. EPA provides two equally acceptable options for standards based on allowable illness rates, as shown in Table 2-1. (See EPA [2012] for a more detailed explanation on interpreting allowable illness rates.) Recommendations for secondary contact (e.g., fishing, some boating) are not included in the RWQC; however, many states have secondary contact standards, typically set at five times the primary contact standard.

Table 2-1. Summary of Current Recreational Water Quality Criteria for Bacteria
(Source: EPA 2012)[1]

Criteria Elements	Recommendation 1 (Est. Illness Rate 36/1,000)		Recommendation 2 (Est. Illness Rate 32/1,000)	
Indicator	**Geometric Mean (cfu/100 mL)**	**STV (cfu/100 mL)**	**Geometric Mean (cfu/100 mL)**	**STV (cfu/100 mL)**
Enterococci (marine & freshwater)	35	130	30	110
E. coli (freshwater)	126	410	100	320

Note: Allowable exceedance frequency is 10% for the STV and 0% for the geometric mean. The recommended assessment period is 30 days. Criteria shown are for culture-based test methods, but equivalent qPCR criteria may be developed.

[1] Note: The swimmer illness risks in Tables 2-1 and 2-2 are equivalent because the NEEAR definition of gastrointestinal illness (NGI) definition is broader than the previously used "highly credible gastrointestinal illness" (HCGI) definition. More illnesses qualify to be counted as "cases" in the NEEAR epidemiological studies than if the older HCGI definition were applied. Therefore, at the same level of water quality, more NGI will be observed than HCGI illnesses. The relative increase in rates of GI illness between the studies (i.e., HCGI versus NGI) is directly attributable to the changes in how illness was defined and not due to an actual increase in the incidence of illness among primary contact recreators at a given level of water quality (EPA 2012).

Table 2-2. Historic Ambient Water Quality Criteria for Bacteria
(Source: EPA 1986)

	Marine Waters	**Fresh Waters**
Primary research basis	Cabelli 1983	Dufour 1984b
Acceptable swimming associated gastroenteritis rate (per 1,000 swimmers)	Increase of 19 illnesses per 1,000 swimmers	Increase of 8 illnesses per 1,000 swimmers
Geometric Mean Limits		
Fecal Coliform *(recommended prior to 1986)*	Fecal Coliform: 200 cfu/100 mL	Fecal Coliform: 200 cfu/100 mL
Enterococci and *E. coli* (EPA 1986)	Enterococci: 35 cfu/100 mL	Enterococci: 33 cfu/100 mL *E. coli*: 126 cfu/100 mL
Single sample limits (not implemented in all states but required for Beach Act-regulated waters [EPA 2004]):		
Designated bathing beach area	Enterococci: 104 cfu/100 mL	Enterococci: 61 cfu/100 mL or *E. coli*: 235 cfu/100 mL
Moderate full body contact recreation	Enterococci: 124 cfu/100 mL	Enterococci: 89 cfu/100 mL or *E. coli*: 298 cfu/100 mL
Lightly used full body contact recreation	Enterococci: 276 cfu/100 mL	Enterococci: 108 cfu/100 mL, or *E. coli*: 406/100 mL
Infrequently used full body contact recreation	Enterococci: 500 cfu/100 mL	Enterococci: 151cfu/100 mL or *E. coli*: 576 cfu/100 mL

In addition to current state variations in selected FIB type and adoption of SSM criteria, other aspects of recreational use classifications that vary by state relate to how considerations such as seasonal use, wildlife influences, natural source exclusions, high-flow recreational use suspensions, and other factors are addressed. Not all receiving waters are assigned primary contract recreation standards, depending on the beneficial use classification of the particular receiving water. However, in most urban areas, waterbodies are typically subject to primary contact recreation standards due to the potential for waterplay by children.

In the 2012 RWQC, EPA also allows states to develop alternative, scientifically-defensible criteria that are equally protective of human health based on the RWQC illness rates; however, this process requires a significant scientific and financial commitment. EPA has provided these options in part because EPA recognizes that the basis of the epidemiological studies used to develop the RWQC were in waters primarily impacted by secondary-treated and disinfected municipal wastewater treatment plant effluent and that these situations may not be representative of all possible fecal contamination combinations that could impact recreational bodies of water (EPA 2012). EPA is allowing states to adopt site-specific alternative criteria that reflect local environmental conditions and human exposure patterns (EPA 2012). These alternative water quality standards may be based on:

1) an alternative health relationship derived using epidemiology, with or without Quantitative Microbial Risk Assessment (QMRA);

2) QMRA to determine water quality values associated with a specific illness rate; or

3) a different indicator/method combination.

Pilot QMRA studies are being conducted in California with regional, state, and federal involvement. These studies have the potential to set precedence for how such alternative criteria will be developed in the future. To be approved by EPA under CWA §303(c), these alternative criteria should reflect the same level of risk regarding illness rate as used by EPA in the 2012 RWQC, be scientifically defensible, and be protective of the recreational use (EPA 2012).

2.2 Regulatory Implications for MS4s

Under the Clean Water Act, states are required to assess attainment of receiving water standards biennially and develop state "303(d) lists" of waters not attaining standards. Once a segment is listed on the 303(d) list, states are required to initiate the TMDL process to address these impairments and assign pollutant load allocations to various sources discharging to the receiving water (Figure 2-1). The basic components of a TMDL include: wasteload allocations (WLAs) for point sources, load allocations (LAs) for non-point sources, and a margin of safety (MOS). Wastewater treatment plant (WWTP) discharges and MS4s are considered point source discharges, with TMDL-related wasteload reductions enforceable under NPDES permit requirements. For MS4 permittees, additional permit requirements related to reducing FIB contributions to receiving waters may result from TMDLs. Nonpoint sources of FIB are typically controlled on a voluntary basis with very limited enforcement mechanisms. See *Protocol for Developing Pathogen TMDLs* (EPA 2001) for more information on the TMDL process.

Although the "formula" for development of a TMDL is relatively straightforward, identification of sources of FIB impairment can be challenging in many urbanized areas. Identifying the source of impairment is essential for developing meaningful measures that reduce FIB in urban environments; however, source identification itself can be complicated and costly. The sources of national stream impairments due to FIB vary considerably and may include human sources, domestic pets, wildlife, and naturalized sources. FIB transport pathways can include sanitary and storm sewer systems, as well as overland flow and direct deposition into waterbodies. Due to the diffuse nature of potential sources of FIB in urbanized areas, multiple approaches are often needed to begin to reduce FIB in waterbodies.

Basic Form of a TMDL

The basic form of a TMDL calculation is:

$$\mathbf{TMDL = \Sigma WLA + \Sigma LA + MOS}$$

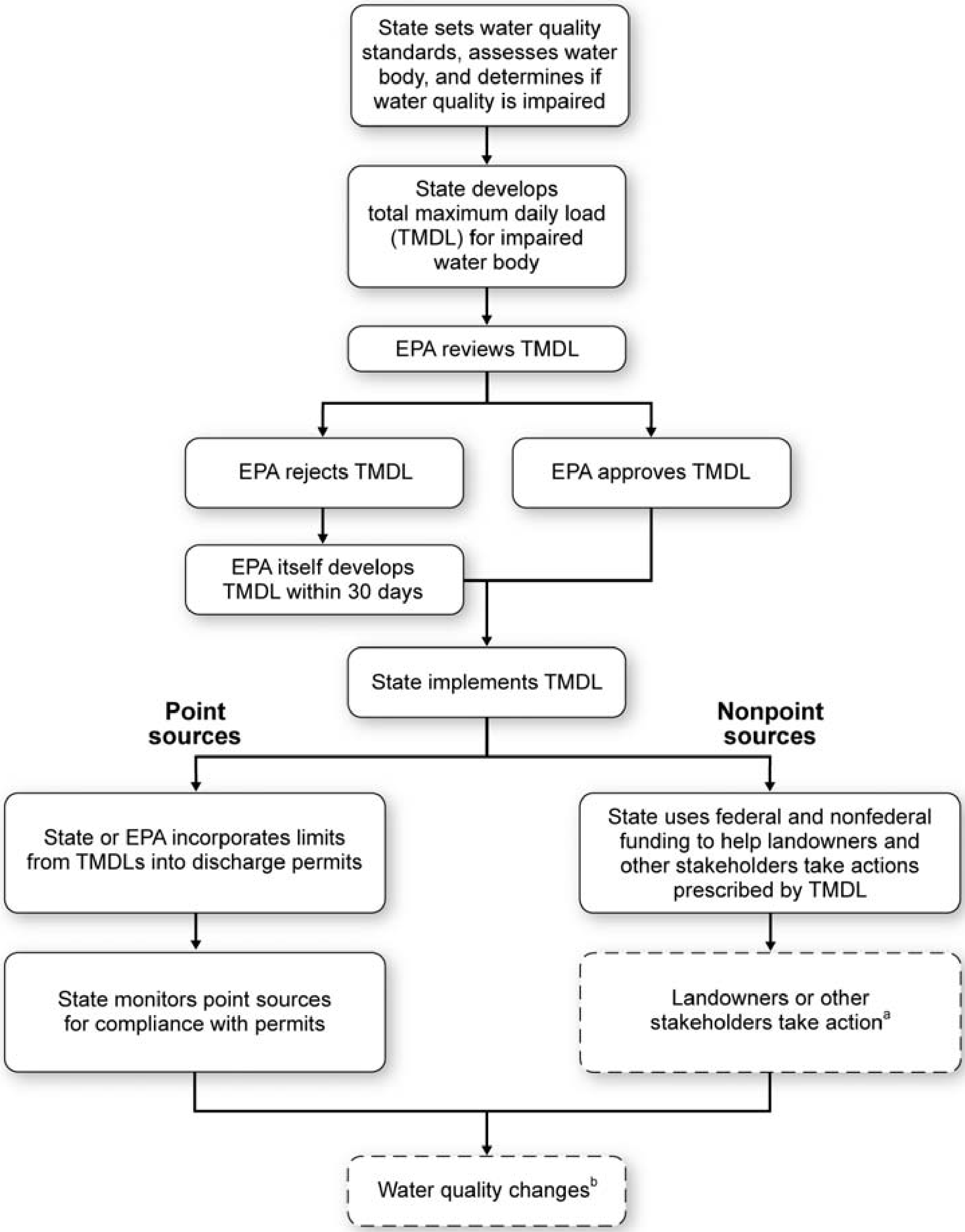
Where:

WLA = the sum of wasteload allocations (point sources such as permitted wastewater and stormwater discharges)

LA= the sum of load allocations (nonpoint sources and background)

MOS= the margin of safety

Figure 2-1. TMDL Development Process
(Source: GAO 2013)



[a]Depends in large part on voluntary participation by private landowners; other actors may include state or local regulatory or non-regulatory programs.
[b]According to EPA officials, it may take years for changes to occur in water quality after implementation of best management practices or other projects and activities prescribed by TMDLs.

In addition to TMDLs, MS4s discharging to FIB-impaired waterbodies may face other MS4 discharge permit conditions associated with permit language that does not allow discharges that "cause or contribute to an exceedance of a receiving water limit" (i.e., stream standard). This is an area of complex and emerging case law that is not fully explored in this report, but can have significant implications in terms of compliance schedules and legal challenges for MS4s. For an example, see the various legal proceedings associated with the Natural Resources Defense Council, Inc. and Santa Monica Baykeeper vs. Los Angeles County Flood Control District and County of Los Angeles.

2.3 Pathogens Found in Surface Water[2]

Human pathogens in surface water include viruses, bacteria, protozoa, and parasitic worms. The primary concern with regard to pathogens in surface waters is incidental human ingestion of contaminated water during recreational contact with the water, resulting in illness; however, other types of exposure to pathogens can also result in respiratory, skin, ear, and eye infections. Enteric pathogens are the group of microorganisms that result in enteric (or gastrointestinal) diseases. Most microbes that inhabit the intestines are harmless, or even beneficial. Others are harmless in normal individuals but can produce disease in the very young, those with weakened immune systems, or in a new host that has no prior exposure to the microbe (EPA 2009a). Exposure to some minimum number of organisms (i.e., an infectious dose) is required for a pathogen to successfully infect a human. In general, enteric viruses and protozoa have very low infectious doses, typically between 1 and 100. Bacterial pathogens tend to require larger doses to cause infection, with an infectious dose ranging from 100 to 1,000,000 (Clark et al. 2010).

Although many different types of pathogens may exist in surface waters from both natural and human sources, the World Health Organization (WHO) and the Centers for Disease Control (CDC) have identified a short list of pathogens expected to be responsible for over 97% of non-foodborne illness. This list includes norovirus, adenovirus, rotavirus, *Cryptosporidium spp., Giardia lamblia, Campylobacter jejuni, Salmonella enterica* and *E. coli* O157:H7. Similar to the list above, the primary focus of recent QMRA research (Soller et al. 2010b) in the context of recreational illness includes these "reference" pathogens: norovirus, *Cryptosporidium spp., Giardia lamblia, Campylobacter jejuni, Salmonella enterica* and *E. coli* O157:H7 (EPA 2010a). These pathogens are considered representative of the transport and fate of other pathogens potentially of concern from the waterborne route of exposure (Ferguson et al. 2009) and have corresponding dose-response relationships in the peer reviewed literature (Soller et al. 2010a). This subset of pathogens is briefly described below, along with brief comments on the life cycle requirements for viruses, bacteria and protozoa.

[2] This section and subsections are adapted from Appendix G of the WERF publication *Sustainable Stormwater Management: Infiltration vs. Surface Treatment, Strategies*, prepared by S.E. Clark, K.H. Baker, D.P. Treese, J.B. Mikula, C.Y.S. Siu, and C.S. Burkhardt with contributions by M.M. Lalor. Water Environment Research Foundation. Project Number 04-SW-3. ISBN: 9781843392811. 2010. 460 pages.

2.3.1 Viruses

Viruses are infectious particles containing nucleic acid (DNA or RNA) encapsulated in a protective protein. They are not independent organisms with independent growth, metabolism, or reproduction. Outside of a host, they are inert; however, they are still infectious. Viruses typically are very host-specific and only infect a limited number of host species (Clark et al. 2010). Over 100 different types of human viruses can be transmitted by water contaminated by fecal matter (Berg 1983). While some viruses cause life-threatening diseases, most cause relatively mild symptoms.

The human enteric virus group, which includes norovirus (or NLV, norwalk-like virus), rotavirus, hepatitis A virus, adenovirus, and enterovirus, is one of the leading causes of human illness. Relatively little is known about many viruses in this group because many enteric viruses are difficult to culture or are not culturable. They also produce diseases that are not readily identifiable (i.e., have symptoms that are common to other pathogens). The main symptoms of viral gastroenteritis are watery diarrhea and vomiting, but the infection usually is self-limiting. The affected person may also have a headache, fever, and abdominal cramps. In general, the symptoms begin one to two days post-infection and may last for one to ten days. Adenoviruses 40 and 41 are important etiological agents in children (Cruz et al. 1990, Uhnoo et al. 1986.). For purposes of near-term QMRA research, the three viruses of primary interest include: norovirus, adenovirus and rotavirus. Although indicators of viruses exist, such as coliphages, they have not experienced widespread use in watershed management or regulatory applications.

2.3.2 Bacteria

Bacteria are the smallest organisms capable of independent existence. Bacteria are essential components of natural ecosystems; however, some are also human pathogens. Most of the pathogenic microorganisms that contaminate surface water enter the water with fecal matter from either animals or humans. For purposes of near-term QMRA research, the primary bacterial pathogens of interest, as described by Clark et al. (2010), include:

- ***Salmonella****:* *Salmonella* includes a very large number of species and serotypes, many of which are infectious to humans. *Salmonella*, in addition to being transmitted by water, is a major foodborne pathogen. Reptiles, birds, and wild rodents can carry *Salmonella*; therefore, detecting *Salmonella* in water does not necessarily reflect contamination with human wastes. *Salmonella* has a high infectious dose for ingestion of about one million (10^6) organisms. Despite this high infectious dose, salmonellosis is one of the most common causes of diarrhea in humans.

- ***Campylobacter:*** *Campylobacter* is generally regarded as one of the most common bacterial causes of gastroenteritis worldwide, accounting for approximately 45% of the cases of diarrhea in the U.S. The infectious dose of *Campylobacter* is significantly lower than that of *Salmonella*, with an infection dose of less than 500 cells potentially causing infection. It can be carried by most mammals and by birds and is especially likely to be found in cattle, sheep, dogs, and poultry. Although rare complications are possible, typical symptoms of the infection include diarrhea (often including the presence of mucus and blood), abdominal pain, malaise, fever, nausea, and vomiting.

- ***E. coli* O157:H7*:*** *E. coli* is one of the most common intestinal bacteria and is a normal part of every mammal's intestinal biota. While most strains of *E. coli* are harmless, there are a few specific types of *E. coli* (enteropathogenic and enterohemorragic strains) that can produce disease. Disease caused by pathogenic strains of *E. coli* is most likely to be seen in cattle, swine, and humans. Infection with *E. coli* O157:H7 results in hemorrhagic colitis. In approximately 10% of the cases, hemorrhagic colitis leads to hemolytic uremic syndrome (HUS), a leading cause of kidney failure in children. The outbreak in Walkerton, Ontario in the last decade, the best know occurrence of waterborne *E. coli* O157:H7 infection, was traced to runoff contamination of wells with manure from nearby cattle farms.

Examples of non-enteric, water-related bacterial diseases include pneumonia (*Legionella pneumophila*), kidney infections, and skin and wound infections (*Staphylococcus aureus, Pseudomonas aeruginosa, Vibrio vulnificus, Leptospira, Aeromonas*). Research has linked water-related viruses to many non-enteric diseases (Clark et al. 2010). For example, *P. aeruginosa* has been reported to be the most abundant pathogenic bacteria organism in urban runoff and streams (Olivieri et al. 1977b). This pathogen is associated with eye and ear infections and is resistant to antibiotics.

2.3.3 Protozoa

In the developed world, the two infectious protozoa associated most often with contaminated water are *Giardia* and *Cryptosporidium*. These protozoa can be transported in both human sewage and from animals. In healthy adults, infections with these organisms typically are self-limiting; however, in immunocompromised individuals, the disease can be chronic and life-threatening. *Cryptosporidium* is a common infection in cattle and therefore, runoff containing livestock manure is a major source of surface water contamination. Other animals that are carriers of *Cryptosporidium* are pigs, cats, deer, guinea pigs, mice, rats, and sheep. *Giardia* is more frequently associated with wild animals than it is with domestic animals although it has been determined that a variety of pet wastes (e.g., dog, cat, bird) can transmit *Giardia* cysts. Improper disposal of pet fecal matter can introduce the organism into urban runoff (Clark et al. 2010).

Infection with *Cryptosporidium*, termed cryptosporidiosis, is the result of ingestion of oocysts of *Cryptosporidium parvum* (now also called *Cryptosporidium homonis)*. It is now estimated that cryptosporidiosis accounts for approximately 5% of infectious intestinal disease in which a causative agent is identified, making it one of the most significant causes of waterborne diseases today (Kramer et al. 1998). Infectious *Cryptosporidium* oocysts are able to persist in the environment for long periods of time; under certain conditions, they may remain infectious for months (Carrington and Miller 1993).

Ingestion of cysts of *Giardia lamblia* (also referred to as *Giardia duodenalis* and *Giardia intestinalis*) can cause giardiasis. As with cryptosporidiosis, the infectious dose is small, with as few as 10 cysts resulting in infection. In healthy individuals, the disease is self-limiting; however, in immunosuppressed individuals, it can be a chronic and debilitating infection.

2.4 Relationships of Pathogens to FIB

As discussed in Section 2.1, FIB are recommended by EPA for use in RWQC because monitoring for the presence of specific pathogens, such as those discussed in Section 2.3, is not practical for routine control purposes. FIB such as *E. coli* and enterococci can be detected and quantified using relatively simple and rapid bacteriological tests. Additionally, EPA has concluded that the use of *E. coli* and enterococci as predictors of gastrointestinal illness is the approach currently best supported by available epidemiological studies (EPA 2012). The fact that the "perfect indicator organism" does not exist has led to much difficulty in sorting out the meaning of elevated FIB in the environment and has led to many complaints related to the current indicators used to establish ambient water quality criteria, especially when it leads to beach/recreational water closures, affecting local businesses. Boehm et al. (2009) provide this concise synopsis of the FIB-pathogen dilemma:

> Although *E. coli* and enterococci are found in high concentrations in human sewage (Maier et al. 1999), they are also highly prevalent in the environment. They are excreted in the feces of numerous warm blood animals (Parveen et al. 1999, Harwood et al. 2000, Ashbolt et al. 2001). There is strong evidence that some genotypes of *E. coli* are naturalized—meaning they are adapted to persisting and even growing in extra-enteric environments like lakes, soils and sediments (Davies et al. 1995, Desmarais et al. 2002, Ishii et al. 2006). *E. coli* and enterococci can be found in a number of environmental reservoirs including soils and sands in tropical, subtropical and temperate climates (Fujioka et al. 1999, Byappanahalli et al. 2006, Yamahara et al. 2007). Enterococci can be found on terrestrial grasses (Ott et al. 2001) and aquatic plants (Whitman et al. 2003). If *E. coli* and enterococci measured at a beach during water quality monitoring emanate from these sources rather than municipal wastewater, then the FIB-gastroenteritis relationship upon which [EPA's] 1986 criteria is based might not hold.
>
> The Missing Causative Link… a positive, correlative relationship between FIB and human pathogen concentrations (for example, human enteric viruses) has remained elusive. In fact, most studies show a striking lack of correlation between the two (e.g., Noble & Fuhrman 2001, Boehm et al. 2003, Jiang & Chu 2004, Pusch et al. 2005). Ultimately, the lack of strong, positive relationships between FIB and pathogens in ambient waters casts doubt on the appropriateness of extrapolating the 1986 criteria to conditions and sites which were not included in the original epidemiological studies used for criteria development.
>
> …*E. coli* and enterococci emanating from naturalized or non-fecal sources may result in the waterbody being incorrectly classified as impaired when the risk of illness is actually not above what had been determined to be acceptable. The development of best management practices and treatment technologies for

> removing FIB from waters where there are no obvious fecal inputs could be costly, destructive to natural ecosystems, and not substantively reduce the health risk of those using the water for recreation.

In response to concerns similar to those expressed by Boehm et al. (2009), EPA considered use of alternative indicators such as *Bacteroidales*, *Clostridium perfringens*, human enteric viruses, and coliphages as part of EPA's 2012 update of the RWQC. EPA ultimately decided to retain *E. coli* and enterococcus as the basis for the RWQC, due to lack of adequate epidemiological data to support an alternative approach. Following the release of the 2012 RWQC, use of QMRA may be one of the more promising options to help to support alternative standards where the currently recommended FIB criteria are not sufficient, or are overly restrictive.

2.5 Epidemiological Studies Related to Stormwater

For urban stormwater managers, a significant question remains regarding the role of stormwater (and nonpoint sources) in recreational waterborne illnesses, since most of the epidemiological research to date has focused on sanitary-impacted locations or at locations where the role of stormwater was not specifically quantified. Wade et al. (2003) conducted an extensive review of the available studies to determine the relationship between recreational water quality, exposure and health effects and found that less than five percent of these studies provided adequate data on the pertinent variables related to the sources of the microorganisms. Thus, many scientific questions remain regarding the specific sources of pathogens causing elevated human health risk in recreational waters in non-sewage related studies. Some examples illustrating these issues, as summarized by Boehm et al. (2009) and EPA (2012), include:

- **New Zealand:** McBride et al. (1998) conducted a study in New Zealand and found that recreational waterborne illnesses were correlated with enterococci at beaches adversely impacted by both rural and urban runoff.

- **Connecticut**: Calderon et al. (1991) studied risk and exposure to enterococci and *E. coli* from non-point animal sources in a lake in Connecticut and found no correlation between recreational waterborne illnesses and FIB concentrations.

- **Santa Monica Bay, CA**: The Santa Monica Bay Project was one of the earlier large-scale investigations that focused directly on the linkage between the discharge of stormwater runoff into recreational waters and human health effects (SMBRP 1996, Jiang et al. 2001). This project combined extensive sampling of stormwater runoff and a survey of over 10,000 individuals involved in water-contact recreation on the same day the water quality samples were obtained. The study found that there was a moderate, and statistically significant, increase in the risk of several adverse health outcomes associated with exposure to urban stormwater runoff (SMBRP 1996).

- **Los Angeles County**: Haile et al. (1999) found swimming in close proximity to urban drains near Los Angeles County discharging FIB led to significant risks of recreational waterborne illnesses at a California beach. In this case, positive, correlative relationships between FIB and numerous recreational waterborne illnesses were apparent. However, Haile et al. (1999) did not specify whether the urban storm drains may have been influenced by sanitary sewer leakage.

- **Mission Bay, CA:** Colford et al. (2007) examined the risk associated with exposure to non-human, non-point FIB sources in a beach without runoff sources in Mission Bay, California, and found no statistical association between traditional FIB and 14 different human health outcomes. Over 8,800 swimmers were surveyed as part of this epidemiological study.

- **Surfside Beach, South Carolina and Boquerón Beach, Puerto Rico:** EPA supported epidemiological studies in two urban-runoff impacted beaches in tropical regions (Boquerón Beach, Puerto Rico) and temperate marine water (Surfside Beach, South Carolina). Neither of these studies showed evidence of increased illness in children or the general population associated with increasing levels of FIB in the recreational waters (EPA 2010a, EPA 2012).

- **Doheny, Avalon and Surfrider Beaches, CA**: The Southern California Coastal Water Research Project (SCCWRP) led three dry weather epidemiology studies considering a range of bacterial sources, with varying degrees of human fecal contribution. (See www.sccwrp.org/ResearchAreas/BeachWaterQuality/CaliforniaEpidemiologicalStudies.aspx.) These studies were conducted between 2007 and 2009. As described by SCCWRP, at the Avalon Beach study on Catalina Island, leaking subsurface sewage infrastructure was expected to be the predominant source, and at Surfrider Beach in Malibu, local septic systems, birds and urban runoff were believed to contribute to the FIB load at the time that the studies began. At the Malibu site, sources were subsequently determined to be birds and other non-point sources (Izbicki 2011). Under the good water quality conditions observed during the epidemiological study, FIB concentrations were not associated with swimmer illness (Arnold et al. 2013). At the third study, Doheny State Beach in Dana Point, FIB inputs were expected to be primarily from nonhuman sources (e.g., birds, urban runoff). However, leaking sanitary lines were subsequently identified at this site. The Doheny Beach study evaluated health-risk relationships between GI illness and enterococci using qPCR-based and culture-based enumeration methods. At Doheny Beach (Colford et al. 2012), study results indicated that when water from an urban creek flowed freely from the freshwater outlet into the marine water (berm open), correlations between gastrointestinal illnesses and enterococcus were observed. This finding is consistent with NEEAR epidemiological studies at WWTP-impacted water bodies (see http://www.epa.gov/neear/). However, when the Doheny FIB source was more diffuse (berm closed), a significant relationship between enterococci and GI illness was not present. These diffuse source results are similar to those observed in the NEEAR Surfside Beach study described above (EPA 2012).

Based on review of these studies, findings regarding the potential human health impacts of urban stormwater with FIB are mixed. Schoen and Ashbolt (2010) state, “Given the existing body of work, there is no clear relationship between illness and any fecal indicator for non-sewage impacted beaches.” Given this unclear relationship between recreational waterborne illnesses and FIB from sources other than municipal wastewater, Boehm et al. (2009) concluded that it may be overprotective to apply the epidemiological relationships between FIB and illness in the human-sewage impacted studies to all U.S. waterbodies. Significantly more research is still needed to better understand the risk of recreating in waterbodies with elevated FIB due to stormwater with FIB from non-human origins (or at least dominated by non-human sources). In particular, studies in inland flowing waters are not well understood in terms of risk to recreators (WERF 2009), particularly where recreation is limited to wading. Studies are needed in a range of geographic and climatic conditions in both freshwater and marine environments before the limited epidemiological findings related to stormwater can be confirmed and applied to other locations.

Regardless of the mixed epidemiological findings related to the strength of the relationship between FIB and illness in urban runoff-impacted waters, research has demonstrated that there is a dramatic difference in FIB concentrations in stormwater-impacted beach water quality and beach water quality during dry weather periods (Noble et al. 2003b, Noble et al. 2004, Griffith 2006b, among others). Epidemiological data during wet weather urban runoff conditions are generally lacking.

2.6 Health Risks from Urban Wildlife Sources and Implications for TMDLs

Closely related to the discussion regarding the epidemiological link between human illness and urban runoff sources (Section 2.5), another common question that MS4 permittees often pose focuses on the contribution of wildlife to elevated FIB in waterbodies in urbanized areas. Given that animals and non-fecal sources of FIB have been documented to cause waterbodies to exceed numeric water quality criteria for FIB (e.g., Cox et al. 2005, Noble et al. 2004, Harwood et al. 2000, Ishii et al. 2006) and can be difficult to control, the question remains as to whether these "natural" sources pose a human health risk and need to be controlled as part of TMDL implementation plans, which may include MS4 permit requirements.

During the 2012 RWQC update, EPA explored the issue of relative risk from non-human sources of pathogens by conducting two literature reviews and sponsoring research related to QMRA. As a result of these efforts, EPA ultimately concluded that a national-level source exclusion for wildlife was not supportable in the 2012 RWQC; however, the criteria also recognized that wildlife sources are generally expected to pose a lower human health risk.

Although the 2012 RWQC should be referenced for a complete discussion of EPA's position on this issue, several key statements excerpted from the RWQC (EPA 2012, pp. 34-37) include:

> EPA further investigated sources of fecal contamination in *Review of Published Studies to Characterize Relative Risks from Different Sources of Fecal Contamination in Recreational Waters* (EPA 2009b) and *Review of Zoonotic Pathogens in Ambient Waters* (EPA 2009a). EPA recognizes the public health importance of waterborne pathogens that can affect both human and other species (zoonotic). However, the state of the science has only recently allowed for the characterization of the potential health impacts from recreational exposures to zoonotic pathogens relative to the risks associated with human sources of fecal contamination. Overall, the aforementioned reviews indicate that both human and animal feces in recreational waters do pose

potential risks to human health, especially in immunocompromised persons and vulnerable individuals. EPA has conducted analyses to characterize the potential differences in magnitude of illness arising from different fecal sources. These analyses indicate that the human health risk associated with exposure to waters impacted by animal sources can vary substantially. In some cases these risks can be similar to exposure to human fecal contamination, and in other cases, the risk is substantially lower.

EPA's research indicates that the source of contamination appears to be an important factor for understanding the human health risk associated with recreational waters and that the potential human health risks from human versus nonhuman fecal sources can vary (Schoen and Ashbolt 2010, Soller et al. 2010b).

Nonhuman sources of fecal contamination and the associated potential human health risks can vary from site-to-site depending on factors such as: the nature of the nonhuman source(s), the fecal load from the nonhuman source(s), and the fate and transport characteristics of the fecal contamination from deposition to the point of exposure. Nonhuman fecal sources can contaminate recreational bodies of water via direct fecal loading into the body of water, and indirect contamination can occur via runoff from the land. The fate and transport characteristics of the zoonotic pathogens and FIB present under these conditions can be different (such as, differences in attachment to particulates or differences in susceptibility to environmental parameters affecting survival) (EPA 2011).

…only a few epidemiological studies have been conducted in waters impacted by nonhuman sources of fecal contamination. The results of these studies are less clear than those conducted in waters impacted by human sources, particularly as related to conventionally enumerated FIB in those types of waters.…These studies collectively suggest that waterbodies with substantial animal inputs may potentially result in human health risks that vary based upon the relative proportion of the human and nonhuman fecal input and the nature of the nonhuman source of infective agent(s).

Microbial risk assessment approaches are available to assist in characterizing potential human health risks from nonhuman sources of fecal contamination (Roser et al. 2006, Soller et al. 2010b, Schoen and Ashbolt 2010, Till and McBride 2004)….The risk presented by fecal contamination from nonhuman sources has been shown in some cases, to be potentially less than the risk presented by fecal contamination from human sources (Schoen and Ashbolt 2010, Soller et al. 2010a&b, WERF 2011).

Because there have been few epidemiological studies, with mixed findings, in waters impacted by nonhuman sources and QMRA shows that risks from some animals may be comparable to humans, EPA is not developing separate national criteria for nonhuman sources. However, since some studies have site-specifically shown less risk in waters impacted by nonhuman sources, states interested in addressing the potential human health risk differences from different sources of fecal contamination on a site-specific basis [have several alternatives].

EPA's 1986 criteria provided an "off-ramp" for site-specific criteria based on completion of a sanitary survey combined with an epidemiological study. Due to the substantial cost and expertise required to conduct epidemiological studies, relatively few communities in the U.S. implemented this alternative. EPA's 2012 RWQC provide a new opportunity for alternative site-specific streams standards if human contamination sources are controlled and further epidemiological studies or QMRA for a waterbody shows that the human health risk in a waterbody is equal to or less than EPA's equivalent illness rate thresholds.

EPA-Sponsored Literature Reviews Related to Health Risks from Non-human Sources

EPA sponsored two literature reviews regarding human health risk from zoonotic pathogens, including:

- **Review of Published Studies to Characterize Relative Risks from Different Sources of Fecal Contamination in Recreational Waters (EPA 2009b)**: This review describes the existing knowledge base available to characterize the relative risks of human illness from various sources of fecal contamination in recreational waters. The review noted that one of the challenges is that most epidemiological studies conducted to date are located in areas where wastewater contamination is present. In recreational waters where wastewater contamination is absent, the studies show mixed results. Drawing upon drinking water illness outbreaks, there is evidence that animal sources of contamination can cause illness in drinking water. The study concluded that the risks to humans from animal feces were unclear based on available literature, stating:

  > "In summary, both human and animal feces in recreational waters continue to pose threats to human health. Although the public health importance of waterborne zoonotic pathogens is being increasingly recognized, it is still not well characterized. Policy makers and researchers have often assumed that the human health risk from pathogens associated with domestic and agricultural animal and wildlife feces is less than the risk from human feces, in large part because viruses are predominately host-specific. This literature review illustrates a lack of detailed and unequivocal information concerning the relative risks of human illness resulting from exposure to various sources of fecal contamination in recreational waters."

- **Review of Zoonotic Pathogens in Ambient Waters (EPA 2009a)**: This review provides a summary of information on waterborne zoonotic pathogens primarily from warm-blooded animals. The study focused primarily on six pathogens: pathogenic *E. coli*, *Campylobacter*, *Salmonella*, *Leptospira*, *Cryptosporidium* and *Giardia.* The appendix to the document tabulates pathogens, animal species that carry the pathogens and studies documenting illness from these sources. Specific bird species are not included in the report. Instead, general categories such as birds, wild birds, poultry, etc., are used. EPA's primary conclusion was:

  > "Although the most common waterborne recreational illnesses are probably due to non-zoonotic human viruses, which typically cause short-term gastroenteritis, the waterborne zoonotic pathogens discussed in this report have the potential to cause serious health effects—especially in immunocompromised persons and subpopulations."

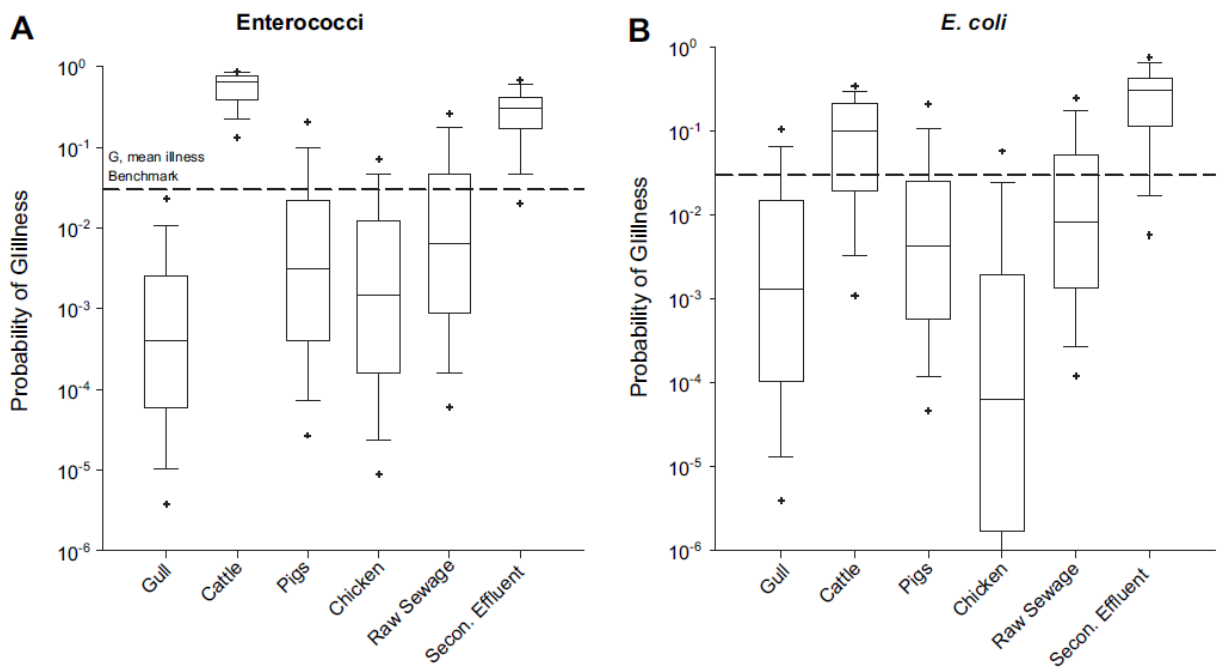
As an example of a key QMRA study, Soller et al. (2010b) estimated the likelihood of pathogen-induced effects by various sources. This work was conducted to determine whether estimated risks following exposure to recreational waters impacted by gull, chicken, pig, or cattle fecal contamination are substantially different than those associated with waters impacted by human sources such as treated wastewater. The probabilities of GI illness were calculated using pathogen dose-response relationships from the literature and Monte Carlo simulations. The primary findings, which may affect recreational water management in areas not affected by human contamination, included:

1) gastrointestinal illness risks associated with exposure to recreational waters impacted by fresh cattle feces may not be substantially different from waters impacted by human sources; and

2) the risks associated with exposure to recreational waters impacted by fresh gull, chicken, or pig feces appear substantially lower than waters impacted by human sources (approximately two orders of magnitude lower than the human-based benchmark) (Figure 2-2).

Other QMRA work by Schoen and Ashbolt (2010) also showed a lower predicted illness risk from seagull impacted waters relative to primary sewage at the same FIB density. These findings are consistent with WHO (2003) policies that assume that in general, sources other than human fecal contamination are less of a risk to human health. WHO (1999) states that “due to the species barrier, the density of pathogens of public health importance is generally assumed to be less in aggregate in animal excreta than in human excreta and may therefore represent a significantly lower risk to human health.”

Figure 2-2. QMRA-based Probability of Gastrointestinal Illness from Ingestion of Water Containing Fresh Fecal Contamination from Various Sources
(Soller et al. 2010b)



Notes (Soller et al. 2010b): QMRA Run 1 probability of GI illness from ingestion of water containing fresh fecal pollution at densities of 35 cfu/100 mL ENT (1A) and 126 cfu/100 mL *E. coli* (1B). Predicted risk for fresh gull, cattle and pig feces, and chicken litter. Human impacts are presented for primary sewage (Human 1) and secondary disinfected effluent (Human 2). The illness benchmark represents a geometric mean probability of illness of 0.03. The higher risk from disinfected wastewater results from a higher proportion of FIB being removed relative to viral and parasitic protozoan pathogens by wastewater treatment and disinfection (Metcalf and Eddy 2003) at the same indicator level.

2.7 Conclusion

EPA establishes and periodically updates RWQC to protect human health. These criteria were last updated in 2012 and continue to recommend use of FIB, namely *E. coli* and enterococci, to assess attainment of recreational use criteria. EPA's criteria are subsequently adopted as water quality standards by states, which use these standards to assess attainment of recreational uses and to support NPDES permit limits for sanitary wastewater.

Pathogens are the top cause of receiving water impairments in the U.S., and receiving water impairments as defined by elevated levels of FIB frequently occurring in urbanized areas. Although the epidemiological linkage between elevated FIB in stormwater and human health risk is less clearly understood than for sanitary sewage-impacted waters, MS4 permit holders and watershed stakeholders must address stormwater system-related contributions of FIB to receiving waters in order to address MS4 stormwater permit related requirements, particularly as required under TMDLs and associated implementation plans. The remainder of this report focuses on information and tools that stormwater managers need to address these requirements, as well as to develop realistic expectations of possible control measures to reduce FIB in urban stormwater.

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3 SOURCES OF FIB IN URBAN AREAS

3.1 Overview of Potential FIB Sources in Urbanized Areas

In order to develop an effective plan for managing and reducing FIB in urbanized areas, it is first necessary to identify the likely sources and associated transport pathways to receiving waters. Effectively targeting source controls requires substantial information about the land uses and activities within the watershed. Sources of pathogens and FIB in MS4s and receiving waters vary widely, originating from both animal and human sources. Representative sources of FIB in urbanized areas include SSOs, CSOs, wet weather (stormwater) discharges from MS4s, illicit connections to storm sewer systems (e.g., sanitary sewer connections to the storm sewer), illicit discharges to storm sewer systems (e.g., power washing), failing or improperly located onsite wastewater treatment systems (septic systems), wastewater treatment plants, urban wildlife, domestic pets, agriculture (e.g., ranchettes), and other sources. Allowed discharges to MS4s such as irrigation runoff and uncontaminated groundwater discharges may also transport FIB originating from other sources. From a regulatory perspective, MS4 permittees are not required to address all of these sources (e.g., CSOs, non-point sources);[3] however, it is beneficial for MS4 permittees to have a broad understanding of the diverse sources of FIB that may be present in impaired waterbodies that receive discharges from the MS4. Table 3-1 provides a summary of potential FIB sources that communities should consider, depending on the conditions potentially present in a specific watershed.

Although agricultural sources are not the focus of this report, both livestock and manure management can be agricultural sources of FIB in watersheds where MS4 permittees are working toward watershed-scale solutions. Secondary sources of persistent FIB include sediments in receiving waters, biofilms in storm sewers and waterbody substrate/sediments, and naturalized FIB associated with plants (e.g., kelp) and soil (Francy et al. 2003, Ran et al. 2013, Byapanahalli et al. 2012, McCarthy 2009, Ellis et al. 1998, Ishii and Sadowsky 2008, among others).

Although some of these sources can be controlled to an appreciable extent (e.g., wastewater discharges, illicit connections), other sources are much more difficult to control. These diffuse and often mobile sources include wildlife such as raccoons, beavers, birds, etc., as well as environmental sources, such as the biofilms and sediments which provide a stable habitat for these organisms to reproduce. Properly accounting for and identifying potential sources is the first step in working toward minimizing FIB contributions from controllable sources. The remainder of this chapter further discusses these sources, closing with a case study providing an example of a source prioritization process for urban areas. Monitoring to identify these sources is discussed in Chapter 5, Chapter 6 discusses statistical analysis of data, and Chapters 7 and 8 discuss management and control.

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[3] Examples of sources that MS4s typically are not required to address include, but are not limited to, CSOs, WWTP effluent, other NPDES-permitted stormwater discharges (e.g., industrial, construction), other NPDES permitted discharges (e.g., dewatering, groundwater treatment systems), non-point source discharges (e.g., state/federal parks, other open space and natural areas not served by the MS4), homeless encampments within receiving water corridors, agricultural runoff, and other sources.

Table 3-1. Potential Sources of FIB in Urbanized Areas and Adjoining Watersheds

General Category	Source/Activity
Municipal Sanitary Infrastructure (piped)	Sanitary sewer overflows (SSOs)
	Combined sewer overflows (CSOs); regulated under NPDES/LTCP
	Leaky sewer pipes (Exfiltration) (see Sercu et al. 2011)
	Illicit Sanitary Connections to MS4
	WWTPs (if inadequate treatment or upsets); regulated under NPDES
Other Human Sanitary Sources (some also attract urban wildlife)	Leaky or failing septic systems
	Homeless encampments
	Porta-Potties
	Dumpsters (e.g., diapers, pet waste, urban wildlife)
	Trash cans
	Garbage trucks
Domestic Pets	Dogs, cats, etc.
Urban Wildlife (naturally-occurring and human attracted)	Rodents/vectors (rats, raccoons, squirrels, opossums)
	Birds (gulls, pigeons, swallows, etc.)
	Open space (coyotes, foxes, beavers, feral cats, etc.)
Other Urban Sources (including areas that attract vectors)	Landfills
	Food processing facilities
	Outdoor dining
	Restaurant grease bins
	Bars/stairwells (washdown areas)
	Piers/docks
Urban Non-stormwater Discharges (Potentially mobilizing surface-deposited FIB)	Power washing
	Excessive irrigation/overspray
	Car washing
	Pools/hot tubs
	Reclaimed water/graywater (if not properly managed)
MS4 Infrastructure	Illegal dumping
	Illicit sanitary connections to MS4 (*also listed above*)
	Leaky sewer pipes (exfiltration) (*also listed above*)
	Biofilms/regrowth
	Decaying plant matter, litter and sediment in the storm drain system
Recreational Sources	Bathers and/or boaters
	RVs (mobile)
Agricultural Sources (potentially including ranchettes within MS4 boundaries)	Livestock, manure storage
	Livestock, pasture
	Livestock, corrals
	Livestock, confined animal feeding operations (CAFO) (NPDES-regulated)
	Manure spreading, pastures/crops
	Municipal biosolids re-use
	Reclaimed water
	Irrigation tailwater
	Slaughterhouses (NPDES-regulated)
Natural Open Space/Forested Areas	Wildlife populations
	Grazing
Other Naturalized Sources	Beach wrackline (flies, decaying plants), plants/algae, sand, soil (naturalized FIB)

Note: this table builds upon previous work by San Diego County (Armand Ruby Consulting 2011).

3.2 Sanitary Sources

Sanitary sources (i.e., human-generated sewage) of fecal contamination of receiving waters can occur under several conditions. These sources of FIB are the highest priority for source controls in impaired watersheds since they are more likely to contain human pathogens. These sources include:

- **Combined Sewer Overflows (CSOs):** During wet weather conditions, stormwater can overwhelm the capacity of CSO infrastructure, including both the piping system and/or the treatment plant. This is a major problem in certain parts of the country, particularly those cities/regions where older infrastructure is in place, combining sanitary and storm drain flows. Depending on the sanitary treatment plant design, flows above the maximum allowable flow for treatment may receive primary treatment or may be bypassed after the headworks. In certain plants, the bypassed flow will be mixed with the treated water prior to disinfection; in other plants, it will be mixed with the treated flow after the treated flow is disinfected and just prior to discharge to the receiving water.

- **Sanitary Sewer Overflows (SSOs):** In separate sewer systems, two sets of pipes exist underground: one to transport sanitary wastewater to the treatment plant and a second to transport stormwater runoff to the receiving water. Sanitary sewer overflows can occur due to excessive inflow & infiltration (I&I), clogging, or due to lift station failures. Most reaches of separate sanitary sewer systems are designed to operate by gravity; therefore, a safety factor is incorporated into the calculation of design flows for the pipes and treatment plant. It is assumed that the gravity piping system will not be leak-proof. In these separate sewer systems, stormwater enters through the sanitary sewer manholes or infiltrates through leaky pipes. In older areas of the country, these piping systems that had a design life of 25 – 50 years have been in the ground for well above this limit and may be cracked and degraded. Stormwater in excess of the I&I assumption can enter the piping system during wet weather and result in overflows, either in the piping system or at the treatment plant.

EPA's CSO Policy

EPA's CSO Control Policy is a national framework for control of CSOs through the NPDES program. The policy resulted from negotiations among municipal organizations, environmental groups, and state agencies. It provides guidance on how to meet the Clean Water Act's pollution control goals as flexibly and cost-effectively as possible. The CSO Policy was published April 19, 1994, at 59 Fed. Reg. 18688. The Policy contains four fundamental principles to ensure that CSO controls are cost-effective and meet local environmental objectives:

- Clear levels of control to meet health and environmental objectives.
- Flexibility to consider the site-specific nature of CSOs and find the most cost-effective way to control them.
- Phased implementation of CSO controls to accommodate a community's financial capability.
- Review and revision of water quality standards during the development of CSO control plans to reflect the site-specific wet weather impacts of CSOs.

- **Publically Owned Wastewater Treatment Facilities (WWTPs):** Although historically WWTPs likely contributed to FIB and pathogen loading at many locations in the U.S., the ready availability of disinfection and NPDES permit requirements have dramatically reduced contributions of FIB from WWTPs. In many urban areas, discharges from WWTPs have much lower FIB concentrations than the receiving water itself and achieve consistently low discharge concentrations through ultraviolet (UV) light disinfection, chlorination or other techniques. Nonetheless, equipment malfunctions and other upsets of WWTPs can and do occur, so WWTPs can still be a source of FIB loading in some watersheds under certain conditions. Additionally, smaller package plants have been identified as a source in some communities (TCEQ 2013).

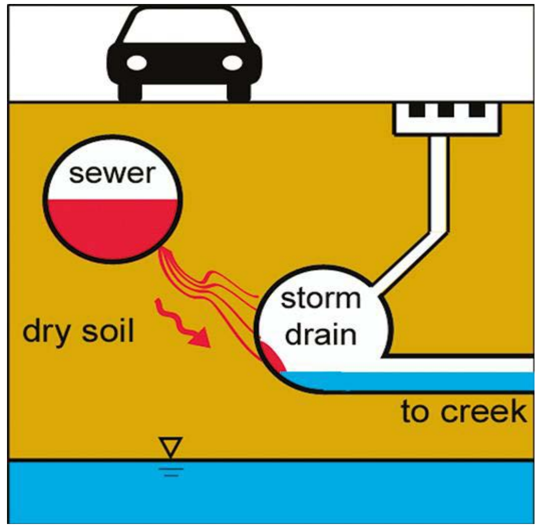


Figure 3-1. Leaking Sanitary Sewer Exfiltrating to Storm Sewer (Source: Sercu et al. 2011[4])

- **Illicit Sanitary Sewer Connections to Storm Sewer System**: In some cases, sanitary sewer pipe connections to the separate storm sewer system occur, either intentionally or inadvertently. Many cities have found that detecting illicit discharges and correcting them have addressed a substantial portion of the sanitary flows into receiving waters. MS4 permittees are required to implement illicit discharge detection and elimination (IDDE) programs to address such sources.

- **Leaking Sanitary Sewer Exfiltration to Storm Sewer Systems**: An often underestimated source of FIB in the storm drain is leaking sanitary sewer lines, where exfiltrated sanitary flow infiltrates into the storm drain (Figure 3-1, Sercu et al. 2011[4]). This can occur in locations where sanitary lines are above the storm drain and flow by gravity into the storm drain (Sercu et al. 2011). Another variation of this condition can occur in older communities where sewer taps have shifted or cracked underground and no longer properly connect to the sanitary sewer main line (Novick 2012).

- **Failing Septic Systems**: Failing on-lot wastewater systems, primarily septic systems, also are a potential source of poorly or untreated sewage into either the storm sewer system or directly to receiving waters. Septic systems and piping can leak and/or allow stormwater to enter and displace sewage/septage into the ground where it can leak into a nearby storm

[4] Figure 3-1 reprinted with permission from: Sercu, B.S., Van De Werfhorst, L.C., Murray, J.L.S., and Holden P.S. (2011). "Sewage Exfiltration As a Source of Storm Drain Contamination during Dry Weather in Urban Watersheds," *Environ. Sci. Technol.*, 45 (17): 7151–7157. 2011. America Chemical Society.

sewer pipe, onto the ground where it is transported via overland flow, or into the groundwater where the pathogens may be transported to a surface receiving water.

- **Direct Human Sources:** There also is the potential for the direct discharge of human waste into receiving waters. These sources can take several different forms such as:
    - Temporary or permanent homeless encampments along waterways where human waste is disposed of in make-shift latrines near the stream or thrown into the stream itself. This can be a common problem in beach communities and highly urbanized areas with an urban stream corridor (City of Santa Barbara 2012).
    - Recreational users of water, particularly young children, who defecate in the water directly.
    - Late-night use of parking lots, sidewalks, planters, and stairwells as latrines, that are later washed into gutters and storm drains (City of Santa Barbara 2012).
- **Recreational Vehicle Dumping/Leaking**: In some communities, particularly vacation destinations, RV dumping or leaking into the storm drains has been identified as a potential source (City of Santa Barbara 2012).

3.3 Wet Weather Discharges to Storm Sewer Systems (Non-sanitary)

Regardless of whether the original source of FIB is natural or human-caused, FIB concentrations in urban stormwater are typically well above primary contact recreation stream standards, regardless of the land use. Urban surfaces are subject to the deposition of contaminants, including FIB, which then may be washed off by rainfall or snowmelt into the storm sewer system. Pitt et al. (2004c) compiled urban stormwater runoff data throughout the U.S. to develop the National Stormwater Quality Database (NSQD), with statistical characterization of fecal coliform by land use shown in Figure 3-2. Even open space areas showed fecal coliform concentrations well above a 200 cfu/100 mL primary contact recreation standard (pre-1986 EPA criteria). Figure 3-2 indicates large variations in fecal coliform observations in all land uses, and generally overlapping boxes. The 25th to 75th percentile fecal coliform values at most monitoring locations are between 1,000 and 20,000 MPN/100 mL. Statistically, transportation-related land uses had the lowest values, and residential areas had the highest values. The large number of data observations (several hundred in each category) enabled significant differences to be statistically identified, although there are obviously large overlaps between the different land uses.

As another example of findings related to wet weather monitoring, Table 3-2 provides constituent load calculations for a subwatershed in the City of San Diego, based on wet weather sampling conducted for various land uses as part of source tracking efforts in the San Diego River watershed. Low-density residential land use was identified as the most significant source of wet weather loads, both in terms of unit loading per acre, as well as with regard to percent of total measured load (Weston 2009a).

Figure 3-2. Box and Whisker Plots of Fecal Coliform in Stormwater Data
(Source: Pitt et al. 2004c)

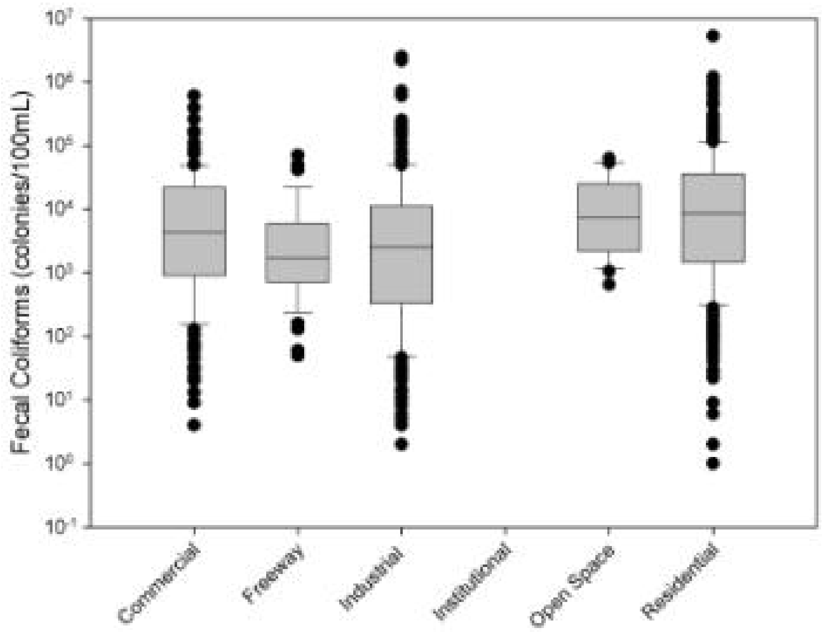
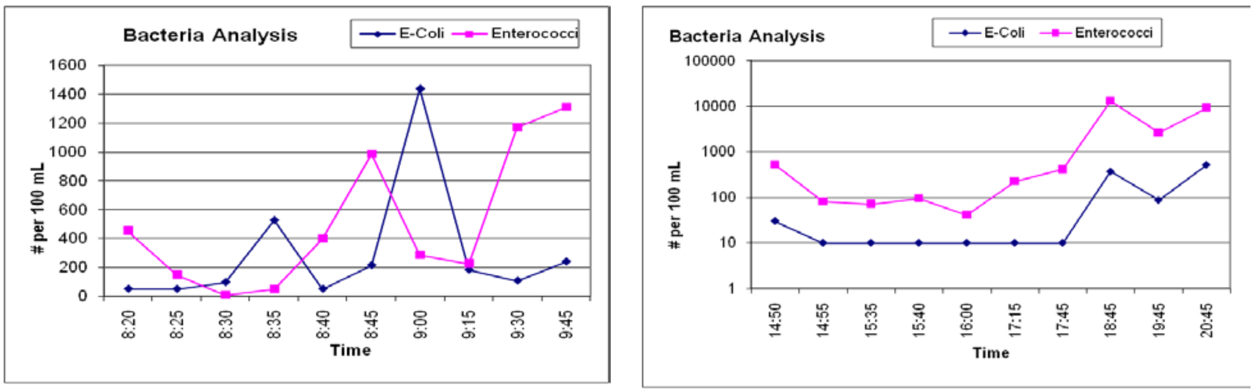


Table 3-2. Wet Weather Enterococci Loads by Land Use for Serra Mesa Subwatershed in the City of San Diego
(Source: Weston 2009a)

Analyte	Land Use	Total Acres	Land Use % of Entire Serra Mesa Drainage	Load/Acre	Total Loads	% of Total Measured Load
Enterococci	Commercial	20.6	1.7%	57,725,587 MPN/a	1,190,301,605 MPN	1.6%
	Open space	219.8	17.7%	9,379,238 MPN/a	2,061,793,336 MPN	2.7%
	High-density residential	111.2	9.0%	18,634,759 MPN/a	2,072,478,945 MPN	2.7%
	Low-density residential	327.6	26.4%	173,830,308 MPN/a	56,942,204,523 MPN	75.5%
	Transportation	196.5	15.8%	45,467,208 MPN/a	8,932,621,939 MPN	11.8%
	Airport	203.9	16.4%	20,572,130 MPN/a	4,194,657,393 MPN	5.6%

When evaluating wet weather data associated with various source areas, an understanding of washoff mechanisms for various land surfaces is helpful. Washoff mechanisms for bacteria are different for paved and non-paved areas, along with differences in their source loadings, and survival characteristics. Figure 3-3 provides plots of *E. coli* and enterococci counts in stormwater from a 0.4 ha mostly paved area during two moderate to large rains in Tuscaloosa, AL. In both cases, the highest counts were observed later in the rains, likely associated with sheetflows originating from landscaped areas surrounding the paved area reaching the monitoring location at the outfall. Being a parking area for a park, it was noted that dogs deposited feces preferentially on the surrounding lawn areas rather than on the pavement. Even during very large rains, this site never had any decreased bacterial levels later in the event, as the bacteria were not likely source limited.

Figure 3-3. FIB Washoff During Moderate Rains for Paved Area (0.4 ha), Tuscaloosa, AL



Maestre (2005) further explored stormwater runoff characteristics, including whether a "first flush" of stormwater constituents existed for various constituents and land uses. Concentrations during the first 30 minutes of the runoff period were compared with the whole runoff period for several hundred events. This investigation indicated that a first-flush effect (increased concentrations at the beginning of an event) was not present for all the land uses, and certainly not for all constituents. Commercial and residential areas were more likely to show this phenomenon, especially if the peak rainfall occurred near the beginning of the event. It is expected that this effect will be more likely to occur in a watershed with a high level of imperviousness, but the data indicated first flush phenomenon less than 50% of the time, even for the most impervious areas. Groups of constituents showed different behavior for different land uses. All of the heavy metals evaluated showed higher concentrations at the beginning of the event in the commercial land use category. Similarly, all of the nutrients showed a higher concentration in residential land uses, except for total nitrogen and ortho-phosphorus. For bacteria, none of the land uses showed a higher amount of bacteria during the beginning of the events compared to the complete events. Other conventional constituents showed elevated initial concentrations in commercial, residential and institutional land uses. Findings for bacteria are also supported by traditional first-flush analyses by McCarthy (2009) and Hathaway and Hunt (2009), who did not find a consistent first-flush effect across a combined five watersheds in Melbourne, Australia, and Raleigh, North Carolina, respectively.

These data question the assumption of first flushes for stormwater bacteria, or that bacteria are source limited in urban areas. High bacteria levels seem to occur even for large rains of several inches in depth, especially when small amounts of debris or landscaped areas are present. Areas having better or more habitats for urban wildlife, or where pets defecate, seem to have higher levels of stormwater bacteria.

For wet weather flows, models may be useful for estimating loads associated with various land uses, particularly when calibrated to local conditions and when land use characteristics have been adequately documented. For example, WinSLAMM includes event mean concentration (EMC) data for fecal coliform associated with various source areas (see www.winslamm.com), and SBPAT includes EMCs for storm runoff for various land uses in southern California (see www.sbpat.net).

In closing, as discussed in Section 2.5, the human health significance of these elevated FIB concentrations in urban runoff remains unclear, given the relatively few studies that have measured pathogen concentrations in non-sanitary impacted urban runoff. For example, Schroeder et al. (2002) investigated the presence of pathogens in urban storm drains and concluded that although pathogens can be found in urban drainage, there did not appear to be a relationship between the presence of human pathogens and the concentration or presence of FIB.

3.4 Dry Weather Discharges to Storm Sewer Systems

There are a variety of dry-weather discharges that can potentially transport pathogens and FIB to receiving waters through the storm sewer system or through overland flow. These can include non-stormwater discharges that transport deposited material to the storm sewer system, as well as improper disposal of FIB-related wastes into the storm sewer system. Examples of these discharges include washwater from cleaning activities such as car washing, window washing, power washing of equipment and buildings, sidewalk cleaning, dumpster washdowns, etc. These washwaters have the potential to dislodge and transport FIB into the storm sewer and eventually to the receiving water. Dry cleanup methods, such as sweeping and leaf blowing, also may transport some fecal matter directly to the storm drain inlet because people may put their leaf residue and dirt in the gutter, assuming that the rain will transport it away from their site. In other cases, pet owners may incorrectly view the storm drain as an appropriate place to dispose of pet waste bags or to dump kitty litter.

As an example of dry weather discharges in urban areas, Figure 3-4 provides a summary of dry weather flow observations in San Diego County for residential and commercial areas (Weston 2009a). This figure indicates that irrigation runoff is the dominant source of dry weather flows in residential and commercial areas in San Diego. A higher frequency of runoff in residential areas was observed compared to commercial land uses. Commercial areas also had frequent runoff from wash down areas and dumpster/grease trap leaks. Water quality samples collected from these runoff sources showed varying levels of FIB, as shown for enterococcus for specific commercial activities in San Diego County in Figure 3-5. The highest average FIB concentrations were associated with dumpster leakage/grease traps (38,291 MPN/100 mL) and wash-down (1,822 MPN/100 mL). These flow sources were commonly associated with restaurant, food outlets, and food distributors. Irrigation leaks and irrigation runoff also had relatively high concentrations of FIB (Weston 2009a).

Figure 3-4. Observed Dry-weather Flow-Related Activities for Residential and Commercial Land Uses in San Diego County (Source: Weston 2009a)

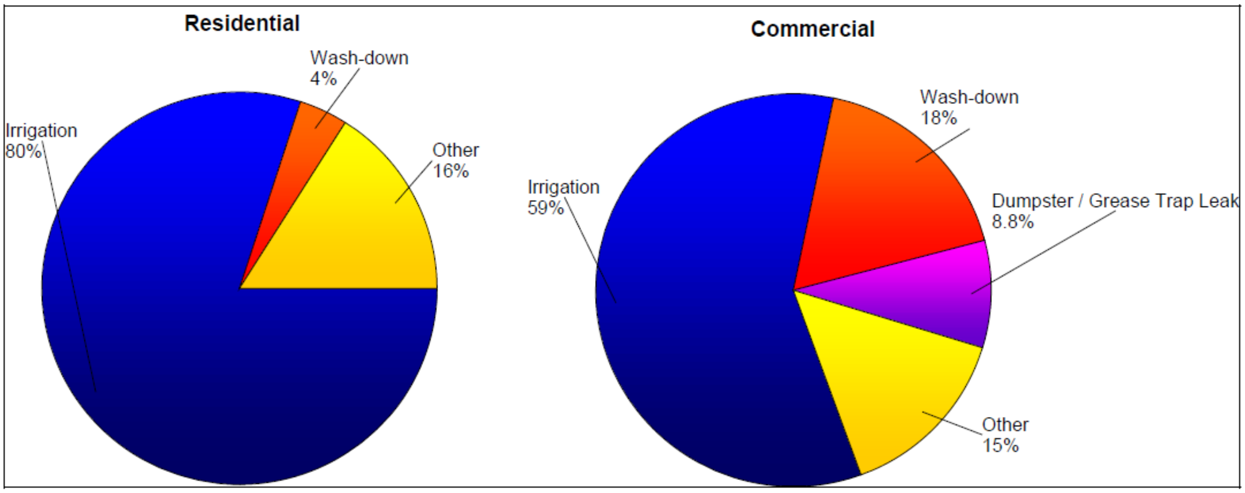
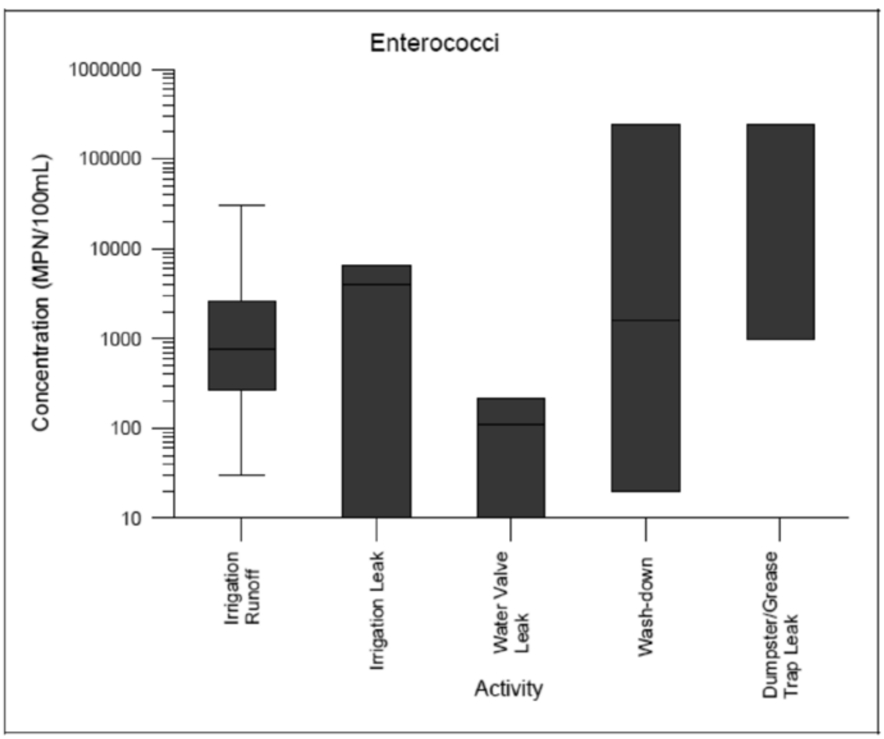


Figure 3-5. Observed Dry-weather Flow-Related Enterococci for Residential and Commercial Land Uses in San Diego County (Source: Weston 2009a)



3.5 Urban Wildlife and Domesticated Animals

Both wild and domestic animals in urban areas are known sources of FIB that present significant management challenges for attainment of stream standards. In *Guidance for Development of Pathogen TMDLs* (EPA 2002), EPA discusses wildlife as an important non-point source of FIB, using beavers, deer, geese, ducks and herons as examples. Table 3-1 provides examples of fecal deposition rates and associated FIB concentrations in feces of various urban mammals and birds. There is a wide range of values, but many researchers have found correlations between feces moisture and bacteria content, with dry feces (such as rabbit feces) being very low in bacteria count, while many birds (especially the water birds) having very moist feces with much greater bacteria content (Pitt 1983). Fecal material can enter waterbodies through direct deposition, as well as from stormwater and dry-weather washing of feces deposited on the ground and other surfaces (e.g., automobiles, sidewalks) into storm sewers and receiving waters. Although urban wildlife can include many animals, the primary focus of research on sources of stormwater pathogens and indicators has been birds. However, raccoons and other mammals may also be significant contributors, particularly in urban areas where open space corridors have been preserved along waterways. Geldreich (1976) reported that large populations of rodents may also contribute significant amounts of fecal material in urban areas. Additional information on birds and other animal sources follows. (Note: urban wildlife can contribute directly to receiving water impairments, without being directly associated with the MS4 itself.)

3.5.1 Birds

Birds are natural sources of FIB loading to streams that may cause waterbodies to exceed RWQC. In particular, geese are considered as public nuisances due to large populations, creating large amounts of feces, especially in open-space areas (e.g., parks, playing fields, ponds) (Manny et al. 1975, French and Parkhurst 2009, Bowen and Valiela 2004, Kear 1963). Clark (2003) reported that non-migratory Canada geese increased eight-fold in a 20-year period (1980s to early 2000s) in North America. Pigeons, blackbirds, starlings, ducks, and other birds also can pose similar problems when they roost on public buildings and bridges.

Geese and other birds can excrete zoonotic pathogens (e.g., *Campylobacter*); however, the association between human illness as a result of recreating in or around waterbodies impacted by bird feces is not well documented (Clark 2004), particularly for urban birds associated with inland waters. As previously discussed in Section 2.5, a QMRA study by Soller et al. (2010b) suggests that risk from birds such as gulls and poultry may be lower than other fecal sources (e.g., cattle, humans).

Swallows nesting under a county bridge over Boulder Creek. (Photo Courtesy Wright Water Engineers.)

Table 3-3. Loading Rates from Mammals Birds (Adapted from Pitt 1983)

Animal	Deposition (grams/ animal/day)	Deposition Rate Reference	Animal	Fecal Coliforms (median MPN/ g feces)	FIB Concentration Reference
Urban Mammals					
Humans	150	Geldreich 1976	humans	13,000,000	Geldreich 1976
Domestic pets					
cats	70	Howe 1969	cats	7,900,000	Geldreich, et al. 1968
dogs	140 23 to 100	Howe 1969 Marron and Senn 1974	dogs	23,000,000	Geldreich 1976
Possible urban wildlife					
rabbits	550	Howe 1969	rabbits	20	Geldreich, et al. 1968
rats	35	Howe 1969	rats	180,000 330,000	Geldreich and Kenner 1969 Geldreich, et al. 1968
			rodents	160,000	Geldreich and Kenner 1969
mice	10	Howe 1969	field mice	330,000	Geldreich 1976
			chipmunks	150,000	Geldreich, et al. 1968
Possible urban birds					
pigeons	25 to 50	Gore & Storrie/Proctor & Redfern 1981a	pigeons	50,000 160,000,000 (*E. coli*/gram= 170,000,000)	Environment Canada 1980 Oshiro and Fujioka 1995
			robins	25,000	Geldreich 1976
			English sparrows	25,000	Geldreich 1976
			Starlings	10,000	Geldreich 1976
			Red-winged blackbirds	9,000	Geldreich 1976
Possible urban water birds					
ducks	70 340	Howe 1969 Geldreich 1976	ducks	33,000,000	Geldreich 1976
geese	160	Howe 1969	geese	300,000-6,000,000 (summer only) 820 - 300,000 (old, dried on docks)	Alderisio and Deluca 1999
gulls	10 to 25	Gould and Fletcher 1978	Herring gulls	71,000,000	Gore and Storrie/Proctor and Redfern 1981a
			Lesser black-backed gulls	370,000,000	Environment Canada 1980
			common gulls	53,000,000	Environment Canada 1980
			Black-headed gulls	27,000,000	Environment Canada 1980
			Lake Merritt bird mixture	200,000	Pitt and Bozeman 1979
			swans	320,000	Environment Canada 1980

As shown in Table 3-3, FIB loading associated with birds can be substantial. Nuisance geese and other water birds (e.g., ducks) in urban-suburban areas tend to occur in areas where lawns abut a waterbody and where they have the ability to detect approaching predators (Conover and Kania 1991). Numerous examples exist of studies that have linked birds to elevated FIB in receiving waters. A few of these studies include the following:

- Alderisio and Deluca (1999), at New York's Kensico Reservoir, showed that sample sites with large numbers of roosting waterfowl had elevated fecal coliform levels compared to sites with no waterfowl present (concentrations of <1/100 mL). Additionally, goose feces concentrations were one to two orders of magnitude higher in summer than the overall average. After a benign waterfowl mitigation program began in 1992, the elevated seasonal fecal coliform concentration attributed to birds was "largely eliminated."

- Using microbial source tracking near Manitou Springs, Colorado, Stoeckel (2010) ruled out humans, pets, and cattle as the likely sources of contamination to Fountain Creek. The study concluded that pigeons were the most likely source of fecal contamination. Power washing of surfaces with heavy pigeon use was identified as a potential FIB transport pathway to the receiving stream.

- In Mission Bay, California, Kolb and Roberts (2009) used microbial source tracking to determine that nearly 70 percent of FIB loading was associated with birds.

- Kirschner et al. (2004), studying six shallow saline habitats, found that wild bird abundance and feces production were significantly correlated to the abundance of FIB in the water. Median concentrations of *E. coli* ranged from 4 cfu/100 mL to 1,200 cfu/100 mL, with maximum concentrations of 1.3×10^4 cfu/100 mL for *E. coli*, 1.8×10^4 cfu/100 mL for fecal coliform, and 6.0×10^4 cfu/100 for enterococci. The other environmental variables that affected the FIB concentrations included rainfall, total suspended solids, chlorophyll-a and total phosphorus, which had significant positive correlations with enterococci concentrations.

- Shergill and Pitt (2004), studying dry and wet weather flows from urban source areas in Alabama, found that *E. coli* and enterococci concentrations greater than 2,400 and 24,000 MPN/100 mL, respectively, were observed, suggesting that urban birds and other animals can be considered significant sources of FIB. FIB levels from tree-covered roofs prone to urban animal use (squirrels and birds) were significantly higher than from roofs not having tree overstories and not exposed to such use. Concentrations varied seasonally, with lower temperatures associated with decreased FIB levels.

- Kadlec and Wallace (2009) noted that treatment wetlands contain numerous animals that excrete FIB. Based on a several-year study example from Tres Rios, Arizona, the authors noted that fecal coliform in the wetland increased from less than 10 cfu/100 mL in the inflow to several thousand in the outflow. Conditions that favored transmission of pathogens and indicators were exposed mudflats with stressed/overcrowded bird populations.

- Hussong et al. (1979), in the Chesapeake Bay, found that overwintering migrant geese and swans were a source of *E. coli* and caused increased coliform counts in the estuarine waters. Fecal coliform in shallow aquatic environments ranged from 10^2 to 10^3 cfu/100 mL in surface water and 10^4 cfu/100 mL in sediment.

These are just a few examples of studies that identified birds as a significant cause of elevated FIB concentrations in waterbodies. Fleming and Fraser (2001) summarize several additional studies, including work by Standridge et al. (1979) in Madison, WI, Benton et al. (1983) in Scotland, Valiela et al. (1991) in Buttermilk Bay, MA, and Levesque et al. (1993) in Quebec. From a municipal stormwater management perspective, the important finding is that there are many studies that demonstrate birds can cause exceedances of FIB standards, so this potential source of FIB loading should not be underestimated. As discussed in Chapter 5, new microbial test methods (e.g., qPCR) are now available to help confirm whether birds are contributors to FIB loading.

3.5.2 Wild and Domestic Mammals in Urban Areas

Urban mammals can be divided into two categories: wildlife (deer, foxes, coyotes, squirrels, raccoons, opossums, mice, rats, and the occasional bears, moose, elk, etc.) and domesticated (cats, dogs, ferrets, pigs, etc.). In some urban areas, livestock, such as chickens, goats, and a few horses and cattle on ranchettes may be sources, particularly when animal pens are adjacent to and drain to nearby receiving waters. In many urban area sampling studies, the contributions of wildlife versus domesticated animals have not been separated. A complete inventory of studies identifying animal-related, non-bird source of FIB has not been completed for this report; however, a few examples are provided below.

In Tuscaloosa, Alabama, Shergill and Pitt (2004) found that FIB concentrations were not significantly different at ground-level areas with varying levels of known animal activity (pet vs. non-pet areas), indicating that urban wildlife is a substantial contributor of FIB. (A relatively small data set may have limited detection of statistically significant differences.) However, areas with higher domestic animal activity generally had higher FIB levels, especially in the warmer months. FIB levels were not affected by total rain depths or rain intensities; however, seasonal effects were observed. In addition, the ratio of *E. coli*/enterococci varied among source areas, dry vs. wet-weather sampling, and seasons, with wet-weather samples having mostly higher enterococci levels than *E. coli*, while dry weather source area samples (such as springs and irrigation runoff) had higher *E. coli* levels. Urban domestic and feral pets also were implicated as substantial sources of high *E. coli* levels in storm sewers draining to the Huron River (Michigan) with isolates from cats appearing more frequently in samples where pet waste was detected. Raccoons as *E. coli* sources were found in late summer and fall (Ram et al. 2007).

Raccoons in an urban storm drain manhole. Photo Courtesy: Andy Taylor, City of Boulder, CO.

In recent sampling in Boulder, Colorado, raccoons were identified as a key source of FIB in the storm sewer system, as evidenced by defecation "latrines" at junctions in the storm sewer system. After the city power washed the pipe and placed controls[5] on the inlet and outlet to the suspect storm sewer, FIB concentrations dropped dramatically from this specific storm drain and have remained low (City of Boulder 2013).

Mass Balance of Animal Fecal Sources Affecting the Lower Rideau River (Pitt 1983)

As part of the Lower Rideau River bacteria studies conducted in the early 1980s in Ottowa (Pitt 1983), Pitt estimated the deposition of bacteria associated with animal feces in an area using feces deposition rates, bacteria content of the feces, and the animal populations, as summarized in the table below, assuming various conveyance fractions for delivery to the river. Obviously, there is substantial uncertainty in this calculation; however, it was useful for focusing attention to the pigeons (on the bridge), ducks and dogs. These analyses directed special efforts to remove the pigeons from the bridge and to enforce dog feces cleanup regulations. The ducks were more difficult to control, as they were protected migratory wildlife. These analyses also pointed out the need to develop more precise methods to quantify the sources of the bacteria and to study their transport and fate.

Animals Present in the Lower Rideau River Watershed	Animal population	Annual feces discharge (g/yr)	Est. fraction to river	Feces discharge to river (g/yr)	FC MPN/g	FC MPN/yr	% of FC discharge to river
Discharge to Land							
Dogs	16,000	$6X10^8$	0.01	$6X10^6$	$2.3X10^7$	$1.4X10^{14}$	15
Cats	16,000	$4X10^8$	0.001	$4X10^5$	$7.9X10^6$	$3.2X10^{12}$	<1
Robins	28,000	$1X10^8$	0.01	$1X10^6$	$2.5X10^4$	$2.5X10^{10}$	<1
Pigeons (land)	4,000	$5X10^7$	0.01	$5X10^5$	$1.0X10^8$	$5.0X10^{13}$	5
Direct Discharge to River							
Pigeons (on bridge)	600	$8X10^6$	0.5	$4X10^6$	$1.0X10^8$	$4.0X10^{14}$	42
Ducks (on river)	100	$2X10^7$	0.5	$1X10^7$	$3.3X10^7$	$3.3X10^{14}$	35
Gulls (on river)	150	$1X10^6$	0.5	$5X10^5$	$5.3X10^7$	$2.7X10^{13}$	3
Swans (on river)	15	$1X10^6$	0.5	$5X10^5$	$3.2X10^5$	$1.6X10^{11}$	<1
Other birds (on river)	10	$4X10^4$	0.5	$2X10^4$	$2.5X10^4$	$5X10^8$	<1
Total Fecal Coliform Annual Discharges							
Stormwater discharges						$1.9X10^{14}$	20
Direct to river						$7.6X10^{14}$	80
Total						$9.5X10^{14}$	

FC = fecal coliform

[5] Any controls placed on the inlet or outlet of a storm drain must be carefully designed to prevent unintentional clogging of the storm drain, which could result in flooding.

3.6 Environmental (Secondary) Sources of FIB

Environmental reservoirs, or secondary sources, of FIB have been the subject of recent research with regard to their role as a persistent source of elevated FIB in receiving waters. Examples of secondary reservoirs enabling persistence and growth of FIB include:

- Sediment deposited within a sewer pipe, treatment device, or waterbody that can be resuspended as a result of a variety of different physical mechanisms such as high flows, wind, recreational activities such as wading and boating, and presumably turnover of a pond or lake. Representative research related to persistence of FIB in sediment includes studies conducted by Byappanahalli et al. (2003), Byappanahalli et al. (2006), Davies et al. (1995), and Monroe (2009).

- Organic matter such as algae, kelp and decaying organic matter that provides a nutrient supply and shelter for FIB. Kolb and Roberts (2009) noted that decaying kelp along beaches appeared to serve as the "perfect incubator for bacterial growth."

- Interstitial waters in shorelines and beach sand adjacent to waterbodies that can be mixed into the water column due to wading and other shallow water recreational activities. For example, Francy et al. (2003) studied the distribution and source of *E. coli* at five Ohio bathing beaches on Lake Erie and one inland lake during 2000 and 2001. The study found that lake-bottom sediments from outside the bathing area were not significant deposition areas for *E. coli*. In contrast, interstitial water and subsurface sediments from near the "swash zone" were enriched with *E. coli*. For example, *E. coli* concentrations were as high as 100,000/100 mL in some interstitial waters.

- Soil adjacent to waterbodies can also be important FIB sources (Fujioka and Byappanahalli 2003, Byappanahalli and Fujioka 2004). Ran et al. (2013) found that some enterococci are able to persist and grow in the Lake Superior watershed, especially in soil, for a prolonged time after being introduced. Byappanahalli et al. (2012) found that there is mounting evidence for widespread extra-enteric environmental sources and reservoirs of enterococci.

- Biofilms (i.e., slime layer) in urban storm sewer systems (e.g., pipes, curbs and gutters). Skinner et al. (2010) summarize recent research indicating that biofilms in storm sewers provide a safe environment for enhanced FIB replication, supply nutrients and water for biofilm FIB, and offer protection against microbial predators, ultraviolet (UV) light, drying, and disinfectants (citing research by Coghlan 1996, Costerton et al. 1995, Donlan and Costerton 2002, Donlan 2002). McCarthy (2009) further suggested such biofilms in urban stormwater sewers may be flushed out during storm events.

- Wetland areas discharging to recreational waterbodies. For example, Grant et al. (2001) conducted a multidisciplinary study to identify sources of enterococci landward of the coastline at Huntington State and City Beaches in Southern California. High concentrations of enterococci were identified in urban runoff, bird feces, marsh sediments, and on marine vegetation. Urban runoff appeared to have relatively little impact on surf zone water quality because of the long time required for this water to travel from its source to the ocean.

Conversely, enterococci generated in a tidal saltwater marsh located near the beach significantly impacted surf zone water quality. As another example, Graczyk et al. (2009) studied constructed subsurface flow and free-surface flow wetlands in Ireland and found that free surface wetlands discharged more pathogens than were delivered to wetlands with incoming wastewater. Among various findings, it was concluded that wildlife can contribute a substantial load of human zoonotic pathogens to wetlands.

- In a microbial source tracking study in the South Nation River Basin in Ontario, waterborne *E. coli* populations that were distinct from fecal isolates were detected by Lyautey et al. (2010) and hypothesized to possibly be naturalized *E. coli* strains.

Sediment and biofilms are discussed further below.

3.6.1 Sediment

Sediment in receiving water, stormwater BMPs and stormwater discharge pipes is increasingly recognized as a potential source of elevated FIB. These sediments, especially those in pipes that are not exposed to the sun, provide a suitable, moist environment for the growth of microorganisms deposited there (Clark et al. 2010, Weston 2010b). During wet weather, much of this microbially enriched sediment is discharged along with the stormwater. These sediments often function as a reservoir in which microorganisms can persist (Jensen 2002, Mermillod-Blondin et al. 2005, Reeves et al. 2004, Davies et al. 1995, Weston 2010b, LaLiberte and Grimes 1981). For example, Davies et al. (1995) studied the survival of several types of culturable FIB in freshwater and marine sediments from sites near sewage outfalls. Studies using in situ membrane diffusion chambers showed that, with the exception of *C. perfringens*, die-off of the test organisms to 10% of their initial numbers occurred in both marine and freshwater sediments within 85 days. Typical exponential decay models applied to FIB in water did not apply to the sediment survival data, with the exception of fecal streptococci. The survival of seeded *E. coli* in marine sediment also indicated that sediment provides a “favorable, non-starvation environment” for FIB. As another example, pond sediments in the Tecolote Creek watershed in San Diego were sampled and analyzed for FIB in 2008, also indicating that sediments served as a potential reservoir for fecal coliforms and enterococci (Weston 2010b).

These findings are important from an FIB modeling perspective since sedimentation is a key process for removing FIB from the water column. However, if FIB persist and/or grow in sediments, the sediment “sink” can become an FIB source. (See additional discussion in Chapter 5.)

3.6.2 Biofilms

Biofilms, which are surface-attached communities of microorganisms that undergo cell attachment, growth, detachment, and sloughing, are ubiquitous in aquatic systems on the surfaces of sediments, rocks, and plants (Costerton et al. 1995). Once in a biofilm, microorganisms excrete a complex mixture of extra-cellular polymeric substances (EPS), which bind cells together and protect them from predation and harsh environmental conditions (de Beer et al. 1993, de Beer et al. 1994). Biofilms found on aquatic surfaces are normally extremely diverse and include a wide variety of bacteria under almost all conditions. Algal-bacterial biofilms form on surfaces with sufficient light to support photosynthesis (Arnon et al. 2007, Barranguet et al. 2005, Costerton et al. 1995, McLean et al. 2000, Rickard et al. 2003). Biofilms are a concern because they can host a wide variety of pathogens and protect resident organisms from environmental stresses, including biocides (Costerton et al. 1995, Costerton et al. 1987, Hall-Stoodley et al. 2004, Parsek and Singh 2003).

The pathogens *E. coli* O157:H7, *Salmonella enterica*, and *Campylobacter jejuni* are all known to form biofilms (Dewanti and Wong 1995, Joshua et al. 2006, Prouty and Gunn 2003, Prouty et al. 2002, Reisner et al. 2006, Rivas et al. 2007, WHO et al. 2004). Further, biofilm formation can favor the survival of all three of these organisms under both typical environmental conditions and under active disinfection (Dykes et al. 2003, Gibson et al. 2006, Joseph et al. 2001, Korber et al. 1997, Latasa et al. 2005, Murphy et al. 2006, Ryu and Beuchat 2005, Cooley et al. 2003, Ryu and Beuchat 2005). Biofilms in flow cells and batch reactors have been shown to capture and concentrate protozoan cysts, bacteria, and viruses (Buswell et al. 1998, Mackay et al. 1998, Searcy et al. 2006a, Storey and Ashbolt 2001). Much less is known about the interactions of these organisms with natural microbial communities, or the potential for these interactions to affect pathogen persistence.

The transport and fate of FIB in natural waters are controlled by a series of events – initial attachment to sediments, colonization, growth, decay, and detachment. Concentrated doses of pathogens are often released following physical or chemical perturbations, e.g., passage of chemical plumes in groundwater, high flows in rivers (Atherholt et al. 1998, Daly et al. 1998, Donnison et al. 2006b, LeChevallier et al. 1991, Muirhead et al. 2004, Parsek and Singh 2003, Ryan and Elimelech 1996, Stott et al. 2007) and human disturbance of rocks and other surfaces. Slow, long-term release of *C. parvum* from sediments has been observed in column experiments, indicating that much greater release of pathogens occurs than suggested by the filtration theory developed to describe the behavior of inorganic colloids (Cortis et al. 2006, Tufenkji et al. 2003).

Unlike the cyst-forming protozoa (e.g., *Cryptosporidium* and *Giardia*), bacterial pathogens may actively grow in natural aquatic biofilms; however, they must compete with indigenous organisms for nutrients and space. Factors that commonly limit the survival of bacterial pathogens in biofilms include low levels of available nutrients, non-favorable oxygen concentrations, and the competitive, antagonistic and predatory activities of the indigenous microbial population (Banning et al. 2002, 2003; John and Rose 2005). As *E. coli*, *Salmonella*, and *Campylobacter* are all known to form biofilms, they can certainly be expected to be able to colonize natural aquatic biofilms when the right environmental conditions are present. Further, the high diversity of sedimentary biofilms should favor the persistence of fecal pathogens, as a wide variety of niche micro-environments can be found within the biofilm structure (Arnon et al. 2007, Costerton et al. 1995, de Beer et al. 1993, Hall-Stoodley et al. 2004), including low-oxygen microenvironments that favor fecal pathogens such as *Campylobacter* (Buswell et al. 1998). However, very little is known about the processes that control the interactions of these bacterial pathogens with natural environmental biofilms.

Biofilms also can develop in urban storm sewer piping and may be a source of FIB. As an example, Kolb and Roberts (2009) summarized findings of microbial source tracking studies in the San Diego area, where biofilms in storm drains appeared to contribute to FIB loading at

Tecolote Creek. Skinner et al. (2010) reported results of their study of street gutters and storm drains in Newport Beach, CA, suggesting that FIB growth and persistence in biofilms in the gutter may result in high FIB levels. Some highlights of their study, including findings from their literature review, include:

- Biofilms can provide a safe environment for enhanced bacterial replication; supply nutrients and water for needed for growth and reproduction; and offer protection against microbial predators, ultraviolet (UV) light, drying, and disinfectants (Citing Coghlan 1996, Costerton et al. 1995, Donlan and Costerton 2002, Donlan 2002).

- Bacteria can detach from the biofilm surface and enter the water column as single planktonic bacteria or small clumps of bacteria attached to fragments of biofilm. The rate of detachment is related to factors such as water flow velocity, shear forces, nutrient availability, and aging of the biofilm.

- Studies in Orange County, CA, determined that enterococci and fecal coliform were multiplying in bacterial biofilms (Ferguson 2006). Follow-up studies involved introducing FIB-free hose water into a dry street gutter and tested for enterococci and fecal coliform at 10 m, 45 m, and 100 m downstream. There was a progressive rise of both enterococci and fecal coliform with the increased flow distance, indicating that biofilms in street gutters could provide suitable habitat for growth of FIB. The levels were 26,000 enterococci/100 mL and 14,000 fecal coliform/100 mL at the 100-m test site.

- Kolb and Roberts (2009) raise the question of whether the use of these indicators, which can persist in biofilms, are suitable since human enteric viruses, one of the primary causes of swimmer-related gastrointestinal illness (Glass et al. 2009), have not been shown to persist in the biofilms found in street gutters.

Other research has reached similar conclusions. For example, in a San Diego County MS4 co-permittee sponsored study in the San Diego River watershed conducted by SCCWRP, Ferguson et al. (2011) found that biofilms provided a beneficial environment for FIB growth in storm drains.

3.7 Natural Background FIB Loading

In many urban watersheds, there will be at least some controllable sources of FIB; however, it is also likely that natural sources (i.e., non-domesticated animals) will also contribute to on-going periodic exceedances of receiving water standards. Some states currently allow regulatory “off-ramps” when natural sources of FIB are determined to be the primary cause of impairment (Meyerhoff et al. 2006). In other cases, a Use Attainability Analysis (UAA) may be conducted that recognizes the influence of natural or irreducible conditions, thereby supporting a change in designated use for a receiving water. In urban areas, where public access to streams occurs (e.g., waterplay by children), changes in use may not be supported by regulatory agencies due to potential human health risk.

In Southern California, natural background loading is recognized through explicit "natural source exclusion" provisions for site-specific wasteload allocation approaches in TMDLs that allow exceedance frequency rates for standards comparable to those expected in natural areas. Two studies supporting expected exceedance rates in natural areas follow.

As one example, SCCWRP conducted reference watershed studies in Southern California to determine exceedance frequencies for FIB under both wet and dry weather conditions. For the dry weather study, Tiefenthaler et al. (2008) reached these conclusions (quoted directly):

- Higher FIB levels observed during the summer suggest that factors existed which promote FIB growth and regrowth in streams. The positive relationship between temperature and FIB levels suggests that heat induced growth may be a contributing factor to seasonally high FIB levels. In addition, warmer temperatures influence the dissolved oxygen content of the water. Decreased oxygen solubility associated with higher temperature may combine with lower dissolved oxygen levels producing algal blooms, which have been shown in previous studies to support growth of *E. coli* and enterococci in freshwater (Byappanahalli et al. 2003a, 2007). These conditions may in turn accelerate death and decomposition of organic matter in the stream, further enhancing in situ FIB growth. Increases in organic decomposition have been shown to increase survival and regrowth of enteric bacteria and viruses (Novotny and Olem 1994).

- There are three possible sources of FIB observed in natural streams: external inputs from sources such as waterfowl, animals, or soil erosion; internal sources of FIB growth and colonization within the stream associated with decomposition of organic matter; or a combination of the two (Byappanahalli et al. 2003b, Toranzos 2007).

- Higher FIB densities and incidence of water quality standard exceedances during the summer is consistent with the observations of others such as Noble et al. (2000) and Sieracki (1980). Nuzzi and Burhans (1998) compared the responses among FIB at 143 New York beach sites and found that survival was longer in the summer, but that the duration could be mediated by exposure to UV radiation from sunlight. More recently, growth or regrowth of FIB in tropical and temperate soils during the summer months has also been reported (EPA 2000, Ishii et al. 2006). Whitman et al. (1999) attributed a gradual increase of *E. coli* in water and sand at beaches during summer to higher survival and growth at warmer temperatures.

- Another explanation for higher FIB levels during the summer could be higher external sources due to different patterns of use by wildlife and birds. A number of studies have shown that wildlife and other animals can be sources of FIB in run-off (Baxter-Potter and Gilliland 1988, Bagshaw 2002, Stein et al. 2007). Previous studies have quantified that wildlife and bird feces contain high levels of FIB. Cox et al. (2005) measured fecal coliform levels of 10^3 - 10^5 cfu/g from native wildlife in Australian watersheds. Ricca and Cooney (1998) reported that droppings from feral populations of pigeons, geese and herring gulls from the environment around Boston Harbor, MA, USA contained up to 10^8 cfu/100 mL of enterococci. Bacteria from wildlife and birds can be associated with FIB levels in streams used by these animals. Noble et al. (2004) found that birds were a likely source of intermittently high levels of FIB observed in the lower Santa Ana River watershed and the nearby surf zone in southern California. Similarly, Harwood et al. (2000) reported that animals were the dominant sources of FIB at Florida sample sites with relatively low anthropogenic impact. Bacterial source tracking studies conducted in Michigan suggested that feces from pets and raccoons were important contributors to FIB levels in streams and storm sewers (Ram et al. 2007). Moreover, levels increased in the late summer and fall coincident with increased raccoon den mobility following breeding.

- Decreased stream flow may have also contributed to higher FIB levels during the summer months. Although there was no statistically significant relationship between flow and FIB densities, in all cases, densities increased exponentially when stream flow decreased below approximately 0.05 m^3/s (2 cfs). In addition, median annual FIB densities were higher in intermittent streams than in perennial, with the differences being mainly due to high levels in the period immediately prior to streams drying up. Despite the differences between perennial and intermittent streams, the annual ranges of observed FIB levels overlapped substantially. Therefore, the combined range of FIB levels for perennial and intermittent streams observed in this study should reflect expected levels in natural streams throughout southern CA.

- Relatively minor perturbations in the contributing watershed can cause sites to quickly deviate from background conditions. Four sites originally considered, but later rejected from the study, had FIB levels 2-3 log units greater than the natural sites retained, but significantly lower than levels observed in the developed Ballona Creek watershed. The watersheds of these four sites were almost entirely natural open space, but had small portions subject to agricultural or transportation related runoff. In one instance, a portion of the contributing watershed was affected by a recent fire. These small perturbations in the watershed led to dramatic changes in FIB levels that moved sites away from reference conditions. Although these sites were not included in the analysis of background conditions, they provide valuable insight into the sensitivity of natural watersheds to small increases in anthropogenic sources.[6]

In another SCCWRP study, Griffith et al. (2006b) conducted the study of wet weather FIB conditions at non-human impacted beaches in Southern California. Six reference beaches were sampled during nine storm events during the 2004-2005 and 2005-2006 wet seasons. Samples were analyzed for total coliform, *E. coli*, and enterococci in the discharge from the undeveloped watershed and in the wave wash where the discharge and surf zone initially mix. Griffith et al. reported these findings (quoted directly):

- Samples collected during wet weather exceeded water quality thresholds established by the State of California greater than 10 times more frequently during wet weather than during

[6]Some scientists have questioned whether these minimally impacted sites should have been excluded from the study and note that the exclusion of these sites may result in an underestimation of exceedance frequencies of FIB standards (Flow Science 2010).

recent dry weather in summer or winter, although the frequency differed by beach. These exceedences were greatest <24 hours following recorded rainfall, then steadily declined for the following three days.

- Early season storms exceeded water quality thresholds more than twice as frequently as late season storms. In addition, over half of these early season storms exceeded thresholds for multiple bacterial indicators, while the vast majority of late season storms only exceeded thresholds for a single bacterial indicator.

- Large storms exceeded water quality thresholds three times more frequently than smaller-sized storms. This was partly due to the breaching of sand berms during large storm events; small storms could not breach these berms and this restricted watershed discharge from entering the surf zone. When watershed discharges did enter the surf zone, FIB concentrations in the wave wash were correlated with watershed FIB flux.

As a result of the Southern California reference watershed studies, TMDLs have been developed with allowable exceedance rates in several TMDLs, as summarized in Table 3-4. These allowable exceedances only pertain to single sample maximum values rather than geometric mean values and remain elusive to attain in most urban areas in Southern California thus far.

Table 3-4. Allowable Exceedance Rates of SSM Criteria in Southern California TMDLs
(as summarized by Brandon Steets, Geosyntec Consultants)

<table>
<tr><th>Waterbody Type</th><th>FIB</th><th>Example TMDL</th><th>Summer-Dry</th><th>Winter-Dry</th><th>Wet</th></tr>
<tr><td>Streams</td><td>E. coli</td><td>Malibu Creek</td><td colspan="2">1.6%</td><td>19%</td></tr>
<tr><td>Beaches</td><td rowspan="2">Enterococcus, Total Coliform (TC), Fecal Coliform (FC), & TC/FC ratio[b]</td><td>Santa Monica Bay Beaches</td><td>0%[a]</td><td>10%</td><td>22%</td></tr>
<tr><td>Estuaries</td><td>Santa Clara River</td><td>5%</td><td>13%</td><td>30%</td></tr>
</table>

[a]Actual reference beach average exceedance rate is 9%; however, the TMDL set the allowed exceedance rate to 0% for the high use period at a specific basin (AB411).
[b]Indicators used in California Ocean Plan.

Although the focus of this report is urban areas, there are also agricultural studies that have been conducted to compare runoff from land under various agricultural conditions against natural land use. As one example, Harmel et al. (2010) studied the effects of agricultural management, land use and watershed scale on *E. coli* concentrations in runoff and streamflow in rural watersheds in Texas. The study found no significant differences in *E. coli* concentrations in “impacted” and “unimpacted” rural streams. In another study in Riesel, Texas, Harmel et al. (2013) also found that mean and median *E. coli* concentrations generally occurred in the following order: cultivated < hayed pasture < native prairie < mixed agricultural land use < grazed pasture. The median *E. coli* concentration for native prairie was 2,000 cfu/100 mL for 22 storm events. The increase in *E. coli* runoff from native prairie relative the hayed pasture was expected to be due to a more abundant wildlife population resulting from the diverse vegetation and habitat on the native prairie. Both studies concluded the likelihood of substantial inputs of FIB by wildlife should be carefully considered when drawing conclusions regarding management options and when evaluating the contribution of agricultural practices to FIB impairments.

3.8 Source Prioritization Process: San Diego County Case Study

Given the many sources of FIB in urban areas and some of the challenges associated with definitively determining sources, it is helpful to develop a source prioritization process. San Diego County developed a formal source prioritization process that provides a framework potentially adaptable to other communities. The process was developed by a work group of San Diego County MS4 co-permittees and their consultants in 2011-2012 (Armand Ruby Consulting 2011) and used to target source control efforts in multiple watersheds across the county.[7] The source prioritization process evolved from work group meetings that initially focused on developing conceptual models for bacteria sources, fate and transport, along with a literature review. Based on the conceptual models and the literature review results, the work group focused on developing a process for prioritizing bacteria sources within watersheds. As a starting point, the conceptual models recognized two overarching, categorical distinctions:

- Wet weather vs. dry weather conditions
- Watersheds (including MS4s, creek and river systems) vs. lagoons (including beaches)

Second, the work group recognized that bacteria sources should be identified by their relationship to human activity and established the following broad categories for bacteria sources:

- Human origin (i.e., from the human body)
- Anthropogenic, non-human origin (resulting from human activities, but not the human body)
- Non-anthropogenic origin (independent of human activity)

Building on these initial frameworks, the work groups developed a rating system using a spreadsheet tool to prioritize efforts. In its initial meetings, the work group produced a lengthy list of potential bacteria sources (similar to Table 3-1), which was used to inform construction of the conceptual model diagrams. The source list was sub-divided into the three main source type categories (human, anthropogenic non-human, non-anthropogenic). Only sources with a potential pathway into an MS4 or a receiving water (creek, river, lagoon, ocean) were allowed on the list. The potential sources were further aggregated according to common characteristics. The draft

[7] Portions of the discussion in Section 3.8 are quoted directly from Source Prioritization Process prepared by Armand Ruby Consulting (2011) for the San Diego County MS4 co-permittees.

lists of sources were then incorporated into the conceptual model diagrams. To support the goal of reducing discharges of pollutants in urban runoff to the maximum extent practicable (MEP), the work group agreed it was important to prioritize sources for further investigation regarding possible application of BMPs (either source controls or local/regional treatment controls).

The work group agreed that prioritization criteria ought to include additional factors other than simply magnitude alone. Temporal variation was identified as a top-level consideration and led to a decision that the prioritization process would be performed separately for dry weather and wet weather sources. Table 3-5 lists factors considered in the source prioritization process, aggregated under the following general themes:

- Human Health Risk
- Magnitude (of loading)
- Geographical Distribution (relative to recreational use locations)
- Controllability/Implementability
- Frequency (of exceedances)

From this exercise, a quantitative ranking scheme was developed for the relative scoring and ranking of sources within a given watershed. The five themes listed above were identified as the factors that would be used in the scoring matrix that was developed into a spreadsheet tool, with example output provided in Table 3-6. Human health risk and magnitude were identified as the most important of the five thematic factors for bacteria source prioritization. Within the scoring scheme, these two factors were given the highest weight, with possible score ranges of 1-10. The other three factors (geographical distribution, controllability, and frequency) were allocated possible score ranges of 1-5. Because of the primary importance of the source type (human, anthropogenic non-human, non-anthropogenic), this factor was given the role of then providing an overall weighting for the source score. The weighting factors for this tool were:

- x 5 for human sources (bacteria derived from the human body)
- x 3 for anthropogenic (resulting from human activity), non-human sources
- x 1 for non-anthropogenic (natural) sources
- x 0 for sources with no apparent transport mechanism from source to MS4 or receiving waters

Table 3-5. Factors Considered in a Source Prioritization Process
(Source: San Diego Co-permittees, as summarized by Armand Ruby Consulting 2011)

SOURCE CATEGORIES TEMPORAL
Temporal Distribution of sources: wet weather vs. dry weather
PRIORITIZATION CRITERIA HUMAN HEALTH RISK
Potential for human pathogens to be present
Potential for human exposure
Dose
MAGNITUDE
Concentration and/or loading
Frequency of occurrence
Variability
GEOGRAPHICAL
Spatial distribution of sources; discrete locations (can map location) or spread out or distributed (e.g., pet waste, soil)
Proximity to REC-1 Uses (beaches)
Proximity to MS4 impermeable surfaces
Land uses, hydrology, soil types, population (design parameters)
Redevelopment opportunities
Ease of transport pathway to receiving waters
CONTROLLABILITY/IMPLEMENTABILITY
Cost, social impact, technological barriers, organizational barriers
Challenge of changing behavior/culturally
How many application sites for BMPs Repetitive nature of behavioral changes
POTENTIAL BENEFITS
Ability to maximize human health improvement
Potential for multiple (secondary/additional) benefits
Other water quality issues
Other benefits (e.g., flood control)
Ability to target underlying water quality issues
Consideration of the benefits of source activities (e.g., flood control)
TECHNICAL/DESIGN
Structural: siting, costs, maintenance
Site-specific flow conditions
POTW capacity for diversions
ORGANIZATIONAL
Regulatory imperative
Code barriers, conflicts w/state-federal regulations
Political opposition/pushback; public support/lack
Organizational ease of implementation
Benefit to public (per cost)
FREQUENCY

Table 3-6. Example Ranking of Weighted Scores for FIB Sources under Dry Weather Conditions Using San Diego Spreadsheet Tool as Applied to the San Diego River
(Source: San Diego Co-permittees, as summarized by Armand Ruby Consulting 2011)

Rank	Human Waste	Dry Score		Rank	Anthropogenic Non-human *(continued)*	Dry Score
1	Sanitary sewer overflows (SSOs)	105		10	MS4s Infrastructure - Biofilm/Regrowth	33
2	Homeless Encampments	105		11	Reclaimed Water	30
3	Leaky Sewer Pipes (Exfiltration)	100		12	Green Waste	27
4	Bathers	95		13	Litter	27
5	Boaters	95		14	Outdoor Dining/ Fast Food	27
6	RVs (mobile)	85		15	Grease Bins	24
7	Porta-Potties	80		16	Soil	18
8	Dumpsters	64		17	Livestock	0
9	Trash cans	64		18	Manure Re-use Non-Ag	0
10	Garbage trucks	60		19	Landfills	0
11	Illegal Dumping	56		20	Livestock	0
12	Leaky or Failing Septic Systems	55		21	Manure Re-use	0
13	Illicit Connections	55		22	Irrigation Tailwater	0
14	Illegal Discharges	40		23	Soil and Decaying Plant Matter	0
15	Gray Water Discharges	40		24	Food Processing	0
16	Pools	36		25	Bio-Tech Manure Management	0
17	Hot Tubs	36			**Non-anthropogenic**	
18	Biosolids Re-use	0		1	Wildlife (Birds and Others)	18
19	Landfills	0		2	Wrackline (Flies, Decaying Plants)	18
	Anthropogenic Non-human			3	Plants	16
1	Pets	72		4	Algae	16
2	Rodents (Mice, Rats), Rabbits, etc.	54		5	Soil	9
3	Birds (Gulls, Pigeons, etc.)	54				
4	Garbage Trucks	42				
5	Dumpsters	36				
6	Trash Cans	36				
7	Manure/Compost	33				
8	Vectors	33				
9	Washwater	33				

3.9 Conclusion

Sources of FIB in urban watersheds range from controllable human sources to naturally occurring sources such as wildlife. Understanding potential sources of FIB and prioritizing primary sources of FIB that can be managed in a watershed is a fundamental step to identifying and implementing control measures to reduce these sources. Once sources of FIB are reasonably understood, steps can begin to be taken to reduce controllable sources of FIB, focusing first on human sanitary sources under dry weather conditions. Based on studies conducted in undeveloped watersheds, it is also likely that some sources of FIB in urban watersheds will be uncontrollable and that some exceedances of RWQC will remain.

4 PREDICTING TRANSPORT AND FATE

A number of common tasks associated with FIB assessment (e.g., source determination, evaluation of treatment alternatives, risk estimation, and model selection) require an understanding of FIB and/or pathogen transport and fate in the environment. Predicting microorganism transport and fate is a highly complex topic, and an exhaustive discussion of the complexities of FIB modeling is beyond the intended scope of this report. However, some useful basic information is provided, related to 1) environmental conditions that affect microorganism survival, 2) transport mechanisms for microorganisms in the environment, and 3) issues which arise when modeling FIB. The intent is to communicate the underlying complexities particular to FIB analysis and to highlight some of the limitations of the current state of practice in predicting FIB behavior.

4.1 Microbial Communities

FIB and pathogens differ from chemical constituents in that they are living organisms that are affected by microbial interactions such as predation and competition. FIB exist as communities of living organisms interacting in a micro-scale ecosystem. The fact that they are living communities increases the complexity involved in attempting to fully describe FIB behavior, since a comprehensive model would reflect the interactions of a variety of living species, each affected by a different set of environmental stressors, including competitor, predatory, or prey species as well as physical/chemical factors. Changes in FIB populations therefore reflect the net result of many concurrent coupled processes, rather than a single causal factor. In principle, the predator/prey or ecosystem models used in other branches of applied biology might be used to describe FIB systems. The problem is that the large numbers of factors, the paucity of data, and the variability of systems from site to site make it unlikely that direct representation of the underlying microbial behaviors will be possible until the state of the art and practice in this area improve substantially. That being the case, the current state of practice requires the use of simplified representations of bulk trends in microbial behavior, the use of careful calibration, and explicit recognition of uncertainty.

The sections below, as they discuss some of the physical factors affecting FIB persistence, should be read with the understanding that they reflect attempts to understand cause and effect relationships between FIB and stressors in global, simplified ways, and that the reality transcends by far what is fully quantifiable at this time.

4.2 Factors Commonly Influencing Microorganism Survival in the Environment[8]

FIB and pathogens may persist in the environment for extended periods of time (outside of a warm-blooded host) in sediments, biofilms, and organic litter in streams, lakes, industrial ponds, and stormwater facilities (e.g., Byappanahalli et al. 2003b, Byappanahalli et al. 2006, Davies et al. 1995, Monroe 2009, Whitman et al. 2003, Kolb and Roberts 2009, Skinner et al. 2010,

[8] The discussion in this section is adapted from previous reports prepared for the International Stormwater BMP Database (Wright Water Engineers and Geosyntec Consultants 2010).

Coghlan 1996, Costerton et al. 1995, Donlan and Costerton 2002, Donlan 2002). The primary characteristics and conditions expected to influence FIB persistence in the environment (and affect treatability in stormwater BMPs) include:

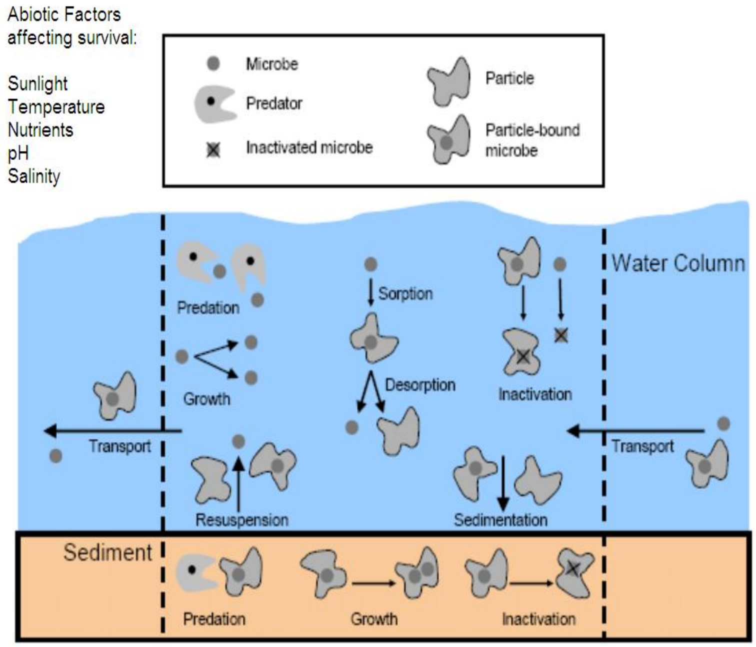
- fluid transport and mixing (discussed in Section 4.4)
- sunlight (solar irradiation)
- temperature
- turbidity
- particle association/partitioning
- nutrient availability
- deposition and suspension
- pH
- salinity
- microbial community (predators, competitors, discussed in Section 4.1)

It should be noted that these factors are generally interdependent. For example, flow affects turbidity via sediment transport, and turbidity affects the efficiency of sunlight penetration, which in turn affects die-off; thus, the effects of sunlight, flow and turbidity can be interrelated. These factors and some of their relationships are discussed further below, with fluid transport and mixing discussed in Section 4.4.

Figure 4-1 illustrates some of the ways that these factors affect the survival, fate and transport of microorganisms in an open waterbody.

Figure 4-1. Potential Fate and Factors that Impact Fate of Microorganisms in Waterbodies and Associated Sediment

(Source: Olivieri et al., in WERF 2007, abiotic notes provided by S.E. Clark)



4.2.1 Sunlight (Solar Irradiation)

Sunlight accelerates the inactivation of FIB transported in water. Studies have shown that sunlight is consistently associated with a decrease in FIB (WERF 2007). Davies and Evison (1991) evaluated the impact of the light source and salinity on FIB survival and showed that the UV component of natural sunlight can impact the survivability in small mesocosms. Although it is generally conceded that sunlight has an effect on inactivation, some studies have indicated that inactivation caused by sunlight may not be permanent, and some bacteria may be able to repair cell damage and regain colony forming potential when no longer exposed to sunlight (WERF 2007).

The degree of exposure affects the degree and rate of FIB inactivation by sunlight. If the fluid is highly turbid, sunlight does not penetrate as well and is therefore less significant in removal. Similarly, if the fluid does not mix well deeper layers will be affected less because light does not penetrate water perfectly. Clumping or association with particulate material can also cause shading that reduces exposure to sunlight. Turbidity is significant enough as a determinant of removal by sunlight that turbidity may be a potential surrogate for determining the effectiveness of sunlight treatment of FIB and pathogens (Tang et al. 2011).

Solar radiation can also indirectly contribute to the inactivation of FIB, since it affects waterbody temperature, which in turn has an effect on microorganism survivability (Joyce et al. 1996, McGuigan et al. 1998) and on growth rates.

The chemical composition of a waterbody can also affect the ability of sunlight to inactivate FIB. For example, sunlight appears to have a greater impact on survival in seawater versus freshwater, likely because the FIB, especially *E. coli*, are under osmotic stress in a saline environment (Fujioka et al. 1981, Korajkic et al. 2013). These seawater studies have shown that different species appear to have different resilience to solar radiation when already stressed. For example, Fujioka et al. (1981) showed that a 90% reduction in fecal coliforms was achieved in 30-90 minutes, whereas it took twice as long to achieve a 90% reduction for fecal streptococci.

Previous studies, particularly those investigating the potential for tropical/equatorial sunlight to disinfect drinking water sources, have shown that sunlight can be an effective and inexpensive disinfectant for FIB (McGuigan et al. 1999). Treatment plants have long used UV light as a disinfectant, so it seems reasonable to suggest that sunlight is a mechanism that will have an impact on FIB removal in suitable BMPs. However, the details of the facility are important in determining the effectiveness of sunlight as an inactivation mechanism. For example, Korajkic et al. (2013) showed no significant impact of sunlight exposure in some freshwater ponds on the survivability of *E. coli*. To be effective, stormwater BMPs that rely, at least partly, on solar irradiation as a treatment mechanism must address factors such as depth of ponding (shallower is better because the natural UV light penetration decreases quickly with water depth), retention time (longer is better), turbidity of water (lower is better), and shading of the water surface (less is better). Mixing of a pond can help to expose more water to sunlight and aerate the pond; however, if flow conditions are too turbulent, resuspension of sediment may occur and increased turbidity may hinder penetration of sunlight through the water column.

4.2.2 Temperature

Most pathogens and FIB are mesophiles, meaning that they prefer warm temperatures (e.g., the temperature of the mammalian gut) with growth possible in the range of 10-50^{o}C and with ideal temperatures in the 20-45^{o}C range. Temperature is commonly identified as a key factor regulating both bacteria growth and die-off rates (WERF 2007, Struck et al. 2006), with sunlight disinfection studies noting that the water temperature had to be raised above 45^{o}C for disinfection/inactivation to occur (McGuigan et al. 1998). In natural water systems, however, temperature-related die-off rate research and seasonal observations of FIB in environmental receiving waters have somewhat contradictory findings.

Research has shown that warmer water temperatures result in faster inactivation of bacteria because warmer temperatures cause faster metabolism and earlier natural inactivation, as well as increased activity (i.e., appetite) of predatory microorganisms. Colder temperatures tend to "preserve" the vitality of bacteria by slowing metabolic processes (Wang and Doyle 1998). In a meta-analysis of 170 datasets on *E. coli* inactivation in the absence of sunlight, Blaustein et al. (2013) evaluated the impact of temperature on inactivation rates for lakes and reservoirs, rivers, and coastal waters. Inactivation rates increased as a function of temperature and could be predicted. Blaustein et al. also noted that part of the impact of temperature was due to the influence of temperature on the other factors that influence inactivation (predator activity, toxic algal products, pH, dissolved oxygen, etc.). In other research, Solic and Krstulovic (1992) found that the time required for a 90% reduction in fecal coliforms decreased by 55 percent for each increase of 10^{o}C. Thomas et al. (1999) observed similar trends for *Campylobacter jejuni*.

Contrary to the findings above, higher bacteria concentrations in natural waters have been correlated with higher water temperatures in the summer and fall. Kadlec and Wallace (2009) noted that bacterial regrowth is fostered by high concentrations of organic matter and by elevated temperatures. Hathaway et al. (2010) also noted that, in North Carolina and other parts of the country, FIB concentrations in surface waters are higher during warmer seasons (Borst and Selvakumar 2003, McCarthy 2008, Young and Thackston 1999, Line et al. 2008, Schoonover and Lockaby 2006). Similarly, bacteria have been found to be significantly lower in snowmelt when compared with warm-weather-rainfall runoff (Clark et al. 2010). Pitt and McLean (1986) found that fecal coliforms, fecal streptococci, and *Pseudomonas aeruginosa* populations were significantly lower (by about tenfold) in snowmelt than in warm weather runoff in Toronto.

Hathaway et al. (2010) concluded that temperature likely acts as a surrogate for seasonal variations and interactions among multiple factors such as moisture and temperature. Hathaway et al. (2010) and others (McCarthy et al. 2008, Crane and Moore 1986, Tiefenthaler et al. 2009) suggest that possible explanations for the increase in FIB concentrations with increased temperatures, even though other studies have shown increased inactivation with increasing temperatures, may include:

1) increased sources of FIB during warm weather due to domestic and wild animal activity, and

2) increased FIB persistence due to seasonal variations in environmental conditions such as temperature, humidity, and rainfall patterns.

4.2.3 Turbidity, Partitioning and Particle Association

Turbidity is a measure of the ability of water to transmit light. As discussed above, turbidity and the associated colloids in water affect the amount of sunlight passing through water, which can reduce the effectiveness of UV radiation in inactivating FIB. However, the particles causing turbidity can also affect FIB inactivation or removal in other ways, as well.

The solids in water can provide a surface for microbial attachment, which may protect the bacteria from harsh environmental conditions and predators, and also act as carriers of attached bacteria to the sediment. Estimates of partitioning and particle association for microorganisms vary greatly between studies, with the fraction that is particle-associated increasing as the suspended solids concentration/turbidity increases. Since bacteria are generally negatively charged, particulates with positive charges on all or part of their surface tend to attract and retain microorganisms; however, bacteria-particulate bonds may be rather weak (Borst and Selvakumar 2003). With regard to bacteria association with specific particle sizes, only a limited number of studies exist (Charaklis and Camper 2009) and their results are not consistent enough to predict particle size associations. As an example, Jeng et al. (2005) found that between 63% and 88% of fecal coliform bacteria in stormwater exist as free-floating in the water column and not associated with suspended sediment. Characklis et al. (2005) reported that the fraction of organisms associated with settleable particles varied by microbe type and flow condition (wet vs. dry weather). For FIB, they found that an average of 20-35% of organisms associated with particles during dry weather and 30-55% associated with particles during runoff events.

Krometis et al. (2007) also reported that partitioning behavior varied across microorganism type, with an average of 40% of FIB associating with settleable particles, whereas 65% of *Clostridium perfringens* spores associated with particles, and only 13% of coliphage associated with particles.

4.2.4 Nutrients

Nutrients in water may affect survival of bacteria. Researchers hypothesize that one reason particulate-bound bacteria survive when compared to free-floating bacteria is due in part to nutrients on particle surfaces. However, the results of recent studies vary regarding the expected role that nutrients play in bacteria survival. For example, Line et al. (2008) showed no correlation between fecal coliform concentrations and nitrate-nitrogen or ammonia-nitrogen in three watersheds in North Carolina. Conversely, McCarthy (2008) showed positive correlations between ammonia-nitrogen and *E. coli* for three out of four watersheds monitored in Melbourne, Australia. In California, Surbeck et al. (2010) found that FIB concentrations were strongly positively correlated with dissolved organic carbon concentration in runoff, and microcosm studies showed that the survival of *E. coli* and enterococci in runoff were strongly dependent on the concentration of both dissolved organic carbon and phosphorus.

4.2.5 pH

Low and high pH are believed to decrease the survival of bacteria. While little research has been conducted on the effect of pH on survivability of stormwater pathogens, one study noted that bacteria thrived near neutral pH (Solic and Krstulovic 1992, from WERF 2007). Wastewater literature states that most bacteria cannot tolerate pH levels above 9.5 or below 4.0, and thrive between 6.5 and 7.5 (Metcalf and Eddy 2003). Therefore, under typical ambient conditions, pH is not a major factor. However, stormwater treatment media having low pH values (such as media having substantial fractions of peat) result in large "removals" of bacteria compared to other materials. Besides the strong sorption properties of peat, the low pH may also affect the effluent bacterial populations (Clark and Pitt 1999).

4.2.6 Salinity

Salinity can affect the survival of bacteria. While this may be a more significant factor in coastal environments, it may also be a factor to consider in streams affected by groundwater inflows that are highly saline and in treatment devices that process snowmelt and salt-laden runoff. Additionally, certain FIB such as *E. coli* lyse in saltwater, which is why *E. coli* is not a recommended fecal indicator for marine water.

4.3 FIB Die-off Rates

As discussed above, many factors influence the "die-off"[9] or decay rates of microorganisms in the environment; however, bacteria die-off typically has been represented as a simple first order

[9] "Die-off" of FIB and pathogens is itself a complex topic since FIB may die, be removed from the water column by sedimentation, or be "inactivated," yet not "dead."

(or pseudo first order) decay relationship (Thomann and Mueller 1987), which can be represented as:

$$\frac{dN}{dt} = -K_b N$$

Where

N = concentration of the organism (typically #/100 mL)

K_b = decay coefficient (usually in 1/day)

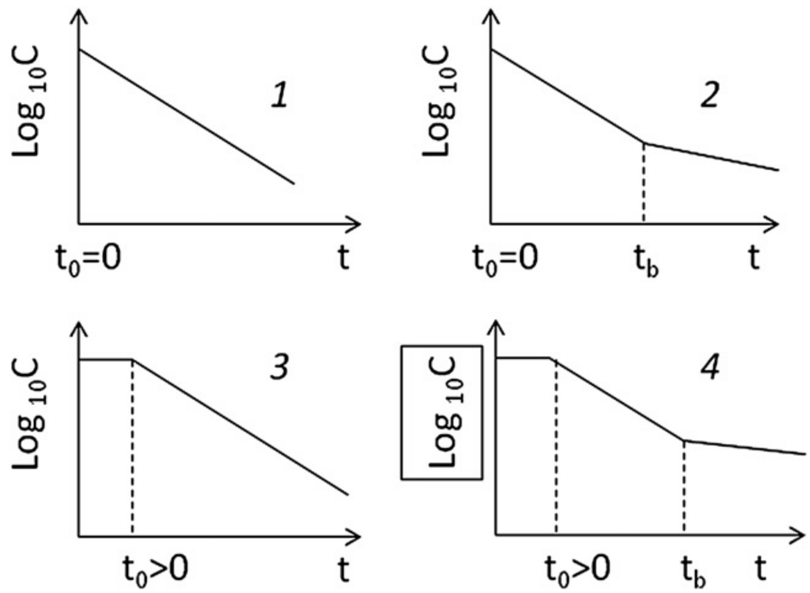
t = time (usually in days)

There are many limitations of this simple conceptualization. Actual behavior of microorganisms is more complicated, based on environmental characteristics and receiving water conditions, with variations of the basic die-off relationship resulting. These variations are one of the reasons that FIB modeling is so challenging. Decay coefficients (K_b) reported in the literature vary substantially from site to site (e.g., EPA [2002] provides values from 0.049/day to 2.0/day for *E. coli*).

Based on a review of 170 FIB data sets, Blaustein et al. (2013) identified generally four different ways that the dependencies of logarithms of *E. coli* concentrations versus time (t) were shaped. Figure 4-2 provides schematics representing these general patterns, which can be represented as first order processes with time-dependent reduction constants. Blaustein describes the four types as follows:

- Type 1 refers to data that are approximately linear throughout the whole range of observation times.
- Type 2 refers to data exhibiting a fast decrease in population until the break time, t_b, after which the slope drastically decreases or becomes close to zero. The data collected after t_b is referred to as the "tail."
- Type 3 refers to data exhibiting an approximately linear dependence of Log C on time after some t_0 substantially greater than 0. The term "shoulder" describes the part of the dataset between experiment start time and t_0.
- Type 4 refers to data with a combination of "shoulder" and "tail" characteristics. The first-order inactivation rate constants were calculated from all data for datasets of Type 1, from data between start time and t_b for the datasets of Type 2, from data after t_0 for Type 3, and for data between t_0 and t_b for Type 4.

Figure 4-2. Patterns Identified by Blaustein et al. (2013) in Data on *E. coli* Inactivation in Waters



As another example of a recent die-off study, Wilson and Pitt (2010, 2011) studied the effects of environmental conditions on die-off of FIB on concrete surfaces. In controlled laboratory tests, Wilson explored the impact of individual and combinations of environmental factors (temperature, moisture, UV light) on dog feces slurry FIB die-off on small concrete blocks. Nearly all of the individual treatments resulted in rapid short-term die-off, followed by reduced decay rates (and, in some cases, regrowth) of the bacteria on the concrete blocks. Figure 4-3 summarizes the results of the factorial experiment for *E. coli* die-off on concrete. Except for the Warm/Wet/Dark conditions, all other combinations of conditions resulted in an initial rapid die-off of the bacteria, with first-order decay rates that were similar to those usually applied for fecal coliform. However, after this initial one or two day period, the die-off rates substantially decreased. The samples subjected to the optimal conditions for survival (warm temperature, high humidity, and no UV light = Warm/Wet/Dark) did not show this two-step die-off and had a more moderate loss rate overall that approximated the long-term rate shown for the other conditions. The model derived parameters applied to these experimental conditions appropriate for use in numeric modeling are presented in Table 4-1.

Figure 4-3. E. coli die-off Results for Household Pet Fecal Sources on Concrete Substrates
(Source: Wilson and Pitt 2011)

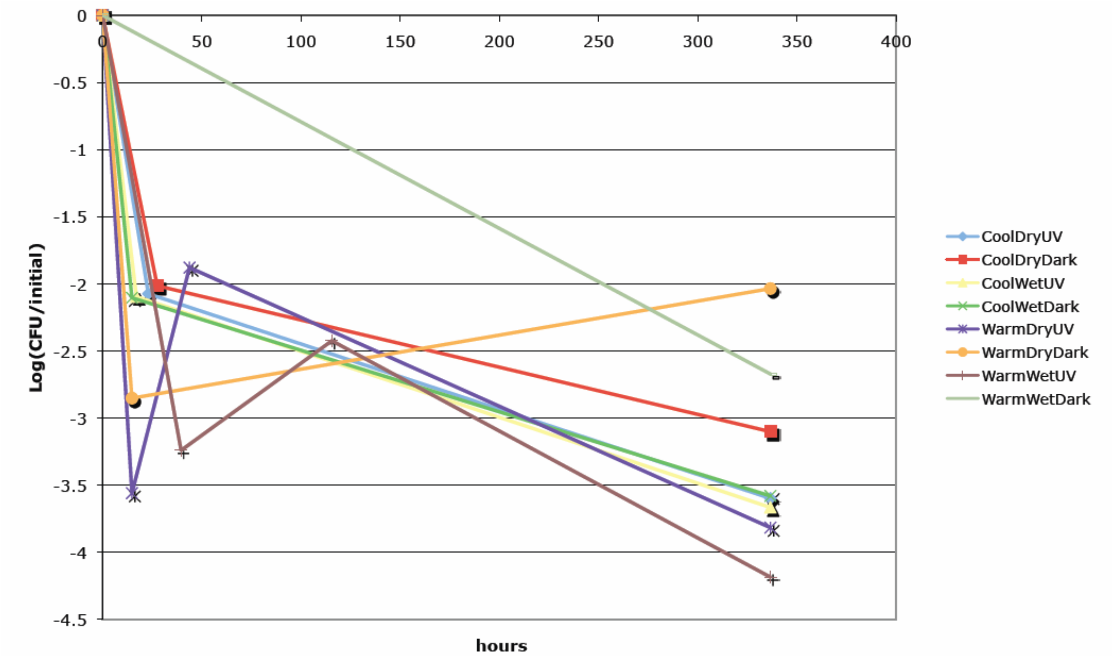


Table 4-1. *E. coli* Modeled Parameters, Applied to Experimental Conditions
(Source: Wilson and Pitt 2011)

	k1 (1/hours)	BP1 (hours)	k2 (1/hours)	BP2 (hours)	k3 (1/hours)
CoolDryUV	-0.109	21.6	0.00221	76.8	-0.00501
CoolDryDark	-0.109	22.1	0.00221	79.0	-0.00501
CoolWetUV	-0.107	21.3	0.00221	83.5	-0.00501
CoolWetDark	-0.107	19.4	0.00221	81.2	-0.00501
WarmDryUV	-0.137	20.4	0.00221	71.0	-0.00501
WarmDryDark	-0.137	19.1	0.00221	77.8	-0.00501
WarmWetUV	-0.0787	27.1	0.00221	91.2	-0.00501
WarmWetDark	-0.0787	22.0	0.00221	84.5	-0.00501

In summary, all treatments exhibit an initial lag or die-off, the rate of which depends on the temperature and humidity. Notably, the warm/wet conditions (those most like the enteric habitat, and exerting the least pressure for adaptation) show the lowest initial rate (k1) of decline, but all inoculants had declined from two to three orders of magnitude within a day or so. The duration of the decline appears to be quite variable (19 h to 27 h). The insensitivities of rates k2 and k3 to environmental factors imply that all adaptive mechanisms available to the inoculant population had been implemented prior to (and caused) the first breakpoint (BP1). The two phase behavior subsequent to BP1 could be attributed to waste buildup in these batch systems or to accumulation of UV-generated thymine dimers. Review of the warm treatment behaviors in the original breakpoint analysis suggests that both factors are involved.

Treatment results for enterococci are shown on Figure 4-4. The warm/wet/dark treatment shows no evidence of a breakpoint (or even a lag), along with a slope essentially equal to zero. The clear trend of greater net survival in other warm treatments seen in the *E. coli* analysis is not evident here, and the timing of breakpoints in treatments (where they occur) is less varied than occurred for *E. coli*. When regrowth phases are recognized, none of the treatments show a net decline of more than about one order of magnitude over a two week period. It was also noted that no population is in decline at the end of the study period. The parameters for enterococci population changes for use in numeric modeling are shown in Table 4-2.

Figure 4-4. Enterococci die-off Results for Pet Fecal Sources on Concrete Substrates
(Source: Wilson and Pitt 2011)

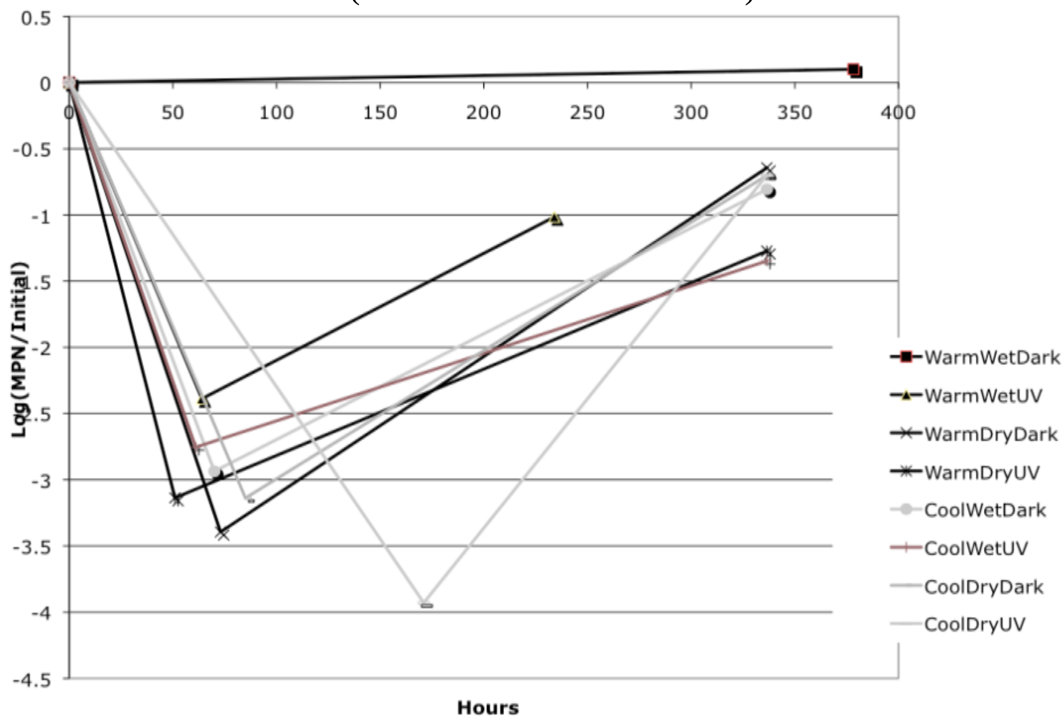


Table 4-2. Enterococci Modeled Parameters, Applied to Experimental Conditions
(Source: Wilson and Pitt 2011)

	k1 (1/hours)	BP (hours)	k2 (1/hours)
CoolDryUV	-0.0501	70.0	0.00652
CoolDryDark	-0.0235	76.7	0.00652
CoolWetUV	-0.0477	66.5	0.00652
CoolWetDark	-0.0211	70.5	0.00652
WarmDryUV	-0.0359	63.2	0.00652
WarmDryDark	-0.0479	70.4	0.00652
WarmWetUV	-0.0233	64.0	0.00652
WarmWetDark	-0.0353	68.6	0.00652

As noted above, all treatments exhibited an initial decline for enterococci, with all three environmental factors (temperature, humidity, and UV exposure) contributing (either as main effects or within interactions). The rates of decline, however, are only about half of those shown by *E. coli*. The adaptation phase of these inoculants lasted about three days before the first breakpoint was observed. Even with the slower rates of decline, most inoculants had been reduced by two or three orders of magnitude in the initial period. The insensitivity of k2 to environmental effects, and the fact that it is positive (indicating net growth) implies that these organisms adapt to impervious environmental surfaces quite well. By the end of the study period (about two weeks) all inoculants had rebounded to within about 10 percent of their original populations.

Wilson also conducted a study of the survival of FIB on pervious environmental surfaces (soil) based on a 2^5 factorial experiment (temperature, humidity, pH, presence/absence of UV-B exposure, and added bioavailable organics). Parallel studies were performed for *E. coli* and enterococci. These tests were a continuation of the bacteria survival studies on impervious surfaces summarized above. Although analyses for this study are still in progress, initial observations include:

- The neutral/no added organics condition showed a similar long-term behavior for all treatments, but with an apparent absence of the initial (first day) rapid die-off.
- Cool/dry conditions were antagonistic to survival, but UV appeared less interactive with the other environmental conditions.
- Overall, the *E. coli* populations in the soil had less rapid changes over time compared to the concrete surfaces.
- Enterococci survival on the soil media exhibited less sensitivity to environmental conditions than *E. coli*, with some treatments showing growth. For the neutral pH/no added organics condition, some samples showed over ten-fold growth during the extended test period.

Examples of other studies evaluating factors affecting die-off include:

- Easton (2000) conducted in-situ field studies of die-off rates in Alabama streams and ponds using equilibrium test chambers holding various mixtures of raw sewage and receiving water. All of the test organisms (*Giardia*, *Cryptosporidium*, and FIB) showed a pattern of leveling off toward an equilibrium population with increasing time, with die-off rates and patterns being organism-specific. Rapid die-off occurred until the carrying capacity of the environment was reached and the organisms were maintained at a level supported by the available nutrients present. An alternative hypothesis was related to “quorum sensing” or genetic programming that enables bacteria to self-regulate their numbers (Easton et al. 1999).
- Hellweger et al. (2009) noted that the decay of fecal bacteria in surface water often follows a biphasic pattern with the apparent first-order rate constant relatively high during a first

phase and lower in a second one. Their study evaluated whether cell density (e.g., quorum sensing) explained this pattern and concluded that the rate constant changes after a certain time, rather than at a certain density, which is inconsistent with a density effect.

- David and Haggard (2011) developed regression based models to predict fecal bacteria numbers at selected sites within the Illinois River Watershed in Arkansas and Oklahoma. The regressions were statistically significant at almost every site for all three bacteria groups. However, the physico-chemical parameters used in the regression equations to explain die-off were very different across sites and fecal bacteria groups.

- Francy et al. (2003) developed predictive models for the distributions and sources of *E. coli* at five Ohio bathing beaches on Lake Erie and one inland lake during 2000 and 2001. The models were shown to be beach specific; that is, different explanatory variables were used to predict the probability of exceeding the standard at each beach. For example, at the three Lake Erie urban beaches, the models included variables such as the number of birds on the beach at the time of sampling, lake-current direction, wave height, turbidity, streamflow of a nearby river, and rainfall. At Mosquito Lake, the model contained the variables rainfall, number of dry days preceding a rainfall, date, wind direction, wind speed, and turbidity.

Key findings associated with this literature are that there is a tendency for removal rates or inactivation to:

1) trend toward a minimum equilibrium concentration,
2) vary over time,
3) vary by species, and
4) vary based on site-specific conditions.

These findings suggest that prediction of FIB behavior requires substantial site-specific data and analysis if unequivocal conclusions regarding die-off rates are to be developed. These findings also raise questions regarding the usefulness of FIB as indicators of pathogens. Given the variability in die-off among species, die-off predictions for FIB may not reflect die-off for pathogens. Hence, if pathogens die-off first and FIB persist, then FIB constitute a false positive that may result in overprotective management decisions. Conversely, if FIB die-off before pathogens, then they may be under protective as an indicator. As has been recognized by others, this is not an ideal situation from a policy and management perspective; however, a better approach has not been developed, leaving FIB as the currently accepted practical surrogate.

4.4 General Transport and Fate Mechanisms

In addition to understanding the factors that affect FIB survival, it is important to understand the transport mechanisms by which FIB reach receiving waters, as well as factors such as sedimentation and resuspension in receiving waters. Pathogens and FIB enter waterbodies from many sources, including shallow overland flow, groundwater discharge, and direct inputs from storm sewers, animals and humans. High concentrations during storms result both from inputs via pipes and overland flow and from resuspension of pathogens retained in streambed sediments from prior storms (Donnison et al. 2006b, Jamieson et al. 2005a&b, Searcy et al. 2006b, Wilkinson et al. 2006). Various organisms may also have differences in distribution, survival and transport behavior (Characklis et al. 2005a, Donnison et al. 2006b). A discussion of FIB and pathogen transport in surface water, vadose zone and groundwater follows. It is important to recognize that the transport and fate characteristics of pathogens may differ from FIB. The remainder of this section provides a fairly technical discussion of transport and fate processes, with a more simplified discussion of implications for modeling provided in Section 4.5.

4.4.1 Physical Transport and Dispersion in Surface Water

Pathogens and FIB in surface waters are subject to a variety of transport processes, with the transport dynamics modified by interaction with soils and suspended sediments, along with sediment beds in waterbodies. Although simplified models or conceptualizations of fluid transport are often applied, it needs to be understood that they are only conditionally applicable, and that more complex phenomena are commonly encountered. Two examples of situations where simple models may be inadequate are as follows:

- In many models, a discharge to a receiving stream is often assumed to mix across the stream cross section almost immediately, and transport effects are assessed in terms of advection and longitudinal mixing below that point. It is not unusual to find that this conceptualization is only an approximation. For example, a discharge from a storm system into a receiving waterbody may be colder than the ambient condition and therefore dive to the bottom before mixing occurs. Or, a discharge from a BMP may be warmer than the water to which it is discharged, and tend to stay near the surface in the receiving body. Sometimes, the lack of mixing between the discharge and the waterbody can persist over significant distances or time periods. During the period where transverse mixing is incomplete, models or tools assuming complete transverse mixing may provide poor approximations of FIB transport in the near field until the temperature of the discharged water cools or warms and more complete mixing can occur.

- Stratification within a receiving waterbody can compartmentalize some parts of a fluid volume, and limit transport to a fraction of the apparent volume of that body. Lower strata in a reservoir, for example, can have completely different physical/chemical characteristics than the surface layers. Models assuming complete mixing across a transverse section may be inappropriate for use in such a situation.

Other examples of these kinds of complexity can be cited, and it is important to understand that they are not uncommon. Those responsible for modeling FIB in practical situations will need sufficient grounding in fluid dynamics and modeling to determine the most appropriate tool and technique for analysis. Despite these complexities, basic advection/dispersion models may be useful for modeling FIB transport, with key transport processes discussed further below.

In streams, the dominant fluid motion term is usually advection. Rapid advection implies short transit times between points and therefore less time for inactivation or removal. Further, rapid advection implies greater shear along a fluid boundary (e.g., a river bottom) and a tendency towards resuspension. Wilkinson et al. (1994) found that the entrainment and deposition of fecal coliforms in streams and rivers appears to be governed by the relationship between flow and the channel bed. Their resultant model assumes that fecal coliforms are associated with low density particles that are entrained when the flow rises and deposited when the flow recedes.

In relatively static fluid bodies (e.g., an extended detention pond or other small body of water) advection may become a small term compared to mixing that occurs through turbulent dispersion or diffusion. In such a situation, a river model may be inappropriate, and a reactor model may be more appropriate. An example of a model of this type is the case where the impoundment is represented as a well-mixed reactor, and an analytic solution of FIB removal is applied.

In theory, mixing can have both positive and negative impacts on bacterial concentrations in the water. Greater mixing associated with greater turbulence suggests less removal by sedimentation, and sufficient turbulence may imply a tendency towards resuspension. At the same time, greater mixing offers the potential for greater exposure to sunlight as water and FIB from deeper in the waterbody move closer to the surface. Increased mixing may increase effective irradiation, albeit for a shorter period of time, and eventually increase the reduction in FIB. The governing terms and results of these positive and negative factors will depend on the particular conditions of the system of interest.

Most receiving water flows are turbulent, which increases the distribution of microorganisms within the waterbody. As a result, it is common to find models that assume complete mixing of pathogens and other fine suspended matter through the water column in receiving streams. However, recent research has shown that a variety of processes favor pathogen deposition in small streams and treatment devices. Pathogens are known to readily associate with soils, fecal and wastewater solids, and other natural sedimentary material (Dai and Boll 2003, Jamieson et al. 2005a&b, Medema et al. 1998, Searcy et al. 2005), which increases the rate of pathogen removal from the water column by settling (Characklis et al. 2005b, Jamieson et al. 2005a, Medema et al. 1998, Searcy et al. 2006b). This lends credibility to models incorporating sediment association as a removal mechanism. Additionally, recent research shows that hydrodynamic surface-subsurface exchange carries pathogens into the shallow subsurface (streambed, hyporheic zone, etc.), where they can be removed by filtration processes (Searcy et al. 2006b), offering another way of incorporating removal in an FIB fate and transport model. Biofilms also play an important role in the retention of pathogens (Searcy et al. 2006a, Stott et al. 2007), and has the potential to enable a wider range of modeled removal or transport phenomena.

Resuspension is strongly dependent on the bacteria associations with particulates and the shear stresses applied to the exposed sediment. Matson et al. (1978), studying river and lake sediments in Connecticut, found that resuspended sediments in shallow waters, due to factors such as elevated flow rates, wind, and human activity, can elevate the water column bacteria concentrations significantly. Davis (1979) stated that bacteria contamination of waterways during and following storm events is a function of the stream sediment bacteria concentrations, the concentrations of bacteria in soils adjacent to the stream (and source areas in an urban watershed), and the stream velocities. There is ample evidence that sediments can contain substantial bacterial concentrations (Davis 1979, Pitt and Bozeman 1979, Geldreich et al. 1980), so this effect can be considered an established contributor to FIB in the water column associated with sediment disturbances.

Many of these deposition mechanisms can readily be parameterized for modeling purposes. Recently, a variety of relatively simple model formulations have been employed to assess the effects of microorganism association with background sediments, sedimentation, and subsurface filtration on pathogen transport in streams and rivers (e.g., Bai and Lung 2005, Dorner et al. 2006, Jamieson et al. 2005b, Searcy et al. 2006b). While these studies indicate a clear need to include these processes in field-scale models for transmission of zoonotic pathogens, very little data are available to evaluate the necessary modeling parameters, such as degree of association with different sediment types and sizes and the resulting settling velocity distributions. Further, the available data are very sporadic and normally focus on only one or two microorganisms.

Many of these processes have been isolated in laboratory experiments; however, there is little data that shows how these processes interact with and affect pathogen transport in surface water systems. The limited field studies conducted to date suggest that association with suspended and bed sediments play an important role in the overall migration of zoonotic pathogens such as *Cryptosporidium* and *Campylobacter spp*., as well as FIB such as enterococci and coliforms, in surface waters (e.g., Characklis et al. 2005b, Davies and Bavor 2000, Donnison et al. 2006a, Jamieson et al. 2005a&b, Kadlec and Knight 1996, Muirhead et al. 2004, Wilkinson et al. 1995, Wilkinson et al. 2006). Association with sediments also influences pathogen survival (Burton et al. 1987, Davies and Bavor 2000, Davies et al. 1995).

In summary, although fluid transport processes are generally understood in concept, and some factors are commonly incorporated into models, there are some additional complexities related to physically based removal and transport processes in surface water for microorganisms that are still not well understood, particularly with regard to particle associations. Additional research and technical development is needed in this area to provide a more comprehensive state of practice for modeling.

4.4.2 Subsurface Transport

Subsurface transport of FIB and pathogens includes movement in the vadose zone, groundwater, and soils. These transport phenomena are discussed below.

4.4.3 Transport in the Vadose Zone

The vadose, or unsaturated, zone is a near-surface dynamic region where the hydrological, chemical and biological processes can occur in the short or long term at the microscopic or macroscopic scale. Pathogen transport in the vadose zone is mostly explained by the pathogen’s size, attachment and adhesion capabilities, sorption, nutrient availability (Bradford et al. 2006, Guber et al. 2005, Hagedorn et al. 1978, Harter et al. 2008, McMurry et al. 1998, Mittal 2004, Powelson and Mills 2001, Unc and Goss 2004), and the presence of predators, all of which have the potential to retard pathogen movement. In addition, properties of the porous media cannot be considered homogeneous and isotropic.

At the soil surface, biological activity (e.g., effects of plants and animals) and environmental factors (e.g., temperature, human activity, humidity, weathering, etc.) can develop fractures and large pore spaces (e.g., biopores) in most soil environments. Rapid transport through soils is typically attributed to macropore flow, especially in soils with significant clay content. The presence of macropores can lead to extensive pathogen inputs to both groundwater and surface waterbodies (Aislabie et al. 2001, Harter et al. 2008, McGechan and Vinten 2004). These large pore spaces and fractures allow water to move (e.g., during rainfall or irrigation events) rapidly from the surface into the soil and transport pathogens and FIB. These organisms, when entrapped in the soil, may need to compete with native microorganisms for nutrients. The top soil is typically rich in ions and nutrients. However, nutrient concentrations typically decrease lower in the soil profile and cyclical aerobic-anaerobic conditions may occur, requiring microorganisms to adapt or die. On the other hand, moving deeper in soils can protect some microorganisms (e.g., less or no UV radiation, less rapid changes in humidity and soil moisture, and few extreme temperatures).

Physical straining as a function of soil pore size and microbial geometry has been viewed as the primary process that retards pathogen transport through the vadose zone (Matthess and Pekdeger 1988, Foppen et al. 2005). Under natural soil conditions, however, many processes affect transport, including biological straining, sorption and preferential flow through macropores and fractures. These macropores develop as a result of nematodes, dead roots, soil aggregation and/or geological processes (e.g., erosion and deposition). In macropores, wetting fronts propagate to significant depths by bypassing matrix pore space (Brusseau et al. 1992, Kladivko et al. 1999, Castiglione et al. 2003), thus rapidly increasing the depth of pathogen penetration in the soil.

Additionally, pathogens are living organisms with attachment capabilities and evolved sensorial mechanisms. These traits allow pathogens to perceive their surroundings and seek the most suitable place (e.g., nutrient availability, other bacteria, biofilms) to attach. Sorption of fecal bacteria in soils had been typically investigated using cultured bacteria suspended in distilled water and found to be proportionally related to the percentage of clay content (Ling et al. 2002). There is a challenge, though, in translating these mono-strain, free cells suspended in inert solutions to field conditions (Guzman et al. 2010a). In addition to dominant soil minerals and clay content, soil organic matter can enhance, suppress and/or decrease sorption of microorganisms as soil organic matter provides additional surface area for sorption, coats clay mineral surfaces and/or favors soil dispersion processes due to changes in pH following manure application, respectively (Guzman et al. 2010).

Garbrecht et al. (2009) demonstrated the importance of flow velocity and soil particle size distribution in *E. coli* transport through soil media with high conductivity plugs, with implications for the design of treatment devices that rely on physical straining for microorganism removal. The introduction of plugs with higher hydraulic conductivities than the surrounding soil increased *E. coli* effluent concentrations and decreased detection times. In their experiments, the plugs acted essentially as preferential flow pathways, allowing for direct transport of *E. coli* through the soil profile. Therefore, these treatment devices should not be designed to maximize drainage, but instead to maximize retention time in the media.

4.4.4 Transport through Soils

Pathogen transport through soils has been primarily viewed as influenced by straining due to pore space size and the bacterium geometry. Matthess and Pekdeger (1988) and Foppen et al. (2005) pointed out the importance of straining as a dominant process and the relationship with the pore size distribution in estimating retention of bacteria. This approach has its roots in the colloid filtration theory (e.g., mechanical straining) in which particles moving through a porous media can be filtered. However, under natural soil conditions, other processes also may occur, including biological straining, sorption and preferential flow. In addition, pathogens are living organisms with attachment capabilities and evolved sensorial mechanisms. These traits allow pathogens to perceive the surrounding and seek the most suitable place (e.g., nutrient availability, other bacteria, biofilms) at the bacteria scale to attach and perhaps adhere.

Sorption isotherms have been proposed and frequently used as a practical means for determining bacteria fate and transport in soils, even though these isotherms cannot theoretically explain the sorption/attachment mechanisms occurring at the soil surface (Pachepsky et al. 2006). Gantzer et al. (2001) used batch soil sorption experiments (specific percentage of clay and organic matter) and observed a nonlinear relationship between free and attached fecal coliforms from wastewater, and suggested both Freundlich and Langmuir isotherms to simulate the equilibrium relationships. Ling et al. (2002) quantified *E. coli* sorption for two different soils (differing clay and organic matter contents) and proposed a linear relationship between the distribution coefficient of the linear Freundlich model, K_d, and the natural logarithm of the clay content (%):

$$K_d\left(\frac{mL}{g}\right) = 10^{-1.6\pm0.9}(\%\ clay)^{1.98\pm0.7}$$

$$K_d\left(\frac{mL}{g}\right) = 10^{-1.6\pm0.9}(\%\ clay)^{1.98\pm0.7}$$

McGechan and Vinten (2003) use K_d = 0.45 mL/g for a sandy loam and 5.0 mL/g for a clay loam for *E. coli* sorption. More complex models have been used at the soil core scale but not deemed practical at larger scales (Pachepsky et al. 2006).

Sorption of fecal bacteria in laboratory studies of soils had been found to be proportionally related to the percentage of clay content (Ling et al. 2002). Under agricultural field conditions, however, increased concentrations of fecal bacteria also are associated with increased amounts of animal waste (initial concentration), which also contain a variety of constituents [nutrients] that can interact with soils. These constituents can result in unique fecal bacteria sorption/attachment mechanisms when compared to isotherms developed using mono-strain, free cells suspended in inert solutions (Guzman et al. 2010a). These variables influence bacteria sorption to soils and many can assist or prevent sorption depending on the environmental conditions. In addition to dominant soil minerals and clay content, soil organic matter can enhance, suppress and/or decrease sorption of *E. coli* as soil organic matter provides additional surface area for sorption, coats clay mineral surfaces and/or favors soil dispersion processes due to changes in pH (Guzman et al. 2010). Barfield et al. (2010) and Hayes et al. (2008) discuss the application pathogen isotherms to sorption in the BRC routine in IDEAL. The sorption process in the IDEAL BRC model is based on the sorption by clays and silt particles. The impact of filtration is considered separately.

Garbrecht et al. (2009) demonstrated the importance of flow velocity and soil particle size distribution in *E. coli* transport through soil media. The introduction of macropores with higher hydraulic conductivities than the surrounding soil increased *E. coli* effluent concentrations and decreased detection times. With macropores, wetting fronts propagate to significant depths by bypassing matrix pore space (Brusseau et al. 1992, Kladivko et al. 1999, Castiglione et al. 2003). Macropores may be subdivided into two major groups based on physical characteristics and origin: natural fractures and cylindrical biopores. Natural fractures originate from soil expansion and contraction or from geological processes. Biopores, on the other hand, are created by tunneling insects, small animals, nematodes and decaying roots (McMahon and Christy 2000). These preferential pathways can interconnect the surface and deeper soils or allow water to move horizontally, transporting solutes and pathogens.

The modeling process for these macropores is difficult. Some progress in this area has been made, but the issues are far from solved. One approach is to assume parallel pathways for flow, one with a high hydraulic conductivity and one which assumes Darcian flow. Preliminary results by Brown (2010) indicate that this research has promise but needs further testing. The preliminary results by Brown were on modeling phosphorus movement and not bacteria, but could hopefully be applied to pathogens. The approach proposed by Brown is used in IDEAL (Hayes et al. 2008, Barfield et al. 2010).

Subsurface drains are becoming integrated into many low-impact development designs in urban areas such as bioretention cells. The influence of macropores increases as soil saturation increases. Therefore, the ability to model the interrelationship between macropore facilitated pathogen transport and subsurface drainage systems, where soil is consistently near saturation, is important for evaluating potential reductions in pathogen removal in media-based stormwater treatment devices. Recent research suggests direct hydrologic connectivity develops between macropores and subsurface drains (Shipitalo and Gibbs 2000, Fox et al. 2004 and 2007). However, only a few research papers have been published on macropore facilitated pathogen transport (Guzman et al. 2010b).

4.4.5 Transport in Groundwater

Fate and transport of pathogens in groundwater is relevant to stormwater management due to the current emphasis on infiltration of stormwater to reduce volume related impacts of runoff. Much of the available research has been conducted in agricultural settings. Groundwater was the source of several documented outbreaks of pathogenic *E. coli* and campylobacteriosis in Canada, the U.S. and Europe (Gallay et al. 2006, Haenninen et al. 2003, Kuusi et al. 2004, Kuusi et al. 2005, Stanley et al. 1998, Unc and Goss 2004). It is often difficult to trace *Campylobacter* spp. and other pathogens in groundwater, as their occurrence appears to be sporadic (Haenninen et al. 2003). Where *Campylobacter* is detected in wells and springs, nearby animal sources are typically identified. Pathogen occurrence in groundwater is limited by its mobility and survival (John and Rose 2005). Transport of microorganisms in groundwater is controlled by the same processes that occur in the vadose zone with the exception of cyclical wetting and drying. In addition, it is common in coastal areas to find that a dynamic balance between freshwater and saline systems exists, which implies differing rates of transport and die-off as a function of the salinity regime being considered. In groundwater, the soil is fully saturated and typically is not aerobic. However, because many pathogens can adapt to lower-oxygen environments, transport of viable organisms is possible, as noted above by the groundwater-associated disease outbreaks. Nonetheless, many studies in the laboratory have shown that microorganisms in groundwater tend to become inactivated over time. In 0.2 m column experiments with alluvial materials, *Salmonella* declined by more than five orders of magnitude over a 12-day time period (Dowd and Pillai 1997). John and Rose (2005) examined numerous studies and found average inactivation rates for coliform bacteria, enterococci, and *Salmonella* to be on the order of 0.07-0.1 $\log_{10}$/day at typical groundwater temperatures. Higher temperatures tend to accelerate inactivation although wide variability is observed.

4.5 Modeling

FIB modeling is challenging and has some substantial limitations due to the many factors affecting bacterial transport and fate, and the fact that these factors vary from site to site and species to species. Therefore, FIB modeling in urban areas should ideally consider bacteria sources, transport and fate on a source-specific basis, instead of applying the same loss rates for all areas. Unfortunately, the availability of data to definitively address even a single source area is usually limited, and data adequate to fully characterize each site in a complex watershed are seldom available. Even if data were not limited, available modeling tools do not typically have the ability to represent all of the phenomena potentially of interest. As a result, modelers are typically faced with a high degree of scatter in observations and limited ability to reproduce the variability that is observed. In theory, sufficient field information and sufficiently detailed models would help to reduce uncertainty, but the costs of data gathering and limitations in the state of the practice suggest that a high degree of uncertainty will be associated with FIB modeling for some time to come.

The net result is that while models can be useful, they must be viewed with caution when it comes to evaluating FIB in the environment. The use of models for the development of TMDLs is common and perhaps adequate for many conventional water quality pollutants. In evaluating factors such as volume reduction as a means of controlling FIB, there may be some confidence in results. In other situations the use of models for FIB can be challenging to the point of having questionable value. This is particularly true given the uncertainty associated with predicting the effectiveness of treatment, (as discussed in Chapter 8), and the problem is further compounded by the significant but often unidentified sources of FIB in the environment. The practitioner is faced with putting uncertain inputs into an uncertain watershed or receiving water model in an uncertain prevailing context, and then attempting to make informed quantitative management decisions on the result.

At the same time, there is a need to consider targets and management requirements for FIB despite these uncertainties. The question is how to go about this. In the EPA (2007a) Expert Scientific Panel report, the modeling task group noted that feedback from some environmental engineers and consultants suggested a high degree of uncertainty in predictions for pathogen and FIB concentrations and fluxes. The full document should be reviewed for a complete understanding of the intent and content of the discussion, but one of their key conclusions was:

> "There is limited understanding regarding the sources of microorganisms and their fate and transport in the aquatic environment, so the use of deterministic, process-based models for criteria development and implementation is not practical for most U.S. water quality managers within the next five years [2012]. Rather, simple heuristic, statistical models that do not necessarily require an understanding of processes and mechanisms are more realistic for criteria development and implementation within the next five years."

While the pragmatic acknowledgement of the limitations in prevailing practice is appropriate, proceeding with tools that "…do not necessarily require an understanding of processes and mechanisms…" may be a preferred option. Case-by-case consideration of the costs of decisions based on inadequate understanding should be a strong determinant of the preferred approach to data gathering, problem determination, and management approaches. There is a need to pursue FIB impairments and address regulatory requirements within the context of technology and current available knowledge, but setting targets or implementing management mechanisms in the absence of adequate knowledge should be tempered by questions regarding the uncertainty of a return on investment.

4.5.1 Model Uncertainty

Understanding the limits of modeling technology and properly accounting for model uncertainty are fundamental to developing a model useful for management decisions. Shirmohammadi et al. (2006) state: "Uncertainty is defined as the estimated amount by which an observed or calculated value may depart from the true value, and it has important policy, regulatory, and management implications." Unfortunately, model results are often reported without recognition of uncertainty. Costly TMDL implementation plans may be implemented as a result of model outputs; therefore, it is important that a phased approach to TMDLs be used that "tests" recommendations of models along the way. Where real-world findings are not consistent with model results, then model outputs need to be reevaluated. Because of the many unknowns associated with FIB modeling, model results should be used cautiously, in terms of general guidelines, rather than as absolutes.

Many published papers address the issue of uncertainty in modeling. As one example, Shirmohammadi et al. (2006) summarize the collective experience of scientists and engineers in the assessment of uncertainty associated with TMDL models. Examples of sources of uncertainty include factors such as input variability, model algorithms, model calibration data and scale. For FIB, all three factors constitute major issues, where inputs are highly variable, significant unknowns exist regarding underlying algorithms, model calibration data is often lacking, and scale issues have been documented (Harmel et al. 2010). For an example of variability in model inputs, Harmel et al. (2010) cites research by McCarthy et al. (2008), who reported that the uncertainty in measured *E. coli* levels introduced by sample storage time averaged ±25% (range ±9% to ±44%), that uncertainty introduced by the Colilert MPN analytical technique averaged ±22% (range ±12% to ±51%), and that the combined uncertainty averaged ±33% (range ±15% to ±67%).

Shirmohammadi et al. (2001, 2006) identified scale-related issues as a source of uncertainty when the scale at which simulated processes are being applied is not consistent with the scale at which they were developed (i.e., plot scale, landscape level, watershed level, etc.). For example, Harmel et al. (2010) found that *E. coli* concentrations consistently decreased as watershed scale increased from field to small watershed to river basin scale. They concluded that there is a need for additional studies that compare *E. coli* fate and transport at multiple watershed scales.

Shirmohammadi et al. (2006) assert that uncertainty in TMDL models is a real issue, requiring explicit quantification, and should be taken into consideration not only during the TMDL assessment phase, but also in the design of BMPs during the TMDL implementation phase. Harmel et al. (2006) recommend that uncertainty inherent in calibration and validation data should also be included in the overall assessment of model uncertainty.

Harmel et al. (2010) provide a concise synopsis of this issue:

> …there remains a large degree of uncertainty in simulating *E. coli* fate and transport, which is due to several factors. First, relatively few *E. coli* data sets are available for model calibration and validation. Data collected from watersheds of varying scales and land uses with different management practices are especially rare, which severely limits the ability of models to predict *E. coli* fate and transport from various sources in response to management alternatives. In addition, the uncertainty in measured *E. coli* data also contributes to the uncertainty in bacterial modeling (Harmel et al. 2006, McCarthy et al. 2008). Second, large variations in reported values for *E. coli* persistence in the environment result largely from a lack of understanding of the fundamental processes controlling fate and transport mechanisms. For example, it is unclear what proportion of *E. coli* cells are transported via surface flow as single cells as opposed to attached to soil particles (Muirhead et al. 2006a&b; Oliver et al. 2007; Mankin et al. 2007; Soupir et al. 2008a, 2010). Similarly, *E. coli* survival kinetics in different environments (Wang et al. 2004; Soupir et al. 2008b), the resuspension of streambed sediment and associated *E. coli* (Rehmann and Soupir 2009), and the potential for establishment of naturalized populations in soils or sediments (Ishii et al. 2006, Jamieson et al. 2004) are not well understood. Despite numerous laboratory and small-scale studies investigating many of these factors, there is a need for additional studies that compare *E. coli* fate and transport at multiple watershed scales.
>
> …increased attention should be given to the basic science of fecal indicator bacteria in the environment. Only with a sound scientific understanding of fundamental processes can the substantial uncertainty associated with bacterial transport assessment and modeling be reduced. Only then can effective and efficient management and regulation of bacterial contamination become a reality.

4.5.2 State of Practice Considerations in FIB Modeling

Despite the above questions related to model capability, state of practice, uncertainty, and other issues raised above, FIB models are needed and will likely continued to be used, despite the known limitations. Until FIB modeling capabilities improve, modelers must recognize current constraints by clearly communicating uncertainty associated with models. Based on the experience of the authors of this report and work by others (e.g., Texas Task Force 2007, Benham et al. 2006), some basic concepts that should be considered when FIB models are needed for TMDL or other purposes have been developed. Although the points below are generally valid, the modeler should consider the specific modeling context before pursuing any particular course of action.

- Models are not simply the technology embodied by a particular software application. A model inherently includes: 1) the chosen software or solution, 2) the data available, 3) the specific nature of the problem, and 4) the skills of the modeler and the way they interpret and represent the system. It is the assembly of these four factors that constitutes a model. In the absence of consideration of all four factors, statements regarding any particular software tool are of limited value. With access to documentation and code, it may be possible to describe the nature of a software tool (scope, algorithms etc.) in isolation from the other factors; however, it is not possible to assess the value of that tool without specific consideration of all of these factors.

- Of these four factors, the most significant aspect underlying the likelihood of eliciting useful results by modeling may be the fourth one, the modelers themselves. Knowing how the data, problem and software can best be knitted together and interpreted in a particular situation is a prerequisite to any successful modeling effort.

- Proprietary modeling software that does not provide accessible source code for computations should generally be avoided unless independent third party review is able to attest to the accuracy and adequacy of documentation. (This comment is focused on the computational engine; availability of code for interfaces, graphics and other elements of a tool which do not determine computational results is not as critical.) A definitive understanding of the computational aspects of a tool is prerequisite to its proper application. It is not necessary that all modelers be able to read and understand the underlying code, although probably desirable; however, it is important that the user has the ability to obtain an unfettered understanding of the strengths and limitations and embedded assumptions inherent in the tool, and it is the source code that is the most definitive way to communicate this.

- A sufficient data set is a critical requirement and often a limiting factor of effective models. While models can apply math and physics and in some cases empiricism to remedy data gaps or other limitations, they cannot represent what is not understood. Without adequate data, calibration, and validation, a model is constrained in value and may be fatally flawed. Given the complexity of the FIB problem and the dependence of FIB parameter estimation on extensive data, this aspect is particularly important for FIB modeling problems.

- Greater complexity does not guarantee improved performance. Depending on context and need, a simple modeling tool may be a preferable option to a complex modeling tool. There is no basis for choosing one over the other *a priori*.

- Generally, a particular software tool for FIB analysis should not be specified for use on a policy basis unless particular conditions demand this be done (e.g., a suitable and useful model already exists and there is simply a need to update its calibration). The modeler, data and context should be considered when a tool is chosen; this consideration is particularly important in a green field situation.

- Modeling techniques and preferred practice must reflect context, and a model suitable for one problem might not be suitable for another. For example, typical approaches to calibration for flood control might reflect an interest in stormwater peak flows. If a tool developed for flood peak prediction is repurposed for FIB assessment, it might be inadequate for assessment of the low flow conditions that often are of interest from an FIB perspective.

- Given the inherent limitations of models for FIB, any model application should be considered a work in progress, to be improved as data are acquired, experience is gained, technologies are improved, and the state of art extends into new areas.

- Model results should be considered to have a life span which ends when it is possible to improve the model that gave rise to them. A treatment requirement, target performance metric or other model result should be revisited as and when the underlying model is or can be improved. Changes in technology, knowledge or practice related to FIB should signal a need to re-examine the appropriateness of TMDL and permit requirements.

- Specific processes affecting FIB transport and/or transformation in the environment may not be present in a particular model. Sedimentation, exposure to sunlight, and other key determinants of microbial behavior are commonly absent from available simulation tools. Consequently, it is commonly necessary to incorporate the effect of a variety of factors in some particular rate constant or modeling technique. In developing and documenting a model, some consideration should be given to the way that the physical reality is compressed into the components available in the modeling technology being applied.

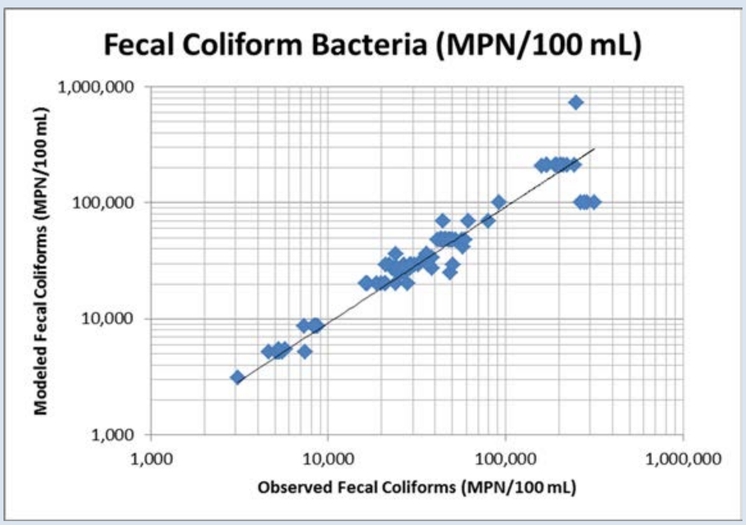
- Representation of organism growth and die-off is a requirement which is fundamental to FIB modeling. To accomplish this, models often condense a large number of factors into a single parameter first order relationship, which has an inherent limited ability to represent die-off only; a changing balance between growth and die-off is often mathematically beyond the scope of such models. At a gross scale, this limitation may be acceptable; however, it is a limitation that should be recognized in FIB modeling applications.

- Analysis of model sensitivity and uncertainty is a complex undertaking. It is useful and appropriate to establish the sensitivity of a model to a particular parameter. Translating that into an understanding of the uncertainty in prediction, however, may be impossible given that the statistical behavior of the variables in question is typically unknown. Uncertainty predictions in such a situation may lead to a false sense of confidence because the determination that a result is accurate plus or minus some amount may be mathematically unjustified.

- If possible, model calibration and verification should both be undertaken, using formal techniques. The predictive power of the calibrated model should be compared to the verification model in order to assess the confidence in the calibrated model. Predictive performance should be assessed for overall (long term) conditions, as well as for high flows, low flows and the range of conditions between them. The data to accomplish this are rarely available; where data do not support proper calibration and verification, the limitations in model validation should be explicitly documented. It should be noted that a recently published study by McCarthy et al. (2011) has made strides toward modeling of FIB in urban watersheds using the MOPUS model. McCarthy et al. (2011) illustrate good performance of the model for estimating event mean concentrations of *E. coli* (Nash-Sutcliffe coefficients from 0.56 to 0.76), and suggest further refinement and improvement of the model is possible.

Urban Runoff Modeling Using WinSLAMM

Urban modeling approaches that rely on sheetflow monitoring observations from each source area in a study area can be useful for prioritizing source areas. As an example, this is the approach utilized in WinSLAMM. Each source area in each land use has a probability plot based on sheetflow quality monitoring results. The calculated outfall quality is then compared to observed outfall quality for verification. The plot below shows calculated vs. observed fecal coliform values using WinSLAMM calibrated with typical source area sheetflows and regional values from the National Stormwater Quality Database for 114 separate locations. Even though this process does not consider each of the separate processes affecting FIB deposition, survival, washoff, and transport, it may be a useful method to predict reasonable outfall discharge values.



4.6 Conclusions and Recommendations

Behavior of microorganisms in the environment is a complex phenomenon that requires thorough site characterization in order to properly understand both the source of FIB and the environmental factors affecting persistence, growth and die-off in the environment. Management strategies for FIB are complicated by natural sources of FIB and environmental factors, requiring a realistic assessment of which sources and environmental factors can be controlled in urban settings. In some cases, regrowth of FIB in the environment or equilibrium conditions with high levels of FIB concentrations can make pinpointing the sources of FIB in urban environments challenging. Transfer of specific findings such as FIB die-off (decay) rates and regression equations between studies is of limited value due to the interactions of multiple environmental factors at study sites.

There are significant limitations associated with use of currently available models to accurately predict FIB loading and reductions associated with various management measures. These limitations are due to multiple factors such as limited understanding of fate and transport mechanisms in the natural environment, scale-related issues, limited data sets for model calibration and verification, and variable performance of stormwater control practices.

Ongoing research related to factors affecting sources, transport and fate of FIB, along with their removals by stormwater controls, may help to improve models in the future. Continued research on each separate process is needed not only for modeling, but also for better source control options and regulations. Until improved information is available, computer models should be used with care and calibrated and verified using local monitoring data reflecting site conditions. It should not be assumed that the most complex available tool is the best, however. Limitations in data, or applicability of the analytical method, or the intent of the project may dictate other options.

Regardless of the approach used to model or analyze the fate and transport of FIB in the environment, it is important to recognize that FIB are used as a surrogate for pathogens, but there may be significant differences in their behavior in the environment. FIB do not necessarily transport or transform in the environment in a way corresponding to the transport or transformation of the pathogens they presumably are intended to represent. For this reason, models predicting success in managing FIB may or may not predict success in managing pathogens. Continued research is needed not only regarding FIB processes and their representation, but also with regard to the relationship between FIB (or other surrogates) and pathogens.

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5 MONITORING AND SOURCE TRACKING

Monitoring strategies to develop an understanding of the sources of FIB can range from simple and relatively inexpensive sample collection and analysis of FIB and basic water quality parameters to more complex microbial source tracking (MST) approaches relying on advanced molecular methods. Generally, it is recommended that entities facing *E. coli* TMDLs begin with simple methods to identify and prioritize reaches of the receiving water of concern, then determine whether advanced methods are warranted or would provide additional benefits in terms of determining sources contributing FIB and pathogens to identified critical reaches. Monitoring and source tracking techniques selected may also be affected by budget constraints, regulatory drivers (e.g., numeric FIB permit limits), and available technical expertise.

In urban areas, initial data collection efforts for FIB-impaired streams typically include instream synoptic sampling combined with dry weather screening of storm sewer outfalls to identify potential illicit connections to storm sewers (CWP et al. 2004). Griffith et al. (2013) recommend a six-step process, as summarized in Figure 5-1. Using this approach, a community would only advance to subsequent steps if the previous step did not provide adequate information to identify sources of FIB pollution with a level confidence needed to develop management strategies to either reduce FIB or support an alternative regulatory resolution. Steps 1-3 are expected to be feasible in most communities, whereas steps 4-6 represent increasing costs that may or may not be justified, depending on the particular watershed, state regulatory requirements, and actual recreational uses present. This chapter provides an overview of monitoring and source tracking strategies that can be used by local governments, including a variety of methods suitable for a range of budgets and technical expertise. A limited discussion of emerging, advanced techniques is provided in this chapter, along with recommended references for more in-depth information.

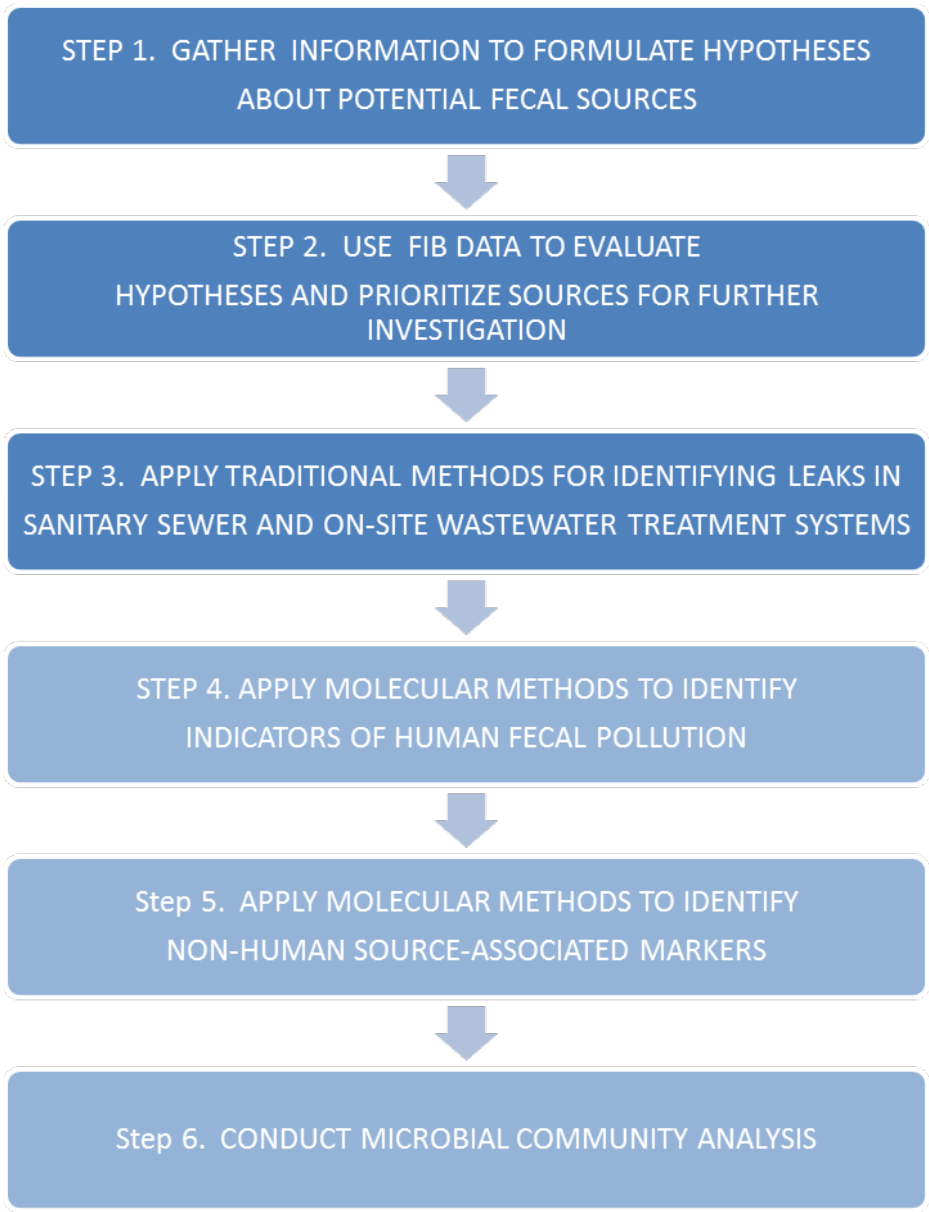
This chapter includes discussion of these topics:

- Basic monitoring for FIB in impaired waterbodies (the starting point for source investigations).
- Source tracking toolbox (multiple approaches that may be considered in refining understanding of FIB sources).
- Dry weather screening of storm drain outfalls, followed by chemical and molecular source tracking approaches.
- Wet weather monitoring.
- Traditional analytical methods for FIB (used by most communities).
- Molecular methods useful for source identification (an emerging area of practice).
- Monitoring to support QMRA (brief introduction to basic process for those considering a QMRA study).

- Targeted analysis using microcosm techniques (useful for refining understanding of factors that may be governing FIB persistence, regrowth and die-off in specific waterbodies).

- Data management.

Figure 5-1. Six-Step Process of Microbial Source Identification
(Based on Recommendations in Griffith et al. 2013)



5.1 Basic FIB Monitoring in Impaired Waters

The starting point for assessing sources of fecal contamination in receiving waters is to collect basic FIB data to determine which portions of the waterbody have elevated FIB. FIB tests are low-cost and can be conducted in-house by most municipal laboratories. One of the keys to an effective FIB monitoring program is to collect samples with adequate spatial and temporal resolution to target stream reaches where FIB targets are exceeded and to identify where significant FIB loading may be occurring. Synoptic sampling of streams, where an upstream to downstream set of locations is sampled on the same day is preferred. Both in–stream sources (such as contaminated sediment resuspension) and watershed sources need to be considered.

Due to very high variability of results in FIB data sets, discerning statistically significant trends with acceptable levels of power and confidence are typically not possible without relatively large data sets. Chapter 6 provides guidance on statistical analysis of FIB data, which may be helpful in determining the numbers of samples needed to meet data quality objectives for a monitoring program. Considerations when collecting and interpreting FIB data include:

- Select initial sampling locations that help to bracket potential FIB sources. Examples might include above and below WWTP discharges, dog parks, areas with heavy bird usage, sewer line crossings of streams, aging sanitary sewers above storm drains (Sercu et al. 2011), etc. (The “below” site for one target area can often serve as the “above” site for another target area.)

- Extreme variations in FIB concentrations can occur at the same location over relatively short time periods, so multiple samples over time are needed to begin to develop an understanding of potential trends and sources.

- Time of day of sample collection can affect FIB concentrations due to inactivation from natural UV light, flow variations that will affect the transport of bacteria discharged upstream through a sampled reach, and discharge variations of bacteria from potential sources. Early morning samples typically have the highest FIB concentrations.

- Seasonal variations in FIB are common, so erroneous conclusions may be drawn if adequate seasonal representation is not provided (e.g., a stream sampled in winter may meet stream standards, whereas a stream sampled in August may not meet standards). Sampling during dry and wet weather will likely also result in quite different results.

- FIB can persist or grow in the environment, so elevated FIB concentrations do not necessarily represent recent fecal contamination. This is particularly true of organic-rich, moist, dark environments such as sediments, decaying organic litter and biofilms. Scour of contaminated sediment and pore water are also known FIB sources associated with previous discharges.

- Unless exceptionally high, FIB concentrations typically do not provide information on the source of the contamination, so additional investigations or source tracking techniques are often needed to follow up initial analyses to identify sources.

- When collecting samples, be sure not to disturb stream sediment during sample collection. For example, collect the water sample first and then perform flow measurements.

- It is important to also collect sediment samples to help determine if sediment resuspension may be contributing to elevated FIB in the water column.

- Results are often above or below certain thresholds (e.g., <10 or >24,192 MPN/100 mL *E. coli*) causing difficulties in data interpretation and statistical analyses. When sewage input is suspected, FIB tests should be conducted at several dilutions, including very high dilutions so that concentrations close to those found in wastewater influent can be obtained (City of Santa Barbara 2012). Consult with the laboratory prior to finalizing the chain of custody in order to ensure an adequate range of quantification for FIB.

For standard operating procedures for FIB sample collection, see Standard Methods (APHA 2012) and CWP et al. (2004). Sample bottles appropriate to the analytical method should be used and samples should be kept cool (4°C) and quickly transported to the laboratory (6 hours is usually noted as a targeted time period between sample collection and analysis). In addition to FIB analyses, it is also often helpful to include analysis for other water quality indicators that

EPA Recommendations for Protecting Health and Safety of Field Staff When Sampling Contaminated Waters
(Source: EPA 2013)

In the *Marine Beach Sanitary Survey User's Manual*, EPA provides good recommendations to improve the safety of field staff involved with sampling potentially contaminated waters. These measures include:

- Limit exposure of *any* open wounds to survey site waters.

- Carry a hand sanitizer, and use it immediately after working at each survey location. (Use care when collecting samples not to make any contact with the inside of the sample containers.)

- Wear latex, nitrile, or other protective gloves; rubber boots; and safety glasses when contact is required or during sampling to minimize the potential for direct exposure to surface waters that are potentially contaminated.

- Carry a spray bottle with dilute bleach solution as part of your survey supplies for immediate disinfection if accidental exposure occurs.

- Practice good personal hygiene.
  - Avoid direct hand-to-mouth, -nose, or -face contact in the field.
  - Avoid eating, drinking, or chewing gum during site surveys. Delay drinking or consuming snacks and meals until you have removed all personal protective equipment and washed your hands and face thoroughly.
  - Promptly shower and wash your clothing with hot water after a day of surveying.

may help to identify human sewage sources and/or conditions that may be contributing FIB growth and persistence in the environment. Table 5-1 lists these parameters; most of these analyses can be conducted in municipal laboratories. Field data, including flow measurements or estimates, are recommended for all sample locations. Some or all of the additionally suggested water quality parameters should be considered based on the objectives of the sampling program since they may assist in identification of sources of discharges from an MS4. While no single parameter in Table 5-1 is a perfectly reliable indicator of sewage contamination, a suite of these parameters may provide an initial weight of evidence to identify potential sources or to identify where more advanced molecular methods should be used to confirm sewage contamination.

Table 5-1. Field and Analytical Parameters for Consideration in Basic FIB Sampling Programs

Field Data	Basic Analytical Parameters
o Flow (either at the sample location or documented from a nearby gage) o pH o Dissolved oxygen o Temperature o Conductivity o Weather conditions o Field observation of sources	o FIB (typically *E. coli*, enterococci or fecal coliform) o Nutrients[2] (e.g., ammonia[1], nitrate/nitrite, total Kjeldahl nitrogen, total phosphorus) o Organic carbon (total and dissolved)[2] o Turbidity o Fluoride[1] o Potassium[1] o Surfactants (typically measured as Methyl Blue Active Substances [MBAS])[1,3] o Optical brighteners (or fluorescence)[1]

[1]These may be sampled instream, at outfalls, or both as part of flow fingerprinting related to sources. See discussion in Section 5.3.1 for additional information on why various analytes are recommended. Advanced analytical parameters are also discussed in Section 9.2.8.
[2]Analytes that have been correlated to elevated FIB in some studies.
[3]Involves hazardous reagents.

Monitoring and Analysis Guidance References

American Public Health Association (APHA). 2012. *Standard Methods for the Examination of Water and Wastewater, 22nd Edition*. Clesceri, L. S., Greenberg, A. E., Eaton, A. D., Eds. Published in conjunction with the American Water Works Association and the Water Environment Federation: New York.

American Water Works Association Research Foundation and U.S. Environmental Protection Agency (AWWARF and EPA). 2006. *Development of Event-Based Pathogen Monitoring Strategies for Watersheds*. IWA Publishers.

Burton and Pitt. 2001. *Stormwater Effects Handbook: A Toolbox for Watershed Managers, Scientists, and Engineers*. http://www.epa.gov/ednnrmrl/publications/books/handbook/index.htm

Center for Watershed Protection and R. Pitt. 2004. *Illicit Discharge Detection and Elimination: A Guidance Manual for Program Development and Technical Assessments*. Center for Watershed Protection, Ellicott City, MD, and University of Alabama, Birmingham, AL. http://www.cwp.org/online-watershed-library/cat_view/64-manuals-and-plans/79-illicit-discharge-detection-and-elimination

EPA. 1992. *National Pollutant Discharge Elimination System Stormwater Sampling Guidance Document*. EPA 833-B-92-001. http://www.epa.gov/npdes/pubs/owm0093.pdf

EPA. 1997. *EPA Monitoring Guidance for Determining the Effectiveness of Nonpoint Source Controls*. EPA 841-B-96-004. http://nepis.epa.gov/EPA/htmL/Pubs/pubtitleOW.htm

EPA. 2013. *Marine Beach Sanitary Survey User's Manual*. http://water.epa.gov/type/oceb/beaches/upload/Marine-Beach-Sanitary-Survey-User-Manual-March-2013.pdf

Federal Highway Administration. 2001. *Guidance Manual for Monitoring Highway Runoff Water Quality*. http://www.fhwa.dot.gov/environment/h2o_runoff/index.htm

Geosyntec and Wright Water Engineers, Inc.2009. *Urban Stormwater BMP Monitoring*. Prepared for EPA and WERF. www.bmpdatabase.org

Granato et al. 2002. *National Highway Runoff Water-Quality Data and Methodology Synthesis, Volume I --Technical Issues for Monitoring Highway Runoff and Urban Stormwater*. http://ma.water.usgs.gov/fhwa/fhwaep.htm

Griffith et al. 2013. *The California Microbial Source Identification Manual: A Tiered Approach to* Identifying *Fecal Pollution Sources to Beaches.* http://www.waterboards.ca.gov/board_info/agendas/2014/jan/012114_11_804_sipp_mst_manual.pdf

Hagedorn, C., Blanch, A.R., V.J. Harwood, V.J. 2011. *Microbial Source Tracking: Methods, Applications, and Case Studies*. Springer Media: NY, NY.

Keith, L.H. (ed.). 1996. *Principles of Environmental Sampling*, 2nd ed. American Chemical Society.

State Water Resources Control Board and the Southern California Stormwater Monitoring Coalition. 2004. *Southern California Coastal Water Research Project, Model Monitoring Program for Municipal Separate Storm Sewer Systems in Southern California. Technical Report #419*. ftp://ftp.sccwrp.org/pub/download/DOCUMENTS/TechnicalReports/419_smc_mm.pdf

U.S. Department of Agriculture-National Resources Conservation Service. 1996. *National Handbook of Water Quality Monitoring.* 450-vi-NHWQM. http://www.wsi.nrcs.usda.gov/products/W2Q/water_qual/docs/wqm1.pdf

U.S. Geological Survey (USGS). (various dates). Techniques of Water-Resources Investigations Reports. Multiple chapters. http://pubs.usgs.gov/twri/

5.2 Source Tracking Toolbox

There are many techniques that communities can use to explore and identify sources of elevated FIB in receiving waters. The selection of techniques should be based on initial hypotheses formed from basic FIB monitoring and in most urban areas should include basic dry weather screening of outfalls in stream reaches with elevated FIB (discussed in Section 5.3). Some of these methods have been available for 20 years or more (e.g., Pitt et al. 1993, CWP et al. 2004), whereas others include recently published methods that integrate significant advances in microbial source tracking (e.g., Griffith et al. 2013). There are strengths and limitations of both the older and more recent approaches, and each community will need to balance their source tracking objectives with available budget and technical resources. These budget-related decisions also need to consider the benefits that a well formulated source tracking program may provide relative to the projected costs of the actions specified in TMDL implementation plans. In some communities, multi-million or billion dollar implementation plans have been developed to work toward addressing FIB impairments; thus, substantial benefits may be gained by a well-developed and clearly targeted monitoring program. In some states, definitely eliminating human sources may enable some regulatory relief for the MS4 (see Section 5.7 for a discussion of QMRA).

Table 5-2 provides a summary or toolbox of potential source tracking methods, ranging from simple to complex. This table integrates findings from earlier EPA-sponsored work by the Center for Watershed Protection et al. (2004) titled *Illicit Discharge Detection and Elimination Manual* and two recently developed key references on source identification approaches that incorporate use of molecular methods. The two latter references include *The California Microbial Source Identification Manual: A Tiered Approach to Identifying Fecal Pollution Sources to Beaches* (Griffith et al. 2013) and *Tools for Tracking Human Fecal Pollution in Urban Storm Drains, Creeks, and Beaches* (City of Santa Barbara 2012a&b). The primary purpose of the tools in Table 5-2 is to identify signals of human waste in creeks, beaches, and storm drains and track these signals to their sources. Several of these techniques are discussed in more detail later in this chapter, but the toolbox concept is addressed first because monitoring programs should ideally be designed considering the big picture of how a study could evolve (i.e., move forward sequentially).

Table 5-2. Source Tracking Tools

(Modeled after *Tools for Tracking Human Fecal Pollution in Urban Storm Drains, Creeks, and Beaches*, City of Santa Barbara 2012a&b; supplemented by Pitt et al. 1993, Center for Watershed Protection et al. 2004)

Tool	Best Use	Caveats and Challenges	Cost
Visual Surveys of Potential Sources	Homeless encampments, sites with frequent daytime use, under bridges, obvious contamination associated with inappropriate discharges.	Feces often contained in newspaper or plastic bags.	$
GIS	Essential for planning and analyzing data in relation to infrastructure. Useful prior to initial field investigations, as well as for targeting areas for more detailed investigations.	Requires accurate data for both storm drains and sanitary sewers, including pipe elevations and inverts, where available.	$$
Dry Weather Outfall Screening	Identification of flowing outfalls for water quality sampling, along with physical observations (odor, color floatables, deposits, stains).	Dry weather flows can originate from both contaminated and uncontaminated sources.	$$
FIB (*E. coli*, enterococci)	Basic indicator of potential fecal contamination tied to regulatory receiving water limits.	Recommended in conjunction with additional chemical or molecular tests. Urban wildlife and pets may be responsible for high values observed. Biofilms and sediment sources may also contribute to elevated FIB. (May be elevated in the absence of human sources.)	$
Chemical Indicators (Basic Flow Fingerprinting/ Non-human Chemistry)	Finding illicit connections. Good for understanding nutrient inputs from any type of illicit connection. Example indicators include: detergents, fluorides, ammonia, and potassium. Others may also be useful.	May not identify direct human deposition (e.g., homeless) and small sewage leaks that are significantly diluted by other flows. Background signal of urban runoff can make fingerprinting sewage difficult in some urban areas.	$$
Chemical Indicators (Advanced Markers of Human Waste)	Finding sewage leaks. Advanced analyses may include: sucralose, caffeine, and cotinine.	Some advanced chemical indicators may be present in the environment from surface deposition, rather than sewage sources (e.g., dumping coffee down storm drains).	$$
Canine Scent Tracking	Best for use when real time results are desired, such as working up storm drain networks with many branches. Also when broad spatial coverage is sought.	Canines may respond to non-human illicit connections, due to training with detergents. Requires specially trained canines with trained staff.	$$

Tool	Best Use	Caveats and Challenges	Cost
CCTV (Closed Circuit Television) of Storm Drains	Best for use where sampling data suggests sustained input of sewage.	Most operators are trained for sanitary sewer pipe inspection, and may seek to clean the lines first. Plan to guide operators to slow down, look carefully at leaks, and do not clean the lines first (in order to see solids on bottom of storm drain).	$$
Electric Current Flow Method	The method uses the variation of electric current flow through the pipe wall to locate defects that are potential water leakage paths either into or out of the pipe.	See ASTM F2550 – 13. Applies only to electrically non-conducting pipes w/ diameters of diameters of 3 to 60 in.	ND
Basic Dye Test	Best for testing laterals or fixtures feeding a single illicit connection that has been observed by CCTV.	Use bright green dye and a UV light to look for dye in storm drains.	$
Smoke Test	Best for limited geographic areas with strong evidence for direct connections (e.g., toilet paper).	Difficult in large pipes and densely populated areas.	$$
Dye with Rhodamine Probe	Best for testing suspected sewage infiltration to storm drains when persistent human-waste markers are found w/out observing solids such as toilet paper.	Difficult to know how long to leave probe in storm drain. Rain events may create a false positive signal.	$$
Automated continuous flow gauges and autosamplers	Best for drains with evidence of higher flows (wet walls, signs of water shooting into creek channel). Supports load estimation.	Check specs carefully to find flow gauges suitable for dry weather flows. Requires confined space entry in most cases.	$$$ (initial)
Temperature Probes	Can be placed in storm drain outfalls to further verify certain types of suspected illegal connections (e.g., flushing/showering patterns).	Does not identify where the illegal connection is located. More useful in smaller drainage areas.	$
Human-specific waste markers (Advanced Technique)	Best tool for quantifying inputs of human waste. Best for sampling in creeks, beaches, storm drain outfalls or major nodes in storm drain network.	Plan repeated sampling to account for variable results. Requires more expertise and cost.	$$$
Community approach, e.g., Phylochip (Emerging Advanced Technique)	Best for sampling along a gradient of suspected inputs, (e.g., to test if septage is entering a creek). May be advantageous in storm drains diluted with clean ground water, due to low detection thresholds.	At this point, results are not conducive to simple interpretation suitable for a nontechnical audience. Requires more expertise and cost.	$$$$

Notes: Cost—increasing $ indicates more expensive techniques. ND = not determined.

5.3 Dry Weather Outfall Screening

Dry weather screening is one of the most important tools available to municipal stormwater managers. Identification and removal of illicit discharges and illegal connections may be the single most important action that municipal stormwater managers can take to reduce human sources of contamination.

The Center for Watershed Protection et al. (2004) prepared *Illicit Discharge Detection and Elimination: A Guidance Manual for Program Development and Technical Assessments* under EPA funding to provide guidance to communities in developing effective management programs and field guidance to reduce illicit discharges. The approximately 200-page manual provides detailed guidance for those embarking on dry weather surveys. The discussion which follows provides a significantly condensed version of steps required to conduct an Outfall Reconnaissance Inventory (ORI) and some aspects of indicator monitoring. ORI field forms, which have been effectively used by many communities are provided in Appendix D of the Center for Watershed Protection et al. (2004) manual (accessible at: http://www.epa.gov/npdes/pubs/idde_manualwithappendices.pdf). The minimum list of monitoring parameters for use in dry weather screening includes flow rate (estimated or measured), water temperature, the regulated FIB parameter, and pH. Additionally recommended parameters for source fingerprinting include ammonia and potassium (for calculating ammonia/potassium ratios), fluoride, phosphorus, surfactants and/or optical brighteners (as summarized in Table 5-1). The basic steps for an outfall reconnaissance inventory include:

1. **Collect background data**. At a minimum, this includes an initial map of storm sewer outfalls. Other background information, when available, can include more detailed sanitary and storm sewer infrastructure mapping, age-related and maintenance information for the sanitary sewer system, citizen complaints, known hotspots draining to the outfall and other information. As GIS is increasingly used by local governments in many urban areas, a significant amount of information can be compiled prior to fieldwork.

2. **Develop outfall descriptions**. This includes information on the size and pipe material of the outfall, among other information.

3. **Conduct quantitative characterization of flowing outfalls.** This includes estimates of flow rates. For techniques useful for measuring or estimating flow rates, see Center for Watershed Protection et al. (2004).

4. **Assess and document physical indicators for flowing outfalls.** Examples of physical indicators of potential FIB contamination include odor, staining, and evidence of sanitary waste (e.g., feces, toilet paper).

5. **Assess and document physical indicators for both flowing and non-flowing outfalls.** Visual indicators present at non-flowing outfalls imply intermittent inappropriate discharges, although water samples for analyses may not be available.

6. **Complete initial outfall designation and follow-up sampling actions.** Based on the initial screening activities, flowing outfalls with indicators of potential FIB contamination should be sampled several times. If an outfall is identified as possibly contaminated, additional sampling and investigations are conducted along the main storm drainage system to isolate the likely reaches of contamination to narrow the watershed investigations to identify the sources. Several different sampling approaches can be used at this stage, including a chemical tracer approach (discussed below), molecular methods, and use of advanced markers.

Prior to discussing various approaches for dry weather investigations, general guidance on dry weather sample collection is important. As is the case with instream sampling, the timing of sample collection from outfalls can affect their results. Center for Watershed Protection et al. (2004) provide these recommendations regarding timing of sample collection:

- Sample in the late fall/early spring because outfalls are easiest to spot during leaf-off or dormant vegetation conditions. Once identified and located, the outfalls should be re-visited at other seasons as inappropriate discharges may be seasonal. It is common for outfalls to continue to be found even after several surveys. Small outfalls draining creek-side businesses may be especially problematic as they are not likely identified on city drainage maps, but have been found to be more frequently contaminated than large outfalls.

- Sample after a dry period of at least 48 hours (trace rainfall activity may be acceptable depending on the size of the watershed). However, periods of regional high groundwater should also be included during surveys to identify possible groundwater intrusion sources.

- Sample in the early morning/late afternoon, when feasible. Checking outfalls when people are home may increase the chances of catching an inappropriate connection (e.g., flushing, showering).

- Avoid conditions during snow melt and/or if salt has been applied to the road system draining to the outfalls. Also note that some field tests (e.g., ammonia, chlorine) are affected by cold temperatures or confounded by the presence of salt (e.g., detergents).

- If outfall monitoring is occurring along a tidal body of water, data collection dates and times should be selected to take advantage of the lowest possible tide, this will allow for the easiest, safest and most accurate and complete assessment of outfalls. If the outfalls are always submerged, sampling needs to occur upgradient in the drainage system above the influence of backwater.

Following initial identification of flowing outfalls, several different source tracking approaches may be used. Examples of several approaches that have been used successfully in various locations follow.

5.3.1 Chemical Tracer Methodologies (Using Basic Flow Fingerprinting)

A chemical tracer methodology can be used to conduct a mass balance of all dry weather flows at an outfall or in a drainage system in order to identify and quantify the flow sources, including sanitary sewage. It is not specifically used to directly identify the sources of FIB, but the presence of wastewaters and other flows that may be contaminated with FIB. An investigation of non-stormwater discharges into storm drainage needs to proceed along a hierarchy of procedures and locations, progressing from exploratory techniques to confirmatory procedures. The methodology briefly summarized here was developed over many years for the EPA and verified in numerous communities (CWP et al. 2004). This procedure recognizes that limited resources are available to municipalities and makes maximum use of information typically available, prior to proceeding to advanced methods.

The purpose of the investigative procedures is to separate storm drain outfalls having dry weather discharges into at least three general categories (with a known level of confidence) to identify which outfalls (and drainage areas) need further analyses and investigations. These categories are outfalls affected by non-stormwater discharges from: (1) pathogenic or toxic pollutant sources, (2) nuisance and aquatic life threatening pollutant sources, and (3) unpolluted water sources. The pathogenic and toxic pollutant source category would be considered the highest priority due to potential human health impacts or significant impacts on receiving water organisms. Nuisance and aquatic life threatening pollutant sources may include laundry wastes, landscaping irrigation runoff, automobile washing, construction site dewatering, and washing of ready-mix concrete trucks. These pollutants can cause excessive algal growths, tastes and odors in downstream water supplies, offensive coarse solids and floatables, and highly colored, turbid or odorous waters. Clean water discharged through stormwater outfalls can originate from natural springs feeding urban creeks that have been converted to storm drains, infiltrating groundwater, infiltrating domestic water from water line leaks, etc.

If the relative amounts of potential components are known, then the importance of the dry weather flows can be determined. As an example, if a baseflow is mostly uncontaminated groundwater, but contains 5% raw sanitary sewage, it would be an important source of potentially pathogenic bacteria. Typical raw sanitary wastewater parameters (such as BOD_5 or suspended solids) would be in relatively low concentrations in the mixture and the sanitary wastewater source would be difficult to detect. Fecal coliform bacteria measurements would not help much because they originate from many possible sources, besides sanitary wastewater. Unique microorganism or biochemical measurements would probably be needed to detect the presence of the wastewater directly, as previously described in this report. Chemical tracers can be used to identify relatively low concentrations of important source flows in storm drain dry-weather flows using various fingerprinting procedures. Ideal tracers should have the following characteristics:

- Significant differences in concentrations between possible pollutant sources;
- Small variations in concentrations within each likely pollutant source category;

- A conservative behavior (i.e., no significant concentration change due to physical, chemical or biological processes); and

- Ease of measurement with adequate detection limits, good sensitivity and repeatability.

Table 5-3 is a summary of the tracer parameter measurements used during the early development of these methods in Birmingham, Alabama (Pitt et al. 1993). This table is a summary of the "library" that describes the tracer conditions for each potential source category based on monitoring 10 to 25 samples of water from each category (the number needed depends on the variability and the desired level of errors). The information shown on this table includes the mean and coefficient of variation (COV) values for each tracer parameter for each source category, along with the probability distribution type uniform, normal, or log-normal). The COV is the ratio of the standard deviation to the mean. A low COV value indicates a much smaller spread of data compared to a data set having a large COV value. As noted above, appropriate tracers are characterized by having significantly different concentrations in flow categories that need to be distinguished. In addition, effective tracers also need low COV values within each flow category. These studies indicated that the COV values were quite low for each category, with the exception of chlorine, which had much greater COV values. Chlorine is therefore not recommended as a quantitative tracer to estimate the flow components. Similar data needs to be collected in each community where these procedures are to be used.

Samples are collected from all flowing outfalls using the procedures described by CWP et al. (2004). That report also has detailed guidance on ancillary observations while in the field. The surveys should be repeated several times during the first year as intermittent flows may change seasonally. After potentially problematic outfalls are identified, similar sampling and analyses is conducted at various manhole locations in a drainage system to isolate the reach where the problem flows are entering the drainage system.

Several options can be used to evaluate the collected screening data. A flow chart method (shown as Figure 5-4) is simple to use and has been shown to be quite accurate, as shown on Table 5-4, which provides an example of verification tests (extensive watershed and drainage system surveys to locate the actual sources) conducted in Birmingham, AL (Pitt et al. 1993). In this case, the flow chart method was used and the number of correctly (and incorrectly) identified discharges was tracked. Tests on ten Birmingham outfalls were mostly favorable, with the flow chart method correctly identifying contaminated discharges in all cases (i.e., washwater or sewage wastewater). At one outfall, the flow chart incorrectly identified sewage as washwater, based on an ammonia (NH3)/potassium (K) ratio of 0.9, which was very close to the breakpoint in the flow chart method (ratio of 1.0).

Table 5-3. Summary of Chemical Characteristics of Source Samples Collected in Birmingham, Alabama (Pitt et al. 1993)

Source	Conductivity (μS/cm)	Fluoride (mg/L)	Hardness (mg/L as $CaCO_3$)	Detergent (mg/L)	Fluorescence % scale	Potassium (mg/L)	Ammonia (mg/L)	Color (units)	Chlorine (mg/L)
Spring Water									
mean	301	0.03	240	0.00	6.80	0.73	0.01	0.0	0.00
COV	0.04	1.00	0.03	n/a	0.43	0.10	2.00	n/a	n/a
distribution	normal	normal	normal	uniform	normal	normal	L-norm	uniform	uniform
Shallow Groundwater									
mean	51.4	0.06	27.3	0.00	29.9	1.19	0.24	8.0	0.02
COV	0.84	0.50	0.39	n/a	1.55	0.44	1.26	1.42	1.62
distribution	normal	L-norm	normal	uniform	L-norm	normal	normal	L-norm	normal
Tap Water									
mean	112	0.97	49.3	0.00	4.63	1.55	0.03	0.0	0.88
COV	0.01	0.01	0.03	n/a	0.08	0.04	0.23	n/a	0.68
distribution	normal	normal	normal	uniform	normal	normal	normal	uniform	bi-modal
Landscaping Irrigation									
mean	105	0.90	40.2	0.00	214	6.08	0.37	10.0	0.03
COV	0.07	0.11	0.04	n/a	0.16	0.26	0.25	0.36	1.02
distribution	normal	normal	normal	uniform	normal	normal	normal	normal	normal
Sewage									
mean	420	0.76	143	1.50	251	5.97	9.92	37.9	.01
COV	0.13	0.23	0.11	0.82	0.20	0.23	0.34	0.55	2.00
distribution	normal	normal	normal	normal	normal	normal	L-norm	normal	L-norm
Septic Tank Discharge									
mean	502	0.93	56.8	3.27	382	18.8	87.2	70.6	0.07
COV	0.42	0.39	0.36	1.33	0.22	0.42	0.40	0.39	1.30
distribution	normal	normal	L-norm	L-norm	normal	normal	normal	normal	normal
Carwash									
mean	485	12.30	157	49.0	1190	42.7	0.24	222	0.07
COV	0.06	0.19	0.05	0.10	0.11	0.37	0.28	0.35	1.14
distribution	normal	normal	normal	normal	normal	normal	normal	normal	bi-modal
Laundry									
mean	563	32.82	36.2	26.9	1024	3.48	0.82	46.7	0.40
COV	0.21	0.38	0.08	0.25	0.12	0.11	0.14	0.27	0.26
distribution	normal	normal	normal	normal	normal	normal	normal	normal	normal
Radiator Waste									
mean	3280	149.32	5.60	15.0	22046	2802	26.3	2999	0.03
COV	0.21	0.16	1.88	0.11	0.04	0.13	0.89	0.01	0.52
distribution	normal	normal	normal	normal	normal	normal	normal	normal	normal
Plating Waste									
mean	10352	5.13	1430	6.81	293	1009	65.6	104	0.08
COV	0.45	0.47	0.32	0.68	0.70	1.24	0.66	0.91	1.20
distribution	normal	normal	normal	normal	normal	L-norm	normal	normal	L-norm

Figure 5-2. Flow Chart to Identify Most Likely Significant Flow Component Contributing to Elevated FIB

(Source: Shergill and Pitt 2004, modifies Pitt et al. 1993)

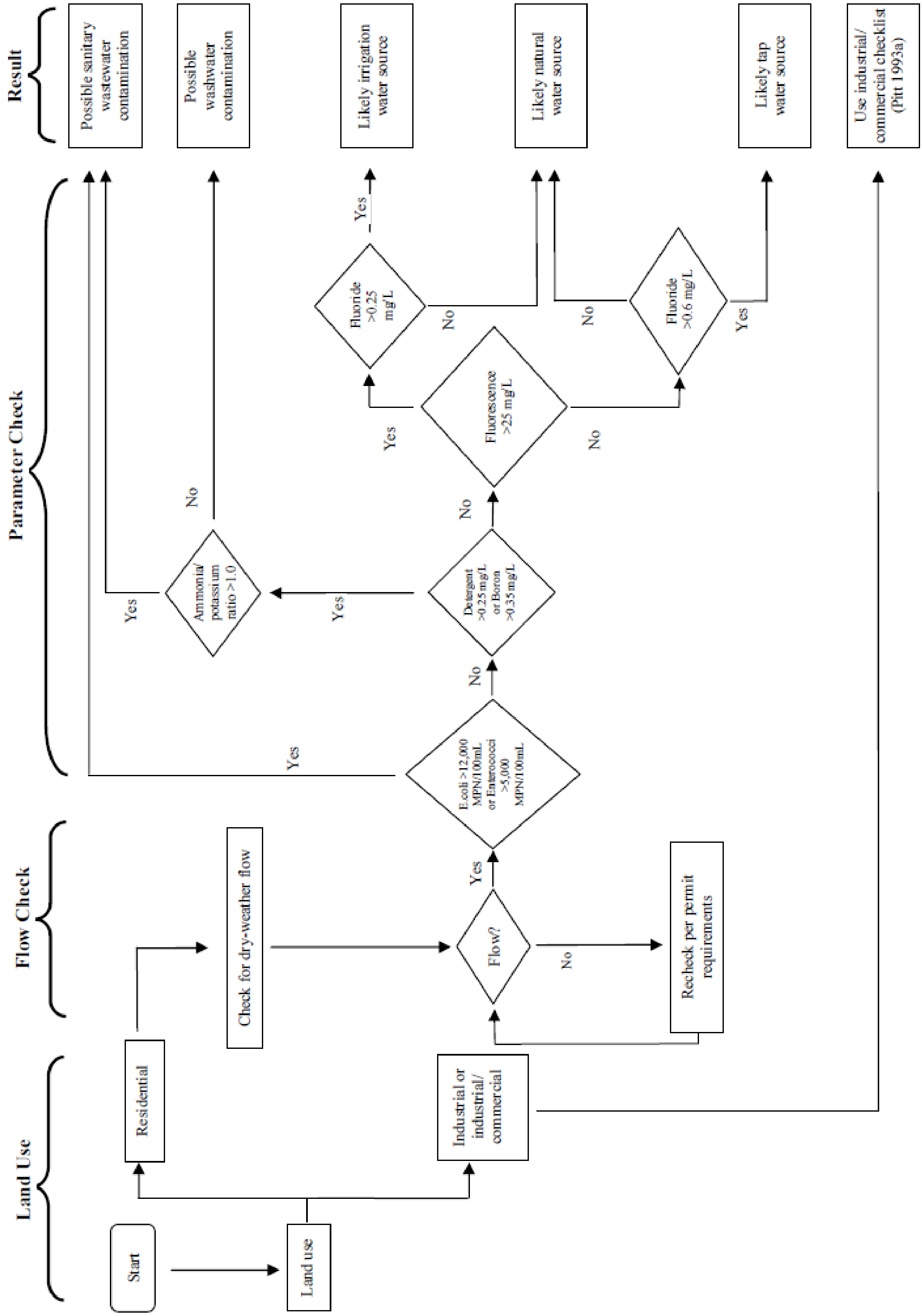


Table 5-4. Evaluation of the Flow Chart Method Using Data from Birmingham, Alabama
(Adapted from Pitt et al. 1993)

Outfall ID	Outfall Concentrations (mg/L)					Predicted Flow Type	Confirmed Flow Type	Result
	Detergents-Surfactants (>0.25 is sanitary or wash water)	NH3	K	NH3/K (>1.0 is sanitary)	Fluoride (>0.25 is tap, if no detergents)			
14	0	0	0.69	0.0	0.04	Natural Water	Spring Water	Correct
20	0	0.03	1.98	0.0	0.61	Tap Water	Rinse Water (Tap) and Spring Water	Correct
21	20	0.11	5.08	0.0	2.80	Washwater	Washwater (Automotive)	Correct
26	0	0.01	0.72	0.0	0.07	Natural Water	Spring Water	Correct
28	0.25	2.89	5.96	0.5	0.74	Washwater	Washwater (Restaurant)	Correct
31	0.95	0.21	3.01	0.1	1.00	Washwater	Laundry (Motel)	Correct
40z	0.25	0.87	0.94	0.9	0.12	Washwater	Shallow Groundwater and Septage	Identifies Contaminant but Incorrect Flow Type
42	0	0	0.81	0.0	0.07	Natural Water	Spring Water	Correct
48	3.0	5.62	4.40	1.3	0.53	Sanitary Wastewater	Spring Water and Sewage	Correct
60a	0	0.31	2.99	0.1	0.61	Tap Water	Landscaping Irrigation Water	Correct

It is also possible to estimate the outfall source flow components using a set of simultaneous chemical mass balance equations. A stochastic version of this procedure, developed by Lalor (1994), enabled the variation within the library values for each source type to be considered using a Monte Carlo simulation approach. Table 5-5 provides a comparison of the predicted flow sources using this tool at these same ten Birmingham area outfall samples compared to the confirmed flow sources. Major flow sources were mostly identified correctly, and in all cases, the problematic waters were correctly identified as needing further investigation.

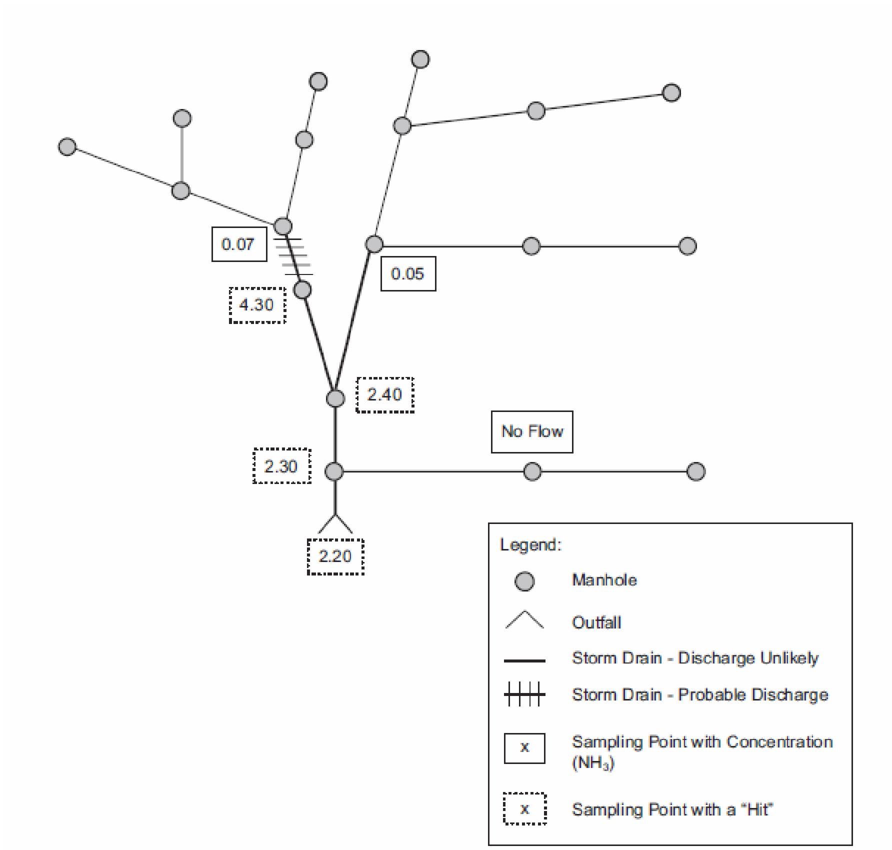
Table 5-5. Analysis of Outfalls Based on Results of the Chemical Mass Balance Program
(Source: Lalor 1994)

Outfall Number	Predicted Flow Source	Confirmed Flow Source
14	88% Spring (7% Sewage) (5% Tap)	100% Spring
20	60% Tap 32% Spring (8% Irrigation)	67% Tap 33% Spring
21	55% Sewage 35% Groundwater (8% Car Wash) (2% Laundry)	100% Washwater (Automotive)
26	74% Spring Water 18% Tap Water (8%Sewage)	100% Spring
28	46% Groundwater 21% Irrigation Water 18% Sewage 10% Spring Water (5%Tap Water)	100% Washwater (Restaurant)
31	55% Sewage 25% Spring Water 18% Laundry (1% Carwash Water)	100% Laundry (Motel)
40z	27% Sewage 23% Tap Water 19% Ground Water 12% Spring Water 11% Septic Tank Discharge (8% Irrigation Water)	Shallow Groundwater and Septic Tank Discharge
42	63% Spring Water 28% Tap Water (9% Sewage)	100% Spring Water
48	79% Sewage 15% Spring Water (5% Carwash Water) (1% Septage)	50% Sewage 50% Spring Water
60a	56% Tap Water 37% Irrigation Water (7% Sewage)	100% Irrigation Water

Once problem outfalls are identified, these fingerprinting techniques can be applied to the contributing storm drain system to further focus source identification and correction measures, as illustrated in Figure 5-3.

Figure 5-3. Use of Ammonia as a Tracer to Identify Drainage System Sections Contributing Contaminated Flows

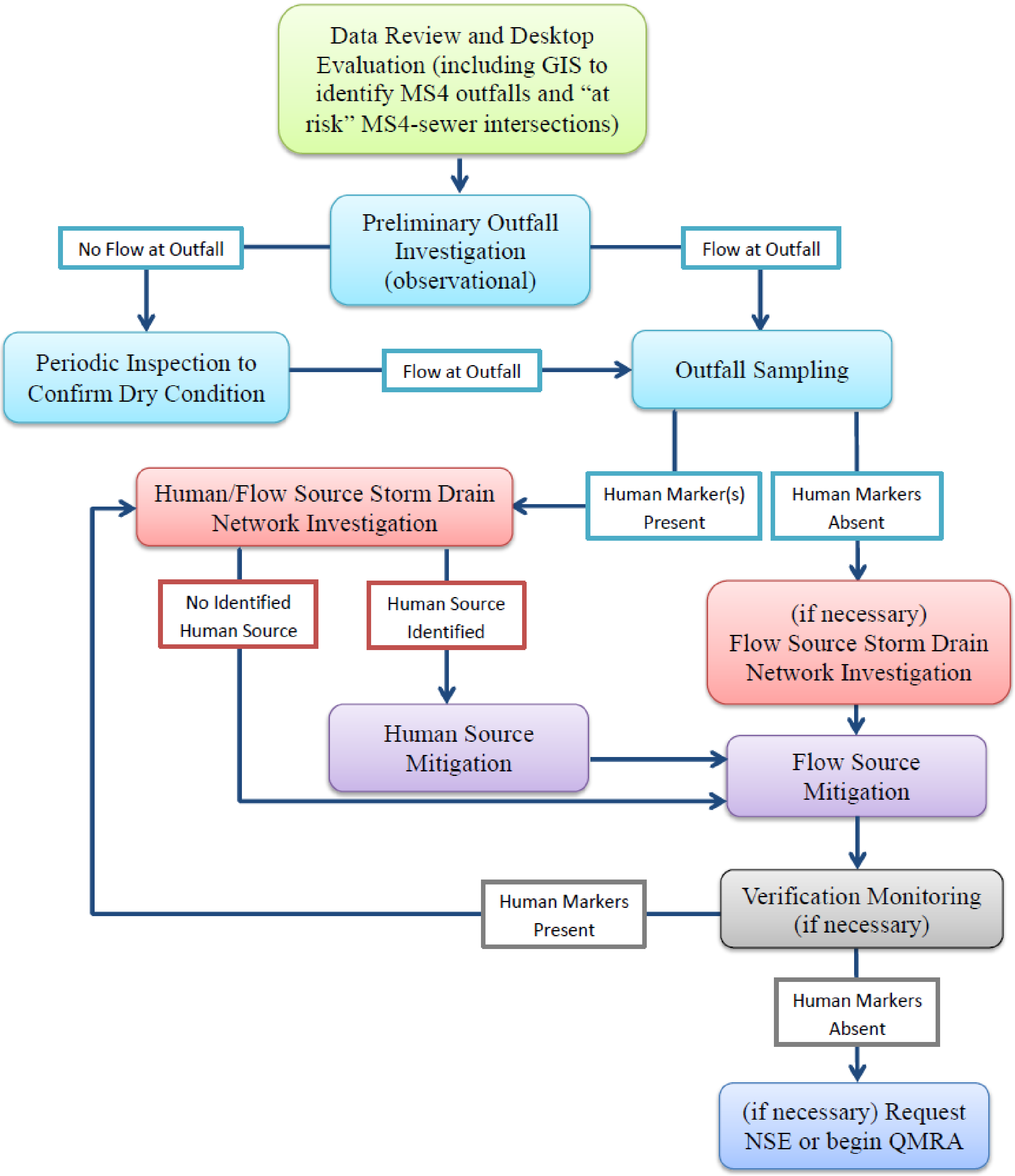
(Source: CWP et al. 2004)



5.3.2 Microbial Source Tracking Toolbox Approach

Given recent progress in the use of microbial methods as part of a source tracking toolbox, some communities have moved towards more routine use of microbial methods as part of dry weather monitoring at storm drains discharging to impaired waterbodies in urban areas. Figure 5-4 provides an example flow chart illustrating how such methods can be applied. Initial steps in this process focus on desktop review of available data, including GIS mapping of sanitary sewers and storm drains, to identify potential problem areas. Next, initial field investigations focus on identifying and mapping outfalls with dry weather flows. This approach then moves directly to sampling outfalls with dry weather flows for human markers using molecular methods, with a key objective being to quickly determine the presence or absence of human sources of fecal contamination. Where human markers are identified, then additional investigations regarding sources of the flows are initiated, using a toolbox of methods. Once sources of human contamination are determined, then mitigation of these sources can begin. Where human sources are absent, it may be possible to reduce flow sources, in some cases (e.g., excessive irrigation).

Figure 5-4. MS4 Microbial Source Identification Investigation Approach
(Source: Brandon Steets, Geosyntec Consultants)



Weber et al. (2013) provide an example application of a microbial source tracking toolbox approach in response to the San Diego River TMDL. A process similar to Figure 5-4 was followed in this investigation. The regulatory driver for the investigation was the San Diego Regional MS4 Permit, which implements requirements of a FIB TMDL. The requirements focus on dry weather sources associated with TMDL compliance goals, with actions oriented toward prioritizing human fecal sources, as well as “controllable anthropogenic” sources (domesticated animals, etc.). (Also see the case study in Chapter 8 for more information on this TMDL.) Reconnaissance activities associated with this study included identifying areas with potential for human fecal inputs and categorizing outfalls based on proximity to receiving waters, and sewer mains/septics. As part of the reconnaissance effort, approximately 110 outfalls were visited, with 19 determined to be flowing. Electronic field forms (i.e., a mobile “app” for use on field tablets) were used to document key field conditions and directly associate collected data with GIS mapping of the storm sewer system (automatically uploaded to a database), which enabled development of consolidated information as shown in Figure 5-5. Tabular review of data enabled a “weight of evidence” approach to assess the likelihood of human sources, as shown in Table 5-6.

Table 5-6. Example Tabulation of Results from Microbial Source Tracking Using a Toolbox Approach in San Diego

(Source: Weber et al. 2013)

Outfall ID	Human - HF183	Human - HumM2	Dog - BacCan	Ammonia as N (mg/L)	Total Phosphate as P (mg/L)	MBAS (mg/L)	Caffeine (ng/L)	Cotinine (ng/L)	Sucralose (ng/L)
SDR-13	0	0	0	0.18	0.28	<0.10	310	38	N/A
SDR-15D	0	0	1	0.06J	0.23	<0.10	6.1	4.5	N/A
SDR-35	0	1	0	<0.02	0.03J	<0.10	150	21	1300
SDR-41	0	0	1	0.08J	0.12	<0.10	52	15	N/A
SDR-696	0	0	0	0.15	0.18	1.4	37	13	91
SDR-714	0	0	0	0.18	0.51	0.32	1000	1200	N/A
SDR-739	0	0	0	0.13	0.1	<0.10	46	30	N/A
SDR-751	0	0	0	1.19	1.07	<0.10	32	8.9	N/A
SDR-754	0	1	0	0.2	0.3	<0.10	14	6.1	N/A
SDR-758	0	0	0	1.06	0.18	<0.10	150	6.3	N/A
SDR-760	0	0	0	0.37	0.12	<0.10	2000	96	N/A
SDR-770 & 769	1	0	0	0.13	0.17	<0.10	9.8	<2	73
SDR-774A	0	0	0	0.07J	0.13	<0.10	7.4	<2	34
SDR-780	0	0	0	0.15	0.12	<0.10	1.8	<2	N/A
SDR-791	0	0	0	0.27	0.61	<0.10	67	12	N/A
SDR-825	0	0	1	0.28	0.52	<0.10	60	24	430
SDR-858	0	1	0	0.13	0.04J	0.1	220	130	420
SDR-939	0	0	0						N/A
SDR-999	0	0	0	0.21	0.33	<0.10	88	17	N/A

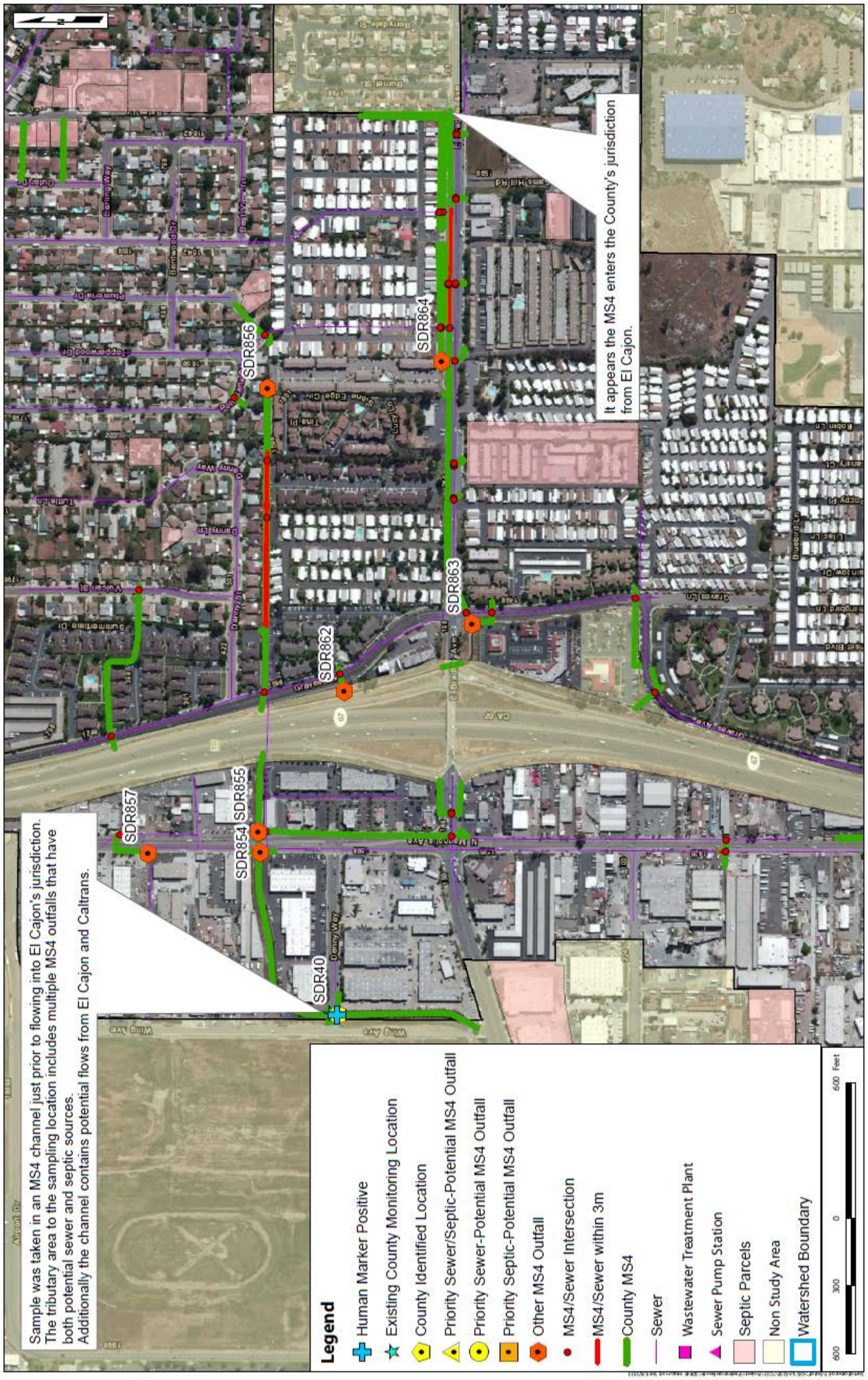
Note: Rows highlighted in blue detected one or more human molecular markers.

Some of the questions that can be asked using a weight of evidence approach, once data are tabulated, include:

- What is the method sensitivity and specificity (likelihood of false positive and false negative rates)?
- What are the marker levels and frequency of detection?
- Do the molecular or chemical marker results correlate with FIB concentrations?
- Was the presence of human marker due to a single fecal contamination event or representative of persistent pollution?
- Do the results match up with visual information?
- Is a “referee lab” needed to increase confidence in sample results?

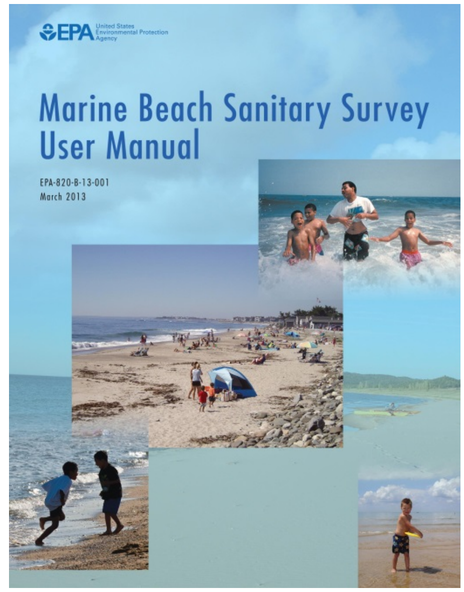
In the example shown in Table 5-6, the appropriate next steps for the project were determined to be follow-up on the four outfalls where human markers were detected. Activities considered for follow-up included additional investigation of flow sources to and within the storm system networks using techniques such as CCTV and dye testing of the target area, as well as additional sampling within the storm system network for FIB, sucralose, markers and human markers. Regrowth studies within the storm drain network may also be considered (Weber et al. 2013). Studies of storm drains in Santa Barbara, San Diego and elsewhere consistently found a lack of correlation between FIB and human markers, confirming the understanding that urban sources of FIB are ubiquitous and typically not controlled by human contamination. At the same time, these and other recent advanced source tracking studies have shown that human fecal contamination within urban stormwater infrastructure is not uncommon; therefore, elevated FIB and detectable human waste may be two persistent but separate issues that urban MS4 permittees may need to address.

Figure 5-5. Example Use of GIS in Microbial Source Tracking Studies
(Source: Weber et al. 2013)



5.3.3 Marine Beach Sanitary Surveys

In 2013, EPA issued *Marine Beach Sanitary Survey User's Manual*, which provides specific guidance for conducting sanitary surveys in beach settings (see http://water.epa.gov/type/oceb/beaches/upload/Marine-Beach-Sanitary-Survey-User-Manual-March-2013.pdf). The purpose of the guidance is to help beach managers in coastal states identify and synthesize beach and watershed information—including water quality data, pollutant source data and land use data—so they can improve water quality for swimming. The intent is to give beach managers a technically sound and consistent approach to identify pollution sources and to share information (EPA 2013). The guidance includes two types of beach sanitary surveys: 1) the Routine On-site Sanitary Survey and 2) the Annual Sanitary Survey. Appendices to the guidance include field and inspection forms help document the information collected.



Types of information requested for routine on-site sanitary surveys include:

- General beach conditions (air temperature, rainfall, wind speed and direction, sky conditions, wave height and intensity, tidal phase, alongshore current speed and direction).
- Water quality (FIB, water temperature, odor, turbidity, salinity, conductivity, dissolved oxygen, total suspended solids).
- Bather load (numbers of people on the beach and in the water).
- Potential pollution sources (visible sources, tidal pools, floatables, algae, dead birds and fish, dogs, wildlife, debris/litter, etc.).

Types of information requested for annual sanitary surveys include:

- Basic information
- Description of land use in the watershed
- Weather conditions and physical characteristics
- Beach dimensions
- Bather load
- Beach cleaning
- Water quality sampling
- Modeling and other studies
- Advisories/closings
- Potential pollution sources
- Description of sanitary facilities and other facilities
- Description of other facilities

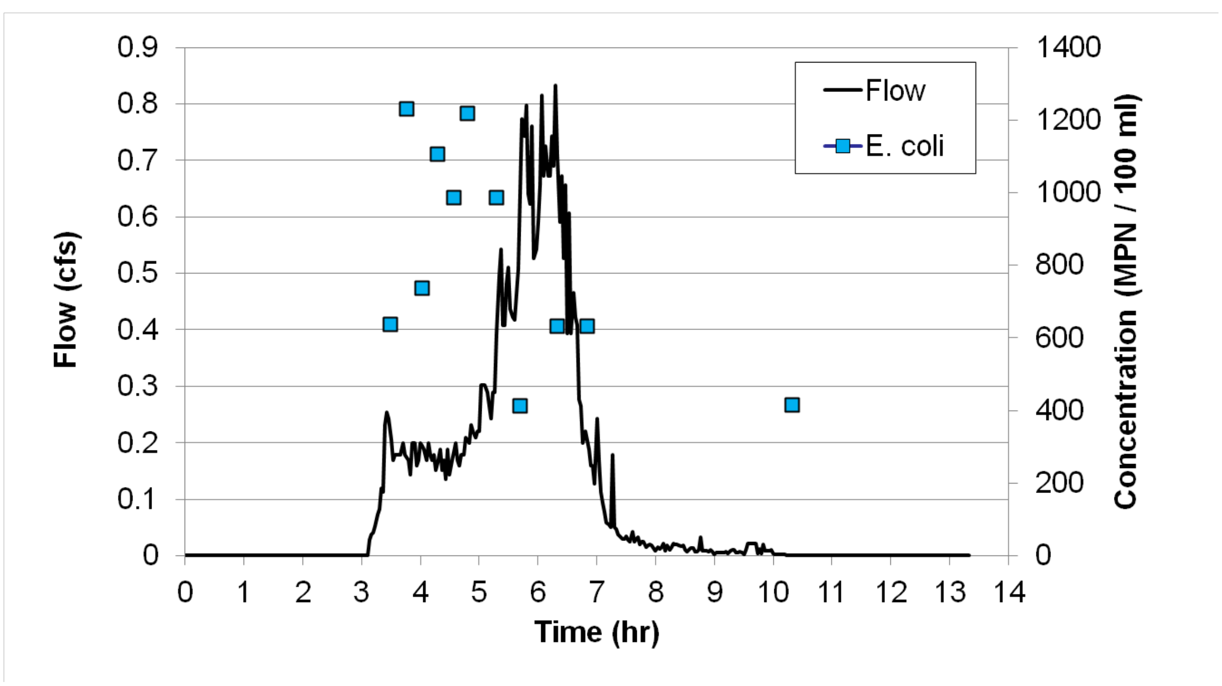
5.4 Wet Weather Runoff Sampling

Wet weather monitoring is typically more costly and more complex than dry weather monitoring. It is used to establish the general bacteria load of the runoff and receiving waters under a variety of conditions, but is limited in its ability to identify bacteria sources. Given that FIB are known to be elevated in urban runoff, many communities choose to focus on understanding dry weather sources first. Two approaches to wet weather monitoring are described below using grab sample techniques and automated monitoring.

5.4.1 Grab Sample Techniques (Applicable to Wet and Dry Weather)

Water quality sampling for pathogens and/or FIB is a meticulous process requiring detailed attention to the sterility of sampling equipment, proper storage of samples, and prompt transport to a laboratory for analysis. Because pathogens are living organisms capable of growth and decay (e.g., cell death, loss of culturability), sample concentrations may vary more than desired if the proper conditions are not maintained during this process. The standard and most recognized monitoring techniques for pathogens typically rely on grab sampling of the water being analyzed (Burton and Pitt 2002). It should be noted that grab samples are generally considered to be a discrete "snapshot" of water quality conditions, and may not accurately depict the substantial variability inherent to FIB concentrations in urban storm drain systems and surface receiving waters, particularly during storm events where concentrations typically vary over orders of magnitude (Figure 5-6). However, contamination is most easily avoided, and sample preservation time most easily assured, when using this strategy.

Figure 5-6. Example *E. coli* Pollutagraph
(Source: Hathaway 2010)



Under a grab sample methodology, bottles can remain under sterile, sealed conditions until just prior to a sample being collected, minimizing the potential for contamination. Pre-sterilized bottles can be purchased, or reusable bottles can be autoclaved at 121^{o}C at 1 atm for 15 minutes and capped (EPA 2012). Care must be taken not to touch the inside of the bottle or bottle cap to avoid contamination. If chlorine is expected in the sample (e.g., drinking water treatment residual), inactivation during transport is a concern. To address this, a small amount of sodium thiosulfate may be added to the sample to neutralize the chlorine residual. Special bacterial sampling bottles can be supplied pre-sterilized, with and without sodium thiosulfate, and are highly recommended.

During transport from the field, microbial samples are placed on ice, and then transferred to a refrigerator at 4^{o}C upon arrival at the laboratory. Hold times vary based on the intent of the sampling data. When samples are being collected to assess the public health impacts of a given source (i.e., recreational waterbody monitoring), samples should be analyzed as soon as possible because up to 24 hours may be required for incubation to obtain results (Noble and Weisberg 2005), which may delay regulatory action (such as beach closure). APHA (2012) specifies that non-potable water for compliance purposes should be analyzed within 6 hours of collection, while a 24 hour hold time is acceptable if other applications for microbial data are intended. For instance, many academic studies performed at nationally recognized public health institutions allow a hold time of 24 hours (e.g., Krometis et al. 2009, Characklis et al. 2005).

Sample Hold Times

Various studies have analyzed the effect of sample hold time on microbial concentrations. Selvakumar et al. (2004) analyzed microbial samples on day 1 of collection and daily thereafter for up to 8 days. Of particular interest in this study was the difference noted between indicator bacteria on day 1 of collection and the day after collection (day 2). For fecal coliform and total coliforms, no significant difference between concentrations was noted from day 1 to 2. However, significant differences were noted for *E. coli* and fecal streptococcus between days 1 and 2. Selvakumar et al. (2004) also analyzed the changes in fecal coliform over a 28 hour period, comparing concentrations determined hourly from 0 to 7 hours to concentrations from 24 to 28 hours. No significant difference in fecal coliform concentration was identified between these two time periods. A similar analysis of the impact of hold times on microbial concentrations was performed by Pope et al. (2003) on samples tested for *E. coli* from 0 to 48 hours after collection. At the majority of the 24 study sites utilized in the study, sample concentrations did not significantly change between 0 and 48 hours. Pope et al. (2003) concluded that *E. coli* samples held below 10^{o}C and not frozen can yield similar results beyond a hold time of 8 hours; however, the authors also conclude that analysis of samples as soon as possible after collection is most desirable. Similarly, McCarthy et al. (2008) investigated the influence of 0 to 24 hours of storage in autosamplers in the field (i.e., hold time) on *E. coli* concentrations for six watersheds in Melbourne, Australia. The authors found no overall significant difference between samples held for up to 24 hours, although a decay trend was noted for two of the sites. The results of these studies suggest that while analyzing microbial concentrations as soon as possible is desirable, hold times of 24 hours are likely acceptable. Beyond 24 hours of storage, differences in concentration may be noted for some indicator bacteria (Selvakumar et al. 2004).

5.4.2 Monitoring With Automated Samplers

Though useful, single grab samples do not capture the inevitable variability in microbial concentrations associated with changes in flow. Although manual collection of multiple grab samples over the course of a storm event is possible, this process is time consuming and limits the number of flow events that can be reasonably and economically sampled (Harmel et al. 2003, Noble et al. 2006, Krometis et al. 2007). Automatic samplers can capture these changes in concentration with flow, while minimizing associated sampling labor time and costs. The obvious challenge when utilizing automatic samples is the need to maintain sterility in sampling equipment. Studies such as Hathaway et al. (2010) have attempted to minimize the risk of contamination when utilizing automatic samples by utilizing autoclaved pie-shaped bottles in a refrigerated sampler and removing, autoclaving, and replacing sampler, pump, and distribution tubing in between each captured storm event.

Other studies have attempted to analyze the influence of automatic samplers on pathogen and FIB concentrations. In an analysis by Line et al. (2008), traditional automatic samplers were utilized to collect samples from multiple watersheds. Line et al. (2008) explored potential contamination of fecal coliform at one of the sampling stations by running distilled (sterile) water through the sampler on two occasions. The resulting samples had minimal contamination, averaging 12 MPN/100 mL of fecal coliform. A study quantifying the quality of water from an agriculturally influenced groundwater source by Boyer and Kuczynska (2003) showed similar results. Distilled water run through the sampling equipment showed no *C. parvum oocysts* and average fecal coliform concentrations of 9.7 cfu/100 mL. The tubing in the Boyer and Kuczynska (2003) study was periodically replaced with new, sterile tubing. Boyer and Kuczynska (2003) also collected grab samples and automatic sampler samples simultaneously during site visits. These samples showed no significant difference.

Despite these studies suggesting minimal bacterial contamination in automatic samplers, unpublished data by Hathaway et al. (2010) suggests contamination may occur in active stormwater BMP inlet and outlet monitoring stations. Hathaway et al. (2014) collected samples of distilled water through 10 active monitoring stations (with no rinse cycle) as part of field equipment blank QA/QC procedures. Fecal coliform concentrations ranged from 15 to >2400 MPN/100 mL, while *E. coli* concentrations ranged from < 1 to 627 MPN/100 mL. Thus, contamination of sampling units may occur; however, further study is needed to determine to what degree such contamination exists, and which sampling protocols and monitoring setups minimize contamination. It is also critical to note that monitoring objectives will determine the seriousness of contamination potential. For example, if the water is heavily contaminated and the monitoring program is designed to support later modeling efforts, the effect of slight overestimates on long-term watershed management decisions may be negligible. However, in other applications this contamination may result in false positives that significantly affect regulatory action.

In summary, due to the time-intensive nature of wet weather sampling, automated equipment is increasingly being used for microbial wet weather sampling. Monitoring installations have been shown to have limited contamination of FIB in some cases, and relatively moderate contamination in other cases. Regardless, common quality assurance/quality controls should be considered when utilizing automatic samplers. These controls include using autoclaved sample bottles, making sure all tubing can drain via gravity flow in between sample collection, ensuring tubing is never “looped” from the sampler to the source, and setting programming to include rinses prior to a sample being collected. Because the extent to which contamination occurs in autosamplers is not well defined, all pathogen/FIB samples that are collected to inform short-term regulatory decisions directly applicable to immediate decisions (e.g., beach closures) should be made using grab samples and analyzed as quickly as possible using available short incubation periods.

5.5 Conventional Culture-Based Analytical Methods

Pathogens and FIB are different from chemical pollutants because they are living, or viable, organisms, capable of surviving and multiplying under the right conditions. While some transport phenomena are similar among pathogens and chemicals (e.g., decay due to degradation or attachment to suspended particles), there is one phenomenon that is unique to pathogens: the ability to grow, or multiply, under certain conditions. This phenomenon is what makes pathogens and FIB quantifiable through laboratory detection techniques that are culture-based. Culture-based methods rely on the ability of existing FIB to grow when adequate nutrients are provided and such physical/chemical factors as temperature and pH maintained at an optimum. The most common methods of measuring FIB are culture-based, but significant progress has been made for molecular methods (see Section 5.6) in recent years. An overview of the traditional culture-based methods follows.

Culture-based measurement methods can be divided into three categories: multiple-tube fermentation (MTF), membrane filtration (MF), and defined-substrate (DS), which included chromogenic and fluorogenic substrates. Other references, such as *Standard Methods for the Examination of Water and Wastewater* (APHA 2012) and Csuros and Csuros (1999), provide detailed procedures on how to conduct the tests. The purpose of this section is to generally explain the concepts of each method for non-laboratory analysts. For more detailed information, see *Standard Methods for the Examination of Water and Wastewater*.

5.5.1 Multiple Tube Fermentation (Most Probable Number)

Most probable number (MPN) techniques provide a method for the statistical estimation of the number of microorganisms within a sample based upon the presence of at least one viable cell in increasingly diluted aliquots of the original sample. The technique is based upon the concept of dilution to extinction. Simply put, a sample is diluted by orders of magnitude (typically 10X) and each of the dilutions is assayed for the presence of the microorganism. When organisms are no longer detected in the dilution, the minimum number of organisms present in the original sample can be calculated. For example if the original sample contained 110 microorganisms, there should be at least one viable cell in both a 10-fold and 100-fold dilution of the original sample. The likelihood of a viable cell in a 1000-fold dilution, however, approaches zero. If multiple replicates are examined for each of the dilutions in the series, it is possible to narrow the estimate of the concentration of organisms in the original sample.

In the MTF method for the enumeration of FIB, multiple test tubes are filled with a sterile liquid culture medium in which the target indicator organism will grow. Each test tube is inoculated with a volume of sample, in decimal dilutions of the original sample. For example, when using five test tubes, the first test tube will contain 9 parts culture medium and 1 part sample, the second test tube will contain 99 parts culture medium and 1 part sample, the third test tube will contain 999 parts culture medium and 1 part sample, and so on, resulting in dilutions of 0.1, 0.01, 0.001, etc. In practice, the dilution range is bracketed by the anticipated concentration of the specific microorganism. The tubes are shaken and incubated for a period dependent on the target organism. After the incubation period, the tubes are checked for gas formation or other reactions according to the target organism. Reactions are recorded along with the dilution and are compared with a most probable number (MPN) table to obtain an organism concentration of MPN/100 mL, within 95% confidence limits.

MTF methods for coliforms often include a presumptive and a confirmed test. The presumptive test preliminarily separates positive and negative results, while the confirmed test confirms the positive results. Usually a third test, the completed test, is also performed. It may require several days to complete this test series through confirmation.

5.5.2 Membrane Filtration

The membrane filtration (MF) method uses the concept that microorganisms (bacteria of interest are about 2 to 5 micrometers in size) can be filtered out of a water sample and counted as colony-forming units (cfu) after incubation on an appropriate solidified growth medium. This method is preferred over MTF because it is faster and uses fewer laboratory supplies. To use the method, the analyst passes a known volume of sample, diluted or undiluted, through a 0.45-µm membrane filter. Bacteria and larger microorganisms will be retained on the membrane filter. The filter is then placed into a petri dish containing a solid growth medium and incubated for an established period of time. After incubation, each bacterium originally on the filter will have developed into a colony large enough to be visible and counted. (However, this assumption is often not valid.) Upon counting the colonies, the result is expressed in cfu/100 mL.

The MF method assumes that each colony formed on the media is the result of a single cell; this assumption, however, is rarely met in natural samples where clumping and attachment of microbial cells to particles (silt) are common. Thus, the MF method provides an estimate of the number of microorganisms present in a sample. In addition, the method will not work if the filter retains other substances from the water samples causing clogging of the filter or causing interference of microbial growth on the solid medium.

5.5.3 Chromogenic Substrate (IDEXX)

The chromogenic substrate or defined-substrate method is newer and considered more convenient than MTF and MF. This method can enumerate total coliform, *E. coli*, and enterococci. The nutrient powder to which the water samples are mixed is proprietary, and the methods are currently marketed by the company IDEXX Laboratories, under the names Colilert® (for the detection of total coliform and *E. coli*) and Enterolert® (for the detection of enterococci).

The method involves mixing the proprietary powder, which contains an indicator-nutrient, with the water sample, or the diluted water sample. The indicator-nutrients for detection of total coliform and *E. coli* are *ortho*-nitrophenyl-ß-D-galactopyranoside (ONGP) and 4-methylumbelliferyl-ß-D-glucuronide (MUG) (Edberg and Edberg 1988). ONGP and MUG are metabolized by enzymes in coliforms and *E. coli*. A similar process occurs for the detection of enterococci, in which a patented indicator-nutrient powder is mixed with a water sample. To quantify the FIB, the mixture of water sample and powder is placed in a Quanti-Tray® (a clear plastic slotted tray measuring approximately 6” x 11”), sealed, and incubated. The Quanti-Tray® contains slots, or wells, in which a color will develop if the target bacteria are present. A count of the wells is compared to a chart, and the count is converted to a bacteria concentration in units of MPN/100 mL, based on a similar statistical basis as the multiple tube method. A comparison study among chromogenic substrate, MTF, and MF tests in storm-affected coastal waters concluded that results from the three tests were within an acceptable 90% agreement (Noble et al. 2004). It should be noted that the defined-substrate method is capable of detecting injured (or viable, but nonculturable [VBNC]) bacteria, while the MTF and MF methods are not (Edberg et al. 1988).

5.5.4 Limitations of Traditional Methods

The traditional methods summarized above have a number of limitations that have led to significant research over the last few years to refine and develop new methods. Some of the limitations of traditional methods include:

- Inability to differentiate sources of FIB.
- Inability to detect viable but not culturable conditions (VBNC) (for some methods).
- Lack of real-time results (which is a consideration for beach closure notifications).

One condition that culture-based methods do not usually account for is the viable but non-culturable state (VBNC). VBNC is the term given to bacteria that are alive but incapacitated. Bacteria in the VBNC state are unable to reproduce and cannot be detected by traditional bacterial culture methods. Bacteria in the VBNC state, though alive, have been shown to be incapable of causing infection in humans. However, bacteria in the VBNC state can return to a viable state, a condition known as resuscitation, and regain infectious ability (McDougald et al. 1998). Detailed reviews of the VBNC state in bacteria are provided in Oliver (2005) and McDougald et al. (1998).

An example of FIB in the VBNC state affecting surface water quality in the environment is provided in a study by Bolster et al. (2005). In this study, *E. coli* in water samples exposed to chlorine (a simulation of disinfected wastewater) were presumed dead after analysis using culture-based methods. However, *E. coli* regained culturability after the water sample was mixed with estuarine waters. Therefore, *E. coli* had reached a VBNC state after exposure to chlorine and regained culturability after contact with the surface water environment. Thus, the VBNC state can potentially complicate water quality assessments when bacteria regain culturability.

Newer, molecular-based methods are able to detect bacteria in the VBNC state because they detect bacteria DNA. However, this can also be a limitation of molecular methods because no determination can be made between dead and living bacteria. These methods are based on polymerase chain reaction (PCR) and are described below.

5.6 Molecular Methods

Deoxyribonucleic acid (DNA) is the double stranded, helix-shaped molecule that carries the genetic code of a cell. Polymerase chain reaction (PCR) is a molecular method that amplifies specific DNA in a sample in order to detect it. In PCR, synthetic DNA primers complementary to the DNA of the target cell are introduced into a sample that has been treated to extract (release) the DNA from the organisms in the sample. Samples are added to small tubes containing the primers and reaction components needed for synthesis of new DNA and placed inside a thermal cycler instrument, in which the temperature changes cyclically to promote a chain reaction in which the DNA primers start the replication of specific DNA, up to millions of times in about two hours. After the reaction time is over, the specific DNA, if present in the original sample, can be detected using gel electrophoresis. More advanced versions of PCR can quantify the amount of target DNA in a water sample. Termed qPCR or real-time PCR, some of these methods incorporate a secondary DNA probe that adds additional specificity to the assay. Several other variations of PCR exist, depending on the target organism.

As noted in Section 5.5, PCR detects cells that are alive, dead, or injured (VBNC) because the DNA in these cells can still be amplified (McDougald et al. 1998). As a result, PCR may indicate the presence of viable bacteria when none are actually present (false positive results). For example, if instream water samples are collected below a disinfected WWTP discharge or in a storm drain influenced by reclaimed water used for irrigation, then “dead” DNA in treated wastewater will likely be detected, which may complicate source identification using PCR in this type of setting.

For purposes of this report, three different contexts for molecular methods are of interest. The first relates to an EPA-approved alternative rapid method for enterococcus ssp. that can be used as an alternative approach to culture-based methods in a regulatory context. The second context is for purposes of source identification, or microbial source tracking (MST). The third is an extension of molecular methods for microbial community analysis (MCA), which is an emerging methodology, as discussed in Section 5.6.3.

5.6.1 EPA Method 1611 for Enterococcus (qPCR)

As part of the 2012 RWQC, EPA approved a molecular method for enterococcus spp. as measured by qPCR (EPA Method 1611), which can detect and quantify enterococci more rapidly than the culture methods. Benefits of this method are that it provides more timely information related to beach closures; however, EPA also recognized that there is a potential for qPCR inhibition (interference) in some waterbodies. Thus, EPA encourages a site-specific analysis of the method’s performance prior to use in a beach notification program or adoption of water quality standards based on the method (EPA 2012). This method is not currently suggested for NPDES permitting or effluent-related monitoring purposes because it detects and enumerates both live and dead enterococci.

5.6.2 Microbial Source Tracking (MST) Molecular Methods

Microbial source tracking using molecular methods is an evolving area of practice. Harwood et al. (2014) provide a review of MST markers for detection of fecal contamination in environmental waters, focusing on the relationships between pathogens and human health outcomes. They note three general areas of challenges related to method selection to identify various genetic markers, including:

1) intrinsic method performance, for example, sensitivity and specificity toward sewage and fecal samples, including geographic range,

2) method performance in the field (sensitivity toward dilute samples, effects of PCR inhibition, and the efficiency of DNA recovery from environmental matrices), and

3) knowledge about ecology of the organisms and persistence of the markers in the environment, and correlation with FIB and pathogens in environmental waters.

Because each potential method has particular strengths and weaknesses, Harwood et al. (2014) recommend the use of multiple methods to identify a particular source. For example, *Bacteroides* HF183, is not completely specific for human waste; however, it has the advantages of being broadly distributed among human populations and of a relatively high concentration in sewage. The more strongly human-associated microorganisms, such as pathogenic viruses, are less concentrated in sewage and are therefore difficult to detect in dilute samples. Analysis for viruses, however, may be useful for verifying findings from other methods.

Harwood et al. (2014) also note that although markers exist for domestic animals such as cattle, poultry, horses, pigs, and dogs, the distribution and performance of animal markers are not as well understood. Markers have not been developed for many domestic and wild animals that can be important contributors to fecal loading in surface waters.

A key resource for those considering use of molecular methods in MST studies is *The California Microbial Source Identification Manual: A Tiered Approach to Identifying Fecal Pollution Sources to Beaches* (Griffith et al. 2013). This manual provides a general framework for MST studies, but also provides an appendix of standard operating procedures for key microbial tests, as summarized in Table 5-7.

Griffith et al. (2013) recommend using human-associated bacterial source markers (e.g., HF183, HumM2) first because of generally high sensitivity and specificity, and because of their relatively low cost and ease of use relative to other molecular markers. When considering use of various methods, there may be circumstances where verification of resulting using these methods is desirable, such as when cross-reactivity of human-associated bacterial markers has been detected or when the cost of mitigation is high enough to warrant additional verification about the presence of human fecal material, particularly municipal sewage. Measurement of human viruses may be considered in those cases; however, there are some challenges associated with measuring human viruses. Because viruses occur at very low densities in the environment and are difficult to concentrate efficiently with current technology, very large sample volumes (e.g., up to 1000 times the typical 100 mL sample) and sample concentration techniques are required (e.g., water filtering in the field), which dramatically increase the cost of sampling and analysis. Specialized research laboratories are also currently required to run these analyses. Nonetheless, standard operating procedures for human adenovirus and human polyomavirus are provided in Griffith et al. (2013), since they are among the more sensitive and robust markers.

After human sources have been ruled out as a dominant source of elevated FIB, MST studies can progress to identification of certain non-human sources. Griffith et al. (2013) report that although source-associated markers are only available for a limited number of species, many of these markers have been shown to be both sensitive and specific for their targets. Reliable source-associated fecal markers are currently available for cattle, dogs, pigs and horses. More general markers that target ruminants and waterfowl are also available. Although samples can be run for multiple markers, care should be taken to run multiple markers only when host sources are numerous enough in the watershed to warrant the added effort and expense. Given limited budgets, it is important to balance the need to collect an adequate number of samples with gaining information about an additional source (Griffith et al. 2013).

When considering microbial analysis, there are a number of practical and logistical considerations to keep in mind prior to embarking on a study. A few examples based on experiences of Weber et al. (2013) include:

- Multiple labs may be required for multiple methods (e.g., chemistry, microbiology and molecular biology laboratories).
- Finding experienced laboratories to conduct the analyses. For example:
    - Most host marker methods involve filtering water samples onto membranes. Some labs do not perform membrane filtration.
    - HumM2 – few laboratories offer this test because of the requirement to purchase a patent license, which is expensive unless routine testing is being conducted.
    - Changes in laboratory equipment, staffing or reagents may result in unexpected changes in method performance (Harwood and Stoeckel 2011).
- Testing Costs: Representative costs of analyses by commercial labs are on the order of several hundred dollars per test per sample, with decreasing cost per analysis when multiple analyses are conducted on a single sample (Source Molecular 2013). Although the costs of advanced methods are higher than traditional water quality constituents, having definitive answers regarding whether human sanitary sources are present can help focus investigations and corrective actions.
- Commercial vs. Research Labs and Laboratory Location: Depending on the study location, research labs may be equipped to conduct analyses; otherwise, samples may need to be shipped to commercial laboratories. Decisions related to shipping water samples vs. archiving MST sample filters (freezing at -80 C) may need to be made.

Table 5-7. Summary of Standard Operating Procedures for MST Marker Methods
(Table developed based on information in Griffith et al. [2013])

MST Marker Method[1]	Description
Human Markers with SOPs	
HF183 Taqman qPCR	Targets *Bacteroides* bacteria in human fecal material. Performed best in method evaluation studies. Recommended by Griffith et al. (2013) as best starting point for detecting human fecal material. However, it has been shown to occasionally cross-react with chicken or dog feces. If those sources are of concern, then it is recommended that HF183 be paired with HumM2.
HumM2 qPCR	Targets *Bacteroides* bacteria in human fecal material. Slightly less sensitive than HF183.
Human Adenovirus qPCR	Targets human adenovirus. Can be used on an as-needed basis to supplement and verify bacterial marker results. More costly and requires more specialized laboratory expertise than the bacterial qPCR methods.
Human Polyomavirus qPCR	Targets human Polyomavirus. Can be used on an as-needed basis to supplement and verify bacterial marker results. More costly and requires more specialized laboratory expertise than the bacterial qPCR methods.
Non-Human Markers with SOPs	
BacCan-UCD qPCR and DogBact qPCR	Targets dog-related fecal sources. Both methods were found to be highly sensitive and specific, though occasional cross reactivity with other species has been observed. Equally recommended by Griffith et al. (2013).
CowM2 qPCR	CowM2 is the recommended marker for cattle because it is expected to become an EPA-approved method.
Rum2Bac qPCR	Recommended for non-bovine ruminants. When both cattle and other ruminants are present in the watershed, then both CowM2 and Rum2Bac are recommended. Rum2Bac occasionally had false positive results with septage, so users should conclusively rule out septage before employing Rum2Bac.
Pig2Bac qPCR	Pig2Bac is the recommended method for detection of pig feces. It may cross-react with human/septage and dog feces, so it is best applied when those sources have been ruled out.
Horse Conventional PCR	This method is recommended when horses are present and other sources have been ruled out. It is not as sensitive as most other host associated assays. This method is not quantitative.
Gull2 Taqman qPCR and Lee Seagull qPCR	Four gull markers were evaluated, with Gull2 Taqman and Lee Seagull markers recommended due to sensitivity and specificity. Bird markers will amplify pigeon and sometimes goose feces, as well as gull. Considered general bird assays and not necessarily specific to gulls. Other new assays may also be available.

[1]SOPs are also provided for procedures including membrane filtration for molecular analysis, DNA EZ ST1 Extraction (GeneRite, LLC), Sketa (Sample Processing Control) qPCR.

Los Angeles River CREST
Bacterial Source Investigation Study

The main stem of the Los Angeles River is approximately 55 miles long and flows through a mostly concrete channel in the Los Angeles area, as do many of its tributaries (CRWQCB 2010). The Los Angeles River Bacteria Source Identification Study was developed by the Cleaner Rivers through Effective Stakeholder-led TMDLs (CREST) and funded by the City of Los Angeles. The overall goal of the BSI Study was to increase the accuracy of the forthcoming LA River Bacteria TMDL and improve the likelihood of success for source control efforts associated with TMDL implementation. This study compiled one of the largest MST datasets known to exist for urban runoff. Over 600 samples were collected and analyzed for traditional FIB (*E. coli* and Enterococcus), *Bacteroidales* (real-time PCR for universal and human), adenovirus and enterovirus (real-time PCR), and flow and other general chemistry parameters.

The study sought to identify which storm drains and/or tributaries were contributing the highest FIB loads and to determine whether the storm drain and tributaries were responsible for significant bacteria loads entering certain reaches and causing standard exceedances. Additionally, the study was designed to determine whether human or non-human sources were responsible for the significant bacteria loads entering certain reaches of the LA River, to assess how human and non-human loading from storm drains and tributaries compared to loading to certain reaches.

CREST found that approximately 85% of the storm drain samples collected exceeded the *E. coli* objective. If human *Bacteroidales* was high in runoff, then *E. coli* was also high, but not vice versa. In the reaches investigated, *E. coli* loading from storm drains and tributaries greatly exceeded the allowable instream loading. The study also found that some of the loading for one of the target reaches (Reach 2) could not be attributed to the measured storm drain inputs. Specifically, only about 10-50% of the FIB measured in Reach 2 of the Los Angeles River during six dry weather sampling events originated from storm drains and tributaries. Another important finding was that the largest dry weather *E. coli* loading increase instream occurred along the downstream portion of Reach 2, whereas a majority of the storm drain loadings occurred along the upstream portion of this reach. The *E. coli* concentrations in this reach increased in a downstream direction by more than an order of magnitude, while human-specific *Bacteroidales* concentrations did not (CREST 2008).

5.6.3 Microbial Community Analysis

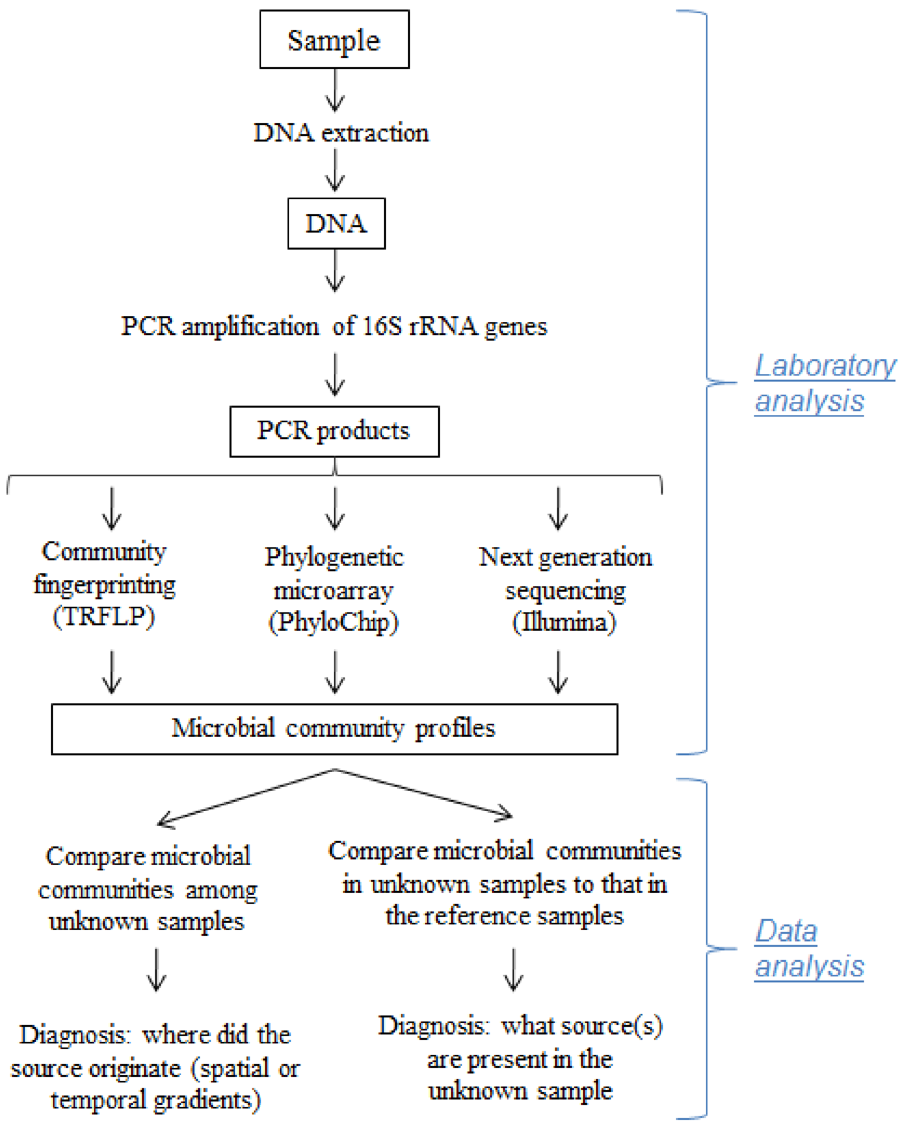
An emerging approach that relies on molecular methods is microbial community analysis (MCA) methods, which can be used where simpler methods are inconclusive. MCA is an evolving practice area, so is only briefly introduced in this report. Griffith et al. (2013) summarize the major steps of MCA in Figure 5-7 and define MCA as follows:

> MCA is a set of tools that provides information about the entire microbial community simultaneously in a sample. While the single marker methods identify a source via detection of a single marker associated with that source, MCA compares the entire microbial community between samples and suspected sources. MCA may help to determine the type of source (i.e. which host) in a library-dependent fashion where environmental samples are compared to local reference samples. MCA can also determine the location of a source in a library-independent fashion by comparing profiles among environmental samples to discern spatial and temporal gradients.

The science supporting MCA is still evolving, but it is expected to be useful in complex situations where costs of TMDL implementation are high. Griffith et al. (2013) should be referenced for more information and provide these examples of situations where MCA may be useful:

- When there is a suspected fecal source for which single-source markers have not yet been developed.
- When a host-associated marker is shared across sources which need to be distinguished for management purposes (e.g., homeless waste vs. sewage).
- When further confirmation is needed to determine if non-fecal sources are major contributors of microbial contamination.
- When further evidence is needed to identify spatial gradients of fecal pollution.

Figure 5-7. Microbial Community Analysis Steps
(Source: Griffith et al. 2013)



5.6.4 Limitations of Molecular Methods

Just as traditional methods have limitations, molecular methods also have limitations. Some of these limitations are associated with the evolving state of the practice and may become less significant in the future. Griffith et al. (2013) provide a good discussion of these issues, with brief highlights including:

- **False Negatives**: The biggest limitation of current source tracking technologies is the possibility of false negative results; that is, the absence of detection when the target is actually present. Factors affecting false negatives generally include inadequate sample numbers, high detection limits, target degradation (i.e., decay or aging) and qPCR inhibition (interference). Many common types of compounds in ambient waters can cause inhibition, such as large organic acids, carbohydrates and metal cations.

- **False Positives**: In rare circumstances, qPCR assays can produce false positive results. False positives result from cross-reactivity, which occurs when a positive result is obtained for a host-associated microbe in a non-target host (e.g., human-associated bacteria present in gull feces). Use of a second confirmatory marker is recommended when one or more of these potential cross-reaction sources are present in the watershed. Cross-reactivity can also occur in cases when animals are present in close association with humans (e.g., pets, gulls feeding at landfills with biosolids). If this kind of site-specific cross-reactivity is suspected, fecal samples from the animals present in the watershed should be collected and tested for the presence of host-associated markers.

- **Source Apportionment**: The ideal outcome of an MST study would be a probability graph indicating relative contributions of FIB from various host sources (with clearly shown uncertainties), rather than just documenting presence/absence, but the technology for this type of source allocation using molecular methods does not exist at present. Ongoing investigations related to methods appropriate for source apportionment are an area in need of continued research.

- **Source Resolution**: Although a number of strong markers exist, methods have not been developed for many common animals, which may limit identification of sources in certain settings. Continued funding of research to develop and verify these markers is needed.

5.7 Monitoring to Support Quantitative Microbial Risk Assessment

As discussed in Chapter 2, quantitative microbial risk assessment (QMRA) is a tool now allowed by EPA to establish site-specific recreational water quality standards, provided that the alternative limits protect human health at levels consistent with EPA's 2012 RWQC. Figure 5-8 provides an overview of the basic steps involved in QMRA. (For more detail, see http://water.epa.gov/scitech/swguidance/standards/criteria/health/recreation/upload/P4-QMRA-508.pdf.) QMRA is generally considered a potentially useful approach in moderately urbanized watersheds where significant compliance efforts have already been implemented and where initial source tracking results demonstrate an absence of or a low percentage of human source contribution (e.g., <10 percent). Streams that are relatively close to meeting TMDL WLAs and underlying EPA RWQC and which have substantially invested in controlling anthropogenic sources are considered potentially good candidates for QMRA.

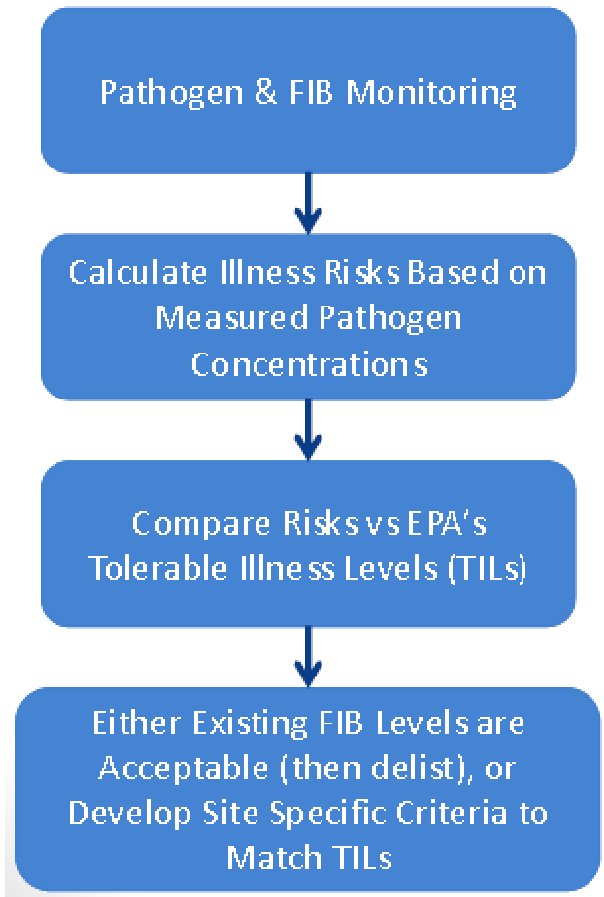
The general premise of QMRA is based on concepts of equivalent risk and the fact that risk varies based on sources of FIB. If the sources of FIB are relatively low risk, then a higher (less restrictive) water quality standard for FIB can be implemented while still protecting human health. Risk is based on exposure and potency. Exposure includes concentration of pathogens and ingestion rate, whereas potency is based on documented dose-response rates of illness in published literature. Simply described, the key steps involve:

1) Monitoring for <u>both</u> FIB and pathogens to develop a data set suitable for conducting QMRA.

2) Calculate expected illness rates associated with measured pathogen concentrations using QMRA methods.

3) Compare calculated risk to EPA's tolerable illness levels (TILs).

Figure 5-8. Overview of QMRA Steps
(Source: Steets 2013)



For examples of QMRA studies, see the Chicago Waterways study (Rijal et al. 2011, Petropolou et al. 2008), Soller et al. (2010b), Schoen (2010), and Ashbolt et al. (2010). EPA is currently in the process of developing technical support materials for use by communities considering QMRA studies.

Communities considering QMRA should be aware that human sources of FIB can be an unanticipated finding that causes a QMRA study to be abandoned, at least until those human sources are corrected. There are multiple examples of sanitary surveys where human markers have been found in storm sewer systems. For example, Sercu (2009, 2011) identified leaking sanitary sewer lines into storm drains as sources of human markers in Santa Barbara, CA. A study by Sauer (2011) identified human markers in at least one sample collected at all 45 outfalls monitored in a Milwaukee study. In Los Angeles, human markers were detected in half of Los Angeles River dry weather storm drain samples (CREST 2008). At Redondo Beach, CA, Slifko et al. (2009) also detected human markers at beach outfalls.

For communities considering QMRA, a potential cost range for a single stream reach or beach site could be on the order of $150,000 to $400,000. Costs would vary depending on the number of sample locations included and whether the study addressed both wet and dry weather conditions. The cost of the Chicago Waterways study, which had a number of complex aspects, was on the order of $1.1 million and included both wet and dry weather conditions. A recent QMRA study begun by EPA in Ventura County, California, which was ultimately abandoned due to evidence of human sources, included costs of approximately $750,000 for a dry weather QMRA at two beaches. This study was likely on the higher end of costs due to a variety of cutting edge components. Additional costs for "first" studies in various states may include an independent expert panel to review results and additional costs related to source investigations.

5.8 Targeted Hypothesis Testing Using Microcosm Techniques

Microcosm experiments are another experimental technique that has been used successfully to improve understanding of factors leading to elevated and persistent FIB. In some cases, unexplained patterns in FIB conditions instream persist, despite no obvious source. Microcosm studies are a tool that can be implemented to further understand and explain FIB concentrations in certain waterbodies. A microcosm is a small-scale reproduction of a real system. In such studies, water samples are collected from the various water sources discharging to the waterbody, mixed in the laboratory to mimic the real mixing scenarios and real temperatures of the water sources, and the mixing combinations (i.e., the microcosms) are monitored in the dark in the lab to assess the rate that FIB concentrations increase (or decrease) over time. The time during which the microcosms are monitored should be at least the estimated time of travel of water in the relevant length of the waterbody in question.

Examples of more detailed questions that can be answered by microcosm studies are:

1) Does a WWTP discharge nutrients into the waterbody that stimulate proliferation of FIB in the base flow of the creek and in the discharge?

2) Does a WWTP discharge injured FIB that are not detectable by current methods and "resuscitate" after discharged to the waterbody?

3) Are nutrients, macroalgae, and sediments playing a role in the increase or decrease of FIB concentrations in this system?

As an example, microcosm studies can be conducted wherein water samples are collected from an impaired river, a discharging WWTP, and a storm drain, following procedures described in Surbeck et al. (2010) and summarized here. A portion of these source waters is analyzed for FIB (e.g., total coliforms, *E. coli* and enterococci), nutrients (e.g., dissolved organic carbon, nitrate, ammonium, phosphate) and temperature. Samples of macroalgae can also be collected from the field site; macroalgae samples should be transported to the laboratory and possibly speciated. A fraction of the macroalgae can be used to determine moisture content (by weighing before and after drying at 100°C).

Microcosm studies are carried out as follows. Mixtures of the different source waters, before or after filter sterilization, are mixed together with or without macroalgae in various combinations designed to answer a set of targeted questions. These mixtures, or "microcosms," are carried out in a lab at environmentally relevant temperatures determined at the time of source water collection. "Filter sterilization" means that the water has been passed through a 0.2 μm filter, which excludes all bacteria and any particulates greater than 0.2 μm in size.

As an example of the kind of targeted questions that can be answered with this study design, consider the possibility that injured or viable bacteria in a WWTP effluent increase in concentration after exposure to dissolved nutrients present in the receiving water. To explore this possibility, a volume of WWTP effluent is mixed with filter-sterilized receiving water, and the concentration of FIB in the resulting mixture is monitored over a period of time, chosen to coincide with river travel times. The volume ratios of the different source waters used in these microcosm studies should be chosen to mimic realistic mixing scenarios (calculated based on actual measurements of base flow, runoff rates, and WWTP effluent discharge rates).

FIB concentrations in the microcosms are monitored using standard methods (described in Section 5.5).

After a time-series of FIB concentrations is obtained, kinetic growth and decay rate constants are calculated by plotting the natural log of the FIB concentration vs. time and calculating the linear regression. The slope of the linear regression equation is the rate constant. Doubling time and half-life are calculated by dividing the natural log of 2 by the kinetic rate constant.

Microcosm in an Erlenmeyer flask being pipetted for FIB analysis using sterile techniques. (Photo Courtesy of Dr. Cristiane Surbeck, University of Mississippi)

The results of the microcosm experiments are reported in terms of rate constants for the increase (regrowth or resuscitation) or decrease (die-off or transfer from culturable to non-culturable states) of FIB as a function of the nature of source waters (and presence/absence of macroalgae), and nutrient status and water chemistry of the source waters. These nutrient and water quality parameters were selected for measurement based on evidence that they affect, singly or in combination, the survival, resuscitation, and regrowth of FIB in environmental matrices (Bolster et al. 2005, Byappanahalli et al. 2003b).

Relationships between growth or decay of FIB and nutrient concentrations can be determined using statistics. Spearman's rank correlations are run for the microcosm studies between the ratio of the final to the initial concentrations of FIB groups ($\log C_f/\log C_o$), the net rate constants (with positive values for growth and negative values for die-off) and the initial nutrient concentrations.

Examples of specific microcosm studies include the following, where A is a sample from the receiving waterbody, B is a sample of treated wastewater, and C is a sample from a storm drain. The letter "f" following A, B, or C denotes that the source water sample will be filter-sterilized.

1. Control Microcosm Af+Bf+Cf. The goal of this microcosm is to confirm that no FIB are present in the three source waters after filter sterilization.

2. Source Water Microcosm Studies: The goal of these microcosm studies is to determine if bacteria present in a specific source water grow, resuscitate, or die-off after exposure to dissolved nutrients from other source waters.

> 2a. Microcosm A+Bf. To evaluate the growth, death, and/or resuscitation of FIB in base flow after addition of dissolved nutrients (but not bacteria and other particulates) from the WWTP effluent.

2b. Microcosm Af+B. To evaluate the growth, death, and/or resuscitation of FIB in WWTP effluent after addition of dissolved nutrients (but not bacteria and other particulates) from the base flow.

2c. Microcosm A+B+Cf. To evaluate the growth, death, and/or resuscitation of FIB in a mixture of WWTP effluent and base flow, after addition of dissolved nutrients (but not bacteria and other particulates) from the basin dry weather runoff.

2d. Microcosm Af+Bf+C. To evaluate the growth, death, and/or resuscitation of FIB in the basin dry weather runoff after addition of dissolved nutrients (but not bacteria and other particulates) from base flow and WWTP effluent.

3. Macroalgae Microcosm Studies: The goal of these microcosm studies is to determine if the macroalgae present in the receiving water are a significant source of FIB after exposure to three different source waters and combinations thereof. For each microcosm experiment described below, separate sub-experiments can be carried out with macroalgae of different initial moisture contents. As noted above, the volume ratios of different source waters is chosen to be consistent with mixing ratios observed in the real system.

3a. Microcosm Af+Macroalgae. To evaluate the release, growth, death, and/or resuscitation of FIB associated with macroalgae collected from the receiving water, after exposure to water and dissolved nutrients (but not bacteria and other particulates) from the base flow in the receiving water.

3b. Microcosm Af+Bf+Macroalgae. To evaluate the growth, death, and/or resuscitation of FIB associated with macroalgae collected from the receiving water, after exposure to water and dissolved nutrients (but not bacteria and other particulates) from base flow in the receiving water and WWTP effluent.

3c. Microcosm Af+Bf+Cf+Macroalgae. To evaluate the growth, death, and/or resuscitation of FIB associated with macroalgae collected from the receiving water, after exposure to water and dissolved nutrients (but not bacteria and other particulates) from base flow in the receiving water, WWTP effluent, and storm drain.

3d. Microcosm A+Macroalgae. To evaluate the release, growth, death, and/or resuscitation of FIB after mixing macroalgae with unfiltered base flow in the receiving water.

3e. Microcosm A+B+Macroalgae. To evaluate the release, growth, death, and/or resuscitation of FIB after mixing macroalgae with unfiltered base flow in the receiving water and unfiltered WWTP effluent.

3f. Microcosm A+B+C+Macroalgae. To evaluate the release, growth, death, and/or resuscitation of FIB after mixing macroalgae with unfiltered base flow in the receiving water, unfiltered WWTP effluent, and unfiltered runoff from the storm drain.

Laboratory Microcosm Studies of Cucamonga Creek, California
(Surbeck et al. 2010)

A study in southern California was conducted to determine the cause(s) of high FIB concentrations in Cucamonga Creek, a concrete-lined urban stream in San Bernardino County. Most of the flow in the creek consisted of treated wastewater, with no FIB but significant concentrations of nutrients (phosphorus, nitrate, and ammonium) and dissolved organic carbon (DOC). A smaller fraction of the flow in the creek consisted of dry and wet weather urban runoff, with elevated concentrations of FIB but low concentrations of nutrients and DOC.

Downstream of the treated wastewater discharge, FIB concentrations often remained high, indicating little dilution effect from the treated wastewater. A simple mass-balance calculation considering flow rates and FIB concentrations did not match the high measured concentrations, indicating that ecological conditions in the creek may have been preventing FIB from dying off as normally expected.

Laboratory microcosm studies of the Cucamonga Creek water sources revealed that the survival of *E. coli* and enterococci bacteria in runoff was strongly dependent on the concentrations of DOC and phosphorus. Above threshold concentrations of 7 and 0.007 mg/L of DOC and phosphorus, respectively, FIB either grew exponentially or exhibited fluctuating concentrations around a steady-state mean. The results showed that the exponential growth may have been promoted by high nutrients in the water and that the steady-state concentrations were maintained by a combination of nutrient concentrations and the presence of predators of FIB. Below the threshold DOC and phosphorus concentrations, FIB died off exponentially.

This microcosm study showed that dry weather FIB impairment in Cucamonga Creek was controlled by ecological conditions present in the urban creek.

5.9 Data Management

An easily corrected, but common, shortcoming of monitoring programs is lack of systematic data management in a manner that enables future access of study data. Data management protocols should be part of any sampling and analysis plan; however, effective data management is often lacking. Some simple considerations that help to maximize investment in monitoring programs include:

- Systematic naming and structure of electronic files supporting the study.
- Timely review of samples to enable identification and correction of errors or follow-up for unusual results (including explanatory comments when unusual results are observed).
- Developing a standard spreadsheet or database format that all data entries will follow, regardless of the individual conducting data entry. Usually, a database format with column headers such as location, date, analytical parameter, result, qualifier, detection limit, comment are needed, along with other explanatory information. If the collected data will

be used as model inputs, then storage of the data in a format easily uploaded to a model can also be helpful.

- Clear identification and nomenclature for sample locations that carries through various study types, even if different entities are conducting the studies. Changing sample location names from year-to-year causes confusion in data analysis. Include latitude and longitude coordinates for all sampling locations and a narrative name to accompany short location labels (e.g., site 120A is located at 120th Avenue upstream of bridge).

- Ensure staff entering or managing data have clear direction on how to record values above or below quantitation limits.

- Record and store field conditions along with water quality data. These anecdotal observations can be critical components for identifying sources of FIB.

- Measurements of flow and precipitation records should be stored along with water quality data.

- Obtain copies of electronic records with clear description of contents from consultants conducting special studies.

Guidance for data management can be found in many references. Two examples include EPA's Water Quality Exchange (STORET) website http://www.epa.gov/storet/wqx/, and in Chapter 6 Data Management, Validation and Reporting of *Urban Stormwater BMP Performance Monitoring* (http://www.bmpdatabase.org/monitoring-guidance.html, Geosyntec and Wright Water Engineers 2009).

5.10 Conclusions

A variety of monitoring approaches are available to help identify potential sources of FIB in urban stormwater systems. New techniques have been developed, expanding the toolbox of techniques available for use by MS4s. Generally, communities should begin with simple, less costly methods focused on identification of human sources of FIB, progressing to more advanced methods where problem areas have been targeted. Identification and removal of human sources of FIB is expected to be most beneficial in terms of reducing human health risks in recreational waters. Once human sources of FIB have been corrected, it is possible (and even likely) that elevated FIB may persist. In such cases, typically where urban wildlife sources are present, significant control of remaining sources of FIB may be challenging. An alternative now available to communities to support development of site specific standards is QMRA. Although QMRA is a potentially helpful technique to provide regulatory relief to MS4s while providing standards at levels providing human health protection, it is costly and requires qualified expertise to properly design and execute a scientifically sound analysis.

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6 STATISTICAL ANALYSES OF STORMWATER FIB DATA

FIB data collected from urban stormwater systems and receiving waters are challenging to statistically analyze. FIB data sets may have one or more of these characteristics:

- highly variable observation levels (more so than for any other commonly monitored stormwater constituent).
- frequent right-censored data (very high levels can exceed the upper limit of the method being used and most analytical methods have a limited range of FIB levels that can be quantified without dilutions).
- potential measurement errors introduced in the laboratory when dilutions are conducted to address right-censored data issues.
- sensitive to environmental conditions (especially temperature).

These issues hinder the types of statistical analyses that can be conducted using most stormwater FIB data. However, many of these problems can be overcome by careful sampling and proper selection of the analytical method, as noted below.

Sampling plans must consider the sampling locations appropriate for the project objectives. Bacteria sources vary greatly in urban areas, and high levels can be observed in many locations. Roof runoff water can have very high levels during summer and spring months if covered extensively by trees, due to the increased numbers of birds and squirrels that can reside above the roof surfaces. During colder months, many of these animals may migrate, hibernate, or become inactive, resulting in significantly decreased FIB levels in the roof runoff. Soil FIB levels can remain high in areas where homeless people, urban wildlife or pets defecate, with runoff FIB levels dependent on the amount of erosion occurring. Bacteria levels in runoff from paved areas can also be high in areas where pets are “walked” or where geese congregate (such as parking lots near parks). Residential areas and park pathways usually have larger FIB levels than industrial areas, for example. Outfall stormwater samples are affected by the relative contributions of flows from the different areas that vary mostly according to rain characteristics. Therefore, an experimental design for FIB sampling needs to address these varying sources and seasonal conditions, requiring many samples over an extended period to draw meaningful conclusions.

Analytical methods used for FIB analyses usually have a limited range of direct quantification. Initial samples are likely to have many “right-censored” observations, with “too numerous to count” or other over-range indications instead of actual values. Most stormwater managers are familiar with “left-censored” data where the observations are below the detection limits. Several data substitution methods may be applied to the non-detected values, if relatively few in number, with minimal effect on the statistical test results. However, simple data substitutions are limited for right-censored data. Substituting the maximum detectable concentration for results reported as greater than the upper quantification limit (e.g., >2,419 MPN/100 mL) can result in a significant underestimate of the central tendency and variability of the underlying distribution. Therefore, statistical approaches, such as log-probability regression, maximum likelihood, and empirical probability (i.e., Kaplan-Meier) are the most appropriate (Helsel 2005). However, all of these approaches require either an assumed statistical distribution or enough uncensored data to characterize the empirical distribution. The best approach is to avoid censoring in the first place by using a wider range of sample dilutions in the FIB analyses. This is done differently for different methods, but will result in additional laboratory analyses, and therefore higher analytical costs (assuming that the lower limit is to be preserved, and not shifted higher). As an example, using the IDEXX methods and Quanti-Tray/2000 chambers, a standard analytical range of <1 to 2,419 MPN/100 mL is available. Most analysts using this method do not dilute the sample, with many stormwater FIB observations exceeding this range. However, this can be supplemented with a second tray (at twice the analytical cost) with a sample diluted 10 to 1 to extend the range to 24,192 MPN/100 mL, a level that is only periodically exceeded. Further dilutions can even be used (such as 100 to 1 for a range up to 241,900), but great care should be taken in the sample dilution process, with the recommended use of replicate trays to reduce errors associated with sample dilution and non-discrete FIB groups. In most cases, having the complete data with minimum uncertainties is worth the extra costs associated with the expanded analytical method, especially if the data set is to be used to identify FIB sources, or to quantify the FIB removal benefits of a stormwater control practice. For compliance purposes, it may only be necessary to know that the permit limit was exceeded; however, the actual value is needed to quantify the geometric mean value from many sampling events, as required by many regulatory agencies or to accurately estimate the variability.

Although water resources statistics texts are readily available for detailed guidance on statistical analysis approaches, the following discussion specifically summarizes some of the issues and solutions that can be applied when statistically analyzing stormwater FIB data. The brief discussions in this chapter can be used as a reminder or brief checklist of statistical techniques to consider for FIB analysis of stormwater and receiving waters. This discussion is not intended to be exhaustive, but can be used to identify some useful tools appropriate for data having high variations and missing observations. Much of this discussion is summarized from the stormwater sampling book by Burton and Pitt (2002), supplemented with various examples from past and on-going research on stormwater FIB sources, transport and fate being conducted by researchers at the University of Alabama.

6.1 Overview of General Steps in the Analysis of Stormwater Bacteria Data

The analysis of data requires at least three elements: 1) quality control/quality assurance of the reported data, 2) an evaluation of the sampling effort and methods (and associated expected errors), and finally, 3) the statistical analysis of the information. Quality control and quality assurance basically involve the identification and proper handling of questionable data (e.g., minimizing errors). When reviewing previously collected data, it is common to find obvious errors that are associated with improper units or sampling locations. Other potential errors are more difficult to identify and correct. In some cases, the identification and rejection of “outliers” may result in the dismissal of rare (but real) data observations that could provide important insight to the problem being investigated.

Experimental design efforts are usually associated with activities conducted prior to formal sample collection. However, many attributes of experimental design can also be used when evaluating previously collected data. This is especially useful when organizing data into relevant groupings for more efficient analyses. In addition, adequate sampling efforts are needed to characterize the information to the desired levels of confidence and power.

A general strategy in data analyses should include several phases and layers of analyses. Graphical presentations of the data (using exploratory data analyses) should be conducted initially. Simple to complex relationships between variables may be more easily identified through visual data presentations for most people, compared to only relying on descriptive statistical summaries. Of course, graphical presentations should be supplemented with statistical test results to quantify the significance of any patterns observed. The comparison of data from multiple situations (upstream and downstream of an outfall, summer vs. winter observations, etc.) is a very common experimental objective. Similarly, the use of regression analyses is also a very commonly used statistical tool. Trend investigations of water quality conditions with time are also commonly conducted. These standard tools can be applied when the database contains few missing or incomplete data. When large fractions of the observations are left-censored, or especially right-censored, then most of these basic tools are not available. The following discussion presents a few of the more useful statistical tools that can be used when data are missing, as in many FIB data sets with right-censored data.

6.2 Selection of Statistical Procedures

Most of the objectives of receiving water studies can be examined through the use of relatively few statistical evaluation tools. The following briefly outlines some simple experimental objectives and a selected number of statistical tests (and their data requirements) that can be used for data evaluation (Burton and Pitt 2001). Rather than relying on a single statistical measure, a combination of graphical and descriptive measures to characterize FIB data are preferred.

6.2.1 Basic Characterizations

One of the first tasks usually conducted with monitoring data is to prepare basic characterization statistics. For most of the examples in this chapter, data collected by Shergill (2004), in support of his master's thesis at the University of Alabama, are used. These were collected during a six month period in 2002 from the campus of the University of Alabama and from surrounding areas in Tuscaloosa, AL. In Table 6-1, *E. coli* data are presented with three different options: the first set of columns reflect the native limited range of the analytical method (<1 to 2,419 MPN/100 mL), the second set of columns includes the results of the additional ten-fold dilutions that were also evaluated, and the third set of columns has data substitutions or 0.5 in place of the <1 low detection limit. Also shown on the table are the statistical summaries for each set of data.

Additional References for More Information on Statistical Analyses Pertinent to FIB

Helsel, D.R. and Hirsch, R.M. (2002). *Statistical methods in water resources—hydrologic analysis and interpretation*, U.S. Geological Survey Techniques of Water-Resources Investigations, Book 4, Chap. A3, 510 p.

Wymer, L.J. and Wade, T.J. (2007). "The Lognormal Distribution and Use of the Geometric Mean and the Arithmetic Mean in Recreational Water Quality Measurement" in *Statistical Framework for Recreational Water Quality Criteria and Monitoring*, John Wiley & Sons Ltd., Chichester.

Table 6-1. Roof Runoff *E. coli* Observations (MPN/100 mL)

	Without dilution		With 10X dilution		With 10X dilution and substitution for <1	
Date	**Birds**	**no birds**	**birds**	**no birds**	**birds**	**no birds**
29-Aug-02	145.5	<1	145.5	<1	145.5	0.5
21-Sep-02	461.1	30.5	461.1	30.5	461.1	30.5
25-Sep-02	18.7	2	18.7	2	18.7	2
25-Sep-02	1,413.6	5.2	1,413.6	5.2	1,413.6	5.2
10-Oct-02	410.6	344.8	410.6	344.8	410.6	344.8
27-Oct-02	>2,419.2	161.6	17,329	161.6	17,329	161.6
5-Nov-02	>2,419.2	29.2	12,033	29.2	12,033	29.2
29-Jan-03	2	<1	2	<1	2	0.5
6-Feb-03	<1	>2,419.2	<1	5,298	0.5	5,298
Minimum	<1	<1	<1	<1	0.5	0.5
Maximum	>2,419	>2,419	17,329	5,298	17,329	5,298
Median	411	29	411	29	411	29
Analyses based on quantifiable data:						
Number of quantifiable observations	6	6	8	7	9	9
Average (mean)	409	95.6	3,977	839	3,535	652
Geometric mean	106	28.3	363	59.8	175	20.7
Standard deviation	529	136	6,772	1,970	2,157	582
Coefficient of Variation (COV)	1.3	1.4	1.7	2.3	1.8	2.7
Shapiro-Wilk test: Significantly different from normal distribution?	Yes (p<0.001)	Yes (p<0.001)	Yes (p<0.001)	Yes (p<0.001)	Yes (p<0.001)	Yes (p<0.001)

These are paired observations obtained from two different residential roofs: one roof had an extensive canopy of trees covering the building, while the other did not. The building with the canopy had a significant amount of observed bird and squirrel activity during the spring and summer months, while few were observed at the uncovered roof. As noted, three of the samples had no response during the test and therefore had <1 MPN/100 mL, while three were over-range. These samples were also analyzed using a ten-fold dilution to extend the upper range of the test to 24,192 MPN/100 mL, resulting in numeric results for the over-range values.

Some basic test statistics are also shown in Table 6-1 for the dilution results. For these calculations, the few non-detectable results at <1 MPN/100 mL were substituted with 0.5 MPN/100 mL values. This substitution was suitable as the detection limits are very low compared to the majority of the data, and the number of non-detected values is relatively low (1 of 9 and 2 of 9), although the “no birds” non-detects data were somewhat larger than the maximum desired non-detected frequency of 15% (Maestre et al. 2005). When substitutions using half of the detection limit exceed this frequency, the effects on the calculated data summaries (the variations and mean) can become larger than desired. One way to observe the effect of the data substitution of the data set is to use the extreme values (and not half of the detection limit), in this case the detection limit of 1 MPN/100 mL and zero. The effects on the means, medians, geometric means, and variation can then be observed. In this example (Table 6-2), the results were very close, except that the geometric means cannot be calculated if the data set includes a zero value and when the difference when substituting 0.5 or 1 had a <5 to 20% effect on the calculated value. Advanced substitution methods are also an option, including the regression on order (ROS) method, Kaplan-Meier, maximum likelihood estimation (MLE), and others. As mentioned above, these methods may also be used to estimate values for the right-censored data.

It should also be noted that ignoring the missing data and not applying substitutions results in a summary of the observed data only. This may be suitable if the number of measured observations is clearly stated, but this method hinders further analyses and complete comparisons to criteria. Appropriate data substitutions are much more useful as they allow more complete statistical tests. However, extensive substitutions can be misleading and it is always best if analytical methods are chosen to minimize non-detected values, or further dilutions be used to reduce the frequency of right-censored values.

Table 6-2. Effect of Various Simple Substitution Methods for Left-Censored Data

	Substituting <1 with 0		**Substituting <1 with 1**		**Substituting <1 with 0.5**	
	Birds	**No birds**	**Birds**	**No birds**	**Birds**	**No birds**
count	9	9	9	9	9	9
average (mean)	3,535	652	3,535	653	3,535	652
median	411	29.2	411	29.2	411	29.2
geometric mean	n/a*	n/a*	189	24.1	175	20.1
st dev	6,471	1,746	6,471	1,746	6,471	1,746
COV	1.8	2.7	1.8	2.7	1.8	2.7

* the geometric mean takes the product of all of the observations and then takes the n^{th} root of that product, where n is the total number of data observations. The presence of zero in the data set therefore does not allow the value to be calculated.

Particularly for FIB, the arithmetic means and geometric means are different when describing the same data set (but not the median), making the selection of the central descriptor critical. Therefore, great care should be taken when summarizing FIB data having large variations, and non-quantified data (as is typical with stormwater FIB observations) to match the objectives of the data. Reporting all three measures is a good practice. Some typical receiving water FIB standards are expressed using geometric means over an extended period of observations. However, when calculating mass discharges of contaminants (such as when determining compliance with TMDL discharge limits, or when calibrating or verifying models), flow-weighted averages should be used. As an example, if several water volumes of different amounts and concentrations are mixed together, the resulting concentration is obviously a flow (or volume) weighted average of the individual values, and not related to medians, the simple average, or the geometric mean of the individual samples.

Because of the wide range of FIB values typically observed during a monitoring period, managers are uncomfortable with the extra effects that the very large values have on resulting calculated average values. The median values are therefore commonly used when describing this type of data as the extremes and uncertain values have little effects on its value (unless the uncertain events number is more than 50% of the data set). Unfortunately, medians are not very useful when comparing to standards written using geometric mean values, unless the data follow log-normal distributions, or when calculating loads. If a data set were symmetrical (not necessarily normally distributed), then the medians and the means would have the same value, but as the distribution skewness increases, the means and medians can vary greatly, as shown in Table 6-2. Methods to moderate the effects of these very large values are typically used for reporting purposes. The median and geometric mean values in Table 6-2 are significantly smaller than the averaged values, with the geometric means being 20+ times smaller than the averages. If geometric means are used in "mass" calculations, the results would cause large errors. Similarly, if geometric mean summaries of past observations are all that are available, the statistical tests that can be applied are limited.

Table 6-1 also shows the results of the Shapiro-Wilk test that was used to test normality of the data. In this example, the test failed for all of the data sets. Therefore, statistical tests that require normally distributed data should not be used with these data unless adequately transformed. In most cases, non-parametric tests that have fewer distribution requirements are usually preferred, if available. This result is common for most stormwater observations (Maestre, et al. 2005, and many others), especially for FIB data. The relatively large coefficient of variation (the standard deviation divided by the mean) values (1.8 and 2.7 for the final data set with substitutions) also indicate likely non-normal behavior.

6.2.2 Determining the Number of Samples for Specified Data Quality Objectives

The comparison of paired data sets is commonly used when evaluating the differences between two situations (locations, times, practices, etc.). An equation to estimate the needed number of samples for a paired comparison is:

$$n = 2\,[(Z_{1-\alpha} + Z_{1-\beta})/(\mu_1 - \mu_2)]^2\sigma^2$$

where:

α = false positive rate (1-α is the degree of confidence. A value of α of 0.05 is usually considered statistically significant, corresponding to a 1-α degree of confidence of 0.95, or 95%)

β = false negative rate (1-β is the power. If used, a value of β of 0.2 is common, but it is frequently ignored, corresponding to a β of 0.5.)

$Z_{1-\alpha}$ = Z score (associated with area under normal curve) corresponding to 1-α

$Z_{1-\beta}$ = Z score corresponding to 1-β value

μ_1 = mean of data set one

μ_2 = mean of data set two

σ = standard deviation (same for both data sets, same units as μ. Both data sets are also assumed to be normally distributed.)

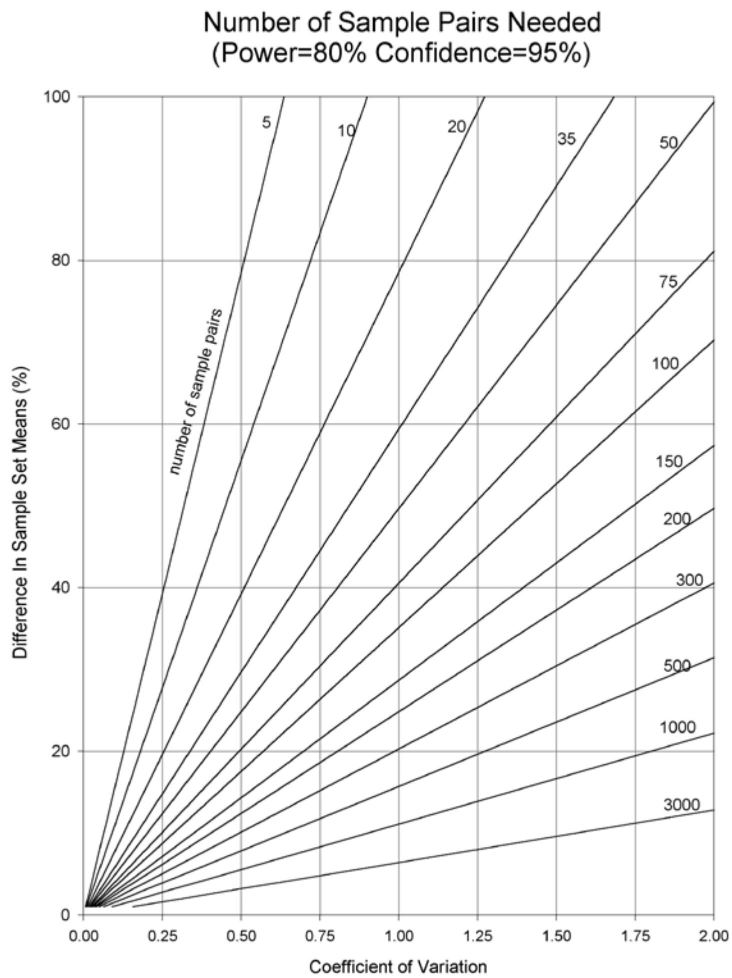
In order to ensure adequate power for a sampling effort (controlled false negatives in addition to confidence, or false positives), the experimental design must include a sufficient sampling effort, as indicated above. Statistical tests only report the confidence (the alpha value) and do not include a separate indication of power. They are therefore somewhat overstating the confidence level as they ignore the power. The most important method to have adequate power and confidence corresponding to the desired data quality objectives is with the experimental design. However, this equation is only approximate as it requires that the two data sets are normally distributed and have the same standard deviations. Most stormwater constituents, including FIB are more likely close to being normally distributed. If the coefficient of variation (COV) values are low (less than about 0.4), then there is little no real difference in the predicted sampling effort. However, the COVs are usually much larger, indicating larger deviations from the calculated sampling effort.

Burton and Pitt (2001) illustrate an example of conducting a log-transformation of the data and applying the above equation. They illustrate that the method results in a range of the difference being detected and not the single value. This range is more symmetrical as the data distribution approaches a normal distribution, with the intended difference close to the median value. As the distribution becomes more biased (larger COV for example), the range becomes more lope-sided, but the desired difference to be detected is still within the range, but not at the median. Therefore, this equation can be applied to the data not having a normal distribution in order to account for both power and confidence, but the uncertainty in the observable difference becomes larger as the data deviate from a normal distribution. The calculated number of samples needed appears to be larger than necessary based on the confidence of a statistical comparison of the data sets, but this is usually due to ignoring the effects of power in addition to the confidence.

Figure 6-1 (Burton and Pitt 2001) is a plot of this equation (normalized using COV and differences of sample means) showing the approximate number of sample pairs needed for an α of 0.05 (degree of confidence of 95%), and a β of 0.2 (power of 80%). As an example, twelve sample pairs will be sufficient to detect significant differences (with at least a 50% difference in the parameter value) for two locations, if the coefficients of variation are no more than about 0.5.

The hypothesis of the experiment using these measurements is that the roof having large numbers of birds would have larger FIB levels in the roof runoff compared to the roof with few birds. The ability of the data to detect these differences is dependent on the power of the experiment, which is a function of the experimental design. (The power of a statistical test is the probability that the test will reject the null hypothesis when the alternative hypothesis is true.) In this case, paired sampling was used to enable paired statistical tests, which have greater power than tests that collect independent data. However, the large variation of the data requires that large numbers of samples be collected to observe small differences, as indicated in Figure 6-1 (Burton and Pitt 2001). As noted previously, this plot assumes normal distributions of the data, which is seldom true, resulting in a range of the targeted difference in sample set means.

Figure 6-1. Sample Effort Needed for Paired Testing
(Power of 80% and Confidence of 95%)
(Source: Burton and Pitt 2001)



This plot (developed for normal data, but useful for non-normal also) indicated that 10 sample pairs with data having COV values of about 2.0 would only be able to detect differences that are much greater than 100% between the sample sets. The data indicate that the observed reductions in FIB levels between the roof with birds compared to the roof without birds is about 80%. The figure indicates that about 75 sample pairs would be needed to statistically validate this hypothesis with the rigorous data quality objectives (DQO) of 80% power and 95% confidence. Therefore, these observations may only indicate possible trends in the observations with much less rigorous DQOs, but they may be useful as a preliminary step in a more rigorous experimental effort. Obviously, one of the problems with wet weather flow sampling is the limited number of sampling opportunities available in a short sampling period. During these experiments, only 9 sample pairs were able to be collected in the six-month sampling period. Many more rains occurred during this time, but the time needed to collect the samples at each of the sampling locations (several other paired locations were also being sampled during these tests) restricted sample collection efforts. During recent monitoring in the Tuscaloosa, AL, area, researchers were able to collect samples from about 25 to 30 rainfall events per year (out of about 100 rains), using automatic samplers. Not all rains produce adequate runoff or meet typical sampling protocol requirements. Bacteria samples are to be manually collected according to most test protocols, and most U.S. locations have many fewer rains than Alabama. Therefore, it would be very difficult to collect large numbers of samples for FIB analyses during most studies, requiring several years, or many concurrent sampling locations, to obtain sufficient data for some experimental objectives. Therefore, because of the large variations and associated large numbers of needed samples, traditionally stringent DQOs may not be achievable for FIB, and the actual confidence limits obtained should always be reported, not just that the data failed by not achieving a traditional 0.05 probability level of statistical significance.

6.2.3 Comparison Tests

A common experimental objective is to compare data collected from different locations, or seasons. Comparison of test site data with reference sites, of influent with effluent, of upstream to downstream locations, for different seasons of sample collection, of different methods of sample collection, can all be made with comparison tests. If only two groups are to be compared (above/below; in/out; test/reference), then the two group tests can be effectively used, such as the simple Student's *t*-test or nonparametric equivalent. If the data are collected in "pairs," such as for concurrent influent and effluent samples, or for concurrent above and below samples, then the more powerful and preferred paired tests can be used. If the samples cannot be collected to represent similar conditions (such as large physical separations exist in sampling location, or different time frames), then the independent tests must be used.

If multiple groupings are used, such as from numerous locations along a stream, but with several observations from each location; or from one location, but for each season, then a one-way ANOVA (or non-parametric equivalent for most FIB data) is needed. If one has seasonal data from each of the several stream locations for multiple seasons, then a two-way ANOVA test can be used to investigate the effects of location, season, and the interaction of location and season together. Three-way ANOVA tests can be used to investigate another dimension of the data (such as contrasting sampling methods or weather for the different seasons at each of the sampling locations), but that would obviously require substantially more data to represent each condition.

There are various data characteristics that influence which specific statistical test can be used for comparison evaluations. The parametric tests require the data to be normally distributed and that the different data groupings have the same variance, or standard deviation (checked with probability plots and appropriate test statistics for normality, such as the Shapiro-Wilk, the Kolmogorov-Smirnov one-sample test, the Chi-square goodness of fit test, or the Lilliefors test). If the data do not meet the requirements for the parametric tests, the data may be transformed to better meet the test conditions (such as taking the $\log_{10}$ of each observation and conducting the normality test on the transformed values). The non-parametric tests are less restrictive, but are not free of certain requirements. Even though the parametric tests have more statistical power than the associated non-parametric tests, they lose any advantage if inappropriately applied. If uncertain, then non-parametric tests should be used. A few example statistical tests are listed below for different comparison test situations:

A) Two Groups

Paired observations

- Parametric tests (data require normality and equal variance)
    - Paired Student's *t*-test (more power than non-parametric tests when the normality and equal variance assumptions hold, but only if these data requirements are met)
- Non-parametric tests
    - Sign test (no data distribution requirements, some missing data accommodated)
    - Friedman's test (can accommodate a moderate number of "non-detectable" values, but no missing values are allowed
    - Wilcoxon signed rank test (more power than sign test, but requires symmetrical data distributions)

Independent observations

- Parametric tests (data require normality and equal variance)
    - Independent Student's *t*-test (more power than non-parametric tests)
- Non-parametric tests
    - Mann-Whitney rank sum test (probability distributions of the two data sets must be the same and have the same variances, but do not have to be symmetrical; a moderate number of "non-detectable" values can be accommodated)

B) Many Groups (use multiple comparison tests, such as the Bonferroni *t*-test, to identify which groups are different from the others if the group test results are significant).

Parametric tests (data require normality and equal variance)

- One-way ANOVA for single factor, but for >2 locations (if 2 locations, use Student's *t*-test)

- Two-way ANOVA for two factors simultaneously at multiple locations
- Three-way ANOVA for three factors simultaneously at multiple locations
- One factor repeated measures ANOVA (same as paired *t*-test, except that there can be multiple treatments on the same group)
- Two factor repeated measures ANOVA (can be multiple treatments on two groups)

Non-parametric tests

- Kruskal-Wallis ANOVA on ranks (use when samples are from non-normal populations or the samples do not have equal variances).
- Friedman repeated measures ANOVA on ranks (use when paired observations are available in many groups).

Nominal observations of frequencies (used when counts are recorded in contingency tables)

- Chi-square (X^2) test (use if more than two groups or categories, or if the number of observations per cell in a 2 x 2 table are > 5).
- Fisher Exact test (use when the expected number of observations is <5 in any cell of a 2 x 2 table).
- McNamar's test (use for a paired contingency table, such as when the same individual or site is examined both before and after treatment)

6.2.4 Data Associations and Model Building

These activities are an important component of the "weight-of-evidence" approach used to identify likely cause and effect relationships. The following list illustrates some of the statistical tools (found in many commercially-available software packages that can be used for evaluating data associations and subsequent model building:

A) Data Associations

Simple

- Pearson correlation (residuals, the distances of the data points from the regression line, must be normally distributed. Calculates correlation coefficients between all possible data variables. Must be supplemented with scatterplots, or scatter plot matrix, to illustrate these correlations. Also identifies redundant independent variables for simplifying models).
- Spearman rank order correlation (a non-parametric equivalent to the Pearson test).

Complex (typically only available in advanced software packages)

- Hierarchical Cluster Analyses (graphical presentation of simple and complex inter-relationships. Data should be standardized to reduce scaling influence. Supplements simple correlation analyses).

- Principal Component Analyses (identifies groupings of parameters by factors so that variables within each factor are more highly correlated with variables in that factor than with variables in other factors. Useful to identify similar sites or parameters).

B) Model building/equation fitting (these are parametric tests and the data must satisfy various assumptions regarding behavior of the residuals)

- Linear equation fitting (statistically-based models)

  - Simple linear regression ($y=b_0+b_1x$, with a single independent variable, the slope term, and an intercept. It is possible to simplify even further if the intercept term is not significant).

  - Multiple linear regression ($y=b_0+b_1x_1+b_2x_2+b_3x_3+\ldots+b_kx_k$, having k independent variables. The equation is a multi-dimensional plane describing the data).

  - Stepwise regression (a method generally used with multiple linear regression to assist in identifying the significant terms to use in the model.)

  - Polynomial regression ($y=b_0+b_1x^1+b_2x^2+b_3x^3+\ldots+b_kx^k$, having one independent variable describing a curve through the data).

- Non-linear equation fitting (generally developed from theoretical considerations, such as solved partial differential equations)

  - Nonlinear regression (a nonlinear equation in the form: $y=b^x$, where x is the independent variable. Solved by iteration to minimize the residual sum of squares).

C) Data Trends

- Graphical methods (simple plots of concentrations versus time of data collection).

- Regression methods (perform a least-squares linear regression and examine ANOVA for the regression to determine if the slope term is significant. Can be misleading due to cyclic data, correlated data, and data that are not normally distributed).

- Mann-Kendall test (a nonparametric test that can handle missing data and trends at multiple stations. Short-term cycles and other data relationships affect this test and must be corrected).

- Sen’s estimator of slope (a nonparametric test based on ranks closely related to the Mann-Kendall test. It is not sensitive to extreme values and can tolerate missing data).

- Seasonal Kendall test (preferred over regression methods if the data are skewed, serially correlated, or cyclic. Can be used for data sets having missing values, tied values, censored values, or single or multiple data observations in each time period. Data correlations and dependence also affect this test and must be considered in the analysis).

As noted previously and as illustrated with the example FIB data set, many of these tools may not be applicable to stormwater (or even dry weather) FIB data. The large data variations hinder sufficient data to verify many of the required data characteristics (generally restricting available methods to some of the non-parametric procedures), and there are typically many missing data in the observed data sets (especially problematic are the over-range observations). In addition, historical FIB data are usually reported as geometric means that do not reflect the flow-weighted values that are needed for load analyses. Therefore, the most obvious methods that can be used to evaluate stormwater FIB data may be restricted to the following:

<u>Basic Data Summaries</u>:

- Central tendency measures appropriate for the project objectives (geometric means for compliance with water quality standards; means for calculating flow-weighted discharges and TMDL compliance and for model calibrations)

- Measures of variation (tests for data normality, standard deviations, COVs, and limitations due to sample numbers)

<u>Exploratory Data Analyses</u>:

- Probability plots (with truncated distributions reflecting missing data)

- Box and whisker plots (possibly only using reported values)

- Trend (time series) plots showing FIB level changes with time

- Line plots contrasting paired data sets

<u>Comparison Tests</u>:

- Sign test for paired observations

- Wilcoxon signed-rank test for paired observations with few missing data

- Mann-Whitney rank sum test for independent observations in two sample sets

- Kruskal-Wallis ANOVA on ranks to detect significant subsets of the data

<u>Correlation Tests</u>:

- Spearman Rank order test for simple correlations of non-normal data

- Cluster and principal component advanced analyses to identify complex relationships of data; requires substantial information and few missing data

Trend Analyses and Model Building:

- Graphical analyses, usually based on time series of observations over long periods of time

- Nonparametric trend tests, depending on available data and their characteristics

- Factorial analyses to identify significant factors affecting observations, if sufficient data are available

A few of the more basic options are described in the following sections.

6.3 Exploratory Data Analysis

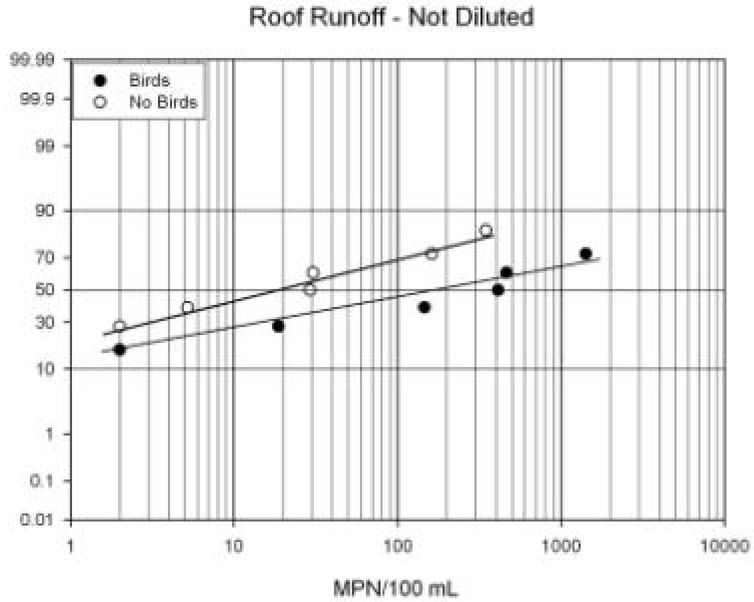
Exploratory data analyses are important tools to quickly review available data before a specific data analysis effort is initiated. It is also an important first step in summarizing collected data to supplement the specific data analyses associated with the selected experimental designs. A summary of the data's variation is most important and can be presented using several simple graphical tools. An important reference for basic analyses is *Exploratory Data Analysis* (Tukey 1977), which is the classic book on this subject and presents many simple ways to examine data to find patterns and relationships. Besides plotting of the data, exploratory data analyses should always include corresponding statistical test results, if available.

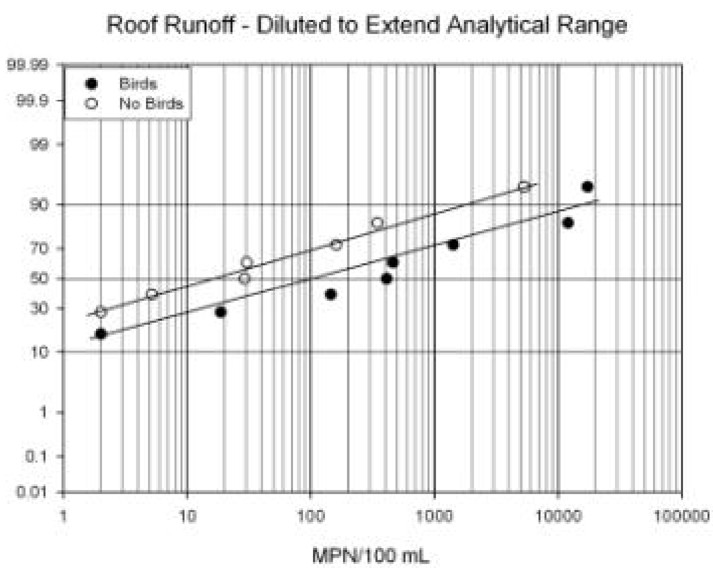
6.3.1 Probability Plots

The most basic and important exploratory data analysis method is to prepare a probability plot of the available data. The plots indicate the possible range of the values expected, their likely probability distribution type, and the data variation. The values and corresponding probability positions are plotted using normal-probability scales. These have a y-axis whose values are spread out for the extreme small and large probability values. When plotted using these scales, the values form a straight line if they are normally distributed (Gaussian). If the points do not form an acceptably straight line (as expected for most FIB stormwater data), they can be plotted using a log scale for the observed values to indicate if they are log-normally distributed.

Figure 6-2 provides log-normal probability plots of the above presented FIB data for the roof runoff from the two sampling locations, with the bird and no-bird data shown on the same graphs for comparison. These are truncated plots not showing information for the non-detected left-censored or right-censored observations. The second plot is for the data set that includes the diluted samples with an extended range and no right-censored observations. These are accurate plots in that they do not include any assumed or substituted data, and reflect the actual observations. They are both log-normal plots and indicate reasonably straight line relationships, indicating that data transformations would possibly be advantageous and allow extended parametric statistical analyses. The lower limits of these plots are truncated and do not show the <1 MPN/100 mL non-detected values that were at about 11 and 22% of the datasets. The upper limits are truncated at the right-censored values. For the first plot, not having the extended range associated with the extra sample dilutions, the data are truncated at about 70 and 80%, respectively. For the extended range plot, the upper limits are only truncated at the maximum values obtained. The plotted median values are seen to shift between the two sets of data, especially for the site having birds.

Figure 6-2. Plots of Roof Runoff with and without Birds Using Two Different Sample Dilutions





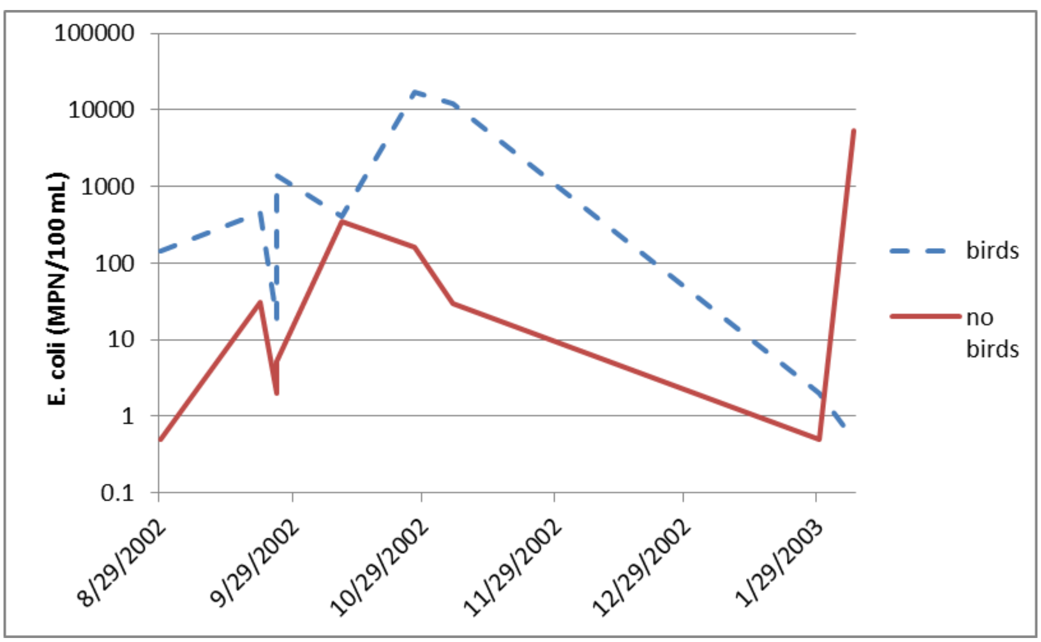
Generally, water quality observations do not form a straight line on normal probability plots, but do (at least from about the 10 to 90 percentile points) on log-normal probability plots, as shown above. This indicates that the samples generally have a log-normal distribution and many parametric statistical tests can probably be used, but only after the data are log-transformed. These plots indicate the central tendency (median) of the data, along with their possible distribution type and variance (the steeper the plot, the smaller the COV and the flatter the slope of the plot, the larger the COV for the data).

Probability plots should be supplemented with standard statistical tests that determine if the data are normally distributed. These tests include the Kolmogorov-Smirnov one-sample test, the chi-square goodness of fit test, the Anderson-Darling test, and the Lilliefors variation of the Kolmogorov-Smirnov test. They basically are paired tests comparing data points from the best-fitted normal curve to the observed data. The statistical tests may be visualized by imagining the best-fitted normal curve data and the observed data plotted on normal probability graphs. If the observed data crosses the fitted curve data numerous times, it is much more likely to be normally distributed than if it only crossed the fitted curve a few times. As indicated previously, these roof runoff FIB data are not normally distributed for statistical test purposes, but may be log-normally distributed.

6.3.2 Time-Series Plots

Berthouex and Brown (1994) point out that since the best way to display data is with a plot, it makes little sense to present the data only in a table. A basic time series plot indicates any obvious data trends with time. Figure 6-3 shows the *E. coli* observations for the roof runoff from the building having birds vs. no birds above the roof, indicating that the presence of the birds (in the absence of any other factor) affected the observed values during much of the study period. However, these effects notably decreased in the late fall, as shown in Figure 6-3.

Figure 6-3. *E. coli* Observations in Roof Runoff



6.3.3 Scatterplots

According to Berthouex and Brown (1994), the majority of the graphs used in science are scatterplots. They stated that these plots should be made before any other analyses of the data are performed. Scatterplots are typically made by plotting the primary variable (such as a water quality constituent) against a factor that may influence its value (such as time, season, flow, another constituent like suspended solids, etc.). Figure 6-4 is a scatterplot relating fecal coliform observations to *E. coli* observations, using data from the Big Dry Creek watershed near Denver, CO. Although not a perfect correlation, this plot indicates a general similarity between these two FIB species in stormwater. (The plot also illustrates a data set constrained by an upper quantitation limit of 2,419 MPN/100 mL for *E. coli*.)

Figure 6-4. Scatterplot Showing Relationship Between Fecal Coliform and *E. coli*

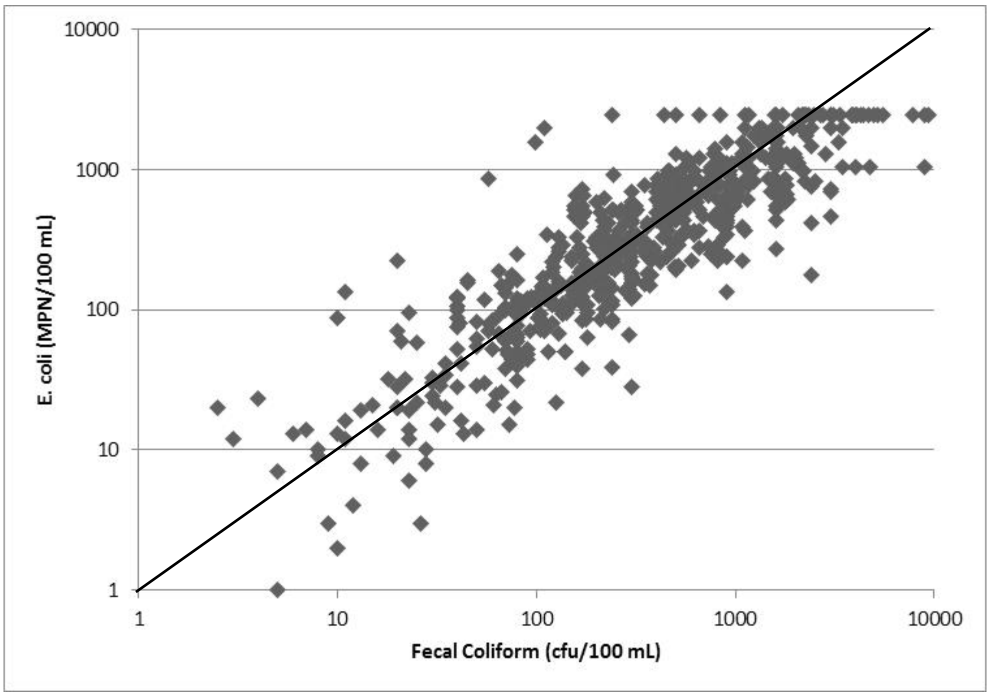
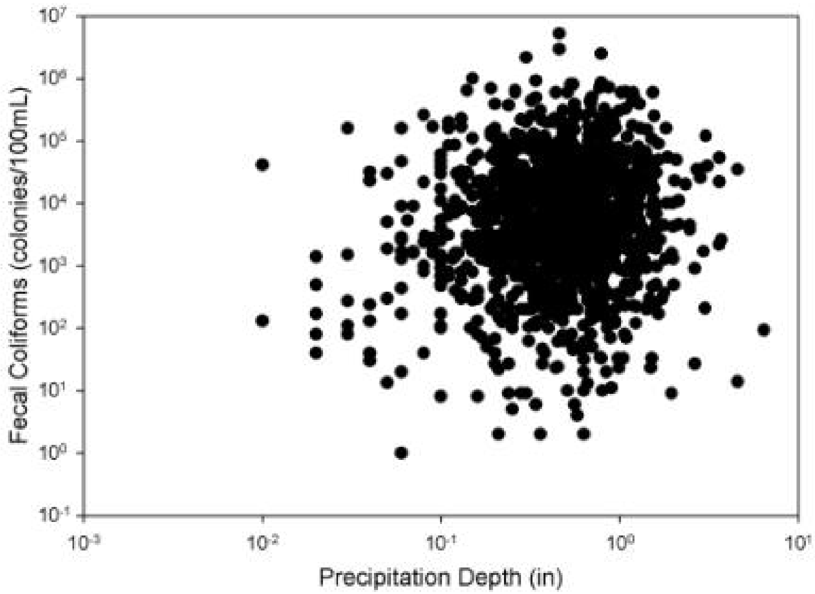


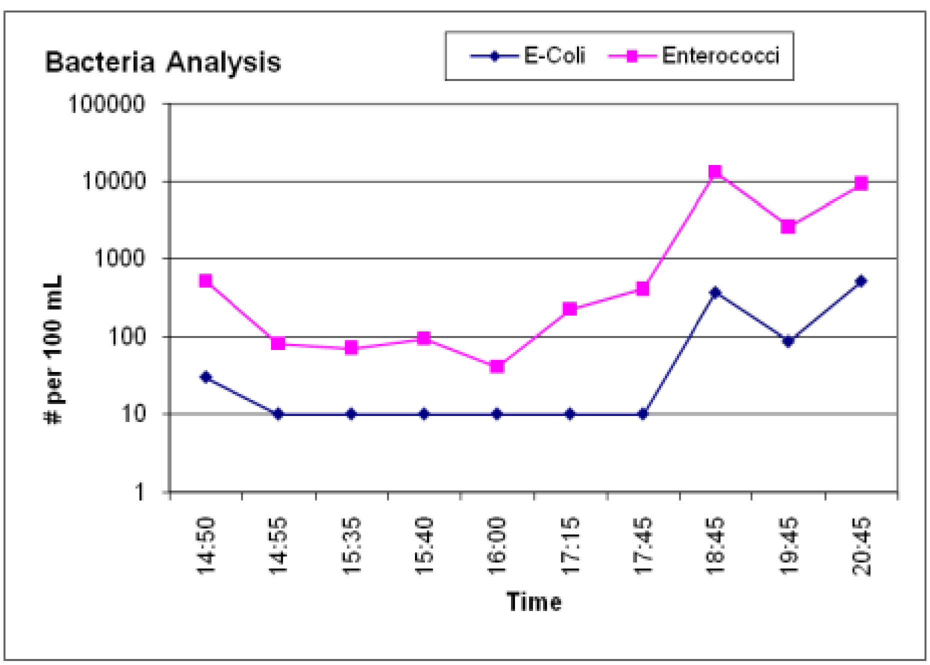
Figure 6-5 provides a scatterplot also using data from the NSQD and relates observed fecal coliform observations from outfall samples to the concurrent rain depth during the sampling period. This plot indicates a random pattern with no observed trends.

Figure 6-5. Scatterplot of Fecal Coliforms vs. Precipitation Depth



As illustrated in Figure 6-6, following a 1-inch rain monitored at a small paved area in Tuscaloosa, AL, the FIB levels steadily increase with time. This was likely associated with the more distant landscaped area flows contributing runoff later in the event as they become saturated and started to produce runoff. These landscaped areas are in a park area where residents exercise their dogs.

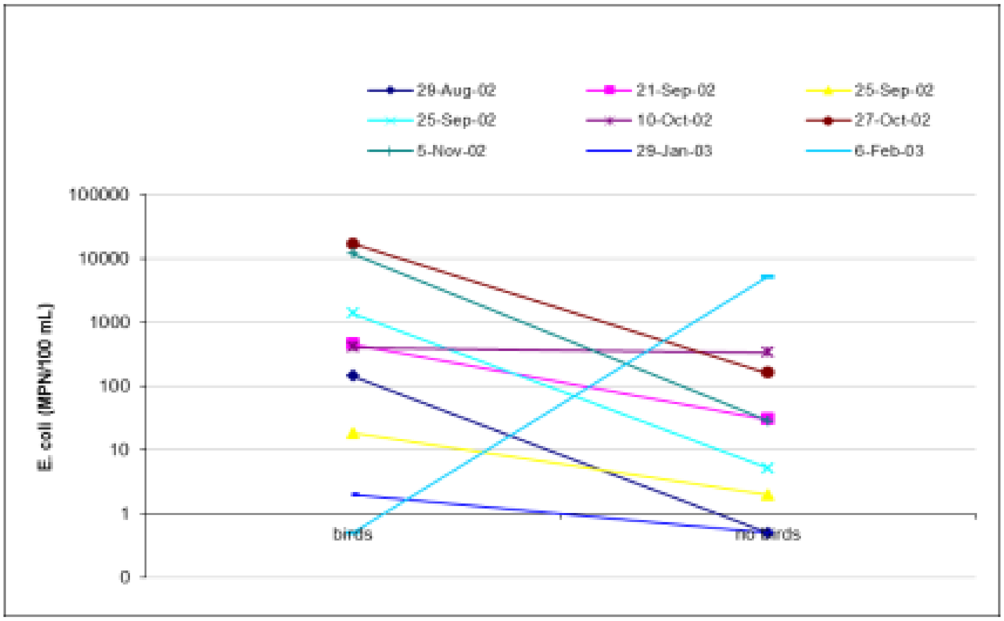
Figure 6-6. Changes in *E. coli* and Enterococci Concentrations over Time



6.3.4 Line Plots

Line plots are a type of scatterplot that contrasts paired observations. Figure 6-7 is a line plot for the roof runoff data contrasting the roofs affected by birds and those not affected by birds. This plot indicates that almost all of the data pairs were higher for the roof having birds than for the roof without birds, with one exception.

Figure 6-7. Line Plots of *E. coli* by Date for Sites with and without Birds

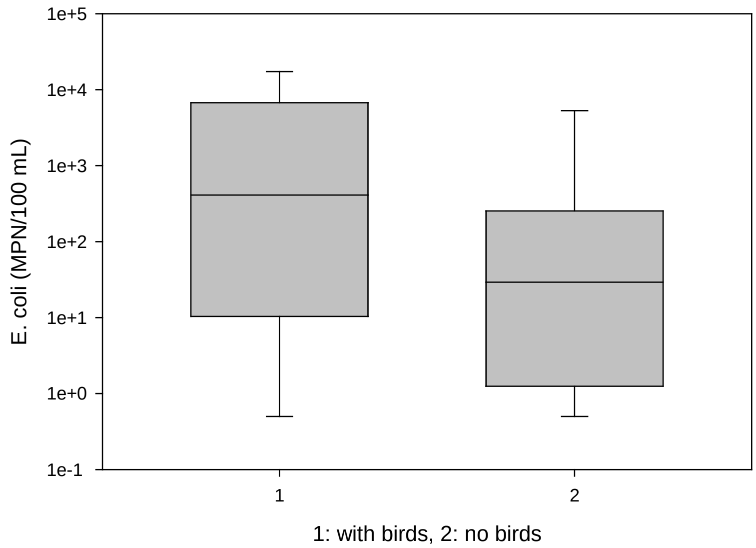


6.3.5 Grouped Box and Whisker Plots

Another primary exploratory data analysis tool, especially when differences between sample groups are of interest, is the use of grouped box and whisker plots. Examples of their use include examining different sampling locations (such as above and below a discharge), influent and effluent of a treatment process, different seasons, etc.

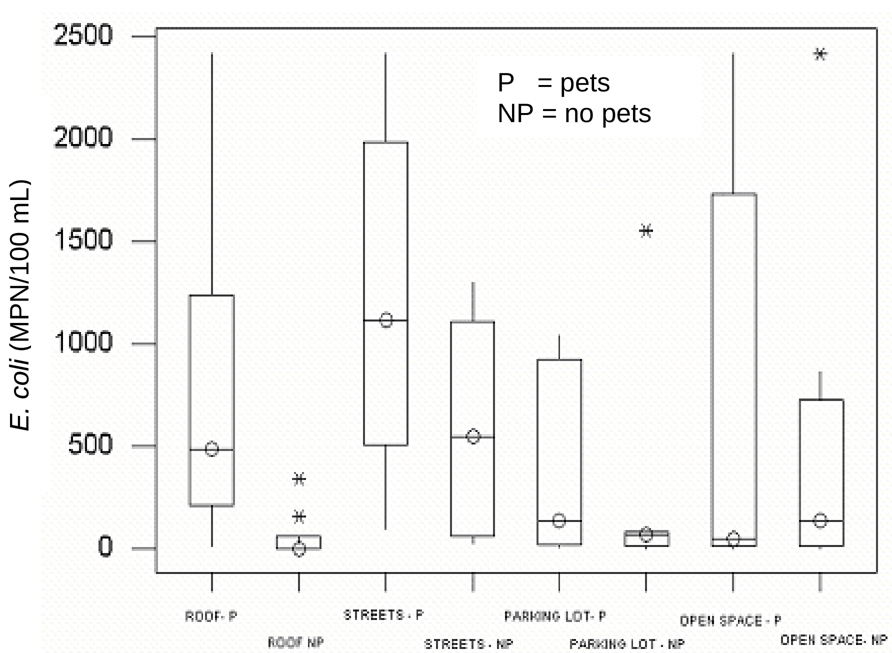
In Figure 6-8, the grouped box and whisker plot (Shergill 2004) contrasts the roof runoff samples collected from buildings affected by birds vs. those without birds. These are all of the data, without regard to season and indicate large variations and overlapping data sets. For small data sets, if the median line in one box (the "central" line in the box) is above or below the corresponding 25th or 75th percentile box ends in the adjacent box, then a statistically significant difference is the data sets is likely (but should be confirmed with the appropriate statistical analysis). The whiskers indicate the 5th and 95th percentile observations of the data sets. In this example, the "with bird" median value is barely larger than the "no bird" 75th percentile, but the "no bird" median is not lower than the "with birds" 25th percentile value. Because of the large variations in these data, the level of confidence in the differences may be marginal at best.

Figure 6-8. Boxplots of *E. coli* in Roof Runoff at Sites with and without Birds



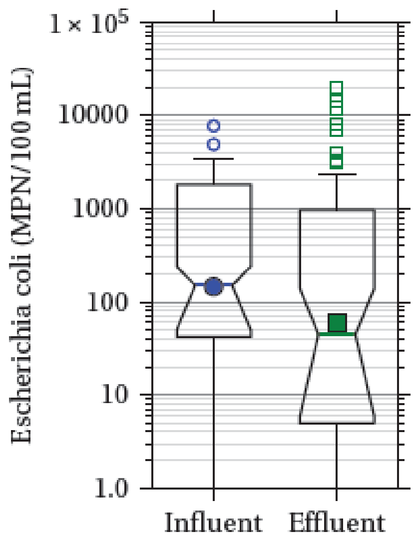
In contrast, Figure 6-9 only examines the warm weather source area *E. coli* values from areas likely affected by urban pets and wildlife compared to other areas. The differences for the roof data are most obvious, but the other areas (streets, parking lots, and open space) also show large decreases (although the large variations and greater overlapping of the boxes indicate that they may not have differences that are statistically significant). The roof and parking lot site that were not prone to animals have much lower values and smaller variations compared to the other sites.

Figure 6-9. Boxplots of *E. coli* by Land Use Type



Use of "notched" box and whisker plots can be a useful variation on standard box and whisker plots. Using an example from the International Stormwater BMP Database, influent and effluent *E. coli* concentrations are shown in Figure 6-10. The "notches" around the median represent the 95% confidence limits for the median. When the confidence intervals between the paired boxplots do not overlap, then statistically significant differences are likely. In this case, the notches overlap, so no statistically significant difference in influent and effluent concentrations is expected, despite the visual suggestion that the effluent median is lower than the influent median. (This finding of no significant difference between influent and effluent was also verified using formal hypothesis testing using the Mann-Whitney and Wilcoxon tests.)

Figure 6-10. Notched Boxplots for Influent and Effluent *E. coli* Data for Bioretention Practices in the International Stormwater BMP Database



To supplement the visual presentation with the grouped box and whisker plots, the Kruskal-Wallis (K-W) ANOVA on ranks test should be conducted to determine if there are any statistically significant differences between the different boxes on the plot (usually used for more than two sets of data). The K-W ANOVA test doesn't identify which sets of data are different from any other, however. A nonparametric multiple comparison procedure can be used to identify significant differences between all cells if the K-W ANOVA finds that a significance difference exists. These tests will identify differences in sample groupings, but similarities (to combine data) are probably also important to know.

6.4 Statistical Tests for Comparing Multiple Sets of Data

Making comparisons of data sets are fundamental objectives of many stormwater investigations. Different locations and seasons can produce significant effects on the observations. Berthouex and Brown (1994) and Gilbert (1987) present excellent summaries of the most common statistical tests that are used for these comparisons in environmental investigations. The significance of the test results (the α value, the confidence factor, along with the β value, the power factor) will indicate the level of confidence and power that the two sets of observations are the same. In most cases, an α level of less than 0.05 has been traditionally used to signify significant differences between two sets of observations, although this is an arbitrary criterion and may not be reasonable for FIB analyses. Unfortunately, β is often ignored (resulting in a default value of 1-β of 0.5), although some use a 1-β value of 0.8. An α value of 0.05 implies that the interpretation will be in error an average of 1 in 20 times. Even if the α level is significant, the magnitude of the difference, such as the pollutant reduction, may not be very important. The importance of the level of pollutant reductions should also be graphically presented using grouped box plots indicating the range and variations of the concentrations at each of the sampling locations, as described previously.

Comparison tests are divided into simple comparison tests between two groups (such as Sign Test) and tests that examine larger numbers of groups and interactions (such as Kruskal-Wallis ANOVA on ranks).

6.4.1 Simple Comparison Tests with Two Groups

The main types of simple comparison tests are separated into independent and paired tests. These can be further separated into tests that require specific probability distribution characteristics (parametric tests) and tests that do not have as many restrictions based on probability distribution characteristics of the data (nonparametric tests). If the parametric test requirements can be met, then they should be used because they have more statistical power. However, if information concerning the probability distributions is not available, or if the distributions do not behave correctly, then the somewhat less powerful nonparametric tests should be used. Similarly, if the data gathering activity can allow for paired observations, then paired analyses should be used preferentially over independent (unpaired) tests.

In many cases, observations cannot be related to each other, such as a series of observations at two locations during all of the rains during a season. Unless the sites are very close together, the rains are likely to vary considerably at the two locations, disallowing a paired analysis. However, if data can be collected simultaneously, such as at influent and effluent locations for a (rapid) treatment process, paired tests can be used to control most factors that may influence the outcome, resulting in a more efficient statistical analysis. Paired experimental designs ensure that uncontrolled factors basically influence both sets of data observations equally (Berthouex and Brown 1994).

The parametric tests used for comparisons are the Student's *t*-tests (both independent and paired *t*-tests). All statistical analyses software and most spreadsheet programs contain both of these basic tests. These tests require that the variances of the sample sets be the same and are constant over the range of the values. These tests also require that the probability distributions be Gaussian (normal). Transformations can be used to modify the data sets to these conditions. Log-transformations can be used to produce Gaussian distributions of most water quality data. Square root transformations are also commonly used to make the variance constant over the data range, especially for biological observations (Sokal and Rohlf 1969). In all cases, it is necessary to confirm these requirements before the standard *t*-tests are used.

Nonparametrics: Statistical Methods Based on Ranks by Lehman and D'Abrera (1975) is a comprehensive general reference on nonparametric statistical analyses. Gilbert (1987) presents an excellent review of nonparametric alternatives to the Student's *t*-tests, especially for environmental investigations from which the following discussion is summarized. Even though the nonparametric tests remove many of the restrictions associated with the *t*-tests, the *t*-tests should be used if justifiable. Unfortunately, seldom are the Student's *t*-test requirements easily met with environmental data and the slight loss of power associated with using the nonparametric tests is much more acceptable than misusing the Student's *t*-tests. Besides having few data distribution restrictions, many of the nonparametric tests can also accommodate a few missing data, or observations below the detection limits. The following paragraphs briefly describe the features of the nonparametric tests used to compare data sets.

Nonparametric Tests for Paired Data Observations. The sign test is the basic nonparametric test for paired data. It is simple to compute and has no requirements pertaining to data distributions. A few "not detected" observations can also be accommodated, as long as the two observations can be examined and the larger value identified (e.g., one non-detect in a pair is useable, but if both are non-detect, then those data are not useable). Two sets of data are compared and the differences are used to assign a positive sign if the value in one data set is greater than the corresponding value in the other data set, or a negative sign is assigned if the one value is less than the corresponding value in the other data set. The number of positive signs are added and a statistical table (such as in Lehman and D'Abrera 1975, Table G) is used to determine if the number of positive signs found is unusual for the number of data pairs examined. This table shows that in order to have at least a 95% confidence that two sets of paired data are significantly different, only one out of eight pairs can have a larger data value in one set compared to the seven larger ones in the other data set. As the number of pairs of observations increase, the allowable number of inconsistent values increases. With 40 pairs of observations, as many as 14 inconsistent values are allowed and still retain a 95% level of confidence.

The roof runoff data sets indicated that the roof most likely affected by birds had larger *E. coli* values than the other roof in almost all cases (only 1 of the 9 pairs of observations was not higher). This level of exceedences and total number of paired observations results in a p value of 0.0195, a significant level when compared to the standard 0.05 level. The non-detected values and the over-range values do not influence this test, as it was always possible to determine which of the two data in each pair was larger. If both data values were uncertain for the same sample pair, then that pair should be discarded (as it is not possible to know which is larger), resulting in fewer overall data.

The Mann-Whitney signed rank test has more power than the sign test, but it requires that the data distributions be symmetrical (but with no specific distribution type). Without transformations, this requirement may be difficult to justify for water quality data. This test requires that the differences between the data pairs in the two data sets be calculated and ranked before checking with a special statistical table (as in Lehman and D'Abrera 1975). In the simplest case for monitoring the effectiveness of treatment alternatives, comparisons can be made of inlet and outlet conditions to determine the level of pollutant removal and the statistical significance of the concentration differences. Table 6-3 provides the result of the Wilcoxon signed rank test using the roof runoff observations for *E. coli*. Again, the uncertain non-detectable and over-range observations did not affect the analysis. However, corrections for ties can be used if non-detectable values or over-range values exist in both data sets (as occurred in this example). The following analysis used the extended range data and ties were corrected for the non-detected values occurring in each data set.

Table 6-3. Non-parametric Wilcoxon Signed Rank Test for Paired Data Observations (Example with and without Birds)

Prior to Wilcoxon test, Shapiro-Wilk normality test result = failed (p < 0.050)					
Group	N	Missing	Median	25%	75%
Roof with birds	9	0	410.6	10.35	6723.3
Roof with no birds	9	0	29.2	1.25	253.2
Wilcoxon Test Results:					
W= -31.000 T+ = 7.000 T-= -38.000					
Z-Statistic (based on positive ranks) = -1.836					
P(est.)= 0.076 P(exact)= 0.074					

The first step in the analysis is to perform a test to check for normality of the data. If the data are normally distributed, then a parametric test would be recommended. In this case, the data are not normally distributed, so the Wilcoxon rank sum test is preferred. The results indicate a less promising conclusion than observed with the Sign Test (p = 0.074). However, this finding is based on a critical p value of 0.05. The calculated p value was slightly larger than 0.05, but less than 0.1, so in many cases, this finding may be considered significant, especially for preliminary data.

Friedman's test is an extension of the sign test for several related data groups. There are no data distribution requirements and the test can accommodate a moderate number of "non-detectable" values, but no missing values are allowed.

Nonparametric Tests for Independent Data Observations. Paired test experimental designs are superior to independent designs for nonparametric tests because of their ability to cancel out confusing properties (which are assumed to occur simultaneously for each observation in the pair). However, paired experiments are not always possible, requiring the use of independent tests. The Wilcoxon rank sum test is the basic nonparametric test for independent observations. The test statistic is also easy to compute and compare to the appropriate statistical table (as in Lehman and D'Abrera 1975). The Wilcoxon rank sum test requires that the probability distributions of the two data sets be the same (and therefore have the same variances). There are no other restrictions on the data distributions (they do not have to be symmetrical, for example). A moderate number of "non-detectable" values can be accommodated by treating them as ties. Table 6-4 is a summary of the test results of the Mann-Whitney rank sum test using the roof runoff data (as if the observations were not paired).

Table 6-4. Non-parametric Test for Independent Data Observations (Birds/No-Birds Example)

Prior to Mann-Whitney test, Shapiro-Wilk normality test result = failed ($p < 0.050$)					
Group	N	Missing	Median	25%	75%
birds	9	0	410.6	10.35	6723.3
no birds	9	0	29.2	1.25	253.2
Mann-Whitney Test Results:					
Mann-Whitney U Statistic= 25.500					
T = 100.500 ($p = 0.199$)					

Again, the first test is to check for normality. Since the data failed this test, the Mann-Whitney rank sum test is the most suitable (assuming the data are not paired, but are independent). The calculated p value ($p = 0.199$) is substantially larger than for the previous sum of ranks paired test, illustrating how the paired test in more likely to detect significant differences in paired data. Also, the Mann Whitney test result is based on a 2-tail distribution, with no pre-test hypothesis on which data set is larger than the other. In this case, the roof affected by the birds was assumed to have larger *E. coli* levels than the other roof, so a one-tail test is suitable. This results in the calculated p value being reduced in half. Even so, the resulting p value is about 0.1, somewhat larger than the p calculated for the paired test (but may still be a suitable level considering the large variability in FIB data and the preliminary nature of this information).

The Kruskal-Wallis ANOVA test on ranks is an extension of the Mann-Whitney rank sum test and allows evaluations of several independent data sets, instead of just two. Again, the distributions of the data sets must all be the same, but they can have any shape. A moderate number of ties and non-detectable values can also be accommodated.

6.4.2 Comparisons of Many Groups

If there are more than two groups of data to be compared (such as in-stream concentrations at several locations along a river, each with multiple observations), one of the analysis of variance (ANOVA) tests should be used. The commonly available one-way, two-way, and three-way ANOVA tests are parametric tests and require that the data in each grouping be normally distributed and that the variances be the same in each group. This can be visually examined by preparing a probability plot for the data in each group displayed on the same chart. The probability plots would need to be parallel and straight. Obviously, log transformations of the data can be used if assumptions are met when the data is plotted using log-normal probability axes.

A non-parametric test usually included in statistical programs for comparing many groups is the Kruskal-Wallis ANOVA on ranks test. This is only a one-way ANOVA test and would be only suitable for comparing data from different sampling locations or seasons, for example. This would be a good test to supplement grouped box and whisker plots.

Grouped comparison tests indicate only that at least one of the groups is significantly different from at least one other, they do not indicate which ones. For that reason, some statistical programs also conduct multiple comparison tests. SigmaStat, for example, offers: the Tukey test, Student-Newman-Keuls test, Bonferroni t-test, Fisher's LDS, Dunner's test, and Duncan's multiple range test. As another example, XLSTAT offers three multiple comparison methods in association with the Kruskal-Wallis test, including procedures by Dunn, Conover and Iman and Steel-Dwass-Critchlow-Fligner. These tests basically conduct comparisons of each group against each other group and identify which are different. Visual inspection of a grouped box and whisker plot of the multiple groups is also especially helpful when combining groups.

6.5 Data Associations

Identifying patterns and associations in data may be considered a part of exploratory data analyses, but many of the tools (especially cluster, principal component, and factor analyses) may require specialized procedures having multiple data handling options that are not available in all statistical software packages, while some are commonly available. The following are possible steps for investigating data associations:

1. Re-examine the hypothesis of cause and effect (an original component of the experimental design previously conducted and was the basis for the selected sampling activities).

2. Prepare preliminary examinations of the data, as described previously (most significantly, prepare scatter plots and grouped box/whisker plots).

3. Conduct comparison tests to identify significant groupings of data. As an example, if seasonal factors are significant, then cause and effect may vary for different times of the year.

4. Conduct correlation matrix analyses to identify simple relationships between parameters. Again, if significant groupings were identified, the data should be separated into these groupings for separate analyses, in addition to an overall analysis.

5. Further examine complex inter-relationships between parameters by possibly using combinations of hierarchical cluster analyses, principal component analyses (PCA), and factor analyses.

6. Compare the apparent relationships observed with the hypothesized relationships and with information from the literature. Potential theoretical relationships should be emphasized.

7. Develop initial models containing the significant factors affecting the parameter outcomes. Simple apparent relationships between dependent and independent parameters should lead to reasonably simple models, while complex relationships will likely require further work and more complex models.

The following sections briefly list these tools that are most commonly available in statistical analyses software packages and may be suitable for stormwater FIB data analyses.

6.5.1 Correlation Matrices

Knowledge of the correlations between data elements is very important in many environmental data analyses efforts. They are especially important when model building, such as with regression analysis. When constructing a model, it is important to include the important factors in the model, but the factors should be independent. Correlation analyses can assist by identifying the basic structure of the model.

6.5.2 Hierarchical Cluster Analyses

Another method to examine correlations between measured parameters is by using hierarchical cluster analyses. A tree diagram (dendogram) illustrates both simple and complex relationships between parameters. Parameters having short branches linking them are more closely correlated than parameters linked by longer branches. In addition, the branches can encompass more than just two parameters. The length of the short branches linking only two parameters is indirectly comparable to the correlation coefficients (short branches signify correlation coefficients close to 1). The main advantage of a cluster analyses is the ability to identify complex correlations that cannot be observed using a simple correlation matrix.

6.5.3 Principal Component Analyses and Factor Analyses

Another important tool to identify relationships and natural groupings of samples or locations is with principal component analyses (PCA). Normally, data are autoscaled before PCA in order to remove the artificially large influence of constituents having large values compared to constituents having small values. PCA is a sophisticated procedure where information is sorted to determine the components (usually constituents) needed to explain the variance of the data. Typically, very large numbers of constituents are needed for PCA and a relatively small number of sample groups are to be identified.

6.6 Summary of Stormwater Bacteria Statistical Analyses

Stormwater FIB data are characterized by large variations and missing data, particularly values above quantification limits. This can be overcome by carefully designing the monitoring program to focus on the most critical elements to monitor so sufficient data can be obtained. In addition, appropriate laboratory methods need to be used to enable the wide range of FIB levels to be quantified, such as expanding the dilution series to avoid data sets with right-censored data.

Data summaries and statistical analyses must be chosen to correspond to the objectives of the research effort. Geometric mean values are commonly used for purposes of comparison to FIB instream standards, but they are misleading when applied to statistical analyses and model building. Flow-weighted average values are most suitable for these analyses. In most cases, nonparametric statistical analyses are needed for analyzing stormwater FIB data. There are many tools that can be used, but data requirements must be verified before their use, especially related to right-censored values. Also, because of the large variability in the data, it may be most suitable to accept somewhat less demanding data quality objectives, especially for initial exploratory investigations.

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7 SOURCE CONTROLS (NON-STRUCTURAL PRACTICES)

Source controls of FIB are the first strategies that should be pursued when FIB impairments are identified. Examples of these strategies are summarized in Table 7-1, with additional discussion of several of these practices provided in the remainder of this report. Education and outreach overarches many of these strategies.

7.1 Education and Outreach

Education and outreach to citizens and businesses is a general overarching source control practice that is needed for other types of source controls to be effective. Education and outreach activities may include brochures, posters, websites, event attendance, utility bill inserts, television advertisements, articles in homeowner association newsletters and other approaches that effectively reach citizens and promote behavioral changes. There may be opportunities for stormwater managers to work in a cross-disciplinary manner with other city utilities to maximize public education dollars. For example, campaigns to reduce water waste by reducing over-irrigation help communities to meet both conservation and water quality objectives. Similarly, drinking water departments often focus on source water protection strategies, but due to the departmental "silos", opportunities for integrating stormwater and drinking water program objectives may not be fully maximized.

7.2 Repair of Aging Infrastructure and Correcting Illicit Connections

In 2013, the American Society of Civil Engineers (ASCE) gave the nation a "D+" on its "Report Card for America's Infrastructure" and estimated that a $3.6 trillion investment was needed by 2020 to address the most pressing aging infrastructure issues. ASCE's report card for wastewater and stormwater sector was a "D" and concluded:

> Capital investment needs for the nation's wastewater and stormwater systems are estimated to total $298 billion over the next twenty years. Pipes represent the largest capital need, comprising three quarters of total needs. Fixing and expanding the pipes will address sanitary sewer overflows, combined sewer overflows, and other pipe-related issues. In recent years, capital needs for the treatment plants comprise about 15%-20% of total needs, but will likely increase due to new regulatory requirements. Stormwater needs, while growing, are still small compared with sanitary pipes and treatment plants. Since 2007, the federal government has required cities to invest more than $15 billion in new pipes, plants, and equipment to eliminate combined sewer overflows.

Consistent with ASCE's findings, in many communities, aging sanitary and storm sewer infrastructure is a major issue. Aging sanitary pipes can be a significant source of FIB loading in urban areas. Many communities have implemented "Asset Management Programs" that provide a systematic strategy to manage, maintain and operate infrastructure. The EPA's Capacity, Management, Operation, and Maintenance (CMOM) is probably the most well-known asset management program. Asset management programs provide a framework for self-evaluation and planning for the function, condition, and performance of a sanitary sewer system (TCEQ 2013).

Table 7-1. Sources and Strategies for Bacteria Reduction

Bacteria Source	Stormwater Control/Management Strategy
Domestic Pets (dogs and cats)	Provide signage to pick up dog waste, providing pet waste bags and disposal containers. Adopt and enforce pet waste ordinances. Place dog parks away from environmentally sensitive areas. Protect riparian buffers and provide unmanicured vegetative buffers along streams to dissuade stream access.
Urban Wildlife (rats, bats, raccoons)	Reduce food sources accessible to urban wildlife (e.g., manage restaurant dumpsters/grease traps, residential garbage, feed pets indoors).
Illicit Connections to MS4s	Implement an IDDE program to identify and remove illicit connections.
Leaking Sanitary Sewer Lines/Aging Sanitary Infrastructure	Conduct investigations to identify leaking sanitary sewer line sources and implement repairs.
Onsite Septic Systems and Package Plants	Implement a program to identify potentially failing septic systems. Enforce discharge permit requirements for small package plants.
Illegal Dumping	Implement a reporting hotline for illegal dumping and educate the public/industries that dumping to storm sewer system is illegal.
Storm Sewer System and Stormwater Quality BMPs	Proper maintenance of the storm sewer system and water quality BMPs is needed for proper functioning of the system. For example, sediment, organic deposits and biofilms in stormwater facilities can be sources of elevated FIB.
Storm Runoff from Urban Areas	Encourage site designs that minimize directly connected impervious areas.
Dry Weather Urban Flows (irrigation, carwashing, powerwashing, etc.)	Implement public education programs to reduce dry weather flows from storm sewers related to lawn/park irrigation practices, carwashing, powerwashing and other non-stormwater flows. Provide irrigation controller rebates. Implement and enforce ordinances related to outdoor water waste. Inspection of commercial trash areas, grease traps, washdown practices, along with enforcement of ordinances.
Birds (e.g., Canada geese, gulls, pigeons)	Identify areas with high bird populations and evaluate deterrents, population controls, habitat modifications and other measures that may reduce bird-associated FIB loading.
Wildlife: (raccoons, beavers, deer, coyotes, field rats, mice)	Consult with state wildlife offices on strategies to reduce food, shelter and habitat for overpopulated urban wildlife. Implement and enforce urban trash management practices.
Homeless Populations	Support of city shelters and services to reduce homelessness. Periodic cleanup of homeless camps near streams. Police enforcement. Providing public restrooms. Partnering with non-governmental organizations to address homelessness.

Aging and leaking sanitary sewer and stormwater conveyance pipes can introduce pollutants to the MS4 through SSOs caused by blockages, line breaks, or other sewer defects; exfiltration of sewage from sanitary sewers; and infiltration of groundwater when the MS4 lies below the water table (Sercu et al. 2011). Upgrading, repairing, or slip-lining faulty sanitary sewer pipes will reduce pollutant loads by eliminating the leaks in those pipes. Additionally, upgrading or repairing storm drain pipes can minimize the infiltration of contaminated groundwater into the MS4 (Geosyntec 2012).

Measures to reduce SSOs include field inspections and using CCTV to inspect sewer lines, which can reveal blockages from debris to roots to grease and show pipeline cracks, breaks, or deterioration. Once such issues are identified, they can be integrated into planning efforts to maintain, rehabilitate or replace aging sanitary infrastructure.

Accelerated repair or upgrade of sanitary sewer and storm drain systems can be a key measure to reduce human sources of FIB. The location and design of upgrades can be optimized to decrease pollutant loads using information gathered in IDDE programs, GIS analysis of high-risk sewers, and/or special source tracking studies. Strategically planning upgrades to older, clay sanitary sewer laterals that cross or run next to and above storm drains is cost-effective and offers multiple benefits, including benefits to water quality and reduced operations and maintenance costs from newer infrastructure (Geosyntec 2012).

Santa Barbara, CA
Source Tracking Case Study and Sanitary Sewer Exfiltration

The City of Santa Barbara partnered with University of California Santa Barbara researchers and Geosyntec Consultants to develop and implement a toolbox approach for source tracking to identify potential human sources of contamination in the storm sewer system (City of Santa Barbara 2012a&b). GIS layers showing the storm sewer and sanitary sewer systems and instream FIB concentrations were used as base mapping to track findings and identify potential sources of contamination. For preliminary screening, trained canines were used to target storm sewers with potential sewage contamination (Van De Werfhorst et al. 2014), then progressively more complex tools were implemented. For example, flow gauges and automated samplers were temporarily installed in suspicious outfalls to identify patterns that could indicate illicit connections. In addition to FIB samples, qPCR analysis for human markers was also conducted (e.g., HF183). CCTV inspections were also used to identify areas of infiltrations into the storm sewers. As a result of these efforts, human sources of bacteria from exfiltrating sanitary sewers above storm sewers were identified and corrected by the City.

For example, the City of Santa Barbara identified four locations through the use of a dye tracer and cutting-edge microbial source tracking study that together contributed roughly 1,500 gallons of raw sewage each day via infiltration into the local MS4 (Sercu et al. 2011).[10] Other studies

[10] Identification of human waste sources in Santa Barbara is not evidence that the area is “more polluted” than other cities. Instead, the findings demonstrate that when a city aggressively pursues research partnerships, grant funding, and community support in an effort to identify and fix sources of human-associated FIB, such sources, which are expected to be common in urban areas, can be identified and corrected.

that have recently applied microbial source tracking analytical techniques to urban storm drains have similarly found (typically low levels of) human fecal markers to be present (Corsi 2014). As another example, Sauer et al. (2011) investigated separate stormwater sewers in the Milwaukee urban area and concluded that leaking sanitary sewer infrastructure was widespread and infiltrating into storm sewers.

Multiple tools for identifying illicit sanitary connections and discharges to the storm drain system were discussed in Chapter 5. Once such sources are identified, they must be corrected because they represent a direct source of raw sewage discharged to receiving waters. To increase the effectiveness of IDDE, enhancements to basic programs may include a tiered dry weather source investigation including: (1) visual surveys of MS4s to identify dry weather flow locations, (2) GIS-based prioritization where aging sewer laterals are above and near storm drains that are observed to occasionally flow during dry weather, (3) video survey of the storm drains to identify leaks from the top of the pipe and/or sewer dye tracing studies, and (4) fecal source tracking studies that use canine scent tracking and/or microbial source tracking (Recommendations from San Diego Comprehensive Load Reduction Plan [CLRP], Geosyntec 2012).

7.3 Maintenance of Storm Sewers and Stormwater Controls

A variety of maintenance activities related to storm drainage infrastructure may help to reduce FIB loading.[11] Unfortunately, quantitative data and evaluation of the benefits of these practices is generally lacking. Practices that may be considered include:

- **Storm Sewer Cleaning:** Storm sewers can accumulate trash, sediment, organic matter and animal waste over time. As a result they can become secondary reservoirs of FIB and other pollutants. Cyclical storm sewer cleaning using water jetting and vacuuming of jetted water is one tool that some communities have implemented as a source control BMP. Storm sewer cleaning is typically done on a several year cycle and can be done more frequently in "priority basins" where elevated FIB at storm sewers is identified.

- **Catchbasin Cleaning:** Catchbasins and drain inlets play an important role in the prevention of trash and other sediment from entering the storm drain system. Catchbasin cleaning is an important institutional BMP, but the FIB load reduction benefits of increased frequency of catchbasin cleaning have not been well documented. A survey conducted as part of the San Diego River source study found that 46% of commercial catchbasins had moderate buildup and 34% had ponded water. Signs of washdown and food scraps were frequently associated with catchbasins near restaurants (Weston 2009a). However, studies to evaluate the potential benefits of catchbasin cleaning did not show significant reductions in FIB (Weston 2009b). However, in a study conducted in the Telecote Creek watershed in San Diego, commercial catchbasins had significantly higher FIB than residential catchbasins (Weston 2010b); thus, if catchbasin cleaning is being considered as a BMP, it may be more beneficial in commercial areas.

[11] Recommendations in Sections 7.3 and 7.4 and portions of Section 7.2 are based on discussion in the San Diego Comprehensive Load Reduction Plan, Attachment E Non-structural BMPs (Geosyntec Consultants 2012).

- **Structural Stormwater BMP Maintenance**: Maintenance of structural stormwater quality BMPs can also help to remove secondary reservoirs of FIB in urban areas. Routine sediment removal from dry detention basins and manufactured devices can reduce the likelihood of sediment resuspension and FIB release during storm events.

7.4 Street Cleaning

Street cleaning removes sediment, debris, and other pollutants from road and parking lot surfaces. The major factors that impact the effectiveness of a street cleaning program in reducing pollutant loads are frequency and timing of cleaning and the type of street cleaning equipment used. Effectiveness is also dependent on the speed the sweeper travels, the amount of sediment on the street, and how much of the street is swept (e.g., whether parked cars prevent sweepers from accessing the curb).

High-efficiency street sweeping equipment, such as regenerative air sweepers or vacuum assisted sweepers can significantly increase the amount of sediment removed from roadways. Geosyntec (2012) summarized findings from several studies comparing mechanical broom sweepers to newer high efficiency alternative equipment. These studies showed increases in sediment removal of 35% (Pitt 2002), 15 to 60% (Minton 1998), and up to 140% (Schwarze Industries). Additionally, regenerative air and vacuum sweepers were designed specifically to better capture fine particles. Bacteria, as well as metals and other pollutants, on sediments are typically associated with smaller sized particles due to a larger surface-to-volume ratio and greater adsorption properties of clay particles (Xanthopoulos and Hahn 1990, Krumgalz et al. 1992). Although measured reductions in discharges of pollutants and FIB to receiving waters due to street cleaning has rarely been observed, street cleaning is an important public works activity to minimize sediment accumulation in drainage systems.

7.5 Downspout Disconnections and Site Designs Minimizing Directly Connected Impervious Area

As discussed in Chapter 6, Shergill and Pitt (2004) found that roofs with birds and squirrels in overhead tree canopy had higher FIB than those without animal activity. An extension of this finding is that rooftops can be a source of FIB loading during wet weather events. Simply disconnecting roof downspouts can help to redirect runoff to pervious areas; thereby, potentially reducing both runoff volumes and FIB loads. Implementation options include redirecting downspouts to lawns, gardens or swales, or installing a rain barrel or cistern to collect roof runoff for later use. Downspout retrofit can be an effective stormwater control for commercial, industrial, and public buildings as well.

In addition to downspout retrofits, new developments or redevelopments can be designed to integrate multiple measures that reduce effective impervious area by disconnecting impervious surfaces. These Low Impact Development site designs can integrate both non-structural practices, as well as structural stormwater controls such as bioretention, permeable pavement and other practices (see Chapter 8). Reducing runoff peaks and volumes during frequently occurring storm events may help to reduce FIB loading, as well as reduce other pollutant loading.

7.6 Pet Waste Disposal and Pet Control Ordinances

The density of pets in urban areas can be quite high; therefore, proper disposal of pet waste is a basic component of FIB control plans in urban areas. Elements of pet control programs may include:

- Providing park and trail signs regarding pet waste disposal requirements and leash laws.
- Providing disposal cans at conveniently spaced intervals on trails and in open space areas. Some communities allow advertising on signs placed at pet waste bag dispensers and disposal cans to partially offset the cost (e.g., Poo Free Parks®).
- Providing and properly maintaining off-leash dog parks, preferably at locations that do not directly drain to receiving waters. Improperly maintained dog parks can become a source of FIB, rather than a stormwater control if not properly managed.
- Allowing natural riparian buffers to grow alongside streams to dissuade pet access.
- Providing educational materials regarding the impact of improperly disposed pet waste. These materials can be made available in locations such as pet stores, animal shelters, veterinary offices, and other sites frequented by pet owners.
- Enforcing pet waste ordinances and leash laws (or developing them, if they do not exist). While most communities have pet waste ordinances "on the books", enforcement of these ordinances may not routinely occur in many communities. In areas with significantly elevated FIB, allocation of resources to park and open space rangers to enforce pet waste disposal controls and leash laws may be needed.

Effectiveness of pet waste control programs is not well documented in terms of instream responses to implementation of such programs; however, Geosyntec (2012) summarized several surveys and reports that attempt to quantify behavioral change associated with such programs. For example, the Phase I Report for the San Diego River Kelp and Dog Waste Management Plan for Dog Beach and Ocean Beach found that public compliance with the "scoop the poop" policy was highly dependent on awareness of the policy and availability of waste disposal bags and trash cans (Weston 2004). Public surveys in the City of Austin indicated their educational campaign resulted in a 9% improvement in the number of pet owners who claim to regularly pick up waste (City of Austin 2008). Studies in San Diego have shown that installation of pet waste stations have resulted in a 37% reduction in the total amount of pet waste in city parks (City of San Diego 2011).

Pet waste cans and signage at a Denver-area park. (Photo Courtesy Jane Clary.)

7.7 Bird Control

Birds are a common source FIB, both at beaches and in inland urban areas. As discussed in Chapter 3, birds are documented sources of elevated FIB in many studies. For this reason, a fairly detailed discussed of potential control strategies for birds follows, since most urban areas are expected to have at least some contribution of FIB from birds.

The University of Nebraska at Lincoln (2010), USDA APHIS (1994a&b, 2003), the Internet Center for Wildlife Damage Management (www.icwdm.org) and others provide guidance on control strategies for geese. Canada geese are protected by federal and state laws. While it is illegal to intentionally kill a wild goose (other than during licensed hunting seasons) or to harm nesting geese and eggs without a permit, there are a number of methods used to discourage geese from congregating in specific areas. Non-lethal control activities do not require federal or state permits, and most non-lethal activities can be conducted throughout the year, except using trained dogs for hazing. Any activities that result in handling, damage, or destruction of geese, or their eggs or nests, require permits (CPW 2014).

Effective goose control often requires early detection of the problem, persistence, and use of multiple methods (CPW 2014). Table 7-2 summarizes measures that have been used for geese control, followed by additional discussion of several of these measures. Overall, USDA APHIS (2009) recommends that the most efficient and effective way to manage resident geese is to harass them before nests are built. If this is not possible, nest destruction and egg oiling are the best options.

Table 7-2. Summary of Selected Waterfowl Management Techniques
(Adapted from Smith et al. 1999, Smith 2006, NYCDEP 2004)

Technique
Public Education
Discontinuance of feeding
Habitat modification
Porcupine wires (for roosting waterfowl and pigeons)
Eliminate shorelines, islands, peninsulas (in constructed waterbodies)
String wire lines or place Mylar tape grids above roosting and pond areas
Fence barriers
Vegetative barriers (taller grasses)
Rock barriers
Floating plastic balls (may wash away during storms)
Reduce or eliminate mowing (adjacent to waterbodies)
Place walking path near water
Place fields away from water
Deterrence Measures
Sprinklers and motion-detected activated sprayers
Pyrotechnics
Sonic Devices: ultrasonics, distress calls, sirens, horns whistles, propane cannons or exploders
Active Visual Deterrents: strobe lights, lasers, light beams
Passive Visual Deterrents: "eye-spot" balloons or kites, flags, scarecrows, floating predator decoys (benefits may be temporary, as waterfowl may habituate over time)
Dispersion Measures
Dogs
Swans (can also be a source of FIB)
Falcons (often impractical to maintain)
Radio-controlled aircraft or boats
Chemical repellents (methyl anthranilate**)**
Reproductive Controls
Removing nesting materials (before egg laying)
Oil/addle/puncture eggs (during incubation)
Replace eggs with dummy eggs
Sterilization (oral contraception or surgical neutering)
Removal
Relocate (may not be effective)
Various lethal measures (e.g., hunts, kill permits)

Additional discussion on several of these methods for geese control includes:

- **Hazing methods**. A permit is not required to scare, repel or herd geese to protect the property, as long as the birds are not harmed or killed. Repeated hazing can cause geese to relocate, but hazing must be reinstated if the geese return. Hazing is most effective when geese first arrive at a location. Examples, as described by Smith et al. (1999), include:

- **Noisemakers and Pyrotechnics**. Check with local authorities before starting a regimen of noise-making, but loud and surprising noises can be a deterrent to resident geese. Where allowed, 12-gauge "cracker shells" and other sharp percussive sounds can prompt geese to move to another, more peaceful location.

- **Trained dogs and clipped swans**. Some golf courses have used highly trained border collies with skilled handlers to chase geese off fairways. This is not a method to be used casually with a canine pet since dogs cannot be allowed to catch or harm geese or other waterfowl. Leash laws in most cities and towns do not allow dogs to run free to chase geese. There are state regulations prohibiting use of dogs during certain times of the year (nesting season.) However, where allowed, this method has had proven successful. Some locations have purchased swans with clipped wings, so they cannot fly away, and released them on a pond or lake to frighten away geese. This method is not recommended where the swans will come in regular contact with people, as they can be aggressive to humans as well as geese. Be aware that swans can also breed and care must be taken to ensure that an overpopulation of swans does not occur in place of geese.

- **Scarecrows, Balloons, Scare Tape.** As a short-term tactic, often used with other methods, geese can sometimes be scared away using various shapes and movements. Scare tape (Mylar tape) is thin, shiny ribbon, usually silver on one side and red on the other. Place the reflective tape where it is visible to the geese and make a low fence across the area where geese exclusion is desired. Tie short lengths of the shiny ribbon on the cross tape; the flashing and rattling of the tape can frighten geese. People, pets and wind can break the tape, so it needs to be inspected and repaired daily to be useful.

- **Feeding:** Do not feed or allow feeding of geese or other waterfowl on the property. Efforts to frighten geese away can be thwarted if nearby neighbors are feeding the geese. If geese are being fed in the area, it will be very difficult to persuade them to move elsewhere.

- **Habitat Alteration:** A variety of habitat alteration measures may be helpful. The purpose of landscape modification around ponds is to disrupt travel and sight lines, with key practices described by Smith et al. (1999):

  - Landscape Modification: Geese dislike visual barriers between ponds and feeding areas. Planting trees, thick bushes, or a dense hedge between grassy areas and water may make the property less attractive to geese. While the living barrier is growing thick enough to be useful, it may be necessary to use other methods, such as temporary fencing or repellents, to keep the geese from establishing in the area. Geese prefer mowed grasses; so leaving a buffer area of tall grass and wildflowers can create a visual and physical barrier to resident geese.

  - Exclusion and Barriers: Physical barriers, such as fences and boulders, can be placed to prevent geese from entering an area. Fences should be at least 2-feet high and have openings no larger than 3 x 3 inches. Chain link, chicken wire, construction fence, and wood can be used.

- Repellents: There are several commercial repellents advertised to keep geese off of lawns. These products must be applied according to label directions to be effective; they may need to be reapplied after rain, or twice weekly in dry conditions. Approved repellents are made from biodegradable, food-grade ingredients and are not toxic to birds, dogs, cats, or humans.

- **Egg Oiling and Manipulation:** Egg oiling involves applying a 100% food-grade corn oil to eggs during the nesting season in early spring. Simply removing the eggs from a nest will cause the female to lay another clutch, while spraying them with oil suffocates the eggs in the nest, but the female continues to incubate them (USDA APHIS 2009). In Colorado, Colorado Parks and Wildlife (CPW) has been issued a special statewide permit by the U.S. Fish and Wildlife Service that allows the DOW to destroy eggs and nests of breeding Canada geese. CPW allows landowners and land managers to conduct egg control activities under the statewide permit and provides training and technical assistance to sub-permittees. Other treatment methods include egg puncturing, and egg shaking (addling). Egg puncturing is done with a long, thin metal object that is punched through the shell, then swirled to break up the material inside. Shaking eggs is the most time consuming, taking 5 to 10 minutes per egg, without a guarantee of success (Swallow et al. 2010).

- **Reproductive Controls:** In addition to destroying eggs, the reproductive inhibitor OvoControl™ G (Innolytics LLC, Rancho Santa Fe, CA) reduces the number of eggs hatched. This chemical is administered, only by licensed specialists, in a bait form. The cost and restrictions associated with this method of management make OvoControl™ G less practical and less efficient than other techniques to control resident goose populations (Swallow et al. 2010).

- **Removal:** If other measures fail, a round-up can be conducted, where geese are netted, removed from the area, and humanely euthanized. In some cases, geese may be live-trapped and relocated (USDA APHIS 2001); however, this is usually ineffective because adult geese will return to the capture site (Holevinski et al. 2006).

- **Lethal Controls:** Hunting is a very effective way of managing goose populations; however, it must be done according to state and federal regulations, as geese are protected under the Migratory Bird Treaty Act of 1918. In suburban and urban areas, however, hunting is usually not an option (CPW 2014).

Questions remain as to the long-term benefits of various control measures. Several case studies suggest that combinations of these control measures can be successful in reducing FIB concentrations. Two examples include:

- The City of New York Department of Environmental Protection (NYCDEP) has had success with a long-term waterfowl management program (beginning in 1993) on their drinking water supply reservoirs, implementing a variety of methods. Their waterfowl management program has included population monitoring, avian deterrence, avian dispersion, and reproductive management methods. Public education related to reducing food sources has also been considered. Representative activities reported by NYCDEP (2004) include:

- Avian deterrence and dispersion measures: techniques to eliminate roosting waterfowl and gulls on the water's surface. Fencing is used to prevent easy access by geese to adjacent feeding areas. Meadow management has converted maintained lawn to tall grass and forbs rendering the vegetation less palatable and creating a less safe environment from predators. NYCDEP also uses Mylar tape grids over docks and other shoreline structures to discourage birds from congregating in these areas.

- Dispersion methods include the use of boats and noisemakers (pyrotechnic devices such as bangers, screamers, etc.). NYCDEP also mentions potential adverse effects of noise on eagle nesting areas. Additionally, empirical studies that indicate that wildlife have the potential to habituate to recurring noise. Distress tapes and red-beam lasers have been explored, but not yet fully tested.

- Reproductive management has included egg puncturing and nest destruction between March and May.

- Capture and removal from reservoirs has also been used.

- In a Pennsylvania study co-sponsored by USDA APHIS, Swallow et al. (2010) monitored three surface water impoundments bi-weekly from May to September along with a single sampling date in both October and November 2009. Two of the impoundments are managed by the USDA, while the third was the unmanaged control site. Measures at the managed ponds included a combination of deterrents (e.g., strobe lights, pyrotechnics, collies), egg oiling, and shrubby or unmown vegetation. Results from fecal coliform testing show strong evidence of the benefits of management with coliform levels up to three times higher in the unmanaged impoundment. Nonetheless, average fecal coliform concentrations remained above primary contact limits at all three ponds (i.e., the managed ponds averaged 438/100 mL and 857/100 mL, with the unmanaged pond at 1,435/100 mL).

The USDA has developed control strategies for other bird species, including pigeons (Williams and Corrigan 1994), blackbirds (Dolbeer 1994), and swallows (Salmon and Gorenzel 2005), as a few examples. Of these birds, pigeons are often a dominant concern in urban areas. Measures listed as alternatives by the USDA-APHIS for pigeons are summarized in Table 7-3. Some of these measures would not be expected to be appropriate in urban areas (e.g., shooting, certain toxicants).

Table 7-3. Summary of Pigeon Control Measures
(Adapted from Williams and Corrigan 1994)

Measure Type	Description
Exclusion	• Screen eaves, vents, windows, doors, and other openings with 1/4-inch (0.6-cm) mesh hardware cloth. • Change angle of roosting ledge to 45 degrees or more. • Attach porcupine wires (Cat ClawTM, Nixalite^{TM}), ECOPIC^{TM}, or Bird Barrier^{TM} to roosting sites. Install electrical shocking device (Avi-Away^{TM}, Flyaway^{TM}, Vertebrate Repellent System [VRS^{TM}]) on roost sites. • Construct parallel or grid-wire (line) systems.
Habitat Modification	• Eliminate food supply. Discourage people from feeding pigeons in public areas. Clean up spilled grain around elevators, feed mills, and railcar clean-out areas. Eliminate standing water.
Frightening	• Visual and auditory frightening devices are usually not effective over long periods of time.
Repellents	• Tactile: various nontoxic, sticky substances (4 -The Birds^{TM}, Hotfoot^{TM}, and Bird-Proof^{TM}, Tanglefoot^{TM}, Roost No More^{TM}). • Odor: naphthalene flakes.
Toxicants	• Consult with local and state agencies on allowed toxicants.
Fumigants	• Generally not practical.
Trapping	• Several live trap designs are effective.
Shooting	• Where legal.
Other Control Methods	• Alpha-chloralose (immobilizing agent used under the supervision of certified personnel only). • Nest removal.

In summary, birds can contribute substantially to FIB loading to receiving waters, posing challenges to MS4 permittees and local governments working toward attainment of numeric water quality limits for FIB. The extent of the impact of birds varies based on site-specific conditions. A variety of source control measures have been developed by state and federal wildlife managers and researchers to help manage the impacts of birds. Selection of control techniques will also vary depending on site-specific conditions. These measures typically require on-going attention and the effectiveness of these measures may vary over time and require adjustments to reduce the likelihood of habituation of the birds to the technique (e.g., harassment measures).

7.8 Urban Wildlife (Mammals)

Urban wildlife can be a key source of FIB loading to urban streams. Fecal matter from wildlife can enter streams through direct overland flow into streams as well as become concentrated by animals living in storm drains and stormwater facilities. Raccoons can be particularly problematic in the storm drain system itself. While it is likely impossible to completely control urban wildlife, there are strategies that can be considered to reduce FIB loading, including:

- Develop a wildlife management plan, working with city wildlife conservation staff and/or state division of wildlife.

- Modify habitat and reduce urban food sources. Raccoon problems may be alleviated by making the habitat less favorable. Because raccoons have fairly large territories, a neighborhood or community-wide effort may be more successful than isolated control measures in urban areas. Removing potential sources of food, water, and shelter is the first step in eliminating the problem. In areas with raccoon activity, garbage cans should be tied down to a solid structure so they cannot be overturned, and lids should be tight fitting, tied or weighted down to deny access to garbage (Pierce 2001). Reduce food sources for urban wildlife through better management of dumpsters, garbage cans and restaurant waste. Additionally, pet food should be stored indoors and pets fed inside (or at least not left out overnight).

- Install storm drain inlet/outlet controls through grates and trash rack. Where raccoons are an issue in storm drains, some communities have successfully reduced end-of-pipe FIB concentrations through installation of grates on storm drain inlets and outlets. These should only be implemented when public safety is not jeopardized by increased flooding or danger of entrapment in a storm sewer. By placing grates on storm sewer inlets, the inlet capacity is reduced, which may require fairly costly retrofitting to maintain design capacities (HDR 2013). The effectiveness of this practice on receiving waters is not well-documented. For example, if grates are only placed on certain drains, then raccoons may simply relocate to other areas, which may also drain to the stream. For example, the home range for male raccoons is 3 to 20 square miles for males, and 1 to 6 square miles for females (Clark 1994), so eliminating a home in one storm drain will likely result in displacement to another nearby location within the home range.

- Clean out storm drains to remove animal waste. When storm drains are power-washed ("jetted"), it is important the discharge be collected by a vacuum truck, otherwise, pollutants are simply flushed into the receiving water.

- Relocate wildlife by trapping. If no other control methods are effective, the problem animals may need to be removed from the area by trapping. In the case of raccoons, there are no poisons or fumigants currently registered for control (Pierce 2001).

For raccoons, there are no chemical repellents registered for controlling or repelling raccoons, although a variety of materials have been tested. Similarly, the use of scare tactics or devices, are not effective or practical in controlling raccoons, particularly in urban areas (Pierce 2001).

When managing urban wildlife, it is important to recognize that states retain primary authority over resident wildlife. When considering possible manipulation of an urban wildlife species, it is important to be aware of the legality of such actions. When in doubt, always contact a wildlife resource agency for consultation (University of Illinois Extension 2014).

Due to uncertainty associated with the effectiveness of these practices on receiving waters, it is important to conduct baseline and follow-up monitoring to assess their effectiveness.

7.9 Irrigation, Car Washing and Power Washing[12]

Over-irrigation, car washing and power washing discharges can mobilize FIB deposited on impervious surfaces, as well as contribute to continually moist conditions in storm sewer systems conducive to biofilms. Public education regarding the water quality impacts of these practices is important for changing public behavior.

Irrigation runoff from lawns, gardens, parks, and other vegetated areas can result in dry-weather nuisance flows with high concentrations of nutrients and also mobilize and transport pollutants accumulated on ground surfaces. The contribution of dry weather inflows from irrigation runoff to a stagnant pool has also been known to foster in-situ bacterial growth (Geosyntec 2010). Effective methods to reduce irrigation runoff may include development of educational outreach, increased inspections, fines for overwatering, tiered water rates, or distribution of smart irrigation controllers and/or other financial incentive programs that decrease watering volume (Geosyntec 2012). By promoting better irrigation runoff management, communities may find that they are able to reduce water waste (increase conservation), as well as improve water quality.

Two studies in Orange County measured the effectiveness of advanced irrigation systems for reducing irrigation runoff. A residential runoff study conducted in five neighborhoods found dry-weather runoff decreased by 50% in areas where weather-based irrigation controllers were installed (IRWD and OCMWD 2004). Berg et al. (2009) found dry-weather runoff reductions of 25% to 50% for a similar study of 4,100 Smart Timers installed in residential and commercial areas. The San Diego River source tracking study (Weston 2009a) found average concentrations of fecal coliforms in dry-weather residential runoff at levels between 100 and 120 MPN/100ml. Besides these concentrations within the irrigation runoff, the increased flows also allow for regrowth in the MS4 and mobilization of pollutants in the MS4 to the receiving waters. Based on these studies, it was assumed that increased irrigation runoff controls, such as inspection, enforcement, and incentives in commercial and residential land uses will generate pollutant load reductions (Geosyntec 2012).

Similarly individual car washing can increase dry weather urban runoff and mobilizes FIB present on impervious surfaces. To reduce FIB loads, educational outreach could be increased to

[12] Recommendations in Sections 7.9 are based on discussion in San Diego Comprehensive Load Reduction Plan, Attachment E Non-structural BMPs, prepared by Geosyntec Consultants (2012).

encourage car owners to minimize washing activities that increase runoff to storm drains (Geosyntec 2012).

7.10 Good Housekeeping/Trash Management (Dumpsters, Restaurants, Garbage Cans)

Good housekeeping practices involve establishing and enforcing ordinances for commercial, industrial and multi-family residential facilities. An ordinance requiring covered trash enclosures and frequent cleaning can help to reduce the FIB load associated with dumpsters.

Programs that address wet weather load reductions may include increased inspection and enforcement of grease removal equipment for restaurants, monitoring trash enclosures for proper waste disposal, and cleaning of private catch basins and drain inlets. The wet weather sources targeted by these BMPs include dumpsters and grease traps. A source tracking study performed in the San Diego River Watershed found that approximately 20% of all dumpsters or grease traps had evidence of liquid leaks. These leaking containers are of especially high importance as a result of the significant concentrations of bacteria in the leaking liquid (Weston 2009a).

Municipalities can also implement restaurant inspection and trash management programs. Uncontained restaurant and grocery store wastes can be a significant FIB source in urban runoff, especially during wet weather. An expanded education and outreach program would increase restaurant and store operator awareness of this potential FIB source and provide solutions to trash management concerns.

7.11 Mobile Sources of Human Waste: Portable Toilets and RV Dumping

Temporary sources of FIB can include portable toilets and illicit RV dumping. The relevance of these sources to FIB impairments is dependent on the particular watershed.

BMPs for portable toilets should address site location cleanout frequency and transportation/hauling requirements. The location where the portable toilet is placed is particularly important. Guidelines for portable toilets placement could include requirements such as:

- Locate portable toilets away from high-traffic vehicular areas.
- Locate portable toilets at least 20 feet away from all storm drains: never locate a portable toilet on top of a storm drain inlet. Place portable toilets on a level ground surface that provides unobstructed access to users and servicing pump trucks.
- Wherever possible, local portable toilets on natural ground and not on or within 5 feet of a paved surface such as asphalt, concrete or similar.

Improperly placed portable toilet with biocide running down gutter toward storm drain. (Photo courtesy of Wright Water Engineers.)

- If portable toilets must be placed on a paved surface exposed to rainwater or stormwater runoff, extra care must be taken during servicing to ensure any wastewater spilled onto the paved surface is rinsed and adequately collected so as not to leave any residue. A wet shop vacuum or similar device would provide for adequate collection.

- As a minimum, portable toilets should not be located within the 75 foot buffer of any stream or lake, or within any other larger stream/lake buffer that may have been established.

For an example of a portable toilet BMP fact sheet, see https://www.gwinnettcounty.com/static/departments/publicutilities/pdf/WQ-04%20Portable%20Toilet%20Management-%20Final.pdf).

Illicit RV dumping to storm drains can be managed in recreational areas by providing public education on appropriate practices, publicizing RV dump locations, by proving a citizen's reporting hotline, and by publicizing fines (e.g., $1,000 fine for illegal dumping in San Diego). Educational materials can include tips such as:

- Use only designated dump stations.

- Never dump into the curb, gutter or sand.

- Connect to sewer with the correct size hose, and an airtight connection.

For an example of an RV dumping brochure, see the brochure developed by "Think Blue San Diego": http://www.sandiego.gov/thinkblue/pdf/rvdumpcard.pdf.

7.12 Septic Systems and Other Onsite Wastewater Treatment Systems

Onsite wastewater treatment systems (OWTSs) include a variety of on-site systems for the collection, storage, treatment, neutralization, or stabilization of sewage that occurs on a property. In some cases, OWTSs are present in urbanized areas, particularly within urban growth boundaries in areas near city limits. OWTSs include traditional septic systems, as well as other small on-site treatment systems.

In addition to approving and tracking OWTS permits, local governments can provide guidance on OWTS maintenance and on signs of failing OWTSs. As an example, Boulder County, Colorado operates a "Septic Smart" program that provides guidance to septic system owners about signs of failing septic systems, including:

- Test results of well water show the presence of bacteria.

- The ground in the area is wet or soggy.

- Grass grows greener or faster in the area.

- Sewage odors in the house or yard.

- Plumbing backups into the house.

- Slowly draining sinks and toilets.
- Gurgling sounds in the plumbing.

If one or more of these warning signs exist, Boulder County recommends that the homeowner should contact a licensed septic system cleaner to have the system inspected and pumped. Additionally, the County recommends that homeowners have septic tanks pumped out by a licensed OWTS cleaner every three years. Additionally, in order to optimize outreach and public education related to potentially problematic OWTSs, the county has inventoried OWTS locations using GIS and ranked and prioritized permitted sites, high risk sites, etc. For more information on this Septic Smart program, see:
http://www.bouldercounty.org/env/water/pages/qandaows.aspx.

7.13 Homeless Encampment Outreach and Enforcement

As discussed in Chapter 3, homeless encampments and gathering areas can be a source of human waste posing potential human health risks in recreational waters. Homelessness is a serious social issue in many communities and often a sensitive public policy issue that stormwater and water resource managers have limited experience in addressing. Based on experience gained in Southern California addressing this issue (Geosyntec 2014), recommendations for an effective homeless encampment enforcement/outreach program may include:

- Collaboration with other agencies.
- Targeted MS4 channel cleanups.
- Enhancing programs to reduce the number of homeless people in encampments.
- Establishing ordinances that reduce encampments.
- Enforcing new and existing laws to decrease the negative impact on water quality.

The Contra Costa County Flood Control and Water Conservation District undertook an extensive research project to understand the best approaches for addressing water quality pollution from homeless encampments (DeVuono-Powell 2013). They found collaboration with other agencies to be the most effective approach for addressing the long-term concerns of homeless encampments. The report lists the following potential stakeholders to include in collaboration efforts:

- Housing
- Homeless community
- Homeless advocates
- Health services: medical, mental
- Community residents, local business
- Politicians
- Engineering: roads, maintenance, drainage
- Law: civil, civil rights, criminal, law enforcement
- Environmental regulations

- Utilities
- Environmental organizations/volunteers
- City/county management
- Social justice
- Family counseling
- Environmental management: trash, chemicals, waste
- Public relations media
- Substance abuse services, programs
- City/county land use and planning
- Parks and open space agencies
- Charitable organization volunteers

Existing targeted cleanup projects in the San Diego River watershed include the Forester Creek Homeless Encampment Removal Project, which involved police sweeps of transient camps and subsequent cleanup. This activity removed 14 cubic yards of debris during fiscal year 2009-2010. Similar sweeps and cleanup events are conducted in the City of Santee in the San Diego Riverbed and have removed 5,000 pounds of trash. The San Diego River Park Foundation works collaboratively with local park rangers, police, and volunteers to identify and remove homeless encampments, and document activities.

Options to reduce water quality impacts of homeless encampments can also be combined with efforts to reduce homelessness. One example is a grant-funded pilot program on Coyote Creek in San José, CA that employs homeless persons living in creek encampments to remove trash and litter and to engage in peer-to-peer outreach with others living in the encampment. Participants are housed temporarily and given food vouchers, case management services, employment skills, and assistance at transitioning to permanent housing (EPA 2011).

Targeted enforcement during the night hours is of special importance, in order to cite and fine those caught camping illegally.

7.14 Conclusions

For cases where human sources of FIB are present, controlling and correcting the source of the contamination is a basic first step for protecting human health. Once these sources are corrected, diffuse non-human sources typically remain and can be challenging to control. Nonetheless, tools such as pet waste ordinances, bird controls, other urban wildlife controls, and storm sewer maintenance activities are tools that should be considered by local governments working to reduce FIB, depending on the sources of elevated FIB in the particular watershed. Limited data are available to evaluate the effectiveness of source controls quantitatively. Monitoring studies related to source controls should be encouraged and submitted to a publically available repository such as the International Stormwater BMP Database, so that stronger performance expectations for these practices can be developed. Effectiveness of source controls on reducing instream FIB is dependent on the dominant sources of FIB in a watershed and the consistency with which source controls are implemented.

8 STRUCTURAL STORMWATER CONTROL PRACTICES

Structural stormwater control practices (or BMPs) are typically implemented to reduce FIB loading to streams under wet weather conditions. This chapter provides an overview of unit treatment processes associated with common passive structural stormwater controls, provides an overview of expected structural BMP performance with regard to FIB, briefly discusses active treatment, and provides several case studies of stormwater control monitoring studies. The majority of these practices are oriented to wet weather flows; however, the brief discussion of active treatment also pertains to dry-weather flows associated with MS4s.

The discussion of passive stormwater control performance in this chapter is based primarily on the International Stormwater BMP Database (www.bmpdatabase.org), which is collaboratively sponsored by the Water Environment Research Foundation (WERF), the Federal Highway Administration (FHWA), the Environmental and Water Resources Institute (EWRI) of the American Society of Civil Engineers (ASCE), and the U.S. Environmental Protection Agency (EPA). The vision for the BMP Database originated with the Urban Water Resources Research Council of EWRI. The BMP Database continually evolves as new stormwater control performance monitoring studies are uploaded each year. The analysis in this chapter is based on the data set available as of January 2014 and draws upon previous discussion and analysis presented in *International Stormwater Best Management Practices (BMP) Database Pollutant Category Summary: Fecal Indicator Bacteria* (Wright Water Engineers and Geosyntec 2010), accessible at www.bmpdatabase.org.

8.1 Unit Treatment Processes in Structural Stormwater BMPs

Removal mechanisms for FIB in stormwater control practices include both passive and active processes. This chapter focuses on passive stormwater treatment in Sections 8.1 through 8.5, discussing active treatment separately in Section 8.6. Based on a literature review conducted for the WERF Stormwater Challenge (Strecker et al. 2009), the dominant passive removal mechanisms for FIB include natural inactivation, predation, inert filtration and sedimentation, sorption and chemical inactivation (via contacting products) and are closely tied to the transport and fate discussion in Chapter 5. Key passive pollutant removal processes that may be present in various stormwater control types are described below (Strecker et al. 2009, Leisenring et al. 2013, WERF 2007).

- **Natural inactivation** is a general removal mechanism that refers to FIB die-off or inactivation due to a wide range of environmental factors. Unless provided with suitable conditions for reproduction, the number of live cells will tend to decrease with time. Growth and decay rates are highly dependent on environmental factors, which are continually changing. The most important environmental factors affecting rate of inactivation are exposure to sunlight, water temperature, and exposure to air (drying or desiccation). Additionally, FIB bound to particulates have been found to be inactivated at slower rates because particulates are hypothesized to provide both nutrients and shelter (WERF 2007).

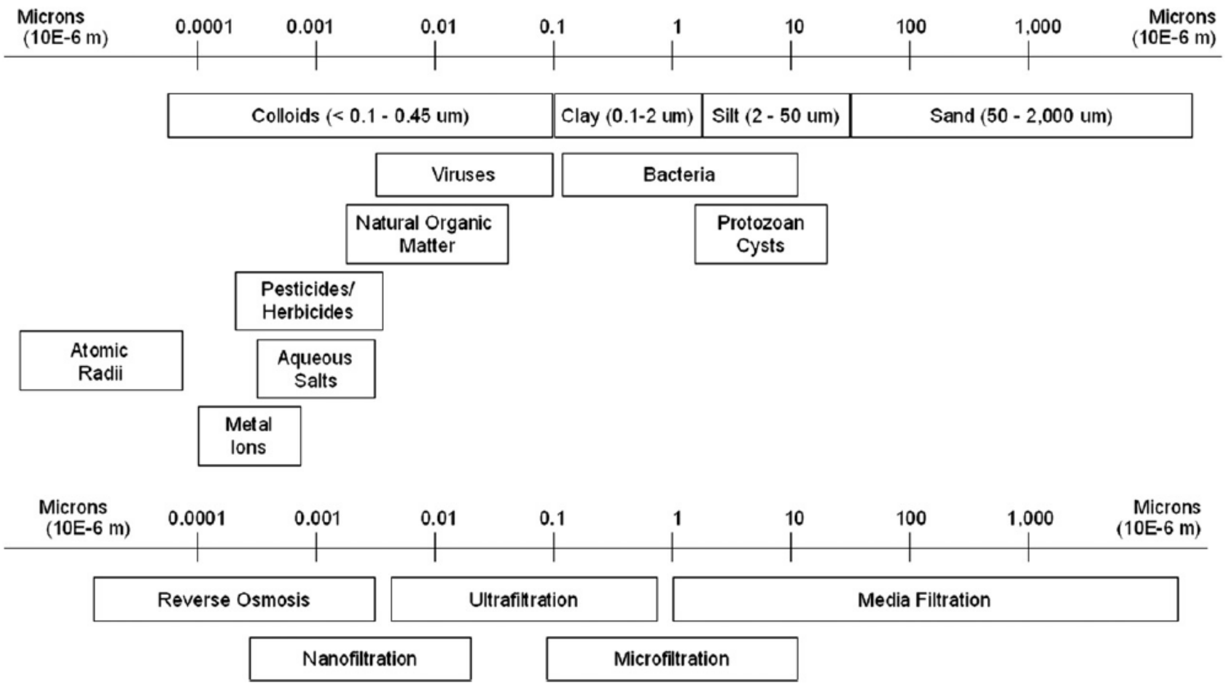
- **Predation** of FIB by other microorganisms is interrelated with natural inactivation and has been found to be a major removal mechanism. The most important predators of FIB are believed to be protozoa and other eukaryotic organisms. Studies have found that predation may account for approximately 90 percent of overall mortality rates of FIB (WERF 2007). Additional studies such as Zhang et al. (2011) have begun to explore changes in microbial ecology in bioretention cells, but more research is needed in this area.

- **Inert filtration[13] and sedimentation** of solids are mechanisms that would be expected to remove FIB bound to particulates from the water column. The effectiveness of particle removal at reducing FIB concentrations is a function of the partitioning of FIB between particulate-bound and free-floating forms, and the association of FIB across the particle size distribution (Figure 8-1). Once again, the removal of FIB from the water column through sedimentation or filtration does not necessarily constitute an ultimate removal mechanism because the survival of FIB is expected to be greater when FIB are bound to sediment, and resuspension of communities of FIB sheltered by sediment could represent a significant later source of FIB in some systems. Barfield et al. (2010) apply a straining model for pathogen trapping in bioretention cells and sand filters, and Hayes et al. (2008) discuss the application of the these processes in the Integrated Design Evaluation and Assessment of Loadings (IDEAL) model. In their predictions, typical trapping efficiencies for sand filters and bioretention cells are in the range of 60 to 80% for well-designed devices, with trapping efficiency decreasing as untreated runoff bypasses the devices and is discharged through the overflow structures during periods of high flows or when the filter is clogged.

  Additionally, Clark and Pitt (2012) note that most bacteria are in the lower limits of the size range for effective physical filtration using a sand medium. However, as the filter ages, removals will tend to increase, partly due to reduction in the effective pore size and due to the exopolymers that many bacteria excrete. These exopolymers provide surface reactive sites, even on a relatively insert sand media. Because of their negative surface charge, bacteria can be removed by attaching to these surface reactive sites. Organic media provide a location for captured bacteria to reside and grow (with potential for predation, as well). The challenge in filtration media selection is to encourage capture and potential growth to create reactive sites, but without excessive growth that sloughs off the media and is flushed out of the media with successive storms.

[13] Inert filtration includes physical filtration processes, but does not encompass sorption and other chemical-physical processes that may occur in filter media.

Figure 8-1. Particle Sizes of Viruses, Bacteria and Protozoan Cysts
(Source: Clark and Pitt 2012)



- **Sorption** (the bonding of microorganisms to the surface of particles) is believed to be controlled by steric, electrostatic, and hydrophobic interactions. As described in WERF (2007), steric interactions arise between macromolecules, electrostatic interactions are based on surface charge, and hydrophobic interactions result from the polarity and non-polarity of organic molecules. Researchers have found that some bonds are "irreversible" but that many bonds that occur between FIB and particulates are reversible if conditions change or if physical forces such as fluid shear forces are applied. Even bonds considered to be irreversible can be broken under high turbulence and fluid shear. Partitioning of FIB to particles is expected to depend on a variety of environmental factors, stormwater characteristics and hydrodynamics and is expected to change drastically with time and likely from site to site.

- **Chemical inactivation** of FIB through contact with antimicrobial products is an approach used in a variety of proprietary BMPs. A common agent in these types of treatment devices is an organosilane derivative (C-18 organosilane quaternary), which is reported to inactivate most FIB without being consumed or dissipated and without producing toxic byproducts (Nolan et al. 2004). It is presumed that effectiveness of stormwater controls relying on a fixed microbial agent would depend on the degree of contact and contact time between stormwater and the microbial agent, dilution, and the amount of FIB bound to particulates. It is not clear whether C-18 organosilane degrades over time and needs to be recharged/replaced. If so, the time since installation or last maintenance would be expected to influence the effectiveness of such proprietary devices. Silt films on the microbial agent would also be expected to decrease their performance.

Although inactivation through addition of chemicals such as chlorine and ozone is known to be effective for sanitary wastewater treatment and some CSO and SSO situations, application is limited in the separate stormwater (MS4) arena due to ongoing operational requirements involving less control than occurs at sanitary facilities and the potential for disinfection byproducts (in the case of chlorine). (See Section 8.6 for additional discussion.)

In addition to these treatment mechanisms, volume-related management practices, such as infiltration, reduce FIB loads reaching waterbodies by controlling the volume component associated with pollutant loading in runoff. For considerations related to groundwater contamination associated with stormwater infiltration, see Pitt et al. (1994).

8.2 FIB Data Summary for the International Stormwater BMP Database

As of January 2014, the International Stormwater BMP Database contained over 5,800 sample results for FIB, including fecal coliform, *E. coli*, fecal streptococus and total coliform. Of these, *E. coli* and enterococcus are of primary interest for purposes of assessing attainment of Recreational Water Quality Criteria; however, some states still use fecal coliform as well. Performance summary reports for FIB were completed in 2010 and 2012 as part of the BMP Database project and are accessible at www.bmpdatabase.org, along with the underlying data sets used for analysis (Wright Water Engineers and Geosyntec 2010, 2012). Since publication of these reports, additional data submissions have resulted in significant growth of the FIB data set; nonetheless, the majority of available data are for fecal coliform and the data sets remain relatively limited for some stormwater control categories. Data available in the BMP Database as of 2014 were queried to prepare updated performance information for this report.

Table 8-1 provides an overview of the number of studies included in the analysis by BMP category and FIB analyte, along with a summary of states where the studies were conducted. Some data sets are dominated by certain regions of the country or certain land use types. For example, most of the bioretention data sets are from monitoring conducted in North Carolina, most of the composite (treatment trains) were in Austin, TX, and most of the manufactured devices are associated with parking lots or highway settings. Conversely, retention ponds, dry detention basins and sand filter data sets are provided for multiple regions of the U.S. More in-depth analysis could be conducted to assess the effects of site-specific conditions on both influent and effluent concentrations, which go beyond the scope of the general characterization included in this report. Generally, stormwater control category data sets with more stormwater controls, more storm events and multiple states with varying climate conditions are considered to be most reliable. There is a clear need for more enterococcus and *E. coli* monitoring data for most stormwater control device categories.

Only one disinfection practice is currently included in the BMP Database. Los Angeles County Department of Public Works installed this treatment system in 2007 in the City of Malibu, CA, at the outlet of the Marie Canyon drainage area. The system is designed to treat 100 gpm of dry weather flows from the storm sewer system. This $1.15 million project incorporates a three-stage treatment process including three dual media filters (sand and anthracite), three activated media filters (organo-clay) and two ultraviolet disinfection units (as described in www.bmpdatabase.org). Although data for this study are included in the tables and figures that follow, it is important to recognize that the data set is based on dry weather monitoring events, as opposed to wet weather flow events monitored in the other data sets in the BMP Database.

Table 8-1. Summary of BMP Database Studies Included in Analysis by FIB Type and State

BMP Code	BMP Type Description	Number of Studies with FIB Data			States
		Enterococcus	*E. coli*	Fecal Coliform	
BI	Biofilter - Grass Strip			2	TX
BR	Bioretention	3	3	2	NC, DE
BS	Biofilter - Grass Swale		5	8	OR, FL, WA
CO	Composite - Treatment Train			5	TX, CA
DB	Detention Basin (Dry) - Surface Grass-Lined Basin	1	3	11	CA, CO, FL, GA, NY, OR, TX
DO	Detention Basin (Dry) - Other, Concrete or Vault			3	CA, TX
FO	Filter - Other Media			4	CA, FL
FS	Filter - Sand	1	1	13	CA, DE, OR, TX
IB	Infiltration Basin			1	CA
MD	Manufactured Device	7		10	CA, DE,TX
MD-Dis	Disinfection System	1		1	CA
RP	Retention Pond (Wet)	2	5	12	CA, FL, GA, NC, ON, TX
WB	Wetland Basin	3	3	3	CA, NC, OR, TX
WC	Wetland Channel			3	CA
Total Studies		**18**	**20**	**78**	

Consistent with previously published analyses for the BMP Database, after the FIB data at inflow and outflow locations for each stormwater control were retrieved from the BMP Database (January 2014), a series of data screening decisions were made with regard to data sets considered appropriate for further analysis based on these criteria:

- Studies with less than five storm events monitored at the outflow from the stormwater control were excluded from the analysis. In part, the five-storm threshold was selected since many states require a minimum of five sampling events for calculation of a geometric mean. Independent researchers may choose alternate thresholds, if desired. Ideally, for statistical hypothesis testing, many more sampling events based on event mean concentrations (EMCs) would be included; however, choosing a higher threshold would result in inclusion of a smaller number of studies in the analysis.

- No further analysis of total coliform or fecal streptococcus data was conducted. Relatively few studies for these two FIB types are present in the BMP Database, and most of these studies also monitored fecal coliform or *E. coli*, so data analysis is focused on the more commonly reported FIB instead.

- Category-level summaries of stormwater control performance were generated for the remaining data sets; however, data sets with only a few installations should be used with caution. In particular, many of the categories have only few studies for enterococcus and *E. coli*. More robust data sets (in terms of study numbers and events) include fecal coliform for grass swales, extended dry detention basins, sand filters, manufactured devices, and retention (wet) pond categories. Although the bioretention studies include relatively few study sites, the sites with available data have a reasonable number of storm events.

In addition to these screening-level decisions, the following limitations of the data set are acknowledged:

- Because much of the FIB data reported were based on grab samples, grab samples were included in this analysis. (Typically, the BMP Database analysis for other constituents is based on EMCs only.) EMCs developed based on samples collected through the duration of the storm hydrograph are considered most appropriate for characterizing stormwater pollutants. In the case of FIB, EMCs are often not collected due to the recommended maximum 6-hour sample hold time for FIB analyses. (See discussion in Section 5.4.1 for additional discussion of hold times.) As a result, most of the data in the BMP Database are grab samples. Some sites may have one grab sample, whereas others may have multiple samples throughout the storm event. This increases uncertainty as to whether the samples are representative of the EMC for the storm, which in turn creates uncertainty when calculating loads or assessing load reduction due to infiltration-oriented practices. For sites where researchers reported multiple, uncomposited grab samples, the average of the samples was used to represent performance for the storm event.

- From a statistical analysis perspective, an additional complication relates to variation in upper and lower quantitation limits (censored data) both within and between studies. Specifically, lab analysts seek a balance when diluting samples to provide characterization of high and low ends of the expected sample result range. Some analysts may reach "too numerous to count" at 2,400/100 mL, whereas other studies may reach this determination at 240,000/100 mL (or greater). These variations in quantitation limits make absolute characterization of influent and effluent concentrations more challenging, as well as comparisons of performance among BMP studies with different upper quantitation limits. (See discussion in Chapter 6).

- Stated more generally, sample collection, processing and culture-based analysis methods for FIB have well known limitations. In addition to the holding time and dilution issues mentioned above, sample contamination can be an issue and culture-based test methods have shortcomings. For example, culture-based methods generally involve collection of a water sample, filtering the water sample through a membrane, placing the membrane in a medium, incubating the sample, then counting and recording the number of FIB colonies. Very small samples are collected (with respect to the storm and with respect to sample volumes for other constituents) and relatively large dilutions are typically needed to obtain a countable number of colonies. The small sample volumes are further split into much smaller subsamples (100 mL), then filtered and counted manually. The nature of the sampling, processing and analysis methods are therefore subject to large amounts of variability. These are in addition to the large variability expected between sites and among different antecedent conditions and storm event properties.

- Widely varying sample results at the same sample location, even during the same day lead to large variation in data sets and wide confidence limits for measures of central tendency. These data characteristics make statistical hypothesis testing challenging without much larger data sets (i.e., it is difficult to draw statistically significant conclusions with highly variable data sets). For guidance on how many samples may be needed to draw statistically significant conclusions at varying levels of confidence and power, see Appendix D of the *Urban Stormwater BMP Monitoring Manual* (http://www.bmpdatabase.org/MonitoringEval.htm). The number of sampling events needed will vary depending on study objectives and site-specific conditions.

- Seasonal distribution of samples may affect conclusions drawn related to stormwater control performance. For example, winter concentrations of FIB may be lower than summer concentrations (Shergill and Pitt 2004, Colorado *E. coli* Work Group and WWE 2010, Hathaway and Hunt 2012).

- All of the monitoring data in the BMP Database and the majority of the monitoring routinely conducted by most communities are targeted to FIB. Much remains unknown with regard to the relationship of FIB to the wide range of human pathogens that may be present in urban runoff.

- Data sets for some stormwater control categories may be dominated by particular types of locations or be limited geographically, as shown in Table 8-2. For example, the manufactured device studies reporting fecal coliform are dominated by two locations: the Delaware Department of Transportation I-95 Service Plaza (6 stormwater controls) and California Department of Transportation (Caltrans) sites (3 stormwater controls).

- Lastly, analyses are limited to stormwater control types voluntarily submitted to the BMP Database and some types of controls are not currently well represented in the BMP Database. For example, only one UV-disinfection control practice is included, and no subsurface gravel wetlands are included. Although some data are included in the BMP Database for permeable pavement and green roofs, these data sets only include a few samples for a few installations, so were excluded from the analysis.

Recommendations for appropriate uses of the BMP Database data set:

- The BMP Database FIB data set can be used for general characterization of stormwater control performance for selected categories. Due to significant variability in the data sets (and the lack of composite data representing complete events), it is important that central tendency statistics (e.g., geometric mean, median) also include measures of variability such as the interquartile range or confidence limits associated with such estimates. Where possible, there may be a benefit to conducting further analyses on the data to select more locally appropriate data sets for evaluation. For example, it may be appropriate to investigate the design parameters for specific control device types using a reduced set of data for the analyses based upon conformance to local design standards.

- For the purposes of FIB modeling, the BMP Database Project Team does not recommend relying solely on the empirical data summaries for estimating the effluent concentrations for control practices for FIB. Rather, this information may be used as a check on the reasonableness of results from more physically based modeling approaches that consider FIB decay and other unit treatment processes (e.g., sedimentation, filtration, etc.). When possible, regional or site-specific data should be used to calibrate and validate physically based models (WWE and Geosyntec 2010).

8.3 FIB Statistical Summary for the International Stormwater BMP Database

Tables 8-2 through 8-4 provide selected summary statistics for data sets included in this analysis, followed by boxplots corresponding to these summary statistics in Figures 8-2 through 8-4. To graphically illustrate the central tendencies and ranges of FIB concentrations observed for the inflow and outflow for each control practice category, boxplots were completed for fecal coliform (Figures 8-2a and 8-2b), enterococcus (Figure 8-3), and *E. coli* (Figure 8-4). In the boxplots, the inflow is provided in the first box and the outflow is provided in the second box (in bold) above each treatment category. Concentrations of FIB are shown on a logarithmic scale. Table 8-5 provides the results of Mann-Whitney hypothesis tests for statistically significant differences between inflow and outflow for each treatment category and FIB type. For purposes of this analysis, $p = 0.10$ was selected to identify significant statistical differences due to the preliminary nature of these analyses and the lack of large amounts of data, in contrast to the normally used critical p value of 0.05. Figure 8-5 provides a cumulative frequency distribution of the outflows for selected stormwater control categories for fecal coliforms to provide supplemental information on the likelihood of effluent from various stormwater control categories meeting various benchmarks (such as instream standards).

Table 8-2. Selected Summary Statistics for Fecal Coliform for BMP Database Studies
(accessed January 2014)

BMP Category	BMP-Flow Type[1]	No. of Events	Geometric Mean	Min	Max	1st Quartile	Median	3rd Quartile	Mean	COV
	Fecal Coliform (#/100 mL); Primary Recreational Contact Geometric Mean Criteria = 200/100 mL									
Biofilter, Grass Strip	BI-In	79	**5,497**	1	2,200,000	1,000	9,889	100,000	115,256	2.6
	BI-Out	100	**26,003**	240	1,890,000	4,100	19,600	181,748	202,790	1.9
Bioretention	BR-In	27	**3,355**	1	160,000	460	5,000	27,000	22,705	1.6
	BR-Out	30	**886**	19	160,000	100	750	4,500	11,390	2.7
Biofilter, Grass Swale	BS-In	71	**3,755**	4	2,000,000	1,255	4,200	24,000	58,397	4.2
	BS-Out	71	**4,777**	19	1,100,000	1,453	5,397	21,000	37,523	3.6
Composite, Treatment Train	CO-In	75	**8,046**	1	282,019	2,530	11,850	28,664	24,302	1.6
	CO-Out	73	**3,738**	9	60,768	973	6,980	18,657	11,547	1.1
Detention Basin (grass, dry)	DB-In	162	**2,218**	1	330,600	505	2,497	18,219	18,860	2.2
	DB-Out	165	**639**	1	138,000	70	700	5,750	8,021	2.3
Detention Basin (other, concrete)	DO-In	36	**8,570**	106	324,893	1,890	11,051	51,224	35,701	1.7
	DO-Out	22	**5,057**	2	551,558	1,123	10,407	45,449	46,491	2.4
Filter, Other Media	FO-In	31	**618**	8	13,000	200	350	4,300	2,635	1.4
	FO-Out	30	**350**	2	3,000	170	515	1,318	884	1.1
Filter, Sand	FS-In	157	**1,463**	2	430,000	200	1,600	11,600	16,533	3.0
	FS-Out	150	**632**	2	98,224	110	593	7,819	7,174	2.0
Infiltration Basin	IB-In	8	**36,257**	800	2,400,000	12,750	37,000	142,500	360,475	2.2
	IB-Out	8	**13,723**	80	280,000	18,033	40,000	97,500	84,276	1.2
Disinfection System	Dis-In	80	**1,158**	80	90,000	450	1,050	2,550	4,318	3.2
	Dis-Out	64	**17**	10	220	10	10	20	28	1.5
Manufactured Device	MD-In	104	**1,478**	13	160,000	200	1,300	5,000	9,706	2.6
	MD-Out	110	**2,504**	80	160,000	325	2,300	10,250	19,368	2.1
Retention Pond (Wet)	RP-In	152	**2,930**	1	964,860	775	3,200	23,224	32,978	3.1
	RP-Out	162	**637**	1	1,770,741	64	1,500	6,570	21,964	6.5
Wetland Basin	WB-In	24	**3,673**	10	41,424	2,125	6,930	15,570	11,096	1.0
	WB-Out	23	**1,115**	10	44,845	105	1,900	15,373	9,209	1.4
Wetland Channel	WC-In	80	**357**	1	2,400	110	933	2,400	1,154	0.9
	WC-Out	53	**247**	2	2,400	33	540	1,600	937	1.0

[1]See Table 8-1 for abbreviations.

Table 8-3. Selected Summary Statistics for Enterococcus for BMP Database Studies (accessed January 2014)

BMP Category	BMP-Flow Type[1]	No. of Events	Geometric Mean	Min	Max	1st Quartile	Median	3rd Quartile	Mean	COV
	Enterococcus (#/100 mL); Primary Recreational Contact Geometric Mean Criteria = 35/100 mL									
Bioretention	BR-In	48	**799**	30	160,000	178	587	2,437	13,672	2.9
	BR-Out	50	**291**	1	160,000	37	234	2,065	8,602	3.3
Detention Basin (grass, dry)	DB-In	13	**5,505**	119	43,700	2,420	10,100	18,400	13,720	1.0
	DB-Out	13	**1,469**	1	198,600	1,120	2,420	4,200	17,626	3.0
Filter, Sand	FS-In	10	**927**	200	50,000	200	550	2,050	6,250	2.4
	FS-Out	10	**504**	200	5,000	200	300	875	1,040	1.4
Disinfection System	Dis-In	80	**1,498**	40	30,000	450	1,300	5,000	6,174	1.6
	Dis-Out	64	**13**	10	40	10	10	20	14	0.6
Manufactured Device	MD-In	65	**3,850**	100	199,000	630	3,000	24,000	33,627	1.7
	MD-Out	68	**4,783**	10	240,000	1,250	5,000	25,000	30,219	1.8
Retention Pond (Wet)	RP-In	33	**729**	2	24,196	160	496	4,839	4,271	1.7
	RP-Out	33	**76**	1	24,196	6	63	870	1,554	2.9
Wetland Basin	WB-In	45	**651**	10	24,196	201	690	2,407	2,606	2.0
	WB-Out	45	**265**	1	29,090	22	284	1,842	2,883	2.3

[1]See Table 8-1 for abbreviations.

Table 8-4. Selected Summary Statistics for E. coli for BMP Database Studies (accessed January 2014)

BMP Category	BMP-Flow Type[1]	No. of Events	Geometric Mean	Min	Max	1st Quartile	Median	3rd Quartile	Mean	COV
	E. coli (#/100 mL); Primary Recreational Contact Geometric Mean Criteria = 126/100 mL									
Bioretention	BR-In	54	145	1	7,701	42	135	1,821	1,121	1.6
	BR-Out	54	60	1	19,863	5	30	965	1,539	2.5
Biofilter, Grass Swale	BS-In	39	1,440	4	41,000	295	3,500	11,000	9,270	1.4
	BS-Out	39	2,365	11	40,000	1,200	4,100	10,000	8,993	1.4
Detention Basin (grass, dry)	DB-In	42	1,011	1	198,600	333	850	4,500	14,184	2.6
	DB-Out	42	283	1	22,800	63	370	1,700	2,167	2.2
Filter, Sand	FS-In	5	2,099	105	15,500	830	2,600	11,605	6,128	1.0
	FS-Out	5	79	10	280	72	98	160	124	0.7
Retention Pond (Wet)	RP-In	87	6,580	10	16,621,000	686	3,466	29,028	799,060	3.2
	RP-Out	84	726	1	12,400,000	23	393	5,225	352,426	4.0
Wetland Basin	WB-In	42	681	5	14,136	257	714	2,509	2,516	1.5
	WB-Out	42	539	6	36,540	65	622	3,577	3,822	2.0

[1]See Table 8-1 for abbreviations.

Figure 8-2a. Boxplots of Fecal Coliform Data from the Stormwater BMP Database (part 1)

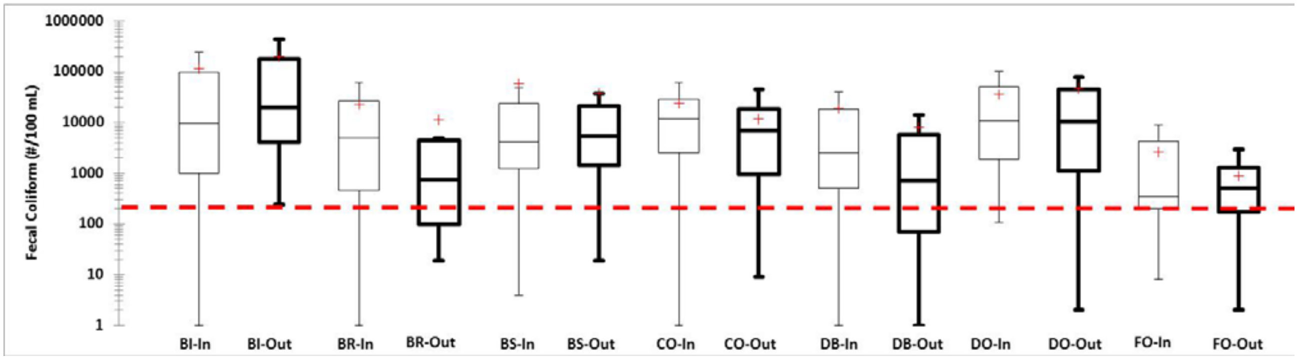


Figure 8-2b. Boxplots of Fecal Coliform Data from the Stormwater BMP Database (part 2)

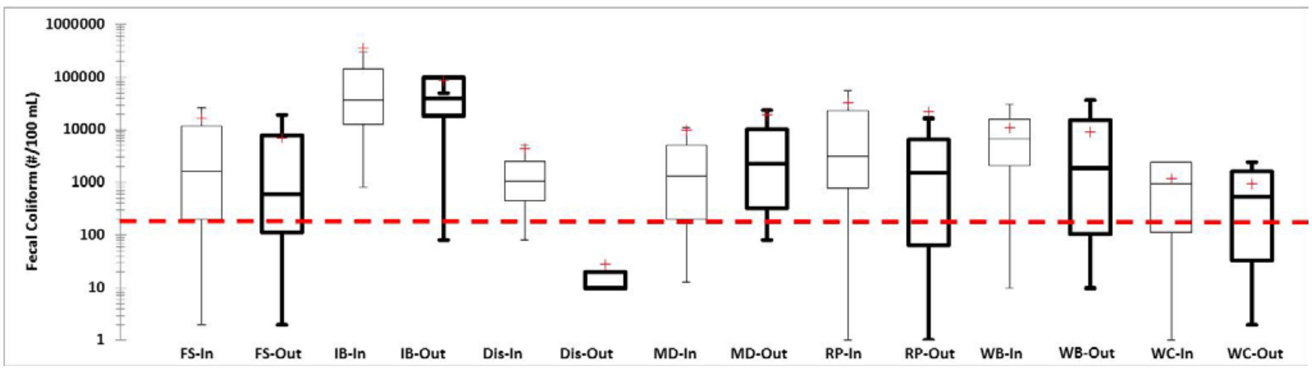
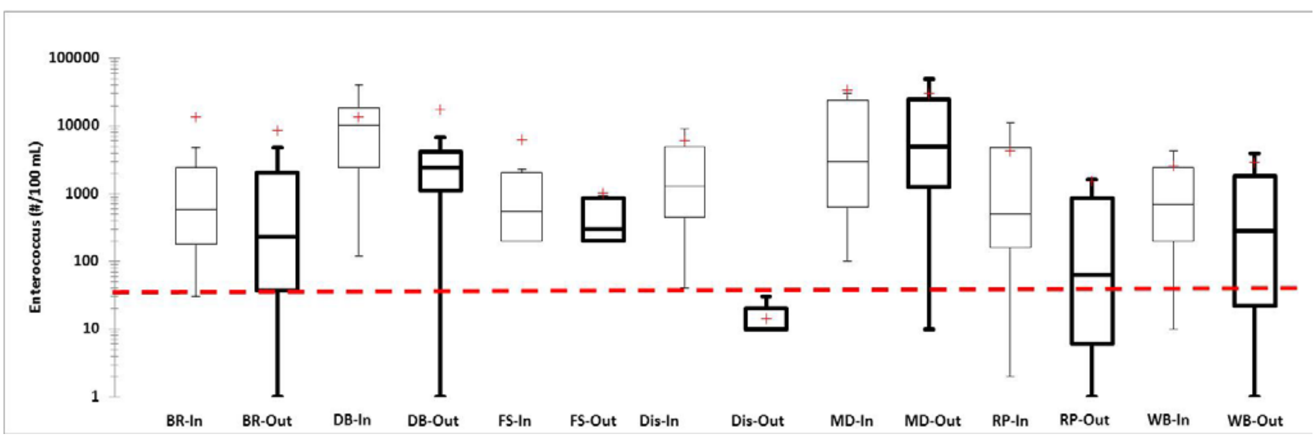
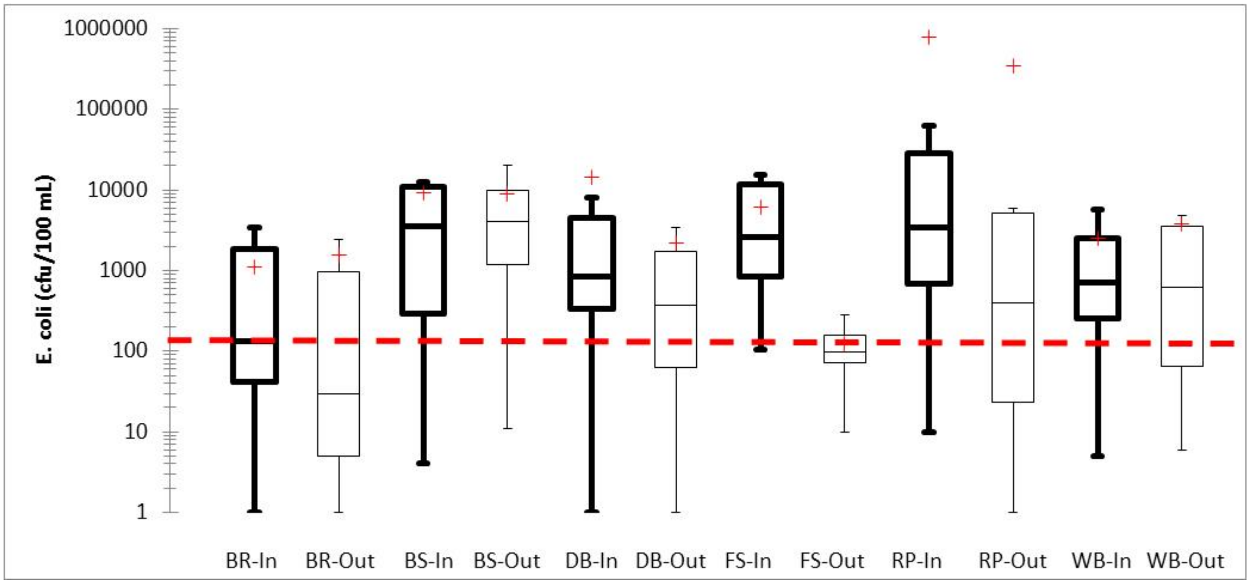


Figure 8-3. Boxplots of Enterococcus Data from the Stormwater BMP Database



Note: See Table 8-1 for abbreviations and Tables 8-2 through 8-3 for corresponding data. Red dashed lines correspond to geometric mean primary contact recreational water quality criteria recommended by EPA, including: 200 cfu/100 mL for fecal coliform (historically recommended), 126 cfu/100 mL for *E. coli* and 35 cfu/100 mL for enterococci.

Figure 8-4. Boxplots of *E. coli* Data from the Stormwater BMP Database



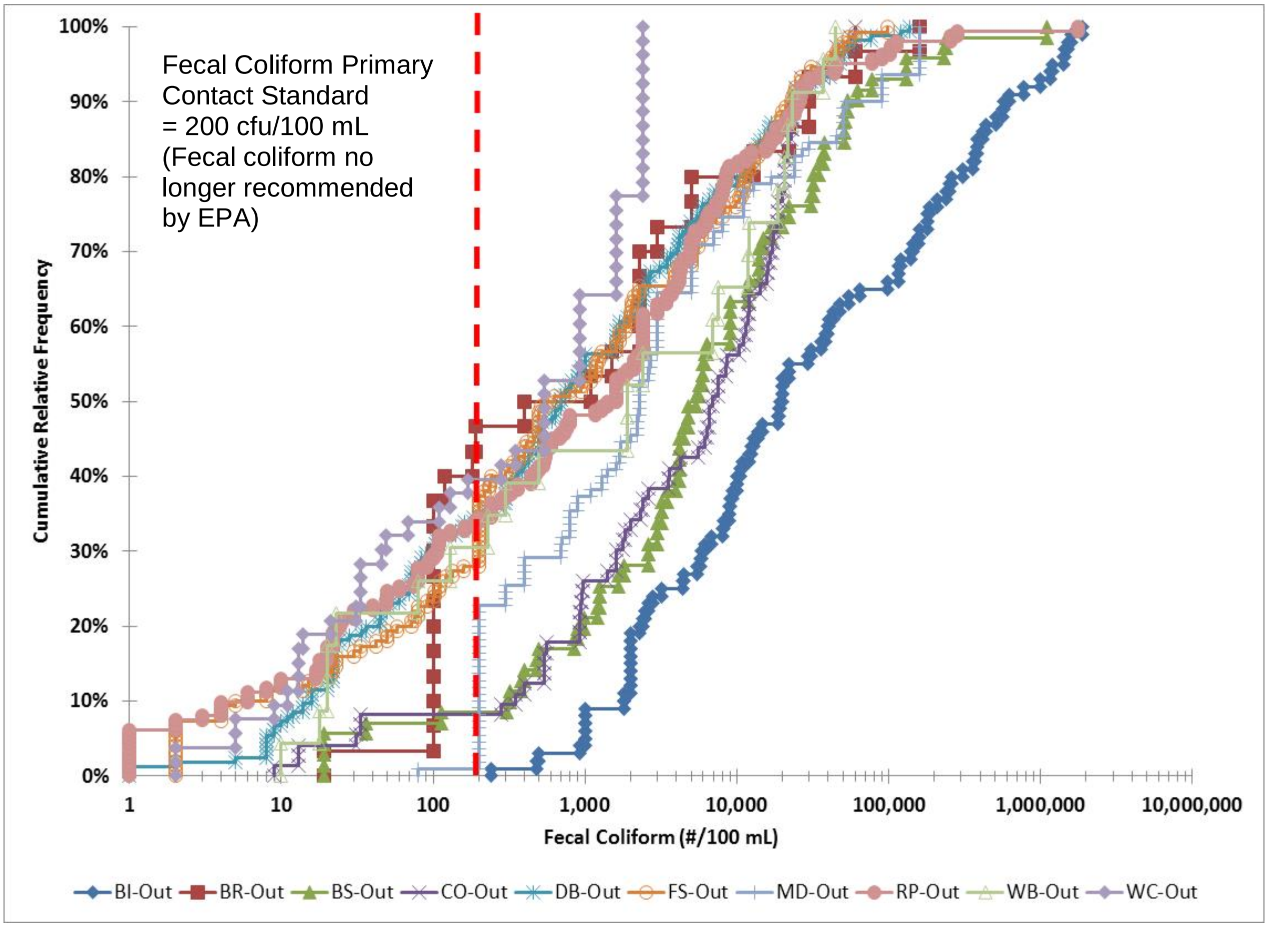
Note: See Table 8-1 for abbreviations and Tables 8-4 for corresponding data. Red dashed line corresponds to geometric mean primary contact recreational water quality criteria recommended by EPA for *E. coli* of 126 cfu/100 mL.

Table 8-5. Mann-Whitney Hypothesis Testing Results to Assess Significant Differences between Inflow and Outflow FIB Concentrations for Various BMP Categories

BMP Type[1]	Fecal Coliform	Enterococcus	*E. coli*
Bioretention	**0.026**	0.117	0.228
Composite, Treatment Train	**0.061**	NA	NA
Detention Basin (grass, dry)	**< 0.0001**	0.252	**0.062**
Detention Basin (other, concrete)	0.622	NA	NA
Biofilter, Grass Strip	**0.003 (exports FIB)**	NA	NA
Biofilter, Grass Swale	0.646	NA	0.480
Infiltration Basin	0.618	NA	NA
Manufactured Device	0.133	0.707	NA
Disinfection System	**0.031**	**< 0.0001**	NA
Filter, Other Media	0.352	NA	NA
Filter, Sand	**0.020**	0.617	**0.049**
Retention Pond (Wet)	**< 0.0001**	**0.002**	**< 0.0001**
Wetland Basin	0.122	0.169	0.719
Wetland Channel	0.495	NA	NA

[1]Substantial surface runoff volume losses may occur at infiltration-oriented practices such as bioretention, infiltration basins and grass swales and strips. From a mass loading perspective, these volume losses would be more important than the changes in surface runoff concentration. Volume losses will vary, depending on site-specific conditions, BMP designs and maintenance practices.

Figure 8-5. Cumulative Relative Frequency Distributions of Fecal Coliform Data from the Stormwater BMP Database for Selected BMP Categories



8.4 Discussion of Findings from FIB Analysis in the Stormwater BMP Database

Conclusions that can be drawn regarding stormwater control device performance for FIB based on this analysis are generally consistent with previous analyses completed for the BMP Database (WWE and Geosyntec 2010, 2012). Key findings and observations based on the data set analyzed include:

- Regardless of FIB type, the available data set shows that concentrations in urban runoff typically exceed primary contact recreation standards, often by one or more orders of magnitude.
- Regardless of stormwater control type or FIB type, both inflow and outflow concentrations are highly variable, typically spanning an order of magnitude or more for the interquartile range.
- Currently available data suggest that it is unlikely that conventional structural stormwater controls using passive treatment can consistently reduce FIB concentrations in runoff to primary contact recreation standards. Sand filters are the only stormwater control

category evaluated with effluent concentrations approaching primary contact stream standards for *E. coli*, and retention (wet) ponds approached the primary contact standard for enterococcus. Although the bioretention data set also achieved *E. coli* concentrations below stream standards, this data set had low *E. coli* in the influent relative to other BMP categories; therefore, these findings are inconclusive for bioretention. Active treatment devices using UV disinfection were able to reduce effluent concentrations to stream standards.

- Bioretention, sand filters, retention (wet) ponds, extended detention basins (dry) and composite (treatment train) stormwater controls appear to be able to reduce FIB concentrations to some extent, based on hypothesis testing summarized in Table 8-5. Unit processes such as sorption and filtration are present in bioretention and media filters, whereas wet ponds may provide long holding times that enable sedimentation, solar irradiation and habitat conducive to natural predation. Detention basins rely primarily on sedimentation; however, scouring and resuspension of sediment deposited in detention basins may be a potential on-going source of FIB loading in the effluent. Review of individual detention basin studies shows that some detention basins export FIB, whereas others reduce FIB concentrations.

- Grass strips and swales do not appear to reduce FIB concentrations in their effluent. Instead, increases in effluent concentrations for fecal coliform are shown for grass strips and some grass swales studies. These stormwater control types may be exporting FIB, either from entrainment of previously deposited FIB or from new sources (e.g., animal excrement). (Note: reductions in FIB loading due to infiltration and evapotranspiration are not evaluated in this analysis.)

- Inadequate data sets are available to evaluate the performance of permeable pavements, and green roofs. Previous review of the green roof data in the BMP Database has shown that even though roofs have relatively few sources of FIB (i.e., birds), sample results an order of magnitude above primary contact stream standards are not uncommon (WWE and Geosyntec 2010).

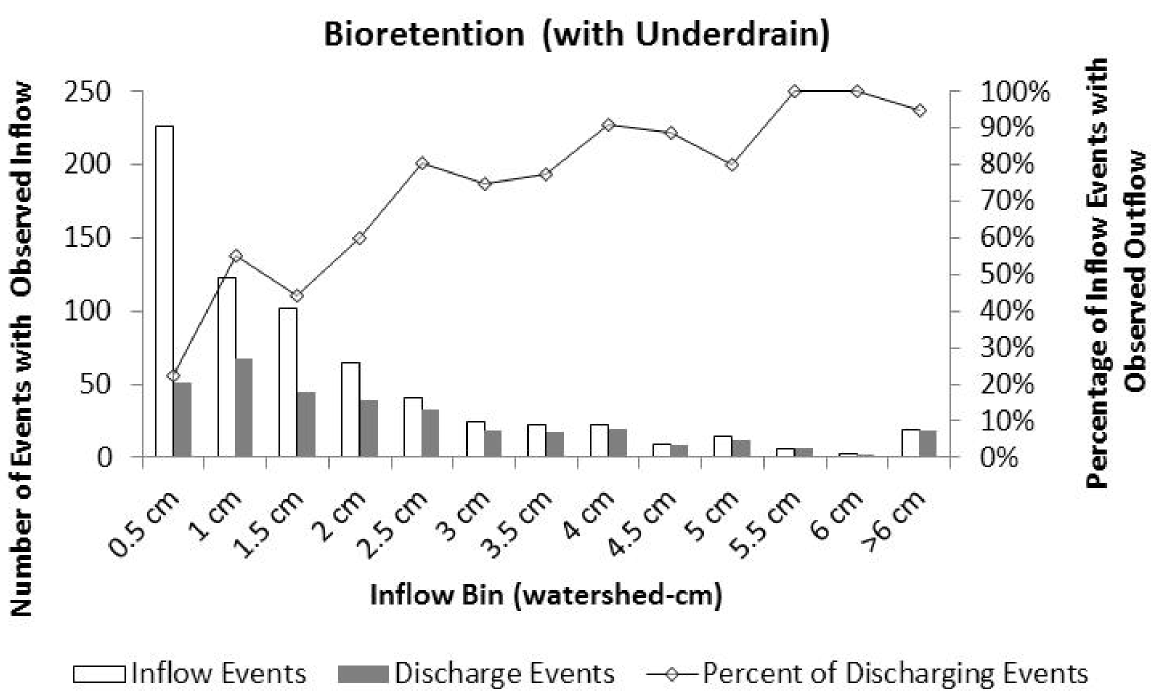
- The manufactured device category includes a range of proprietary devices that rely on various unit treatment processes; therefore, performance should be evaluated on a unit treatment process basis for purposes of stormwater control device selection. Nonetheless, the manufactured device studies currently included in the BMP Database did not result in FIB effluent concentrations attaining stream standards. Significant overlap of interquartile ranges for inflows and outflows is present for the majority of the manufactured devices, with nearly statistically significant increases (export) of FIB for this overall stormwater treatment device category ($p = 0.13$). Due to ongoing innovation regarding unit processes provided in manufactured devices, general conclusions about manufactured devices, or subcategories of manufactured devices, should be used with caution.

- Review of the cumulative frequency distribution for fecal coliform in Figure 8-5 for selected treatment categories with larger data sets indicates that all of the categories analyzed exceed the 200/100 mL fecal coliform threshold for the majority (>50%) of outflows.

- The concentration-based analysis does not account for load reductions that may result from reduced surface volumes discharged from the various stormwater control types. For more information on volume reduction benefits of BMPs, see *International Stormwater Best Management Practices (BMP) Database Technical Summary: Volume Reduction* (Geosyntec and Wright Water Engineers 2011) and *Addendum 1 Expanded Analysis of Volume Reduction in Bioretention BMPs* (Geosyntec and Wright Water Engineers 2012b) for a discussion of volume reduction analyses conducted for the BMP Database. Practices that infiltrate runoff can help to reduce the number of runoff events discharged from a stormwater control device and reduce runoff volumes, which may help to reduce the number of exceedance days associated with wet weather conditions, and will reduce in-stream final concentrations. Figure 8-6 provides an example of analysis of discharge events for bioretention with underdrains, with inflow concentrations grouped into inflow bins of runoff normalized to watershed-centimeters.

Figure 8-6. Binned Presence/Absence of Discharge Plots for Bioretention Sites with Underdrains

(Source: Geosyntec and Wright Water Engineers 2012b)



8.5 Other Passive Structural Stormwater Controls Not Currently Analyzed in BMP Database

Several stormwater control types that communities may consider using to reduce FIB loading are not currently well represented in the BMP Database. These include subsurface flow wetlands with upstream detention, permeable pavement, and emerging manufactured device products. These are discussed briefly below.

8.5.1 Subsurface Flow Wetlands with Detention

Subsurface flow wetlands with detention are engineered, below-ground treatment wetlands that include many of the natural treatment processes of surface flow constructed wetlands as well as the filtration mechanisms of media filters. Water flows through a granular matrix, which typically supports the growth of emergent wetland vegetation on the surface. The matrix provides a significant surface area for the filtration of particulate-bound constituents and the growth of bacterial biofilms that metabolize and degrade pollutants. Due to the low treatment flow rates, an equalization basin is typically needed upgradient of the wetlands to handle peak flows and provide a near constant discharge to the facility.

Currently, no subsurface flow wetland performance studies for FIB are included in the BMP Database; however, published research is available that suggests that subsurface flow wetlands may be effective at reducing FIB (Kadlec and Knight 1996, EPA 1993, Puigagut et al. 2007, Sleytr 2007). Implementation of a subsurface flow wetland is dependent on adequate hydrology, which may not be available in all settings, particularly in semi-arid and arid climates. Additionally, adequate land area for equalization basins is needed. Subsurface flow wetlands are a treatment alternative often considered in Southern California TMDL plans (Geosyntec 2009).

8.5.2 Engineered Media in Advanced Stormwater Controls

An area of current research relates to optimizing filtration media in various stormwater controls such as bioretention (biofilters) and media filters. Effluent concentrations for fecal coliform and *E. coli* in bioretention facilities vary depending on the filter media, vegetation, exposure to sunlight, climate conditions (dry/humid) and hydraulic retention time. Previous studies have shown fecal coliform and *E. coli* removal rates generally greater than 50% (Barrett 2003, Hunt et al. 2008, Rusciano and Obropta 2007, Zhang et al. 2010, Kim et al. 2012, Chandrasena et al.2012). Two major removal mechanisms of bacteria from bioretention facilities include straining and sorption, however, sorption is the most likely removal process for *E. coli* due to its small size (Zhang et al. 2010, Kim et al. 2012). Optimization of media to promote sorption may be an area where future opportunities exist to improve stormwater control performance. A detailed discussed of this emerging research is beyond the scope of this report, but may be useful in urban areas in the future. For examples of pertinent research, see Clark and Pitt (2012, 1999).

8.5.3 Permeable Pavement

Currently, insufficient permeable pavement studies for FIB have been submitted to the BMP Database for analysis. To the extent that permeable pavement sites reduce runoff volumes from a site, they would be expected to help reduce discharged pollutant loads under wet weather conditions and to reduce the frequency of exceedance days, similar to bioretention.

8.5.4 New Proprietary Manufactured Devices

The proprietary market for stormwater controls is continually evolving. A systematic evaluation of manufactured devices designed to reduce FIB and pathogens was not completed for purposes of this report. When proprietary devices are considered as a treatment alternative, care should be taken to ensure proper maintenance, since proprietary devices are often underground (out-of-sight). When proper maintenance is not conducted, sediment and organic materials captured in the device can become a source of FIB. Similarly, if devices allow resuspension and scouring of sediment, then export of FIB may be an issue. Additionally, when reviewing performance data and literature associated with manufactured devices, it is preferable to review independently measured quantitative results for each monitored event (including effluent concentrations), rather than simplified percent removal tabulations. Independently conducted or verified field-based studies that present influent and effluent concentrations for the monitored storm events, precipitation and flow data associated with monitored events, and information on the sampling plan should be provided for careful review and applicability to site-specific applications. "Real" stormwater (including natural organic matter and suspended sediment) should be used in such evaluations rather than synthetic stormwater; otherwise, performance results may not be representative of installed conditions.

Examples of detailed proprietary device evaluations based on field installations can be obtained from the International Stormwater BMP Database (www.bmpdatabase.org), the New Jersey Cooperative for Advance Technology (NJCAT) program (http://www.njcat.org/), the Technology Acceptance Reciprocity Program (TARP), and other sources such as in-depth academic dissertations and publications (e.g., Cai et al. 2014), as few examples.

8.6 Active Treatment (Disinfection) of MS4 Flows

Sanitary wastewater disinfection technologies are well documented to achieve acceptably low FIB concentrations for municipal wastewater treatment plants (WWTPs) and CSOs and SSOs (Field et al. 2004) through the use of active treatment such as UV light irradiation, chlorination, chlorine dioxide, and ozonation disinfection. Effectiveness in CSO/SSO contexts is enhanced by mixing (Field et al. 2004). Many questions remain regarding active treatment of FIB in separate stormwater systems (i.e., MS4s), although examples of disinfection of urban runoff date back to the mid-1960s with a hypochlorination project in New Orleans, LA (USEPA 1973a). UV and ozone disinfection of separate stormwater system flows have been used in communities where frequent beach closures due to elevated FIB occur. Although these approaches are costly, they control FIB at the point of discharge. These techniques are typically used for low flows where there is a defined point source discharge that can be treated close to a swimming area. Communities considering use of disinfection should be aware that even though disinfection can effectively treat flows for pathogens, the downstream receiving water may not necessarily attain recreational water quality criteria since new sources of FIB (e.g., wildlife, birds) may be introduced following treatment (Murray and Steets 2009). The types of disinfection that could be considered for stormwater treatment (and that have also been successful for CSO/SSOs [Field et al 2004]), include:

- **UV light irradiation**: UV bulbs used for wastewater disinfection emit energy at a wavelength of about 254 nm, which penetrates the cell wall of a microorganism and is absorbed by cellular materials such as nucleic acids. This absorption will either keep the cell from reproducing or destroy the cell entirely. UV disinfection requires a relatively high level of pretreatment to reduce suspended solids (Field 1996), typically using sand filtration or another method, a pump, and backwashing for filter maintenance. UV disinfection effectiveness requires careful design with regard to flow rate, which is a principal determinant of the dosage of UV light necessary for effective disinfection (Wojtenko et al. 2001). UV disinfection is a safer option than chemical disinfectants and has no known downstream ecological affects. The system may be placed in a pump house and does not require additional land. The capital cost is low compared to other active treatment alternatives. Operation and maintenance (O&M) includes regular inspection, cleaning, bulb replacement, and an energy supply (Geosyntec 2009). Because fouling materials deposited on quartz sleeves of UV bulbs decrease transmittance of UV light and associated disinfection capability (Oliver and Gosgrove 1975), an in-place cleaning system should be considered to remove fouling materials from the quartz sleeves.

- **Ozonation:** Ozone disinfection facilities include an on-site ozone production chamber, a contactor tank, and an ozone destruction device. Due to ozone's molecular instability and dangers associated with having the gas stored on location, an on-site ozone production facility is necessary to produce the chemical throughout treatment. Ozone does not produce disinfection residuals and dissipates when exposed to air. Some pretreatment is also typically required to reduce suspended solids to minimize disinfection interferences. Depending on the influent's chemical composition, ozone treatment could produce brominated disinfection byproducts. The capital cost is greater than UV, and O&M includes inspection, cleaning, and an energy supply (Geosyntec 2009).

- **Peracetic Acid:** Peracetic acid disinfects through oxidation. The chemical mixture is a combination of glacial acetic acid, hydrogen peroxide, and water (EPA 1999). It deactivates bacteria and virus cells by instigating electron transfer when oxidizing a microorganism's cell wall. It has primarily been used in the food and beverage industry. This disinfection alternative is less safe than UV or ozone because of the compound's explosive nature and lacks implementation examples in the stormwater field. The footprint, capital cost, and O&M would be similar to that of a chlorine facility due to its comparable configuration (Geosyntec 2009).

- **Chlorine:** Chlorine is the most widely used chemical disinfectant for wastewater in the United States and is highly effective as a disinfectant. However, it poses on-site chemical storage risks and results in residuals that can threaten aquatic life downstream. Compared to the other alternatives, the land requirements, capital cost, and O&M are low. Capital costs include the treatment tank and initial chlorine supply. O&M consists of regular cleaning of the system and chlorine re-supply. However, due to the risks associated with chlorination, it is typically not a preferred alternative (Geosyntec 2009). (Chlorination/dechlorination is still predominantly used for CSO/SSO disinfection, but is generally viewed as a less desirable alternative for MS4 flows.)

When considering use of active treatment, it is generally recommended that source controls should be implemented as a primary stormwater treatment strategy first, followed by carefully selected passive-treatment structural stormwater controls (BMPs). In cases where these practices are not effective and uses such as recreational beaches are present, disinfection may be a viable alternative, particularly if human sources of FIB have been confirmed and not controlled by other measures. Disinfection can be implemented in the form of low-flow diversions to a treatment system or through diversions to a municipal WWTP.

Effectiveness of disinfection is influenced by the water quality (e.g., turbidity and organic matter content), type of disinfectant used, disinfectant dosage and disinfectant contact time. Challenges arise in disinfecting urban runoff due to the extreme variability between dry and wet weather flow volumes (Stinson and Perdek 2003). Disinfection of urban runoff requires some form of filtration, sedimentation prior to introduction of disinfecting chemicals or irradiation (EPA 1973a&b). High levels of particulate matter in urban runoff (particularly during wet weather conditions) can provide a “shielding effect” in which particles protect the microbes from the disinfecting agent (Sakamoto and Cairns 1997). To enhance treatment of wet weather flows, it is essential that mechanical or chemical pretreatment processes are applied prior to disinfection and are subjected further to high-rate filtration processes prior to discharge to waterbodies. Thus, the costs of active treatment for disinfecting stormwater include additional costs for pretreatment and flow equalization prior to the disinfection process itself.

Regarding contact times, wet weather flow disinfection can be achieved at shorter contact times relative to conventional contact times, termed “high-rate disinfection” (EPA 1979a&b, Stinson et al. 1998). Use of conventional contact time of about 30 min for disinfection of wet weather flows is extremely costly because of their relatively high flow rates and intermittently occurring volumes. High-rate disinfection is accomplished by: (1) increased mixing intensity, (2) use of higher concentrations of disinfectant, (3) use of chemicals or irradiation with higher oxidizing rates or microorganism-kill potential, or (4) combinations of these (Field 1990). The use of increased mixing with any disinfection technology provides better dispersion of the disinfectant and forces disinfectant contact with a greater number of microorganisms per unit time.

Use of disinfection for low-flow diversions under dry weather conditions is discussed briefly below. Additionally, blending concepts in CSO situations are also briefly described, although CSO treatment and management is generally beyond the scope of this report. Most of the wet weather flow disinfection studies to date have been conducted in the context of CSOs (EPA 2002).

8.6.1 Low Flow (Dry Weather) Diversions for MS4s

In recreational beach settings in California, low-flow diversion and treatment has been implemented in many locations. For example, Los Angeles County Flood Control District operates over 20 low flow diversions at Santa Monica Bay, Marina del Rey, and Long Beach, with approximately 15 additional facilities operated by other agencies. The approximate cost for the 23 facilities in the Los Angeles County Flood Control District facilities is approximately $18.4 million for engineering and construction, with ongoing annual operation and maintenance costs of approximately $1.4 million (as of 2009-2010) (Orange County Public Works 2014). Cases where these costs of active disinfection are warranted typically include situations where source controls and passive treatment are not sufficient or effective, where receiving water numeric limits are applicable at the point of discharge, and where costs of beach closures are substantial.

Geosyntec (2013) summarized costs of several individual low-flow diversions with treatment facilities, with costs differing between projects depending on design flow, the amount of bacterial loads, and ultimately the size of the system itself. Many of these UV treatment projects were designed to treat about 150 gpm (Moonlight Beach and Westside SURF projects) to 170 gpm (Aliso Beach) and ranged from $1.3 million (Westside SURF project) to $2 million (Poche Beach CBI project) in total project costs. Annual operation and maintenance costs ranged from $18,000 (Westside SURF project) to $250,000 (Poche Beach CBI project). UV treatment consistently performed well and significantly reduced the bacteria concentrations in the treated effluent. However, based on a review of active disinfection of low flows in California, Geosyntec (2009) concluded that a consistent lesson learned in almost all of the treatment projects reviewed is that although treatment is successful at the facility itself, bacteria loads increased downstream of the facility due to regrowth, regeneration, animal inputs into the open channel, and bird droppings along the wrackline. These studies stated that the opportunity for bacteria regrowth and regeneration can be reduced by installing the treatment system further downstream and closer to the discharge site (Geosyntec 2009). Discussions of operations and experiences at a few specific facilities follow.

The Moonlight Beach study in Encinitas, California looked at the effectiveness of dry weather diversion and disinfection on receiving water benefits. Moonlight Beach experienced frequent beach postings and closures due to FIB at the compliance point at the mouth of Cottonwood Creek. A UV treatment process was implemented under the Clean Beaches Initiatives grant program. The treatment facility was constructed in 2002 and consisted of a wet well and pumping station, a series of filters, including a two dual media (sand and anthracite) pressure filters, and a disinfection unit. The disinfection unit consisted of two UV disinfection chambers approximately 48 inches in length and 8 inches in diameter. Each chamber has four low-pressure, high intensity UV lamps. The system is operated from a programmable logic controller (PLC). System controls were set to shut the entire system down on three operating conditions: high level in the wet well, high pump discharge pressure, and high effluent turbidity. Treated flow is returned to Cottonwood Creek, the receiving water discharge point upstream of the beach. The entire treatment facility is housed in a 24 feet long, 10 x 10 foot prefabricated steel enclosure. The system was designed to operate primarily during dry weather conditions via stream diversion.

The City of Encinitas reported the Moonlight Beach project to be a success in terms of treatment efficiency (i.e., >99% reduction) at the treated discharge and reduction of beach closures (Rasmus and Weldon 2003); however, it also noted that increases in FIB occurred downstream of the treated effluent. Subsequently, special studies to evaluate the increases in FIB following treatment found:

- an increase (~180%) in fecal coliform concentrations immediately after treatment in open “natural” channel, likely due to animals;
- nearly a 200% increase in all three indicators in the 72-inch pipes downstream of open channel thought to be due to ideal conditions for FIB growth in the pipes (dark, wet, with organic matter); and
- continued 57% increase from treatment effluent in enterococcus concentrations only (on

beach) thought to be due to birds and typical wrack line accumulation on the beach.

Selected recommendations, among others, included regular cleaning of system piping to reduce media for FIB growth and locating the system as close to the receiving water as possible to limit opportunity for regrowth after treatment.

Similar efforts have been implemented in other Southern California communities such as Aliso Creek in Orange County, CA where a runoff treatment plant was installed using granular activated carbon and UV disinfection. The Aliso Creek project showed rapid regrowth in natural channels after treatment of flows, even though the UV filtration/disinfection produced >99% removal in urban runoff entering Aliso Creek (Anderson 2005, Orange County Stormwater Program 2014). Fecal coliform increased from 317 cfu/100mL at the stormwater control outlet to 2,575 cfu/100 mL in the natural receiving water channel 35 feet downstream of the stormwater control (Flow Science 2010).

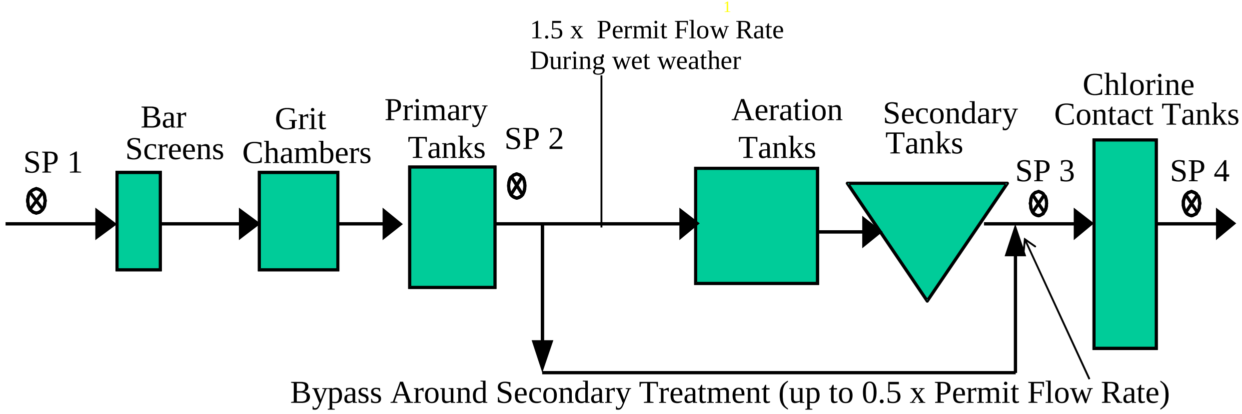
The Poche Beach project in the San Clemente and Dana Point coastal area in Orange County, California, provides another case study of low-flow diversion and treatment. This facility treats urban runoff from Prima Deshecha Channel prior to its discharge to the surf zone at Poche Beach. The facility implementation cost was approximately $3 million, with ongoing operation and maintenance costs. Urban runoff from the channel is diverted into the treatment facility by an inflatable diversion gate within the channel and can treat flows up to 694 gpm (1 MGD) using filter panel screening, sand media filtration, and UV light disinfection. The project began official operation in July 2010. Over the first two years of operation, the facility treated average flows of 403 to 521 gpm (0.58 to 0.75 MGD), and produced average FIB removal efficiencies of 98 to 99%. Despite facility performance, a corresponding improvement in surf zone water quality has not occurred due to rebound of FIB concentrations in the channel prior to reaching the surf zone. The County is presently seeking resource agency approval for permanent relocation of treated outflow to a more effective discharge site. Additionally, it is increasingly apparent that poor surf zone quality may also be attributable to inordinately large numbers of shorebirds congregating at the intertidal area of the beach. The County and City of San Clemente are further investigating the contribution of shorebirds to beach FIB levels in order to develop appropriate pollution prevention strategies (Orange County Public Works 2014).

Santa Monica's Urban Runoff Recycling Facility (SMURRF) is a joint effort of the City of Los Angeles and the City of Santa Monica. SMURRF treats dry weather urban runoff by conventional and advanced treatment systems to remove sediment, oil and grease, and pathogens. Treated water is then used for beneficial purposes such as irrigation and in dual plumbing systems for commercial buildings. The system includes coarse and fine screening to remove trash, plant material and debris, degritting systems to remove sand and grit, dissolved air flotation to remove oil and grease, microfiltration to remove turbidity and UV radiation to kill pathogens. The facility is designed to treat 0.5 MGD (347 gpm). The SMURRF cost approximately $12 million, including the distribution system for the recycled water (http://www.smgov.net/Departments/PublicWorks/ContentCivEng.aspx?id=7796, Santa Monica Public Works website).

8.6.2 Peak Flow Blending

As previously mentioned, CSOs are not the focus of this report; however, strategies that can be effective at the treatment point within combined sewer systems are briefly introduced below. While applying to systems that typically bypass wet weather flows at the treatment plant (as opposed to using regulators that discharge at points upstream of the treatment plant; see examples below), blending is a practice that diverts a portion of the peak wet-weather flow at WWTPs, after primary treatment. The process typically diverts primary treatment effluent around biological treatment and combines it with both primary and secondary treatment prior to disinfection and subsequent discharge from a permitted outfall. For combined sewer systems, EPA's 1994 CSO Policy encourages delivery of maximum flows to WWTPs, while ensuring that bypasses do not result in NPDES permit exceedences. In addition, in December 2005, the EPA proposed, for public comment, a new policy for addressing peak flow events at municipal WWTPs served by separate sewer systems, also through flow maximization. This policy is still open with EPA. The blending practice at treatment plants prevents inactivation or washout of the vulnerable secondary biological process during peak flows. Blending is resorted to as an economically feasible alternative for flow maximization and CSO control. However, the blending practice raises concerns, shared by the EPA and general public, that the disinfection process and pollutant removal may be compromised, and public health and water quality are not well protected. Therefore, the impact of blending on disinfection and on the quality of the receiving water has been the subject of recent studies by EPA and WERF.

Figure 8-7. Schematic of a Wastewater Treatment Process with Bypass/Blending in New York City, NY
(Source: EPA 2010c, EPA/600/R-10/003)



Notes:

1) Permit FlowRate = 12 Month Rolling Average Flow
2) SP1, SP2, SP3, SP4 –Sampling Location

Several case studies are available regarding bypass and blending. For example, in the EPA study titled *Impact of Wet-Weather Peak Flow Blending on Disinfection and Treatment: A Case Study at Three Wastewater Treatment Plants* (EPA 2010c, Stinson et al. 2009), the authors evaluated the effects of wet-weather blending on the concentrations of fecal coliform and enterococcus, protozoa, and viruses in the WWTP final effluent. Three NYC WWTPs were evaluated. The results showed that during blending, the evaluated WWTPs removed, on average, between 97% and 99% of coliphage and enteric viruses, approximately 71% of *Cryptosporidium*, and between 40% and 88% of *Giardia*. Enterovirus, reovirus and adenovirus were the "top three" viruses detected in WWTP samples; on a few occasions, rotavirus was also detected. However, in the final effluent samples, only enterovirus, reovirus, and adenovirus were detected. The geometric mean for fecal coliform concentrations in disinfected effluent during blending at the three WWTPs ranged from 520 to 19,000 MPN/100 mL and the corresponding geometric mean for enterococcus effluent concentrations ranged from 870 to 17,000 MPN/100 mL. All blended FIB concentrations in disinfected blended effluent were much greater than the acceptable values, which are 200 MPN/100 mL for fecal coliform and 33 MPN/100 mL for enterococcus. During blending, effluent BOD and total suspended solids concentrations remained below 30 mg/L (a monthly average permit limit for both parameters) at two out of three WWTPs; the third WWTP, that had results above 30 mg/L for both parameters, was undergoing a partial construction at the time of sampling.

The study showed the need for development and improvement of pathogen detection techniques in the WWTPs effluents. Use of maceration, a sample preparation technique, showed statistically significant higher FIB levels in the final (chlorinated) effluent. The study recommends achieving better understanding of the transport and fate of pathogens and related indicators discharged from WWTPs during blending by: (1) evaluating sampling protocols and test methods, (including maceration, sonication, and tissue homogenization) for determining more accurate concentrations of specific microorganisms in WWTP effluents during normal dry weather conditions and blending, (2) determining the major factors impacting fate and transport of pathogens in blended effluents, including die-off and after-growth potential of specific microorganisms in waters receiving effluents from WWTPs during blending, and (3) assessing the effects of the discharge of effluents from WWTPs during blending.

The major strength of this study is that it gathered information at three full-scale WWTPs functioning as usual during actual dry-weather non-blending and wet-weather blending operation. Also, this study represents a first detailed effort to analyze the impact of blending during wet weather. The limitation of the study is that it represents only one geographical location for the three plants studied and the wet-weather blending ratios or flow rates were measured in only one of the three plants. Thus, the geographical proximity and the limited number of facilities evaluated during the study suggest that these results should be viewed as plant-specific.

In another case study titled *Characterizing the Quality of Effluent and Other Contributory Sources During Peak Wet Weather Events* (WERF 2009), the authors evaluated the impacts of blending practices at WWTPs on effluent and receiving water quality, and estimated the public health risk associated with recreation in surface waters receiving blended flows. Field samples were collected at four WWTPs for in-plant processes and receiving waters during wet-weather blending, wet weather non-blending, and dry-weather events. Laboratory analyses for *Giardia*, *Cryptosporidium*, viruses (adenovirous, enteric viruses, rotavirus, norovirus), pathogen indicator organisms, (fecal coliform, *E. coli*, enterococcus, and male specific coliphage), and other water quality parameters were analyzed. Data from the East Bay Municipal Utility District's (EBMUD) Main Wastewater Treatment Plant (MWWTP) were used to develop hydrodynamic and water-quality computer models to predict receiving-water conditions and QMRA to evaluate increased risks of gastrointestinal and respiratory infections for people recreating in waters receiving blended flows. Results indicated that only *Giardia* and adenovirus concentrations in plant final effluent increased during wet weather blending events out of all the organisms tested in this study, and so receiving water modeling was conducted for these organisms. It should be noted that EBMUD practices double disinfection, while the NYC plants practice a single disinfection. The change to double disinfection at EBMUD results in primary and secondary effluent flows disinfected separately prior to blending, whereas all three NYC plants disinfect downstream of the blending point. The WERF study identified some alternatives to reduce or eliminate blending, including rainfall derived infiltration and inflow reduction, peak stormflow storage, and treatment capacity expansion.

8.7 Stormwater Control Case Studies

Many published papers containing stormwater control performance monitoring results are available. Several examples are provided below addressing a range of bacteria removal in different technologies. The first is a bioretention case study. Bioretention is a key component of many Green Infrastructure/Low Impact Development (GI/LID) projects. For this reason, a case study providing findings from bioretention studies conducted by North Carolina State University researchers in North Carolina is provided. The second case study is provided for a construction site sedimentation basin in South Carolina, with interesting findings regarding prevalence of *E. coli* in an essentially undeveloped area (Sawyer et al. 2010).

8.7.1 Bioretention Research in North Carolina

Bioretention is increasingly being used as part of watershed management strategies in urbanizing watersheds in North Carolina. Although bioretention has been shown to remove metals, nutrients, sediment, and other pollutants from stormwater runoff, relatively little has been published regarding its ability to sequester FIB. However, bioretention areas have multiple treatment mechanisms for FIB control. Filtration, sorption, exposure to sunlight (UV radiation), desiccation, and predation all occur in bioretention areas. The Department of Biological and Agricultural Engineering at North Carolina State University studied five bioretention areas described below.

The experimental sites were located in Charlotte (Hathaway et al. 2009), Graham (Passeport et al. 2009), and Wilmington, North Carolina (Hathaway et al. 2011). Two bioretention cells were present at both the Graham and Wilmington sites. In Graham, the two bioretention areas varied by underlying soil type, media depth, and drainage configuration (Table 8-6). The south cell (Graham-S) was underlain by loamy clay, while sandy loam soils were noted below the north cell (Graham-N). In Wilmington, the two cells differed by media depth. One cell had an average media depth of approximately 25 cm (Wilmington-S), while the other had a depth of 60 cm (Wilmington-D). General site characteristics are provided in Table 8-6. Monitoring at each site was conducted by grab sampling from inlet and outlet locations using sterilized bottles. Samples were collected only when the systems were operating under storm flow. The FIB selected for analysis varied based on location, with fecal coliform, *E. coli*, and enterococci analyzed among the three locations.

Table 8-6. General Characteristics of North Carolina Bioretention Areas

Characteristic	Charlotte	Graham-N	Graham-S	Wilmington-D	Wilmington-S
Approximate Year Constructed	2003	2005	2005	2006	2006
Drainage Area (ha)	0.37	0.35	0.35	0.10	0.05
Watershed Composition	Municipal (parking lot)	High School (parking lot)	High School (parking lot)	Commercial (parking lot)	Commercial (parking lot)
Estimated Imperviousness	100%	40%	40%	100%	98%
Surface Area (ha)	0.023	0.010	0.010	0.006	0.006
Storage Depth (cm)	18	23	23	28	28
Media Composition	80% sand, 20% fines and organics	80% expanded slate fines, 15% sand, 5% organics	80% expanded slate fines, 15% sand, 5% organics	87% sand, 4% silt, 4% clay	88% sand, 5% silt, 5% clay
Estimated Average Media Depth (cm)	120	60	90	30	60
Drainage Configuration	Standard with Underdrains	45 cm internal storage zone	75 cm internal storage zone	Standard with Underdrains	Standard with Underdrains


(a)


(b)


(c)

Bioretention areas in (a) Charlotte, (b) Graham, (c) Wilmington, NC. (Photos courtesy of Jon Hathaway and Bill Hunt)

The inlet and outlet FIB concentrations for each site are presented in Table 8-7. Substantial reductions in FIB concentrations were observed for all bioretention cells other than Wilmington-S. Although data were not tested for significant differences for the Graham site, significant differences between inlet and outlet concentrations were noted for both fecal coliform and *E. coli* for the Charlotte site, and for enterococci for the Wilmington-D site ($p < 0.05$).

Although these data suggest bioretention can sequester FIB, the magnitude of effluent FIB concentrations is not always desirable. Average effluent FIB concentrations were not lower than EPA primary surface water standards at one of three sites for fecal coliform, two of three sites for *E. coli*, and zero of two sites tested for enterococci. The probability of not exceeding EPA primary surface water standards was estimated for the Wilmington sites. The non-exceedance probabilities for *E. coli* and enterococci standards were 63 and 47% for Wilmington-D, and 43 and 9% for Wilmington-S, respectively. It should also be noted that performance appears to vary depending on FIB. Non-exceedance probabilities for enterococci were lower for both Wilmington sites. Enterococci are generally regarded as more persistent in the environment, potentially leading to the observed results; however, enterococci have not been monitored for many stormwater control devices. Thus, these results should be verified through further research.

Bioretention are also capable of infiltrating relatively large volumes of stormwater runoff, effectively reducing pollutant loads to surface waters even when effluent concentrations are less than desirable. The influence of such infiltration on groundwater microbial quality is yet to be determined, and remains somewhat of a concern. Nonetheless, such mass reduction should be considered in watershed restoration models.

Table 8-7. FIB (cfu/100 mL) at Inlet and Outlet of NC Bioretention Areas
(Source: Hathaway and Hunt 2010)

Location	Fecal coliform		*E. coli*		Enterococci	
	inlet	outlet	inlet	outlet	inlet	outlet
Charlotte[1]	2420	258	241	20	-	-
Graham-N[2]	4172	125	-	-	-	-
Graham-S[2]	4172	646	-	-	-	-
Wilmington-S[1]	-	-	130	284	375	378
Wilmington-D[1]	-	-	130	39	375	39

[1] Geometric mean concentration
[2] Arithmetic mean concentration

A final observation regarding the data in Table 8-7 is the contrast in performance between Wilmington-D and Wilmington-S. These two bioretention areas were adjacent to one another with somewhat varied watershed areas as noted in Table 8-6. However, the most obvious difference in site design specifications was varied media depth. Other potential explanations for differences in performance between the two cells including temperature, soil moisture, soil chemistry, and soil FIB concentrations (Hathaway et al. 2009). Only modest differences in temperature and soil moisture were identified between the cells, with other investigated properties being similar. Thus, the hydrologic properties of the system provide the most obvious explanation for the observed results. Deeper soil media results in decreased soil-water velocity. Excess soil-water velocity has been shown to result in reduced soil column sequestration of FIB in laboratory analyses. This is due to reduced contact time and increased shear stresses on FIB. Thus, shallow bioretention appears to exhibit poor FIB retention and, in the case of Wilmington-S, export of FIB. FIB have been shown to persist in stream, wetland, and estuarine sediments, suggesting the same potential exists for bioretention cells. Such persistence may leave FIB available for export due to being stripped from the soil matrix. In short, bioretention design specifications may influence FIB sequestration; however, there are many facets of bioretention microbe removal mechanisms that are poorly understood. Research addressing such knowledge would allow more effective bioretention designs that promote FIB sequestration.

In conclusion, five experimental locations in North Carolina have furthered the understanding of FIB sequestration in bioretention areas. Bioretention showed the ability to effectively reduce FIB concentrations for four of five bioretention areas. However, average effluent FIB concentrations were sometimes higher than EPA standards for primary contact waters. Thus, the overall influence of bioretention on the watershed scale must incorporate estimated effluent concentrations from these systems coupled with volume reductions. Further, these studies suggest that design parameters may be adjusted to influence bioretention performance for FIB. It appears as though bioretention media depth has a minimum functional depth, below which FIB performance suffers.

8.7.2 Construction Site Sediment Basin, South Carolina

Although the focus of this report is primarily post-construction conditions, stormwater managers should also be aware that construction site runoff can also be a source of FIB loading. Sawyer et al. (2010) evaluated *E. coli* concentrations in construction-derived runoff in the Piedmont of South Carolina to assess whether the sediment basins at construction sites acted as sources, sinks or reservoirs for potential pathogens and to examine relationships between these observed FIB concentrations and corresponding environmental variables. A summary of results from data analysis and conclusions from the study follow.

Photos of Construction Sediment Basin in South Carolina Study, after and prior to vegetation, respectively. Photos courtesy of John Hayes and Calvin Sawyer.

Water quality data were collected from seven construction site sediment basins associated with permitted land disturbance activities in Anderson, South Carolina, as summarized in Table 8-8. Located in the Piedmont physiographic province, site soils are characterized primarily by the Cecil series, which is a moderately to well-drained clay loam having dominantly clay subsoil and a clay content ranging between 5-35% (USDA-SCS 1993). Mean annual precipitation is 127 cm and mean annual temperature is 16.3°C. More detailed site descriptions and designed surface area and storage for all basins for the 2-year storm elevation are provided in Sawyer (2009).

Table 8-8. Mean Water Quality Results for Samples Collected at Sediment Basins
(Source: Sawyer 2009)

			Measured Variables									
Site	Sample Location	*n*	*E. coli* Density (MPN/100 ml)		TSS (mg/l)	Rainfall (cm)	DSLR[a] (days)	Temp (°C)	pH	DO (mg/l)	Cond (µS/cm)
AHC	WC	14	282	+/- 587	106.3	1.40	2.11	26.0	6.3	6.1	177.0
	Inlet	5	515	+/- 727	332.3	2.01	0.10				
	Outlet	7	913	+/- 1,146	151.2	2.67	0.29				
	Sediment	14	37,103	+/- 60,539		1.40	2.11				
C81-A	WC	21	627	+/- 783	92.7	0.96	2.48	28.7	5.8	5.0	59.4
	Inlet					0.00					
	Outlet	5	1,642	+/- 846	147.7	3.07	0.30				
	Sediment	21	117,488	+/- 188,477		0.97	2.48				
C81-B	WC	5	1,297	+/- 696	75.2	1.85	0.60	26.4	5.3	5.3	47.8
	Inlet	2	114	+/- 91	17.7	3.19	0.00				
	Outlet	4	1,551	+/- 662	33.7	2.32	0.00				
	Sediment	5	180,124	+/- 194,105		1.85	0.60				
CH1	WC	6	1,274	+/- 1,257	106.4	2.49	0.50	27.7	5.7	5.6	94.9
	Inlet	2	289	+/- 5	81.1	2.76	0.00				
	Outlet	2	1,594	+/- 1,166	153.3	2.76	0.00				
	Sediment	6	488,923	+/- 423,009		2.49	0.50				
CH2	WC	6	1,405	+/- 1,148	82.7	2.49	0.50	27.8	5.3	5.2	59.2
	Inlet	2	1,123	+/- 1,221	193.0	2.76	0.00				
	Outlet	2	1,534	+/- 1,252	82.1	2.76	0.00				
	Sediment	6	390,654	+/- 398,179		2.49	0.50				
RP	WC	12	1,084	+/- 1,106	69.0	1.63	1.63	27.8	5.7	4.4	59.6
	Inlet	3	1,721	+/- 1,209	26.8	2.56	0.00				
	Outlet	5	1,728	+/- 717	125.6	2.17	0.10				
	Sediment	12	295,038	+/- 427,261		1.63	1.63				
SC	WC	9	1,254	+/- 1,033	128.6	2.08	1.72	26.4	5.5	4.8	61.7
	Inlet					0.00					
	Outlet	4	989	+/- 972	115.6	2.49	0.13				
	Sediment	9	119,914	+/- 128,968		2.08	1.72				

E. coli density values shown as MPN/100 ml +/- standard deviation
[a] Days since last rainfall; *n*, number of samples; WC = water column

Key findings from data analysis included:

- Analysis of grab and composite samples collected throughout the period of research indicate conditions associated with monitored sediment basins may be a source of elevated *E. coli* for relevant receiving waterbodies. Site-specific *E. coli* concentration means across dates for water samples (inlet, outlet, water column) exhibited substantial variability. Inlet means ranged from a low of 114 MPN/100 mL at C81-B inlet to a high of 1728 MPN/100 mL at the RP outlet. Sediment-associated *E. coli* means were several orders of magnitude higher than those of their corresponding water columns, and ranged from a low of 37,103 MPN/100 mL at AHC to a high of 488,923 MPN/100 mL at CH1. Only one sampling location (C81-B inlet) of all sites evaluated fell below the single sample maximum criterion of 235 MPN/100 mL.

- To determine FIB contribution from the construction sites, inlet *E. coli* concentrations were isolated and analyzed. The intent of this aspect of the research was to quantify and test whether there was a significant difference between recommended EPA criteria and *E. coli* concentrations measured from construction site runoff. The t-test results indicate the true mean for inlet samples across sites and dates was significantly greater than 235 (mean = 771 MPN/100 mL; t-stat = 2.16; $p = 0.025$; $n = 14$).

- To quantify *E. coli* concentrations in sediment basin discharge, outlet data were similarly tested against EPA criteria. T-test results showed mean *E. coli* concentrations for outlet samples collected across dates and sites was significantly greater than 235 (mean = 1,368 MPN/100 mL; t-stat = 6.70; $p < 0.0001$; $n = 29$).

These data indicate *E. coli* was present at evaluated construction sites and entered sediment basins at concentrations exceeding EPA recommended thresholds. It is also evident *E. coli* concentrations in the sediment basin discharges also significantly exceeded recommended values. To assess the potential difference between *E. coli* concentrations found in construction site runoff and those found in sediment basin discharge, a paired samples t-test was employed. Results confirmed that *E. coli* concentrations in basin discharge were significantly higher than corresponding concentrations in site runoff (t-stat = 3.54; $p = 0.0036$; $n = 14$). Thus, construction site sediment basin systems may be acting as reservoirs for *E. coli* in addition to serving as net sources of FIB loadings to receiving waters.

Analyses were conducted on basin water column (WC) and sediment-associated *E. coli* concentrations, confirming both contained *E. coli* concentrations significantly above the recommended EPA water quality threshold. The overall mean water column concentration was 877 MPN/100 mL, whereas the mean sediment-associated concentration was substantially higher at 188,828 MPN/100 mL. Site averages for sediment-associated *E. coli* varied by an order of magnitude yet all site means were in excess of 37,103 MPN/100 mL (Table 8-7). Although statistical tests prove neither growth nor decay, they imply an abundant reservoir of *E. coli* available for resuspension given the necessary physical conditions.

In summary, based on analysis of data collected at seven construction site basins in South Carolina, construction site sedimentation basins can be a source of *E. coli* loading to receiving waters, with effluent concentrations exceeding recreational water quality criteria. With such a large reservoir of viable *E. coli* associated with bottom sediments (mean = 188,828 MPN/100 mL), remobilization of sediments during relatively turbulent rain events seems likely and may be a contributing factor to elevated *E. coli* in the effluent. Although the sediment basins reduced TSS concentrations discharged from the construction sites, preferential association of *E. coli* with smaller clay particles discharged through the site outlet could be one factor that helps to explain elevated FIB concentrations. Ehrhart et al. (2002) demonstrated that preferential settling within sediment basins of larger eroded particles produced effluent containing a higher proportion of finer suspended sediments downstream, as measured by particle size distribution. Controlled research over eight years conducted in experimental sediment basins found that on average, 24% of sediment lost through discharge represented resuspension of previously deposited bottom sediments (Jarrett 2001, Fennessey and Jarrett 1996).

Although MST was beyond the scope of this project, potential sources of *E. coli* were routinely observed and photographically documented. Various feces deposited directly within basin catchments were visually confirmed as deer, raccoon, bird, cat and dog. Animal tracks both entering and exiting areas subject to storm-related inundation were also detected on a regular basis. Additionally, three of the sites were hydro-seeded either during, or previous to, the onset of sampling. Aside from grass seed and surfactant, these mixtures often contain "proprietary" blends of fertilizer, including manure.

Direct fecal input however, does not sufficiently account for the observed difference in FIB density between bottom sediments and those of the overlying water column or corresponding basin discharge. While additional research should undertake a mass balance accounting of water and sediments, it appears likely that *E. coli* is persisting for months within basin substrates. Results of research generated through this project, while focused on construction-derived soils and associated man-made hydrologic systems, would appear to confirm similar findings by Whitman et al. (2006) that have shown that *E. coli* can be abundant in undisturbed soils, streams and interstitial water.

8.8 Conclusions and Recommendations for Selection of Stormwater Controls

Performance data for structural stormwater controls remain relatively limited; therefore, only general inferences regarding the selection of stormwater controls are appropriate at this time. General recommendations resulting from analyses in the International Stormwater BMP Database (WWE and Geosyntec 2010) and reinforced by more recent analyses contained in this report include:

- Those working to address pathogen impairments on streams should focus first and foremost on source controls. This requires clear identification of the primary sources of FIB relative to site-specific conditions. Focusing on controllable sources of FIB, particularly those of human origin, is believed to be the most important first step in protecting human health (Pitt 2004a, Clary et al. 2009, Steets 2013) although source control alone may not be sufficient to meet ambient water quality standards.

- The majority of conventional stormwater controls in the BMP Database do not appear to be able to reduce FIB concentrations to primary contact stream standards. Because the data are limited, both in the number of data points and the representativeness of the data (i.e., grab samples, bias from quantitation limits, etc.), rigorous statistical conclusions cannot be drawn based on the available data. Significantly more studies and more representative data (i.e., flow-weighted composites and/or multiple grab samples during an event) are needed for all BMP types to increase the confidence of performance estimates with regard to FIB.

- In terms of reducing overall FIB *loads* to receiving waters, site designs and individual BMPs that reduce runoff volumes should reduce FIB loading from urban runoff. (However, this does not necessarily mean that the receiving waters will attain stream standards if runoff is retained onsite, since the cause of the impairment may include other sources such as birds.)

- At the stormwater control category level, retention (wet) ponds, and various types of media filters may help to reduce FIB concentrations, although not necessarily or consistently to instream standards. Individual bioretention studies also appear to reduce FIB concentrations (as discussed in Section 8-7), but more studies are needed for this category of stormwater controls to draw category-level conclusions. Based on the unit treatment processes provided in retention ponds, media filters, and bioretention, FIB reductions are expected, so the data, for the most part, support the theory.

- In general, grass swales/strips and detention basins do not appear to provide meaningful reduction in FIB concentrations and often show increases in FIB concentrations. These stormwater control types may require enhancements to improve specific additional treatment processes such as filtration and sedimentation. However, it should be noted that volume reductions may be significant, so these controls may be effective at reducing FIB loadings to receiving waters (Fox et al. 2011).

- The manufactured devices in the BMP Database include a range of unit treatment processes, requiring case-by-case evaluation of performance. As an overall category, the individual studies currently included in the Database do not demonstrate significant FIB removals, regardless of the unit treatment process.

- Disinfection is a costly active treatment alternative that may be considered for specific situations such as swim beaches where runoff is consolidated into a single discharge pipe near the receiving water. Disinfection has been used most effectively in the context of low-flow or baseflow diversions, rather than for treatment of all runoff events. Although disinfection is a proven method to reduce FIB at the point of treatment (discharge), studies have shown that the benefits to receiving waters may be limited in some cases due to sources of FIB introduced below the treated discharge. For the diffuse storm drain networks present in much of the U.S., disinfection is not a realistic alternative due to high cost and numerous outflow points that are typical for inland flowing waters.

- Various individual stormwater controls may provide reductions in FIB. Representative examples include individual bioretention studies, a wetland basin and a few detention basins. Care should be taken to understand both site-specific and stormwater control design characteristics in these studies before assuming that similar performance will occur at other locations.

8.9 Additional Research Needs Regarding Stormwater Control Device Performance

Research needs related to stormwater control performance identified in analyses of the International Stormwater BMP Database (WWE and Geosyntec 2010) and reinforced in this report include:

- More studies with larger numbers of storm events and additional within-storm sample collection and analyses for EPA's currently recommended FIB in a range of geographical locations would be helpful in drawing more statistically rigorous conclusions for all stormwater control types.

- Studies that document performance of stormwater controls under various hydraulic conditions to assess the effect of resuspension of sediment on FIB concentrations in treated effluent could be beneficial in design enhancements. This could also include further exploration of the relationship between sediment particles and FIB.
- Paired watershed studies of non-structural stormwater control practices such as pet waste controls, urban wildlife management programs, storm sewer cleaning, etc., could help to target source controls that are most effective in urban watersheds. The BMP Database is structured to accept these types of studies, but none have been submitted to date.
- More studies that help to elucidate transport and fate related issues such as the relationship between FIB and sediment sizes, various nutrients, presence of biofilms, and other factors would be helpful to support modeling.

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9 BACTERIA TMDLS AND URBAN CASE STUDIES

In December 2013, the U.S. Government Accounting Office (GAO) issued a report to Congress titled "Clean Water Act: Changes Needed if Key EPA Program is to Help Fulfill the Nation's Water Quality Goals." Despite completion of over 50,000 TMDLs, the GAO found that few impaired waterbodies had fully attained water quality standards although pollutants had been reduced in many waters. Additionally, GAO concluded that TMDLs seldom contained all features key to attaining water quality standards. Citing recommendations by EPA and the National Research Council (NRC), the needed TMDL elements include: identifying pollution-causing stressors and showing how addressing them would help attain standards; specifying how and by whom TMDLs will be implemented; and ensuring periodic revisions as needed (GAO 2013). The GAO concluded that TMDLs can often achieve targets for point source pollution control through permits, but less than 20 percent achieved their targets for nonpoint source pollution. The GAO recommended that EPA issue new regulations for TMDL development, add additional requirements and consider revising its approach to non-point source pollution (GAO 2013). For FIB TMDLs, it is extremely difficult to reliably include the GAO-recommended elements due to the diffuse and mobile nature of FIB sources and due to limitations associated with treatment technologies.

Currently, although the EPA tracks information on development of TMDLs, a centralized repository of information on TMDLs that have successfully attained water quality standards is not readily accessible (GAO 2013). Originally, the intent of this chapter was to provide examples of FIB TMDLs which had been successfully implemented in urban areas, resulting in attainment of instream recreational water quality standards; however, none were readily identified.[14] This is not surprising, given the many challenges associated with conclusively determining the cause of elevated FIB, challenges in controlling sources of impairment, and treatment limitations associated passive structural BMPs.

An additional factor related to TMDL development in some states is lawsuit-driven development of TMDLs under Consent Decrees (GAO 2013). In such cases, information may or may not have been adequate to develop TMDLs, but the TMDL was required to be completed under pressure to meet a legal deadline. These legal constraints may not result in TMDLs that are developed based on an adequate understanding of sources of FIB.

FIB TMDLs are developed at the watershed scale and often have load reduction targets for both waste load allocations (WLAs) and load allocations (LAs) in excess of 90% over a baseline load. MS4 permittees are finding that they are responsible for a significant portion of the WLA component of the TMDL. Often, much of the land area within older MS4 boundaries has little or no stormwater infrastructure that provides water quality treatment. Moreover, MS4s may find that a portion of their WLA includes an urban wildlife source. Although some BMP types in certain settings are capable of reducing FIB loads (as discussed in Chapter 8), many challenges exist when effectively implementing these practices at a watershed scale. A multi-faceted strategy is typically required and varies from watershed to watershed, depending on the sources

[14] A list of TMDLs having attained water quality standards is not currently available in a centralized repository. Urban successes may exist that were not identified during development of this report.

of FIB loading identified in the watershed. Even when such strategies are implemented, it is unclear whether instream recreational water quality criteria are consistently attainable in urban areas, given technical, practical and economic considerations.

This chapter provides a general strategy suggested for development of TMDL plans, based largely on approaches implemented in Southern California, followed by case studies on various topics addressed earlier in this report.

9.1 Prioritizing TMDL Implementation Plan Activities

Regardless of whether FIB stream standards are attained, most would agree that it is important to control human sanitary sources of stream contamination in urban areas, along with managing pet waste and other reasonably controllable sources of pathogens posing human health risk. Beyond such measures, policy level discussions are needed to ensure wise use of public funds and to answer the question of how far a community needs to go to address FIB exceedances. The following strategy is recommended for impaired streams in urban areas (in non-CSO settings):

1. Conduct targeted studies to identify sources of FIB. Proper identification of the sources of FIB is fundamental to selecting solutions that can help reduce FIB loading. TMDLs completed without a reasonable understanding of pollution sources have a low likelihood of success. Documentation and data collection should be conducted in a systematic manner, thinking forward to potential solutions and implementation strategies. Using GIS supplemented by field reconnaissance, problems and potential solutions can be initially prioritized based on knowledge about the relative severity of the source, implementation opportunities and constraints, and funding availability.

2. Focus improvements in areas where actual recreational uses are documented to exist and where stream hydrology supports recreation. Public recreation on urban streams is weighted toward tranquil flow, and warm weather conditions, making summertime impairments highest priority in most communities (Stiles 2008). (Some exceptions may exist on creeks and rivers where fishing or kayaking occurs during other periods of the year.)

3. Focus on dry-weather compliance first, then wet-weather. Dry weather sources of FIB are the most likely to be identified and effectively managed and are most likely to include on-going human sources. Elevated FIB in wet weather conditions is often very costly to address and stream standards are potentially unattainable. Additionally, during high flow conditions, flow-based physical risks (e.g., drowning) are often substantial; therefore, high-flow recreational use suspensions may be prudent to protect public safety.

4. Focus on identification of human fecal sources first, then consider non-human sources. Human sources of FIB are most likely to pose human health risks in urban areas, relative to diffuse natural sources such as birds and urban wildlife. (See discussion in Section 5.7 related to QMRA.)

5. Implement source controls first, then consider structural controls if source controls are unsuccessful. Source controls related to correcting human sanitary sources are particularly important, including practices such as correcting illicit sanitary connections

to the storm sewer system, repairing leaking sanitary infrastructure, addressing failing septic systems or poorly functioning package plants. Additionally, public education and enforcement of pet waste ordinances and leash laws, and equipping parks and trails with proper pet waste disposal cans in open space areas are basic source control steps. Work with local wildlife managers to assess the need for population controls or active management of urban wildlife.

6. Before embarking on capital investments in structural stormwater controls and treatment programs, conduct a systematic evaluation to prioritize implementation of practices that are most likely to provide meaningful benefits and reduce human health risk. Water quality models can help to select and screen potential alternatives and help to develop cost estimates. Effectiveness of control measures will vary depending on a variety of factors, beginning with the degree to which FIB sources have been correctly identified. Assuming that sources are correctly targeted, the expected effectiveness of structural controls will also vary and many unknowns remain.

In summary, a phased program of implementation is recommended, focusing first on correcting and removing human sources of FIB from dry weather sources. Once human sources are corrected and basic source controls are implemented, municipalities enter a phase of significant uncertainty regarding attainability of recreational stream standards. At this stage, regulatory and policy discussions are needed regarding use attainability, regulatory off-ramps and evaluation of risks to human health.

9.2 Case Studies

To illustrate issues discussed throughout this report, a set of case studies has been selected to illustrate strategies, challenges and costs associated with FIB impairments in urban waterbodies. Some case studies illustrate a single point, where others provide more detailed descriptions of multiple aspects of TMDL implementation plans, including costs.

9.2.1 San Diego River, California Comprehensive Load Reduction Plan

The San Diego River Watershed Comprehensive Load Reduction Plan (Geosyntec Consultants 2012) provides an example of a recent complex FIB TMDL and implementation plan. The San Diego River Watershed is located in central San Diego County, California. The river extends over 52 miles with a watershed area of approximately 434 square miles. The river ultimately discharges to the Pacific Ocean at Dog Beach in the Ocean Beach community within the City of San Diego. In response to requirements in an FIB TMDL, the Cities of San Diego, El Cajon, La Mesa, and Santee, County of San Diego, and Caltrans were required to prepare a Load Reduction Plan (LRP) outlining a proposed program of activities that will be capable of achieving TMDL-specified FIB load reductions. To qualify for an extended 20-year wet weather compliance timeline, a Comprehensive Load Reduction Plan (CLRP) was developed to address multiple pollutants. Load reductions are required during both dry weather and wet weather conditions within a 10- and 20-year compliance timeline, respectively. The compliance points for this watershed are the Pacific Ocean shoreline at the mouth of the San Diego River, as well as two locations within the main stems of the San Diego River and Forester Creek.

To identify a program of activities to achieve TMDL-required FIB load reductions during wet weather, the Structural BMP Prioritization and Analysis Tool (SBPAT) water quality model (http://sbpat.net/) was used to estimate the target FIB load reductions for various BMP implementation scenarios predicted to achieve compliance with the TMDL's allowable exceedance day-based WLAs. SBPAT screens areas based on need (i.e., pollutant load generation and downstream impairments), and then identifies opportunities (i.e., appropriateness of the area, adjacent storm drains) for BMP implementation. These opportunities are ranked based on factors such as effectiveness, cost, and maintenance requirements. In this case, non-structural BMPs were emphasized as the preferred implementation approach, particularly in the initial phases of CLRP implementation, because they are expected to be the most cost-effective way of reducing pollutant loading. BMPs ultimately implemented may require economic justifications related to available funding and perceived holistic benefit to taxpayers and residents.

To select non-structural BMPs for the plan, the parties first identified and prioritized FIB sources by considering various factors, including: 1) the magnitude and prevalence of the sources, potential threat to public health, and proximity to receiving water bodies; 2) results from microbial source tracking studies conducted in the watershed and region; and 3) best professional judgment. As a result, candidate non-structural BMPs identified in this CLRP include:

- Irrigation Runoff Reduction
- Residential/Small-Scale Low Impact Development (LID) Incentive Program
- Pet Waste Management
- Onsite Wastewater Treatment Source Reduction
- Identification and Control of Sanitary Sewer Discharge to the Municipal Separate Storm Sewer System (MS4) (which may include sewer upgrades)
- Commercial/Industrial Good Housekeeping Enhancements
- Animal Facility Waste Management Enhancements
- Homelessn Waste Management Program
- Redevelopment and New Development LID Implementation (Standard Urban Stormwater Mitigation Plan [SUSMP])
- Drain Inlet and Conveyance System Cleaning
- Street Sweeping

For structural BMPs, both regional and distributed controls were considered, recognizing that very few types are capable of effectively reducing FIB. As a result, the candidate regional structural BMP technologies focused on:

- Subsurface Flow Wetlands
- Infiltration Basins and Underground Infiltration Galleries
- Wet Ponds

Distributed structural BMPs considered included green streets, rainwater harvesting, and other Low Impact Development-type solutions. Other structural controls also considered in the plan included low-flow diversions to the sanitary sewer as a structural option to treat dry weather flows, streambank stabilization, and other practices.

The TMDL requires full compliance with the allowable exceedance frequencies (22% for wet weather and 0% for dry weather) by 2021 for dry weather and 2031 for wet weather. Tables 9-1 and 9-2 provide a summary of load reductions based on the various BMPs considered for wet and dry weather loads, respectively. Table 9-3 provides a cost estimate associated with implementation of the program elements. As shown in Table 9-3, the total projected cost range is $590 million to $1.34 billion for the entire watershed. An updated analysis by the City of San Diego identified a cost of $483 million for the City's portion of the watershed (Tetra Tech 2013).

Table 9-1. Summary of Wet Weather Load Reductions
(Source: Geosyntec 2012)

Stormwater Control/ BMP Category	FC Load Reduction (10^{12} MPN/YEAR) 1993 WY Load[1] [Low-High Range]
Regional Structural BMPs	880 [510 – 1,000]
Stream Restoration Projects	95 [22 – 170]
Distributed Structural BMPs	1,400 [780 – 1,600]
Non-structural BMPs	2,000 [710 – 3,200]
Private Property BMPs[2]	490 [280 – 560]
Subtotal	4,800 [2,300 – 6,600]
Overlapping Benefits Adjustment[3]	-620 [-280 - -880]
Load Reduction Effective Fraction[4]	0.23
Load Reduction Sum	**970 [460 – 1,300]**
Target Load Reduction[5]	**1,150**

[1] Range of WY1993 water quality benefits represent 25th and 75th percentile results. Average WY1993 water quality benefits are represented by 50th percentile results. Range reflects variability in baseline pollutant loading (primarily driven by land use EMC's) as well as variability in BMP effectiveness.
[2] Private property BMPs are an optional strategy and may be considered at the discretion of individual jurisdictions only if needed to meet load reduction targets.
[3] Adjustment made to avoid double counting of overlapping load reductions between non-structural and structural BMPs and between distributed and regional BMPs; improves reliability of results.
[4] Adjustment made to account for fraction of load reduction that is considered to be "effective" for reducing likelihood of exceedance in non-Annual Exceedance Days, therefore more improves reliability for comparing with Target Load Reduction.
[5] Target Load Reduction was estimated to achieve compliance with TMDL Annual Exceedance Days for fecal coliform for TMDL compliance year 1993.

Table 9-2. Summary of Dry Weather Load Reductions[1]
(Source: Geosyntec 2012)

Stormwater Control/ BMP Category	% of MS4 Area
Stream Restoration/Enhancement	1.7% - 9.4%
Non-structural BMPs	7.9% - 39%
Low Flow Diversions[2]	42% - 22%
Regional Structural BMPs[2]	40% - 24%
Distributed Structural BMPs[2]	2.8% - 1.7%
Filter + UV Treatment or similar (if needed)	0% - 3.7%
Load Reduction/Geographical Coverage	94% - 100%
Target Load Reduction	**>94% - 95%**

[1] Estimates are based on an assumption that non-structural BMPs are between 8% and 43% effective.
[2] Adjusted for overlapping coverage/benefits among various BMP types.

Table 9-3. 20-Year Cost Estimate to Achieve Bacteria TMDL Compliance in 2011 Dollars
(Source: Geosyntec 2012)

Cost Category	Lower Limit ($M)	Upper Limit ($M)
Non-structural BMPs	$38M	$104M
Infrastructure Improvement	$144M	$423M
Regional Structural BMPs	$59M	$141M
Distributed Structural BMPs	$66M	$219M
Stream Restoration Projects	$42M	$42M
Dry-Weather Diversion/Treatment	$19M	$43M
Private Property BMPs[1]	$216M	$360M
Special Studies	$3M	$6.5M
Monitoring	$3M	$3M
Total Cost Estimates	**$590M**	**$1,340M**

[1] Private property BMPs are an optional strategy and may be considered at the discretion of individual jurisdictions if needed to meet load reduction targets.

The overall cost of TMDL implementation for the City of San Diego over a 20-year period is estimated at $3.7 billion dollars, as reported in a study by the Point Loma University Fermian Institute (2011). This economic study also reports an expected economic benefit to San Diego residents of $617 million, primarily associated with reduced economic losses due to beach closures and human health-related expenses. This translates to a $57/yr benefit per resident and a $351/yr cost per resident for 20 years. (See the Fermian Institute report for details on methods used to estimate costs and benefits.) Given the extreme cost of the plan, the economic analysis recommended that a gradual, phased approach was needed, with low-cost practices implemented first.

The cost estimates in San Diego are within the range of costs estimated in other Southern California communities (as well as within the range of costs of "green infrastructure" being constructed in CSO cities). For example, the estimated cost for the Ballona Creek FIB TMDL implementation plan is over $1 billion (City of Beverly Hills et al. 2009), as summarized in Table 9-4. Ballona Creek is a relatively small (130 square miles), ultra-urban watershed located in the Los Angeles area. Additionally, for the 467 square mile Los Angeles River watershed, the Los Angeles River bacteria TMDL implementation cost was estimated at up to $5.4 billion by the Los Angeles Regional Water Quality Control Board (2010). Despite these significant expenditures, concerns were raised that the river may still not attain standards following the implementation activities (Flow Science 2010).

Table 9-4. Estimated TMDL Implementation Plan Costs for the Ballona River TMDL
(Source: City of Beverly Hills et al. 2009)

Ballona Creek Watershed BMPs	Treated Acres [2]	Capital Cost per Treated Acre	Total Capital Cost	O&M Costs per acre	Annual O&M
Structural BMPs					
Distributed BMPs	10,100[3]	$68,000	$686,800,000	*$2,800*	$18,180,000
Regional BMPs	1,840	$22,500	$41,400,000	*$600*	$1,100,000
Low Flow Diversion-1 (NOTF)			$10,600,000		$1,060,000
Low Flow Diversion-2 (Oval St)			$14,700,000		$1,470,000
Institutional BMPs					
Enhanced Street Sweeping			$840,000		$600,000
Downspout Disconnection			$88,400,000		$0
Enhance Pet Waste Pickup and Education Program			$2,000,000		$200,000
Subtotal			**$840,000,000**		**$22,600,000**
Program Management, Engineering, Administration, and Monitoring (20% of capital cost)[4]			$170,000,000		$4,500,000
Program Contingency (30%)			$250,000,000		$6,800,000
Total Cost			**$1,260,000,000**		**$34,000,000**

[1] Selected BMPs will address multiple pollutants including sediment, bacteria, metals and toxicity, and some will also result in reduced runoff volume discharges.
[2] Treated Acres based on draft Implementation Plan selected scenario assuming distributed BMP deployment as required to meet Bacteria TMDL load reduction target and eight regional BMP facilities.
[3] Excludes the acres that will be retrofit through the SUSMP program, as these costs would not be the responsibility of the responsible jurisdictions.
[4] The responsible agencies will require additional resources in order to manage the BMP implementation described in the Implementation Plan. The costs associated with this include administration, engineering, and ongoing monitoring of the program. The costs are estimated to be 20% of the total capital costs, or $160,000,000 through 2021. This cost would include increased staff for oversight of the design and implementation of the structural BMPs as well as implementation of the institutional BMPs (reviewing and enhancing existing policies, etc.).

9.2.2 Antelope Creek, Nebraska, Watershed Management Plan

Antelope Creek flows through the urbanized portion of Lincoln, Nebraska near the downtown area and the University of Nebraska, Lincoln campus. The stream is an urban concrete-lined channel, which is accessible for shallow wading; however, it is not used for swimming. In 2007, elevated FIB was identified and the Nebraska Department of Environmental Quality (NDEQ) developed a TMDL based on a Load Duration Curve approach (Cleland 2003, 2007; EPA 2007d). The recreation season geometric mean concentration of *E. coli* at the confluence with Salt Creek measured by NDEQ in 2004 used to develop the TMDL was 3,433 cfu/100 mL, relative to a stream standard of 126/100 mL. After accounting for a margin of safety, the 2007 TMDL identified 113 cfu/100 mL as the reduction goal for Antelope Creek. Using more recent data collected during the 2010-2011 sampling activities, 1,511 cfu/100 mL was considered to represent the baseline from which a 93% reduction in the *E. coli* would be needed to attain the TMDL target.

Antelope Creek near downtown, Lincoln, NE, showing paths and small amphitheaters as part of flood-proofing reconstruction projects. (Photos courtesy Bob Pitt.)

In 2010, the City embarked on the development of a watershed plan (EA Engineering et al. 2012) to address FIB and improve overall water quality. Dry weather sampling of stormwater outfalls indicated that the most probable source of elevated FIB was urban wildlife (e.g., pigeons, raccoons), along with possible domestic pets and other natural sources. In order to develop alternatives for reducing FIB loads, WinSLAMM was calibrated to local conditions and a variety of alternatives were developed (Pitt 2011a,b). The primary approach to reducing FIB loading focused on curb-cut bioretention retrofits in order to infiltrate large amounts of runoff and simultaneously retain the associated FIB. The cost of fully implementing the approach was estimated at $57 million over 40 years for a 7.7 square mile area ($7.4 million/square mile, or about $12,000/acre). Given the major total cost of such a plan, the city will begin by focusing on source controls and a limited number of pilot projects under a 5-year plan (Table 9-5), then reevaluate future steps for possible long-term implementation, including potential coordination with regular neighborhood capital improvement projects.

Table 9-5. Antelope Creek Bacteria TMDL Phase 1 Cost Estimate
(Source: EA Engineering et al. 2012)

Phase One: Structural Stormwater Controls	**Estimated Cost ($)**
P01: Antelope Park: Van Dorn St to Sheridan Blvd	125,000
P02: Antelope Park: South St to Van Dorn St	125,000
P03: Antelope Park: SW of 33rd and South St	125,000
P04: Antelope Park: A Street to South Street	250,000
P06: Lincoln Children's Zoo	425,000
Sub-total	**$1.1 million**
Phase One: Non-Structural Stormwater Controls	**550,000**
Phase One: Review, Monitoring, Plan Revision	**50,000**
Grand Total	**$1.7 million**

The primary components of the non-structural stormwater control program include a combination of activities implemented by the City and/or Lower Platte South Natural Resources District:

- Retrofitting older bridges and overpasses crossing Antelope Creek to limit bird activity (pigeon roosting on bridges over the creek was the most noticeable FIB source observed).
- Sanitary sewer line inspection program expansion.
- Dry weather storm drain screening.
- Enforcement of existing pet waste ordinances.
- Supplying and maintaining additional pet waste containers.

Other non-structural stormwater controls listed below would be implemented by residents and property owners through programs offered by the City and/or LPSNRD (some of these are oriented to general water quality improvement, not just *E. coli*):

- Low/no-phosphorus fertilizer program.
- Rooftop disconnection incentive program.
- Rain garden program.
- Rain barrel program.

Using GIS, potential locations for Phase 1 pilot projects were inventoried and selected based on a ranking system that considered factors such as land ownership (publically owned property was preferred), potential for public education/demonstration, adequate space available to implement the stormwater controls, locations with the greatest potential for pollutant removal, and size of drainage area treated. The majority of the pilot projects focused on various bioretention applications and modification of existing drainage infrastructure to improve water quality. Results from the initial phase of the implementation plan are not yet available.

9.2.3 Rock Creek, Montgomery County, Maryland, TMDL Implementation Plan

The Rock Creek Watershed comprises approximately 76 square miles (48,640 acres), with approximately 80% of the drainage area within Montgomery County, Maryland, and the remaining 20% within Washington, D.C. (EPA 2007). The Montgomery County portion of the wasteload allocation was identified as 40% of the bacteria TMDL target, and wildlife portion of the TMDL WLA for the MS4 comprises 52% of the baseline load (Table 9-6).

Table 9-6. Montgomery County, MD Rock Creek Bacteria TMDL Allocations

Load Component	Components of Baseline	Billion MPN/yr	As % of Respective Total
Total Baseline Load		1,443,575	
	Human	170,342	11.8%
	Domestic	277,166	19.2%
	Livestock	435,960	30.2%
	Wildlife	557,220	38.6%
Total Mont. County (MS4) Baseline Load		453,669	31%
	Domestic	218,961	48%
	Wildlife	234,708	52%
TMDL Target Load		45,625	
Mont. County TMDL Target WLA		18,195	40%

The practical implication of the WLA target is that even if Montgomery County retrofitted the developed areas within the MS4, the limit of the technology makes it impossible to meet the target. Added challenges include how to manage the “urban wildlife” loads. These sources are diffuse, difficult to target and are impractical to target for treatment, unless eradication programs are implemented. Given these practical constraints, the TMDL appears to be established in a manner that non-compliance is inevitable, despite the development of an implementation plan, as described below.

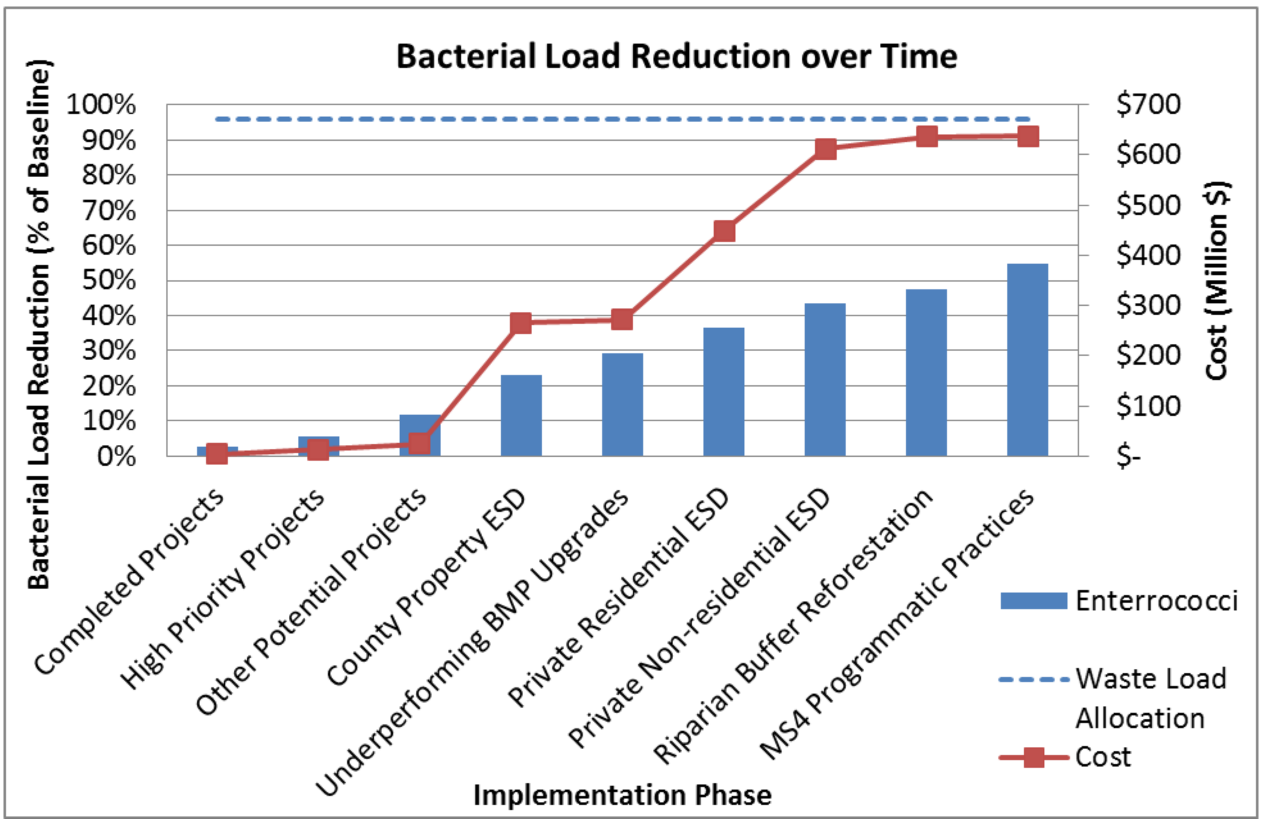
The Watershed Treatment Model (WTM), a simple spreadsheet loading model developed by the Center for Watershed Protection (2001) was run iteratively using a series of assumed structural and programmatic strategies targeting FIB load reduction throughout the watershed and within the MS4 jurisdictional area. Strategies included larger structural practices such as wet ponds and constructed wetlands (new and retrofits); smaller low impact development (LID) practices such as green streets, parking lot bioretention, and rooftop runoff capture; residential lot practices such as rain gardens; habitat restoration initiatives such as reforestation; and education initiatives that target pet waste management. These strategies were also targeted to help comply with other TMDL and MS4 permit compliance requirements. Associated planning level costs with each approach were also applied to provide a sense of the relative cost/benefit of the various strategies. Table 9-7 below provides a summary of the analysis, corresponding to Figure 9-1.

Table 9-7. Summary of Bacteria Load Reduction and Associated Costs for Rock Creek

Implementation Phase	Enterococci Loading (% reduction toward Baseline)	Description of Activities	Cumulative Cost ($ million)
Baseline Load	**0%**		$ -
Strategy 1	12%	Larger structural practices	$ 24.2
Strategy 2	43%	LID strategies (public and private property)	$ 612.5
Strategy 3	47%	Habitat restoration	$ 636.3
Strategy 4	55%	MS4 programmatic practices (education)	$ 637.5
TMDL WLA	**96%**		

The Montgomery County restoration strategy is further illustrated in Figure 9-1, where the implementation phases are shown in order with their resulting FIB load in comparison to the WLA. The cost for each implementation phase is also shown. The greatest reduction is attributed to environmental site design (ESD) strategies, while pet waste education (an MS4 programmatic practice) was the most cost-efficient strategy. Even by implementing the maximum realistic restoration potential of all four strategies, the County plan is only capable of reaching about 60% of the target reduction.

Figure 9-1. Rock Creek Bacteria Loading Over Time of Restoration Implementation



9.2.4 The Houston Metropolitan Area Bacteria TMDL Implementation Plan

FIB are the most common water quality impairment in the Houston-Galveston region of Texas. Waterbodies designated as impaired are required by the Clean Water Act to develop a bacteria TMDL for each waterbody's segment. Once a TMDL is completed, development of an Implementation Plan (I-Plan) is required to identify and recommend strategies, controls and practices to reduce the pollutant and restore the waterbody. The Texas Commission on Environmental Quality (TCEQ) requested that a stakeholder group be formed to develop the I-Plan(s) for the numerous bacteria TMDLs under study in the region. With stakeholder preference for a common I-Plan, TCEQ grouped several impaired segments together to create these bacteria TMDL projects. The project areas also shared local jurisdictions which provided a suitable platform for the stakeholders to develop an integrated I-Plan. The I-Plan project area is about 2,204 square miles with a population of approximately 4 million.

Seventy-two TMDLs for bacteria were adopted by the TCEQ for the Houston-Galveston metropolitan area over a 10-county area during the 2009 through 2011 time period using the Load Duration Curve (LDC) approach (Cleland 2003, 2007; EPA 2007d). These are described below:

- Eighteen TMDLs in Buffalo and Whiteoak Bayous and their tributaries were adopted on April 8, 2009.
- Nine TMDLs were adopted in Clear Creek and its tributaries were adopted on September 10, 2008.
- Eight TMDLs for bacteria were adopted in the Greens Bayou watershed on June 2, 2010.
- TMDLs for 18 segments in Brays, Sims, Halls and in eastern Houston Bayous were adopted on September 15, 2010.
- TMDLs for watersheds upstream of Lake Houston were adopted on April 6, 2011.

The stakeholder group, known as the Bacteria Implementation Group (BIG), included members representing municipalities and county governments; special districts; local, state and federal resource agencies; business, agricultural, and engineering interests; conservation and watershed groups; and the public. The I-Plan's recommendations are representative of the work of the BIG members and hundreds of interested citizens who actively participated on numerous BIG subcommittee workgroups and in the public process over a two year period.

The goal of the BIG is to assist in the reduction of bacteria in the region's impaired waterways and become suitable for their current designated uses. The BIG is to accomplish this goal by coordinating, implementing, assessing and revising the I-Plan. The BIG is the decision making body for the I-Plan.

As stated in the document, the I-Plan is a flexible tool that governments and non-governmental organizations will use to guide program management through voluntary or regulatory measures. Progress is to be evaluated on a regular basis with updates and changes being made to the I-Plan, as needed. The I-Plan provides:

- Steps the TCEQ and the stakeholders will take to achieve the pollutant reductions identified in the TMDL reports.
- The schedule for implementation activities.
- A description of the legal authority under which the participating agencies may require implementation of the recommended activities.
- A tracking and monitoring plan to determine the effectiveness of the implementation activities.
- Measureable outcomes for assessing progress.
- Communication strategies that will be used.

Of note is the I-Plan's provision that allows for the addition of watersheds into the existing program footprint if TCEQ adopts new bacteria TMDLs for waterways located near or adjacent to the BIG project area.

The BIG proposes an adaptive management approach that allows for the implementation of practical controls while additional data collection and analysis are conducted. The I-Plan states that the cost-effectiveness of the recommendations will need to be tested early during implementation so the overall strategy can be can be adapted to emphasize those measures. Results from the numerous sub-committee workgroups culminated into the recommended implementation strategies comprising the I-Plan. A summary of these workgroup results and the BIG's recommended implementation strategies are presented in Table 9-8.

Table 9-8. Summary of Recommended Implementation Strategies for the I-Plan
(Source: TCEQ 2013)

I-Plan Section	Activity Category	Focus of Implementation Activities
Implementation Strategy 1.0	Wastewater Treatment Facilities	Increase monitoring requirements, impose stricter bacteria limits, require updates to facilities not able to comply with limits, and increase enforcement.
Implementation Strategy 2.0	Sanitary Sewer Systems	Require all systems to develop and implement a utility asset management program and to protect against power outages at lift stations.
Implementation Strategy 3.0	On-Site Sewerage Facilities	Address failing systems and inadequate maintenance.
Implementation Strategy 4.0	Stormwater and Land Development	Expand stormwater management programs, develop a recognition program, and petition the TCEQ to facilitate reimbursement of bacteria reduction measures.
Implementation Strategy 5.0	Construction	Improve compliance and enforcement of existing stormwater management permits
Implementation Strategy 6.0	Illicit Discharges and Dumping	Increase efforts to address direct and dry-weather discharges, and better control waste hauler activities.
Implementation Strategy 7.0	Agricultural and Animal	Expand existing cost-share programs and the management of feral hog populations.
Implementation Strategy 8.0	Residential	Expand public education efforts.
Implementation Strategy 9.0	Monitoring and I-Plan Revision	Maintain databases of ambient and non-ambient water quality monitoring data and implementation activities, review I-Plan progress, and update I-Plan.
Implementation Strategy 10.0	Research	Examine effectiveness of stormwater activities, bacteria persistence and regrowth, and appropriate indicators for use in water quality monitoring.
Implementation Strategy 11.0	Geographic Priority Framework	Consider recommended criteria when selecting geographic locations for projects.

9.2.5 South Platte River, Colorado, FIB Control under MS4 Permit Requirements

In 1998, the South Platte River (Segment 14) was placed on the Colorado 303(d) List as impaired by *E. coli*, with an *E. coli* TMDL issued by the Colorado Water Quality Control Division in 2007. The river flows through the metro Denver area and is used for kayaking in some areas, with a few access locations where water play may occur. The South Platte *E. coli* TMDL was the first FIB TMDL issued in Colorado and basically focused on dry weather discharges from the storm sewer system, which are enforced through requirements in Denver's MS4 permit. Primary requirements associated with the TMDL include: 1) monitoring to identify outfalls of concern (primary outfalls); 2) implementing a storm sewer system maintenance program, 3) marking storm drain inlets, 4) education and outreach, 5) implementing other stormwater controls as needed and 6) conducting annual analysis of monitoring data. A 10-year compliance schedule was allowed. To date, Denver's primary focus has been on maintenance activities including cleaning (jetting) storm and sanitary sewers, eliminating illicit connections to the storm sewers, identifying and eliminating cross connections between the storm and sanitary sewers, and repairing damaged sanitary infrastructure and disconnected taps.

In six priority basins, Denver's efforts have been successful in reducing *E. coli* in discharges from storm drains to below instream standards, as shown in Figure 9-2 and Table 9-9. In other basins, elevated *E. coli* still exist at outfalls. Despite reductions at multiple outfalls, instream *E. coli* remains elevated. The Denver example illustrates a situation where sanitary sources of *E. coli* existed and were in need of correction; however, the instream response to these improvements remains inconclusive, and the stream still does not meet primary contact recreation standards, despite these reductions at outfalls.

Figure 9-2. Reductions in *E. coli* Discharges from an Example Priority Outfall in Denver (Source: Novick 2013)

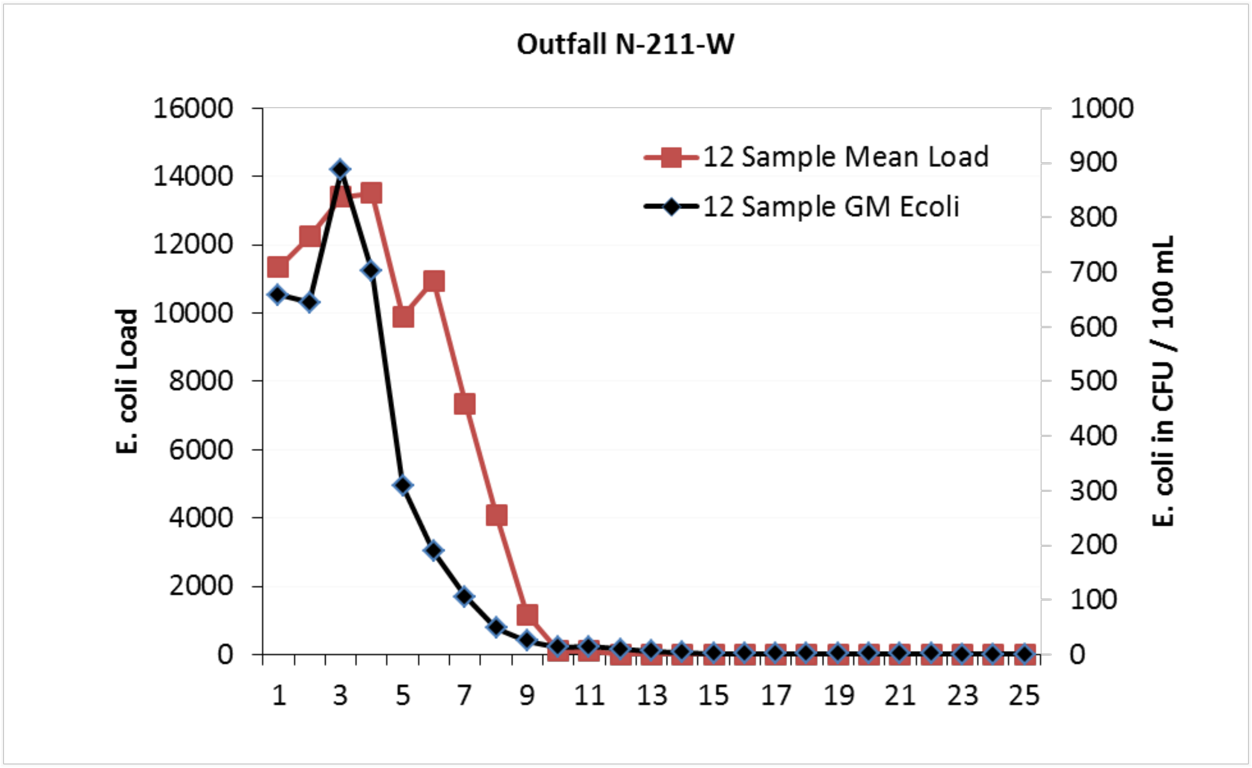


Table 9-9. Comparison of *E. coli* Discharges in Denver's Priority Outfalls Before and After Implementation Activities
(Source: Novick 2013)

	Before Implementation			**After Implementation**		**Statistically Significant Difference?**
Outfall ID	**Number of Samples**	**Median *E. coli* (CFU / 100 mL)**		**Number of Samples**	**Median *E. coli* (CFU / 100 mL)**	**p**
S-242-E	37	200		0	NA	NA
S-191-W	46	240		3	230	0.84
N-42-W	15	510		9	230	0.28
N-201-W	**15**	**660**		**38**	**1**	**0.038**
N-211-W	**13**	**440**		**36**	**1**	**0.002**
N-221-W	**15**	**3600**		**36**	**710**	**0.001**
N-311-W	**17**	**6600**		**6**	**810**	**0.002**
N-411-E	9	1090		60	465	0.428
N-433-E	17	2700		55	900	0.126
N-453-E	**25**	**140**		**20**	**1**	**0.000**

9.2.6 Boulder Creek, Colorado, TMDL Implementation Plan and Raccoon Controls

Boulder Creek flows through the City of Boulder, Colorado and includes a subreach identified as impaired for *E. coli*. In 2011, the city completed a third-party TMDL using a load duration curve approach and developed an implementation plan (TetraTech 2011). Special studies of the creek identified few clear relationships between *E. coli*, human *Bacteroides* and a toolbox of other indicators, but generally ruled out human sources of elevated *E. coli*. The city embarked on pilot projects related to controlling raccoons in storm drain systems, which successfully reduced *E. coli* in dry weather flows at end of pipes in initial pilot projects. However, statistically significant reductions in instream *E. coli* have not occurred, with *E. coli* remaining elevated above primary contact recreation standards.

The pilot project basically consisted of retrofitting inlets with grates to keep raccoons from entering the storm drain, retrofitting outlets with spring-shut grates, and cleaning (jetting) the storm drain. A challenge of this approach included keeping the outlet grate from clogging with debris and litter, along with significant cost in retrofitting inlets to maintain the design capacity of the inlet for purposes of public safety. The cost of the initial pilot project to retrofit 20 inlets, install three check valves near outlets and implement two curb extension practices was approximately $310,000. Expanding this to the full subbasin to address 72 inlets was $1.2 million (also including 12 curb-cut bioretention locations) (HDR 2013). Due the extremely high cost of the retrofits, the next phase of the pilot projects has not yet been implemented, pending exploration of more cost-effective source controls that may help to reduce *E. coli* instream. Additional instream monitoring is needed to assess whether the success at end-of-pipe is present in the stream itself.

Raccoons inhabiting storm sewer system (a) and retrofitted storm drain inlet (b).
(Photos courtesy of Andy Taylor, City of Boulder, CO)

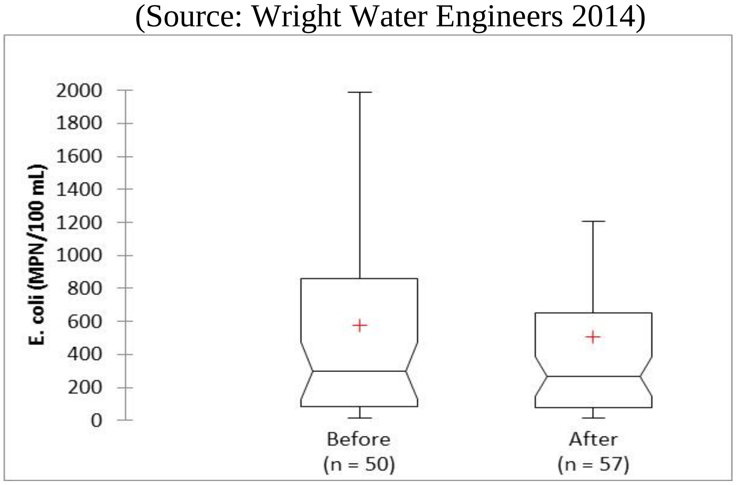
9.2.7 Big Dry Creek, Colorado

Big Dry Creek flows through the northwestern suburbs of Denver, Colorado and experiences highly managed hydrology due to Standley Lake, which is a drinking water and irrigation water reservoir. The stream itself is not typically used for recreation; however, it has a "potential primary contact" standard due to unrestricted access. The stream is not suitable for boating or swimming, but could be accessed for wading in some locations since the stream is not fenced to preclude access.

A TMDL is currently under development for Big Dry Creek, but has not yet been completed, in part due to challenges associated with using the Load Duration Curve approach in the context of highly managed (unnatural) stream hydrology. Nonetheless, a voluntary watershed association, consisting of six cities and counties has undertaken several special studies to identify and attempt to reduce elevated instream *E. coli*. As part of these voluntary efforts, dry weather screening of storm outfalls was completed for the urbanized portion of the watershed. Following dry weather screening procedures developed by the Center for Watershed Protection et al. (2004), one illegal sanitary sewer connection to the storm drain was identified based on a combination of indicators such as significantly elevated *E. coli* (e.g., 7,000 MPN/100 mL), evidence of toilet paper and deposits at the outfall, and monitoring of temperature fluctuations using in situ temperature probes (which suggested showering, toilet flushing, etc.). The illegal connection (due to a plumbing error when the house was constructed) was corrected; however, seven years later, no statistically significant difference ($p = 0.76$) in *E. coli* has occurred instream based on the Mann-Whitney analysis of before versus after removal of the illegal connection, as shown in Figure 9-3.

The reach of stream in the urbanized portion of the watershed with elevated *E. coli* is in a stream segment where the adjacent cities have preserved a wide open space corridor along the stream, including unmanicured native riparian vegetation. Wildlife (e.g., beavers, coyotes, geese, swallows) inhabiting the riparian corridor is suspected to be the primary source of the elevated *E. coli* instream, with possible contributions from dogs; however, molecular source tracking costs have been outside of the voluntary association's operating budget to date.

Figure 9-3. Big Dry Creek Instream *E. coli* Before and After Correction of an Illicit Sanitary Connection to the Storm Sewer System
(Source: Wright Water Engineers 2014)



9.2.8 Chemical Source Tracking in Kitsap, Washington

The costs of identifying and correcting pollution sources can be significant, a problem that is magnified when dealing with large areas or multiple small watersheds. Effective use of investigative tools can help reduce these costs by focusing limited resources on those areas with demonstrated anthropogenic sources of bacteria pollution. This recent chemical source tracking study in Kitsap, Washington, provides an overview of the strengths and limitations of this approach, including factors that can lead to false negatives and false positives.

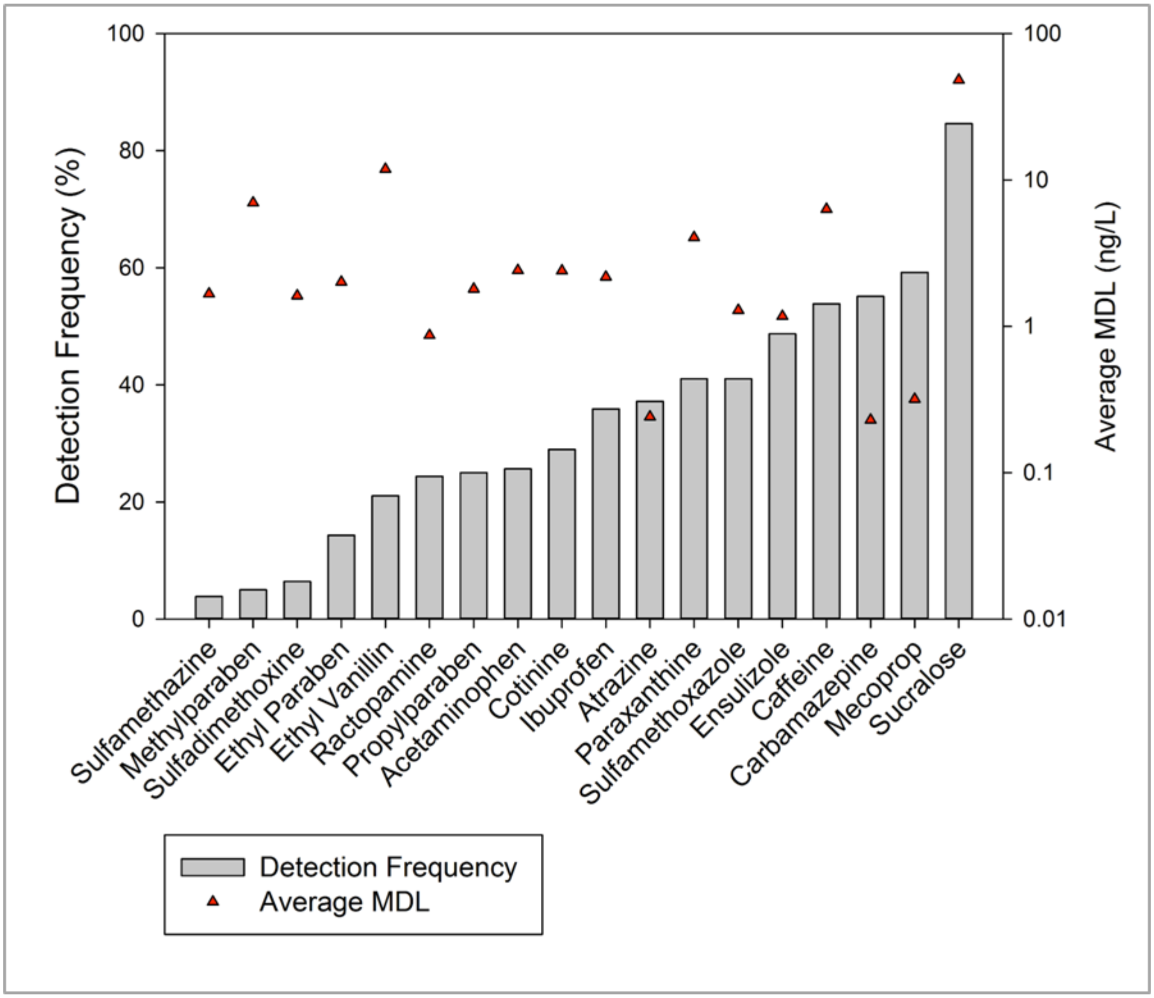
In 2013, the Kitsap Public Health District worked in partnership with the University of Washington Tacoma to evaluate the utility of a suite of chemical compounds, referred to as contaminants of emerging concern (CEC), for identifying sources of fecal coliform bacteria contamination (Kitsap Public Health District 2013). These chemicals include caffeine, nicotine, artificial sweeteners, as well as those present in personal care products, medicines, and some used in lawn care such as herbicides.

To test this method, six sampling events were performed at approximately 20 sites representing a range of impacts from bacterial contamination. Sites were all freshwater discharges to the Puget Sound, generally occurring at the shoreline. These consisted of small, local drainages discharging as seeps through bulkheads, localized drainage culverts, or small streams. Site evaluation was performed to ensure sampling sites represented a range of conditions with regard to anthropogenic bacterial contamination based on historical sampling information. General site classifications included those impacted by failing septic systems, leaking sewer systems, and agricultural activities. There were also some sites with a record of bacterial contamination but without an identifiable source. Control sites were streams with a record of no bacterial contamination in watersheds with minimal development.

Samples were processed prior to analysis using filtration, stabilization of pH, and solid phase extraction followed by elution and evaporation. Efficiency of the extraction was accounted for in the final concentrations using a set of isotopically labeled surrogate standards. Chemical analysis was done by High Performance Liquid Chromatography-Tandem Mass Spectrometry (HPLC-MS/MS, triple quadrupole). Quantification of CECs was achieved via isotope dilution, and calibration was done using inverse concentration weighting. This sample extraction and concentration and HPLC-MS/MS instrumentation is critical to achieve the low detection limits of ng/L (parts per trillion) required for CEC work. The CEC are very dilute in water samples, and analysis done with higher detection limits would likely result in non-detects.

Analytical results for all samples were compiled to estimate the extent and magnitude of CEC input into the Puget Sound from these sources. A plot of detection frequency and method detection limit (MDL) for selected analytes is shown in Figure 9-4. Figure 9-5 shows the detection frequency and measured concentrations of CECs for all sites and sampling events.

Figure 9-4. Summary of Detection Frequencies and Method Detection Limits for All Sites and Samples (n ~ 80)
(Source: Kitsap Public Health District 2013, prepared by A. James, University of Washington)



Using CEC to determine the extent of anthropogenic sources of bacterial pollution in a watershed is promising, but further work is needed to refine which compounds correlate best with different types of sources, and the detection limits that will provide useful information with minimal risk of false positives. With that in mind, the following are preliminary findings of the research project described above:

- In this investigation, higher concentrations of CEC are generally associated with impacts from known sources of sewage. These concentrations are also strongly influenced by type of source, pathway, and site conditions that effect the dilution and/or degradation of chemical compounds. In addition, concentrations of CEC were found to vary depending on the time of day at some locations. Due to these factors, setting specific concentration limits to determine which sites are impacted by anthropogenic pollution sources could create false negatives.

- Since a given CEC may not be used in individual homes, no one compound is adequate to determine impacts from septic systems. A group of CECs shows better promise in identifying failing systems. This group includes commonly-used compounds such as acetaminophen, ibuprofen, and caffeine. In addition, metabolites such as paraxanthine and theobromine (from caffeine), and cotinine (from nicotine) should only be present from wastewater impacts.

- Other compounds which are persistent in the environment were found at low levels even in the cleanest streams used as a control group. Examples include sucralose and ensulizole, which only correlated well with known bacterial sources when found in higher concentrations.

- The presence of labile CECs (those which likely degrade relatively quickly in the environment) may indicate proximity of human wastewater source.

- Infrequently used compounds, such as certain prescription drugs, may be useful for identifying influence of leaking sewer pipes. Examples include carbamazepine and sulfamethoxazole. These are relatively rare in individual septic system effluent but were consistently detected in sewage treatment plant influent, even from small WWTPs (~0.2 MGD). In trunk lines, larger collection areas should result in higher probability of detection.

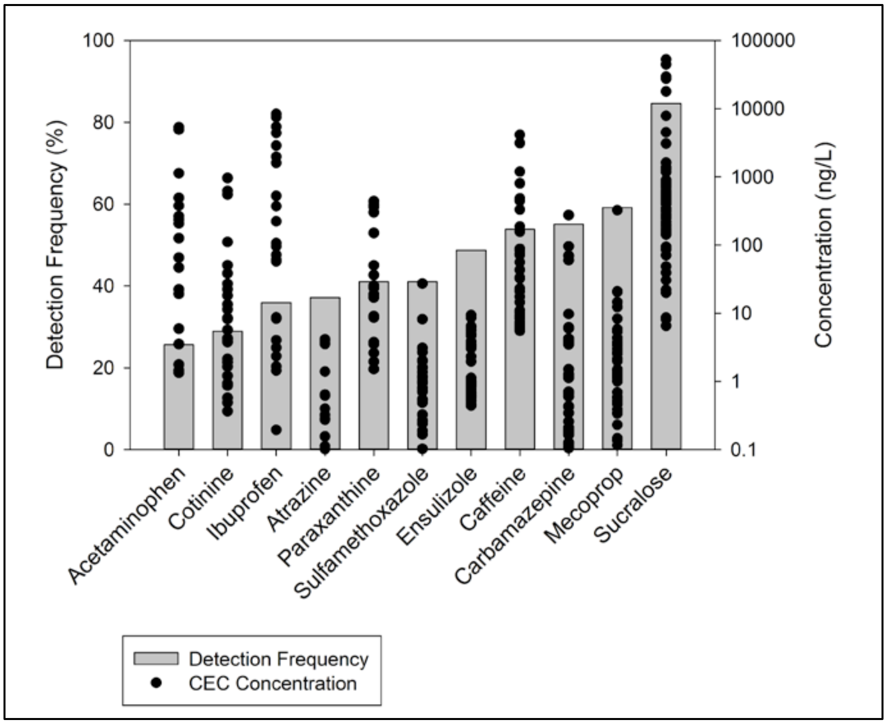
- Sucralose may also be a good conservative tracer of human wastewater. As mentioned earlier, this compound has been found to be nearly ubiquitous in the environment. It has minimal degradation through wastewater treatment plants and slow environmental degradation. With a median concentration in sewage around 30,000 ng/L, elevated levels of sucralose may predict the occurrence of other CECs.

- Additional compounds may be useful for identifying impacts from livestock and dairy sources. Examples could include veterinary medicines and animal feed additives.

- Sites with no CECs strongly suggest no anthropogenic influence.

Figure 9-5. Detection Frequency and Measured Concentrations of CECs for All Sites and Sampling Events

(Source: Kitsap Public Health District 2013, prepared by A. James, University of Washington)



9.3 Conclusions

The costs of implementation measures expected to be necessary to attain FIB TMDLs are staggering, exceeding billions of dollars in metropolitan areas, even in the absence of CSO issues. Additionally, the likelihood of attaining FIB standards is highly uncertain, even with such expenditures. Combined with uncertainty related to the human health effects associated with elevated FIB in urban runoff, there is a need for a phased approach to TMDL development in urban areas. Ideally, TMDLs should be developed after reasonable identification and quantification of sources has been completed so that better informed WLAs and LAs can be implemented. Without reasonable identification and quantification of sources, WLAs may not be achievable or equitable. For example, some bacteria TMDLs have been developed that require a blanket percent reduction of existing loadings (e.g., Willamette River TMDL in Oregon) because the source loadings could not be reasonably quantified. Percent reduction TMDLs do not consider that treatment becomes more difficult as concentrations are reduced or that baseline concentrations for some sources may already be below the water quality standard responsible for the impaired listing. Given these and other challenges associated with FIB TMDLs, care should be taken to develop TMDLs that clearly recognize constraints of available data at the time of TMDL development, allow for phased implementation, and allow principles of adaptive management to be applied in implementation plans.

10 CONCLUSIONS AND RESEARCH NEEDS

The single most frequent cause of water quality impairment in the U.S. is elevated fecal indicator bacteria (FIB) (EPA 2014). FIB-related impairments can have significant and costly implications for local governments, businesses, and watershed stakeholders due to beach closures and TMDL compliance and implementation requirements to address these impairments. TMDLs and associated MS4 NPDES permit requirements for FIB load reductions pose unique challenges relative to TMDLs for chemical constituents. FIB are living organisms that occur naturally in the environment and whose sources can move freely throughout watersheds and storm drain systems, even when anthropogenic sources of FIB are controlled. Furthermore, FIB are generally not a direct cause of human health impacts; instead, they are easy-to-measure surrogate parameters that are intended to infer that fecal wastes and associated pathogens may be present. Nonetheless, FIB are currently considered to be the best available practical alternative to monitoring for multiple pathogens associated with human and animal wastes. Although the human health risk associated with exposure to waters impacted by untreated or poorly treated human sewage is well documented, the health risk from recreational exposure to elevated FIB in urban runoff-impacted receiving waters is less well known.

The state of the art and practice in modeling transport and fate of FIB (and pathogens) involves significant uncertainty, more so than traditional water quality constituents. This uncertainty carries forward into evaluation of FIB management strategies, development of appropriate wasteload and load allocations for TMDLs, and regulatory decisions. Nonetheless, MS4 owners/operators are often assigned wasteload allocations in urban FIB TMDLs and may face significant wasteload reduction requirements, which are enforceable through MS4 discharge permits. Although management and correction of human sources of FIB (e.g., leaking sanitary infrastructure, illicit connections, dumpster drainage) to storm sewer systems can reduce FIB loads posing human health risk, many MS4s will need to reduce FIB from other sources as well to meet wasteload reduction targets. Identifying the sources of FIB and their relative contributions can be complex and costly. Load reductions are difficult, especially for the natural, non-human FIB sources, for multiple reasons (e.g., ubiquitous nature of FIB, current limits of technology related to urban stormwater controls, magnitude of reductions targeted). For these and other reasons, there are real questions regarding the attainability of FIB water quality standards in urban watersheds and in MS4 discharges. Depending on the sources of FIB affecting a particular receiving water and the manner in which MS4 permit compliance is assessed, dry weather standards may be attainable in some cases, but consistently attaining standards under wet weather conditions may be infeasible.

This report provides those involved with urban FIB TMDLs and other FIB-related receiving water impact issues with this information:

1. A consolidated, understandable synopsis of the underlying science associated with current Recreational Water Quality Criteria (RWQC) and the tools available for seeking site-specific criteria.

2. An understanding of potential sources of FIB in urban areas and tools to identify these sources.

3. Guidance on monitoring FIB and interpreting monitoring results.

4. An overview of source controls and structural stormwater controls that may be considered to reduce FIB loads.

5. A practical strategy for prioritizing activities associated with FIB TMDLs.

Due to the breadth of topics addressed, this report has not addressed any single topic exhaustively, but instead points users to other resources where more in-depth information can be obtained. Recommendations for additional research are also provided, given the billions of dollars of investments that MS4s and others are anticipated to require in order to address FIB impairments.

A summary of key background information and findings of this report includes:

1. In 2012, EPA updated the RWQC, which established health-based water quality criteria intended to protect human health in the context of primary contact recreation in streams and lakes. These criteria serve as guidance for states for purposes of developing water quality standards. The criteria are based on epidemiological studies conducted primarily at lake and ocean beaches at locations affected by FIB and pathogens associated with sources mostly of sanitary (human) origin.

2. Epidemiological and QMRA studies regarding human health risks associated with recreational activities in urban runoff impacted receiving waters, particularly during wet weather, remain limited, and conclusions regarding human health risks associated with urban stormwater systems are mixed. Additionally, EPA-sponsored literature reviews and QMRA studies have shown that human health risks associated with zoonotic (animal) sources of FIB and pathogens may vary depending on a variety of factors. Although many experts agree that non-human sources of FIB and pathogens generally pose a lower risk of human illness than human sources, EPA did not have adequate information to provide national source-based exclusions in the 2012 RWQC, and instead developed risk-based criteria based on specific gastrointestinal illness rates.

3. Receiving waters with primary contact recreation use classifications in most urbanized areas must comply with standards based on the RWQC, regardless of the source of FIB. However, under the 2012 RWQC, EPA allows options for development of site-specific standards that provide equivalent protection to EPA's recommended criteria. These alternative standards generally become a viable option only after human sources of FIB have been controlled. Scientific methods that can be used to support alternative standards generally include either epidemiologic studies or QMRA. Although QMRA is less costly than an epidemiological study, both approaches require significant scientific expertise and are expensive to implement. Sanitary surveys, possibly including microbial source tracking techniques, are also important evidence needed for developing site-specific standards in urban areas.

4. Sources of FIB in urban environments can include both human and non-human sources. A variety of source identification approaches can be used, depending on local conditions and budgets. The first step in addressing FIB impairments is to inventory the various FIB

sources specific to the watershed, and prioritize human FIB sources first, given the greater public health risks they may present. Although municipal WWTPs are not typically a significant source of elevated FIB in urban receiving waters, sanitary sewer collection systems can contribute human waste, particularly in areas with aging infrastructure (e.g., leaky sewer lines), SSOs, or CSOs. Other urban sources of human waste include homeless, RV discharges, and septic systems. The second management priority is control of non-human anthropogenic sources contributing to FIB loading, which include pet waste, fertilizers, trash, and dumpster leaks, to the extent that they are controllable. The third and lowest priority of FIB control is non-anthropogenic sources, which include urban wildlife, plants, soils, and decaying organic materials. Recent scientific advances in MST allow fecal sources to be more reliably and quantitatively identified, with validated source markers available for such categories as human, canine, gull, horse, pig, and ruminant. Such tools can be used to support a comprehensive source identification investigation, where conditions warrant advanced investigations.

5. FIB concentrations in wet weather urban discharges from separate storm sewer systems are typically orders of magnitude above primary contact recreation standards, regardless of the land use. FIB in dry weather urban runoff may also be elevated, depending on site-specific conditions. FIB in waters receiving runoff from natural areas may also sometimes exceed primary contact standards. Regulatory flexibilities based on high-flow recreational use suspensions and allowable exceedances frequencies based on reference (natural) watershed conditions vary depending on state regulations, but are not explicitly addressed in the federal RWQC.

6. FIB monitoring results, given their large variability, do not provide the statistical confidence or power necessary to form statistically significant conclusions, such as regarding spatial or temporal patterns, unless very large numbers of samples are available. FIB sources, fate, and transport dynamics contribute to this large variability in concentrations. FIB are living organisms that die-off, grow, and persist, depending on environmental conditions. For example, particle-associated FIB may settle out of the water column and persist (and reproduce) in sediments for long periods of time, then be resuspended in the water column during periodic high flows. Additionally, FIB sources vary seasonally and may change over short time periods. For example, illicit discharges may be intermittent, and stormwater discharges occur episodically. For this reason, it is critically important that decisions for TMDLs and proposed control strategies be based on robust data sets that represent each critical period. Monitoring to identify or confirm the absence of human sources should be a high priority. This typically includes dry-weather sampling of storm drain outfalls, visual and/or CCTV inspection of storm drain networks, and receiving water monitoring programs to identify areas where more intensive source monitoring may be needed.

7. Urban stormwater quality mathematical/computer models, such as watershed models that are typically used for TMDL development and/or implementation, have more limited predictive capability for FIB than for other conventional urban stormwater pollutants. This is due to the relatively smaller input datasets (such as regional land use event mean concentrations), as well as the greater uncertainty regarding FIB sources, fate and transport (parameters which, unless directly measured, require calibration to match

receiving water monitoring data). Robust monitoring datasets are needed for model setup, calibration, and verification; however, watershed-specific datasets are often costly to develop. Where regional or national datasets are used (such as for land-use based concentrations), interpretation of model results should carefully consider results of sensitively and uncertainty analyses, and should recognize current limitations of the state of the practice. Thus, watershed modeling studies for FIB should place an emphasis on the development of robust and representative input and calibration datasets, as well as on analysis of output sensitivity and uncertainty, wherever feasible. The same recommendations apply to the application of risk-based models (e.g., QMRA).

8. Based on stormwater control performance data from the International Stormwater BMP Database, consistent attainment of concentration-based primary contact recreational standards at end of pipe during all discharge conditions is unlikely for most passive stormwater controls (excluding disinfection). However, stormwater controls have many other water quality benefits and may still reduce FIB loads (especially through volume reductions), even if concentration-based limits are not consistently attainable. When selecting structural stormwater controls, both concentration and volume reduction benefits should be considered, focusing on practices with unit treatment processes that may be effective at reducing FIB loads.

9. Disinfection through chlorination, ultraviolet radiation, and ozonation are well documented to effectively reduce both FIB and pathogen concentrations in wastewater and drinking water. Chlorination and ozonation are typically impractical for urban stormwater applications due to needs for dechlorination (to prevent byproduct formation or discharge of toxic residuals) and risks of chemical storage. Ultraviolet radiation of dry weather MS4 discharges has been implemented in some locations, although long-term operation and maintenance costs are significant. Examples of disinfection of urban low-flows are typically limited to MS4 discharges to receiving waters where recreational exposure (i.e., potential public health impact) and economic impacts of beach closures are significant. Generally, disinfection is considered an option when source controls and stormwater controls have not resulted in attainment of FIB standards and elevated human health risks are present. In some cases, disinfection has been effective at point of treatment, but FIB regrowth has been observed shortly downstream, thereby potentially reducing its benefits (at least in terms of compliance with FIB limits).

10. Although the primary focus of this report is not CSOs, urban stormwater controls (e.g., green infrastructure controls that emphasize infiltration) that provide volume reduction can play a significant role in reducing the frequency and magnitude of CSO events and are often a component of long-term control plans (LTCPs). Additionally, principles of integrated planning of stormwater and sanitary municipal programs may be transferable to MS4 permits. Regulatory flexibilities that have been approved under LTCPs for CSOs may be helpful in formulating practical regulatory solutions to receiving water impairments once reasonable steps have been taken to reduce controllable sources of FIB. For example, some LTCPs have allowed use-attainability analysis to modify the recreational designated use (classification) of a waterbody receiving wet weather discharges from CSOs during wet weather conditions. Even in the absence of LTCPs,

some regulations allow high-flow suspension of recreational uses, which is conceptually similar to the use of a sizing criterion for an end-of-pipe retention or treatment system.

11. Given the issues and constraints described in this report, additional policy-level dialogue is needed to determine the most effective approach for developing and implementing urban FIB TMDLs and to determine TMDL "endpoints" that may differ from 100% compliance with RWQC, while still protecting public health. Once human sources of FIB are addressed, site-specific criteria, such as based on QMRA, are one alternative, particularly for large metropolitan areas with high exposure or high value recreational use waters; however, the cost of conducting these studies at multiple smaller waterbodies is beyond the reach of many smaller municipalities across the country. An alternative, cost-effective compliance approach that is protective of public health and that also recognizes economic constraints of local governments and practical limitations of technology and/or controllability of FIB sources is needed.

Based on the issues explored in this report, the following list of applied research and policy needs is suggested:

1. Continued FIB performance monitoring studies for urban stormwater structural controls. Such monitoring should be conducted in the field, not the laboratory, so that real-world variables are incorporated. Where possible, composite sampling should be used. FIB parameters should include enterococcus and *E. coli* consistent with the RWQC, with sufficient dilutions to allow for a wide analytical range of detection so that censored results (such as "too numerous to count") do not inhibit data interpretation. In particular, FIB performance datasets are needed for the following less studied, but potentially high performing stormwater control types: bioretention (with or without underdrains), media filters with bacteria-targeting media (e.g., antimicrobial media), subsurface flow wetlands, and infiltration-based systems (with "effluent" being subsurface samples). Data should be collected so that they are compatible with reporting protocols from the International Stormwater BMP Database. When budget allows, pathogen influent/effluent data should also be collected to allow for a better understanding of the treatability of these parameters.

2. Rigorous FIB performance monitoring studies for urban stormwater non-structural controls, such as source controls, catchbasin and storm drain cleaning, and other education, outreach, inspection, and enforcement based programs. Proper study design should incorporate the use of "control" watersheds. Such studies should be consolidated into a publically-accessible centralized performance database, such as the International Stormwater BMP Database.

3. Land use monitoring studies for pathogens, to potentially allow watershed models to expand beyond prediction of FIB concentrations to prediction of receiving water recreational illness risks. More robust land use monitoring datasets for enterococcus and *E. coli* are also needed for many regions of the country, with an emphasis on composite sampling techniques.

4. Epidemiological studies to assess human health risks for activities common to inland flowing waters, such as wading and waterplay by children in shallow water. Such studies could potentially provide the basis for new site-specific criteria for levels of exposure associated with such uses. This is an important question related to Use Attainability Analyses because many states assign primary contact standards to shallow urban streams where there is potential access by children, but there may actually be little actual use of the stream for recreation. In addition, wet weather epidemiological studies and QMRAs are needed for urban runoff receiving waters.

5. Regional "reference" watershed studies, such as those implemented in Southern California, to better characterize the range of naturally occurring FIB during wet and dry weather conditions.

6. Development of additional molecular methods to identify fecal wastes from common urban wildlife sources not well studied such as raccoons, along with QMRA capability to model risks from such fecal sources.

7. Additional research to refine understanding of source deposition, survivability, washoff, transport, and fate issues for FIB to support improved model algorithms.

8. A national policy-level dialogue regarding regulatory options that are protective of human health, while recognizing practical economic constraints facing local governments. The central questions that elected officials, regulators, non-governmental organizations, and regulated parties must somehow reach agreement on are:

    a. What level of control for FIB is practical and attainable, and reflects "acceptable" levels of public health protection based on actual pathogenicity and exposure (i.e., duration and frequency of exposure, and associated rates of ingestion) under real-world recreational use scenarios?

    b. How can measurable water quality compliance metrics (e.g., for TMDLs, MS4 permits) be expressed so that practical constraints are recognized, while still promoting meaningful water quality improvement?

11 REFERENCES

Aislabie, J., Smith, J. J., Fraser, R., and McLeod, M. (2001). “Leaching of bacterial indicators of faecal contamination through four New Zealand soils.” *Australian Journal of Soil Research*, 39(6), 1397-1406.

Alderisio, K.A., DeLuca, N. (1999). “Seasonal enumeration of fecal coliform bacteria from the feces of ring-billed gulls (*Larus delawarensis*) and Canada Geese (*Branta canadensis*).” *Applied and Environmental Microbiology*. 65(12), 5628-5630.

American Public Health Association (APHA). (2012). “Standard Methods for the Examination of Water and Wastewater, 22nd Edition.” L. S. Clesceri, A. E. Greenberg, A. D. Eaton, eds. Published in conjunction with the American Water Works Association and the Water Environment Federation, New York.

American Society for Testing and Materials (ASTM) (2013). ASTM F2550 - 13 Standard Practice for Locating Leaks in Sewer Pipes by Measuring the Variation of Electric Current Flow Through the Pipe Wall. Accessible at: http://www.astm.org/Standards/F2550.htm

American Society of Civil Engineers (ASCE). (2013). “Report Card for America’s Infrastructure.” http://www.infrastructurereportcard.org/a/#p/wastewater/overview

American Water Works Association Research Foundation and U.S. Environmental Protection Agency (AWWARF and U.S. EPA). (2006). *Development of Event-Based Pathogen Monitoring Strategies for Watersheds*. IWA Publishers.

Anderson, M.A. (2005). *Aliso Beach Clean Beaches Initiative, J01P28 Interim Water Quality Improvement Package Plant Best Management Practice*. County of Orange Resources and Development Management Department, Watershed and Coastal Resources, January.

Armand Ruby Consulting (ARC). (2011). “Source Prioritization Process for Bacteria.” Draft. December 12, 2011.

Armand Ruby Consulting (ARC). (2012). “Summary of Literature Review, Bacteria Source Identification.” Technical Memorandum. Prepared by: Armand Ruby Consulting in Association with AMEC. March 12, 2012.

Arnold, B.F., Schiff, K.C., Griffith, J.F., Gruber, J.S., Yau, V., Wright, C.C., Wade, T.J., Burns, S., Hayes, J.M., McGee, C., Gold, M., Cao, Y., Weisberg, S.B., Colford, J.M. Jr. (2013). Swimmer illness associated with marine water exposure and water quality indicators: impact of widely used assumptions. *Epidemiology* 24(6):845-53.

Arnon, S., Gray, K. A., and Packman, A. I. (2007). “Biophysicochemical process coupling controls nitrate use by benthic biofilms.” *Limnology and Oceanography*, 52(4), 1665-1671.

Ashbolt, N. J., Grabow, W.O. and Snozzi, M. (2001). “Indicators of microbial water quality. In Water Quality: Guidelines, Standards and Health.” *Risk Assessment and Management for Water-Related Infectious Disease*, L. Fewtrell and J.Bartram, eds., 289–315. IWA Publishing, London, UK.

Ashbolt, N. J., Schoen, M. E., Soller, J. A., and Roser, D. J. (2010). ”Predicting pathogen risks to aid beach management: the real value of quantitative microbial risk assessment (QMRA).” *Wat. Res.,* 44(16), 4692-703.

Atherholt, T., LeChevallier, M., and Rosen, J. (1998). “Effect of rainfall on Crypto and Giardia.” *Journal of American Water Works Association*, 90(9), 66-80.

Bagshaw, C.S. (2002). “Factors influencing direct deposition of cattle fecal material in riparian zones.” MAF Technical Paper No: 2002/19. University of Auckland, Department of Psychology. Wellington, New Zealand.

Bai, S., and Lung, W. S. (2005). “Modeling sediment impact on the transport of fecal bacteria.” *Water Research*, 39(20), 5232-5240.

Banning, N., Toze, S., and Mee, B. J. (2002). “Escherichia coli survival in groundwater and effluent measured using a combination of propidium iodide and the green fluorescent protein.” *Journal of Applied Microbiology*, 93(1), 69-76.

Banning, N., Toze, S., and Mee, B. J. (2003). “Persistence of biofilm-associated Escherichia coli and Pseudomonas aeruginosa in groundwater and treated effluent in a laboratory model system.” *Microbiology-Sgm*, 149, 47-55.

Barfield, B. J., Harp, S.L., Hayes, J.C., Chalavadi, M., Yeri, S., Brown, G. A., and Stevens, E. (2010). “Filtration and Sorption BMP Modeling for Water Quantity and Quality.” *2010 21st Century International Water Quality Conference, American Society of Biological and Agricultural Engineering*, Earth University, Costa Rica.

Barranguet, C., Veuger, B., Van Beusekom, S. A. M., Marvan, P., Sinke, J. J., and Admiraal, W. (2005). “Divergent composition of algal-bacterial biofilms developing under various external factors.” *European Journal of Phycology*, 40(1), 1-8.

Barrett, M. E. (2003). “Performance, cost, and maintenance requirements of Austin sand filters.” *J. Water Resour. Plann. Manage*., 129 (3): 234–242.

Baxter-Potter, W.K. and M.W. Gilliland. (1988). “Bacterial pollution in runoff from agricultural lands.” *Journal of Environmental Quality,* 17, 27-34.

Benham, B.L., Baffaut, C., Zeckoski, R.W., Manikn, K.R., Pachepsky, Y.A., Sadeghi, A.M., Brannan, K.M., Soupir, M.L., Habersack, M.J. (2006). “Modeling Bacteria Fate and Transport in Watersheds to Support TMDLS.” *Transactions of the ASABE,* 49(4), 987−1002.

Benton, C., Khan, F., Monaghan, P., and Richards, W.N. (1983). "The contamination of a major water supply by gulls (Larus sp.)." *Water Research*., 17(7), 789-798.

Berg, G. (ed.). (1983). *Viral pollution of the environment*. CRC Press, Boca Raton, Fla.

Berg, J.M., Hedges, S.C. and Jakubowski, S.D. (2009). "Evaluating Irrigation Effectiveness – Effects of runoff-reducing weather based irrigation (Smart Timers)." *Water Efficiency*. July-August.

Berthouex, P.M., and Brown, L.C. (1994). *Statistics for Environmental Engineers*. Lewis Publishers, Boca Raton, FL.

Blaustein, R.A., Pachepsky,Y., Hill, R.L., Shelton, D.R., Whelan, D. (2013). "Escherichia coli survival in waters: Temperature dependence." *Water Research*, 47, 569-578.

Boehm, A. B., Fuhrman, J. A., Mrse, R. D. & Grant, S. B. (2003). "A tiered approach for identification of a human fecal pollution source at a recreational beach: case study at Avalon Bay, Catalina Island, California, USA." *Environ. Sci. Technol.*, 37(4), 673–680.

Boehm, A.B., Ashbolt, N., Colford, J., Dunbar, L., Fleming, L., Gold, M., Hansel, J., Hunter, P.R., Ichida, A., McGee, C., Soller, J.A., Weisberg, S.B. (2009). "A Sea Change Ahead for Recreational Water Quality Criteria." *Journal of Water and Health*, 7(1), 9-20.

Bolster, C.H., Bromley, J.M., and Jones, S.H. (2005). "Recovery of chlorine-exposed Escherichia coli in estuarine microcosms." *Environ. Sci. Technol*., 39, 3083-3089.

Borst, M., and Selvakumar, A. (2003). "Particle Associated Microorganisms in Stormwater Runoff." EPA/600/J-03/262. http://www.epa.gov/ORD/NRMRL/pubs/600j03262/600j03262.pdf.

Bowen, J.L., and Valiela, I. (2004). "Nitrogen Loads to Estuaries: Using Loading Models to Assess the Effectiveness of Management Options to Restore Estuarine Water Quality." *Estuarine Research Federation*, 27, 482-500.

Boyer, D.G. and E. Kuczynska. (2003). "Storm and seasonal distributions of fecal coliforms and cryptosporidium in a spring." *Journal of the American Water Resources Association*, 39(6), 1449-1456.

Bradford, S.A., Morales, V.L., Zhang, W., Harvey, R.W., Packman, A.I., Mohanram, A. and Welty, C. (2013). "Transport and Fate of Microbial Pathogens in Agricultural Settings." *Critical Reviews in Environmental Science and Technology*, 43, 775-893.

Bradford, S.A., Simunek, J., and Walker, S.L. (2006). "Transport and straining of E-coli O157: H7 in saturated porous media." *Water Resources Research*, 42(12), 12.

Brown, G.A. (2010). Personal communication to B. J. Barfield by G. Brown, Professor, Biosystems and Agricultural Engineering, OK State University, Stillwater, OK

Brusseau, M. L., Gerstl, Z., Augustijn, D., and Rao, P. S. C. (1992). “Modeling solute transport influenced by multi-process nonequilibrium and transformation reactions.” *Water Resources Research*, 28, 175-182.

Burton, G.A. and Pitt, R. (2001 and 2002). “Stormwater effects handbook: A toolbox for watershed managers, scientists and engineers.” U.S. EPA Report (2001) and CRC Press, Inc., Boca Raton, FL (2002). 911 pages.

Burton, G.A., Gunnison, D., and Lanza, G.R. (1987). “Survival of Pathogenic Bacteria in Various Fresh-Water Sediments.” *Applied and Environmental Microbiology*, 53(4), 633-638.

Buswell, C.M., Herlihy, Y.M., Lawrence, L.M., McGuiggan, J.T., Marsh, P.D., Keevil, C.W., and Leach, S.A. (1998). “Extended survival and persistence of Campylobacter spp. in water and aquatic biofilms and their detection by immunofluorescent-antibody and -rRNA staining.” *Appl. Environ. Microbiol*., 64(2), 733-741.

Byappanahalli, M.N. and Fujioka, R.S. (1998). “Evidence that tropical soil environment can support the growth of Escherichia coli.” *Water Science and Technology*, 38, 171-174.

Byappanahalli, M.N. and Fujioka, R.S. (2004). “Indigenous soil bacteria and low moisture may limit but allow faecal bacteria to multiply and become a minor population in tropical soils.” *Water Sci. Technol.*, 50, 27-32.

Byappanahalli, M.N., Fowler, M., Shively, D., and Whitman, R. (2003a). “Ubiquity and persistence of Escherichia coli in a Midwestern coastal stream”. *Appl. Environ. Microbiol.*, 69, 4549-4555.

Byappanahalli, M.N., Nevers, M.B., Korajkic, A., Staley, Z.R., and Harwood, V.J. (2012). “Enterococci in the Environment Microbiol.” *Mol. Biol. Rev.*, 76, 4685-706.

Byappanahalli, M.N., Richard, L.W., Shivelya, D.A., Ferguson, J., Ishii, S., and Sadowsky, M. J. (2007). “Population structure of cladophora-borne *Escherichia coli* in nearshore water of lake Michigan.” *Water Research*, 41, 3649–3654.

Byappanahalli, M.N., Shively, D.A., Nevers, M.B., Sadowsky, M.J., and Whitman, R.L. (2003b). “Growth and survival of Escherichia coli and enterococci populations in the macro-alga Cladophora (Chlorophyta).” *FEMS Microbiology Ecology*, 46, 203-211.

Byappanahalli, M.N., Whitman, R.L., Shively, D.A., Sadowsky, M.J., and Ishii, S. (2006). “Population structure, persistence, and seasonality of autochthonous Escherichia coli in temperate, coastal forest soil from a Great Lakes watershed.” *Environ. Microbiol.*, 8, 504-513.

Cabelli, V.J. (1983). “Health Effects Criteria for Marine Recreational Waters.” U.S. Environmental Protection Agency, Health Effects Research Laboratory, Research Triangle Park, NC.

Cai, Y., Pitt, R.E., Togawa, N., McGee, K., Osei, K. and R. Andoh. (2014). "Full Scale Up-Flo Filter Field Verification Tests." *CHI Journal of Water Management Modeling* 2014;C365.

Calderon, R.L., Mood, E.W. and Dufour, A.P. (1991). "Health effects of swimmers and nonpoint sources of contaminated water." *Int. J. Environ. Health Res.*, 1(1), 21–31.

California Regional Water Quality Control Board, Los Angeles Region. (2010). *Los Angeles River Watershed Bacteria Total Maximum Daily Load, Revised Staff Report*.

California State Water Resources Control Board (CSWRCB). (2006). "Stormwater Panel Recommendations to the California State Water Resources Control Board: The Feasibility of Numeric Effluent Limits Applicable to Discharges of Stormwater Associated with Municipal, Industrial and Construction Activities." June 19. (http://www.cacoastkeeper.org/stormwater-pollution.php)

Carrington, E.G., Miller, D.G. (1993). "The occurrence and origins of Cryptosporidium oocysts in source waters." *Water Supply*, 11, 91.

Castiglione, P., Mohanty, B.P., Shouse, P.J., Simunek, J., van Genuchten, M. T., and Santini, A. (2003). "Lateral water diffusion in an artificial macroporous system: modeling and experimental evidence". *Vadose Zone J.*, 2, 212-231.

Center for Watershed Protection, and Pitt, R. (2004). "Illicit Discharge Detection and Elimination: A Guidance Manual for Program Development and Technical Assessments." Center for Watershed Protection, Ellicott City, MD, and University of Alabama, Birmingham, AL. U.S. Environmental Protection Agency, Office of Water and Wastewater. EPA Cooperative Agreement X-82907801-0. Washington, D.C., 357 pgs. Oct. 2004. http://www.cwp.org/online-watershed-library/cat_view/64-manuals-and-plans/79-illicit-discharge-detection-and-elimination

Center for Watershed Protection. (2001). Watershed Treatment Model. Accessible at: http://www.cwp.org/online-watershed-library/cat_view/65-tools/91-watershed-treatment-model

Center for Watershed Protection. (2008). *Monitoring to Demonstrate Environmental Results: Guidance to Develop Local Stormwater Monitoring Studies Using Six Example Designs*. www.cwp.org

Chandrasena, G.I., Filip S., Zhang K., Osborne C.A., Deletic A., and McCarthy, D.T. (2012). "Pathogen and indicator microorganism removal in field scale stormwater biofilters." *7th international WSUD conference*, Melbourne, Australia, February 21-23, 2012.

Characklis, G., and Camper, A. (2009). "Microbial Partitioning to Settleable Particles in Stormwater." Seminar hosted by Urban Wet-Weather Flow Research Program, National Risk Management Research Lab, EPA, January 26.

Characklis, G.W., Dilts M.J., Simmons, O.D., Likirdopulos, C.A., Krometis, L.H., and Sobsey, M.D. (2005). "Microbial partitioning to settleable particles in Stormwater." *Water Research*, 39, 1773–1782.

City of Austin. (2008). *Scoop the Poop Education Campaign, Summary Report*.

City of Beverly Hills, CalTrans, City of Culver City, City of Los Angeles, City of Inglewood, City of Santa Monica, and City of West Hollywood Beverly Hills. (2009). *Total Maximum Daily Load for Bacterial Indicator Densities in Ballona Creek, Ballona Estuary, and Sepulveda Channel Implementation Plan*.

City of Boulder (2013). City of Boulder Water Resources Advisory Board, Information Item: Boulder Creek TMDL and Marine Street Basin Pilot *E. coli* Reduction Project Update, June 17. City of Boulder, CO.

City of Encinitas. (2006). "Moonlight Beach Urban Runoff Treatment Facility Final Report." City of Encinitas, California Clean Water Program. February. http://www.ci.encinitas.ca.us/NR/rdonlyres/5612D387-D6D9-48A7-AF7F-A9D039C78547/0/Moonlight_Beach_Urban_Runoff_030806.pdf

City of San Diego. (2011). *Phase II Pet Waste Bag Dispenser Station Design and Implementation.*

City of Santa Barbara, Creeks Division, in Partnership with University of California, Santa Barbara. (2012a). *Tools for Tracking Human Fecal Pollution in Urban Storm Drains, Creeks, and Beaches.* September 1, 2012. Accessible at: http://www.santabarbaraca.gov/civicax/filebank/blobdload.aspx?BlobID=16722

City of Santa Barbara, Creeks Division. (2012b). "Source Tracking Protocol Development Project, Final Grant Report, Clean Beaches Grant Agreement 07-583-550-2." August 20, 2012. http://www.santabarbaraca.gov/civicax/filebank/blobdload.aspx?BlobID=16724

Clark, L. (2003). "A review of pathogens of agricultural and human health interest found in Canada Geese." In: *Proceedings of the 10th wildlife damage management conference*. K. A. Fagerstone and G. W. Witmer, eds., 6–9 April 2003; Hot Springs, AR. Fort Collins, CO: The Wildlife Damage Management Working Group of The Wildlife Society, 326-334.

Clark, L. (2004). "Avian Diseases: Carriage of Bacterial Pathogens by Canada Geese and Blackbirds." *USDA Animal and Plant Health Inspection Service (APHIS)*, Wildlife Services Center, Fort Collins, CO.

Clark, S.E. and R. Pitt. (1999). *Stormwater Treatment at Critical Areas, Vol. 3: Evaluation of Filtration Media for Stormwater Treatment*. U.S. Environmental Protection Agency, Water Supply and Water Resources Division, National Risk Management Research Laboratory. Cincinnati, Ohio. 442 pgs.

Clark, S.E., Pitt, R., and Field, R. (1994). "Stormwater treatment: inlet devices and filtration." Proc. Eng. Foundation Conf. Stormwater NPDES Related Monitoring Needs. *American Society of Civil Engineers*, Reston, VA. 641-650.

Clark, S.E. and Pitt, R. (2012). "Targeting treatment technologies to address specific stormwater pollutants and numeric discharge limits." *Water Research*, 46, 6715-6730.

Clark, S.E., Baker, K.H., Treese, D.P., Mikula, J.B., Siu, C.Y.S., Burkhardt, C.S., with contributions by Lalor, M.M. (2010). "Sustainable Stormwater Management: Infiltration vs. Surface Treatment Strategies." *Water Environment Research Foundation.* Project Number 04-SW-3. ISBN: 9781843392811. 460 pages.

Clary, J., Jones, J. and Urbonas, B. (2009). "Challenges in Attaining Recreational Stream Standards for Bacteria: Setting Realistic Expectations for Management Policies and BMPs." *Proc. of American Society of Civil Engineers Environmental and Water Resources Institute World Environmental and Water Resources Congress*, Kansas City, Missouri, May 17–21.

Cleaner Rivers through Effective Stakeholder-led TMDLs (CREST). (2008). *Los Angeles River Bacteria Source Identification Study: Final Report.* November 2007.

Cleland, B.R. (2003). "TMDL Development from the Bottom Up – Part III: Duration Curves and Wet-weather Assessments." *National TMDL Science and Policy 2003 -- WEF Specialty Conference*. Chicago, IL. November.

Cleland, B.R. (2007). "TMDL Development from the Bottom Up – Part IV: Connecting to Storm Water Management Programs." *National TMDL Science and Policy 2007 – WEF Specialty Conference.* Bellevue, WA. June.

Coghlan, A. (1996). "Slime City." *New Scientist*, 151(2045).

Colford, J.M., Schiff, K.C., Griffith, J.F., Yau, V., Arnold, B.F., Wright, C.C., Gruber, J.S., Wade, T.J., Burns, S., Hayes, J., McGee, C., Gold, M., Cao, Y., Noble, R.T., Haugland, R., Weisberg, S.B. (2012). "Using rapid indicators for Enterococcus to assess the risk of illness after exposure to urban runoff contaminated marine water." *Water Research*, 46, 2176-2186.

Colford, J.M., Wade, T.J., Schiff, K.C., Wright, C.C., Griffith, J.F., Sandhu, S.K., Burns, S., Sobsey, M., Lovelace, G., and Weisberg, S.B. (2007). "Water quality indicators and the risk of illness at beaches with nonpoint sources of fecal contamination." *Epidemiology*, 18(1), 27-35.

Colorado *E. coli* Work Group and Wright Water Engineers. (2009). *Synopsis of Recreational Water Quality Issues in Colorado: White Paper Summarizing Results of E. coli Work Group 2007-2009*.

Colorado Parks and Wildlife. (2014). "Canada Goose: Managing Human-Goose Conflicts in Urban and Suburban Areas." Accessible at: http://cpw.state.co.us/learn/Pages/LivingwithWildlifeGoose.aspx

Conover, M., and Kania, G. (1991). "Characteristics of feeding sites used by urban-suburban flocks of Canada geese in Connecticut." *Wildlife Society Bulletin*, 19, 36-38.

Cooley, M.B., Miller, W.G., and Mandrell, R E. (2003). "Colonization of Arabidopsis thaliana with Salmonella enterica and enterohemorrhagic Escherichia coli O157 : H7 and competition by Enterobacter asburiae." *Applied and Environmental Microbiology*, 69(8), 4915-4926.

Corsi, S.R., Borchardt, M.A., Spencer, S.K., Hughes, P.E., and Baldwin, A.K. (2014). Human and bovine viruses in the Milwaukee River watershed: Hydrologically relevant representation and relations with environmental variables. *Science of the Total Environment*. 490: 849–860.

Cortis, A., Harter, T., Hou, L.L., Atwill, E.R., Packman, A.I., and Green, P.G. (2006). "Transport of Cryptosporidium parvum in porous media: Long-term elution experiments and continuous time random walk filtration modeling." *Water Resources Research*, 42(12), 12.

Costerton, J.W., Lewandowski, Z., Caldwell, D.E., Korber, D.R., and Lappin-Scott, H.M. (1995). "Microbial Biofilms." *Annual Review of Microbiology*, 49, 711-745.

Costerton, W.J., Cheng, K.J., Geesey, G.G., Ladd, T.I., Nickel, J.C., Dasgupta, M., and Marrie, T.J. (1987). "Bacterial biofilms in nature and disease." *Annual Review of Microbiology*, 41, 435-464.

Cox, P., M. Griffith, M. Angles, D. Deere and C. Ferguson. (2005). "Concentrations of pathogens and indicators in animal feces in the Sydney watershed." *Applied and Environmental Microbiology*, 71, 5929-5934.

Crane, S. R., and Moore, J. (1986). "Modeling enteric bacterial die−off: A review." *Water, Air, and Soil Poll.*, 27(3−4), 411−439.

Cruz, J., Caceres, R., Cano, F., Flores, J., Bartlett, A., Torun, B. (1990). "Adenovirus types 40 and 41 and rotaviruses associated with diarrhea in children from Guatemala." *J. Clin. Microbiol.*, 28, 1780.

Csuros, M., and Csuros, C. (1999). *Microbiological Examination of Water and Wastewater*. Lewis Publishers, Boca Raton, FL.

Dai, X., and Boll, J. (2003). "Evaluation of attachment of Cryptosporidium parvum and Giardia lamblia to soil particles." *Journal of Environmental Quality*, 32, 296-304.

Daly, B., Betts, W.B., Brown, A.P., and O'Neill, J.G. (1998). "Bacterial loss from biofilms exposed to free chlorine." *Microbios*, 96(383), 7-21.

David, M.M. and Haggard, B.E. (2011). “Development of Regression-Based Models to Predict Fecal Bacteria Numbers at Select Sites within the Illinois River Watershed, Arkansas and Oklahoma, USA.” *Water, Air, and Soil Pollution,* 215(1), 525-547.

Davies, C.M., and Bavor, H.J. (2000). "The fate of stormwater-associated bacteria in constructed wetland and water pollution control pond systems." *Journal of Applied Microbiology*, 89(2), 349-360.

Davies, C.M., and Evison, L.M. (1991). “Sunlight and survival of enteric bacteria in natural waters.” *Journal of Applied Bacteriology,* 70, 265–279.

Davis, E.H. (1979). *Maximum Utilization of Water Resources in a Planned Community. Bacterial Characteristics of Stormwaters in Developing Rural Areas.* USEPA Rept. No. EPA/600/2-79-050F. NTIS No. PB 80-129091.

de Beer, D., Stoodley, P., and Lewandowski, A. (1993). “Oxygen distribution and mass transport in biofilms.” *Biotechnology and Bioengineering*, 43, 1131-1138.

de Beer, D., Stoodley, P., and Lewandowski, Z. (1994). “Liquid Flow in Heterogeneous Biofilms.” *Biotechnology and Bioengineering*, 44(5), 636-641.

Desmarais, T.R., Solo-Gabriele, H.M. and Palmer, C.J. (2002). “Influence of soil on fecal indicator organisms in a tidally influenced subtropical environment.” *Appl. Environ. Microbiol.,* 68(3), 1165–1172.

DeVuono-Powell, S. (2013). “Homeless Encampments in Contra Costa County” *Contra Costa County Flood Control and Water Conservation District*. Summer 2013.

Dewanti, R., and Wong, A.C.L. (1995). “Influence of Culture Conditions on Biofilm Formation by Escherichia-Coli O157-H7.” *International Journal of Food Microbiology*, 26(2), 147-164.

Dolbeer, R.A. (1994). “Blackbirds.” *The Handbook: Prevention and Control of Wildlife Damage*, Internet Center for Wildlife Damage Management, University of Nebraska - Lincoln DigitalCommons@University of Nebraska – Lincoln.

Donlan, R.M. (2002). “Biofilms: Microbial Life on Surfaces.” *Emerging Infectious Disease*, 8(9), 881-90.

Donlan, R.M., and Costerton, J.W. (2002). “Biofilms: Survival Mechanisms of Clinically Relevant Microorganisms.” *Clinical Microbiology Review*, 15, 167–193.

Donnison, A.M., Ross, C.M., and Davies-Colley, R.J. (2006). "Campylobacter as indicated by faecal microbial contamination in two streams draining dairy country." *Water 2006 Conference*, Auckland, New Zealand.

Dorner, S.M., Anderson, W.B., Slawson, R. M., Kouwen, N., and Huck, P.M. (2006). “Hydrologic modeling of pathogen fate and transport.” *Environmental Science & Technology*, 40(15), 4746-4753.

Dowd, S. E., and Pillai, S.D. (1997). “Survival and transport of selected bacterial pathogens and indicator viruses under sandy aquifer conditions.” *Journal of Environmental Science and Health Part a-Environmental Science and Engineering & Toxic and Hazardous Substance Control*, 32(8), 2245-2258.

Dufour, A.P. (1984). “Health effects criteria for fresh recreational waters.” *U.S. Environmental Protection Agency*.

Dykes, G.A., Sampathkumar, B., and Korber, D.R. (2003). “Planktonic or biofilm growth affects survival, hydrophobicity and protein expression patterns of a pathogenic Campylobacter jejuni strain.” *International Journal of Food Microbiology*, 89(1), 1-10.

EA Engineering, JEO Consulting Group and Wright Water Engineers (2012). *Antelope Creek Watershed Basin Management Plan*. Prepared for City of Lincoln, NE, and Lower Platte South Natural Resources District.

Easton, J. (2000). *The Development of Pathogen Fate and Transport Parameters for use in Assessing Health Risks Associated with Sewage Contamination*. Ph.D. dissertation, the Dept. of Civil and Environmental Engineering, the University of Alabama at Birmingham.

Easton, J.H., Lalor, M., Gauthier, J. J., Pitt, R., Newman, D.E., and Meyland, S. (1999). “Determination of Survival Rates for Selected Bacterial and Protozoan Pathogens from Wet Weather Discharges.” *WEFTEC 1999, Water Environmental Federation Annual Conference and Exposition*, New Orleans, October 1999

Edberg, S.C., and Edberg, M.M. (1988). “A defined substrate technology for the enumeration of microbial indicators of environmental pollution.” *The Yale J. of Biol. and Medicine*, 61, 389-399.

Ehrhart, B.J., Shannon, R.D., and Jarrett, A.R. (2002). “Effects of Construction Site Sedimentation Basins on Receiving Stream Ecosystems.” *Transactions of the ASAE*, 45(3), 675-680.

Ellis, A., Newman, D., and Pitt, R. (1998). “The use of peepers to measure nutrient and bacteria stratification in urban sediments.” Presented at the *34th Annual American Water Resources Association Conference: Applications of Water Use Information*. Point Clear, AL. November

Federal Highway Administration. (2001). *Guidance Manual for Monitoring Highway Runoff Water Quality*.

Fennessey, L.A. and Jarrett, A.R. (1996). "Influence of Principal Spillway Geometry and Permanent Pool Depth on Sediment Retention of Sedimentation Basins." *Transactions of the ASAE*, 40(1), 53-59.

Ferguson, C.M., Charles, K., Deere, D.A. (2009). "Quantification of microbial sources in drinking-water catchments." *Crit. Rev. Environ. Sci. Technol.*, 39, 1-40.

Ferguson, D. (2006). "Growth of *E. coli* and Enterococcus in Storm Drain Biofilm." Presentation at *2006 US EPA National Beaches Conference*.

Ferguson, D., Cao, Y., and Griffith, J. (2011). "Use of concrete coupons to assess attachment, growth and diversity of Enterococcus in storm drains." *American Society of Microbiology,* May 2011.

Field, R. (1990). "Combined Sewer Overflows: Control and Treatment." In: *Control and Treatment of Combined-Sewer Overflows,* Editor: P. E. Moffa. Van Nostrand Reinhold, New York, NY. 119-190.

Field, R. (1996). "Stormwater Pollution Abatement Technologies." In: *The Control and Treatment of Industrial and Municipal Stormwater,* Editor: P. E. Moffa. Van Nostrand Reinhold, New York, NY. 151-237.

Field, R.D., Wang, Y., Roswintiarti, O., and Guswanto, (2004). "A drought-based predictor of recent haze events in western Indonesia." *Atmos. Environ.*, 38, 1869-1878.

Fleming, R. and Fraser, H. (2001). "The Impact of Waterfowl on Water Quality." *Literature Review*. University of Guelph, Canada.

Flow Science. (2010). Comments prepared in response to the Public Hearing Notice Proposed amendment to the Water Quality Control Plan for the Los Angeles Region (Basin Plan) to incorporate Total Maximum Daily Load for Bacteria in the Los Angeles River FSI 037033. June 4 Letter from Dr. Susan Paulsen to Ms. Renee Purdy and Man Voong, California Regional Water Quality Control Board, Los Angeles Region.

Foppen, J.W., Mporokoso, A., and Schijven, J.F. (2005). "Determining straining of Escherichia coli from breakthrough curves." *J. Contam. Hydrol.,* 76, 191– 210.

Fox, G.A., Malone, R.W., Sabbagh, G.J., and Rojas, K. (2004). "Interrelationship of macropores and subsurface drainage for conservative tracer and pesticide transport." *J. Environ. Qual.,* 33, 2281-2289.

Fox, G.A., Matlock, E.M., Guzman, J.A., Sahoo, D. and Stunkel, K.B. (2011). "*Escherichia coli* load reduction from runoff by vegetative filter strips: A laboratory-scale study." *J. Environ. Qual*. 40(3):980-988.

Fox, G.A., Sabbagh, G.J., Malone, R.W., and Rojas, K. (2007). "Modeling parent and metabolite fate and transport in subsurface drained fields with directly connected macropores." *J. Am. Water Res. Assoc.*, 43(6), 1359–1372.

Francy, D.S., Gifford, A.M., Darner, R.A. (2003). *Escherichia coli beaches—distribution, sources, wastewater indicators, and predictive modeling*. US Geological Survey Water-Resources Investigations Report 02-4285. Columbus, Ohio.

French, L., and Parkhurst, J. (2009). “Managing Wildlife Damage: Canada Goose (*Branta canadensis*).” Virginia Cooperative Extension Publication 420-203.

Fujioka, R.S. and Byappanahalli, M.R. (2003). “Tropical Water Quality Indicator Workshop: Proceedings and Report.” Special Report SR-2004-01, *Water Resources Research Center*, University of Hawaii, 95 pp.

Fujioka, R.S., Sian-Denton, C., Borja, M., Castro, J., Morphen, K. (1999). “Soil: the environmental source of *Escherichia coli* and enterococci in Guam’s streams.” *J. Appl. Microbiol. Symp. Suppl.*, 85, 835.

Fujioka, R.S., Hashimoto, H.H., Siwak, E.B., and Young, R.H.F. (1981). “Effect of sunlight on survival of indicator bacteria in seawater.” *Appl. Environ. Microbiol*., 41, 690.

Gallagher, M. A., Karthikeyan, R., and Mukhtar, S. (2012). “Growth Kinetics of Wildlife E. coli Isolates in Soil and Water.” *Journal of Environmental Protection, 3,* 838-846.

Gallay, A., De Valk, H., Cournot, M., Ladeuil, B., Hemery, C., Castor, C., Bon, F., Megraud, F., Le Cann, P., and Desenclos, J. C. (2006). “A large multi-pathogen waterborne community outbreak linked to faecal contamination of a groundwater system, France, 2000.” *Clinical Microbiology and Infection*, 12(6), 561-570.

Gantzer, C., Gillerman, L., Kuznetsov, M., and Oron, G. (2001). “Adsorption and survival of faecal coliforms, somatic coliphages and F-specific RNA phages in soil irrigated with wastewater.” *Water Sci. Technol*., 43 (12), 117–124.

Garbrecht, K., Fox, G.A., Guzman, J.A., and Alexander, D. (2009). “E. coli transport through soil columns: Implications for bioretention cell removal efficiency.” *Transactions of the ASABE*, 52(2), 481-486.

Garcia-Armisen, T., and Servais, P. (2009). “Partitioning and Fate of E. coli in River Waters.” *Water Environmental Research*, 81(1), 21-28

Geldreich, E.E. and Kenner, B.A. (1969). “Concepts of fecal streptococci in stream pollution.” *Journal of the Water Pollution Control Federation,* 41(8), R336-R352.

Geldreich, E.E., Best, L.C., Kenner, B.A., and Van Donsel, D.J. (1968). “The bacteriological aspects of stormwater pollution.” *Journal of the Water Pollution Control Federation,* 40(11), 1861-1872.

Geldreich, E.E. (1976). "Fecal Coliform and Fecal Streptococcus Density Relationships in Waste Discharges and Receiving Waters." *Critical Reviews in Environmental Control*, 6(4), 349. Oct.

Geldreich, E.E., Nash, H.D., Spino, D.F., and Reasoner, D.J. (1980). "Bacterial Dynamics in a Water Supply Reservoir: A Case Study." *Journal AWWA*. Jan.

Geosyntec Consultants and Wright Water Engineers. (2009). *Urban Stormwater BMP Performance Monitoring Guidance.* Prepared for U.S. Environmental Protection Agency, Water Environment Research Foundation and the Federal Highway Administration. (http://www.bmpdatabase.org/MonitoringEval.htm)

Geosyntec Consultants and Wright Water Engineers. (2011). *International Stormwater Best Management Practices (BMP) Database Volume Reduction Technical Summary*. January. (accessible at www.bmpdatabase.org).

Geosyntec Consultants and Wright Water Engineers. (2012a). *International Stormwater Best Management Practices (BMP) Database Technical Summary: Manufactured Devices Performance Summary.* Prepared under Support from WERF, FHWA, EWRI/ASCE and EPA. July.

Geosyntec Consultants and Wright Water Engineers. (2012b). *International Stormwater Best Management Practices (BMP) Database Addendum 1 to Volume Reduction Technical Summary (January 2011), Expanded Analysis of Volume Reduction in Bioretention Facilities.* May. (accessible at www.bmpdatabase.org).

Geosyntec Consultants. (2009). *Laguna Watershed Study and Water Quality Improvement Feasibility Analysis Laguna Channel Santa Barbara, California*. Prepared with assistance from with assistance from UCSB Bren School of Environmental Science & Management; laboratory of Dr. Patricia Holden.

Geosyntec Consultants. (2010). *Dry Weather Source Characterization and Control Summary, Santa Monica Bay Beaches Bacteria TMDL Implementation*. Revised June, 2011.

Geosyntec Consultants. (2012). *San Diego River Watershed Comprehensive Load Reduction Plan, Attachment E: Non-Structural BMP Memo*. Prepared for the County of San Diego, the Cities of El Cajon, La Mesa, Santee, San Diego, and the California Department of Transportation.

Geosyntec Consultants. (2014). *Draft Task Order No. 24: Dry Weather Microbial Source Tracking Study Findings Report, San Diego River Watershed*. February 2014.

Gibson, D.L., White, A.P., Snyder, S.D., Martin, S., Heiss, C., Azadi, P., Surette, M., and Kay, W.W. (2006). “Salmonella produces an O-antigen capsule regulated by AgfD and important for environmental persistence.” *Journal of Bacteriology*, 188(22), 7722-7730.

Gilbert, R.O. (1987). *Statistical Methods for Environmental Pollution Monitoring*. New York, NY: Van Nostrand Reinhold, 1987.

Glass, R.I., Parashar, U.D. and Estes, M.K. (2009). “Norovirus Gastroenteritis.” *New England Journal of Medicine*, 361, 1776–1785, October.

Gore & Storrie Ltd./Proctor & Redfern Ltd. (1981). *Executive Summary Report on Rideau River Stormwater Management Study, Phase I.* Rideau River Stormwater Management Study, Ottawa, and the Ontario Ministry of the Environment, Kingston, Ontario.

Graczyk, T.K., Lucy, F.E., Tamang, L., Mashinski, Y., Broaders, M.A., Connolly, M., and Cheng, H.W.A. (2009). "Propagation of human enteropathogens in constructed horizontal wetlands used for tertiary wastewater treatment." *Applied and Environmental Microbiology*, 75(13), 4531-4538.

Granato, G.E., Zenone, C., and Cazenas, P.A (eds.) (2002). *National Highway Runoff Water-Quality Data and Methodology Synthesis, Volume I --Technical Issues for Monitoring Highway Runoff and Urban Stormwater.* http://ma.water.usgs.gov/fhwa/fhwaep.htm

Grant, S.B., Sanders, B.F., Boehm, A.B., Redman, J.A., Kim, J.H., Mrse, R.D., Chu, A.K., Gouldin, M., McGee, C.D., Gardiner, N.A., Jones, B.H., Svejkovsky, J., and Leipzig, G.V. (2001). "Generation of Enterococci bacteria in a coastal saltwater marsh and its impact on surf zone water quality." *Environmental Science and Technology*, 35, 2407-2416.

Griffith, J. F., Aumand, L. A., Lee, I. M., McGee, C. D., Othman, L. L., Ritter, K. J., Walker, K. O., Weisberg, S. B. (2006a). "Comparison and verification of bacterial water quality indicator measurement methods using ambient coastal water samples." *Environ. Monit. Assess.*, 116, 335–344.

Griffith, J. F., Schiff, K. C., and Lyon, G. S. (2006b). "Microbiological water quality at non-human impacted reference beaches in southern California during wet weather." Technical report 495. *Southern California Coastal Water Research Project*, Westminster, CA.

Griffith, J.F., Layton, B.A., Boehm, A.B., Holden, P.A., Jay, J.A., Hagedorn, C., McGee, C.D., Weisberg, S.B. (2013). *The California Microbial Source Identification Manual: A Tiered Approach to Identifying Fecal Pollution Sources to Beaches*. Southern California Coastal Water Resources Project, Technical Report 804. December. Accessible at: http://www.waterboards.ca.gov/board_info/agendas/2014/jan/012114_11_804_sipp_mst_manual.pdf.

Guber, A.K., Shelton, D.R., and Pachepsky, Y.A. (2005). "Effect of manure on *Escherichia coli* attachment to soil." *Journal of Environmental Quality*, 34(6), 2086-2090.

Guzman, J.A., Fox, G.A., and Penn, C. (2010). "Escherichia coli sorption in artificial and natural soils after swine manure application." *2010 Land Grant and Sea Grant National Water Conference*, Hilton Head, SC, 22 February 2010.

Habersack, M.J., Dillaha, T., and Hagedorn, C. (2011). "Common Snapping Turtles (Chelydraserpentina) as a Source of Fecal Indicator Bacteria in Freshwater Systems." *Journal of the American Water Resources Association (JAWRA)*, 47(6), 1255-1260.

Haenninen, M.L., Haajanen, H., Pummi, T., Wermundsen, K., Katila, M.L., Sarkkinen, H., Miettinen, I., and Rautelin, H. (2003). "Detection and typing of *Campylobacter jejuni* and

Campylobacter coli and analysis of indicator organisms in three waterborne outbreaks in Finland." *Appl. Environ. Microbiol*., 69(3), 1391-1396.

Hagedorn, C., Blanch, A.R., Harwood, V.J. (2011). *Microbial Source Tracking: Methods, Applications, and Case Studies.* Springer Media: NY, NY.

Hagedorn, C., Hansen, D. T., and Simonson, G. H. (1978). "Survival and movement of fecal indicator bacteria in soil under conditions of saturated flow." *Journal of Environmental Quality*, 7(1), 55-59.

Haile, R.W., Witte, J.S., Gold, M., Cressey, R., McGee, C., Millikan, R.C., Glasser, A., Harawa, N., Ervin, C., Harmon, P., Harper, J., Dermand, J., Alamillo, J., Barrett, K., Nides, M.,Wang, G. (1999). "The health effects of swimming in ocean water contaminated by storm drain runoff." *Epidemiology*, 10(4), 355-63.

Hall-Stoodley, L., Costerton, J.W., and Stoodley, P. (2004). "Bacterial biofilms: From the natural environment to infectious diseases." *Nature Reviews Microbiology*, 2(2), 95-108.

Harmel, R.D., Cooper, R.J., Slade, R.M., Haney, R.L., and Arnold, J.G. (2006). "Cumulative uncertainty in measured streamflow and water quality data for small watersheds." *Trans. ASABE*, 49(3), 689-701.

Harmel, R.D., King, K.W. and Slade, R.M. (2003). "Automated storm water sampling on small watersheds." *Applied Eng. In Agric.* 19(6): 667-674.

Harmel, R.D., Karthikeyan, R., Gentry, T., Srinivasan, R. (2010). "Effects of agricultural management, land use, and watershed scale on E. coli concentrations in runoff and streamflow." *Transactions of the ASABE*, 53(6), 1833-1841.

Harmel, R.D., Wagner, K.L., Martin, E., Gentry, T., Karthikeyan, R., Dozier, M., Coufal, C. (2013). "Impact of poultry litter application and land use on *E. coli* runoff from small agricultural watersheds." *Biological Engineering Transactions (ASABE)*, 6(1), 3-16.

Harter, T., Atwill, E.R., Hou, L.L., Karle, B.M., and Tate, K.W. (2008). "Developing risk models of *Cryptosporidium parvum* transport in soils from vegetated, tilted soil box experiments." *Journal of Environmental Quality*, 37, 245-258.

Harwood, V.J. and Stoeckel, D. M. (2011). "Performance Criteria. In Microbial Source Tracking: Methods, Applications and Case Studies." C.H. Hagedorn, A.R. Blanch and V.J. Harwood, eds., Springer, New York.

Harwood, V.J., Staley, C., Badgley, B.D., Borges, K., and Korajkic, A. (2014). "Microbial source tracking markers for detection of fecal contamination in environmental waters: relationships between pathogens and human health outcomes." *FEMS Microbiology Reviews*, 38(1): 1-14.

Harwood, V. J., Whitlock, J. and Withington, V. (2000). “Classification of antibiotic resistance patterns of indicator bacteria by discriminant analysis: use in predicting the source of fecal contamination in subtropical waters.” *Appl. Environ. Microbiol.*, 66(9), 3698–3704.

Hathaway, J.M. (2010). *An evaluation of indicator bacteria transport in stormwater runoff and removal in stormwater control measures*. Doctoral dissertation, North Carolina State University, Raleigh, NC.

Hathaway, J.M. and Hunt, W.F. (2008). “Removal of Pathogens in Stormwater, North Carolina State University Extension Urban Waterways Fact Sheet.”

Hathaway, J.M. and Hunt, W.F. (2011). “Evaluation of first flush for indicator bacteria and total suspended solids in urban stormwater runoff.” *Water, Air, and Soil Pollution*, 217, 135-147.

Hathaway, J.M., and Hunt, W.F. (2012).”Indicator Bacteria Performance of Stormwater Control Measures in Wilmington, North Carolina.” *Journal of Irrigation and Drainage Engineering*, 138(2), 185-197.

Hathaway, J.M., Hunt, W.F., and Jadlocki, S. (2009). “Indicator bacteria removal in storm-water best management practices in Charlotte, North Carolina.” *Journal of Environmental Engineering*, 135(12), 1275 – 1285.

Hathaway, J.M., Hunt, W.F., and Simmons, O.D. (2010). “Statistical evaluation of factors affecting indicator bacteria in urban storm-water runoff.” *Journal of Environmental Engineering*, 136(12), 1360-1368.

Hathaway, J.M., Hunt, W.F., Guest, R.M., and McCarthy, D.T. (2014). “Residual indicator bacteria in autosampler tubing: a field and laboratory assessment.” *Water Science & Technology*, 69(5), 1120 - 1126

Hayes, J.C., Barfield, B.J., Harp, S.L., Chalavadi, M., Stevens, E., Alexander, M.D., Bates, B.T. (2008). “Modeling Impacts of Post Development Water Quality BMP.” 2008 21st Century International Water Quality Conference, American Society of Biological and Agricultural Engineering, University of Concepcion University, Concepcion, Chile.

HDR (2013). Technical Memorandum Re: Raccoon Storm System Controls - University Hill. Memorandum to B. Linenfelser, City of Boulder, October 29.

Hellweger, F.L., Bucci, V., Litman, M. R., Gu, A.Z., and Onnis-Hayden, A. (2009). “Biphasic decay kinetics of fecal bacteria in surface water: not a density effect.” *ASCE Journal Environmental Engineering*, 135, 372-376.

Helsel, D.R. (2005). *Nondetects and data analysis*. New York: John Wiley & Sons, Inc.

Helsel, D.R. and Hirsch, R.M. (2002). “Statistical methods in water resources—hydrologic analysis and interpretation.” *U.S. Geological Survey Techniques of Water-Resources Investigations*, Book 4, Chap. A3, 510 p.

Holevinski, R.A., Malecki, R.A., Curtis, P.D. (2006). "Can Hunting of Translocated Nuisance Canada Geese Reduce Local Conflicts?" *Wildl. Soc. Bull.*, 34(3), 845-849.

Hunt, W.F., Smith, J.T., Jadlocki, S.J., Hathaway, J.M., and Eubanks, P. (2008). "Pollutant removal and peak flow mitigation by a bioretention cell in urban Charlotte, NC." *J. Environ. Eng*., 134 (5): 403–408.

Hussong, D., Damare, J.M., Limpert, R.J., Sladen, W.J.L, Weiner, R.M., Colwell, R.R. (1979). "Microbial impact of canada geese (Branta canadensis) and whistling swans (Cygnus columbianus columbianus) on aquatic ecosystems." *Applied and Environmental Microbiology*, 37(1), 14-20.

International Association on Water Pollution Research and Control (IAWPRC) Study Group on Health Related Water Microbiology. (1991). "Bacteriophages as model viruses in water quality control." *Water Res.*, 25(5), 529–545.

International Stormwater BMP Database website: www.bmpdatabase.org.

Irvine Ranch Water District (IRWD) and Orange County Metropolitan Water District (OCMWD). (2004). *Residential Runoff Reduction Study*. Irvine Ranch Water District and Municipal Water District of Orange County. July.

Irvine Ranch Water District (IRWD). (2005). Orange County Green Vision Presentation. September 15. Accessible at: www.lgc.org

Ishii, S. and Sadowsky, M.J. (2008). "*Escherichia coli* in the Environment: Implications for water quality and human health." *Microbes Environ*., 23, 101-108.

Ishii, S., Ksoll, W. B., Hicks, R. E., Sadowsky, M. J. (2006). "Presence and growth of naturalized *Escherichia coli* in temperate soils from Lake Superior watershed." *Appl. Environ. Microbiol.*, 72, 612-621.

Izbicki, J. (2011). *Distribution of Fecal Indicator Bacteria along the Malibu, California, Coastline*. U.S. Geological Survey Open-File Report 2011-1091.

Jamieson, R.C., Gordon, R., Joy, D., and Lee, H. (2004). "Assessing microbial pollution of rural surface waters: A review of current watershed-scale modeling approaches." *Agric. Water Mgmt.*, 70(1), 1-17.

Jamieson, R.C., Joy, D. M., Lee, H., Kostaschuk, R., and Gordon, R. (2005a). "Transport and deposition of sediment-associated *Escherichia coli* in natural streams." *Water Research*, 39(12), 2665-2675.

Jamieson, R.C., Joy, D.M., Lee, H., Kostaschuk, R., and Gordon, R.J. (2005b). "Resuspension of sediment-associated Escherichia coli in a natural stream." *Journal of Environmental Quality*, 34(2), 581-589.

Jarrett, A.R. (2001). “Designing Sedimentation Basins for Better Sediment Capture.” Soil Erosion Research for the 21st Century, *Proceedings of the International Symposium ASAE*. 701P007:63-66.

Jeng, H.A.C., England, A.J., Bakeer, R.M., and Bradford, H.B. (2005). “Impact of urban stormwater runoff on estuarine environmental quality.” *Estuarine, Coastal and Shelf Science,* 63, 513–526.

Jensen, P., Rifai, H., Battenfield, T. (2002). “Lessons learned from Texas bayous identifying bacteria sources.” *Public Works*, 133(5), 308-310.

Jiang, S.C. and Chu, W. (2005). “PCR detection of pathogenic viruses in southern California urban rivers.” *J. Appl. Microbiol*., 97, 17-28.

Jiang, S.C., Noble, R., Chu, W. (2001). “Human adenoviruses and coliphages in urban runoff - impacted coastal waters of Southern California.” *Appl. Environ. Microbiol*., 67(1), 179.

Jin, G., Englande, A.J., Bradford, H., Jeng, H.W. (2004). “Comparison of *E. coli,* enterococci, and fecal coliform as indicators for brackish water quality assessment.” *Water Environ. Res*., 76(3), 245.

John, D.E., and Rose, J. B. (2005). “Review of factors affecting microbial survival in groundwater.” *Environmental Science & Technology*, 39(19), 7345-7356.

Joseph, B., Otta, S.K., Karunasagar, I., and Karunasagar, I. (2001). “Biofilm formation by Salmonella spp. on food contact surfaces and their sensitivity to sanitizers.” *International Journal of Food Microbiology*, 64(3), 367-372.

Joshua, G.W.P., Guthrie-Irons, C., Karlyshev, A.V., and Wren, B.W. (2006). “Biofilm formation in *Campylobacter jejuni*.” *Microbiology-Sgm*, 152, 387-396.

Joyce, T.M., McGuigan, K.G., Elmore-Meegan, M. and Conroy, R.M. (1996). “Inactivation of fecal bacteria in drinking water by solar heating.” *Appl. Environ. Microbiol*. 62(2): 399-402.

Kadlec, R. and Wallace, S. (2009). *Treatment Wetlands*, Second Edition. CRC Press.

Kadlec, R.H. and Knight, R.L. (1996). *Treatment Wetlands*. Lewis Publishers, CRC Press, Inc. Boca Raton, Fla.

Kear, J. (1963). “The agricultural importance of wild goose droppings.” *The Waterfowl Trust, 14th annual Report*, 1961-1962, 72-77.

Keith, L.H., ed. (1996). *Principles of Environmental Sampling*. 2nd ed. American Chemical Society.

Kim, H.J., Phenrat, T., Tilton, R.D., and Lowry, G.V. (2012). “Effect of kaolinite, silica fines and pH on transport of polymer-modified zero valent iron nanoparticles in heterogeneous porous media.” *Journal of Colloid and Interface Science.* 370, 1-10.

Kirschner, A., Zechmeister, T., Kavka, G., Beiwl, C., Herzig, A., Mach, R., Farnleitner, A. (2004). "Integral Strategy for Evaluation of Fecal Indicator Performance in Bird-Influenced Saline Inland Waters." *Applied and Environmental Microbiology*, 70(12), 7396-7403.

Kitsap Public Health District. (2013). Kitsap Public Health and University of Washington Team Up on "Emerging Contaminants" Study, Press Release. May 28. (Additional study description provided by Shawn Ultican, Kistap Public Health District.)

Kladivko, E.J., Grochulska, J., Turco, R.F., Van Scoyoc, G.E., and Eigel, D.E. (1999). "Pesticide and nitrate transport into subsurface tile drains of different spacings." *J. Environ. Qual.*, 28, 997-1004.

Kolb, R. and Roberts, G. (2009). "Further Understanding of the Bacterial Dynamic, Lessons from Microbial Source Tracking." Powerpoint presentation prepared by Weston Solutions for the "Think Blue San Diego" Program. San Diego, CA.

Korajikic, A., Wanjugi, P., and Harwood, V. (2013). "Indigenous Microbiota and Habitat Influence Escherichia coli Survival More than Sunlight in Simulated Aquatic Environments." *Appl. Environ. Microbiol.*, 79(17), 5329–5337.

Korber, D.R., Choi, A., Wolfaardt, G.M., Ingham, S.C., and Caldwell, D.E. (1997). "Substratum topography influences susceptibility of *Salmonella enteritidis* biofilms to trisodium phosphate." *Appl. Environ. Microbiol*., 63(9), 3352-3358.

Kramer, M.H., Sorhage, F.E., Goldstein, S.T., Dalley, E., Wahlquist, S.P., Herwaldt, B.L. (1998). "First reported outbreak in the United States of cryptosporidiosis associated with a recreational lake." *Clin. Infec. Dis*., 26, 27-33.

Krometis, L.A., Characklis, G.W., Simmons III, O.D., Dilts, M.J., Likirdopulos, C.A., and Sobsey, M.D. (2007). "Intra-storm variability in microbial partitioning and microbial loading rates." *Water Research*, 41(2): 506 - 516.

Krometis, L.A., Drummey, P., Characklis, G., and Sobsey, M. (2009). "Impact of Microbial Partitioning on Wet Retention Pond Effectiveness." *J. Environ. Eng*., 135(9), 758–767.

Krumgalz, B.S., Fainshtein, G., Cohen, A. (1992). "Grain size effect on anthropogenic trace metal and organic matter distribution in marine sediments." *Sci. Total Environ*., 116, 15-30.

Kuusi, M., Klemets, P., Miettinen, I., Laaksonen, I., Sarkkinen, H., Hanninen, M. L., Rautelin, H., Kela, E., and Nuorti, J. P. (2004). "An outbreak of gastroenteritis from a non-chlorinated community water supply." *Journal of Epidemiology and Community Health*, 58(4), 273-277.

Kuusi, M., Nuorti, J.P., Hanninen, M.L., Koskela, M., Jussila, V., Kela, E., Miettinen, I., and Ruutu, P. (2005). "A large outbreak of campylobacteriosis associated with a municipal water supply in Finland." *Epidemiology and Infection*, 133(4), 593-601.

LaLiberte, P. and Grimes, D.J. (1981). “Survival of *Escherichia coli* in Lake Bottom Sediment.” *Applied and Environmental Microbiology*, 43(3), 623-628.

Lalor, M. (1994). “Assessment of non-stormwater discharges to storm drainage systems in residential and commercial land use areas.” Ph.D. thesis. Department of Environmental and Water Resources Engineering. Vanderbilt University. Nashville, TN.

Latasa, C., Roux, A., Toledo-Arana, A., Ghigo, J. M., Gamazo, C., Penades, J. R., and Lasa, I. (2005). “BapA, a large secreted protein required for biofilm formation and host colonization of *Salmonella enterica serovar Enteritidis*.” *Molecular Microbiology*, 58(5), 1322-1339.

LeChevallier, M.W., Norton, W., and Lee, R. (1991). “Occurrence of Giardia and *Cryptosporidium spp.* in surface-water supplies.” *Applied and Environmental Microbiology*, 57(9), 2610-2616.

Lehman, E.L. and D'Abrera, H.J.M. (1975). *Nonparametrics: Statistical Methods Based on Ranks*. Holden-Day and McGraw-Hill.

Leisenring, M., Barrett, M., Pomeroy, C., Poresky, A., Roesner, L. Rowney, A.C., and Strecker, E. (2013). *Linking BMP systems performance to receiving water protection, BMP performance algorithms*. WERF: Alexandria, VA.

Levesque, B., Brousseau, P., Simard, P., Dewailly, E., Meisels, M., Ramsay, D., and Joly, J. (1993). “Impact of the ring-billed gull (*Larus delawarensis*) on the microbiological quality of recreational water.” *Applied and Environmental Microbiology*, 59(4), 1228-1230.

Line, D.E., White, N.M., Kirby-Smith, W.W., and Potts, J.D. (2008). “Fecal coliform export from four coastal North Carolina areas.” *Journal of the American Water Resources Association*, 44(3), 606-617.

Ling, T.Y., Achberger, E.C., Drapcho, C. M., and Bengtson, R. L. (2002). “Quantifying adsorption of indicator bacteria in a soil-water system.” *Trans. ASAE*, 45(3), 669-674.

Lyautey, E., Lu, Z., Lapen, D. R., Wilkes, G., Scott, A., Berkers, T., and Topp, E. (2010). “Distribution and diversity of *Escherichia coli* populations in the South Nation River drainage basin, eastern Ontario, Canada.” *Applied and Environmental Microbiology*, 76(5), 1486-1496.

Mackay, W., Gribbon, L., Barer, M., and Reid, D. (1998). “Biofilms in drinking water systems- a possible reservoir for *Helicobacter pylori*.” *Water Science and Technology*, 38(12), 181-185.

Maestre, A. (2005). “Stormwater Characteristics as Described in the National Stormwater Quality Database.” Ph.D. Dissertation. The University of Alabama, Tuscaloosa.

Maestre, A. and Pitt, R. (2005). The National Stormwater Quality Database, Version 1.1, A Compilation and Analysis of NPDES Stormwater Monitoring Information. U.S. EPA, Office of Water, Washington, D.C. (final draft report) August 2005.

Maier, R.M., Pepper, I.L. and Gerba, C.P. (1999). *Environmental Microbiology*. Academic Press, New York, USA.

Mankin, K.R., Wang, L., Hutchinson, S.L., and Marchin, G.L. (2007). "*Escherichia coli* sorption to sand and silt loam soil." *Trans. ASABE*, 50(4), 1159-1165.

Manny, B.A., Wetzel, R.G., and Johnon, W.C. (1975). "Annual contribution of carbon, nitrogen and phosphorus by migrant Canada geese to hardwater lake." *Ver. Internat. Verein. Limnol*., 19, 949-951.

Maryland Department of the Environment (MDE). (2007). *Rock Creek TMDL*.

Matson, E.A., Hornor, S.G., and Buck, J.D. (1978). "Pollution indicators and other microorganisms in the river sediment." *Journal WPCF*. Jan.

Matthess, G., and Pekdeger, A., (1988). "Survival and transport of pathogenic bacteria and viruses in groundwater." In: *Groundwater Quality*. C. H. Ward, W. Giger, P. L. McCarty, eds., John Wiley, New York, 472– 482.

McBride, G.B., Salmond, C.E., Bandaranayake, D.R., Turner, S.J., Lewis, G.D. and Till, D.G. (1998). "Health effects of marine bathing in New Zealand." *Int. J. Environ. Health Res*., 8(2), 173–189.

McCarthy, D.T. (2009). "A traditional first flush assessment of *E. coli* in urban stormwater runoff." *Water Science and Technology*, 60(11), 2749–2757.

McCarthy, D.T., Deletic, A., Mitchell, V.G., and Diaper, C. (2011). "Development and testing of a model for micro-organism prediction in urban stormwater (MOPUS)." *Journal of Hydrology*, 409, 236-247.

McCarthy, D.T., Deletic, A., Mitchell, V.G., Fletcher, T.D., and Diaper, C. (2008). "Uncertainties in stormwater E. coli levels." *Water Research*, 42(6-7), 1812-1824.

McDougald, D., Rice, S., Weichart, D., and Kjelleberg, S. (1998). "Nonculturability: adaptation or debilitation?" *FEMS Microbiology Ecology*, 25, 1-9.

McGechan, M.B. and Vinten, A. J.A. (2003). "Simulation of transport through soil of *E. coli* derived from livestock slurry using the MACRO model." *Soil Use and Management*, 19, 321-330.

McGechan, M.B., and Vinten, A. J.A. (2004). "Simulating transport of *E. coli* derived from faeces of grazing livestock using the MACRO model." *Soil Use and Management*, 20(2), 195-202.

McGuigan, K.G., Joyce, T.M., Conroy, R.M., Gillespie, J.B., and Elmore-Meegan, M. (1998). “Solar disinfection of drinking water contained in transparent plastic bottles: Characterizing the bacterial inactivation process.” *Journal of Applied Microbiology*, 84(6), 1138-1148.

McLean, B.M., Baskaran, K., and Connor, M.A. (2000). “The use of algal-bacterial biofilms to enhance nitrification rates in lagoons: experience under laboratory and pilot-scale conditions.” *Water Science and Technology*, 42(10-11), 187-194.

McMahon, M.J., and Christy, A.D. (2000). “Root growth, calcite precipitation, and gas and water movement in fractures and macropores: A review with field observations.” *Ohio J. Sci.*, 100(3-4), 88-93.

McMurry, S.W., Coyne, M.S., and Perfect, E. (1998). “Fecal coliform transport through intact soil blocks amended with poultry manure.” *Journal of Environmental Quality*, 27(1), 86-92.

McQuaig, S., Griffith, J. and Harwood, V. (2012). Association of Fecal Indicator Bacteria with Human Viruses and Microbial Source Tracking Markers at Coastal Beaches Impacted by Nonpoint Source Pollution. *Applied and Environmental Microbiology*. 78 (18): 6423–6432.

Medema, G.J., Schets, F.M., Teunis, P.F.M., and Havelaar, A.H. (1998). “Sedimentation of free and attached *Cryptosporidium* oocysts and *Giardia* cysts in water.” *Applied and Environmental Microbiology*, 64(11), 4460-4466.

Mermillod-Blondin, F., François-Carcaillet, F., and Rosenberg, R. (2005). “Biodiversity of benthic invertebrates and organic matter processing in shallow marine sediments: an experimental study.” *Journal of Experimental Marine Biology and Ecology*, 315(2), 187-209.

Metcalf and Eddy. (1991). *Wastewater Engineering: Treatment, Disposal, Reuse*. 3rd ed. McGraw-Hill, Inc., New York.

Meyerhoff, R.D., Bounds, D.G., Moore, T.F. (2006). “Innovative Strategies to Apply Water Quality Criteria for Bacteria: a 50-state Survey of Approved Approaches for Freshwaters.” *Proceedings WEFTEC 2006*.

Minton, G. and R. Sutherland. (2010). “Street Dirt: A Better Way of Measuring BMP Effectiveness.” *Stormwater Magazine*. p. 12-22. March/April.

Mittal, G.S. (2004). “Characterization of the effluent wastewater from abattoirs for land application.” *Food Reviews International*, 20(3), 229-256.

Monroe, M. (2009). “A Multi-faceted Approach to Microbial Source Tracking within Secondary Environments.” Master’s Thesis, Colorado School of Mines.

Muirhead, R.W., Collins, R.P., and Bremer, P.J. (2006a). "Interaction of *Escherichia coli* and soil particles in runoff." *Appl. Environ. Microbiol.*, 72(5), 3406-3411.

Muirhead, R.W., Collins, R.P., and Bremer, P. J. (2006b). "Numbers and transported state of *Escherichia coli* in runoff direct from fresh cowpats under simulated rainfall." *Letters Appl. Microbiol.*, 42(2), 83-87.

Muirhead, R.W., Davies-Colley, R. J., Donnison, A.M., and Nagels, J.W. (2004). "Faecal bacteria yields in artificial flood events: quantifying in-stream stores." *Water Research*, 38(5), 1215-1224.

Murphy, C., Carroll, C., and Jordan, K.N. (2006). "Environmental survival mechanisms of the foodborne pathogen *Campylobacter jejuni*." *Journal of Applied Microbiology*, 100(4), 623-632.

Murray, J. and Steets, B. (2009). "Dry-weather Hydrology, DNA-based Microbial Source Tracking and Feasibility Analysis for the Laguna Watershed." In *Proceedings of StormCon 2009*, Los Angeles, CA.

New York City Department of Environmental Protection. (2004). *Final Environmental Impact Statement for the New York City DEP's Expanded Waterfowl Management Program.* April 13.

Noble, R.T. and Fuhrman, J.A. (2001). "Enteroviruses detected by reverse transcriptase polymerase chain reaction from the coastal waters of Santa Monica Bay, California: Low correlation to bacterial indicator levels." *Hydrobiologia*, 460, 175-184.

Noble, R.T. and Weisberg, S.B. (2005). "A review of technologies for rapid microbial detection in recreational waters." *Journal of Water and Health,* 3(4), 381-392.

Noble, R.T., Allen, S.M., Blackwood, A.D., Chu, W.P., Jiang, S.C., Lovelace, G.L., Sobsey, M.D., Stewart, J.R., and Wait, D.A. (2003a). "Use of viral pathogens and indicators to differentiate between human and non-human fecal contamination in a microbial source tracking comparison study." *Journal of Water and Health*, 1, 195-207.

Noble, R.T., Dorsey, J.H., Leecaster, M., Orozco-Borbon, V., Reid, D., Schiff, K., and Weisberg, S. B. (2000). "A regional survey of the microbiological water quality along the shoreline of the southern California bight." *Environmental Monitoring and Assessment* , 64, 435-447.

Noble, R.T., Griffith, J.F., Blackwood, A.D., Fuhrman, J.A., Gregory, J.B., Hernandez, X., Liang, X., Bera, A.A., Schiff, K. (2006). "Multi-tiered approach using Quantitative Polymerase Chain Reaction for tracking sources of fecal pollution to Santa Monica Bay, CA." *Appl. Environ. Microbiol*. 72 (2): 1604-1612.

Noble, R.T., Leecaster, M.K., McGee, C.D., Weisberg, S.B., and Ritter, K. (2004). "Comparison of bacterial indicator analysis methods in stormwater-affected coastal waters." *Water Res*., 38(5), 1183-1188.

Noble, R.T., Weisberg, S.B., Leecaster, M.K., McGee, C.D., Dorsey, J.H., Vainik, P., and Orozco-Borbon, V. (2003b). “Storm effects on regional beach water quality along the southern California shoreline.” *Journal of Water and Health,* 1, 23-31.

Nolan, S., Jones, S., Landry, N. (2004). “Evaluating the Stormwater Treatment Performance of AbTech Industries Smart Sponge(R) Plus in Seabrook, New Hampshire.” New Hampshire Estuary Project, University of New Hampshire, and New Hampshire Department of Environmental Services. August, 2004.

Novick, J. (2012). “Assessing Progress towards Reducing *E. coli* Levels in Dry Weather Discharges from Denver’s MS4.” Poster presented at the *Eighth National Monitoring Conference*, Portland, Oregon, April 30 – May 4, 2012.

Novick, J. (2013). Reducing *E. coli* Levels in Dry Weather Discharges from Denver’s MS4: How Effective are System Maintenance BMPs After Four Years of Implementation? Presentation at *Colorado Association of Stormwater and Floodplain Managers 24th Annual Conference.* September 10-13, 2013, Steamboat Springs, CO.

Novotny, V.H., and Olem, H. (1994). *Water quality: Prevention, identification, and management of diffuse pollution.* Van Nostrand Reinhold.

Nuzzi, R. and Burhans, R. (1998). “The use of enterococcus and coliforms in characterizing bathing-beach waters.” *Journal of Environmental Health,* 60, 16-22.

Oliver, B.G. and Cosgrove, E.G. (1975). “The disinfection of sewage treatment plant effluents using ultraviolet light.” *The Canadian Journal of Chemical Engineering,* 53 (2): 170–174.

Oliver, D.M., Clegg, C.D., Heathwaite, A.L., and Haygarth, P.M. (2007). “Preferential attachment of Escherichia coli to different size fractions of an agricultural grassland soil.” *Water, Air, and Soil Pollut*., 185, 369-375.

Oliver, J.D. (2005). “The viable but nonculturable state in bacteria.” *J. Microbiol*., 43, 93-100.

Olivieri, A.W., Boehm, A., Sommers, C.A., Soller, J.A., Eisenberg, J.N.S., and Danielson, R. (2007). *Development of a Protocol for Risk Assessment of Microorganisms in Separate Stormwater Systems.* Water Environment Research Foundation, Project 03-SW-2, Final Project Report.

Olivieri, V.P., Kruse, C. W., and Kawata, K. (1977b). *Microorganisms in Urban Stormwater*. USEPA Rept. No. EPA-600/2-77-087 (July).

Orange County Public Works. (2014). Orange County Public Works Environmental Engineering, Water Quality Projects, Urban Runoff Diversion Facilities. Accessed at: http://media.ocgov.com/gov/pw/watersheds/programs/waterways/enveng/default.asp.

Orange County Stormwater Program. (2014). *Aliso Creek Watershed Runoff Management Plan Water Quality Data Assessment 2013 Annual Report*.

Oshiro, R. and Fujioka, R. (1995). “Sand, soil, and pigeon droppings: sources of indicator bacteria in the waters of Hanauma Bay, Oahu, Hawaii.” *Water Science Technology*, 31(5-6), 251-254.

Ott, E., Muller, T., Muller, M., Franz, C.., Ulrich, A., Gabel, M. and Seyfarth, W. (2001). “Population dynamics and antagonistic potential of enterococci colonizing the phyllosphere of grasses.” *J. Appl. Microbiol.*, 91(1), 54–66.

Pachepsky, Y.A., Sadeghi, A.M., Bradford, S.A., Shelton, D.R., Guber, A.K., and Dao, T. (2006). “Transport and fate of manure-borne pathogens: Modeling perspective.” *Agric. Water Mgmt*., 86, 81-92.

Padia, R., Karthikeyan, R., Mukhtar, S., and Parker, I. (2012). “Occurrence and fate of *E. coli* from various non-point sources in a subtropical watershed.” *Journal of Natural & Environmental Sciences*, *3*(1), 9-18.

Parker, I. D., Lopez, R.R., Padia, R., Gallagher, M., Karthikeyan, R., Cathey, J.C., Silvy, N.J., and Davis, D.S. (2013). “Role of free-ranging mammals in the deposition of *Escherichia coli* into a Texas floodplain.” *Wildlife Research*. 40(7): 570-577.

Parsek, M.R., and Singh, P.K. (2003). “Bacterial biofilms: An emerging link to disease pathogenesis.” *Annual Review of Microbiology*, 57, 677-701.

Parveen, S., Portier, K.M., Robinson, K., Edmiston, L. and Tamplin, M.L. (1999). “Discriminant analysis of ribotype profiles of *Escherichia coli* for differentiating human and nonhuman sources of fecal pollution.” *Appl. Environ. Microbiol*,. 65(7), 3142–3147.

Passeport, E., Hunt, W.F., Line, D.E., Smith, R.A., and Brown, R.A. (2009). “Field study of the ability of two grassed bioretention cells to reduce storm-water runoff pollution.” *Journal of Irrigation and Drainage Engineering*, 135(4), 505-510.

Petropolou, C., Tolson, J., DeFlaun, M., Lanyon, R., Granato, T., Rijal, G., Gerba, C., Lue-Hing C., and Patterson Environmental Consultants. (2008). “Disinfection vs. Non-disinfection Microbial Risk Assessment for Recreational Use of Chicago Area Waterways.” *WEFTEC 2008*, October 21.

Pierce, R. (2001). “Managing Raccoon Problems in Missouri, University of Missouri Extension.” Publication G9453.

Pitt, R. (1983). *Urban Bacteria Sources and Control in the Lower Rideau River Watershed*, Ottawa, Ontario, Ontario Ministry of the Environment, ISBN 0-7743-8487-5. 165 pgs.

Pitt, R. (1985). *Characterizing and Controlling Urban Runoff through Street and Sewerage Cleaning*. U.S. Environmental Protection Agency, Storm and Combined Sewer Program, Risk Reduction Engineering Laboratory. EPA/600/S2-85/038. PB 85-186500. Cincinnati, Ohio. 467 pgs. June.

Pitt, R. (2002). “Emerging stormwater controls for critical source areas.” *Management of Wet-Weather Flow in the Watershed*. Sullivan, D. & Field R. (Eds). p. 14-16. CRC Press, Boca Raton, FL.

Pitt, R. (2004). *Control of Microorganisms in Urban Waters*. (http://unix.eng.ua.edu/~rpitt/Class/ExperimentalDesignFieldSampling/MainEDFS.html).

Pitt, R. (2011a). “Lincoln, Nebraska Standard Land Use Characteristics and Pollutant Sources.” Prepared in support of *Antelope Creek Watershed Basin Management Plan*. Prepared for City of Lincoln, NE, and Lower Platte South Natural Resources District. Prepared with EA Engineering, JEO Consulting Group and Wright Water Engineers.

Pitt, R. (2011b). “Lincoln, Nebraska, Retrofit Stormwater Management Options.” Prepared in support of *Antelope Creek Watershed Basin Management Plan*. Prepared for City of Lincoln, NE, and Lower Platte South Natural Resources District. Prepared with EA Engineering, JEO Consulting Group and Wright Water Engineers.

Pitt, R. and Bozeman, M. (1979). *Lake Merritt Management Plan.* City of Oakland, Ca. May.

Pitt, R. and Bozeman, M. (1982). “Sources of Urban Runoff Pollution and Its Effects on an Urban Creek.” EPA-600/S2-82-090, U.S. Environmental Protection Agency, Cincinnati, Ohio.

Pitt, R. and McLean, J. (1986). Toronto Area Watershed Management Strategy Study: Humber River Pilot Watershed Project. Ontario Ministry of the Environment, Toronto, Ontario. 486 pgs.

Pitt, R., and Rittenhouse, B. (2001). *Methods for detection of inappropriate discharges to storm drainage systems, background literature and summary of findings*. EPA X-82907801-0. Washington, DC: United States Environmental Protection Agency. Prepared for USEPA, Office of Water-Water Permits Division, USA.

Pitt, R., Chaturvedula, S., Karri, V., Nara, Y. (2004a). “Source Verification of Inappropriate Discharges to Storm Drainage Systems.” *Proceedings of the 72nd Annual Water Environment Federation Technical Exposition and Conference* [CD-ROM]; New Orleans, Louisiana, Oct 10-13; Water Environment Federation: Alexandria, Virginia.

Pitt, R., Clark, S. and Parmer, K. (1994). *Potential Groundwater Contamination from Intentional and Nonintentional Stormwater Infiltration.* EPA Office of Research and Development: Cincinnati, OH. EPA/600/SR-94/051. http://www.p2pays.org/ref/07/06744.pdf

Pitt, R., Clark, S., Johnson, P., Morquecho, R., Gill, S., and Pratap, M. (2004b). “High level treatment of stormwater heavy metals.” *Proc. 2004 World Water Environmental Resources Congress: Critical Transitions in Water and Environmental Resources Management*. American Society of Civil Engineers, Reston, VA. 917-926.

Pitt, R., Field, R., Lalor, M,. and O’Shea, M. (1992). “U.S. EPA’s Manual of Practice for the Investigation and Control of Cross-connection Pollution into Storm Drainage Systems.”

In *International Perspectives on Integrated Stormwater Management* (Edited by R. Field, M. O'Shea, and K.K. Chin). pp. 355-368. Lewis Publishers.

Pitt, R., Lalor, M., Field, R., Adrian, D.D., and Barbe, D. (1993). *A User's Guide for the Assessment of Non-Stormwater Discharges into Separate Storm Drainage Systems*. U.S. Environmental Protection Agency, Storm and Combined Sewer Program, Risk Reduction Engineering Laboratory. EPA/600/R-92/238. PB93-131472. Cincinnati, Ohio. 87 pgs. January.

Pitt, R., Maestre, A. and Morquecho, R. (2008). National Stormwater Quality Database, Version 3. February.

Pitt, R., Maestre, A., Morquecho, R., Brown, E., Schueler, T., Cappiella, K. and Sturm, P. (2004c). *National Stormwater Quality Database (NSQD), Research Progress Report.* January. Accessible at: http://chesapeakestormwater.net/wp-content/uploads/downloads/2012/02/National-SW-Quality-Database-report.pdf

Point Loma Nazarene University Fermian Institute for Business and Economic Institute. (2011). *Meeting Water Quality Standards for San Diego's Recreational Waters: A Cost-Benefit Analysis*. Commissioned by the City of San Diego Transportation and Stormwater Department.

Pope, M.L., Bussen, M., Feige, M. A., Shadix, L., Gonder, S., Rodgers, C., Chambers, Y., Pulz, J., Miller, K., Connell, K., and Standridge, J. (2003). "Assessment of the effects of holding time and temperature on *Escherichia coli* densities in surface water samples." *Applied and Environmental Microbiology*, 69(10), 6201-6207.

Powelson, D.K., and Mills, A.L. (2001). "Transport of *Escherichia coli* in Sand Columns with Constant and Changing Water Contents." *J. Environ. Qual.*, 30(1), 238-245.

Prouty, A.M., and Gunn, J.S. (2003). "Comparative analysis of Salmonella enterica serovar typhimurium biofilm formation on gallstones and on glass." *Infection and Immunity*, 71(12), 7154-7158.

Prouty, A.M., Schwesinger, W.H., and Gunn, J.S. (2002). "Biofilm formation and interaction with the surfaces of gallstones by Salmonella spp." *Infection and Immunity*, 70(5), 2640-2649.

Puigagut J., Villaseñor J., Salas J.J., Bécares E, García, J. (2007). "Subsurface-flow constructed wetlands in Spain for the sanitation of small communities: a comparative study." *Ecol Eng* 30:312–319.

Pusch, D., Oh, D.Y., Wolf, S., Dumke, R., Schroter-Bobsin, U., Hohne, M., Roske, I. and Schreier, E. (2005). "Detection of enteric viruses and bacterial indicators in German environmental waters." *Arch. Virol.*, 150(50), 929–947.

Ram, J., Thompson, B., Turner, C., Nechvatal, J., Sheehan, H., and Bobrin, J. (2007). "Identification of pets and raccoons as sources of bacterial contamination of urban storm sewers using a sequence-based bacterial source tracking method." *Water Research,* 41(16), 3605–3614.

Ran Q., Badgley, B.D., Dillon, N., Dunny, G.M., and Sadowsky, M.J. (2013). "Occurrence, Genetic Diversity, and Persistence of Enterococci in a Lake Superior Watershed." *Appl. Environ. Microbiol*., 79(9), 3067-3075.

Rasmus, J. and Weldon, K. (2003). "Moonlight Beach Urban Runoff Treatment facility: Using Ultraviolet disinfection to reduce bacteria counts." *Stormwater*, May/June 2003. Available at: http://www.forester.net/sw_0305_moonlight.html

Rasmussen, P. (2003). *Comparison and Continuous Estimates of Fecal Coliform and Escherichia Coli Bacteria in Selected Kansas Streams, May 1999 Through April 2002.* 03-4056, 87 p. USGS WRI 03-4056. http://ks.water.usgs.gov/pubs/abstracts/wrir.abstract.03-4056.html

Reeves, R.L., Grant, S.B., Morse, R.D., Copil Oancea, C.M., Sanders, B.F., Boehm, A.B. (2004). "Scaling and management of fecal indicator bacteria in runoff from a coastal urban watershed in southern California." *Environ. Sci. Technol*., 38(9), 2637-48

Rehmann, C.R., and Soupir, L.M. (2009). "Importance of interactions between the water column and the sediment for microbial concentrations in streams." *Water Research*, 43(18), 4579-4589. MS thesis. College Station, Tex.: Texas A&M University, Department of Biological and Agricultural Engineering.

Reisner, A., Krogfelt, K.A., Klein, B.M., Zechner, E.L., and Molin, S. (2006). "In vitro biofilm formation of commensal and pathogenic *Escherichia coli* strains: Impact of environmental and genetic factors." *Journal of Bacteriology*, 188(10), 3572-3581.

Rickard, A.H., McBain, A. J., Ledder, R.G., Handley, P.S., and Gilbert, P. (2003). "Coaggregation between freshwater bacteria within biofilm and planktonic communities." *Fems Microbiology Letters*, 220(1), 133-140.

Rijal, G., Tolson, J. K., Petropoulou, C., Granato, T. C., Glymph, A., Gerba, C., Deflaun, M. F., O'Connor, C., Kollias, L., and Lanyon, R. (2011). "Microbial risk assessment for recreational use of the Chicago Area Waterway System," *Journal of Water and Health*, January 9, 2011.

Rivas, L., Dykes, G.A., and Fegan, N. (2007). "A comparative study of biofilm formation by Shiga toxigenic *Escherichia coli* using epifluorescence microscopy on stainless steel and a microtitre plate method." *Journal of Microbiological Methods*, 69(1), 44-51.

Roser, D. and Ashbolt, N. (2006). "Microbial exposure assessment of an urban recreational lake: a case study of the application of new risk-based guidelines." *Water Science and Technology*. 54:245-252.

Rusciano, G.M. and Obropta, C.C. (2007). "Bioretention column study: Fecal coliform and total suspended solids reductions." *Transactions of the ASABE*, 50: 1261-1269.

Rusciano, G.M., Obropta, C.C. (2007). “Bioretention column study: Fecal coliform and total suspended solids reductions.” *Trans. ASABE.*, 50(4), 1261-1269.

Ryan, J.N., and Elimelech, M. (1996). “Colloid mobilization and transport in groundwater.” *Colloids and Surfaces a-Physicochemical and Engineering Aspects*, 107, 1-56.

Ryu, J.H., and Beuchat, L.R. (2005). “Biofilm formation by *Escherichia coli* O157:H7 on stainless steel: Effect of exopolysaccharide and curli production on its resistance to chlorine.” *Appl. Environ. Microbiol.*, 71(1), 247-254.

Sakamoto, G. and Cairns, W. L. (1997). “UV Dose-response Determination: Collimated Beam vs. Pilot Studies.” In: *Proceedings of Hawaii Water Environment Association Conference*, Honolulu, HI.

Salmon, T.P., and Gorenzel, W.P. (2005). “Cliff swallows.” Publication 7482. University of California Agriculture and Natural Resources, Davis, California, USA.

Santa Monica Bay Restoration Project (SMBRP). (1996). “An epidemiological study of possible adverse health effects of swimming in Santa Monica Bay.” *Santa Monica Bay Restoration Project*, Monterrey Park, CA.

Santa Monica Public Works (ND). SMURRF: Santa Monica Urban Runoff Recycling Facility website: http://www.smgov.net/departments/publicworks/contentciveng.aspx?id=7796, accessed August 2014.

Sauer, E.P., VandeWalle J.L., Bootsma M.J., McLellan S.L. (2011). Detection of the human specific *Bacteroides* genetic marker provides evidence of widespread sewage contamination of stormwater in the urban environment. *Water Res*. 45(14):4081–91.

Sawyer, C.B. 2009. *Characterization of Escherichia coli Related to Construction Site Sediment Basins.* Doctoral Dissertation. Department of Forestry and Natural Resources, Clemson University, Paper 498, 126. pp.

Sawyer, C.B., English, R., and Hayes, J.C. (2010). “*Escherichia coli* in Construction Site Runoff and Sediment Basin Discharge in Piedmont South Carolina.” *Proceedings, EWRI World Congress.*

Schiff, K., Griffith, J., Lyon, G. (2005). *Microbiological water quality at reference beaches in southern California during wet weather.* Technical Report 448. Southern California Coastal Water Research Project. Westminster, CA.

Schilling, J.G. (2005). *Street Sweeping – Report No. 1, State of the Practice*. Prepared for Ramsey-Washington Metro Watershed District. North St. Paul, Minnesota. June.

Schoen, M. and Ashbolt, N. (2010). “Assessing pathogen Risk to Swimmers at Non-sewage impacted Recreational Beaches.” *Environmental Science and Technology*, 44(7), 2286-2291.

Schoonover, J.E. and Lockaby, B.G. (2006). "Changes in nutrient and bacteriological properties of stream water across an urban-rural gradient in western Georgia." *Journal of Hydrology*, 331, 371-382.

Schroeder, E.D., Stallard, W.M., Thompson, D.E., Loge, F.J., Deshussess, M.A., Cox, H.J. (2002). *Management of Pathogens Associated with Storm Drain Discharge, Results of Investigations of the Presence of Human Pathogens in Urban Storm Drains*. Prepared for the Division of Environmental Analysis, California Department of Transportation Interagency Agreement No. 43A0073. CTSW-RT-02-025. May. http://www.dot.ca.gov/hq/env/stormwater/pdf/CTSW-RT-02-025.pdf

Schueler, T. and Holland, H. (2000). *Microbes and Urban Watersheds: Concentrations, Sources and Pathways, The Practice of Watershed Protection*. The Center for Watershed Protection, Ellicott City, MD.

Schwarze Industries. (2004). "Virginia test further documents pickup of high efficiency sweepers." *American Sweeper*. 8(1).

Searcy, K.E., Packman, A.I., Atwill, E.R., and Harter, T. (2006a). "Capture and retention of *Cryptosporidium parvum* oocysts by *Pseudomonas aeruginosa* biofilms." *Appl. Environ. Microbiol*., 72(9), 6242-6247.

Searcy, K.E., Packman, A.L., Atwill, E.R., and Harter, T. (2006b). "Deposition of Cryptosporidium oocysts in streambeds." *Appl. Environ. Microbiol*., 72(3), 1810-1816.

Selvakumar, A. and Borst, M. (2006). "Variation of microorganism concentrations in urban stormwater runoff with land use and seasons." *J. Water Health*, 4(1), 109–124.

Selvakumar, A., Borst, M., Boner, M., and Mallon, P. (2004). "Effects of Sample Holding Time on Concentrations of Microorganisms in Water Samples." *Water Environment Research*, 76 (1): 67-72.

Sercu, B.S., Van De Werfhorst, L.C., Murray, J.L.S., and Holden P.S. (2009). Storm drains are sources of human fecal pollution during dry weather in three urban southern California watersheds. *Environ. Sci. Technol.* 43(2):293-8.

Sercu, B.S., Van De Werfhorst, L.C., Murray, J.L.S., and Holden P.S. (2011). "Sewage Exfiltration As a Source of Storm Drain Contamination during Dry Weather in Urban Watersheds," *Environ. Sci. Technol*., 45 (17): 7151–7157.

Shergill, S.S. (2004). *Quantification of Escherichia Coli and Enterococci Levels Wet Weather and Dry Weather Flows*. MSCE thesis, Dept. of Civil and Environmental Engineering, University of Alabama.

Shergill, S.S. and Pitt, R. (2004). "Quantification of Escherichia coli and enterococci levels in wet weather and dry weather flows." Presented at the *77th Annual Water Environment Federation Technical Exposition and Conference*. New Orleans, LA. Oct 4 – 7.

Shipitalo, M.J., and Gibbs, F. (2000). "Potential of earthworm burrows to transmit injected animal wastes to tile drains." *Soil Sci. Soc. Am. J.*, 64, 2103-2109.

Shirmohammadi, A., Chaubey, I., Harmel, R.D., Bosch, D.D., Muñoz-Carpena, R., Dharmasri, C., Sexton, A., Arabi, M., Wolfe, M.L., Frankenberger, J., Graff, C., Sohrabi, T.M. (2006). "Uncertainty in TMDL Models." *Transactions of the American Society of Agricultural and Biological Engineers*, 49(4), 1033−1049.

Shirmohammadi, A., Montas, H.J., Bergstrom, L., and Knisel, W.G., Jr. (2001). "Water quality models." In *Agricultural Nonpoint-Source Pollution: Watershed Management and Hydrology*, 233-256. W. F. Ritter and A. Shirmohammadi, eds., Washington, D. C.: Lewis Publishers.

Sieracki, M. (1980). *The effects of short exposures of natural sunlight on the decay rates of enteric bacteria and coliphage in a simulated sewage outfall microcosm*. Master of Science Thesis. Microbiology, University of Rhode Island. Kingston, RI.

Skinner, J.F., Kappeler, J., and Guzman, J. (2010). "Regrowth of enterococci and coliform in biofilm." *Stormwater*, Santa Barbara, California.

Sleytr, K., Tietz A., Langergraber, G., Haberl. R. (2007). "Investigation of bacterial removal during the filtration process in constructed wetlands." *Sci Total Environ* 380:173–80.

Slifko, T.R., Heil, A., Walker, K., Reinke, R., Gregory, J., Padilla, M. and Shay, M. (2009). *Model Program for Bacterial Source Identification and Abatement: Redondo Beach Pier Pilot Project*. Accessible at: http://water.epa.gov/type/oceb/beaches/upload/SlifkoTerri.pdf.

Smith, A.E., Craven, S.R. and Curtis, P.D. (1999). *Managing Canada Geese in Urban Environments.* Jack Berryman Institute Publication 16, and Cornell University Cooperative Extension, Ithaca, N.Y.

Smith, C.D. (2006). *Waterfowl Management in Stormwater Management Ponds*, Thesis, University of Waterloo, Waterloo, Ontario, Canada.

Sokal, R.R and Rohlf, F. J. (1969). *Biometry: The Principles and Practice of Statistics in Biological Research.* W.H. Freeman and Co. New York.

Solic, M., and Krstulovic, N. (1992). "Separate and combined effects of solar radiation, temperature, salinity, and pH in the survival of fecal coliforms in seawater." *Mar. Pollut. Bull.*, 24, 411–416.

Soller, J.A., Bartrand, T., Ashbolt, N.J., Ravenscroft, J., and Wade, T.J. (2010a). "Estimating the primary etiologic agents in recreational freshwaters impacted by human sources of faecal contamination." *Water Res.*, 44(16), 4736-4747.

Soller, J.A., Schoen, M.E., Bartrand, T., Ravenscroft, J., and Ashbolt, N.J. (2010b). "Estimated human health risks from exposure to recreational waters impacted by human and non-human sources of faecal contamination." *Water Res.*, 44(16), 4674-4691.

Solo-Gabriele, H.M., Wolfert, M.A., Desmarais, T.R. and Palmer, C.J. (2000). "Sources of *Escherichia coli* in a coastal subtropical environment." *Applied and Environmental Microbiology*, 66, 230-237.

Soupir, M.L., Mostaghimi, S., and Dillaha, T. (2010). "Attachment of *Escherichia coli* and enterococci to particles in runoff." *J. Environ. Qual.*, 39(3), 1019-1027.

Soupir, M.L., Mostaghimi, S., and Lou, J. (2008b). "Die-off of *E. coli* and enterococci in dairy cowpats." *Trans. ASABE*, 51(6), 1987-1996.

Soupir, M.L., Mostaghimi, S., and Love, N.G. (2008a). "A method to partition between attached and unattached *E. coli* in runoff from agricultural lands." *J. American Water Resources Assoc.*, 44(6), 1591-1599.

Source Molecular Corporation (2013). 2013 price list for microbial source tracking analysis (personal communication with Grace Anderson, Source Molecular). Website: http://www.sourcemolecular.com/.

Southern California Water Resources Research Board (ND). Project: Wet and Dry Weather Beach Epidemiology Studies (website, accessible at: http://www.sccwrp.org/ResearchAreas/BeachWaterQuality/CaliforniaEpidemiologicalStudies.aspx. Accessed August 20, 2014.

Standridge, J.H., Delfine, J.J., Kleppe, Butler, R. (1979). "Effect of waterfowl (*Anas platyrhynchos*) on indicator bacteria populations in a recreational lake in Madison, Wisconsin." *Applied and Environmental Microbiology*, 38(3), 547-550.

Stanley, K., Cunningham, R., and Jones, K. (1998). "Isolation of Campylobacter jejuni from groundwater." *Journal of Applied Microbiology*, 85(1), 187-191.

State Water Resources Control Board and the Southern California Stormwater Monitoring Coalition. (2004). *Southern California Coastal Water Research Project, Model Monitoring Program for Municipal Separate Storm Sewer Systems in Southern California.* Technical Report #419. ftp://ftp.sccwrp.org/pub/download/DOCUMENTS/TechnicalReports/419_smc_mm.pdf

Steets, B. (2013). "Lessons from California Bacteria TMDLs." Presentation to Colorado *E. coli* Work Group, Denver, CO. January 2013.

Stein, E.D., Tiefenthaler, L.L., and Schiff, K.C. (2007). *Sources, Patterns and Mechanisms of Storm Water Pollutant Loading From Watersheds and Land Uses of the Greater Los Angeles Area, California, USA.* Southern California Research Project (SCCWRP) Technical Report, 510. March 2007.

Stiles, T.C. (2008). “Kansas Thinking on Bacteria Water Quality: Uses, Criteria and TMDLs”, Kansas Department of Health and Environment. Accessed at: http://www.bayoupreservation.org/html/StilesPresentationSymposium.pdf.

Stinson, M.K. and Perdek, J.M. (2003). “Chapter Three: Management and Control of Pathogens.” In *Managing Urban Watershed Pathogen Contamination*, EPA/600/R-03/111, EPA Office of Research and Development, Cincinnati, OH.

Stinson, M.K., Field, R., Moffa, P.E., Goldstein, S.L., Smith, K.J., and Delva, E. (1998). “High-Rate Disinfection Technologies for Wet Weather Flow.” In *Advances in Urban Wet Weather Pollution Reduction. Water Environment Federation Specialty Conference,* Cleveland, OH.

Stinson, M.K., Field, R., Rukovets, R.B. (2009). “Impact of Wet-Weather Peak Flow Blending on Disinfection and Treatment: A Case Study at Three Wastewater Treatment Plants. National Risk Management Research Laboratory, Office of Research and Development.” In *Proceedings of Disinfection 2009, Water Environment Federation Specialty Conference*, February 28 – March 3, 2009, Atlanta, Georgia.

Stoeckel, D. (2010). “Presentation: Microbial Source Tracking—Sources of *E. coli* in Upper Fountain Creek.” Colorado Dept. Public Health and Environment E. coli Work Group. April 6, 2010.

Stoeckel, D.M. and Harwood, V.J. (2007). “Performance, design, and analysis in microbial source tracking studies.” *Appl. Environ. Microbiol.*, 73, 2405–2415.

Stoeckel, D.M., Mathes, M.V., Hyer, K.E., Hagedorn, C., Kator, H., Lukasik, J., O’Brien, T.L., Fenger, T.W. (2004). “Comparison of seven protocols to identify fecal contamination sources using *Escherichia coli*.” *Environ. Sci. Technol.,* 38, 6107-6117.

Storey, M. and Ashbolt, N. (2001). “Persistence of two model enteric viruses (B4-8 and MS-2 bacteriophages) in water distribution pipe biofilms.” *Water Science and Technology*, 43(12), 133-138.

Stott, R., Davies-Colley, R., Nagels, J. and Packman, A. (2007). “Flume studies of uptake of faecal microbes and turbidity in gravel streambeds.” *Proceedings of the 11th International IWA Conference on Diffuse Pollution*, Brazil, August, 2007.

Strecker, E., Leisenring, M., Poresky, A., Pomeroy, C., Mattson, J., and Barrett, M. (2009). “Recommended Initial Set of Water Quality Constituents and BMPs to Address in BMP Algorithms - Task 3A Technical Memo of Linking BMP Performance to Receiving Water Protection to Improve BMP Selection and Design.” WERF Project SW1R06. Submitted to Project Issue Area Team August 24, 2009.

Struck, S.D., Selvakumar, A., Borst, M. (2006). “Performance of Stormwater Retention Ponds and Constructed Wetlands in Reducing Microbial Concentrations.” *United States Office of Environmental Protection and Development*, September, 2006.

Structural BMP Prioritization and Analysis Tool (SBPAT) website (ND). Developed by Geosyntec Consultants and GreenInfo Network for Heal the Bay, City of Los Angeles, and County of Los Angeles Department of Public Works and Orange County Transportation Authority. Accessible at: http://sbpat.net/index.html.

Surbeck, C.Q., Jiang, S.C., and Grant, S.B. (2010). "Ecological control of FIB in an urban stream." *Environ. Sci. Technol.*, 44(2), 631-637.

Swallow, M., Huffman, J., Van Why, K. and D'Angelo. G. (2010). *The Effect of Goose Management on Water Quality*. Northeast Wildlife DNA Laboratory, East Stroudsburg University, East Stroudsburg, Pennsylvania and USDA APHIS Wildlife Services. Harrisburg, Pennsylvania.

Tang, K.W., Dziallas, C., and Grossart, H. (2011). "Zooplankton and aggregates as refuge for aquatic bacteria: protection from UV, heat and ozone stresses used for water treatment." *Environmental Microbiology*, 13 (2): 378–390.

Teague, A., Karthikeyan, R., Babbar-Sebens, M., Srinivasan, R., and Persyn, R.A. (2009). "Spatially explicit load enrichment calculation tool to identify potential E. coli sources in watersheds." *Transactions of the ASABE*, 52(4), 1109-1120.

TetraTech, Inc. (2011). *Boulder Creek, Colorado Segment 2b: From 13th Street to the Confluence with South Boulder Creek Total Maximum Daily Load Escherichia coli*. Prepared for the City of Boulder, CO. February 7.

TetraTech, Inc. (2013). *San Diego River Watershed: Comprehensive Load Reduction Plan – Phase II*. Submitted to City of San Diego Transportation & Storm Water Department. July 24, 2013.

Texas Commission on Environmental Quality (TCEQ). (2013). *Implementation Plan for Seventy-Two Total Maximum Daily Loads for Bacteria in the Houston-Galveston Region.* January 30, 2013.

Texas Water Quality Task Force. (2007). *Bacteria Total Maximum Daily Load Task Force Report*. June 4, 2007. Prepared for Texas Commission on Environmental Quality and Texas State Soil and Water Conservation Board.

Thomann, R.V. and Mueller, J.A. (1987). *Principles of Surface Water Quality Modeling and Control*. Harper and Row: New York.

Thomas, C., Hill, D.J., and Mabe, M. (1999). "Morphological changes of synchronized Campylobacter jejuni populations during growth in single phase liquid culture." *Letters in Applied Microbiology*, 28, 194–198.

Tiefenthaler, L., Stein, E.D., Lyon, G.S. (2008). *Fecal Indicator Bacteria (FIB) Levels During Dry Weather from Southern California Reference Streams*. SCCWRP Technical Report 542. January 2008.

Till, D.G. and McBride, G.B. (2004). "Potential public health risk of Campylobacter and other zoonotic waterborne infections in New Zealand." In *Waterborne Zoonoses: Identification Causes and Control*, Cotruvo, J. A., Dufour, A., Rees, G., Bartram, J., Carr, R., Cliver, D. O., Craun, G. F., Fayer, R. & Gannon, V. P. J. (eds.). World Health Organisation, IWA, London, pp. 191–207. Available from: www.who.int/water_sanitation_health/diseases/zoonosessect4.pdf.

Toranzos, G.A. (2007). "Current and possible alternate indicators of fecal contamination in tropical waters: A short review." *Environmental Toxicology and Water Quality*, 6, 121-130.

Tufenkji, N., Redman, J.A., and Elimelech, M. (2003). "Interpreting deposition patterns of microbial particles in laboratory-scale column experiments." *Environmental Science & Technology*, 37(3), 616-623.

Tukey, J.W. (1977). *Exploratory Data Analysis*. Addison-Wesley Publishing Co.

U.S. Department of Agriculture Animal and Plant Health Inspection Services, Wildlife Services. (2001). "Assistance with waterfowl." Factsheet. United States Department of Agriculture, Animal and Plant Health Inspection Service, Wildlife Services. November.

U.S. Department of Agriculture, Animal and Plant Health Inspection Service, Wildlife Services. (2009). "Management of Canada goose nesting." August 2009. 8 pp. http://www.aphis.usda.gov/wildlife_damage/downloads/canada_goose.pdf

U.S. Department of Agriculture, Animal and Plant Health Inspection Service, Wildlife Services. (2010). "Assistance with waterfowl damage." Factsheet. United States Department of Agriculture, Animal and Plant Health Inspection Service, Wildlife Services. May 2010. 3 pp.

U.S. Department of Agriculture, Soil Conservation Service. (1993). *Soil Survey Manual*. Soil Conservation Service. U.S. Department of Agriculture Handbook 18.

U.S. Department of Agriculture-National Resources Conservation Service. (1996). *National Handbook of Water Quality Monitoring.* 450-vi-NHWQM. http://www.wsi.nrcs.usda.gov/products/W2Q/water_qual/docs/wqm1.pdf

U.S. Environmental Protection Agency. (1973a). *Hypochlorination of Polluted Stormwater Pumpage at New Orleans.* EPA-670/2-73-067, Office of Research and Development, Washington, DC.

U.S. Environmental Protection Agency. (1973b). *Microstraining and Disinfection of Combined Sewer Overflows - Phase II.* EPA-R2-73-124 Office of Research and Development, Washington, DC.

U.S. Environmental Protection Agency. (1979a). *Combined Sewer Overflow Abatement Program, Rochester, NY, Vol. II Pilot Plant Evaluations*. EPA/600/2-79-031b. EPA Office of Research and Development, Cincinnati, OH.

U.S. Environmental Protection Agency. (1979b). *Disinfection/Treatment of Combined Sewer Overflows, Syracuse, New York*. EPA/600/2-79-134. Office of Research and Development, Cincinnati, OH.

U.S. Environmental Protection Agency. (1986). *Ambient Water Quality Criteria for Bacteria – 1986*. EPA440/5-84-002. Washington, DC: US EPA. http://www.epa.gov/waterscience/beaches/files/1986crit.pdf

U.S. Environmental Protection Agency. (1992). *National Pollutant Discharge Elimination System Stormwater Sampling Guidance Document*. EPA 833-B-92-001. http://www.epa.gov/npdes/pubs/owm0093.pdf

U.S. Environmental Protection Agency (EPA). (1993). *Subsurface Flow Constructed Wetlands for Wastewater Treatment: A Technology Assessment*, EPA 832-R-93-008.

U.S. Environmental Protection Agency. (1994). *Combined Sewer Overflow Control Policy, 59 Federal Register 18688*. April 19, 1994.

U.S. Environmental Protection Agency. (1996). *EPA Monitoring Guidance for Determining the Effectiveness of Nonpoint Source Controls*. EPA 841-B-96-004. http://nepis.epa.gov/EPA/htmL/Pubs/pubtitleOW.htm

U.S. Environmental Protection Agency. (1997). *EPA Monitoring Guidance for Determining the Effectiveness of Nonpoint Source Controls*. EPA 841-B-96-004. http://nepis.epa.gov/EPA/html/Pubs/pubtitleOW.htm

U.S. Environmental Protection Agency. (1999). *Preliminary Data Summary of Urban Stormwater Best Management Practices*. U.S. EPA 821-R-99-012. Office of Water, Washington, D.C.

U.S. Environmental Protection Agency. (2000). Improved Enumeration Methods for the Recreational Water Quality Indicators: Enterococci and *Escherichia coli*. EPA Office of Science. PA/821/R-97/004.

U.S. Environmental Protection Agency. (2001). *Protocol for Developing Pathogen TMDLs*. EPA 841-R-00-002. Washington, DC: US EPA Office of Water. http://www.epa.gov/owow/tmdl/pathogen_all.pdf

U.S. Environmental Protection Agency. (2002). *Guidance for Quality Assurance Project Plans*. EPA QA/G-5, EPA, Office of Environmental Information, Washington, D.C. www.epa.gov/quality/qs-docs/g5-final.pdfwww.epa.gov/quality/qs-docs/g5-final.pdf

U.S. Environmental Protection Agency. (2004). *Water Quality Standards for Coastal and Great Lakes Recreation Waters*. Federal Register: November 16, 2004 (Volume 69, Number 220, page 67217-67243).

U.S. Environmental Protection Agency. (2007a). *Report of the Experts Scientific Workshop on Critical Research Needs for the Development of New or Revised Recreational Water*

Criteria. March 26-30, 2007. EPA 823-R-07-006. http://www.epa.gov/waterscience/criteria/recreation/

U.S. Environmental Protection Agency. (2007b). *Critical Path Science Plan for the Development of New or Revised Recreational Water Criteria*. EPA 823-R-08-002. http://www.epa.gov/waterscience/criteria/recreation/plan/cpsplan.pdf

U.S. Environmental Protection Agency. (2007c). *Criteria Development Plan for the Development of New or Revised Recreational Water Criteria*. EPA 823-R-08-003. http://www.epa.gov/waterscience/criteria/recreation/plan/developmentPlan.pdf

U.S. Environmental Protection Agency. (2007d). *An Approach for Using Load Duration Curves in the Development of TMDLs*. EPA 841-B-07-006. United States Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds.

U.S. Environmental Protection Agency. (2009a). *Review of Zoonotic Pathogens in Ambient Waters*. Office of Water Health and Ecological Criteria Division. EPA 822-R-09-002. February.

U.S. Environmental Protection Agency. (2009b). *Review of Published Studies to Characterize the Relative Risk from Different Sources of Fecal Contamination*. Office of Water Health and Ecological Criteria Division. EPA 822-R-09-001. http://www.epa.gov/waterscience/criteria/recreation/pdf/fecalcontamrecreationalwaters.pdf

U.S. Environmental Protection Agency. (2010a). *Report on 2009 National Epidemiologic and Environmental Assessment of 60 Recreational Water Epidemiology Studies (NEEAR 2010 - Surfside & Boquerón).* EPA-600-R-10-168. Available at: http://www.epa.gov/neear/files/Report2009v5_508comp.pdf

U.S. Environmental Protection Agency. (2010b). *Comparison and Evaluation of Epidemiological Study Designs of Health Effects Associated with Recreational Water Use*.

U.S. Environmental Protection Agency. (2010c). *Impact of Wet-Weather Peak Flow Blending on Disinfection and Treatment: A Case Study at Three Wastewater Treatment Plants*. National Risk Management Research Laboratory, Office of Research and Development. EPA/600/R-10/003.

U.S. Environmental Protection Agency. (2011). "A grant-funded pilot program focusing on cleaning up Coyote Creek. Clean Creeks." *Healthy Communities Program Snapshot*. May 31.

U.S. Environmental Protection Agency. (2012). *Recreational Water Quality Criteria*. Office of Water 820-F-12-058. http://water.epa.gov/scitech/swguidance/standards/criteria/health/recreation/

U.S. Environmental Protection Agency. (2013). *Marine Beach Sanitary Survey User's Manual.* http://water.epa.gov/type/oceb/beaches/upload/Marine-Beach-Sanitary-Survey-User-Manual-March-2013.pdf

U.S. Environmental Protection Agency. (2014). "National Summary of Impaired Waters and TMDL Information." Website accessed August 21, 2014 at: http://iaspub.epa.gov/waters10/attains_nation_cy.control?p_report_type=T#tmdl_by_pollutant.

U.S. Geological Survey. (various dates). *Techniques of Water-Resources Investigations Reports*. http://pubs.usgs.gov/twri/http://pubs.usgs.gov/twri/

U.S. Government Accounting Office. (2013). *Clean Water Act: Changes Needed if Key EPA Program is to Help Fulfill the Nation's Water Quality Goals*.

Uhnoo, I., Wadell, G., Svensson, L., Olding-Stenkvist, E., Mobby, R. (1986). "Aetiology and epidemiology of acute gastroenteritis in Swedish children." *J. Infect*., 13, 73–89.

Unc, A., and Goss, M.J. (2004). "Transport of bacteria from manure and protection of water resources." *Applied Soil Ecology*, 25(1), 1-18.

Valiela, I., Alber, M., LaMontagne, M. (1991). "Fecal coliform loadings and stocks in Buttermilk Bay, Massachusetts, USA, and Management Implications." *Environmental Management*, 15(5), 659-674.

Van De Werfhorst, L.C., Murray, J.L.S., Reynolds, S., Reynolds, K., and Holden, P.A. (2014). "Canine Scent Detection and Microbial Source Tracking of Human Waste Contamination in Storm Drains," *Water Environment Research*, 86 (6): 550-558.

Wade, T.J., Pai, J., Eisenberg, J.N., Colford, J.M. (2003). "Do US Environmental Protection Agency Water Quality Guidelines for Recreational Waters Prevent Gastrointestinal Illness? A Systematic Review of Meta-Analysis." *Environmental Health Perspectives*, 111(8), 1102-1109.

Wang, G., and Doyle, M.P. (1998). "*Heat* shock response enhances acid tolerance of Escherichia coli." *Lett. Appl. Microbiol.*, 26, 31-4.

Wang, L., Mankin, K.R., and Marchin, K.R. (2004). "Survival of fecal bacteria in dairy cow manure." *Trans. ASAE*, 47(4), 1239-1246.

Water Environment Research Foundation (WERF). (2007). *Development of a Protocol for Risk Assessment of Microorganisms in Separate Stormwater Systems*. Prepared by Olivieri, A., Boehm, A., Sommers, C., Soller, J., Eisenberg, J., and Danielson, R. WERF publication No: 03-SW-2. Copublishers: WERF, Alexandria, VA, and IWA Publishing, Colchester, UK.

Water Environment Research Foundation (WERF). (2009). *Report on the Expert Scientific Workshop on Critical Research and Science Needs for the Development of Recreational*

Water Quality Criteria for Inland Waters. Dallas/Ft. Worth, Texas, February 18-20, 2009. Accessible at: http://www.werf.org/AM/Template.cfm?Section=Research_Profile&Template=/CustomSource/Research/PublicationProfile.cfm&id=PATH4W09

Water Environment Research Foundation (WERF). (2011). *Final Report: Quantification of Pathogens and Sources of Microbial Indicators for QMRA in Recreational waters*. PATH2R08.

Weber, J.A., Kampalath, R., Ferguson D., and Steets, B. (2013). “MS4 Outfall Dry Weather Bacteria Source Tracking Study.” Presented at *2013 CASQA Workshop*. Prepared by Geosyntec Consultants.

Weston Solutions, Inc. (2004). *Phase I Report for the San Diego River Kelp and Dog Waste Management Plan for Dog Beach and Ocean Beach*. Prepared for the City of San Diego.

Weston Solutions, Inc. (2009a). *San Diego River Source Tracking Investigation – Phase I, Final Report, Revision 1*. Prepared for the City of San Diego. December 2009.

Weston Solutions, Inc. (2009b). *San Diego County Municipal Copermittees 2007–2008 Urban Runoff Monitoring.* Prepared for the County of San Diego. January 2009.

Weston Solutions, Inc. (2010a). *San Diego County Municipal Copermittees 2008–2009 Urban Runoff Monitoring*. Prepared for the County of San Diego. January 2010.

Weston Solutions, Inc. (2010b). *Tecolote Creek Microbial Source Tracking Summary Phases I, II, and III.* Prepared for the City of San Diego. June 2010.

Weston Solutions. (2004). *San Diego River – Ocean Beach Water Quality Improvement Project – Phase II Kelp and Dog Waste Management Plan.* Prepared for the City of San Diego. San Diego, CA. September.

Weston Solutions. (2009a). *San Diego River Source Tracking Investigation – Phase I, Final Report Revision 1*. Prepared for City of San Diego Storm Water Department. San Diego, CA.

Weston Solutions. (2009b). *San Diego River Source Tracking Investigation – Phase II, Final Report Revision 1*. Prepared for City of San Diego Storm Water Department. San Diego, CA.

Weston Solutions. (2010). *Tecolote Creek Microbial Source Tracking Summary: Phases I, II, and II*. Prepared for City of San Diego Storm Water Department.

Whitman, R.L., Shively, D.A., Pawlik, H., Nevers, M.B. and Byappanahalli, M.N. (2003). “Occurrence of *Escherichia coli* and enterococci in Cladophora (Chlorophyta) in nearshore water and beach sand of Lake Michigan.” *Appl. Environ. Microbiol.*, 69(8), 4714–4719.

Whitman, R.L., Nevers, M.B., and Byappanahalli, M.N. (2006). "Examination of the Watershed-Wide Distribution of *Escherichia coli* along Southern Lake Michigan: An Integrated Approach." *Applied and Environmental Microbiology*, 72(11), 7301-7310.

Whitman, R.L., Nevers, M.B., and Gerovac, J. (1999). "Interaction of ambient conditions and fecal coliform bacteria in southern Lake Michigan waters: monitoring program implications." *Natural Areas Journal*, 19, 166-171.

Wilkinson, J., Jenkins, A., Wyer, M., and Kay, D. (1995). "Modeling Fecal-Coliform Dynamics in Streams and Rivers." *Water Research*, 29(3), 847-855.

Wilkinson, J., Kay, D., Wyer, M., and Jenkins, A. (2006). "Processes driving the episodic flux of faecal indicator organisms in streams impacting on recreational and shellfish harvesting waters." *Water Research*, 40(1), 153-161.

Williams, D.E. and Corrigan, R.M. (1994). "Pigeons (rock doves)." Pp. 87-96 in S. E. Hygnstrom, R.M. Timm, and G.E. Larson, eds., *Prevention and control of wildlife damage.* University of Nebraska, Lincoln, NE.

Wilson, B. and Pitt, R. (2010). "Bacterial indicator-species survival on impervious surfaces." *WEFTEC, the 83rd Annual Water Environment Federation Technical Exhibition and Conference***.** New Orleans, LA, October 2–6, 2010. Conference CD.

Wilson, B. and Pitt, R. (2011). "Survival of Bacterial Indicator-species on Impervious Surfaces." *ASCE/EWRI World Environment and Water Resources Congress*. Palm Springs, CA, May 22-26, 2011.

Wilson, B. and Pitt, R. (2013). "Survival of Bacterial Indicator-Species on Impervious Environmental Surfaces in Tuscaloosa, Alabama." Monograph 21. *Pragmatic Modeling of Urban Water Systems*, James, W., K.N. Irvine, James Y. Li, E.A. McBean, R.E. Pitt, and S.J. Wright (editors). Computational Hydraulics International. Guelph, Ontario. 2013.

WINSLAMM Website (ND). WinSLAMM (Source Loading and Management Model for Windows). Developed by PV Associates. Accessible at: http://www.winslamm.com/default.html

Wojtenko, I., Stinson, M.K., and Field, R. (2001). "Challenges of Combined Sewer Overflow Disinfection with Ultraviolet Light Irradiation." *Critical Reviews in Environmental Science and Technology*, 31(3), 223-239.

World Health Organization (WHO). (1999). *Health‑based Monitoring of Recreational Waters: The Feasibility of a New Approach (the "Annapolis Protocol").* WHO.SDE/WSH/99.1, Protection of the Human Environment, Water Sanitation and Health Series, WHO, Geneva

World Health Organization (WHO). (2003). *Emerging issues in water and infectious disease.* World Health Organization, accessible at: http://www.who.int/water_sanitation_health/emerging/emerging.pdf.

World Health Organization (WHO). (2004). *Waterborne Zoonoses: Identification, Causes, and Control.* J. A. Cotruvo, A. Dufour, G. Rees, J. Bartram, R. Carr, D. O. Cliver, G. F. Craun, R. Fayer and V. P. J. Gannon, eds., World Health Organization and IWA Publishing, London.

Wright Water Engineers (WWE) and Geosyntec Consultants. (2010). *International Stormwater Best management Practice (BMP) Database Pollutant Category Summary Bacteria*. Prepared for Water Environment Research Foundation, Federal Highway Administration, and Environment and Water Resources Institute of the American Society of Civil Engineers. Accessible at: www.bmpdatabase.org.

Wright Water Engineers, Inc. (WWE). (2014). *Big Dry Creek 2013 Annual Water Quality Report*. Prepared for the Big Dry Creek Watershed Association. Accessible at: www.bigdrycreek.org

Wu, J., Rees, P., Storrer, S., Alderisio, K., and Dorner, S. (2009). "Fate and Transport Modeling of Potential Pathogens: The Contribution of Sediments." *Journal of the American Water Resources Association (JAWRA)*, 45(1), 35-44.

Wymer, L.J. and Wade, T.J. (2007). "The Lognormal Distribution and Use of the Geometric Mean and the Arithmetic Mean in Recreational Water Quality Measurement" in *Statistical Framework for Recreational Water Quality Criteria and Monitoring*, John Wiley & Sons Ltd., Chichester.

Xanthopoulos, C., and Hahn, H.H. (1990). "Pollutants attached to particles from drainage areas." *Science of the Total Environment*, 93, 441-448.

Yamahara, K. M., Layton, B.A., Santoro, A. E., Boehm, A.B. (2007). "Beach sands along the California coast are diffuse sources of fecal bacteria to coastal waters." *Environ. Sci. Technol.*, 41, 4515–4521.

Young, K.D., Thackston, E.L. (1999). "Housing density and bacterial loading in urban streams." *J. Environ. Eng.*, 125(12), 1177-1180.

Zhang, L., Seagren, E.A., Davis, A.P., and J.S. Karns. (2010). "The capture and destruction of *Escherichia coli* from simulated urban runoff using conventional bioretention media and iron oxide-coated sand." *Water Environment Research*. 82: 701-714.

Zhang, L., Seagren, E.A., Davis, A.P., and Karns, J.S. (2011). "Long-term sustainability of *Escherichia coli* removal in conventional bioretention media." *Journal of Environmental Engineering*, 137(8), 669–677.